DRIVERS OF CHANGE IN ESTUARINE-COASTAL ECOSYSTEMS: DISCOVERIES FROM FOUR DECADES OF STUDY IN SAN FRANCISCO BAY

James E. Cloern[1] and Alan D. Jassby[2]

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[1] Poised at the interface of rivers, ocean, atmosphere and dense human settlement, estuaries are driven by a large array of natural and anthropogenic forces. San Francisco Bay exemplifies the fast-paced change occurring in many of the world's estuaries, bays, and inland seas in response to these diverse forces. We use observations from this particularly well-studied estuary to illustrate responses to six drivers that are common agents of change where land and sea meet: water consumption and diversion, human modification of sediment supply, introduction of nonnative species, sewage input, environmental policy, and climate shifts. In San Francisco Bay, responses to these drivers include, respectively, shifts in the timing and extent of freshwater inflow and salinity intrusion, decreasing turbidity, restructuring of plankton communities, nutrient enrichment, elimination of hypoxia and reduced metal contamination of biota, and food web changes that decrease resistance of the estuary to nutrient pollution. Detection of these changes and discovery of their causes through environmental monitoring have been essential for establishing and measuring outcomes of environmental policies that aim to maintain high water quality and sustain services provided by estuarine-coastal ecosystems. The many time scales of variability and the multiplicity of interacting drivers place heavy demands on estuarine monitoring programs, but the San Francisco Bay case study illustrates why the imperative for monitoring has never been greater.

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1. INTRODUCTION

[2] Four decades ago, the study of ecosystems was emerging as a scientific discipline to understand how biological communities and their physical environment are organized spatially and how they change over time. A classic paper from that era was Eugene P. Odum's "The strategy of ecosystem development," which depicts ecosystem change over time as an orderly process of community development culminating in a stable system and where stability is maintained by the evolution of complex biological structure and its "increased control of, and homeostasis with, the physical system" [*Odum*, 1969, p. 262]. Odum described ecosystem development as successional stages from immature to mature communities, where the stages are *predictable*, include *subtle* changes in food webs, and climax to a *steady state*. But Odum also pointed out that "[s]evere stress or rapid changes brought about by outside forces can, of course, rob the system of these protective mechanisms" and "[m]ost physical stresses introduced by man are too sudden, too violent, or too arrhythmic for adaptations to occur at the ecosystem level, so severe oscillation rather than stability results" [*Odum*, 1969, p. 264, p. 268] Odum's narrative description of unperturbed mature ecosystems provides a benchmark from which we can judge the extent to which ecosystems are altered by severe outside forces, including anthropogenic ones.

[3] This review is about change in marine ecosystems connected to land, such as estuaries, bays, and lagoons. Odum and his contemporaries had access to long-term (multidecadal) records of biological and environmental variability from terrestrial [*Baltensweiler*, 1964] and marine [*Southward*, 1995] ecosystems, but in the late 1960s there were few observational records documenting decadal-scale changes in estuaries. Numerous coastal research and monitoring programs began in the 1970s and 1980s, however, and have been sustained long enough to provide empirical bases for comparison against Odum's attributes of mature unperturbed ecosystems. We

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[1] U.S. Geological Survey, Menlo Park, California, USA.

[2] Department of Environmental Science and Policy, University of California, Davis, California, USA.

Corresponding author: J. E. Cloern, U.S. Geological Survey,
345 Middlefield Rd., MS 496, Menlo Park, CA 94025, USA.
(jecloern@usgs.gov)

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show 10 examples (Figure 1) to illustrate changes in sediment supply to the Yangtze Estuary, China [*Li et al.*, 2012]; dissolved oxygen concentration and phosphorus input to the Potomac Estuary, USA [*Jaworski et al.*, 2007]; diatom productivity (Si uptake) in northern San Francisco Bay, USA [*Kimmerer*, 2005]; optical properties of Chesapeake Bay, USA [*Gallegos et al.*, 2011]; areal extent of submerged aquatic vegetation in the lower Potomac River, USA [*Orth et al.*, 2010]; biomass of benthic invertebrates in Denmark's Ringkøbing Fjord [*Petersen et al.*, 2008]; mercury content of mussels in the Forth Estuary, UK [*Dobson*, 2000]; annual landings of pelagic fish in the Limfjorden, Denmark [*Riisgård*, 2012]; and abundance of shorebirds (black-tailed godwits) in the Tagus Estuary, Portugal [*Catry et al.*, 2011].

[4] These examples are representative of the substantial and rapid physical, biogeochemical, and biological changes that have occurred in many of the world's estuarine-coastal ecosystems in recent decades. They reveal complex and diverse patterns of change as monotonic trends or abrupt shifts (up or down), oscillations, and multiyear peaks in plant and animal abundance, and they show that estuarine-coastal environments and their biological communities are changing at a fast pace. Many of these changes were surprises and did not occur as *predictable* successional stages, none could be classified as *subtle*, and the large trends, step changes, and high-amplitude oscillations are not characteristic of *steady state*. Therefore, empirical observations from the world's estuaries, bays, and lagoons reveal dynamics distinctly different from Odum's description of how ecosystems develop in the absence of unusual external forces.

[5] Our purpose is to use a suite of observations collected in a particularly well-studied estuary to review what has been learned in recent decades about the external forces that drive diverse and rapid changes at the land-sea interface. We describe changes in hydrology, sediment dynamics, biological communities, and water quality that have been captured in sampling programs sustained over multiple decades in San Francisco Bay, its watershed, and the adjacent coastal Pacific Ocean. We selected examples where change can be attributed to a specific driver or pressure—a human action or climatic forcing that brings about fundamental change at the ecosystem scale. Identification of drivers is a key to understanding past changes, and it provides the foundation for anticipating and adapting to future changes [*Jackson*, 2007]. Aided by rich observational records, we use the San Francisco Bay case study to illustrate responses to six drivers that are common agents of change in the world's estuarine-coastal systems: consumption and diversion of fresh water, modification of sediment supply, introduction of nonnative species, sewage input, environmental policy, and climate shifts.

2. DESCRIPTION OF THE ESTUARY

[6] San Francisco Bay is an estuary, a coastal bay where seawater is measurably diluted with fresh water from land drainage [*Pritchard*, 1967]. Seawater enters through the narrow deep channel at the Golden Gate (Figure 2), and its chemical and biological constituents are influenced by seasonal upwelling in the adjacent coastal boundary current. Fresh water is delivered primarily by the Sacramento and San Joaquin rivers, which carry runoff produced in the 163,000 km^2 watershed bounded by the Cascade and Sierra Nevada mountains. Annual runoff is highly variable (Figure 3). During the last century, for example, annual runoff ranged from a low of 7.6 km^3 in 1977 to a high of 65 km^3 in 1983, both El Niño years. Runoff is also highly seasonal, reflecting a climate of wet winters and dry summers.

[7] San Francisco Bay is the defining geographic feature of the "Bay Area," home to 7.5 million people (Figure 4). California's urban population centers and agricultural production are largely dependent upon water diverted from the estuary. The Bay moderates regional climate, assimilates wastewater from 50 municipal sewage treatment plants [*van Geen and Luoma*, 1999], is a center of commercial shipping, serves as both nursery and migration route for ocean-harvested fish and crabs, and includes the largest tidal wetland restoration project in the western United States [*Thébault et al.*, 2008]. San Francisco Bay supports 30% of shorebird populations and up to half of diving duck populations in the Pacific Flyway [*Takekawa et al.*, 2001]. The Bay and its tributary rivers and wetlands provide habitat for threatened and endangered species of fish (Chinook salmon, *Oncorhynchus tshawytscha*; steelhead trout, *Oncorhynchus mykiss*; delta smelt, *Hypomesus transpacificus*; longfin smelt, *Spirinchus thaleichthys*) and birds (western snowy plover, *Charadrius alexandrinus nivosus*; California clapper rail, *Rallus longirostris obsoletus*). This estuary has been radically transformed by human actions that began soon after the 1848 discovery of gold in California and included, for example, near-complete (95%) diking and filling of tidal marsh habitat [*Nichols et al.*, 1986]. As in all the world's estuarine-coastal ecosystems, changes continue in response to human disturbances and climatic variability. Understanding the drivers of these changes requires a broad landscape perspective from mountains to ocean because, as we show, processes of change originate far into the watershed, within the Bay, and in the Pacific Ocean.

[8] The San Francisco Bay system (Figure 2) comprises the North Bay (including Suisun and San Pablo Bays), a partially stratified estuary of the Sacramento–San Joaquin Rivers, and the South Bay, a marine lagoon situated in a densely populated urban setting. We use observations from Suisun Bay, just downstream of the confluence of the Sacramento and San Joaquin rivers, as an example of an estuarine system strongly influenced by seasonal and annual fluctuations of runoff from an agricultural watershed. We use observations from South Bay as an example of a marine lagoon strongly influenced both by inputs from an urban landscape and connectivity to a coastal ocean.

[9] Suisun Bay is a turbid, low-salinity embayment with high nutrient (N, P, Si) concentrations but low phytoplankton biomass (chlorophyll *a*) and primary production (Table 1). South Bay is a larger, nutrient-enriched embayment with higher salinity, lower turbidity, and higher phytoplankton biomass and primary production. Both embayments are broad expanses of intertidal and shallow subtidal habitat incised by a relict river channel. Tidal currents are strong (peak velocity

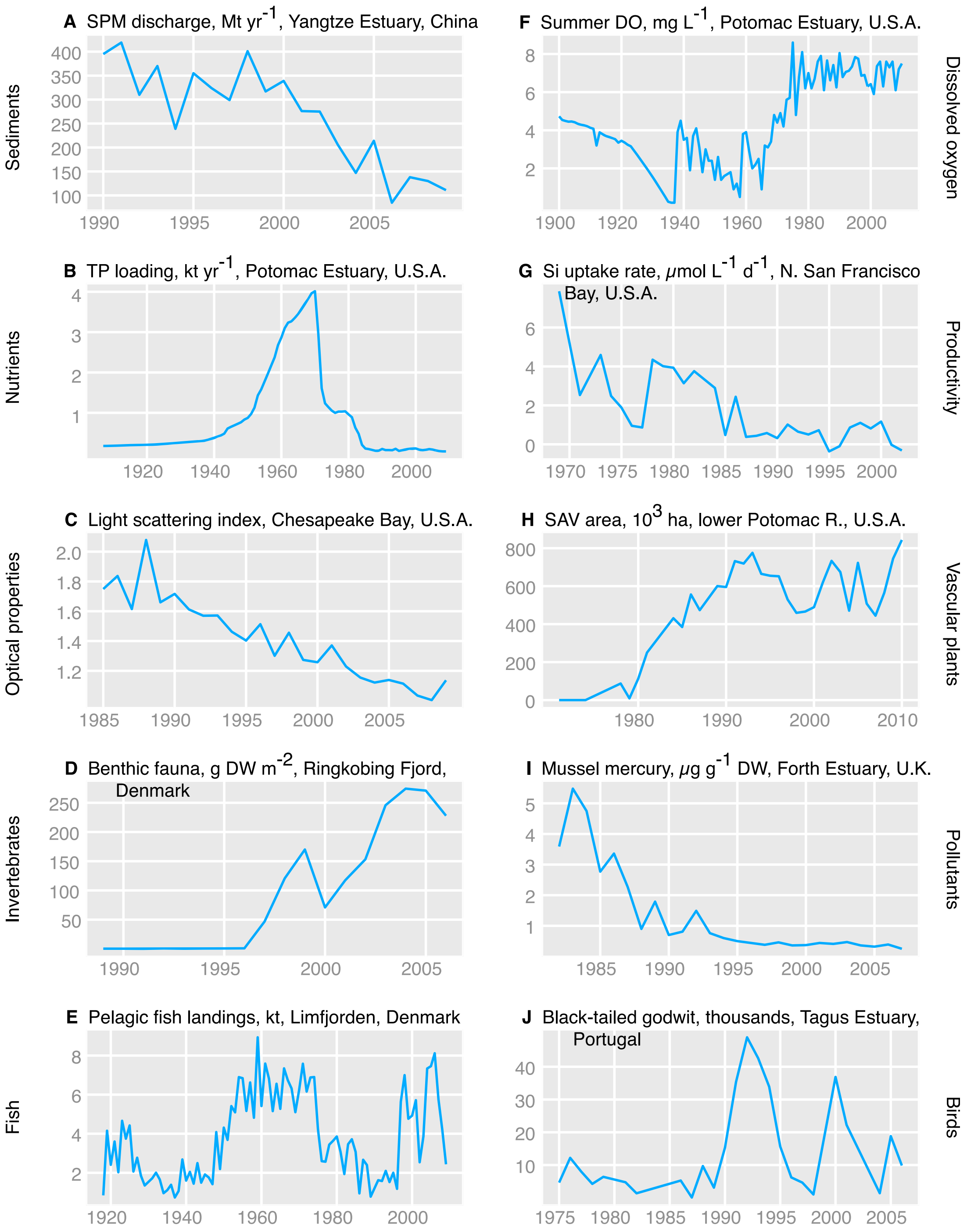

Figure 1. Examples of change in estuaries, as altered: (a) sediment supply to the Yangtze Estuary, (b) total phosphorus loading to the Potomac Estuary, (c) index of light scattering in Chesapeake Bay, (d) biomass of benthic invertebrates in Ringkøbing Fjord, (e) landings of pelagic fish in Limfjorden, (f) dissolved oxygen concentration in bottom waters of the Potomac Estuary during summer, (g) uptake of silicate as an index of diatom primary productivity in northern San Francisco Bay, (h) area of submerged vascular plants (SAV) in the lower Potomac River, (i) concentrations of mercury in mussels collected in the Forth Estuary; and (j) abundance of black-tailed godwits in the Tagus Estuary. Data provided by Shilun Yang (East China Normal University) (Figure 1a), Norbert Jaworski (U.S. EPA, retired) (Figures 1b and 1f), Charles Gallegos (Smithsonian Environmental Research Center) (Figure 1c), Jens Würgler Hansen (Aarhus University, Denmark) (Figure 1d), Hans Ulrik Riisgård (University of Southern Denmark) (Figure 1e), Wim Kimmerer (San Francisco State University) (Figure 1g), David Wilcox (Virginia Institute of Marine Science) (Figure 1h), Judith Dobson (Scottish Environment Protection Agency) (Figure 1i), and Teresa Catry (CESAM/Museu Nacional de História Natural, Portugal) (Figure 1j).

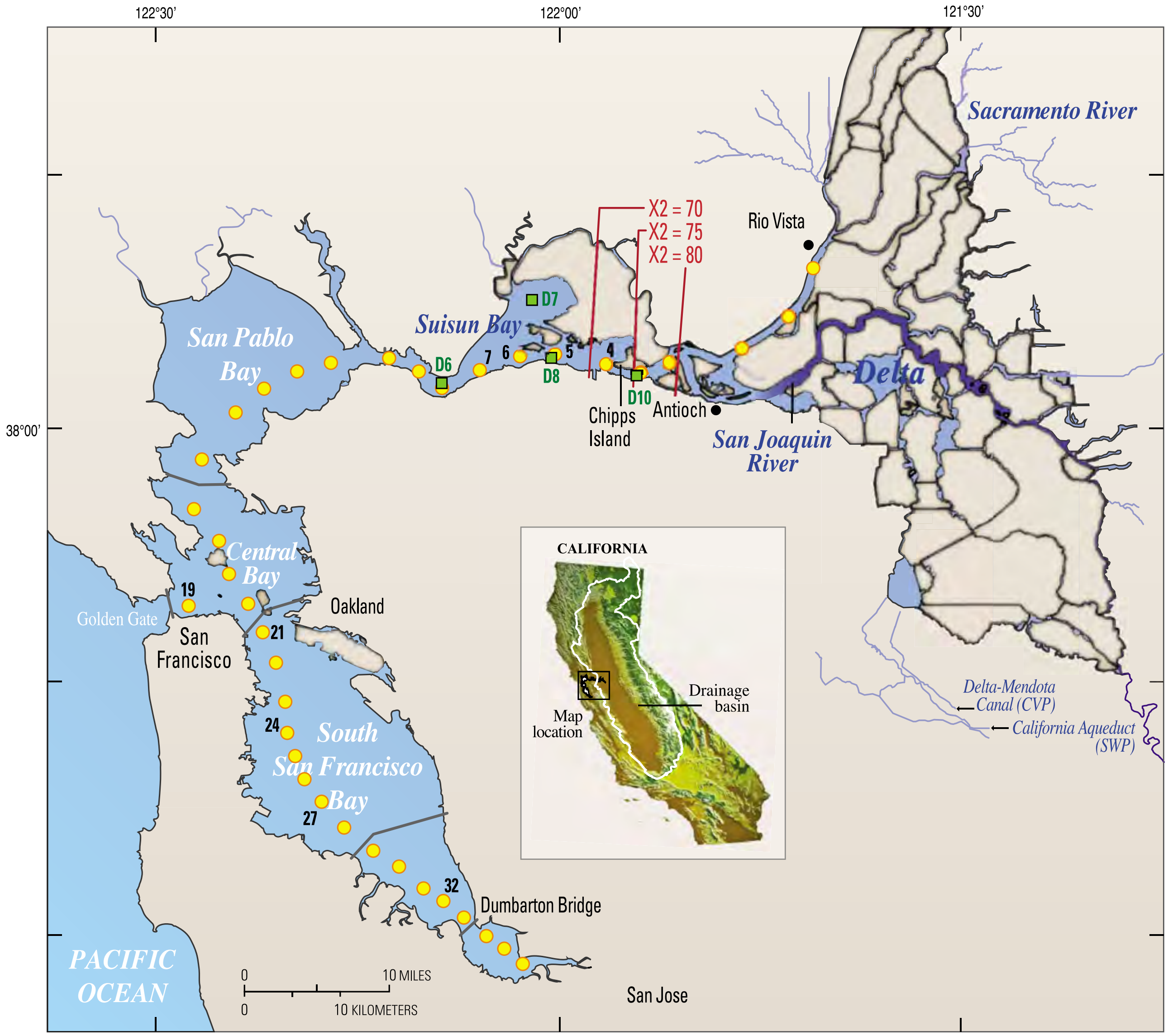

Figure 2. San Francisco Bay, fed by the waters of the Sacramento–San Joaquin River Delta and connected to the Pacific Ocean at the Golden Gate. The boundary between the Bay and the Delta is specified to be at Chipps Island. Water is exported from the southern Delta via state (SWP) and federal (CVP) water project canals. Numbers labeled "X2" represent distances (km) along the axis of the estuary from the Golden Gate. We use observations at sampling sites shown in South Bay and Suisun Bay to illustrate drivers of change detected over the past four decades.

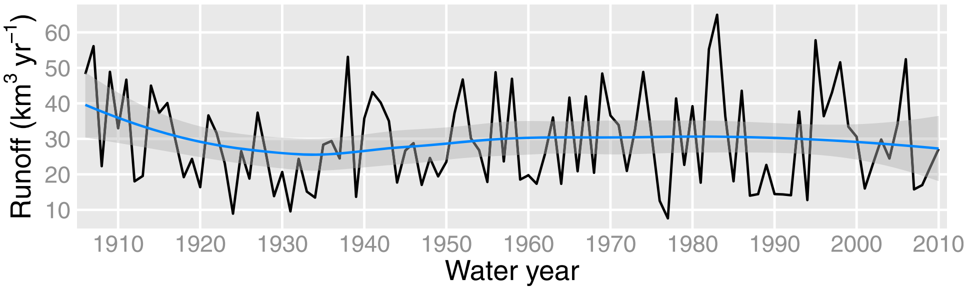

Figure 3. Unimpaired runoff for the water year (October through September), based on measured flows in the major tributaries to San Francisco Bay upstream of storage and diversion points. The blue line represents a loess smoother with 0.95 confidence interval.

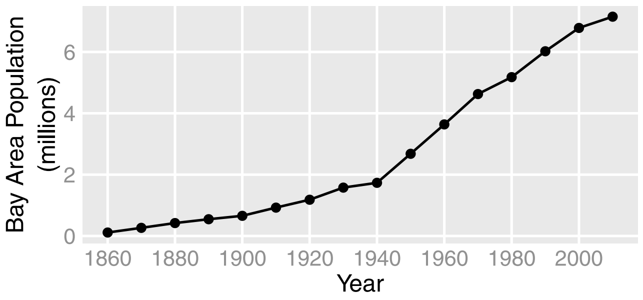

Figure 4. Growth of the total population of San Francisco Bay Area counties, 1860–2010.

$\sim$175 cm s^{-1} at the Golden Gate), and tidal amplitude is damped as the tidal wave propagates into North Bay but amplified along the semi-enclosed South Bay [*Walters et al.*, 1985]. Water residence time in Suisun Bay ranges from less than a day during large floods to about a month during the dry season and from weeks to months in South Bay. San Francisco Bay is turbid because of large river inputs of suspended particulate material (SPM), mostly mineral sediments. Unlike Chesapeake Bay and many other nutrient-enriched estuaries, San Francisco Bay is not currently impaired by harmful algal blooms, excessive phytoplankton production, or hypoxia (Table 1; see section 8.2).

[10] Water quality and biological communities are sampled regularly in San Francisco Bay by the U.S. Geological Survey (USGS) and the Interagency Ecological Program (IEP), a consortium of state and federal agencies. These research and monitoring programs are motivated by the common needs of resource managers and policy makers around the world's coastlines to understand how environmental changes are brought about by climate variability and human disturbance. These sampling programs have been sustained over four decades, providing one of the longest and most comprehensive records of environmental and biological variability in a U.S. coastal ecosystem. Combined sampling by USGS, IEP, and other agencies has produced a valuable observational record for capturing large environmental changes as trends over time or abrupt shifts and for identifying their underlying causes. We use these records to illustrate six drivers of change that are common in marine systems influenced by connectivity to land. The next six sections follow a common format of: background information about a specific driver of change, measured responses to that driver in San Francisco Bay, and discussion of the significance of those changes from both a local and global perspective.

3. WATER CONSUMPTION AND DIVERSION

3.1. Background

[11] The explosive population growth and economic development that began in California after the 19th century gold rush [*Nichols et al.*, 1986] required a stable water supply. That supply was met with construction of a massive infrastructure that includes reservoirs to capture water produced by runoff during the wet winter-spring and canals to carry that water from the humid northern region of California to the drier south, where 75% of water demand is concentrated. This infrastructure provides flood protection and water for urban centers and California's agricultural industry, which annually produces crops valued at $36 billion [*U.S. Department of Agriculture*, 2010]. The two major systems of water infrastructure are the Central Valley Project (CVP) operated by the U.S. Bureau of Reclamation and the State Water Project (SWP) operated by the California Department of Water Resources (Figure 2 shows their delivery facilities exiting the Delta). Although the CVP and SWP storage facilities account

TABLE 1. Attributes of Suisun Bay and South Bay as Contrasting Estuarine Habitats of San Francisco Bay, Including Their Dimensions, Water Residence Time, Phytoplankton Primary Production, and Quartile Values of Water-Quality Constituents From Sampling by the USGS (Stations 4–7 (Suisun Bay) and 20–36 (South Bay)) and IEP (Stations D6, D7, D8, D10 (Suisun Bay)) From 1969 Through 2010

	Suisun Bay				South Bay			
	Value	Median	25th Percentile	75th Percentile	Value	Median	25th Percentile	75th Percentile
Surface area (km^2), MSL[a]	170				430			
Mean depth (m), MSL[a]	4.6				5.8			
Mean tidal range[b] (m)	0.9–1.3				1.3–2.3			
Residence time[c] (d)	0.5–35				14–160			
Primary production[d] (g C m^{-2} yr^{-1})	20–130				130–210			
Salinity		5.8	1.0	10.7		27.4	23.8	30.0
Temperature (°C)		17.1	12.4	20.0		15.1	12.8	17.7
Chlorophyll *a* (μg L^{-1})		2.0	1.4	3.0		4.1	2.4	7.3
NO_3 + NO_2 (μM)		23.6	15.7	31.4		21.7	14.7	32.1
NH_4 (μM)		4.3	2.3	7.1		6.1	3.4	8.7
PO_4 (μM)		2.3	1.8	2.9		4.8	2.9	8.6
SiO_4 (μM)		201	163	240		83	60	109
Suspended particulate matter (mg L^{-1})		39	25	63		17	10	31
Attenuation coefficient k (m^{-1})		2.7	1.9	3.8		1.4	1.0	2.0
Dissolved oxygen (mg L^{-1})		8.7	8.2	9.4		7.9	7.1	8.6

[a] U.S. Geological Survey, San Francisco Bay bathymetry, 2007, available at http://sfbay.wr.usgs.gov/sediment/sfbay/geostat.html.

[b] NOAA, 2007 NOAA tide predictions: San Francisco, 2006, available at http://tidesandcurrents.noaa.gov/get_predictions.shtml?year=2007&stn=1813+San+Francisco.

[c] *Walters et al.* [1985].

[d] *Alpine and Cloern* [1992], *Cloern* [1987], and *Cloern et al.* [1985].

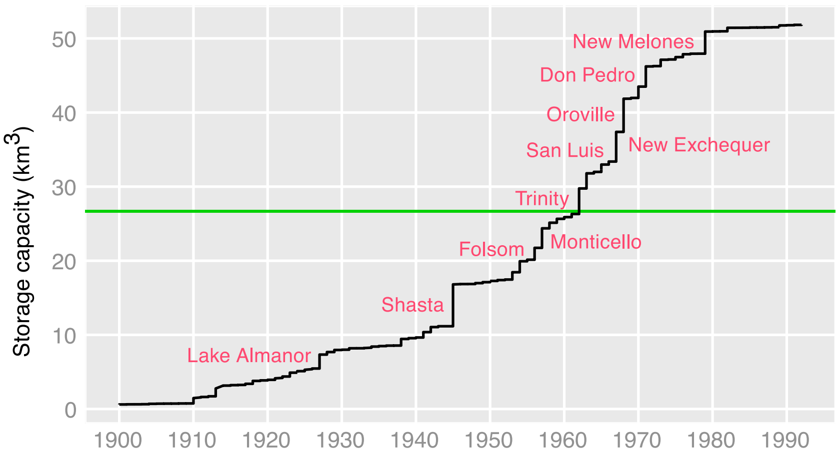

Figure 5. Growth in California reservoir capacity since 1900. The 10 largest reservoirs are labeled next to the corresponding step increase in capacity. The green line represents mean unimpaired runoff for water years 1906–2010 (Figure 3).

for more than half the upstream water storage capacity, many other upstream facilities and numerous small water users also affect water supply to San Francisco Bay [*Arthur et al.*, 1996].

[12] Construction of the CVP began with Friant Dam on the San Joaquin River in 1942, and the last major facility constructed was the New Melones Dam on the Stanislaus River, a tributary of the San Joaquin, in 1979 (Figure 5). The largest CVP reservoir is Lake Shasta (5.62 km^3) on the Sacramento River, formed by Shasta Dam in 1945. While the CVP impounds water from five major rivers (American, Sacramento, San Joaquin, Stanislaus, and Trinity), the SWP draws primarily from the Feather River, the main tributary of the Sacramento. The largest SWP reservoir (4.36 km^3) is Lake Oroville, formed by Oroville Dam in 1968. Every large river in the Sierra Nevada (except the Cosumnes River) has a large terminal storage reservoir. The cumulative capacity (27 km^3) of the 10 largest reservoirs mediating flow into San Francisco Bay is approximately the same as the (1906–2010) median annual runoff in the Sacramento–San Joaquin drainage (Figure 5). The CVP and SWP are two of the largest water diversions in the world, and we show how their operations have changed the quantity and seasonal pattern of freshwater inflow to San Francisco Bay.

3.2. Reduced Amount and Altered Timing of Freshwater Inflow

[13] The net outflow from the Delta is the most important freshwater input to San Francisco Bay. Delta outflow is what remains of Delta inflow after exports to various water projects and depletions within the Delta. Measurements of outflow, inflow, exports, and depletions are available for water years 1956–2010 (Table A1, Dayflow Program flow data). In addition, we can estimate *unimpaired* Delta inflow, which is runoff that would have occurred had water flow remained unaltered in rivers and streams upstream of the Delta instead of being stored in reservoirs, imported, exported, or diverted [*California Department of Water Resources* (*CADWR*), 2007]. We used these particular estimates of unimpaired inflow, even though they are available only through 2003, because they include accretions from the Sacramento and San Joaquin valley floors in addition to runoff from higher elevations. The difference between measured and unimpaired Delta inflow provides an estimate of the upstream effect on water losses, which can then be compared to the Delta effect. During 1956–2003, a median 61% of unimpaired inflow from the watershed flowed out of the Delta into Suisun Bay (Figure 6), while upstream and Delta effects accounted for 21% and 13%, respectively.

[14] Although the upstream effect exhibited no long-term trend for 1956–2003, both exports and Delta outflow changed systematically during this period. The trend in the Delta effect ($+4.0\ m^3\ s^{-1}\ yr^{-1}$, $p < 0.001$) essentially mirrored that in outflow ($-3.6\ m^3\ s^{-1}\ yr^{-1}$, $p < 0.001$). The Delta effect therefore increased over time, at the expense of outflow to San Francisco Bay. The trend in Delta effect is due to a trend in water exports from the Delta (also $+4.0\ m^3\ s^{-1}\ yr^{-1}$, $p < 0.001$), as opposed to within-Delta depletion that contributes a median of only 19% of the Delta effect and has no long-term time trend. The long-term increase in exports, from approximately 5% to 30% of Delta inflow, is obvious in Figure 7, and it appears to end by the 1990s.

[15] Delta inflow for the months July and August increased significantly from 1956 to 2010 (Figure 8a). Presumably, this is the result of storage-and-release patterns from impoundments upstream of the Delta, which store excess supply from spring runoff and postpone releases until drier summer conditions [*Knowles*, 2002]; there was no change in the seasonal pattern of unimpaired Delta inflow for 1956–2003. Monthly exports from the Delta also increased every month except May (Figure 8b). Because of the high inflow typical of winter months there was no detectable change in Delta outflow during January through April, and the enhanced supply from upstream ameliorated any effect of exports on Delta outflow during July through August. During September through December, however, changes in the upstream supply no longer compensated for increased export losses and, as a result, outflow from the Delta to San Francisco Bay declined (Figure 8c). As in the case of annual exports (Figure 7), the downward trend in Delta outflow for September through December was nonlinear and

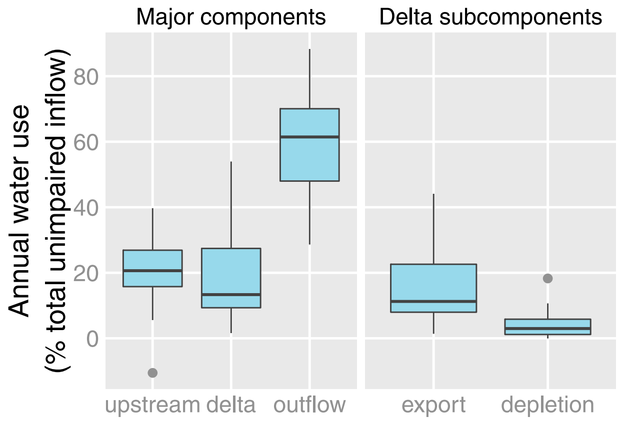

Figure 6. The fate of tributary water to San Francisco Bay as a percentage of total unimpaired inflow to the Delta during 1956–2003, i.e., the inflow to the Delta that would have occurred in the absence of upstream human activities. (left) The major fates include net upstream use (including consumption, reservoir storage or release, and import or diversion), Delta use, and outflow from the Delta to the Bay. (right) Uses in the Delta can be classified as exports to state and federal water projects and depletions within the Delta (the net result of consumption, precipitation, and evapotranspiration).

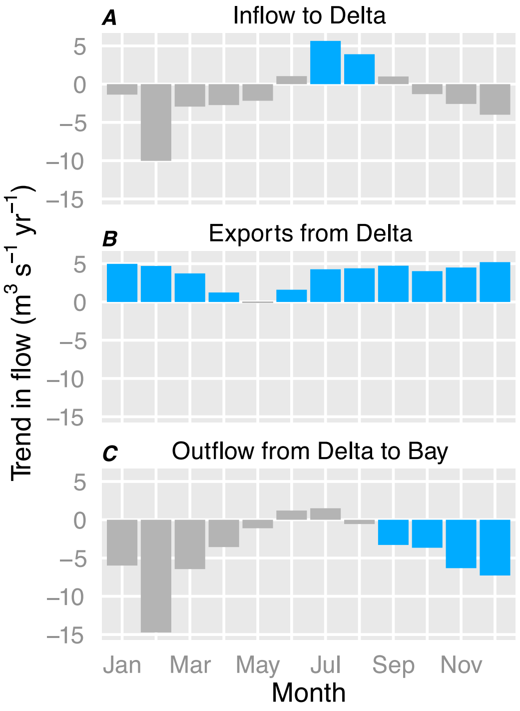

Figure 8. Long-term trends (1956–2010) in three important flow variables for San Francisco Bay. (a) Total measured inflow to the Delta. (b) Exports from the Delta to state, federal, and local water projects. (c) Net outflow from the Delta past Chipps Island (see Figure 2) to San Francisco Bay. The blue shading represents significant trends ($p < 0.05$).

essentially over by about 1990. Stabilization of the water export ratio reflects a 1994 policy agreement on Bay-Delta environmental protection (Bay-Delta Accord) between government agencies and diverse stakeholders to set monthly quotas for water export (CALFED, History of CALFED Bay-Delta Program, 2012, available at http://calwater.ca.gov/calfed/about/History/Detailed.html).

[16] In response to these September through December flow trends, salinity now moves further upstream during the latter part of the calendar year. The salinity gradient of North San Francisco Bay can be characterized by X2, the distance (kilometers) from the Golden Gate where near-bottom salinity is 2 [*Jassby et al.*, 1995]. We determined X2 from Delta outflow using a steady state model [*Monismith et al.*, 2002]. As implied by the negative trend in September through December outflow from the Delta (Figure 8c), there is a corresponding positive trend in September through December X2, i.e., an increase in the autumn intrusion of salinity into the estuary. Table 2 illustrates this trend using decadal averages of actual X2 based on Delta outflow and "unimpaired" X2 based on unimpaired Delta outflow [*CADWR*, 2007] for September through December. The average difference ΔX2 between them was negative through the 1970s, indicating how reservoir operations initially shifted the water supply to San Francisco Bay from the earlier wet to the later dry months. But exports from the Delta eventually dominated, and salinity intrusions

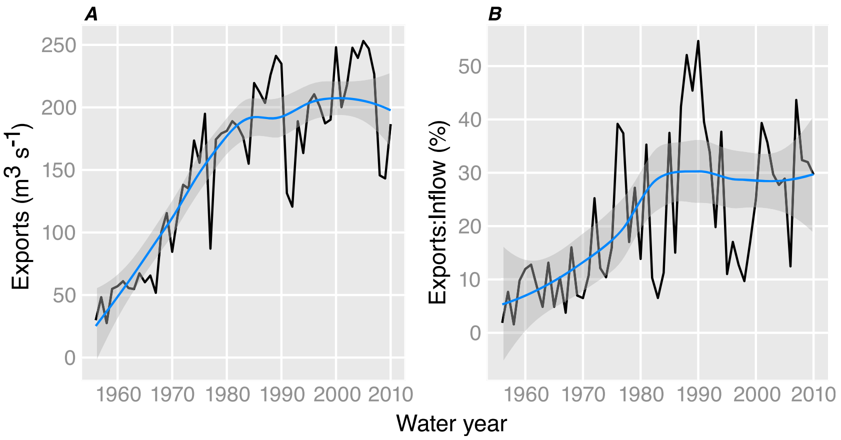

Figure 7. (a) Water-year mean exports from the Delta. (b) Exports as a percent of total inflow to the Delta. The blue lines represent loess smoothers with 0.95 confidence intervals.

TABLE 2. Decadal Averages of X2 (km) for September Through December[a]

Decade	X2	X2*	ΔX2	SD
1950–1959		73.7		
1956–1959	73.2	75.9	−2.7	2.1
1960–1969	71.3	73.3	−2.0	2.8
1970–1979	73.3	73.7	0.5	5.2
1980–1989	75.1	72.5	2.6	4.7
1990–1999	78.6	75.9	2.7	2.9
2000–2003	79.9	74.2	5.6	0.7
2000–2010	80.5			

[a] X2 measured in kilometers. X2, estimated from outflow; X2*, estimated from unimpaired outflow; ΔX2, difference between them; SD, standard deviation of difference.

during September through December have become greater than they would have been under unimpaired conditions, i.e., ΔX2 > 0 (Table 2). The estuarine salinity gradient has thus been displaced landward relative to unimpaired conditions. Despite these overall trends, there is high variability from year to year within each decade. In fact, interannual outflow variability may be larger now than in pre-European times, when flows were dampened by large wetland and floodplain areas [*Enright and Culberson*, 2010].

3.3. Significance of the Changes

[17] Flow management in the San Francisco Bay–Delta watershed is so pronounced that a median 39% of its unimpaired runoff is consumed upstream or diverted from the estuary (Figure 6), and the Sacramento–San Joaquin River system is thus classified as "strongly affected" by fragmentation [*Dynesius and Nilsson*, 1994]. Responses to this fragmentation include annual exports sometimes exceeding 50% of inflow (Figure 7b), shifts in the seasonal hydrograph (Figure 8), and a landward displacement of the estuarine salinity gradient during autumn (Table 2). The era of increasing water exports from the Sacramento–San Joaquin Delta (Figure 7) has been marked by population declines of native aquatic biota across trophic levels from phytoplankton [*Alpine and Cloern*, 1992] to zooplankton [*Winder et al.*, 2011] to pelagic fish [*Sommer et al.*, 2007], and large shifts in biological communities [*Winder and Jassby*, 2011]. These signs of ecosystem disturbance are related, at least partly, to altered flow regimes from water consumption and exports [*Bennett*, 2005; *National Research Council* (*NRC*), 2010; *Sommer et al.*, 2007]. Attribution of specific biological changes to flow modification is difficult because of data gaps (water exports began before biological monitoring), and confounding effects of other drivers of change such as climate variability, pollutant inputs, introductions of nonnative species, and landscape modifications [*Mac Nally et al.*, 2010]. However, modifications of inflow and salinity are contributing factors to population declines of native species in low-salinity habitats of the San Francisco Bay system [*Moyle et al.*, 2010] and to the remarkably successful establishment of nonnative species [*Winder et al.*, 2011], including species that have restructured food webs and their productivity [*Winder and Jassby*, 2011]. Water export from the Sacramento–San Joaquin Delta is a direct source of mortality to fish, including imperiled species such as delta smelt and longfin smelt [*Grimaldo et al.*, 2009; *NRC*, 2010], and export plus within-Delta depletion alters system energetics of an already low-productivity ecosystem by removing phytoplankton biomass equivalent to 30% of Delta primary production [*Jassby et al.*, 2002]. Reduced autumn inflows and associated salinity increases (Table 2) have lowered habitat quantity and quality for species endemic to the upper estuary, such as the endangered delta smelt [*Feyrer et al.*, 2011].

[18] These linkages between water diversion and sustainability of native fishes and their supporting food webs are now recognized in policy. First, an ambitious biological conservation plan having coequal goals of water supply reliability and ecosystem restoration was created [*Bay Delta Conservation Plan* (*BDCP*), 2010]. Second, California's State Water Resources Control Board recently determined that current flows to the San Francisco Bay–Delta "are insufficient to protect public trust resources" and proposed flow criteria based on its conclusion that "[f]low modification is one of the few immediate actions available to improve conditions to benefit native species" [*State Water Resources Control Board*, 2010, p. 2, p. 40].

[19] Outcomes of this policy recommendation are uncertain, but the San Francisco Bay example illustrates the extent to which humans have modified hydrologic systems and the global challenge of measuring and balancing the societal benefits and environmental costs of different water management actions and policies. At least 90% of total river discharge in the United States is strongly affected by channel fragmentation from reservoir operations, interbasin diversions, and irrigation consumption [*Jackson et al.*, 2001]. Flow management has had particularly large effects at middle latitudes, where the cumulative discharge of many rivers declined 60% [*Milliman et al.*, 2008] in the 1951–2000 era of accelerated dam construction and irrigation, which dominates U.S. water use [*Gleick and Palaniappan*, 2010]. Large-scale fragmentation of river systems has been a significant disturbance to estuarine-coastal ecosystems. Iconic examples include extensive losses of wetlands and bivalve mollusks in the Gulf of California after completion of the Glen Canyon Dam on the Colorado River [*Baron et al.*, 2002]; collapse of Egypt's coastal fishery when Nile flows to the Mediterranean Sea were reduced by 90% after completion of the Aswan Dam [*Nixon*, 2003]; salinity increase and restructuring of Florida Bay's seagrass and fish communities after freshwater inflows from the Everglades were reduced by 60% [*Herbert et al.*, 2011]; and greatly reduced nutrient supply, primary production, diversity, and biomass of fish communities in the Bohai Sea after Huanghe (Yellow) River discharge was reduced 73% between the 1950s and 1990s [*Fan and Huang*, 2008].

4. HUMAN MODIFICATION OF SEDIMENT SUPPLY

4.1. Background

[20] One of the first scientific investigations of San Francisco Bay was published as a USGS professional paper by G. K.

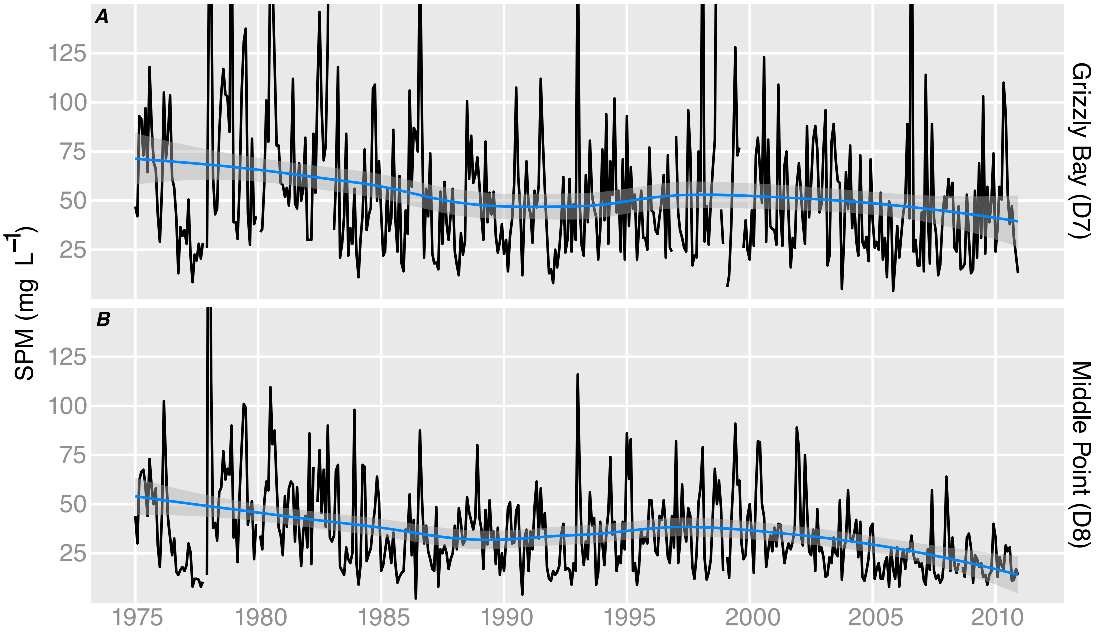

Figure 9. Monthly mean suspended particulate matter concentrations measured in surface waters at two locations in Suisun Bay. (a) Grizzly Bay, a shallow subembayment location near Suisun Slough (Figure 2, site D7). (b) In the deep channel off of Middle Point (site D8). The blue lines represent loess smoothers with 0.95 confidence intervals.

Gilbert [*Gilbert*, 1917]. This remarkably detailed and comprehensive study included measurements in San Francisco Bay, its tributary rivers, and their watersheds to assess impacts of hydraulic gold mining in the Sierra Nevada Mountains on sediment supply to and deposition in San Francisco Bay. Gilbert's conclusion was startling and accurate: mobilization of sediments by hydraulic mining during the period 1849–1914 delivered nearly a billion cubic meters of sediments to San Francisco Bay. A comparison of the 1856 and 1887 bathymetric surveys of San Pablo Bay (Figure 2) confirmed that the estuary accumulated sediments during this period, when some regions filled by more than 4 m and intertidal mudflats expanded 60% [*Jaffe et al.*, 2007]. Hydraulic mining was prohibited in 1884, and the late 19th century era of sediment deposition was followed by a gradual shift to the current state of San Francisco Bay as an erosional system [*Jaffe et al.*, 2007]. This shift was driven by multiple processes including erosion of the hydraulic-mining debris deposited in the river system, diking the rivers and disconnecting them from floodplains, and retention of sediments behind large dams constructed in the 20th century (Figure 5) [*Schoellhamer*, 2011; *Wright and Schoellhamer*, 2004]. As a response to these changes in the watershed, sediment supply to the estuary has been halved since the mid-20th century [*Wright and Schoellhamer*, 2004]. Sediment supply peaked at about 12 Mt yr^{-1} in the late 19th century, then declined as the era of large-dam building progressed and is now <1 Mt yr^{-1} [*Schoellhamer*, 2011]. Changes in sediment supply of this magnitude have reshaped San Francisco Bay's geomorphology [*Jaffe et al.*, 2007]. We focus here on another important consequence—reduced concentrations of SPM and turbidity in the upper estuary.

4.2. Decreasing Sediment Concentrations

[21] We chose two sampling locations in Suisun Bay to illustrate SPM trends (Figure 9): D7 in the center of Grizzly Bay, a shallow subembayment, and D8 in the deep channel (Figure 2). Both sites have been sampled at about 1 m depth approximately monthly from 1975 through 2010. We calculated trends in water-year mean SPM concentration, using a model to separate out the variability due to year-to-year changes in flow. This effect of flow on water quality constituents is often separated out with an additive model that includes terms for both flow and long-term trend [*Cohn et al.*, 1992]. For our application, net Delta outflow was used as the flow variable and water year was used as the trend variable. Rating curves for SPM are often nonlinear, and our preliminary exploration suggested a nonlinear transform for flow within the additive model. Both SPM and outflow also required log-transformation to ensure normality of residuals. The resulting model is

$$\ln M = c_0 + c_1 T + s(\ln Q_{\text{out}}) + \varepsilon, \tag{1}$$

where for each water year T, M is the mean SPM concentration (mg L^{-1}); Q_{out} is the mean outflow (m^3 s^{-1}); c_0 and c_1 are constant coefficients; s represents a natural spline; and ε is the residual. We summarize the fit to equation (1) graphically for site D8 (Figure 10).

[22] Both water year (long-term trend) and outflow are significant sources of SPM variability, and the effect sizes of trend and outflow are similar (Figure 10). The outflow effect on SPM is positive and reaches a maximum between 500 and 1000 m^3 s^{-1}. There is typically an estuarine turbidity maximum in this embayment, and gravitational circulation is

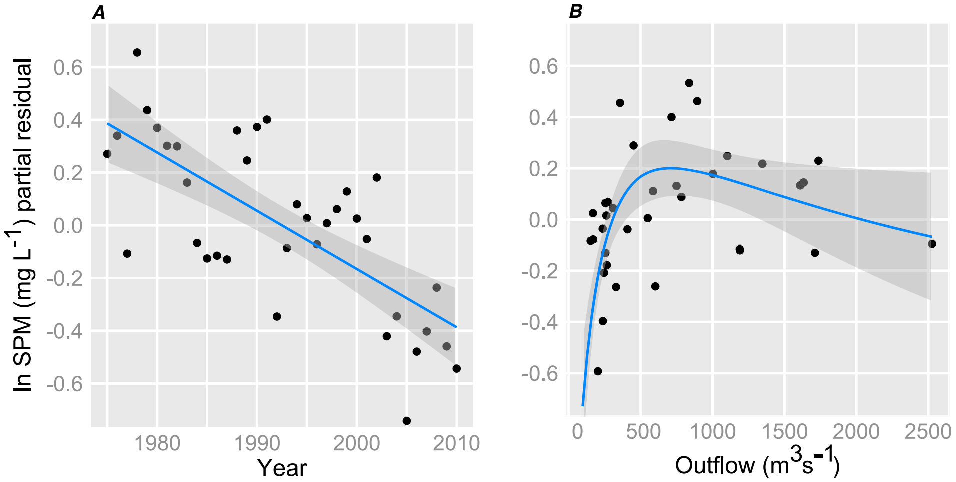

Figure 10. Partial residual plots for a regression model that accounts for variability in annual mean suspended particulate matter in Suisun Bay (D8) as a result of a long-term trend plus variability in annual mean outflow from the Delta. (a) The linear effect of trend. (b) The nonlinear effect of outflow. The blue lines represent loess smoothers with 0.95 confidence intervals.

strongest in this outflow range. The trend effect is equivalent to an annual SPM loss of 2.0% yr^{-1}. With outflow set to its long-term median, the model implies that SPM concentrations dropped from 54 mg L^{-1} in 1975 to 27 mg L^{-1} in 2010, consistent with regional trends of declining turbidity in the upper estuary [*Kimmerer*, 2004]. Results for D7 are qualitatively similar (71 to 46 mg L^{-1}), although the annual SPM loss was only 1.2% yr^{-1}, perhaps reflecting the greater importance of resuspension at this shallow site. Loss rates of SPM concentration in Suisun Bay compare to a 1.3% yr^{-1} decline of sediment supply during the last half of the 20th century [*Wright and Schoellhamer*, 2004].

4.3. Significance of the Changes

[23] The reduced sediment supply to San Francisco Bay (Figure 10a) has important ecological implications for this estuary, including changes in the transport of sediment-bound contaminants, exposure of legacy contaminants (e.g., mercury; see section 7.3) as surface sediments continue to erode [*Jaffe et al.*, 2007], and a “bleak prognosis” for long-term sustainability of tidal marshes in this urban setting where marshes cannot migrate upland to accommodate anticipated sea level rise and low sediment supply [*Stralberg et al.*, 2011]. Here we consider implications of decreasing SPM concentrations on turbidity and light availability to primary producers, an underexplored response to human modifications of sediment supply to estuaries. San Francisco Bay has high nutrient concentrations (see section 6), but Suisun Bay has unusually low phytoplankton biomass (Table 1) because of fast water filtration by clams (see section 7.2) and turbidity from high SPM concentrations leading to light limitation of photosynthesis [*Alpine and Cloern*, 1992]. What, then, are implications of a 50% reduction of SPM concentration (Figure 9) for primary productivity? To address this question we developed a multiple regression model relating the attenuation coefficient for photosynthetically active radiation, k (m^{-1}), to SPM concentrations and salinity. The model was linear in SPM and salinity, the latter serving as a proxy for dilution of terrestrial-derived, colored dissolved organic matter [*Twardowski and Donaghay*, 2001]. Setting salinity to its median of 5.1 during 1975–2010, and substituting flow-adjusted values for SPM in 1975 and 2010, the model implies a drop in flow-adjusted k from 3.67 to 2.35 m^{-1} in the deep channel of Suisun Bay (D8). This implies a corresponding increase in photic-zone depth from 1.3 to 2.0 m. Similar calculations for the shallow subembayment (D7) yield an increase in photic-zone depth from 1.1 to 1.5 m.

[24] This decrease in light attenuation has a direct effect on primary production. Phytoplankton primary productivity in San Francisco Bay [*Cole and Cloern*, 1984] and the upstream Delta [*Jassby et al.*, 2002] are well described with a simple model of biomass and light availability:

$$P_{\mathrm{g}} = \psi B I_0 z_{\mathrm{p}}, \tag{2}$$

where P_{g} (mg C m^{-2} d^{-1}) is gross primary productivity, i.e., the photosynthetic incorporation rate of carbon by phytoplankton beneath a square meter of water surface; ψ (mg C [mg Chl a]$^{-1}$ [Einstein m^{-2}]$^{-1}$) is an efficiency factor; B (μg L^{-1}) is chlorophyll a concentration; I_0 (Einstein m^{-2} d^{-1}) is incident photosynthetically active radiation (PAR); and z_{p} is photic-zone depth (m), the depth at which PAR falls to 1% of its surface value I_0. Given that primary productivity is proportional to z_{p}, the increases in z_{p} calculated here imply a 54% increase in phytoplankton productivity per unit biomass (P_{g}/B) at the channel site D8 and a 38% increase at the shallow site D7.

[25] Turbid estuaries are inherently low-productivity ecosystems [*Cloern*, 1987]. Annual net primary production measured in the Suisun Bay channel during 1980 was only 80 g C m^{-2} yr^{-1}, compared to 160 g C m^{-2} yr^{-1} in the South Bay channel where SPM concentrations are lower [*Cloern et al.*, 1985]. However, Suisun Bay waters have become more transparent as SPM concentrations declined (Figure 9),

so the water column mean irradiance to support photosynthesis has increased significantly since those measurements were made. This illustrates an unintended consequence of river impoundment—reduced sediment supply to estuaries, leading to smaller SPM concentrations, deeper light penetration, and increased light availability to primary producers.

[26] Changes in water clarity have effects on other biological communities. Abundance of delta smelt has fallen to critically low values, and sustainability of this endemic species is a priority management goal. Delta smelt are most abundant in turbid, low-salinity habitats, and their association with turbidity may be an adaptation to minimize predation risk [*Nobriga et al.*, 2008]. An index of delta smelt habitat suitability declined 78% from 1967 through 2008 as a response to trends of increasing salinity (section 3.2) and water transparency. Reduced habitat quantity and quality are important factors contributing to population declines of this endangered species [*Feyrer et al.*, 2011]. On the other hand, increasing water clarity has expanded habitat area for rooted macrophytes including the nonnative *Egeria densa*, introduced in the 1940s and a target for removal as an invasive pest [*Santos et al.*, 2009]. *Egeria* now dominates shallow regions of the upstream Delta, and its expansion provides increasing habitat for nonnative fish, such as centrarchids (bluegill, sunfish), whose populations have grown since the 1980s [*Brown and Michniuk*, 2007]. The trend of diminishing sediment supply therefore has important ecological implications through its effect on turbidity, light availability, and photosynthesis of primary producers, and habitat suitability for native and nonnative plant and fish species.

[27] Over half of the world's large river systems (172 of 292) are affected by dams [*Nilsson et al.*, 2005]. Many of the world's major rivers have experienced similar drops in sediment discharge due to dams, followed by ecological effects that are long-lived and significant. Sediment discharge of the Yangtze River dropped from 490 to 150 Mt yr^{-1} after closure of Three Gorges Dam (Figure 1a), and the estuary downstream has become sediment-starved with corresponding submersion of salt marshes and erosion of the coastal delta [*Yang et al.*, 2011]. The largest coastal wetland loss in the United States is the 25% loss of Mississippi Delta wetlands after dam construction in the upper watershed reduced sediment supply to the lower Mississippi River from 400 to 500 MT yr^{-1} to 205 MT yr^{-1} [*Blum and Roberts*, 2009]. Louisiana's coastal wetlands are now sediment-starved and, with subsidence and accelerating sea level rise, a further 10,000–13,500 km^2 are projected to be submerged by 2100 [*Blum and Roberts*, 2009]. On the Nile River, the Aswan Dam has limited the amount of nutrient-rich sediments reaching the delta and negatively affected both agriculture and the functioning of coastal ecosystems [*Hamza*, 2009]. Dams on the Ebro River in Spain trap almost all suspended sediment and bed load in reservoirs, causing ongoing riverbed incision downstream [*Vericat and Batalla*, 2006]. Worldwide, nearly a third of sediment moving from land to the world oceans is now trapped behind dams [*Syvitski*, 2003] and, as a result, coastal wetlands are subsiding and river deltas are eroding. For regulated basins, more than half of the sediment is trapped [*Vörösmarty et al.*, 2003]. However, worldwide sediment discharge may actually have been only half its current level before widespread agriculture and deforestation began two to three millennia ago [*Milliman and Syvitski*, 1992].

5. INTRODUCED SPECIES

5.1. Background

[28] Accelerating globalization of commerce has had the unintended consequence of translocating species of microbes, plants, and animals, and human redistribution of life forms on Earth is now recognized as a powerful component of global environmental change [*Vitousek et al.*, 1996]. Biological invasions challenge the integrity of natural plant and animal communities and confound conservation plans to preserve endangered species. The most important vector for transferring marine species is movement of ship ballast water that is usually taken from and discharged into bays and estuaries. U.S. ports alone receive >79 million tons of ballast water annually from foreign ports [*Ruiz et al.*, 1997]. As a result, the world's bays, estuaries, and inland waters with deep-water ports are described as marine analogs of highly invaded oceanic islands and among the most threatened ecosystems on the planet [*Carlton and Geller*, 1993]. San Francisco Bay stands out as a coastal ecosystem transformed by introduced species that contribute up to 97% of the individuals and 99% of the biomass of some communities. The rate of biological invasions is accelerating and estimated at one new species introduced to the San Francisco Bay–Delta system every 14 weeks from 1961 through 1995 [*Cohen and Carlton*, 1998]. As a result, this "may be the most invaded estuary and possibly the most invaded aquatic ecosystem in the world" [*Cohen and Carlton*, 1998, p. 556]. We describe here one of the most far-reaching of these invasions: a restructuring of the Suisun Bay planktonic food web following introduction of the nonnative clam *Corbula amurensis* that quickly established itself as a "keystone" species. Regular sampling provided early detection of the clam's arrival and measurement of its rapid dispersal, making this one of the best-documented invasions of any estuary [*Carlton et al.*, 1990].

5.2. Restructured Planktonic Food Web

[29] *Corbula amurensis* is a small clam native to rivers and estuaries of East Asia. It was first discovered in Suisun Bay in October 1986 and was probably introduced as larvae discharged in ship ballast water [*Carlton et al.*, 1990; *Nichols et al.*, 1990]. By summer 1988 *Corbula* dominated the benthic community, exceeding 95% of the total in both numbers and biomass, and it has reached abundances as high as 16,000 individuals m^{-2} and biomass (ash-free dry weight) as high as 131 g m^{-2} [*Chauvaud et al.*, 2003]. The remarkably fast colonization and dominance of the Suisun Bay benthos by *C. amurensis* is attributed to its capacity to utilize a broad range of food resources [*Parchaso and Thompson*, 2002] and its adaptability to a wide range of salinities, including tolerance of salinity <1 [*Nichols et al.*, 1990]. The annual freshening of Suisun Bay during the wet season precludes colonization by marine bivalves, so *C. amurensis* rapidly

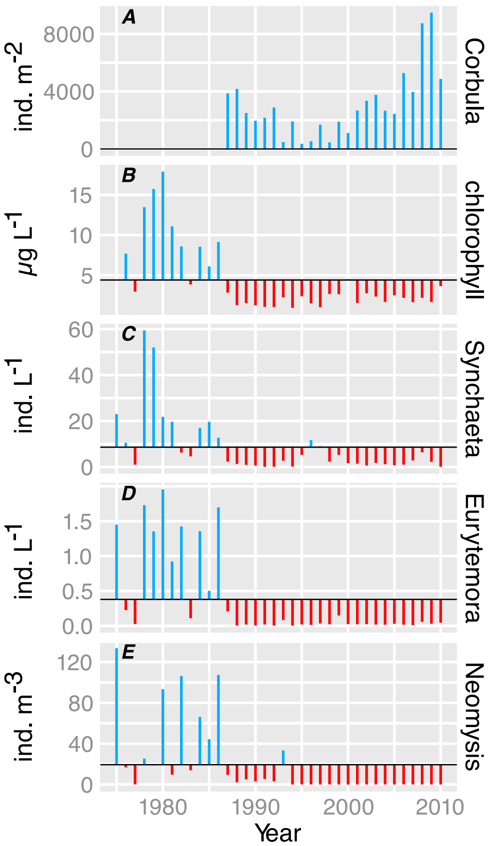

Figure 11. Response of the planktonic food web in Suisun Bay to an introduced clam, *Corbula amurensis*. (a) *Corbula* abundance. (b) Phytoplankton biomass as chlorophyll *a* concentration. (c) Density of the rotifer *Synchaeta bicornis*. (d) Density of the copepod *Eurytemora affinis*. (e) Density of the mysid shrimp *Neomysis mercedis*. The horizontal line in Figures 11b–11e is the long-term mean.

occupied and filled a vacant niche. Abundance of this clam has fluctuated markedly since its establishment (Figure 11a), and the single most prominent reason is salinity variability in response to inflow changes [*Peterson and Vayssieres*, 2010]. In particular, clam abundance in Suisun Bay tends to increase as X2 shifts upstream (section 3.2) [*Nichols*, 1985; *Nichols et al.*, 1990; *Winder et al.*, 2011].

[30] *Corbula amurensis* is a suspension-feeding bivalve that efficiently assimilates phytoplankton cells [*Cole et al.*, 1992; *Werner and Hollibaugh*, 1993]. Its annual mean filtration rate of Suisun Bay (~0.1–0.25 d^{-1}) is about twice the growth rate of phytoplankton (~0.05–0.1 d^{-1}), so clam consumption exceeds local production of phytoplankton biomass [*Thompson*, 2005]. As a result, average chlorophyll *a* concentration decreased abruptly after the clam introduction (Figure 11b), from 11 ± 2 $\mu g\ L^{-1}$ during 1975–1986 to 2.2 ± 0.2 $\mu g\ L^{-1}$ during 1987–2010. This biomass drop is ecologically significant because chlorophyll *a* concentrations of about 10 $\mu g\ L^{-1}$ represent a threshold below which zooplankton reproduction can become food limited [*Kimmerer et al.*, 2005; *Müller-Solger et al.*, 2002]. Introduction of *C. amurensis* changed the seasonal pattern of phytoplankton biomass because its grazing effect is strongest during summer (Figure 12). Prior to the introduction, Suisun Bay sustained high phytoplankton biomass, usually >10 μg chlorophyll *a* L^{-1}, during May through September when freshwater inflow is low and residence time is long enough for biomass to accumulate [*Cloern et al.*, 1983]. Since the introduction, chlorophyll *a* concentration is now regularly <3 $\mu g\ L^{-1}$, even during the low-flow season (Figure 12).

[31] Once established, *Corbula amurensis* quickly transformed Suisun Bay by reducing phytoplankton biomass and primary production fivefold [*Alpine and Cloern*, 1992], redirecting much of the remaining primary production from pelagic (zooplankton) to benthic (clam) consumers [*Thompson*, 2005], and creating a persistent state of low phytoplankton biomass and potential food limitation of herbivorous zooplankton. Other sources of organic matter are available to fuel production in food webs. The largest source to San Francisco Bay is river input of detritus [*Jassby et al.*, 1993], but this is largely refractory and the labile components must be converted through the inefficient microbial loop into forms accessible to zooplankton [*Jassby and Cloern*, 2000; *Müller-Solger et al.*, 2002; *Sobczak et al.*, 2002]. This pathway has also been disrupted because *C. amurensis* consumes all components of the microbial loop, including bacteria [*Werner and Hollibaugh*, 1993], ciliates, and flagellates [*Greene et al.*, 2011]. Introduction of a nonnative clam therefore reduced the microplankton food resource available to zooplankton and forced a shift toward their greater reliance on low-quality detritus [*Jassby*, 2008].

[32] These fundamental changes at the base of the food web provoked a cascade of responses, beginning with abrupt population declines of some zooplankton species [*Kimmerer et al.*, 1994]. Average abundance of the rotifer *Synchaeta bicornis* decreased from 23,500 ± 6,700 to 1,600 ± 360 individuals m^{-3} (Figure 11c). Average abundance of the calanoid copepod *Eurytemora affinis* dropped from 700 ± 140 to 35 ± 11 individuals m^{-3} (Figure 11d), and abundance of another calanoid copepod, *Acartia* spp., also declined sharply after 1987 [*Kimmerer*, 2004]. Average abundance of the mysid shrimp *Neomysis mercedis* dropped from 32 ± 10 to 2.5 ± 2.1 individuals m^{-3} and this species has virtually disappeared after a temporary population rebound in 1993 (Figure 11e). Near extinction of these previously abundant zooplankton species is attributed to depletion of the phytoplankton food resource and, in the case of *Eurytemora affinis*, predation on its larvae by the introduced clam [*Kimmerer*, 2004]. These well-documented observations in Suisun Bay before and after colonization by *Corbula amurensis* illustrate the power of bivalve mollusks to alter ecosystem production, pathways of energy flow, and food web structure through their predation upon and competition with zooplankton.

[33] The abrupt population declines of *Synchaeta bicornis*, *Eurytemora affinis*, *Acartia* spp., and *Neomysis mercedis* (Figures 11c–11e) followed population declines of other zooplankton taxa in Suisun Bay that began before the introduction of *Corbula* [*Kimmerer et al.*, 1994]. Comparing mean biomass from the 1970s with the period after 1990, calanoid copepods fell from 14 to 4 $\mu g\ C\ L^{-1}$, rotifers from 10 to 1 $\mu g\ C\ L^{-1}$, and cladocerans from 1.2 to 0.2 $\mu g\ C\ L^{-1}$

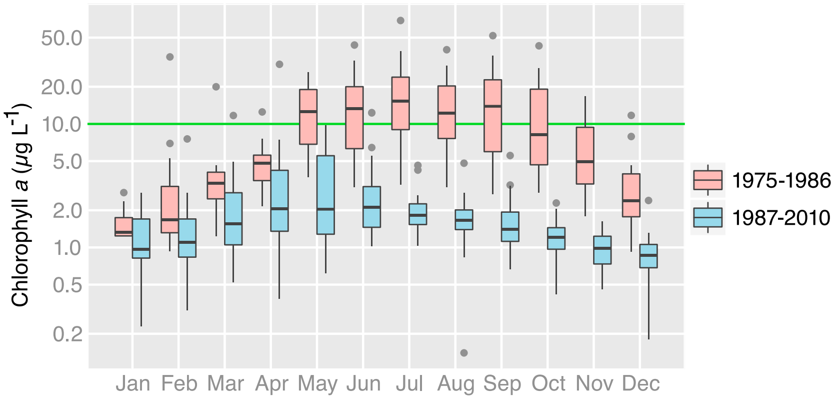

Figure 12. Boxplot summaries of monthly mean chlorophyll *a* in a shallow subembayment of Suisun Bay (Figure 2, site D7) before and after introduction of the clam *Corbula amurensis* in 1987. The green line represents characteristic threshold concentration at which zooplankton growth or reproduction can become food limited.

[*Winder and Jassby*, 2011]. These taxa were replaced by eight species of nonnative copepods and two species of nonnative mysids that became established in the upper estuary as a sequence of introductions during periods of low freshwater inflow, particularly during the 1987–1992 drought when salinity intrusion facilitated establishment of introduced species adapted to brackish habitat, including *C. amurensis* [*Winder et al.*, 2011]. The cumulative changes since the 1970s have produced a remarkable and perhaps unprecedented transformation of a zooplankton community from one having large components of mysid shrimp, rotifers, and calanoid copepods to one dominated by introduced copepods indigenous to East Asia [*Winder and Jassby*, 2011]. This transformation included emergence of smaller cyclopoid copepods that contributed less than 2% of zooplankton biomass before 1987 but more than 24% of biomass after 1994.

5.3. Significance of the Changes

[34] Losses of rotifers, calanoid copepods, and mysid shrimp have contributed to the collapses of fish populations in low-salinity regions of San Francisco Bay because these are essential dietary components for resident fish. Rotifers are preferred prey of larval delta smelt [*Nobriga*, 2002], and many planktivorous fish, including adult delta smelt, longfin smelt, and early life stages of other species, selectively prey on calanoid copepods that are larger and have higher nutritional quality than cyclopoid copepods [*Winder and Jassby*, 2011]. Other species such as American shad, starry flounder, and juvenile striped bass feed primarily on mysids when available [*Feyrer et al.*, 2003]. Losses of these zooplankton components provoked adaptations by their fish predators. Fish reliant on mysids shifted their diets to other prey, and those with the largest dietary shifts had the largest population declines in Suisun Bay marshes after 1987 [*Feyrer et al.*, 2003]. Northern anchovy (*Engraulis mordax*) is the biomass-dominant pelagic fish in San Francisco Bay; summer abundance of this species fell 94% in the low-salinity region of the estuary as anchovies adapted to the decreased food supply in Suisun Bay by migrating seaward [*Kimmerer*, 2006]. In addition to food web transformations, the fivefold decrease in primary production implies a comparable decrease in the energetic carrying capacity for fish in Suisun Bay based on its primary production [*Nixon*, 1988].

[35] Ecologists struggle to understand how and why some indigenous communities are displaced by nonnative species, but the near-complete restructuring of the zooplankton community in low-salinity regions of San Francisco Bay during the past four decades appears to be the synergistic result of multiple drivers [*Winder and Jassby*, 2011; *Winder et al.*, 2011]: introduction of the clam *Corbula amurensis*, an extended period of low freshwater inflow and salinity intrusion, and amplification of the drought effect by water diversions (section 3.2). Ecosystem disruptions by species introduced to San Francisco Bay by transoceanic shipping have shaped policy by motivating passage of California's Marine Invasive Species Act, considered the strictest regulation of ship ballast discharge in the United States to prevent or minimize release of nonindigenous species from commercial vessels [*Takata et al.*, 2011].

[36] More than 500 nonnative species have become established in U.S. coastal waters, and accelerating species introductions rank as one of the "most pervasive threats to native ecosystems and human economies" [*Grosholz*, 2005, p. 1088]. Brackish estuarine waters are vulnerable not only because so many ports are situated in estuaries and brackish-water species are tolerant of ballast-water tank conditions, but also because brackish waters tend to have fewer indigenous species so that aliens can establish more easily [*Wolff*, 1998]. Subsequently, the effects of international shipping reverberate up and down coasts as species are introduced into more isolated estuaries by intraregional transport [*Wasson et al.*, 2001]. Alien species can disrupt ecosystems by altering biogeochemical processes (e.g., silica cycling in the Bay of Brest [*Chauvaud et al.*, 2000]), by amplifying bioaccumulation of toxic contaminants in food webs (e.g., selenium in San Francisco Bay [*Stewart et al.*, 2004]), and through disruption of ecosystem functions that support native populations. Ecological regime shifts similar to that in Suisun Bay followed introductions of other bivalve mollusks, such as the soft-shell clam *Mya arenaria* to Denmark's Ringkøbing

TABLE 3. Annual Inputs of Dissolved Inorganic Nitrogen (DIN) to U.S. Estuaries and Bays From Treated Sewage and the Sewage Contribution to Total DIN Input

Estuary	Sewage DIN Input (mmol N m^{-2} yr^{-1})	Sewage (%)	Total DIN Input (g N m^{-2} yr^{-1})
Tomales Bay[a]	0	0	2
Apalachicola Bay[b]	10	2	8
Mobile Bay[b]	80	7	18
Chesapeake Bay[c]	NA	NA	14
Narragansett Bay[b]	390	41	13
Potomac Estuary[b]	390	48	11
Delaware Bay[b]	650	50	18
Long Island Sound[b]	270	67	6
New York Bay[b]	27,230	82	447
South San Francisco Bay[d]	1,860	92	28

[a] *Smith et al.* [1996].
[b] *Nixon and Pilson* [1983].
[c] *Kemp et al.* [2005].
[d] *McKee and Gluchowski* [2011].

Fjord [*Petersen et al.*, 2008] and zebra mussel (*Dreissena polymorpha*) and quagga mussel (*D. rostriformis bugensis*) introductions to lakes and rivers [*Higgins and Vander Zanden*, 2010]. In all cases, phytoplankton and zooplankton abundance declined significantly, and biological communities were restructured across multiple trophic levels.

[37] Species introductions are the most important cause of bird extinctions and second most important cause of fish extinctions globally [*Clavero and García-Berthou*, 2005], and as many as 80% of the endangered species in some regions of the world are at risk because of pressures from nonnative organisms [*Pimentel et al.*, 2005]. These disruptions have economic costs through loss or reduced productivity of species harvested from coastal waters. For example, extensive losses of the oyster *Crassostrea virginica* along the mid-Atlantic coast of the United States in the 1950s was the result of disease (multi-nucleated sphere unknown (MSX)) caused by the parasite *Haplosporidium nelsoni* introduced from Asia [*Carnegie and Burreson*, 2011]. Stocks of the most abundant fish species in the Caspian Sea (anchovy kilka) virtually collapsed after 2001 because of predation and food competition from the introduced ctenophore *Mnemiopsis leidyi* [*Daskalov and Mamedov*, 2007]. Given the scope of biodiversity and economic losses from species introductions, coastal scientists now advocate national policies to manage introduced species with the same efforts currently applied to reduce chemical pollution and restore wetlands and fisheries stocks [*Williams and Grosholz*, 2008].

6. SEWAGE INPUT

6.1. Background

[38] San Francisco Bay has been described as "The Urbanized Estuary" [*Conomos*, 1979], reflecting the landscape setting of its South Bay between the cities of San Francisco, San Jose (Silicon Valley), and Oakland (Figure 2). The rate of urbanization accelerated in the mid-20th century as the regional population grew from 2.7 million in 1950 to 7.2 million in 2010 (Figure 4). Humans and their industrial, commercial, and agricultural enterprises generate wastes delivered to coastal waters through atmospheric deposition, land runoff, groundwater, and point sources such as discharges from municipal sewage treatment plants (STPs). Sewage effluent contains an array of pollutants including nutrients (nitrogen and phosphorus), organic matter (and its biochemical oxygen demand (BOD)), toxic metals, pharmaceuticals, and pathogens that pose risks to human and ecosystem health [*NRC*, 1993]. Nutrient enrichment can provoke excessive production of algal biomass and sustain harmful algal blooms [*Cloern*, 2001]. Metabolism of algal biomass and organic matter from wastewater can deplete water of dissolved oxygen, and the severity and occurrence of hypoxic dead zones are expanding across the world's coastal waters as a response to anthropogenic nutrient enrichment [*Diaz and Rosenberg*, 2008]. Toxic metals [*Luoma and Rainbow*, 2008] and endocrine-disrupting chemicals (e.g., synthetic estrogen) are commonly found in sewage effluent [*Duffy et al.*, 2009] and can impair growth, reproduction, and immune systems of fish and invertebrates. South San Francisco Bay receives 500,000 m^3 of municipal wastewater annually from 12 STPs serving 4 million people [*McKee and Gluchowski*, 2011]. We use nutrient concentrations as an indicator of how sewage inputs alter the chemistry and potential productivity of this urbanized estuary.

6.2. Nutrient Enrichment

[39] Sewage discharges deliver 11,200 tons of dissolved inorganic nitrogen (DIN) and 1860 tons of dissolved inorganic phosphorus (DIP) to South San Francisco Bay annually [*McKee and Gluchowski*, 2011]. These inputs are 92% and 96% of the combined land-based and atmospheric loadings of DIN and DIP, respectively, and South San Francisco Bay apparently ranks as the U.S. estuary having the largest sewage component of DIN loading (Table 3). The spatial distributions of DIN and DIP concentration show progressive N and P enrichment from the Golden Gate to lower South Bay, reflecting mixing between lower-nutrient ocean water and higher-nutrient Bay water (Figure 13). On an areal basis, STP loadings to South San Francisco Bay are 1860 mmol DIN m^{-2} yr^{-1} and 140 mmol DIP m^{-2} yr^{-1}. As an index of the urbanization effect on nutrient input, we compare these with nutrient loads to Tomales Bay, a smaller estuary just north of San Francisco Bay with a similar latitude and climate but situated in a rural watershed. Inputs to Tomales Bay from atmospheric deposition, groundwater, and surface water inflows are 154 mmol DIN m^{-2} yr^{-1} and 6.8 mmol DIP m^{-2} yr^{-1} [*Smith et al.*, 1996]. Therefore, sewage input to South San Francisco Bay contributes more than 10 times the total DIN input and more than 20 times the total DIP input to Tomales Bay per unit estuary area. As a result, DIN and DIP concentrations are highly elevated in South San Francisco Bay. We show the enrichment effect of wastewater by comparing DIN and DIP concentrations in South San Francisco Bay with those in Tomales Bay and Willapa Bay, another estuary on the U.S. West Coast situated in a rural landscape (Figure 14). The median DIN concentration in South San Francisco Bay is enriched 7.6- and 10-fold compared to these estuaries not having direct sewage inputs. The median

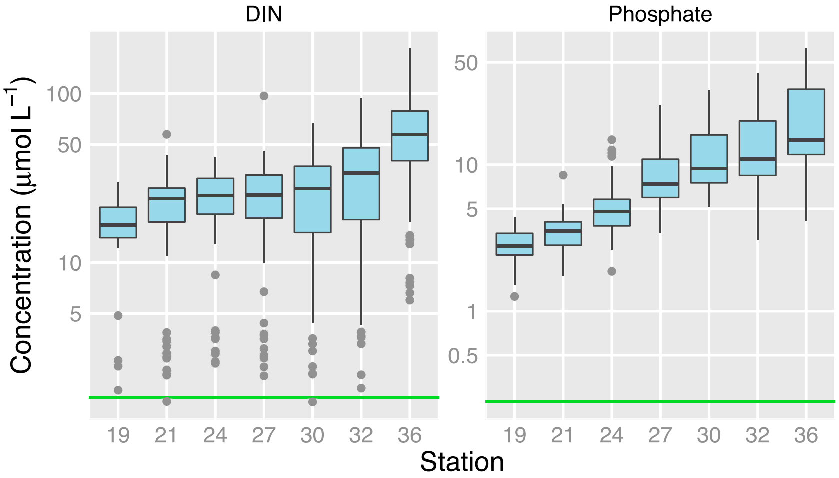

Figure 13. Boxplots showing spatial distributions of dissolved inorganic nitrogen (DIN) and phosphate (dissolved inorganic phosphorus, DIP) in surface waters (0–3 m) of South San Francisco Bay, 1969–2010 (sampling locations shown in Figure 2). Five extreme DIN values >200 or <1 μmol L^{-1} are omitted. The green lines represent characteristic half-saturation constants for DIN and phosphate uptake, respectively, as indices of nutrient levels that potentially limit phytoplankton growth.

DIP concentration in South San Francisco Bay is 3.8 and 8.8 times the median values in Tomales and Willapa bays, respectively. As a result of its setting in an urban landscape, South San Francisco Bay is highly enriched with sewage-derived nitrogen and phosphorus.

6.3. Significance of the Changes

[40] The nutrient concentrations in South San Francisco Bay are typically well above those that limit the growth rate of algae. This is illustrated by comparing DIN and DIP concentrations to the half-saturation constants (K_N, K_P) for phytoplankton growth as an index of potential nutrient limitation (Figure 13). Of 4096 DIN measurements made in South San Francisco Bay from 1969 to 2010, only 126 (0.03%) were smaller than the mean K_N for marine diatoms (1.6 μM [*Sarthou et al.*, 2005]). Only 1 of 4330 DIP measurements was smaller than the mean K_P (0.24 μM [*Sarthou et al.*, 2005]).

[41] Based on these high N and P concentrations, South San Francisco Bay has the potential to produce phytoplankton biomass at levels that severely impair other nutrient-enriched estuaries, such as Chesapeake Bay, where occurrences of large algal blooms have led to summer hypoxia in bottom waters, loss of submerged vascular plants and alteration of biogeochemical processes such as denitrification [*Kemp et al.*, 2005]. The nitrogen input to South San Francisco Bay from sewage disposal is almost twice the total N input from all sources to Chesapeake Bay and its tributaries (Table 3). As a result, N and P concentrations are substantially higher in South San Francisco Bay than in Chesapeake Bay (Figure 14). However, South San Francisco Bay paradoxically has low phytoplankton biomass relative to other enriched estuaries. The median chlorophyll *a* concentration in South Bay is only 4.1 μg L^{-1} (Table 1), but the potential chlorophyll *a* concentration—that expected if the median DIN concentration were converted into phytoplankton biomass—is about 28 μg L^{-1} (assuming a chl-*a*:N ratio of 1 [*Gowen et al.*, 1992]). This high-nutrient low-chlorophyll state implies that San Francisco Bay is inefficient at converting nutrients into algal biomass and, therefore, resistant to the harmful consequences of enrichment observed in other estuaries such as Chesapeake Bay (we show in section 8, however, that this resistance is weakening).

[42] San Francisco Bay has (at least) three attributes that confer resistance to the harmful consequences of nutrient enrichment. First, its strong tidal currents generate sufficient turbulence to break down stratification caused by surface heating and freshwater inflow. Chesapeake Bay has weaker tides, weaker turbulent mixing, and stratification that persists long enough (months) for bottom waters to become and remain hypoxic or anoxic. Salinity stratification can develop in South San Francisco Bay during weak neap tides, and these stratification events promote fast growth of phytoplankton biomass in the surface layer. But the surface blooms dissipate on the subsequent spring tide when the water column is mixed [*Cloern*, 1996]. Second, San Francisco Bay is more turbid than Chesapeake Bay because it receives large river inputs of sediments and is shallow, so sediments are maintained in suspension by wind waves and tidal currents [*May et al.*, 2003]. As a result, the median light attenuation coefficient in South San Francisco Bay (1.4 m^{-1}; Table 1) corresponds to a photic depth of only 3.3 m, and phytoplankton growth rate is limited by low availability of sunlight energy [*Cloern*, 1999]. Third, accumulation of phytoplankton biomass is controlled by bivalve mollusks (clams, mussels) that can filter a volume of water equal to the South San Francisco Bay volume each day during summer [*Cloern*, 1982]. In Chesapeake Bay, this filter-feeding function was provided historically by an oyster population that could filter that bay's volume in less than 4 d. That filtration time is now hundreds of days because the oyster population has been decimated by overharvest, disease, and hypoxia [*Kemp et al.*,

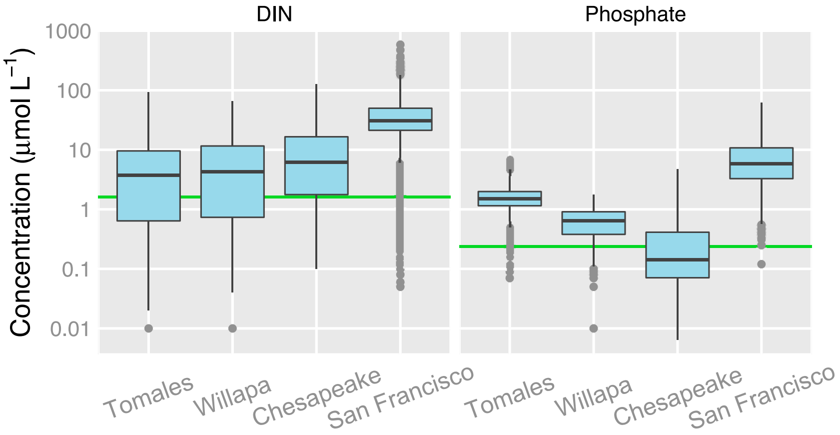

Figure 14. Boxplots of dissolved inorganic nitrogen (DIN) and phosphate in South San Francisco Bay (1969–2010), Tomales Bay (1987–1995), Willapa Bay (1991–2006), and the deep channel of Chesapeake Bay (2006–2010). The data are from all available depths. The green lines represent characteristic half-saturation constants for phytoplankton growth rate.

2005]. Comparative analyses of Chesapeake Bay and San Francisco Bay reveal that estuaries have inherent attributes, such as hydrodynamic, optical and biological properties, that control the efficiency with which nutrients are converted into phytoplankton biomass and, therefore, the expression of nutrient enrichment as a driver of environmental change.

[43] Nutrient pollution from municipal wastewater is a globally significant problem that has degraded water quality, reduced biological diversity, and altered biogeochemical functioning of urban coastal areas such as Boston and New York harbors [*NRC*, 1993], Tampa Bay [*Greening et al.*, 2011], Osaka Bay [*Yasuhara et al.*, 2007], Mersey Estuary [*Jones*, 2006], Hong Kong's Tolo Harbor [*Xu et al.*, 2011], Rio de Janeiro's Guanabara Bay [*Kjerfve et al.*, 1997], Turkey's Golden Horn Estuary [*Taş et al.*, 2006], and Australia's Swan-Canning Estuary [*Hamilton and Turner*, 2001]. Environmental degradation by nutrient overenrichment has motivated local, national and multinational policies to reduce nutrient inputs from urban and agricultural sources to coastal ecosystems. For example, a goal of the Chesapeake 2000 Agreement is to reduce N and P inputs to Chesapeake Bay by 48% and 53%, respectively [*Kemp et al.*, 2005]. These are similar to goals of multinational agreements to halve nutrient inputs to the Baltic Sea and North Sea [*Conley et al.*, 2002]. The Danish government has enacted even more aggressive plans to reduce N inputs to its aquatic environments by 50% and point sources of P by 80% [*Conley et al.*, 2002].

[44] The establishment of such quantitative targets for nutrient reduction is a challenging policy application of estuarine science. Early responses of the Dutch Wadden Sea, Chesapeake Bay, and Danish fjords to nutrient reduction strategies have not all met the expectations of policy makers [*Carstensen et al.*, 2011]. The contrasting responses of San Francisco Bay and Chesapeake Bay to N and P enrichment teach that nutrient loading rate alone is not a good predictor of algal biomass or the impairments associated with high algal biomass, such as hypoxia and harmful blooms. This lesson appears to be general because a broad range of empirical relationships exists between nutrient (e.g., total N) and chl-*a* concentrations measured in 28 coastal systems, providing "overwhelming evidence that system-specific attributes modulate the response of phytoplankton to nutrient enrichment" [*Carstensen et al.*, 2011, p. 9127]. As explained above, these system-specific attributes go far beyond just hydraulic retention time, noted long ago as a factor differentiating water bodies with respect to nutrient loading [*Vollenweider*, 1975]. Policies to remediate overfertilized coastal waters therefore might be most effective and cost efficient if they are tailored to the attributes of individual estuaries and bays. The urgency for place-based nutrient-reduction strategies will likely accelerate in step with continued urbanization and population and economic growth: global sewage emissions are projected to increase from 6.4 Tg of N and 1.3 Tg of P in 2000 to emissions as high as 15.5 Tg N and 3.1 Tg P by 2050, with the fastest increases in southern Asia [*Van Drecht et al.*, 2009].

7. ENVIRONMENTAL POLICY: THE U.S. CLEAN WATER ACT

7.1. Background

[45] In 1972 the U.S. Congress unanimously passed Public Law 92-500, which we know as the Clean Water Act (CWA), to "restore and maintain the chemical, physical and biological integrity of the nation's waters" and attain "fishable and swimmable waters" across the United States. This landmark legislation established the first federal regulation of sewage disposal by requiring secondary treatment of municipal wastewater to reduce inputs of solids, oxygen-consuming chemicals, and pathogens to the nation's waters. The CWA provided funding for construction and improvement of STPs, and it established effluent standards for BOD, suspended solids, fecal coliform bacteria, and pH. Enactment of the CWA and similar policies in other countries reflected growing public concern about the accelerating and increasingly visible degradation of water quality caused by municipal and industrial sources of pollution. The Potomac Estuary of "the Nation's River" was an iconic example of environmental

degradation from sewage pollution, manifested as noxious algal blooms, hypoxia (Figure 1f), fish kills, loss of water clarity and waters unsafe for swimming because of high counts of fecal coliform bacteria [*Jaworski*, 1990]. By the 1970s, regions of nutrient-enriched Tampa Bay lost complete benthic communities and half their seagrasses because of low DO and turbidity caused by high algal biomass [*Greening and Janicki*, 2006]. Perhaps the most infamous sign of the state of U.S. water quality in this era occurred in June 1969 when Cleveland's Cuyahoga River ignited because of its flammable pollutants.

[46] Pollution effects became increasingly visible in San Francisco Bay during the 20th century as the surrounding population grew (Figure 4). The San Francisco Bay–Delta was once the foremost fishing center on the U.S. west coast, but its commercial fisheries for sturgeon, salmon, striped bass, shad, and clams all ended by the 1950s because of habitat degradation, overharvest, and poor water quality [*Smith and Kato*, 1979]. Harvest of oysters—the Atlantic species *Crassostrea virginica* reared primarily in South San Francisco Bay—was the most valuable California fishery in the late 19th century. But oyster culture ended by the 1930s because of poor growth and conditions attributed to urban pollution, and shellfish harvest from San Francisco Bay was quarantined in 1932 to protect human health from waterborne pathogens. At the time of passage of the CWA, fish kills and skin tumors on fish were common in San Francisco Bay, extremes of contamination for toxic metals in clams were equal to or higher than anywhere in the world, and regions of South Bay had seasonal anoxia [*Luoma and Cloern*, 1982]. We use data from sampling programs that began before and continued after passage of the CWA to show how these pollution indicators in South San Francisco Bay changed after national and local policies mandated enhanced sewage treatment.

7.2. Recovery From Sewage-Derived Pollutants

[47] The first measurements of dissolved oxygen (DO) in San Francisco Bay were made in the late 1950s and they showed recurrent summer anoxia in the southernmost region of South Bay (below Dumbarton Bridge, Figure 2). City and regional policies mandated secondary treatment of sewage to reduce BOD inputs to this region even before passage of the CWA. Secondary treatment was fully implemented by all STPs discharging to lower South Bay by 1973. Prior to 1973 municipalities discharged untreated or primary treated sewage, and inputs of oxygen-consuming organic matter and ammonium overwhelmed the assimilation capacity of this region. Summer anoxia was eliminated in the 1970s as secondary treatment was implemented, but DO concentrations still fell below 5 mg L^{-1} (Figure 15c), a common standard to protect marine fish sensitive to low oxygen. The Clean Water Act provided incentives for further improvements in wastewater treatment, and by 1980 all STPs discharging into this region implemented processes to remove 99% of BOD and nitrification to convert ammonium into nitrate. From 1978 to 1980, BOD input from the largest STP (San Jose–Santa Clara) dropped from 3700 to 400 t yr^{-1}, and ammonium-N input dropped from 2800 to 40 t yr^{-1} (Figure 15a). In response, hypoxia was eliminated from San Francisco Bay and DO concentrations are now consistently near or above 5 mg L^{-1} (Figure 15c).

[48] Environmental effects of sewage-derived metals were detected in the 1970s when sediments and clams (*Macoma balthica*) sampled on a mudflat near the Palo Alto Regional STP discharge were found to be highly contaminated with copper, silver, and other metals [*Hornberger et al.*, 2000; *Luoma and Cloern*, 1982]. Copper (310 $\mu g\ g^{-1}$) and silver (103 $\mu g\ g^{-1}$) in clam tissues (Figure 15d) reached levels that impaired reproduction; histological analyses confirmed that clams were nonreproductive; and the invertebrate community had low diversity and was dominated by small forms, diagnostics of disturbance by toxic contaminants [*Hornberger et al.*, 2000]. Although the primary target of advanced wastewater treatment was removal of BOD, incremental additions of new treatment processes (e.g., biological nutrient removal in the 1990s), combined with industrial pretreatment at the source, were also highly effective at reducing metal inputs from STPs. Annual copper loading from the Palo Alto Regional STP was 5800 kg in 1979 but dropped continuously through the 1980s and has been <300 kg since 1995 (Figure 15b). Annual silver inputs declined from 92 kg in 1989 to <10 kg since 1995. As loadings decreased, metal contamination of sediments and biota decreased proportionately. By the 1990s copper and silver concentrations in the clam *Macoma balthica* had dropped 10- and 30-fold, respectively, from their peaks of the 1970s (Figure 15d). With greatly reduced metal contamination, clams became reproductive and larger forms of invertebrates recolonized mudflats near the STP outfall, both evidence that environmental stresses from metals have been greatly reduced since the 1970s. Monthly sampling that began in South San Francisco Bay in 1975 provided one of the first observational records in the U.S. to demonstrate (1) correlation between metal levels in organisms and metal inputs from municipal wastewater and (2) recovery of physiological impairment and biological communities after metal inputs to an estuary were reduced through advanced wastewater treatment [*Hornberger et al.*, 2000; *Luoma and Cloern*, 1982].

7.3. Significance of the Changes

[49] In 1950, 40% of U.S. municipal sewage collection systems discharged untreated sewage. By 1996 virtually all of the nation's 16,000 STPs were using secondary or advanced treatment. As a result, STP discharge of BOD declined nationwide from 6900 t d^{-1} to 3800 t d^{-1} between 1968 and 1996, an era when the population served by STPs increased from 140 to 190 million and the BOD influent to STPs increased 35% [*U.S. Environmental Protection Agency*, 2000]. The responses documented in San Francisco Bay from measurements of DO and metal contamination before and after implementation of the CWA exemplify the measurable improvements in water quality seen in other U.S. urban estuaries. The first (National Pollutant Discharge Elimination System) permits under the CWA were issued in

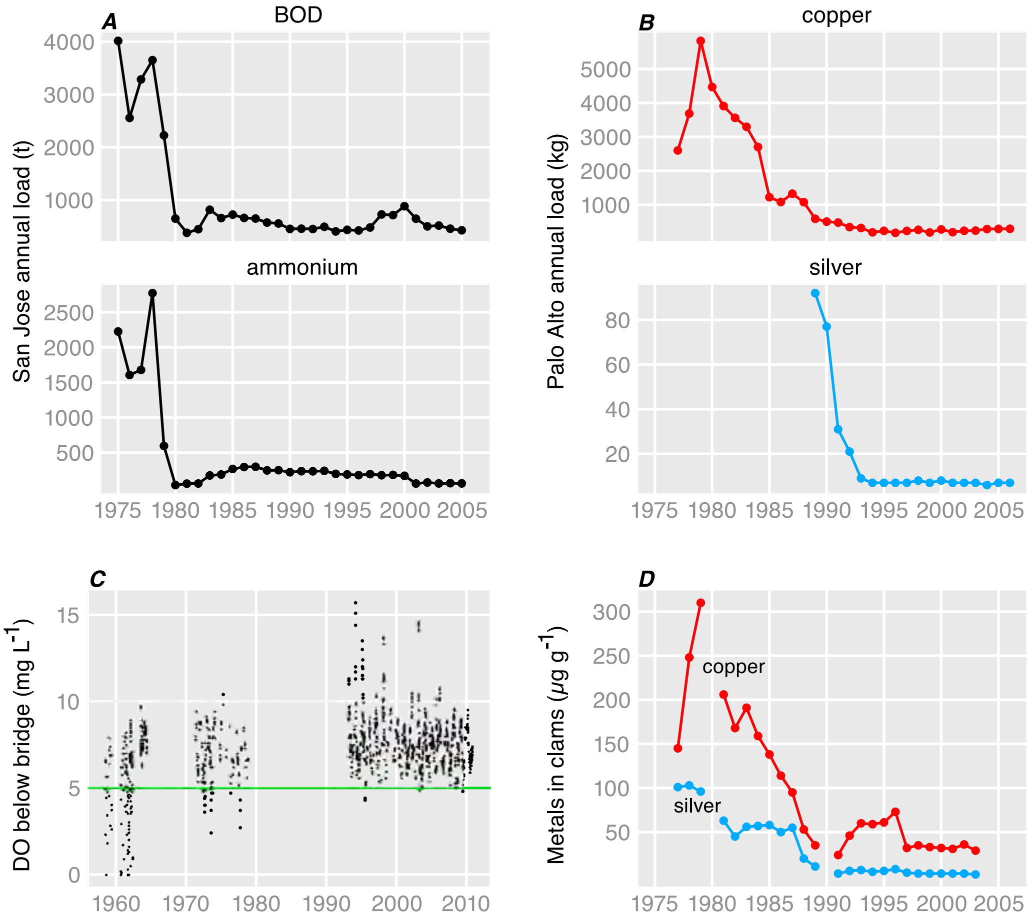

Figure 15. Responses in South San Francisco Bay to improvements in sewage treatment efficiency. (a) Annual loads of BOD and ammonium from the San Jose–Santa Clara wastewater treatment plant. (b) Annual loads of copper and silver from the Palo Alto Regional wastewater treatment plant. (c) Dissolved oxygen south of the Dumbarton Bridge. The green line represents a common standard to protect marine fish sensitive to low oxygen. (d) Copper (red) and silver (blue) concentrations in clam (*Macoma balthica*) tissue from a Palo Alto mudflat.

1974 to STPs discharging to the Potomac Estuary. Treatment processes targeted removal of BOD and phosphorus. Between 1954 and 1985, BOD loadings decreased from 91,000 to 5400 kg d^{-1} and P loadings decreased from 10,900 to 270 kg d^{-1} [*Jaworski*, 1990]. In response, phosphorus concentrations in the estuary fell by 80%, algal biomass decreased 60%, DO increased to >5 mg L^{-1} (Figure 1f) and fish kills no longer occurred. Elsewhere, coliform bacteria decreased tenfold in the lower Hudson-Raritan Estuary from 1968 to 1993 [*Brosnan and O'Shea*, 1996], and metal (copper, cadmium, nickel) concentrations in the Hudson River estuary decreased 36% to 90% after the 1970s [*Sañudo-Wilhelmy and Gill*, 1999]. Seagrass cover expanded more than 2000 ha in Tampa Bay after nitrogen inputs were reduced 60%, and seagrass recovery continues as algal biomass decreases and water clarity increases [*Greening et al.*, 2011]. By these kinds of measures, the Clean Water Act was a highly successful policy to reduce point-source pollution of estuaries through water treatment technology.

[50] Policies similar to the CWA were implemented elsewhere to reduce wastewater pollutant loadings at the scale of municipalities (e.g., Hong Kong's Water Pollution Control Ordinance [*Xu et al.*, 2011]; Perth's Swan Canning Cleanup Program [*Hamilton and Turner*, 2001]; Golden Horn Rehabilitation Project [*Taş et al.*, 2006]), or through national policies (UK Clean Rivers (Estuaries and Tidal Waters) Act [*Matthiessen and Law*, 2002]; Denmark's Action Plan on the Aquatic Environment [*Conley et al.*, 2002]), or through multinational agreements such as the European Community's Urban Wastewater Directive [*Hering et al.*, 2010]. Perhaps the most publicized rehabilitation was of the river Thames, which historically supported runs of Atlantic salmon but had regions devoid of oxygen and fish from 1920 to 1964, largely because of sewage inputs from London. Fish, invertebrates, and water birds returned to the Thames Estuary after London's sewage works were upgraded in the 1960s, and by 1976 a cumulative total of 112 fish species had returned, including adult salmon for the first time in 140 years [*Attrill*, 1998]. Similarly, dissolved oxygen concentrations increased in Victoria Harbor after Hong Kong implemented secondary sewage treatment in 2001 [*Xu et al.*, 2011], metal and polychlorinated biphenyl (PCB) concentrations in sponges decreased in Cortiou Cove on the French Mediterranean after Marseille implemented primary

sewage treatment [*Perez et al.*, 2005], and benthic invertebrates and fish returned to Spain's heavily polluted Nervión Estuary [*Borja et al.*, 2006] and the UK's Mersey Estuary [*Hawkins et al.*, 2002] after metal and BOD inputs were reduced with sewage treatment.

[51] These case studies illustrate the success of policies to rehabilitate estuarine-coastal ecosystems from the severe degradation of water quality and disruption of biological communities caused by disposal of untreated municipal waste. However, the standards prescribed in the Clean Water Act have not been fully met. For example, a 2004 assessment of 141 U.S. estuaries determined that the majority have moderate or high symptoms of eutrophication expressed as low DO, loss of submerged vascular plants, proliferation of macroalgae, or harmful algal blooms [*Bricker et al.*, 2007]. Nutrient pollution remains the largest pollution problem in U.S. coastal rivers and bays [*Howarth et al.*, 2002]. In some estuaries, such as those of the northeast United States [*Whitall et al.*, 2007], sewage input remains the largest source of nitrogen, reflecting the design of secondary sewage treatment to reduce BOD (not nutrients) in effluent. For others, such as Australia's Sydney estuary, storm water runoff from urban watersheds is the primary source of nutrients and metals [*Beck and Birch*, 2012]. The largest sources of nitrogen to most estuaries, however, are the diffuse nonpoint sources generated by agriculture and fossil fuel combustion [*Howarth et al.*, 2002]. The risk for coastal eutrophication will likely continue to grow in many world regions because anthropogenic activities will increase river nutrient loading and shift nutrient ratios toward those favoring blooms of harmful algae [*Seitzinger et al.*, 2010].

[52] Certain contaminants produced in the past remain in watersheds and persist in estuarine sediments. For example, mercury and PCBs remain priority pollutants in South San Francisco Bay because they are persistent and accumulate in food webs to levels that are health risks to birds, harbor seals, and humans [*Grenier and Davis*, 2010]. Contemporary sources include atmospheric deposition (mercury) and urban runoff (PCBs). Moreover, accelerating erosion—a consequence of reduced sediment supply (section 4)—is now exposing buried sediments having high concentrations as legacy contaminants from the gold mining era (mercury) and later era of PCB manufacture. A second priority is new contaminants that persist and accumulate in food webs, such as polybrominated diphenyl ethers (PBDEs) (brominated flame retardants). These compounds were undetected in the 1980s, but residues are now common in water, sediments, and biota of South San Francisco Bay where concentrations in harbor seals, bird eggs, and humans are among the highest recorded [*Grenier and Davis*, 2010]. The sources and environmental effects of PBDEs are largely unknown, but their presence illustrates the challenge of maintaining the chemical and biological integrity of estuaries when new contaminants emerge faster than our capacity to identify their sources and assess their effects.

[53] The Clean Water Act and similar policies of other countries have greatly reduced inputs of organic matter, pathogens, and toxic contaminants to coastal waters, with demonstrable improvements in water and habitat quality. However, standards prescribed in the CWA have not been fully met. Many of the world's estuaries, bays, and inland seas are still not fishable and swimmable. Further rehabilitation, or even maintenance of the status quo, will require innovative strategies to solve the much more difficult problems of nonpoint sources of nutrients and toxic contaminants [*Brown and Froemke*, 2012; *Smith et al.*, 1987], legacy contaminants from the past, and new contaminants of the future.

8. SHIFT IN THE OCEAN-ATMOSPHERE SYSTEM

8.1. Background

[54] Fishermen have known for centuries that fish abundance in the sea fluctuates between eras of good and poor catch that are tied to climate variability. Four centuries of catch records from the English Channel show oscillations of herring and sardine stocks that are synchronized with shifts between cold and warm periods [*Southward et al.*, 1988]. More recently, marine and atmospheric scientists have discovered that population fluctuations of fish and their food resources are synchronized with shifting patterns of atmospheric pressure over ocean basins that are represented by climate indices: recruitment of yellowfin tuna is highest in the tropical Pacific after El Niño events [*Lehodey et al.*, 2006], salmon stocks in Alaska and California fluctuate inversely with the Pacific Decadal Oscillation (PDO) [*Mantua et al.*, 2002], and cod recruitment in the North Sea is high when the North Atlantic Oscillation (NAO) is positive [*Stige et al.*, 2006]. Progress is advancing to understand the linkage mechanisms between fish abundance and these climate patterns through their influence on ocean currents, temperature, and primary and secondary production [*Lehodey et al.*, 2006].

[55] Discovery of climate-related variability of marine fish populations is rooted in historical observations, including catch records from some fisheries that have been maintained for a century or longer and therefore capture variability over multiple periods of the NAO, PDO, and other multidecadal climate patterns. Most observational records in the world's estuarine-coastal systems are much shorter, but they are becoming long enough that we can begin to ask if and how variability in estuaries, bays, and lagoons is related to interdecadal shifts in atmospheric forcing across ocean basins. Given the intense human modification of estuarine-coastal systems through changes in freshwater and sediment input, introductions of alien species, and nutrient enrichment, there is uncertainty that ecological responses to oscillating climate patterns can be detected in observations having the large and varied signals of human disturbance shown above [*Cloern and Jassby*, 2008]. Here we summarize an ecological regime shift that occurred in South San Francisco Bay after a shift in atmospheric pressure patterns across the North Pacific Ocean. Detection of this regime shift and its attribution to a climatic process was possible because of observations sustained more than two decades before and a decade after the climate shift.

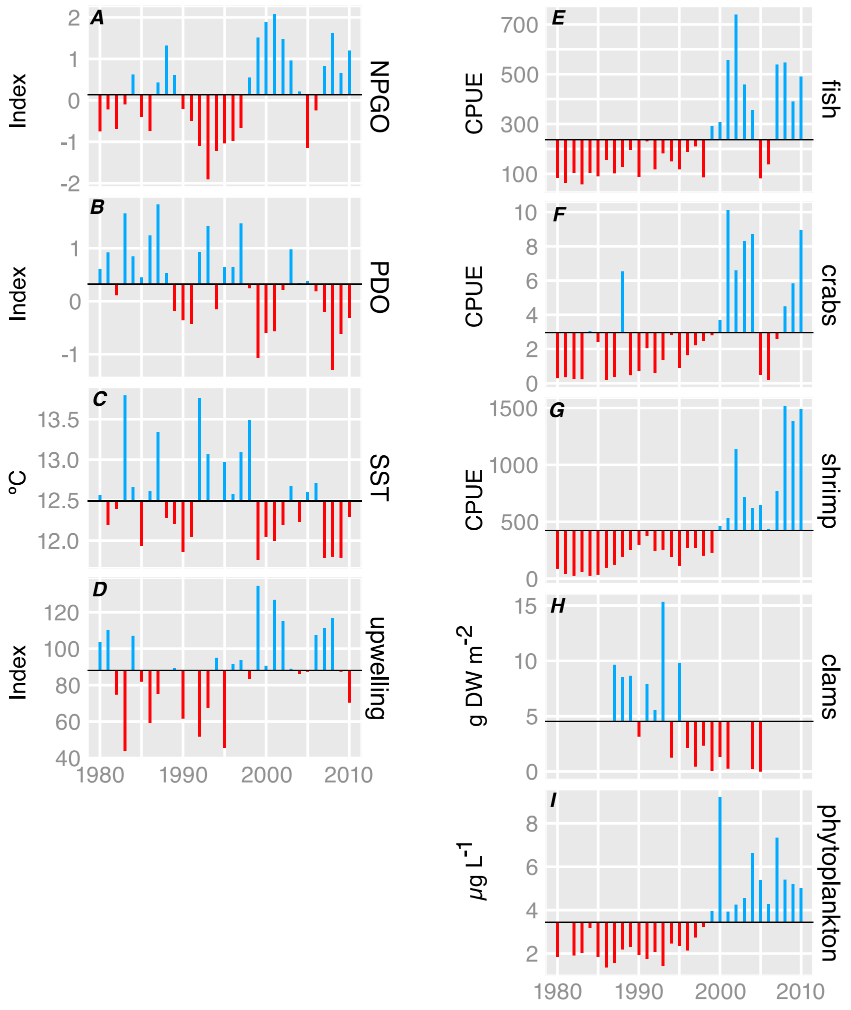

Figure 16. Time series of annual mean climate indices, ocean conditions near the mouth of San Francisco Bay, and annual mean abundances of various biota within San Francisco Bay, shown as anomalies about the long-term means. (a) North Pacific Gyre Oscillation. (b) Pacific Decadal Oscillation. (c) Sea surface temperature at Farallon Islands. (d) Upwelling index at 39°N. Sum of catches per unit effort in the marine subembayments (South, Central, and San Pablo bays) for (e) five species of demersal fish (age-0 English sole, speckled sanddab, plainfin midshipman, bay goby, and Pacific staghorn sculpin), (f) three species of crabs (age-0 Dungeness, slender, and brown rock crab), and (g) two species of shrimp (blacktail bay shrimp and Stimpson coastal shrimp). (h) Dry weight of clams from all available shallow sampling sites in South Bay. (i) Annual mean phytoplankton biomass (chlorophyll *a*) in surface waters of the three South Bay stations sampled most frequently (24, 27, and 30).

8.2. Ecosystem Regime Shift

[56] The largest observed El Niño event occurred in 1997–1998, and it was followed by an equally strong La Niña in 1999. This abrupt transition appears to demarcate a climatic regime shift in the North Pacific manifested as a change in atmospheric pressure and wind patterns, ocean temperature, and biological productivity. This shift was "possibly the most dramatic and rapid episode of climate change in modern times" [*Peterson and Schwing*, 2003, p. 1899]. It was expressed as sign changes in the two prominent modes of sea surface temperature and sea level pressure across the North Pacific [*Chenillat et al.*, 2012]: the PDO, which became strongly negative, and the North Pacific Gyre Oscillation (NPGO), which became strongly positive in 1999 (Figures 16a and 16b). Regional responses to these ocean-basin scale indices were measured in the coastal ocean adjacent to San Francisco Bay as cooling of surface waters and increase in the upwelling index (Figures 16c and 16d). Intensification of upwelling and cooling at this latitude are responses to strengthened equatorward winds and equatorward transport in the California Current that are most strongly correlated with the NPGO. Because of this, the NPGO is a primary indicator of upwelling, nutrient supply to phytoplankton, and primary production in the California Current System (CCS) [*Di Lorenzo et al.*, 2008]. Regional ocean models simulate higher coastal nitrate concentration, chlorophyll *a*, and zooplankton biomass in the central CCS, and the differences result from both earlier and stronger upwelling in NPGO$^+$ compared to NPGO$^-$ regimes

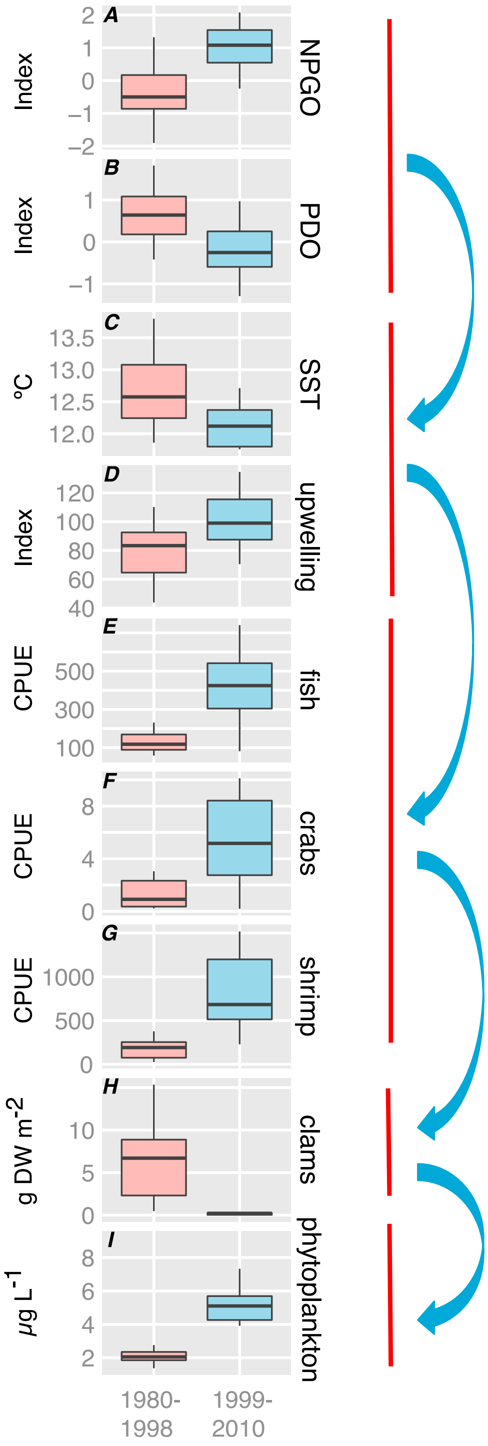

Figure 17. (a–i) Ecosystem regime shift, depicted in boxplot distributions of the time series in Figure 16, divided into years before and after the 1998–1999 climate shift.

[*Chenillat et al.*, 2012]. Therefore, large-scale processes of ocean-atmosphere coupling captured in Pacific climate indices like the NPGO have important ramifications for biological productivity in the coastal waters adjacent to San Francisco Bay.

[57] Unexpectedly, major changes in biological communities inside San Francisco Bay followed the shift of the northeast Pacific to its cool phase [*Cloern et al.*, 2007]. We show examples as abundance indices of demersal marine fish, crabs, and shrimp species (Figures 16e–16g) that migrate into estuaries, either as adults to reproduce (shrimp) or as juveniles produced in the coastal ocean (e.g., English sole, Dungeness crab). Populations in each of these communities reached record-high levels during or soon after the 1998–1999 climate shift, and abundances have remained above their 1980–2010 means since, except for 2005 and 2006 when the NPGO was weak. Synchronous with these changes was an increase of phytoplankton biomass (chlorophyll *a*) in South Bay that has exceeded the long-term mean each year since 1999. Bivalve mollusks disappeared from shallow regions of South Bay in 1999 (Figure 16h), and bivalves have remained scarce in these regions during the past decade (J. Thompson, USGS, personal communication, 2012).

[58] These observations reveal an ecological regime shift in San Francisco Bay that was coherent with a climate shift in the north Pacific, suggesting a previously unrecognized linkage between interdecadal variability of the ocean-atmosphere system and biological communities inside this estuary. We compare the two regimes as box plots of NPGO, PDO, upwelling and SST in the coastal ocean, and abundances/biomass of organisms in marine-influenced regions of San Francisco Bay for the periods 1980–1998 (warm) and 1999–2010 (cool). The arrows (Figure 17) depict a cascade of responses from the ocean-basin scale (shift to $NPGO^+/PDO^-$), to the regional scale (cooling, strengthening of upwelling), and to San Francisco Bay as population changes of organisms across a range of trophic levels. The demersal fish, crabs, and shrimp are all predators that feed on bivalve mollusks, so the absence of bivalves since 1999 may be a result of increased predation mortality, as observed in other estuaries [*Beukema and Dekker*, 2005]. The phytoplankton increase is presumably, then, a response to decreased bivalve grazing [*Cloern et al.*, 2007]. Therefore, South San Francisco Bay's biological communities were reorganized through a trophic cascade that was initiated by an abrupt increase in abundance of predators whose populations most closely track the NPGO [*Cloern et al.*, 2010]. These predators include juvenile forms that migrate into San Francisco Bay, so their high abundances since 1999 must reflect changes in their production rate in the coastal ocean. A likely mechanism is the enhanced plankton food supply to early life stages of flatfish and crabs by the shift from a warm, low-production to a cool, high-production regime of the northeast Pacific that amplifies abundances of zooplankton, pelagic fish, and seabirds in the CCS [*Peterson and Schwing*, 2003]. The last climate shift in the north Pacific occurred in 1976, before regular biological sampling began in South San Francisco Bay, so we have no observations (or even proxies) to determine if the current ecosystem regime existed in earlier cool regimes of the northeast Pacific (e.g., 1948–1976). Validation of the linkages hypothesized in Figure 17 will require sampling through the next warm regime, so we tell students that a single career is not long enough.

[59] Every measured component of phytoplankton dynamics changed in South Bay after the climate shift. The earliest sign of change was a surprising bloom in October 1999 (Figure 18a), the first occurrence of an autumn bloom and a

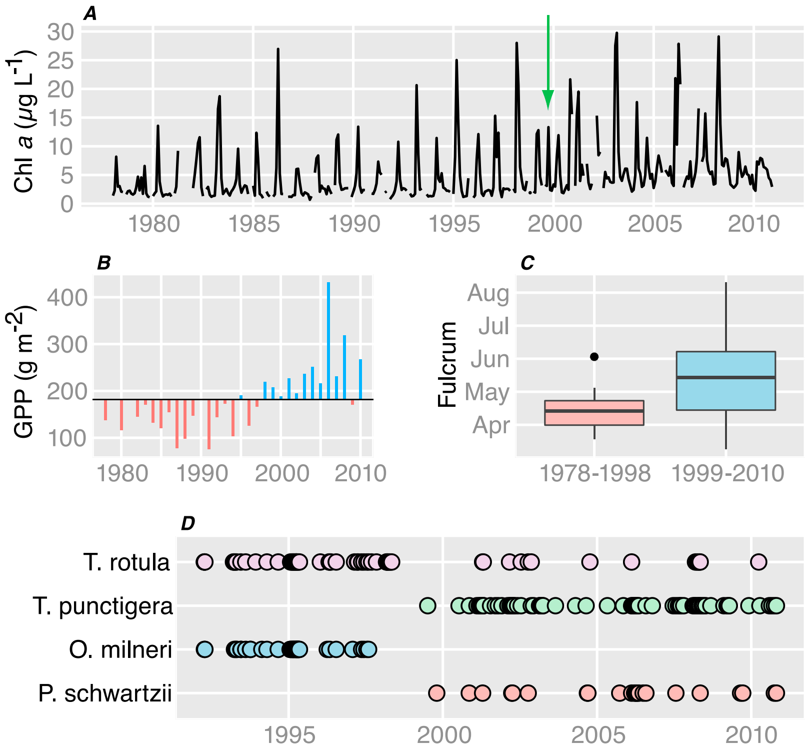

Figure 18. Behavior of the South San Francisco Bay phytoplankton community around the time of a Pacific Ocean regime shift in 1998–1999. (a) Monthly mean phytoplankton biomass (chlorophyll *a*) in surface waters of the three South Bay stations sampled most frequently (24, 27, and 30). Arrow shows the appearance of the first autumn-winter bloom in October 1999. (b) Calculated annual gross primary production averaged for the same three stations. (c) Boxplot distributions of the fulcrum, i.e., the timing of the center of gravity of the annual chlorophyll pattern, for the two eras. (d) Occurrence of four phytoplankton species in South and Central bays. Circles are plotted at each date when the indicated species was detected. *Thalassiosira rotula* and *Thalassiosira punctigera*, both centric diatoms, were the dominant species before and after the shift, respectively. *Oxytoxum milneri* gave way to *Polykrikos schwartzii*, both heterotrophic dinoflagellates.

departure from the canonical 1978–1998 pattern of one (spring) bloom each year. Other autumn-winter blooms have occurred since, including the first observed dinoflagellate red tide in September 2004 [*Cloern et al.*, 2005]. Calculated gross primary production has increased during the past decade, and it exceeded the long-term mean every year since 1997, except one (Figure 18b). As a simple index of the seasonal development of phytoplankton biomass, we calculated the day each year when cumulative chlorophyll-*a* reached the midpoint ("fulcrum") of annual cumulative chlorophyll-*a*. This index shifted +31 days (from mid-April to mid-May) between 1978–1998 and 1999–2010 (Figure 18c), reflecting the new occurrences of autumn-winter blooms and overall increases in summer biomass.

[60] We illustrate phytoplankton community changes as occurrence timelines of two marine diatoms (*Thalassiosira rotula*, *T. punctigera*) and two heterotrophic (nonphotosynthetic) dinoflagellates (*Oxytoxum milneri*, *Polykrikos schwartzii*). *Thalassiosira rotula* occurred commonly and was the biomass-dominant species in San Francisco Bay during 1992–2001 [*Cloern and Dufford*, 2005]. It occurred less frequently in the past decade (Figure 18d), and its contribution to biomass during 1999–2010 was ranked only 54th. The biomass dominant after 1999 was *T. punctigera*, a species not observed previously. *Oxytoxum milneri* occurred commonly before but was never observed after 1997. Conversely, *Polykrikos schwartzii* first appeared in 1999 and has been observed regularly since (Figure 18d). The synchrony of these species appearances and disappearances with the 1998–1999 climate shift suggests that they are related to phytoplankton species changes in the coastal ocean, perhaps analogous to the switch from warm- to cold-water copepod species in the northern CCS after 1998 [*Peterson and Schwing*, 2003]. However, our knowledge of the life cycles, biogeography, and physiological ecology of marine phytoplankton is not sufficient to explain why one *Thalassiosira* species would nearly completely replace another or why one heterotrophic dinoflagellate would completely replace another at about the same time. We also know surprisingly little about the ecological significance of these kinds of species changes, although they could be substantial because of differences in cell size, behavior, and nutritional quality among phytoplankton species [*Cloern*, 1996]. The diminished biomass of *Thalassiosira rotula*, for example, is intriguing because this marine diatom produces oxylipins that arrest hatching of copepod eggs in laboratory experiments [*Carotenuto et al.*, 2011]. The environmental

significance of these experiments is debated, but they highlight the critical need for increased knowledge of biological interactions at the species level to understand the significance of climate-driven community changes at the ecosystem level.

8.3. Significance of the Changes

[61] The San Francisco Bay response exemplifies an emerging principle: variability in coastal oceans can be a powerful driver of variability inside estuaries and bays, and shifts in large-scale ocean-atmosphere dynamics can induce ecological regime shifts in estuaries and bays through their connectivity to coastal oceans. Other examples are accumulating. Species diversity, abundance, and growth rates of juvenile marine fish in the Thames Estuary are significantly correlated with the NAO index [*Attrill and Power*, 2002]. Recruitment of Atlantic croaker (*Micropogonias undulatus*) along the U.S. East Coast is highest during the warm (+) phase of the NAO, when estuarine water temperature and overwinter survival of juveniles are high [*Hare and Able*, 2007]. Shellfish toxicity in Puget Sound is associated with the warm (+) phase of the PDO because warming promotes growth of the toxin-producing dinoflagellate *Alexandrium catenella* [*Moore et al.*, 2010]. Perhaps the most striking climate-driven transformation of an estuarine ecosystem has occurred in Narragansett Bay, where a 1.7°C winter warming of coastal waters since 1970 has been accompanied by loss of the traditional winter phytoplankton bloom, 40%–50% reduction in primary production, reduced supply of organic matter to sediments, decreased benthic metabolism and abundance of demersal fish, and a switch of N cycling from net denitrification to net N fixation [*Nixon et al.*, 2009]. Historical management of estuarine ecosystems has been based from a landward-looking perspective because of human disturbances to surrounding landscapes (sections 3–7). However, water quality, system production, biological communities, and biogeochemical cycling in estuaries also respond to oceanographic processes influenced by large-scale climate patterns.

[62] The oceanic drivers of estuarine variability add to the already complex challenge of managing water quality and living resources of nearshore coastal ecosystems because they can confound outcomes of actions to reduce effects of human disturbance, such as nutrient enrichment. In the Hood Canal of Puget Sound, where scientific investigations are focused on the genesis of bottom water hypoxia, the coastal ocean is a significant source of nutrients providing, for example, more than 90% of the nitrogen input [*Steinberg et al.*, 2011]. Recent studies in the Columbia River Estuary documented multiple intrusions of deep, low-DO coastal water brought to the surface by wind-driven upwelling and transported into the estuary by tidal advection and estuarine circulation [*Roegner et al.*, 2011]. Tidal dispersion drives inputs of phytoplankton biomass into San Francisco Bay during the upwelling season [*Martin et al.*, 2007], and metabolism (and therefore oxygen consumption) in Tomales Bay is fueled by inputs of phytoplankton biomass produced in the adjacent upwelling system [*Smith et al.*, 1996]. Harmful algal blooms in coastal waters provide inocula for blooms of harmful species to develop in estuaries [*Cloern et al.*, 2005]. Therefore, inputs from the ocean and especially from upwelling systems can have similar consequences to those of anthropogenic nutrient enrichment: high nutrients and phytoplankton biomass, low DO, and harmful algal blooms.

9. THE MONITORING IMPERATIVE

9.1. Patterns of Temporal Change

[63] The study of temporal change has always been an important part of ecology. Diel, tidal, annual, and longer cycles in population behavior and abundance are the most obvious and a matter of study since the early days of "modern" ecology [*Elton*, 1927]. Noncyclic change is also a fundamental component of ecological understanding, such as the continuous species colonization and extinction within communities revealed by island biogeographical studies [*MacArthur and Wilson*, 1963] and the maturation of undisturbed whole ecosystems so well summarized by *Odum* [1969]. In the last few decades, the possibility of chaotic dynamics in populations [*Hastings et al.*, 1993] and biotic responses to decadal ocean-atmosphere regimes (section 8) have been a focus of investigation. Accordingly, stationarity of ecological time series—which implies constant mean and variance—has never been a dominant idea in ecology, even for ecosystems not subject to anthropogenic impacts. Although a characteristic average long-term state has traditionally been a useful principle in related fields such as hydrology, even that assumption is now recognized as untenable in the face of strong climate change [*Milly et al.*, 2008].

[64] A key feature of many temporal patterns observed in nature—including both environmental [*Steele*, 1985] and biotic [*Pimm and Redfearn*, 1988] time series—is that the general appearance of the pattern is more or less unchanged when observed at different time scales. Although known for a long time, this scale invariance became a subject of study only in the 1970s when the needed mathematical tools became available [*Gisiger*, 2001]. Now we know that ecological variability can often be described by a power law spectrum proportional to $(1/f)^{\nu}$, where f is frequency (cycles yr^{-1}) and $0 \le \nu \le 2$ [*Cyr and Cyr*, 2003; *Halley*, 1996]: $\nu = 0$ corresponds to white noise, which emphasizes short time scales; $\nu = 2$ corresponds to a random walk or brown noise, which emphasizes long time scales; and $\nu = 1$ corresponds to $1/f$ or pink noise, which is not biased toward any particular time scale.

[65] Pink noise, as opposed to the traditional white noise, appears to be the most suitable null model for ecological time series [*Halley and Inchausti*, 2004]. The prevalence of pink noise has implications for estuarine monitoring: what distinguishes pink and even "redder" noise with $\nu > 1$ is that variance continues increasing, no matter how long the time series. Given that marine environmental variables such as temperature tend to have reddened spectra, and terrestrial whitened, at least for time periods up to 50–100 years [*Vasseur and Yodzis*, 2004], the variability spectrum for estuaries may therefore depend on the relative importance of

oceanic and terrestrial drivers. In particular, estuaries with strong ocean influence, such as San Francisco Bay, may have redder spectra and be more influenced by relatively slow or rare environmental fluctuations with long periods. The data requirements to reliably distinguish $1/f$-noise from alternative noise models and to estimate ν [*Fleming*, 2008] are greater than currently available for San Francisco Bay biotic and water quality variables. Nonetheless, we should anticipate continuous and unexpected long-term changes and trends in the estuary's drivers.

9.2. Implications for Monitoring

[66] These patterns of temporal change imply several lessons for monitoring, which we summarize briefly in the context of San Francisco Bay and Delta. First, discrete monitoring programs must take into account variability on scales shorter than the sampling interval because of potential uncertainty and bias due, for example, to tidal and diel cycles [*Jassby et al.*, 2005; *Lucas and Cloern*, 2002; *Lucas et al.*, 2002]. Moreover, the assumption of white or Gaussian noise may not be appropriate to account for this uncertainty. Monitoring programs must, at some point, include focused, higher-frequency studies to understand the effects of shorter time scales. *Milly et al.* [2008] also caution that the nature of variability on these shorter scales may itself be changing, requiring more attention and adjustments to our current assumptions and sampling designs.

[67] Second, pink noise implies the presence of slow processes that cannot be identified without a commitment to ongoing sampling. Of our case studies, the gradual recovery of water clarity over decades after a short period of hydraulic mining was perhaps the slowest process. This recovery was a "noisy" one (Figure 9), which could not be identified with a data set much shorter than the 35 years available. One can assume that there are many slow trends underway which we cannot yet see. Inclusion of higher-frequency automated monitoring cannot substitute for sites that already have a long record, even if at a much lower frequency. Commitments are therefore essential to continue sampling into the indefinite future at a small number of sentinel sites that are the most representative of their subregions and have the longest monitoring records [*Burt et al.*, 2010; *Jassby*, 1998]. Uninterrupted and long data records are key to the most informative monitoring programs [*Southward*, 1995].

[68] Many estuaries do have monitoring programs that are sustained because of mandates. In the San Francisco Bay–Delta, for example, some long-term monitoring programs have persisted through budget shortfalls because they are mandated in water rights decisions and biological opinions about the long-term operations of the Central Valley and State Water projects. Mandated monitoring thus improves the chances for long-term survival. Unfortunately, it can also be difficult to modify a mandated program even if its design could be made more informative or if better sampling and analysis methods become available. The cost of long-term, historically mandated programs may also make it difficult to establish new, potentially more useful programs. Moreover, data management, analysis, synthesis, and communication have been much less sustainable than the sampling itself, especially during budget shortfalls [*Hughes and Peck*, 2008]. Accordingly, monitoring programs themselves need to be "adaptively managed" while recognizing two essential principles: a corresponding capacity for useful data synthesis must accompany changes in sampling, and at least a small number of sentinel sites must survive changes in the programs.

[69] Third, some processes may be so rare that we have no experience with them, but they may be large, have a disproportionate effect, and need immediate attention. In the world of finance statistics, the term "black swan" has been coined for these events, which have had such widespread and devastating consequences [*Taleb*, 2007]. Perhaps the pelagic organism decline described below could be considered such an event. But there is evidence that such catastrophic ecosystem events have statistical early warning indicators [*Carpenter et al.*, 2011; *Scheffer et al.*, 2009] that could be observed in routine monitoring. Indeed, the word *monitoring* derives from the Latin *monere*, to warn. Black swans require both continuity of long-term monitoring and a high enough sampling frequency to calculate these early warning indicators.

[70] Finally, some processes are the result of multiple drivers that cannot be sorted out with short data sets. For example, a nonlinear regression model with three coefficients (in addition to the intercept) was required to disentangle the decline of suspended particulate matter from interannual variability in flow (equation (1) and Figure 10). Simulation studies suggest that 10–20 observations are required per regression coefficient [*Harrell*, 2001], which means that 30–60 years are needed just for this simple model.

[71] Yet multiple drivers and more complex interactions are probably the rule. Perhaps the most prominent example from this estuary is the marked decrease during the past decades of some estuarine-dependent fish, some to the point of near extinction. Particular interest in the past decade has been given to four pelagic fish species whose populations all declined significantly in the early 2000s: delta smelt, longfin smelt, striped bass (*Morone saxatilis*), and threadfin shad (*Dorosoma petenense*) [*Sommer et al.*, 2007]. Population declines of multiple species across multiple trophic levels are symptoms of an intensely disturbed ecosystem, and the most urgent questions posed to the scientific community are, What are the drivers of these population declines, and what actions can be taken to promote recovery and then sustain populations of fish endemic to this estuary? These questions have social and economic significance because "the solutions under consideration include major investments in infrastructure, changes in water management, and rehabilitation of species' habitats that collectively will cost billions of dollars" [*Thomson et al.*, 2010, p. 1432].

[72] Often implicit in these questions is the expectation (or hope) that stressors on the estuary originate from a single or small set of drivers whose effects can be mitigated. However, the fish and plankton species of concern have distinct life histories and varying patterns of population collapse over time, suggesting that stressors arise from multiple, interacting drivers of change associated with

human disturbances and climate anomalies such as sustained drought. Results from two independent analyses (multivariate autoregressive modeling and change point analysis) are consistent with this interpretation. They reveal significant associations between population declines of individual species during the period 1967–2007 and each of the drivers described in sections 3–5, e.g., negative associations between longfin smelt abundance and salinity (X2), striped bass abundance and water clarity, delta smelt abundance and water exports during winter, and density of *Corbula* and calanoid copepods [*Mac Nally et al.*, 2010; *Thomson et al.*, 2010]. Our preceding remarks about the suspended particulate matter decline should make it clear that many decades of observation may be required to understand the relative importance and modes of operation of so many drivers.

9.3. Monitoring and Environmental Policy

[73] A consensus has emerged from these and other analyses [*BDCP*, 2010; *Brown and Moyle*, 2005; *NRC*, 2010] that the challenge of sustaining communities of native species will require an integrated strategy to mitigate the cumulative effects of water consumption and diversions, river impoundments, introductions of nonnative species, and other human disturbances such as land-use change [*Cloern*, 2007] and inputs of toxic pollutants [*Brooks et al.*, 2012]. The challenge will grow as the effects of global warming drive further changes in freshwater inflow, sea level, water temperature, salinity, and sediment inputs [*Cloern et al.*, 2011]. A general lesson, relevant to other damaged ecosystems, emerges from decades of sustained and careful observation in San Francisco Bay: “a holistic approach to managing the ecology of imperiled fishes in the delta will be required if species declines are to be reversed” [*NRC*, 2010, p. 42].

[74] In the past, sustained monitoring gave rise to the Clean Water Act and then proved its efficacy. More recently, monitoring of the *Corbula* invasion has spurred rules regarding discharge of ballast water into San Francisco Bay and other estuaries. The same sustained monitoring is now enabling us to anticipate emerging problems, exemplified by the changing role of nutrients in South San Francisco Bay, as we now describe. Marine species use estuaries as nursery habitats, and recruitment from estuaries can strongly drive marine population dynamics of many commercial fish species [*Attrill and Power*, 2002]. Observations in San Francisco Bay revealed that the immigration of marine fish and invertebrates can also drive biological community changes inside estuaries (section 8). These community changes have altered the balance between phytoplankton production and consumption, leading to increases in phytoplankton biomass and primary production during the past decade. The trend of increasing primary production from 1978 to 2010 is ecologically significant because it spans the ranges defining oligotrophic (low-production, <100 g C m^{-2} yr^{-1}), mesotrophic (moderate-production, 100–300 g C m^{-2} yr^{-1}), and eutrophic (high-production, >300 g C m^{-2} yr^{-1}) estuaries [*Nixon*, 1995]. Thus, South San Francisco Bay shifted from an oligo-mesotrophic estuary to a meso-eutrophic estuary after 1997 (Figure 18b). This upward shift signals an increased efficiency in the conversion of nutrients into algal biomass and a weakening of the estuary’s resistance to the harmful consequences of nutrient enrichment. Nutrient enrichment of San Francisco Bay was not a concern to water-quality managers in the past, but it is now and they ask, Is South Bay on a trajectory toward the impairments seen in Chesapeake Bay, what standards are appropriate for protecting the ecological integrity of this estuary, and will policies be required to mandate additional wastewater treatment processes to remove nutrients?

[75] Policies to reduce nutrient loadings to other estuarine-coastal systems have had mixed and sometimes disappointing results. These policies are usually established from an assumed functional relationship between responses, such as amplified algal biomass, and nutrient loading rate [*Carstensen et al.*, 2011]. Target responses are selected, and appropriate nutrient loading rates are then prescribed from the functional relationship. For some estuaries, such as Tampa Bay, this approach has been highly successful, and steady progress has been made to reduce algal biomass and recover seagrasses [*Greening et al.*, 2011]. In other cases, costly programs to reduce nutrient inputs have had unexpected results as algal responses have been muted and shown lags and hysteresis, patterns interpreted as manifestations of “shifting baselines” [*Carstensen et al.*, 2011]. Observations in San Francisco Bay illustrate a shifting baseline after biological communities were restructured through a climate regime shift (Figure 17). These links between climate regimes, biological communities and water quality are newly emerging themes of estuarine-coastal research [e.g., *Cloern et al.*, 2007; *Nixon et al.*, 2009], so they have not yet been considered in most nutrient-management strategies.

9.4. Concluding Perspective

[76] In this review we used data sets collected in a well-studied estuary to illustrate how the coupling of regular sampling with ongoing analyses and retrospective syntheses has become a powerful research approach for understanding ecosystem dynamics at time scales longer than the duration of individual grants. This approach has been used broadly to reveal that the long-term behavior of estuarine habitats, biological communities, and biogeochemical processes is nonstationary and includes abrupt shifts driven by local-scale processes such as species introductions (e.g., Figure 11) and global-scale processes such as climate shifts (e.g., Figure 17). Empirical observations over the past half century have documented fast, large-amplitude changes in the world’s estuarine-coastal ecosystems that depart radically from Eugene P. Odum’s depiction of the natural evolution of biological communities and their habitats in unperturbed environments. These changes reflect today’s human domination of the Earth’s ecosystems as “most aspects of the structure and functioning of Earth’s ecosystems cannot be understood without accounting for the strong, often dominant influence of humanity” [*Vitousek et al.*, 1997, p. 494].

[77] Monitoring is essential for managing the human dimension of ecosystem dynamics because it detects environmental changes, provides insights into their underlying

TABLE A1. Sources of Data Used in This Review

Description	Source	Date Accessed
Census populations of Bay Area counties and incorporated cities	http://www.dof.ca.gov/research/demographic/state_census_data_center/historical_census_1850-2010/view.php	2012-01-12
Measured unimpaired runoff to the Sacramento and San Joaquin valleys	http://cdec.water.ca.gov/cgi-progs/iodir/wsihist	2011-10-28
Reservoir storage volume in California	[*CADWR*, 1993]	
Estimated unimpaired inflow to the Delta	[*CADWR*, 2007]	
Dayflow Program flow data	http://www.water.ca.gov/dayflow/output/Output.cfm	2011-02-01
IEP Environmental Monitoring Program discrete water quality data	http://www.water.ca.gov/bdma/meta/Discrete/data.cfm	2011-03-20
USGS Water Quality of San Francisco Bay discrete water quality data	http://sfbay.wr.usgs.gov/access/wqdata/query/index.html	2011-06-03
IEP Environmental Monitoring Program discrete benthic data	http://www.water.ca.gov/bdma/meta/benthic/data.cfm	2011-10-10
IEP Environmental Monitoring Program discrete zooplankton data	http://www.water.ca.gov/bdma/meta/zooplankton.cfm	2011-09-07
Chesapeake Bay nutrients	http://www.chesapeakebay.net/data/downloads/cbp_water_quality_database_1984_present	2012-02-17
Tomales Bay nutrients	http://lmer.marsci.uga.edu/tomales/	2011-12-29
Willapa Bay nutrients	http://www.ecy.wa.gov/programs/eap/mar_wat/mwm_intr.html	2012-01-27
SERL DO in South Bay	[*Harris et al.*, 1961; *McCarty et al.*, 1962]	
San Jose/Santa Clara Water Pollution Control Plant performance summary	Neal Van Keuren, City of San Jose, Environmental Services	
Palo Alto Regional Water Quality Control Plant metal loadings	Michelle Hornberger, USGS, personal communication, January 2012	
USGS Ecology and Contaminants Program, metals in Palo Alto clams	http://wwwrcamnl.wr.usgs.gov/tracel/	2012-02-09
PDO index	http://jisao.washington.edu/pdo/	2012-01-05
NPGO index	http://www.o3d.org/npgo/data/NPGO.txt	2012-01-05
Sea surface temperature, southeast Farallon Island	http://www.sccoos.org/query/	2012-01-05
Upwelling index	http://www.pfeg.noaa.gov/products/pfel/modeled/indices/upwelling/upwelling.html	2012-01-10
California Department of Fish and Game, San Francisco Bay Study	Kathy Hieb, DFG, personal communication, 24 January 2012 and earlier	

causes, can provide early warning signs of impending state shifts [*Carpenter et al.*, 2011; *Scheffer et al.*, 2009], prompts mitigation and adaptation policies (such as the Clean Water Act), and measures outcomes of those policies. Yet, commitments to monitoring programs are difficult to secure and sustain because their value accrues over the long-term and precise benefits cannot be prescribed in advance. The stakes are growing, however, as the cumulative effects of fast-paced change across all global ecosystems might be driving a trajectory toward a planetary state shift with large social and economic consequences [*Barnosky et al.*, 2012]. Given the scope and breathtaking pace of change occurring in the world's estuarine-coastal ecosystems (e.g., Figure 1), the imperative for monitoring data and their analysis has never been greater.

APPENDIX A

A1. Data Sources

[78] The water quality data used for analysis here were obtained primarily by the Interagency Ecological Program's (IEP) Environmental Monitoring Program (EMP) and the USGS Water Quality of San Francisco Bay Program. The IEP is a consortium of 10 member agencies cooperating on research in the Bay and Delta since 1970, and its EMP activities of primary interest here—discrete water quality, zooplankton, and fish monitoring—are operated by the California Department of Water Resources and Department of Fish and Game (DFG). The EMP samples water quality and zooplankton mostly in the Delta and Suisun Bay, while the USGS program samples water quality mostly in San Francisco Bay (including Suisun Bay). The California DFG San Francisco Bay Study of fish and shellfish includes all of the Bay and much of the Delta. Many other types and sources of data are also used, all of which are summarized in Table A1.

A2. Data Analysis

[79] Except in the cases where individual observations were called for, we binned water quality data for each variable and station by month using the mean to form a collection of monthly time series. For zooplankton and *Corbula amurensis*, we imputed missing monthly data using the long-term mean for the month. Annual averages for zooplankton used March through November data only. The 1987 *Corbula* average is based on May through December values only.

[80] Fish counts for all five indicated species for all marine subembayments (South, Central, and San Pablo bays) were summed for each tow, converted to catch per unit effort (CPUE) = 1000 × count/tow area, and then averaged over each year (February to October). A similar procedure was used for three crab and two shrimp species, except that CPUE = count/5 min tow.

[81] When estimating trends in time series, we used a robust measure sometimes known as the Theil-Sen slope, unless otherwise noted. This is simply the median slope of the lines joining all pairs of points in the series. We assessed the significance of these trends using the Mann-Kendall test. Tests were conducted only if at least 50% of the total possible number of values in the beginning and ending fifths of the record were present [*Helsel and Hirsch*, 2002].

[82] In the graphs, boxplots are traditional boxplots, i.e., the line within the box represents the median, the boxes extend from the first through third quartiles, and the vertical lines extend to all points within 1.5 times the interquartile distance (box height). Smoothing lines in graphs are local polynomial regressions, in particular, loess smooths with span equaling 0.75 and degree equaling 2 [*Helsel and Hirsch*, 2002].

[83] Restricted cubic, or natural, splines were used as transforms for predictors in regression relationships. To minimize the number of parameter estimates, a restricted cubic spline with only three knots was used, requiring only two parameters. The knot positions—at the 0.1, 0.5, and 0.9 quantiles—were chosen based on general recommendations from simulation studies and were not tailored in any way for these particular data sets. We conducted analyses of variance to determine if the nonlinear part of each predictor transform actually improved the regression model or if it could be replaced by a simple linear term [*Harrell*, 2001].

[84] Multivariate regression model results are illustrated as partial residual plots, which show the relationship between a given independent variable and the response variable, while accounting for the other independent variables in the model. Partial residuals for any predictor are formed by omitting that term from the model [*Chambers*, 1992].

[85] We used the R language [*R Development Core Team*, 2012] for all calculations and graphs, including extensive use of the ggplot2 [*Wickham*, 2009] and wq (A. D. Jassby and J. E. Cloern, wq: Exploring water quality monitoring data, R package version 0.3–6, 2012, available at http://cran.r-project.org/web/packages/wq/index.html) packages.

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