



The interannual variabilities of chlorophyll and nutrients in San Francisco Bay: a modeling study

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Abstract

San Francisco Bay (SFB) is a complex ecosystem that has been heavily impacted by human activities. It has experienced strong year-to-year variations in physical, chemical, and biological conditions, due to both natural climate variation and human activities. There is a need to investigate this long-term variation, with a focus on management practices and needs. The study aims to construct a coupled hydrodynamic and ecosystem model SCHISM/CoSiNE, to investigate interannual variability of chlorophyll and nutrient dynamics in SFB over the 10-year period of 2005–2014. The coupled SCHISM/CoSiNE model captures the long-term observations well. It shows that high chlorophyll concentrations in South Bay persist during warm months, while chlorophyll in North Bay is low and has strong interannual variation. Nutrient concentrations are higher in North Bay where large river outflow influences their distribution and interannual variation, and in lower South Bay. The model results show that phytoplankton blooms in Suisun Bay tend to occur when flow rates are between 100 and 250 m³/s and ammonium concentration is in the range of 1–2 mmol/m³. Model sensitivity experiments show that benthic grazing can potentially reduce phytoplankton biomass, and its effect on chlorophyll concentration is modulated by river outflow. We demonstrate the potential of this open-source model for exploring water quality conditions and options for river flow management of SFB.

Keywords San Francisco Bay · Interannual variabilities · Ecosystem modeling · Chlorophyll · Nutrient flux

Key points

- The biogeochemical model captures the interannual variabilities in San Francisco Bay with high model skills.
- Analyses show that river flow plays an important role in the Bay ecosystem, along with benthic grazing and nutrients.

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1 Introduction

San Francisco Bay (SFB) is the largest estuary on the U.S. West Coast. It provides habitat for many birds as well as serving as a nursery ground for juvenile fish and invertebrates including shellfish (Warnock and Takekawa 1995). Unlike many other coastal/estuarine systems that are experiencing heavy eutrophication problems such as high chlorophyll and low dissolved oxygen (Conley et al. 2009; Kemp et al. 2005; Rabalais et al. 2002), the phytoplankton biomass (and primary production) in SFB remains low despite high concentrations of nutrients: N, Si, and P (Cloern and Jassby 2012; Dugdale et al. 2013, 2016; Glibert et al. 2014a; Liu et al. 2018), and SFB has been described as a high-nutrient low-chlorophyll ecosystem (Cloern 2001).

The cause of this low-chlorophyll, low-productivity phenomenon in SFB is open to debate. Currently, there are several working hypotheses, including the following: (1) the poor light conditions related to the high concentration of suspended particulate matter (SPM) (Cloern 1987); (2) benthic grazing by the invasive clam *Potamocorbula amurensis* that entered SFB in the 1970s (Alpine and Cloern 1992; Carlton et al. 1990; Cloern 1982; Dugdale et al. 2016; Lucas et al. 2016; Nichols et al. 1990; Prins et al. 1997); (3) short residence time and high flow (Glibert et al. 2014a; Lucas et al. 2009; Wang et al. 2019); and (4) anthropogenic ammonium (NH_4) that represses phytoplankton nitrate (NO_3) uptake (Cochlan and Harrison 1991; Conway 1977; Dortch 1990) such that the larger pool of dissolved inorganic nitrogen (DIN) in SFB (i.e., NO_3) is not accessed fully and chlorophyll concentration is held low (Dugdale et al. 2013; Parker et al. 2012a; Parker et al. 2012b).

The role of NH_4 repression on NO_3 uptake was proposed by Yoshiyama and Sharp (2006) to explain the low productivity in the Delaware Estuary and by Waiser et al. (2011) for the low productivity in Wascana Creek, Canada. In SFB, this phenomenon is found to occur both in the field and experimental enclosures filled with SFB water (Dugdale et al. 2007; Glibert et al. 2014a, b; 2016; Parker et al. 2012a; b; Wilkerson et al. 2015). In addition, the interaction between NH_4 and NO_3 is common in higher plants and algae (Britto and Kronzucker 2002). The way that high NH_4 concentration can limit phytoplankton growth has been comprehensively investigated and well documented in the literatures (Cochlan and Harrison 1991; Conway 1977; Cresswell and Syrett 1979; Dortch 1990; Glibert et al. 2016; Nakamura 1985; Syrett and Morris 1963).

SFB is composed of several subembayments (Fig. 1), including South Bay, Central Bay as well as Suisun Bay and San Pablo Bay in northern SFB. In this paper, we denote the combination of Suisun Bay and San Pablo Bay as North Bay. With large fresh water input from the Sacramento and San Joaquin rivers, North Bay is partially mixed (Cloern 1987). In contrast, South Bay is a well-mixed lagoon-type estuary

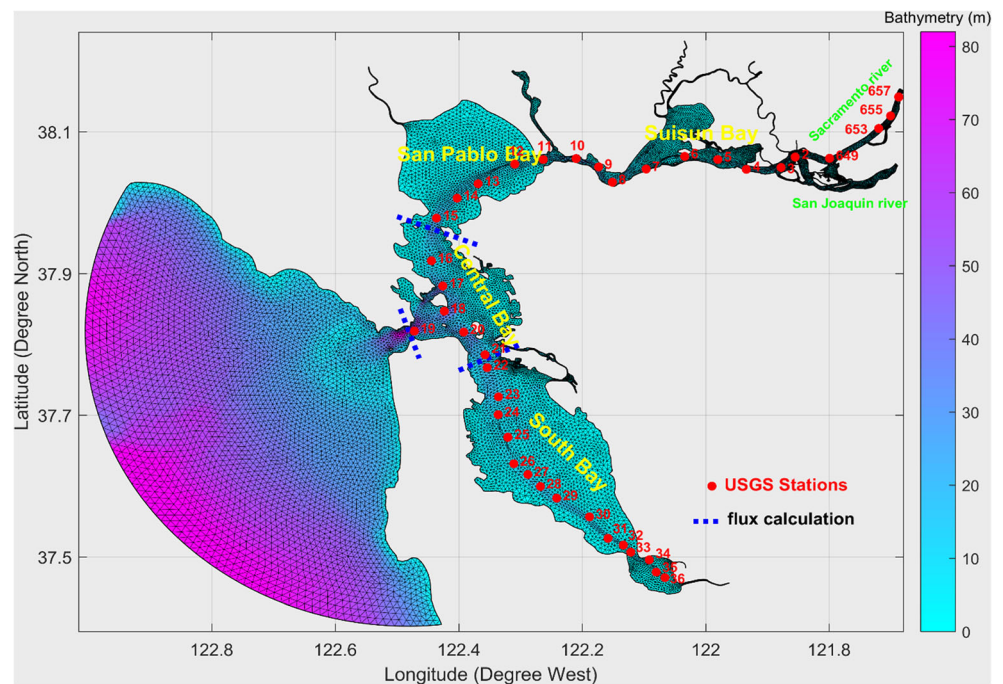
with very limited fresh water input (Cloern 1987). From Central Bay to North Bay, gravitational circulation develops as a result of the density gradient between the salty water close to the coastal ocean and brackish water close to the rivers (Cloern 1987).

Fresh water input into SFB has a strong influence on the hydrology, nutrient, and phytoplankton dynamics in SFB (Cloern 1996; Dugdale et al. 2013; Kimmerer 2002; Novick and Senn 2014). For the two largest rivers in the north, namely the Sacramento River and San Joaquin River, average river discharges are about 800 and 150 m^3/s , respectively (U.S. Geological Survey 2014). In the other areas of SFB, a small amount of fresh water discharge is mainly from local streams and creeks (Liu et al. 2018; Novick and Senn 2014). Normally, the river outflow (sum of Sacramento and San Joaquin flows) peaks in winter and spring, and diminishes in summer and fall with outflow rates varying widely from year to year. In wet years, the average river outflow can reach up to 2000 m^3/s ; however, in dry years, it can drop below 200 m^3/s (U.S. Geological Survey 2014). How river outflow affects the interannual variability of the SFB ecosystem, especially the lower trophic levels, is a goal of this research.

There are several sources of nutrient loads into SFB: Waste Water Treatment Plants (WWTPs), rivers, watershed runoff, atmospheric deposition, and oceanic input through the Golden Gate (Liu et al. 2018; Novick and Senn 2014; Sobota et al. 2009, 2011). Among these sources, watershed runoff poses a large uncertainty to the total nutrient budget of the bay as there are no accurate estimates, although this uncertainty has been suggested to be as large as $7.71 \times 10^5 \text{ mol}[\text{N}]/\text{day}$ (Novick and Senn 2014; Sobota et al. 2009, 2011). Atmospheric deposition is negligible in SFB (Novick and Senn 2014). Oceanic nutrient sources are normally dominated by terrigenous loading inside SFB, resulting in a net nutrient flux out of the bay through the Golden Gate, although it may become important under upwelling events in the outer bay (Largier and Stacey 2014). Here, we highlight the loads from the WWTPs and river sources of nutrients. On average, SFB receives about $1.89 \times 10^6 \text{ m}^3/\text{day}$ waste water from the WWTPs (Novick and Senn 2014). The associated nitrogen loads are mostly in the form of NH_4 as few WWTPs practice nitrification (Dugdale et al. 2013), except those in South Bay that discharge NO_3 as a result. The annual mean NH_4 load to SFB is estimated to be $2.45 \times 10^6 \text{ mol}[\text{N}]/\text{day}$, while the annual mean NO_3 load is estimated to be $8.21 \times 10^5 \text{ mol}[\text{N}]/\text{day}$ (Novick and Senn 2014). The nutrient load delivered by the major rivers is estimated to be $3.79 \times 10^5 \text{ mol}[\text{NH}_4]/\text{day}$ and $7.43 \times 10^5 \text{ mol}[\text{NO}_3]/\text{day}$ based on river flow and nutrient concentration. Since the river outflow varies greatly, the associated nutrient load is also highly variable, whereas the load from the WWTPs is fairly constant.

To achieve a comprehensive understanding of the SFB ecosystem and address the different hypotheses for the low-

Fig. 1 Model grid for San Francisco Bay. The unstructured grid is composed of triangles with coarser resolution (1 km) in the coastal region and finer resolution inside the bay. The shading represents bathymetry. The red dots are the USGS water quality stations from Rio Vista (657) in the north to Coyote Creek in the south. The dotted blue lines denote the transects used for nutrient fluxes calculation



chlorophyll condition, a number of models have been applied to evaluate different aspects of the SFB ecosystem (MacWilliams et al. 2016). Gross et al. (2009) used the proprietary model TRIM to investigate SFB hydrodynamics. As part of the CASCaDe 2 project in SFB, Martyr-Koller et al. (2017) applied the D-Flow FM hydrodynamic model component of the Delft3D model to investigate tidal, seasonal and annual dynamics of water levels, river flows and salinity under historical environmental and infrastructural conditions; and Knowles et al. (2018) used numerical models to project managed flows in the system. Chao et al. (2017b) used the semi-implicit cross-scale hydroscience integrated system (SCHISM) model to conduct a numerical hydrodynamic simulation of interannual variability of temperature and salinity patterns in different regions of SFB, and revealed the importance of atmospheric forcing on the heat budget and fresh water discharge on the salt budget.

Dugdale et al. (2013) used a simple biogeochemical box model to investigate flow (residence time) and nutrient impacts on phytoplankton, demonstrating that high biomass could only be achieved when NH_4 concentration was within a range determined by the flow and discharge of the WWTP effluents (Dugdale et al. 2012). Numerical models have also been employed to study the effects of benthic grazing on phytoplankton communities (Kimmerer and Thompson 2014; Lucas et al. 2016) and on spring blooms (Dugdale et al. 2016). Kimmerer et al. (2014) modeled the tidal migration and retention of zooplankton in SFB with a particle tracking model. Most of the aforementioned modeling efforts focused on one aspect of the SFB ecosystem. The first study that integrated both hydrodynamics and biogeochemistry into an

ecosystem model of SFB was conducted by Di Toro et al. (1977). Their model also included different forms of nitrogen (NO_3 versus NH_4) and their impacts on phytoplankton production.

More recently, Liu et al. (2018) used the biogeochemical model carbon, silicate, and nitrogen ecosystem (CoSiNE) coupled to the hydrodynamic model SCHISM to carry out a seasonal ecosystem simulation for SFB over 2 years. This coupled model considered the interaction of nutrients, phytoplankton, zooplankton, and hydrodynamics. It reproduced the seasonal and spatial patterns of nutrients and chlorophyll in North Bay. The authors also conducted a series of sensitivity tests to evaluate phytoplankton responses to different environmental factors, such as river outflow, WWTP, NH_4 inhibition, and benthic grazing, with a focus on the period of 2011–2012.

It is necessary to run such a model for a longer time period in the highly dynamic system of the SFB to capture the observed interannual variabilities in chlorophyll and nutrients (Cloern and Jassby 2012). In this study, the model described in Liu et al. (2018) was run from 2005 to 2014 to encompass different meteorological and hydrological conditions, especially the variation in fresh-water outflow. First, a 10-year calibration was conducted for hydrodynamics, nutrient and chlorophyll conditions in North, Central and South Bays. Second, the spatial distribution of chlorophyll and mean seasonal variations of chlorophyll and nitrogen were compared with monthly observations of 10-year time series to assess the fidelity of the model in simulating phytoplankton processes. Third, the interannual variabilities of nutrients and phytoplankton concentration were analyzed, followed by calculation of the nutrient fluxes around Central Bay (indicated by

the dashed blue lines in Fig. 1) under different flow conditions. Finally, sensitivity tests were conducted to study the effects of flow and benthic grazing in different regions of SFB. In the future, this open source model is planned to be run operationally at the California Department of Water Resources and could be used for other estuaries.

2 Materials and methods

2.1 Model framework

We adopted the model framework of Liu et al. (2018), which consists of a hydrodynamic model of SCHISM, and a biogeochemical model of CoSiNE. These two models are coupled, to simulate the water circulation and ecosystem processes simultaneously. When integrating the coupled model, the hydrodynamic equations are solved first, followed by the treatment of the biogeochemical kinetics. In this study, some new features were added to the Liu et al. (2018) version, including options for benthic grazing, specifying SPM and flexible model outputs at specific stations. The ecosystem simulation of SCHISM/CoSiNE was conducted on the “viz3” High Performance Computing (HPC) system at the University of Maine. A one-year simulation took about 20 h when using 128 computation cores.

2.1.1 Hydrodynamic model SCHISM

SCHISM is a derivative of the original SELFE model (Zhang and Baptista 2008), and has many new enhancements and upgrades including an extension to large-scale eddying regimes. It is an open source community model based on unstructured grids (triangles, quadrilaterals, or their mix) designed for seamless simulation of 3D baroclinic circulation across creek-lake-river-estuary-shelf-ocean scales (Ye et al. 2016, 2018; Zhang et al. 2015, 2016, 2017). The cross-scale, semi-implicit scheme in SCHISM allows for large time steps aided by higher- and lower-order numerical methods (Ye et al. 2018). More information about this model can be found on the SCHISM website: schism.wiki. The application of the SCHISM model in SFB ensures accurate hydrodynamics, which is needed for the transport and diffusion of nutrients and phytoplankton.

2.1.2 Biogeochemical model CoSiNE

Originally developed for studying biogeochemical processes in the equatorial Pacific Ocean (Chai et al. 2002; Chai et al. 2007; Dugdale et al. 2002), different versions of CoSiNE models have evolved to address different research foci (Chai et al. 2003; Liu et al. 2018; Xiu and Chai 2014). The one used in this study has 13 state variables (Table 1), including two phytoplankton

Table 1 List of CoSiNE model variables

Name of state variables	Symbol	Unit
Nitrate	NO ₃	mmol[N]/m ³
Silicate	SiO ₄	mmol[Si]/m ³
Ammonium	NH ₄	mmol[N]/m ³
Small Phytoplankton	S1	mmol[N]/m ³
Diatoms	S2	mmol[N]/m ³
Microzooplankton	Z1	mmol[N]/m ³
Mesozooplankton	Z2	mmol[N]/m ³
Detrital Nitrogen	DN	mmol[N]/m ³
Detrital Silicon	DSi	mmol[Si]/m ³
Phosphate	PO ₄	mmol[P]/m ³
Dissolved Oxygen	DOX	mmol[O ₂]/m ³
Total Dioxide Carbon	CO ₂	mmol[C]/m ³
Alkalinity	ALK	meq/m ³

functional groups (S1 and S2), two zooplankton species (Z1 and Z2), three nitrogen forms (NO₃, NH₄ and detrital nitrogen DN), two silicon forms (SiO₄ and detritus silicon DSi), one phosphorus form (PO₄), dissolved oxygen (DOX), total carbon dioxide (CO₂), and total alkalinity (ALK). Phytoplankton biomass as the summation of small phytoplankton S1 and diatoms S2 has a unit of millimoles [N] per cubic meter in the model, which is converted to micrograms [chl] per liter with a 1:1 conversion ratio. This conversion is based on previous studies that found 1 μmol nitrogen removed produces 1 μg chlorophyll in SFB (Dugdale and Goering 1970; Dugdale et al. 2012; Marra et al. 1990). The ecological process diagram of the model and its linkage to physical processes can be found in Liu et al. (2018) along with the documentation of equations and parameter values. This includes physiological interactions, e.g., light limitation of photosynthesis and NH₄ repression of NO₃ uptake. The latter is included specifically as it has been shown to play a role in SFB phytoplankton production and influence chlorophyll level (Glibert et al. 2014b; Parker et al. 2012a; b; Wilkerson et al. 2015).

2.1.3 SFB model grid

The model covers SFB from Coyote Creek in South Bay to Rio Vista in North Bay near the confluence of the Sacramento and San Joaquin rivers, adjacent to the Bay Delta. It has a length of ~100 km and a width varying from 5 to 20 km. Figure 1 shows the model grid of pure triangles superposed with bathymetry. The unstructured grid enables a good representation of the geomorphology varying from the coastal ocean to the estuarine subembayments and rivers/creeks in SFB. The grid has a coarse resolution up to 1 km in the coastal ocean where the depth reaches over 100 m. Inside the estuary, finer resolution is applied with grid size varying from several

hundreds of meters to as small as 10 m, accompanied by bathymetry changing from over 10 m in the channels to less than 3 m in the shallow shoals. In addition, more grid cells are placed in the rivers/creeks to represent small features and in the ship channels that guide the flows from the rivers to ocean. In the vertical, 23 terrain-following sigma layers are used (see Zhang and Baptista 2008 for details).

2.1.4 Physical model specification

The specification for the physical model SCHISM is based on Chao et al. (2017b) and Zhang (at UCLA, personal communication). It includes atmospheric forcing, river forcing, and coastal ocean forcing from 2005 to 2014. The atmospheric forcing is from the Coupled Ocean Atmosphere Mesoscale Prediction System (Doyle et al. 2009). It provides wind, air temperature, atmospheric pressure, humidity, precipitation, and solar radiation. The coastal ocean forcing contains elevation, velocity, salinity, and temperature at the ocean boundary from Chao et al. (2017a). The river forcing includes boundary conditions for the three rivers in SFB (Sacramento, San Joaquin, and Napa) and a small creek (Coyote) in the lower South Bay. SFB fresh water input is taken from USGS measurements (<https://waterdata.usgs.gov/nwis>).

2.1.5 Biogeochemical specification

The specification for the CoSiNE model follows the model setup in Liu et al. (2018). The coastal ocean boundary conditions for the 13 CoSiNE state variables are based on the results from the ROMS/CoSiNE (Chao et al. 2017a; Xiu et al. 2018). At the river boundary, concentrations of CoSiNE state variables are inferred from the USGS water quality measurements in SFB (hereafter as WQDATA), using observations at the nearest stations: USGS 657 for Sacramento River, USGS 3 for San Joaquin, USGS 9 for Napa River, and USGS 36 for Coyote Creek. If water quality measurements are missing for a certain period, we replace them with data from nearby stations. Since the USGS data are generally monthly, interpolation is used for preparing the boundary condition at 1-day interval. Through the river boundary, fresh water with associated nutrients and seed phytoplankton enters the SFB model system. Another important source of nutrients in SFB is the WWTPs. Following the model setup in Liu et al. (2018), we only consider the major WWTPs that result in a total NO_3 flux of 10.7 mol/s and a total NH_4 flux of 27.5 mol/s.

To simulate the influence of turbidity on light penetration and, therefore, on phytoplankton biomass and productivity (Cloern 1987), Liu et al. (2018) applied a constant SPM concentration of 20 mg/L in their base experiment, even though the WQDATA and previous observations (Bever and MacWilliams 2013; Cloern and Jassby 2012) showed large differences in SPM between North and South Bays. To

improve the light field for phytoplankton growth, we apply spatially and temporally varying SPM in the multiple-year simulation by interpolating observations from the WQDATA.

2.2 Observational data

In this study, the WQDATA are extensively used for model calibration and analysis. The data were downloaded from the USGS website: <https://sfbay.wr.usgs.gov/access/wqdata/index.html>, which include temperature, salinity, chlorophyll, NO_3 , NH_4 , and SPM. The measurements were made nearly monthly from 1969 to the present. The dataset has a good spatial coverage with 37 fixed stations located in the central deep channel from South to North Bays with 3–6 km spacing (represented by the red dots in Fig. 1). For comparison between model results and observations, only surface data are used because bottom nutrient data are limited, and the differences between surface and bottom chlorophyll concentrations are small in SFB (Schraga and Cloern 2017).

3 Results

3.1 Model validation

Figure 2 compares surface temperature and salinity of model results and USGS observations from 2005 to 2014. The sampling frequency for model results is two hours, and that for WQDATA observations is one month. Thus, we linearly interpolate the monthly data to the same time base as the model with a 2-hour interval. We also linearly interpolate the observational data in space, to display them in Fig. 2. The y -axis represents the distances of USGS water quality stations aligned from south to north, from USGS 36.

The model captures well the seasonal variation of temperature with higher temperatures (above 20 °C) in summer and lower temperatures (around 5 °C) in winter. The model underestimates the summer and winter temperatures in South Bay (below USGS 21). This discrepancy may be partially due to the lack of short-term (hourly) observations and the interpolations of monthly data to two hourly. However, the model output using two-hourly intervals is useful as it provides more details. There is little temperature difference between the south and north in SFB, but cold temperature signals caused by the intrusion of coastal upwelling water (from USGS 16 to USGS 21) are clearly seen in both observations and model results.

For surface salinity, both model results and observations (Fig. 2c, d) show strong seasonal cycles from south to north, with good match in patterns, although more short-term salinity variabilities exist in the modeled results (Fig. 2d). In general, SFB experiences low salinity in winter and spring due to large river outflow (Chao et al. 2017b). In summer and fall, higher

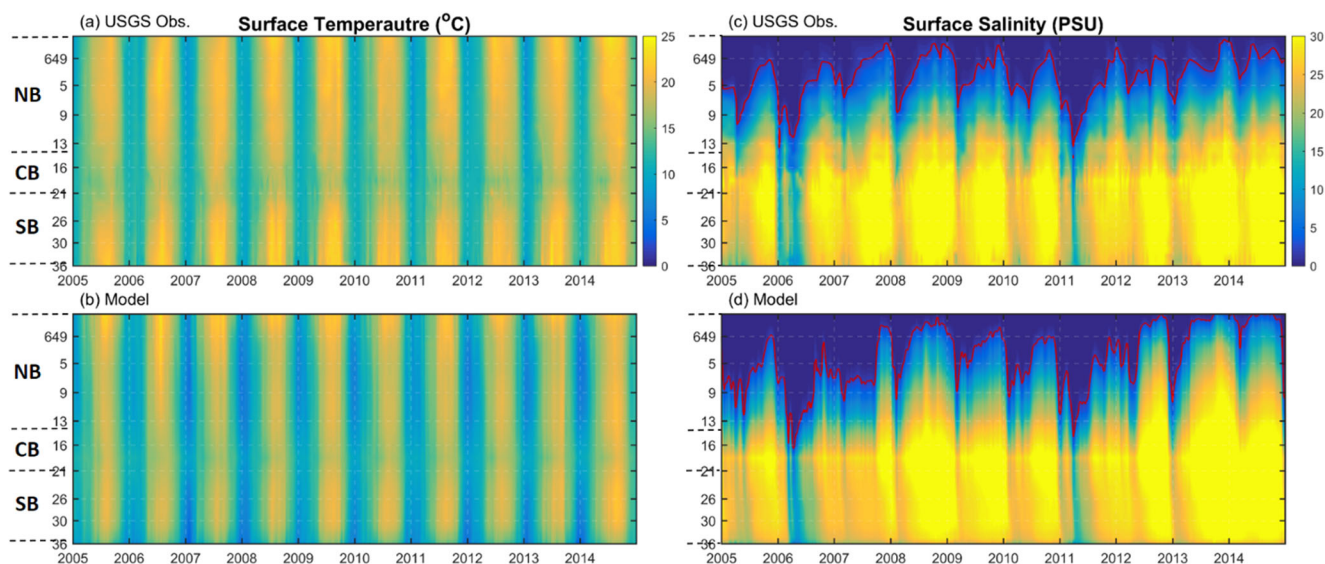


Fig. 2 Comparison of surface temperature (**a**, **b**) and salinity (**c**, **d**) between SCHISM model and USGS observations. Both temperature and salinity are expressed as a function of time and space. The x -axis represents time from 2005 to 2014, and the y -axis is the transect from

south to north along the USGS water quality stations (NB, North Bay; CB, South Bay; SB South Bay). The red line in (**c**, **d**) delineates the location where salinity is 2 PSU

salinity water appears in the bay because of lower fresh water input in these dry seasons. Also, North Bay (north of USGS 16, Fig. 1) is much fresher than South Bay since the major rivers (Sacramento and San Joaquin) are located in the north. The spread of fresh water (Fig. 2c, d), and the transition or low salinity zone (indicated by the 2 PSU red line contour) varying with time and space is clear. Two exceptional spring precipitation events resulting in very large river outflow in 2006 and 2011 can be identified where the river influence penetrated across Central Bay and into South Bay. The water in South Bay was heavily diluted, and the salinity dropped below 10 PSU; and the river influence was dominant in the entire SFB. The physical model successfully captures these two events (Fig. 2d).

The simulation of nutrients and plankton in CoSiNE was validated in Liu et al. (2018) at seasonal time scales. To further validate model results over more climatologically relevant interannual time scales, we compare chlorophyll, NO_3 and NH_4 of the model and WQDATA observations at three stations selected to represent North Bay (USGS 6), Central Bay (USGS 18), and South Bay (USGS 27) from 2005 to 2014 (Fig. 3). Chlorophyll concentrations increase in spring and summer, begin to drop in fall and remain low in winter. For most of the years, chlorophyll stays higher in the warm periods than in the cold ones. The magnitude of modeled chlorophyll matches that of observations at all three locations. Generally, it is less than $10 \mu\text{g/L}$ in North Bay and Central Bay and greater than $10 \mu\text{g/L}$ in South Bay (in the warm period). Visually, the chlorophyll time series matches the USGS measurements best in South Bay, does well in North Bay, and does less well in Central Bay where greater chlorophyll values also occur in the winter months. The modeled

minimum concentrations systematically underestimate the observations during winter in Central Bay and South Bay. There are also overestimates in the model, mostly in North and South Bays from 2010 to 2014, which may be related to the sampling interval of monthly observational data that are likely to fail to capture transient phytoplankton blooms that occur on daily to weekly scales (Cloern et al. 2005).

The modeled nutrients (NO_3 and NH_4) over the 10 years show seasonal cycles with peak values in winter and minimum concentrations in summer, matching the USGS observations. Stronger seasonal cycles are observed in North Bay and South Bay than in Central Bay. The CoSiNE model simulates well the interannual variations of NO_3 and NH_4 observations (WQDATA) that show a large amplitude in North and South Bays, compared with being fairly constant in Central Bay. For example, the model captures the lower NO_3 concentrations during 2006 and 2011 in North Bay. Even though discrepancies between model results and observations exist in particular years, the overall model performance is good (see also the following quantitative assessment) in reproducing the interannual and spatial variabilities of nutrients in SFB.

To quantitatively assess the model performance, we use statistics including correlation coefficient (R) and model skill (SS) in Table 2 for surface salinity, temperature, chlorophyll, NO_3 , and NH_4 . Model skill is defined as $SS = 1 - \frac{\sum_i (\text{model}_i - \text{obs}_i)^2}{(|\text{model}_i - \overline{\text{obs}}_i| + |\text{obs}_i - \overline{\text{obs}}_i|)^2}$ (Willmott 1981), where model_i and obs_i are time series of modeled and observed variables, respectively, and overbar denotes time average. $SS = 1.0$ means perfect model prediction, while $SS = 0$ means total disagreement. Taylor diagrams for each variable are shown in Fig. 4. The Taylor diagram integrates three statistics (correlation

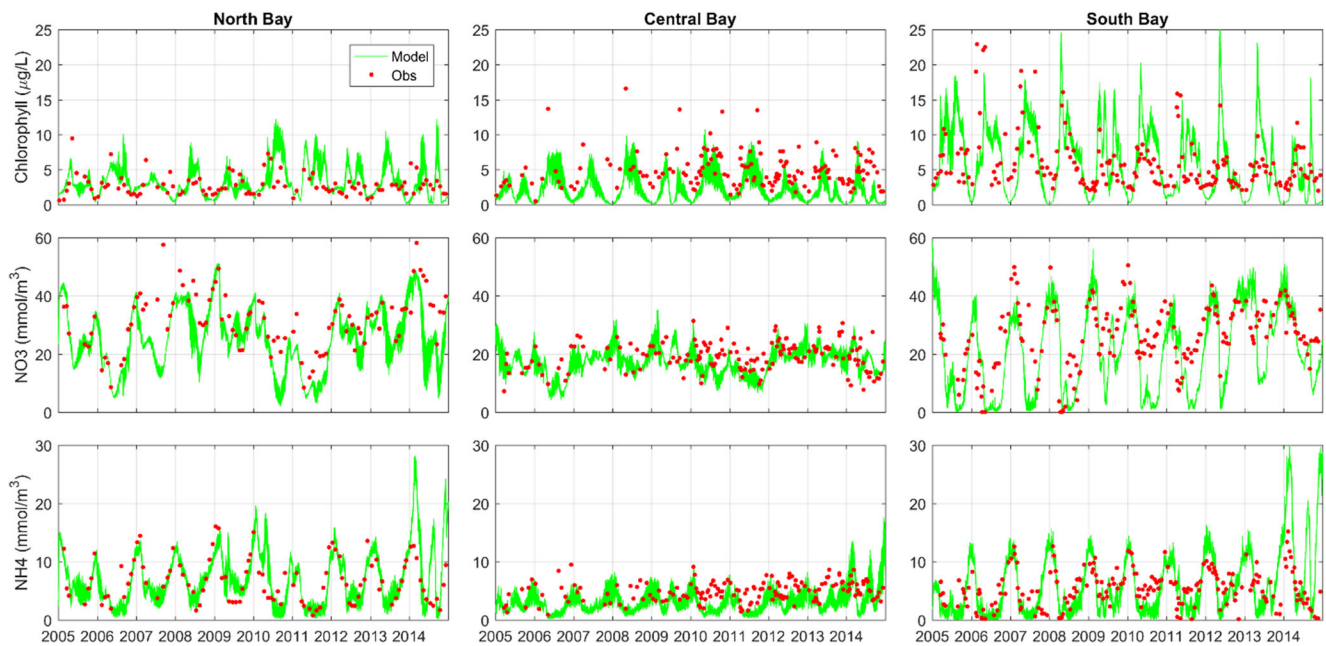


Fig. 3 Comparison of modeled chlorophyll (top row), NO_3 (middle row), and NH_4 (bottom row) with USGS observations in North Bay (USGS 6), Central Bay (USGS 18), and South Bay (USGS 27). Green line shows model results, while red dots represent the observational data

coefficient, standard deviation, and root mean square difference) in one plot, and one can visually evaluate how a model result matches the observation by the distance of a data point to the perfect simulation (the black dot). These statistics are based on

the “best matches” over ± 3 days around the time point because river boundary conditions for CoSiNE are based on monthly data, and the model might not be simulating true conditions on the exact day.

Table 2 Statistics concerning SCHISM/CoSiNE model calibration for salinity, temperature, chlorophyll, NO_3 , and NH_4

Variable\station	649	3	6	9	13	15	18	21	24	27	30	32
SAL (PSU)												
R	0.80	0.79	0.91	0.96	0.98	1.00	1.00	0.96	0.94	0.90	0.90	0.91
SS	0.846	0.823	0.944	0.979	0.988	0.998	0.998	0.980	0.963	0.942	0.944	0.950
TEMP (°C)												
R	0.99	0.98	0.97	0.98	0.98	0.99	0.99	0.99	0.98	0.98	0.97	0.96
SS	0.994	0.980	0.964	0.959	0.976	0.991	0.994	0.991	0.979	0.970	0.949	0.901
CHL ($\mu\text{g/L}$)												
R	0.54	0.63	0.69	0.49	0.67	0.76	0.67	0.79	0.73	0.67	0.59	0.69
SS	0.600	0.777	0.809	0.638	0.795	0.846	0.718	0.849	0.845	0.813	0.754	0.831
NO_3 (mmol/m^3)												
R	0.95	0.89	0.76	0.66	0.58	0.62	0.75	0.67	0.61	0.61	0.76	0.89
SS	0.971	0.932	0.838	0.751	0.681	0.713	0.843	0.798	0.749	0.744	0.849	0.942
NH_4 (mmol/m^3)												
R	0.85	0.78	0.82	0.77	0.69	0.65	0.67	0.77	0.66	0.60	0.69	0.58
SS	0.919	0.880	0.901	0.853	0.780	0.732	0.758	0.858	0.802	0.744	0.796	0.713

Note: Statistics are based on the best match in ± 3 days. Twelve USGS water quality stations with observations were chosen to evaluate model performance using correlation coefficient (R) and model skill (SS)

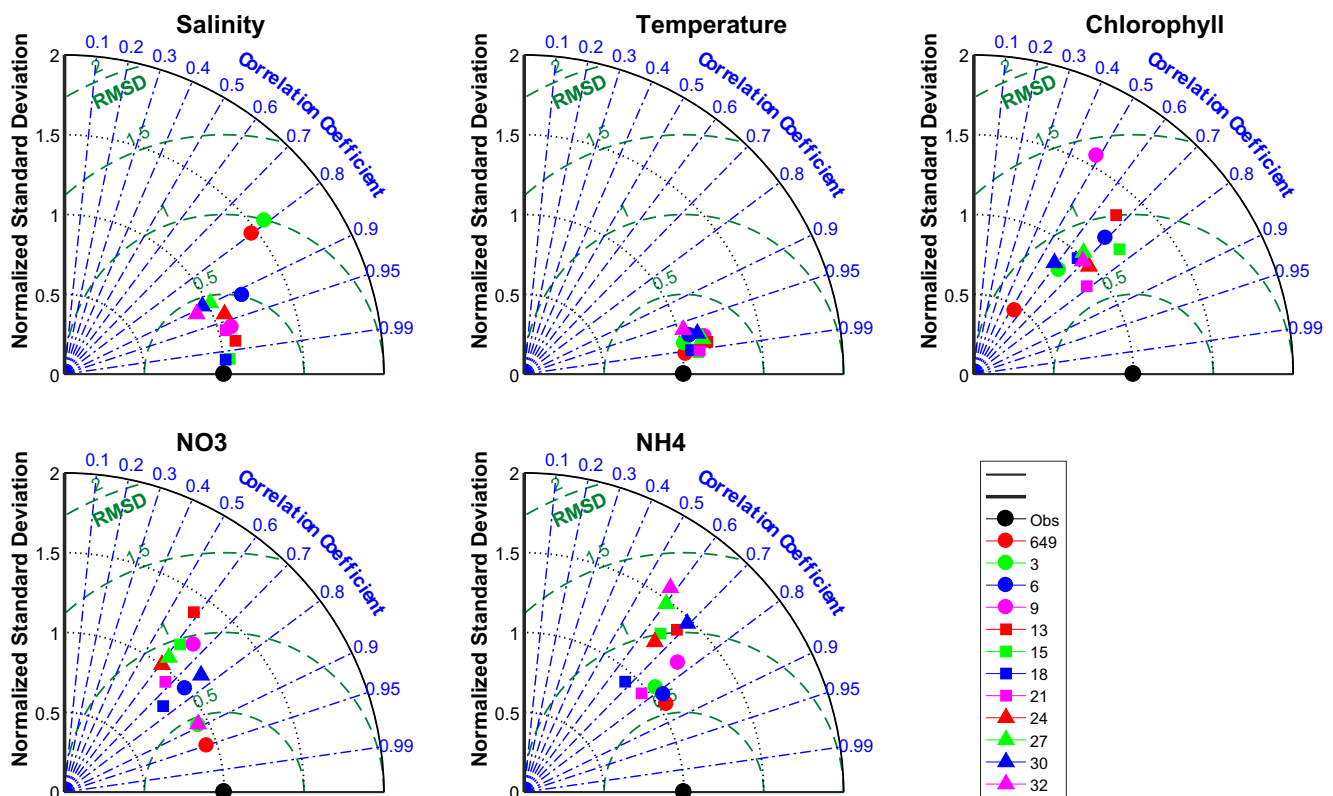


Fig. 4 Taylor diagrams illustrating model performances in terms of salinity, temperature, chlorophyll, NO₃, and NH₄. The model values are best matches with the data points within ± 3 days around the sampling

time. Each variable is normalized by the standard deviation of the corresponding observed field

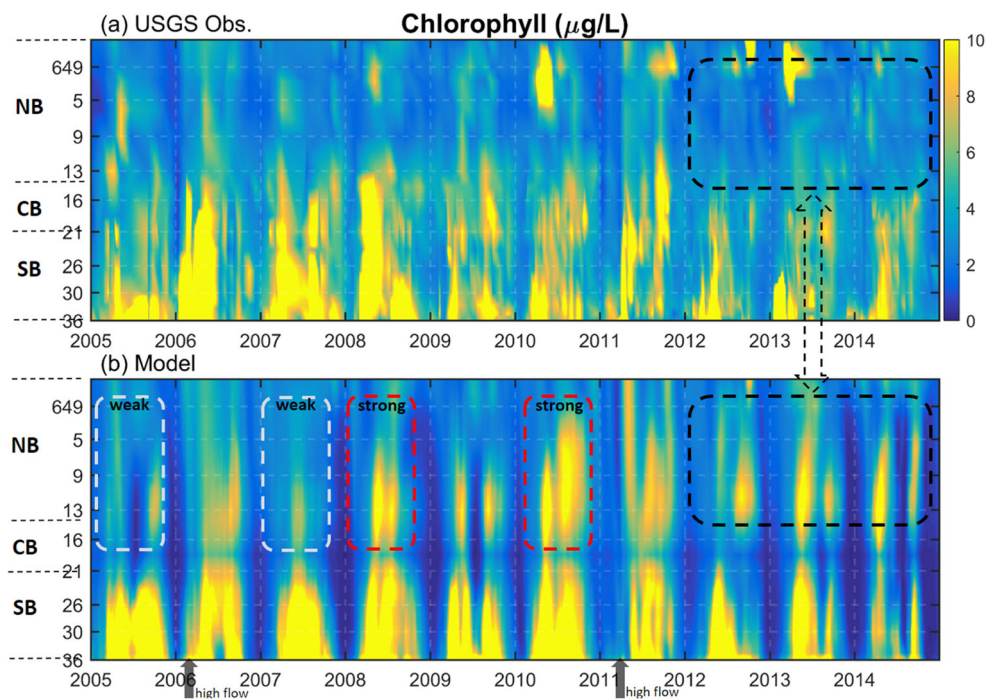
Table 2 and Fig. 4 show that the model performance is very good for physical parameters. For salinity, R exceeds 0.9 for most stations, while the lowest $R = 0.79$ appears in the upper North Bay at station USGS 3. R for temperature is high at all stations. SS ranges from 0.823 to 0.998 for salinity and from 0.901 to 0.994 for temperature. As might be expected, the model performance for biological parameters is not as good as for physical parameters. For chlorophyll, R is mostly greater than 0.5. Similar model performances are seen at most stations, except at two stations USGS 649 and USGS 9 where relatively larger deviations exist (Fig. 4). SS of chlorophyll ranges from 0.600 to 0.849. The overall model performances for NO₃ and NH₄ are similar to that of chlorophyll. R for NO₃ varies from 0.58 to 0.95, while R for NH₄ varies from 0.58 to 0.85. SS for NO₃ ranges from 0.681 to 0.971, and for NH₄ from 0.713 to 0.919. Overall, the validation across different parameters at different stations lends confidence to the modeling results. Discrepancies in chlorophyll and nutrients between model results and observations may be related to multiple factors, including the omission of benthic grazing by invasive clams, oversimplification of WWTP inputs and no nutrient feedback from sediment remineralization (Cornwell et al. 2014; Dugdale et al. 2013, 2016; Liu et al. 2018). Furthermore, monthly WQDATA are used (and interpolated) to drive the biogeochemical model at river boundaries, while the model output is

sampled every two hours. The temporal mismatch between model and observational data is an important source of error.

3.2 Interannual variability

Figure 5 compares modeled chlorophyll with WQDATA observations at the USGS stations from 2005 to 2014. Model results reproduce reasonably well the spatial pattern, and the seasonal and interannual variations of the observations. Spring and fall blooms are often evident in SFB, although chlorophyll always remains low in winter. Consistent with the observations, modeled chlorophyll is persistently high in South Bay during warm months until 2012 when concentrations decrease. In contrast, chlorophyll in North Bay has strong interannual variability, and the overall concentrations are lower in North Bay than in South Bay. Chlorophyll concentrations in North Bay were higher in 2008 and 2010 (peak chlorophyll $> 10 \mu\text{g/L}$), and lower in 2005 and 2007. Large differences in chlorophyll between model results and observations can be seen in 2012, 2013, and 2014. Another difference between South Bay and North Bay is the extent of the high chlorophyll concentration region. When there are phytoplankton blooms, the high chlorophyll concentration spreads over almost the entire South Bay. In North Bay, the blooms tend to occur in the more seaward part (San Pablo Bay) although occasionally they may appear in the

Fig. 5 Chlorophyll as a function of time and space (the same format as used in Fig. 2). The top panel (a) is interpolated from the USGS observational data, and the bottom panel (b) is model results. The years with lower chlorophyll (dashed white line) and higher chlorophyll (dashed red outline) are marked. The large difference in chlorophyll between model results and observations in North Bay from 2012 to 2014 are marked by a dashed black line



landward region. Similarly, Figs. 6 and 7 show the nutrient variations in time and space. Visually, the model captures well the observations of both NO_3 and NH_4 (e.g., 2005–2006 and 2011–2012). The nutrient concentrations are higher towards the upper North Bay and lower South Bay, and relatively low in Central Bay. Although the WWTP nutrient input is predominantly NH_4 (Novick and Senn 2014), the magnitude of NO_3

concentration is larger (50 mmol/m^3) than that of NH_4 concentration (15 mmol/m^3), as it is largely un-used by phytoplankton. In addition, there is high NO_3 towards the lower South Bay, which is fairly stable every year because of the large amount of NO_3 input from the local WWTPs (Novick and Senn 2014). In North Bay, there were higher values in 2008, 2009, and 2014 (strong NO_3 years), and low values in 2006 and 2011 (weak

Fig. 6 NO_3 as a function of time and space (the same format as used in Fig. 2). The top panel is interpolated from the USGS observational data, and the bottom panel is model results. The years 2005–2006 and 2011–2012 are outlined to show the agreement between model results and observations

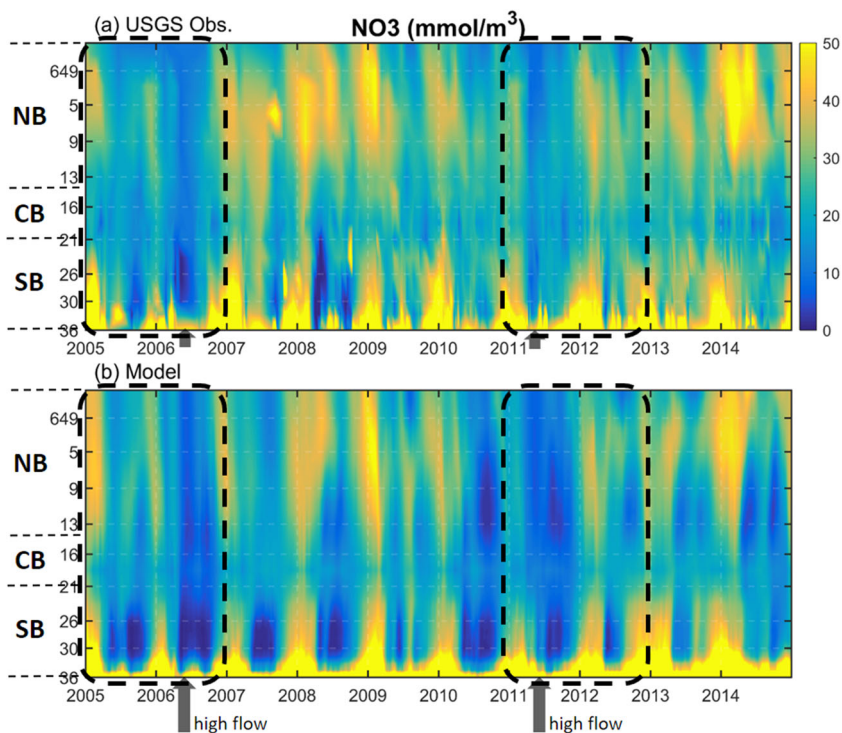
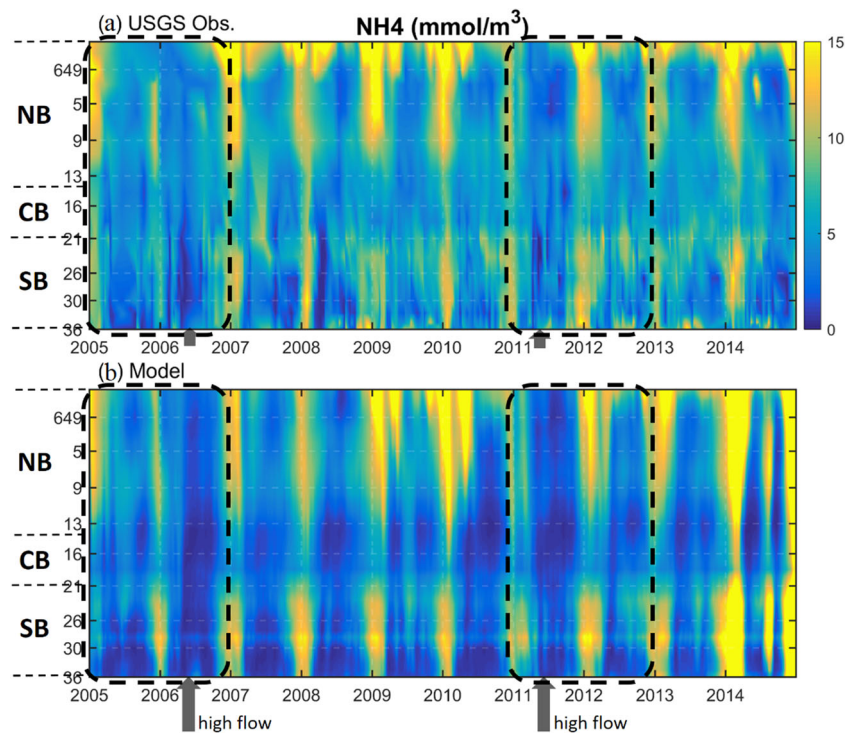


Fig. 7 Same as Fig. 6, except for NH_4



NO_3 years). The interannual variability of NH_4 generally follows that of NO_3 . However, NH_4 concentration is usually higher towards the upper North Bay.

To further examine the interannual variation, the anomalies of chlorophyll and nutrients are shown in Fig. 8, where anomaly is defined as the model result after removing mean modeled annual variation. For chlorophyll, many positive anomalies ($\sim 5 \mu\text{g/L}$) appeared in South Bay, especially before 2012, while they appeared sporadically in North Bay after 2008. For NO_3 and NH_4 , the anomalies are consistent with the nutrient variations in Figs. 6 and 7, and most positive anomalies are present either in the upper North Bay or towards the lower South Bay. Large positive NO_3 anomalies ($\sim 20 \text{ mmol/m}^3$) appeared mainly in 2008, 2009, 2012, 2013, and 2014, while large positive NH_4 anomalies ($\sim 5 \text{ mmol/m}^3$) appeared mainly in 2009, 2010, and 2014. Large negative anomalies exist in the two wet years of 2006 and 2011 for both NO_3 and NH_4 . The negative nitrogen anomalies coincide with the negative salinity anomalies shown in Chao et al. (2017b) and the elevated peak river discharges in 2006 and 2011 in Fig. 9, which suggests that the very low NO_3 and NH_4 concentrations are mainly caused by river dilutions, not by ecological processes.

3.3 Nutrient fluxes

To illustrate changes of nutrient fluxes under different flow conditions, the DIN (the sum of NO_3 and NH_4) fluxes across the boundaries of Central Bay (indicated by the dashed lines in Fig. 1) from 2006 to 2008 are plotted in Fig. 10. These include the

flux into the coastal ocean from Central Bay, the flux from North Bay to Central Bay, the flux from South Bay to Central Bay, and a constant WWTP input. The bottom panels in Fig. 10 are the averaged fluxes for 2006, 2007, and 2008, along with the components of NO_3 and NH_4 . The statistical values are listed in Table 3. For the DIN flux from North Bay to Central Bay, NO_3 is the dominant nitrogen form and the percentage varied from 87.7% in 2006 to 93.1% in 2007. However, for DIN from South Bay to Central Bay, NH_4 is the dominant nitrogen form and the percentage of NO_3 varies from 15.2 to 37.8%. For the constant DIN from the WWTPs (see Table B1 in Liu et al. (2018)), NH_4 is also dominant, with NH_4 percentage at 88.9%. The total nitrogen influx into Central Bay was 46.2 mol/s in 2006, 33.4 mol/s in 2007, and 27.8 mol/s in 2008. The total influx was largely balanced by the outflux into the ocean, with 39.3 mol/s in 2006, 32.6 mol/s in 2007, and 25.1 mol/s in 2008.

The pattern of DIN flux into the ocean and the flux from North Bay resemble each other (Fig. 10a), suggesting North Bay as the major source of DIN that is exported to the ocean, as NO_3 is not taken up by the estuarine phytoplankton. There also exists year-to-year variation. In particular, DIN flux from North Bay was much larger in 2006 (high flow year) than in 2007 (intermediate flow year) and 2008 (low flow year). The DIN flux into the ocean in 2006 reached a peak value of around 200 mol/s, while the DIN flux from South Bay, normally $< 20 \text{ mol/s}$, does not change much. In addition, DIN flux is high in spring and low in summer with some oscillations, which follow the general pattern of major river discharge (see Fig. 9).

Fig. 8 Chlorophyll, NO₃ and NH₄ anomalies from 2005 to 2014. For each variable, the anomaly is computed by removing annual mean. Large negative anomalies of NO₃ and NH₄ in 2006 and 2011 are outlined, and gray arrows along the x-axis show high flow years

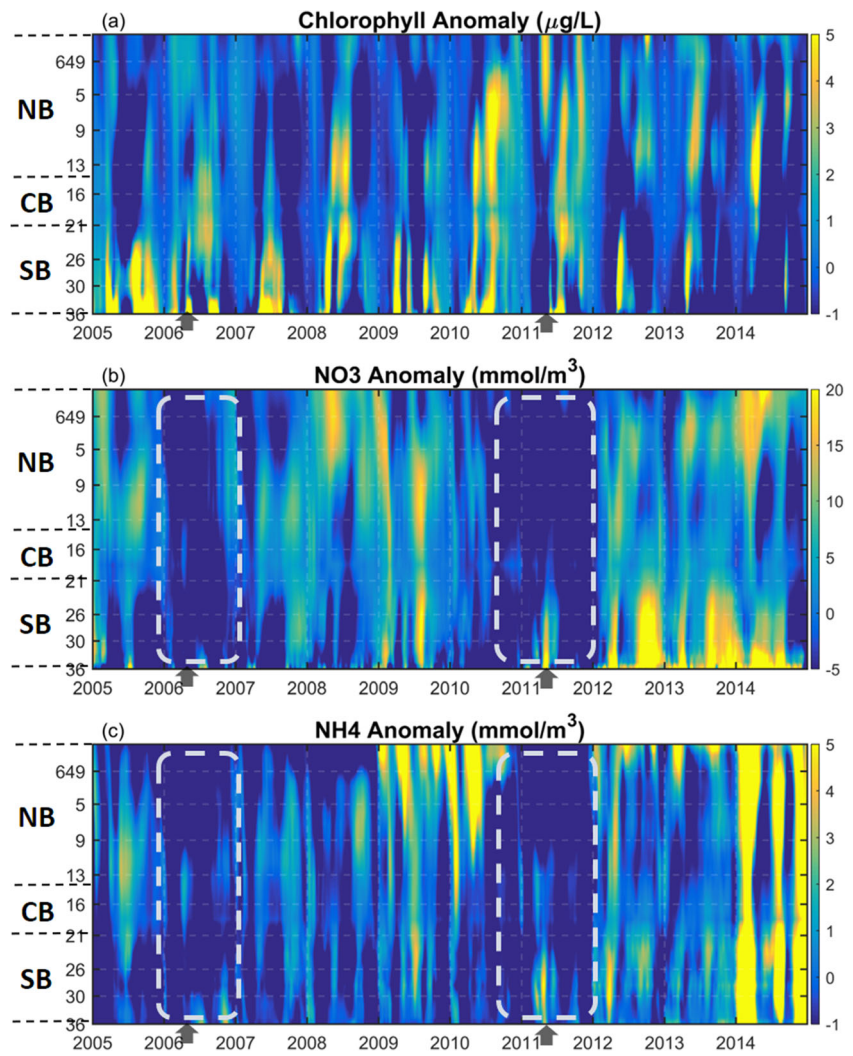


Fig. 9 Major river outflows (Sacramento River + San Joaquin River) from 2005 to 2014. The annual mean river outflow for each Water Year (from 1st Oct to 30th Sep) is also given

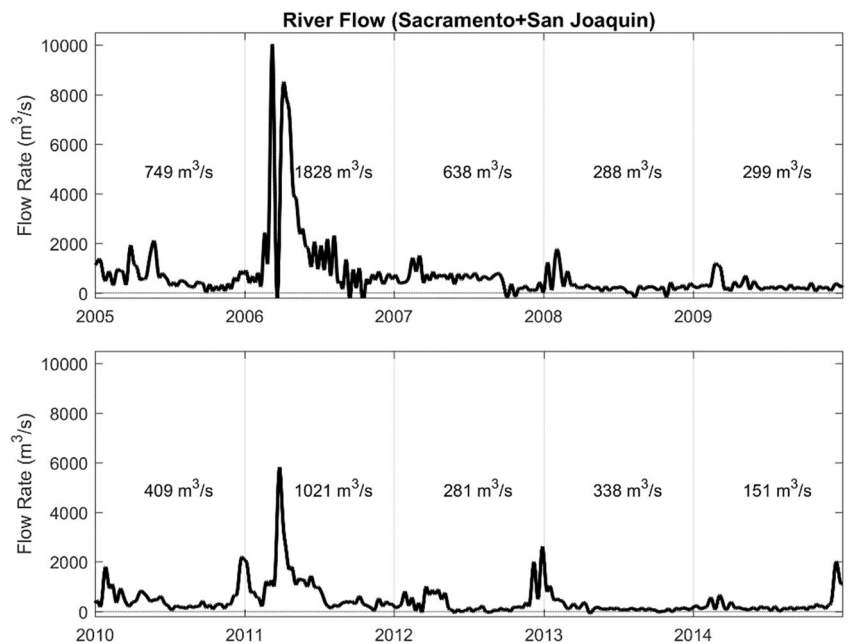
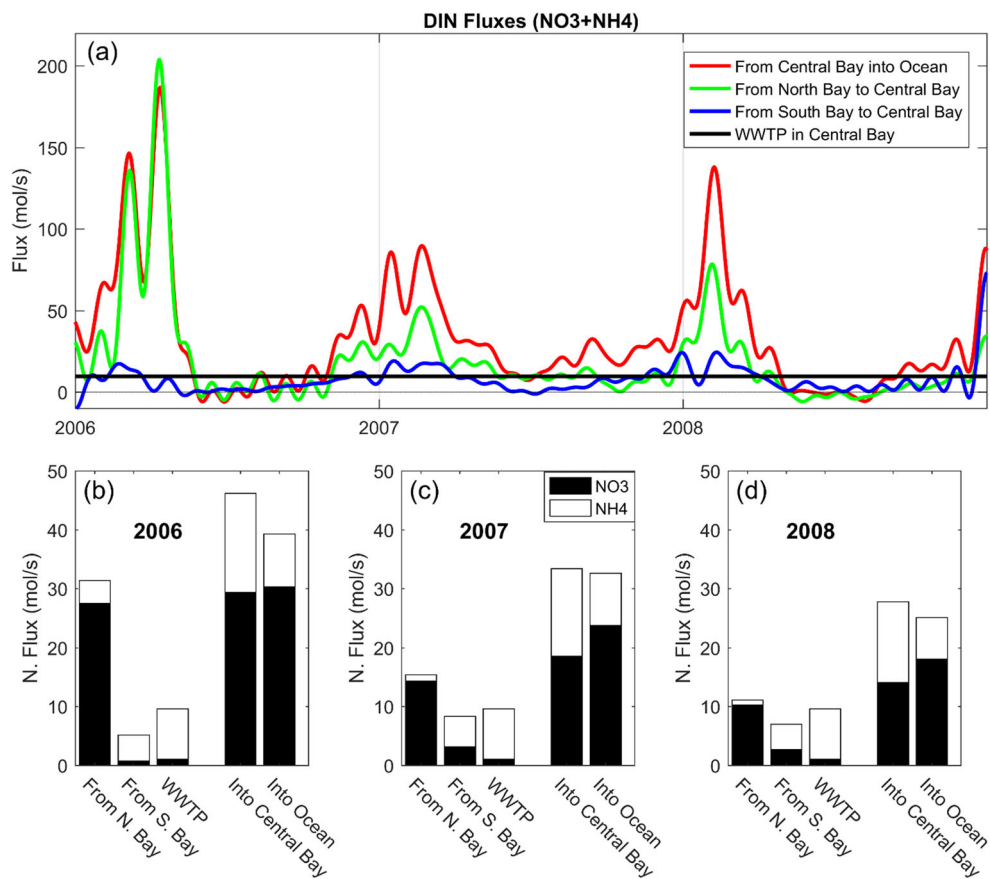


Fig. 10 Top panel: DIN flux as the sum of NO_3 flux and NH_4 flux into and out of the Central Bay in 2006, 2007, and 2008. The constant WWTP input in Central Bay is shown by the black line. The 3 years represent variable flow conditions from high flow in 2006 to low flow in 2008. The bottom panels are the annual mean nitrogen flux (black for NO_3 and white for NH_4) for the 3 years. The total flux into the coastal ocean through the Golden Gate is compared with the total fluxes into Central Bay as the sum of fluxes from both North Bay and South Bay and WWTPs



For Central Bay, the difference between total DIN influx and outflux is the nitrogen that goes either into sediment or is assimilated as phytoplankton biomass. Also, nitrification that converts NH_4 to NO_3 accounts for some of the higher NO_3 percentage in the outflux from North Bay (and so into Central Bay and the ocean), which was more evident in 2007 and 2008 (Fig. 10c, d). The large increase of nitrogen influx into the ocean in 2006 was caused by the high flow condition. The flow rate in 2006 ($1828 \text{ m}^3/\text{s}$) was more than six times of that in 2008 ($288 \text{ m}^3/\text{s}$). However, the nitrogen flux from North Bay in 2006 was only about 2.8 times of that in 2008, which implies that the nitrogen concentration associated with the high river outflow in 2006 dropped to less than half of the concentration in 2008. The flux from South Bay was actually smaller in 2006, probably because the large flow from the north suppressed the transport of nutrients from South Bay into Central Bay. For comparison, Smith and Hollibaugh (2006) adopted a box model approach to estimate DIN fluxes in North Bay and South Bay under steady state, although the method could not be applied in Central Bay. They reported the nitrogen flux into North Bay, varying from 7.1 mol/s ($6.1 \times 10^5 \text{ mol/day}$) in the summer months of the driest year 1994 to 69.8 mol/s ($6.033 \times 10^6 \text{ mol/day}$) in the winter months of the wettest year 1995, and the nitrogen flux into South Bay was around 13.8 mol/s ($1.2 \times 10^6 \text{ mol/day}$). In general, our model

estimates are in agreement with these published results, regarding the magnitude of DIN flux (Table 3), and the DIN seasonal variation with high flux in winter and low flux in summer (Fig. 10), although our calculation of DIN flux is focused on Central Bay. In addition, our model estimates are consistent with the DIN fluxes in Smith and Hollibaugh (2006), in which the DIN flux from North Bay is generally larger than that from South Bay.

3.4 Benthic grazing

For phytoplankton concentration in North Bay, grazing rates by clams are important, but reports of this contribution are mixed depending on the grazing rates applied (Chappelle et al. 2019; Kimmerer and Thompson 2014; Kraus et al. 2017), suggesting that more research is needed on benthic grazing. To investigate the potential effect of clam grazing on phytoplankton dynamics under different hydrographic conditions, we simulated this effect in the CoSiNE model by increasing the mortality rates of planktons in the bottom layer as outlined in Liu et al. (2018). Focus was placed on the effect of clam grazing on phytoplankton concentration in the bottom water as we did not intend to study the complex interaction between benthic clam dynamics and the dynamics of pelagic phytoplankton and nutrients. Two experiments

Table 3 Mean (standard deviation) for DIN (sum of NH₄ and NO₃) flux (mol/s) across the transects in Central Bay (blue dashed lines in Fig. 1), and nitrogen contributions (%) from NH₄ and NO₃

Nitrogen fluxes in Central Bay	2006	2007	2008
	Flow rate (m ³ /s)		
	1828	638	288
Flux from North Bay			
DIN (mol/s)	31.4 (47.2)	15.4 (11.6)	11.1 (18.9)
NH ₄ (%)	12.3%	6.9%	7.2%
NO ₃ (%)	87.7%	93.1%	92.8%
Flux from South Bay			
DIN (mol/s)	5.2 (5.0)	8.4 (6.1)	7.0 (11.3)
NH ₄ (%)	84.8%	62.2%	61.3%
NO ₃ (%)	15.2%	37.8%	38.7%
WWTP			
DIN (mol/s)	9.6	9.6	9.6
NH ₄ (%)	88.9%	88.9%	88.9%
NO ₃ (%)	11.1%	11.1%	11.1%
Influx into Central Bay			
DIN (mol/s)	46.2 (41.1)	33.4 (13.7)	27.8 (24.0)
NH ₄ (%)	36.4%	44.4%	49.2%
NO ₃ (%)	63.6%	55.6%	50.8%
Flux into ocean			
DIN (mol/s)	39.3 (46.8)	32.6 (21.2)	25.1 (33.7)
NH ₄ (%)	22.8%	27.1%	28.0%
NO ₃ (%)	77.2%	72.9%	72.0%

were conducted by increasing the mortality rates (as a proxy for benthic grazing) in North Bay by three or ten times, which resulted in lower chlorophyll as expected (Fig. 11). When three-times mortality rate was applied, chlorophyll dropped slightly compared with the baseline model result (Fig. 11a–c). However, when ten-times mortality rate was applied, the chlorophyll decreases differed markedly for different years. In 2006 (high flow), chlorophyll decreased by 38.6%; in 2007 (intermediate flow), by 52.2%; and in 2008 (dry flow), by 69.5%.

The two experiments (Fig. 11) suggest that the benthic grazing effect is modulated by the river outflow. High river flow tends to suppress the effect by reducing the retention time of water in the system. The benthic grazing effects under different flow conditions in different years can also be seen in Fig. 11d–f, where changes of chlorophyll flux from North Bay to Central Bay relative to the baseline condition are shown. The largest changes of chlorophyll flux occur in summer when the river flow is low. However, the changes are depressed in winter and spring when the river flow is high. The chlorophyll flux change due to benthic grazing presents an opposite seasonal variation to the nutrient fluxes, as shown in Fig. 10. For the interannual variability of chlorophyll flux

changes when ten-times mortality rate was used, the decrease in chlorophyll flux was the largest in 2006 with a maximum value of ~ 11 mol/s, while the decreases in 2007 and 2008 were smaller with maximum value ~ 7 mol/s for both years. Considering that the river flow in 2006 was about three to six times of those in 2007 and 2008 (Fig. 9), we conclude that the benthic grazing effect is more significant in low flow years.

4 Discussion

Fresh water input to SFB plays an important role in regulating the ecosystem. Figure 9 shows how the river outflow from the major rivers (Sacramento + San Joaquin) for our study period 2005–2014 varied strongly with distinct seasonal patterns. The majority of fresh water input into SFB is in spring, while summer and fall appear as dry seasons. There is also a large interannual variation. The annual mean river outflow (depending on Water Year, Buchanan and Ganju (2007)) can vary from 151 m³/s in the drought year 2014 to 1828 m³/s in the wet year 2006. Two high flow events in 2006 and 2011 stood out, both in spring when the peak flow rate approached 1.0 × 10⁴ m³/s. To examine the responses of the ecosystem to different flow conditions, we first presented the results from the 10-year run, and then examined three consecutive years with varying flow rates in later sections: 2006 as a high flow year, 2007 as an intermediate flow year and 2008 as a low flow year (annual mean river flow of 1828, 638, and 288 m³/s, respectively). Different levels of chlorophyll, DIN flux, and effects of benthic grazing appeared to be related to the different flow regimes.

High chlorophyll concentration appeared in 2008 (Fig. 5), but it did not appear in 2014, although the river flows in both these 2 years were low (see Fig. 9). This suggests that river flow is not the only driver for the chlorophyll variation. Other factors can be equally important in controlling the phytoplankton dynamics, e.g., turbidity, benthic grazing, and NH₄ repression of NO₃ uptake (Liu et al. 2018). Particularly, river effects are often interwoven with many physical and biogeochemical processes. Low river flow can increase the residence time for phytoplankton growth, but it may also lead to larger benthic grazing effect on phytoplankton, as shown in Sect. 3.4. Under low-flow conditions, any nutrient dilution effect is weakened, resulting in higher nutrient concentrations for phytoplankton growth; however, elevated NH₄ concentration may also repress NO₃ uptake by phytoplankton. Both physical and biogeochemical factors combine to regulate the phytoplankton growth in SFB (Dugdale et al. 2013; Wilkerson and Dugdale 2016).

River flow and NH₄ concentration are interconnected in their effect on phytoplankton concentration in North Bay (Dugdale et al. 2012). When river flow is very low, the NH₄ concentration could become relatively high because of the

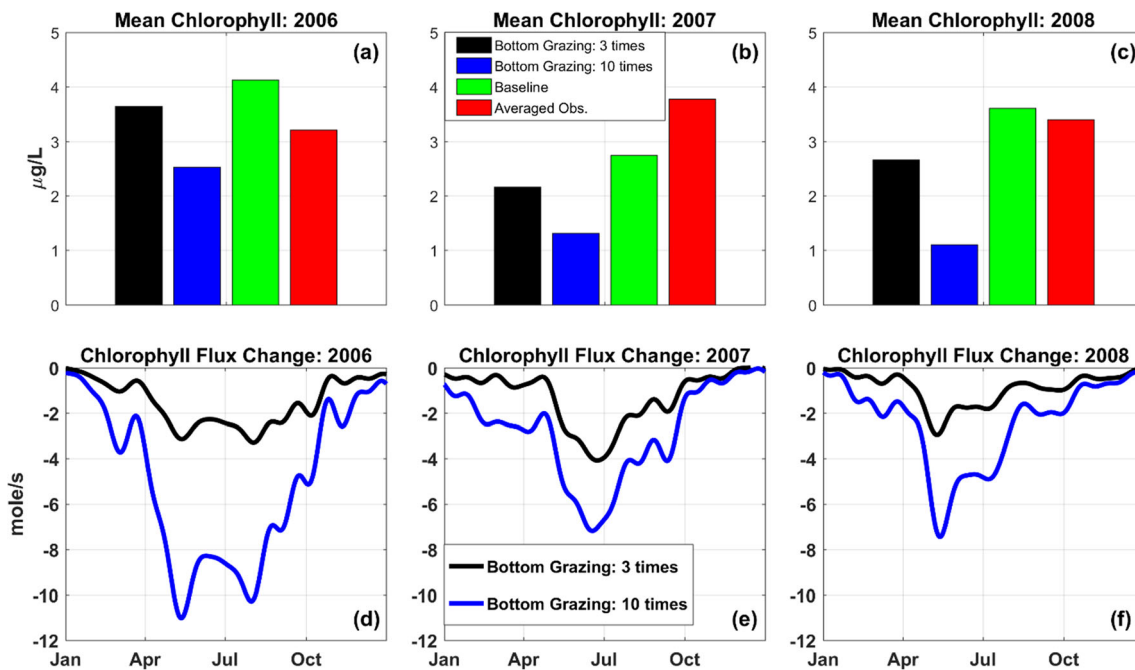


Fig. 11 Sensitivity tests for bottom grazing in 2006, 2007, and 2008. The bottom grazing rate is simulated by increasing the mortality rates of phytoplankton and zooplankton in North Bay to three times (black) and ten times (blue) the base rates. The top panels are the annual mean

chlorophyll concentrations in 2006 (a), 2007 (b), and 2008 (c). The bottom panels show the changes of chlorophyll fluxes from North Bay to Central Bay (indicated by the northernmost dotted line in Fig. 1) with respect to baseline conditions

nutrient input from upstream WWTPs, which could lead to low chlorophyll concentration due to NH_4 inhibition on phytoplankton (Wilkerson et al. 2015). When river flow is very large, it can dilute the NH_4 concentration in alleviating NH_4 inhibition; at the same time, it flushes the estuary to reduce chlorophyll concentration downstream. Consequently, phytoplankton blooms usually occur under moderate values of river flow and NH_4 concentration. In Fig. 12, modeled chlorophyll concentration ($> 5 \mu\text{g/L}$) is plotted versus flow rate and NH_4 concentration in Suisun Bay using all data from all months of the 10-year outputs. For each chlorophyll value, we associate it with the corresponding river flow averaged for the prior 2 weeks. It is clear that high chlorophyll ($\sim 10 \mu\text{g/L}$) occurs during an optimal window when NH_4 is $1\text{--}2 \text{ mmol/m}^3$ and flow rate is $100\text{--}250 \text{ m}^3/\text{s}$. This flow rate for the higher chlorophyll condition is consistent with the conclusion of Ball and Arthur (1979) who observed blooms happened only in the lower flow range of $[110, 700] \text{ m}^3/\text{s}$.

Before the 1980s, the primary production in Suisun Bay was high, which provided food for fish (Ball and Arthur 1979). The high-nutrient low-chlorophyll phenomenon in SFB since the late 1980s is related to many factors (Sect. 1), including light limitation, clam grazing and NH_4 repression (Cloern 1987; Dugdale et al. 2007; Lucas et al. 2016). In Suisun Bay, the anthropogenic nutrient input may drive the low chlorophyll condition as the nitrogen loading is mainly in the form of NH_4 (Novick and Senn 2014). With NH_4 continuously added to the system, at a rate higher than

phytoplankton can take it up (remove it), the high NH_4 concentration restricts phytoplankton from using NO_3 (Dortch 1990), which means that the major DIN pool in the form of NO_3 (as shown in Figs. 6, 7, and 8) is actually not used by phytoplankton (Dugdale et al. 2012; Glibert et al. 2014a). This mechanism contributes to the lack of high chlorophyll data when NH_4 concentration is $> 2 \text{ mmol/m}^3$ in Fig. 12. It also

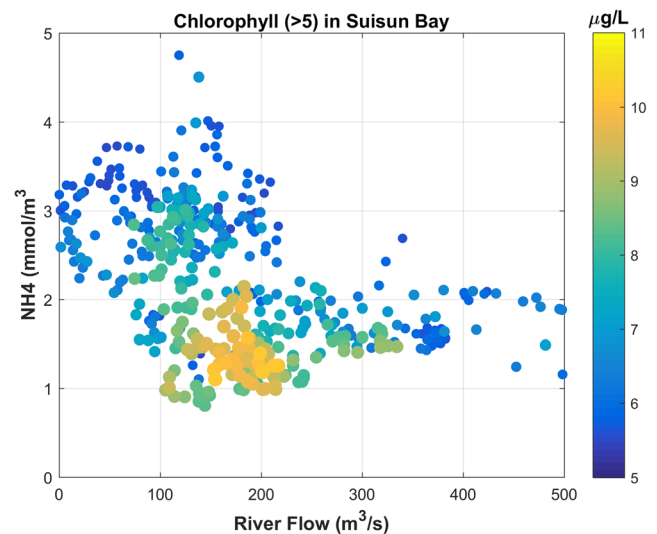


Fig. 12 Scatter plot for chlorophyll ($> 5 \mu\text{g/L}$) in Suisun Bay. Ten-year model results are used for this analysis. The x-axis represents the flow rate from major rivers, while the y-axis represents NH_4 concentration. For each chlorophyll point, the corresponding flow rate is obtained by averaging the river flow over the past 2 weeks

explains why un-used NO_3 is exported from North Bay to Central Bay and the ocean (Fig. 10). The major WWTPs influencing the SFB nutrients are currently upgrading their sewage treatment infrastructures to reduce both the DIN concentration and the proportion of NH_4 in the discharge (Chappelle et al. 2019), which will provide an opportunity to further validate the interplay between river flow and NH_4 elucidated in this modeling study.

Variation in river flow may have other cumulative indirect effects on the ecosystem. Under low river flow conditions, less sediments are transported into the bay that could potentially improve the light condition and possibly increase chlorophyll concentration. However, the way that changing sediment transport influences chlorophyll concentration is still unknown as tides and wind waves also regulate SPM concentration (Bever and MacWilliams 2013). Incorporating a sediment transport model into our SFB ecosystem model may shed light on this issue. Conversely, high river flow can bring a large amount of sediment into SFB (Moftakhari et al. 2015), resulting in poor light conditions. This effect would be additive to high flow washout effects (Liu et al. 2018), which could further reduce chlorophyll concentration under high flow conditions.

Moreover, remineralization of benthic detritus in the CoSiNE model is simulated in the bottom layer with a large remineralization coefficient, which resembles a reflective boundary condition (Soetaert et al. 2000). In this way, most of the deposited detrital nitrogen returns to the water column as NH_4 . Sediment efflux of NH_4 is important in estuaries and in SFB (Cornwell et al. 2014) and may act to delay NO_3 uptake and phytoplankton biomass accumulation (Dugdale et al. 2016). Denitrification is omitted in the model because SFB is an oxic region (based on the WQDATA), and denitrification is inhibited under high oxygen concentration conditions (Gypens et al. 2008; Soetaert et al. 1996). Explicit modeling of benthic biogeochemical processes and their feedback to water column will involve a dynamic sediment flux model (Di Toro 2001; Wang et al. 2020), which is under development, and should potentially improve the model simulation of nutrients in SFB.

5 Conclusions

Many physical (river flow and turbidity) and biological (nutrients, phytoplankton growth and benthic grazing) factors are interconnected ecosystem drivers. Coupled physical-biogeochemical modeling integrates these drivers in a systematic approach to give a holistic understanding for the SFB ecosystem. We have applied the coupled SCHISM/CoSiNE model to the entire SFB, and conducted model simulations for a 10-year period to study the interannual variabilities of phytoplankton and nutrients and their relationships to river flow.

The model captured the SFB interannual variability with high model skills. The modeled temperature and salinity match the WQDATA observations from 2005 to 2014 with high correlation, and are consistent with the results in Chao et al. (2017b), which demonstrated that the SCHISM performs well in simulating physics in SFB. The CoSiNE model shows a strong correlation ($R = 0.49\text{--}0.79$) between modeled and observational chlorophyll in all subembayments, and the modeled NO_3 and NH_4 concentrations were also highly correlated ($R = 0.58\text{--}0.95$ and $0.58\text{--}0.85$, respectively), on both seasonal and interannual time scales.

The 10-year ecosystem simulation shows that high chlorophyll concentration persists in South Bay during warm months, whereas elevated chlorophyll displays strong interannual variability in North Bay. Spring and fall blooms are often observed in SFB model results but often do not reach the extra high levels 32 and 60 $\mu\text{g/L}$ reported by Glibert et al. (2014a) and Wilkerson et al. (2015), respectively. The 10-year simulation of nutrients, consistent with the USGS observations, reproduced the higher concentrations towards the more landward part of North Bay and lower South Bay, and the lowest values in Central Bay. In North Bay, nutrient concentrations were low in the high flow years 2006 and 2011, and the negative anomalies in these years resembled the low salinity anomalies discussed by Chao et al. (2017b), which indicates river influence on nutrient dilution and distribution in SFB. The nitrogen budget in Central Bay was dominated by DIN flux from North Bay and the flux into the ocean. The fluxes have strong interannual variation, attributed to the changes from the major river outflows. In 2006, the large river flow brought more nitrogen into Central Bay mainly in the form of NO_3 , which eventually entered the coastal ocean. The high flow also suppressed DIN transport from South Bay.

The results of two model experiments in Fig. 11 showed that benthic grazing in North Bay can potentially reduce phytoplankton concentration, but this is modulated by river outflow. The effect of benthic grazing was more significant during low flow conditions. Modeled occurrences of phytoplankton blooms in Suisun Bay were restricted to conditions with a flow rate range of 100–250 m^3/s and NH_4 concentration of 1–2 mmol/m^3 . This matches the simple model of Suisun Bay productivity linking flow and NH_4 (Dugdale et al. 2013), which showed that flow set the range of NH_4 concentration that allowed high phytoplankton biomass to develop. The model results suggest that for higher chlorophyll concentration in North Bay to occur, lower flows could be favorable, but not too low. The observation of blooms in North Bay associated with lower flow conditions was also reported by Glibert et al. (2014a) and Wilkerson et al. (2015).

Our analyses show that SFB ecosystem has distinct characteristics in different subembayments. North Bay is under strong influence of river outflows from the Sacramento River and San Joaquin River, while South Bay is a more stable

semi-enclosed system except under large flow events. The model shows the strong influence of river outflow on nutrient concentration inside SFB and nutrient transport to the coastal ocean. Overall, this study provides a comprehensive description for the SFB ecosystem on the decadal time scale by integrating most of the important physical and biological factors. The model highlights the interannual variabilities of chlorophyll and nutrients in SFB, and reinforces the importance of river flow on the ecosystem. As river flow management is a priority for California to improve the health of SFB ecosystem, this model can be useful in suggesting the flow criteria and evaluating effects of management practices. The knowledge gained from this study enriches our understanding of SFB about its response to future changes of river outflow, light, clam grazing, and nutrients. Finally, this study comparing model outputs with a large database of observations provides a new and higher level of validation and confidence in the application of the SCHISM/CoSiNE coupled hydrodynamic biogeochemical model for studying SFB ecosystem.

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