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IN SAN FRANCISCO BAY

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Integrated Watershed-Bay Modeling Strategy: Lower South Bay Dissolved Tracer Pilot Study

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CONTRIBUTION No. 1250 / March 2026

Preface

This report is a companion to two other reports that are being published at approximately the same time. Héberger and McKee (2025) reports on a strategy to improve watershed models to remain aligned and appropriately paced to support in-Bay dynamic and food web modeling of contaminants and sediment. Scheu et al. (2025) reports on development of a linked watershed-Bay model for sediment and PCBs in San Leandro Bay. This report complements the other two reports, describing a linked watershed-Bay model for dissolved constituents in Lower South Bay.

Suggested Citation:

King, A., McKee, L., McWilliams, S., Scheu, K., Jones, C., Avellaneda, P., Héberger, M., Yee, D., and Davis, J.A. 2026. Integrated Watershed-Bay Modeling Strategy: Lower South Bay Dissolved Tracer Pilot Study. SFEI Contribution #1250. San Francisco Estuary Institute (SFEI), Richmond CA.

Acknowledgements

Ezra Miller and Rebecca Sutton (SFEI) provided invaluable expert advice about comparing model results to measured contaminant concentrations. This product was supported by funding from the San Francisco Bay Regional Monitoring Program Supplemental Environmental Project (R2-2022-1004).

Executive Summary

Study Motivation

Understanding how watershed contaminants affect water quality in San Francisco Bay is key to addressing management questions from the Regional Monitoring Program for Water Quality in San Francisco Bay (RMP). By combining watershed and Bay models with monitoring data, we can better grasp watershed-Bay connections, support actions to reduce contaminant impacts and increase sea level rise resilience and habitat protection, and optimize use of monitoring resources. The objective of this pilot study was to explore the coupling of the watershed and Bay models as a test case for examining a set of possible management questions (MQs) related to the dispersion of watershed-derived dissolved phase contaminants in the Bay.

Study Design

The Stevens Creek-Bay interface was selected for evaluating the strengths and weaknesses of existing data and models due to the availability of data for emerging contaminants, slower tidal mixing in the Lower South Bay compared to other subembayments, and low inflows from other parts of the Bay. The output from the Watershed Dynamic Model (WDM), a regional watershed model developed by the RMP to simulate streamflows and contaminant loads in Bay Area watersheds, was used to generate boundary conditions for the in-Bay hydrodynamic model, which was based on the Nutrient Management Strategy (NMS) open Bay model.

For this study, improved bathymetry data were incorporated into the NMS open Bay model, with an increased grid resolution extending into the tidal channels of watersheds. The model was then set up, run, and evaluated for Water Year (WY) 2023. The WY 2023 hydrodynamic model output was then used to conduct a numerical conservative tracer study to explore the contribution of water from various sources at sites across the Lower South Bay (LSB), and idealized model runs were conducted to examine the fate of a tracer pulse entering LSB from Stevens Creek during a storm.

Responses to Management Questions

MQ1 For different parts of the Bay and times of the year, what are the relative contributions of contaminants to Bay concentrations from wastewater treatment plants (WWTPs), LSB tributaries, other Bay tributaries, and Central Valley (Sacramento and San Joaquin River) inputs?

We addressed this question using results of a simulation based on WY 2023 conditions, examining the source of water (termed provenance) at site P34, a long-standing RMP monitoring location in the main channel of the LSB. Water provenance at P34 varied significantly over the water year. Stevens Creek contributed minimally (ranging from 0-3.5%) year-round, the contribution from WWTPs was also small (5-8%), as was the

contribution from other tributaries north of the Dumbarton Bridge (<1-5%). The largest and most variable freshwater contributions came from the major LSB tributaries, especially Guadalupe River and Coyote Creek, which could contribute up to 54%, and Central Valley Rivers which peaked at 30% in May and June before dropping to about 20% in the summer. Depending on the time of the year, between 35-75% of the water at site P34 came from the ocean.

MQ2 Is there a steady-state baseline established during the wet season or a build-up and/or dilution phenomenon due to successive runoff events?

No steady-state baseline was established for any freshwater sources during the wet season. Flow from WWTPs was consistently diluted by other freshwater sources during storms. Other sources showed increases during the wet season, with variations depending on their size and location. Contributions from Stevens Creek and other LSB tributaries peaked after each rainstorm, particularly after the largest storm of the year, and gradually diminished over the next few months. Although no steady state was reached, multiple consecutive storms maintained higher contributions from these watersheds. Tributaries north of the Dumbarton Bridge also showed increased contributions with subsequent storms. The Central Valley River contribution was highest during large storms but generally increased through the wet season, peaking in June after a significant Delta flow in March and sustained high flows through May.

MQ3: What are the maximum short-term concentrations of dissolved phase contaminants [anytime] during the wet season in different habitats of interest (channel, mudflat, subtidal), where in the Bay are these maximum concentrations, and how long do these elevated concentrations persist before returning to “background”?

Based on the idealized simulation of a conservative tracer release from Stevens Creek, the highest concentrations of the tracer were predicted to be found in the tidal channels on the mudflat. Within the first 24 hours, concentrations in these channels could reach 100% of those in terrestrial stormwater but would dilute rapidly to <5% by 7 days post-storm. On the mudflats, concentrations were predicted to peak at over 50% within 24 hours, then decline to around 5% or less by 7 days across all mudflat areas. In subtidal sites, concentrations could briefly reach near 100% within the first 6 hours but would decline to around 10% within 24 hours and <5% by 4 days. Generally, concentrations were expected to be <5% at all sites by 7 days post-storm release.

MQ4: How should we sample in the Bay to capture the dispersion gradients away from treated wastewater discharges and tributary inputs at different locations in the Bay and times of year, to effectively characterize local exposure risk, and model fate and transport of various legacy and emerging contaminants?

The proposed sampling program for dissolved-phase contaminants will depend on the relationship between peak terrestrial stormwater concentrations and contaminant detection limits in Bay water. If detection limits are not an issue, and Stevens Creek is representative of other stormwater-Bay interfaces, the program could involve sampling in the tidal channel and adjacent mudflats within 72 hours of the storm peak, and at subtidal and Bay spine sites (Sampled using the R/V David H. Peterson USGS research vessel) ideally within 7 days. However, if detection limits pose a challenge, a faster sampling program would be required. This would involve sampling the tidal channel and mudflats during flood tides within 24 hours of the storm peak, and subtidal and Bay spine sites within 72 hours.

Comparison of Observed and Simulated Contaminant Concentrations

For a pilot comparison of observations with model results we chose hexamethoxymethylmelamine (HMMM) measured during the 2021 - 2024 pilot study (Miller et al. 2025). HMMM is a cross-linking agent used in the production of rubber and coatings, is grouped under vehicle-derived contaminants in the RMP tiered risk-based framework for emerging contaminants, and is present at high concentrations in stormwater. It is primarily a dissolved phase contaminant and likely has low concentrations in wastewater, the Delta, and the ocean. HMMM is known to decay with a moderate half-life (estimated to be 23-67 days) so its ambient concentration relative to a modeled conservative tracer is likely to be reduced. Nonetheless, it provides useful empirical data to compare to expectations for the modeled ideal tracer. For comparison, according to our model, the time for mass of a conservative tracer released from Stevens Creek to fall to one half the original mass within Lower South Bay is 8 days.

Across five storm events, in which HMMM concentration was measured in the tidal portion of the Stevens Creek channel during or shortly after the storm, and HMMM concentration was measured in the subtidal channel of Lower South Bay (Peterson sites 33 and 34) between 6 and 26 days after the storm, the open-water concentrations ranged from fractions of 0.10 to 0.29 compared to the stormwater concentration. This is within the range of model water fraction attributed to Lower South Bay tributaries at USGS Peterson Site P34 which is approximately 0.10 to 0.45. Note these model fractions are attributed to all Lower South Bay tributaries, not just Stevens Creek. The model Stevens Creek fraction alone ranged from only 0.01 to 0.04. This comparison of observed and simulated concentrations confirms our model finding that water in the subtidal region of Lower South Bay is composed of many sources, not just the closest tributary.

Implications for Monitoring Design

The RMP Status and Trends sampling design for near-field (locations near expected loading pathways) wet weather sampling includes stations in Stevens Creek (S01) and the USGS P34 station in the open Bay. As expected, during a runoff event, and for several tidal cycles following the event, modeled tracer concentrations in the Stevens Creek channel and locations near the mouth of the channel are dominated by the loading from

Stevens Creek. Modeling results over subsequent days indicate a reduced influence from Stevens Creek, and greater influence of a mixed signal integrating nearby local tributary and wastewater inputs combined with dilution from the South Bay. Concentrations at P34 similarly represent a regional mix of inputs; for many contaminants of interest, we would expect the concentrations discharged to be similar among surrounding watersheds, as the mix of watershed characteristics and land uses are similar. If there were a contaminant unique to Stevens Creek, it would be most easily measured at the in-channel station and the intertidal zone near the mouth during a discharge event and for a few tidal cycles after. However, by the time the discharge reaches P34, the signal could be diluted to concentrations that might not be easily detected. The modeling also highlights that monitoring of in-channel tidal areas may suffice for estimating localized short-term concentrations and exposure risk, given the challenges of sampling mudflats around tributary mouths due to difficult access during inclement weather. Once the discharge reverts to baseflow, the concentrations at the channel mouth at low tide and in the channel during the following high tide are of a similar magnitude. A continued focus on monitoring tidal areas of urban creeks and continued refinement of models is recommended to estimate exposure fields for relevant habitats and beneficial uses. However, if stakeholders desire more certainty, a modified sampling approach could be used for open water areas if there is a need to focus on inputs from a specific tributary rather than just a mixed regional signal. Such monitoring in open water areas would need to occur nearer the tributary input and within a few days after a discharge using a reactive on-demand logistically difficult sampling plan, rather than relying on sampling with a fixed schedule that could end up several weeks after a storm.

Study Limitations

This study was limited to dissolved phase contaminants only. During the pilot study we became aware of some model limitations that could be addressed in future studies. The first is that the relative contributions of various sources to the water volume in the Lower South Bay has an eight-month spin-up time due to distance from the Delta and the ocean sources. Future model runs could include an 8-month spin-up period instead of the 2-month spin-up period we usually use for salinity and nutrients. Second, the upland portions of the tributary channels were not adequately resolved by the model grid. We found that this matters only to predictions of concentrations within the stream channels, not on the mudflats, and not within the Bay proper. Future studies should include finer resolution in the tidal portion of the tributaries if contaminant concentrations within this region are of interest.

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1. Introduction

Learning about how watershed contaminants impact water quality in San Francisco Bay is crucial for addressing management questions identified by the Regional Monitoring Program for Water Quality in San Francisco Bay (RMP). By combining watershed and Bay modeling tools with monitoring data, we can better understand watershed-Bay connections and inform actions to reduce the impacts of contaminants including PCBs, mercury, nutrients, and contaminants of emerging concern (CECs). This approach also supports sediment management for sea level rise resilience and habitat protection. With 30 years of monitoring data, the RMP has informed many water quality improvements but as monitoring costs increase and management questions become more spatially and temporally specific and future-focused, modeling also provides a cost-effective way to learn how to optimize monitoring resources.

There are many management questions (MQs) relating to how contaminants enter the Bay and cause biotic exposure. As part of this pilot study, the authors met with Richard Looker at the Water Board and Tom Mumley to discuss the types of questions that a coupled watershed-Bay modeling framework could answer, and then Richard Looker and Tom Mumley prepared a 2-page draft that outlined 10 questions and a set of sub-questions. These questions have not been reviewed by the RMP committees, but provide a good starting point for the Emerging Contaminants Workgroup (ECWG) and Microplastics Workgroup (MPWG) to discuss. In this pilot study, we focused on a few examples rather than addressing the more exhaustive list. The questions were grouped into two types: (1) questions that link external sources with Bay processes to characterize seasonal contaminant concentrations (time averaged) that might be more akin to a chronic exposure scenario, and (2) questions that relate the potential for field sampling designs to capture the short-term response of the Bay to storm events (event-focused gradient and dispersion questions) that might be more akin to an acute exposure scenario.

Time-averaged provenance (water source contribution) and exposure questions

- MQ1: For different parts of the Bay and different times of the year, what are the relative contributions to Bay concentrations from Lower South Bay (LSB) tributaries, wastewater treatment plants (WWTPs), other Bay tributaries, and Central Valley (Sacramento and San Joaquin River) inputs?
- MQ2: Is there a steady-state baseline established during the wet season or a build-up and/or dilution phenomenon due to successive runoff events?

Event-focused gradient and dispersion (dilution) questions

- MQ3: What are the maximum short-term concentrations of dissolved phase contaminants during the wet season in different habitats of interest (channel, mudflat, subtidal), where in the Bay are these maximum concentrations, and how long do these elevated concentrations persist before returning to “background”?
- MQ4: How should we sample in the Bay to capture the dispersion gradients away from treated wastewater discharges and tributary inputs at different locations in the Bay

and times of year, to effectively characterize local exposure risk, and model fate and transport of various legacy and emerging contaminants?

The answers to these four questions may be unique for contaminants with differing source characteristics. A few examples are:

1. Contaminants that build up in watersheds over the dry season and are flushed out into the Bay during storms with lower concentrations during later or larger winter season storms,
2. Contaminants that are mobilized all winter season during every storm, and
3. Contaminants that are associated with dry weather or groundwater flow.

Until data are collected repeatedly at fixed stations strategically located in selected watersheds over a range of storm and dry weather conditions, we will not be in a position to empirically investigate how contaminants disperse into the Bay on a seasonal basis. In the meantime, in the absence of these data, dynamic models (those that simulate processes in relation to time) can be used to explore these questions conceptually.

In 2017, the SFEI Bay Area Hydrological Model (BAHM) was coupled with the Nutrient Management Strategy (NMS) “Open Bay” three-dimensional hydrodynamic model via 44 pour points at the perimeter of the Bay model domain (Holleman et al., 2017). BAHM provided daily freshwater flows at these 44 pour points to the first WY 2013 and WY 2017 Open Bay hydrodynamic model runs. Later on, BAHM was replaced by the Watershed Dynamic Model (WDM) (SFEI, 2020; 2023). WDM is higher resolution than BAHM but for the NMS modeling work it is aggregated to the original 44 BAHM pour points. At this time, WDM predicts daily freshwater flows from calendar years 1994 - 2023 and the NMS hydrodynamic model has been updated and run from WY 2013 through WY 2023 (King et al., in preparation).

Although dynamic models for San Francisco Bay and its watershed exist and are synchronized in time for WY 2013 to WY 2023, the watershed model is currently calibrated through to WY 2024 whereas the Bay model is not. In addition, the watershed model has been calibrated on a daily time step. To explore the interaction between watershed flood flows and tides in the Bay using a Bay model, we used an hourly output from the watershed model.

The previously published Bay model runs used daily inflows from the watershed model at 40+ pour points which are located at the edge of our model grid, which is below the head of tide. To run watershed-Bay simulations to explore tidal channels and upper mudflat habitats, the models needed to be connected with new pour-points that connect through a higher resolution grid with more finely resolved bathymetry. These factors provide examples of limitations to the immediate application and effectiveness of the watershed models coupled with Bay models for addressing specific management needs.

The objectives of this pilot study were to investigate the coupling of the watershed and Bay models as a test case for exploring management questions associated with the

dispersion of watershed-derived dissolved phase contaminants in the Bay. Insights gained from this pilot, along with findings from a similar effort focused on sediment and PCBs in San Leandro Bay (McWilliams et al., 2024; Scheu et al., 2025) have informed the refinement and finalization of the accompanying watershed-Bay modeling strategy and workplan (Héberger et al., 2025).

2. Methods

2.1 Selection of the Pilot Study Area

This pilot study in the Stevens Creek area of LSB provides a valuable opportunity for local-scale integration of RMP watershed flow modeling with in-Bay fate modeling.

Stevens Creek was chosen for three main reasons:

1. The existence of contaminant (e.g., HMMM) data measured during a winter-season storm in the watershed, a few days later in the tidal area of the Stevens Creek-Bay interface, and a few weeks later out in the open Bay;
2. The recognition that the LSB has slower tidal mixing and therefore offers a better chance of obtaining detectable ratios of watershed freshwater flows mixed with the Bay water volume than the other more rapidly mixed margins of South and Central Bay; and
3. The constrained nature of the geography of the LSB with lesser inflow from other parts of the Bay.

Given these valuable attributes, this pilot study done at the interface of Stevens Creek and the Bay establishes a foundation for exploring the weaknesses and strengths of the existing data and models to answer management questions of interest. Given these unique Lower South Bay properties, we note that the results may not be representative of tributaries to other parts of the Bay where mixing with open Bay water differs.

2.2 Watershed Model Setup

Output from the Watershed Dynamic Model (WDM) was used as boundary conditions to the hydrodynamic model (discussed below). The WDM is a regional watershed model developed by SFEI staff for the RMP that simulates streamflows, sediment, and water quality in the Bay Area watersheds (Zi et al., 2021, 2022).

2.2.1 Watershed Hydrography

The watershed draining into the Lower South Bay (LSB), defined as the area of the Bay south of Dumbarton Bridge, mainly emanates from Santa Clara County. The drainage area includes Coyote Creek, Guadalupe River, Stevens Creek, San Tomas Aquino Creek, and other smaller streams that discharge to the Bay (Table 1 and Figure 1). These watersheds encompass both urban and rural areas. Urbanization in the lower portions of all these watersheds, particularly in San Jose, has increased impervious surfaces, impacting the natural flow regime and water quality. This increase has been primarily due to agricultural and grazing land being converted into residential communities in the

southern region of the Santa Clara Valley. However, agriculture and open space are still found in some parts of that southern region, and steeper portions of all the tributaries remain relatively undeveloped. On average Guadalupe River produces the largest proportion of total annual average flow (29%) from local tributaries into the LSB, and Stevens Creek ranks 5th, producing 7%. Collectively, five watersheds produce 77% of the annual average flow and >90% emanates from 9 watersheds (Table 1). These all discharge to the southern and southwestern shorelines of the LSB. In contrast, the eastern shoreline tributaries are estimated to produce just 10% of the total annual average flow.

Table 1. Basic properties of the sub-watersheds considered for the Lower South Bay model.

Subwatershed ID (WDM)	Stream	Outlet #	Drainage Area (km ²)	Directly Connected Impervious Area (%)	Simulated average daily streamflow (m ³ /s)
4200	Guadalupe River	6	442	26	6.08
4000	Coyote Creek	5	830	9	4.91
4300	San Tomas Aquino Creek	7	114	35	2.00
4900	San Francisquito Creek	14	18	41	1.59
4500	Stevens Creek	10	82	20	1.49
4322	Calabazas Creek	8	37	59	0.82
3800	Mowry Slough	1	49	56	0.79
4600	Permanente Creek	11	48	24	0.65
4120	Berryessa Creek	4	74	33	0.62
3900	Agua Caliente Creek	2	63	31	0.45
4312	Sunnyvale East Channel	9	53	47	0.44
4700	Adobe Creek	12	37	25	0.43
4800	Matadero Canal	13	31	27	0.34
4110	Agua Fria Creek	3	23	30	0.19

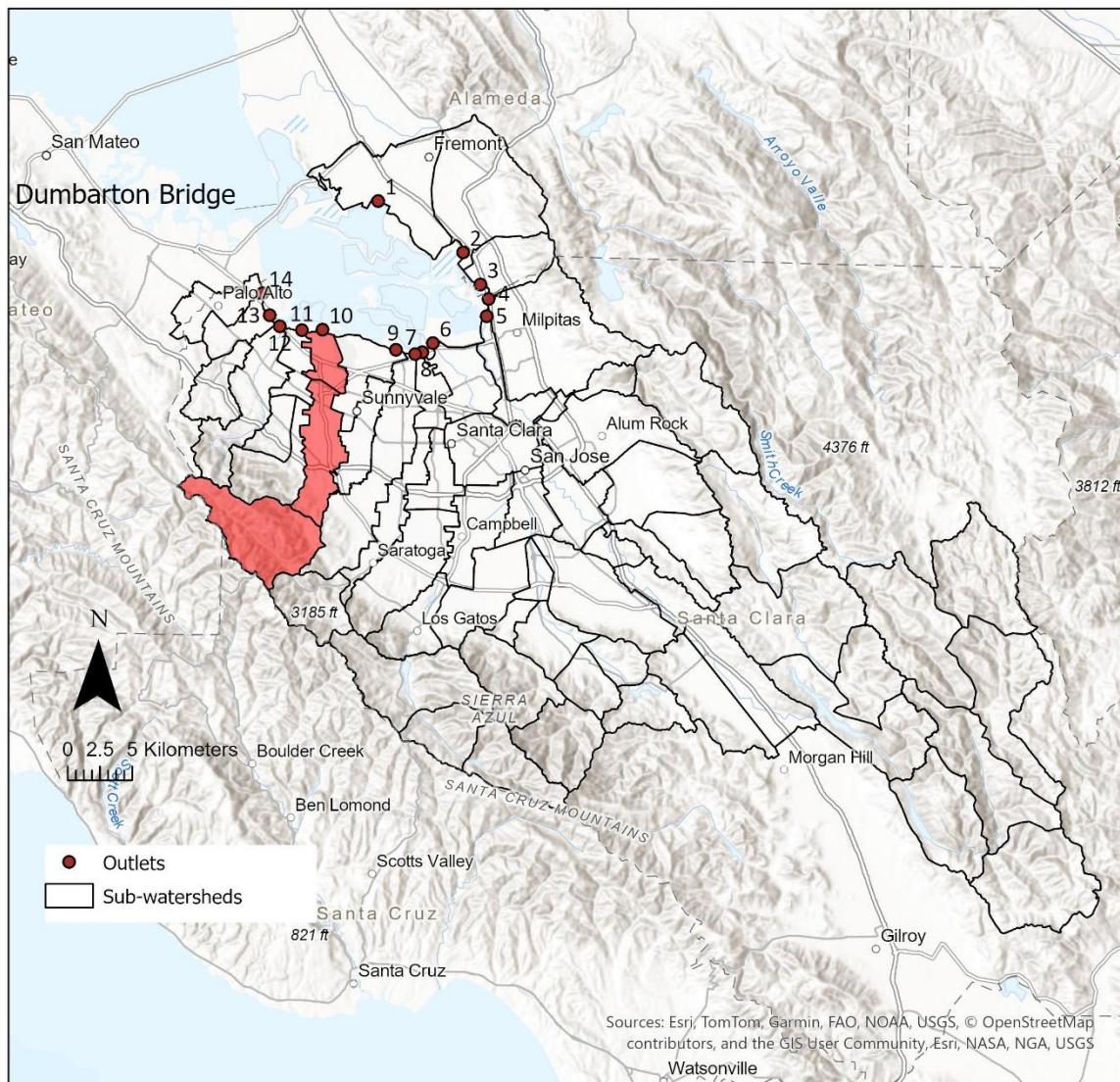


Figure 1. Locations of the sub-watersheds and outlets (red dots) that were considered for the Lower South San Francisco Bay model. The Stevens Creek sub-watershed is highlighted in red.

2.2.2 Model Structure

The RMP has conducted hydrological modeling in the LSB watershed as part of the regional watershed dynamic model (WDM) (Zi et al., 2021, 2022). The WDM has been calibrated and validated for water and sediment at the regional scale and generates streamflow and suspended sediment loads with a daily time step; PCB and Hg model development is in progress. The modeling work utilizes the Loading Simulation Program in C++ (LSPC) as its computational engine. LSPC is a watershed modeling framework designed to simulate watershed hydrology, sediment transport, and water quality

processes originating from upland contributing areas and receiving streams. The WDM output was used to provide streamflow inputs to the in-Bay hydrodynamic model.

LSPC simulates surface runoff as a function of precipitation, evapotranspiration, infiltration, soil moisture change, and groundwater flow. In the LSPC model, a watershed is represented by different storage units. These storage units are connected and simulate the water cycle based on a water balance equation. The upper zone storage unit is defined as the water that is available for direct evaporation as opposed to transpiration by plants (i.e., surface-depression storage plus shallow-soil moisture). When water infiltrates, it moves down to the lower zone storage unit and gets stored as groundwater. The major pathways to streamflow are overland flow, interflow (i.e., lateral movement of groundwater), and groundwater storage. The described water balance is evaluated for each hydrologic response unit (HRU). HRUs are areas with similar land features, slope, and soil characteristics. Potential evapotranspiration was calculated using the Penman approach (Mahrt and Ek, 1984), and infiltration was estimated using Philip's equation (Philip, 1969). The streamflow estimated at a sub-watershed level was then routed through the river systems using a kinematic wave model.

Based on topography and the stream network, the drainage area was divided into 62 sub-watersheds (Figure 1), and 14 watershed outlets represented pour points to the in-Bay hydrodynamic model. We adopted the 55 HRUs developed for the WDM (Zi et al., 2022, 2021). To develop the hydrologic response units, SFEI developed the 'Old' (before 1968 for industrial and 1974 for commercial and residential) and 'New' urban land uses based on a layer prepared by the Metropolitan Transportation Commission (MTC) in 2020 (Figure 2). Most industrial and commercial areas are found in the watershed's downstream portions near the margins of the LSB (Figure 3).

2.2.3 Surface Hydrology

Meteorological inputs included precipitation, potential evapotranspiration (PET), temperature, dew point temperature, wind speed, cloud cover, and solar radiation. Most weather inputs were derived from PRISM (Parameter-elevation Relationships on Independent Slopes Model) climate data and the North American Land Data Assimilation System Version 2 (NLDAS-2) data (Zi et al., 2022, 2021). We processed the meteorological information from 10/01/2013 to 10/01/2023 and generated streamflow at the watershed outlets for the Water Year (WY) 2023.

We evaluated model performance using observed streamflows at four USGS stations. The discrepancy between observed and simulated streamflow values was measured using the Nash-Sutcliffe efficiency (NSE) and the coefficient of determination (R^2). NSE varies from $-\infty$ and 1, with $NSE = 1$ indicating a perfect fit. $NSE < 0$ indicates that the observed mean streamflow is a better predictor than the model output. We used the following NSE thresholds to characterize model performance: $NSE > 0.8$ ($R^2 > 0.85$), very good; $0.7 < NSE < 0.8$ ($0.70 < R^2 < 0.85$), good; $0.5 < NSE < 0.7$ ($0.50 < R^2 < 0.70$), satisfactory; and

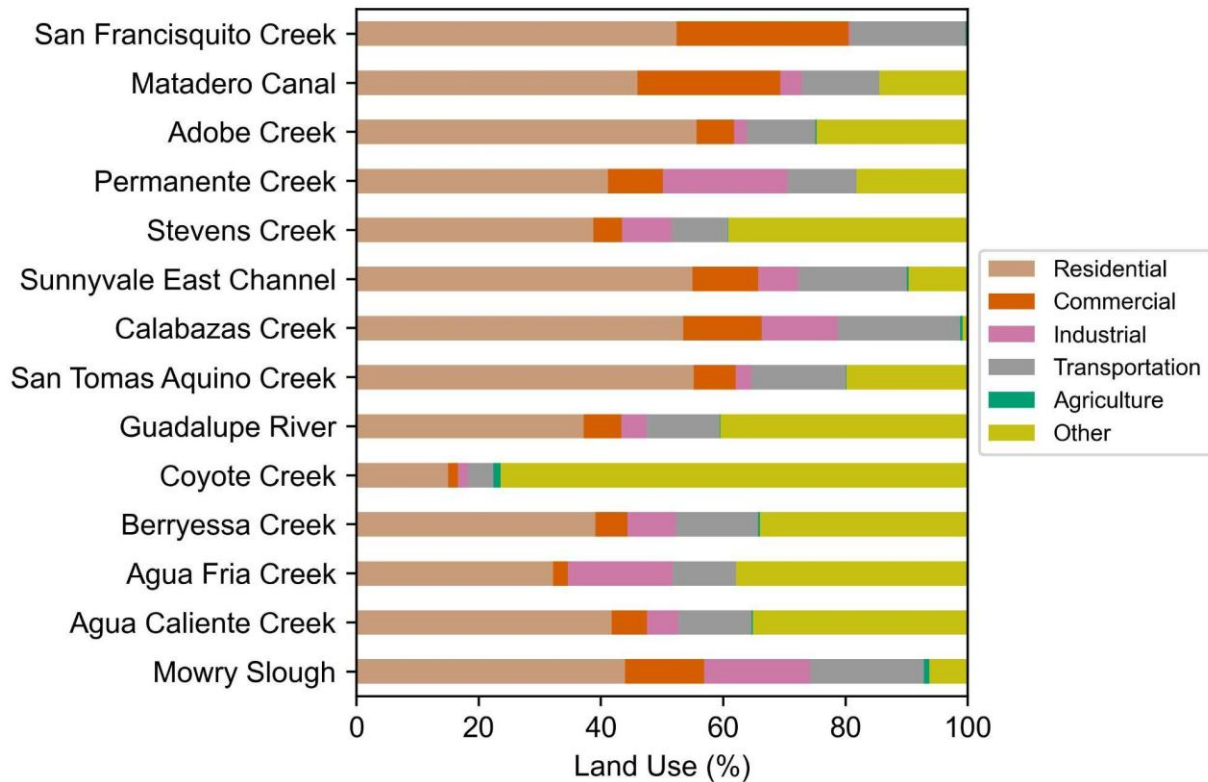


Figure 2. Land uses across the model domain. Forest, grassland, wetlands, and water bodies are grouped into the “Other” land use category.

NSE < 0.5 (R^2 < 0.50), poor (Moriassi et al., 2015). The highest model performance was observed at the USGS station on the Guadalupe River (Table 2, Figure 4, and Figure 5). The lowest model performance was observed at the USGS station on Coyote Creek. SFEI staff is continuing to update meteorological forcing data and hydrologic parameters to improve model performance.

Flows from the watershed model were output at an hourly time scale to support boundary conditions for the Bay model. Although the watershed model was calibrated at a daily time scale, the resulting hourly flows have not been fully evaluated.

For this pilot study, this limitation was not considered problematic. The time-averaged provenance (water-source contribution) analyses focused on processes operating over monthly time scales, for which precise hourly timing was not critical. In addition, for the event-focused gradient and dispersion (dilution) analyses, we intentionally adjusted the timing of watershed inflows to align with tides. As a result, and discussed more later, known weaknesses in the watershed model did not materially affect the study results or conclusions.

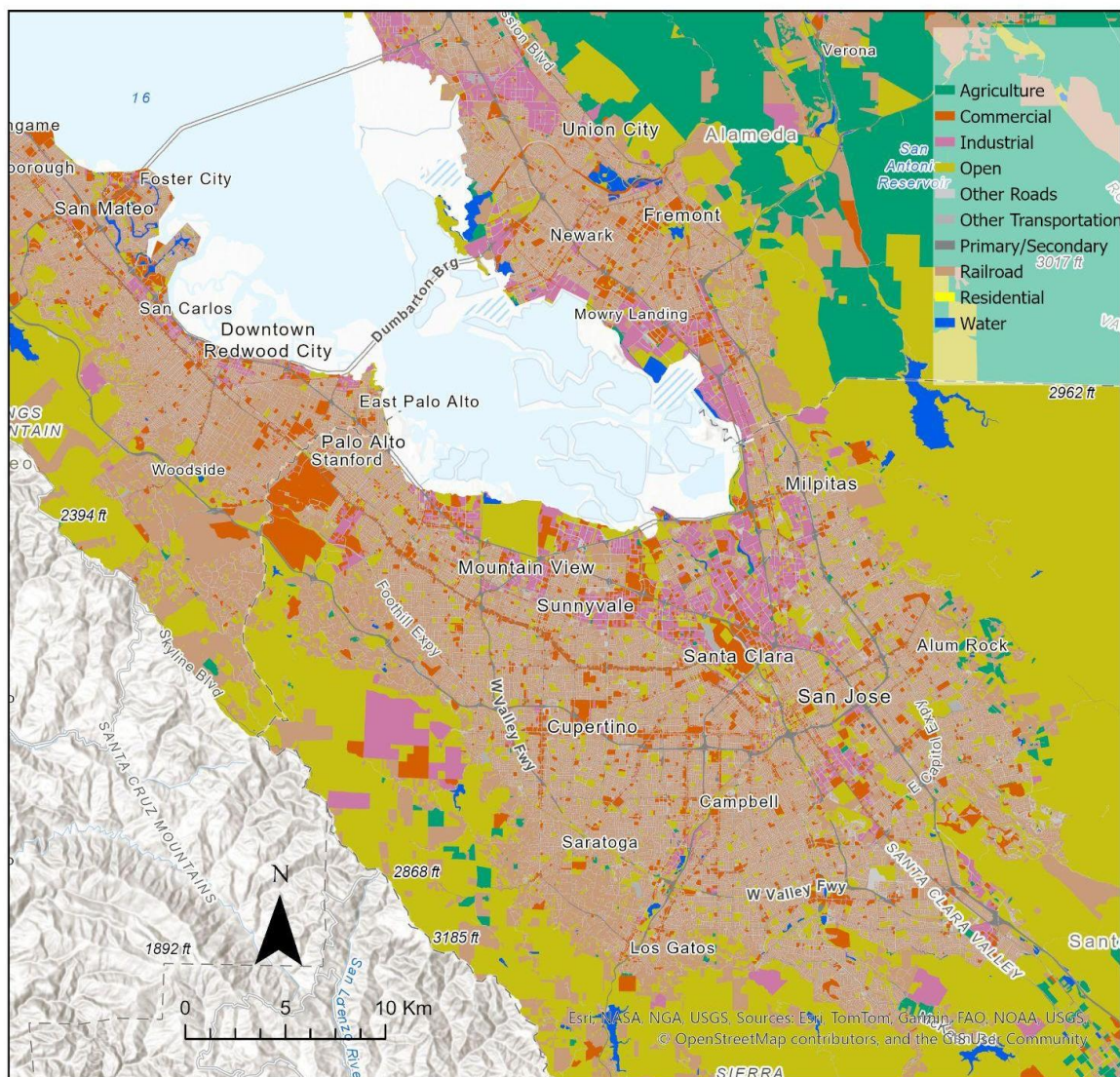


Figure 3. Land use distribution across the watershed model domain, zoomed to show the South Bay. “Primary/Secondary” refers to interstate and state highways.

Table 2. Model performance at calibration stations (daily time scale).

Stream	USGS Station	USGS Record	Calibration Period	NSE (-)	R ² (-)
Coyote Creek	11172175	1999-2024	1999-2023	0.30	0.41
Guadalupe River	11169025	2002-2024	2002-2023	0.61	0.80
Matadero Canal	11166000	1994-2017	1995-2017	0.57	0.61
San Francisquito Creek	11164500	1994-2024	1995-2023	0.67	0.68

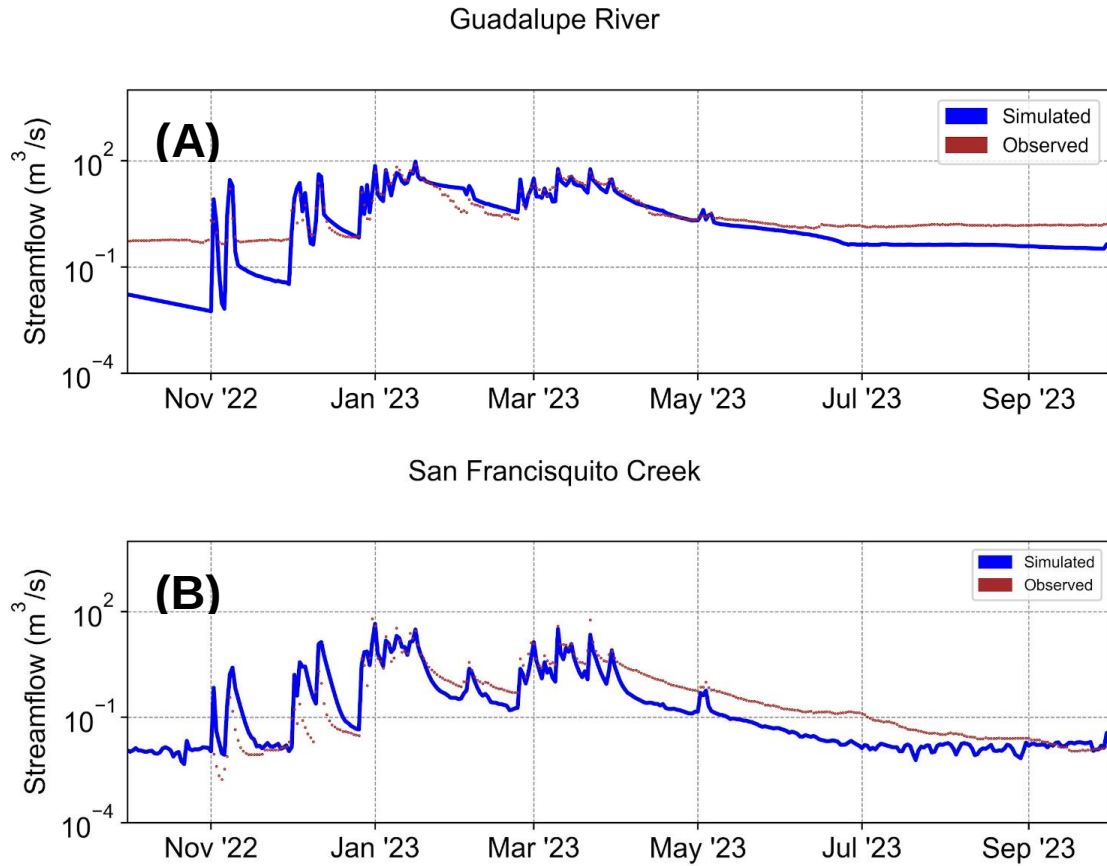


Figure 4. Simulated and observed streamflows for the Guadalupe River and San Francisquito Creek.

2.3 Bay Hydrodynamic Model Setup for Numerical Tracer Studies

The in-Bay hydrodynamic model was based on the open Bay model developed for the Nutrient Management Strategy (NMS) as described in Holleman et al. (2017), Nuss et al. (2018), SFEI (2020), and SFEI (2023). Note we did not use the newer version of the model described in King et al. (in preparation) because it was still in development at the beginning of the Stevens Creek project. For this project, we modified the NMS open Bay model grid within the LSB and set up, ran, and evaluated the modified model for WY 2023. The WY 2023 hydrodynamic model output was then used to conduct a tracer study to explore the contribution of water from different sources at sites across LSB. In addition to the WY 2023 model based on real-world conditions, we modified the WY 2023 model to conduct two idealized runs to examine the fate of a pulse of tracer entering LSB from Stevens Creek.

In this section we first give a broad overview of the existing NMS open Bay model, referring the reader to prior reports for greater detail. Then we describe the changes we

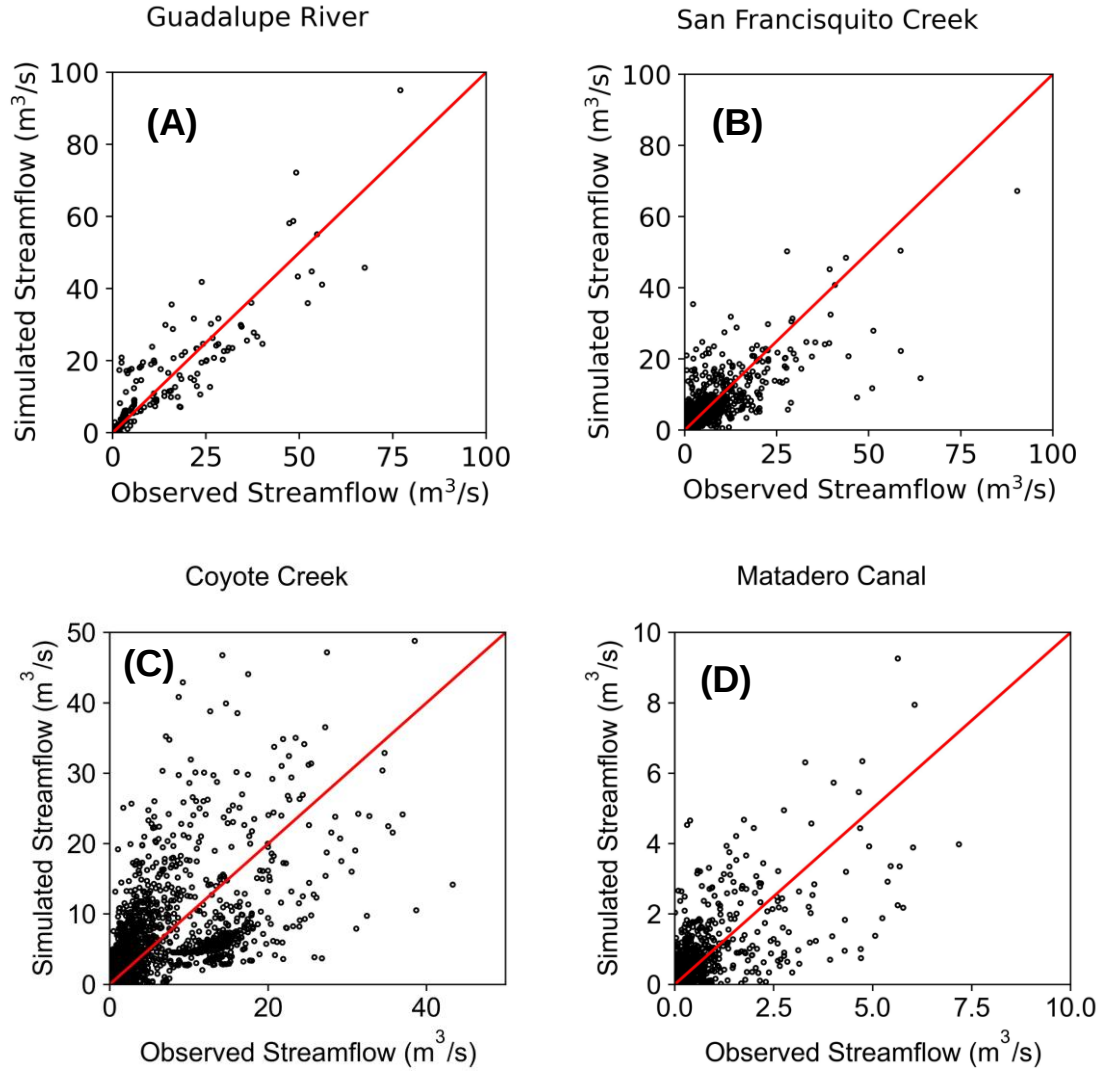


Figure 5. Scatterplots of simulated and observed streamflows for the calibration watersheds. Each panel shows simulated and observed streamflows for the calibration periods displayed in Table 2.

have made to the grid and present the model forcing and performance evaluation for WY 2023. Finally, we describe the tracer study based on the WY 2023 hydrodynamic model results as well as the two idealized hydrodynamic runs and tracer studies focusing on Stevens Creek.

2.3.1 NMS Open Bay Hydrodynamic Model

The NMS open Bay hydrodynamic model is built on the open-source modeling platform D-Flow Flexible Mesh (DFM). DFM is a finite-volume, three-dimensional, unstructured grid hydrodynamic model (Deltares, 2023a). The unstructured grid was ideal for this project since it allows for efficient and flexible resolution of flow features ranging from the tidal

portions of the local tributaries of the LSB up to a regional representation of the coastal ocean. Vertical gradients are resolved using 10 sigma layers – the vertical layer thickness adjusts to water depth over time and space as a percentage of water depth. Processes include flow induced by water level gradients as well as horizontal gradients in water density (barotropic and baroclinic), bottom stress, and wind stress (but not wind waves). Turbulent mixing in the vertical direction is parameterized with a k-epsilon model (Pope, 2000). Salt and heat are each transported via an advection-diffusion equation. Evaporation, precipitation, and heat transfer occur in the surface cell except for shortwave radiation which penetrates the water column.

The NMS open Bay model domain covers San Francisco Bay, including portions of Coyote Creek and Guadalupe River at the southernmost extent of the Bay, and extending north to the Sacramento and San Joaquin Rivers at Rio Vista and Jersey Point, respectively (Figure 6). Model forcings include the tidal ocean water surface elevation, direct precipitation, evaporation, stormwater runoff, treated wastewater discharge, tidal Delta flow, wind, humidity, air temperature, and cloudiness. From these inputs, DFM calculates water level, salinity, temperature, velocity, and vertical turbulent diffusivity (the rate at which turbulence smooths vertical gradients in scalar concentration) throughout the Bay. The locations of treated wastewater discharges are labeled in Figure 6. Pour points for stormwater inflows are shown as triangles in the same figure. These pour points were created to correspond to the Bay Area Hydrologic Model (BAHM) (Clear Creek Solutions, 2014) which was used to generate daily inflows in the first two NMS open Bay model runs (Holleman et al., 2017; SFEI, 2020). For subsequent runs, the BAHM was dropped and the WDM was used to generate daily inflows and WDM output was aggregated to correspond to the BAHM pour points (SFEI, 2020). We refer the reader to Holleman et al. (2017), SFEI (2020), and SFEI (2023) for detailed descriptions of all model boundary conditions.

Model performance was evaluated using water level data from NOAA moorings, salinity data from USGS and SFEI moorings, salinity transects taken using the USGS R/V David H. Peterson research vessel (hereafter referred to as “the USGS Peterson”), and for WY 2013, depth-averaged velocity data from NOAA's acoustic Doppler current profilers. We refer the reader to Holleman et al. (2017), SFEI (2020), and SFEI (2023) for an overview of model performance for water level and salinity and to Holleman et al. (2017) for performance for velocity. Since tidal exchange under the Dumbarton Bridge is crucial for the present tracer fate and transport study, we highlight the model performance for the current speed at this location. The observed and modeled current speeds have been depth-averaged and compared in the principal and secondary directions (perpendicular to the principal) (Figure 7). Figure 7 shows the directions of the observed and modeled current speeds, respectively. The principal (along-stream) current speed is positive in the direction of the arrows, and the secondary (across-stream) current speed is positive in the direction perpendicular and to the right of the arrows. The scatter plot shows observed vs. modeled principal current speeds. See Holleman et al. (2017, p. 11-12) for definitions of the following statistics.

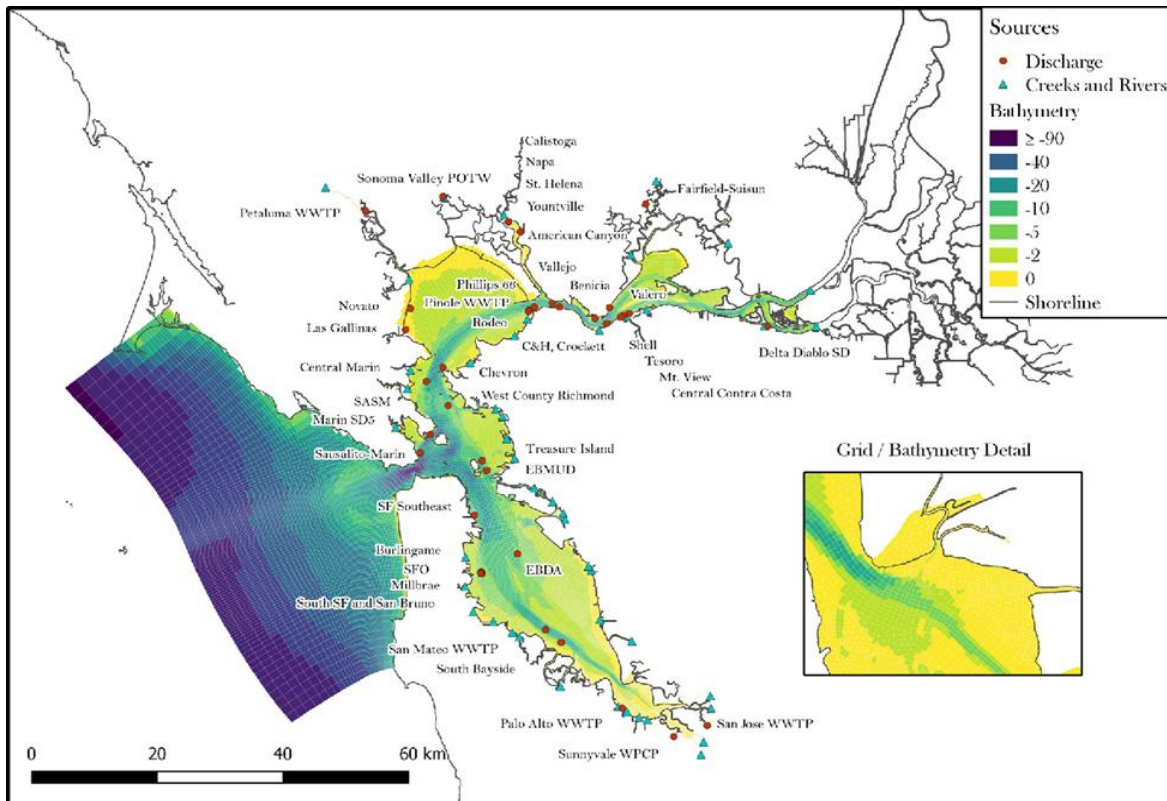


Figure 6. The NMS open Bay hydrodynamic model mesh as well as locations of treated wastewater discharge and tributary inputs from the hydrologic model.

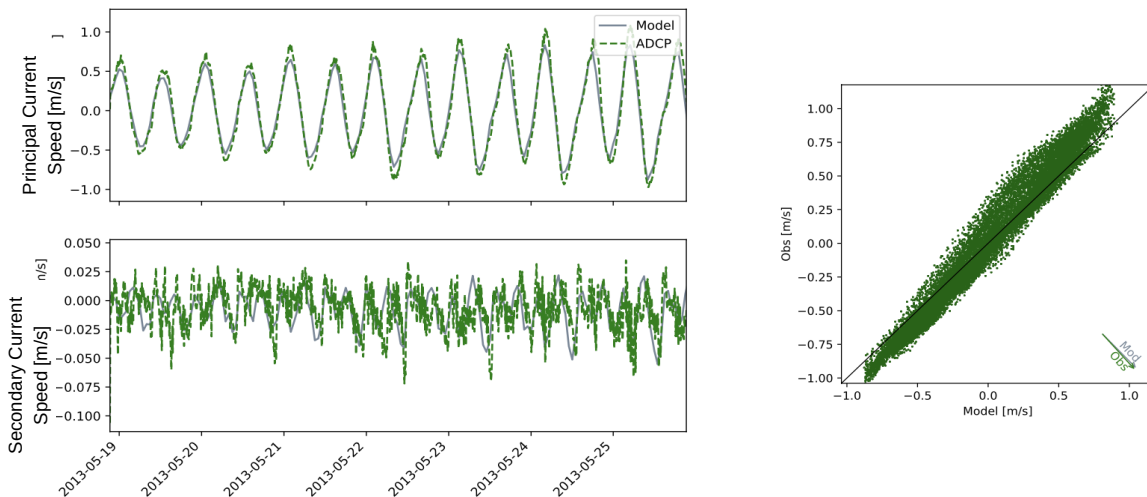


Figure 7. Comparison of measured and modeled depth-averaged current speed at the Dumbarton Bridge during a portion of WY 2013 when an ADCP was deployed by NOAA. The green and gray arrows in the lower right corner of the scatter plot on the right are not meant to share the axis with principal current speed – they simply point to compass direction, i.e., they both point to the southeast.

For principal current speed, the Willmott skill score was 0.985 (Willmott, 1981), the squared correlation coefficient was 0.968, bias was -0.013 m/s, root mean square error was 0.119 m/s, lag was -15.6 min, and amplification factor was 0.85. This is considered very good performance, for example MacWilliams et al. (2016) define skill greater than 0.975 as “accurate” for current. However, an amplification factor less than one implies a slight underprediction of the tidal current speed oscillation amplitude (amplification factor is the ratio of the standard deviations of modeled and observed time series). The link between modeled tidal current speed amplitude and net transport across the Dumbarton Bridge depends on mixing efficiency on either side of the Bridge, thus we do not know if our model underpredicts or overpredicts net mass transport through the bridge cross-section.

2.3.2 Grid Refinement

South of Dumbarton Bridge, the NMS open Bay model grid captured the gross features of the system but lacked the finer details needed to meet project study objectives. To support this current work, the grid was refined by making the following modifications:

- extension into watershed channels of the WDM pour points,
- increased unstructured grid resolution along the shoreline, and
- updated bathymetry in the revised areas.

Revisions were made to the base grid along the shoreline where San Francisquito Creek, Matadero Canal, Adobe Creek, Permanente Creek, and Stevens Creek meet the Bay (Figure 8: Areas outlined in yellow). The goal was to ensure that the output locations of the watershed model were captured in the hydrodynamic model to better represent the tidal prism and reduce uncertainty in boundary condition temporal alignment.

2.3.3 Extended Grid Boundaries and Resolution Modifications

To better represent the distribution of a passive tracer within the study area, the grid boundaries were modified to capture intertidal areas extending up the creeks and more closely following shoreline features not captured in the NMS open Bay model grid (Figure 9). Refinements were made using the RGFGGRID software provided by Deltares. Rectangular cells were used to define each of the channels with a grid resolution of ~15x25 m. Each revised channel is at least three cells wide. While this may not capture the full complexity of bathymetric features in the system due to the relatively narrow active portion of the channels, the resolution should be sufficient to capture the gross properties of the system and balance computational model run time to address the study questions.

Areas between the revised channels were represented with unstructured grid cells with the finest areas along the shoreline at 50x50 meters and expanding to connect to the original grid areas with a resolution of 150x150 m. These resolutions are approximate as the grid allows for changes to capture features and connect between areas of varied refinement (Figure 10). An omission in both the baseline and updated model grid was the



Figure 8. South San Francisco Bay grid revision area and watershed inputs.

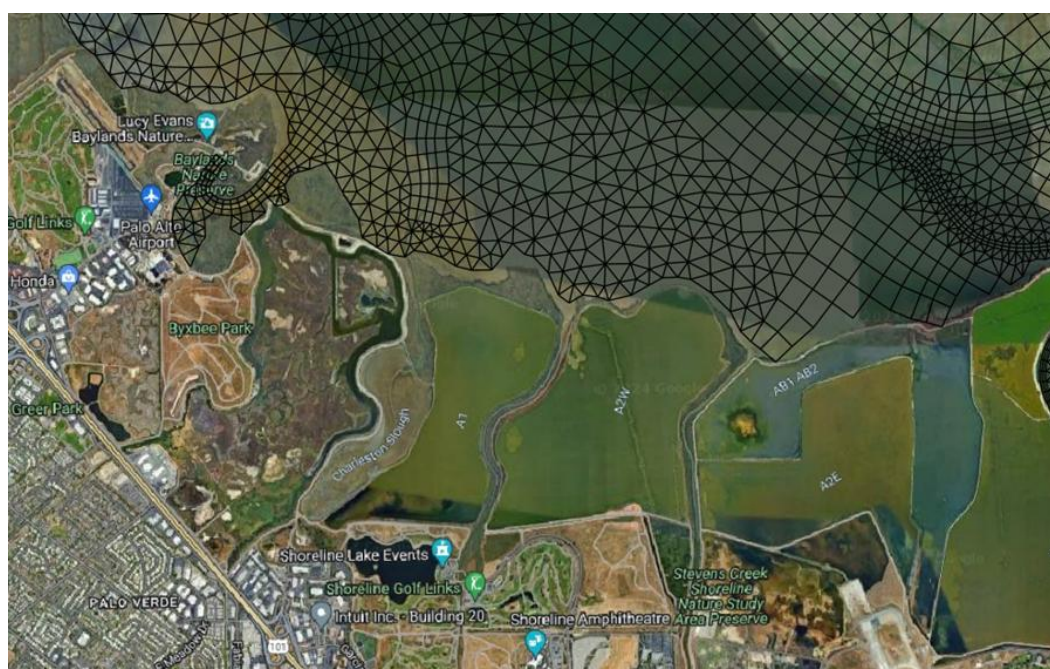


Figure 9. Baseline grid along the area of interest.



Figure 10. Modified grid extending to shoreline and into creeks.

influence and exchange of flow between the Bay and the salt ponds. It was deemed that the effort required to acquire the data on fine-scale salt pond bathymetry and levee features and parameterize the model to quantify these complex dynamics was not warranted because the volumes of exchange are estimated to be small due to the shallow nature and muted exchange (small and infrequent levee openings) relative to the stormwater, wastewater, and wider Bay exchange volumes. Instead, the grid modifications followed the levees outlining the salt ponds in the areas identified for refinement. The modified portions of the grid integrate well into the wider South Bay grid and extend to each watershed model pour point (Figure 10 and 11).

2.3.4 Bathymetry Revisions

Because the base model did not extend into the shallow intertidal and channel regions of interest, updated bathymetry had to be sourced and interpolated onto the new grid. This project relied on the continuously updated digital elevation model (CUDEM) for San Francisco Bay. The CUDEM is a multi-agency product that combines elevation data from multiple sources to develop a 1/9th arc-second (~3 m) resolution topo-bathy dataset throughout San Francisco Bay. Data were subset in the region of interest (black region of Figure 12) and converted to x-y-z points, referenced to NAD83 UTM Zone 10N with a vertical datum of NAVD88. The data were then interpolated onto the model grid.

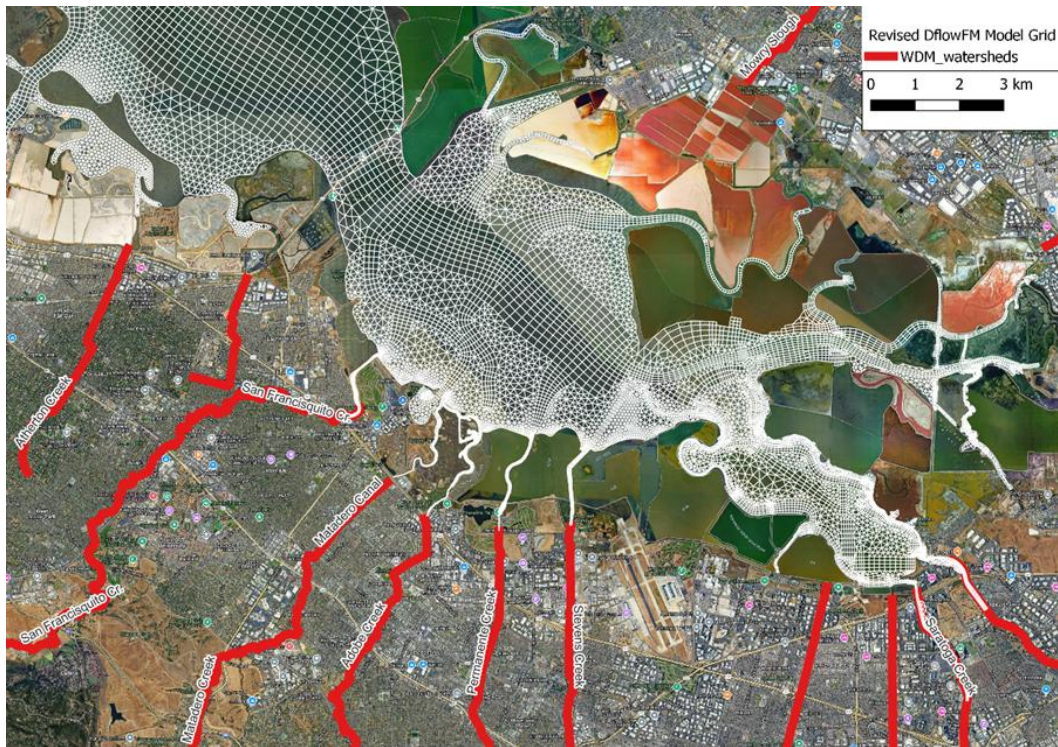


Figure 11. Modified South San Francisco Bay model grid.

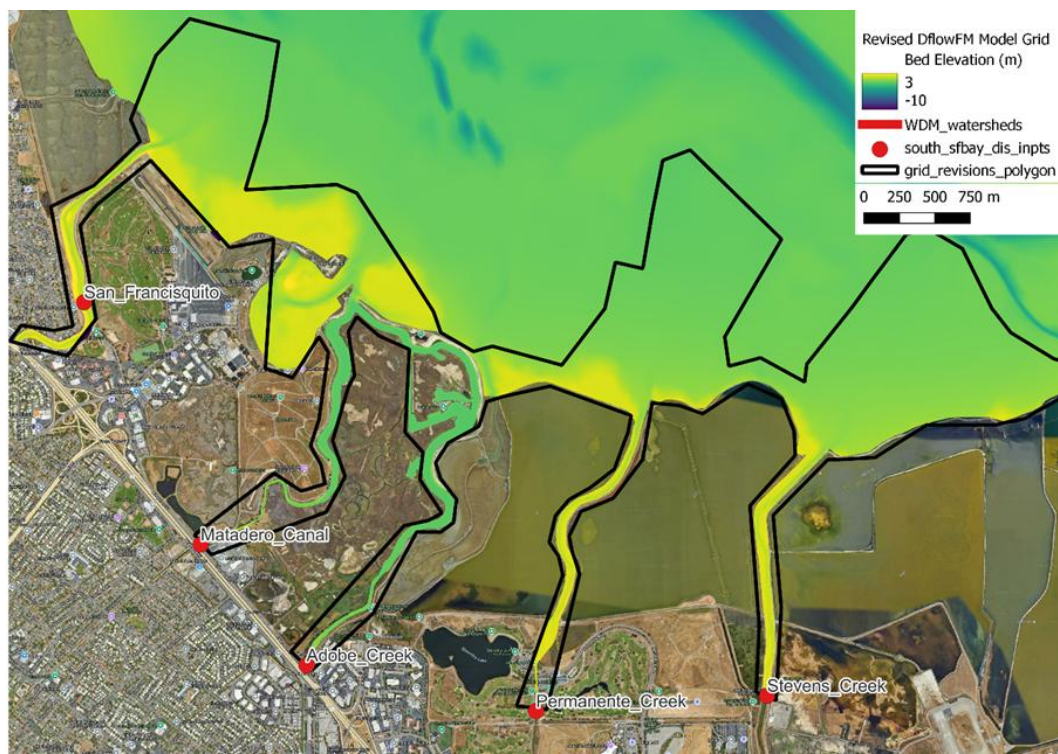


Figure 12. Bathymetry of the study area in South San Francisco Bay.

The CUDEM dataset (Amante et al., 2023) was obtained from NOAA. Metadata for the CUDEM is here:

https://ch.s.coast.noaa.gov/htdata/raster2/elevation/NCEI_ninth_Topobathy_2014_8483/

The revised grid and inclusion of CUDEM bathymetry were thought to adequately represent the upland channels. This was supported for most of the tributaries, including Stevens Creek, by an initial check of the WY 2023 simulation that showed cumulative inflow from each pour point equaled cumulative outflow from the tributary to the Bay for all tributaries except Adobe Creek, Permanente Creek, and Mowry Slough. Connectivity was also observed for the idealized flood-tide tracer release in Stevens Creek. However, for the idealized ebb tide tracer release in Stevens Creek, we observed stranding in puddles which were an artifact of underresolved bathymetry in the upland portion of the stream – the grid cells in this region are about three times wider than the channel.

To work our way around these problems and advance our understanding of passive tracer mixing in this current study, the connectivity of Adobe Creek, Permanente Creek, and Mowry Slough was attained by flattening the upper reaches until all upstream flow reached the Bay. For the WY 2023 simulation based on real world conditions, connectivity was attained for Stevens Creek with the data-based bathymetry, but for our idealized pulse tracer release components of this current study, we flattened the interior portion of the Stevens Creek channel to an elevation 1.1 m above MLLW to ensure mass was successfully conveyed during and after the ebb tide as well as the flood tide. While this did artificially increase the tidal prism slightly in these regions, it ensured that discharge, even at low flows, introduced a passive tracer into the Bay. In contrast to ebb-tide tracer release results, flood-tide tracer release results were not sensitive to the change in upstream bathymetry, likely due to deeper water levels during the release preventing puddling for the original bathymetry. In future studies, if the dynamics of the intertidal stream habitat are of concern, the grid should be refined in these regions to better resolve the channel. However, our tests showed that results on the mudflats and subtidal regions of the Bay are not sensitive to the details of the grid in the upper reaches of the streams.

2.3.5 Water Year 2023 Hydrodynamic Model and Tracer Release Study

The NMS open Bay model with the modified LSB grid (as described) was run for a simulation period that spanned July 1, 2022, through September 30, 2023 (that included a 2-month spinup before the first available field data for HMMM on September 2022, just before the beginning of WY 2023). The key boundary conditions for the WY 2023 hydrodynamic model run include tidal exchanges with the Pacific Ocean, Delta inflow from the large Central Valley Rivers, local tributary inflow from the Bay Area counties north of the Dumbarton Bridge, LSB local tributaries, treated wastewater discharges, and the net of precipitation minus evaporation (Figure 13). Relative to other years, WY 2023 was a wet year (Figure 14). That year allows us to model about 5 larger storm events and a number of minor storms in local smaller tributaries. Delta outflows in excess of 6000 m³/s (Figure 13a) have only occurred about 12 times in the past 30 years.

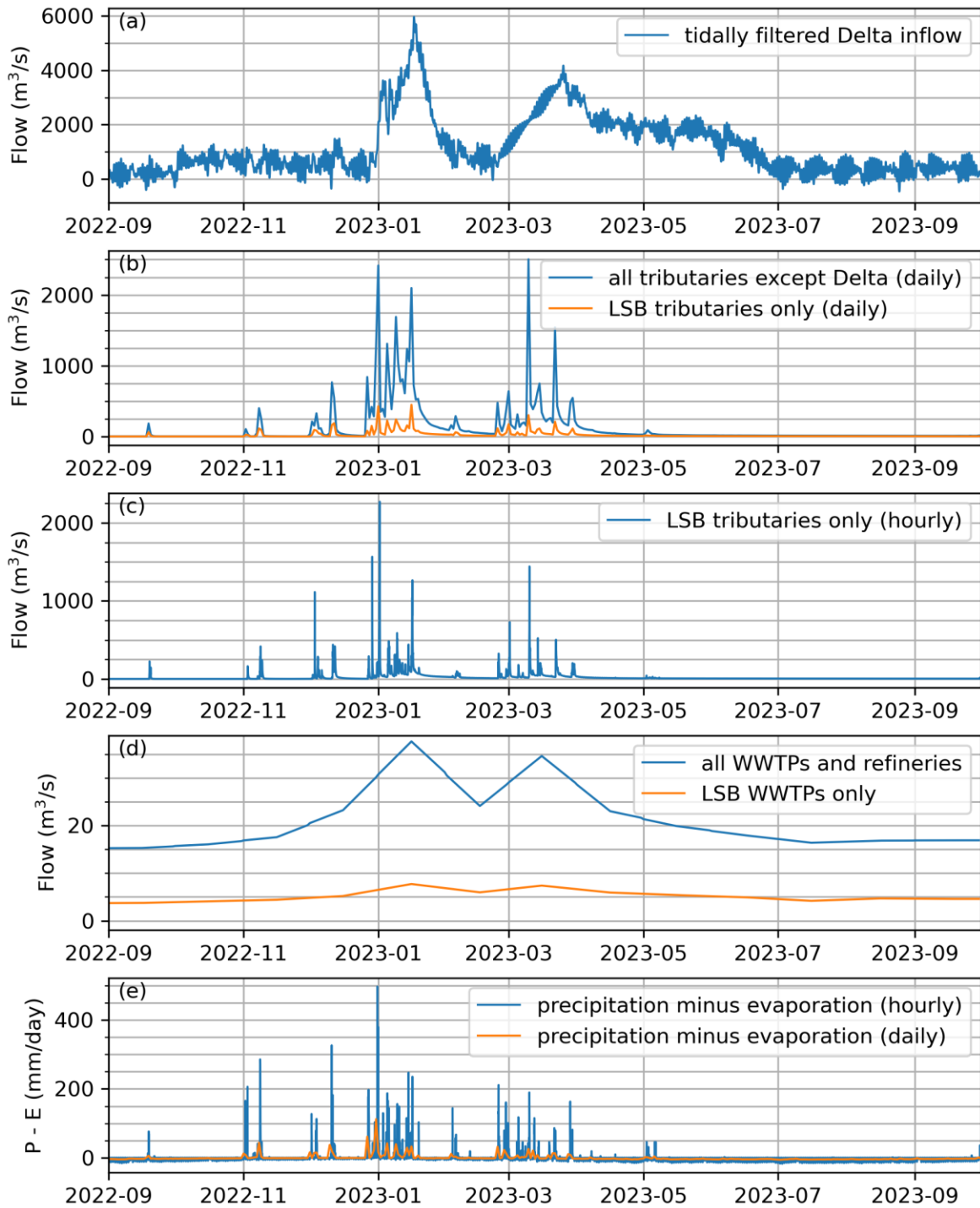


Figure 13. Boundary conditions for the WY 2023 hydrodynamic model. (a) Tidally filtered Delta inflow (negative values indicate landward flow). (b) Daily averaged flows for small tributaries and Lower South Bay (LSB) tributaries. (c) Hourly LSB tributary flows. (d) Total and LSB wastewater treatment plant (WWTP) discharges; monthly scales reflect a twofold winter increase due to stormwater augmentation. (e) Hourly and daily averaged precipitation minus evaporation.

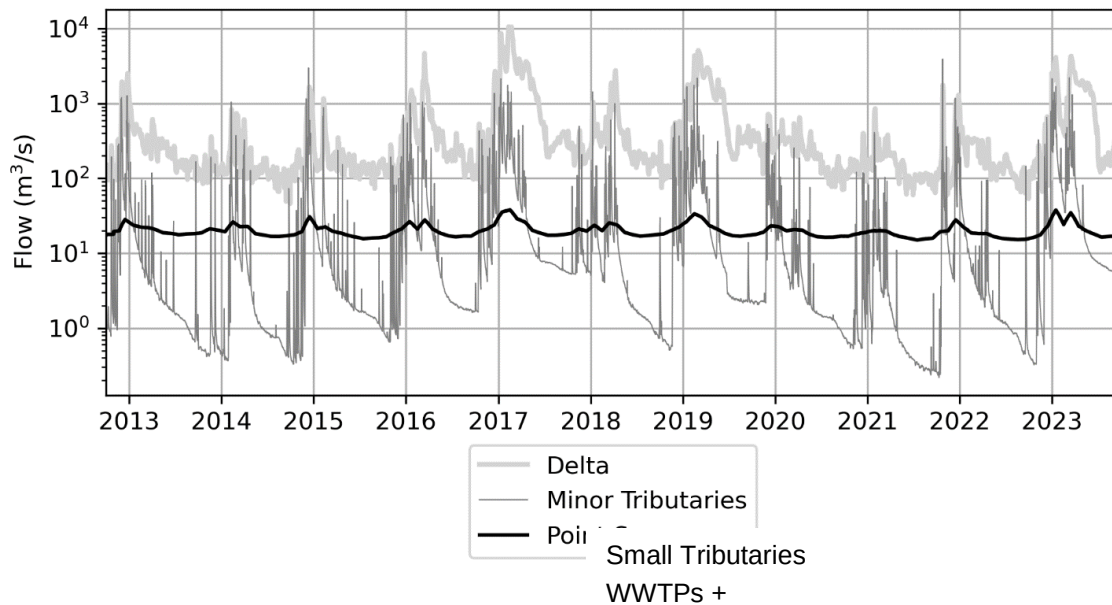


Figure 14. Freshwater inflows to the Bay from the Delta (Sacramento and San Joaquin Rivers), small tributaries (all except the Delta) and treated wastewater discharges (WWTPs and refineries) over 11 years.

We used 15-minute tidally varying inflows from the Sacramento and San Joaquin Rivers to drive the model, but in Figure 13 we show the tidally filtered flow (similar to the daily average flow) for Sacramento plus San Joaquin (the Delta inflow). Delta inflow is the largest inflow to the Bay. Smalltributaries made a lesser contribution. Daily inflows from small tributaries north of the Dumbarton Bridge were deemed appropriate. South of the Dumbarton Bridge, hourly inflows were used for the model boundary. In the case of treated wastewater discharges, monthly average flows were used due mainly to data availability. This will cause an underestimation the relative contributions of treated wastewater during storms since flows from these sources are known to increase during wet weather due to stormwater augmentation. Precipitation minus evaporation was input at an hourly time step and was used for the atmospheric boundary condition. Boundary conditions that are not shown included (1) tidal ocean water surface elevation (this would be too blurry to plot for 13 months) (2) spatially and temporally varying wind field, and (3) spatially and temporally varying relative humidity, air temperature, and cloudiness. These were included as described in Holleman et al. (2017), SFEI (2020), and SFEI (2023). Atmospheric pressure gradients across the bay were not included because they are small compared to the tides (on the order of a few millibar, or a few cm of water).

We ran standard NMS validation scripts to evaluate the WY 2023 model, following Holleman et al. (2017), SFEI (2017), and SFEI (2023). See Holleman et al. (2017, p. 11-12) for definitions of validation statistics. These validation statistics are shown in Tables 3 and 4 for WY 2023. Hydrodynamic model validation Figures, water level and salinity

Table 3. Hydrodynamic model validation statistics for water surface elevation. Units: m.

Station	St. Dev. Observed	St. Dev. Modeled	Lag (min)	Amplitude	Intercept	RMSE	Skill	r2
NOAA Richmond	0.569	0.580	-6	1.015	-0.000	0.054	0.998	0.996
NOAA Alameda	0.619	0.628	-6	1.011	0.000	0.052	0.998	0.997
NOAA Redwood City	0.765	0.761	-12	0.990	-0.000	0.072	0.998	0.996
NOAA Point Reyes	0.548	0.496	6	0.906	0.000	0.054	0.997	0.999
NOAA Port Chicago	0.459	0.761	-30	1.597	0.000	0.343	0.920	0.963

Table 4. Hydrodynamic model validation statistics for salinity time series. Units: PSU.

Station	Mean Obs	St. Dev. Obs	Mean Mod	St. Dev. Mod	Lag (min)	Amp	Inter.	Bias	Ub RMSE	Skill	r2
USGS Richmond-San Rafael Bridge (Upper)	25.7	5.43	25.4	4.24	15	0.71	7.21	-0.27	2.38	0.94	0.91
USGS Richmond-San Rafael Bridge (Lower)	27.5	3.72	26.4	3.58	-15	0.84	3.50	-1.04	1.90	0.91	0.87
USGS Alcatraz	27.2	4.69	26.3	3.88	0	0.77	5.37	-0.88	1.79	0.95	0.93
USGS San Mateo Bridge (Upper)	25.1	5.93	25.4	4.42	-15	0.71	7.68	0.35	2.21	0.95	0.95
USGS San Mateo Bridge (Lower)	25.0	6.18	25.6	4.27	-15	0.64	9.45	0.56	2.69	0.93	0.93
USGS Dumbarton Bridge (Upper)	23.5	6.76	23.5	5.33	0	0.75	6.01	0.07	2.43	0.96	0.95
USGS Dumbarton Bridge (Lower)	23.9	6.53	24.2	4.94	0	0.71	7.16	0.34	2.48	0.95	0.94

SFEI San Mateo Bridge	25.5	5.79	25.5	4.42	-15	0.72	7.08	-0.08	2.19	0.95	0.94
SFEI Shoal	25.4	5.95	25.0	4.69	-15	0.76	5.80	-0.41	1.99	0.96	0.96
SFEI Hayward	25.8	5.87	25.3	4.89	-45	0.77	5.43	-0.45	2.27	0.95	0.93
SFEI San Leandro Marina	24.6	6.62	23.5	6.29	-15	0.90	1.34	-1.10	2.11	0.97	0.95
SFEI Alviso Slough	11.5	8.70	14.5	9.61	-45	0.94	3.71	3.04	5.06	0.90	0.85
SFEI Guadalupe Slough	13.5	8.12	8.4	7.77	60	0.58	0.62	-5.12	7.09	0.72	0.60
SFEI Coyote Creek	13.5	8.12	19.8	7.23	60	0.75	9.66	6.29	4.38	0.79	0.84
SFEI Newark Slough	21.8	7.88	21.5	7.06	420	0.82	3.63	-0.38	3.26	0.95	0.91
SFEI Pond A8 Outlet	10.1	8.76	11.8	10.03	-30	0.98	1.91	1.71	5.20	0.91	0.86

validation statistics, and station maps are in the Appendix. Model performance was consistent with previous NMS Open Bay applications (see SFEI, 2023; SFEI, 2020; Holleman et al. 2017). Water surface elevation predictions were excellent, except at Port Chicago, which is close to the truncated Delta boundary and not expected to perform well. We typically exclude validation for SFEI mooring stations in LSB (Alviso Slough, Guadalupe Slough, Coyote Creek, Newark Slough, Pond A8 Outlet) because the model was not designed to resolve all the relevant bathymetry and fine-scale structures (gates, levee breaches) in these regions, and many salt ponds are left out of the model to ensure reasonable run times, but we have included model performance at these stations here because the region of interest was LSB. Salinity performance was generally good (skill over 0.9), except at Guadalupe Slough, Alviso Slough, and Coyote Creek (Table 4). Strong negative salinity bias during the early months of the simulation and in USGS Peterson cruise transects (Appendix) likely stemmed from inaccurate initial conditions. In early model runs, including this one, we used an initial salinity field from WY 2011. An updated model version improves the salinity initial condition by creating a salinity field from USGS Peterson cruise data at the appropriate time, resulting in better salinity predictions early in the simulation

The WY 2023 hydrodynamic model run produced a set of binary output files every 30 minutes for every cell in the model: cell volume, cross-sectional area of the cell faces, flow between the cells, salinity and temperature within the cells, and shear stress at the vertical interfaces between the cells as well as the bed and the surface. This output from DFM was fed into the Deltares water quality module, Delwaq (also called DWAQ). With Delwaq, we conducted a passive tracer study to investigate the provenance (the relative contribution of different source water) to various locations across LSB. In an estuarine context, the term “provenance” describes the origin and changing relative contributions of water from multiple sources—and their associated materials—as they mix and vary over tidal to seasonal timescales.

To determine provenance, separate tracers, all with concentration 1 (unitless), were released continuously throughout the July 1, 2022 through September 30, 2023 time period. In this study, we tracked provenance spatially and temporally for a large number of sources:

1. Ocean
2. San Joaquin River + Sacramento River (Delta)
3. LSB tributaries:
 - a. Stevens Creek
 - b. Guadalupe River
 - c. Berry Coyote
 - d. San Francisquito Creek
 - e. Adobe Creek
 - f. Permanente Creek
 - g. Mowry Slough
 - h. Agua Caliente Creek
 - i. Calabazas Creek
 - j. Sunnyvale East Lower
 - k. Saratoga Creek
 - l. Matadero Canal + Agua Fria Creek (combined)
4. LSB treated wastewater discharges:
 - a. San Jose
 - b. Palo Alto
 - c. Sunnyvale

5. All other tributary inflows (north of the Dumbarton Bridge, excluding the Delta)

6. All other treated wastewater discharges (north of the Dumbarton Bridge)

The concentration of an additional tracer was set to 1 for the initial condition across the entire domain to track the contribution of the initial water. Concentrations of water for each source were tracked throughout the simulation. In the Delwaq input file, we specified advection scheme 24 to minimize numerical diffusion. Scheme 24 is an explicit second-order scheme that applies different time steps to different parts of the domain to ensure stability (Deltares, 2023b). Numerical diffusion with scheme 24 is greater than numerical diffusion within DFM, the hydrodynamic solver, by about a factor of 4, but it is within the broad range we might reasonably expect for diffusion within the Bay. Numerical studies of a Gaussian tracer release found dispersion $K=30 \text{ m}^2/\text{s}$ in South Bay proper for scheme 24, compared to $8 \text{ m}^2/\text{s}$ for DFM. We did not intentionally add horizontal diffusion to the model as numerical diffusion was thought to be adequate. Exploratory modeling of the WY2022 harmful algal bloom in South Bay found better agreement with satellite imagery with lower diffusion schemes (SFEI internal communication). We took care to ensure that mass was conserved throughout the simulation. If mass is conserved, the sum of the concentrations of all the sources, including the initial water concentration, at all locations within the model domain, and for all time, will equal 1. It is very unlikely the concentrations will sum to 1 if mass is not conserved, so this is a check of mass conservation. However, in our simulation, there is evaporation and precipitation, which result in concentration or dilution such that the sum of the source concentrations is not necessarily 1. To verify that mass was conserved in our simulation, we conducted a special simulation with no precipitation or evaporation and verified that the concentrations for all sources did sum to 1 over all space and time. In the final simulations, which did include precipitation and evaporation, the concentrations did not necessarily sum to 1, so we normalized each concentration by the sum of all concentrations to obtain the fractions of the different water sources.

2.3.6 Idealized Tracer Release Study for Stevens Creek

We used the WY 2023 hydrodynamic simulation described above as the basis for two idealized tracer release studies in Stevens Creek. The model grid was modified slightly to level the interior portion of Stevens Creek to an elevation of 1.13 m above MLLW as this was found to ensure the intended mass entered the domain without compromising the validity of the model results within Stevens Creek (without this modification, the mass would not enter the domain when the source cell was dry). We explored other methods for ensuring all the mass entered the domain. For example, only dredging the source cell did solve this problem but the solution was invalid in the near field.

All boundary conditions for the idealized runs matched those used in the Water Year 2023 simulation, with the exception of the Stevens Creek inflow. For Stevens Creek, we generated a synthetic inflow time series featuring a base flow of $0.01 \text{ m}^3/\text{s}$ and a 6-hour pulse inflow of $107.2 \text{ m}^3/\text{s}$. Based on the WDM annual maximum series (Figure 15), an $8 \text{ m}^3/\text{s}$ storm—corresponding to a 1-year return event—has only a 1% chance of occurring

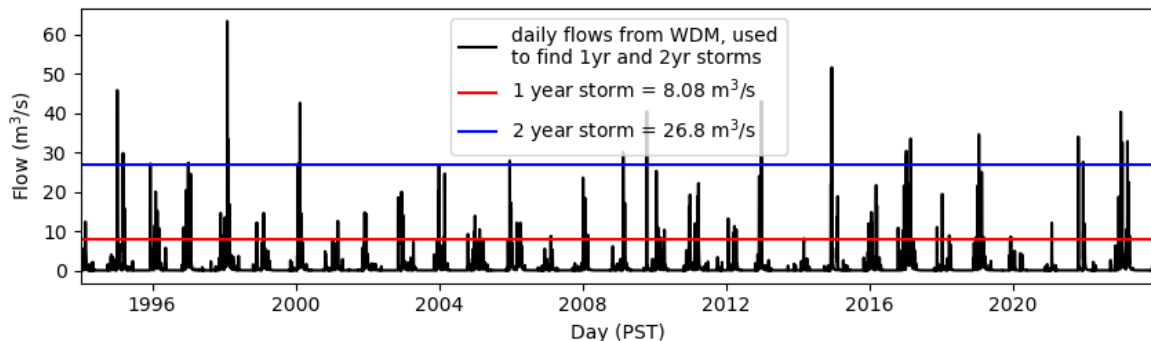


Figure 15. Estimated 1 and 2-year return frequency storms for Stevens Creek based on an exceedance probability formula.

in any given year over the past 25 years. To simulate a larger, less frequent event, we used a 26.8 m³/s flow sustained over 24 hours, which represents an estimated 2-year return storm. This flow has the same total volume as the 107.2 m³/s inflow lasting 6 hours (roughly half a tidal cycle) that was used for our model runs. Although a number of factors could affect contaminant discharge, such as storm intensity and duration, antecedent days since the last storm, number and timing of previous storms, soil saturation, and mass load tends to be nonlinear with flow, this high-flow scenario was chosen to reflect conditions where a larger storm events typically result in larger contaminant discharge. However, while this approach was developed to reflect a plausible scenario for an acute toxic event, it was not based on or tied in any way to any specific contaminant, toxicity threshold, storm return frequency, or event duration (e.g., 1 hour, 6 hours, or 24 hours) of concern. It was designed to be purely illustrative.

To model the dispersion of the 2-year synthetic storm, a 6-hour time window for the pulse flow was selected based on tidal water surface elevations and velocity at USGS Peterson site P35 (The “P” prefix refers to a Peterson cruise that is done along a transect of the deep channel of San Francisco Bay, from the Lower South Bay up into the Sacramento River system approximately monthly, collecting discrete water samples and sensor profiles – see Appendix for a map of the site locations) to occur during a full ebb tide and a full flood-tide to capture the most and least dispersive conditions, respectively. Separate model runs were conducted for the ebb tide pulse and the flood-tide pulse. We looked for a time period with fairly symmetric ebb and flood-tides. We chose 9/11 21:15 through 9/12 03:15 for the ebb tide pulse and 9/12 03:15 through 9/12 09:15 for the flood-tide pulse (Figure 16).

We simulated from July 1, 2022, through September 30, 2022 only. In the future, we could run a longer simulation, but we would have to modify more of the boundary conditions as later in WY 2023 there were very large inflows not representative of typical conditions. DFM output was fed to Delwaq, where we applied a concentration of 1 to Stevens Creek

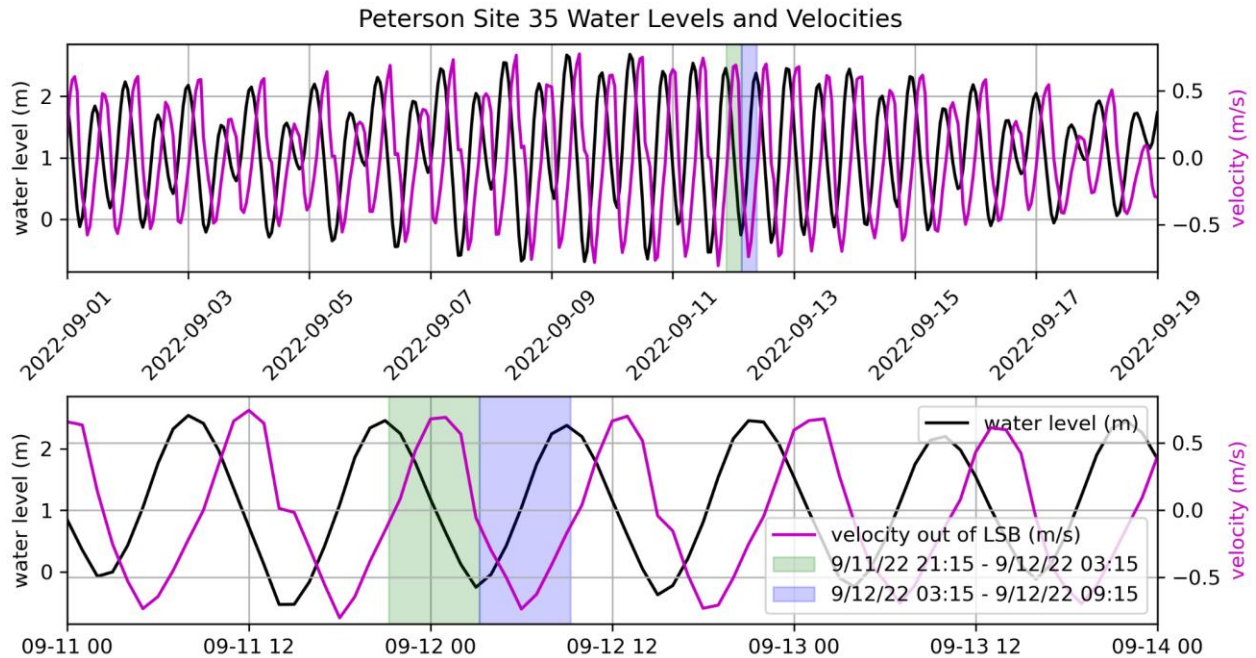


Figure 16. Water surface elevation and velocity (positive velocity is out of LSB) at USGS Peterson Site 35 within Lower South Bay. Shaded regions highlight periods of the ebb tide Stevens Creek pulse flow (green, 9/11 21:15 through 9/12 03:15) and the flood-tide pulse flow (blue, 9/12 03:15 through 9/12 09:15). The lower plot is a zoomed in version of the upper plot.

during the 6-hour pulse. The concentration of the Stevens Creek source water was set to zero outside this time window. Again, we used integration scheme 24. We took care to ensure that all the intended mass entered the model and that mass was conserved throughout the simulation. For the idealized tracer release studies, we created a set of sampling sites (Figure 17; Table 5) to capture the evolution of the Stevens Creek Plume approximately along its centerline as well as in habitats identified in the management questions. These sampling sites were included in the WY 2023 simulation as well.

3. Results and Discussion

The model begins on 1 July 2022. A two-month spin-up period was used to initialize the salinity and temperature field (water surface elevation and velocity spin up faster). Water provenance was evaluated at the sampling sites using plots similar to Figure 18. This figure shows the provenance of water at USGS Peterson Site 35. We tidally filtered the model results for readability. A tidal filter removes tidal oscillations and higher frequency oscillations from a time series. We used a 6th order low-pass Butterworth filter with a 30-hour cutoff period and constant padding. This is similar to a rolling average with a 30-hour period, but produces less spurious oscillations near sharp transitions (Lathi and Green, 1998).

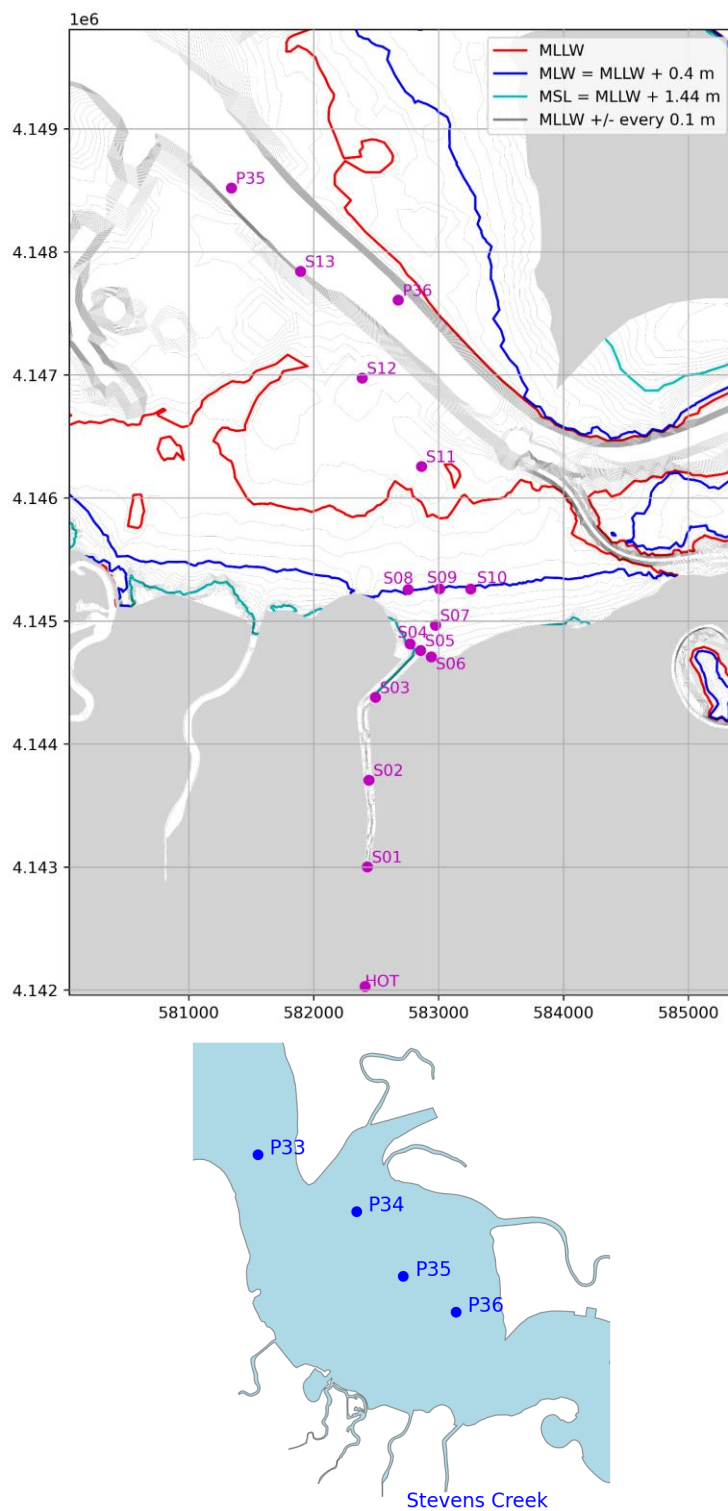


Figure 17. Locations of the sampling sites defined in Table 2.4.2.5 and the USGS Peterson Cruise sampling sites (labeled with a “P”). Tidal datums are plotted as colored lines: MLLW = Mean Low Low Water, MLW = Mean Low Water, MSL = Mean Sea Level, HOT = Head of Tide; thin grey lines represent 0.1 m increments on either side of MLLW.

Table 5. New sampling points were created for the idealized Stevens Creek tracer release study. Sites highlighted in yellow are approximately at mean sea level (MSL) and sites highlighted in blue are approximately at mean low water (MLW).

Station	x (UTM, m)	y (UTM, m)	Bed Level w.r.t. MLLW (m)
S01	582430	4143002	2.11
S02	582442	4143706	1.82
S03	582494	4144380	1.39
S04	582772	4144815	1.44
S05	582856	4144760	0.82
S06	582941	4144708	1.32
S07	582976	4144966	0.63
S08	582757	4145255	0.36
S09	583007	4145261	0.41
S10	583257	4145259	0.40
S11	582863	4146256	-0.15
S12	582389	4146974	-0.25
S13	581894	4147842	-1.80

Initially, the water already in the Bay (the unknown starting mixture of ocean, wastewater, and watershed water) was 100% (Figure 18; dark blue). It took approximately eight months for the initial Bay water to dilute to <1% of the volume at site 35 (exact time was 2023-02-28 17:00, 242.7 days into the simulation). The time for dilution to reach <0.1% at Site 35 was 319.1 days. This is considerably longer spin-up compared to salinity and temperature, which require only two months, and velocity and water level, which require days or weeks.

The following sections first discuss water provenance at sites within Stevens Creek and across the LSB over the real-world Water Year 2023. We then examine how a conservative tracer enters from the watersheds during an idealized storm simulation. Lastly, we compare our real-world water CEC results with the results of the dilution simulation to determine if the measured CEC concentrations behave similarly to the simulated tracers. The archived data from the model setup and runs stored on SFEI's internal server (richmond@sfei.org:/hpcvol2/stevens_creek/) can be obtained by request.

3.1 Provenance of Freshwater at Stations in Lower South Bay

3.1.1 Provenance Characteristics During Water Year 2023

The proportion of freshwater contributions from small tributaries and treated wastewater varies throughout the year based on rainfall-runoff events, Delta inflows, tides, and wind. At any given time and location, freshwater provenance is a unique mixture and was estimated by the model.

Water Year 2023 was marked by a late December/early January storm series and a late February/March storm series with some minor flows in September, November, and early December (Figure 19). In the Stevens Creek tidal channel locations (sites S02 and S03; Figure 17), flow from Stevens Creek was the dominant source during the wet season (Figure 20). From November 1, 2022, to April 30, 2023, any sampling would have primarily yielded freshwater from the Stevens Creek watershed, with Bay water, flow from other tributaries, and WWTPs contributing up to 30% altogether, but usually less than a few percent.

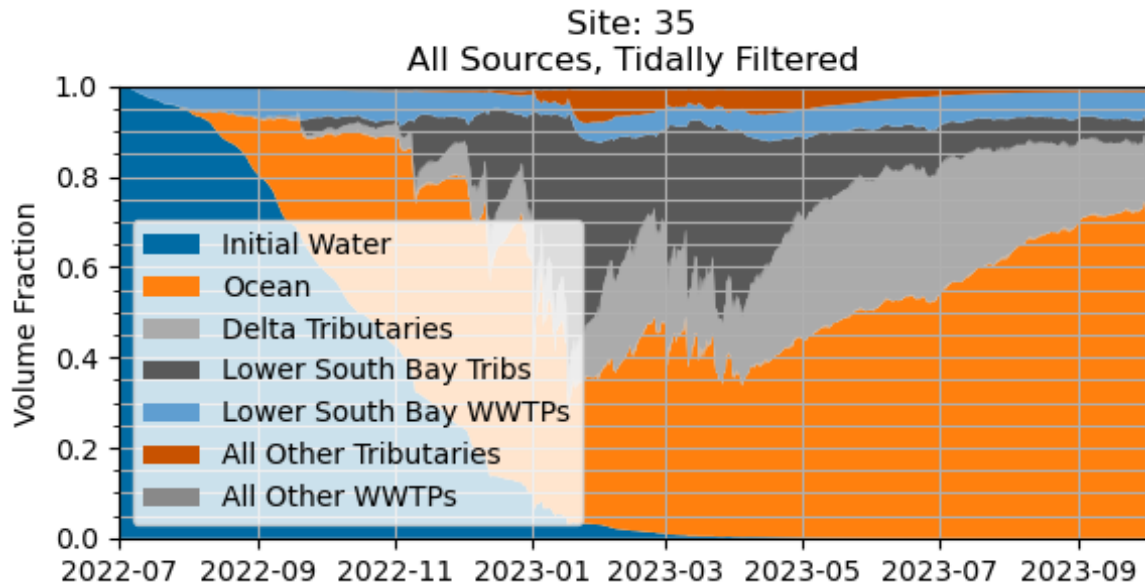


Figure 18. The provenance of water at site P35 (see appendix for a map of the USGS Peterson sites) during Water Year 2023. This is a stack plot, in which contributions from the various sources are stacked on top of each other, such that the contribution of a given source at a given time is represented by the thickness of its color in the vertical direction. For example, on May 1, 2023, the volume fraction of the Delta Tributaries is approximately 0.25. Also note that the contribution of all other WWTPs is so small it is not easily seen at the top of this stack plot.

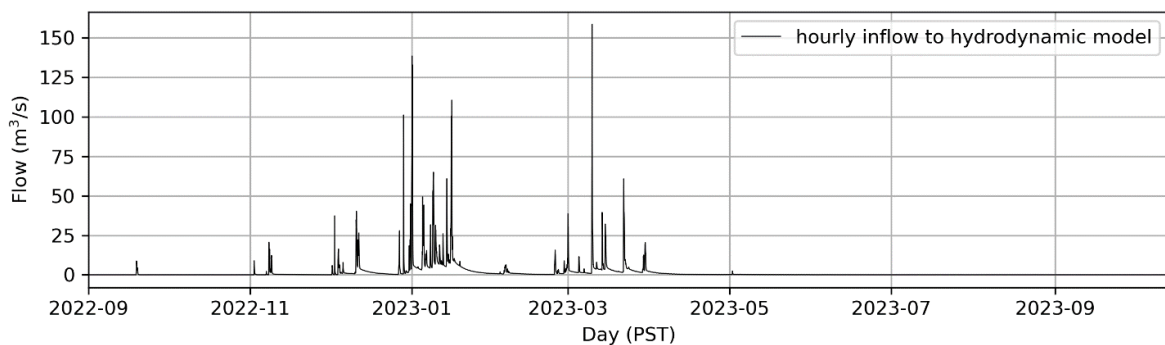


Figure 19. Flow in Stevens Creek at the upstream boundary (the entry point into the Bay modeling domain) during Water Year 2023.

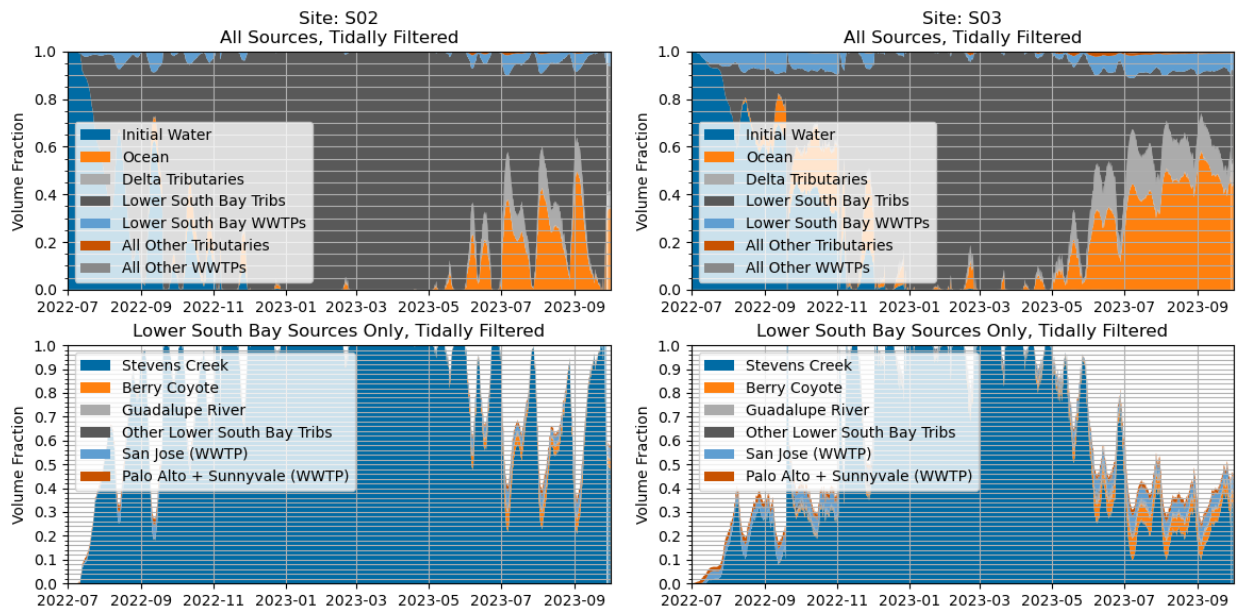


Figure 20. The provenance of fresh water within the Stevens Creek tidal channel as illustrated by two hypothetical sampling points (S02 and S03) during Water Year 2023. Please refer to Table 5 and Figure 17 for the geographic locations of these points. For flow conditions throughout the year see Figure 13.

This trend extended a short distance out into the Bay along a tidal channel of Stevens Creek that is incised into the mudflat. At sites S05 and S07 (Figure 17), flow from Stevens Creek remained the dominant source during the wet season (Figure 21). Though flow from Stevens Creek was nearly 100% only during wet season storm events, from November 1, 2022, to April 30, 2023, sampling would have still primarily yielded fresh water from Stevens Creek, with Bay water or flow from other tributaries and WWTPs contributing up to 60% but less than 10% most of the time.

On the mudflat near the entrance of Stevens Creek channel to the Bay at an elevation of approximately mean sea level (MSL) as illustrated by sites S04 and S06 (Figure 17), flow derived from Stevens Creek reached nearly 100% during a mid-January storm but was usually less than 20% of the volume (Figure 21). From November 1, 2022, to April 30, 2023, any sampling would have been dominated by fresh water, but with a highly variable mixture of Stevens Creek, Guadalupe River, other LSB tributaries, and Ocean water (up to 45%, typically 20%), and typically <5% contribution from WWTPs. Based on the model results, to capture an acute response from Stevens Creek, a sampling design responsive to the timing of large storm events would have been needed for either of these two upper mudflat locations.

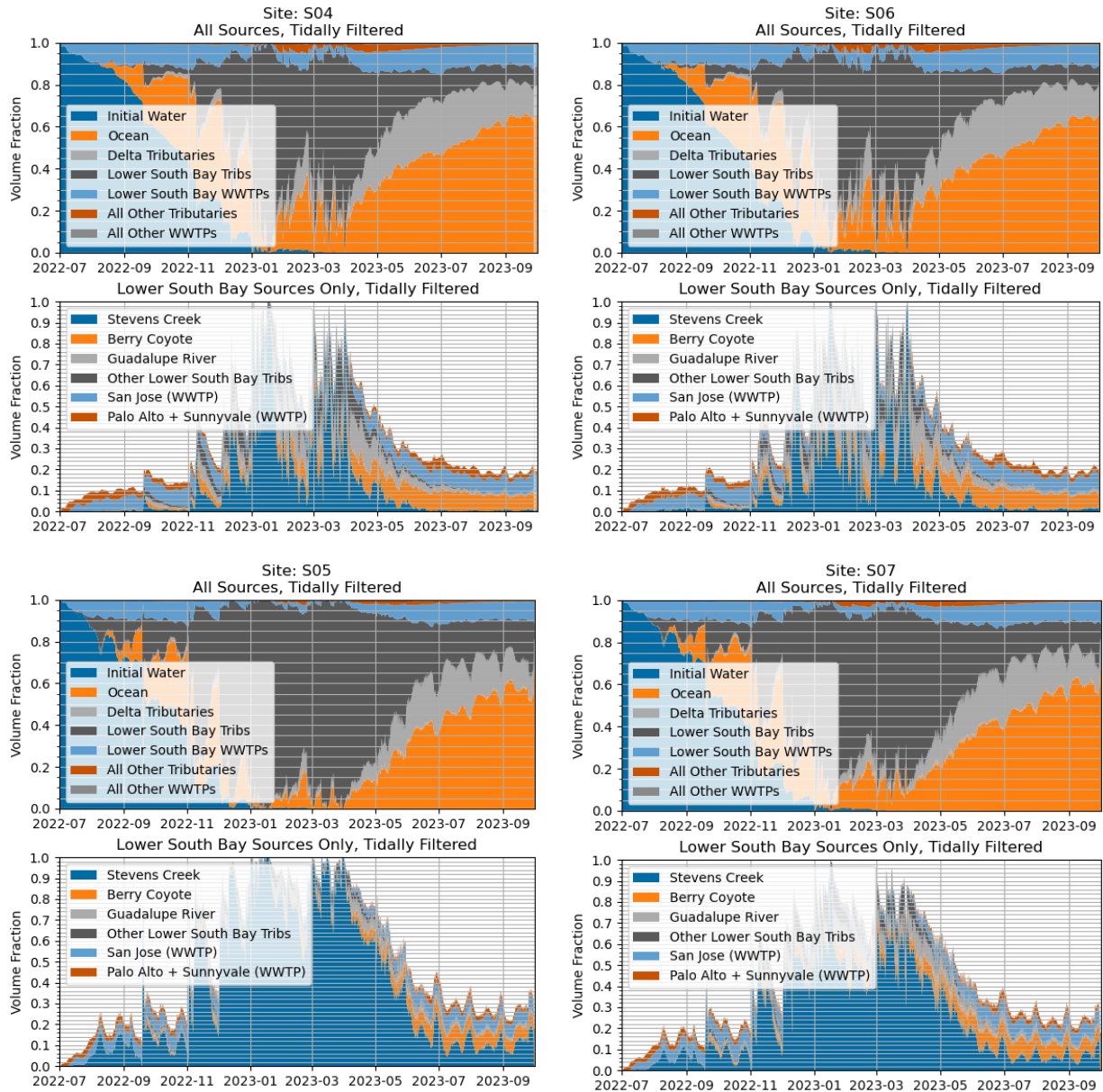


Figure 21. The provenance of fresh water at the Stevens Creek tidal channel outlet to the Bay is shown through four hypothetical sampling points: two in the tidal channel (S05 and S07) and two on the adjacent mudflats (S04 and S06), 100m on each side, during Water Year 2023. Mass conservation is illustrated by the total water sources summing to 1. Refer to Table 5 and Figure 17 for the locations of these points, and Figure 13 for flow conditions throughout the year.

Further out on the mudflats at mean low water (MLW) elevation (sites S08, S09, and S10; Figure 17), flow from Stevens Creek reached upward of 50% during a mid-January storm (Figure 22) but was usually less than 35%. From November 1, 2022, to April 30, 2023, any

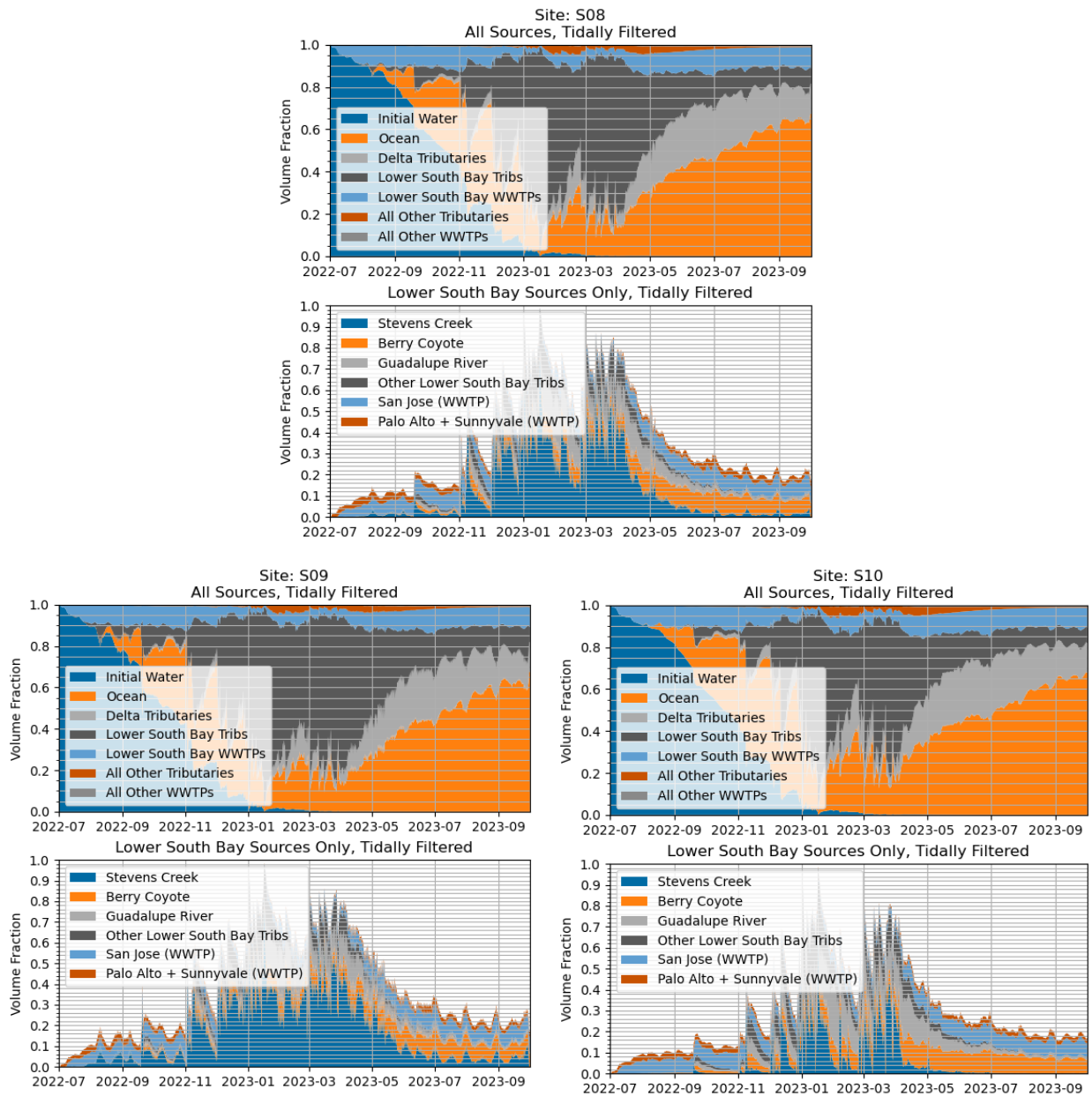


Figure 22. The provenance of fresh water on the mudflat out from the mouth of Stevens Creek at a depth of approximately mean low water as illustrated by three hypothetical sampling points (S08, S09, and S10) during Water Year 2023. Please refer to Table 5 and Figure 17 for the geographic locations of these points. For flow conditions throughout the year see Figure 13.

sample taken would have mainly consisted of fresh water, including up to 10% Delta water, with a highly variable mixture of Stevens Creek, Guadalupe River, Coyote Creek, other LSB tributaries, and Ocean water (up to 45%, typically 30%), and less than 10%

from WWTPs. Based on model results, it would have been nearly impossible to capture a volume of water dominated by flow from Stevens Creek at these outer mudflat sites.

In deeper parts of the LSB near the Bay spine (S11, 0.15 m below MLLW; S12, 0.25 m below MLLW; S13, 1.8 m below MLLW; Figure 17), and at the Bay spine location where the closest CEC sampling was done (P34, 5.7 m below MLLW), the provenance of water is very similar to sites S08, S09, and S10. During the WY 2023 wet season, with a +/- 10% variation, these open water regions of LSB consisted of about 40% Ocean water (Figure 23). Except for S11 where water from Stevens Creek occasionally exceeded 10%, at these locations, the volume was <5% Stevens Creek, with variable amounts of flow from Guadalupe River, Coyote Creek, other LSB tributaries, and WWTPs (5%) making up the balance. Based on model results, at S11, S12, and S13, flow and any dissolved phase pollution emanating from Stevens Creek would have been diluted at least 20-fold most of the time. Dilution would have been even greater at P34 (see appendix for a map of the USGS Peterson sites). As mentioned earlier, and discussed more below, dilution would have even been greater if increased flow from WWTP facilities had been input to the model at daily rather than monthly timescales.

3.1.2 Answers to Provenance Management Questions

MQ1 For different parts of the Bay and times of the year, what are the relative contributions to Bay concentrations from wastewater treatment plants (WWTPs), LSB tributaries, other Bay tributaries, and Central Valley (Sacramento and San Joaquin River) inputs?

As shown in the previous results, flow contributions vary greatly depending on the Bay location and water year. For illustration, we focused on site P34, a long-standing RMP monitoring location where CEC samples have been collected in recent years. At this site, Stevens Creek's flow contribution remained minimal (~0-3.5%) throughout the year (Table 6). A dissolved-phase contaminant load from Stevens Creek would be diluted by 30 to over 100 times. Flow from WWTPs was small (5-8%) and varied slightly, as did flow from other tributaries north of the Dumbarton Bridge (<1-8%). It is worth noting that although 5-8% may sound small, for chemicals at high concentrations in wastewater, there might still be impacts even after 12.5 to 20-fold dilution a considerable distance from discharge points. Also, since stormwater inflow and infiltration can substantially augment flows, in some cases increasing WWTP influent by a factor of two to three, the estimates would likely be higher if actual wet weather flows were used as inputs rather than monthly averages. The largest and most variable contributions came from major LSB tributaries, mainly Coyote Creek and Guadalupe River, which could contribute up to 54%. Central Valley Rivers influence grew more slowly than other sources as the initial water was displaced, due to distance from LSB, and peaked at around 30% in May and June, then dropped to about 20% in the summer (Table 6). Therefore, the answer to the question is that at P34 and likely other Bay locations, relative flow contributions from WWTPs, LSB tributaries, other Bay tributaries, and Central Valley rivers vary greatly throughout the year.

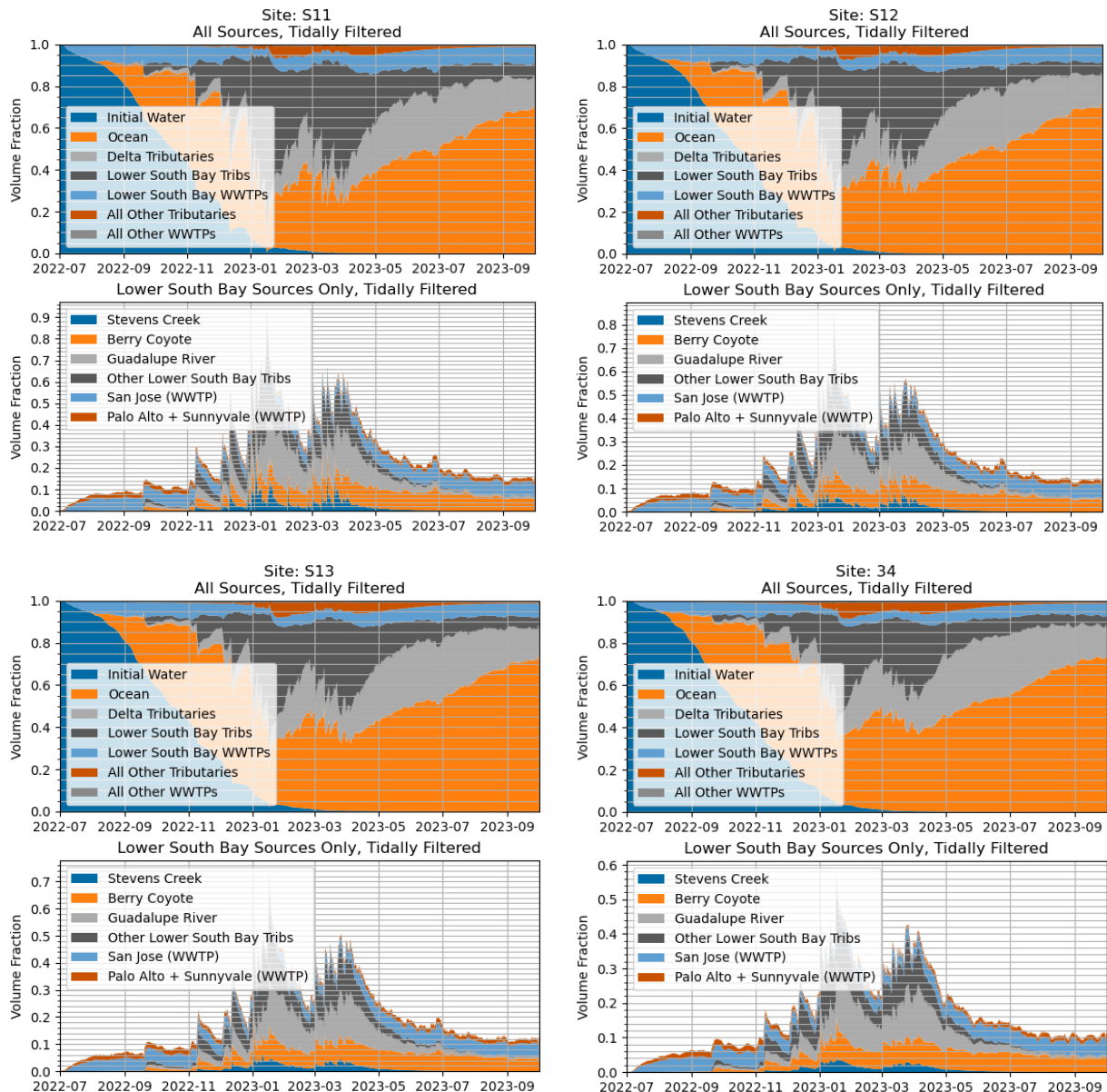


Figure 23. The provenance of fresh water at selected hypothetical sampling locations (S11, S12, S13) and at the USGS Peterson Cruise sampling site (P34) where CEC sampling was done during Water Year 2023. Please refer to Table 5 and Figure 17 for the geographic locations of these points. For flow conditions throughout the year see Figure 13.

MQ2 Is there a steady-state baseline established during the wet season or a build-up and/or dilution phenomenon due to successive runoff events?

Similar to MQ1, the responses to this question vary greatly depending on the location in the Bay and the modeled water year. However, based on the results for site P34 during WY 2023, no steady-state baseline was established for any water sources during the wet

Table 6. Provenance of water during storms at the USGS Peterson Cruise sampling site (P34) where CEC sampling was done during Water Year 2023.

Average Flow (m ³ /s)	Early wet season (November)	Mid wet season (January)	Late wet season (April)	Summer (August)
Delta	558	3161	2120	284
Stevens Creek	0.50	9.22	0.51	0.01

Provenance	Early wet season (November)	Mid wet season (January)	Late wet season (April)	Summer (August)
Wastewater treatment plants	9%	4%	6%	6%
Stevens Creek	<1%	3%	2%	~0%
Other Lower South Bay tributaries	10%	35%	24%	5%
Other Bay Tributaries	<1%	5%	5%	<1%
Central Valley Rivers	7%	11%	22%	20%

season, and the response varied by source. Flow from WWTPs was unique, diluted by other freshwater runoff sources during each storm and generally throughout the wet season. While dilution throughout the wet season is likely real, it is possible that the degree of dilution during storms is somewhat overestimated, as daily WWTP inflows and discharges also increase during storm periods, and this is not captured by our monthly time step for WWTP inflows. Other freshwater sources showed increases during the wet season, with variation depending on the source's location and size. For Stevens Creek and other LSB tributaries, contributions peaked after each rainstorm, being greatest during the largest storm of the year. These contributions diminished after each storm at a rate of about 20% in the first month, 10% in the second, and 2% in the third month. Although no steady state was established, multiple storms in succession tended to maintain an overall higher contribution from these watersheds in the wet season. Contributions from tributaries north of the Dumbarton Bridge also appeared to increase with each subsequent storm. The Central Valley Rivers contribution was highest during larger storms but generally increased through the wet season, peaking in June after a flow of 151,000 cfs from the Delta on March 18th and sustained flows of >60,000 cfs until tailing off after May. Of all the freshwater sources, the Delta would have taken the longest to reach Lower South Bay, so prior to January, when most of the initial water was displaced, Delta contributions are underrepresented by an unknown amount.

3.2 Dispersion of a Conservative Tracer After an Idealized Storm

To avoid confounding factors from real-world flow conditions, two idealized simulations were conducted using a synthetic storm and conservative tracer during the stable flow period from 9/1/22 to 9/19/22 using a plug flow of 107.2 m³/s over 6 hours and a 0.01 m³/s base flow (see methods section 2.3.6 for details). In the first simulation, the plug flow coincided with a flood tide, and in the second simulation, an ebb tide. A tracer with constant concentration of 1 was released during the 6-hour plug flow period. Despite not mirroring any specific real-world event or toxicity threshold, this synthetic storm tracer study is highly useful in illustrating local transport and dispersion processes to determine water provenance in the period immediately following a storm event.

3.2.1 Model Results

The results of the idealized simulations were examined over 7 days (168 hours) or about 13.5 tides. Model predictions suggest that the dilution characteristics of a flood-tide release would become similar to an ebb-tide release after about 24 hours, meaning a flood-tide release would be a slightly delayed version of an ebb-tide release. In Figure 24 we examine the sites along the channel (for a reminder of where these sites are located see Table 5 and Figure 17). Site S01 is at the model boundary and is included to ensure the release occurred as intended, with concentration equal to one for 6 hours after the beginning of the release. In Figure 25 we examine the correlation between tracer concentration and water depth at mudflat channel site S07. At all sites except S01, which is influenced by boundary effects, our model simulations suggest a similar pattern to that shown at S07, where tracer concentration drops during the rising tide and remains low and relatively constant around high tidal elevations, as the plug of tracer is pushed up into the tributaries, and tracer concentration increases again as tidal elevations drop and reach a high stable concentration on either side of and during low tide, as the tracer is drawn out into the open water. Superimposed upon this oscillation, we see a gradual dilution of concentration peaks as the tracer mixes with water from other sources. This process of oscillation and dilution can be seen in Figure 26, where we show tracer concentrations across LSB during four slack tides following the ebb tide release. At site S03, far enough from the boundary to be free of boundary effects but still within the tidal channel, the tracer diluted to 55% after 24 hours and 40% after 36 hours. In the tidal mudflat channel (S09), it was estimated that it would take 42 hours to dilute the tracer to 40%, mainly mirroring the repeated oscillation of the release into and out of the channel.

This was consistent with results from cross-channel sites (S04 and S06) (Figure 27), where concentrations were predicted to drop to nearly 10% after 48 hours. Within 6 hours of release, a plume from Stevens Creek would likely reach sites S11 and S12, diluting to <20% after 18 hours and <10% after 42 hours (Figure 24). The plume was predicted to reach the closest USGS Peterson site (P36), where concentrations peaked at 18% of the tracer release concentration at 6 hours post-release and dropped to 4% by 42 hours (Figure 28). At site P34 (CEC sampling location), maximum concentrations were estimated to reach just 5.5% around 18 hours post-release.

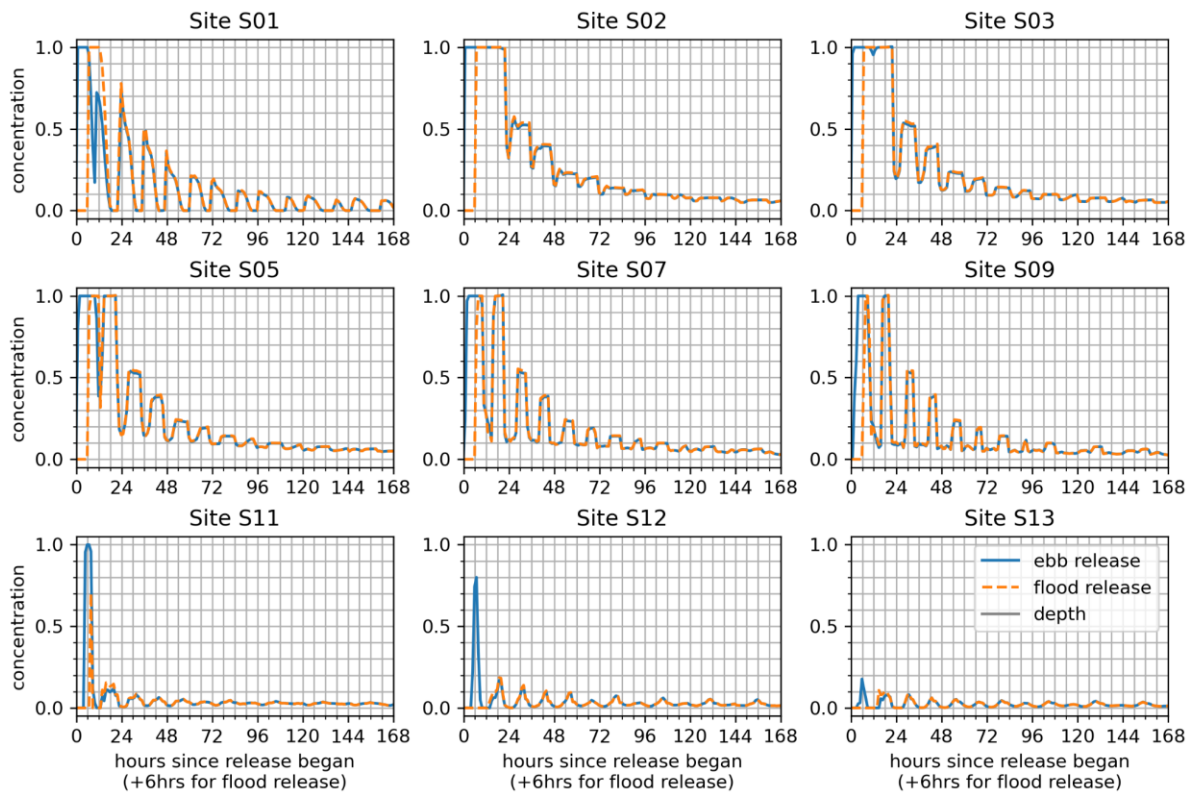


Figure 24. Dilution of a conservative tracer introduced into the Lower South Bay during ebb- and flood-tides using a 2-year return 24-hour synthetic storm in Stevens Creek watershed at the channel center-line sites and the subtidal sites. Please refer to Table 5 and Figure 17 for the geographic locations of these points.

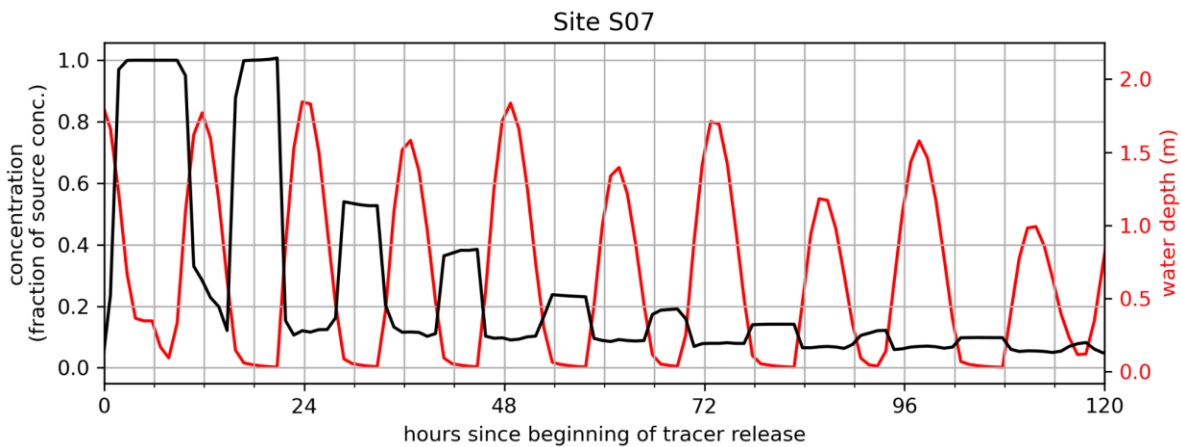


Figure 25. Concentration and water depth at site S07 for the ebb tide release.

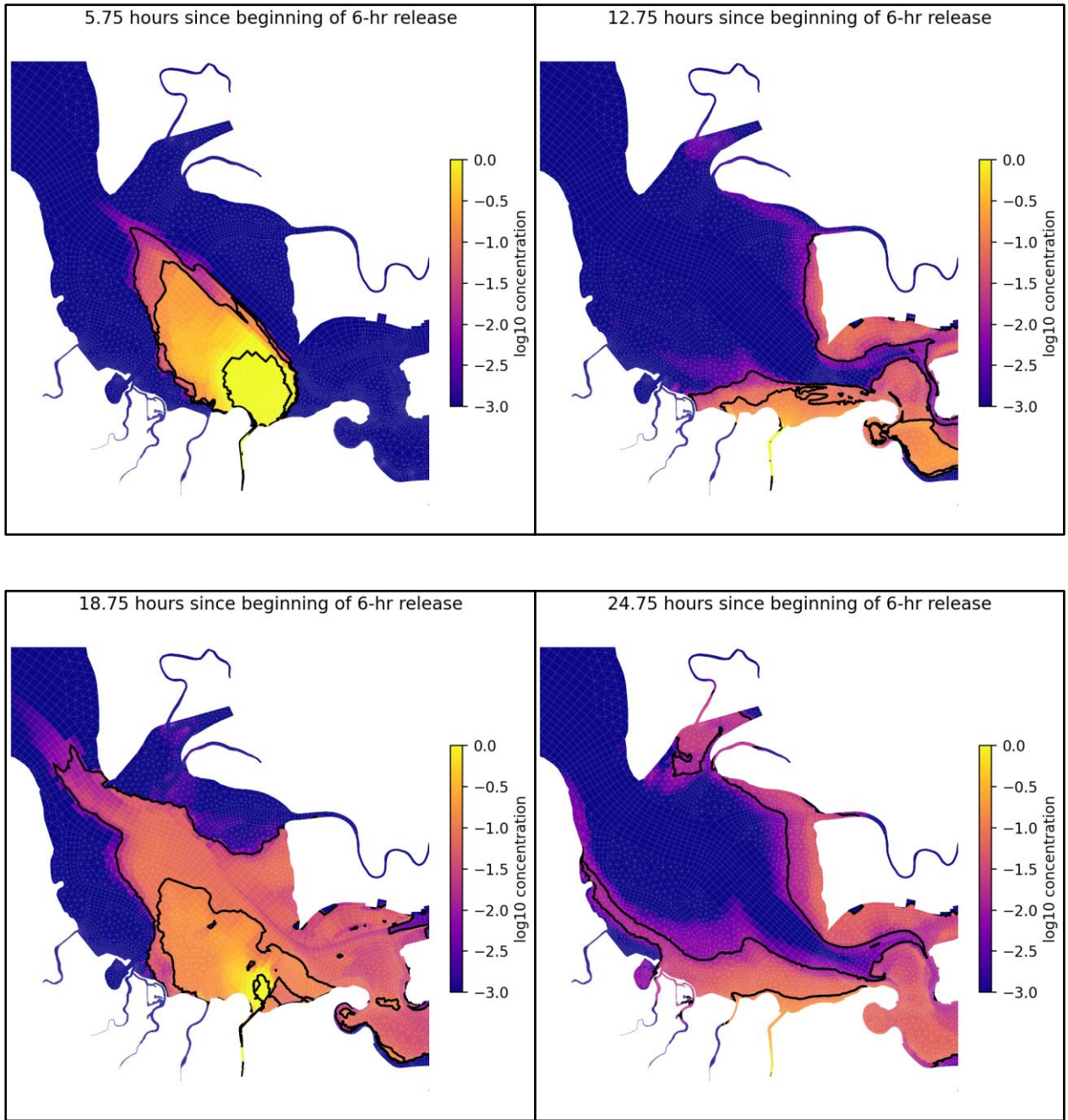


Figure 26. Tracer concentrations for the ebb tide release, at four times measured from the beginning of the 6-hour release: approximately 6.2 hours (low slack tide), 12.4 hours (high slack tide), 18.6 hours (low slack tide), and 24.8 hours (high slack tide). Times are slightly off from true slack due to hourly model output. Black contour lines are drawn at concentrations of 1, 0.1, and 0.01.

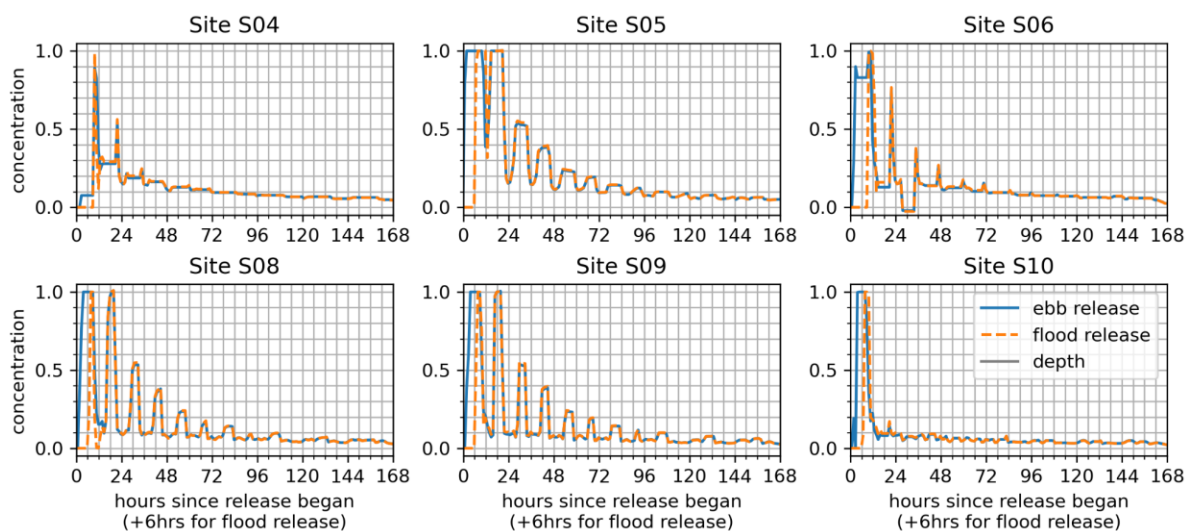


Figure 27. Dilution of a conservative tracer introduced into the Lower South Bay during ebb- and flood-tides using a 2-year return 24-hour synthetic storm in Stevens Creek watershed at the cross-channel sites. Please refer to Table 5 and Figure 17 for the geographic locations of these points.

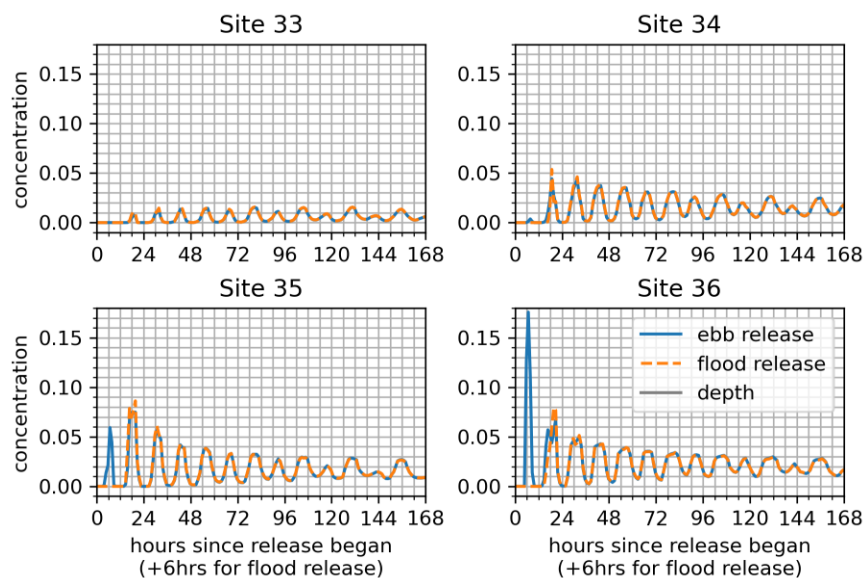


Figure 28. Dilution of a conservative tracer introduced into the Lower South Bay during ebb- and flood-tides using a 2-year return 24-hour synthetic storm in Stevens Creek watershed at the USGS Peterson sampling sites. Please refer to Table 5 and Figure 17 for the geographic locations of these points.

3.2.2 Responses to Dispersion Management Questions

MQ3: What are the maximum short-term concentrations of dissolved phase contaminants [anytime] during the wet season in different habitats of interest (channel, mudflat, subtidal), where in the Bay are these maximum concentrations, and how long do these elevated concentrations persist before returning to “background”?

Based on the idealized simulation with a synthetic storm of 1:2-year return frequency and the release of a conservative tracer from Stevens Creek into the Bay margin, the highest concentrations would likely be found in the tidal channels on the mudflat. In these channels, concentrations were predicted to be as high as 100% of those in terrestrial stormwater within the first 24 hours but would dilute to <40% by 2 days, <10% by 4 days, and <5% by 7 days post-storm. On the mudflat surfaces, concentrations were predicted to peak at >50% of terrestrial stormwater concentrations within the first 24 hours, but rapidly decline to 10% within 2 days at the upper mudflat sites and 4 days at the outer mudflats, reaching around 5% or less in all mudflat areas by 7 days. The maximum short-term concentrations at subtidal sites could be near 100% in the first 6 hours post-storm but would likely decline to around 10% within 24 hours and <5% within 4 days. In subtidal areas, concentrations may peak at >80% within 6 hours but rapidly dilute thereafter. Concentrations were predicted to be <5% at all sites at 7 days post-storm release.

MQ4: How should we sample in the Bay to capture the dispersion gradients away from treated wastewater discharges and tributary inputs at different locations in the Bay and times of year, to effectively characterize local exposure risk, and model fate and transport of various legacy and emerging contaminants?

The answer depends on the relationship between the peak terrestrial stormwater concentration and the detection limits of the contaminant of interest in Bay water. If detection limits are not a barrier to quantification, and Stevens Creek is a reasonable model for other stormwater-Bay interfaces, a proposed sampling program for dissolved-phase contaminants could involve collecting samples in a safe location in the tidal channel and adjacent mudflats within 72 hours of the storm peak, and at subtidal and Bay spine (USGS Peterson) sites within 7 days. If detection limits are a barrier to quantification, a faster program would be needed, sampling the tidal channel and adjacent mudflats during flood-tides within 24 hours of the storm peak, and subtidal and Bay spine sites within 72 hours.

3.2.3 Comparison of Observed Concentrations with Model-Simulated Dilution

For a pilot comparison of observations with model results we chose hexamethoxymethylmelamine (HMMM) measured during the 2021 - 2024 RMP Status and Trends wet season pilot study (Miller et al. 2025). HMMM is a cross-linking agent used in the production of rubber and coatings, is grouped under vehicle-derived contaminants in the RMP tiered risk-based framework for emerging contaminants and is present at high concentrations in stormwater. It is primarily a dissolved phase contaminant, and likely has

low concentrations in wastewater (e.g., Johannessen et al., 2022) although we do not yet have local Bay Area wastewater measurements. As for many other emerging contaminants, North Bay and Delta concentrations are thought to be low, as are ocean concentrations. HMMM is known to decay in the environment. The degradation half-life reported in the literature ranges from 23 days (Johannessen and Parnis, 2021) to 67 days (Eberhard et al., 2015). This is longer than the time for mass of a conservative tracer to fall to one half of the original mass released from Stevens Creek in our idealized tracer release study, which was approximately 8 days, as shown in Figure 29. But the lower end estimate for the degradation half-life is the same order of magnitude as this residence time scale. In this analysis we treat HMMM as a conservative tracer, recognizing the modeled HMMM coming from Lower South Bay stormwater is consequently overestimated. HMMM released from more remote sources, such as the tributaries north of Dumbarton or the Delta is also not modeled, but these remote sources are likely more reduced by ocean exchange and by longer degradation times before reaching Lower South Bay.

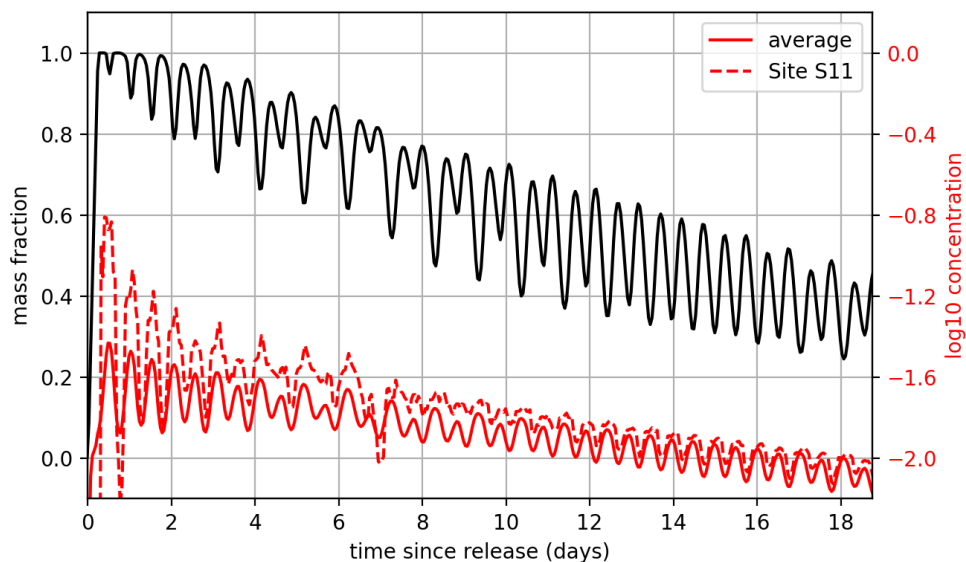


Figure 29. Total mass remaining within Lower South Bay (everything south of the Dumbarton Bridge) as a fraction of the total mass released is plotted on the left axis against time since the release for the flood tide idealized tracer release in Stevens Creek. Average concentration within Lower South Bay and concentration at Site S11 are plotted on the right axis.

In Figure 30, HMMM concentrations from Miller et al. (2025) are plotted as a time series along with flow rate from Guadalupe River plus Coyote Creek during the pilot study time window. During the wet season, concentrations were measured in the tidal channel of Stevens Creek (the same as Site S01 in Figure 17 and Table 5) during or shortly after a storm, and concentrations were measured in the subtidal channel of the Bay at USGS Peterson Sites 33 and 34 between 6 and 26 days after each wet season Stevens Creek

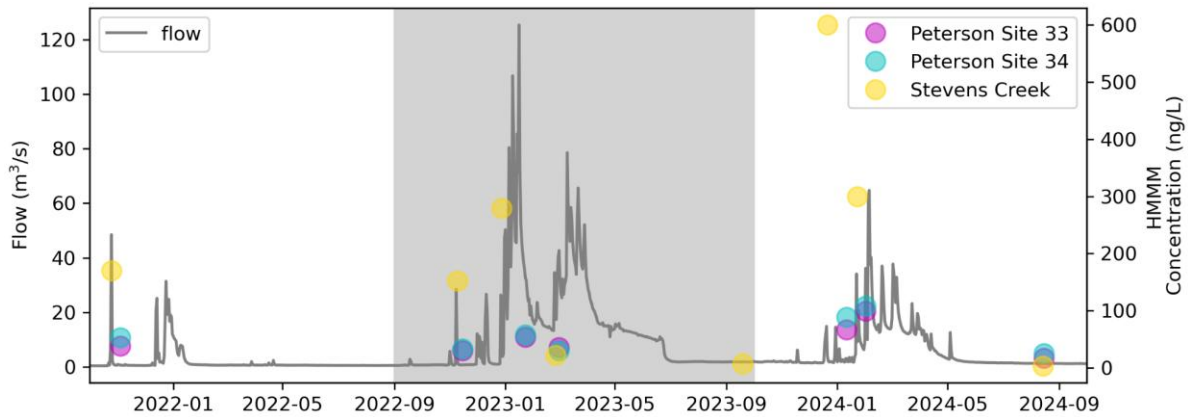


Figure 30. HMMM concentrations at the tidal channel site (Stevens Creek, which is the same as S01) and at USGS Peterson sites 33 and 34 are shown on the right axis. See Figure 17 and Table 5 for sampling site locations. Daily flow from the Guadalupe River plus Coyote Creek (measured at USGS sites 11169025 and 11172175, respectively) is plotted on the left axis. The time period covered by our WY 2023 simulation is shaded in light grey.

measurement. Concentrations were also measured during the dry season. Here we consider only the wet season measurements, and we exclude the February/March 2023 measurements because they occurred a long time after a large inflow event, and it appears the tidal channel water had mixed with the in-Bay water and no longer represented the watershed. Data for the remaining five storm events are shown in Table 7. For three out of these five storms, salinity was also measured at the Stevens Creek site. The median salinity was 5 psu indicating that during storm sampling, the Stevens Creek site was mostly fresh water but included some ocean water. Assuming an ocean salinity of 34 psu and ocean HMMM concentration of zero, we estimate the freshwater concentration at the Stevens Creek site by multiplying the measured concentration by $34 \text{ psu} / (34 \text{ psu} - \text{Salinity})$. Dividing the HMMM concentration measured in the subtidal channel (average of sites 33 and 34) by the preceding freshwater HMMM concentration for each storm event, we get fractions ranging from 0.10 to 0.29 (Table 7). These numbers fall within the range of model water fraction attributed to Lower South Bay tributaries at site P34 (Figure 23) which is approximately 0.10 to 0.45 (the low end 0.10 is approximate due to difficulty interpreting model results prior to full spin-up).

This comparison of model and observational data is imperfect for many reasons:

1. we have very little observational data;
2. the observational data only partially overlap with the model time frame;
3. we assume all Lower South Bay tributaries have the same HMMM stormwater concentrations as Stevens Creek;
4. we assume HMMM concentrations are zero in ocean water, water from the Delta, and wastewater;

5. we ignore small tributaries north of the Dumbarton (Figure 23 suggests the contribution from these is small); and
6. we assume HMMM is conservative, or at least that it does not decay or deposit significantly between Lower South Bay tributaries and USGS Peterson sites 33 and 34.

Table 7. HMMM concentrations measured (a) in the tidal channel of Stevens Creek and (b) at USGS Peterson sites 33 and 34, within the subtidal channel of Lower South Bay (average of these two in-Bay sites is also given), and (c) the in-Bay fraction with respect to stormwater (in-Bay average concentration divided by Stevens Creek concentration). In (a) we estimate the freshwater concentration by multiplying by 34psu / (34 psu - Salinity). When there was no salinity measurement we used the median of 5 psu.

(a) Site Stevens Creek				(b) Site P33 Site P34 In-Bay Avg.				(c)
Date	Conc. (µg/L)	Salinity (psu)	Freshwater Conc. (µg/L)	Date	Concentration (µg/L)			In-Bay Fraction
10/25/2021	170	-	199	11/4/2021	38	53	45.5	0.23
11/9/2022	152	-	178	11/15/2022	30	33	31.5	0.18
12/28/2022	279	5	327	1/23/2023	54	58	56	0.17
12/21/2023	600	9	816	1/11/2024	67	89	78	0.10
1/23/2024	300	5	352	2/1/2024	99	108	103.5	0.29

While sparse data will always be a problem for CEC sampling, many of these assumptions are reasonable for HMMM and it is encouraging that the observations generally agree with model predictions of provenance. This helps to illustrate the type of questions we could use the modeling to answer and also how improvements in observational data might help to address weaknesses associated with the assumptions and calibrate or verify any future modeling exercises. The model in its current form could be used to estimate dilution of CECs that are present in wastewater, the ocean, and/or the Delta provided those source concentrations are measured along with stormwater concentrations. Estimating dilution of CECs with significant decay, precursors, or partitioning would require further development of the model.

3.2.4 Implications for the Status and Trends Sampling Design

The RMP Status and Trends sampling design for near-field wet weather sampling includes one station in Stevens Creek at S01 and the P34 station in the open Bay. Similar

configurations have also been sampled in three other areas (San Leandro Bay, Redwood Creek, and Palo Alto) (Trinh et al., 2025). The design called for sampling at the Bay margin stations within hours of the end of a storm event and sampling in the open Bay within three weeks of a storm event, USGS Peterson cruise schedule permitting. The concept of this design was to assess the gradient and exposure field of concentrations emanating from these tributaries.

This integrated modeling exercise and the comparison to results obtained to date for HMMM was very enlightening. As expected, during a runoff discharge event, and for several tidal cycles following the event, modeled tracer concentrations in the Stevens Creek channel station and locations near the mouth of the channel are dominated by the loading from Stevens Creek. Modeling results over subsequent days indicate a reduced influence from Stevens Creek, and greater influence of a mixed signal integrating nearby local tributary inputs combined with dilution from the South Bay. The current sampling design at P34 similarly is representative of a regional mix of inputs. If there were a CEC unique to Stevens Creek, it would be most easily measured at the in-channel station and the intertidal zone near the mouth during a discharge event and for a few tidal cycles after. However, by the time the discharge reached P34, the signal would be diluted to concentrations that might not be easily detected. The modeling also highlights that monitoring of in-channel tidal areas may suffice for estimating localized short-term concentrations and exposure risk, given the challenges of sampling mudflats around tributary mouths due to difficult access during inclement weather. Once the discharge reverts to baseflow, the concentrations at the channel mouth at low tide and in the channel during the following high tide are of a similar magnitude.

A continued focus on monitoring tidal areas of urban creeks and continued refinement of models is recommended to estimate exposure fields for relevant habitats and beneficial uses. However, if stakeholders desire more certainty, a modified sampling approach could be used for open water areas if there is a need to focus on inputs from a specific tributary rather than just a mixed regional signal. Such monitoring in open water areas would need to occur nearer the tributary input and within a few days after a discharge using a reactive on-demand logistically difficult sampling plan, rather than relying on sampling with a fixed schedule that could end up several weeks after a storm.

4. Conclusions

This pilot study coupled WDM outputs with an in-Bay hydrodynamic model with a much higher degree of integration and much higher temporal and spatial resolution than most prior efforts. Using these integrated models, we explored management questions regarding the dispersion of an idealized dissolved-phase contaminant from watersheds into the Bay. Insights from this pilot, along with findings from the San Leandro Bay study on sediment and PCBs (Scheu et al. 2025), are helping to refine the overall RMP watershed-Bay modeling strategy and workplan (Heberger et al. 2025).

Using the integrated models, we estimated the provenance of freshwater in different parts of the LSB throughout the year, including the contributions from Stevens Creek, WWTPs, other LSB tributaries, Bay tributaries in other segments, and the Central Valley rivers (the Delta). We also simulated the dispersion of a dissolved contaminant for an idealized storm discharge from Stevens Creek, showing how the contaminant would be transported and dispersed by tides as it moves from its initial tidal discharge channel, through the mudflat, to the shallow subtidal Bay, and out to the deeper RMP Status and Trends long-term sampling sites on the Bay spine.

This model study serves as a complementary companion to the San Leandro Bay sediment and PCB model study (Scheu et al., 2025), which considers a simplified idealized constituent at the other extreme: a conservative contaminant present only on sediment particles. In that study, the provenance of sediment (rather than water) distributed in the water body was characterized over time. Because the transport of sediment particles is highly sensitive to flow velocity, more finely resolved flows may be needed to understand the fate of sediment-associated contaminants. For sediment-associated contaminants, the residence time in biologically active sediment may be decades before they are sufficiently buried, transported, degraded, or diluted out to avoid impacts, so temporally this represents the opposite extreme from a purely dissolved-phase contaminant.

Real-world contaminants are often a mix of dissolved and particulate forms, but these simplified model studies are valuable initial steps in developing tools to assess contaminants of future interest. Even with highly detailed models at fine time scales (e.g., hourly or event-based), modeling multiple years of future scenarios would be inefficient. While a future year may have similar total rainfall, the distribution and intensity of individual storms are unpredictable and cannot be reliably forecasted. Extreme events beyond our current monitoring range add further uncertainty. However, despite these uncertainties, models offer a useful starting point for understanding how the system might respond to extreme conditions.

The approach used in these two studies—tracking the provenance of water and sediment in receiving waters through unique conservative tracers from each discharge point—has proven to be a flexible and valuable tool. Although the loads and concentrations of contaminants from various pathways are uncertain or unknown, modeling the provenance of water or sediment at different times or locations allows us to explore the expected distribution of any contaminant as a combination of the proportions and concentrations from each pathway. For instance, if ambient concentrations are too high to be explained by extrapolated concentrations from known watersheds, we can determine how high concentrations would need to be in unmeasured watersheds to account for the difference. This approach is also helpful for modeling potential management actions, such as reducing or eliminating concentrations from specific pathways. These scenarios can be easily explored by adjusting input values in a spreadsheet, without the need to re-run complex models.

A logical next step could be to expand the spatial scale of the study using a similar approach as has been done for nutrients. While applying unique tracers to all individual pathways at the whole Bay scale is impractical, focusing on the largest loads and grouping smaller pathways may offer a manageable and flexible way to examine loading and fate processes on a larger scale for selected CECs where mass balance maybe of interest. This approach would allow for similar spreadsheet-based scenario explorations to explore potential management outcomes across the broader spatial area.

These simple tracer models will not address all the needs for modeling contaminants. For example, there can be transfers between water and sediment, which may lower concentrations in one compartment while increasing them in the other. Degradation or transformation processes can reduce concentrations of individual compounds while creating more toxic by-products (e.g., from 6PPD to 6PPD-Q, Tian et al., 2021). Although unlikely, losses to the atmosphere or direct deposition through gas transfer, rainfall, or particulate matter deposition could also be important pathways for some contaminants and may be necessary to explain observed concentrations.

This model of integrated watershed-Bay transport processes represents a good initial effort and foundation upon which more complex models can be built. If the simple models prove inadequate to explain observed concentrations and distributions, we could add to model complexity (and resultant additional data to inform it) as necessary, as outlined in the RMP In-Bay modeling strategy (Jones et al., 2022). By adding complexity to our models only as needed, we can most efficiently evaluate the transport and fate of various legacy and emerging contaminants, and the resultant exposure risks they pose to resident biota, informing possible management actions.

5. Recommendations for Future Similar Modeling Studies

Based on this pilot modeling study, possible areas for improvement in the model setup and supporting data include the following.

1. Despite revisions to the grid and bathymetry, inflow and outflow did not always match in Stevens, Adobe Creek, Permanente Creek, and Mowry Slough. Issues such as stranding in puddles arose due to under-resolved bathymetry in the upland sections of the streams. These problems were addressed by artificially flattening the bathymetry because we did not have time to refine the grid to capture the deepest part of the channel. While flood-tide tracer release results were unaffected by these adjustments, ebb-tide results remained sensitive to the original bathymetry. To predict dynamics within intertidal channel habitats, future studies should employ a more refined grid within this region, though results in the Bay's mudflat and subtidal areas were not notably influenced by the grid's resolution in the upper reaches.
2. Related to 1, to predict concentrations within the tidal channels in the hours and days following a discharge, the model boundary should be extended to the

observational sampling point furthest upstream. Sample concentrations at this point can be used as a boundary condition.

3. Recommendations 1 and 2 are important for predicting exposure within the tidal channels. However, the results of this pilot suggest they have little or no effect on predictions within the subtidal regions of Lower South Bay; for this use, the effort made to refine the grid may have been unnecessary. A comparison of the refined and un-refined model grid results for USGS Peterson site concentrations could be used to evaluate the impacts of grid improvements and help us select the appropriate grid to match study question needs.
4. The NMS open Bay hydrodynamic model uses a spin-up period of two months, which is sufficient for water level, velocity, temperature, and salinity, but two months was not sufficient to spin up water provenance. For water provenance simulations of Lower South Bay, a spin-up period of eight months should be used. This should increase computation time from approximately 3 days to 4.3 days, which is not an unreasonable burden.
5. The WDM was calibrated for streamflow at a daily time step. Although the WDM produces hourly output, further refinement of model parameters is needed to improve model performance at an hourly time step. However, sensitivity of Bay model predictions of water provenance to the WDM model time step should be assessed prior to investing time in improvements to the hourly time step WDM model.
6. The in-stream hydraulic simulation of the WDM relies on channel geometry. Channel geometry (i.e., depth and width) was estimated from empirical and local watershed area-geometry relationships; however, these relationships may not apply to urban streams. LiDAR topography and channel geometry data from local agencies could help enhance the model's accuracy (Heberger et al., 2025).

Lessons Learned from this Effort

Through the implementation of this study, we have found some challenges with model configurations and sensitivity, and spatial or temporal resolution interacting with model characteristics to cause artifacts such as unrealistic pooling in intertidal channels. Through this process, we are identifying ways to improve the models individually and better integrate the watershed and in-Bay models at priority areas and the whole-Bay scale.

1. The inclusion of tidal wetlands is likely not needed to accurately simulate provenance and dilution characteristics at the segment scale; tidal wetland flow volumes are small relative to other flows. However, for individual channels, the adjacent tidal wetlands may have major impacts on in-channel processes (e.g. hydrodynamics, oxygen demand, etc.).
2. The timing of discharge relative to the tidal phase was shown to have little impact on concentration after several days. Separate simulations of discharges occurring on ebb- and flood-tide are generally not needed.
3. Related to the above finding, hourly resolution of tributary inputs is not necessary for in-Bay concentration studies, but for estimating concentrations in the intertidal

portion of the channels and the intertidal shallow areas in the hours and days following discharges, at least sub-daily resolution of inputs is helpful.

4. For many purposes, a simplified or idealized “worst case” simulation may be sufficient, and is useful for assessing model sensitivity and possible artifacts, before the model includes too many complex interactions. For example, if even under an idealized case, a contaminant will not reach a toxic threshold even nearby for short periods, we may be less concerned. If the prediction is ambiguous and concern remains, then introducing complexity to obtain better model fidelity with environmental processes is warranted.

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7. Appendix

Link to electronic appendix: https://www.sfei.org/sites/default/files/biblio/2026-10/Stevens_Ck_Watershed-Bay_Model_FINAL_Report_Appendix_Hydro_Model_Validation.pdf