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Lower South Bay Nutrient Synthesis

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Executive Summary

Lower South Bay (LSB), a shallow subembayment of San Francisco Bay (SFB), is situated south of the Dumbarton Bridge, and is surrounded by, and interconnected with, a network of sloughs, marshes, and former salt ponds undergoing restoration (Figure ES.1). LSB receives 120 million gallons per day of treated wastewater effluent from three publicly owned treatment works (POTWs) that service San Jose and the densely populated surrounding region. During the dry season, when flows from creeks and streams are at their minimum, POTW effluent comprises the majority of freshwater flow to Lower South Bay. Although LSB has a large tidal prism, it experiences limited net exchange with the surrounding Bay, because much of the water that leaves on ebb tides returns during the subsequent flood tides. The limited exchange leads to distinctly different biogeochemical conditions in LSB compared to other SFB subembayments, including LSB having the highest nutrient concentrations and highest phytoplankton biomass.

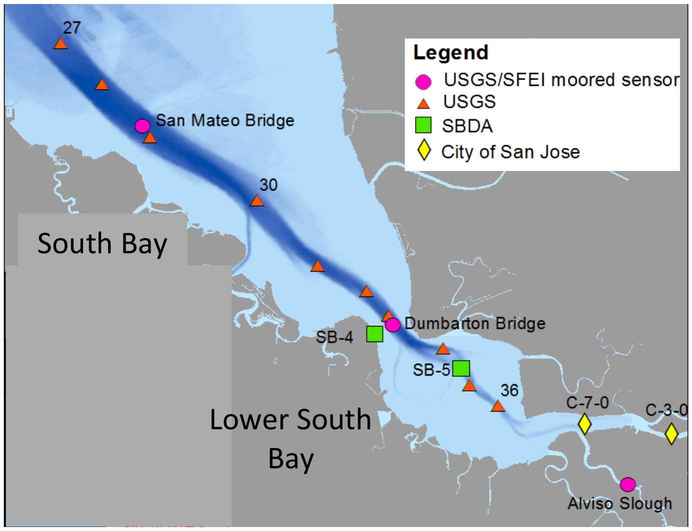


Figure ES.1 Location and bathymetry of Lower South Bay and the southern half of South Bay region, with locations of water quality monitoring sites used in this report. Benthos sampling and fish trawl sites are not shown here, but are described in Sections 4 and 7, respectively

This report was prepared as one in a series of reports that explore the current state of knowledge on nutrient-related issues in San Francisco Bay, in support of the San Francisco Bay Nutrient Management Strategy (NMS). The report's main goals are:

- Synthesize data and observations in LSB related to nutrient inputs and cycling in LSB, major indicators of ecosystem response to nutrients, and the range of physical, chemical, and biological factors that regulate ecosystem response;
- Identify major data or conceptual gaps across.

The report is organized as follows:

- Section 2: Nutrients: loads, seasonal, spatial, and ambient concentrations, and evidence for the importance of in situ biogeochemical transformations (Section 2)
- Section 3: Suspended sediment concentrations, and their influence on light levels
- Section 4: Abundance of benthic grazers
- Section 5: Phytoplankton biomass
- Section 6: Dissolved oxygen concentrations
- Section 7: Fish abundance
- Section 8: Priority science questions, and proposed activities to address those questions.

Nutrient Loads: Treated wastewater effluent is the primary source of nutrient loads to LSB, and loads are fairly constant throughout the year. All three POTWs nitrify before discharging; thus, throughout most of the year, DIN inputs occur primarily in the form of NO_3^-. POTW nutrient loads to LSB have decreased substantially over the past 30 years (Figure ES.2). From 1980 to 2011, annual average POTW DIN loads decreased by 35% and o-PO_4 loads have decreased by 70%. The DIN and o-PO_4 loads decreased over a time period when the population served by these POTWs increased 40%. The DIN load decreases resulted primarily from changes in treatment technology at SJSC. POTW o-PO_4 loads to LSB also decreased between 1980 and 2011, due to both P removal from household detergents and changes in SJSC's treatment technology.

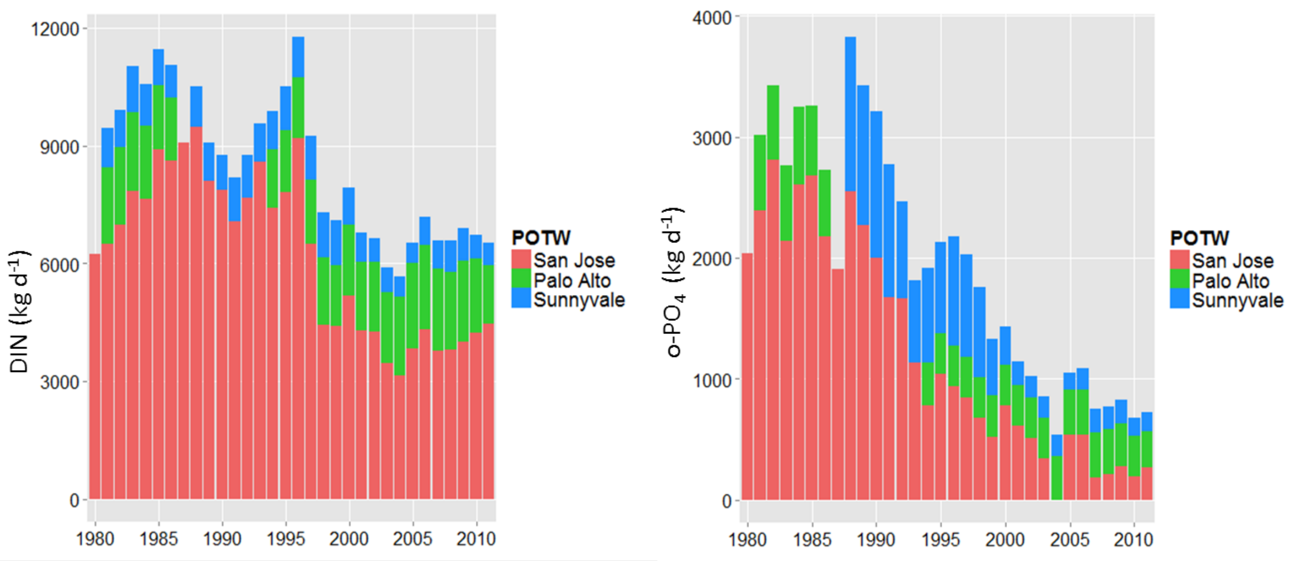


Figure ES.2 Annual average DIN (a) and o-PO_4 (b) loads from each of the three wastewater discharges to Lower South Bay, 1980-present. Sunnyvale did not measure o-PO_4 prior to 2012, so estimates of o-PO_4 were made using total phosphorous (TP) measurements and average % TP as o-PO_4 data from 2012. Note: Missing values for DIN or o-PO_4 for a POTW during a given year indicates that monitoring for that analyte did not occur, not that load is 0.

Changes in LSB nutrient concentrations: The substantial decreases in nutrient loads to LSB have resulted in proportional decreases in ambient concentrations of DIN and o-PO4 concentrations (Figure ES.3). Median DIN and o-PO4 concentrations have decreased by approximately 40% and 70%, respectively.

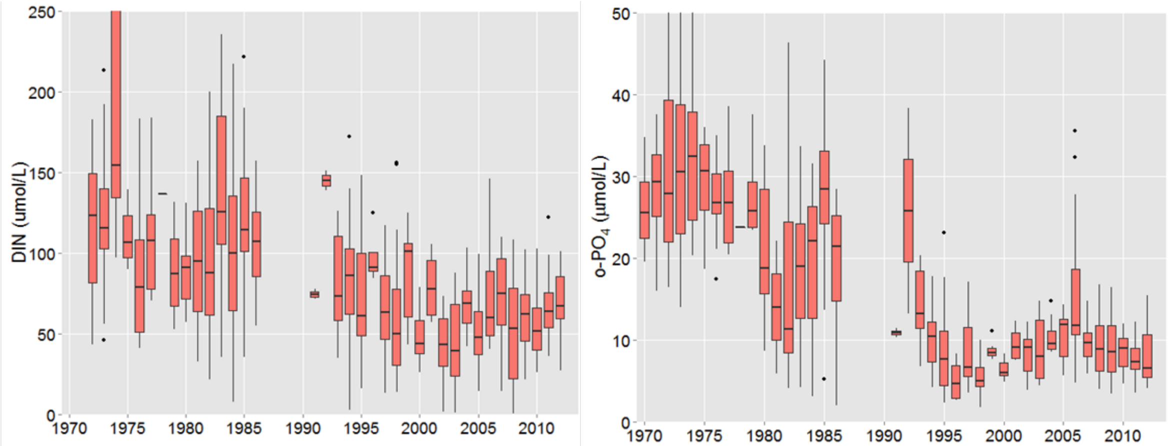
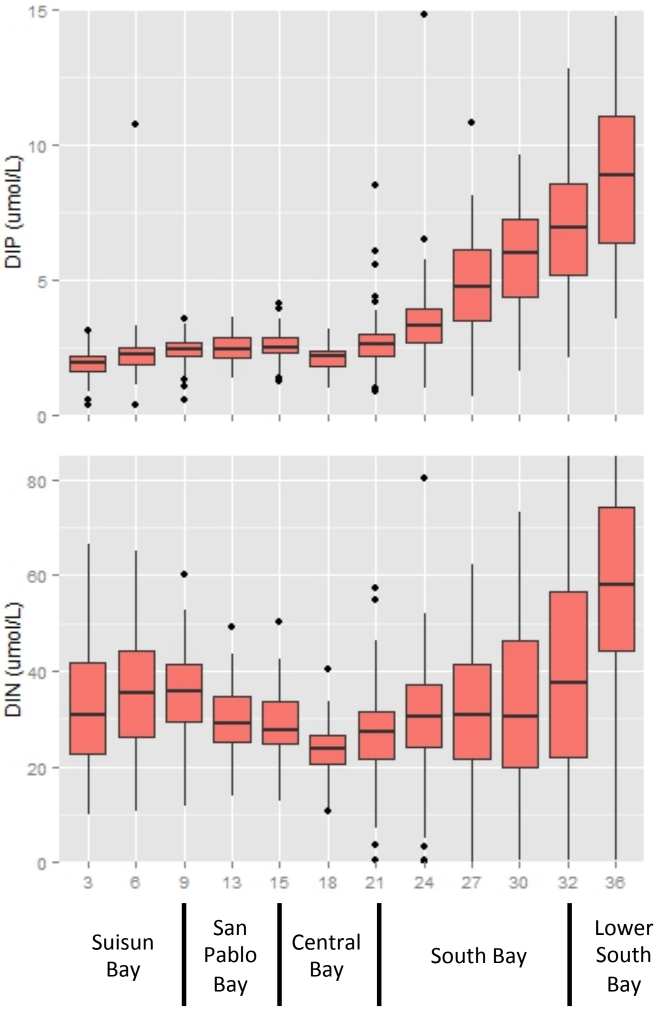


Figure ES.3 Long-term changes in nutrient concentrations in Lower South Bay (USGS stations 34-36 and SBDA station SB-5). The horizontal line represents the median, and the box extends to the 25th and 75th percentiles. The whiskers extend to 1.5*IQR from the 25th and 75th percentiles.



Ambient N and P in LSB compared to other areas of SFB: Although nutrient loads to LSB have been reduced, ambient DIN and o-PO_4 concentrations in LSB remain up to 2.5 and 4 times higher, respectively, than other SFB subembayments (Figure ES.4), and also substantially greater than those observed in many other estuaries (Figure 2.7, Cloern and Jassby 2012). LSB's elevated nutrient concentrations result from a combination of several factors, including the substantial direct loads LSB receives and its relatively small volume and slow flushing rate.

Figure ES.4 Current (2005-2012) nutrient concentrations along the San Francisco Bay's longitudinal axis in the deep channel at selected USGS stations extending from Suisun Bay (station 3) to Lower South Bay (station 36). Year-round data from this period are presented, and seasonal variability in concentrations contributes to the wide distribution at some stations.

N and P spatial heterogeneity within LSB

DIN and o-PO_4 concentrations increased along a southerly transect moving from the Dumbarton Bridge towards the confluence of Coyote Creek and Artesian Slough (Figure ES.5), near the SJSC effluent outfall. Both DIN and DIP concentrations increased by a factor ~4 between the open water station s36 and C-7-0, and DIN increased by another factor of ~1.5 between C-7-0 and C-3-0 (see Figure ES.1 for locations). That spatial variation of nutrient concentrations in LSB is likely due to a combination of factors, including: proximity to source and dilution, uptake by algae, and loss through denitrification. Most of our current knowledge about nutrient concentrations in LSB is based on measurements from the open-Bay, with considerably less information available for slough, creek, and salt pond environments.

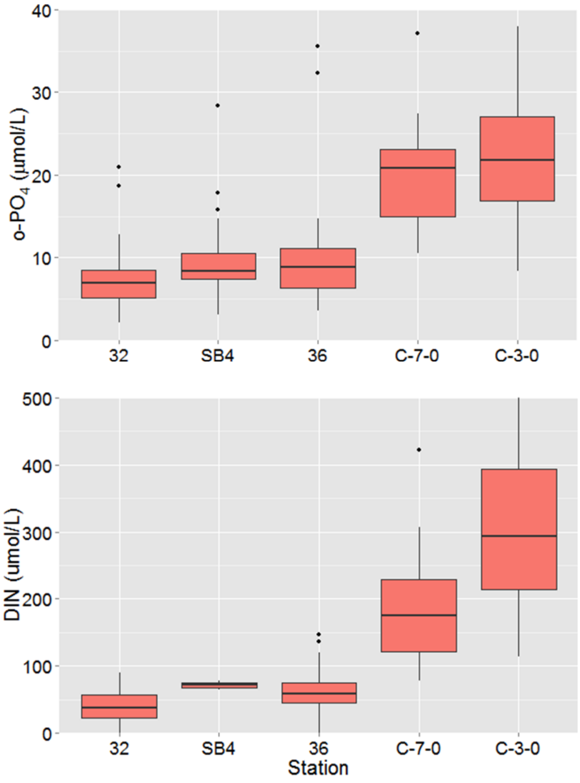


Figure ES.5 Current (2005-2012) nutrient concentrations in Lower South Bay. Stations 32, 34 and 36 are USGS stations in the open Bay, and stations C-7-0 and C-3-0 are monitored by SJSC and are located in Coyote Creek (see Figure ES.1).

N and P seasonal variability and transformations: Strong seasonal variations in nitrate, ammonium and o-PO$_4$ concentrations indicate that in situ biogeochemical processes play an important role regulating ambient concentrations (Figure ES.6). Important processes likely include remineralization of organic matter (liberating NH4 and o-PO4), N and P flux from sediments, nitrification in the water column or sediments, denitrification in sediments, and assimilation of N and P by phytoplankton. A simplified mass balance for DIN during summer months suggests that ~60% of the DIN that enters LSB is lost, either by assimilation or denitrification. While more advanced biogeochemical modeling will undoubtedly be needed to accurately predict nutrient fate, this estimate suggests that DIN losses can be substantial. There is currently little empirical data available for LSB related to nutrient transformation rates, and such data will likely be needed to calibrate models.

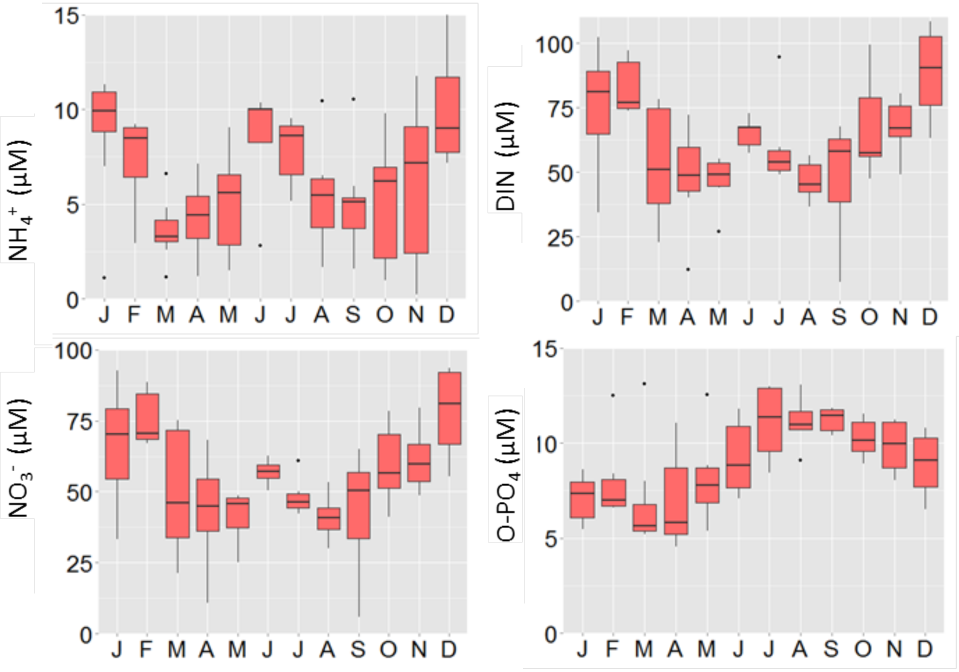


Figure ES.6 Seasonal variability in nutrient concentrations in South Bay. Data: USGS

Phytoplankton biomass

LSB and southern South Bay experience the highest phytoplankton biomass levels of deep subtidal areas throughout SFB. Chlorophyll-a concentrations (chl-a; measure of phytoplankton biomass) tend to peak in March-April, with median peak concentrations of 10-15 µg/L (Figure ES.7). Past investigations in LSB and South Bay have observed that phytoplankton growth is limited by light availability, and that blooms commonly occur during short-lived periods (days to weeks) when water column is vertically-stratified. Stratification alleviates light-limitation by allowing phytoplankton to reside in light-rich surface layers. Available data indicate that, in the open Bay areas of LSB and South Bay, phytoplankton blooms are seldom limited by nutrient concentrations, although short-duration (days-weeks) drawdown to potentially rate-limiting concentrations may occur during some blooms.

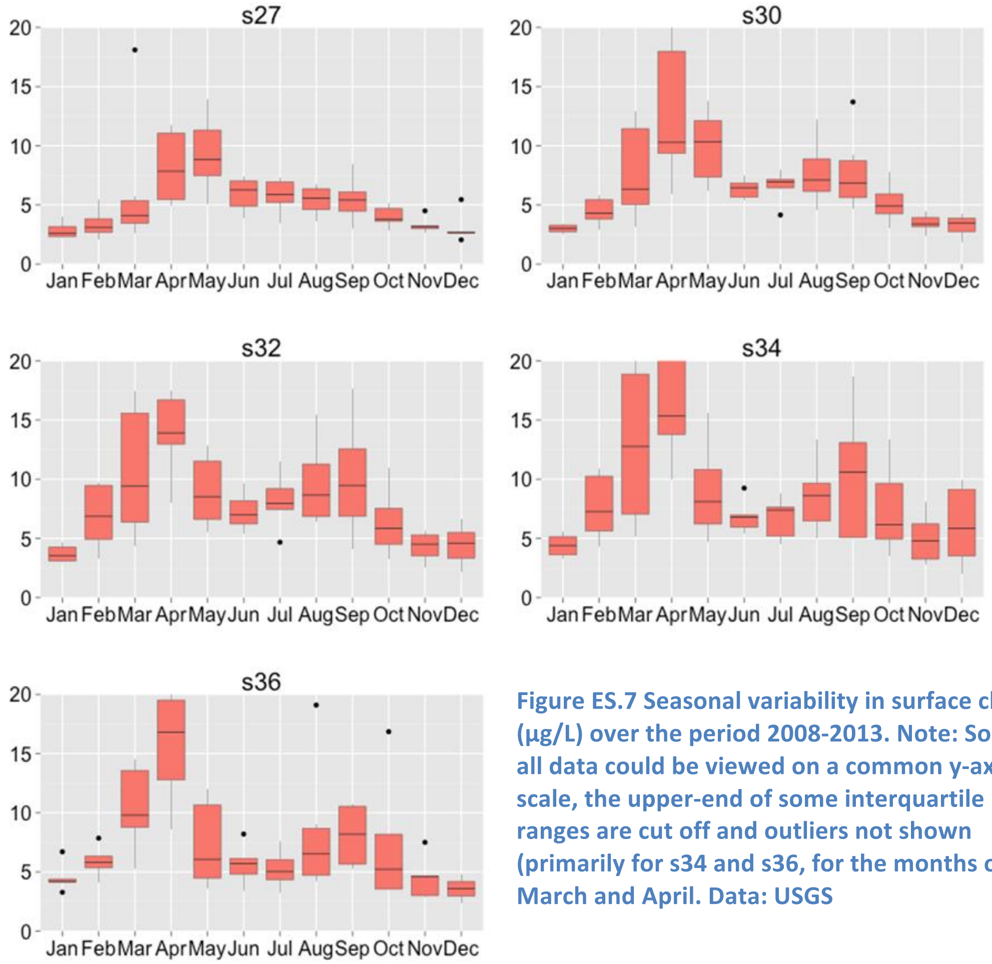


Figure ES.7 Seasonal variability in surface chl-a (µg/L) over the period 2008-2013. Note: So that all data could be viewed on a common y-axis scale, the upper-end of some interquartile ranges are cut off and outliers not shown (primarily for s34 and s36, for the months of March and April. Data: USGS

Substantial interannual variability, and potential for very large blooms: The magnitude of spring blooms (area, concentration) also vary substantially year to year. Extremely large winter/spring blooms have been observed during some years, with chl-a concentrations in the range of 10s-100 µg/L over large areas and lasting for periods of weeks (Figure ES.8).

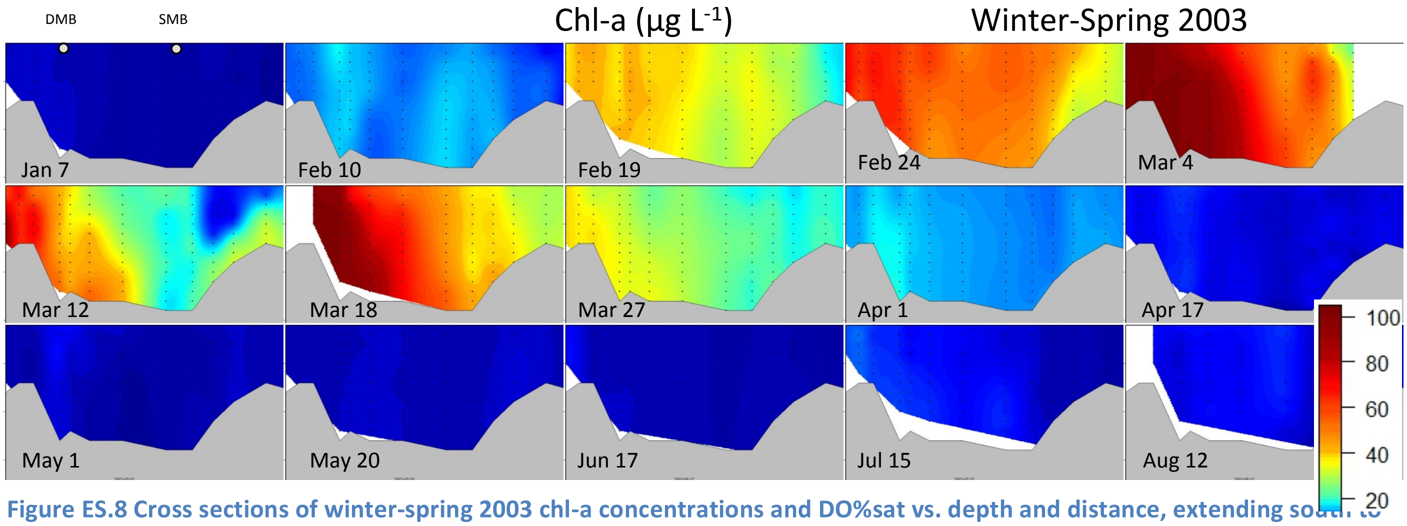


Figure ES.8 Cross sections of winter-spring 2003 chl-a concentrations and DO%sat vs. depth and distance, extending south to north station from s36 in LSB to s27 in South Bay (near the San Bruno Shoal). Grey area indicates approximate bottom elevation. Black dots indicate locations where chl-a and DO was measured on each date. DMB and SMB indicate approximate locations of the Dumbarton and San Mateo Bridges, respectively. Data: USGS. http://sfbay.wr.usgs.gov/access/wqdata/

Biomass increase and plateau:

Cloern et al. (2007, 2010) brought together 3 decades of phytoplankton biomass data for South Bay and observed that, between the late 1990s and 2005, late-summer/fall biomass concentrations had increased 3-fold. To further examine trends in phytoplankton biomass in South Bay and LSB, we extended the time series of Cloern et al. (2007) through 2013. Over the period of 2005-2010, biomass concentrations appear to have plateaued, establishing a new median fall concentration of 5-6 µg/L (Figure ES.9A). Biomass values for 2012-2013 were lower than 2005-2010, but still substantially greater than biomass levels prior to 1999. The 5+ years of relatively flat biomass concentrations from 2005-2010 may represent a new and more sensitive biological response level for South Bay to nutrients, even if that new plateau only lasts for a short period of time (several years) before shifting to another state. It may be tempting to speculate that the lower concentrations of 2012-2013 signal a return to lower sensitivity. However, 2012 and 2013 are only two years in a system that exhibits high interannual variability in its biological response to nutrients. Phytoplankton response at sites more closely associated with LSB were also examined (Figure ES.9B)

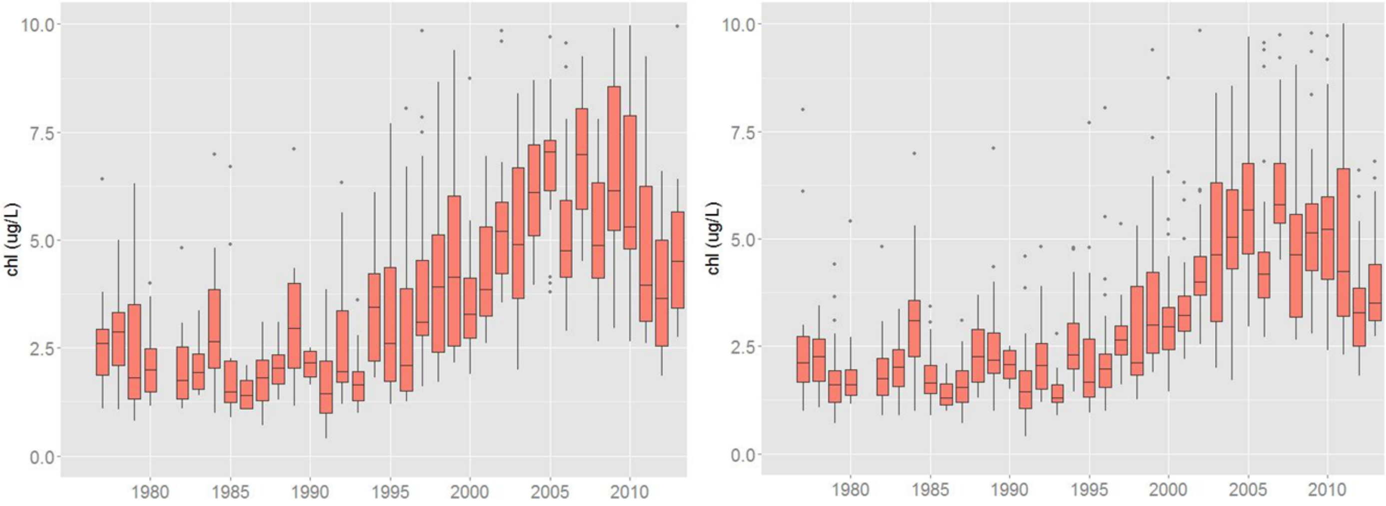
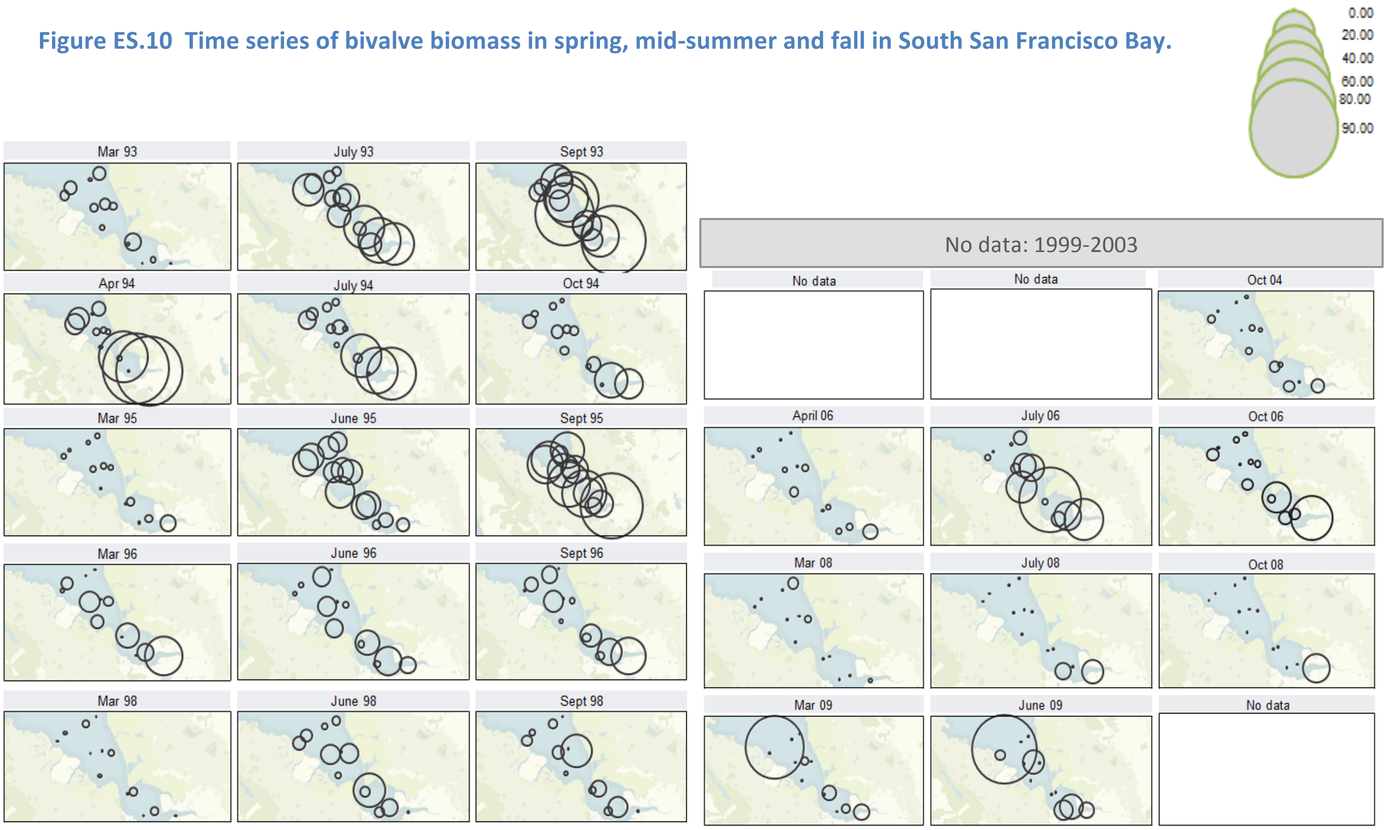


Figure ES.9 A. Phytoplankton biomass for the months Aug-Dec, using the same stations as Cloern et al. (2007) (limited to South Bay) and extending the time series through 2013. B. Phytoplankton biomass for the months Aug-Dec, similar to Figure 5.12 A but using stations 27, 30, 32, 34, 36. Data: USGS

Possible explanations for increased biomass

The phytoplankton biomass increase occurred during a period of time when nutrient loads to the system either remained constant (in some parts of South Bay) or decreased (in Lower South Bay). Changes in nutrients could therefore not explain the phytoplankton biomass increases. Cloern et al. (2007, 2010) argued that the phytoplankton biomass increase resulted from a loss of benthic bivalves that had strongly regulated phytoplankton biomass prior to the mid-1990s by filtering the water column. Two decades of compiled data from South Bay and LSB are strongly suggestive of a substantial decrease in benthic grazer biomass after 1998 (Figure ES.10). However, there remain large gaps in the data post-1998. In addition, the potential impact of this loss on phytoplankton biomass still needs to be quantified through modeling

Figure ES.10 Time series of bivalve biomass in spring, mid-summer and fall in South San Francisco Bay.



Analysis of a 20-year high-frequency SSC record at the Dumbarton Bridge indicates that the SSC decreased by 40% over that time period (Figure ES.11). That decrease began abruptly in the mid- to late-1990s, and appears to have plateaued at a new level by the early 2000s. Phytoplankton growth is considered to be light-limited in deep subtidal areas of LSB and South Bay due to high suspended sediment concentrations (Cloern, 1995). LSB's photic zone - the depth at which light levels are 1% of incident light – is a relatively thin layer of the water column, typically only 1-2 m (Cloern et al., 1985). The 40% decrease in SSC translates into roughly a 40% increase in photic depth, which in turn translates to a 40% increase in phytoplankton potential growth rates.

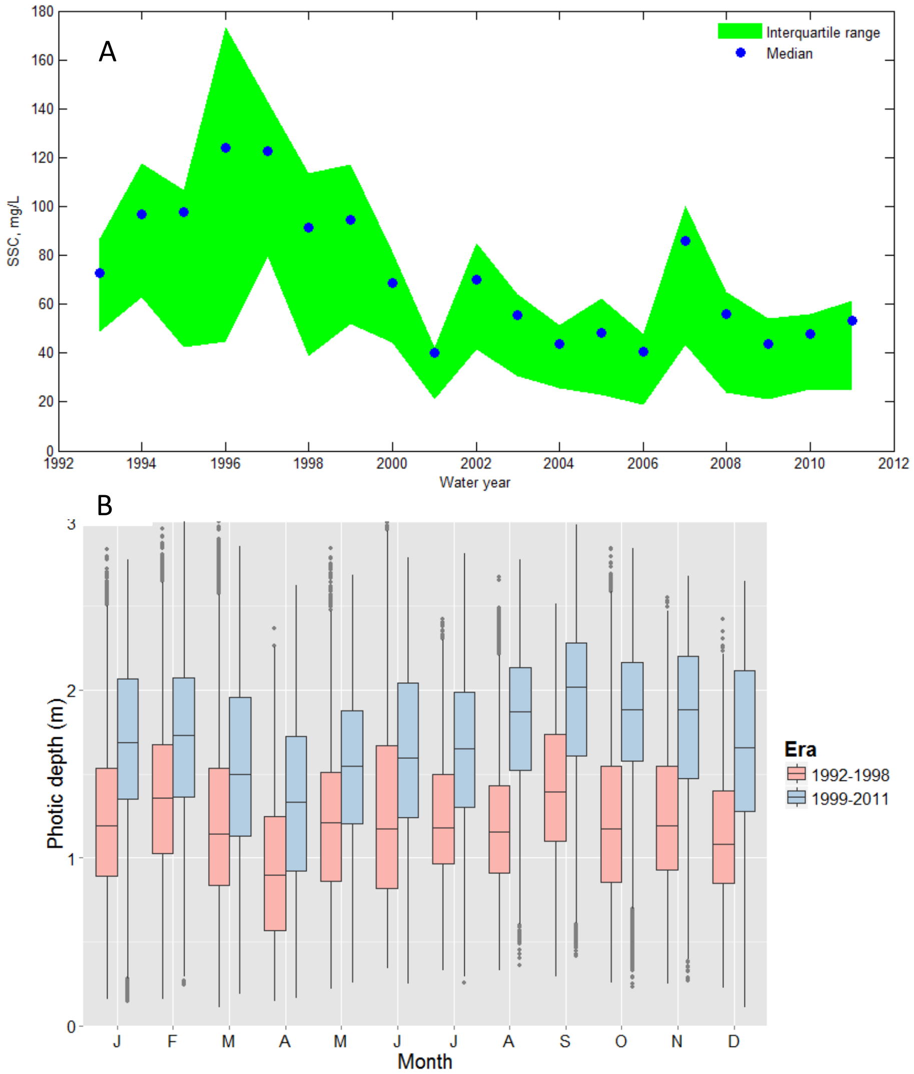


Figure ES.11 A. Annual suspended sediment concentrations based on high-frequency (15-min) records at Dumbarton Bridge (See Section 3). B. Photic depth calculated from monthly-binned high-frequency data from the Dumbarton Bridge, divided into two eras: 1992-1998 and 1999-2011.

To a first approximation, the loss of benthic grazers and increased light levels have the potential to impact phytoplankton biomass to a similar degree (Table ES.1). Those estimates are based on very basic steady-state mass balance assumptions, and are thus highly uncertain. More advanced modeling is needed explore the relative effects and to inform future monitoring

	pre-1999 (d^{-1})	post-1999 (d^{-1})
k_{graze}	1-1.5	0-0.2
k_{grow}	0.5	0.75
k_{settle}	0.2-0.3	
k_{flush}	0.06	

Table ES.1 Relative importance of potential drivers of phytoplankton biomass, expressed as estimated first order rate constants.

Additional sources of phytoplankton biomass? High frequency chl-a measurements at Dumbarton Bridge and at select slough sites suggest that margin habitats may be a major and previously unaccounted for source of organic matter, based on systematic elevated chl-a levels on the outgoing tide (Figure ES.12). There are several potential sources for this fresh organic matter, including marshes, resuspended benthic algae, and exchange with highly-productive restored salt ponds. Restoration efforts over the past 5-10 years have now connected an area of restored salt ponds to the Bay that is equal to the area of the open Bay. Because of the shallower depth and higher light levels in restored salt ponds, they could serve as important areas for primary production. The influences of restored salt ponds on nutrient cycling, primary production, and dissolved oxygen are major information gaps that need further examination.

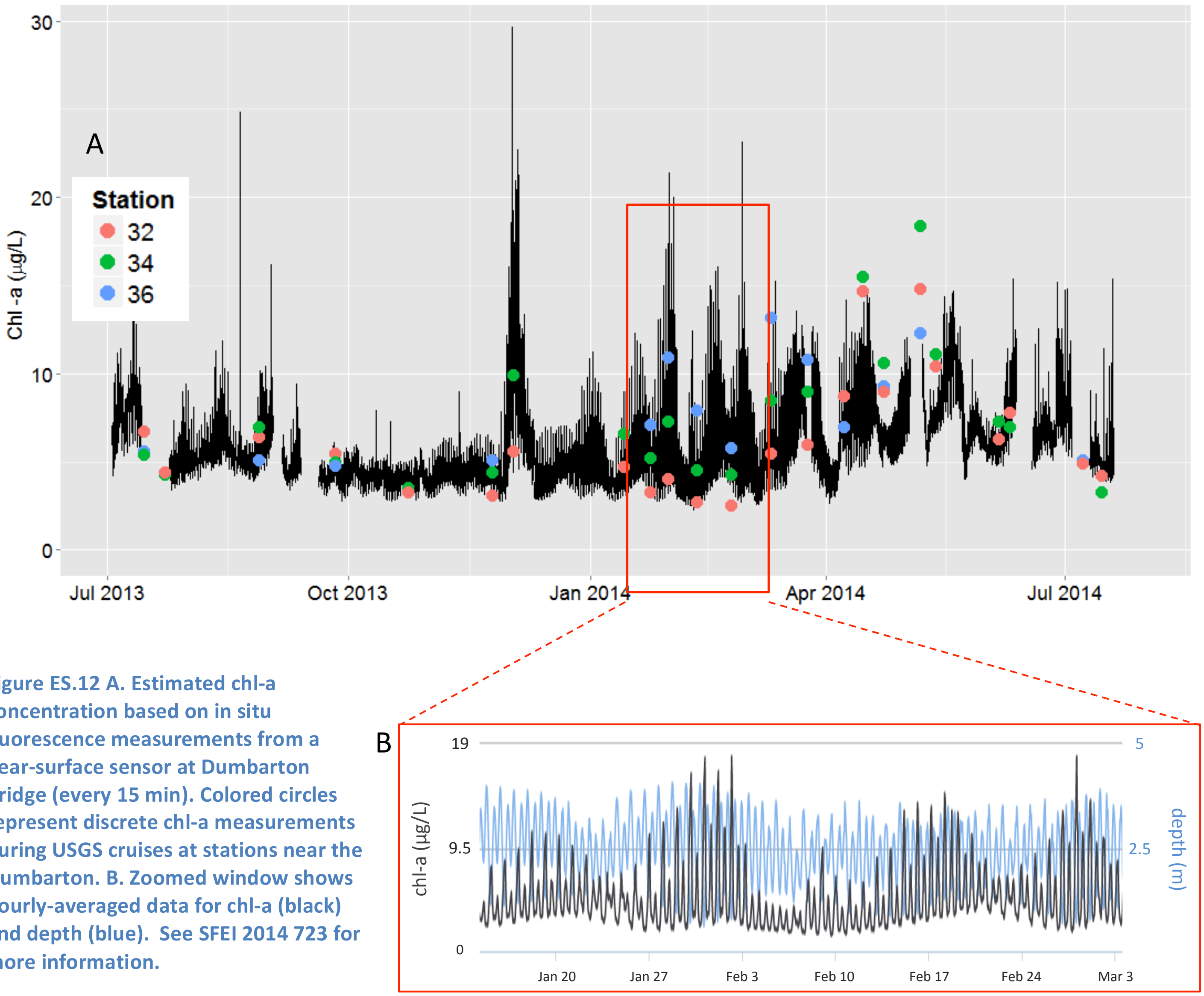


Figure ES.12 A. Estimated chl-a concentration based on in situ fluorescence measurements from a near-surface sensor at Dumbarton Bridge (every 15 min). Colored circles represent discrete chl-a measurements during USGS cruises at stations near the Dumbarton. B. Zoomed window shows hourly-averaged data for chl-a (black) and depth (blue). See SFEI 2014 723 for more information.

Dissolved Oxygen: Deep subtidal
Long-term ship-based data indicate that dissolved oxygen concentrations at deep subtidal sites of LSB and South Bay are above 80% saturation and 5 mg/L, the Basin Plan standards, during the vast majority of biweekly-to-monthly monitoring events over the past ~20 years (Figure ES.13).

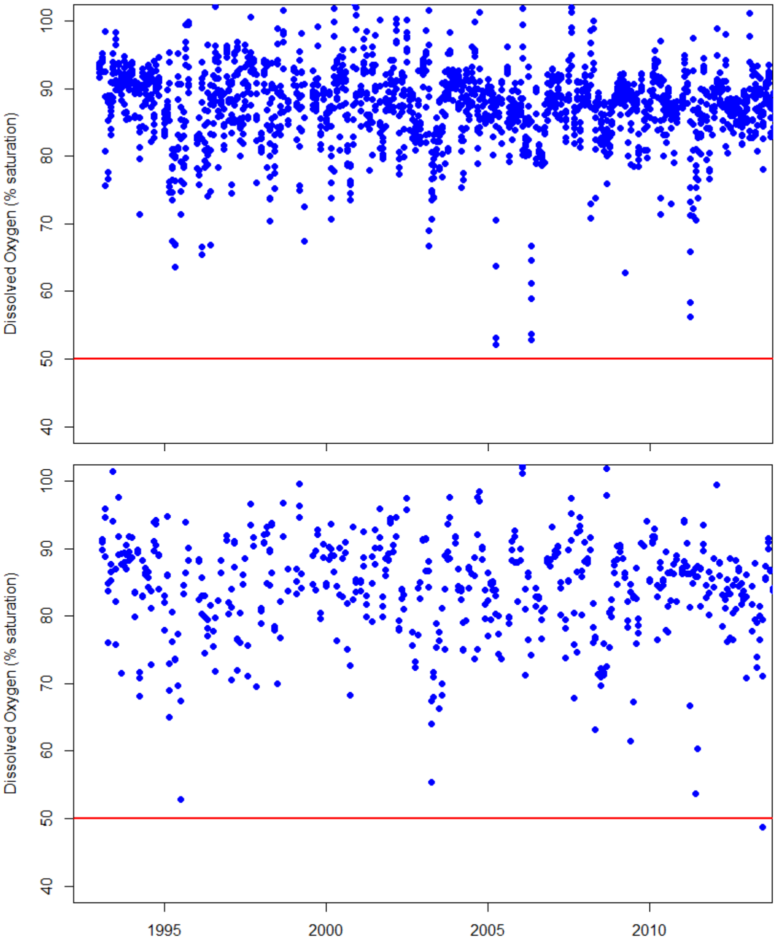
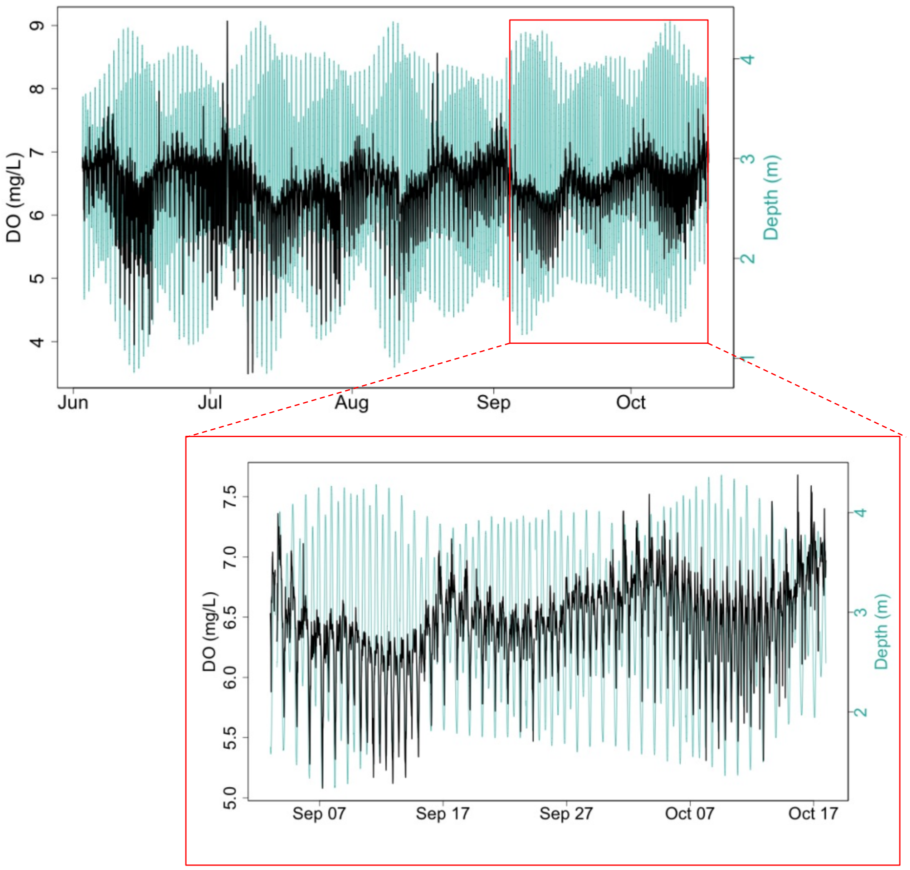


Figure ES.13 A Minimum DO measured during each vertical profile during weekly to biweekly sampling at USGS stations in A. South Bay (Bay Bridge to Dumbarton Bridge; s21, s22, s24, s25, s27, s29, s30, s32) and B. Lower South Bay (south of Dumbarton Bridge; s34, s35, s36)



Lower DO at low tide: Recent high-frequency DO measurements at Dumbarton Bridge show that DO decreased by 1-1.5 mg/L between low and high tide during late-spring, summer, and early fall(Figure ES.14). These observations suggest that there was a source of low DO water mixing into LSB during ebb tides.

Figure ES.14 Dissolved oxygen concentrations (black) and depth (blue) at Dumbarton near-surface sensor, June –October 2014. Note the two patterns with different periods: 1. semi-diurnal tides leading to local DO minima at low tide; 2. DO minima and maximum values decrease over spring tides, and increase increase again during neap tides.

Dissolved Oxygen: margin habitats

Compared to monitoring in SFB's deep subtidal areas, limited systematic monitoring has occurred in the sloughs and creeks that drain into Lower South Bay. Data from a site in Alviso Slough suggest that DO commonly dropped below 5 mg/L there (Figure ES.15A) and that varies strongly with diurnal and semi-monthly (i.e., spring/neap) tides (Figure ES.15B).

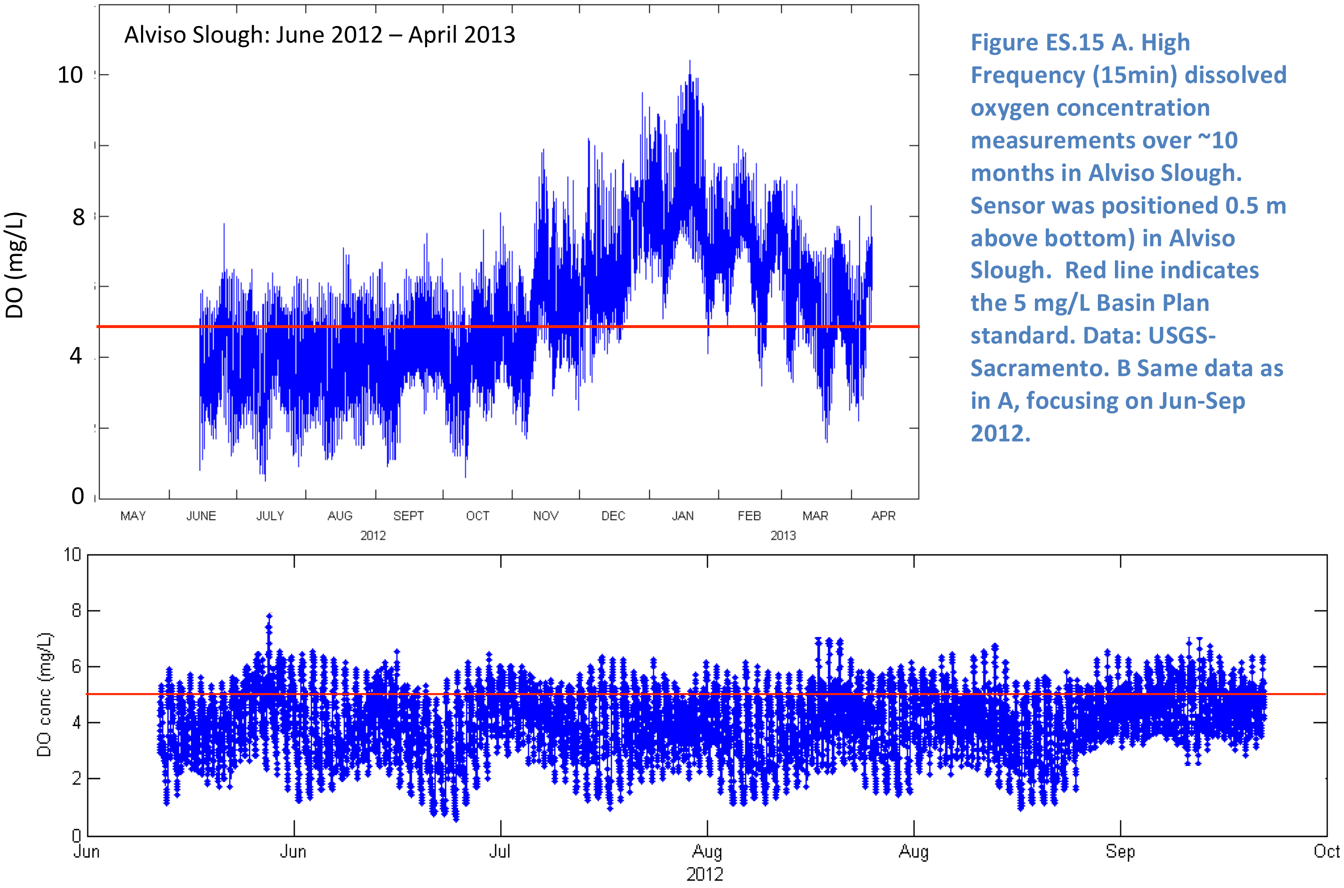


Figure ES.15 A. High Frequency (15min) dissolved oxygen concentration measurements over ~10 months in Alviso Slough. Sensor was positioned 0.5 m above bottom) in Alviso Slough. Red line indicates the 5 mg/L Basin Plan standard. Data: USGS-Sacramento. B Same data as in A, focusing on Jun-Sep 2012.

Factors regulating DO in sloughs and open Bay

Based on the slough and open-Bay observations, we developed the following conceptual model consisting of four linked mechanisms, intended as hypotheses that require further examination through additional field investigations and modeling.

1. Dissolved oxygen concentration decreases rapidly in slough bottom waters when the slough water column becomes vertically-stratified during neap tides. (Figure ES.15B) Neap tides have lower tidal velocities than spring tides. The lower velocities during neap tides have weaker mixing energy, and allow vertical salinity stratification to develop when less-dense fresher water meets more-dense saltier water near the site. DO is consumed during the mineralization of organic matter; since the bottom layer is cutoff from the atmosphere, DO concentration in that layer decreases over time. The water column experiences periodic vertical mixing as tidal energy increases, causing DO concentrations to increase. On shorter time scales (once or twice per day) DO varies due to due to a combination of shorter-lived periods of stratification/mixing on ebb and flood tides, respectively, and longitudinal movement of water having different chemistry.

2. Less flushing of the slough occurs during neap tides than spring tides, i.e., less exchange with higher-DO open Bay water. As a result, slough DO concentrations (minimum and mean) tend to decrease over time during neap tides. (Figure ES.15B) During spring tides, a larger portion of 'old' slough water is flushed and replaced by higher DO water from the open Bay, whereas, during neap tides, there is less net flushing and gradual consumption of DO and decreasing average DO concentrations.

3. A substantial source of labile (fresh) organic matter enters the sloughs, leading to sufficient oxygen demand to cause observed DO conditions. A chl-a sensor deployed in Alviso Slough beginning in 2013 indicated that chl-a levels are substantially higher than those in the open Bay, consistent with a local source of fresh organic matter. A portion of this organic matter accumulates in slough sediments, from where it exerts on-going oxygen demand on DO in the water column. Hypothesized sources include exchange with restored salt ponds, resuspension of benthic algae, or dislodging of periphyton from marsh plants. Production of new algal biomass in the slough water column has not yet been strictly ruled out, but is likely small because of low light levels.

4. *Water quality differs substantially between open Bay and margin habitats of LSB.* Margin waters containing lower-DO are drawn into the open-Bay and past the Dumbarton sensors during ebb tides due to LSB's large tidal prism (Figures ES.16 and ES.15B). The net exchange of low-DO water (or oxygen demand) from sloughs to the open-Bay appears to be non-trivial, as evidenced by decreases in maximum DO levels at Dumbarton over the course of spring tides, when the greatest flushing of sloughs is expected (mechanism #2).

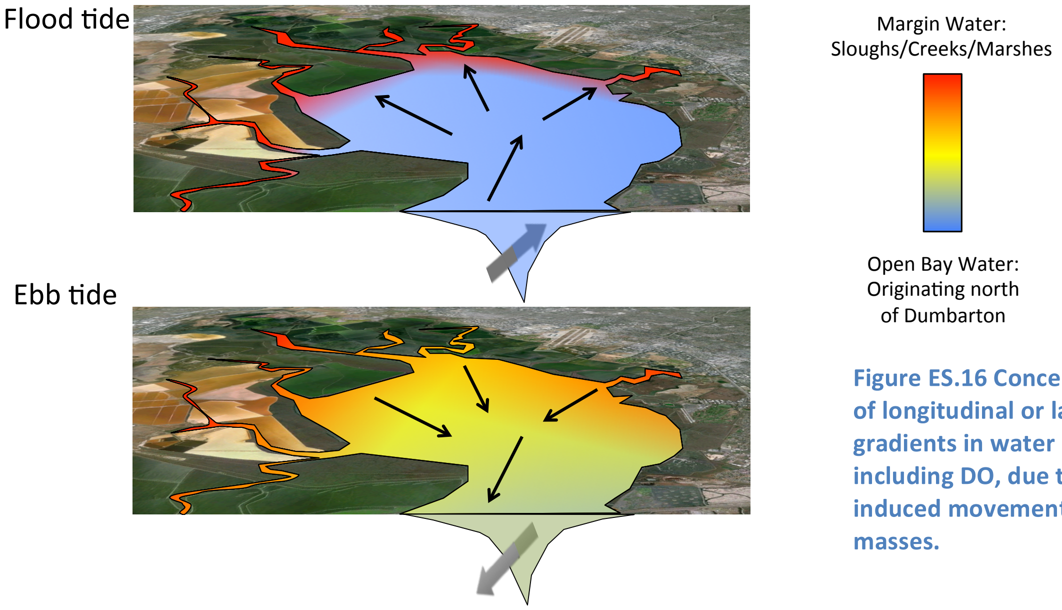


Figure ES.16 Conceptualization of longitudinal or lateral gradients in water quality, including DO, due to tidally-induced movement of water masses.

Fishes of LSB and South Bay, and fish-nutrient conceptual model

A set of conceptual models was developed for LSB and South Bay describing habitat-types, fish assemblages, fish-habitat associations, and potential effects of nutrients on fish, in particular as it pertains to dissolved oxygen. Four fish monitoring datasets, for South Bay and Lower South Bay, were also explored to identify species and abundance patterns in space and time.

The frequency, magnitude and duration of low-oxygen events can have profound effects on aquatic organisms (Diaz and Breitburg 2009). Organisms can exhibit responses to hypoxia - defined as DO < 2-3 mg/L (CENR 2010) - that can operate at many levels of biological organizations, from effects on molecular and biochemical pathways to individual behavior, to population demography to community dynamics and ecosystem structure and function (Figure ES.17).

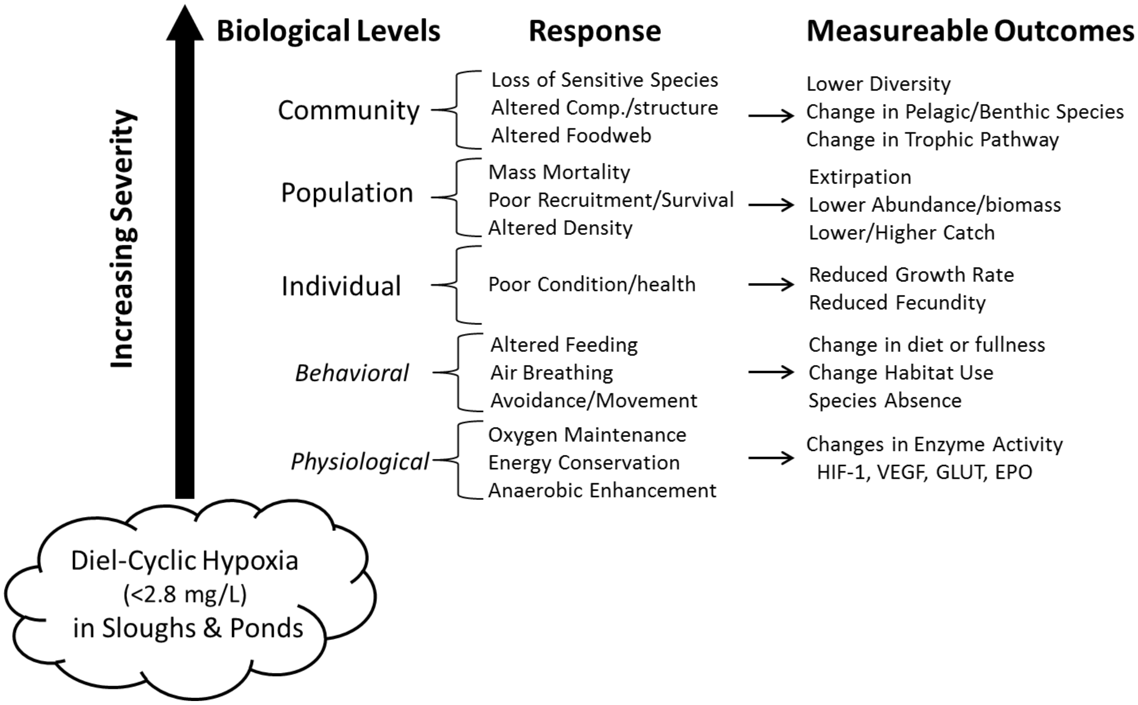


Figure ES.17 Conceptual model of hypoxia across a gradient of increasing severity and associated biological levels affected and their responses linked to measurable outcomes.

The tolerance of South Bay and LSB species to low DO is currently poorly known. The information regarding tolerance that is available exists only for non-native species that occur in other estuaries that experience frequent hypoxic conditions. Therefore, although habitat quality under high and very-low DO may be extrapolated with reasonable confidence from other species (good and bad, respectively), assessing the effects of low and intermediate DO on the specific fishes and macro-invertebrates that occur in South Bay and LSB is difficult given our limited understand of what drives population trends.

Although true DO tolerance for some species is currently unknown, the two figures below, which present conditions under which each species was captured in recent LSB trawls, provide an imperfect but still meaningful indirect measure of DO tolerance.

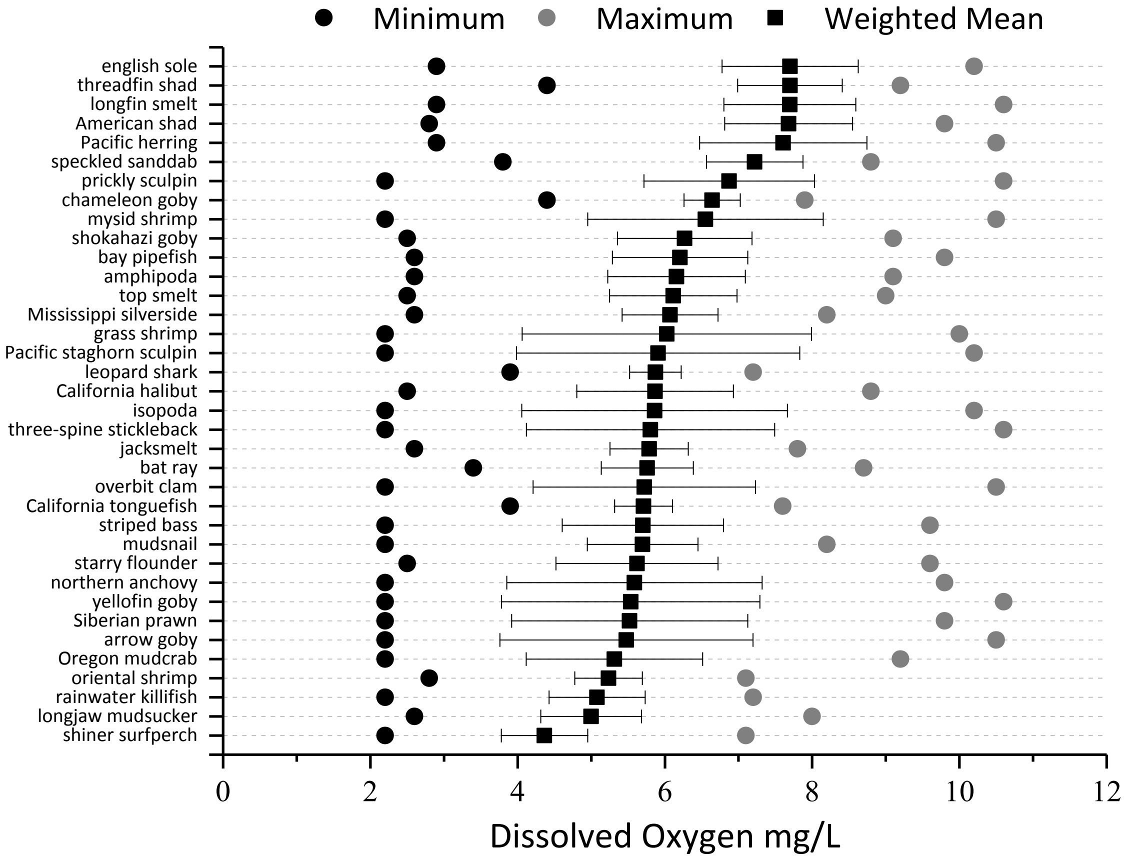


Figure ES.18 Mean dissolved oxygen concentration weighted by the frequency of occurrence for the most common fish and macro-invertebrate species found in South Bay marsh habitats.

Figure ES.18 uses the same underlying data as in Figure ES.17, but illustrates the distribution of trawls vs. DO concentration, and the catch per unit effort for each species. This presentation of the data offers some additional perspective on when species were and were not caught, and helps identify some data gaps. For example, while the weighted mean DO concentration for northern anchovy was ~5 mg/L (Figure ES.17), they were commonly caught at high abundance at DO <3-4 mg/L (Figure ES.18). Leopard shark, on the other hand, had a similar weighted mean DO as northern anchovy (~5.2 mg/L), but were evenly distributed between 4 and 7 mg/L but only rarely captured (n=10). Several species appear to have rather sharp thresholds, e.g., longfin smelt, threadfin shad, and English sole. However, the absence of longfin smelt and threadfin shad at lower DO may have as much or more to do with temperature tolerance (and co-occurrence of higher T and lower DO) than DO tolerance. Therefore, caution is needed when interpreting these data.

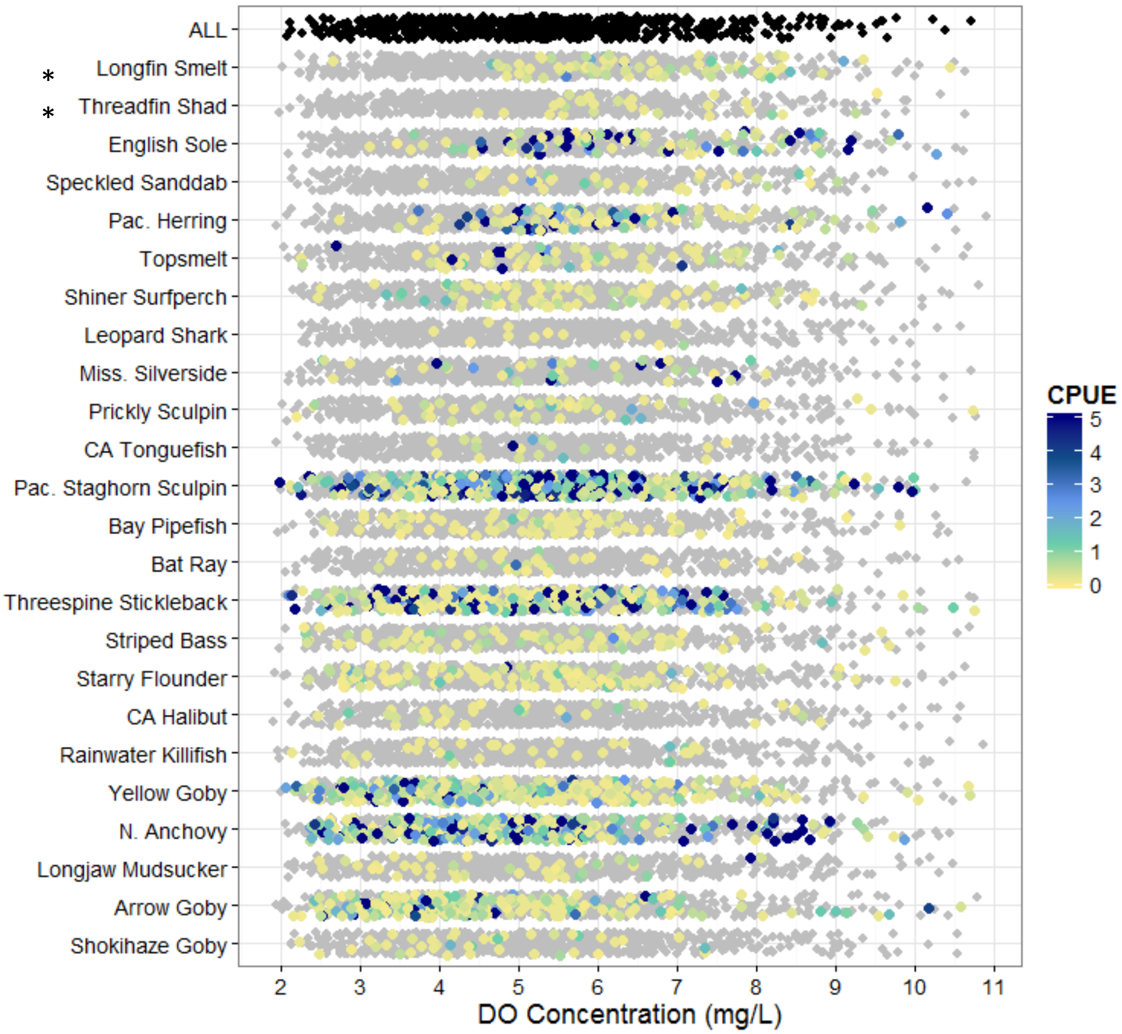


Figure ES.19 All trawls and catch per unit effort each trawl vs. DO concentration for each species. Black symbols in the first row display the distribution of trawls relative to DO. Trawls are repeated for each species (rows), with a grey symbol indicating that species was not caught; other colors indicate the species was caught, with the color denoting the catch per unit effort. * Indicates species whose absence may also be strongly driven by other factors such as seasonally-varying temperatures.

Priority Science Questions and Recommendations

Nutrients

1. How do nutrient concentrations and forms vary spatially and temporally, particularly in margin habitats where limited monitoring has occurred to date?
2. What are the dominant processes controlling nutrient fate in Lower South Bay, and how do their magnitudes vary spatially and temporally?
3. What nutrient loads can Lower South Bay assimilate without adverse impacts (e.g., with respect to chl-a, DO, or algal toxins)?

Phytoplankton biomass

1. What combination of factors regulate phytoplankton productivity and biomass, and how do the relative importance of those factors vary spatially and seasonally?
2. What combination of factors can explain the fall biomass increase in the late-1990s (e.g., loss of filter-feeding benthos, decreasing suspended sediments)?
3. How important are margin habitats as a source of organic matter to the open Bay, in particular restored salt ponds?
4. How important is benthic algae production to overall productivity and organic matter accumulation in Lower South Bay?
5. What effects would potential management actions have on biomass, DO, and algal toxins? E.g.,
    a. Decreased nutrient loads by 25%, 50%, 75%?
    b. Operation of restored salt ponds (e.g., including optimization for nutrient removal and beneficial habitat condition)
    c. Managed oyster or mussel reefs

Phytoplankton community composition

(not explored in this report because of data limitations, but a priority nonetheless)

1. What factors most strongly regulate phytoplankton community composition in Lower South Bay (e.g., light availability, temperature, nutrients, benthic grazing, exchange with salt ponds)?
2. To what extent do conditions in Lower South Bay select for either potentially harmful algae or algae that poorly support the food web?
3. What are source(s) of algal toxins in Lower South Bay?

Dissolved oxygen

1. What are the frequency, duration, spatial extent, and severity of low DO events in shallow margin habitats?
2. What causes the substantial tidal-variations in DO concentrations at Dumbarton Bridge?
    a. What are conditions like elsewhere (south of Dumbarton) during these times?
3. Are biota being adversely impacted by low(er) DO in the margins or open Bay?
4. What factors most strongly regulate DO in sloughs and creeks, and what data collection is needed to best predict DO condition?
    a. Organic matter source(s)
    b. Role of anthropogenic nutrients
    c. Physics: stratification, salt pond exchange, slough-open Bay exchange

Fish and benthos

1. What are the DO preferences or tolerances of key fish species that are observed, or expected, in Lower South Bay?
2. How do fish populations and diversity respond to spatial and temporal variability in DO?
3. Do current conditions support or adversely impact benthos abundance or assemblage?

Effects of salt pond restoration

1. What effects are salt pond restoration activities having on nutrient, carbon and DO budgets in the margins? The open Bay?
2. What effects are salt pond restoration activities having on habitat conditions in Lower South Bay?
3. Are restored salt ponds a substantial source of harmful algal species and algal toxins?

Future scenarios

1. What levels of phytoplankton production and biomass are plausible under future scenarios in Lower South Bay? Have we reached a new plateau or will concentrations rise further?
    a. How will controls on phytoplankton biomass (i.e. light availability, benthic grazing) change in the future?
2. What would be protective nutrient levels in terms of biomass, DO, and phytoplankton assemblage or toxins?

Recommendations

The recommendations below emerged from considering current data or conceptual gaps. Since some of these investigations would be resource- and time-intensive undertakings, some level of prioritization is still needed.

R 1: Systematically investigate DO in the margins

- High frequency DO monitoring and ancillary data
- Evaluate relative importance of mechanisms that control DO through data interpretation and modeling
- Gather data to use in model calibration/validation

R 2: Develop improved quantitative understanding of controls of phytoplankton biomass, i.e. grazers and suspended sediment

- Determine whether additional monitoring and field investigations are needed, specifically benthos surveys and light levels.
- Modeling and sensitivity analysis

R 3: Gather high-spatial resolution data through biogeochemical mapping

- Characterize spatial and temporal heterogeneity
- Assess condition across the region – extent and severity of potentially problematic events
- Gather data for model calibration/validation

R 4: Conduct mechanistic field investigations to quantify important processes related to physical processes, nutrient cycling and phytoplankton and benthic algae production (slough ←→ open Bay, salt pond ←→ slough, stratification in open Bay and sloughs)

R 5: Develop and apply a coupled hydrodynamic and biogeochemical model for Lower South Bay, including sloughs and margins

- Evaluate mechanisms that control phytoplankton biomass and DO concentrations through sensitivity analysis
- Examine the role of anthropogenic nutrients and quantify nutrient fate
- Forecast ecosystem response under potential future conditions, including changing environmental factors (sediment concentrations, bivalves)
- Quantify how potential management actions, such as nutrient load reductions and salt pond operation, will influence ecosystem response (phytoplankton biomass, DO)
- Characterize and quantify uncertainty

R 6: Characterize phytoplankton community composition in Lower South Bay, and explore mechanisms that influence community composition, including potential sources of harmful algal species

R 7: Further analyze existing fish data to better characterize spatial and temporal variability in fish populations, and drivers of that variability

R 8: Explore the feasibility of using existing benthos survey data to assess habitat condition with respect to DO

R 9: Identify appropriate protective DO conditions for fish and other biota

- Literature review to determine:
  - What are we trying to protect?
  - What conditions would be protective?
  - What is the uncertainty in our understanding for species of interest?
- Compare existing conditions with protective conditions

R 10: Conduct fish and benthos surveys, in conjunction with habitat surveys (DO, T, food abundance and quality, etc.) to assess condition

Acknowledgements

This synthesis report was prepared as part of the San Francisco Bay Nutrient Management Strategy (NMS). The datasets used were collected by researchers at a number of agencies: USGS-Menlo Park (Section 2, Section 3, Section 4, Section 5, Section 6); San Jose-Santa Clara Regional Wastewater Facility (Section 2); City of Palo Alto (Section 2); Sunnyvale Water Pollution Control Plant (Section 2); South Bay Dischargers Authority (Section 2, Section 6, Section 7); USGS-Sacramento (Section 3, Section 6); CA Department of Fish and Wildlife (Section 7); Marine Science Institute (Section 7); South Bay Salt Pond Restoration Project, UC-Davis (Section 7)

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Table of Contents

1. Introduction

Lower South Bay (LSB), a shallow subembayment of San Francisco Bay (SFB), is situated south of the Dumbarton Bridge, and is surrounded by, and interconnected with, a network of sloughs, marshes, and former salt ponds undergoing restoration. LSB receives 120 million gallons per day of treated wastewater effluent from three publicly owned treatment works (POTWs) that service San Jose and the densely populated surrounding region. During the dry season, when flows from creeks and streams are at their minimum, POTW effluent comprises the majority of freshwater flow to Lower South Bay. Although LSB has a large tidal prism, it experiences limited net exchange with the surrounding Bay, because much of the water that leaves on ebb tides returns during the subsequent flood tides. The limited exchange leads to distinctly different biogeochemical conditions in LSB compared to other SFB subembayments, including LSB having the highest nutrient concentrations and highest phytoplankton biomass.

This report was prepared as one in a series of reports that explore the current state of knowledge on nutrient-related issues in San Francisco Bay, in support of the San Francisco Bay Nutrient Management Strategy (NMS). The report's main goals are:

- Synthesize data and observations in LSB related to nutrient inputs and cycling in LSB, major indicators of ecosystem response to nutrients, and the range of physical, chemical, and biological factors that regulate ecosystem response;
- Identify major data or conceptual gaps across.

The report is organized as follows:

- Section 2: Nutrients: loads, seasonal, spatial, and ambient concentrations, and evidence for the importance of in situ biogeochemical transformations (Section 2)
- Section 3: Suspended sediment concentrations, and their influence on light levels
- Section 4: Abundance of benthic grazers
- Section 5: Phytoplankton biomass
- Section 6: Dissolved oxygen concentrations
- Section 7: Fish abundance
- Section 8: High priority science questions, and proposed activities to address those questions.

2. Nutrients in Lower South Bay

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2.1 Introduction

The main goals of Section 2 are:

- Collect and synthesize historical and current data on nutrient loads to and concentrations within Lower South Bay.
- Examine how loads and concentrations have changed over time, and how concentrations vary spatially within Lower South Bay
- To the extent possible, assess the fate of N and P in Lower South Bay
- Identify major data gaps or uncertainties

Data from several research and monitoring programs over the period of 1970-2012 were analyzed to characterize seasonal, temporal and spatial variations in nutrient loads and receiving water concentrations, and to the extent possible, to explore the underlying causes of these variations. The fate of DIN and DIP in LSB and the importance of losses or transformation were assessed through a conservative mixing analysis. The section ends with a discussion on major data gaps and recommendation for future research needs.

2.2 Sources of Data

Nutrient Loads Monthly effluent concentrations and flow data from publicly owned treatment works (POTWs) were obtained from the three publicly-owned treatment works (POTW) that discharge directly into Lower South Bay - San Jose/Santa Clara (SJSC), City of Palo Alto, and City of Sunnyvale WPCP (Tables 2.1-2.2). Monthly data for nitrate (NO_3^-), ammonium (NH_4^+) and orthophosphate (o-PO_4) or total P (TP), reported by POTWs as part of permit requirements. Data from 1975-2011 were available for SJSC and 1981-1986 and 1994-2011 were available for Palo Alto. For Sunnyvale, NO_3^- and NH_4^+ data were available from 1982 to 2011, and total phosphorus, as opposed to o-PO_4, data was available beginning in 1988. For comparison with other POTWs and ambient water quality data, Sunnyvale o-PO_4 concentrations were estimated using total phosphorous (TP) measurements and average % TP as o-PO_4, based on an analysis of monthly TP and o-PO_4 from 2012-2013 (Table 2.2). Monthly nutrient loads from each POTW were estimated by combining concentrations when available with flow data. Average annual and monthly stormwater NO_3^-, NH_4^+ DIN, and DIP loads are based on estimates from a recent study that quantified external nutrient loads to SFB (SFEI 2014, #704).

Ambient Nutrient Concentrations Ambient nutrient concentration data were obtained from several sources (Table 2.2). USGS maintains a science and monitoring program in San Francisco Bay that measures a range of water quality parameters (e.g., salinity, temperature, turbidity, suspended sediments, nutrients, dissolved oxygen and chlorophyll a) on a bi-weekly to monthly basis along the Bay's axis (Lower South Bay to the lower Sacramento River). Three stations (34, 35, and 36) are located in LSB (Figure 2.1) and monthly concentration data are available for measured parameters at these stations over the period 1970-1980 and 1988 to 2012; no nutrient data are available between 1981 and 1987. Nutrient concentrations at other USGS-monitored SFB stations were also analyzed, both for comparison of LSB with other subembayments, and because water quality at stations between the Dumbarton Bridge and the San Bruno Shoal are believed to be strongly influenced through tidal exchange by conditions south of the Dumbarton. The South Bay Dischargers Authority (SBDA; an entity that carried out nutrients receiving water monitoring from 1963-1993, and nutrients monitoring from 1975-1993) and individual wastewater agencies have also carried out ambient water quality monitoring (Larry Walker Associate, Inc, 1983). Between SBDA, and the three POTWs, receiving water quality was monitored at 24 stations along tidal sloughs and creeks, and at two stations within LSB (Figure 2.1) with varying completeness between 1975 and the present. Monitored parameters included: nutrient concentrations, salinity, temperature, turbidity, pH, and Secchi depth. The eight stations (C-1-0 through C-8-0) along Artesian slough have the most complete long-term records, with continuous monthly data available from 1975 to 1993, and three of them (C-3-0, C-7-0, and C-8-0) also have monthly data available from 2003 to 2012. Nutrient concentration data are only available from 1981 to 1986 for Palo Alto and Sunnyvale receiving water stations. SB4 has data from two time periods -1981 to 1986 and 2002 to 2009, and SB5 only has six years of data, from 1981 to 1986. Data from these stations were analyzed to characterize spatial variability of nutrient concentrations moving away from Coyote Creek/Artesian slough toward Dumbarton Bridge, as well as the spatial and seasonal variability of nutrient concentrations in SJSC's near-field receiving waters.

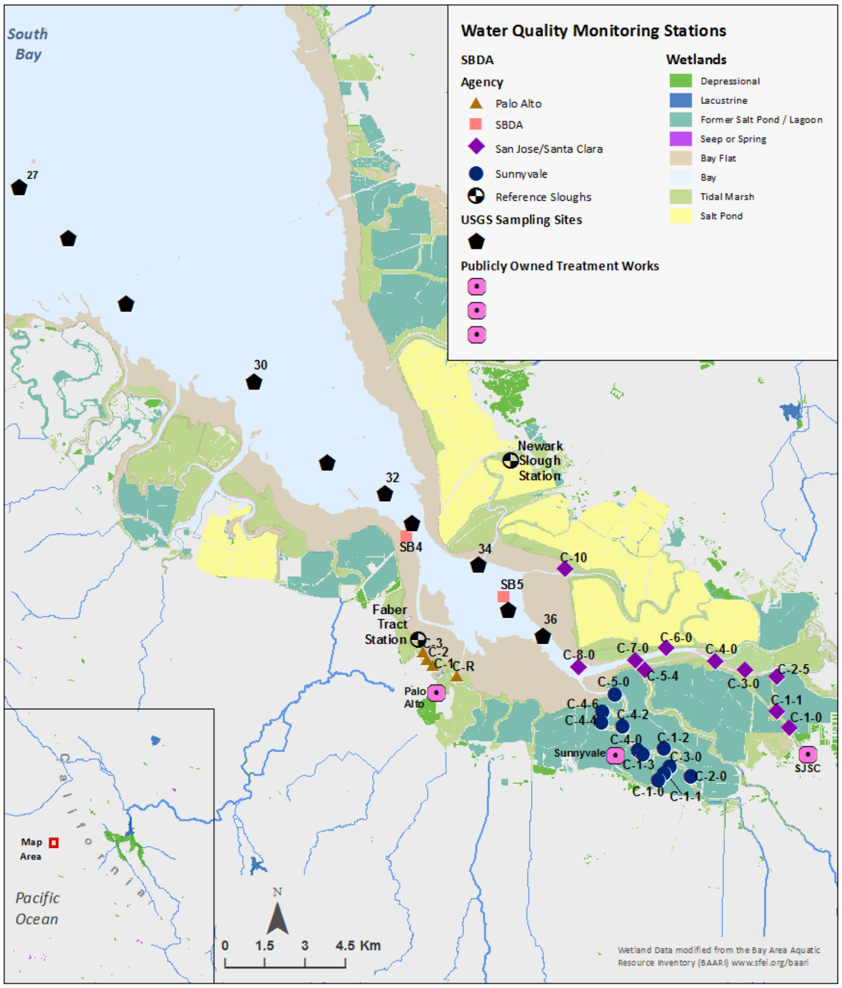


Figure 2.1. Monitoring locations for wastewater effluent and water quality data used in this report

Table 2.1. Operation details on the three wastewater dischargers to Lower South Bay

Treatment plant	**Treatment Level**	**Technology**	Population served	Rated capacity (mgd)	Summary of major upgrade
San Jose/ Santa Clara	Advanced	Biological nutrient removal (BNR) with filtration	1,365,000	167	February 1979 - installed and initiated nitrification and filtration processes. NH3 was nearly fully converted to NO3 after this point. August 1997 - converted to Step-Feed BNR Process which reduced NO3 concentrations by roughly 40%.
Palo Alto	Advanced	Activated sludge +fixed film, dual media filters	228,500	38	The advanced treatment facilities (tower-type roughing filters and dual media filters) were completed in 1980.
Sunnyvale	Advanced	Ox ponds, fixed film reactor for N, dual media filtration	136,000	29.5	Advanced treatment facilities (nitrification towers and dual media filters) wer completed in 1978 and were fully operational in 1979.

Table 2.2. Available wastewater effluent and nutrient water quality data for Lower South Bay used in this report

Data	Source	NH4		NO3		PO4		Flow	
		Date	# samples	Date	# samples	Date	# samples	Date	# samples
Effluent[1]	San Jose/Santa Clara	1965-2011	576	1975-2011	452	1974-2011	447	1957-2011	648
	Palo Alto	1981-1986	60	1981-1986	60	1981-1986	60	1981-1986	60
		1994-2011	845	1994-2011	220	1994-2011	204	1994-2011	6326
	Sunnyvale	1982-2011	2589	1982-2011	1298	1988-2011[2]	444	1982-2011	2708
Ambient Water quality	USGS[3]	1972-1980	738	1970-1980	852	1970-1980	916		
		1988-2012	1593	1988-2012	1599	1988-2012	1595		
	San Jose stations	1975-1993[4]	3772	1975-1993[4]	1304	1975-1993[4]	972		
		2003-2012[5]	307	2003-2012[5]	409	2003-2012[5]	359		
	Palo Alto stations	1981-1986	337	1981-1986	286	1981-1986	290		
	Sunnyvale stations	1981-1986	347	1981-1986	349	1981-1986	349		
	SB4	1981-1986	117	1981-1986	118	1981-1986	118		
		2002-2006	25	2002-2009	50	2002-2009	76		
	SB5	1981-1986	117	1981-1986	118	1981-1986	118		

[1] Data from 2012-2013 are available and has been analyzed in SFEI (2014, #704), but are used in this report

[2] Phosphorus is reported as TP

[3] For stations 27-36 (shown on Figure 2.1)

[4] Monitoring occurred as early as 1963 for temp, secchi depth, pH and dissolved solids, but nutrient monitoring began in 1975

[5] no data at C-4-0, C-5-0 or C-6-0

2.3 Results and Discussion

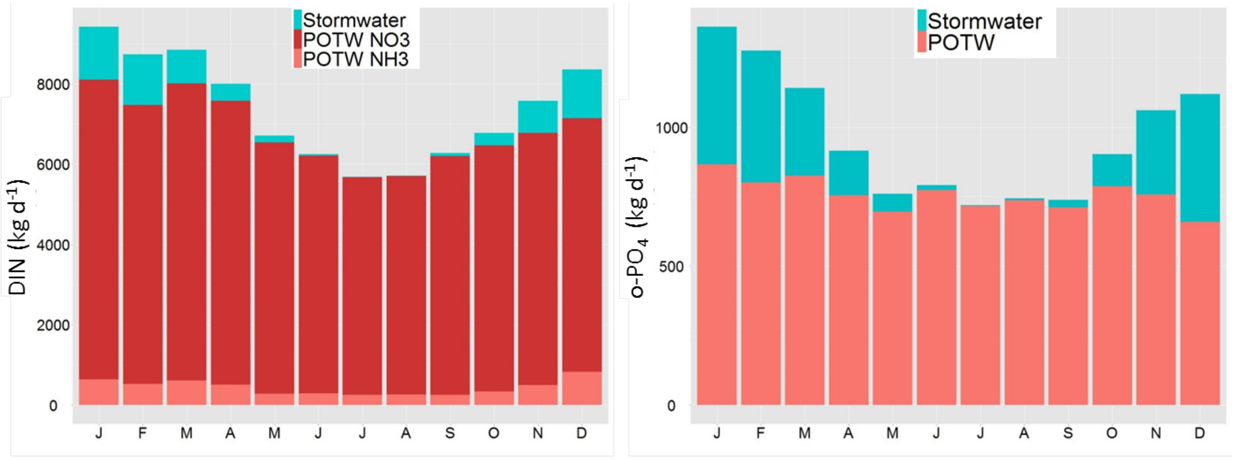
2.3.1 Nutrient loads

Nutrient loads to LSB and other subembayments of SFB are explored in detail in another recent report (SFEI #704, 2014), and are summarized briefly below.

2.3.1.1 Current (2006-2011) nutrient loads to LSB

Lower South Bay receives among the highest area-normalized nutrient loads in San Francisco Bay (SFEI 2014, #704). On average, 7500kg dissolved inorganic nitrogen (DIN,=NO_3^- + NH_4^+) and 1000kg o-PO_4 are delivered to LSB each day, with POTWs being the dominant sources (Figure 2.2). NO_3^- is the predominant form of DIN discharged to LSB (90%) because LSB POTWs carry out nitrification prior to discharge. POTW loads to LSB exhibited some seasonality (Figure 2.2). From the dry season to the wet season, DIN loads from POTWs increased by 20%, with the highest loads observed in January-April. o-PO_4 loads remained relatively constant year-round, with slightly lower loads during summer (May-September).

Stormwater flows deliver seasonally-varying N and P loads to LSB. During the summer months, when there is little or no rainfall, stormwater nutrient loads were estimated to be insignificant. However, from October to April, when storm events typically occur, estimates suggest that stormwater contributes minimally to DIN loads (<10%) and more substantially to o-PO_4 loads (up to 30%) during peak runoff periods (Figure 2.2). The current load estimates are highly uncertain due to data limitations, and thus the relative uncertainty in the magnitude of stormwater derived N and P loads is high. Despite the uncertainty, it seems unlikely that stormwater loads to LSB could rival POTW loads at the subembayments scale, in particular for DIN. A nontrivial contribution of stormwater-derived o-PO_4 to overall o-PO_4 loads cannot be ruled out during wet months



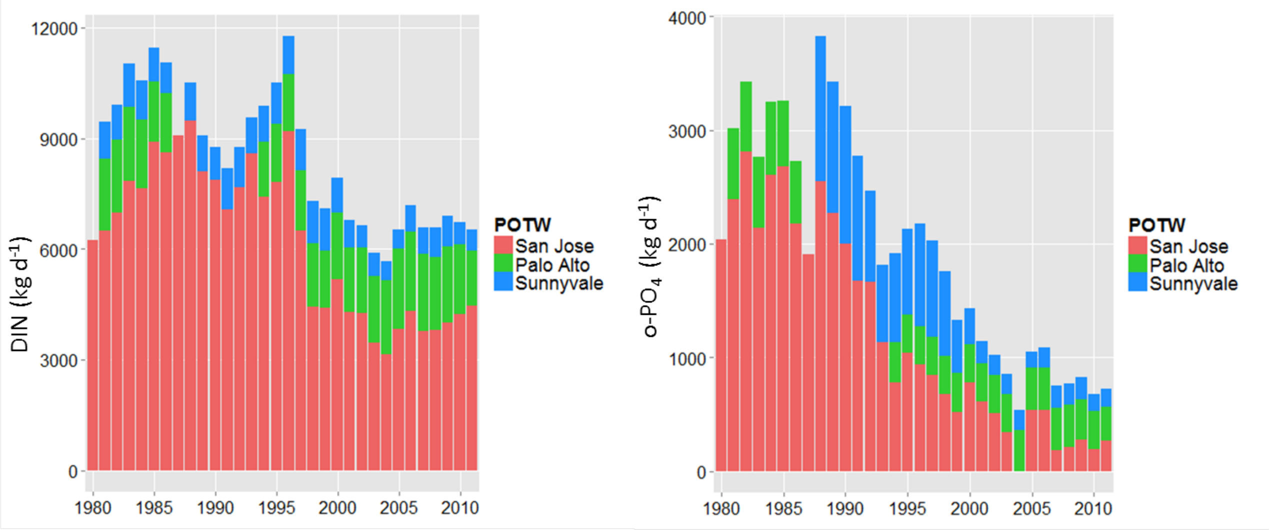
(a) DIN

(b) o-PO_4

Figure 2.2. Current (2006-2011) nutrient loads to Lower South Bay, averaged by month. POTWs are the dominant source of both DIN (predominately of the form of NO_3^-) and o-PO_4. Note that Sunnyvale did not measure o-PO_4 prior to 2012, so estimates of o-PO_4 were made using total phosphorous (TP) measurements and average % TP as o-PO_4 data from 2012

2.3.1.2 Changes in POTW loads: 1980-2011

POTW nutrient loads to LSB have decreased substantially over the past 30 years (Figure 2.3). From 1980 to 2011, annual average POTW discharges of DIN decreased ~35%, from ~10000 kg d^{-1} in the early 1980s to ~6500 kg d^{-1} in recent years (Figures 2.3a). o-PO_4 loads have decreased more substantially than DIN, dropping 70% from 3000 kg d^{-1} to ~700 kg d^{-1} between 1980 and 2011(Figure 2.3b). The decrease in DIN and o-PO_4 loads occurred over a time period when the population served by these POTWs increased by 40%. SJSC is the largest-capacity POTW discharging to LSB, and currently accounts for ~80%, ~60%, and 30% of effluent flows, DIN loads, and o-PO_4 loads, respectively. The annual-average decreases in DIN loads between 1980 and 2011 were due primarily to changes in treatment technology at SJSC. o-PO_4 loads decreased from all POTWs, likely due in part to the removal of P from household detergents. In addition, the larger proportional drop in o-PO_4 load from SJSC was the result of changes to its treatment technology.



(a)

(b)

Figure 2.3. Annual average DIN (a) and o-PO_4 (b) loads from each of the three wastewater discharges to Lower South Bay, 1980-present. Sunnyvale did not measure o-PO_4 prior to 2012, so estimates of o-PO_4 were made using total phosphorous (TP) measurements and average % TP as o-PO_4 data from 2012. Note: Missing values for DIN or o-PO_4 for a POTW during a given year indicates that monitoring for that analyte did not occur, not that load is 0.

Changes in the amount and composition of nutrient loads over the period of 1980-present from each of the three POTWs discharging to LSB are described below and illustrated in Figures 2.4-2.6.

SJSC

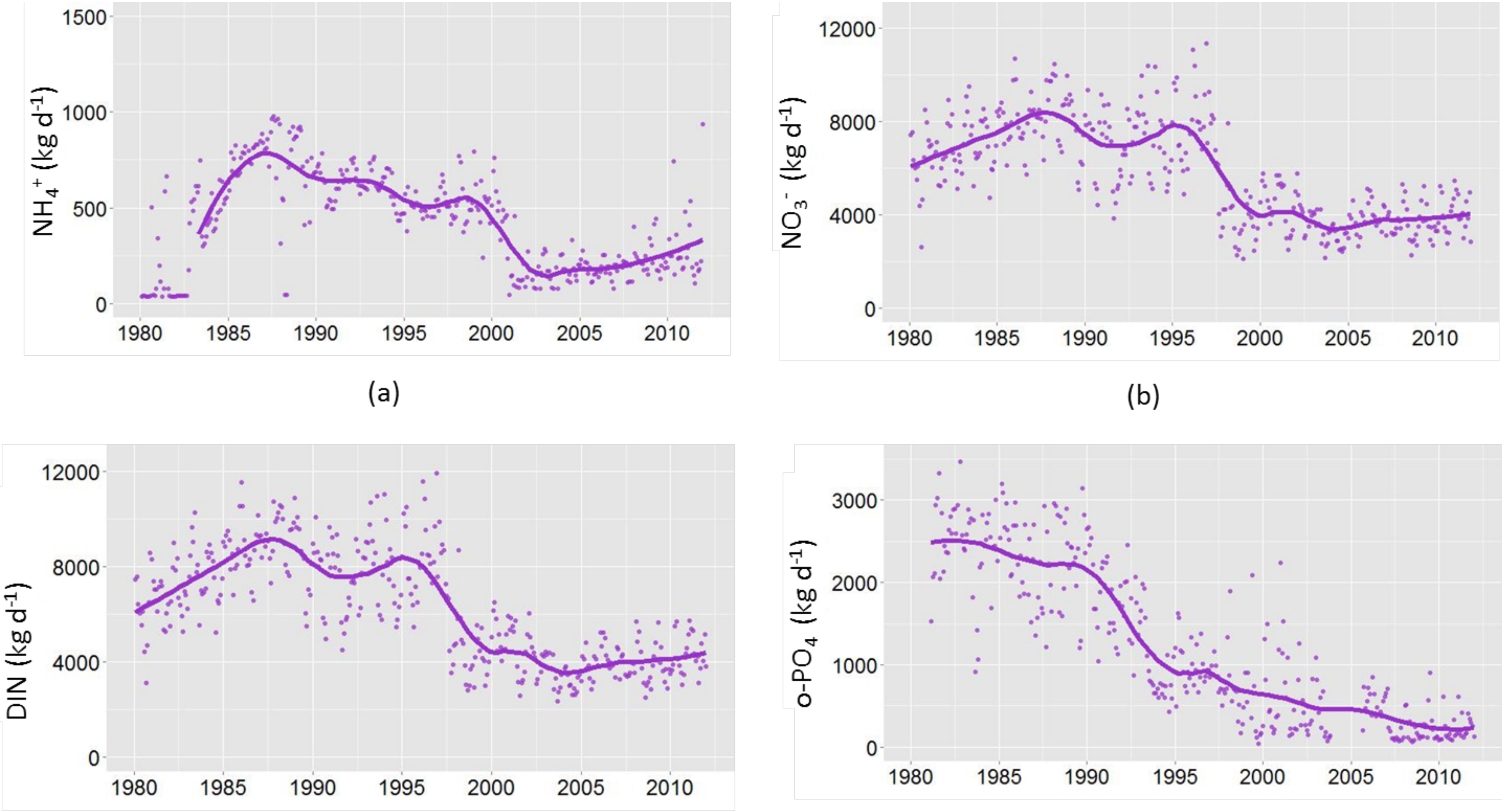
SJSC initiated nitrification in 1979, and since then NO_3^- has been the predominant form of DIN exiting the plant, with limited residual NH_4^+ (Figure 2.4a and 2.4b). In 1997, SJSC implemented a step-feed biological nutrient removal (BNR) process that reduced DIN loads by roughly 40% (Figure 2.4b, 2.4c). Currently, DIN loads are ~4000 kg d^{-1}, with substantial variability (±30-40%) around this central tendency value. Although NH_4^+ now represents only ~10% of SJSC's N load, there appears to have been a trend of increasing NH_4^+ over the past 10 years. This load increase was apparently due to increases in effluent NH_4^+ concentrations, since flows have actually decreased over this same time period (SFEI 2014, #704). o-PO_4 loads from SJSC have decreased by ~90%, from ~2500kg d^{-1} in 1980 to ~250 kg d^{-1} in 2011(Figure 2.4d). The reduction was steepest during 1990-1995, due in large part to changes in treatment technology.

Palo Alto

Palo Alto also nitrifies its effluent before discharge, and the majority of its DIN occurred was in the form of NO_3^- (Figure 2.5a, 2.5b). In general, NH_4^+ loads have accounted for <5% of DIN loads. Palo Alto's DIN loads have increased by approximately 30% since 1995, due to increases in effluent concentrations (Figure 2.5c). o-PO_4 loads remained more or less constant from 1995 to 2005 but increased by approximately 20% between 2005-2009, and decreased back to 1995 levels since 2009 (Figure 2.5d).

Sunnyvale

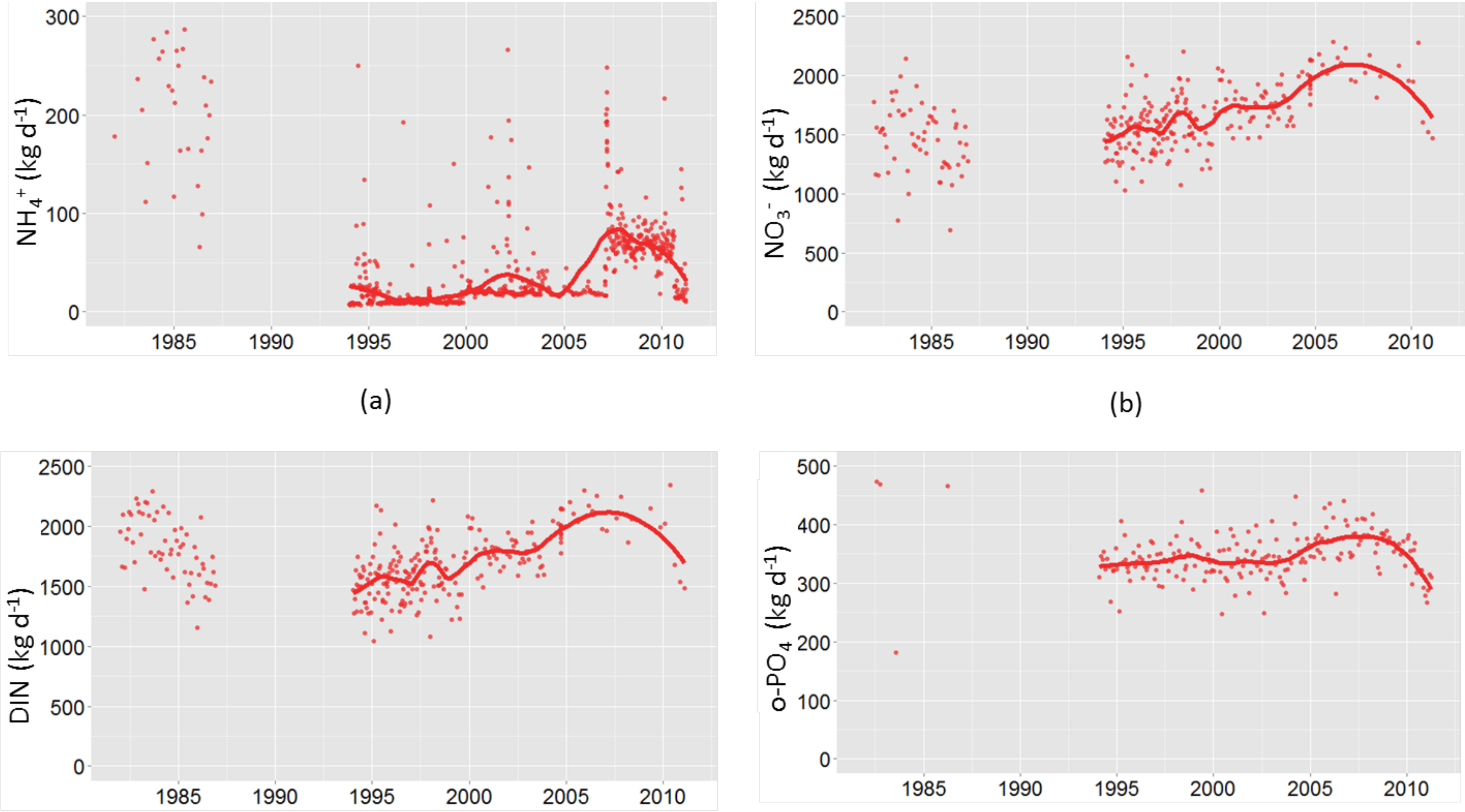
Strong seasonality was evident in Sunnyvale's NH_4^+ and NO_3^- loads (Figure 2.6). This is apparently due Sunnyvale's use of oxidation ponds in secondary treatment and fixed growth reactors to nitrify, and strong temperature dependence of the associated biological processes (T. Hall, EOA Inc., pers. comm.). Beyond this seasonality there was no apparent trend in baseline NH_4^+ loads. Although DIN loads varied by nearly 100% around the central tendency, average DIN loads appear to have decreased by >30% since 2000. Phosphorus was measured as TP at Sunnyvale and o-PO_4 loads were calculated as 93% of TP, a factor that is estimated based on 2012-2013 effluent monitoring data when both forms of P were monitored. o-PO_4 loads remained centered around ~350kg d^{-1} from 1990 to 1998, but decreased by ~50% from 1998 to 2002 (Figure 2.6d). Over the past 10 years, average o-PO_4 loads have remained relatively constant, hovering around 150 kg d^{-1}.



(a) (b)

(c) (d)

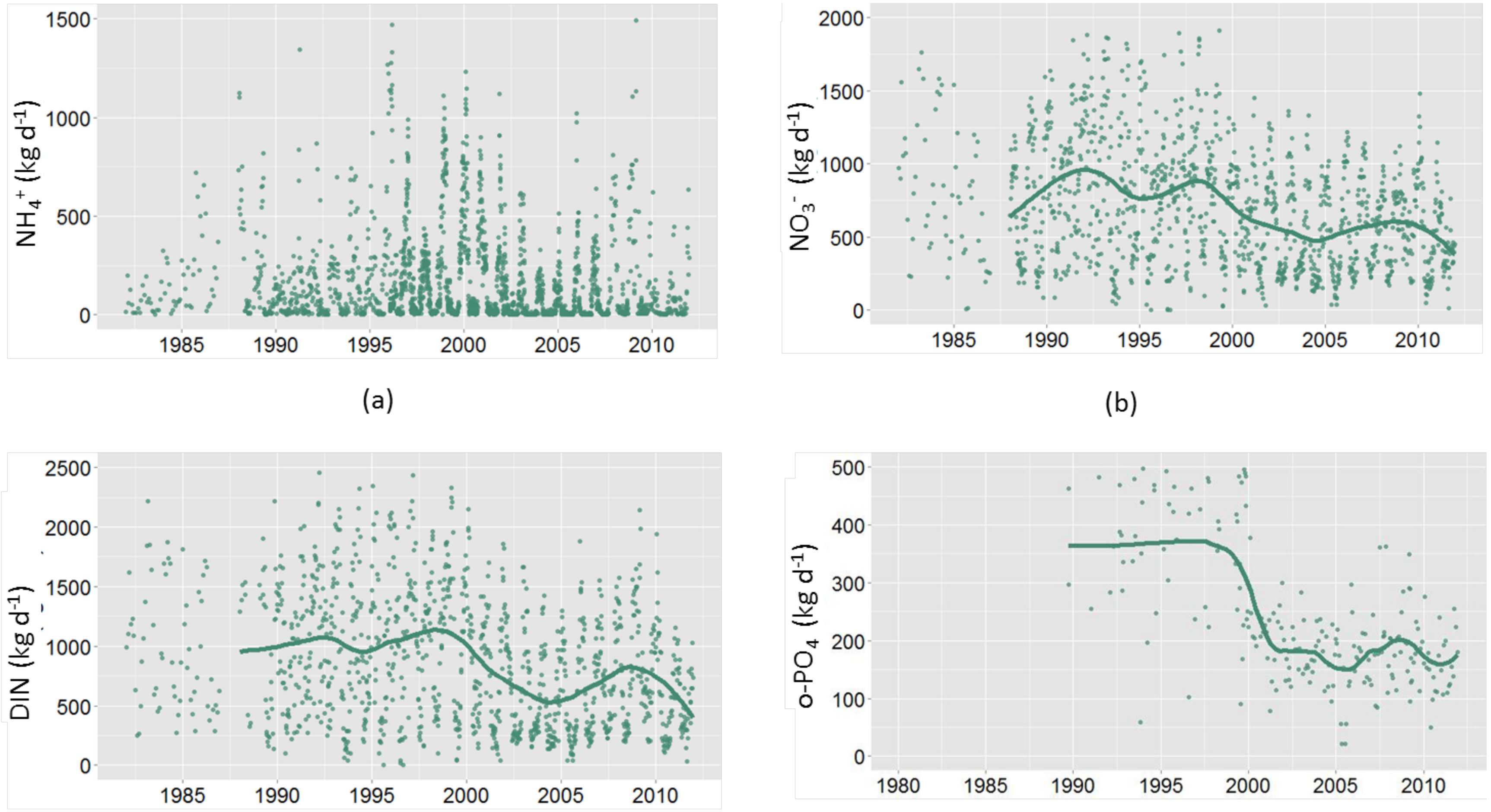
Figure 2.4. Effluent (a) NH_4^+ (b) NO_3^- (c) DIN and (d) o-PO_4 loads from SJSC, 1980-present, monitored approximately monthly. A loess line (smoothing parameter = 0.3) was added to some Figures in order to show a general pattern, but is not intended as a rigorous trend analysis



(a) (b)

(c) (d)

Figure 2.5. Effluent (a) NH_4^+ (b) NO_3^- (c) DIN and (d) o-PO_4 loads from Palo Alto, 1981-present, monitored approximately monthly. A loess line (smoothing parameter = 0.3) was added to some Figures in order to show a general pattern, but is not intended as a rigorous trend analysis



(a)

(b)

(c)

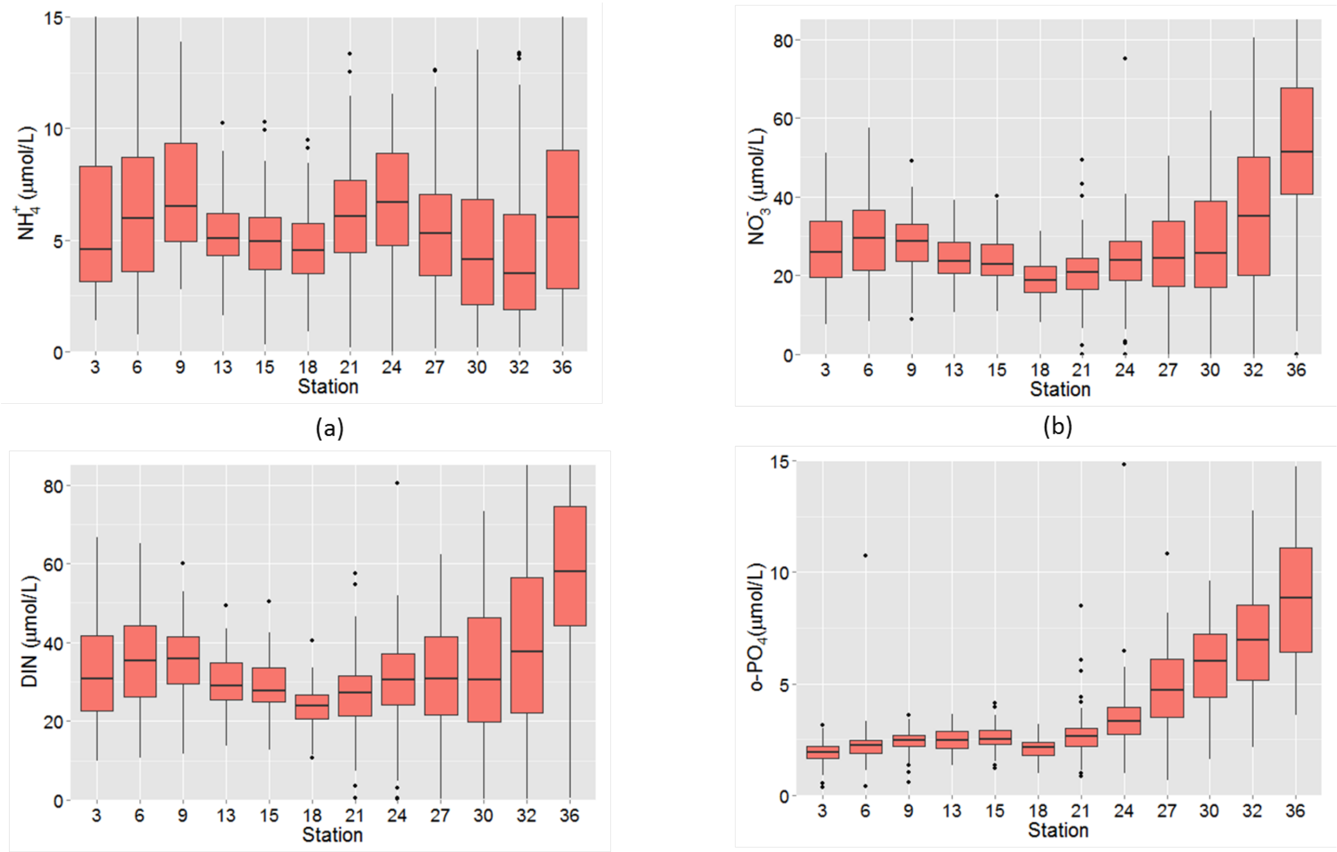
(d)

Figure 2.6. Effluent (a) NH_4^+ (b) NO_3^- (c) DIN and (d) o-PO_4 loads from Sunnyvale, 1980-present, monitored approximately monthly. o-PO_4 is scaled as 0.93*TP load, based on 2013-2013 effluent monitoring data where both were monitored. A loess line (smoothing parameter = 0.3) was added to some Figures in order to show a general pattern, but is not intended as a rigorous trend analysis

2.3.2 Current ambient nutrient concentrations in Lower South Bay

2.3.2.2 Spatial variability in current ambient nutrient concentrations

DIN and o-PO$_4$ concentrations measured in LSB were 1.5-4 times higher than DIN and DIP measured in other SFB subembayments (Figure 2.7), and also substantially greater than those observed in many other estuaries (Figure 2.8; Cloern and Jassby 2012). The individual forms of DIN, NO_3^- and NH_4^+, were also generally the highest in LSB (NH_4^+ concentrations in Suisun Bay exceeded those in LSB during November-January). The elevated DIN and o-PO$_4$ concentrations in LSB result from a combination of several factors. First, LSB receives among the highest area-normalized N and P loads in SFB (SFEI 2014, #704). Second, LSB is relatively shallow (average depth = 3 m), which translates to higher concentrations per unit load. Finally, water south of the San Bruno Shoal (near San Mateo Bridge), including Lower South Bay, has a fairly long residence time compared to other areas of SFB, allowing nutrients to accumulate to higher concentrations.



(a)

(b)

(c)

(d)

Figure 2.7. Current (2005-2012) nutrient concentrations along the San Francisco Bay's longitudinal axis in the deep channel at selected USGS stations extending from Suisun Bay (station 3) to Lower South Bay (station 36). All data from this period are presented, and seasonal variability in concentrations contributes to the wide distribution at some stations. The horizontal line represents the median, and the box extends to the 25th and 75th percentiles. The whiskers extend to 1.5*IQR from the 25th and 75th percentiles.

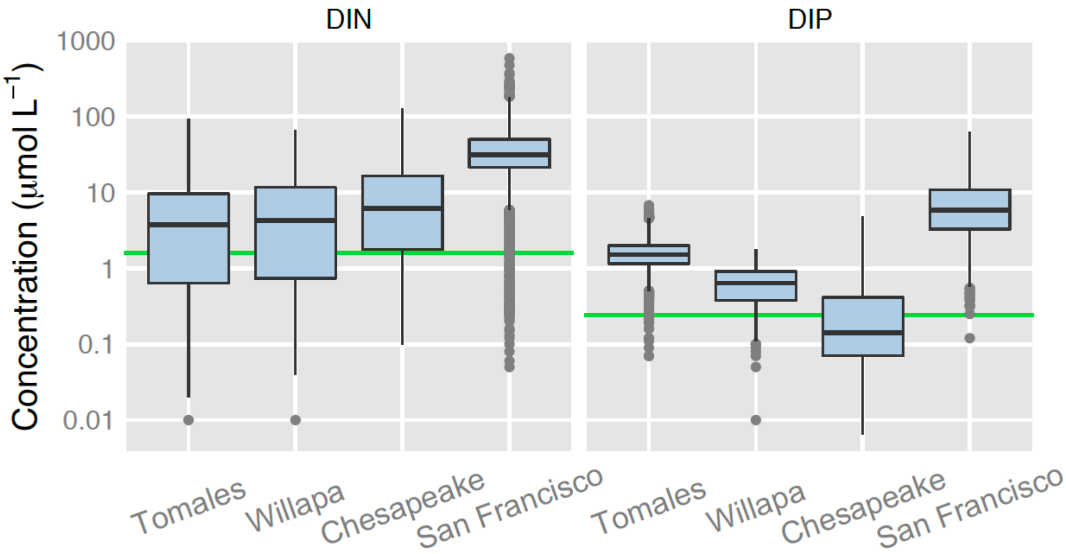
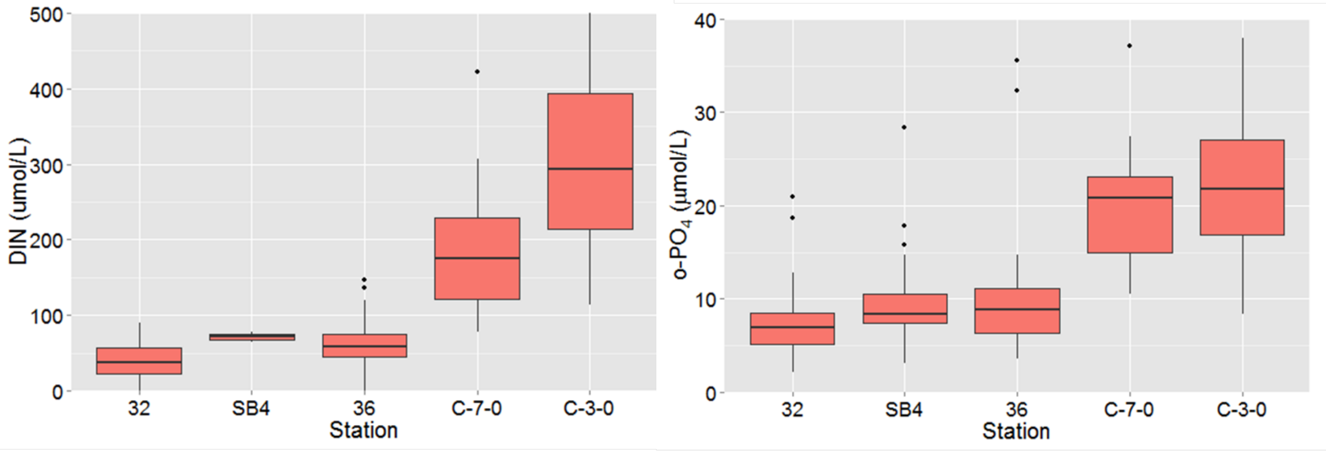


Figure 2.8. Nutrient concentrations in South Bay (stations 21-36, Figure 2.1) compared to other estuaries. The green line represent characteristic half-saturation constants for phytoplankton growth. Source: Cloern and Jassby (2012)

DIN and o-PO_4 concentrations increased along a southerly transect moving from the Dumbarton Bridge towards the Coyote Creek and Artesian Slough confluence (near SJSC outfall) (Figure 2.9). Both DIN and DIP concentrations increased by a factor ~4 between the open water station s36 and C-7-0, and DIN increased by another factor of ~1.5 between C-7-0 and C-3-0 (Figure 2.1, Figure 2.9). That spatial variation of nutrient concentrations in LSB is likely due to a combination of factors including dilution, uptake by algae, and loss through denitrification.



(a)

(b)

Figure 2.9. Current (2005-2012) nutrient concentrations in Lower South Bay. Stations 32, 34 and 36 are USGS stations in the open Bay, and stations C-7-0 and C-3-0 are monitored by SJSC and are located in Coyote Creek (see Figure 2.1). Box and whisker descriptions are the same as in Figure 2.7

2.3.2.1 Seasonal fluctuations in ambient nutrient concentrations

DIN concentrations exhibited strong seasonal variability in LSB (Figure 2.10). The majority of DIN was present as NO_3^-, and most of the variability in DIN resulted from changes in NO_3^-, so the two are discussed together. Median DIN concentrations ranged from 50-100 µM. In aquatic ecosystems, N undergoes numerous transformations and can experience multiple fates (denitrification and loss of N_2 to the atmosphere, uptake by phytoplankton and plants and conversion to organic forms, settling and accumulation in sediments, remobilization of NH_4^+ from the sediments). At s36, DIN concentrations were lowest from March to September/October, and higher during November through February. The lower DIN concentrations in September-October could be due to a combination of in situ processes, including elevated uptake rates by phytoplankton, or higher rates of denitrification due to longer days (sunlight) and warmer temperatures. Seasonally-lower DIN loads may also contribute somewhat to the seasonal variation in ambient concentrations: DIN loads from Sunnyvale were ~1000 kg d^{-1} lower in summer than winter (Figure 2.6), a ~15% decrease in overall DIN loads; loads from SJSC are also lower in summer than winter by ~10%. At three frequently monitored stations immediately north of the Dumbarton Bridge (s27, s30, s32), the summer DIN minimum concentrations were roughly the same as those measured at s36, but the winter maxima were 25-50% lower.

NH_4^+ concentrations also varied seasonally in LSB, and ranged from 3 µM to 10 µM at s36. The NH_4^+ concentration minima in April and September coincided with periods of highest phytoplankton biomass, and may indicate NH_4^+ uptake by phytoplankton. Since the vast majority of N loaded directly to LSB is in the form of NO_3^-, much of the NH_4^+ observed in the LSB water column was likely regenerated from the sediments. The relative NH_4^+ maximum in June-July likely results from higher rates of mineralization of organic matter in the sediments (due to higher water temperatures). Stations north of the Dumbarton Bridge show little or no increase in NH_4^+ concentrations during summer months. Compared to other SFB subembayments, sediment sources of NH_4^+ may have a larger effect on water column NH_4^+ concentrations in LSB because of its shallow depth: t any flux from LSB sediments would be mixed over a relatively smaller volume of water, causing a larger increase in concentration.

$o\text{-}PO_4$ concentrations exhibited a different pattern in LSB (s36) than was observed for DIN. From January-May, $o\text{-}PO_4$ concentrations were in the range of 6-8 µM, with the suggestion of a modest minimum during March-April, perhaps due to uptake by phytoplankton. $o\text{-}PO_4$ increased to 10-12 µM from June-December. The increase in $o\text{-}PO_4$ may be partially due to mobilization of $o\text{-}PO_4$ from sediments, similar to the observed NH_4^+ increases. Seasonal $o\text{-}PO_4$ patterns are similar at stations north of the Dumbarton Bridge, but concentrations are approximately 30-50% lower throughout the year.

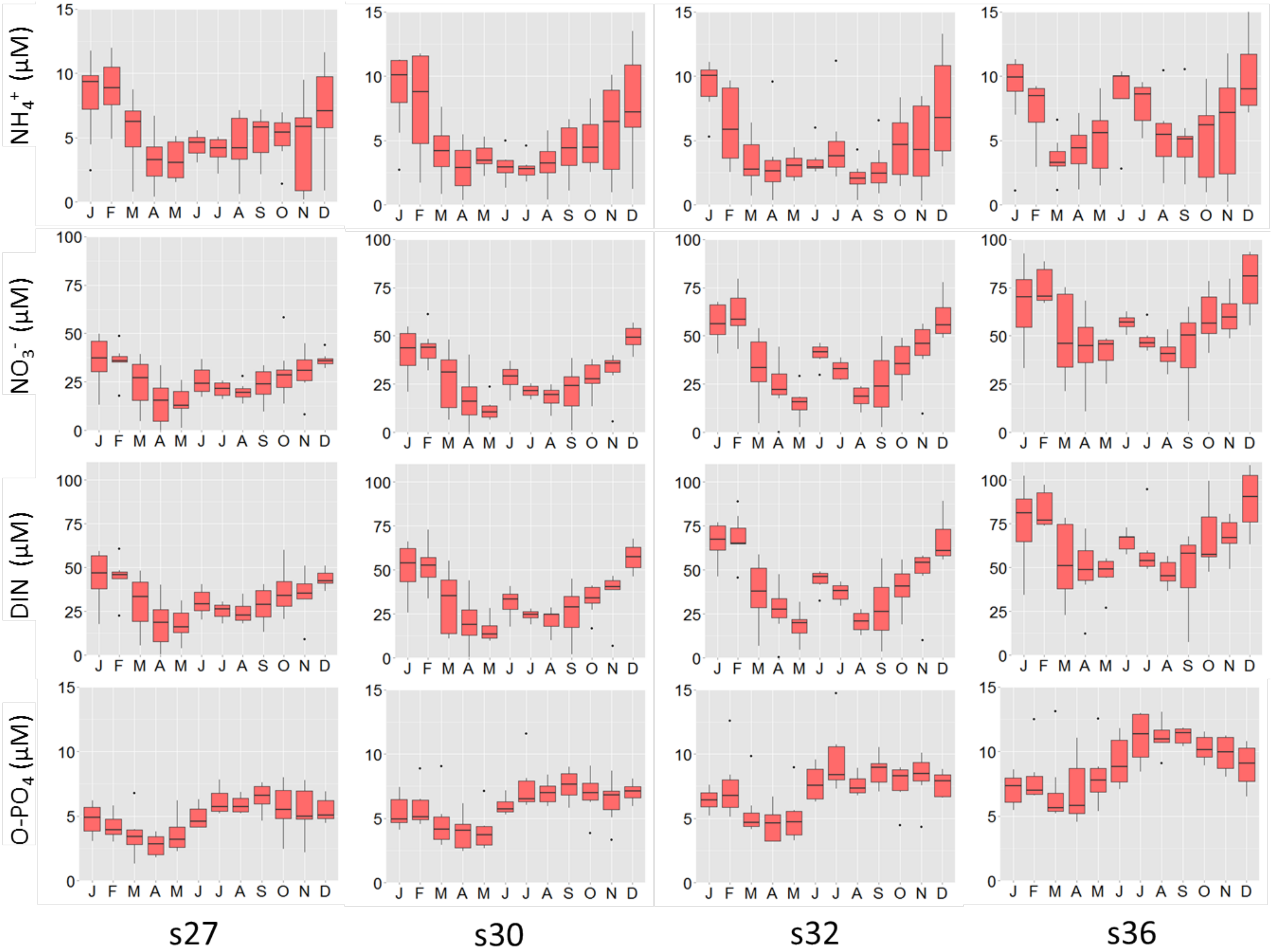


Figure 2.10. Seasonal variability in nutrient concentrations in South Bay. Data from USGS

2.3.2.2 Nutrient fate

Strong seasonal variability in DIN, o-PO_4′, and NH_4^+ concentrations in Lower South Bay (Figure 2.10) suggest that substantial nutrient transformations occur. To explore this issue further, we estimated the fate of DIN in LSB and the importance of losses or transformation using a box modeling approach. The analysis focused on summer months (July-August), when POTW loads to LSB and observed DIN concentrations within LSB are relatively constant (Figure 2.2, 2.10), justifying a steady-state assumption (i.e., DIN mass within LSB remains constant over time). In addition, freshwater inputs to LSB can be considered neglible during summer months, and do not influence residence time.

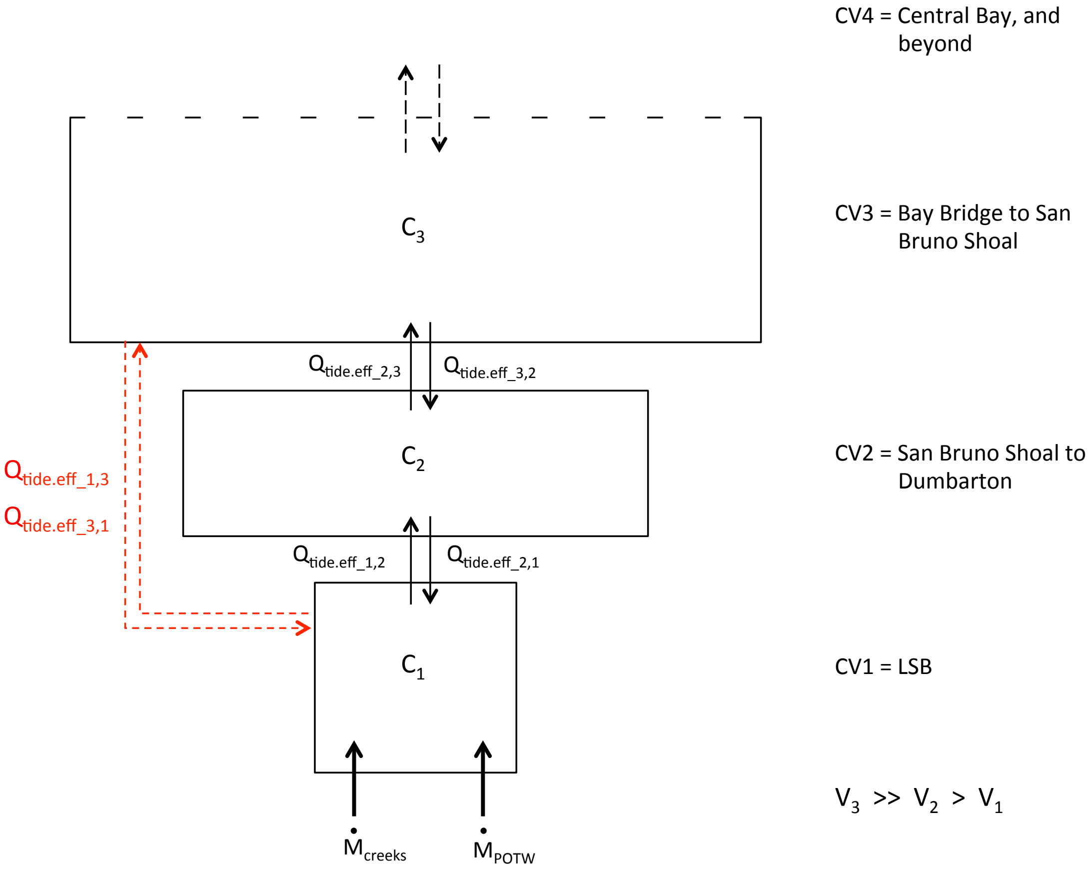


Figure 2.11. Conceptual diagram illustrating the connections between major regions of South Bay and Lower South Bay. Although conceptualized exchange presented includes 3 control volumes (CV) or boxes, not all 3 boxes are explicitly used in the mass balance. CV2 is strongly influenced by water that exits (CV1) LSB. Modeling indicates that much of the water that enter LSB on flood tides is water that had previously exited LSB on prior ebb tides.

We set up the mass balance equation as follows:

$$\frac{dM}{dt} = \dot{M}_{POTW} + \dot{M}_{creeks} + \dot{M}_{tide,in} - \dot{M}_{tide,out} - \dot{M}_{loss/trans}$$

During summer months $\dot{M}_{creeks}$ is expected to be small relative to POTW inputs. At steady-state, $\frac{dM}{dt} = 0$, i.e., mass is not changing as a function of time within the control volume. Thus, the equation becomes

$$\dot{M}_{loss/trans} = \dot{M}_{POTW} + \dot{M}_{tide,in} - \dot{M}_{tide,out}$$

where:

$\dot{M}_{loss/trans}$ = mass of DIN lost/transformed (kg/d)
$\dot{M}_{POTW}$ = DIN loading from POTWs (kg/d), 2005-2011 average
$\dot{M}_{tide,in}$ = DIN entering LSB on incoming tides (kg/d)
= $Q_{tide,eff\_3,1} * C_3$
$\dot{M}_{tide,out}$ = DIN exported from LSB on outgoing tides (kg/d)
= $Q_{tide.eff\_1,3} * C_1$
$Q_{tide.eff\_3,1}$, $Q_{tide.eff\_1,3}$ = the amount of 'new' water that enters LSB on a tidally-averaged basis, not including water that was previously in LSB and left on an earlier flood tide; or the amount of LSB water that leaves on an ebb tide and does not return on subsequent flood tides.
= $\frac{k_{flush}}{V_3}$, where k_{flush} is a first order rate constant for flushing derived from model simulations (as described below), and V_3 is the volume of LSB.
C_3 = DIN concentration of the 'new' water entering LSB
= $C_{s27,avg}$
C_1 = DIN concentration within LSB, assuming a well-mixed control volume. Used average concentration data from station s36.

Estimating the "net" DIN export from LSB by tidal exchange, i.e., $\dot{M}_{tide,in} - \dot{M}_{tide,out}$, is challenging because the waters immediately north of the Dumbarton Bridge are strongly influenced by LSB: much of the water that enters LSB during flood tide is water that had exited LSB on prior ebb tides. Thus, net export of DIN is much smaller than might be expected based on the nominal volume that actually enters and leaves LSB over a full tidal period. To estimate the "effective" tide (water permanently leaving LSB, replaced by new water), we applied a 2D model developed by RMA, which included a conservative tracer introduced at a constant concentration to all waters south of the Dumbarton Bridge at t = 0 (see Appendix A.1 for more details). The "sloshing" of LSB water back and forth past the Dumbarton Bridge is evident based on model tracer contours (Figure 2.12) and the time series of mass remaining within LSB (Figure 2.13). While approximately 60% of the tracer leaves LSB on the first ebb tide, most of that mass re-enters on the subsequent flood tide (Figure 2.13). The effective flushing rate can be estimated using a best-fit exponential curve of the form $M = M_0 e^{-k_{flush}t}$, where $Q_{tide.eff\_1,3} = \frac{k_{flush}}{V_3}$. The exponential fit to the full time series and the maximum daily peaks yields k_{flush} values of 0.05-0.06 d^{-1} (i.e., 5-6% of LSB water is effectively exchanged with 'new' water each day).

When tidal exchange is treated this way ($Q_{tide.eff\_1,3}$ = $Q_{tide.eff\_3,1}$ = 5.4 x 10^6 m^3 d^{-1}), $\dot{M}_{loss/trans}$= 4400 kg d^{-1}. Based on these estimates, 60% of total DIN inputs ($\dot{M}_{POTW} + \dot{M}_{tide,in}$) are lost or transformed (uptake by phytoplankton, transformation), and only 40% leaves by net tidal exchange. More detailed biogeochemical modeling and field investigations are required to fully characterize DIN fate (e.g., distinguish between uptake vs. denitrification, more accurately model exchange and spatially-varying concentrations within LSB). However, this estimate suggests that the majority of DIN that enters LSB either undergoes uptake by phytoplankton or denitrification.

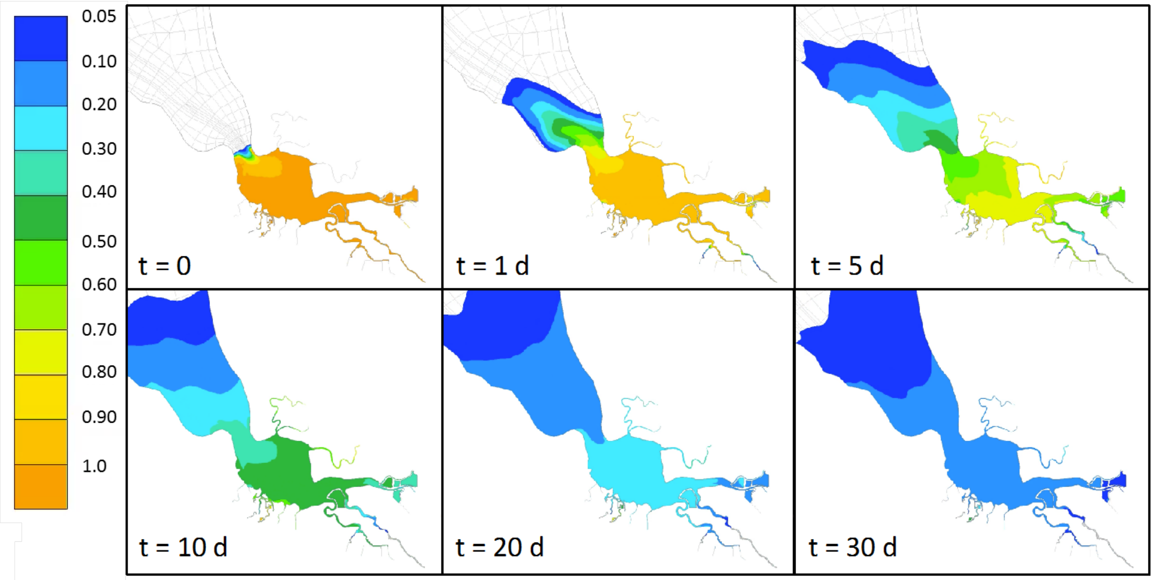


Figure 2.12 Contour plots of tracer concentration remaining in LSB over time. At t=0, concentration in LSB = 1 and concentration elsewhere = 0

2.3.3 Long-term changes in ambient concentrations from 1970 to 2012

2.3.3.1 Open-Bay stations

Nutrient concentrations in Lower South Bay have decreased substantially over the past 40 years as a result of reduced POTW nutrient loads to LSB. NH_4^+ concentrations decreased sharply from 1970 to 1979, and have generally remained below 10uM since 1979 (Figure 2.14a) when SJSC and Sunnyvale upgraded to nitrification (Palo Alto began nitrification in 1980). NO_3^- concentrations increased from ~50 uM in 1970 to ~100 µM in 1985; most of this increase was presumably due to nitrification of effluent prior to discharge (Figure 2.14b). The presence of

NO_3^- at comparable concentrations to NH_4^+ in the early 1970s, despite most of the DIN load originating as NH_4^+, indicates that *in situ* nitrification was an important process in LSB. Compared to changes in the individual N forms, DIN concentrations remained relatively constant from 1970-1985 (Figure 2.14c), although DIN does appear to have increased modestly between 1979 and 1985, consistent with the apparent load increase from SJSC during this time (Figure 2.4c). In the late 1990s, DIN and NO_3^- concentrations decreased to near their current values (80 µM and 50 µM, respectively), coincident with implementation of biological nitrogen removal (BNR) at SJSC in 1997 which reduced DIN loads by roughly ~50% (Figure 2.4c). DIP concentrations have decreased by a factor of 3, from ~30 µM to ~10 µM, between1970 and 2012 (Figure 2.14d), consistent with the decreased DIP loads (Figure 2.4d, 2.6d).

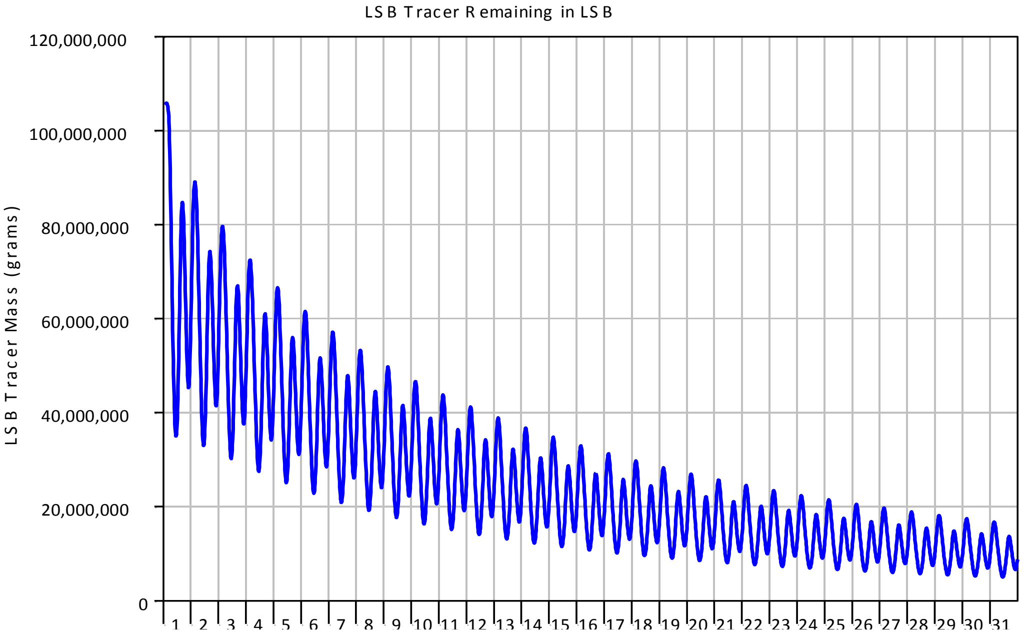
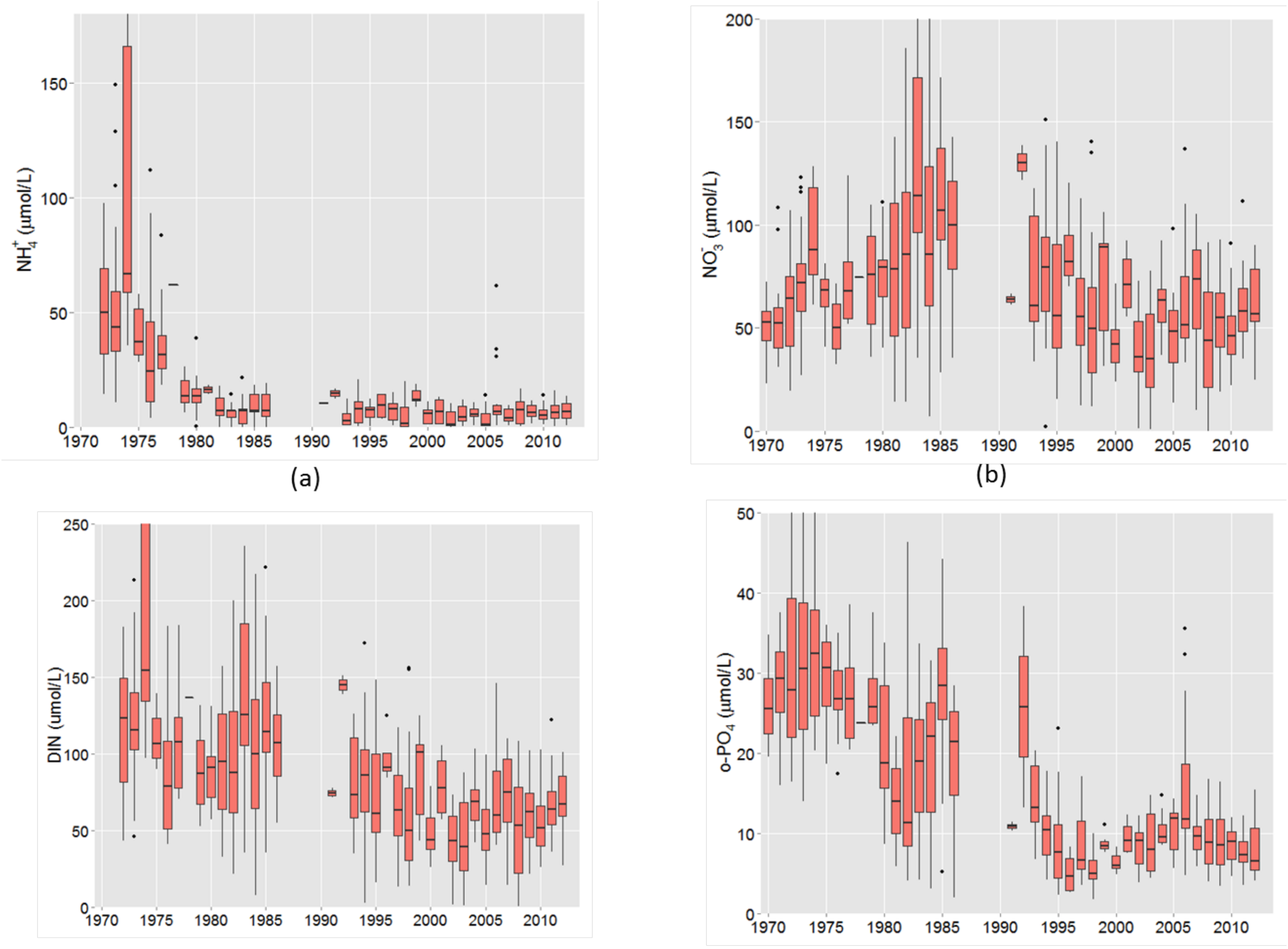


Figure 2.13. Time series of mass of tracer remaining in LSB over time. A significant amount of the mass that exits on one ebb tide re-enters LSB on the next flood tide

2.3.3.2 Slough stations near SJSC's receiving waters

Water quality data collected at biweekly to monthly frequency at multiple stations along Artesian Slough and Coyote Creek from 1979 to 1989 offer useful information on nutrient concentrations and transport in this area of LSB. Nutrient loads from SJSC varied seasonally between 5000-10000 kg d^{-1} during this time (Figure 2.15a), which were substantially higher than present day loads of 4000 kg d^{-1}. Ambient DIN concentrations ranged from <500 µM to 2000 µM, and showed both strong seasonal and spatial variability (Figure 2.15b). DIP concentrations ranged between <50 µM and 300 µM (Fig 2.15e). Salinity data (Figure 2.17c) illustrates that the decreasing DIN and DIP concentrations gradients , and their seasonal variability, along the 12 km transect were in large part the result of dilution both by tidal mixing (mixing of saline water upstream along Coyote Creek and Artesian Slough) and by fluvial freshwater inputs to Coyote Creek (especially during the wet season; e.g. near-zero salinity along all of Artesian Slough and Coyote Creek in early 1983 and 1986). We performed conservative mixing calculations using this data to assess the degree to which DIN and DIP behaved conservatively or nonconservatively; however, the results were inconclusive. Given the data richness, a more sophisticated treatment (e.g., a coupled hydrodynamic and water quality model) may provide valuable insights into nutrient transformations both during this period of time and under current conditions.



(a) (b)

(c) (d)

Figure 2.14. Long-term changes in nutrient concentrations in Lower South Bay (USGS stations 34-36 and station SB-5). Box and whiskers are the same as described in Figure 2.7

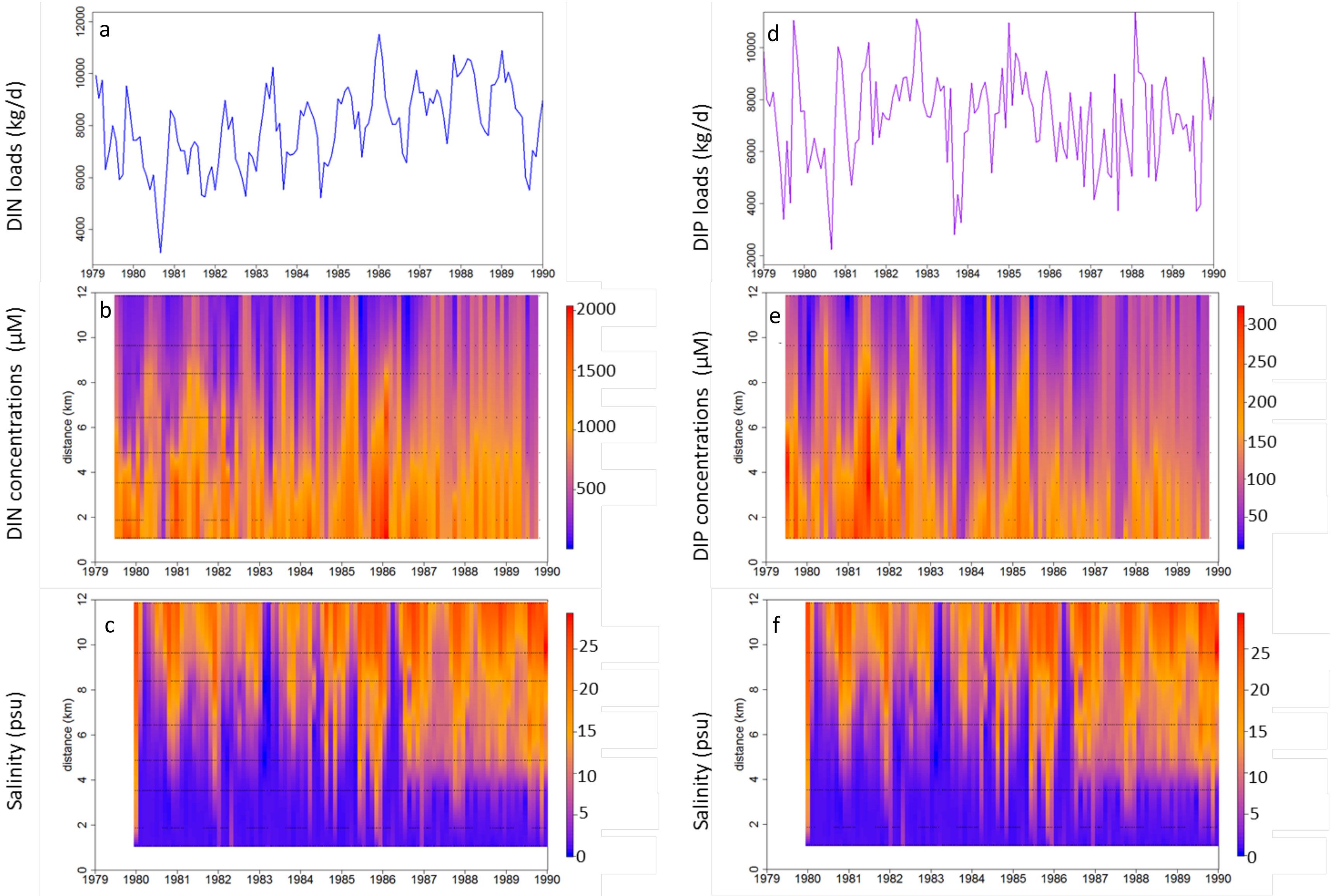


Figure 2.15. Seasonal and interannual variation in DIN and DIP loads, and seasonal, interannual and spatial patterns in DIN and DIP concentration and salinity along a transect extending from SJSC outfall (distance = 0) along Artesian slough and Coyote Creek to C8 where Coyote Creek opens into Lower South Bay. Black points in bottom two panels indicate actual data upon which contours were calculated. Data from SBDA and provided in electronic format by J Ervin (personal communication).

2.4 Major Data Gaps and Recommendations

In assessing the state of the science with regards to nutrient concentrations and loads in Lower South Bay, we have identified the following major knowledge gaps:

1. How do nutrient concentrations and forms vary spatially and temporally, particularly in margin habitats where limited monitoring has occurred to date?
2. What are the dominant processes controlling nutrient fate in Lower South Bay, and how do their magnitudes vary spatially and temporally?
3. What nutrient loads can Lower South Bay assimilate without adverse impacts (e.g., with respect to chl-a, DO, or algal toxins)?
4. What effects are salt pond restoration activities having on nutrient in the margins? The open Bay?
5. What would be protective nutrient levels in terms of biomass, DO, and phytoplankton assemblage or toxins?

We propose a number of high-priority activities to address these knowledge gaps. Since some of these investigations would be resource- and time-intensive undertakings, some level of prioritization is still needed:

- Gather high-spatial resolution data through biogeochemical mapping
  - Characterize spatial and temporal heterogeneity
  - Gather data for model calibration/validation
- Conduct mechanistic field investigations to quantify important processes related to nutrient cycling (slough ←→ open Bay, salt pond ←→ slough, stratification in open Bay and sloughs)
- Develop and apply a coupled hydrodynamic and biogeochemical model for Lower South Bay, including sloughs and margins
  - Examine the role of anthropogenic nutrients and quantify nutrient fate
  - Quantify how potential management actions, such as nutrient load reductions and salt pond operation, will influence ecosystem response (phytoplankton biomass, DO)
  - Characterize and quantify uncertainty

3. Review of suspended sediment in lower South Bay relevant to light attenuation and phytoplankton blooms

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3.1 Introduction

In San Francisco Bay (Fig 3.1), suspended sediment limits light in the water column which in turn limits phytoplankton growth (Cloern, 1987). Thus, suspended-sediment concentration (SSC) and phytoplankton biomass are inversely related. In San Francisco Bay beginning in 1999, SSC decreased (Schoellhamer, 2011), chlorophyll concentrations increased, and autumn phytoplankton blooms occurred for the first time since at least 1978 (Cloern et al., 2007). These observations indicate that the Bay crossed a threshold and fundamentally changed in 1999. San Francisco Bay has been transformed from a low-productivity estuary to one having primary production typical of temperate-latitude estuaries. Cloern et al. (2007) also state that a shift in currents in the Pacific Ocean, improved wastewater treatment, reduced sediment inputs, and introductions of new species may be responsible for the chlorophyll increase. If these trends of increasing Bay clarity and phytoplankton growth continue, eutrophication and other adverse impacts from high nutrient loads become more likely, especially in the lower part of South San Francisco Bay (south of the Dumbarton Bridge, called lower South Bay or LSB) which receives discharge from several wastewater treatment plants, has relatively long (several months in summer) residence times (Walters et al. 1985), and experienced summer depletion of dissolved oxygen prior to improvements in sewage treatment beginning in the 1960s (Nichols et al. 1986).

The purpose of this chapter is to review our understanding of processes affecting suspended sediment in LSB relevant to potential eutrophication. We review previous results and conduct simple analyses of existing data to describe:

- The relation between light attenuation and suspended-sediment concentration
- Vertical dynamics of settling, erosion, and stratification
- Time scales of variability in suspended-sediment concentration
- Spatial gradients of suspended sediment in LSB
- Watershed effects on suspended sediment in LSB including freshwater flows from local tributaries and the Central Valley and tidal restoration of South Bay salt ponds

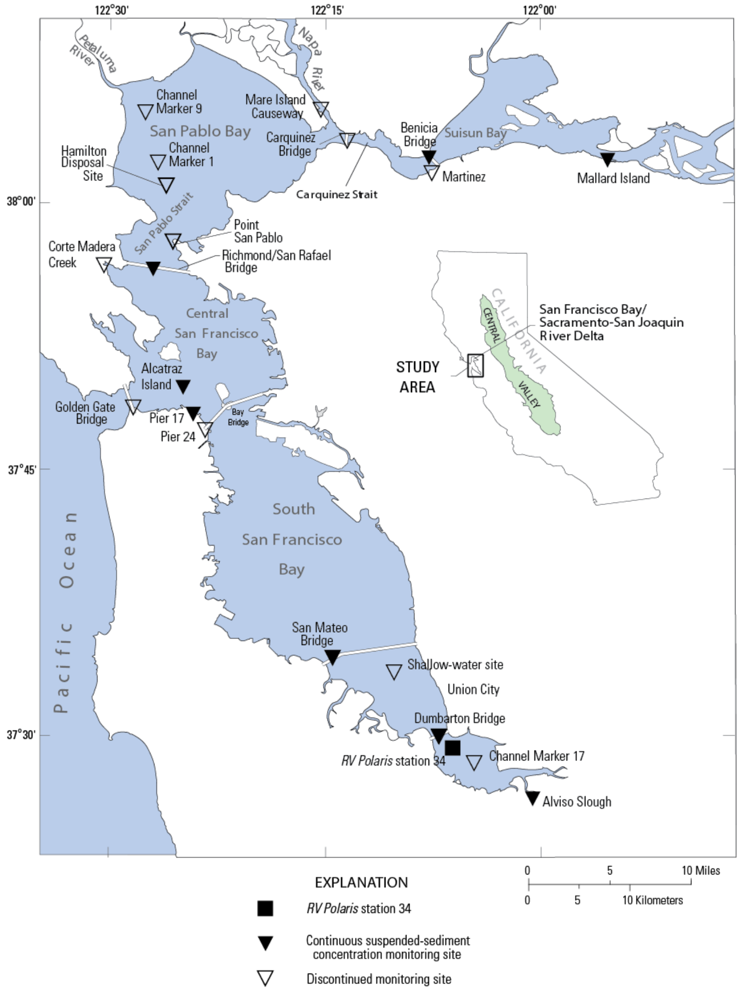

Figure 3.1. Map of San Francisco Bay and continuous suspended-sediment concentration monitoring stations.

3.2 Relation between light attenuation and suspended sediment

Suspended particles like sediment absorb and scatter light, reducing the depth of sunlight penetration in the water column (Cloern, 1996). Utilizing data collected from USGS research vessel cruises near Newark Slough, South San Francisco Bay (*R/V Polaris* station 34, Figure 3.1), between January 2000 and October 2013, we explored the relationship between near-surface suspended particulate matter (SPM) concentration and light extinction coefficient (http://sfbay.wr.usgs.gov/access/wqdata/query/index.html). The extinction coefficient is a measure of the rate at which light is attenuated with depth in a water column. A linear model demonstrated a trend of increasing light extinction with increasing SPM concentration (Figure 3.2) and explained much of the variance (R^2=0.81). Because the growth of phytoplankton near the water surface can affect light extinction, we also analyzed the relationship between near-surface chlorophyll *a* concentration and light extinction coefficient. However, no obvious relationship was observed (Figure 3.3), suggesting that self-shading by phytoplankton was not an important process affecting light attenuation in this region. Total suspended solids concentration, SPM, and SSC are different names for the same quantity in San Francisco Bay (Gray et al. 2000). Because light extinction coefficient and SSC are well correlated, in this chapter we assume that SSC is a surrogate for light extinction coefficient and henceforth we will discuss SSC.

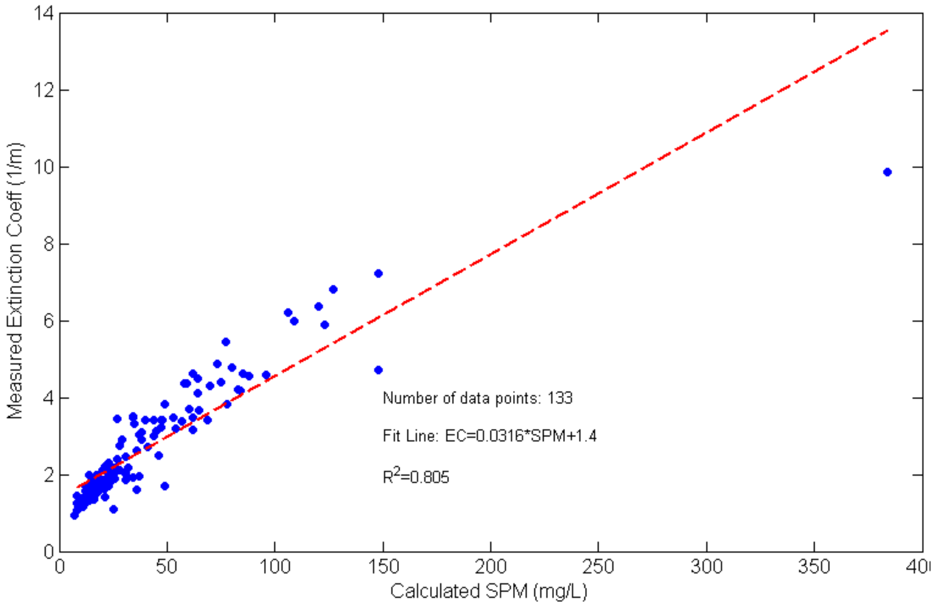

Figure 3.2. Relation between calculated suspended particulate matter (SPM) concentration at 1 m depth and measured light extinction coefficient near Newark Slough (*RV Polaris* station 34) in LSB. Data accessed online 12/17/13.

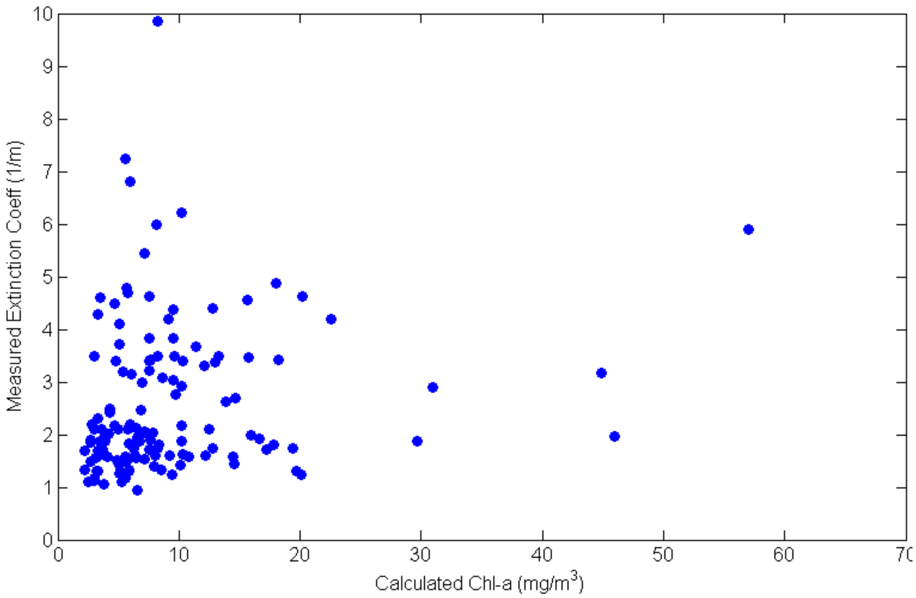

Figure 3.3. Relation between calculated chlorophyll *a* concentration at 1 m depth and measured light extinction coefficient near Newark Slough (Station 34) in South San Francisco Bay. Data accessed online 12/17/13.

3.3 Vertical dynamics

The density of suspended sediment is greater than water, so particles tend to settle downward in the water column and onto the bed. Particles that have settled onto the bed can be resuspended up into the water column by shear stresses from wind waves and tidal currents. Thus, the bed acts as a source and sink of suspended sediment. The extent to which resuspended sediment can be mixed vertically up into the water column may be limited by stratification. In this section we quantify settling of suspended sediment, erodibility of bed sediment, and water column stratification. An understanding of these processes is essential for interpreting SSC observations, predicting spatial and temporal variability in SSC, and estimating light availability and phytoplankton growth rates in water quality models

3.3.1 Flocculation and settling

Suspended sediment in San Francisco Bay is composed almost entirely of fine, cohesive sediment which includes silts and clays. Inter-particle forces attract these cohesive particles to one another and they form flocs composed of the particles and inter-particle voids (Figure 3.4). The size and density of suspended flocs determines their settling velocity, which in turn affects how long suspended particles remain in suspension and the clarity of the water column. Manning and Schoellhamer (2013) conducted a Bay-wide transect from the RV Polaris on June 17, 2008, to measure floc diameter, settling velocity, and density with a floc camera. In this section we summarize the results for the South San Francisco Bay leg of the transect (Figure 3.5).

3.3.1.1 Floc Measurement Methods

Ten nearbed floc populations were sampled within South Bay on June 17th, 2008 using the INSSEV-LF: IN-Situ Settling Velocity instrument. Manning and Schoellhamer (2013) describe the measurement methods in detail and they are summarized here. The LF (LabSFLOC) version of INSSEV is a hybrid system which combines two key components: i) the low intrusive LabSFLOC system, a high resolution video-based device to measure the individual floc properties; ii) an in-situ estuarine floc sampler. The LabSFLOC – Laboratory Spectral Flocculation Characteristics – instrument (Manning, 2006) enables individual floc sizes and settling velocities to be measured simultaneously and was set up on *RV Polaris*. The LabSFLOC camera resolution can practically view flocs down to 10 μm in size. A 2.2L Van Dorn horizontal sampling tube with a 14 kg torpedo-shaped weight suspended from the underside of the tube was used to collect a water sample nominally 0.7 m above the estuary bed. A small sub-sample containing a floc population was carefully extracted from the horizontal Van Dorn using a modified pipette. This sample was immediately transferred to the LabSFLOC settling chamber, whereby the flocs passed from the vertically held pipette to the chamber and settled solely under gravity. Settling velocity and diameter of individual flocs were determined from the video and floc density and porosity were calculated. A diameter of 160 μm is a convenient demarcation between smaller microflocs and larger macroflocs.

Turbulence in the water column is an important factor affecting floc size and thus must be estimated for our floc analysis. Water turbulence during the research cruise was not directly measured, so we used the depth-averaged current speed as a surrogate to estimate turbulence and shear stress parameters in the nearbed region. Depth-averaged current speed (u) at sampling stations was calculated using the UnTRIM three-dimensional numerical model which was calibrated to data from San Francisco Bay by MacWilliams et al. (2008, also discussed by Kimmerer et al. 2009). The model was run with a 3-minute time step through June 17th 2008, and vertical velocity profiles and u at the locations of cruise sampling were extracted. The nearbed frictional (shear) velocity (U*) was calculated from u using a formula offered by Delo (1988), and used to determine values of average local shear stress (τ). Appropriate values of Manning's bed roughness coefficient were chosen based on Cheng et al. (1993). This procedure neglects turbulence damping by vertical stratification.

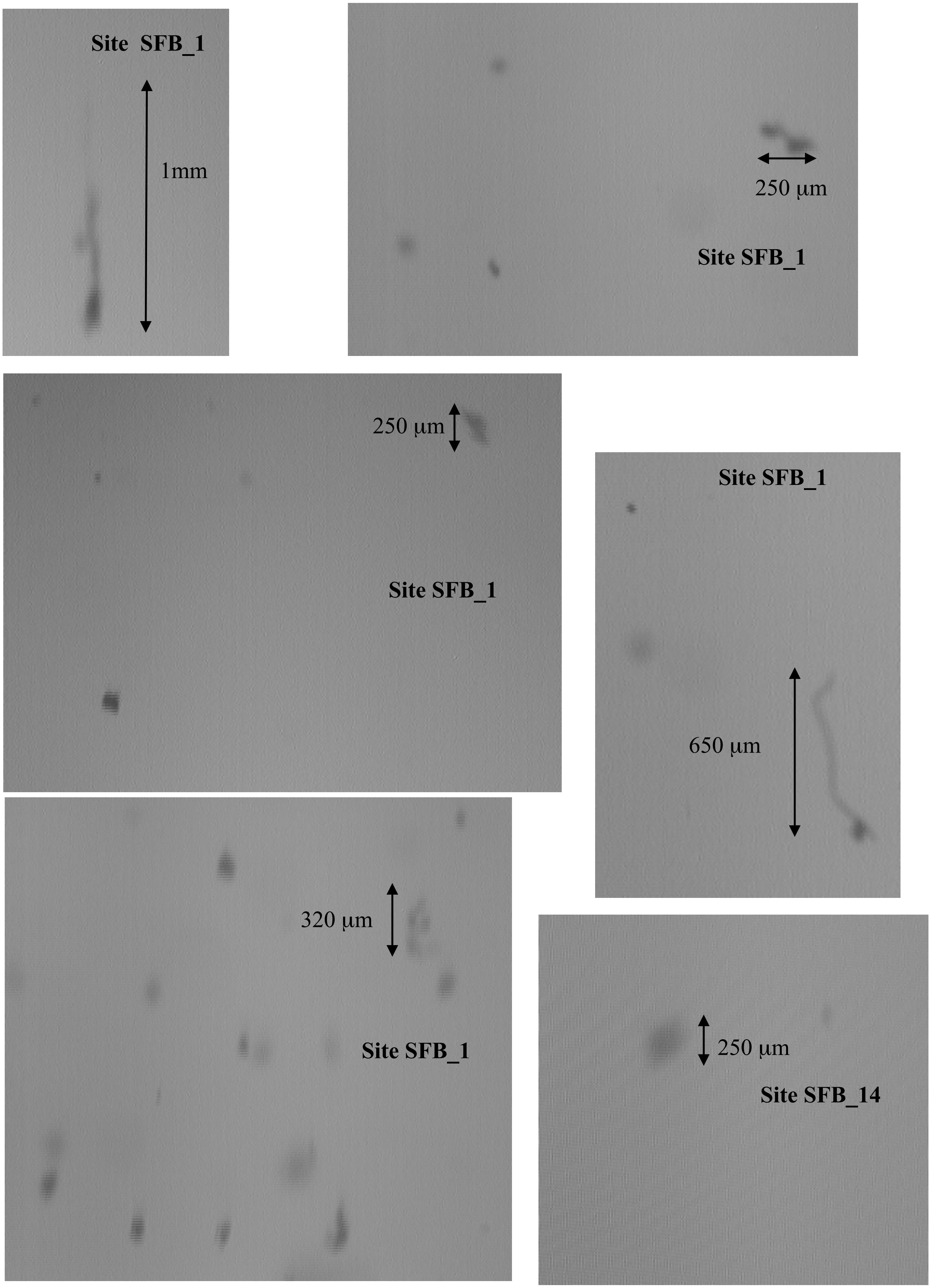

Figure 3.4. Examples of various South Bay floc types observed (as sillouetted images) by the LabSFLOC video camera at site SFB_1 (Newark Slough) and SFB_14 (Potrero Point).

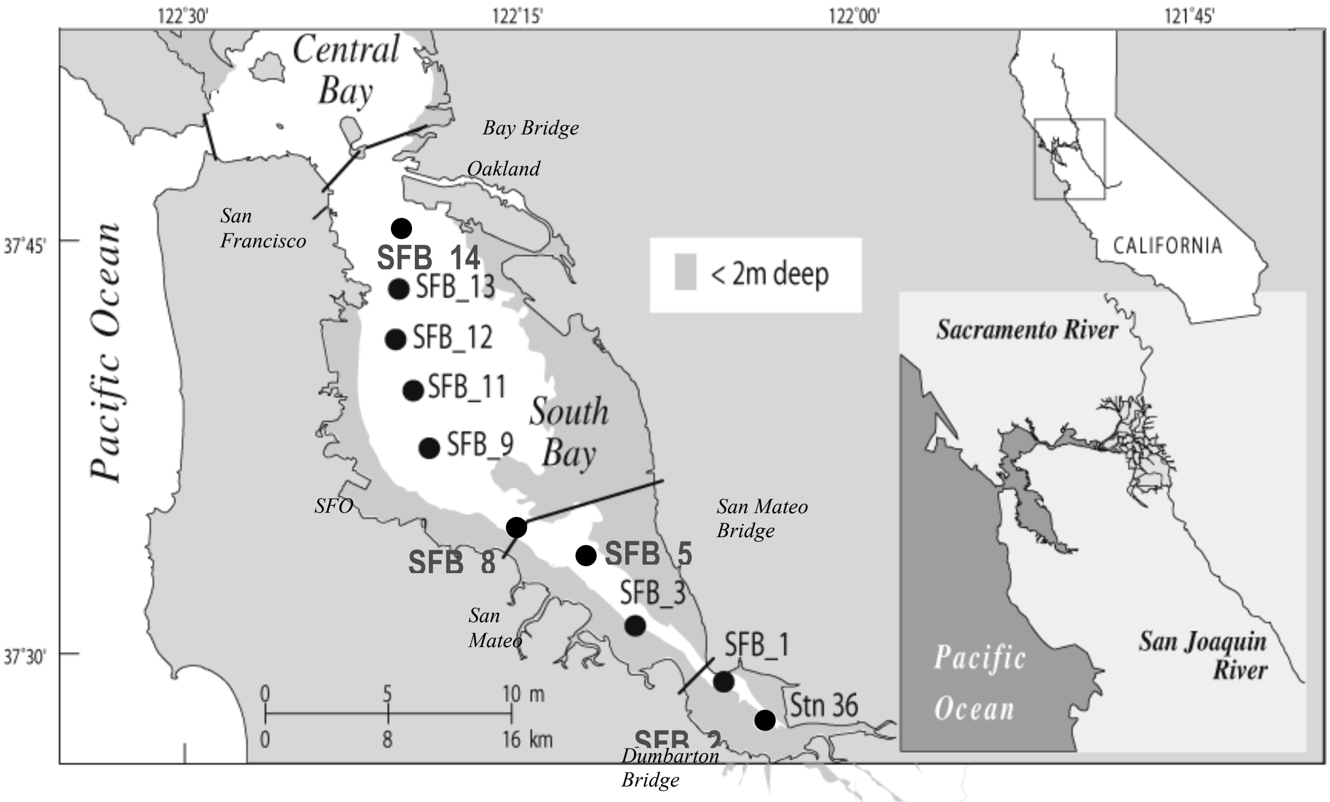

Figure 3.5. South Bay indicating the 10 floc sampling stations. All transect distances in table 1 are referenced from *RV Polaris* Station 36 (shown on map as Stn 36).

3.3.1.2 Floc Observation Results

If we consider the average floc characteristics from all ten South Bay populations, the flocs had a mean settling velocity (Ws_{mean_ALL}) of 2.66 mm/s and a mean floc diameter (D_{mean_ALL}) of 115 µm (Table 3.1). Macroflocs comprised 26.3% of the total population by number but 29.7% by mass. An even larger percentage, 56.4% of the mass settling flux (MSF, 1180 mg/m^2/s) was due to macroflocs because of their higher settling velocity, $Ws_{Macro\_}$ of 2.72 mm/s.

Station	all	SFB_1	SFB_3	SFB_9	SFB_14
Distance from station 36 (km)	-	3.5	8.2	26.2	43.9
SSC (mg/L)	19-230	230	20	30	47
Shear stress τ (Pa)	0.01-0.58	0.24	0.03	0.11	0.58
Velocity u (m/s)	0.04-0.82	0.37	0.13	0.27	0.80
Number of flocs	1613	522	75	97	135
Number of macroflocs	425	299	0	19	24
D_{mean} (µm)	115	170	81	114	112
W_{smean} (mm/s)	2.66	4.4	0.5	0.61	6.4
Macroflocs (% by mass)	36.7	61	0	62	30.7
Mass setting flux (mg/m^2/s)	2100	1100	11	18	293

Table 3.1. Environmental variables and floc population properties, South San Francisco Bay, June 17, 2008

We observed an abundance of fine-grained cohesive sediment at the southern station SFB_1. The floc population from this site demonstrates that smaller flocs are denser than larger flocs. The effective density (ρ_e) of the flocs less than 100 μm in diameter were all greater than 160 kg/m^3 while ρ_e for macroflocs larger than 300 μm were all less than 160 kg/m^3. In fact the eighteen flocs larger than 500 μm, just 3.5% of the total population, predominantly demonstrated $\rho_e < 30$ kg/m^3 (porosity > 98.5%). Thus, this large size fraction represented over 10% of both the ambient SSC and total MSF.

Floc sample SFB_3 was acquired at Ravenswood Point where lower SSC and less turbulent environment near slack tide reduced the potential for constructive floc inter-particle collisions, and this was reflected by a small D_{mean} of 81 μm (Table 3.1). Similarly the Ws_{mean} reduced to 0.55 mm/s. No macroflocs were present. The SFB_3 MSF was two orders of magnitude lower than at SFB_1.

North of San Mateo Bridge near San Francisco International Airport the aquatic environment became more turbulent during this cruise. Flocs were larger than those present in the very quiescent conditions of Ravenswood Point (SFB_3). The SFB_9 macrofloc fraction had a mean effective density of just 27 kg/m^3 and were over 98% porous, which suggests they were extremely fragile and possibly primarily organic in composition. Macrofloc density at SFB_1 and SFB_9 were 164 and 27 kg/m^3 while microfloc densities were 851 and 239 kg/m^3. These differences suggest that the macrofloc and microflocs may have different compositional matrices at different locations throughout South Bay.

On reaching the northern region of South Bay at Potrero Point (SFB_14), SSC increased and turbulence was greatest. The high turbulence likely limited floc growth, however, the flocs had the highest settling velocities within South Bay (Table 3.1).

If we attempt to parameterize the floc settling behavior within South Bay in order to produce estimates of MSF for modeling purposes, a constant Ws of 0.5 mm·s^{-1} (i.e. a value typically employed for cohesive sediment settling parameterization in numerical modeling) only provided realistic flux estimates when fluxes were under 20 mg.m^{-2}s^{-1}, but still underpredicted the South Bay flux at those locations by 10-14%. Contrastingly, in the northern region of South Bay (SFB_13 and SFB_14) and the southern most station (SFB_01), a constant Ws of 0.5 mm·s^{-1} underpredicted the MSF at each station by more than an order of magnitude. As a whole, the 0.5 mm·s^{-1} Ws parameter under-estimates depositional flux by a mean of 14.5%, and a range of 8% to 91% at the South Bay stations. Thus this demonstrates that a data set that includes distributions of floc size, settling velocity, and hydrodynamics in time and space is a significant improvement over a more simplified 'guess' of the settling velocity for accurate applied modeling purposes.

Collectively, all floc populations observed within South Bay demonstrated a fairly wide range in Ws for a constant D (Figure 3.6), and similarly a varying span in D for a constant Ws. This indicates the influence of varying floc effective density and its effect on mass and MSF. Thus it is extremely importance to use an instrument (such as the INSSEV-LF) which is capable of measuring the variability of Ws and the relation of density with D and Ws.

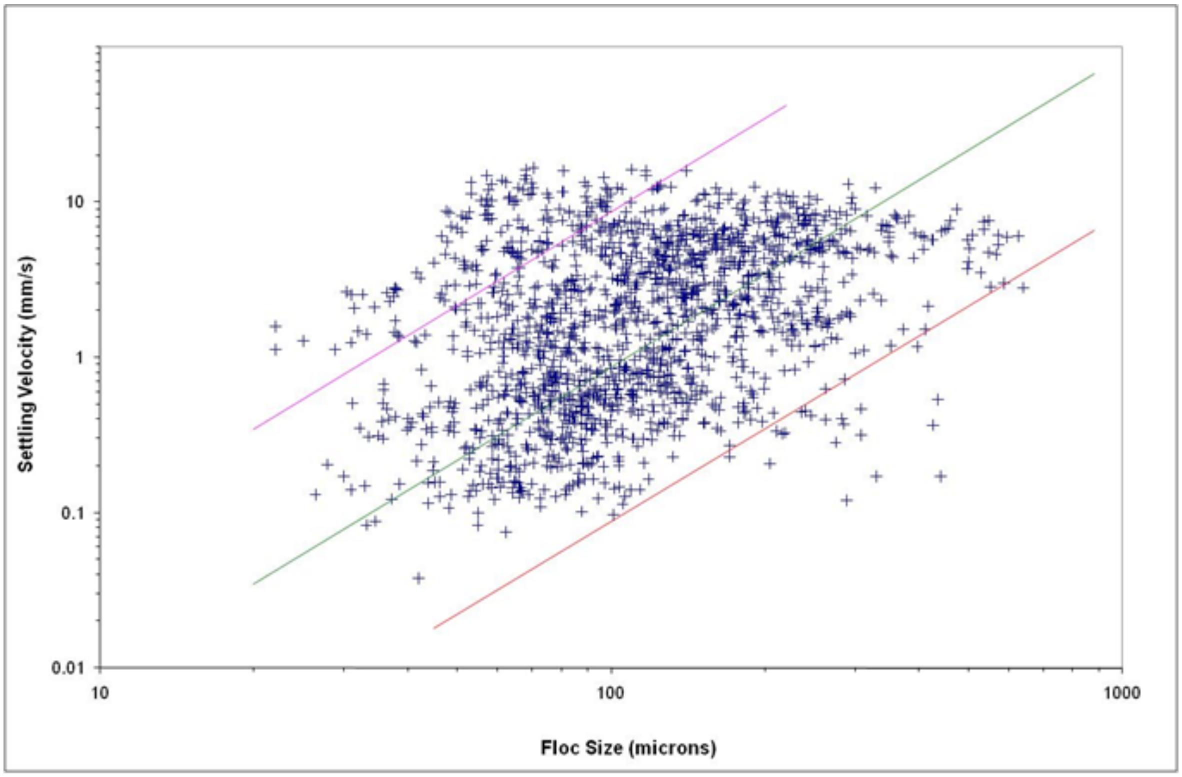

Figure 3.6. Combined distribution of settling velocity and floc size for all ten samples SFB_01 to SFB_14 (acquired nominally 0.7 m above the estuary bed). The diagonal lines on the Ws vs. D scatterplot represent contours of constant Stokes equivalent effective density: the upper pink line is 1600 kg/m^3, the middle green line is 160 kg/m^3, and the lower red line is 16 kg/m^3.

3.3.2 Erodibility

Only recently have there been studies of the erosion of South Bay bed sediment. Brand et al. (2010) measured SSC south of the San Mateo Bridge and observed elevated resuspension during calm periods after wind-induced resuspension, which likely indicates that newly deposited sediment is more erodible than sediment that has rested on the bed long enough to consolidate. They also measured vertical sediment flux at a site south of San Mateo Bridge and estimated the critical shear stress for sediment resuspension to be about 0.1 Pa.

Jones and Jaffe (2013) collected sediment cores from four sites in South Bay, two of which were from sites with relatively low hydrodynamic energy, and thus erosional force, south of the Dumbarton Bridge. The cores were placed in the bottom of a laboratory flume (Sedflume), exposed to water flow, and eroded under these controlled conditions. Erosion properties are measured down to about 20 cm depth within the core. They found that the two sites south of the Dumbarton Bridge with lower energy allowed fine silts to settle and incorporate into the bed sediment, increasing the strength of the bed relative to the sites north of the Bridge. In addition, south of Dumbarton Bridge, one of the sites was intertidal and experienced cycles of wetting and drying while the other was subtidal and always submerged. The intertidal site had lower water content and greater strength than the subtidal site.

Schoellhamer (2013) collected two cores at a site south of the Dumbarton Bridge and eroded them in a University of Maryland – Gust Erosion Microcosm System (UGEMS, Dickhudt et al. 2011). UGEMS applies much smaller shear stresses than Sedflume and applies them only to the bed surface. Whereas Sedflume determines erosion properties down many centimeters into the core, UGEMS only erodes less than one millimeter of sediment. Preliminary results are that about 0.1 kg/m^2 was eroded from both cores which would increase SSC about 100 mg/L in 1 m of water, which is typical variability due to tides and wind waves at the site. Critical shear stress at the sediment/water interface was about 0.075 Pa, similar to the estimate of 0.1 Pa by Brand et al. (2010), and increased with depth. Erosion rate (mass per unit area per unit time) is the product of the excess shear stress (shear stress minus critical shear stress) and an empirical erosion rate constant (Dickhudt et al. 2011). The erosion rate constant increased with depth from about $0.5*10^{-4}$ to $3*10^{-4}$ kg/m^2/Pa/s possibly because the solids fraction increased or biofilm stabilized the sediment surface.

Given the importance of resuspension of bed sediments on SSC and thus phytoplankton growth, more measurements of erosion properties of South Bay bed sediments are needed to assess spatial and temporal variations. To assess these variations on the appropriate time scale (i.e., tidal), the UGEMS methodology should be utilized.

3.3.3 Stratification

Density stratification of the water column is an important process that affects mixing in estuaries, and is a fundamental control on primary production (e.g., Cloern, 1996). Vertical density stratification is caused by processes including tidal straining, surface heating, and freshwater inputs and occurs at various time scales from tidal to seasonal (Simpson et al., 1990). Stable density stratification, where a higher density water mass underlies a lower density water mass, inhibits vertical mixing between the water masses, which affects the distribution of dissolved and suspended constituents (Simpson et al., 1990). Such stratification can lead to phytoplankton blooms by trapping phytoplankton in the upper water column, where irradiance is higher and benthic grazing is inhibited (Cloern, 1996). In the deeper channels of South San Francisco Bay, stratification is strongest during winter storms when freshwater inflow is greatest, giving rise to a recurring spring bloom of phytoplankton (Cloern, 1996). In the shallower shoals, vertical density stratification caused by a vertical gradient in SSC has been observed during strong summer wind events (Lacy et al., 2014).

We used continuous sensors at Dumbarton Bridge (USGS station 373015122071000) to assess vertical stratification effects on SSC, with the hypothesis that stratification episodes would cause lower SSC in the upper water column. Classic estuarine physics theory predicts that estuaries are well mixed on flood tides and become stratified on ebb tides (e.g., Simpson et al., 1990). This is evident at Dumbarton Bridge during 2 Jun-13 Jun 2011, where strongest stratification (indicated by highest magnitude of bottom minus top salinity difference) was observed on late ebb ($dQ/dt < 0$ and $2000 > Q > 0$) and early flood ($dQ/dt < 0$ and $-4000 < Q < 0$), with weak stratification at all other times (Figure 3.7). We observed highest magnitudes of SSC at mid-depth and near-bottom sensors near the end of the stronger daily ebb tide (Figure 8), indicating advection of a sediment mass from LSB. Approaching slack after ebb, SSC at both sensors decreased, likely due to particle settling in low water velocities. In most cases, there was a second peak in SSC at the lower sensor during early flood, suggesting resuspension of the sediment mass and advection back past the sensor. This increase in SSC was not seen at the upper sensor; since this coincides with a period of increased stratification (Figures 3.8 and 3.9), it possibly indicates that stratification has greater effect than tidal shear and suppresses the vertical mixing of sediment in the upper water column. In addition, during this period we observed local SSC maxima at both near-bottom and mid-depth sensors during peak flood tide, indicating resuspension at peak tidal velocity. In general, the magnitude of the SSC peak caused by resuspension during mid-flood tide was smaller than that caused by advection during late ebb tides. The magnitude of SSC appeared to be affected by the spring-neap cycle, with higher SSC during spring tides, possibly due to increased velocity and associated resuspension and increased tidal excursion causing greater transport of sediment sourced from the estuary perimeter.

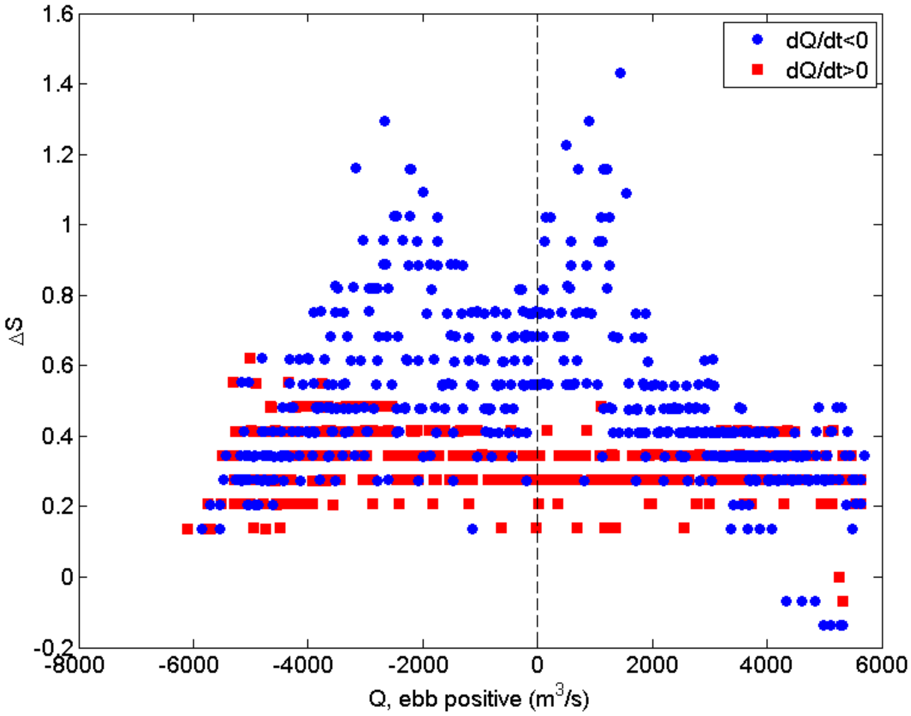

Figure 3.7. Scatter plot of discharge (Q) versus bottom-top salinity difference (ΔS) at Dumbarton Bridge during 2 - 13 June 2011. Stratification is greatest during late ebb and early flood. The lower sensor is located near the bottom (1.2 meters above the bed) and the upper sensor is located near mid-depth (7.6 meters above the bed). For the blue dots (dQ/dt<0) time progresses from right to left beginning at maximum ebb and ending at maximum flood. For the red dots (dQ/dt>0) time progresses from left to right, beginning at maximum flood and ending at maximum ebb.

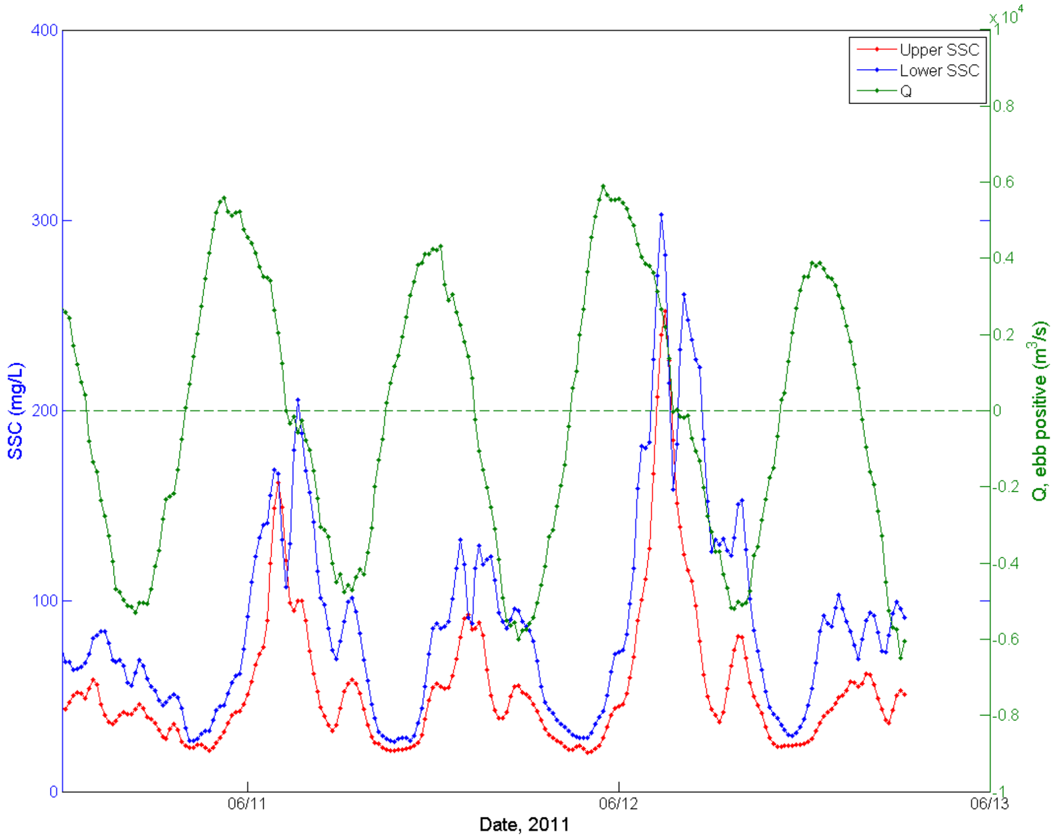

Figure 3.8. Time series of suspended-sediment concentrations (SSC) and discharge (Q, ebb positive) at Dumbarton Bridge during 10-13 June 2011. Highest SSC at the lower sensor is observed during the late phase of stronger daily ebb and subsequent early flood, with a temporary decrease at slack tide; this is indicative of advection of a sediment mass past the sensor. Highest SSC at the upper sensor is observed at late ebb; however the elevated SSC does not continue into early flood tide, when stratification is elevated (Figure 9). There is a secondary peak in SSC at upper and lower sensors at maximum flood discharge, indicative of resuspension at maximum tidal velocity. The lower sensor is located near the bottom (1.2 meters above the bed) and the upper sensor is located near mid-depth (7.6 meters above the bed).

Although we observed weak stratification and possible corresponding reduction of SSC in the upper water column using near-bottom and mid-depth sensors, a sensor located at the water surface would be more useful to test this hypothesis. A sensor has been deployed near the surface to better identify stratification in the upper water column. To address this lack of data in the upper water column at the time of this analysis, we attempted to utilize vertical profiles of salinity and SSC for the entire water column, which were collected from the *RV Polaris* at Station 34 between 2000 and 2014. However, in general the measurements are taken near slack after flood or early ebb tide due to water depth restrictions on the vessel, giving rise to a tidal bias in sampling. As explained previously and seen in Dumbarton Bridge data, estuaries tend to be well mixed on flood tides; thus for samples taken from the *RV Polaris*, we would expect small differences in water density between top and bottom. This is the case, as demonstrated in the histogram of bottom minus top density for all cruises during 2000-2014 (Figure 3.10). Thus, existing data make it difficult to quantify the effect of density stratification in the upper water column for the channels of South Bay, however, continuous data from moored sensors at Dumbarton Bridge suggest stratification should have an effect. Our understanding of this phenomenon could be improved by vertical profiles taken at all phases of the tide from a smaller vessel, or by continuous observations from moored sensors located both near bed and near surface.

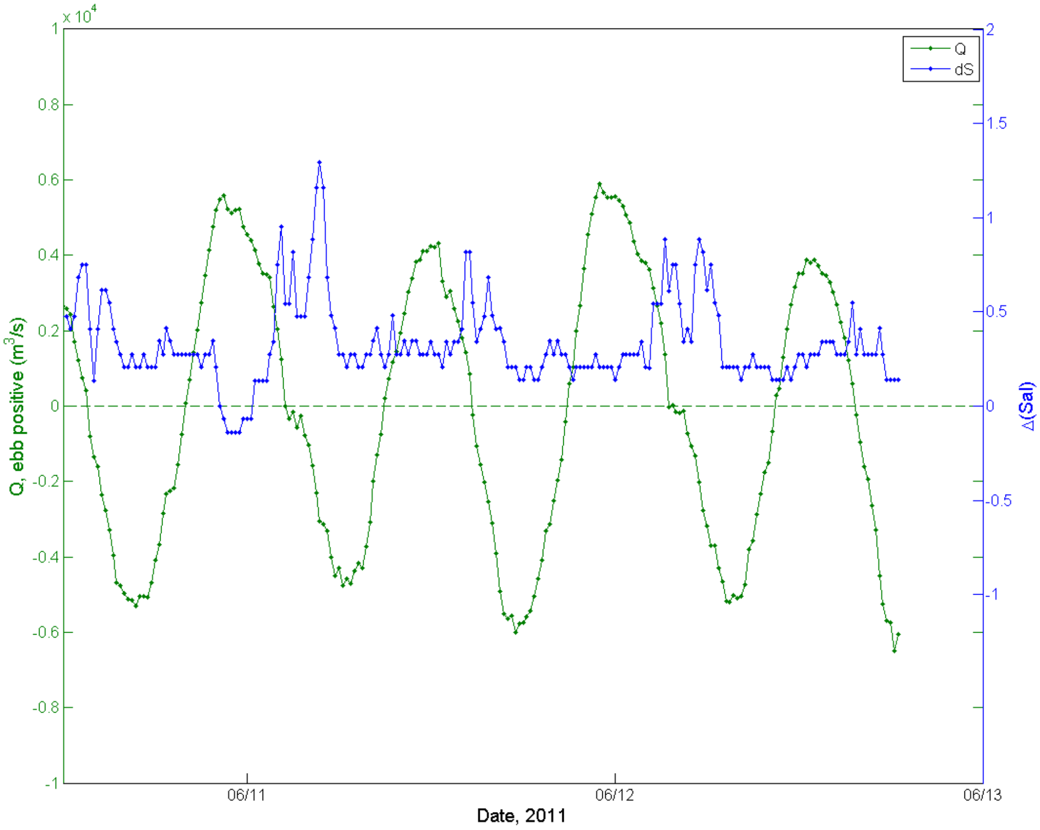

Figure 3.9. Time series of discharge (Q, ebb positive) and bottom-top salinity difference (ΔS) at Dumbarton Bridge during 10-13 June 2011. Highest ΔS is observed during the late phase of each ebb tide and subsequent early flood, indicating strongest stratification during these periods. The lower sensor is located near the bottom (1.2 meters above the bed) and the upper sensor is located near mid-depth (7.6 meters above the bed).

3.4 Time scales of suspended sediment variability and forcing mechanisms

SSC in San Francisco Bay varies over different time scales (Schoellhamer 1996, 2002). SSC generally increases as tidal velocities and wind generated waves increase. Semidiurnal and diurnal tides vary in strength over the semimonthly spring/neap, lunar month, and semiannual solstice/equinox cycles. Wind varies diurnally and seasonally. In addition, SSC decreases as the estuary adjusts to decreasing sediment supply (Schoellhamer et al., 2013). In this section, time series of mid-depth SSC at the Dumbarton Bridge collected at a 15 minute interval from 1992-2011 (Buchanan and Morgan, 2012, Figure 3.11) will be analyzed to determine the time scales of variability and associated forcing mechanisms. Data collection at the Dumbarton Bridge began in October 1992 and temporarily stopped in September 2011 because the data collection station had to be removed from the Bridge to allow seismic repairs. Fifty-two percent of the potential data are valid during this 19 year period. Invalid data are mostly due to biological fouling of the optical instruments used to measure SSC. Data collection was temporarily moved from the highway bridge to the railroad bridge until April 2013 when the original station was redeployed. Analyses are specific to the Dumbarton Bridge and utilize the most recent data available, which are the longest SSC station period of record available in San Francisco Bay.

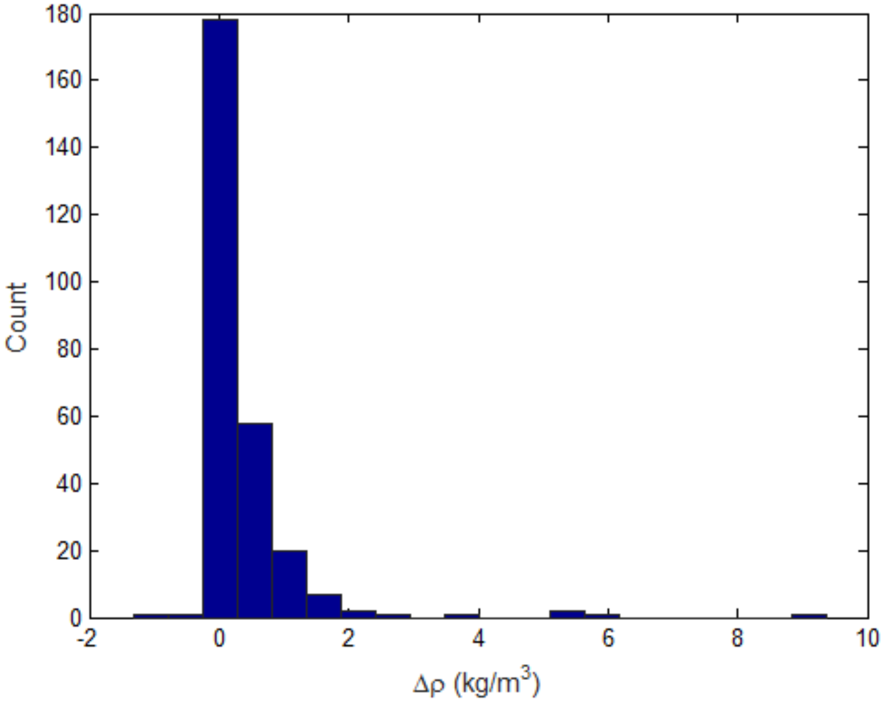

Figure 3.10. Histogram of bottom-top density difference (Δρ) for *R/V Polaris*, St. 34 during 2000-2014. Density difference was calculated as bottom minus top for each vertical profile.

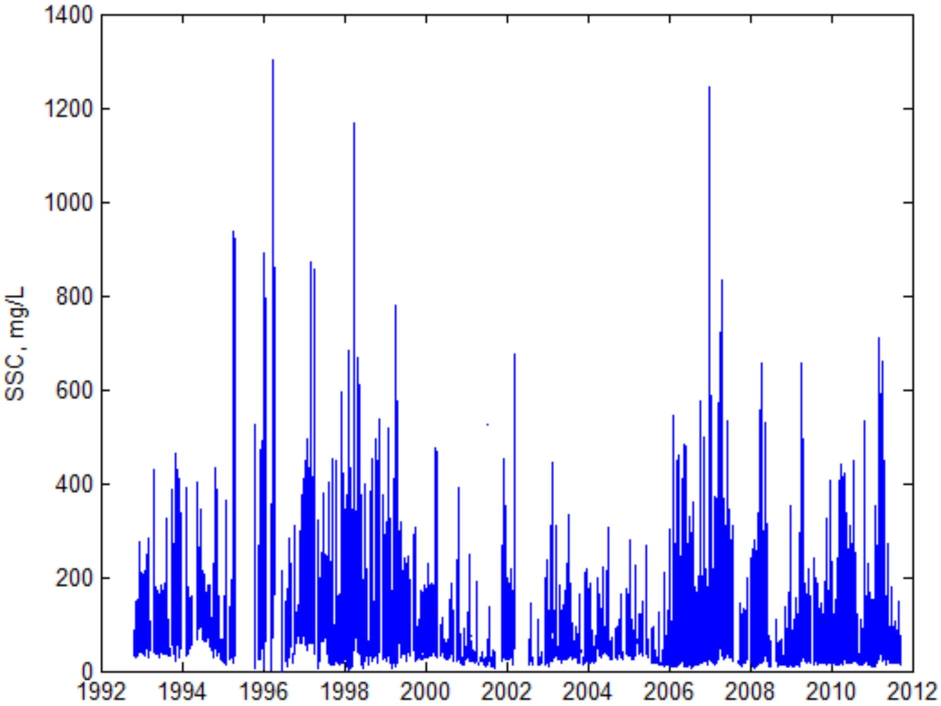

Figure 3.11. Suspended-sediment concentration at Dumbarton Bridge, South San Francisco Bay during 1992-2011. Sensor is located near mid-depth (7.6 meters above the bottom).

3.4.1 SSC components with time scales from tidal to annual

To quantify the time scales of variability of SSC at mid-depth at the Dumbarton Bridge, singular spectrum analysis for time series with missing data (SSAM, Schoellhamer, 2001) was used to reconstruct components of the 19-year time series. SSAM was applied in a sequential manner to calculate reconstructed components with time scales of variability that ranged from tidal to annual as described in Schoellhamer (2002). The resulting time scales can be associated with physical forcing. Reconstructed components are pseudo-periodic. Schoellhamer (2002) similarly applied SSAM to interpret a 6-year SSC time series from Point San Pablo in Central San Francisco Bay (USGS Station 11181360); more methodological details can be found there. South Bay experiences more biofouling than Central Bay so the fraction of valid data at Dumbarton Bridge (52.0%) was less than at Point San Pablo (73.5%). SSAM uses a data window for which a user specified fraction of data must be valid. In previous applications of SSAM, a threshold (f) of at least half (50%) of the data in a window being valid was applied. When applied to the Dumbarton Bridge data, a 50% threshold limited identification of longer periods of variability. For this analysis, a threshold of f= 1/12 (8.333%) was used. Data are collected every 15 minutes, so this threshold is equivalent to having one valid data point every 3 hours. A low threshold (f=1/12 in this case) enables SSAM to act as a data interpolator that fills missing values in time series while a high threshold (f=0.50 in this case) reduces the number of valid values in the results, especially for longer periods. The two thresholds produced virtually the same results for up to monthly periods and the f=1/12 threshold provided information on longer periods.

Physical processes that controlled SSC and their contribution to the total variance of SSC were (A) quarter diurnal tidal resuspension and deposition (6%), (B) semidiurnal tides (13%), (C) diurnal tides (7%), (D) other higher frequency tidal constituents (8%), (E) semimonthly tidal cycles (11%), (F) monthly tidal cycles (15%), (G) annual cycle (4%), and (H) subannual variation with a periodicity greater than 2.14 years (781.25 days) (10%) (Figure 3.12). Of the total variance 74% was explained and subtidal variability (time scales greater than tidal, 40%) was greater than tidal variability (34%). Processes at subtidal time scales accounted for more variance of SSC than processes at tidal time scales because sediment accumulated in the water column and the supply of easily erodible bed sediment increased during periods of increased subtidal energy. In other words, sediment accumulates in the water column and the supply of erodible sediment increases during periods of greater tidal energy such as spring tides. A semiannual cycle was found for the Point San Pablo analysis (Schoellhamer, 2002) but could not be clearly identified at Dumbarton Bridge. The annual cycle had peak SSC in April or May 61% of the years (Figure 3.13) due to high mean wind speeds (Figure 3.14) and greater availability of erodible bed sediment during these months (Schoellhamer, 1996). In addition, later it will be shown that April is the month with greatest SSC at Dumbarton Bridge and we will present a hypothesis that the spring phytoplankton bloom may also play a role. The annual cycle had minimum SSC in October or November 61% of the years (Figure 3.15) due to low wind speed, weak tides, and winnowing of fine sediment from the bed during the summer dry season (Schoellhamer, 1996). Later it will be shown that September is the month with smallest SSC at Dumbarton Bridge and autumn is the season with the smallest SSC. The subannual reconstructed component is similar to but less noisy than a running mean with a 2.14 year window (Figure 3.16). This subannual time series shows the 2000 step decrease in SSC, somewhat smeared due to the large window. Increased SSC in 2007 may be due to 2006 having the largest sediment supply from the watershed in the 2000s (McKee et al., 2013) and subsequent resuspension of that sediment.

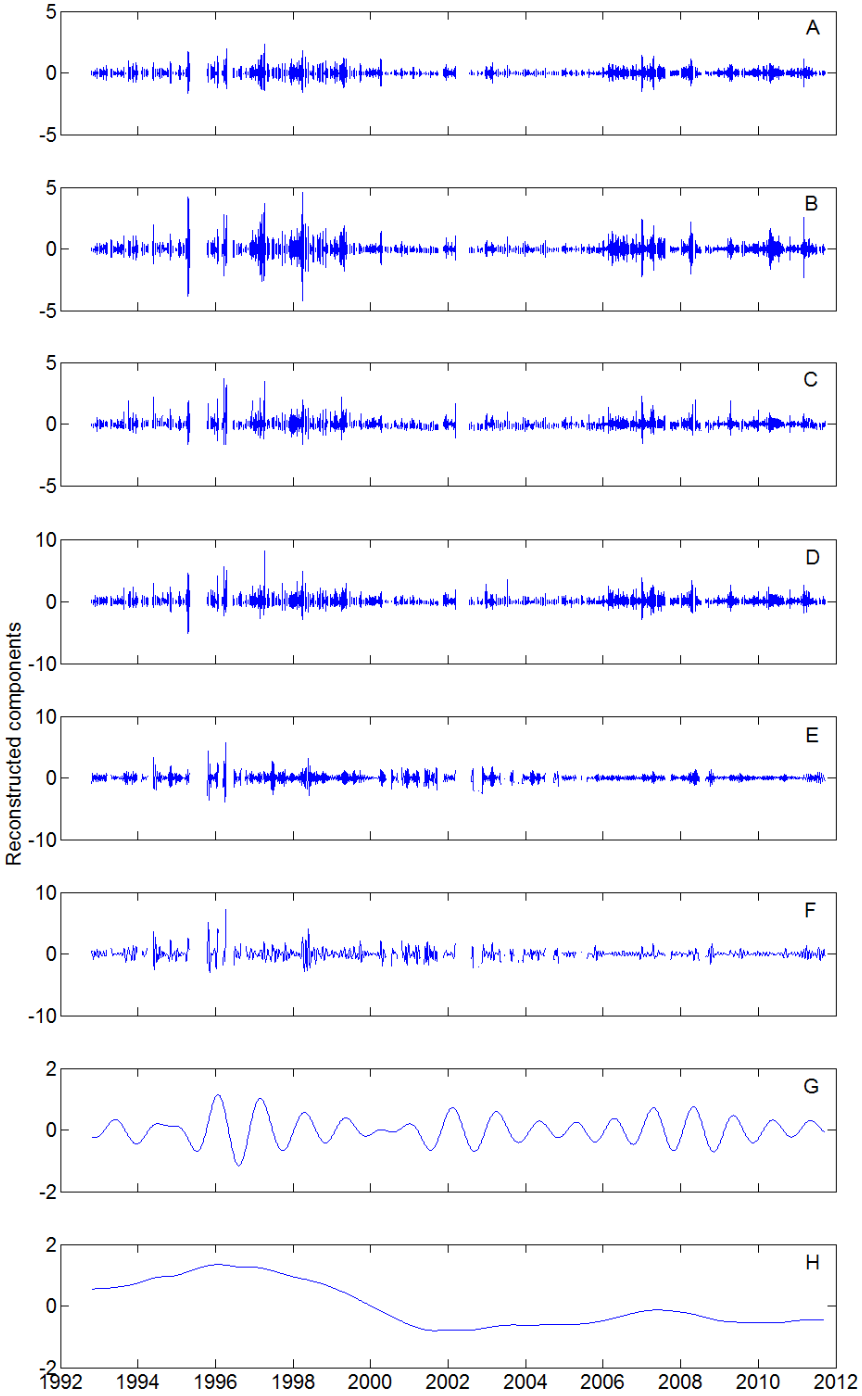

Figure 3.12. Reconstructed components from sequential Singular Spectrum Analysis for time series with missing data applied to Dumbarton Bridge SSC at mid-depth. The following signals were identified: A) quarter diurnal (6% of total variance), B) semidiurnal (13%), C) diurnal (7%), D) other higher frequency tidal constituents 8%), E) semimonthly (11%), F) monthly (15%), G) annual (4%), and H) subannual (10%). Reconstructed components are dimensionless.

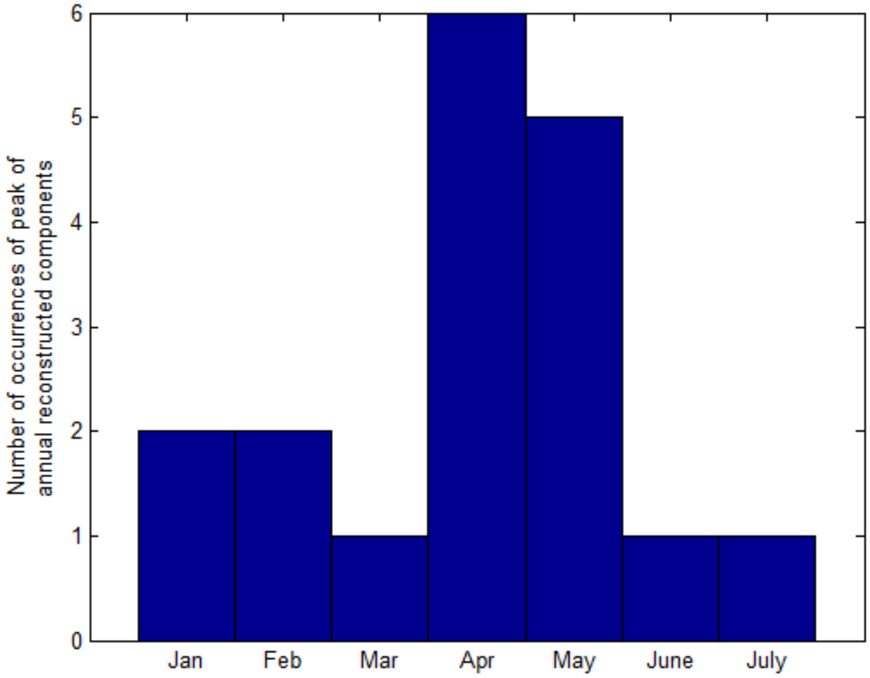

Figure 3.13. Number of occurrences by month of local maxima (peaks) of the annual reconstructed components of mid-depth SSC at the Dumbarton Bridge.

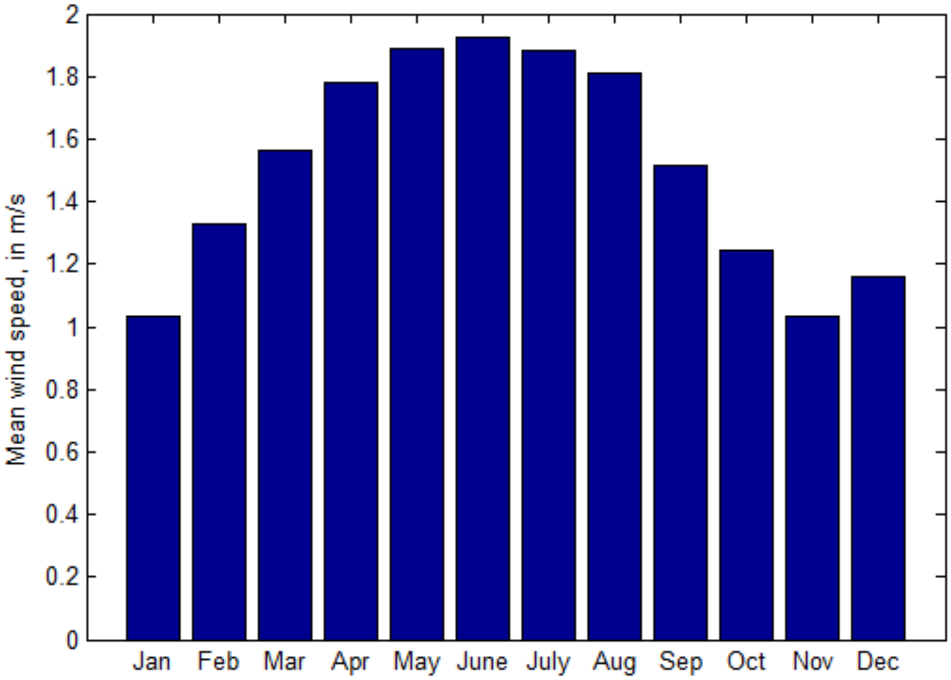

Figure 3.14. Monthly mean wind speed, Union City, February 2001-September 2011. Data from California Irrigation Management Information System, http://wwwcimis.water.ca.gov/cimis/

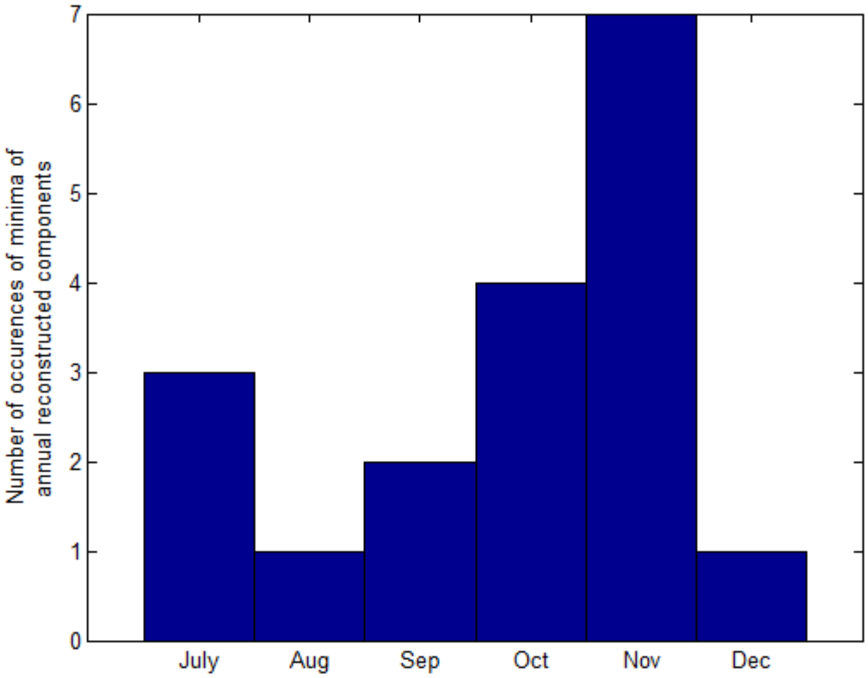

Figure 3.15. Number of occurrences by month of local minima of the annual reconstructed components of mid-depth SSC at the Dumbarton Bridge.

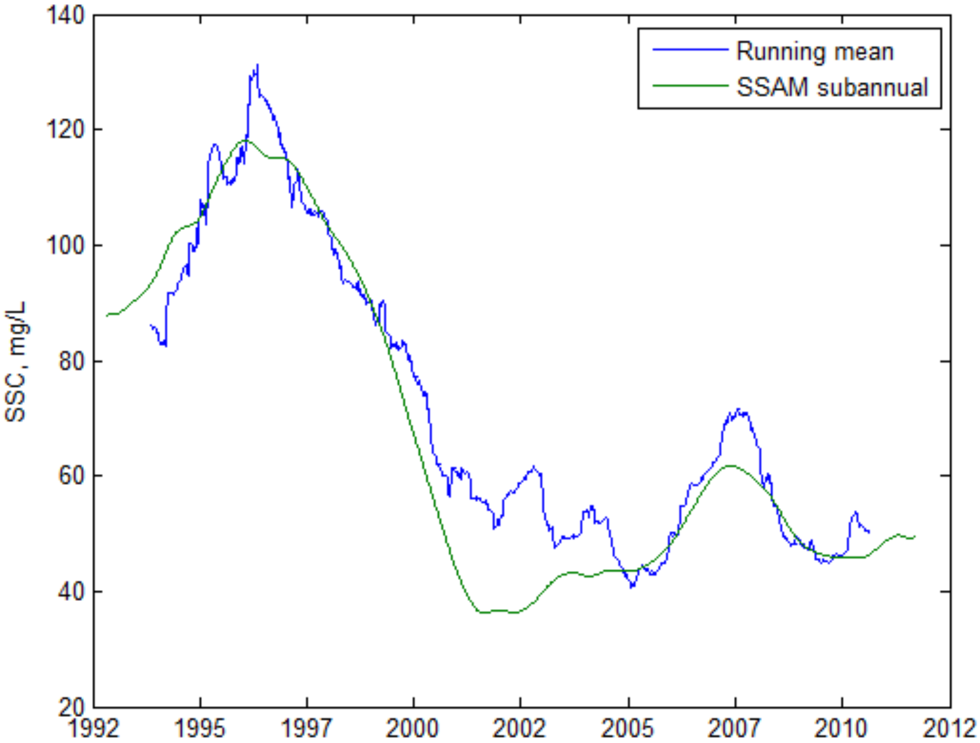

Figure 3.16. Running mean mid-depth SSC at Dumbarton Bridge calculated with a 2.14 year (781.25 day) window and SSAM dimensional subannual reconstructed component. The dimensionless reconstructed component in Figure 12H was converted to its dimensional form and the mean SSC (68.4 mg/L) added to make it comparable to the running mean.

3.4.2 SSC step decrease

Water year median SSC at mid-depth at the Dumbarton Bridge has decreased from the 1990s to the 2000s (Figures 3.16 and 3.17). Water years start October 1 and end September 30. From 1993 to 1999 median SSC was 80 mg/L and from 2000-2011 median SSC was 41 mg/L. These values update those presented by Schoellhamer (2011), who hypothesized that a Bay wide step decrease in SSC in 1999 was caused when the Bay crossed a threshold from transport to supply limitation of sediment transport. The running mean and SSAM dimensional subannual reconstructed component have a similar but more gradual decrease from 1997 to 2001 because of the 2.14 year long data window (Figure 3.16).

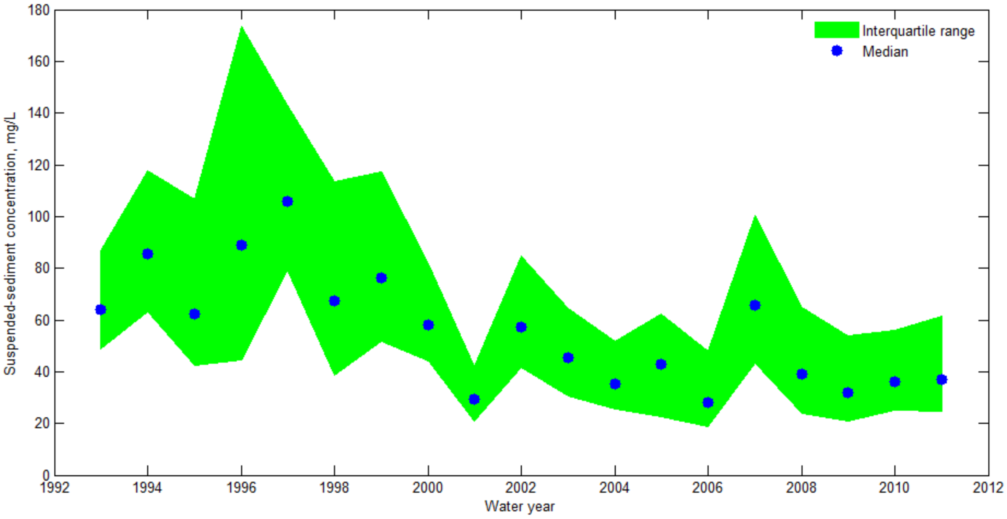

Figure 3.17. Water year median suspended-sediment concentration (SSC) and interquartile range, mid-depth, Dumbarton Bridge.

Monthly medians for the 1992-1999 and 2000-2011 periods indicate that the greatest percentage decrease in SSC was August – November (58-64%, Table 3.2). In addition, minimum median SSC occurs in September. Maximum SSC occurs in April, similar to the S
SAM annual reconstructed component (Figure 3.16).

Table 3.2. Monthly median suspended-sediment concentration (mg/L), mid-depth, Dumbarton Bridge.

	1992-1999	2000-2011	Percentage decrease
January	76	41	46%
February	65	38	42%
March	82	52	37%
April	107	63	41%
May	83	47	43%
June	81	45	44%
July	78	43	45%
August	80	33	59%
September	69	25	64%
October	80	33	59%
November	78	33	58%
December	91	44	52%

3.4.3 Hypothesis: Spring phytoplankton bloom increases SSC

An annual phytoplankton bloom may increase SSC in South San Francisco Bay. A predictable spring phytoplankton bloom occurs following periods of strong vertical salinity stratification in the water column (Cloern, 1996). The peak of the bloom is indicated by maximum chlorophyll *a* concentration which typically is observed late March or early April during weekly to monthly cruises of the USGS *RV Polaris* (Figure 3.18). The annual maximum of SSC typically is during the spring tide following the end of the spring phytoplankton bloom, typically mid-April. Ruhl and Schoellhamer (2001) present an example for water year 1998. The greatest SSC at Dumbarton Bridge occurs in April (Table 3.2) and after the observed maximum chlorophyll-a concentration (Figure 3.19). Possible explanations include increased erodibility of bottom sediment, bioturbation from benthos or birds feeding on the bloom detritus, and increasing wind speed and wind waves from March to May. Wind speed is greater in May than April (Figure 3.14), but SSC is less in May than April, so the erodibility of the bottom sediment must be greater in April than May. Winnowing of fine sediment is one possible explanation. We hypothesize that increased erodibility in April triggered by the phytoplankton bloom is another possible explanation.

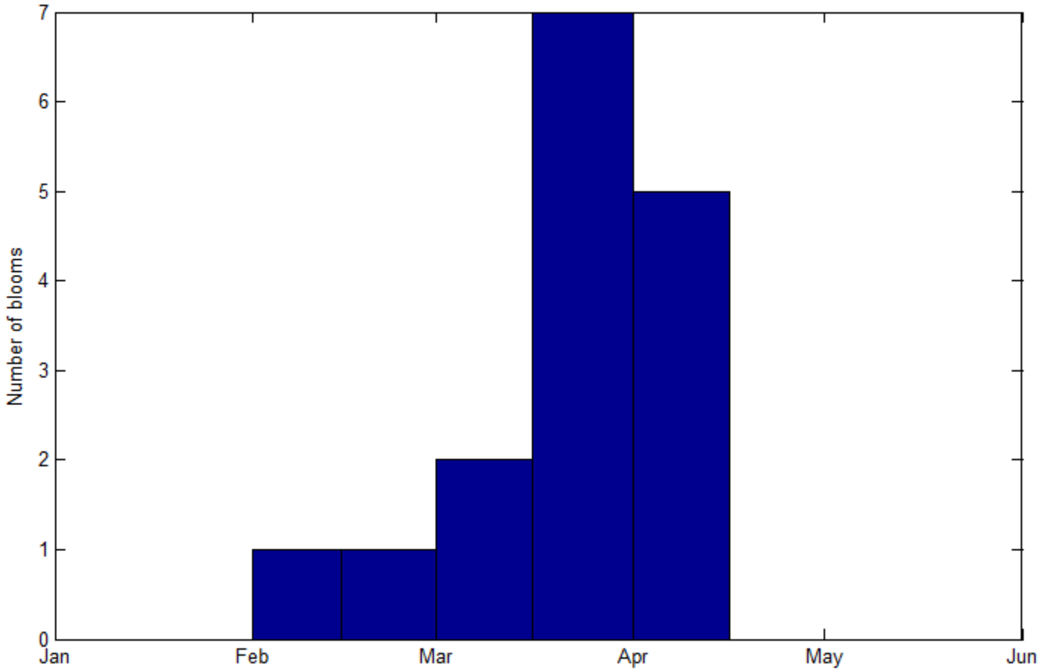

Figure 3.18. Histogram of dates of when the USGS *RV Polaris* observed maximum chlorophyll *a* concentration at Dumbarton Bridge water surface (top 1 meter at station 33) during its weekly to monthly cruises 1992-2011. Data from http://sfbay.wr.usgs.gov/access/wqdata/index.html .

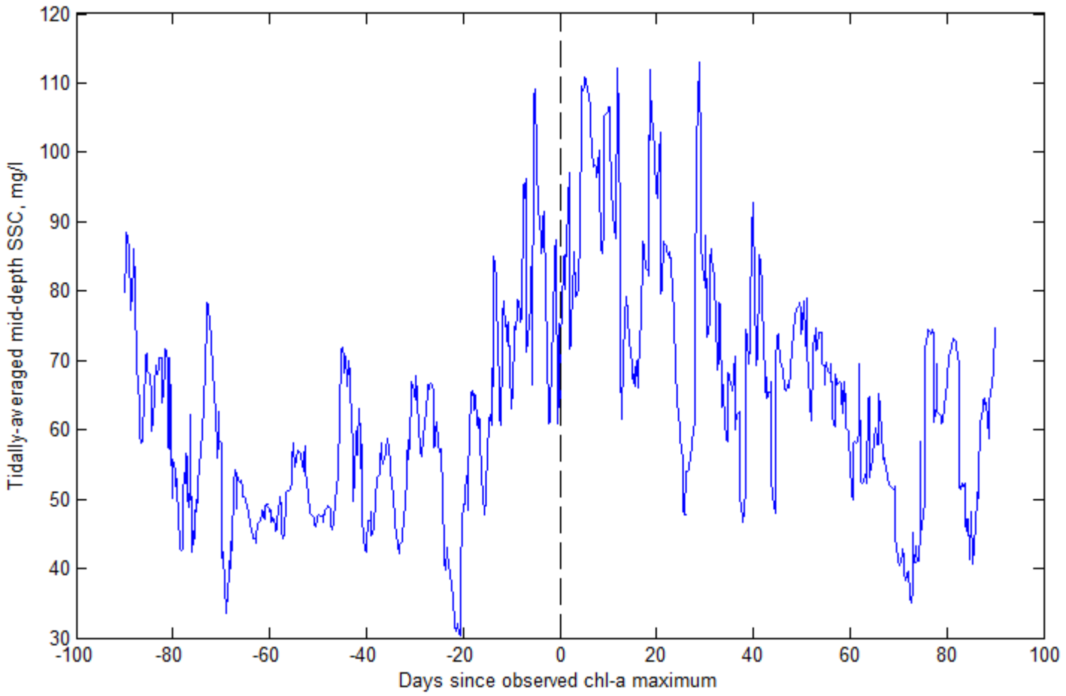

Figure 3.19. Median of tidally averaged Dumbarton Bridge mid-depth SSC. Time zero is the time the *RV Polaris* observed maximum chlorophyll *a* concentration at Dumbarton Bridge water surface (top 1 meter at station 33) during its weekly to monthly cruises.

3.5 Spatial gradients of suspended-sediment concentration

Suspended-sediment concentration is not spatially uniform in South San Francisco Bay. In this section we describe the general spatial gradients of SSC.

3.5.1 Longitudinal gradient

In general, SSC increases from north to south in South San Francisco Bay. Continuous monitoring data from 1993-2001 indicates that the median and interquartile range of mid-depth SSC increased from Central Bay (San Francisco Pier 24) to LSB (Channel Marker 17) (Figure 3.20) which was originally published in Schoellhamer et al. 2007). Factors creating this gradient include mixing with relatively clear ocean water in Central Bay and finer bed sediment and shallower depths in LSB (Barnard et al. 2013) that promote wind wave resuspension (Schoellhamer 1996).

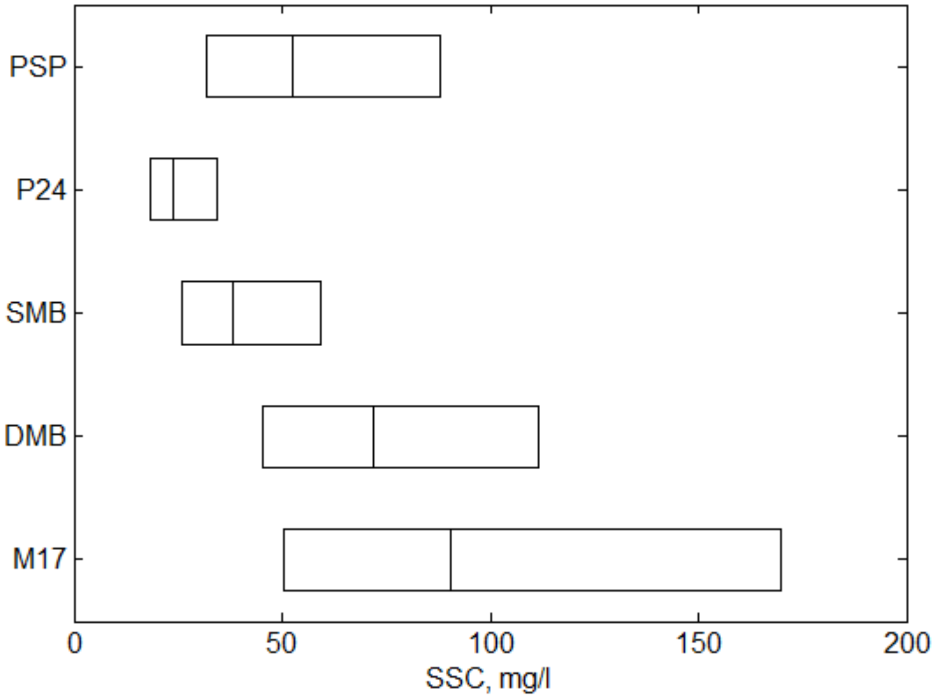

Figure 3.20. Box plots of mid-depth suspended-sediment concentration data, San Francisco Bay, 1993–2001. The left edge of each box indicates the lower quartile, the middle line indicates the median, and the right edge indicates the upper quartile. Stations on the vertical axis are arranged from north to south (Figure 1). Point San Pablo (PSP) is at the boundary between Central and San Pablo Bays, San Francisco Pier 24 (P24) is in Central Bay and is closest to the Pacific Ocean, San Mateo Bridge (SMB), Dumbarton Bridge (DMB), and Channel Marker 17 (M17) are in South Bay. From Schoellhamer et al. (2007).

3.5.2 Lateral gradient

SSC is usually greater in the shallows of South San Francisco Bay than in the deeper channel because of wind wave resuspension (Schoellhamer 1996). For example, SSC was usually about 50 mg/L at San Mateo Bridge and 100 mg/L at a shallow water site 6 km away on March 15, 1994 (Figure 3.21). One exception was during the lower low tide when SSC and San Mateo Bridge increased to 100 mg/L as more turbid shallow water entered the channel. Another exception was at the end of flood tides when SSC at the shallow water site decreased to 50 mg/L as relatively clearer channel water completed moving into shallow water (Figure 3.21). Thus, a simple conceptual model is that the deep channel in South Bay has clear water, the shallows have turbid water, and the two water masses displace one another during the tidal cycle.

Fifteen years later Brand et al. (2010) found similar results from more detailed measurements at two stations south of San Mateo Bridge. Instruments were deployed at stations 1000 and 2000 m from the middle of the deep channel February 24 to March 16, 2009. They found that SSC at the site closer to the channel was generally less than 50 mg/L and at the further site SSC would increase up to 100 mg/L during high wind-wave induced shear. Thus, SSC was greater closer to shore when wind waves resuspended bottom sediment. Most wind-induced resuspension events may have led to horizontal sediment transport directed toward the shoreline. During calm periods, SSC was greater on ebb tide than flood tide. Data from the two stations indicated that spatial heterogeneity of sediment dynamics is important in South Bay. While there was a step decrease in SSC between the Schoellhamer (1996) and Brand et al. (2010) studies, the basic process of wind-wave resuspension in shallow waters of South Bay was similar.

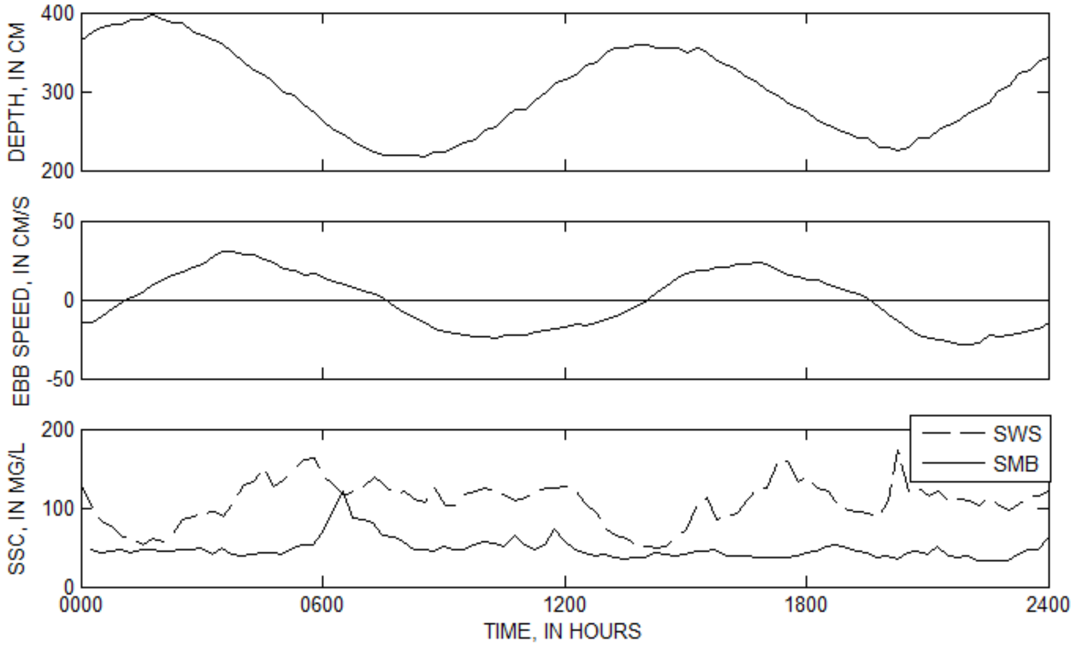

Figure 3.21. Variation of SSC at middepth at the San Mateo Bridge (SMB) and at the shallow-water site (SWS) with water depth and ebb current speed at the shallow-water site on March 15, 1994. The direction of ebb tide at the shallow-water site is to the northwest. From Schoellhamer (1996).

Although wind waves may increase SSC in the shallows, wind duration and lateral density gradient determine whether it reaches the channel. Lacy et al. (2014) found that sediment was transported from the eastern shallows to the channel in South Bay by sustained (greater than 24 hours) wind-driven return flow and a density-driven current when water on the shallows was denser due to being colder or more turbid. In the absence of a lateral density gradient or sustained wind, sediment resuspended by observed diurnal sea breezes was not transported to the channel.

3.5.3 Bay and sloughs

The tidal sloughs around the margin of South Bay experience significant strong vertical and longitudinal gradients of a number of variables. The sloughs serve as a mixing zone between tributary and Bay water and often receive freshwater flow at the upstream edge, which can dramatically alter the slough temperature, salinity, and other physical and chemical properties. Under certain conditions, the water column in the sloughs can stratify (typically based on salinity), which decreases turbulent mixing and alters transport processes. In fact, salinity gradients and stratification could lead to the formation of an Estuarine Turbidity Maximum (ETM) in the sloughs that can move up or down slough as the tides and freshwater inflow interact (Burchard and Baumert 1998; Schuttleaars et al. 2002). The USGS has begun studying the behavior of Alviso Slough, with respect to salinity, turbidity, and dissolved oxygen. This slough is the downstream reach of the Guadalupe River that drains the Santa Clara Valley and parts of the Coastal Range. Early results from the slough suggest that tidal dynamics have a large effect on overall transport processes.

Figure 3.22 shows an estimate of suspended-sediment flux (the product of water discharge and SSC, a mass per unit time) at a near-bottom sensor in Alviso Slough (USGS station 11169750). Large positive fluxes (bayward) in November and December 2012 were the result of precipitation events from storms, which transported sediments from the watershed to the Bay. In contrast, there are periods of negative (landward) flux, particularly in May and August-September 2012, where net transport is from the Bay into the slough. Such landward sediment flux has been documented during the summer months in a different San Francisco Bay tributary, Corte Madera Creek near the town of Larkspur (USGS station 11460090, Downing-Kunz and Schoellhamer, 2013). In Corte Madera Creek, the summertime landward sediment flux is the result of a longitudinal concentration gradient in SSC generated by wind-wave resuspension in the shallow, open waters of the Bay.

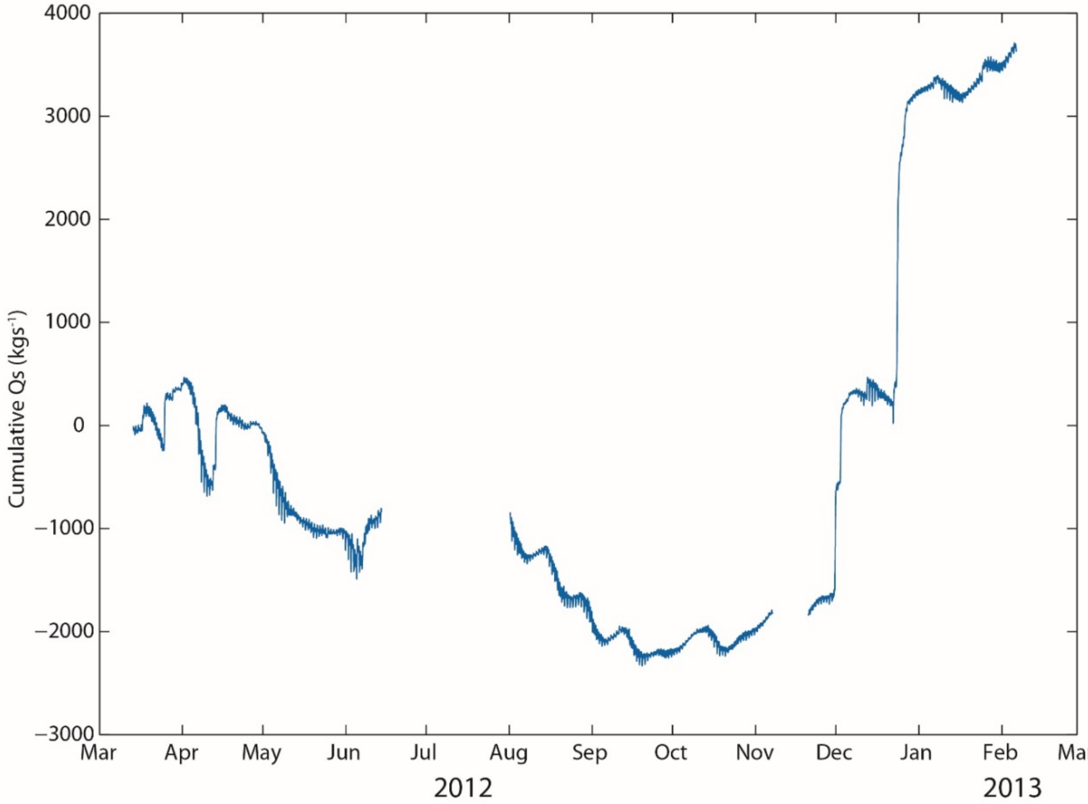

Figure 3.22. Cumulative suspended-sediment flux in Alviso Slough for parts of 2012 and 2013. Positive fluxes are bayward, negative fluxes are landward.

We hypothesize that water column stratification affects suspended sediment transport and DO in Alviso Slough. Figure 3.23 shows two 4-day periods (one spring-tide period and one neap-tide period) of depth, salinity, SSC, and DO data from the Alviso Slough station. There are some commonalities between the spring and neap tidal periods. Spring tides display local DO maxima associated with both high and low tidal stages. Neap tides have the local DO maxima associated with high tide before and low tide after only the strong ebbs. Local maxima for SSC occur immediately prior to each low tide during spring tides, while maxima for SSC during neap tides occur immediately prior to low tide following the strong ebb. Both periods exhibit relatively low SSC during early to mid-ebb tides and moderate SSC during most flood tides. The major difference between the two tidal periods is associated with the weak high tide and ebb during neap tides. Late during the weak flood and continuing until mid-strong flood (Figure 3.23B), salinity is nearly constant, SSC is depressed, and DO exhibits a steady decrease. The lowest values of both SSC (< 50 mg/L) and dissolved oxygen concentration (< 2 mg/L) occur during these periods. About midway through the daily strong ebb, salinity and SSC both increase and DO dramatically recovers. A plausible explanation for this pattern is water column stratification - the weak ebb and flood tides are not energetic enough to breakdown stratification that has formed in the slough. The stratification has two main effects: 1) it suppresses turbulence and sediment resuspension and 2) it isolates the bottom layer of water from the atmospheric oxygen supply and sunlight. The trapped water is still subject to oxygen demand from the water column and sediments, so DO concentration continues to drop until the stratification breaks down and the water column fully mixes. The increase in DO co-occurs with an increase in salinity suggesting that this higher DO water has a bayward source. Local minima of SSC, steadily decreasing DO to a local minima, and nearly constant salinity are also seen during early- to mid-ebb tides in spring tidal periods (Figure 3.23A). This may also be a function of water-column stratification, albeit for a shorter period of time because the high energy associated with the spring tides will increase turbulence and reduce stratification. Asymmetry in the flood/ebb SSC that results from water column stratification can lead to asymmetries in net flux, and subsequent net upstream transport of sediment.

3.5.4 Vertical gradient

Suspended-sediment particles are denser than water so they tend to settle to the bottom. This settling is counteracted by turbulence that keeps particles suspended and results in vertical dispersion. The net result of settling and vertical dispersion is that the theoretical vertical profile of suspended sediment has minimum SSC at the surface and maximum at the bottom. Vertical profiles from LSB confirm that this is generally the case (Figure 3.24). Most of the profiles measured by the USGS *RV Polaris* in 2012 at Polaris station 34 in the main South Bay channel south of Dumbarton Bridge near Newark Slough have greatest SSC near the bed.

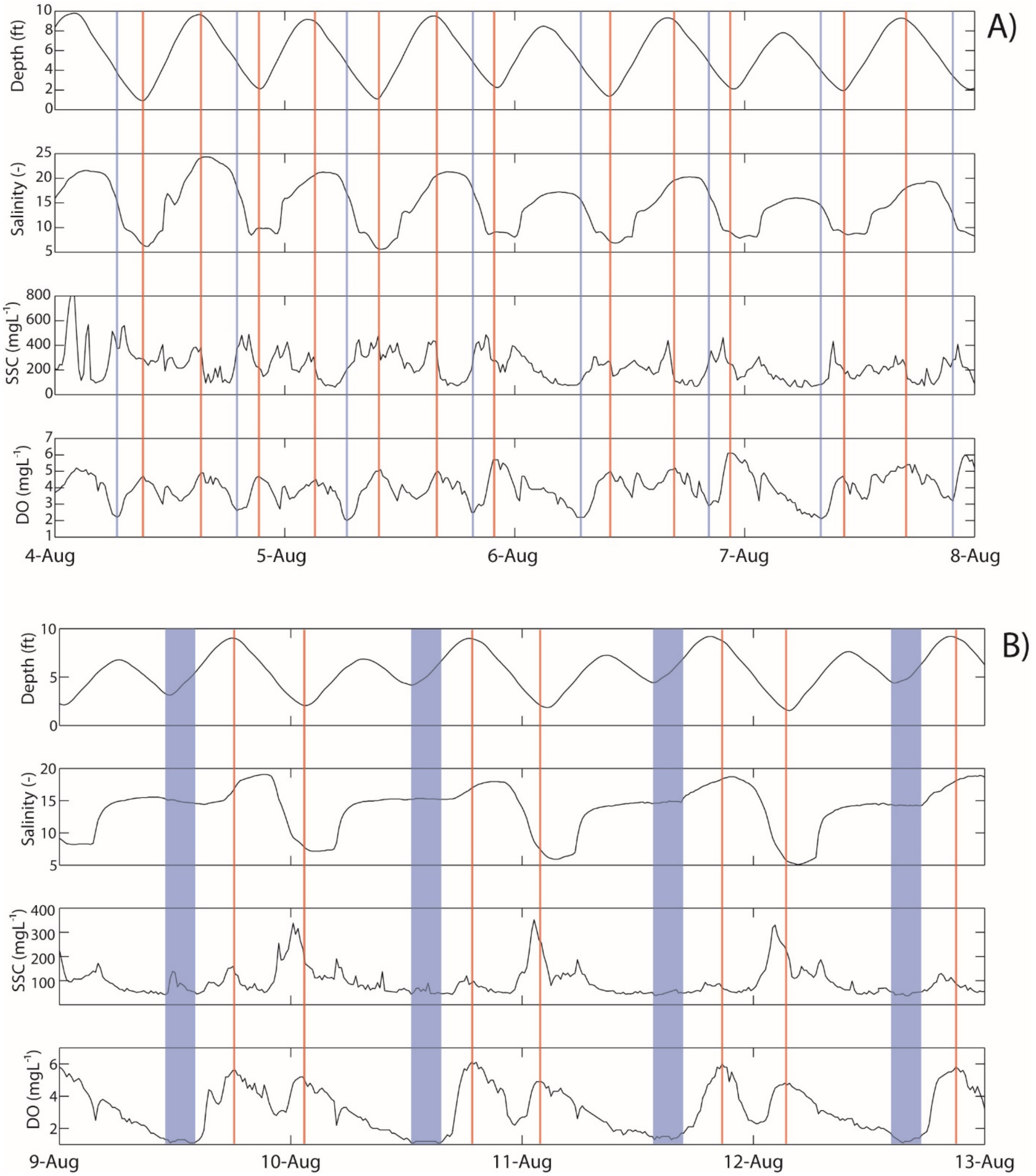

Figure 3.23. Depth, salinity, suspended-sediment concentration (SSC), and dissolved oxygen (DO) in Alviso Slough during A) a spring tide in early August 2012 and B) a neap tide in mid-August 2012. The red lines mark local maxima for DO, and the blue lines and blue shaded bars mark local minima for DO. During spring tides, DO concentrations are lowest for a brief period near the end of each ebb; local maxima are observed at nearly all depth extrema. During neap tides, DO concentrations are lowest for an extended period beginning late during the daily weak ebb and extending through the middle of subsequent flood; we hypothesize these low concentrations may be attributed to water column stratification.

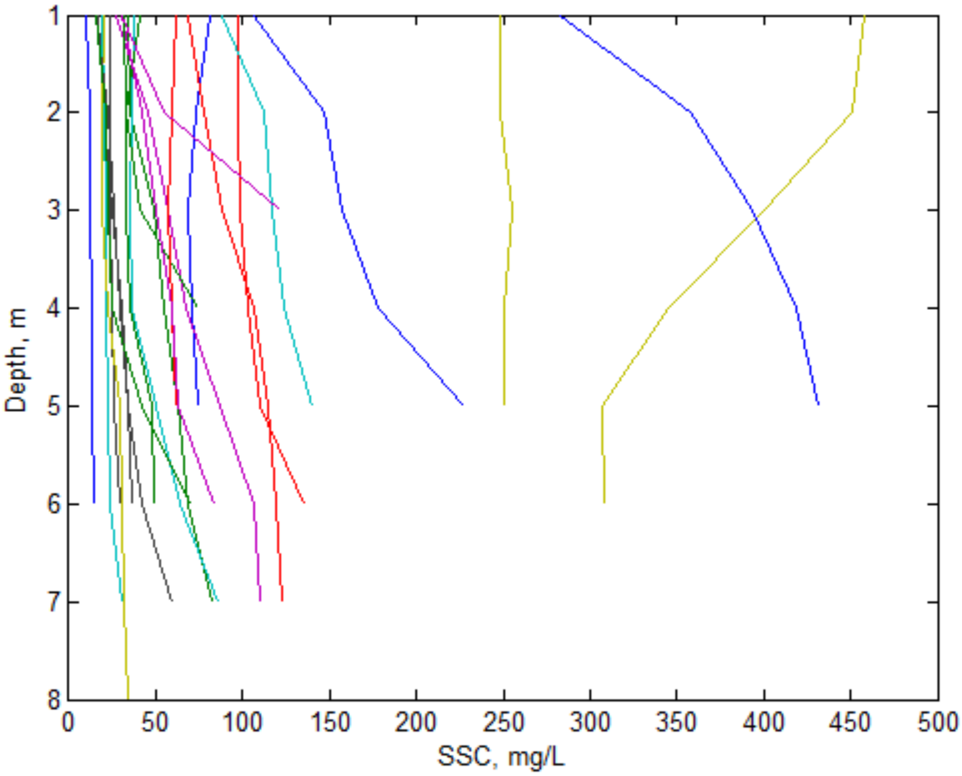

Figure 3.24. Vertical profiles of suspended-sediment concentration, USGS *RV Polaris* station 34 south of Dumbarton Bridge in the main channel near Newark Slough, in 2012. Data from http://sfbay.wr.usgs.gov/access/wqdata/index.html .

3.6 Watershed effects on suspended sediment

3.6.1 Local freshwater flow

There are at least ten tributaries that flow into LSB. The two main tributaries are Guadalupe River and Coyote Creek, which together deliver about 55% of the tributary inflow into this reach (Shellenbarger et al., 2013). Both of these tributaries have decade-long gaging stations that are located just above the head of the tide (Guadalupe River: USGS station 11169025; Coyote Creek: USGS station 11172175). The Guadalupe River delivers more freshwater to the Bay than Coyote Creek; peak storm flows during 2006 were 2,500 cfs and1,500, respectively (Figure 3.25). The other tributaries to the reach - including San Francisquito Creek, Stevens Creek, Calabasas Creek, and others – have not been as reliably gaged and thus provide less useful data. Combined peak storm flow for the large storm in 2006 was about 4,000 cfs from the gaged tributaries and estimated to be about 3,300 cfs from ungaged areas for a combined tributary inflow around 7,300 cfs. There are also three wastewater treatment plants (WWTP) that discharge fresh water to this reach of the bay. The San/Jose/Santa Clara WWTP, the largest of the three, is estimated to have a mean annual discharge rate of 152 cfs (City of San Jose 2012).

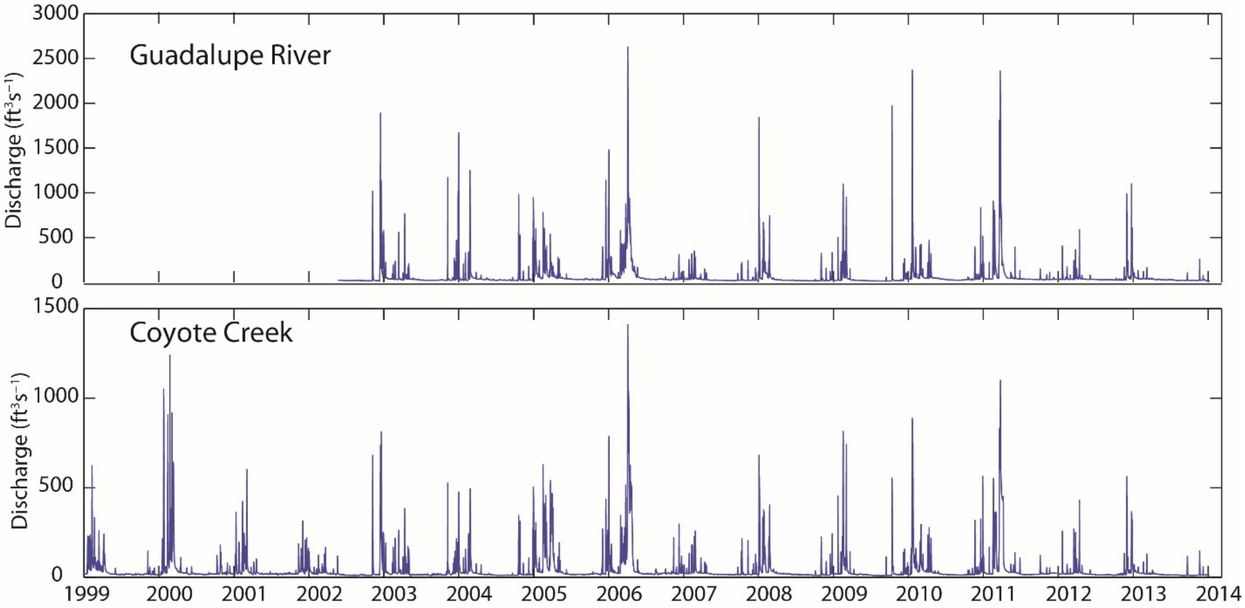

Figure 3.25. Discharge from the two main LSB tributaries for the past decade. Note the different scales on the y-axis.

The pattern of discharge from the tributaries is normal for this part of California and follows precipitation patterns. Runoff events typically correspond directly to storm systems and rarely continue with much flow beyond a few days after rain. Over the period of record in Figure 3.25, summer conditions are dry with only base flow of a few cubic feet per second in each stream. Summertime flows typically result from agricultural irrigation and urban use return flows and are not considered to support much sediment transport. Thus, the three WWTP are the dominant source of fresh water flow into LSB during summer. Figure 3.26 shows a longer discharge time series from a Guadalupe River station (USGS station 11169000) several miles upstream of the above mentioned gaging station. These data reinforce that storm-responsive, seasonal flows are the normal condition in these tributaries.

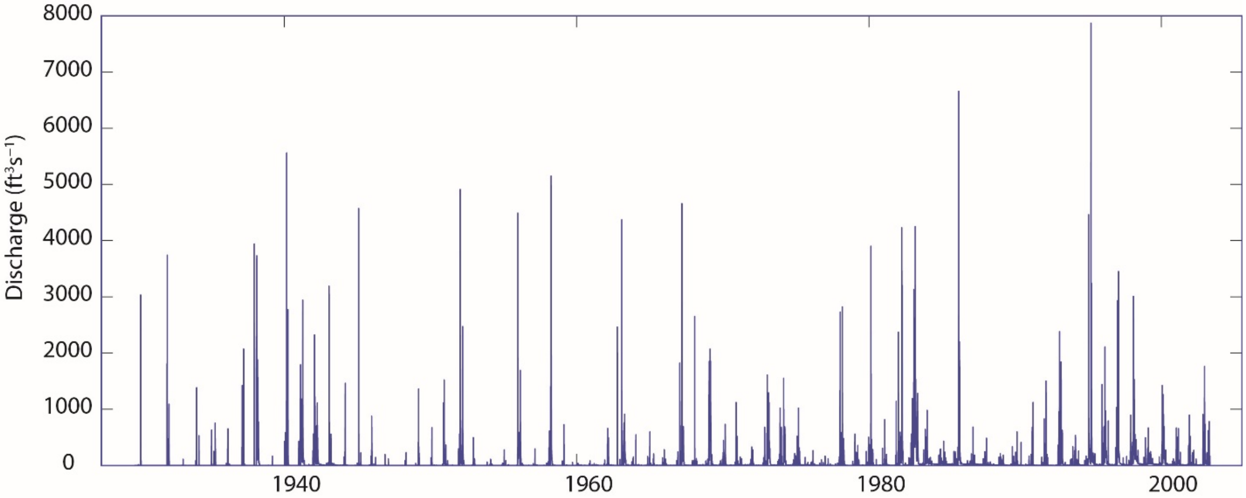

Figure 3.26. Long-term discharge record from USGS station 11169000 on the Guadalupe River that shows more than 70 years of the annual and seasonal patterns of tributary flow in South Bay.

The supply of suspended sediment from the two major tributaries to LSB has recently only been gaged seasonally (October – April), reflecting the local precipitation patterns and low streamflow conditions that typically exist during the late spring and summer. Since 2004, these tributaries have contributed between 2- 27 kt (kilotonnes) of suspended sediment annually to LSB (Figure 3.27). For four of the six years that both sites have data, Coyote Creek has delivered more sediment to the Bay than the Guadalupe River, in spite of larger discharge in the Guadalupe River (Figure 3.25). This is likely due to differences between the two watersheds. Coyote Creek drains the west side of the coastal range on the east side of the Santa Clara Valley, while the Guadalupe River drains the more vegetated east side of the Santa Cruz Mountains. Precipitation amount does not correlate particularly well with stream discharge and suspended-sediment flux (SSF), since a number of factors control the relationship between precipitation and runoff, such as soil moisture, permeability, slope, vegetation, antecedent rain, etc. The relationship between SSF and discharge is more closely linked, with a least-squares regression coefficient of determination R^2 = 0.49 (n = 1273, p<0.001, Coyote Creek) and R^2 = 0.70 (n = 1698, p<0.001, Guadalupe River).

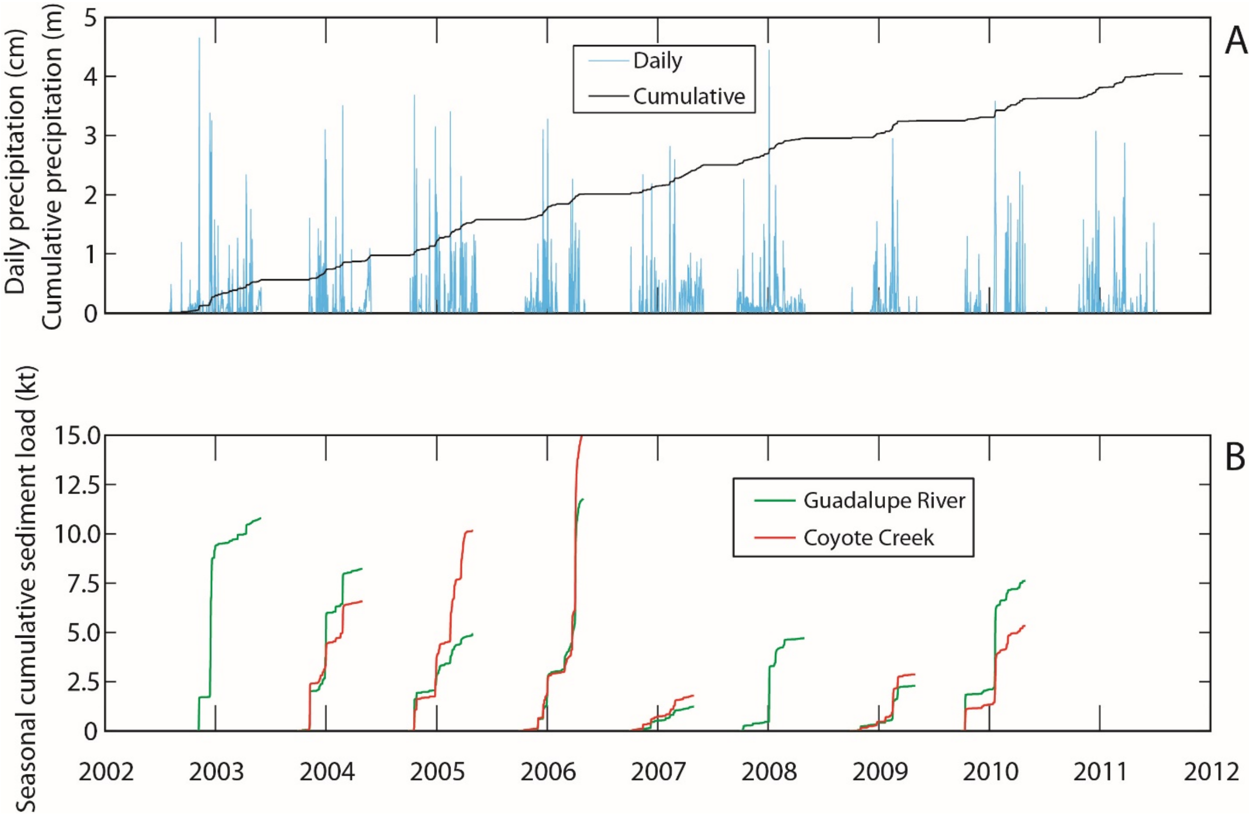

Figure 3.27. Time series of a) precipitation at Union City, CA and B) seasonal (October – April) cumulative tributary sediment load since Water Year 2003.

3.6.2 North Bay freshwater flow

Flow from the Delta into the Bay is the largest single source of freshwater to the Bay. Figure 3.28 shows a daily time series of outflow from the Delta since 1955 (California Department of Water Resources 1986). The long-term average flow is 27,000 cfs (standard error = 300 cfs), while the long-term median flow is about 11,000 cfs. Variability in seasonal flow is very high, with most precipitation falling between October – April of a given Water Year. Interannual variability creates wet and dry years; 1986 was a year of flooding and the greatest daily flow from the Delta, while the next five years 1987-1991 show extended drought conditions with very little Delta flow (Figure 28). Since 2000, there have been four years classified as 'above normal' or 'wet' and nine years 'below normal' or 'dry' and one critically dry year (using the Sacramento Valley Water Year Index, http://cdec.water.ca.gov/cgi-progs/iodir/wsihist).

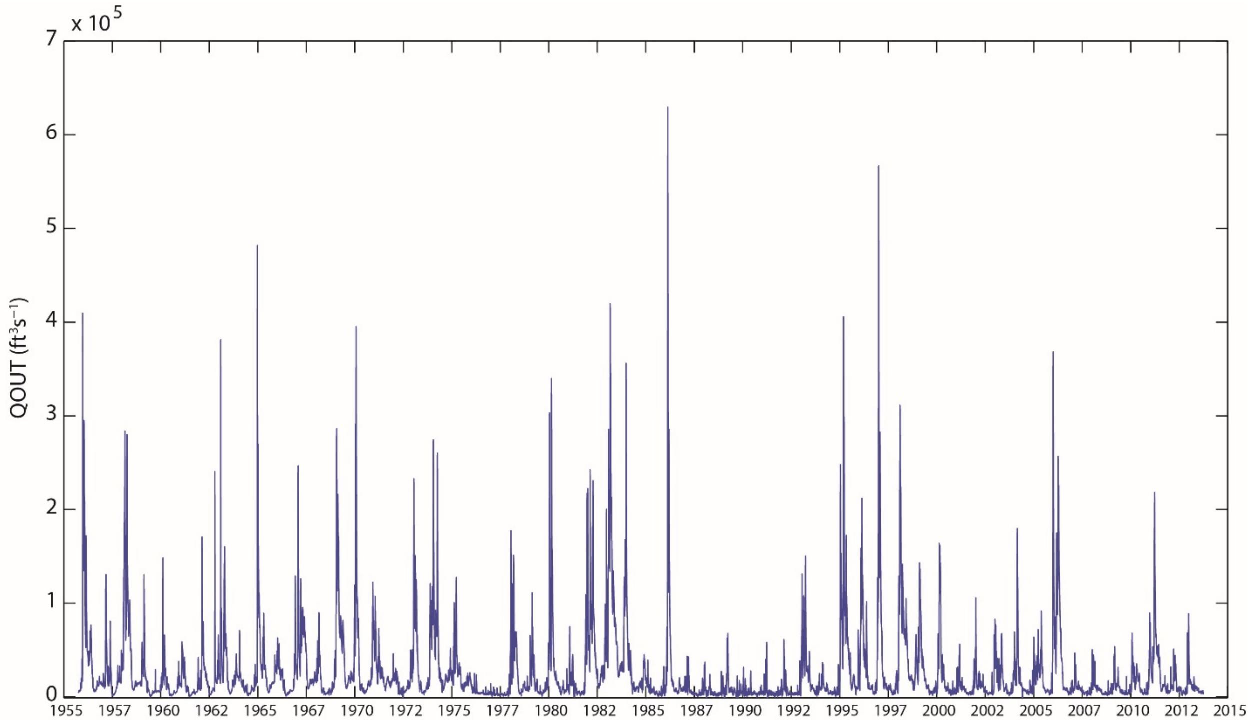

Figure 3.28. Time series of the estimate of QOUT, daily discharge from the Delta to the Bay (California Department of Water Resources 1986).

Under typical estuarine conditions, regions of a bay that are closest to the mouth of the estuary have higher salinities than regions closer to the freshwater inflow. The San Francisco Estuary has two lobes, with the main source of freshwater flow coming into the northern lobe (North Bay) and only local tributary and WWTP inflow in the southern lobe (South Bay). Typically, areas at the head of the lobes (i.e., Mallard Island in the north and Alviso in the south) experience salinities that are lower than that in Central Bay – this is a normal estuarine condition. In years of high Delta outflow, however, Central Bay can become fresh relative to South Bay. This leads to an inverse estuarine condition for South Bay with saltier water near the head and fresher water near the mouth. This condition can create a reversal in the direction of gravitational circulation in South Bay, promoting net residual flux of material in the lower water column northward and flushing South Bay (McCulloch et al., 1970, Shellenbarger et al., 2013).

Unlike rivers, where peak SSC generally correlates with precipitation and flow events, SSC at Dumbarton Bridge is decoupled from tributary flow; instead peaks are generally observed in April (Table 3.2). The specific mechanism that controls the springtime SSC at Dumbarton is unknown; however, it appears to be linked to the spring phytoplankton bloom in South Bay as discussed previously. Given the large spring SSF at Dumbarton, the specific timing of an inverse estuarine event in the Bay during the spring could have a dramatic impact on the net annual direction of material flux. Water year (WY) 2009 was classified as a dry year, WY2010 as below normal, and WY2011 was a wet year. Figure 3.29 shows Dumbarton SSF during these three types of year. In the dry year, SSF was into LSB, a below normal year showed almost no net flux, and a wet year (2011) showed a strong SSF northward out of LSB. This suggests that flow from the Delta might be controlling the direction of flux at Dumbarton during the spring. Using monthly spring (April, May, and June) Dumbarton sediment flux and outflow from the Delta, a significant relationship is seen (Figure 3.30). More data are needed to confirm this relationship.

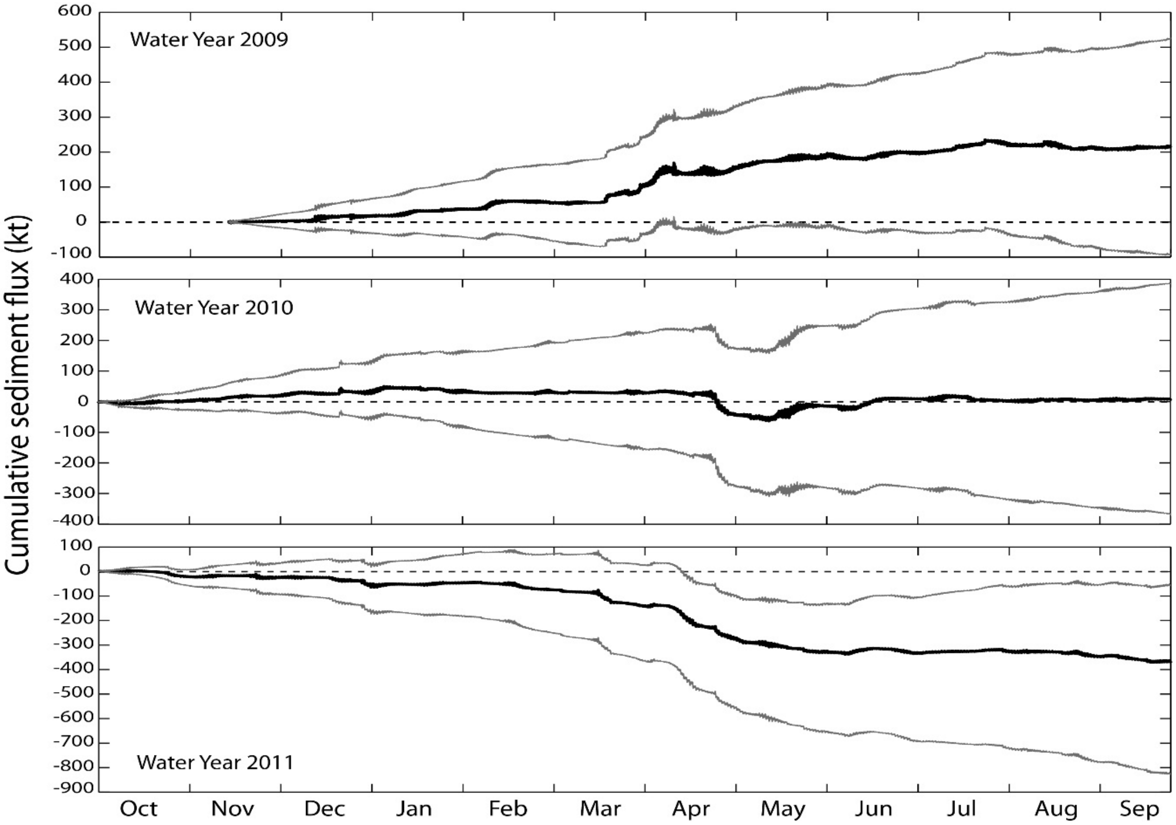

Figure 3.29. Cumulative sediment flux past the Dumbarton Bridge from WY2009-WY2011. Positive fluxes are into LSB from the rest of the Bay and negative fluxes are seaward out of LSB into the rest of the Bay. Black lines are the estimate of flux and grey lines are ± 1 standard deviation. From Shellenbarger et al. (2013).

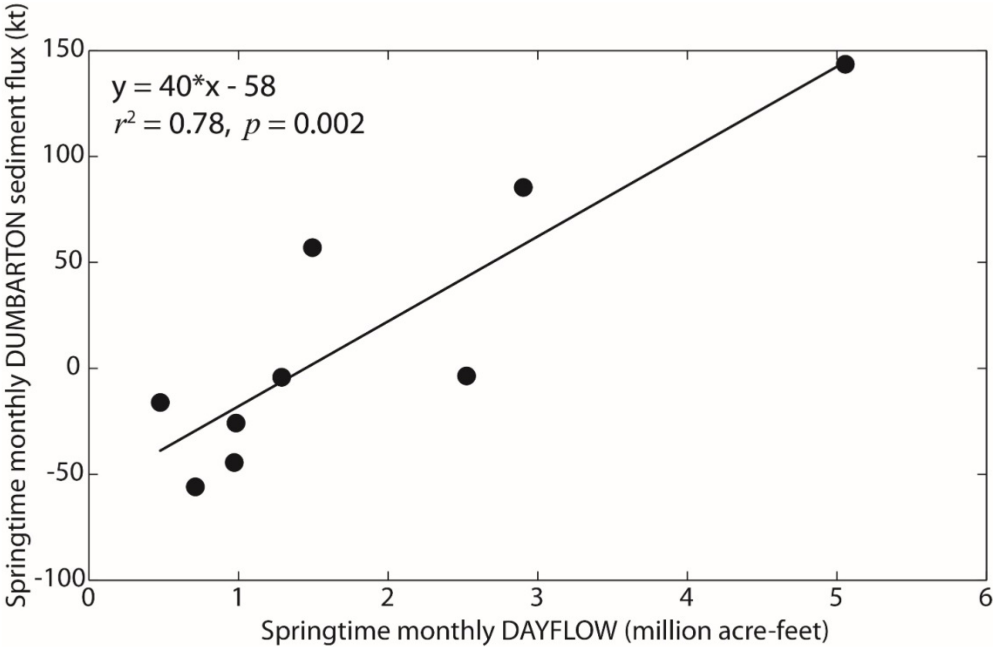

Figure 3.30. The relationship between springtime monthly Dumbarton suspended-sediment flux and springtime monthly outflow from the Delta, DAYFLOW.

3.6.3 Salt pond restoration

The South Bay Salt Pond Restoration Project (www.southbayrestoration.org), a major wetland habitat restoration project taking place along the margins of South Bay, plans to turn about 6,000 ha of former commercial evaporative salt ponds into a mix of tidal marsh and managed ponds to support wildlife. Many of the project ponds have subsided below mean sea level due to groundwater overdrafts in the mid-20th century. These ponds will require sediment to raise their bottom elevations up to mean tide level to allow for colonization of tidal marsh plants. It has been estimated that this subsided or 'accommodation' space will require 29-45 Mm^3 of sediment (U.S. Fish and Wildlife Service and California Department of Fish and Game 2007). Assuming a sediment bulk density of 617 kg m^{-3} (an average of seven core samples collected in two different studies, Caffrey 1995, and Love et al. 2003), it is estimated that between 18,000-28,000 kt of sediment are required to fill the accommodation space.

A suspended-sediment budget for LSB was calculated for a three-year period by Shellenbarger et al. (2013, Figure 3.29). The sediment sources for the budget include input by the local tributaries and WWTP and input from the rest of the Bay through the Dumbarton Narrows. Suspended sediment is lost from the system only from the export of sediment to the rest of the Bay northward through the Dumbarton Narrows. The input from the wastewater treatment plants is insignificant compared to the other terms in the budget, so the net gain or loss of sediment in this reach depends on local tributary input and the flux at Dumbarton. Table 3 details these inputs and losses for 2009-2011. While local tributaries deliver sediment to South Bay in wet or dry years, the supply can vary by at least a factor of four. The flux through the Dumbarton Narrows is also highly variable (Table 3.3) and can be an order of magnitude larger than the tributary flux. In drier years, the net sediment flux is into LSB from the rest of the Bay, while wet years appear to reverse the direction of flux. Although this study did not include collection of water column stratification data, a reasonable hypothesis is that gravitational circulation (described previously) in South Bay controls the flux direction.

Table 3.3. Total amount of suspended sediment gained or lost from different sources in LSB over three Water Years in kilotonnes. Positive values are inputs to LSB, and negative values are losses to the rest of the Bay. The local tributaries include data from the two gaged streams and scaled loads for the other streams.

Water Year	Local Tributaries (kt)	Dumbarton (kt)	Net to Project Area (kt)
2009	9	220	229
2010	24	11	35
2011	42	-440	-398

This estimate of annual natural supply of sediment to LSB is small compared to the amount of sediment needed to fill the accommodation space in the ponds. Given that more sediment was exported from LSB in WY 2011 than was imported in WY 2009 and 2010 combined, predicting the time it would take to fill the accommodation space through the natural sediment supply is problematic. If the sediment supply is solely from the tributaries, the sediment budget data suggest that it could take between 400 years (larger flux, smaller need) and 3,300 years (smaller flux, larger need) to fill the subsided areas. If we look at the total net flux only for WY 2009 and 2010 (when there was net import of sediment to LSB), natural sediments could take 90-600 years to fill the space. These estimates are suggestive of the potential effects of regional long-term sediment supply, assuming that the Project sediment needs do not change, and there is no net loss of marshes and mudflats in the reach. To date the Project has restored several ponds and accretion time scales have been faster perhaps because only a small fraction of the subsided volume has been tidally connected to the Bay.

The use of flux estimates that are based on less than a decade of data may not truly reflect the actual long-term potential for larger sediment loads. First, the weather in California is influenced by El Nino-Southern Oscillation (ENSO) patterns that switch between predominately wet and predominantly dry years at a frequency of about three to seven years (McKee et al. 2013). Shellenbarger et al. (2013) computed a sediment budget for only three years (2009-2011). These years are classified as ‘Dry’, ‘Below Normal’, and ‘Wet’, respectively (for the Sacramento Valley Index, http://cdec.water.ca.gov/cgi-progs/iodir/wsihist). Although these three years cover a range of hydrologic conditions from wet to dry, they do not represent the total range of variability in the system. Since 1906, there have been 21 years where the water year index was higher than in 2011 (wetter) and 21 years where the water year index was lower than in 2009 (drier). Second, Mckee et al. (2013) estimates that, for 1995-2010, small tributaries around SFB contributed 61% of the suspended-sediment flux to the Bay – more than comes from the Central Valley. The main explanation for this is that the small tributaries in the steep, more erodible, tectonically active watersheds around the Bay have a higher mean yield of sediment than the lower gradient, granitic rock watersheds of the Sacramento and San Joaquin Rivers. In fact, the mean yield of the smaller tributaries is 12-times greater than the mean yield from the Sacramento-San Joaquin Rivers. For these reasons, Shellenbarger et al.

(2013) likely underestimates the contribution of the tributaries to the long term sediment supply to the project area.

The loss of suspended sediments in the Project area through sedimentation or export could decrease the concentration of suspended sediment in the water column. This would mean a decrease in turbidity and increasing light penetration in South Bay. So, one potential effect of opening ponds to tidal action could be an increase in the potential for phytoplankton blooms. Shellenbarger et al. (2004) estimated that opening Alviso Ponds A9-16 (the most subsided ponds) to tidal action would result in a loss in SSC from 118 mg/L to 110 mg/L – a net loss in turbidity of about 10%. May et al. (2003) developed a model that defined two water column clearing rates, vertical and horizontal, that can control bloom dynamics. Decreasing the water column turbidity by 10% has the same effect as increasing the horizontal and vertical clearing rates in the model by 10%. The increase in the clearing rates can lead to more favorable conditions for a shoal-supported bloom; however, this effect is small compared to the effect that the inter-annual variability in the benthic grazing rate has on phytoplankton bloom dynamics. Since the opening of ponds to tidal action will be phased instead of en masse, the impact of opening the ponds on the water column SSC should be less than 10%. This suggests that the restoration may affect South Bay SSC, but it is unlikely to have any strong influence on potential for phytoplankton blooms.

3.7 Summary

In this chapter we reviewed the present state of the knowledge of processes affecting suspended sediment in LSB. Suspended sediment in an estuary can control phytoplankton bloom dynamics in the water. Light attenuation in the water column is proportional to suspended-sediment concentration, so decreased SSC increases the probability of sustained phytoplankton blooms and the potential for eutrophication in LSB.

Vertical processes in the water column control the exchange of suspended sediment with the bed. These processes include particle settling (which is affected by flocculation), erosion from the bed, and resuspension into the water column. How much sediment is mixed throughout the water column is directly affected by density stratification of the water column.

In general, SSC in San Francisco Bay periodically varies on time scales from tidal to annual, with semidiurnal, semimonthly, and monthly tidal cycles dominating; however, subtidal time scale processes account for more SSC variance than the tidal time scale processes. In South Bay, maximum SSC typically occurs in April or May and minimum SSC typically occurs in October or November. Of greatest concern is the 49% step decrease in suspended sediment at Dumbarton Bridge around 2000. This appears to co-occur with a period of increasing size and frequency of phytoplankton blooms.

SSC also varies longitudinally and laterally in South Bay due to differing horizontal processes and spatial variations in sediment supply. In general, there is more SSC in South Bay closer to the tributaries than to the mouth of the Bay resulting in increasing SSC to the south. SSC is usually higher over the shallow water shoals and mudflats than in the deeper channel. This is because sediment resuspension is greater in the shallower areas where wind waves impart a greater force on the bay floor.

At least ten tributaries and three wastewater treatment plants provide inflow and sediment to South Bay. The tributaries provide storm-related sediment inputs mainly during the winter, while the treatment plants operate year-around. Two main tributaries to South Bay delivered 2-27 kt suspended sediment annually during the period 2004-2011. On an annual basis, LSB can be either a source or sink of sediment for the rest of San Francisco Bay. Hundreds of kilotonnes of suspended sediment pass under the Dumbarton Bridge each year – typically an order of magnitude more sediment than supplied by the tributaries. However, the direction of flux appears to be related to water-year type, where dry years promote flux to the south and wet years promote flux to the north. It is not known if the same is true for sediment flux past the San Mateo Bridge. Local tidal marsh restoration activities could reduce SSC though deposition in the restored areas, but this effect should be small relative to the total amount of sediment in the system.

3.8 Acknowledgements

Jessica Lacy, David Senn, and Paul Work provided helpful comments on initial drafts of this chapter. The San Francisco Estuary Institute, US Army Corps of Engineers/Regional Monitoring Program, and the USGS Priority Ecosystem Science Program financially supported the writing of this chapter. Professor Manning's contribution to the preparation of this chapter was partly funded by both the US Geological Survey Co-operative Agreement Award (G11AC20352) with HR Wallingford (DDS0280), and the HR Wallingford Company Research projects 'Sediment in Transitional Environments — SiTE' (DDY0427) and ADCP-Sediment Profiling (DDY0460). Professor Manning would like to thank his HR Wallingford Coasts & Estuaries colleagues Prof. Richard Whitehouse and Tim Chesher, for their continued support.

4. Benthic grazers in Lower South Bay

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4.1 Introduction

Benthic communities are monitored because the individuals reflect the water quality of a system by using and sometimes controlling available carbon resources. They are also a good indicator of stressful environments because they accumulate contaminants, and respond, sometimes dramatically, to low and high phytoplankton biomass as well as low oxygen conditions. Pearson and Rosenberg (1978) published a conceptual model that hypothesized that the benthic community would respond to increasing organic enrichment and pollution with the following succession of stages: (1) no change in species composition but slight increases in biomass; (2) larger increases in biomass and loss of some species that will be replaced by opportunistic species; and (3) appearance of anoxia with loss of all benthic species. As predicted, Pearson and Rosenberg's changes in biomass and community composition have been noted with eutrophication but in most cases a decade or more of data is needed to see the change (Heip 1995). For that reason we concentrate here on data sets that could be used to compare before and after changes should nutrient-related impairment arise.

Benthic communities can also affect water quality by grazing pelagic food resources and increasing the rate of nutrient regeneration through feeding and bioturbating the sediment. South Bay is a system dependent on phytoplankton as the base to the food web (Jassby et al 1993). Despite abundant nutrients, South Bay has limited phytoplankton production due to poor light conditions and high grazing losses. Thus, the system only rarely experiences anoxia that is usually associated with high nutrient systems (Cloern 2001). Our conceptual model for phytoplankton growth in South Bay includes a delicate balance between light availability, grazing losses (primarily in the shallow water) and physical mixing of the water column (Lucas et al 2009). This balance has maintained the phytoplankton in South Bay at low biomass levels relative to other high nutrient urban estuaries (Cloern 2001). Increases in light availability, decreases in benthic grazing rates, and/or a reduction in mixing can and have resulted in unusually high phytoplankton biomass on some occasions. Two such occasions occurred in 1993 and 1998 when periods of increased light availability, caused by density stratification of the water column, increased the phytoplankton growth rate (Thompson et al 2008, Cloern 1996). Other periods of elevated chlorophyll *a* concentrations were seen in summer and fall 1994 (Thompson et al 2008) and in 1999-2004 (Cloern et al 2007) resulting from a sharp reduction in benthic grazers (dominated by bivalves in this system).

Our primary goal here is to first compare how water quality and physical habitat have structured the benthic invertebrate community in the South Bay in the past, (Nichols and Thompson 1985a, 1985b, Nichols and Pamatmat 1988, Lee et al. 2003) and in more recent work. Our analysis of the benthic community includes abundance, biomass/grazing rate where appropriate, and the relative dominance of functional traits (feeding mode, habitat, motility, and structures such as tubes and shells) within a community. These analyses will help signal if there has been a shift in the impact of the benthic community on the pelagic producers and nutrients.

Benthic species distributions are dependent on physical habitat (substrate and depth), physiological limits (salinity in this system, Lee et al. 2003), and predators (Cloern et al. 2007). Therefore seasonal and interannual differences in freshwater flow result in both seasonal and episodic patterns in species abundance and community composition (Nichols and Thompson 1985a, 1985b). At a larger time scale, episodic events such as invasive species introductions can have lasting effects on the benthic community structure and function. Therefore to understand the benthic community, analyses must be done at time scales to capture seasonal changes as well as longer term events. Here, we use these analyses to examine if the composition and function of the benthic community has sufficiently changed in the last 20 years to change its grazing impact on phytoplankton, reflect a change in nutrient/phytoplankton biomass state, and to alert us to future changes in benthic grazing that might affect our understanding of the nutrient concentrations and patterns of the ecosystem.

4.1.1 The importance of non-indigenous species

Nichols and Thompson (1985a) described a 10 year data set at an intertidal location in South Bay and summarized the state of our knowledge on the Bay benthos in 1985 (Nichols and Thompson 1985b). In both papers the authors acknowledged the high percentage of non-indigenous species in the benthic community and how the traits of those species might determine community structure and persistence. The fifteen most common species in the 10 year study were all non-indigenous; the authors did not know that the one native species that was listed, *M. balthica*, was it fact the exotic bivalve, *Macoma petalum.* Nichols and Thompson also stressed the importance of physical and possibly chemical disturbance in maintaining populations of these opportunistic species and the importance of seasonal weather and hydrologic extremes in controlling the seasonal patterns of growth, reproduction, and mortality of individual species. Despite the variability that they observed in time and space, the authors concluded that benthic communities sampled "during the past three decades in San Francisco Bay provide no evidence that the qualitative distribution of benthic macroinvertebrate species in the bay had(s) changed perceptibly" (Nichols and Thompson 1985b). Lee et al. (2003) analyzed more recent benthic community data from four monitoring programs (two in the south bay) and concluded that the introduction of *Potamocorbula amurensis* (Carlton et al. 1990) and its effect on the composition of the benthic community in the northern bay (Nichols et al. 1990) in addition to its reduction of phytoplankton (Alpine and Cloern 1992) and zooplankton (Kimmerer et al 1994) biomass had rapidly changed the benthic community structure from the period of the Nichols and Thompson studies and that their conclusion needed to be revisited. It is not apparent that the introduction of *Potamocorbula* resulted in large benthic community changes in the South Bay because we have limited data before 1988 but it is possible that the benthic community today in South Bay is a result of competition with and facilitation by *P. amurensis*.

4.2 Methods and Data Analysis

Most South Bay benthic data through 2006 and through 2009 in one case is available on the "San Francisco Bay Macroinvertebrate Atlas" website (http://www.werc.usgs.gov/Project.aspx?ProjectID=210). Of that data, the broadest coincidently collected spatial data and longest running temporal data were collected by the USGS, National Research Program, Menlo Park, CA (J. Thompson Project Lead) and that data will be the source of most of the discussion here.

4.2.1 Sampling

Samples for intertidal benthic community analysis were collected with an 8.5-cm diameter by 20-cm deep hand-held core. Three replicate samples were taken arbitrarily, within a square-meter area, during each sampling date. Three seasonal, spatially intensive benthic community sampling studies with major emphasis on shallow water locations was conducted in 1993-1995, using a 0.05 m^2 van Veen Figure 4.1). Seasonal studies were done (1) prior to or during the early stages of the spring bloom, (2) in mid-summer when there was no bloom, and (3) during the fall equinox. Benthic community samples were washed on a 500-µm screen, fixed in 10-percent formalin and then later preserved in 70-percent ethanol. Samples were stained with rose bengal solution. All animals in all samples were sorted to species level where possible (some groups, such as the oligochaetes, are still not well defined in the bay), and individuals for each species were enumerated. Taxonomic work was performed in conjunction with a private contractor familiar with the taxonomy of San Francisco Bay invertebrates (Susan McCormick, Colfax, California). McCormick also compared and verified her identifications with previously identified samples.

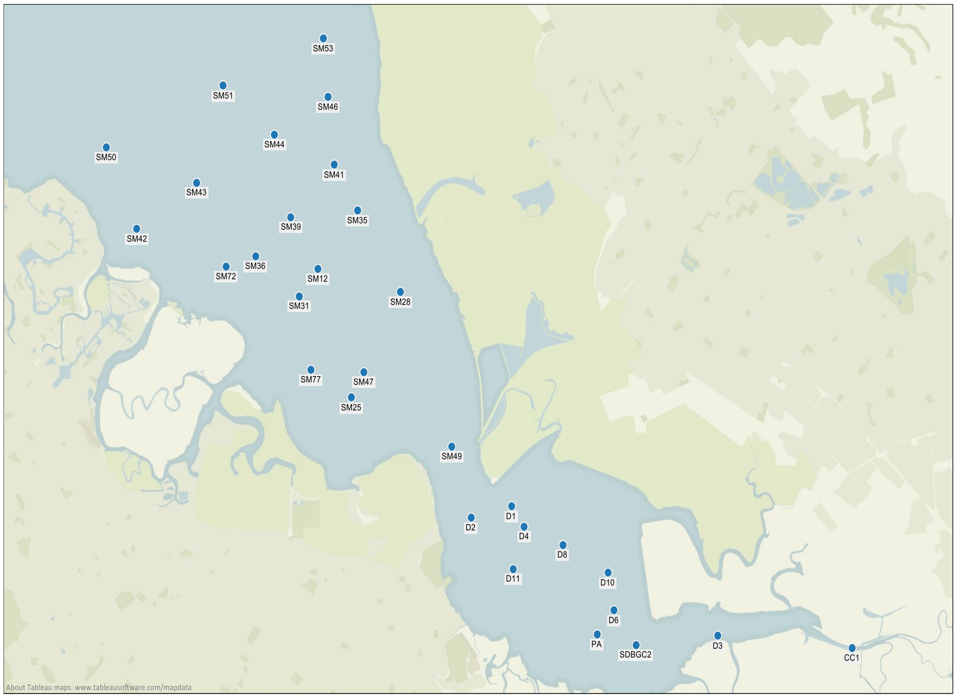

Figure 4.1. Station locations of benthic sampling in South Bay.

4.2.2 Benthic grazing estimates

All filter-feeding bivalve individuals (*Corbula (*formerly *Potamocorbula) amurensis, Venerupis japonica, Musculista senhousia, Mytilus c.f. edulis, Mya arenaria* and *Macoma petalum*) were removed from each sample and counted, and the longest length of each individual was measured. A size range (1.5 mm long and greater) of live animals was collected at each site during each month to calculate dry tissue weight for each size of animal. Animals were measured, dried at 60˚C (dry weight), weighed, ashed at 500˚C in a muffle furnace, and re-weighed (ash weight). A relationship between animal length and ash-free dry weight (AFDW=dry wt. – ash wt.) was then used to convert the measured animals from each benthic sample into biomass estimates for each species.

Bivalve community grazing rate estimates (*BG*) were based on bivalve biomass and species specific pumping rates (*PR's*) which were adjusted for a concentration boundary layer. Community *PR's* were based on published relationships: *C. amurensis*, 400L/g AFDW/day (Cole et al., 1992); *V. japonica*, 200L/g AFDW/day, (O'Riordan et al., 1993); *M. arenaria*, PR=aw^b, a=11.6, b=0.7, where w is tissue weight (Mohlenberg and Riisgard, 1979); and *M. edulis*, *PR*= aw^b, a=7.45, b=0.66 (Mohlenberg and Riisgard, 1979). Because there is no published relationship for *M. senhousia*, the *M. arenaria PR* relationship was used based on laboratory observations of excurrent siphon velocities for the two species. The *PR* for *M. petalum* was based on a relationship for a congener, *Macoma nasuta* (Meyhofer, 1985), and may be an overestimate because M*. nasuta* frequently lives in higher velocity environments than *M. petalum* and may be better adapted for the filter-feeding mode than *M. petalum*. The *PR* relationships were chosen because they were from studies where bivalve sizes were similar to those seen in this study. *PR* was adjusted for seasonal changes in water temperature using coefficients (Q_{10}) from studies of *M. edulis*: Q_{10} of 2.2 at 5˚C, 1.2 at 10˚C, and 1 at 15˚-20˚C (Winters, 1978).

Community pumping rates were converted to grazing rates by reducing *PR* to adjust for the presence of a concentration boundary layer. This adjustment was based on O'Riordan's (1995, Figure 7b) refiltration relationship, $n_{max} = 2.5/(s/d_0)$, where $n_{\max}$ is the maximum refiltration proportion (i.e. the proportion of water previously filtered), s is the distance between siphon pairs, and d_0 is the diameter of the excurrent siphon. The diameter of the excurrent siphon was changed throughout each year to reflect the change in average size of animals as the year progressed, and the distance between siphon pairs was based on density of animals observed in our benthic sampling assuming equidistant spacing within the 0.05 m^2 grab. The use of maximum refiltration proportion maximizes the effect of the concentration boundary layer resulting in a conservative grazing rate estimate. We assumed all bivalves grazed continuously.

4.2.3 Qualitative Description of the benthic community in South Bay (Thompson et al. 2007)

The main channel sediment is mostly mud and muddy sand, with sporadic occurrences of the bivalves *P. amurensis*, *Mya arenaria*, *Venerupis japonica, Macoma petalum,* and *Musculista senhousia* . These are all filter-feeding bivalves except *M. petalum* which is capable of filter feeding and deposit feeding. *Ampelisca abdita*, the tube-dwelling amphipod, can reach extremely high abundances in the channels (>40,000/m^2). The large populations of the maldanid polychaete *Sabaco elongatus*, a head down deposit feeder with tubes that can be close to a meter in length can structure the rest of the community because of their very dense populations and relatively large tubes (5-6mm in diameter). Surprisingly, *A. abdita* and the bivalves mentioned above can live among *S. elongatus* tubes which extend 1-2 cm above the substrate surface. Many of the species in this assemblage are patchy in space and time with some, like *M. senhousia* and *A. abdita*, having very high abundance one year and low abundance the next year. Other tube-dwelling species, including several species of *Corophium* amphipods and spionid worms (mostly *Streblospio benedicti*) can reach moderate abundances. The cumacean *Nippoleucon hinumensis*, the polychaete *Cirriformia spirabrancha*, the oligochaete *Tubificoides* species, and the bivalve *Theora lubrica* are common surface and subsurface deposit feeders in the channel.

Benthic communities in the South Bay sloughs are similar to channel communities with the addition of large populations of the amphipods *Monocorophium alienense* and *Grandidierella japonica*, the bivalve *Macoma petalum* and the polychaete *Neanthes succinea*.

The benthic community composition in the shallow water is similar to that seen in the main channel with the exception of *S. elongatus* which is less common in the shallows and the presence of *Gemma gemma* which is more common in the shallow water. The bivalves *P. amurensis*, *Mya arenaria*, *Venerupis japonica, Macoma petalum,* and *Musculista senhousia* can be common inhabitants in the shallow water depending on the year, but show very strong seasonal patterns with declines in bivalve abundances to near zero each winter/early spring. The shallow water bivalves are therefore mostly annual species in this habitat with peaks in abundance occurring in late spring/early summer. The amphipods (*Corophium heteroceratum* and *Ampelisca abdita*) have two annual abundance peaks most years except during dry years when *A. abdita* seems to persist through the winter. As seen in the channel, the cumacean *Nippoleucon hinemensis* is common and peaks in spring of most years.

4.3 Results/Discussion

4.3.1 Benthic Community Dynamics in the Intertidal

Nichols and Thompson (1985a) described the composition of the benthic community during 10 years of study at three intertidal sites near Palo Alto as opportunistic species that persisted because of their rapid response to physical and chemical disturbances of the mudflat. The numerically dominant species were tube dwelling amphipods (*Ampelisca abdita*) and polychaetes (*Streblospio benedicti*) that switch between filter feeding and deposit feeding (Riisgard and Kamermans 2001), and a tiny bivalve, *Gemma gemma*, that lives near the sediment surface and filter feeds what is likely to be a mix of pelagic based food and resuspended particles from the mudflat surface. Abundance fluctuations of each of these three species were significantly correlated between the three stations at both the intra- and inter-annual time scale. However, only *A. abdita* abundance varied with a seasonally predictable pattern, which was the result of recruitment twice per year. It was hypothesized by these authors that species fluctuations are likely a response to factors in addition to temperature in this temperate estuary and therefore a clear seasonal pattern may be difficult to discern. Disturbances that disrupted the community included seasonal erosion and accretion of the mudflat surface and transport and deposition of macroalgae onto the site from other locations. It took most of a year for the community to return to the mudflat following the anoxia from the macroalgae deposit and the classic opportunistic species, *Capitella capitata*, made a short appearance at that time before the previous species returned. Prescient of what was to become the pattern in the following years, abundance of two of the opportunistic species, *A. abdita* and *S. benedicti,* declined significantly in 1982 and 1983, respectively, and returned in the mid-1980's to intermediate densities. The authors attributed the short declines in these two species to stress and mortality caused by an increase in freshwater inflow and rainfall during these two wet years.

The highest intertidal station has continued to be sampled with a break between 1991 and 1998 (Figure 4.2) and this extended dataset allows us to examine community changes over multiple decades. The number of species at this station has shown a lot of variation but does not show a significant trend. As was seen in 1982 and 1983, the two opportunistic species *S. benedicti* and *A. abdita* abundance declined between 1990 and 2000. Unlike before, the abundance of both species has remained low since 1990 (Figure 4.3). Two non-indigenous species have invaded the benthic community of South Bay during this period and they may have some effect on *S. benedicti* and *A. abdita* although there is no hypothesized mechanism for the interference. The filter-feeding bivalve *Potamocorbula amurensis* invaded in 1988 (Carlton et al. 1990) as did the small surface-deposit feeding cumacean *Nippoleucon hinnumensis*. *P. amurensis* has been a sporadic member of the benthic community at this station but *N. hinnumensis* has consistently maintained its presence since it invaded. The period of decline for *A. abdita* and *S. benedicti* has also been a period when the deep deposit feeding polychaete *Heteromastus filiformis* has been successful in the mudflat (Figure 4.4a) and when *G. gemma* has shown some periods of large abundance increases (Figure 4.4b). Although *H. filiformis* may be considered opportunistic (Can et al 2009), it is usually considered a species that arrives at the second to third successional stage after an initial invasion by opportunists following a disturbance(van Colen et al. 2008). *H. filiformis* has recently declined in abundance but there has been no coincident increase in other species. The substrate at this station is frequently covered by microphytobenthos and it is likely that the high nutrient concentration helps maintain that community. The persistence of *Macoma petalum,* a deep burrowing non-obligate filter-feeder*,* at this location is likely related to this benthic primary production as the mudflat is covered with star patterns made by *M. petalum's* siphons vacuuming the surface of the mudflat (Figure 4.4c). We have also seen periodic growths of patchy macroalgae that could increase if turbidity declines sufficiently and the nutrient concentration is high. Although we have seen several changes in the community over the years, none of the changes would indicate that there is a recurring anoxia problem that forces the community to re-establish itself. The disappearance of *A. abdita* and *S. benedicti* and appearance of *H. filiformis* may signal periods of stress followed by less stress. If so it has never gotten so stressed that *S. benedicti* and *A. abdita* return.

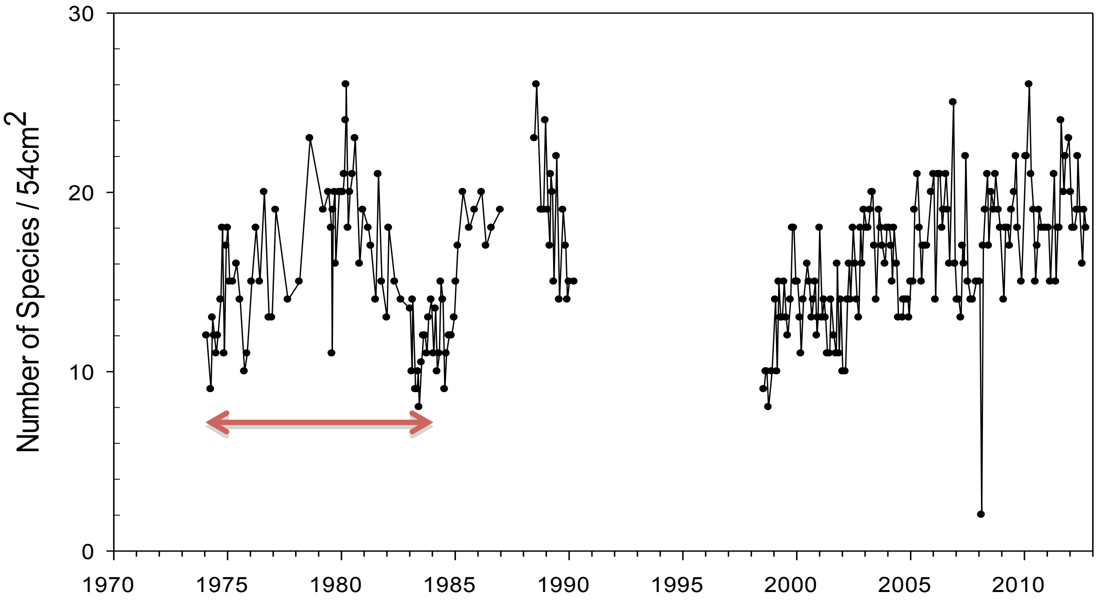

Figure 4.2. Number of species present at station FN45 (intertidal mudflat), Palo Alto, Calif., 1974–2012. Red arrow shows the period reported in Nichols and Thompson (1985a). No collections were made between 1991 and 1998.

4.3.2 South Bay benthic studies – San Mateo Bridge to Coyote Creek

We examined the benthic community composition in these studies to determine if there is any indication between 1993 and 2009 of a sufficient change in organic carbon loading to cause a shift in the functional feeding groups of the benthos, or cause an increase in biomass of the bivalve community. We also monitored the bivalve grazing rates during this period to help understand losses or gains in phytoplankton biomass; without this analysis we might interpret a change in phytoplankton biomass to be due to nutrient effects whereas phytoplankton biomass may also reflect increases and decreases in benthic filter-feeders and biomass.

4.3.2.1 Community Description 1993-2009

Because the benthic community is so dependent on hydrology and salinity to structure its composition, we compare similar water years in the discussion below. We will compare the benthic community in two critically dry years (1994 vs. 2008), two near normal hydrologic periods (1993 vs. 2004) and two wet years (1995 vs. 2006). We will examine each pairing of data by looking at Multi-Dimensional Scaling (MDS), a method that allows us to visualize the similarity in the benthic community composition at different stations for each pair of years. We will then discuss if shifts in feeding function is reflected in the MDS differentiation of the data.

The normal and dry hydrological years showed the most separation in the MDS analyses of the benthic community (Figure 4.5). The differences in the normal year comparison were driven by the presence of bivalves in 1993 and not in 2004. Potamocorbula in particular appears to structure the community and many surface deposit feeders do not co-occur with this bivalve. Once the bivalves were missing, the increase in surface deposit feeders (particularly in areas north of Dumbarton Bridge, Figure 4.6) was likely due to the accumulation of phytoplankton on the bottom which is normally consumed by the large filter feeders before it reaches the mudflat surface. The dry hydrological year comparison was between two years when Potamocorbula were absent from both years. Small filter feeders/surface deposit feeders were more common in 2008 than in 1994 It is not surprising that the benthic communities in the wet years (1995 and 2006) would be the most similar because much of the species selection is based on physiological tolerance and the low salinity in wet years likely narrows the species to a limited number of freshwater tolerant species. In addition, bivalves are present in both years although with lower abundance in 2006. Similar to the other comparisons between years the benthic communities varied most by an increase in small filter-feeders and surface deposit feeders in 2006 relative to 1995.

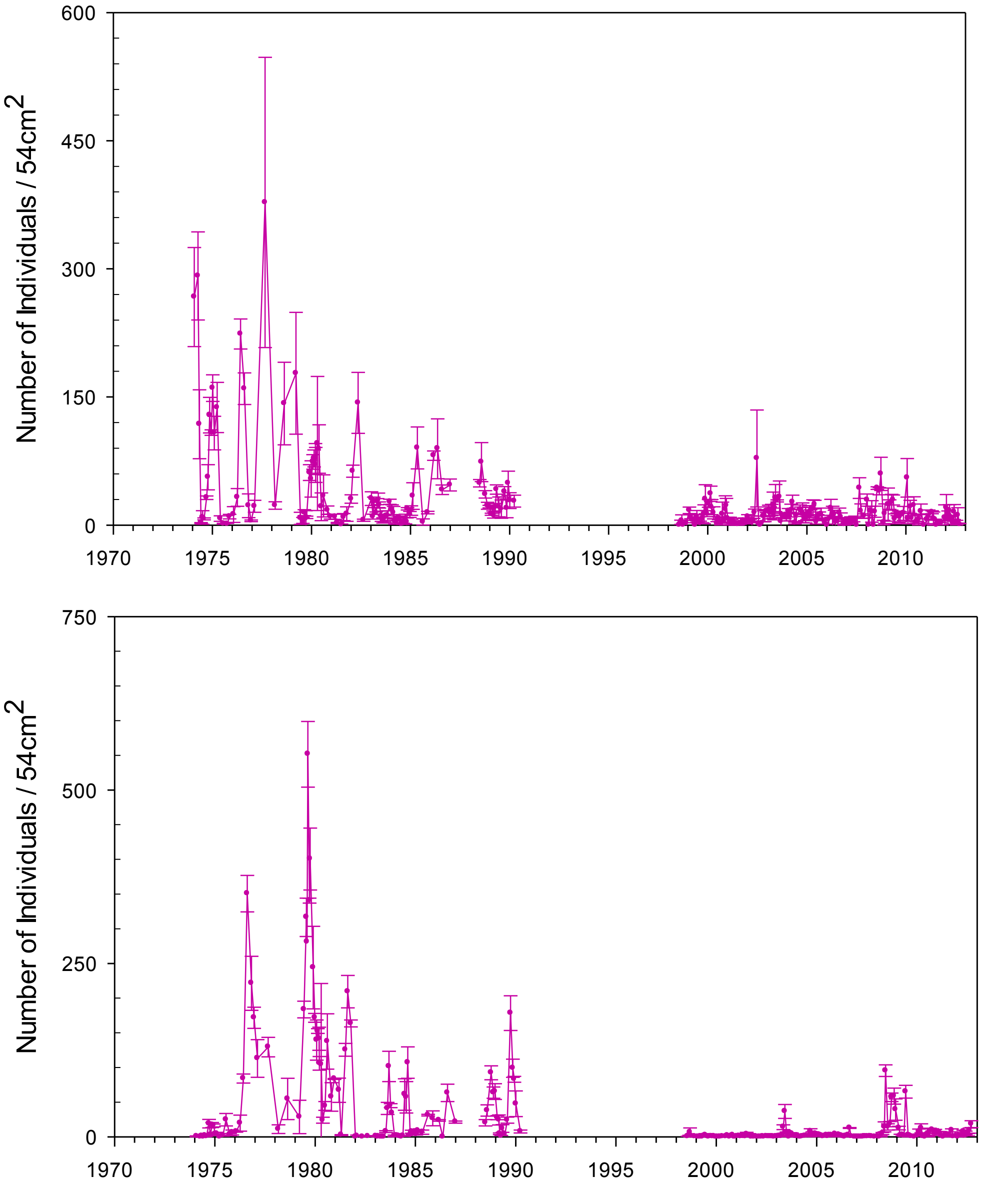

Figure 4.3. Monthly average abundance of All Species at station FN45 (intertidal mudflat), Palo Alto, Calif., 1974–2012. Error bars are standard deviation of 3 samples collected at the site. No collections were made between 1991 and 1998. Top = *Streblospio*; Bottom = *Ampelisca.*

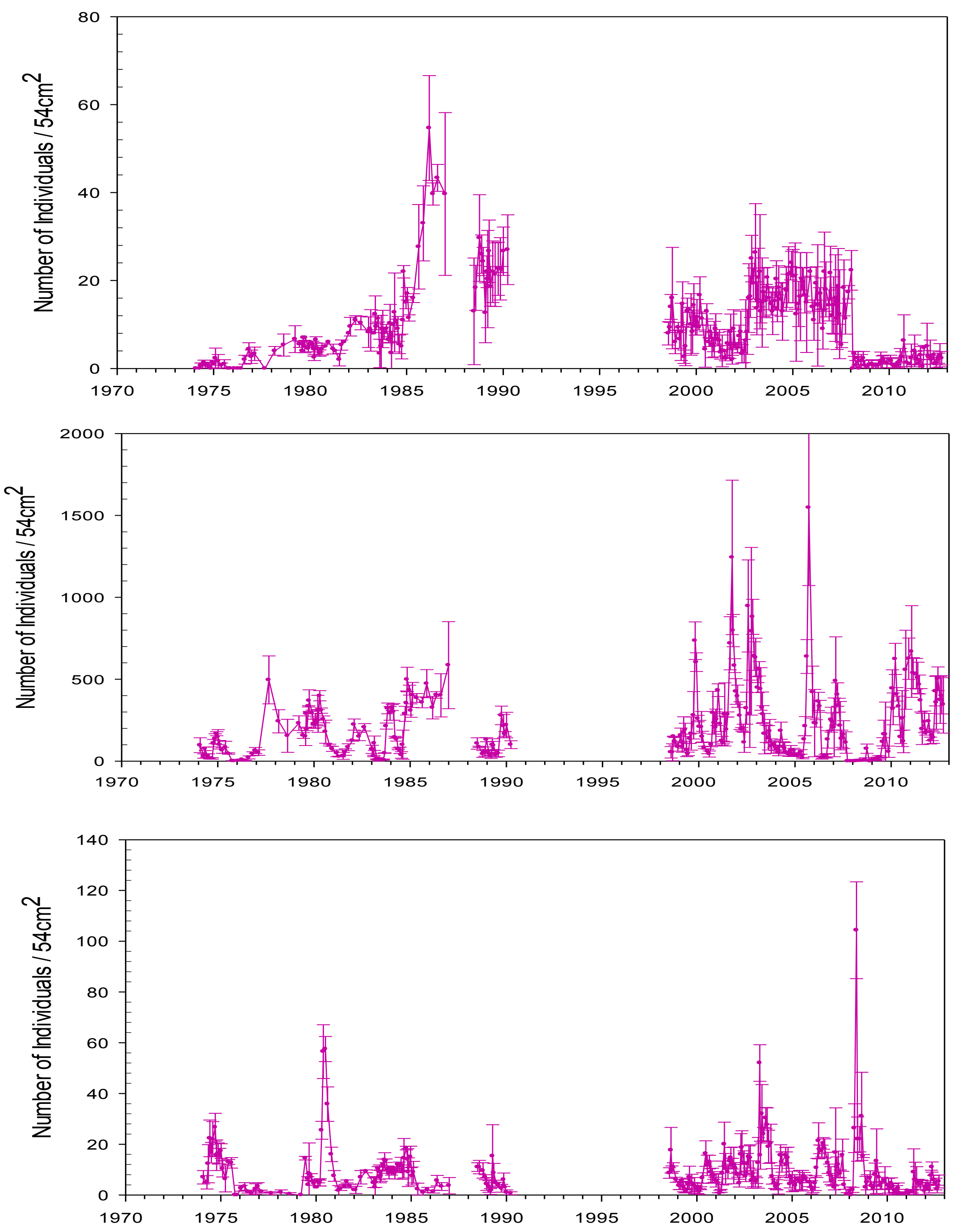

Figure 4.4. Monthly average abundance of All Species at station FN45 (intertidal mudflat), Palo Alto, Calif., 1974–2012. Error bars are standard deviation of 3 samples collected at the site. No collections were made between 1991 and 1998. Top = *Heteromastus filiformis;* Middle = *Gemma*; Bottom = *Macoma petalum*

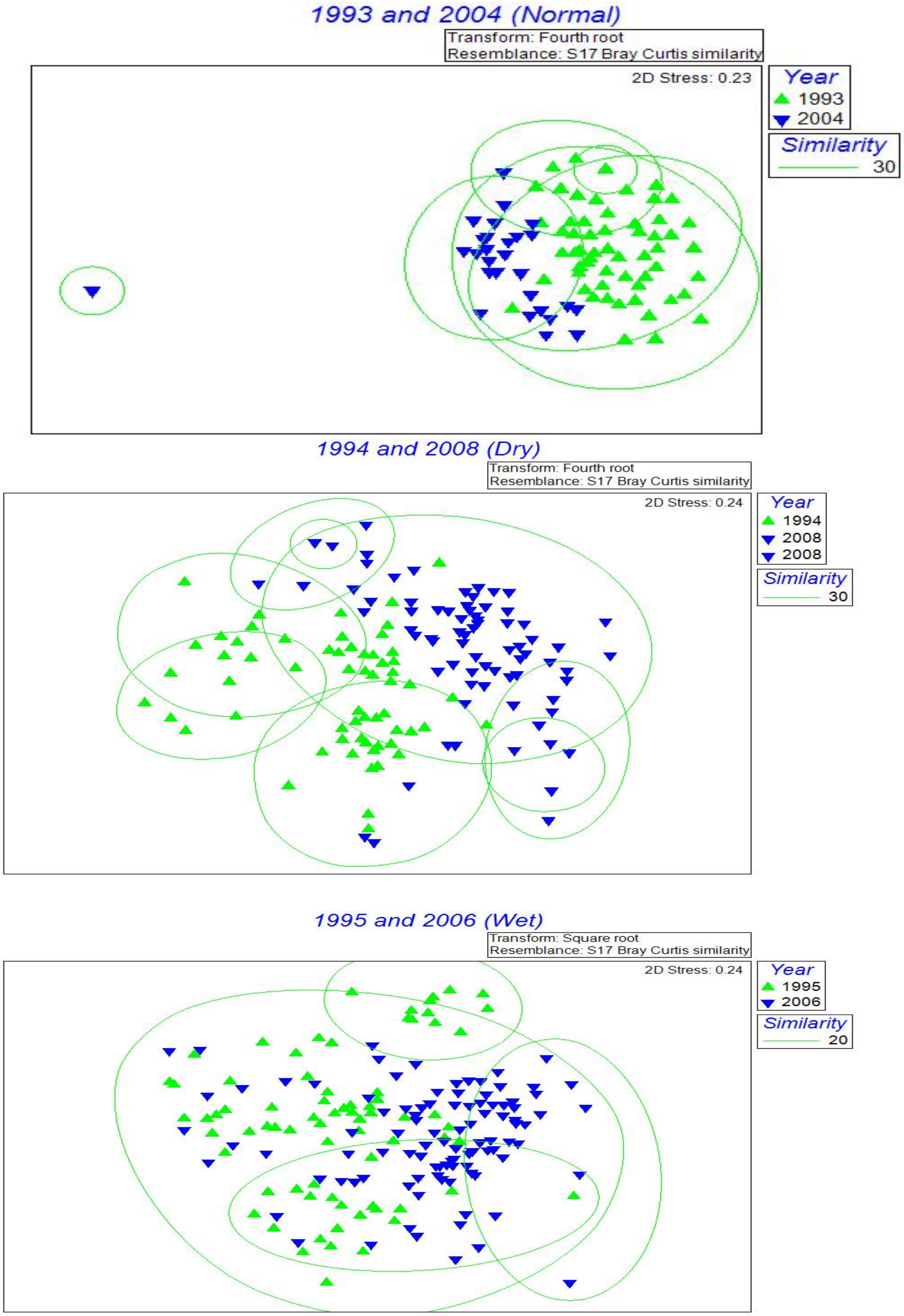

Figure 4.5. Multi-Dimensional Scaling (MDS) Plots comparing Normal (top), Dry (middle) and Wet (bottom) water years.

Figure 4.6. Change in percentage of functional feeding groups over time.

4.3.2.2 Comparing Benthic Community Function Over the Long Term

A comparison of functional feeding groups shows that filter feeders dominate the abundance of individuals in this system in all years and locations and that large filter feeders (large bivalves and ascidians) declined from 2004 onward (Figure 4.6). The most consistent differences in the benthic community functional feeding groups between the northern and southern regions (reference point is Dumbarton Bridge) is the higher percentage of filter feeders in the southern region and the lower percentage of subsurface deposit feeders in the southern than the northern communities. This was a bit of a surprise as the southern reach is known to be depositional and we expected that buried labile organic carbon might be in higher concentrations in this region.

When the large bivalves and large ascidians are removed from the percentage of filter feeders, the percentage of the total abundance that are small filter-feeding species (mostly amphipods) remains variable in time in both the northern and southern shoals. However, the benthic community in the northern shoal had an increase in surface deposit feeding species (mostly polychaetes and amphipods) coincident with the decrease in bivalves. This shift in functional feeding groups in the north probably reflects the increase in food availability on the mud surface as the phytoplankton sinks to the bottom. This shift may also occur in the southern shoal but the data as presented here uses species abundance, not biomass, which would be a better measure of these differences.

4.3.2.3 Dynamics of filter-feeding bivalve function in the ecosystem

The spatial studies in South Bay have given us a time series of bivalve biomass in the spring, mid-summer, and fall, which gives us an opportunity to examine grazing rates during the spring bloom, after the spring bloom and during the fall bloom. The advantage of showing the data spatially, as shown in Figures 4.7 and 4.8, instead of as an annual average, is that we can see the importance of the spatial changes. The pattern until 1999 (Figure 4.7) consistently showed low biomass in spring except in 1994 in the southern reach, followed by increasing biomass in summer and fall. We see that there has been a shift in bivalve biomass starting in 2004 (Figure 4.8). With the exception of one location in the northern channel (due entirely to a localized population of Musculista) the biomass levels have been very low north of the DB since 1998. The only period when bivalves moved into the northern embayment, July 2006, did not include the entire bay and was short lived with the biomass greatly declining in fall.

The regular fall reduction in bivalves in the shallow water each year prior to 1999 allowed the spring phytoplankton bloom to develop in the South Bay. Each fall, predation by migratory and resident birds (Thompson et al 2008), fish, and invertebrates (Cloern et al 2007) decimated the shallow water bivalve communities in South and San Pablo Bays (Poulton et al 2002, 2004, Richman and Lovvorn 2004). This elimination of bivalve grazing in the shallow water allows the phytoplankton to grow if light and mixing are not limiting (Thompson et al. 2008). Bivalves in the shallow water are thus essentially an annual species with larvae settling each spring followed by rapid growth which allows them to become a controlling factor on the phytoplankton by late spring and summer. However, as shown by increases in phytoplankton biomass in South Bay during the strong upwelling events after 1998 (Cloern et al 2007), the relationship between benthic grazers and phytoplankton is not always straightforward. (Cloern et al 2007). This is a cautionary story, as it is important that we not misinterpret increases in phytoplankton biomass such as were seen after 1998 as caused entirely by water quality changes.

Bivalve biomass, shown in Figures 4.7 and 4.8, is a good proxy for benthic grazing rates. The biomass levels seen here can be converted to grazing rates ranging from 1-20 $m^3m\text{-}^2d^{-1}$ in 1993-1995, 1-10 $m^3m\text{-}^2d^{-1}$ in 1996 and 1994, and 1-15 $m^3m\text{-}^2d^{-1}$ in the post 2000 period (using the method of Thompson et al. 2008). In all but a few locations the grazing rates after 2000 were in the 1-2 $m^3m\text{-}^2d^{-1}$ range. We have shown biomass distribution of bivalves only and we realize it would be useful to do the same for other filter feeding species, such amphipods but biomass has never been calculated for these species. Jones et al. (2009) used very liberal pumping rates for amphipods, the most likely candidates for imposing a large grazing loss to the phytoplankton, and showed the maximum grazing rate attained in Suisun Marsh there was <10 $m^3m\text{-}^2d^{-1}$. This is of the same order as the grazing loss from the bivalves and thus it may be worth pursuing amphipod grazing as an important factor in our understanding of phytoplankton dynamics in the south bay system.

4.4 Data Gaps

What can we learn?

To fill the present data gaps we propose:

1. Process remaining seasonal, spatially intensive benthic community sampling studies samples from 2009 and 2010 to present (Table 4.1).
2. Continue three, seasonal, spatially intensive benthic community sampling studies for species composition and biomass. Sort samples, initially, to the family level to identify a species' functional feeding group.
3. Process samples from wet years 1997-1998.
4. Biomass estimates of other members of the benthic community.

These will help answer the following:

1. The potential role grazing rate of benthos plays in regulating phytoplankton biomass.
2. Identify changes in the function and composition of the benthic community to make sure another species has not supplanted the role of bivalves (e.g. an exotic that is non palatable to predators would certainly change things and decrease the phytoplankton – i.e. exotic sponge with high filtration rate).
3. How smaller scale function (smaller animals) contribute to the true loss of pelagic food to the benthos.

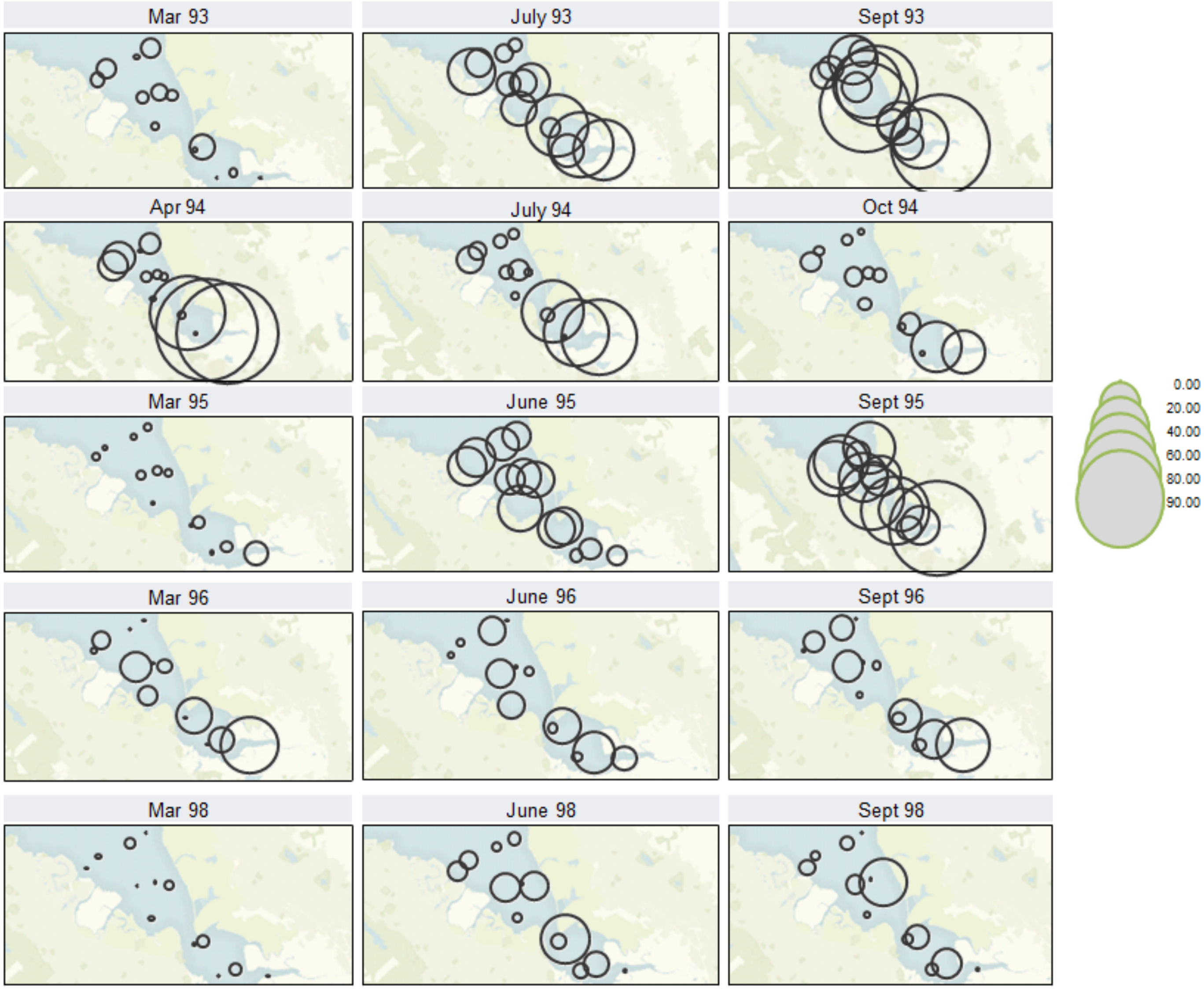

Figure 4.7. Time series of bivalve biomass in spring, mid-summer and fall in South San Francisco Bay.

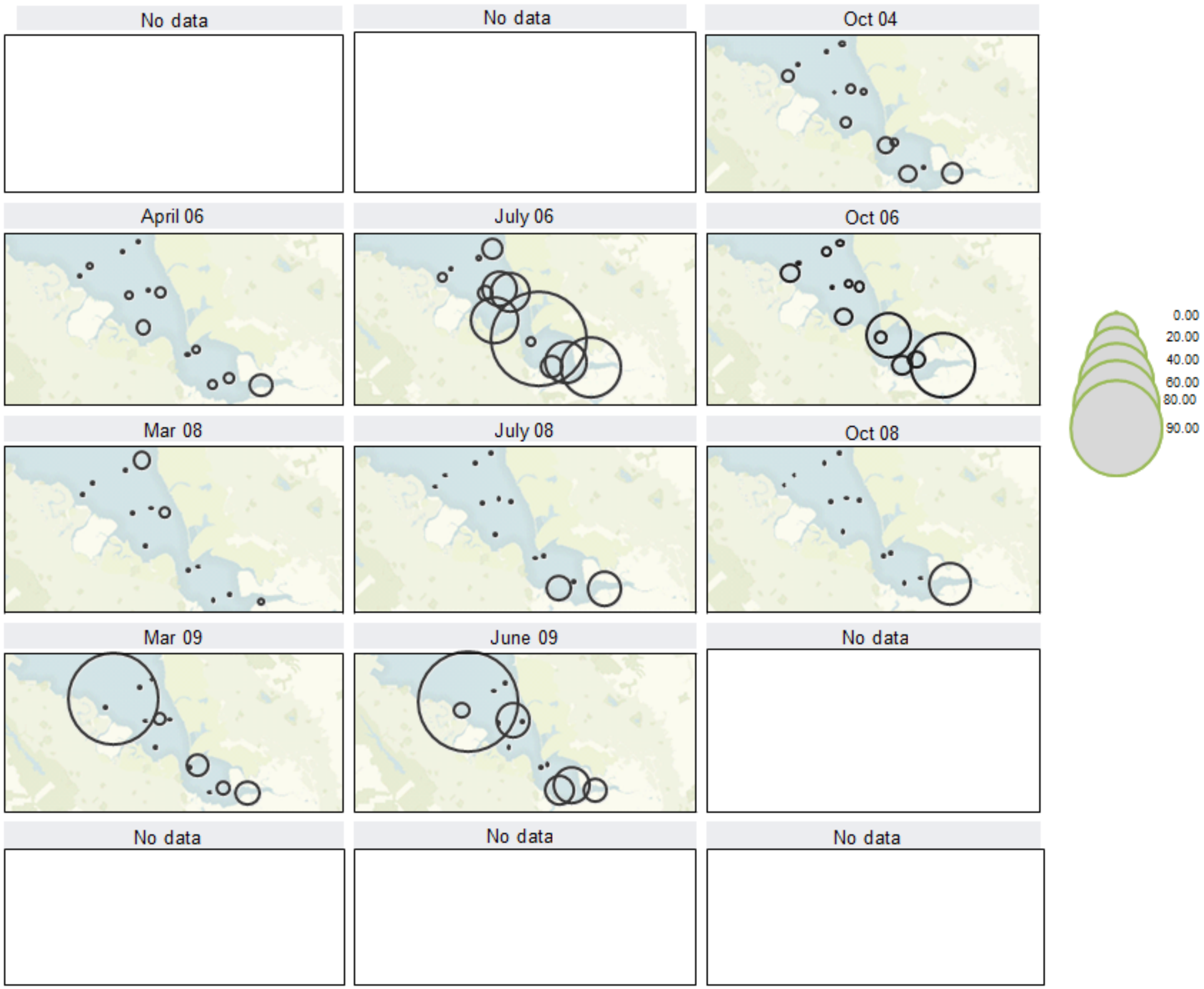

Figure 4.8. Time series of bivalve biomass in spring, mid-summer and fall in South San Francisco Bay.

6/1/1997	4/26/2006	10/11/2006	10/30/2007	10/15/2009	3/16/2010	7/7/2010	10/5/2010	3/15/2011	6/23/2011	9/7/2011	4/3/2012	6/13/2012	10/11/2012	4/17/2013	5/7/2013	8/1/2013
								South of Dumbarton								
D1 D10 D11 D2 D3 D4 D6 D8 D9 PA			D10 D11 D2 D3 D6 PA	CC1 D10 D11 D2 D3 PA SDBGC2	CC1 D11 D2 D3 D6 PA SDBGC2	CC1 D10 D11 D2 D3 D6 PA SDBGC2	D1 D10 D11 D2 D4 D6 D8 PA	CC1 D1 D10 D11 D2 D3 D4 D6 D8 PA SDBGC2	CC1 D1 D10 D11 D2 D3 D4 D6 D8 PA SDBGC2	CC1 D1 D10 D11 D2 D3 D4 D6 D8 PA SDBGC2	CC1 D1 D10 D11 D2 D3 D4 D6 D8 D9 PA SDBGC2	CC1 D1 D10 D11 D2 D3 D4 D6 D8 PA SDBGC2	CC1 D1 D10 D2 D3 D4 D6 D8 PA SDBGC2	D1 D4 D8	CC1 D1 D11 D2 D3 D4 D6 D8 PA SDBGC2	CC1 D1 D10 D11 D2 D3 D4 D6 D8 PA SDBGC2
								North of Dumbarton								
SM06 SM13 SM23 SM25 SM27 SM28 SM30 SM31 SM35 SM37 SM38 SM39 SM41 SM42 SM43 SM44 SM46 SM47 SM49 SM50 SM51 SM53 SM54	SM5 SM50 SM51 SM52 SM53 SM54 SM55 SM56 SM58 SM59 SM6 SM60 SM61 SM62 SM65 SM70 SM71 SM72 SM77 SM81 SWH1	SM5 SM50 SM51 SM52 SM53 SM54 SM55 SM56 SM58 SM59 SM6 SM60 SM61 SM62 SM65 SM70 SM71 SM72 SM77 SM81 SWH1	SM12 SM25 SM28 SM31 SM35 SM36 SM39 SM41 SM42 SM43 SM44 SM46 SM47 SM49 SM50 SM51 SM53 SM72 SM77	SM12 SM25 SM28 SM35 SM39 SM41 SM42 SM44 SM46 SM51 SM53 SM72 SM77	SM12 SM25 SM31 SM35 SM36 SM39 SM41 SM42 SM43 SM44 SM47 SM49 SM50 SM51 SM53 SM72 SM77	SM12 SM25 SM28 SM31 SM35 SM36 SM39 SM41 SM42 SM43 SM44 SM46 SM47 SM49 SM50 SM51 SM53 SM53 SM72 SM77	SM12 SM25 SM28 SM31 SM35 SM36 SM39 SM41 SM42 SM43 SM44 SM46 SM47 SM49 SM50 SM51 SM52 SM53 SM72 SM77	SM12 SM25 SM28 SM31 SM35 SM36 SM39 SM41 SM42 SM43 SM44 SM46 SM47 SM49 SM50 SM51 SM53 SM72 SM77	SM12 SM25 SM28 SM31 SM35 SM36 SM39 SM41 SM42 SM43 SM44 SM46 SM47 SM49 SM50 SM51 SM53 SM72 SM77	SM12 SM25 SM28 SM31 SM35 SM36 SM39 SM41 SM42 SM43 SM44 SM46 SM47 SM49 SM50 SM51 SM53 SM72 SM77	SM12 SM25 SM28 SM31 SM35 SM36 SM39 SM41 SM42 SM43 SM44 SM46 SM47 SM49 SM50 SM51 SM53 SM72 SM77	SM12 SM25 SM28 SM31 SM35 SM36 SM39 SM41 SM42 SM43 SM44 SM46 SM47 SM49 SM50 SM51 SM53 SM72 SM77	SM12 SM25 SM28 SM31 SM35 SM36 SM39 SM41 SM42 SM43 SM44 SM46 SM47 SM49 SM50 SM51 SM53 SM72 SM77	SM31 SM36 SM43 SM47 SM49 SM50	SM12 SM25 SM28 SM31 SM35 SM36 SM39 SM41 SM42 SM43 SM44 SM46 SM47 SM49 SM50 SM51 SM53 SM77	SM12 SM25 SM28 SM31 SM35 SM36 SM39 SM41 SM42 SM43 SM44 SM46 SM47 SM49 SM50 SM51 SM53 SM72 SM77

** This is just the Blitz data

This doesn't include the higher temporal resolution data (up to 13 stations monthly,1990-1998 and 2004-present)

🟥 Archived / need sorting

Table 4.1. Archived samples that need to be sorted/processed in order to fill data gaps.

5. Phytoplankton Biomass: Trends and Potential Drivers

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5.1 Introduction

This section examines available data and past studies related to phytoplankton biomass, and factors that affect production rates or biomass accumulation, for Lower South Bay and southern South Bay. The section's specific goals include:

a. Characterize current conditions of phytoplankton abundance in LSB, including spatial and seasonal variability, based on chl-a concentration.
b. Analyze long-term trends in phytoplankton biomass, including how biomass varies seasonally and spatially
c. Estimate the relative importance of several regulating factors such as light levels and grazing, and the the degree of uncertainty in with those estimates
d. Identify major data gaps and recommend future monitoring or studies needed to address these gaps and inform management decisions

5.2 Data Sources

Table 5.1 and Figure 5.1 present the main data sources and station locations used for Section 5. Data analyzed include measurements taken during bi-weekly to monthly surveys of water quality parameters and phytoplankton abundance along a transect of the entire Bay conducted by the USGS starting in the 1970s (Figure 5.1). These surveys primarily sampled stations from deep waters following the main channels of the Bay. We also discuss results from published studies in the deep channel and along the shallow shoals of South Bay and LSB that examined factors that regulate biomass (e.g., Cloern, 1989; Powell, 1989; Huzzey, 1990; Thompson et al., 2008; Lucas et al., 2009). High frequency data from stations at Dumbarton Bridge and in Alviso Slough are also utilized.

Table 5.1 Data Sources

Organization	Location	Dates	Notes
USGS-Menlo Park[1]	N. of Dumbarton Br.	1975-present	Discrete chl-a; 1-2x monthly; main channel
	S. of Dumbarton Br.	1975-1980 1992-present	Discrete chl-a; 1-2x monthly; main channel
SFEI	Dumbarton Br.	Jul 2013-present	Chl-a fluorescence; high-frequency; main channel
	Alviso Slough	Sept 2013-present	Chl-a fluorescence; high-frequency; slough

[1] sfbay.wr.usgs.gov/access/wqdata/

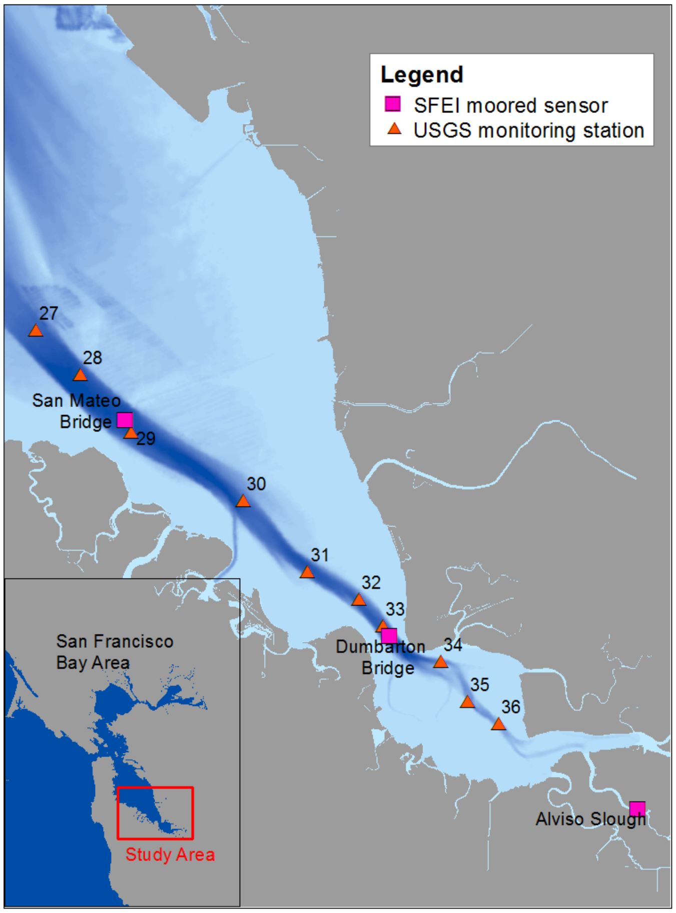

Figure 5.1 Sampling stations that are part of bi-weekly to monthly sampling aboard the USGS *R/V Polaris*

5.3 Results and Discussion

5.3.1 Spatial variability of phytoplankton biomass

Figure 5.2 summarizes year-round surface phytoplankton biomass concentrations (as chl-a) at Central Bay, South Bay, and LSB stations based on bi-weekly to monthly data from 2008-2013. Median biomass concentrations were greater in LSB and at stations south of s27 than those north of s27. At the stations s33 and s34, which had the highest biomass, median concentrations were ~50% greater than stations north of s27, while 25%ile and 75%ile values were nearly a factor of 2 greater. Station s27 is situated near the San Bruno Shoal, a shallow bottom feature that strongly limits longitudinal mixing, a factor that likely plays a role in the observed differences in concentrations north and south of this location.

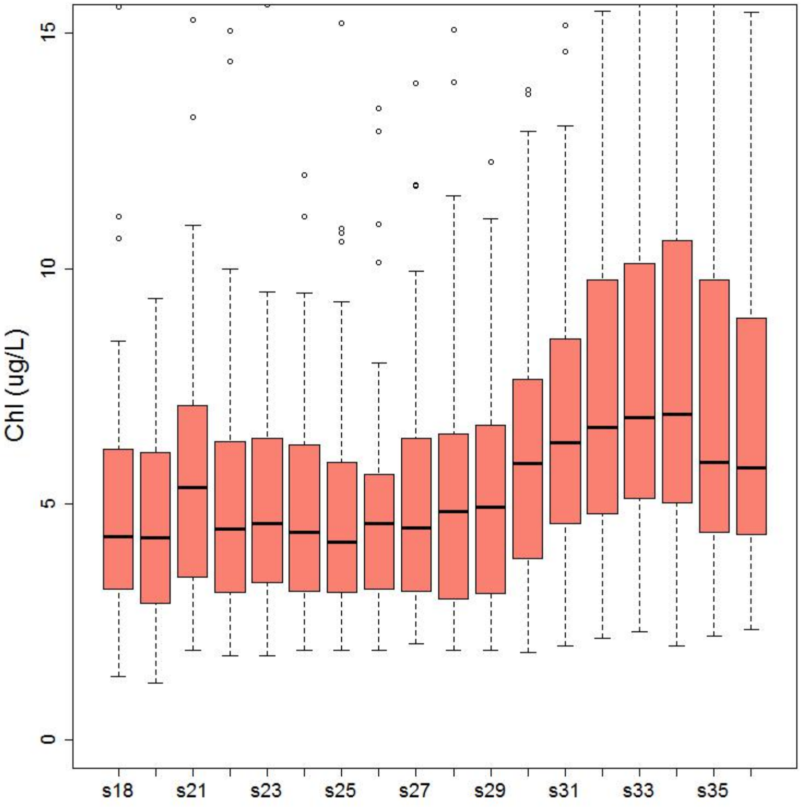

Figure 5.2 Surface chl-a concentrations at USGS stations along the deep channel extending from Lower South Bay (s36) to Central Bay (s18). Black lines represent medians; extents of red bars indicate the 25% and 75% percentiles; error bars are ±95%; and points indicate statistical outliers. Some outliers are cutoff in this graphic because of the maximum for the y-scale. Data: USGS

5.3.2 Temporal variability in biomass and in the mechanisms that regulate productivity

Time series data at LSB and South Bay stations show that phytoplankton biomass concentrations varied in a highly-periodic manner, with low baseline levels interspersed with sharp peaks each year (Figure 5.3). While data availability is much greater for stations s32-s27 that s36-s34 for years earlier than 1992, both groups have comparably dense data from 1992-present.

Maximum chl-a levels in LSB and South Bay have typically been observed during spring blooms in March or April (Figure 5.4). A second feature is slightly higher chl a concentrations at a number of the south Bay stations in August and September as compared to other summer and fall months (Figure 5.4; Cloern et al. 2007; Cloern and Jassby 2012). These modest fall blooms appear to be a fairly new phenomenon, as discussed in Section 4.3.3.

Observed biomass concentration –at any point in space and time - is the result of the instantaneous balance of multiple processes whose rates vary: productivity; mixing and export/import; grazing; and settling. Observations over the past 3 decades create the basis for a conceptual model of factors that regulate phytoplankton blooms in SFB (e.g., Cloern 1996; see SFEI 2014 for an overview). This conceptual model is most relevant to South Bay, and with some applicability to LSB; however, more investigation is needed to quantify the relative importance of processes in LSB.

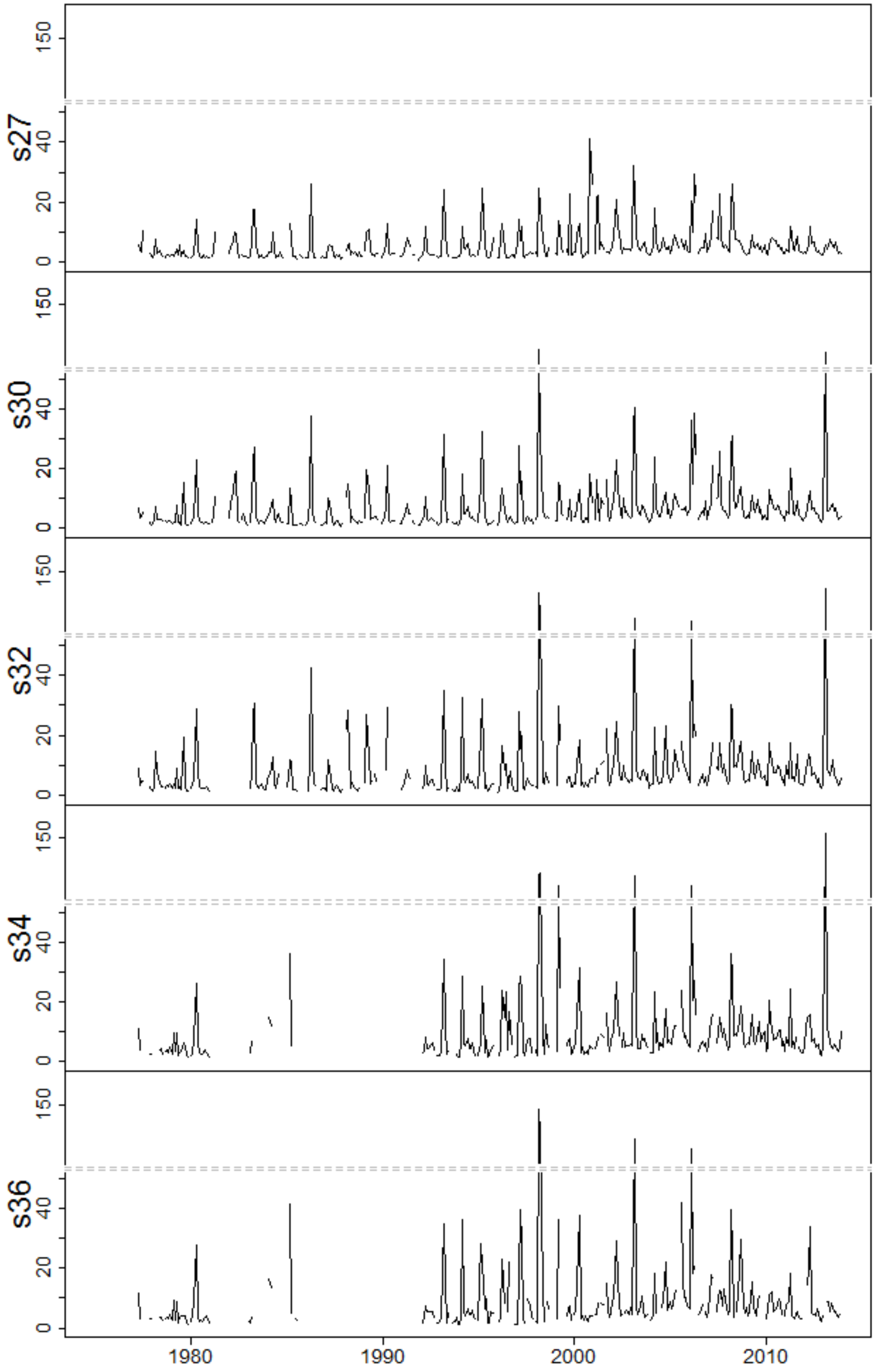

Figure 5.3 Time series of phytoplankton biomass (chl-a, µg/L) at stations s36-s27. When multiple samples were collected in a single month, monthly average values are plotted. Data: USGS.

Light availability: High suspended sediment concentrations in SFB make its photic zone - the depth at which light levels are 1% of incident light – a relatively thin layer of the water column, typically only 1-2 m (Cloern et al., 1985). Throughout many of its habitats and much of the year, phytoplankton growth rates in South Bay and Lower South Bay are often considered light limited (ref). Factors that influence suspended sediment concentrations in turn can influence phytoplankton growth rates (e.g., wind- and tidally-driven sediment resuspension and mixing).

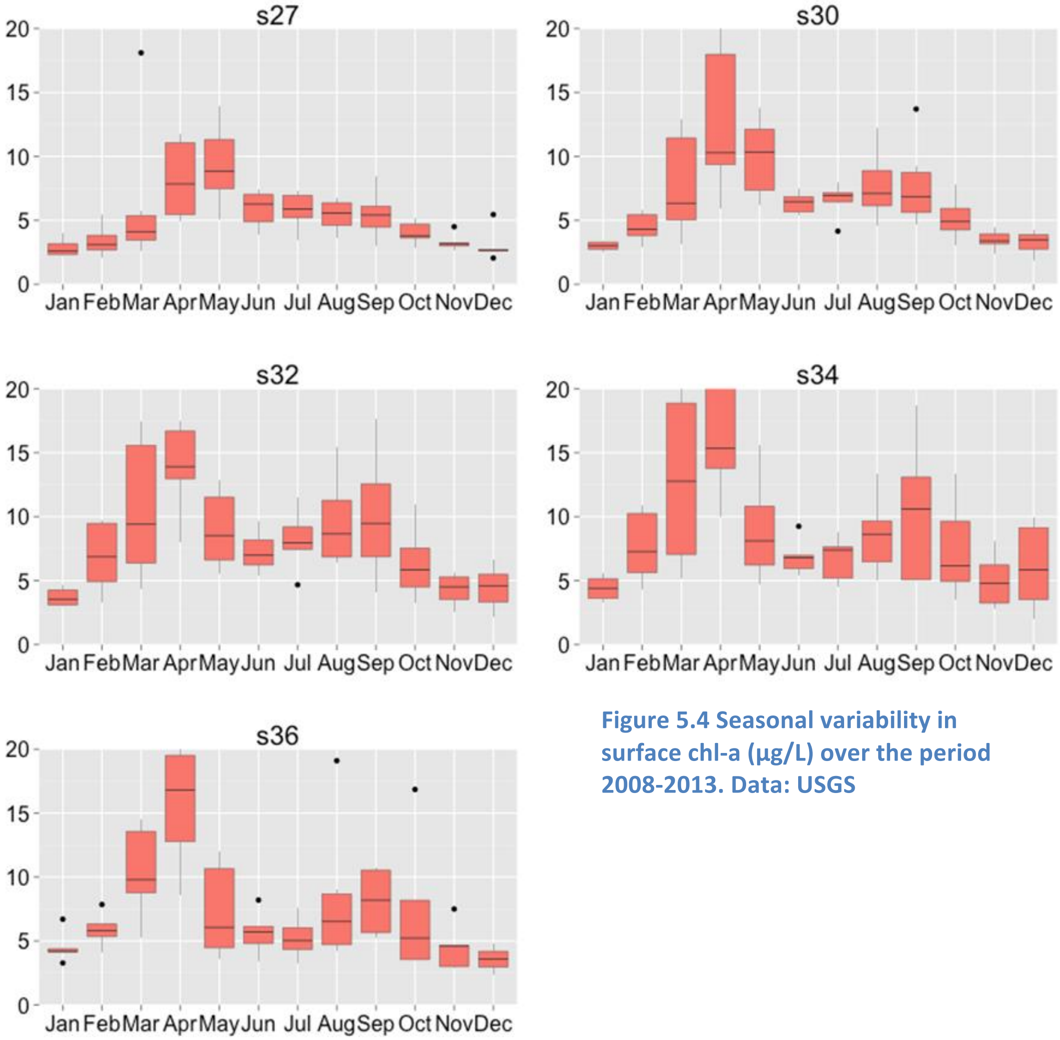

Figure 5.4 Seasonal variability in surface chl-a (µg/L) over the period 2008-2013. Data: USGS

Vertical Mixing: Vertical mixing works in conjunction with low light penetration to control biomass production. Vertical water movement controls the average amount of time phytoplankton remain within the light-rich surface photic zone, and thereby may determine if, when, and where phytoplankton blooms develop, and also when they terminate (Cloern 1991; Lucas et al., 1998). When the water column is vertically well-mixed, the amount of time phytoplankton spend in the photic zone decreases in proportion to water column depth. Stratification - vertical layering of the water column– develops when less dense freshwater layers overlay more-dense salty layers, results most commonly in SFB from differences in salinity. The density difference limits vertical mixing and allows phytoplankton to reside in the relatively thin (e.g., 1-3 m), light-rich surface layer, as opposed to being moved over the entire water column. When confined to the surface layer, phytoplankton harvest more light, resulting in higher growth rates. Factors that influence whether stratification occurs, and how long it persists, therefore have an important influence on productivity and biomass accumulation. SFB experiences strong tidal mixing which acts to break down stratification by vertically-mixing the water column (Cloern, 1991). Cloern (1996) observed that blooms along the deep channel of South Bay generally developed in March, when periods of weak tidal mixing co-occurred with sufficient freshwater input to allow stratification to develop and persist for 10-14 days. The termination of these blooms corresponded with increased tidal energy that vertically-mixed the water column (Cloern 1996). Tidal mixing intensity varies periodically: two tidal cycles per day with different mixing energies; the spring/neap cycle over which tides vary in magnitude on a ~14 day cycle; and twice-annual periods of lowest sustained tidal mixing energy (March, September) and maximum sustained mixing energy (December, June). Assuming there is sufficient freshwater input (or lateral or longitudinal gradients in salinity) for salinity gradients to be develop, stratification/destratification can occur with the same periodicity and duration as these tidal mixing intensity. Thus, the duration of stratification events can vary from hours (semi-diurnal to diurnal stratification) to days and weeks (during the weakest tides twice per year) depending on the strength of stratification relative to the tidal mixing energy.

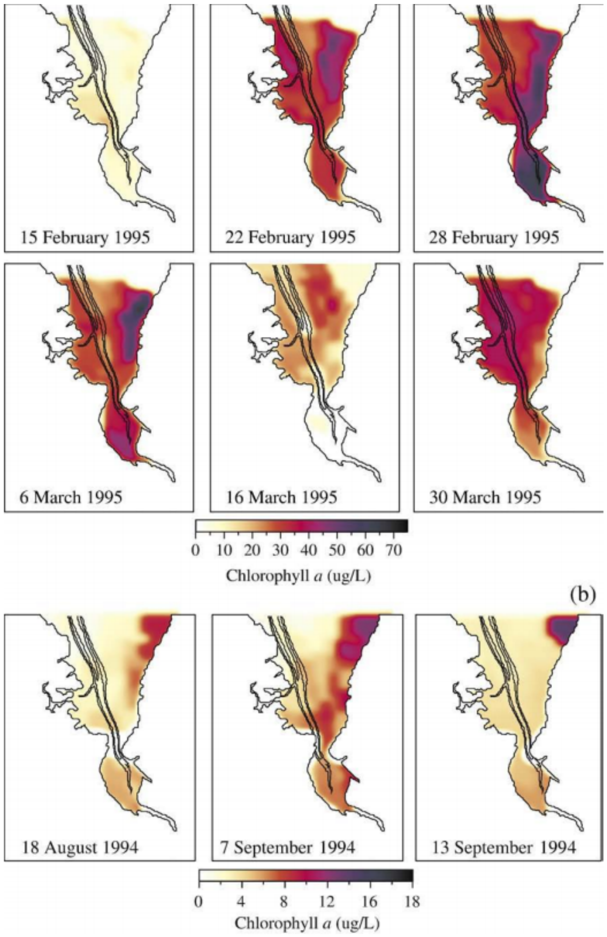

Figure 5.6 Chl-a concentrations in South Bay and Lower South Bay in 1994-1995 showing blooms developing along the shoals. Source: Thompson et al. 2008

Lateral mixing: South Bay's shoals are considered important zones for phytoplankton production because light penetrates over a greater portion of the shallower water column there. Large proportions of South Bay and Lower South Bay have water depths of <2 m. Field and modeling studies in South Bay indicate that, under appropriate lateral mixing conditions sustained phytoplankton blooms develop along the shoals (Figure 5.6), exploiting the relatively light-rich conditions of the shallow water column (Cloern et al., 1989; Huzzey et al., 1990; Lucas et al, 1999; Lucas et al., 2009 ; Thompson et al., 2008). Because sediments are more readily resuspended in shallow environments, higher turbidity, resulting from tidally- or wind-driven local resuspension of sediments, can decrease productivity on the shoals (Lucas et al., 2009). Furthermore, filter feeding by clams can more efficiently clear the shoal water column than the deep channel water column, and reign in shoal blooms (discussed below). Despite the likely importance of productivity along the shoals, there is limited recent data available from these areas. In addition to the studies in the mid-1990s (Thompson et al. 2008; Figure 5.6) there have been some shorter term (~1 year or less) studies including transects into shallow water {Cloern, 1989, Powell 1989, Huzzey, 1990). Those studies found similar patterns as described in later work, with shoal phytoplankton concentrations often higher than in the channel, especially at the start of bloom events. These earlier studies did not include any transects in LSB, so the importance of these mechanisms in LSB is unclear.

Grazing: Benthic grazing can play an important and sometimes dominant role in regulating the amount of biomass that accumulates in the water column of some SFB subembayments (e.g., Thompson et al. 2008; Kimmerer and Thompson, 2014; Cloern et al., 2007; Lucas and Thompson, 2013). The effect of benthic grazing rates on phytoplankton biomass is dependent on the filtration rates ($m^3\ g^{-1}\ d^{-1}$) of the species present and the abundance of grazers ($g\ m^{-2}$). As described in Section X, benthic grazer abundance varies seasonally and spatially based on individual species' life histories, predation, and habitat preference (salinity, sediment type, etc.). Grazer abundance is also tightly coupled to their food supply: i.e., the biomass of grazers at any point in is related to the amount of food available prior to that time. The influence of the filtration rate on phytoplankton concentrations in the overlying water column also depends on water column depth: at a given filtration rate (which is proportional to clam biomass), a shallow water column will be cleared of its phytoplankton faster than a deep water column. The effect of benthos on phytoplankton biomass also depends on other factors such as benthic boundary layer thickness and stratification, which are themselves influenced by turbulent mixing energy. Stratification positively influences biomass accumulation in the sense that filter-feeding benthos cannot access phytoplankton in the surface layer.

Pelagic grazing rates by zooplankton are dependent on the types of zooplankton, their abundance, and their biomass-normalized grazing rates. Copepods, mesozooplankton that are an important food resource in SFB and the Delta, derive most of their energy from phytoplankton as opposed to detrital organic matter (Mueller-Solger et al 2002; Sobczak et al 2002, 2004). Despite mesozooplankton's reliance on phytoplankton, modeling estimates by Kimmerer and Thompson (2014) suggest that they have only a limited effect on phytoplankton biomass in Suisun Bay. Cloern (1982) reached the same conclusion for South Bay. However, microzooplankton have the potential to substantially influence phytoplankton biomass in Suisun Bay (Kimmerer and Thompson, 2014). Outside of Suisun Bay there are limited data on mesozooplankton and microzooplankton biomass and feeding rates. While it may be reasonable to expect that the effect of mesozooplankton grazing on phytoplankton biomass is small Bay-wide, microzooplankton could play a substantial role, based on the observations in Suisun Bay.

Nutrient Concentrations: In many estuaries, pulse nutrient inputs stimulate blooms, nutrient concentrations influence primary production rates, and the depletion of nutrients determines when blooms terminate. However, in deep subtidal areas of South Bay and LSB, nutrients tend to be replete much of the year (Dugdale and Cloern 2010), and the conventional wisdom is that nutrients seldom limit phytoplankton growth rates. Nutrient concentrations do exhibit periodic drawdowns. However, at least in deep channel environments where most data is available, concentrations infrequently dip to levels that would be expected to substantially slow overall production rates (Dugdale and Cloern 2010). Instead, field observations and modeling studies in SFB suggest that phytoplankton bloom termination at the subembayment scale more commonly occurs due to other factors, especially break-down in stratification (Cloern 1991), and sometimes increase in grazing pressure (Thompson et al., 2008). In general, time series of biomass, DIN, orthophosphate, and dissolved Si at stations in LSB and South Bay are consistent with the notion that nutrients seldom limit growth rates (Figures 5.7-5.8). The half-saturation constant, K_s, for phytoplankton uptake of nutrients is a commonly used metric against which the degree of nutrient-related growth limitation can be evaluated (e.g., Dugdale and Cloern 2010; Cloern and Jassby 2012). Nutrient concentrations greater than $10 * K_s$ may serve as an approximate threshold above which nutrient-limited growth would not be expected. At stations s36 and s32, the majority of observations indicate that nutrient concentrations far exceed $10*K_s$. However, during several bloom periods, DIN and DSi did drop below $10*K_s$ suggesting the potential for growth limitation by nutrients. Similar observations can be made with data at other South Bay and LSB stations (Figures A.x – A.y). While it may be reasonable to assert, at the subembayment scale, that nutrients seldom limit growth, the current spatial and temporal frequency of sampling is likely insufficient to say this with confidence about conditions in areas of localized high growth rates (e.g., shoals; intense short-lived blooms).

Blooms when the water column remains well-mixed

A major bloom event in Spring 2003, lasting for more ~1 month, reached chl-a concentrations of ~100 µg/L chl-a, and extended over the full-depth of the well-mixed water column from station s36 to the San Bruno Shoal (s27) (Figure 5.9). Although blooms of this magnitude do not happen every year, there are other examples in the recent record (e.g., 1998). Typical light penetration in the deep channel (below 1-2 m) is considered insufficient to maintain biomass at these concentrations. The conceptual model for phytoplankton primary production would explain the bloom as follows (see Figure 7.2 in SFEI 2014): i) Production rates in shallow areas (shoals, margin habitats) must have been high enough to compensate for the low production rates in deep areas, while also offsetting losses (i.e., grazing, settling, flushing); ii) To maintain this biomass production rate, the rate of lateral mixing between shallow and deep habitats must have been near-optimal for both resupplying sufficient nutrients to the production zone, and transporting sufficient biomass to the channel to sustain high concentrations without washing out the bloom on the shoals (e.g., Cloern, 2006). Nutrient levels were obviously sufficient during 2003 to support this bloom, and, in general, would have been present at similar levels in other years. The fact that blooms like this do not occur every year suggests that some shift occurred in the magnitudes of drivers that regulate phytoplankton growth rates and accumulation in 2003, the cause of which has not been identified. The winter 2003 bloom might be thought of as the “potential bloom”: what could develop based on the nutrient supply, but only occurs under the right combination of physical and biological factors.

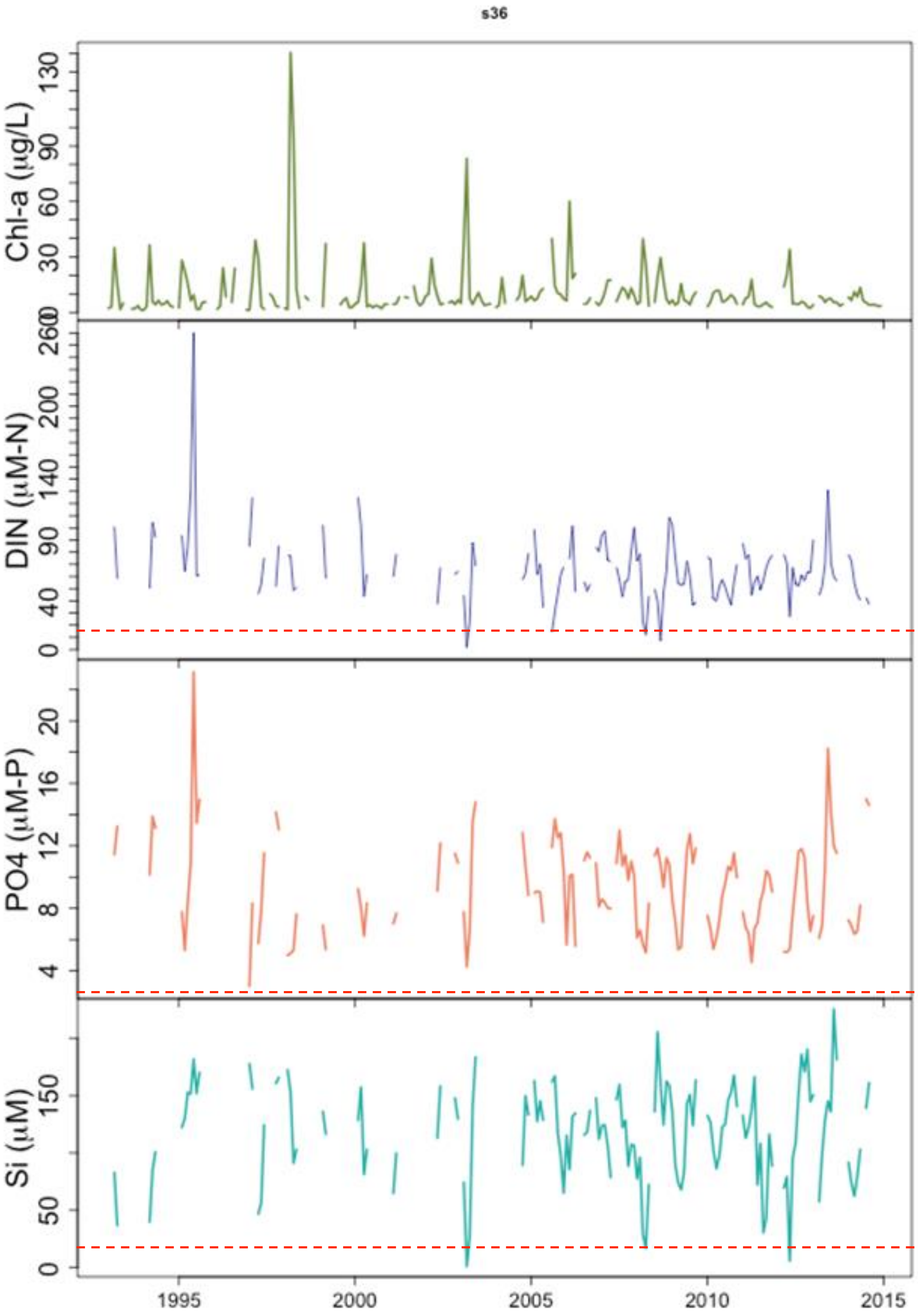

Figure 5.7 Station 36 time series of Chl, dissolved inorganic N, dissolved orthophosphate (PO4), and dissolved Si (DSi) concentrations from 1992-2013. Red dashed lines correspond to 10 times estimated half-saturation constants, K_s, for DIN, PO4, and DSi (Dugdate and Cloern 2010; Cloern and Jassby 2012). When nutrient concentrations are greater than K_s, phytoplankton growth rates should not be kinetically limited by nutrients. Data: USGS

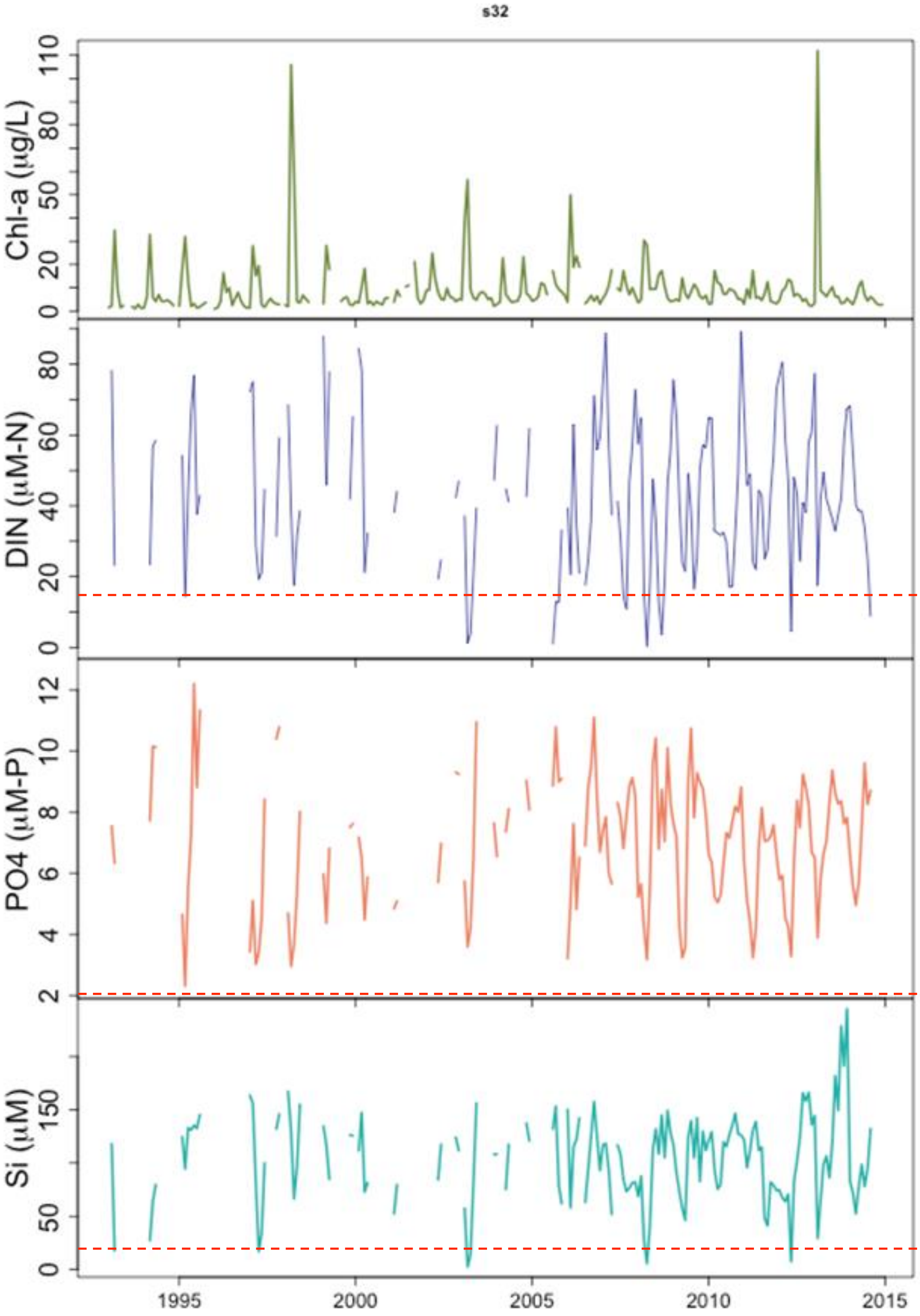

Figure 5.8 Station 32 time series of Chl, dissolved inorganic N, dissolved orthophosphate (PO4), and dissolved Si (DSi) concentrations from 1992-2013. Red dashed lines correspond to 10 times estimated half-saturation constants, K_s, for DIN, PO4, and DSi (Dugdate and Cloern 2010; Cloern and Jassby 2012). When nutrient concentrations are greater than K_s, phytoplankton growth rates should not be kinetically limited by nutrients. Data: USGS

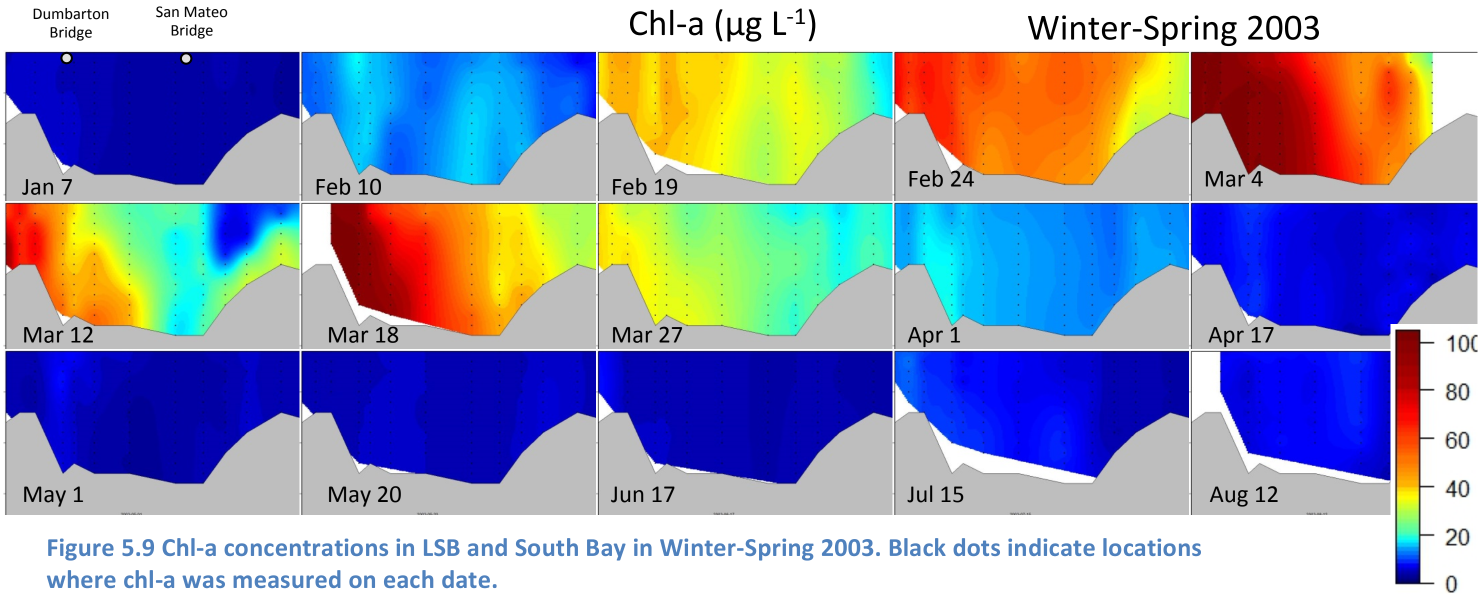

Figure 5.9 Chl-a concentrations in LSB and South Bay in Winter-Spring 2003. Black dots indicate locations where chl-a was measured on each date.

5.3.3 Observations based on high frequency measurements

The discussion in Sections 4.3.1-4.3.2 was based on measurements of discrete samples collected once per week to once per month. Phytoplankton biomass and growth rates respond to environmental factors that change over shorter time-scales (such as hours to days levels). To evaluate the potential importance of variability at shorter time-scales, we examined data from a recent set of high-frequency near-surface sensors that were recently deployed at Dumbarton Bridge. A ~1-year record of calculated chl-a (μg/L), measured every 15 minutes, is plotted in Figure 5.10 A. The chl-a estimates, calculated using a calibration curve for this site (SFEI 2014 xxx), agree reasonably well with *R/V Polaris* discrete samples from stations near the Dumbarton Bridge (colored circles in Figure 5.10). While biweekly to monthly ship-based sampling often identified when chl-a was elevated – and is more accurate in terms of true concentration - the cruise data often missed peak values of an event or did not capture an event's duration.

The *in situ* sensor data also reveals high-frequency patterns in chl-a concentration that point to another potentially-important mechanism influencing biomass concentrations in the open-water areas of LSB. Much of what appears to be 'noise' in Figure 5.10 A emerges as strongly-periodic signals after zooming to a 1-month window (Figure 5.10.B). In January-February 2014, sharp peaks in chl-a concentrations at Dumbarton coincided with low tide. We hypothesize that this elevated chl-a resulted from relatively high-biomass water draining from shallow margin habitats (sloughs, creeks) and mixing with open-Bay water. The semi-diurnal tidal pattern appears to be superposed on a lower-frequency spring-neap pattern: at least for the time period in Figure 5.10.B, and in some other winter and spring periods, peak chl-a at low tide tended to increase over spring tides, while both max and min chlorophyll-a tended to be lowest during neap periods. Similar to observations related to dissolved oxygen in Section 6, we hypothesize that the increasing peaks during spring tide are related to greater cumulative

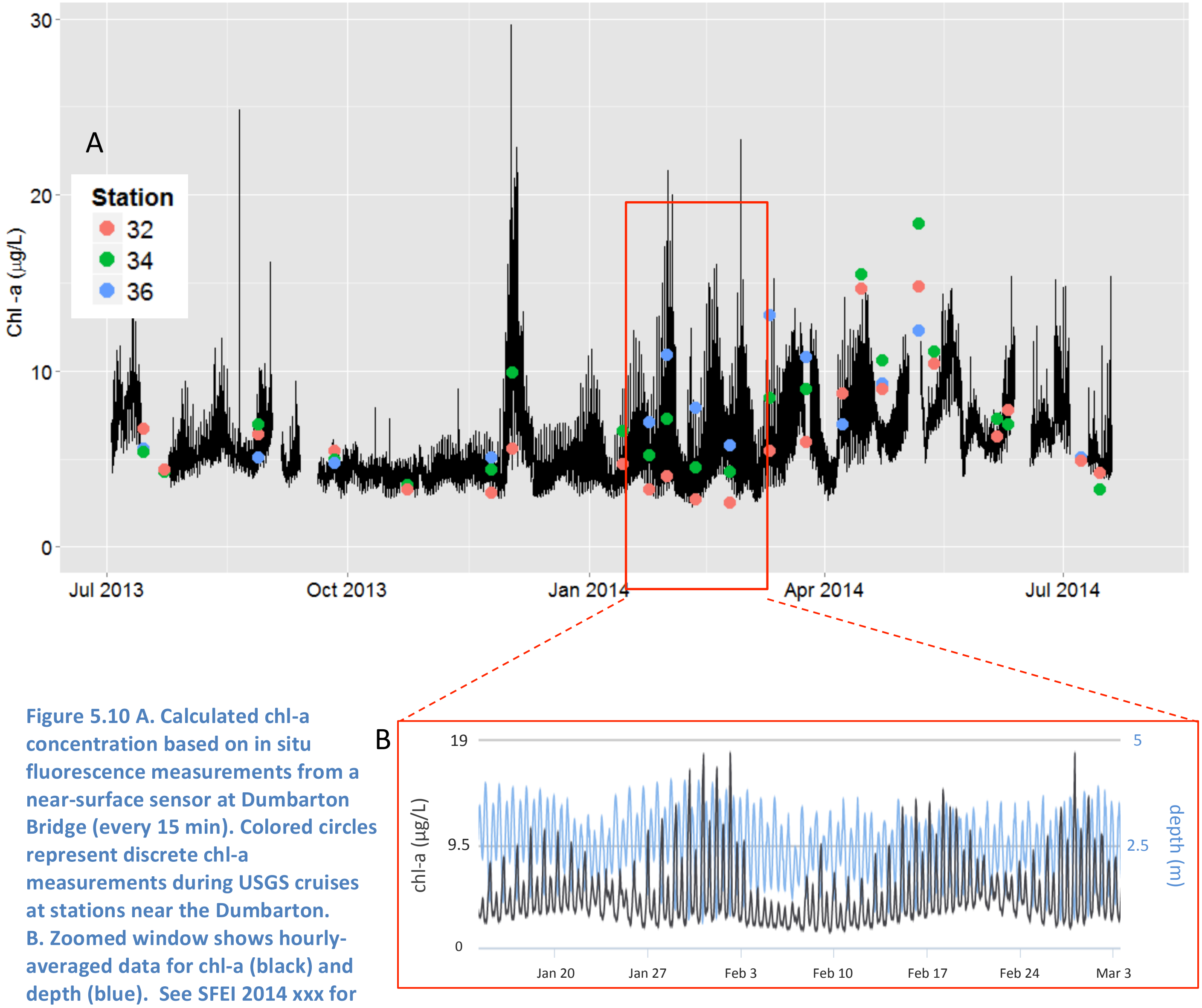

Figure 5.10 A. Calculated chl-a concentration based on in situ fluorescence measurements from a near-surface sensor at Dumbarton Bridge (every 15 min). Colored circles represent discrete chl-a measurements during USGS cruises at stations near the Dumbarton. B. Zoomed window shows hourly-averaged data for chl-a (black) and depth (blue). See SFEI 2014 xxx for more information.

exchange between the margins and the open Bay during the window around spring tides. These semi-diurnal, and sometimes large (e.g., 15-20 µg/L), chl-a peaks had not been previously documented, and we currently do not know how important this process might be as a previously unaccounted for phytoplankton biomass source to LSB. Similarly, we do not yet know how important the observed semidiurnal/daily/weekly variability identified by high-frequency measurements will be for estimates of phytoplankton production in LSB, compared to what would be inferred from the ship-based data alone (compare the ship-based samples and continuous data in Figure 4.10 A). A combination of continued data collection, time-series analysis, and hydrodynamic/biogeochemical modeling will be needed to quantify their importance.

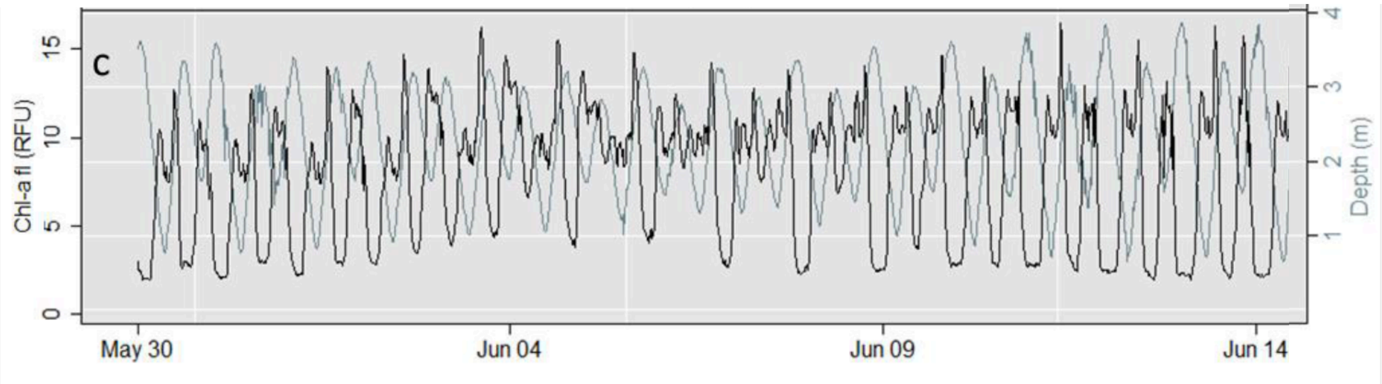

Figure 5.11 Chl-a (relative fluorescent units) and depth (m) for a ~2 week period in late spring 2014.

Through high-frequency measurements in Alviso Slough (far south of LSB), we also found evidence of high chl-a levels in some margin habitats (Figure 5.11). Chl-a fluorescence readings (RFU) in Alviso Slough were regularly 5-10 fold greater than observations at the Dumbarton Bridge, and showed strong variations with tidal stage. Initial calibration data suggests that the relationship between RFU and true chl-a concentration is much different between the two sites; nonetheless chl-a still appears to be substantially higher in Alviso Slough than the open Bay. One hypothesis for the source of increased phytoplankton biomass is that restored salt ponds deliver substantial amounts of biomass to sloughs due to tidal exchange.

5.3.4 Interannual variability in phytoplankton biomass

Cloern et al. (2007, 2010) brought together 3 decades of phytoplankton biomass data for South Bay and observed that, between the late 1990s and 2005, late-summer/fall biomass concentrations had increased 3-fold (Figure 5.12). This increase was the result of either an increase in fall baseline biomass, or as a fall bloom (Cloern and Jassby 2012) that was not a common feature of the system prior to the late 1990s. The phytoplankton biomass increase occurred during a period of time when nutrient loads to the system either remained constant (in some parts of South Bay) or decreased (in Lower South Bay), thus changes in nutrients could not explain the phytoplankton increases. Cloern et al. (2007, 2010) argue that the phytoplankton biomass increase resulted from a loss of benthic

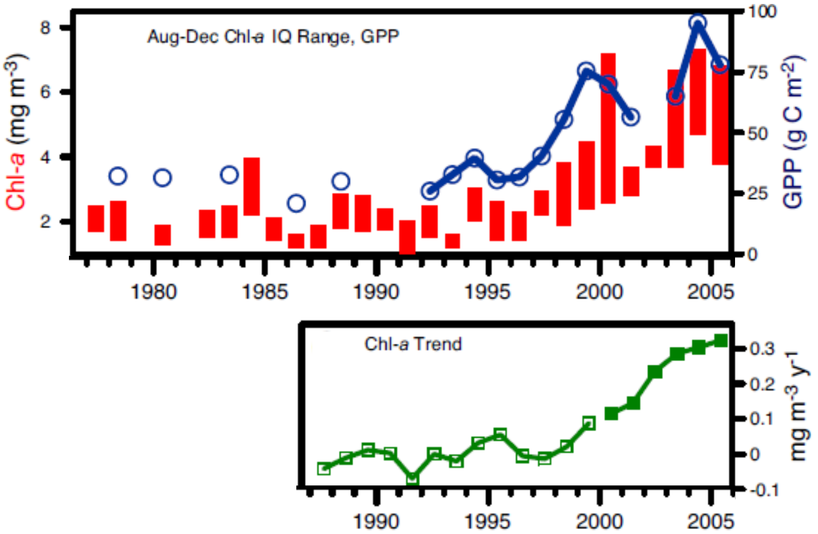

Figure 5.12 From Cloern et al 2007. Top: Aug-Dec chl-a interquartile range and gross primary production, all South Bay stations Bay (s32-s21). Bottom: 10-year rolling window trend, with positive numbers indicating that chl-a increase during that window. Filled symbols indicate values significantly different from zero. Note that trend was only calculated for 1987 onward since data was only available beginning in 1977.

bivalves that, prior to the late 1990s, fed on and strongly regulated phytoplankton biomass by filtering the water column. The loss of benthic grazers occurred due to shifts in two large-scale climate forcings, namely the North Pacific Gyre Oscillation (NPGO) and the Pacific Decadal Oscillation (PDO). The shifts in NPGO and PDO had the combined effect of facilitating the southward expansion of several bottom-feeding oceanic species (e.g., Dungeness crab, English sole), causing an increase in their abundance that coincided with both the decrease in benthic grazers and increase in phytoplankton biomass (Cloern et al., 2010).

To further examine trends in phytoplankton biomass in South Bay, we extended the time series of Cloern et al. (2007) through 2013, using same set of stations and months as in Figure 5.12. Over the period of 2005-2010, biomass concentrations appear to have plateaued, establishing a new median fall concentration of 5-6 µg/L (Figure 5.13). Biomass values for 2012-2013 were lower than 2005-2010, but still substantially greater than biomass levels prior to 1999. The 5+ years of relatively flat biomass concentrations from 2005-2010 might reasonably be construed as representing a new and more sensitive biological response level for South Bay to nutrients, even if that new plateau only lasts for a short period of time (several years) before shifting to another state (higher or lower sensitivity). It may be tempting to also speculate that the lower concentrations of 2012-2013 signal a return to lower sensitivity. However, 2012 and 2013 are only two years in a system that exhibits high interannual variability in its biological response to nutrients (i.e., Figure 5.3). The 2012-2013 distributions of biomass concentrations at South Bay sites are comparable to those observed in 2006, which was immediately followed by several higher biomass years.

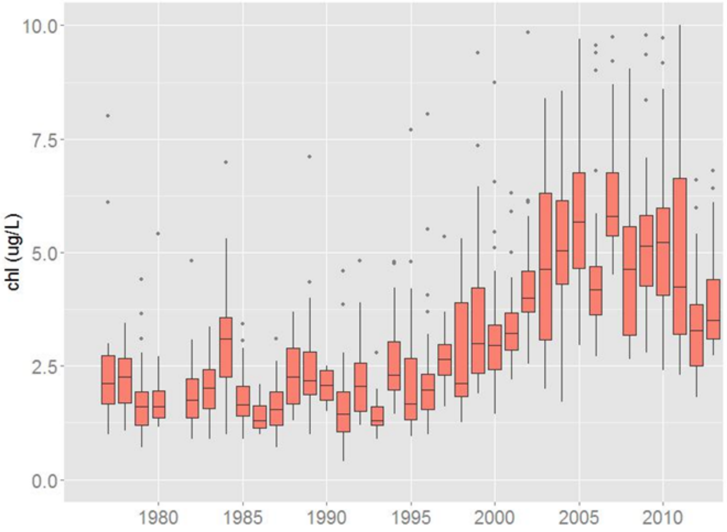

Figure 5.13 Phytoplankton biomass for the months Aug-Dec, using the same stations as Cloern et al. (2007) and extending the time series through 2013. Annual interquartile ranges (IQR) of collected monthly data for these stations aggregated together are shown as red bars, with the whiskers extending to 1.5* IQR, and anything beyond that marked by points. Data: USGS

We also examined post-2005 phytoplankton response for a set of stations that are more closely associated with conditions in LSB ,i.e., s36-s27 (Figure 5.1). These LSB and southern South Bay stations showed a similar pattern of increasing fall biomass (Figure 5.14) as the South Bay stations (Figure 5.13). Biomass actually began increasing earlier at stations s36-s27, and plateaued at slightly higher levels. A similar decrease was also evident in 2011-2013.

Figures 5.15 displays seasonal variations in phytoplankton biomass, and changes over time at individual stations. The left panels summarize monthly data broken into three eras (1975-1999, 2000-2008, 2009-2013), and Figure 4.11 F-J presents trends by month. All of the stations experienced increases in monthly chl-a (Figure 4.11 f-j), and at all stations those changes were statistically significant ($p < 0.05$, Mann-Kendall test) for at least half the year. All stations experienced statistically significant increases in Sep-Nov, similar to observations made by Cloern and Jassby (2012), and to the suggestion of a fall bloom. Interestingly, during the two months when the largest blooms typically occur – March and April – the changes were not statistically significant at any stations. The lack of significance may owe in part to the high interannual variability of the spring bloom magnitude.

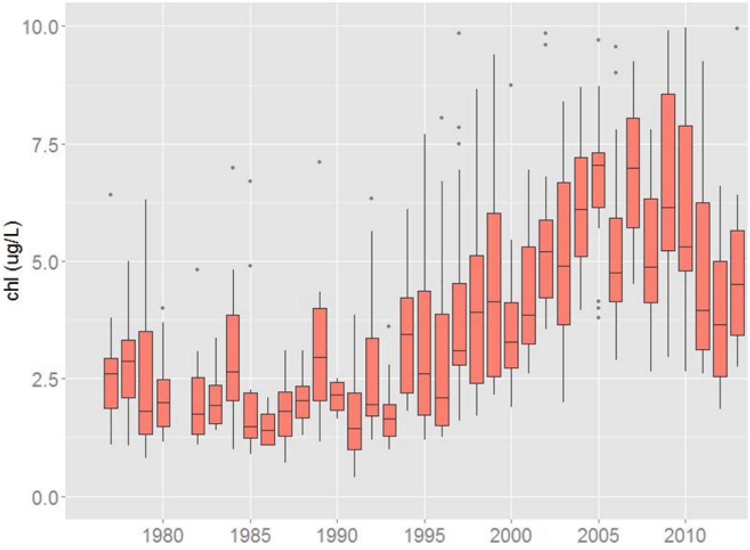

Figure 5.14 Phytoplankton biomass for the months Aug-Dec, similar to Figure 4.12 but using stations s27,30,32,34,36. Annual interquartile ranges (IQR) of collected monthly data for these stations aggregated together are shown as red bars, with the whiskers extending to 1.5* IQR, and anything beyond that marked by points. Data: USGS

5.3.5 Potential explanations for increased biomass in LSB and South Bay

Loss of benthic grazers and increased light levels (and growth rates) are both plausible explanations for the observed increases in phytoplankton biomass in South Bay and LSB between the mid-1990s and 2013. Cloern et al. (2007, 2010) presented evidence that benthic grazers declined rapidly in the late 1990s after shifts in the PDO and NPGO initiated increases in the abundance of several benthos-feeding organisms. On-going benthic sampling has found that bivalve populations remain low relative to early-1990 levels (Thompson et al., Section 4). Jassby (2008) documented substantial decreases in suspended sediment concentrations in Suisun Bay and the Delta. Schoellhamer (2011) presented evidence of decreased suspended sediment concentrations in the Bay, with a particular focus on San Pablo Bay; and, in Section 2 of this report, Schoellhamer et al (2015) illustrate how suspended sediment concentrations have decreased substantially at the Dumbarton Bridge since the mid-1990s.

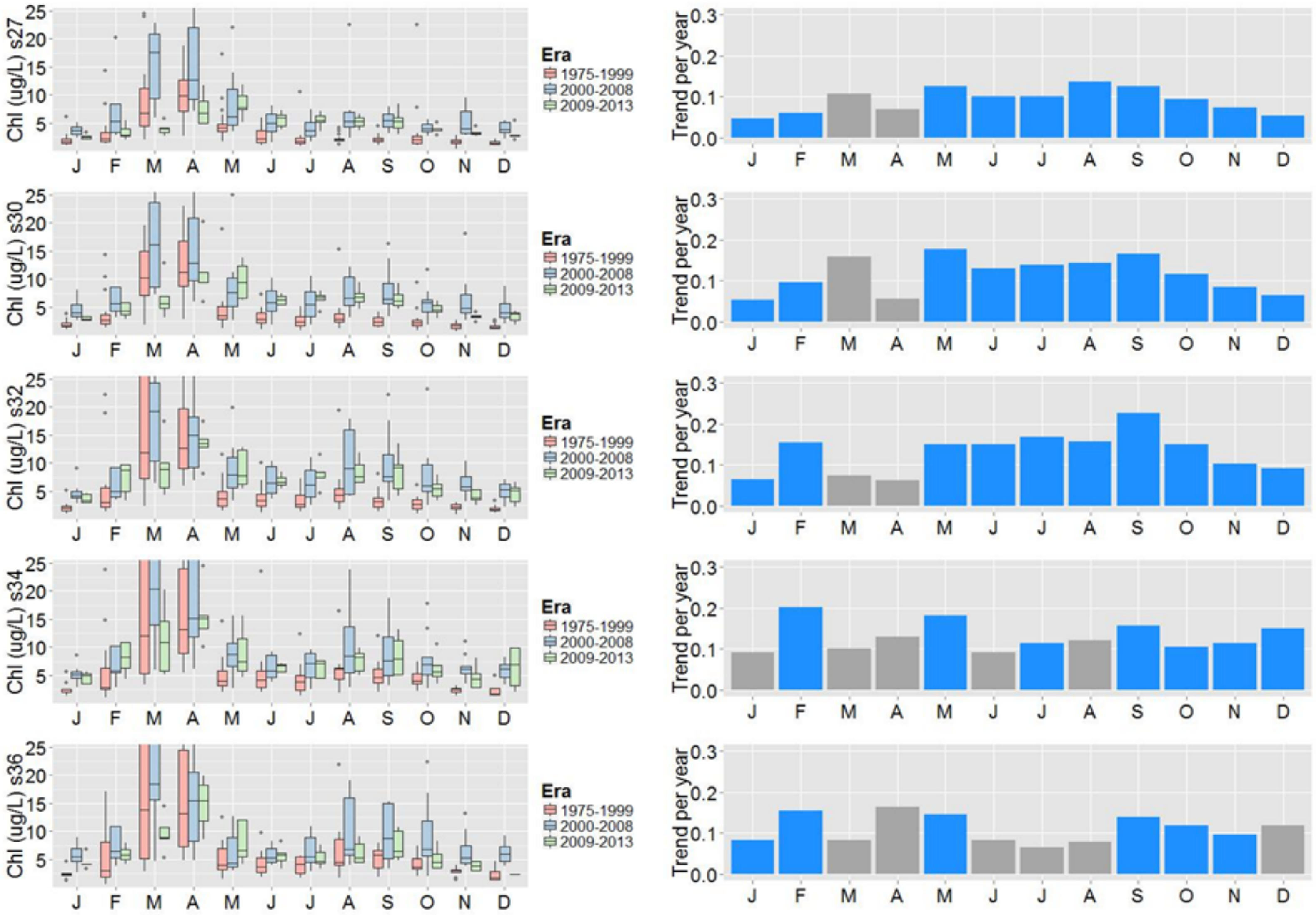

Figure 5.15 Left: Monthly plots of chl-a interquartile ranges from 1975-2013, divided into 3 eras at s36-s27. Right: Trend analysis (Theil-Sen slope) over the entire time period, with blue indicating statistically significance (*p* < 0.05, Mann-Kendal test).

In the case of both benthic grazing and suspended sediment concentrations, however, the potential quantitative effect of decreased grazing rates and increased light levels on phytoplankton biomass concentrations have not been evaluated. An understanding of the relative importance of these factors on biomass will help prioritize among data collection needs, and offer insights into how conditions may change in the future. Although these processes ultimately need to be thoroughly evaluated within a coupled hydrodynamic-biogeochemical model, we examine them here in a simplified way here to provide some initial estimates. Changes in biomass concentration over time in a control volume can be modeled as a set of first order processes: growth, grazing, settling and physical exchange of water (flushing), similar to the approach used in Section 2:

$$\frac{dB}{dt} = +k_{grow} \cdot B - k_{graze} \cdot B - k_{settle} \cdot B - k_{flush} \cdot B + \ k_{flush} \cdot B_{ext}$$

which can be simplified to

$$\frac{dB}{dt} = (+k_{grow} - k_{graze} - k_{settle} - k_{flush}) \cdot B + \ k_{flush} \cdot B_{ext}$$

where:

B = biomass concentration (e.g., µg/L organic carbon, or as µg/L chl-a)

$\frac{dB}{dt}$ = change in biomass over time (µg L^{-1} d^{-1})

k_{grow} = first-order growth rate constant (d^{-1})

k_{graze} = first-order grazing rate constant (d^{-1}) = biomass specific filtration rate (m^3 d^{-1} g^{-1}) * bivalve biomass density (g m^{-2}) / water column depth (3 m)

k_{settle} = phytoplankton settling rate = settling velocity (0.5-1 m d^{-1}) / water column depth (3 m) = 0.2-0.3 d^{-1}, but as high as >1 d^{-1} for higher settling velocities.

k_{flush} = first-order flushing rate constant (d^{-1}) = Volume of LSB (m^3) / Q_{flush} (m^3 s^{-1}) effective tidal flushing rate = 0.06 d^{-1} during summer. See Section 2.

B_{ext} = biomass in the external water volume that exchanges with LSB, in this case assume station s27 = 2-3 µg/L chl-a

Although this approach treats LSB as a well-mixed single box, a major oversimplification, it allows us to directly assess the relative influence of processes on biomass concentrations by comparing their first order rate constants. Section 2 describes an approach for estimating k_{flush} that is also appropriate for this analysis, which yielded a value for k_{flush} of 0.06 d^{-1} for summer/fall conditions. Phytoplankton settling velocities vary widely depending on taxa, growth phase or condition, and other factors (Ball and Arthur, 1981; Cloern et al. 1983). Settling velocities for diatoms are most relevant here, since diatoms tend to comprise most of the biomass in LSB. For the purposes of this analysis, values of 0.5-1 m d^{-1} will be used, resulting in k_{settle}= 0.2-0.3 d^{-1}.

The discussions of bivalve abundance and suspended sediment concentrations presented in Sections X and Y, respectively, provide the basis for estimating k_{graze} and k_{grow}. Bivalve biomass values from Section 4 were converted to filtration rates (m^3 m^{-2} d^{-1}) using biomass specific filtration rates relevant for LSB and South Bay, then spatially-interpolated. The spatially interpolated filtration rates were combined with water column depth to yield spatially-interpolated values for k_{graze}. Figure 5.16 illustrates how k_{graze} varied in both space and time in LSB and southern South Bay. It should be noted that there are large gaps in data (1995-2004), and therefore substantial uncertainty due to these gaps, and that uncertainty is in addition to what was already substantial uncertainty due to spatial and interannual variability, natural biological variability of "true" filtration rates, and extrapolation of limited samples over space and time. Best estimates for k_{graze} pre-1995 and post-2000 are 1-1.5 d^{-1} and 0-0.2 d^{-1}, respectively.

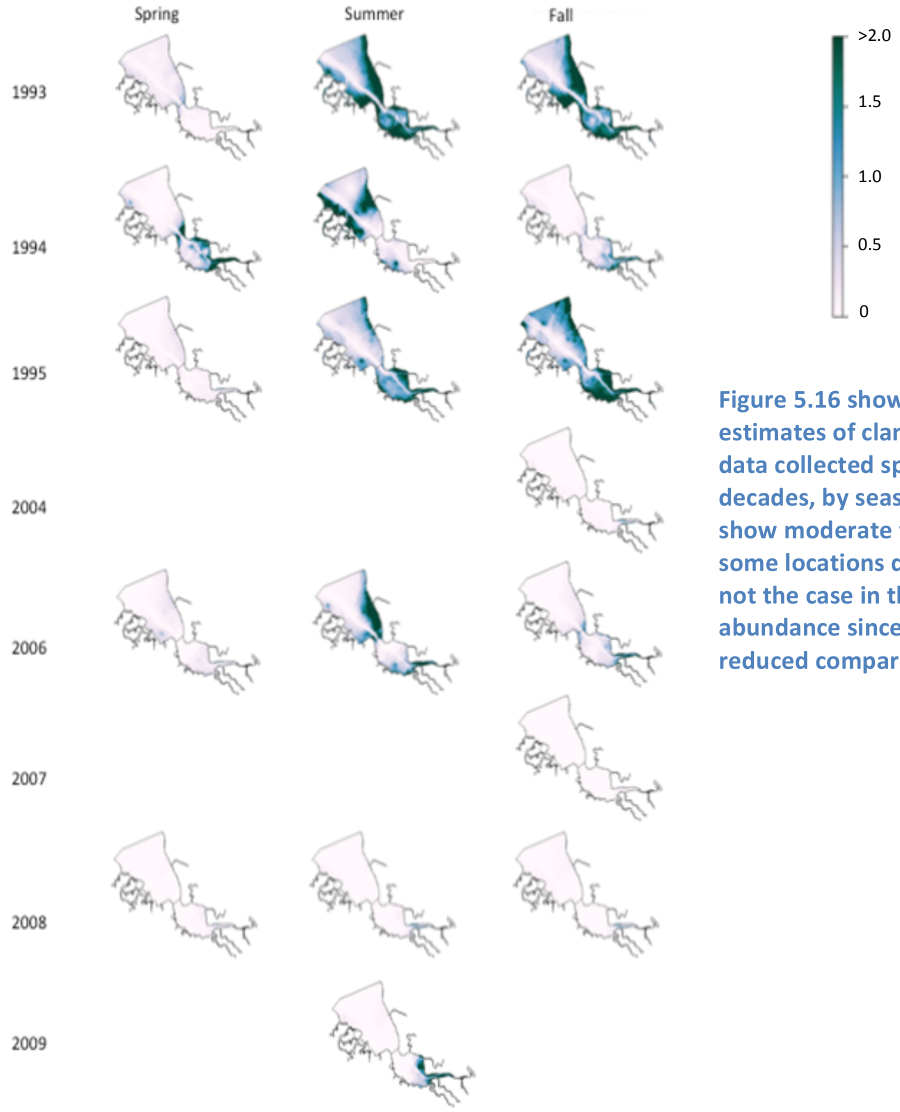

Figure 5.16 shows spatially interpolated estimates of clam abundance from USGS data collected sporadically over the past two decades, by season. Although data in 2006 show moderate to high clam abundance in some locations during the summer, this is not the case in the fall and for the most part abundance since 2004 has been greatly reduced compared to the prior decade.

Changes to k_{grow} were estimated based on changes in suspended sediment concentrations (SSC). The analysis of the long-term record of high frequency SSC data discussed in Section X found that SSC concentrations decreased by ~40% at the Dumbarton Bridge since 1999, with most of that decrease occurred between 1997 and 2001 (Figure 5.17 A). We examined how this change varied seasonally and over time by binning the high-frequency SSC data by month and then into two eras, 1992-1998 and 1999-2011, and then using the relationship with SSC presented in Figure 3.2 to compute photic depth each 15-min SSC estimate (Figure 5.17 B). On average, photic depth has increased by at least 25% in all months, and by up to 50% during August through October. Using the method developed by Cole and Cloern (1987) to k_{grow} based on photic depth (assumes light-lighted growth), k_{grow} during late-summer/fall has increased from 0.5 d^{-1} to 0.75 d^{-1} over the last two decades.

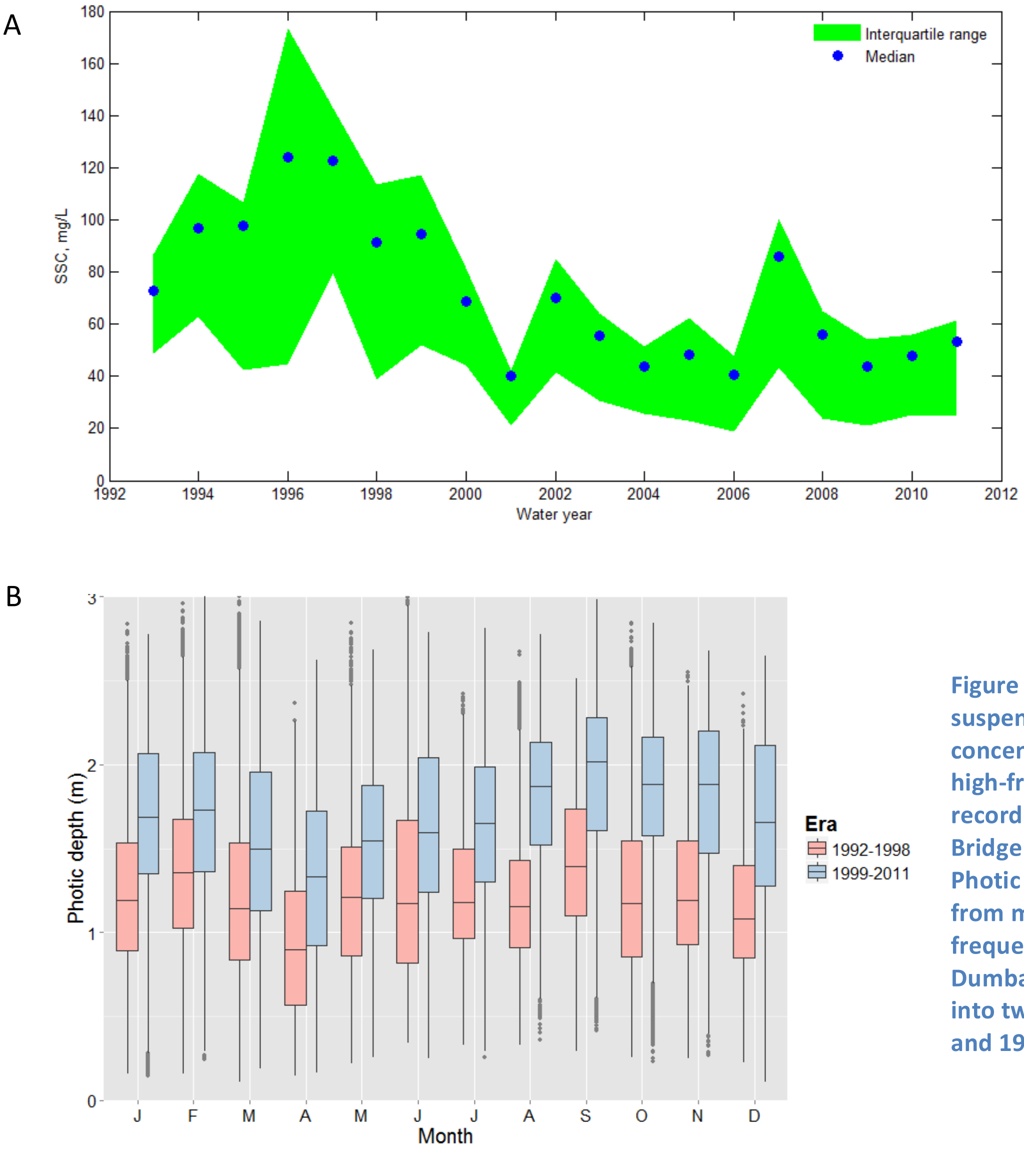

Figure 5.17 A. Annual suspended sediment concentrations based on high-frequency (15-min) records at Dumbarton Bridge (See Section 3). B. Photic depth calculated from monthly-binned high-frequency data from the Dumbarton Bridge, divided into two eras: 1992-1998 and 1999-2011.

The estimated rate constants are summarized to the right (Table 5.2). Although these estimates indicate that k_{graze} and k_{grow} both changed substantially, they suggest a greater influence of decreased grazing compared to increased growth rate. That said, a 50% increase in growth rate due to relaxation in light limitation in a historically light-limited system is noteworthy. The comparison of rate constants also suggests that flushing of biomass from LSB is an insignificant loss term relative to grazing and settling. Settling, on the other hand, has the potential to be an important loss term that could be of similar magnitude as the production term and comparable to pre-1999 grazing rates. The uncertainty of the settling term is also large, and one that needs to be better constrained if accurate biogeochemical models are to be developed.

Table 5.2 Relative importance of different potential drivers of phytoplankton biomass, expressed as estimated first-order rate constants

	pre-1999 (d^{-1})	post-1999 (d^{-1})
k_{graze}	1-1.5	0-0.2
k_{grow}	0.5	0.75
k_{settle}	0.2-0.3	
k_{flush}	0.06	

5.4 Major data gaps and recommendations

In assessing the state of the science with regards to phytoplankton biomass in Lower South Bay, we have identified the following major knowledge gaps:

1. What combination of factors regulate phytoplankton productivity and biomass, and how do the relative importance of those factors vary spatially and seasonally?
2. What combination of factors can explain the fall biomass increase in the late-1990s (e.g., loss of filter-feeding benthos, decreasing suspended sediments)?
3. How important are margin habitats as a source of organic matter to the open Bay, in particular restored salt ponds?
4. How important is benthic algae production to overall productivity and organic matter accumulation in Lower South Bay?
5. What effects would potential management actions have on biomass and algal toxins? E.g.,
    a. Decreased nutrient loads by 25%, 50%, 75%?
    b. Operation of restored salt ponds (e.g., including optimization for nutrient removal and beneficial habitat condition)
    c. Managed oyster or mussel reefs
6. What levels of phytoplankton production and biomass are plausible under future scenarios in Lower South Bay? Have we reached a new plateau or will concentrations rise further?
    a. How will controls on phytoplankton biomass (i.e. light availability, benthic grazing) change in the future?

Also, though not explored within this report, there are also a number of high priority science questions related to phytoplankton community composition, including:

1. What factors most strongly regulate phytoplankton community composition in Lower South Bay (e.g., light availability, temperature, nutrients, benthic grazing, exchange with salt ponds)?
2. To what extent do conditions in Lower South Bay select for either potentially harmful algae or algae that poorly support the food web?
3. What are source(s) of algal toxins in Lower South Bay?
4. Are restored salt ponds a substantial source of harmful algal species and algal toxins?

We propose a number of high-priority activities to address these knowledge gaps. Since some of these investigations would be resource- and time-intensive undertakings, some level of prioritization is still needed:

- Gather high-spatial resolution data through biogeochemical mapping
    - Characterize spatial and temporal heterogeneity
    - Assess condition across the region – extent and severity of potentially problematic events
    - Gather data for model calibration/validation
- Conduct mechanistic field investigations to quantify important processes related to phytoplankton and benthic algae production (slough ←→ open Bay, salt pond ←→ slough, stratification in open Bay and sloughs)
- Develop and apply a coupled hydrodynamic and biogeochemical model for Lower South Bay, including sloughs and margins
    - Evaluate mechanisms that control phytoplankton biomass and DO concentrations through sensitivity analysis
    - Examine the role of anthropogenic nutrients and quantify nutrient fate
    - Forecast ecosystem response under potential future conditions, including changing environmental factors (sediment concentrations, bivalves)
    - Quantify how potential management actions, such as nutrient load reductions and salt pond operation, will influence ecosystem response (phytoplankton biomass, DO)
    - Characterize and quantify uncertainty
- Characterize phytoplankton community composition in Lower South Bay, and explore mechanisms that influence community composition, including potential sources of harmful algal species

6. Dissolved Oxygen

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6.1 Introduction

Dissolved oxygen (DO) concentration serves as an important indicator of estuarine habitat condition, because all aquatic macro-organisms require some minimum DO level to survive and prosper. The instantaneous DO concentration, measured at a specific location in the water column, results from a balance between multiple processes that add or remove oxygen (Figure 6.1): primary production produces O2; aerobic respiration in the water column and sediments consumes O2; abiotic or microbially-mediated biogeochemical reactions utilize O2 as an oxidant (e.g., oxidation of ammonium, sulfide, and ferrous iron); O2 exchange occurs across the air:water interface in response to under- or over-saturated DO concentrations in the water column; and water currents and turbulent mixing transport DO into and out of zones in the water column. If the oxygen loss rate exceeds the oxygen production or input rate, DO concentration decreases. When DO losses exceed production or input over a prolonged enough period of time, hypoxia (<2-3 mg/L) or anoxia can develop. Persistent hypoxia or anoxia causes stress or death in aquatic organism populations, or for organisms that can escape a hypoxic or anoxic area, the loss of habitat. In addition, sulfide, which is toxic to aquatic organisms and causes odor problems, escapes from sediments under low oxygen conditions.

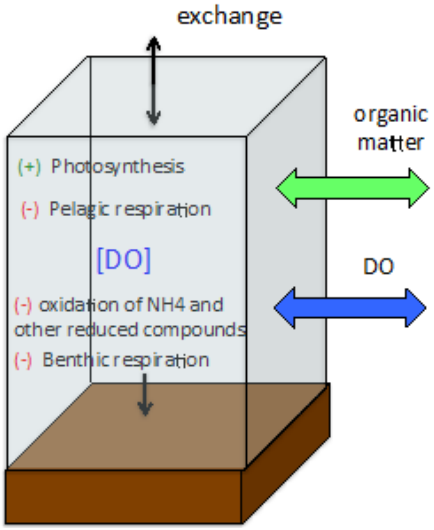


Figure 6.1 Major processes influencing DO concentration within a control volume.

Low dissolved oxygen is a common aquatic ecosystem response to elevated organic matter inputs, including organic matter produced in situ as a result of high N and P loads to a system. The San Francisco Bay Basin Plan specifies a standard for a minimum DO concentration of 5 mg/L, or 3 month rolling median of >80% saturation [ref]. Several decades ago (1960s-1970s), areas of Lower South Bay did experience low DO (Cloern and Jassby, 2012) due to the discharge of wastewater containing high biochemical oxygen demand (BOD) and ammonium (NH_4). Changes in wastewater treatment practices in the 1970s (secondary treatment to remove biochemical oxygen demand, and nitrification of NH4 prior to discharge) resulted in major improvements in DO levels in deep subtidal areas of LSB. Low DO also occurs naturally in some habitats; it is therefore relevant to ask what portion of observed DO deficits may be due to natural phenomena and what portion is the result of anthropogenic impacts, including high nutrients. However, there has been limited systematic investigation of DO levels in margin habitats (sloughs, tidal wetlands, and restored salt ponds). In addition, a comprehensive analysis of DO concentrations in deep subtidal areas of Lower South Bay has not recently been conducted.

The goals of this section are to:

1. Summarize DO data in deep subtidal habitats based on long-term monitoring data, and explore seasonality and changes over time from 1990s-present
2. Summarize DO observations in shallow margin habitats, including salt ponds and sloughs
3. Examine recently collected continuous DO observations at Dumbarton Bridge,
4. Characterize mechanisms that influence DO concentrations in sloughs and in deep subtidal areas based on existing data; and
5. Identify priority data gaps or knowledge gaps.

6.2 Data sources

The USGS has measured DO during biweekly to monthly cruises aboard the *R/V Polaris* along the deep center axis of South Bay and LSB (Figure 6.2) over the past ~25 years, measured both in discrete samples and using a vertical profiling sensor. USGS data were downloaded from http://sfbay.wr.usgs.gov/access/wqdata/, and further processed and analyzed using R, including the *wq* package. Throughout this section, dissolved oxygen levels are presented in units of either concentration (mg/L) or % saturation (%sat). (Note: saturation concentration (DO_{sat}) is a function of temperature and salinity. For example, at T = 20C, DO_{sat} = 9.1 mg/L and 7.6 mg/L for freshwater and salinity = 30 ppt, respectively). Historical DO data (1960s-early-1990s, South Bay Dischargers Authority; recently converted to electronic format; J Ervin, City of San Jose, pers. comm.), measured weekly to biweekly in discrete samples along a transect in Artesian Slough and Coyote Creek, are also explored to provide a historical perspective in an area of LSB that is not sampled as part of the USGS program.

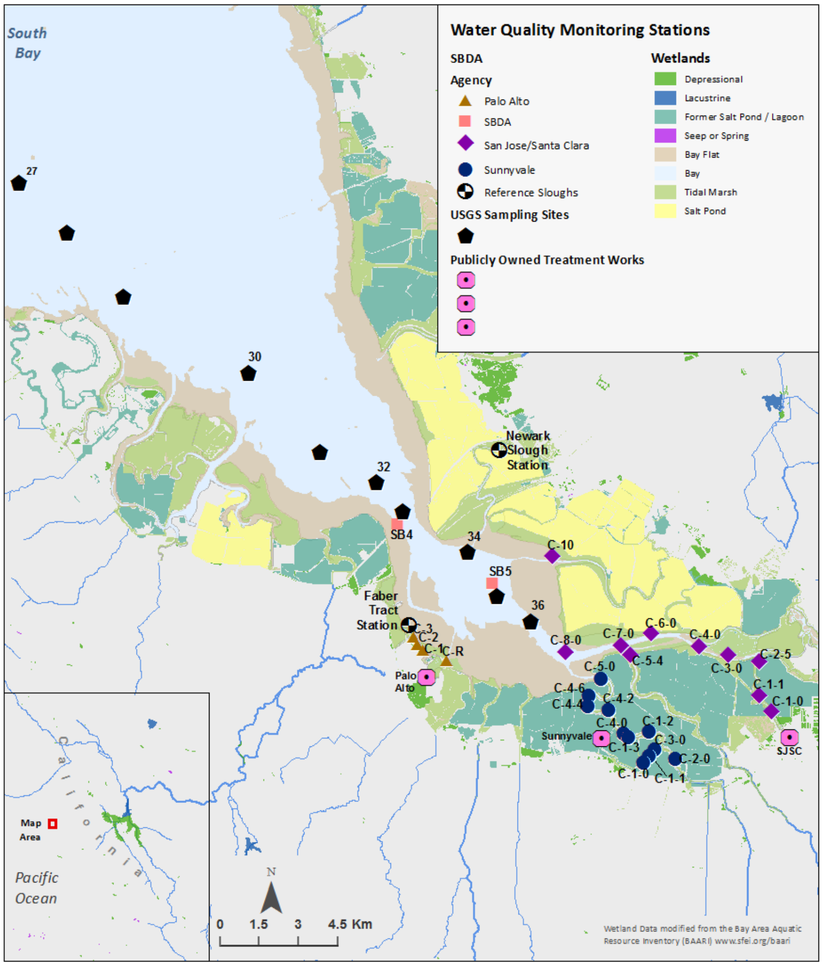


Figure 6.2 Location of monitoring stations.

High-frequency DO data from moored sensors were also analyzed. At the Dumbarton Bridge, DO data were available from an *in situ* sensor deployed near the surface (depth = 1-3 m; September 2013-present). High-frequency

DO data has also been collected for several years at a station located in Alviso Slough (ALV), approximately 4 km upstream of the Alviso-Coyote confluence (Figure 6.2). That station is maintained by USGS, and, more recently, by USGS and SFEI. More information about those in situ sensor records are available in a recent report (SFEI #723 2014). DO data from several monitoring efforts that deployed *in situ* sensors in shallow margin habitats (sloughs, creeks, and restored salt ponds) were also examined. That data are summarized in Appendix A.2, and summary figures are presented below. Field data from published reports and unpublished data from other on-going work were also analyzed, as noted below.

6.3 Results and Discussion

6.3.1 DO concentrations in Lower South Bay and South Bay

6.3.1.1 Deep subtidal areas

Dissolved oxygen concentration exhibited a high degree of variability across stations in both South Bay and LSB over the period 1993-2013 (Figure 6.3A and 6.3B). Individual DO measurements ranged from a few values close to 50% to >100% of saturation, and the central tendency values for DO%$_{sat}$ at South Bay stations over time (~90%) were slightly greater than LSB stations (80-85%). Some of the lowest DO values (e.g., <80%) in South Bay appear to be associated with periods when recently-upwelled water from the coastal ocean (cold, relatively low DO) intruded as far south as the San Bruno Shoal (Figure 6.4). Evidence suggests these low-DO intrusion layers tend to be relatively short-lived (weeks), as indicated by them seldom being observed during consecutive bi-weekly or monthly surveys; the stratification is likely broken down by strong tidal mixing, and lower-DO bottom water is mixed with higher-DO surface waters.

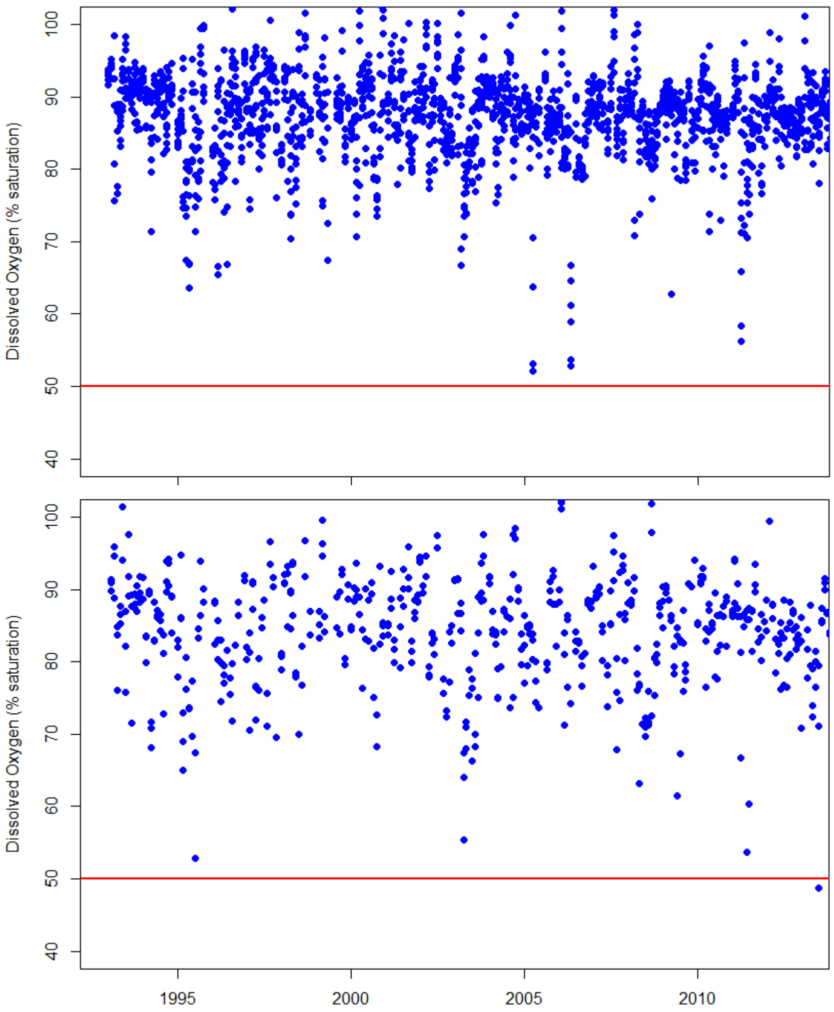
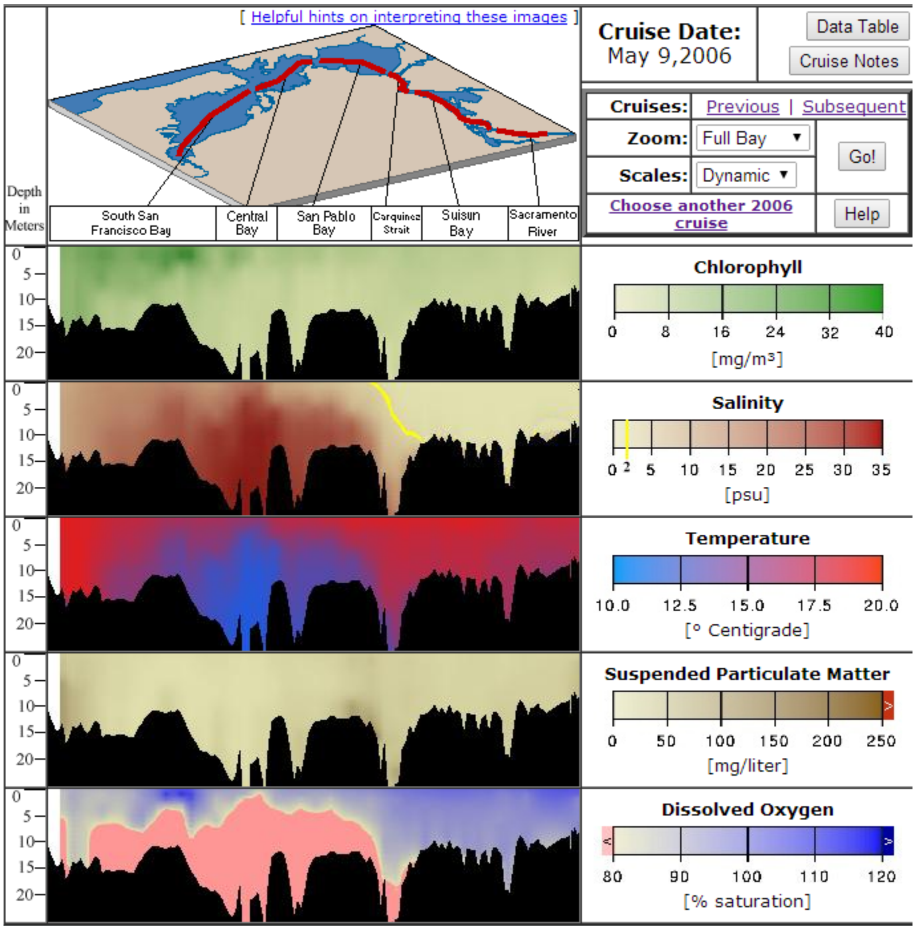


Figure 6.3 A Minimum DO measured during each vertical profile during weekly to biweekly sampling at USGS stations in A. South Bay (Bay Bridge to Dumbarton Bridge; s21, s22, s24, s25, s27, s29, s30, s32) and B. Lower South Bay (south of Dumbarton Bridge; s34, s35, s36)



Cruise Date: May 9,2006

Zoom: Full Bay

Scales: Dynamic

Figure 6.4 Contour plots of water quality parameters in LSB and South Bay on May 9 2006 showing a bottom layer of denser (higher salinity, lower T) waterh having lower-DO. Such layers are thought to result from the intrusion of recently-upwelled coastal water through the Golden Gate. High Delta outflows tend to increase the likelihood of such intrusions by strengthening the salinity (density) gradient that encourages density-driven exchange.

$DO\%_{sat}$ in surface water samples varied seasonally at most stations in LSB and South Bay, with $DO\%_{sat}$ generally lower in spring-summer (Apr-Aug) than in winter months (Figure 6.5). At each station, changes in $DO\%_{sat}$ over time were explored through visual inspection after dividing monthly data at each station into 3 eras, and more formally by calculating the change in DO over time and testing that the slope was significantly different from zero (see Figure 6.4 caption for more details). Although it is difficult to make generalizations that hold across all stations, at many stations the data suggest a modest decrease (0.3-0.5%sat per year) during summer and fall, but only a minority of those negative slopes were statistically significant. When stations are grouped by subembayment and examined across the entire record using a test that is robust for seasonally-varying data (Mann-Kendall seasonal test, applied to data in Figure 6.3), a slight but statistically-significant decreasing trend is detected for minimum DO for South Bay (slope = -0.12%/yr, p = 0.009). The slope for minimum DO for LSB stations was not significantly different from 0 (slope = -0.05%, p = 0.41). Additional rigorous analysis of these data may be warranted to determine if changes in DO have occurred. In a related effort, a significant relationship between increased probability of summertime DO < 80%sat and annual chlorophyll-a was detected (Sutula et al., in prep).

Data from Jan-Aug 2003 illustrate how, during some years, DO concentrations in LSB and southern South Bay exhibit strong seasonal and spatial variability (Figure 6.6). Chl-a concentrations, a measure of algal biomass (See Section 5), increased to high levels over the full water column depth from mid-February through mid-March over a large area extending from below the Dumbarton Bridge to north of the San Mateo Bridge. By mid-April, chl-a dropped to levels < 10 µg/L, and remained fairly low through August. DO increased from 90-100% saturation to >140% saturation in areas where the bloom was most intense, and in early March DO ≥ 100% over the entire water column volume from south of the Dumbarton to the San Bruno Shoal. By mid-March, bottom DO concentrations appeared to decrease to ~90% saturation between the Dumbarton and San Mateo Bridges. By mid-April, DO decreased below 80% over most of the water volume between s36 and s27, and continued in the range of 75-85% through August (with the exception of May 1, which was inexplicably high).Phytoplankton blooms of this magnitude, and the subsequent lagged response in DO concentration, did not occur during all years; in fact the sustained high chl-a levels in Winter-Spring 2003 stand out as a period of high, but not necessarily anomalous, production.

While ship-based measurements indicate that DO levels in deep subtidal areas generally fall above the Basin Plan DO standards (80% saturation) in LSB and South Bay (Figure 6.2A and 6.2B), continuous monitoring data from a near-surface sensor at the Dumbarton Bridge show that DO concentrations experienced sharp departures of 1-1.5 mg L^{-1} within narrow time windows (few hours; Figure 6.7). Lowest DO concentrations were observed at low tide, and values occasionally dipping below 5 mg L^{-1} (SFEI 2014, #723). The association between low DO and low tide suggests that there were strong spatial gradients in DO concentration within LSB, and that tidal action moved these different water masses past the stationary sensor at Dumbarton Bridge. This hypothesis is discussed further in Section 3.2.2. It is also noteworthy that, over the longer record presented in Figure 6.7, ship-based sampling at stations near the Dumbarton Bridge seldom captured the lower DO conditions even when they were observed hours before or after the cruises sampled in this area. Ship-based measurements at s34 during two cruises in summer 2014 did, however, observe this condition. Both of these cruises began at low tide, and s36 was not sampled because water levels were too low for the *R/V Polaris* to sample there. These observations may suggest that there may be a tendency for ship-based data to be biased-high for LSB during some times of the year because of navigational limitations to sampling further south in LSB.

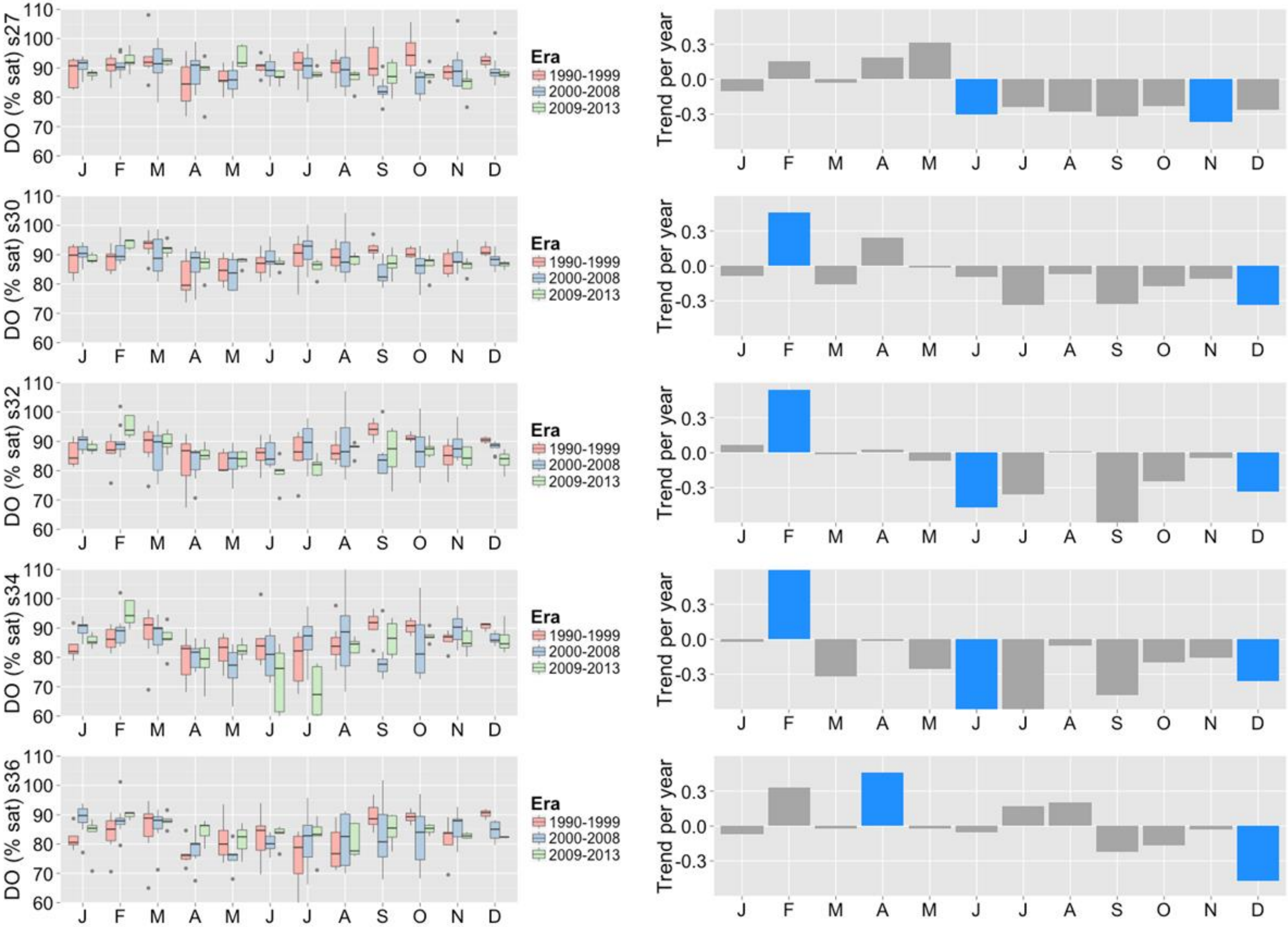


Figure 6.5 Left: Monthly plots of DO%sat interquartile ranges from 1975-2013, divided into 3 eras at s36-s27. Right: Trend analysis (Theil-Sen slope) over the entire time period, with blue indicating statistically significance (*p* < 0.05, Mann-Kendal test).

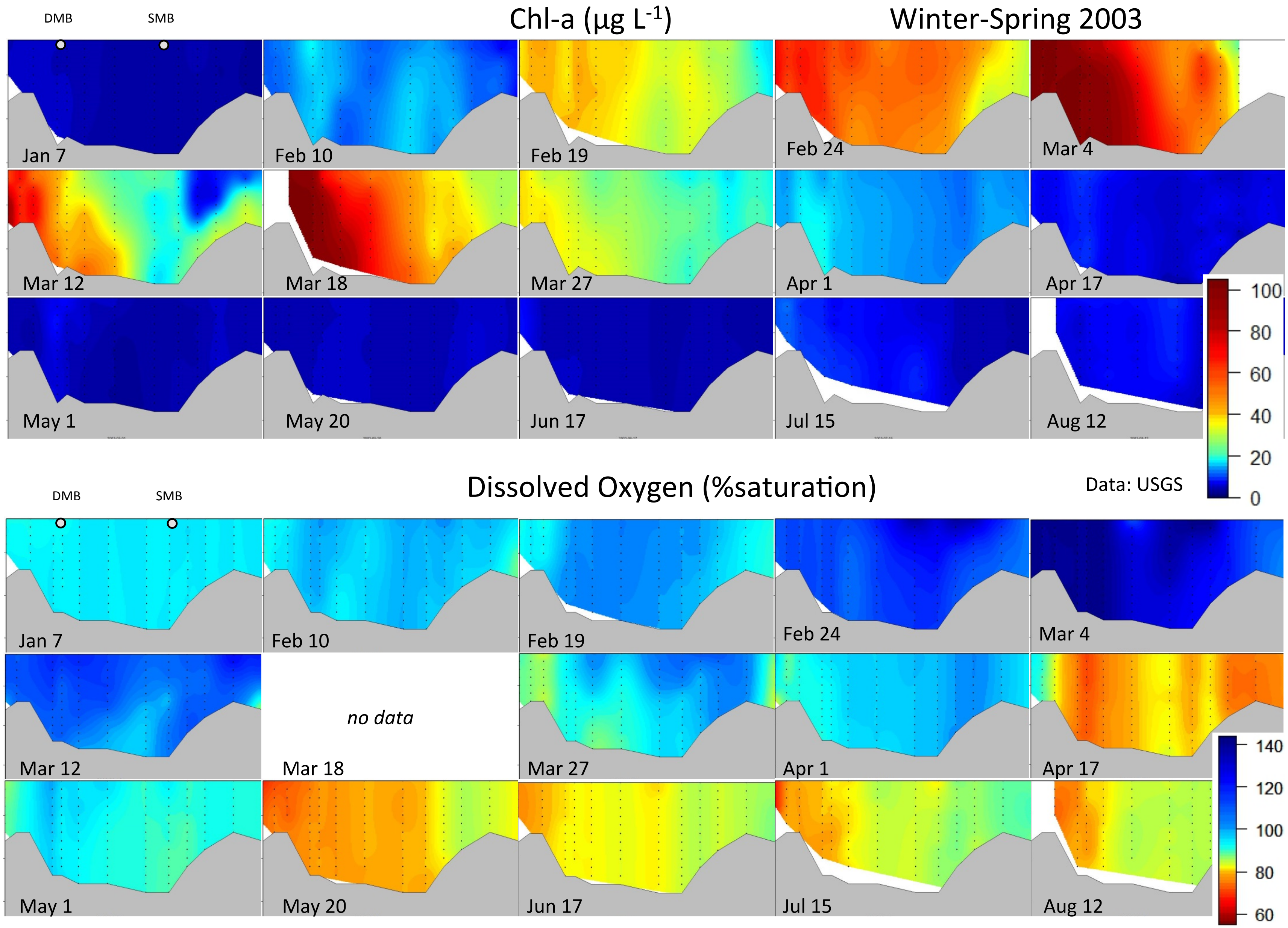


Figure 6.6 Chl-a concentrations and DO%sat in LSB and South Bay in Winter-Spring 2003. Black dots indicate locations where chl-a was measured on each date. Data: USGS. http://sfbay.wr.usgs.gov/access/wqdata/

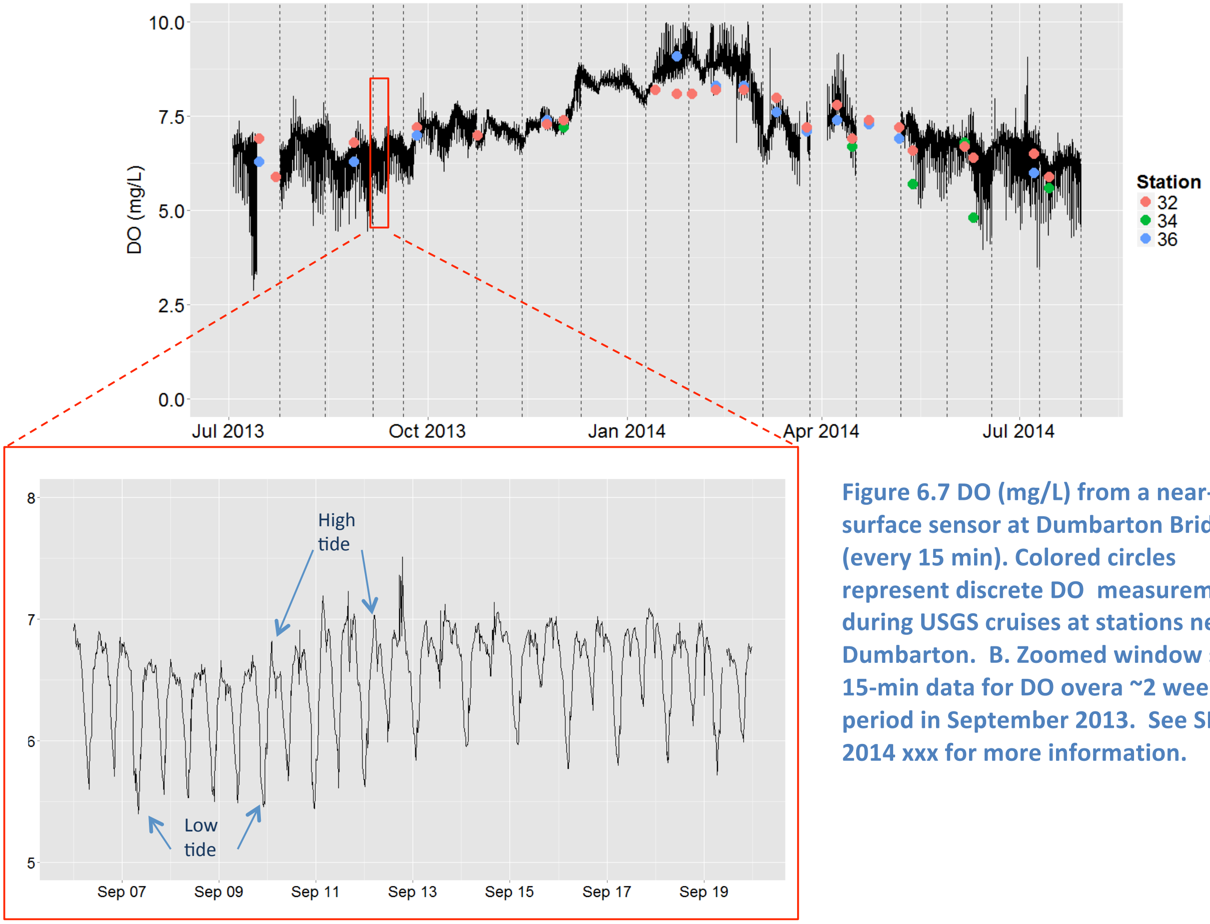


Figure 6.7 DO (mg/L) from a near-surface sensor at Dumbarton Bridge (every 15 min). Colored circles represent discrete DO measurements during USGS cruises at stations near the Dumbarton. B. Zoomed window shows 15-min data for DO overa ~2 week period in September 2013. See SFEI 2014 xxx for more information.

6.3.1.2 Sloughs, salt ponds, and Coyote Creek

A long-term DO record was available for a 12 km transect beginning near SJSCCWTP's discharge into Artesian Slough and continuing along Coyote Creek to where it opens into LSB, collected at weekly-to-monthly frequency for 25 years (1964-1990) (Figure 6.2 and Figure 6.8). During low flow months (May-Nov), SJSCWTP's discharge into Artesian Slough has generally been the primary source of freshwater to this area. During this time period, the stretch between station C-8-0 and C-1-0 was a mixture of LSB water and effluent, with salinities typically ranging between 10 and 25 ppt (See Section 2 Figure 2.15). Between 1964 and the late 1970s, low DO (e.g., < 2-3 mg/L) was commonly observed over extended stretches. DO concentrations increased markedly after the late 1970s, apparently in response to major upgrades at SJSCWTP's that resulted in major decreases in biochemical oxygen demand (BOD) and ammonium loads. The data in Figure 6.8 do not take into account tidal stage at the time of sampling, which depending on when sampling occurred could contribute some systematic bias to the data. Currently, effluent from SJSCCWTP is fully aerated (i.e., $DO\%_{sat}$ = 100%) prior to entering Artesian Slough. While SJSCCWTP effluent is now low in BOD, it nonetheless carries potential BOD in the form of inorganic nutrients that can be converted into algal biomass that subsequently exerts BOD, as discussed below and elsewhere in the report. Over the past 2 decades, data were not collected with the same consistency along this transect Coyote Creek.

Since wastewater effluent quality has changed substantially since the late 1970s, the data in Figure 6.8 does not necessarily provide the same types of insights into current conditions as some of the other datasets used in this report. It does, however, provide valuable context for how DO conditions have changed over time in Coyote Creek. DO measurements in Coyote Creek, collected as part of on-going investigations indicate that surface DO continues to periodically dip below 3-4 mg/L (J Hobbs, pers. comm.; SFEI, unpublished data) at a frequency that suggests a strong tidal influence.

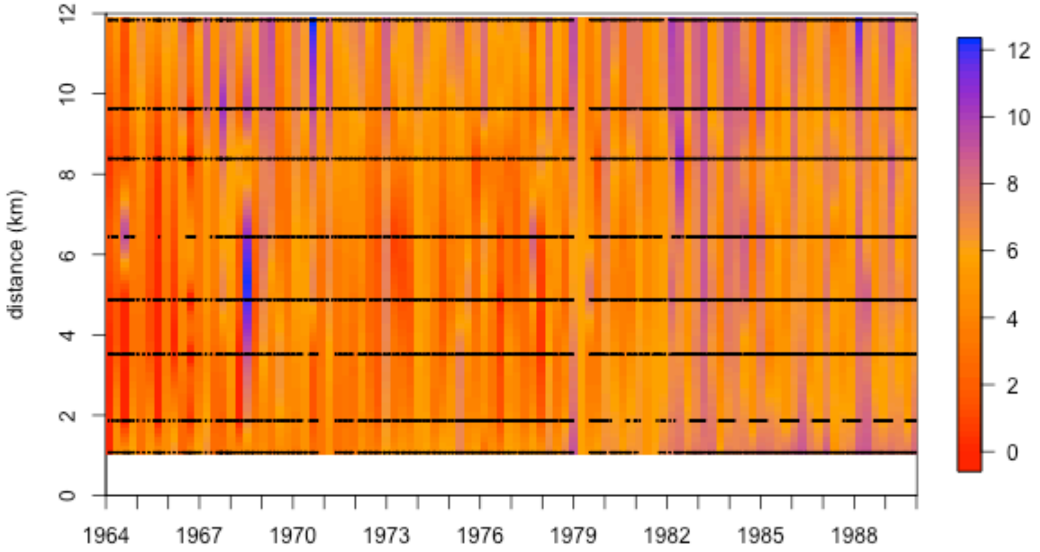


Figure 6.8 Seasonal and interannual variation DO (mg/L) along a transect extending from SJSC outfall (distance = 0) along Artesian slough and Coyote Creek to C-8-0 where Coyote Creek opens into Lower South Bay. Black points indicate actual data upon which contours were calculated. Data from SBDA and provided in electronic format by J Ervin (personal communication).

Compared to the abundant monitoring data available for deep subtidal habitats in LSB South Bay (Figure 6.2A and 2B) and the record for Artesian Slough and Coyote Creek, systematically collected DO data for sloughs and salt ponds are relatively limited. However, the observations that are available suggest that DO concentrations commonly dip below 5 mg L^{-1} in these shallow margin habitats, and frequently reach much lower values. A few studies have investigated DO and phytoplankton biomass in LSB salt ponds and sloughs. Salt ponds undergoing restoration have delicately balanced O_2 budgets. Some ponds have high phytoplankton biomass, fast phytoplankton growth rates, and therefore high O_2 production rates during daylight hours (Thebault et al., 2008), the result of high average light levels (lower SPM, shallower depth) and sufficiently high nutrient concentrations. Thebault et al. (2008) observed in one salt pond (A18) that DO concentrations could plummet in response to, for example, several consecutive days of day-time cloud cover or elevated T, both of which resulted in lower primary production rates that could no longer supply oxygen at rates that exceeded the competing high respiration rates (Figure 6.9A). As a result, large diurnal fluctuations in DO levels occurred (Figure 6.9). Topping et al. (2009) found that Pond A3W also experienced large diurnal DO fluctuations (Figure 6.9B).

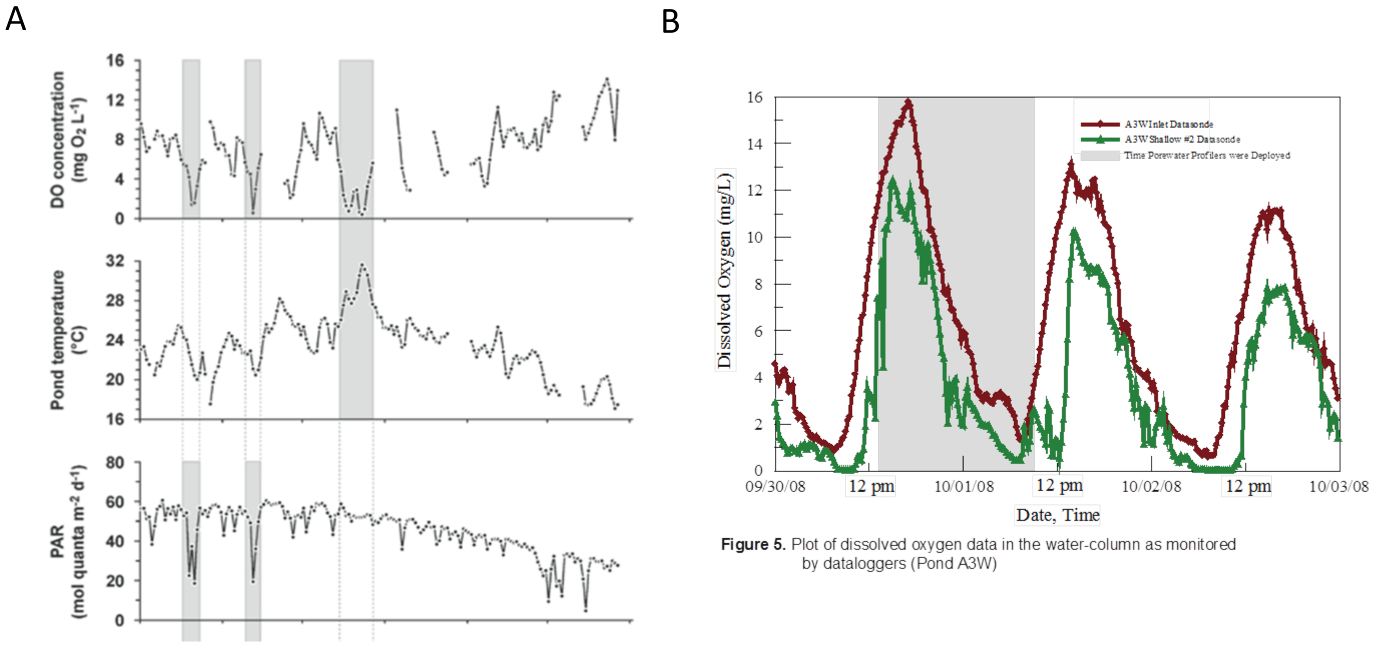


Figure 5. Plot of dissolved oxygen data in the water-column as monitored by dataloggers (Pond A3W)

Figure 6.9 A. Dissolved oxygen concentration in LSB salt pond A18. Grey bars indicate time periods when incident light was low (clouds) or temperatures were high enough to inhibit primary production. These factors lead to sustained periods of low DO. Source: Thebault et al. 2008. B. Dissolved oxygen concentrations in LSB salt pond A3W undergoing restoration Source: Topping et al. 2008.

A recent meta-analysis examined DO data from sloughs collected through several independent monitoring efforts in LSB and South Bay between 2004 and 2012 (Appendix 1). The analysis focused on data from in situ sensors that had been deployed at 10 slough sites and 14 restored salt pond sites. The frequency with which depressed DO concentrations were observed varied substantially among slough sites. DO < 5mg/L was observed with a median frequency (among sites) of 40%; in other words, half of the sites experienced DO <5 mg/L more than 40% of the time. Much of the variability among sites may be related to the fact that instruments were deployed at sites during different seasons or years, and for different durations. In general, DO deficits were observed more frequently in restored salt ponds than in sloughs (median frequency <5 mg/L = 55%).

Shellenbarger et al. 2008 measured DO concentrations at 3 slough sites during a set of short-term deployments in August 2007 (Figure 6.10). In two sloughs (Newark, Mowry), DO concentrations were near 5.5-6 mg/L during flood tide and dropped to 3-4 mg/L during both daily ebb tides, with the largest decreases occurring during the weaker of the two ebbs. In Guadalupe Slough, the maximum DO was closer to 5 mg/L, the low DO period lasted longer than the other two sloughs, and the minimum DO value was similar during both the weaker and stronger daily ebbs. Although there are clear differences between the DO time series, it is difficult to attribute those differences to slough-specific rates or processes because site selection was not based on a consistent set of criteria (e.g., tidal excursion, a discharge-adjusted distance, etc.).

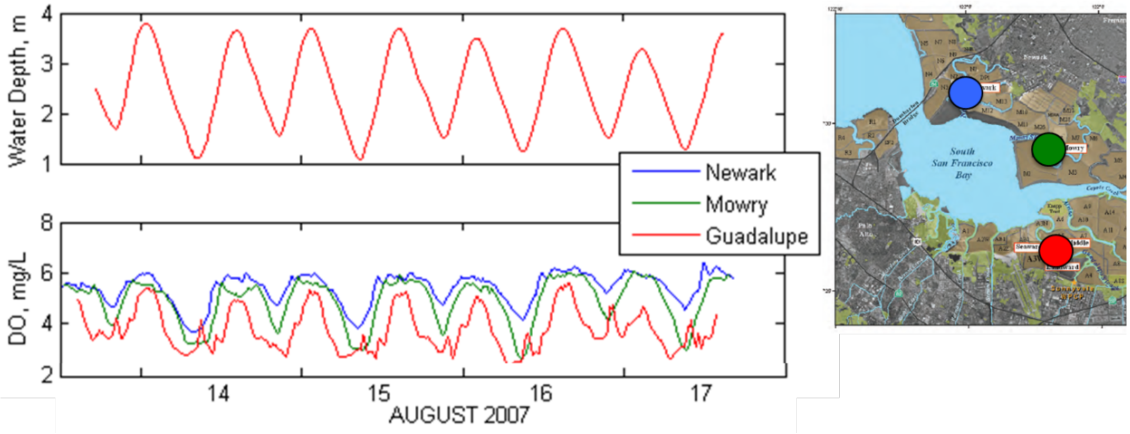


Figure 6.10 Dissolved Oxygen in three slough habitats in Lower South Bay measured using continuous monitoring sensors. Top panel shows water depth. Dissolved oxygen concentrations increase during the flood tide due to water with higher DO from LSB being tidally advected into sloughs. DO concentrations gradually decrease over the outgoing tide interval, likely caused, at least in part, by sediment oxygen demand within the sloughs, and lower DO water from up-slough moving back over the sensor. Colors of lines correspond to location denoted by circle colors in map. Source: Schellenbarger et al., 2008

The longest continuous DO record available for a slough habitat is from a station in Alviso Slough, approximately 4 km upstream of the Alviso-Coyote Creek confluence (Figure 6.2). At this site sensors were positioned ~0.5 m above channel bottom, and DO data have been collected continuously from June 2012-present. During the 10 month period shown in Figure 6.11 (Jun 2012-Mar 2013), measurements showed strong variability in DO concentrations over an annual cycle, and DO < 5 mg/L the majority of the time during June-October 2012. Zooming into a 3 month period in Summer 2012 (Figure 6.12), what may appear as 'noise' in Figure 6.11 appears as a strongly-periodic DO signal. Additional sensors were added to this site beginning in Fall 2013. DO concentrations again exhibited strongly periodic minima during Summer 2014 (SFEI 2014 #723). However, sharp peaks in DO, sometimes exceeding 100%, were also evident. In situ chl-a fluorescence measurements also varied strongly with tidal stage in Summer 2014, and the periodic maximum estimated chl-a concentrations reached levels that were several-fold greater than chl-a observed during the same time period at Dumbarton Bridge (see Section 5, Figure 5.9).

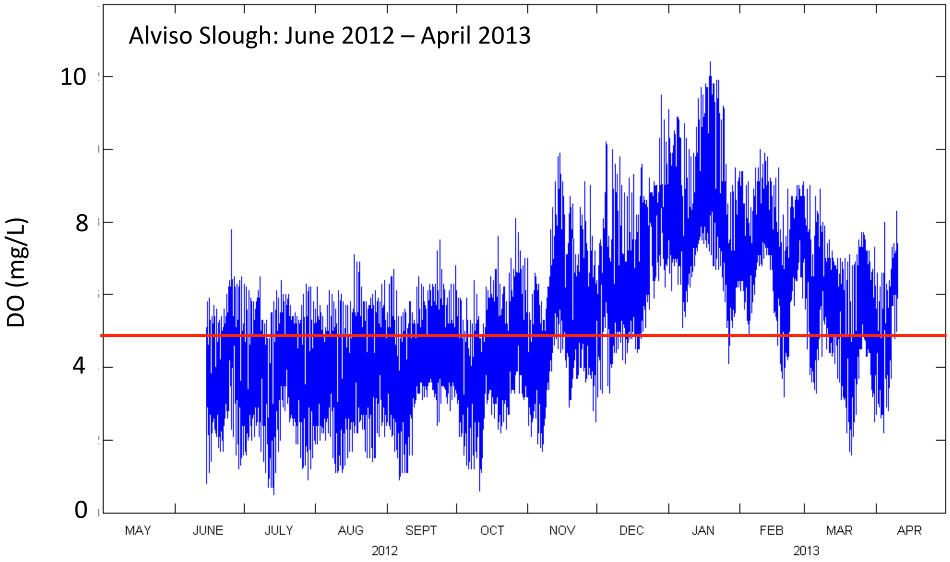


Figure 6.11 15-min dissolved oxygen concentration measurements over ~10 months in Alviso Slough. Sensor was positioned 0.5 m above bottom) in Alviso Slough. Red line indicates the 5 mg/L Basin Plan standadard. Data: USGS-Sacramento

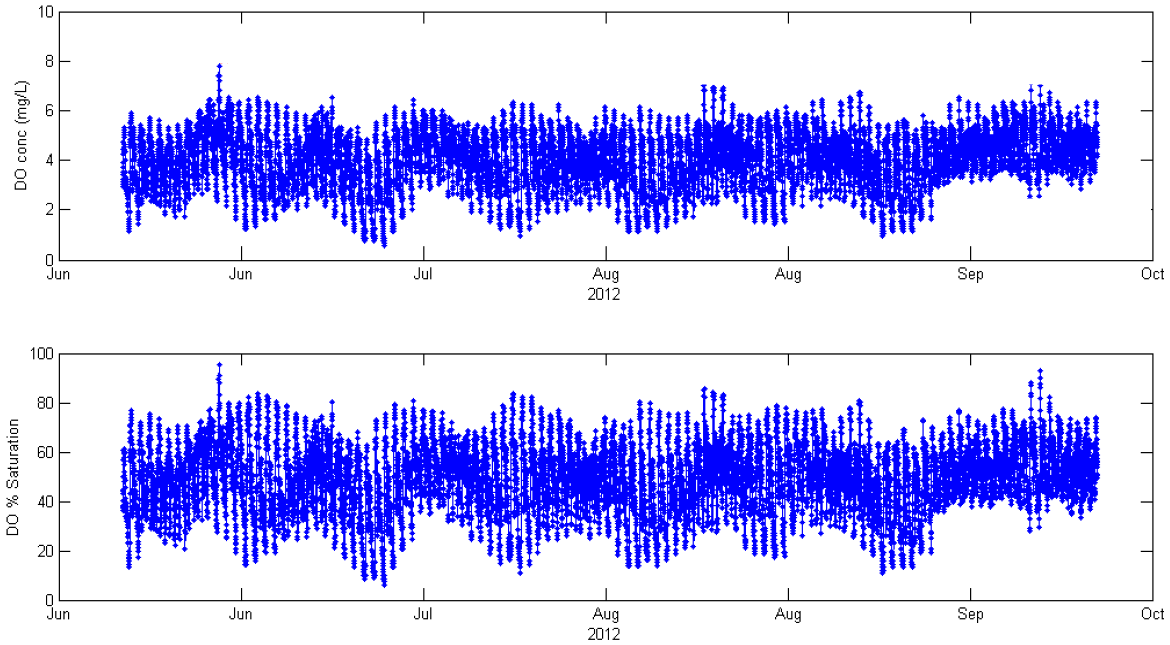


Figure 6.12 Dissolved oxygen concentrations in Alviso Slough, zooming in on a 3 month summer window of the data in Figure 6.11.

6.3.2 Factors contributing to observed DO variability

6.3.2.1 DO in sloughs

To further examine the variability in the Alviso Slough DO observations (Figure 6.11), we plotted DO concentration as a function of both time of day and day of year for summer 2012 (Figure 6.13). If DO concentration at the Alviso site had been driven primarily by the diurnal cycle of DO

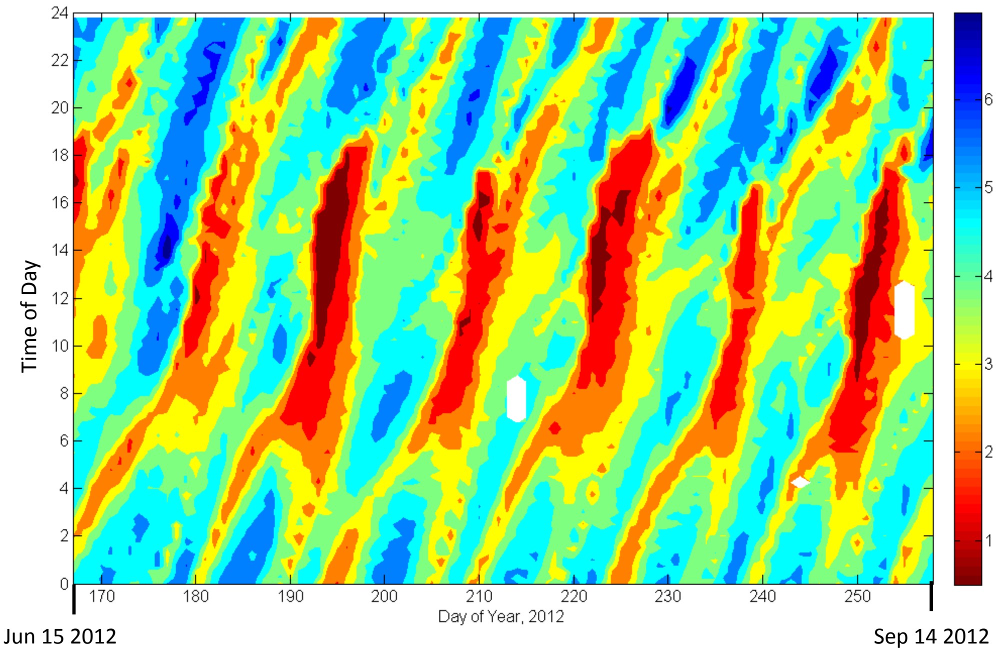


Figure 6.13 Dissolved oxygen concentration at Alviso Slough site, plotting data from Figure 6.12 as a function of both time of day and day of year. Contours represent DO in mg/L.

production, maximum DO would have been expected during mid to late afternoon. Instead, a different pattern emerged. Red-orange bands, signifying lowest DO levels, extended for 8-12 hours (early morning through early evening) were bounded by higher DO concentrations in the very early morning and late night. The low DO conditions re-occurred during this time window over ~5 days, were replaced for several days by higher DO conditions, and then returned 10-14 days later. An examination water level data recorded at the site revealed that the low DO periods co-occurred with neap tides, and that each daily minimum DO spanned the weaker of the two ebb tides.

We propose the following conceptual model, consisting of three linked mechanisms, to explain the DO concentration patterns at the Alviso site, and potentially other slough sites. The conceptual model and the mechanisms are intended as hypotheses that require further examination through additional field investigations and modeling.

1. *Dissolved oxygen concentration decreases rapidly in bottom waters when the slough water column becomes vertically-stratified during neap tides.* Neap tides have lower tidal velocities than spring tides. Lower velocities translate into weaker mixing energy, allowing vertical salinity stratification to develop when less-dense fresher water meets more-dense saltier water near the site. DO is consumed during the mineralization of organic matter; since the bottom layer is cutoff from the atmosphere, DO concentration in that layer decreases over time.

This hypothesized mechanism was identified through examining depth, salinity, and DO time series data, zooming in on narrower windows in time, e.g., during the neap tide between 7/10/2012-7/15/2012 (Figure 6.14). Maximum DO coincided with each day's highest tide. Over

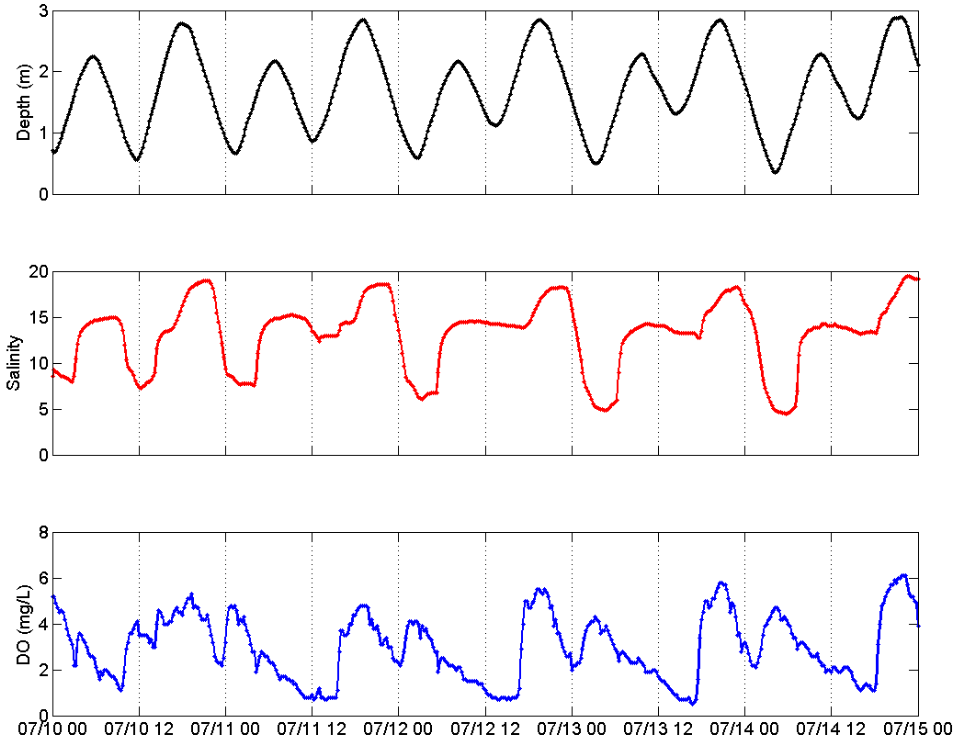


Figure 6.14 Time series of depth, salinity, and dissolved oxygen (DO) concentration during neap tide at Alviso Slough, 10 Jul 2012-15 Jul 2012. Tick marks on x-axis are at midnight (00) and noon (12) of each day. Lowest DO occurred on weaker daily ebb during period of near-constant salinity. Subsequently, DO concentration increased rapidly, coincident with an increase in salinity.

the subsequent ebb tide, salinity dropped sharply, with saltier water (~18 ppt) water gradually replaced by fresher water (~8 ppt) at the site. DO began decreasing before salinity, and actually rebounded slightly as depth reached its minimum, and then began decreasing again over the weaker flood tide and the subsequent weaker ebb tide, remaining at DO < 2 mg/L for several hours. Sediment oxygen demand (SOD) may have been fairly constant over this entire 5-day period. However, even a constant SOD would translate to a greater rate of concentration change during stratification since the mass is being removed entirely from the bottom layer and because O2 in the bottom layer could not be replenished through vertical transport or atmospheric exchange. A lower-salinity water mass was observed to move downstream at least as far as the Alviso sensor package. During each day's weaker flood tide salinity increased, likely due to a saltier water mass moving upstream, yet DO concentration continued to decrease. During the early stages of the subsequent flood tide, salinity returns to the previous day's maximum value, and DO increases from <1 mg/L to 5-6 mg/L within a couple hours, consistent with increased turbulent energy during the flood tide vertically mixing the water column and breaking down vertical stratification. While other processes undoubtedly contribute to the observed DO and salinity (e.g. longitudinal movement of water masses with different composition), these data suggested that vertical stratification may play an important role. However, no salinity or DO depth profile data was available during this time period to confirm that stratification was occurring.

To test the above hypothesis, vertical profile measurements of DO and salinity were performed along a 3.5 km transect from ALV site toward Coyote Creek (Figure 6.15) during a neap tide period in August 2014. Vertical salinity stratification was evident at ALV site, with the bottom 0.5-1m layer containing higher salinity, denser water. A sharp DO gradient was also evident, decreasing from >5 mg/L at depth = 1m to DO < 4 mg/L depth = 2m. The higher-salinity/lower-DO bottom layer (DO < 4.5 mg/L) extended ~1 km downstream from ALV, beyond which concentrations increased above 5 mg/L, although vertical DO gradients were still evident.

2. *Less flushing of the slough (i.e., less exchange with higher-DO open Bay water), occurs during neap tides than spring tides. As a result, DO concentrations (minimum, and mean) tend to decrease over time during neap tides.* Smaller differences in water level between high and low tides during neap compared to spring tides translates to smaller volumes of water being exchanged with the open-Bay during neaps tides than spring tides. During spring tides, a larger portion of 'old' water is flushed from the slough and replaced by higher DO water from the open Bay, whereas, during neap tides, there is less net flushing, and less injection of higher DO water. During spring tides in Summer 2012, minima DO concentrations tended to be greater than 2 mg/L and 30% saturation, whereas minima DO concentrations steadily decreased during neap tides to <1-2 mg/L and <20% (Figure 6.12).

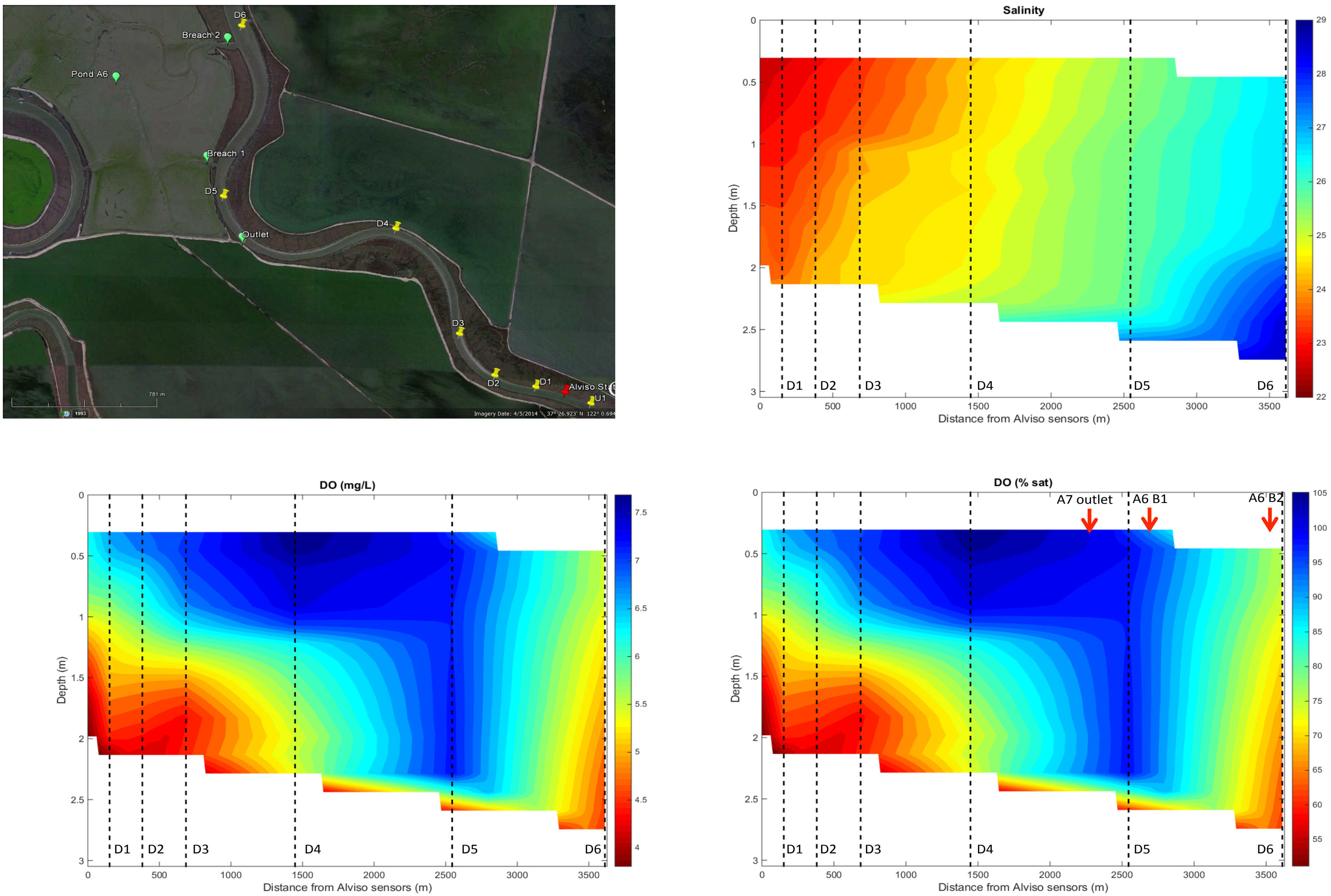


Figure 6.15 Salinity, DO (mg/L) and DO%sat measuread along a transect in Alviso Panel. Red marker in the upper left panel indicates the location of the Alviso moored station., and yellow markers indicate vertical profile stations. Measurements were made around slack low tide on the weaker of the two daily low tides.

3. A substantial source of labile (fresh) organic matter enters the sloughs or is produced in situ, leading to sufficient oxygen demand through pelagic or benthic respiration. Although no chl-a sensors were present at ALV during the period of June 2012-April 2013, chl-a sensors were deployed beginning in September 2013. In general, in fall 2013, winter 2013, and spring 2014, estimated chl-a concentrations in Alviso Slough were substantially elevated (5-10 fold) above those observed in the open Bay (e.g., Dumbarton Bridge; see Section 5 Figure 5.14 and SFEI 2014)), indicating that were was non-trivial source of fresh organic matter. Turbidity levels are fairly high at ALV, suggesting that in situ pelagic production rates may be low. However, Alviso Slough is in hydraulic communication with two adjacent salt ponds that could be major sources of phytoplankton biomass. The saturated and slightly super-saturated near-surface DO concentrations over the distance 0.6-2 km downstream of the Alviso moored sensor (Figure 6.15) offer evidence consistent with the notion of a recently-productive water mass entering the slough in this vicinity. Two salt pond connections (A7 outlet, A6 B1; Figure 6.15) are 0.75-1 km downstream of the surface maximum of DO%$_{sat}$, and water with maximum DO%$_{sat}$ could have easily been transported upstream during the prior flood tide and only partially migrated back downstream during subsequent weak ebb tide. The vertically well-mixed water column with near-100%sat DO at station D5 (Figure 6.15), which is located between those two outlets, was likely the result of DO-rich water entering Alviso Slough.

Benthic algae production along intertidal mudflats could also conceivably be a substantial source of new organic matter. Benthic algae could undergo degradation as algal mats die and decay. In addition, some of benthic algal biomass could be mixed into the water column if there is sufficient mixing energy at the bed during flood or ebb tides. Periphyton mobilized from marsh vegetation due to tidal mixing is another potential source of fresh organic matter.

Although the mechanisms described above were developed primarily for the observations at the Alviso site, some of them may be applicable to other sloughs. However, some of the mechanisms have certain prerequisites: e.g., connections to restored salt pond or other source of organic matter, source of fresher water such as watershed discharge or WWTP effluent to allow for stratification to set up. To a first approximation, the similarity between the tidally-driven DO patterns observed at the ALV in July 2012 and those at the Newark, Mowry, and Guadalupe Sloughs in August 2007 (Figure 6.10) suggest that, indeed, at least similar measurable “symptoms” develop, whether or not all the underlying causes are exactly the same.

6.3.2.2 DO in the open Bay

Based on the high-frequency DO data at Dumbarton Bridge (Figure 6.7), we hypothesized that spatial DO gradients occur in LSB and are detected at or near the Dumbarton Bridge during ebb tide. The magnitude of these gradients and their causes may vary seasonally, but could take the form of either

1. A substantial north-south gradient in DO concentration, the leading edge of which is measured by Dumbarton DO sensors during ebb tides, as conceptualized in Figure 6.16; or

2. Large differences in DO concentrations between the open Bay at high tide and the sloughs and creeks in the tidal margins, with lower DO margin waters draining into and mix with open Bay water during ebb tide. Even if there is no measurable north-south gradient in LSB at high tide, drainage of water from the margins could create a north-south gradient during ebb tides (Figure 6.16).

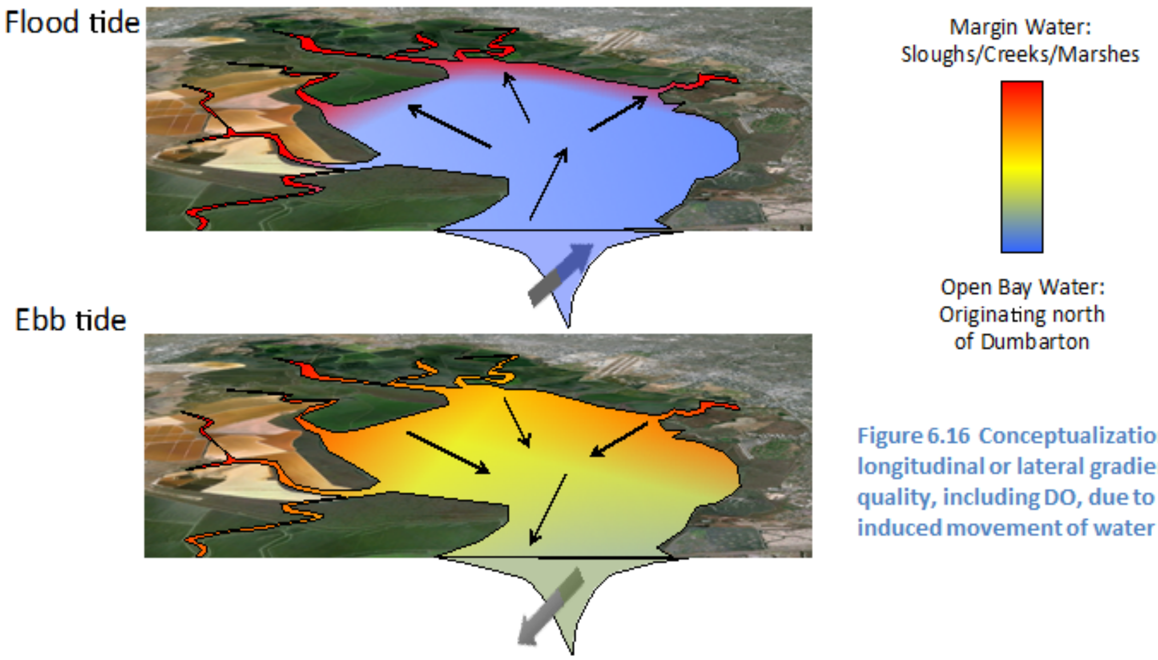


Figure 6.16 Conceptualization of longitudinal or lateral gradients in water quality, including DO, due to tidally-induced movement of water masses.

Since the data in Figure 6.7 are from a near-surface sensor, it is possible that the water column was not well-mixed, and that the observations were caused by a surface lens of less-dense fresher water with low DO moving past the sensor (e.g., draining from a nearby slough with a perennial freshwater source). Modest (5%) but sharp decreases in SpC did coincide with low tide and low DO. However, data from a sensor deployed 1.2 m above the bottom at Dumbarton Bridge also showed a similar DO pattern as the surface sensor, albeit with slightly smaller DO decreases (Figure 6.17). The similar DO relationship measured with near-surface and near-bottom sensors indicates that the water column was reasonably well-mixed. Vertical profiles of DO concentration along a lateral transect just south of the Dumbarton Bridge at both low and high water during spring tide in July 2014 provide further evidence of decreased DO levels at low water (end of ebb) throughout a vertically well-mixed water column (Figure 6.18).

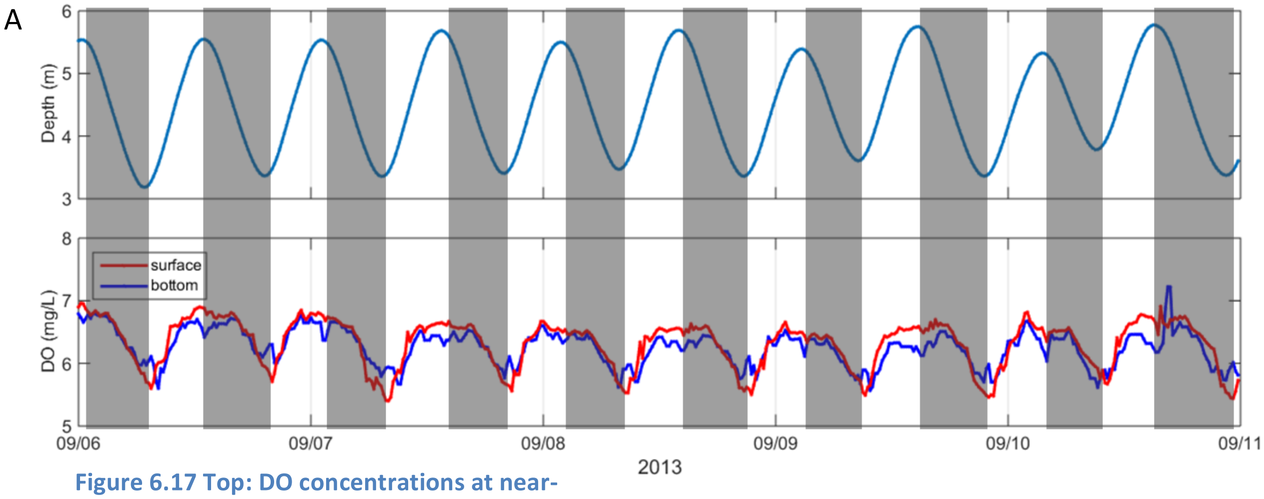
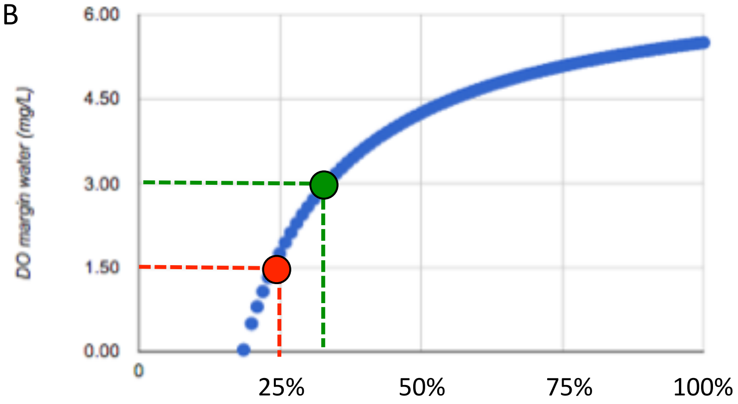


Figure 6.17 Top: DO concentrations at near-surface and near-bottom DO sensors at Dumbarton, indicating that the water column was reasonably well-mixed with lower-DO water occurring measured over the full depth. Right: A simple conservative mixing model to explore the question: If DO measured at Dumbarton at low tide (~5.5 mg/L) in the top panel was a mixture of water from the open Bay (6.5-7.0 mg/L) and water from margin habitats, what would the DO concentration of the margin water have been (y-axis) for different percentages of margin water (x-axis)?



At higher high water, DO concentrations over the entire water column exceeded 6 mg/L, with slightly higher surface concentrations (6.5 mg/L) than at depth, consistent with daytime production of DO in surface waters. Measurements performed along the same transect at lower low water show that the water column was reasonably well-mixed, with DO concentrations ranging between 5.0 and 5.5 mg/L for the majority of the cross section. There was a pronounced east-west DO gradient, with lower DO concentrations on the eastern edge suggesting that, at least on this particular ebb tide, a major source of lower DO water flowed along/from the eastern side. While the time series in Figure 6.17 and the transect profiles in Figure 6.18 present short glimpses or snapshots of condition, the longer record presented in Figure 6.7 indicates that the sharp DO decreases were a common occurrence during other periods in summer (2014) and early fall (2013).

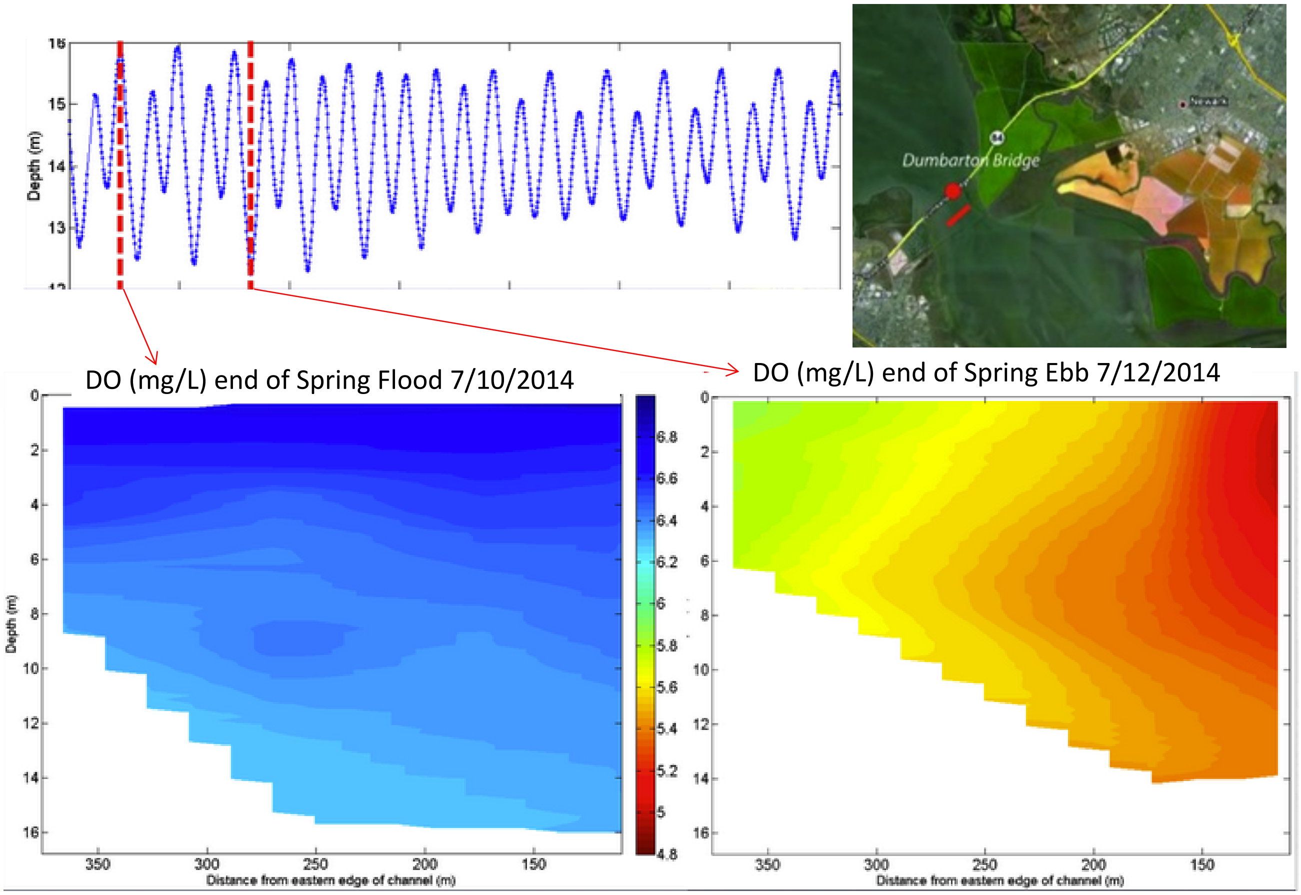


Figure 6.18 Vertical cross-sections of dissolved oxygen concentrations (bottom panels) measured along the same transect (red line in map) at two different tidal stages (left = high tide; right = low tide), as indicated by the detpth time series (upper left panel).

Beyond the vertical profiles described above and the *R/V Polaris* sampling - which at least in 2013-2014 did not capture most the apparent instances of lower DO (Figure 6.7) - there is currently limited data available on DO concentrations in LSB at locations south of Dumbarton Bridge, especially at low tide. Therefore the magnitude of horizontal DO gradients in the open Bay, or the DO conditions in margin habitats that mix into the open Bay, can only be inferred. It is possible, however, to begin identifying bounding DO concentration estimates using simple conservative mixing assumptions. If the tidally-driven DO decreases observed at Dumbarton Bridge are assumed to have resulted from mixing of higher-DO open Bay water with lower-DO slough or creek water, the average slough DO concentration can be estimated as a function of the mixing ratio of slough water to open-Bay water (Figure 6.17 B). If, for example, the fraction of slough water at Dumbarton Bridge at low tide was 0.25, the average slough DO concentration must have been ~1.5 mg/L to have caused the observed DO decrease. If the slough fraction had been 0.5, the slough DO concentration would have been 4.3 mg/L. Salinity or electrical conductance (EC) can also used as a tracer to further constrain the range of potential mixing ratios. No EC data are available from sloughs close to Dumbarton Bridge during the time periods in Figure 6.18. If the slough EC is assumed to be similar to maximum late fall 2013 EC values from Alviso Slough, the water at Dumbarton Bridge at low tide would have been 25-30% water from sloughs, suggesting that the average slough-water DO concentration was in the range of 1.5-3 mg/L. However, because of limited slough EC data, a contribution as high as 50%, corresponding to DO = 4.2 mg/L, cannot be strictly ruled out. To constrain the slough/creek contribution based on system bathymetry, we estimated the volume of water stored in sloughs, creeks, and marshes around the LSB margin as a function of tidal stage, and compared this with the volume stored in LSB. Based on these estimates, the water volume stored in sloughs/creeks/marshes at high tide is approximately 50% of the LSB open-bay volume at low tide. Thus, margin water could easily comprise a non-trivial portion of open-Bay water at low tide.

Patterns in DO concentration changes at Dumbarton Bridge over longer periods (days to weeks) suggest that open-Bay DO concentrations are also influenced by processes related to spring-neap tidal cycles (Figure 6.19). The difference between max (high tide) and min (low tide) DO concentrations tends to be greater during spring tides than ebb tides. More striking, however, is the clear decrease in maximum DO concentration during the approach to spring tides (difference of ~0.5 mg/L), and the gradual increase in max DO over the subsequent transitions to neap tides. We hypothesize that the decrease in max DO is caused by the gradual increased fraction of margin waters containing lower DO and potentially other reducing compounds (e.g., labile organic matter) from the margins into the open Bay where it exerts oxygen demand. Throughout the period of decreasing DO concentration, the increased fraction of low DO margin water and respiration of organic matter that it brings into the system - in addition to on-going oxygen demand within the open bay area - are competing with O2 production by photosynthesis and reaeration (since DO hovered around ~80-90% saturation) that add O2 back to the water column. As exchange with the margins decreases during the transition to neap tide, the combined rates of O2 production and reaeration exceed the rate of DO decrease, and max DO concentrations increase. This pattern - and the proposed mechanism - is the direct complement to the mechanism proposed for why max DO concentrations at ALV decreased during the approach to neap tides, and increased again during spring tides. While their complementary nature is in fact required - because they hydraulically linked - it is noteworthy that its effect on DO concentrations can be readily observed measured in both compartments.

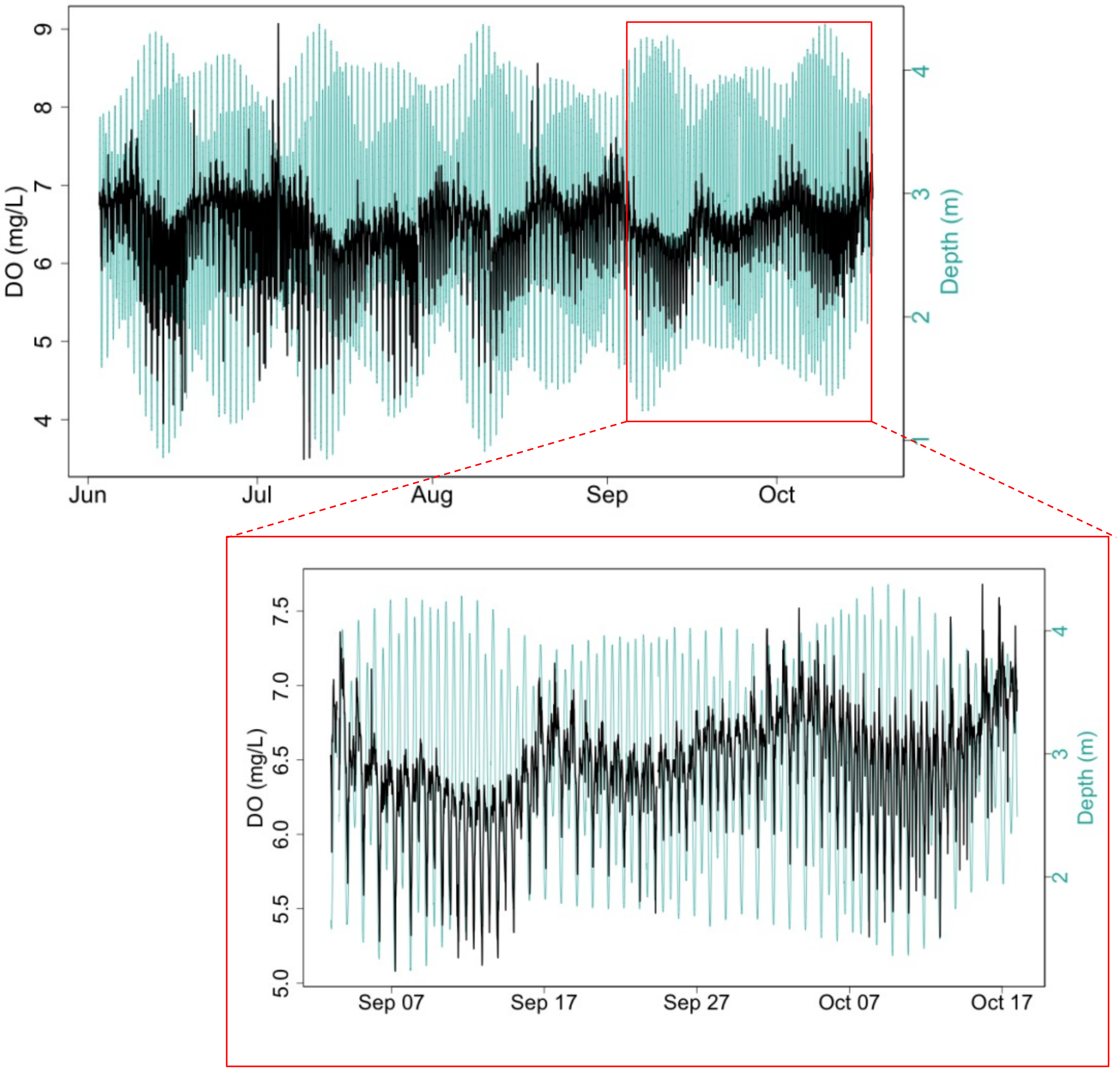


Figure 6.19 Dissolved oxygen concentrations (black) and depth (blue) at Dumbarton near-surface sensor, June – October 2014. Note the two patterns with different periods: 1. semi-diurnal tides leading to local DO minima at low tide; 2. DO minima and maximum values decrease over spring tides, and increase increase again during neap tides.

6.4 Summary and Major Research and Management Questions

In assessing the state of the science with regards to dissolved oxygen in Lower South Bay, we have identified the following major knowledge gaps:

1. What are the frequency, duration, spatial extent, and severity of low DO events in shallow margin habitats?
2. What causes the substantial tidal-variations in DO concentrations at Dumbarton Bridge?
    a. What are conditions like elsewhere (south of Dumbarton) during these times?
3. Are biota being adversely impacted by low(er) DO in the margins or open Bay?
4. What factors most strongly regulate DO in sloughs and creeks, and what data collection is needed to best predict DO condition?
    a. Organic matter source(s)
    b. Role of anthropogenic nutrients
    c. Physics: stratification, salt pond exchange, slough-open Bay exchange
5. What effects are salt pond restoration activities having on DO budgets in the margins? The open Bay?
6. What effects would potential management actions have on DO? E.g.,
    a. Decreased nutrient loads by 25%, 50%, 75%?
    b. Operation of restored salt ponds (e.g., including optimization for nutrient removal and beneficial habitat condition)
    c. Managed oyster or mussel reefs

We propose a number of high-priority activities to address these knowledge gaps. Since some of these investigations would be resource- and time-intensive undertakings, some level of prioritization is still needed:

- Systematically investigate DO in the margins
    - High frequency DO monitoring and ancillary data
    - Evaluate relative importance of mechanisms that control DO through data interpretation and modeling
    - Gather data to use in model calibration/validation
- Gather high-spatial resolution data through biogeochemical mapping
    - Characterize spatial and temporal heterogeneity
    - Assess condition across the region – extent and severity of potentially problematic events
    - Gather data for model calibration/validation
- Conduct mechanistic field investigations to quantify important processes related to physical processes, nutrient cycling and phytoplankton and benthic algae production (slough ←→ open Bay, salt pond ←→ slough, stratification in open Bay and sloughs)
- Develop and apply a coupled hydrodynamic and biogeochemical model for Lower South Bay, including sloughs and margins
    - Evaluate mechanisms that control phytoplankton biomass and DO concentrations through sensitivity analysis
    - Examine the role of anthropogenic nutrients and quantify nutrient fate

- Forecast ecosystem response under potential future conditions, including changing environmental factors (sediment concentrations, bivalves)
- Quantify how potential management actions, such as nutrient load reductions and salt pond operation, will influence ecosystem response (phytoplankton biomass, DO)
- Characterize and quantify uncertainty

7. Fishes of South Bay and Lower South Bay: Synthesis of existing data and conceptual model for fish response to nutrients

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7.1 Introduction

This section provides an overview of the current knowledge regarding fish in South Bay and Lower South Bay (LSB). The San Francisco Bay Estuary is fortunate that several long-term monitoring efforts to quantify the distribution and relative abundance of fish and macro-invertebrates have been going on for several decades. In addition many shorter term studies have been conducted in South Bay for wastewater discharge permit compliance and scientific investigations for restoration. However, very little of this information has been published in the peer reviewed literature. Here we provide:

- an overview of the different monitoring programs and studies that have taken place in South Bay and LSB,
- describe the methods of sample collections and provide summaries of species found in South Bay and LSB,
- explore seasonal and inter-annual trends, identify different habitat types used by species and discuss life history characteristics that are important to take into consideration when developing indicators for nutrient related impacts.
- Conceptual models of nutrient and hypoxia effects on fish; and
- A semi-quantitative assessment of DO tolerance levels for the most frequently observed species.

7.2 Approach

7.2.1 Data Sources

Fish species abundance and distribution trends were derived from several sources, described below. There are several other sources of fisheries data for Lower South Bay, but were not used in this review due primarily to different sampling methods, availability of raw data or missing key data regarding sampling efforts and locations.

San Francisco Bay Study- (SFBS), California Department of Fish and Wildlife (1980-present)

The San Francisco Bay Study (SFBS) was established in 1980 to determine the effects of freshwater outflow on the abundance and distribution of fish and mobile crustaceans in the San Francisco Estuary, primarily downstream of the Sacramento-San Joaquin Delta (Baxter et al. 1980). Sampling ranges from south of the Dumbarton Bridge in Lower South San Francisco Bay to the confluence of the Sacramento and San Joaquin Rivers in the North Bay. Data from this study utilized trawl data from South Bay stations from 1980-2012. The South Bay is defined as the area south of Candlestick Point on the west shore and the entrance to the Alameda Ship Channel on the east shore down to the Dumbarton Bridge and contains 8 stations, 3 in the deep channel (108, 107, and 101) running down the axis of the bay and 5 in shallow shoal habitats (106, 105, 104, 103 and 102). The SFBS uses two gear types, a benthic otter trawl and a mid-water trawl. The otter trawl, which samples demersal fishes, shrimp, and crabs, is towed against the current at a standard engine rpm for 5 minutes then retrieved. The otter trawl is equipped with a 4.9 m head rope, a 2.5 cm stretch mesh body, and a 0.55 cm (opening across the widest dimension) delta knotless mesh codend. A 5:1 scope (ratio of cable out to water depth) was used to keep the otter trawl on the bottom. . The midwater trawl, which samples pelagic fishes, is towed with the current at a standard engine rpm for 12 minutes and retrieved obliquely such that all depths are sampled equally. The midwater trawl mouth was 3.7 m x 3.7 m. The mesh graduated in nine sections from 20.3 cm stretch mesh at the mouth to 1.3 cm (½") stretch mesh at the codend. A 5:1 scope was used when setting the midwater trawl. It was towed with the current for 12 minutes and retrieved obliquely. Fish, caridean shrimp, and brachyuran crabs are identified, measured, and counted. Shrimp and crabs are also sexed. Sampling effort is quantified (i.e. distance towed, volume of water filtered) and salinity, water temperature, Secchi depth, and station depth are measured; wave height, tide, cloud cover, and tow direction are categorized. The length, catch, and effort data is used to calculate catch-per-unit-effort (CPUE) by species and age class. The CPUE data is used to calculate monthly and annual abundance indices. Annual Status and Trends reports summarize recent changes for the most commonly collected species. These reports are published in the Spring issue of the IEP Newsletter, which can be found at http://iep.water.ca.gov/report/newsletter. The 1999 IEP Technical Report, "Report on the 1980-1995 Fish, Shrimp, and Crab Sampling in the San Francisco Estuary, California", is a good source of basic information. This report is out-of-print, but can be found at www.water.ca.gov/iep/docs/tech_rpts/tech_rprt_63_toc.html.

Marine Science Institute – (MSI) (1980-present)

The Marine Science Institute (MSI) is a non-profit science education organization based out of Redwood City, California. MSI has been conducting science based cruises with an educational focus in South San Francisco Bay for 35 years. Typically two cruises are conducted per day with three otter trawls conducted per cruise daily between June and October in South San Francisco Bay, offshore of Bair Island. The otter trawl had a 4.9-m headrope length, 3.8cm mesh in the body and 0.64-cm mesh cod end. Trawls were conducted for 10 minute and all fish were sorted, identified by a biologist, counted and measured for standard lengths. Salinity and temperature 1-m below the surface was done with a thermometer and hydrometer. The MSI trawled monthly (from Oct. 1970 to 1981 (missing only 9 months during this period) , intermittently in 1985-1986 and monthly from Oct. 1992 to 2012 (missing 33 months, primarily Jan-Feb during this period). Over the course this study, MSI had conducted 9,936 trawls collecting almost 90,000 fish. A summary report titled "Trends in South San Francisco Bay Fish Populations from 1972-2002" can be found here. http://www.werc.usgs.gov/

South Bay Salt Pond Restoration Project-(SBSPR), UC Davis (2010-2013)

The fish monitoring program for the South Bay Salt Pond Restoration Project was initiated in summer of 2010 to monitor the response of fish and macroinvertebrates to restoration of former commercial salt ponds. The project utilized a variety of methods to monitor fish and macroinvertebrates in several habitat types, including shallow sloughs, tidally restored ponds, tidally muted ponds and marsh plains, among four marsh sites in South San Francisco Bay (Alviso Marsh, Eden Landing Marsh, Bair Island Marsh and Ravenswood Marsh) (Figure 7.2). Shallow slough tidal ponds were monitored using a four-seam otter trawl with a 1.5-m depth-4.3-m width opening, a length of 5.3-m, and a mesh size of 35-mm stretch in the body and 6-mm stretch in the cod end. A 5:1 scope (ratio of cable out to water depth) was used to keep the otter trawl on the bottom. Tows were conducted against the current for 5-minutes at an engine speed of 3,500 revolutions per second. Tidally muted ponds were monitored with a 30-m beach seine with 6.4-mm mesh and minnow traps. Marsh plains and creeks were also monitored with minnow traps. All fish, invertebrates are identified to species, counted and measured and released. For each site, temperature (degrees Celsius, °C), salinity (approximated by practical salinity units, PSU), dissolved oxygen parameters (percent saturation, and milligrams per liter, mg/L), and specific conductance (microSiemens, µS) were recorded using a Yellowstone Springs Instruments (YSI) model 85 meter. Water clarity was measured using a Secchi disk and recorded in centimeters (cm). Depths at which the trawl was towed were also recorded. This study provides species habitat use information among many different habitat types that are not sampled by the California Department of Fish and Wildlife's long-term monitoring studies. Data reports can be found on the South Bay Salt Pond Restoration Project's website (http://www.southbayrestoration.org/).

South Bay Dischargers Authority (SBDA) -, Phillip Williams and Associates (1981-1986)
The South Bay Dischargers Authority project was a five year study on the water quality and biological resources of Lower South Bay, downstream of the San Jose-Santa Clara Wastewater Facility located at the top of Artesian Slough in the Alviso Marsh. Monthly surveys for fish and macroinvertebrates were conducted in the Lower South Bay from Artesian Slough and Coyote Creek in the Alviso Marsh to the base of the Dumbarton Bridge in Lower South Bay, from December 1981 to November of 1986. Monitoring was performed pursuant to requirements of the San Francisco Bay Regional Water Quality Control Board. The project conducted three 10-minute otter trawls at 5 stations. All fish and six of the dominant invertebrate taxa were identified to species and counted. Water quality, salinity and temperature were measured at each station during trawls. Hard copies of the final report are available from Phillip Williams and Associates.

7.2.2 Site Description

For the purposes of this section, South Bay (SB) is defined as the waters extending from around the city of Oakland to the Dumbarton Bridge (Figure 7.1); where the bay is constricted on both sides by land masses jetting out into the bay from the east and west sides of the bay (Conomos 1979). Lower South Bay (LSB) extends from the Dumbarton Bridge up into the Alviso Marsh which has two of the main freshwater inputs in South Bay; the Guadalupe River and Coyote Creek. As described in Section 2, three wastewater treatment plants discharge to LSB, including the San Jose-Santa Clara Regional Wastewater Facility which contributes ~90110 million gallons per day of treated effluent to the estuary. Because of its geographic position within San Francisco Bay, and the predominant flow of freshwater originating from the rivers feeding into the Delta, tidal and river flows disconnect South Bay from the rest of estuary, resulting in an estuary that has features similar to coast lagoons, with low freshwater input and high residence time of waters within South Bay (Conomos 1979). South Bay and Lower South Bay contain a vast diversity of habitat types, with vast shallow shoals and intertidal mudflats.

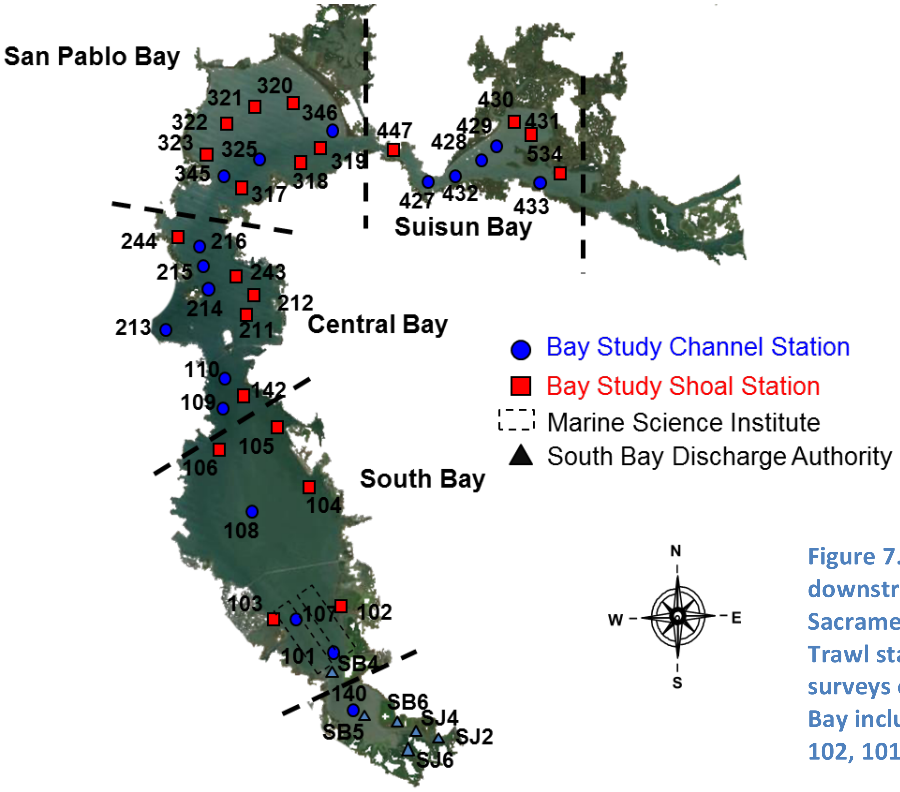

Figure 7.1. Map of San Francisco Bay downstream from the confluence of the Sacramento and San Joaquin Rivers. Trawl stations are identified for the four surveys examined. SFBS sites for South Bay included 106, 105, 104, 108, 103, 102, 101 and 140.

7.2.3 Methods

We examined fish count data for the four fish monitoring studies conducted in South San Francisco Bay from 1972 to 2013. Each survey used a benthic otter trawl, thus some catch abundance similarity did exist, although the SFBS, and SBSPR utilized additional gear types to sample different habitats. Comparisons of catch between surveys was not our focus for this report, rather we sought to combine data to determine the general abundance status and numbers of species encountered, explore the species-habitat associations and document the species, lifestage and monthly distribution of species to support the assessment of nutrient related impacts to the fish assemblages of South San Francisco Bay. We calculated the catch per trawl (standardized by number of minutes per trawl) for each species across all trawl stations and years and scored rank abundance (ordered most numerous to least) for the each survey. We then characterized each species abundance status by summing ranks across all surveys. Natural breaks in relative abundance trends were used to score species as abundant, common, uncommon, and rare. To characterize distribution between different habitat types and marshes we calculated the proportion of the total catch found in different habitat types surveyed by each study, for example the SBSPR sampled sloughs and pond habitats and we report proportions of total catch in each habitat type for a species. Lastly, we characterized the relative abundance and lifestage for common and abundant species by month and between habitat types for surveys that sampled multiple habitats, thus characterizations are between habitat types within surveys.

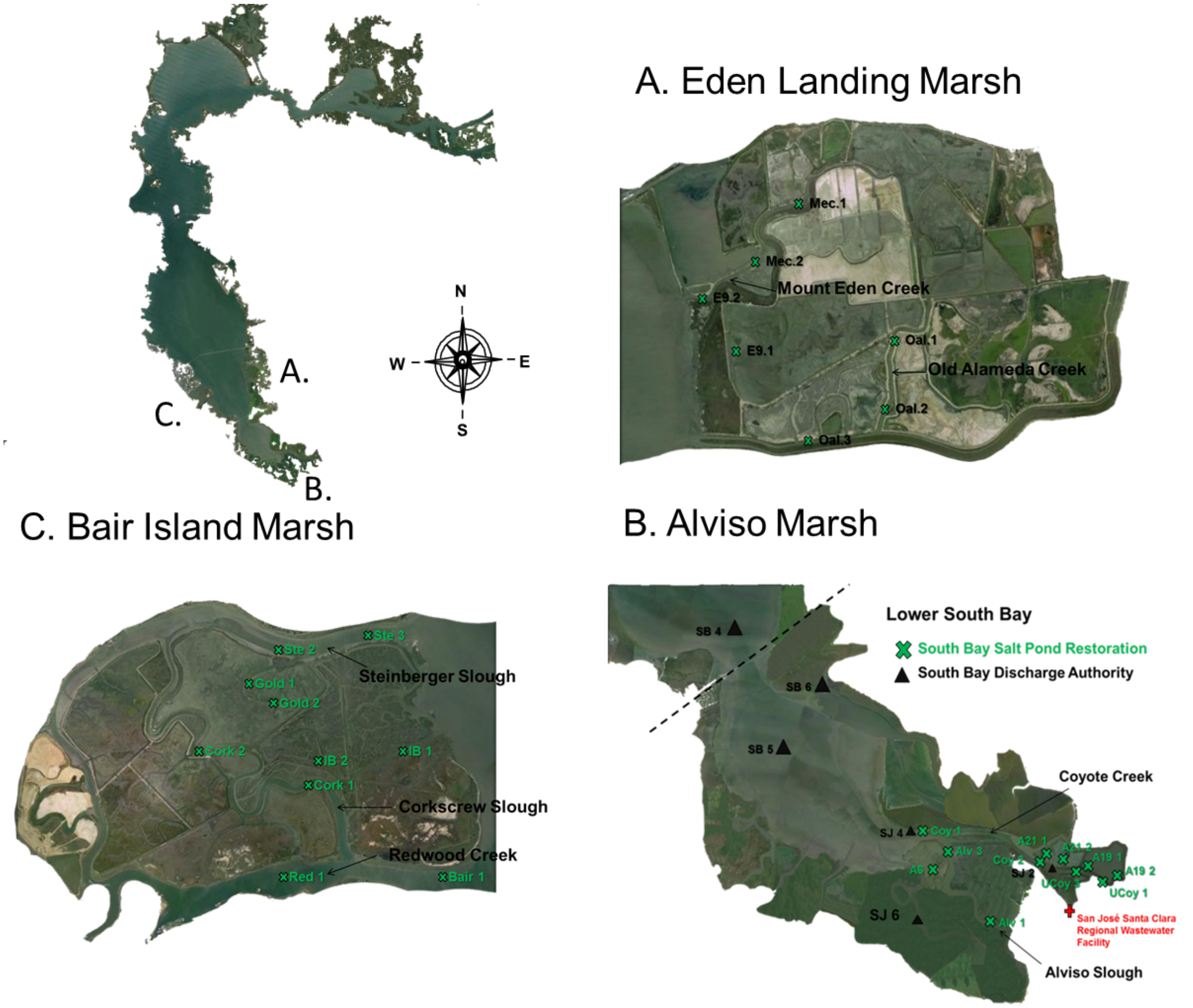

Figure 7.2. Map of the South Bay Salt Pond Restoration Fish Monitoring Project marsh sites. A. The Eden Landing Marsh sampling consisted of sampling in Mount Eden Creek, Old Alameda Creek and the tidal restoration of Pond E9. B. The Alviso Marsh consisted of sampling Alviso Slough, Coyote Creek and the tidal restoration ponds A6,8, 19-21. Also on this graphic are the sites for the SBDA. C. Bair Island Marsh consisted of sites in Redwood Creek, Steinberger Slough, Corkscrew Slough and tidal restoration pond in Outer Bair Island on the Steinberger Slough.

7.4 Fish Assemblage Classification

Fish species assemblages can be characterized by a variety of ecological, biological or life history traits. Several attempts have been made to characterize fish assemblages of estuarine systems based on life histories as well as spatial and temporal distribution, however the application of these attempts often are specific to difference estuarine systems, and may not appropriately serve our needs to evaluate the role of nutrient impacts on fish populations. In general we follow the classification scheme established by Allen et al. (2006), with minor modifications. Species are classified based into five major categories, (1) freshwater taxa are those that occur at salinities typically less than 2-ppt, but can be infrequently found at slightly higher salinities, (2) diadromous taxa are those that migrate between marine and freshwater (or brackish water) for spawning purposes. Most common of these are anadromous species, which mature in the ocean and enter freshwater to spawn. (3) Estuarine residents are taxa that exhibit significant variability in salinity (euryhaline) and temperature (eurythermal) tolerance and complete the lifecycle in the estuary. (4) Marine migrants are dissected into two classes, one that includes adults that migrate into bays and estuaries to spawn or give birth, and those taxa that enter the bay as larvae or early juvenile stages that are spawned offshore. We make this distinction because (*i*), it is well recognized that larval mortality is high in fishes and population regulation can occur during the early life stages, which in this case, occurs in the ocean rather than in the estuary. (*ii*) larval stages can be sensitive to stressors associated with nutrient effects. (5) Marine species are those taxa that occur in all life stages in the nearshore environment and can also be found in estuaries for short periods of time. Our classification of some species differs from Allen et al. (2006), and may reflect species taking advantage of unique habitats or open niches in the San Francisco Estuary. For example, Allen at al. (2006) classified the Threespine Stickleback as a freshwater species, while we classified this species as an estuarine resident, with significant numbers being found in salinities as high as 18-ppt in South Bay. Below we present examples of our classification of South Bay's fish assemblages. See Table 7.1 for a complete list of species and their classification.

7.4.1 Freshwater Species

In South Bay freshwater taxa are not commonly found in the bay, primarily only in high outflow years. Freshwater taxa are found in the upper ends of slough habitats along the bay margin and Lower South Bay's Alviso Marsh, where freshwater input, at time can be high. Freshwater taxa include, the Sacramento Sucker, Common Carp. Largemouth Bass and Threadfin Shad.

7.4.2 Diadromous Species

Very few species found in South Bay are considered diadromous. Many of the anadromous species of the estuary are species that have experienced declines in abundance and are not commonly found in South Bay surveys but may use the habitats found there for brief periods of time, including Green Sturgeon, Steelhead Trout, Chinook Salmon, American Shad, Striped Bass and Longfin Smelt.

7.4.3 Estuarine Residents.

The estuarine resident species are abundant taxa in the South Bay. The most abundant estuarine resident species include the Pacific Staghorn Sculpin, Shiner Surfperch as well as other seaperchs, Three-spine Stickleback, Topsmelt, Jacksmelt, Inland Silversides, Arrow Goby, Longjaw Mudsucker, Bay Goby and Yellowfin Goby. Within the estuarine resident group, some species are more associated with low salinity habitats, such as Three-spine-Stickleback and Inland Silverside and usually have a positive population abundance trend with more freshwater inflow from the watershed. The Pacific Staghorn Sculpin is found in many different salinity environments and can range from near full strength sea water to freshwater as juveniles.

7.4.4 Marine Migrants

7.4.4.1 Ocean Recruits

Ocean recruits are primarily species characterized as marine or marine migrant species (Allen et al. 2006 classification scheme) that are found in South Bay as juveniles, but are typically spawned offshore, in the ocean. These species often immigrate into San Francisco Bay in the spring, when coastal upwelling currents deliver larvae and early juveniles near shore, where they are entrained into the San Francisco Bay plume. Marine-Ocean Recruits typically only spend a few months in South Bay before migrating back into the ocean. Distinguishing ocean recruits from other groups is important because the population trends of ocean-supplied recruits in the estuary are dependent on the status of adult populations in the ocean and mortality in the ocean during the early larval stage and thus when evaluating their populations in regards to nutrient impacts, the supply of recruits from the ocean must be taken into consideration. Moreover, ocean circulation patterns often undergo multi-year cycles (e.g. Pacific Decadal Oscillation) influencing delivery of larvae to the estuary, thus population trends of recruits in the estuary may be influenced by ocean circulation patterns. Species we classified as ocean recruits include, English Sole, California Halibut, , ,and rockfishes. No known spawning of these species has occurred in the San Francisco Bay Estuary, although we speculate that California Halibut spawn in the bay.

7.4.4.2 Estuarine Recruits

Estuarine recruits are species often characterized as marine migrants that come into San Francisco Bay from the ocean as adults to spawn or give birth to eggs, larvae or developed juveniles. The young may rear in the estuary for only a few months, such as Pacific Herring , Starry Flounder or several years as the case with White Croaker, Leopard Shark, Bat Ray and Brown Smoothhound before migrating to the ocean. Like ocean-derived recruits, the population status of estuarine-derived recruits depends on adult populations in the ocean, however; they differ in that the supply of recruits is dependent upon adults migrating into the estuary and not necessarily on ocean currents.

7.4.5 Marine Fishes

The marine fish taxa are characterized by the uncommon and rare species found in South Bay. These species are typically found in the bay in the summer and fall months when bay salinities are greatest. Species that comprise this group include Pacific Sardine, Big Skate, and Shovelnose Guitarfish.

7.5 Species Status

South San Francisco Bay harbors great fish species diversity with 89 fish species encountered by the four studies examined in this report. (Table 7.2a-d). The Marine Science Institute (MSI), the oldest ongoing survey recorded a total of 121 fish species encountered since 1972, although several taxa were only identified to the family level or recorded as unidentified taxa thus the total number is likely less. The San Francisco Bay Study (SFBS) encountered 87 species of fish since 1980 in the South Bay stations. If we include Central Bay, the species count was greater than 100 species and for the entire San Francisco Bay. The shorter studies encountered fewer species: the South Bay Discharger Authority encountered 39 fish species in five years of trawling and the South Bay Salt Pond Restoration Project encountered 54 fish species. There were differences in the number of species encountered between marsh restorations of differing age by the SBSPR, with Eden Landing Marsh being the newest restoration project and harboring the fewest species. Invertebrate taxa encountered by these studies were also extremely diverse, although not all taxa are identified and counted during the surveys. Typically only macro-invertebrates (i.e. larger than 1-mm) were counted and many only down to major taxonomic levels, making quantification of species diversity difficult.

We characterized 15 species of fish as abundant, 21 species as common, 20 species as uncommon and 29 species as rare among the four surveys in South Bay (Table 7.2a-d). The Northern Anchovy was by far the most abundant species making up approximately 60% of the catch in South Bay, (MSI and SFBS). The Pacific Staghorn Sculpin was the second most abundant species overall and was the top ranking species in Lower South Bay (SBDA and SBSPR). The species assemblages collected by the SFBS and MSI studies in bay habitats of South Bay were similar, although some rare species were collected in only one of the two surveys. There was less overlap between the South Bay surveys and the Lower South Bay, although this is likely due to the many more years of sampling conducted in the South Bay by the MSI and SFBS compared to the shorter SBDA and SBSPR studies. A great majority of species were found in more than one habitat type emphasizing the importance of habitat diversity and having monitoring surveys operating in different habitats (Table 7.2a-d).

7.6 Fish-Habitat Associations

The South Bay and Lower South Bay contain a variety of habitat types from open waters to shallow mudflats and a patchwork mosaic of bay margin habitats including production salt ponds, managed ponds, restoration ponds and extent tidal marsh. Salt pond habitats comprise a relatively novel ecosystem of shallow high salinity habitat that did not exist before the development of the bay area. The fish communities of South Bay and LSB can arranged by habitat type with different habitat features within the major habitat types supporting different fish communities, for example the shallow shoal habitats have a variety of benthic habitat types such as shell mounds, rip rap and other structural debris, bridge pilings, hard sandy bottoms and fine silt soft bottoms, which all support different invertebrate communities and subsequent fish communities. Oyster beds and sea grass meadows were once in great abundance in South Bay. Many of the species that utilize these habitats still occur in San Francisco Bay, and many still utilize the small remnant patches of habitat.

7.6.1 Deep Channel

Channel habitats are the parts of the open bay that are greater than 5.5 meters below the Mean Lower Low Water (MLLW) tidal datum. A deep channel habitat runs the axis of the bay down to just below the Dumbarton Bridge (Figure 7.3). The sediments of the channel habitats vary widely from course sands to very fine clays and silts. A majority of the channel habitat of South Bay consists primarily of mud, a mixture of more than 80% fine silts and clays. The deep channel serves as habitat for most of the species in San Francisco Bay. Species commonly found in deep channel habitats include small pelagic species (Northern Anchovy, Longfin Smelt, Pacific Sardine), benthic species (White Croaker, Plainfin Midshipmen, Pacific Staghorn Sculpin, California Tonguefish, Yellowfin Goby) and larger species of fish including Bat Ray, Leopard Shark, and Big Skate (Table 7.2). Very little is known about the foraging activities of fish in the deep channel habitats. Species occupying deep channel habitats may in fact make short foraging excursions into the shallow shoals as the same species are often found in both habitats. The deep channel habitats may actually concentrate the species into a narrow area and thus would appear to be more abundant in bottom trawl surveys compared to adjacent shallow shoals which are much broader, affording the fish to spread out and thus be less dense and less likely to be captured in great abundance in bottom trawls.

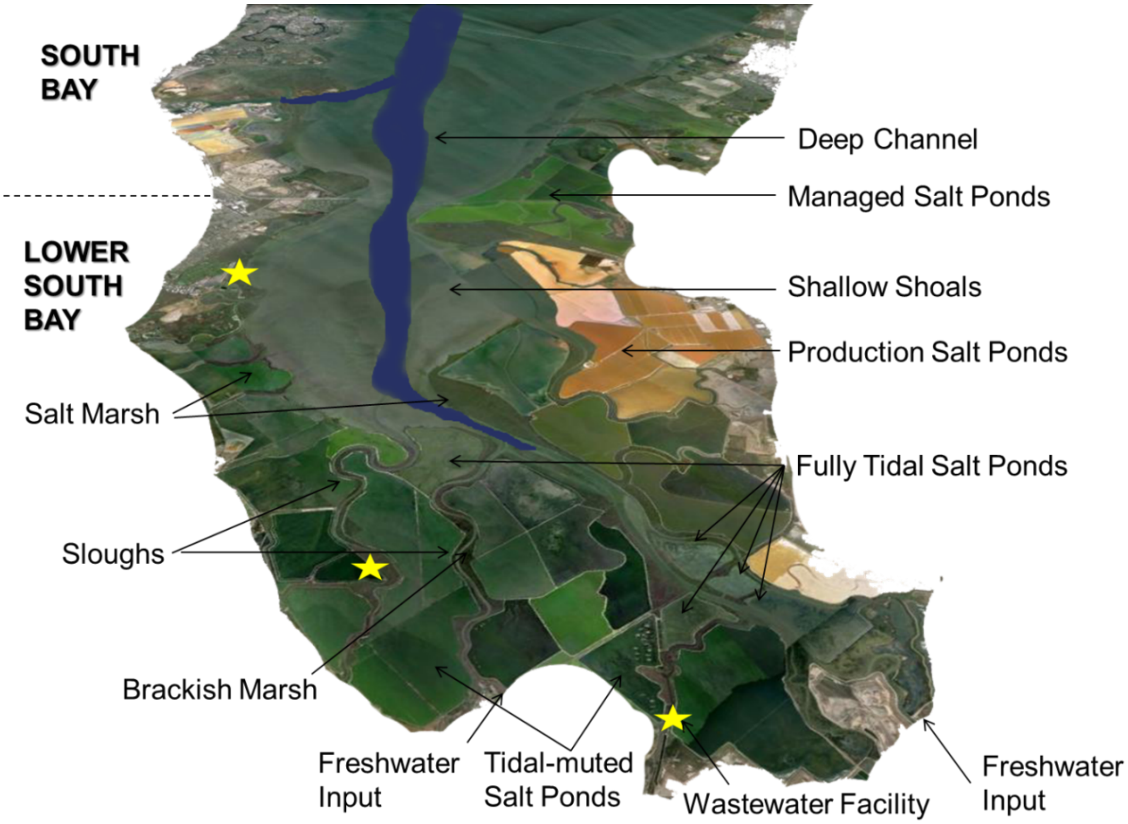

Figure 7.3. South San Francisco Bay and its major habitat types. (We may put some up close pictures of these habitats in this graphic, but it might be too big)

7.6.2 Shallow Shoals

The shallow shoals include portions of the open bay that are below 5.5 meters-MLLW. The sediments are comprised primarily of mud, with areas of remnant shell fragments from historical oyster farms for the native and introduced oysters. The shallow shoal makes up a large portion of the open bay habitat in South Bay and is almost the entirety of Lower South Bay. The shoals are utilized by a plethora of species for feeding on abundant in-faunal invertebrate communities. The assemblage of fishes found in the shallow shoals habitats of South Bay are generally the same composition as the deep channel habitats however; several species are often more abundant or frequently encountered in the shoals, as is the case for many of the more abundant species found in bottom trawl surveys, including Pacific Herring, Cheekspot Goby, Shiner Surfperch, Bay Goby, English Sole, Jacksmelt, Topsmelt, Speckled

Sanddab,and Chameleon Goby (Table 7.2). These species are found to be foraging on the abundant infaunal invertebrate communities occupying the shallow shoal habitats, and are particularly abundant in the spring through fall. The shoals have several micro-habitat types, making this habitat structurally diverse, supporting a large number of species. The shoal micro-habitats include the remnants of once abundance sea grass beds and oyster reefs, sandy patches with interspersed shell fragments, soft silt mud patches and hard muddy clay patches. Species like the Speckled Sanddab and English sole tend to be associated with a sandy substrate where they can use their coloration to blend in with their environment, while gobys and smelt are more likely to be found in muddier bottom types where species can sift through the soft silts for prey or use the more turbid habitats to avoid predation. Many of the rare species are known to be associated with shell mounds and sea grass beds in other estuaries, (Bay Pipefish, rockfishes, greenlings, gunnels to list a few).

7.6.3 Sloughs

Slough habitats are those small often sinuous shallow channels that form the connection between the open water bay habitats and the tidal marsh habitats. Slough habitats are often in the range of 5.5 to less than 1 meter depth and exhibit large swings in depth with the tides. The bottoms of sloughs can have a variety of mud, clays silts and shells depending on the width and tidal energy with areas with high tidal energy and flow having more clay and large debotage type bottoms and smaller slower moving slough with fin silts and detrital materials. Many fish species can be found in slough habitats, which are often used as pathways between the open bay habitat and tidal marshes where foraging occurs. Many of the species that are commonly found in shallow shoals are also found in slough habitats. The most common species found in slough habitats of South Bay include the Pacific Staghorn Sculpin, Three-spine Stickleback, Northern Anchovy, English Sole, American Shad and Pacific Herring. The Northern Anchovy is the most abundant species in South Bay and is in greatest abundance in the summer months in Lower South Bay sloughs, while Pacific Herring are more abundant in the winter to spring months when water temperatures are cooler. The pelagic species feed primarily on mysid shrimps and copepods, while benthic species feed on infaunal invertebrates, amphipods and detrital materials. The shallow littoral sections of the sloughs are dominated by the Three-Spine Stickleback, Top Smelt, Jacksmelt and the introduced Inland/Mississippi Silverside. Like the pelagic species, these assemblages feed mostly on planktonic organisms such as copepods and mysid shrimps when available. The benthic portions of the slough are dominated by Pacific Staghorn Sculpin, English Sole, Yellowfin Goby, Leopard Sharks, Bat Rays and Arrow Goby who mainly feed on small shrimps, clam siphons, worms and amphipods, with Leopard Sharks and Bay Rays also feeding on fish. These species tend to be more abundant in spring and fall in Lower South Bay sloughs. Most species found in slough habitats utilize these habitats in different seasons, with summer assemblages and winter assemblages. This may be due to the strong seasonal variability in environmental conditions such as water temperature and salinity.

7.6.4 Tidal Flats

The tidal flats, include broad flat shallow intertidal habitats composed of mud, sands and shell deposits. The mudflats comprise the largest area of tidal flats and one of the largest habitats of South Bay and support high densities of infuanal invertebrate communities, including policheate and oligocheate worms, amphipods, and small clams. These tidal flats are extremely important for the productivity of the bay by creating vast surfaces for diatom mats to flourish, providing fuel, in addition to phytoplankton production, to the base of the South Bay foodweb. At high tides the tidal flats are major foraging areas for flatfishes, such as Starry Flounder, English Sole, Speckled Sanddabs and Pacific Halibut, as well as other benthic foraging species such as Pacific Staghorn Sculpin, Top Smelt and a variety of other less common species. These flats are also areas of high predation by larger fishes such as Leopard Sharks and Bat Rays. Unfortunately, the tidal flats are the least studied in terms of fish assemblages and very little information exists even though this habitat type is one of the dominant features of South Bay.

7.6.5 Tidal Marsh

Tidal marsh is defined as those intertidal areas dominated by vegetation from the lowest reaches with twice daily tidal inundation to the highest tideline where vascularized land plants dominate. The margins of the bay were once one of the largest tidal marsh habitats on the west coast, however due to the reclamation of marshlands for residential, agricultural and salt production, greater than 90% of these habitats where lost (Atwater 1979). Tidal marshes can be delineated by the salinity regimes experienced with the highest salinity tidal marshes deemed salt marsh, lower salinity areas, brackish marsh, and in upstream areas freshwater marshes. Tidal salt marsh habitats have experienced that greatest change with the shorelines of San Francisco Bay being transformed for urban, agriculture and production salt ponds. Remnants of the original tidal salt marsh still exist (e.g. lower Newark Slough Marsh) however greater than 90% if the historic tidal salt marsh was lost (Atwater 1979). Unlike the other habitats of South Bay, relatively few species are found to occur in the intertidal shallow creek habitats that interlace the tidal salt ponds and brackish marshes (Table 7.3). This is likely due to the highly ephemeral nature of these habitats, since they are inundated only during high tides. The most common species occurring in the intertidal creeks of tidal marsh habitats is the Longjaw Mudsucker. The Longjaw Mudsucker is the only species that can consistently occupy these intertidal habitats, being capable of tolerating long periods of time exposed to air (Moyle 2002). Other species found opportunistically taking advantage of the flooded creek and marsh plains include, Pacific Staghorn Sculpin, Three-Spine Stickleback and Top Smelt. Leopard shark and Bat Rays are often found just outside the creeks where they enter the slough to feed on fishes and shrimps vacating the marsh as the tides recede. Macro-Invertebrates are also abundant in tidal marsh, species include the Caridean shrimps such as the grass shrimp *Crangon franciscanum* and Asian prawn *Paleomon macrolepidotus* and a brachyran crab the Oregon mud crab *Hemigrapsus oregonensis*.

7.6.6 Salt Ponds

One of the novel habitats in the South Bay are salt ponds. Before Spanish settlement of California, Native Americans visited the shorelines of South Bay to the salt pannes, where salt water on marsh planes would form ponded water and evaporate, leaving behind the salts from the bay waters (ver Plank 1958). Because of the vast flat low lying areas surrounding the bay, tidal marsh habitats were leveed from the bay and connected with water culverts to create large networks of ponded habitats that extended from shore where salinities would be similar to the adjacent bay and then progressively shunted landward where evaporation would occur and salinity levels would increase until reaching the terminal landward pond where salt would precipitate out and be harvested for commercial use. Salt ponds provide habitat for a select few fish species that can tolerate high salinities for short or long periods of time, including Top Smelt, Longjaw Mudsucker, and Three-Spine Stickleback. The salt ponds support large population of migratory shorebirds and waterfowl, including the endangered western snowy plover.

7.6.7 Restored Salt Ponds.

Another novel habitat type in South Bay is the restored salt pond. With the creation of the South Bay Salt Pond Restoration Project, several thousand acres of former production salt ponds were reconnected to the bay resulting in tidal inundation of former gypsum flats. Inundation of these former salt production ponds has resulted in the creation of large expanses of off-channel shallow, often intertidal pond areas. The restoration includes ponds that were fully tidal, whereby the levees were breached to MLLW and the tide is allowed to freely enter and retreat, and ponds where the tide is muted by a water control structure keeping the water level above MLLW. The progress of transformation of different ponds within the landscape of the South Bay has created a variety of salt ponds in varying stages of restoration. Ponds located close to the bay margin have had rapid sediment deposition and colonization of saltmarsh plant species, including the pickleweed *Sarcocornia sp*. and cordgrass *Spartina sp*, while pond further form the bay and presumably receiving less sediment are much slower to accrete a marsh plane were plants can become established. These ponds resemble nearby tidal mudflats and often support a difference infaunal invertebrate community, and function differently than ponds that have exhibited rapids marsh plant colonization. The restored salt ponds support a similar species assemblage to the adjacent slough habitats (Table 7.3). Common inhabitants include American Shad, Arrow Goby, Bat Ray, Bay Pipefish, English Sole. Jacksmelt, Longfin Smelt, Longjaw Mudsucker, Inland/Mississippi Silverside, Northern Anvhovy, Pacific Herring, Pacific Staghorn Sculpin, Rainwater Killifish, Shiner Surfperch, Starry Flounder, Striped Bass, Three-Spine Stickleback, Top Smelt and Yellowfin Goby and have barrow ditches ringing the marsh plan, leftover of the construction of the surrounding levees with a clam shell dredge. Often fish catches using bottom trawl gears can be higher in the barrow ditches within the restored salt ponds compared to slough habitats however; due to the much narrower barrow ditch this may be the result of confining the fish into a smaller space and easier catchability compared to wider sloughs. Nonetheless, soon after reopening the ponds to the slough, the species assemblages utilizing the restored ponds resembles the fish assemblages found in the slough. The one exception being a greater incidence of capturing the Longjaw Mudsucker and these barrow ditch slough like habitats are very close to the tidal creeks and are often shallow and narrow enough to resemble large tidal creeks.

7.6.8 Sea Grass and Oyster Reefs.

Sea grass meadows and native oyster reefs were a prominent habitat feature in South Bay historically, but have been significantly reduced or all but lost, in the case of native oyster reefs. Several species of fish have been associated with seagrass meadows such as the iconic Bay Pipefish, which has a morphology that closes resembles a leaf of seagrass. Other common species found in seagrass beds in Northern California estuaries include, the sea perches, including Shiner Seaperch, Black Seaperch, Rubberlip Seapearch, to name a few. Many of the seaperches are known to also utilized wharf or bridge pillings and other debotage found in shallow shoals. The Pacific Staghorn Sculpin, English Sole, Speckled Sanddab and juvenile lifestages of nearshore species such as the rockfishes, cabezon, lingcod are often found in sea grass habitats. Oyster reefs also support similar assemblages of fish including the seaperches as well as cryptic species including Arrow Goby, Fringehead, Buffalo Sculpin and Plainfin Midshipmen (Allen et al. 2006).

7.7 Monthly Patterns, Lifestage and Restoration Types

We examined monthly catches of the abundant and common species that were collected in the SFBS and SBSPR to determine temporal patterns in species habitat usage and for those abundant species with multiple lifestages and habitat patterns (Table 7.3). Monthly patterns can be difficult to interpret due to the shorter time series of sampling conducted by the SBSPR, but comparisons between the deep channel, shoals samples by SFBS and the sloughs and restoration ponds revealed interesting patterns. In the deep channel and shoal habitats most species were caught year round and some species exhibited season shifts in abundance and distribution between the deep channel and the shoals, e.g. Brown Rockfish. Fish tended to be more abundant in the spring and summer months, when juvenile lifestages f are recruiting into the size range catchable by the trawl gears, e.g. English Sole, Northern Anchovy, Brown Rockfish, California Halibut, Diamond Turbot, Speckled Sanddab, and Starry Flounder. Seasonal patterns also reflected the reproductive timing of spring spawning species in the bay, e.g. Bat Ray, Leopard Shark, and Pacific Herring. Fewer species were more abundant in the winter months and consisted of adult or sub-adult lifestages, e.g. American Shad in the sloughs and ponds, Bay Pipefish in the channels and shoals, California Halibut in deep channels, Longfin smelt, Pacific Staghorn Sculpin in the deep channels, adult Pacific Herring, adult Plainfin Midshipmen, adult Starry Flounder.

Few species were exclusive to either study, which reflected habitat preferences by species, e.g. Inland/Mississippi Silverside were not collected in the deep channels or shallow shoals by the SFBS, but are one of the most abundant species in the sloughs and salt ponds. These species are characterized as fish that live in very shallow waters. Many of the larger bodied marine species were not encountered in the shallow slough or pond habitats, such as Big Skate and Shovelnose Guitarfish. The slough habitats and tidal restoration ponds had similar species and relative abundances and overall fewer species than the deep channel and shoals reflecting habitat selections but also the lack of sampling conducted in the sloughs and ponds. The muted tidal ponds had fewer species than the tidal ponds, and tended to support greater numbers of invasive species, such as the Yellowfin Goby and Inland/Mississippi Silverside.

7.8 Conceptual Model of Fish Population Drivers

While the pelagic fish community in the northern San Francisco Bay Estuary has been extensively studied over the past few decades, South Bay and LSB fish communities have received limited attention. The abundance of pelagic fishes in North Bay have undergone significant declines in recent years, largely attributed to the effects of increased grazing by invasive claims on phytoplankton and zooplankton populations (Sommer et al. 2007; Baxter et al. 2010, Mac Nally et al. 2010). While many of the species associated with the Pelagic Organism Decline in the North Bay are not commonly found in South Bay, pelagic species common to South Bay have also undergone declines, such as the Northern Anchovy and Pacific Herring, the numerically most abundant species in the estuary (Kimmerer et al. 2006; Fish et al. 2013). Meanwhile some species that recruit to San Francisco Bay from the ocean have increased due to changes in the Pacific Decadal Oscillation and other ocean-climate related productivity cycles (Cloern et al. 2007; Fish et al. 2013).

The conceptual model below was designed to provide a framework to aid evaluation of the potential effects of nutrients on the distribution and abundance of fish in South Bay and Lower South Bay. The model is not meant to be an exhaustive evaluation of fish populations and the effects of excess nutrients on estuarine ecosystems; rather it aims to provide a structure to develop testable hypotheses given the potential outcomes of nutrient addition to the estuary. To do so, we start with an overarching conceptual model of factors that influence the distribution and/or abundance of fish, including physical factors and biological factors and how they interact. We the develop a nested conceptual model that illustrates the potential effects of excess nutrients on the estuaries structure and functions, including the effects low dissolved oxygen, as this is the most likely impact on fishes in Lower South Bay and South Bay.

The fish population model is largely derived from early versions of the Pelagic Organism Decline conceptual models (Baxter et al 2010), that emphasizes top down and bottom up classical ecological regulation of populations, as well as incorporates a stock-recruitment relationship with prior abundance linkages, since most of the species we encounter in monitoring surveys are young-of-year fish (Figure 7.4). Biological effects through predation and food web effects are considered in context of the underlying physical and chemical environment and acknowledge that fish populations can often be driven entirely by physical processes rather than biological one. In our model we include nutrients as a specific chemical driver as well as outline distinct habitat types in South Bay.

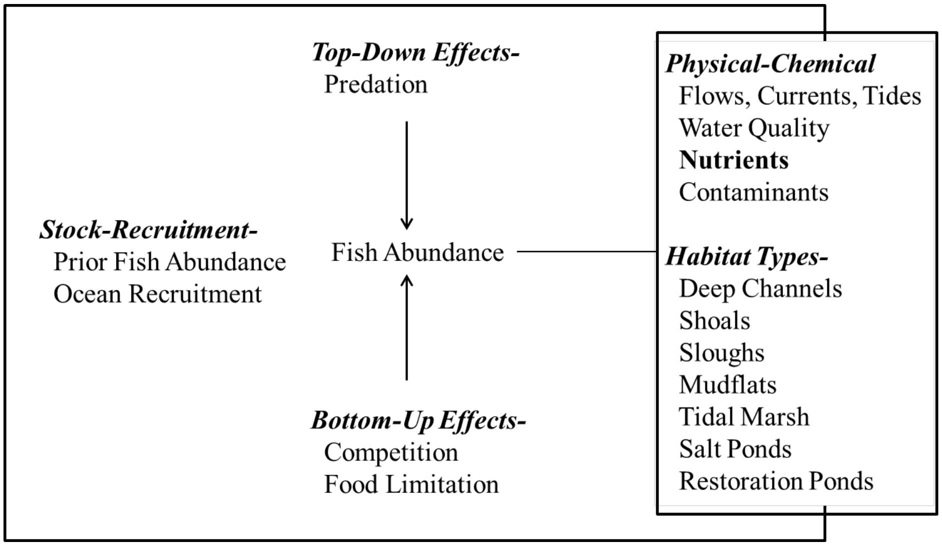

Figure 7.4. Conceptual model of drivers of fish population dynamics.

7.8.1 Physical and Chemical Habitat:

The distribution and abundance of estuarine fishes can be driven by many environmental factors such as water quality, freshwater outflow and contaminants (Jassby et al. 1995; Feyrer et al. 2007). Estuaries are places with dynamic salinity regimes and variable freshwater flows. The Mediterranean climate of the San Francisco Estuary leads to extreme inter-annual variability in the amount and timing of freshwater flows to the estuary as well as large seasonal fluctuations in temperature, which can have significant consequences for the distribution and abundance of estuarine fish (Sommer et al 2007; Baxter et al. 2010). The physical and chemical environment of the water column, along with the physical benthos form the habitat types in LSB and South Bay, and provides the overall framework for the biological drivers of fish populations. In the diagrammatic representation of Figure 7.4, a large box surrounds the model representing the habitat linkages to fish populations. For example nutrient inputs from wastewater can influence the foodweb through production of phytoplankton at the base of the foodweb, as well as through potential toxicity effects in the contaminants module, thus linkages can operate vertically as well and laterally in this conceptual model. Dissolved oxygen falls broadly under ‘water quality’ here, but is treated separately below because of its potential importance as a driver.

7.8.2 Stock-Recruitment:

This sub-model represents the relationship between adult spawning populations and the numbers of young subsequently recruiting to the estuary. A majority of the species found in South Bay are juveniles, highlighting the nursery function this habitat provides. We make the distinction between species that recruit as larvae or juvenile stages from the ocean and those that are born in the estuary. This distinction was made in our fish classification scheme and is important in this module as fish population regulation can be largely driven during the larval phase and this can take place in the ocean rather than the estuary, thus we need to keep this in consideration when evaluating impacts exclusive to San Francisco Estuary as opposed to the ocean. Oceanic decadal and longer term cyclic pattern in productivity and current patterns have been found to have profound effects on nearshore productivity and recruitment of marine fishes (Cloern et al. 2007). These effects would largely be independent from potential nutrient effects in the estuary. Species that migrate into the estuary as adults to spawn or give birth to young also deserve distinction here as adult populations again may be regulated by ocean cycles, although the recruitment of young can be influence by drivers within the estuary. Lastly estuarine resident species would experience potential nutrient effects throughout the life history and would likely exhibit the greatest impact on long-term abundance patterns.

7.8.3 Foodweb Effects:

Fish populations can fluctuate from predation “top-down” and food limitation “bottom-up” foodweb effects. Predation by larger piscivorous fishes, mammals or birds are considered top-down effects on the population by direct removal of individuals to predation events. The “top down” drivers on fish populations have not been adequately examined in South Bay and LSB. Abundance and distribution of predators, (i.e., larger piscivorous fishes) are rarely collected in long-term monitoring projects, thus data is insufficient for determining predation effects on fish populations in general. Avian predation is also likely to be a significant contributor to top down effects in the shallow shoal and ponds. To fill this data gap, we recommend diet studies be conducted on large piscivorous species collected in the monitoring surveys. This can be readily accomplished using non-lethal gastric lavage techniques. The South Bay Salt Pond Restoration Project has conducted limited diet studies using this technique on Leopard Shark collected in South Bay marshes (*Hobbs et al. 2014 draft final report*). Since trawl sampling is ongoing, addition of this component to those studies would add minimal costs. To get a better understanding of predation effects on fish populations, a new monitoring study targeting larger fishes with gill nets or long-lines would be required. Given the limited funding for ongoing monitoring surveys and minimal scope for additional tasks, the current monitoring projects are not well equipped to add this task. The California Department of Fish and Wildlife conducts gill netting for Striped Bass and sturgeon in the north and central bays, and has on occasion sampled in South Bay. Expansion of these efforts into South Bay on a regular basis would be the best approach to acquiring better information on the distribution and abundance of predatory fishes and aid evaluating the overall effect of predation on fish populations in the San Francisco Estuary

Bottom-up drivers, i.e., phytoplankton and zooplankton abundance, can have a significant effect on fish population dynamics. There is evidence that suggests food limitation in North Bay and Delta habitats may be a major factor limiting fish abundance (MacNally et al. 2010; Baxter et al . 2010; Kimmerer et al. 2006; Sommer et al. 2007). As with predation, food limitation has not been studied in South Bay and we do not know the extent to which food is a limiting resource for fish populations in these habitats. Nutrient additions could stimulate zooplankton productions in the estuary, however; food production is limited by other factors, such as clam predation on phytoplankton and zooplankton (Kimmerer 2004; Kimmerer and Thompson 2014), or suspended sediments limiting light penetration (Alpine and Cloern 1992; Cloern 1999; Cloern and Jassby 2012), thus it is unlikely nutrients would have much effect without interactions with other physical factors and foodweb effects. Data on fish diets in South Bay is very limited, and could provide valuable information on the overall structure of the food web as well as linkages to top down effects. Studies on feeding and growth for South Bay fishes would allow us to determine whether food limitation is a key factor limiting fish in South Bay.

7.8.4 Conceptual for Fish Responses to Nutrients

This subsection does not present a conceptual model for nutrient cycling, but rather picks up at the point where nutrient-related effects influence fish. For a complete nutrient conceptual model for SFB, the reader is referred to SFEI (2014, #731).

Fish can respond in multiple ways to nutrients in an estuary. On the one hand, sufficient nutrients are needed to support phytoplankton blooms which in turn can result in a “bottom-up” effect by stimulating secondary production of pelagic zooplankton (e.g. copepods and mysid shrimps), which could ultimately lead to an increase in the abundance of pelagic fishes or a shift in distribution to areas of high primary production (Breitburg 2002; Nixon 2002). Phytoplankton production in the pelagic environment could also lead to increased abundance of benthic filter feeders such as clams and thus short circuit the link to pelagic fishes. In South Bay and LSB, such an increase in filter feeding invertebrates could increase feeding, growth and abundance of benthic species that feed on these organisms such as the Leopard Shark and Bat Ray. In many areas of SFB phytoplankton production is considered to be light-limited, with nutrients already present at sufficient levels to support sizable blooms; therefore, increased nutrient inputs would not necessarily translate to higher fish production. Macroalgal blooms, fostered by sufficient nutrients, can have positive effects on littoral and benthic fishes by providing structural habitat in the shoals and tidal flats, where seagrass beds once dominated. Species such as the sea perches (e.g. Shiner Surfperch, Barred Surfperch) would benefit from the added structure macroalgae may provide. Like the pelagic species, the sea perches have declined in abundance in recent years (Fish et al. 2013).

On the other hand, nutrients have adverse impacts on fish when present at excessive levels. Excessive phytoplankton production leads to high rates of microbial respiration that consumes oxygen, leading to low DO levels.. Large expansive macroalgal blooms could also have negative effects on the benthos by covering bottom habitats where benthic fishes (e.g. English Sole, Pacific Staghorn Sculpin, California Halibut) would forage for invertebrates buried in the sediments. High levels of nutrients have also been associated with the production of harmful algal blooms such as *Microcystis* spp., which exudes a toxic compound which can harm zooplankton and fishes (Lehman et al. 2005). Salinity levels in South Bay are typically too high for *Microcystis* blooms, although some have been observed in wetter years, and other noxious species of algae and algal toxins have been found in salt ponds and in South Bay and LSB (SFEI 2014, #731). Harmful algal blooms can have effects on the foodweb, reducing the abundance of prey for fishes or having more direct effects on fish resulting (Lehman et al. 2010).

7.8.5 Low Dissolved Oxygen Effects on Fish Conceptual Model.

The frequency, magnitude and duration of low-oxygen events can have profound effects on aquatic organisms (Diaz and Breitburg 2009). Organisms can exhibit responses to hypoxia - defined as DO < 2-3 mg/L (CENR 2010) - that can operate at many levels of biological organizations, from effects on molecular and biochemical pathways to individual behavior, to population demography to community dynamics and ecosystem structure and function. The effects of hypoxia at the different levels of organization are structured as state responses to the hypoxic conditions and subsequent outcome from the response, which are measurable (Figure 7.6). Since the biological levels are nested hierarchically, the responses and measureable outcomes can be also considered cause and effect vertically through biological integration.

While the effects below are described as they pertain to hypoxia, many of these adverse impacts can also be exerted at higher DO, with sensitivity being species specific and influenced by other factors (e.g., temperature, food availability). The DO standard for habitats in South Bay and LSB is 5 mg/L, or rolling 3-month median of 80% saturation.[1] The tolerance of South Bay and LSB species to low DO is currently poorly known. The information regarding tolerance that is available exists only for non-native species that occur in other estuaries that experience frequent hypoxic conditions. Ascertaining effects of low DO on the specific fishes and macro-invertebrates that occur in South Bay and LSB will be difficult given our limited understand of

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[1] http://www.waterboards.ca.gov/sanfranciscobay/water_issues/programs/planningtmdls/basinplan/web/docs/BP_all_chapters.pdf

what drives population trends. For that reason, much of the discussion below is based on our general understanding of the effects of low DO from other ecosystems.

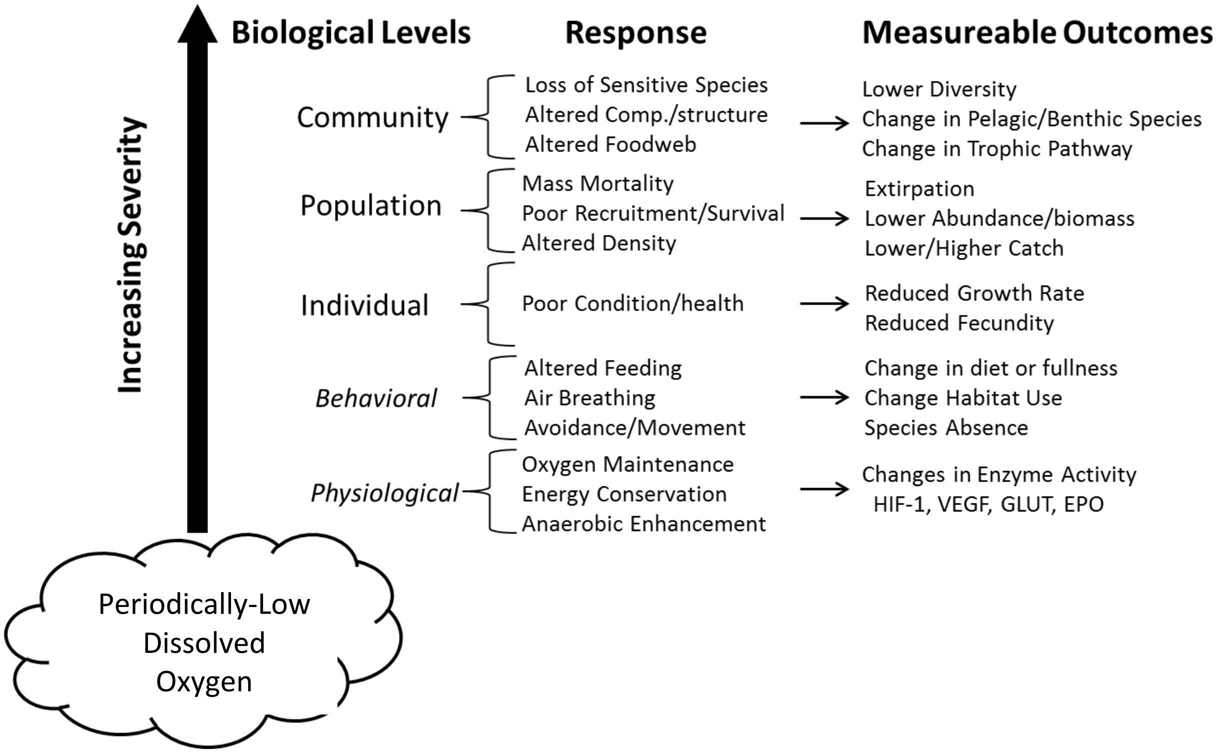


Figure 7.6. Conceptual model of hypoxia across a gradient of increasing severity and associated biological levels effected and their responses linked to measurable outcomes.

7.8.5.1 Physiological

Aquatic organisms exposed to hypoxia exhibit similar physiological compensatory responses as the biochemical pathways are highly genetically conserved (Wu 2002; Richards 2009). Fish physiologically respond to hypoxic stress by attempting to maintain oxygen delivery to tissues (Wu and Woo 1984), conserving energy reserves (Dalla Via et al. 1998) and by enhancing the anaerobic energy pathway for energy supply (Pollock et al. 2007). The cascade of biochemical responses is triggered by the DNA-binding protein known as hypoxia-inducible factor 1 (HIF-1). The enzyme becomes active when the organism is exposed to low dissolved oxygen concentrations, resulting in transcription of a number of hypoxia-inducible genes producing enzymes which facility the organisms ability to tolerate the hypoxic stress. The HIF-1 facilitates proliferation of blood vessels for increasing oxygen supply (VEGF), production of red blood cells (EPO) glucose transporters (GLUT) to enhance glucose transport for energy demands and a variety of glycolytic enzymes for anaerobic energy cycling (see review by Wu 2002).

7.8.5.2 Behavioral

Behavioral responses to low oxygen conditions can be categorized into four principal activities, (1) changes in activity rates, (2) increased use of air breathing, (3) increased use of aquatic surface respirations, and (4) avoidance via vertical or horizontal movement (Kramer 1987; Chapman 2009). Fish first experiencing low oxygen water will attempt to conserve energy, which is most typically accomplished through reduced activity. Swimming speeds are often reduced in hypoxic conditions (Schurmann and Stefensen 1994; Crocker and Cech 1997), movements can be diminished (Dalla Via et al. 1998) and in extreme condition immobility or paralysis has been observed (Fischer et al 1992). However, Domenici et al. (2000) found Atlantic herring increased swimming speeds under conditions of gradually decreasing oxygen concentrations. Concomitant with reduced activity, feeding is often reduced during periods of hypoxia (Pihl 1992; Pichavant et al. 2001). Species that are better adapted to low oxygen waters have been found to opportunistically take advantage of increased prey availability during periods of hypoxia when benthic prey are stressed and come to the sediment surface (Diaz et al. 1992). Hypoxia can also have effects on prey resulting in changes in diet composition of fishes capable of tolerating hypoxia (Pihl 1994). Fish can take advantage of higher oxygen concentrations at the aquatic surface boundary layers were oxygen diffusion with the atmosphere is greatest, or near aerated features in the habitat such as spilling culverts or drainage pipes. The Central Mudminnow can take advantage of hypoxic conditions by engulfing air bubbles (Rahel and Nutzman 1994). Fish can detect and actively avoid hypoxia by migrating or moving away from low oxygen waters (Pihl et al. 1991). This is the most commonly documented behavioral response by fishes to hypoxic conditions. There have been many field studies documenting lower fish abundance and distributional shifts related diel-cyclic hypoxia in estuarine habitats (Eby and Crowder 2004; Bell and Eggleston 2005; Tyler and Targett 2007) and reductions in species richness and diversity (Howell and Simpson 1994; Diaz and Breitburg 2009)

7.8.5.3 Organism

The physiological and behavior responses are conceptually and biologically nested under the organismal level responses which focus the effects of hypoxia on fitness. Fitness can be defined here as individual condition and growth to support survival as well as reproductive condition of fish experiencing hypoxic conditions. The physiological responses of fish to hypoxia can be energetically costly and result in poor somatic condition or reduced growth (Wang et al. 2009). Hypoxia has also been shown to reduce feeding in a variety of fish. Numerous laboratory studies investigating hypoxia induced effects on feeding and growth have ubiquitously shown that low oxygen levels can have direct impacts on feeding rates and subsequent growth rates (Thetmeyer et al. 1999; Pichavant et al. 2001; Zhou et al. 2001; Brandt et al. 2009; Roberts et al. 2011). The common carp, a model organism for hypoxia investigations, reduces feeding rate and exhibits reduced growth along with a suite of biochemical responses when exposed to hypoxic conditions (Mustafa et al. 2011). Hypoxia can also have significant effects on reproductive success, through effects on gamete formation, timing, and sex determination (see review by Wu 2009). In Gulf Killifish, hypoxia has been observed to reduce feeding, growth and the gonadosomatic index and egg production, particularly under condition of food limitation (Landry 2007; Cheek 2011).

7.8.5.4 Population

Hypoxic effects on reduced growth or reproductive fitness can ultimately result in lower population abundance, however; direct evidence of effects in the wild is difficult to measure as many other environmental covariates can obfuscate such patterns and a long-term baseline would be required to disentangle hypoxia effects from other effects on population dynamics. Population declines and reduced fish biomass has been observed in many systems experiencing hypoxic conditions (Breitburg 2002, 2009; Wu 2002; Pollock et al, 2007; Diaz and Breitburg 2009). Mass die-offs of fish from rapid reductions in dissolved oxygen are the most obvious effect due to their visual nature and have been reported in many different ecosystems worldwide (Diaz and Breitburg 2008). Hypoxia can also have sublethal effects that are much less pronounced but can have significant impacts on recruitment or survival. Effects on vital rates such as growth and survival are difficult to identify and often require more advanced modelling exercises. In the Chesapeake Bay Estuary, individual based models have been used to describe the effect of hypoxia on egg and larval survival and subsequent population level effects for Bay Anchovy (Adamack et al. 2007, 2012; Rose et al. 2009). Hypoxia can also result in habitat "squeeze" or a reduction in available habitat (Coutant 2012). When habitat limitation occurs over small spatial or temporal scales, increased density of fish can occur as fish are rapidly confined to small areas (Eby and Crowder 2002).

7.8.4.5 Community

Hypoxia can ultimately result in changes in fish community structure and composition (Pollock et al. 2007; Diaz and Breitburg 2009). The tolerance of individual fish species to dissolved oxygen concentrations has been studied extensively for estuarine species, with most species capable of surviving hypoxia for short period of time (Miller et al. 2002). Prolonged or cyclic hypoxia can lead to sensitive species avoiding an area for long periods of time or permanently resulting in lowered species diversity (Chapman and McKenzie 2009). Benthic species are often more sensitive to low dissolved oxygen concentrations in estuaries and changes in the ratio of benthic to pelagic fishes has been observed (Caddy 2000; Breitburg 2002). This pattern was observed for the fisheries landings in the Chesapeake Bay with landing of pelagic Atlantic Menhaden increasingly dominant through time as eutrophication and hypoxia increased (Kemp et al. 2005). Hypoxia can also effect the food web structure and flow of energy and nutrient through the system. The loss of benthic prey in areas with persistent hypoxia can alter diets and trophic interactions (Pihl et al.1992, 1994). Species avoiding low dissolved oxygen areas are often concentrated along the edges and can be subjected to higher predation rates, transferring energy away from the affected areas (Eby et al. 2005).

7.10 Is Lower South Bay Currently Experiencing Nutrient Related Impacts?

Lower South Bay historically experienced poor water quality during the warm summer months: dissolved oxygen concentrations in Coyote Creek were often near or below 1 mg/L, when raw sewage was discharged into the bay on a daily basis (Figure 7.7; Figure 6.8). Since then, wastewater treatment methods have improved considerably resulting in substantially lower oxygen demand and, in general, higher DO concentrations in Coyote Creek (Yigzaw 2014; Figure 7.7). Very little historical data inventorying the abundance and distribution of aquatic organisms in Lower South Bay exists, thus we cannot directly examine many of the outcomes predicted from our conceptual models based on comparisons of fish assemblages pre- and post-improvements. The San Francisco Estuary has experienced declines in most fish and macro-invertebrate species since the inception of long-term monitoring studies (Baxter et al. 1999). Changes in fish abundance over time have been ascribed to the long-term reduction in freshwater flows to the estuary.   Utilizing available time series data from monitoring studies for a single mechanism cause/effect relationship may obfuscate in fish abundance trends. Thus to evaluate the effects of nutrients in South Bay and LSB a multi-parameter approach is required. Moreover, linking model variables to conceptual models, such as those presented above, will provide for a parsimonious examination of existing monitoring data.

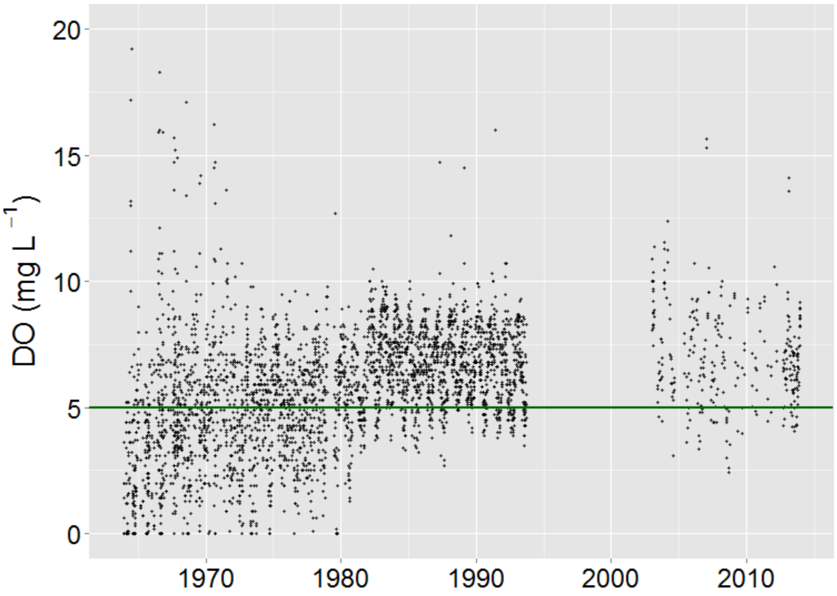


Figure 7.7 DO concentrations for the five stations in the lower stretch of Coyote Creek, for the years during which data were collected (with a data gap during the period from the early 1990s to the early 2000s). The green horizontal line represents the 5 mg/L water quality objective for DO for the San Francisco Basin. (Reproduced from Yigzaw 2014)

Dissolved oxygen conditions in LSB and South Bay are described in Section 6. Ship-based monitoring in open Bay areas of LSB and South Bay has observed that, in general, DO was >5 mg/L (or 80% saturation) throughout the year (Figure 6.3). Recent high frequency data collected at the Dumbarton Bridge, however, suggest that, at low tide, summer and fall DO concentrations further south in LSB may approach or dip below 5 mg/L. Limited data suggests that some sloughs may experiences frequent hypoxic events: for example, during summer months, DO concentrations in Alviso Slough dipped below 2 mg/L for several hours each day during some years (Section 6, Figure 6.13). In addition, the SBSPRP conducted deployments of a water quality sonde in the breach at pond A21 in Coyote Creek (Figure 7.9). Over a five day period in June 2013, dissolved oxygen concentrations inside the restoration pond consistently dropped below 2 mg/L in the early morning hours following a nocturnal high tide. DO concentrations in Alviso Slough appear to be strongly regulated by tidal action while DO in salt ponds may be more subject to diurnal variations in DO production

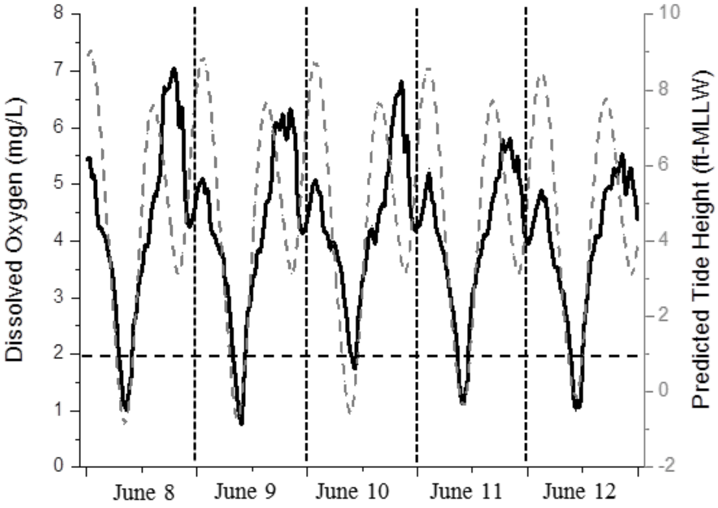


Figure 7.8. Continuous dissolved oxygen concentration (mg/L) measured every 20-min over a 5 day period in June 2013 in the breach of pond A21 in Coyote Creek. The grey dashed line is 15-min predicted tide height data from the Coyote Creek NOAA station. Horizontal dashed line shows the 2 mg/L hypoxia threshold.

Very little information regarding dissolved oxygen tolerance levels exists for fish and macro-invertebrates found in South San Francisco Bay. Monitoring for fish abundance has not been conducted in South Bay and LSB specifically for the purpose of assessing potential effects of nutrients or low DO. However, the South Bay Salt Pond Restoration Project has been quantifying the abundance of fishes and macro-invertebrates in the Alviso Marsh on a monthly basis since the summer of 2010. As an initial attempt to characterize dissolved oxygen tolerance, we used catch data and associated dissolved oxygen concentration measured at the time of sample collection to calculate a catch-weighted mean and standard deviation using frequency of occurrence from presence-absence data as a weighting factor for 23 species of fish and 8 species of invertebrates (Figure 7.10). This provides an environmentally relevant dissolved oxygen concentration occurrence metric for species frequently found in the Alviso Marsh. The mean dissolved oxygen concentrations varied by species and assemblage, with the Shokahazi Goby having the lowest value 4.3 mg/L and Longfin Smelt the highest 7.8 mg/L. All, but the Shimofuri Goby and Longjaw Mudsucker had mean values greater than 5 mg/L.

Assemblages common in the winter and early spring months tended to havehigher values, e.g. English Sole, Threadfin Shad ,Longfin Smelt, American Shad and Pacific Herring, , Speckled Sanddab.

Several species had relatively high standard deviations, reflecting their being found at a wide range of dissolved oxygen concentrations, (e.g. Arrow Goby, Three-spine Stickleback, crangon shrimp, Pacific Staghorn Sculpin, Yellowfin Goby, Asian clams, Paleomon shrimp and Northern Anchovy). These species are the most abundant taxa found in the summer months when dissolved oxygen concentrations are low. While these data may not represent a true physiological tolerance, they likely represent behaviorally relevant values. For most species the minimum dissolved oxygen concentrations were at least one individual was captured was significantly lower than the mean, with most species being collected at the lowest dissolved oxygen concentration recorded 2.2 mg/L. Thus species can be found below their mean value, and may be experiencing physiological effects of low dissolved oxygen exposure.

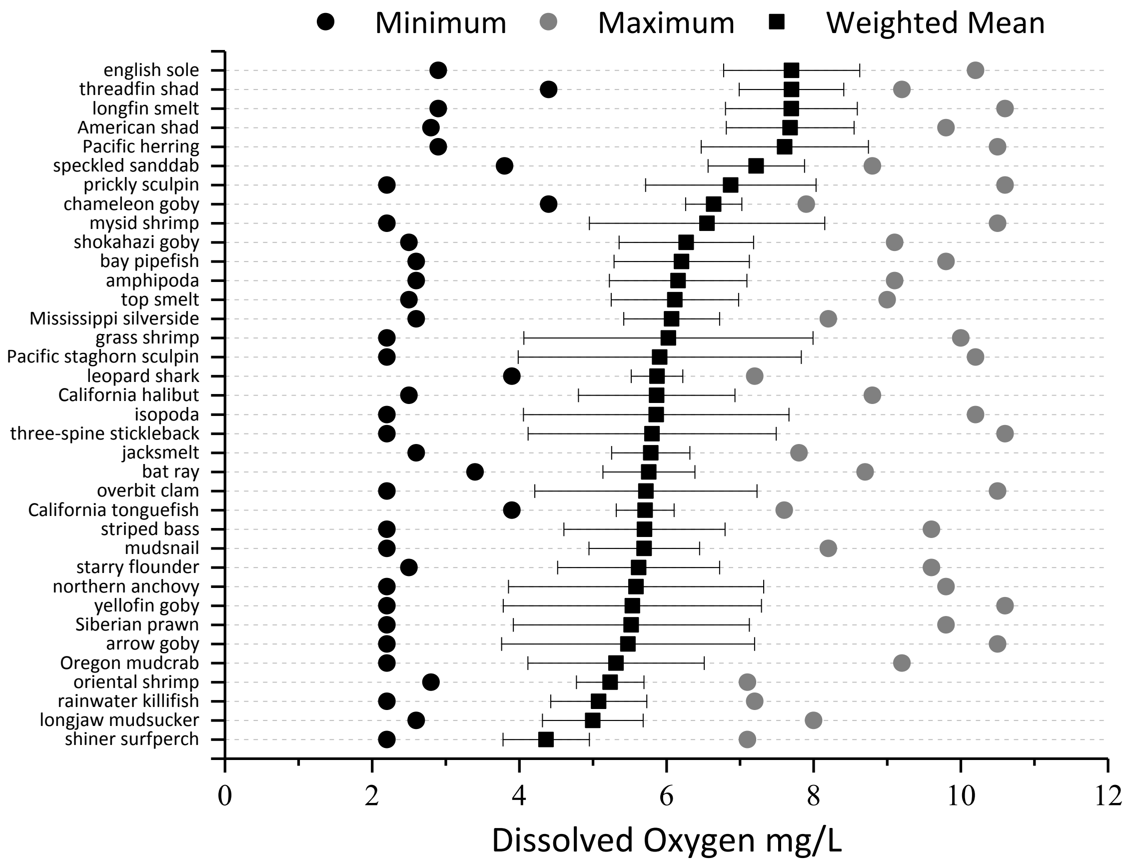


Figure 7.10. Mean dissolved oxygen concentration weighted by the frequency of occurrence for the most common fish and macro-invertebrate species found in South Bay marsh habitats.

Figure 7.11 uses the same underlying data as in Figure 7.10, but illustrates the distribution of trawls vs. DO concentration, and the catch per unit effort for each species. This presentation of the data offers some additional perspective on when species were and were not caught, and helps identify some data gaps. For example, while the weighted mean DO concentration for northern anchovy was ~5 mg/L (Figure 7.10), they were commonly caught at high abundance at DO <3-4 mg/L. Leopard shark, on the other hand, had a similar weighted mean DO as northern anchovy (~5.2 mg/L), but were evenly distributed between 4 and 7 mg/L but only rarely captured (n=10). Several species appear to have rather sharp thresholds, e.g., longfin smelt, threadfin shad, and English sole. However, the absence of longfin smelt and threadfin shad at lower DO may have as much or more to do with temperature tolerance (and co-occurrence of higher T and lower DO) than DO tolerance.

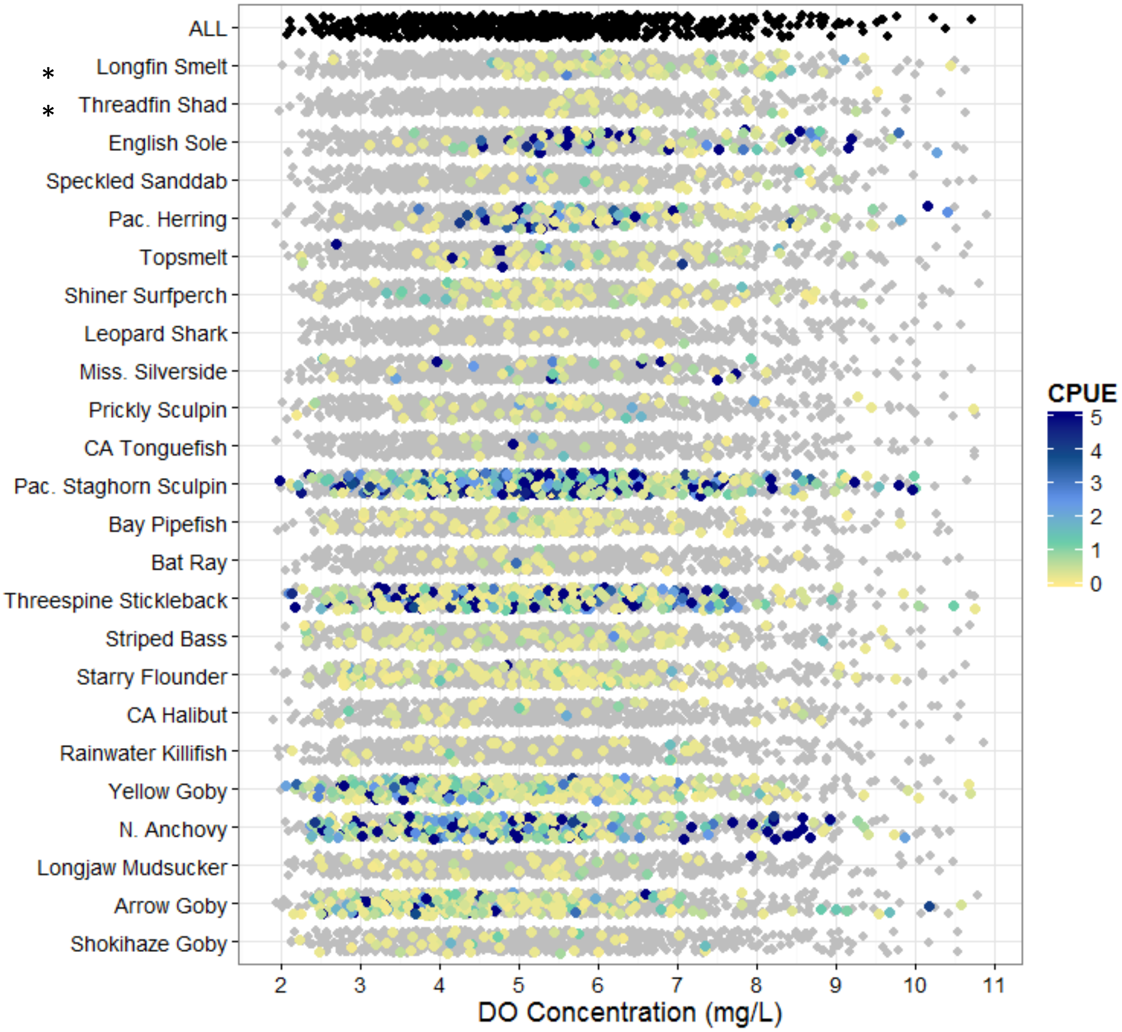


Figure 7.11 All trawls and catch per unit effort each trawl vs. DO concentration for each species. Black symbols in the first row display the distribution of trawls relative to DO. Trawls are repeated for each species (rows), with a grey symbol indicating that species was not caught; other colors indicate the species was caught, with the color denoting the catch per unit effort. * Indicates species whose absence may also be strongly driven by other factors such as seasonally-varying temperatures.

7.11 Recommendations

In assessing the state of the science with regards to fish populations in Lower South Bay, we have identified the following major knowledge gaps:

1. Are biota being adversely impacted by low(er) DO in the margins or open Bay?
2. What are the DO preferences or tolerances of key fish species that are observed, or expected, in Lower South Bay?
3. How do fish populations and diversity respond to spatial and temporal variability in DO?
4. Do current conditions support or adversely impact populations benthos abundance or assemblage?

We propose a number of high-priority activities to address these knowledge gaps. Since some of these investigations would be resource- and time-intensive undertakings, some level of prioritization is still needed:

- Further analyze existing fish data to better characterize spatial and temporal variability in fish populations, and drivers of that variability
- Identify appropriate protective DO conditions for fish and other biota
    - Literature review to determine:
        - What are we trying to protect?
        - What conditions would be protective?
        - What is the uncertainty in our understanding for species of interest?
    - Compare existing conditions with protective conditions
- Conduct fish and benthos surveys, in conjunction with habitat surveys (DO, T, food abundance and quality, etc.) to assess condition

Table 7.1. Summary of fish species rank abundance and life history types from the top 70 species collected by the San Francisco Bay Study 1980-2012, and the proportion of total catch for each species in deep channel versus shallow shoal stations.

Family	CommonName	ScientificName	Native Status	Assemblage Classification	South Bay Status
Acipenseridae	green sturgeon	Acipenser medirostris	N	D	rare
-	white sturgeon	Acipenser transmontanus	N	D	uncommon
Atherinidae	Mississippi Silverside	Menidia Berrylina	I	F/E	abundant
Atherinopsidae	California grunion	Leuresthes tenuis	N	M	common
-	jacksmelt	Atherinopsis californiensis	N	E	abundant
-	topsmelt	Atherinops affinis	N	E	abundant
Batrachoididae	plainfin midshipman	Porichthys notatus	N	E	abundant
Carangidae	jack mackerel	Trachurus symmetricus	N	M	rare
Catostomidae	Sacramento Sucker	Catostomus occidentalis	N	F	rare
Centrarchidae	Largemouth Bass	Micropterus salmoides	I	F	rare
Clupeidae	American shad	Alosa sapidissima	I	D	common
-	Pacific herring	Clupea pallasii	N	ME	abundant
-	Pacific sardine	Sardinops sagax	N	M	abundant
-	threadfin shad	Dorosoma petenense	I	F	common
Cottidae	bonyhead sculpin	Artedius notospilotus	N	M	uncommon
-	Pacific staghorn sculpin	Leptocottus armatus	N	E	abundant
-	prickly sculpin	Cottus asper	N	E	rare
Cynoglossidae	California tonguefish	Symphurus atricaudus	N	MO	common
Cyprinidae	Common Carp	Cyprinus carpio	I	F	rare
Embiotocidae	barred seaperch	Amphistichus argenteus	N	E	common
-	black seaperch	Embiotoca jacksoni	N	E	uncommon
-	calico seaperch	Amphistichus koelzi	N	E	rare
-	dwarf seaperch	Micrometrus minimus	N	E	common
-	pile seaperch	Rhacochilus vacca	N	E	uncommon
-	redtail seaperch	Amphistichus rhodoterus	N	E	rare
-	rubberlip seaperch	Rhacochilus toxotes	N	E	rare
-	shiner seaperch	Cymatogaster aggregata	N	E	abundant
-	silver seaperch	Hyperprosopon ellipticum	N	E	rare
-	striped seaperch	Embiotoca lateralis	N	E	rare
-	walleye seaperch	Hyperprosopon argenteum	N	E	common
-	white seaperch	Phanerodon furcatus	N	E	uncommon
Engraulidae	northern anchovy	Engraulis mordax	N	E/M	abundant
Fundulidae	rainwater killifish	Lucania parva	I	E	rare
Gadidae	Pacific tomcod	Microgadus proximus	N	M	common
Gasterosteidae	threespine stickleback	Gasterosteus aculeatus	N	E	uncommon
Gobiidae	arrow goby	Clevelandia ios	N	E	common
-	bay goby	Lepidogobius lepidus	N	E	abundant
-	chameleon goby	Tridentiger trigonocephalus	I	E	abundant
-	cheekspot goby	Ilypnus gilberti	N	E	abundant
-	longjaw mudsucker	Gillichthys mirabilis	N	E	rare
-	shimofuri goby	Tridentiger bifasciatus	I	E	rare
-	shokihaze goby	Tridentiger barbatus	I	E	common
-	yellowfin goby	Acanthogobius flavimanus	I	E	common
Hexagrammidae	lingcod	Ophiodon elongatus	N	M	uncommon
Hexanchidae	broadnose sevengill shark	Notorynchus cepedianus	N	M	rare
Kyphosidae	halfmoon	Medialuna californiensis	N	M	rare
Liparidae	showy snailfish	Liparis pulchellus	N	M	uncommon
Moronidae	striped bass	Morone saxatilis	I	D	common

Myliobatidae	bat ray	Myliobatis californica	N	ME	common
Ophichthidae	yellow snake eel	Ophichthus zophochir	N	M	rare
Ophidiidae	spotted cusk-eel	Chilara taylori	N	M	rare
Osmeridae	eulachon	Thaleichthys pacificus	N	M	rare
-	longfin smelt	Spirinchus thaleichthys	N	D	abundant
-	night smelt	Spirinchus starksi	N	M	uncommon
-	surf smelt	Hypomesus pretiosus	N	M	uncommon
-	whitebait smelt	Allosmerus elongatus	N	M	uncommon
Paralichthyidae	California halibut	Paralichthys californicus	N	E/MO	common
-	Pacific sanddab	Citharichthys sordidus	N	M	uncommon
-	speckled sanddab	Citharichthys stigmaeus	N	MO	abundant
Petromyzontidae	Pacific lamprey	Lampetra tridentata	N	D	uncommon
-	river lamprey	Lampetra ayresii	N	F	rare
Pholidae	saddleback gunnel	Pholis ornata	N	M	uncommon
Platyrhynidae	thornback	Platyrhinoidis triseriata	N	M	rare
Pleuronectidae	C-O sole	Pleuronichthys coenosus	N	M	rare
-	curlfin sole	Pleuronichthys decurrens	N	M	rare
-	diamond turbot	Pleuronichthys guttulatus	N	MO	common
-	English sole	Parophrys vetulus	N	MO	abundant
-	hornyhead turbot	Pleuronichthys verticalis	N	M	rare
-	sand sole	Psettichthys melanostictus	N	M	uncommon
-	starry flounder	Platichthys stellatus	N	ME	common
Rajidae	big skate	Raja binoculata	N	M	uncommon
Rhinobatidae	shovelnose guitarfish	Rhinobatos productus	N	M	uncommon
Salmonidae	Chinook salmon	Oncorhynchus tshawytscha	N	D	uncommon
-	Steelhead Trout	Oncorhynchus mykiss	N	D	rare
Sciaenidae	queenfish	Seriphus politus	N	M	rare
-	white croaker	Genyonemus lineatus	N	E	abundant
-	white seabass	Atractoscion nobilis	N	M	rare
Scorpaenidae	black rockfish	Sebastes melanops	N	MO	rare
-	brown rockfish	Sebastes auriculatus	N	MO	common
-	vermilion rockfish	Sebastes miniatus	N	MO	rare
-	yellowtail rockfish	Sebastes flavidus	N	M	rare
Squalidae	spiny dogfish	Squalus acanthias	N	M	uncommon
Stromateidae	Pacific pompano	Peprilus simillimus	N	M	common
Syngnathidae	bay pipefish	Syngnathus leptorhynchus	N	E	common
Synodontidae	California lizardfish	Synodus lucioceps	N	M	uncommon
Torpedinidae	Pacific electric ray	Torpedo californica	N	M	rare
Triakidae	brown smoothhound	Mustelus henlei	N	M	common
-	leopard shark	Triakis semifasciata	N	ME	common

Table 7.2a. Fish species encountered during San Franicsco Bay Study monitoring surveys conducted in South and Lower San Francisco Bay between 1980 and 2012. B=benthic, P=pelagic, L=litorral, N=native, I=invasive

San Francisco Bay Study

South Bay & Lower South Bay 1980-2012

CommonName	Family	ScientificName	P, B or L	N or I	Rank	Status	%Channel	%Shoal
American shad	Clupeidae	*Alosa sapidissima*	P	I	25	uncommon	█	█
arrow goby	Gobiidae	*Clevelandia ios*	B	N	35	uncommon	█	█
barred surfperch	Embiotocidae	*Amphistichus argenteus*	L	N	28	uncommon	█	█
bat ray	Myliobatidae	*Myliobatis californica*	B	N	17	common	█	█
bay goby	Gobiidae	*Lepidogobius lepidus*	B	N	6	common	█	█
bay pipefish	Syngnathidae	*Syngnathus leptorhynchus*	L	N	23	uncommon	█	█
big skate	Rajidae	*Raja binoculata*	B	N	46	rare	█	█
black perch	Embiotocidae	*Embiotoca jacksoni*	L	N	50	rare	█	█
black rockfish	Scorpaenidae	*Sebastes melanops*	B	N	75	rare		█
bonyhead sculpin	Cottidae	*Artedius notospilotus*	B	N	42	rare	█	█
broadnose sevengill shark	Hexanchidae	*Notorynchus cepedianus*	B	N	67	rare	█	█
brown rockfish	Scorpaenidae	*Sebastes auriculatus*	B	N	27	uncommon	█	█
brown smoothhound	Triakidae	*Mustelus henlei*	B	N	21	uncommon	█	█
buffalo sculpin	Cottidae	*Enophyrs bison*	B	N	94	rare		
cabazon	Cottidae	*Scorpaenichthys marmoratus*	B	N	95	rare		
calico surfperch	Embiotocidae	*Amphistichus koelzi*	L	N	68	rare		█
California grunion	Atherinopsidae	*Leuresthes tenuis*	L	N	32	uncommon	█	█
California halibut	Paralichthyidae	*Paralichthys californicus*	B	N	20	uncommon	█	█
California lizardfish	Synodontidae	*Synodus lucioceps*	B	N	52	rare	█	█
California tonguefish	Cynoglossidae	*Symphurus atricaudus*	B	N	16	common	█	█
chameleon goby	Gobiidae	*Tridentiger trigonocephalus*	B	I	11	common	█	█
cheekspot goby	Gobiidae	*Ilypnus gilberti*	B	N	3	abundant	█	█
Chinook salmon	Salmonidae	*Oncorhynchus tshawytscha*	L	N	40	rare	█	█
C-O sole	Pleuronectidae	*Pleuronichthys coenosus*	B	N	76	rare		█
copper rockfish	Sebastidae	*Sebastes caurinus*	B	N				
curlfin sole	Pleuronectidae	*Pleuronichthys decurrens*	B	N	61	rare	█	█
diamond turbot	Pleuronectidae	*Pleuronichthys guttulatus*	B	N	29	uncommon	█	█
dwarf perch	Embiotocidae	*Micrometrus minimus*	L	N	33	uncommon	█	█
English sole	Pleuronectidae	*Parophrys vetulus*	B	N	7	common	█	█
eulachon	Osmeridae	*Thaleichthys pacificus*	P	N	69	rare	█	
grass rockfish	Sebastidae	*Sebastes rastrelliger*	B	N				
green sturgeon	Acipenseridae	*Acipenser medirostris*	B	N	70	rare		█
halfmoon	Kyphosidae	*Medialuna californiensis*	P	N	77	rare		█
hornyhead turbot	Pleuronectidae	*Pleuronichthys verticalis*	B	N	71	rare	█	
jack mackerel	Carangidae	*Trachurus symmetricus*	P	N	78	rare		█
jacksmelt	Atherinopsidae	*Atherinopsis californiensis*	P	N	8	common	█	█
kelp greenling	Hexagrammidae	*Hexagrammos decagrammus*	B	N				
leopard shark	Triakidae	*Triakis semifasciata*	B	N	22	uncommon	█	█
lingcod	Hexagrammidae	*Ophiodon elongatus*	B	N	49	rare	█	█
longfin smelt	Osmeridae	*Spirinchus thaleichthys*	P	N	14	common	█	█
longjaw mudsucker	Gobiidae	*Gillichthys mirabilis*	B	N	79	rare	█	
Mississippi/Inland silverside	Atherinopsidae	*Menidia berrylina*	L		111	rare		
night smelt	Osmeridae	*Spirinchus starksi*	L	N	39	rare	█	█
northern anchovy	Engraulidae	*Engraulis mordax*	P	N	1	abundant	█	█
Pacific electric ray	Torpedinidae	*Torpedo californica*	B	N	58	rare	█	
Pacific herring	Clupeidae	*Clupea pallasii*	P	N	2	abundant	█	█
Pacific lamprey	Petromyzontidae	*Lampetra tridentata*	B	N	53	rare	█	█
Pacific pompano	Stromateidae	*Peprilus simillimus*	P	N	36	uncommon	█	█
Pacific sanddab	Paralichthyidae	*Citharichthys sordidus*	B	N	54	rare	█	█
Pacific sardine	Clupeidae	*Sardinops sagax*	P	N	15	common	█	█
Pacific staghorn sculpin	Cottidae	*Leptocottus armatus*	B	N	13	common	█	█
Pacific tomcod	Gadidae	*Microgadus proximus*	P	N	34	uncommon	█	█
penpoint gunnel	Pholidae	*Apodichthys flavidus*	B	N				
pile perch	Embiotocidae	*Rhacochilus vacca*	L	N	37	rare	█	█
plainfin midshipman	Batrachoididae	*Porichthys notatus*	B	N	12	common	█	█
prickly sculpin	Cottidae	*Cottus asper*	B	N	80	rare	█	

Table 7.2a (continued)

San Francisco Bay Study

South Bay & Lower South Bay 1980-2012

CommonName	Family	ScientificName	P, B or L	N or I	Rank	Status
queenfish	Sciaenidae	*Seriphus politus*	L	N	62	rare
rainbow surfperch	Embiotocidae	*Hypsurus caryi*	B	N	120	rare
rainwater killifish	Fundulidae	*Lucania parva*	L	I	57	rare
red irish lord	Cottidae	*Hemilepidotus hemilepidotus*	B	N	122	rare
redtail surfperch	Embiotocidae	*Amphistichus rhodoterus*	L	N	72	rare
rex sole	Pleuronectidae	*Glyptocephalus zachirus*	B	N		
river lamprey	Petromyzontidae	*Lampetra ayresii*	B	N	73	rare
rubberlip seaperch	Embiotocidae	*Rhacochilus toxotes*	L	N	64	rare
saddleback gunnel	Pholidae	*Pholis ornata*	B	N	43	rare
sand sole	Pleuronectidae	*Psettichthys melanostictus*	B	N	48	rare
shimofuri goby	Gobiidae	*Tridentiger bifasciatus*	B	I	59	rare
shiner perch	Embiotocidae	*Cymatogaster aggregata*	L	N	4	abundant
shokihaze goby	Gobiidae	*Tridentiger barbatus*	B	I	30	uncommon
shovelnose guitarfish	Rhinobatidae	*Rhinobatos productus*	B	N	55	rare
showy snailfish	Liparidae	*Liparis pulchellus*	B	N	44	rare
silver surfperch	Embiotocidae	*Hyperprosopon ellipticum*	L	N	65	rare
speckled sanddab	Paralichthyidae	*Citharichthys stigmaeus*	B	N	10	common
spiny dogfish	Squalidae	*Squalus acanthias*	B	N	51	rare
spotted cusk-eel	Ophidiidae	*Chilara taylori*	B	N	74	rare
starry flounder	Pleuronectidae	*Platichthys stellatus*	B	N	24	uncommon
striped bass	Moronidae	*Morone saxatilis*	P	I	26	uncommon
striped seaperch	Embiotocidae	*Embiotoca lateralis*	L	N	81	rare
surf smelt	Osmeridae	*Hypomesus pretiosus*	L	N	47	rare
thornback	Platyrhynidae	*Platyrhinoidis triseriata*	B	N	60	rare
threadfin shad	Clupeidae	*Dorosoma petenense*	P	I	31	uncommon
threespine stickleback	Gasterosteidae	*Gasterosteus aculeatus*	L	N	45	rare
topsmelt	Atherinopsidae	*Atherinops affinis*	L	N	9	common
vermilion rockfish	Scorpaenidae	*Sebastes miniatus*	B	N	82	rare
walleye surfperch	Embiotocidae	*Hyperprosopon argenteum*	L	N	19	common
white croaker	Sciaenidae	*Genyonemus lineatus*	B	N	5	abundant
white seabass	Sciaenidae	*Atractoscion nobilis*	P	N	83	rare
white seaperch	Embiotocidae	*Phanerodon furcatus*	L	N	38	rare
white sturgeon	Acipenseridae	*Acipenser transmontanus*	B	N	56	rare
whitebait smelt	Osmeridae	*Allosmerus elongatus*	L	N	41	rare
yellow snake eel	Ophichthidae	*Ophichthus zophochir*	B	N	84	rare
yellowfin goby	Gobiidae	*Acanthogobius flavimanus*	B	I	18	common
yellowtail rockfish	Scorpaenidae	*Sebastes flavidus*	B	N	66	rare

Table 7.2b Fish species encountered during Marine Science Institute monitoring surveys conducted in South and Lower San Francisco Bay between 1990 and 2013. B=benthic, P=pelagic, L=litorral, N=native, I=invasive

					Marine Science Institute	
					South Bay 1990-2013	
CommonName	**Family**	**ScientificName**	**P, B or L**	**N or I**	Rank	Status
American shad	Clupeidae	*Alosa sapidissima*	P	I	31	uncommon
arrow goby	Gobiidae	*Clevelandia ios*	B	N	56	uncommon
barred surfperch	Embiotocidae	*Amphistichus argenteus*	L	N	18	uncommon
bat ray	Myliobatidae	*Myliobatis californica*	B	N	16	uncommon
bay goby	Gobiidae	*Lepidogobius lepidus*	B	N	15	uncommon
bay pipefish	Syngnathidae	*Syngnathus leptorhynchus*	L	N	34	uncommon
big skate	Rajidae	*Raja binoculata*	B	N	62	uncommon
black perch	Embiotocidae	*Embiotoca jacksoni*	L	N	40	uncommon
black rockfish	Scorpaenidae	*Sebastes melanops*	B	N	-	-
bonyhead sculpin	Cottidae	*Artedius notospilotus*	B	N	36	uncommon
broadnose sevengill shark	Hexanchidae	*Notorynchus cepedianus*	B	N	70	rare
brown rockfish	Scorpaenidae	*Sebastes auriculatus*	B	N	46	uncommon
brown smoothhound	Triakidae	*Mustelus henlei*	B	N	23	uncommon
buffalo sculpin	Cottidae	*Enophyrs bison*	B	N	103	rare
cabazon	Cottidae	*Scorpaenichthys marmoratus*	B	N	59	uncommon
calico surfperch	Embiotocidae	*Amphistichus koelzi*	L	N	94	rare
California grunion	Atherinopsidae	*Leuresthes tenuis*	L	N	106	rare
California halibut	Paralichthyidae	*Paralichthys californicus*	B	N	7	common
California lizardfish	Synodontidae	*Synodus lucioceps*	B	N	-	-
California tonguefish	Cynoglossidae	*Symphurus atricaudus*	B	N	25	uncommon
chameleon goby	Gobiidae	*Tridentiger trigonocephalus*	B	I	11	common
cheekspot goby	Gobiidae	*Ilypnus gilberti*	B	N	19	uncommon
Chinook salmon	Salmonidae	*Oncorhynchus tshawytscha*	L	N	109	rare
C-O sole	Pleuronectidae	*Pleuronichthys coenosus*	B	N	83	rare
copper rockfish	Sebastidae	*Sebastes caurinus*	B	N	96	rare
curlfin sole	Pleuronectidae	*Pleuronichthys decurrens*	B	N	104	rare
diamond turbot	Pleuronectidae	*Pleuronichthys guttulatus*	B	N	27	uncommon
dwarf perch	Embiotocidae	*Micrometrus minimus*	L	N	12	uncommon
English sole	Pleuronectidae	*Parophrys vetulus*	B	N	3	abundant
eulachon	Osmeridae	*Thaleichthys pacificus*	P	N	-	-
grass rockfish	Sebastidae	*Sebastes rastrelliger*	B	N	84	rare
green sturgeon	Acipenseridae	*Acipenser medirostris*	B	N	89	rare
halfmoon	Kyphosidae	*Medialuna californiensis*	P	N	108	rare
hornyhead turbot	Pleuronectidae	*Pleuronichthys verticalis*	B	N	107	rare
jack mackerel	Carangidae	*Trachurus symmetricus*	P	N	-	-
jacksmelt	Atherinopsidae	*Atherinopsis californiensis*	P	N	48	uncommon
kelp greenling	Hexagrammidae	*Hexagrammos decagrammus*	B	N	60	uncommon
leopard shark	Triakidae	*Triakis semifasciata*	B	N	22	uncommon
lingcod	Hexagrammidae	*Ophiodon elongatus*	B	N	47	uncommon
longfin smelt	Osmeridae	*Spirinchus thaleichthys*	P	N	43	uncommon
longjaw mudsucker	Gobiidae	*Gillichthys mirabilis*	B	N	52	uncommon
Mississippi/Inland silverside	Atherinopsidae	*Menidia berrylina*	L		-	-
night smelt	Osmeridae	*Spirinchus starksi*	L	N	41	uncommon
northern anchovy	Engraulidae	*Engraulis mordax*	P	N	1	abundant
Pacific electric ray	Torpedinidae	*Torpedo californica*	B	N	97	rare
Pacific herring	Clupeidae	*Clupea pallasii*	P	N	5	common
Pacific lamprey	Petromyzontidae	*Lampetra tridentata*	B	N	95	rare
Pacific pompano	Stromateidae	*Peprilus simillimus*	P	N	114	rare
Pacific sanddab	Paralichthyidae	*Citharichthys sordidus*	B	N	17	uncommon
Pacific sardine	Clupeidae	*Sardinops sagax*	P	N	26	uncommon
Pacific staghorn sculpin	Cottidae	*Leptocottus armatus*	B	N	4	common
Pacific tomcod	Gadidae	*Microgadus proximus*	P	N	39	uncommon
penpoint gunnel	Pholidae	*Apodichthys flavidus*	B	N	72	rare
pile perch	Embiotocidae	*Rhacochilus vacca*	L	N	21	uncommon
plainfin midshipman	Batrachoididae	*Porichthys notatus*	B	N	14	uncommon
prickly sculpin	Cottidae	*Cottus asper*	B	N	-	-

Table 7.2b (continued)

					Marine Science Institute	
						South Bay 1990-2013
CommonName	**Family**	**ScientificName**	**P, B or L**	**N or I**	Rank	Status
queenfish	Sciaenidae	*Seriphus politus*	L	N	-	-
rainbow surfperch	Embiotocidae	*Hypsurus caryi*	B	N	42	uncommon
rainwater killifish	Fundulidae	*Lucania parva*	L	I	-	-
red irish lord	Cottidae	*Hemilepidotus hemilepidotus*	B	N	69	rare
redtail surfperch	Embiotocidae	*Amphistichus rhodoterus*	L	N	67	uncommon
rex sole	Pleuronectidae	*Glyptocephalus zachirus*	B	N	75	rare
river lamprey	Petromyzontidae	*Lampetra ayresii*	B	N	68	uncommon
rubberlip seaperch	Embiotocidae	*Rhacochilus toxotes*	L	N	-	-
saddleback gunnel	Pholidae	*Pholis ornata*	B	N	79	rare
sand sole	Pleuronectidae	*Psettichthys melanostictus*	B	N	6	common
shimofuri goby	Gobiidae	*Tridentiger bifasciatus*	B	I	29	uncommon
shiner perch	Embiotocidae	*Cymatogaster aggregata*	L	N	2	abundant
shokihaze goby	Gobiidae	*Tridentiger barbatus*	B	I	117	rare
shovelnose guitarfish	Rhinobatidae	*Rhinobatos productus*	B	N	90	rare
showy snailfish	Liparidae	*Liparis pulchellus*	B	N	73	rare
silver surfperch	Embiotocidae	*Hyperprosopon ellipticum*	L	N	66	uncommon
speckled sanddab	Paralichthyidae	*Citharichthys stigmaeus*	B	N	10	common
spiny dogfish	Squalidae	*Squalus acanthias*	B	N	51	uncommon
spotted cusk-eel	Ophidiidae	*Chilara taylori*	B	N	-	-
starry flounder	Pleuronectidae	*Platichthys stellatus*	B	N	20	uncommon
striped bass	Moronidae	*Morone saxatilis*	P	I	44	uncommon
striped seaperch	Embiotocidae	*Embiotoca lateralis*	L	N	-	-
surf smelt	Osmeridae	*Hypomesus pretiosus*	L	N	50	uncommon
thornback	Platyrhynidae	*Platyrhinoidis triseriata*	B	N	91	rare
threadfin shad	Clupeidae	*Dorosoma petenense*	P	I	38	uncommon
threespine stickleback	Gasterosteidae	*Gasterosteus aculeatus*	L	N	80	rare
topsmelt	Atherinopsidae	*Atherinops affinis*	L	N	30	uncommon
vermilion rockfish	Scorpaenidae	*Sebastes miniatus*	B	N	101	rare
walleye surfperch	Embiotocidae	*Hyperprosopon argenteum*	L	N	33	uncommon
white croaker	Sciaenidae	*Genyonemus lineatus*	B	N	8	common
white seabass	Sciaenidae	*Atractoscion nobilis*	P	N	86	rare
white seaperch	Embiotocidae	*Phanerodon furcatus*	L	N	24	uncommon
white sturgeon	Acipenseridae	*Acipenser transmontanus*	B	N	57	uncommon
whitebait smelt	Osmeridae	*Allosmerus elongatus*	L	N	92	rare
yellow snake eel	Ophichthidae	*Ophichthus zophochir*	B	N	-	-
yellowfin goby	Gobiidae	*Acanthogobius flavimanus*	B	I	13	uncommon
yellowtail rockfish	Scorpaenidae	*Sebastes flavidus*	B	N		

Table 7.2c Fish species encountered during South Bay Discharge Authority surveys conducted in South and Lower San Francisco Bay between 1982 and 1986. B=benthic, P=pelagic, L=litoral, N=native, I=invasive

					South Bay Discharge Authority			
					Lower South Bay and Coyote Creek 1982-1986			
CommonName	**Family**	**ScientificName**	**P, B or L**	**N or I**	Rank	Status	%Shoal	%Slough
American shad	Clupeidae	*Alosa sapidissima*	P	I	23	uncommon	█	█
arrow goby	Gobiidae	*Clevelandia ios*	B	N	-	-	-	-
barred surfperch	Embiotocidae	*Amphistichus argenteus*	L	N	-	-	-	-
bat ray	Myliobatidae	*Myliobatis californica*	B	N	25	rare	█	█
bay goby	Gobiidae	*Lepidogobius lepidus*	B	N	16	uncommon	█	█
bay pipefish	Syngnathidae	*Syngnathus leptorhynchus*	L	N	28	rare	█	█
big skate	Rajidae	*Raja binoculata*	B	N	34	rare	█	-
black perch	Embiotocidae	*Embiotoca jacksoni*	L	N	30	rare	█	█
black rockfish	Scorpaenidae	*Sebastes melanops*	B	N	-	-	-	-
bonyhead sculpin	Cottidae	*Artedius notospilotus*	B	N	26	rare	█	█
broadnose sevengill shark	Hexanchidae	*Notorynchus cepedianus*	B	N	-	-	-	-
brown rockfish	Scorpaenidae	*Sebastes auriculatus*	B	N	21	uncommon	█	-
brown smoothhound	Triakidae	*Mustelus henlei*	B	N	20	uncommon	█	█
buffalo sculpin	Cottidae	*Enophyrs bison*	B	N	-	-	-	-
cabazon	Cottidae	*Scorpaenichthys marmoratus*	B	N	-	-	-	-
calico surfperch	Embiotocidae	*Amphistichus koelzi*	L	N	-	-	-	-
California grunion	Atherinopsidae	*Leuresthes tenuis*	L	N	-	-	-	-
California halibut	Paralichthyidae	*Paralichthys californicus*	B	N	19	uncommon	█	█
California lizardfish	Synodontidae	*Synodus lucioceps*	B	N	-	-	-	-
California tonguefish	Cynoglossidae	*Symphurus atricaudus*	B	N	8	common	█	█
chameleon goby	Gobiidae	*Tridentiger trigonocephalus*	B	I	15	uncommon	█	█
cheekspot goby	Gobiidae	*Ilypnus gilberti*	B	N	24	rare	█	█
Chinook salmon	Salmonidae	*Oncorhynchus tshawytscha*	L	N	-	-	-	-
C-O sole	Pleuronectidae	*Pleuronichthys coenosus*	B	N	-	-	-	-
copper rockfish	Sebastidae	*Sebastes caurinus*	B	N	-	-	-	-
curlfin sole	Pleuronectidae	*Pleuronichthys decurrens*	B	N	-	-	-	-
diamond turbot	Pleuronectidae	*Pleuronichthys guttulatus*	B	N	29	rare	█	█
dwarf perch	Embiotocidae	*Micrometrus minimus*	L	N	13	uncommon	█	█
English sole	Pleuronectidae	*Parophrys vetulus*	B	N	9	common	█	█
eulachon	Osmeridae	*Thaleichthys pacificus*	P	N	35	rare	-	-
grass rockfish	Sebastidae	*Sebastes rastrelliger*	B	N	-	-	-	-
green sturgeon	Acipenseridae	*Acipenser medirostris*	B	N	37	rare	-	█
halfmoon	Kyphosidae	*Medialuna californiensis*	P	N	-	-	-	-
hornyhead turbot	Pleuronectidae	*Pleuronichthys verticalis*	B	N	-	-	-	-
jack mackerel	Carangidae	*Trachurus symmetricus*	P	N	-	-	-	-
jacksmelt	Atherinopsidae	*Atherinopsis californiensis*	P	N	-	-	-	-
kelp greenling	Hexagrammidae	*Hexagrammos decagrammus*	B	N	-	-	-	-
leopard shark	Triakidae	*Triakis semifasciata*	B	N	17	uncommon	█	█
lingcod	Hexagrammidae	*Ophiodon elongatus*	B	N	-	-	-	-
longfin smelt	Osmeridae	*Spirinchus thaleichthys*	P	N	4	common	█	█
longjaw mudsucker	Gobiidae	*Gillichthys mirabilis*	B	N	33	rare	-	█
Mississippi/Inland silverside	Atherinopsidae	*Menidia berrylina*	L		-	-	-	-
night smelt	Osmeridae	*Spirinchus starksi*	L	N	-	-	-	-
northern anchovy	Engraulidae	*Engraulis mordax*	P	N	2	abundant	█	█
Pacific electric ray	Torpedinidae	*Torpedo californica*	B	N	-	-	-	-
Pacific herring	Clupeidae	*Clupea pallasii*	P	N	14	uncommon	█	█
Pacific lamprey	Petromyzontidae	*Lampetra tridentata*	B	N	-	-	-	-
Pacific pompano	Stromateidae	*Peprilus simillimus*	P	N	-	-	-	-
Pacific sanddab	Paralichthyidae	*Citharichthys sordidus*	B	N	-	-	-	-
Pacific sardine	Clupeidae	*Sardinops sagax*	P	N	-	-	-	-
Pacific staghorn sculpin	Cottidae	*Leptocottus armatus*	B	N	1	abundant	█	█
Pacific tomcod	Gadidae	*Microgadus proximus*	P	N	27	rare	█	█
penpoint gunnel	Pholidae	*Apodichthys flavidus*	B	N	-	-	-	-
pile perch	Embiotocidae	*Rhacochilus vacca*	L	N	-	-	-	-
plainfin midshipman	Batrachoididae	*Porichthys notatus*	B	N	22	uncommon	█	█
prickly sculpin	Cottidae	*Cottus asper*	B	N	-	-	-	-

Table 7.2c (continued)

					South Bay Discharge Authority			
					Lower South Bay and Coyote Creek 1982-1986			
CommonName	**Family**	**ScientificName**	**P, B or L**	**N or I**	Rank	Status	%Shoal	%Slough
queenfish	Sciaenidae	*Seriphus politus*	L	N	-	-	-	-
rainbow surfperch	Embiotocidae	*Hypsurus caryi*	B	N	-	-	-	-
rainwater killifish	Fundulidae	*Lucania parva*	L	I	-	-	-	-
red irish lord	Cottidae	*Hemilepidotus hemilepidotus*	B	N	-	-	-	-
redtail surfperch	Embiotocidae	*Amphistichus rhodoterus*	L	N	-	-	-	-
rex sole	Pleuronectidae	*Glyptocephalus zachirus*	B	N	-	-	-	-
river lamprey	Petromyzontidae	*Lampetra ayresii*	B	N	-	-	-	-
rubberlip seaperch	Embiotocidae	*Rhacochilus toxotes*	L	N	-	-	-	-
saddleback gunnel	Pholidae	*Pholis ornata*	B	N	-	-	-	-
sand sole	Pleuronectidae	*Psettichthys melanostictus*	B	N	-	-	-	-
shimofuri goby	Gobiidae	*Tridentiger bifasciatus*	B	I	-	-	-	-
shiner perch	Embiotocidae	*Cymatogaster aggregata*	L	N	3	abundant	■	■
shokihaze goby	Gobiidae	*Tridentiger barbatus*	B	I	-	-	-	-
shovelnose guitarfish	Rhinobatidae	*Rhinobatos productus*	B	N	-	-	-	-
showy snailfish	Liparidae	*Liparis pulchellus*	B	N	31	rare	■	■
silver surfperch	Embiotocidae	*Hyperprosopon ellipticum*	L	N	-	-	-	-
speckled sanddab	Paralichthyidae	*Citharichthys stigmaeus*	B	N	11	uncommon	■	■
spiny dogfish	Squalidae	*Squalus acanthias*	B	N	-	-	-	-
spotted cusk-eel	Ophidiidae	*Chilara taylori*	B	N	-	-	-	-
starry flounder	Pleuronectidae	*Platichthys stellatus*	B	N	5	common	■	■
striped bass	Moronidae	*Morone saxatilis*	P	I	10	common	■	■
striped seaperch	Embiotocidae	*Embiotoca lateralis*	L	N	-	-	-	-
surf smelt	Osmeridae	*Hypomesus pretiosus*	L	N	38	rare	■	
thornback	Platyrhynidae	*Platyrhinoidis triseriata*	B	N	-	-	-	-
threadfin shad	Clupeidae	*Dorosoma petenense*	P	I	12	uncommon	■	■
threespine stickleback	Gasterosteidae	*Gasterosteus aculeatus*	L	N	-	-	-	-
topsmelt	Atherinopsidae	*Atherinops affinis*	L	N	18	uncommon	■	■
vermilion rockfish	Scorpaenidae	*Sebastes miniatus*	B	N	-	-	-	-
walleye surfperch	Embiotocidae	*Hyperprosopon argenteum*	L	N	36	rare	■	■
white croaker	Sciaenidae	*Genyonemus lineatus*	B	N	6	common	■	■
white seabass	Sciaenidae	*Atractoscion nobilis*	P	N	-	-	-	-
white seaperch	Embiotocidae	*Phanerodon furcatus*	L	N	-	-	-	-
white sturgeon	Acipenseridae	*Acipenser transmontanus*	B	N	32	rare	■	■
whitebait smelt	Osmeridae	*Allosmerus elongatus*	L	N	-	-	-	-
yellow snake eel	Ophichthidae	*Ophichthus zophochir*	B	N	-	-	-	-
yellowfin goby	Gobiidae	*Acanthogobius flavimanus*	B	I	7	common	■	■
yellowtail rockfish	Scorpaenidae	*Sebastes flavidus*	B	N				

Table 7.2d Fish species encountered during South Bay Salt Pond Restoration Project surveys conducted in South and Lower San Francisco Bay between 1982 and 1986. B=benthic, P=pelagic, L=litorral, N=native, I=invasive

					Alviso-Coyote Sloughs 2011-2013				*Bair Island Marsh- Redwood Cr. & Steinberger Slough*				*Eden Landing Marsh- Mt. Eden Cr. and Old Alameda Cr.*			
CommonName	**Family**	**ScientificName**	**P, B or L**	**N or I**	Rank	Status	% Pond	%Slough	Rank	Status	% Pond	%Slough	Rank	Status	% Pond	%Slough
American shad	Clupeidae	*Alosa sapidissima*	P	I	12	common	█	█	26	rare	-	█	17	rare	-	█
arrow goby	Gobiidae	*Clevelandia ios*	B	N	6	common	█	█	6	common	█	█	3	common	█	█
barred surfperch	Embiotocidae	*Amphistichus argenteus*	L	N	-	-	-	-	-	-	-	-	15	rare	-	█
bat ray	Myliobatidae	*Myliobatis californica*	B	N	19	uncommon	█	█	20	rare	-	█	12	uncommon	-	█
bay goby	Gobiidae	*Lepidogobius lepidus*	B	N	36	rare	█	█	5	abundant	█	█	-	-	-	-
bay pipefish	Syngnathidae	*Syngnathus leptorhynchus*	L	N	14	uncommon	█	█	27	rare		█	16	rare	-	█
big skate	Rajidae	*Raja binoculata*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
black perch	Embiotocidae	*Embiotoca jacksoni*	L	N	47	rare		█	14	uncommon	█	█	-	-	-	-
black rockfish	Scorpaenidae	*Sebastes melanops*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
bonyhead sculpin	Cottidae	*Artedius notospilotus*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
broadnose sevengill shark	Hexanchidae	*Notorynchus cepedianus*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
brown rockfish	Scorpaenidae	*Sebastes auriculatus*	B	N	50	rare	█	-	-	-	-	-	-	-	-	-
brown smoothhound	Triakidae	*Mustelus henlei*	B	N	46	rare	-	█	12	uncommon	-	█	-	-	-	-
buffalo sculpin	Cottidae	*Enophrys bison*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
cabazon	Cottidae	*Scorpaenichthys marmoratus*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
calico surfperch	Embiotocidae	*Amphistichus koelzi*	L	N	-	-	-	-	-	-	-	-	-	-	-	-
California grunion	Atherinopsidae	*Leuresthes tenuis*	L	N	-	-	-	-	-	-	-	-	-	-	-	-
California halibut	Paralichthyidae	*Paralichthys californicus*	B	N	31	uncommon	█	█	-	-	-	-	-	-	-	-
California lizardfish	Synodontidae	*Synodus lucioceps*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
California tonguefish	Cynoglossidae	*Symphurus atricaudus*	B	N	20	uncommon	█	█	32	rare	-	█	13	uncommon	-	█
chameleon goby	Gobiidae	*Tridentiger trigonocephalus*	B	I	43	rare	█	-	10	uncommon	-	█	7	uncommon	-	█
cheekspot goby	Gobiidae	*Ilypnus gilberti*	B	N	42	rare	█	-	-	-	-	-	-	-	-	-
Chinook salmon	Salmonidae	*Oncorhynchus tshawytscha*	L	N	-	-	-	-	-	-	-	-	-	-	-	-
C-O sole	Pleuronectidae	*Pleuronichthys coenosus*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
copper rockfish	Sebastidae	*Sebastes caurinus*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
curlfin sole	Pleuronectidae	*Pleuronichthys decurrens*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
diamond turbot	Pleuronectidae	*Pleuronichthys guttulatus*	B	N	38	rare	█	█	17	uncommon	█	-	18	rare	-	█
dwarf perch	Embiotocidae	*Micrometrus minimus*	L	`	-	-	-	-	18	uncommon	█	█	-	-	-	-
English sole	Pleuronectidae	*Parophrys vetulus*	B	N	4	common	█	█	8	common	█	█	2	common	█	█
eulachon	Osmeridae	*Thaleichthys pacificus*	P	N	-	-	-	-	-	-	-	-	-	-	-	-
grass rockfish	Sebastidae	*Sebastes rastrelliger*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
green sturgeon	Acipenseridae	*Acipenser medirostris*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
halfmoon	Kyphosidae	*Medialuna californiensis*	P	N	-	-	-	-	-	-	-	-	-	-	-	-
hornyhead turbot	Pleuronectidae	*Pleuronichthys verticalis*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
jack mackerel	Carangidae	*Trachurus symmetricus*	P	N	-	-	-	-	-	-	-	-	-	-	-	-
jacksmelt	Atherinopsidae	*Atherinopsis californiensis*	P	N	16	uncommon	█	█	23	rare	-	█	-	-	-	-
kelp greenling	Hexagrammidae	*Hexagrammos decagrammus*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
leopard shark	Triakidae	*Triakis semifasciata*	B	N	41	rare	█	█	19	uncommon	-	█	9	uncommon	-	█
lingcod	Hexagrammidae	*Ophiodon elongatus*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
longfin smelt	Osmeridae	*Spirinchus thaleichthys*	P	N	11	common	█	█	29	rare	-	█	-	-	-	-
longjaw mudsucker	Gobiidae	*Gillichthys mirabilis*	B	N	24	uncommon	█	█	-	-	-	-	-	-	-	-
Mississippi/Inland silverside	Atherinopsidae	*Menidia berrylina*	L		8	common	█	█	21	rare	█	█	8	uncommon	█	█
night smelt	Osmeridae	*Spirinchus starksi*	L	N	-	-	-	-	-	-	-	-	-	-	-	-
northern anchovy	Engraulidae	*Engraulis mordax*	P	N	3	common	█	█	2	abundant	█	█	5	uncommon	-	█
Pacific electric ray	Torpedinidae	*Torpedo californica*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
Pacific herring	Clupeidae	*Clupea pallasii*	P	N	5	common	█	█	3	abundant	█	█	11	uncommon	-	█
Pacific lamprey	Petromyzontidae	*Lampetra tridentata*	B	N	30	uncommon	█	█	-	-	-	-	-	-	-	-

Table 7.2d (continued)

					Alviso-Coyote Sloughs 2011-2013				*Bair Island Marsh- Redwood Cr. & Steinberger Slough*				*Eden Landing Marsh- Mt. Eden Cr. and Old Alameda Cr.*			
CommonName	**Family**	**ScientificName**	**P, B or L**	**N or I**	Rank	Status	% Pond	%Slough	Rank	Status	% Pond	%Slough	Rank	Status	% Pond	%Slough
Pacific pompano	Stromateidae	*Peprilus simillimus*	P	N	-	-	-	-	-	-	-	-	-	-	-	-
Pacific sanddab	Paralichthyidae	*Citharichthys sordidus*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
Pacific sardine	Clupeidae	*Sardinops sagax*	P	N	-	-	-	-	-	-	-	-	-	-	-	-
Pacific staghorn sculpin	Cottidae	*Leptocottus armatus*	B	N	1	abundant	█	█	1	abundant	█	█	1	abundant	█	█
Pacific tomcod	Gadidae	*Microgadus proximus*	P	N	-	-	-	-	-	-	-	-	-	-	-	-
penpoint gunnel	Pholidae	*Apodichthys flavidus*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
pile perch	Embiotocidae	*Rhacochilus vacca*	L	N	-	-	-	-	-	-	-	-	-	-	-	-
plainfin midshipman	Batrachoididae	*Porichthys notatus*	B	N	39	rare	█	█	15	uncommon	-	█	-	-	-	-
prickly sculpin	Cottidae	*Cottus asper*	B	N	18	uncommon	█	█	-	-	-	-	-	-	-	-
queenfish	Sciaenidae	*Seriphus politus*	L	N	-	-	-	-	-	-	-	-	-	-	-	-
rainbow surfperch	Embiotocidae	*Hypsurus caryi*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
rainwater killifish	Fundulidae	*Lucania parva*	L	I	22	uncommon	█	█	30	rare	█	█	19	rare	-	█
red irish lord	Cottidae	*Hemilepidotus hemilepidotus*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
redtail surfperch	Embiotocidae	*Amphistichus rhodoterus*	L	N	-	-	-	-	-	-	-	-	-	-	-	-
rex sole	Pleuronectidae	*Glyptocephalus zachirus*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
river lamprey	Petromyzontidae	*Lampetra ayresii*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
rubberlip seaperch	Embiotocidae	*Rhacochilus toxotes*	L	N	-	-	-	-	-	-	-	-	-	-	-	-
saddleback gunnel	Pholidae	*Pholis ornata*	B	N	45	rare	█	█	-	-	-	-	-	-	-	-
sand sole	Pleuronectidae	*Psettichthys melanostictus*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
shimofuri goby	Gobiidae	*Tridentiger bifasciatus*	B	I	40	rare		█	24	rare	-	█	21	rare	-	█
shiner perch	Embiotocidae	*Cymatogaster aggregata*	L	N	13	uncommon	█	█	4	abundant	█	█	6	uncommon	-	█
shokihaze goby	Gobiidae	*Tridentiger barbatus*	B	I	29	uncommon	█	█	34	rare	-	█	-	-	-	-
shovelnose guitarfish	Rhinobatidae	*Rhinobatos productus*	B	N	37	rare	█	█	-	-	-	-	-	-	-	-
showy snailfish	Liparidae	*Liparis pulchellus*	B	N	52	rare		█	-	-	-	-	-	-	-	-
silver surfperch	Embiotocidae	*Hyperprosopon ellipticum*	L	N					-	-	-	-	-	-	-	-
speckled sanddab	Paralichthyidae	*Citharichthys stigmaeus*	B	N	17	uncommon	█	█	13	uncommon	█	█	10	uncommon	-	█
spiny dogfish	Squalidae	*Squalus acanthias*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
spotted cusk-eel	Ophidiidae	*Chilara taylori*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
starry flounder	Pleuronectidae	*Platichthys stellatus*	B	N	10	common	█	█	22	rare	-	█	20	rare	-	█
striped bass	Moronidae	*Morone saxatilis*	P	I	15	uncommon	█	█	-	-	-	-	-	-	-	-
striped seaperch	Embiotocidae	*Embiotoca lateralis*	L	N	-	-	-	-	-	-	-	-	-	-	-	-
surf smelt	Osmeridae	*Hypomesus pretiosus*	L	N	-	-	-	-	-	-	-	-	-	-	-	-
thornback	Platyrhynidae	*Platyrhinoidis triseriata*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
threadfin shad	Clupeidae	*Dorosoma petenense*	P	I	25	uncommon	█	█	25	rare	█	█	-	-	-	-
threespine stickleback	Gasterosteidae	*Gasterosteus aculeatus*	L	N	2	abundant	█	█	35	rare	█	-	-	-	-	-
topsmelt	Atherinopsidae	*Atherinops affinis*	L	N	9	common	█	█	7	common	█	█	14	rare	-	█
vermilion rockfish	Scorpaenidae	*Sebastes miniatus*	B	N	-	-	-	-	-	-	-	-	-	-	-	-
walleye surfperch	Embiotocidae	*Hyperprosopon argenteum*	L	N	-	-	-	-	-	-	-	-	-	-	-	-
white croaker	Sciaenidae	*Genyonemus lineatus*	B	N	49	rare		█	11	uncommon	-	█	-	-	-	-
white seabass	Sciaenidae	*Atractoscion nobilis*	P	N	-	-	-	-	-	-	-	-	-	-	-	-
white seaperch	Embiotocidae	*Phanerodon furcatus*	L	N	-	-	-	-	-	-	-	-	-	-	-	-
white sturgeon	Acipenseridae	*Acipenser transmontanus*	B	N	54	rare		█	-	-	-	-	-	-	-	-
whitebait smelt	Osmeridae	*Allosmerus elongatus*	L	N	-	-	-	-	-	-	-	-	-	-	-	-
yellow snake eel	Ophichthidae	*Ophichthus zophochir*	B	N	-	-	-	-	-	-	-	-	4	uncommon	-	█
yellowfin goby	Gobiidae	*Acanthogobius flavimanus*	B	I	7	common	█	█	9	uncommon	█	█	-	-	-	-
yellowtail rockfish	Scorpaenidae	*Sebastes flavidus*	B	N												

Table 7.3a Monthly relative abundance for the Abundant, Common and Uncommon species, including the target species during the San Francisco Bay Study, 1980-2012. South Bay Stations only

San Francisco Bay Study 1980-2012 - South Bay Stations Only

Species	Family	ScientificName	*Deep Channel Habitat*												*Shoals Habitat*											
			1	2	3	4	5	6	7	8	9	10	11	12	1	2	3	4	5	6	7	8	9	10	11	12
American Shad	Clupeidae	*Alosa sapidissima*	•	•	•	•	•	•	●	●	●	●	●	•	•	•	•	•	•	•	●	●	●	●	●	•
Arrow Goby	Gobiidae	*Clevelandia ios*	•	•	•	•	•	•	•	•	•	•	•	•	●	●	•	•	•	●	●	•	•	•	•	●
Barred Surfperch	Embiotocidae	*Amphistichus argenteus*	•				•		•		•	•	•		•	•	•	•	●	●	●	●	●	●	•	•
Bat Ray	Myliobatidae	*Myliobatis californica*	•	•	•	•	•	•	●	●	●	•	•	•	•	•	●	●	●	●	•	•	•	●	●	●
Bay Goby	Gobiidae	*Lepidogobius lepidus*	•	•	•	•	•	•	•	•	•	•	•	•	●	●	•	•	•	●	●	•	•	•	•	●
Bay Pipefish	Syngnathidae	*Syngnathus leptorhynchus*	●	●	•	•	•	•	•	•	•	●	●	●	●	●	●	●	•	•	•	●	●	●	●	●
Big Skate	Rajidae	*Raja binoculata*	●	●	●	●	●	●	●	●	●	●	●	•	•	•	•	•	•	•	•	•	•	•	•	•
Bonyhead Sculpin	Cottidae	*Artedius notospilotus*	●	●	●	●	●	●	●	●	●	●	•	●	•	•	•	•	•	•	•	•	•	•	•	•
Brown Rockfish	Scorpaenidae	*Sebastes auriculatus*	●	●	●	●	•	•	•	•	•	•	•	•	•	•	•	•	●	●	●	●	●	●	•	•
Brown Smoothhound	Triakidae	*Mustelus henlei*	•	•	•	•	●	●	●	●	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•
California Halibut	Paralichthyidae	*Paralichthys californicus*	●	●	●	•	•	•	•	•	•	•	•	•	●	●	●	●	●	●	•	•	•	●	●	●
California Tonguefish	Cynoglossidae	*Symphurus atricaudus*	•	•	•	•	•	●	●	●	•	•	•	•	•	•	•	•	•	●	•	•	•	•	•	•
Chameleon Goby	Gobiidae	*Tridentiger trigonocephalus*	•	•	•	•	•	•	•	•	•	•	•	•	●	●	●	●	•	•	•	•	•	●	●	●
Cheekspot Goby	Gobiidae	*Ilypnus gilberti*	•	•	•	•	•	•	•	•	●	●	•	•	•	•	•	●	•	•	●	●	●	●	●	●
Diamond Turbot	Pleuronectidae	*Pleuronichthys guttulatus*	●	•	•	•	•	•		•	•	•	•	•	•	●	●	●	●	●	●	●	•	•	•	•
Dwarf Perch	Embiotocidae	*Micrometrus minimus*	•	●	•	•	•	•	•	•	•	•	•	•	●	•	•	•	•	•	•	•	•	•	•	●
English Sole	Pleuronectidae	*Parophrys vetulus*	•	•	•	•	•	•	•	•	•	•	•	•	•	●	●	●	●	●	•	•	•	•	•	•
Jacksmelt	Atherinopsidae	*Atherinopsis californiensis*	a,j	a,j	a,j	a,j	a,j	a,j	a,J	a,J	a,J	a,J	a,J	j	a,j,	a,j	A,j	A,j	A,j	A,j	A,J	a,J	a,J	a,J	a,J	a,j
Leopard Shark	Triakidae	*Triakis semifasciata*	●	●	●	●	●	●	●	●	●	●	●	●	•	•	•	•	•	•	•	•	•	•	•	•
Longfin Smelt	Osmeridae	*Spirinchus thaleichthys*	A	A	A	a	a	a,j	a,J	j	j	j	a,j	a,J	a	a	a	a	a,j	j	j	j	j	a,j	a,j	a ,J
Longjaw Mudsucker	Gobiidae	*Gillichthys mirabilis*	▨	▨	▨	▨	▨	▨	▨	▨	▨	•	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
Mississippi Silverside	Atherinopsidae	*Menidia audens*	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
Northern Anchovy	Engraulidae	*Engraulis mordax*	j	j	a,J	A,J	A,J	A,J	A,J	A,J	A,j	a,j	a,j	j	j	j	a,j	A,j	A,J	A,J	A,J	A,J	a,J	a,j	a,j	j
Pacific Herring	Clupeidae	*Clupea pallasii*	a	A	a,j	a,J	a,J	j	j	j	j	j	j	a,j	A	a	a,j	A,J	a,J	a,J	j	j	j	j	a	a
Pacific Sardine	Clupeidae	*Sardinops sagax*	a,j	a,j	j	a,J	a,J	a	A	▨	a	a,j	j	a,j	j	j	j	J	a,J	A	a	a	a	a,j	a,J	a,j

KEY

●	more abundant (species specific)
•	less abundant (species specific)
▨	No catch

A	Adults abundant (1+ aged fish)
a	Adult less abundant
J	Juveniles more abundant (species specific, age 0 fish)
j	Juveniles less abundant (species specific, age 0 fish)

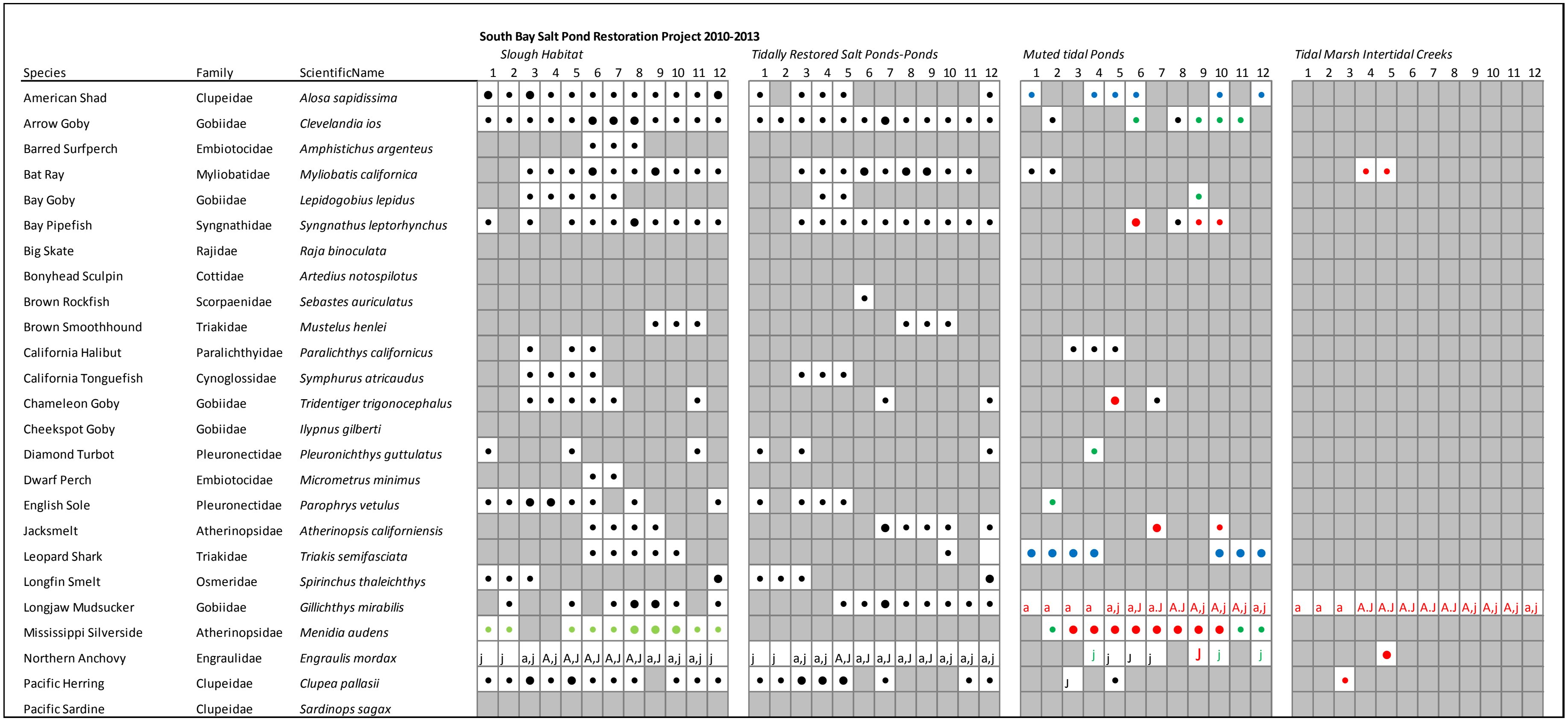
black	Otter Trawls
red	Minnow Traps
green	Beach Seines
blue	Gill Nets

Table 7.3a (continued)

San Francisco Bay Study 1980-2012 - South Bay Stations Only

			Deep Channel Habitat												*Shoals Habitat*											
Species	Family	ScientificName	1	2	3	4	5	6	7	8	9	10	11	12	1	2	3	4	5	6	7	8	9	10	11	12
Pacific Staghorn Sculpin	Cottidae	*Leptocottus armatus*	A,j	A,j	A,J	A,J	A,J	AJ	a,J	j	j	j	j	j	a,j	a,j	a,j	a,J	a,J	a,J	a,J	a,J	j	a,j	j	j
Pacific Tomcod	Gadidae	*Microgadus proximus*	a	a	a,j	a	a	a	▨	▨	▨	▨	j	j	a	a	a,j	a,j	a	a	a	a	▨	▨	▨	j
Pile Perch	Embiotocidae	*Rhacochilus vacca*	▨	▨	•	▨	▨	▨	▨	▨	▨	▨	•	•	•	•	•	•	•	•	•	•	•	•	•	•
Plainfin Midshipman	Batrachoididae	*Porichthys notatus*	A	A	A	A	A,j	A,J	A,J	J	J	J	J	j	a	a	a,j	A,J	a	a,j	a,J	J	J	j	j	j
Rainwater Killifish	Fundulidae	*Lucania parva*	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
Shimofuri Goby	Gobiidae	*Tridentiger bifasciatus*	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
Shiner Surfperch	Embiotocidae	*Cymatogaster aggregata*	A	A	A	A	a,j	a,j	a,j	a,j	a,j	a,j	a,j	a,j	a	a	a	A	A,j	A,J	A,J	a,J	a,J	a,j	a,j	a,j
Shokihaze Goby	Gobiidae	*Tridentiger barbatus*	●	•	•	•	•	•	•	•	•	•	•	●	•	▨	▨	▨	•	▨	▨	▨	▨	▨	▨	•
Speckled Sanddab	Paralichthyidae	*Citharichthys stigmaeus*	A,J	A,J	A,J	A,J	A,J	A,J	a,J	a,j	a,j	a,j	a,j	a,j	a,j	A,J	A,J	A,J	A,J	A,J	a,j	a,j	a,j	a,j	a,j	A,J
Starry Flounder	Pleuronectidae	*Platichthys stellatus*	A	A	a	a	a,j	a,j	j	a,j	a,j	j	a	a,j	A	a	a	A,j	A,j	A,J	a,j	A,j	a,j	a,j	a,j	a,j
Striped Bass	Moronidae	*Morone saxatilis*	a,j	a,j	a,j	a,j	a,j	a	a	a	a	a	▨	j	a,j	a,j	a,j	a,j	a,j	a,j	a,j		a,j	a,j	a,j	a,j
Threadfin Shad	Clupeidae	*Dorosoma petenense*	•	•	•	•	▨	▨	▨	▨	▨	▨	•	•	•	•	▨	▨	▨	▨	▨	▨	▨	•	•	•
Threespine Stickleback	Gasterosteidae	*Gasterosteus aculeatus*	•	•	▨	•	•	•	•	•	•	▨	▨	•	•	▨	•	▨	•	•	▨	▨	▨	▨	▨	▨
Topsmelt	Atherinopsidae	*Atherinops affinis*	a,J	a,J	a,J	a,j	a,j	a,j	a,j	a,J	a,J	a,J	a,J	a,J	a,J	a,J	a,J	a,j	a,j	a,j	a,j	a,j	a,J	A,J	A,J	A,J
Walleye Surfperch	Embiotocidae	*Hyperprosopon argenteum*	▨	•	▨	▨	▨	▨	▨	▨	▨	▨	▨	•	•	•	•	•	●	●	●	•	•	•	▨	•
White Croaker	Sciaenidae	*Genyonemus lineatus*	a,j	a,j	A,j	A,J	A,J	A,J	A,J	a,j	a,j	a,j	a,j	a,j	a,j	a,j	a,j	A,J	A,J	A,J	A,J	A,J	a,j	a,j	a,j	a,j
White Surfperch	Embiotocidae	*Phanerodon furcatus*	•	▨	▨	▨	▨	▨	▨	•	▨	▨	▨	▨	•	•	•	▨	●	●	●	●	•	•	•	•
Yellowfin Goby	Gobiidae	*Acanthogobius flavimanus*	a	a	a	a,j	j	J	j	j	a,j	a,j	a,j	a	a	a	a	a	a,j	a,j	a,j	j	j	j	j	j

Table 7.3b Monthly relative abundance for the Abundant, Common and Uncommon species, including the target species during the South Bay Salt Pond Restoration Project Monitoring Study 2010-2013



South Bay Salt Pond Restoration Project 2010-2013

Species	Family	ScientificName	*Slough Habitat*												*Tidally Restored Salt Ponds-Ponds*												*Muted tidal Ponds*												*Tidal Marsh Intertidal Creeks*											
			1	2	3	4	5	6	7	8	9	10	11	12	1	2	3	4	5	6	7	8	9	10	11	12	1	2	3	4	5	6	7	8	9	10	11	12	1	2	3	4	5	6	7	8	9	10	11	12
American Shad	Clupeidae	*Alosa sapidissima*	●	•	●	•	•	•	•	•	•	•	•	●	•		•	•	•							•	🔵			🔵	🔵	🔵				🔵		🔵												
Arrow Goby	Gobiidae	*Clevelandia ios*	•	•	•	•	•	●	●	●	•	•	•	•	•	•	•	•	•	•	●	•	•	•	•	•		•				🟢		•	🟢	🟢	🟢													
Barred Surfperch	Embiotocidae	*Amphistichus argenteus*						•	•	•																																								
Bat Ray	Myliobatidae	*Myliobatis californica*			•	•	•	●	•	•	●	•	•	•			•	•	•	●	•	●	●	•	•		•	•														🔴	🔴							
Bay Goby	Gobiidae	*Lepidogobius lepidus*			•	•	•	•	•									•	•																🟢															
Bay Pipefish	Syngnathidae	*Syngnathus leptorhynchus*	•		•		•	•	•	●	•	•	•	•			•	•	•	•	•	•	•	•	•	•						🔴		•	🔴	🔴														
Big Skate	Rajidae	*Raja binoculata*																																																
Bonyhead Sculpin	Cottidae	*Artedius notospilotus*																																																
Brown Rockfish	Scorpaenidae	*Sebastes auriculatus*																		•																														
Brown Smoothhound	Triakidae	*Mustelus henlei*									•	•	•									•	•	•																										
California Halibut	Paralichthyidae	*Paralichthys californicus*			•		•	•																					•	•	•																			
California Tonguefish	Cynoglossidae	*Symphurus atricaudus*			•	•	•	•									•	•	•																															
Chameleon Goby	Gobiidae	*Tridentiger trigonocephalus*			•	•	•	•	•				•								•					•					🔴		•																	
Cheekspot Goby	Gobiidae	*Ilypnus gilberti*																																																
Diamond Turbot	Pleuronectidae	*Pleuronichthys guttulatus*	•				•						•		•		•									•				🟢																				
Dwarf Perch	Embiotocidae	*Micrometrus minimus*						•	•																																									
English Sole	Pleuronectidae	*Parophrys vetulus*	•	•	●	●	•	•		•				•	•		•	•	•									🟢																						
Jacksmelt	Atherinopsidae	*Atherinopsis californiensis*						•	•	•	•										●	•	•	•		•							🔴			🔴														
Leopard Shark	Triakidae	*Triakis semifasciata*						•	•	•	•	•												•			🔵	🔵	🔵	🔵						🔵	🔵	🔵												
Longfin Smelt	Osmeridae	*Spirinchus thaleichthys*	•	•	•									●	•	•	•									●																								
Longjaw Mudsucker	Gobiidae	*Gillichthys mirabilis*		•			•		•	●	●	•		•					•	•	●	•	•	•	•	•	a	a	a	a	a,j	a,J	a.J	A.J	A,j	A,j	A,j	a,j	a	a	a	A.J	A.J	A.J	A.J	A.J	A,j	A,j	A,j	a,j
Mississippi Silverside	Atherinopsidae	*Menidia audens*	🟢	🟢			🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢														🟢	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🔴	🟢	🟢												
Northern Anchovy	Engraulidae	*Engraulis mordax*	j	j	a,j	A,j	A,J	A,J	A,J	A,J	a,J	a,j	a,j	j	j	j	a,j	a,j	A,J	a,J	a,J	a,J	a,J	a,j	a,j	a,j				j	j	J	j		J	j		j					🔴							
Pacific Herring	Clupeidae	*Clupea pallasii*	•	•	●	•	●	•	•	•		•	•	•	•	•	●	●	●		•				•	•			J		•										🔴									
Pacific Sardine	Clupeidae	*Sardinops sagax*																																																

KEY

●	more abundant (species specific)
•	less abundant (species specific)
▨	No catch

A	Adults abundant (1+ aged fish)
a	Adult less abundant
J	Juveniles more abundant (species specific, age 0 fish)
j	Juveniles less abundant (species specific, age 0 fish)

black	Otter Trawls
red	Minnow Traps
green	Beach Seines
blue	Gill Nets

Table 7.3b (continued)

South Bay Salt Pond Restoration Project 2010-2013

Slough Habitat

Species	Family	ScientificName	1	2	3	4	5	6	7	8	9	10	11	12
Pacific Staghorn Sculpin	Cottidae	*Leptocottus armatus*	•	•	●	●	●	●	●	●	●	●	●	●
Pacific Tomcod	Gadidae	*Microgadus proximus*												
Pile Perch	Embiotocidae	*Rhacochilus vacca*						•	•	•				
Plainfin Midshipman	Batrachoididae	*Porichthys notatus*						•	•	•	•			
Rainwater Killifish	Fundulidae	*Lucania parva*	•	•	•				●	•				
Shimofuri Goby	Gobiidae	*Tridentiger bifasciatus*				•		•	●		•			
Shiner Surfperch	Embiotocidae	*Cymatogaster aggregata*	•		●	•	•	●	•			●	•	•
Shokihaze Goby	Gobiidae	*Tridentiger barbatus*						•		•		●	•	•
Speckled Sanddab	Paralichthyidae	*Citharichthys stigmaeus*	•	●	•	•	•	•		•		•		•
Starry Flounder	Pleuronectidae	*Platichthys stellatus*	●	•	●	•	•	•	●	●	•	●	●	●
Striped Bass	Moronidae	*Morone saxatilis*			•	●	●	•	•	•		•	•	
Threadfin Shad	Clupeidae	*Dorosoma petenense*	•		•									●
Threespine Stickleback	Gasterosteidae	*Gasterosteus aculeatus*	•	●	●	•	•	●	●	●	●	●	●	●
Topsmelt	Atherinopsidae	*Atherinops affinis*	•	•	•		•	•		•	•	●	•	●
Walleye Surfperch	Embiotocidae	*Hyperprosopon argenteum*												
White Croaker	Sciaenidae	*Genyonemus lineatus*			•	•	•	•				•		
White Surfperch	Embiotocidae	*Phanerodon furcatus*												
Yellowfin Goby	Gobiidae	*Acanthogobius flavimanus*	●	•	•	•	•	●	●	●	•	●	●	●

Tidally Restored Salt Ponds-Ponds

Species	1	2	3	4	5	6	7	8	9	10	11	12
Pacific Staghorn Sculpin	•	•	●	●	●	●	•	•	●	●	●	•
Pacific Tomcod												
Pile Perch												
Plainfin Midshipman							•					
Rainwater Killifish							🟢	•	🟢	🟢	🟢	•
Shimofuri Goby												•
Shiner Surfperch	•		•	•	•	•	●	•		•		
Shokihaze Goby					•							
Speckled Sanddab	•	•										•
Starry Flounder	•		•	•	•	●	•	•	•	●		
Striped Bass			•	•	•	•	•					
Threadfin Shad	•	•	•									•
Threespine Stickleback	•	•	•	•	•	•	●	●	●	●	•	●
Topsmelt	•	•	🟢	🟢	🟢	🟢	🟢	🟢	●	●	●	●
Walleye Surfperch												
White Croaker										•		
White Surfperch												
Yellowfin Goby	•	•	●	•	•	●	●	•	•	•	•	•

Muted tidal Ponds

Species	1	2	3	4	5	6	7	8	9	10	11	12
Pacific Staghorn Sculpin			a	A,j	A,j	a,J	a,J	a,J	a,J	J	j	j
Pacific Tomcod												
Pile Perch												
Plainfin Midshipman								•	●			•
Rainwater Killifish					🟢	🔴	🔴	🔴	🟢	🔴	🔴	🔴
Shimofuri Goby												
Shiner Surfperch						J	j		j	j		
Shokihaze Goby												
Speckled Sanddab										🟢		
Starry Flounder												
Striped Bass										🔵		🔵
Threadfin Shad												
Threespine Stickleback			🟢		🔴	🔴	🔴			🔴	🔴	🟢
Topsmelt					🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢
Walleye Surfperch												
White Croaker												
White Surfperch												
Yellowfin Goby		🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢	🟢

Tidal Marsh Intertidal Creeks

Species	1	2	3	4	5	6	7	8	9	10	11	12
Pacific Staghorn Sculpin		a	a,J	a,J	a,J	j	a.j	A	A	A	A	A
Pacific Tomcod												
Pile Perch												
Plainfin Midshipman												
Rainwater Killifish					🔴		🔴					
Shimofuri Goby												
Shiner Surfperch				🔴								
Shokihaze Goby												
Speckled Sanddab												
Starry Flounder												
Striped Bass												
Threadfin Shad												
Threespine Stickleback	j	A,J	A,J	A,J	a,J	a,J	j	j	j	j	j	j
Topsmelt		🔴	🔴	🔴	🔴	🔴		🔴		🔴		
Walleye Surfperch												
White Croaker												
White Surfperch												
Yellowfin Goby			a.j	a,J	a,J	a,J	a,j	a	a	a	a	a

8. Key science questions and recommendations

In this report, we have summarized the state of the science related to the potential for adverse impacts from high nutrient concentration in Lower South Bay and have identified the following key knowledge gaps:

Nutrients

1. How do nutrient concentrations and forms vary spatially and temporally, particularly in margin habitats where limited monitoring has occurred to date?
2. What are the dominant processes controlling nutrient fate in Lower South Bay, and how do their magnitudes vary spatially and temporally?
3. What nutrient loads can Lower South Bay assimilate without adverse impacts (e.g., with respect to chl-a, DO, or algal toxins)?

Phytoplankton biomass

1. What combination of factors regulate phytoplankton productivity and biomass, and how do the relative importance of those factors vary spatially and seasonally?
2. What combination of factors can explain the fall biomass increase in the late-1990s (e.g., loss of filter-feeding benthos, decreasing suspended sediments)?
3. How important are margin habitats as a source of organic matter to the open Bay, in particular restored salt ponds?
4. How important is benthic algae production to overall productivity and organic matter accumulation in Lower South Bay?
5. What effects would potential management actions have on biomass, DO, and algal toxins? E.g.,
    a. Decreased nutrient loads by 25%, 50%, 75%?
    b. Operation of restored salt ponds (e.g., including optimization for nutrient removal and beneficial habitat condition)
    c. Managed oyster or mussel reefs

Phytoplankton community composition

(not explored in this report because of data limitations, but a priority nonetheless)

1. What factors most strongly regulate phytoplankton community composition in Lower South Bay (e.g., light availability, temperature, nutrients, benthic grazing, exchange with salt ponds)?
2. To what extent do conditions in Lower South Bay select for either potentially harmful algae or algae that poorly support the food web?
3. What are source(s) of algal toxins in Lower South Bay?

Dissolved oxygen

1. What are the frequency, duration, spatial extent, and severity of low DO events in shallow margin habitats?
2. What causes the substantial tidal-variations in DO concentrations at Dumbarton Bridge?
    a. What are conditions like elsewhere (south of Dumbarton) during these times?
3. Are biota being adversely impacted by low(er) DO in the margins or open Bay?
4. What factors most strongly regulate DO in sloughs and creeks, and what data collection is needed to best predict DO condition?
    a. Organic matter source(s)
    b. Role of anthropogenic nutrients
    c. Physics: stratification, salt pond exchange, slough-open Bay exchange

Fish and benthos

1. What are the DO preferences or tolerances of key fish species that are observed, or expected, in Lower South Bay?
2. How do fish populations and diversity respond to spatial and temporal variability in DO?
3. Do current conditions support or adversely impact benthos abundance or assemblage?

Effects of salt pond restoration

1. What effects are salt pond restoration activities having on nutrient, carbon and DO budgets in the margins? The open Bay?
2. What effects are salt pond restoration activities having on habitat conditions in Lower South Bay?
3. Are restored salt ponds a substantial source of harmful algal species and algal toxins?

Future scenarios

1. What levels of phytoplankton production and biomass are plausible under future scenarios in Lower South Bay? Have we reached a new plateau or will concentrations rise further?
    a. How will controls on phytoplankton biomass (i.e. light availability, benthic grazing) change in the future?
2. What would be protective nutrient levels in terms of biomass, DO, and phytoplankton assemblage or toxins?

Recommendations

The recommendations below emerged from considering current data or conceptual gaps. Since some of these investigations would be resource- and time-intensive undertakings, some level of prioritization is still needed.

R 1: Systematically investigate DO in the margins

- High frequency DO monitoring and ancillary data
- Evaluate relative importance of mechanisms that control DO through data interpretation and modeling
- Gather data to use in model calibration/validation

R 2: Develop improved quantitative understanding of controls of phytoplankton biomass, i.e. grazers and suspended sediment

- Determine whether additional monitoring and field investigations are needed, specifically benthos surveys and light levels.
- Modeling and sensitivity analysis

R 3: Gather high-spatial resolution data through biogeochemical mapping

- Characterize spatial and temporal heterogeneity
- Assess condition across the region – extent and severity of potentially problematic events
- Gather data for model calibration/validation

R 4: Conduct mechanistic field investigations to quantify important processes related to physical processes, nutrient cycling and phytoplankton and benthic algae production (slough ←→ open Bay, salt pond ←→ slough, stratification in open Bay and sloughs)

R 5: Develop and apply a coupled hydrodynamic and biogeochemical model for Lower South Bay, including sloughs and margins

- Evaluate mechanisms that control phytoplankton biomass and DO concentrations through sensitivity analysis
- Examine the role of anthropogenic nutrients and quantify nutrient fate
- Forecast ecosystem response under potential future conditions, including changing environmental factors (sediment concentrations, bivalves)
- Quantify how potential management actions, such as nutrient load reductions and salt pond operation, will influence ecosystem response (phytoplankton biomass, DO)
- Characterize and quantify uncertainty

R 6: Characterize phytoplankton community composition in Lower South Bay, and explore mechanisms that influence community composition, including potential sources of harmful algal species

R 7: Further analyze existing fish data to better characterize spatial and temporal variability in fish populations, and drivers of that variability

R 8: Explore the feasibility of using existing benthos survey data to assess habitat condition with respect to DO

R 9: Identify appropriate protective DO conditions for fish and other biota

- Literature review to determine:
    - What are we trying to protect?
    - What conditions would be protective?
    - What is the uncertainty in our understanding for species of interest?
- Compare existing conditions with protective conditions

R 10: Conduct fish and benthos surveys, in conjunction with habitat surveys (DO, T, food abundance and quality, etc.) to assess condition

9. References

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Section 6

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Appendix 1 Synthesis of Existing Dissolved Oxygen Data in Southern San Francisco Bay

SYNTHESIS OF EXISTING DISSOLVED OXYGEN DATA IN SOUTHERN SAN FRANCISCO BAY

October 2013

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Executive Summary

The San Francisco Bay Nutrient Management Strategy, established to guide activities in support of the development of nutrient objectives for San Francisco Bay, calls for the analysis of existing dissolved oxygen (DO) data in the Bay margins and diked bayland habitats, among numerous other projects. Previous reports and published papers of DO conditions along the longitudinal axis of the Bay show few instances of low dissolved oxygen occurrences; however, additional data were available for the margins of the Lower South Bay that have not yet been compiled and synthesized.

The purpose of this project was to compile and analyze available monitoring data for dissolved oxygen (DO) within Bay margin habitats, diked Bayland habitats, and subtidal open-water habitats in areas south of the San Mateo Bridge. Data from continuous sensors and discrete measurements were reviewed to: 1) estimate the prevalence of low oxygen events at monitored stations; and 2) describe the frequency and duration of low oxygen events at stations where they are documented. We compared observed concentrations to two DO benchmark concentrations: 1) low-DO conditions as defined by the San Francisco Bay Basin Plan water quality objectives for tidal waters downstream of the Carquinez Bridge (< 5.0 mg O_2 L^{-1}); and 2) hypoxic conditions, i.e., severely oxygen-depleted situations (<2.8 mg O_2 L^{-1}). More than 650,000 data points collected between 1994 and 2012 were included in this analysis, the majority of which came from continuous sensors measuring at high frequency (2-15 min intervals). For the purposes of data analysis, the observations were grouped into three habitat types: 1) open water subtidal habitat, 2) tidal slough, and 3) restored salt pond habitat.

In the open water habitat, no instances of DO < 5 mg L^{-1} were documented after 1998 and between 1994-1998, DO < 5 mg L^{-1} was observed only on two occasions. Low-DO was a fairly common occurrence at the tidal slough and former salt ponds sites where monitoring occurred. Generalizing observations across all sloughs is problematic because of the uneven distribution of measurements and their timing among sites. With that caveat noted, DO was < 5 mg L^{-1} at a mean frequency of 38% of the time and < 2.8 mg L^{-1} 10% of the time in tidal slough stations where continuous monitoring data were available. At former salt ponds sites, DO was < 5 mg L^{-1} at a mean frequencies of 55% of the time and < 2.8 mg L^{-1} 20% of the time. The frequency, duration, and severity (i.e., amount below thresholds) of low-DO varied substantially among the different sloughs and salt ponds as well as among stations within a given waterbody. Most of the documented low-DO events were short in duration (less than six hours), pointing to the importance of diurnal and semi-diurnal processes, such as oxygen production during photosynthesis and tidally-driven movement of water masses having different DO levels. However, several instances of extended low-oxygen events (>12 days) were documented.

Although the compiled dataset serves as a valuable starting point for exploring DO conditions in shallow habitats south of the San Mateo Bridge, the results need to be interpreted with caution. First, the data were compiled from multiple programs that had different monitoring objectives. As a result, the timing (e.g., seasons) and duration of measurements and the locations of sites were not designed to provide a representative system-wide view of condition. Second, while some basic data quality assurance/quality control (QA/QC) occurred during the initial data compilation step, no information was available on calibration, validation, or other observations (e.g., evidence of biofouling or instrument drift). Finally, low-DO is a common natural phenomenon in tidal creek and wetland habitats. It is not yet known whether observed DO deficits occurred more frequently or were more severe than would have occurred in the absence of anthropogenic impacts, or the role that anthropogenic nutrient loads may have played in creating the low-DO conditions. These issues need to be explored in more detail in future studies.

1. Introduction

The San Francisco Bay Regional Water Quality Control Board (Regional Board) is developing nutrient water quality objectives for San Francisco Bay. This effort is aligned with a broader effort by the State Water Board to develop nutrient water quality objectives for the State's surface waters, using an approach known as the Nutrient Numeric Endpoint (NNE) framework. The NNE establishes a suite of numeric endpoints based on the ecological response of an aquatic waterbody to nutrient overenrichment (e.g. eutrophication and other nutrient-related adverse effects). The NNE framework is intended to serve as numeric guidance to translate narrative water quality objectives into quantitative targets, and is currently under development for all California estuaries (Sutula 2011). Because San Francisco Bay represents California's largest estuary (70% by area of estuarine habitat statewide), it merits development of an estuary-specific NNE framework.

Work on the San Francisco Bay NNE began with an extensive literature review that identified candidate NNE indicators, summarized the status of condition based on these indicators in San Francisco Bay, reviewed available nutrient loading data, and identified key data gaps and next steps (McKee et al. 2011). Subsequently, the Regional Board worked collaboratively with stakeholders to develop the San Francisco Bay Nutrient Management Strategy, which established a framework for building the scientific basis for setting objectives and managing nutrients in the Bay. Both the NNE literature review and the Nutrient Management Strategy call for the analysis of existing dissolved oxygen (DO) data in the Bay margins and diked bayland habitats. This recommendation was echoed in a recent nutrient conceptual model report for San Francisco Bay, which laid out high-priority data gaps and science questions, and identified dissolved oxygen concentrations in shallow margin habitats as topics for further monitoring and analysis.

The purpose of this project was to compile and conduct a preliminary analysis of available monitoring data for dissolved oxygen (DO) within Bay margin habitats, diked Bayland habitats, and subtidal open-water habitats in areas south of the San Mateo Bridge. Data from continuous sensors and discrete measurements were reviewed to: 1) estimate the prevalence of low oxygen events at monitored stations; and 2) describe the spatial and temporal patterns for stations at which low oxygen events are documented. We compared observed concentrations to two DO benchmarks concentrations: 1) low-DO conditions as defined by the San Francisco Bay Basin Plan water quality objectives for tidal waters downstream of the Carquinez Bridge ($<$ 5.0 mg O_2 L^{-1}); and 2) hypoxic conditions, i.e., severely oxygen-depleted situations ($<$2.8 mg O_2 L^{-1}).

It is well known that, under natural conditions, shallow tidal habitats frequently experience low-oxygen conditions, as well as diurnal oscillations between low and high DO. Therefore, a key issue that needs to be addressed is whether observed DO conditions across these habitats are similar to natural conditions, or are worse (e.g., lower DO, more frequent or longer durations of low DO), and whether conditions constitute an impaired state. However, addressing those issues are beyond the scope of this report. This report's summary of DO observations is considered a first step, which needs to be followed by other investigations that go into mechanistic interpretations and evaluate whether impairment is occurring.

2. Data Description and Approach

Dissolved oxygen data from sites south of the San Mateo Bridge were gathered by the Regional Board from multiple agencies that have been monitoring water quality in these habitats over the past 10-20 years. The agencies include: California Department of Fish and Wildlife (DFW), City of Palo Alto, City of San Jose, City of Sunnyvale, San Francisco Bay Regional Monitoring Program (RMP), Santa Clara Valley Urban Runoff Pollution Prevention Program (SCVURPPP), and the U.S. Geological Survey (USGS). The data were organized into a consistent database format and analyzed the data. Some basic data QA/QC occurred during the data compilation step, including removal of data that appeared erroneous. While a number of checks were performed to catch erroneous data, we cannot strictly rule out the possibility of additional erroneous data. Furthermore, no information was available on calibration, validation, or other observations (e.g., evidence of biofouling or instrument drift) to allow further QA/QC. A full data summary table is provided in Appendix B, and the electronic database will be made publicly available.

Figure 1a-c shows the sampling locations. Discrete data from 1993-2009 were included. Continuous data was available at some sites beginning in 2004; Table 1 provides an overview of the stations and timing for which continuous DO data were available.

For the purposes of data analysis, stations were grouped into three habitat classes (Figure 1a-c):

1. Open Bay subtidal habitat
2. Tidal creeks and sloughs
3. Former (restored) salt ponds

We compared measured DO data to two benchmark concentrations: < 5.0 mg O_2 L^{-1} and <2.8 mg O_2 L^{-1}. The value of 5 mg O_2 L^{-1} is equivalent to the San Francisco Bay Basin Plan objectives for tidal waters downstream of the Carquinez Bridge (SFRWQCB 2013) and is a value below is generally considered to be oxic but low quality waters (Vaquer-Sunyer and Duarte 2008, Sutula et al. 2012). Waters with DO concentrations < 2.8 mg O_2 L^{-1} are considered hypoxic and acutely toxic to fish (Sutula et al. 2012).

Different types of DO data provide different types of information about condition. For example, DO measured in discrete samples collected synoptically at multiple sites provides a snap shot of oxygen conditions at a point in time over a broad area; however it fails to capture temporal (e.g., diurnal or semi-diurnal) variations in concentration. Continuous measurements with in situ sensors, on the other hand, can identify high-temporal resolution trends in DO levels and provide a means for characterizing the frequency and duration of hypoxia, but only represent a single point in space. For discrete samples, the time of sampling can have strong influence on observed concentration due to influences of tidal stage and O_2 production during photosynthesis, and this information was not considered as part of the current analysis. Time of day is not a problem for continuous measurements; however, uneven distribution in the months or seasons during which instruments were deployed may influence interpretations when data are aggregated.

Where continuous data were available, the frequency and duration of events < 2.8 and < 5 mg O2 L^{-1} were characterized. In addition, where discrete data records longer than 6 months were available, seasonal variability in percentage of low oxygen events occurred was determined.

A major limitation of this study is that the data were compiled from multiple programs with differing monitoring objectives. Therefore, the timing and duration of measurements (Table 1), and the locations of sites, do not necessarily provide a representative view of condition. Conclusions can be drawn about the monitored areas during periods when sampling occurred, but cannot necessarily be extrapolated to other sites or other time periods. For that reason, the data are considered a starting point for exploring the linkages between DO and contributing factors in this region of San Francisco Bay.

3. Results

3.1 Open Bay Subtidal Habitat

During 1993-1998, DO <5 mg L^{-1} was observed in only two instances in the discrete data available for open Bay subtidal habitat. Since 1998 there have been no documented observations of DO < 5 mg L^{-1} at subtidal sampling locations (Table 2). Most of the data included in the subtidal habitat dataset are from discrete samples collected by the San Francisco Bay Regional Monitoring Program (RMP). DO measurements from the USGS monthly Polaris cruises were not included in this analysis; however the DO concentrations observed on those cruises were typically >5 mg L^{-1}.

3.2 Tidal Creek and Slough Habitat

Low oxygen events were a fairly common occurrence at a majority of sampling locations in tidal creeks and sloughs (Table 3). Generalizing observations across all sloughs is problematic because of the uneven distribution of measurements and their timing among sites (Table 1). With that caveat noted, for tidal slough stations at which continuous monitoring data were available, DO was < 5 mg L^{-1} at a mean frequency of 38% of the time (20-90% interquartile range) and < 2.8 mg L^{-1} 10% of the time (1-36% interquartile range) in tidal slough stations where continuous monitoring data were available.

There was considerable variability in the frequency of low DO observations among different channels, as well as among stations within the same channel for some of the channels. For example, Guadalupe Slough had the highest proportion of records below the benchmarks, but that proportion varied widely among the different Guadalupe Slough monitoring locations (Table 3). Both continuous and discrete data were available for Guadalupe Slough. Low DO occurred most frequently at three continuous monitoring stations deployed during ~1 month in August 2007 (Guadalupe Landward, Middle, and Seaward; 90-95% <5 mg O_2 L^{-1}, 30-38% < 2 mg O_2 L^{-1}). DO values < 5mg L^{-1} were also consistently observed at stations sampled discretely (15-54%), and at a fourth continuous monitoring station (C-1-3; 34%) deployed across a range of months/seasons. Similar frequencies of DO < 5 mg L^{-1} were observed at continuous (35%) and discrete (12-33%) sites along Moffett Channel, which is a tributary to Guadalupe Slough. C-5 was the only station along Guadalupe slough where DO < 5 mg L^{-1} was never observed.

Compared to Guadalupe Slough, Artesian Slough had low percentages of records below the thresholds: 1% of values fell below 5 mg O_2 L^{-1} and 0% below mg O_2 L^{-1} (Figure 2). The Artesian Slough site (Figure 1b) experiences conditions that are influenced by discharge from the San Jose/Santa Clara Regional Wastewater Facility, discharges from two former salt ponds, and water movements into and out of the channel due to tides. The data record at Artesian Slough extends from 2006 to 2012, and the sampling period is April/May through October during each of these years (Table 1), and was the most consistently monitored of all the slough sites. Measures of low DO occurred in Artesian slough during the months of June, July, August, and September.

Conditions at continuous DO stations in Alviso Slough were more comparable to sites in Guadalupe Slough than Artesian Slough (Table 3). In Alviso Slough, DO < 5 mg L^{-1} occurred 29% of the time at the furthest upstream site (205AVSMBD), and only 15% of the time at downstream site (205AVSPA6). A ~10-month DO record (June 2012-April 2013, data not shown) at another site along Alviso Slough (near 205AVSMBD) shows a high frequency (>50% of measured values) of DO < 5 mg L^{-1} and strong seasonality, with pronounced lower frequency observed in November-February (M. Downing, USGS, personal communication). These data will be considered in a future report.

Across all tidal sloughs, roughly half of all low-DO events (47%, n= 1217) lasted less than one hour, and the vast majority (80%) lasted less than six hours (Figure 2). For ~32% and ~62% of the 1 h and 1-6 h events, respectively, DO was actually less than 2.8 mg L^{-1}. The longest low-DO event was observed in Moffett Channel at station MC-2000 (Figure 3) and lasted 6 days (March 13-18, 2011), with several periods of anoxic conditions, of which two lasted more than 24 hrs. The longest continuous period of anoxic conditions was documented at station MC-2000 on March 22-23, 2011.

3.3 Former Salt Ponds

Continuous DO data were available for 16 salt ponds, and in some cases sensors were deployed at multiple sites within a pond (Table 4), although the timing and duration of deployments varied considerably (Table 1). Similar to sloughs, generalization of observations across all ponds is problematic because of the uneven distribution of measurements and their timing among sites (Table 1). For ponds at which continuous monitoring data was available, the mean frequencies with which DO < 5 mg L^{-1} and DO < 2.8 mg L^{-1} were 55% (48-70% interquartile range) and 20% (10-35% interquartile range), respectively.

There was substantial variability in DO conditions among different salt ponds, as well as among stations within a salt pond, but that variability was somewhat less pronounced than in tidal sloughs (Figure 5). Notable exceptions are the records for B6A where 100% of measured values <2.8 mg L^{-1}, while at BW-15 100% of measured values were >5 mg L^{-1}. The results at B6A are biased by the fact that the data utilized were collected during one short time window (Apr-Jun 2006) during which conditions may have been fairly constant, whereas most other ponds were sampled over multiple seasons. Similarly, data for pond BW-15 may reflect a bias in that only 13 discrete samples were collected between 1997 and 2001, and may not represent the range of conditions that occur there.

The majority of recorded low-DO events (65%) in former salt ponds lasted less than six hours (Figure 6). There were a higher proportion of 6-12h and 12-24h events with DO < 5mg L^{-1} in restored ponds than in sloughs (Figure 2). Four low-DO events in restored ponds lasted more than 12 days, with anoxic conditions lasting for the entire period of these events. These events were the longest documented periods of continued hypoxic conditions and occurred in Ponds B2C and 6A (Figure 7).

4. Discussion and Next Steps

The results presented in this report, compiled from a range of monitoring efforts, indicate that both tidal slough and former salt pond habitats south of the San Mateo Bridge experienced DO < 5 mg L^{-1} with fairly high frequency, and DO < 2.8 mg L^{-1} with low to moderate frequency (Figure 5). Data available for open Bay subtidal stations indicate these habitats rarely experience DO < 5 mg L^{-1}.

This summary should be considered a first step, which needs to be followed by additional data analysis and possibly field investigations to assess the degree to which anthropogenic nutrients may be contributing to low DO conditions, and whether those conditions are impairing beneficial uses. While several important caveats noted below need to be considered, low DO is a classic symptom of nutrient-overenrichment, and it is reasonable to hypothesize that anthropogenic nutrient loads – either on their own or acting in concert with other ecosystem disturbances – contribute to the observed low DO events. For example, anthropogenicallysourced nutrients enter the salt ponds that are undergoing restoration. While nutrients that enter the subtidal habitats often go unutilized by phytoplankton due to light limitation, the salt ponds are shallow, high-light environments where phytoplankton production occurs more rapidly. The biomass produced would exert oxygen demand within the salt ponds and in sloughs and creeks to which they drain. Low DO occurred with fairly high frequency at the slough and pond sites considered in this report, and also at sites not included here (e.g., Alviso Slough, M. Downing-Kunz, personal communication; Coyote Creek, J Hobbs, personal communication). Furthermore, recent catch data at some margin sites in this region of the Bay suggest that while some fish species (e.g., anchovies) may tolerate these low DO values, other species may be avoiding low DO areas (leopard sharks; J Hobbs, personal communication) resulting in periodically lost habitat.

Although the compiled dataset serves as a valuable starting point for exploring DO conditions in shallow habitats south of the San Mateo Bridge, the results need to be interpreted with caution. First, the data were compiled from multiple programs that had different monitoring objectives. As a result, the timing (e.g., seasons) and duration of measurements and the locations of sites were not designed to provide a representative system-wide view of condition. This is well-illustrated by Table 5a and 5 b, which present graphical summaries of the percent of time data fell below 5 mg O_2 L^{-1} and 2.8 mg O_2 L^{-1} thresholds, respectively, distributed across the months and years when data was collected at continuous monitoring sites. For tidal sloughs, it is difficult to discern seasonal patterns because of large gaps in data availability. The one site that has regularly-collected over multiple seasons and multiple years is in channel that is heavily influenced by well-oxygenated freshwater inputs of treated wastewater effluent, and apparently is not representative of other sloughs. While there is substantially more data available for former salt ponds, data discontinuities in these systems nonetheless make it difficult to generalize across seasons and among ponds.

Second, while some basic data QA/QC occurred during the data compilation step, and several checks were performed to catch erroneous data (including removal of sizable portions of data that appeared erroneous), we cannot strictly rule out the possibility that some erroneous data remains in the database. This is in part due to the fact that no information was available on calibration, validation, or other observations (e.g., evidence of biofouling or instrument drift).

Finally, low-DO is a common and naturally occurring condition in tidal creek and wetland habitats (MacPherson et al., 2007). It is not yet known whether DO deficits occurred more frequently or were more severe in these Bay margin habitats than would have occurred in the absence of anthropogenic impacts, nor whether they constitute an impairment of beneficial uses. Furthermore the role that anthropogenic nutrient loads may have played in creating the low-DO conditions has not yet been quantified.

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Figures:

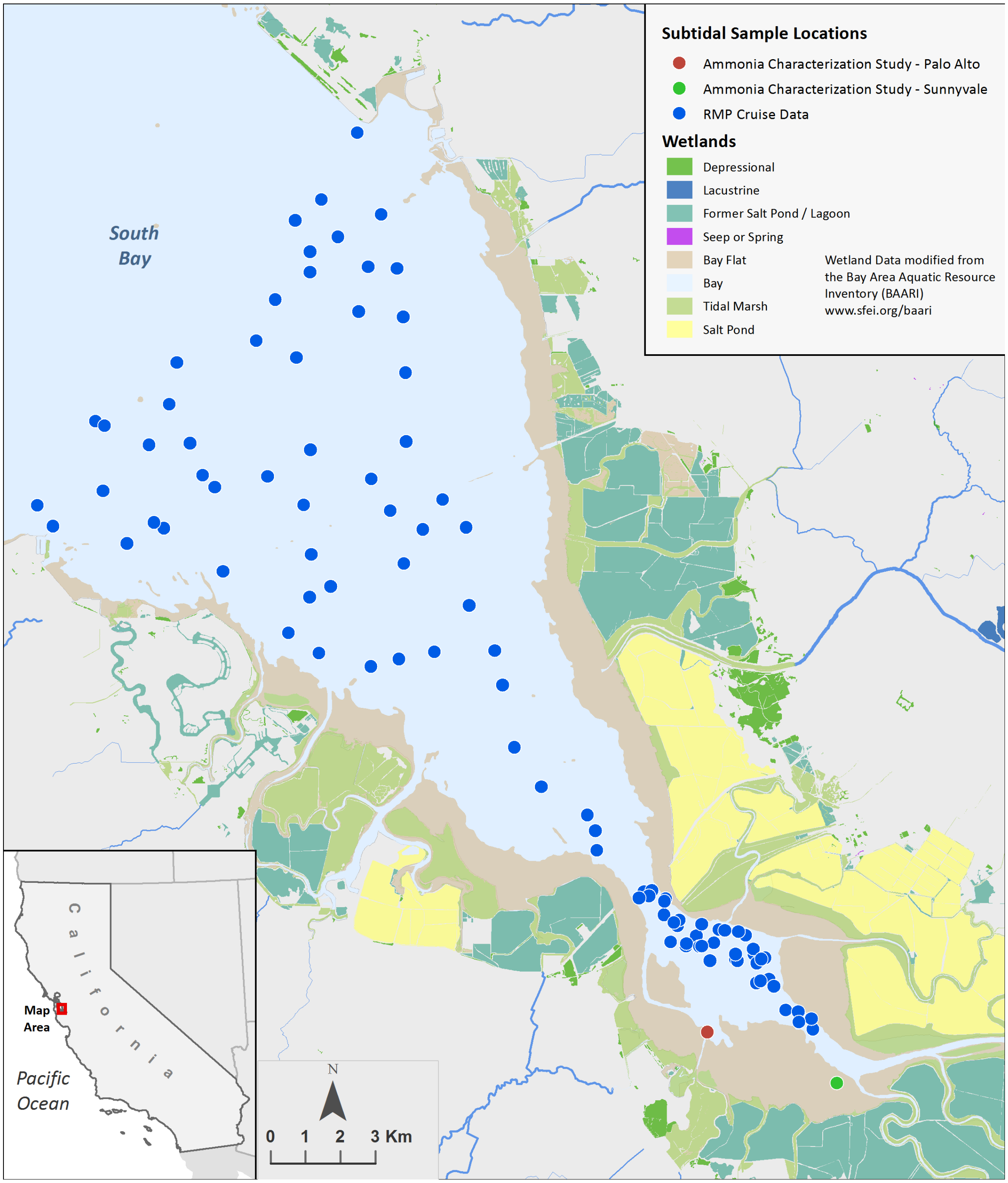

Figure 1a. Lower San Francisco Bay open water sampling locations.

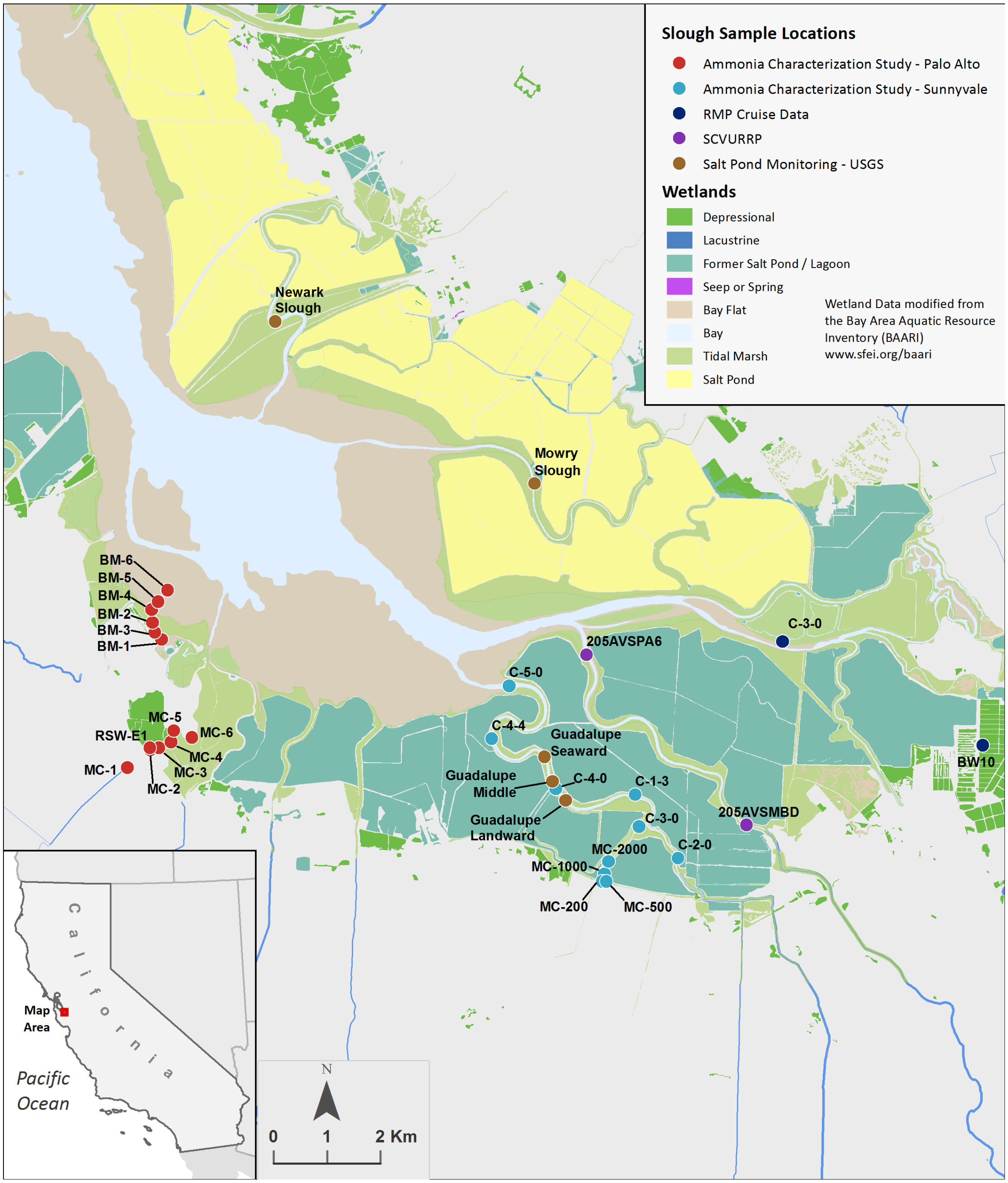

Figure 1b. Tidal slough/creek sampling locations.

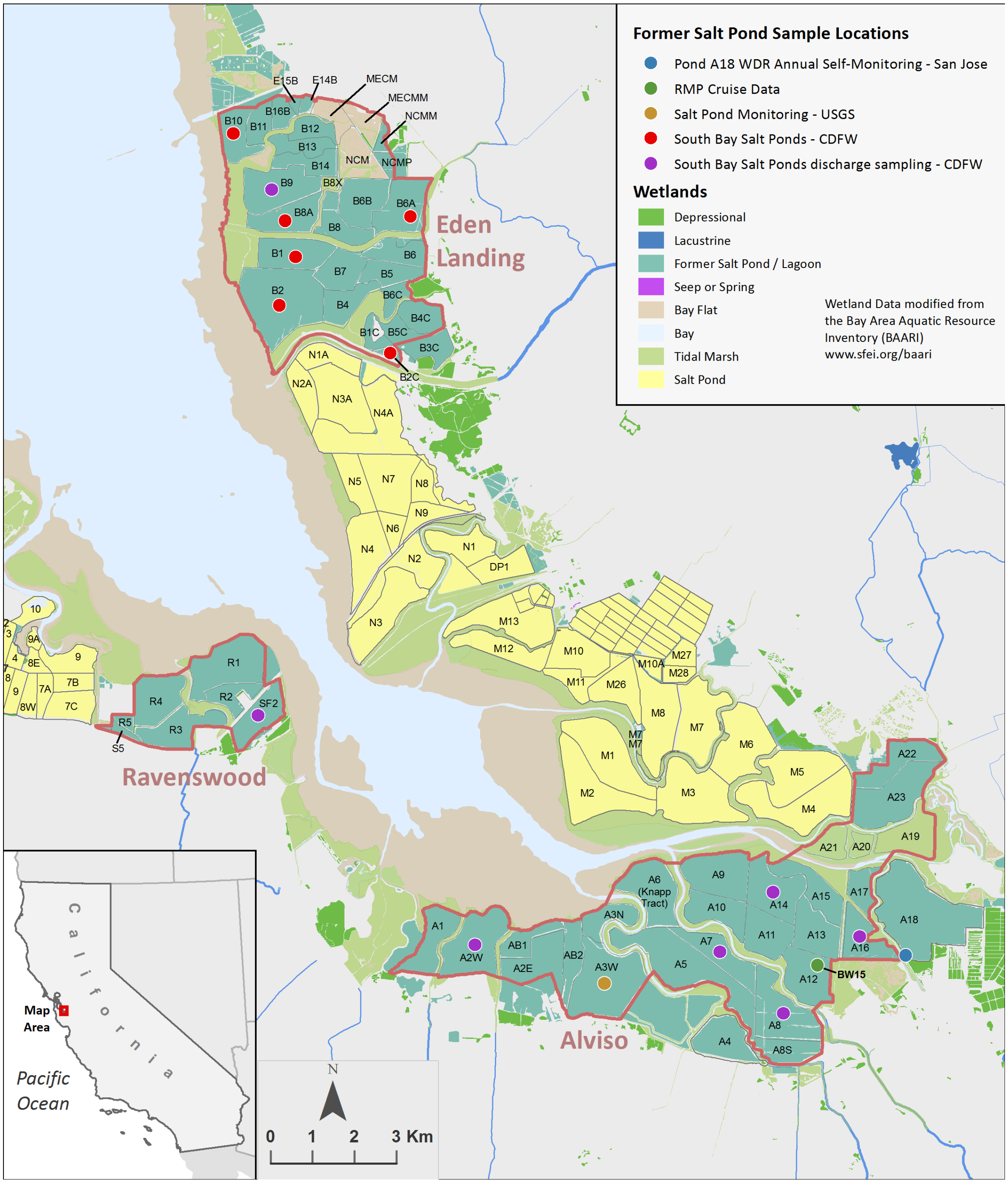

Figure 1c. Former salt pond sampling locations.

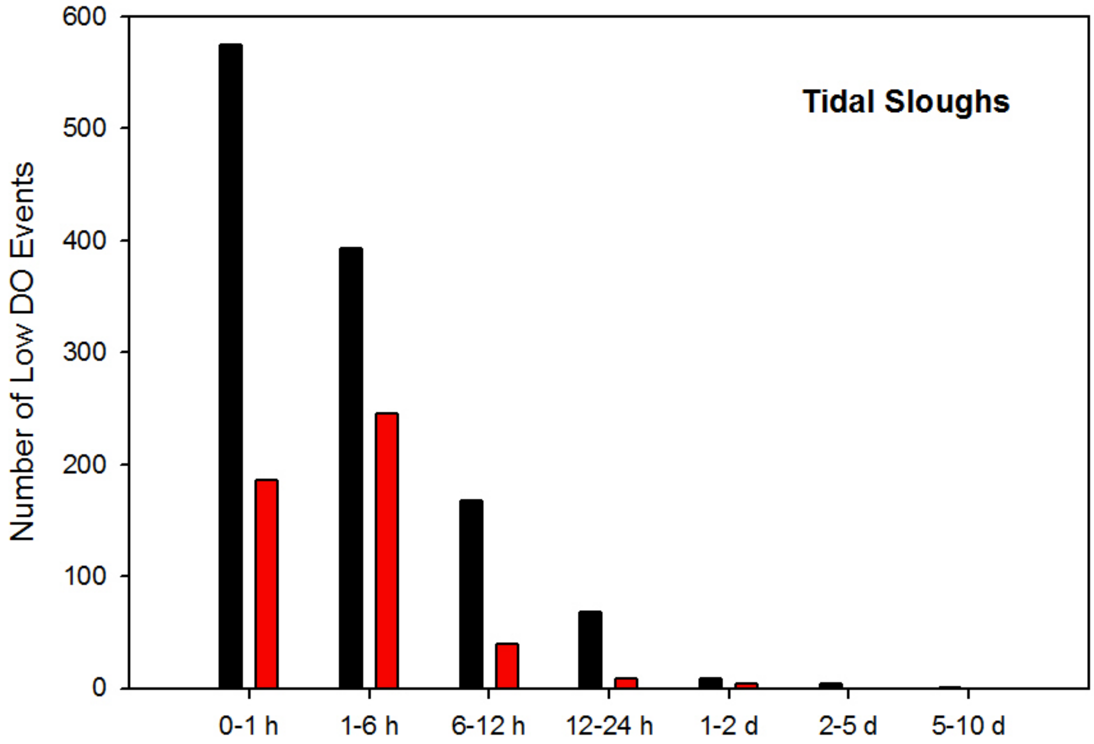

Figure 2. Duration of low-DO events in Lower South Bay tidal sloughs. [2].

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[2] Note that the event categories are mutually exclusive, i.e. 1-6 h events are not included in the count of 0-1 h events, and vice versa, i.e. a 6 day event does not encompass any shorter events

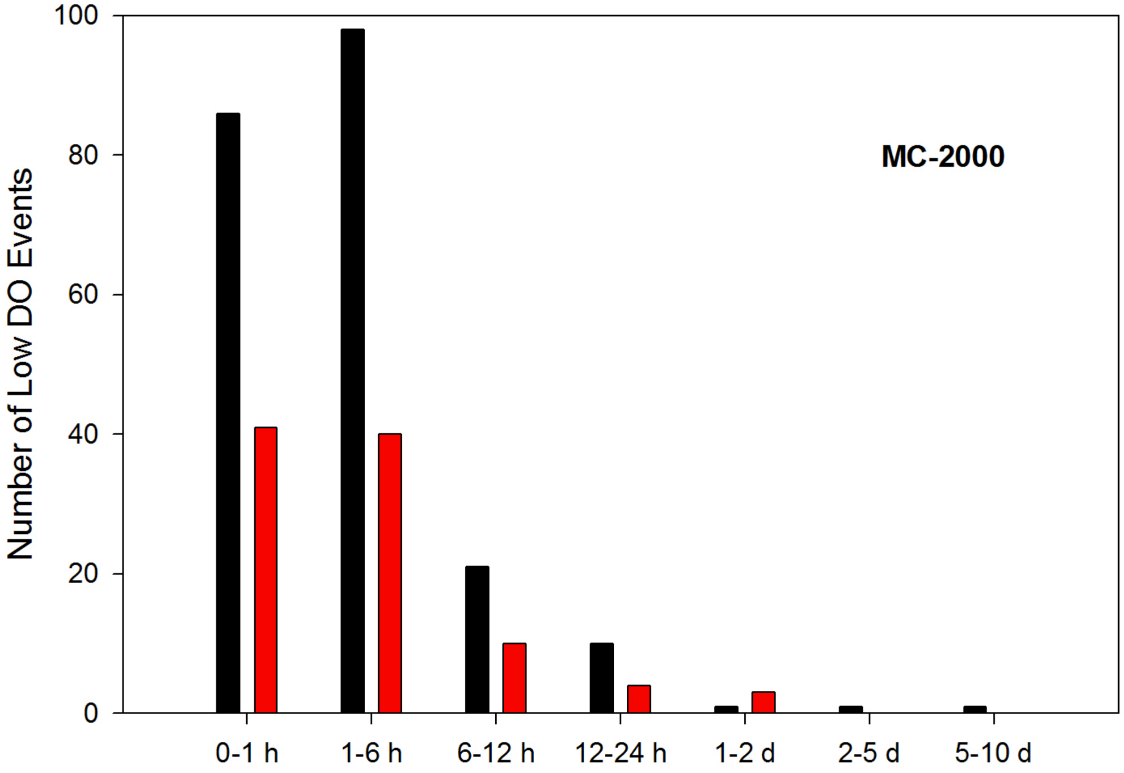

Figure 3. Duration of low-DO events at Station MC-2000 in Moffett Channel. [3].

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[3] Note that the event categories are mutually exclusive, i.e. 1-6 h events are not included in the count of 0-1 h events, and vice versa, i.e. a 6 day event does not encompass any shorter events

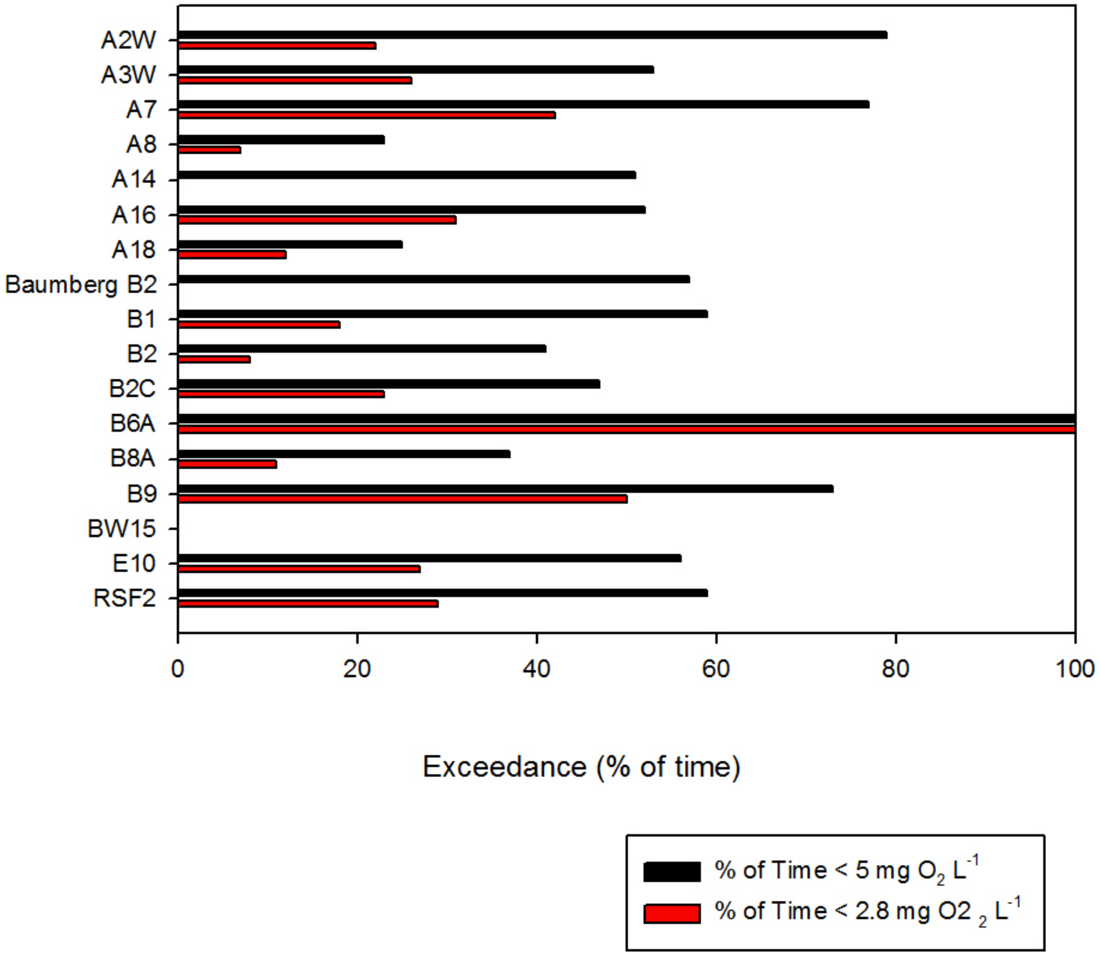

Figure 4. Former salt ponds, percent of DO records below thresholds (black: < 5 mg/L O_2, red: < 2.8 mg/L O_2).[4]

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[4] For former salt ponds with several monitoring sites, the results are aggregated across all sites within a pond. E.g., A3W aggregates the results for all A3W sites combined.

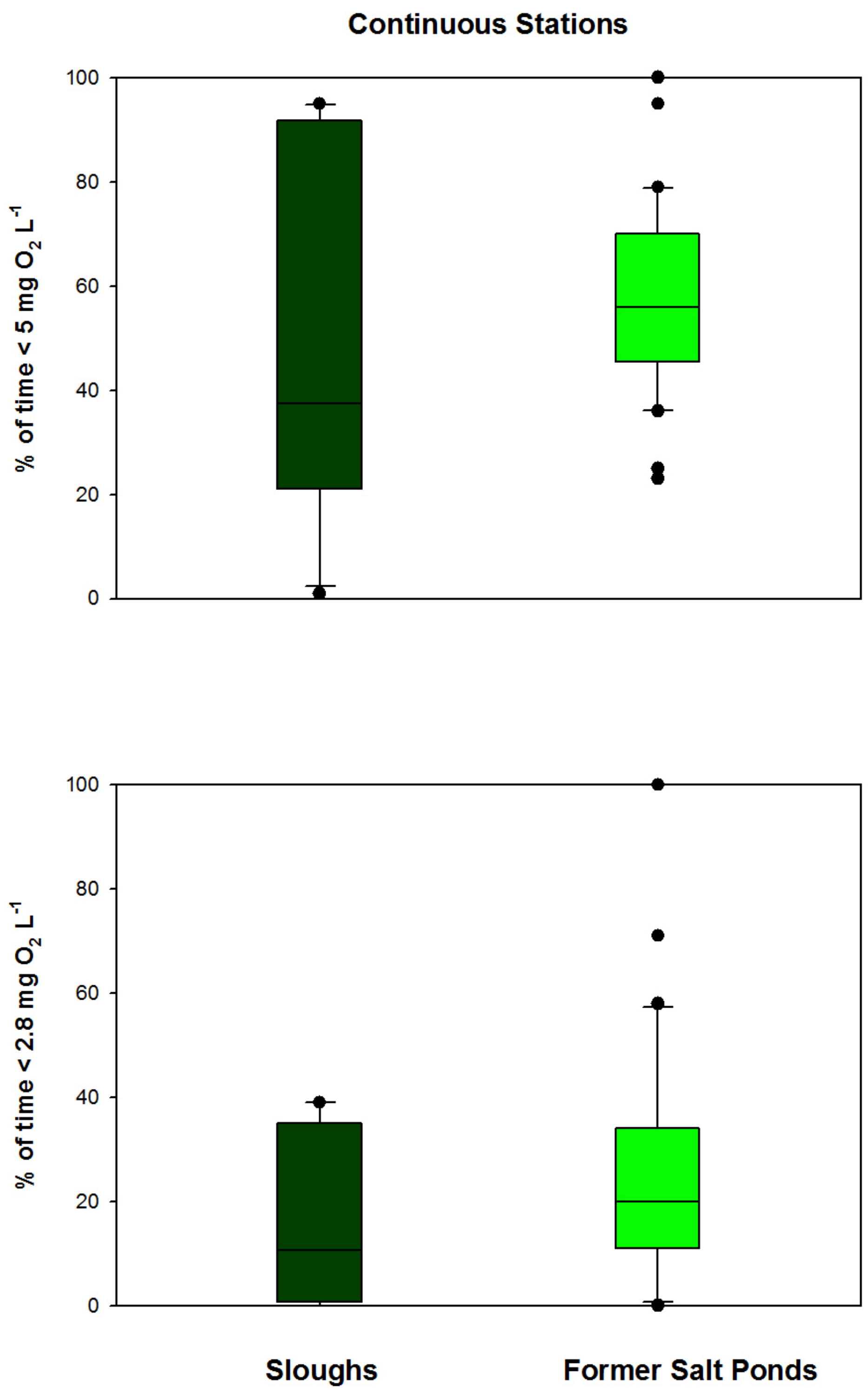

Figure 5. Frequency of exceedance (%) vs. habitat type (box plots). For calculating the mean (horizontal line inside each box), each station's frequency was considered as an individual value. Upper and lower edges of boxes are the upper and lower quartiles, and error bars represent ± 1 standard deviation.

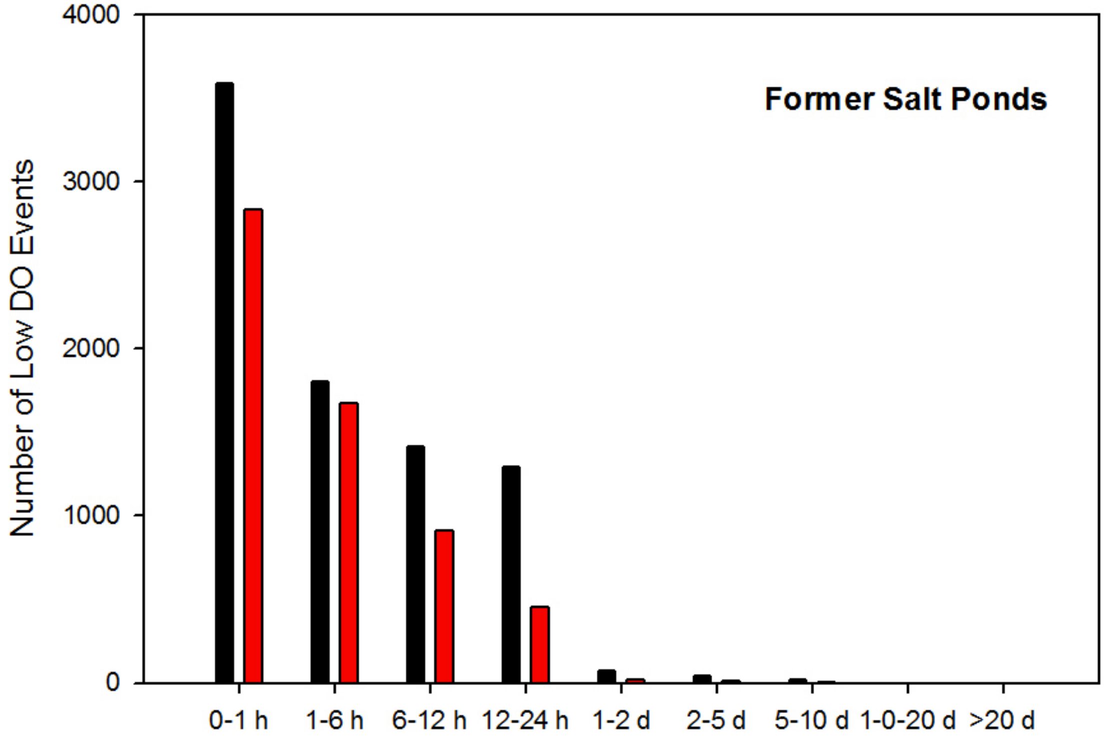

Figure 6. Duration of low-DO events in Lower South Bay former salt ponds. [5].

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[5] Note that the event categories are mutually exclusive, i.e. 1-6 h events are not included in the count of 0-1 h events, and vice versa, i.e. a 6 day event does not encompass any shorter events

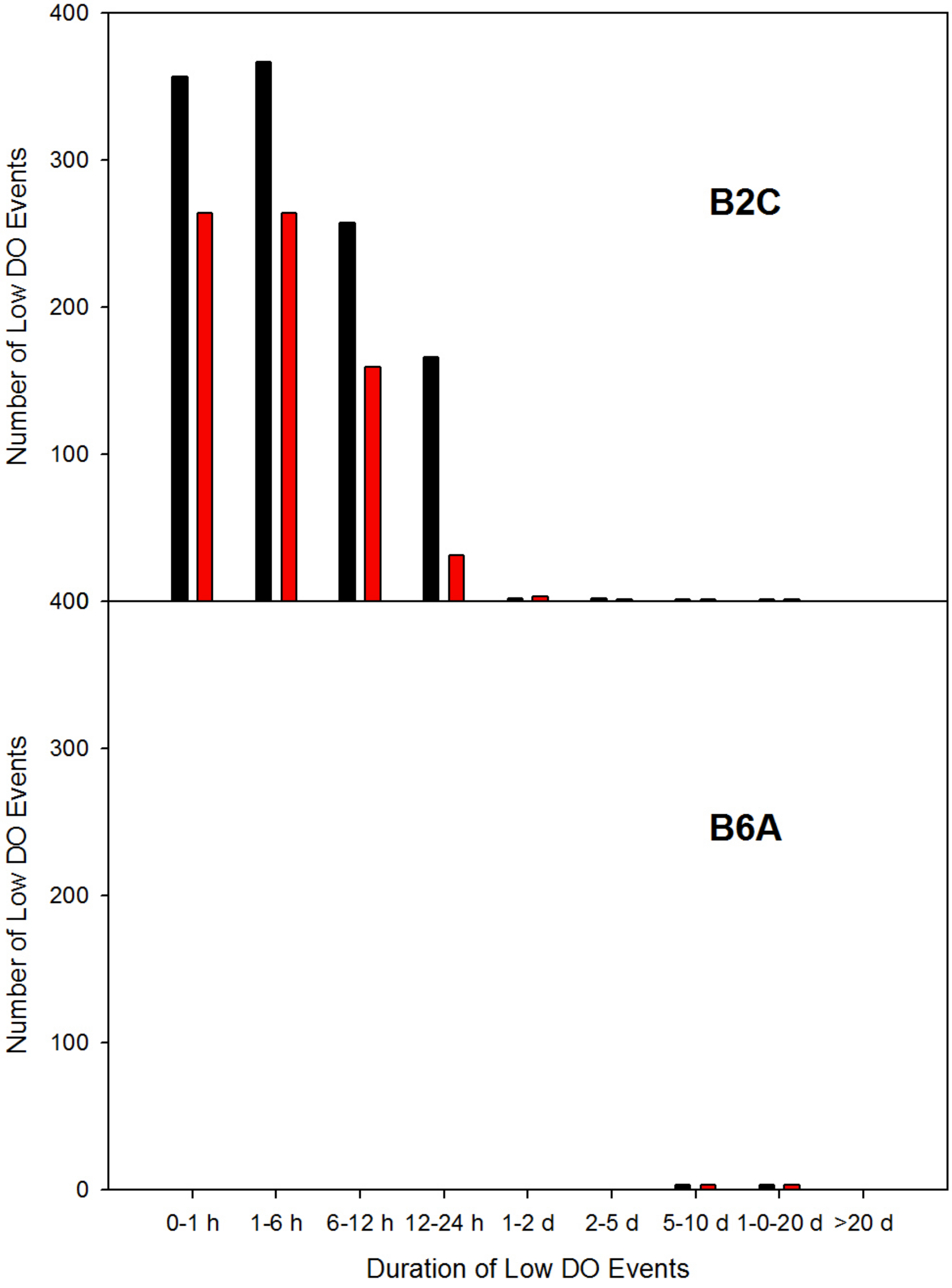
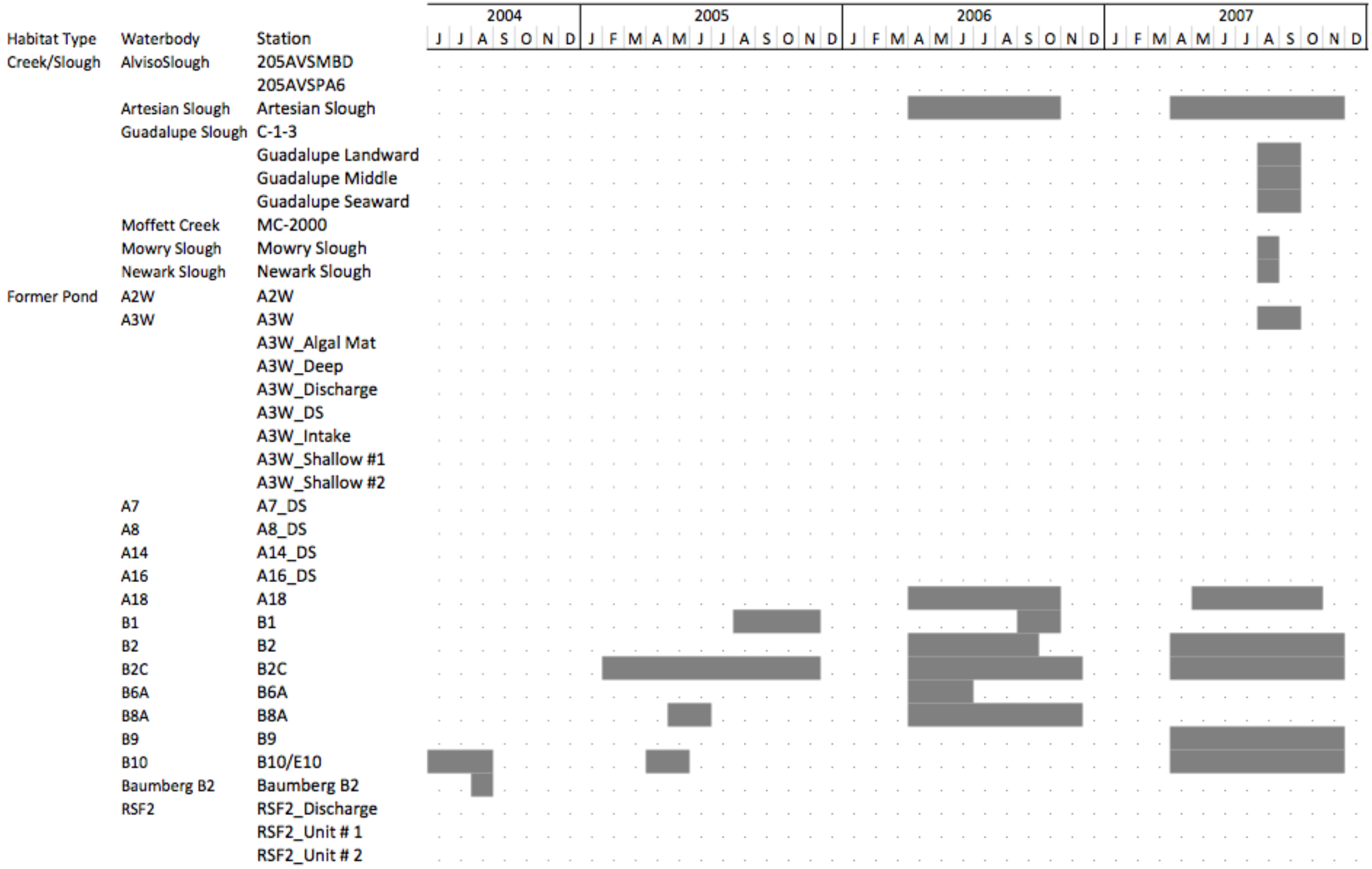


Figure 7. The four longest documented periods of continued hypoxic conditions occurred in Ponds B2C and 6A, each lasting more than 12 days.

Tables:

Table 1. Temporal extent of the available continuous datasets.



			2004							2005												2006												2007											
Habitat Type	Waterbody	Station	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D
Creek/Slough	AlvisoSlough	205AVSMBD																																											
		205AVSPA6																																											
	Artesian Slough	Artesian Slough																							■	■	■	■	■	■	■						■	■	■	■	■	■	■	■	
	Guadalupe Slough	C-1-3																																											
		Guadalupe Landward																																							■	■			
		Guadalupe Middle																																							■	■			
		Guadalupe Seaward																																							■	■			
	Moffett Creek	MC-2000																																											
	Mowry Slough	Mowry Slough																																							■				
	Newark Slough	Newark Slough																																							■				
Former Pond	A2W	A2W																																											
	A3W	A3W																																							■	■			
		A3W_Algal Mat																																											
		A3W_Deep																																											
		A3W_Discharge																																											
		A3W_DS																																											
		A3W_Intake																																											
		A3W_Shallow #1																																											
		A3W_Shallow #2																																											
	A7	A7_DS																																											
	A8	A8_DS																																											
	A14	A14_DS																																											
	A16	A16_DS																																											
	A18	A18																							■	■	■	■	■	■	■							■	■	■	■	■	■		
	B1	B1															■	■	■	■											■	■													
	B2	B2																							■	■	■	■	■	■							■	■	■	■	■	■	■	■	
	B2C	B2C									■	■	■	■	■	■	■	■	■	■					■	■	■	■	■	■	■	■					■	■	■	■	■	■	■	■	
	B6A	B6A																							■	■	■																		
	B8A	B8A												■	■										■	■	■	■	■	■	■	■													
	B9	B9																																			■	■	■	■	■	■	■	■	
	B10	B10/E10	■	■	■								■	■																							■	■	■	■	■	■	■	■	
	Baumberg B2	Baumberg B2			■																																								
	RSF2	RSF2_Discharge																																											
		RSF2_Unit # 1																																											
		RSF2_Unit # 2																																											

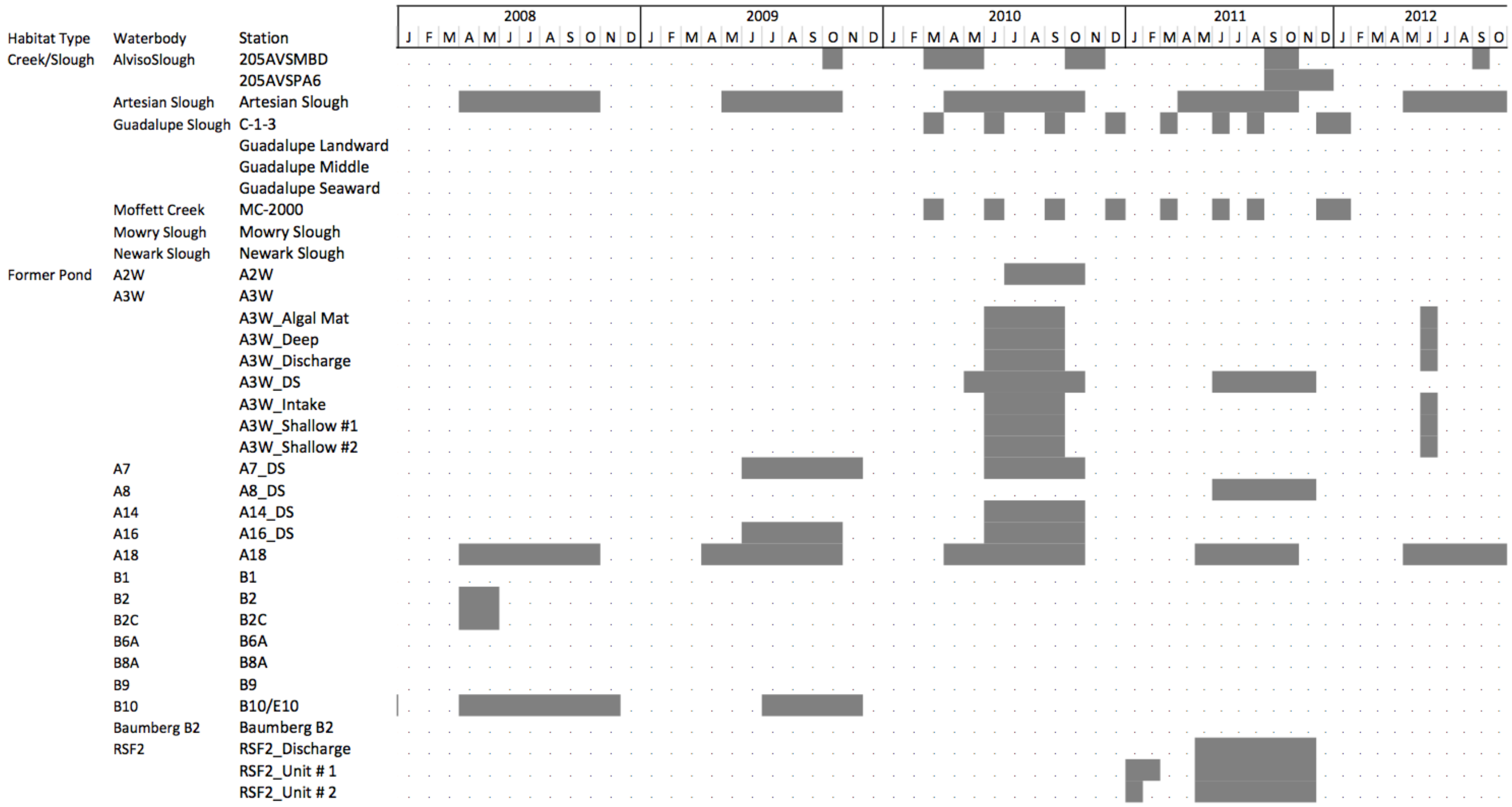

2008
2009
2010
2011
2012
J F M A M J J A S O N D
J F M A M J J A S O
Habitat Type
Waterbody
Station
Creek/Slough
AlvisoSlough
205AVSMBD
205AVSPA6
Artesian Slough
Artesian Slough
Guadalupe Slough
C-1-3
Guadalupe Landward
Guadalupe Middle
Guadalupe Seaward
Moffett Creek
MC-2000
Mowry Slough
Mowry Slough
Newark Slough
Newark Slough
Former Pond
A2W
A2W
A3W
A3W
A3W_Algal Mat
A3W_Deep
A3W_Discharge
A3W_DS
A3W_Intake
A3W_Shallow #1
A3W_Shallow #2
A7
A7_DS
A8
A8_DS
A14
A14_DS
A16
A16_DS
A18
A18
B1
B1
B2
B2
B2C
B2C
B6A
B6A
B8A
B8A
B9
B9
B10
B10/E10
Baumberg B2
Baumberg B2
RSF2
RSF2_Discharge
RSF2_Unit # 1
RSF2_Unit # 2

Table 2. Summary table (frequency of exceedances) for Lower South Bay subtidal habitat dissolved oxygen data.

Year	**Number of Records/Stations**	**% of Time < 2.8 mg O_2 L^{-1}**	**% of Time < 5 mg O_2 L^{-1}**
1993	8	0	0
1994	15	0	0
1995	15	0	0
1996	15	0	7
1997	15	0	0
1998	15	0	7
1999	13	0	0
2000	8	0	0
2001	10	0	0
2002	16	0	0
2003	15	0	0
2004	15	0	0
2005	16	0	0
2006	15	0	0
2007	8	0	0
2008	9	0	0
2009	9	0	0
Total	***217***	***0***	***1***

Table 3. Summary table (frequency of exceedances) for dissolved oxygen data collected at monitoring stations located in tidal sloughs and creeks of the Lower South Bay. See the data summary table in Appendix B for additional detail.

Station	*Data Type*	*Length of Record*	*Number of Records*	*% of Time < 2.8 mg O_2 L^{-1}*	*% of Time < 5 mg O_2 L^{-1}*
Alviso Slough					
205AVSMBD	Continuous	Sep 2009 - Sep 2012	22,752	5%	29%
205AVSPA6	Continuous	2011	7,549	1%	15%
Artesian Slough	Continuous	2006-12	117,670	0%	1%
Coyote Creek	Discrete	1994-2002	23	0%	44%
Guadalupe Slough					
C-1-3	Continuous Discrete	2010-2012 1994-2008	10,968 49	8% 8%	34% 41%
C-2-0	Discrete	2006-2008	26	12%	46%
C-3-0	Discrete	2006-2008	26	15%	54%
C-4-0	Discrete	2006-2008	27	0%	41%
C-4-4	Discrete	2006-2008	27	0%	15%
C-5	Discrete	2006-2008	27	0%	0%
Guadalupe Landward	Continuous	2007	2840	34%	94%
Guadalupe Middle	Continuous	2007	2834	38%	95%
Guadalupe Seaward	Continuous	2007	2853	34%	91%
Matadero Creek					
MC-1	Discrete	2009-2010	12	0%	0%

MC-2	Discrete	2009-2010	12	0%	0%
MC-3	Discrete	2009-2010	12	0%	8%
MC-4	Discrete	2009-2010	12	0%	17%
MC-5	Discrete	2009-2010	12	0%	8%
MC-6	Discrete	2009-2010	12	0%	17%
RSW-E1	Discrete	2009-2010	11	0%	0%
Moffett Channel					
MC-200	Discrete	2006-2008	26	0%	12%
MC-500	Discrete	2006-2008	26	0%	12%
MC-1000	Discrete	2006-2008	26	4%	27%
MC-2000	Continuous Discrete	2010-2012 2006-2008	10,952 26	13% 12%	33% 35%
Mowry Slough	Continuous	2011	400	1%	41%
Newark Slough	Continuous	2011	405	0%	23%

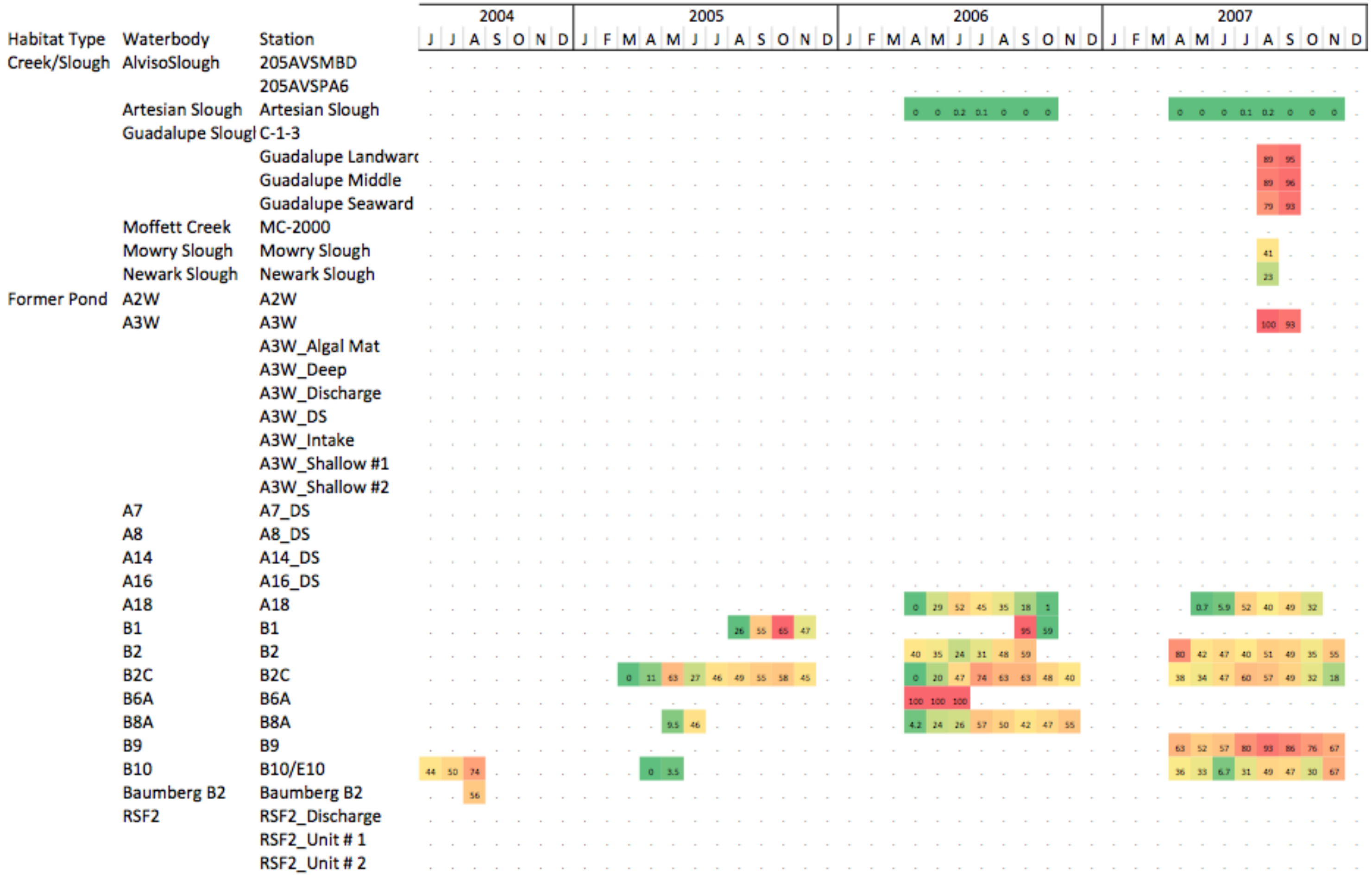
Table 4. Summary table (frequency of exceedances) for dissolved oxygen data collected at monitoring stations located in former salt ponds of the Lower South Bay.

Station	*Data Type*	*Length of Record*	*Number of Records*	*% of Time < 2.8 mg O_2 L^{-1}*	*% of Time < 5 mg O_2 L^{-1}*
A2W	**Continuous**	**2010**	**185**	**22%**	**79%**
A3W					
A3W_AlgalMat	Continuous	2010-2012	8484	34%	58%
A3W_Alviso	Continuous	2010-2011	94	13%	70%
A3W_Deep	Continuous	2010-2012	6936	17%	47%
A3W_Discharge	Continuous	2010-2012	8598	21%	61%
A3W_DS	Continuous	2010-2011	6797	22%	52%
A3W_Intake	Continuous	2010-2012	9346	11%	39%
A3W_Shallow#1	Continuous	2010-2012	8633	13%	36%
A3W_Shallow#2	Continuous	2010-2012	7675	58%	72%
Pond A3W	Continuous	2007	1873	71%	95%
A7					
A7_Alviso	Continuous	2010-2011	82	10%	57%
A7_DS	Continuous	2009-2010	13,294	42%	77%
A8					
A8_Alviso	Continuous	2011	10	30%	70%
A8_DS	Continuous	2011	11,560	7%	23%
A14					
A14_Alviso	Continuous	2010-2011	36	0%	56%

A14_DS	Continuous	2010	221	0%	50%
A16					
A16_Alviso	Continuous	2010-2011	60	13%	40%
A16_DS	Continuous	2009-2010	13,331	31%	53%
A18	Continuous	**2006-2012**	**116,804**	**12%**	**25%**
Baumberg B2	Continuous	2004	2275	0%	57%
B1	Continuous	2005-2006	11,074	18%	59%
B2	Continuous	2006-2008	32,699	8%	41%
B2C	Continuous	2005-2008	57,041	23%	47%
B6A	Continuous	2006	5846	100%	100%
B8A	Continuous	2005-2006	23,753	11%	37%
B9	Continuous	2007	16,752	50%	73%
BW15	Discrete	1997-2001	13	0%	0%
E10/(B10)	Continuous	2004-2009	45,295	27%	56%
RSF2					
RSF2_Discharge	Continuous	2011	20,186	34%	69%
RSF2_Unit #1	Continuous	2011	23,589	19%	53%
RSF2_Unit #2	Continuous	2011	15,286	37%	56%

Table 5. Graphic summary of percent exceedances of the 5 mg O_2 L^{-1} (a) and 2.8 mg O_2 L^{-1} (b) thresholds. The color gradient ranges from 0% (green) over 50% (yellow) to 100% exceedance (red). Only continuous (high-frequency) datasets are included.

5a.



			2004							2005											
Habitat Type	Waterbody	Station	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D
Creek/Slough	AlvisoSlough	205AVSMBD																			
		205AVSPA6																			
	Artesian Slough	Artesian Slough																			
	Guadalupe Sloug	C-1-3																			
		Guadalupe Landwar																			
		Guadalupe Middle																			
		Guadalupe Seaward																			
	Moffett Creek	MC-2000																			
	Mowry Slough	Mowry Slough																			
	Newark Slough	Newark Slough																			
Former Pond	A2W	A2W																			
	A3W	A3W																			
		A3W_Algal Mat																			
		A3W_Deep																			
		A3W_Discharge																			
		A3W_DS																			
		A3W_Intake																			
		A3W_Shallow #1																			
		A3W_Shallow #2																			
	A7	A7_DS																			
	A8	A8_DS																			
	A14	A14_DS																			
	A16	A16_DS																			
	A18	A18																			
	B1	B1															26	55	65	47	
	B2	B2																			
	B2C	B2C										0	11	63	27	46	49	55	58	45	
	B6A	B6A																			
	B8A	B8A												9.5	46						
	B9	B9																			
	B10	B10/E10	44	50	74								0	3.5							
	Baumberg B2	Baumberg B2			56																
	RSF2	RSF2_Discharge																			
		RSF2_Unit # 1																			
		RSF2_Unit # 2																			

			2006												2007											
Habitat Type	Waterbody	Station	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D
Creek/Slough	AlvisoSlough	205AVSMBD																								
		205AVSPA6																								
	Artesian Slough	Artesian Slough				0	0	0.2	0.1	0	0	0						0	0	0	0.1	0.2	0	0	0	
	Guadalupe Sloug	C-1-3																								
		Guadalupe Landwar																				89	95			
		Guadalupe Middle																				89	96			
		Guadalupe Seaward																				79	93			
	Moffett Creek	MC-2000																								
	Mowry Slough	Mowry Slough																				41				
	Newark Slough	Newark Slough																				23				
Former Pond	A2W	A2W																								
	A3W	A3W																				100	93			
		A3W_Algal Mat																								
		A3W_Deep																								
		A3W_Discharge																								
		A3W_DS																								
		A3W_Intake																								
		A3W_Shallow #1																								
		A3W_Shallow #2																								
	A7	A7_DS																								
	A8	A8_DS																								
	A14	A14_DS																								
	A16	A16_DS																								
	A18	A18				0	29	52	45	35	18	1							0.7	5.9	52	40	49	32		
	B1	B1									95	59														
	B2	B2				40	35	24	31	48	59							80	42	47	40	51	49	35	55	
	B2C	B2C				0	20	47	74	63	63	48	40					38	34	47	60	57	49	32	18	
	B6A	B6A				100	100	100																		
	B8A	B8A				4.2	24	26	57	50	42	47	55													
	B9	B9																63	52	57	80	93	86	76	67	
	B10	B10/E10																36	33	6.7	31	49	47	30	67	
	Baumberg B2	Baumberg B2																								
	RSF2	RSF2_Discharge																								
		RSF2_Unit # 1																								
		RSF2_Unit # 2																								

			2008											
Habitat Type	Waterbody	Station	J	F	M	A	M	J	J	A	S	O	N	D
Creek/Slough	AlvisoSlough	205AVSMBD												
		205AVSPA6												
	Artesian Slough	Artesian Slough				0	0.1	0.7	0.8	4	2.6	0.1		
	Guadalupe Sloug	C-1-3												
		Guadalupe Landwar												
		Guadalupe Middle												
		Guadalupe Seaward												
	Moffett Creek	MC-2000												
	Mowry Slough	Mowry Slough												
	Newark Slough	Newark Slough												
Former Pond	A2W	A2W												
	A3W	A3W												
		A3W_Algal Mat												
		A3W_Deep												
		A3W_Discharge												
		A3W_DS												
		A3W_Intake												
		A3W_Shallow #1												
		A3W_Shallow #2												
	A7	A7_DS												
	A8	A8_DS												
	A14	A14_DS												
	A16	A16_DS												
	A18	A18				0	0.5	6.2	65	82	76	34		
	B1	B1												
	B2	B2				6.3	37							
	B2C	B2C				19	37							
	B6A	B6A												
	B8A	B8A												
	B9	B9												
	B10	B10/E10				0	64	0	86	91	77	70	93	
	Baumberg B2	Baumberg B2												
	RSF2	RSF2_Discharge												
		RSF2_Unit # 1												
		RSF2_Unit # 2												

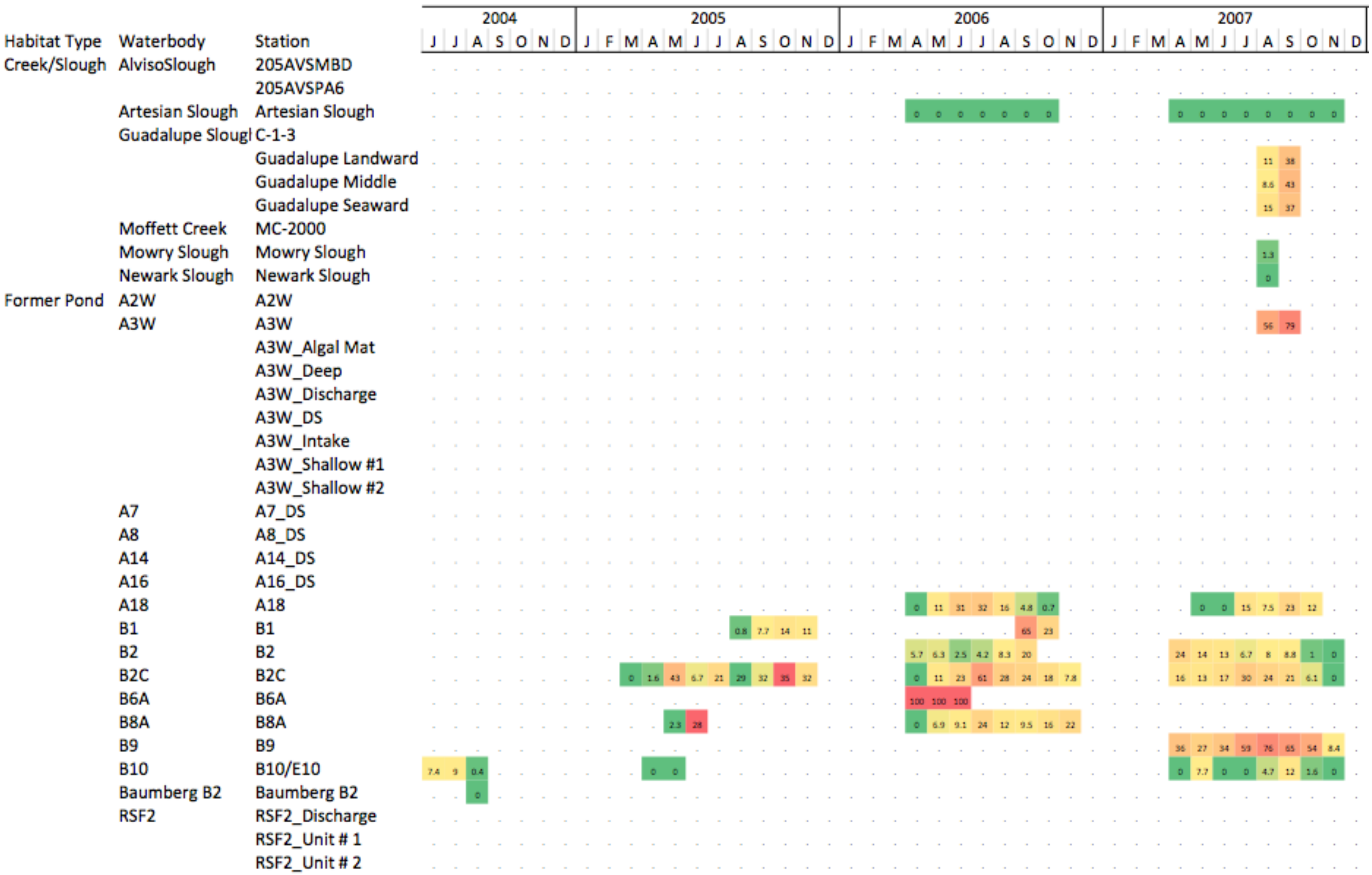
			2009											
Habitat Type	Waterbody	Station	J	F	M	A	M	J	J	A	S	O	N	D
Creek/Slough	AlvisoSlough	205AVSMBD										26		
		205AVSPA6												
	Artesian Slough	Artesian Slough					0	0	0	0.8	4.2	0.2		
	Guadalupe Sloug	C-1-3												
		Guadalupe Landwar												
		Guadalupe Middle												
		Guadalupe Seaward												
	Moffett Creek	MC-2000												
	Mowry Slough	Mowry Slough												
	Newark Slough	Newark Slough												
Former Pond	A2W	A2W												
	A3W	A3W												
		A3W_Algal Mat												
		A3W_Deep												
		A3W_Discharge												
		A3W_DS												
		A3W_Intake												
		A3W_Shallow #1												
		A3W_Shallow #2												
	A7	A7_DS						69	86	93	84	60	69	
	A8	A8_DS												
	A14	A14_DS												
	A16	A16_DS						33	43	65	75	50		
	A18	A18				0	1.2	0	19	68	55	18		
	B1	B1												
	B2	B2												
	B2C	B2C												
	B6A	B6A												
	B8A	B8A												
	B9	B9												
	B10	B10/E10							74	56	56	73	79	
	Baumberg B2	Baumberg B2												
	RSF2	RSF2_Discharge												
		RSF2_Unit # 1												
		RSF2_Unit # 2												

			2010											
Habitat Type	Waterbody	Station	J	F	M	A	M	J	J	A	S	O	N	D
Creek/Slough	AlvisoSlough	205AVSMBD			0.5	1	0					56	1.6	
		205AVSPA6												
	Artesian Slough	Artesian Slough				0	0	0.2	0.6	2.5	2.2	0.1		
	Guadalupe Sloug	C-1-3			0			63			91			19
		Guadalupe Landwar												
		Guadalupe Middle												
		Guadalupe Seaward												
	Moffett Creek	MC-2000			1.3			47			45			
	Mowry Slough	Mowry Slough												
	Newark Slough	Newark Slough												
Former Pond	A2W	A2W							100	75	100	37		
	A3W	A3W												
		A3W_Algal Mat						47	58	65	78			
		A3W_Deep						37	42	46	92			
		A3W_Discharge						52	50	66	75			
		A3W_DS					100	100	100	100	100			
		A3W_Intake						23	33	45	86			
		A3W_Shallow #1						16	36	50	85			
		A3W_Shallow #2						61	74	84	83			
	A7	A7_DS						0	50	65	100	50		
	A8	A8_DS												
	A14	A14_DS						25	75	25	67	77		
	A16	A16_DS						40	0	60	82	25		
	A18	A18				0	0	0	3.7	75	62	43		
	B1	B1												
	B2	B2												
	B2C	B2C												
	B6A	B6A												
	B8A	B8A												
	B9	B9												
	B10	B10/E10												
	Baumberg B2	Baumberg B2												
	RSF2	RSF2_Discharge												
		RSF2_Unit # 1												
		RSF2_Unit # 2												

			2011											
Habitat Type	Waterbody	Station	J	F	M	A	M	J	J	A	S	O	N	D
Creek/Slough	AlvisoSlough	205AVSMBD									63	65		
		205AVSPA6									18	28	3.1	0.4
	Artesian Slough	Artesian Slough				0	0.3	3.7	2.5	1	0.5	0.1		
	Guadalupe Sloug	C-1-3			15			51		14				22
		Guadalupe Landwar												
		Guadalupe Middle												
		Guadalupe Seaward												
	Moffett Creek	MC-2000			84			35		41				7.7
	Mowry Slough	Mowry Slough												
	Newark Slough	Newark Slough												
Former Pond	A2W	A2W												
	A3W	A3W												
		A3W_Algal Mat												
		A3W_Deep												
		A3W_Discharge												
		A3W_DS						35	37	41	65	74	100	
		A3W_Intake												
		A3W_Shallow #1												
		A3W_Shallow #2												
	A7	A7_DS												
	A8	A8_DS						3.9	7.6	21	32	35	0	
	A14	A14_DS												
	A16	A16_DS												
	A18	A18				0	0	0	0	0	3.1	45		
	B1	B1												
	B2	B2												
	B2C	B2C												
	B6A	B6A												
	B8A	B8A												
	B9	B9												
	B10	B10/E10												
	Baumberg B2	Baumberg B2												
	RSF2	RSF2_Discharge					4.7	44	74	83	88	92	100	
		RSF2_Unit # 1	1.4	9.6			27	52	63	68	70	54	100	
		RSF2_Unit # 2	41				6	38	50	67	70	77	98	

			2012									
Habitat Type	Waterbody	Station	J	F	M	A	M	J	J	A	S	O
Creek/Slough	AlvisoSlough	205AVSMBD									60	
		205AVSPA6										
	Artesian Slough	Artesian Slough					0.1	0.1	0.2	1.7	0	0
	Guadalupe Sloug	C-1-3	5.9									
		Guadalupe Landwar										
		Guadalupe Middle										
		Guadalupe Seaward										
	Moffett Creek	MC-2000	17									
	Mowry Slough	Mowry Slough										
	Newark Slough	Newark Slough										
Former Pond	A2W	A2W										
	A3W	A3W										
		A3W_Algal Mat						24				
		A3W_Deep						0				
		A3W_Discharge						72				
		A3W_DS										
		A3W_Intake						34				
		A3W_Shallow #1						8.9				
		A3W_Shallow #2						33				
	A7	A7_DS										
	A8	A8_DS										
	A14	A14_DS										
	A16	A16_DS										
	A18	A18					0	0	0.1	5.3	12	59
	B1	B1										
	B2	B2										
	B2C	B2C										
	B6A	B6A										
	B8A	B8A										
	B9	B9										
	B10	B10/E10										
	Baumberg B2	Baumberg B2										
	RSF2	RSF2_Discharge										
		RSF2_Unit # 1										
		RSF2_Unit # 2										

5b.

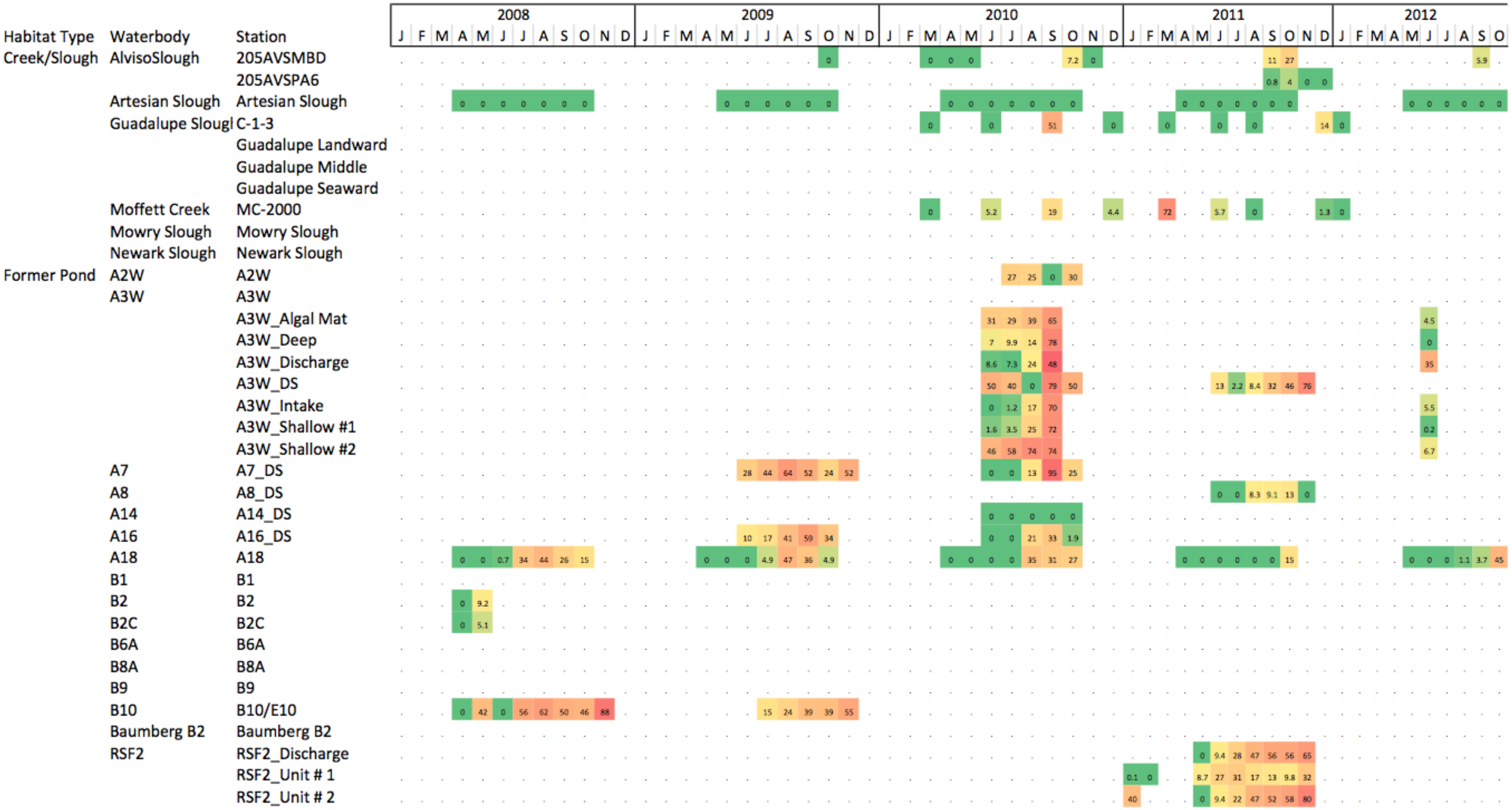


			2004						
Habitat Type	Waterbody	Station	J	J	A	S	O	N	D
Creek/Slough	AlvisoSlough	205AVSMBD							
		205AVSPA6							
	Artesian Slough	Artesian Slough							
	Guadalupe Sloug	C-1-3							
		Guadalupe Landward							
		Guadalupe Middle							
		Guadalupe Seaward							
	Moffett Creek	MC-2000							
	Mowry Slough	Mowry Slough							
	Newark Slough	Newark Slough							
Former Pond	A2W	A2W							
	A3W	A3W							
		A3W_Algal Mat							
		A3W_Deep							
		A3W_Discharge							
		A3W_DS							
		A3W_Intake							
		A3W_Shallow #1							
		A3W_Shallow #2							
	A7	A7_DS							
	A8	A8_DS							
	A14	A14_DS							
	A16	A16_DS							
	A18	A18							
	B1	B1							
	B2	B2							
	B2C	B2C							
	B6A	B6A							
	B8A	B8A							
	B9	B9							
	B10	B10/E10	7.4	9	0.4				
	Baumberg B2	Baumberg B2			0				
	RSF2	RSF2_Discharge							
		RSF2_Unit # 1							
		RSF2_Unit # 2							

			2005											
Habitat Type	Waterbody	Station	J	F	M	A	M	J	J	A	S	O	N	D
Creek/Slough	AlvisoSlough	205AVSMBD												
		205AVSPA6												
	Artesian Slough	Artesian Slough												
	Guadalupe Sloug	C-1-3												
		Guadalupe Landward												
		Guadalupe Middle												
		Guadalupe Seaward												
	Moffett Creek	MC-2000												
	Mowry Slough	Mowry Slough												
	Newark Slough	Newark Slough												
Former Pond	A2W	A2W												
	A3W	A3W												
		A3W_Algal Mat												
		A3W_Deep												
		A3W_Discharge												
		A3W_DS												
		A3W_Intake												
		A3W_Shallow #1												
		A3W_Shallow #2												
	A7	A7_DS												
	A8	A8_DS												
	A14	A14_DS												
	A16	A16_DS												
	A18	A18												
	B1	B1								0.8	7.7	14	11	
	B2	B2												
	B2C	B2C			0	1.6	43	6.7	21	29	32	35	32	
	B6A	B6A												
	B8A	B8A					2.3	28						
	B9	B9												
	B10	B10/E10				0	0							
	Baumberg B2	Baumberg B2												
	RSF2	RSF2_Discharge												
		RSF2_Unit # 1												
		RSF2_Unit # 2												

			2006											
Habitat Type	Waterbody	Station	J	F	M	A	M	J	J	A	S	O	N	D
Creek/Slough	AlvisoSlough	205AVSMBD												
		205AVSPA6												
	Artesian Slough	Artesian Slough				0	0	0	0	0	0	0		
	Guadalupe Sloug	C-1-3												
		Guadalupe Landward												
		Guadalupe Middle												
		Guadalupe Seaward												
	Moffett Creek	MC-2000												
	Mowry Slough	Mowry Slough												
	Newark Slough	Newark Slough												
Former Pond	A2W	A2W												
	A3W	A3W												
		A3W_Algal Mat												
		A3W_Deep												
		A3W_Discharge												
		A3W_DS												
		A3W_Intake												
		A3W_Shallow #1												
		A3W_Shallow #2												
	A7	A7_DS												
	A8	A8_DS												
	A14	A14_DS												
	A16	A16_DS												
	A18	A18				0	11	31	32	16	4.8	0.7		
	B1	B1									65	23		
	B2	B2				5.7	6.3	2.5	4.2	8.3	20			
	B2C	B2C				0	11	23	61	28	24	18	7.8	
	B6A	B6A				100	100	100						
	B8A	B8A				0	6.9	9.1	24	12	9.5	16	22	
	B9	B9												
	B10	B10/E10												
	Baumberg B2	Baumberg B2												
	RSF2	RSF2_Discharge												
		RSF2_Unit # 1												
		RSF2_Unit # 2												

			2007											
Habitat Type	Waterbody	Station	J	F	M	A	M	J	J	A	S	O	N	D
Creek/Slough	AlvisoSlough	205AVSMBD												
		205AVSPA6												
	Artesian Slough	Artesian Slough				0	0	0	0	0	0	0	0	
	Guadalupe Sloug	C-1-3												
		Guadalupe Landward								11	38			
		Guadalupe Middle								8.6	43			
		Guadalupe Seaward								15	37			
	Moffett Creek	MC-2000												
	Mowry Slough	Mowry Slough								1.3				
	Newark Slough	Newark Slough								0				
Former Pond	A2W	A2W												
	A3W	A3W								56	79			
		A3W_Algal Mat												
		A3W_Deep												
		A3W_Discharge												
		A3W_DS												
		A3W_Intake												
		A3W_Shallow #1												
		A3W_Shallow #2												
	A7	A7_DS												
	A8	A8_DS												
	A14	A14_DS												
	A16	A16_DS												
	A18	A18					0	0	15	7.5	23	12		
	B1	B1												
	B2	B2				24	14	13	6.7	8	8.8	1	0	
	B2C	B2C				16	13	17	30	24	21	6.1	0	
	B6A	B6A												
	B8A	B8A												
	B9	B9				36	27	34	59	76	65	54	8.4	
	B10	B10/E10				0	7.7	0	0	4.7	12	1.6	0	
	Baumberg B2	Baumberg B2												
	RSF2	RSF2_Discharge												
		RSF2_Unit # 1												
		RSF2_Unit # 2												



			2008												2009												2010												2011												2012									
Habitat Type	Waterbody	Station	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O	N	D	J	F	M	A	M	J	J	A	S	O
Creek/Slough	AlvisoSlough	205AVSMBD																						0					0	0	0					7.2	0										11	27											5.9	
		205AVSPA6																																													0.8	4	0	0										
	Artesian Slough	Artesian Slough				0	0	0	0	0	0	0							0	0	0	0	0	0						0	0	0	0	0	0	0						0	0	0	0	0	0	0							0	0	0	0	0	0
	Guadalupe Slougl	C-1-3																											0			0			51			0			0			0		0				14	0									
		Guadalupe Landward																																																										
		Guadalupe Middle																																																										
		Guadalupe Seaward																																																										
	Moffett Creek	MC-2000																											0			5.2			19			4.4			72			5.7		0				1.3	0									
	Mowry Slough	Mowry Slough																																																										
	Newark Slough	Newark Slough																																																										
Former Pond	A2W	A2W																															27	25	0	30																								
	A3W	A3W																																																										
		A3W_Algal Mat																														31	29	39	65																					4.5				
		A3W_Deep																														7	9.9	14	78																					0				
		A3W_Discharge																														8.6	7.3	24	48																					35				
		A3W_DS																														50	40	0	79	50								13	2.2	8.4	32	46	76											
		A3W_Intake																														0	1.2	17	70																					5.5				
		A3W_Shallow #1																														1.6	3.5	25	72																					0.2				
		A3W_Shallow #2																														46	58	74	74																					6.7				
	A7	A7_DS																		28	44	64	52	24	52							0	0	13	95	25																								
	A8	A8_DS																																										0	0	8.3	9.1	13	0											
	A14	A14_DS																														0	0	0	0	0																								
	A16	A16_DS																		10	17	41	59	34								0	0	21	33	1.9																								
	A18	A18				0	0	0.7	34	44	26	15						0	0	0	4.9	47	36	4.9						0	0	0	0	35	31	27						0	0	0	0	0	0	15							0	0	0	1.1	3.7	45
	B1	B1																																																										
	B2	B2				0	9.2																																																					
	B2C	B2C				0	5.1																																																					
	B6A	B6A																																																										
	B8A	B8A																																																										
	B9	B9																																																										
	B10	B10/E10				0	42	0	56	62	50	46	88								15	24	39	39	55																																			
	Baumberg B2	Baumberg B2																																																										
	RSF2	RSF2_Discharge																																									0	9.4	28	47	56	56	65											
		RSF2_Unit # 1																																					0.1	0			8.7	27	31	17	13	9.8	32											
		RSF2_Unit # 2																																					40				0	9.4	22	47	52	58	80											

Appendix A. Data Summary Table

Agency/Source	Dataset	Number of Records	Type of Sampling	Sampling Frequency	Time Period of Record	Locations	Total Stations	Other Parameters	Notes
SCVURPPP	MRP Creek Data (BAASMA RMC, Fish - Water Quality: YSI sonde)	>100,000	Continuous	every 2-5 min,1-9 events/station, 4 days up to 3 weeks/event	Sep 2009 - Sep 2012	Alviso Slough, Guadalupe River	9	General WQ	RWQCB2_ContMon_v1.0
Palo Alto	Ammonia Characterization Study	~100	Discrete	quarterly/monthly	Nov 2009 - Oct 2010	Matadero Creek and Bay discharge points	13	Field measurements, ammonia	
SFEI	RMP Cruise Data	955	Discrete	3x/yr (early 90s) to annually (present)	1994 - 2010	Lower South Bay, South Bay	113	Other ancillary parameters	
San Jose	Pond A18 WDR Annual Self-Monitoring	~250,000	Continuous	every 15 min (late April/May thru Oct)	2006 - 2012	Artesian Slough, Pond A18	2	Temp, salinity, pH	
CDFW	South Bay Salt Ponds	~150,000	Continuous	every 15 min, 1-3 events per station, 3 months/event (Aug-Nov)	2005 - 2009	B1, B2, B2C, B6A, B8A, B10, E10	7	SC, salinity, Temp, pH, tide	DO and LDO data, raw data and QA/QC'd data available; median/event calculated; for some datasets daily, weekly, and seasonal mean values are calculated and summarized
CDFW?	South Bay Salt Ponds discharge sampling	~100,000	Continuous	every 15 min	2009-2012 (seasonal, summer/fall)	A2W, A3W, A7, A14, A16,A 19, RSF2	11	SC, salinity, Temp, pH, tide	
CDFW?	South Bay Salt Ponds 2010 discharge	~1,100	Continuous	every 1 min, 1x12 min event/week	Jun-Oct 2010	A2W, A3W, A7, A14,	5	SC, salinity, Temp, pH, tide	

	sampling					A16			
CDFW?	South Bay Salt Ponds Alviso Slough RW sampling	~120	Discrete	monthly	Jun-Oct 2010, Aug & Oct 2011	A3W, A7, A14, A17	~250	SC, salinity, Temp, pH, tide	
Sunnyvale	Receiving Water Ammonia Characterization Study	~20,000	Continuous	every 15 min (14 days/quarter, 8 deployments)	2010-11	Moffett Channel, Guadalupe Slough	2	Salinity, Temp, pH	
SWAMP	R2 South Bay Continuous Monitoring	~20,000	Continuous	every 15 min (1-4 deployments/station, 4d-1mo/deployment)	2002-2010	Coyote Creek, Permanente Creek Steven Creek	7	Temp, pH, SC, turbidity	
USGS	Slough and pond data	~17,500	Continuous	August: every 15 min, 4d-deployment September: every 7.5 min, 13d-deployment	2007	Newark, Mowry, and Guadalupe sloughs; Pond A3W	6	Temp, pH, SC	