Appendix A: Copies of Cited Reports and Scientific Literature

Limnol. Oceanogr., 51(5), 2006, 2018–2032

Evaluating the biogeochemical cycle of selenium in San Francisco Bay through modeling

Shannon L. Meseck[1] *and Gregory A. Cutter*
Old Dominion University, Department of Ocean, Earth, and Atmospheric Sciences, Norfolk, Virginia 23529

Abstract

A biogeochemical model was developed to simulate salinity, total suspended material, phytoplankton biomass, dissolved selenium concentrations (selenite, selenate, and organic selenide), and particulate selenium concentrations (selenite + selenate, elemental selenium, and organic selenide) in the San Francisco Bay estuary. Model-generated estuarine profiles of total dissolved selenium reproduced observed estuarine profiles at a confidence interval of 91–99% for 8 different years under various environmental conditions. The model accurately reproduced the observed dissolved speciation at confidence intervals of 81–98% for selenite, 72–91% for selenate, and 60–96% for organic selenide. For particulate selenium, model-simulated estuarine profiles duplicated the observed behavior of total particulate selenium (76–93%), elemental selenium (80–97%), selenite + selenate (77–82%), and organic selenide (70–83%). Discrepancies between model simulations and the observed data provided insights into the estuarine biogeochemical cycle of selenium that were largely unknown (e.g., adsorption/desorption). Forecasting simulations investigated how an increase in the discharge from the San Joaquin River and varying refinery inputs affect total dissolved and particulate selenium within the estuary. These model runs indicate that during high river flows the refinery signal is undetectable, but when river flow is low (70-day residence time) total particle-associated selenium concentrations can increase to $>2\ \mu g\ g^{-1}$. Increasing the San Joaquin River discharge could also increase the total particle-associated selenium concentrations to $>1\ \mu g\ g^{-1}$. For both forecasting simulations, particle-associated selenium was predicted to be higher than current conditions and reached levels where selenium could accumulate in the estuarine food web.

Extensive research has been done on modeling how physical, biological, or chemical parameters in an estuary individually affect the distribution and speciation of a trace element (e.g., Paucot and Wollast 1997; Mwanuzi and De Smedt 1999), but little work has been done using models to simulate the complete biogeochemical cycle of a trace element (i.e., coupling physical, biological, and chemical processes). With recent advances in estuarine modeling, more extensive simulations of biogeochemical cycles of an element are now possible. Coupling empirical observations with modeling enables estuarine processes to be more completely elucidated than using either approach individually. In this respect, selenium presents some compelling reasons for use as the "test" element. Human activities (e.g., irrigation, petroleum refining, power production, and mining), have increased the input of selenium to some aquatic systems. This mobilization has been implicated in elevated concentrations of selenium in waterfowl, fish, and bivalves of some estuaries like the San Francisco Bay (Ohlendorf et al. 1986).

Selenium exists in four oxidation states (II, 0, IV, and VI), and in different chemical forms (i.e., organic and inorganic) within these oxidation states. In oxygenated marine and fresh waters, dissolved selenium is found as selenite (Se+IV, ca. 35% of the total selenium; Measures et al. 1980), selenate (Se+VI), and organic selenides (Se-II), with some of the organic species as selenium-containing amino acids and peptides (Cutter and Bruland 1984) and methylated forms (Cooke and Bruland 1987). Significantly, the biotic uptake and toxicity of dissolved selenium depends not only on its concentration, but also on its chemical speciation (Riedel et al. 1996). Thus, any modeling efforts with selenium must include the capability for accurate speciation predictions.

The biogeochemical cycle of dissolved selenium in estuarine waters (Takayanagi and Cossa 1985; Cutter 1989; Cutter and Cutter 2004) and sediments (Belzile and Lebel 1988; Velinsky and Cutter 1991) and its bioavailability in the food web (Doblin et al. 1999) have all been examined. The biogeochemical cycle of dissolved selenium in an estuary (Fig. 1) includes inputs via rivers, anthropogenic sources, and exchange with the open ocean. Advection and diffusion move selenium through the estuary, while internal transformations occur through biotic and abiotic reactions during transport. The transformation reactions (biotic and abiotic) include the oxidation of dissolved organic selenide to selenite and selenite to selenate. Biotic reactions affecting selenium in an estuary include dissolved selenite, selenate, and organic selenide uptake by phytoplankton, and incorporation into various biochemical components (Fig. 1).

The sources of particulate selenium to an estuary are particles from rivers (biogenic and mineral detritus), biogenic particles produced in the water column (phytoplankton detritus), and sediment resuspension. Suspended particulate organic selenide can undergo remineralization

[1] Present address: U.S. Department of Commerce/National Oceanic & Atmospheric Administration/National Marine Fisheries Service, 212 Rogers Avenue, Milford, Connecticut 06460.

Acknowledgments

We thank Lynda Cutter and Martina Doblin for sampling and providing the dissolved and particulate selenium speciation data, and Robin Stewart of U.S. Geological Survey for the acquisition and processing of refinery samples.

This project was supported by funds from the National Science Foundation Environmental Geochemistry and Biogeochemistry (EBG) Program (OCE-9707946 to G.A.C.) and CALFED (98-2015000-00096).

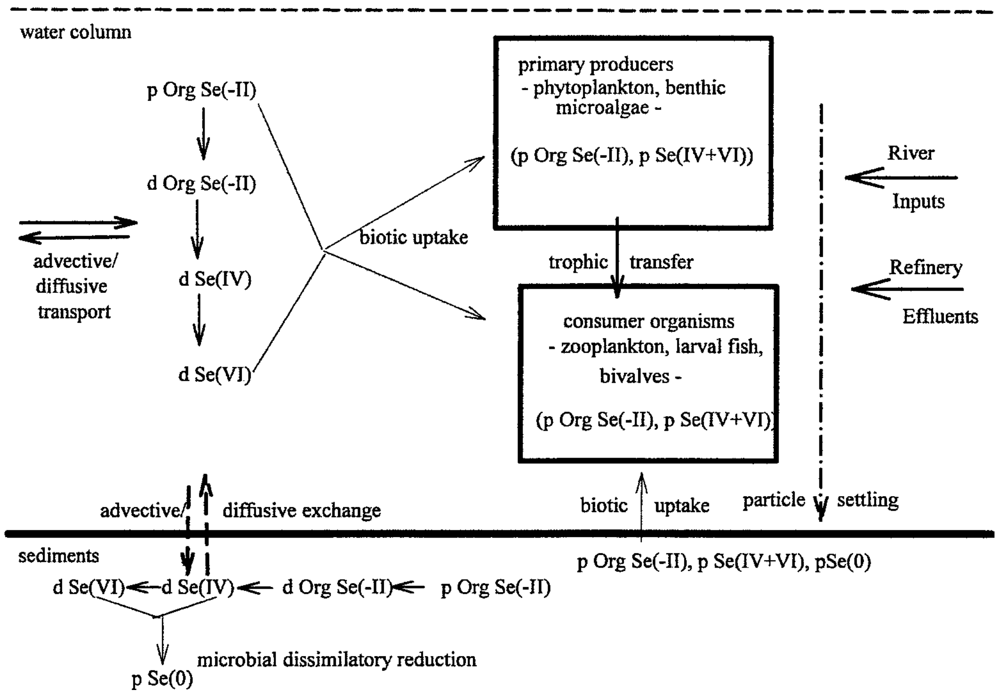


Fig. 1. Conceptual diagram of selenium's biogeochemical cycle in the San Francisco Bay estuary, with the major chemical speciation of particulate selenium in primary producers and consumer organisms indicated in brackets. Arrows represent fluxes or transformations; p is particulate, and d is dissolved.

to dissolved organic selenide or it can sink and become part of the sedimentary record (Fig. 1). In sediments, particulate selenium can undergo a variety of oxidation-reduction reactions that may cause selenium to become mobile or permanently buried (Velinsky and Cutter 1991). For example, particulate selenium in the bed of an estuary can undergo regeneration to dissolved organic selenide. In this way, sediments can become a source of dissolved selenium to the estuary via pore-water exchange with the overlying water. A biogeochemical model of selenium needs to include all of the processes in Fig. 1, including physical forcing (tidal and riverine transport, resuspension) that moves it through an estuary. Such a model can be used to test or evaluate the understanding of this complex cycle (e.g., the relative importance of phytoplankton versus sediment resuspension in controlling suspended particulate selenium).

Selenium concentrations in the particulate (Doblin et al. 2006), dissolved (Cutter 1989; Cutter and Cutter 2004), and sedimentary phases (Meseck and Cutter unpubl. data) have been determined in the San Francisco Bay estuary. These studies were conducted during a 16-year time period and provide data that were used to construct a biogeochemical model of selenium in the estuary based on Fig. 1. This article describes the model and its performance with past and present-day data (so-called "validation") and then presents future simulations of dissolved and particulate selenium in the San Francisco Bay under altered conditions of river flows and industrial inputs.

Methods

Study area—The San Francisco Bay is divided into what is known as the Northern Reach and the South Bay. The Northern Reach includes Central Bay, San Pablo Bay, and Suisun Bay (Fig. 2) and is the focus of this modeling effort; the South Bay will not be discussed further. Seawater enters the bay through the Golden Gate and proceeds into the Northern Reach, whereas freshwater from the Sacramento River and the San Joaquin River enters the bay through a complex series of channels, embayments, and marshes known as the "Delta." The Northern Reach has many features in common with other estuaries, namely, short residence time, strong tidal influences, a well to partially mixed water column, and natural and anthropogenic inputs of nutrients and trace elements. Freshwater residence times in the San Francisco Bay range from about 1 week during a high flow period to 3–5 weeks during low flow periods (Cutter 1989).

Model description—The Center for Coastal and Marine Sciences at the Plymouth Marine Laboratory, United Kingdom, developed the biogeochemical model ECoS 3, which can simulate biological productivity, total suspended material, salinity, nutrients, and trace metal behavior in the Tamar estuary. It is commercially available, and therefore only modifications made to the model to simulate the biogeochemical cycle of selenium in the San Francisco Bay (Fig. 1) are discussed. However, Web Appendix 1 (http://

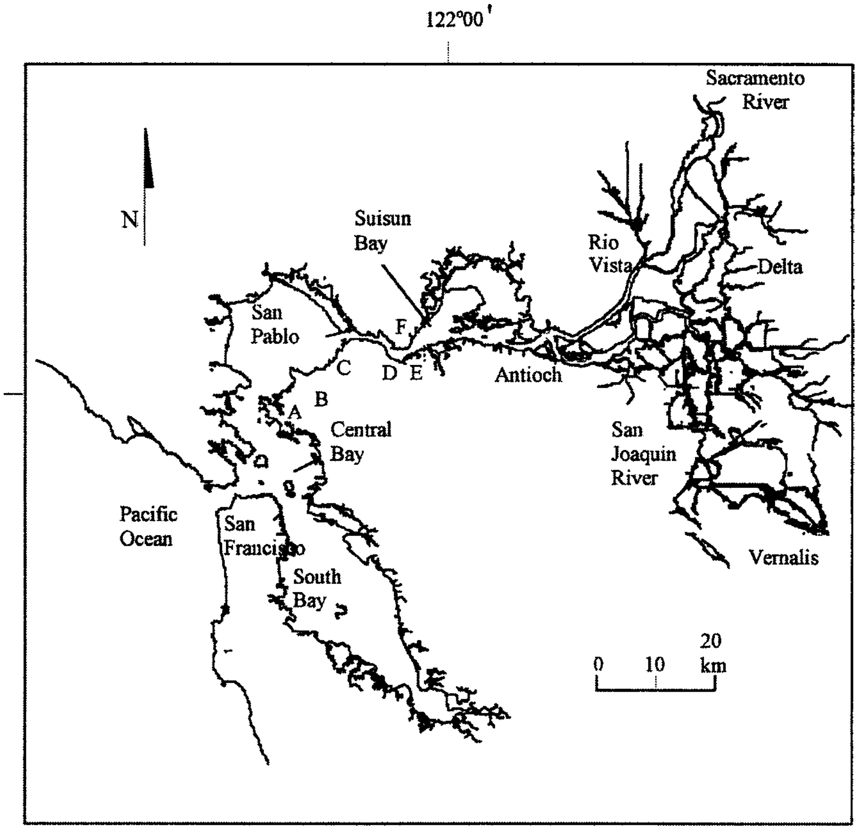


Fig. 2. The San Francisco Bay estuary, with the model domain being from the Golden Gate (between Central Bay and the Pacific Ocean) to Rio Vista on the Sacramento River. Refinery discharges into San Francisco Bay: (Fig. 2 The San Francisco Bay and Sacramento -San Joaquin Delta and the location of six major refineries. A) Chevron USA, Richmond Refinery; (B) Pacific Refining, Hercules; (C) Unocal, Rodeo; (D) Shell Oil, Martinez; (E) Tosco Refining, Martinez; and (F) Exxon USA, Benicia.

www.aslo.org/lo/toc/vol_51/issue_5/2018a1.pdf) has more thorough descriptions of the model and its parameters.

Modeling morphology and hydrology—The San Francisco Bay estuary was modeled as a multibox (33 boxes, each 3.3 km in length), one-dimensional estuary such that for any solute, s,

$$\frac{\partial s}{\partial t} = u\frac{\partial s}{\partial x} - \frac{\partial}{\partial x}\left(K_x \frac{\partial s}{\partial x}\right) - \Gamma \qquad (1)$$

where t is time, u is the tidal velocity, x is the axis of the estuary, K_x is the coefficient of longitudinal eddy diffusion, and Γ are other processes/reactions (e.g., biological uptake) that may affect the transport of a constituent (e.g., selenium) and are discussed following. Further information on how the model calculates the movement of a solute from one box to the next is provided in Web Appendix 1.

ECoS 3 has mathematical equations to define the shape of an estuary and its tidal movements (Harris and Gorely 1998), and because these equations were not modified (only the parameters were adjusted for the San Francisco Bay), further discussion on how the model defined the bathymetry and tidal components are given in Web Appendix 1. Although there are two major river inputs, approximately 98% of the flow from the San Joaquin River is diverted for irrigation practices, with the lower part of the river dependent on freshwater from agricultural drainage (Presser and Piper 1998). During most of the year, the San Joaquin discharge rate is low, with little or no water entering the estuary. Thus, the Sacramento River largely defined the riverine input of selenium into the Northern Reach and was the single freshwater end member (input) of the model.

However, the elevated dissolved selenium concentrations in the San Joaquin River (Cutter and San Diego-McGlone 1990; Cutter and Cutter 2004) can be an important input of selenium to the estuary when it does flow into the bay. Therefore, the San Joaquin River was treated as a point source (to a specific box) with variable flows over time. Discharge rates for both the Sacramento and San Joaquin rivers were obtained from the Interagency Ecological Program (http://iep.water.ca.gov/dayflow/).

Phytoplankton dynamics—Modeling phytoplankton biomass is difficult because of seasonal variations in community composition and productivity within an estuary but is needed since the biogeochemical cycle of dissolved and particulate selenium (Fig. 1) is influenced by biological uptake. In a well mixed estuary, phytoplankton biomass is modeled as

$$\frac{\partial B}{\partial t} = -U\frac{\partial B}{\partial x} + K_x\frac{\partial^2 B}{\partial x^2} + \mu_n B - GB - P_b B - \frac{\partial}{\partial z}(w_s B) - RB + B_{river} \quad (2)$$

where U is the water velocity (m d^{-1}), B is the phytoplankton biomass (g chlorophyll α [Chl *a*] L^{-1}), K_x is the dispersion coefficient along the axis of the estuary (m^2 d^{-1}), μ_n is the net biomass-specific growth rate (d^{-1}), G (d^{-1}) is loss rate attributed to zooplankton grazing, P_b (d^{-1}) is a loss caused by benthic grazing, z is the depth (m), w_s is the sinking rate (m d^{-1}), R (d^{-1}) is nonspecific natural mortality of phytoplankton caused by anything other than grazing (i.e., respiration), and B_{river} is the riverine input of phytoplankton (g Chl *a* L^{-1} d^{-1}). Sinking of phytoplankton tends to be small (0.5–0.9 m d^{-1}) and was set to a constant value based on literature values for the San Francisco Bay (Lucas et al. 1998). Mortality caused by respiration can be up to 10% of the maximum rate of photosynthesis at optimal light intensity (P_m) and was held at 10% of P_m in this model (Cole and Cloern 1984).

The net biomass-specific growth rate, μ_n in Eq. 2, is calculated as

$$\mu_n = \frac{P}{C:Chl} \quad (3)$$

where P is the biomass-specific rate of photosynthesis (mg C mg Chl a^{-1} d^{-1}), and C : Chl is the carbon to Chl *a* ratio (51 mg C mg Chl a^{-1}; Cloern and Alpine 1991) for the bay. The biomass-specific rate of photosynthesis, P, in the San Francisco Bay estuary is light-limited (Cole and Cloern 1984), and thus is determined from the photosynthesis-irradiance equation of Platt and Jassby (1976). However, an accurate simulation of the suspended particulate matter is required to accurately reproduce the in situ irradiance. Total suspended material (TSM) in the water column was defined in ECoS 3 (Harris and Gorley 1998), and the settling and resuspension rates were adjusted to those found in the San Francisco Bay. Further details of the parameters used for simulating TSM in the San Francisco Bay are given in Web Appendix 1.

Grazing in the San Francisco Bay includes zooplankton and benthic grazing. The specific loss of phytoplankton per day by zooplankton grazing (G in Eq. 2) is simulated from Cloern et al. (1985). Benthic grazing of phytoplankton (P_b in Eq. 2) changed largely with the introduction of the invasive clam *Potamocorbula amurensis* (Werner and Hollibaugh 1993). Before the introduction of *P. amurensis*, the main control on phytoplankton populations was zooplankton grazing, but the introduction of *P. amurensis* potentially increased phytoplankton grazing rates to values greater than specific growth rates of phytoplankton (Werner and Hollibaugh 1993). Data for the Northern Reach indicate that the largest number of benthic grazers are located in Suisun Bay (Thompson 2000); therefore, benthic grazing rates were increased from 0.04 d^{-1} to 0.05 d^{-1} (25%) for Suisun Bay.

Dissolved selenium—Results from Cutter and Cutter (2004) were used to parameterize dissolved selenium inputs from the Sacramento and San Joaquin. However, data for the San Joaquin River were taken at Vernalis (Cutter and Cutter 2004), which is approximately 60 km from where the San Joaquin River enters the Delta. Samples taken in the Delta in 1998 and 2000 (Cutter unpubl. data) indicate that the concentration of selenium may be reduced by 60–80% as it is being transported from Vernalis through the Delta and into the estuary at Antioch (Fig. 2). Based on this information, a removal constant of 60%, henceforth referred to as the "Delta removal constant," was applied to the input of selenium from the San Joaquin River.

The refineries along the San Francisco Bay have been a major source of selenium input (Cutter and San Diego-McGlone 1990). The concentration and speciation of selenium from the refineries has varied significantly during the last 10 years (Cutter and Cutter 2004). Refinery inputs of dissolved selenium were treated as point sources in the model, with inputs corresponding to each refinery location identified in Fig. 2. Total selenium output fluxes were obtained from each refinery for the years of interests (San Francisco Bay Regional Water Quality Control Board pers. comm.). Cutter and Cutter (2004) and Cutter and San Diego-McGlone (1990) determined the speciation of selenium from the refinery output, and their data were used in the model.

The in situ processes, Γ, for modeling dissolved selenium in the bay include production and removal terms (Fig. 1). These processes for selenite, selenate, and organic selenide are described in the model as

$$\frac{\partial D\mathrm{Se(IV)}}{\partial t} = k_3[\mathrm{DSe(IV)}] - k_5[\mathrm{DSe(VI)}] \quad (4)$$

$$\frac{\partial D\mathrm{Se(IV)}}{\partial t} = k_2[\mathrm{DSe(-II)}] - k_3[\mathrm{DSe(IV)}] - k_4[\mathrm{DSe(IV)}] \quad (5)$$

$$\frac{\partial D\mathrm{Se(-II)}}{\partial t} = k_1[\mathrm{PSe(-II)}] - k_2[\mathrm{DSe(-II)}] - k_6[\mathrm{DSe(-II)}] \quad (6)$$

Previously determined rate constants (Cutter and Bruland 1984; Cutter 1992) were used for k_1, k_2, and k_3. The rate constants k_4, k_5, and k_6 are controlled by phytoplankton (Fig. 1). Typically, Michaelis–Menton uptake kinetics would be used, but few are available for selenium. However, the data that are available (e.g., Vandermeulen and Foda 1988; Riedel et al. 1996) suggest that for the selenium concentrations in the bay, uptake would be in the linear region and thus can be modeled as first-order reactions (*see Web Appendix 1*). There are many factors affecting dissolved uptake rates, such as the species of phytoplankton that are actually present, but the best available constants were selected for this model. The model sensitivity to these rate constant values is described in Web Appendix 2 (http://www.aslo.org/lo/toc/vol_51/issue_5/2018a2.pdf). For selenite (k_4) and selenate (k_5) the first-order uptake rate constants from Riedel et al. (1996) were used, since the bay can be dominated by freshwater diatoms in the upper reaches (Lehman 2000). For organic selenide, Baines et al. (2001) found that organic selenide uptake is about half the rate of selenite uptake using phytoplankton species found in the San Francisco Bay. Based on this, the uptake rate constant for organic selenide (k_6) was set at half of the selenite value.

Pore-water exchange can be a significant source or sink of dissolved selenium to the estuary. Pore-water exchange is modeled as

$$\frac{\partial Se_{porewater}}{\partial t} = A \times J_{Se} \tag{7}$$

where A is the area (m^2) of the sediment, and J_{Se} is the diffusive flux (nmol m^{-2} yr^{-1}). Diffusive fluxes were calculated based on the overlying water selenium concentration (model-generated) and measured pore-water concentrations that vary little with season (Meseck and Cutter unpubl. data).

Particulate selenium—Modeling particulate selenium in the bay requires the ability to model the transport of suspended sediments within the water column as described previously. Particulate selenium in the water column is derived from sediment resuspension, sediment loads from the Sacramento River, and in situ production (e.g., phytoplankton uptake of selenium). This can be expressed as:

$$\frac{\partial PSe}{\partial t} = Se_{SED}\frac{\partial BEPS}{\partial t} + Se_{river}\frac{\partial PSM}{\partial t} - \Gamma \tag{8}$$

where PSe is the particulate selenium concentration (nmol L^{-1}), Se_{SED} is the selenium concentration in the uppermost sediment that can be resuspended (nmol g^{-1}; Meseck and Cutter unpubl. data), BEPS is the load of resuspended sediment (g L^{-1} generated in the model through tidal movement and river flows), Se_{river} is the concentration of selenium in riverine particles (nmol g^{-1}; Doblin et al. 2006), PSM is permanently suspended material in the river (g L^{-1}), and Γ is all the in situ reactions/processes, which will be defined below for each species of particulate selenium.

With respect to the speciation of particulate selenium and in situ processes, elemental selenium is primarily generated by dissimilatory selenite + selenate reduction (Oremland et al. 1989; Cutter 1992). Since the water column of San Francisco Bay is oxic, the presence of particulate elemental selenium in total suspended material can be attributed to either sediment resuspension or riverine particulate inputs (i.e., $\Gamma = 0$). For particulate selenite + selenate, besides sediment inputs, in situ adsorption/desorption processes can affect this concentration in the water column. The in situ adsorption/desorption of selenite + selenate was modeled by the distribution coefficient $K_d = a'/b$, where a' is the intrinsic adsorption rate constant (L g^{-1} d^{-1}), and b is the rate constant (d^{-1}) of desorption (Nyffeler et al. 1984). The value of K_d was obtained from Zhang and Moore (1996), and values of a' were obtained from Nyffeler et al. (1984). The rate constant b was obtained by rearranging the K_d equation to $b = a'/K_d$. Similarly, sediment inputs and in situ processes control the concentration of particulate organic selenide. Once phytoplankton take up dissolved selenium, it is converted into particulate organic selenide as controlled by k_4, k_5, and k_6 (Eqs. 4–6).

Results

Model validation—Sensitivity analyses of the model are discussed in Web Appendix 2, and only the validation results are presented here to illustrate its performance. Three statistical tests were used to determine the ability of the model to reproduce observed behaviors of the various modeled parameters. They included the linear correlation coefficient, the mean cumulative error (*M*, the bias of the model), and the confidence interval (CI). For the linear correlation coefficient, the 95% CI was used ($p < 0.05$). The mean cumulative error indicates if the model was underpredicting relative to the observed values (a negative value) or overpredicting (a positive sign). The confidence interval is not affected by outliers or data variation because it measures the absolute difference between the predicted concentrations of the model and the actual data and is probably one of the better measures of model performance (Perrin et al. 2001).

Estuarine profiles of salinity, phytoplankton biomass, total suspended material, and dissolved and particulate selenium from 1986 and 1998 were used for validation (1999 data were used for calibration) because they represent extremes in several parameters, including river discharge and an increase in benthic grazing from the invasive clam *P. amurensis*. Furthermore, these years represent periods of low (38 mol Se d^{-1}, 1998) and high (99 mol Se d^{-1}, 1986) refinery discharge of total dissolved selenium, and the speciation in their effluents changed from primarily selenite in 1986 to selenate in 1998 (Cutter and San Diego-McGlone 1990; Cutter and Cutter 2004). The 1986 and 1998 data sets also have data available during high flow months (April 1986, June 1998) and low flow months (September 1986, October 1998), providing a variety of conditions within the estuary to validate the model.

Salinity, TSM, and phytoplankton—The model must be able to simulate the physical transport in an estuary (i.e., salinity), phytoplankton growth, and TSM, because these are critical parts of the biogeochemical cycle of selenium (Fig. 1). The model was able to reproduce the observed salinity profiles at *r* values >0.95 (Table 1) for the observed salinities in 1986 and 1998 (Fig. 3). More specifically, the model underpredicted salinity (−0.07) for 23 April 1986 and overpredicted it for 23 September 1986, 12 June 1998, and 12 October 1998 (Table 1). The confidence intervals of the model to simulate salinities ranged from 80% to 97% (Table 1). Overall, Fig. 3 and the *r*, CI, and *M* results show that the model was able to accurately reproduce the salinity

Table 1. Summary of validation results for various years for salinity, phytoplankton biomass, and total suspended material. The table gives the linear correlation (r), mean cumulative error (M), and confidence interval (CI) between the observed data and model simulations.*

Year	Salinity			Phytoplankton			TSM		
	r	M†	CI (%)	r	M† (μg Chl a L^{-1})	CI (%)	r	M† (mg L^{-1})	CI (%)
23 Apr 1986	0.966‡	−0.07	92	0.789‡	+0.97	84	0.678‡	+2.8	93
23 Sep 1986	0.995‡	+0.06	95	0.635‡	+0.71	79	0.626‡	−3.5	86
08 Oct 1987	0.979‡	−0.08	96	NA	NA	NA	NA	NA	NA
15 Mar 1988	0.952‡	−0.27	80	NA	NA	NA	NA	NA	NA
11 May 1988	0.979‡	+0.35	97	NA	NA	NA	NA	NA	NA
06 Nov 1997	0.984‡	−0.83	86	0.469‡	−0.24	81	0.589‡	+1.1	96
12 Jun 1998	0.974‡	+0.13	80	0.207	+0.65	75	0.527‡	+8.3	74
12 Oct 1998	0.960‡	+0.08	86	0.561‡	+0.41	78	0.879‡	−1.3	96

* NA, data not available.
† −, model was underpredicting relative to the observed; +, model was overpredicting.
‡ Linear correlation was significant at $p < 0.05$.

profiles for the various conditions in 1986 and 1998. As a result, all remaining figures were plotted against salinity so that removal/production processes can be observed (e.g., as in Cutter and Cutter 2004).

Particulate selenium profiles in the estuary are affected by processes controlling total suspended material, making it essential that the model reproduce the estuarine profiles of TSM. Model-generated simulations show maxima in the upper reaches of the estuary that agree with the observed data in 1986 and 1998 (Fig. 3), and the high linear correlation coefficients, high CI, and low M (Table 1) indicate that the model validation for TSM was fully acceptable. Nevertheless, it should be noted that the model does not simulate wind mixing, so these events will not be included in the simulations (but also do not appear to be important for these validation periods as the model captures most of the observed behaviors).

The uptake of dissolved selenium and the production of particulate organic selenide are largely a function of phytoplankton biomass, and therefore accurate model reproductions of the observed phytoplankton distributions are required. Model simulations of phytoplankton biomass were similar to the observed data for 1986 and 1998 (Fig. 3), although the model did overpredict phytoplankton biomass in both years (from 0.41 μg Chl a L^{-1} to 0.97 μg Chl a L^{-1}; Table 1). The linear correlation was significant ($p < 0.05$) for all the months simulated except for 12 June 1998 (Table 1). Model-generated phytoplankton biomass in June was overpredicted for the entire estuary, and sensitivity analyses of the model (Web Appendix 2) indicate that phytoplankton biomass was largely controlled by grazing. Therefore, it is possible that for June 1998, grazing of phytoplankton biomass was slightly underestimated. However, even though the linear correlation was not significant, the model was still able to predict 75% of the observed data (Table 1). Other than this month, the simulation results show that the model is accurately predicting phytoplankton biomass (Table 1). Significantly, the validation periods included an extreme change in benthic grazing rates, and the model confirmed the empirical observations of lowered phytoplankton biomass in the estuary because of this change (Cloern and Alpine 1991).

Dissolved selenium, high flow months—For 1986, high flow samples were taken in April, whereas in 1998 they were taken in June. Total dissolved selenium in the estuary displayed conservative mixing behavior in April 1986 (Fig. 4A) and appears nonconservative in June 1998 (Fig. 4B). Model simulations reproduced the conservative and nonconservative behavior of total dissolved selenium for both months (r, CI, and M in Table 2).

Selenite concentrations in the estuary ranged from 0.2 nmol L^{-1} to 0.6 nmol L^{-1} and showed nonconservative behavior (Fig. 4C,D) that the model was able to reproduce for both months. On 23 April 1986, the confidence of the model was 95%, it slightly underpredicted selenite (−0.02 nmol L^{-1}), and the correlation coefficient was significant (r = 0.902). For 12 June 1998, the correlation coefficient was not significant (r = 0.435), but the confidence interval was 99% and the mean cumulative error was 0.00 nmol L^{-1}, indicating that there was an excellent fit between the observed and modeled concentrations (Table 2).

Observed selenate concentrations varied from 1.0 nmol L^{-1} to a maximum of 2.0 nmol L^{-1} (Fig. 4E,F), with selenate displaying nonconservative behavior in April 1986 (Fig. 4E) and conservative in June 1998 (Fig. 4F). Simulated estuarine profiles reproduced these behaviors, with a 76% confidence on April 1986 and a 75% confidence in June 1998 (Table 2). The linear correlation coefficient was significant for April (r = 0.832) and June (r = 0.574), with a cumulative error of −0.25 nmol L^{-1} for 23 April 1986 and +0.38 nmol L^{-1} for 12 June 1998 (Table 2).

Dissolved organic selenide concentrations varied from nondetectable to approximately 2.0 nmol L^{-1} for the months examined and displayed nonconservative behavior in the estuary (Fig. 4G,H). The correlation coefficient for 23 April 1986 was not significant (r = 0.319), but the high confidence interval (92%) and low mean cumulative error (−0.05 nmol L^{-1}; Table 2) indicate that the model was able to reproduce the observed dissolved organic selenide

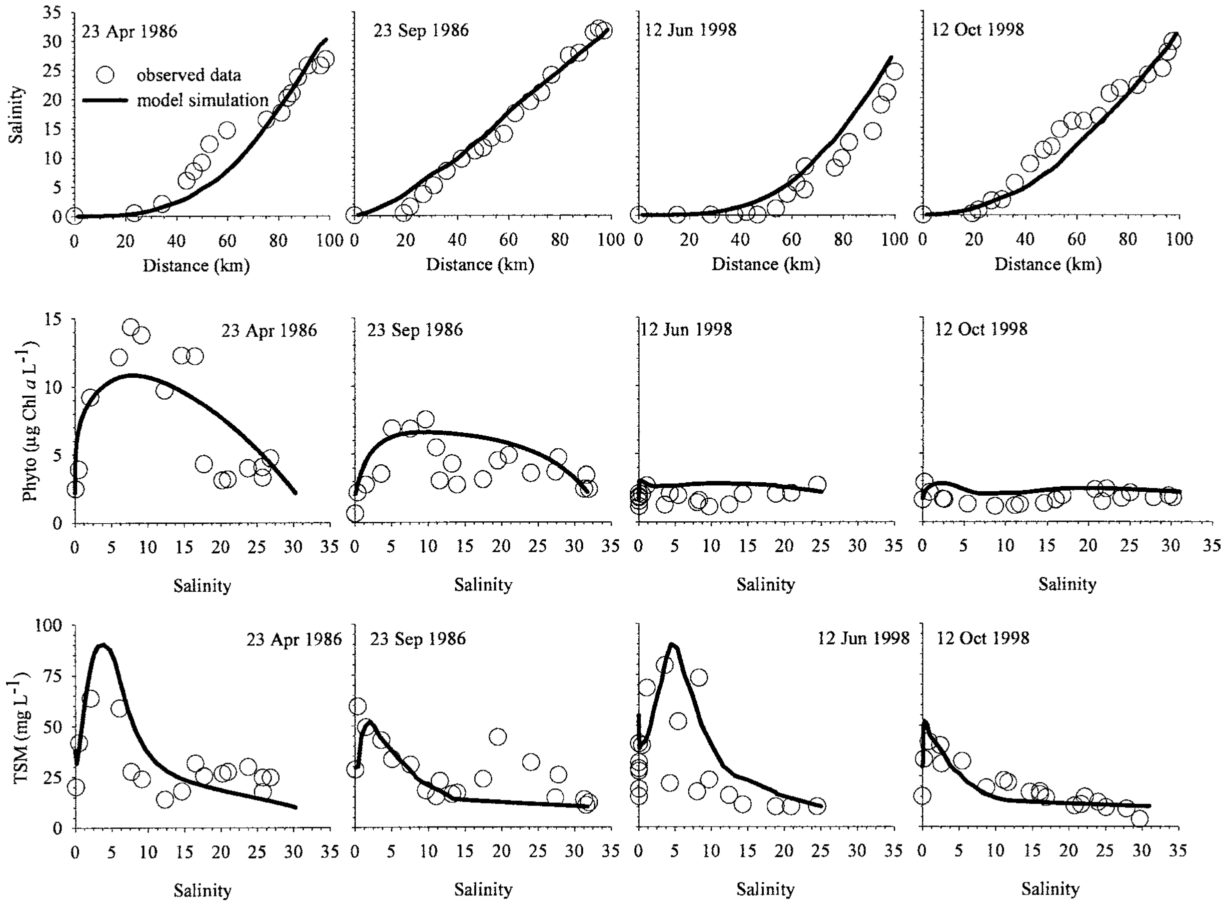


Fig. 3. Estuarine profiles of model-generated and observed salinity, TSM, and phytoplankton biomass (as Chl *a*) for 1986 and 1998 in the San Francisco Bay.

profile. For the high flow month of 12 June 1998, the model was able to predict 80% of the dissolved organic selenide in the estuary (Table 2).

Dissolved selenium, low flow months—Samples taken on 23 September 1986 and 12 October 1998 for dissolved selenium represent the low river flow months in the San Francisco Bay. Observed total dissolved selenium during these periods displayed nonconservative behavior within the estuary, although the concentration in 1986 reached a maximum of 4 nmol L^{-1}, whereas in 1998 the maximum was only 2.0 nmol L^{-1}. Model simulations were able to reproduce these nonconservative behaviors (Fig. 5A,B) for both years (r, CI, and M values in Table 2).

Selenite concentrations on 23 September 1986 reached a mid-estuary maximum of 1 nmol L^{-1} (Fig. 5C), whereas on 12 October 1998 the mid-estuary maximum was only 0.5 nmol L^{-1} (Fig. 5D) because of decreased selenite fluxes from refineries (Cutter and Cutter 2004). For September 1986 the model underpredicted selenite (-0.13 nmol L^{-1}), but the correlation coefficient was significant ($r = 0.946$) and was able to reproduce 85% of the observed data (Table 2). The fit for October 1998 was just as good, with a confidence of 81%, a mean cumulative error of -0.06 nmol L^{-1}, and a significant correlation coefficient ($r = 0.688$).

Estuarine concentrations in 1986 and 1998 for selenate were similar and ranged from 0.2 nmol L^{-1} to 1.2 nmol L^{-1} (Fig. 5E,F). The correlation coefficient for 23 September 1986 was not significant ($r = 0.296$), but the model reproduced the observed estuarine profile of selenate at a confidence interval of 91% and slightly underpredicted selenate (-0.06 nmol L^{-1}; Table 2). For 12 October 1998, the correlation coefficient was significant ($r = 0.589$), and although the model underpredicted selenate (-0.19 nmol L^{-1}), it was still able to simulate 77% of the observed selenate behavior (Table 2).

The estuarine profiles for organic selenide in the low flow months display nonconservative profiles within the estuary (Fig. 5G,H). Model simulations of organic selenide reproduced the observed estuarine profiles at a 96% confidence interval in 23 September 1986 and 63% in October 1998 (Table 2). For both years the linear correlations were significant, but for the 23 September 1986 simulation the model slightly underpredicted organic selenide (-0.02 nmol L^{-1}) and overpredicted organic selenide on 12 October 1998 (+0.13 nmol L^{-1}; Table 2).

Table 2 summarizes the r, CI, and M values for all the dissolved selenium data that were available for validating the model. The confidence interval of the model varied

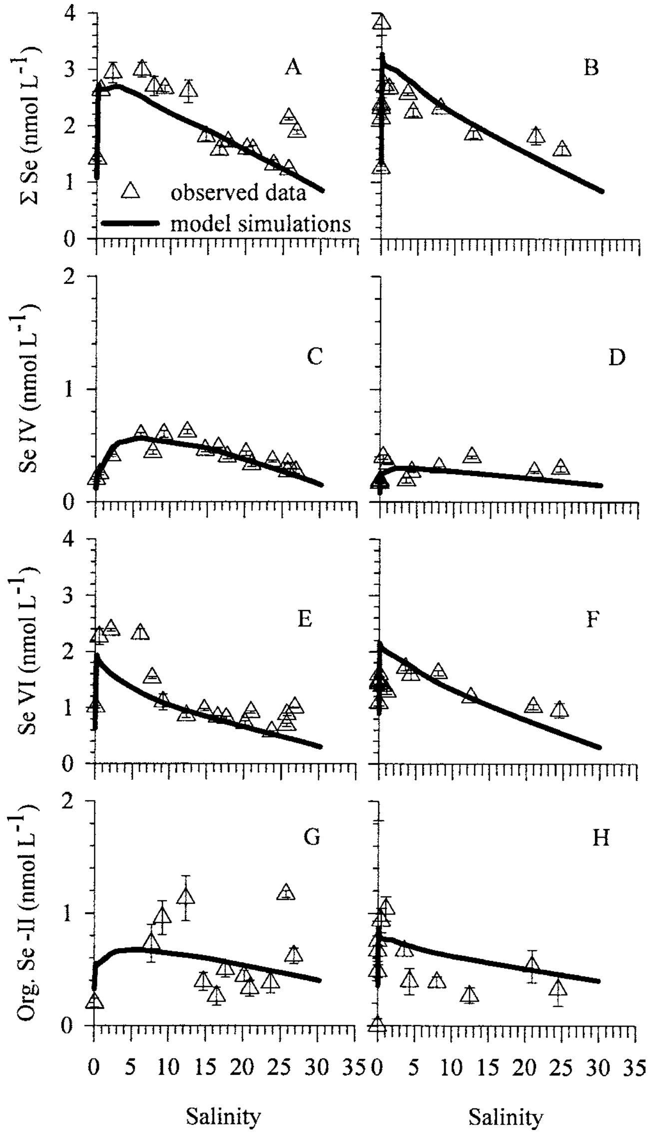


Fig. 4. High flow model-generated (A) total dissolved selenium, (C) selenite, (E) selenate, and (G) organic selenide and observed data for 23 April 1986 and (B) total dissolved selenium, (D) selenite, (F) selenate, and (H) organic selenide and observed data for 12 June 1998.

from a low of 60% to a high of 99% of the observed behavior (Table 2). Furthermore, most of the correlation coefficients were significant, and for those that were not, there was usually a low mean cumulative error (e.g., 0.00 nmol L^{-1} for selenite on 12 June 1998) and a high model confidence interval (Table 2). Overall, the model performance for simulating the behavior of dissolved selenium under extremely variable environmental conditions is very good.

Particulate selenium—For both high (12 June 1998) and low flows (12 October 1998), total particulate selenium concentrations ranged from 0.1 nmol L^{-1} to 0.3 nmol L^{-1} (Fig. 6A,B), with higher concentrations located near the maxima of total suspended material (Fig. 3). Model-derived total particulate selenium concentrations reproduced the observed upper estuarine maxima (Fig. 6A,B) for both years (r, CI, and M results in Table 3).

Table 2. Model validation results for dissolved selenium and its speciation in the San Francisco Bay. The linear correlation (r), mean cumulative error (M), and confidence interval (CI) between the observed data and model simulations are given.

Year	r	M* (nmol L^{-1})	CI (%)
23 Apr 1986			
Total selenium	0.791†	−0.18	91
Selenite	0.902†	−0.02	95
Selenate	0.832†	−0.25	76
Organic selenide	0.319	−0.05	92
23 Sep 1986			
Total selenium	0.882†	−0.16	92
Selenite	0.946†	−0.13	85
Selenate	0.296	−0.06	91
Organic selenide	0.576†	−0.02	96
08 Oct 1987			
Total selenium	0.568†	−0.12	96
Selenite	0.543†	+0.06	95
Selenate	0.561†	−0.35	72
Organic selenide	0.111	+0.01	98
15 Mar 1988			
Total selenium	0.991†	+0.03	99
Selenite	0.890†	+0.01	98
Selenate	0.497†	−0.10	90
Organic Selenide	0.625	+0.09	80
11 May 1988			
Total selenium	0.589†	+0.06	97
Selenite	0.807†	−0.11	89
Selenate	0.197	−0.13	87
Organic Selenide	0.110	+0.11	76
06 Nov 1997			
Total selenium	0.550†	−0.22	91
Selenite	0.551†	−0.08	87
Selenate	0.622†	−0.48	63
Organic selenide	0.525†	+0.25	60
12 Jun 1998			
Total selenium	0.445	+0.06	96
Selenite	0.435	0.00	99
Selenate	0.574†	+0.38	75
Organic selenide	0.532	+0.03	80
12 Oct 1998			
Total selenium	0.613†	−0.11	93
Selenite	0.688†	−0.06	81
Selenate	0.589†	−0.19	77
Organic selenide	0.607†	+0.13	63

* −, model underpredicted; +, the model overpredicted.
† Linear correlation is significant at $p < 0.05$.

Unlike dissolved selenium, speciation data for particulate selenium for high and low flow months were only available in 1998 (Doblin et al. 2006). There was little variation in the estuarine profiles of particulate selenite + selenate (Fig. 6C,D) for both June and October, with the concentrations ranging from nondetectable to

BDCP1673

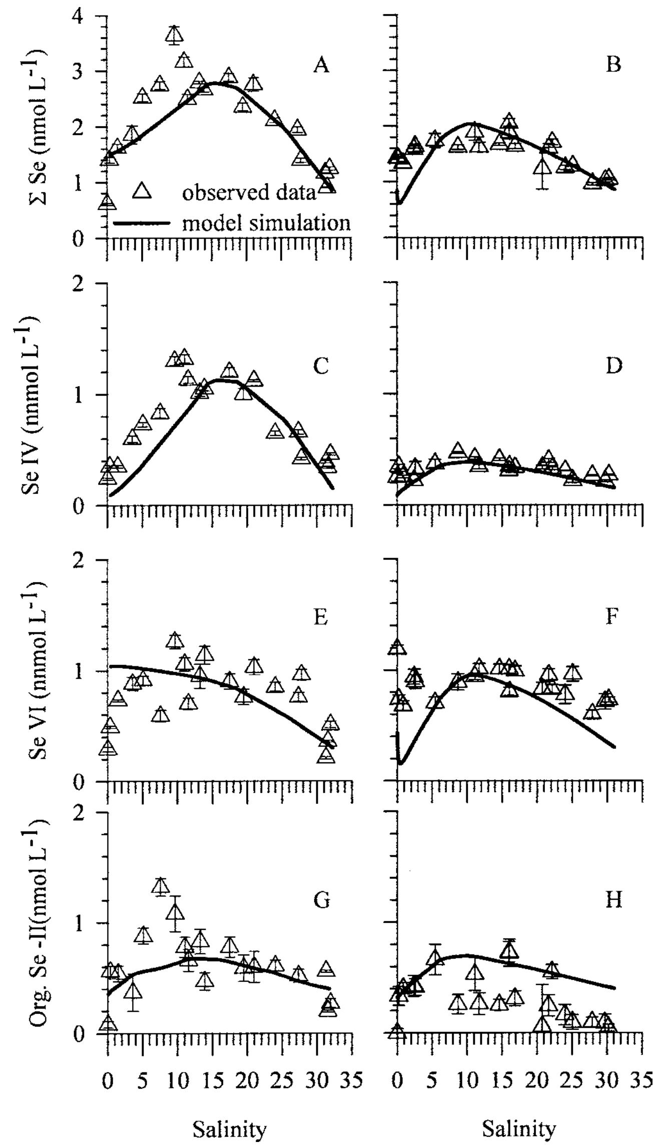


Fig. 5. Low flow model-generated (A) total dissolved selenium, (C) selenite, (E) selenate, and (G) organic selenide and observed data for 23 September 1986 and (B) total dissolved selenium, (D) selenite, (F) selenate, and (H) organic selenide and observed data for 12 October 1998.

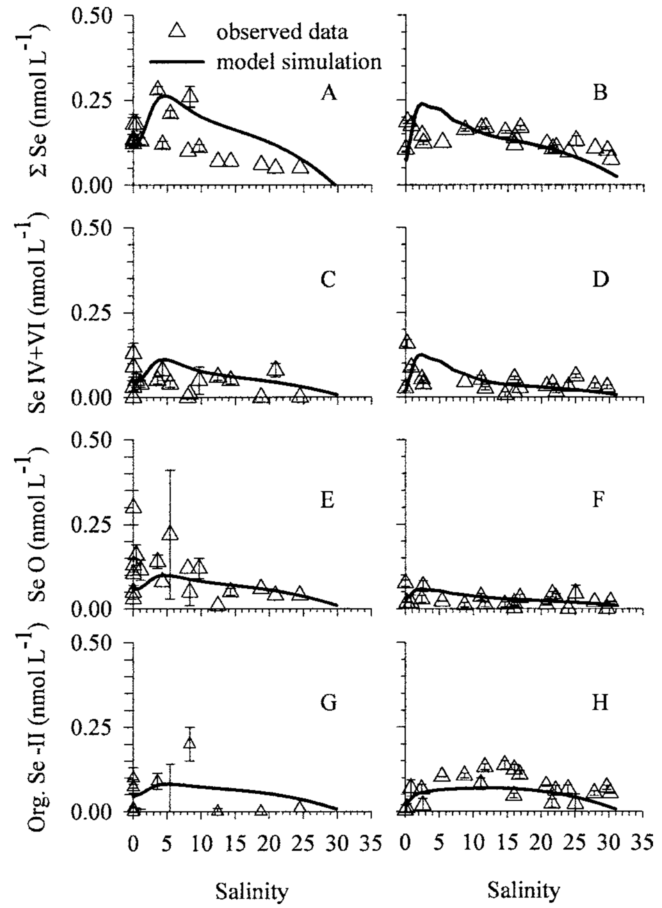


Fig. 6. Model-generated (A) total particulate selenium, (C) selenite, (E) selenate, and (G) organic selenide and observed data for 12 June 1998 and model-generated (B) total particulate selenium, (D) selenite, (F) selenate, and (H) organic selenide and observed data for 12 October 1998.

0.15 nmol L^{-1}. Simulated particulate selenite + selenate concentrations are within the errors of the observed data (Fig. 6C,D). The linear correlations were insignificant for June and October 1998 simulations; however, the confidence interval in June 1998 was 80% and 77% in October 1998, and the model slightly overpredicted particulate selenite + selenate in October (+0.01 nmol L^{-1}), and slightly underpredicted the concentrations in June (−0.02 nmol L^{-1}; Table 3). Differences between the observed and simulated data may be caused by difficulties in quantifying adsorption/desorption in the model. For example, absorption/desorption studies of selenite + selenate were done in freshwater (Zhang and Moore 1996) and may not be applicable to an estuarine environment. Sensitivity analyses of adsorption/desorption process indicate that adsorption/desorption coefficients may significantly affect predictions of estuarine particulate selenite + selenate (Web Appendix 2). Further studies of the adsorption/desorption of selenite + selenate on estuarine particles are certainly needed.

Observed particulate elemental selenium concentrations ranged from nondetectable to 0.25 nmol L^{-1} (Fig. 6E,F), with higher concentrations located in the upper estuary. Model-derived concentrations of elemental selenium produced an estuarine distribution similar to the observed data for both June 1998 and October 1998 (Fig. 6E,F, respectively). The linear correlation for both months was not significant, but the confidence interval was 80% in June and 98% in October (Table 3). The model slightly overpredicted elemental selenium in June (0.01 nmol L^{-1}), but agreed perfectly for October (0.00 nmol L^{-1}; Table 3).

Doblin et al. (2006) found that in October 1998 there was a mid-estuary maximum of particulate organic selenide (Fig. 6H), whereas in June most of the observed concentrations were below the detection limits (Fig. 6G). As with dissolved organic selenide, particulate organic selenide is

Table 3. Model validation of particulate selenium and its speciation for San Francisco Bay. The linear correlation (r), mean cumulative error (M), and confidence interval (CI) between the observed data and model simulations are given.

	r	M* (nmol L^{-1})	CI (%)
23 Apr 1986			
Total selenium	0.742†	+0.01	93
23 Sep 1986			
Total selenium	0.778†	+0.03	84
06 Nov 1997			
Total selenium	0.439†	+0.02	85
Selenite + selenate	0.683†	−0.01	82
Elemental	−0.261	−0.01	87
Organic selenide	−0.115	−0.01	83
12 Jun 1998			
Total selenium	0.425†	+0.03	76
Selenite + selenate	0.185	−0.02	80
Elemental	−0.288	+0.01	80
Organic selenide	0.211	−0.02	70
12 Oct 1998			
Total selenium	0.623†	−0.03	78
Selenite + selenate	0.152	+0.01	77
Elemental	0.124	0.00	98
Organic selenide	0.446†	−0.01	77

* −, model underpredicted relative to the observed; +, the model overpredicted.
† Linear correlation is significant at $p < 0.05$.

determined by difference [ΣPartSe – Se(0) – Se(IV + VI)] and often results in larger error bars in the observed data (Fig. 6G,H). Model simulations were able to reproduce the observed behavior by 70% in June 1998 and 77% in October 1998 (Table 3). For all simulations of particulate organic selenide, the model underpredicted the amount of organic selenide relative to the observed data (Table 3). Sensitivity analyses (Web Appendix 2) showed that phytoplankton uptake constants (k_4, k_5, k_6) are important variables for simulating particulate organic selenide, but as noted above, are poorly constrained for phytoplankton in the San Francisco Bay (uptake may be even higher than that used in the model). By improving these rate constants, the ability to predict particulate organic selenide should improve, although with the currently available data the model can simulate at least 70% of the observed behavior.

Overall, the model reproduced the majority of selenium's estuarine behavior under extreme changes in river and refinery inputs and in the ecosystem structure (changing from pelagic- to benthic-dominated grazing). However, it also provides insights into the biogeochemical cycle of selenium in the San Francisco Bay estuary that were not readily apparent or studied. On a simplistic level, discrepancies between model simulations and observed data are the result of either the model not including important processes or the observations being insufficient (e.g., sampling density or timing) to reveal other processes. An excellent example of a model-derived insight is the

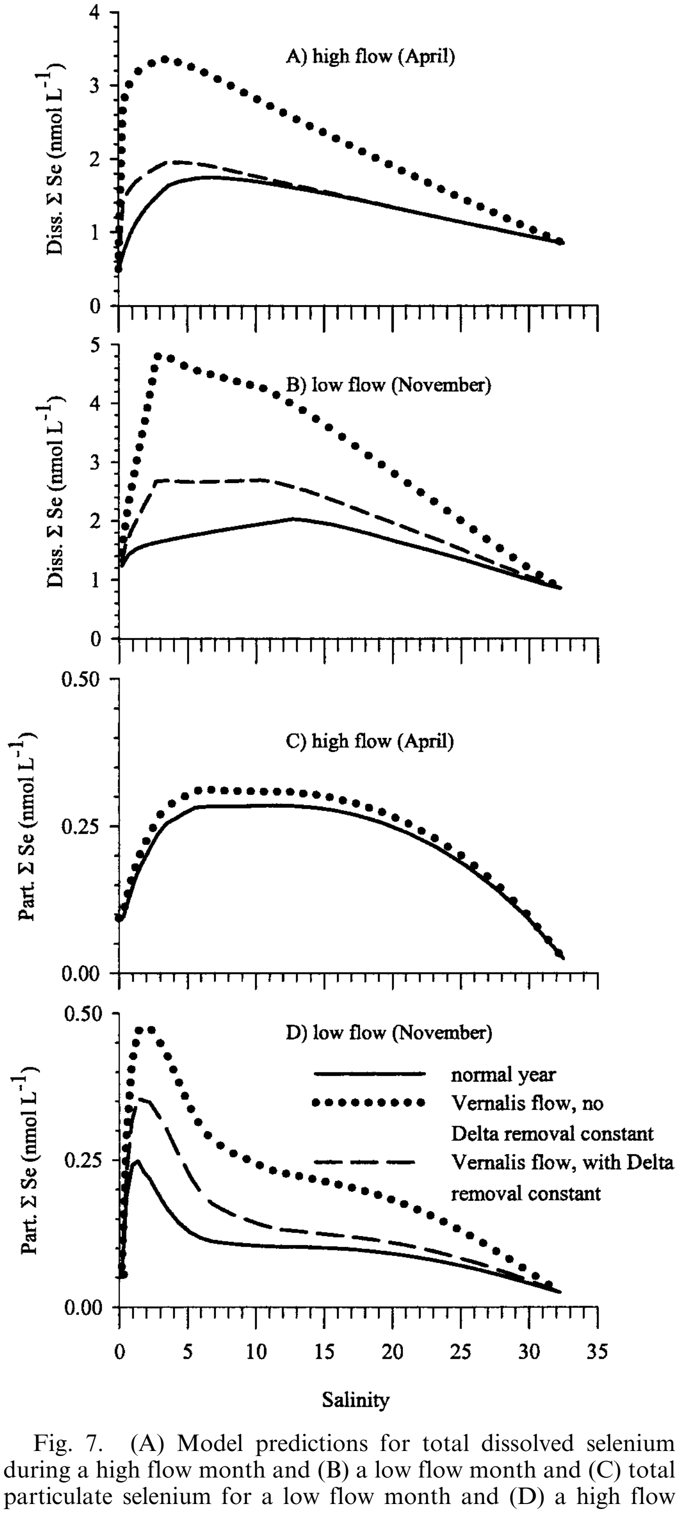


Fig. 7. (A) Model predictions for total dissolved selenium during a high flow month and (B) a low flow month and (C) total particulate selenium for a low flow month and (D) a high flow month in the San Francisco Bay under varying flows from the San Joaquin River. The "normal year" simulation used the Vernalis flow minus Delta withdrawals and imposed a Delta removal constant of 60% for all selenium inputs. "Vernalis flow, no Delta removal constant" simulations used the full flow at Vernalis without any withdrawals and no 60% selenium removal. "Vernalis flow, with Delta removal constant" simulations used the full flow at Vernalis without any withdrawals, but used the 60% selenium removal constant in the Delta.

Table 4. Predicted particle-associated selenium concentrations (as averages for the entire estuary) during a typical high flow (April) and low flow month (November) for different river discharge from the San Joaquin River and different refinery inputs.

	Particulate Σ Se ($\mu g\ g^{-1}$)
April	
Vernalis flow, no Delta removal constant	0.44±0.28
Vernalis flow, with Delta removal constant	0.44±0.26
Normal San Joaquin flow	0.43±0.26
November	
Vernalis flow, no Delta removal constant	1.00±0.32
Vernalis flow, with Delta removal constant	0.64±0.16
Normal San Joaquin flow	0.51±0.11
April	
99 mol d^{-1} total selenium	0.43±0.26
38 mol d^{-1} total selenium	0.43±0.26
No refinery inputs	0.43±0.26
November	
99 mol d^{-1} total selenium	0.62±0.14
38 mol d^{-1} total selenium	0.51±0.11
No refinery inputs	0.43±0.09

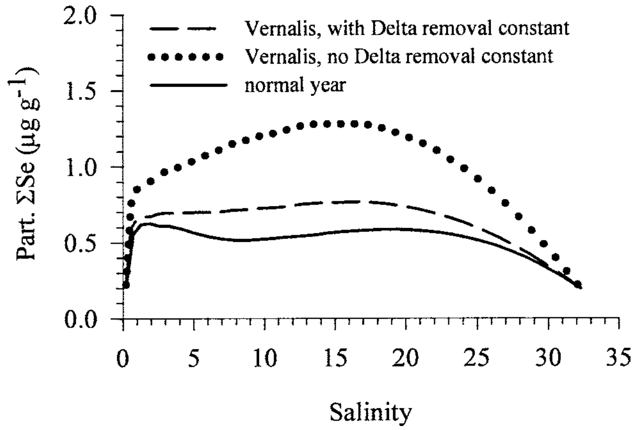


Fig. 8. Model predictions of particle-associated selenium in the San Francisco Bay for three different San Joaquin flow scenarios. The "normal flow" simulation used the Vernalis flow minus Delta withdrawals and imposed a Delta removal constant of 60% for all selenium inputs. "Vernalis flow, no Delta removal constant" used the full flow at Vernalis without any withdrawals and no 60% selenium removal; "Vernalis flow, with Delta removal constant" used the full flow at Vernalis without any withdrawals, but used the 60% selenium removal constant in the Delta.

importance of selenite and selenite adsorption/desorption onto particles. Without adsorption/desorption in the model, particulate selenite + selenate were underpredicted, and a 25% variation in the adsorption/desorption constant resulted in a 50% change in the particulate selenite + selenate within the estuary during low flow months. Relevant adsorption constants are not available for selenium in an estuarine environment, and the results of the model argue that they should be obtained. Another powerful use of such a model is assessing the relative importance of processes or inputs. For the San Francisco Bay the sources of suspended particulate selenium (in situ, riverine, sediment resuspension) are critically important as this phase is what is consumed by grazers and made available to the estuarine food web (e.g., Stewart et al. 2004; Luoma and Rainbow 2005). In this respect, the model results and sensitivity analyses found that during high flow months particles within the estuary are primarily controlled by riverine inputs; during low flow months, in situ processes account for most of the variation in particulate selenium within the estuary. For example, during a low flow month, varying the river discharge by 25% resulted in only a 22% change in particulate organic selenide within the estuary, whereas varying phytoplankton productivity by only 25% resulted in a 157% change. By dividing the observed total particulate selenium (nmol L^{-1}) by the total suspended material concentration (mg L^{-1}) the particle-associated selenium ($\mu g\ g^{-1}$) can be calculated. We find that during high flow (23 April 1986) the particle-associated selenium is $0.41 \pm 0.07\ \mu g\ g^{-1}$, which is almost two times lower than at low flow ($0.74 \pm 0.24\ \mu g\ g^{-1}$; 23 September 1986). These empirical results confirm the conclusions drawn from model-derived "observations" (i.e., sensitivity analyses). Simulation models can also be used in a predictive or forecasting mode, and this application is discussed next.

Predictive modeling—Having shown that the model was able to accurately simulate the observed behavior of dissolved and particulate selenium under a variety of environmental conditions, it can be used for predictive purposes. Because the behavior of selenium in the San Francisco Bay is largely controlled by river flow and refinery inputs (Cutter and Cutter 2004), two scenarios were examined: higher San Joaquin River flow and higher refinery discharges. For brevity, only the predicted total dissolved and particulate selenium results are discussed in this article.

Increasing San Joaquin River discharge—The State of California has a goal to "reduce the impacts of water diversion on the Bay-Delta system" and thus increase the discharge from the San Joaquin River into the bay (see http://www.baydeltawatershed.org/pdf/prog_plan.pdf). To evaluate the potential effects of increasing the San Joaquin River discharge, simulations were done using the full discharge of water at Vernalis (the freshwater end member of the San Joaquin, before the water is diverted for irrigation practices) rather than the current flow, which is lower because of withdrawals (see previous). In addition, if the flow from the San Joaquin River increased, the residence time of water within the Delta would substantially decrease. A decrease in the water residence time of the Delta would likely reduce the magnitude of the Delta

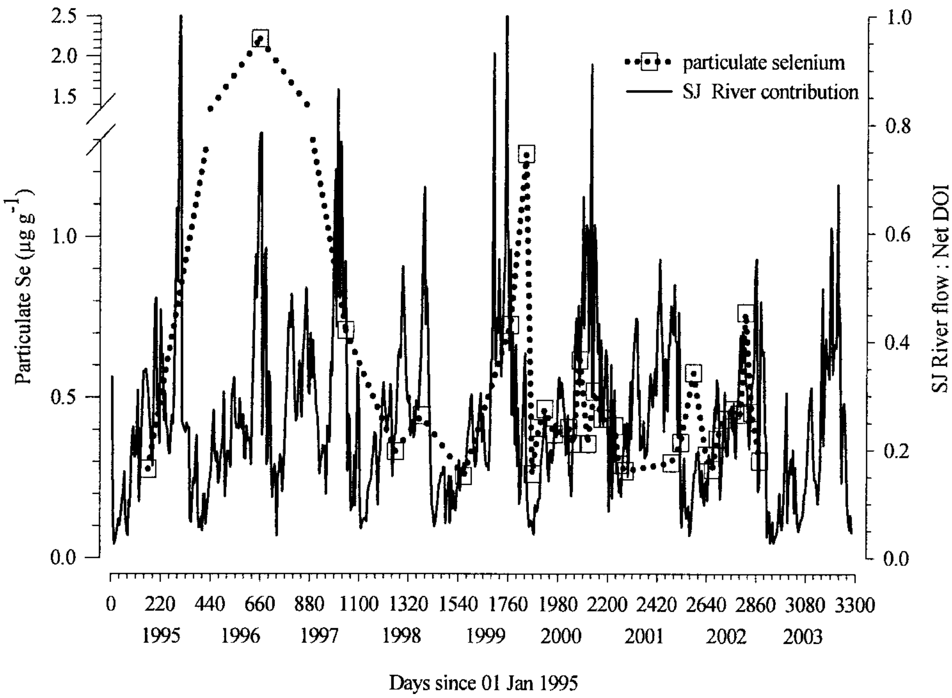


Fig. 9. Measured discharge from the San Joaquin River at Vernalis relative to the total net flow from the Delta (SJ River Flow: Net DOI) compared to particle associated selenium at a fixed site in Suisun Bay (38°1.89′N, 122°8.39′W) over a nine year period. The SJ River Flow: Net DOI total net flow (Net DOI) is a ratio of the San Joaquin river discharge to the net delta outflow where Net DOI is defined as the total freshwater discharge from the Delta.

removal constant described previously. Therefore, simulations were run for both overall high and low flow months (driven by the Sacramento River) under the following conditions: normal flow from the San Joaquin River (including withdrawals) as used in the validations, high San Joaquin River discharge (i.e., discharge from Vernalis with no withdrawals) with the Delta removal constant still turned on, and high San Joaquin River discharge with the Delta removal constant turned off (i.e., extremes of potential residence time effects).

During a high flow month (April) total dissolved selenium in the estuary displayed nonconservative behavior under current conditions, and increasing the flow from the San Joaquin River and removing the Delta removal constant of 60% did not change the general shape of the total dissolved selenium profile (Fig. 7A). However, under current conditions the model predicted a maximum of 1.8 nmol L^{-1} at a salinity of 5, whereas under high San Joaquin River flow the maximum increased to 2 nmol L^{-1} and increased up to 3.2 nmol L^{-1} when the Delta removal constant was also turned off (Fig. 7A). For a low flow simulation (November), a mid-estuary peak was observed, and as the flow from the San Joaquin River increased, the mid-estuary selenium concentration maximum increased and moved closer to the river end members (Fig. 7B). As with high flow conditions, the model predicted the highest concentrations of dissolved selenium (5.0 nmol L^{-1}) when the flow from Vernalis was increased and the Delta removal constant was turned off. This concentration is as high as those measured in the mid 1980s and 1990s (Cutter and Cutter 2004) when refineries were discharging their greatest selenium concentrations.

Total particulate selenium profiles in April (high flow) showed that increasing the flow from the San Joaquin resulted in a slight increase of total particulate selenium in the estuary (Fig. 7C), whereas turning off the Delta removal constant had no effect on total particulate selenium (Fig. 7C; lines overlap). However, increasing the flow in the San Joaquin River during a low flow month (November) resulted in a maximum total particulate selenium concentration of 0.35 nmol L^{-1} with the Delta removal constant on, which is greater than the maximum total particulate selenium observed under current conditions (0.25 nmol L^{-1}; Fig. 7D). Without the Delta removal constant, the total particulate selenium concentration maximum rose to 0.49 nmol L^{-1} (Fig. 7D).

The increased total particulate selenium could be caused by either an increase in sediment resuspension or in situ production of particles (i.e., via phytoplankton uptake). If the increase is due to sediment resuspension, the particle-associated selenium (μg g^{-1}) should be similar for each simulation. In April there is no difference between the estuarine averages of particle-associated selenium for the three scenarios (Table 4), thus suggesting that the predicted increase in total particulate selenium is largely caused by sediment resuspension. However, when the simulation was run for November (low flow), particle-associated selenium concentration increased as more San Joaquin River water reached the bay (Table 4). When the Delta removal constant was turned off, the particulate-associated selenium exceeded 1 μg g^{-1} over a salinity range of 5 to 20 (Fig. 8). Significantly, this concentration of particle-associated selenium is one that has a direct effect on consumer organisms, such as clams, and higher trophic levels in the estuarine food web (Luoma et al. 1992). These simulation results (Figs. 7, 8; Table 4) show that the concentration of particle-associated selenium varies with the different flow scenarios, suggesting that these changes are attributed to in situ production of particulate selenium (by phytoplankton) and not just sediment resuspension. Although the model has been validated under past and present conditions (i.e., excellent accuracy), these low flow (November) predictions are rather surprising. Essentially, the model predicts higher suspended particulate selenium in the upper estuary during higher San Joaquin discharge rates. In support of this prediction, field data from a fixed site in Suisun Bay across a 7-year, albeit discontinuous, period (Fig. 9) show a positive San Joaquin River flow-particulate selenium trend (r^2 = 0.633, n = 27), although the flows and concentrations are not always in phase.

Altered refinery inputs—In 1986, the refineries were discharging 99 mol d^{-1} of total selenium with 64% of the total as selenite (Cutter 1989), whereas in 1999 they were only discharging 38 mol d^{-1} with 13% as selenite, 57% as selenate, and 30% as organic selenide (Cutter and Cutter 2004). To examine how changing the refineries' discharges and speciation of selenium can affect selenium behavior (e.g., if they returned to higher discharge rates), three different flux rates using the current speciation (Cutter and Cutter 2004) were run: 38 mol d^{-1} as a reference (current conditions), 99 mol d^{-1} (old rate), and no discharge.

Under high flow (April) and low flow (November) conditions the predicted estuarine behaviors for total dissolved selenium were apparently nonconservative (Fig. 10A,B). During a high flow month, varying the refinery discharge from 0 mol d^{-1} to 99 mol d^{-1} resulted in the total dissolved selenium maxima ranging from 1.0 nmol L^{-1} to 2.2 nmol L^{-1} (Fig. 10A). In November (low flow month), total dissolved selenium concentrations increased to a maximum of 2.8 nmol L^{-1} as the refinery inputs increased (Fig. 10B). Under pristine conditions (no refinery inputs), total dissolved selenium had no mid-estuary maximum, compared to when the refineries were discharging selenium (Fig. 10B).

The model predicted no change in total particulate selenium within the estuary due to an increase/decrease in refinery inputs during a high flow month (Fig. 10C). However, for a low flow month (November) total particulate selenium increased when the refinery discharge was increased (Fig. 10D). Any increase in total particulate selenium must be caused by in situ production since the discharge from the refineries has no effect on the amount of sediment resuspended in the bay. The estuarine average particle-associated selenium increased from 0.43 μg g^{-1} under pristine conditions to 0.62 μg g^{-1} with the highest refinery inputs (Table 4). This is below the 1 μg g^{-1} of particle-associated selenium that has caused elevated concentrations of selenium in tissues of benthic consumers (Luoma et al. 1992; Luoma and Rainbow 2005), but is still above the typical background levels (0.2 μg g^{-1}; Doblin et al. 2006).

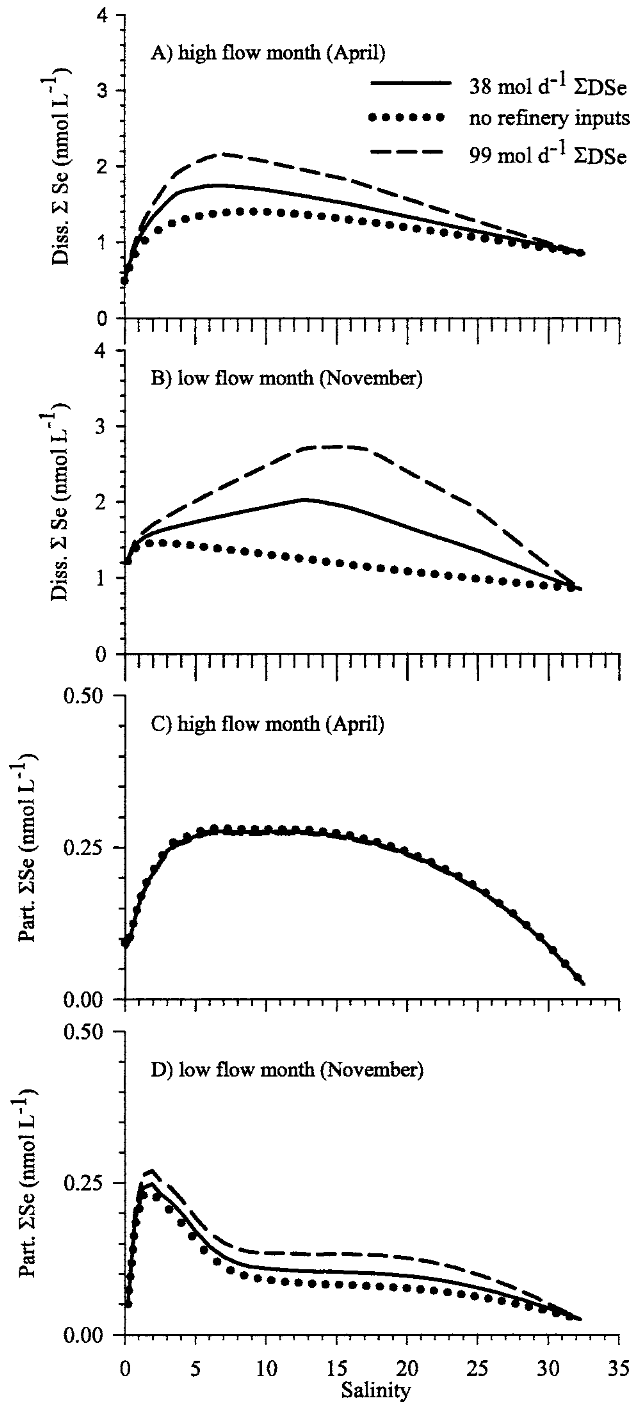


Fig. 10. (A) Model predictions for total dissolved selenium during a high flow month and (B) a low flow month and (C) total particulate selenium for a low flow month and (D) a high flow month in the San Francisco Bay under varying refinery discharge rates. The total discharge rates (38 mol total dissolved selenium d^{-1} and 99 mol total dissolved selenium d^{-1}) were from all the refineries in Fig. 2 with 13% of the total as selenite, 57% of the total as selenate, and 30% of the total as organic selenide.

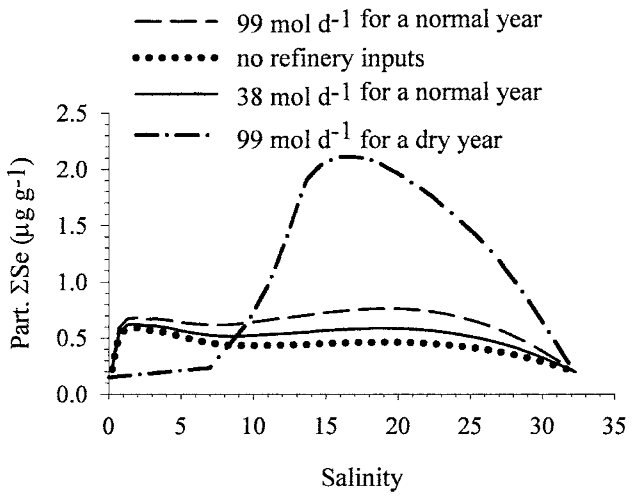


Fig. 11. Model predictions for particle-associated selenium in San Francisco Bay when selenium discharges from refineries are varied under normal river flow conditions and during a dry year. The total discharge rates (38 mol total dissolved selenium d^{-1} and 99 mol total dissolved selenium d^{-1}) were from all the refineries in Fig. 2 with 13% of the total as selenite, 57% of the total as selenate, and 30% of the total as organic selenide.

Although it might seem that refinery discharge has only a minor effect, it should be remembered that these simulations were run under a normal flow year (when summer discharge is $<400\ m^3\ s^{-1}$, but $>200\ m^3\ s^{-1}$). According to the hydrological classification of the San Francisco Bay estuary in the last 20 years (1983–2003), five have been dry years and five have been critical (i.e., drought) conditions. If the simulation were run using the flow from a critical year (1977) and the higher refinery inputs (99 mol d^{-1}), the model predicted particle-associated selenium could reach a maximum of 2.2 $\mu g\ g^{-1}$ (Fig. 11). This is greater than what would occur if the flow from the San Joaquin River was increased, indicating that the effects from the refineries are magnified depending on the freshwater residence time in the estuary (Cutter 1989; Cutter and San Diego-McGlone 1990).

The confidence associated with the ability of the model to predict future scenarios is dependent on how well known each parameter in the model is and the number of validation runs that can be done. The model results indicate that in situ processes (phytoplankton uptake of selenium and adsorption/desorption of selenite and selenate) need better parameterization. In particular, the uptake rates of different dissolved selenium forms by relevant estuarine phytoplankton species need to be quantified using Michaelis–Menton kinetics. As an example, recent studies within the Delta (Baines et al. 2004) found the biotic uptake rate of dissolved organic selenide may be much faster than previously thought.

The model simulations under various scenarios of San Joaquin River flow also show the need for studies of selenium reactivity as it is transported from Vernalis, through the Delta, to the bay. For example, sensitivity analysis found that varying the Delta removal constant by only 25% could result in an increase of dissolved selenate within the estuary by 51% and particulate organic selenium could increase by 16% within the estuary (Web Appendix 2). If selenium concentration measured by Cutter and Cutter (2004) at Vernalis were to be introduced to the estuary, the particle-associated selenium would be $>1\ \mu g\ g^{-1}$and have cascading effects through the estuarine food web (Luoma and Rainbow 2005). Finally, the model provides an excellent demonstration of the effect that reducing refinery inputs has on the concentrations of dissolved and particulate selenium under conditions seldom found in empirical studies (e.g., extremes of river flows; Cutter 1989; Cutter and Cutter 2004).

Although the model applications described here were specific for selenium and the San Francisco Bay, the steps for adapting ECoS 3 were not. For example, the processes depicted in Fig. 1 and incorporated into the model could be used to simulate the behavior of arsenic in San Francisco Bay by adjusting or eliminating some of the transformation and uptake rate constants, in addition to adjusting the concentrations of inputs; all of the physical processes are the same. The model also could be used for studying selenium in another well mixed estuary, for example Delaware Bay. In this application, the physical parameters such as those for tides and bathymetry would have to be adjusted, as well as the phytoplankton growth and grazing parameters, but not those for selenium. Regardless of the element or venue, the full integration of large data sets and biogeochemical models of estuaries such as San Francisco Bay enable an expanded understanding of estuarine processes and simulations of effects that could result from water/ecosystem management decisions.

References

Baines, S. B., N. S. Fisher, M. A. Doblin, and G. A. Cutter. 2001. Uptake of dissolved organic selenides by marine phytoplankton. Limnol. Oceanogr. **46:** 1936–1944.

———, ———, ———, ———, and L. S. Cutter. 2004. Light dependence of selenium uptake by phytoplankton and implications for predicting selenium incorporation into food webs. Limnol. Oceanogr. **49:** 566–578.

Belzile, N., and J. Lebel. 1988. Selenium profiles in the sediments of the Laurentian trough (Northwest North Atlantic). Chem. Geo. **68:** 99–103.

Cloern, J. E., and A. Alpine. 1991. *Potamocorbula amurensis*, a recently introduced Asian clam, has had dramatic effects on the phytoplankton biomass and production in northern San Francisco Bay. J. Shellfish Re. **10:** 258–259.

———, B. E. Cole, L. Raymond, J. Wong, and A. E. Alpine. 1985. Temporal dynamics of esturine phytoplankton: A case study of San Francisco Bay. Hydrobiologia **129:** 153–176.

Cole, B. E., and J. E. Cloern. 1984. Significance of biomass and light availability to phytoplankton productivity in San Francisco Bay. Mar. Ecolo. Prog. Ser. **17:** 15–24.

Cooke, T. D., and K. W. Bruland. 1987. Aquatic chemistry of selenium: Evidence of bimethylation. Environ. Sci. Technol. **21:** 1214–1219.

Cutter, G. A. 1989. The estuarine behaviour of selenium in San Francisco Bay. Estuar. Coast. Shelf Sci. **28:** 13–34.

———. 1992. Kinetic controls on metalloid speciation in seawater. Mar. Chem. **40:** 65–80.

———, AND K. W. BRULAND. 1984. The marine biogeochemistry of selenium in a simple system. Geochim. Cosmochim. Acta **48:** 1417–1433.

———, AND L. S. CUTTER. 2004. Selenium biogeochemistry in the San Francisco Bay estuary: Changes in water column behavior. Estuar. Coast. Shelf Sci. **6:** 463–476.

———, AND M. L. C. SAN DIEGO-MCGLONE. 1990. Temporal variability of selenium fluxes in the San Francisco Bay. Sci. Total. Environ. **97:** 235–250.

DOBLIN, M. A., S. BAINES, L. S. CUTTER, AND G. A. CUTTER. 2006. The biogeochemistry of selenium in San Francisco Bay estuary: seston and phytoplankton. Est. Coastal Shelf Sci **67:** 681–694.

DOBLIN, M. A., S. I. BLACKBURN, AND G. M. HALLEGRAEFF. 1999. Comparative study of selenium requirements of three phytoplankton species: *Gymnodinium catenatum*, *Alexandrium minutum* (Dinophyta) and *Chaetoceros* cf. *tenuissimus* (Bacillariophyta). J. Plankton Res. **21:** 1153–1169.

HARRIS, J. R. W., AND R. N. GORLEY. 1998. An introduction to modeling estuaries with ECoS 3. Plymouth Marine Laboratory.

LEHMAN, P. W. 2000. The influence of climate on phytoplankton community biomass in San Francisco Bay Estuary. Limnol. Oceanogr. **45:** 580–590.

LUCAS, L. V., J. E. CLOERN, J. R. KOSEFF, S. G. MONISMITH, AND K. J. THOMPSON. 1998. Does the Sverdrup critical depth model explain bloom dynamics in estuaries? J. Mar. Res. **56:** 375–415.

LUOMA, S. N., C. JOHNS, N. S. FISHER, N. A. STEINBERG, R. S. OREMLAND, AND J. R. REINFELDER. 1992. Determination of selenium bioavailability to a benthic bivalve from particulate and dissolved pathways. Environ. Sci. Technol. **26:** 485–491.

LUOMA, S. N., P. S. RAINBOW, 2005. Why is metal bioaccumulation so variable? Biodynamics as a unifying concept. Environ. Sci. Technol. **39:** 1921–1931.

MEASURES, C. I., R. E. MCDUFF, AND J. M. EDMONDS. 1980. Selenium redox chemistry at GEOSECS 1 re-occupation. Earth Planet Sci. Lett. **29:** 120.

MWANUZI, F., AND F. DE SMEDT. 1999. Heavy metal distribution model under estuarine mixing. Hydrol. Process. **13:** 789–804.

NYFFELER, U. P., Y. H. LI, AND P. H. SANTSHI. 1984. A kinetic approach to describe trace-element distribution between particles and solution in natural systems. Geochim. Cosmochim. Acta **48:** 1513–1522.

OHLENDORF, H. M., R. W. LOWE, P. R. KELLY, AND T. E. HARVEY. 1986. Selenium and heavy metals in San Francisco diving ducks. J. Wildl. Manage. **50:** 64–71.

OREMLAND, R. S., J. T. HOLLIBAUGH, M. A. MAEST, T. S. PRESSER, L. G. MILLER, AND C. W. CULBERTSON. 1989. Selenate reduction to elemental selenium, by anerobic bacteria in sediments and cultures: Biogeochemical significance of a novel, sulfate-independent respiration. Appl. Environ. Microbiol. **55:** 2333–2343.

PAUCOT, H., AND R. WOLLAST. 1997. Transport and transformation of trace metals in the Scheldt Estuary. Mar. Chem. **58:** 229–244.

PERRIN, C., C. MICHEL, AND V. ANDRÉASSIAN. 2001. Does a large number of parameters enhance model performance? Comparative assessment of common catchments model structures on 429 catchments. J. Hydrol. **242:** 275–301.

PLATT, T., AND A. D. JASSBY. 1976. The relationship between photosynthesis and light for natural assemblages of coastal marine phytoplankton. J. Phycol. **12:** 421–430.

PRESSER, T. S., AND D. Z. PIPER. 1998. Mass balance approach to selenium cycling through the San Joaquin Valley, sources to river to Bay, p. 153–182. *In* W. Grankenberger and R. A. Engberg [eds.], Environmental chemistry of selenium. Marcel Dekker.

RIEDEL, G. F., J. G. SANDERS, AND C. C. GILMOUR. 1996. Uptake, transformation, and impact of selenium in freshwater phytoplankton and bacterioplankton communities. Aquat. Microb. Ecol. **11:** 43–51.

STEWART, A. R., S. N. LUOMA, C. E. SCHLEKAT, M. A. DOBLIN, AND K. A. HIEB. 2004. Foodweb pathway determines how selenium affects aquatic ecosystems: A San Francisco Bay case study. Environ. Sci. Technol. **38:** 4519–4526.

TAKAYANAGI, K., AND D. COSSA. 1985. Speciation of dissolved selenium in the upper St. Lawrence Estuary, p. 275–284. *In* A. C. Siglo and A. Hatteri [eds.], Marine and estuarine geochemistry. Lewis Publishers.

THOMPSON, J. K. 2000. Two stories of phytoplankton control by bivalves in San Francisco Bay: The importance of spatial and temporal distribution of bivalves. J. Shellfish Res. **19:** 612.

VANDERMEULEN, J. H., AND A. FODA. 1988. Cycling of selenite and selenate in marine phytoplankton. Mar. Biol. **98:** 115–123.

VELINSKY, D. J., AND G. A. CUTTER. 1991. Geochemistry of selenium in a coastal salt marsh. Geochem. Cosmochim. Acta **55:** 179–191.

WERNER, I., AND J. T. HOLLIBAUGH. 1993. *Potamocorbula amurensis*: Comparison of clearance rates and assimilation efficiencies for phytoplankton and bacterioplankton. Limnol. Oceanogr. **38:** 949–964.

ZHANG, Y., AND J. N. MOORE. 1996. Selenium fractionation and speciation in a wetland sediment. Env. Sci. Technol. **30:** 2613–2619.

Received: 29 August 2005
Accepted: 6 March 2006
Amended: 10 April 2006

State of San Francisco Bay 2011
Appendix F

LIVING RESOURCES - Fish Indicators and Index
Technical Appendix

Prepared by Christina Swanson
July 2011

I. Background

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

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

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

II. Data Source

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

[1] Native species are those that have evolved in the Bay and/or adjacent coastal or upstream waters. Non-native species are those that have evolved in other geographically distant systems and have been subsequently transported to the Bay and established self-sustaining populations in the estuary.
[2] Information on the CDFG Bay Study is available at http://www.dfg.ca.gov/delta/projects.asp?ProjectID=BAYSTUDY

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

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

III. Indicator Evaluation

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

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

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

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

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

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

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

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

IV. Indicators

A. Fish Community Attributes

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

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

B. Abundance Indicators

1. Rationale

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

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

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

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

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

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

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

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

2. Methods and Calculations

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

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

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

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

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

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

> # fish/10,000 m^3 = [(# of fish)/(# of trawls x av. trawl volume, m^3)] x (10,000)
> (for Midwater trawl)
>
> # fish/10,000 m^2 = [(# of fish)/(# of trawls x av. trawl area, m^2)] x (10,000)
> (for Otter trawl)

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

3. Reference Conditions

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

4. Results

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

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

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

Based on the abundance of pelagic fishes, CCMP goals to "recover" and "reverse declines" of estuarine fishes have not been met.
Both current levels (expressed as the 2004-2008 average) and trends in pelagic fish abundance are below the 1980-1989 reference period for all sub-regions of the estuary (t-test or Mann-Whitney, $p<0.05$, all regions). However, in the most recent two years there is some evidence of increases in pelagic fish abundance in all sub-regions of the San Francisco Estuary except Suisun Bay.

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

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

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

Based on the abundance of northern anchovy, CCMP goals to "recover" and "reverse declines" of estuarine fishes have not been met in the upstream sub-regions of the estuary.
The abundance of northern anchovy, the most common fish in the San Francisco Estuary, has declined throughout the upstream regions of the estuary to levels that significantly below the 1980-1989 average reference conditions (t-test or Mann-Whitney, $p<0.05$ for San Pablo and Suisun Bays). In contrast, in Central and San Pablo Bays, recent northern anchovy abundance levels are comparable to levels measured in the 1980s (t-test or Mann-Whitney, $p>0.05$, both regions). As with demersal fishes, the markedly different trends between the upstream sub-regions (Suisun and San Pablo Bays) and downstream sub-regions (Central and South Bays) suggest that different environmental drivers are influencing northern anchovy in different sub-regions of the estuary: ocean conditions in the downstream sub-regions and watershed conditions, in particular hydrological conditions and planktonic food availability, in the upstream sub-regions.

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

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

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

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

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

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

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

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

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

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

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

C. Diversity Indicators

1. Rationale

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

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

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

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

2. Methods and Calculations

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

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

The **Estuary-dependent Fish Species Diversity** indicator was calculated for each year and for each of four sub-regions of the estuary as the number of estuary-dependent species collected (see Table 4), expressed as the percentage of the maximum number of estuary-dependent species ever collected in that sub-region, using catch data from the Bay Study Midwater and Otter Trawl surveys. The indicator was calculated as:

% of species assemblage =
(# estuary-dependent species/maximum # of estuary-dependent species reported) x 100

3. Reference Conditions:

For the two diversity indicators, the primary reference condition was based on the average diversity (expressed as % of the native fish assemblage present), measured for the first ten years of the Bay Study, 1980-1989, and for all four sub-regions combined. Diversity levels that were greater than the 1980-1989 average were considered to reflect “good” conditions. The average percentage of the native fish assemblage present during the 1980-1989 period diversity differed slightly among the four sub-regions for the Native Fish Species Diversity indicator (1980-1989 average: 49%; Suisun Bay diversity was lower than that in the other three sub-regions) and significantly for the Estuary-dependent Fish Species Diversity indicators (1980-1989 average: 72%; Suisun Bay was lowest and Central and South Bay were highest). This approach tended to reflect the relatively lower species diversity observed in Suisun Bay in the indicator results. Table 5 below shows the quantitative reference conditions that were used to evaluate the results of the two diversity indicators.

4. Results

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

Maximum native species diversity differs among the four sub-regions of the estuary.
The greatest numbers of native fish species are found in Central Bay (94 species) and the fewest are in Suisun Bay (48 species). A maximum of 73 native species have been collected in South Bay and 66 native species have been found in San Pablo Bay.

The percentage of the native fish species assemblage present differs among the sub-regions.
In addition to having a smaller native fish species assemblage, Suisun Bay has a significantly lower percentage (44%) of that assemblage present each year compared to all other sub-regions (48% in Central Bay; 49% in South Bay and 51% in San Pablo Bay) (ANOVA: $p<0.001$, all pairwise comparisons: $p<0.01$). In recent years (2004-2008), native fish diversity has been highest in Central Bay (ANOVA: $p<0.05$ for Central Bay compared to Suisun Bay).

Trends in native species diversity differ among the sub-regions.
Native species diversity has increased significantly in Central Bay (regression: $p<0.01$) with an average of six more species in the most recent five-year period compared to the 1980-1989 reference period. Native fish species diversity decreased significantly in San Pablo Bay (regression: $p=0.05$), with an average of four fewer species in the 2005-2008 period compared to the 1980-1989 period. Native fish species diversity fluctuated in both South and Suisun bays.

Based on the diversity of the native fish community, CCMP goals to “recover” and “reverse declines” of estuarine fishes have been met in all sub-regions of the estuary.
Comparison of average native fish species diversity in the most recent five years (2004-2008) to that measured during the 1980-1989 period shows no significant differences except for Central Bay, where diversity is significantly higher (t-test: $p<0.05$).

Results of the **Estuary-dependent Fish Species Diversity** indicator are shown in Figure 8.

The diversity of estuary-dependent species is lower in Suisun Bay than in other sub-regions of the estuary.

Although roughly the same number of estuary-dependent species are found in each sub-region (38 species in San Pablo Bay; 36 species in Central and South Bays; and 31 species in Suisun Bay), a significantly smaller percentage of the estuary-dependent fish assemblage occurs in Suisun Bay (49% of the assemblage) than in all other regions of the San Francisco Estuary (84% in Central Bay; 80% in South Bay; and 69% in San Pablo Bay) (ANOVA: $p<0.001$, all pairwise comparisons, $p<0.05$).

Diversity of Bay-dependent species is generally stable in most sub-regions of the estuary.
Estuary-dependent species diversity has declined slightly in San Pablo Bay (regression: $p<0.05$, for a decrease of 2 species from the 1980-1989 period to the 2004-2008 period) and South Bay (regression: $p<0.05$, for an average decrease of 1.5 species). In all other regions, estuary-dependent diversity has fluctuated but remained relatively stable over the 29-year period.

Based on the diversity of the estuary-dependent fish community, CCMP goals to "recover" and "reverse declines" of estuarine fishes have been met in all sub-regions of the estuary except South Bay.
Comparison of average estuary-dependent fish species diversity in the most recent five years (2004-2008) to that measured during the 1980-1989 period shows no significant differences, except for South Bay, where diversity of estuary-dependent fishes was significantly lower (Mann-Whitney Rank Sum test: $p<0.05$).

D. Species Composition Indicators

1. Rationale

The relative proportions of native and non-native species found in an ecosystem is an important indicator of ecosystem health (May and Brown, 2002; Meador et al., 2003). Non-native species are most prevalent in ecosystems that have been modified or degraded with resultant changes in environmental conditions (e.g., elevated temperature, reduced flood frequency), pollution, or reduction in area or access to key habitats (e.g., tidal marsh, seasonal floodplain). The San Francisco Estuary has been invaded by a number of non-native fish species. Some species, such as striped bass, were intentionally introduced into the estuary; others have arrived in ballast water or from upstream habitats, usually reservoirs.

The Fish Index includes two different indicators for species composition. The **Percent Native Species** indicator uses Midwater and Otter trawl survey data to measure what percentage of the fish species collected in each sub-region of the estuary are native species. The **Percent Native Fish** uses the survey data to measure what percentage of the individual fish collected in each sub-region of the estuary are native species.

2. Methods and Calculations

The **Percent Native Species** indicator was calculated for each year and for each of four sub-regions of the estuary as the percentage of fish species collected in the estuary that are native to the estuary and its adjacent ocean and upstream habitats using the equation below.

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

The Percent Native Fish indicator was calculated for each year and for each of four sub-regions of the estuary as the percentage of fish collected in the estuary that are native to the estuary and its adjacent ocean and upstream habitats using the equation below.

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

3. Reference Conditions:

There is an extensive scientific literature on the relationship between the presence and abundance of non-native species and ecosystem conditions and the length of the available data record for the San Francisco Estuary allows for establishment of "reference conditions". In general, ecosystems with high proportions of non-natives (e.g., >50%) are considered to be seriously degraded. Furthermore, non-native fish species have been present in the San Francisco Estuary Bay for more than 100 years; therefore, 100% native fish species is unrealistic. Among the four sub-regions, the 1980-1989 average percentage of native species was 87% and the average percentage of native fish was 90%. For both indicators, Suisun Bay values were lowest. Based on this information, the primary reference condition for both indicators was established at 85%. Percent Native Species levels that were greater than this value were considered to reflect "good" conditions. Table 6 below shows the quantitative reference conditions that were used to evaluate the results of the two species composition indicators.

4. Results

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

The percentage of native species in the fish community differs among the four sub-regions of the estuary.
For the past 29 years, non-native species have been most prevalent in Suisun Bay, where in most years less than 75% of species are natives, intermediate in South and San Pablo Bays (88% and 86% native, respectively), and the least prevalent in Central Bay (92%) (Kruskal Wallis One-way ANOVA of Ranks: $p<0.001$, all pairwise comparisons: $p<0.05$).

Trends in the percentage of native species differ among the sub-regions.
The percentage of native species is declining in all sub-region of the estuary except Central Bay. In San Pablo Bay, the percent native species declined significantly (regression: $p<0.001$) from 90% in the 1980-1989 period to 81% in the most recent five-year period. Percent native species declined in Suisun Bay from 77% to 69% (regression: $p<0.01$) and in South Bay the percentage of native species declined from 89% to 85% (regression: $p<0.05$).

Trends in the percentage of native species in Bay fish assemblages are driven by declines in the numbers of native species and increases in non-native species.
During the past 29 years, the number of native species in San Pablo Bay declined by three species and the number of non-native species increased by three, to an average of seven non-

native species of the 2004-2008 period. The number of non-native species collected in Suisun Bay increased by an average of three species, from six species in the 1980-1989 period to nine species in the most recent five years. In South Bay, native species declined by one and non-natives increased by one. In Central, the total number of native species collected increased by six species.

Based on fish species composition, CCMP goals to "recover" and "reverse declines" of estuarine fishes have not been met in Suisun and San Pablo Bays.
Compared to the 1980-1989 period and the biologically based 85% native species reference condition, recent measurements (2004-2008) of the fish species composition indicate significantly poorer condition for San Pablo Bay (Mann-Whitney Rank Sum test: $p<0.01$) and Suisun Bay (t-test: $p<0.01$). Although both a long-term (1980-2008) and recent (2004-2008) decline were evident in South Bay, the average percentage of native species for the most recent five year period was not significantly different than that for the 1980-1989 reference period.

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

The percentage of native fish in the fish community differs among the four sub-regions of the estuary.
For the past 29 years, non-native fish have dominated the Suisun Bay sub-region, where in most years less than 50% of fish collected are natives (1980-2008 average: 49%). Non-native fish are rare in the other three sub-regions. Central Bay has the least (1980-2008 average: 0.1%), South Bay has just 1% non-native fishes and San Pablo Bay less than 3% non-native fishes (Kruskal Wallis One-way ANOVA of Ranks: $p<0.001$, all pairwise comparisons: $p<0.05$).

Trends in the percentage of native fish differ among the sub-regions.
The percentage of native fishes is declining in the Suisun and South Bay sub-region of the estuary but not in Central or San Pablo Bays. In Suisun Bay, the percent native fish declined significantly (regression: $p<0.001$) from 63% in the 1980-1989 period to just 37% in the most recent five-year period. Percent native fish declined in South Bay from more than 99% to 96%% (regression: $p<0.01$). The increases in the numbers of non-native fish in South Bay in 2007 and 2008 were largely attributable to higher catches of two non-natives, striped bass and chameleon goby.

Based on fish species composition, CCMP goals to "recover" and "reverse declines" of estuarine fishes have been met in all sub-regions of the estuary except Suisun Bay.
In all regions of the estuary except Suisun Bay, native fish comprise the vast majority of the fish community, exceeding 95% of the total fish present in nearly all years. In Suisun Bay, the percentage of the fish community that is comprised of non-native fish is extremely high and increasing, indicating that the condition of this region of the estuary is poor and deteriorating.

E. Distribution Indicators

1. Rationale

The distribution of native fishes within a habitat is an important indicator of ecosystem condition (May and Brown, 2002; Whitfield and Elliott, 2002; Nobriga et al., 2005). Native fishes may be excluded or less abundant in degraded habitats with unsuitable environmental conditions and/or those in which more tolerant non-native species have become established. The Fish Index includes two indicators to assess the distribution of native fishes within the estuary. The **Pelagic Fish Distribution** indicator uses Midwater trawl survey data to measure the percentage of the survey's sampling stations at which native species were regularly collected. The **Demersal Fish Distribution** indicator uses Otter trawl survey data to make a similar measurement for bottom-oriented native fishes.

5. Methods and Calculations

The **Pelagic Fish Distribution** indicator was calculated for each year and for each of four sub-regions of the estuary as the percentage of Midwater trawl survey stations at which at least one native fish was collected in at least 60% of the surveys conducted in that year.

Pelagic Fish Distribution =
(# survey stations with native fish in 60% of surveys)/(# survey stations sampled) x 100

The **Demersal Fish Distribution** indicator was calculated identically using Otter trawl survey data.

6. Reference Conditions:

There is an extensive scientific literature on the relationship between the presence and abundance of non-native species and ecosystem conditions. The length of the available data record for the San Francisco Estuary allows for establishment of "reference conditions". For the two Distribution indicators, the primary reference condition was established based on the number of stations sampled by the Bay Study surveys (8-12 stations per sub-region; therefore the maximum resolution of this indicator is limited to 8-13% increments depending on sub-region) and the average percentage of stations with native species present for the first ten years of the Bay Study, 1980-1989 (~96%). Distribution levels that were greater than the reference condition were considered to reflect "good" conditions. Table 7 below shows the quantitative reference conditions that were used to evaluate distribution indicators.

7. Results

Results of the **Pelagic Fish Distribution** indicator are shown in Figure 11.

The percentage of Midwater trawl survey stations that regularly have native fish differs among the four sub-regions of the estuary.
For the past 29 years, native fish have been consistently present at nearly all Midwater trawl survey stations in all sub-regions of the estuary except Suisun Bay. During the 1980-2008 period, native fish were present at 98-99% of survey stations in South, Central and San Pablo Bays. In contrast, native fish were present in only an average of 81% stations in Suisun Bay (Kruskal Wallis One-way ANOVA of Ranks: $p<0.001$, Suisun v all other sub-regions; $p<0.05$).

Trends in the percentage of native fish differ among the sub-regions.
The percentage of survey stations with native fish was stable in all sub-regions of the estuary except Suisun Bay. In Suisun Bay, distribution of native fishes declined significantly from 88% of stations (1980-1989) to 63% in the most recent five years (2004-2008) (Mann-Whitney Rank Sum test; $p<0.01$; regression: $p<0.05$). This decline in distribution occurred abruptly in 2003 and is largely drive by low distribution in 2005, when native fish were collected in only five of 12 stations (42%). Prior to 2003, distribution of native pelagic fish in Suisun Bay was generally stable at 86% of stations (1980-2002 average) but since 2003 native pelagic fish were present at only 63% of Suisun Bay stations (2003-2008 average). Native fish were most frequently absent from survey stations located in the lower San Joaquin River and the western region of Suisun Bay.

Based on native pelagic fish distribution, CCMP goals to "recover" and "reverse declines" of estuarine fishes have been met in all sub-regions of the estuary except Suisun Bay.
In all regions of the estuary except Suisun Bay, native pelagic fish are regularly collected at all Midwater trawl survey stations. In contrast, native fish are increasingly absent from the western region of Suisun Bay, the most upstream region of the estuary, suggesting that the condition of this region of the estuary is deteriorating.

Results of the **Demersal Fish Distribution** indicator are shown in Figure 12.)

The percentage of Otter trawl survey stations that regularly have native fish differs among the four sub-regions of the estuary.
For the past 29 years, native fish have been consistently present at nearly all Otter trawl survey stations in all sub-regions of the estuary except Suisun Bay. During the 1980-2008 period, native fish were present at 98-100% of survey stations in South, Central and San Pablo Bays. In contrast, native fish were present in only an average of 81% stations in Suisun Bay (Kruskal Wallis One-way ANOVA of Ranks: $p<0.001$, Suisun v all other sub-regions; $p<0.05$).

Trends in the percentage of native fish differ among the sub-regions.
The percentage of survey stations with native fish was stable in all sub-regions of the estuary except Suisun Bay. In Suisun Bay, distribution of native fishes declined briefly but significantly in the early 1990s, from 91% of stations (1980-1991) to just 64% of stations (1992-1994), and then recovered to 89% (1995-2000). In 2001, distribution declined significantly again, falling to 62% of stations (2001-2007) before returning to 91% in 2008 (Mann-Whitney Rank Sum test; $p<0.05$ both tests). For the most recent five years (2004-2008), native demersal fish have been present at 62% of stations. Similar to pelagic fish, native demersal fish were most frequently absent from survey stations located in the western region of Suisun Bay.

Based on native demersal fish distribution, CCMP goals to "recover" and "reverse declines" of estuarine fishes have been met in all sub-regions of the estuary except Suisun Bay.
In all regions of the estuary except Suisun Bay, native demersal fish are regularly collected at all Otter trawl survey stations. In contrast, native fish are increasingly absent from the western region of Suisun Bay, the most upstream region of the estuary, suggesting that the condition of this region of the estuary is deteriorating.

V. Fish Index

The Fish Index aggregates the results of the four abundance indicators (Pelagic Species, Demersal Species, Northern Anchovy, and Sensitive Species), two diversity indicators (Native Species and Estuary-dependent Species), two species composition indicators (Percent Native Species and Percent Native Fish) and the two distribution indicators (Pelagic Fish and Demersal Fish Distribution).

A. Index Calculation

For each year and for each sub-region, the Fish Index is calculated by combining the results of the ten indicators into a single number. First, results of the indicators in each fish community attribute (i.e., abundance, diversity, species composition and distribution) were combined by averaging the quantitative scores of each of the component indicators. Within the fish community attribute, each indicator was equally weighted. Next the average scores for each fish community attribute were combined by averaging, with each fish community attribute equally weighted. An index score greater than 3 was interpreted to represent “good” conditions and an index score less than 1 was interpreted to represent “very poor” conditions.

B. Results

Results of the Fish Index for each sub-region~~s~~ are shown in Figure 13.

The Fish Index differs among the four sub-regions of the estuary.
For the 29-year survey period, the Fish Index was highest in the Central Bay (1980-2008 average: 3.14), lowest in Suisun Bay (1.77), and intermediate in South and San Pablo Bays (3.01 and 2.78, respectively) (Kruskal Wallis One-way ANOVA of Ranks: $p<0.05$; Central>South and San Pablo>Suisun). For the most recent five years, the differences among the regions are even greater. The Fish Index was highest in Central (2004-2008 average: 3.025), lowest in Suisun (1.28) and intermediate in South and San Pablo Bays (2.84 and 2.56, respectively). Lower Fish Index values for Suisun Bay at the beginning of the survey period were attributable to lower diversity (i.e., smaller percentages of the sub-region’s species assemblage were present) and species composition (i.e., high prevalence of non-native species and non-native fish).

Trends in the Fish Index differ among the sub-regions.
During the 29-year survey period, the Fish Index has declined significantly in Suisun, San Pablo and South Bays but not in Central Bay (regression 1980-2008: $p<0.005$ all sub-regions except Central Bay). The overall condition of the fish community in Suisun Bay has declined from “fair” in the early 1980s (1980-1989 average: 2.21) to consistent “poor” conditions throughout the 1990s and 2000s. In 2006, when diversity, species composition and distribution all dropped, condition of the fish community in Suisun Bay was “very poor.” In San Pablo Bay, the Fish Index has declined steadily, from mostly “good” conditions in the early 1980s to “fair” conditions by the 1990s: since then, the San Pablo Bay Fish Index has not fallen to “poor” levels and has continued to decline. The decline in the Fish Index in South Bay, while significant, is

not as severe. In Central Bay, the Fish Index has been relatively stable with generally "good" fish community conditions.

Based on Fish Index, CCMP goals to "recover" and "reverse declines" of estuarine fishes have been met in only the Central Bay sub-region.
The overall condition of the fish community is "good" in Central Bay, the most downstream region of the San Francisco Estuary. In all other sub-regions of the estuary, the condition of fish community is declining. In Suisun Bay, the most upstream region of the estuary most directly affected by watershed degradation, alteration of freshwater inflows and declines in the quality and quantity of low-salinity habitat, the fish community is in "poor" condition. These declines in the Fish Index are largely driven by declines in fish abundance (all three sub-regions), declining diversity (South and San Pablo Bays), increasing prevalence of non-native species (all three sub-regions), and declines in the distribution of native fish within the sub-region (Suisun Bay).

C. Summary and Conclusions

Collectively, the ten indicators and the Fish Index provide a reasonably comprehensive assessment of status and trends San Francisco Estuary fish community. The results show substantial geographic variation in both the composition and condition of the fish community within the estuary and in the response of specific indicators over time. Table 8 below summarizes the indicator and Index results by sub-region. In addition, the following general conclusions can be made:

1. The San Francisco Estuary fish community differs geographically within the estuary in fish community composition, fish abundance, and trends in various attributes of its condition over time.
2. Different indicators show different responses over time, some demonstrating clear declines in condition over time, others no change, and a few increases. In some cases, the same indicators measured in different sub-regions of the estuary show different responses over time. These results suggest that different physical, chemical or biological environmental variables (or combinations of these variables) influence the fish community response in different sub-regions.
3. Overall condition, as measured individually by the fish indicators and by the Fish Index for the community response, is poorest in upstream region of estuary, Suisun Bay; best in Central Bay, the region most strongly influenced by ocean conditions and with a predominantly marine fish fauna; and intermediate in San Pablo and South Bays. However, condition of the fish community in San Pablo and South Bays is declining and, for San Pablo Bay, could deteriorate to "poor" condition if the current rate of decline continues for the next two decades.
4. Even 30 years ago, the condition of the fish community in Suisun Bay was poorer than in all other sub-regions of the estuary. The fish community was less diverse with relatively lower percentages of the native fish assemblage present, and dominated by high percentages of non-native species.
4. The abundance of pelagic fishes in the estuary (which include Northern anchovy and most of the sensitive species measured in those two indicators) has shown the greatest changes over time, indicating this component of the fish community has low resilience and/or is tightly linked to just one or a few environmental drivers that have also experienced substantial change in conditions during the sampling period.

VI. Peer Review

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

VII. References

CDFG (2002) Pacific herring commercial fishing regulations. Final Supplemental Environmental Document. California Department of Fish and Game, SCH No. 98052052.

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

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

Harrison, T. D. and A. K. Whitfield (2004) A multi-metric fish index to assess the environmental condition of estuaries. J. Fish Biology. 65:283-710.

Nobriga, M., F. Feyrer, R. Baxter, and M. Chotkowski. 2005. Fish community ecology in an altered river delta: spatial patterns in species composition, life history strategies, and biomass. Estuaries 28:776-785.

Jassby, A. D., W. J. Kimmerer, S. G. Monismith, C. Armor, J. E. Cloern, T. M. Powell, J. R. Schubel, and T. J. Vendlinski (1995) Isohaline position as a habitat indicator for estuarine populations. Ecol. Appl. 5:272-280.

Karr, J. R. (1981). Assessment of biotic integrity using fish communities. Fisheries 6, 21–27.

Karr, J. R., J. D. Allan, and A. C. Benke (2000) River conservation in the United States and Canada: science, policy and practice. In: P. J. Boon, B. R. Davies, and G. E. Petts (eds.), *River Conservation: Science Policy and Practice*. J. Wiley & Sons, New York, pp. 502-528.

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

May, J. T. and L. R. Brown (2002) Fish communities of the Sacramento River Basin: implications for conservation of native fishes in the Central Valley, California. Env. Biol. Fish. 63:373-388.

Meador, M. R., L. R. Brown, and T. Short (2003) Relations between introduced fish and environmental conditions at large geographic scales. Ecological Indicators 3:81-92.

Moyle, P. B. (2002) Inland Fishes of California. University of California Press, Berkeley. 502 pp.

SFEIT (2008) Assessment Framework as a tool for integrating and communication watershed health conditions for the San Francisco Estuary. Technical Report #1, submitted by the San Francisco Estuary Indicators Team to the California Department of Water Resources, September 30, 2008.

Wang, L. and J. Lyons (2003) Fish and benthic macroinvertebrate assemblages as indicators of stream degradation in urbanizing watersheds. In *Biological Response Signatures. Indicator Patterns Using Aquatic Communities*, (ed. T. P. Simon), pp. 227-249. CRC Press: New York.

Whitfield, A. K. & Elliott, M. (2002). Fishes as indicators of environmental and ecological changes within estuaries: a review of progress and some suggestions for the future. Journal of Fish Biology 61 (Suppl. A), 229–250. doi: 10.1006/jfbi.2002.2079.

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

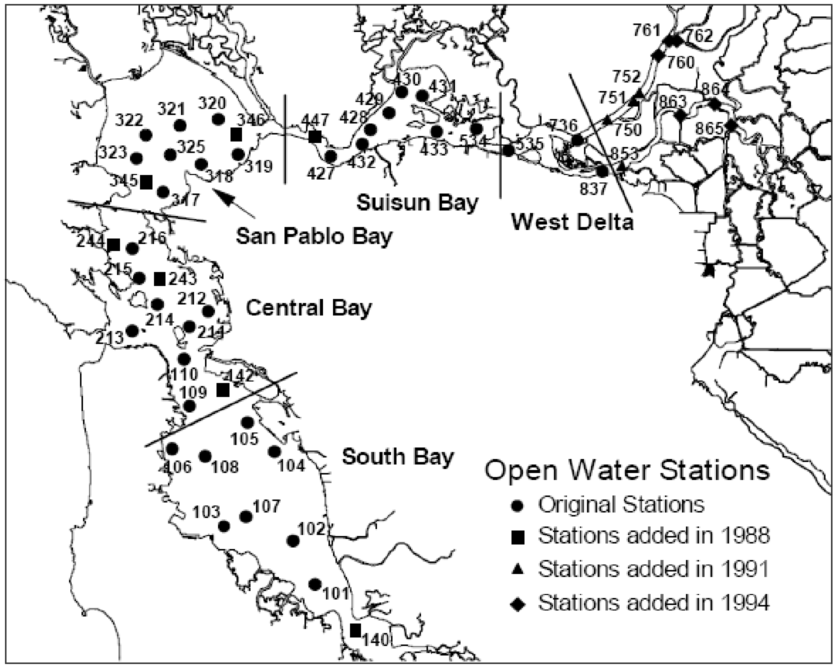


Figure 2. Locations of the sampling stations for the CDFG Bay Study Midwater Trawl and Otter Trawl surveys in different sub-regions of the San Francisco Bay. For the 2007 Fish Index, only data from the "original stations" (sampled continuously for 1980-2006 period) were used to calculated indicators for four sub-regions: South Bay, Central Bay, San Pablo Bay, and Suisun Bay (which for this study includes the West Delta sub-region).

Table 1. Sampling stations and total numbers of surveys conducted per year (range for the 1980-2006 period, excludes 1994) by the CDFG Bay Study Survey in each of four sub-regions of San Francisco Bay. MWT=Midwater Trawl survey; OT= Otter Trawl survey. See Figure 1 for station locations.

Sub-region	Sampling stations	Number of surveys (range for 1980-2005 period)
South Bay	101, 102, 103, 104, 105, 106, 107, and 108	64-96 (MWT) 64-96 (OT)
Central Bay	109, 110, 211, 212, 213, 214, 215, and 216	64-96 (MWT) 64-96 (OT)
San Pablo Bay	317, 318, 319, 320, 321, 322, 323, and 325	64-96 (MWT) 64-96 (OT)
Suisun Bay (includes West Delta sub-region shown in Figure 1)	425, 427, 428, 429, 430, 431, 432, 433, 534, 535, 736, and 837	87-132 (MWT) 88-132 (OT)

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

Fish Community Characteristic	Indicators
Abundance	• Pelagic Fish Abundance • Northern Anchovy Abundance • Demersal fish Abundance • Sensitive Species Abundance
Diversity	• Native Fish Diversity • Estuary-dependent Fish Diversity
Species Composition	• Percent Native Species • Percent Native Fish
Distribution	• Pelagic Fish Distribution • Demersal Fish Distribution

Table 3. Quantitative reference conditions and associated interpretations for the results of the fish abundance indicators. The primary reference condition, which corresponds to “good” conditions, is in bold.

Abundance Indicators (Pelagic Fish, Northern Anchovy, Demersal Fish, Sensitive Species)		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>150% of 1980-1989 average	“Excellent”	4
>100% of 1980-1989 average	**“Good”**	**3**
>50% of 1980-1989 average	“Fair”	2
>15% of 1980-1989 average	“Poor”	1
≤15% of 1980-1989 average	“Very Poor”	0

Figure 3. Changes in the Pelagic Fish Abundance indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition (1980-1989 average).

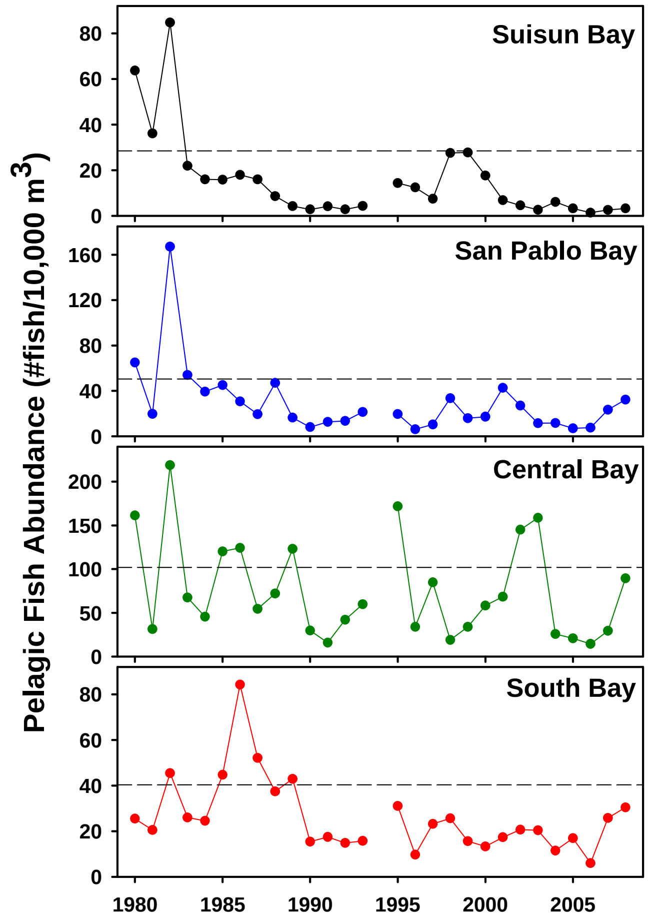


Figure 4. Changes in the Northern Anchovy Abundance indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition (1980-1989 average).

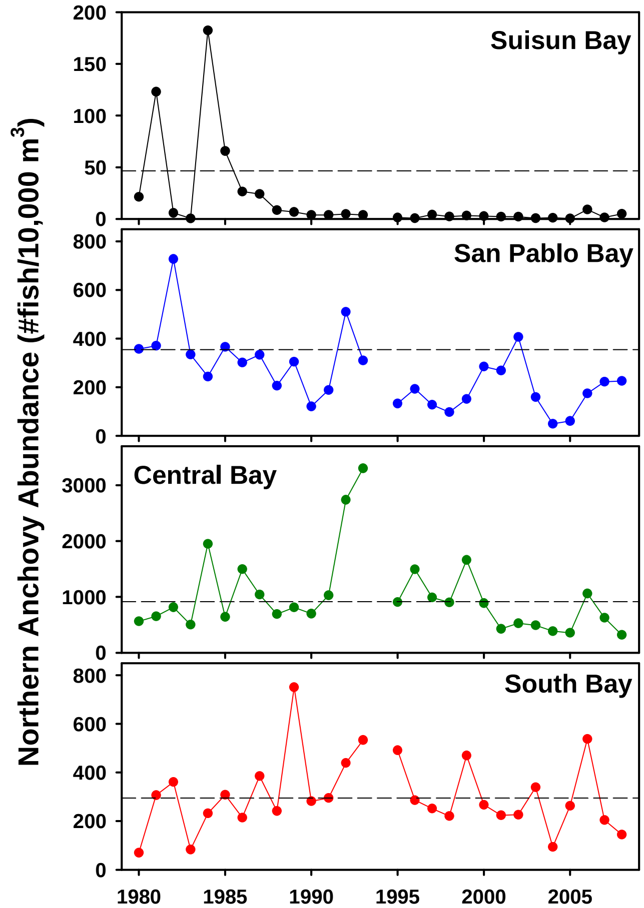


Figure 5. Changes in the Demersal Fish Abundance indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition (1980-1989 average).

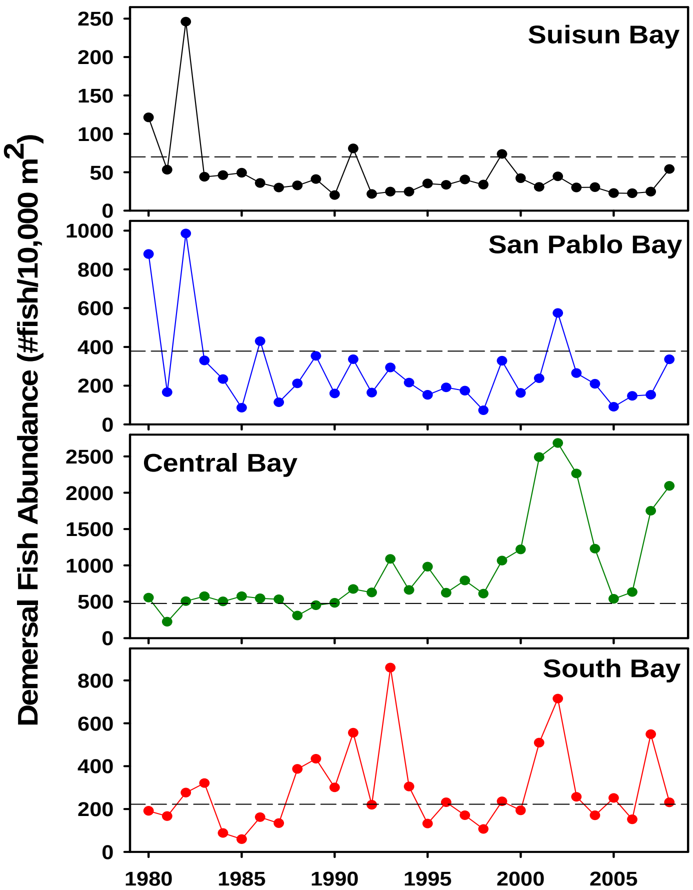


Figure 6. Changes in the Sensitive Fish Species Abundance indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition (1980-1989 average).

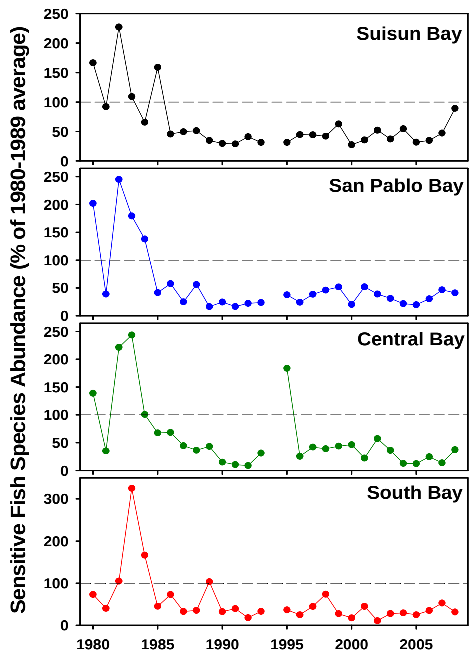


Table 4. San Francisco Estuary-dependent fish species collected in the CDFG Bay Study Midwater Trawl and Otter Trawl surveys.

Estuary-dependent fish species (common names)	
Estuary resident species Species with resident populations in the estuary and/or estuary-obligate species that use the estuary as nursery habitat	**Seasonal species** Species regularly use the estuary for part of their life cycle but also have substantial connected populations outside the estuary
Arrow goby	Barred surfperch
Bat ray	Black perch
Bay goby	Bonehead sculpin
Bay pipefish	California halibut
Brown rockfish	California tonguefish
Brown smoothhound	Diamond turbot
Cheekspot goby	English sole
Delta smelt	Northern anchovy
Dwarf surfperch	Pacific sandab
Jack smelt	Pacific tomcod
Leopard shark	Plainfin midshipman
Longfin smelt	Sand sole
Pacific herring	Speckled sanddab
Pacific staghorn sculpin	Spiny dogfish
Pile perch	Splittail
Shiner perch	Starry flounder
Threespine stickleback	Surfsmelt
Topsmelt,	Walleye surfperch
Tule perch	
White croaker	
White surfperch	

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

Diversity Indicators		
Native Fish Species Diversity		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>60%	"Excellent"	4
>50% (~1980-1989 average)	**"Good"**	**3**
>40%	"Fair"	2
>30%	"Poor"	1
≤30%	"Very Poor"	0
Estuary-dependent Fish Species Diversity		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>85%	"Excellent"	4
>70% (~1980-1989 average)	**"Good"**	**3**
>55%	"Fair"	2
>40%	"Poor"	1
≤40%	"Very Poor"	0

Figure 7. Changes in the Native Fish Species Diversity indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition.

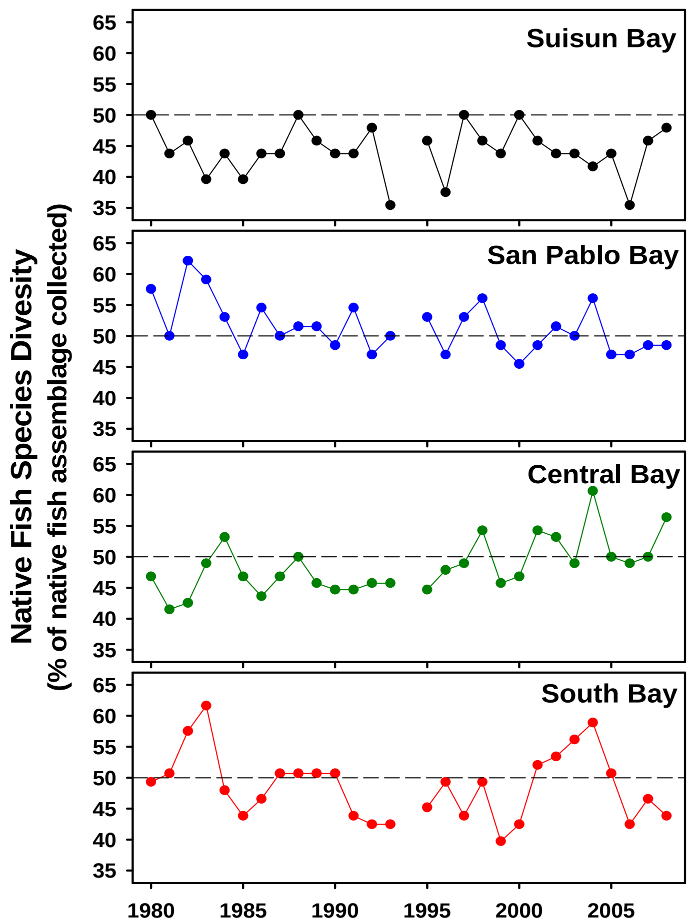


Figure 8. Changes in the Estuary-dependent Fish Species Diversity indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition (1989-1989 average).

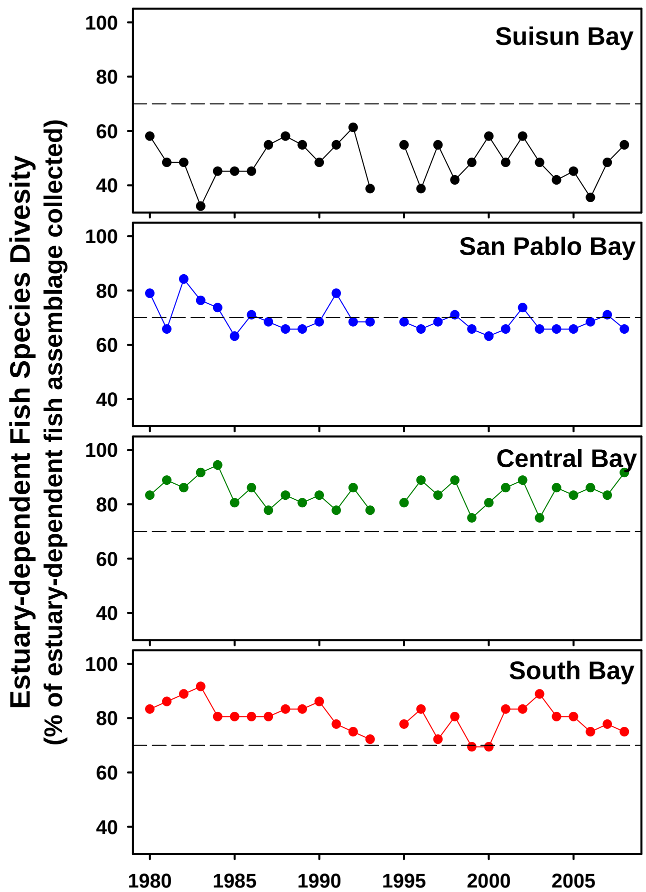


Table 6. Quantitative reference conditions and associated interpretations for the results of the species composition indicators. The primary reference condition, which corresponds to “good” conditions, is in bold.

Species Composition Indicators (Percent Native Species, Percent Native Fish)		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
>95%	“Excellent”	4
>85% (~1980-1989 average)	**“Good”**	**3**
>70%	“Fair”	2
>50%	“Poor”	1
≤50%	“Very Poor”	0

Figure 9. Changes in the Percent Native Species indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition.

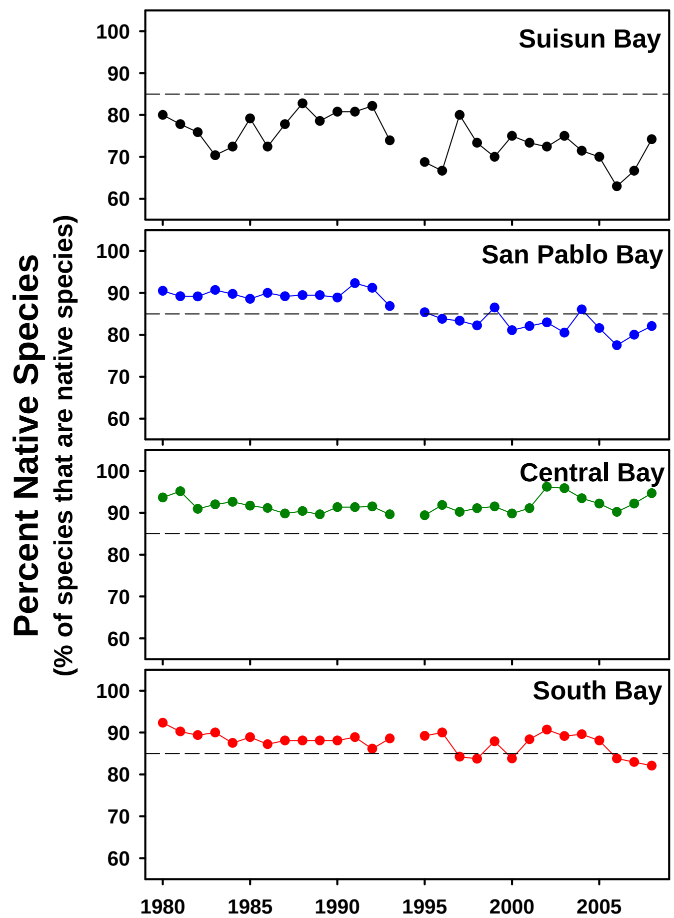


Figure 10. Changes in the Percent Native Fish indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition.

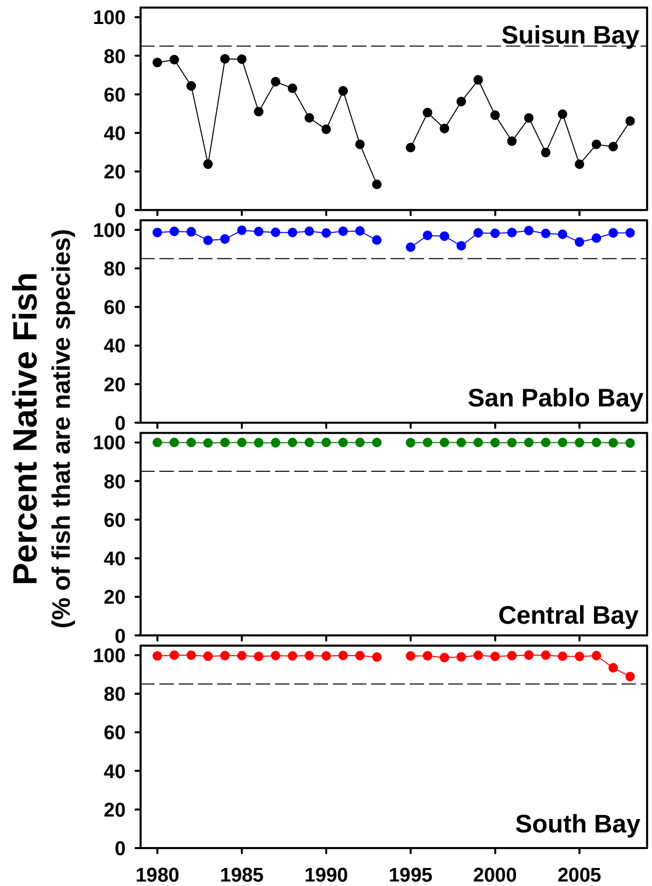


Table 7. Quantitative reference conditions and associated interpretations for the results of the distribution indicators. The primary reference condition, which corresponds to "good" conditions, is in bold.

Distribution Indicators (Pelagic Fish, Demersal Fish)		
Quantitative Reference Condition	**Evaluation and Interpretation**	**Score**
100%	"Excellent"	4
>80% (~1980-1989 average)	**"Good"**	**3**
>60%	"Fair"	2
>40%	"Poor"	1
≤40%	"Very Poor"	0

Figure 11. Changes in the Pelagic Fish Distribution indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition.

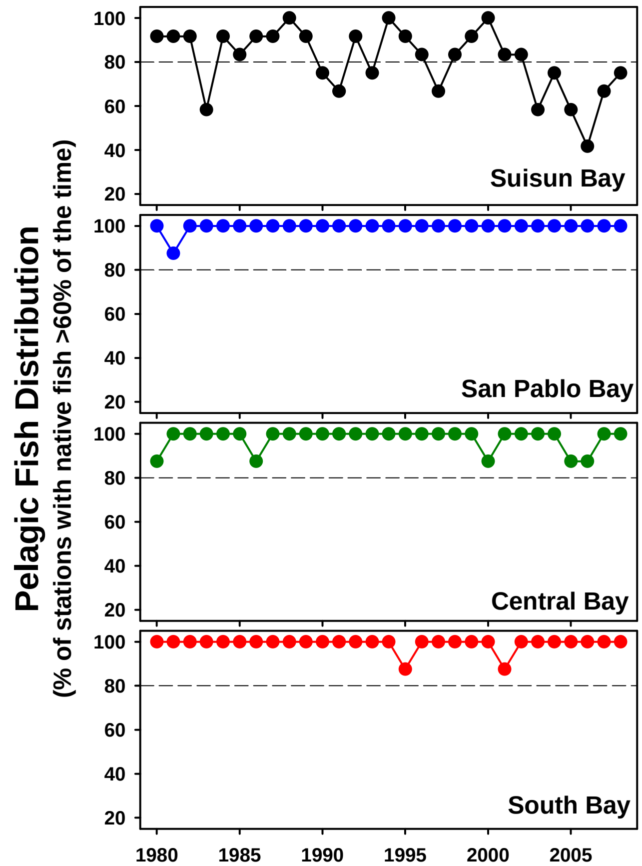


Figure 12. Changes in the Demersal Fish Distribution indicator in each of four sub-regions of the San Francisco Estuary from 1980-2008. Horizontal dashed line shows the reference condition.

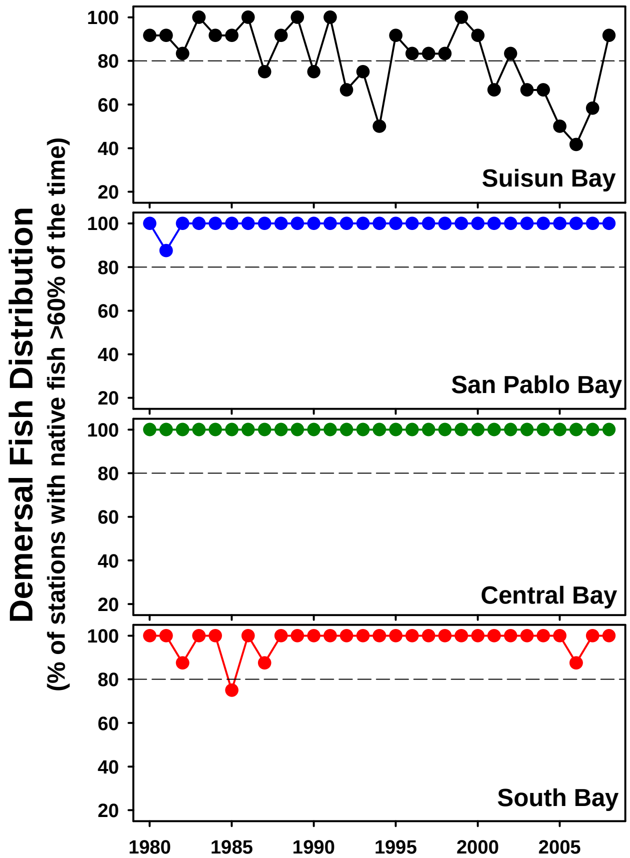


Figure 13. Changes in the Fish Index in each of four sub-regions of the San Francisco Estuary from 1980-2008. Black lines and symbols show the annual Index values, solid red line shows the linear regression for the 1980-2008 period. Horizontal dashed lines shows the reference conditions and associated interpretations.

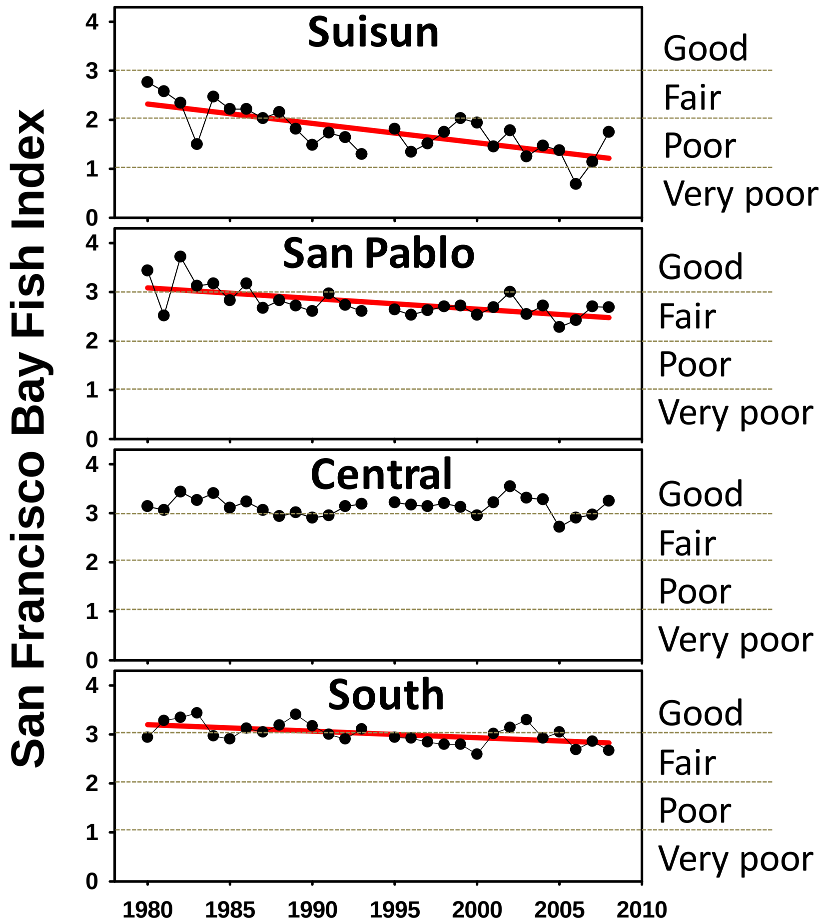


Table 8. Summary of results, relative to the CCMP goals to "recover" and "reverse declines" of estuarine fishes, of the seven fish indicators for each of the four sub-regions of the San Francisco Estuary.

Indicator or Index	Sub-region	CCMP Goal Met (yes or no)	Trend long-term (29 yrs)	Trend short-term (last 5 yrs)
Pelagic Fish Abundance	Suisun	No	Decline	Stable
	San Pablo	No	Decline	Stable
	Central	No	Stable	Stable
	South	No	Decline	Stable
Northern Anchovy Abundance	Suisun	No	Decline	Stable
	San Pablo	No	Decline	Increase
	Central	Yes	Stable	Stable
	South	Yes	Stable	Stable
Demersal Fish Abundance	Suisun	Yes	Decline	Stable
	San Pablo	Yes	Stable	Stable
	Central	Yes	Increase	Stable
	South	Yes	Stable	Stable
Sensitive Fish Species Abundance	Suisun	Yes	Decline	Stable
	San Pablo	Yes	Decline	Stable
	Central	No	Decline	Stable
	South	No	Decline	Stable
Native Fish Species Diversity	Suisun	Yes	Stable	Stable
	San Pablo	Yes	Decline	Stable
	Central	Yes	Increase	Stable
	South	Yes	Stable	Stable
Estuary-dependent Fish Species Diversity	Suisun	Yes	Stable	Stable
	San Pablo	Yes	Decline	Stable
	Central	Yes	Stable	Stable
	South	No	Decline	Stable
Percent Native Species	Suisun	No	Decline	Stable
	San Pablo	No	Decline	Stable
	Central	Yes	Stable	Stable
	South	Yes	Decline	Decline
Percent Native Fish	Suisun	No	Decline	Stable
	San Pablo	Yes	Stable	Stable
	Central	Yes	Stable	Stable
	South	Yes	Decline*	Decline
Pelagic Fish Distribution	Suisun	No	Decline	Stable
	San Pablo	Yes	Stable	Stable
	Central	Yes	Stable	Stable
	South	Yes	Stable	Stable
Demersal Fish Distribution	Suisun	No	Decline	Stable
	San Pablo	Yes	Stable	Stable
	Central	Yes	Stable	Stable
	South	Yes	Stable	Stable
Fish Index	**Suisun**	**No**	**Decline**	**Stable**
	San Pablo	**No**	**Decline**	**Stable**
	Central	**Yes**	**Stable**	**Stable**
	South	**No**	**Decline**	**Stable**

AUTHOR QUERY FORM

ELSEVIER	Journal: MARGO Article Number: 4860	Please e-mail or fax your responses and any corrections to: E-mail: Corrections.ESCH@elsevier.spitech.com Fax: +1 619 699 6721

Dear Author,

Please check your proof carefully and mark all corrections at the appropriate place in the proof (e.g., by using on-screen annotation in the PDF file) or compile them in a separate list. Note: if you opt to annotate the file with software other than Adobe Reader then please also highlight the appropriate place in the PDF file. To ensure fast publication of your paper please return your corrections within 48 hours.

For correction or revision of any artwork, please consult http://www.elsevier.com/artworkinstructions.

Any queries or remarks that have arisen during the processing of your manuscript are listed below and highlighted by flags in the proof. Click on the 'Q' link to go to the location in the proof.

Location in article	Query / Remark: click on the Q link to go Please insert your reply or correction at the corresponding line in the proof
Q1	Please confirm that given names and surnames have been identified correctly.
Q2	Please check the telephone/fax number of the corresponding author, and correct if necessary.
Q3	Uncited reference: This section comprises references that occur in the reference list but not in the body of the text. Please position each reference in the text or, alternatively, delete it. Thank you. Please check this box if you have no corrections to make to the PDF file. ☐

Thank you for your assistance.

BDCP1673
Marine Geology xxx (2012) xxx

Contents lists available at SciVerse ScienceDirect

Marine Geology

journal homepage: www.elsevier.com/locate/margeo

Highlights

Marine Geology xxx (2012) xxx–xxx

Integration of bed characteristics, geochemical tracers, current measurements, and numerical modeling for assessing the provenance of beach sand in the San Francisco Bay Coastal System

Patrick L. Barnard [a,*], Amy C. Foxgrover [a], Edwin Elias [a,b], Li H. Erikson [a], James R. Hein [a], Mary McGann [a], Kira Mizell [a], Robert J. Rosenbauer [a], Peter W. Swarzenski [a], Renee K. Takesue [a], Florence L. Wong [a], Donald L. Woodrow [a]

[a] United States Geological Survey, Pacific Coastal and Marine Science Center, 400 Natural Bridges Drive, Santa Cruz, CA 95060, USA
[b] Deltares, P.O. Box 177, 2600 MH Delft, Rotterdamseweg 185, 2629DH Delft, Netherlands

► 9 provenance techniques are integrated to track sand in a coastal–estuarine system. ► Cross-validation results in robust model of beach-sized sand transport pathways. ► Sources and pathways highlight scale of anthropogenic influences in the system. ► Historical decline of primary sediment source linked to erosion of ocean beaches ► Unique approach applicable for coastal–estuarine systems worldwide

0025-3227/$ – see front matter. Published by Elsevier B.V.
http://dx.doi.org/10.1016/j.margeo.2012.11.008

Marine Geology xxx (2012) xxx–xxx

Contents lists available at SciVerse ScienceDirect

Marine Geology

journal homepage: www.elsevier.com/locate/margeo

Integration of bed characteristics, geochemical tracers, current measurements, and numerical modeling for assessing the provenance of beach sand in the San Francisco Bay Coastal System

Patrick L. Barnard [a,*], Amy C. Foxgrover [a], Edwin Elias [a,b], Li H. Erikson [a], James R. Hein [a], Mary McGann [a], Kira Mizell [a], Robert J. Rosenbauer [a], Peter W. Swarzenski [a], Renee K. Takesue [a], Florence L. Wong [a], Donald L. Woodrow [a]

[a] United States Geological Survey, Pacific Coastal and Marine Science Center, 400 Natural Bridges Drive, Santa Cruz, CA 95060, USA
[b] Deltares, P.O. Box 177, 2600 MH Delft, Rotterdamseweg 185, 2629DH Delft, Netherlands

ARTICLE INFO

Article history:
Received 3 July 2012
Received in revised form 16 November 2012
Accepted 16 November 2012
Available online xxxx

Keywords:
provenance
bedforms
sediment transport
isotopes
foraminifera
heavy minerals
rare earth elements
numerical modeling

ABSTRACT

Over 150 million m^3 of sand-sized sediment has disappeared from the central region of the San Francisco Bay Coastal System during the last half century. This enormous loss may reflect numerous anthropogenic influences, such as watershed damming, bay-fill development, aggregate mining, and dredging. The reduction in Bay sediment also appears to be linked to a reduction in sediment supply and recent widespread erosion of adjacent beaches, wetlands, and submarine environments. A unique, multi-faceted provenance study was performed to definitively establish the primary sources, sinks, and transport pathways of beach-sized sand in the region, thereby identifying the activities and processes that directly limit supply to the outer coast. This integrative program is based on comprehensive surficial sediment sampling of the San Francisco Bay Coastal System, including the seabed, Bay floor, area beaches, adjacent rock units, and major drainages. Analyses of sample morphometrics and biological composition (e.g., Foraminifera) were then integrated with a suite of tracers including $^{87}Sr/^{86}Sr$ and $^{143}Nd/^{144}Nd$ isotopes, rare earth elements, semi-quantitative X-ray diffraction mineralogy, and heavy minerals, and with process-based numerical modeling, in situ current measurements, and bedform asymmetry to robustly determine the provenance of beach-sized sand in the region.

Published by Elsevier B.V.

1. Introduction

A definitive understanding of sediment sources, sinks, and pathways in urbanized coastal–estuarine systems is essential for assessing the current and future effects of sediment-impacting activities, such as dredging operations, aggregate mining, shoreline armoring, and watershed modifications (Duck et al., 2001). More informed management of sediment resources can promote the sustainability of fringing tidal wetlands and beaches, the first line of defense as sea level rises (Vermeer and Rahmstorf, 2009) and potentially larger storms (Graham and Diaz, 2001) increase the vulnerability of coastal environments over the next century and beyond (Jevrejeva et al., 2012), enhancing threats to public safety, vital infrastructure, and ecosystems (Nicholls and Cazenave, 2010).

The physical, biological, geochemical, and mineralogical composition of coastal sediment is a product of multiple factors, including river catchment petrology (Cho et al., 1999), cliff and seafloor geology, biogenic contributions (Lackschewitz et al., 1994), oceanographic and climatic conditions (Bernárdez et al., 2012), residence time, grain size, shape, density, and local hydrodynamics (Steidtmann, 1982). Therefore, understanding the sources of beach sediment can yield important information about transport pathways and anthropogenic impacts, littoral transport directions, and local erosion.

Spatial variations in grain size parameters (i.e., mean grain size, sorting, and skewness) have been used as tool for decades to infer sediment transport pathways, with insight into local sources and sinks (e.g., McLaren and Bowles, 1985; Gao and Collins, 1992; Le Roux, 1994). However, this approach suffers from severe limitations, such as lack of validation data sets for the multiple approaches, uncertainty as to whether the grain size variability is associated with a modification of the hydrodynamic energy or with sediment reworking processes, input uncertainties such as sampling and measurement error, and model uncertainties (Poizot et al., 2008). Preferential sorting on beaches has established heavy mineral analysis as a common tracer for establishing provenance (e.g., Rao, 1957; Morton, 1985; Frihy et al., 1995), where storms, frequent washing of sediments, and wind erosion can focus more dense, darker grains in distinct layers (Da Silva, 1979; Li and Komar, 1992). However, from source to sink, the effects of weathering, transportation, deposition and diagenesis must be considered in interpretations (Morton, 1985), and the mechanisms of beach

* Corresponding author. Tel.: +1 831 427 4756; fax: +1 831 427 4758.
E-mail address: pbarnard@usgs.gov (P.L. Barnard).

0025-3227/$ – see front matter. Published by Elsevier B.V.
http://dx.doi.org/10.1016/j.margeo.2012.11.008

deposition are still poorly understood (Gallaway et al., 2012). Andrews and Eberl (2012) used quantitative X-ray diffraction (qXRD) and SedUnMix, an Excel Macro program, to gain a greater understanding of provenance in a complicated glacial marine system, but were not able to capture exact source rock compositions, a common shortcoming of qXRD. Magnetic properties of sediment have been used as a fast, low cost means to explore sediment provenance in estuaries (Jenkins et al., 2002) and beaches (Rotman et al., 2008), although magnetic signatures are not useful if the magnetic susceptibility of source areas is not distinct, and the results are complicated by the natural particle size variability of the samples (Oldfield and Yu, 1994). Rare earth elements (REE) have been used as a tracer to determine sediment transport pathways (Ronov et al., 1967; Piper, 1974), with numerous studies using REEs to determine coastal sediment provenance (e.g., Munksgaard et al., 2003; Prego et al., 2012), but their universal applicability can be limited by natural abundance. Isotopic analysis (e.g., $^{87}Sr/^{86}Sr$ and $^{143}Nd/^{144}Nd$) has often been used in recent years, particularly for mud-dominated seafloor sediments and eolian dust (e.g., Lee et al., 2005; Saitoh et al., 2011), due to their stability and reflection of minerals and rocks with different ages and compositions (Grousset and Biscaye, 2005), but the analysis is expensive and the results can be difficult to interpret (Farmer et al., 2003).

The only means to implement effective local and regional sediment management plans that promote the sustainability of coastal environments is to understand the entire coastal system, from source to sink. However, because any given provenance technique limits the relevance and applicability of the results to discrete portions or processes within a complex coastal–estuarine system, recent studies have utilized multiple techniques. For example, Duck et al. (2001) used bedform asymmetry, grain size distribution, and magnetic susceptibility measurements in an attempt to distinguish the relative contribution of marine and fluvially derived bedload in a channel of the Tay Estuary, Scotland. Bernárdez et al. (2012) incorporated grain size, total carbon, particulate organic and inorganic carbon, particulate organic nitrogen, X-ray diffraction, heavy mineral separation, and flame atomic absorption spectrometry for metals analysis to determine the provenance of marine sediments off the coast of the northwest Iberian Peninsula. The results of these provenance studies clearly were strengthened by the use of multiple techniques, but the integration of the results in these prior studies was only qualitative.

In this study we present a uniquely extensive, complex, and robust approach to determining sediment provenance in the San Francisco Bay Coastal System, focusing on the pathways for the movement of beach-sized sand from the watershed, through the estuary, and onto open-coast beaches. This study was motivated by major anthropogenic changes to the Bay that began with the influx of hydraulic mining-related sediment from the Gold Rush in the 19th century (Gilbert, 1917), and have continued to the present with extensive indirect and direct impacts on the Bay sediment supply, including widespread watershed modifications (e.g., Wright and Schoellhamer, 2004), and Bay floor aggregate mining and dredging (Dallas and Barnard, 2011), reflected by ~150 million m^3 of erosion from the floor of San Francisco Bay over the last half of the 20th century (Barnard and Kvitek, 2010). This significant erosion of the Bay floor is temporally correlated with similarly high volumes of erosion of the ebb-tidal delta at the mouth of San Francisco Bay (Hanes and Barnard, 2007; Dallas and Barnard, 2009), as well as widespread erosion of adjacent, open-coast beaches (Hapke et al., 2006; Dallas and Barnard, 2011; Barnard et al., 2012a). However, a quantitative physical or geochemical connection has not been established between sediments inside and outside the Bay, nor a definitive causal link driving regional coastal erosion.

Using extensive regional sediment sampling, geochemical and mineralogical analyses, multibeam bathymetry mapping, physical process measurements, and numerical modeling, we developed a semi-quantitative method to integrate and cross-validate the results of nine separate techniques for establishing sand provenance:

1) Grain size morphometrics
2) $^{87}Sr/^{86}Sr$ and $^{143}Nd/^{144}Nd$ isotopic ratios
3) Rare earth element (REE) composition
4) Heavy minerals
5) Semi-quantitative X-ray diffraction (XRD)
6) Biologic, anthropogenic, and volcanic constituents
7) Bedform asymmetry
8) Acoustic Doppler velocity measurements
9) Modeled residual sediment transport

The multifaceted approach results in a definitive understanding of sand movement in the coastal–estuarine system, thereby providing essential information to promote more efficient management of sediment resources. This unique and complex approach can serve as a model for provenance studies worldwide.

2. Study area

2.1. Physical setting

San Francisco Bay is the largest estuary on the U.S. West Coast (Conomos et al., 1985), and is among the most developed and human-altered estuaries in the world (Knowles and Cayan, 2004). The San Francisco Bay Coastal System comprises four sub-embayments, as well as the open coast littoral cell, extending from Pt. Reyes to Pt. San Pedro, the ebb-tidal delta (i.e., San Francisco Bar) at the mouth of San Francisco Bay, the inlet throat (i.e., Golden Gate), and the Sacramento–San Joaquin Delta mouth (Fig. 1). The region is subjected to highly energetic physical forcing, including spatially and temporally variable wave, tidal current, wind, and fluvial forcing. The open coast at the mouth of San Francisco Bay is exposed to swell from almost the entire Pacific Ocean, with annual maximum offshore significant wave heights (h_s) typically exceeding 8 m, and mean annual $h_s = 2.5$ m (Scripps Institution of Oceanography, 2012). Inside the Bay, wave forcing is less important, except on shallow Bay margins where local wind-driven waves, and occasionally open ocean swell can induce significant turbulence and sediment transport (Talke and Stacey, 2003; Hanes et al., 2011). Tides at the Golden Gate (NOAA/Co-ops station 9414290) are mixed, semi-diurnal, with a maximum tidal range of 1.78 m (MLLW–MHHW, 1983–2001 Tidal Epoch), but due to the large Bay surface area (1200 km^2 at MSL), the Golden Gate strait serves a spring tidal prism of 2×10^9 m^3. This powerful tidal forcing results in peak ebb tidal currents that exceed 2.5 m/s in the Golden Gate, peak flood tidal currents of 2 m/s just inside Central Bay, and even 1 m/s on the edge of the ebb-tidal delta, 10 km from the inlet throat (Rubin and McCulloch, 1979; Barnard et al., 2007). The strongest tidal currents throughout the other sub-embayments are focused in the main tidal channels, commonly approaching 1 m/s (e.g., Wright and Schoellhamer, 2004). Bedforms dominate the substrate (Rubin and McCulloch, 1979; Chin et al., 2004; Barnard et al., 2006, 2011b, 2012b) where sand is prevalent among the highly energetic areas throughout the region, including at the mouth of San Francisco Bay and the deeper portions of Central Bay, San Pablo Bay, and Suisun Bay (Fig. 1), particularly within the main tidal channels. The bottom sediments are mud-dominated in South Bay and in the shallower (<4 m), lower tidal energy areas of Central Bay, San Pablo Bay, and Suisun Bay (Conomos and Peterson, 1977).

Sediments are derived from watersheds of the Sacramento–San Joaquin Delta (i.e., Sierran, notably granitic) and local tributaries, and the local coast range that outcrops along the open coast, in the Golden Gate and Central Bay (i.e., Franciscan Complex, notably chert and serpentine, and younger volcanic and sedimentary rocks) (Gilbert, 1917; Yancey and Lee, 1972; Schlocker, 1974; Porterfield, 1980; McKee et al., 2003; Graymer et al., 2006; Keller, 2009). The modern Bay floor and adjacent open coast seafloor are primarily composed of sand and

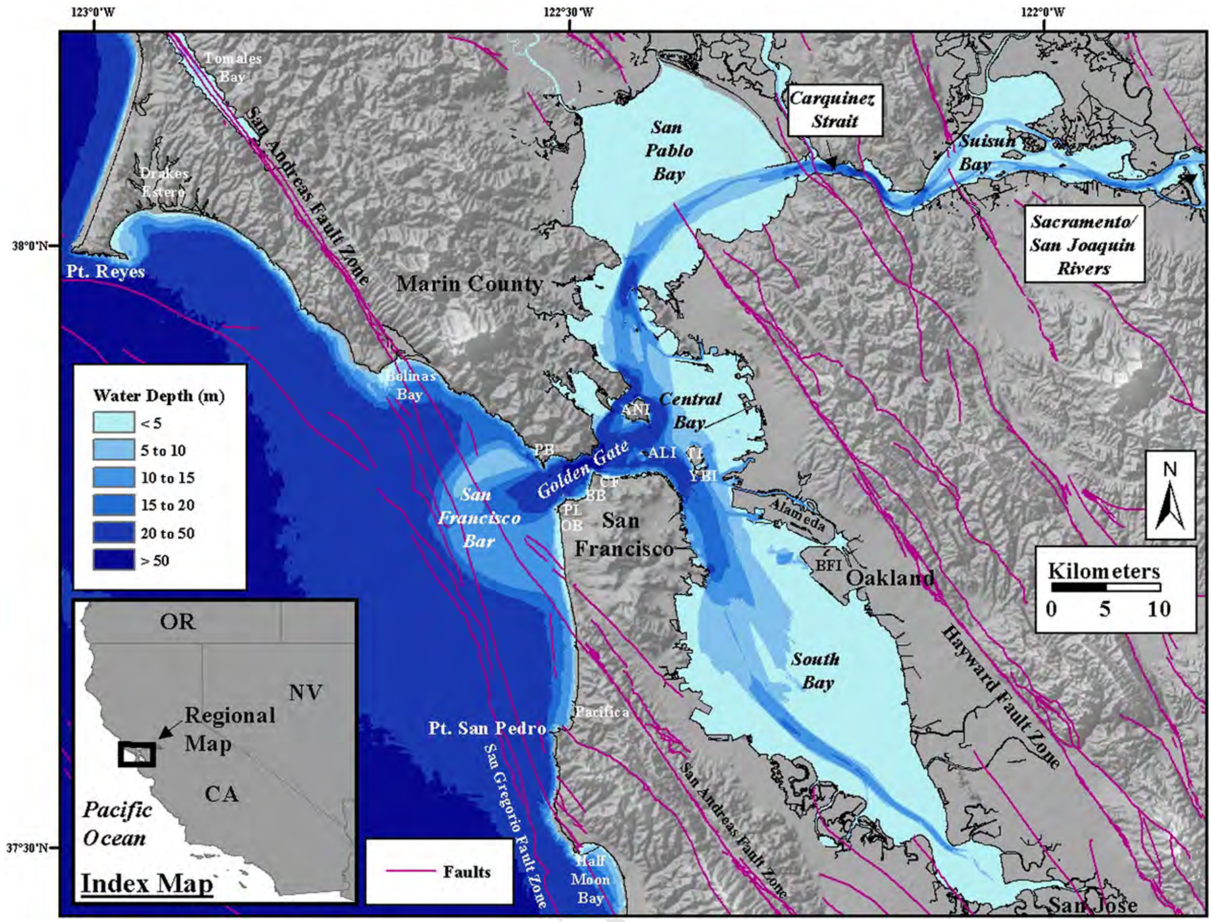


Fig. 1. The San Francisco Bay Coastal System. (ALI = Alcatraz Island, ANI = Angel Island, BB = Baker Beach, BFI = Bay Farm Island, CF = Crissy Field, OB = Ocean Beach, PB = Pt. Bonita, PL = Pt. Lobos, TI = Treasure Island, YBI = Yerba Buena Island).
Fault lines from USGS (2006).

mud of Sierran and Franciscan origin that is actively transported into the region (Keller, 2009), overlying metamorphic and sedimentary bedrock: the shallowest depths to bedrock and intermittent bedrock exposures are most common in Central Bay (Trask, 1956; Goldman, 1969; Carlson and McCulloch, 1970; Chin et al., 2004), within the Golden Gate (Barnard et al., 2006), the northern open coast, and Carquinez Strait (Jachens et al., 2002). The framework geology for the San Francisco Bay Coastal System is described extensively in Elder (this issue).

2.2. Prior work-sediment transport

Historically, the majority of the sediment load to San Francisco Bay was supplied from the Sacramento–San Joaquin Delta (Krone, 1979; Porterfield, 1980), with the Sacramento River producing seven times the sediment yield of the San Joaquin River (Oltmann et al., 1999), a ratio that is still valid (Wright and Schoellhamer, 2005). Prior to the Gold Rush in 1849, Gilbert (1917) estimated sediment supply from the Delta to the Bay was ~1.3 Mt/yr. Ganju et al. (2008) estimated a decrease in mean annual sediment loads to the Delta from a high of greater than 10 Mt/yr in the late 19th century to less than 3 Mt/yr in the latter half of the 20th century, with a dramatic decrease after 1910 attributed to the onset and subsequent cessation of hydraulic mining, followed by major Delta modifications (Knowles and Cayan, 2004). Recent estimates of suspended loads entering the estuary from the Sacramento–San Joaquin Delta range from 1.2 Mt/yr (McKee et al., 2006) to 4 Mt/yr (Shvidchenko et al., 2004), with most of this likely mud-sized, with a comparable amount coming from local tributaries (Lewicki and McKee, 2010). However, newly updated estimates of suspended supply for the period 1995–2010 from the Delta are 0.89 Mt/yr, and 1.43 Mt/yr from local tributaries, indicating that local watersheds are now the dominant source of sediment feeding the Bay (McKee et al., this issue). Suspended sediment loads decreased by 50% from the Sacramento River from 1957 to 2001, from ~2–3 Mt/yr to 1–2 Mt/yr, or, assuming a linear decrease over that time period, a total reduction of ~25 Mt (Wright and Schoellhamer, 2004; Singer et al., 2008). From water years 1991–1998 to 1999–2007, there was an abrupt, 36% step decrease in suspended sediment concentrations observed inside the Bay, broadly attributed to the depletion of the 'erodible sediment pool' created by hydraulic mining and possibly urbanization, and further reduced by river bank protection, and sediment trapping behind dams and in flood bypasses (Schoellhamer, 2011). However, the transport pathways and ultimate sink of these historically varying sediment loads has never been established.

The net direction of sediment transport across the Golden Gate, the critical interface that connects the Bay and the open coast, is poorly understood, but paramount to understanding limits on sediment supply within the San Francisco Bay Coastal System. Fram et al. (2007) measured root-mean-squared instantaneous discharges across the inlet throat of 60,000 m^3/s, mean discharges of 600 m^3/s (net seaward), and a mildly stratified channel, while Martin et al. (2007) noted that the direction of the net advective flux of chlorophyll was always seaward. The only direct estimates of suspended sediment transport using in situ measurements across the Golden Gate were conducted by Teeter et al. (1996), who performed repeated inlet cross-sectional transects using boat-mounted acoustic Doppler profiler systems. They

Please cite this article as: Barnard, P.L., et al., Integration of bed characteristics, geochemical tracers, current measurements, and numerical modeling for assessing the provenance..., Marine Geology (2012), http://dx.doi.org/10.1016/j.margeo.2012.11.008

observed a clear net seaward transport of suspended sediment of 188,000 Mt over a two week period, with fluxes during ebb flows 44% higher on average than during flood flows. Although direct measurements of bedload transport across the Golden Gate have not been performed, an extensive study of bedform asymmetry covering west-central San Francisco Bay and the mouth of San Francisco Bay suggests a net seaward flux of bedload through the Golden Gate, further confirmed by applying a hydrodynamically-validated numerical model to estimate the net flux of suspended load and bedload across the inlet throat (Barnard et al., 2012b, this issue-a). A complete summary of sediment transport research in the region can be found in Barnard et al. (this issue-b).

2.3. Prior work-sediment provenance

A number of sediment provenance studies in the San Francisco Bay area have focused exclusively on the mud fraction (e.g., Knebel et al., 1977; Griggs and Hein, 1980; Hornberger et al., 1999; Ingram and Lin, 2002), with fewer studies providing information on sand sources and littoral transport, but typically just the fine and very fine sand fraction (~0.063–0.25 mm; e.g., Moore, 1965; Cherry, 1966; Wong, 2001). Yancey and Lee (1972) identified five distinct heavy mineral assemblages for the Central California coast. This study linked the majority of bottom sediments in North Bay (i.e., Suisun and San Pablo Bays), Central Bay, and the mouth of San Francisco Bay south to Pacifica to a Sierran source delivered to the Bay by the San Joaquin–Sacramento drainage basins (see Fig. 1), suggesting that the dominant regional direction of transport is from the Bay seaward toward the ebb-tidal delta, and then primarily to the south, which the Sierran sedimentary petrographic province of Moore (1965) also strongly suggests. Locally-derived heavy mineral assemblages are more evident for South Bay, and in the immediate vicinity of Pt. Reyes and Bolinas Bay.

Conomos (1963) used heavy and light minerals to determine that most of the sandy sediment in the southern half of South Bay was derived from the Franciscan rocks of local tributaries (primarily Alameda Creek, which enters South Bay along the southeastern shoreline: Fig. 1) entering the sub-embayment, with no sediment from the Sacramento–San Joaquin Delta. The fine fraction of the northern portion of South Bay is well-mixed with the majority of sediment inflow originating from other sub-embayments to the north, but the sand fraction appears locally-derived (Gram, 1966), evidence that the mud and sand fractions are transported by a different set of processes and cannot be used as tracers for each other. Based on surficial grain size distributions and the multibeam, backscatter and sidescan data of Greene and Bizarro (2003), Chin et al. (2010) suggest that the sand in Central Bay is derived from either outside San Francisco Bay, shoreline sediments and outcrops in the vicinity of the Golden Gate (the coarser sands), or from San Pablo Bay (finer sands), with little mixing of the two fractions.

Along the open coast, a major potential source of sediment north of Pt. Reyes is the Russian River mouth, but heavy mineral analysis of beach and inner shelf sediments document a sharp decrease in abundance south of Bodega Head (Cherry, 1964; Minard, 1971; Demirpolat, 1991). The Russian River thus is unlikely to be a significant source of sediment to the San Francisco Bay Coastal System. Cherry (1966) used heavy mineral distribution on several beaches to track littoral sand movement near Pt. Reyes, finding negligible net movement of sand, with most of the beach material locally derived from the less resistant beach-backing cliffs, and inactive transport beyond ~27 m water depth. Wilde et al. (1969) collected over 60 cliff, beach and inner shelf samples in the Bolinas Bay region, finding the major supply of heavy minerals being a granitic source extending directly from the ebb-tidal delta at the mouth of San Francisco Bay, with secondary sources from Bolinas Lagoon and adjacent cliffs. Landward of this lobe, to the north and northeast, Franciscan minerals become increasingly more concentrated. They also established a counter-clockwise transport of sediment within Bolinas Bay with flux of 220,000 m^3, and bottom sediments in a state of quasi-equilibrium.

The actively eroding Franciscan bluffs bordering the Golden Gate are likely a significant local source of coarse sediment, with diagnostic minerals and mineral assemblages found on the ebb-tidal delta (Gilbert, 1917), the ocean floor of the Golden Gate, beaches along the open coast (Moore, 1965), and from west-central San Francisco Bay (Keller, 2009). Two local Quaternary sedimentary formations with Sierran material (Merced and Colma formations) are exposed on Angel Island, from Ocean Beach to Pacifica, and may underlie sediment offshore (Schlocker, 1974; Bruns et al., 2002). Schlocker (1974) interpreted the sand at Ocean Beach as derived locally from these two formations, with mineralogy atypical of the Franciscan Complex. Particularly diagnostic of the Colma Formation is the abundance of magnetite along the heavily eroding section of southern Ocean Beach (Hansen and Barnard, 2010). Based on the physical and mineralogical properties of extensive regional beach and shelf sediment sampling (n = ~200), Moore (1965) concluded that the sand on the ebb-tidal delta and inner shelf to the south in depths less than ~30 m reflected the mineralogy of San Francisco Bay sediments (similar to channel sands west of Carquinez Strait), and was notably distinct from beach and nearshore sediments to the north. He further noted that the littoral zone in this region is largely composed of sediment locally derived from proximal headlands, cliffs, watersheds, and bays, and that littoral zone mineralogy changes alongshore when local source rock changes or physical boundaries occur. However, the composition of beach sands south of the Golden Gate are less variable than the local cliffs, suggesting only minor inputs from that local source, but with distinct southerly littoral transport. Schatz (1963) integrated the grain size and heavy mineral work of Trask (1953) and Kamel (1962) to suggest a possible pathway of sand from north to south across the crest of the ebb-tidal delta, and then toward shore at the southern end of Ocean Beach, a pathway that was later hypothesized by Battalio and Trivedi (1996).

Wong (2001) isolated the fine sand fraction (0.063 to 0.250 mm) of heavy minerals from samples collected on the continental shelf from approximately Pt. Reyes to Half Moon Bay, identifying two primary heavy mineral assemblage groups that dominated the region: 1) sand derived from granitic rocks, particularly Sierran, extending from approximately Bolinas Bay to Half Moon Bay, broadly similar to the region designated as the Sierran heavy mineral province by Yancey and Lee (1972), and 2) sand derived from Franciscan rocks, found predominantly from Bolinas Bay to Pt. Reyes. However, most of the sediment samples are well outside the active littoral zone, and believed to be relict deposits from at least the mid-Holocene. These prior studies can offer only broad guidance to our present work, as none of this research isolated the beach-sized sand fraction and traced it from source to sink, including the Bay, open-coast beaches, and the littoral zone.

3. Methods

Below is a brief summary of the methods used in this study. For a more comprehensive description of the methods for each individual technique please refer to the references listed, particularly within this special issue.

3.1. Pilot study of bulk geochemistry

Prior to the full beach-sized sand provenance study, eight surficial sediment samples were collected from beaches in the vicinity of the Golden Gate and nearshore to determine if bulk sediment chemistry could distinguish sources along the open coast (Fig. 2). Bulk sediment samples were ground to <0.15 mm and decomposed with a four-acid total digestion (Briggs and Meier, 2002). Thirty-seven major, minor, trace and rare earth elements were analyzed on a Perkin Elmer Elan

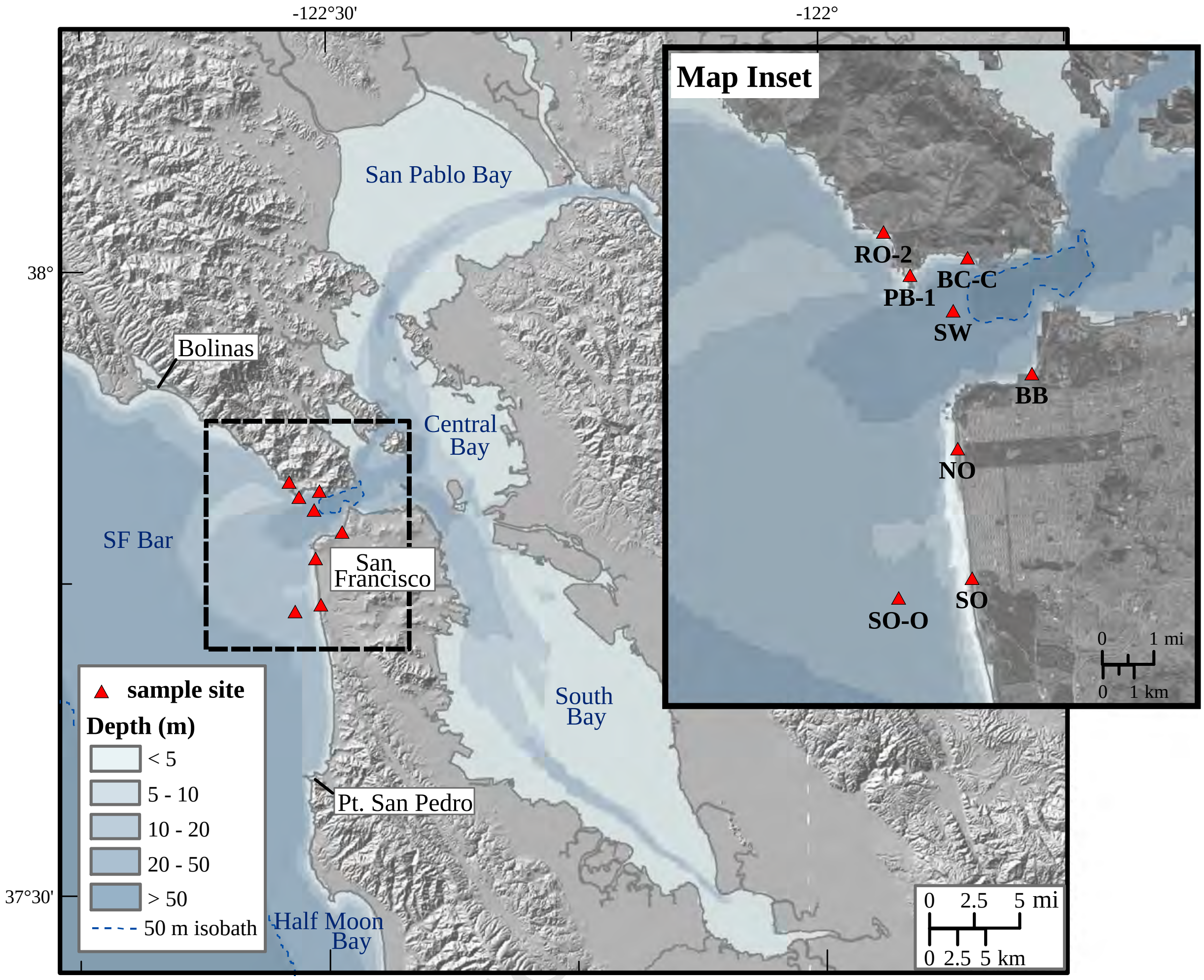


Fig. 2. Location of the pilot samples analyzed for bulk geochemistry. (RO = Rodeo Beach, PB = Point Bonita, BC = Bonita Cove, BB = Baker Beach, NO = north Ocean Beach, SO = south Ocean Beach, SW = sand wave field, SO-O = south Ocean Beach offshore).

6000 inductively-coupled plasma mass spectrometer. Limits of determination, defined as five times the standard deviation of the blank, are ≤0.01 wt.% for major elements and <1 ppm for most minor and most trace elements.

3.2. Sediment sampling and geochemistry

A total of 423 sediment and/or rock samples were collected from major Bay tributaries, the Bay and seafloor, Bay and outer coast beaches, and bedrock outcrops within the San Francisco Bay Coastal System and associated watersheds (Fig. 3). The majority of samples used in this study (n = 253) were collected over the course of 3 cruises (Table 1), with seafloor grab samples collected in early 2010 and beach and tributary samples between 2010 and 2012. An additional 170 seafloor grab samples were collected in late 2011/early 2012 and were incorporated solely for the grain size morphometrics portion of this study (see Section 3.2.1). Grain size of surface samples from a series of earlier studies (mostly collected from 2005 to 2008) throughout the region (n = 290) were also incorporated into the grain size analyses.

To characterize the geochemical signature of potential source materials, bed sediment was collected from the Sacramento River (3 sites), San Joaquin River (2 sites), and from nine smaller local tributaries that drain directly into the Bay (Napa and Guadalupe rivers; Alameda, Calaveras, Corte Madera, Del Presidio, San Francisquito, Sonoma, and Wildcat creeks) as well as the Russian River, which drains to the Pacific Ocean north of Bodega Bay (Fig. 3). Tributary samples were extracted from the top ~10 cm of sediment deposits. All of the tributary samples were collected along the river's edge by hand trowel, with the exception of two Sacramento and one of the San Joaquin River samples which were collected in the center of the channel. Source rock samples were extracted using a rock hammer at subaerial outcrops along the open coast from the major geologic rock sources (i.e., granite, basalt, chert, sandstone, and serpentinite). Forty-two surface sediment samples were collected from beaches throughout the study area. To assess transport from these sources and potential mixing and redistribution throughout the study area, surface sediment (top ~10 cm) was collected using a clam shell grab sampler from a total of 170 bay/ocean floor samples throughout the Bay and along the open coast. The surficial sediment sampling strategy was intended to capture the most active sediment layer, and therefore reflect the modern provenance of sediment. However, in some cases, the upper 10 cm of the substrate may penetrate into eroding sediments that are more representative of historical rather than contemporary conditions, and therefore the integration of co-located proxy provenance techniques (i.e., bedform asymmetry, numerical modeling, and/or velocity measurements) will be particularly effective in reducing impact of this potential bias. Prior to standard grain size processing, a small fraction of select sediment samples was selected for biologic, anthropogenic, and volcanic constituent analyses (McGann et al., this issue). The remaining fraction of all sediment samples were then cleaned with hydrogen peroxide to remove organics, disaggregated in an ultrasonic bath, washed with deionized water to

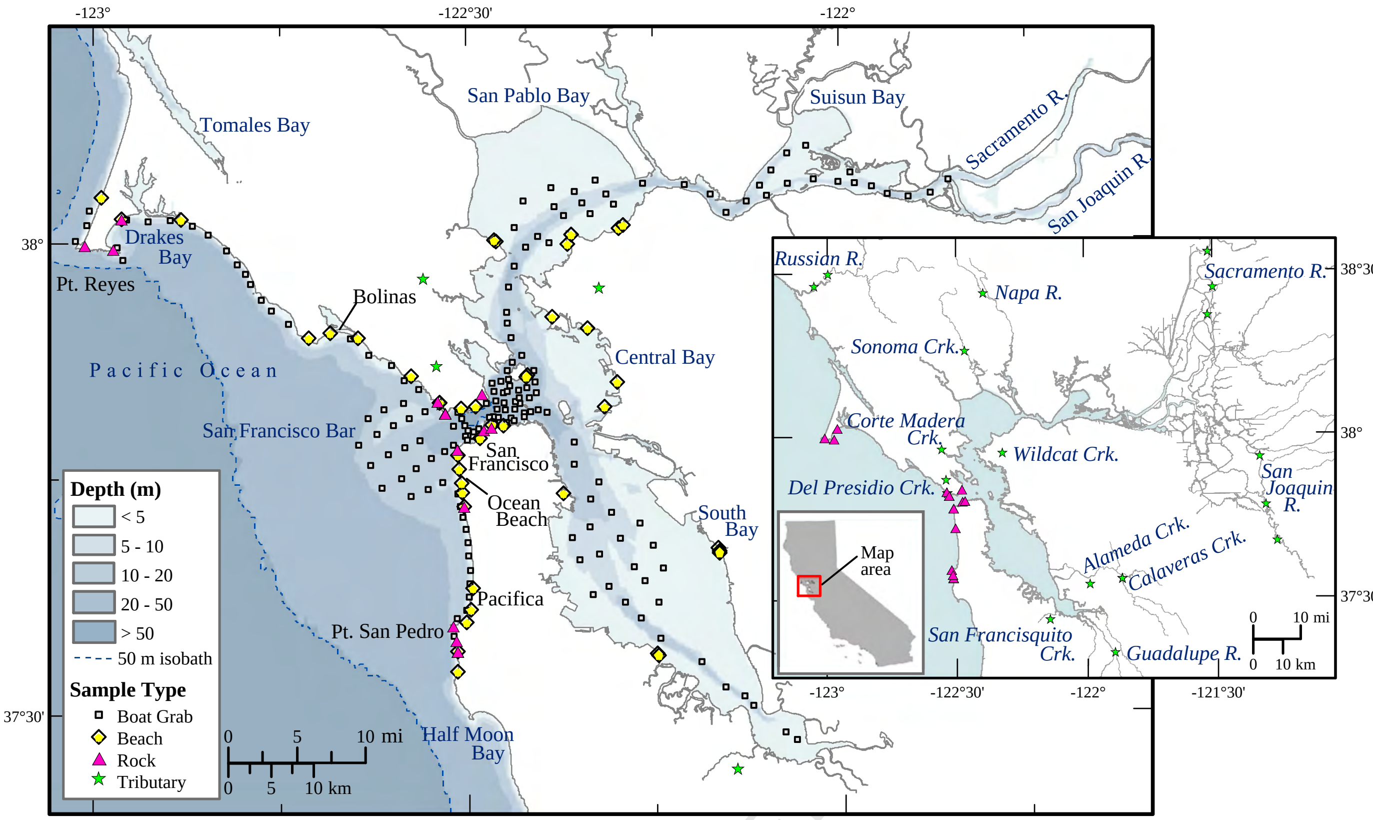


Fig. 3. Location and source of sediment samples included in this study.

remove salt, and the gravel fractions isolated from sand and mud fractions via wet sieving. Particle size analysis on the mud and sand fractions was performed using a laser diffraction particle size analyzer, and gravel size was determined by wet sieving.

Based on the mean D_{10} to D_{90} range of the open coast beach samples ($n=19$) of 0.15–0.5 mm, the beach-sized sand fraction of 101 sediment samples was split for geochemical analyses (Fig. 4). From the first split, two size fractions were isolated, 0.063–0.25 mm and 0.25–0.5 mm, and the target weight measured out for heavy mineral analysis:

- Fraction 1a — ~50 g (10 g or more) for fine sand heavy mineral analysis
- Fraction 1b — ~50 g (10 g or more) for medium sand heavy mineral analysis.

Using the second split of the sand fraction, the particle size range from 0.15 to 0.5 mm was isolated, shell was removed by acid leaching and the sample was rinsed thoroughly with ultra-pure deionized water. After being pulverized to a fine powder, bedrock samples ($n=18$) were also leached and cleaned. The cleaned, salt-free, shell-free samples were split to get target weights for additional analyses:

- Fraction 2 — ~5 g for $^{143}Nd/^{144}Nd$ and $^{87}Sr/^{86}Sr$ analyses (min. 1 g or more)
- Fraction 3 — ~10 g for semi-quantitative XRD analysis (min. 5 g or more)
- Fraction 4 — ~10 g for rare earth element analysis.

Table 2 lists the total number of samples analyzed in this study, with their locations plotted in Fig. 5.

Table 1
Cruise dates and number of samples collected (USGS, 2010, 2011).

Cruise ID	Dates	Description	Count
S-7-10-SF	1/2010	USGS cruise, SF Bay grab samples	59
S-8-10-SF	3/2010	USGS cruise, SF Bay and coastal grab samples	111
B-2-10-SF	3/2010–3/2012	Sediment collected from beaches and tributaries, rock from outcrops	83
B-5-11-SF[a]	8/2011	RMP sediment cruise coordinated by SFEI and run by Applied Marine Sciences, Inc., SF Bay grab samples	51
S-1-12-SF[a]	1/2012	USGS cruise, SF Bay and coastal grab samples	119

[a] Samples from these cruises were used solely for grain size analyses.

3.2.1. Grain size morphometrics

A simplified sediment trend analysis was performed by evaluating spatial variations in grain size parameters (mean grain size, sorting, and skewness) throughout the study area using a Geographic Information System (GIS). Surface grab samples were processed using standard procedures. Particle size distributions of the mud and sand fractions were analyzed separately using a Beckman Coulter LS100Q and the gravel fraction by wet sieves. Statistics were calculated using the method of moments for the 170 surface grab samples collected in early 2010 (Table 1) and for an additional 170 samples collected in August 2011 and January 2012. Mean grain size was also compiled from a series of earlier studies (samples mostly collected from 2005 to 2008) focusing primarily on western Central Bay, the Golden Gate and the San Francisco Bar ($n=290$). The data sets were combined and interpolated to create continuous surface representations of each of three statistics of interest (mean grain size, sorting, and skewness) using a triangular interpolated network (TIN) algorithm. The TINs were then converted to raster surfaces with a horizontal resolution of 300 m. The Flow Direction tool in the ArcGIS Spatial Analyst Toolbox was used to create surfaces of

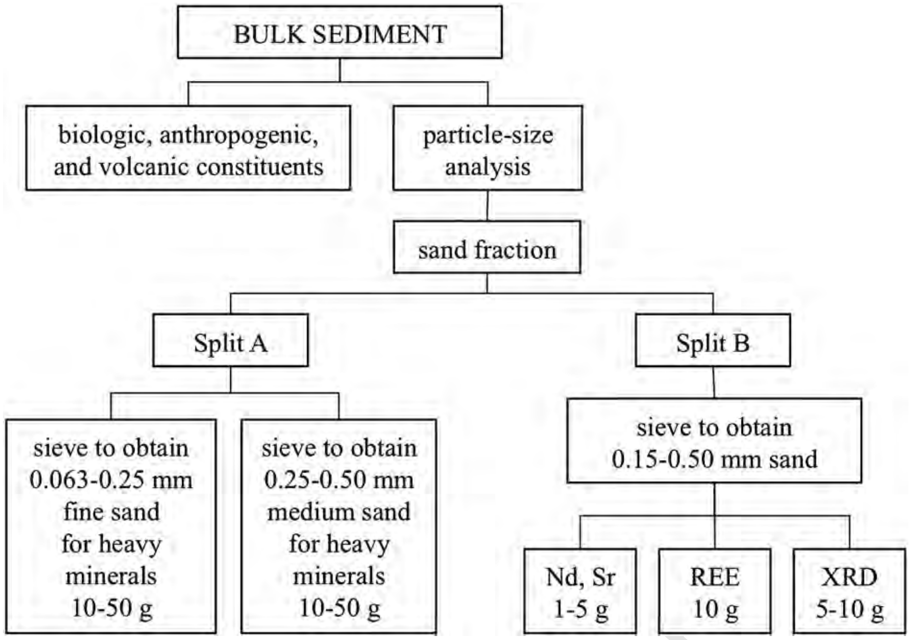


Fig. 4. Flow chart of geochemical analyses.

inferred sediment transport direction for each of the three statistics. The Flow Direction tool evaluates each individual grid cell within a raster and assigns a direction to that central cell based upon the greatest decrease in value between it and the eight surrounding grid cells. In this instance, flow direction for the three separate surfaces are derived from the greatest decrease in: (1) mean grain size (sediment fining), (2) standard deviation (better sorting), and (3) skewness (more negatively skewed in phi units, indicating a tail of coarser sediments). To assimilate results from the three different statistics, the study area was divided into 3×3 km blocks and the dominant transport direction within each block assigned. In blocks where the inferred sediment transport directions from at least two of the three parameters were within the same 90 degree quadrant of one another, the directions were averaged to calculate transport direction.

3.2.2. $^{87}Sr/^{86}Sr$, $^{143}Nd/^{144}Nd$ isotopic ratios and trace elements

Solid phase $^{87}Sr/^{86}Sr$ and $^{143}Nd/^{144}Nd$ isotopic ratios were determined following procedures presented in Weis et al. (2006). Solid phase isotopic ratios were measured using Thermal Ionization Mass Spectrometry (TIMS) and the isotopic ratios were normalized to correct for mass fractionation using reference $^{87}Sr/^{86}Sr$ and $^{143}Nd/^{144}Nd$ activity ratios. The normalized $^{143}Nd/^{144}Nd$ ratios were converted to ε_{Nd} using a value of 0.512636 for CHUR (chondritic uniform reservoir) (Rosenbauer et al., this issue).

Table 2
Number of samples used for each type of analysis by sample origin.

Analysis	Sample origin				Total
	Seafloor	Beach	Rock	Tributary	
Grain size	339	42	0	24	405
X-ray diffraction	61	27	18	13	119
Rare earth elements	58	27	16	16	117
$^{87}Sr/^{86}Sr$, $^{143}Nd/^{144}Nd$	46	16	10	15	87
Heavy minerals	27	8	1	6	42
Biologic/anthropogenic[a]	294	0	0	0	294

[a] Analyses included additional samples collected during earlier USGS and SFEI cruises. See McGann et al. (this issue).

3.2.3. Rare earth elements

A complete trace element characterization, including the suite of rare earth elements (REE) was carried out (Rosenbauer et al., this issue). Each sediment sample was fused by lithium metaborate, dissolved using dilute HNO_3, and analyzed by high-resolution inductively-coupled plasma mass spectrometry (HR-ICP-MS) on a Thermo Scientific Element 2. Precision with known calibration materials was within 2σ error of literature and recommended values. Procedural duplicates and replicate measurements showed excellent agreement, with relative standard deviations (RSD) less than 5%. REE values were chondrite normalized using values reported in Anders and Grevesse (1989), except for yttrium (Y) whose chondrite normalizing value was obtained from Bau et al. (1996). Cerium (Ce) and europium (Eu) anomalies were calculated using the formulas provided by Bau et al. (1996).

3.2.4. Heavy minerals

Sediment samples were selected from the 0.063 and 0.25-mm size fraction (or, if not enough sample was available, from the 0.25 to 0.50 mm size fraction) for heavy mineral analysis. Samples were separated in tetrabromoethane diluted to a specific gravity of 2.90; both the light and heavy (floating and sinking, respectively) grains were retrieved. The heavy grains were microsplit to about 1000 grains and mounted on glass slides. Grains were identified and counted by optical properties determined on a petrographic microscope for 42 samples. The counts were normalized as percent of total non-opaque grains and a cluster analysis was applied (Wong et al., this issue).

3.2.5. Semi-quantitative X-ray diffraction bulk sand mineralogy

The samples ($n=119$) were powdered, X-rayed, and mineral peak height counts multiplied by published weighting factors and summed to 100%. Samples were analyzed using a Philips XRD with graphite monochromator and XRD digital scans were analyzed using Philips X'Pert High Score search and match function to identify peaks and qualitative mineral composition. Cluster analysis was performed on raw scan data using Philips X'pert High Score with default settings. Cluster analysis is an automatic four-step procedure that compares each scan with all other scans and then generates a distance matrix that determines the number of "meaningful" clusters of the most representative member and of the furthermost members of each cluster.

Please cite this article as: Barnard, P.L., et al., Integration of bed characteristics, geochemical tracers, current measurements, and numerical modeling for assessing the provenance..., Marine Geology (2012), http://dx.doi.org/10.1016/j.margeo.2012.11.008

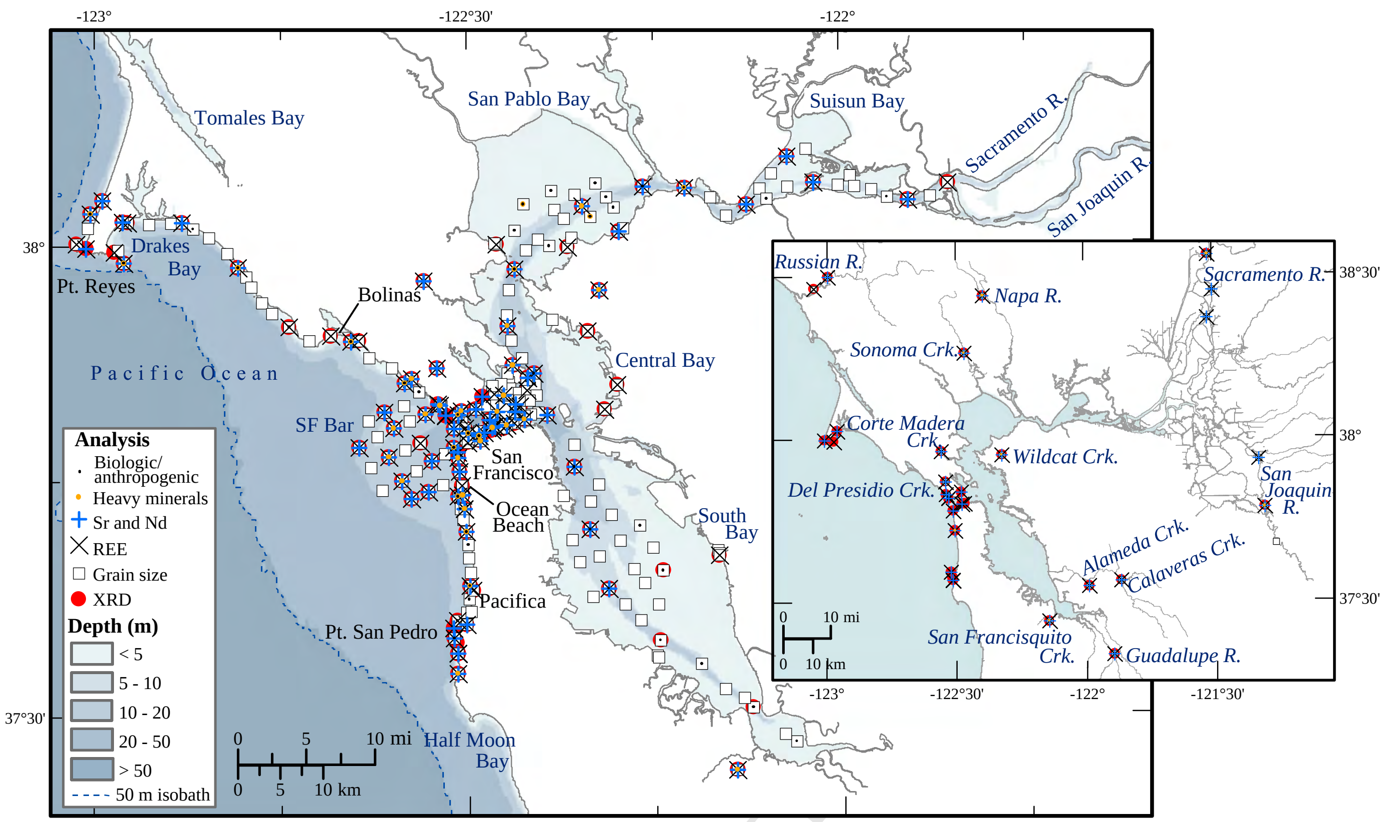


Fig. 5. Location of sediment samples and type of geochemical analyses performed.

Principal Components Analysis (PCA), which is an independent method of visualizing and judging the quality of the clustering, was also used. The scans were compared using the matching algorithm provided for qualitative phase identification (Hein et al., this issue).

3.2.6. Biologic, anthropogenic, and volcanic constituents

Bulk sediment samples for constituent analysis were collected from 294 sites by the San Francisco Estuary Institute (SFEI) from 1995 to 1998 and by the U.S. Geological Survey (USGS) in 1998 and 2010 (Fig. 6). Benthic foraminifera and representative specimens of other organic and inorganic constituents were picked from the sieved sediment and identified. Of the 294 samples, 251 were picked of all or a split (>300) of the foraminifera present; the remaining 43 samples were scanned for the presence of foraminiferal species. Relative foraminiferal species abundances from the 1995–1998 SFEI and 1998 USGS studies were converted to presence/absence data to be analogous to the 2010 USGS data. Once converted, a Q-mode cluster analysis was utilized to describe the relationship between the benthic foraminiferal assemblages. The cluster analysis grouped the samples according to their degree of similarity. Clustering was based on a square root transformation of the data, a Sørenson similarity coefficient, and amalgamated by a group-averaged linkage strategy. In addition, volcanic glass from five sites was described petrographically and analyzed by electron microprobe. The results were compared to the USGS tephra geochemical database to identify their source (McGann et al., this issue).

3.3. Bedform asymmetry

The asymmetry of ~45,000 bedforms was measured from 13 multibeam bathymetry surveys performed between 1999 and 2010 in the San Francisco Bay Coastal System (Fig. 7) to infer the bedload transport directions. Point measurements were spatially-averaged into 25,450 2500-m^2 grid cells (50 m $\times$ 50 m) using a standard inverse distance weighting technique. The inferred transport direction (ebb or flood) was based on the assumption that bedforms migrate in the direction of the steep lee face (e.g., Van Veen, 1935; Stride, 1963; Allen, 1968; McCave and Langhorne, 1982; Knaapen, 2005), an assumption that has been broadly validated in the San Francisco Bay region by near-bottom current measurements (Rubin and McCulloch, 1979) and numerical modeling (Barnard et al., this issue-a).

3.4. Measured residual currents

The long-term (months to years) net sediment transport direction is often assumed to coincide with the residual current direction. Several long-term measurements of current velocities have been made within San Francisco Bay and the immediate open coast. We synthesize some previously reported residual current analyses in South and Suisun Bays and present results of measurements obtained at the seaward end of the shoals outside the Golden Gate, along Ocean Beach to the south of the Golden Gate, and in the vicinity of Crissy Field, immediately east of the Golden Gate along the north shore of San Francisco (Fig. 7).

Cheng and Gartner (1984) and Walters et al. (1985) presented residual current directions from a suite of current meters deployed in Suisun and South Bays. Mechanical current meters were mounted on rigid moorings or tethered partially through the water column at numerous stations throughout the Bay during the years 1979 through 1982. Current meter sampling rates were set to one sample every 10 min for the 1979 and 1980 deployments and increased to every 2 min for the later measurements. In waters 10 m or deeper, two mechanical current meters were deployed simultaneously at each station, one within ~3 m of the bed and one at 7 m above the bed. Data collection at each station used in the residual analysis ranged between two and three months.

Please cite this article as: Barnard, P.L., et al., Integration of bed characteristics, geochemical tracers, current measurements, and numerical modeling for assessing the provenance..., Marine Geology (2012), http://dx.doi.org/10.1016/j.margeo.2012.11.008

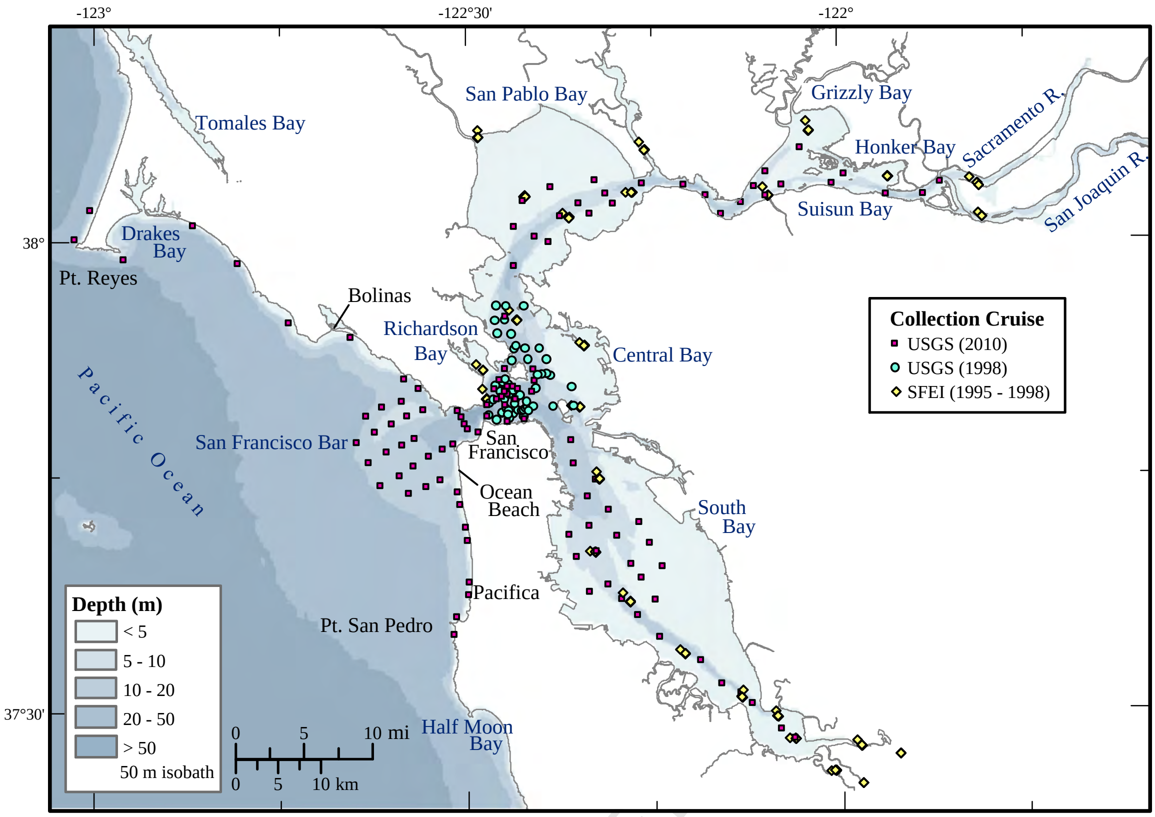


Fig. 6. Location of samples analyzed for biologic, anthropogenic, and volcanic constituents.
Sites include those collected by San Francisco Estuary Institute (SFEI) from 1995 to 1998, as well as those collected by the USGS in 1998 and 2010 (USGS, 1998, 2010).

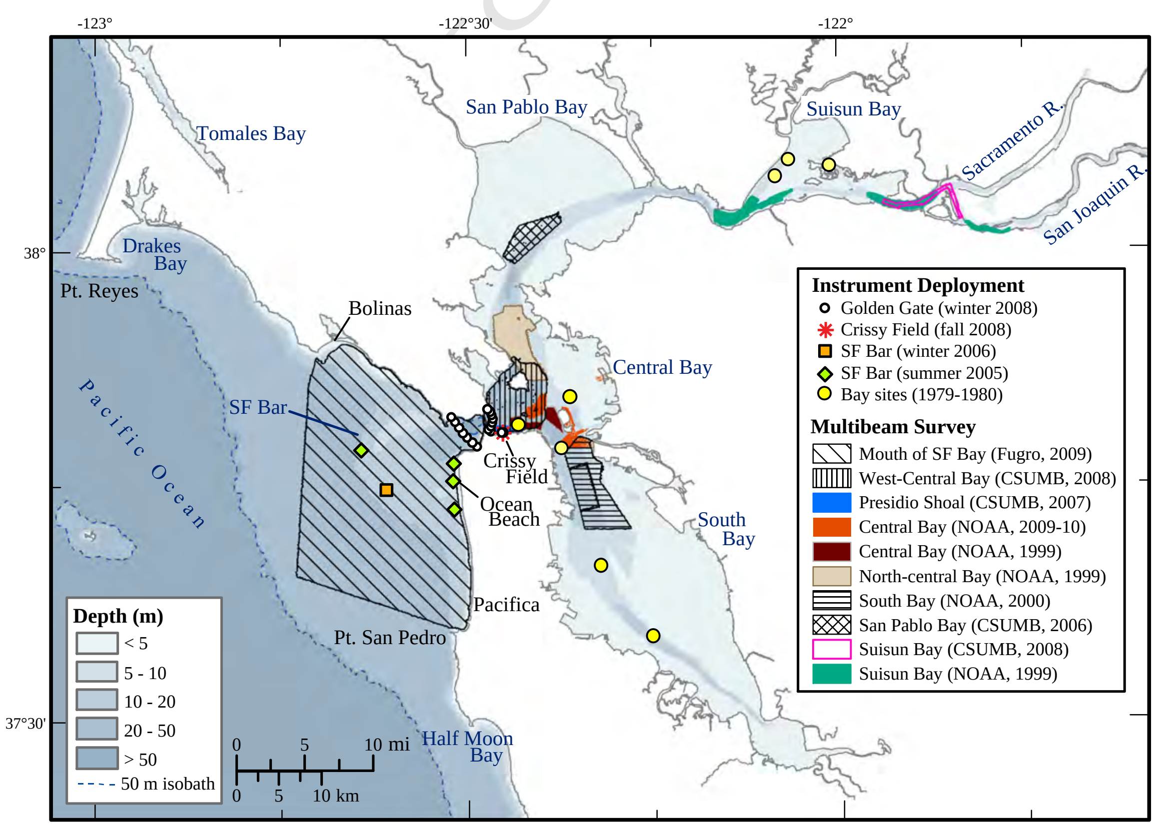


Fig. 7. Multibeam bathymetry data coverage and in situ current measurement locations.

Measurements along the outer coast and Crissy Field were obtained in 2005–2008 (Barnard et al., 2007; Hansen and Barnard, 2010; Hanes et al., 2011) with acoustic Doppler profilers (ADPs; Table 3, Fig. 7). Residual currents were calculated for the bottom bins (within 3 m of the seabed) and depth-averaged over all available bins (bin heights ranged from 0.25 m to 1 m, depending on total water depth) as measured with the current profilers. Exploiting the modern technology of ADPs, sediment flux estimates were also calculated with the acoustic backscatter intensity following the method described by Gartner (2004). Backscatter intensity data were corrected for beam spreading and water absorption, and suspended sediment concentrations computed with calibration parameters obtained from a measurement campaign at the Golden Gate (Erikson et al., this issue). The difference between residual current directions calculated with vector averaged currents and those multiplied by estimated suspended sediment concentrations was small (<15°).

Residual current results presented herein were computed using a low pass filter (cut-off frequency = 8.4175×10^{-6} Hz) for the 2005–2008 data. Calculations were done on time-series data reduced to an available maximum even multiple of the M_2 tidal period as this is by far the most dominant constituent in the Bay.

3.5. Numerical modeling

To investigate physical processes and sediment transport in the San Francisco Bay Coastal System, a coupled Delft3D hydrodynamic model FLOW and SWAN (Simulating WAves Nearshore) wave numerical model was created (Elias and Hansen, this issue). Delft3D FLOW forms the core of the model system simulating water motion due to tidal and meteorological forcing by solving the unsteady shallow water equations (Stelling, 1984; Lesser et al., 2004). The FLOW model consists of six two-way coupled domains of varying resolution for optimal computational efficiency. Given the large spatial scale involved with solving the inlet dynamics, and to achieve acceptable model run times, all flow grids were run in depth-averaged mode (2DH). The spectral wave model SWAN (version 40.72ABCDE; Holthuijsen et al., 1993; Booij et al., 1999; Ris et al., 1999) was applied in stationary, third-generation mode to propagate waves from well offshore of the continental shelf to the coastline and into the Bay. The hydrodynamic and wave models were run in quasi-nonstationary mode, a two-way coupling (15-minute intervals) of a nonstationary hydrodynamic calculation in combination with regular stationary wave simulations. The Online Morphology addition to Delft3D is used to compute sediment transport in the flow domains (Lesser et al., 2004). The TRANSPOR2004 transport equations are used to model the movement of non-cohesive sand fractions due to suspended and bed-load sediment transports. The bed was schematized as a single sediment fraction (representative for the ebb-tidal delta deposits) with a D_{50} of 0.25 mm.

Long term (multi-year) simulations would be needed to create representative sediment transport patterns, but such simulations are computationally unfeasible given the high resolution and spatial extent of the model. Instead, input schematization techniques (De Vriend et al., 1993; Lesser, 2009) were used to schematize the wave and tidal boundary forcing to create a representative set of wave conditions and a single 24.8 hour tidal cycle derived from the calibrated constituents. The total wave-averaged transports are obtained by running the coupled wave-flow model for each of the 24 wave cases over one 24.8 hour representative tidal cycle. The tide-cycle-averaged velocity and sediment transport for each simulation were then weighted by the normalized probability of occurrence of each wave case. The probability weighted results were then summed to generate an ensemble of all 24 wave cases to calculate the residual sediment transport. For additional information on modeling details, including calibration and validation, see Elias and Hansen (this issue).

3.6. Integration of techniques

In order to assimilate the results of all provenance approaches and develop a best estimate of beach-sized sand transport pathways, a semi-quantitative user-interface tool was developed using GIS software. The study area was divided into 3×3 km blocks (n = 216), and for each block the user could choose from 8 compass directions for inferred transport direction, and 3 levels of confidence (high (3), medium (2), low (1)), based on the data available for each technique (Fig. 8, Table 4). In <10% of the grid cells there was not enough sampling data available locally or regionally to make an entry for any technique. After the results for each of the individual techniques were input into separate data files, the results were compiled and outliers removed by eliminating individual transport vectors falling outside of a 180 degree radius of the majority of data. The mean transport direction (weighted by confidence) and average confidence values for each block was calculated. A final weighted confidence was assigned based on the number of entries, i.e., greater weight was given to blocks with entries from a greater number of techniques driving the result, such that:

$$\text{weighted confidence} = (\text{number of entries}) \times (\text{mean confidence score}).$$

The weighted confidence for each block was reflected in the size of the arrow in the final map of beach-sized sand (i.e., 0.15–0.50 mm for isotopes, REEs, and XRD, 0.063 mm–0.5 mm for heavy minerals) transport pathways.

4. Results and interpretation

4.1. Pilot study of bulk geochemistry

There was distinct bulk geochemical differences among sediments from the outer coast pilot study. In the vicinity of San Francisco,

Table 3
Sampling sites and instrumentation used for 2005–2008 current measurements at the outer coast and Crissy Field. See Fig. 7 for mapped locations.

Site ID	Deployment dates	Depth (m)	Lat (DD)	Long (DD)	ADP mfg. & frequency
Ocean Beach					
Site 2	06/21/05–08/16/05	11.5	37.7560	122.5200	RDI 1.2 MHz ADCP
Site 3	06/21/05–08/16/05	14.6	37.7260	122.5180	RDI 1.2 MHz ADCP
Site 4	06/21/05–07/26/05	21.1	37.7890	122.6430	Nortek AWAC 1 MHz
Site 3	01/12/06–02/06/06	13.4	37.7260	122.5180	Nortek AWAC 1 MHz
Site 5	01/12/06–02/11/06	13.9	37.7470	122.6090	RDI 1.2 MHz ADCP
TV1	01/16/08–05/19/08	12.4	37.7404	122.5210	Nortek AWAC 1 MHz
Central Bay–Crissy Field					
CF2s	09/08/08–09/26/08	4.9	37.8070	122.4507	Nortek AWAC 1 MHz
CF1j	01/14/08–01/30/08	4.9	37.8085	122.4679	Nortek AWAC 1 MHz
CF2j	01/14/08–01/30/08	4.4	37.8070	122.4507	Nortek AWAC 1 MHz

Please cite this article as: Barnard, P.L., et al., Integration of bed characteristics, geochemical tracers, current measurements, and numerical modeling for assessing the provenance..., Marine Geology (2012), http://dx.doi.org/10.1016/j.margeo.2012.11.008

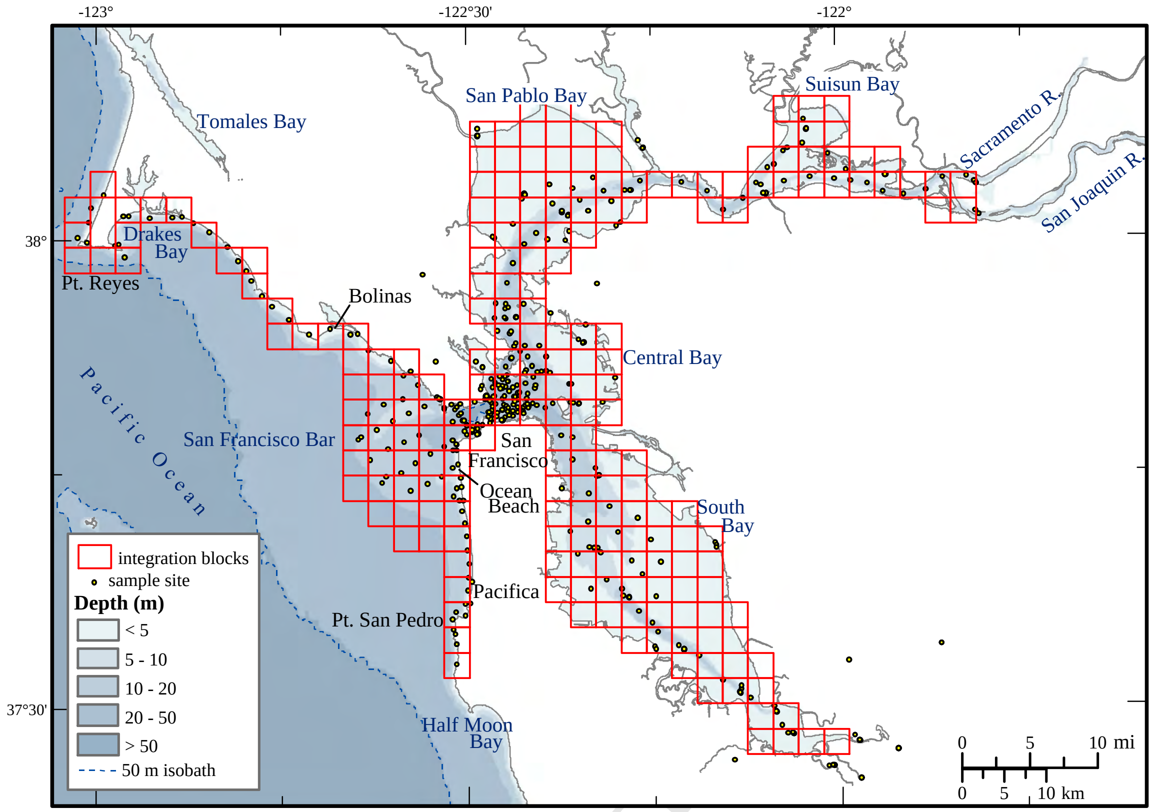


Fig. 8. Sand provenance integration grid based on 3-×3-km square blocks.

sediment at Baker Beach, northern Ocean Beach, and the sand wave field in the Golden Gate had iron to aluminum ratios (Fe/Al) more similar to granitic and felsic volcanic rocks than Franciscan rocks (Fig. 9), suggesting a Sierran source. North of the Golden Gate (Rodeo Beach, Bonita Cove, Point Bonita) and south of San Francisco (southern Ocean Beach, offshore of southern Ocean Beach) sedimentary Fe/Al ratios fall along a mixing line with Franciscan chert and average sandstone at one end, and Franciscan shale, average basalt and average shale at the other. Sediment at Point Bonita, southern Ocean Beach, and offshore of southern Ocean Beach were enriched in chromium (Cr) relative to Franciscan rocks (data not shown). The enrichments at and offshore of southern Ocean Beach are consistent with the input of Cr-enriched heavy minerals such as Cr-magnetite or chromite from the Colma Formation which outcrops along the coast. At Point Bonita the Cr enrichment was accompanied by high vanadium (V) content, and could be related to the metamorphic history of this site. In summary, based on bulk sediment geochemistry it appears that local sediment sources predominate along the coast north of the Golden Gate and south of San Francisco, while a Sierran source supplies sediment to northern San Francisco beaches (i.e., Baker Beach, north Ocean Beach) and the seafloor of the Golden Gate.

4.2. Grain size morphometrics

The only spatially coherent transport patterns that emerged from the analysis of grain size parameters (Fig. 10) were west of the Golden Gate, where the inferred transport direction in 82% of the 3×3 km blocks fell within ±90° of the average transport direction calculated using all of the techniques applied in this study. Agreement east of the Golden Gate was not as good, with only 62%, 52%, and 48% of the cells in South, Central, and North Bays, respectively, falling within the same ±90 degree window. Inferred transport patterns west of the Golden Gate are consistent with ebb-dominated flow

Table 4
Confidence intervals for the sand provenance techniques. (SQA = semi-quantitative assessment).

Technique	Diagnostic	High (3)	Medium (2)	Low (1)	No entry
Grain size morphometrics	Grain size, standard deviation and skewness	N/A	N/A	Agreement of two or more metrics in same quadrant	Agreement of less than two metrics in the same quadrant
$^{87}Sr/^{86}Sr$, $^{143}Nd/^{144}Nd$	Cluster analysis	SQA	SQA	SQA	SQA
REE composition	Cluster analysis	SQA	SQA	SQA	SQA
Heavy minerals	Cluster analysis	SQA	SQA	SQA	SQA
XRD	Cluster analysis	SQA	SQA	SQA	SQA
Misc. constituents	Cluster analysis	SQA	SQA	SQA	SQA
Bedform asymmetry	Asymmetry (%)	$A \geq 20\%$	$20\% > A \geq 10\%$	$10\% > A \geq 5\%$	$A < 5\%$
Residual current measurements[a]	Duration of deployment	$D \geq 3$ months	3 months $> D \geq 1$ month	1 months $> D \geq 2$ weeks	$D < 2$ weeks
Model — outer coast and Central Bay	Rate (m^3/d/m)	$S \geq 10^{-6}$	$10^{-6} > S \geq 10^{-8}$	$10^{-8} > S \geq 10^{-10}$	$S < 10^{-10}$
Model — South Bay and North Bay	Rate (m^3/d/m)	N/A	$S \geq 10^{-7}$	$10^{-7} > S \geq 10^{-9}$	$S < 10^{-9}$

[a] All 1979–1982 data was assigned a confidence value of 2.

Please cite this article as: Barnard, P.L., et al., Integration of bed characteristics, geochemical tracers, current measurements, and numerical modeling for assessing the provenance..., Marine Geology (2012), http://dx.doi.org/10.1016/j.margeo.2012.11.008

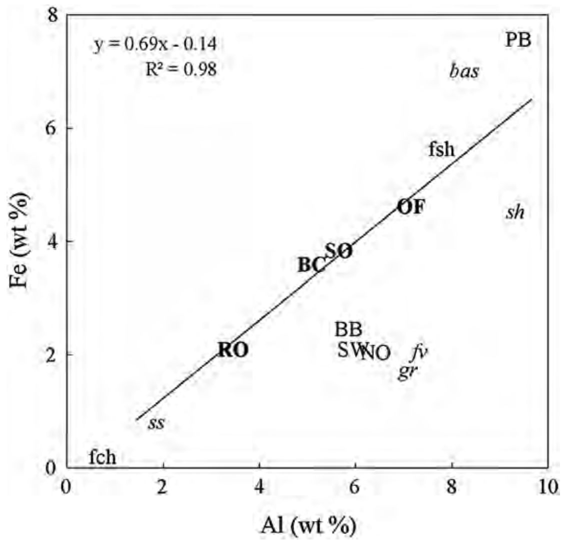


Fig. 9. Iron (Fe) relative to aluminum (Al) contents of sediment at outer coast sites (upper case), Franciscan chert and shale (lower case; Murray et al., 1991), and common rock types (italics; Condie, 1993). Line shows the least-squares regression of four outer coast sites (bold). See Fig. 2 for sample locations. (RO = Rodeo Beach, BC = Bonita Cove, SO = southern Ocean Beach, OF = offshore of southern Ocean Beach, PB = Point Bonita, BB = Baker Beach, SW = sand wave field, NO = northern Ocean Beach, fch = Franciscan chert, fsh = Franciscan shale, ss = sandstone, gr = granite, fv = felsic volcanic, sh = shale, bas = basalt).

patterns with sediments traveling in a southwesterly direction through the mouth and over the ebb-tidal delta.

Sediment transport within San Francisco Bay is very complex, and the relatively poor performance of grain size in predicting transport direction is likely due to the numerous limitations and uncertainties of this approach (see Poizot et al., 2008). A fundamental concern is whether the grain size variability captured is associated with a modification of the hydrodynamic energy or with sediment reworking processes. Additional input uncertainties stem from sampling depths (ideally capturing only the time-scale of the depositional process of interest), density of the samples, and the duration over which they were collected. Limitations due to model uncertainties of this simplified trend analysis were not quantified, and as a result, only the transport directions for west of the Golden Gate, where results were validated by independent analyses, were incorporated into the synthesis of this larger project. Because of the many limitations associated with this analysis, all transport directions inferred from grain-size measurements were assigned a low confidence rating.

4.3. Geochemical analyses

4.3.1. Isotopes and rare earth elements

The normalized $^{143}Nd/^{144}Nd$, $^{87}Sr/^{86}Sr$, and Nd/Sr isotope ratios and to a lesser extent the total amounts and ratios of trace and rare earth elements (REE) and high field strength elements (HFSE, such as Y, Zr, Nb, Ta), were used to infer beach-sized sand transport pathways in the region. The Nd and Sr isotope ratios indicate that the sediment within the San Francisco Bay Coastal System can be complexly sourced both locally and distally. Using the Jenks (1967) optimization method, a data classification method for determining the ideal grouping of values into distinct classes, the Nd and Sr isotope and REE anomaly data were classified into five distinct groups (Figs. 11–12) that were correlated relative to likely geographical sources. Based on the most robust isotopic indicator (ε_{Nd} — for more information see Rosenbauer et al., this issue), the predominant source of beach-sized sand to Suisun Bay, San Pablo Bay, and Central Bay is likely derived from the Sierras via the Sacramento River with additional local contributions to San Pablo Bay from the Napa River. The REE data also imply that some sediment is

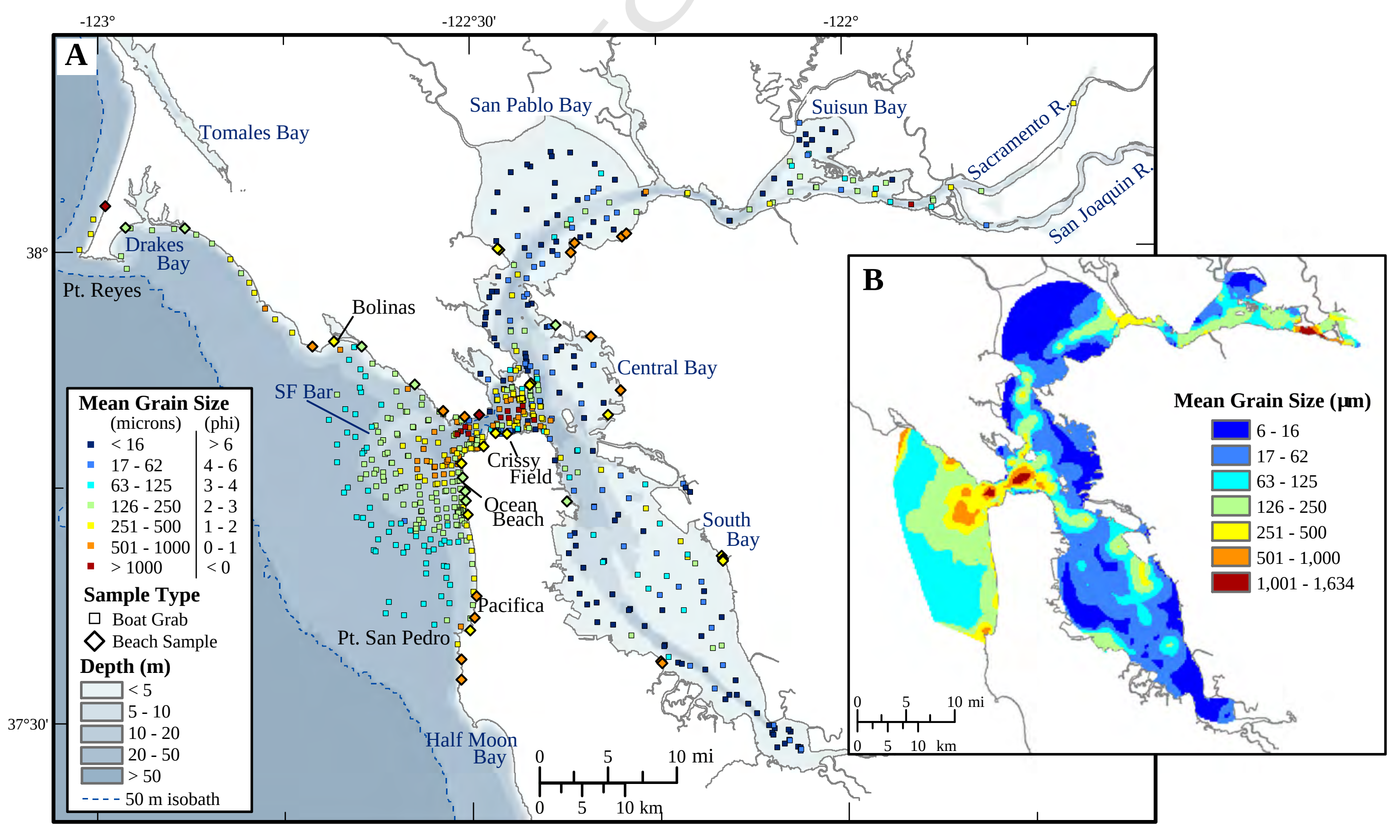


Fig. 10. A) Mean grain size of surface sediment samples, and B) interpolated surface of mean grain size.

Please cite this article as: Barnard, P.L., et al., Integration of bed characteristics, geochemical tracers, current measurements, and numerical modeling for assessing the provenance..., Marine Geology (2012), http://dx.doi.org/10.1016/j.margeo.2012.11.008

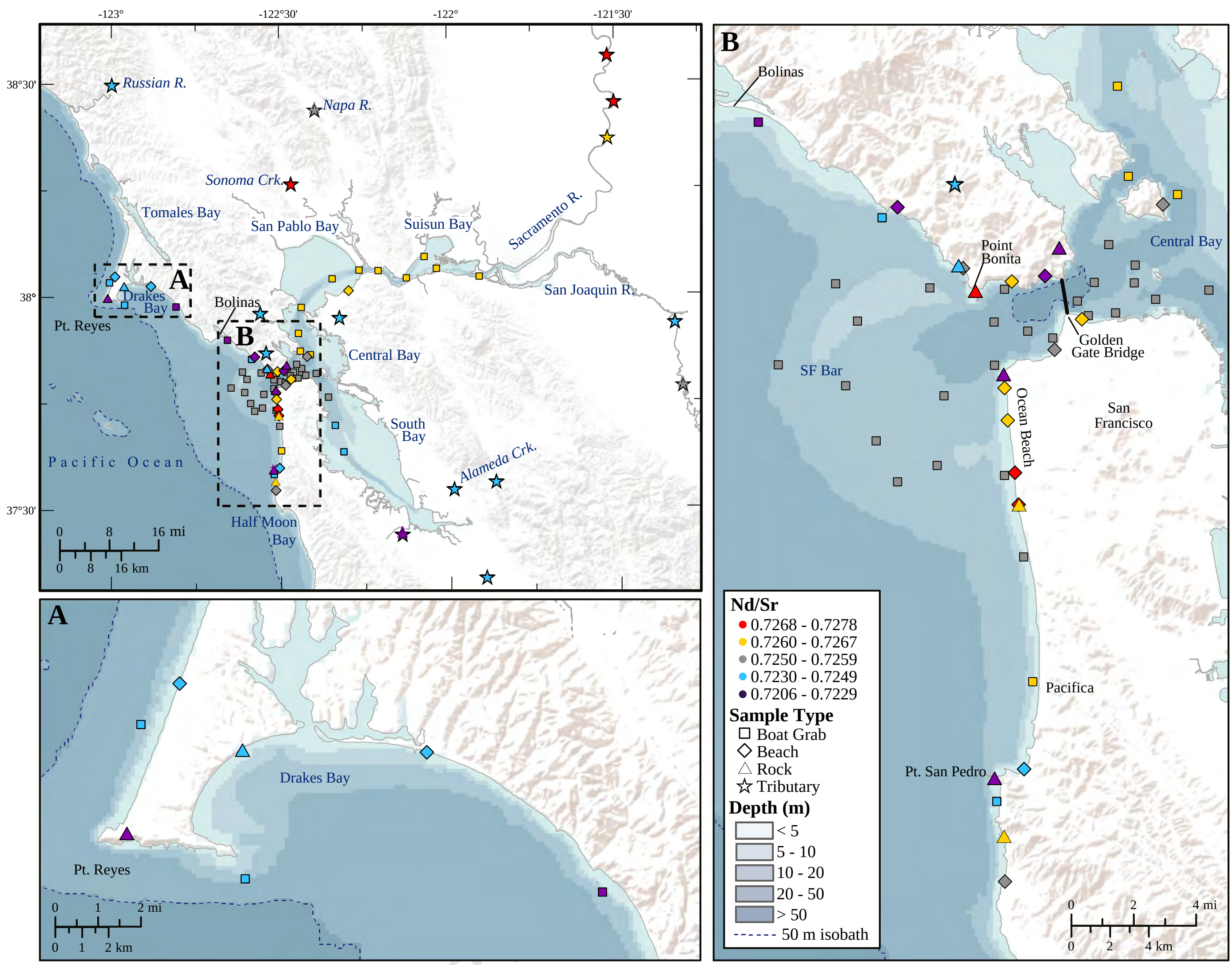


Fig. 11. Nd:Sr isotopic composition ratio of samples (Rosenbauer et al., this issue).

introduced into the Suisun Bay and San Pablo Bay from the San Joaquin River. Based on the isotopic signatures, some component of sand-sized sediment exits San Francisco Bay proper and is then carried southward along the outer coast by prevailing currents. Nd/Sr isotopic ratios also reveal regions of localized sediment accumulation such as basalt being uniquely deposited around the Golden Gate Bridge. The sandy sediment in the southern half of South Bay is derived from local tributaries, primarily Alameda Creek, with no sediment evident from the Sacramento or San Joaquin Rivers.

On the outer coast of Pt. Reyes north of the Golden Gate Bridge, beach-sized material appears to be derived from the discharge of the Russian River with smaller contributions likely from local streams and sandstone outcrops. On the inner coast south of Pt. Reyes there is some material derived from the erosion of the granitic headland that seems contained within Drakes Bay. Most of the sediment on and offshore from Pt. Reyes to the Golden Gate Bridge is consistent with sandstone outcrops at Pt. Reyes and likely other locally-derived geochemically-similar material along the northern open coast. This material mixes with sediment transiting the Golden Gate from within the Bay and some of this material is carried back into Central Bay and partly into South Bay through tidal currents, and some transported southward along and onto Ocean Beach. The beach and offshore sands along the coast south of the Golden Gate are an amalgamation of material transported alongshore from north of the Golden Gate mixed with sediment derived from within the Bay, primarily from the Sacramento River, as well as material derived from local outcrops and creeks (Rosenbauer et al., this issue). Distinct transport pathways were not discernible from the REE results alone, but aided in interpretation of the isotopic data.

4.3.2. Heavy minerals

Samples from beaches, seafloor, local drainages and cliff outcrops are grouped into two major and three minor classes on the basis of cluster analysis of the heavy mineral abundance (Fig. 13). Twenty-two of the 42 samples fall into class 1 (Sierran), which is characterized by hornblende, hypersthene, and zircon, and occurs throughout the estuary west of Carquinez Strait, through the Golden Gate and southward along the coast. Class 2 (Golden Gate) consists of six samples and is similar to class 1, but has far less hypersthene, more zircon, and a more restricted geography near the Golden Gate. The remaining 14 samples are in geographically restricted areas (Franciscan, Bay streams, and Marin classes) or are outliers unrelated to any other samples. The wide distribution of samples from class 1 indicates that the sand is present throughout the estuary and out of the Golden Gate, but no directional trend is evident in either the abundance of the individual minerals or the weighting from the cluster analysis (Wong et al., this issue).

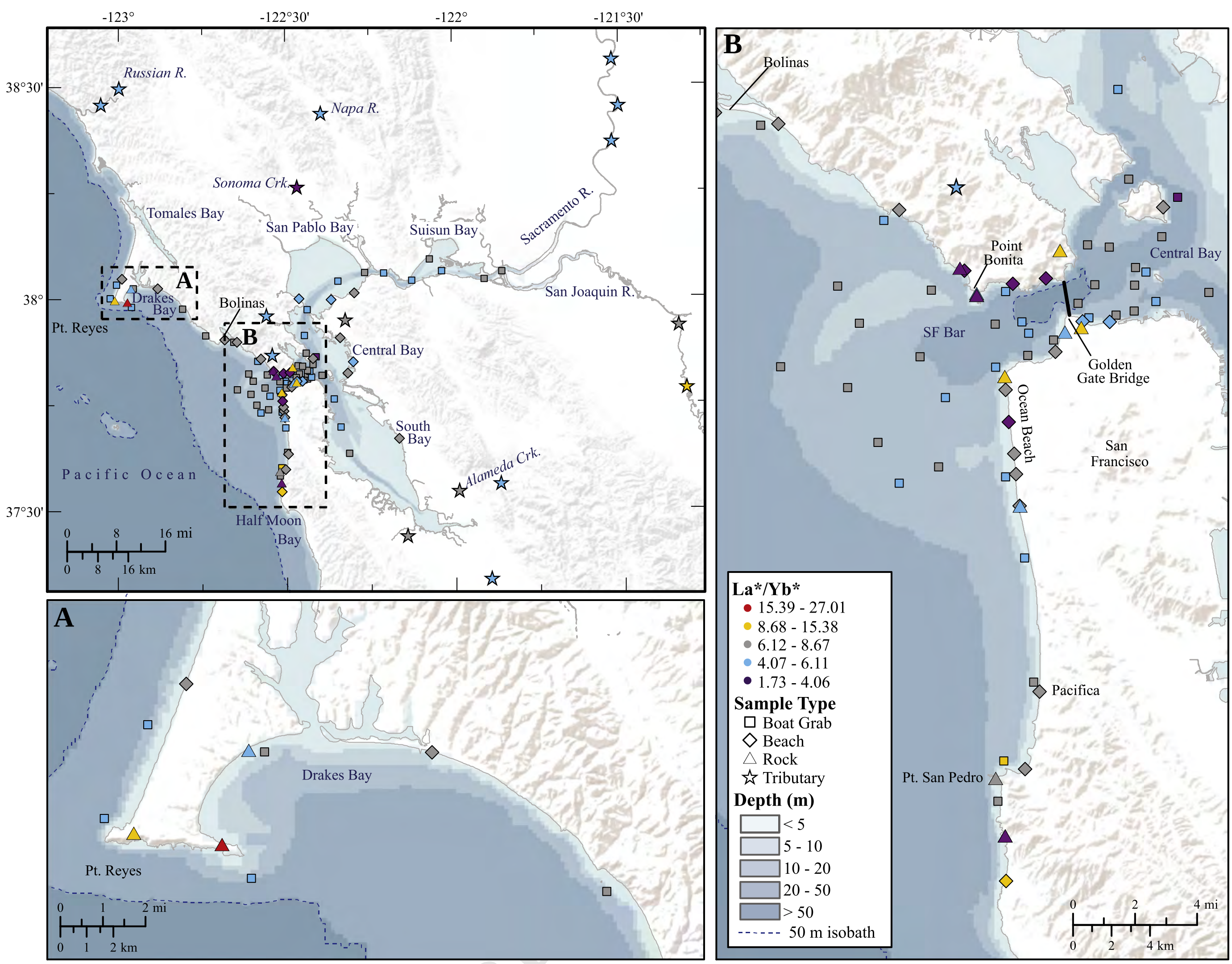


Fig. 12. Representative rare earth element (REE) ratio of samples, lanthanum (La):ytterbium (Yb) (Rosenbauer et al., this issue).

4.3.3. Semi-quantitative X-ray diffraction bulk sand mineralogy

Beach and offshore sands north of the Golden Gate are derived predominantly from Franciscan rocks eroded by local streams and by the larger Russian River. Sediment from the Russian River moves south along the coast and around Point Reyes (Fig. 14). Rock outcrops provide sources for a component of sand for local beaches and near-shore samples, but are diluted with other sources from longshore transport. The general sediment signature north of the Golden Gate can be traced into Central Bay and across the Bay mouth to the south. Most beach and offshore sands south of the Golden Gate are derived from local outcrops and creeks, longshore transport from north of the Golden Gate, and sediment from the Sacramento and San Joaquin Rivers that transits through San Francisco Bay. Local sources or more distant sources can dominate at any particular beach south of the Golden Gate.

The area around the Golden Gate Bridge is a zone of mixing of sediment from various sources including longshore transport from north of the Golden Gate, westward transport from the Sacramento–San Joaquin and Napa–Sonoma drainages, and northward transport from the area of north Ocean Beach into the southern Bay mouth along Crissy Field. Local sources are prominent for beaches along the Marin Headlands. Beaches just southeast of the Golden Gate receive sand from erosion of local Franciscan sandstone, mixed sediment of the Sacramento–San Joaquin Rivers, and the coast north of the Golden Gate, and from Ocean Beach. The remainder of San Francisco Bay receives sediment predominantly from the Sacramento and San Joaquin Rivers. However, sediment from Napa River and Sonoma Creek can be identified in San Pablo Bay and the South Bay area, and likely forms a small component of Central Bay sediment. Local streams flowing into the southernmost portion of South Bay are recognized in nearby sediments. Sediment from Suisun Bay also receives sediment derived from erosion of the Franciscan Complex, perhaps delivered through small creeks (Hein et al., this issue).

4.4. Biologic, anthropogenic, and volcanic constituents

Organic and inorganic sediment constituents were recovered in 294 samples collected in the San Francisco Bay Coastal System from 1995 to 2010. Both naturally-occurring and displaced remains are used to identify pathways of sediment transport and sites of deposition in the region (Fig. 15). Offshore water commonly intrudes into Central Bay, to the southern end of South Bay and the middle of San Pablo Bay, and occasionally as far east as Suisun Bay, as evidenced by the presence of marine-indicating organisms such as benthic and planktic foraminifera, ostracods, diatoms, and radiolaria. In contrast, estuarine waters flow from San Francisco Bay out onto the San Francisco Bar and along the coast, as demonstrated by the recovery of estuarine ostracods and benthic foraminifera in nearshore marine

Please cite this article as: Barnard, P.L., et al., Integration of bed characteristics, geochemical tracers, current measurements, and numerical modeling for assessing the provenance..., Marine Geology (2012), http://dx.doi.org/10.1016/j.margeo.2012.11.008

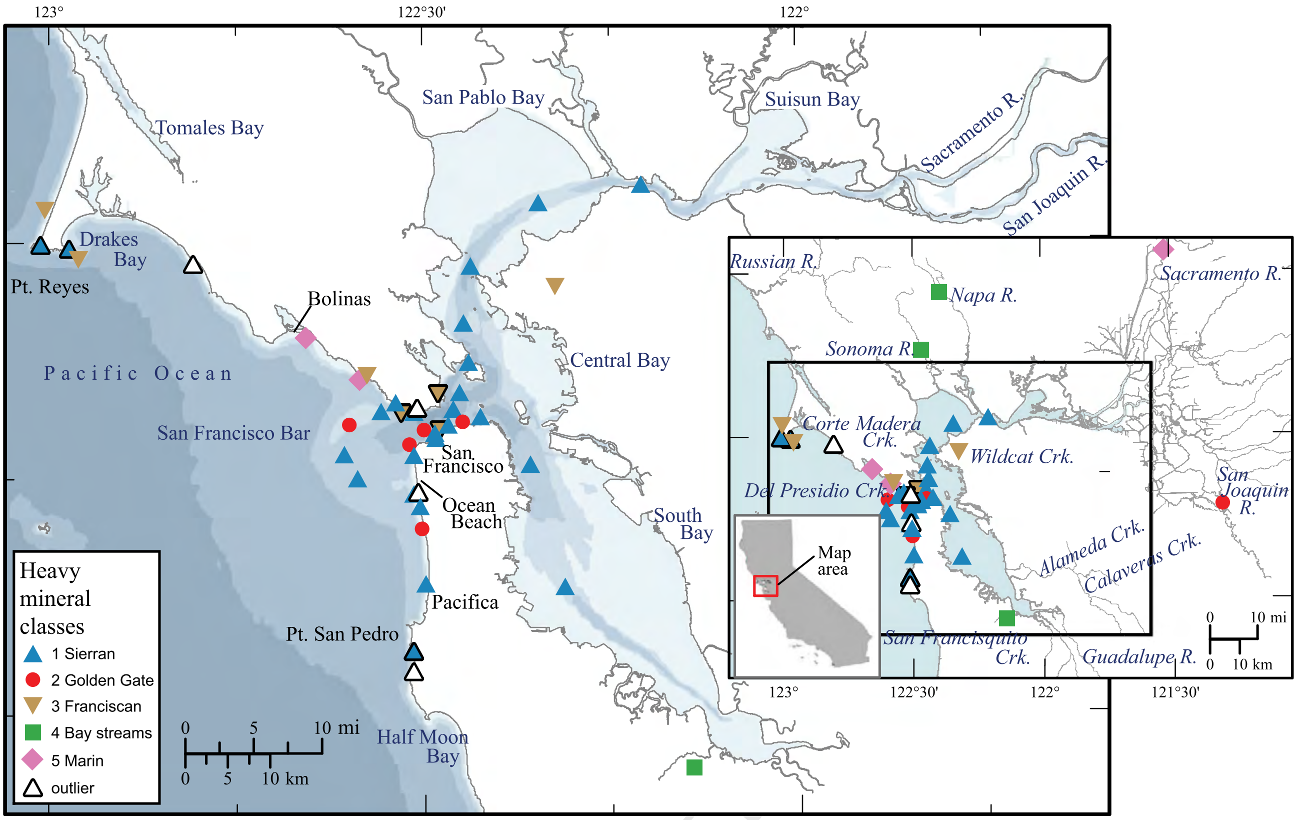


Fig. 13. Distribution of primary heavy mineral classes as determined by cluster analysis. Symbols for cliff rock samples that can be assigned to classes of unconsolidated sediment are outlined in black (Wong et al., this issue).

samples. Biota which inhabits the periphery of the Bay, such as marsh benthic foraminifera and freshwater gastropods and ostracods, commonly are transported to the middle of the sub-embayments of the estuary. Similarly, terrestrially-derived welding slag and glass microspheres are found in the middle of sub-embayments and outside along the outer coast, far from any docks or roads that are presumed to be their source. Lastly, volcanic glass shards originating from the Sacramento and San Joaquin River watersheds were recovered throughout the Bay, including the extreme end of South Bay and along the open coast south to Pt. San Pedro. From these data, we can conclude that sediment is transported from the Delta to all regions of the Bay and out into the offshore realm, as well as from the marine realm back into San Francisco Bay. The channel in Suisun Bay, San Pablo Bay, Central Bay, and South Bay, and the Golden Gate, are conduits for sediment movement and sites where scouring occurs. However, the transport directions inferred from the biologic, anthropogenic, and volcanic constituents utilized in this portion of the study should be considered with caution, as they are derived from bulk sediment samples, and the hydraulic properties of the constituents considered here are not necessarily consistent with the beach-sized sand fraction isolated for the other techniques. Therefore, while we have used these constituents for supporting evidence in the development of the conceptual sand transport model, we have not included the results with the other eight techniques in the semi-quantitative integration. Nevertheless, this technique clearly demonstrates the well-mixed nature of the estuary, and that fresh, brackish, and marine constituents penetrate into all reaches of the Bay.

4.5. Bedform asymmetry

The mean grain size of bedform sediment samples ranged from 0.014 mm to 1.54 mm (Fig. 10; mean = 0.34 mm, $\sigma = 0.28$), indicating that bedform sediment is a potential source of beach sand, defined here as 0.15–0.50 mm. The direction and degree of bedform asymmetry are indicative of sediment transport direction; bedform asymmetry calculations suggest an ebb-dominated system (Fig. 16), with a mean net ebb asymmetry for the entire system of 5%, and significantly ebb-oriented bedforms at the mouth of San Francisco Bay (11% ebb asymmetry), in San Pablo Bay (7% ebb asymmetry) and Suisun Bay (8% ebb asymmetry). Only South Bay exhibits slight flood-orientation (2% flood asymmetry), while Central Bay exhibits only a slight ebb preference (1% ebb asymmetry). Cross-sections of bedform asymmetry across the narrowest section of Suisun Bay (20% ebb asymmetry), the entirety of Central Bay (12% ebb asymmetry), and the inlet mouth (5% ebb asymmetry) all suggest that the Bay is a net exporter of sand to the open coast. In addition to mean overall ebb orientation of the bedforms, there are a number of large regions where ebb- or flood-directed transport is clearly dominant, such as the southern portion of Central Bay (ebb), through the center of the Golden Gate (ebb), and along the southern margin of the Golden Gate (flood). The asymmetry measurements significantly agree (up to ~76%) with annual residual transport directions derived from numerical modeling (see Section 4.7), and the orientation of adjacent, flow-sculpted seafloor features such as mega-flute structures (Barnard et al., this issue-a).

4.6. Measured residual currents

Current measurements show that residual currents were predominantly ebb-oriented in the central and northern portions of Suisun Bay (Fig. 17A; Walters and Gartner, 1985; Walters et al., 1985). Measurements in September 1978 showed that during spring tides, a down-estuary flow across the northern portion of Suisun Bay resulted from the tidally-driven residual flow dominating over the density-driven up-estuary flow. During neap tides, the density-driven flow dominated because of decreased vertical mixing and weakened residual flow. A comparison of meteorological

Please cite this article as: Barnard, P.L., et al., Integration of bed characteristics, geochemical tracers, current measurements, and numerical modeling for assessing the provenance..., Marine Geology (2012), http://dx.doi.org/10.1016/j.margeo.2012.11.008

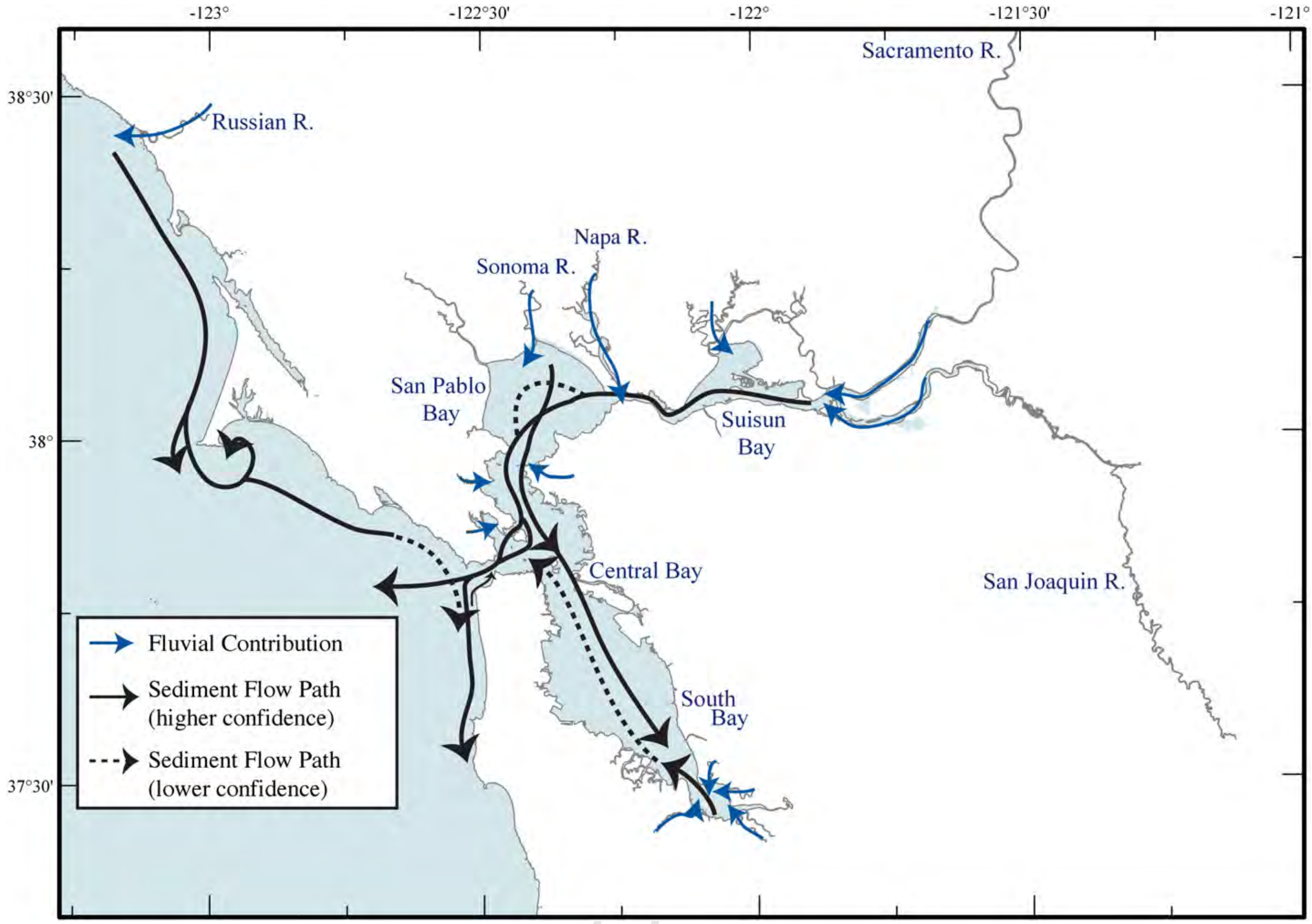


Fig. 14. Summary of beach-sized sand transport pathways based on semi-quantitative X-ray diffraction results. Modified from Hein et al. (this issue).

and current meter data did not reveal any wind-driven component in the residual circulation of Suisun Bay.

In South Bay, measurements obtained with mechanical current meters during the years 1979 to 1982 indicated a residual tidally-driven current northward along the west side of the main channel and eastward along the northern slope of San Bruno shoals (Fig. 17B; Cheng and Gartner, 1985; Walters et al., 1985). Residual flows over the shoals along the eastern part of South Bay were strongly affected by wind-driven currents; under conditions of a north wind, surface flows in the shallow regions were southward with an ebb-directed return flow in the channel. Although these current measurements are ~thirty years old, previous modeling efforts (Gross, 1997) have shown that while winds contribute significantly, tidal currents are the primary forcing responsible for creation of residual circulation in South San Francisco Bay. Unless wind patterns and magnitudes have changed substantially or changes in bathymetry, freshwater loading, and the tidal prism has significantly altered the tidal regime, it is likely that the measurements are still largely representative of circulation in this sub-embayment.

ADP measurements at the two sites along Crissy Field indicate ebb-directed residual currents for both the January and September 2008 deployments (Fig. 17C). Residuals at the westward site were oriented alongshore while at the eastward site (CF2 — near an inlet to a restored tidal wetland), residual currents were ~20° from the shore-normal direction. The shore-normal current component was almost always directed onshore and likely is responsible for the observed sedimentation and frequent closure of the marsh inlet (Hanes et al., 2011). Surface wind stress (Fig. 17D, bottom panel), has a good correlation ($r = 0.65$) with westward- (ebb) directed residual currents, but no correlation with the north–south or shore-normal residual currents. However, the onshore-directed residual currents show a strong ($r = 0.80$) correlation with significant wave heights measured at the San Francisco Bar outside the Golden Gate (Fig. 17D, third panel). The occurrence of large ocean waves has been shown to coincide with marsh closure events (Hanes et al., 2011), further indicating that the ocean swell penetrating through the Golden Gate is largely responsible for the nearshore residual currents and sedimentation along Crissy Field beach.

Along the outer edge of the ebb-tidal delta, residual currents were directed seaward (Fig. 17C). For the winter measurement period in 2006, waves averaged 2.6 m with a maximum of 5.6 m in ~14 m water depth at Site 5, but the depth-averaged residual current (consistently offshore) was poorly correlated with wave height, indicating the dominant influence of the ebb jet emanating from the Golden Gate. Residual current measurements along Ocean Beach (Sites 2, 3 and TV1, 11 m–14 m water depth) showed a consistent north–north-west direction.

4.7. Numerical modeling

Modeled residual transport is dominantly seaward at the mouth of San Francisco Bay, including through the center of the Golden Gate, and across the ebb-tidal delta (Fig. 18). However, there is a narrow but distinct pathway for flood-directed transport from the northern section of Ocean Beach, around Pt. Lobos, and along Baker Beach,

Please cite this article as: Barnard, P.L., et al., Integration of bed characteristics, geochemical tracers, current measurements, and numerical modeling for assessing the provenance..., Marine Geology (2012), http://dx.doi.org/10.1016/j.margeo.2012.11.008

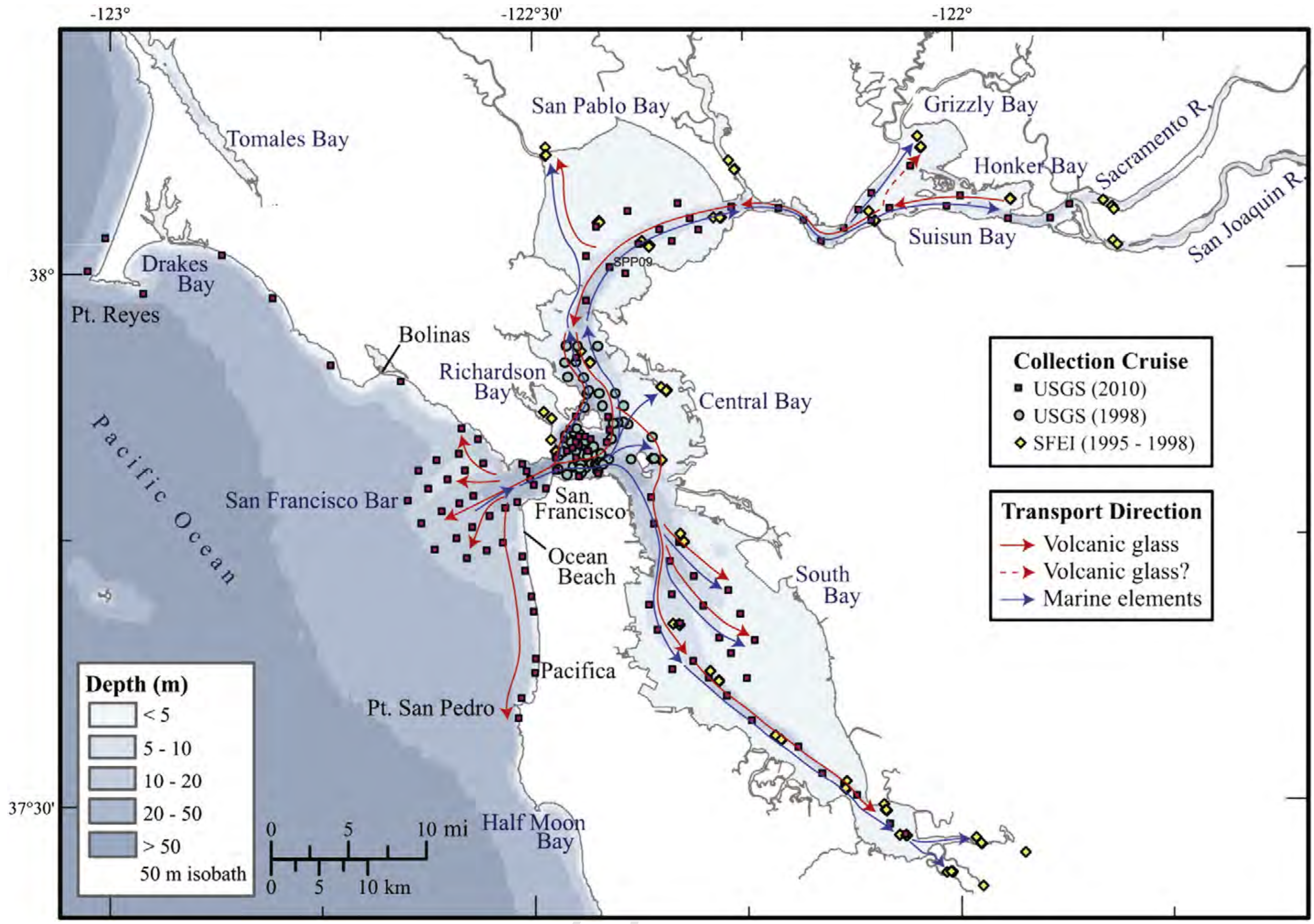


Fig. 15. Location of the sediment constituent study sites and pathways of sediment transport in the San Francisco Bay Coastal System inferred by the presence of marine elements from the offshore realm as well as volcanic glass originating in the Sierras.
Modified from McGann et al. (this issue).

which also is suggested by many of the other analyses in this paper. Inside Central Bay, ebb-directed transport dominates along the periphery, including the southern and northwest sections. Closer to the Golden Gate strait, flood-directed-transport is more prevalent in the center of the inlet, while transport patterns are more complex to the east, but generally agree with the bedform asymmetry patterns for the majority of locations (see Fig. 16). South Bay transport directions are uncertain and often conflict with the other analyses (e.g., only 38% agreement with the bedform asymmetry), but the disagreement is not unexpected as wind-driven gravitational circulation, known to be a key driver of transport patterns in this sub-embayment (Conomos et al., 1985), is not incorporated in the model. Similarly, although modeled transport directions in San Pablo Bay (76% agreement with bedform asymmetry) and to a lesser extent Suisun Bay (65% agreement with bedform asymmetry) are well aligned with the other analyses, the model results are given less weight as density-driven estuarine circulation processes are not simulated and are known to be important in those areas (Monismith et al., 2002).

4.8. Integration of techniques

The consensus beach-sized sand transport directions based on the results for eight of the nine provenance techniques are synthesized in Fig. 19 for each 3×3 km cell. The confidence intervals applied for each technique are listed in Table 4. In the center of the San Francisco Bay Coastal System (i.e., Central Bay, Golden Gate, and ebb-tidal delta), the transport directions and pathways are more robust (i.e., higher confidence: Fig. 19A) and delineated due to the greater sediment sampling density, numerical model calibration and validation, bedform distribution, in situ current measurements, and prevalence of sand-sized material. However, there is substantial regional sampling and geochemical evidence to confidently determine the broad-scale sediment transport pathways throughout the entire system, ranging from the distal sources in the Sacramento, San Joaquin, Napa, and Russian Rivers and Sonoma Creek, through each sub-embayment of the Bay, and along the entire open coast study area.

5. Synthesis and discussion

5.1. Primary sediment sources, sinks and pathways

Through the quantitative integration of eight distinct provenance techniques, the results (Fig. 19) are simplified in a conceptual model of beach-sized sand transport for the San Francisco Bay Coastal System (Fig. 20).

In the northern sub-embayments of San Francisco Bay, Suisun Bay exports sandy sediment to San Pablo Bay, sourced from the Sierras primarily via the Sacramento River, and to a lesser extent the San Joaquin River, in line with previous studies that note the far greater contribution of Sacramento River-derived sediments (Krone, 1979; Porterfield, 1980; Oltmann et al., 1999; Wright and Schoellhamer, 2005). In addition to Sierran sand transported from Suisun Bay, San

Please cite this article as: Barnard, P.L., et al., Integration of bed characteristics, geochemical tracers, current measurements, and numerical modeling for assessing the provenance..., Marine Geology (2012), http://dx.doi.org/10.1016/j.margeo.2012.11.008

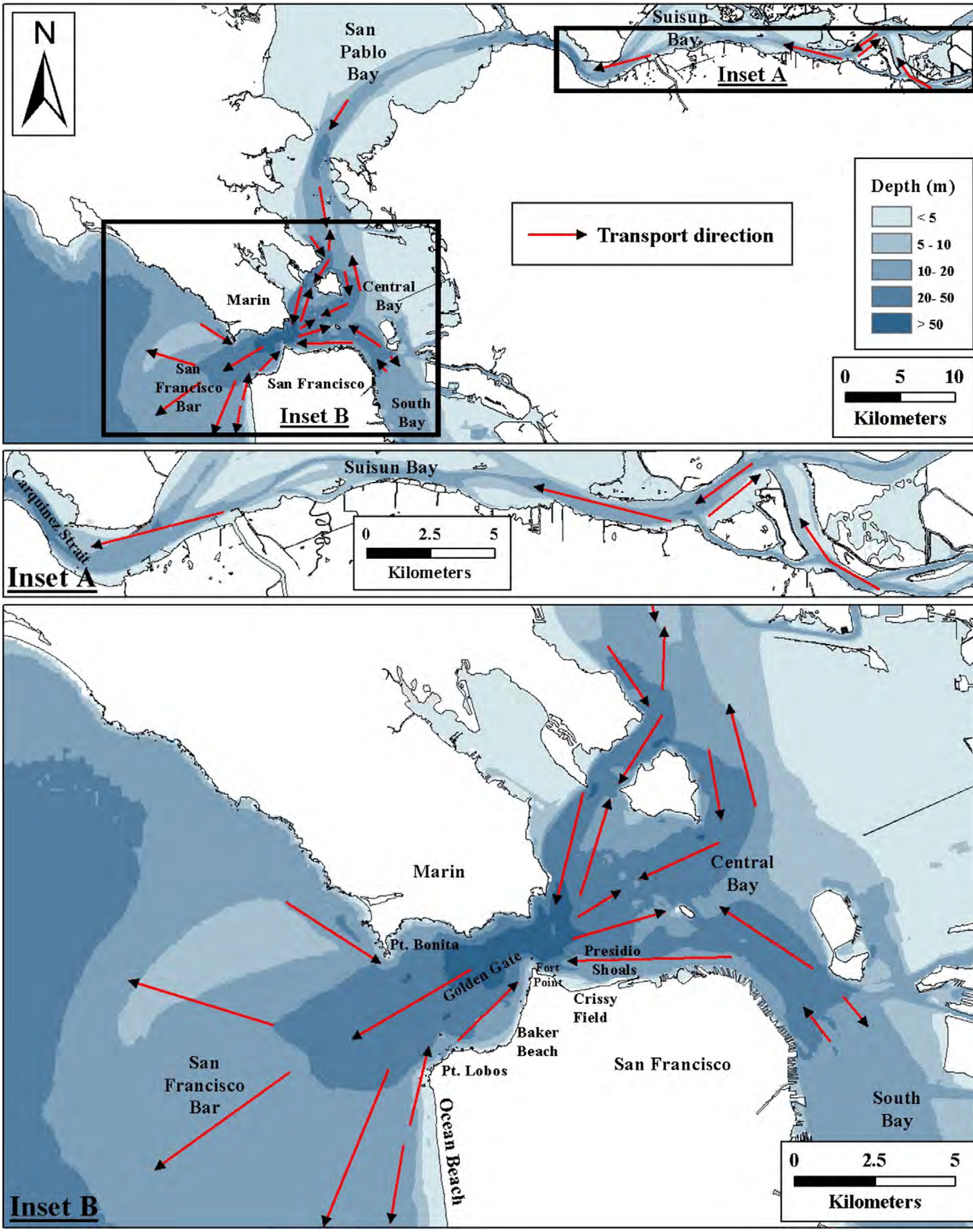


Fig. 16. Inferred sand transport pathways based on agreement between bedform asymmetry and numerical modeling results, simplified from Barnard et al. (this issue-A). Arrow length represents spatial coverage of transport direction agreement.

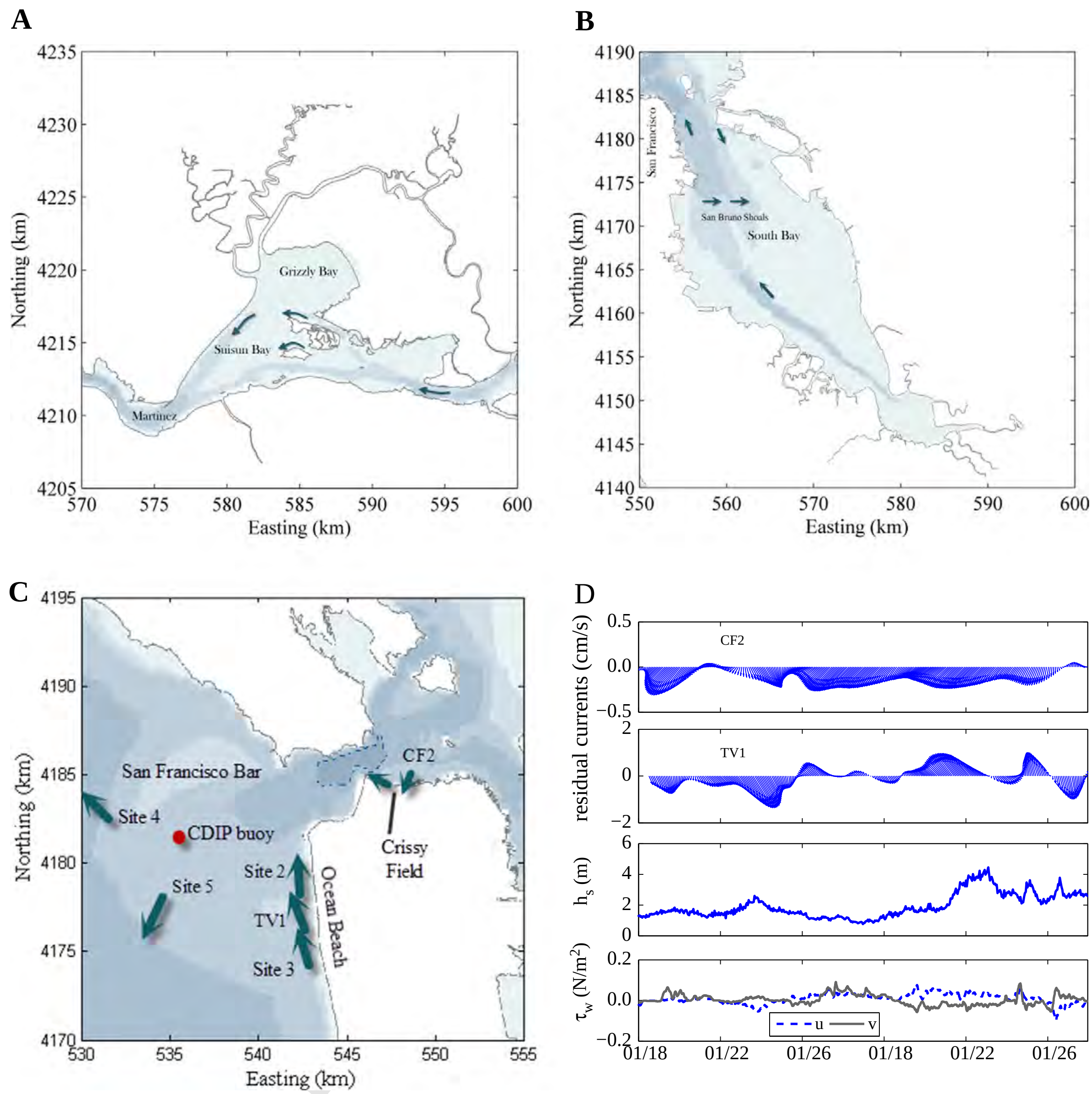


Fig. 17. Depth-averaged residual currents calculated from current measurements in A) Suisun Bay (Walters et al., 1985), B) South Bay (Walters et al., 1985), and C) at the open coast and Crissy Field. Arrow size is not indicative of magnitude or confidence, only direction. Dashed lines show the 50-m isobath. D) Residual currents at CF2 and TV1 (upper 2 panels); third panel — significant wave heights at the San Francisco Bar CDIP buoy; bottom panel — wind stress components calculated using data from the NOAA Ft. Point tide station near CF2 and methods described in Smith (1988).

Pablo Bay receives notable contributions from the Napa and Sonoma drainages, with a net export of sediment to Central Bay. This is the first study we know of documenting sand contributions in the Bay from these two local tributaries, although Porterfield (1980) measured significant quantities of sand in the suspended load (estimate sand transport = ~40 t/day) 10's of kilometers upstream from the Bay outlet of each tributary.

The provenance results demonstrate that South Bay is primarily a sink for beach-sized sand, consistent with the multi-decadal accretionary trend for this sub-embayment (Foxgrover et al., 2004). From the limited number of samples collected within South Bay, it appears that sandy sediment in the southern half of South Bay is derived entirely from local tributaries, particularly Alameda Creek, which is consistent with earlier findings of Conomos (1963). The northern section of South Bay includes sediment derived from both the Central Bay region and the Napa River and Sonoma Creek that enter initially into San Pablo Bay. This is in contrast to the postulation by Gram (1966) that the sand fraction here is entirely locally-derived, and also conflicts with Yancey and Lee (1972) who clearly designate South Bay as a distinct mineral province with sediments derived exclusively from the adjacent tributaries. No evidence of a significant Sierran source in South Bay for beach-sized sand has been detected in the present or prior studies (Conomos, 1963; Gram, 1966; Yancey and Lee, 1972).

Central Bay comprises an amalgamation of sources, but the primary origin of beach-sized sand is from the Sierras via the Sacramento River–Suisun Bay–San Pablo Bay transport pathway, with minor contributions evident from the San Joaquin River, Napa River, Sonoma Creek, local Franciscan sources from the Golden Gate region, and from the open coast north of the Golden Gate. A portion of this sediment is exported to South Bay along the eastern section of the main tidal channel connecting the two sub-embayments, as indicated by bedform asymmetry, current measurements, and XRD. Conversely, along the western end of the channel, South Bay exports sediment

Please cite this article as: Barnard, P.L., et al., Integration of bed characteristics, geochemical tracers, current measurements, and numerical modeling for assessing the provenance..., Marine Geology (2012), http://dx.doi.org/10.1016/j.margeo.2012.11.008

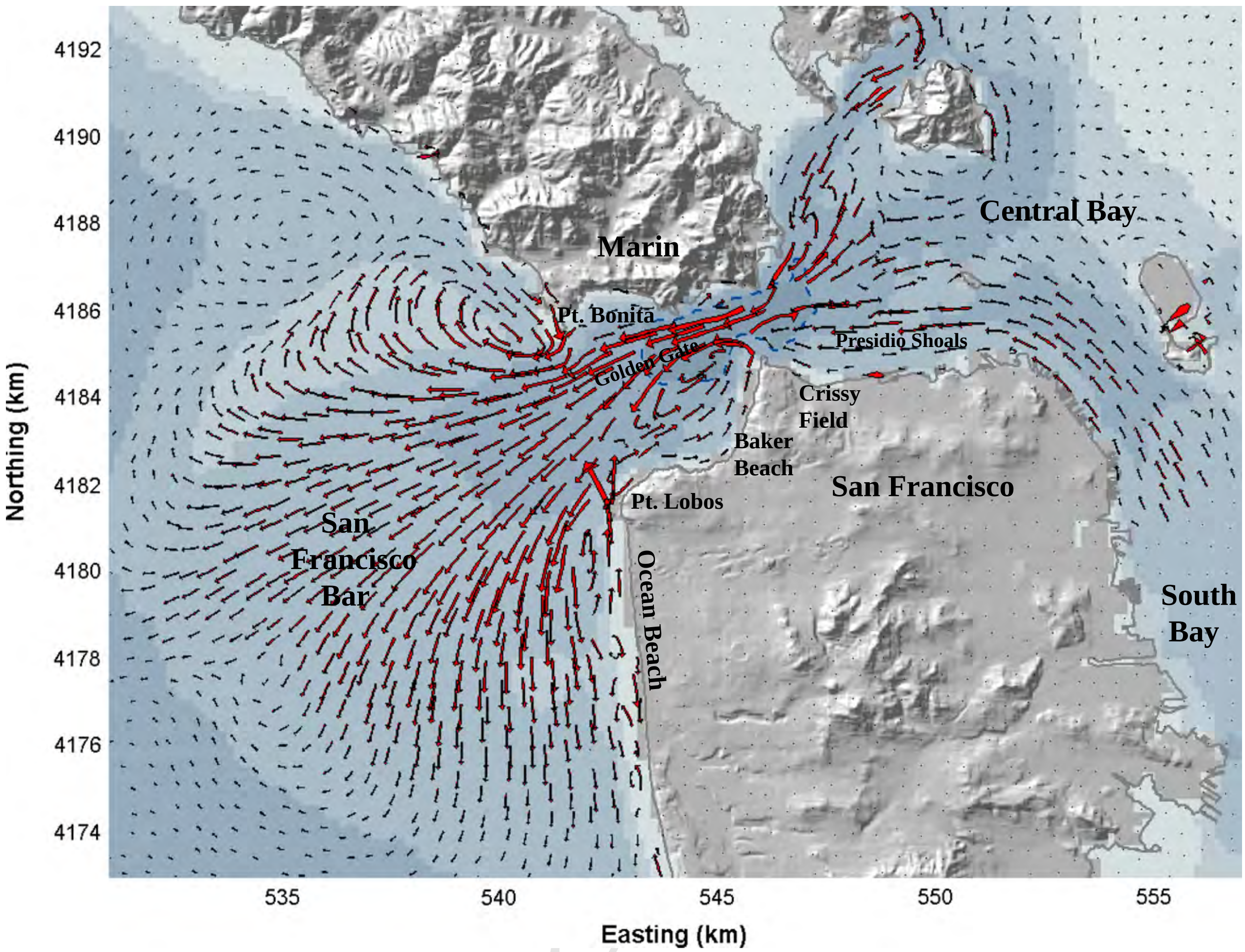


Fig. 18. Modeled residual sediment transport in the San Francisco Bay Coastal System. Axes in UTM coordinate system. Dashed lines show the 50-m isobath. Size of arrows indicates relative magnitude of residual transport.
Modified from Elias and Hansen (this issue).

to Central Bay along a distinct pathway that wraps around the northeastern and northern perimeter of the San Francisco peninsula toward the Golden Gate, clearly delineated through numerical modeling, bedform asymmetry, grain size, current measurements, and XRD.

Beach-sized sand in Central Bay, the Golden Gate, the ebb-tidal delta and southern open coast is strongly geochemically linked. This link is further reinforced by bedform asymmetry, numerical modeling, and current measurements. Through the center of the Golden Gate, net transport is dominantly seaward to the ebb-tidal delta, with among the highest weighted confidence values in the entire study area. The sediment is derived from numerous locations, most prominently Sierran from the Sacramento River, with additional contributions from the San Joaquin River, Napa River, and Sonoma Creek. Local Franciscan sources are particularly evident on local pocket beaches fed by adjacent outcrops of basalt, chert, and serpentinite. The samples collected from this zone of intense mixing also incorporate sand that moves south via longshore transport from north of the Golden Gate, and northward transport from the area of Ocean Beach, along Baker Beach, and westward along the northern shoreline of San Francisco (i.e., Crissy Field).

The ebb-tidal delta receives sediment primarily from the Golden Gate (dominantly Sierran), and secondary inputs that move south from the northern coast, derived chiefly from the sandstone outcrops near Pt. Reyes, and more proximal Franciscan outcrops. From the ebb-tidal delta, the majority of sand-sized material moves both alongshore to the south and offshore onto the inner continental shelf.

Along the northern outer coast, sand is derived from the Russian River, particularly north of Pt. Reyes, mixing with granitic and sandstone outcrops near Pt. Reyes, and moving south with additions from Franciscan rocks in cliffs and drained by local streams closer to the Golden Gate. This material moves south by longshore transport, with some material entering Central Bay, possibly around Pt. Bonita, while the rest moves across the ebb-tidal delta toward the southern open coast. The beaches immediately north of the Golden Gate are sourced almost entirely from locally-derived Franciscan outcrops of chert, basalt, and shale.

Beach and nearshore sediment along the southern open coast represents a complex mixture of sand from the northern coast combined with sediment sourced primarily from the Sacramento River (i.e., Sierran) via the Bay, as well as material derived from local outcrops and creeks, with the source contributions varying with alongshore location. Sediment found at northern Ocean Beach is linked geochemically to Baker Beach (and the adjacent Golden Gate sand wave field), and Crissy Field, representative of the dominant Sierran source, and consistent with the geochemistry, numerical modeling, in situ measurements, and bedform asymmetry that document a distinct pathway for sediment into San Francisco Bay along the northern shoreline of the San Francisco peninsula. However, sand at southern Ocean

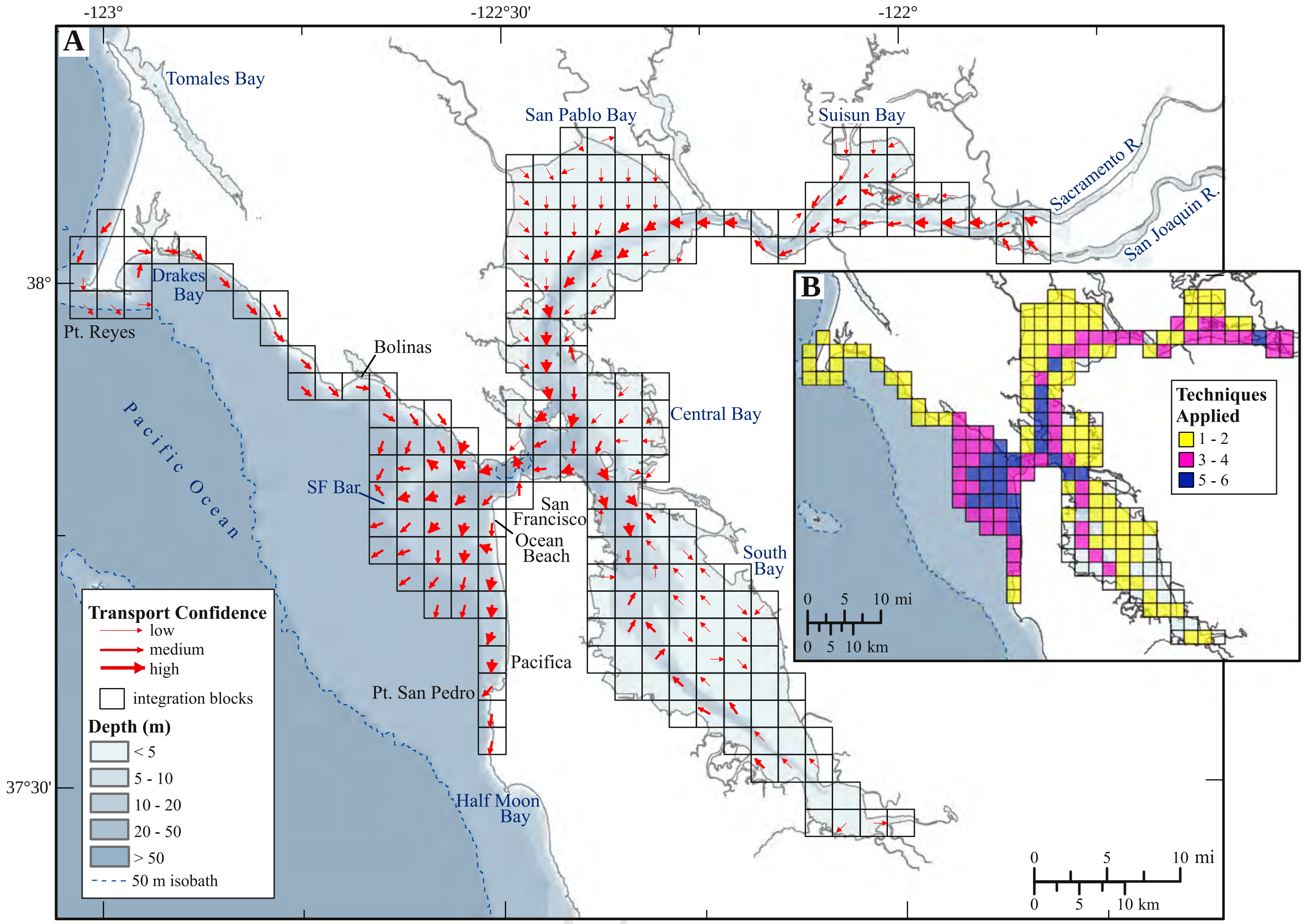


Fig. 19. A) Calculated transport directions based on the integration of the provenance techniques. B) Number of techniques applied for each grid cell to determine the final transport directions.

Beach and offshore are consistent with sand locally eroded from beach-backing cliffs comprising the Colma Formation, distinguished by relatively high magnetite concentrations.

5.2. Implications for regional sediment management

From the above assimilation, a suite of distinct and important transport pathways emerge that have significant implications for regional sediment management (Fig. 20).

5.2.1. Sacramento/San Joaquin Rivers (i.e., Sierran source) → Suisun Bay → San Pablo Bay → Central Bay → Golden Gate → ebb-tidal delta → southern open coast and continental shelf (sink)

For the San Francisco Bay Coastal System, based on the multiple techniques for assessing sand provenance described herein, the Sierra Nevada Range is the dominant source of beach-sized sand, which is actively transported into and through the Bay to the mouth of San Francisco Bay, and along the southern open coast, robustly supporting evidence of this source and pathway from earlier studies that looked at different grain sizes (Gilbert, 1917; Moore, 1965; Yancey and Lee, 1972). Clearly, the sharp reduction in sediment supply from the Sierras over the last century (Wright and Schoellhamer, 2004; Ganju et al., 2008; Singer et al., 2008; Schoellhamer, 2011) via the Sacramento and San Joaquin Rivers, due to the cessation of the hydraulic mining signal and major watershed modifications (Gilbert, 1917; Knowles and Cayan, 2004), has had a significant impact on the sediment supply to the entire region. This dominant pathway for beach-sized sand material destined for the open coast directly intersects the two major active aggregate mining regions in San Francisco Bay, Suisun Bay and Central Bay (Hanson et al., 2004). Also within the 20th century, over 200 million m^3 (~170 Mt, assuming a bulk density of 850 kg/m^3 per Porterfield, 1980) of sediment was directly removed from the San Francisco Bay Coastal System through dredging, aggregate mining, and borrow pit mining, including at least 54 million m^3 of sand-sized or coarser sediment from Central Bay (Dallas and Barnard, 2009, 2011). Together, these changes have contributed to ~240 million m^3 of sediment loss to the San Francisco Bay Coastal System, as estimated from bathymetric change surveys spanning the last fifty years (Capiella et al., 1999; Foxgrover et al., 2004; Jaffe and Foxgrover, 2006; Hanes and Barnard, 2007; Jaffe et al., 2007; Fregoso et al., 2008; Barnard and Kvitek, 2010). Over 150 million m^3 of measured volume loss during this period is from the sand-dominated substrates of Central Bay, the Golden Gate, and ebb-tidal delta (Hanes and Barnard, 2007; Fregoso et al., 2008; Barnard and Kvitek, 2010). Coastal erosion along the outer coast south of the Golden Gate during this same period is the highest for the entire coast of California (Hapke et al., 2006, 2009), and has accelerated by 50% between Ocean Beach and Pt. San Pedro since the 1980s (Dallas and Barnard, 2011). As further evidence of the continued reduction in sediment supply within the system, Schoellhamer (2011) observed a 36% step decrease in suspended sediment concentrations inside the Bay between water years 1991–1998 and 1999–2007. At the mouth of San Francisco Bay, Barnard et al. (2012a) documented a fining of mean

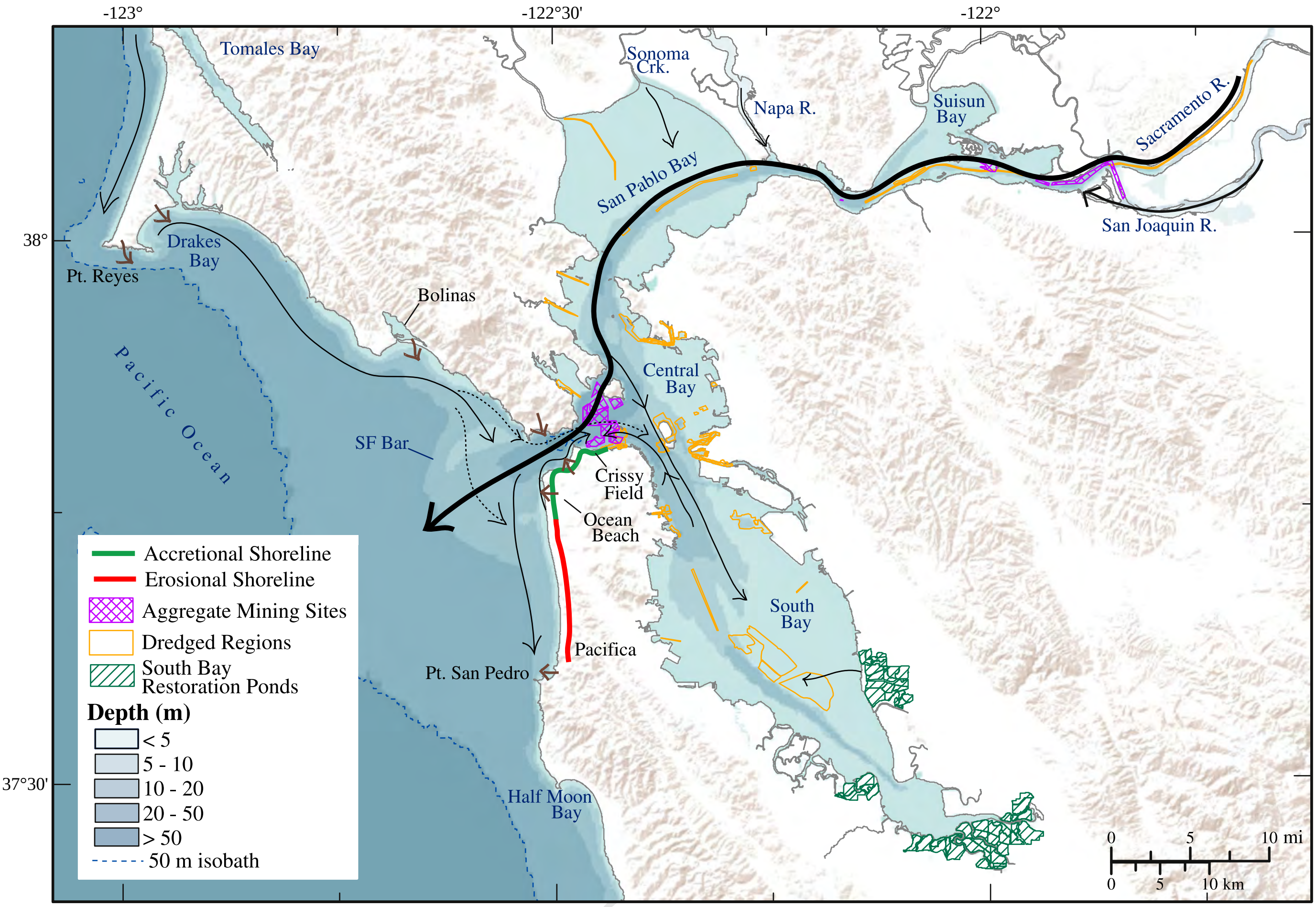


Fig. 20. Final conceptual model of the primary beach-sized sand transport pathways in the San Francisco Bay Coastal System, based on the integration of the provenance techniques. Notable anthropogenic activity locations and significant shoreline change trends are also plotted.

grain size by ~0.025 mm from 1997 to 2008, in particular progressively finer sediment along the outer reaches of the ebb-tidal delta between 2002 and 2007, indicating a reduction in the coarser sand supply.

Looking forward over the next century, the National Research Council (2012) projects 92 cm (range 42–166 cm) of sea level rise by 2100 for San Francisco Bay. Outer coast and Bay beaches, an important line of defense against storm impacts and rising sea levels, will require increasingly higher rates of sand supply to prevent erosion and landward migration, which in many locations would threaten fringing development. Using Global Climate Models linked to regional physical and ecological models in the San Francisco Bay area through 2100, Cloern et al. (2011) projected reduced fluvial discharge from the Sacramento–San Joaquin Delta, a further decline in suspended sediment concentration, and a marked increase in the frequency of extreme water levels. At present, aggregate mining removes approximately 0.9 million m^3/yr of sand and gravel-sized sediment in Central Bay and Suisun Bay (Hanson et al., 2004), while dredging removes about 3 million m^3/yr of sediment, with the majority of this material permanently removed from the San Francisco Bay Coastal System (Dredged Material Management Office, 2008; Keller, 2009; San Francisco Estuary Institute, 2009). Together, these losses exceed the present annual sediment supply from the Sierras and local watersheds combined (Schoellhamer et al., 2005; McKee et al., this issue). Therefore, management of the current sediment inventory in the Bay will be critical.

5.2.2. Ocean Beach → Baker Beach → Crissy Field

Multi-decadal erosion and contraction of the ebb-tidal delta (Hanes and Barnard, 2007; Dallas and Barnard, 2011) have modified sediment transport patterns along Ocean Beach, effectively driving more sediment toward the northern end of the beach and less toward the southern end (Hansen et al., this issue). The modeled patterns are supported by observed beach and nearshore changes over interannual (Hansen and Barnard, 2010) and multi-decadal time scales (Dallas and Barnard, 2011; Barnard et al., 2012a), including an ~3 fold increase in the rates of shoreline accretion at the north end over the last several decades, and correspondingly higher rates of erosion at the south end that have led to significant infrastructure damage (Barnard et al., 2011a). As the northern shoreline has continued to extend seaward, increasingly higher volumes of northward-moving sand are no longer trapped by Pt. Lobos at the north end of Ocean Beach, and instead move toward Baker Beach and eventually into Central Bay at Crissy Field (Fig. 20). For example, over the last decade, sedimentation forced the relocation of a tide gauge and caused shoaling within the adjacent yacht harbor. These three sites have now been linked geochemically in this study, and recently accelerating rates of shoreline accretion at Baker Beach and Crissy Field correlate temporally with observed changes at northern Ocean Beach (Dallas and Barnard, 2011). These trends and correlative impacts are expected to continue (Hansen et al., this issue) as higher sea levels and further reductions in sediment supply drive further contraction of the ebb-tidal delta.

5.2.3. Northwest South Bay → southern Central Bay → Golden Gate

This distinct pathway, substantiated by a wide range of provenance techniques (i.e., XRD, bedform asymmetry, current residuals, numerical modeling), intersects three lease sites on Presidio Shoals in southern Central Bay (see Fig. 18), where active aggregate mining takes place (Fig. 20). Bathymetric change analysis from 1997 to 2008 across the lease sites records a volume loss of ~2.3 million m^3; most of this attributed to sand and gravel removal by aggregate mining (Barnard and Kvitek, 2010), significantly reducing the sediment available for transport to the mouth of San Francisco Bay and adjacent beaches.

5.2.4. South Bay local tributaries (source) → South Bay (sink)

The integrated provenance results demonstrate that South Bay is primarily a sink of beach-sized sand (with the notable exception of the northwest portion as described in the previous section), particularly the southern half, where local tributaries, namely Alameda Creek and its tributary, Calaveras Creek, are the primary sources, with no evidence of a Sierran component. As South Bay is the only sub-embayment with a recent accretionary trend (Foxgrover et al., 2004), and is the site for the largest tidal wetland restoration on the west coast, the prospects that the newly created tidal wetlands will keep up with sea level rise are greater than for regions that rely directly on a Sierran source where sand supply continues to trend downward.

5.2.5. Russian River (source) → Pt. Reyes → ebb-tidal delta → southern open coast (sink)

In contrast to earlier analyses of heavy minerals contained in beach and inner shelf sediments that suggested that the Russian River was not a major source of sediment in the vicinity of Pt. Reyes south to the Golden Gate (Cherry, 1964; Minard, 1971; Demirpolat, 1991), the geochemical evidence here definitively links the Russian River-derived sand to beach sand immediately north of Pt. Reyes. XRD analyses further suggest that the Russian River influence may extend as far downcoast as the ebb-tidal delta and southern open coast. It is possible that the finer sand grain sizes (<0.25 mm heavy minerals) in the prior studies would have been more easily advected offshore at the Russian River mouth and at Bodega Head, effectively removing them from the littoral system, although the density of these heavy minerals would make them more hydraulically comparable to coarser, more commonly-occurring beach mineral grains. Nevertheless, depending on the impact of future climate change on Russian River discharge rates, this source may help to mitigate coastal erosion pressures on outer coast beaches driven by rising sea levels and the projected continued reduction in the Sierran sediment supply.

6. Conclusions

Through the unique integration of nine separate provenance techniques, the sources and pathways for beach-sized sand in a complex coastal–estuarine system have been robustly established. The consensus results highlight the regional impact of a sharp reduction in the primary sediment source to the San Francisco Bay Coastal System over the last century – the Sierras – in driving massive erosion of the Bay floor, ebb-tidal delta, and the highest regional shoreline retreat rates in California along the adjacent outer coast. In addition, this work also highlights the need to more efficiently manage existing in-Bay sediment resources, as active aggregate mining and dredging occurs along well-defined sand transport pathways that carry sediment toward outer coast beaches, at removal rates that exceed the present-day sediment supply rates from all San Francisco Bay watersheds. Given the observed reduction in contributions from the Delta, and the relative increase of the sediment supply from local tributaries which may be enhanced in the coming decades due to flood control strategies within local watersheds, future beach-sized sand provenance should evolve over the course of the next century to represent these more proximal sources. The comprehensive approach introduced here also definitively established other, previously unresolved secondary sources of sand input to the system that may contribute to the sustainability of beaches on a local and system-wide scale, including the Russian and Napa Rivers, and eroding cliff and bluff sources, such as in the vicinity of Pt. Reyes, within and adjacent to the Golden Gate (e.g., Franciscan Formation), and along the southern open coast (e.g., Colma Formation). Cross-validating geochemical analyses, numerical modeling, physical process measurements, and proxy-based techniques (e.g., bedform asymmetry, grain size morphometrics) is an effective approach for confidently defining sources, pathways and sinks of sand in complex coastal–estuarine systems.

7. Uncited reference Q3

Walters, 1982

Acknowledgements

This research was supported by the United States Army Corps of Engineers, San Francisco and Los Angeles Districts, the California Sediment Management Workgroup, and the United States Geological Survey Coastal and Marine Geology Program. Thanks to Scott Wright (USGS Water Resources Division), the San Francisco Estuary Institute, and Applied Marine Sciences, Inc., for collecting samples used in this research. Many thanks to the crew of the R/V *Snavely*, Pete Dal Ferro, Jackson Currie, Jamie Grover, Tim Elfers, Jenny White, and Greg Gable (all of the USGS), and to the numerous folks that assisted with sample collection including: Dan Hoover, Jeff Hansen, Kira Runtzel, Julia Klofas, and Leticia Diaz (all of USGS), Isaac Jones (San Francisco State University), and Brenda Goeden, Carolynn Box, Pascale Soumoy, and Jenny Quay (all of BCDC). Thanks to Josh Logan (USGS) for automating the averaging of directional transport data. Sample preparation and processing were performed by Angela Lam and Tracey Conrad at USGS.

References

Allen, J.R.L., 1968. The nature and origin of bedform hierarchies. Sedimentology 10, 161–182.
Anders, E., Grevesse, N., 1989. Abundances of the elements; meteoritic and solar. Geochimica et Cosmochimica Acta 53, 197–214.
Andrews, J.T., Eberl, D.D., 2012. Determination of sediment provenance by unmixing the mineralogy of source-area sediments: the "SedUnMix" program. Marine Geology 291–294, 24–33.
Barnard, P.L., Kvitek, R.G., 2010. Anthropogenic influence on recent bathymetric change in west-central San Francisco Bay. San Francisco Estuary and Watershed Science 8 (3) (13 pp.).
Barnard, P.L., Hanes, D.M., Rubin, D.M., Kvitek, R.G., 2006. Giant sand waves at the mouth of San Francisco Bay. Eos Transactions of the American Geophysical Union 87 (29), 285–289.
Barnard, P.L., Eshleman, J.L., Erikson, L.H., Hanes, D.M., 2007. Coastal processes study at Ocean Beach, San Francisco, CA: summary of data collection 2004–2006. U.S. Geological Survey Open-File Report 2007-1217 (165 pp.).
Barnard, P.L., Erikson, L.H., Kvitek, R.G., 2011a. Small-scale sediment transport patterns and bedform morphodynamics: new insights from high resolution multibeam bathymetry. Geo-Marine Letters 31 (4), 227–236.
Barnard, P.L., Allan, J., Hansen, J.E., Kaminsky, G.M., Ruggiero, P., Doria, A., 2011b. The impact of the 2009–10 El Niño Modoki on U.S. West Coast beaches. Geophysical Research Letters 38 (L13604) (7 pp.).
Barnard, P.L., Hansen, J.E., Erikson, L.H., 2012a. Case study of an erosion hot spot, Ocean Beach, CA (USA). Journal of Coastal Research 28 (4), 903–922.
Barnard, P.L., Erikson, L.H., Rubin, D.M., Dartnell, P., Kvitek, R.G., 2012b. Analyzing bedforms mapped using multibeam sonar to determine regional bedload sediment transport patterns in the San Francisco Bay coastal system. In: Li, M.Z., Sherwood, C.R., Hill, P.R. (Eds.), Sediments, Morphology and Sedimentary Processes on Continental Shelves: Advances in Technologies, Research and Applications: International Association of Sedimentologists (IAS) Special Publication, 44, pp. 272–294.
Barnard, P.L., Erikson, L.H., Elias, E., Dartnell, P., this issue-a. Cross-validation of bedform asymmetry and modeled residual sediment flux to determine sediment transport patterns in the San Francisco Bay Coastal System. Marine Geology, Special Issue San Francisco Bay, 50 pp.
Barnard, P.L., Schoellhamer, D.H., Jaffe, B.E., McKee, L.J., this issue-b. Sediment transport in the San Francisco Bay Coastal System: an overview. Marine Geology, Special Issue San Francisco Bay, 65 pp.
Battalio, R.T., Trivedi, D., 1996. Sediment transport processes at Ocean Beach, San Francisco, CA. Proceedings of the 25th International Conference on Coastal Engineering, American Society of Civil Engineers, Orlando, pp. 2691–2707.

Bau, M., Koschinsky, A., Dulski, P., Hein, J.R., 1996. Comparison of the partitioning behaviours of yttrium, rare earth elements, and titanium between hydrogenetic marine ferromanganese crusts and seawater. Geochimica et Cosmochimica Acta 60, 1709–1725.

Bernárdez, P., Prego, R., Giralt, S., Esteve, J., Caetano, M., Parra, S., Francés, G., 2012. Geochemical and mineralogical characterization of surficial sediments from the Northern Rias: implications for sediment provenance and impact of the source rocks. Marine Geology 291–294, 63–72.

Booij, N., Ris, R.C., Holthuijsen, L.H., 1999. A third-generation wave model for coastal regions 1. Model description and validation. Journal of Geophysical Research 104 (C4).

Briggs, P.H., Meier, A.L., 2002. The determination of forty-two elements in geological materials by inductively coupled plasma-mass spectrometry. In: Taggart Jr., J.E. (Ed.), Analytical Methods for Chemical Analysis of Geologic and Other Materials: Chapter "I", U.S. Geological Survey Open-File Report 02-223.

Bruns, T., Cooper, A., Carlson, P., McCulloch, D., 2002. Structure of the submerged San Andreas and San Gregorio fault zones in the Gulf of the Farallones off San Francisco, California, from high-resolution seismic-reflection data. In: Parsons, T. (Ed.), Crustal Structure of the Coastal and Marine San Francisco Bay Region, California: U.S. Geological Survey Professional Paper 1658.

Capiella, K., Malzone, C., Smith, R., Jaffe, B., 1999. Sedimentation and bathymetry changes in Suisun Bay (1867–1990). U.S. Geological Survey Open-File Report 99-563.

Carlson, P.R., McCulloch, D.S., 1970. Bedrock-surface map of central San Francisco Bay. U.S. Geological Survey-HUD Basic Data Contribution.

Cheng, R.T., Gartner, J.W., 1984. Tides, tidal and residual currents in San Francisco Bay California—results of measurements, 1979–1980. Parts I–V. U.S. Geological Survey Water Resources Investigations Report 84-4339.

Cheng, R.T., Gartner, J.W., 1985. Harmonic analysis of tides and tidal currents in south San Francisco Bay, CA. Estuarine, Coastal and Shelf Science 21, 57–74.

Cherry, J.A., 1964. Sand Movement Along a Portion of the Northern California Coast. Hydraulic Engineering Laboratory, University of California, Berkeley.

Cherry, J.A., 1966. Sand movement along equilibrium beaches north of San Francisco. Journal of Sedimentary Petrology 36 (2), 341–357.

Chin, J.L., Wong, F.L., Carlson, P.R., 2004. Shifting shoals and shattered rocks—how man has transformed the floor of west-central San Francisco Bay. U.S. Geological Survey Circular 1259. (30 pp.).

Chin, J.L., Woodrow, D.L., McGann, M., Wong, F.L., Fregoso, T., Jaffe, B.E., 2010. Estuarine sedimentation, sediment character, and Foraminiferal distribution in central San Francisco Bay, California. U.S. Geological Survey Open-File Report 2010-1130 (63 pp.).

Cho, Y.-G., Lee, C.-B., Choi, M.-S., 1999. Geochemistry of surface sediments off the southern and western coasts of Korea. Marine Geology 159, 111–129.

Cloern, J.E., Knowles, N., Brown, L.R., Cayan, D., Dettinger, M.D., Morgan, T.L., Schoellhamer, D.H., Stacey, M.T., van der Wegen, M., Wagner, R.W., Jassby, A.D., 2011. Projected evolution of California's San Francisco Bay-Delta–River system in a century of climate change. PloS One 6 (9) (13 pp.).

Condie, K.C., 1993. Chemical composition and evolution of the upper continental crust: contrasting results from surface samples and shales. Chemical Geology 104, 1–37.

Conomos, T.J., 1963. Geologic aspects of the recent sediments of South San Francisco Bay. Master's Thesis, San Jose State College, San Jose, CA, 118 pp.

Conomos, T.J., Peterson, D.H., 1977. Suspended-particle transport and circulation in San Francisco Bay, an overview. Estuarine Processes 2, 82–97.

Conomos, T.J., Smith, R.E., Gartner, J.W., 1985. Environmental setting of San Francisco Bay. Hydrobiologia 129 (1), 1–12.

Da Silva, M.A.M., 1979. Provenance of heavy minerals in beach sands, southeastern Brazil: from Rio Grande to Chuí (Rio Grande do Sul State). Sedimentary Geology 24, 133–148.

Dallas, K., Barnard, P.L., 2009. Linking human impacts within an estuary to ebb-tidal delta evolution. Journal of Coastal Research Special Issue 56, 713–716.

Dallas, K., Barnard, P.L., 2011. Anthropogenic influences on shoreline and nearshore evolution in the San Francisco Bay coastal system. Estuarine, Coastal and Shelf Science 92 (1), 195–204.

De Vriend, H., Capobianco, M., Chester, T., de Swart, H.E., Latteux, B., Stive, M.J.F., 1993. Approaches to long-term modelling of coastal morphology: a review. Coastal Engineering 21, 225–269.

Demirpolat, S., 1991. Surface and near-surface sediments from the continental shelf off the Russian River, northern California. Marine Geology 99, 163–173.

Dredged Material Management Office, 2008. 2007 Annual Report.

Duck, R.W., Rowan, J.S., Jenkins, P.A., Youngs, I., 2001. A multi-method study of bedload provenance and transport pathways in an estuarine channel. Physics and Chemistry of the Earth (B) 26 (9), 747–752.

Elder, W.P., this issue. Bedrock geology of the San Francisco Bay Area: a local sediment source for bay and coastal systems. Marine Geology, Special Issue San Francisco Bay, 53 pp.

Elias, E., Hansen, J.E., this issue. Understanding processes controlling sediment transports at the mouth of a high-energetic inlet system (San Francisco Bay, CA). Marine Geology, Special Issue San Francisco Bay, 61 pp.

Erikson L.E., Wright, S., Elias, E., Hanes, D.H., Schoellhamer, D.H., this issue. The use of numerical modeling and physical process measurements for identifying net sediment transport directions due to tides in the San Francisco Bay Coastal System. Marine Geology, Special Issue San Francisco Bay, 50 pp.

Farmer, G.L., Barber, D.C., Andrews, J.T., 2003. Provenance of Late Quaternary ice-proximal sediments in the North Atlantic: Nd, Sr and Pd isotopic evidence. Earth and Planetary Science Letters 209, 227–243.

Foxgrover, A.C., Higgins, S.A., Ingraca, M.K., Jaffe, B.E., Smith, R.E., 2004. Deposition, erosion, and bathymetric change in South San Francisco Bay, 1858–1983. U.S. Geological Survey Open-File Report 2004-1192 (25 pp.).

Fram, J.P., Martin, M.A., Stacey, M.T., 2007. Dispersal fluxes between the coastal ocean and a semi-enclosed estuarine basin. Journal of Physical Oceanography 37, 1645–1660.

Fregoso, T.A., Foxgrover, A.C., Jaffe, B.E., 2008. Sediment deposition, erosion, and bathymetric change in central San Francisco Bay: 1855–1979. U.S. Geological Survey Open-File Report 2008-1312 (46 pp.).

Frihy, O.E., Lotfy, M.F., Komar, P.D., 1995. Spatial variations in heavy minerals and patterns of sediment sorting along the Nile Delta, Egypt. Sedimentary Geology 97, 33–41.

Gallaway, E., Trenhaile, A.S., Cioppa, M.T., Hatfield, R.G., 2012. Magnetic mineral transport and sorting in the swash-zone: northern Lake Erie, Canada. Sedimentology 59 (4), 1–17.

Ganju, N.K., Knowles, N., Schoellhamer, D.H., 2008. Temporal downscaling of decadal sediment load estimates to a daily interval for use in hindcast simulations. Journal of Hydrology 349, 512–523.

Gao, S., Collins, M., 1992. Sand sediment transport patterns inferred from grain-size trends based upon definition of "transport vectors". Sedimentary Geology 80, 47–60.

Gartner, J.W., 2004. Estimating suspended solids concentrations from backscatter intensity measured by acoustic Doppler current profiler in San Francisco Bay, California. Marine Geology 211, 169–187.

Gilbert, G.K., 1917. Hydraulic-mining debris in the Sierra Nevada. U.S. Geological Survey Professional Paper 105 (154 pp.).

Goldman, H., 1969. Geologic and engineering aspects of San Francisco Bay fill. California Division of Mines and Geology Special Report 97.

Graham, N.E., Diaz, H.F., 2001. Evidence for intensification of North Pacific Winter Cyclones since 1948. Bulletin of the American Meteorological Society 82, 1869–1893.

Gram, R., 1966. The marine geology of the recent sediments of Central San Francisco Bay. Master's Thesis, San Jose State College, San Jose, CA.

Graymer, R.W., Moring, B.C., Saucedo, G.J., Wentworth, C.M., Brabb, E.E., Knudsen, K.L., 2006. Geologic Map of the San Francisco Bay Region. U.S. Geological Survey Scientific Investigations Map 2918, 1 map sheet, 36×48 in., scale 1:275,000, http://pubs.usgs.gov/sim/2006/2918/.

Greene, H.G., Bizarro, J.J., 2003. Benthic Habitat Maps of San Francisco Bay Interpreted from Multibeam Bathymetric Images and Side-scan Sonar Mosaics, 2003. National Oceanic and Atmospheric Administration, Western Administrative Center, National Ocean Service.

Griggs, G.B., Hein, J.R., 1980. Sources, dispersal, and clay mineral composition of fine-grained sediment off Central and Northern California. Journal of Geology 88 (5), 541–566.

Gross, S., 1997. Numerical modeling of hydrodynamics and scalar transport in an estuary. Ph.D. dissertation, Stanford University.

Grousset, F., Biscaye, P., 2005. Tracing dust sources and transport patterns using Sr, Nd and Pb isotopes. Chemical Geology 222, 149–167.

Hanes, D.M., Barnard, P.L., 2007. Morphological evolution in the San Francisco Bight. Journal of Coastal Research Special Issue 50, 469–473.

Hanes, D.M., Ward, K., Erikson, L.H., 2011. Waves and tides responsible for the intermittent closure of the entrance of a small, sheltered tidal wetland at San Francisco, CA. Continental Shelf Research 31, 1682–1687.

Hansen, J.E., Barnard, P.L., 2010. Sub-weekly to interannual variability of a high-energy shoreline. Coastal Engineering 57 (11–12), 959–972.

Hansen, J.E., Elias, E., Barnard, P.L., this issue. Evolution in nearshore morphodynamics resulting from multi-decadal contraction of a large ebb-tidal delta Marine Geology, Special Issue San Francisco Bay, 37 pp.

Hanson, C., Coil, J., Keller, B., Johnson, J., Taplin, J., Monroe, J., 2004. Assessment and Evaluation of the Effects of Sand Mining on Aquatic Habitat and Fishery Populations of Central San Francisco Bay and the Sacramento–San Joaquin Estuary. Hanson Environmental Inc. (Available from: http://www.hansonenvironmentalinc.com/reports.htm).

Hapke, C.J., Reid, D., Richmond, B.M., Ruggiero, P., List, J., 2006. National assessment of shoreline change: part 3: historical shoreline changes and associated coastal land loss along the sandy shorelines of the California coast. U.S. Geological Survey Open-File Report 2006-1219.

Hapke, C.J., Reid, D., Richmond, B., 2009. Rates and trends of coastal change in California and the regional behavior of the beach and cliff system. Journal of Coastal Research 25 (3), 603–615.

Hein, J.R., Mizell, K., Barnard, P.L., Klofas, J.M., this issue. Beach and seabed sand provenances and transport pathways for the San Francisco Bay coastal system, based on X-ray diffraction mineralogy. Marine Geology, Special Issue San Francisco Bay, 32 pp.

Holthuijsen, L.H., Booij, N., Ris, R.C., 1993. A spectral wave model for the coastal zone. 2nd International Symposium on Ocean Wave Measurement and Analysis, New Orleans, pp. 630–641.

Hornberger, M.I., Luoma, S.N., van Green, A., Fuller, C., Anima, R., 1999. Historical trends of metals in the sediments of San Francisco Bay, California. Marine Geochemistry 64, 39–55.

Ingram, B.L., Lin, J.C., 2002. Geochemical tracers of sediment sources to San Francisco Bay. Geological Society of America Bulletin 30 (6), 575–578.

Jachens, R., Wentworth, C., Zoback, M., Bruns, T., Roberts, C., 2002. Concealed stands of the San Andreas Fault system in the central San Francisco Bay region, as inferred from aeromagnetic anomalies. In: Parsons, T. (Ed.), Crustal Structure of the Coastal and Marine San Francisco Bay Region: U.S. Geological Survey Professional Paper 1658, pp. 43–61.

Jaffe, B.E., Foxgrover, A., 2006. Sediment deposition and erosion in South San Francisco Bay, California from 1956 to 2005. U.S. Geological Survey Open-File Report 2006-1287.

Jaffe, B.E., Smith, R.E., Foxgrover, A.C., 2007. Anthropogenic influence on sedimentation and intertidal mudflat change in San Pablo Bay, California: 1856–1983. Estuarine, Coastal and Shelf Science 73, 175–187.

Please cite this article as: Barnard, P.L., et al., Integration of bed characteristics, geochemical tracers, current measurements, and numerical modeling for assessing the provenance..., Marine Geology (2012), http://dx.doi.org/10.1016/j.margeo.2012.11.008

Jenkins, P., Duck, R., Rowan, J., Walden, J., 2002. Fingerprinting of bed sediment in the Tay Estuary, Scotland: an environmental magnetism approach. Hydrology and Earth System Sciences 6 (6), 1007–1016.

Jenks, G.F., 1967. The data model concept in statistical mapping. International Yearbook of Cartography 7, 186–190.

Jevrejeva, S., Moore, J.C., Grinsted, A., 2012. Sea level projections to AD2500 with a new generation of climate change scenarios. Global and Planetary Change 80–81, 12–20.

Kamel, A.D., 1962. Littoral studies near San Francisco using tracer techniques. Beach Erosion Board Technical Memo No. 131 (86 pp.).

Keller, B.R., 2009. Literature review of unconsolidated sediment in San Francisco Bay and nearby Pacific Ocean Coast. San Francisco Estuary and Watershed Science 7 (1) (25 pp.).

Knaapen, M.A.F., 2005. Sand wave migration predictor based on shape information. Journal of Geophysical Research—Earth Surface 110 (FO4S11) (9 pp.).

Knebel, H.J., Conomos, T.J., Commeau, J.A., 1977. Clay-mineral variability in the suspended sediments of the San Francisco Bay System, California. Journal of Sedimentary Petrology 47 (1), 229–236.

Knowles, N., Cayan, D.R., 2004. Elevational dependence of projected hydrologic changes in the San Francisco estuary and watershed. Climate Change 62, 313–336.

Krone, R.B., 1979. Sedimentation in the San Francisco Bay system. In: Conomos, T.J. (Ed.), San Francisco Bay: The Urbanized Estuary. American Association for the Advancement of Science, San Francisco, pp. 85–96.

Lackschewitz, K.S., Wallrabe-Adams, H.-J., Garbe-Schönberg, D., 1994. Geochemistry of surface sediments from the mid-oceanic Kolbeinsey Ridge, north of Iceland. Marine Geology 121, 105–119.

Le Roux, J.P., 1994. An alternative approach to the identification of sand sediment transport paths based on a grain-size trends. Sedimentary Geology 94, 97–107.

Lee, J.I., Park, B.-K., Jwa, Y.-J., Yoon, H.I., Yoo, K.C., Kim, Y., 2005. Geochemical characteristics and the provenance of sediments in the Bransfield Strait, West Antarctica. Marine Geology 219, 81–98.

Lesser, G.R., 2009. An Approach to Medium-term Coastal Morphological Modelling. UNESCO-IHE Institute for Water Education, Delft, The Netherlands.

Lesser, G.R., Roelvink, J.A., van Kester, J.A.T.M., Stelling, G.S., 2004. Development and validation of a three-dimensional morphological model. Coastal Engineering 51, 883–915.

Lewicki, M., McKee, L., 2010. New Methods for Estimating Annual and Long-term Suspended Sediment Loads from Small Tributaries to San Francisco Bay, 337. IAHS Publication . (5 pp.).

Li, M.Z., Komar, P.D., 1992. Longshore grain sorting and beach placer formation adjacent to the Columbia River. Journal of Sedimentary Petrology 62, 429–441.

Martin, M.A., Fram, J.P., Stacey, M.T., 2007. Seasonal chlorophyll a fluxes between the coastal Pacific Ocean and San Francisco Bay. Marine Ecology Progressive Series 337, 51–61.

McCave, I.N., Langhorne, D.N., 1982. Sand waves and sediment transport around the end of a tidal sand bank. Sedimentology 29, 95–110.

McGann, M., Erikson, L., Wan, E., Powell, C. II, Maddocks, R.F., this issue. Distribution of biologic, anthropogenic, and volcanic constituents as a proxy for sediment transport in the San Francisco Bay Coastal System. Marine Geology, Special Issue San Francisco Bay, 89 pp.

McKee, L., Leatherbarrow, J., Pearce, S., Davis, J., 2003. A review of urban runoff processes in the Bay Area. San Francisco Estuary Institute Contribution, 66 . (Oakland, CA).

McKee, L., Ganju, N., Schoellhamer, D., 2006. Estimates of suspended sediment flux entering San Francisco Bay from the Sacramento–San Joaquin Delta, San Francisco Bay, California. Journal of Hydrology 323, 335–352.

McKee, L.J., Lewicki, M., Schoellhamer, D.H., Ganju, N.K., this issue. Comparison of sediment supply to San Francisco Bay from coastal and Sierra Nevada watersheds. Marine Geology, Special Issue San Francisco Bay, 57 pp.

McLaren, P., Bowles, D., 1985. The effects of sediment transport on grain-size distributions. Journal of Sedimentary Petrology 55 (4), 457–470.

Minard, C.R. Jr, 1971. Quaternary Beaches and Coasts Between the Russian River and Drakes Bay, California. Hydraulic Engineering Laboratory, University of California, Berkeley.

Monismith, S.G., Kimmerer, W., Burau, J.R., Stacey, M.T., 2002. Structure and flow-induced variability of the subtidal salinity field in Northern San Francisco Bay. Journal of Physical Oceanography 32, 3003–3019.

Moore, D.B., 1965. Recent coastal sediments, Double Point to Point San Pedro, California. Technical Report HEL 2-14. Hydraulic Engineering Laboratory, University of California, Berkeley (95 pp.).

Morton, A.C., 1985. Heavy minerals in provenance studies. In: Zuffa, G.G. (Ed.), Provenance of Arenites. Reidel, Dordrecht, pp. 249–277.

Munksgaard, N.C., Lim, K., Parry, D.L., 2003. Rare earth elements as provenance indicators in North Australian estuarine and coastal marine sediments. Estuarine, Coastal and Shelf Science 57, 399–409.

Murray, R.W., Buchholtz ten Brink, M.R., Gerlach, D.C., Russ III, G.P., Jones, D.L., 1991. Rare earth, major and trace elements in chert from the Franciscan Complex and Monterey Group, California: assessing REE sources to fine-grained marine sediments. Geochimica et Cosmochimica Acta 55, 1875–1895.

National Research Council, 2012. Sea-level rise for the coasts of California, Oregon, and Washington: past, present, and future. Committee on Sea Level Rise in California, Oregon, and Washington. The National Academies Press, Washington, D.C. (260 pp.).

Nicholls, R.J., Cazenave, A., 2010. Sea-level rise and its impact on coastal zones. Science 328, 1517–1520.

Oldfield, F., Yu, L., 1994. The influence of particle size variations on the magnetic properties of sediments from the north-eastern Irish Sea. Sedimentology 41, 1093–1108.

Oltmann, R.N., Schoellhamer, D.H., Dinehart, R.L., 1999. Sediment inflow to the Sacramento–San Joaquin Delta and the San Francisco Bay. Interagency Ecological Program Newsletter 12 (1), 30–33.

Piper, D.Z., 1974. Rare earth elements in the sedimentary cycle: a summary. Chemical Geology 14, 285–304.

Poizot, E., Méar, Y., Biscara, L., 2008. Sediment Trend Analysis through the variation of granulometric parameters: a review of theories and applications. Earth-Science Reviews 86, 15–41.

Porterfield, G., 1980. Sediment transport of streams tributary to San Francisco, San Pablo and Suisun Bays, California, 1909–66. U.S. Geological Survey Water Resources Investigations 80-64 (92 pp.).

Prego, R., Caetano, M., Bernárdez, P., Brito, P., Ospina-Alvarez, N., Vale, C., 2012. Rare earth elements in coastal sediments of the northern Galician shelf; influence of geological features. Continental Shelf Research 35, 75–85.

Rao, C.B., 1957. Beach erosion and concentration of heavy-mineral sands. Journal of Sedimentary Petrology 27 (2), 143–147.

Ris, R.C., Booij, N., Holthuijsen, L.H., 1999. A third-generation wave model for coastal regions, part II: verification. Journal of Geophysical Research 104, 7649–7666.

Ronov, A.B., Balashov, Y.A., Migdisov, A.A., 1967. Geochemistry of the rare earths in the sedimentary cycle. Geochemistry International 4, 1–17.

Rosenbauer, R.J., Foxgrover, A.C., Hein, J.R., Swarzenski, P., this issue. Efficacy of rare earth element concentrations and $^{143}Nd/^{144}Nd$ and $^{87}Sr/^{86}Sr$ isotope ratios for assessing provenances of sand-sized sediment in San Francisco Bay. Marine Geology, Special Issue San Francisco Bay, 50 pp.

Rotman, R., Naylor, L., McDonnell, R., MacNiocaill, C., 2008. Sediment transport on the Freiston Shore managed realignment site: an investigation using environmental magnetism. Geomorphology 100, 241–255.

Rubin, D.M., McCulloch, D.S., 1979. The movement and equilibrium of bedforms in Central San Francisco Bay. In: Conomos, T.J. (Ed.), San Francisco Bay: The Urbanized Estuary. American Association for the Advancement of Science, San Francisco, pp. 97–113.

Saitoh, Y., Tamura, T., Kodama, Y., Nakano, T., 2011. Strontium and neodymium isotopic signatures indicate the provenance and depositional process of loams intercalated in coastal dune sand, western Japan. Sedimentary Geology 236, 272–278.

San Francisco Estuary Institute, 2009. The pulse of the estuary: monitoring and managing water quality in the San Francisco Estuary. San Francisco Estuary Institute Contribution, 583 . (Oakland, CA, 96 pp.).

Schatz, B., 1963. A restudy of bottom sediments near the entrance of the Golden Gate. Technical Report HEL 4-3. Hydraulic Engineering Laboratory, University of California, Berkeley (7 pp.).

Schlocker, J., 1974. Geology of the San Francisco north quadrangle, California. U.S. Geological Survey Professional Paper 782.

Schoellhamer, D.H., 2011. Sudden clearing of estuarine waters upon crossing the threshold from transport to supply regulation of sediment transport as an erodible sediment pool is depleted: San Francisco Bay, 1999. Estuaries and Coasts 34 (5), 885–899.

Schoellhamer, D.H., Lionberger, M.A., Jaffe, B.E., Ganju, N.K., Wright, S.A., Shellenbarger, G.G., 2005. Bay sediment budgets: sediment accounting 101. The pulse of the estuary: monitoring and managing contamination in the San Francisco Estuary. San Francisco Estuary Institute Contribution, 411 . (Oakland, CA, 84 pp.).

Scripps Institution of Oceanography, 2012. The Coastal Data Information Program. Integrative Oceanography Division, San Diego . (http://cdip.ucsd.edu. Accessed 2012 Mar 27).

Shvidchenko, A.B., MacArthur, R.C., Hall, B.R., 2004. Historic sedimentation in Sacramento–San Joaquin Delta. Interagency Ecological Program for the San Francisco Estuary Newsletter 17 (3), 21–30.

Singer, M.B., Aalto, R., James, L.A., 2008. Status of the lower Sacramento Valley flood-control system within the context of its natural geomorphic setting. Natural Hazards Review 9, 104–115.

Smith, S.D., 1988. Coefficients for sea surface wind stress, heat flux, and wind profiles as a function of wind speed and temperature. Journal of Geophysical Research 93 (C12), 15, 467–15, 472.

Steidtmann, J.R., 1982. Size density sorting of sand size spheres during deposition from bedload transport and implications concerning hydraulic equivalence. Sedimentology 29, 877–883.

Stelling, G.S., 1984. On the construction of computational methods for shallow water flow problems. Rijkswaterstaat Communications 35.

Stride, A.H., 1963. Current swept sea floors near the southern half of Great Britain. Quarterly Journal of the Geological Society of London 119, 175–199.

Talke, S.A., Stacey, M.T., 2003. The influence of oceanic swell on flows over an estuarine intertidal mudflat in San Francisco Bay. Estuarine, Coastal and Shelf Science 58, 541–554.

Teeter, A.M., Letter, J.V. Jr, Pratt, T.C., Callegan, C.J., Boyt, W.L., 1996. San Francisco Bay Long-Term Management Strategy (LTMS) for dredging and disposal. Report 2. Baywide Suspended Sediment Transport Modeling, Army Engineer Waterways Experiment Station, Hydraulics Lab Technical Report A383413, Vicksburg, MS (96 pp.).

Trask, P.D., 1953. San Francisco Bay Sediment Study, Inshore Survey. University of California Institute of Engineering Research, San Francisco Bay.

Trask, P.D., 1956. Geology of some American estuarine harbors. Journal of Hydraulics Division, Proceedings of the American Society of Civil Engineers 82 (2), 1–18 (Paper No. 956).

USGS, 1998. U.S. Geological Survey sample collection survey J-1-98-SF. http://walrus.wr.usgs.gov/infobank/j/j198sf/html/j-1-98-sf.meta.html.

USGS, 2006. Quaternary fault and fold database for the United States. Updated 11/3/2010 http://earthquakes.usgs.gov/regional/qfaults/ (Accessed 2012 Mar 27).

USGS, 2010, U.S. Geological Survey sample collection surveysB-2-10-SF http://walrus.wr.usgs.gov/infobank/b/b210sf/html/b-2-10-sf.meta.htmlS-7-10-SF http://walrus.wr.usgs.gov/infobank/s/s1710sf/html/s-17-10-sf.meta.htmlS-8-10-SF http://walrus.wr.usgs.gov/infobank/s/s810sf/html/s-8-10-sf.meta.html.

USGS, 2011, U.S. Geological Survey sample collection surveysB-5-11-SF http://walrus.wr.usgs.gov/infobank/b/b0511sf/html/b-05-11-sf.meta.htmlS-1-12-SF http://walrus.wr.usgs.gov/infobank/s/s0112sf/html/s-01-12-sf.meta.html.

Van Veen, J., 1935. Sandwaves in the North Sea. International Hydrographic Review 12, 21–29.

Vermeer, M., Rahmstorf, S., 2009. Global sea level linked to global temperature. Proceedings of the National Academy of Sciences 106 (51), 21,527–21,532.

Walters, R.A., 1982. Low frequency relations in sea level currents in South San Francisco Bay. Journal of Physical Oceanography 12, 658–668.

Walters, R.A., Gartner, J.W., 1985. Subtidal sea level and current variations in the northern reach of San Francisco Bay. Estuarine, Coastal and Shelf Science 21, 17–32.

Walters, R.A., Cheng, R.T., Conomos, T.J., 1985. Time scales of ciruclation and mixing processes of San Francisco Bay waters. Hydrobiologia 129, 13–36.

Weis, D., Kieffer, B., Maerschalk, C., Barling, J., De Jong, J., Williams, G., Hanano, D., Pretorius, W., Mattielli, N., Scoates, J.S., Goolaerts, A., Friedman, R., Mahoney, J.B., 2006. High-precision isotopic characterization of USGS reference materials by TIMS and MC-ICP-MS. Geochemistry, Geophysics, Geosystems 7, Q08006.

Wilde, P., Isselhardt, C., Osuch, L., Lee, T., 1969. Recent marine sediments of Bolinas Bay, California: part C — interpretation and summary of results. Technical Report HEL 2-23. Hydraulic Engineering Laboratory, University of California, Berkeley (5 pp.).

Wong, F., 2001. Heavy-mineral provinces on the continental shelf. In: Karl, H.A., Chin, J.L., Ueber, E., Stauffer, P.H., Hendley II, J.W. (Eds.), Beyond the Golden Gate—Oceanography, Geology, Biology, and Environmental Issues in the Gulf of the Farallones: U.S. Geological Circular 1198, pp. 116–121.

Wong, F.L., Woodrow, D.L., McGann, M., this issue. Heavy mineral analysis for assessing the provenance of sandy sediment in the San Francisco Bay Coastal System. Marine Geology, Special Issue San Francisco Bay, 15 pp.

Wright, S.A., Schoellhamer, D.H., 2004. Trends in the sediment yield of the Sacramento River, California, 1957–2001. San Francisco Estuary and Watershed Science 2 (2) (14 pp.).

Wright, S.A., Schoellhamer, D.H., 2005. Estimating sediment budgets at the interface between rivers and estuaries with application to the Sacramento–San Joaquin River Delta. Water Resources Research 41 (W09428).

Yancey, T., Lee, J., 1972. Major heavy mineral assemblages and heavy mineral provinces of the central California Coast region. Geological Society of America Bulletin 83 (7), 2099–2104.

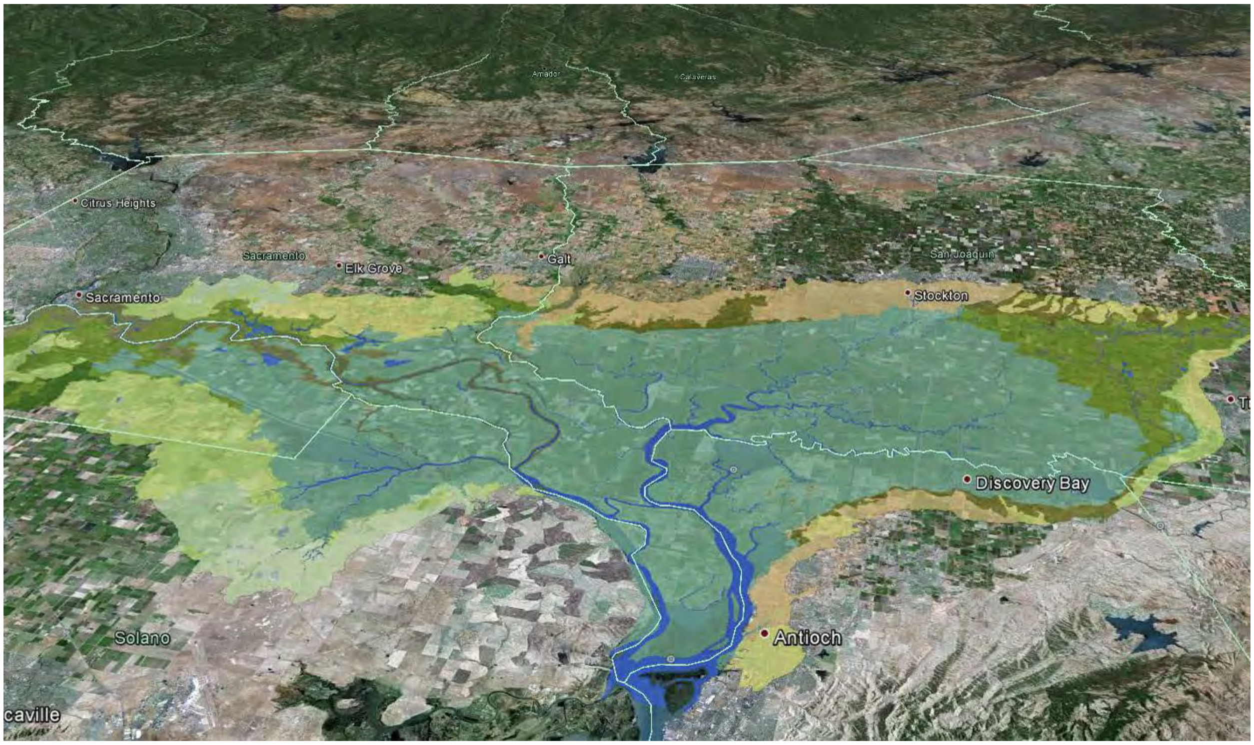


Image from Whipple et al. 2013, Sacramento-San Joaquin Delta Historical Ecology Investigation: EXPLORING PATTERNS AND PROCESS

Delta Science Program Independent Review Panel Report

BDCP Effects Analysis Review, Phase 3

A report to the

Delta Science Program

Prepared by

Alex Parker, Ph. D., California Maritime Academy, California State University (Panel Chair)
Charles "Si" Simenstad, M.S., University of Washington (Lead Author)
T. Luke George, Ph.D., Colorado State University
Nancy Monsen, Ph.D., Stanford University
Tom Parker, Ph.D., California State University San Francisco
Greg Ruggerone, Ph.D., Natural Resources Consultants, Inc.
John Skalski, Ph.D., University of Washington

March 2014

Delta Stewardship Council
Delta Science Program

Delta Science Program Independent Review Panel Report: BDCP Effects Analysis Review, Phase 3

Table of Contents

List of Tables

Executive Summary

Under the auspices of the Delta Science Program, the seven-member Independent Scientific Review Panel (Panel) reviewed the adequacy of the Effects Analysis component of the Bay Delta Conservation Plan (BDCP or Plan). This report represents the third phase of the Effects Analysis review; the Phase 1 (completed in November 2011) and Phase 2 reviews (completed June 2012) were partial reviews of the Effects Analysis and were completed as the Conceptual Foundation and Analytical Approach were still under development. These documents are available online at: http://deltacouncil.ca.gov/science-program/independent-review-draft-bay-delta-conservation-plan-effects-analysis. The present, Phase 3 review covers the first complete public draft of the BDCP Chapter 5 Effects Analysis and its associated technical appendices, made available in December 2013.

Four broad themes emerged from the Panel's review of the BDCP Effects Analysis. Firstly, the long, highly detailed document was difficult to review and comprehend. The vastness of the Effects Analysis report and appendices are both its strength and weakness. Although highly improved from the documents that the Panel reviewed during Phase 2, Chapter 5 continues to be fragmented in its presentation and sometimes inconsistent with the technical appendices. While the sheer scope of the analysis is impressive, the inefficient organization and incomplete cross-referencing among sections within the Effects Analysis (e.g., the 8 supporting appendices, totaling ~4500 pages) as well as with the larger BDCP planning documents make interpretation of anticipated net effects of BDCP implementation difficult at best. The 745-page Chapter 5: Effects Analysis does not represent a stand-alone document and it relies extensively on the associated appendices and other chapters for the presentation of scientific information, with insufficient guidance for the reader. As concluded from the Phase 2 report, the Panel universally believes that by itself, Chapter 5: Effects Analysis inadequately conveys the fully integrated assessment that is needed to draw conclusions about the Plan, in part because of incomplete information on factors affecting the covered species.

The second theme in the Panel's review is an apparent disconnect between the assessments of the levels of scientific uncertainty presented in Chapter 5 versus what is characterized in the technical appendices. In many cases, the Panel felt that there was appropriate characterization of high uncertainty within the technical appendices but Chapter 5 did not sufficiently acknowledge or articulate this reality, especially when using professional judgment to reach overall net effects of the BDCP on key species. In particular, the Panel observed that the critical uncertainties associated with presumed beneficial effects of tidal wetland restoration were not recognized in the Chapter 5 summary. Given the magnitude of the BDCP, the inherent natural and anthropogenic complexity in the Bay-Delta ecosystem, and the long time horizon for BDCP implementation and rehabilitated community development, most of the potential BDCP effects carry a relatively high level of uncertainty. For these reasons, the Effects Analysis must provide clear guidance for conceptual models, monitoring, metrics that assess underlying ecosystem processes, explicit thresholds and triggers, alternative hypotheses, special studies to address critical information gaps, and structured decision making in the form of a rigorously institutionalized adaptive management process.

The third major theme of this review is the lack of an integrated or quantitative assessment of net effects, echoing a similar review comment in the Phase 2 review. The Panel acknowledges that considerable effort has been made in documenting the complex information used to determine net effects. However, in the case of covered species, effects could not be quantified and only two of the sixteen existing life cycle models were deemed to be relevant to BDCP. For these and other reasons, a systematic approach to synopsize the overall net effect on each species was not used. Instead, professional judgment was used instead of a ranking approach to quantify a synthesis of cumulative effects and associated certainty in the projected outcome. Finally, in one paragraph, Chapter 5 accurately portrayed the anticipated BDCP effects: "*These expectations represent a working hypothesis of the relationship between actions, stressors, and biological performance*". However, this statement was not emphasized throughout the document.

The fourth major theme reflected on the need to address the extensive uncertainties associated with the assumptions and predictions of the beneficial effects of the BDCP conservation measures. While the Phase 2 Effects Analysis accurately reflected the detailed process and implementation structure to apply an adaptive management approach to resolve uncertainties, the Panel was concerned that it defaulted to rather "passive learning" instead of a rigorous, institutionalized adaptive management process that resolved effects on covered species and their requisite ecosystems through an active, experimental approach.

Together with background obtained during Phase 1 and 2 of the BDCP Effects Analysis review, the Panel provides the following synopsis of the Panel's responses to their General Charge Questions; further responses to specific issues and the adequacy of supporting documents are provided in the body of the report.

1. How well does the Effects Analysis meet its expected goals?

The Phase 3 review-version of the Effects Analysis is a much improved and impressive compilation of background material and scientific and technical knowledge about the Bay-Delta that provides a plausible basis for the conservation measures. The Panel concluded that much of the available data and arguments for the rationale behind the Effects Analysis assumptions and conclusions are contained within the BDCP documents. However, we suggest that the Effects Analysis (Chapter 5) itself is still poorly substantiated and leaves too much to appendices and other BDCP chapters without explicit cross-references. The lack of accessibility to information within the chapter or clear reference to supporting detail inhibits rather than elucidates comprehension of the findings and thus conveys an unsatisfying "trust us" message.

Our conclusion of the Effects Analysis is that many of the critical assumptions in modeling effects and justifications behind the supposed benefits of the conservation measures are highly uncertain. Much of the conservation measures center around restoration activities and management actions to improve current conditions. Our impression, therefore, is that the foundation of the BDCP is weak in many respects and the default burden to ensure covered species benefit, if not recovery, depends on adaptive management. The adequacy of the BDCP therefore rests not in the intent and development of the conservation measures, but in the rigor and application of adaptive management to ensure that the critical uncertainties are addressed and strategically incorporated into a progressively refined Plan.

2. How complete is the Effects Analysis; how clearly are the methods described?

Chapter 5 provides a comprehensive overview of the spatial and temporal scope of the analysis, definitions of project baselines that differ depending on regulatory authority, recognition of climate change information, identification of a variety of models used to evaluate effects, treatment of viable salmon population criteria, and the approach to determining net effects on fish and wildlife. As might be expected, with the size of the Effects Analysis task, the quality of the assessments ranged in scientific rigor based on the amount of available data and best available science. Some aspects of the assessment, such as water quality and flow, were quantitatively assessed using sophisticated mathematical models. Some aspects of the Chinook salmon assessments were also based on empirical data and process-based models. However, for many of the other fish species and their potential stressors, conceptual models supported by the scientific literature were the only recourse.

3. Is the Effects Analysis reasonable and scientifically defensible? How clearly are the net effects results conveyed in the text, figures and tables?

The approach to net effect conclusions needs to be reconsidered and revamped. The Effects Analysis assessment of net effects, particularly for covered fish, tries to incorporate information on potentially beneficial or detrimental effects covering 12 different stressors, 32 attributes, and multiple life stages using best available information and science. Only a perfect life-cycle model with perfect information on all the effects and their interactions could possibly weight the results correctly and draw unambiguous conclusions. A serious limiting factor of the current consolidation of Net Effects is a near complete absence of any weighting of the biological importance to particularly sensitive life history stages of the many attributes under consideration. As a result, whether and how any critical life stages or attributes are being adversely affected by the BDCP is generally unclear. The net effects conclusions for a fish species needs to therefore take into account the relative importance of the various life history stages, make them explicit, and interpret Plan effects within that context on a species-by-species basis. Similarly, the simple summation of the number of acres of suitable habitat that are removed or restored for each species by the conservation measures does not consider landscape-level effects such as connectivity and patch size, nor does it take into account variation in habitat quality.

The net effects analysis tends to overreach conclusions of positive benefits for covered fish species, given the inability to quantify the over-all net effects and the realization of high uncertainty. In particular, it does not adequately defend conclusions regarding the net effects of habitat restoration. Restoration of tidal wetlands (and other communities) is highly uncertain and at least an extremely long process. The Effects Analysis does not adequately justify the critical assumption of the benefit of tidal wetland restoration as a food web subsidy for covered pelagic fish given the uncertainties of tidal wetland restoration itself. A critical issue is the implicit expectation that restoration activities will result in increases in abundance of lower trophic levels, but it is uncertain whether the resulting increased production will result in food web pathways supporting covered species. The presentation of phytoplankton-based and tidal wetland macrophyte detritus-based food webs as alternative ecosystem processes, rather than as an integrated system, also significantly complicates the interpretation of the potential benefit of BDCP restoration. For foraging salmonids, the Effects Analysis did not evaluate the reduced extent to which salmonids would have access to rehabilitated habitat when the north Delta intakes are operating and flows are reduced.

Only one configuration of Restoration Opportunity Areas (ROAs) were modeled by the hydrodynamic models and the locations of these assumed Restoration Opportunity Areas are not available. Some details of the hydrodynamic modeling, especially where 1D and 2D models did not agree or situations where counter-intuitive results were reported, could not be evaluated due to the limited information provided.

4. How well is uncertainty addressed? How could communication of uncertainty be improved?

A broad consensus exists among the Panel that Chapter 5 does not adequately acknowledge the extensive uncertainty associated with the BDCP's assumptions and predictions. In its current form, at the level of detail conveyed, in the models used, and in the verbal assessments and conclusions, the level of uncertainty is often downplayed. Within appendices sometimes more explicit discussion of uncertainties can be found, but there is a disconnect between the summary pages with the conclusions drawn in Chapter 5. In situations in which an array of outcomes may be possible, only the more beneficial outcomes are used in conclusions about the BDCP. Communication of uncertainty would be improved by consideration of a range of potential outcome values in models.

5. How well does the Effects Analysis describe how conflicting model results and analyses in the technical appendices are interpreted?

The Panel found models describing salmonid Delta passage and habitat suitability for terrestrial species to be appropriate and any conflicting results adequately explained. Because hydrodynamic models are sensitive to how the open water regions are represented and how they are connected to the adjacent channels, and because the panel was not provided the bathymetric configuration of the ROAs or the order in which the ROAs were established, it is not feasible for the Panel to evaluate the sensitivity of the models to the placement of the Restoration Opportunity Areas.

Overall, the Panel found the Chapter 5 text describing the two life cycle models (IOS and OBAN), which provide alternative views of BDCP effects compared with other analyses, to be complicated and somewhat confusing. It was not clear whether or not the models were appropriately applied to evaluate a portion of the BDCP attributes.

The Effects Analysis modeling of salmon sensitivity to water temperature during egg incubation in the Sacramento River is not clear, given that the BDCP has no effect on upstream conditions according to some sections of Chapter 5. The Chapter 5 evaluation needs clarification, including a clear description of how the BDCP might affect flow and temperature in this area.

6. How well does the Effects Analysis link to the adaptive management plan and associated monitoring programs?

While both the need for and operative structure of adaptive management is identified considerably more in the Phase 3 review version of the Effects Analysis, it remains characterized as a silver bullet but without clear articulation about how key assumptions will be vetted or uncertainties resolved to the point that the BDCP goals and objectives are more assured. The concept of adaptive management is appropriately described and allocated a prominent role in the implementation structure. However, the commonly acknowledged process of adaptive management is easily misunderstood and misapplied, often resulting in a loss of rigor and commitment in application. Because of the extensive uncertainties surrounding the assumptions and predictions of the BDCP, the Panel strongly emphasizes institutionalizing an exceedingly rigorous adaptive management process. This is critical in order to avoid the high risk associated with ecological surprises that will be difficult or impossible to reverse once they have occurred. BDCP must make a commitment to the fundamental process, and specifically the required monitoring and independent science review, not just the concept of adaptive management.

Introduction

This report describes the results of an independent scientific review of the Bay Delta Conservation Plan (BDCP) Effects Analysis. At the request of the BDCP participants, the Delta Science Program convened an Independent Science Review Panel (Panel) to assess the scientific soundness of the BDCP Effects Analysis, guided by a Panel Charge with explicit questions to address.

Background and History

The BDCP Working Draft was initially released November 18, 2010 without a detailed effects analysis. This review has been conducted in three phases and was initiated in October 2011. The Panel's initial (Phase 1) review was conducted on the Draft BDCP Effects Analysis' Conceptual Foundation and Analytical Framework and the Entrainment Appendix as an example of the application of the conceptual understanding, methods and analyses discussed in the Conceptual Foundation and Analytical Framework. In the most recent drafts of the BDCP Effects Analysis, the Foundation and Framework (originally Appendix A) concepts were incorporated into Chapter 5: Effects Analysis. During Phase 2, the Panel reviewed drafts of the BDCP Chapter 5: Effects Analysis and drafts of many of the associated technical appendices. Appendices 5.E: Habitat Restoration and 5.G: Fish Life Cycle Models were not reviewed during the Phase 2 review. The BDCP Chapter 5: Effects Analysis and all of its associated technical appendices were reviewed during the Phase 3 review that is summarized in this report.

BDCP Goals and Role of Effects Analysis

The overall goal of the BDCP is to restore and protect ecosystem health, water supply, and water quality within a stable regulatory framework. Component goals include:

- provide for the conservation and management of Covered Species within the Plan Area;
- preserve, restore and enhance aquatic, riparian and associated terrestrial natural communities and ecosystems that support Covered Species within the Plan Area through conservation partnerships;
- allow for projects to proceed that restore and protect water supply, water quality, and ecosystem health within a stable regulatory framework;
- provide a means to implement Covered Activities in a manner that complies with applicable State and Federal fish and wildlife protection and laws, including California Endangered Species Act and Federal Endangered Species Act, and other environmental laws, including the California Environmental Quality Act and National Environmental Policy Act;
- provide a basis for permits necessary to lawfully take Covered Species;
- provide a comprehensive means to coordinate and standardize mitigation and compensation requirements for Covered Activities within the Planning Area;
- provide a less costly, more efficient project review process which results in greater conservation values than project-by-project, species-by-species review; and,
- provide clear expectations and regulatory assurances regarding Covered Activities occurring within the Planning Area.

The Effects Analysis is a critical component for the BDCP. Its purpose is to provide the best scientific assessment of the likely effects of BDCP actions on the species of concern and ecological processes of the Bay-Delta system. The Effects Analysis will, out of necessity, rely heavily on the application of models to quantify the likely results of the BDCP. These include conceptual, numerical, hydrodynamic, operational, and species models. The BDCP Effects Analysis is being conducted and documented through Chapter 5: Effects Analysis and a series of technical appendices centered on common stressors or groups of similar effects. The draft appendices reviewed in Phase 1 of the Effects Analysis review included the Conceptual Foundation and Analytical Framework Appendix (Foundation and Framework) and the Entrainment Technical Appendix. The Foundation and Framework described the high-level vision, purpose, and regulatory foundation for the Effects Analysis. It also provided an overview of the proposed methods to accomplish the analysis. In the most recent drafts of the BDCP Effects Analysis, the Foundation and Framework (originally Appendix A of the BDCP) concepts have been incorporated into Chapter 5: Effects Analysis.

Panel Members

- Alex Parker, Ph. D., California Maritime Academy, California State University (Panel Chair)
- Charles "Si" Simenstad, M.S., University of Washington (Lead Author)
- T. Luke George, Ph.D., Colorado State University
- Nancy Monsen, Ph.D., Stanford University
- Tom Parker, Ph.D., California State University San Francisco
- Greg Ruggerone, Ph.D., Natural Resources Consultants, Inc.
- John Skalski, Ph.D., University of Washington

The Panel member's biographies are included in Appendix A of this report.

Charge to Panel

The Panel was charged with assessing the scientific soundness of Chapter 5: Effects Analysis and the associated technical appendices, including recommendations for how these might be improved with respect to achieving their stated goals (Appendix B). The charge directed the Panel to address six general questions on Chapter 5: Effects Analysis and review of eight specific topics that had been formulated by the BDCP agencies. In addition, seven other questions were addressed on the approach, analysis and models described in the Chapter 5 technical appendices.

Review Schedule

- October 2011
    - The Panel convened in Sacramento to discuss the Foundation and Framework and Entrainment Technical Appendix and made initial recommendations.
- November 2011
    - Phase 1 Panel report completed November 28, 2011.
- April/ May 2012
    - The Panel reconvened in Sacramento to discuss BDCP Chapter 5: Effects Analysis and the many of the technical appendices. Appendices 5.E:

Habitat Restoration and 5.G: Fish Life Cycle Models were not reviewed at this time.

- June 2012
  - Phase 2 Partial Review Panel report completed.
- December 2013
  - An informational briefing was provided for the Panel. It included an overview of changes to the Effects Analysis and associated technical appendices since the Phase 2 review, including the changes made in response to the Panel's previous comments.
- January 2014
  - The Panel convened in Sacramento to discuss the BDCP Chapter 5: Effects Analysis and technical appendices on January 28-29.
- March 2014
  - Phase 3 report completed.

Organization of Report

We have sought to organize the Panel's review comments and recommendations around the questions framing the Charge to the Panel (Appendix B). Given the extensive volume of review material in Chapter 5 and its associated appendices, our ability to draw on other chapters in the entire BDCP document and all other supplemental material provided to the Panel was considerably limited and inconsistent. However, we attempted to reduce our own uncertainties by exploring the whole body of the BDCP as much as was feasible within the constraints on our time and resources.

For each of the Panel Charge questions we provide a brief ***summary*** section, a series of bulleted ***recommendations***, and a ***comments*** section with more detailed discussion. In order to maintain this structure throughout the report, there is some redundancy, particularly between the summary comments and detailed comments sections.

Summary observations

Reponses to Phase 1 and Phase 2 Panel recommendations

Many of the recommendations from the Phase 2 report should still be referenced while developing the adaptive management plan and initial rules for operating the north Delta diversion facility. Highlighted below are some Phase 2 recommendations that are relevant in this Phase 3 report.

Recommendation 1: Analysis of biological effects needs more consistency and specificity

In some respects, the current draft of the Effects Analyses lacks even more specificity than before, although it may be that sections were moved to other chapters. The 'multi-author' problem is apparent in the variation in assessments found in different locations. Most biological objectives for covered fishes were not fully evaluated in Chapter 5 because information was deemed to be insufficient (Table 5.2.8). Requests for full aquatic food webs were followed and a reasonable conceptual food web was provided, but it was incomplete.

Recommendation 2: Net Effects Analysis needs greater objectivity

Regardless of the degree of uncertainty and the number of linkages without analyses, the conclusion is often overstated as the most beneficial result. Many biological models were analyzed without any sensitivity analyses; consultants would say, 'there's no data,' but they could have said, 'what if we were just 90% correct here, or 60% correct', or 'what if the benefits of restoring wetlands are delayed 10-15 years over our most positive perspective' – but none of those alternative scenarios were considered.

Recommendation 3: Increase consistency of stressor analysis across covered species, and provide more detail.

Chapter 5 identified a ranking approach that addressed: 1) importance of attribute to the population; 2) effect of stressor on individuals; and, 3) certainty of 1 & 2. However, the analysis did not transparently follow through with this approach.

Recommendation 4: Chapter 5 must be a "stand alone" document

The synthesis quality of the Effects Analysis was improved. But reference to specific sections of technical appendices and other supporting documentation could be improved in many sections. Given uncertainty in effects analysis, more description of monitoring and adaptive management would be worthwhile to show that the BDCP would adequately address the uncertainty.

Recommendation 5: Clarify the baseline

The baseline(s) was described, although the baselines vary with regulatory agency. This complicated an already very complicated and lengthy Effects Analysis.

Recommendation 6: Provide systematic understanding and planning for conservation actions, especially restoration

Achieving beneficial conservation measures requires understanding limiting factors, ecosystem processes, sequencing, adaptive management responses, thresholds for certain actions, and interactions and other consequences of these actions. Otherwise, this isn't a conservation plan, but rather a conservation menu that generally fails to describe how major uncertainties will be resolved. For instance, while the Effects Analysis recognizes that suspended sediment has been declining in the Sacramento River and that the new diversions would remove an additional 8-9%, all analyses used a high and constant amount with no mention of downstream sediment effects on either Suisun or San Francisco Bay. Similarly, the uncertainty about being able to remove *Egeria* or other invasive species is not directly addressed in Chapter 5. *Egeria* is certainly poorly considered in the context of the aquatic food webs. Bivalves are not incorporated into aquatic food web analyses, although they're mentioned as 'uncertainties'.

While the conceptual model of food web enhancement support of covered species through restoration of tidal wetlands is more thoroughly covered, potential changes in the contributions of different food web sources and subsidies are still treated as disparate. Discussion of the Delta's potential food web structure and dynamics under BDCP conservation measures still fails to treat the Delta as a system, with spatially and temporally integrated sources of phytoplankton-based and detritus-based secondary production. There remains the need to provide a synthetic view of the potential benefit of restoration to the covered species that represents the integrated ecosystems and processes that fuel that food web, and potentially enhance it under the BDCP.

No additional detail has been provided for the Restoration Opportunity Areas (ROAs), other than their general locations. There is very little mention of how they will be connected, interact or be sequenced. What criteria have been developed to provide that guidance, or is it entirely dependent on opportunity (real estate costs, availability, public land, etc.)? Ultimately, adaptive management incorporating an extensive management structure and large representation of stakeholders will need to be implemented in order to resolve issues and uncertainties. There is a tremendous trust embodied in an ill-defined adaptive management process.

Recommendation 7: Include indirect effects of contaminants as part of Appendix 5.D: Contaminants

Indirect effects of contaminants on covered species via food web effects (i.e., contaminant effects on the microorganisms that make up the food web that covered species depend on) are almost certainly important.

Recommendation 8: Accurately characterize food resources and food webs

While there is now more comprehensive assessment of both phytoplankton- and detritus-based food web pathways proposed to be enhanced by BDCP conservation measures, the Effects Analysis still leaves the impression that phytoplankton and macrophyte (e.g., tidal marsh) production are separate, almost “opposing” alternative food webs. Only a simple depth model is used for phytoplankton production, nothing else incorporated. Many things are now mentioned in the text, no analyses incorporated, no discussion of potentially modified planktonic composition, etc.

Recommendation 9: The hydrodynamic modeling needs to capture the entire domain of effects

1) New guidelines will need to be put in place to regulate tidal (and maybe tidally averaged reverse flows) in the north Delta channels including Steamboat, Sutter, and Georgiana Sloughs. The operation of the Delta Cross Channel also needs to be rethought. These new guiding regulations need to be in place *before* exports start to occur in the system.

2) The current Effects Analysis does not consider the influence of shifting timing of withdrawals on San Francisco Bay circulation patterns and ecology. This is a significant omission with ecologically important implications.

Recommendation 10: Incorporate life cycle models for all species, as quantitatively as possible

Appendix 5.G identified a number of life cycle models, but eliminated all but two to be used in the effects analysis. The Panel questioned whether some models were inappropriately dismissed. The two models used in Chapter 5 both involved winter Chinook salmon. Thus, the large majority of covered species were not evaluated with life cycle models. The Panel asks why the BDCP did not develop life cycle models when beginning the process.

Recommendation 11: Consider salmonids at stock and life history scale

This aspect of the Effects Analysis was also improved. Each salmonid stock was evaluated. “Forager” versus “migrant” life histories were compared and evaluated, but proportions of each life history type did not seem to be considered in the analysis of net effects. Furthermore, the relative proportion of wild versus hatchery fish contributing to each life history type was not considered.

Recommendation 12: Identify analytical tools that need to be developed to address the extremely high uncertainty involved with calculating sediment supply and turbidity

Multiple statements within Chapter 5 and Appendix 5.C indicate that turbidity distribution is largely unknown.

Recommendation 13: Levels of uncertainty are not adequately addressed

The Effects Analysis provides an improved recognition of uncertainty, but there’s not better resolution of uncertainty than in previous drafts and the more complete discussion of uncertainty is often buried in the appendices. As a result, Chapter 5 reflects the lowest common denominator in terms of uncertainty. The level of uncertainty was often mentioned when evaluating the effect of a stressor on a species. Uncertainty was also mentioned when estimating net effects. However, conclusions regarding covered fish often overstated potential beneficial effects while not adequately addressing the lower-end effects.

Recommendation 15: Include sensitivity analyses and model validation in the effects analysis for covered fish species

While sensitivity analyses would have informed the Effects Analysis in the case of some of the biological models, this recommendation was generally not followed.

Recommendation 16: Provide more detail about the specific approaches that will be used when implementing adaptive management

Given the tremendous levels of uncertainty associated with critical assumptions and predictions inherent in the Effects Analysis, the burden of sustaining or enhancing covered species will seemingly fall almost entirely on adaptive management, particularly “active” adaptive management where explicit interventions may be required. However, it remains unclear how many of the critical uncertainties can or will be addressed as explicit experiments. While the Adaptive Management Plan is appropriately, and often effectively, designed to specifically address the major uncertainties, thresholds, triggers and alternative measures need to be explicitly derived from conceptual and numerical models. In some cases, metrics or success criteria have yet to be identified (e.g., Table 3.D.2). Furthermore, the critical monitoring that would be required for effective decision making and adjustments are often relegated to research actions rather than mandated effectiveness monitoring, which presents potential lack of commitment or delay in timely resolution of critical uncertainties. Given the critical importance of monitoring and adaptive management to BDCP success, it would be worthwhile to have an explicit section within Chapter 5 that specifies how monitoring and adaptive management has been designed and implemented to address specific uncertainties, test critical assumptions and predictions and sequenced to improve the chance of success.

Recommendation 17: Ensure a declining fish population (e.g., longfin smelt) does not decline further while waiting for possible beneficial effects of habitat restoration

The key assumption is that food production will be the primary benefit to longfin smelt from habitat restoration measures. Winter-spring flow is also believed to be key factor affecting abundance. Chapter 5 states that the key question is the extent to which abundance can be increased through improved food production and how these improvements interact with the spring outflow-abundance relationship. Recognition of the length of time needed to restore habitats and increase food production for longfin smelt could be strengthened in Chapter 5.

Accessibility of Effects Analysis elements

The Panel recognizes that the complexities involved in the process to develop and the ultimate structure of the BDCP are enormous, and as a consequence reviewing one component such as the Effects Analysis can be inhibited by lack of clear knowledge of the other components, expanded detail or underlying rationale. Furthermore, the Panel found it difficult to readily track down key information in the 745 page Effects Analysis (Chapter 5), which was supported with eight appendices containing an additional 4,500 pages. In general, in spite of its length, we often found assumptions or conclusions stated in the Effects Analysis to be lacking in sufficient detail to stand alone without links to Effects Analysis appendices or other BDCP chapters that provided the necessary detail or background. Although outside the charge of the Panel, we often found after digging further into the BDCP documents that the Effects Analysis was supported with some information. We recommend that for recognition of the voluminous and detailed information supporting the Effects Analysis, and ease of migrating through it, a simple system of (appendix/chapter and page-line number) cross referencing be employed to point the reader to that supporting information.

General Charge Questions

1. How well does the Effects Analysis meet its expected goals?

Summary

Compared to the initial development of the BDCP Effects Analysis, the Panel consensus is that the Phase 3 version is a much improved and impressive compilation of background material and scientific and technical knowledge about the Bay-Delta that provided a plausible basis for the conservation measures. The Panel concluded that all of the available data and arguments for the rationale behind the Effects Analysis assumptions and conclusions are contained within the BDCP documents, although we suggest that the Effects Analysis (Chapter 5) itself is still poorly substantiated and leaves too much to appendices and other BDCP chapters without explicit cross references. The lack of accessibility to information conveys a “trust us” message.

Evaluation of BDCP effects was typically systematic in that it attempted to identify key attributes affecting Covered Species and described, to the extent possible, the importance of that attribute, the potential effect of the BDCP on the attribute, and uncertainty regarding the evaluation. Findings from multiple approaches taken to assess potential effects were described and strengths and shortcomings were identified when possible. However, this level of detail, which sometimes included conflicting information, inhibits rather than elucidates comprehension of the findings.

The tenuous conclusion drawn from the Effects Analysis is that <u>many of the critical justifications behind the supposed benefits of the conservation measures are highly uncertain</u>. Other than the impression that the foundation of the BDCP is weak in many respects, the default burden to ensure Covered Species benefit, if not recovery, rests on adaptive management. The adequacy of the BDCP therefore rests not in the intent and development of the conservation measures, but in the rigor and application of adaptive management to ensure that the critical uncertainties are addressed and strategically incorporated into a progressively refined Plan.

There is great potential in the area of decreasing invasive aquatic vegetation (IAV) abundance. Control of extremely invasive IAV, such as *Egeria densa* (Brazilian waterweed) and *Eichhornia crassipes* (water hyacinth), could be substantial and effective if the Plan follows through on its actions. The prospects of success with predator control appear marginal and then only if hotspot actions are followed through year after year. The effects of water withdrawals by the Plan may lead to expanded populations of the non-indigenous, invasive clams *Potamocorbula amurensis* and *Corbicula fluminea* without further direct actions to control their population growth. The fate of *Microcystis aeruginosa* is also not promising. Between trends in climate warming and planned water withdrawals, the prospects for *Microcystis* blooms appear to remain unchanged or slightly worse under the Plan, although the direction of these potential outcomes is highly uncertain.

The Effects Analysis develops a robust conceptual model of aquatic food webs and the diverse linkages that may impact the net production of food for covered fish species. Yet, the Effects Analysis contains a number of assumptions, some of which are inappropriate (such as the magnitude and location of invasive clam depression of phytoplankton production), and others highly uncertain. Uncertainties are mentioned, but no effort was made to include conservation efforts reaching only a portion of the biological objectives and goals. Thus the analysis of effects further assumes only the most beneficial potential results, but doesn't incorporate other possibilities. Other aspects of food webs in aquatic habitats are described but remain unanalyzed, some of which may enhance, while others may inhibit achievement of biological objectives. While the overall conceptual model is adequate, integration and synthesis is lacking. Consequently the conclusions and net effects are not appropriate given the gaps in analyses and the uncertainties.

For terrestrial communities and covered species, the Effects Analysis provides a simple accounting of the number of acres of natural communities and suitable habitat that will be removed and restored but very little information is provided about the management actions that will be implemented to maintain them over the duration of the conservation plan.

Recommendations

- Provide detailed cross-referencing and indexing between Chapter 5 and the associated technical appendices as well as other chapters of the BDCP, especially the Adaptive Management Plan.
- Improve reporting of uncertainty levels within Chapter 5 Effects Analysis, including within the Executive Summary.
- Identify the most relevant monitoring indicators necessary to evaluate the trajectory of outcomes with respect to the biological objectives,
- Complete work on biological objectives.
- Provide triggers for adaptive management
- Guide the scientific community by highlighted research priorities to address critical information gaps.
- Improve on the systematic approach for integrating net effects for Covered Species.
- Develop life cycle models for each of the Covered Species in order to evaluate BDCP effects

Comments

The length and detail of the text and accompanying tables indicate considerable effort to document information used to determine the net effects. However, this level of detail, which sometimes included conflicting information, inhibits rather than elucidates comprehension of the Effects Analysis findings.

Overall, the BDCP and the 22 conservation measures have the goal to enhance fish and wildlife species in the Plan Area. Twenty-one of the conservation measures involve actions intended to restore habitat and benefit Covered Species. Conservation Measure 1 (Water Facilities and Operation) also has the goal to benefit covered species but this specific action involves activities that may adversely impact species (e.g., water removal and construction activities) while also benefiting some species (e.g., reduced entrainment at the south Delta pumps). Therefore, a key goal of the BDCP Effects Analysis is to determine whether the overall positive effects of the conservation measures outweigh the adverse effects of water removal and project construction, and if so, to what degree.

The Effects Analysis attempted to evaluate the effects of the BDCP on each covered fish species in an open, unbiased manner. Sixteen life-cycle models for Covered Species were examined for applicability to the BDCP, but only two were deemed to be relevant. It was not clear why life cycle models were not developed for the specific purpose of evaluating BDCP effects on each of the Covered Species. Quantitative effects could not be described, rather effects of each attribute were ranked as zero, low, moderate, or high effect. A systematic approach to synopsize the overall net effect on each species was not used even though a ranking approach that could have been used in a systematic roll-up was described. Instead, professional judgment was used to assess the overall net effect.

If there is one area of general scientific consensus among the Panel about the implementation of the Bay Delta Conservation Plan is that its <u>outcomes remain highly uncertain</u>. As such, one would expect that the Effects Analysis would reflect this general conclusion by stressing a high level of uncertainty around all of its conclusions. There is also general consensus among stakeholders that the high level of uncertainty should not be an impediment to any action in the restoration of the Bay Delta ecosystem. The only way to address the highly uncertain outcomes of BDCP implementation is through rigorous monitoring and adaptive management. The BDCP Effects Analysis should better integrate where uncertainty exists, identify the most relevant monitoring indicators necessary to evaluate the trajectory of the outcome, provide triggers for adaptive management and guide the scientific community by highlighted research priorities to address critical information gaps. On these points the Effects Analysis as a stand-alone document falls short.

Table 5.2-8 identifies the biological objectives for each of the covered fish species and whether or not the Effects Analysis was able to assess the likelihood of the BDCP achieving the objectives. Some of the biological objectives were quantitative, thereby providing a specific metric that could be evaluated both prior to BDCP implementation and after implementation. For example, for winter-run Chinook originating in the Sacramento River, the objective is to achieve a 5-yr geometric mean survival through the Delta of 52% by year 19 (from an estimated 40% at present), to 54% by year 28, and to 57% by year 40. Although the table notes that this objective is interim and subject to possible change as new data are collected, the Review Panel complements the BDCP team for developing quantitative biological objectives to be achieved within specific time periods. Ideally, the Effects Analysis should evaluate likelihood of the BDCP achieving each biological objective.

The inability to fully evaluate the likelihood of achieving each biological objective at this time highlights the need for a rigorous monitoring and Adaptive Management Plan. Chapter 5 seems to recognize this need in light of the incomplete evaluation of biological objectives. The Panel was not tasked with reviewing monitoring and adaptive management plans. Nevertheless, monitoring efforts should be designed to quantify whether or not the biological objectives are being achieved. The adaptive management plan needs to be linked to monitoring with identified trigger points and actions to steer the effort towards achievement of the biological objectives.

For terrestrial communities and covered species, the Effects Analysis, for the most part, provides a simple accounting of the number of acres of natural communities and suitable habitat that will be removed and restored but very little information about the management actions that will be implemented to maintain them over the duration of the conservation plan. The estimates of habitat restoration assume that restoration targets for the different habitats will be achieved with certainty, an assumption that unlikely to be met. In addition, the contribution of natural community restoration to species habitat restoration is estimated by multiplying the percentage of modeled habitat comprising the natural community by the total acres of natural community restoration in the plan area. This approach, however, confounds the spatially explicit nature of many of the species distributions within the Plan Area. For instance, only the riparian woodland south of Highway 4 within the Plan Area is considered potential riparian woodrat habitat which makes sense given their current distribution. The riparian woodland in this region currently comprises approximately 12.1% of the riparian woodland in the entire Plan Area. It is inappropriate to apply this percentage the estimate the amount of restored habitat in the Plan Area that will be available to riparian woodrats. If none of the restored habitat occurs south of Highway 4 then none of it will be potentially available to riparian woodrats. It makes much more sense to identify only riparian woodland restored south of Highway 4 as potential riparian woodrat habitat. Because the distribution of many of the species in the Plan Area is limited by their current distribution and dispersal abilities, the potential for colonization of restored areas should be identified using spatially explicit information. In the case of the riparian brush rabbit and riparian woodrat, a specified number of acres of riparian woodland should be restored within their potential range in the Plan Area.

The issue of the management of terrestrial communities and covered species is addressed in very broad terms in Chapter 5. In some cases there is mention of maintaining communities in a successional state that will make it suitable for a particular species (e.g., early successional riparian forest for riparian brush rabbits and western yellow-billed cuckoo), but many of the uncertainties surrounding long-term management of species and habitats are subsumed into adaptive management. Adaptive management is unlikely to succeed unless clear targets and thresholds for alternative management approaches are identified.

2. How complete is the Effects Analysis; how clearly are the methods described?

Summary

The Effects Analysis is a monumental effort incorporating over 745 pages of text and another 4,500 page of supporting appendices. The assessment covers potential changes in the physical environment, natural communities (12), fish (11 species), wildlife (25) and plant (12) species associated with BDCP. For fish species, 12 different categories of stressors and 32 attributes were examined over four different life stages. As many as 14 different operating scenarios were examined from the status quo to the long-term effects of BDCP implementation with climate change. For terrestrial species, areas of habitat loss and gained through management actions were examined.

Chapter 5 provides an overview of the spatial and temporal scope of the analysis, definitions of project baselines that differ depending on regulatory authority, recognition of climate change information, identification of a variety of models used to evaluate effects, treatment of viable salmon population criteria, and the approach to determining net effects on fish and wildlife. Biological goals and objectives were identified; this is important because the Effects Analysis should address each biological objective.

As might be expected, with the size of the Effects Analysis task, the quality of the assessments ranged in scientific rigor based on the amount of available data and best available science. Some aspects of the assessment, e.g., such as water quality and flow, were quantitatively assessed using sophisticated mathematical models. Aspects of the Chinook salmon assessment were also based on empirical data and process-based models. However, for many of the other fish species and their potential stressors, conceptual models supported by the scientific literature were the only recourse. In the case of Effects Analysis on fish, a workshop of professional biologists was used to incorporate feedback and to better express levels of uncertainty associated with assessment conclusions. The distinction between conclusions drawn from quantitative models and conceptual models was made clear.

The vastness of the Effects Analysis report and appendices is both its strength and weakness. In order to draw conclusions regarding effects of individual stressors or net effects on a species, it was often necessary in the report to draw on information from a number of appendices or other sections of the report. In many cases, these sections were not referenced or the specific findings of those sections not restated. This leaves the reader to hunt for the pertinent facts. It also appears at times that conclusions are based on a select subset of the facts that influence both the strength and certainty of the conclusions.

Because the variety of topics that the BDCP covers, how clearly the methods are described varies between topics. Several panelists gave input into Question 2 based on their areas of expertise.

Covered Fish

Approximately 72% of the objectives for covered fish could not be fully evaluated at this time due to insufficient information. The overall net effects conclusion for each species seemed to be based on the judgment of the authors, rather than a systematic ranking of attribute importance, change in response to the BDCP, and uncertainty in the rankings. Sixteen life cycle models for Covered Species were examined for applicability to the BDCP, but only two were deemed to be relevant, although the Panel is concerned about the exclusion of some life-cycle models. A systematic approach for synthesizing the net effect on each Covered Species was not used even though a ranking system was described that could have been used as a semi-quantitative scoring approach. Instead, professional judgment was used to assess the overall net effect.

In section 5.5, the text describes a numeric ranking for evaluating the importance of the attribute to the species, and the effect of the BDCP action on the attribute. The summary table (e.g., Fig. 5.5.1-5) was extremely difficult to read, used text to describe the effect (zero to high) and color to describe certainty. A small, essentially illegible "-" sign identified negative rankings. This summary table needs to be redesigned to improve readability.

No major omissions for the scientific literature or failure to use best available data were found in the Effects Analysis. However, the Effects Analysis did not develop new methods when gaps in assessment capabilities were encountered. For example, no attempt was made to modify any of the existing delta smelt models for the express purpose of this assessment.

An inevitable risk in using any mathematical model is extrapolation outside the range of the model. This extrapolation is likely whenever projecting to environmental conditions that have not yet occurred such as the changes that could be brought about by the BDCP. It is imperative that model-based assessments clearly state when such extrapolation is occurring and the potential direction of bias that might likely arise.

Hydrodynamics

The coupling of the multi-D, DSM2, and CALSIM II models is not a standard method that would naturally be understood by the reader. The documentation for this coupling is part of the EIS documentation, not part of the BDCP documentation. A short summary of the method should be included in Chapter 5.

Terrestrial species

The methods for the terrestrial species are adequately described in the various appendices (but see specific comments on the description of the methods for the habitat restoration in Appendix 5.J.B).

Recommendations

Over-arching recommendations

- Include a table of cross-references for each section or appendix referenced in the Net Effects.
- Add formal comparisons of model results in the Effects Analysis and appendices.
- Include within the Net Effect sections, discussions of contradictions or non-supportive facts in order to better capture some of the uncertainty in the conclusions.
- Emphasize the following Effects Analysis statement: "*These expectations represent a working hypothesis of the relationship between actions, stressors, and biological performance.*"

Covered fish

- Model-based assessments should clearly state when extrapolation is occurring and the potential direction of bias that might likely arise.
- Redo the format of the effects on attributes summary tables (e.g., Fig. 5.5.1-5) to improve readability.

Hydrodynamics

- A short summary of the method to inter-link multi-D hydrodynamic models, 1-D (DSM2) models, and CALSIM II should be included in Chapter 5.

Comments

Effects on Covered Fish

Chapter 5 addressed topics that it should address given information available at this time. Chapter 5 provides an overview of the:

- spatial and temporal scope of the analysis
- definitions of project baselines that differ depending on regulatory authority
- recognition of climate change effects on future conditions
- identification of BDCP actions
- identification of a variety of models and their limitations for evaluating BDCP effects
- an ESA take assessment including effects on viable salmon population criteria
- a qualitative approach for determining effects of each attribute on species habitat and performance
- an approach for classifying certainty of the effects analysis, and
- a description of the approach for evaluating overall net effects of the BDCP on each fish and wildlife species.

Additionally, biological goals and objectives were identified in Chapter 5. Identification of biological goals and objectives in Chapter 5 is important because the Effects Analysis should address the ability of the BDCP to achieve each biological objective. However, Chapter 5 states that approximately 72% of the objectives for covered fish could not be fully evaluated at this time due to insufficient information. It is noted in Chapter 5 that these information needs would be incorporated into monitoring and research actions, which are described in Section 3.6 (not reviewed by the Panel). Given the incomplete information, the Effects Analysis states, "*These expectations represent a working hypothesis of the relationship between actions, stressors, and biological performance.*" This is an important statement that should be highlighted in Chapter 5 rather than in the middle of a paragraph on page 5.2-36.

Implementation of methods for evaluating BDCP effects was not readily transparent. Section 5.5 describes a numeric ranking approach for evaluating 1) the importance of the attribute to the species, and 2) the effect of the BDCP action on the attribute. Rankings reportedly ranged from -4 to +4. These two values were reportedly multiplied to develop a ranking of effect for each attribute. Certainty was reportedly evaluated using the same numerical ranking approach for both the importance of the attribute on the species and the BDCP effect on the species attribute. This approach seems reasonable given the limitations of existing information, and the evaluation would be transparent. However, the numeric values of these rankings were not presented or discussed in the BDCP. Instead, figures were presented (e.g., Fig. 5.5.1-5) that used text to describe the effect (zero to high) and color to describe certainty. A small, essentially illegible "-" sign identified negative rankings. It was not clear whether this summary figure incorporated the importance of the attribute to the population, although importance of the attribute was often described in the text.

The numeric ranking approach described above was not used to evaluate net effects of the BDCP on each species, even though it seems that it could have been used and compared with the professional judgment evaluations. Instead, the overall net effects conclusion for each species seemed to be based on the judgment of the authors, rather than a systematic ranking of attribute importance, change in response to the BDCP, and uncertainty in the rankings. Chapter 5 notes that its conclusions were compared with professional judgments of agency personnel provided during a series of workshops in August 2013. This is worthwhile, but a table showing the variability in the judgments would have been useful as a means for indicating variability in the assessment rankings.

The Panel does not provide comments on methodologies presented in the technical appendices, except when discussed below. The level of detail in the descriptions of methodologies in the appendices varies considerably. In many cases, the original document must be consulted for a description of the methodology. Given the variety of information sources, referral to the original report for methodology was not unexpected.

Hydrodynamics

One of the issues that had to be worked through with the hydrodynamic models for the Effects Analysis was how to use hydrodynamic models that were designed for the current bathymetric configuration of the Delta and the watershed. The CALSIM II model is a watershed optimization model that has operational criteria based on salinity intrusion into the Delta. Changing main point of diversion in Conservation Measure 1, adding ROAs in Conservation Measure 3, and factoring in climate change (especially sea level rise), all change the circulation patterns in the Delta and the associated salinity intrusion. It is necessary to use the physically based multi-dimensional hydrodynamic models to first calculate hydrodynamic parameters (stage and flow) and salinity throughout the system. Because the multi-dimensional models are computationally intensive to run, the results of the multi-dimensional models are used to calibrate the DSM2 (1-D) model. The DSM2 (1-D) model is then used to create the relationship between salinity intrusion and river input flows. This river inflow-salinity intrusion relationship is what CALSIM II needs for optimization.

The coupling of the multi-D, DSM2, and CALSIM II models is not a standard method that would naturally be understood by the reader. The documentation for this coupling is part of the Environmental Impact Statement documentation, not part of the BDCP documentation. A short summary of the method should be included in Chapter 5.

Effects on Terrestrial Species

The methods for the terrestrial species are adequately described in the various appendices (but see specific comments on the description of the methods for the habitat restoration in Appendix 5.J.B).

3. Is the Effects Analysis reasonable and scientifically defensible? How clearly are the net effects results conveyed in the text, figures and tables?

Summary

The effects analysis covers a multitude of topics. Each panelist provided input into Question 3 based on their areas of expertise.

Overall approach to determine net effects

The Effects Analysis, particularly for covered fish, tries to incorporate information on potentially beneficial or detrimental effects covering 12 different stressors, 32 attributes, and multiple life stages using best available information and science. Only a perfect life cycle model with perfect information on all the effects and their interactions could possibly weight the results correctly and draw unambiguous conclusions. Any and all actual effects analyses are far from that measure of perfection, including the BDCP. The effect summary figures (e.g., Figure 5.5.2-5) attempt to illustrate the multidimensional aspects of the assessment process and, along with the Net Effect narratives, try to convey an overall assessment conclusion. A serious limiting factor of the current cumulative Net Effects is a near complete absence of any explicit weighting (in summary tables) of the biological importance of the many attributes under consideration (e.g., Figure 5.5.1-5). Size and direction of anticipated effects on the attributes is provided in the summary figures, along with color coding levels of certainty. Even though summary tables show values for each life stage, what cannot be discerned is whether any <u>critical</u> life stages or attributes are being adversely affected by the BDCP. Consequently, it is also unclear whether the Net Effects conclusions are correctly taking critical life stages into account when deriving overall Net Effects conclusions.

The approach to net effect conclusions needs to be reconsidered and revamped. The net effect summary figure (e.g., Figure 5.5.2-5) does not include the relative importance of the categories (e.g., food, entrainment, etc.). Without incorporating their relative importance in the summary figure, net effect conclusions are potentially meaningless and uncertainty cannot be characterized. The net effect conclusions for a fish species need to therefore take into account the relative importance of the various categories, make them explicit, and interpret Plan effects within that context on a species-by-species basis.

Covered Fish

The Effects Analysis does not adequately defend conclusions regarding the net effects of the BDCP, including habitat restoration. Habitat restoration certainly has the potential to increase the productivity of species such as salmonids, but the literature contains relatively few studies documenting the population response of salmonids to habitat restoration. The conclusion statements from Chapter 5 (and/or the Executive Summary) tend to overstate the beneficial effects of BDCP for many different fish populations (e.g., salmonids, delta smelt, green and white sturgeon). The net effects analysis tends to over-reach conclusions of positive benefits for covered fish species, given the inability to quantify the overall net effect and the realization of high uncertainty.

Key issues/questions that still need to be address for covered fish include:

1. The importance of interactions between BDCP flows and habitat restoration.
2. Will the migrant life history sufficiently benefit from conservation measures to offset moderate negative impacts related to reduced spring flows? Migrant salmonids may benefit less from conservation measures, and may experience a negative net effect.
3. To what extent is foraging habitat and exposure of foraging salmonids to predators affected by reduced spring flows?
4. The text does not distinguish between hatchery versus wild salmonids in the analysis.

Conceptual Models

In general, the conceptual models for dissolved oxygen and contaminants are well developed, although consideration of nutrient form and nutrient ratios (e.g., Glibert *et al.* 2011) would be a nice addition given the interest and recent publications on these topics. Also, algal toxins could be an attribute for monitoring to reduce uncertainty in contaminants and food web conceptual models.

Although there are good synthetic conceptual models developed for the Bay-Delta longfin smelt population encapsulated in the Effects Analysis (e.g., Baxter 2010; Rosenfield 2010), the conceptual model is still constrained by the lack of a life-history model that would elucidate the role of prey composition and abundance in population dynamics.

Food Webs

Restoration of tidal wetlands (and other communities) is highly uncertain and at least an extremely long process. The Effects Analysis does not adequately justify the critical assumption of the benefit of tidal wetland restoration as a food web subsidy for covered pelagic fish given the uncertainties of tidal wetland restoration itself. The conceptual model of the food web appears to include many of these processes. However, within the narrative current understanding as well as the implications of inherent uncertainties are not fully explored.

Organic matter subsidies to the Delta Food Web

There is an expectation that restoration activities will result in increases in abundance of lower trophic levels but the structure of the lower food web will be critical in whether this increased production can support covered species. Not only quantity, but also quality of the primary production that is supported by restoration activities is important. Water residence time within ROAs and other characteristic transport timescales for Delta channels are not the only factors to consider. The type of phytoplankton primary production that is stimulated is highly uncertain and likely dependent upon water temperature, nutrient concentrations, vertical mixing and grazing. In addition, an increased residence time may promote toxigenic cyanobacteria (*Microcystis aeruginosa*).

Hydrodynamics and physical changes at export facilities

For hydrodynamic modeling, only one set of ROAs were modeled. Because the locations of these assumed ROAs are not being presented to the public, there are details of the hydrodynamic modeling that cannot be factored into the Panel's evaluation of the Effects Analysis.

Conservation Measure 1 now includes significant modifications to Clifton Court Forebay. This region has been identified as a predation hot spot by multiple studies. Reduction in predation hot spots should be considered in the physical design.

Terrestrial species

The Effects Analysis for terrestrial species focuses almost exclusively on a simple summation of the number of acres of suitable habitat that are removed or restored for each species by the conservation measures. The simple accounting approach does not consider landscape-level effects such as connectivity and patch size nor does it take into account differences in habitat quality.

Recommendations

Overall approach to determine net effects

- Clearly indicate on effect summary figures (e.g., Figure 5.5.2-5) both beneficial (+) and detrimental (–) effects.
- In order to incorporate biological importance into the Net Effects process, the rows (i.e., categories, attributes) of the effects figures (e.g., Figure 5.5.21-5) could be ranked or rearranged in clusters according to biological importance for the specific species (e.g., high, medium, low). In this way, it would be easier to assess whether any biologically important attributes are likely to be negatively impacted and at what level of impact. It will also allow readers to discern whether any biologically important attributes also have high levels of uncertainty assigned to them.
- From the August 2013 Covered Fish workshops, it would be valuable to include in the Net Effects summary, what fraction of the attendees agreed with the Net Effects conclusions (i.e., direction, amplitude and level of certainty).

Covered fish

- Examine and re-write conclusion statements about population net effects in both Chapter 5 and the Executive Summary to objectively express the range in anticipated population effects.
- Evaluate effects of conservation measure attributes on species while considering all other potentially interacting conservation measures.
- Consider relative abundance of salmon life histories when evaluating net effects on each species.
- “Wild” salmonids should be considered separately from hatchery fish whenever possible.

Conceptual Models

- Consideration of nutrient form and nutrient ratios (e.g., Dugdale *et al.* 2007; Glibert *et al.* 2011) would be a nice addition to food web models given the interest and recent publications on these topics.
- Algal toxins could be an attribute for monitoring to reduce uncertainty in contaminants and food web conceptual models.

Food Web

- A simple surface area versus water volume calculation would provide a first-order estimate of potential food subsidy to open water habitats of the low salinity zone.
- Evaluate and compare the magnitude and temporal and spatial variation in the multiple organic matter subsidies to the Delta food web.
- Incorporate into the Effects Analysis the idea that tidal wetland restoration may mitigate some of the nutrient loading into Delta by acting as a nutrient sink through emergent vegetation production, phytoplankton production as well as fluxes to the atmosphere via denitrifcation.
- Estimate the potential food web subsidy attained based on the degree to which habitats are connected hydraulically to Suisun and Grizzly Bays. These areas could serve as “proof of concept” for other, unidentified Restoration Opportunity Areas.

Hydrodynamics and physical changes at export facilities

- When Conservation Measure 3 is implemented, the details of the connection between the Restoration Opportunity Areas and the adjacent channels and the order in which the Restoration Opportunity Areas are established need to be top design criteria.
- Since Conservation Measure 1 is proposing significant physical changes be made to Clifton Court Forebay, the identified predation hot spots within Clifton Court Forebay should be considered in the re-design.

Terrestrial species

- Landscape-level effects should be considered.

Comments

Effects on Covered Fishes

A Comprehensive Summary Figure Would Be Useful*.* For specific actions affecting covered fishes, the Effects Analysis summarizes findings of multiples investigations when available and often qualifies the findings with opinion statements of how important the attribute might be and how certain the finding is. This assessment by the authors of the Effects Analysis is often compared with a summary of conclusions, including a statement of uncertainty, developed from a workshop with agency personnel in August 2013. This approach is reasonable given the information available, but as noted elsewhere, improvements could be made to systematically summarize 1) the relative importance of the attribute, 2) the level of change caused by BDCP implementation, and 3) the certainty of this evaluation. The relative importance of an attribute was often provided within the narrative of Chapter 5, but a comprehensive table or figure summarizing this metric was not presented along with the effect of the BDCP on the attribute and the certainty associated with the rankings. A comprehensive summary figure is a key step leading to the overall net effect determination for each species. This figure would also enhance transparency in the final professional judgment of net effects*.* Furthermore, some sections of the Effects Analysis did not seem to reach a conclusion or describe the certainty about the findings, e.g., text description of Feather River flow effects on spring Chinook (see Feather River discussion below).

Salmonid Life History Increases Uncertainty. Salmonids have a variety of juvenile life history types that result in differential use of Delta habitats over time. The Effects Analysis characterized these life history types as foragers and migrants. Foraging juvenile salmonids are younger, smaller and typically inhabit shallower habitats compared with larger, older yearling salmonids that pass through the Delta relatively quickly. Recognition and consideration of these two life history strategies in the BDCP Effects Analysis (e.g., Fig. 5.5.3-4) is important. However, as noted below, the complex life history of salmonids, including life history differences between wild and hatchery origin fish, leads to greater uncertainty in the overall net effect of the BDCP actions on salmonid populations.

Literature Shows Major Restoration Needed to Improve Fish Populations*.* The Effects Analysis does not adequately defend conclusions regarding the net effects of the BDCP, such as habitat restoration, on fish species. Habitat restoration certainly has the potential to increase the productivity of species such as salmonids, but the literature, including published papers and technical reports, contains relatively few studies documenting the population response of salmonids to habitat restoration (see reviews by Roni *et al.* 2008, 2011). Findings in the literature on the response of salmonid populations to habitat restoration was not adequately addressed in the Effects Analysis when describing the net effect of each species, although the methods section (5.2.7.10.3) did provide a reference by NMFS stating that quantitative linkages between specific habitat actions and viable salmonid population criteria is difficult*.* The difficulty in documenting population responses to habitat restoration should be recognized and addressed with large and strategic habitat restoration projects and detailed monitoring. For example, in a comprehensive evaluation of salmon responses to habitat restoration in Puget Sound, Roni *et al.* (2011) concluded:

> "Given the large variability in fish response (changes in density or abundance) to restoration, 100% of the habitat would need to be restored to be 95% certain of achieving a 25% increase in smolt production for either species. Our study demonstrates that considerable restoration is needed to produce measurable changes in fish abundance at a watershed scale."

Conclusions Often Overstate Beneficial Effects. The Panel believes that the net effects analysis tends to over-reach conclusions of positive benefits for covered fish species, given the uncertainty and inability to quantify the overall net effect. Given the findings of Roni *et al.* (2011), it may be inappropriate to extend an uncertain but potentially positive effect conclusion to statements about species conservation, especially under future climate scenarios. For example, the following grand conclusion statements from Chapter 5 (and/or the Executive Summary) tend to overstate the beneficial effects of BDCP:

> "The magnitude of benefits for winter-run Chinook salmon at the population level cannot be quantified with certainty. Nonetheless, the overall net effect is expected to be a positive change that has the potential to increase the resiliency and abundance of winter-run Chinook salmon relative to existing conditions."

Statements about increased resiliency and abundance are inappropriate given the high uncertainty expressed in the initial sentence. The statements tend to focus on the upper end of beneficial effects rather than a balanced analysis that might capture the range in net effects. The Panel underlined the questionable text.

"The BDCP should help conserve the species in the Plan Area and help it cope with expected climate change…." The term "conserve" implies a large beneficial population effect for salmon that may help the population recover from ESA listing. Maybe the BDCP will lead to a positive effect, but the magnitude of the effect is uncertain, as stated above, so it seems inappropriate to imply the BDCP would eliminate attributes in the Delta that cause lower population viability. The life cycle models suggested climate change effects would overwhelm the evaluated BDCP actions on winter Chinook salmon.

The following conclusion for delta smelt overstates and over-emphasizes the potential for significant beneficial effects (by emphasizing great potential) while also noting the conflicting conclusion of high uncertainty in the net effect: "While there is great potential for large benefits for delta smelt, there is a high level of uncertainty regarding the resulting effects. However, combined with the Fall X2 decision tree, the BDCP will have at least a minor beneficial effect on the species, but a great potential for larger benefits depending on actual food production and location of delta smelt population in relation to restored areas." The high-end benefit is emphasized in the BDCP text. Perhaps there is higher certainty for a positive versus negative net effect but there is high uncertainty for the net effect of actions on the delta smelt population, ranging from little to high population effect. This evaluation would benefit by the removal of "great".

For green and white sturgeon, the BDCP concluded: "Therefore, the BDCP is expected to conserve both species in the Plan Area through improvements in abundance, productivity, life history diversity, and spatial diversity." The term "conserve" implies a large beneficial population effect that was not supported by the evaluation. The conclusion statement also implies and therefore overstates measureable positive changes to four population viability criteria. These benefits may reflect the goals of the BDCP, but the uncertain magnitude of benefits to sturgeon should not be described as improving abundance, productivity, life history diversity, and spatial diversity.

Interactions between conservation measures. Interactions between BDCP flows and habitat was not adequately addressed in the report. For example, Table 5.5.3-4 shows that habitat units typically increased for foraging salmonids in response to habitat restoration, but the habitat analysis did not appear to consider whether salmonids would have access to the habitat during reduced flows under the BDCP scenarios (see Table 5.E.4-1). For example, flows were expected to be ~15% to 20% lower during January to April when many foraging salmonids are rearing in the Delta area. In other words, how much rearing habitat is available and what is the habitat quality for foraging salmonids when flows have been reduced 10-20%? The Cache Slough region is one example where key habitat restoration sites might be affected by reduced river flows. Perhaps tidal fluctuations overwhelm river flows in some of the lower habitats, but this should be stated in the report. For foraging salmonids, do reduced flows of the BDCP negate the reported habitat gains from some restoration activities? Recommendation: evaluate effects of conservation measure attributes on species while considering all other potentially interacting conservation measures. This approach was taken for some measures (e.g., Delta Passage Model evaluations) but not all.

Migrant salmonids may benefit less from conservation measures and may experience a negative net effect. The effect of each attribute on migrant versus forager salmonids was examined in Chapter 5, but summary Figure 5.5.3-2 did not capture differences in the assumed relative abundances of these life histories among the species. Plan area flows were typically ranked as a moderate negative effect on migrant salmonids in the Sacramento River and a low negative effect on foragers. However, this attribute was ranked the same for each salmonid species regardless of the proportion migrants versus foragers assumed in the population. The negative impact of reduced plan area flows should have been greater on Sacramento River species such as spring Chinook and steelhead that are dominated by migrant life histories.

Migrant life histories are less likely to benefit from habitat restoration activities, which are a key focus of the BDCP conservation measures. This implies that spring Chinook and steelhead may experience less benefit from BDCP actions than other salmonid species, or they may even experience a negative net effect in response to reduced spring flows. The key question, which deserves more attention in the BDCP, is whether the migrant life history will sufficiently benefit from conservation measures to offset moderate negative impacts related to reduced spring flows. This question is key for spring Chinook and steelhead that are composed mostly of migrant life histories.

Characterize uncertainty in plan area flow effects on salmonid life history types. The Delta Passage Model (DPM) is a key tool for this evaluation because it predicts survival of migrant salmonids while considering river flows, passage into interior areas, entrainment to pumps, and passage into the Yolo Bypass. The survival model is largely based on Chinook salmon exceeding 140 mm in fork length, therefore the DPM does not represent foragers or smaller migrants, which are the target of the habitat restoration activities.

The Effects Analysis states that it was assumed with moderate certainty that flow has high importance to foraging winter Chinook salmon, then notes that the moderate level of uncertainty reflects the relative lack of investigation on the influence of flows on smaller salmonids (Page 5.5.3-24, line 39-41). Moderate uncertainty is quite different from moderate certainty, which is also concluded in each salmonid summary figure (e.g., 5.5.3-4). If there is no information on how flows affect survival of smaller foraging salmonids in the Delta, it is difficult to accept a moderate level of certainty associated with the low negative impact of flows on foraging juveniles salmonids, especially when data suggest flow has a significant effect on larger salmonid (migrant) survival (Fig. 5C.5.3-4). To what extent is foraging habitat and exposure of foragers to predators affected by reduced spring flows? For winter Chinook and fall Chinook, the forager life history is the dominant type, indicating less certainty about the net effect of BDCP flows on these species compared with species dominated by migrant life histories that have been tagged and analyzed, e.g., Fig. 5C.5.3-4.

Hatchery versus "wild" origin salmonids. The presence of hatchery salmonids is typically noted in the introductory descriptions of each salmonid species in Chapter 5. The degree to which hatchery salmonids contribute to the two life history types was not described, though hatchery fish are released as migrants. For example, 80% of juvenile spring Chinook were assumed to be migrants. To what extent was this due to the release of migrants from hatcheries given that some of the natural population produces primarily foragers? The text does not otherwise distinguish between hatchery versus wild salmonids in the analysis. Although some hatchery stocks are protected by the ESA, it would seem that wild salmonids would have a higher priority than hatchery-produced salmonids, even though hatchery runs provide important role in the Central Valley and ocean fisheries. Perhaps resolution of effects and uncertainty inhibit analyses specific to wild salmonids. Nevertheless, wild salmonids should be considered independently from hatchery salmonids when possible.

Do habitat actions only affect salmonid capacity and not productivity? Fig. 5.5.3-2 shows BDCP effects on productivity of each salmonid species by attribute. No effect is shown for habitat attributes such as channel margin, floodplain, riparian, etc. In contrast, these attributes are scored in other Figures for each species, e.g., Fig. 5.5.3-4. Does this reflect an opinion that these habitat actions only increase the capacity of the habitat to support salmonids rather than habitat quality?

Obtain more information from life cycle models. Life cycle simulations were only performed for winter-run Chinook salmon using the OBAN and IOS models. Comparison of through-delta survival and adult returns by management scenario (Table 5.G-2) was very useful. One way to compare and evaluate the two models is to assess consistency in the management scenario rank (best to worst) for the various response variables. For instance, if the same management scenario always ranks first, then that would indicate high level of consistency and support for that conclusion. On the other hand, if management scenario rankings varied greatly between assessments then conclusions would have high degrees of uncertainty (See Table 1, below).

Some life cycle models inappropriately excluded. Appendix 5G excluded delta smelt life cycle models in the Effects Analysis without adequate justification. Based on the premise of using the "best available science," it is unclear how none of the delta smelt models could have reached that level of acceptance. One justification was that none of the models used zooplankton data; however, the BDCP Net Effects assessment indicated zooplankton was only of moderate importance to delta smelts (Figure 5.5.1-5). It would therefore seem that some assumptions about zooplankton could have been made, allowing life-cycle modeling to be performed. Robustness studies could have accompanied the modeling process. Furthermore, if the BDCP team felt none of the delta smelt models to be adequate, why was there no investment made in model development for such an important species of interest?

Net Effects

The Net Effects summary figures (e.g., Figures 5.5.1-5, 5.5.2-5, etc.) are very useful for synopses for each fish species, but they are incomplete. It would be visually helpful to explicitly include both positive (+) and negative (–) signs for each combination of life stage and category. There continue to be discrepancies between conclusions regarding certainty and level of effect between the text and summary tables. The quantitative scoring method described on page 5.5.1 seems to be largely ignored. Instead, a qualitative ocular assessment of the summary tables seems to be applied separately to the certainty and level of effect dimensions. For salmonid species, weighting is discussed for migrant vs. foraging forms, but this too is seemingly ignored (or at least not mentioned) in the Net Effect conclusions.

The approach to Net Effects conclusions needs to be reconsidered and revamped. The Net Effects summary figures (e.g., Figure 5.5.2-5) do not include the relative importance of the categories (e.g., food, entrainment, etc.). Without incorporating their relative importance, Net Effects conclusions are potentially meaningless and uncertainty cannot be characterized. Levels of uncertainty have different weight depending on the importance of the various categories. An assessment might have high uncertainty for all low importance categories and still have high overall certainty if all the important categories carry with them high certainty. Conversely, the overall assessment would have low certainty, if one or more of the high importance categories carry high uncertainty. The Net Effects conclusions for a fish species needs to therefore take into account the relative importance of the various categories, make them explicit, and interpret Plan effects within that context on a species-by-species basis. Uncertainty plus uncertainty is more uncertainty. Uncertainty never averages or cancels out uncertainty; any more than noise plus noise is less noise. One graphical approach to conveying importance of the various categories and attributes is to order or group the rows of the figures according to their importance for a particular fish species. It would then be possible to see if any detrimental effects of the BDCP are associated with any important biological processes or not.

Life-cycle simulations were only performed for winter-run Chinook salmon (i.e., models OBAN and IOS). Comparison of through-Delta survival and adult returns by management scenario (Table 5.G-2) was very useful. One way to characterize model consistency is to assess how consistent the management scenarios rank (best to worst) across the models and different response variables. For instance, if the same management scenario always ranks first, then that would indicate a high level of consistency and support for that conclusion. On the other hand, if management scenario rankings varied greatly between assessments, conclusions would have a high degree of uncertainty.

<u>Restoration of tidal wetlands (and other communities) is highly uncertain or at least an extremely long process</u>

Restoration of tidal wetlands is considered in detail in the section on aquatic food webs (Question 12). In general, tidal wetland restoration of biological function is quite difficult with respect to ecosystem processes beyond tidal flux and especially with respect to ecological equivalency to comparable natural wetlands. This has been reviewed in a number of studies and conclusions have remained consistent over the past two or three decades (e.g., Kentula 1996, Simenstad and Thom 1996, Zedler and Callaway 1999, BenDoer *et al.* 2009, Moilanen *et al.* 2009).

<u>Lack of specificity in Restoration Opportunity Areas limits conclusions of many aspects of Effects Analysis</u>

For the hydrodynamic modeling, only one set of Restoration Opportunity Areas were modeled. (See discussion of implementation of models in Question 2.) Because the locations of these Restoration Opportunity Areas are not being presented to the public, there are details of the modeling that cannot be factored into the Panels evaluation of the Effects Analysis. As examples: 1) in Panel Question 7, the placement of the Restoration Opportunity Areas influences reverse flows in Georgiana Slough, 2) the calibration of the 1-D model based on the 2-D model results is sensitive to Delta Cross Channel operations, which could be the result of Restoration Opportunity Areas representation in the system. (See question 5 Restoration Opportunity Areas modeling discussion.) When Conservation Measure 3 is implemented, the details of the connection between the Restoration Opportunity Areas and the adjacent channels and the order in which the Restoration Opportunity Areas are established need to be top design criteria.

<u>Clifton Court Forebay physical changes need more evaluation before implementation because of its reputation as a predation hotspot</u>

Conservation Measure 1 now includes significant modifications to Clifton Court Forebay. These modifications include building a wall in Clifton Court Forebay to create two separate regions, the north region would receive water from the North Delta pump facilities and the south region would receive water from the existing south Delta channels. In addition, the current size of the Clifton Court Forebay would also be enlarged by flooding an adjacent tract of land to the south. Based on the public panel discussion with ICF and the Fish agencies on January 29, 2014, the philosophy behind the modifications is that the water coming from the North Delta facilities will have already been pre-screened for critical fish species. Therefore, there would be significant savings in not filtering north Delta diversion (NDD) water through the south Delta fish screening facility.

ICF acknowledged that this is a newer element of the design for Conservation Measure 1. There was no documentation in Appendix 5.H (Aquatic Construction and Maintenance Effects) regarding this construction. The building of a dam in the center of Clifton Court Forebay and dredging another tract should be considered in Appendix 5.H.

Clifton Court Forebay has been identified as a predation hot spot by multiple studies. The Fish Predation science panel (Grossman *et al.* 2013) stated in their final report that: “Clifton court Forebay (CCFB) has been identified by multiple sources as an inhospitable location for salmonids. Within CCFB several areas are particularly hazardous including: 1) the deep scour hole just inside CCFB by the radial gates; 2) the trash gates in front of the Tracy Fish Collection Facility; and 3) section of Old River adjacent to the radial gates." Since Conservation Measure 1 is proposing significant physical changes be made to Clifton Court Forebay, these predation hot spots should be considered in the re-design.

Delta Food Web

5.3.38 Cache Slough and Suisun Marsh Restoration Opportunity Areas are suggested as areas of substantial increase in Prod-Acres. Given that these Restoration Opportunity Areas are defined, some work could be done to estimate the potential food web subsidy attained based on the degree to which habitats are connected hydraulically to Suisun and Grizzly Bays. These areas could serve as "proof of concept" for other, unidentified Restoration Opportunity Areas. An interesting outcome of such an exercise would be a determination of the potential for export and trophic transfer (a positive outcome) versus localized cultural eutrophication, increased biochemical oxygen demand and dissolved oxygen sags in tidal sloughs (negative outcome).

The discussion of water residence time throughout the Delta (Section 5.3.36) suggests an increase of 3 to 4 days as compared to the current configuration. But this analysis is also site-specific. The approach used to calculate residence time is also of concern. The residence time in each Restoration Opportunity Areas is a function of bathymetry, the exchange between the Restoration Opportunity Area and the adjacent channels. The 1-D DSM2 model does not have the capability to calculate this parameter. In addition, because the specific locations and configurations of the Restoration Opportunity Areas are not presented in the Effects Analysis, the panel has no basis to comment on the validity of the approach.

The phytoplankton productivity model that results in Prod-Acres is limited in terms of prediction or certainty in outcomes. Again, it comes down to a question not only of quantity but also quality of the primary production that is supported. The result of longer residence time is likely to increase phytoplankton primary production (i.e., "slower is greener") this may not hold when invasive clams are introduced to the system (Lucas and Thompson, 2012). Additionally, the type of phytoplankton primary production that is stimulated is highly uncertain and likely dependent upon water temperature, nutrient concentrations, vertical mixing and grazing. Lehman *et al.* (2013) suggested that increased residence and warmer water temperatures in excess of 19 - 20° C will promote toxigenic cyanobacteria including *Microcystis aeruginosa*. It should be recognized that *Microcystis* is only one potentially important toxigenic cyanobacteria in the Bay-Delta – *Aphanizomenon* was abundant in 2011 and 2012 in the Bay-Delta (Karobe *et al.* 2013).

Tidal wetland restoration may mitigate some of the nutrient loading into the Delta by acting as a nutrient sink through emergent vegetation production, phytoplankton production as well as fluxes to the atmosphere via denitrifcation. These ideas are not considered within the Effects Analysis. The decay of large amounts of invasive aquatic vegetation (a result of control measures) also has the potential to increase biochemical oxygen demand and inorganic and organic nutrient supply; this may shift phytoplankton community composition and promote local eutrophication. This issue is raised in a single bullet point on page 5.F-130, line 26

Terrestrial Species

Rather than using current estimates of habitat occupancy within the Plan Area to estimate occupancy of restored habitat, we recommend using spatially explicit occupancy models (see comments under question 4). In addition, the minimum width and maximum distance of riparian habitat corridors should be identified for terrestrial mammals that are restricted to riparian habitats (riparian woodrat and riparian brush rabbit). Persistence of these species in the Plan Area requires riparian habitat patches that are sufficiently large to support stable populations as well as riparian corridors that will allow movement between suitable habitat patches. Both the minimum patch size and minimum corridor parameters (width, distance, overstory cover) should be specified to ensure long-term occupancy of restored riparian habitat.

4. How well is uncertainty addressed? How could communication of uncertainty be improved?

Summary

A broad consensus exists among the Panel that Chapter 5 does not adequately address uncertainty. In its current form, at the level of detail conveyed, in the models used, and in the verbal assessments and conclusions, the level of uncertainty is downplayed. Within appendices sometimes more explicit discussion of uncertainties can be found, but a disconnect exists between the summary pages with the conclusions drawn in Chapter 5. In situations in which an array of outcomes may be possible, only the more beneficial outcomes are quantitatively assessed or used in conclusions about the BDCP. Communication of uncertainty would be improved by consideration of a range of potential outcome values in models.

The Panel cannot determine whether the conclusions about covered fish species or other species in the BDCP are accurate. Detailed monitoring is needed to evaluate the BDCP conclusions, in addition to the outcomes for the biological objectives that could not be fully evaluated at this time in the BDCP. The BDCP effects analyses are qualitative and conclusions regarding net effects on each species typically reflect professional opinion. Therefore, the Effects Analysis does not lend itself to evaluation of chained statistical uncertainties. The tremendous length of the documents did not reduce the uncertainty in the overall net effects.

Recommendations

- Unknowns and research needs should be incorporated into the BDCP as explicit conservation measures, in other words, as a required part of the BDCP.
- Monitoring needs, timing and intensity also need more explicit incorporation into the BDCP. While often well explicated in an appendix (e.g., within Appendix 5.F-Biological stressors on covered fish), they are frequently absent within the material discussed in Chapter 5 or treated as an uncertainty.
- Research needs are often mentioned as sections within appendices. These should be consolidated within Chapter 5. This would help guide future research priorities for the Delta.

Comments

Effects on Covered Fishes

For covered fishes, when evaluating the importance of an attribute to a species and evaluating the effect of the BDCP on that attribute, the Effects Analysis was typically careful to describe the level of certainty associated with this evaluation. The evaluation of certainty was typically a judgment by the BDCP authors rather than a quantitative measure of certainty (e.g., standard deviation), therefore estimates of certainty have their own level of uncertainty. The Effects Analysis did not lend itself to evaluation of "chained statistical uncertainties" as identified in the charge questions addressed to the Panel because the effects analyses were not quantitative. Nevertheless, the judgments of certainty have value, though they could be improved upon (see below).

Judgments of certainty were also compared with judgments provided by California agency scientists at the August 2013 workshops. However, identification of agency certainty seemed to be the interpretation by the BDCP authors of the agency response rather than a systematic evaluation of certainty scores. At the January 2014 Effects Analysis Panel meeting, ICF noted that they did not think it was possible to consistently document variability in Effects Analysis evaluations by agency personnel at the August 2013 workshops. As a result, evaluation of certainty of BDCP effects on attributes of each species is limited to the interpretation of the BDCP authors.

Please see discussion above on the overall net Effects Analysis for each species. Although conclusion statements typically stated high uncertainty in the overall net effects, they also tend to ignore uncertainty when highlighting the potential benefits to conservation without also stating the lower end of the effects range.

Monitoring and Research

As an example of the high uncertainty in the BDCP to achieve biological goals and objectives, many of the sections of appendices have sections on monitoring and research needs. These often highlight impacts of conservation measures in which the outcomes may have a range of positive to negative impacts. The unknowns and research needs should be better incorporated into the analyses of biological impacts of the BDCP. At a minimum they should be required as an explicit conservation measure. In a number of instances, especially in Appendices, for example Appendix 5.F, needs are highlighted for a robust monitoring and evaluation program, coupled with a detailed, prescriptive adaptive management plan. BDCP success will depend on monitoring and evaluations and responding to issues as they emerge. Furthermore, high uncertainty in the outcomes for the covered species means that budgets for monitoring and adaptive management must be developed with uncertainty in mind.

Disconnect between uncertainty and BDCP conclusions

Frequently, explicit modeling is reduced to small portions of conceptual models. When a range of potential outcomes may result from uncertainties in multiple conditions, only the most beneficial outcome is considered when coming up with a conclusion or summary. Some of these are discussed in other sections of this report. One example can be found in Appendix 5.F. When considering the impacts of some of the conservation measures, for example, Conservation Measure 13, removal of *Egeria* is discussed with multiple potential effects (Appendix 5.F, p. 5.F-48 and following), some beneficial, such as removing habitat for predators of covered fish, while others may exacerbate populations problems for covered fish, such as cascading effects through the food chain of the loss of some invertebrates that feed on *Egeria*, shifts in aquatic web linkages, and the rapid replacement of *Egeria* by other invasive submerged aquatic vegetation. Nonetheless, these uncertainties are simply ignored when it comes to conclusions, where it is determined that only the beneficial results of control invasive aquatic vegetation will result from the BDCP (pp. 5.F-48-49). To be fair, occasionally the poorer results dominate conclusions; for example, *Microcystis* may increase due to management activities inside and outside the region but these conclusions fail to emerge in the discussion of the aquatic food webs within Chapter 5.

The discussion of the aquatic food webs is based on a good conceptual model, but the dynamics of the food web are ignored and only a single component, phytoplankton productivity, is modeled as a result of restoration efforts in the relatively near- and far-term. Detrital contributions could also enhance food webs, but are not considered in any detail. Phytoplankton productivity is unrealistically modeled, and assumed to essentially be consumed along linkages that connect directly to covered fish. Chapter 5 does mention invasive bivalves, but fails to incorporate their potential as direct competitors for plankton within the food web, even though that potential is discussed. In other words, the BDCP is inconsistent in how models and analyses handle uncertainty and model assumptions, making it difficult to complete assessment.

Restoration

Because this is discussed in other sections, we will only mention that there is great uncertainty associated with the restoration of the wide range of ecosystems slated for restoration. Many of these systems have a poor record of achieving restoration, especially in short-to-moderate time periods. This range of ecosystems also varies considerably in the degree of difficulty of restoring functions. Nonetheless, the outcomes for conservation measures and their interaction and effectiveness are glossed over and uncertainties are not apparent in conclusions and summary discussions. For example, wetland restoration will require considerable input of sediment in the short-term to meet the outcomes described in the BDCP. Yet Chapter 5 models tidal wetland restoration with a constant concentration of suspended sediment, even though the document discusses the fact that sediment has been declining over the past decades, and further that the operations of the north Delta pumps may remove 8-9% more. In other words, there is considerable inconsistency between a discussion of uncertainty and how uncertainty is incorporated into the conclusions.

Similarly, restoration of many of the terrestrial habitats for other covered species also involves considerable uncertainty, especially as to the rate at which function will return that will be recognized by covered species. Consequently uncertainty of the occupancy targets for terrestrial species are not addressed. In all cases, a single value of number of acres that will be occupied is provided. No estimates of the uncertainty of achieving stated restoration goals nor uncertainty of the proportion of the restored habitat that will be occupied are included.

North Delta Diversion

In addition, the validity of the primary assumption that there will be no entrainment of fish at the north Delta diversion (NDD) should be evaluated. In reality, there will be some fish lost at the transfer point; therefore, the empirical relationship would be altered including this additional transfer point.

Water Clarity and Suspended Sediments

Section 5.3-24 (lines 31-38) correctly identifies a low level of certainty around changes in water clarity but does not include the potential positive or negative implications for changes in water clarity.

Suspended sediment is one of two key components driving the development of tidal wetlands in the Delta, especially under sea level projections, yet Delta inflow has been experiencing a decline in suspended sediment and operations of the NDD may remove 8-9% more. BDCP indicates there may not be sufficient sediment for marsh restoration (Chap. 5, p. 109).

The NDD operations should factor in suspended sediment into the operational criteria. Adaptive management should consider the possibility operating the NDD such that the first flush, which contains a large sediment load, is not exported.

5. How well does the Effects Analysis describe how conflicting model results and analyses in the technical appendices are interpreted?

Summary

The Effects Analysis covers a multitude of topics. Each panelist gave input into Question 5 based on their areas of expertise.

Hydrodynamics

Hydrodynamic models are sensitive to how the open water regions are represented and how they are connected to the adjacent channels. Because the panel was not provided the bathymetric configuration of the Restoration Opportunity Areas or the order in which the Restoration Opportunity Areas were established, it is not feasible to evaluate the sensitivity of the models to the placement of the Restoration Opportunity Areas. DSM2 (1-D) and RMA/TRIM (mult-D) hydrodynamic models represent Restoration Opportunity Areas is differently. This could be a significant source of error, especially when Delta Cross Channel gates configuration is open.

Life cycle models: winter Chinook salmon

No formal comparison of output from the OBAN and IOS models was provided, either on an absolute scale or relative scale. It should be acknowledged that adult escapement differs between models by a factor of 5. Through-Delta survival projects were also fractionally different between models. In neither case was an explanation for the discrepancy provided. The relative ranking of the different BDCP scenarios (Table 5.G-2) between models should be provided in the report, and certainly should be assessed, in part, based on the degree of consistency in predictions of the BDCP scenario ranks between models.

Salmonids: Delta Passage Model

For salmonids, the Delta Passage Model Salvage Estimates and the Salvage Density methods produced reasonably consistent estimates. Variance calculations need to be corrected. There appear to be analytical errors in expressing uncertainty.

Salmonids: Temperature Model

The text is not clear how the models predict these changes associated with the BDCP during egg incubation, if the BDCP has no effect on upstream conditions, as reported in sections of Chapter 5. In spite of these conflicting results, Figure 5.5.4-1 shows that there would be zero effect on eggs in the Sacramento River with moderate to high certainty in this conclusion. This evaluation needs clarification and should be consistent with the Appendix.

Terrestrial Species

Suitable habitat for each species in the Plan Area was based on expert opinion and therefore there are no model results to interpret. The plan adequately addresses conflicting estimates of the number of sandhill cranes that may be killed by collisions with powerlines.

Recommendations

Covered fish

- A direct comparison of the output from competing models should be presented.
- Clarify confusing and conflicting text related to salmon models.
- Explanation for the large discrepancies in predictions in adult returns (i.e., factor of 5) should be provided and possible consequences to Effects Analysis. Use of relative effects does not eliminate systematic biases of models.

Hydrodynamics

- Identify which Restoration Opportunity Areas are represented differently between the DSM2 and the RMA/TRIM models, especially in the Mokelumne system, which is sensitive to Delta Cross Channel operations.
- Publications from that CASCaDE (http://cascade.wr.usgs.gov/index.shtm) would be resources to guide the evaluation of propagation errors in the BDCP Effects Analysis.

Comments

Life-cycle models

When discussing IOS and OBAN life cycle modeling results, the Effects Analysis stated:

> "*The results of both models suggest future climate change effects would dominate changes in adult winter-run Chinook salmon escapement in the future, which is of appreciable concern for the species. Factoring in climate change, relatively small differences in upstream conditions between the BDCP LLT scenarios and EBC2_LLT resulted in greater adult escapement under HOS_LLT or lower adult escapement under ESO_LLT and LOS_LLT. These results reflect what appears to be appreciable model sensitivity to relatively small changes in estimated upstream conditions because, as noted above, <u>the BDCP does not change Shasta Reservoir and upper Sacramento River operating criteria</u>, so that changes in upstream areas derived from modeling, be they positive or negative, may not be fully reflective of the nature of actual changes that could occur*." (pg. 5.5.3-45, lines 38-46)

The above statement about climate change impacts on Chinook abundance is clear and noteworthy, but the text below it is confusing and should be clarified (did the model receive inaccurate information for upstream conditions?).

Chinook salmon

For egg incubation of spring Chinook, Chapter 5 describes conflicting results (pg. 5.5.4-14). The text states, “Several models show no change in upstream condition as a result of BDCP”. In the same paragraph, it states that SacEFT predicts a 12% reduction in egg incubation “condition” based on water temperature effects on egg survival. In contrast, the Reclamation Egg Mortality model predicts no effect due to the BDCP except in below normal water years (12% reduction in survival). SALMOD predicts negligible impacts of the BDCP on eggs, fry and smolt. The text concludes that the adverse impacts are related to high sensitivity of some models to small changes in upstream conditions. The text is not clear when describing how the models might predict these changes during egg incubation, if the BDCP has no effect on upstream conditions as reported in portions of Chapter 5. In spite of these conflicting results, Figure 5.5.4-1 shows that there would be zero effect on eggs in the Sacramento River with moderate to high certainty in this conclusion. This evaluation needs clarification.

- Habitat and flow modeling efforts in the Delta were not linked. As noted above, habitat suitability modeling indicates somewhat large habitat increases for foraging salmonids in response to restoration activities. However, these estimates of habitat did not account for reduced flows that would occur when juvenile salmonids are present in the Delta area, especially in wet years. In other words, will reduced BDCP flows affect access by juvenile salmonids to the habitat identified in Table 5.5.3-4, or do tidal fluctuations overwhelm river flows in all of these habitats?

Lack of consideration of propagation of errors or sensitivity analysis in linked models

A direct comparison of the output from competing models is rarely presented. Results from different models are rarely formally compared on either an absolute or a relative scale. When different models projections exist, the BDCP rarely attempts to explain why the discrepancies are occurring or describe the direction of the expected errors.

Uncertainty plus more uncertainty produces even more uncertainty. Uncertainty never averages or cancels uncertainty any more than noise plus additional noise produces less noise. The propagation of errors will not be a simple sum of uncertainties in most cases. One can use variance in stages formula

$$Var(\hat{\theta}) = E_2[Var_1(\hat{\theta}|2)] + Var_2[E_1(\hat{\theta}|2)]$$

to propagate errors over multiple processes or sequentially linked models and where 1 and 2 denote sources of error in estimating the parameter θ by $\hat{\theta}$. Levels of uncertainty have different credence depending on the importance of biological stressors or attributes. An assessment might have high uncertainty for all low-importance attributes and still have overall high certainty if all the important attributes carry with them high certainty. Conversely, the overall assessment would have low certainty if one or more high-importance attributes carry high uncertainty. Overall uncertainty will never be less than the highest level of uncertainty for the more important biological attribute being considered.

There are several different cases in the Effects Analysis where multiple models are linked together. Each model has inherent errors either due to assumptions made in the modeling or numerical method errors. One of the best examples of how to link models in the Delta system is the U.S. Geological Survey's CASCaDE project (http://cascade.wr.usgs.gov/index.shtm). Publications from that project would be resources to guide the evaluation of propagation errors in the BDCP Effects Analysis.

The assumptions made in hydrodynamic models TRIM/ RMA versus DSM2 or CALSIM2 result in a range of outcomes; their analysis is limited to only one set of ROA configurations

During the hydrodynamics presentation on 1/28, the calibration of the DSM2 (1-D) model compared to the TRIM/RMA (multi-d) models showed that the models agreed better when the Delta Cross Channel was closed than when the Delta Cross Channel was open. When the Delta Cross Channel is open, transport is influenced more by the circulation in the Mokelumne channels on the east side of the Delta.

The fact that the two models do not match well when the Delta Cross Channel is open indicates that the representation of Restoration Opportunity Areas is different between the 1-D and 2-D models. Hydrodynamic models are sensitive to how the open water regions are represented and how they are connected to the adjacent channels.

Because the panel was not provided the bathymetric configuration of the Restoration Opportunity Areas or the order in which the Restoration Opportunity Areas were established, it is not feasible to evaluate the sensitivity of the models to the placement of the Restoration Opportunity Areas.

6. How well does the Effects Analysis link to the adaptive management plan and associated monitoring programs?

Summary

While the adaptive management plan is considerably more developed in the BDCP Phase 3, it remains characterized as a silver bullet but without clear articulation about exactly how key assumptions will be vetted or uncertainties resolved to the point that the BDCP goals and objectives are more assured. The concept of adaptive management is appropriately described and allocated a prominent role in the implementation structure. However, as is increasingly documented, the commonly acknowledged process of adaptive management continues to be misunderstood and misapplied (Allen *et al.* 2011; Fontaine 2011; Westgate *et al.* 2013), often resulting in a loss of rigor and commitment in application. The consequence hasn't improved much since Walter's (1986) description of the adaptive management process as beginning:

> "…*with the central tenet that management involves a continual learning process that cannot conveniently be separated into functions like research and ongoing regulatory activities, and probably never converges to a state of blissful equilibrium involving full knowledge and optimum productivity*."

In the case of the uncertainties surrounding the assumptions and predictions of the BDCP, the Panel emphasizes that BDCP needs to recognize the risks of **not** institutionalizing an exceedingly rigorous adaptive management process in order to avoid ecological surprises that will be difficult or impossible to reverse once they have established (Lindenmayer *et al.* 2010; Westgate *et al.* 2013). BDCP must make a commitment to the fundamental process, and specifically the required monitoring, not just the concept of adaptive management. As Murphy and Weiland (2014) counsel:

> *"…adaptive management that targets listed species represents a complex process that can be resource intensive, including in its demand for guidance from research, monitoring, and modeling, therefore requiring substantial technical and institutional capacity. That considered, adaptive management has a great potential to improve the effectiveness and efficacy of resource management actions provided it is properly implemented."*

In the final assessment of the Effects Analysis, the Panel found the cautionary conclusion of Olden *et al.* (2014) about large-scale flow experiments to be particularly germane:

> "…*managers and policy makers must embrace both the scientific uncertainty and surprise learning opportunities that inevitably arise from these experiments, and not purposely ignore uncertainty to avoid complicating their message to stakeholders, only to later invoke this issue when flow experiments fail to deliver expected ecological or social outcomes*."

Recommendations

- The Effects Analysis effectively communicates the important principles and implementation stages of adaptive management, but the specific process whereby adaptive management would be utilized to ensure BDCP meets its goals and objectives by rigorous adaptive management need to be described in much more detail. There needs to be a more obvious commitment to active adaptive management.
- There is explicit linkage between key uncertainties underlying the assumptions of the Effects Analysis and the monitoring and research that need to address them through adaptive management. However, many of the critically uncertain ecosystem processes, population responses, etc. that are identified as adaptive management targets are delegated to research, rather than monitoring. Any metric upon which decisions about the expected or predicted performance of a management measure will be made should be a foundational monitoring metric, not a focus of research, which is often vulnerable to competing priorities.
- To facilitate an active adaptive management plan that has some chance of ensuring the beneficial result of BDCP conservation measures, each and every key uncertainty should be "fleshed out" into implementable adaptive management "experiments" where the following are specifically described: (1) a conceptual model, or components of an existing model, that characterizes the uncertainty and what it influences; (2) assessment of the relationship between the uncertainty and the BDCP goals and objectives; (3) sensitivity of the proposed implementation to the uncertainty; (4) success criteria, monitoring metrics, baseline levels, thresholds and trigger points that will identify whether or when the performance of the conservation measure is deviating significantly from the anticipated target or prediction; (5) alternative hypotheses and how they affect the original conceptual model; and, adaptation of the (6) implementation action or (7) adaptation of the goals and objectives.

- Linkages between scientific development of the Effects Analysis and adaptive management should continue, if not expand, with implementation of the BDCP. At the minimum, consider the necessity to guarantee independent science review at the interface between the Adaptive Management Team and the Implementation Office, to ensure close to real time tracking of adaptive management experiments and decisions.

Comments

Perhaps the largest challenge to achieving the stated goals and objectives of the BDCP is how many of these critical uncertainties can be addressed by adaptive management given the baseline and the required monitoring? For example, some of the key uncertainties identified in the Effects Analysis (Appendix 3.D), often associated with conservation measures 4, 5, 7, and 11, include:

- The ability of the restored habitat to meet the objectives and expected outcomes, including the time it takes to meet the biological objectives. (Can this be addressed by both magnitude and siting of restoration action?)
- The risk that the restored habitat will be colonized by invasive species such as nonnative submerged vegetation, nonnative predatory fish, and/or clams. (Hardly uncertain, but controllable?)
- The change in magnitude of predation mortality on covered fish. (Doesn't this require an existing reliable estimated of predation mortality?)
- Food web responses to habitat restoration actions on both a local and a regional scale.
- The risk of adverse effects resulting from unsuitable changes in water quality and exposure to toxic contaminants. (How much can be modeled?)
- The proportion of the covered species population that actively inhabit restored habitats and the change in growth rate, survival, abundance, life-history strategies, and population dynamics. (A very difficult baseline to quantify!)

The Effects Analysis provided explicit associations of such key uncertainties with each conservation measure and linked these to "potential research actions" (BDCP, Table 3.D-3).

The context of a "phased approach to serve as a large-scale experimental program" in adaptive management context implies conceptual models, baselines and thresholds?

Linkages between scientific development of the Effects Analysis and adaptive management should continue, if not expand, with implementation of the BDCP. In particular, it will be important to ensure that there is direct science input to the adaptive management process, and preferably an independent science body that has no conflict of interest in interpreting and adapting conservation measures. In the proposed implementation structure, the Science Manager chairs the Adaptive Management Team and coordinates with the Delta Science Program, and the Delta Independent Science Board may also be consulted about *"…matters relating to these monitoring activities and research efforts*." (Chap. 7-25, pp. 7-25). However, the Delta Independent Science Board is not engaged to the extent that they could deal with extensive monitoring and research results and adaptive management decisions in real time. We would doubt that the adaptive management process would be efficient, timely and evaluated without an independent scientific advisory body that reports to the Adaptive Management Team, Science Manager, Program Manager and the Delta Science Program.

Review of Specific Analyses

7. Are the analyses related to the north Delta diversion facilities appropriate and does the Effects Analysis reasonably describe the results? In particular:

Q. Was existing empirical information such as Perry *et al.* 2010 and Newman 2003 incorporated appropriately into the modeling? Where model runs required extrapolation beyond existing data ranges, were assumptions and interpretations appropriate?

Summary

The empirical information in Perry (2010) and Newman (2003) must be guardedly and cautiously applied in the modeling in future cases when north Delta diversion is operational. These empirical relationships are based on the best available information regarding current physical and operational configuration of the Delta. We assessed the validity of four model assumptions. The panel concluded: 1) the assumption of a 3-day moving average to characterize flow on the Sacramento below Georgiana Slough is not valid in the new configuration, 2) exporting water at the north Delta diversion facilities will change circulation patterns at the important north Delta channel junctions (i.e. Steamboat, Sutter, Delta Cross Channel, Georgiana), 3) an additional transfer point out of the Sacramento at the north Delta diversion will alter the empirical relationship, and 4) there are issues with original assumptions in Newman (2003). The concerns raised above, at best, add additional uncertainty to the conclusion drawn by BDCP. At worst, these concerns may result in systematic biases in the model projections. The direction of the net effect of these biases is unknown.

Recommendations

- Consult with Russell Perry and Ken Newman on their perspectives regarding the applicability of their models to the Effects Assessment.
- Perform more hydrographic modeling below the anticipated north Delta diversion to determine whether the nature of the outflow will violate assumptions or parameterizations of the Perry (2010) model and alter model output.
- Additive simulations should be performed varying the parameterization and possible structure of the relationships with Perry (2010) and Newman (2003) to determine robustness of the model results to changes in Sacramento River outflow under the BDCP.

Comments

The empirical relationships, derived in Perry (2010) and Newman (2003), are based on the best available information regarding current physical and operational configuration of the Delta. For these relationships to be useful, they also need to describe the Delta under BDCP. To assess the validity of these relationships, we must examine how the system will change with the addition of the north Delta diversion. There are four primary sets of questions to address: 1) Will the system continue to have a "quasi-steady state" condition or the will the timescale of flow variance change? Is a 3-day moving average to characterize flow on the Sacramento below Georgiana Slough a legitimate assumption?, 2) Will the circulation patterns change at the important channel junctions (i.e., Steamboat, Sutter, Delta Cross Channel, Georgiana) as a result of north Delta diversion operations?, 3) Will the north Delta diversion be another transfer point out of the Sacramento river migration corridor?, and 4) Are the assumptions used in the original analysis valid?

Will the system continue to have a "quasi-steady state" condition or will the timescale of flow variance change as the result of north Delta diversion operations?

In the current configuration of the system, the north Delta is in a quasi-steady state. In general, flows on the Sacramento at Freeport change slowly over time (i.e., on the order of days). The only operation that can dramatically alter circulation patterns is the opening or closing of the Delta Cross Channel gates. The position of this gate is not frequently changed. And, when changed, the system reaches a different quasi-steady state condition after about a day. A visual example of this step change is found in Perry (2010, Fig. 3). Therefore, the assumption of a three-day moving average to characterize flow on the Sacramento below Georgiana Slough seems reasonable for the current configuration (flow and operations) of the North Delta.

When the north Delta diversion facilities become operational, the North Delta will no longer be in a quasi-steady state condition. The flows will behave more like what is currently observed in the South Delta as the pumping will not be continuous throughout the day. And, pump volume will also change at least daily. The timescale of flow variance will change more rapidly over time (i.e., on the order of hours). Therefore, the three-day moving average flow assumption is not valid in the new configuration with the north Delta diversion.

Will the circulation patterns change at the important channel junctions (i.e., Steamboat, Sutter, Delta Cross Channel, and Georgiana) as a result of north Delta diversion operations?

We know that opening and closing the Delta Cross Channel changes the circulation patterns in the north Delta. Exporting water at the north Delta diversion facilities will also change circulation patterns at the important channel junctions (i.e., Steamboat, Sutter, Delta Cross Channel, Georgiana). The DSM2-Hydro simulations that were used for the analysis of this issue in section 5C.5.3.5 are capable of outputting data even on a 15 minute time step. This model resolution should be able to quantify these differences. If the circulation patterns change, the proportion of fish distributed to each downstream channel will be altered as well. Therefore, the empirical relationship created for the current configuration of the Delta is not valid for the future configuration.

Will the north Delta diversion be another transfer point out of the Sacramento migration corridor?

Throughout the analysis in 5C.5.3.5, there is an assumption of zero entrainment of as a result of 100% effective diversion screens. However, the north Delta diversion will be pumping water. Therefore, empirical relationship between the flow at Sacramento below

Georgiana and the number of fish present will be different from the current empirical relationship using the current (no north Delta diversion) configuration.

In addition, the validity of the primary assumption that there will be no entrainment of fish at the north Delta diversion should be evaluated. In reality, there will be some fish lost at the transfer point, therefore, the empirical relationship would be altered including this additional transfer point.

Are the assumptions used in the original analysis valid?

Newman (2003), Table 2 presents a summary of the covariates used in his modeling. There are two columns, mean and sample standard deviation. In this table, he reports a mean value for Delta Cross Channel gates of 0.61 with a sample standard deviation of 0.49. The Delta Cross Channel gate signal is a binary signal. It should be either open (1) or closed (0). Under no circumstances should that variable be reported as something other than 0 or 1. This analysis should have been broken into two time periods: gate open and gate closed conditions. This table raises a significant concern that the author did not have a basic understanding of how the Delta Cross Channel gate changes flow patterns (and migration patterns) in the Delta.

The concerns raised above, at best, add additional uncertainty to the conclusion drawn by the Plan. At worst, these concerns may result in systematic biases in the model projections. The direction of the net effect of these biases is unknown.

Q. Does the analysis of the frequency of reverse flows at Georgiana Slough accurately characterize changes in hydrodynamics due to changes in river stage, sea level rise, and Delta habitat restoration?

Modified question based on 1/29/2014 meeting discussion: Will the operation of the north Delta diversion change the circulation patterns around the Sacramento junctions with the Delta Cross Channel and Georgiana Slough such that fish (particularly migrating fish) have a higher likelihood of being diverted into the interior of the Delta via Georgiana Slough or the Delta Cross Channel due to tidal flood/ebb flows in this region?

Summary

We know, based on long-term field observations and hydrodynamic modeling, that the transition point from uni-directional flow and bi-directional flow at the tidal timescale occurs somewhere between Sacramento River above the Delta Cross Channel (RSAC128) and Sacramento River below Georgiana (RSAC123) for the current configuration and operations of the Delta. The operation of the north Delta diversion facility will reduce the amount of freshwater flow in the region of the Delta Cross Channel and Georgiana junctions. Hydrodynamic modeling will likely show that transition point between uni-directional and bi-directional flow will move upstream as a result of north Delta diversion operations. This transition location is also a function of whether the Delta Cross Channel is open or closed. If bi-directional flow occurs more frequently near the Sacramento junctions with the Delta Cross Channel and Georgiana Slough, fish will have a higher likelihood of being diverted into the interior of the Delta via Georgiana Slough or the Delta Cross Channel.

Recommendations

The DSM2 simulations should be re-run for the ELT and LLT simulations with bathymetry that does not include the Restoration Opportunity Areas but driven with ELT or LLT river flow and tidal stage boundary conditions and operations. These simulations would clearly show how north Delta diversion operations change circulation patterns near Georgiana Slough and the Delta Cross Channel.

Comments

During the Effects Analysis Panel presentation on 1/29/2014, one of the Panel members (N. Monsen) asked for clarification of Question 7b. Based on that discussion, we concluded that the main questions that the Fish Agencies would like to see the panel address were:

> *"Will the operation of the north Delta diversion change the circulation patterns around the Sacramento junctions with the Delta Cross Channel and Georgiana Slough such that fish (particularly migrating fish) have a higher likelihood of being diverted into the interior of the Delta via Georgiana Slough or the Delta Cross channel due to tidal flood/ebb flows in this region?*
>
> *Will this change in flow regime as a result of north Delta diversion operations result in fish encountering this junction multiple times rather than just once, thus increasing the probability of the fish being diverted into the interior Delta?"*

It should be noted that these rephrased questions are very different than what the analysis in Sections 5C.4.3.2.6 and Section 5C.5.3.8.1 of the Effects Analysis addressed. The following suggest an approach to answer the modified question and comment on the analysis in Sections 5C.4.3.2.6 and Section 5C.5.3.8.1.

Part A: Suggested approach to address the modified 7b question

For this discussion, please refer to the Draft Environmental Impact Report/Environmental Impact Statement Appendix 5A that has examples of observed tidal stage and flow time series data from three key locations along the Sacramento River (Appendix C of this document).

The Sacramento River throughout the Delta has a tidal signal for both stage and flow. The Sacramento observation station at Freeport (RSAC155), above the proposed north Delta diversion intakes, has a tidal flow signal (Appendix 5A-D1, p. 128). At Freeport, both the tidal and tidally-averaged flow is always uni-directional downstream. Therefore, a neutrally-buoyant particle going with the flow at this location will always be traveling downstream, although the velocity at which it moves is dependent on the phase of the tides.

In the current bathymetric configuration and operations of the Delta Cross Channel (no north Delta diversion facilities), the observation station on the Sacramento above the Delta Cross Channel (RSAC128, Appendix 5A-D1, p. 129) also has downstream uni-directional flow both for the tidal and the tidally-averaged timescale. However, the flow signal on the Sacramento below Georgiana Slough (RSAC123, Appendix 5A-D1, p. 130) has reversing tidal flows. Therefore, even though the tidally-averaged flow at RSAC123 is downstream. A particle moving with the velocity field in the region of RSAC123 will flow both upstream and downstream. Therefore, the tidal excursion or range that a neutrally-buoyant particle will move upstream and downstream, at

RSAC123 is important to determine how many times the particle will encounter junctions (such as Georgiana and Delta Cross Channel).

The Sacramento River above the Delta Cross Channel (RSAC128) and the Sacramento River below Georgiana (RSAC123) are only 5 river km apart and yet the flow signals at these stations are very different. These flow signals are distinctly different because there are two junctions, the Delta Cross Channel and Georgiana Slough, between these measurement stations where a portion of the water is diverted towards the Central Delta. The flow signal at RSAC123 also changes depending on whether the Delta Cross Channel is open or closed.

Therefore, we know, based on long-term field observations and hydrodynamic modeling, that the transition point between uni-directional flow and bi-directional flow at the tidal timescale occurs somewhere between RSAC123 and RSAC128 for the current configuration and operations of the Delta.

To determine how the north Delta diversion operations will change circulation patterns around the Delta Cross Channel and Georgiana Slough, the DSM2 model can be used to determine the location along the Sacramento where the flow transitions from unidirectional and bi-directional tidal flows. This transition location will also be a function of whether the Delta Cross Channel is open or closed. It is also useful to determine the extent of tidal excursion to determine whether particles would encounter either the Delta Cross Channel junction or the Georgiana Slough junction multiple times.

The operation of the north Delta diversion facility will reduce the amount of freshwater flow in the region of the Delta Cross Channel and Georgiana junctions. Modeling will likely show that transition point between unidirectional and bi-directional flow will be moved upstream. This transition point may be even as far upstream as RSAC128 (Sacramento above DCC).

Part B: Comments related to the analysis in Sections 5C.4.3.2.6 and 5C.5.3.8.1

The approach taken for the analysis in Sections 5C.4.3.2.6 and 5C.5.3.8.1 focused only on the exchange between the Sacramento River with Georgiana Slough. The approach of analyzing flow direction every 15 minutes was a reasonable approach given the original 7b question. However, the analysis did not attempt to also look at the exchange through the Delta Cross Channel, which should be done for the modified 7b question.

The bigger issue with this particular analysis is the assumed Delta bathymetry used for the ELT and the LLT simulations. For both the ELT and LLT simulations, Restoration Opportunity Areas are included in the bathymetry. The tidal field is significantly changed by the inclusion of these Restoration Opportunity Areas. Note that these Restoration Opportunity Areas are only one possible configuration. As of this BDCP draft, the final locations of the Restoration Opportunity Areas, the order of construction the Restoration Opportunity Areas, and the bathymetric connections between the Restoration Opportunity Areas and the adjacent channels have not been established.

In the BDCP conclusion for this analysis states:

> *"Ongoing research is investigating link is between the distribution of energy dissipation and the distribution of tidal prism within the context of Plan Area restoration and other factors (DeGeorge pers. comm.). … it is unknown whether the presently limiting conveyance capacity of a number of Delta channels for tidal*

> *flows may become enlarged by scouring in response to Plan Area changes in geometry resulting from habitat restoration. These factors may have consequences for the hydrodynamics at the Sacramento River-Georgiana Slough divergence and other locations."* (5C.53-331, lines 22-29)

This conclusion indicates that the present hydrodynamic modeling does not separate the effects of the north Delta diversion from the preliminary Restoration Opportunity Areas configuration in the ELT and LLT simulations.

One of the best reasons to use hydrodynamic modeling as an analysis tool is that models have the capability of isolating individual effects. The DSM2 simulations should be re-run for the ELT and LLT simulations with bathymetry that does not include the Restoration Opportunity Areas but does have the ELT or LLT river flow and tidal stage boundary conditions and operations. These simulations would clearly show how north Delta diversion operations change circulation patterns near Georgiana Slough and the Delta Cross Channel.

8. How should the effects of changes in Feather River flows on fish spawning and rearing be characterized? In particular, how should the trade-off between higher spring flows and lower summer flows be interpreted? Does the analysis adequately capture the expected benefits of CM 2, Yolo Bypass Fishery Enhancement?

Summary

Chapter 5 correctly recognized that flow/habitat relationships are necessary for evaluating changes in Feather River flow and temperature on salmonids. However, relationships between flow and habitat were not presented in Chapter 5, therefore it was not possible for the Panel to evaluate changes in spawning and rearing habitat. Most salmonids reportedly inhabit the low flow channel which will reportedly experience little change. BDCP effects relate primarily to the fraction of salmonid populations inhabiting the high flow channel plus fish exposure to the high flow reach during upstream and downstream migrations.

Chapter 5 provides a reasonable discussion of the approximate benefits of increasing flow into Yolo Bypass and allowing more juvenile salmon, especially foragers, to utilize this rearing habitat. Potential adverse effects on migrating adults should be monitored.

Recommendations

- Develop flow/habitat relationships for salmonids in the Feather River high flow channel, approximate the proportion of the population that uses this habitat, and correct inconsistencies in the text and summary figure.
- The Yolo Bypass evaluation should recognize that natural origin Chinook salmon have a higher fraction of foraging type juveniles compared with migrant Chinook produced in hatcheries. Natural origin juveniles would likely benefit more than hatchery fish.

Comments

Feather River

Salmon and Steelhead. Chapter 5 provided a summary of beneficial and adverse effects of Feather River flows on juvenile and spawning spring Chinook salmon. The analysis was based on expected changes in monthly flows in the low and high flow channels and associated changes in water temperature. The text recognizes that salmon habitat area and quality are important (see introductory paragraph), but the evaluation did not attempt to convert predicted flow and temperature scenarios to habitat units for steelhead and Chinook salmon. Lack of habitat data for each species reduces the certainty of the anticipated effects, except when flows and temperature are expected to experience little change, as in the low flow channel. Key to this analysis is the reportedly high use by salmonids of the low flow channel relative to the high flow channel, given that the low flow channel is expected to experience relatively little change.

The text states that juvenile spring Chinook salmon may be present in the Feather River from November through June. Chapter 5 also concludes that juvenile migration would not be affected by BDCP flows, which are higher in spring and lower in summer in the high flow channel during BDCP operations. Why is juvenile migration not affected by higher spring flows and lower summer flows? To what extent is rearing habitat in the high flow channel affected by higher flows and to what extent are juveniles using this habitat? There is no mention of the actual temperature experienced by the fish in the Feather River.

It is not clear how the low positive effect with moderate certainty (Figure 5.5.4-1) was derived, given that there was no presentation on flow/habitat relationships, which were discussed as being key to the analysis. Chapter 5 states that real-time operations could be used to minimize adverse effects in the Feather River, but there is no mention of whether this will be done and what the criteria might be to protect salmon. The Chapter 5 description of Feather River effects on salmonids did not incorporate information related to exceedance of minimum flows that was discussed in Appendix 5C.5.2.

For steelhead, the analysis and text involving Feather River flows are somewhat more conclusive. A key statement is that the vast majority of steelhead reportedly spawn and rear in the low flow channel which would receive little effect from the BDCP (what percentage of steelhead rear in the high flow channel?). Adult and juvenile steelhead may experience somewhat higher flows during migration, but there is no judgment of whether this is beneficial or not. The text also states that summer flows in the high flow channel would be reduced by 50%, a period that includes year-round rearing of steelhead. The Panel notes that steelhead prefer higher velocities than other salmonids, but changes in the amount of habitat in relation to velocity was not presented. The text concludes with moderate certainty that there would be a <u>low negative effect</u> in the Feather River (the text should clearly identify that it is the rearing stage in the high flow channel that is affected). However, Figure 5.5.6-1 shows zero effect on rearing steelhead and <u>low positive</u> effect on migration. The results in this figure are not consistent with the text.

Yolo Bypass

Chapter 5 provides a reasonable discussion of the approximate benefits of increasing flow into Yolo Bypass and allowing more juvenile salmon, especially foragers, to utilize this rearing habitat. Reported data indicate only ~12% of the juvenile population would utilize the habitat. For spring Chinook salmon, the analysis assumed 80% of the juveniles were migrant rather than foraging Chinook. These values apparently included hatchery spring Chinook salmon which are mostly migrants and less likely to utilize rearing habitat and benefit from Yolo Bypass compared with wild Chinook salmon that are more likely to be foragers that benefit from the Yolo Bypass. Yolo Bypass is more likely to benefit wild Chinook (to the extent that they are "foragers") than hatchery Chinook salmon, and it would be worth discussing this in Chapter 5.

Potential adverse effects of Yolo Bypass on juveniles, such as stranding, were described. Potentially adverse temperature effects or predation affects (if predators are attracted to the Bypass) were not described, but BDCP authors stated at the January meeting that temperature and predator attraction are not likely to pose a problem within Yolo Bypass. Adult salmonids could be adversely affected in Yolo Bypass, as discussed in Chapter 5; these fish should be monitored to ensure safe migration.

9. Does the analysis adequately describe the predation and other screen-related effects of the proposed north Delta diversion structures? Is the application of the observed mortality rate at the fish screen of the Glenn-Colusa Irrigation District (GCID) an appropriate assumption for expected mortality at the proposed BDCP north Delta intakes? Are there other studies on salmonid survival at positive barrier fish screens that would be appropriate to apply?

Summary

Chapter 5 concluded that there is a low negative impact related to contact and impingement of salmonids with the north Delta diversion screens, but the technical appendix states that this effect could not be evaluated. Regarding predation, the Panel believes that there is uncertainty about the extent to which juvenile salmon and predators will aggregate near the intakes, and this is an issue that must be monitored. Positive barrier fish screens are widely used throughout the Pacific Northwest to protect juvenile salmonids from entrainment into water diversions, and this information should be readily available to the BDCP team.

Recommendations

- Correct inconsistency in conclusions in Chapter 5 and the Appendix regarding impingement.
- Monitor predator aggregation and predation rates at north Delta intakes.
- Conduct literature search on positive barrier fish screens, which are widely used.

Comments

Screen contact and impingement

The Effects Analysis stated in regard to fish contact and impingements at the north Delta intakes:

"It is concluded with moderate certainty that there will be a low negative change to the north Delta intakes attribute to foraging and migrating juvenile salmonids as a result of contact and impingement at the north Delta diversions".

A reasonable summary of information leading to this conclusion was presented, although more information on relative abundances of foraging Chinook (smaller & more susceptible fish) versus migrant Chinook could have been presented. It was stated that monitoring would occur during operation as a means to ensure low adverse effects. This monitoring is important because debris build-up might alter contact and impingement rates. However, Appendix 5.B: Entrainment stated:

"Because of the lack of an established relationship between passage time, screen contact rate and injury or mortality, it is not possible to conclude with certainty what the effects of the north Delta intakes may be on juvenile Chinook salmon or indeed on juvenile steelhead…".

Therefore, information presented in Chapter 5 on injuries related to the north delta intakes was inconsistent with information presented in the supporting Appendix. This inconsistency needs to be corrected.

Predation at north delta intakes. The Effects Analysis presents some findings that indicate mortality of salmonids associated with predation is uncertain at the north delta intakes and that monitoring and adaptive management would address this issue. The use of monitoring and adaptive management to address the predation issue is important, and implementation of these activities is key to minimizing predation risk. The Panel believes that there is uncertainty about the extent to which juvenile salmon and predators will aggregate near the intakes.

One of the predation analyses relied upon information collected in relation to salmon losses at the Glenn Colusa diversion and screen. Application of the Glenn Colusa analysis to the north delta intake suggested a cumulative loss of 12% of the juvenile winter-run Chinook salmon at the north Delta intake, a value that is high for a relatively short reach of river. Relatively few details about the Glenn Colusa predation study were presented in Chapter 5 or in the supporting appendix (5F: Biological stressors), therefore the Review Panel cannot directly address the question above on this issue. Nevertheless, the Glenn Colusa study seems to indicate that predators may aggregate near fish screens and consume many salmonids. The study at Glenn Colusa highlights the need to monitor fish predation at the north Delta intakes.

Positive barrier fish screens are widely used throughout the Pacific Northwest to protect juvenile salmonids from entrainment into water diversions, and fish screening criteria are widely applied. The BDCP team could access relevant documents on the web. However, regarding predation at the north intake, salmon and predator behavior in response to flow and habitat conditions along the screen intakes will likely be the key determinants of salmon mortality at the intakes. This information must be gathered during project implementation.

10. Does the Effects Analysis provide a complete and reasonable interpretation of the results of physical models as they relate to upstream spawning and rearing habitat conditions, particularly upstream water temperatures and flows resulting from proposed BDCP operations?

Summary

A valid approach was used to calculate daily flow and daily temperatures in the upstream locations. However, the presentation of the temperature results and the synthesis of the results should be improved to aid understanding. The Fish Agencies should also refine the types of analysis they need to best show the temperature impact on fish as the result of BDCP actions. Currently, the temperature analysis includes: 1) a comparison of *mean monthly* temperatures categorized by water year type, 2) exceedances of water temperature thresholds for the different fish species calculated for each month and categorized by water year type, and 3) the number of years where the exceedance occurred categorized by the level of concern (Table 5C.4-4, pgs. 5C4-19, example Table 5C.5.2-42, pgs. 5C5.2-79).

Recommendations

- Question 10 is one of the topics in the Effects Analysis where the data is presented in individual species and life stage sections. It is very hard to synthesize the results in this format.
- To help the reader understand what locations, which species, what life stages are most likely to be impacted by temperature as a result of upstream reservoir operations in response to north Delta diversion requirements, a synthesis section in the main Effect Analysis Chapter 5 should be included. This synthesis should address the summary of the problem presented in Section 5C.4 (5C.4-16 lines 26-32).
- Most charts in this section are hard to visually synthesize the temperature data. Color coding the charts would help guide the reader. Table 5C.5.2-197 (pg. 5C.5.2-364) is a good example of how to improve chart readability.
- Table 5C.5.2-32 (p. 5.C.5.2-79) show compares the level of exceedance for the different scenarios. This table is not effective at communicating that the level of exceedance is shifting between different categories. For example, less “orange” classifications may mean that there are more “red” classifications. It would be helpful to re-visit how this information is presented.
- Another potential key statistic that could be extracted from the model data is the number of *consecutive* days in which water temperature is greater than the threshold level.

Comments

Approach to calculating upstream flows and water temperatures:

The CALSIM II watershed model was used to specify the monthly flows in each of the upstream rivers. These monthly results were then “downscaled” to daily values based on the historical records at three historical locations in the watershed. These flows are used as inputs into the Sacramento River Water Quality Model (SRWQM) or the Reclamation Temperature model, depending on the location. This downscaling approach seems to be reasonable approach to estimate flows. The temperature models used are specific to this region and have been used in other applications.

The temperature analysis included: 1) a comparison of *mean* monthly temperatures categorized by water year type; 2) exceedances of water temperature thresholds for the different fish species calculated for each month and categorized by water year type; and, 3) the number of years where the exceedance occurred categorized by the level of concern (Table 5C.4-4, pgs. 5C4-19, example Table 5C.5.2-42, pgs. 5C5.2-79).

Analysis and synthesis of the Temperature modeling:

Question 10 is one of the topics in the Effects Analysis where the way the data is presented makes it very hard to synthesize the results. The topic of temperature was evaluated in the Upstream Habitat Results Section 5C.5.2 (548 pages long) for each species and life stage. In many cases the description of the results were very repetitive and did not explain how the results differed from other species.

To help the reader understand what locations, which species, what life stages are most likely to be impacted by temperature as a result of upstream reservoir operations in response to north Delta diversion requirements, a synthesis section in the main Effect Analysis Chapter 5 should be included. The current summary of upstream temperature (Table 5.3-5, p. 5.3-21) is too general to be useful. It is not a sufficient synthesis of the information contained in Section 5C.5.2. This synthesis should address the summary of the problem presented in Section 5C.4 (5C.4-16 lines 26-32).

11. Does the Effects Analysis use a reasonable method for “normalizing” results from the salvage-density method to the population level for salmonid species?

Summary

The normalization approach seems to simply adjust entrainment values based on relative population size over the years of observation so that entrainment values <u>relative to water export</u> may be more comparable from year to year. The normalization should be used for qualitative purposes but not for modeling purposes, because it will mask some of the variation and uncertainty. This standardization has utility for the purpose of calculating entrainment per volume of exported water, but it provides only a partial view of the pumping effect on fish populations. The percent of the populations entrained is more important. This value has more relevance to Effects Analysis on the population. It also appears the variance calculations for salvage abundance and entrainment index are being calculated incorrectly.

Recommendations

- Calculation of salvage density and entrainment need to be revisited and the variance calculations corrected. Current variance calculations for salvage density are underestimating actual variance and uncertainty.

Comments

The salvage-density method was developed to provide an index to entrainment that reflects the volume of export, taking into account fish species abundance. The method assumes a linear relationship between entrainment and export flows. There is some evidence this assumption of linearity may not be correct over the total range of conditions (Kimmerer 2008).

An estimate of total salvage abundance (S_i) for year i is estimated by the product

$$\hat{S}_i = \widehat{D}_i \cdot V_i$$

where

$\widehat{D}_i$ = estimate of fish salvages per volume of water export,

V_i = volume of water export.

The estimate of salvage loss is then "normalized" for an average population size of the fish according to the formula

$$\tilde{S}_i = \left(\frac{S_i}{N_i}\right)\overline{N}$$

where

N_i = fish abundance for the ith year,

$\overline{N}$ = average fish abundance over the years of inference.

Ideally, the fish abundance values should be based on the same population as the fish being salvaged. For example, winter-run Chinook where normalization is based on juvenile production estimates. In the case of fall and late fall-run and spring-run Chinook salmon, the normalization is based on adult run size and in the case of longfin smelt, a trawl index. For each of these latter cases, there is the additional assumption that juvenile abundance is proportional to either adult abundance or the trawl index, i.e.,

$$N_i = cA_i \;\; V_i$$

or

$$N_i = cT_i \;\; V_i$$

where

A_i = adult abundance in year i,

T_i = trawl index in year i, and

V_i = water volume in year i.

The normalized values, $\tilde{S}_i$, can be used in indices of annual salvage numbers but should not be used in subsequent simulations or the calculations of interval estimates. The normalization process has dampened the variability among annual values such that any subsequent variance calculations will underestimate the actual magnitude of the uncertainty (i.e., confidence interval [CI] width).

The entrainment index (E_i) is calculated

$$E_i = \frac{\hat{S}_i}{V_i}$$

per Section 5.B.5.4.3. It is unclear whether the actual salvage abundance $\left(\hat{S}_i\right)$ estimate or the normalized value $\left(\tilde{S}_i\right)$ is used in these calculations.

The variance calculations for the entrainment index (Section 5.B.5.4.3, lines 8–17) appear to be wrong. Based on the description, the average index value is calculated by taking the entrainment density for all relevant water years $(D_i, i = 1, \cdots, n)$ multiplying these values by alternative water volumes from CALSIM $(V_j, j = 1, \cdots, m)$, then averaging over all nm. The variance is based on the empirical variance using the nm values, i.e.,

$$\widehat{\text{Var}}\left(\hat{\hat{S}}\right) = \frac{s_{S_{ij}}^2}{nm},$$

per the plan, and where the S_{ij} are all possible values over n and m, then

$$E\left(\frac{s_{S_{ij}}^2}{nm}\right) = \frac{\bar{V}^2\sigma_D^2}{nm} + \frac{\bar{D}^2\sigma_V^2}{nm} + \frac{\sigma_V^2\sigma_D^2}{nm}.$$

However, based on the stratified nature of the calculations, the correct variance has the form

$$\text{Var}\left(\hat{\hat{S}}\right) = \frac{\bar{V}^2\sigma_D^2}{n} + \frac{\bar{D}^2\sigma_V^2}{nm} + \frac{\sigma_V^2\sigma_D^2}{nm}$$

where

$\bar{V}$ = average water volume,

σ_V^2 = variance in water volume values,

$\bar{D}$ = average density,

σ_D^2 = variance in density values.

The report variance is too small.

The variance of the total salvage estimate also appears to be wrong (pages 5.B-65 and 66). The calculation of total salvage (S) was based on the description to be:

$$S = \widehat{\text{density}} \cdot \text{Volume}$$

where the estimator of density was based on a linear regression of log salvage density vs. day of inundations. The report then states that the confidence intervals were then computed using the 95% confidence levels of the estimates of the regression." This calculation, as described, is wrong. The calculations should be based on the variance estimate for the back-transformed estimate of density from the regression, i.e.,

$$\text{Var}(\hat{S}) = \text{Var}\left(\widehat{\text{density}} \cdot \text{Volume}\right)$$

$$= \text{Volume}^2 \, \text{Var}\left(e^{\hat{y}}\right)$$

$$\doteq \text{Volume}^2 \, \text{Var}\,(\hat{y})\left(e^{\hat{y}}\right)^2$$

where $y = \ln(\text{density}) = \alpha + \beta x$.

See Appendix D for appropriate variance calculations for the salvage model.

12. Are the assumptions of the analysis of aquatic habitat restoration food web effects appropriate for covered fish species? Are the conclusions and net effects appropriate?

Summary

The BDCP develops a robust conceptual model of aquatic food webs and the diverse linkages that may impact the net production of food for Covered Fish. Yet the BDCP contains a number of assumptions, some of which are inappropriate, others of which contain considerable uncertainty. Uncertainties are mentioned, but no effort was made to include whether conservation efforts reach only a portion of the goals of biological objectives. Thus the analysis of effects further assumes only the most beneficial potential results in any calculations, but doesn't incorporate other possibilities. Other processes of food webs in aquatic habitats are described but remain unanalyzed, some of which may enhance, while others of which would inhibit their biological objectives. While the overall conceptual model is adequate, integration and synthesis is lacking. Consequently the conclusions and net effects are not appropriate given the gaps in analyses and the uncertainties.

Recommendations

- Model the potential flow of energy through the pelagic food web – baseline information
- Assume a variety of primary production flows to covered species due to competitors or environmental issues – to what extent might their optimistic scenarios vary from equally potential realities
- Assume shifts in composition of plankton from favorable to unfavorable species (with respect to covered species) – even with potentially higher productivity by plankton, what happens if energy flows into other pathways other than nearly immediately into the covered species
- Incorporate a detrital energy flow – this might shift energy flow back toward covered species
- The direction of restoration in these systems that would support phytoplankton is not simple and linear, adaptive management would need to be an aggressive component of the BDCP with authority to take immediate actions, regardless of what those might be

Comments

The conceptual model of the food web appears to contain all the significant compartments required for an adequate assessment of the impact of the BDCP. The BDCP contains a number of conservation efforts that have the potential to provide considerable enhancement of the populations of covered fish. These include increasing habitat, providing a diversity of habitat conditions that may enhance different life history stages, as well as allowing for potential increases in food web services for covered species. However, other than estimates made for phytoplankton production, no other assessments are made. First we review some of the assumptions inherent in the BDCP consideration of food webs.

An overarching assumption is that Conservation Measures have rapid and positive impacts. With respect to food webs, wetland and aquatic systems restoration are assumed to be effectively restored and functional immediately or in a short time frame and meet the biological objectives of the BDCP. This result is based on a number of additional assumptions, all of which contain considerable uncertainty. Similarly, while potentially negative impacts on the success of restoration are considered in passing, e.g., invasive bivalves, none of their potential effects are incorporated into their analyses or conclusions. The simplest effects perspective of the BDCP is that it edits out all potential outcomes except for the most favorable one.

Restoration of natural ecosystems, however, is difficult and fraught with great uncertainties and some systems that are assumed to have a positive influence on covered species are particularly difficult. The contingency of ecological communities means they will not automatically assemble in some predictable manner (Parker 1997). Chapter 5 contains even less information this time concerning details about timing and sequencing required to evaluate potential impacts. Understanding the sequences is also critical because they have major influences (Drake 1990, 1991; Hobbs and Cramer 2008). For example, the BDCP implies a consistent increase in restoration acreage through time, but without strong management intervention prior to opening of new wetland or shallow aquatic habitat, submerged aquatic invasive species such as bivalves, *Egeria*, or other newly detected species may expand rapidly into the new tidal habitat. The result would be a much larger management problem without the food web benefits proposed by the BDCP.

The assumption of rapid positive food web benefits from restoration of aquatic habitat is a potential benefit, but the degree of benefit, its timing, and even whether benefits will accrue is uncertain. Restoration even may be on a pathway to achieving desired biological objectives, but the time frame may be considerable and beyond the 50-year period of the BDCP. Similarly, changing the order of different conservation measures may push ecological systems onto different trajectories. Usually these cannot be predicted, and requires an integrated monitoring and adaptive management with considerable authority and manpower. Restoration rarely achieves immediate conservation or biodiversity goals (Hobbs and Cramer 2008, Hobbs *et al.* 2011). While tidal water as a process can be achieved by opening dikes, restoration of biological function is actually quite difficult with respect to ecosystem processes beyond tidal flux and especially with respect to ecological equivalency to comparable natural wetlands (Kentula 1996; Simenstad and Thom 1996; Zedler and Callaway 1999; Lockwood and Pimm 1999). More recent studies substantiate these evaluations (Burgin 2008; BenDoer *et al.* 2009; Moilanen *et al.* 2009).

The BDCP further ignores critical data that should have been incorporated into trajectories concerning the restoration of wetland and associated aquatic habitat. This is a crucial piece because the restoration that is planned is critical key to increasing suitable habitat and food web productivity. The issue is sediment supply for these restorations. The BDCP assumes a constant sediment concentration for the time period of the plan (Appendix 5.E, pp. 43-44: turbidity held constant in models and interpretations), yet they indicate that sediment concentration has been declining over the past 50 years (p. 109) and that the BDCP conservation measures will further reduce the sediment supply by an additional 8-9%. While in their discussion of sediment supply, they also conclude that declining sediment concentration and the impact of CM1 will mean much lower sediment supply, these issues have no impact on the BDCP analysis and inference. Yet the loss of sediment supply creates great uncertainties in the rate and potential for restoration of these habitats, while only the most optimal circumstances are modeled or estimated.

Similarly, the BDCP uses a simple depth-productivity model to quantify how habitat restoration may impact primary production (Figure 5.E.4-85, Relationship between Phytoplankton Growth Rate and Depth, in Appendix 5.E, Habitat Restoration). This assumes the relationship between phytoplankton growth rate and depth developed by

Lopez *et al.* (2006) is accurate. The analysis focused solely on the relationship between phytoplankton and depth, while recognizing that other factors may influence phytoplankton production in particular locations (p. 121).

Ironically, the literature they rely on, Lopez *et al.* (2006) and Lucas and Johnson (2012), indicate that biomass and production of phytoplankton in the Delta do not fit this simple model expectations. A major limitation of the depth-productivity model is the impact bivalve grazing on available net production. Net phytoplankton production (in excess of potential grazing) peaked at different depths and at much lower rates depending on overall habitat depth and water residence time. Assumptions of phytoplankton production and their conversion to zooplankton and invertebrates as food sources for covered species in aquatic systems consequently lack realism.

A third assumption involves the production of food for covered fish. Food produced in the restoration areas is assumed to directly benefit covered fish and indirectly by export. The restoration of these areas are predicted to create better habitat and food for juvenile Chinook salmon, splittail, sturgeon, delta smelt, and longfin smelt. Two issues arise from this assumption, one is their analysis of phytoplankton production and the second is that the analysis never includes potential competitors.

In contrast to their assumption, they cite literature that models the impact of introduced clams and their rate of filtering of phytoplankton and other aquatic organisms. These models suggest 1) that the depth-productivity model they used is completely inaccurate in the context of invasive clams and 2) remind us that while the potential impact of clams are mentioned as an uncertainty, only the most optimal scenario without clams is used for conclusions about the short and long-term benefits of the BDCP.

Beyond the analysis of assumptions, the other compartments of the food web are not incorporated into their analyses. For example, the potential for detritus as a major source of food web production was reviewed at some point and mentioned during the discussion of food webs. However, no incorporation or estimation of potential detritus production was made, nor was the detrital web discussed any further. Ironically, this could be a significant and positive impact on covered species.

Similarly, the role of SAV and emergent vegetation were not assessed although they were mentioned. The issue of competitors was not assessed. No incorporation was made of anthropogenic nitrogen influences on phytoplankton community composition (for example increasing the proportion of *Microcystis*). While the BDCP generally has a review of most of these compartments that they illustrate in the conceptual model, no quantitative models, nor estimates derived from the literature review were developed to allow a variety of scenarios that might indicate the potential robustness of the impacts of the conservation measures. Thus, some quantitative detail on one or a few compartments, complete with large tables showing all the numbers produced, lacks significant meaning when other compartments are merely discussed. The overall impression is that these compartments live in conceptual isolation, lacking the integration of multiple and linked processes/interactions together into a synthesis. Consequently the BDCP analyses are ambiguous and conclusions and estimates of net effects overestimate the net positive impacts of conservation measures.

13. Is the analysis of food web benefits to longfin smelt from habitat restoration appropriate? How well do the analyses link intended food web improvements to improvement in the longfin smelt spring Delta outflow/recruitment relationship?

Summary

While the Effects Analysis develops an appropriate logic train suggesting that restoration actions (e.g., CM4) would result in the production and export of increased longfin smelt "food", this objective is constrained by considerable uncertainty (acknowledged as only "Partial" assessment) because the data is lacking to quantitatively estimate the relationship between longfin smelt production and what might be exported from tidal wetland restoration and converted to food web linkages to the smelt. Although there are good, synthetic conceptual models developed for the Bay-Delta longfin smelt population encapsulated in the Effects Analysis (e.g., Baxter *et al.* 2010; Rosenfield 2010), this uncertainty is further constrained by the lack of a life-history model that would elucidate the role of prey composition and abundance in population dynamics. Delta smelt are principally planktivorous, feeding on copepods, cladocerans and mysids in the Bay-Delta (Moyle 2002; Feyrer *et al.* 2003; Hobbs *et al.* 2006). A potentially significant change in the viability of food web support of longfin smelt by the shift from the native *Eurytemora affinis* to non-indigenous species such as *Pseudodiaptomus forbesi* and *Sinocalanus doerri* is implicated in declining availability of natural prey for longfin smelt. However, these changes were also confounded by flow diversions and restriction of the mixing zone and potential increased entrainment into water diversions and the increased predation of the overbite clam *Potomocorbula amurensis* on mysids and other zooplankton prey after its introduction in 1986 (Alpine and Cloern 1992; Kimmerer 2002).

Recommendations

- Strengthen the documented data and other evidence supporting the presumption that export of detrital matter would specifically contribute to food web linkages supporting longfin smelt.

Comments

While there is viable evidence that poor survival and growth are a major cause of longfin smelt decline (Bennett and Moyle 1996; Sommer *et al.* 2007), the mechanism and magnitude of increased production of desired longfin smelt prey contributed by restoring tidal natural communities and other proposed BDCP restoration actions is still highly uncertain (see response, above, to Question 12). As discussed elsewhere, the contribution of restoring shallow water tidal wetlands to net phytoplankton production and increased plankton abundance available to longfin smelt is basically hypothetical because of the uncertainties of primary consumption within the restoring ecosystems, especially by non-indigenous clams, and whether these systems would be sources or sinks for any increased production. The Effects Analysis does acknowledge that tidal wetland restoration is also likely to export detrital organic matter, as well as macroinvertebrates, but the potential contribution of these food web sources to longfin smelt production is equally uncertain without more explicit and quantitative linkages to the longfin smelt prey potentially involved, such as mysids.

From that standpoint of linking food web benefits to the longfin smelt spring Delta outflow/recruitment relationship, the Effect Analysis does provide a reasonable rationale for smelt post-larvae and juveniles to benefit from exported production from the Suisun Marsh ROA, albeit with the same uncertainty associated with the utility of that exported production. Current understanding of juvenile longfin smelt occupancy of the Suisun Bay and West Delta subregions during March through June, before moving further into San Francisco Bay proper, suggests that linking the outflow/recruitment relationship to the management of spring (March-May) Delta outflow (Chap. 2, Section 2.4.1.4.4 Decision Trees) could be a management strategy.

14. How well does the analysis address population-level effects of the BDCP on white sturgeon?

Summary

The analysis does an excellent job of summarizing what is currently known about the life history and ecology of white sturgeon (southern distinct population segment) using the most recent analyses and peer-reviewed publications. In addition, the conclusions regarding the level of certainty about the effects of the different conservation measures on white sturgeon, based the expert panel convened in August 2013, are thoroughly discussed in the text and well summarized in Figure 5.5.8-2.

Estimating the effects of the BDCP on white sturgeon population levels is very difficult because of: 1) the lack of a thorough understanding of the effects of flow regimes on downstream migration and year class recruitment; 2) considerable uncertainty about white sturgeon sensitivity to water quality and whether current water quality conditions constitute negative impacts; (3) a poor understanding of the role of intertidal and subtidal habitat on food availability for migrating juveniles; and 4) little information about factors influencing growth and survival of adults in San Francisco Bay and the ocean. Given these limitations, the Effects Analysis does an adequate job of using existing information to predict the effect of the various conservation measures on white sturgeon.

Recommendations

- Implement measures to improve estimates (reduce uncertainty) of adult survival and population size of white sturgeon in the Delta.
- Undertake research studies to identify the reason(s) for the observed association between high flows and high recruitment.
- Initiate studies to understand the links (or lack thereof) between water quality and intertidal and subtidal habitat on growth and survival of 1) migrating juveniles and 2) adults.

Comments

The life history of white sturgeon, high adult survival and fecundity in combination with episodic recruitment in high water years, suggests that the multiple approach to conservation measures should promote increased adult survival and ensuring high recruitment when conditions are favorable. We agree with the conclusions of the Effects Analysis that reduction of illegal harvest (CM 17) and reduction of entrainment at the Fremont weir (CM 2) are both highly likely to have a positive effect on adult survival. Similarly, we agree that the restoration of tidal wetlands under CM4 are very likely to provide significantly increased rearing habitat and epibenthic and benthic food resources. Perhaps more than the pelagic covered species, white sturgeon could also derive significant benefits from enhanced and exported detrital organic matter from tidal wetland restoration because much, if not most, of their natural (and unnatural given the non-indigenous clams contributions to their diets) prey on mudflats and in adjacent channels are detritivores.

Quantitatively estimating the effects of these conservation measures on adult survival will require more rigorous, focused sampling efforts. The large confidence intervals associated with recent estimates of adult survival will make it nearly impossible to document effects of the conservation measures. The effects of water diversion and changes in flow regimes on white sturgeon recruitment are much more difficult to predict and will require a more thorough understanding of the mechanisms behind the correlation between recruitment and flow volume.

Adequacy of Technical Appendices

Appendix 5.B—Entrainment

Summary

Section 5.B.4.1 (p. 5.B-11 lines 18-23) has the most important statement of the entire appendix. This conclusion that should be the first conclusion in the executive summary:

“Under the ESO (Evaluated Starting Operations), in the wetter water years (wet and above-normal water years…), most of the combined total exports would come from the new north Delta facility and exports from the south Delta facility would be lower than existing biological conditions … The use of the north Delta pumps would be lower in the dryer years with most pumping going from the south Delta pumps in dry and critical water year… Less use of the north Delta pumps in drier water years reflects requirements to maintain adequate bypass flows at the north Delta diversions.” (5.B-11, lines 18-23)

This conclusion is the basis of most of the entrainment analysis in Appendix 5.B for the South Delta facilities. There may be different approaches to come up with the regression between export rate and salvage, but the simplistic conclusion is that when the pump operations are lower, so is the entrainment of fish. However, in the dry and critical years, entrainment at the South Delta facilities will be higher because the north Delta facilities’ operations will be limited.

The next question to ask, therefore, is how often we will be under dry or critical year conditions. Will California have more frequent dry water years, resulting in fewer times when the north Delta diversion facilities can be operated?

Recommendations

- The conclusion stated above in the summary Section 5.B.4.1 (p. 5.B-11 lines 18-23) should be the first conclusion in the Appendix 5.B executive summary and should be included in Chapter 5.

- The Climate Change (Appendix 5.A) portion of the Effects Analysis needs to address the question for frequency of dry/critical water years and relate it back Appendix 5B.
- The documentation of the DSM2 and particle tracking model (PTM) model in this appendix should be greatly expanded to provide clarity in their approach.

Comments

Section 5.B.4.1 (p. 5.B-11 lines 18-23) has the most important statement of the entire appendix. This conclusion that should be the first conclusion in the executive summary:

"Under the ESO (Evaluated Starting Operations), in the wetter water years (wet and above-normal water years…), most of the combined total exports would come from the new north Delta facility and exports from the south Delta facility would be lower than existing biological conditions … The use of the north Delta pumps would be lower in the dryer years with most pumping going from the south Delta pumps in dry and critical water year… Less use of the north Delta pumps in drier water years reflects requirements to maintain adequate bypass flows at the north Delta diversions." (p. 5.B-11, lines 18-23)

This conclusion is the basis of most of the entrainment analysis in Appendix 5.B for the South Delta facilities. There may be different approaches to come up with the regression between export rate and salvage, but the simplistic conclusion is that when the pump operations are lower, so is the entrainment of fish. However, in the dry and critical years, entrainment at the South Delta facilities will be higher because the north Delta facilities operation will be limited.

The next question to ask, therefore, is how often we will be under dry or critical year conditions. Are we going to have more frequent drier water years, resulting in fewer times when the north Delta diversion facilities can be operated? The Climate Change (Appendix 5.A) portion of the Effects Analysis needs to address this question and relate it back to this Appendix.

In this appendix, the first conclusion stated is: "The BDCP would substantially change the amount and pattern of water exports from the south Delta SWP/CVP facilities, which generally would be expected to lower the number of fish of all species entrained relative to existing biological conditions." (Appendix 5.B, p. 5.B-iii, lines 38-40)

We agree that the south Delta export patterns will change substantially, especially in wet and above normal years. However, it is also important to look at how the flow patterns will also change in the north Delta. This is an equally important piece of evaluation that should be included in the entrainment analysis. The use of the DSM2 PTM is a first attempt at this type of analysis. However, the documentation of the DSM2 PTM model in this appendix should be greatly expanded to provide clarity in their approach. Some of this documentation may already be in Appendix 5.C, however, the present documentation is not sufficient to allow Appendix 5.B to act as a stand-alone document.

Appendix 5.C—Flow, Passage, Salinity, and Turbidity

Summary

Appendix 5.C has been a catch-all appendix ever since Phase 1 of this Effects Analysis review. Unlike the Entrainment or Contaminants appendices, this appendix does not have an individual issue that it is trying to address. This appendix is 2,636 pages long and spans a laundry list of topics including flows in river, salmon migration through the Delta, Delta Cross Channel and Georgiana Slough circulation, non-physical barriers, temperature modeling, water clarity, turbidity, invasive species, nutrients, dissolved oxygen, and algae. This appendix should have been divided into multiple appendices in previous iterations of the BDCP document. At this point, the division of the appendix will likely never happen. As a result, this is a very difficult appendix to review. In general, the Panel read through portions of this appendix to answer specific questions for the main charge questions for Chapter 5.

Recommendations

- Most Appendix 5.C recommendations are included in the Chapter 5 questions.
- Guiding operational rules in place for the current configuration of the Delta, such as E/I ratios, need to be reviewed to see if they still make sense for the combined system.
- The calculation of transport time scales should be done with relation to a particular question being addressed rather than calculated as a bulk parameter.
- Improve the synthesis of results in Section 5C.5.3.1: Passage, Movement, and Migration Results, Flow Summary.
- Water clarity and suspended sediment should have been in an appendix all its own rather than being buried in Part 6 of Appendix 5.C.

Comments

Baseline operations (Section 5C.2.2)

The Effects Analysis used two different baseline conditions, one that was consistent with the USGFWS BiOp RPA actions (EBC2) and one in which the USFWS RPA (Fall X2 action) was not included (EBC1). The panel will not comment the details of the baseline operations that were used to represent current conditions because this level of detail is beyond the area of expertise of the panel. We defer this issue to public comments by interested stakeholders, state and federal agency personnel that have more understanding of these details.

Proposed operations, Maximum Allowable Export Rules (Section5C.2.2.2.1)

Before the north Delta diversion facility is operational, the operating criteria for both the North and South facilities need to be established. Guiding operational rules in place for the current configuration of the Delta, such as E/I ratios, need to be reviewed to see if they still make sense for the combined system. For instance:

"For the BDCP cases, the [Export/Import] E/I ratio was assumed to apply only to south Delta exports; the north Delta intake diversions were assumed to exempt form E/I rule because the north Delta diversions are controlled by the bypass flow rules. The south Delta pumping was limited by the E/I calculated with the inflow minus the north Delta diversions; this would allow slightly higher total exports during periods when Sacramento River flows are high and north Delta diversion are high.” (p. 5C.2-3, lines 41-42; p. 5C.2-4 lines 1-3)

Residence Time (Section 5C.4.4.7)

The residence times calculated using 38 particle release sites using the DSM2 PTM model is of limited use. The calculation of transport time scales should be done with relation to a particular question being addressed. For example, how long will water reside in a specific Restoration Opportunity Area and how does that transport timescale compare to other important timescales, such as phytoplankton growth rates, contaminant reaction time, etc.

The Delta is a very diverse mosaic of regions. Each sub-section of the Delta has unique characteristics. Transport timescales in each sub-region is a function of operations (such as the operation of the Delta Cross Channel and the placement of temporary barriers, flooding in the Yolo Bypass), bathymetry, and connectivity to adjacent regions. Transport timescales calculated in sub-regions rather than full Delta “average” residence time will give much more detailed information about changes in circulation patterns as a result of alterations to the system as a result changes in operations and additions of restoration opportunity areas.

Passage, Movement, and Migration Results, Flow Summary (Section 5C.5.3.1, Pages 5C.5.3-1 through 5C.5.3-64)

Please improve the synthesis of results in this section. These pages contain only charts with no dialogue or graphs to aid the reader. This section likely contains very important information about how the circulation changes in the Delta will change as a result of the Conservation Measures at key locations throughout the Delta.

Attachment 5C.D (Water Clarity-Suspended Sediment Concentration and Turbidity) (5C.D-1 through 5C.D-64)

Water clarity and suspended sediment should have been in an appendix all its own rather than being buried in Part 6 of Appendix 5.C. This is a topic is as important as Entrainment and Contaminants. This section is a good resource to read for background on issues related to sediment transport in the Delta.

Appendix 5.D—Contaminants

Summary

Currently, the contaminants section of Chapter 5 comprises 1 ½ pages of a 745 page document with most of the information related to contaminant effects contained in a single table. There are many caveats to consider with contaminants and this topic should get more attention within Chapter 5. Appendix 5D has a very well written introduction that lays out the key issues related to both mercury and selenium in the Delta. This introduction should be included in Chapter 5 where it will be read and considered. This list of potential contaminants seems reasonable and the conceptual model for contaminants (Fig 5D.3-1) is well developed. The growing list of contaminants of emerging concern is a clear sign that additional contaminants may need consideration in the future.

The Executive Summary of Appendix 5.D (p. 5.D-i, lines 24 -29) states that quantitative analyses were applied where available but were not sufficient to fully examine the potential for contaminant effects. This statement is important for characterizing the level for which potential contaminant effects can be assessed, however this is not part of the bulleted summary within the Executive Summary (p. 5.D.ii, lines 35-42).

The Contaminants Appendix is limited to direct contaminant effects on covered species even though it is recognized that both direct and indirect contaminant effects must be considered (p. 5.2.3, lines 5-7). The Effects Analysis authors indicate that indirect contaminant effects are handled within Appendix 5.F: Biological Stressors on Covered Fish. Given the degree to which indirect contaminant effects are presently covered in Appendix 5.F this is not satisfactory. A Phase II Panel recommendation was to incorporate grey literature where needed in the contaminants section, especially for indirect contaminant effects. These recommendations were not taken and stand from the original review.

The separation of direct and indirect contaminant effects lead to strange splits in organization, including for *Microcystis* which is included as a “contaminant” in the contaminant conceptual model but is not part of the discussion in Appendix 5.D: Contaminants. Rather, *Microcystis* is considered in Appendix 5.F.

Both Conservation Measure 15: Methylmercury Management (pp. 4-257) and AMM27 Selenium Management (p. 5.D-37, line 18) should be evaluated by contaminants experts to determine if these approaches will be acceptable for mitigation. The modeling of Methylmercury effects are highly uncertain due in large part to site-specific characteristics that cannot be modeled at present.

Recommendations

- Provide more information with Chapter 5: Effects Analysis rather than relying heavily on Appendix 5.D: Contaminants.
- Include both indirect and direct contaminant effects within Contaminants Appendix (Phase II recommendation).
- Methylmercury Management and Selenium Management should be evaluated by contaminants experts.
- Incorporate grey literature where needed (especially herbicide application for control of Invasive Aquatic Species).
- Provide clear statements within Chapter 5 and the Executive Summary of Appendix 5.D about the high level of uncertainty associated with contaminant effects as a result of site-specific details that cannot be modeled without explicit information about the location and connectivity of ROAs.

Comments

The Contaminants Appendix is limited to direct effects of contaminants on covered species despite the recognition (Chap. 5, pg. 5.2-3, lines 5-7) that that both direct and indirect contaminant effects must be considered. Appendix 5.D states that with the exception of herbicides used to control Aquatic Vegetation, the BDCP does not add any contaminants to the Plan Area. Nonetheless, as stated (Chapter 5, page 5.3-26, lines 29-30) BDCP activities will alter freshwater flow and alter water residence times at various locations in the Delta. These changes can result in major changes in how contaminants interact with the Delta ecosystem by changing the local concentration of a given contaminant or duration of exposure. For these reasons, restricting the analysis to direct effects on covered species is inadequate.

The inherent challenges in navigating a document of this size could be overcome by placing all of the contaminant effects under the Appendix entitled "Contaminants". This was a recommendation made during the Phase 2 review. Indirect effects are handled elsewhere in the Effects Analysis (Appendix 5.F: Biological Stressors on Covered Fish) however at present discussion of potential indirect contaminant effects are not sufficient in scope, detail, or characterization of uncertainty. Ammonia (NH_3) / ammonium (NH_4) effects, as written in Appendix 5.D, appear to consider indirect effects of ammonia/ium which is inconsistent with the authors' intent for Appendix 5.D.

Appendix 5.D has a very well written introduction that lays out the key issues related to both mercury and selenium in the Delta. The analysis was very careful to separate out the effects of Conservation Measure 1 (north Delta diversion facilities) from the effects of Conservation Measure 2 (Establishment of ROAs). In general, the environmental effects related to constructing ROAs are a bigger concern for contaminants than the north Delta diversion. However, in the case of selenium, changing the pumping operation location in conjunction with the establishment of ROAs in the South Delta has a potential significant effect. Changing to the north Delta diversions shifts the primary source of water in the South Delta to San Joaquin derived water rather than Sacramento source water under certain conditions.

It is recognized that Methylmercury concentrations would continue to exceed criteria under the BDCP and restoration actions are likely to increase production, mobilization and bioavailability of Methylmercury (5.D-24, lines 41-44). There is considerable uncertainty related to Methylmercury production resulting from BDCP activities. This is due in large part to site-specific information needed to construct reasonable models and trophic interactions from various sources are not easily modeled (5.D-22, lines 11-17)

DSM2 is a one-dimensional model that represents open water areas as well-mixed, continuously stirred tank reactors. In addition, the location of the ROAs and how these areas are connected to the adjacent channels is unknown.

Currently, dissolved Se in the San Joaquin is an order of magnitude higher than in the Sacramento River. (Monsen *et al.* 2007) Therefore, even if the proportion of San Joaquin discharge relative to the Sacramento River is low, the increase in Se concentration could still be significant. This conclusion should be reviewed. There is much uncertainty in the DSM2 results, especially for residence times in the newly established open water regions.

Section 5.D.43 (lines 8-10) on the impact of restoration on ammonium suggest that restoration will not have an impact on NH_4 concentrations – This is overly simplistic as tidal wetlands are known to be important in nitrogen biogeochemistry, acting as a source via sediment re-mineralization (Cornwell *et al.* 2014) or clam excretion (Kleckner 2009) or as a sink via organic matter production or coupled nitrification – denitrifcation (Cornwell *et al.* 2014).

Conservation Measure 13: Invasive Aquatic Vegetation Control is discussed in Section 5.F-6. There is little consideration of the potential effects on lower trophic levels (algal primary producer) due to herbicide applications. This issue is raised in a single bullet on page 5.F-130 Line 24-25. While the literature is not well developed for the SFE there is at least some indication that herbicide applications are detrimental to photosynthetic organisms (phytoplankton). This should be addressed as a possible effect with implications for adaptive management.

Appendix 5.F—Biological Stressors on Covered Fish

Summary

Appendix 5.F examines the effects of 10 conservation measures on four key biological stressors: invasive aquatic vegetation (IAV), predation, invasive mollusks, and *Microcystis*. Effects of these actions on fishes was largely based on professional opinion while utilizing available information. While intentions of these actions is good, the outcome for fishes is uncertain, indicating the need to monitor and adapt. Key issues include expansion of invasive clams that consume phytoplankton, more favorable conditions for *Mycrocystis and harmful algal blooms,* and continuous effort needed to control invasive aquatic vegetation and predator abundances.

Recommendations

- Page 5.F-107, last paragraph, first sentence, and Executive Summary: The 1% to 12.8% range in predation effects due to the north Delta diversion is a mixture of population-level and localized effects and should not be treated as measuring the same quantity. That range estimate is deceptive and technically incorrect.
- Monitor progress and maintain efforts to control invasive species than impact covered fishes.

Comments

Biological stressors can result from “competition, herbivory, predation, parasitism, toxins and disease.” The objective of the conservation measures is to reduce the negative effects of key biological stressors on covered fish species. Appendix F examines the effects of 10 conservation measures on four key biological stressors: invasive aquatic vegetation (IAV), predation, invasive mollusks, and *Microcystis*. This review is designed around the four biological stressors and the prospects for change under the BDCP plan.

<u>Invasive Aquatic Vegetation (IAV).</u> The plan states controlling IAV is expected to reduce densities of largemouth bass but could enhance open water conditions favorable to striped bass. The control of IAV should increase turbidity which should be beneficial to foraging by juvenile fish and reduce predation. Brazilian waterweed (*Egeria densa*) and water hyacinth (*Eichhornia crassipes)* are the two most abundant IAV in the Delta. The CM13 proposes to treat approximately 1,700–3,400 acres of *Egeria* per year in and near restored habitat. Currently, *Egeria* is increasing at a rate of approximately 15% per year. Efforts will need to be sustained and focused to be effective.

Assessments of the benefits of IAV control were based on “scientific literature,” consultations with local experts, and conceptual models of key processes, habitat, and covered fish species. There is also practical experience to draw from. At Franks Tract, *Egeria* control was 47% effective (5.F-40), while Delta-wide *Egeria* continues to expand at about 15%/year. Annual treatment of 1500 acres/year would be expected to maintain the status quo.

Figure 5.F.5-3 projects it would take approximately 10 years to eradicate *Egeria* under a high treatment scenario and a 20% annual expansion rate. Some of this benefit may be offset by the fact that habitat restoration under the Plan would also create susceptible *Egeria* habitat. Water hyacinth control, on the other hand, appears to be already successful.

Predation. Predation control is to be locally focused on predator hotspots. Ten spots have been specified, along with the new north Delta water diversion facilities and nonphysical barriers. It is unclear how effective these localized remodels will be because the predators being controlled (i.e., largemouth bass and striped bass) are moderately to highly mobile.

For the north Delta diversion facilities, two approaches were used to estimate predation-related effects: bioenergetics modeling and fixed estimate of 5% predation loss at each of three intakes screens. The Executive Summary states predation losses at north Delta intakes should be from less than 1% to 12.8%. However, this range is contradicted by the simple fixed estimate model: Assuming three intakes each with a 5% independent rate of loss, then the overall rate is $1 – (1 – 0.05)^3 = 0.1426$ or 14.26%. The bioenergetics model was considered the Plan's best approach to assessing predation near the intakes. However, the fourth assumption of this model (p. 5.F-15) states predation was assumed to be proportional to the prey's relative abundance. This is in contrast with most energetics models that assume consumption has a lower threshold dependent on the predator's physiology and size. Predation is then proportional to predator abundance. The analysis also apparently ignores smaller size prey (assumption 6, p. 5.F-16). This analysis was also based on guesstimates of expected predator abundance at the future north Delta intake facilities. The model also assumes all prey are at equal risk, regardless of their location in the channel.

Using the bioenergetics models to express the effects of predation at the north Delta intakes as a percentage of total juvenile predation can be misleading (p. 5.F-75). Localized predation rates are more useful and can be compared to the 5% design specifications. Alternatively, the effect of predation at the intakes could be expressed in terms of proportional change in through-delta survival. Under the fixed predation loss method, it is unclear how proportions of 11.7%, 12.1%, and 12.8% for various fish stocks are estimated (p. 5.F-77) when a simple model based on independent intake events estimates $(1 – (1 – 0.05)^3 \times 100\% = 14.26\%$.

The predator removal program at the north Delta intakes and elsewhere is projected to remove 8,840 striped bass annually. The net effect is a project reduction in 13,320 juvenile salmonids being consumed. The Plan does not estimate the fraction of striped bass removal in the delta (i.e., another measure of relative reduction in predation). The Plan states it is uncertain how long such a removal effort could be sustained, and that predator removal treatments are likely short lived.

The effects of habitat restoration on predator control are uncertain. Effects on turbidity, flow, etc., may be much localized. In addition, it is unclear whether restoration actions will benefit prey, predators, or both.

Invasive Mollusks. The overbite clam (*Potamocorbula amurensis*) currently dominates the brackish transition zone of the delta estuary. Its presence has dramatically altered the zooplankton community. It can filter the entire water column once a day in delta channels. The decline in phytoplankton has been subsequently correlated with declines in copepods and mysid shrimp, a food source of the delta smelt and longfin smelt. The overbite clam has a salinity range of tolerance that could be affected by the Plan's water operations. There is expected to be "generally little difference (25%) in average suitable habitat for the clam between EBC2 scenarios and ESO scenarios . . . ." However, there is risk of *Potamocorbula* expansion:

> "*For ESO without Fall X2 (modeled as ALT1_ELT and ALT1_LLT), the area of suitable abiotic habitat for Potamocorbula would increase 7 to 9% in wet water-year types compared with the EBC1 baseline, but would be little different for all other water-year types. Suitable abiotic habitat for clams would increase in wet and above normal water-year types by about 18 to 28% in early long-term compared with EBC2 baselines (EBC2, EBC2_ELT) and increase 11 to 30% in late long-term*." (Appendix 5.f, page 5.F-117, lines 7-11)

Restoration actions to produce more shallow water habitat may not have a net positive effect. While shallow water habitat produce phytoplankton, the presence of *Corbicula* may result in a phytoplankton sink (p. 5.F-121). One of the few management options is to manipulate salinity which is a function, in part, of river flow. The water withdrawals from the north Delta Diversion should not help the situation. Decision whether to implement the Fall X2 will affect the area of notable colonization by *Potamocorbula*.

Mycrocystis. *Microcystis* blooms can have an adverse effect on phytoplankton, zooplankton, and fish. Factors associated with blooms include high water temperature, high water transparency, low flows, high nutrient concentration, and high nitrogen/phosphorus (N/P) ratios. Runoff from land use contributes to these favorable conditions. *Microcystis* affects fish populations through declines in food sources, mortality, and reduced fecundity. Water operations that reduce flow and increase water residence time may promote *Microcystis*. Shallow water habitat reduction may also promote *Microcystis*. Actions that increase water velocity and turbidity are helpful in controlling *Microcystis* blooms. ESO_ELT and LOS_ELT scenarios are projected to increase average water residence time (Table 5.F.8-2), which would have a detrimental effect in trying to control *Myrcocystis*. Submerged aquatic vegetation (SAV) control may produce water conditions unfavorable to *Microcystis*. Climate warming may be a significant driver in *Microcystis* trends in the future.

Appendix 5.G—Fish Life Cycle Models

Summary

It is not clear to the Panel why life cycle models were not developed specifically for the evaluation of the BDCP. The Panel previously identified a number of expectations for the life cycle model appendix, which had yet to be released. The Panel also recognized that these expectations might not be achieved, and noted that the inability to achieve these expectations would indicate higher uncertainty in the ability of the BDCP to achieve the biological goals and objectives.

Recommendations

- Provide more detailed description of the 14 different scenarios modeled (Table 5.G-2) than shown on p. 5.G-17. For instance, specify what are the low- and high-flow operations specified in scenarios HOS and LOS.
- Check survival estimates. The 94-98% or 96-98% survival values (inconsistent text, p. 5.6-42 and Table 5.G-3) between ocean entry and age 2 seem very high. Rechisky *et al.* (2009), for instance, found early ocean survival of yearling Chinook salmon smolts from the Columbia River to be as low as 0.28 within the first month. Rechisky *et al.* (2012) reported early ocean survival of yearling Chinook salmon smolts to range from 0.04–0.29.
- Clarify what information and how the information from Michel (2010) and Perry *et al.* (2013) were incorporated in the IOS models (page 5.G-44).
- Perform a sensitivity analysis at to generate confidence intervals at the north delta intakes using mortality values at existing structures (Perry 2010) (p. 5.G-46). The 95% survival value used in simulations of the north Delta intake is an engineering specification.
- Consider describing extinction rates. OBAN – Adult Escapement (pp. 5.G-51 to 5.G-61). Examination of plots (Figure 5.G-15, p. 5.G-19) suggests extinction rates for winter-run Chinook salmon would be very high for all long-term (LLT) scenarios and not insignificant for short-term (ELT) scenarios.
- Compare model output as described below. Escapement values for OBAN (Tables 5.G-8 and 5.G-12) and IOS (Table 5.G-24) models differ by roughly a factor of 5. No formal comparison of the model projections from the IOS and OBAN models was presented. A ranking of model output for median adult escapement of the two models shows reasonable agreement (see Table 1 below). The two models flip the number 1 and 2 ranks of scenarios EBC1 and EBC2. The largest discrepancy was in scenario HOS-LLT with alternative rankings of 5 and 8. Such a table should be included in the report, along with an analogous comparison of through-Delta survival. A comparison of scenarios ranks is in keeping with the sentiment that only the relative output of the models be considered.

Table 1 Relative ranking of alternative model scenarios for medial adult escapement based on the IOS and OBAN models (1 = highest, 10 = lowest).

	EBC 1	EBC 2	EBC2 -ELT	EBC2 -LLT	ESO-ELT	ESO-LLT	HOS-ELT	HOS -LLT	LOS-ELT	LOST -LLT
IOS	1	2	3	7	6	10	4	5	8	9
OBAN	2	1	3	7	4	9	5	8	6	10

- Define ES0 95 ELT. Sensitivity analysis (p. 5.G-79) refers to a model (i.e., ES0 95 ELT) not defined in Table 5.G-2 at the beginning of the Appendix.
- Evaluate and compare sensitivity of populations to a broader range in mortality at the north delta intakes and passage through the Delta. A 5% mortality at the north Delta intake is projected to cause a 58 to 61% reduction in adult escapement (i.e., EBC2- ELT or EBC2-LLT vs. ESO-95-ELT or ESO-95-LLT). This is a huge effect

that would have to be mitigated by other BCDP conservation actions. Presently, 5% entrainment is based on engineering specifications and is lower than at other intake facilities (Perry 2010). These results are also in sharp contrast when through-Delta mortality was increased by 5% and escapement changed by only 0 to 4.6% in the OBAN model. Additional analyses ***must*** be done over a wider range of mortality values, 1% to 10%, to assess how bad the intake problem could be and how well must the intake function. In addition, the discrepancy between the effects of the 5% north Delta intake mortality and the 5% through-Delta mortality needs to be reconciled. It is unclear why these sensitivity results noted in the Conclusion (5.G.4) were not reconciled. They appear to be an important finding of the life cycle analysis.

Comments

A total of 17 candidate life cycle models were considered for use in the Effects Analysis (seven Chinook, eight smelt, one splittail, and one steelhead model). Appendix 5.G reviewed a number of life history models in the Central Valley, but concluded that only two of the Chinook models (i.e., Interactive object-oriented simulation [IOS] model and *Oncorhynchus* Bayesian analysis [OBAN]) were applicable to the BDCP. The OBAN model for winter Chinook involved factors such as water temperature in the Sacramento River (Bend Bridge), exports at the south Delta pumps, days of flow in Yolo Bypass, Delta Cross Channel operation, striped bass (predator) abundance, ocean harvest and ocean upwelling. None of the smelt models were selected, despite the fact that four models (state-space, multivariate autoregression, Bayesian change point, and smolt survival regression) met their five selection criteria. Given the relative importance of the delta smelt, it is unclear how none of the models met the criteria of best available science. It is also unclear, given the important of BDCP, why the plan did not invest in independent model developed tailored to its objectives or invest in modifying one or more of the existing models to better meet the objectives of the plan. The IOS and OBAN models were used to assess effects only on winter-run Chinook salmon.

Under the BDCP, the ISO and OBAN models were used to simulate the projected effects of:

a. Benefits of CM 2 Yolo Bypass Fisheries Enhancement
b. Benefits of SM 15 Nonphysical Barriers (assumed 67% diversion away from Georgiana Slough)
c. Detrimental effects of juvenile entrainment at north Delta intakes (assumed 5% mortality)

No other BDCP conservation measures were considered. How the benefits of Yolo Bypass Fisheries Enhancement were modeled is unclear.

The OBAN model “cannot account for north Delta exports” and “does not include any Delta flow-based covariates other than export (EXPT) and Yolo Bypass inundation (YOLO) and, therefore, cannot account for any potential changes in survival below the north Delta diversions, e.g., because of changes in water velocity” (p. 5.G-32). Consequently, the effect of lower flows due to water withdrawal or slower water velocities and subsequent increased smolt predation were not incorporated in the OBAN modeling. Appendix 5.G goes on to state that because of these modeling limitations, all performance measures should be compared on a relative basis.

However, ratios of model output (i.e., relative differences) will not eliminate biases due to structural defects in the model under alternative scenarios.

The IOS model also assumed “survival and travel times during River Migration are independent of flow” (p. 5.G-44). However, the IOS model does model the effects of flow and route selection and water exports on smolt survival in the Delta (p. 5.G-33). Such assumptions are very important because water withdrawals will affect flows which, in turn, are known to affect the travel time and survival of salmon smolts.

Calibration of the models was limited by available data which, in turn, can limit the range in valid model response. Nevertheless, model descriptions are generally adequate as a whole. Primary model outputs considered median through-Delta survival and annual escapement. In population assessments of endangered or listed species, it is common to include 50-year or 100-year extinction rates. Increasing median escapement has limited value if a salmon population continues to have an unexceptionally high probability of extinction in the future. The simulations should also be summarized in terms of extinction rates under the 14 different operational/environmental scenarios (Table 5.G-2).

The appendix does not include a formal comparison of model output for OBAN and IOS, either on an absolute scale or relative scale. It should be acknowledged that adult escapement differs between models by a weighting factor of 5. More importantly, the relative ranking of the different BDCP scenarios (Table 5.G.-2) between models should be included in Appendix 5.G. Certainty should be assessed, in part, based on the degree of consistency in model predictions.

Appendix 5.J—Effects on Natural Communities, Wildlife, and Plants

Summary

In general, the Panel felt that the information in Appendix 5.J was clearly presented in the tables and figures. Because so much of the information in the appendix depends on the accuracy of the GIS database, the authors should provide a reference or preferably a link to a description of the database and an analysis of its accuracy. As discussed in other sections of our review, providing a single value for the number of acres of habitat that will be occupied by each species is scientifically questionable.

Recommendations

- The description of the methods used to arrive at the number of acres of restored habitat that will be occupied needs to be revised.
- Consider including a range of values (minimum and maximum) of potential occupied habitat rather than a single value.

Comments

Appendix 5.J is divided into five sections each of which addresses a different conservation issue related to natural communities. Our comments on some sections are rather brief and some questions are not relevant to a section so we have included our comments on each section under each question. If there are no comments on a section under a particular question, we felt there was no need to address it.

a. *How well are the proposed analytical tools defined, discussed and integrated?*

Construction-Related Nitrogen Deposition on BDCP Natural Communities

The analysis of construction-related nitrogen deposition is thorough and sufficient. It is clear that the amount of nitrogen produced by construction-related activities of the BDCP will be negligible relative to the amount that is currently being contributed by the surrounding urban and agricultural areas.

Natural Community Restoration and Protection Contributing to Covered Species Conservation

The estimates of the current distribution of natural vegetation types in the Plan Area depend on the accuracy of the GIS database that used for the analysis. Provide a citation for the database and a brief discussion of the error associated with the different community types. In addition, the description of the approach that was used to estimate the amount of habitat for each species (pp. 5J.B-1 and 5J.B-2) is poorly worded and needs revising. The description should state that the details of the approaches used to develop the species-specific habitat models are provided in the species accounts in Appendix 2A.

Analysis of Potential Bird Collisions at Proposed BDCP Powerlines

The authors did an excellent job of integrating spatially explicit information about roost and foraging sites in the Plan Area to estimate the number of potential encounters with power lines and combining this with information in the scientific literature on mortality estimates from each encounter.

Indirect Effects of the Construction of the BDCP Conveyance Facility on Sandhill Crane

The authors considered all of the important indirect effects of the construction on sandhill cranes in the Plan Area. The analytical tools they used were appropriate for the analyses. Most of the estimates of indirect effects came from studies in other regions but that is unavoidable because no detailed studies have been conducted in the Plan Area.

Estimation of BDCP Impact on Giant Garter Snake Summer Foraging Habitat (Acreage of Rice) in the Yolo Bypass

This section is a simple accounting of the number of acres that are planted to rice within the Yolo bypass that may be removed when the bypass is inundated. Rice fields are used as foraging habitat by giant garter snakes and therefore could result in a loss of this habitat for the snake in the Plan Area. By intersecting the maximum amount of rice that was planted in area with the inundation level that results in the maximum amount of rice removed, the analysis provides an estimate of the maximum amount of potential foraging habitat that will be removed. We feel this approach is adequate to address this very specific question.

b. How clear and reasonable is the scale of analysis?

Natural Community Restoration and Protection Contributing to Covered Species Conservation

The scale of vegetation distribution information (1 acre, from Appendix 2A) is reasonable for most species. Although some wildlife species may use habitat patches that are < 1 acre, it is unlikely that those patches contribute significantly to the amount of suitable habitat in the Plan Area.

c. How well were the Panel's earlier comments addressed and applied in the technical appendices/analyses?

Natural Community Restoration and Protection Contributing to Covered Species Conservation

Earlier comments were addressed to some degree. The previous version of this appendix did not have any text at the beginning describing the methods that were used to arrive at the numbers presented in the tables. The description, however, needs to be edited and should specify that the assumptions behind the approaches used when developing habitat models can be found in Appendix 2A.

The other sections of this appendix were not previously reviewed.

d. How well did the technical appendix evaluate the effects of potential BDCP conservation measures on the specified variable(s)?

Natural Community Restoration and Protection Contributing to Covered Species Conservation

As discussed in our review of Chapter 5, the estimate of the amount of habitat that will be occupied by a species following restoration is questionable. The number of acres of suitable habitat that are temporarily or permanently removed and restored are clearly conveyed in the tables in Appendix 5.J. But, the approach used in Appendix 5.J assumes that the proportion of the appropriate habitat that is within the current range of the species in the Plan Area is an appropriate estimate of the proportion of suitable habitat that will be occupied when habitat restoration measures are completed. However, if habitat restoration does not occur within the potential range of the species in the Plan Area, none of it will be occupied. The best way to address this is to set specific goals for habitat restoration within the potential range of the species in the Plan Area and to identify occupancy thresholds.

e. Were the conclusions drawn from the results accurate and did these conclusions appropriately consider uncertainty, including chained statistical uncertainties?

Natural Community Restoration and Protection Contributing to Covered Species Conservation

As discussed in our review of Chapter 5, uncertainty was not considered when estimating the number of acres of restored habitat that a species would occupy following restoration.

f. Were appropriate models used in the technical appendices? If model results conflicted, was this clearly stated and was the conflict appropriately addressed?

Analysis of Potential Bird Collisions at Proposed BDCP Powerlines

The authors considered all 12 bird species that are covered by the BDCP when addressing collision risk. They concluded, and we concur, that the only species that may suffer significant mortality from BDCP-related power lines in the areas is the sandhill crane. The authors used the highest estimate of the probability of mortality due to power line collisions from the published literature when making their computations. In addition, their estimates of the number of potential encounters between cranes and power lines were based on spatially explicit data from the BDCP region. We feel their estimate of potential crane mortality from new power lines that will be constructed is appropriate based on the information available from the site and the literature. We also feel that the estimates of the reduction in crane mortality due to placing bird diverters on existing lines are appropriate. We emphasize, however, that crane mortality from power line collisions should be closely monitored in the Plan Area and additional bird diverters should be put in place if targets for overall reduction in crane collisions are not achieved.

g. How well are the models and analyses described, interpreted and summarized?

Analysis of Potential Bird Collisions at Proposed BDCP Powerlines. The results of their analyses are well described and are well summarized in Tables 2-7 of Appendix 5.J.C. Their estimates of the mitigation from marking power lines are also well described and summarized in section 5.0 of Appendix 5.J.C.

References Cited

Allen, C. R., J. J. Fontaine, K. I. Pope, and A. S. Garmestani. 2011. Adaptive management for a turbulent future. *J. Environ. Mgmt.* **92**:1339-1345.

Alpine, A. E., and J. E. Cloern. 1992. Trophic interactions and direct physical effects control phytoplankton biomass and production in an estuary. *Limnol. Oceanogr.* **37**:946–955.

Baxter, R. D., R. Breuer, L. R. Brown, L. Conrad, F. Freyer, S. Fong, K. Gehrts, L. Grimaldo, B. Herbold, P. Hrodey, A. Mueller-Solger, T. Sommer, and K. Souza. 2010. *Interagency Ecological Program 2010 Pelagic Organism Decline Work Plan and Synthesis of Results: Interagency Ecological Program for the San Francisco Estuary*. Sacramento, CA.

BenDor, T., J. Sholtes, and M. W. Doyle. 2009. Landscape characteristics of a stream and wetland mitigation banking program. *Ecol. Appl.* **19**:2078–2092.

Bennett, W. A., and P. B. Moyle. 1996. Where have all the fishes gone? Interactive factors producing fish declines in the Sacramento–San Joaquin Estuary. Pp. 519–542 in J.T. Hollibaugh (Ed.), **San Francisco Bay: The Ecosystem**. Pac. Div. Am. Assoc. Adv. Sci.

Burgin, S. 2008. BioBanking: An environmental scientist's view of the role of biodiversity banking offsets in conservation. *Biodiv. Conserv.* **17**:807–816.

Cornwell, J. C, P. M. Glibert, and M. S. Owens. 2014. Nutrient fluxes from sediments in the San Francisco Bay Delta. *Est. Coasts*. DOI 10.1007/s12237-013-9755-4

Drake, J. A. 1991. Community-assembly mechanics and the structure of an experimental species ensemble. *Am. Nat.* **137**:1-26.

Drake, J. A. 1990. Communities as assembled structures: do rules govern pattern? *Trends Ecol. Evol.* **5**:159-164.

Dugdale, R. C., F. P. Wilkerson, V. E. Hogue and A. Marchi. 2007. The role of ammonium and nitrate in spring bloom development in San Francisco Bay. *Est. Coastal Shelf Sci.* **73**:17-29.

Feyer, F., B. Herbold, S. A. Matern and P. B. Moyle. 2003. Dietary shifts in a stressed fish assemblage: Consequences of a bivalve invasion in the San Francisco Estuary*. Environ. Biol. Fish.* **67**: 277-288.

Fontaine, J. J. 2011. Improving our legacy: Incorporation of adaptive management into state wildlife action plans. *J. Environ. Mgmt.* **92**:1403-1408.

Glibert, P. M., D., Fullerton, J. M., Burkholder, J. Cornwell, and T. M. Kana. 2011. Ecological stoichiometry, biogeochemical cycling, invasive species, and aquatic food webs: San Francisco Estuary and comparative systems*. Rev. Fish. Sci.* **19**: 358-417.

Grossman, G. D., T. Essington, B. Johnson, J. Miller, N. E. Monsen and T. N. Pearsons. 2013. Effects of fish predation on salmonids in the Sacramento-San Joaquin Delta and associated ecosystems. 25 September 2013, Submitted to the California Department of Fish and Wildlife, Delta Science Program, and NOAA's National Marine Fisheries Service, 71 pp. (www.dfg.ca.gov/er/predation.asp)

Hobbs, J. A., W. A. Bennett, and J. E. Burton. 2006. Assessing nursery habitat quality for native smelts (Osmeridae) in the low-salinity zone of the San Francisco estuary. *J. Fish Biol.* **69**:907–922.

Hobbs, R. J. and V. A. Cramer. 2008. Restoration ecology: interventionist approaches for restoring and maintaining ecosystem function in the face of rapid environmental change. *Ann. Rev. Environ. Res.* **33**:39-61.

Hobbs, R. J., L. M. Hallett, P. R. Ehrlich, and H. A. Mooney. 2011. Intervention ecology: applying ecological science in the twenty-first century. *BioScience* **61**:442-450.

Kentula, M. E. 1996. Wetland restoration and creation, pp. 87-92. In: J. D. Fretwell, J. S. Williams and P. J. Redman (eds.) **National Water Summary on Wetland Resources**. USGS Water Supply Paper 2425. Reston, Virginia, USA.

Kimmerer, W. J. 2002. Effects of freshwater flow on abundance of estuarine organisms: physical effects or trophic linkages? *Mar. Ecol. Prog. Ser.* **243**:39–55

Kimmerer, W. J. 2008. Losses of Sacramento River Chinook salmon and delta smelt to entrainment in water diversions in the Sacramento-San Joaquin Delta. *San Fran. Est. Watershed Sci.* 6(2).

Kleckner, A. 2009. The role of an invasive clam, *Corbula amurensis,* in the cycling of nitrogen in Suisun Bay, CA. M.S. Thesis, San Francisco State Univ.

Kurobe, T., D. Baxa, C.E. Mioni, R. M.Kudela, T.R. Smythe, S. Waller, A. Chapman, A, S. Teh. 2013. Identification of harmful cyanobacteria in the Sacramento-San Joaquin Delta and Clear Lake, California by DNA barcoding. *SpringerPlus* 2: 1-12. 10.1186/2193-1801-2-491.

Lehman, P.W., K. Marr, G.L. Boyer, S. Acuna, S.J. Teh. 2013. Long-term trends and causal factors associated with *Microcystis* abundance and toxicity in San Francisco Estuary and implications for climate change impacts. *Hydrobiologia* **718**:141-15.

Lindenmayer, D.B., G. E. Likens, C. J. Krebs, and R. J. Hobbs. 2010. Improved probability of detection of ecological ''surprises''. *Proc. Natl. Acad. Sci. USA* **107**: 21957–21962.

Lockwood, J. L. and S. L. Pimm. 1999. When does restoration succeed? Pp. 363-392. In: E. Weiher and P.A. Keddy (eds.) **Ecological Assembly Rules: Perspectives, Advances and Retreats**. Cambridge Univ. Press, Cambridge, UK.

Lopez, C. B., J. E. Cloern, T. S. Schraga, A. J. Little, L. V. Lucas, J. K. Thompson and J.R. Burau. 2006. Ecological Values of Shallow-Water Habitats: Implications for the Restoration of Disturbed Ecosystems. *Ecosystems* **9**:422-440.

Lucas, L. V. and J. K. Thompson 2012. Changing restoration rules: Exotic bivalves interact with residence time and depth to control phytoplankton productivity. *Ecosphere* 3 (12). DOI.org/10.1890/ES12-00251.1

Michel, C. J. 2010. River and estuarine survival and migration of yearling Sacramento River Chinook salmon *(Oncorhynchus tshawytscha)* smolts and the influence of environment. Master's thesis. University of California, Santa Cruz. Santa Cruz, CA.

Moilanen, A., A. J. A. van Teeffelen, Y. Ben-Haim, and S. Ferrier. 2009. How much compensation is enough? A framework for incorporating uncertainty and time discounting when calculating offset ratios for impacted habitat. *Restor. Ecol.* **17**:470–478.

Monsen, N. E., J. E. Cloern, and J. R. Burau. 2007. Effects of flow diversion on water and habitat quality: Examples from California's highly manipulated Sacramento-San Joaquin Delta. *San Francisco Est. Watershed Sci.* **5**: Article 2.

Moyle, P. B. 2002. **Inland fishes of California**. University of California Press, Berkeley.

Murphy, D. D., and P. S. Weiland. 2014. Science and structured decision making: fulfilling the promise of adaptive management for imperiled species. *J. Environ. Stud. Sci.* (on-line; DOI 10.1007/s13412-014-0165-0)

Newman, K. B. 2003. Modelling paired release-recovery data in the presence of survival and capture heterogeneity with application to marked juvenile salmon. *Stat. Modelling* **3**:157-177.

Olden, J. D., C. P. Konrad, T. S. Melis, M. J. Kennard, M. C. Freeman, M. C. Mims, E. N. Bray, K. B. Gido, N. P. Hemphill, D. A. Lytle, L. E. McMullen, M. Pyron, C. T. Robinson, J. C. Schmidt, and J. G. Williams. 2014. Are large-scale flow experiments informing the science and management of freshwater ecosystems? *Front. Ecol. Environ.* doi:10.1890/130076.

Parker, V. T. 1997. The scale of successional models and restoration objectives. *Restor. Ecol.* **5**:301-306.

Perry, R. W. 2010. Survival and migration dynamics of juvenile chinook salmon (*Oncorhynchus tshawytscha*) in the Sacramento-San Joaquin River delta. Ph.D. dissertation, University of Washington, Seattle, WA.

Perry, R., P. Brandes, J. Burau, A. Klimley, B. MacFarlane, C. Michel, and J. Skalski. 2013. Sensitivity of survival to migration routes used by juvenile Chinook salmon to negotiate the Sacramento-San Joaquin River Delta. *Environ. Biol. Fishes* **96**:381-392.

Rechisky, E. L., D. W. Welch, A. D. Porter, M. C. Jacobs, and A. Ladouceur. 2009. Experimental measurement of hydrosystem-induced delayed mortality in juvenile Snake River spring Chinook salmon (*Oncorhynchus tshawytscha*) using a large-scale acoustic array. *Can. J. Fish. Aquat. Sci.* **66**:1019-1024.

Rechisky, E. L., D. W. Welch, A. D. Porter, M. C. Jacobs-Smith, and P. M. Winchell. 2013. Influence of multiple dam passage on survival of juvenile Chinook salmon in the Columbia River estuary and coastal ocean. *Proc. Natl. Acad. Sci. United States* **110**:6883-6888.

Rechisky, E. L., D. W. Welch, A. D. Porter, M. C. Jacobs-Scott, P. M. Winchell, and J. L. McKern. 2012. Estuarine and early-marine survival of transported and in-river migrant Snake River spring Chinook salmon smolts. *Sci. Rep.* **2**, Article number: 48, doi:10.1038/srep00448.

Roni, P., K. Hanson and T. J. Beechie. 2008. Global review of the physical and biological effectiveness of stream habitat rehabiliataion techniques. *N. Am. J. Fish. Mgmt.* **28**:856-890.

Roni, P., G. R. Pess, T. J. Beechie and S. A. Morley. 2011. Estimating salmon and steelhead response to watershed restoration: How much restoration is enough? *N. Am. J. Fish. Mgmt.* **30**:1469-1484.

Rosenfield, J. A. 2010. Life History Conceptual Model and Sub-Models for Longfin Smelt, San Francisco Estuary Populations: Delta Regional Ecosystem Restoration Implementation Plan (DRERIP). Sacramento, CA.

Simenstad, C. A. and R. M. Thom. 1996. Functional equivalency trajectories of the restored Gog-Le-Hi-Ti estuarine wetland. *Ecol. Appl.* **6**:38-56.

Sommer, T., C. Armor, R. Baxter, R. Breuer, L. Brown, M. Chotkowski, S. Culberson, F. Feyrer, M. Gingras, B. Herbold, W. Kimmerer, A. Müller-Solger, M. Nobriga, and K. Souza. 2007. The collapse of pelagic fishes in the Upper San Francisco Estuary. *Fisheries* **32**:270–277.

Walters, C. J. 1986. **Adaptive Management of Renewable Resources**. McMillan, New York, NY, USA.

Westgate, M. J., G. E. Likens, and D. B. Lindenmayer. 2013. Adaptive management of biological systems: A review. *Biol. Conserv.* **158**:128-139.

Zedler, J. B., and J. C. Callaway. 1999. Tracking wetland restoration: do mitigation sites follow desired trajectories? *Restor. Ecol.* **7**:69-73.

Appendices

Appendix A—BDCP Effects Analysis Scientific Review Panel members biographies

Nancy Monsen – Delta Hydrodynamics, Stanford University

Dr. Monsen's research has focused on multi-dimensional hydrodynamic modeling of the Sacramento-San Joaquin Delta for twenty years. Her PhD research was based on the TRIM3D hydrodynamic model. She also has consulting experience with the DELFT3d hydrodynamic model. She is currently Visiting Scholar in the Environmental Fluid Mechanics Laboratory, part of the Civil and Environmental Engineering Department, at Stanford University. Over the prior two years, Dr. Monsen worked as a Stanford Research Associate on a Delta Science program funded research project to develop a multi-dimensional hydrodynamic model of the Sacramento-San Joaquin Delta using Stanford's SUNTANS model. Prior to working at Stanford, she worked for ESA PWA (formerly Philip Williams and Associates) for a year and a half and at the U.S. Geological Survey (Menlo Park, National Research Program) for ten years. Dr. Monsen earned her doctorate in Civil and Environmental Engineering at Stanford University in 2001.

Greg Ruggerone – Anadromous Fish

Dr. Ruggerone has investigated population dynamics, ecology, and management of Pacific salmon in Alaska and the Pacific Northwest since 1979. He was the Project Leader of the Alaska Salmon Program, University of Washington, from the mid-1980s to early 1990s where he was responsible for conducting and guiding research at the Chignik and Bristol Bay field stations. Most of his research involves factors that affect survival of salmon in freshwater and marine habitats, including climate shifts, habitat degradation, predator-prey interactions, and hatchery/wild salmon interactions. He is currently a member of the Columbia River Independent Scientific Advisory Board and the Independent Scientific Review Panel. He recently served as the fish ecologist on the Secretary of Interior review of dam removal on the Klamath River.

(http://www.nrccorp.com/staff/staff_ruggerone.htm).

Charles (Si) Simenstad – Pelagic/Native Fish

Charles ("Si") Simenstad, Research Professor at the University of Washington's School of Aquatic and Fishery Science (SAFS), is an estuarine and coastal marine ecologist and coordinator of the Wetland Ecosystem Team (WET). Si has studied the organization and function of estuarine and coastal marine ecosystems throughout Puget Sound, Washington, Oregon and California, and Alaska for over forty years. Much of this research has focused on the functional role of estuarine and coastal habitats to support juvenile Pacific salmon and other biotic communities, and the associated ecological processes and community dynamics that are responsible for enhancing their production and life history diversity. Recent research has integrated such ecosystem interactions with applied issues such as restoration of estuarine and coastal wetland ecosystems, and ecological approaches to evaluating the success of coastal wetland restoration from ecosystem to landscape scales. He is presently Co-Editor in Chief of

Estuaries and Coasts, on the Editorial Board of *San Francisco Estuary and Watershed Science*, volume co-editor for the "Treatise on Estuarine and Coastal Science", a standing member of the Scientific Advisory Group (SAG) of the Interagency Ecological Program (IEP) in the San Francisco Bay-Delta, and was recently appointed to Environmental Advisory Board to the Chie of Engineers, US Army Corps of Engineers; (http://fish.washington.edu/people/simenstd/).

John Skalski – Fishery population dynamics and modeling

Dr. Skalski is a Professor of Biological Statistics in the School of Aquatic & Fishery Sciences, College of the Environment, at the University of Washington. He is also an adjunct professor in Quantitative Ecology and Resource Management and Wildlife Sciences, and an instructor in the Center for Quantitative Sciences. His expertise is in sampling theory, parameter estimation, mark-recapture theory, and population dynamics. His research focuses on the development of sampling methodology, field designs, and statistical tests for human-induced and natural effects on organismic and ecological systems. He is the statistician in charge of survival compliance testing at all 13 major hydroprojects in the Snake-Columbia River system. He has authored or coauthored over 100 technical reports on salmonid survival studies and over 40 peer-reviewed articles on tagging studies. Dr. Skalski is a member of the American Statistical Association, The Wildlife Society, and the American Fisheries Society. He is also a Certified Wildlife Biologist through The Wildlife Society.

Alex Parker – Aquatic Ecology/Food Webs

Alex Parker is an Assistant Professor of Oceanography at the California Maritime Academy, CSU and a Research Associate at the Romberg Tiburon Center, San Francisco State University. His Ph.D. work (College of Marine Studies, University of Delaware) focused on microbial biogeochemistry in the Delaware Estuary, a highly modified estuary on the US East Coast. Dr. Parker was a CALFED Post-Doctoral Science Fellow. His work in the San Francisco Estuary includes the study of pelagic phytoplankton rate processes, wetland primary producers, the dynamics of heterotopic bacteria and the carbon and nitrogen physiology of cyanobacteria in the SFE Delta. Additionally, Dr. Parker has carried out research in coastal and equatorial upwelling areas as well as polar environments.

Tom Parker, Plant Communities

Thomas Parker is Professor of Ecology and Evolution at San Francisco State University who studies the ecology and evolution of plant communities, focusing on their dynamics. Current research includes the effects of climate change on tidal wetlands of the San Francisco Bay-Delta, and the ecology and evolution of *Arctostaphylos* species in chaparral and other communities (http://bio.sfsu.edu/people/v-thomas).

T. Luke George, Terrestrial Ecology

Dr. George has been a faculty member in the Department of Wildlife at Humboldt State University since 1991. He specializes in the design, implementation, and analysis of demographic, population monitoring, and habitat selection studies of terrestrial vertebrates. His recent work has focused on estimating demographic parameters and modeling habit selection of threatened and at risk species including the San Clemente sage sparrow, northern spotted owl, greater sage grouse, and tricolored blackbird. Dr. George assisted with the development of a population viability analysis (PVA) of the San Clemente sage sparrow and has served as an advisor on PVAs of Western snowy plovers and San Clemente loggerhead shrikes. He has conducted research on habitat selection and space use of Steller’s jays and common ravens in Redwood National and State Parks and has advised state and federal agencies on strategies to reduce nest predation by corvids on marbled murrelets, Western snowy plovers, and other threatened and endangered species in California.

Appendix B—Charge to the Delta Science Program Independent Review Panel for the BDCP Effects Analysis Review, Phase 3 (dated 2/12/2014)

The Panel will be charged with assessing the scientific soundness of Chapter 5: Effects Analysis and the associated technical appendices. The Panel will make recommendations for how these might be improved with respect to achieving their stated goals. Specific attention will be given to the following questions:

Chapter 5: Effects Analysis

General Questions

1. How well does the Effects Analysis meet its expected goals?
2. How complete is the Effects Analysis; how clearly are the methods described?
3. Is the Effects Analysis reasonable and scientifically defensible? How clearly are the net effects results conveyed in the text, figures and tables?
4. How well is uncertainty addressed? How could communication of uncertainty be improved?
5. How well does the Effects Analysis describe how conflicting model results and analyses in the technical appendices are interpreted?
6. How well does the Effects Analysis link to the adaptive management plan and associated monitoring programs?

Review of Specific Analyses

7. Are the analyses related to the north Delta diversion facilities appropriate and does the effects analysis reasonably describe the results? In particular:

- Was existing empirical information such as Perry *et al.* 2010 and Newman 2003 incorporated appropriately into the modeling? Where model runs required extrapolation beyond existing data ranges, were assumptions and interpretations appropriate?
    - Does the analysis of the frequency of reverse flows at Georgiana Slough accurately characterize changes in hydrodynamics due to changes in river stage, sea level rise, and Delta habitat restoration?

8. How should the effects of changes in Feather River flows on fish spawning and rearing be characterized? In particular, how should the trade-off between higher spring flows and lower summer flows be interpreted? Does the analysis adequately capture the expected benefits of CM 2, Yolo Bypass Fishery Enhancement?
9. Does the analysis adequately describe the predation and other screen-related effects of the proposed north Delta diversion structures? Is the application of the observed mortality rate at the fish screen of the Glenn-Colusa Irrigation District (GCID) an appropriate

assumption for expected mortality at the proposed BDCP north Delta intakes? Are there other studies on salmonid survival at positive barrier fish screens that would be appropriate to apply?

10. Does the effects analysis provide a complete and reasonable interpretation of the results of physical models as they relate to upstream spawning and rearing habitat conditions, particularly upstream water temperatures and flows resulting from proposed BDCP operations?
11. Does the effects analysis use a reasonable method for “normalizing” results from the salvage-density method to the population level for salmonid species?
12. Are the assumptions of the analysis of aquatic habitat restoration food web effects appropriate for covered fish species? Are the conclusions and net effects appropriate?
13. Is the analysis of food web benefits to longfin smelt from habitat restoration appropriate? How well do the analyses link intended food web improvements to improvement in the longfin smelt spring Delta outflow/recruitment relationship?
14. How well does the analysis address population-level effects of the BDCP on white sturgeon?

Technical Appendices

For each Chapter 5 technical appendix:

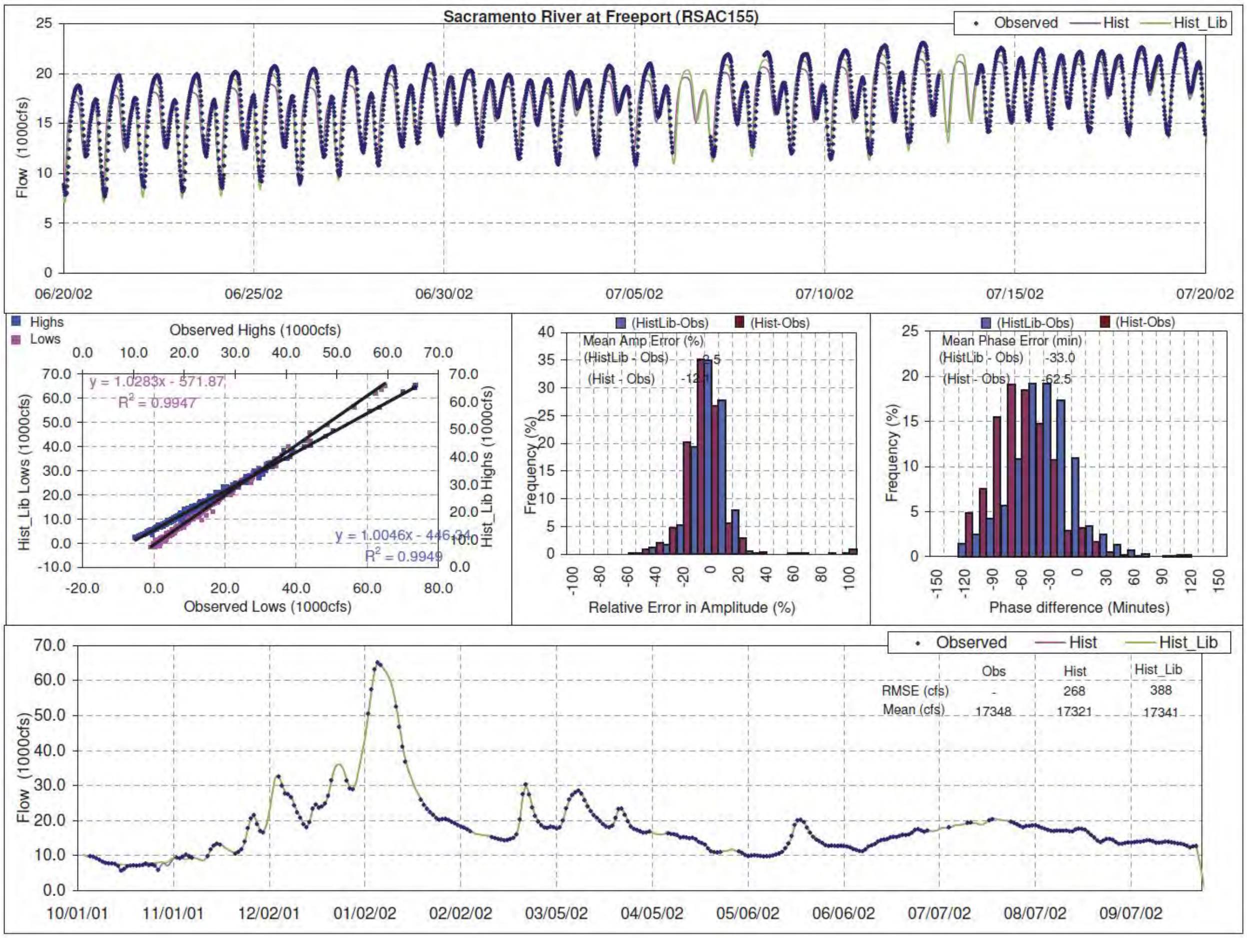
Approach and Analysis

a. How well are the proposed analytical tools defined, discussed and integrated?
b. How clear and reasonable is the scale of analysis?
c. How well were the panel’s earlier comments addressed and applied in the technical appendices/analyses?
d. How well did the technical appendix evaluate the effects of potential BDCP conservation measures on the specified variable(s)?
e. Were the conclusions drawn from the results accurate and did these conclusions appropriately consider uncertainty, including chained statistical uncertainties?

Models

f. Were appropriate models used in the technical appendices? If model results conflicted, was this clearly stated and was the conflict appropriately addressed?
g. How well are the models and analyses described, interpreted and summarized?

Appendix C—Observed tidal stage and flow time series data from three key locations along the Sacramento River (from BDCP Appendix 5A-D1, pp. 128-129)



Appendix D—Variance Calculations Associated with Salvage Model

Estimator of average salvage:

$$\hat{\bar{S}} = \frac{\sum_{i=1}^{n}\sum_{j=1}^{m} D_i V_j}{nm} \qquad (1)$$

Then, the variance of this average salvage value is as follows:

$$\text{Var}\left(\hat{\bar{S}}\right) = \text{Var}\left[\frac{\sum_{i=1}^{n}\sum_{j=1}^{m} D_i V_j}{nm}\right]$$

$$= \text{Var}_n\left[E_m\left[\frac{\sum_{i=1}^{n}\sum_{j=1}^{n} D_i V_{ij}}{nm}\middle| n\right]\right] + E_n\left[\text{Var}_m\left[\frac{\sum_{i=1}^{n}\sum_{j=1}^{n} D_i V_{ij}}{nm}\middle| n\right]\right]$$

$$= \text{Var}_m\left[\frac{\sum_{i=1}^{n} D_i \bar{\bar{V}}}{n}\right] + E_m\left[\frac{\sum_{j=1}^{m} \bar{D}_i^2 \sigma_i^2}{n^2 m}\right]$$

$$= \frac{\bar{V}^2\sigma_D^2}{n} + \frac{\sigma_V^2}{m} E\left[\frac{\sum_{i=1}^{n} \bar{D}_i^2}{n^2}\right]$$

$$= \frac{\bar{V}^2\sigma_D^2}{n} + \frac{\sigma_V^2}{m}\cdot\frac{\left(\sigma_{D_i}^2 + \bar{D}^2\right)}{n}$$

$$= \frac{\bar{V}^2\sigma_D^2}{n} + \frac{\bar{D}^2\sigma_V^2}{mn} + \frac{\sigma_V^2\sigma_D^2}{m}. \qquad (2)$$

However, if the variance of $\hat{\bar{S}}$ is calculated based on the empirical variance of the nm values, the variance has the expected value as follows:

$$E\left(\frac{s_{S_{ij}}^2}{nm}\right) = \frac{E\left(s_{ij}^2\right)}{nm} = \frac{\text{Var}(S_{ij})}{nm}$$

$$\frac{\text{Var}(\hat{S}_{ij})}{nm} = \frac{1}{nm}\{\text{Var}_n[E_m(S_{ij}|n)] + E_n[\text{Var}_m(S_{ij}|n)]\}$$

$$= \frac{1}{nm}\{\text{Var}_n[E_m(D_i V_{ij}|n)] + E_n[\text{Var}_m(D_i V_{ij}|n)]\}$$

$$= \frac{1}{nm}\left\{Var_n[D_i\bar{V}] + E_n\left[D_i^2\sigma_{V_{ij}}^2\right]\right\}$$

$$= \frac{1}{nm}\{\bar{V}^2\sigma_D^2 + \sigma_V^2\cdot E[D_i^2]\}$$

$$= \frac{1}{nm}\{\bar{V}^2\sigma_D^2 + \sigma_V^2\cdot E[\sigma_D^2 + \bar{D}^2]\}$$

$$= \frac{\bar{V}^2\sigma_D^2}{nm} + \frac{\bar{D}^2\sigma_V^2}{nm} + \frac{\sigma_V^2\cdot\sigma_D^2}{nm} \qquad (3)$$

Note variance as calculated (3) is smaller than the correct variance (2). The first term of Equation (3) is inappropriately divided by m. Hence, CI width and uncertainty will be underestimated.

This document is a compilation of the fish agency "red flag" comments and ICF's responses thereto regarding the BDCP draft Effects Analysis. These informal comments were developed by agency staff to flag quickly issues that need to be resolved prior to final submittal of the plan. As such, they do not reflect an official agency position or decision. ICF's responses are preliminary and intended to facilitate further discussion and resolution of issues. ICF and the agencies will be working to address the red flag issues in the coming weeks.

DFG April 2012 BDCP EA (Ch. 5)
Staff "Red Flag" Review
Comprehensive List

STURGEON

Methodological

- The logic of section 5.5.5.4 (Net Effects) is difficult to follow and does not attempt to prioritize Plan outcomes relative the magnitude of their likely impacts on sturgeon production. The largely Best Professional Judgment discussion seems to miss rough quantification opportunities that might be derived from flow abundance-relationships, adult migration straying rates into the Yolo Bypass, and known survival and harvest rates (as they might, for example, relate to illegal harvest reduction). The conclusions in the paragraph beginning on line 29 seem essentially unsupported.

  ICF Response: We will make this discussion clearer.

- The assessment effects seems to turn the notion of uncertainty upside down. In general, the Plan reduces winter-spring outflow, and in some regards Sacramento River Flow. There is a strong historical association between flow conditions and sturgeon production, which the EA seems to dismiss, citing a lack of understanding of the mechanisms underlying the association. This would seem to be a very risky approach from a species conservation point of view, given that the anticipated offsets to the potential flow impact are Plan attributes that address "stressors" that have not been clearly associated with variation in production (e.g. food supply).

  ICF Response: We will make this discussion clearer.

- The EA seems to suggest that a reduction in entrainment of juvenile sturgeon at the south Delta offsets (justifies) the effects of reduction in winter-spring outflows. While the statement that "Entrainment of juvenile sturgeon at the south Delta pumping facilities, however, is considered an important stressor for this life stage." may be true, it is not considered to be a more important stressor on sturgeon than reduced winter-spring outflow. Entrainment of juvenile white sturgeon at the south Delta pumping facilities is not a significant stressor, when compared to the loss of winter-spring outflow. Although entrainment of green sturgeon is a somewhat different matter, reducing it in exchange for reducing winter-spring outflow is still not preferred.

  ICF Response: We will make this discussion clearer.

- There is a general tendency section 5.5.5.1 (Beneficial Effects) to overstate Plan benefits. An example, can be found in the sentence beginning at line 8 on page 5.5-114, which concludes that Plan-related changes in DCC operations will reduce entrainment and improve the ability of adult sturgeon to cue in on Sacramento River flows. These conclusions seem to ignore that adult sturgeon are rarely entrained, and that overall the Plan substantially reduces lower Sacramento River flows.

  ICF Response: We will make this discussion clearer.

Flows

- River flows are important to sturgeon production in the Sacramento River system and Delta, and PP operations are predicted to result in significant occurrences of river flow reduction during the sturgeon spawning and early rearing periods. Reductions are most pronounced in the mainstem Sacramento River downstream of the Fremont Weir and the proposed northern delta intakes, but occurrences of substantial flow reductions are also predicted in more upstream river reaches.

  As identified in the December, 2011 version of Appendix C, the PP is predicted to expose green sturgeon larvae to substantial reductions in July-September Feather River flows in most years. In addition, predicted juvenile white sturgeon migration period flows at Verona are sometimes lower under PP operations, and white sturgeon larval transport flows at Wilkins Slough fall more frequently below thresholds in dry years.

  The collective predicted negative river flow effects of the PP create the risk of a depressive effect on sturgeon production that may not be overcome by more favorable PP aspects (e.g. reduced entrainment, increased food production supply). This suggests the need to modify the PP to reduce the magnitude and frequency of river flow reduction occurrences, in both upstream and downstream areas.

  ICF Response: Changes to operations are currently under discussion with the agencies.

SALMONIDS

Effects Analysis

- Combining all salmonids into one net effects analysis is not appropriate and “averages” out the adverse effects of individual runs. The net effects analysis needs to differentiate between Sacramento and San Joaquin river salmonids; salmon and steelhead; and individual runs of salmon (i.e. winter-run, spring-run, fall and late fall-run).

  ICF Response: We agree as was noted in the Chapter 5 Admin Draft. We plan to work closely with the fish and wildlife agencies to develop separate analyses for each salmonid run and, where appropriate, each population.

- Analysis of the reduction in Sutter Bypass floodplain acreage has not been addressed in the effects analysis. This issue has been raised previously and still not been responded to. Data shows that there could be a significant reduction in floodplain habitat in the lower Sutter Bypass based on the preliminary proposal due to lowering the river stage at Verona, which will lead to a direct reduction in Butte Creek spring-run Chinook salmon rearing habitat (and splittail).

  ICF Response: We will work with the agencies to develop this analysis.

- The rationale for the degree of certainty seems unfounded for some of the stressors (e.g. transport flows, flow regulation, and flow-associated habitat (5.5-55-59)). The tables show a high degree of uncertainty regarding the effects of flow on salmon on the basis that there is no quantitative analysis or little applicable literature, which is unjustified.

  ICF Response: We can work with agencies to gather information to better justify the certainty of stressor rankings.

- Table 5.5-16 is contradictory to the statements made at spring-run egg mortality and winter-run redd dewatering.

  ICF Response: We will work with the agencies to correct this.

Implementation

- The decision on phasing of proposed North Delta Diversions (NDD) intakes needs to be determined. From a fishery management perspective it would be best to build some (e.g., two) of the intakes and operate them prior to building the rest. This phasing approach would allow us to learn and potentially correct any unforeseen issues.

  ICF Response: This is a policy-level decision.

- The timeline to complete the required environmental documentation and permitting for Conservation Measure 2 is much longer than necessary to complete this critical measure. It should not require more than three to five years to complete environmental compliance and an additional two years to acquire the necessary permits.

  ICF Response: We agree that an aggressive timeline for CM2 is needed to ensure that the substantial benefits of those actions are realized as soon as possible. The current timeline is based on the likely need to design and permit many CM2 actions separately. We will consider ways to accelerate the schedule. Changing the assumed timeline is a policy-level decision.

Upstream

- The preliminary proposal shows a reduction in the end of September storage (cold water pool storage) which is unacceptable and needs to be addressed.

- Winter-run redd dewatering and lower weighted usable spawning habitat in the Sacramento River under the preliminary proposal is not acceptable. This would lead to a significant decline in the population (as estimated by the JPE).

- Spring-run egg mortality in the mainstem of the Sacramento River is near 100 percent during dry and critical dry years. This type of egg mortality could lead to the extirpation of spring-run Chinook salmon from the mainstem of the Sacramento River during one drought cycle.

  ICF Response: We propose exploring the inclusion of upstream temperature controls in the modeling done for the effects analysis to reduce uncertainty of these effects and to offset CALSIM's modeling approach to better reflect the actual operations of the project.

North Delta Flow

- Reduction in flows below proposed NDD could have significant impacts on the transport flows for juvenile fish species and the upstream migration cues of adults.
- The net effects analysis shows that there would be increased reverse flows in the Sacramento River below the proposed NDD due to the preliminary proposal (5.3-4, line 10-13), this is not protective and doesn't appear to account for real time operations to minimize these effects.

  ICF Response: We will work to better explain this issue and work with the fish and wildlife agencies to find a diversion scheme that can included in the public draft BDCP.

SJR Flows at Antioch (5.3.1.2.9)

- The continuation of zero and (-) SJR flows at Antioch is not protective of San Joaquin Basin fish. While the PP_ELT and PP_LLT show an increase in OMR and SJR flows due to a reduction in south Delta exports, the continuation of low flows in August and September followed by 0 cfs in October and November and (-) 2000 cfs in December is not protective. Positive SJR flows during this time are important and necessary to cue upstream adult migration, reduce straying, and to help address water quality concerns (e.g., DO and temperature).

  ICF Response: Our analysis did not explicitly identify this as an area of concern, but we will work with the agencies to further examine this issue goals.

Entrainment Issues

- Increasing entrainment in the south delta compared to EBC in dry and critical years is a concern and should be avoided. Due to the lack of discussion on this issue, it leads the reader to believe that there will be more water export than existing conditions under the preliminary proposal.

  ICF Response: The PP does not include an E/I ratio that currently governs reverse flows (in combination with OMR) and therefore in drier years, exports from south Delta, and associated entrainment, are increased. Our analysis does not take into account the real-time operations management groups that have been effective at reducing the risk of entrainment, and therefore may overestimate the entrainment under the PP. Nonetheless, we are evaluating how to minimize entrainment in drier years.

SMELT(S)

(Delta Smelt, Section 5.5.1)

Methodological

- The paragraph beginning at the bottom of page 5.5-24 (and at other locations in Section 5.5.1) notes that there is no change anticipated in Fall abiotic habitat when comparing the PP with EBC1 (existing condition, sans the Fall X2 RPA action). This may be a problematic PP outcome in the context of a NCCP. Reasonable arguments have been made that recent changes in Delta water management have substantially degraded Fall abiotic habitat conditions, particularly in Falls following Above Normal and Wet water years (roughly half of all years, historically), contributing to the POD condition for delta smelt. This suggests that the "no change" outcome produced by the PP would make it difficult to demonstrate a PP contribution to species recovery.

  ICF Response: This method accounts for only a portion of the population and recovery of the species doesn't necessarily need to be driven by this one component. We believe that the Effects Analysis shows at least a minor beneficial effect on the species relative to existing conditions, and that it has great potential for larger benefits depending on actual food production and location of delta smelt population in relation to those areas. While there is uncertainty about these conclusions, we attempted to address this uncertainty in Chapter 5 by including focused studies prior to the new intake operation as well as describing how adaptive limits could be used if needed, to increase fall outflows. We hope to continue discussions with the agencies regarding how to address the Fall X2 issues.

- The paragraph beginning at line 16 on page 5.5-17 introduces the approach of examining Plan Fall abiotic habitat effects based on Feyrer et al. (2011). The text then goes on to identify several "concerns" DWR and applicants have regarding the approach. This expression of concern is reasonably presented, other than the fact that the similar concerns of other parties regarding the investigations critical of Feyrer et al. are not presented. The overarching "red flag" here is that the key technical concerns surrounding this aspect of the effects analysis are not be addressed in a systematic way, other than through non-collaborative production of "combat science." This approach is not effectively reducing uncertainty about Plan outcomes, and places a particular burden on permitting agencies who will have no choice but to assess the uncertainties and conservatively mold the permits around their perception of uncertainty.

  ICF Response: In the revised Appendix C, we have more clearly defined the issues with the method and as stated above, we hope to continue discussions with the agencies regarding how to address the Fall X2 issues.

Plan Concerns

- As Figure 5.5-1 clearly shows, the role up for delta smelt is about balancing the uncertain benefits of food , predation, and tidal habitat benefits against the uncertain negative effects of Fall abiotic habitat degradation. This is not a very comfortable assessment for such a key species. Some improvement of the Fall habitat situation would go a long way towards improving the ability of the project to achieve the conserve standard for an NCCP.

  ICF Response: We hope to continue discussions with the agencies regarding how to address the Fall X2 issues.

- Table 5.5-4 (and other similar tables) shows essentially no existing habitat in the southern Delta. This is counter-intuitive, given that the same southern Delta had lots of smelt in it in the early 1970s. This is part of a general problem that the southern Delta may be getting short shrift in considering potential restoration potential.

  ICF Response: The analysis conducted for tidal habitat restoration in the ROAs is at a broad landscape level and may not capture the full range of potential habitat benefits or suitability because of the coarse level of the analysis. As such, individual restoration areas and some existing areas may provide suitable habitat for smelt and other species. However, the relatively poor tidal wetland habitat quality assessed for the south Delta is consistent with its habitat characteristics for smelts, in particular water clarity. It is uncertain whether or not the south Delta could regain some of the more desirable habitat characteristics.

(Longfin smelt)

- Population effect of reduced winter-spring outflow identified in the effects analysis.

- On line 11 of page 5.5-48 the text raises the notion of "bottlenecks" between lifestages. The examination of existing data does not suggest the existence of such a population dynamics effect. Age 2 fish appear to be suffering the greatest effects of food limitation, but it is still the case that there is roughly a linear stock-recruitment relationship between the two age classes. It should not be assumed that benefits to one lifestage will not be realized in subsequent stages.

  ICF Response: Analyses of species population dynamics available in the scientific literature have identified life-stage bottlenecks that have an effect on setting year-class strength and dynamics. Kimmerer hypothesized a population bottleneck for delta smelt during the summer months based on high levels of population abundance variation earlier in the life history with substantially less variation in abundance of older life stages such that early life stage abundance was not a good predictor of abundance at later life stages. Rosenfield and Baxter also hypothesized that there may be a population bottleneck for longfin smelt. The discussion can be revised to add literature and other support to the notion of a bottleneck between lifestages for a species and acknowledge that benefits of conservation actions for one lifestages may not translate into population level

benefits to later lifestages in the event that a population bottleneck exists (e.g., a density-dependent mechanism).

- The conclusion of “no net effect” with “low certainty” found at line 4 on page 5.5-50 does not quite capture the essence of the accompanying analysis. Although the statement is not entirely unreasonable, it does not capture the notion of species RISK when an easily foreseeable negative outcome is matched against a pretty speculative benefit. Whereas it may suffice in the EA to have a best guess as to the net effect of the project, I think the NCCP will have to grapple with the downside risk of a likely flow impact, which is to be offset by reasonable, but highly uncertain speculation about food supply improvements.

  ICF Response: Adaptive management, coordination with agencies during permitting and design, and maintenance, the risks associated with the project can be reduced and the benefits can be enhanced.

- Section 5.5.2 devotes considerable space to discussing the expansion of subtidal (“suitable”) habitat and its potential benefits. Given the severe decline in species abundance it seems highly unlikely that expanding the amount of this very general habitat type will benefit the species. To be fair, the Plan characterizes this attribute as only a slightly positive benefit.

  ICF Response: Aquatic habitat restoration under BDCP would result in the expansion of access to substantial areas of intertidal, subtidal, and seasonally inundated floodplain habitat. These inundated habitats would also be expected to produce organic material that support the food supplies for longfin smelt and other covered fish species. Although at current levels of population abundance of delta smelt, longfin smelt, and salmonids expansion of access to subtidal habitat may provide little physical habitat benefit food production in these areas but may be important in rebuilding population growth, survival, and abundance of covered fish. The additional access to suitable shallow water subtidal habitat in the future is expected to be greater as population abundance of covered species recovers to higher levels.

FWS BDCP Effects Analysis red flags for March, 2012

Elements marked by an asterisk are provisional, and may change after review of the outstanding Chapter 5 appendices.

Issue Area 1: Incomplete conceptual foundation for the Effects Analysis

***The effects analysis deals with the critical concept of uncertainty inconsistently and does not effectively integrate, use, and report uncertainty in the Net Effects.** The BDCP Independent Science Advisors, the National Research Council review panel, the Delta Science Program panel, and we have all commented on the inherent uncertainty in the scientific understanding of certain aspects of the Bay-Delta ecosystem. This extends to difficulty predicting how the ecosystem might respond to BDCP implementation. Uncertainty needs to be used objectively and consistently, and the appendices and Net Effects need to develop and propagate uncertainty through the threads of the effects analysis. Highly important variation in the value and uncertainty of individual conservation measure features will occur over space and time as a function of implementation strategy and other factors. Many of the current conservation measures and issues are, or appear to be, overly simplified or otherwise superficially analyzed. The list includes OMR management, fish-habitat relationships, the habitat-for-flow trade-off, predator suppression, nuisance vegetation suppression, and others. Each of the foregoing issues raises uncertainties that propagate through the threads of analysis and must be reckoned with in the "net" conclusions. To the extent we can form our own conclusions about the Net Effects without having access to all the revised documents, it appears that inconsistency in dealing with uncertainty has resulted in conclusions that overly optimistically predict Preliminary Project benefits for almost all of the target fish species almost everywhere. As such, we are reluctant to rely on the conclusions of the present effects analysis. We await receipt of the outstanding appendices, and look forward to working closely with our partners to provide technical assistance as these matters are resolved.

> **ICF Response: ICF has attempted to document certainty in multiple ways throughout the EA. For example, the description of each method in each appendix highlights its limitations and assumptions made. We have also tried to explain in the results and conclusion sections the uncertainties associated with the analysis. Also, in Chapter 5, we've attempted to document the basis for each conclusion and its level of certainty. However, ICF acknowledges that additional work can be done to improve this component of the analysis, including a more robust linkage to adaptive management, research and monitoring to address specific areas of uncertainty.**

***A key missing piece from the Analytical Framework document is how the Effects Analysis will be framed in the context of fish population dynamics.** We expected this to occur in the draft Technical Appendix on the subject of fish populations, but that document did not fully analyze long-term and recent population trends in the target fishes. There is clear evidence that most of the covered fish species have been trending downward. The document should clearly and accurately lay out what is known of the foundations of each species' population dynamics (e.g., density-dependent under some circumstances?, trends in carrying capacity?, etc.) as mechanistically as possible and discuss how BDCP actions will influence these processes. Because the conceptual foundations presented to date do not frame the effects in the context of historical and present-day fish population dynamics and the most parsimonious explanations of their causes, it is unclear how the net effects should be interpreted. We await receipt of the life cycle modeling appendix to complete our review of this issue, and look forward to continuing to work with our partners to help ensure that the best available science is used in the effects analysis.

> **ICF Response: Each appendix provides a background on the specific issues associated with that topic. For example, the entrainment appendix describes the historical entrainment trends and how that's affected fish. However, a more comprehensive description of each covered fish species population trends and ecological status could be developed to the extent it is necessary to understand the effects of the BDCP on each covered species. This could be included in Appendix 2.A, Species Accounts. We look forward to discussing with the Agencies ways in which this comment can be addressed with the best available data.**

Issue Area 2: Inadequate conceptual models and analysis of estuarine fish habitat, and project issues resulting from same

***The objectives for restoring habitat addressed in the Chapter 5's Restoration Appendix are simply described, but it is not clear whether the plan will or can achieve them.** The draft Appendix E states that BDCP's habitat restoration has two objectives[1]. The first is to "increase the amount of available habitat for covered fish species." This first objective is reasonable, but does not clearly articulate that new habitat needs to be good habitat. We know quite a bit about what determines habitat value to covered fish species. This knowledge is partly reflected in the habitat suitability indices that are currently under development, but is often discounted elsewhere in the Chapter 5 documents. The habitat for BDCP target fishes, and all estuarine fishes for that matter, is fundamentally created by the interaction of tidal and river channel flows with the broader estuary landscape. The Preliminary Project proposes to extract larger volumes of fresh water from the Delta than are currently exported against a backdrop of rising sea level and a re-design of the estuary landscape that will change tidal flows. Whether this can be accomplished while other parts of the plan simultaneously contribute to recovery of covered species is an unanswered question of central importance. Fully incorporating existing science on the interplay of freshwater flow and the Plan Area landscape and its constituent species would provide more accurate and defensible conceptual models for the Effects Analysis. We also suggest consulting the Department of Interior Adaptive Management Technical Guide and other adaptive management resources on the role of (potentially conflicting or alternative) conceptual models in the adaptive management process. We look forward to working with our partners and providing technical assistance toward the resolution of this issue.

The second objective is "to enhance the ecological function of the Delta." This formulation is not clear. The Delta provides multiple ecological services, and alterations to different parts of the Delta may potentially contribute to them in different ways. There have been several large-scale,

[1] We note that these objectives are more akin to goals. They are not at present specific enough to function as objectives in the context of performance evaluation or adaptive management.

unintentional or quasiintentional "wetland restoration projects" in the Bay-Delta since 1920. These include Franks Tract in the 1930s, Mildred Island in the early 1980s, Liberty Island in the latter 1990s, and Napa River marsh in the past decade to name a few. There is also the seasonal fish habitat generated by large-scale floodplain restoration along the lower Cosumnes River that started in the mid-1990s. The draft appendix never mentions these events or synthesizes what is known about them. This is a critical aspect of the analysis, and needs to be done credibly. We believe these "unintended experiments" provide useful lessons in what we may expect from actions on similar spatial scales in similar circumstances in various restoration scenarios.

A close look at the estimated elevations of restored habitats shows that much of the acreage is not at intertidal elevation and thus will not readily produce the dendritic channel mosaics on a tidal marsh plain that are frequently espoused in the appendix for their fish production benefits. Particularly by the late long-term, there is a lot of the subtidal habitat types in the model outputs[2]. We do not know if unintentional habitat restorations that have occurred have increased the productivity of the Delta beyond what it would have been without them. In a pure carbon-productivity sense they might have – because productivity is just creation of biological carbon per unit of time. However, these and other "wetland restorations" have not noticeably increased the capacity of the Delta to produce BDCP-covered native fishes. As achieving this is a key premise of the BDCP, understanding these examples and learning from what has happened in each case is a matter of great importance. We look forward to providing assistance to our partners as these comments are addressed.

ICF Response: a. Regarding the first objective, the HSI approach used measures both the quality and quantity of habitat using habitat units as the unit of measurement. This analysis is based on CALSIM, DSM2, and RMA Bay Delta models that incorporate climate change effects and restoration in the Delta over time so that the effects of changes in flows are captured by the analysis in terms of habitat units. The larger question regarding how flow and habitat restoration interact in terms of effects on covered fish, the information and tools we would need to address this issue in the EA do not exist. Therefore, this needs to be handled with adaptive management, which requires additional coordination to develop sufficient clarity and rigor. We would like to coordinate with the agencies regarding the development and application of conceptual models, as well as a more robust adaptive management plan, to address these issues.

Regarding the second objective, ICF has reviewed and will continue to review information available from these unintentionally restored wetlands and incorporate relevant information as appropriate. However, the habitat restoration proposed under the BDCP is substantially more than any other restoration effort implemented before. In addition, BDCP tidal restoration will be purposefully designed to maximize benefits to covered species and minimize adverse effects (e.g., submerged aquatic vegetation), and will rely on adaptive

[2] It may be possible to manage subsided lands to raise them back to sea-level so that they can support self-sustaining intertidal marshes. However, that process can be very slow and the full realization of potential physical morphology could take many decades.

> **management and information collected from successful restoration sites, such as Liberty Island, to achieve this goal.**

***The modeling shows a gain of shallow, intertidal habitats in the Plan Area by the early long-term, which is a goal of the BDCP. However, it also shows that there is a net loss of intertidal habitat and a large increase in deep water habitat by the late long-term.** The Bay-Delta is not currently limited in terms of deep water habitats, and some relevant historical experience suggests deeper off-channel habitats are likely to be more favorable habitat to exotic species than to natives, so an increase in the depth of restored habitats does not appear to be a desirable outcome. Thus the benefits attributed to creating the proposed habitat acreages may be quite optimistic. We look forward to providing technical assistance on this issue; a good start would be a more in-depth investigation of the expected depth distribution in potentially restored areas in the early and late long-term time periods.

> **ICF Response: The current HSI analysis actually indicates substantially more intertidal habitat than under existing conditions, although there is a decreases in the quality of this habitat as a result of the combined effects of climate change and the BDCP. It describes the depths of habitats that would be created in each ROA and the benefits attributed to the BDCP take these depths into account. However, ICF is currently in the process of analyzing the habitat units Delta-wide using more accurate modeling that incorporates depth based on RMA modeling so that a better comparison of the change in habitat type units can be made. This analysis will be reviewed with the agencies.**

***The effects analysis underemphasizes Bay-Delta water flows as a system-wide driver of ecosystem services to the San Francisco Estuary.** While climate and associated hydrology affect the magnitude of watershed runoff, system hydrodynamics downstream of the big dams (e.g., exports, OMR flows, X2, gate operations, etc.) are largely driven by coordinated water operations. All of these influence the habitats and population dynamics of listed species. It is critical that the BDCP effects analysis identify changes in operations that will importantly alter hydrodynamics, and address in depth the dependency of the ecosystem and its constituent species on flows. Reduction of flows (in full consideration of timing, magnitude, variability) is the most fundamental cause of stress and driver of change to the fishes and food web that have adapted to the tidal and freshwater mixing environment that is the Bay-Delta ecosystem. In addition, some of the other stressors listed and assumed to be addressed through the conservation measures are either directly or indirectly influenced by Delta inflows, exports, and outflows. Until the roles of flows and flow alteration, for which there is substantial literature, are adequately represented in conceptual models and developed in the effects analysis, we are reluctant to rely on its conclusions. We look forward to providing technical assistance on this issue as it is resolved.

> **ICF Response: As described above, ICF looks forward to working with the agencies to develop and apply conceptual models as appropriate. The current effects analysis does include analyses using over 30 different models and methods that link biological effects to changes in flows. Appendix C, Flow, Temperature, Turbidity, and Salinity, describes the methods, results, and conclusions of these models and Chapter 5 attempts to integrate these results into a net effects analysis on each species. ICF looks forward to working with the agencies to revise the net effects analysis to more clearly describe how these**

> **results interact with the results from other stressors and how together, they form the net effect on each species.**

***The Low Salinity Zone (LSZ) is a dynamic habitat defined by the tides and freshwater flow that requires a globally tailored conservation strategy.** It is widely recognized that estuarine habitat suitability is driven by the interaction of a flow regime with a brackish, tidally influenced landscape. Changing this interaction by reducing outflow can set a series of ecosystem changes in motion that degrade expected ecological services. In the Bay-Delta, both the flow regime and the landscape are highly altered, and the Preliminary Project proposes new changes. It is well established that variation in Delta outflow or X2 is correlated with many important ecosystem processes and the abundance or survival of estuarine biota. It is also well established that the most important mechanisms and seasons for species that use the LSZ vary. Chapter 5 does not directly grapple with the conservation implications of these and other relevant facts, arguing that the *mechanisms* causing flow effects on certain fish species are not "well-understood". But the phenomena of species-flow responses are well-developed in the scientific literature. Unless there are concerns about the adequacy of the underlying data, which there may be, flow relationships developed in the scientific literature should be used as the initial basis to predict the effects of changes in flow regime. The effects of flow regime on species and ecosystem processes in the LSZ have been an important subject of study for a long while, and, in addition to their role in the water operations consultations form part of the basis for regulatory processes underway or contemplated by the State Board and EPA. We look forward to working with our partners on resolving the framing of the LSZ habitat analysis.

> **ICF Response: While outflow is reduced in some months of some years, the biological meaning of these reductions is not always clear, even with the application of existing scientific literature. In the case of delta smelt, we attempted to address this uncertainty in Chapter 5 by including focused studies prior to the new intake operation as well as describing how adaptive limits could be used if needed, to increase fall outflows. We hope to continue discussions with the agencies regarding how to address the Fall X2 issues. However, it is important to understand that changes in water temperatures in the overall Delta are solely driven by atmospheric temperatures and therefore delta smelt will experience increased exposure to lethal and sublethal temperatures even without the BDCP. Regarding longfin smelt, we are looking much more closely as how outflow interacts with this species to more clearly examine the actual changes from the project and how those translate into a biological change. Similarly, we are taking into account the changes in migration flows for other species. We look forward to working with the agencies once we have information to share on this topic to discuss how the project might be modified to provide adequate flows. Additionally, we are currently working on a revised analysis of changes in turbidity.**

***The Low Salinity Zone (LSZ) is the primary habitat for delta smelt and the primary rearing habitat for larval longfin smelt and juvenile to adult splittail.** The Preliminary Proposal modeling indicates that Delta outflows during February-June will more frequently be near the minima required by the SWRCB under D-1641. This will represent a substantial negative project effect on longfin smelt. The effects analysis and Net Effects only partly address this issue, reporting that Preliminary Project is expected to provide a large, positive impact to food resources that will offset the negative impact to "transport flows". But there are multiple mechanisms by which Delta outflow can affect longfin smelt recruitment; transport flow is only one of them. Transport flows might be managed via gates or other engineering solutions. The other mechanisms for which there is stronger scientific support are kinetic energy mechanisms (low-salinity zone habitat area and retention from gravitational circulation in the estuary). The problems that reduced outflow creates by changing these processes do not have reasonable engineering solutions, and at present appear to be manageable only via outflow. Thus, although some of the potential impact of outflow reductions is reported, the analysis is too narrowly focused.

Both projected sea level rise and the Preliminary Proposal are also anticipated to cause the average location of X2 to move upstream during the summer and fall. The modeling indicates that intra-annual variability would be lost for several months in the late summer and fall in all water year types; even wet years would functionally become dry years for a third of delta smelt's life cycle. The effects analysis acknowledges this result, but the Net Effects concludes that habitat restoration and food web enhancement will greatly offset this loss of habitat value. The conclusion is in part speculation and in part does not reflect current scientific understanding.

This has several implications for delta smelt. First, under the preliminary project delta smelt habitat would less frequently lie in Suisun Bay and Marsh during summer and fall. The habitat suitability modeling shows that this would limit the capacity of tidal marsh restoration in the Suisun region to contribute to delta smelt production. Second, lower summer outflows would increase the length of time that seasonal delta smelt habitat constriction occurs and overlaps with physiologically stressful water temperatures. This means that more food production would be required to maintain current delta smelt growth and survival rates, even in areas where temperatures remain suitable. In areas where temperatures exceed physiologically suitable levels during the summer (~ 24°C), no amount of food production will increase growth or survival rates. Third, the restricted distribution of delta smelt during most summers and essentially all falls would increase the chance that a localized catastrophic event could pose a serious threat to the survival of the delta smelt population.

Turbidity is another important component of delta smelt habitat suitability. Section C.4.1.4 ("Turbidity") states: "[f]irm conclusions regarding changes in turbidity in the BDCP Plan Area are difficult to make." But some large-scale changes in sediment fluxes might affect turbidity on scales important to smelt, and should be straightforward to analyze. The Sacramento River is the most important contributor of sediment to the Bay-Delta. According to the Effects Analysis it contributes an estimated 80% of its load during high flow events. The North Delta diversions in the Preliminary Project have the ability to take up to 15,000 cfs during high flow events. For a 70,000 cfs event, this could be 20% of the Sacramento River water including its suspended sediment load. The effects analysis makes no attempt to analyze how much sediment loss per year that would represent and whether it would change the ratio of supply to loss of sediment from the estuary. The same calculations should be done for the south Delta to give the results full context.

In summary, the current Effects Analysis does not appropriately deal with critical issues involving the role of the Low Salinity Zone as habitat for longfin smelt, delta smelt, and splittail. Until it addresses the right questions regarding flow, LSZ location, and turbidity, we are reluctant to rely on its conclusions. We look forward to working with our partners as these issues are resolved.

ICF Response: Same response as above.

***There is no reason to expect that invasive vegetation will not proliferate in the East and South Delta ROAs, and no reason to expect a meaningful increase in south Delta turbidity if vegetation could be successfully controlled.** There should not be an a priori assumption that SAV can be controlled via ecologically sound methods in the east, central and south Delta. These are comparatively low turbidity, high vegetation areas already, under the existing hydrodynamic regime. There is nothing in the Preliminary Proposal that would dramatically change channel geometry, increase SJR flows, or increase sediment inputs that could be expected to change the turbidity of the entire southern half of the Delta.

ICF Response: A quantitative analysis is being undertaken to examine the issue of reduced sediment input from the Sacramento River in relation to the proposed north Delta intakes. ICF agrees that a full analysis of this issue would include consideration of inputs from other tributaries such as the San Joaquin River and will investigate the potential to do so following completion of the Sacramento River analysis.

***Chapter 5 is deficient in its descriptions of channel margin, riparian, and floodplain habitat restoration outside of Yolo Bypass.** The Yolo Bypass tends to benefit native fishes because (1) it floods frequently with major inundation events; (2) it floods during times of year that BDCP target fishes can, and have evolved to, use it; and (3) upon drying it leaves very little permanent habitat for non-native fishes to colonize and reproduce in, because most non-native fishes are late spring/summer spawners. The original habitat analysis attributed seasonal floodplain benefits along the San Joaquin River that we do not believe are plausible; however, we understand there is now general agreement on this point and we will not comment on it further. However, the Sacramento River from Sacramento to about Rio Vista is also highly constrained, in this case by levees rather than regulated hydrology, and there are strict flood control capacity requirements that are enforced by USACOE. The effects analysis does not describe how this constrained reach of the river can support the proposed changes, where they will be, or assess their feasibility.

ICF Response: The January 2012 version of the habitat appendix (Appendix E) did not include an analysis of floodplain or riparian habitat restoration benefits. The revised appendix will include this analysis. If the comment is in reference to the conservation measures themselves, we can work with the agencies to identify more specific areas where this restoration can occur to both benefit covered fish and avoid interruption to the flood control system.

***Increased residence times and reduced flushing of the Delta by Sacramento River water appear likely to result in interior-Delta channels that are further dominated by agricultural runoff, invasive aquatic vegetation, warmer temperatures, and increased algal productivity with its associated dissolved oxygen swings.** These environmental conditions favor non-native/invasive species (e.g. *Egeria densa*, largemouth bass, water hyacinth, *Microcystis*) and disfavor native fishes. The Delta is already more biologically similar to a lake than it once was, due to the historical accumulation of human modifications. We expect that by reducing Delta flows, the Preliminary Project would likely facilitate the spread of habitat conditions that are unfavorable to delta smelt, and and less favorable to other target fish species survival and recovery.

> **ICF Response: There may be potential adverse effects related to increased residence times, but there may also be beneficial effects such as increased production of phytoplankton that result from increased residence times. ICF will consult with DWR and the fish agencies in order to determine the elements needed to produce a more robust characterization of potential changes in estuarine habitat that may be caused by residence time changes.**

Issue Area 3: The Effects Analysis relies on selective use and interpretation of statistical and mathematical models

***The effects analysis did not use the available splittail life cycle model at all to support its Net Effects conclusion.** There is a published stage-based life cycle model for splittail where the effects of various environmental variables were examined for their effects on long-term trajectory of population abundance. This model helped frame the preferred time-interval for floodplain activation necessary to ensure splittail persistence in the Central Valley. This available approach to an Effects Analysis for a listed species of native fish was not discussed in the present Effects Analysis.

> **ICF Response: This comment was provided to ICF on the life cycle models appendix in December 2011. This appendix is currently under revision and we are currently working on how to incorporate more and better models for all of the covered fish species, including the splittail model. However, many of the life cycle models currently available are limited in their application to the effects analysis because they cannot easily (or at all) incorporate a changed configuration of the Delta, as is proposed by the BDCP. ICF agrees that the splittail model developed by Moyle et al. (2004) was a useful effort to characterize population dynamics of the species. However, aspects of that model pose significant challenges for its use in the BDCP effects analysis. As noted by Moyle et al. (2004: 36), "While the model can be made to simulate population dynamics that mimic the natural situation, actual numbers for mortality and survival rates are lacking for the most part, so it is hard to distinguish among various sources of mortality." To our knowledge, such data shortcomings remain for splittail.**
>
> **Further, Moyle et al. (2004: 37) noted that the ability of the model to estimate consequences of entrainment loss of splittail at the south Delta pumps would require the model to be sectored into spatial segments. South Delta entrainment is an important example of a stressor that would be changed because of BDCP. The required model restructuring would be a substantial effort that may be challenging to apply, given the lack of proportional entrainment loss estimates for the species. We also note that the model uses year type as a proxy for changes in spawning habitat availability and therefore does not estimate the effect of increasing inundated floodplain acreage at a given flow, which is a potential important effect of BDCP with respect to splittail.**

> **In conclusion, ICF will acknowledge the model's main findings in the life cycle models appendix and the net effects analysis; however, without significant additional efforts to further refine the model, its application in the effects analysis would be challenging and is unlikely to alter the main conclusion for this species, i.e., that BDCP has the potential to produce substantial benefits based primarily on increased floodplain availability. However, ICF will continue to investigate the applicability of the splittail model, and other life cycle models, for use in the effects analysis.**

***The effects analysis did not use the best available longfin smelt statistical models to support its net effects conclusion.** The newest published statistical analyses of longfin smelt are quasi-life cycle models that account for prior abundance and spring flow influences (among other factors) on this species. These models were discussed and discounted as not being 'life cycle models'. Dismissing them because they are not 'life cycle models' is unhelpful: they are the best available scientific tools to evaluate project effects on longfin smelt. The older regression models that were used in the effects analysis are published, but can easily be shown not to perform as well as the newer models. The older models also average the flow influence on longfin smelt across half a calendar year, which likely affects conclusions about the reduction in springtime outflow seen in modeling outputs for the Preliminary Proposal. We look forward to working with our partners and providing technical assistance as this issue is resolved.

> **ICF Response: In developing the effects analysis for longfin smelt consideration was given to three types of statistical approaches that included (1) simple linear regression analyses that had previously been published in the literature depicting relationships between average Delta outflow and/or X2 location during the late winter and spring months and subsequent indices of fall abundance; (2) more sophisticated statistical analyses of various potential covariates on indices of longfin smelt abundance at various life stages; and (3) statistical lifecycle models depicting the effects of various covariates on the abundance and survival of longfin smelt over their lifecycle. In reviewing the various approaches and supporting data and information consideration was given to using multivariate statistical analyses such as those developed by Thompson et al. (2010) and Mac Nally et al. (2010). These analyses, however, focus on individual life stages of longfin smelt and do not reflect the species lifecycle. In addition, these statistical models include a number of covariates (e.g., indices of zooplankton food supplies) that have been based on prior monitoring but are uncertain in the future. The effects of BDCP actions such as changes in hydrodynamics can be predicted using existing tools such as CALSIM which are compatible with the simple outflow vs. abundance relationships and were used in the analysis. Analytical tools are not available to predict the response of many other covariates such as the ability of BDCP tidal habitat restoration to produce quantitative estimates of zooplankton densities available as a food resource for various lifestages of longfin smelt in the future. Similarly, although there are currently several efforts underway to develop statistically based lifecycle models for longfin smelt no lifecycle model exists that could be applied to the Effects Analysis. In addition, many of the lifecycle model statistical analyses also include various covariates that will be difficult to predict as a response to BDCP**

conservation actions. As a result of the difficulties in predicting many of the needed covariates that may respond to BDCP actions in the future and the lack of a lifecycle model that could be applied to the longfin smelt analysis these approaches (2 and 3 above) were not used in the longfin smelt effects analysis.

We agree that additional consideration can be given to refining the simple regression approach by using more biologically meaningful seasonal time periods, alternative sample data to develop indices of population abundance (e.g., CDFG bay otter trawl collections), and other refinements to the statistical tools. Further consideration can also be given to the use of focused sensitivity analysis of the multivariate covariate statistical tools to inform the range of uncertainty in the effects of various levels of response of zooplankton populations and other covariates to future BDCP actions. Further consideration can also be given to various focused experimental and monitoring efforts that could be implemented in the near-term to provide better information on the response of longfin smelt and other covariates to changes in environmental conditions such as Delta outflow. Consideration can also be given to results of longfin smelt lifecycle model analyses in the near-term when they become available and can be used as a basis for further qualitative and quantitative analysis of the potential response of longfin smelt to BDCP actions. We look forward to participating in discussions with BDCP partners to discuss these and other potential analyses and additional data collection efforts that can be conducted over the near-term to improve the Effects Analysis for longfin smelt.

***The effects analysis continues to insist on an analytical approach to entrainment that does not reflect the best available science.** The current Draft Effects Analysis (as of September 13, 2011) downplays the potential effects of entrainment to the delta smelt population: (e.g., Section B.1.1.1), "[H]owever, analyses to date have not found correlation between entrainment and population level responses of delta smelt ..." The delta smelt population is now at historically-low abundance and population losses due to entrainment may have significant population effects depending on their magnitude and frequency. While it is true that some regression-based analyses have failed to reveal an export affect to the delta smelt population, other approaches that more effectively investigate the role of fish distribution to entrainment have revealed an important relationship between water operations and the risk of population-level entrainment effects to delta smelt. Kimmerer (2011) demonstrated that entrainment losses averaging 10% per year can be "…simultaneously nearly undetectable in regression analysis, and devastating to the population." We look forward to working with our partners to ensure that the best model-based analyses of proportional entrainment for both South- and North-Delta diversion facilities are brought to bear to resolve this issue.

ICF Response: The revised appendix (March 2012) incorporates a regression that reflects the FWS's 2008 approach while also adjusting per Kimmerer 2011. Additionally, the appendix more clearly describes the various studies that have been performed relative to the relationship between entrainment and delta smelt. While there are no statistically established links between delta smelt abundance and entrainment, there is an appendix dedicated to entrainment in which delta smelt are thoroughly analyzed.

***We think that the delta smelt state-space model is a useful framework to explore hypotheses about what drives delta smelt abundance.** However, the Maunder-Deriso model is a new application that needs additional collaborative work before it reaches maturity. We are concerned that the present model may have identifiability problems, as we discussed in our technical comments last fall. Until that concern is resolved, we are unsure whether the parameter estimates developed in that model represent what they are described to represent. We are also unsure why the model uses the official DFG Fall Midwater Trawl Abundance indices for delta smelt, but does not use the official DFG Summer Townet Survey or 20 mm Survey abundance indices. The rationale for this (which may be simple) is not explained. The model also assumes a specific form of density dependence between generations. We have questioned the appropriateness of this choice, because on very thin ground it limits the universe of plausible explanations for delta smelt reproductive success that can be derived from the model.

The intent of this new model was to explain a specific historical dataset, and other than some broad assumptions it does not contain much of the mechanism presented in current delta smelt conceptual models (like DRERIP, or POD conceptual model, or the Fall Outflow Adaptive Management Plan conceptual model). The published version of the model used data through 2006. The model was updated for the Effects Analysis to include data through 2010. When this was done, the model fit deteriorated dramatically relative to what was reported in the paper. While this does not (at all) cause us to think it should be discarded, it does underscore questions about the maturity of the tool. The current model's success in fitting a specific set of historical data may not translate to good predictions of the the effects of flow and habitat change. The current model may perform still more poorly when CALSIM II water operations outside the envelope of historical experience are used as input.

It is important for the Effects Analysis to acknowledge that some data that may prove to be essential to modeling delta smelt or longfin smelt dynamics have been collected only recently. There are a number of studies now underway that address questions about fall outflow processes and delta smelt ecology as a whole. The novelty of the Maunder-Deriso model, and existence of other tools and analyses taking a process-oriented approach to to predicting the effects of flow and habitat changes, make the framing of the effects analysis very important. It is equally – possibly more – important that uncertainty at all levels be properly developed and acknowledged. Achieving these things, which are important to having an effects analysis we can rely on, will require work and a willingness to adapt on the part of ICF. We look forward to continuing to work with ICF and our other partners to ensure that the best science is identified and used defensibly in the effects analysis.

ICF Response: ICF is currently coordinating with the model developers, to establish a mechanism for further review of this model as well as running sensitivity analyses to better inform the effects analysis. ICF also agrees that new information is constantly emerging that must be incorporated into the effects analysis, but just as important-or more so-must have a framework for incorporation into implementation of the BDCP. ICF looks forward to developing that framework in collaboration with the agencies to ensure a process is in place to utilize new information to benefit covered species throughout the life of the plan.

<u>Issue Area 4: The BDCP's net effects conclusions rest on an equivocal food web conceptual model</u>

***The FWS agrees that the pelagic food web that historically supported greater abundance of estuarine fishes including longfin smelt and delta smelt has been impaired and that contributing to its restoration is a key component of a conservation strategy for the Bay-Delta. However, food limitation is a ubiquitous feature of ecology in the Bay-Delta.** It affects non-native species as well as the BDCP target species. Thus, the issue is not really "food limitation" *per se*. Rather, the issue is food web pathways and the number of steps in a food chain between primary producers (phytoplankton and plants) and the BDCP covered fishes. For the smelts, the desired food pathway would be dominated by this short food chain: diatoms → calanoid copepods and mysids → low-salinity zone fishes. The short food chain outlined above dominated the historical low-salinity zone food web. Longfin and delta smelt are highly dependent on it (and minor variations of it). The other BDCP target fishes also use it, but have more generalized diets that often include benthic organisms and riparian and floodplain insects. The draft appendix has a very long section on food web changes when a simpler summary of the major points would be more effective.

The focus of food web restoration in the effects analysis is on floodplain and tidal marsh restoration. The production of diatoms may have been limited by disconnecting floodplains from their rivers and by reclaiming tidal marshes. These are the primary hypotheses behind the BDCP habitat restoration conservation measures. However, the two best-substantiated drivers of diatom suppression are overbite clam grazing and ammonium concentrations in the estuary. The suppression of diatoms is hypothesized to have provided a competitive advantage to lower quality primary producers and primary producers like *Egeria densa* and *Microcystis* that have virtually no food web value to the BDCP target fishes. This change in the base of the food web has reduced the amount of fish production that can be supported by the historical diatom-based food chain, and forced the fish to rely on other longer and more energy-limited food pathways. Longer food chains are less productive, and do not support as many fish. Because splittail and young Chinook salmon are the covered species that most extensively utilize floodplains and tidal marsh networks, they should be expected to gain the greatest food web benefits that restoration of these habitats can provide. However, this is not what the Net Effects concluded. Rather, it concluded that habitat restoration would provide *greater* benefit for the smelts despite their limited overlap and more restricted diets.

Shortcomings in the Net Effects resulting from mischaracterization of processes limiting transfer of production in new habitat areas to native fish biomass renders the present analysis inconsistent with best available science, and we are reluctant to rely on it to judge the design of the preliminary project. As with other modeling issues, we look forward to working collaboratively with our partners as these issues are resolved.

> **ICF Response: The key role of clams on the delta food web is discussed extensively in the effects analysis. The role of ammonia is acknowledged though its source is presently beyond the scope of the BDCP. The discussion of food web effects is currently under revision and will include discussion of these issues to be as clear as possible how these factors interact with the food productivity potential for BDCP. In regard to the differences in food selection between smelt and salmon and the relative differences in response in the HSI, it is important to**

regard each of the HSI models as independent---comparisons of response between species is not appropriate as rating curves have been established for each species independently and there is no attempt to calibrate response between species. The limitations of the existing analysis are acknowledged in the effects analysis.

While multiple observers have noted the desirability of a more complete food web analysis, no method has been advanced. Most observers acknowledge that the limited quantitative analysis in the effects analysis must be combined with a firm qualitative analysis and discussion that addresses these limitations and reaches qualitative conclusions regarding the benefits of restoration on the food web.

Issue Area 5: The analysis and interpretation of BDCP are hindered by indeterminate model baselines and related issues

***A key point of continuing analytical confusion is the use of multiple baselines.** The current set-up for the BDCP employs two 'base case' model runs (EBC1 and EBC2). The EBC1 does not include the full suite of elements in the current FWS and NMFS OCAP RPAs. The EBC2 attempts to include the RPAs in their present-day form, but it does not accurately capture them all. There are numerous cases in Chapter 5 where it is not clear what Project model result is being compared to which baseline condition. This generates confusion. We look forward to continuing to work with our partners to be sure that baselines used in the effects analysis are appropriately constructed and are used clearly and correctly.

ICF Response: Please clarify what is meant that not all components of the RPA are 'adequately captured' in EBC2. Also, ICF will strive to be as clear as possible regarding which baseline is used for comparison in Chapter 5 and its appendices.

***CALSIM II demand representation in 2060 studies should have some justification.** Some explanation for, or error estimate of, assuming a 2020 level water demand for a 2060 climate change simulation should be made. Presumably portions of the State (Southern California, the American River Basin, etc.) are going to continue to grow through 2060. Some estimate in the change of cropping patterns over the 40 years (2020 – 2060) should also be made (or at least a write-up of why it cannot be made) should be included. Without clear resolution of this issue, it appears to us that the modeling may underestimate water demand in the late long-term. We are unable to provide technical assistance on this issue, but look forward to its resolution.

ICF Response: a. The water demands in the CALSIM II model are based on 2030 projected level of development. The Sacramento Valley hydrology used in the CALSIM II model reflects 2020 land-use assumptions associated with Bulletin 160-98. The San Joaquin Valley hydrology reflects draft 2030 land-use assumptions developed by Reclamation.

The water demands for Late Long-Term (Year 2060) conditions are assumed to remain the same as those at 2030 level in CALSIM II, because the demands assumed for 2030 level of development in CALSIM II for both CVP and SWP are already at the full build-out for the contracts. CVP and SWP agricultural demands south of the Delta are fixed at maximum contract amount and do not

> **vary year to year. Full Table A water demands are assumed for the SWP contractors. Also, full water rights are assumed for the water rights holders to account for future growth. In CALSIM II, only the demands assumed in the north-of-the-Delta are dependent on land-use as reported in the Bulletin 160-98. The implications of the resulting changes in the water deliveries on the current land use are evaluated in the Socioeconomics chapter of the EIR/EIS.**

***The proposed restoration in each "Restoration Opportunity Area" (ROA) is only compared against the lands bounded within the ROAs, which themselves lie in larger regions.** These comparisons of present-day ROA habitat to future ROA habitat are inappropriate – especially in cases like the east and south Delta ROAs, which are currently dry land. Mathematically, if a terrestrial habitat is subsequently flooded, the improvement for target fishes increases by an infinite percentage even if the habitat performs poorly because a habitat suitability index that is even a tiny fraction of 1 is still infinitely higher than zero, which is the suitability of dry land to fishes. Habitat analyses need to be based on comparisons against currently available aquatic habitat acreages in the entire regions containing the ROAs. They also need to be synthesized and integrated into Plan Area-wide totals, with river flow and climate changes incorporated, in order for the analyses to be meaningful.

> **ICF Response: ICF is currently revising the HSI modeling to include the entire Plan Area to address this concern. We have been working with the agencies to also adjust the curves used for each species and together, these refinements should substantially improve the analysis of habitat restoration.**

NMFS List of Issues Unresolved in BDCP Administrative Draft

(4/2/2012)

- **Hood Diversion Bypass Flows**

The Effects Analysis of the Preliminary Proposal (PP) raises concerns over reduced flows downstream of the North Delta diversions, especially in winter and spring months. These flows relate to:

A. Increased frequency of reversed Sacramento River flows at the Georgiana Slough junction. The January 2010 PP rules included a provision that north Delta pumping would not increase these reverse flows. Calsim II results provided by CH2M-Hill indicate that the PP will increase the percent of time Sacramento River flows are reversed, causing increased entrainment of juvenile salmonids into the Central Delta. If the frequency of reverse flows increases due to the PP, then the diversion amounts allotted under the PP could not be implemented. The DSM2 analysis of reverse flows in the DPM suggests that tidal marsh restoration in the Delta will nearly offset both the effects of sea-level rise and large water diversions from the Sacramento River, a conclusion which needs much more explanation in the EA (see comment on tidal marsh effects).

B. Long-term viability of sturgeon populations. There are concerns that Sacramento River flow reductions will impact the reproductive success of white and green sturgeon, which have been documented to produce strong year classes mostly in years with high flows in April and May (AFRP study). We do not know if this has been addressed in revised Appendix C.

1. Further explanation and analysis of the reverse flow issue.

2. Work with the Services to find a diversion scheme that is still likely to be permittable after adequate modeling and analysis has been conducted.

> **ICF Response: We agree and will work to better explain this issue and work with the fish and wildlife agencies to find a diversion scheme that can move the project forward.**

- **Salmonid Net Effects**

All salmonid species are grouped together, with no separate evaluations for the separate ESUs of Chinook salmon or for steelhead. It is important for the net effects analysis to describe individual ESUs/species, and provide full consideration of the life-history diversity and timing exhibited by each ESU/species. We also need the Sacramento River populations and San Joaquin populations for Spring-run Chinook, Fall-run Chinook, and Central Valley steelhead summarized by river basin, prior to the roll-up by ESU/DPS. Steelhead life-history and ecology especially warrant a separate evaluation. “Net effects” is useful for comparing alternative operations, but will not provide the robust effects analysis needed for ESA purposes (see comment on ESA baseline).

Separate all Chinook by ESU, by San Joaquin and Sacramento populations, and separate steelhead in all analyses and discussion.

ICF Response: We agree as was noted in the Chapter 5 Admin Draft. We plan to work closely with the fish and wildlife agencies to develop separate analyses for each salmonid run and, where appropriate, each population.

- **ESA Baseline, Future Conditions, and Climate Change**

In order to conduct the ESA jeopardy analysis on the PP, the baseline condition and projections of future baseline conditions, including effects of climate change, need to be re-written to be consistent with the 2009 Biological Opinion and current case law. ESA regulations define the environmental baseline as "the past and present impacts of all Federal, State, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process." Implicit in this definition is a need to anticipate the future baseline, which includes future changes due to natural processes and climate change. For the ESA jeopardy analysis we add the effects of the proposed action to the environmental baseline to determine if there will be an appreciable reduction in the likelihood of survival and recovery of the species (by reducing its reproduction, numbers or distribution).

Upstream effects associated with climate change need to be in the baseline and future conditions, with any effects of the project (in the Delta or associated with upstream operations) added to that future condition to determine jeopardy. A project proposed in this type of baseline conditions needs to more than offset its effects in order to alleviate a jeopardy finding.

ICF Response: This is an issue that legal staff from the fish and wildlife agencies should address with DWR legal counsel. It is critical that this issue be resolved quickly because of its implications for the effects analysis.

- **Analysis of Water Temperature Impacts**

Lethal and sub-lethal water temperature thresholds need to be examined at a finer scale. Currently the effects analysis relies heavily on a Reclamation water temperature model which can only estimate monthly values, which have limited value for predicting project effects on fish. In addition, the effects analysis has only presented frequencies of temperature threshold exceedances, while the magnitude and duration of exceedance is also very important. We do not know if this has been addressed in revised Appendix C.

1. Provide tables and probability plots of magnitude and duration of temperature exceedances at certain upstream locations, by water year type and month.

2. Technical discussion with Reclamation and CH2MHill about how to post-process data.

3. Investigate the use of SWFSC's Sacramento River temperature model to predict project effects and make hindcasts of empirical temperatures.

4. Investigate the use of the new American River temperature (and storage and flow?) model

ICF Response: Regarding the additional temperature exceedences analysis, this comment was also made on Appendix C and we would like to further discuss how this analysis would contribute to the overall net effects analysis and how it could be done in a way that is clear and useful. ICF will work with the agencies to determine the potential application of the SWFSC and American River models.

- **Assumption of Habitat Restoration CM Success**

In several places, the EA assumes that adverse impacts of the PP will be offset by unsubstantiated benefits of habitat restoration. The EA assumes that all restoration will be successful and work as predicted, with little or no evidence to support this prediction and no attempt to analyze the potential outcomes of less than perfect success.

1. It is imperative to avoid language such as "This conservation measure will...", because the anticipated CM outcomes are based on conceptual thinking, not execution. To be able to comprehensively think through the adaptive management and monitoring plan, implementers need to try to anticipate a range of responses that must be managed in order to be prepared for the uncertainty of the response.

2. Alternative outcome scenarios should be evaluated to bracket the range of possible outcomes from proposed habitat restoration.

ICF Response: We can be clearer about the assumptions that create the foundational analysis of habitat restoration benefits. With little empirical data, no site-specific plans, and a long-term planning period, even ranges of potential outcomes would not provide more meaningful analysis. However, we can be clearer that a range of outcomes can be expected, and develop the adaptive management plan, including monitoring and research, to address those outcomes. We can work with the agencies to describe what that potential range may be. We can also clarify that the success of restoration effects is expected to increase over time as more projects are implemented and we learn from each project.

- **Overreliance on Real-time Operations and Adaptive Management**

In several places, the EA assumes that adverse impacts of the PP will be fully resolved through the implementation of real-time operations and adaptive management. This may not always be possible. For example, long-term trends towards reduced carryover storage may not be able to be mitigated using real-time operations. How adaptive management might work in this situation has not been fully assessed. There are going to be limitations on what adaptive management and real time operations can accomplish.

Examine recent (five to ten years) real-time management of the cold water pool in Shasta Reservoir to determine both the effectiveness of real-time operations and a range of adaptive management options.

ICF Response: We agree that recent years can be evaluated to determine how well cold-water pool and temperature standards in upstream areas can operate. Additionally, we propose exploring the inclusion of upstream temperature

controls in the modeing of the effects analysis to reduce uncertainty of these effects and to offset CALSIM's modeling approach to better reflect the actual operations of the project.

- **North Delta Diversion Effects**

  Mortality rates from predation and other screening effects are difficult to predict, as there is a high level of uncertainty associated with predation and other effects on juvenile salmonids. The estimate of <1% loss at all 5 screens is not sufficient without giving additional consideration to higher estimates of mortality (GCID empirical studies showed a 5% per screen loss rate, much higher than the <1% used in the DPM).

  1. Bracket the analysis of screen related mortality around a 5% per screen loss assumption.

  2. Investigate the use of DWR's hydrodynamic model to assess local flow alterations at the proposed diversion structures, including the creation of predator holding areas. Specific questions are whether the model can simulate on-bank structures and the additional hydrodynamic effects of active pumping.

  ICF Response: We would like to review and discuss with you the empirical data from GCID to develop the appropriate range of predation that should be evaluated for the north Delta intakes.

- **Predator Control Conservation Measure**

  We agree that predation is a significant risk factor to the listed species, but the assumed positive results of this CM are questionable and unsupported **(see F.5.4.1.4 in Appendix F).** As an example, localized control of striped bass may not be feasible as this species exists throughout the Plan area and are highly mobile. Few specific details have been presented on how the CM will be implemented, and an aggressive predator removal program could result in significant incidental take of listed species. Due to the high level of uncertainty, we find it very unlikely that we could rely on this measure for any benefits during the permit process.

  Remove this CM measure from the plan, and move it to an experimental research program and link to adaptive management. Reflect this appropriately in the EA.

  ICF Response: We propose discussing with the agencies which areas are most important targets for predator removal and further develop a description to reduce uncertainties about its effectiveness in those key areas.

- **Delta Passage Model**

  DPM is used as the sole predictor of smolt survival in baseline and PP scenarios. However, the assumptions, inputs, and results are still being validated and reviewed. The datasets used in this model are very limited and largely based on results from hatchery late-fall run Chinook, which are then being applied to other runs of Chinook.

  Continue refinement and development of DPM. Weigh validity of results against those of other models and relationships. The use of Newman, 2003 may be another tool to use for assessing the survival of fall and spring run smolts through the Delta.

ICF Response: We agree and appreciate the collaborative nature in which we've been working to move this analysis forward. We will investigate the use of Newman 2003.

- **Deficient Analysis of Fry Passage/Survival**

Because the DPM model is only for smolt sized fish, the salmonid analysis is insufficient as it provides no information on fry-sized salmonid passage/survival.

Add qualitative analysis of fry survival based on best available data. Perhaps add time/added mortality to a modified version of an updated DPM model.

ICF Response: We agree and recently submitted the revised Flow Appendix (Appendix 5.C), which included a new model (Yolo Bypass Fry Growth) for analyzing the differences in survival and growth among scenarios. Although this model is currently specific to fall-run Chinook, it could be expanded to all salmonids. ICF looks forward to continued collaboration on this effort.

- **PTM Runs Inadequately Capture Altered North Delta Hydrodynamics**

PTM model runs did not include conditions in which ND diversions would be at the upper limits of allowable pumping (high proportion of total river flow). The technical memo from NMFS and USFWS highlighted the issue and the resolution to the problem. We will need additional modeling runs to adequately assess ND diversion impacts on salmonid travel time and route entrainment.

Do additional PTM analysis following guidelines outlined in NMFS/USFWS memo.

ICF Response: We plan to work with the agencies to develop more informative PTM runs for this issue as well as others in the north Delta subregion (i.e., agricultural diversions).

- **D1641 Export/Inflow Ratio**

Combined north and south Delta exports under the PP exceed the current D-1641 Delta Export/Inflow standard. (The PP calculation method measures Sac River inflow below the North Delta diversions and does not include ND diversions as part of total exports).

1) Provide summary analysis of differences between PP and EBC by month and water year type using alternate E/I calculations.

2) Show resulting flow data for both calculation methods.

ICF Response: We will work with the agencies to develop this analysis.

- **Yolo Bypass**

Yolo Bypass has great potential for fisheries benefits, but the current EA may be overstating the benefits without adequate studies or data to support these conclusions. Without project specific plans to help quantify the effects, concerns remain about issues such as sturgeon passage, juvenile salmonid survival under lower flow regimes, ability to get juveniles into the floodplain through notch and reduction of flows in the mainstem

Sacramento River to accommodate additional flooding in Yolo Bypass. Also, some races/runs of salmon may not have access to Yolo Bypass.

Provide project specific plans and consider the risks of managing the floodplain under lower flows related to issues above.

ICF Response: Project-specific plans for the bypass have not yet been developed, but through adaptive management, coordination with agencies during permitting and design, and maintenance, the uncertainties associated with CM2 can be reduced. Additionally, we propose exploring a sturgeon rescue program as part of this CM to ensure reduced uncertainities.

- **Channel Margin Habitat**

  Altered flows resulting from the North Delta diversions may result in reduced water levels affecting the percentage of time that current wetland and riparian benches are inundated.

  Compare anticipated water levels under future scenarios with those in the design documents of restored wetlands and riparian benches to analyze potential dewatering of those features.

  ICF Response: We agree and this analysis is included in the revised Appendix C.

- **Construction and Maintenance Impacts**

  The EA does not adequately address the potential for adverse impacts on sturgeon, fall-run Chinook adults, and steelhead adults, which are generally present in the project area during the proposed in-river work windows described for construction and maintenance of North Delta facilities.

  Discuss ways of minimizing impacts and implementing mitigation for species not protected by work windows.

  ICF Response: We can discuss additional methods for minimization besides restoration.

- **Tidal Marsh Impacts on Riverine Flow**

  The effect analysis assumes that restored tidal marsh will act to decrease flow reversals, which has not been well explained. It seems that tidal marsh restoration was modeled as a single configuration; there has been no description of that configuration to indicate how they were implemented in the hydrodynamic models. Therefore, there is a lot of uncertainty regarding model results.

  Document changes to hydrodynamic models that were implemented to characterize tidal marsh restoration.

  ICF Response: While some information will not be made available to the public, and therefore won't appear in the EA, we have substantially expanded the discussion of assumptions and modeling efforts used for the analyses. Some of this information is in the revised App C.

- **Cumulative Effects Show Long-Term Viability Concerns for Salmon**

  The analysis indicates that the cumulative effects of climate change along with the impacts of the PP may result in the extirpation of mainstem Sacramento River populations of winter-run and spring-run Chinook salmon over the term of the permit.

  1) Incorporate operational criteria into the PP that will protect and conserve suitable habitat conditions in the upper river for the species under the 50 year HCP (these operational criteria should be designed to meet the performance criteria in the NMFS BiOp RPA).

  2) Convene a 5-agency team of experts specialized in Shasta operations and temperature management to develop the above described operational criteria.

  ICF Response: As NMFS and others have pointed out, the projected adverse temperature regimes under both existing conditions and the PP in early and late-long term, are unlikely to occur under current real-time operation practices. As a result, the potential cumulative effect of both climate change and the project may be misinterpreted. As described above, we would like to discuss the inclusion of temperature controls in the modeling, which would likely eliminate or substantially reduce the actual likelihood of BDCP contribution to extinction, and may help to offset some of the climate change effects under some circumstances.

- **Holistic Estuarine Evaluation**

  The effect analysis should examine synergistic and cumulative ecological impacts associated with reducing inflows to an estuary that is already severely degraded, and discuss the importance that water quantity, quality, and the natural hydrograph have to the ecosystem, as well as the direct impacts on native fish species. So far, the impacts to fish have mostly been examined in a piecemeal fashion (e.g., examining impacts of flow reduction on adult homing).

  Incorporate a holistic evaluation of impacts on the estuarine ecosystem. Include discussion of the importance of water quantity, quality, and the natural hydrograph to the ecosystem, and the direct impact that changes to these conditions have on native fish species.

  ICF Response: We would like to discuss this comment and request additional detail about what is meant. The net effects analysis is an attempt to weave together all of the various effects on the species, including the interaction of various effects. For example, we examined how changes in the location of the low salinity zone could affect distribution of delta smelt and how that would change their exposure to microcystis. Likewise, the hydrodynamic modeling integrates the changed tidal exchange based on a restoration configuration. Because the net effects analysis is qualitative in nature, our ability to evaluate synergistic or interactive effects is mostly limited to pairwise comparisons or linear sequences of effects, as opposed to multivariate effects. However, we agree that more information can be developed to demonstrate the similarities and differences between the project and historical conditions. If the fish and wildlife agencies

could provide examples of the synergies that are missing from the effects analysis, we could focus on those effects.

- **Burden of Proof**

  Deference should be given to known population drivers and documented relationships (e.g., sturgeon recruitment relationship with flows is well documented, though the exact mechanism is not completely understood). Since flow is a key component of habitat for aquatic species, do not assume that it can be substituted for by other actions.

  Do not assume that incremental benefits in a conservation measure will compensate for known population drivers related to flow.

  ICF Response: The analysis considers all of the potential effects together to determine the total effect on the species. We can work with the agencies to determine how to weight different analyses or include an improved description or justification for certainty ratings.

- **Incomplete Analyses and Documentation**

  The full appendices were not released concurrently with Chapter 5 which makes review of the results problematic.

  Provide all appendices/analysis simultaneously so Services can have all pertinent information used in Effects Analysis summaries without having to backtrack weeks later.

  ICF Response: We have been coordinating with the agencies to develop these revisions and expect that these revisions address agency concerns. A revised Appendix C was released on 4/13/12 and the revised Appendix B was released on 3/30/12.

- **Insufficient Biological Goals and Objectives**

  The conservation measures are sometimes defining the BDCP species objectives, which is insufficient. 30% juvenile through-Delta survival is not a suitable goal for a 50 year conservation plan.

  The BDCP objectives should be biological, species-level outcomes.

  ICF Response: We are coordinating with the agencies to refine the BGOs.

- **OMR Flows Unimproved in Drier Water Years**

  Improved OMR flows under the PP occur during wetter years when OMR is less of an issue for covered fish. PP OMR flows are often worse than, or similar to, EBC in drier years. Sacramento Basin fish are most vulnerable to entrainment into the central Delta in drier years when Sacramento River flows have the potential to reverse and OMR levels are below -2,500 cfs. San Joaquin basin fish are best protected by increased Vernalis flows and/or a HORB which the PP does not address.

  1. Analyze the risk in different water year types and with different flow levels in the Sacramento River.

2. Implement Scenario-6 to help address the adverse impacts seen under the PP.

> **ICF Response: We are working on a proposal to address water operations issues, including those occurring in dry years in the south Delta. We plan to coordinate with the agencies soon.**

- **Non-Physical Barriers**

  Assessment of non-physical barriers is inadequate, and the potential negative effects of predation associated with non-physical barriers haven't been assessed.

  Include analysis of potential adverse effects of non-physical barriers.

  > **ICF Response: Appendix F includes this analysis, but we agree that additional information could be gleaned from the HORB and Georgiana Slough studies.**

- **Carry-over of OCAP RPA's on technological improvements to the South Delta Facilities**

  By not carrying forward technological fixes in the South Delta called for in the OCAP RPAs into the Conservation Measures, we would expect the effects analysis to specifically flag this and analyze it as a degradation to future conditions (as compared to the baseline which should include the RPA improvements) .

  Add south Delta technological improvement RPA's to Conservation Measures

  > **ICF Response: We will work with the agencies to determine how to integrate this into the project.**

- **Feasibility of 65K acres of Habitat Restoration**

  Recent evaluation of land available for habitat restoration indicates potential roadblocks to acquiring all the land proposed in the PP. DWR's own analysis suggests that 65K acres is very unlikely.

  Analyze the potential effects of partial implementation of habitat restoration and incorporate alternative actions or measures to compensate for this possibility.

  > **ICF Response: We believe that 65,000 acres of tidal restoration is feasible based on recent assessments. We would like to discuss these results with the fish and wildlife agencies and ways to improve the documentation to demonstrate feasibility.**

BDCP1673

SAN FRANCISCO ESTUARY & WATERSHED SCIENCE
Sponsored by the Delta Science Program and the UC Davis John Muir Institute of the Environment

Title:

Ecosystem-scale Selenium Model for the San Francisco Bay-Delta Regional Ecosystem Restoration Implementation Plan

Journal Issue:

San Francisco Estuary and Watershed Science, 11(1)

Author:

Presser, Theresa S., U.S. Geological Survey
Luoma, Samuel N., John Muir Institute for the Environment, University of California, Davis

Publication Date:

2013

Publication Info:

San Francisco Estuary and Watershed Science

Permalink:

http://www.escholarship.org/uc/item/2td0b99t

Acknowledgements:

This work was supported by CALFED (in the project's earlier phases), the U.S. Geological Survey National Research and Toxics Substances Hydrology Programs, and the U.S. Environmental Protection Agency (San Francisco, California, Region 9). Jeanne DiLeo (U.S. Geological Survey, Menlo Park Publishing Service Center) prepared the outstanding graphics. We thank William Beckon and Thomas Maurer (U.S. Fish and Wildlife Service, Sacramento, California) for provision of species at risk for the Bay-Delta. We also thank William Beckon, Susan De La Cruz (U.S. Geological Survey, Vallejo, California) and Regina Linville (Office of Environmental Health Hazard Assessment, Sacramento, California) for assistance with species-specific effects models and human health considerations. Thanks also go to Mark Huebner (U.S Geological Survey, Menlo Park, California) for exceptional assistance with manuscript preparation.

Keywords:

selenium, biodynamics, bioaccumulation, food webs, ecotoxicology, ecology

Local Identifier:

jmie_sfews_11157

Abstract:

Environmental restoration, regulatory protections, and competing interests for water are changing the balance of selenium (Se) discharges to the San Francisco Bay–Delta Estuary (Bay–Delta). The model for Se described here as part of the Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) draws both from the current state of knowledge of the Bay–Delta and of environmental Se science. It is an ecosystem-scale methodology that is a conceptual and quantitative tool to (1) evaluate implications of Se contamination; (2) better understand protection for fish and aquatic-dependent wildlife; and (3) help evaluate future restoration actions. The model builds from five basic principles that determine ecological risks from Se in aquatic environments: (1) dissolved Se transformation to particulate material Se, which is partly driven by the chemical species of dissolved Se, sets dynamics at the base of the food web; (2) diet drives bioavailability of Se to animals; (3) bioaccumulation differs widely among invertebrates, but not necessarily among fish; (4) ecological risks differ among food webs and predator species; and (5) risk for each predator is driven by a combination of exposures via their specific food web and the species' inherent sensitivity to Se toxicity. Spatially and temporally matched data sets across media (i.e., water, suspended particulate material, prey, and predator) are needed for initiating modeling and for providing ecologically consistent predictions. The methodology, applied site-specifically to the Bay–Delta, includes use of (1) salinity-specific partitioning factors based on empirical estuary data to quantify the effects of dissolved speciation and phase transformation; (2) species-specific dietary biodynamics to quantify foodweb bioaccumulation; and (3) habitat use and life-cycle data for Bay–Delta predator species to illustrate exposure. Model outcomes show that the north Bay–Delta functions as an efficient biomagnifier of Se in benthic food webs, with the greatest risks to predaceous benthivores occurring under low flow conditions. Improving the characterization of ecological risks from Se in the Bay–Delta will require modernization of the Se database and continuing integration of biogeochemical, ecological, and hydrological dynamics into the model.

Supporting material:

Appendix A

Copyright Information:

MARCH 2013

SAN FRANCISCO ESTUARY & WATERSHED SCIENCE

Sponsored by the Delta Science Program and the UC Davis John Muir Institute of the Environment

Ecosystem-Scale Selenium Model for the San Francisco Bay-Delta Regional Ecosystem Restoration Implementation Plan (DRERIP)

Theresa S. Presser[1,†] and Samuel N. Luoma[1,2]

ABSTRACT

Environmental restoration, regulatory protections, and competing interests for water are changing the balance of selenium (Se) discharges to the San Francisco Bay–Delta Estuary (Bay–Delta). The model for Se described here as part of the Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) draws both from the current state of knowledge of the Bay–Delta and of environmental Se science. It is an ecosystem-scale methodology that is a conceptual and quantitative tool to (1) evaluate implications of Se contamination; (2) better understand protection for fish and aquatic-dependent wildlife; and (3) help evaluate future restoration actions. The model builds from five basic principles that determine ecological risks from Se in aquatic environments: (1) dissolved Se transformation to particulate material Se, which is partly driven by the chemical species of dissolved Se, sets dynamics at the base of the food web; (2) diet drives bioavailability of Se to animals; (3) bioaccumulation differs widely among invertebrates, but not necessarily among fish; (4) ecological risks differ among food webs and predator species; and (5) risk for each predator is driven by a combination of exposures via their specific food web and the species' inherent sensitivity to Se toxicity. Spatially and temporally matched data sets across media (i.e., water, suspended particulate material, prey, and predator) are needed for initiating modeling and for providing ecologically consistent predictions. The methodology, applied site-specifically to the Bay–Delta, includes use of (1) salinity-specific partitioning factors based on empirical estuary data to quantify the effects of dissolved speciation and phase transformation; (2) species-specific dietary biodynamics to quantify foodweb bioaccumulation; and (3) habitat use and life-cycle data for Bay–Delta predator species to illustrate exposure. Model outcomes show that the north Bay–Delta functions as an efficient biomagnifier of Se in benthic food webs, with the greatest risks to predaceous benthivores occurring under low flow conditions. Improving the characterization of ecological risks from Se in the Bay–Delta will require modernization of the Se database and continuing integration of biogeochemical, ecological, and hydrological dynamics into the model.

KEY WORDS

Selenium, biodynamics, bioaccumulation, food webs, ecotoxicology, ecology.

† Corresponding author: *tpresser@usgs.gov*
1 U.S. Geological Survey, 345 Middlefield Road, Menlo Park, CA 94025
2 Current affiliation: John Muir Institute of the Environment, University of California Davis, Davis, CA 95616

INTRODUCTION

The Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) process focuses on construction of conceptual models that describe and define the relationships among the processes, habitats, species, and stressors for the Bay-Delta (DiGennaro and others 2012). The models use common elements and are designed to interconnect to achieve the goals of evaluating and informing Bay-Delta restoration actions. Selenium is recognized as an important stressor in aquatic environments because of its potency as a reproductive toxin and its ability to bioaccumulate through food webs (Chapman and others 2010; Presser and Luoma 2010a). Selenium's role is well documented in extirpation (i.e., local extinctions) of fish populations (Lemly 2002) and in occurrences of deformities of aquatic birds in affected habitats (Skorupa 1998). For Se, exposure is specific to a predator species' choice of food web and physiology, making some predators more vulnerable and, thus, more likely than others to disappear from moderately contaminated environments (Lemly 2002; Luoma and Presser 2009; Stewart and others 2004).

Concern about Se as a stressor in the Bay-Delta watershed originates from the damage to avian and fish populations that resulted when an agricultural drain to alleviate subsurface drainage conditions in the western San Joaquin Valley released Se into the Kesterson National Wildlife Refuge in the 1980s (Presser and Ohlendorf 1987). Later it was recognized that (1) some aquatic predators in the Bay-Delta were bioaccumulating sufficient Se to threaten their reproductive capabilities (SWRCB 1987, 1988, 1989, 1991) and; (2) primary Se sources included not only organic enriched sedimentary deposits in the San Joaquin Valley and elsewhere, but also their anthropogenic by-products such as oil (Cutter 1989; Presser 1994; Presser and others 2004). Proposals in 1978 and 2006 to extend an agricultural drain from the western San Joaquin Valley directly to the Bay-Delta as a way of removing Se from the valley were found both times to present substantial and broad ecological risks (e.g., USBR 1978, 2006; Presser and Luoma 2006).

Currently, Se contamination is spatially distributed from the Delta through the North Bay (Suisun Bay, Carquinez Strait, and San Pablo Bay) to the Pacific Ocean, mainly from oil-refining discharges internal to the estuary, and agricultural drainage discharges exported via the San Joaquin River. Regulatory and planning processes have intervened in the cases of both existing Se sources resulting in a decline in contamination since 1986-1992 when concentrations were maximal (SWRCB 1987, 1988, 1989, 1991; Presser and Luoma 2006; USBR 1995, 2001, 2009). However, the North Bay, the Delta, and segments of the San Joaquin River and some of its tributaries and marshes remain designated as impaired by Se (SWRCB 2011). More recently, the State initiated a Se Total Maximum Daily Load (TMDL) process to target both agricultural and oil refinery sources of Se (SFBRWQCB 2007, 2011) in coordination with development and implementation of site-specific water quality Se criteria for the protection of fish and wildlife by the U.S. Environmental Protection Agency (USEPA 2011a). The presence of a major oil-refining industry in the North Bay, and the substantial accumulated reservoir of Se in the soils and aquifers of the western San Joaquin Valley suggest that the potential for ecological risk from Se within the Bay-Delta watershed will continue into the foreseeable future as Se management and mitigation efforts take place (Presser and Luoma 2006; Presser and Schwarzbach 2008; USBR 2008; Appendix A.1).

Historic and more recent data show that certain predator species are considered most at risk from Se in the Bay-Delta (e.g., white and green sturgeon, scoter, scaup) because of high exposures obtained when they consume the estuary's dominant bivalve, *Corbula amurensis*, an efficient bioaccumulator of this metalloid (Stewart and others 2004; Presser and Luoma 2006). The latest available surveys of Se concentrations in *C. amurensis* and white sturgeon (*Acipenser transmontanus*) that were feeding (based upon isotopic evidence) in Carquinez Strait, Suisun Bay, and San Pablo Bay (Stewart and others 2004; Linares and others 2004; Kleckner and others 2010; Presser and Luoma 2010b; SFEI 2009) continue to show concentrations exceeding U.S. Fish and Wildlife Service (USFWS) dietary and tissue toxicity guidelines (Skorupa and others 2004; Presser and Luoma 2010b). Sturgeon contain higher concentrations of Se than any other fish species, reflecting their position as a top benthic predator (Stewart and others 2004). Surveys of surf scoter (*Melanitta perspicillata*) and greater scaup (*Aythya marila*) that feed voraciously on *C. amurensis* as they overwinter in Suisun Bay (SFEI 2005; De La Cruz and others 2008; De La Cruz 2010; Presser and Luoma 2010b) show Se has bioaccumulated to levels in muscle and liver tissue that may affect their ability to successfully migrate and breed (Heinz 1996; USDOI 1998; Ohlendorf and Heinz 2011).

Endangered Species Act requirements led to a number of species being determined as jeopardized by Se in the Bay-Delta under a proposed chronic aquatic life Se criterion of 5 $\mu g\ L^{-1}$ (USFWS and NOAA Fisheries 2000), including delta smelt (*Hypomesus transpacificus*); longfin smelt (*Spirinchus thaleichthys*); Sacramento splittail (*Pogonichthys macrolepidotus*); Sacramento perch (*Archoplites interruptus*); tidewater goby (*Eucyclogobius newberryi*); green sturgeon (*Acipenser medirostris*) and its surrogate white sturgeon (*Acipenser transmontanus*); steelhead trout (*Oncorhynchus mykiss*); Chinook salmon (*Oncorhynchus tshawytscha*); California clapper rail (*Rallus longirostris obsoletus*); California least tern (*Sterna antillarum browni*); bald eagle (*Haliaeetus leucocephalus*); California brown pelican (*Pelecanus occidentalis californicus*); marbled murrelet (*Brachyramphus marmoratus*); and giant garter snake (*Thamnophis gigas*). Recent analysis by the USFWS (2008a) of 45 species assumed the species most at risk depended on benthic food webs: greater scaup; lesser scaup (*Aythya affinis*); white-winged scoter (*Melanitta fusca*); surf scoter; black scoter (*Melanitta nigra*); California clapper rail; Sacramento splittail; green sturgeon; and white sturgeon. Not enough species-specific information is currently available for consideration of Se exposures for the giant garter snake, an endangered aquatic predator (USFWS 2006, 2009); the Dungeness crab (*Cancer magister*), an invertebrate that consumes *C. amurensis* (Stewart and others 2004); or for species that are within the Dungeness-crab food webs.

Human health advisories currently are posted for the Bay-Delta for the consumption of scoter, greater scaup, and lesser scaup based on elevated Se concentrations in their muscle and liver tissue (CDFG 2012, 2013). Selenium was found to be below the level of human health concern for consumption of edible tissue in certain species of fish, including white sturgeon, from the estuary (OEHHA 2011). White sturgeon contained the highest levels of Se among species of fish surveyed. Some individual white sturgeon sampled from North Bay locations had Se concentrations that exceeded Se advisory levels, based on specific consumption rates (see later detailed discussion under "Human Health" on page 23). Additionally, white sturgeon recreational fishing is limited, based on a decreasing species population (CDFG 2012).

It was recently suggested that the traditional regulatory approach to managing Se contamination is deeply flawed (Reiley and others 2003; Luoma and Presser 2009; Chapman and others 2010), and that a new conceptual model of the processes that control its toxicity is needed for regulatory purposes, especially in estuarine environments like the Bay-Delta. In recognition of the issues with the traditional approach to deriving a criterion for Se, the USEPA is leading a cooperative effort to develop site-specific fish and wildlife Se criteria for habitats affected by Se in California. Specifically for the Bay-Delta, the effort includes protection of Federally listed species and designated critical habitat (USFWS and NOAA Fisheries 2000; USEPA 2011a). Development of Se criteria for the Bay-Delta is proceeding first in this effort because the estuary is considered a sensitive hydrologic system and habitat in terms of Se and it was thought that protection here would elicit regulatory compliance upstream (USEPA 2011a). On the broader scale, Se is considered a general stressor of the estuary, and a constituent that should be analyzed as part of management and restoration planning and implementation (USEPA 2011b; NRC 2010, 2011, 2012).

The cooperative regulatory effort specifically recognizes that the new conceptual model must consider (1) the inaccuracies of deriving toxicity from waterborne Se concentrations; (2) the bioaccumulative nature of Se in aquatic systems; (3) Se's long-term persistence in aquatic sediments and food webs; and (4) the importance of dietary pathways in determining toxicity (USEPA 1992, 2000a; USFWS and NOAA Fisheries 2000; Luoma and Presser 2009; Presser and Luoma 2006, 2010a, 2010b). Revisions by USEPA also are occurring at the national level to incorporate into the basis for regulation recent advances in the environmental science of Se. For example, a fish tissue Se criterion and implementation plan are being proposed to better integrate dietary exposure pathways into regulatory frameworks, and ensure an adequate link to toxicity (USEPA 2004, 2011b). During this transitional period when species may be jeopardized and while Se criteria are being revised, USEPA has applied the national chronic freshwater Se criterion of 5 µg L^{-1} to the estuary (USEPA 1992, 2000a).

We present here an ecosystem-scale Se conceptual model for the Bay-Delta that addresses the needs of both the DRERIP process and the USEPA. Quantitative applications of the model are also possible. Quantification provides an opportunity to evaluate site-specific Se risks under different circumstances, using field data combined with a systematic quantification of each of the influential processes that link source inputs of Se to toxicity. The methodology is presented in terms of specified DRERIP components (i.e., drivers, linkages, and outcomes). As an example of how quantitative applications can be used, we calculate the dissolved ambient Se concentrations that would result in compliance with a chosen fish or bird tissue guideline under different assumptions or environmental conditions. Uncertainties and model sensitivities are illustrated by comparing outcomes of different exposure scenarios. The scenario approach could facilitate the model's use by decision-makers for quantitative evaluation of restoration alternatives for ecosystem management and protection.

MODEL OVERVIEW

The DRERIP Ecosystem-Scale Selenium Model for the Bay-Delta (Figure 1) has five interconnected modules that depict drivers (sources and hydrology), linkages (ecosystem-scale processes), concentration outcomes (Se concentrations in water, particulates, and organisms), and food web exposure outcomes (effects on fish, wildlife, and human health). Model outcomes in Figure 1 are further refined to critical choices for modeling and species-specific risk scenarios for the Bay-Delta. Together the five modules consider the essential aspects of environmental Se exposure: biogeochemistry, food web transfer, and effects. They also take into account the estuary's ecology and hydrology as well as the functional ecology, physiology and ecotoxicology of the most vulnerable predator species. The modules define relationships that are important to conceptualizing and quantifying how Se is processed from water through diet to prey and predators, and the resulting effect on components of the food web. Thus, the DRERIP Ecosystem-Scale Selenium Model combines fundamental knowledge of Se behavior in ecosystems (Se drivers, linkages, and outcomes) with site-specific knowledge of the Bay-Delta (Bay-Delta drivers, linkages, and outcomes) to define site-specific Se risk (Figure 1).

The DRERIP Se submodels provide details for

- Sources and Hydrology (submodel A, Figure 2);
- Ecosystem-Scale Se Modeling (submodel B, Figure 3);
- Exposure: Food Webs, Seasonal Cycles, Habitat Use (submodels C, D; Figures 4, 5);
- Fish and Wildlife Health: Ecotoxicology and Effects (submodels E, F; Figures 6, 7); and
- Human Health (submodel G, Figure 8).

A human health pathway is designated, but emphasis here is on Se pathways to fish and wildlife health. The North Bay and the Delta are emphasized because the important Se sources have the potential to most affect those habitats and ecosystems (submodel A, Figure 2).

The quantitative DRERIP Ecosystem-Scale Selenium Model is based upon concepts and parameters developed elsewhere for a wide variety of aquatic systems and their food webs (submodel B, Figure 3; submodel E, Figure 6) (Luoma and Rainbow 2005; Luoma and Presser 2009; Chapman and others 2010; Presser and Luoma 2010a). To quantitatively apply the rela-

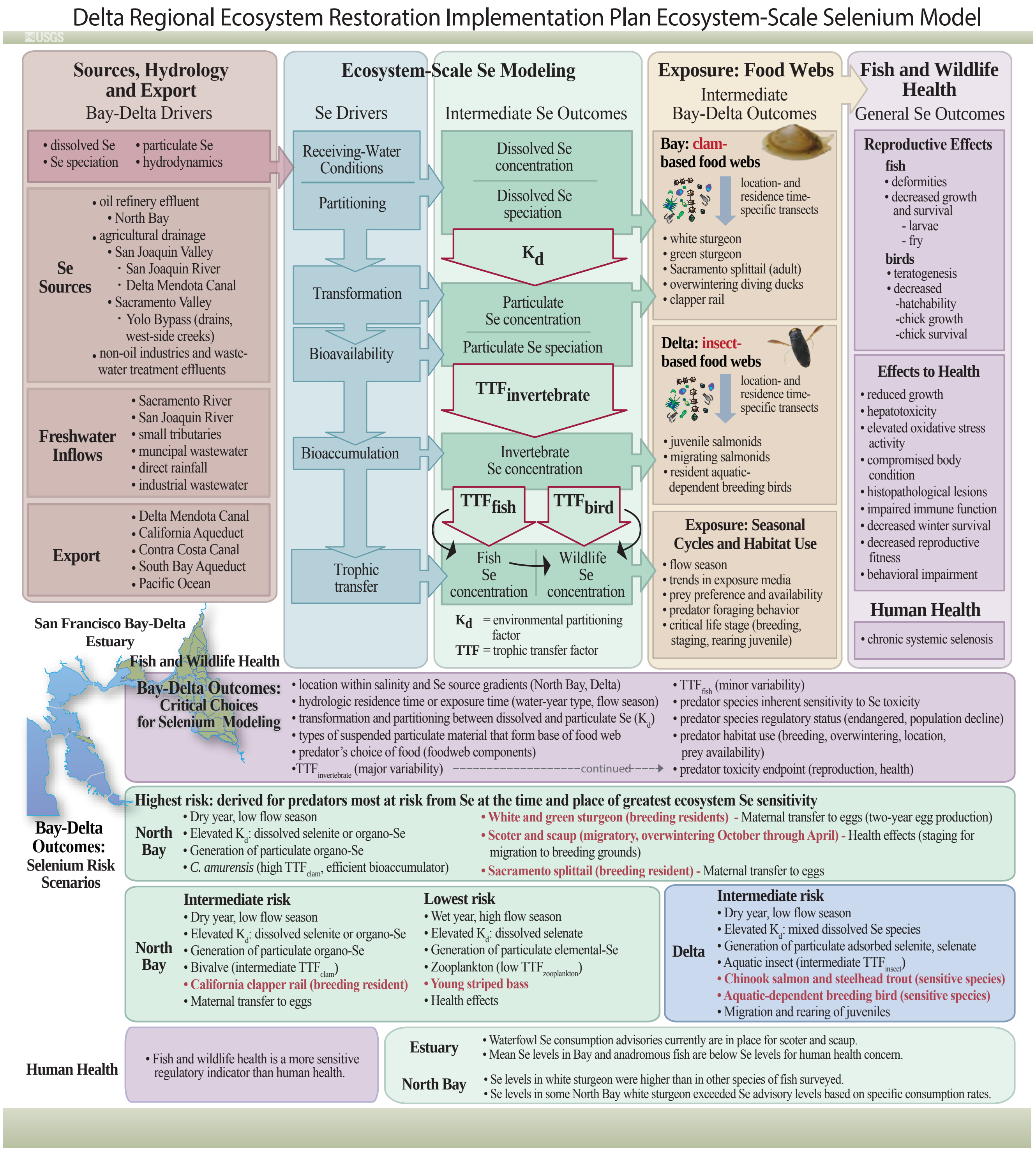


Figure 1 The DRERIP Ecosystem-Scale Selenium Model illustrates five interconnected modules that depict essential aspects of the Bay-Delta's hydrology, biochemistry, and ecology and of the exposure and ecotoxicology of predators at risk from selenium. These modules, and the detailed sub-models that follow, conceptualize (1) how selenium is processed from water through diet to predators and (2) its effects on ecosystems. Critical choices for modeling are summarized, and a quantitative application of the model for the estuary is derived for predators most at risk from Se at the time and place of greatest ecosystem Se sensitivity.

tionships in the conceptual model, we use empirical data from the Bay-Delta (e.g., Cutter and Cutter 2004; Presser and Luoma 2006, 2010b) to (1) help define environmental partitioning factors (K_ds) that quantify transformation of dissolved Se into particulate forms; and (2) help define biodynamic trophic transfer factors (TTFs) that quantify uptake by consumer species and their predators (submodel C, Figure 4; submodel D, Figure 5; submodel F, Figure 7). The broader, ecosystem-scale Se modeling approach was validated by comparing model forecasts with field data, across both a range of common food webs and hydrologic environments (Luoma and Rainbow 2005; Presser and Luoma 2010a) and specifically for the Bay-Delta and Newport Bay (Presser and Luoma 2006, 2009, 2010b).

The organizing principle for quantification is the progressive solution of a set of simple equations, each of which quantifies a process important in Se exposure (submodel B, Figure 3). The interaction of Se loading from different sources, hydrology, and hydrodynamics determine dissolved Se concentrations in the Bay-Delta. Transformation of Se from its dissolved form to a particulate form (represented here operationally as K_d) ultimately determines bioavailability to the food web. In a given environment, Se is taken up much faster from food than from solution by animals. Thus, the entry of Se into the food web can be estimated by a TTF for each trophic level. $TTF_{invertebrate}$ defines dietary uptake by a consumer species, which occurs when invertebrates (or herbivorous fish), feed on primary producers, detritus, microbes, or other types of particulate materials. Selenium bioaccumulation differs widely among invertebrate species because of different physiologies (Luoma and Rainbow 2005). These differences are captured by employing species-specific TTFs (Luoma and Presser 2009). Species-specific TTFs for predaceous fish and birds ($TTF_{predator}$) also are applied to the transfer of Se from invertebrate prey species to their predators (Presser and Luoma 2010a).

For the Bay-Delta, Stewart and others (2004) showed that Se concentrations differ widely among predators that live in the same environment. The main reason for those differences lies in the prey preferences of predators. For example, bass eating from the water-column food web consume invertebrates with much lower Se concentrations than sturgeon eating benthic invertebrates, especially bivalves (Stewart and others 2004). The differences in Se uptake among predator species ($C_{predator}$) can be captured only if the correct prey species (or class of prey species) is included in the equation (submodel B, Figure. 3) and the conceptualization (submodel C, Figure 4). This also means that the choice of predator species is critical in assessing risks from Se contamination.

Selenium concentrations in predators can be predicted with surprisingly strong correlation to observations from nature if particulate Se concentrations are known and an appropriate food web is used for the predator (Luoma and Presser 2009; Presser and Luoma 2010a). One use of these calculations might be to quantify the degree to which different species of birds and fish might be threatened by Se in a specified environment, for example. The correspondence between observed $C_{predator}$ and predictions of $C_{predator}$ from the series of equations that begins with dissolved concentrations (submodel B, Figure 3) depends upon how closely the partitioning between dissolved and particulate Se used in the model matches that occurring in the ecosystem of interest. One use of quantification in this instance is to run the model in the reverse direction to determine the dissolved Se concentration in a specific type of hydrologic environment and food web that would result in a specified Se concentration in the predator. Later, we present a detailed example of how the latter might be applied to real-world issues.

In the final step, effects on the reproduction and health of predaceous fish and birds are determined from bioaccumulated Se concentrations. Selenium is one of the few trace elements for which tissue concentrations have been correlated to these adverse effects in both dietary toxicity tests and field studies. The toxicity data for some of the key species in the Bay-Delta are limited or non-existent. The necessity of establishing effects thresholds from surrogate species adds some uncertainty to assessments of risk. Therefore, in our examples, we use different possible choices for such thresholds.

Additionally, modeling here is within a specified location and flow condition to provide context for exposure and to help narrow the uncertainties in quantifying the ecological and physiological potential for bioaccumulation (Presser and Luoma 2010b).

MODULES

Sources, Hydrology, and Export

Estuary Mass Balance

The major portion of the estuary from the rivers to the Golden Gate Bridge is termed the Northern Reach, with Suisun Bay near the head of the estuary (submodel A, Figure 2). Selenium sources and their hydraulic connections within that reach have been documented in a number of publications (Cutter 1989; Cutter and San Diego–McGlone 1990; Cutter and Cutter 2004; Meseck and Cutter 2006; Presser and Luoma 2006, 2010b; SFBRQWCB 2011) (Figure 1; submodel A, Figure 2). In brief, the most important regulated estuarine sources of Se are (1) internal inputs of oil refinery wastewaters from processing of crude oils at North Bay refineries; and (2) external inputs of irrigation drainage from agricultural lands of the western San Joaquin Valley conveyed mainly through the San Joaquin River. (submodel A, Figure 2). These and other potential Se sources are described in detail in Appendix A.1. These details reflect the depth of history for Se management within the Bay-Delta watershed and the continuing tradeoffs that accompany their presence.

The Sacramento and San Joaquin rivers are the main sources of freshwater inflow to the Bay-Delta, with the Sacramento River being the dominant inflow under most conditions (Conomos and others 1979; Peterson and others 1985). The rivers provide 92% of the freshwater inflows to the Bay-Delta, with small tributaries and municipal wastewater providing approximately 3% each (McKee and others 2008).

In general, Se concentrations in the Sacramento River (above tidal influence, e.g., at Freeport) are low and relatively constant (1998 to 1999 average: 0.07 µg L^{-1}; range 0.05 to 0.11 µg L^{-1}) (Cutter and Cutter 2004). Dissolved Se concentrations in the San Joaquin River (above tidal influence, e.g., at Vernalis) were about an order-of-magnitude higher than those in the Sacramento River in 1999 (1998 to 1999 average: 0.71 µg L^{-1}; range 0.4 to 1.07 µg L^{-1}) (Cutter and Cutter 2004) and are much more variable. In the late 1980s and early 1990s concentrations above 5 µg L^{-1} were observed occasionally in the San Joaquin River (Presser and Luoma 2006), but in-valley source control efforts have reduced Se loads and concentrations (Appendix A.1).

In the present configuration of the Bay-Delta, the San Joaquin River is predominantly re-routed and exported back to the San Joaquin Valley (submodel A, Figure 2; Appendix A.1). Hence, for the purposes of evaluating Se contamination sources, the simplest assumption is that the "baseline" Se concentrations (undisturbed by human activities) in the Delta would be close to the Se concentrations in the Sacramento River. The pre-disturbance baseline Se concentrations in the Bay or tidal reaches of the rivers would be concentrations in the Sacramento River mixed with concentrations in coastal waters, as reflected by the salinity of the sampling location. Deviations from that baseline reflect inputs of Se internal to the Bay (industrial or local streams) (Cutter and San Diego-McGlone 1990; Cutter and Cutter 2004) or input of Se to the Bay from the San Joaquin River.

The current San Joaquin River contributions to the Bay, thought to be minimal during most flow conditions, are especially difficult to measure (Appendix A.1). However, that could change. Under some proposals for modifications in water infrastructure, increased diversion of the Sacramento River through tunnels or canals would be accompanied by greater inflows from the San Joaquin River to the Delta and the Bay. In simulations available of the implications of such a change, Meseck and Cutter (2006) found that Se concentrations doubled in particulate material in the Bay.

The conceptual model described above suggests that parameters critical in determining the mass balance model for Se inputs for the Bay-Delta are (1) total river discharge (Sacramento River and San Joaquin River); (2) water diversions or exports (i.e., pumping at Tracy and Clifton Court Forebay south to the Delta–Mendota Canal and the California Aqueduct); (3) proportion of the San Joaquin River directly

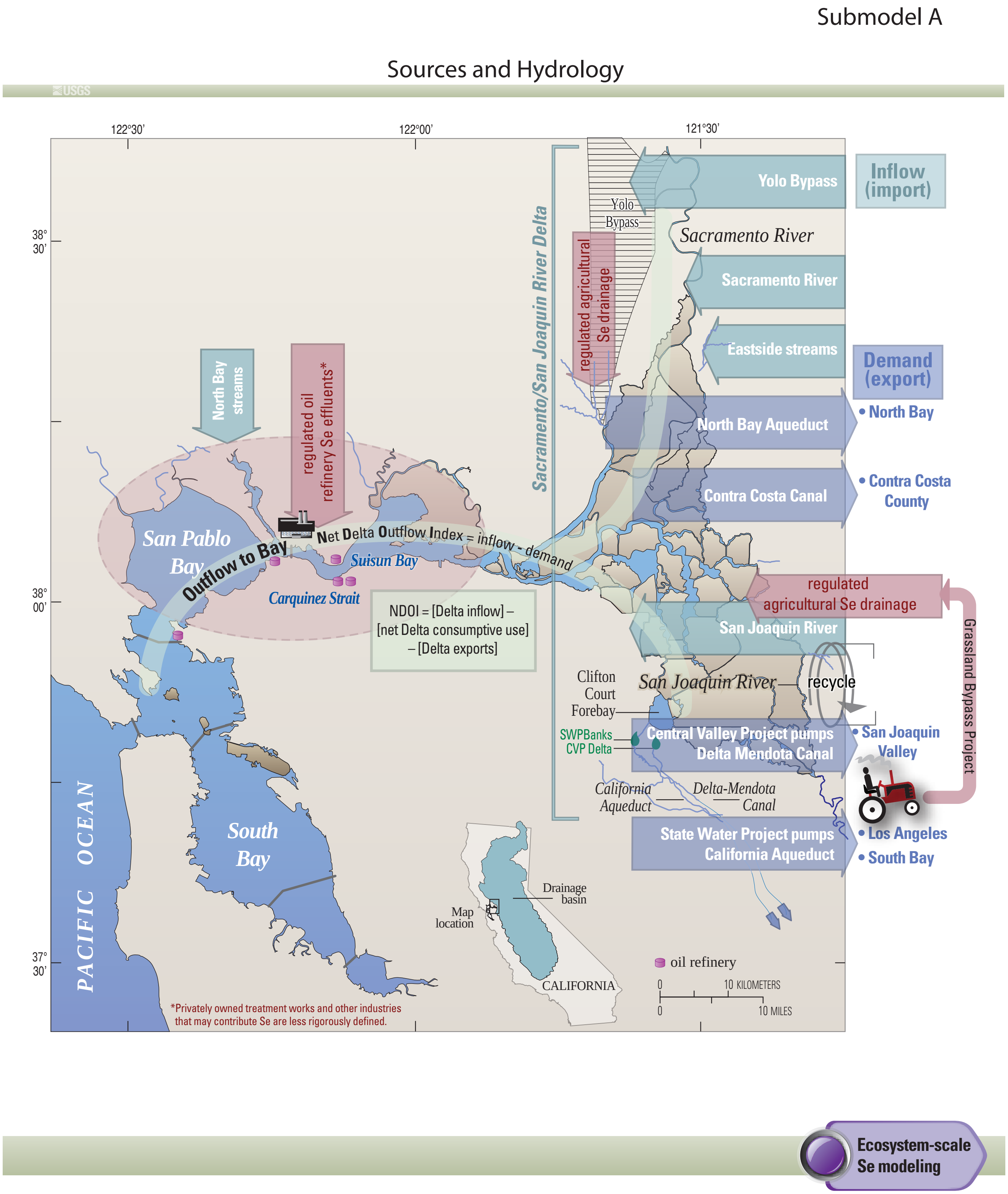


Figure 2 Submodel A. Sources and Hydrology

recycled south before it enters the Bay; 4) Se concentrations in each of the internal and external sources; and 5) total outflow of the rivers to the Bay or Net Delta Outflow Index (NDOI).

There are several uncertainties in quantification of the Se mass balance. One is the difficulty of precisely defining the contribution of the San Joaquin River to the NDOI, and hence the agricultural component of Se inputs to the Bay. Diversions and Delta hydrodynamics are sufficiently complex that every method available to determine that contribution has serious uncertainties (e.g., subtracting Sacramento River flow at Rio Vista from NDOI). Simple water accounting suggests minimal potential for flow from the San Joaquin River to enter the Bay (i.e., as measured by the percent by which river flow at Vernalis exceeds total export) during many months of the year (USBR 2012). Inputs are possible during spring months (April and May), wet and above normal years, and times of low capture efficiency (e.g., when river barriers are in-place) or when the ratio of the Sacramento River and San Joaquin River discharges is lowest in the fall.

A second uncertainty is that the strong tidal circulation in the Bay and the Delta mixes dissolved and particulate Se through the entire tidal reach, distorting spatial patterns that might otherwise help identify important sources of Se input (Ganju and others 2004). The three-dimensional nature of tidally driven hydrodynamics dissociates distributions of dissolved and particulate Se as well, adding complexity. One important outcome of this is that particulates contaminated with Se from industrial sources in Suisun Bay could feasibly be found throughout the full tidal range in both rivers, including otherwise uncontaminated segments of the Sacramento River. Riverine endmember concentrations of particulate Se, therefore, must be defined from landward of the reach of the tides, although river discharge at those locations does not necessarily represent riverine outflow to the Bay. Collecting an adequate mass of suspended particulate material for Se analysis in non-tidal freshwaters is challenging; therefore, few such data exist for the Sacramento River and even for some of the areas possibly affected by agricultural drainage. Hydrodynamic models of varying complexity are available that can approximate water movements in this complex situation (e.g., Delta Simulation Model II). But modeling the distribution of particulate material (crucial for understanding implications of Se) is much more difficult (Ganju and others 2004).

Links Between Source Inputs and Water Inflows

Both Sacramento River and San Joaquin River discharges vary dramatically during the year depending on runoff, water management, and diversions. Residence (or retention) time is affected by river discharges (e.g., Cutter and Cutter 2004), but the strong tidal influences make that difficult to precisely define. Nevertheless, even a coarse differentiation of seasonal periods (low flow and high flow) and classification by water year (critically dry, dry, below normal, normal, above normal and wet) can be useful in evaluating influences on processes important to the fate and bioavailability of Se (Presser and Luoma 2006). Empirical data suggest processes such as dilution of local inputs and phase transformations that incorporate Se into organic particulate material appear to be affected by changes in retention time in the estuary, at least to some extent (Cutter and Cutter 2004; Doblin and others 2006; Presser and Luoma 2006, 2010a, 2010b). For example, Cutter and San Diego-McGlone (1990) found that a peak in selenite concentrations was centered around the area of inputs from oil refineries during low riverine inflows to the Bay in the 1980s; but that peak disappeared during periods of high riverine discharge. They used a one-dimensional model of the water and a Se mass balance to show that the mass of Se discharged by the refineries was the dominant source of selenite during low flows, but that it was insignificant compared to the mass of Se input from the Sacramento River during high flows. The selenite peak was reduced and replaced by a different pattern of dissolved Se speciation when Se discharges from the refineries were reduced by about half in 1999 (Cutter and Cutter 2004). Similarly, high Se concentrations in the southernmost Delta (Stockton) reflect San Joaquin River inputs, but concentrations seaward of this location decline as they are diluted by the large volumes of Se-poor Sacramento River water channeled into the Delta for export (Lucas and Stewart 2007). Local tributaries could be an internal source of Se to the Bay, but these inputs occur almost entirely during high riverine inflow periods when their Se loads are insignificant compared to the large mass of Se carried into the Bay by high discharge from the Se-poor Sacramento River.

The NDOI, essentially inflow minus demand, is often used to indicate hydrologic influences on Se concentrations, including differences in retention time of a parcel of water in the Bay and Delta (Cutter and Cutter 2004). Increased exposure time (i.e., the cumulative amount of time a particle spends within a domain, taking into consideration repeated visits over multiple tidal cycles; L. Doyle, W. Fleenor, and J. Lund, University of California, Davis, pers. comms.; 2012) at the lowest inflows may explain why NDOI is a relevant indicator of the effect of flow on processes such as conversion of Se from dissolved to particulate forms.

Exports

The Delta–Mendota Canal, California Aqueduct, Contra Costa Canal, and South Bay Aqueduct all export water from the Delta. Thus, all are secondary recipients of the Se sources considered here (submodel A, Figure 2). The Delta–Mendota Canal also receives agricultural drainage directly, with that source proposed to be under regulatory control (USFWS 2009; USBR 2011). In general, however, few data are available to assess a mass balance for Se through the State Water Project, Central Valley Project, and other water-delivery systems.

In terms of export of Se to the Pacific Ocean from the Bay, some data are available for seaward locations in the Bay. Dissolved concentrations at these locations are among the lowest observed in the system when not under flood flows (Cutter 1989; Cutter and San Diego–McGlone 1990; Cutter and Cutter 2004); particulate concentrations are occasionally high, however. Under shorter residence times during high flows, increased dissolved concentrations near the Golden Gate Bridge (Cutter and Cutter 2004) suggest sources internal to the Bay affect ocean-dissolved Se concentrations. Outflows to the sea have been estimated in simple mass balance models (Cutter and San Diego-McGlone 1990) although there are some uncertainties in such estimates. Ocean disposal was considered as one of the alternatives for comprehensive agricultural drainage management from the western San Joaquin Valley (USBR 2006). However, efficient Se recycling within productive ocean ecosystems and the opportunities for Se biomagnification in complex marine food webs suggest serious risks are likely (Cutter and Bruland 1984); hence, there are reasons for careful study before such options are considered.

Ecosystem-Scale Selenium Modeling

Dissolved Selenium Concentrations, Speciation, and Transformation

Total dissolved Se concentrations within the Bay range from 0.070 to 0.303 $\mu g\ L^{-1}$, with a mean of $0.128 \pm 0.035\ \mu g\ L^{-1}$ and a median of $0.125\ \mu g\ L^{-1}$ across 128 samples collected since 1997 (Doblin and others 2006; Lucas and Stewart 2007). The mean concentration is only approximately two times higher than Se concentrations in the dominant freshwater endmember (the Sacramento River). In all surveys since the 1980s, Se concentrations in the tidal Bay and Delta are highest in Suisun Bay, with a downward spatial trend from Carquinez Strait toward the ocean. The latter suggests that dissolved concentrations in the ocean endmember are about the same as those in the Sacramento River.

The dissolved gradients of Se concentration are not necessarily the best indicators of the distribution of Se effects. Ecological implications depend upon the biogeochemical transformation from dissolved to particulate Se. Phase transformation of Se is of toxicological significance because particulate Se is the primary form by which Se enters food webs (Figures 1, 3 and 4) (Luoma and others 1992). Speciation of dissolved Se into its three dominant oxidation states is an important component in many conceptual models. In the Bay-Delta, speciation of dissolved Se is important because it influences the type and rate of phase transformation reaction that creates particulate Se. Examples of phase transformation reactions include (1) uptake by plants and phytoplankton of selenate, selenite, or dissolved organo-Se and transformation to particulate organo-Se by

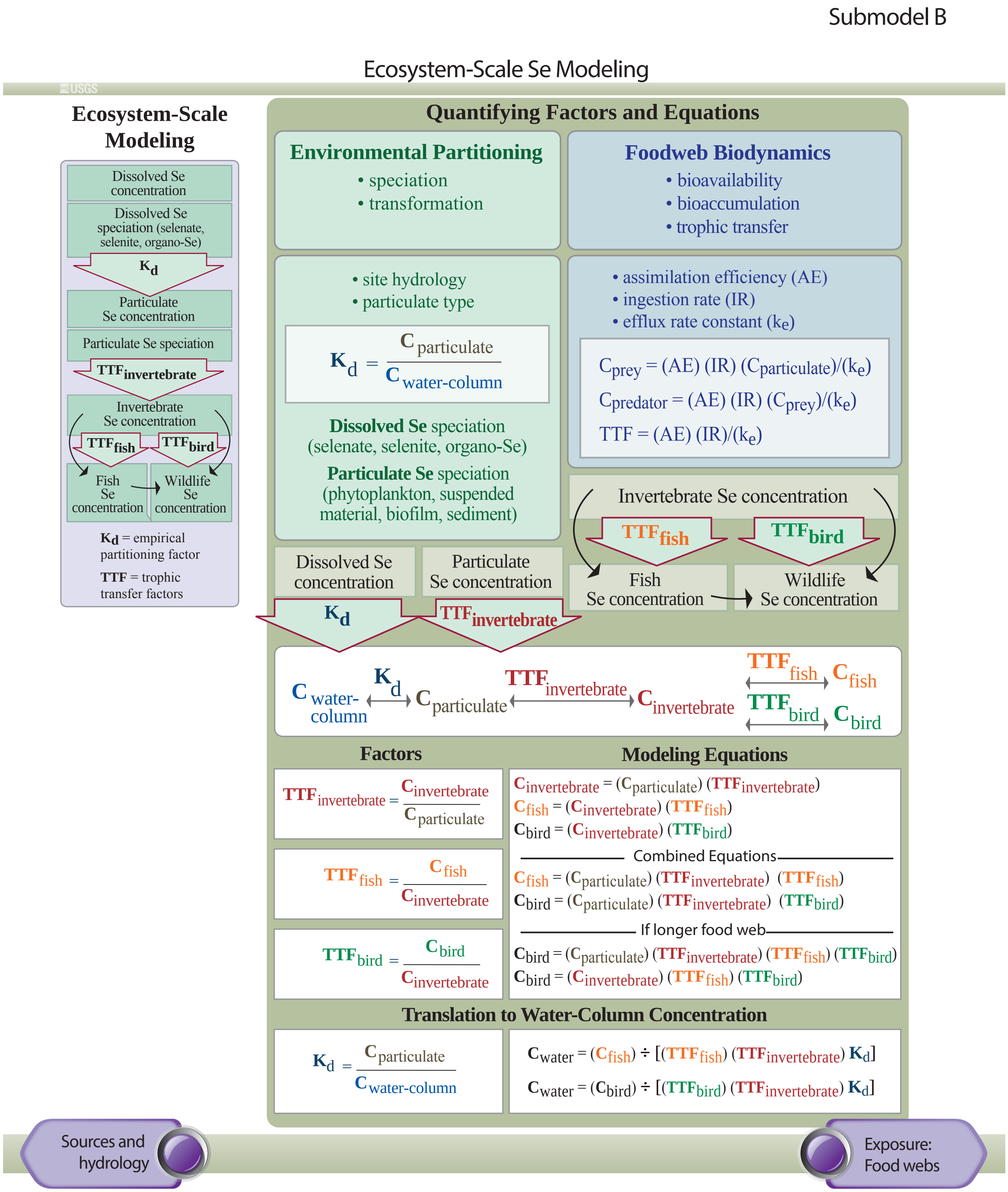


Figure 3 Submodel B. Ecosystem-Scale Se Modeling

assimilatory reduction, where uptake of selenate is considerably slower than uptake of the other two forms (e.g., Sandholm and others 1973; Riedel and others 1996; Wang and Dei 1999; Fournier and others 2006); (2) sequestration of selenate into sediments as particulate elemental Se by dissimilatory biogeochemical reduction (e.g., Oremland and others 1989); (3) adsorption as co-precipitated selenite through reactions with particle surfaces; and (4) recycling of particulate phases back into water as detritus or as dissolved organo-Se, after organisms die and decay (e.g., Velinsky and Cutter 1991; Reinfelder and Fisher 1991; Zhang and Moore 1996).

These different biogeochemical transformation reactions result in different forms of Se in particulate material: organo-Se, adsorbed Se, or elemental Se. Although only a few studies have determined speciation of particulate Se (e.g., Doblin and others 2006), such data can greatly aid in understanding bioavailability. Experimental studies show that particulate organo-Se is the most bioavailable form when it is eaten by a consumer species (Luoma and others 1992). Detrital or adsorbed Se is also bioavailable when ingested by animals, although to a lesser extent than organo-Se (Wang and others 1996). Non-particle associated elemental Se is not bioavailable (Schlekat and others 2000).

Concentrations of Se in particulate materials (per unit mass material) within the Bay and tidal freshwaters range widely from 0.1 to 2.2 $\mu g\ g^{-1}$ dry weight (dw), with a mean of $0.56 \pm 0.32\ \mu g\ g^{-1}$ dw and a median of 0.45 $\mu g\ g^{-1}$ dw ($n = 128$) since 1997 (Doblin and others 2006; Lucas and Stewart 2007). The 15-fold range in particulate concentrations contrasts sharply with the 4-fold range in dissolved concentrations, as do the contrasts in standard deviations. Not only are particulate concentrations much more dynamic than dissolved concentrations, but they also are about four times higher if expressed in common units. Both reflect biogeochemical transformation processes and, perhaps, inorganic adsorption. The latter is probably more important in soils than in the aquatic environment. Given the different dynamics and the variability of dissolved and particulate Se, it is not surprising that the ratio of the two also is quite variable.

Geochemical models that attempt to capture phase transformations of Se under different conditions are problematic. In fact, no models are available that can predict particulate Se concentrations based solely upon dissolved concentrations and biogeochemical conditions. One reason is that conventional thermodynamic equilibrium-partitioning models are inadequate for Se. Critical Se transformation processes are biological, and not predictable from thermodynamics. Some model approaches predict the particulate Se added on to a pre-existing particulate concentration, using a combination of phytoplankton productivity and re-suspension (Meseck and Cutter 2006; SWRCB 2011; Tetra Tech, Inc. 2010). While such models provide interesting estimates of temporal and spatial distributions of particulate Se, their major limitations lie in the basis upon which the pre-existing concentration is chosen and their inability to comprehensively account for all the processes involved in transformation.

The choice of the (pre-existing) baseline particulate Se concentration is critical to the questions models can address. Local data can be used for choosing pre-existing Se concentrations at the seaward and landward boundaries in the Bay-Delta. But the data used to date are from tidally affected reaches of the river, and are likely to be biased by redistribution of already contaminated particles from tidal pumping. As noted above, few data exist for particulate Se concentrations above the tidal reach of the Sacramento River; nor are there adequate determinations of Se concentrations on particulates from the coastal zone. In such a case, answers to questions about changing the internal Se inputs to the Bay are biased in that the boundary condition already includes such inputs (SWRCB 2011; Tetra Tech, Inc. 2010). On the other hand, this modeling approach appears to be well suited to test the influence of changing inputs from one boundary or from primary production alone (Meseck and Cutter 2006; Tetra Tech, Inc. 2010).

Observations of environmental partitioning of Se between dissolved and particulate phases can be employed to estimate transformation efficiencies in lieu of a comprehensive approach to modeling biogeochemical phase transformation for Se. Presser and Luoma (2006) first used field observations to quantify partitioning, which they described by the somewhat controversial term K_d. Luoma and Presser (2009) were careful to emphasize that their K_ds represented conditional observations from the Bay-Delta at a specific time and place; and were not meant to be equilibrium constants. Thermodynamic equilibrium constants would be inappropriate to describe an inorganic to organic transformation. They pointed out that no single constant could be expected to apply to all environmental conditions either in the Bay-Delta or elsewhere. Site hydrology, dissolved speciation, and the type of particulate material are all influential, although specific influences were not necessarily predictable in quantitative terms. An operational approach was therefore chosen to try to estimate influences of such processes.

They defined K_d as the ratio of particulate material Se concentration (in dw) to the dissolved Se concentration observed at any instant in simultaneously collected samples. The specific equation is

$$K_d = (C_{\text{particulate material}}, \mu g\ kg^{-1}\ dw) \div (C_{\text{water}}, \mu g\ L^{-1}) \quad (1)$$

Of interest here is the particulate matter at the base of the food web. As sampled in the environment that can include suspended particulate Se (which is a physically inseparable mix of phytoplankton, periphyton, detritus and inorganic suspended material), biofilm, sediment and/or attached vascular plants. Feeding characteristics of the organisms in question and data availability dictate the best choice among these. For example, for a filter-feeding bivalve in the Bay-Delta, Se concentrations determined in suspended particulate material (in $\mu g\ g^{-1}$ dw) are the preferred parameter for modeling because these animals filter their food from the water-column.

Some broad generalizations are possible about K_ds for Se (Presser and Luoma 2010a). For example, if all other conditions are the same, K_d will increase as selenite and dissolved organo-Se concentrations increase relative to selenate. Calculations using data from laboratory microcosms and experimental ponds show speciation-specific K_ds of 140 to 493 where selenate is the dominant form; 720 to 2,800 when an elevated proportion of selenite exists; and 12,197 to 36,300 for 100% dissolved seleno-methionine uptake into algae or periphyton (Besser and others 1989; Graham and others 1992; Kiffney and Knight 1990). Compilations of K_ds also show different general ranges for rivers, streams, lakes, ponds, wetlands, and estuaries that are affected by Se inputs (Presser and Luoma 2010a), although with some overlap. Exposure time for phase transformation is probably an important factor driving differences among such systems. Estuaries are among the sites with the highest values (range of medians from 4,000 to 21,500) indicating efficient conversion of dissolved Se to particulate Se. Finally, although the influence of exposure time for a particle within an estuary is challenging to understand precisely, especially in the Bay-Delta because of the dominance of tidally driven circulation, K_ds seem to be higher during conditions where more time is available for transformation reactions to occur (Presser and Luoma 2010b).

The most recent transects of the Bay that provide spatially and temporally matched data for derivation of K_ds from dissolved and particulate Se concentrations were from June 1998 to November 1999 (Cutter and Cutter 2004; Doblin and others 2006). In these studies, samples were collected at 1 meter below the surface, and included dissolved Se concentrations, suspended particulate material Se concentrations, dissolved Se speciation, suspended particulate Se speciation, salinity, and total suspended material. These data were collected in four different transects across the salinity gradient in the Northern Reach under a variety of river discharge and presumed residence time conditions. The full range of dissolved Se concentrations in these transects was 0.070 to 0.303 $\mu g\ L^{-1}$. The suspended particulate material Se concentrations were more variable: 0.15 to 2.2 $\mu g\ g^{-1}$ dw. Calculated K_ds ranged from 712 to 26,912. The degree of variability across this whole data set is large. However, the largest part of the variability was driven by very high values in the landward-most and seaward-most samples, where dissolved concentrations were very low. Such ratios can be artificially inflated when values become very low in the denominator, if the numerator does not decline as rapidly. Tidal pumping of contaminated particles from the Bay upstream into the less contaminated Sacramento River water is a possible cause of such an effect.

Downstream transport of highly contaminated particles from the San Joaquin River into Bay or Delta water could also be a cause. Finally, seaward, where residence times are elevated in Central and San Pablo bays, biological transformation could enrich Se in particles while depleting it from the water column. If the goal is to find conditions where there is sufficient linkage between dissolved and particulate Se to be useful in forecasts of one from the other, none of these conditions would apply. Presser and Luoma (2010b) avoided such biases and thereby constrained variability by restricting K_ds geographically to the middle range of the salinity zone in Suisun Bay. This also focused the modeling on the most contaminated segment of the estuary.

If location is restricted to Carquinez Strait–Suisun Bay—eliminating freshwater and ocean interfaces—then the range of dissolved Se concentrations is narrowed to 0.076 to 0.215 µg L^{-1} and the range of suspended particulate material Se concentrations is narrowed to 0.15 to 1.0 µg g^{-1} dw. The variation of K_d is narrowed to a range of means of 1,180 to 5,986 (or of individual measurements, 712 to 7,725). Because this data set is still large, median or mean concentrations, or a given percentile, can be used as viable indicators of partitioning in modeling scenarios.

Seasonality also is important, and restrictions to specific flow regimes also can be used to constrain variability. For example, the highest mean K_ds occur during periods of the lowest river inflows (and highest residence times). Constrained to Suisun Bay, the mean K_d was 1,180 ± 936 in June 1998. This was a high flow season wherein Cutter and Cutter (2004) estimated a residence time of 11 days. The mean K_d was 5,986 ± 1,353 in November 1999. This was a low flow season with an estimated residence time of 70 days. The mean K_d among all constrained samples was 3,317, and the mean for low flow seasons was 4,710.

Transects in the Delta were also conducted between 1998 and 2004 in different flow regimes (Doblin and others 2006; Lucas and Stewart 2007). Dissolved Se concentrations among all these samplings ranged from 0.083 to 1.0 µg L^{-1}, with a mean of 0.25 ± 0.24 (n = 72). Particulate concentrations ranged from 0.27 to 6.3 µg g^{-1} dw, with a mean of 0.98 ± 0.94 (n = 71). As in the Bay transects, the range in particulate concentrations (23-fold) exceeds the range in dissolved concentrations (12-fold). Concentrations and variability, thus, were even greater in the Delta, overall, than in the Bay. In the Delta, K_ds ranged from 554 to 38,194, with the range of means from 1,886 ± 1,081 in January 2003 (a high flow season) to 7,712 ± 3,282 in July 2000 (a low flow season). Sets of dissolved and particulate Se concentrations determined as part of focused research for the Delta in September 2001, the low flow season of a dry year, yielded some especially elevated K_ds (>10,000) (Lucas and Stewart 2007). In general, these elevated K_ds may reflect tidal pumping, or represent times and areas where Se is concentrating in particulate material because of differing hydrologic environments (e.g., slow-moving backwaters with high productivity). Constraining variability is more difficult in the Delta, hence, quantifying phase transformation from empirical data is more uncertain in this system.

Given the degree of variability in both the Bay and the Delta, modeling that requires linking dissolved Se to particulate Se should include several scenarios using different K_ds that are within a range of values constrained, as described above.

Uptake Into Food Webs

Kinetic bioaccumulation models (i.e., biodynamic models, Luoma and Fisher 1997; Luoma and Rainbow 2005, 2008) account for the now well-established principle that Se bioaccumulates in food webs principally through dietary exposure. Uptake attributable to dissolved exposure makes up less than 5% of bioaccumulated Se in almost all circumstances (Fowler and Benayoun 1976; Luoma and others 1992; Roditi and Fisher 1999; Wang and Fisher 1999; Wang 2002; Schlekat and others 2004; Lee and others 2006). Biodynamic modeling (submodels B and C, Figures 3 and 4) shows that Se bioaccumulation (the concentration achieved by the organism) is driven by physiological processes specific to each species (Reinfelder and others 1998; Wang 2002; Baines and others 2002; Stewart and others 2004). Biodynamic models have the further advantage of providing a basis for deriving a simplified measure of the linkage between trophic levels: TTFs. For each species, a TTF can be derived from either experimental studies or field observations.

Experimental derivation of TTFs is based on the capability of a species to accumulate Se from dietary exposure as expressed in the biodynamic equation (Luoma and Rainbow 2005):

$$dC_{species}/dt = (AE)\ (IR)\ (C_{food}) - (k_e + k_g)\ (C_{species}) \quad (2)$$

where $C_{species}$ is the contaminant concentration in the animals (µg g^{-1} dw), t is the time of exposure in days (d), AE is the assimilation efficiency from ingested particles (%), IR is the ingestion rate of particles (g g^{-1} d^{-1}), C_{food} is the contaminant concentration in ingested particles (µg g^{-1} dw), k_e is the efflux rate constant (d^{-1}) that describes Se excretion or loss from the animal, and k_g is the growth rate constant (d^{-1}). Key determinants of Se bioaccumulation are the ingestion rate of the animal, the efficiency with which Se is assimilated from food, and the rate constant that describe Se turnover or loss from the tissues of the animal (Luoma and Rainbow 2005; Presser and Luoma 2010a). Experimental protocols for measuring such parameters as AE, IR, and k_e are now well developed for aquatic animals (Luoma and others 1992; Wang and others 1996; Luoma and Rainbow 2005). The rate constant of growth is significant only when it is comparable in magnitude to the rate constant of Se loss from the organism. Consideration of the complications of growth can usually be eliminated if the model is restricted to a long-term, averaged accumulation in adult animals (Wang and others 1996).

In the absence of rapid growth, a simplified, resolved biodynamic exposure equation for calculating a Se concentration in an invertebrate (submodel B, Figure 3) is

$$C_{invertebrate} = [(AE)(IR)(C_{particulate})] \div [k_e] \quad (3)$$

For modeling, these physiological parameters can be combined to calculate a $TTF_{invertebrate}$, which characterizes the potential for each invertebrate species to bioaccumulate Se. $TTF_{invertebrate}$ is defined as

$$TTF_{invertebrate} = [(AE)(IR)] \div k_e \quad (4)$$

Similarly, foodweb biodynamic equations for fish or birds are

$$C_{fish\ or\ bird} = [(AE)\ (IR)\ (C_{invertebrate})] \div k_e \quad (5)$$

and

$$TTF_{fish\ or\ bird} = [(AE)\ (IR)] \div k_e \quad (6)$$

Where laboratory data are not available, TTFs can be defined from field data, where the TTF defines the relationship between Se concentrations in an animal and in its food in dw. The field $TTF_{invertebrate}$ must be defined from spatially and temporally matched data sets (in dw or converted to dw) of particulate and invertebrate Se concentrations (submodel B, Figure 3) as

$$TTF_{invertebrate} = C_{invertebrate} \div C_{particulate} \quad (7)$$

A field derived species-specific TTF_{fish} is defined as

$$TTF_{fish} = C_{fish} \div C_{invertebrate} \quad (8)$$

where $C_{invertebrate}$ is for a known prey species, C_{fish} is reported as muscle or whole-body tissue, and both Se concentrations are reported in µg g^{-1} dw (submodel B, Figure 3).

Whether the TTFs are determined from the laboratory or the field, the modeling approach is sufficiently flexible to represent complexities such as mixed diets. For example, a diet that includes a mixed proportion of prey in the diet can be addressed using the equation

$$C_{fish} = (TTF_{fish})\ [(C_{invertebrate\ a})\ (\text{prey fraction}) + (C_{invertebrate\ b})\ (\text{prey fraction}) + (C_{invertebrate\ c})\ (\text{prey fraction})] \quad (9)$$

Equations are combined to represent step-wise bioaccumulation from particulate material through invertebrates to fish (submodel B, Figure 3) as

$$C_{fish} = (TTF_{invertebrate})\ (C_{particulate})\ (TTF_{fish}) \quad (10)$$

Similarly, for birds, the combined equation is

$$C_{bird} = (TTF_{invertebrate})\ (C_{particulate})\ (TTF_{bird}) \quad (11)$$

Modeling can accommodate longer food webs that contain more than one higher trophic level consumer (e.g., forage fish being eaten by predatory fish) by incorporating additional TTFs. One equation for this type of example (submodel B, Figure 3) is

$$C_{\text{predator fish}} = (TTF_{\text{invertebrate}})\ (C_{\text{particulate}})\ (TTF_{\text{forage fish}})\ (TTF_{\text{predator fish}}) \tag{12}$$

Modeling for bird tissue also can represent Se transfer through longer or more complex food webs (e.g., TTFs for invertebrate to fish and fish to birds) as

$$C_{\text{bird}} = (TTF_{\text{invertebrate}})\ (C_{\text{particulate}})\ (TTF_{\text{fish}})\ (TTF_{\text{bird}}) \tag{13}$$

Variability or uncertainty in processes that determine AEs or IRs can be directly accounted for in sensitivity analysis (Wang and others 1996). This is accomplished by considering the range in the experimental observations for the specific animal in the model. Field-derived factors require some knowledge of feeding habits, and depend on available data for that species. Laboratory and field factors for a species can be compared and refined to reduce uncertainties in modeling (Presser and Luoma 2010a).

A substantial number of species-specific TTFs are available (Luoma and Presser 2009; Presser and Luoma 2010a). These are enough data at least to begin to model important food webs. Across invertebrate species, TTFs range from 0.6 to 23. Of the 29 species studied, 27 species have TTFs > 1. Thus, most invertebrate species bioaccumulate as much as or more Se than concentrated in the trophic level below them. In other words, the concentration of Se biogeochemically transformed into algae, microbes, seston, or sediments is preserved and/or (bio)magnified as Se passes up food webs. In general, TTFs for bivalves (clams, mussels, oysters) and for barnacles are the highest among species of invertebrates (i.e., an experimentally determined TTF range of approximately 4 to 23) (Presser and Luoma 2010a).

Trophic transfer factors from the available data for fish have a median of approximately one, and vary much less than among invertebrates: from 0.5 to 1.8 (Presser and Luoma 2010a). Compilations show that TTFs derived from laboratory biodynamic experiments range from 0.51 to 1.8; TTFs for different fish species derived from field studies are similar, ranging from 0.6 to 1.7.

Trophic transfer factors for aquatic birds (diet to bird egg) are less well developed, and laboratory data are limited (Presser and Luoma 2010a). The most robust data from the laboratory relate Se concentrations in the diet (as seleno-methionine) to egg Se concentrations from controlled feeding of captive mallards (*Anas platyrhynchos*). The range of $TTF_{\text{bird egg}}$ calculated from the compilation of nominal experimental diet Se concentrations and mean egg Se data given in Ohlendorf (2003) for mallards is 1.5 to 4.5. Using the detailed data from Heinz and others (1989) narrows this range to 2.0 to 3.9, with a mean of 2.6. Field data could be used to refine $TTF_{\text{bird egg}}$ on a site-specific basis, but variability in food sources and habitat use may add uncertainty to such data, and limits applications among habitats.

Exposure: Food Webs, Seasonal Cycles, and Habitat Use

Selenium is at least conserved and usually biomagnified at every step in a food web (Presser and Luoma 2010a). Selenium toxicity is generally assumed to be observed first in specific predator species as differences in food web exposure are propagated up trophic pathways (Luoma and Rainbow 2005; Stewart and others 2004). Some invertebrate species also may be susceptible to environmentally relevant Se concentrations (Conley and others 2009, 2011). Selenium is usually not detoxified in animal tissues by conjugation with metal-specific proteins or association with non-toxic inclusions (Luoma and Rainbow 2008). Hence, general mechanisms that semi-permanently sequester metals in non-toxic forms and lead to progressive accumulation with size or age probably are less applicable to the metalloid Se than to metals in general (Luoma and Presser 2009).

Predator population distribution, feeding preference, prey availability, life stage, gender, physiology, and species sensitivity are all variables that influence how a predator is affected by Se. Field factors such as varying weather, water depth, human disturbance, and food dispersion also affect foraging energetics, and accessibility of contaminants in foods on a localized level. Despite these complexities, some generalizations are possible at the present state of understanding. Predator species for the Bay-Delta, their food webs, and potential exposure are shown in submodels C and D (Figures 4 and 5), with further supporting information compiled in Appendix A.2 and A.3.

Based upon studies of invertebrate bioaccumulation the greatest exposures to Se will occur in predators that ingest bivalves in the Bay-Delta (Stewart and others 2004; Presser and Luoma 2006, 2010b). The estimated maximum percentages of diet that are clam-based for various benthic predators were estimated by the USFWS (2008a) (submodel C, Figure 4): lesser scaup 96%; surf scoter 86%; greater scaup 81%; black scoter 80%; white-winged scoter 75%; California clapper rail 64%; bald eagle 23%; white sturgeon (and assumed for green sturgeon) 41%; and Sacramento splittail (2-year olds) 34%. Dietary estimates are not specific to *C. amurensis*, but a bivalve component to diet in general. Bald eagles are an example of a predator with a diet wherein 23% are those waterfowl (scaups and scoters) that primarily feed on benthic mollusks (USFWS 2008a). Clapper rails feed on benthic food webs, but are littoral feeders that usually do not eat *C. amurensis*, which is mostly subtidal. Figure 4 (submodel C) also shows potential food webs for Dungeness crab. Diet component data and kinetic loss rates are not documented for life stages of this crustacean, but isotopic data indicate that clams such as *C. amurensis* would be expected to be an important food for this species (Stewart and others 2004). Selenium concentration data, in turn, indicate that predators of this crab would be subjected to elevated dietary Se concentrations (submodel C, Figure 4).

Food webs illustrated for Delta inhabitants include aquatic insects to salmonids (submodel C, Figure 4). The diets of salmon and steelhead trout are dominated by species with TTFs lower than bivalves. These species thereby incur less dietary Se exposure than molluscivores. Field data for Se concentrations are limited to 1986 to 1987 for Chinook salmon (Saiki and others 1991) and absent for steelhead trout that inhabit the estuary and migration corridors. Although their exposures are not exceptionally high, these species may be vulnerable because of their toxicological sensitivity to Se (USFWS 2008a, 2008b; Janz 2012). Delta smelt are endemic to the estuary and are included here because population numbers for the Delta smelt are alarmingly low. Thus, the USFWS (2008a) concluded that this species is particularly vulnerable to any adverse effect. It should be noted, however, that the feeding habits of Delta smelt would not suggest high exposures compared to other species, and sensitivity or bioaccumulation data are not available.

Not all predators reside in the estuary throughout their lives. When a predator is present across flow seasons and during critical life stages may influence Se exposure and effects. Predator seasonal cycle diagrams are shown for migratory birds (scoter and scaup); breeding birds (California clapper rail, bald eagle); migrating/rearing juveniles (Chinook salmon, steelhead trout); and breeding fish (green sturgeon, white sturgeon, and Sacramento splittail) (submodel D, Figure 5). The North Bay is part of the migration corridor and feeding ground for anadromous fish such as white sturgeon, Chinook salmon, and striped bass. The estuary also serves seasonally as a nursery area for species that spawn either in freshwater (e.g., Sacramento splittail) or in the ocean (e.g., Dungeness crab). Migrating diving ducks on the Pacific flyway winter and feed in the estuary as they stage for breeding in the freshwater ecosystems of the boreal forests of Canada and Alaska (De La Cruz and others 2009). As migratory waterfowl move north to breed in the spring, there is the potential for depuration of Se (USFWS 2008a; Appendix A.2 and A.3).

Some of the highest *C. amurensis* Se concentrations of the annual cycle occur when overwintering scoter and scaup actively feed in Suisun Bay and San Pablo Bay during the fall and early winter, (Linville and others 2002; Kleckner and others 2010) (submodel D, Figure 5). Long-lived white sturgeon feed predominantly on *C. amurensis* and have a two-year internal egg maturation that makes them particularly vulnerable to loading of Se in eggs and reproductive effects (Linville 2006). As an indication of this potential, Linares and others (2004) found Se concentrations as high as 47 $\mu g\ g^{-1}$ dw in immature gonads of 39 white sturgeon captured in the estuary. In earlier studies, Kroll and Doroshov (1991) reported that Se concentrations in developing ovaries

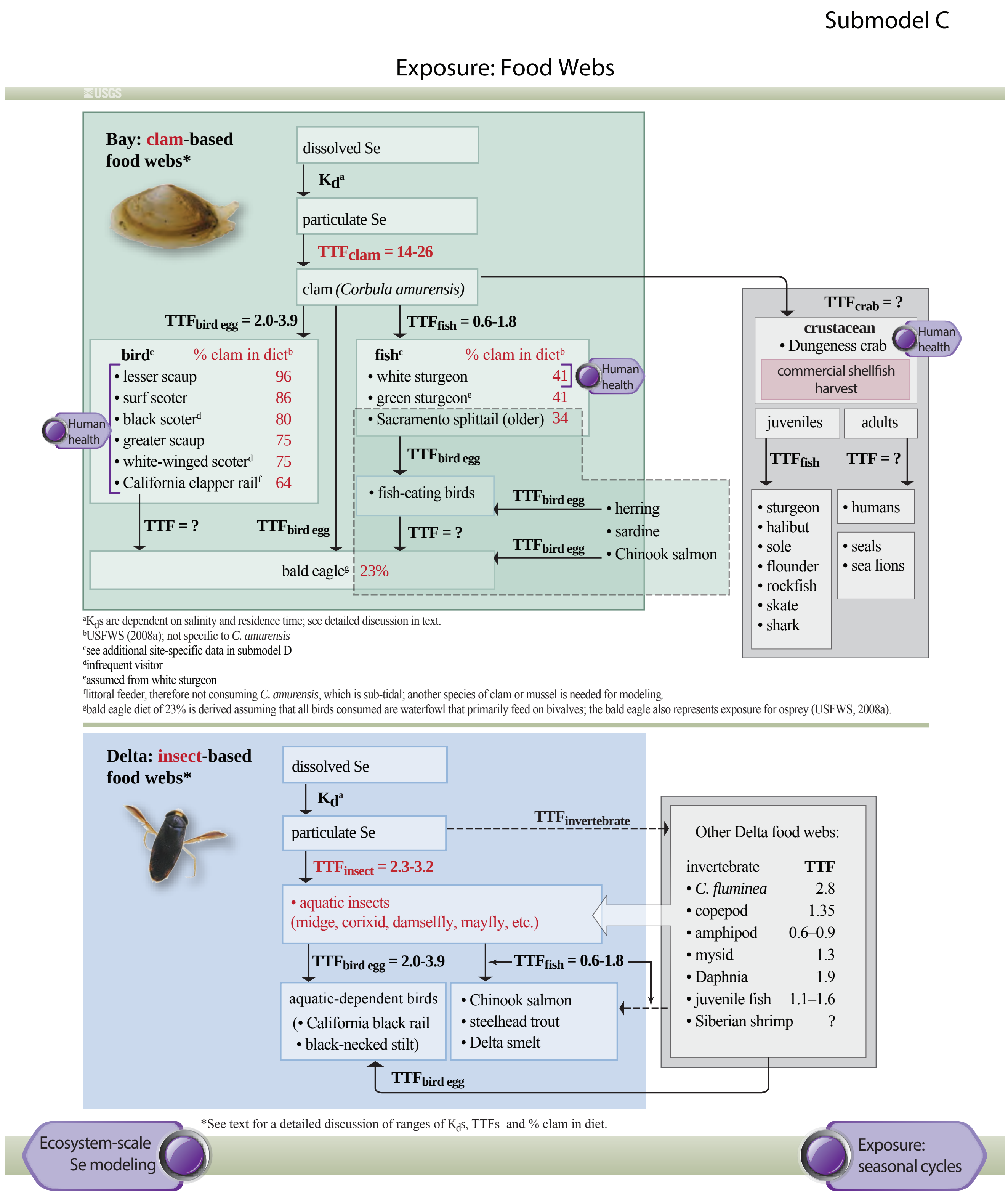


Figure 4 Submodel C. Exposure: Food Webs

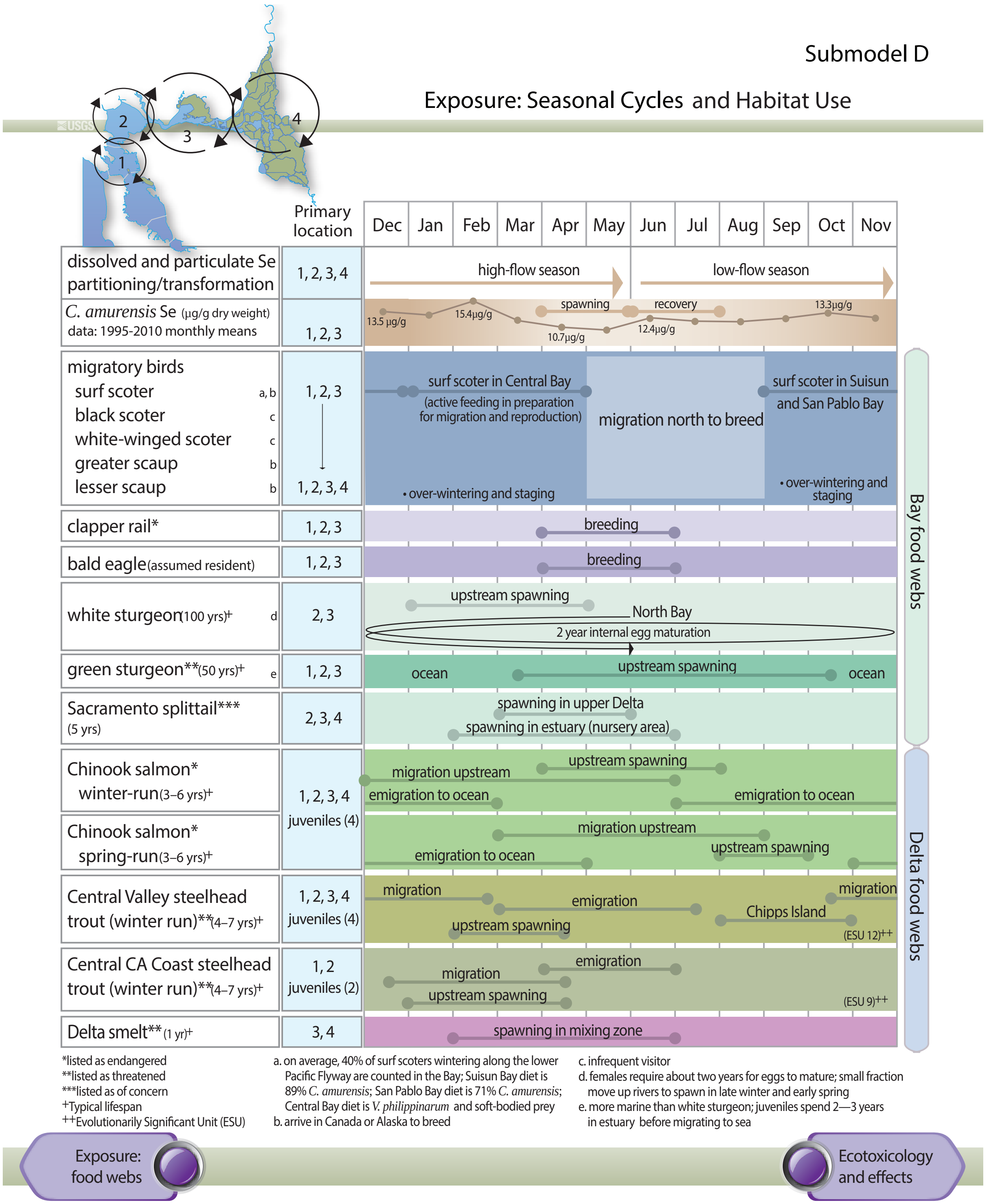


Figure 5 Submodel D. Exposure: Seasonal Cycles and Habitat Use

of white sturgeon from the Bay contained maxima of 72 µg $g^{-1}$and 29 µg g^{-1}. This range of wild white sturgeon reproductive tissue Se concentrations approach or exceed levels that cause severe deformities and mortalities in newly hatched larvae (Lemly 2002; Linville 2006). Larger, older Sacramento splittail also feed on *C. amurensis* and they are known to spawn both in the upper Delta and estuary (Stewart and others 2004). Modeling for species such as clapper rail would need specifics of diet composition (i.e., which species of clam, mussel, or crab is consumed), and whether prey species are efficient bioaccumulators of Se. Formalized, detailed knowledge such as this (submodel D, Figure 5), in turn, helps set choices in comparative modeling scenarios.

Fish and Wildlife Health: Ecotoxicology and Effects

Toxicity arises when dissolved Se is transformed to organic-Se by bacteria, algae, fungi, and plants (i.e., synthesis of Se-containing amino acids *de novo*) and then passed through food webs. It is generally thought that animals are unable to biochemically distinguish Se from sulfur, and therefore excess Se is substituted into proteins and alters their structure and function (Stadtman 1974). Other biochemical reactions also can determine and mediate toxicity (Chapman and others 2010). The effect of these reactions is recorded, most importantly in birds and fish, as failures in hatching or proper development (teratogenesis or larval deformities) (submodel E, Figure 6). Other toxicity endpoints include growth, winter survival, maintenance of body condition, reproductive fitness, and susceptibility to disease (submodel E, Figure 6; Appendix A.3). Specifically, Se can alter hepatic glutathione metabolism to cause oxidative stress (Hoffman and others 1998, 2002; Hoffman 2002) and diminished immune system function (Hoffman 2002).

Details of general ecotoxicological pathways of Se for fish and birds and effects of concern for Se are shown in submodel E (Figure 6). As represented here, birds and fish differ in how Se taken up from diet distributes among tissues (submodel E, Figure 6). Physiological pathways shown here for birds emphasize an exogenous dietary pathway and for fish an endogenous liver pathway. Species-specific Se effect models for the Bay-Delta are shown for breeding clapper rail; migratory scoter and scaup; white sturgeon; downstream-migrating juvenile salmonids; and upstream-migrating adult salmonids (submodel F, Figure 7). Details of Se-specific toxicological information for predator species considered here are compiled in Appendix A.3.

Such health effects are important to the overall ability of birds and fish to thrive and reproduce. But the consequences of Se transfer from the mother to her progeny via each reproductive stage are the most direct and sensitive predictors of the effects on birds and fish (Heinz 1996; Lemly 2002; Chapman and others 2010). Ultimately, it would be expected that effects on reproduction, especially in slowly reproducing, demographically vulnerable species (e.g., sturgeon), could lead to effects on populations and community changes.

To translate exposure into toxicity, effects levels are needed for predator species. Traditionally, guidelines relate Se concentrations in water to effects. But it is increasingly recognized that the concentrations of Se bioaccumulated in fish and bird tissues are more strongly related to signs of toxicity in nature, and would provide less ambiguous guidelines (Chapman and others 2010). The best correlations occur between Se in reproductive tissue and effects on reproductive processes. To assess implications of Se contamination in water from such relationships a bioaccumulation model is, then, necessary.

Experimental determination of tissue Se concentrations at which adverse effects occur is influenced by choice of endpoint, life-stage, dietary form, route of transfer, and choice of effect concentration. Another consideration in determining the guideline is the steepness of the Se dose-response curves and the choice of mathematical models to describe the curve (Skorupa 1998; Ohlendorf 2003; Lemly 2002; Environment Canada 2005; Beckon and others 2008; Chapman and others 2010). Effect guidelines that focus on a combination of the most sensitive assessment measures might include, for example, a selenomethionine diet, parental exposure, and embryonic or larval life-stage effect (Presser and Luoma 2006).

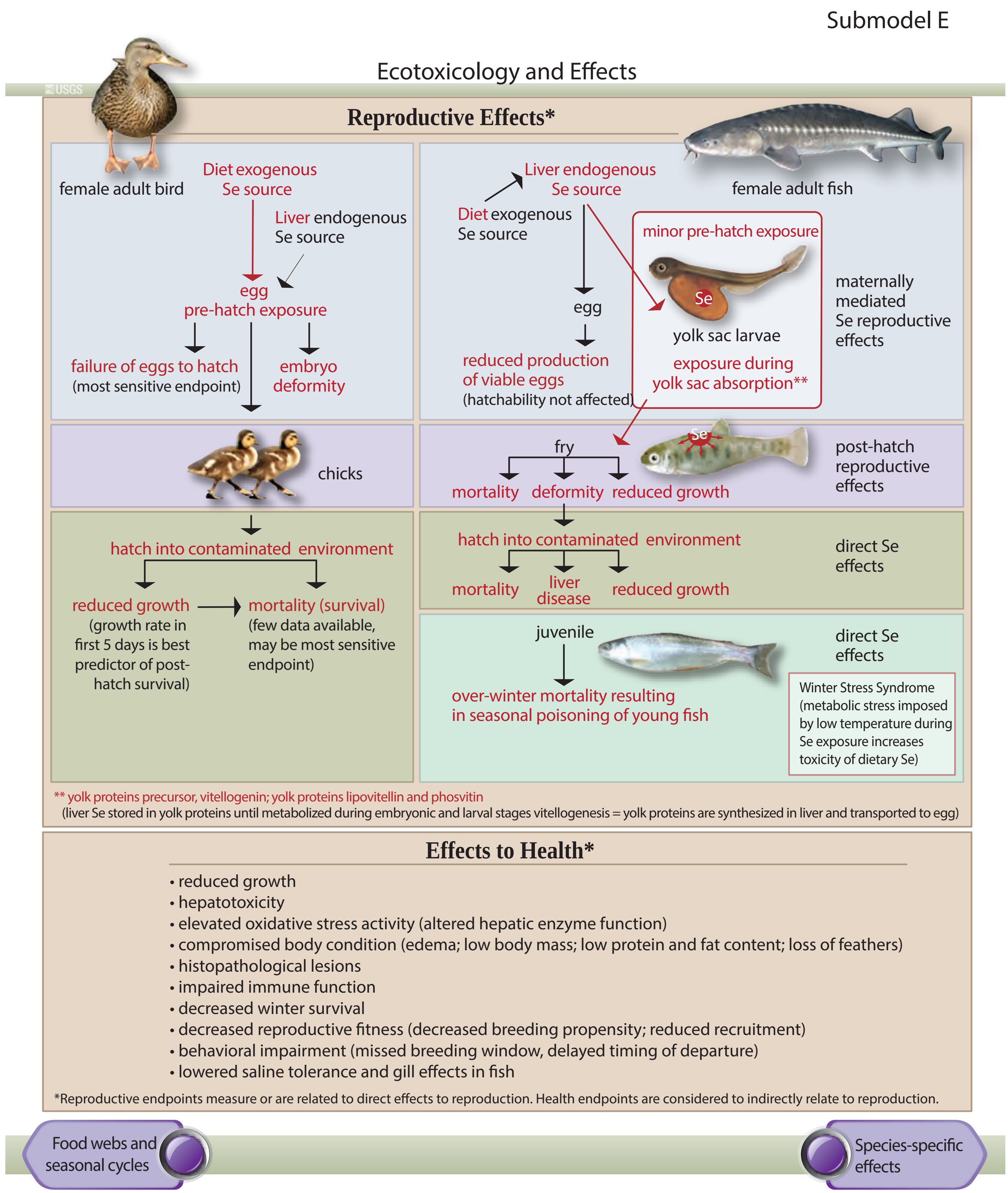


Figure 6 Submodel E. Ecotoxicology and Effects

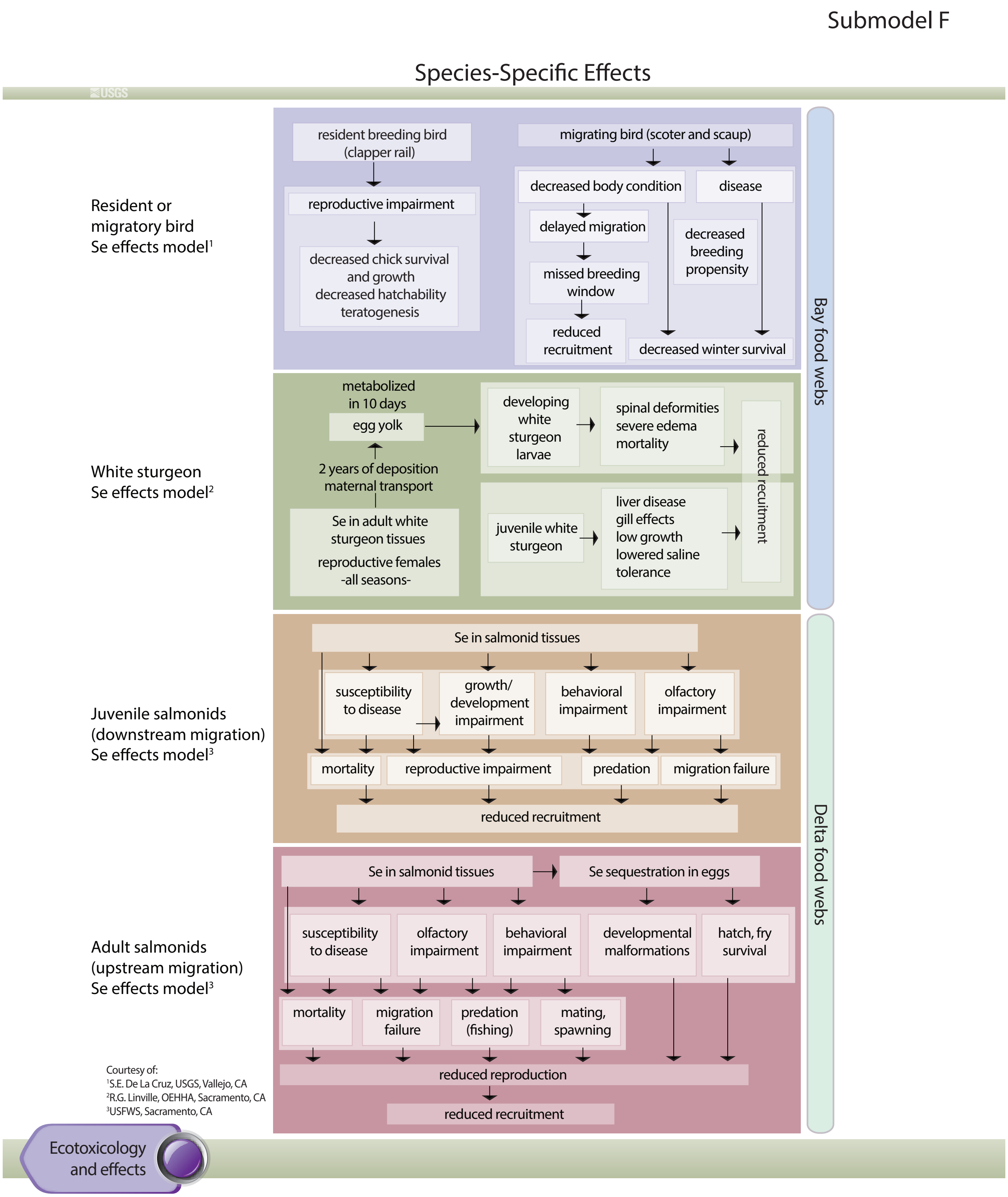


Figure 7 Submodel F. Species-Specific Effects

Even then the choice of statistical analysis and effect level can lead to disagreement about effect guidelines.

Human Health

A number of species from the Bay-Delta are consumed by humans (submodel G, Figure 8). Human health advisories against consumption of greater scaup, lesser scaup, and scoter because of elevated Se levels have been in effect since 1986 (Presser and Luoma 2006) for Suisun Bay, San Pablo Bay, Central Bay, and South Bay (CDFG 2012, 2013). The health warning states that no one should eat more than four ounces of scaup meat per week or more than four ounces of scoter meat in any two week period. Further, no one should eat the livers of ducks from these areas.

Fish consumption advisories, including for white sturgeon, exist for the Bay because of the effect of mercury and PCBs (OEHHA 2011, 2012). Pesticides, flame retardants, and Se also were tested, but a mean concentration calculated for each fish species collected from locations throughout the Bay-Delta over a range of years was found to be below that chemical's advisory tissue level (OEHHA 2011, 2012). Specifically for Se, concentrations in white sturgeon ($n = 56$ during 1997 to 2009, or 4.3 fish per year) were higher than other species of fish tested; and some Se concentrations for white sturgeon collected in North Bay locations (maximum 18.1 µg g^{-1} dw) exceeded Se advisory levels (e.g., 10.4 µg g^{-1} dw or 2.5 µg g^{-1} wet weight based on consumption of three 8-ounce meals per week (OEHHA 2011, 2012). Length restrictions (117 to 168 cm) and a bag limit of one fish per day are in effect for legal fishing of white sturgeon in the Bay, with a mean of 134 cm measured in fish collected for advisories.

A median per angler consumption rate of 16 g d^{-1} was determined specifically for Bay fish during 1998 and 1999 (SFEI 2000). This site-specific rate can be compared to a national recreational fisher consumption rate of 17.5 g d^{-1} and a national per capita rate of 7.5 g d^{-1} (USEPA 2000b).

Nutritional guidelines, toxicity symptoms, and national guidance concerning human health risk for consumption of fish are shown in submodel G (Figure 8). The details of how guidelines shown in Figure 8 were determined and how they might be linked to regulation of Se in wildlife and to fish health are presented in Appendix A.4.

QUANTITATIVE MODELING

This section presents an example of an application of the quantitative DRERIP Ecosystem-Scale Selenium Model. The questions addressed in this example are: What are the implications for ecosystem concentrations of Se if a fish tissue and/or wildlife Se guideline is implemented (a guideline based upon Se concentrations in a predator)? More specifically, what changes in dissolved or particulate Se concentration in the Bay-Delta would be necessary to achieve the selected tissue concentrations in predators? Agencies have traditionally regulated contaminants on the basis of dissolved concentrations, and managed inputs from different sources based upon their implications for dissolved concentrations (e.g., total mass daily loadings). This example shows a methodology that ties the new concept of tissue guidelines to the traditional concept of dissolved-concentration-based management. Inherent in every regulatory guideline are assumptions about the environment being regulated. The model allows an explicit evaluation of the implications of different assumptions.

The generalized equations for prediction of a dissolved Se concentration from a tissue Se concentration are given in submodel B (Figure 3). Table 1 gives the specific combinations of choices for food web, guideline, location, hydrologic condition, K_d, and TTFs used for the Bay-Delta application. In this example, several alternatives for a tissue guideline were chosen from among those that have been discussed in the regulatory context. Then, the invertebrate, particulate, and dissolved Se concentrations were calculated that would be expected if the tissue concentrations were in compliance with each choice of a guideline. Calculations also were conducted under different assumptions about K_d, food web, and TTFs. Finally, the calculated dissolved, particulate,

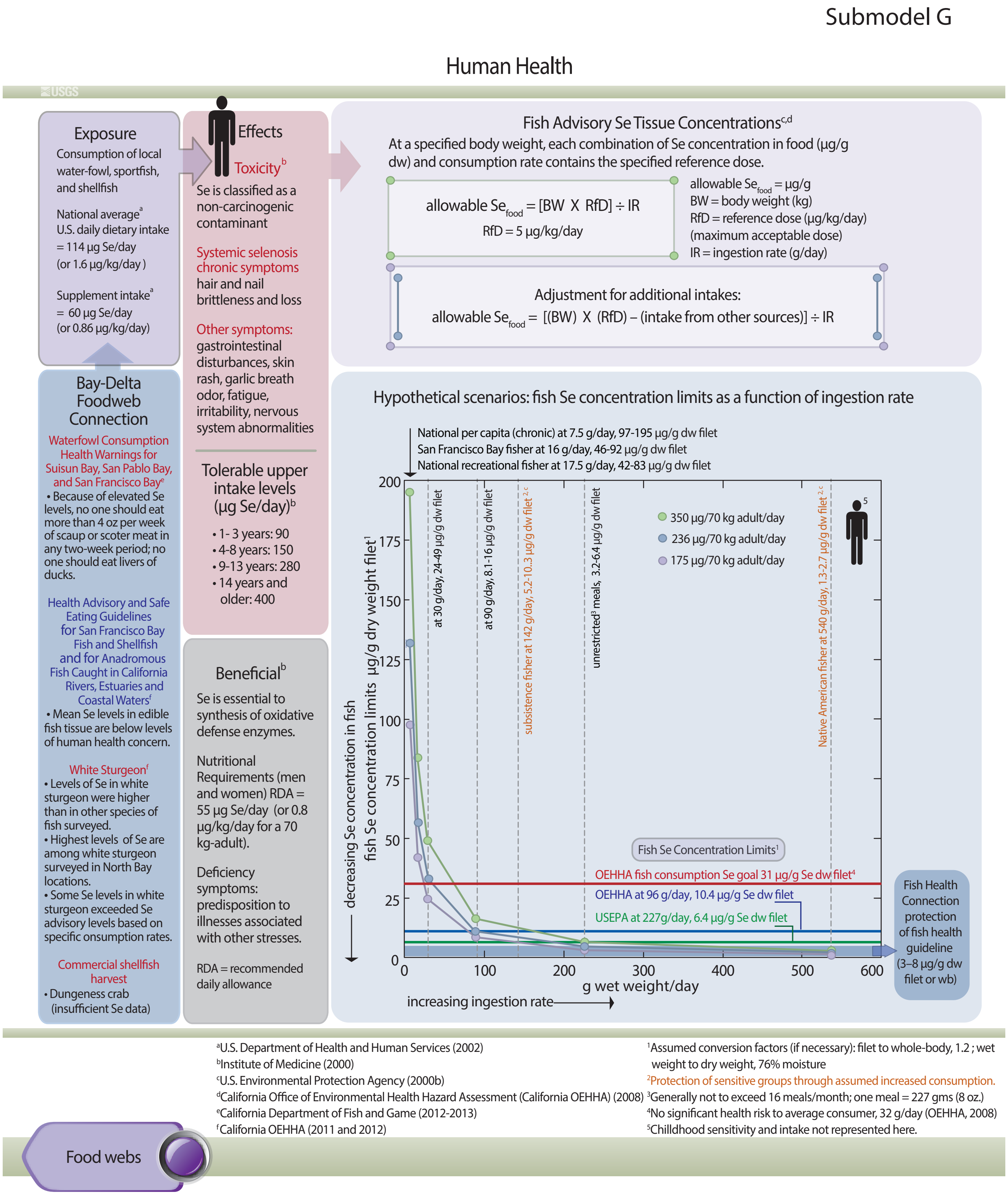


Figure 8 Submodel G. Human Health. See additional explanation in Appendix A.4.

Table 1 Locations, food webs, and model parameters for quantitative modeling examples

Location	Predator	Food web	Predator tissue target ($\mu g\ g^{-1}$ Se, dw)	$TTF_{predator}$	Prey	TTF_{prey}	Particulate phase as base of food web	K_d	Flow condition
San Francisco Bay (Carquinez Strait – Suisun Bay)	sturgeon	clam-based	5 or 8 whole-body	1.3	50% *C. amurensis* 50% [amphipods plus other crustaceans]	9.2	suspended particulate material	5,986	low flow (Nov 1999)
	sturgeon	clam-based	5 or 8 whole-body	1.3	50% *C. amurensis* 50% [amphipods plus other crustaceans]	9.2	suspended particulate material	3,317	average condition
	young striped bass	zooplankton-based	8 whole-body	1.1	zooplankton	2.4	suspended particulate material	3,317	average condition
	bird	clam-based	7.7, 12.5, or 16.5 egg	2	50% *C. amurensis* 50% [amphipods plus other crustaceans]	9.2	suspended particulate material	5,986	low flow (Nov 1999)
	bird	clam-based	7.7, 12.5, or 16.5 egg	2	50% *C. amurensis* 50% [amphipods plus other crustaceans]	9.2	suspended particulate material	3,317	average condition
Sacramento–San Joaquin Delta	fish	insect-based	5 or 8 whole-body	1.1	aquatic insects	2.8	suspended particulate material	3,680	average condition
	bird	insect-based	7.7, 12.5, or 16.5 egg	2	aquatic insects	2.8	suspended particulate material	3,680	average condition
San Joaquin River (main stem at Vernalis)	fish	insect-based	5 or 8 whole-body	1.1	aquatic insects	2.8	suspended particulate material	1,212	generalized (July 2000)

and invertebrate Se concentrations were compared with observations of those values from the Bay-Delta to assess how much existing conditions would be need to change to achieve compliance with the chosen guidelines (Table 2). Implicitly, comparisons of outcomes with data from nature tests how well model predictions match reality (Luoma and Rainbow 2005). Comparisons under different assumed conditions test the sensitivity of the model to changes within a few critical parameters.

The method, as indicated in the conceptual model (Figures 3 and 4, especially) includes the following steps: (1) selection of tissue guidelines to test; (2) selection of places and times of interest; (3) derivation of K_d using spatially and temporally matched dissolved and particulate Se concentrations constrained within the selected place and time; (4) selection of a food web of interest to each locality; (5) determination of species-specific TTFs for invertebrates and their specific predators that are relevant to the place and food web; (6) prediction of invertebrate, particulate and dissolved Se concentrations; (7) comparison of predicted values to field observations of Se concentrations in these media in the Bay-Delta; and (8) conclusions about implications for compliance.

Modeling Parameters and Variables

Guidelines

The effect guidelines chosen for evaluation were 5 and 8 $\mu g\ g^{-1}$ dw fish whole-body; as well as 7.7, 12.5, and 16.5 $\mu g\ g^{-1}$ dw for bird eggs (Presser and Luoma 2010b) (Table 1). The regulatory community is debating appropriate critical tissue values that relate bioaccumulated Se concentrations and toxicity in predators (see previous discussion). We are not

Table 2 Predicted dissolved and particulate Se concentrations and percent exceedances for example scenarios

Location	Flow condition and tissue guideline ($\mu g\ g^{-1}$ Se, dw fish whole-body or bird egg)	Predicted invertebrate concentration ($\mu g\ g^{-1}$ Se, dw)	Predicted particulate concentration ($\mu g\ g^{-1}$ Se, dw)	Percent particulate Se exceedance in ecosystem	Predicted dissolved concentration ($\mu g\ L^{-1}$ Se)	Percent dissolved Se exceedance in ecosystem
San Francisco Bay: Carquinez Strait – Suisun Bay						
Bay sturgeon	low flow – 5.0	3.8	0.42	59	0.070	100%
	average – 5.0	3.8	0.42	59	0.126	47%
	low flow – 8.0	6.2	0.67	27	0.112	66%
	average – 8.0	6.2	0.67	27	0.202	3%
Bay striped bass	average – 8.0	7.3	3.0	0	0.914	0%
Bay birds	low flow – 7.7	3.9	0.42	59	0.070	100%
	average – 7.7	3.9	0.42	59	0.126	47%
	low flow – 12.5	6.3	0.68	25	0.113	64%
	average – 12.5	6.3	0.68	25	0.205	2%
	low flow – 16.5	8.3	0.90	11	0.150	23%
	average – 16.5	8.3	0.90	11	0.270	1%
Sacramento–San Joaquin Delta						
Delta fish	average – 5.0	4.5	1.6	7	0.441	19%
	average – 8.0	7.3	2.6	3	0.706	10%
Delta birds	average – 7.7	3.9	1.4	16	0.374	19%
	average – 12.5	6.3	2.2	3	0.607	11%
	average – 16.5	8.3	2.9	3	0.801	6%
San Joaquin River (main stem at Vernalis)						
River fish	July 2000 – 5.0	4.5	1.6	No data	1.3	16%
	July 2000 – 8.0	7.3	2.6	No data	2.1	3%

suggesting these are the best choices for guidelines; but they are within the range of those that are being discussed. In particular, the fish whole-body target of 5 $\mu g\ g^{-1}$ and a bird egg target of 7.7 $\mu g\ g^{-1}$ have been derived to provide additional protection for endangered species (Skorupa and others 2004; Skorupa 2008). The illustrated scenarios also considered the differences in the changes required if a bird egg-based guideline were used instead of a whole-body fish-based guideline.

Place and Time

The modeling scenarios compared two locations: a brackish-water Bay environment and a tidal freshwater Delta environment. For the Bay, we constrained consideration to the geographic area of Carquinez Strait and Suisun Bay (Presser and Luoma 2010b) (Table 1). In terms of drivers, this location is affected by oil-refinery effluents that contain Se, and also could be influenced by inputs from the San Joaquin Valley. As noted previously, Se concentrations in at least some predators (sturgeon and diving ducks) at this location now exceed USFWS Se guidelines (Presser and Luoma 2010b). For the Delta, the area considered was from Stockton westward through the Delta, and was constrained to the freshwater environment. We also compared scenarios for average conditions across the year(s) in the Bay, to a specific example of conditions for one low flow season

(November 1999). An average condition for the Delta was modeled.

Partitioning and K_ds

The approach of Presser and Luoma (2006, 2010b) was used to select two K_ds for the scenarios from the Bay and one for the Delta (Table 1). The data for the Bay were narrowed to a Carquinez Strait–Suisun Bay location (Cutter and Cutter 2004; Doblin and others 2006; Presser and Luoma 2010b) to focus on the most contaminated area in the estuary, and to exclude the extreme K_ds at the ocean and freshwater interfaces. We selected the mean of co-collected dissolved and particulate Se concentrations from a transect for November 1999 (K_d = 5,986) to represent low flow conditions. Average conditions in the Bay across all seasons and several years were represented by the grand mean of all transects through the Carquinez Strait–Suisun Bay area during 1998-1999 (K_d =3,317) and the freshwater Delta during 2003-2004 (K_d = 3,680). For comparison, the Delta grand mean K_d for low flow transects was 2,613 and for high flow transects 5,283. As discussed earlier, the value that describes transformation, even when constrained, is the most variable of any of the model parameters. The uncertainty associated with the choice of this value could be avoided if environmental guideline were based upon empirically determined particulate Se, but cannot be avoided if it is necessary to relate tissue Se to dissolved Se.

Food Webs and TTFs

For the Bay, the food web used was for suspended particulate material to *C. amurensis* to clam-eating fish or aquatic-dependent clam-eating bird (submodel C, Figure 4 and Table 1). The diet for both predators was assumed to be 50% clam and 50% benthic crustaceans. The bivalve food web is the most efficient at accumulating Se in the system, in both the field and in the quantitative model; therefore, it is the most environmentally protective to use in evaluating a tissue guideline. Different assumptions, of course, could be used for the percentage of diet that is clam-based (e.g., 75% to 96% for scoter and scaup, submodel C, Figure 4). Data on variability of benthic assemblages with time, Bay location, and hydrologic condition also can be used to adjust dietary considerations (Peterson and Vayssieres 2010). If migrating scoter and scaup were modeled, a guideline based on body-condition endpoint, rather than a direct reproductive guideline, would be appropriate. To test the sensitivity of the choice of predator, one comparative simulation was calculated for a pelagic food web in the Bay: suspended material to zooplankton to young striped bass. The food web for the Delta was suspended particulate material to aquatic insects to juvenile salmon or steelhead trout.

Only a few recent data sets from the Bay-Delta are available that analyze Se concentrations across a reasonably complete food web (e.g., Stewart and others 2004). Some important food webs have not been assessed at all (e.g., aquatic insects and Chinook salmon or steelhead trout) (Presser and Luoma 2010b). However, studies of Se concentrations in enough individual predator and prey species are available to assess the predictions from the model and to derive, in a few instances, some critical trophic transfer relationships (e.g., Linville and others 2002; Stewart and others 2004; Schwarzbach and others 2006; Lucas and Stewart 2007; De La Cruz and others 2008; De La Cruz 2010). For the Bay, the dominant bivalve in the Carquinez Strait–Suisun Bay area is *C. amurensis*. This species strongly bioaccumulates Se (Linville and others 2002). A species-specific $TTF_{C.\ amurensis}$ of 17 (a range of 14 to 26 over different estuary conditions) was used here based on the field calibration that Presser and Luoma (2010b) describe. Benthic crustaceans, like amphipods and isopods, are much less efficient than clams in bioaccumulating Se; TTFs can range from 0.8 for amphipods to 2.0 for other crustaceans (Presser and Luoma 2010a). Under the assumption of a mixed diet of *C. amurensis* ($TTF_{C.\ amurensis}$ = 17) and benthic crustaceans ($TTF_{benthic\ crustacean}$ = 0.8 and 2.0), the combined diet TTF used here is 9.2.

An important benthic predator, white sturgeon, was chosen for the example, because the Se biomagnifier *C. amurensis* is an important food source for this species in the Bay. White sturgeon accumulate higher concentrations of Se than any other fish in the Bay (Stewart and others 2004; OEHHA 2011), making it the environmentally conservative choice for evaluating a guideline. From studies in the late 1980s, field TTFs derived specifically for white sturgeon from the Bay that used bivalves as prey, showed a range from 0.6 to 1.7, with a mean of 1.3 (Presser and Luoma 2006); similar to the value of 1.1, which is the mean among all fish species studied. Calculations from more recent data sets for *C. amurensis* at Carquinez Strait, and seaward white sturgeon, showed a somewhat lower TTF of 0.8 (Presser and Luoma 2010b).

For the Delta food web, Se TTFs for freshwater aquatic insects were selected from data from literature sources (submodel C, Figure 4). For example, Presser and Luoma (2010a) derived a mean Se TTF_{insect} of 2.8 (range 2.3 to 3.2) based on matched field data sets for particulate and insect Se concentrations in freshwater environments for several species of aquatic insect larvae including mayfly, caddisfly, dragonfly, midge, and waterboatman. These values generally compare well to laboratory-derived TTFs for aquatic insect larvae (Conley and others 2009). TTFs for other potential invertebrates in Delta food webs (range 0.6 to 2.8) also are shown in submodel C, Figure 4 (Presser and Luoma 2010a).

Much less data are available to evaluate bioaccumulation in avian food webs. Data from the study of toxicity in mallards (Heinz and others 1989, 1990) are the most comprehensive studies available to use for modeling dietary exposure. From these studies, the laboratory-derived $TTF_{bird\ egg}$ of 2.6 was assumed for transfer of Se from prey to bird eggs (which correlate best with toxicity). For the model, this choice of TTF for bird species was lowered to 2.0 to illustrate the possible effect of field variables on exposure factors that encompass habitat use and feeding behavior. A diet of 50% clams and 50% crustaceans was assumed for a clam-eating bird.

Implications of Model Choices and Estuary Conditions

Details of the calculations to evaluate implications of different guidelines, under different conditions, are summarized in Table 2. To compare the implications of these choices, we determined the percentage Se concentrations in dissolved and particulate form that exceeded the value predicted to be necessary to meet the tissue guideline. All published dissolved (n = 168) and particulate Se (n = 168) data from the Bay and from the Delta, collected after 1997, are employed in this estimate. Together, the scenarios depict a Bay for which there is ecological risk from Se contamination, but the degree of risk, judged by the degree of compliance with the guidelines, depends heavily upon assumptions about toxicity (the guideline), transformation, and choice of food web.

The occurrence of 8 µg g^{-1} dw Se in sturgeon muscle from the contaminated area of San Francisco Bay (Linares and others 2004) is one of several lines of evidence that ecological risks from Se are occurring in the Bay. When this concentration was used for a predator guideline (Table 2), the model predicted Se concentrations in invertebrates and suspended particulate material and a dissolved Se concentration that were within the range typical of the Bay-Delta (Table 2). Thus, the model results appear to successfully capture the links between Se concentrations in different ecosystem components of the Bay, in general [also see Presser and Luoma (2010b) for further validation details]. This also suggests that the use of calibrated mean K_ds to reduce uncertainties about transformation adequately captures and constrains the variability in these processes. The agreement between ecosystem observations and the predicted Se concentrations in invertebrates and predators similarly points to the validity of the TTFs.

The most remarkable conclusion from the calculations is that fish tissue Se concentrations typical of risks to reproductive toxicity (the selected guideline examples) occur in the Bay at dissolved Se concentrations more than ten times less than the traditional water quality regulatory guideline of 5 µg L^{-1} (Table 2). At least some food webs in the Bay and the Delta are particularly vulnerable to small changes in bioavailable Se concentrations. The very high K_ds consistently observed in both the Bay and the Delta, compared to many other ecosystems (Presser and Luoma 2010a), may be one reason for this sensitivity. Also influential is the strong ability of invertebrates such as *C. amurensis* to bioaccumulate Se when compared to other prey species. It appears that ecosystems wherein dissolved Se is efficiently transformed to particulate Se, and in which particulate Se is propagated up a food web to predators, will amplify relatively small changes in concentrations of dissolved Se concentrations to levels that could affect predators.

Under low flow conditions, 23 to 66% of dissolved Se determinations in the Bay exceeded the value predicted to be necessary to meet the higher sturgeon-based guideline or the higher bird-based guidelines (Table 2). Under guidelines chosen to protect endangered species, 100% exceedance occurs at low flow conditions. Clearly, low flow conditions, like those in November 1999, are the time of greatest ecosystem sensitivity to Se inputs (as suggested by Presser and Luoma 2006). It is notable that the example presented here does not represent the most extreme condition of a low flow season of a dry year or critically dry year.

If annual average conditions are assumed (the mean of spatially constrained K_ds), compliance is much more sensitive to the choice of guideline. Few if any exceedances (1 to 3%) are observed if the higher fish or bird egg guidelines are implemented under that assumption. For endangered species protection under an average condition, exceedance is approximately 47% for both the fish and bird guidelines. Of course, regulations based upon average conditions run the risk of under-protecting species sensitive to Se exposure during the protracted time in every year (especially drier years) when Se is most bioavailable.

Considering the choice of different guidelines, if a 5 $\mu g\ g^{-1}$ guideline is implemented that uses sturgeon as the target organism, the entire Bay would be out of compliance. The model calculation suggests nearly all anthropogenic Se would have to be removed to drive sturgeon tissues to concentrations as low as 5 $\mu g\ g^{-1}$, especially during a low flow condition. The projected dissolved Se concentration necessary to reach that level in sturgeon tissue is approximately the value for the Sacramento River, and hence the pre-disturbance baseline condition for the Bay. The modeling results suggest that if it is assumed that 5 $\mu g\ g^{-1}$ represents the toxicity threshold for sturgeon, and if it were applied using concentrations in sturgeon from the field, then there is no room for any deviation from concentrations in the Sacramento River without risk to the species. It is important to remember, however, that this toxicity guideline was derived for the most sensitive fish species. So, the use of the most sensitive surrogate in the toxicity guideline combined with field determinations from the fish with the greatest exposure results in an ultra-sensitive outcome.

These model results also illustrate how sensitive the implementation of a tissue guideline can be to the choice of predator. For example, many of the differences between sturgeon-based guidelines and bird egg-based guidelines are relatively small. Both appear to be sensitive indicators of ecological risks. However, the outcomes of guidance based upon striped bass, a water-column predator, are quite different from outcomes based upon bird eggs or sturgeon. The model showed that while aquatic birds and sturgeon are at risk under most assumptions, few or no exceedances of Se concentrations occur if the choice of regulatory indicator is based upon striped bass tissues. The differences are the result of the different invertebrate prey of the two species. Sturgeon eat a diet that includes strong Se bioaccumulator species (bivalves); striped bass eat from prey that live in the water-column and do not strongly bioaccumulate Se.

Selenium concentrations in the water column or particulate material of the Delta are higher and more variable than in the Bay. Average K_ds are similar between the Delta and the Bay. Nevertheless, few exceedances of dissolved and particulate Se concentrations (3% to 19%) are predicted in the Delta, even when the most sensitive fish guideline is used. This is consistent with the observation of low Se concentrations in the few fish that have been sampled from the Delta (e.g., Foe 2010). Use of the local food web is extremely influential in this outcome. Bioaccumulation of Se in the aquatic insect larvae (and other arthropods) that are the primary prey species of most Delta fish and birds is much lower than bioaccumulation by bivalves. As a result, it appears that the Delta food webs are easier to protect from adverse effects of Se than benthic food webs in the Bay, even if it is assumed that the most sensitive fish guideline applies. Nevertheless, the actual concentrations of dissolved Se predicted to be necessary to meet the tissue guidelines range from 0.37 to 0.80 μg L^{-1}, far below the Se concentrations typical of most existing dissolved guidelines for Se (Luoma and Presser 2009). This reflects the unusually high K_ds consistently observed in this freshwater environment.

Few determinations of Se concentrations in particulate material in the incoming rivers to the Bay are available outside the tidal range. Lucas and Stewart (2007) reported matched dissolved and particulate Se concentrations from which one K_d could be calculated (a value of 1,212) for the San Joaquin River during transect sampling in 2000. The example in Table 2 shows that if that were typical of the river, and the food web was mainly based upon arthropods, then compliance with a tissue guideline could occur at dissolved Se concentrations ten times higher than would be the case in the Bay. This river simulation is based on very limited data; it is given here for comparative purposes to show the sensitivity of the model to the choice of hydrologic setting. Comprehensive modeling of the San Joaquin River system would require data collection and analysis specific to the river's settings, predator species, food webs, and habitats. Percentage exceedance (Table 2) is based on weekly sampling of total Se for the river at Vernalis from water year 1995 through water year 2010 (SWRCB 2012)

CONCLUSIONS

The DRERIP Ecosystem-Scale Selenium Model outcomes for the Bay-Delta show critical choices for Se modeling, and derived risk scenarios that illustrate varying degrees of risk, depending on those choices (Figure 1; Tables 1 and 2). In general, the conceptual model for Se shows that the focus of concern for this contaminant is the top of the food web. Quantitative model calculations show that enough is known to adequately characterize the distribution of Se through the Bay-Delta ecosystem, although the available data from which to validate the outcomes is dated and does not include conditions within a low flow season of a dry year or critically dry year. Presser and Luoma (2010b) give additional specifics for updated data collection and model refinements.

Selenium concentrations in fish or bird tissues alone appear to be good indicators of ecological risks from Se. Key invertebrates (e.g., the bivalve *C. amurensis* in the Bay) may be a more pragmatic indictor for frequent monitoring. Given that (1) suspended particulate material Se concentrations are key to accurate prediction of prey and predator Se concentrations; and (2) dissolved Se concentrations are constrained to a narrow dynamic range within the estuary, a suspended particulate material Se concentration also may be a sensitive parameter on which to assess change. Dissolved Se concentrations appear to be the variable of choice for regulatory agencies, however, because of links to total maximum daily loads.

The ability to quantitatively characterize distributions among all these ecosystem components from field determination of only one component allows great flexibility in future monitoring whatever the choice of indicator. The detailed site-specific conceptual model, and the ability to quantitatively apply that model, also provide perspective on the processes that are most influential in determining Se contamination in the predators of this Se-sensitive environment (Figure 1).

The quantitative example (Tables 1 and 2) provides some lessons for implementing regulations to manage Se in this system. First, it is notable that extremely small changes in dissolved Se concentrations, in absolute terms, have strong implications for compliance with the tissue guidelines. A regulatory program that focuses on dissolved Se would require an extremely rich data set to reliably detect the differences between compliance and non-compliance, based upon the translation from tissue to dissolved Se. This is another reason why regulation of suspended particulate material Se concentration may be a more sensitive parameter on which to assess change.

Second, if compliance is determined from tissue concentrations in a predator, the choice of that predator is crucial. Predators of bivalves in benthic food webs are much more at risk than predators from pelagic food webs. The former should be the basis of tissue monitoring in the Bay.

Third, any decision as to whether reductions in ambient concentrations of Se would be required to comply with the tissue guidelines depends upon the choice of guideline and assumed environmental conditions. For example, the modeling suggests that a fish tissue guideline of 5 µg g^{-1} would ultimately require essentially all enriched Se inputs to the Bay to be eliminated if the guideline were applied using Se concentrations in sturgeon. According to the calculations, dissolved Se concentrations in the Bay would have to decline to nearly those in the Sacramento River to comply with such a guideline. If a guideline of 8 µg g^{-1} was used, the Bay would be near compliance under average conditions; but 66% out of compliance in a situation like November 1999 (i.e., low flow). Calculating in the opposite direction from a traditional dissolved Se concentration guideline, allowing dissolved concentrations of Se in the Bay to reach 5 µg L^{-1} (the current regulatory guideline) or even 2 µg L^{-1} would result in tissue concentrations (potentially greater than 100 µg g^{-1} in *C. amurensis*) that could threaten many of the predators in the Bay, if other conditions stay as they are.

Fourth, the current food webs in the Delta are less at risk from Se than the benthic food webs of the Bay, because of the differences in food webs. The differences between the Delta and the Bay are not the result of the freshwater versus brackish water nature of the systems of interest because, on average, transformation efficiencies are similar in the two. Where transformation processes are greatly different between two ecosystems, then a different outcome from implementing the same tissue guideline might be expected. The San Joaquin River example shows how a less efficient transformation of dissolved Se to particulate Se in the river can result in less sensitivity of the ecosystem to changes in Se concentrations.

Finally, the more specificity added to the model, the less uncertainty in predictions. If, for example, the geographic range is narrowed by using data only from Carquinez Strait–Suisun Bay, then freshwater and ocean interfaces are avoided. If the temporal range is narrowed to low flow seasons of dry years (i.e., high residence time or high exposure time), then focus can be on times when the transformative nature of the estuary is elevated. Juxtaposition of times when suspended particulate material or prey species achieve maximum Se concentrations with critical life stages of species at risk being present allows regulatory considerations to focus on times that govern Se's ecological effects (i.e., ecological bottlenecks) (Figure 1).

The greatest strength of the analytical and modeling processes is that it is an orderly, ecologically consistent approach for assessing different aspects of the fate and effects of Se. Assessments such as the examples shown here can represent a starting point for initiating management decisions. Application of the DRERIP Ecosystem-Scale Selenium Model shows that management of Se requires incorporation of the complexity of dietary exposures and the systematic consideration of critical aspects of hydrology, biogeochemistry, physiology, ecology, and ecotoxicology to define ecosystem protection. Although this is complex, scenarios can be developed from specific questions that arise in the planning and implementation of restoration actions for the Bay-Delta. Quantitative evaluation of those scenarios is feasible. However, the Se database and monitoring program need to be modernized (e.g., refocused and expanded). Specifically, monitoring should include (1) representation of conditions in dry and critically dry years; and (2) collection of spatially and temporally matched data sets across media (i.e., water, suspended particulate material, prey, and predator) to ensure that derived site-specific factors are current for the ecological and hydrological dynamics of the Bay-Delta. Only then will predictions from the model remain relevant and realistic to a constantly evolving estuary.

ACKNOWLEDGEMENTS

This work was supported by CALFED (in the project's earlier phases), the U.S. Geological Survey National Research and Toxics Substances Hydrology Programs, and the U.S. Environmental Protection Agency (San Francisco, California, Region 9). Jeanne DiLeo (U.S. Geological Survey, Menlo Park Publishing Service Center) prepared the outstanding graphics. We thank William Beckon and Thomas Maurer (U.S. Fish and Wildlife Service, Sacramento, California) for provision of species at risk for the Bay-Delta. We also thank William Beckon, Susan De La Cruz (U.S. Geological Survey, Vallejo, California) and Regina Linville (Office of Environmental Health Hazard Assessment, Sacramento, California) for assistance with species-specific effects models and human health considerations. Thanks also go to Mark Huebner (U.S Geological Survey, Menlo Park, California) for exceptional assistance with manuscript preparation.

REFERENCES

Baines SB, Fisher NS, Stewart R. 2002. Assimilation and retention of selenium and other trace elements from crustacean food by juvenile striped bass (*Morone saxatilis*). Limnology and Oceanography 47(3):646–655.

Beckon WN, Parkins C, Maximovich A, Beckon AV. 2008. A general approach to modeling biphasic relationships. Environmental Science and Technology 42(4):1308–1314.

Besser JM, Huckins JN, Little EE, La Point TW. 1989. Distribution and bioaccumulation of selenium in aquatic microcosms. Environmental Pollution 62(1):1–12.

[CDFG] California California Department of Fish and Game. 2012–2013. Waterfowl and upland game hunting and hunting and other public uses on state and federal lands regulation. Sacramento (CA): California Department of Fish and Game. 80 p.

[CDFG] California California Department of Fish and Game. 2012. Current ocean fishing regulations: Pigeon Point to Point Conception: sturgeon. Sacramento (CA): California Department of Fish and Game. 6 p. Available from: *http://www.dfg.ca.gov/marine/mapregs4.asp* Accessed 11 June 2012.

[OEHHA, California OEHHA] California Office of Environmental Health Hazard Assessment. 2008. Development of fish contaminant goals and advisory tissue levels for common contaminants in California sport fish: chlordane, DDT's, dieldrin, methylmercury, PCB's, selenium and toxaphene. Sacramento (CA): California Environmental Protection Agency. 115 p. Available from: *http://www.oehha.ca.gov/fish/gtlsv/crnr062708.html* Accessed 11 June 2012.

[OEHHA, California OEHHA] California Office of Environmental Health Hazard Assessment. 2011. Health Advisory and safe eating guidelines for San Francisco Bay Fish and Shellfish. Sacramento (CA): California Environmental Protection Agency. 53 p. Available from: *http://oehha.ca.gov/fish/nor_cal/pdf/SFBayAdvisory21May2011.pdf* Accessed 11 June 2012.

[OEHHA, California OEHHA] California Office of Environmental Health Hazard Assessment. 2012. Health Advisory and safe eating guidelines for American shad, Chinook (King) salmon, steelhead trout, striped bass, and white sturgeon caught in California rivers, estuaries, and coastal waters. Sacramento (CA): California Environmental Protection Agency. 27 p. Available from: *http://www.oehha.ca.gov/fish/special_reports/pdf/AnadromousSppAdvisory.pdf* Accessed 18 March 2013.

[SFBRWQCB] California San Francisco Bay Regional Water Quality Control Board. 2007. Project plan north San Francisco Bay selenium TMDL. Oakland (CA): California San Francisco Bay Regional Water Quality Control Board. 30 p. Available from: *http://www.swrcb.ca.gov/rwqcb2/water_issues/programs/TMDLs/seleniumtmdl.shtml* Accessed 11 June 2012.

[SFBRWQCB] California San Francisco Bay Regional Water Quality Control Board 2011. Total Maximum Daily Load Selenium in North San Francisco Bay, Preliminary Project Report. Oakland (CA): California San Francisco Bay Regional Water Quality Control Board. 103 p. Available from: *http://www.swrcb.ca.gov/rwqcb2/water_issues/programs/TMDLs/seleniumtmdl.shtml* Accessed 11 June 2012.

[SWRCB] California State Water Resources Control Board. 1987. Selenium verification study, 1986. Sacramento (CA): California State Water Resources Control Board. 79 p + 9 appendices.

[SWRCB] California State Water Resources Control Board. 1988. Selenium verification study, 1986–1987. Sacramento (CA): California State Water Resources Control Board. 60 p + 8 appendices.

[SWRCB] California State Water Resources Control Board. 1989. Selenium verification study, 1987–88. Sacramento (CA): California State Water Resources Control Board. 81 p + 11 appendices

[SWRCB] California State Water Resources Control Board. 1991. Selenium verification study, 1988–1990. Sacramento (CA): California State Water Resources Control Board. 79 p + 9 appendices.

[SWRCB] California State Water Resources Control Board. 2011. California Clean Water Act Section 303(d) list of Water Quality Limited Segments for 2010. Sacramento (CA): California State Water Resources Control Board. 53 p.

[SWRCB] California State Water Resources Control Board. 2012. Data retrieval, California Environmental Data Exchange Network (CEDEN). Sacramento (CA): California State Water Resources Control Board. Available from: *http://ceden.org/ceden_data.shtml* Accessed 18 March 2013.

Chapman PM, Adams WJ, Brooks ML, Delos CG, Luoma SN, Maher WA, Ohlendorf HM, Presser TS, Shaw DP. 2010. Ecological assessment of selenium in the aquatic environment. Pensacola (FL): SETAC Press. 358 p.

Conley JM, Funk DH, Buchwalter DB. 2009. Selenium bioaccumulation and maternal transfer in the mayfly Centroptilum triangulifer in a life-cycle, periphyton-biofilm trophic assay. Environmental Science and Technology 43(20):7952–7957.

Conley JM, Funk DH, Cariello NJ, Buchwalter DB. 2011. Food rationing affects dietary selenium bioaccumulation and life cycle performance in the mayfly Centroptilum triangulifer. Ecotoxicology 20 (8):1840–1851.

Conomos TJ, Smith RE, Peterson DH, Hager SW, Schemel LE. 1979. Processes affecting seasonal distributions of water properties in the San Francisco Bay estuarine system. In: Conomos TJ, editor. San Francisco Bay: the urbanized estuary. San Francisco (CA): AAAS Pacific Division. p. 115–142. Available from: *http://downloads.ice.ucdavis.edu/sfestuary/conomos_1979/archive1022.PDF* Accessed 12 June 2012.

Cutter GA.1989. The estuarine behavior of selenium in San Francisco Bay. Estuarine, Coastal, and Shelf Science 28(1):13–34.

Cutter GA, Cutter LS. 2004. Selenium biogeochemistry in the San Francisco Bay estuary: changes in water column behavior. Estuarine, Coastal and Shelf Science 61(3):463–476.

Cutter GA. Bruland KW. 1984. The marine biogeochemistry of selenium a re-evaluation. Limnology and Oceanography 29(6):1179–1192.

Cutter GA, San Diego–McGlone MLC. 1990. Temporal variability of selenium fluxes in San Francisco Bay. Science of the Total Environment 97/98:235–250.

De La Cruz SE. 2010. Habitat, diet, and contaminant relationships of surf scoters wintering in San Francisco Bay: implications for conservation in urban estuaries. [Ph.D. dissertation]. Davis (CA): University of California Davis. 215 p.

De La Cruz SEW, Takekawa JY, Miles A, Eadie JM, Palm EC, Wilson MT. 2008. The influence of winter habitat use and diet on selenium risks for surf scoters (Melanitta perspicillata) in the San Francisco Bay Estuary [absract]: State of the San Francisco Estuary Conference. **[dates?]** Oakland (CA): The San Francisco Estuary Project. 1 p.

De La Cruz SEW, Takekawa JY, Wilson MT, Nysewander DR, Evenson JR, Esler D, Boyd WS, Ward DH. 2009. Spring migration routes and chronology of surf scoters (*Melanitta perspicillata*): a synthesis of Pacific coast studies. Canadian Journal of Zoology 87(11):1069–1086.

DiGennaro B, Reed D, Swanson C, Hastings L, Hymanson Z, Healey M, Siegel S, Cantrell S, Herbold B. 2012. Using conceptual models and decision-support tools to guide ecosystem restoration planning and adaptive management: an example from the Sacramento–San Joaquin Delta, California. San Francisco Estuary and Watershed Science [Internet]. Available from: *http://www.escholarship.org/uc/item/3j95x7vt* Accessed 18 March 2013.

Doblin MA, Baines SB, Cutter LS, Cutter GA. 2006. Sources and biogeochemical cycling of particulate selenium in the San Francisco Bay estuary. Estuarine, Coastal and Shelf Science 67(4):681–694.

Environment Canada. 2005. Guidance document on statistical methods for environmental toxicity tests (EPS 1/RM/46)/ Method Development and Application Section. Ottawa (ON): Environmental Technology Centre, Environment Canada. 170 p. and appendices A–R, with June 2007 amendments.

Foe C. 2010. Selenium concentrations in largemouth bass in the Sacramento–San Joaquin Delta. Sacramento (CA): Central Valley Regional Water Quality Control Board. 24 p.

Fournier E, Adam C, Massabuau J–C, Garnier–Laplace J. 2006. Selenium bioaccumulation in *Chlamydomonas reinhardtii* and subsequent transfer to *Corbicula fluminea*: role of selenium speciation and bivalve ventilation. Environmental Toxicology and Chemistry 25(10):2692–2699.

Fowler SW, Benayoun G. 1976. Influence of environmental factors on selenium flux in two marine invertebrates. Marine Biology 37(1):59–68.

Ganju NK, Schoellhamer DH, Warner JC, Barad MF, Schladow GF. 2004. Tidal oscillation of sediment between a river and a bay: a conceptual model. Estuarine Coastal and Shelf Science 60: 81–90.

Graham RV, Blaylock BG, Hoffman FO, Frank ML. 1992. Comparison of selenomethionine and selenite cycling in freshwater experimental ponds. Water, Air, and Soil Pollution 62(1–2):25–42.

Heinz GH. 1996. Selenium in birds. In: Beyer WN, Heinz GH, Redmon–Norwood AW, editors. Environmental contaminants in wildlife: interpreting tissue concentrations. New York, (NY): Lewis Publishers. p. 447–458.

Heinz GH, Hoffman DJ, Gold LG. 1989. Impaired reproduction of mallards fed an organic form of selenium. The Journal of Wildlife Management 53(2):418–428.

Heinz GH, Pendleton GW, Krynitsky AJ, Gold LG. 1990. Selenium accumulation and elimination in mallards. Archives of Environmental Contamination and Toxicology 19(3):374–379.

Hoffman DJ. 2002. Role of selenium and oxidative stress in aquatic birds. Aquatic Toxicology 57(1–2):11–26.

Hoffman DJ, Ohlendorf HM, Marn CA, Pendleton GW. 1998. Association of mercury and selenium with altered glutathione metabolism and oxidative stress in diving ducks from the San Francisco Bay Region, USA. Environmental Toxicology and Chemistry 17(2):167–172.

Hoffman, DJ, Marn C, Marois K, Sproul E, Dunne M, Skorupa JP. 2002. Sublethal effects in avocet and stilt hatchlings from selenium-contaminated sites. Environmental Toxicology and Chemistry 21(3):561–566.

Institute of Medicine. 2000. Dietary reference intakes for vitamin C, vitamin E, selenium and carotenoids. Washington (DC): National Academy Press. 506 p.

Janz, DM. 2012. Selenium. In: Wood CW, Farrell AP, Brauner CJ, editors. Fish Physiology Volume 31A, Homeostasis and toxicology of essential metals. San Diego (CA): Elsevier. p. 327–374.

Kiffney P, Knight A. 1990. The toxicity and bioaccumulation of selenate, selenite, and seleno–L–methionine in the cyanobacterium *Anabaena flos-aquae*. Archives of Environmental Contamination and Toxicology 19(4):488–494.

Kleckner AE, Stewart AR, Elrick K, Luoma SN. 2010. Selenium concentrations and stable isotopic compositions of carbon and nitrogen in the benthic clam *Corbula amurensis* from Northern San Francisco Bay, California, May 1995 to February 2010. U.S. Geological Survey Open-File Report 2010–1252. 34 p. Available from: *http://pubs.usgs.gov/of/2010/1252/* Accessed 11 June 2012.

Kroll KJ, Doroshov SI. 1991. Vitellogenin: potential vehicle for selenium bioaccumulation in oocytes of the white sturgeon (*Acipenser transmontanus*). In: Williot P, editor. *Acipenser*: Proceedings of the 1st International Sturgeon Conference, October 3–6, 1989, Bordeaux, France. *Centre National du Machinisme Agricole du Genie Rural des Eaux et des Forets*, Bordeaux, France. p. 99–106

Lee B–G, Lee J–S, Luoma SN. 2006. Comparison of selenium bioaccumulation in the clams *Corbicula fluminea* and *Potamocorbula amurensis*: a bioenergetic modeling approach. Environmental Toxicology and Chemistry 25(7):1933–40.

Lemly AD. 2002. Selenium assessment in aquatic ecosystems: a guide for hazard evaluation and water quality criteria. New York (NY): Springer–Verlag. 180 p.

Linares J, Linville RG, Van Eenennaam J, Doroshov S. 2004. Selenium effects on health and reproduction of white sturgeon in the Sacramento–San Joaquin Estuary. Final Report, Project No. ERP–02–P35. Davis (CA): University of California, Department of Animal Science. 56 p.

Linville RG. 2006. Effects of excess selenium on the health and reproduction of white sturgeon (*Acipenser transmontanus*): implications for San Francisco Bay–Delta. [Ph.D. dissertation]. Davis (CA): University of California Davis. 232 p.

Linville RG, Luoma SN, Cutter L, Cutter GA. 2002. Increased selenium threat as a result of invasion of the exotic bivalve *Potamocorbula amurensis* into the San Francisco Bay–Delta. Aquatic Toxicology 57(1–2):51–64.

Lucas L, Stewart AR. 2007. Transport, transformation, and effects of selenium and carbon in the Delta of the Sacramento–San Joaquin Rivers: implications for ecosystem restoration: Final report to CALFED. Ecosystem Restoration Program Agreement No. 4600001955. Project No. ERP–01–C07. 515 p.

Luoma SN, Fisher N. 1997. Uncertainties in assessing contaminant exposure from sediments. In: Ingersoll CG, Dillon T, Biddinger GR, editors. Ecological risk assessment of contaminated sediments. Pensacola (FL): SETAC Press. p. 211–237.

Luoma SN, Presser TS. 2009. Emerging opportunities in management of selenium contamination. Environmental Science and Technology 43(22):8483–8487.

Luoma SN, Rainbow PS. 2005. Why is metal bioaccumulation so variable? biodynamics as a unifying concept. Environmental Science and Technology 39(7):1921–1931.

Luoma SN, Rainbow PS. 2008. Metal contamination in aquatic environments: science and lateral management. New York (NY): Cambridge University Press. 573 p.

Luoma SN, Johns C, Fisher NS, Steinberg NA, Oremland RS, Reinfelder JR. 1992. Determination of selenium bioavailability to a benthic bivalve from particulate and solute pathways. Environmental Science and Technology 26(3):485–491.

McKee L, Oram J, Yee D, Davis J, Sedlack M. 2008. Sources, pathways, and loadings workgroup five-year workplan (SFEI Contribution #657). Richmond (CA): San Francisco Estuary Institute. 35 p.

Meseck SL, Cutter GA. 2006. Evaluating the biogeochemical cycle of selenium in San Francisco Bay through modeling. Limnology and Oceanography 51(5):2018–2032.

[NRC] National Research Council. 2010. A scientific assessment of alternatives for reducing water management effects on threatened fishes in California's Bay-Delta. Washington D.C.: National Academy Press. 93 p.

[NRC] National Research Council. 2011. A review of the use of science and adaptive management in California's draft Bay-Delta conservation plan. Washington D.C.: National Academy Press. 69 p.

[NRC] National Research Council. 2012. Sustainable water and environmental management in the California Bay–Delta. Washington D.C.: National Academy Press. 220 p.

Ohlendorf HM. 2003. Ecotoxicology of selenium. In: Hoffman DJ, Rattner BA, Burton Jr GA, Cairns Jr J, editors. Handbook of Ecotoxicology, 2nd ed. Boca Raton (FL): Lewis Publishers. p 465–500.

Ohlendorf HM, Heinz GH. 2011. Selenium in birds. In: BeyerWN, MeadorJP, editors. Environmental contaminants in biota: interpreting tissue concentrations, 2nd ed. Boca Raton (FL): CRC Press. p 669–701.

Oremland RS, Hollibaugh JT, Maest AS, Presser TS, Miller LG, Culbertson CW. 1989. Selenate reduction to elemental selenium by anaerobic bacteria in sediments and culture: biogeochemical significance of a novel, sulfate-independent respiration. Applied and Environmental Microbiology 55(9):2333–2343.

Peterson DH, Smith RE, Hager SW, Harmon DD, Herndon RE, Schemel LE. 1985. Interannual variability in dissolved inorganic nutrients in northern San Francisco Bay estuary. Hydrobiologia 129(1):37–58.

Peterson HA,Vayssieres M. 2010. Benthic assemblage variability in the upper San Francisco Estuary: a 27-year retrospective. San Francisco Estuary and Watershed Science [Internet]. Available from: *http://www.escholarship.org/uc/item/4d0616c6* 18 March 2013.

Presser TS. 1994. The Kesterson effect. Environmental Management 18(3):437–454.

Presser TS, Luoma SN. 2006. Forecasting selenium discharges to the San Francisco Bay–Delta Estuary: ecological effects of a proposed San Luis Drain extension. U.S. Geological Survey Professional Paper 1646.196 p. Available from: *http://pubs.usgs.gov/pp/p1646/* Accessed 12 June 2012.

Presser TS, Luoma SN. 2009. Modeling of selenium for the San Diego Creek watershed and Newport Bay, California. U.S. Geological Survey Open-File Report 2009–1114. 48 p. Available from: *http://pubs.usgs.gov/of/2009/1114* Accessed 12 June 2012.

Presser TS, Luoma SN. 2010a. A methodology for ecosystem-scale modeling of selenium. Integrated Environmental Assessment and Management 6(4):685–710.

Presser TS, Luoma SN. 2010b. Ecosystem-scale modeling in support of fish and wildlife selenium criteria development for the San Francisco Bay–Delta Estuary, California, U.S. Geological Survey Administrative Report prepared for (2010) and released by USEPA (2011), Region 9, San Francisco, California. 101 p + appendices A–D. Available from: *http://www.epa.gov/region9/water/ctr* Accessed 12 June 2012.

Presser TS, Ohlendorf HM. 1987. Biogeochemical cycling of selenium in the San Joaquin Valley, California, USA. Environmental Management 11(6):805–821.

Presser TS, Schwarzbach SE. 2008. Technical Analysis of In-Valley Drainage Management Strategies for the Western San Joaquin Valley, California: U.S. Geological Survey Open-File Report 2008–1210. 37 p. Available from: *http://pubs.usgs.gov/of/2008/1210/* Accessed 12 June 2012.

Presser TS, Piper DZ, Bird KJ, Skorupa JP, Hamilton SJ, Detwiler SJ, Huebner MA. 2004. The Phosphoria Formation: a model for forecasting global selenium sources to the environment. In: Hein J, editor. Life cycle of the Phosphoria Formation: from deposition to the post-mining environment. New York (NY): Elsevier p. 299–319. Available from: *http://wwwrcamnl.wr.usgs.gov/Selenium/Library_articles/chap11_global_Se-optimized.pdf* Accessed 12 June 2012.

Reiley MC, Stubblefield WA, Adams WJ, Di Toro DM, Hodson PV, Erickson RJ, Keating Jr FJ, editors. 2003. Reevaluation of the state of the science for water-quality criteria development. Pensacola (FL): SETAC Press. 224 p.

Reinfelder JR, Fisher NS. 1991. The assimilation of elements ingested by marine copepods. Science 251(4995):794–796.

Reinfelder JR, Fisher NS, Luoma SN, Nichols JW, Wang W–X. 1998. Trace element trophic transfer in aquatic organisms: a critique of the kinetic model approach. Science of the total environment 219(2–3):117–135.

Riedel GF, Sanders JG, Gilmour CC. 1996. Uptake, transformation, and impact of selenium in freshwater phytoplankton and bacterioplankton communities. Aquatic Microbial Ecology 11(1):43–51.

Roditi HA, Fisher NS. 1999. Rates and routes of trace element uptake in zebra mussels. Limnology and Oceanography 44(7):1730–1749.

Saiki MK, Jennings MR, Hamilton SJ. 1991 Preliminary assessment of the effects of selenium in agricultural drainage on fish in the San Joaquin Valley. In: Dinar A, Zilberman, D, editors. The economics and management of water and drainage in agriculture. Boston (MA): Kluwer Academic Publishers. p 369–385.

Sandholm M, Oksanen HE, Pesonen L. 1973. Uptake of selenium by aquatic organisms. Limnology and Oceanography 18(3):496–499.

[SFEI] San Francisco Estuary Institute. 2000. San Francisco Bay seafood consumption study. Richmond (CA): San Francisco Estuary Institute. 84 p + 1 appendix. Available from: *http://www.sfei.org/node/2022* Accessed 12 June 2012.

[SFEI] San Francisco Estuary Institute. 2005. Data retrieval, tissue, surf scoter, scaup, muscle, selenium: San Francisco Estuary Institute Regional Monitoring Program, Region-wide Science for Ecosystem Management. Oakland (CA): San Francisco Estuary Institute. Available from: *http://sfei.org/tools/wqt* Accessed 12 June 2012.

[SFEI] San Francisco Estuary Institute. 2009. Data retrieval, tissue, white sturgeon, muscle, selenium: San Francisco Estuary Institute Regional Monitoring Program, Region-wide Science for Ecosystem Management. Oakland (CA): San Francisco Estuary Institute. Available from: *http://sfei.org/tools/wqt* Accessed 12 June 2012.

Schlekat CE, Dowdle PR, Lee BG, Luoma SN, Oremland RS. 2000. Bioavailability of particle-associated Se to the bivalve *Potamocorbula amurensis*. Environmental Science and Technology 34(21):4504–4510.

Schlekat CE, Purkerson DG, Luoma SN. 2004. Modeling selenium bioaccumulation through arthropod food webs in San Francisco Bay, California, USA. Environmental Toxicology and Chemistry 23(12):3003–3010.

Schwarzbach SE, Albertson JD, Thomas CM. 2006. Impacts of predation, flooding, and contamination on the reproductive success of the California clapper rail (*Rallus longirostris obsoletus*) in San Francisco Bay. The Auk 123(1):45–60.

Skorupa JP. 1998. Selenium poisoning of fish and wildlife in nature: lessons from twelve real-world examples. In: Frankenberger Jr WT, Engberg RA, editors. Environmental chemistry of Selenium. New York (NY): Marcel Dekker Inc. p 315–354.

Skorupa JP. 2008 Great Salt Lake selenium standard: written recommendations: U.S. Fish and Wildlife Service, Division of Environmental Quality, Branch of Environmental Contaminants, Arlington, Virginia, 9 p. Available from: *http://wwwrcamnl.wr.usgs.gov/Selenium/Library_articles/Great_Salt_Lake_Selenium_Standard.pdf*

Skorupa JP, Presser TS, Hamilton SJ, Lemly AD, Sample BE. 2004. EPA's draft tissue-based criterion: a technical review. U.S. Fish and Wildlife Report presented to U.S. Environmental Protection Agency, dated June 16, 2004. 35 p. *http://wwwrcamnl.wr.usgs.gov/Selenium/Library_articles/skorupa_et_al_2004.pdf* Accessed 18 March 2013

Stadtman TC. 1974. Selenium biochemistry. Science 183(4128):915–922.

Stewart AR, Luoma SN, Schlekat CE, Doblin MA, Hieb KA. 2004. Food web pathway determines how selenium affects ecosystems: a San Francisco Bay case study. Environmental Science and Technology 38(17):4519–4526.

[Tetra Tech, Inc.] Tetra Tech Incorporated. 2010. Technical Memorandum 6: application of ECoS3 model for simulation of selenium fate and transport in North San Francisco Bay. Lafayette (CA): Tetra Tech Incorporated. 103 p. Available from: *http://www.swrcb.ca.gov/rwqcb2/water_issues/programs/TMDLs/seleniumtmdl.shtml* Accessed 12 June 2012.

[USBR] U.S. Bureau of Reclamation. 1978. Special task force report on the San Luis Unit as required by Public Law 94–46, Central Valley Project, California. Sacramento (CA): U.S. Bureau of Reclamation Mid-Pacific Region. 331 p + 11 maps.

[USBR] U.S. Bureau of Reclamation. 1995. Finding of no significant impact and supplement environmental assessment, Grassland Bypass Channel Project, Interim use of a portion of the San Luis Drain for conveyance of drainage water through the Grassland Water District and adjacent Grassland areas. Sacramento (CA): U.S. Bureau of Reclamation Mid-Pacific Region. 36 p + appendices.

[USBR] U.S. Bureau of Reclamation. 2001. Grassland Bypass Project environmental impact statement and environmental impact report. Sacramento (CA): U.S. Bureau of Reclamation Mid-Pacific Region. Volume I (main text; appendices A and I) and volume II (appendices B–H).

[USBR] U.S. Bureau of Reclamation. 2006. San Luis Drainage Feature Re-evaluation Environmental Impact Statement. Sacramento (CA): U.S. Bureau of Reclamation Mid-Pacific Region. Executive summary, sections 1–24, appendices A–P. 2,785 p.

[USBR] U.S. Bureau of Reclamation. 2008. San Luis Drainage Feature Re-evaluation Feasibility Report. Sacramento (CA): U.S. Bureau of Reclamation Mid-Pacific Region. 193 p + appendices A–M. 2,247 p.

[USBR] U.S. Bureau of Reclamation. 2009. Grassland Bypass Project (2010–2019) environmental impact statement and environmental impact report. Sacramento (CA): U.S. Bureau of Reclamation Mid-Pacific Region. 391 p + appendices A–H. Available from: *http://www.usbr.gov/mp/nepa/documentShow.cfm?Doc_ID=4412* Accessed 12 June 2012.

[USBR] U.S. Bureau of Reclamation. 2011. Delta–Mendota Canal water quality monitoring report, quarterly report of flows, concentrations and loads of salts and selenium (on-going series). Sacramento (CA): U.S. Bureau of Reclamation Mid-Pacific Region. 39 p.

[USBR] U.S. Bureau of Reclamation. 2012. Central Valley Operations Office Water Accounting Reports. Sacramento (CA): U.S. Bureau of Reclamation Mid-Pacific Region. Available from: *http://www.usbr.gov/mp/cvo/pmdoc.html* Accessed 12 June 2012

[USDOI] U.S. Department of the Interior. 1998. Guidelines for interpretation of effects of selected constituents in biota, water, and sediment – selenium. 47 p. Available from: *http://www.usbr.gov/niwqp/guidelines/pdf/Selenium.pdf* Accessed 12 June 2012

[USDHHS] U.S. Department of Health and Human Services. 2002. Dietary intake of macronutrients, micronutrients and other dietary constituents. United States National Center for Health Statistics, 1988–1994. Vital Health Statistics 11(245).

[USEPA] U.S. Environmental Protection Agency. 1992. Rulemaking, water quality standards, establishment of numeric criteria for priority toxic pollutants, states' compliance (National Toxics Rule), Final Rule: Federal Register 57 (242) p. 60848.

[USEPA] U.S. Environmental Protection Agency. 2000a. Water Quality Standards: Establishment of Numeric Criteria for Priority Toxic Pollutants for the State of California (California Toxics Rule): Federal Register 65(97), p. 31681–31719. Available from: *http://regulations.vlex.com/vid/environmental-protection-agency-23304844* Accessed 12 July 2012

[USEPA] U.S. Environmental Protection Agency. 2000b. Guidance for assessing chemical contaminant data for use in fish advisories. Volume2: risk assessment and fish consumption limits. Third edition, EPA 823-B-00-008. November 2000.

[USEPA] U.S. Environmental Protection Agency. 2004. Notice of draft aquatic life criteria for selenium and request for scientific information, data, and views: Federal Register 69 (242):75541–75546. Available from: *http://www2.epa.gov/sites/production/files/documents/actionplan.pdf* Accessed 12 July 2012.

[USEPA] U.S. Environmental Protection Agency. 2011a. Update on California Toxics Rule and Endangered Species Act (September, 2011): Selenium. San Francisco, (CA): U.S. Environmental Protection Agency, Region 9. Available from: *http://www.epa.gov/region9/water/ctr/* Accessed 12 June 2012.

[USEPA] U.S. Environmental Protection Agency. 2011b. Water Quality Challenges in the San Francisco Bay/Sacramento–San Joaquin Delta Estuary. Advanced Notice of Proposed Rulemaking: Federal Register 76(35):9709–9714. Available from: *http://www2.epa.gov/sfbay-delta/bay-delta-action-plan* Accessed 18 March 2013.

[USFWS] U.S. Fish and Wildlife Service. 2006. Giant garter snake (*Thamnophis gigas*) 5-year review: summary and evaluation. Sacramento (CA): U.S. Fish and Wildlife Service. 46 p.

[USFWS] U.S. Fish and Wildlife Service. 2008a. Species at risk from selenium exposure in the San Francisco estuary. Sacramento (CA): U.S. Fish and Wildlife Service, Sacramento Fish and Wildlife Office, Environmental Contaminants Division March 2008. 81 p.

[USFWS] U.S. Fish and Wildlife Service. 2008b. Potential effects of selenium contamination on federally-listed species resulting from delivery of federal water to the San Luis Unit. Sacramento (CA): U.S. Fish and Wildlife Service, Sacramento Fish and Wildlife Office, Environmental Contaminants Division. March 2008. 46 p.

[USFWS] U.S. Fish and Wildlife Service. 2009. Endangered species consultation on the proposed continuation of the Grassland Bypass Project, 2010–2019. Sacramento (CA): U.S. Fish and Wildlife Service, Sacramento Fish and Wildlife Office. December 2009. 198 p.

[USFWS and NOAA Fisheries] U.S. Fish and Wildlife Service and National Oceanic and Atmospheric Administration National Marine Fisheries Service. 2000. Final biological opinion on the effects of the U.S. Environmental Protection Agency's "Final Rule for the Promulgation of Water Quality Standards: Establishment of Numeric Criteria for Priority Toxic Pollutants for the State of California. U.S. Fish and Wildlife Service and National Marine Fisheries Service. 323 p.

Velinsky DJ, Cutter, GA. 1991. Geochemistry of selenium in a coastal salt marsh. Geochimica et Cosmochimica Acta 55(1):179–191.

Wang W-X. 2002. Interactions of trace metals and different marine food chains. Marine Ecology Progress Series 243(1):295–309.

Wang W-X, Dei RCH. 1999. Kinetic measurements of metal accumulation in two marine macroalgae. Marine Biology 135(1):11–23.

Wang W-X, Fisher NS. 1999. Delineating metal accumulation pathways for marine invertebrates. Science of the Total Environment 237/238:459–472.

Wang W-X, Fisher NS, Luoma SN. 1996. Kinetic determinations of trace element bioaccumulation in the mussel Mytilus edulis. Marine Ecology Progress Series 140(1-3):91–113.

Zhang YQ, Moore JN. 1996. Selenium fractionation and speciation in a wetland system. Environmental Science and Technology 30(8):2613–2619.

SAN FRANCISCO ESTUARY & WATERSHED SCIENCE
Sponsored by the Delta Science Program and the UC Davis John Muir Institute of the Environment

Title:
Agricultural Losses from Salinity in California's Sacramento-San Joaquin Delta

Journal Issue:
San Francisco Estuary and Watershed Science, 12(1)

Author:
Medellín-Azuara, Josué, University of California, Davis
Howitt, Richard E., University of California, Davis
Hanak, Ellen, Public Policy Institute of California
Lund, Jay R., University of California, Davis
Fleenor, William E., University of California, Davis

Publication Date:
2014

Publication Info:
San Francisco Estuary and Watershed Science

Permalink:
http://escholarship.org/uc/item/4b7295m9

Acknowledgements:
This study draws on earlier work supported by the UC Davis Center for Watershed Sciences and the Public Policy Institute of California. Funding from the S.D. Bechtel, Jr. Foundation supported additional research undertaken for this article. Research assistance from Molly Ferrell, Michelle Lent, and Duncan MacEwan in preparing and/or providing datasets for this work is appreciated. We are also thankful for the comments from peer reviewers on earlier versions of this work, and on the manuscript submitted to this journal, which helped improve it substantially.

Author Bio:
Civil and Environmental Engineering, Research Scientist, Center for Watershed Sciences

Agricultural and Resources Economics, Professor

Senior Policy Fellow

Civil and Environmental Engineering, Professor, Center for Watershed Sciences, Director

Civil and Environmental Engineering, Research Engineer, Center for Watershed Sciences

Keywords:
Sacramento-San Joaquin Delta, Salinity, Positive Mathematical Programming, Calibration, California, Hydro-Economic Models, Agricultural Production, Drought Analysis, Economic Impacts, Economics, Engineering, Agronomy

---

Abstract:
Sea level rise, large-scale flooding, and new conveyance arrangements for water exports may increase future water salinity for local agricultural production in California's Sacramento–San Joaquin Delta. Increasing salinity in crop root zones often decreases crop yields and crop revenues. Salinity effects are nonlinear, and vary with crop choice and other factors including drainage and residence time of irrigation water. Here, we explore changes in agricultural production in the Delta under various combinations of water management, large-scale flooding, and future sea level rise. Water management alternatives include through-Delta water exports (current conditions), dual conveyance (through-Delta and a 6,700 $Mm^3 yr^{-1}$ [or 7500 cfs] capacity peripheral canal or tunnel) and the flooding of five western islands with and without peripheral exports. We employ results from previous hydrodynamic simulations of likely changes in salinity for irrigation water at points in the Delta. We connect these irrigation water salinity values into a detailed agro-economic model of Delta agriculture to estimate local crop yield and farm revenue losses. Previous hydrodynamic modeling work shows that sea level rise is likely to increase salinity from 4% to 130% in this century, depending on the increase in sea level and location. Changes in water management under dual conveyance increase salinity mostly in the western Delta, and to a lesser extent in the north, where current salinity levels are now quite low. Because locations likely to experience the largest salinity increases already have a lower-value crop mix, the worst-case losses are less than 1% of total Delta crop revenues. This result also holds for salinity increases from permanent flooding of western islands that serve as a salinity barrier. Our results suggest that salinity increases could have much smaller economic effects on Delta farming than other likely changes in the Delta such as retirement of agricultural lands after large-scale flooding and habitat development. Integrating hydrodynamic, water salinity, and economic models can provide insights into controversial management issues.

MARCH 2014

SAN FRANCISCO ESTUARY & WATERSHED SCIENCE

Sponsored by the Delta Science Program and the UC Davis John Muir Institute of the Environment

Agricultural Losses from Salinity in California's Sacramento–San Joaquin Delta

Josué Medellín–Azuara*[1], Richard E. Howitt[2], Ellen Hanak[3], Jay R. Lund[1], and William Fleenor[1]

ABSTRACT

Sea level rise, large-scale flooding, and new conveyance arrangements for water exports may increase future water salinity for local agricultural production in California's Sacramento–San Joaquin Delta. Increasing salinity in crop root zones often decreases crop yields and crop revenues. Salinity effects are nonlinear, and vary with crop choice and other factors including drainage and residence time of irrigation water. Here, we explore changes in agricultural production in the Delta under various combinations of water management, large-scale flooding, and future sea level rise. Water management alternatives include through-Delta water exports (current conditions), dual conveyance (through-Delta and a 6,700 $Mm^3\ yr^{-1}$ [or 7500 cfs] capacity peripheral canal or tunnel) and the flooding of five western islands with and without peripheral exports. We employ results from previous hydrodynamic simulations of likely changes in salinity for irrigation water at points in the Delta. We connect these irrigation water salinity values into a detailed agro-economic model of Delta agriculture to estimate local crop yield and farm revenue losses. Previous hydrodynamic modeling work shows that sea level rise is likely to increase salinity from 4% to 130% in this century, depending on the increase in sea level and location. Changes in water management under dual conveyance increase salinity mostly in the western Delta, and to a lesser extent in the north, where current salinity levels are now quite low. Because locations likely to experience the largest salinity increases already have a lower-value crop mix, the worst-case losses are less than 1% of total Delta crop revenues. This result also holds for salinity increases from permanent flooding of western islands that serve as a salinity barrier. Our results suggest that salinity increases could have much smaller economic effects on Delta farming than other likely changes in the Delta such as retirement of agricultural lands after large-scale flooding and habitat development. Integrating hydrodynamic, water salinity, and economic models can provide insights into controversial management issues.

KEY WORDS

Sacramento–San Joaquin Delta; salinity; positive mathematical programming; calibration; California; hydro-economic models; agricultural production; drought analysis; economic effects; climate change, sea level rise, large area flooding

* Corresponding author: *jmedellin@ucdavis.edu*
1 University of California, Davis, Department of Civil and Environmental Engineering, Davis, CA USA
2 University of California, Davis, Agricultural and Resource Economics, Davis, CA USA
3 Public Policy Institute of California, San Francisco, CA USA

INTRODUCTION

Salinity-driven reductions in agricultural production have long been a policy concern for California's Sacramento–San Joaquin Delta (CDPW 1931; Lund et al. 2007; DPC 2012; Medellín–Azuara et al. 2012a). In this study we quantify the economic effects on local agriculture of changes in localized Delta water salinity for a range of sea level and water management conditions during the irrigation season. We employ the Delta Agricultural Production model (DAP, after Lund et al. 2007; Howitt et al. 2012), an agro-economic model for crops in the Delta that accounts for crop yield response to changes in irrigation water salinity. This work demonstrates the combined application of hydrodynamic, water salinity, and agro-economic modeling to provide policy and management insights for a major water resources problem in California.

The economic effects of changes in irrigation water salinity vary in magnitude by crop, location, and the initial level of water salinity. By connecting hydrodynamic simulations with the crop production model, we find that small changes in salinity generally cause little change in Delta crop yields and revenues. Land use surveys indicate that higher-value and generally less salt-tolerant crops tend to be grown in areas of the Delta that currently have lower-irrigation water salinity; these areas do not experience major salinity changes in the simulated scenarios. These conditions allow lower-cost adaptation of cropping patterns, irrigated areas and the intensity of production factors per unit area within the Delta in response to the modeled salinity changes.

Salt accumulation has affected agriculture since ancient times in Mesopotamia and Egypt, and modeling of crop salinity response has been in the literature for some decades. Crop production response to salinity also has a history in the economics modeling literature at various temporal (short and long run) and spatial scales (from crop to farm and regional levels) (Feinerman 2000). Models usually involve optimization to maximize profits or minimize costs in farming under different salinity scenarios. Also, Cardon and Letey (1992) applied Darcy's law on "flow through a porous medium" to model plant water uptake under salinity conditions. Knapp and Wichelns (1990) review dynamic optimization methods, finding that initial conditions matter and that large enough drainage disposal costs make water recycling more attractive.

This paper uses results from Delta hydrodynamic and salinity transport modeling to provide irrigation water salinity levels for various locations in California's Sacramento–San Joaquin Delta under a variety of sea level and water management conditions; we use these values as inputs to an agro-economic model of crop production that includes the effects of soil salinity (Figure 1). Our modeling framework, presented in Figure 1, shows the flow of information among models. The hydrodynamic models (Water Analysis Module [WAM] and Resource Management Associates [RMA] 2-D model) provide water salinity data for different locations in the Delta. The DAP model takes crop production information from the Statewide Agricultural Production model (SWAP, Howitt et al. 2012), crop response to salinity models (Hoffman 2010), and land use information from the Department of Water Resources for each water salinity scenario to produce economically optimal cropping patterns for each Delta island. Sensitivity analyses for more recent Delta export periods and fixed salinity scenarios are also part of the modeling framework.

Several underlying assumptions are worth discussing. First, our approach assumes that soil salinity in the root zone is the same as that of irrigation water applied in the surface. Second, following Hoffman (2010), we assume sufficient drainage exists in irrigated areas to avoid salt accumulation in the root zone (a problem in the southern Central Valley studied by Medellín–Azuara et al. [2008] and Howitt et al. [2009]). Hoffman (2010) concluded that many factors influencing soil salinization in general, including leaching requirements, crop salt tolerance at growth stages, shallow groundwater table, effective rainfall, irrigation efficiency and uniformity, climate, soil

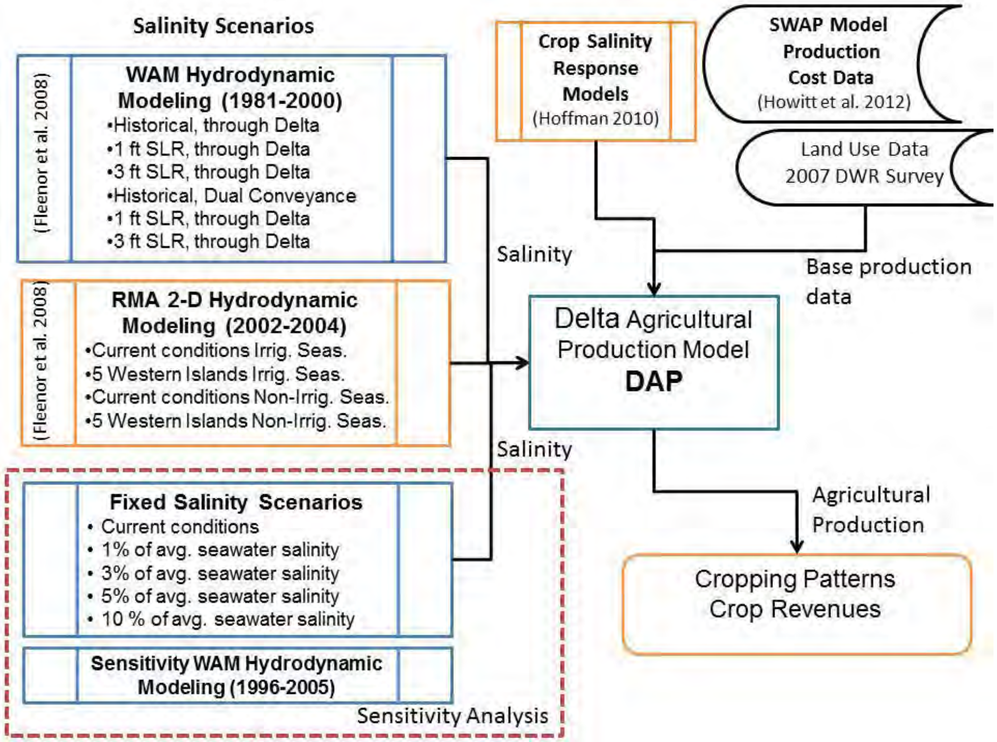


Figure 1 Modeling framework

bypass flow, salt precipitation and dissolution, are not major factors for salt accumulation in soils in the southern Delta. In Delta locations where drainage is a concern for crop productivity, subsurface drainage has been already installed.

Third, we use a sigmoidal approach (Van Genuchten 1983) for crop salinity response, as it is the best developed and well-suited for non-linear cropping optimization models like the one employed in this paper. In addition, the sigmoidal response-function approach showed good performance compared to the threshold-linear and exponential approaches (Van Genuchten and Hoffman 1984).

Mass and Hoffman (1977) pioneered comprehensive assessment of crop response to soil salinity. Mass (1990) provided a threshold approach in which different crop types (within a range of tolerant and sensitive) have relative yields constant up to thresholds in soil salinity. Beyond a threshold, relative yields decline at a constant rate. Another approach (Van Genuchten 1983; Hillel 2000) describes crop response to soil salinity in the root zone using a sigmoidal function that calibrates to a soil salinity at which crop yields are reduced by 50 percent.

Other factors that may affect crops include drainage and irrigation water salinity. Drainage salinity is closely related to soil salinization, because poor soil drainage conditions retain salts. A rising groundwater table with brackish or saline water can degrade soil at the root zone with prolonged exposures. Salinity in irrigation water decreases yields for many crops. However, brackish or slightly saline irrigation water may not affect yields for some crops if the appropriate drainage exists, in which case salts do not accumulate in the root zone.

Below, we present the DAP model structure and data sets, the water salinity scenarios and hydrodynamic modeling work, and model results for the water salinity scenarios (Figure 1). We conclude with a summary of the findings and some suggestions for further research.

DELTA AGRICULTURAL PRODUCTION MODEL

The Delta Agricultural Production Model (DAP) estimates the irrigated crop area and the crop mix that maximizes total net revenues on land areas within the Delta, taking into account production costs, crop prices, water use, and yield changes from irrigation water salinity (Lund et al. 2007, Appendix D). DAP is a customized version of the SWAP agro-economic model of California agriculture, augmented to examine the effects of irrigation water salinity. SWAP uses positive mathematical programming (PMP, after Howitt 1995) to calibrate a base case to observed values of input use, namely land, water, labor and supplies. SWAP has been used for numerous agricultural modeling applications in California including water markets, soil salinity in the Central Valley (Howitt et al. 2009; Medellín–Azuara et al. 2008), climate change (Medellín–Azuara et al. 2012b), and regional economic impacts of water markets and drought in the Central Valley (Howitt et al. 2012).

Model Formulation

DAP expands the SWAP model to incorporate crop yield changes from varying irrigation salinity following Van Genuchten and Hoffman (1984) and Hoffman (2010). PMP calibration for SWAP is presented in detail elsewhere (Howitt et al. 2012). PMP is a multi-stage calibration method developed by Howitt (1995) to represent agricultural production, land and water. It calibrates to a base data set of production costs, production volume and factor use. PMP takes the opportunity cost of land and water and a linear profit maximization program and uses these values and the first-order condition identities to parameterize a non-linear cost function. In a last stage, these PMP cost functions and the resource constraints conform to the base case, which calibrates exactly to observed values of crop production inputs such as land, water, labor and supplies. PMP (Howitt 1995) is one of the most common approaches in optimization models of agricultural production. Here we describe the steps in the last stage of optimization, relevant for assessing irrigation water salinity effects on crop yields. The DAP objective function is to maximize net financial returns of land and management using the following equation:

$$Maximize\ Z = \sum_g \sum_i v_{gi} \left[Y_{red}\, \tau_{gi} \sum_j X_{gij}^{\rho} \right]^{1/\rho} - \sum_g \sum_i \sum_j \left(\alpha_{gij} X_{gij} + \gamma X_{gij}^2 \right) \tag{1}$$

In this formulation, v_{gi} is the price for crop i on Delta island g; Y_{red} is the relative yield under each water salinity scenario following Van Genuchten and Hoffman (1984), τ_{gi} is the CES scale parameter for a constant elasticity of substitution production function. X_{gij} is the quantity of production factor j (land, water, labor and supplies) allocated to crop i, and α_{gij} and γ_{gij} are the parameters of a PMP cost function (Howitt 1995).

Y_{red}, the relative yields, are given by:

$$Y_{red} = \frac{1}{1 + \left(C/C_{50}\right)^{\rho}} \tag{2}$$

Where C is the reference salinity and C_{50} is the salinity at which the crop yields are reduced by 50% of the base yield. The parameters C_{50} and *rho* were obtained empirically by Van Genuchten and Hoffman (1984). *Rho* was estimated using a maximum entropy approach (Shannon 1948) detailed below. Land and water are the limiting resources such that:

$$\sum_i X_{gij} \le b_{gj} \quad \forall g,\ j \in \{land, water\} \tag{3}$$

DAP Projections for Year 2030

To estimate a 2030 case for farming in the Delta, we scaled down 2050 projections for California agriculture in Medellín–Azuara et al. (2012b) through interpolation. These projections include yield increases from improved technology (Brunke et al. 2005) and changes in market conditions and crop demands that increase prices of most crops. For prices we assumed that:

1. The Delta does not have market power; commodity prices elsewhere in the state are not affected by production in the Delta, thus the Delta is a commodity price taker;
2. California will maintain market power for specialty crops (e.g., fruits, nuts, vegetables) but will be a price taker for global crops such as rice, corn, and wheat; and
3. Shifts in demands—and consequently prices—for the specialty crops, are linked to income and population growth projections, whereas global crops are influenced by future world demand trends.

Crop Yield Response to Salinity

Figure 2 shows the relative yield as a function of salt concentration in the root zone for different parameter values. Root zone salinity is assumed to be the same as the irrigation water salinity.

We estimated the value of the *rho* parameter in Equation 2 using a compilation of studies by Hoffman (2010) for crops in the southern Delta. We employed a maximum entropy algorithm that used the experimental relative yield observations in Figure 2 to obtain *rho*. With a maximum entropy approach it is possible to obtain a parameter probability distribution even with small data sets. From there, *rho* was used in Equation 2 to obtain the entropy-adjusted relative yield curve (Figure 2). As in Hoffman's study, we assume that irrigation efficiency in the Delta is 85%, with a 15% leaching fraction. We assume no long-term salinity accumulation in Delta soils, because Delta farmers can drain their soils to avoid long-term salinity build-up. This contrasts with closed basins such as the Tulare Basin, where imported salts accumulate because salts cannot be exported (Medellín–Azuara et al. 2008).

We used a maximum entropy estimation to obtain a probability distribution and the expected value of the *rho* parameter in the nonlinear response function shown in Lund et al. (2007). With respect to the entropy-estimated *rho* parameter, we grouped the Delta crops into three categories: "sensitive" to salinity in the root zone, "moderately sensitive," and all other crops. The sensitive group includes almonds and pistachios, some vegetables (truck crops), and subtropical fruits. The moderately sensitive group includes alfalfa, irrigated and non-irrigated pasture, tomatoes, other deciduous, cucurbits, and vine crops. For all other DAP crop groups (corn, grain, other field, rice, and sugar beet) we used parameter information for the relative yield equation above from Lund et al. (2007).

Model Data Sets

The DAP model requires a base data set that includes average land and water use, labor and supplies, crop prices, yields and production costs using these factors. The DAP model has been revised from

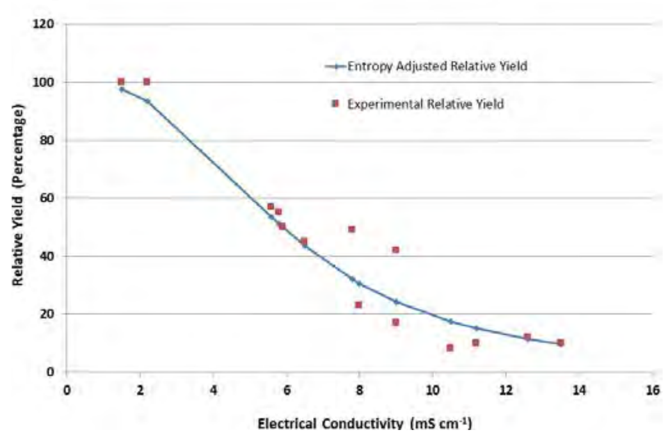


Figure 2 Empirical data on relative yield response to changes in electrical conductivity for dry beans in the South Sacramento–San Joaquin Delta. (Adapted from Medellín–Azuara et al. 2012a.)

previous versions (Lund et al. 2007), with more recent land use information, salinity response functions and production costs. DAP base model information on prices and other factors is for the average of 2005 through 2008, which we applied to 2007 land use as described below.

Land Use

This latest version of DAP employs preliminary land use estimates from the California Department of Water Resources (CDWR) 2007 field survey of the Delta (Medellín–Azuara et al. 2012a). We disaggregate production in the area defined as the "Legal Delta" into 70 different Delta islands and mainland areas, which are treated as individual farming units. The DAP model includes about 57% of the total Legal Delta area (nearly 299,000 ha). Thus total area in the modeled DAP area is 169,159 ha (418,000 ac), of which nearly 106,432 ha (263,000 ac) are farmed.

Figure 3 illustrates the study area, showing the salinity sampling stations, and breaking the Delta into sub-regions to describe the salinity results for the 70 modeled land units (detailed results by island are shown in Appendix Tables 1-4[1]). The Far West (dark red) is an area where salinity is already too high to support farming. The Delta's core agricultural areas are in the North (purple), South (orange), East (dark green), Central (light green), and West (light red). As discussed below, baseline salinity levels (and cropping patterns) vary considerably across these sub regions, as do the salinity effects of the scenarios examined here. The hatched blue area within the Western sub-region represents the five western islands flooded in the 2-D hydrodynamic modeling (hatched area in Figure 3). These five islands were chosen for an analysis of the effects of flooding in the state's Delta Risk Management Strategy study because of their role as a salinity barrier (Fleenor et al. 2008; Medellín–Azuara et al. 2012a).

1 *https://watershed.ucdavis.edu/project/water-quality-and-hydrodynamics*

Table 1 Annual revenues per acre (2008 $USD) in the base year and projected for 2030 [a]

Crop group	2005–08	2030	Increase (%)
Perennial fruits and nuts			
Almond and pistachio	5,054	5,533	9
Other deciduous	4,401	5,084	16
Subtropical	5,983	6,825	14
Vine	4,632	5,479	18
Vegetables and other truck farming			
Tomato [b]	1,940	2,668	38
Other truck	4,120	6,234	51
Field crops and pasture			
Alfalfa	1,004	1,207	20
Corn	853	1,242	46
Grain	464	470	1
Irrigated pasture	597	691	16
Non-irrigated grain and pasture	464	470	1
Other field crops	1,000	1,135	13
Rice	1,333	1,486	11
Sugar beet	1,891	2,043	8

a. Sources: author estimates using SWAP; Howitt et al. 2012; adapted from Medellín–Azuara et al. 2012a
b. Price is based on processing tomato, which constitutes 95% of the value.

Crop Prices, Yields, and Production Costs

Both DAP and SWAP use a 20-crop group classification established by CDWR (Howitt et al. 2012). Within the Delta, only 14 of these groups are present in large acreages (Table 1).

Cost information is from SWAP for Central Valley Production Model (CVPM) Region 9, which corresponds to the Sacramento–San Joaquin Delta. Information on CVPM regions is detailed in Howitt et al. (2012) and on the SWAP website (*http://swap.ucdavis.edu*).

SWAP crop production budgets are updated regularly using University of California–Davis' cost studies (*http://coststudies.ucdavis.edu/*). The SWAP crop prices and yields and resulting revenues per acre for the baseline period (2005–2008 average) are from a

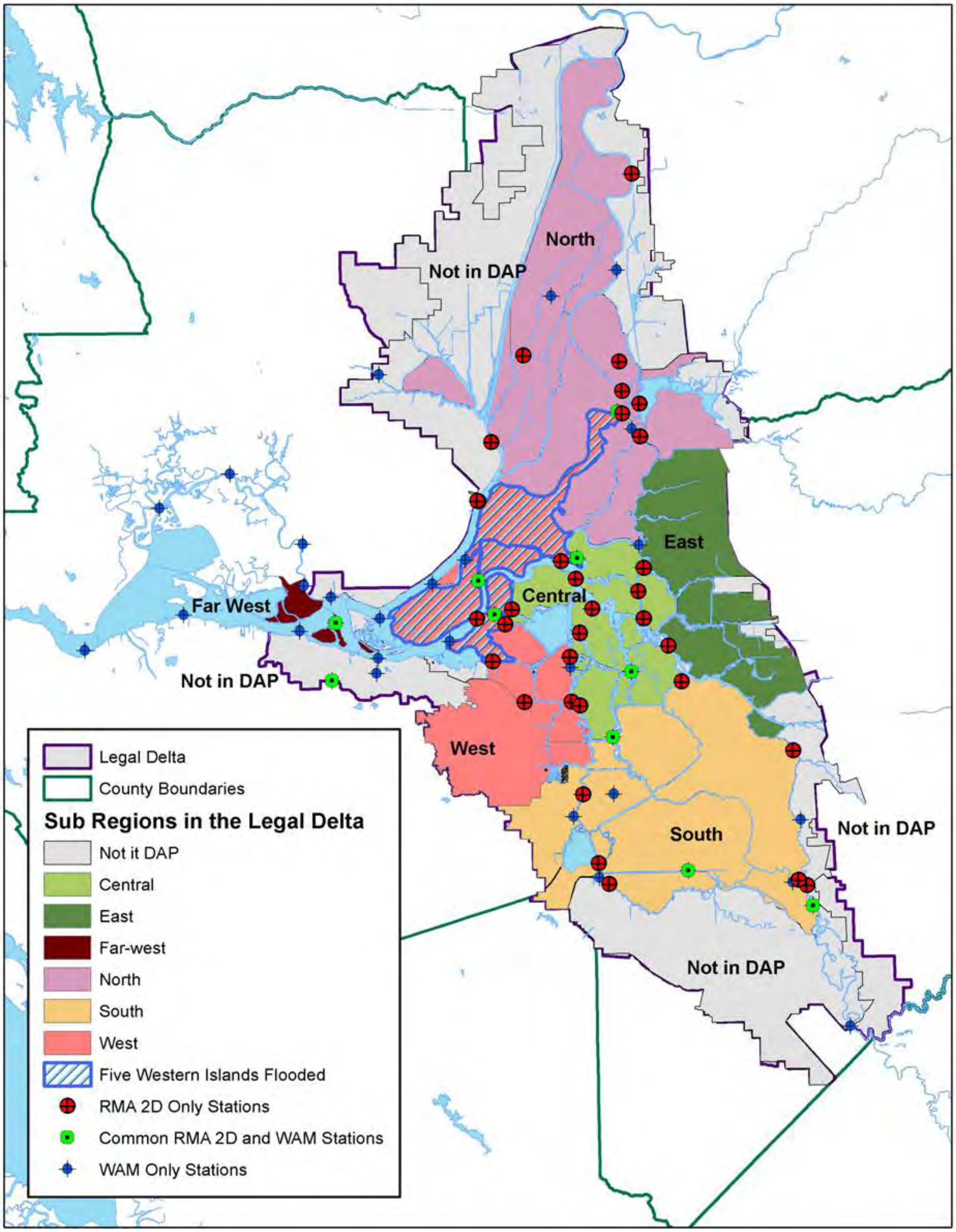


Figure 3 Land use coverage in the Delta Agricultural Production model and location of water salinity model output stations in the legal Delta. (Adapted from Medellín–Azuara et al. 2012a.)

recent analysis by CH2M Hill, which used USDA–NASS County Agricultural Commissioners' reports, adjusted in some cases to ensure positive returns to land and management. Whereas land use and applied water in SWAP can be represented at relatively fine scale, SWAP compiles production costs for each of the 20 CDWR crop groups at a regional scale, because this information is not available for all commodities for all counties and years.

Table 1 also compares of baseline revenues per acre from 2005 through 2008 and in 2030 in the Delta for its 14 crop groups, taking into account farmers' responses to prices, yields, and baseline salinity conditions, as discussed earlier in this section. As a result of yield and price changes, some crops maintain about the same irrigated land area (e.g., alfalfa and corn). Most vegetable and fruit crops (orchards, tomatoes, other vegetables, sugar beet, and vine crops) experience a slight increase (1% to 7%) in irrigated area. Lastly, almonds and pistachios, some grains, some field crops, irrigated pasture, and rice and subtropical crop groups have acreage reductions from 2% to 10%.

HYDRODYNAMIC MODELING AND WATER SALINITY INFORMATION FOR DAP

The hydrodynamic modeling used to estimate salinity changes of Delta waters is based on two models developed by Resource Management Associates, Inc. for the state-commissioned Delta Risk Management Strategies (DRMS) study and reported in Fleenor et al. (2008). Development, verification, calibration and validation of both models can be found in Fleenor et al. (2008), Bombardelli et al. (2010) and (2011), and Fleenor and Bombardelli (2013).

First, the one-dimensional Water Analysis Module (WAM) is used to estimate salinity changes with the introduction of dual conveyance of water exports and sea level rise. Fleenor et al. (2008) performed simulations with WAM over water years 1981 through 2000. Second, a two-dimensional RMA Bay-Delta model (referred to here as the "RMA 2-D" model) was used to estimate salinity changes from the permanent flooding of five western islands that serve as a salinity barrier at the Delta's western edge (hatched in Figure 3). RMA performed these 2-D simulations that spanned the April 2002 through December 2004 hydrologic period for the DRMS study. Permanent flooding represents conditions where the islands have either been flooded for some time or during winter months when considerable freshwater flows are available, but not the near-term results of a "Big Gulp" of salt water flowing into the Delta that might occur with catastrophic island failures during the summer or fall. We summarize these modeling results and show the model output water salinity sampling locations (Figure 3). To assign irrigation water salinity for each island and water salinity scenario we located the two closest sampling locations (Figure 3 and Medellín–Azuara et al. 2012a) and then selected the sampling station with the highest monthly average salinity during the irrigation season. (This choice was made to avoid under-estimating the salinity effects farmers might experience.) The supplementary tables in the project website (Appendices 1-4[2]) provide detailed information on the sampling stations used and simulated monthly average salinity levels by island and hydrodynamic modeling scenario.

To account for the largest possible monthly average salinity levels, we explored salinity conditions within a relatively long irrigation season (April 1 to September 30). This choice also likely overstates the average salinity conditions most farmers face when irrigating their crops, because salinity tends to be highest in the late summer and fall, when most irrigation is finished except for pasture and hay crops.

Salinity Scenarios

WAM simulations contrast 1981–2000 salinity conditions for three sea levels (current conditions, and for 1 and 3 feet of sea level rise). The sea level rise projections are within the range the California Ocean Protection Council (2011) recommends for long-term planning purposes, based on recent model projections for the mid- and late-21st century

2 *https://watershed.ucdavis.edu/project/water-quality-and-hydrodynamics*

(Vermeer and Rahmstorf 2009). Some projections anticipate the potential for higher sea level rise (55 in) by the end of the century, and these would likely generate higher salinity levels than those shown here. WAM simulations also include two Delta export configurations (current through-Delta exports and a dual conveyance system in which a 212.4 $m^3 s^{-1}$ [7,500 $ft^3 s^{-1}$] capacity peripheral facility is added that draws water from the Sacramento River at a point upstream of the Delta). RMA 2-D simulations contrast a 2002–2004 base salinity case with all islands intact and a scenario with five western islands flooded (Bradford, Brannan–Andrus, Jersey, Sherman, and Twitchell), the hatched area in Figure 3. RMA 2-D runs do not consider sea level rise. For WAM, we also contrast a base case (through-Delta exports with no sea level rise) and a dual conveyance case for critically dry years within the modeled time period (1987–1991 and 1994). For both WAM and RMA 2-D runs, all cases assume the same daily hydrology and water system operations (reservoir releases, Delta export volumes) as those which actually occurred during the modeled periods. In the case of dual conveyance, the model draws exports through the new conveyance system unless these exports would cause Sacramento River flows to fall below a minimum environmental flow of 283.2 $m^3 s^{-1}$ (10,000 $ft^3 s^{-1}$). This environmental constraint is introduced to avoid reverse flows at the intake points that could harm fish (Burau 2007). Average export levels during the 1981–2000 reference period used for WAM were 5.96 billion $m^3 yr^{-1}$ (4.83 million ac-ft yr^{-1}) (and 5.74 billion $m^3 yr^{-1}$ or 4.65 million ac-ft yr^{-1} for the dry and critical years), and 7.14 million ac-ft yr^{-1}(5.79 million ac-ft yr^{-1}) for the 2002–2004 reference period used for RMA 2-D. Reference salinity for each hydrodynamic model run are shown in supplementary tables in the project website (*https://watershed.ucdavis.edu/project/water-quality-and-hydrodynamics*, hydrodynamic modeling results used in DAP).

Figure 4 shows salinity as electrical conductivity during the irrigation season for the five agricultural sub regions within the Delta under different export conveyance and sea level rise cases. Baseline salinity (the solid blue "current conditions" bar) is highest in the western Delta and lowest in the northern Delta. At current sea level, dual conveyance would increase salinity in most regions (particularly in the west), though not necessarily in the eastern and central parts of the Delta (hatched blue bar). Sea level rise increases salinity in most cases (again, particularly in the west). However, dual conveyance operations combined with sea level rise may not increase salinity in the eastern and central Delta (hatched green and red bars). During dry years, salinity is generally higher than during other years in the modeled time period, and dual conveyance increases average salinity at least marginally in all regions in both the irrigation (April to September) and non-irrigation (October to March) seasons, as shown in Figure 5.

The permanent flooding of western islands does not result in large increases in salinity over the base case during the irrigation season, although it does increase salinity somewhat more in the non-irrigation season (Figure 6). The lack of major effects in either season reflect the nature of the modeling scenario: recall that these islands are treated as "pre-flooded"—with salinity levels set the same as the surrounding channels; this corresponds to long-term conditions or near-term flooding under high river-flow conditions within the Delta, not the near-term effect of a "Big Gulp" of saltwater that might occur if the islands flood in the summer or fall or a very dry winter or spring. The contrast between the irrigation and non-irrigation seasons may reflect the effects of the D-1641 regulations (adopted in the mid-1990s), which include requirements to maintain low "X2" salinity standards in the western Delta from February until June. As a result, water exporters responded by increasing pumping in the fall for storage and urban uses, drawing more saline water toward the pumps. Permanent flooding of western islands greatly increases the volume of flood tide inflows and reduces the ability of the out-flowing water to restrain salinity intrusion.

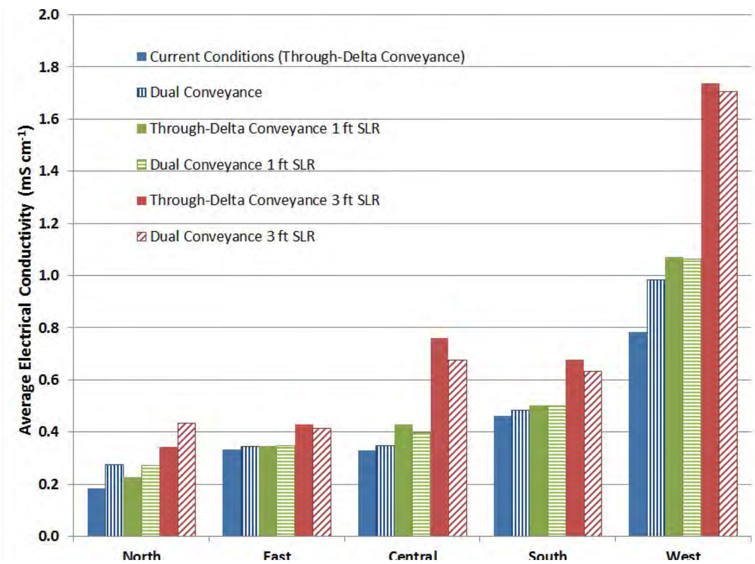


Figure 4 Electrical conductivity during the irrigation season (April to September) by WAM modeling scenario for five Delta sub-regions. Solid bars indicate through-Delta conveyance and hatched bars refer to dual conveyance.

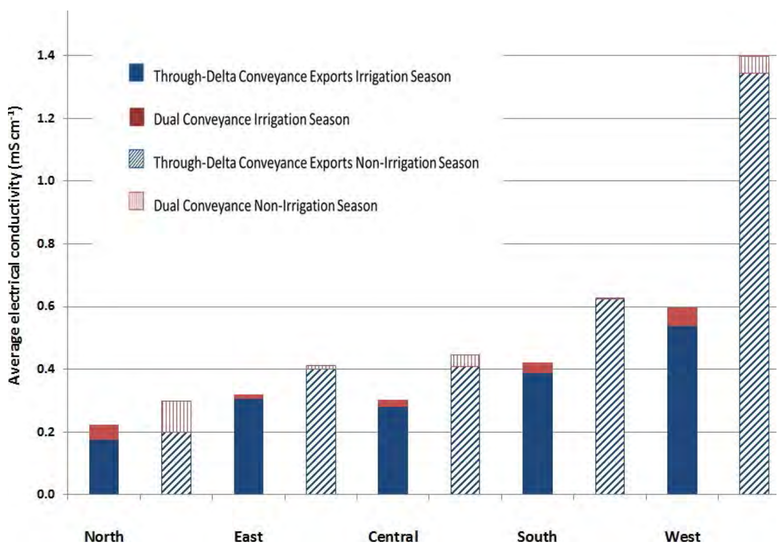


Figure 5 Electrical conductivity during dry years for through-Delta exports and dual conveyance configurations (current sea level)

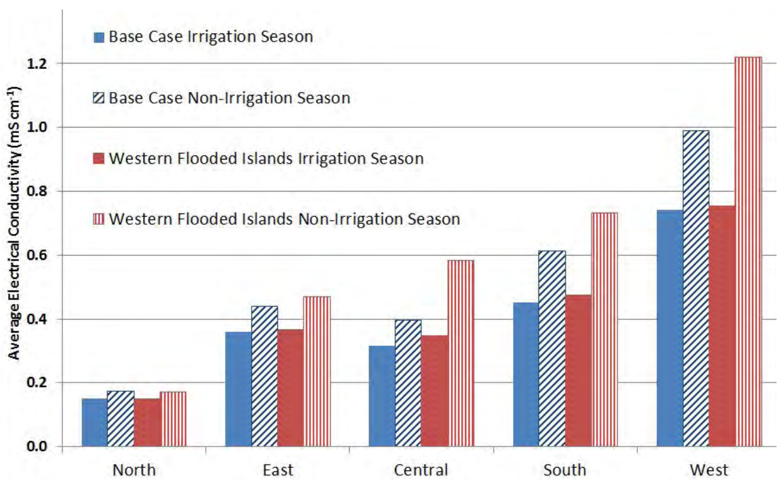


Figure 6 Electrical conductivity from the RMA 2-D hydrodynamic model for a base case and five western islands flooded during the irrigation season (solid bars) and non-irrigation season (hatched bars)

DAP MODELING RESULTS

In general, higher salinity reduces the relative yield of crops in the Delta. However, a large enough change to cause major yield losses throughout the Delta seems unlikely even under three feet of sea level rise or the flooding of the five western islands. Similarly, we do not find major changes in baseline cropping patterns or crop revenues with any of the hydrodynamic modeling salinity scenarios analyzed.

Hydrodynamic-Based Salinity Model Simulations of Revenue Losses

Sea level rise leads to limited crop revenue losses in the Delta, both with dual conveyance and through-Delta exports (Figure 7). Dual conveyance for Delta exports generally increases total revenue losses somewhat relative to through-Delta exports, but these losses remain well under 1% of total revenues. During dry years, when Delta waters are more saline, dual export conveyance gives the highest revenue losses, slightly above 0.7% (third bar from the left in Figure 7), roughly \$4.5 million yr^{-1}, with most losses occurring in the western Delta.

Results using the RMA 2-D hydrodynamic modeling for salinity with permanently flooded western islands (Figure 6) also show little revenue loss during the irrigation season. Some areas in the north of the Delta may even see slight decreases in water salinity and corresponding increases in crop revenues. Because most salinity changes occur outside the main irrigation season, crop yield and revenue effects are largely confined to acreage planted to winter crops such as wheat; thus the absolute revenue losses are very small because the acreage of winter plantings is itself small (less than 0.2% in the affected areas in the western and southern Delta).

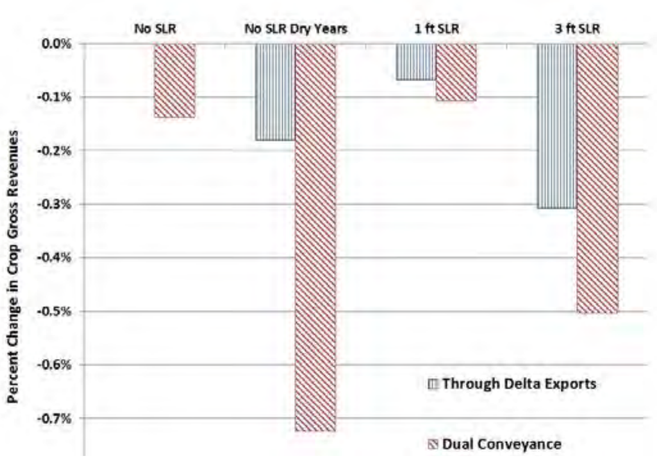


Figure 7 Percent change in agricultural crop revenues across WAM water salinity scenarios

Of course, beyond their effects on water salinity in the Delta, the permanent flooding of the five western islands would also lead to losses from flooded land being taken out of production. Elsewhere, we used DAP to show that farm revenue losses from the permanent flooding of 19 western and central Delta islands would far exceed the salinity-related losses shown here—roughly \$66 to \$90 million yr^{-1}—10% or more of baseline crop revenues (Medellín–Azuara et al. 2012a). (Suddeth et al. [2010] had earlier shown that these 19 islands would not merit repair after flooding based on the costs of repair and the value of agricultural production and other assets on the islands.) Changes of this magnitude would also ripple through the regional economy (multiplier effects), causing additional losses in revenues and value added. Large conversions of farmland to habitat could also have more substantial local and regional economic effects than the salinity changes modeled here (Medellín–Azuara et al. 2012a).

Sensitivity Analysis

To test the robustness of both the hydrodynamic and salinity transport simulations and the crop yield response model (DAP), we conducted two separate sensitivity analyses. The first tests for changes in water salinity in more recent levels of water exports in the Delta. The second tests the sensitivity of the crop production model to higher levels of irrigation water salinity than those obtained from hydrodynamic and salt transport modeling with WAM and RMA2/RMA11.

Because the historical 1981–2000 water year, average export of 5.96 billion $m^3\ yr^{-1}$ (4.83 million ac-ft yr^{-1}) might not be representative of more recent, higher export levels, we replicated Fleenor et al. (2008) WAM hydrodynamic modeling runs using the 1996–2005 water years, when average exports were 7.28 billion $m^3\ yr^{-1}$ (5.9 million ac-ft yr^{-1}). We found no major increase in salinity for any of the 52 sampling stations considered during the irrigation season. The largest increase was 2% at the Mokelumne River station near Terminous Tract, and the average electrical conductivity across all stations in the Delta was generally lower than during the 1981–2000 period. This is because the 1996–2005 period was fully covered in the D-1641 requirements in operation from the mid-1990s; under these requirements, the isohaline line of 2 ppt must be maintained in the far western Delta (around Chipps Island) from February to June to support delta smelt. During the non-irrigation season of the 1981–2000 time period, however, dual conveyance may increase salinity in some areas, including those near Old and Middle Rivers, which are intake points to supply Delta water for urban uses in Contra Costa Water District. The change in salinity in these locations is about 15%, which would increase water treatment costs for the Contra Costa Water District service area if the utility were unable to store water during lower salinity periods for later use.

To test the sensitivity of the DAP agro-economic model, we also examine cases with uniform values of irrigation water salinity for all islands at 1%, 3%, 5% and 10% of seawater salinity (set at 33 practical salinity units, psu: 50.4 mS cm^{-1} in surface at 25 °C (77 °F) or 33 ppt. DAP responds more abruptly to electrical conductivity levels beyond one percent of seawater. This analysis supports conclusions from earlier modeling (Medellín–Azuara et al. 2012a): crop revenue losses from salinity increases caused by dual conveyance and sea level rise are relatively low because most higher value crops are not located in parts of the Delta that experience the highest salinity increases.

When identical, higher levels of irrigation water salinity are assumed for all Delta islands and sub regions, DAP reports generally higher agricultural revenue losses than those shown in the previous section. This results from two factors: first, the proportional salinity increases are much higher on islands and sub regions in the Delta where salinity is currently low; second, these areas also tend to have greater concentrations of higher-value crops. Thus, increased salinity conditions and losses of higher value crops increase revenue losses substantially (Figure 8). The absolute revenue losses are highest in the northern and southern Delta, where such higher value crops predominate (Figure 9).

CONCLUSIONS

In the coming decades, the Sacramento–San Joaquin Delta is likely to experience changing land and water conditions as a result of a variety of natural and anthropogenic forces (Lund et al. 2010). Sea level rise, permanent flooding of some islands that lie below sea level, and altered water export operations are likely to affect water salinity and crop farming in the Delta.

In this paper, we quantified changes in agricultural crop revenues in the Delta for a range of water salinity changes using the DAP model. We used geo-referenced land use and water salinity information from field data and existing hydrodynamic modeling was employed to assess agricultural production under the different water salinity scenarios. We also tested the sensitivity of the model using more recent, higher export levels and a set of fixed scenarios with higher salinity levels, based on percentages of seawater salinity.

Several conclusions arise from this work:

1. Salinity changes from sea level rise, estimated by hydrodynamic modeling, reduce total agricultural crop revenues in the Delta by less than 1% of current revenues.

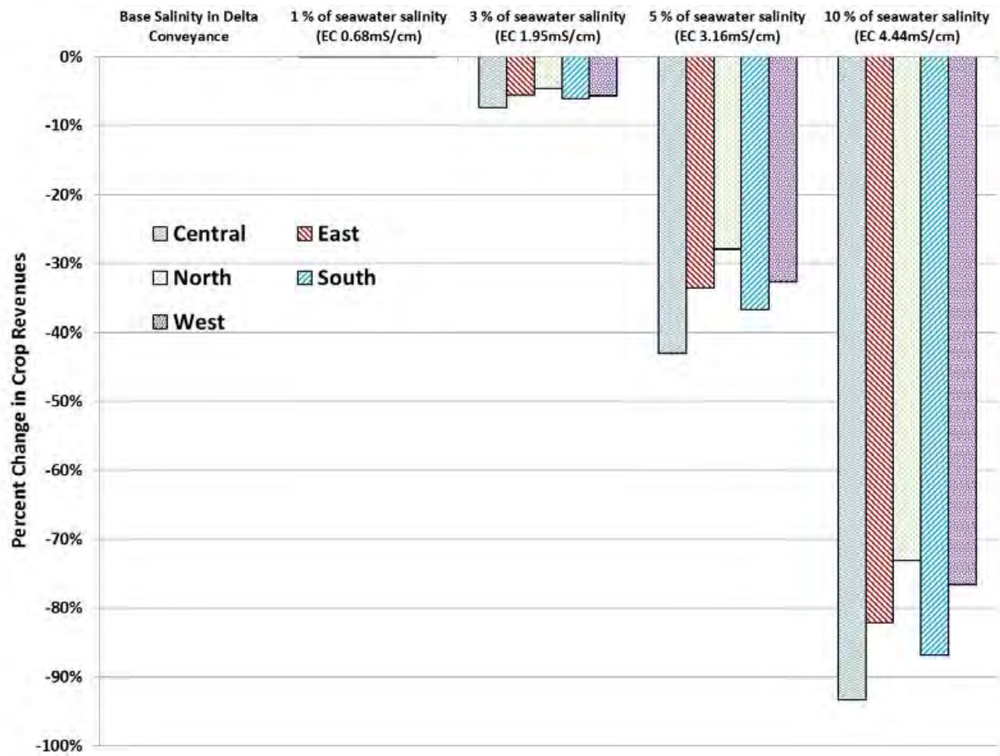


Figure 8 Change in crop revenues by percent of seawater salinity level (sensitivity analysis)

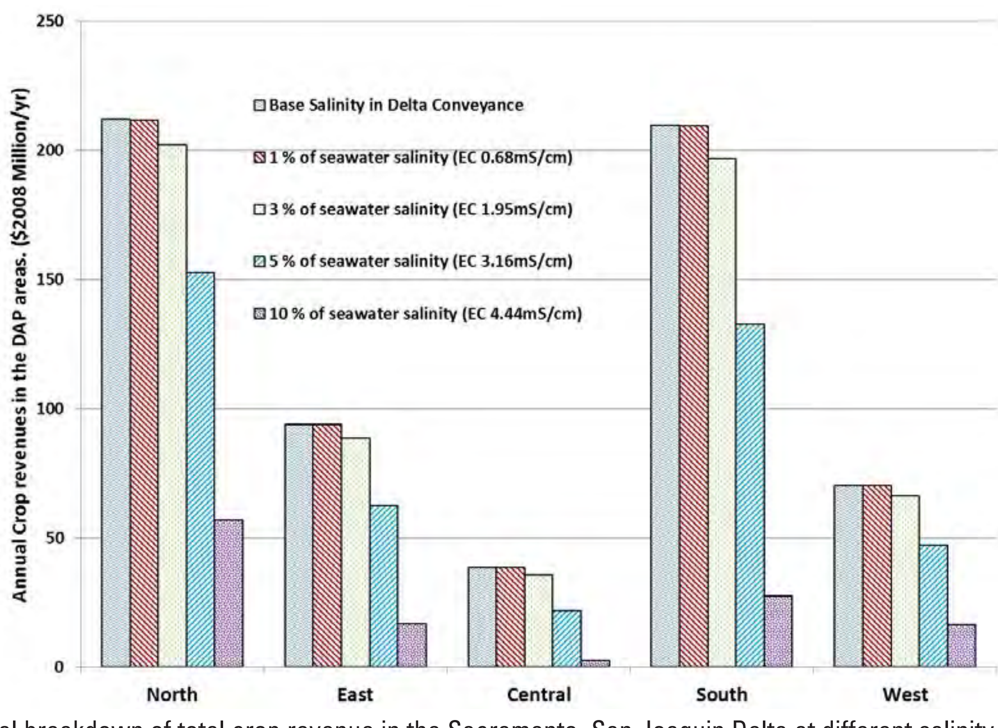


Figure 9 Sub-regional breakdown of total crop revenue in the Sacramento–San Joaquin Delta at different salinity levels (sensitivity analysis)

2. Water export operations with dual conveyance (including a peripheral canal or tunnel intake in the northern Delta) would slightly decrease crop revenue, especially during dry years, but these losses would remain within 1% of total projected Delta crop revenues under current salinity conditions.

3. Similar conclusions also hold for salinity losses arising from the permanent flooding of the five western islands that serve as a salinity barrier; salinity increases more during the non-irrigation season. Direct agricultural revenue losses from island flooding are much greater.

4. Total crop revenue losses from these salinity increases generally remain small because areas in the Delta with the greatest salinity effects now mostly grow lower-value crops. Farmers' ability to vary crop mix in response to salinity increases also reduces crop revenue losses. Economic losses from the permanent removal of agricultural lands because of island flooding or habitat conversions shown elsewhere (Medellín–Azuara et al. 2012a) are potentially much higher than the salinity effects found here.

5. Sensitivity analyses show that large increases in salinity for all Delta islands, beyond 3% of seawater, would greatly reduce Delta crop revenues. The greatest losses would be in areas that currently grow more salt-sensitive, higher-value crops; these areas are further inland from San Francisco Bay.

6. A better understanding of the hydrodynamics of Delta water salinity is needed through the aid of 3-D models, both to assess the combined effects of island flooding and sea level rise and to assess additional water operation alternatives. However, this study demonstrates the insights and potential from more detailed integrated analysis of crop production, adaptation, and revenue losses from salinity for a wide range of salinity and management conditions.

ACKNOWLEDGEMENTS

This study draws on earlier work supported by the UC Davis Center for Watershed Sciences and the Public Policy Institute of California. Funding from the S.D. Bechtel, Jr. Foundation supported additional research undertaken for this article. Research assistance from Molly Ferrell, Michelle Lent, and Duncan MacEwan in preparing and/or providing datasets for this work is appreciated. We are also thankful for the comments from peer reviewers on earlier versions of this work, and on the manuscript submitted to this journal, which helped improve it substantially.

REFERENCES

Bombardelli FA, Kohne JR, Behrens DK. 2010. Comparing Delta flow and transport models: theoretical and numerical basis. Report to the California Water Resources Control Board, Center for Watershed Sciences, UC Davis. Davis (CA): Center for Watershed Sciences, University of California, Davis. p. 1–92.

Bombardelli FA, Reddy S, Kohne JR. 2011. Comparing delta flow and transport models: numerical results of flow and salinity. Report to the California Water Resources Control Board, Center for Watershed Sciences, UC Davis. Davis (CA): Center for Watershed Sciences, University of California, Davis. p. 1–422.

Brunke H, Howitt R, Sumner D. 2005. Future food production and consumption in California under alternative scenarios. California water plan update 2005. Vol. 4, Reference guide. Sacramento (CA): California Department of Water Resources. p. 1055–1080.

Burau JR. 2007. DRERIP hydrodynamic/transport conceptual model. Draft report dated 20 December 2007.

[CDPW] California Department of Public Works. 1931. Bulletin No. 27. Variation and control of Salinity in Sacramento San Joaquin Delta and Upper San Francisco Bay. Sacramento (CA): California State Printing Office. 440 p.

California Ocean Protection Council. 2011. Resolution of the ocean protection council on sea level rise. Available from: *http://www.opc.ca.gov/2011/04/resolution-of-the-california-ocean-protection-council-on-sea-level-rise*.

Cardon GE, Letey J. 1992. Soil-based irrigation and soil salinity management Model I: plant water uptake. Soil Sci Soc Am J 56:1881–1887.

[DPC] Delta Protection Commission. 2012. Economic sustainability plan for the Sacramento–San Joaquin Delta. West Sacramento (CA): Delta Protection Commission. [cited 28 February 2012]. Available from: *http://www.delta.ca.gov*.

Feinerman E. 2000. Salinity management for sustainable irrigation. In: Hillel D, editor. Salinity management for sustainable irrigation: integrating science, environment, and economics. Washington, D.C.: The World Bank. p. 63–87.

Fleenor WE, Bombardelli F. 2013. Simplified 1-D hydrodynamic and salinity transport modeling of the Sacramento–San Joaquin Delta: sea level rise and water diversion effects. San Fran Est Water Sci [Internet]. [cited 24 January 2014];11(4). Available from: *http://www.escholarship.org/uc/item/3km0d0kt*.

Fleenor WE, Hanak E, Lund JR, Mount JF. 2008. Appendix C: Delta hydrodynamics and salinity conditions. In: Lund J, Hanak E, Fleenor W, Bennett W, Howitt R, Mount J, Moyle P. 2010. Comparing futures for the Sacramento–San Joaquin Delta. San Francisco (CA): Public Policy Institute of California. [cited 28 February 2012]. Available from: *http://www.ppic.org/content/pubs/other/708ehr_appendixc.pdf*.

Hillel D. 2000. Salinity management for sustainable agriculture: integrating science, environment and economics. Washington, D.C.: The World Bank. 104 p.

Howitt RE. 1995. Positive mathematical programming. Am J Ag Econ 77(2):329–342.

Howitt RE, Kaplan J, Larson D, MacEwan D, Medellín–Azuara J, Horner G, Lee NS. 2009. Central Valley Salinity Report. Report for the State Water Resources Control Board. Davis (CA): University of California, Davis. [cited 24 February 2012]. Available from: *http://swap.ucdavis.edu*.

Howitt RE, Medellín–Azuara J, MacEwan D, Lund JR. 2012. Calibrating disaggregate economic models of agricultural production and water management. Environ Model Soft 38:244–258.

Knapp KC, Wichelns D. 1990. Dynamic optimization models forsalinity and drainage management. In: Tanji KK, editor. Agricultural salinity assessment and management. New York (NY): American Society of Civil Engineers. p. 530–548.

Lund J, Hanak E, Fleenor W, Howitt R, Mount J, Moyle P. 2007 Envisioning futures for the Sacramento–San Joaquin Delta. San Francisco (CA): Public Policy Institute of California. 300 p.

Lund J, Hanak E, Fleenor W, Bennett W, Howitt R, Mount J, Moyle P. 2010 Comparing futures for the Sacramento–San Joaquin Delta. Berkeley (CA): University of California Press. 229 p.

Medellín–Azuara J, Hanak E, Howitt R, Lund JR. 2012a. Transitions for the Delta economy. San Francisco (CA): Public Policy Institute of California. [cited 2012 February 28]. Available from: *http://www.ppic.org/main/publication.asp?i=913*.

Medellín–Azuara J, Howitt RE, MacEwan D, Lund JR. 2012b. Economic impacts of climate-related yield changes in California. Climatic Change 109(Supplement 1):S387–S405.

Medellín–Azuara J, Howitt RE, Lund J, Hanak E. 2008. Appendix I: Economic effects on agriculture of water export salinity south of the Sacramento–San Joaquin Delta. In: Lund J, Hanak E, Fleenor W, Bennett W, Howitt R, Mount J, Moyle P. 2010. Comparing futures for the Sacramento–San Joaquin Delta. San Francisco (CA): Public Policy Institute of California. [cited 2012 February 28]. Available from: *http://www.ppic.org/content/pubs/other/708EHR_appendixI.pdf*.

Mount J. 2007. Sea level rise and Delta planning. Memo from the CALFED Independent Science Board to Mike Healey, CALFED Lead Scientist, dated 6 September 2007. *http://calwater.ca.gov/science/pdf/isb/meeting_082807/ISB_response_to_ls_sea_level_090707.pdf*.

Shannon CE. 1948. A mathematical theory of communication. Bell System Tech J 27:379–423.

Suddeth R, Mount J, Lund JR. 2010. Levee decisions and sustainability for the Sacramento–San Joaquin Delta. San Fran Estuar Water Sci [Internet]. [cited 01 February 2012]; 8(2). Available from: *http://www.escholarship.org/uc/item/9wr5j84g*.

Van Genuchten MT. 1983. Analyzing crop salt tolerance data: model description and user's manual. Riverside (CA): Agricultural Research Service, U.S. Salinity Laboratory, U.S. Department of Agriculture. p. 1–47.

Van Genuchten MT, Hoffman GJ. 1984. Analysis of crop salt tolerance data. In: Shainberg I, Shalhevet J, editors. Soil salinity under irrigation, processes and management. Ecological Studies No. 51. Berlin: Springer–Verlag. p. 258–271.

Title:
Subsidence, sea level rise, and seismicity in the Sacramento-San Joaquin Delta

Journal Issue:
San Francisco Estuary and Watershed Science, 3(1)

Author:
Mount, Jeffrey, University of California, Davis
Twiss, Robert, University of California, Berkeley

Publication Date:
2005

Publication Info:
San Francisco Estuary and Watershed Science

Permalink:
http://escholarship.org/uc/item/4k44725p

Keywords:
Sacramento-San Joaquin Delta, subsidence, levee integrity, seismicity, accomodation space, levee failure

Local Identifier:
jmie_sfews_10966

Abstract:
Anthropogenic accommodation space, or that space in the Delta that lies below sea level and is filled neither with sediment nor water, serves as a useful measure of the regional consequences of Delta subsidence and sea level rise. Microbial oxidation and compaction of organic-rich soils due to farming activity is the primary cause of Delta subsidence. During the period 1900-2000, subsidence created approximately 2.5 billion cubic meters of anthropogenic accommodation space in the Delta. From 2000-2050, subsidence rates will slow due to depletion of organic material and better land use practices. However, by 2050 the Delta will contain more than 3 billion cubic meters of anthropogenic accommodation space due to continued subsidence and sea level rise. An Accommodation Space Index, which relates subaqueous accommodation space to anthropogenic accommodation space, provides an indicator of past and projected Delta conditions. While subsidence and sea level rise create increasing anthropogenic accommodation space in the Delta, they also lead to a regional increase in the forces that can cause levee failure. Although these forces take many forms, a Levee Force Index can be calculated that is a proxy for the cumulative forces acting on levees. The Levee Force Index increases significantly over

the next 50 years demonstrating regional increases in the potential for island flooding. Based on continuing increases in the Levee Force Index and the Accommodation Space Index, and limited support for Delta levee upgrades, there will be a tendency for increases in and impacts of island flooding, with escalating costs for repairs. Additionally, there is a two-in-three chance that 100-year recurrence interval floods or earthquakes will cause catastrophic flooding and significant change in the Delta by 2050. Currently, the California Bay-Delta Authority has no overarching policy that addresses the consequences of, and potential responses to, gradual or abrupt landscape change in the Delta.

MARCH 2005

SAN FRANCISCO ESTUARY & WATERSHED SCIENCE
Published by the California Bay-Delta Authority Science Program and the John Muir Institute of the Environment

Subsidence, Sea Level Rise, and Seismicity in the Sacramento-San Joaquin Delta

Jeffrey Mount
University of California, Davis
mount@geology.ucdavis.edu

Robert Twiss
University of California, Berkeley

ABSTRACT

Anthropogenic accommodation space, or that space in the Delta that lies below sea level and is filled neither with sediment nor water, serves as a useful measure of the regional consequences of Delta subsidence and sea level rise. Microbial oxidation and compaction of organic-rich soils due to farming activity is the primary cause of Delta subsidence. During the period 1900-2000, subsidence created approximately 2.5 billion cubic meters of anthropogenic accommodation space in the Delta. From 2000-2050, subsidence rates will slow due to depletion of organic material and better land use practices. However, by 2050 the Delta will contain more than 3 billion cubic meters of anthropogenic accommodation space due to continued subsidence and sea level rise. An Accommodation Space Index, which relates subaqueous accommodation space to anthropogenic accommodation space, provides an indicator of past and projected Delta conditions. While subsidence and sea level rise create increasing anthropogenic accommodation space in the Delta, they also lead to a regional increase in the forces that can cause levee failure. Although these forces take many forms, a Levee Force Index can be calculated that is a proxy for the cumulative forces acting on levees. The Levee Force Index increases significantly over the next 50 years demonstrating regional increases in the potential for island flooding. Based on continuing increases in the Levee Force Index and the Accommodation Space Index, and limited support for Delta levee upgrades, there will be a tendency for increases in and impacts of island flooding, with escalating costs for repairs. Additionally, there is a two-in-three chance that 100-year recurrence interval floods or earthquakes will cause catastrophic flooding and significant change in the Delta by 2050. Currently, the California Bay-Delta Authority has no overarching policy that addresses the consequences of, and potential responses to, gradual or abrupt landscape change in the Delta.

KEYWORDS

Sacramento-San Joaquin Delta, subsidence, levee integrity, seismicity, accommodation space, levee failure

SUGGESTED CITATION

Mount J, Twiss R. 2005. Subsidence, sea level rise, seismicity in the Sacramento-San Joaquin Delta. San Francisco Estuary and Watershed Science. Vol. 3, Issue 1 (March 2005), Article 5. http://repositories.cdlib.org/jmie/sfews/vol3/iss1/art5

INTRODUCTION[1]

The CALFED Bay-Delta Program (CALFED) is an outcome of a 1994 agreement among agencies and environmental and water user stakeholders (the so-called "Delta Accord") that was intended to provide interim environmental guidelines while CALFED worked with the agencies and stakeholders to develop a long-term solution to environmental and water supply problems in the Sacramento-San Joaquin Delta (Delta). The Delta provides at least a portion of the water supply for about two-thirds of California's population, and provides a migratory pathway for four fish that are listed as endangered or threatened pursuant to the federal Endangered Species Act. Two of the overriding CALFED goals are to maintain the reliability of water supplies from the Delta and to restore the Delta ecosystem and that of its watershed. More information about the CALFED Program can be found at http://calwater.ca.gov/.

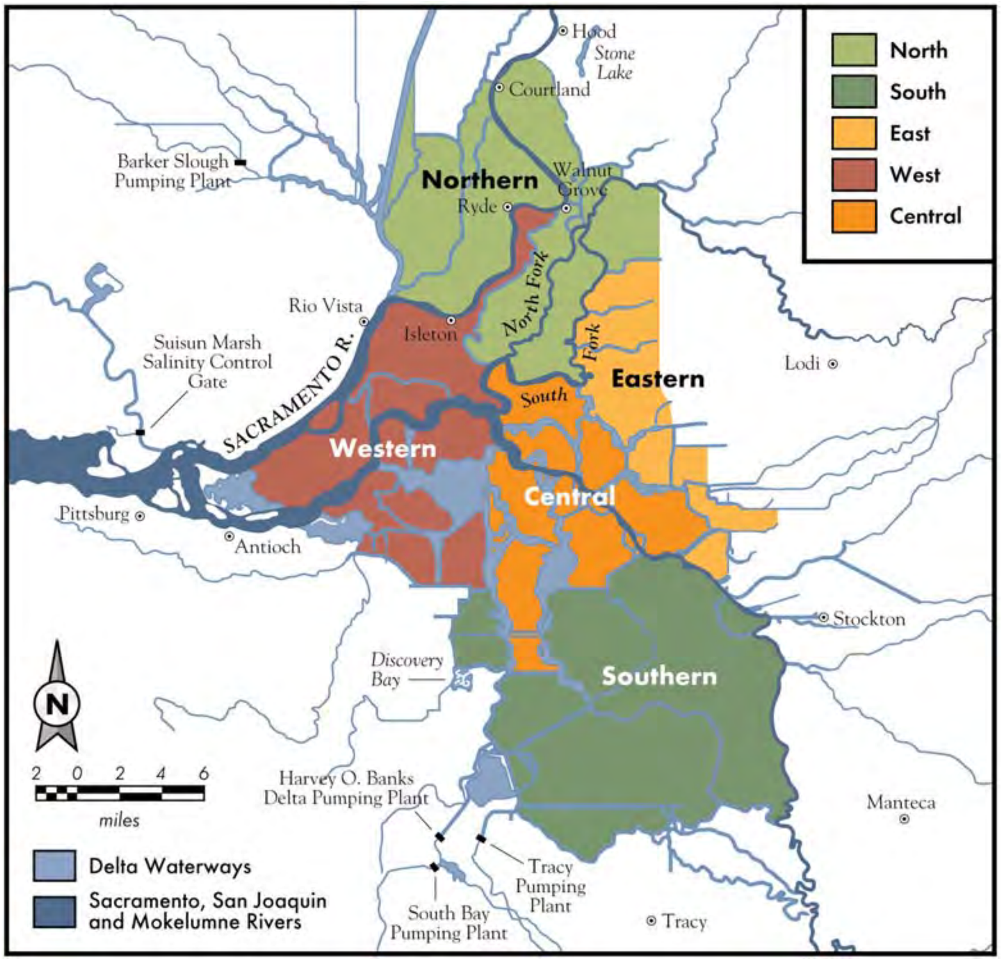


Figure 1. Generalized map of subsided portion of the Sacramento-San Joaquin Delta indicating regions discussed in text.

The hydraulic integrity of the Sacramento-San Joaquin Delta is maintained by more than 1700 km of levees, most of which are privately owned and maintained (DWR 1995). Microbial oxidation and consolidation of organic-rich soils on Delta islands is causing widespread subsidence (Figure 1), with island elevations in the west and central Delta locally more than 8 m below mean sea level (Ingebritsen et al. 2000). Island subsidence has reduced the stability of Delta levees, increasing the risk of failure (DWR 1986, 1989). Embankment and foundation materials for most Delta levees are substandard, adding the risk of failure during seismic events (Torres et al. 2000). It is generally acknowledged that the current channel network of the Delta and the hydraulic disconnection between islands and surrounding channels is necessary for meeting water quality standards at the south Delta pumping plants that support the Central Valley Project, State Water Project and Contra Costa Water District (NHI 1998; CALFED 2000). CALFED (2000) and the California Department of Water Resources (DWR 1986, 1989, 1995) have noted that failure of the levees and the flooding of subsided islands, particularly during the

1. The following article is the first in our new category, Policy and Program Analyses. The paper itself has been adapted from a report the authors submitted to the Independent Science Board, a standing panel of distinguished scientists and engineers convened to help the CALFED Bay-Delta Authority (Authority) establish an independent and objective view of the science issues underlying important policy decisions. The authors are members of the Independent Science Board.

The Authority itself arose out of a 1994 accord among federal and state agencies and stakeholders designed to improve the reliability of water supplies diverted from the Sacramento-San Joaquin Delta and to restore the health of the San Francisco Estuary and its watershed. The Authority is charged with meeting the water supply and ecosystem goals. More details about Authority goals and programs can be found at http://calwater.ca.gov/

spring and summer months, has the potential to significantly degrade Delta water quality by (1) drawing brackish water into the Delta during rapid flooding of Delta islands and (2) changing the dynamics of the tidal prism in the west Delta. Additionally, CALFED's Ecosystem Restoration Program (CALFED 2004) has concluded that subsided islands and deeply flooded islands provide poor quality habitat for native aquatic plant and animal communities, and are generally viewed as undesirable.

With the exception of recognizing the impacts of population growth and increased water demand, federal and state programs that seek to improve water quality, water supply reliability, and ecosystem health in the Delta are predicated upon maintaining the existing levee and channel network. We found no comprehensive CALFED plan or policy that addresses response to gradual or abrupt changes in hydrologic, geomorphic, geotechnical and cultural factors that influence levee integrity. In this report we present low-resolution simulations of potential changes in Delta levee integrity through 2050. These simulations assume business-as-usual approaches to management of the Delta, principally for agriculture. Continued island subsidence, coupled with eustatic rise in sea level, will threaten levee stability significantly by 2050, leading to increased potential for island flooding. Additionally, it is likely that a seismic event or regional flood will impact the levee network of the Delta. Landscape change, whether gradual or abrupt, will affect CALFED programs in the San Francisco Bay, Sacramento-San Joaquin Delta, and the watershed, and should be considered by the California Bay-Delta Authority Independent Science Board.

BACKGROUND

Historic accommodation space

Sediment core analyses indicate that the Sacramento-San Joaquin Delta has been a tidal freshwater marsh, with a network of channels, sloughs and islands, for more than 6,000 years (Shlemon and Begg 1975; Atwater 1982). The persistence of intertidal conditions reflects a dynamic equilibrium between processes that regulated the influx of sediment into the Delta, the production of organic sediment within the Delta, and the export of sediment to the San Francisco Bay. A preserved stratigraphic record of intertidal conditions indicates that regional tectonic subsidence and sea level rise were sufficient to allow net accumulation of sediment in the Delta during that time (Atwater et al. 1979; Atwater and Belknap 1980; Orr et al. 2003). This record reflects the long-term formation of accommodation space, or space that is available for the accumulation and preservation of deposited sediment. The concept of accommodation space is well-established within the geologic literature and forms the underpinnings of modern concepts of depositional sequence stratigraphy (Emery and Meyers 1996).

In estuarine settings like the Sacramento-San Joaquin Delta, the formation and destruction of accommodation space controls the distribution and character of sediment deposition and related environmental conditions at large scales. For any given interval of time, accommodation space is created by eustatic (global) sea level rise and subsidence of the bed, typically associated with sediment compaction and tectonic subsidence of the crust. The eustatic rise (or fall) of sea level and the rate of subsidence control the rate at which accommodation space is either created or, in the case of falling sea level or crustal uplift, lost. In intertidal systems, accommodation space is filled with water and sediment.

Where rates of organic and inorganic sediment deposition keep pace with accommodation space formation, intertidal conditions persist; where rates of accommodation space formation exceed sediment deposition, there is a landward shift in sedimentary environments (known as transgression) and subtidal conditions expand. In deltaic or estuarine settings, sediment will tend to move through or bypass areas of low available accommodation space (supratidal or high intertidal) and accumulate in areas with higher accommodation space (low intertidal or subtidal). This process, which is governed in part by tidal energy and wind waves, regulates the movement of sediment through estuarine depositional systems and is responsible for large-scale lateral shifts in sedimentary environments (Pethick 1996; Pethick and Crook 2000; Reed 2002a, 2002b).

Anthropogenic accommodation space

Prior to the conversion of the Delta to farms, the creation of accommodation space was balanced by sedimentation, maintaining persistent tidal marsh conditions. Sedimentation on marsh platforms consisted of sub-equal mixes of inorganic material, derived from the watershed, and locally-derived organic material from highly-productive tule marshes. Beginning in the late 1800s, there were substantial changes in the balance between the creation of accommodation space and sedimentation patterns. In the 1880s the Delta was impacted by a wave of hydraulic mining sediment (Gilbert 1917). Since accommodation space was limited within the Delta, the bulk of this material by-passed the region, eventually accumulating in San Pablo Bay and other portions of the San Francisco Bay (Jaffe et al. 1998). During and immediately following the arrival of the hydraulic mining sediment, widespread reclamation of Delta tule marsh islands began. By 1930, virtually all of the marshes of the Delta had been reclaimed (Thompson 1957). This reclamation involved construction of more than 1700 km of levees and stabilization of the channel network in the configuration much like that seen today.

Farming of the Delta islands required the construction of extensive drainage ditches to lower water tables below crop root zones. Draining tule marsh soils initiated a sustained period of land subsidence that continues today (Prokopovitch 1985; DWR 1995; Ingebritson et al. 2000). Subsidence of Delta histosols is related to their organic content and farming practices (Figure 2). Draining of organic-rich soils leads to compaction and microbial oxidation of organic matter. Deverel et al. (1998) and Deverel and Rojstaczer (1996) demonstrated that gaseous CO_2 flux associated with microbial oxidation accounts for approximately 75% of

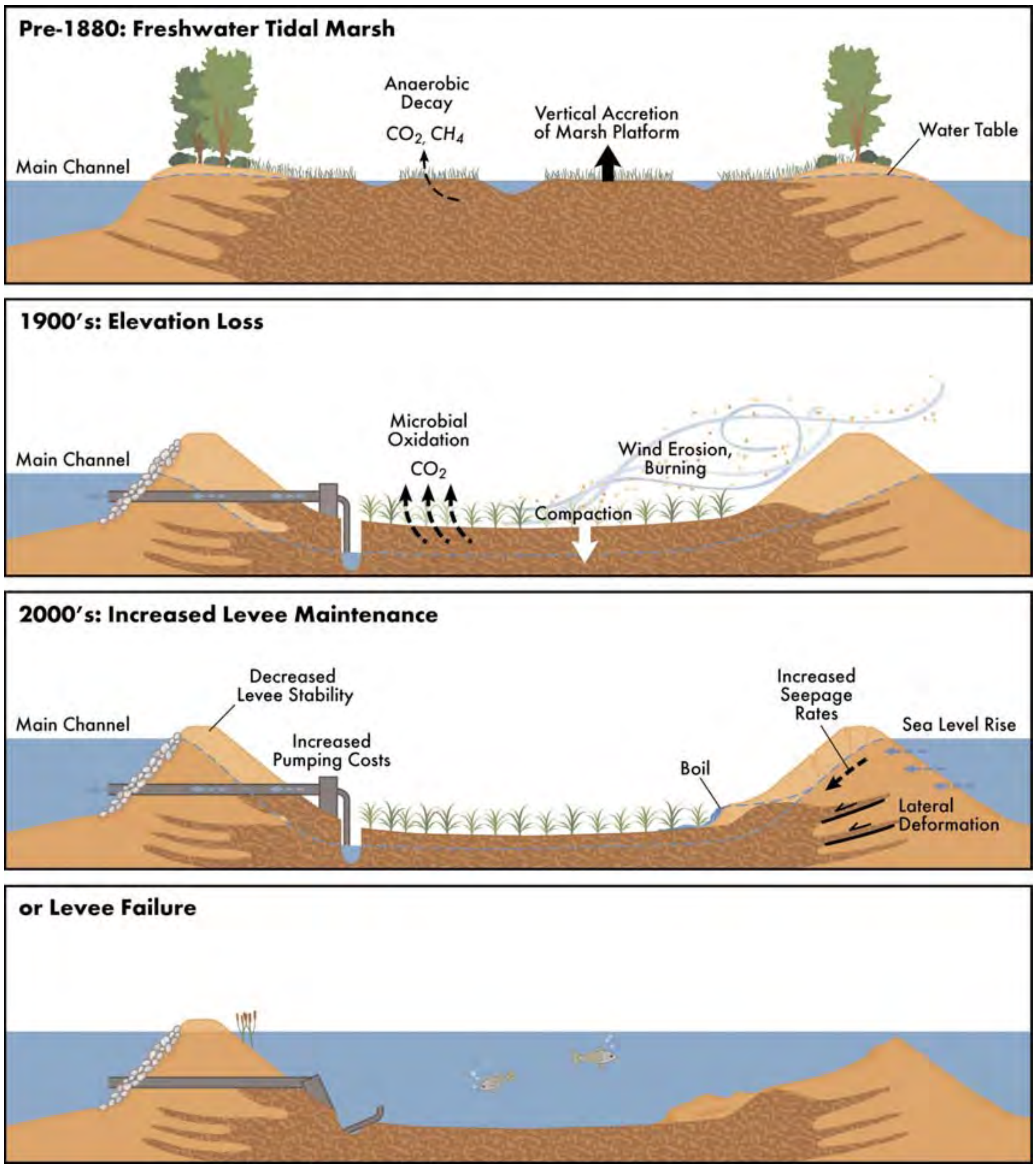


Figure 2. Conceptual diagram illustrating evolution of Delta islands due to levee construction and island subsidence. Modified from Ingebritsen et al. (2000).

current elevation losses, while the remaining 25% is associated with consolidation due to dewatering of the soils and compaction of saturated, underlying soils. Prior to 1950, poor land use practices, including burning of peat soils and wind erosion, exacerbated soil losses due to microbial oxidation (summary in Deverel 1998). Today, the Delta is a mosaic of levee-encased subsided islands with elevations locally reaching more than 8 m below mean sea level.

Subsidence of Delta islands created a new form of accommodation space. This *anthropogenic accommodation space* is distinguished by the fact that it is filled with neither sediment nor water, yet lies below mean sea level. The current levee system imperfectly isolates this space from processes that seek to fill it throughout the Delta. We suggest here that the amount of anthropogenic accommodation space is a 3-dimensional, landscape-scale measure of potential consequence of subsidence within the Delta. When levee breaches occur on deeply-subsided islands, rapid filling draws brackish water into the Delta, temporarily degrading water quality over a large region (DWR 2002). Known colloquially as the "Big Gulp," the water quality impact of island filling is principally a function of the magnitude and location of anthropogenic accommodation space. Island flooding directly affects tidal prism dynamics within the Delta (DWR 2002), with the potential for long-term degradation of water quality. The magnitude of the impact depends upon the location of flooded islands, the volume of water within the island, and the geometry of breach openings.

Levee instability

While regional increases in anthropogenic accommodation space in the Delta increase the consequence of island flooding, there is increase in the concomitant force that acts to destabilize levees and introduce water and sediment into available accommodation space. At the local scale, the processes that cause levee failure are diverse and commonly exacerbated by island subsidence. The increase in head difference between the water surface of the Delta channels and the interior of the islands increases hydrostatic forces on levees and seepage rates through and beneath levees. Depending upon location and magnitude, subsidence increases levee foundation problems by reducing lateral support and shear resistance, promoting settling or deformation of underlying peat layers (Foote and Sisson 1992; Enright 2004). This leads to lateral spreading, slumping and cracking of levees, which increases the likelihood of their failure due to seepage erosion or overtopping.

Susceptibility of Delta levees to failure is highly variable and, to date, poorly-documented (Torres et al. 2000; CALFED 2004). This variability and poor understanding make it difficult to address precisely the level of risk associated with island subsidence at the landscape scale. However, generalizing over the regional scale, the forces that are acting on Delta levees derive, in some form, from the differences in elevation between the water surface of the channels and the interior of the subsided island. For this reason, hydrostatic force for any length of levee can be used as a proxy for the potential to destabilize that levee. In order to apply this as a landscape-scale measure that can capture regional differences at various scales, hydrostatic force needs to be summed over the length of levees. The potential for levee failure on an island, or group of islands, is therefore a function of the magnitude of subsidence and the length of levee that the hydrostatic forces are acting on. Although not precisely recording the processes that cause levee failures at the local scale, we suggest that *cumulative hydrostatic force* provides a useful landscape-scale measure of levee failure potential in the Delta.

ACCOMMODATION SPACE AND LEVEE FORCE INDICES

To evaluate historic, current and projected landscape changes in the Delta, we developed two indices: the Accommodation Space Index, an index that captures the consequence of island subsidence and flooding, and the Levee Force Index, an index that is a proxy for the potential for levee failure and island flooding.

For any given time the Accommodation Space Index (ASI) is calculated as:

$$ASI = (A_s + A_a)/(A_s) \qquad (1)$$

where A_s = subaqueous accommodation space, or the volume of the Delta that is filled with water and lies below mean sea level, and A_a = anthropogenic accommodation space, or the subaerial volume of the Delta that lies below mean sea level. Up until the late 1800s, all accommodation space that was generated by sea level rise or regional subsidence in the Delta was filled with water and sediment. Thus, the ASI in the late 1800s, prior to the construction of high levees and the initiation of widespread subsidence, was approximately 1. As discussed below, by the early 1900s island subsidence created rapid increases in anthropogenic accommodation space, dramatically increasing the ASI. This rate of increase in the ASI has been slowed somewhat by the abandonment of some islands within the Delta, such as Franks Tract and Mildred Island, since these flooded islands are counted as subaqueous accommodation space.

The Levee Force Index (LFI), a concept and method suggested by Jack Keller of the CALFED Independent Science Board, records the cumulative hydrostatic force acting on the levees of the Delta, indexed to an estimated force in 1900, immediately prior to widespread subsidence of the Delta. To simplify the calculation of this index, each levee is considered as a wall, with the difference between the average elevation of water in the channel and the average elevation of the adjacent island as the control on the magnitude of hydrostatic force. Based on this simplification, the cumulative hydrostatic force (CF) for an island is represented by

$$CF = P \times A \times L \quad (2)$$

Where P is average hydrostatic pressure on the island levee, A is area of the unit length of levee (1 m x H), and L is levee length of the island. Since

$$P = 0.5\rho gH \quad (3)$$

where ρ is the density of water, g is gravitational acceleration and H is the difference between the average channel water surface elevation and the average elevation of the island, then

$$CF = 0.5\rho gH^2L \quad (4)$$

The cumulative hydrostatic force acting on an island's levee is therefore a function of the square of the depth of subsidence in the island. In contrast to arithmetic increases in accommodation space, hydrostatic forces due to subsidence increase with the square of subsidence depth. Cumulative hydrostatic force, as defined here, captures two general processes that influence the regional stability of levees. Islands that are deeply subsided are more prone to levee failure due to greater force acting on the levees. Additionally, when coupled with deep subsidence, islands with relatively long levee lengths are more prone to levee failure because hydrostatic forces are acting over a greater levee surface, increasing the likelihood of exposing weaknesses in levee construction, maintenance and foundation.

Based on these calculations, the LFI for the Delta is

$$LFI = CF_t/CF_{1900} \quad (5)$$

where CF_t and CF_{1900} are the sum of the estimated cumulative hydrostatic force throughout the Delta at time *t* and 1900, respectively. The two islands that are filled, Mildred Island and Franks Tract, are not counted in these totals since their cumulative force is effectively zero. In addition, islands with mean elevations at or above MSL are not included in this calculation since their LFI = 0.

METHODS

For the purposes of this report, we used a simplified approach for reconstructing historic and projected changes in the ASI and LFI. An elevation model of the Delta was constructed from the Shuttle Radar Topography Mission (SRTM) data obtained from the Global Land Cover Facility (USGS 2004). This dataset was collected in February 2000 at approximately 1:100,000 scale, with reported +/-1 meter vertical resolution and 1 arc-second/30-meter horizontal resolution. Delta island maps were acquired from the Research Program in Environmental Planning and GIS (REGIS), at the University of California, Berkeley, http://www.regis.berkeley.edu/, which digitized the island-forming levees from the DWR Delta Atlas and USGS maps. Zonal statistics for each island were then used to calculate mean island elevations in the year 2000. Based on area/elevation relationships, the average elevation and accommodation space was estimated for each island in year 2000.

It is important to note that the resolution of the SRTM data within the Delta has not been established. Efforts at the Global Land Cover Facility are testing the resolution of SRTM data. We conducted a first-order assessment of the SRTM data through comparison with multiple data sources. Recent, unpublished surveys have been performed on Bacon Island by private consultants (personal communication, Delta Wetlands, December 2004). These surveys re-established historic transects across the island and were used to calculate average elevation losses due to subsidence. Based on these surveys, conducted in the summer of 2000, the average elevation of the island was estimated to be -5.06 m; calculated mean elevation based on SRTM data is -4.82 m. Given the different methods used to estimate average elevation (transect versus zonal statistics) these results are surprisingly comparable. In addition, we compared SRTM data with local high-resolution LIDAR surveys supplied to us by DWR. These surveys covered Staten Island and McCormick-Williamson Tract in the north Delta (flown in February/March 2002). For all datasets we used zonal statistics to calculate average island elevation. The mean difference in average elevation between LIDAR and SRTM data is +0.31 m, with a maximum difference of +0.49 m on Staten Island and a minimum difference of +0.13 m on McCormack-Williamson Tract. This cursory analysis of SRTM data indicates that areal averaging of elevations on islands provides a reasonable method for estimating accommodation space and total subsidence.

To derive the time-averaged subsidence, we made the assumption that the average elevation of the interior of Delta islands prior to reclamation was approximately current mean sea level (MSL). This is based on the distribution of topographic features, including tidal channels and tule marsh, which make up the marsh platform, and the limited change in sea level over the past century. Based on this information, we calculated an average annual subsidence rate for each island for the period 1900-2000. Because detailed information about individual islands is relatively sparse, the year 1900 was chosen as an average year for the initiation of subsidence throughout the Delta, recognizing that subsidence may have begun as early as 1880 on some islands (e.g. Jersey Island) and as late as 1930 on some smaller islands (Thompson 1957).

Rojstaczer and Deverel (1993, 1995), Deverel and Rojstaczer (1996), Deverel et al. (1998) and Deverel (1998) conducted detailed studies of the rates of subsidence on several Delta islands. Based on field experiments and analysis of historic survey data, they suggest that rates of subsidence have been declining since the 1950s due to improved land use practices and decreasing organic content of island soils. For this reason, projecting average 1900-2000 subsidence rates into the future will result in significant overestimation of future subsidence. To address this issue, we reanalyzed elevational data summarized by Deverel et al. (1998) for Mildred Island, Bacon Island and Lower Jones Tract. Survey transects on these islands were reoccupied 18 times between 1925 and 1981, with average island depth estimated for each survey. We used linear regression analysis to establish average subsidence rates for each island during the survey period. To estimate the decline in subsidence rates associated with better land use practices, we regressed post-1950 island elevations separately (Figure 3). The post-1950 subsidence rates range from 20% to 40% less than the averaged rate of subsidence for the period 1925-1981. To simulate subsidence of Delta islands from 2000-2050, we applied the more conservative rate of 40% reduction in subsidence rates to the calculated 1900-2000 subsidence rates based on the SRTM data.

Future subsidence in the Delta is constrained by the thickness of organic-rich sediments, deposited since the mid-Holocene. Using 500 m grid point data provided by DWR, spline interpolation was used to derive a surface representing the base of the organic-rich sediments. Subsequently, we were able to use this surface in conjunction with subsiding land surface elevations to calculate depth to the base of the peat layer through time. Average interior island subsidence and anthropogenic accommodation space were simulated in annual time steps. Annual subsidence at 40% less than the 1900-2000 average for each island was held constant for each time step until depth of subsidence equaled the depth of organic-rich soils, at which point subsidence ceased for the remaining time steps.

Subaqueous accommodation space and average channel depth were calculated from bathymetry maps supplied by the California Department of Fish and Game (DFG 2004) using ArcGIS 3D Analyst. With the exception of space added by flooding of Franks Tract and Mildred Island, subaqueous accommodation space was assumed to be constant since the late 1800s. This volume may overestimate the subaqueous accommodation space during the late 1800s and early 1900s, since channel dredging and re-alignment may have increased the total channel volume. With local exceptions, channel depth is typically greater than the elevation difference between the water surface and the average elevation of the subsided island.

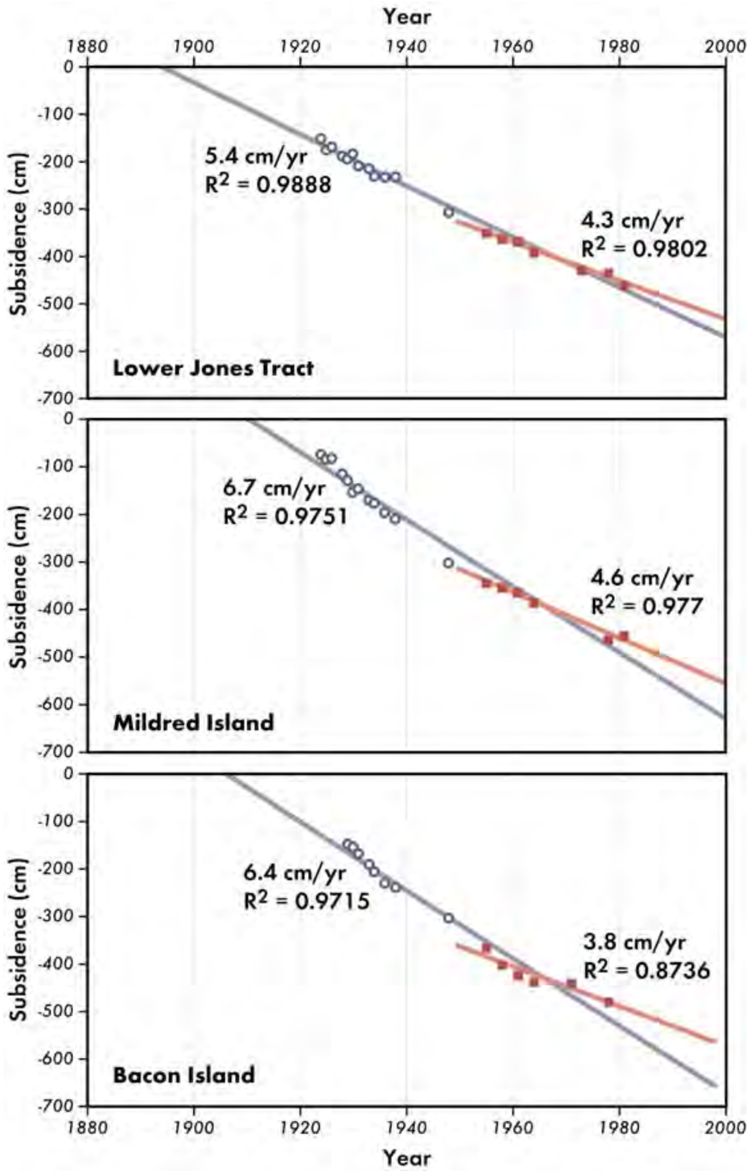


Figure 3. Linear regression of elevation data from three Delta islands to assess changes in rates of subsidence. Blue line depicts best fit for subsidence data from 1925-1981: red line represents post-1950 data. See text for discussion. Data from Deverel (1998; personal communication, S. Deverel, 2004).

Since accommodation space and difference in elevation between the channel and the island is a function of subsidence and sea level change, we adjusted our simulations for sea level rise over the period 2001-2050. Eustatic sea level rise in the latter parts of the 20th century and the present is being driven by a combination of thermal expansion of the oceans due to global warming and increases in ocean mass associated with melting of continental ice. A recent discussion (Miller and Douglas 2004) notes significant disparity among current estimates of sea level rise. Most estimates range from 1.5 to 2.0 mm/yr, based on analysis of historic gage and dynamic ocean height data, to approximately 2.5 mm/yr based on satellite altimetric estimates from the 1990s. We used an average of the range of reported sea level rise values of 2 mm/yr for this study. Modeling efforts summarized by the IPCC (2001) indicate variable rates of projected sea level rise, ranging from as little as 1 mm/year to as much as 5.1 mm/yr by 2050. For the purposes of this simulation, we assumed a conservative linear increase in sea level rise from 2 mm/yr in 2001 to 3 mm/yr in 2050. This reflects an approximate average of six different global climate models (IPCC 2001) and may underestimate total sea level rise.

The results of this modeling effort are summarized in the maps shown in Figure 4, depicting the current elevations within the Delta and simulated elevations in 2050. The 2050 map elevations reflect a systematic lowering of relative inner island elevations by an average rate of subsidence and an increase in sea level.

This simplified approach to estimation of the ASI and LFI makes multiple assumptions that should be taken into account in interpreting the results of this study. First, projections to 2050 assume business-as-usual approaches to management of the Delta. That is, Delta islands will continue to be farmed using current best management practices and levees will continue to be maintained in their current configuration.

Second, this approach does not accurately model anticipated asymptotic declines in rates of subsidence that should occur as the inorganic fraction of some island soils increases over time. For that reason, the estimates of accommodation space given here should be viewed as conservative maxima. However, it is important to note that if farming continues to be the dominant land use in the Delta, subsidence will continue and accommodation space will increase. There is no known or anticipated technologically feasible method to eliminate or reverse subsidence in land that is being farmed. As the regression analyses of subsidence data from Bacon and Mildred islands and Jones Tract show, improved land use practices have only slowed subsidence rates by 40% or less (Figure 3). Additionally, the impact of increased concentration of inorganic content of the soils appears to only impact subsidence once the organic-matter content of the soils is less than 20% (Deverel 1998). In many central and west Delta islands the organic matter content of the soils is unlikely to reach concentrations below 20% during the next 50 years.

Finally, it is important to note that the methods used here cannot resolve local-scale complexities of historic or projected subsidence in the Delta. Detailed studies by Rojstaczer and Deverel (1995) and Deverel and Rojstaczer (1996), showed order-of-magnitude variation in subsidence within individual islands. Areas near the margins of the islands tend to be organic-poor, recording the influence of natural levee deposition prior to reclamation. Conversely, the center of the islands, which were covered by marsh plain and were most isolated from channel influences, tend to be most organic rich. Differential rates of subsidence occur on every island, with generally less subsidence near the margins and higher

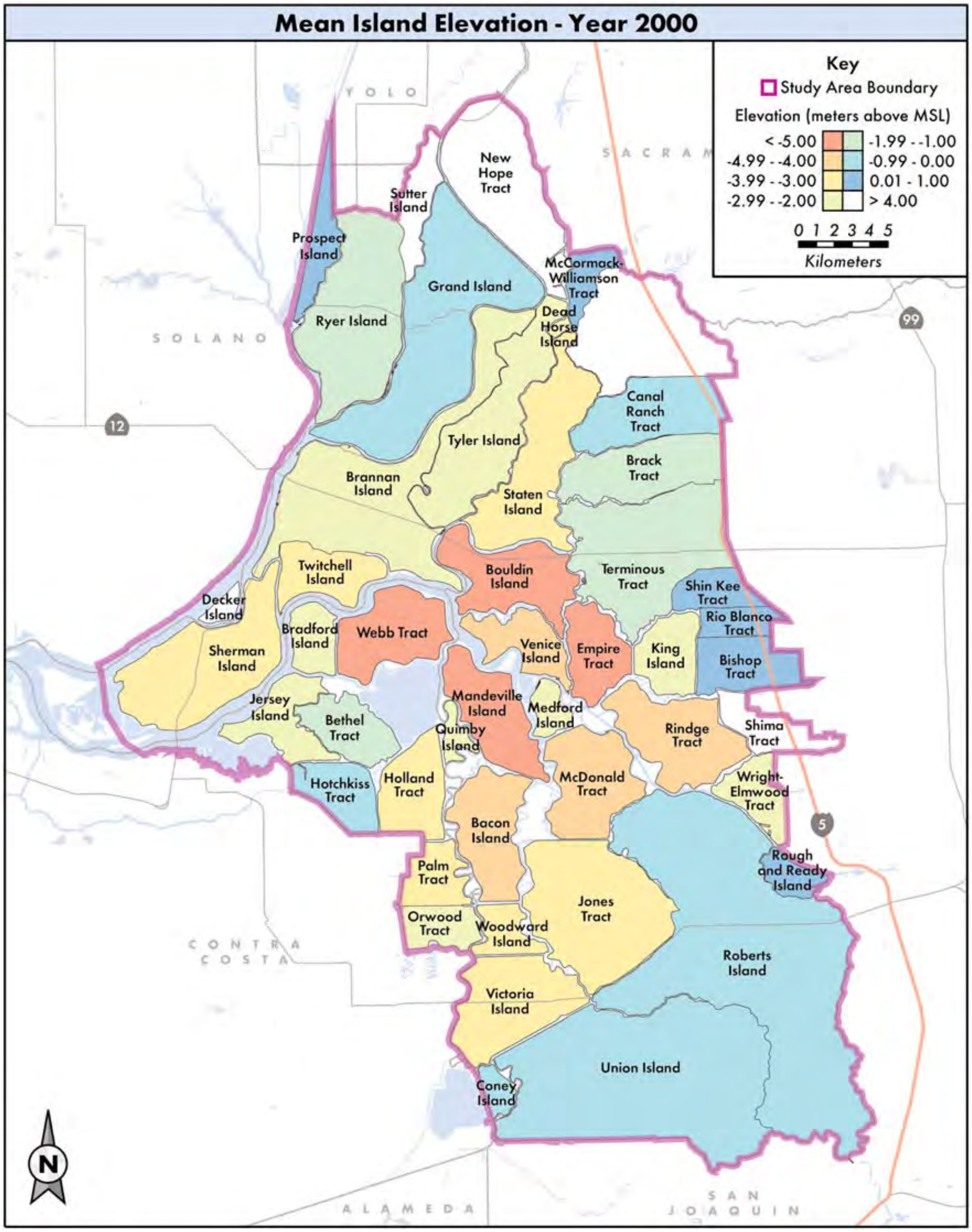


Figure 4A. Calculated average island elevations for 2000. Methods described in text.

subsidence near the center. Acknowledging the limits of resolution of SRTM data described above, the approach taken here averages subsidence for the entire island and should not be used to interpret processes within a specific island. This approach may also overstate the cumulative levee force on some islands since the LFI is based on the average elevation, rather than elevations immediately adjacent to the levee.

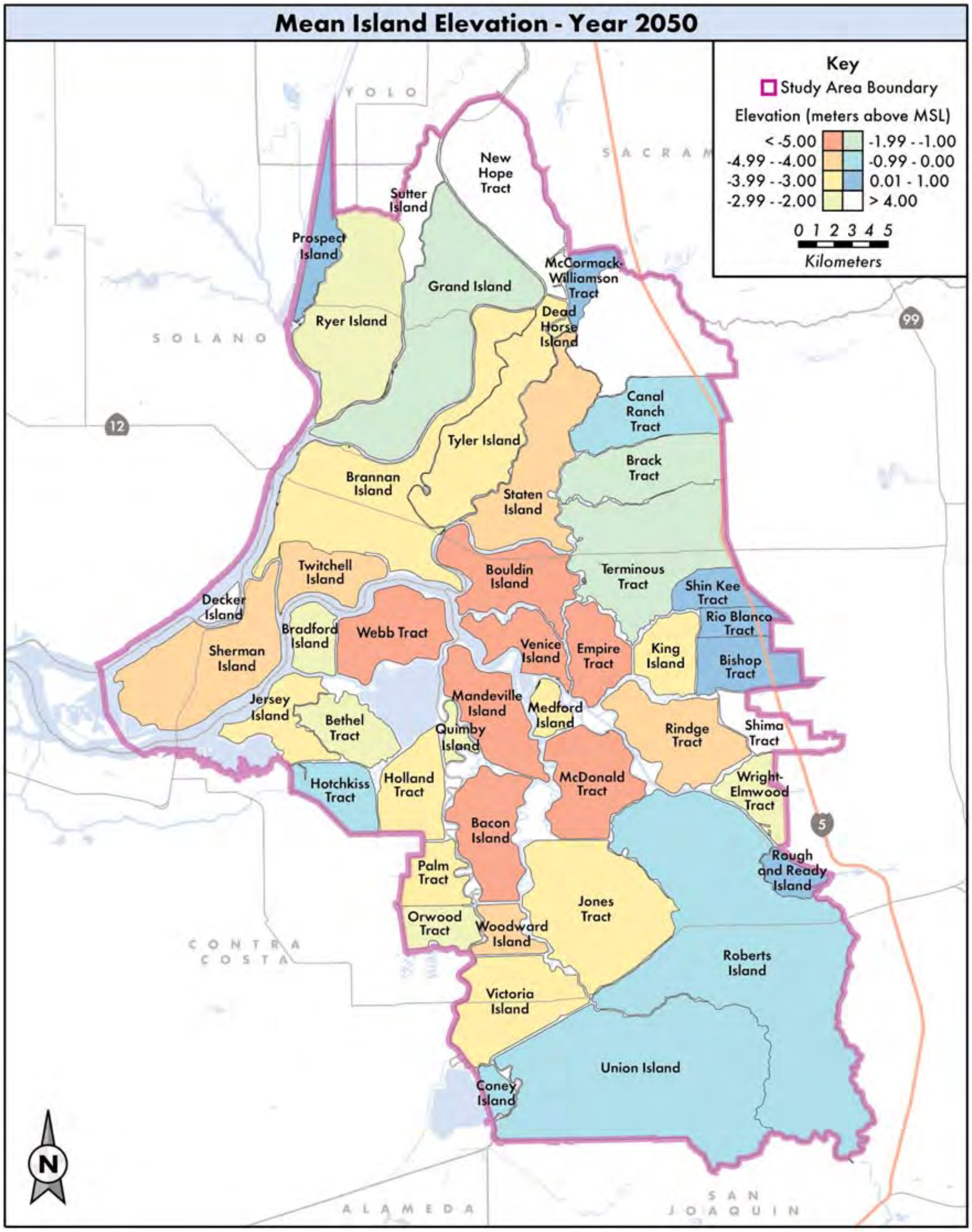


Figure 4B. Simulated elevations for 2050. Methods described in text.

RESULTS

Wherever there are organic-rich soils in the Delta that have been farmed, there has been significant subsidence and the formation of anthropogenic accommodation space. The magnitude of anthropogenic accommodation space generation varies in space and time (Figure 5A). As noted above, rates of subsidence are a function of organic content of the soils and land use practices. The organic-rich soils of the central and west Delta, for example, exhibit the highest historic average rates of subsidence, 3.2 and 4.8 cm/yr respectively. More than half the total 2.5 billion cubic meters of anthropogenic accommodation space formed during the past century occurs in the central and west Delta. Simulations of future accommodation space generation also reflect the distribution and thickness of organic-rich soils. In the east and south Delta, historic subsidence has reduced or eliminated the organic-rich soils. In these areas, anthropogenic accommodation space formation will be dominated by the effects of eustatic sea level rise, rather than continued subsidence. In contrast, the central and west Delta, which contains thick organic-rich soils, will continue to subside. Although the north Delta retains the thickest organic-rich soils of the Delta, the lower subsidence rate reflects the lower total organic content.

Similar to changes in anthropogenic accommodation space, historic and future cumulative levee force varies substantially in the Delta (Figure 5B). The lowest cumulative levee forces are in the east Delta, where relatively high island elevations and correspondingly smaller levees predominate. The Central Delta dominates cumulative levee force, approximately equaling all other regions of the Delta combined. The disproportionate cumulative levee force of the Central Delta is a function of both the high regional rates of subsidence and the large levee lengths relative to total island area. Unlike anthropogenic accommodation space, future cumulative levee force in the central, west and north Delta increases substantially in the period 2000-2050.

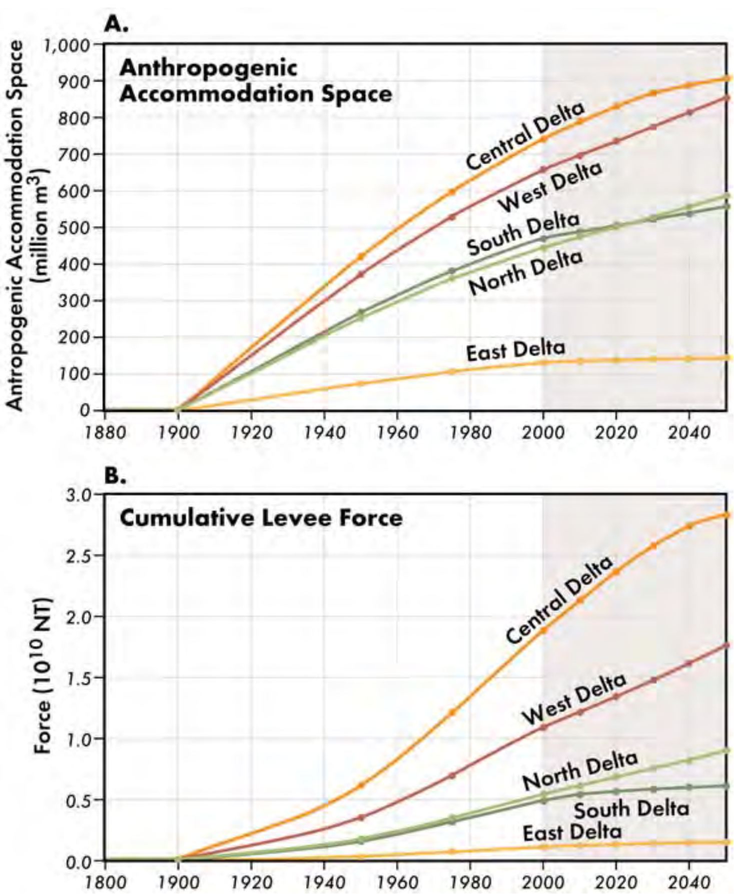


Figure 5. Calculated and simulated Anthropogenic Accommodation Space and Cumulative Hydrostatic Force for regions of the Delta shown in Figure 1.

To establish anthropogenic accommodation space and cumulative levee force for the 1950 and 1975 data points we adjusted individual island subsidence rates for the periods 1900-1950 and 1951-1975 based on an average of relative rate changes noted on Lower Jones Tract and Mildred and Bacon islands, as shown in Figure 3.

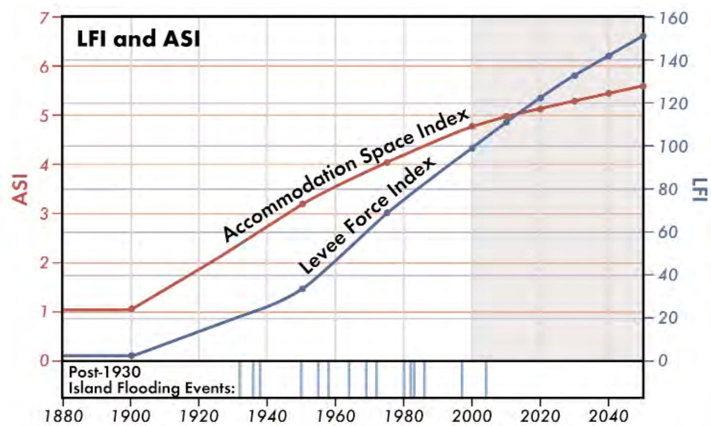


Figure 6. Accommodation Space Index (ASI) and Levee Force Index (LFI) for the subsided portion of the Delta. See text for discussion.

The ASI and the LFI for the Delta are depicted in Figure 6. These indices provide a landscape-scale proxy for current and future consequence of levee failure in the Delta (ASI) and the relative risk of island flooding (LFI). As noted above, these indices are dominated by the impacts of central and west Delta subsidence and, in the case of the LFI, relative levee lengths. Both indices show substantial increases in the future, due to continued subsidence and sea level rise.

LANDSCAPE CHANGE IN CONTEXT

During the past 100 years, farming activity in the Delta has resulted in the loss of approximately 2.5 billion cubic meters of soil—an average of 25 million cubic meters per year. The amount of anthropogenic accommodation space generated from subsidence and sea level rise is projected to increase to more than three billion cubic meters in 2050, an annual average of approximately 10 million cubic meters per year. Sea level rise accounts for approximately 30% of the increase in the anthropogenic accommodation space during this period.

It is important to place the amount of anthropogenic accommodation space into historic perspective. The volume of organic-rich sediment that accumulated within the Delta during the mid- to late Holocene can be approximated by summing the volume of anthropogenic accommodation space and the volume of organic-rich soils that underlie the islands. This underestimates the total volume because it does not account for material that underlies the current channel network. Based on this approach, we estimate that approximately 5.1 billion cubic meters of tidal marsh sediment filled accommodation space within the Delta during the past 6000 years. This represents an average annual rate of accumulation of approximately 850,000 cubic meters. During the past 100 years, oxidation, compaction, erosion and burning have reduced the volume of accumulated sediment by almost one half—an annual rate of loss almost 30 times the rate of historic accretion. Over the next 50 years rates of anthropogenic accommodation space generation will decline, but will remain more than an order of magnitude greater than historic rates of accretion, substantially increasing the forces acting on the Delta levee systems.

In his seminal study of the impacts of 19th century hydraulic mining on the Bay-Delta watershed, G.K. Gilbert (1917) estimated that mining introduced 1.2 billion cubic meters of sediment into the Sacramento River system. As noted above, when the hydraulic mining sediment waves entered the Delta in the late 1800s, there was little accommodation space and the material by-passed the Delta. The volume of sediment created by hydraulic mining, considered one of the most destructive land use practices in the history of the Bay-Delta watershed (Mount 1995), is less than half of the volume of accommodation space created by subsidence to date, and approximately one-third of the projected total volume in 2050.

Alternatively, levee and dam construction throughout the Bay-Delta watershed limits the current sediment inputs into the Delta. Wright and Schoellhamer (2004) estimate that approximately 6.6 million metric tons of sediment enter the Delta annually, with 2.2 million metric tons leaving the Delta and 4.4 metric tons deposited within the Delta. Assuming a bulk density of 850 kg/m^3, annual deposition in the Delta is approximately 1.7 million cubic meters. This volume is less than 7% of the rate of historic anthropogenic accommodation space generation and only 17% of future rates. If sea level remained unchanged, subsidence in the Delta were stopped, and current rates of inorganic deposition in the Delta were maintained, it would take 1470 years to restore elevations to mean sea level. However, projected annual accommodation space created by sea level rise alone is roughly twice the amount that could be filled by inorganic sedimentation.

The goal of these comparisons is to illustrate that subsidence and associated anthropogenic accommodation space generation is the dominant landscape-forming process in the Delta during the past 100 years and will remain so for the indefinite future. All CALFED programs that relate to the Delta are being affected in some manner by this process, yet, with the exception of the Levee System Integrity Program (CALFED 2004), no programs appear to fully recognize the potential impacts and implications.

PUNCTUATED LANDSCAPE CHANGE

The above discussion illustrates that the landscapes of the Delta are dynamic, with change occurring incrementally. However, change in the Delta is not limited to gradual shifts. Punctuated, or sudden landscape change has a high probability of occurring within the Delta during the period simulated here, posing a considerable policy challenge for the CBDA and its member agencies. Punctuated change can be derived from two sources: seismicity and extreme flood events.

The levees of the Delta are at significant risk of failure due to seismicity. This stems from poor foundation soils prone to settling or liquefaction, or poor-quality engineering and construction materials (DWR 1995). Although there have been no significant quakes in or closely adjacent to the Delta since high levees were originally constructed, there are at least five major faults within the vicinity of the Delta capable of generating peak ground acceleration values that would likely lead to levee failures. A preliminary analysis of the risk of levee failure due to seismicity was prepared for the CALFED Levee System Integrity Program (Torres et al. 2000). Based on standard methods and local expertise, Torres et al. (2000) estimated the magnitude and recurrence intervals of peak ground accelerations throughout the Delta. Two competing fault models were evaluated for this study, producing a wide range of potential accelerations. Then, based on local knowledge and limited geotechnical information, Damage Potential Zones were established for the Delta (Figure 7). The zones of highest risk lie in the central and west Delta where tall levees are constructed on unstable soils that are at high risk of settling or liquefaction during an earthquake. This also coincides with areas of the Delta that have the highest cumulative hydrostatic force and anthropogenic accommodation space.

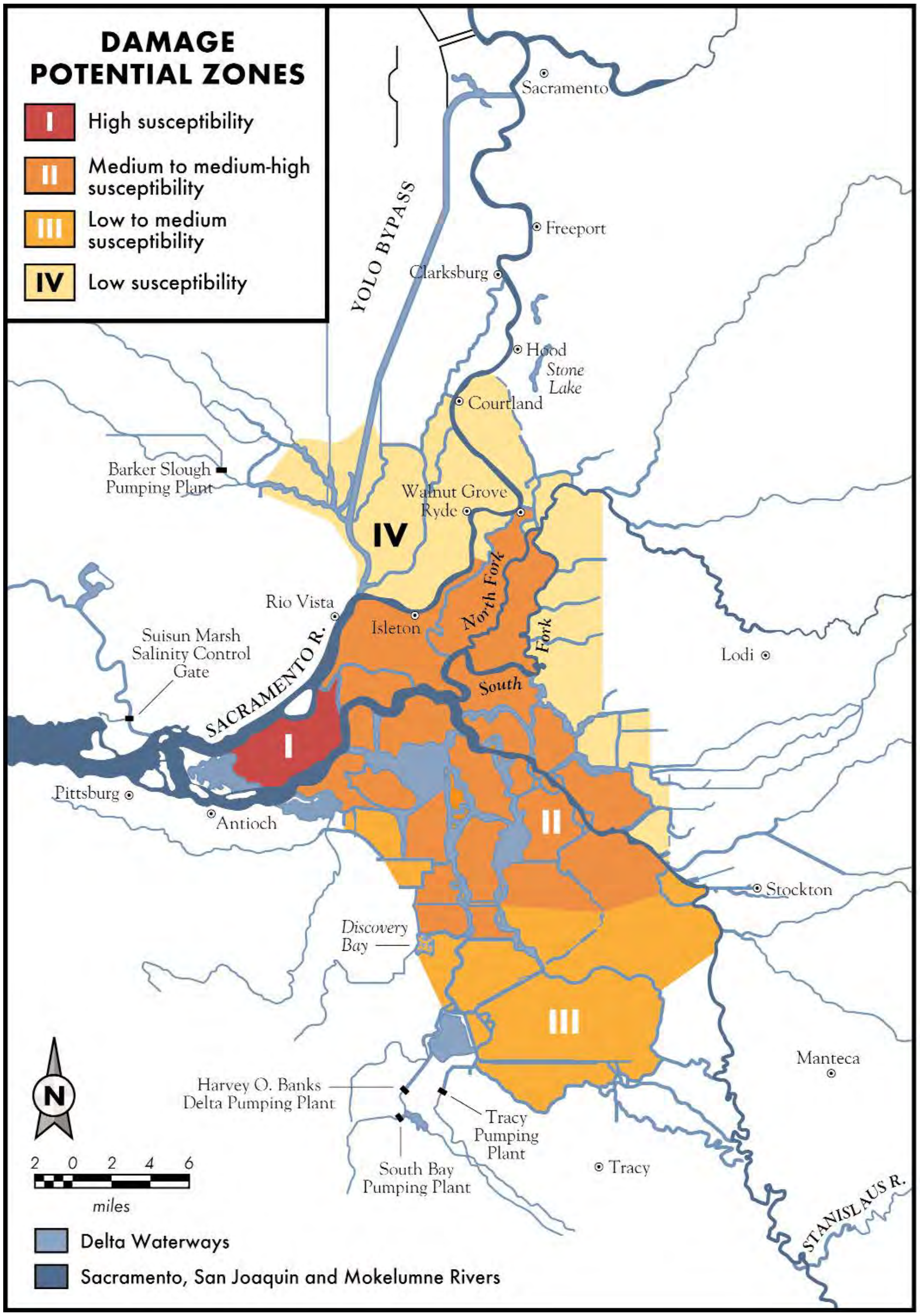


Figure 7. Zones of varying potential damage due to seismically-induced liquefaction and levee collapse. Modified from Torres et al. (2000).

Torres et al. (2000) estimated recurrence intervals for ground accelerations and the number of potential levee failures in each Damage Potential Zone. It is useful to examine their estimates of the number of failures that might occur during a 100-year event, or an event with a 0.01 probability of being equaled or exceeded in any given year (Figure 8). As in any probabilistic analysis of this sort, the range of potential responses to this kind of earthquake are broad and difficult to predict with precision. Based on their estimates, it is a roughly 50-50 chance that 5 to 20 levee segments (equal to one standard deviation around a mean of seven) will fail during a 100-year event in the Delta. This does not imply that 5 to 20 islands will flood, but just that 5 to 20 levee segments will fail. The loss of 5 to 20 levee segments in the Delta con-

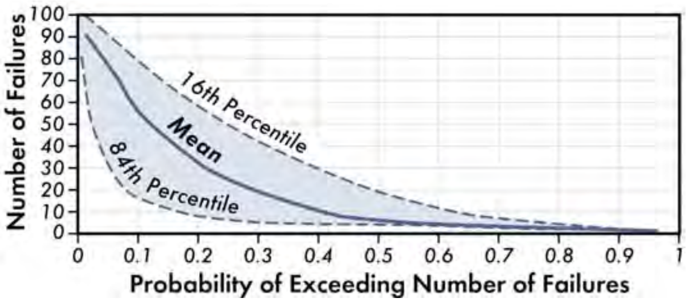


Figure 8. Figure 8. Probabilities of number of levee failures expected in 100-year recurrence interval event impacting Delta. Modified from Torres (2000).

stitutes considerable and abrupt landscape change, since island flooding is likely to be widespread and, as discussed below, persistent for a long period of time.

The high likelihood of abrupt change during seismic events is compounded by the potential for change during and immediately following major winter runoff events. Following the 1986 flood event, the State legislature developed target elevations and cross sections for levees throughout the Delta. Under Senate Bill 34, the State established the Subventions Program to support maintenance and levee upgrades. Under this program, the elevation of the levee crowns were to be upgraded to one foot above the U.S. Army Corps of Engineers' estimated 100-year flood stage (DWR 1995). Although this target elevation is tied to the 100-year flood stage, it does not imply that there is 100-year flood protection for Delta levees. There is insufficient freeboard or levee cross section to withstand sustained flows of this stage. The National Flood Insurance Program maps of the Delta reflect this vulnerability, indicating that all the major islands have less than 100-year flood protection. It is reasonable to assume, therefore, that a flood of 100-year recurrence interval will produce substantial, widespread, and as discussed below, possibly permanent flooding of islands in the Delta comparable to that associated with seismic events.

The risk of abrupt change in the Delta during the 50-year simulation period can be evaluated probabilistically using standard methods (review in Mount 1995). In any year, the probability that a flood with a 100-year recurrence interval will occur is 0.01. However, the probability that such a 100-year event will occur sometime in the next 50 years is 0.40, or a two-in-five chance. Since either a 100-year flood or 100-year seismic event can produce significant change in the Delta, it is more appropriate to estimate the probability that either event would occur in the 50-year time interval. When evaluated this way, the odds of either event occurring is 0.64: a roughly two-in-three chance. This discussion is meant to highlight the fact that punctuated landscape change in the Delta is not a remote, hypothetical possibility, but is highly likely during the simulated period of 50 years. This is especially pertinent to the risk of seismicity where continued accumulation of strain on local fault zones may increase the risk of an earthquake with time.

DISCUSSION: FUTURE TENDENCIES

The approach used here to assess historic and projected changes in the Delta does not offer the resolution necessary for island-by-island assessments or prediction of future levee failure. Thus, this paper is not intended to be used as a planning tool. Rather, this approach offers a landscape-scale assessment of processes that are increasing the overall consequences of, and potential for island flooding in the Delta over the next 50 years. However, given the relative magnitude of increases in the ASI and LFI and the high probability of seismic or flood events that will result in levee failure, it is reasonable to assume that there will be an increasing tendency for island flooding events, with the consequences of any flooding event also increasing.

Local island flooding events are a relatively common occurrence in the Delta (Figure 5). Since the 1930s there have been more than 15 such flooding events (DWR 1995). Several State and federal programs, including the Subventions and Special Projects Programs (DWR) and the Base Level Protection and Special Improvements Programs (ACOE) have improved maintenance of many private levees within the Delta and have upgraded multiple at-risk levee segments. Although improvements have been made within the Delta and reduced the risk of flooding, the current level of risk is largely unknown. Levee programs are focused principally on maintaining current levels of protection, set in 1986, rather than assessing and planning for future conditions. The Levee System Integrity Program Plan (CALFED 2000) notes that

885 km of levees will require upgrading to meet Federal PL 84-99 standards at a cost of more than \$1 billion in today's dollars. Recently signed federal legislation authorizing the CALFED Bay-Delta Program includes \$90 million for levee projects in the Delta for the next five years. However, this represents less than 10% of the current backlog and is unlikely to address future needs. Levee upgrades to meet existing standards typically cost \$1.0 to 1.7 million/km, with costs rising to near \$3.4 million/km where extensive reconstruction is required (DWR staff, personal communication, 2004). Given the high costs and historic trends in funding, the Delta levee system, which is already well behind in maintenance, repairs and upgrades, will continue to fall behind under future, business-as-usual landscape change scenarios.

Although maintenance and upgrade of levees represents a significant, on-going cost in the Delta, island flooding events have the potential to dramatically impact local and government resources. The June 3, 2004, flooding of Jones Tract in the south Delta created substantial costs for repair, flood fighting, emergency services, and island pumping. According to DWR staff, costs to government alone for this break exceeded \$44 million. This does not account for crop losses, job losses, farm infrastructure repair or carriage water releases to maintain water quality. Estimates of total costs of the Jones Tract failure reported in the *Sacramento Bee* and *Contra Costa Times* approach \$90 million (quoted from California Office of Emergency Services sources): a figure equal to the total amount allocated for levees in the 2004 federal authorization of CALFED.

Limited funding for levee maintenance and upgrades, high costs of emergency levee repairs, and projected increasing instability of the Delta indicate that local island flooding will impact the Delta significantly during the next 50 years. Climate change and changes in runoff conditions (which are, for the most part, beyond the scope of this report) may exacerbate these conditions. There are multiple potential policy responses to this projected trend. However, to date, there has been no comprehensive assessment of the effects of increased island flooding on CALFED programs. Rather, current policies appear to be predicated upon the unlikely prospect of maintaining fixed hydraulic conditions.

The impact of regional flooding associated with seismic events or large floods poses an additional challenge to CALFED programs. These events have the capability to significantly and permanently change conditions within the Delta over a very short period of time. To illustrate, currently there is one contractor, Dutra Corporation, with the equipment necessary for repairing levee breaks in the Delta. According to DWR staff, this contractor is capable of restoring two to three levee breaches in a single season. If regional island flooding results in numerous levee breaches, it is unlikely that levee integrity can be restored for many years, with protracted disruption of water supply and loss of farm income. Moreover, if a seismic event leads to levee failures in the Delta, it is likely to be associated with significant damage to infrastructure in the San Francisco Bay Area, creating competition for resources necessary for restoring levee integrity.

To our knowledge, the California Bay-Delta Authority and its member agencies have not articulated a policy regarding regional flooding in the Delta and the possibility of permanent, abrupt change. It is important to note, however, that the Levee System Integrity Program has initiated a comprehensive, multi-year study of the risks due to seismicity in the Delta (CALFED 2003). This program, which is being run by DWR, is in its nascent stage, but will address some of the key issues raised here and provide more precision on estimates of risk.

CONCLUSIONS

The results of the simulations conducted for this report indicate that microbial oxidation and compaction of organic-rich soils in the Delta have led to significant regional subsidence in the Delta. Although slowing substantially, subsidence is likely to continue into the indefinite future, particularly in the central and west Delta. When coupled with rising sea level over the next 50 years, continued subsidence will magnify the instability of the Delta levee network, leading to increased potential for and consequence of island flooding. Additionally, there is significant likelihood of regional flooding in the Delta during the next 50 years due to earthquake-induced levee failures or sustained large floods. These events are likely to result in dramatic change in the Delta.

The implication of future Delta landscape change is, at present, largely unknown and speculative. Outside of initial efforts by the Levee System Integrity Program, there are no systematic assessments of risk to CALFED program elements. There have been efforts to assess methods of subsidence reversal in the Delta, but these have been stalled by on-going contract issues at DWR. In our view, there is no comprehensive scientific effort to address this issue and to provide the necessary information to inform policymakers.

ACKNOWLEDGMENTS

The GIS data assembly and analysis, including the subsidence modeling were conducted by Joshua Johnson of the UC Davis Information Center for the Environment. Joel Dudas of the California Department of Water Resources provided DWR's GIS data and assistance. Dr. Steve Deverel of Hydrofocus provided raw data and guidance on projecting future subsidence rates. Jack Keller of the California Bay-Delta Authority Independent Science Board developed the concept for the Levee Force Index used in this report and guided the authors in its analysis. Janice Fong of the UC Davis Department of Geology prepared the illustrations. Drs Johnnie Moore, Denise Reed and Tom Dunne provided comment on the manuscript. All errors of omission or commission are, however, entirely our own. This work was supported by the California Bay-Delta Authority Independent Science Board and the UC Davis Center for Integrated Watershed Science and Management.

REFERENCES

Atwater BF. 1982. Geologic maps of the Sacramento-San Joaquin Delta, California. Menlo Park (CA): U.S. Geological Survey MF-1401.

Atwater BF, Belknap DF. 1980. Tidal-wetland deposits of the Sacramento-San Joaquin Delta, California. In: Field ME, Bouma AH, Colburn IP, Douglas RG, Ingle JC, editors. Quaternary depositional environments of the Pacific Coast. Proceedings of the Pacific Coast Paleogeography Symposium 4. Los Angeles (CA): Society of Economic Paleontologists and Mineralogists. p 89-103.

Atwater BF, Conard SG, Dowden JN, Hedel CW, MacDonald RL, Savage W. 1979. History, landforms, and vegetation of the estuary's tidal marshes. In: Conomos TJ, Leviton AE, Berson M, editors. San Francisco Bay: the urbanized estuary. San Francisco (CA): Pacific Division, AAAS. p 347-385.

[CALFED] California Bay-Delta Program. 2000. Programmatic record of decision. Sacramento (CA): California Bay-Delta Program. 118 p.

[CALFED] California Bay-Delta Program. 2003. Levee System Integrity Program: multi-year program plan (years 4-7). Sacramento (CA): California Bay-Delta Program. 13 p.

[CALFED] California Bay-Delta Program. 2004. Ecosystem Restoration Program: multi-year program plan (years 5-8). Sacramento (CA): California Bay-Delta Program. 29 p.

[CDFG] California Department of Fish and Game. 2004. Combined bathymetry of San Francisco Bay and the San Joaquin Delta, based on source data from California Department of Water Resources and the U.S. Geological Survey. Sacramento (CA): California Department of Fish and Game.

Deverel SJ. 1998. Subsidence reversal in the Sacramento-San Joaquin Delta. Report to CALFED Bay-Delta Program. 44 p.

Deverel SJ, Rojstaczer S. 1996. Subsidence of agricultural lands in the Sacramento-San Joaquin Delta, California: role of aqueous and gaseous carbon fluxes. Water Resources Research 32:2359-2367.

Deverel SJ, Wang B, Rojstaczer SA. 1998. Subsidence of organic soils, Sacramento-San Joaquin Delta. In: Borchers JW, editor. Land subsidence case studies and current research. Proceedings of the Joseph Poland Subsidence Symposium. Sudbury (MA): Association of Engineering Geologists. p 489-502.

[DWR] California Department of Water Resources. 1986. Delta subsidence investigation progress report. p 1-53.

[DWR] California Department of Water Resources. 1989. Delta subsidence investigation progress report. p 1-6.

[DWR] California Department of Water Resources. 1995. Sacramento-San Joaquin Delta atlas. Sacramento (CA): California Department of Water Resources. 121 p.

[DWR] California Department of Water Resources et al. 2002. Demonstration of techniques for reversing the effects of subsidence in the Sacramento-San Joaquin Delta, CALFED Ecosystem Restoration Program (ERP-98-C01). Draft annual report.

Emery D, Myers K, editors. 1996. Sequence stratigraphy. Cambridge (MA): Blackwell Science. 297 p.

Enright C. 2004. Levee integrity and subsidence: tied at the hip for the future of the Delta [abstract]. Presented at the 2004 CALFED Science Conference. Available at: http://cain.nbii.gov/regional/calfed/calfedabstracts/.

Foote R, Sisson R. 1992. Threatened levees on Sherman Island. In: Proceedings on Stability and Performance of Slopes and Embankments. ASCE, Geotechnical Division. p 756-774.

Gilbert GK. 1917. Hydraulic mining in the Sierra Nevada. U.S. Geological Survey Professional Paper 105. Washington, DC: Government Printing Office.

Ingebritsen SE, Ikehara ME, Galloway DL, Jones DR. 2000. Delta subsidence in California: the sinking heart of the state. U.S. Geological Survey FS-005-00. 4 p.

[IPCC] Intergovernmental Panel on Climate Change. 2001. Climate change 2001: the scientific basis. Houghton JT, Ding Y, Griggs DJ, Noguer M, van der Linden PJ, Dai X, Maskell K, Johnson CA, editors. Cambridge: Cambridge University Press. 881 p.

Jaffe BE, Smith RE, Torresan L. 1998. Sedimentation and bathymetric change in San Pablo Bay, 1856-1983. U.S. Geological Survey Open-File Report 98-759.

Miller L, Douglas BC. 2004. Mass and volume contributions to global sea level rise. Nature 428:406-409.

Mount JF. 1995. California rivers and streams: the conflict between fluvial process and land use. Berkeley (CA): University of California Press. 359 p.

[NHI] Natural Heritage Institute. 1998. An environmentally optimal alternative for the Bay-Delta: a response to the CALFED Program. Available at: http://www.n-h-i.org/Publications/Publications.html

Orr M, Crooks S, Williams PB. 2003. Will restored tidal marshes be sustainable? In: Brown LR, editor. Issues in San Francisco Estuary tidal wetlands restoration. San Francisco Estuary and Watershed Science. Vol. 1, Issue 1 (October 2003), Article 5. http://repositories.cdlib.org/jmie/sfews/vol1/iss1/art5/

Pethick JS. 1996. The geomorphology of mudflats. In: Nordstrom KF, Roman CT, editors. Estuarine shores: evolution, environment and human health. Chichester (UK): John Wiley. p 185-211.

Pethick JS, Crook S. 2000. Development of a coastal vulnerability index: a geomorphological perspective. Environmental Conservation 27:359-367.

Prokopovitch NP. 1985. Subsidence of peat in California and Florida: Bulletin Association of Engineering Geologists 22:395-420.

Reed DJ. 2002a. Sea-level rise and coastal marsh sustainability: geological and ecological factors in the Mississippi delta plain. Geomorphology 48:233-243.

Reed DJ. 2002b. Understanding tidal marsh sedimentation in the Sacramento-San Joaquin Delta, California. Journal of Coastal Research Special Issue 36:605-611.

Rojstaczer S, Deverel SJ. 1993. Time dependence in atmospheric carbon inputs from drainage of organic soils: Geophysical Research Letters 20:1383-1386.

Rojstaczer S, Deverel SJ. 1995. Land subsidence in drained histosols and highly organic mineral soils of the Sacramento-San Joaquin Delta. Soil Science Society of America Journal 59:1162-1167.

Shlemon RJ, Begg EL. 1975. Late Quaternary evolution of the Sacramento-San Joaquin Delta, California. In: Suggate RP, Cressel MM, editors. Quaternary studies. Bulletin 13, The Royal Society of New Zealand. p 259-266.

Thompson J. 1957. The settlement geography of the Sacramento-San Joaquin Delta, California [Ph.D. dissertation]. Available from Stanford University.

Torres RA, et al. 2000. Seismic vulnerability of the Sacramento-San Joaquin Delta levees. Report of levees and channels technical team, seismic vulnerability sub-team to CALFED Bay-Delta Program. 30 p.

[USGS] U.S. Geological Survey. 2004. 1 arc second SRTM elevation, reprocessed to GeoTIFF. College Park (MD): The Global Land Cover Facility. Version 1.0.

Wright SA, Schoellhamer DH. 2004. Trends in the sediment yield of the Sacramento River, California, 1957-2001. San Francisco Estuary and Watershed Science. Vol. 2, Issue 2 (May 2004), Article 2. http://repositories.cdlib.org/jmie/sfews/vol2/iss2/art2

Projected Evolution of California's San Francisco Bay-Delta-River System in a Century of Climate Change

James E. Cloern[1]*, Noah Knowles[1], Larry R. Brown[2], Daniel Cayan[3], Michael D. Dettinger[3], Tara L. Morgan[2], David H. Schoellhamer[2], Mark T. Stacey[4], Mick van der Wegen[5], R. Wayne Wagner[4], Alan D. Jassby[6]

1 U.S. Geological Survey, Menlo Park, California, United States of America, 2 U.S. Geological Survey, Sacramento, California, United States of America, 3 Climate Research Division, The Scripps Institution of Oceanography, U.S. Geological Survey, La Jolla, California, United States of America, 4 Department of Civil and Environmental Engineering, University of California, Berkeley, California, United States of America, 5 UNESCO-IHE (Institute for Water Education, United Nations Educational, Scientific and Cultural Organization), Delft, the Netherlands, 6 Department of Environmental Science and Policy, University of California Davis, Davis, California, United States of America

Abstract

Background: Accumulating evidence shows that the planet is warming as a response to human emissions of greenhouse gases. Strategies of adaptation to climate change will require quantitative projections of how altered regional patterns of temperature, precipitation and sea level could cascade to provoke local impacts such as modified water supplies, increasing risks of coastal flooding, and growing challenges to sustainability of native species.

Methodology/Principal Findings: We linked a series of models to investigate responses of California's San Francisco Estuary-Watershed (SFEW) system to two contrasting scenarios of climate change. Model outputs for scenarios of fast and moderate warming are presented as 2010–2099 projections of nine indicators of changing climate, hydrology and habitat quality. Trends of these indicators measure rates of: increasing air and water temperatures, salinity and sea level; decreasing precipitation, runoff, snowmelt contribution to runoff, and suspended sediment concentrations; and increasing frequency of extreme environmental conditions such as water temperatures and sea level beyond the ranges of historical observations.

Conclusions/Significance: Most of these environmental indicators change substantially over the 21st century, and many would present challenges to natural and managed systems. Adaptations to these changes will require flexible planning to cope with growing risks to humans and the challenges of meeting demands for fresh water and sustaining native biota. Programs of ecosystem rehabilitation and biodiversity conservation in coastal landscapes will be most likely to meet their objectives if they are designed from considerations that include: (1) an integrated perspective that river-estuary systems are influenced by effects of climate change operating on both watersheds and oceans; (2) varying sensitivity among environmental indicators to the uncertainty of future climates; (3) inevitability of biological community changes as responses to cumulative effects of climate change and other drivers of habitat transformations; and (4) anticipation and adaptation to the growing probability of ecosystem regime shifts.

Citation: Cloern JE, Knowles N, Brown LR, Cayan D, Dettinger MD, et al. (2011) Projected Evolution of California's San Francisco Bay-Delta-River System in a Century of Climate Change. PLoS ONE 6(9): e24465. doi:10.1371/journal.pone.0024465

Editor: Zoe Finkel, Mt. Alison University, Canada

Received July 26, 2011; **Accepted** August 10, 2011; **Published** September 21, 2011

Funding: CALFED Science Program grant SCI-05-G01-84, California Energy Commission, NOAA RISA, USGS Programs of Toxic Substances Hydrology, Priority Ecosystem Science, and National Research Program of the Water Discipline. The funders had no role in study design, data collection and analysis, decision to publish, or preparation of the manuscript.

Competing Interests: The authors have declared that no competing interests exist.

* E-mail: jecloern@usgs .gov

Introduction

Planet Earth is warming at an accelerating rate. The latest assessments show the 2000s to be the third consecutive decade of record high global-average surface temperature [1], and 2010 tied with 2005 as the warmest year since records began in 1880 (http://www.ncdc.noaa.gov/sotc/global/2010/13). This warming is attributed with high probability to increasing human emissions of greenhouse gases [2]. Global warming has altered water supplies through changes in precipitation, evapotranspiration, runoff and river discharge [3]. Risks to coastal communities and infrastructure are growing as the rate of sea level rise accelerates [4] and as the intensity of tropical storms is projected to increase [5]. Surface temperatures of inland water bodies [6], rivers [7] and oceans [1] have all increased significantly. Warming of streams and rivers contributes to local species extinctions and facilitates colonization by introduced species [7]. Spring warming of temperate lakes disrupts the synchrony between zooplankton and their phytoplankton food supply [8]. Warming of the world oceans strengthens thermal stratification and has contributed to a 1% per year loss of oceanic primary production over the past century [9]. Therefore, evidence is accumulating on a global scale of strong links between climate warming and changes in availability of fresh water, risks to humans from coastal flooding and storms, and altered biological diversity and productivity of aquatic ecosystems.

Simulations with global climate models (GCMs) under a plausible range of greenhouse gas emissions scenarios all project substantial warming through the 21st century [2]. Continued warming will have important consequences for social and natural systems, but these consequences will not be felt uniformly across the planet [1,3,6]. Therefore, strategies for adaptation to climate change require quantitative projections of how altered global patterns of temperature, precipitation and sea level will cascade to regional and local scales. We illustrate here one approach for developing quantitative projections by linking models of processes computed at sequentially smaller scales, from global to regional to local.

Our study is focused on California's San Francisco Estuary-Watershed (SFEW), which includes San Francisco Bay, the Sacramento-San Joaquin Delta (Delta) and the Sacramento and San Joaquin river drainages (Fig. 1). The SFEW has social and economic significance as the source of runoff that provides drinking water to 25 million people [10] and irrigation water to a million hectares of farmland producing crops valued at $36 billion per year [11]. It also has large ecological significance because the river system is habitat for native fishes including Pacific salmon and steelhead trout. San Francisco Bay is the largest estuary on the US west coast, providing habitat for endemic species (e.g. delta smelt, salt marsh harvest mouse) and marine species supporting fisheries (e.g. English sole, Dungeness crab). Fourteen species of migratory or Delta-resident fishes are imperiled, and their population declines motivate ambitious and costly programs of environmental conservation [12] and habitat rehabilitation [13]. On the shores of this estuary, 270,000 people and $62 billion of development are at risk of flooding as sea level continues to rise [14]. Regional planning and conflicts of resource allocation in the SFEW are already great challenges. These challenges are likely to grow as the regional effects of global climate change and other changes accumulate through this century. Here we develop integrated scenarios of the future SFEW by projecting a suite of environmental responses to climate change and assessing their implications for sustainability of native biota, water supplies, and risks of coastal flooding.

Regional setting

The San Francisco Estuary-Watershed is composed of an interconnected airshed, watershed, river network, estuary and coastal ocean (Fig. 1). The 163,000-km^2 watershed is bounded by the Sierra Nevada and Cascade mountains. Regional climate is characterized by a winter wet season and summer-autumn dry season. An average of forty percent of annual runoff to the river network is produced from snowmelt [15]. Reservoirs are managed to capture this late-season runoff as a resource, while water reaching the reservoirs during the earlier rainy season is managed as a hazard and allowed to pass through the reservoirs to maintain flood control space. Runoff and reservoir outflows collect in the Sacramento and San Joaquin Rivers, which converge in the Delta (Fig. 1). Tides propagate through the Golden Gate to the Delta, and the extent of salinity intrusion into northern San Francisco Bay is determined primarily by sea level height and river inflow. California's hydrology has followed the climate-driven patterns of change observed across the western United States and attributed to human-induced warming [16]. These patterns include trends of increasing winter and spring air temperatures and lengthened growing seasons [17], decreasing contributions of snow to annual precipitation [18], and advancement of spring snowmelt by 5 to 30 days [19]. Mean sea level at the entrance to San Francisco Bay has increased about 2.2 cm decade^{-1} since the 1930s, and the frequency of extreme tides has increased 20-fold since 1915 [20].

Future climates have been evaluated for the California region, where air temperatures are projected to increase 1.5 to 4.5°C this century in a range of scenarios [21]. Projected responses to warming include further declines of snow accumulation, decreasing hydropower generation, reduced viability of many species of fruit trees, high susceptibility of alpine and subalpine forests to warming, and increasing fire frequency [22]. Global sea level rise, expected to be a close index for that in California [20], is projected to be 70–185 cm above the present-day level [23]. Climate-driven changes in the California region are therefore expected to increase risks to the sustainability of native plant and animal communities and to human health, infrastructure, water supply and food production [24]. Here, we build from these past regional assessments to investigate how the combined effects of rising sea level and hydroclimatic changes could transform California's large watershed-river-estuary-ocean system through the 21st century. Our projections suggest that climate-driven changes to the SFEW could require adaptations to an interconnected suite of responses including: a diminishing water supply, continued shifts toward wetter winters and drier summers, sea level rising to higher levels than were projected only a few years ago, salt water intrusion, reduced habitat quality for native aquatic species, and expanding envelopes of environmental variability into regimes we have not experienced. Adaptations to these responses would require integrated and flexible planning to cope with growing risks to humans and the increasingly difficult challenge of meeting demands for fresh water and sustaining native biota and their supporting ecosystem functions.

Methods

We chose to evaluate two very different scenarios selected from the GCM projections used in the IPCC Fourth Assessment Report [2]. The PCM-B1 climate scenario portrays the B1 emissions scenario (representing a future where GHG emissions are curtailed by mid-century) as modeled by the Parallel Climate Model (PCM), a model with relatively low sensitivity to GHG emissions [25]. The GFDL-A2 climate scenario represents the A2 emissions scenario (corresponding to a future of continually increasing atmospheric greenhouse gases) as modeled by the medium-sensitivity NOAA Geophysical Fluid Dynamics Laboratory (GFDL) CM2.1 model [26]. These model-emissions scenario combinations were chosen to span a wide range of possible futures with regard to amount of warming and precipitation change, providing a comparison between a projection of a warmer future with little change in precipitation (PCM-B1) and that of a much warmer and drier future (GFDL-A2).

Our approach was to use linked models, each representing a different component of the system, to propagate the effects of the climate scenarios described above through the watershed-river-estuary system. Ultimately we portrayed these effects with a series of environmental indicators representing multiple components. These indicators were developed for the current century (2010–2099) and for a baseline period, defined as 1970–1999 to capture recent historical behavior (1999 is the end year of the "historical" GCM runs—see below). For all indicators, observation-based and model-based indicators were produced for the historical period to allow for model evaluation and to provide a baseline for assessing scenario projections.

For those indicators calculated directly from GCM output (air temperature, precipitation, and sea level), "historical" GCM simulations (driven by historical GHG forcings but otherwise unconstrained by observations) from the PCM and GFDL models were used to produce "model-based" historical indicators. Since

BDCP1673

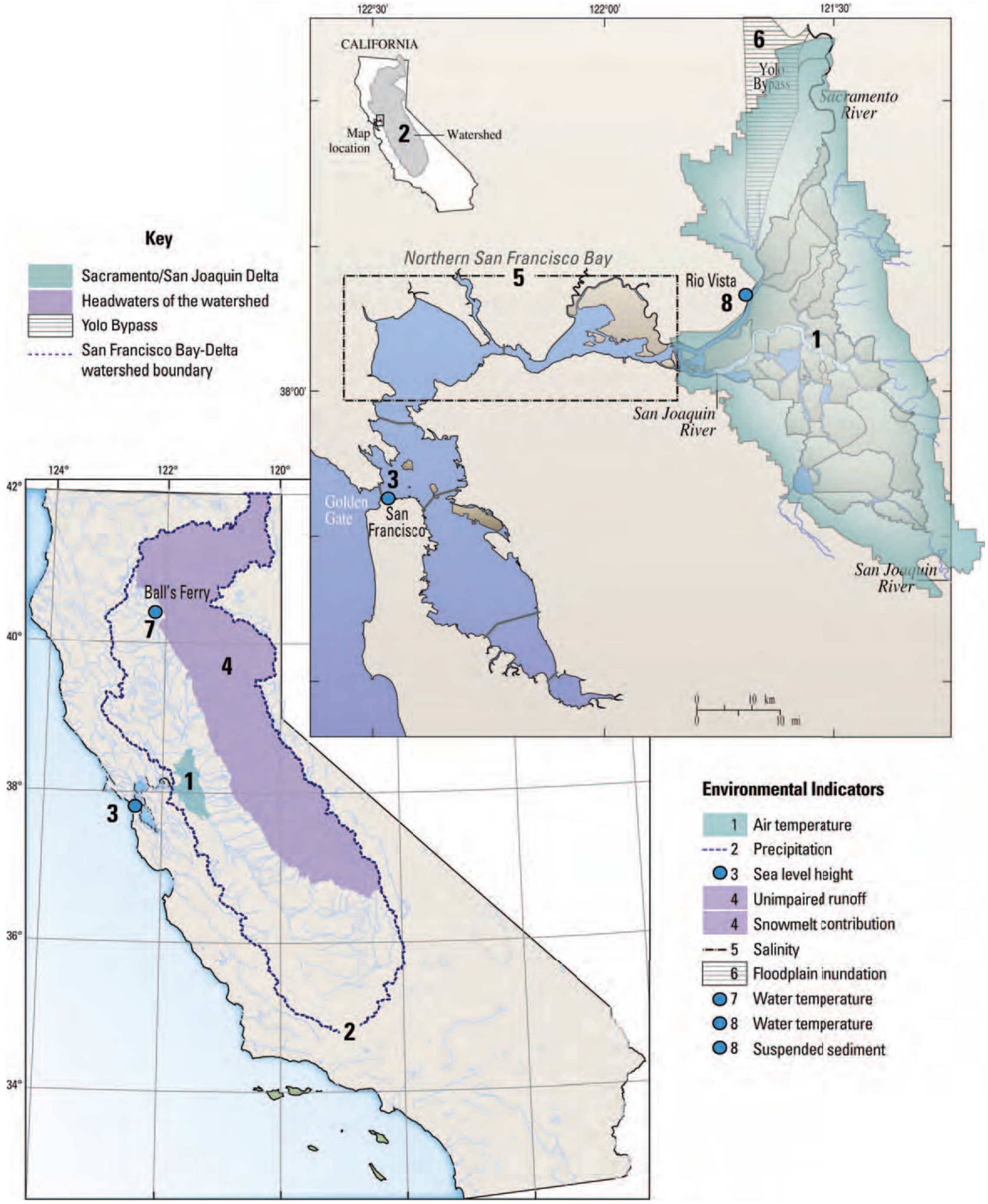


Figure 1. Spatial domains of environmental indicators. Shaded or hatched areas represent spatial domains of indicators representing areal averages or pertaining to a broad area, and blue dots represent locations of indicators corresponding to specific sites. Key shows geographic descriptions, and legend on lower-right shows corresponding indicators; compare to Figs. 2–3 and Table 1.
doi:10.1371/journal.pone.0024465.g001

the GCMs are freely running atmosphere-ocean-land models constrained only by observed GHG concentrations, these indicators will not agree on a year-to-year basis with the corresponding observation-based indicators (Fig. 2). Thus, the GCMs should be evaluated based on their statistical agreement with the observations, including model bias and variance. The model historical measures are essential to provide a baseline against which to compare the corresponding projections.

For indicators derived from the chain of models downstream from the GCMs, the model-based historical indicators are ultimately based on observed meteorological forcings, but they also reflect errors introduced by the linked models used to produce them. As such, these indicators allow for direct model evaluation by comparison with the corresponding "observation-based" time series, as well as providing a model-based baseline against which to compare the projections.

BDCP1673

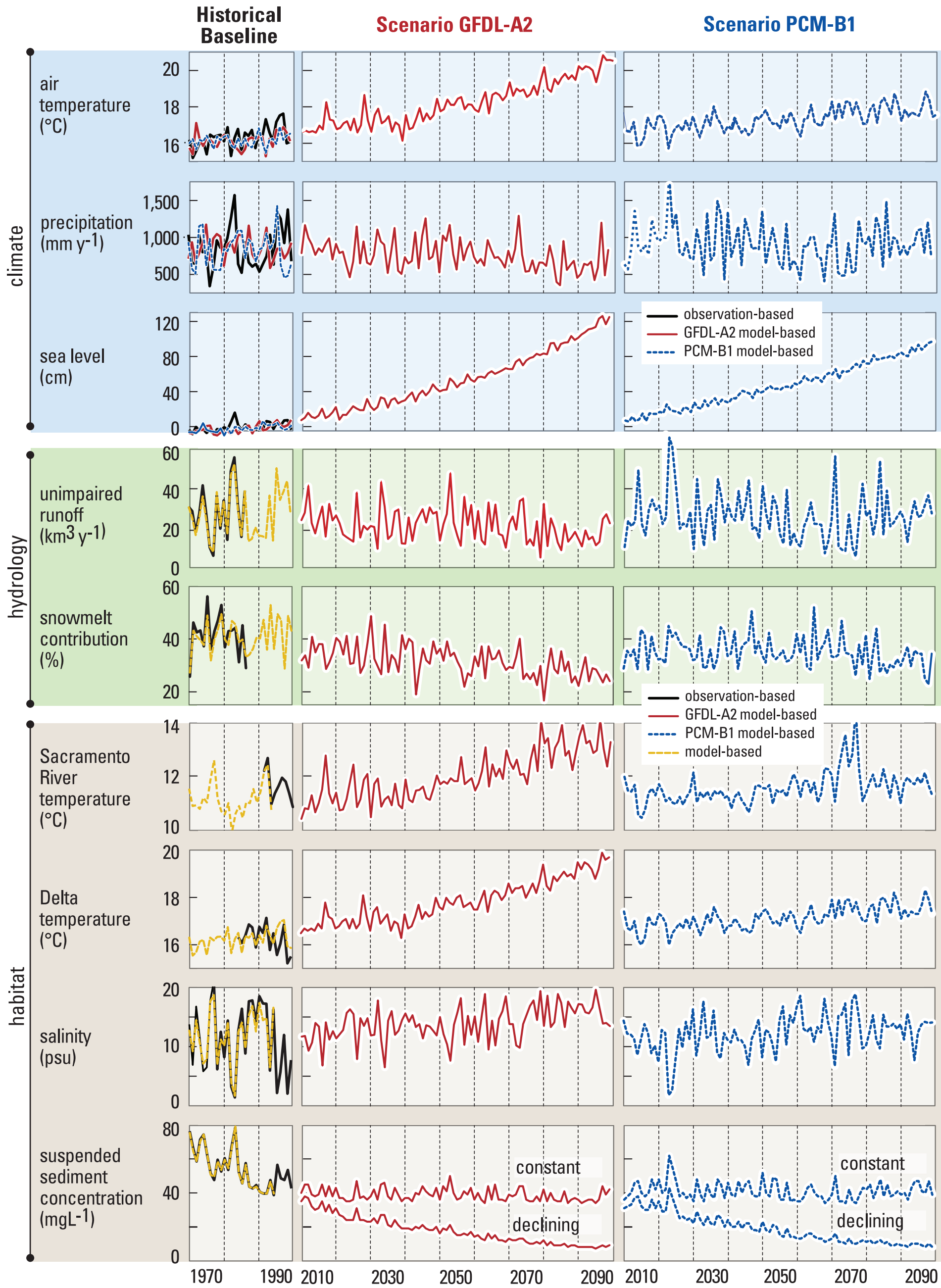


Figure 2. Projected 2010–2099 changes in annual mean values of nine environmental indicators for the A2 (red lines) and B1 (blue lines) scenarios compared to modeled and observed values during the 1970–1999 baseline period (left panels). The indicators measure changes in regional climate, regional hydrology, and habitat quality in the San Francisco Estuary-Watershed system. The GFDL-A2 and PCM-B1 "historical" data represent simulated realizations of possible climates constrained only by historical GHG forcing, and thus are not expected to track observed historical variability on a year-to-year basis.
doi:10.1371/journal.pone.0024465.g002

The trend slope for each indicator time series (Fig. 3) was calculated using the approach of Theil [27] and Sen [28]. Trend significance was determined using the modified Mann-Kendall approach of Yue and Pilon [29] which corrects for serial correlation. The confidence interval on the trend was calculated using the method described by Sen [28].

Descriptions of the individual component methods follow. An expanded methods section is in Supporting Information (Methods S1).

Meteorology

Daily values of the climate variables for the GFDL-A2 and PCM-B1 climate scenarios, and for historical PCM and GFDL model runs (forced using historical GHG concentrations) were obtained from the Program for Climate Diagnosis and Intercomparison at the Lawrence Livermore National Laboratory ([30]; www-pcmdi.llnl.gov). The GCM simulations were made on global grids with about 2 to 3° latitude and longitude resolution (about 250 km at the latitude of the Delta), and thus the original GCM

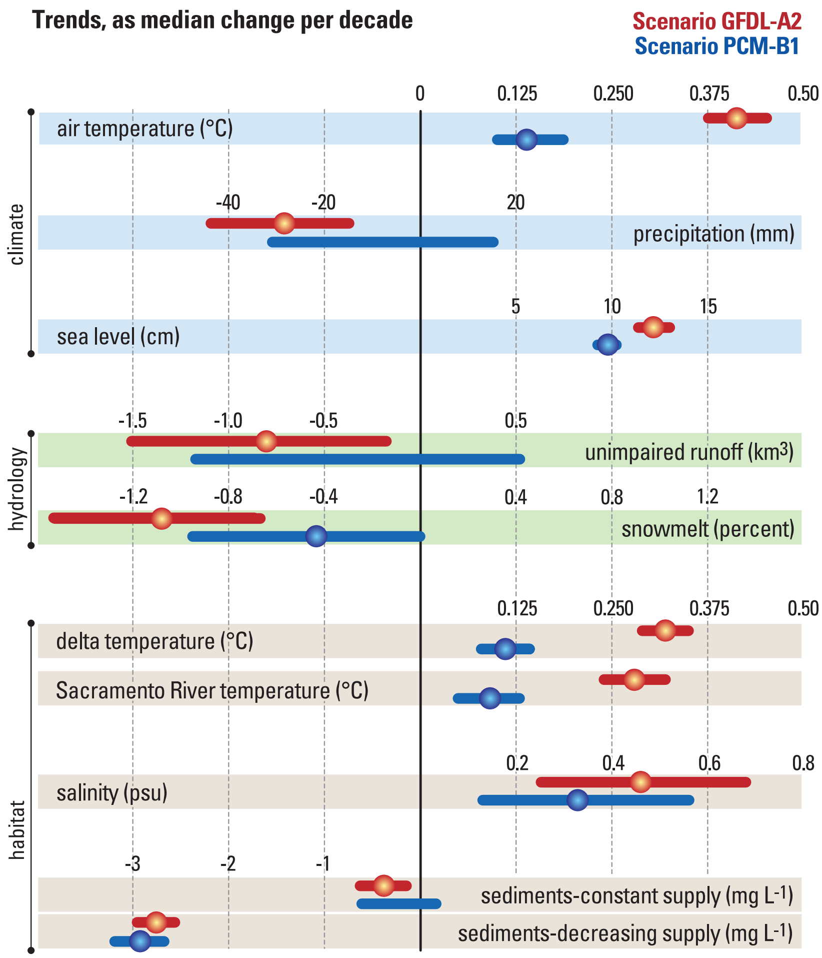


Figure 3. Projected 2010–2099 changes in nine environmental indicators, expressed as median trend per decade, for the A2 scenario (red) and B1 scenario (blue). Statistically significant ($p<0.05$) trends are indicated with solid circles; horizontal lines show 95% confidence limits of the trend estimates.
doi:10.1371/journal.pone.0024465.g003

scenarios were too spatially coarse for the purposes of this study. The GCM temperatures and precipitation values were downscaled onto a 1/8° latitude-longitude grid over the study area by a method called Constructed Analogs [31]. This method is designed to ensure that daily weather simulated by the GCM is consistently carried down to the 12-km scale, and also to yield realistic temperatures across areas with sharp geographic gradients, as in California. The method was applied to climate simulations spanning the period from 1970–2099 to obtain daily, gridded temperature and precipitation patterns over California, from which watershed- and Delta-average (see Fig. 1) values were extracted. The corresponding averages based on historical observations were derived from the gridded meteorological dataset of Maurer et al. [32].

Sea level

A model [20] was adopted to investigate sea level trends and extremes. The model was trained from historical data and used to project future water levels at the San Francisco Golden Gate tide gage location (Fig. 1). The model consists of four components: predicted astronomical tides, synoptic meteorologically-forced sea level fluctuations (based on local sea-level pressure and regional wind stress), ENSO-related monthly-to-interannual fluctuations, and long-term sea level rise associated with global warming. The synoptic and ENSO components were produced with regression models based on historical data [20] and applied to GCM outputs. The climate-change component was based on the method of Vermeer and Rahmstorf [23]. Simulated sea level at the Golden Gate was constructed by superposing these four components, yielding a time series of hourly sea levels from 1970 through 2099 for each climate scenario. Historical observations for 1970–1999 were obtained for the Golden Gate tide gage from NOAA (tidesandcurrents.noaa.gov).

Hydrology and management

A combination of models was used to simulate the watershed's hydrologic behavior. Downscaled meteorological fields (see "Meteorology" above) were used to drive the VIC watershed model [33,34], configured for the Sacramento River and San Joaquin River watersheds using the same parameters applied in several prior studies of the area [16,21,35]. This resulted in daily estimates of unimpaired reservoir inflows for each scenario. A simulation was also performed for the baseline period, driven using historical meteorology [32] to produce the model-based historical hydrological indicators. Estimates of unimpaired flow at major reservoirs throughout the watershed were obtained from the California-Nevada River Forecast Center (www.cnrfc.noaa.gov) and the California Data Exchange Center (CDEC, cdec.water.ca.

gov). Data covering the period 1970–1986 were available, allowing total watershed unimpaired runoff and snowmelt fraction of annual runoff to be calculated for this period, providing the observation-based historical time series for those indicators.

These inflows were used to drive a model of freshwater management operations—the California Department of Water Resources' CALSIM II model [36]. CALSIM is a management optimization model in which, given inputs of reservoir inflows, a set of freshwater management decisions is determined at each time step that optimally satisfy operational goals and constraints. The results are estimates of managed freshwater flows at points throughout the watershed. CALSIM has been applied in other climate-change studies [37,38,39,40,41]. In this study, a new configuration of CALSIM II was used to produce projections for the coming century, and an existing configuration (configured for runs only up to 1994) was used to produce historical estimates (1970–1994). Finally, monthly historical and projected stream temperatures were simulated throughout the watershed using the U.S. Bureau of Reclamation's CALSIM-driven stream-temperature model. This model has also been applied in other climate-change studies (e.g., [40]).

Estuarine salinity

Two complementary models were used to project changes in estuarine salinity due to climate change. The Uncles-Peterson (U-P) model, a 2D box model of San Francisco Bay (Fig. 1), accurately reproduces salinities at weekly to interannual time scales over a wide range of flow regimes [42,43]. Importantly, the U-P model is very economical computationally, enabling the 90-year runs needed to evaluate estuarine variability under the climate scenarios. The U-P model was driven using daily freshwater inflows derived from CALSIM outputs described above, producing daily salinities along the estuary's axis for the historical baseline period and for each future scenario. A simulation was also performed for the baseline period using observed inflows (www.water.ca.gov/dayflow) to derive "observation-based" historical salinity values.

While the U-P simulations provide a representation of the influence of changing upstream hydrology on estuarine salinities, the U-P model does not capture the effects of sea level rise on salinity. The Delft3D model of San Francisco Bay [44] is a 3D process-based model that is sophisticated enough to capture these effects. Delft3D is, however, too computationally demanding to evaluate full 90-year scenarios, and was thus applied in a complementary manner with the U-P model. Multiple runs of Delft3D were used to develop a regression model of salinity changes based on amount of sea level rise (see Supporting Information, Methods S1 for details), which was then driven by historical values of mean sea level for the baseline period, and by sea level projections through the end of this century (see "Sea level" above). The changes were added to the corresponding U-P salinities, and the final results represent our estimate of salinity changes throughout the estuary due to the combination of upstream hydrologic forcing and sea level rise.

Suspended sediment

To evaluate suspended sediment changes under the climate-change scenarios, we developed a rating curve of suspended sediment concentration (SSC) at Rio Vista (Fig. 1) versus Sacramento River discharge (Fig. S1). For each scenario, daily discharges (see "Hydrology and management" above) were used to calculate the daily median SSC, which was then annually averaged. Sediment delivery from the Sacramento River watershed to San Francisco Bay has decreased by about one-half between 1957 and 2001 [45]. As these changes in sediment delivery have occurred, the turbidity and associated SSC within the Delta have also decreased by approximately 40% (Fig. S2). Because it is unclear whether this trend will continue, we developed two sediment-supply scenarios (Fig. S3). The first scenario assumes that the historical rating curve applies in the future, and the second assumes that SSC decreases at 1.6% yr^{-1}, the Delta-wide average rate of SSC decrease from 1975–2008 (data from the Interagency Ecological Program's Environmental Monitoring Program at www.water.ca.gov/bdma; Seasonal Kendall test [46]). Since little observed SSC data exist for the baseline period, the rating curve was applied to produce a hindcast of SSC, using observed discharges (www.water.ca.gov/dayflow) and the historical trend in sediment delivery. This is presented in Fig. 2 as the "observation-based" time series of SSC during the baseline period. The historical "model-based" indicator was produced by applying the rating curve to the CALSIM-based daily discharge estimates (see "Hydrology and management" above), and using the historical trend in sediment delivery.

Delta water temperature

Water temperature data were obtained from the Interagency Ecological Program for the Sacramento River at Rio Vista, where water temperatures were collected from May 1983 through September 2002 (1984–1999 annual averages of these data constitute the observation-based historical baseline). Historical air temperature and insolation data were also acquired (www.cimis.water.ca.gov, www.calclim.dri.edu/data.html). A regression was developed to relate the daily-averaged water temperature to the air temperature and insolation from the same day and water temperature from the preceding day [47]. To project water temperatures for the coming century, the model was applied to the downscaled climate data (see "Meteorology" above), using the mean annual insolation cycle. Similarly, to hindcast water temperatures for 1970–1999, the model was forced with the long-term historical air temperatures and the mean annual insolation cycle, providing the "model-based" historical indicator for Delta water temperature. Annual averages were calculated from the daily model output (see Methods S1 for additional discussion).

Biological indicators

Delta smelt (*Hypomesus transpacificus*) is endemic to the San Francisco Estuary [48,49]. It is listed as endangered by the state of California, and a change in status from threatened to endangered has been deemed warranted under the US Endangered Species Act. Thus, maintaining the population of delta smelt has become a key goal in managing the estuary [50]. To assess the effects of climate change on delta smelt, the frequency of mean daily water temperatures above 25°C was determined from modeled water temperatures at Rio Vista (see "Delta water temperature" above), a location within one tidal excursion of a large portion of delta smelt habitat in the Sacramento River. Multiple studies indicate that mean daily temperature of 25°C is a threshold for high mortality of delta smelt [48,51,52].

Winter-run Chinook salmon (*Oncorhynchus tshawytscha*) is endemic to the Sacramento River system of California and is listed as endangered under both state and US endangered species legislation [49]. Most of the population is subject to water temperature regulation by Shasta Reservoir. Winter-run Chinook salmon begin spawning in the spring. Developing embryos and pre-emergent fry are expected to be in the gravel from May through October. The effects of climate change on winter-run Chinook salmon were assessed by comparing projected mean monthly water temperatures (see "Hydrology and management" above) for the period May–October against a threshold of 16°C, which would result in high mortality of eggs and pre-emergent fry. This is likely a conservative comparison since in a month with a mean of 16°C, approximately half the days would have higher temperatures. Comparisons were made for the Sacramento River at Balls Ferry (Fig. 1), which is at the lower end of the spawning reach. Historical temperature data were obtained for 1991–1999 from CDEC and were used to produce the corresponding observation-based historical indicator. Stream temperature data from the historical run of the stream temperature model (1970–1994; see "Hydrology and management" above) were used to produce the model-based historical indicator.

Sacramento splittail (*Pogonichthys macrolepidotus*) is a large cyprinid, endemic to the San Francisco estuary and watershed [49,53]. Splittail are true floodplain spawners and production of strong year-classes is associated with flooding of Sutter and Yolo bypasses, floodways designed to protect urban areas from flooding. Yolo Bypass (Fig. 1) provides benefits to native fishes, including Chinook salmon and splittail [54]. Floodplains must remain continuously flooded for a minimum of about 30 days [55] for splittail to successfully spawn, and longer inundation periods result in greater production of young splittail [53]. Yolo Bypass provides appropriate spawning conditions at flows above about 113 m^3 s^{-1}. Therefore, for each scenario we counted the number of floods each year in which flows continuously exceeded 113 m^3 s^{-1} for at least 30 days.

Results

Projected responses to climate change in the 21st Century

Our objective was to develop quantitative visions of the SFEW system in two contrasting future climates and to communicate those visions in a way that makes them useful for planning adaptation strategies. Therefore, from the many outputs of models described above we selected nine (Table 1) to use as indicators of changing climate, hydrology and habitat quality. The climate indicators are air temperature over the Delta, precipitation over the Sacramento-San Joaquin River basin, and water elevation at the entrance to San Francisco Bay (Fig. 1). Hydrologic indicators, modeled using the climate projections as inputs, are unimpaired runoff from the headwater basins of the Sierra Nevada and Cascade ranges and the snowmelt contribution to runoff. Habitat indicators, modeled using the climate and hydrologic projections as inputs, are salinity in northern San Francisco Bay, water temperature in the upper Sacramento River, and water temperature and suspended sediment concentration (SSC) in the Delta (Fig. 1). We show future visions of the SFEW as yearly mean values of each environmental indicator for the period 2010–2099 and compared to the 1970–1999 baseline period (Fig. 2). To simplify presentation of results we use "B1 scenario" to denote projections from the PCM model using B1 GHG emissions, and "A2 scenario" to denote projections from the GFDL model using A2 emissions.

Most indicators show good agreement between historical model-based and observation-based time series (Fig. 2, left panels). The climate indicators are not necessarily expected to agree in this sense because the "historical" GCM runs do not correspond to actual historical variations, but instead reflect a realization of climate given historical GHG forcings. Of the remaining indicators, Sacramento River water temperature has only three years of overlap between observations and simulations, though agreement is good during that time. Annually averaged Delta water temperature shows poor agreement ($r = 0.41$) during the period shown. This is a result of three high-flow years near the end of the comparison period, which cause errors in the annual averages (see Methods S1 for more details). The effect of high flow on Delta temperatures (Fig. S4) does not create significant biases in the projections because unimpaired runoff changes little (B1) or declines (A2) for the climate scenarios presented. At the daily timescale, which is critical to fish survival, the comparison of modeled and observed temperatures yielded very high correlations ($r = 0.98$). Unimpaired runoff, snowmelt fraction of annual runoff, north Bay salinity, and suspended sediment concentrations all have high correlations ($r = 0.99$, 0.87, 0.98, and 0.997, respectively) that are strongly statistically significant ($p < 0.00001$).

Air temperature increases steadily in both future scenarios (Fig. 2), but the rate of change is faster in the A2 scenario (maximum annual temperature reaches 21°C) than in the B1 scenario (maximum annual temperature of 18.6°C). Annual precipitation declines steadily in the A2 scenario and is persistently below the modeled 1970–99 baseline by the latter part of the century. There is no apparent secular trend of precipitation change in the B2 scenario, but this projection has large interannual variability that includes years of extreme high precipitation and a simulated multi-year drought in the 2070 decade (Fig. 2). These two future climates span much of the range of temperature and precipitation projections made within a larger ensemble of climate models and GHG emissions [21]. Our projections of sea level rise are within the range of global sea level rise developed in recent studies [4] and reach 125 cm (A2) and 96 cm (B1) above the observed and modeled baselines by the end of this century (Fig. 2).

The hydrologic indicators reflect combined effects of changing air temperature and precipitation. Projections of unimpaired runoff largely reflect changes in precipitation. Runoff in the A2 scenario is 11–12% below the baseline during the first two–thirds of the century. Then, coincident with the simulated end-of-century drought, runoff drops another 16% and persists at this low level for nearly 15 years. Runoff in the B1 scenario exhibits the same large interannual variability of precipitation, including an extremely wet year in 2023 and two very wet years and large droughts between 2065 and 2085. The snowmelt contribution to annual runoff declines steadily in the A2 scenario, but it shows no obvious trend in the B1 scenario until the last two decades when runoff is consistently below the historical mean (Fig. 2). These changes imply continuing shifts toward earlier runoff as a declining fraction of annual runoff occurs during the snowmelt season.

We used these climate and hydrologic projections to develop the first quantitative assessments of how habitat quality in the SFEW will be altered by climate change. As a response to both sea level rise and reduced runoff, computed salinity in northern San Francisco Bay increases 4.5 (A2 scenario) and 2.2 psu (B1 scenario) above the 1979–1999 baseline during the last third of the century. Mean annual water temperature in the upper Sacramento River approaches or exceeds 14°C regularly toward the end of the A2 scenario, and also during the projected 2070s drought in the B1 scenario. Delta water temperatures also increase steadily in both future climates, most rapidly in the A2 scenario. Suspended sediment concentrations in the Delta were calculated as a function of river inflow, assuming that either (a) the supply of erodible sediments in the river system remains constant, or (b) supply decreases as the declining trend of recent decades [56] continues. Sediment concentrations decline slightly under assumption (a), but rapidly under assumption (b) in both climate scenarios (Fig. 2).

We emphasize that such model-based projections are not predictions but instead are plausible depictions of how this

Table 1. Environmental indicators analyzed directly (top 10; see Figs. 2–3) or exceedences of thresholds (bottom 4; see Fig. 4), with corresponding spatial domains (see Fig. 1), units of measurement, and social/ecological significance.

Indicator	Spatial Domain	Metric	Significance
Air temperature	Sacramento-San Joaquin Delta	°C (annual mean)	Water supply; water & habitat quality; human health
Precipitation	Sacramento-San Joaquin watershed	mm yr^{-1}	Water supply; water & habitat quality
Sea level height	San Francisco Bay entrance	cm	Flood risk; water & habitat quality
Unimpaired runoff	Sacramento-San Joaquin headwaters	km^3 yr^{-1}	Water supply; flood protection; reservoir operations; water & habitat quality
Snowmelt contribution	Sacramento-San Joaquin headwaters	percent (of annual runoff)	Seasonal hydrology; flood protection; water & habitat quality
Salinity	Northern San Francisco Bay	psu (April–June mean)	Estuarine habitat quality; drinking-water quality
Water temperature	Upper Sacramento River	°C (annual mean)	Habitat quality
Water temperature	Sacramento-San Joaquin Delta	°C (annual mean)	Habitat quality
Suspended sediment - constant supply	Delta, Lower Sacramento River	mg L^{-1} (annual mean)	Habitat & water quality; estuary geomorphology; wetland sustainability
Suspended sediment - decreasing supply	Delta, Lower Sacramento River	mg L^{-1} (annual mean)	Habitat & water quality; estuary geomorphology; wetland sustainability
Extreme water level	San Francisco Bay entrance	h yr^{-1}>99.99th percentile	Flood risk
Lethal water temperature	Upper Sacramento River	months yr^{-1}>16°C	Sustainability of winter-run Chinook salmon
Lethal water temperature	Sacramento-San Joaquin Delta	days yr^{-1}>25°C	Sustainability of delta smelt
Floodplain inundation	Yolo Bypass	flow>113 m^3 s^{-1}, duration>29 d	Ecosystem restoration (floodplain habitat management)

doi:10.1371/journal.pone.0024465.t001

complex landscape might respond to prescribed model- and emissions-specific future climates. Importantly, we have not considered potentially confounding effects of changing water resource management objectives, rules or infrastructure. We also have not considered changes in land use or infrastructure that might occur through planned actions or catastrophic events such as major levee breaks. However, even considering these constraints and caveats, our projections from two different climate scenarios include years with mean air and water temperature, sea level height and estuarine salinity well above observed and modeled values in the 1970–99 baseline period (Fig. 2). They also include years with annual precipitation, snowmelt contribution to runoff and suspended sediment concentrations well below modeled and observed historical values.

Trends of the environmental indicators

Indicators of climate-driven environmental change will be most useful to policy makers and resource managers if they measure rates of change and indicator sensitivity to different climate scenarios. We extracted this information from the time series of each indicator shown in Fig. 2 by calculating an overall trend for the period 2010–2099 and measuring its statistical significance. The trends represent median rates of change over the 90-year series, and are expressed as rates of change per decade. Results in Fig. 3 present an integrated view of how the SFEW system will respond to global climate change as realized in two future scenarios. Among the climate indicators, air temperature and sea level increase significantly in both scenarios. Air temperature increases 0.42°C decade $^{-1}$ in the A2 scenario, but only 0.14°C decade^{-1} in the B1 scenario (Fig. 3). Sea level increases 12.3 and 9.9 cm decade^{-1} in the A2 and B1 scenarios, respectively. Precipitation declines significantly (-28 mm decade^{-1}) in the A2 scenario, but does not have a significant trend in the B1 scenario. The hydrologic indicators respond to these changes in precipitation and air temperature. Unimpaired runoff, like precipitation, has a significant negative trend in the A2 scenario (-0.80 km^3 decade^{-1}) but not in the B1 scenario (Fig. 3). However, the snowmelt contribution to runoff declines significantly in both scenarios, at -1.1% decade^{-1} (A2 scenario) and -0.4% decade^{-1} (B1 scenario).

Water temperatures in the Sacramento River respond to two factors, both of which trend significantly: 1) increasing air temperature, and 2) decreasing snowmelt runoff reducing the amount of cold water in the upstream reservoirs available to manage downstream temperatures. Water temperatures in the Delta, well removed from the effects of the major reservoirs, respond primarily to increasing air temperature. Sacramento and Delta water temperatures increase significantly, and at roughly the same rate, in both scenarios (Fig. 3). Salinity in northern San Francisco Bay (Fig. 3) also increases significantly in both scenarios (+0.46 psu decade^{-1} for A2, +0.33 psu decade^{-1} for B1), due to sea level rise in both scenarios and the added effect of declining runoff in A2. Suspended sediment concentrations in the Delta change only slightly if sediment supply in the river system remains constant, but they fall rapidly (-2.7 and -2.9 mg L^{-1} decade $^{-1}$) in both climates if sediment supply continues to decline. Therefore, projections of suspended sediment concentrations in the Delta, and consequently sediment transport to San Francisco Bay, are driven more by prescribed changes in sediment supply than by climate-driven changes in river discharge (Fig. 3).

Increasing frequency of extreme events

Some important ramifications of climate change are not captured in annual mean indices because these don't depict changes in the frequency of extreme events [3]. We computed four environmental indicators as exceedence frequencies of threshold values chosen to measure risks to humans or native biota. Projected water levels at the Golden Gate were compared to the historical 99.99th percentile of water elevation (141 cm, relative to the recent historical mean sea level). Both climate scenarios project marked increases in the frequency of extreme water heights over the historical rate of approximately 8 hours decade^{-1}, amounting to increases to 2,000 (A2) and 1,200 (B1) hours decade^{-1} by mid century, and 30,000 (A2) and 15,000 (B1) hours decade^{-1} by the end of the century (Fig. 4).

As an indicator of habitat quality for delta smelt, we calculated number of days each decade when projected water temperature in the Delta exceeds 25°C. The frequency of occurrence of temperatures greater than 25°C increases gradually in the B1 scenario but rapidly in the A2 scenario (Fig. 4). The frequency of occurrence of lethal temperatures for Chinook salmon (>16°C) grows modestly in the B1 scenario, except during the simulated drought of the 2070-decade when this threshold is exceeded in 17 months (Fig. 4). River temperatures above 16°C become common (>20 months decade^{-1}) after 2080 in the A2 scenario. The final habitat indicator is number of years each decade in which spring floods are large enough to inundate the Yolo Bypass (Fig. 1) for at least 30 consecutive days, a minimum threshold for successful spawning of Sacramento splittail. Spring flooding continues through the 21st century in the B1 scenario. But the warmer and drier climate in the A2 scenario reduces the frequency of spring floods having duration long enough for successful spawning and rearing of this species (Fig. 4).

Discussion

California's San Francisco Estuary-Watershed system is the focus of continuing policy debates centered around the challenge of meeting multiple and sometimes conflicting objectives of resource management [57] . Our projections show how those conflicts and the challenge of resource management could intensify as the water supply, sea level, and habitats are transformed by global climate change. We highlight five conclusions that emerge from our study, and end with general lessons to guide strategies of climate-change adaptation in this and other coastal landscapes.

Uncertainty about how SFEW will evolve in the future

The two scenarios used in this study were chosen to explore possible futures and, at the same time, illustrate uncertainty. Different projected futures arise from differences among GCMs in their sensitivity to greenhouse gas emissions and from a range of possible GHG emissions trajectories. Propagation of this uncer-

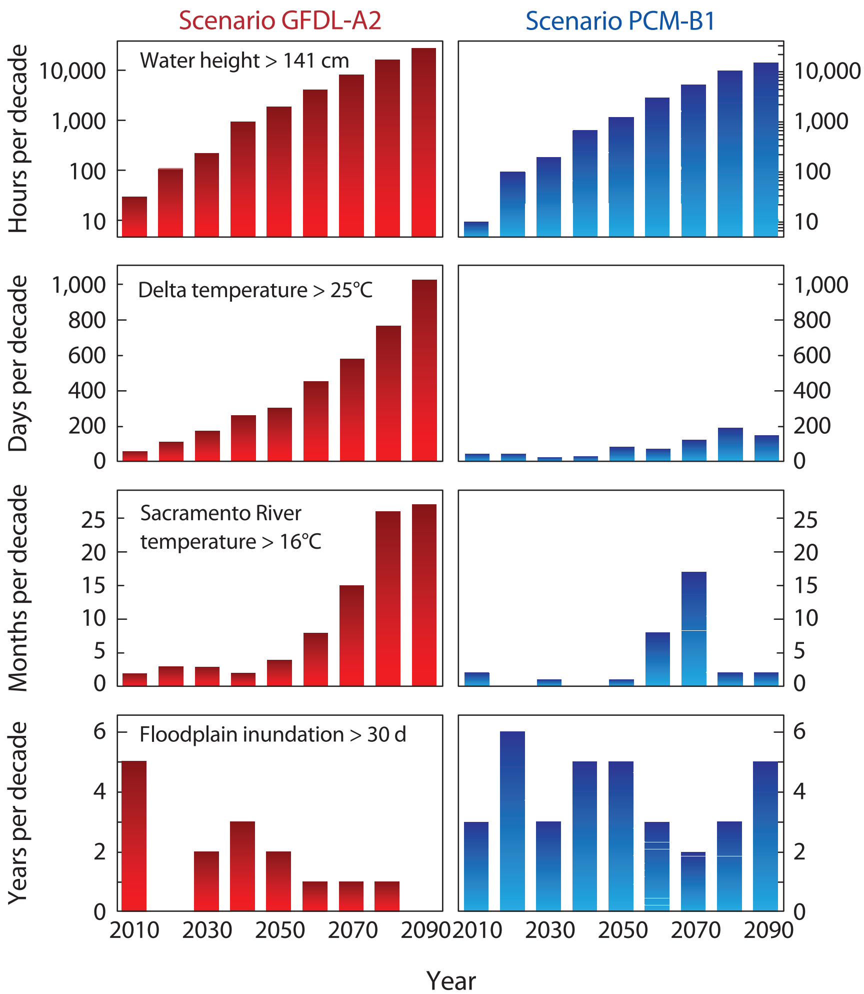


Figure 4. Projected 2010–2099 changes in the occurrence of extreme environmental conditions in the San Francisco Estuary-Watershed system for the A2 (left) and B1 (right) scenarios. The indicators count projected exceedences each decade of threshold values based on historical extreme water elevation or having significance for sustainability of native species of fish (lethal water temperatures) or habitat restoration through management of floodplain habitats.
doi:10.1371/journal.pone.0024465.g004

tainty into the physical and biological systems in SFEW varies among environmental indicators that fall into two classes. First are those with non-significant trends in the B1 scenario, but with large and significant trends in the A2 scenario: precipitation and unimpaired runoff (Fig. 3). Future changes of these indicators will depend on how much climate change is realized and thus on how sensitive the climate system proves to be to greenhouse gases and how future emissions evolve—neither of which can be predicted yet. If realized, the significant trends toward reduced precipitation and runoff in the A2 scenario would have important implications for California's future water supply. The second class of indicators includes those with significant trends in both scenarios, indicating that these represent likely regional responses to global warming. Within this class are two subclasses having different sensitivities to the uncertainty of climate projections. The projected trends of salinity increase, snowmelt decline, and SSC with decreasing supply have comparable magnitudes (overlapping confidence intervals) in the A2 and B1 scenarios (Fig. 3). Therefore, changes in these indicators are relatively insensitive to the uncertainty arising from differences among GCMs and emissions trajectories. The other subclass includes trends of air and water temperature and sea level, which have non-overlapping confidence intervals in the two scenarios. Therefore, changes in these indicators are likely, but the rates of change are strongly tied to projected rates of global warming, so these indicators are particularly sensitive to model- and emissions-specific scenarios.

This classification of projected responses to climate change suggests that regional planners and resource managers should consider: (a) strategies for adaptation to progressively increasing air and water temperature, sea level and salinity intrusion in the SFEW, and further shifts toward more runoff in winter and less in spring-summer; but (b) planning for a broad range of future water supply because GCMs differ widely in their projections of precipitation trends. Effective strategies will be flexible and responsive to new data and assessments of climate change as they emerge [58]. For example, projections of global sea level rise are evolving rapidly [4,23] and are likely to undergo further revisions in the future. Therefore, our projections of environmental change are best viewed as a starting place; each will be modified as new information and tools emerge for assessing regional responses to global change [3].

Today's extremes could become tomorrow's norms

These projections highlight an important manifestation of climate change: changes in mean values of hydroclimatic variables can induce relatively large changes in the frequency of extreme events [3]. As examples, we show projections of increasing frequency of exceptional sea level and water temperature in both scenarios, and of decreasing floodplain inundation in the A2 scenario (Fig. 4). These imply growing risks of coastal flooding, extinction of native fishes, and decreasing feasibility of some ecosystem restoration actions. Therefore, regional resource planning and risk assessments should anticipate shifts into regimes of environmental conditions unprecedented in the period of our social and economic development. This challenge is daunting because of large uncertainty reflected in the variability among indicators in their sensitivity to climate scenario (Fig. 4), and because changing frequency of extreme conditions implies that the indicators will fluctuate within new envelopes of variability over time – i.e., their underlying drivers become non-stationary. Today's resource-management tools are grounded in the assumption of stationary processes of natural variability. Climate change undermines that assumption [3], so adaptation will require development of new probabilistic models to assess environmental changes and their uncertainty in a nonstationary world.

It's not just climate change

Our projections illustrate how responses to climate change could transform the SFEW into a very different system by mid-century (Fig. 2). Transformative change is not new to this ecosystem, which has been altered over the past 150 years by massive landscape modifications, water development, pollutant inputs and introductions of alien species [59]. We selected SSC as one example of an environmental indicator that is more sensitive to landscape change than to climate change. Cessation of hydraulic mining, flood management, and damming the large rivers have decreased sediment delivery to the estuary by about half [56]. Whether this decline continues or abates will have a much greater effect on the future trajectory of SSC than climate change (Fig. 2). This trajectory has important ecological implications because further reductions in sediment supply will increase vulnerability of tidal marshes and mudflats to sea level rise [60], reduce habitat quality for some native fishes, and might promote blooms of toxic cyanobacteria [61] that will be increasingly favored as nutrient-enriched Delta waters warm [62]. Assessments of climate-change impacts must therefore be placed in the broad context of all the drivers that will continue to transform coastal ecosystems [60], including population growth and urbanization, nutrient enrichment, catastrophic levee failures from storms or earthquakes, modified reservoir operations and water conveyances, and implementation of ecosystem restoration plans. Planning will be most challenging with regard to environmental indicators, such as sediment supply, which contain uncertainties in their responses to both climate change and these other drivers of change.

Biological community changes are inevitable

Programs of biodiversity conservation will face an increasingly difficult challenge as environmental conditions in the SFEW diverge from those to which its native species are adapted [13]. Expected outcomes include increasing extinction risk of native species and continuing emergence of nonnative species as dominant components of biological communities. Fishes endemic to the Delta, such as delta smelt, are adapted to cool, turbid, low-salinity habitats [63]. Sustaining populations of these species will become increasingly difficult as Delta waters warm, clear and become more saline (Fig. 2). Of the four runs of Chinook salmon that spawn in the Sacramento-San Joaquin drainage, the winter run is at exceptional risk because its spawning is timed such that eggs develop in summer, when projected river temperatures reach lethal levels (Fig. 4). Communities of fish and their zooplankton prey in the Delta have become increasingly dominated by nonnative species whose successful invasions have been facilitated by synergistic effects of climate anomalies (extended drought) and flow management [64]. Our projections include significant departures from the contemporary climate and flow regimes in the future, so environmental conditions might continue to move toward those that select for nonnative biota.

We have learned from other studies that small perturbations can trigger ecosystem regime shifts. A recent example occurred in Denmark's Ringkøbing Fjord, where mean salinity increased 1.6 psu after actions were taken to enhance water exchange with the North Sea. This small salinity change was followed by sudden and unanticipated reorganization of biological communities at all trophic levels, from phytoplankton to macrobenthos and water-birds [65]. We project larger salinity increases in San Francisco Bay by the end of the 21st century (Fig. 2). Therefore, conservation plans should expect surprises and include monitoring to detect and contingencies for adapting to unexpected shifts in habitats and their biological communities. And, they should be designed to accommodate a range of future climates. Feasibility and outcomes of proposed habitat restoration actions, such as creation of seasonal floodplain habitat (Fig. 4), low-salinity aquatic habitats and thermal refugia for native species [13], will be very different as seasonal hydrology and water temperature change.

The challenge of meeting California's water demands will intensify

California's water supply (annual unimpaired runoff) is projected to decline or remain steady (Fig. 3), and demands are likely to increase as populations and temperatures rise. Deficits of surface runoff are now met with groundwater pumping. However, pumping between 1998 and 2010 depleted 48.5 km^3 of water from the Central Valley groundwater system, and continued groundwater depletion at this rate is unsustainable [66]. Future strategies of water management will require adaptations such as aggressively increasing water-use efficiency, reducing surface water deliveries, capturing more runoff in surface storage or groundwater recharge, and implementing programs of integrated regional water management [67]. Model results suggest that the inherent large annual variability of precipitation will persist (Fig. 2), even as longer-term trends of warming and possibly drying take hold. Therefore, water-resource planning should also include contingencies for longer dry seasons, extended droughts, and extreme floods due to shifts from snow to rain. Diminishing snow packs result in earlier reservoir inflow, so reservoir operations must adapt to a shift toward more water being managed as a hazard (flood control) and less as a resource (reservoir storage). Additional freshwater releases to mitigate increased salinity intrusion into the estuary will be required to maintain quality of drinking water to communities that use the Delta as their municipal water supply. These adaptations to maintain water supply for human consumptive uses will potentially constrain availability of water to meet objectives of habitat conservation plans, such as restoring natural flow and salinity variability to promote recovery of native biota in the Delta [13].

General lessons to guide climate-change adaptation planning

To our knowledge, this is the first attempt at an integrated quantitative assessment of how global signals of climate change would cascade to modify runoff, river discharge, water temperature, sea level, salinity intrusion and suspended sediments in a large watershed-river-estuary-ocean system. Although our projections of climate-driven change are specific to SFEW, lessons from this place-based study can be used as a starting place to guide adaptation strategies elsewhere:

1. Outputs from complex models can be explored by simplifying into a small set of environmental indicators chosen to develop an integrated view of how climate change will be manifested across landscapes.
2. Climatic, hydrologic and habitat indicators vary in their sensitivity to uncertainty about the future; measures of that sensitivity provide important information for assigning priorities and including contingencies in adaptation planning.
3. Results from climate simulations and resulting assessments of climate-change impacts will continue to evolve as the underlying science improves, so adaptation planning must be responsive to the continuing emergence of new models, analyses and insights.
4. Assessments of climate-change impacts are best placed in the broader context of all the drivers of change because some environmental indicators are more sensitive to other drivers such as landscape transformations, species introductions, pollution and water development.
5. Biological community changes are inevitable, and programs of ecosystem rehabilitation and biodiversity conservation will be most likely to meet their objectives if they are designed from projections of the future climate rather than today's climate.
6. Environmental planning should anticipate and adapt to ecosystem regime shifts; monitoring is essential for detecting and responding to regime shifts.
7. Warming in regions such as the western United States implies that sustainability of reliable water supplies will require changes in water management. These adaptations will potentially exacerbate conflicts of water allocation to meet human demands and goals of biological conservation plans.

Finally, our results are consistent with other model-based projections that California's climate will continue to warm through the 21st century. There is uncertainty about how much global temperature will rise in response to increases in greenhouse gases, but it is clear that the rate of warming will increase with higher greenhouse gas emissions [21,24]. Environmental indicators considered here respond more rapidly and more strongly to the A2 scenario than to the B1 scenario (Figs. 2, 3). Collectively, these indicators depict climate-driven changes in the reliability of California's water supply, in risks to humans and ecosystems due to coastal flooding, and in likely outcomes of ecosystem restoration programs. Contrasting futures in the A2 and B1 scenarios show that mitigation steps that slow greenhouse gas emissions in the first half of the 21st century would reduce the requirements for adaption to climate-change impacts through the end of the century. However, regardless of the greenhouse gas emissions trajectory, substantial global and regional warming is likely, so successful climate-change adaptation will require other near-term mitigation actions aimed at buffering some of the long-term climate-change effects depicted by our indicators.

Supporting Information

Figure S1 Sediment rating curve for the Sacramento River at Rio Vista, 1998–2002.
(TIF)

Figure S2 Mean annual turbidity, declining throughout the Sacramento-San Joaquin Delta from 1975–2008. From monthly data provided by California Department of Water Resources, Environmental Monitoring Program.
(TIF)

Figure S3 GFDL and PCM scenarios for suspended sediment concentration (SSC) in the Sacramento River at Rio Vista for constant and decreasing sediment supply. Each band represents the interquartile range of SSC.
(TIF)

Figure S4 Effects of high river flows on errors in modeled annual average Delta water temperatures. Difference between modeled and observed yearly average water temperature is compared to the annually averaged Sacramento River flow; model-observation deviations occur in years with high river flow.
(TIF)

Methods S1 Expanded description of methods with supporting references.
(RTF)

Acknowledgments

This paper is dedicated to our friend and colleague David Peterson who attracted us to work together on problems of integrated science requiring collaborations across the disciplines of atmospheric and ocean sciences, estuarine oceanography, hydrology, ecology and sediment dynamics. We thank Anke Mueller-Solger, Michael Healey and Cliff Dahm for review comments and Jeanne Dileo for illustrations. This is CASCaDE publication number 21.

Author Contributions

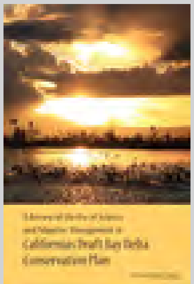
Conceived and designed the experiments: JEC NK LRB DC MDD TLM DHS MTS MvdW RWW ADJ. Performed the experiments: NK LRB DC MDD TLM DHS MTS MvdW RWW. Analyzed the data: JEC NK LRB DC MDD TLM DHS MTS MvdW RWW ADJ. Wrote the paper: JEC NK LRB DC MDD TLM DHS MTS MvdW RWW ADJ.

References

1. Hansen J, Ruedy R, Sato M, Lo K (2010) Global surface temperature change. Reviews of Geophysics 48: RG4004.
2. IPCC (2007) Climate Change 2007 - Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the IPCC. Cambridge: Cambridge University Press.
3. Milly PCD, Betancourt J, Falkenmark M, Hirsch RM, Kundzewicz ZW, et al. (2008) Climate change. Stationarity is dead: Whither water management? Science 319: 573–574.
4. Rahmstorf S (2010) A new view on sea level rise. Nature Reports Climate Change 4: 44–45.
5. Knutson TR, McBride JL, Chan J, Emanuel K, Holland G, et al. (2010) Tropical cyclones and climate change. Nature Geoscience 3: 157–163.
6. Schneider P, Hook SJ (2010) Space observations of inland water bodies show rapid surface warming since 1985. Geophysical Research Letters 37: L22405.
7. Kaushal SS, Likens GE, Jaworski NA, Pace ML, Sides AM, et al. (2010) Rising stream and river temperatures in the United States. Frontiers in Ecology and the Environment 8: 461–466.
8. Winder M, Schindler DE (2004) Climate change uncouples trophic interactions in an aquatic system. Ecology 85: 3178–3178.
9. Boyce DG, Lewis MR, Worm B (2010) Global phytoplankton decline over the past century. Nature 466: 591–596.
10. Service RF (2007) Environmental restoration: Delta blues, California style. Science 317: 442–445.
11. US Department of Agriculture website. Available: http://www.nass.usda.gov/Statistics_by_State/California/Publications/California_Ag_Statistics/Reports/2009cas-all.pdf. Accessed 2011 Aug 17.
12. Bay Delta Conservation Plan website. Available: http://bdcpweb.com/Files/Highlights_of_the_BDCP_FINAL_03-17-11.pdf. Accessed 2011 Aug 17.
13. Moyle PB, Lund JR, Bennett WA, Fleenor WE (2010) Habitat variability and complexity in the upper San Franciso Estuary. San Francisco Estuary and Watershed Science 8(3): 1–24. Available: http://escholarship.org/uc/item/0kf0d32x. Accessed 2011 Aug 17.
14. San Francisco Bay Conservation and Development Commission website. Available: http://www.bcdc.ca.gov/proposed_bay_plan/bp_1-08_cc_draft.pdf. Accessed 2011 Aug 17.
15. Knowles N (2000) Modeling the Hydroclimate of the San Francisco Bay-Delta Estuary and Watershed [Ph. D]. La JollaCA: Scripps Institution of Oceanography, University of California - San Diego.
16. Barnett TP, Pierce DW, Hidalgo HG, Bonfils C, Santer BD, et al. (2008) Human-induced changes in the hydrology of the western United States. Science 319: 1080–1083.
17. Cayan DR, Kammerdiener SA, Dettinger MD, Caprio JM, Peterson DH (2001) Changes in the onset of spring in the western United States. Bulletin of the American Meteorological Society 82: 399–415.
18. Knowles N, Dettinger MD, Cayan DR (2006) Trends in snowfall versus rainfall in the Western United States. Journal of Climate 19: 4545–4559.
19. Stewart IT, Cayan DR, Dettinger MD (2005) Changes toward earlier streamflow timing across western North America. Journal of Climate 18: 1136–1155.
20. Cayan DR, Bromirski PD, Hayhoe K, Tyree M, Dettinger MD, et al. (2008) Climate change projections of sea level extremes along the California coast. Climatic Change 87: 57–73.
21. Cayan DR, Maurer EP, Dettinger MD, Tyree M, Hayhoe K (2008) Climate change scenarios for the California region. Climatic Change 87: 21–42.
22. Cayan DR, Luers AL, Franco G, Hanemann M, Croes B, et al. (2008) Overview of the California climate change scenarios project. Climatic Change 87: S1–S6.
23. Vermeer M, Rahmstorf S (2009) Global sea level linked to global temperature. Proceedings of the National Academy of Sciences of the United States of America 106: 21527–21532.
24. Hayhoe K, Cayan D, Field CB, Frumhoff PC, Maurer EP, et al. (2004) Emissions pathways, climate change, and impacts on California. Proceedings of the National Academy of Sciences of the United States of America 101: 12422–12427.
25. Washington WM, Weatherly JW, Meehl GA, Semtner Jr. AJ, Bettge TW, et al. (2000) Parallel climate model (PCM) control and transient simulations. Climate Dynamics 16: 755–774.
26. Delworth TL, Broccoli AJ, Rosati A, Stouffer RJ, Balaji V, et al. (2006) GFDL's CM2 Global Coupled Climate Models. Part I: Formulation and Simulation Journal of Climate 19: 643–674.
27. Theil H (1950) A rank-invariant method of linear and polynomial regression analysis. I Nederlands Akad Wetensch Proc 53: 386–392.
28. Sen PK (1968) Estimates of the regression coefficient based on Kendall's Tau. Journal of the American Statistical Association 63: 1379–1389.
29. Yue S, Pilon P, Phinney B, Cavadias G (2002) The influence of autocorrelation on the ability to detect trend in hydrological series. Hydrological Processes 16: 1807–1829.
30. Meehl GA, Covey C, Delworth TL, Latif M, McAvaney M, et al. (2007) The global coupled model dataset: A new era in climate change research. Bull Amer Meteor Soc 88: 1383–1394.
31. Hidalgo H, Dettinger M, Cayan D (2008) Downscaling with constructed analogues—Daily precipitation and temperature fields over the United States. California Energy Commission PIER Final Project Report CEC-500-2007-123. 48 p.
32. Maurer EP, Wood AW, Adam JC, Lettenmaier DP, Nijssen B (2002) A long-term hydrologically-based data set of land surface fluxes and states for the conterminous United States. J Clim 15: 3237–3251.
33. Cherkauer KA, Bowling LC, Lettenmaier DP (2003) Variable infiltration capacity cold land process model updates. Global Plan Change 38: 151–159.
34. Liang X, Lettenmaier DP, Wood EF, Burges SJ (1994) A simple hydrologically based model of land surface water and energy fluxes for general circulation models. J Geophys Res 99: 14415–14428.
35. Maurer EP, Hidalgo HG, Das T, Dettinger MD, Cayan DR (2010) The utility of daily large-scale climate data in the assessment of climate change impacts on daily streamflow in California. Hydrol Earth Syst Sci 14: 1125–1138.
36. Draper AJ, Munévar A, Arora SK, Reyes E, Parker NL, et al. (2004) CalSim: generalized model for reservoir system analysis. Journal of Water Resources Planning and Management 130(6): 480–489.
37. Brekke LDMN, Bahford KE, Quinn NWT, Dracup JA (2004) Climate change impacts uncertainty for water resources in the San Joaquin river basin, California. Journal of the American Water Resources Association 40(1): 149–164.
38. Dracup JA, Vicuna S, Leonardson R, Dale L, Hanneman M (2005) Climate Change and Water Supply Reliability. California Energy Commission, PIER Energy-Related Environmental Research, CEC-500-2005-053.
39. Vicuna S, Maurer EP, Joyce B, Dracup JA, Purkey D (2007) The Sensitivity of California Water Resources to Climate Change Scenarios. Journal of the American Water Resources Association 43(2): 482–498.
40. Anderson J, Chung FI, Anderson M, Brekke L, Easton D, et al. (2008) Progress on incorporating climate change into management of California's water resources. Climatic Change 87, suppl. 1: 91–108.
41. Brekke LD, Maurer EP, Anderson JD, Dettinger MD, Townsley ES, et al. (2009) Assessing reservoir operations risk under climate change. Water Resources Research 45: W04411.
42. Knowles N (1996) Simulation and Prediction of Salinity Variability in San Francisco Bay. [Master's Thesis]. La JollaCA: University of California, San Diego.
43. Peterson D, Cayan D, Dileo J, Noble M, Dettinger M (1995) The role of climate in estuarine variability. American Scientist 83: 58–67.
44. van der Wegen M, Jaffe BE, Roelvink JA (2011) Process-based, morphodynamic hindcast of decadal deposition patterns in San Pablo Bay, California, 1856–1887. Journal of Geophysical Research 116: F02008.
45. Wright SA, Schoellhamer DH (2004) Trends in the sediment yield of the Sacramento River, California, 1957–2001. San Francisco Estuary and Watershed Science 2(2): 1–14. Available: http://escholarship.org/uc/item/891144f4. Accessed 2011 Aug 17.
46. Helsel DR, Hirsch RM (1992) Statistical methods in water resources. Amsterdam: Elsevier.
47. Wagner R, Stacey MT, Brown L, Dettinger M (2011) Statistical models of temperature in the Sacramento-San Joaquin Delta under climate-change scenarios and ecological implications. Estuaries and Coasts 34: 544–556.
48. Bennett WA (2005) Critical assessment of the delta smelt population in the San Francisco Estuary, California. San Francisco Estuary and Watershed Science 3(2): 1–73. Available: http://escholarship.org/uc/item/0725n5vk. Accessed 2011 Aug 17.
49. Moyle PB (2002) Inland fishes of California. Berkeley: University of California Press.
50. Sommer T, Armor C, Baxter R, Breuer R, Brown L, et al. (2007) The collapse of pelagic fishes in the upper San Francisco Estuary. Fisheries 32: 270–277.

51. Swanson C, Reid T, Young PS, Cech JJ (2000) Comparative environmental tolerances of threatened delta smelt (*Hypomesus transpacificus*) and introduced wakasagi (*H. nipponensis*) in an altered California estuary. Oecologia 123: 384–390.
52. Nobriga ML, Sommer T, Feyrer F, Fleming K (2008) Long-term trends in summertime habitat suitability for delta smelt, *Hypomesus transpacificus*. San Francisco Estuary and Watershed Science 6(1, Available: http://repositories.cdlib.org/jmie/sfews/vol6/iss1/art1. Accessed 2011 Aug 17.
53. Moyle PB, Baxter RD, Sommer T, Foin TC, Matern SA (2004) Biology and population dynamics of Sacramento Splittail (*Pogonichthys macrolepidotus*) in the San Francisco Estuary: a review. San Francisco Estuary and Watershed Science 2(2): 1–4. Available: http://escholarship.ucop.edu/uc/item/61r48686. Accessed 2011 Aug 17.
54. Sommer T, Harrell B, Nobriga M, Brown R, Moyle P, et al. (2001) California's Yolo Bypass: Evidence that flood control can be compatible with fisheries, wetlands, wildlife, and agriculture. Fisheries 26: 6–16.
55. Sommer T, Baxter R, Herbold B (1997) Resilience of splittail in the Sacramento-San Joaquin Estuary. Transactions of the American Fisheries Society 126: 961–976.
56. Schoellhamer DH (2011) Sudden clearing of estuarine waters upon crossing the threshold from transport- to supply- regulation of sediment transport as an erodible sediment pool is depleted: San Francisco Bay, 1999. Estuaries & Coasts 34: 885–899.
57. CALFED Science Program website. Available: http://science.calwater.ca.gov/publications/sbds.html. Accessed 2011 Aug 17.
58. Dettinger MD, Culberson S (2008) Internalizing climate change — Scientific resource management and the climate change challenges. San Francisco Estuary and Watershed Science 6(2): 1–17. Available: http://escholarship.org/uc/item/64z013q3. Accessed 2011 Aug 17.
59. Nichols FH, Cloern JE, Luoma SN, Peterson DH (1986) The modification of an estuary. Science 231: 567–573.
60. Scavia D, Field JC, Boesch DF, Buddemeier RW, Burkett V, et al. (2002) Climate change impacts on US coastal and marine ecosystems. Estuaries 25: 149–164.
61. Lehman PW, Teh SJ, Boyer GL, Nobriga ML, Bass E, et al. (2009) Initial impacts of *Microcystis aeruginosa* blooms on the aquatic food web in the San Francisco Estuary. Hydrobiologia 637: 229–248.
62. Paerl HW, Huisman J (2008) Climate. Blooms like it hot. Science 320: 57–58.
63. Feyrer F, Newman K, Nobriga M, Sommer T (2010) Modeling the effects of future outflow on the abiotic habitat of an imperiled estuarine fish. Estuaries and Coasts 34: 120–128.
64. Winder M, Jassby AD, MacNally R (2011) Synergies between climate anomalies and hydrological modifications facilitate estuarine biotic invasions. Ecology Letters;in press: DOI 10.1111/j.1461-0248.2011.01635.x.
65. Petersen JK, Hansen JW, Laursen MB, Clausen P, Carstensen J, et al. (2008) Regime shift in a coastal marine ecosystem. Ecological Applications 18: 497–510.
66. Famiglietti JS, Lo M, Ho SL, Bethune J, Anderson KJ, et al. (2011) Satellites measure recent rates of groundwater depletion in California's Central Valley. Geophysical Research Letters 38: L03403.
67. California Department of Water Resources website. Available: http://www.water.ca.gov/climatechange/docs/ClimateChangeWhitePaper.pdf. Accessed 2011 Aug 17.

BDCP1673



A Review of the Use of Science and Adaptive Management in California's Draft Bay Delta Conservation Plan

ISBN
978-0-309-21231-1

100 pages
6 x 9
PAPERBACK (2011)

Panel to Review California's Draft Bay Delta Conservation Plan; National Research Council

BDCP1673

A Review of the Use of Science and Adaptive Management in California's Draft Bay Delta Conservation Plan

Panel to Review California's Draft Bay Delta Conservation Plan

Water Science and Technology Board

Ocean Studies Board

Division on Earth and Life Studies

NATIONAL RESEARCH COUNCIL
OF THE NATIONAL ACADEMIES

THE NATIONAL ACADEMIES PRESS
Washington, D.C.
www.nap.edu

THE NATIONAL ACADEMIES PRESS 500 Fifth Street, N.W. Washington, DC 20001

NOTICE: The project that is the subject of this report was approved by the Governing Board of the National Research Council, whose members are drawn from the councils of the National Academy of Sciences, the National Academy of Engineering, and the Institute of Medicine. The members of the panel responsible for the report were chosen for their special competences and with regard for appropriate balance.

Support for this study was provided by the Department of the Interior under contract no. 80221-A-G100. Any opinions, findings, conclusions, or recommendations expressed in this publication are those of the author(s) and do not necessarily reflect the views of the organizations or agencies that provided support for the project.

International Standard Book Number 13: 978-0-309-21231-1
International Standard Book Number 10:0-309-21231-6

Photo on the cover is courtesy of David Policansky.

Additional copies of this report are available from the National Academies Press, 500 5th Street, N.W., Lockbox 285, Washington, DC 20055; (800) 624-6242 or (202) 334-3313 (in the Washington metropolitan area); Internet, *http://www.nap.edu*.

Printed in the United States of America.

THE NATIONAL ACADEMIES

Advisers to the Nation on Science, Engineering, and Medicine

The **National Academy of Sciences** is a private, nonprofit, self-perpetuating society of distinguished scholars engaged in scientific and engineering research, dedicated to the furtherance of science and technology and to their use for the general welfare. Upon the authority of the charter granted to it by the Congress in 1863, the Academy has a mandate that requires it to advise the federal government on scientific and technical matters. Dr. Ralph J. Cicerone is president of the National Academy of Sciences.

The **National Academy of Engineering** was established in 1964, under the charter of the National Academy of Sciences, as a parallel organization of outstanding engineers. It is autonomous in its administration and in the selection of its members, sharing with the National Academy of Sciences the responsibility for advising the federal government. The National Academy of Engineering also sponsors engineering programs aimed at meeting national needs, encourages education and research, and recognizes the superior achievements of engineers. Dr. Charles M. Vest is president of the National Academy of Engineering.

The **Institute of Medicine** was established in 1970 by the National Academy of Sciences to secure the services of eminent members of appropriate professions in the examination of policy matters pertaining to the health of the public. The Institute acts under the responsibility given to the National Academy of Sciences by its congressional charter to be an adviser to the federal government and, upon its own initiative, to identify issues of medical care, research, and education. Dr. Harvey V. Fineberg is president of the Institute of Medicine.

The **National Research Council** was organized by the National Academy of Sciences in 1916 to associate the broad community of science and technology with the Academy's purposes of furthering knowledge and advising the federal government. Functioning in accordance with general policies determined by the Academy, the Council has become the principal operating agency of both the National Academy of Sciences and the National Academy of Engineering in providing services to the government, the public, and the scientific and engineering communities. The Council is administered jointly by both Academies and the Institute of Medicine. Dr. Ralph J. Cicerone and Dr. Charles M. Vest are chair and vice chair, respectively, of the National Research Council.

www.national-academies.org

BDCP1673

PANEL TO REVIEW CALIFORNIA'S DRAFT BAY DELTA CONSERVATION PLAN *

HENRY J. VAUX, JR., Chair, Professor Emeritus, University of California
MICHAEL E. CAMPANA, Oregon State University, Corvallis
JEROME B. GILBERT, Consultant, Orinda, California
ALBERT E. GIORGI, BioAnalysts, Inc., Redmond, Washington
ROBERT J. HUGGETT, Professor Emeritus, College of William and Mary, Seaford, Virginia
CHRISTINE A. KLEIN, University of Florida College of Law, Gainesville, Florida
SAMUEL N. LUOMA, U.S. Geological Survey, Emeritus, Menlo Park, California
THOMAS MILLER, University of Maryland Center for Environmental Science, Chesapeake Biological Laboratory, Solomons
STEPHEN G. MONISMITH, Stanford University, California
JAYANTHA OBEYSEKERA, South Florida Water Management District, West Palm Beach
HANS W. PAERL, University of North Carolina, Chapel Hill
MAX J. PFEFFER, Cornell University, Ithaca, New York
DESIREE D. TULLOS, Oregon State University, Corvallis

NRC Staff

LAURA J. HELSABECK, Staff Officer
DAVID POLICANSKY, Scholar
STEPHEN D. PARKER, Director, Water Science and Technology Board
SUSAN ROBERTS, Director, Ocean Studies Board
ELLEN DE GUZMAN, Research Associate
SARAH BRENNAN, Senior Program Assistant

* Biographical information for panel members is in Appendix H. This project was organized and overseen by the NRC's Water Science and Technology Board (lead) and Ocean Studies Board, whose rosters are in Appendixes F and G, respectively.

Preface

This panel's review of the draft Bay Delta Conservation Plan (BDCP) has occurred alongside myriad activities in the Delta to facilitate a secure water future for California, including an environmental future, and alongside related activities of the National Research Council (NRC). I particularly want to make clear the distinction between the Delta Plan and the BDCP, and between this panel's report and two related NRC reports, one already published, one still in preparation.

The Delta Plan (formally the Delta Stewardship Plan) is a comprehensive umbrella plan mandated by the California Delta Protection Act of 2009 to advance the goals of improving the reliability of California's water supply and restoring, protecting, and enhancing the Delta ecosystem. It is overseen by the state of California and a broadly represented council of stakeholders as authorized by statute. Although the Delta Plan was not part of this review and is mentioned only incidentally in this report, it is related to the BDCP to some degree by intent and to some degree by statute (those relationships are briefly discussed in the body of this report). Readers should understand from the outset, however, that it is the BDCP, and only the BDCP that is reviewed in this report.

The related NRC activities are being conducted by the Committee on Sustainable Water and Environmental Management in the California Bay-Delta. The NRC appointed that committee in response to a request from Congress and the Department of the Interior to provide advice on two topics: (1) the scientific basis of actions identified in two biological opinions by the National Marine Fisheries Service and the U.S. Fish and Wildlife Service to protect threatened and endangered species in the Delta, and (2) how to most effectively incorporate science and adaptive management into a holistic program for managing and restoring the Delta. Advice on the first topic was provided in a report published in March 2010 titled *A Scientific Assessment of Alternatives for Reducing Water Management Effects on Threatened and Endangered Fishes in California's Bay-Delta*. The committee expects to release its advice on the second topic late in 2011.

While the committee was working on its second report, the U.S. Secretaries of Interior and Commerce asked the NRC to review the draft BDCP in terms of its use of science and adaptive management. In response, the NRC established a separate Panel to Review California's Draft Bay Delta Conservation Plan, which is the author of this report. Although there is considerable overlap between the membership of the committee and this panel, the two groups were appointed separately, have separate statements of task, and have worked independently of each other.

This report was reviewed in draft form by individuals chosen for their diverse perspectives and technical expertise in accordance with the procedures approved by the NRC's Report Review Committee. The purpose of this independent review is to provide candid and critical comments that will assist the NRC in making its published report as sound as possible, and to ensure that the report meets NRC institutional standards for objectivity, evidence, and responsiveness to the study charge. The review comments and draft manuscript remain confidential to protect the integrity of the deliberative process.

We thank the following for their review of this report: Frank Davis, University of California, Santa Barbara; Holly Doremus, University of California, Berkeley; Peter Gleick, Pacific Institute for Studies in Development, Environment, and Security; George Hornberger, Vanderbilt University; Cynthia Jones, Old Dominion University; Jay Lund, University of California, Davis; Judy Meyer, University of Georgia; and Lynn Scarlett, Resources for the Future.

Although these reviewers provided constructive comments and suggestions, they were not asked to endorse the report's conclusions and recommendations, nor did they see the final draft of the report before its release. The review of this report was overseen by Michael Kavanaugh, Geosyntec Consultants, who was appointed by the NRC's Report Review Committee and by Paul Risser, University of Oklahoma, who was appointed by the NRC's Division on Earth and Life Studies. They were responsible for ensuring that an independent examination of this report was conducted in accordance with NRC institutional procedures and that all review comments received full consideration. Responsibility for this report's final contents rests entirely with the authoring committee and the NRC.

Henry J. Vaux, Jr.
Chair
Panel to Review California's Draft Bay Delta Conservation Plan

Contents

Summary

The San Francisco Bay Delta Estuary (Delta, for short) is a large, complex estuarine ecosystem in California (Figure 1). It has been substantially altered by dikes, levees, channelization, pumps, human development, introduced species, dams on its tributary streams, and contaminants. The Delta supplies water from the state's wetter northern regions to the drier southern regions and also serves as habitat for many species, some of which are threatened and endangered. The restriction of water exports in an attempt to protect those species together with the effects of several dry years have exacerbated tensions over water allocation in recent years, and have led to various attempts to develop comprehensive plans to provide reliable water supplies and to protect the ecosystem.

One of those plans is the Bay Delta Conservation Plan (BDCP), the focus of this report. The BDCP is technically a habitat conservation plan (HCP), an activity provided for in the federal Endangered Species Act that protects the habitat of listed species in order to mitigate the adverse effects of a federal project or activity that incidentally "takes"[1] (includes actions that "harm" wildlife by impairing breeding, feeding, or sheltering behaviors) the listed species. It similarly is a natural community conservation plan (NCCP) under California's Natural Community Conservation Planning Act (NCCPA). It is intended to obtain long-term authorizations under both the state and federal endangered species statutes for proposed new water operations—primarily an "isolated conveyance structure," probably a tunnel, to take water from the northern part of the Delta for export to the south, thus reducing the need to convey water through the Delta and out of its southern end.

The U.S. Secretaries of the Interior and Commerce requested that the National Research Council (NRC) review the draft BDCP in terms of its use of science and adaptive-management (see Appendix A for the full statement of task). In response, the NRC established the Panel to Review California's Draft Bay Delta Conservation Plan, which prepared this report. The panel reviewed

[1] *Take* means "to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct." ESA, Section 3, 16 U.S.C. 1532.
Harm, within the statutory definition of "take" has been further defined by regulation: "Harm in the definition of take in the Act means an act which actually kills or injures wildlife. Such act may include significant habitat modification or degradation where it actually kills or injures wildlife by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering." 50 C.F.R. 17.3.

BDCP1673

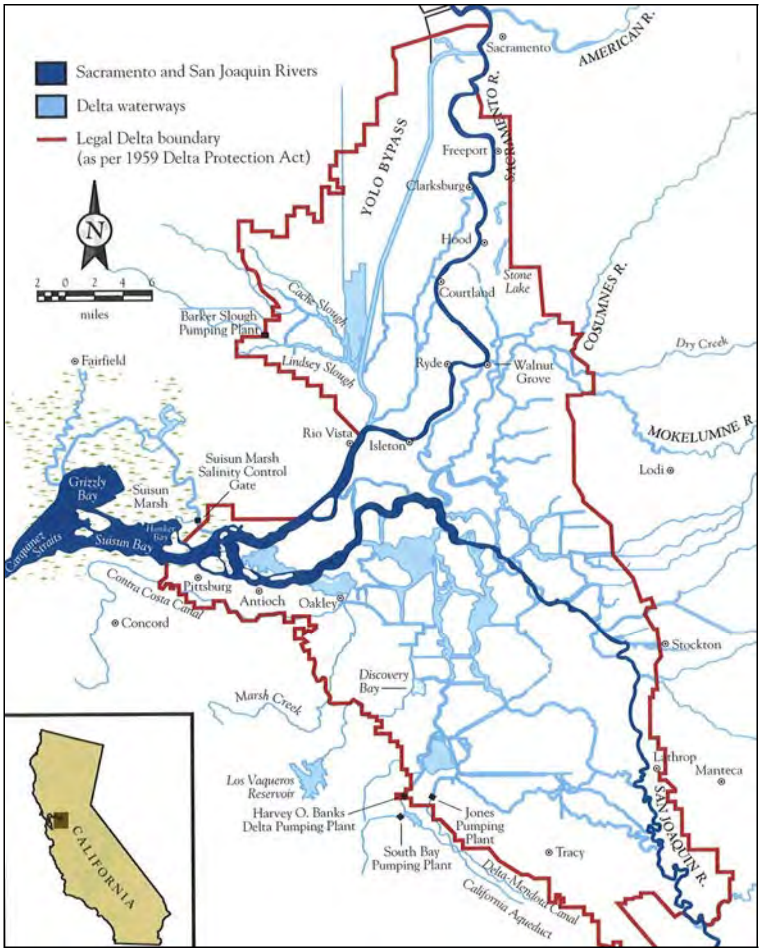


FIGURE 1. The Sacramento-San Joaquin Delta in California. San Francisco Bay, an integral part of the system, is just to the west. SOURCE: Reprinted, with permission, from Lund et al. (2010). Copyright by Public Policy Institute of California.

the draft BDCP, which was posted on the BDCP website: (*http://www.resources.ca.gov/bdcp/*) on November 18, 2010. [2] The panel determined that the draft BDCP is incomplete in a number of important areas and takes this opportunity to identify key scientific and structural gaps that, if addressed, could lead to a more successful and comprehensive final BDCP. Yet science alone cannot solve the Delta's problems. Water scarcity in California is very real, the situation is legally and politically complex, and many stakeholders have differing interests. The effective management of scarcity requires not only the best science and technology, but also consideration of public and private values, usually through political processes, to arrive at plans of action that are scientifically based but also incorporate and reflect the mix of differing personal and group values.

CRITICAL GAPS IN THE SCOPE OF THE DRAFT BDCP

At the outset of its review, the panel identified a problem with the geographical and hydrologic scope of the draft BDCP. The BDCP aims to address management and restoration of the San Francisco Bay Delta Estuary, an estuary that extends from the Central Valley to the mouth of San Francisco Bay. Thus, given that the BDCP describes a *bay* delta conservation plan, the omission of analyses of the effects of the BDCP efforts on San Francisco Bay (aside from Suisun Bay) is notable.

The Lack of an Effects Analysis

The draft BDCP describes an effects analysis as:

> "the principal component of a habitat conservation plan. . . . The analysis includes the effects of the proposed project on covered species, including federally and state listed species, and other sensitive species potentially affected by the proposed project. The effects analysis is a systematic, scientific look at the potential impacts of a proposed project on those species and how those species would benefit from conservation actions." (draft BDCP, p. 5-2)

Clearly, such an effects analysis, which is in preparation, is intended to be the basis for the choice and details of those conservation actions. Its absence in the draft BDCP, therefore, is a critical gap in the science in the BDCP and the corresponding conservation actions. Nevertheless, the panel takes this opportunity

[2] BDCP (Bay Delta Conservation Plan Steering Committee). 2010. Bay Delta Conservation Plan Working Draft. November 18. Available online at: *http://www.resources.ca.gov/bdcp/*. Last accessed April 26, 2011.

to present its vision of a successful effects analysis, which includes an integrated description of the components of the system and how they relate to each other; a synthesis of the best available science; and a representation of the dynamic response of the system.

The term "effects analysis" also applies to an analysis of what is causing the listed (and other ecologically important) species to decline. In such a case, the logical sequence would be to perform the effects analysis on the causes of the species' declines, then design a proposed alternative to current operations to help reverse those declines, and then perform a second effects analysis on the probable effects of the proposed alternative. This aspect of an effects analysis is not mentioned in the current draft of the BDCP, and its absence brings the panel to a second critical gap in the scope of the draft BDCP, namely, a lack of clarity of the BDCP's purpose.

The Lack of Clarity as to the BDCP's Purpose

The legal framework underlying the BDCP is complex, as are the challenges of assembling such a large habitat conservation plan. Nonetheless, the BDCP's purpose or purposes need to be clearly stated, because their nature and interpretation are closely tied to the BDCP's scientific elements. The lack of clarity makes it difficult for this panel and the public to properly understand, interpret, and review the science that underlies the BDCP.

The central issue is to what extent the BDCP is only an application for a permit to incidentally take listed species, and to what extent it also is designed to achieve the two co-equal goals of providing for a more reliable water supply for the state of California and protecting, restoring, and enhancing the Delta ecosystem specified in recent California water legislation. To obtain an incidental take permit, it is logical to identify a proposed project or operation and design conservation methods to minimize and mitigate its adverse effects. But if the BDCP were largely a broader conservation program, designed to protect the ecosystem and provide a reliable water supply, then a more logical sequence would be to choose alternative projects or operating regimes only after the effects analysis was complete. Under that scenario, choosing the alternative first would be like putting the cart before the horse, or *post hoc* rationalization; in other words, choosing a solution before evaluating alternatives to reach a preferred outcome.

A related issue is the lack of consideration of alternatives to the preferred proposal (i.e., the isolated conveyance system). To the degree that the reasons for not considering alternatives have a scientific (as opposed to, for example, a financial) basis, their absence makes the BDCP's purpose less clear, and the panel's task more difficult.

THE USE OF SCIENCE AND SYNTHESIS IN THE BDCP

Many scientific efforts are and have been under way to understand and monitor hydrologic, geologic, and ecological interactions in the Delta, efforts that constitute the BDCP's scientific foundation. But overall it is not clear how the BDCP's authors synthesized the foundation material and systematically incorporated it into the decision-making process that led to the plan's conservation actions. For example, it is not clear how the Delta Regional Ecosystem Restoration Implementation Plan has been incorporated into the draft BDCP (see Appendix F of the draft BDCP). It also is not clear whether and how the draft BDCP incorporated the analyses for the Delta Risk Management Strategy and the framework developed by the Interagency Ecological Program related to factors affecting pelagic organism decline.

Furthermore, some of the scientific efforts related to the BDCP were incomplete at the time of this review. For example, warming, sea level rise, and changes in precipitation patterns and amounts will play a central role in Delta water allocation and its effects. Although the draft BDCP does mention incorporation of climate variability and change and model uncertainty, such information was not included in the draft BDCP that was provided.

Several other conservation efforts have been undertaken in the Delta in response to consultations with the National Marine Fisheries Service and the U.S. Fish and Wildlife Service concerning the potential for project operations (e.g., pumping) to jeopardize the listed species. The link between the BDCP and these other efforts is unclear. For example, the Delta Plan is a comprehensive conservation, restoration, and water-supply plan mandated in recent California legislation. That legislation also provided for potential linkage between the BDCP and the Delta Plan, but the draft BDCP does not make clear how this new relationship will be operationalized.

Much of the analysis of the factors affecting the decline of smelt and salmonids in the Delta has focused on water operations there, in particular, the pumping of water at the south end of the Delta for export to other regions. However, a variety of other significant environmental factors ("other stressors") have potentially large effects on the listed fishes. In addition, there remain considerable uncertainties surrounding the degree to which different aspects of flow management in the Delta, especially management of the salinity gradient, affect the survival of the listed fishes. Indeed, the significance and appropriate criteria for future environmental flow optimization have yet to be established, and are uncertain at best. The panel supports the concept of a quantitative evaluation of stressors, ideally using life-cycle models, as part of the BDCP.

The lack of clarity concerning the volume of water to be diverted is a major shortcoming of the BDCP. In addition, the BDCP provides little or no information about the reliability of supply for such a diversion or the different reliabilities associated with diversions of different volumes. It is nearly impossible to evaluate the BDCP without a clear specification of the volume(s) of water to be diverted, whose negative impacts the BDCP is intended to mitigate.

The draft BDCP is little more than a list of ecosystem restoration tactics and scientific efforts, with no clear over-arching strategy to tie them together or to implement them coherently to address mitigation of incidental take and achievement of the co-equal goals and ecosystem restoration. The relationships between scientific programs and efforts external to the BDCP and the BDCP itself are not clear. Furthermore scientific elements within the BDCP itself are not clearly related to each other. A systematic and comprehensive restoration plan needs a clearly stated strategic view of what each major scientific component of the plan is intended to accomplish and how this will be done. The separate scientific components should be linked, when relevant, and systematically incorporated into the BDCP. Also, a systematic and comprehensive plan should show how its (in this case, co-equal) goals are coordinated and integrated into a single resource plan and how this fits into and is coordinated with other conservation efforts in the Delta, for example, the broader Delta Plan.

ADAPTIVE MANAGEMENT

Numerous attempts have been made to develop and implement adaptive management strategies in environmental management, but many of them have not been successful, for a variety of reasons, including lack of resources; unwillingness of decision makers to admit to and embrace uncertainty; institutional, legal, and political preferences for known and predictable outcomes; the inherent uncertainty and variability of natural systems; the high cost of implementation; and the lack of clear mechanisms for incorporating scientific findings into decision making. Despite all of the above challenges, often there is no better option for implementing management regimes, and thus the panel concludes that the use of adaptive management is appropriate in the BDCP. However, the application of adaptive management to a large-scale problem like the one that exists in California's Bay-Delta will not be easy, quick, or inexpensive. The panel concludes that the BDCP needs to address these difficult problems and integrate conservation measures into the adaptive management strategy before there can be confidence in the adaptive management program. In addition, the above considerations emphasize the need for clear goals and integrated goals, which have not been provided by the draft BDCP. Although no adaptive management program can be fully described before it has begun, because such programs evolve as they are implemented, some aspects of the program could have been laid out more clearly than they have been.

Adaptive management requires a monitoring program to be in place. The draft BDCP does describe its plan for a monitoring program in considerable detail. However, given the lack of clarity of the BDCP's purpose and of any effects analysis, it is difficult to evaluate the motivation and purpose of the monitoring program. An effective monitoring program should be tied to the effects analysis, its purpose should be clear (e.g., to establish reference or baseline conditions, to detect trends, to serve as an early-warning system, to monitor management regimes for effectiveness), and it should include a mechanism for linking the information gained to operational decision making and to the monitoring itself. Those elements are not clearly described in the draft BDCP.

In 2009, the BDCP engaged a group of Independent Science Advisors to provide expertise on approaches to adaptive management. The panel concludes that the Independent Science Advisors provided a logical framework and guidance for the development and implementation of an appropriate adaptive management program for the BDCP. However, the draft BDCP lacks details to demonstrate that the adaptive management program is properly designed and follows the guidelines provided by the Independent Science Advisors. The panel further concludes that the BDCP developers could benefit significantly from adaptive management experiences in other large-scale ecosystem restoration efforts, such as the Comprehensive Everglades Restoration Program. The panel recognizes that no models exactly fit the Delta situation, but this should not prevent planners from using the best of watershed-restoration plans to develop an understandable, coherent, and data-based program to meet California's restoration and reliability goals. Even a soundly implemented adaptive management program is not a guarantee of achieving the BDCP's goals, however, because many factors outside the purview of the adaptive-management program may hinder restoration. However, a well-designed and implemented adaptive management program should make the BDCP's success more likely.

MANAGEMENT FRAGMENTATION AND A LACK OF COHERENCE

The absence of scientific synthesis in the draft BDCP draws attention to the fragmented system of management under which the plan was prepared—a management system that lacks coordination among entities and clear accountability. No one public agency, stakeholder group or individual has been made accountable for the coherence, thoroughness, and effectiveness of the final product. Rather, the plan appears to reflect the differing perspectives of federal, state, and local agencies, and the many stakeholder groups involved. Although this is not strictly a scientific issue, fragmented management is a significant impediment to the use and inclusion of coherent science in future iterations of the BDCP. Different science bears on the missions of the various public agencies, and different stakeholders put differing degrees of emphasis on specific pieces of science. Unless the management structure is made more coherent and unified, the final product may continue to suffer from a lack of integration in an attempt to satisfy all discrete interests and not, as a result, the larger public interests.

IN CONCLUSION

The panel finds the draft BDCP to be incomplete or unclear in a variety of ways and places. The plan is missing the type of structure usually associated with current planning methods in which the goals and objectives are specified, alternative measures for achieving the objectives are introduced and analyzed, and a course of action is identified based on analytical optimization of economic, social, and environmental factors. Yet the panel underscores the importance of a credible and a robust BDCP in addressing the various water management problems that beset the Delta. A stronger, more complete, and more scientifically credible BDCP that effectively integrates and utilizes science could indeed pave the way toward the next generation of solutions to California's chronic water problems.

1
Introduction

The San Francisco Bay Delta Estuary encompasses the deltas of the Sacramento and San Joaquin Rivers as well as the eastern margins of San Francisco Bay. Although the area has been extensively modified over the past 150 years, it remains biologically diverse while functioning as a central element in California's water supply system. The Delta system is subject to several forces of change, including seismicity, land subsidence, sea level rise, and changes in flow magnitudes as well as such societal changes as increased urbanization, population growth, growing water demands, and changing agricultural practices. These changes threaten the integrity of the Delta and its capacity to function both as an important link in the state's water supply system and as habitat for many species, some of which are threatened and endangered. In anticipation of the need to manage and respond to changes that have already and are likely to beset the Delta, a variety of planning activities have been undertaken. One such activity entails the development of a Bay Delta Conservation Plan (BDCP) by a consortium of federal, state, and local government agencies, environmental organizations, water supply entities, and other interested parties as a habitat conservation plan (see Appendix B). The BDCP covers 11 fish, 6 mammal, 12 bird, 2 reptile, 3 amphibian, 8 invertebrate, and 21 plant species (see Appendix C).

The present volume is the report of a panel appointed by the National Research Council at the request of the U.S. Secretaries of Interior and Commerce to review a working draft of the BDCP, dated November 18, 2010.[3] Specifically, the panel was charged with providing a short report assessing the adequacy of the use of science and adaptive management in the draft BDCP (see Appendix A). The panel met on December 8 and 10, 2010 in San Francisco, California. On the first day the panel heard presentations from the various authors and sponsors of the draft BDCP and commentary from interested stakeholders. The panel spent the remainder of the meeting time as well as the intervening weeks examining, evaluating, and analyzing the draft BDCP. In the course of this review, the panel delved into supporting documents such as the Delta Risk Management Strategy and other relevant documents. This report refers to and comments on those documents in the context of the BDCP; however, this report is not a review of those documents.

The use of science has been emphasized in recent legislation, and science is

[3] BDCP (Bay Delta Conservation Plan Steering Committee). 2010. Bay Delta Conservation Plan Working Draft. November 18. Available online at: *http://www.resources.ca.gov/bdcp/*. Last accessed April 26, 2011.

undoubtedly essential to the development of Delta plans generally. But science is only a starting point in the development of an integrated watershed-based plan, and it must be broadly applied. Moreover, science by itself cannot generate solutions to the myriad problems of the Delta that will satisfy the interests of all parties. Water scarcity in California is very real and science is not necessarily the sole solution to California's water problems. There is simply not enough water to serve all desired uses. The situation surrounding the Delta is a symptom of scarcity. The effective management of scarcity requires not only the best science and technology, but also consideration of public and private values, usually through political processes, to arrive at plans of action which are scientifically sound but also incorporate and reflect the mix of differing societal values.

This review contains a background section describing the geography, hydrology, and history of the Delta and more detailed explications of the points noted above. Then the discussion is organized according to: (1) critical gaps in the scope of the draft BDCP, (2) the use of science in the draft BDCP (3) adaptive management in the BDCP, and, (4) the fragmentation of management that appears to characterize the effort.

2
Background

The BDCP has been developed in an environment characterized by complexity and uncertainty. Furthermore, the BDCP context is dynamic, with underlying conditions themselves in flux. Complexity and uncertainty characterize the biophysical environment, including complexities and changes in the hydrologic system, such as interactions of altered freshwater discharge regimes of tidal influences, changes in the composition and numbers of many species, variability and changes in precipitation, nutrient and sediment input, and changes in the built environment. They also characterize the human environment, particularly with regard to population growth; people's livelihoods and lifestyles; political, financial, and economic conditions; changes in technology; and changes in people's understanding of these systems. Uncertainty is inherent in many of the above factors. The panel did not consider all of the above factors during its review because to do so would be difficult, time-consuming, and beyond the panel's charge. Nevertheless, recognition of the difficult environment in which the BDCP is being developed is helpful in gaining an understanding and appreciation of the difficulties surrounding it and other attempts to improve the reliability of water supplies in California and to restore the Delta ecosystem. The panel thus briefly summarizes the history and the human and biophysical environment of the region.

The San Francisco Bay Delta Estuary (Delta, for short) includes the lower reaches of the two most important rivers in California and the eastern estuary and associated waters of San Francisco Bay. The Sacramento and San Joaquin Rivers and their tributaries include all of the watersheds that drain to and from the great Central Valley of California's interior, as shown in Figure 2. The respective deltas of these rivers merge into a joint delta at the eastern margins of the San Francisco Bay estuary. The Delta proper is a maze of canals and waterways flowing around more than 60 islands that are protected by levees. The islands themselves were historically reclaimed from marshlands as agricultural lands, and most of them are still farmed.

Today, the Delta is among the most modified deltaic systems in the world (Kelley, 1989; Lund et al., 2010). The Sacramento-San Joaquin Delta, as shown in Figure 3, is an integral part of the water supply delivery system of California. Millions of acres of arid and semi-arid farm lands depend upon the Delta for supplies of irrigation water, and approximately 25 million Californians depend upon transport of water through the Delta for their urban water supplies. Population growth anticipated for the first half of the 21st century is likely to create additional water demands despite significant reductions in per capita consump-

BDCP1673

FIGURE 2. The Bay Delta Watershed. SOURCE: Reprinted, with permission, from National Resources Defense Council (*http://www.nrdc.org*).

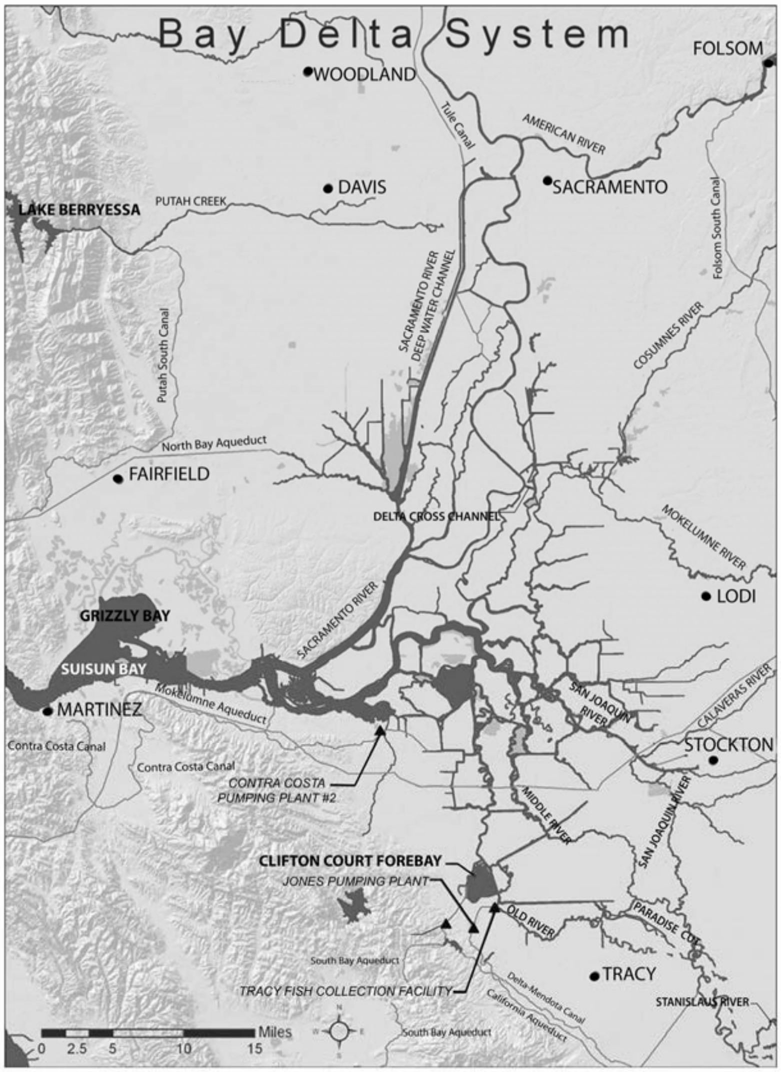


FIGURE 3. The Sacramento-San Joaquin Delta in California. San Francisco Bay, an integral part of the system, is just to the west. SOURCE: Reproduced from NRC (2010b), modified from FWS (2008).

tive uses. In addition, the Delta provides habitat for fish and wildlife, some speciesof which are listed as threatened or endangered under the federal Endangered Species Act and the California Endangered Species Act. The Delta is also an important recreational resource supporting significant boating and fishing activities.

Unimpaired inflows of water to the Delta originate in the watersheds of the Sacramento and San Joaquin Rivers. In an average year those flows are estimated to be 40.3 million acre feet (MAF) or 48.8% of California's average annual total water resource of approximately 82.5 MAF. Of the total unimpaired average inflow, 11.4 MAF are diverted upstream of the Delta for agricultural (83.8%), urban (15.0%), and environmental (1.2%) uses. Diversions from the Delta itself average 6.35 MAF, a little more than a third of all diversions in the Sacramento-San Joaquin system. Diversions from the Delta are dominated by exports to the irrigation service areas of the federal Central Valley Project (CVP) and the State Water Project (SWP) service areas, which include southern portions of the San Francisco Bay Area, the western side of the San Joaquin Valley, and much of southern California. Significant amounts of water are diverted to irrigate Delta lands, and irrigation return flow is discharged into Delta channels. The average yearly outflow from the Delta remaining after the diversions equals 22.55 MAF (Lund et al., 2010).

The quantities of water reported above are for an average year, but hardly any year in California is an "average" water year. Moreover, averages mask the fact that water supplies are highly variable from one year to another. Thus, for example, in the Merced River, which drains the watershed including most of Yosemite National Park and is a tributary of the San Joaquin River, the average annual flow is 1.0 MAF. Yet the low flow of record for the Merced River is 150,000 acre feet, only 15% of the average flow, while the high flow of record is 2.8 MAF, 280% of the average flow. The variability in flows, which is characteristic of all of the state's rivers, is largely a function of the interannual patterns of California's Mediterranean climate, which has a wet and a dry season with precipitation falling mainly in the late fall and winter months. In addition, there is considerable variability in the proportion of the precipitation that falls in the mountains as snow, which adds to the variability of the hydrologic regime.

Until recently, planning for water shortage was based on a five-year dry cycle from the 1930s, or on 1977, the driest year of record. However, recent analyses of potential precipitation resulting from different anticipated climate conditions have changed the criteria employed by the state to project water availability. Despite statewide conservation efforts, which are particularly pronounced in the urban sector, increasing restrictions on diversions have reduced the amount of water available for delivery under the terms of SWP and CVP water supply contracts. These projects, which export water to regions of the state that have experienced persistent water scarcity for many decades, are particularly important features of the California waterscape.

The CVP withdraws water from the Delta and conveys it southward into the San Joaquin Valley through a system of canals built and operated by the federal

Bureau of Reclamation and various water user groups. Most of this water is used for agricultural purposes throughout the San Joaquin Valley and the Tulare subbasin at the southern end of the Valley. A minor amount is contracted for domestic use. The SWP withdraws water separately from the Delta and conveys it southward to agricultural users on the west side and at the very southern end of the San Joaquin Valley and subsequently over the Tehachapi Mountains into the conurbation of the South Coast Basin. Los Angeles and San Diego are among the water users in the South Coast Basin. The SWP supplies domestic water users in southern California (and a minor amount of domestic use in the southern San Francisco Bay Area) as well as Central Valley agriculture in proportions that are determined in any given year by climatic factors and the availability of alternative sources of supply. Total available supplies have been constrained in recent years by drought and court decisions.

Changes in the hydrologic and physical integrity of the Delta would constrain and threaten the ability of state and federal water managers to continue exporting water in accustomed quantities through the two major projects. This is a concern because the structure of the Delta is changing and will continue to change. Lund et al. (2010) identify several factors that today pose significant threats to the Delta, including: (1) continued subsidence of the agricultural lands on the Delta islands; (2) changing inflows of water to the Delta, which appear to increase flow variability and may skew flows more in the direction of earlier times in the water year in the future; (3) sea level rise that has been occurring over the past 6,000 years and is expected to accelerate in the future; and (4) earthquakes, which threaten the physical integrity of the entire Delta system. There is a long history of efforts to solve these physical problems as well as persistent problems of flood control and water quality (salinity). Salinity intrusion from San Francisco Bay now requires a specific allocation of Delta inflows to repel salinity and to maintain low salinity water at the Delta's western margin. This is done by monitoring and managing the average position of the contour line identifying acceptable levels of salinity, known as "X2". Controlling salinity requires outflow releases from reservoirs that could be used for other demands.

Resolution of these problems is complicated by water scarcity generally and because alternative solutions impose differing degrees of scarcity on different groups of stakeholders. Additional allocation problems arise from a complex system of public and private water rights and contractual obligations to deliver water from the federal CVP and California's SWP. Some of these rights and obligations conflict, and in most years there is insufficient water to support all of them. This underscores the inadequacy of Delta water supplies to meet demands for various consumptive and instream uses as they continue to grow. Surplus water to support any new use or shortfalls in existing uses are unavailable, and any change in the Delta's hydrologic, ecological, or physical elements could reduce supplies further. The risks of change, which could be manifested either by increases in the already substantial intra-seasonal and intra-annual variability or through an absolute reduction in available supplies, underscore the existence of water scarcity and illustrate ways in which such scarcity could be intensified.

In its natural state, the Delta was a highly variable environment. The volume of water inflows changed dramatically from season to season and from year to year. Water quality also varied. In wet periods both salinity and chemical inputs (naturally occurring) were diluted. The species that occupied Delta habitats historically were adapted to accommodate variability in flow, quality, and all of the various factors that they help determine. The history of human development of the Delta, both of land use and water development, is a history of attempts to constrain this environmental variability, to reduce environmental uncertainty and to make the Delta landscape more suitable for farming and as a source of reliable water supplies. A full understanding of the historical pervasiveness and persistence of environmental variability underscores the need to employ adaptive management in devising future management regimes for the Delta (Healey et al., 2008).

The history of water development and conflict in California focuses in part on the Sacramento-San Joaquin Delta. Beginning with the California gold rush in 1848, early settlers sought to hold back the seasonal influx of water and to create agricultural lands. The construction of levees played a central role in this effort, which was threatened in the late 1800s and early 1900s by the movement of hundreds of millions of cubic yards of debris from upstream hydraulic mining that passed through the Delta. There followed throughout the first third of the 1900s further work that helped to stabilize a thriving Delta agriculture (Jackson and Patterson, 1977; Kelley, 1989). The CVP, which began operations in the 1940s (Thompson, 1957), and the SWP of the 1960s required conveyance of water from mainstream river channels through the channels and sloughs of the Delta to the extraction points located in the southern Delta from where water is pumped into the Delta-Mendota Canal (CVP) and the California Aqueduct (SWP) for transport south as illustrated in Figure 4. Once these projects became operational there was a need to control salinity, which became an issue that was decided by the courts (Hundley, 2001; Lund et al., 2010).

Since the beginning of CVP operations, diversions of water to users outside the Delta have been limited to ensure that salinity intrusion does not adversely affect local domestic water diverters in the western margins of the Delta. Additionally, the California's constitution requires that the waters of the state be put to "beneficial use," and this criterion is subject to judicial review and determination. The importance of environmental uses of water has been reflected in many state regulatory decisions and, more recently, in judicial interpretations of the federal Endangered Species Act and the California Endangered Species Act. Several species of Delta fishes and anadromous fishes that migrated through the Delta have been listed as threatened and endangered. The courts became involved, and specific operational restrictions followed from their findings. The maze of federal and state laws as well as the interests of dozens of stakeholder groups have combined to create a gridlock, which at times appeared penetrable

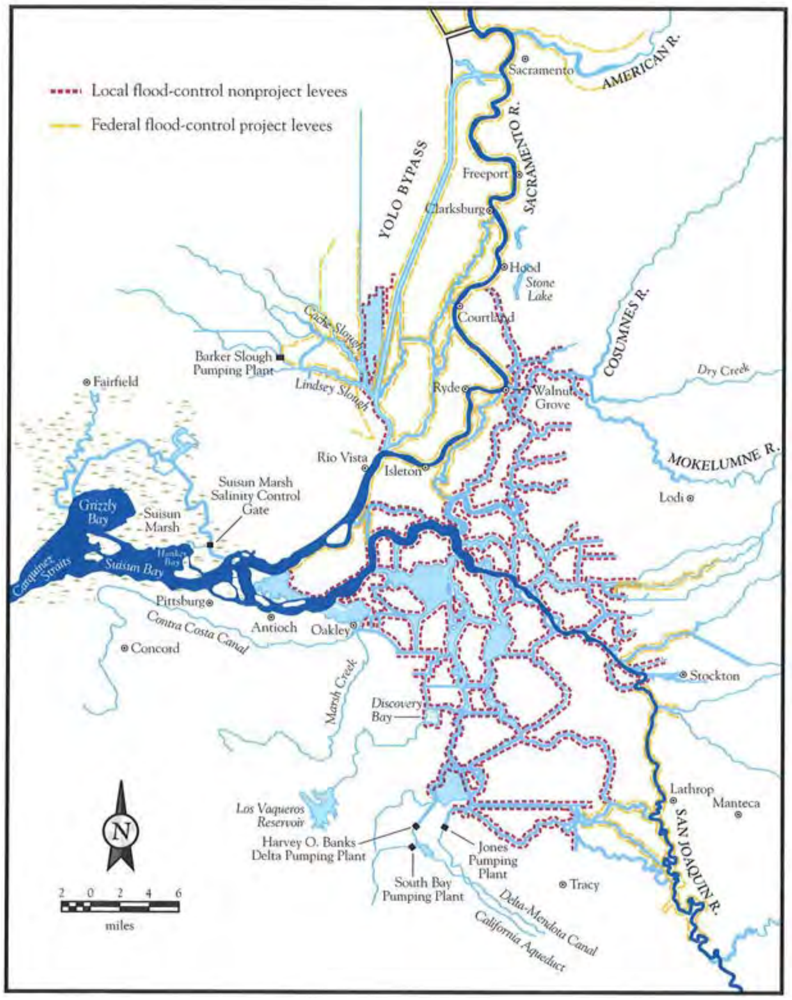


FIGURE 4. The Sacramento-San Joaquin Delta in California, highlighting the Delta levees, 2006. San Francisco Bay, an integral part of the system, is just to the west. SOURCE: Reprinted, with permission, from Lund et al. (2010). Copyright by Public Policy Institute of California.

only by the state and federal courts (Lund et al., 2010). As a result, most recent water operations have tended to be based on legislative requirements and judicial decisions mandating the protection of individual species rather than the optimization of water allocation among all purposes.

There have been several efforts to resolve differences, find areas of agreement, and identify solutions to the problems of the Delta and the operation and allocation of the waters that flow through it. These efforts assumed particular urgency as California was beset by severe droughts in the periods 1987-1992 and again in the first decade of the 2000s. A collaboration of 25 state and federal agencies called the CALFED program was created in 1994 with the mission "to improve California's water supply and the ecological health of the San Francisco Bay/Sacramento-San Joaquin River Delta" (*http://calwater.ca.gov/calfed/about/index.html*). State and federal agencies quickly developed a proposal for water quality standards titled Principles for Agreement on Bay-Delta Standards between the State of California and the Federal Government, otherwise known as the Bay Delta Accord. State and federal agencies with responsibilities in the Delta and stakeholders engaged in a decade-long CALFED process, but they did not alter the strategy of relying on the Delta to convey crucial elements of the water supply to California. The CALFED process also would be used to attain the four main goals of water supply reliability, water quality, ecosystem restoration, and enhancing the reliability of the Delta levees (CALFED, 2000).

The Bay-Delta Accord, which was signed in 2000, began to unravel mid-decade as environmentalists and water users came to believe that their interests were not being well served (Lund et al., 2010) and as federal resources declined. There followed an attempt by the governor to develop a Delta Vision Strategic Plan or "Delta Vision" with the aid of an independent Blue Ribbon Task Force. The Delta Stewardship Plan, which is referred to in this report as the Delta Plan, resulted from this effort. The Delta Plan is a broad umbrella plan mandated by the California Delta Protection Act of 2009 (California Water Code, 85300) to advance the co-equal goals of providing a more reliable water supply for California and protecting, restoring and enhancing the Delta ecosystem. The act requires development and implementation of the plan by January 2012 and specifies that a Delta Stewardship Council, whose membership must reflect broad California water interests, oversee the effort. Also beginning in mid-decade, federal, state, and local water agencies, state and federal fishery management agencies, environmental organizations, and other parties began work on the Bay Delta Conservation Plan (BDCP), a draft of which is the subject of the present report. In addition to the activities already mentioned, many other efforts are ongoing in the Delta such as, for example, a recent report of the State Water Resources Control Board on flows, recent biological opinions concerning listed species, The California Water Plan, The Recovery Plan for Central Valley Salmonids, and the Interim Federal Action Plan.

The BDCP is a habitat conservation plan that can be incorporated into the Delta Plan described above if specific criteria specified in California's water legislation are met (draft BDCP, pp. 1-6). The organizations involved in the

BDCP process have formed a steering committee that includes representatives from the various agencies and interest groups involved in the collaboration (see Appendix B). The BDCP planning effort began in 2006 with a completion goal of 2013. The completed plan also is intended to be implemented over the next 50 years (*http://baydeltaconservationplan.com/Home.aspx*). As of November 22, 2010 close to $150 million has been spent in developing the plan (Sagouspe, 2010).

The BDCP is to be supported by the Environmental Impact Report (EIR)/Environmental Impact Statement (EIS) that will evaluate the range of alternatives for providing ecosystem restoration, water conveyance and other management alternatives identified in the BDCP. The EIR/EIS is currently being prepared by the California Department of Water Resources, the U.S. Bureau of Reclamation, the U.S. Fish and Wildlife Service, and the National Oceanic and Atmospheric Administration's National Marine Fisheries Service in cooperation with California's Department of Fish and Game, the U.S. Environmental Protection Agency, and the U.S. Army Corps of Engineers (*http://baydeltaconservationplan.com/Home.aspx*).

The subsequent sections of this report describe and analyze prominent features of the BDCP while identifying and discussing the critical gaps in the document.

3
Critical Gaps in the Scope of the Draft BDCP

The panel concludes that the draft BDCP is missing critical elements, including an effects analysis, a description of how and where scientific information was used in the draft BDCP, and a description of the BDCP's relationship to other ongoing efforts. In addition, the draft has several structural or systematic problems, including lack of clarity as to the purpose of the BDCP; an unclear linkage of various parts of the BDCP to the effects analysis[4] and among its other components; and lack of detail about analyses of various future scenarios, including a lack of analyses of tradeoffs among the BDCP's goals in various scenarios. The panel offers some guidance on how these systematic problems might be addressed and how the draft BDCP might be completed more usefully.

At the outset of its review the panel identified a problem with the geographical and hydrologic scope of the draft BDCP. The BDCP aims to address management and restoration of the San Francisco Bay Delta Estuary, an estuary that extends from the Central Valley to the mouth of San Francisco Bay. Thus, given that the BDCP purports to describe a *Bay* Delta Conservation Plan, the omission of analyses of the effects of the BDCP efforts on San Francisco Bay (aside from Suisun Bay) is notable. This omission should be of concern to all BDCP parties because the Bay-Delta system is an estuary, and there are significant physical, biogeochemical, and ecological connections between the various sub-embayments as well as between the Bay-Delta and the Pacific Ocean (e.g., Cloern et al., 2010). In particular, changes in outflows and in the tidal prism associated with changing water-project operations and restoration actions would be expected to cause changes in San Francisco Bay, and not only in the Delta. A plan intended to be comprehensive should incorporate these fundamental features of the system. Although the statutory basis of the BDCP may argue against consideration of the effects outside the statutory Delta, the BDCP's failure to address issues related to San Francisco Bay is a significant flaw that should be corrected in subsequent versions of the plan.

[4] Even though the effects analysis is not yet complete, the BDCP's authors should at least be able to describe how the completed parts of the BDCP will be linked to the effects analysis.

THE LACK OF AN EFFECTS ANALYSIS

The draft BDCP describes an effects analysis as:

> "the principal component of a habitat conservation plan [HCP]. . . . The analysis includes the effects of the proposed project on covered species, including federally and state listed species, and other sensitive species potentially affected by the proposed project. The effects analysis is a systematic, scientific look at the potential impacts of a proposed project on these species and how these species would benefit from conservation actions" (draft BDCP p. 5-2).

Clearly, such an effects analysis, which is in preparation, is intended to be the basis for the choice and details of those conservation actions. Its absence in the BDCP, therefore, is critical gap in the scope of the science and the conservation actions. Nevertheless, the panel presents its vision of the structure and content of a useful effects analysis.

The above description of the effects analysis to be included in the BDCP is rather narrowly cast, because it focuses on the BDCP as a habitat conservation plan (HCP), that is, as an application for an incidental take permit. It thus *presupposes* the choice of the project to be permitted. By contrast, a broadly focused conservation strategy, which the draft BDCP also says it is[5], requires a similarly broadly focused, comprehensive effects analysis. Such an effects analysis would include a systematic analysis of the factors affecting species and ecosystems of concern and the likely contribution of human-caused changes in the system. Such an analysis would then lead to the informed choice of options for reversing the decline of the ecosystem and its components, rather than only analyzing a pre-chosen option. What would such an effects analysis look like?

Effects analyses are used in a range of disciplines to understand complex systems. As noted in the quote above, their main attribute is that they are systematic scientific analysis. Their precise form is not critical. For example, failure mode and effects analysis (FMEA) is commonly applied in the automotive, aerospace, and software industries to understand whether and how the failure of individual components impact the reliability of the overall system (Gilchrest, 1993; McDermott et al., 2009). In the environmental field, effects analyses are used to understand and compare likely responses to alternative management schemes (e.g., Marcot et al., 2001). The National Research Council has reviewed the application of effects analysis within the environmental arena (NRC, 2009). In addition, several NRC reports have discussed or applied the techniques of effects analysis even though they were not necessarily called "effect

[5] The following statement appears on p. 1-1 of the draft BDCP: "The [BDCP] sets out a comprehensive conservation strategy for the Delta designed to advance the co-equal planning goals of restoring ecological functions of the Delta and improving water supply reliability to large portions of the state of California."

analysis" (e.g., NRC, 1995, 2002, 2004a, 2004b, 2005; Appendix E of this report provides an example of an effects analysis from NRC 2004b). Effects analyses are commonly used because they integrate empirical data and expert opinion to guide management decisions (e.g., NRC, 2004b). The analytical approaches used in the different types of effects analyses vary from classical risk priority numbers, to simulation modeling (e.g., Legault, 2005), to complex Bayesian network models (Ellison, 1996; Uusitalo, 2007). However, certain important elements are common to all of these analyses, including the need to describe how individual components in the system are connected. It is an effects analysis of this scope that the panel envisions for the BDCP. Here, the panel provides guidance regarding the structure and essential elements that it would expect to see in the completed effects analysis for the draft BDCP. The panel draws on a recent paper by Murphy and Weiland (2011) for a description of a useful effects analysis, itself based to some degree on NRC (2009), because it sets forth specifics for an effects analysis that would be appropriate for the Delta. The panel agrees with Murphy's and Weiland's general approach.

An effects analysis is an essential element of the final BDCP, because it will help meet the legal requirement for a habitat conservation plan to evaluate whether the preferred action aids in the recovery of the species (state requirement) and does not appreciably reduce the likelihood or the survival and recovery of the listed species in the wild (federal requirement). These requirements are initially triggered because as an HCP/NCCP (natural communities conservation plan), the BDCP deals with listed species. However, even if this were not the case, an effects analysis provides the framework within which the impacts of alternative management options can be compared and thus could be justified from a purely logical point of view. An effects analysis is further justified because it also may inform the adaptive management process by identifying which components or processes are the most sensitive indicators of the status and structure of the ecosystem (McCann et al., 2006).

Once the goal of the effects analysis has been defined, the first element of any effects analysis must be an integrated description of the components of the system and how they relate to one another. This description should include a clear statement of the alternative management actions proposed, including that of no action. The activities in this first section naturally lead to a clear definition of the management goal and the temporal and spatial domain of the impacted area. At this introductory level, it is not necessary to quantify the relationships. One needs to mainly indicate the connections. Such a description is essential for several reasons. Most important, it formalizes the understanding of the connections among processes and components in the system. It defines which processes and components are expected to respond to any perturbation and which ones will not. Secondarily, in formulating the problem, a conceptual diagram can serve to identify and rank in importance data on different processes and components within the system. Finally, the system description provides a broader context into which information on the status and trends of species covered by state and federal statues can be placed—such that the dependencies of these listed species on processes and components of the system are identified.

The second stage of the effects analysis should be the collection, review and critical assessment of the best relevant scientific information available. The determination of which data need to be assembled is guided largely by the conceptual framework identified in the first stage. It is neither necessary nor helpful for the assembled data to be encyclopedic in coverage. However, it is essential that data on those processes and components identified in the first stage are compiled, assessed and summarized. This information may be in the form of empirical data or in instances where data are unavailable, in the form of expert opinion. Expert and stakeholder opinion has been successfully used in several management questions involving water use or fish stocks (Borsuk et al., 2001; Miller et al., 2010). The objectives of the data assembly phase are to clearly describe the baseline or reference condition[6] and to quantify the expected relationships among system processes and components. An important feature of this stage is the need to include information on the uncertainties around estimates of processes or component levels. Additionally, the spatial and temporal scale of processes and components under consideration are a vital concern. Different processes and components likely respond at characteristic spatial and temporal scales. For example, the response of many chemical or physical variables might scale with the residence time of water in the system, whereas the response of biological variables might scale with the generation time of the organisms involved. Similarly, salinity gradients affect much of the central and western Delta, while some organisms like salmon, which spend a portion of their life cycles in sea water, occupy much of the North Pacific as well as the Delta and its tributaries. Within the biological realm, rates of primary production, nutrient and oxygen cycling, as well as microbial growth may respond rapidly to ecosystem conditions whereas the abundance of long-lived animals such as sturgeon is expected to integrate ecosystem dynamics over extended periods. The Comprehensive Everglades Restoration Plan (CERP) provides a good example of the use of measurable outcomes for these purposes (NRC, 2008, 2010c).

The next stage of the effects analysis is the most challenging–that of representing the dynamic response of the system. For simple systems, this may be in the form of a simple model. For example, decisions regarding quota levels in fisheries management are often made with guidance from a single assessment model, albeit one with hundreds of parameters (Miller et al., 2010). However, even in simple systems, the level of uncertainty present in individual processes and components of the system may be of such magnitude that state-variable models are unreliable. In these cases probabilistic models have been developed (Legault, 2005). More recently, Bayesian approaches have been used to guide management in the face of uncertainty for complex environmental questions

[6] Large restoration programs usually include methods for assessing their effects so that adaptive management can occur. The basic prerequisite for such assessments is the establishment and characterization of a reference condition against which future conditions and proposed alternatives can be compared.

(Borsuk et al., 2004; McCann et al., 2006; Rieman et al., 2001). For an example of incorporating uncertainty into management options, see Box 1.

In the case of the BDCP, it is unlikely that a single analytical framework, even one as flexible as Bayesian network analysis, will be adequate. Thus, it is likely that multiple models will be used to assess the response of different system components to each management alternative. Ultimately a range of integrated scenarios should be developed that link the models' outputs to an integrated response. It is particularly important that each set of the models and analyses be clearly related back to the original conceptual framework generated in the first stage of the effects analysis. Analysts should be explicit about the model inputs and assumptions for each stage of the process. One of the risks of this approach is error propagation, that is, that uncertainty inherent in the forecasts made for one component are not fully carried forward to models of other components.

It would be highly advantageous if outcomes in the effects analysis were quantifiable empirically and could thus become components of the BDCP's Monitoring and Evaluation Program (e.g., NRC, 2000, 2008; Orians and Policansky 2009). As noted above, the CERP has considered and described these issues in considerable detail (NRC, 2008 and references therein). This information, when gathered in the BDCP's Monitoring and Evaluation Program, could then be used to conduct statistical analyses and calibrate models and the modeling framework to inform the adaptive management phase over the decades following implementation of the BDCP actions.

BOX 1
The 2008 Federal Columbia River Power System Biological Opinion

A suitable example of an attempt to incorporate uncertainty is evidenced in the 2008 Federal Columbia River Power System (FCRPS) Biological Opinion (NOAA, 2008) and in the 2010 Supplemental FCRPS Biological Opinion (NOAA, 2010) prepared after the 2008 opinion was voluntarily remanded. The comprehensive analysis in this biological opinion focused on determining the effects of different dam operation alternatives, on key ESA-listed anadromous salmonid populations in the Columbia River Basin. In that analysis, water delivery and dam operation models create conditions that route juvenile salmon through different routes at eight dams in the FCRPS, resulting in net smolt survival downstream of the last dam (Bonneville). Changes in smolt system survival associated with different operation-alternatives are then linked to a broader life-cycle analysis to assess the potential for population level responses to selected management actions.

During the meeting on December 8, 2010, in San Francisco, presenters indicated that the effects analysis that will be included in the BDCP will be only a first step, that is, that it would be iteratively updated as empirical data from the operation of the approved alternatives become available. This approach is certainly compatible with the use of the effects analysis framework as the foundation of the adaptive management framework. If this is indeed how the BDCP developers intend to use the effects analysis, the panel recommends that the final version of the plan articulate a clear vision of how the effects analysis will be updated and how these results will be used to generate the ranges that will be the foundation for subsequent adaptive management.

As an example, much of the recent discussion of changes in the Delta ecosystem has focused on declining planktonic primary production in the Delta and Northern San Francisco Bay (Jassby et al., 2002) as driving food-web changes, notably declines in planktonic grazers (secondary producers), that may underlie to some extent the decline of pelagic fish species like delta and longfin smelt (Baxter et al., 2008). Accordingly, significant elements of the BDCP involve efforts to enhance primary and secondary production through creation of additional tidal wetlands mostly around the edges of the Delta, a plan that strongly echoes CALFED's earlier focus on the creation of shallow water habitat (c.f. Brown, 2003). The bases for this strategy are twofold: (1) in the face of light limitations, shallow water habitats for which the photic zone is a greater fraction of the water column should have higher rates of primary production than deeper waters, e.g., channels (Cloern, 2007); and (2) empirically it is observed that the periodically flooded shallow waters of the Yolo Bypass can support high rates of export of phytoplankton biomass (Schemel et al., 2004).

However, if an effects analysis is indeed "the principal component of a habitat conservation plan" (draft BDCP p. 5-2), then it is difficult to see how these and other conservation strategies described in the BDCP can be scientifically justified before the effects analysis is completed.

THE LACK OF CLARITY AS TO THE BDCP'S PURPOSE

The legal framework underlying the BDCP is extraordinarily complex. In attempting to comply with all relevant laws and regulations, the BDCP's authors have undertaken to develop a habitat conservation plan of great importance, scope, and difficulty. The panel recognizes that the authors face significant challenges and that the BDCP is a work in progress. With these caveats in mind, the panel observes that it would be helpful for the draft BDCP to clarify and place into context a number of legal issues, because their nature and interpretation are closely tied to the BDCP's scientific elements. Any lack of legal clarity makes it difficult for the panel and the public to properly understand, interpret, and review the science of the BDCP.

Ambiguous Role of Co-Equal Goals and Their Relationship to the BDCP

According to the draft BDCP (p. 1-8), it:

> "has been prepared as a joint [habitat conservation plan] HCP/ [Natural Communities Conservation Plan] NCCP, which will support the issuance of incidental take authorizations from the US [Fish and Wildlife Service] FWS and [National Marine Fisheries Service] NMFS pursuant to Section 10 of the [federal Endangered Species Act] ESA and take authorizations from the California Department of Fish and Game (DFG) under Section 2835 of the [Natural Communities Conservation Planning Act] NCCPA to the non-federal applicants. The BDCP has also been designed to meet the standards of Section 2081 of the California Endangered Species Act (CESA). The BDCP will further provide the basis for biological assessments (BA) to support the issuance of incidental take authorizations from USFWS and NMFS to [the Bureau of] Reclamation pursuant to Section 7 of the ESA, for its actions in the Delta."

Thus, the BDCP is clearly and specifically an application for the incidental take of listed species as set forth in federal and state statutes.

To apply for an exemption from the § 9 "take"[7] prohibition of the federal Endangered Species Act (ESA), the water users must submit a habitat conservation plan (here, the BDCP) that will minimize and mitigate the harmful impacts of their water usage. HCPs prepared as part of an application for an incidental take permit under federal law are not required to help listed species recover, but they must demonstrate that "the taking will not appreciably reduce the likelihood of the survival and recovery of the species in the wild" (ESA § 10).[8] Under state law, the water users must submit a Natural Community Conservation Plan (NCCP) that, among other things, "aids in the recovery of the species." (Natural Communities Conservation Planning Act [NCCPA], Cal. Fish and Game Code §§ 2800-2835). Neither the ESA nor the NCCPA specifically requires applicants to advance the "co-equal goals."

Despite this, the first paragraph of the draft BDCP (p. 1-1) states that it "sets out a comprehensive conservation strategy for the Delta designed to ad-

---

[7] *Take* means "to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct." ESA, Section 3, 16 U.S.C. 1532.
Harm, within the statutory definition of "take" has been further defined by regulation: "Harm in the definition of take in the Act means an act which actually kills or injures wildlife. Such act may include significant habitat modification or degradation where it actually kills or injures wildlife by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering." 50 C.F.R. 17.3.

[8] ESA § 10 also requires successful applicants to demonstrate that (1) " the taking will be incidental [to an otherwise lawful activity]," (2) "the applicant will, to the maximum extent practicable, minimize and mitigate the impacts of such taking," (3) "the applicant will ensure that adequate funding for the plan will be provided," and (4) "[such other measures that the Secretary may require as being necessary or appropriate for purposes of the plan] will be met." 16 USC § 1539(a)(2)(B).

vance the co-equal planning goals of restoring ecological functions of the Delta and improving water supply reliability to large portions of the state of California." This and similar statements throughout the plan make it difficult to understand and evaluate the purposes of HCPs and NCCPs, and the methods of implementing them. Moreover, the methods of implementation are considerably different from the purposes and methods for achieving the two co-equal goals specified in California statutes. Indeed, California has begun to develop a broader "Delta Plan" in accordance with a recent state statute (Cal. Water Code §§ 85300-85309). Thus, the question arises as to the degree of importance to the BDCP of its purpose as an HCP/NCCP and of its purpose as a broader conservation plan designed to achieve California's two co-equal goals. The BDCP and the Delta Plan address the same ecosystem and are somewhat overlapping, but their goals and legal requirements are not identical. Unless the BDCP's relationship to the Delta Plan is clearly described, and its purposes clearly delineated, it will be difficult to assess the BDCP's underlying scientific basis, because the purposes of a broad conservation plan like the Delta Plan are not necessarily the same as those of a habitat conservation plan.

The body of the BDCP contains some elements of both purposes, but not in a coherent and consistent way. For example, despite the statement that achieving the two co-equal goals is one of its purposes, the BDCP focuses on one of the goals at the expense of the other. Additional sources of the confusion are multiple, but two stand out. First, the BDCP document lists some eight planning goals of which providing a "basis for permits necessary to lawfully take covered species" is only one of these eight goals (draft BDCP, p. 1-6). Yet, the remainder of the BDCP appears to focus disproportionately on this goal. As such, much of the BDCP appears to be a post-hoc rationalization of the water supply elements contained in the BDCP.

A consequence of the lack of clarity is related to this post-hoc rationalization. To the extent that the BDCP is simply a request for an incidental take permit then the water users would first identify their desired action (such as construction of a specifically configured "alternative conveyance"), and then analyze its impacts and to develop measures to minimize and mitigate adverse effects. However, to the extent that the BDCP seeks incorporation into the broader Delta Plan, then an effects analysis would precede the choice of all conservation and alternative-operation options, and only then would an effects analysis of those options be performed. That is, if the proposed conveyance system and other measures such as wetlands restoration have been developed as measures to further the restoration of the Delta ecosystem, then one would expect that the effects analysis would be completed before coming to a conclusion as to the preferred type of water delivery system. The absence of an effects analysis and of consideration of water supply alternatives (other than the 45 mile tunnel or possibly an open canal; see section below on alternatives) suggests that the BDCP's major purpose is to provide the basis for an application for an incidental take permit. Yet, this is contrary to what is stated throughout the plan with respect to the attainment of co-equal goals.

Despite these ambiguities, the draft BDCP has concluded that an "isolated conveyance facility" should be constructed consisting of a 45-mile tunnel or pipeline, capable of conveying 15,000 cubic feet per second (cfs) of Sacramento River water around the Delta to the south Delta's existing water export pumping plants, to allow for "dual operation" with the existing south Delta diversion facilities (draft BDCP, Chapter 4.2.2.1.1 and Table 4-1). (Again, the "note to reviewers" on p. 4-14 of the draft BDCP suggests that the conveyance system might be a canal, but there is no analysis of a canal in the draft BDCP or even a statement as to whether the findings from the analysis of a canal would differ from the analysis of a tunnel system.)

Alternative Actions

To support the issuance of an ESA § 10 take permit, the BDCP must specify "what alternative actions to such taking the applicants considered and the reasons why such alternatives are not being utilized" (ESA § 10, 16 U.S.C. § 1539(a)(2)(A)). Even if the proposed action has been decided on, an analysis of alternatives is still required. This analysis does not appear prominently in the draft BDCP. Not only is the analysis a legal requirement, but it also is important scientifically, because to the degree that the reasons for not utilizing the alternatives are *scientific* reasons, the absence of the analysis hinders the ability to evaluate the BDCP's use of science. If the BDCP also seeks incorporation into the Delta Plan (and thereby qualifying for state funding of public benefits), then it should also include an analysis of "conveyance" alternatives. As a prerequisite to incorporation, the BDCP must undertake "a comprehensive review and analysis of . . . [a] reasonable range of Delta conveyance alternatives, including through-Delta, dual conveyance, and isolated conveyance alternatives and including further capacity and design options of a lined canal, an unlined canal, and pipelines" (Cal. Water Code, § 85320). Finally, the federal approval process also will require an environmental impact statement that considers alternatives to the "proposed action," which includes construction of the alternative conveyance (National Environmental Policy Act, 42 U.S.C. § 4332(2)(C)(iii)). Once again, this legally required analysis of alternatives is scientifically important. Therefore, to permit a complete scientific evaluation of the BDCP, it should include an analysis of such alternatives to "take" and to the construction and design of the contemplated isolated conveyance.

4
Use of Science in the BDCP

The panel recognizes the body of scientific information available to support some actions within the BDCP. For example, the compilation of the Delta Regional Ecosystem Restoration Plan (DRERIP, see Appendix F in the draft BDCP) demonstrates that the community has invested considerable effort in establishing a scientific foundation for the numerous actions proposed in the draft BDCP. The participation of 50 analysts and scientists in the construction and scoring of the scientific evaluation worksheets indicates the large effort devoted to identifying ecologically founded actions. The massive DRERIP reflects the collective wisdom and insight of the region's most knowledgeable and respected scientists.

However, it is not clear how the BDCP's authors synthesized the foundation material and systematically incorporated it into the decision-making process that led to the suite of actions selected for implementation. As a unit, the draft BDCP combines a catalog of overwhelming detail with qualitative analyses of many separate actions that often appear disconnected and poorly integrated. Thus, although the biological descriptions and scenarios reflect a strong understanding of the scientific basis for many individual actions by the BDCP authors, there is no obvious distillation, synthesis and integration of the material into a cohesive decision-making process. The BDCP's authors may have performed this critical exercise, but it is not described in the BDCP itself. The panel expects that the pending and critical effects analysis document could provide that convincing clarifying synthesis, relying on the DRERIP to provide the grist. Importantly, the participants who contributed to the DRERIP identified many uncertainties and deficiencies that need to be addressed by the community. Addressing these concerns presumably should happen before the plan is accepted as an ecologically sound path. The following excerpt from the DRERIP emphasizes these points:

> "Collectively, the synthesis team concluded that a number of the conservation measures have the potential for additional synergistic effects that can raise or lower the value of some individual conservation measures when implemented concurrently with other actions. The complexity of various trade-offs between expected positive and negative effects make it difficult to predict the biological responses to concurrent multiple measures. The Synthesis Team recommended that refinements could be made to the proposed modification of the Fremont Weir and Yolo Bypass inundation, North Delta diversions with bypass criteria, and Cache slough restoration to op-

> timize ecological benefits and water supply goals. They also identified the need for better information and modeling of the survival and growth of covered species and predators to establish baseline conditions against which benefits can be assessed…" (DRERIP, see Appendix F-1 of the draft BDCP, p. 17).

This is just one example of the strong body of scientific information that is available to support specific actions within the plan. Nevertheless, there is a deficiency in the scientific synthesis that is needed to support the collective actions specified in the BDCP. Some examples of opportunities for demonstrating that scientific synthesis are described below.

INCORPORATING RISK ANALYSIS

The analyses for the Delta Risk Management Strategy (DRMS, 2009) have been performed to better understand the various risks to the integrity of levees and the local and statewide consequences of levee failure. Although there are limitations to this analysis, the results can offer guidance for prioritizing actions within the BDCP. For example, the DRMS study indicates that the benefits of the restorative conservation measures could be lost if levees failed and concludes that current levee management strategies in the Delta are unsustainable because of seismic risk, high water conditions, sea level rise and land subsidence. In addition to these broad conclusions, the report offers specific estimates of land impacts (e.g., economic costs of more than $15 billion due to earthquake-derived levee failures and associated flooding of 20 islands) (DRMS, 2009).

California continues to invest in levee restoration, and additional restoration is included in the BDCP. However, levee repairs are not prioritized with regard to objectives such as habitat restoration, salinity management, drinking water protection, and preserving agriculture and historic Delta communities. Thus, any effects analysis should explicitly consider the interactions and tradeoffs between infrastructure and ecosystem goals. These interactions and tradeoffs may be considered in a risk-based framework, which could be complemented by analysis of the system reliability (the likelihood that a hydrosystem will fail to achieve some target), resilience (the ability of a system to accommodate, survive, and recover from unanticipated perturbation or disturbance), and vulnerability (the severity of the consequences of failure) (Fiering, 1982; Hashimoto, 1982; Moyle et al., 1986).

Furthermore, decision frameworks have recently been demonstrated in the Delta that highlight the economic tradeoffs of levee repair against the value of land and assets protected by those levees (Suddeth et al., 2010). The results suggest that, even with doubling of property values, repair of levees is not economically justifiable for most of the islands within the Delta's Primary Zone. Although decisions regarding levees, habitat, land use, and water allocations will certainly be based on more than economic motivations, the use of existing decision analysis tools, and development of new ones to address specific needs, may be invaluable in justifying prioritization of actions and geographical areas of emphasis within and outside of the Delta.

In developing such risk-based approaches, BDCP partners may also identify unacceptable outcomes and evaluate their likelihood, a task that would be valuable in comparing the ability of various management strategies to reduce the likelihood of hydrosystem deterioration, as has been suggested for climate change adaptation (NRC, 2010a). Therefore, the panel recommends that the BDCP partners select and apply a formal analytical framework to investigate the outcomes of proposed activities, including quantitative projections and existing science. Such an analysis—the effects analysis described in some detail above—should occur in advance of selecting the conservation and management actions, and should link specific restoration goals and undesirable system outcomes to the costs, benefits, and reliability of the proposed actions. To do so will require use of the extensive science developed in the basin, recognizing the limitations of its application and the implications of scientific uncertainty in prioritizing actions.

INTEGRATION OF CLIMATE CHANGE ANALYSIS

Climate change has been and will continue to be a major driver of hydrologic and landscape changes in the Delta. Projected changes in the primary drivers of climate change—namely rising temperatures, changing patterns of precipitation, and sea level rise—are expected to result in significant impacts to the ecosystems of both the Delta region and its tributary watersheds and will adversely impact the water supplies that are critical to both urban and agricultural users who depend on the Delta, the major reservoirs and the water conveyance systems (Chung et al., 2009). Therefore, climate change could pose significant threats to the success of the BDCP's ecological goals and could increase the need for additional conservation measures such as construction of additional surface and aquifer storage facilities, demand management such as conservation programs and pricing, and changes in operating strategies (Lund et al., 2010), and it could affect economic factors and water operations (for example, Tanaka et al., 2008).

California's climate change research has generated a wealth of information (Franco et al., 2008), which indicates potential impacts of climate change in the Delta region (e.g., Cayan et al., 2000; Climate Action Team, 2010; DWR, 2010; Field et al., 1999). The work to date has included a systems approach to understanding natural variability including (1) the potential global interconnections to the region's climate (Gershunov et al., 2000; Redmond and Koch, 1991); (2) detection and attribution of historical change in climate (Bonfils et al., 2008); (3) quantification of potential changes in primary stressors of climate through analyses of general circulation model (GCM) predictions (Cayan et al., 2009) and statistical downscaling (Hidalgo et al., 2008; Maurer and Hidalgo, 2008); (4) impacts of projected sea level rise (Knowles, 2008) and effects of rising temperatures on Delta water temperatures (Wagner et al., 2011); and (5) the sensitivity of the water resource system to climate change and sea level rise (USBR, 2008).

Although significant research on climate change vulnerabilities exists in the literature and in various reports produced by numerous agencies and institutions, the panel could not find evidence that such information has been used effectively in the development of the BDCP. Climate change analysis is legally required to obtain an incidental take permit, per NRDC vs. Kempthorne, 506 F.Supp.2d 322 (E.D. Cal. 2007). Yet the draft BDCP's treatment of the topic of climate change, including warming and sea level rise, is fragmented: climate change is addressed only in the descriptions of existing biological conditions (Chapter 2), and sparsely in the Conservation Strategy section (Chapter 3). Furthermore, these discussions are limited to qualitative assessment of potential vulnerabilities and how the conservation strategy might be able to accommodate such impacts. The panel could not find a quantitative analysis of the specific hydrological and biological consequences of potential changes in the primary drivers and consequent changes in the tributary watersheds, aquifers, demands, risks of levee failure, and ecology of the BDCP plan area. Neither could the panel find a statement indicating that such analyses are not available or feasible at this scale. In spite of the brief quantitative summary of potential changes described in Section 2.3.3.2 (pp. 2-36-2-37), there is no evidence that such estimates have been incorporated into the effects analysis and the design of conservation strategy elements. Chapter 5 of the draft BDCP (p. 5-3) says:

> "The effects of climate change (e.g., sea level rise, temperature, and hydrology) were evaluated for early and late points in time of BDCP implementation based on climate change scenarios developed by the consultant team, technical staff from the lead agencies, and outside climate change experts (see Appendix K, *Climate Change Evaluation Methods*, for a discussion of this analysis),"

which appears to address some of the panel's concerns. However, such information was not included in the draft BDCP that was provided.

In the presentation ("Incorporating Climate variability, Change, and Model Uncertainty in Scenarios for California Water Planning") to the panel during its open session on December 8, 2010, Armin Munevar, a consultant from the firm CH2M HILL, did include the aforementioned analysis. A summary of this work appears in a December 2010 report entitled *Climate Change Characterization and Analysis in California Water Resources Planning Studies* (DWR, 2010, pp. 58-67). The climate change study of the BDCP is summarized in the above report and constitutes a reasonable approach for incorporating the current information regarding future climate projections, as predicted by the climate models, and the corresponding hydrologic impacts. Recognizing that precipitation projections are more uncertain (p. 2-36, draft BDCP) than temperature projections, the BDCP's approach includes five scenarios: (1) drier, less warming; (2) drier, more warming; (3) wetter, more warming; (4) wetter, less warming; and (5) a central tendency scenario, which aggregates the majority of model projections (DWR, 2010, p. 62). A further addition to this approach is the concept of the "nearest neighbor" method to select subgroups of models that represent the above five scenarios. Groups of GCM predictions and the corresponding down-scaled information demonstrate a significant spread in both precipitation and temperature, and the above approach of using five scenarios to select a set of model runs bracketing the potential changes in precipitation and temperature appears to be adequate until better methods become available.

The above scenarios for climate change and sea level rise have been combined with a variety of hydrologic, operational, and hydrodynamic models to investigate the performance of numerous BDCP scenarios with respect to such metrics as changes in the timing and magnitude of watershed run-off, reservoir storage, flows in the southern part of the Delta, and seasonal variations in the salinity gradient (the position of X2). This analysis appeared to address the hydrologic and hydrodynamic impacts of climate change, incorporating a sequence of linked models to propagate the effects throughout the system.

The panel did not see clear evidence of the use of these hydrologic and hydrodynamic effects to assess the corresponding impacts on ecological processes in the BDCP plan area. According to the DWR 2010 report, the operational simulations of the BDCP using DWR's CALSIM II model have not been completed. Such an analysis is extremely important for investigating the feasibility of meeting future demands associated with the environmental, agricultural, and urban subsystems connected to the greater Bay Delta system. The panel could not find a clear discussion of the extent to which such demands may or may not be met under future climate change scenarios. In addition, there were no quantitative estimates of trade-offs between the co-equal goals of the plan under climate change scenarios, which is discussed below.

Incorporation of the following key elements would strengthen the BDCP's treatment of climate change: (1) Provide a detailed documentation of the approach, analysis, and conclusions, with emphasis on uncertainties and their implications. The lack of discussion in the material provided to the panel of the plan's approach to climate change makes it difficult to more definitively evaluate the scientific basis for climate change projections. (2) Continue efforts to select models with better skills, including models with the ability to reproduce ocean-atmosphere teleconnections, including regime shifts, in the California region (Brekke, 2008, 2009); (3) Quantify the impacts of warming, changes in watershed hydrology and sea level rise on the ecology of the Delta system though the use of ecological models (e.g., CASCaDE, 2010) and quantify the effects on the plan's co-equal goals; and (4) clearly address the role of climate change in the adaptive management strategy. Considering the length of the planning horizon and the importance of climate change to the plan's success, the panel concludes that the BDCP should include a separate chapter on this subject. In view of the importance of the climate change implications in the planning and

BDCP1673

implementation of the BDCP, the panel recommends that this work be reviewed in detail by an independent expert panel assembled by the Delta Science Program or the Interagency Ecological Program.

A FRAMEWORK FOR LINKING DRIVERS AND EFFECTS

The comprehensive conceptual framework developed by the Interagency Ecological Program related to the drivers of pelagic organism decline in the Delta is an important example of supporting science (Mueller-Solger, 2010). This framework identifies and links, in the context of both ecosystem structure and functioning, the key stressors that help to explain the decline of pelagic organisms. The "drivers of change" (Figure 5) are quantifiable, "suitable for model evaluation" and directly linked to hydrologic, biogeochemical and biotic changes that accompany diversion of freshwater from the Delta and parallel increases in nutrient and other pollutants resulting from upstream anthropogenic activities. This is an example of how the individual components could be functionally and conceptually linked and of how climate-change modeling should be integrated into other aspects of the BDCP, including regime shifts.

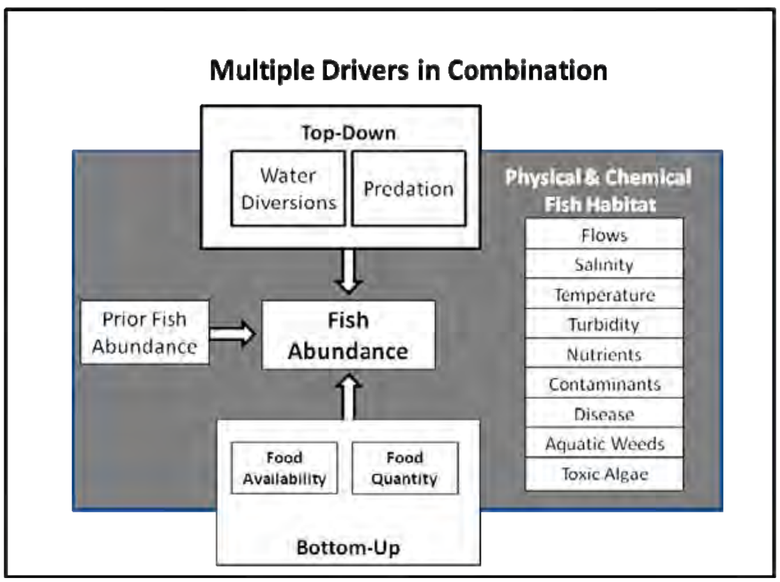


FIGURE 5. Conceptual framework, providing example of supporting science for linking drivers of ecological change to fish community responses. This figure could be a starting point for establishing and rationalizing these linkages. SOURCE: Reprinted, with permission, from Interagency Ecological Program (2010) as modified from Sommer et al. (2007). Available online at: *http://www.water.ca.gov/iep/docs/FinalPOD2010Workplan12610.pdf*.

The types of stressors identified are integrative, reflecting co-occurring physical, chemical, and biotic changes. They also apply to multiple structural (food web structure, biodiversity) and functional (food transfer changes, biogeochemical cycling) changes taking place in the Delta. The framework and associated detail are both comprehensive and useful in terms of linking these drivers to changes taking place at multiple levels of the food web. This type of conceptual approach will also be useful for examining other drivers and impacts of ecological change, including observed changes in fish community structure and production; specifically, how these changes are affected and influenced by changes in physico-chemical factors (e.g., salinity, temperature, turbidity, nutrients/contaminants) and at lower trophic levels (phytoplankton, invertebrate grazers, and prey).

Such a conceptual framework is a necessary precursor to the more holistic integrated analyses for which this panel has identified a need. It may well be impossible to develop a single, integrated model that simultaneously addresses all sources of uncertainty. However, the panel identifies the need for clearer connections among the currently disparate analyses as part of a more synthetic BDCP.

SIGNIFICANT ENVIRONMENTAL FACTORS AFFECTING LISTED SPECIES

Much of the analysis of factors affecting the decline of smelt and salmonids in the Delta has focused on water operations, in particular, the pumping of water at the south end of the delta for export to other regions. This is in part because the pumping can be shown to kill some fish and in part because proposed changes in water operations were the focus of biological assessments and biological opinions developed by NOAA and USFWS (NRC, 2010b). However, many scientists and others in the region have recognized that other significant environmental factors ("other stressors") have potentially large effects on the listed fishes (e.g., NRC, 2010b). Recent studies have suggested that some of these other factors might be of critical importance to fish (e.g., Baxter et al., 2010; Baxter, 2010; Glibert, 2010). In addition, there remain considerable uncertainties regarding the degree to which different aspects of flow management in the Delta, especially X2 management, affect the survival of the listed fishes (e.g., NRC, 2010b). Indeed, the significance and appropriate criteria for future environmental flow optimization have yet to be established, and are uncertain at best.

The panel supports the concept of a quantitative evaluation of the significance of stressors, ideally using life-cycle models, as part of the BDCP, but such a quantitative evaluation is not part of the draft of the BDCP. The panel concludes that in addition to being incomplete, the absence of a data-based, quantitative assessment and analysis of stressors, ideally using life-cycle models, that supports the effects analysis and adaptive management, is a significant scientific flaw in the current version of the BDCP. A sound, data-supported, quantitative analysis of stressors should be one part of the planning process and should provide the foundation for the effects analysis, adaptive management, and ultimately the choice of conservation measures.

SYNTHESIS

The panel finds the BDCP to be a long list of ecosystem management tactics or incomplete scientific efforts with no clear over-arching strategy to integrate the science, or implement the plan. Furthermore, the BDCP does not tie proposed actions together, in terms of addressing the co-equal goals in a unified way or in terms of ecosystem restoration. On the ecosystem side alone, the plan lists more than 100 restoration actions but provides no guidance on which actions are most important, which actions are more or less feasible, which species are more or less susceptible to extinction, which restoration efforts are most difficult, and which actions might be most easily and immediately addressed. In other words, there is a list but not a synthesized plan for the restoration activities. A systematic and comprehensive plan needs a clearly stated strategic view of what each major component of the plan is trying to accomplish, how it is going to do it, and why it is justified. Also, a systematic and comprehensive plan would show how the co-equal goals are coordinated and integrated into a single resource plan.

THE RELATIONSHIP OF THE BDCP TO OTHER SCIENTIFIC EFFORTS

A cohesive conservation plan should provide a clear picture of how the different efforts in the Delta fit together. Indeed, such a synthesis could be valuable not only to the BDCP but also to other conservation efforts in the region. As noted above, the BDCP does not provide adequate perspective on how it fits into, for example, the broader Delta Plan, or on how documents such as the Delta Risk Management Strategy fit into the BDCP. Also, aspects of the BDCP fundamental to understanding how and what science was applied are not yet developed. The inadequacies of ingredients such as the effects analysis, or the details of adaptive management or monitoring, lead the panel to ask, how will these tools be employed to assure effective implementation of the BDCP? How specifically will they be tied to the proposals for conservation and infrastructure change? Evidence of a *coordinated* conservation and water management strategy is the first step in establishing public trust that this is a scientifically credible effort.

Clarification of the volume of water to be diverted or mention of how it will be diverted is crucial to a scientific analysis. Moreover, it is unclear how the upper capacity limit of the isolated conveyance structure of 15,000 cfs (draft

BDCP Chapter 4.2.2.1.1 and Table 4-1) was established. The BDCP cannot be properly evaluated if it does not clearly specify the volume of water deliveries whose negative impacts are to be mitigated. The draft BDCP suggests that the water requirements are based on the amount of acreage and crops that contractors have grown, or on the maximum deliveries specified by the SWP contracts—up to 4.173 MAF/year by 2021 (draft BDCP, Chapters 4.3.1 and 5.1). There is no mention that quantities diverted may be constrained by various provisions of California water law, by possible changes in the extent of irrigated agriculture south of the Delta, and by potential changes in cropping patterns fueled by globalizing forces of supply and demand for food. The draft BDCP also fails to identify and integrate demand management actions with other proposed mitigation actions. A conservation plan should address issues of water use efficiency and should account for future trends in other variables that drive the demand for agricultural and urban water supplies. These issues are directly pertinent to the establishment of a water use strategy and they bear importantly on the costs of restoration actions intended to minimize adverse ecological effects. The BDCP's lack of attention to these issues constitutes a significant omission, given the intensifying scarcity of water in California.

In short, synthesis at all levels is a key ingredient in converting a document into a plan. The lack of synthesis constitutes a systemic problem in the draft BDCP. The panel recognizes that the challenge of linking tactics and strategy with a problem this complex is great, but no plan is either complete or likely to point the way toward success without meeting that challenge.

5
Adaptive Management in the BDCP

Adaptive management is a formal, systematic, and rigorous program of learning from the outcomes of management actions, accommodating change, and thereby improving management (Holling, 1978; NRC, 2003). It has been recommended as part of the solution to many environmental problems (e.g., NRC, 2004a), and it is quite appropriately an important part of the draft BDCP. Adaptive management was developed in response to the difficulty of predicting the outcome of management alternatives in natural systems, because of the many uncertainties involved. Current models, typically used for formulating restoration plans, often lack predictive power. Adaptive management, at least in theory, provides resource managers with an iterative strategy to deal with uncertainties and use science, with a heavy emphasis on monitoring, for planning, implementation, and assessment of restoration efforts (Williams et al., 2009). The BDCP has correctly recognized the importance of adaptive management in its various conservation measures and its developers should be commended for emphasizing this aspect of the plan.

Despite numerous attempts to develop and implement adaptive environmental management strategies, many of them have not been successful (Gregory et al., 2006; Walters, 2007). Walters (2007) concluded that most of more than 100 adaptive management efforts worldwide have failed primarily because of institutional problems that include lack of resources necessary for expanded monitoring; unwillingness of decision makers to admit and embrace uncertainties in making policy choices; and lack of leadership in implementation. Thus many issues affecting the successful implementation of adaptive management programs are attributable to the context of how they are applied and not necessarily to the approach itself (Gregory et al., 2006). In addition, the aims of adaptive management often conflict with institutional and political preferences for known and predictable outcomes (e.g., Richardson, 2010) and the uncertain and variable nature of natural systems (e.g. Pine et al., 2009). The high cost of adaptive management, and the large number of factors involved also often hinder its application and success (Lee, 1999; NRC, 2003). Thus, adaptive management, although often recommended, is not a silver bullet and it is not easy, quick, or inexpensive to implement.

In addition to the above difficulties, Doremus (forthcoming) has advocated an analysis of conditions to determine whether adaptive management is an appropriate strategy before it is undertaken. This is good advice, and by implication it could be followed as a method of evaluating existing adaptive management programs. Doremus argues that three conditions favor the use of adaptive management: the existence of information gaps, good prospects for learning at an appropriate time scale compared to management decisions, and opportunities for adjustment. This panel has not performed a formal analysis of the BDCP's situation in regard to these three conditions, and is not aware of any such analysis, but it does draw some preliminary conclusions. Clearly, the first condition (the presence of information gaps) exists, and the second condition (good prospects for learning) seems likely to exist if the program is designed well. The third condition (opportunities for adjustments) is more problematic. There are pressures for management guarantees; for example, the draft BDCP makes clear that one of its aims is a reliable water supply, and Sagouspe (2010) points out that the Planning Agreement that led to the BDCP provides assurances that "no additional restrictions on the use of land, water, or financial resources" beyond the agreed-on amounts will be required without the agreement of the water users (c.f. Richardson, 2010, cited above). Such agreements on their face seem to reduce opportunities for adjustments, although they do not necessarily preclude them altogether.

All of the above considerations lead as well to a reminder of the need for clear goals, cited in many appraisals of adaptive management (e.g., Milon et al., 1998), and this returns the panel to its earlier concern, namely, that the goals of the BDCP are multiple and not clearly integrated with each other. Despite all of the above challenges, there often is no better option for implementing management regimes, and thus the panel concludes that the use of adaptive management is appropriate for the BDCP.

In light of the above, this panel further concludes that the BDCP needs to address these difficult problems and integrate conservation measures into the adaptive management strategy before there can be confidence in the adaptive management program. In addition, an important step in adaptive management that is often given less attention than the others is the need for a mechanism to incorporate the information gained into management decision-making (e.g., NRC, 2003, 2006, 2008). This matter is critical; it also was raised by the Bay Delta Conservation Plan Independent Science Advisors (draft BDCP, Appendix G) and is discussed further below.

In 2009, the BDCP's developers engaged a group of Independent Science Advisors to provide expertise on approaches to adaptive management in the BDCP (draft BDCP, Appendix G-3). Their advice has been incorporated into the adaptive management program presented in Section 3.7 of the draft BDCP. The Independent Science Advisors' report to the BDCP Steering Committee identified key missing elements in the available documentation at the time, including the formal setting of goals based on problems; more effective use of conceptual or simulation models; a properly designed monitoring strategy to evaluate the effectiveness of conservation measures; and more effective assessment, synthesis, and assimilation of information collected during the implementation. Further, their report recommended an adaptive management framework for the BDCP (Bay Delta Conservation Plan Independent Science Advisors' Report on Adaptive Management, 2009, Figure 1, p. 3). The panel concludes that the Independent Science Advisors have provided a logical framework and guidance for the development and implementation of an appropriate adaptive management program for the BDCP.

Much of the information on the adaptive management program is contained in Section 3.7 of the draft BDCP. A brief description of the management of the adaptive management program is presented in Section 7.35. Identification of uncertainties, a critical step in any adaptive management program, is discussed under each of the Conservation Measures (Section 3.4) and adaptive management considerations are shown in Table 3-20, which is part of Section 3.6, Monitoring and Research Program. Because the details of the adaptive management program are fragmented and occur throughout the BDCP without clear linkages of critical components in one section of the document, it is difficult to obtain an overall assessment of the promise of the adaptive management program. The information is not sufficient to demonstrate that the adaptive management plan is properly designed and follows the guidelines provided by the Independent Science Advisors.

Although the adaptive management framework provided by the Independent Science Advisors recommended a logical, stepwise approach for flow of information (Bay Delta Conservation Plan Independent Science Advisors' Report on Adaptive Management, 2009, Figure 1, p. 3), the adaptive management framework shown in Figure 3-63 of the BDCP (also shown in Appendix E of this report) is significantly different and is missing some key elements. It is not clear how the monitoring and "targeted research" programs were designed using goals and objectives, desired outcomes, and performance metrics to select and evaluate steps outlined in the Independent Science Advisors' report. More important, clearly defined uncertainties at various scales starting with the ecosystem level are not presented adequately in the BDCP. In particular, the role of models is not clearly identified in the adaptive management framework, except in Figure 3-63. Box 5b of that figure simply suggests a refinement of models without identifying them. Also, the BDCP does not make clear whether adaptive management applies to broad, ecosystem goals or narrower goals related to specific natural communities or specific conservation measures, or both. Without this distinction and a clear discussion of the role of adaptive management at the ecosystem level, the draft BDCP does not provide assurance that it will successfully use adaptive management to make adjustments during the planning, design, and operational stages of the project.

The Independent Science Advisors correctly pointed out the need for an emphasis on when and where the active versus passive approaches should be used during the design phase. A passive approach is used when the projects are irreversible in nature, as in the case of a dual conveyance facility whereas an active approach involves experiments to test competing hypotheses in cases of significant uncertainties in ecosystem response. The BDCP lacks details of the types of adaptive management approaches and the specifics of the experimental testing that would be conducted to reduce uncertainties. Passive adaptive management is used when there is a high confidence regarding the anticipated ecosystem response, often predicted by reliable models. However, the BDCP does not explicitly rationalize the particular selections in the adaptive management framework, for example, with regard to proposed creation of wetlands, levee restoration, and conveyance options.

The lack of detail about the adaptive management program's details makes evaluating it difficult. Many details of adaptive management are needed to perform a thoughtful review of it, and in some cases, those details emerge only as the plan is implemented. For these reasons, the panel is unable to provide a detailed review of the adaptive management plan at this stage. However, some comments and suggestions are in order.

First, as mentioned above, an adaptive management program requires clear goals. This point often is overlooked. If the project's management goals are not clear, then it will not be evident how to adapt management in the face of new information. The BDCP does not explain how its multiple goals are to be integrated, but the problem goes deeper: some agreed-on goals, such as sustainability of the ecosystem or having a healthy ecosystem, may no longer be acceptable to all parties when they become more specific or when it becomes clear that not all aspects of the ecosystem can be rehabilitated simultaneously. This problem is not unique to the Delta: it affects other large restoration efforts as well, for example, the Everglades (e.g., Milon et al., 1998; NRC, 2010).

Second, adaptive management requires a monitoring program to be in place. The draft BDCP describes its monitoring plan in considerable detail: Table 3-20, which describes the monitoring for effectiveness of conservation actions, runs more than 80 pages, implying a large amount of monitoring activity. However, because there is no effects analysis, it is difficult to evaluate the scientific basis or to justify the appropriateness of individual elements of the monitoring program, elements which clearly should be tied to the results of the effects analysis. In addition, the panel questions the availability of resources necessary to accomplish the all monitoring described in Table 3-20, especially because additional baseline, compliance, and other monitoring also are described in the BDCP as being necessary.

Third, although all of the elements of an adaptive management program are present in the draft BDCP, some of them are not described in detail and some do not appear to be incorporated into the framework in Figure 3-63 (shown in Appendix E of this report). The panel emphasizes again how important it is for a meaningful adaptive management program to be tied to the results of the effects analysis, or at least related to the same issues being addressed by the effects analysis. If it is not, then it is difficult to see how the monitoring and adaptive management program can inform the implementation of the plan and inform decision makers.

The draft BDCP appropriately cites the Independent Science Advisors' Report on Adaptive Management conclusion that:

> "the weakest aspect of most adaptive management plans is in the sequence of steps required to link the knowledge gained from implementation monitoring and research and other sources to decisions about whether to continue, modify, or stop actions, refine objectives, or alter monitoring" (draft BDCP, p. 3-577).

This issue has been addressed by NRC reports on the Everglades restoration (e.g., NRC, 2006, 2008), and it is taken seriously by the Comprehensive Everglades Restoration Program. The panel recognizes the difficulty of understanding from the outside how decisions actually are made, and those elements of the BDCP's adaptive management program that require publication of scientific results and provision of the resulting scientific advice to program managers are a good step in that direction. However, a clearer description of the mechanisms that will enable the scientific results to inform management decisions would be helpful.

Details of two other aspects of adaptive management, stakeholder engagement and interagency coordination, are vague. The way that agencies coordinate their activities and that stakeholders participate in the process can have significant consequences. For example, Linkov and his colleagues (Linkov et al., 2006a,b) have described the use of multicriteria decision analysis to enhance adaptive management, and the NRC (2004b) has provided worked examples of such an approach applied to restoring Atlantic salmon in Maine. Those approaches all depend on input from stakeholders. The concepts of a stakeholder committee to receive public input and a "Decision Body" to adjust water operations are too vague and their functions appear to be too limited to provide guidance. The panel recommends that the BDCP take advantage of the literature on this topic—beginning, but not ending, with the material cited above—to inform its processes.

Finally, the importance of action-related triggers related to environmental conditions or the status of covered species is briefly mentioned in the draft BDCP (draft BDCP, Section 3.7.4, pp. 3-586-3-587), but there is no discussion of their importance and role in the adaptive management program and their relation to the effects analysis.

The essence of adaptive management is to identify major uncertainties about the efficacy of policy actions, then to design field tests or management experiments to directly measure efficacy. Such tests can include field evaluation of alternative feedback decision rules that do or do not include thresholds or triggers for action. Initial adaptive management modeling exercises may screen out policies that require triggers by illustrating the challenges associated with uncertainty about the best triggering conditions. In some cases, however, triggers for action can and have been used, often in conjunction with multi-objective structured decision analysis that includes the values and alternatives preferences of the various stakeholders involved (e.g., Karl et al., 2007; Kiker et al., 2008; Miller et al., 2010).

One such example is a recent effort on the Colorado River, where managers are seeking to establish flow releases to control non-native fish below Glen Canyon Dam[9] (Runge et al., 2011). Through the decision-analysis process, objectives were identified (e.g., manage resources to protect tribal sacred sites and spiritual values, maintain and promote local economies and public services, operate within the authority, capabilities, and legal responsibility of the Bureau of Reclamation). In addition, management strategies were evaluated against the objectives, and tradeoffs between strategies were considered. The process identifies specific triggers (e.g. following High-Flow experimental floods, abundance of native or introduced fish species, flow and sediment load) for management actions (e.g., removal of non-native species, fine sediment slurry, release of stranding flows), while other actions (e.g., mechanical or chemical disruption of fish spawning areas, augmentation of fine sediment) are recommended without triggers. The value of triggers is in the efficiency of managing the system, minimizing expensive actions to when and where they are thought to be necessary for and beneficial to species recovery. Such triggers also would help to design a more-focused monitoring program. However, the challenge of using triggers is in the uncertainty in establishing thresholds for triggering actions. Thus, (Runge et al., 2011) caution that their results do not provide the final decision but instead provide guidance for further consultation by the decision makers. That consultation is likely to require experimentation, modeling, and continued adaptive management.

In summary, the BDCP's adaptive management program is not fully developed. In addition, there remain significant scientific, policy, and management uncertainties about the BDCP's purpose and organization. The panel concludes that the BDCP's developers can benefit significantly from experiences in adaptive management attempted in other large-scale ecosystem restoration efforts. One such example is the CERP, where adaptive management has been a key component since its inception in 1999 (USACE & SFWMD, 1999). As recognized by the NRC (2006), the CERP adaptive management strategy provides a sound organizational model for the execution of a passive approach. More recent activities also include examples of active approaches where field tests have played a major role in the early phases of selected projects (RECOVER, 2010). Key components of the CERP adaptive management program are:

- CERP Adaptive Management Strategy (RECOVER, 2006a);
- Monitoring and Assessment Plan and an Assessment Strategy designed to monitor system-wide responses to determine how well CERP is achieving its goals (RECOVER, 2004; 2006a,b; 2009); and

---

[9] The panel provides this example as a good use of action-related triggers. The success of adaptive management in Glen Canyon in general has been questioned (Susskind et al., 2010).

- CERP Adaptive Management Integration Guide (available in draft form) (RECOVER, 2010).

The above documents detail more than five years of progress in implementing adaptive management in the CERP. The CERP's program includes nine activities, which have been effectively integrated into the standard practice of project planning and life-cycle analysis (NRC, 2006). The integration guide describes how to apply adaptive management concepts to the CERP program and related projects through the identification of key uncertainties and the incorporation of activities into the existing CERP planning and implementation process. Even a soundly implemented adaptive management program is not a guarantee of a successful restoration effort, however. As described in several NRC reports and other documents, several factors outside the purview of the adaptive-management teams and even the program managers have hindered restoration progress in the Everglades. They include financial, political, bureaucratic, legal and other obstacles (e.g., NRC 2006, 2008, 2010), factors certain to influence the implementation of the BDCP as well. But a well-designed and implemented program should improve the likelihood of success in implementing the BDCP.

6
Management Fragmentation and a Lack of Coherence

The management of any science-based process has profound impact on the use of science and adaptive management within that process. The panel was charged with evaluating the use of science and adaptive management, and therefore management of the enterprise falls appropriately within this charge. The absence of any synthesis in the draft BDCP draws attention to the fragmented system of management under which it was prepared—a management system that lacks coordination among entities and clear accountability. No one public agency, stakeholder group, or individual has been accountable for the coherence, thoroughness, and scientific integrity of the final product. Rather, the plan appears to reflect the differing perspectives of federal, state and local agencies, and the many stakeholder groups involved, as noted in the introduction to this report. This is not strictly a scientific issue, but fragmented management is a significant impediment to the use and inclusion of coherent science in future iterations of the BDCP. Different science bears on the missions of the various public agencies; different stakeholders put differing degrees of emphasis on specific pieces of science; and different geographical entities require different kinds of science. The panel concludes that without more coherent and unified, the BDCP's final product, like the current draft, will rely on bits and pieces of science that are not well integrated. Moreover, the lack of coherence in the management of the preparation of the BDCP helps to explain the fragmentation of science and the lack of synthesis.

The discussion of the implementation structure in Chapter 7 of the draft BDCP suggests that the fragmented management that characterizes the preparation of the draft plan is also likely to be a feature of the implementation of the plan that finally emerges. The appointment of a single program manager and creation of an Implementation Office, as envisioned in the draft BDCP, are unlikely—even taken together—to result in a well-integrated, coherent implementation program. The public agencies that are involved in the planning and implementation of the BDCP are a mix of operating and regulatory state and federal agencies. Moreover, their interests are intertwined with those of the stakeholder groups, most obviously water-using and environmental groups. These agencies and stakeholders have differing missions and agendas that are almost certain to conflict from time to time and yet the BDCP has no formal mechanism to deal with such conflicts.

Indeed, the BDCP appears to carve out territorial boundaries that make fragmented, and even perhaps antagonistic, management of the plan's implementation more likely. Thus, for example, the BDCP states, "The [Implementation Office] will not be involved in the development or operation of the [State Water Project] and/or [Central Valley Project] facilities" (draft BDCP, p. 7-5). Further, the plan states, "No general delegation of authority by [the California Department of Water Resources] or the [Bureau of] Reclamation to the Program Manager or one of their employees assigned to the [Implementation Office] will occur" (draft BDCP p. 7-7). The plan also proposes that agency personnel be assigned to populate various BDCP implementation committees. This seems to further ensure that inter-agency conflicts and traditional turf battles will be strongly internalized in the management arrangements. The plan, then, envisions that traditional agency missions and turf will be protected, leaving the program manager to navigate through a maze of conflicting interests without any real authority or capacity to resolve conflicts and otherwise ensure that the management approach is integrated.

There is an important literature on the problem of management fragmentation in the planning and operations management of large water schemes (Conca, 2005; Feldman, 2011; Scholz and Siftel, 2005). There is additional helpful literature on network governance (Kettl and Goldsmith, 2004) and collaborative federalism (Emerson and Murchie, 2010). This work underscores the importance of collaboration, the sharing of authority and power, and acknowledgment of the interests of all stakeholders if the large-scale management of water is to be integrated and successful. The panel recommends that the BDCP's authors give this matter careful attention.

Development and implementation of large restoration and conservation programs such as the BDCP often require a complex structure to incorporate technical, political, and legal realities and the evolving dynamics of both the physical and organizational environments. The panel recommends that the agencies responsible for implementing the BDCP review other examples of large scale restoration programs that have been developed and implemented. One such example is the Boston Harbor Islands National Recreation Area where management coordinates through a General Management Plan executed with several cooperative agreements. Although CalFed dissolved, the former CalFed institutional structure dealt with some of the same management issues. The CalFed experience and associated body of literature could be a useful source of positive and negative lessons.

Another example is the Everglades restoration program (CERP; www.evergladesplan.org), with which several committees of the National Research Council have been involved for many years (NRC, 2006, 2008, 2010c). Since its authorization in the Water Resources Development Act of 2000, the CERP has necessitated the development of a number of coordination processes, agreements, and carefully designed planning and implementation efforts (Figure 6 in Box 2 of this report) to incorporate the unprecedented scope and complexity of the final plan, regulations of the federal and state governments, and stakeholder interests. However, unlike the BDCP, the CERP's focus was more on ecosystem restoration than on concerns about endangered and threatened species.

Unlike the seemingly fragmented structure for the BDCP implementation, the authority for implementing the Everglades program lies with both federal and state agencies with a carefully designed planning process and inter-agency agreements in each step. The Everglades management system has accountability in that the federal and state agencies have a formal agreement on cost-sharing of the entire restoration program and the authority to execute the restoration plan. Furthermore, they have coordination mechanisms, such as the South Florida Ecosystem Restoration Task Force which is a coordination mechanism for many entities involved in the restoration. Specifically, the U.S. Army Corps of Engineers (USACE) and the U.S. Department of the Interior (DOI), in partnership with the lead state agency, the South Florida Water Management District (SFWMD), are responsible for undertaking the CERP's implementation. A continuously evolving Integrated Delivery Plan sets the priority projects that must be implemented. Central to the planning and implementation of a particular project is the Project Implementation Report (PIR) developed by a Project Delivery Team, which constitutes a multi-agency team with strong stakeholder participation (Box 2). Active participation by all agencies with authority and pre-approved CERP Guidance Memoranda (CGMs) ensure agreement on the plan, scientific basis, and the expected benefits in the PIR before it is submitted for approval and authorization for funding (see Figure 3-3 of NRC, 2006). The PIR includes an evaluation of alternative designs and operations for environmental benefits, the costs, and the engineering feasibility (NRC, 2006). Once a project

BOX 2
Implementation of Everglades Restoration:
Structure for Inter-agency Collaboration and Stakeholder Involvement

The U.S. Army Corps of Engineers (USACE), Department of the Interior (DOI), and the South Florida Water Management District (SFWMD) are currently implementing a planning process that provides significant opportunity for local, state, federal and tribal governments, as well as public and non-governmental stakeholders to participate in the projects that are being designed and implemented. For each project, an interagency, interdisciplinary Project Delivery Team (PDT) is established. The PDT is led by the USACE and SFWMD Project Managers and includes members from various local, state, federal and tribal governments. Figure 6 illustrates the typical composition and entities that provide input and feedback to the PDTs. Although much work is accomplished in a PDT, additional agency stakeholder and public in-

box continues

BOX 2 Continued

put are received at scheduled points in the planning process. Specifically, such advice is sought as development of project objectives, identification of performance measures, selection of evaluation models, and development and evaluation of alternative plans. Additional opportunities for governmental agencies, stakeholders, and the public to provide input and feedback during the planning process are provided at publicly noticed meetings of the following established groups (a) Governing Board of the SFWMD; (b) South Florida Ecosystem Restoration Task Force (SFERTF); (c) South Florida Ecosystem Restoration Working Group; and (d) The Water Resources Advisory Commission (WRAC).

To ensure that the development and implementation of CERP is based on the best and most recent science available, and to ensure that the restoration program is implemented with an adaptive management approach, a multiagency, multidisciplinary science team called RECOVER has been formed. In addition, the USACE and SFWMD have established an Interagency Modeling Center (IMC) to function as a single point of service for the modeling needs of CERP. As the primary organization responsible for regional and subregional modeling for CERP modeling, the IMC conducts system-wide evaluations of CERP implementation plans and updates, and provides modeling support for PDTs.

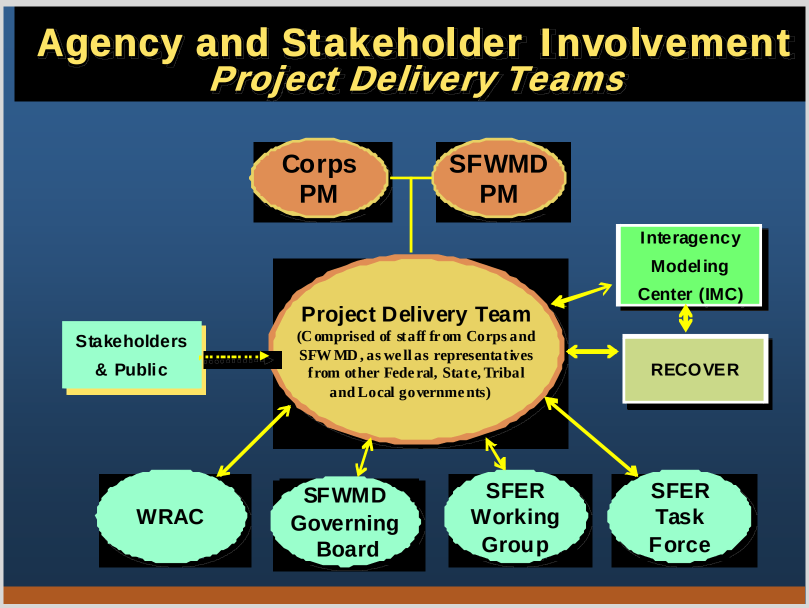


FIGURE 6. Agency and stakeholder involvement in the project delivery teams (PDT). Figure courtesy of the South Florida Water Management District

is authorized, depending on the funding, a series of technical refinements beginning with detailed designs and ending with construction occurs prior to its operation. Project Cooperation Agreements between the federal and the state partner are obtained prior to the initiation of construction. The current progress of CERP has demonstrated the need for formal agreement among partners. One example of such as agreement is the Design Agreement between the USACE and SFWMD (*http://www.evergladesplan.org*). Implementation of the agreement is ensured by an interagency unit known as the Design Coordination Team (DCT), which oversees the schedules and budgets, plans and specifications, and contractual work.

However, no matter how good the management structure may be, it is no guarantee of progress; it is a necessary but not a sufficient condition. Experience with large restoration projects elsewhere, and especially in the Delta, reveals that progress will be affected by lawsuits, economic crises, unexpected (and expected) environmental events, cost overruns, political changes, and so on. Yet the literature and examples mentioned here show that management of complicated systems, where more than one agency has management responsibilities, can be successful as long as there is adequate coordination and clear accountability. Apparently, the new deputy secretary of the California Natural Resources Agency has the BDCP as his major responsibility, which is an encouraging development. The panel recommends that the BDCP's authors give this matter careful attention, because an appropriate system of management is necessary but not sufficient for the use of coherent, synthesized science in future iterations of the BDCP and a successful adaptive management program.

7
In Conclusion

The panel finds the draft BDCP to be incomplete or unclear in a variety of ways and places. The plan is missing the type of structure usually associated with current planning methods in which the goals and objectives are specified, alternative measures for achieving the objectives are introduced and analyzed, and a course of action is identified based on analytical optimization of economic, social, and environmental factors. The lack of an appropriate structure creates the impression that the entire effort is little more than a post-hoc rationalization of a previously selected group of facilities, including an isolated conveyance facility, and other measures for achieving goals and objectives that are not clearly specified. Furthermore, unless goals are not only stated but also prioritized, it is impossible to forecast the effects of projects that would achieve the goals because it is impossible to identify the projects or the consequences that would be deemed acceptable. One symptom of the absence of appropriate structure is the systemic lack of synthesis in the BDCP. Frequently, the plan appears to be little more than a list of tactics or management options that are not strategically integrated. It is unclear how these tactics would be knitted together to achieve the objectives of the plan which are themselves not always clear; and there is no indication of how the various tactics and elements in the plan could be implemented in a logical and strategic fashion.

Several errors of omission also complicate this review. First, there is no effects analysis that describes the impacts of the proposed project or alternatives on target species, even though the BDCP notes that the effects analysis would be "...the principal component of a habitat conservation plan..." Without an effects analysis it is exceedingly difficult to evaluate alternative mitigation and conservation actions. In addition, the plan remains silent on the probable effects of proposed actions on target species. Second, the descriptions of the BDCP's purpose lack clarity. The confusion arises because it is unclear to what extent or whether the BDCP is exclusively a habitat conservation plan, which is to be used as an application for a permit to "take" listed species incidentally, or to what extent or whether it is also intended to be a plan to achieve the co-equal goals of providing reliable water supply and protecting and enhancing the Delta ecosystem. Third, the proposed adaptive management plan is incomplete. Any adaptive management plan requires a monitoring program and, although one is described, it is unclear what purposes it is intended to achieve. The proposed monitoring program has not been linked to the adaptive management plans in a way that would allow managers to account for lessons learned from previous experience, and more important, it is not linked to the effects analysis. In short, there is no compelling information that would allow the panel to conclude that the adaptive management program has been properly designed.

The lack of integrated management and coherence in developing the BDCP is also a shortcoming. The plan reflects the perspectives of various public agencies at the federal, state, and local levels and the many stakeholder groups involved. Although this is not strictly a scientific issue, the panel concluded that fragmented management is a significant impediment to the use and inclusion of coherent science in future iterations of the BDCP. Moreover, the proposed BDCP implementation arrangements appear unlikely to result in a well-integrated, coherent implementation program because of the conflicting agency and stakeholder interests and objectives that are built into the structure of the proposed Implementation Office.

The panel underscores the importance of a credible and a robust BDCP in addressing the various water management problems that beset the Delta. A stronger, more complete, and more scientifically credible BDCP that effectively integrates and utilizes science could indeed pave the way toward the next generation of solutions to California's chronic water problems.

References

Baxter, R. 2010. Are Juvenille Longfin Smelt Abandoning the Suisun Bay Neighborhood? …The Rest of the Story. Abstract. 6th Biennial Bay-Delta Science Conference. Ecosystem Sustainability: Focusing on Science and Managing California's Water Future. September 8, 2010.

Baxter, R. R. Breuer, L. Brown, M. Chotkowski, F. Feyrer, M. Gingras, B. Herbold, A. Mueller-Solger, M. Nobriga, T. Sommer, and K. Souza. 2008. Pelagic Organism Decline Project Report: 2007 Synthesis of Results. Sacramento, CA: Interagency Ecological Program for the San Francisco Estuary.

Baxter, R., R. Breuer, L. Brown, L. Conrad, F. Feyrer, S. Fong, K. Gehrts, L. Grimaldo, B. Herbold, P. Hrodey, A. Mueller-Solger, T. Sommer, and K. Souza. 2010. Interagency Ecological Program, 2010 Pelagic Organism Decline Work Plan and Synthesis of Results. Sacramento, CA: Interagency Ecological Program for the San Francisco Estuary. Available online at: *http://www.water.ca.gov/iep/docs/FinalPOD2010Workplan12610.pdf*. Last accessed March 10, 2011.

BDCP (Bay Delta Conservation Plan Steering Committee). 2010. Bay Delta Conservation Plan Working Draft. November 18. Available online at: *http://www.resources.ca.gov/bdcp/*. Last accessed April 26, 2011.

Bonfils, C., B.D. Santer, D.W. Pierce, H.G. Hidalgo, G. Bala, T. Das, T.P. Barnett, D.R. Cayan, C. Doutriaux, A.W. Wood, A. Mirin, and T. Nozaw. 2008. Detection and attribution of temperature changes in the mountainous western United States. Journal of Climate 21: 6404–6424.

Borsuk, M.E., R. Clemen, L. Maquire, and K.H. Reckhow. 2001. Stakeholder values and scientific modeling in the Neuse River watershed. Group Decision and Negotiation 10: 355–373.

Borsuk, M.E., C.A. Stow, and K.H. Reckhow. 2004. A Bayesian network of eutrophication models for synthesis, prediction, and uncertainty analysis. Ecological Modelling 173: 219–239.

Brekke, L.D., E.P. Maurer, J.D. Anderson, M.D. Dettinger, E.S. Townsley, A. Harrison, and T. Pruitt. 2009. Assessing reservoir operations risk under climate change. Water Resources Research 45:W04411.

Brekke, L.D., M.D Dettinger., E.P. Maurer, and M. Anderson. 2008. Significance of model credibility in estimating climate projection distributions for regional hydroclimatological risk assessments. Climate Change 89:371–394.

Brown, L.R. 2003. Will tidal wetland restoration enhance populations of native fishes? San Francisco and Watershed Science 1. Available online at: *http://repositories.cdlib.org/jmie/sfews/vol1/iss1/art2*. Last accessed April

13, 2011.

CALFED. 2000. California's Water Future: Framework for Action. Sacramento, CA: CALFED Bay Delta Program.

CASCaDE. 2010. Computational Assessments of Scenarios of Change for the Delta Ecosystem. Available online at: *http://cascade.wr.usgs.gov/index.shtm*. Last accessed March 10**,** 2011.

Cayan, D., M. Tyree, M. Dettinger, H. Hidalgo, T. Das, E. Maurer, P. Bromirski, N. Graham, and R. Flick. 2009. Climate change scenarios and sea level rise estimates for the California 2008 climate change scenarios assessment. CEC-500-2009-014-D. Sacramento, CA: California Climate Change Center.

Chung, F.I., J. Anderson, S. Aurora, M. Ejeta, J. Galef, T. Kadir, K. Kao, A. Olson, C. Quan, E. Reyes, M. Roos, S. Seneviratne, J. Wang and H. Yin. 2009. Using Future Climate Projections to Support water Resources Decision Making in California. California Energy Commission, PIER Program Report, CEC-500-2009-052-D. Sacramento, CA: California Energy Commission.

Climate Action Team. 2010. Climate Action Team Report to Governor Schwarzenegger and the California Legislature. California Environmental Protection Agency. April 2010. Available online at: *http://www.energy.ca.gov/2010publications/CAT-1000-2010-005/CAT-1000-2010-005.PDF*. Last accessed March 10**,** 2011.

Cloern, J.E. 2007. Habitat Connectivity and ecosystem productivity: implications from a simple model. American Naturalist 169: E21–E33.

Cloern, J.E., K.A. Hieb, T. Jacobson, B. Sansó, E. Di Lorenzo, M.T. Stacey, J.L. Largier, W. Meiring, W.T. Peterson, T.M. Powell, M. Winder, and A.D. Jassby. 2010. Biological communities in San Francisco Bay track large scale climate forcing over the north Pacific. Geophysical Letters 37, L21602, doi.1029/2010GLO447744.

Conca, K. 2005. Growth and fragmentation in expert networks: The elusive quest for integrated water resources management. Pp. 432–470 in Handbook of Global Environmental Politics, P. Dauvergne, ed. Cheltenham, UK: Edward Elgar Publishers.

Delta Risk Management Strategy (DRMS). 2009. Executive Summary, Phase 1 Report. Available online at: *http://www.water.ca.gov/floodmgmt/dsmo/sab/drmsp/docs/drms_execsum_ph1_final_low.pdf*. Last accessed March 10**,** 2011.

Doremus, H. Forthcoming. Adaptive management as an information problem. North Carolina Law Review. UC Berkeley Public Law Research Paper No. 1744426. Available online at: *http://ssrn.com/abstract=1744426***.** Last accessed on April 11, 2011.

DWR (California Department of Water Resources). 2010. Climate Change Characterization and Analysis in California Water Resources Planning Studies. Sacramento, CA: California Department of Water Resources.

Ellison, A.M. 1996. An introduction to Bayesian inference for ecological research and environmental decision-making. Ecological Applications 6:

1036–1046.

Emerson, K., and P. Murchie. 2010. Collaborative governance and climate change. Pp. 141–161 in The Future of Public Administration, Public Management and Public Service Around the World: The Minnowbrook Perspective, R. O'Leary, S. Kim, D. Van Slyke, eds. Washington, D.C.: Georgetown University Press.

Feldman, D.L. 2011. Water: A Volume in the Geopolitics of Resources Series. Boston, MA: Polity Books.

Field, C. B., G. C. Daily, F. W. Davis, S. Gaines, P. A. Matson, J. Melack, and N. L. Miller. 1999. Confronting Climate Change in California: Ecological Impacts on the Golden State. Cambridge, MA: Union of Concerned Scientists and Washington, DC: Ecological Society of America.

Fiering, M. 1982. A screening model to quantify resilience. Water Resources Research 18(1): 27–32.

Franco, G., D.R. Cayan, A. Luers, M. Hanemann, and B. Croes. 2008. Linking climate change science with policy in California. Climatic Change 87(Suppl 1):S7–S20.

FWS (Fish and Wildlife Service). 2008. Biological Opinion on Coordinated Operations of the Central Valley Project and State Water Project. Available online at: *http:www.fws.gov/sacramento/ed/documents/SWP-CVP_OPs_BO_12-15_ final_OCR.pdf.* Last accessed March 15, 2010.

Gershunov, A., T.P. Barnett, D. R. Cayan, T. Tubbs, and L. Goddard. 2000. Predicting and downscaling ENSO impacts on intraseasonal precipitation statistics in California: The 1997/98 Event. Journal of Hydrometeorology 1:201–210.

Glibert, P.M. 2010. Long-term changes in nutrient loading and stoichiometry and their relationships with changes in the food web and dominant pelagic fish species in the San Francisco Estuary, California. Reviews in Fisheries Science 18 (2):211–232.

Gilchrest W. 1993. Modeling failure modes and effects analysis. International Journal of Quality and Reliability Management 10(5):16–23.

Gregory, R., D. Ohlson, and J. Arvai. 2006. Deconstructing adaptive management: criteria for applications to environmental management. Ecological Applications 16(6):2411–2425.

Hashimoto, T. 1982. Reliability, resiliency, and vulnerability criteria for water resource system performance evaluation. Water Resources Research 18(1):14-20. Available online at: *http://www.climatechange.ca.gov/climate_action_team/reports/index.html***.** Last Accessed on April 11, 2011.

Healey, M., M. Dettinger, and R. Norgaard. 2008. The State of Bay-Delta Science, 2008: Summary. Available online at: *http://www.science.calwater.ca.gov/publications/sbds.html.* Last accessed on April 11, 2011.

Holling, C.S. 1978. Adaptive Environmental Assessment and Management. Chichester, UK: John Wiley and Sons, Ltd.

Hundley, N., Jr. 2001. The Great Thirst. Californians and Water: A History. Revised Edition. Berkeley, CA: University of California Press.

Independent Science Advisors. 2009. Bay Delta Conservation Plan Independent Science Advisors' Report on Management, prepared for the BDCP Steering Committee, February 2009. Available at: *http://baydeltaconservationplan.com/Libraries/Background_Documents/BDCP_Adaptive_Management_ISA_report_Final.sflb.ash*. Last accessed on April 11, 2011.

Interagency Ecological Program (IEP). 2010. Interagency Ecological Program 2010 Pelagic Organism Decline Work Plan 2. Available online at: *http://www.water.ca.gov/iep/docs/FinalPOD2010Workplan12610.pdf*. Last accessed April 25, 2011.

Jackson, W. T., and A. M. Patterson. 1977. The Sacramento-San Joaquin Delta: The Evolution and Implementation of Water Policy, An Historical Perspective. California Water Resources Center Contribution no. 163. Davis, CA: California Water Resources Center. Available online at: *http://escholarship.org/uc/item/36q1p0vj#page-20*. Last accessed April 25, 2011.

Jassby, A.D., J.E. Cloern, and B.E. Cole. 2002. Annual primary production: Patterns and mechanisms of change in a nutrient-rich tidal ecosystem. Limnology and Oceanography 2002. 47:712.

Karl, H.A., L.E. Susskind, and K.H. Wallace. 2007. A dialogue not a diatribe—effective integration of science and policy through joint fact finding. Environment 49(1):20–34.

Kelley, R. 1989. Battling the Inland Sea. Berkeley, CA: University of California Press.

Kettl, D. F., and S. Goldsmith, eds. 2009. Unlocking the Power of Networks: Keys to High Performance Governance. Washington, D.C.: Brookings Institution Press.

Kiker, G. A., T. S. Bridges, and J. Kim. 2008. Integrating comparative risk assessment with multi-criteria decision analysis to manage contaminated sediments: an example for the New York/New Jersey harbor. Human and Ecological Risk Assessment 14(3):495–511.

Knowles, N. 2008. Potential Inundation Due to Rising Sea Levels in the San Francisco Bay Region. California Climate Change Center. CEC-500-2009-023-F, California Energy Commission, PIER Energy-Related Environmental Research. Available online at: *http://www.energy.ca.gov/2009publications/CEC-500-2009-023/CEC-500-2009-023-D.PDF*. Last accessed March 10, 2011.

Lee, K.N. 1999. Appraising adaptive management. Conservation Ecology 3(2):3. Available online at: *http://www.consecol.org/vol3/iss2/art3*. Last accessed on April 11, 2011.

Legault, C.M. 2005. Population viability analysis of Atlantic salmon in Maine, USA. Transactions of the American Fisheries Society 134:549–562.

Linkov, I., F.K. Satterstrom, G. Kiker, C. Batchelor, T. Bridges, and E. Ferguson. 2006a. From comparative risk assessment to multi-criteria decision analysis and adaptive management: Recent developments and applications. Environment International 32:1072–1093.

Linkov, I., F. K. Satterstrom, G. Kiker, T.S. Bridges, S.L Benjamin, and D.A.

Belluck. 2006b. From optimization to adaptation: Shifting paradigms in environmental management and their application to remedial decisions. Integrated Environmental Assessment and Management 2(1):92–98.

Lund, J., E. Hanak, W. Fleenor, W. Bennett, R. Howitt, J. Mount, and P. Moyle. 2010. Comparing Futures for the Sacramento–San Joaquin Delta. Berkeley, CA: University of California Press.

Marcot, B.G., R.S. Holthausen, M.G. Raphael, M.M. Rowland, and M.J. Wisdom. 2001. Using Bayesian belief networks to evaluate fish and wildlife population viability under land management alternatives from an environmental impact statement. Forest Ecology and Management 153: 29–42.

Maurer, E. P., and H. G. Hidalgo. 2008. Utility of daily vs. monthly large-scale climate data: An intercomparison of two statistical downscaling methods. Hydrology and Earth System Sciences 12:551–563.

McCann, R.K., B.G. Marcot, and R. Ellis. 2006. Bayesian belief networks: applications in ecology and natural resource management. Canadian Journal of Forest Research-Revue Canadienne De Recherche Forestiere 36: 3053–3062

McDermott, R.E., R.J. Mikulak, and M.R. Beauregard. 2009. The Basics of FMEA. New York, NY: CRC Press

Miller, T. J., J. A. Blair, T. F. Ihde. R. M. Jones, D. H. Secor and M. J. Wilberg. 2010. FishSmart: an innovative role for science in stakeholder-centered approaches to fisheries management. Fisheries 35(9):424–433

Milon, W,J, C.F. Kiker, and D.L. Lee. 1998. Adaptive ecosystem management and the Florida Everglades: more than trial-and-error? Water Resources Update 113:37–46.

Moyle, P.B., H.W Li, and B.A. Barton. 1986. The Frankenstein effect: impact of introduced fishes on native fishes in North America. Pp. 415–426 in Fish Culture in Fisheries Management, R.H. Stroud, ed. Bethesda, MD: American Fisheries Society.

Mueller-Solger, A. 2010. Drivers of change in the San Francisco Estuary. Presentation to the NRC Committee/Panel. December 8, 2010. San Francisco, CA.

Murphy, D.D., and P.S. Weiland. 2011. The route to best science implementation of the Endangered Species Act's consultation mandate: The benefits of structured effects analysis. Environmental Management 47(2):161–172.

NOAA (National Oceanic and Atmospheric Administration). 2008. Federal Columbia River Power System Biological Opinion. Available online at: *http://www.fws.gov/pacific/finalbiop/BiOp.pdf.* Last accessed March 10, 2011.

NOAA. 2010. Endangered Species Act Section 7(a)(2) Consultation Supplemental Biological Opinion: Supplemental Consultation on Remand for Operation of the Federal Columbia River Power System, 11 Bureau of Reclamation Projects in the Columbia Basin and ESA Section 10(a)(I)(A) Permit for Juvenile Fish Transportation Program. Available online at: *https://pcts.nmfs.noaa.gov/pls/pctspub/sxn7.pcts_upload.download?p_file=*

F25013/201002096_FCRPS Supplemental_2010_05-20.pdf. Last accessed April 25, 2011.

NRC (National Research Council). 1995. Science and the Endangered Species Act. Washington, D.C.: National Academy Press.

NRC. 2000. Ecological Indicators for the Nation. Washington, D.C.: National Academy Press.

NRC. 2002. Ecological Dynamics on Yellowstone's Northern Range. Washington, D.C.: National Academy Press.

NRC. 2003. Adaptive Monitoring and Assessment for the Comprehensive Everglades Restoration Plan. Washington, D.C.: The National Academies Press.

NRC. 2004a. Endangered and Threatened Fishes in the Klamath River Basin: causes of Decline and Strategies for Recovery. Washington, D.C.: The National Academies Press.

NRC. 2004b. Atlantic Salmon in Maine. Washington, D.C.: The National Academies Press.

NRC. 2005. Developing a Research and Restoration Plan for Arctic-Yukon Kuskokwim (Western Alaska) Salmon. Washington, D.C.: The National Academies Press.

NRC. 2006. Progress Toward Restoring the Everglades: The First Biennial Review. Washington, D.C.: The National Academies Press.

NRC. 2008. Progress Toward Restoring the Everglades: The Second Biennial Review. Washington, D.C.: The National Academies Press.

NRC. 2009. Science and Decisions. Washington, D.C.: The National Academies Press.

NRC. 2010a. Adapting to the Impacts of Climate Change. America's Climate Choices: Panel on Adapting to the Impacts of Climate Change. Washington, D.C.: The National Academies. Press.

NRC. 2010b. A Scientific Assessment of Alternatives for Reducing Water Management Effects on Threatened and Endangered Fishes in California's Bay-Delta. Washington, D.C.: The National Academies Press.

NRC. 2010c. Progress Toward Restoring the Everglades: The Third Biennial Review, 2010. Washington, D.C.: The National Academies Press.

National Research Defense Council. Map of the Bay Delta Watershed. Available online at: *http://www.nrdc.org/greengate/water/diverted.html.* Last accessed on April 11, 2011.

Orians, G., and D. Policansky. 2009. Scientific bases of macroenvironmental indicators. Annual Reviews of Environment and Resources 34:375–404.

Pine, W.E., S.J.D. Martell, C. J. Walters, J.E. Kitchell. 2009. Counterintuitive responses of fish populations to managements actions: some common causes and implications for predictions based on ecosystem modeling. Fisheries 34(4):165–180.

RECOVER. 2004. CERP Monitoring and Assessment Plan: Part 1, Monitoring and Supporting Research. Restoration Coordination and Verification, c/o U.S. Army Corps of Engineers, Jacksonville District, Jacksonville, FL and

South Florida Water Management District, West Palm Beach, FL. Available online at: *http://www.evergladesplan.org/pm/recover/recover_map.aspx.* Last accessed March 10, 2011.

RECOVER. 2006a. Comprehensive Everglades Restoration Plan Adaptive Management Strategy. Restoration Coordination and Verification Program, c/o U.S. Army Corps of Engineers, Jacksonville District, Jacksonville, FL and South Florida Water Management District, West Palm Beach, FL. April 2006. Available online at: *http://www.evergladesplan.org/pm/recover/recover_docs/am/rec_am_stategy_brochure.pdf.* Last accessed March 10, 2011.

RECOVER. 2006b. CERP Monitoring and Assessment Plan (MAP): Part 2, 2006 Assessment Strategy for the MAP. Restoration Coordination and Verification, c/o U.S. Army Corps of Engineers, Jacksonville District, Jacksonville, FL and South Florida Water Management District, West Palm Beach, FL.

RECOVER. 2009. CERP Monitoring and Assessment Plan (MAP). Restoration Coordination and Verification, C/O U.S. Army Corps of Engineers, Jacksonville District, Jacksonville, FL and South Florida Water Management District, West Palm Beach, FL. Available online at: *http://www.evergladesplan.org/pm/recover/recover_map.aspx.* Last accessed March 10, 2011.

RECOVER. 2010. CERP Adaptive Management Integration Guide (draft), (2010). C/O U.S. Army Corps of Engineers, Jacksonville District, Jacksonville, FL and South Florida Water Management District, West Palm Beach, FL. Available online at: *http://www.evergladesplan.org/pm/pm_docs/adaptive_mgmt/090110_cerp_amig_v3_4.pdf.* Last accessed March 10, 2011.

Redmond, K.T., and R.W. Koch. 1991. Surface climate and streamflow variability in the western United States and their relationship to large-scale circulation indices. Resources Research 27(9):2381–2399.

Richardson, Jr., J.J. 2010. Conservation easements and adaptive management. Sea Grant Law and Policy Journal, 3(1):31–58.

Rieman B., J.T. Peterson, J. Clayton, P. Howell, R. Thurow, W. Thompson, and D. Lee. 2001. Evaluation of potential effects of federal land management alternatives on trends of salmonids and their habitats in the interior Columbia River basin. Forest Ecology and Management 153:43–62.

Runge, M.C., E. Bean, D.R. Smith, and S. Kokos. 2011. Non-native fish control below Glen Canyon Dam—Report from a structured decision-making project: U.S. Geological Survey Open-File Report 2011–1012. Available online at: *http://pubs.usgs.gov/of/2011/1012/.* Last accessed March 10, 2011.

Sagouspe, J. 2010. Letter to the Honarable David Hayes from Jean Sagouspe, President, Westlands Water District, to David Hayes, Deputy Secretary of the Interior, November 22nd, 2010. Available online at: *http://www.westlandswater.org/wwd%5Cpr%5C2010-11-22_sagouspe_ltr_to_hayes.pdf.* Last accessed on June 21, 2011.

Schemel, L.E., T.R. Sommer, A.B. Muller-Solger, and W.C. Harrell. 2004.

Hydrologic variability, water chemistry, and phytoplankton biomass in a large floodplain of the Sacramento River, CA. Hydrobiologia 513:129–139.

Scholz, J.T., and B. Stiftel. 2005. The challenges of adaptive governance. Pp. 1–11 in Adaptive Governance and Water Conflict: New Institutions for Collaborative Planning. Washington, D.C.: Resources for the Future.

Sommer, T., C. Armor, R. Baxter, R. Breuer, L. Brown, M. Chotkowski, S. Culberson, F. Feyrer, M. Gingras, B. Herbold, W. Kimmerer, A. Mueller-Solger, M. Nobriga, and K. Souza. 2007. The collapse of pelagic fishes in the upper San Francisco Estuary. Fisheries 32(6):270–277.

Suddeth, R., J.F. Mount, and J.R. Lund. 2010. Levee decisions and sustainability for the Sacramento-San Joaquin Delta. San Francisco Estuary and Watershed Science 8(2):1–23.

Susskind, L., A.E. Camacho, and T. Schenk. 2010. Collaborative planning and adaptive management in Glen Canyon: a cautionary tale. Columbia Journal of Environmental Law: 35:1–54.

Tanaka, S. K., C. R. Connell, K. Madani, J. Lund, E. Hanak, and J. Medellin-Azuara 2008. The economic costs and adaptations for alternative Delta regulations. Chapter in Comparing Futures for the Sacramento-San Joaquin Delta, J. Lund, E. Hanak, W. Fleenor, W. Bennett, R. Howitt, J. Mount, and P. Moyle, ed. San Francisco, CA: Public Policy Institute of California.

Thompson, J. 1957. Settlement geography of the Sacramento-San Joaquin Delta, California. Ph.D. Dissertation. Available from: Stanford University.

USACE (U.S. Army Corps of Engineers) and SFWMD (South Florida Water Management District). 1999. The Central and Southern Florida Flood Control Project Comprehensive Review Study Final Integrated Feasibility Report and Programmatic Impact Statement (PEIS). West Palm Beach, FL: South Florida Water Management District. Available online at: *http://www.evergladesplan.org/pub/restudy_eis.cfm#mainreport.* Last accessed March 10, 2011.

USBR (U.S. Bureau of Reclamation). 2008. Appendix R. Sensitivity of Future Central Valley Project and State Water Project Operations to Potential Climate Change and Associated Sea Level Rise, OCAP-BA, July, 2008. Available online at: *http://www.usbr.gov/mp/cvo/OCAP/sep08_docs/Appendix_R.pdf*. Last accessed March 10, 2011**.**

Uusitalo, L. 2007. Advantages and challenges of Bayesian networks in environmental modelling. Ecological Modelling 203:312–318.

Wagner, R.W., M. Stacey, L.R. Brown, and M. Dettinger. 2011. Statistical models of temperature in the Sacramento–San Joaquin Delta under climate-change scenarios and ecological implications. Estuaries and Coasts DOI 10.1007/s12237-010-9369-z. Available on line at: *http://tenaya.ucsd.edu/~dettinge/wagner_delta_temps.pdf.* Last accessed April 13, 2011.

Walters, C.J. 2007. Is adaptive management helping to solve fisheries problems? AMBIO: A Journal of the Human Environment 36(4):304–307.

Williams, B.K., R.C. Szaro, and C.D. Shapiro. 2009. Adaptive Management:

The U.S. Department of the Interior Technical Guide. Washington, D.C.: Department of the Interior, Adaptive Management Working Group.

BDCP1673

Appendixes

Appendix A
Statement of Task

At the request of the U.S. Departments of Interior and Commerce, a National Research Council panel of independent experts will review the draft Bay Delta Conservation Plan (BDCP), which is being prepared through a collaboration of state, federal, and local water agencies, state and federal fish agencies, environmental organizations, and other interested parties to restore the California Bay-Delta ecosystem and protect water supplies, i.e., provide for both species/habitat protection and improved reliability of water supplies.

Specifically, the panel will provide a short report assessing the adequacy of the use of science and adaptive management in the initial public draft of the Bay Delta Conservation Plan (BDCP) by April 2011. This draft, which is currently scheduled for release on November 24th, 2010, will identify a set of water flow and habitat restoration actions to contribute to the recovery of endangered and sensitive species and their habitats in California's Sacramento-San Joaquin Delta while improving water supply reliability.

The panel's review will be related to but be conducted separately from the on-going, more broadly focused National Research Council *Committee on Sustainable Water and Environmental Management in the California Bay-Delta*. The panel's report is expected to contribute to the broader study which will be completed in late 2011.

Appendix B
BDCP Steering Committee Members and Planning Agreement Signature Dates

Entities	***Original Signature Date***	***Amendment Signature Date***
State and Federal Agencies		
California Natural Resources Agency	October 24, 2006	October 27, 2009
California Department of Water Resources	November 14, 2006	December 3, 2009
State Water Resources Control Board (*ex officio*)	See Note	See Note
U.S. Bureau of Reclamation	November 13, 2006	October 30, 2009
U.S. Army Corps of Engineers (*ex officio*)	See Note	See Note
Potential Regulated Entities (PREs)		
Kern County Water Agency	December 6, 2006	January 29, 2010
Metropolitan Water District of Southern California	November 2, 2006	December 3, 2009
Mirant Delta, LLC	December 6, 2006	October 5, 2009
San Luis & Delta-Mendota Water Authority	December 6, 2006	December 6, 2009
Santa Clara Valley Water District	November 20, 2006	November 30, 2009
Westlands Water District	December 6, 2006	December 1, 2009
Zone 7 Water Agency	October 26, 2006	November 30, 2009
Environmental Organizations		
American Rivers	November 8, 2006	January 21, 2010
Defenders of Wildlife	March 15, 2007	January 29, 2010
Environmental Defense Fund	October 30, 2006	January 21, 2010
Natural Heritage Institute	October 25, 2006	November 3, 2009
The Nature Conservancy	November 14, 2006	December 1, 2009
The Bay Institute	July 26, 2007	December 7, 2009
Other Member Agencies		
California Farm Bureau Federation	March 30, 2007	November 11, 2009
Contra Costa Water District	August 3, 2007	January 4, 2010
Friant Water Authority	March 9, 2009	November 18, 2009
North Delta Water Agency	March 12, 2009	October 5, 2009
Fishery Agencies		
California Department of Fish and Game (*ex officio*)	October 24, 2006	October 5, 2009
U.S. Fish and Wildlife Service (*ex officio*)	November 6, 2006	December 3, 2009
National Marine Fisheries Service (*ex officio*)	November 14, 2006	December 3, 2009
Other *Ex Officio* Member Agencies		
Delta Stewardship Council		
Note: The SWRCB and USACE are not signatories of the Planning Agreement.		

SOURCE: Draft BDCP (November 2010).

Appendix C
BDCP Proposed Covered Species and Associated Habitats

No.	*Common Name/ Scientific Name*	*Status (Federal/ State/CNPS)*[1]	*Natural Communities Supporting Species Habitat*
Fish (11 species)			
1	Central Valley steelhead *Oncorhynchus mykiss* DPS	T/-/- DPS Critical Habitat, Recovery Plan[11]	Tidal perennial aquatic, tidal mudflats, tidal brackish emergent wetland, tidal freshwater emergent wetland
2	Sacramento River winter-run Chinook salmon *Oncorhynchus tshawytscha* Evolutionarily Significant Unit (ESU)	E/E/- ESU Critical Habitat, Recovery Plan[11, 12]	Tidal perennial aquatic, tidal mudflats, tidal brackish emergent wetland, tidal freshwater emergent wetland
3	Central Valley spring-run Chinook salmon *Oncorhynchus tshawytscha* ESU	T/T/- ESU Critical Habitat, Recovery Plan[11, 13]	Tidal perennial aquatic, tidal mudflats, tidal brackish emergent wetland, tidal freshwater emergent wetland
4	Central Valley fall- and late fall-run Chinook salmon *Oncorhynchus tshawytscha*	-/SSC/- Recovery Plan[13]	Tidal perennial aquatic, tidal mudflats, tidal brackish emergent wetland, tidal freshwater emergent wetland
5	Delta smelt *Hypomesus transpacificus*	T/T/- Critical Habitat, Recovery Plan[13]	Tidal perennial aquatic, tidal mudflats, tidal brackish emergent wetland, tidal freshwater emergent wetland
6	Longfin smelt *Spirinchus thaleichthys*	-/T/- Recovery Plan[13]	Tidal perennial aquatic, tidal mudflats, tidal brackish emergent wetland, tidal freshwater emergent wetland
7	Sacramento splittail *Pogonichthys macrolepidotus*	-/SSC/- Recovery Plan[13]	Tidal perennial aquatic, tidal mudflats, tidal brackish emergent wetland, tidal freshwater emergent wetland
8	White sturgeon *Acipenser transmontanus*	-/-/-	Tidal perennial aquatic, tidal mudflats, tidal brackish emergent wetland, tidal freshwater emergent wetland

BDCP1673

No.	***Common Name/ Scientific Name***	***Status (Federal/ State/CNPS)[1]***	***Natural Communities Supporting Species Habitat***
9	North American green sturgeon *Acipenser medirostris* Southern DPS	T/SSC/- Southern DPS *Proposed* Critical Habitat, Recovery Plan[13]	Tidal perennial aquatic, tidal mudflats, tidal brackish emergent wetland, tidal freshwater emergent wetland
10	Pacific lamprey *Entosphenus tridentatus*	-/-/-	Tidal perennial aquatic, tidal mudflats, tidal brackish emergent wetland, tidal freshwater emergent wetland
11	River lamprey *Lampetra ayresii*	-/-/-	Tidal perennial aquatic, tidal mudflats, tidal brackish emergent wetland, tidal freshwater emergent wetland
Mammals (6 species)			
12	San Joaquin kit fox *Vulpes macrotis mutica*	E/T/-Recovery Plan[2]	Grassland, Agricultural habitats
13	Riparian woodrat *Neotoma fuscipes riparia*	E/SSC/- Recovery Plan[2]	Valley/foothill riparian
14	Salt marsh harvest mouse *Reithrodontomys raviventris*	E/E,FP/- Recovery Plan[3, 4]	Tidal brackish emergent wetland, managed wetlands, grassland
15	Riparian brush rabbit *Sylvilagus bachmani riparius*	E/E/-Recovery Plan[2]	Valley/foothill riparian
16	Townsend's big-eared bat *Corynorhinus townsendii*	-/SSC/-	All natural communities
17	Suisun shrew *Sorex ornatus sinuosus*	-/SSC/- Recovery Plan[3]	Tidal brackish emergent wetland, managed wetlands
Birds (12 species)			
18	Tricolored blackbird *Agelaius tricolor*	-/SSC/-	Tidal brackish emergent wetland, tidal freshwater emergent wetland, valley/foothill riparian, alkali seasonal wetland complex, managed wetlands, other natural seasonal wetlands, grassland, agricultural habitats
19	Suisun song sparrow *Melospiza melodia maxillaries*	-/SSC/- Recovery Plan[4]	Tidal brackish emergent wetland, tidal freshwater emergent wetland, managed wetlands
20	Yellow-breasted chat *Icteria virens*	-/SSC/-	Valley/foothill riparian
21	Least Bell's vireo *Vireo bellii pusillus*	E/E/- Recovery Plan[5]	Valley/foothill riparian

No.	***Common Name/ Scientific Name***	***Status (Federal/ State/CNPS)*** [1]	***Natural Communities Supporting Species Habitat***
22	Western burrowing owl *Athene cunicularia hypugaea*	-/SSC/-	Grassland, alkali seasonal wetland complex, vernal pool complex, managed wetland, other natural seasonal wetlands, agricultural habitats
23	Western yellow-billed cuckoo *Coccyzus americanus occidentalis*	C/E/-	Valley/foothill riparian
24	California least tern *Sternula antillarum browni*	E/E/-Recovery Plan[6]	Tidal perennial aquatic
25	Greater sandhill crane *Grus canadensis tabida*	-/T,FP/-	Agricultural habitats, alkali seasonal wetland complex, vernal pool complex, managed wetlands, other natural seasonal wetlands, grassland
26	California black rail *Laterallus jamaicensis coturniculus*	-/T,FP/- Recovery Plan[4]	Tidal brackish emergent wetland, tidal freshwater emergent wetland, nontidal freshwater permanent emergent wetland
27	California clapper rail *Rallus longirostris obsoletus*	E/E,FP/- Recovery Plan[3, 4]	Tidal brackish emergent wetland
28	Swainson's hawk *Buteo swainsoni*	-/T/-	Valley/foothill riparian, agricultural habitats, grassland, alkali seasonal wetland complex, vernal pool complex, managed wetlands, other natural seasonal wetlands
29	White-tailed kite *Elanus leucurus*	-/FP/-	Valley/foothill riparian, agricultural habitats, grassland, alkali seasonal wetland complex, vernal pool complex, managed wetlands, other natural seasonal wetlands
Reptiles (2 species)			
30	Giant garter snake *Thamnophis gigas*	T/T/-Recovery Plan[6]	Tidal perennial aquatic, tidal freshwater emergent wetland, nontidal perennial aquatic, nontidal freshwater permanent emergent wetland, alkali seasonal wetland complex, vernal pool complex, managed wetlands, other natural seasonal wetlands, grassland, agricultural habitats

No.	Common Name/ Scientific Name	Status (Federal/ State/CNPS)[1]	Natural Communities Supporting Species Habitat
31	Western pond turtle *Actinemys* (formerly *Clemmys* and *Emys*) *marmorata*	-/SSC/-	Tidal perennial aquatic, tidal freshwater emergent wetland, tidal brackish emergent wetland, nontidal perennial aquatic, nontidal freshwater permanent emergent wetland, valley/foothill riparian, alkali seasonal wetland complex, vernal pool complex, managed wetlands, other natural seasonal wetlands, grassland, agricultural habitats
Amphibians (3 species)			
32	California red-legged frog *Rana draytonii*	T/SSC/-Critical Habitat, Recovery Plan[8]	Valley/foothill riparian, nontidal freshwater permanent emergent wetland, tidal freshwater emergent wetland, nontidal perennial aquatic, managed wetlands, grassland, alkali seasonal wetland complex, vernal pool complex, other natural seasonal wetlands, agricultural habitats
33	Western spadefoot toad *Spea hammondii*	-/SSC/- Recovery Plan[9]	Grassland, alkali seasonal wetland complex, vernal pool complex, other natural seasonal wetlands, nontidal perennial aquatic
34	California tiger salamander *Ambystoma californiense* Central Valley Distinct Population Segment (DPS)	T/T/-Central Valley DPS Critical Habitat	Vernal pool complex, alkali seasonal wetland complex, other natural seasonal wetlands, grassland
Invertebrates (8 species)			
35	Lange's metalmark butterfly *Apodemia mormo langei*	E/-/-Recovery Plan[15]	Inland dune scrub
36	Valley elderberry longhorn beetle *Desmocerus californicus dimorphus*	T/-/-Recovery Plan[14]	Valley/foothill riparian, grassland
37	Vernal pool tadpole shrimp *Lepidurus packardi*	E/-/-Critical Habitat Recovery Plan[9]	Vernal pool complex
38	Conservancy fairy shrimp *Branchinecta conservatio*	E/-/-Critical Habitat Recovery Plan[9]	Vernal pool complex
39	Longhorn fairy shrimp *Branchinecta longiantenna*	E/-/-Recovery Plan[9]	Vernal pool complex
40	Vernal pool fairy shrimp *Branchinecta lynchi*	T/-/-Critical Habitat Recovery Plan[9]	Vernal pool complex

No.	*Common Name/ Scientific Name*	*Status (Federal/ State/CNPS)[1]*	*Natural Communities Supporting Species Habitat*
41	Midvalley fairy shrimp *Branchinecta mesovallensis*	-/-/- Recovery Plan[9]	Vernal pool complex
42	California linderiella *Linderiella occidentalis*	-/-/- Recovery Plan[9]	Vernal pool complex
Plants (21 species)			
43	Alkali milk-vetch *Astragalus tener* var. *tener*	-/-/1B Recovery Plan[9]	Vernal pool complex
44	Heartscale *Atriplex cordulata*	-/-/1B	Alkali seasonal wetland complex, vernal pool complex, grassland
45	Brittlescale *Atriplex depressa*	-/-/1B	Alkali seasonal wetland complex, vernal pool complex, grassland
46	San Joaquin spearscale *Atriplex joaquiniana*	-/-/1B	Alkali seasonal wetland complex, vernal pool complex, grassland
47	Slough thistle *Cirsium crassicaule*	-/-/1B	Valley/foothill riparian
48	Suisun thistle *Cirsium hydrophilum var.hydrophilum*	E/-/1B Critical Habitat Recovery Plan[4]	Tidal brackish emergent wetland
49	Soft bird's-beak *Cordylanthus mollis ssp. mollis*	E/R/IB Critical Habitat Recovery Plan[4]	Tidal brackish emergent wetland
50	Dwarf downingia *Downingia pusilla*	-/-/2	Vernal pool complex
51	Delta button-celery *Eryngium racemosum*	-/E/1B	Alkali seasonal wetland complex, vernal pool complex, valley/foothill riparian, grassland
52	Contra Costa wallflower *Erysimum capitatum var. angustatum*	E/E/1B Critical Habitat Recovery Plan[15]	Inland dune scrub
53	Boggs Lake hedge-hyssop *Gratiola heterosepala*	-/E/1B Recovery Plan[9]	Vernal pool complex
54	Carquinez goldenbush *Isocoma arguta*	-/-/1B	Alkali seasonal wetland complex, grassland
55	Delta tule pea *Lathyrus jepsonii var. jepsonii*	-/-/1B Recovery Plan[4]	Tidal brackish emergent wetland, tidal freshwater emergent wetland, valley/foothill riparian
56	Legenere *Legenere limosa*	-/-/1B Recovery Plan[9]	Vernal pool complex
57	Heckard's peppergrass *Lepidium latipes var. heckardii*	-/-/1B	Vernal pool complex
58	Mason's lilaeopsis *Lilaeopsis masonii*	-/R/1B	Tidal mudflats, tidal brackish emergent wetland, tidal freshwater emergent wetland, valley/foothill riparian

No.	***Common Name/ Scientific Name***	***Status (Federal/ State/CNPS)[1]***	***Natural Communities Supporting Species Habitat***
59	Delta mudwort *Limosella subulata*	-/-/2	Tidal mudflats, tidal brackish emergent wetland, tidal freshwater emergent wetland, valley/foothill riparian
60	Antioch Dunes evening-primrose *Oenothera deltoides ssp. howellii*	E/E/1B Critical Habitat Recovery Plan[15]	Inland dune scrub
61	Side-flowering skullcap *Scutellaria lateriflora*	-/-/2	Valley/foothill riparian
62	Suisun Marsh aster *Symphyotrichum* (formerly *Aster lentus) lentum*	-/-/1B	Tidal brackish emergent wetland, tidal freshwater emergent wetland, valley/foothill riparian
63	Caper-fruited tropidocarpum *Tropidocarpum capparideum*	-/-/1B	Grassland

Note: This table provides the current list of proposed covered species. Additional species may be added and some of the species presented here may be removed from the covered species list as per continuing development of the BDCP.

1Status:
Federal
E = Listed as endangered under ESA
T = Listed as threatened under ESA
C = Candidate for listing under ESA
State
E = Listed as endangered under CESA
T = Listed as threatened under CESA
R = Listed as rare under the California Native Plant Protection Act
SSC = California species of special concern
FP = Fully protected under the California Fish and Game Code
California Native Plant Society (CNPS)
1B = rare or endangered in California and elsewhere
2 = rare and endangered in California, more common elsewhere

2U.S. Fish and Wildlife Service. 1998. Recovery plan for upland species of the San Joaquin Valley, California. Region 1, Portland,
OR. 319 pp.

3U.S. Fish and Wildlife Service. 1984. Salt marsh harvest mouse and California clapper rail recovery plan. Portland, OR.

4U.S. Fish and Wildlife Service. 2009. Draft Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California.
Sacramento, California. xviii+636 pp.

5U.S. Fish and Wildlife Service. 1998. Draft recovery plan for the least Bell's vireo. U.S. Fish and Wildlife Service, Portland, OR.
139 pp.

6U.S. Fish and Wildlife Service. 1985. Recovery plan for the California least tern, Sterna antillarum browni. U.S. Fish and
Wildlife Service, Portland, OR. 112 pp.

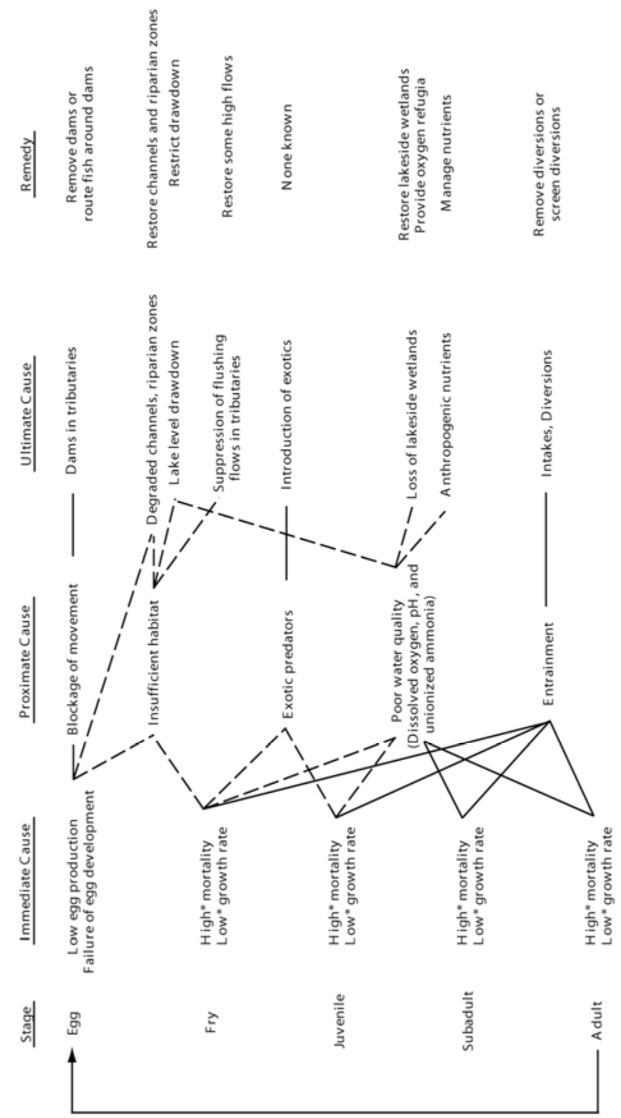
7U.S. Fish and Wildlife Service. 1999. Draft Recovery Plan for the Giant Garter Snake (Thamnopsis gigas). U.S. Fish and Wildlife
Service, Portland, Pregon. ix+192 pp.

8U.S. Fish and Wildlife Service. 2002. Recovery Plan for the California Red-legged Frog (Rana aurora draytonii). U.S. fish and
Wildlife Service, Portland, Oregon. viii+173 pp.

9U.S. Fish and Wildlife Service. 2005. Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon. Portland,
Oregon. xxvi + 606 pages.
10California Tiger Salamander distinct population segments are federally listed as endangered in Sonoma and Santa Barbara
counties.
11National Marine Fisheries Service. 2009. Public Draft Recovery Plan for the Evolutionarily Significant Units of Sacramento
River Winter-run Chinook Salmon and Central Valley Spring-run Chinook Salmon and the Distinct Population Segment of
Central Valley Steelhead. Sacramento Protected Resources Division. October 2009.
12National Marine Fisheries Service. 1997. NMFS Proposed Recovery Plan for the Sacramento River winter-run Chinook Salmon.
NMFS Southwest Region. Long Beach, CA.
13U.S. Fish and Wildlife Service. 1995. Sacramento-San Joaquin Delta Native Fishes Recovery Plan. U.S. Fish and Wildlife
Service, Portland, Oregon.
14U.S. Fish and Wildlife Service. 1984. Valley elderberry longhorn beetle Recovery Plan. U.S. Fish and Wildlife Service, Portland,
Oregon. 62 pp.
15U.S. Fish and Wildlife Service. 1984. Revised recovery plan for three endangered species endemic to Antioch Dunes, California.
16U.S. Fish and Wildlife Service, Portland, Oregon

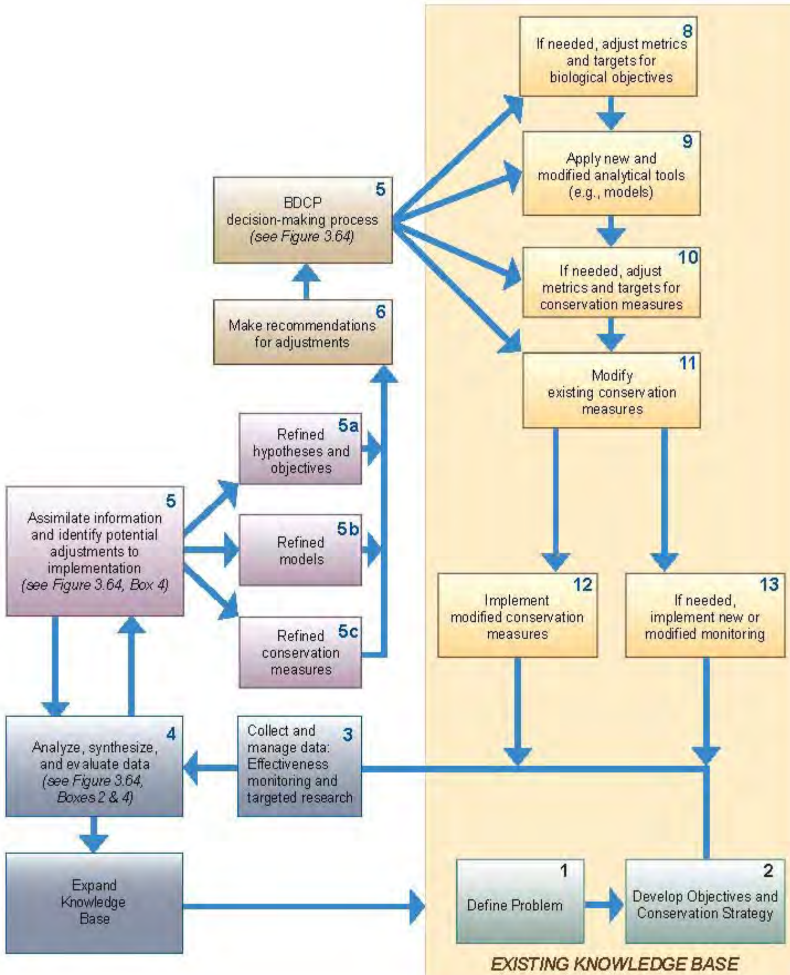
SOURCE: BDCP (Bay Delta Conservation Plan Steering Committee). 2010. Bay Delta Conservation Plan Working Draft. November 18. Available online at: *http://www.resources.ca.gov/bdcp/*. Last accessed April 26, 2011.

Appendix D
Possible Causal Connections in Suppression of Populations of Endangered Suckers in Upper Klamath Lake



NOTE: Solid line = connection verified scientifically; dashed line = connection under study. "High mortality" and "low growth rate" are relative to rates in stable populations.
SOURCE: NRC, 2004a

Appendix E
BDCP Adaptive Management Process Framework



SOURCE: Draft BDCP (November 2010).

Appendix F
Water Science and Technology Board

Appendix G
Ocean Studies Board

DONALD F. BOESCH, *Chair*, University of Maryland Center for Environmental Science, Cambridge
EDWARD A. BOYLE, Massachusetts Institute of Technology, Cambridge
CORTIS K. COOPER, Chevron Corporation, San Ramon, CA
JORGE E. CORREDOR, University of Puerto Rico, Mayaguez
KEITH R. CRIDDLE, University of Alaska Fairbanks, Juneau
JODY W. DEMING, University of Washington, Seattle
ROBERT HALLBERG, NOAA/GFDL and Princeton University, NJ
DEBRA HERNANDEZ, SECOORA, Mt. Pleasant, SC
ROBERT A. HOLMAN, Oregon State University, Corvallis
KIHO KIM, American University, Washington, DC
BARBARA A. KNUTH, Cornell University, Ithaca, NY
ROBERT A. LAWSON, Science Applications International Corporation, San Diego, CA
GEORGE I. MATSUMOTO, Monterey Bay Aquarium Research Institute, Moss Landing, CA
JAY S. PEARLMAN, The Boeing Company (retired), Port Angeles, WA
ANDREW A. ROSENBERG, Conservation International, Arlington, VA
DANIEL L. RUDNICK, Scripps Institution of Oceanography, La Jolla, CA
ANNE M. TREHU, Oregon State University, Corvallis
PETER L. TYACK, Woods Hole Oceanographic Institution, MA
DON WALSH, International Maritime Incorporated, Myrtle Point, OR
DAWN J. WRIGHT, Oregon State University, Corvallis
JAMES A. YODER, Woods Hole Oceanographic Institution, MA
Ex-Officio
MARY (MISSY) H. FEELEY, ExxonMobil Exploration Company, Houston, TX

Staff
SUSAN ROBERTS, Board Director
CLAUDIA MENGELT, Senior Program Officer
KIM WADDELL, Senior Program Officer
DEBORAH GLICKSON, Senior Program Officer
MARTHA MCCONNELL, Program Officer
SHUBHA BANSKOTA, Financial Associate
PAMELA LEWIS, Administrative Coordinator
SHERRIE FORREST, Research Associate
JEREMY JUSTICE, Senior Program Assistant
EMILY OLIVER, Program Assistant
PETER THOMPSON, Mirzayan Fellow

Appendix H
Panel Biographical Information

HENRY J. VAUX, JR., ***Chair***, is Professor Emeritus of Resource Economics at both the University of California in Berkley and Riverside. He is also Associate Vice President Emeritus of the University of California system. He also previously served as director of California's Center for Water Resources. His principal research interests are the economics of water use, water quality, and water marketing. Prior to joining the University of California, he worked at the Office of Management and Budget and served on the staff of the National Water Commission. Dr. Vaux has served on the NRC committees on Assessment of Water Resources Research, Western Water Management, and Ground Water Recharge, and Sustainable Underground Storage of Recoverable Water. He was chair of the Water Science and Technology Board from 1994 to 2001. He is a National Associate of The National Academies. Dr. Vaux received an A.B. from the University of California, Davis in biological sciences, an M.A. in natural resource administration, and an M.S. and Ph.D. in economics from the University of Michigan.

MICHAEL E. CAMPANA is Professor of Geosciences at Oregon State University (OSU), former Director of its Institute for Water and Watersheds, and Emeritus Professor of Earth and Planetary Sciences at the University of New Mexico. Prior to joining OSU in 2006 he held the Albert J. and Mary Jane Black Chair of Hydrogeology and directed the Water Resources Program at the University of New Mexico, was a research hydrologist at the Desert Research Institute, and taught in the University of Nevada-Reno's Hydrologic Sciences Program. He has supervised 70 graduate students. His research and interests include hydrophilanthropy, water resources management and policy, communications, transboundary water resources, hydrogeology, and environmental fluid mechanics, and he has published on a variety of topics. Dr. Campana was a Fulbright Scholar to Belize and a Visiting Scientist at Research Institute for Groundwater (Egypt) and the IAEA in Vienna. Central America and the South Caucasus are the current foci of his international work. He has served on six NRC-NAS committees. Dr. Campana is Founder, President, and Treasurer of the Ann Campana Judge Foundation (www.acjfoundation.org), a 501(c)(3) charitable foundation that funds and undertakes projects related to water, sanitation, and hygiene (WASH) in Central America. He operates the WaterWired blog and Twitter. He earned a B.S. in geology from the College of William and Mary, and M.S. and Ph.D. degrees in hydrology from the University of Arizona.

JEROME B. GILBERT is a consulting engineer and founder of J. Gilbert, Inc. His interests include integrated water supply and water quality planning and management. Mr. Gilbert has managed local and regional utilities, and he has developed basin/watershed water quality and protection plans. He has supervised California's water rights and water quality planning and regulatory activities, chaired the San Francisco Bay Regional Water Quality Control Board, and led national and international water and water research associations. Areas of experience include: authorship of state and national water legislation on water rights, pollution control, water conservation and urban water management; optimization of regional water project development; groundwater remediation and conjunctive use; economic analysis of alternative water improvement projects; and planning of multipurpose water management efforts including remediation. He has served on national panels related to control and remediation of ground and surface water contamination, and the National Drinking Water Advisory Council. Mr. Gilbert is a member of the National Academy of Engineering. He received his B.S. from the University of Cincinnati and an M.S. from Stanford University.

ALBERT E. GIORGI is President and Senior Fisheries Scientist at BioAnalysts, Inc. in Redmond, Washington. He has been conducting research on Pacific Northwest salmonid resources since 1982. Prior to 1982, he was a research scientist with NOAA in Seattle, Washington. He specializes in fish passage migratory behavior, juvenile salmon survival studies, and biological effects of hydroelectric facilities and operation. His research includes the use of radio telemetry, acoustic tags, and PIT-tag technologies. In addition to his research, he acts as a technical analyst and advisor to public agencies and private parties. He regularly teams with structural and hydraulic engineers in the design and evaluation of fishways and fish bypass systems. He also has served on the NRC Committee on Water Resources Management, Instream Flows, and Salmon Survival in the Columbia River. He received his B.A. and M.A. in biology from Humboldt State University and his Ph.D. in fisheries from the University of Washington.

ROBERT J. HUGGETT is an independent consultant and Professor Emeritus and former Chair of the Department of Environmental Sciences, Virginia Institute of Marine Sciences at the College of William and Mary, where he was on the faculty for more than 20 years. He also served as Professor of Zoology and Vice President for Research and Graduate Studies at Michigan State University from 1997 to 2004. Dr. Huggett is an expert in aquatic biogeochemistry and ecosystem management whose research involved the fate and effects of hazardous substances in aquatic systems. From 1994 to 1997, he was the Assistant Administrator for Research and Development for the U.S. Environmental Protection Agency (EPA, where his responsibilities included planning and directing the agency's research program. During his time at the EPA, he served as Vice Chair of the Committee on Environment and Natural Resources and Chair of the Subcommittee on Toxic Substances and Solid Wastes, both of the White House Office of Science and Technology Policy. Dr. Huggett founded the EPA Star Competitive Research Grants program and the EPA Star Graduate Fellowship program. He has served on the National Research Council's (NRC) Board on Environmental Studies and Toxicology, the Water Science and Technology Board, and numerous study committees on wide ranging topics. Dr. Huggett earned an M.S. in marine chemistry from the Scripps Institution of Oceanography at the University of California at San Diego and completed his Ph.D. in marine science at the College of William and Mary.

CHRISTINE A. KLEIN is the Chesterfield Smith Professor of Law at the University of Florida Levin College of Law, where she has been teaching since 2003. She offers courses on natural resources law, environmental law, water law, and property. Previously, she was a member of the faculty of Michigan State University College of Law, where she served as Environmental Law Program Director. From 1989 to 1993, she was an Assistant Attorney General in the Office of the Colorado Attorney General, Natural Resources Section, where she specialized in water rights litigation. She has published widely on a variety of water law and natural resources law topics. She holds a B.A. from Middlebury College, Vermont; a J.D. from the University of Colorado School of Law; and an LL.M. from Columbia University School of Law, New York.

SAMUEL N. LUOMA is an emeritus Senior Research Hydrologist in the Water Resources Division of the U.S. Geological Survey, where he worked for 34 years. Dr. Luoma's research centers on sediment processes, both natural and human-induced, particularly in the San Francisco Bay area. He served as the first lead on the CALFED Bay-delta program and is the Editor-in-Chief of San Francisco Estuary & Watershed Science. Since 1992, he has published extensively on the bioavailability and ecological effects of metals in aquatic environments. He has helped refine approaches to determine the toxicity of marine and estuarine sediments. In 1999, he was invited to discuss how chemical speciation influences metal bioavailability in sediments for the European Science Foundation. He has served multiple times on the EPA's Science Advisory Board Subcommittee on Sediment Quality Criteria and on several NRC committees. Dr. Luoma received his B.S. and M.S. in zoology from Montana State University, Bozeman, and his Ph.D. in marine biology from the University of Hawaii, Honolulu.

THOMAS MILLER is Professor of Fisheries and Bioenergetics and Population Dynamics at the Chesapeake Biological Laboratory, University of Maryland Center for Environmental Science (UMCES-CBL), where he has been teaching since 1994. Prior to UMCES-CBL, he was a postdoctoral fellow at McGill University, Montreal, Canada, and research specialist with the Center for Great Lakes Studies, University of Wisconsin, Milwaukee. His research focuses on population dynamics of aquatic animals, particularly in understanding recruitment, feeding and bio-physical interactions, and early life history of fish and crustaceans. He has been involved in the development of a Chesapeake Bay fishery ecosystem plan, which includes detailed background information on fisheries, foodwebs, habitats and monitoring required to develop multispecies stock assessments. Most recently, he has developed an interest in the sub-lethal effects of contamination on Chesapeake Bay living resources using population dynamic approaches. He received his B.Sc. (hons) in human and environmental biology from the University of York, UK, and his M.S. in ecology and Ph.D. in zoology and oceanography from North Carolina State University.

STEPHEN G. MONISMITH is Professor of Environmental Fluid Mechanics and directs the Environmental Fluid Mechanics Laboratory at Stanford University. Prior to coming to Stanford, he spent 3 years in Perth (Australia) as a research fellow at the University of Western Australia. Dr. Monismith's research in environmental and geophysical fluid dynamics involves the application of fluid mechanics principles to the analysis of flow processes operating in rivers, lakes, estuaries and the oceans. Making use of laboratory experimentation, numerical modelling, and field measurements, his current research includes studies of estuarine hydrodynamics and mixing processes, flows over coral reefs, wind wave-turbulent flow interactions in the upper ocean, turbulence in density stratified fluids, and physical-biological interactions in phytoplankton and benthic systems. He received his B.S., M.S., and Ph.D. from the University of California at Berkeley.

JAYANTHA OBEYSEKERA directs the Hydrologic & Environmental Systems Modeling Department at the South Florida Water Management District, where he is a lead member of a modeling team dealing with development and applications of computer simulation models for Kissimmee River restoration and the restoration of the Everglades Ecosystem. Prior to joining the South Florida Water Management District, he taught courses in hydrology and water resources at Colorado State University, Fort Collins; George Washington University, Washington, DC; and at Florida Atlantic University, Boca Raton, Florida. Dr. Obeysekera has published numerous research articles in refereed journals in the field of water resources. Dr. Obeysekera has more than 20 years of experience practicing water resources engineering with an emphasis on both stochastic and deterministic modeling. He has taught short courses on modeling in the Dominican Republic, Colombia, Spain, Sri Lanka, and the United States. He was a member of the Surface Runoff Committee of the American Geophysical Union and is currently serving as a member of a Federal Task Group on Hydrologic Modeling. He served as member of NRC's Committee on Further Studies of Endangered and Threatened Fishes in the Klamath River. Dr. Obeysekera has a B.S. degree in civil engineering from University of Sri Lanka; M.E. in hydrology from University of Roorkee, India; and Ph.D. in civil engineering with specialization in water resources from Colorado State University.

HANS W. PAERL is Kenan Professor of Marine and Environmental Sciences, at the University of North Carolina Chapel Hill Institute of Marine Sciences, Morehead City. His research includes microbially mediated nutrient cycling and primary production dynamics of aquatic ecosystems, environmental controls of harmful algal blooms, and assessing the causes and consequences of man-made and climatic (storms, floods) nutrient enrichment and hydrologic alterations of inland, estuarine, and coastal waters. His studies have identified the importance and ecological impacts of atmospheric nitrogen deposition as a new nitrogen source supporting estuarine and coastal eutrophication. He is involved in the development and application of microbial and biogeochemical indicators of aquatic ecosystem condition and change in response to human and climatic perturbations. He heads up the Neuse River Estuary Modeling and Monitoring Program, and ferry-based water quality monitoring program, FerryMon, which employs environmental sensors and a various microbial indicators to assess near real-time ecological condition of the Pamlico Sound System, the nation's second largest estuarine complex. In 2003 he was awarded the G. Evelyn Hutchinson Award by the American Society of Limnology and Oceanography for his work in these fields and their application to interdisciplinary research, teaching and management of aquatic ecosystems. He received his PhD from the University of California-Davis.

MAX J. PFEFFER is International Professor of Development Sociology and senior Associate Dean of the College of Agriculture and Life Sciences at Cornell University. His teaching concentrates on environmental sociology and sociological theory. His research spans several areas including farm labor, rural labor markets, international migration, land use, and environmental planning. The empirical work covers a variety of rural and urban communities, including rural/urban fringe areas. Research sites include rural New York and Central America. He has been awarded competitive grants from the National Institutes of Health, the National Science Foundation, the U.S. Environmental Protection Agency, the U.S. Department of Agriculture's National Research Initiative and its Fund for Rural America, and the Social Science Research Council. Dr. Pfeffer has published a wide range of scholarly articles and has written or co-edited four books. He recently published (with John Schelhas) *Saving Forests, Protecting People? Environmental Conservation in Central America.* He also previously served as the Associate Director of both the Cornell University Agricultural Experiment Station and the Cornell University Center for the Environment. He received his Ph.D. degree in sociology from the University of Wisconsin, Madison.

DESIREE D. TULLOS is Assistant Professor in the Department of Biological and Ecological Engineering, Oregon State University, Corvallis. Dr. Tullos also consulted with Blue Land Water Infrastructure and with Barge, Waggoner, Sumner, and Cannon before joining the faculty at Oregon State University. Her research areas include ecohydraulics, river morphology and restoration, bioassessment, and habitat and hydraulic modeling. She has done work on investigations of biological responses to restoration and engineered applications in riverine ecosystems; development and evaluation of targeted and appropriate bioindicators for the assessment of engineered designs in riverine systems; assessing effects of urban and agricultural activities and management practices on aquatic ecosystem stability in developing countries. She received her B.S. in civil engineering from the University of Tennessee, Knoxville, and her MC.E. in civil engineering and Ph.D. in biological engineering from North Carolina State University, Raleigh.

STAFF

LAURA J. HELSABECK is a Staff Officer with the National Research Council's Water Science and Technology Board. Her interests include the use of scientific information to enhance water policy and management decisions pertaining to water quality and quantity. Since joining the National Research Council, she has directed studies for a variety of topics including the Committee on Challenges and Opportunities in the Hydrology Sciences and the Committee on U.S. Geologic Survey's Water Resources Research. Dr. Helsabeck received her B.A. from Clemson University, her M.S. from Vanderbilt University, and Ph.D. from The Ohio State University in Environmental Science. Her dissertation work, Ibuprofen photolysis: Reaction kinetics, chemical mechanism, and byproduct analysis, was awarded the Ellen C. Gonter Environmental Chemistry Award by the American Chemical Society.

DAVID POLICANSKY is a Scholar with the Board on Environmental Studies and Toxicology at the National Research Council, where he directs studies on applied ecology and natural resource management. He chairs the Advisory Council for the University of Alaska's School of Fisheries and Ocean Sciences and was a 2001 Harriman Scholar on the retracing of the 1899 Harriman Alaska Expedition. His research interests include genetics; evolution; and ecology, including the effects of fishing on fish populations; ecological risk assessment; natural resource management; and how science is used in informing policy. He has directed more than 30 projects at the National Research Council on natural resources and ecological risk assessment.

BDCP1673