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San Francisco Bay Watershed Dynamic Model (WDM) Progress Report, Phase 2

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Preface

The San Francisco Bay total maximum daily loads (TMDLs) call for a 50% reduction in mercury (Hg) loads by 2028 and a 90% reduction in PCBs loads by 2030. In support of these TMDLs, the Municipal Regional Permit for Stormwater (MRP) (SFBRWQCB, 2009, SFBRWQCB, 2015, SFBRWQCB, 2022) called for the implementation of control measures to reduce PCBs and Hg loads from urbanized tributaries. In addition, the MRP has identified additional information needs associated with improving understanding of sources, pathways, loads, trends, and management opportunities of pollutants of concern (POCs). In response to the MRP requirements and information needs, the Small Tributary Loading Strategy (STLS) was developed, which outlined a set of management questions (MQs) that have been used as the guiding principles for the region's stormwater-related activities. In recognition of the need to evaluate changes in loads or concentrations of POCs from small tributaries on a decadal scale, the updated 2018 STLS Trends Strategy (Wu et al., 2018) prioritized the development of a new dynamic regional watershed model for POCs (PCBs and Hg focused) loads and trends. This regional modeling effort will provide updated estimates of POC concentrations and loads for all local watersheds that drain to the Bay. The Watershed Dynamic Model (WDM) will also provide a mechanism for evaluating the impact of management actions on future trends of POC loads or concentrations.

As a multi-use modeling platform, the WDM is being developed to include other pollutants, such as contaminants of emerging concern (CECs), sediment, and nutrients and to be coupled with a Bay fate model to form an integrated watershed-Bay modeling framework to address Regional Monitoring Program (RMP) management questions. As this model is developed, flexibility to link with other models will be an important consideration.

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Executive Summary

A new, dynamic regional watershed model (Watershed Dynamic Model [WDM]) for Bay Area hydrology, sediment, and stormwater contaminant loads and trends is being developed by the Regional Monitoring Program for Water Quality in San Francisco Bay (RMP). The watershed of this new model is the area that drains to San Francisco Bay (SF Bay) from the nine adjacent counties around the SF Bay. The hydrological model development and calibration was documented in a Phase 1 report (Zi et al, 2021).

This is Phase 2 of the modeling effort focused on suspended sediment simulation. The WDM sediment module (Phase 2) provides the total suspended sediment load (SSL) estimate for the Bay area by considering both upland and channel processes at an appropriate time step to represent the physical processes (e.g., rainfall detachment, sediment settling). The WDM sediment estimate takes into account the spatial and temporal heterogeneities of the erosion and sediment transport processes in the region by differentiating landscape features as well as continuously simulating the physical processes driven by the weather data.

Several modifications have been made to Phase 2 of the WDM, including watershed delineation, channel geometry, land use and imperviousness, and hydrologic response units. The hydrologic calibration was extended to 26 years (1995-2020). The hydrology and sediment simulation was conducted at the hourly time step. Modeled sediment load was calibrated and validated against SSL, suspended sediment concentration (SSC), and suspended sediment particle distribution data to develop a robust estimate of annual average SSL from local tributaries to the SF Bay. In general, the current WDM sediment module has a solid simulation performance on the total sediment load from local tributaries to the SF Bay. The model has good performance at calibrated watersheds during the calibration period. The model also shows consistently good performance in interannual sediment load variation, discharge-SSC relationships, and particle size distribution at different locations. For uncalibrated watersheds, the model still shows reasonable representations of the physical processes. High temporal and spatial variability in SSL estimates generated by the WDM are comparable to the variability derived from long-term monitoring data.

The estimated 26-year total SSL from local tributaries to the Bay (1995 to 2020) was 33.64 Mt. The estimated annual average total sediment supply from local tributaries to the Bay for this time period was 1.29 Mt with a standard deviation of 1.06 Mt. The modeled average annual SSL from local tributaries is close to the average annual SSL derived from monitoring data. San Pablo Bay had the largest sediment supply, 697,003 metric tonnes per year, 56% of the total SSL from local tributaries. Suisun Bay received the second highest amount, about 16% of the total load. Central Bay and South Bay each received about 12% of total load, and Lower South Bay 4% of the total load. A Mann-Kendall monotonic trend analysis on the 26-year modeled sediment load did not identify a significant trend of average annual sediment load at the regional scale.

This dynamic watershed sediment model is now available for the Bay Area to estimate the total

sediment and specific sediment classes (sand, silt, clay) load for the whole region and for specific watersheds. It can estimate the sediment yield from different land uses and simulate sediment dynamics within channels. Aided by monitoring data, this numerical model can be used to better understand the sediment transport processes and budgets of San Francisco Bay. This sediment modeling tool can be used to refine watershed sediment management and serves as a solid basis for sediment-associated contaminant load modeling in the next phase of the WDM and will be used as watershed boundary condition inputs to the in-Bay PCB fate model being developed by the PCB Workgroup.

The sediment calibration in Phase 2 is generally more uncertain than the hydrologic calibration in Phase 1 because it is difficult to simulate varied and localized sediment processes, and because there is a lack of sufficient sediment data to accurately calibrate the model. Although the WDM has been calibrated against monitoring data at several locations around the region, some major data gaps, the level of complexity of sediment dynamics, and the lack of understanding of the details of related physical processes were major sources of uncertainties.

A few recommendations are proposed to reduce the uncertainties of sediment load estimation and better understand the sediment dynamics at the SF Bay area.

1. Verifying and improving the accuracy of land use and vegetation cover inputs to the model (the current model impervious HRU area is relatively low compare to previous RAA model at the same region) and to include temporal change over the model calibration period (currently 1995-2020). These parameters have a large influence on the production of flow, sediment and contaminants from the landscape and is important on quantifying the contributions of different HRUs. This is also important for contaminant (PCBs, Hg, CECs) load estimation.
2. Gathering monitoring data at urban regions (impervious surfaces) for sediment accumulation, washoff rate, and removal rate (such as street sweeping) to better parameterize the sediment transport process at impervious surfaces. These monitoring data can improve sediment associated contaminant load estimation.
3. Monitoring processes such as bank erosion, landslides, debris flows at large spatial scale, identifying these events spatially, and qualitatively or quantitatively describing the magnitude of those processes. Doing this can help quantify the sediment supply from these events which the model does not simulate currently. It can also help parameterize and verify the bank erosion module of the sediment model.
4. Conducting monitoring to understand some key modeling parameters, such as sediment particle size distribution at land surfaces and at channel beds, channel geometries and cross-sectional area.
5. Completing sediment gauging in the watersheds of Sonoma Creek, Napa River, and Walnut Creek, three large watersheds that produce a lot of sediment but that have no or limited recent data to support either load estimates for specific years or model calibration.
6. Evaluating the impacts of wild fire through some land processes such as soil sealing and removal of vegetative cover. More post-fire monitoring is recommended for regions that are prone to wild fire events to help the WDM better represent the wild fire impacts on

sediment dynamics. Wildfire can have large impacts on sediment dynamics, especially in the Napa River and Sonoma Creek watersheds.

7. Conducting a thorough model sensitivity and uncertainty analysis on different factors to prioritize the key factors that affect the modeling results most and to quantify the modeling uncertainties by cross-validation and sensitivity analysis.
8. Extending the model for more years (backwards to capture more of the existing sediment data from the 1970s and 1980s or forwards if coupled with more sediment gauging in the Napa, Sonoma, and Walnut Creek watersheds) and using it for trend analysis or future predictions in relation to land use and climate change.
9. Conducting thorough trend analysis based on the results model sensitivity analysis and uncertainty analysis. The aim of the trend analysis is to explore the sediment load responses given different possible future changes (e.g., climate, land use, management actions) and to support management actions.
10. Utilizing multiple monitoring resources, such as remote sensing images and high frequency turbidity sensors, to increase the spatial and temporal coverages of monitoring data.
11. Recalibrating the sediment model every few years to keep the model representations up to date with the changes of erosion and sediment transport processes.
12. Adding monitoring activities that are designed for different pathways. The main purpose of the sediment simulation is to support estimation of contaminant loads (sediment and pollutant) by different pathways. A key factor is representing the right balance between pathways – e.g., between upland erosion and channel erosion. Direct evidence on the fraction of sediment load that is derived from source areas in recent contact with the atmosphere (e.g., tillage, roadway solids, and upland sheet and rill erosion) and those that have not (e.g., bank failure, gully formation, mass wasting) can be obtained through atmospheric radionuclide isotope analysis (Pb, Be, Cs).

The goal of this report is to document the erosion and suspended sediment transport modeling development and calibration effort, provide a sediment load estimate from local tributaries to the Bay, and facilitate discussions that lead to consensus among stakeholders on future model development. This report also sets the stage for Phase 3 of the modeling, which will focus on pollutants of concern. The overall goals of the WDM Phase 2 development include a better understanding of suspended sediment loads for stormwater contaminant load estimation. The model and data collected for the model development can be used to further support watershed management and stormwater management practices. The WDM model is intended to be a tool for all RMP stakeholders.

Contents

Abbreviations and Acronyms

ACCWP	Alameda Countywide Clean Water Program
BAHM	Bay Area Hydrologic Model
BAMSC	Bay Area Municipal Stormwater Collaborative
BASMAA	Bay Area Stormwater Management Agencies Association
BCDC	San Francisco Bay Conservation and Development Commission
CDWR	California Department of Water Resources
CCCWP	Contra Costa Countywide Clean Water Program
CECs	contaminants of emerging concern
CONUS	The contiguous United States
CSO	combined sewer overflow
CSS	combined sewer system
CWA	Clean Water Act
DCIA	directly connected impervious area
DEM	digital elevation model
EBMUD	East Bay Municipal Utility District
ECWG	Emerging Contaminants Workgroup
EIA	effective impervious area
EMC	event mean concentration
EPA	U.S. Environmental Protection Agency
ET	evapotranspiration
GGB	Golden Gate Bridge
GIS	geographical information system
GSI	green stormwater infrastructure
HRU	hydrologic response unit
HSPF	Hydrologic Simulation Program – FORTRAN
HUC	Hydrologic Unit
LSPC	Loading Simulation Program in C++
MAE	mean absolute error
MQs	management questions
MRP	Municipal Regional Permit
MS4	municipal separate storm sewer system
MTC	Metropolitan Transportation Commission
NCDC	National Climatic Data Center
NLDAS	North American Land Data Assimilation System
NLCD	National Land Cover Dataset
NMS	Nutrient Management Strategy
NOAA	National Oceanic and Atmospheric Administration
NPDES	National Pollutant Discharge Elimination System
NSE	Nash-Sutcliffe Efficiency
PAHs	polycyclic aromatic hydrocarbons
PCBs	polychlorinated biphenyls
PCBWG	PCB Workgroup

PET	potential evapotranspiration
POCs	pollutants of concern
PRISM	Parameter Elevation Regression on Independent Slope Model
RAA	Reasonable Assurance Analysis
RE	relative error
RMP	Regional Monitoring Program for Water Quality in San Francisco Bay
RMSE	root mean squared error
RWSM	Regional Watershed Spreadsheet Model
SCVWD	Santa Clara Valley Water District
SFEI	San Francisco Estuary Institute
SFPUC	San Francisco Public Utilities Commission
SFBRWQCB	San Francisco Bay Regional Water Quality Control Board
SPLWG	Sources, Pathways, and Loadings Workgroup
STLS	Small Tributary Loading Strategy
SCC	California State Coastal Conservancy
SedWG	Sediment Workgroup
SWMM	Stormwater Management Model
TIA	total impervious area
TMDL	total maximum daily load
TRC	Technical Review Committee
TSS	total suspended solids
USDA	U.S. Department of Agriculture
USGS	U.S. Geological Survey
USLE	Universal Soil Loss Equation
WBD	Watershed Boundaries Datasets
WDM	Watershed Dynamic Model

1. Introduction

1.1 Background

The San Francisco Bay TMDLs call for a 50% reduction in mercury (Hg) loads by 2028 and a 90% reduction in polychlorinated biphenyls (PCB) loads by 2030 relative to 2002, the baseline year. In support of these TMDLs, the Municipal Regional Permit for Stormwater (MRP) (SFBRWQCB, 2009; SFBRWQCB, 2015) called for the implementation of control measures to reduce PCB and Hg loads from urbanized tributaries. In addition, the MRP identified additional information needs associated with improving the understanding of sources, pathways, loads, trends, and management opportunities of contaminants. In response to the MRP requirements and information needs, the Small Tributary Loading Strategy (STLS) was developed and outlined a set of management questions (MQs) that guide the stormwater-related activities conducted by the Sources, Pathways, and Loadings Workgroup (Table 1, SFEI, 2009; Wu et al., 2018). Zi et al. (2021) completed a hydrologic module setup and calibration of the Watershed Dynamic Model (WDM) for the watersheds draining to the Bay from the nine surrounding counties (excluding the Sacramento and San Joaquin rivers). This report covers Phase 2 of the WDM development, which is the setup of the sediment module. The sediment module, after calibration, will provide an estimate of suspended sediment load from local tributaries to the Bay, and together with the flow module, will serve as the basis for stormwater contaminant load modeling and trends evaluation. Example applications of the model for water and sediment loads (Table 1) will later be extended to stormwater contaminants in those environmental matrices.

Table 1. Management questions used by the Sources, Pathways, and Loadings Workgroup (SPLWG) to prioritize studies, as well as example application of the WDM to these questions.

Management Question	**Example Information Application**
Q1: What are the loads or concentrations of Pollutants of Concern (POCs) from small tributaries to the Bay?	The model will produce an estimate of flow, sediment, and POC concentrations and loads for each individual watershed.
Q2: Which are the high-leverage small tributaries that contribute or potentially contribute most to Bay impairment by POCs?	Estimates of concentration, load, or yield produced by the WDM at each watershed can be compared to explore relative loading rates and their relationship to specific priority margin areas (Yee et al., 2019), operational landscape units (Beagle et al., 2019), or Bay segments.
Q3: How are loads or concentrations of POCs from small tributaries changing on a decadal scale?	Time series of flow, sediment, and POC loads for 1995-2020 can be used to assess trends for individual watersheds and the region. The model could be extended in the future to include additional water years.
Q4: Which sources or watershed source areas provide the greatest opportunities for reductions of POCs in urban stormwater runoff?	Model outputs of flow, sediment, and POCs will help identify high-yield areas within watersheds that can be targeted for management actions.
Q5: What are the measured and projected impacts of management action(s) on loads or concentrations of POCs from small tributaries, and what management action(s) should be implemented in the region to have the greatest impact?	Management actions, both existing and planned or anticipated, could be evaluated in the model through scenario runs to predict future loads based on land-use management scenarios and climate changes.

Over the past decade, considerable effort, including both field monitoring and modeling, has been made by the RMP and Bay Area county municipal stormwater programs to address these management questions. These efforts have mostly focused on addressing Q1, Q2, and Q4. Questions remain as to how loads at the regional scale have changed and will change as a result of decades-long management actions in relation to TMDL goals (Q3). In recognition of the need to answer Q3, the updated 2018 STLS Trends Strategy (Wu et al., 2018) prioritized the development of a new dynamic regional watershed model for pollutant of concern (POC: PCBs and Hg) loads and trends and developed a multi-year plan to obtain initial answers by 2022. Zi et al. (2021) started the WDM model development in 2020 and will have the regional PCBs and Hg baseline load estimation available by the end of 2022 following three planned phases (hydrology, sediment, and POCs). In addition to addressing Q3, this WDM effort will also directly support Q1, Q2, and Q4 by providing updated estimates of PCBs and Hg concentrations and loads for all watersheds in the region. The WDM could also provide a mechanism to evaluate current or planned management impact on trends of PCBs and Hg loads or concentrations in support of Q5. The study will provide information essential to understanding spatial and temporal characteristics of hydrology, suspended sediment and contaminant loads, at the scales of individual watersheds and the whole region to address the SPLWG high-level management questions.

Phase 1 of model development focused on the hydrology of the watersheds (Zi et al. 2021). Phase 2 modeling efforts focused on sediment erosion and suspended sediment transport module development and simulation. Sediment is an important constituent in the Bay, targeted for research and management actions to inform sea level rise adaptation (Schoellhamer et al., 2018), contaminant transport (Davis et al., 2014; Jones et al., 2022), and light-limited primary productivity (SFEI, 2014). The RMP Sediment Workgroup (SedWG) has identified estimating sediment loads from Bay Area watersheds as a research need (McKee et al., 2020) and the NMS is using estimates and trends of sediment loads to help support modeling efforts to estimate future algal biomass and bloom occurrence in the Bay (White et al., 2021). Legacy contaminants entering the Bay, such as PCBs and Hg are mainly sediment-associated. The completion of the sediment module is a prerequisite for the planned Phase 3 PCBs and Hg modeling.

In addition, the WDM can also be extended for other contaminants, such as contaminants of emerging concern (CECs) and nutrients. The Emerging Contaminants Workgroup (ECWG) has developed a CEC Strategy that identifies stormwater as a significant pathway for many CECs and calls for a combined modeling and monitoring approach to estimate their loads (Lin et al., 2018). A CECs load model exploration project is assessing the functionalities of different modeling platforms, including the WDM, for CEC stormwater loads estimation. A stormwater CECs groundwork project has been proposed to further the CEC stormwater load estimation with monitoring and data analysis for modeling.

The WDM has already been applied to support other programs and projects in the Bay Area beyond the RMP, such as evaluating sand budgets from local tributaries to the Bay (SFEI-ASC, 2022) and providing hydrological boundary conditions for NMS in-Bay modeling. As this model is developed, flexibility to link with other models will be considered, such as providing the boundary conditions to the in-Bay fate modeling (Jones et al., 2022). The WDM together with an in-Bay model can represent the full pathway from upland sources to receiving water and simulate the fate of contaminants to and within the Bay. Supplemental Environmental Project (SEP) funds are currently supporting the development of an integrated watershed-Bay modeling strategy and pilot study. The SEP project is developing the road map for integrating watershed and in-Bay models to help address RMP management questions. The strategy will document the rationale of applying the suite of integrated modeling frameworks to address RMP management questions, suggest future modeling efforts, and use a case study to demonstrate proof of concept.

1.2 Previous Watershed Sediment Supply Studies

The WDM sediment model estimates the suspended sediment load from local tributaries to the Bay. A few sediment budgets have been developed for San Francisco Bay as a whole (Krone 1996; Schoellhamer et al. 2005; Perry et al. 2015). Krone (1996) computed a sediment budget for the Bay for the period 1955-1990 and suggested a net flux to the ocean of 3.35 million cubic yards (2.6 Mm^3) of total sediment per year. Calculating a total sediment budget to the Bay consists of three major terms: 1. sediment supply from the Central Valley; 2. sediment supply from the ring of watersheds surrounding the Bay; and 3. sediment fluxes at the Golden Gate Bridge. The sediment contribution from local tributaries is now thought to be the dominant source of sediment to the Bay, with an annual average load of 1.4 Mt/y (60%) compared with just 0.9 Mt/y from the Central Valley (McKee et al., 2013). These estimates are based on rating curve methods developed for gauged watersheds and land-use based extrapolation to ungauged watersheds. The previous estimates were challenged by limited monitoring data, the use of rating curves based on an annual time step, and a simple land use-based extrapolation method. However, sediment production in the small tributary watersheds of the Bay Area is known to vary considerably between differing geological types, slopes, and climatic conditions (McKee et al. 2013, SFEI-ASC 2017; East et al., 2018).

Several sediment supply estimations from local tributaries have been conducted in the past. In 2000, stormwater flows, sediment, and pollutant loads were estimated using a simple rainfall/runoff model (Davis et al., 2000). The simple model is a deterministic model that uses a linear relationship between actual total stormwater volume, annual rainfall amount, and land use, as well as a linear relationship between load and stormwater volume using an average concentration for each distinct land use type. McKee et al. (2003) developed a conceptual model of sediment sources, pathways, and loads from small tributaries and suggested the total local tributary sediment supply to the Bay could be 2-3 times the value estimated by the Davis et al. (2000) simple model. Building from that work, Lewicki and McKee (2010) evaluated hydrologic, physical, and land use characteristics of the San Francisco Bay watersheds to predict relationships between watershed sediment loads and geomorphic processes, and, ultimately, to provide an updated estimate of regional suspended sediment loads from small tributaries. They modified the load estimates by a delivery ratio to simulate edge-of-stream loads (the loads reaching the boundary of the stream) from the surrounding watershed area (NRCS, 1983), whereby the proportion of sediment delivery to the channel decreases as watershed size increases. Annual average loads were estimated to be 1.3 M metric tonnes, 4-fold greater than estimated by Davis et al. (2000) and 1.67-fold greater than estimated by McKee et al. (2003). McKee et al. (2013) extended the work of Lewicki and McKee (2010) for a longer period of time and used the results to discuss the variable relationship between suspended sediment supply from the Central Valley versus local tributaries in the nine counties that surround the Bay in relation to climate. These concepts were further explored by

Schoellhamer et al. (2018) who for the first time added an estimate of bed load and explored the influences of local tributaries flood control channel management and Delta navigational dredging on sediment loads. Although a lot was learned about watershed sediment supply, all of these previous efforts used statistical and simple regression modeling to estimate loads, based on simplifications of watershed sources, pathways and erosional processes.

A dynamic watershed model is essentially a series of algorithms applied to watershed characteristics and meteorological data to simulate land-based processes including rainfall-runoff, interflow, groundwater flow, flow routing, and suspended sediment loadings over an extended period. A well-calibrated watershed model can be used to characterize loadings from the upland watersheds of the Bay, ensuring that all major watershed sources and pathways are represented. Numerical models have been applied to represent the dynamic erosion and sediment transport processes. The Bay Area Hydrologic Model (BAHM) was originally developed by AQUA TERRA Consultants for the Brake Pad Partnership (BPP) to estimate the copper from brake pad wear debris released to the Bay (Donigian et al., 2007). The model is built on HSPF, a continuous simulation model capable of estimating flow and pollutant loads for mixed land-use watersheds. The watershed models developed for Reasonable Assurance Analysis (RAA) also provided suspended sediment load estimation for San Mateo and Santa Clara County (SCVURPPP 2020; SMCWPPP 2020). While the existing modeling efforts have been applied to different local regions around the Bay, the total suspended sediment load from local small tributaries has not been modeled with a dynamic model. Phase 1 of the WDM provides a consistent modeling framework for the regional hydrological processes and has been developed to provide an improved understanding of interannual variability, errors and uncertainties, and facilitates more reliable estimates of sediment load from local tributaries. The WDM sediment module (Phase 2) provides the total suspended sediment load (SSL) estimate for the Bay area by considering both the upland and channel processes at an appropriate time step to represent the physical processes (e.g., rainfall detachment, sediment settling). The WDM sediment estimate takes into account the spatial and temporal heterogeneities of the erosion and sediment transport processes at the region, by differentiating landscape features (such as geological characteristics, land use and land covers, soil types, imperviousness, and slopes) as well as continuously simulating the physical processes driven by the weather data. The sediment module can provide the sediment load estimates with separate grain sizes, which benefits the refined management plans for sediment reuse and habitat restoration purposes.

2. Sediment Modeling Approach

The WDM is based on the LSPC model, which is a comprehensive watershed model that can serve as a dynamic hydrologic and water quality tool. It provides a dynamic, continuous simulation of hydrology, sediment, and water-quality processes. Since it is a watershed model, sediment transport from the Bay to the channels by tides and waves (a process known to occur in the tidal portions of our flood control channels: Schoellhamer et al., 2018) cannot be accounted for.

The model includes these major modules:

PERLND/IMPLND: modules of hydrologic processes on pervious/impervious land
SEDMNT/SOLIDS: modules of sediment production and removal from pervious/impervious land
PQUAL/IQUAL: modules of pollutant production/removal from pervious/impervious land
RCHRES: module of in-stream flow and water quality processes
SEDTRN: module of sediment transport, deposition, and scour in streams
GQUAL: module of pollutant fate and transport in streams

All these modules include various submodules and options for both simplified and complex process representations. The WDM is capable of simulating sediment erosion from the land surface in PERLND and IMPLND modules and the deposition and scour of sediment in river reaches. The WDM simulates sediment dynamics with two processes: 1) overland processes of soil erosion and transport to streams and channels; and 2) in-stream processes of deposition, scour, and transport. The equations used to produce and remove sediment was based on a sediment model developed by Negev (1967) and a Universal Soil Loss Equation (USLE)-like soil rainfall detachment method and management factors are used for the soil erosion simulation on pervious land surface (Bicknell et al., 2001). A buildup-wash off-based method is used for solids accumulation and delivery on impervious land surfaces. A transport capacity and critical shear stress threshold-based method is used for sediment transport within channels.

2.1 Upland Erosion

2.1.1 Soil Erosion from Pervious Land

Sediment yield to streams was simulated in the WDM in two stages (Figure 1). First, the detachment rate of sediment by rainfall is calculated within the WDM using the equation:

$$DET = (1\text{-}COVER) \times SMPF \times KRER \times P^{JRER}$$

where DET is the detachment rate (tons/acre), COVER is the dimensionless factor accounting for the effects of land cover on the detachment of soil particles, SMPF is the dimensionless management practice factor, KRER is the coefficient in the soil detachment equation, JRER is the exponent in the soil detachment equation, and P is precipitation depth in inches over the simulation time interval. KRER and JRER are two main calibration parameters to adjust the erosion potential across different vegetation and soil types, land surface roughness, and geologic types, among other factors.

Second, the detached soil particles are transported to streams by overland flow. The detached soil particles are stored in detached sediment storage. Actual detached sediment storage available for transport (DETS) is a function of the reincorporation rate (AFFIX) and accumulation/removal over time (NVSI). Transport capacity is the upper limit of sediment that the overland flow can carry. The transport capacity for detached sediment from the land surface (STCAP) is represented as a function of overland flow:

$$STCAP = KSER \times ((SURS + SURO)/DELT60)^{JSER} ,$$

where KSER is the coefficient for transport of detached sediment, SURS is surface water storage (inches), SURO is surface outflow of water (in/hr), and JSER is the exponent for transport of detached sediment. DELT60 is time step in hours.

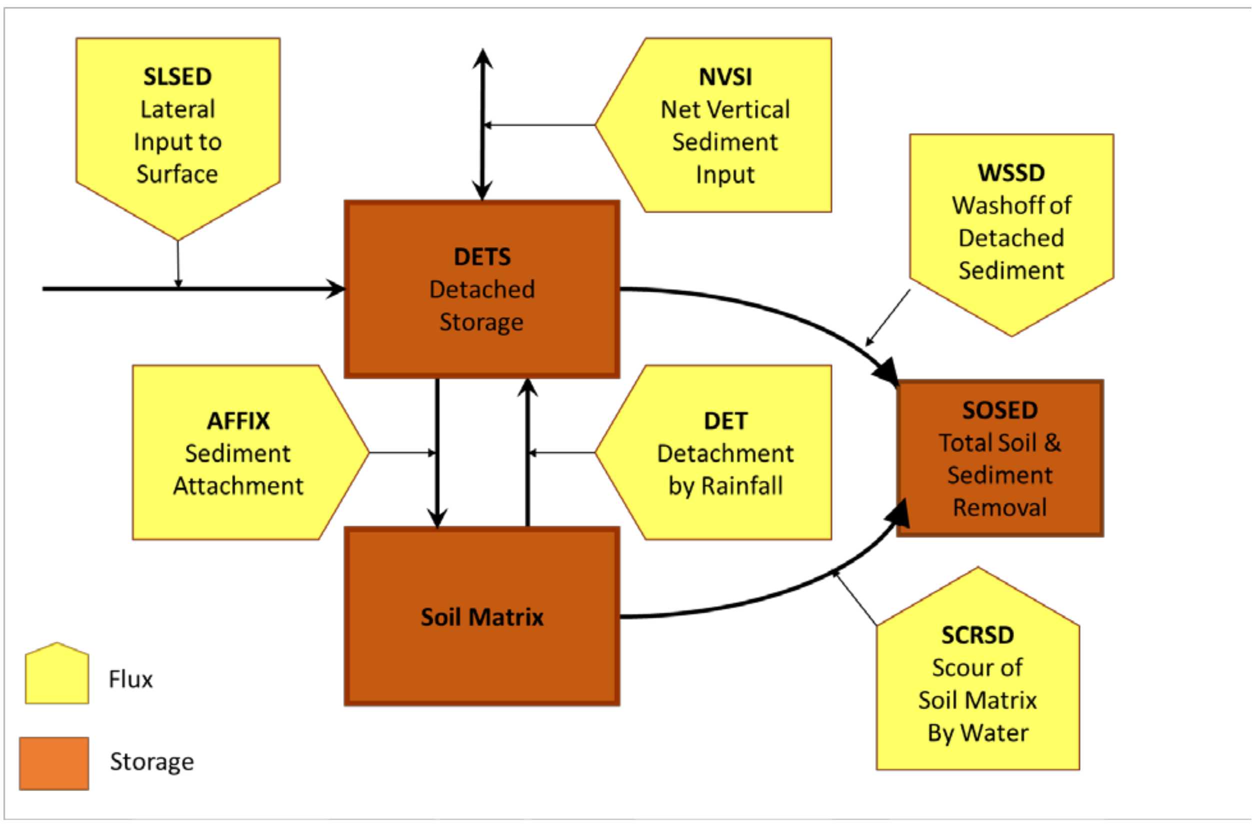

Figure 1. Upland soil erosion scheme of the Watershed Dynamic Model (WDM).

Figure 1 shows the upland sediment scheme of the WDM model. The soil erosion processes on pervious and impervious land are simulated with different modules. Soil detachment from pervious surfaces is simulated using a method like the Universal Soil Loss Equation (USLE). The soil erosion and detachment from impervious surfaces is simulated using a buildup-wash off method, described below.

2.1.2 Sediment Simulation from Impervious Land

Sediment yield from impervious land surfaces is simulated with a buildup and wash-off method. The detached sediment accumulates in the surface storage (SLDS) on days when precipitation did not occur, using the equation:

$$SLDS = ACCSDP + SLDSS(1.0 - REMSDP),$$

where ACCSDP is accumulation rate of the solids storage, SLDSS is solids in storage at start of day, REMSDP is unit removal rate of solids in storage and removal of solids by runoff and other means from the impervious land segment. The solids outflow may be used to simulate quality constituents associated with particulates.

Wash-off of solids is limited by transport capacity of overland flow. The transport capacity is calculated by the equation:

$$STCAP = DELT60 \times KEIM \times ((SURS + SURO)/DELT60)^{JEIM},$$

where STCAP is capacity for removing solids, KEIM and JEIM are a coefficient and exponent for transport of solids, SURS is surface water storage, and SURO is surface outflow of water. The transport capacity is estimated and compared to the amount of solids available. When STCAP is greater than the amount of solids in storage, wash-off is calculated by:

$$SOSLD = SLDS \times SURO / (SURS + SURO),$$

If the storage is sufficient to fulfill the transport capacity, then the following relationship is used:

$$SOSLD = STCAP \times SURO / (SURS + SURO),$$

where SOSLD is wash-off of solids and SLDS is solids storage.

2.2 In-Stream Processes

Once the sediment loading simulation results from upland areas are provided to the stream channels, the sediment simulation then focuses on the channel processes of deposition, scour, and transport that determine both the total sediment load and the outflow sediment concentrations. The sediment loads from upland are divided into appropriate size fractions. Based on the size of particles, the sediment is grouped into two main categories, non-cohesive sediment and cohesive sediment. Cohesive sediment behaves differently than non-cohesive sediment as suspended cohesive particles tend to bond together and form flocs or aggregates. In this study, silt and clay (<63 um) were considered as cohesive sediment. The size threshold between cohesive and non-cohesive particles can be different from the size thresholds of silt and clay. More sediment size classes can be represented in WDM model to better separate cohesive and non-cohesive sediments with the support of sediment particle size distribution monitoring data. The transport of the sand (non-cohesive) fraction is calculated as a power function of the average velocity in the channel reach for each time step (Bicknell et al., 2001). This transport capacity is compared to the available inflow and storage of sand and larger particles; the bed is scoured if there is excess capacity to be satisfied, and sand is deposited if the transport capacity is less than the available sand in suspension within the channel reach. For the silt and clay (cohesive) fractions, shear stresses from different reach segments are determined as a function of the slope and hydraulic radius of the reach, as follows:

$$TAU = SLOPE \times GAM \times HRAD$$

where TAU is stream bed shear stress (lb/ft^2), SLOPE is the slope of the channel segment (-), GAM is unit weight, or density, of water (62.4 lb/ft^3), and HRAD is hydraulic radius (ft or m).

The hydraulic radius is calculated as a function of average water depth (AVDEP) and mean top width (TWID):

$$HRAD = (AVDEP \times TWID) / (2 \times AVDEP + TWID)$$

This equation is essentially an estimated cross-sectional area divided by an estimated wetted perimeter. The hydraulic radius and shear stress equations used in WDM assumes a rectangle cross-section shape and the shape of channel is uniform within each subwatershed. The simple representation of channel geometry in WDM could be a potential source of errors in simulating channel processes.

The shear stresses from channel segments are then compared to user-defined critical values for deposition and scour for silt and clay. If the shear stress is greater than the critical value for scour, the bed is scoured. If the shear stress is less than the critical deposition value, the silt or clay fraction deposits. If the shear stress falls between the critical scour and deposition values, the incoming suspended sediment is transported through the reach (Figure 2).

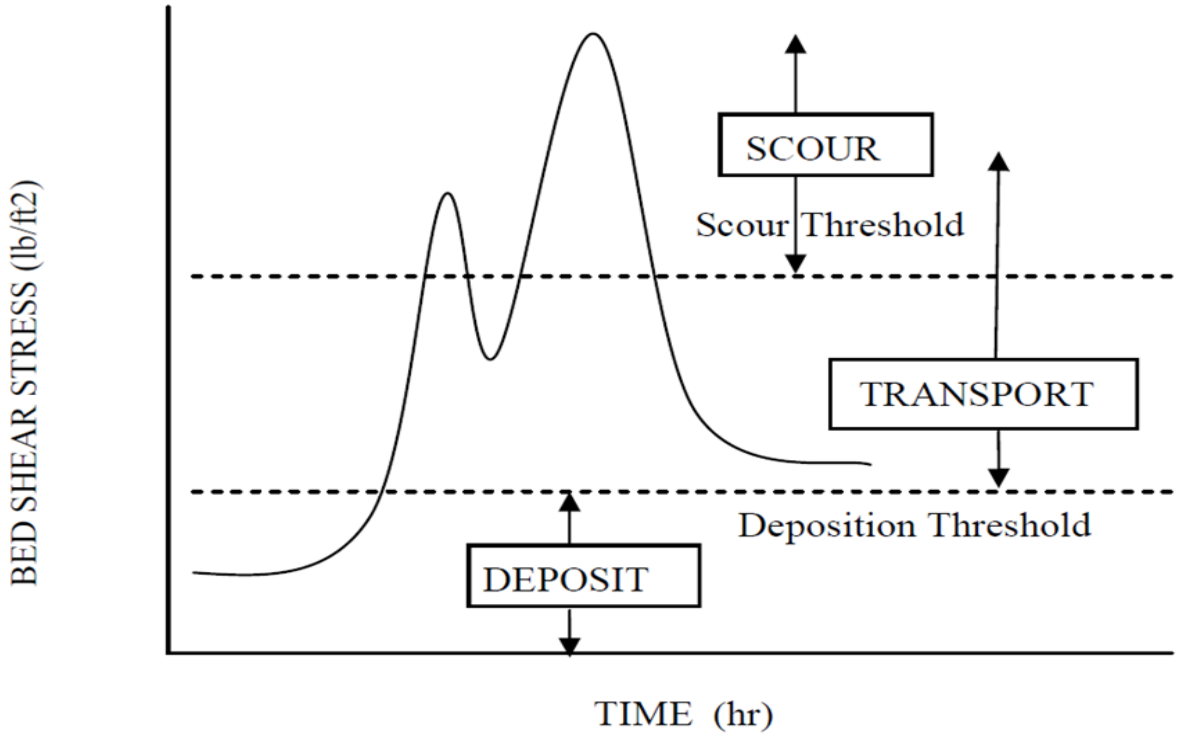

Figure 2. The illustration of sediment transport/settling/scour processes based on critical shear stress thresholds in the Watershed Dynamic Model. (Adopted from EPA BASINS Tech Note 8: USEPA, 2000).

Following these calculations, the depth of sediment on the bed is determined. The WDM uses a single sediment bed layer. Initial bed composition in each reach segment is based on available field data or best professional judgment. The simulation continuously updates the bed composition in each reach based on relative amounts of scour or deposition of the three defined size classes (sand, silt, and clay). In general, both upland and instream processes are simulated in the WDM to capture the hydrology and sediment dynamics at the subwatershed scale.

3. WDM Sediment Model Setup and Calibration

The WDM sediment model consists of two major parts, upland process representation and in-stream process representation. The Hydrologic Response Unit (HRU) is the basic modeling land unit of the WDM. Each subwatershed (SWS) consists of a combination of different HRUs. The rainfall-runoff, erosion, and sediment transport to streams are simulated at the HRU level. Each HRU type represents a land type with similar physical characteristics, such as land cover, land use, soil type, slope, and imperviousness. There are currently 55 HRU types in this model. The hydrologic, erosion, and sediment transport processes are considered similar for land units with the same HRU type. The HRU responses are then aggregated at the subwatershed scale. Each subwatershed in the WDM is associated with a major stream/channel segment within the subwatershed. In-stream sediment transport processes (settling, resuspension, and transport) are simulated at each stream segment. Both upland and in-stream processes were simulated at hourly time step in this study.

To accomplish the ultimate goal of stormwater contaminant load estimation to the Bay, the sediment module needs to estimate the total suspended sediment load from local tributaries to the Bay. Three characteristics of sediment related variables must be examined successively: 1) flow rates and volumes of local tributaries; 2) suspended sediment load (SSL) at calibration watersheds; and 3) suspended sediment concentration (SSC) at calibration watersheds. At each calibration station, simulated and observed flows, SSL, and SSC are examined and key hydrologic and sediment related parameters are adjusted to attain acceptable model performance. Calibration for sediment modeling follows the sequence of hydrology, then sediment. Parameters are adjusted iteratively to improve the SSL estimation and the relationships between SSC and flow rate. Model parameter adjustment follows the guidance and ranges in BASINS Technical Note 8 (USEPA, 2000). The WDM was calibrated and validated for a longer period (1995-2020) than originally planned by the RMP with aid from the SCC/BCDC sand budget project (SFEI-ASC, 2022).

3.1 Updates from Phase 1

Phase 2 of the WDM includes some modifications to Phase 1 following the comments and suggestions received from the SPLWG scientific advisors and RMP stakeholders. This section highlights major changes incorporated into the Phase 2 model.

3.1.1 Channel geometry and watershed delineation

The sediment transport process can be greatly affected by channel hydraulics. The in-stream hydraulic simulation of the WDM relies on the channel characteristics, especially channel geometry. Channel depths and widths in the Phase 1 model were estimated using the empirical relationships developed by Leopold and Maddock (1953). There are several local and more recent studies available to derive the relationships between drainage area and channel geometries, such as Marin-Sonoma regional curves (Collins and Leventhal, 2013). Table 2 shows the published local relationships between drainage area and the channel geomorphology. USGS rating curves and field survey data, when available, were used to define the geometry of channels. For channels without rating curves available, the depths and widths were re-derived using the relationships from corresponding local studies and then put into the WDM. The details of the channel representations in WDM can be found in Phase 1 report (Zi. et al., 2021). These relationships are for natural streams and may not be applicable to urban streams. The cross-section areas from surveys if available were used for urban streams and different roughness parameter were applied to concrete channels.

Table 2. Summary of local channel geometry studies.

	Drainage Area (sq mi)	**Width (ft)**	**Depth (ft)**	**Area (sq ft)**	**Reference**
South Bay	0.1	6.16	0.35	1.73	Hecht et al., 2013
	100	45.7	3.64	233	
Bay Area for 30” of rainfall	0.1	7	0.8	5.5	Dunne and Leopold, 1978
	100	80	5	450	
East Bay	0.1			4.2	Riley, 2003
	100			310	
Marin-Sonoma	0.1	4.41	0.44	1.95	Collins and Leventhal, 2013
	100	110	5.57	617	

Watershed segmentation refers to the subdivision of the entire modeling extent into smaller, discrete subwatersheds and reaches for modeling and analysis. The watershed delineation of the Phase 1 WDM was based primarily on existing hydrologic boundaries (Federal standards and procedures for the National Watershed Boundary Dataset (WBD), 2013), watersheds delineated with stormwater network information (BAARI stream and stormwater network, watersheds delineated for RWSM model (SFEI, 2018)), watershed layers from local sources (e.g., Valley Water), and elevation models, including LiDAR and topo-bathy derivations. Other sources, such as the stormwater pipe network from Oakland Library and the Zone 7 water agency were used in the Phase 2 model to refine the watershed delineation in urban regions. Three more subwatersheds were delineated, increasing the total number of subwatershed to 240. The final watershed segmentations are shown in Figure 3.

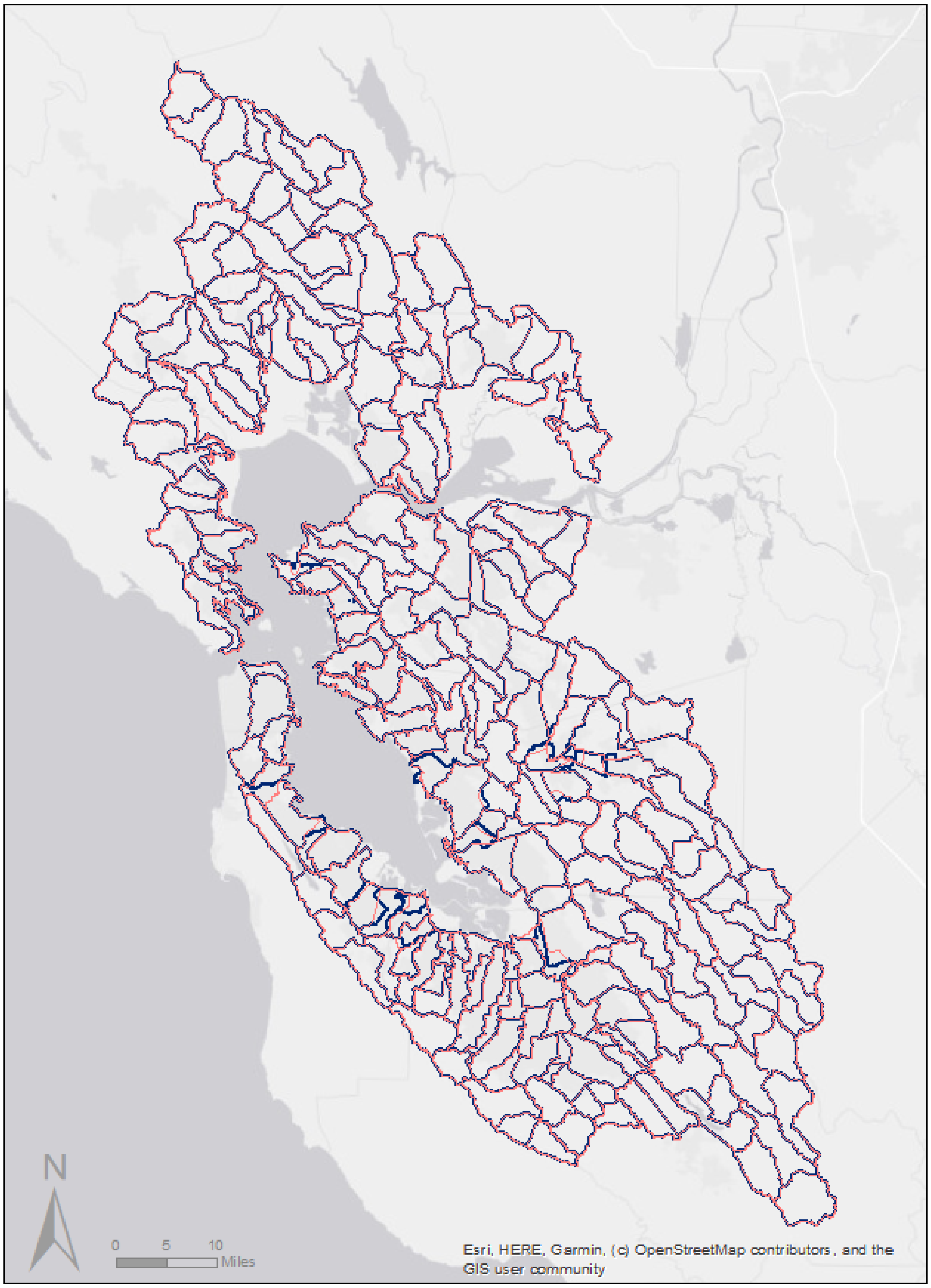

Figure 3. Watershed delineation of Phase 2 Watershed Dynamic Model; the dark blue lines identify the changes in watershed delineation boundaries compared to Phase 1.

3.1.2 Land use, imperviousness, and Hydrologic Response Units

The hydrologic response unit (HRU) is the basic modeling land unit, which is a generalized representation of land space with a common and defined combination of land use/cover, soil type, slope, imperviousness and other characteristics. Each defined combination of land features represents specific physical/chemical processes (e.g., rainfall-runoff, soil detachment and transport, pollutant accumulation) across subbasins. Each subbasin has its own combination of HRUs and summarizes the processes on each HRUs at subbasin scale. The HRU approach allows incorporation of a high degree of detail into the model while also allowing for efficient simulation and relatively short model run times. The HRUs developed in the Phase 1 model account for land use/cover, soil type, imperviousness, slope, and geology. In Phase 2, the land use and land cover data source was changed to a different period to cover the longer simulation period, directly connected impervious surfaces were recalculated based on a local study, and the HRU were regrouped for the purpose of erosion and sediment transport simulation.

Land use and land cover are important input data for watershed models as they represent key characteristics of a watershed. Two sources of land use data are available for the region around the baseline year—the USGS National Land Cover Database (NLCD, Vogelmann et al., 2001) and ABAG (2005). NLCD includes land use and percent imperviousness surface in 30-meter resolution and is updated every five years. In contrast, the ABAG land use layer was developed in 2005 (with data from 2001-2003). The latest ABAG land-use data layer (data gathered around 2018-2021) is still under development. The Phase 3 of WDM is expected to utilize the latest land-use data layer from ABAG, once available, for stormwater contaminants modeling. The Phase 2 WDM extended the simulation period from 1995 to 2020. To better represent the land use and land cover for the simulation period, a hybrid approach was used to develop the land use and land-cover layer—ABAG 2005 (latest available regional urban land use data) for urban land using reclassification and grouping and NLCD 2011 for open and rural area land cover and imperviousness. Finer natural area classification was applied to the HRU grouping. Forest land cover in Phase 1 was reclassified into three categories: evergreen forest, mixed forest, and shrubland.

As a highly urbanized region, impervious surfaces are a major contributor to stormwater runoff and contaminant loads. Not all impervious surfaces are directly connected to streams. Effective impervious area (EIA), defined as the subset of the total impervious area (TIA) often hydrologically connected to stream networks via stormwater infrastructure, is required to better model the runoff volume and timing in the patches with a high percentage of impervious surface (e.g., urban areas). The estimated Directly Connected Impervious Area (DCIA) is usually considered a surrogate for EIA. In the Phase 1 model, Sutherland’s method (1995) was used to estimate DCIA. TIA estimation method derived from a local study (URS 2006) was applied to the DCIA estimation to adjust the area of DCIA from Sutherland’s method.

For $0\% <= TIA_{NLCD} <= 10\%$,
$TIA = 1.0971 \times TIA_{NLCD} + 1.5494$, and
For $TIA_{NLCD} > 10\%$,
$TIA = 1.0428 \times TIA_{NLCD} + 9.5745$.
Where TIA_{NLCD} is the NLCD percent imperviousness estimate, and TIA is the adjusted percent imperviousness.

Using the localized TIA estimation method, an additional 4940 acres of DCIA area was added to the Phase 2 model, a nearly 10% increase of the total DCIA area from the Phase 1 model. The DCIA percentage is 3.4% for the whole modeling domain, 9.4% for the watersheds downstream of reservoirs. The updates in land use classification and impervious areas for Phase 2 model HRUs are summarized in Table 3.

Table 3. The Hydrologic Response Unit (HRU) classification of Phase 2 Watershed Dynamic Model.

HRU ID	Land Use Land Cover	Hydrologic Soil Group	Slope	Impervious (DCIA)	Area %
1	Agriculture	A/B	Low	N	0.83%
2	Agriculture	C	Low	N	1.17%
3	Agriculture	D	Low	N	0.67%
4	Evergreen Forest	A/B	High	N	1.83%
5	Evergreen Forest	C	High	N	2.15%
6	Evergreen Forest	D	Moderate	N	0.79%
7	Evergreen Forest	D	High	N	2.44%
8	Grass and Barren	A/B	Low	N	1.26%
9	Grass and Barren	C	Low	N	1.56%
10	Grass and Barren	C	Moderate	N	2.95%
11	Grass and Barren	C	High	N	2.24%
12	Grass and Barren	D	Low	N	2.99%
13	Grass and Barren	D	Moderate	N	3.72%
14	Grass and Barren	D	High	N	2.54%
15	Mixed Forest	A/B	High	N	1.25%
16	Mixed Forest	C	Moderate	N	2.39%
17	Mixed Forest	C	High	N	7.26%

18	Mixed Forest	D	Moderate	N	1.82%
19	Mixed Forest	D	High	N	5.41%
20	Shrubland	A/B	Moderate	N	0.73%
21	Shrubland	C	Moderate	N	6.40%
22	Shrubland	C	High	N	0.41%
23	Shrubland	D	Low	N	0.64%
24	Shrubland	D	Moderate	N	3.02%
25	Shrubland	D	High	N	5.06%
26	Water	-	-	N	1.13%
27	Wetland	-	-	N	0.37%
28	Mixed Urban	C/D	Low	N	2.43%
29	Mixed Urban	C/D	Moderate	N	1.29%
30	Mixed Urban	C/D	High	N	0.72%
31	New Commercial	C/D	Low	N	1.18%
32	New Industrial	C/D	Low	N	1.43%
33	New Residential	C/D	Low	N	2.30%
34	New Residential	C/D	Moderate	N	1.26%
35	New Residential	C/D	High	N	0.53%
36	Old Commercial	C/D	Low	N	3.01%
37	Old Industrial	C/D	Low	N	2.45%
38	Old Residential	C/D	Low	N	7.28%
39	Old Residential	C/D	Moderate	N	2.12%
40	Old Residential	C/D	High	N	0.88%
41	Other Roads	C/D	Low	N	1.94%
42	Other Roads	C/D	Moderate	N	0.77%
43	Other Transportation	C/D	Low	N	2.91%
44	Other Transportation	C/D	Moderate	N	1.06%
45	Primary and Secondary Roads	-	Low	Y	0.46%
46	Mixed Urban	-	Low	Y	0.28%

47	New Commercial	-	Low	Y	0.02%
48	New Industrial	-	Low	Y	0.02%
49	New Residential	-	Low	Y	0.28%
50	Old Commercial	-	Low	Y	0.07%
51	Old Industrial	-	Low	Y	0.05%
52	Old Residential	-	Low	Y	1.22%
53	Other Roads	-	Low	Y	0.37%
54	Other Transportation	-	Low	Y	0.61%
55	Other Impervious Surface	-	Low	Y	0.00%

Other factors such as microclimate, geologic, and tectonic features could impact rainfall-runoff, sediment accumulation and transport processes. Four parameters groups were set to account for the geological characteristics (Zi. et al., 2021) at subwatershed scale. To account for the heterogeneity of features that are not covered by HRUs, the Phase 2 model divided the Bay area into seven subregions based on the geological map, watershed delineation, and embayment (Figure 4) to better capture the spatial variation of physical processes across the Bay area. The boundaries of subregions were mainly followed by the boundaries of watersheds of streams and counties and each subregion has a comparable size. The setting of subregions was used to inform the extension of calibrated parameters to uncalibrated subwatersheds and to avoid overstretch of HRU parameters for a large region. Based on the monitoring data analysis, to avoid over-extrapolation of calibrated parameters, the North Bay subregions have similar erosion potential, the subregion of Contra Costa County and northern Alameda County has the largest erosion potential, the South Bay and San Mateo County subregions have similar and relatively lower erosion potential. These general patterns were applied to guide the model calibration processes.

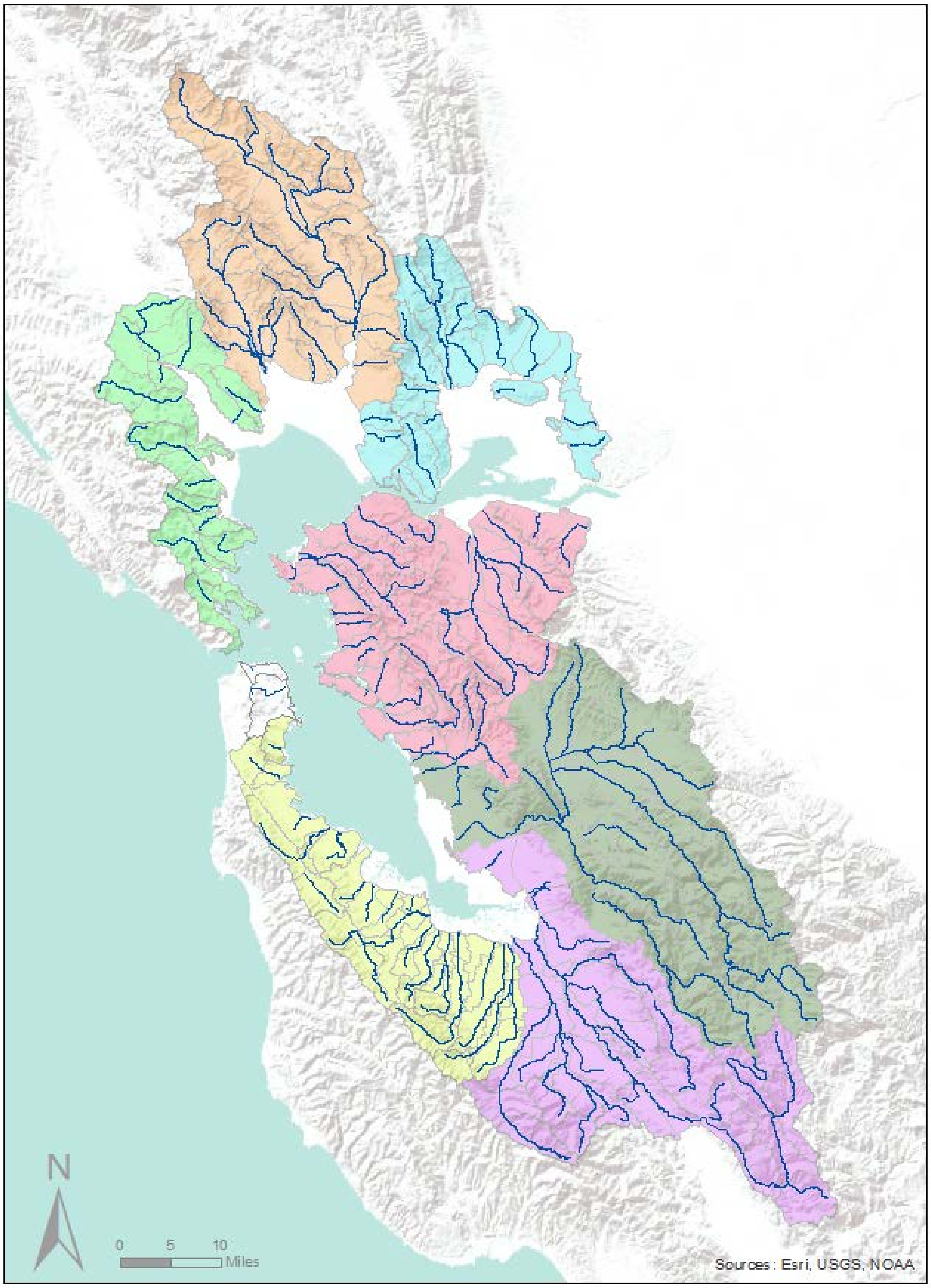

Figure 4. Seven subregions in Phase 2 Watershed Dynamic Model (Watersheds in San Francisco were not included due to the combined stormwater sewer system).

3.1.3 Hydrologic calibration

The Phase 1 model hydrologic calibration was conducted against the daily discharge records from 13 USGS stream gauges from 2000 to 2006. The Phase 2 model extended the simulation period from 1995 to 2020 and made previously mentioned changes in the land use, land cover, and imperviousness data to generate HRUs. WDM was recalibrated to achieve a performance that was about the same or better than the previous calibration. Three statistical indices were used to evaluate model performance. Percent of bias (PBIAS) was selected to evaluate the water budget simulation. The ratio between RMSE and standard deviation (RSR) standardizes the RMSE using the standard deviation of monitoring data. RSR is an index that incorporates the error information with a scaling factor, thus the RSR value can be compared to different components (e.g., sites with different flow rates and variances). The optimal value of RSR is 0, which indicates zero RMSE or residual variation and perfect model simulation. The lower the RSR, the lower the RMSE, and the better the model simulation performance. The Nash-Sutcliffe efficiency (NSE) is a widely used and reliable statistic for assessing the goodness of fit of hydrology calibration (Nash and Sutcliffe, 1970). NSE is a normalized statistic that determines the relative magnitude of the residual variance compared to the measured data variance and reflects the overall fit of a hydrograph. Values vary from $-\infty$ to 1.0. A value of NSE = 1.0 indicates a perfect fit between modeled and observed data, whereas values less than 0 indicate that model predictions of temporal variability in observed flows are worse than using the average of observed data. The same criteria as the Phase 1 model (Zi et al., 2021) was used to rank the model performance into ‘Very Good’, ‘Good’, ‘Fair’, and ‘Poor’ classes. The hydrologic simulation performance of the WDM is summarized in Table 4.

The model captures the flow dynamics very well. The values of NSE, RSR, and total PBIAS are at a ‘Very Good’ level for almost all calibrated subwatersheds. Only three subwatersheds have a PBIAS slightly larger than the ‘Very Good’ criteria (11% vs 10%). All subwatersheds have NSE and RSR values that meet or exceed ‘Good’ criteria, except for the calibration site Guadalupe, whose NSE is just shy of ‘Good’ criteria (0.64 vs 0.65). In general, the model has very good performance on flow volumes and peaks (timing and quantity) at the regional scale. The solid model performance in estimating flows sets up a good foundation for estimating sediment load.

Table 4. The summary of WDM hydrological simulation performance.

Gauge ID	Gauge Name	Calibration Period	NSE	RSR	PBIAS	High flow PBIAS	Wet Season PBIAS
11164500	SAN FRANCISQUITO C A STANFORD UNIVERSITY CA	1995-2020	0.89	0.33	9%	2%	11%
11166000	MATADERO C A PALO ALTO CA	1995-2017	0.91	0.3	-6%	1%	-2%
11169500	SARATOGA C A SARATOGA CA	1995-2020	0.72	0.53	-11%	1%	-6%

Gauge ID	Gauge Name	Calibration Period	NSE	RSR	PBIAS	High flow PBIAS	Wet Season PBIAS
11172175	COYOTE C AB HWY 237 A MILPITAS CA	1999-2020	0.87	0.36	6%	-11%	5%
11169025	GUADALUPE R ABV HWY 101 A SAN JOSE CA	2002-2020	0.64	0.6	-3%	-14%	-13%
11179000	ALAMEDA CREEK NILES	1995-2020	0.78	0.47	-11%	-25%	-13%
11180825	SAN LORENZO C AB DON CASTRO RES NR CASTRO V CA	1997-2020	0.96	0.19	-7%	5%	14%
11180960	CULL C AB CULL C RES NR CASTRO VALLEY CA	1995-2020	0.88	0.34	4%	-1%	12%
11181040	SAN LORENZO C A SAN LORENZO CA	1995-2020	0.85	0.38	8%	6%	-2%
11182500	SAN RAMON C A SAN RAMON CA	1999-2006	0.77	0.48	-11%	-10%	-5%
11456000	NAPA R NR ST HELENA CA	1995-2020	0.95	0.23	6%	-8%	2%
11458000	NAPA R NR NAPA CA	1995-2020	0.89	0.33	-5%	-11%	4%
11458500	SONOMA C A AGUA CALIENTE CA	1995-2020	0.94	0.25	1%	-10%	4%

* High90 PBIAS is the relative error of high flows (above 90th percentile), Wet Season PBIAS is the relative error of wet season flows.

** Color code: Green -> Very Good, Yellow -> Good, Pink -> Fair.

3.2 Model Setup and Assumptions

The WDM represents the erosion and sediment transport process at two spatial scales: 1) soil detachment, transport, erosion and settling at the HRU level of subwatersheds; 2) detached soil particles from HRUs that reach the stream are then subjected to the hydraulic forces within the stream segment for transport, resuspension, and settling. The goal of the WDM sediment model is to have a reasonable estimation of sediment load from local watersheds into the Bay for given hydrological conditions and to represent the overall sediment behavior of the watershed that is consistent with conceptual models and observed concentration and loading data. The suspended sediment supply is a combination of model estimated upland erosion and suspended sediment transport within the channels.

In the WDM, 240 subwatersheds consist of numerous HRUs—the basic modeling unit for upland processes—and reflect the spatial heterogeneity of the upland erosion processes. We considered a range of geological characteristics, soil types, land use and land covers, and slopes to determine hydrological, erosion, and sediment transport processes at the HRU level (i.e., each HRU had its own representations of hydrologic and sediment transport processes). The upland erosion is rainfall driven and is estimated with the USLE-based method with consideration of sediment delivery ratio (SDR, described in Section 3.3.2). Gully erosion estimation was applied to HRUs with agricultural, grass, and barren land covers. Sediment was classified into three different classes based on the particle size distribution. The particle size range of sand is from 63 um to 2 mm; silt from 4 um-63 um and clay less than 4 um. The portions of different sediment classes (sand, silt, clay) from natural and rural areas were derived from the soil survey data and SDR estimation. In urban areas, the street sweeping sediment sampling data were used to estimate the portions of different sediment classes (BASMAA, 2017).

Each subwatershed within the WDM is assigned only one stream segment. However, there are channels that exist that are not defined in the model. For these channels that are not included in the stream network, the drainage area is retained and runoff from the drainage area is routed to the stream segment assigned to that subwatershed. Length and slope data for natural reaches are estimated using the USGS National Elevation Dataset DEMs (10 m) and digitized reach lengths. Four different parameter sets were used to distinguish the different types of reach segments (natural fluvial, unnatural fluvial, natural tidal, unnatural tidal). Function tables (discharge-area-depth table) were built into the WDM for the channels with USGS rating curves and channels with related information collected from water agencies (as shown in Figure 5).

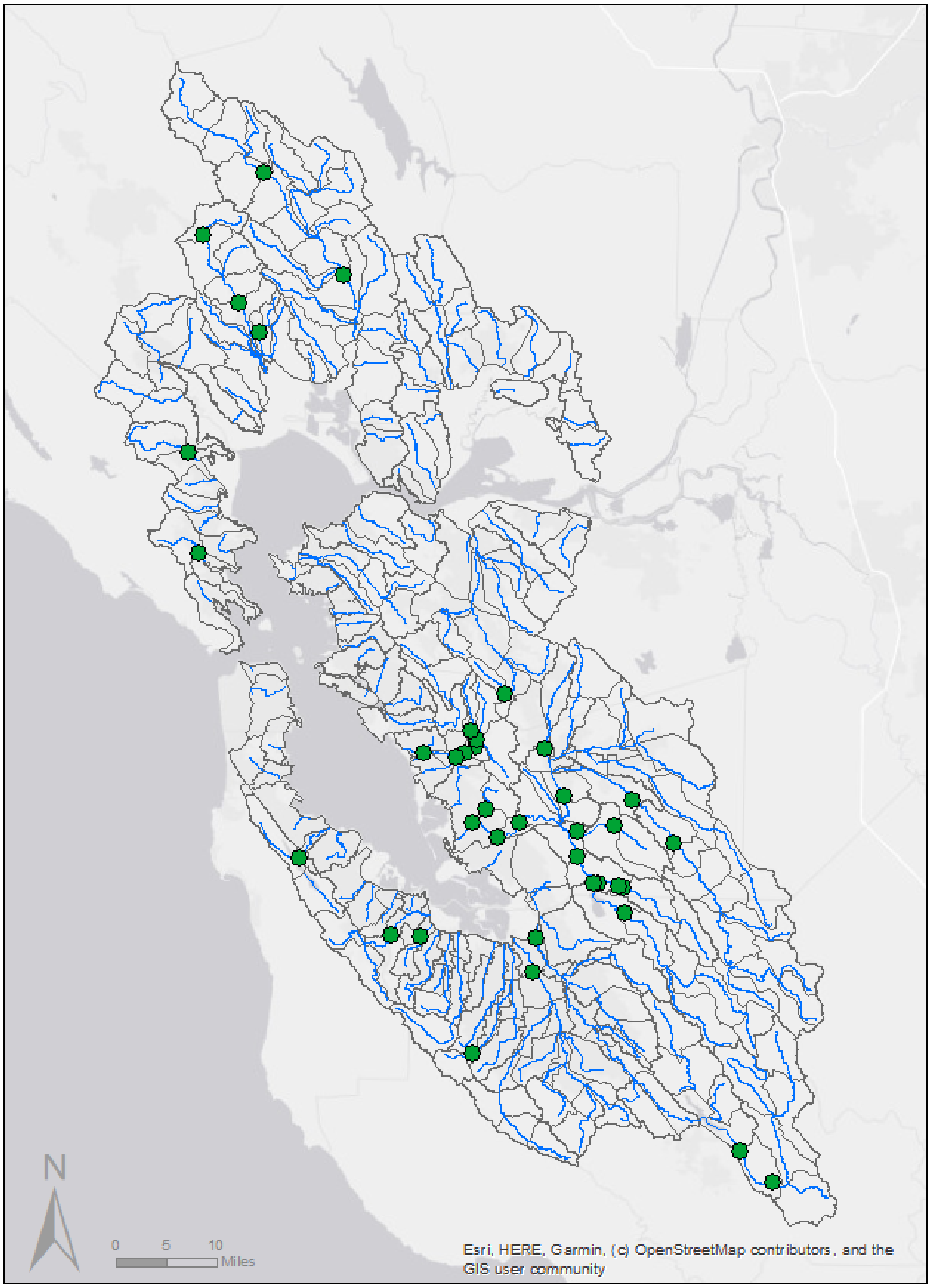

Figure 5. Channels with USGS rating curves available illustrated by the green dots.

The sediment size class distribution of each stream segment was initialized by aggregating sediment size distribution and the sediment delivery ratio of its drainage area. Bank erosion was approximated for the highly urbanized drainage systems by adjusting the scouring processes within streams. The sediment transport from the Bay to the channels by tides and waves was not accounted for in the WDM. The change of land use and channel geometries over the simulation period was not represented in the Phase 2 sediment modeling. The impacts on channel hydraulics from the changes on stream/channel cross-sections due to sedimentation, dredging, and scouring were not assessed and represented during the modeling period. Other sediment sources, including landslide, reservoir fill and release, and post-fire debris flows were not represented in the model. The modeling uncertainties caused by these missing pieces and simplifications are discussed in Section 4.

The sediment model calibration process follows a two-step approach. First, the sediment yields from different HRUs were estimated (Section 3.3). This step focuses on adjusting erosion-related parameters of each HRU. This calibration step is aimed at representing reasonable trends and rankings of sediment yields from different HRUs and establishing relative sediment load levels across all sources of sediment to develop summarized model performance at the subwatershed scale. The modeled Edge-of-Stream sediment yield were based on the soil erodibility from different land uses and land covers using the USLE-like method (Section 3.3.1) and taking account to the spatially explicit Sediment Delivery Ratio (SDR, Section 3.3.2). The spatially explicit SDR was summarized by the HRUs and the rank of the SDR values for different HRUs was used to adjust the HRU sediment yield. The products of HRU specific erodibility and SDR were then calibrated to have a reasonable range of Edge-of-Stream sediment yield data comparing to literature values. The erodibility values were adjusted for four parameter groups based on geological characteristics. The sediment yield values were also adjusted for different subregions given the patterns we found from monitoring data. The second step is to calibrate the model against the in-stream suspended sediment load monitoring data (Section 3.4). This step focuses on adjusting erosion and sediment transport parameters for both HRUs and channel segments and is aimed at having a good suspended sediment load estimation at calibrated watersheds. The Edge-of-Stream sediment yield from each HRU was summarized by subwatershed and then was converted to three different sediment size groups and transported into the associated reach segment. The particle size distributions of sediment from different HRUs were derived from soil survey data and street dirt sampling data. The spatially explicit SDR for three sediment size groups were used to estimate the PSD for the sediment reach the Edge-of-Stream at each subwatershed. The sediment size distribution of bed sediment were also derived based on the input from each subwatershed. The instream processes were calibrated based on the SSL estimate at all calibration sites, PSD of the suspended sediment at calibration sites, and the bed sediment dynamics.

3.3 Edge-of-Stream Sediment Yield Estimation

3.3.1 Soil erodibility

Edge-of-stream sediment yield is based on the estimation of two processes, the soil detachment at field level and the sediment delivery ratio. The WDM applied a USLE-based method to estimate the soil detachment from pervious surfaces by rainfall.

$$D = K \times C \times S \times P^{J}$$

where D is soil detachment rate, K is soil erodibility factor, C is land cover factor, S is land management factor, P is rainfall intensity and J is the exponent for rainfall detachment. The USLE equation is derived from experiments at field scale. The slope length factor in the USLE equation is applied to adjust the soil detachment rate of fields with different slopes and scales. The WDM method does not include the slope length factor, instead the soil erodibility is retrieved from SSURGO soil survey data at each grid cell (30 m × 30 m). The SSURGO soil layer contains the K-factor attribute, which is a measure of soil erodibility. The K-factor was mapped on each pixel of the HRU raster (Figure 6a).

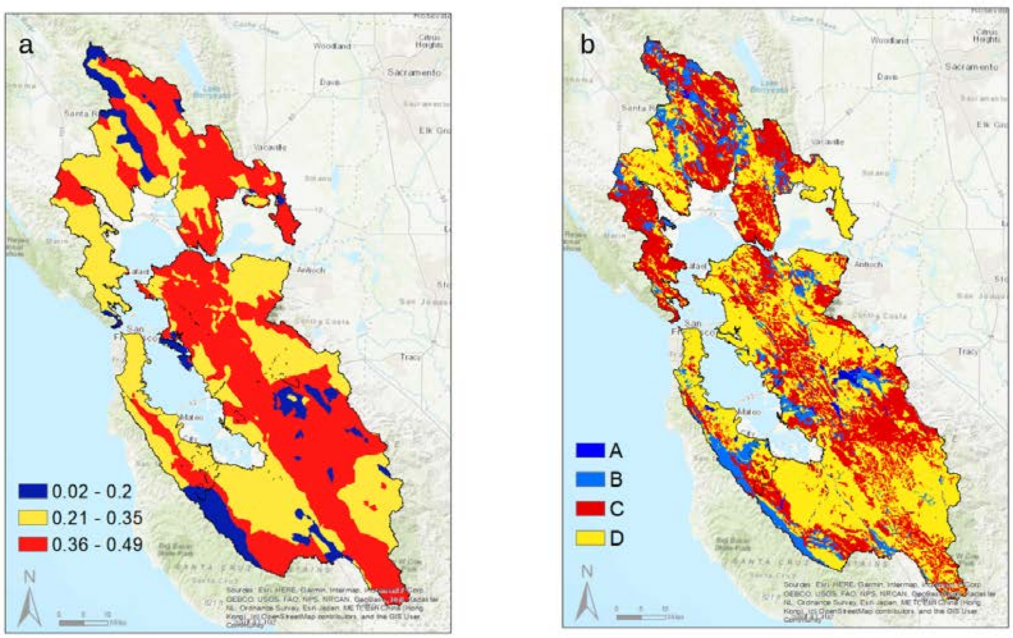

Figure 6. (a) Soil Erodibility K-Factor (Source: USDA SSURGO), (b) Hydrologic Soil Groups (A,B,C,D).

Soil erodibility varies with soil texture, hydrologic soil groups, and other factors such as geological characteristics. Soils with high silt content are the most erodible and are associated with relatively high K values. Soils with high clay content or sandy soils tend to have relatively low K values, because either they are resistant to detachment or they are hard to mobilize via runoff. Soil texture also impacts the infiltration rate, and subsequently the rainfall-runoff process, thus the hydrologic soil group classification. The spatial distribution of K values has a good correlation with the distribution of hydrologic soil groups (Figure 6b). Table 5 shows the summary of area-weighted K-factor values for different hydrologic soil groups. C soils were generally more erodible than the other three hydrologic soil groups. This ranking was used to inform parameter adjustment for each HRU.

Table 5. Area-weighted K-factor values for hydrologic soil groups.

HSG	K factor
AB	0.29
C	0.34
D	0.32

Geologic features can have a large impact on hydrologic and sediment accumulation transport processes. Different geologic landscapes of the Bay Area produce different amounts of sediment. There are five main geologic types in the Bay Area: Franciscan Formation, Mesozoic-Tertiary sedimentary, Great Valley, Sonoma Volcanics and Quaternary deposits. The Sonoma Volcanics and Quaternary deposits have the lowest erosion potential and therefore were grouped together. The varied geologic features were represented as four different parameter groups in the WDM. Each geologic type has its own specific parameter group. All HRUs were subject to geologic types and can be calibrated differently to account for the geology controls across the landscapes. In general, the erosion potential of different geologic features from high to low is: Franciscan Formation, Great Valley, Mesozoic-Tertiary sedimentary, and Sonoma Volcanics and Quaternary deposits, with some exceptions of a few geologic subunits. This general trend is used to guide the calibration of KRER and JRER coefficients at the HRU level.

The erosion processes on impervious surfaces were simulated as a buildup-washoff process. Sediment accumulation rates were assigned to different impervious HRUs based on reference values from the literature and previous modeling studies.

3.3.2 Sediment Delivery Ratio

The soil erodibility value represents the maximum soil particles detached from rainfall and the erosion potential of certain HRUs. Not all the detached soil particles reach the concentrated flows and are transported to the main channel segment of each subwatershed. To account for the loss of sediment during the transport process from field to the edge of the channel, sediment delivery ratio (SDR) is applied to HRU parameterization. The goal of a SDR is to identify the overland sediment generated and delivered to target locations. Factors that influence the SDR include sediment source, texture, nearness to the main channel, channel density, basin area, slope, length, land use/land cover, and rainfall-runoff factors (Sharp et al., 2020; Ouyang and Bartholic, 1997). Bicknell et al. (2001) present an equation developed by NRCS (1983) to estimate SDR as a function of watershed area:

$$SDR = 0.417762 \times A^{-0.134958} - 0.127079$$

where A is the upstream area in square miles.

For the purposes of this project, an estimated spatially-variable SDR was created in ArcGIS for clay, silt, and sand, derived from an Index of Connectivity (IC) raster layer.

Sediment connectivity characterizes the degree of linkage that controls sediment fluxes, in particular between sediment sources and downstream target areas. The index of connectivity (IC) is an explicit variable when calculating the SDR as proposed by Vigiak et al. (2012). IC was derived following the approach of Borselli et al. (2008) who defined IC as:

$$IC = log10\ (D_{up}/D_{dn})$$

where D_{up} and D_{dn} are the upslope and downslope components of connectivity, respectively. IC can have any positive/negative value, with connectivity increasing for larger IC values. This was an innovative approach as previous efforts assessed only the upslope component or downslope component individually. This approach was further modified by Cavalli et al. (2013) to better suit mountainous environments and high-resolution digital terrain models.

The upslope component is the potential for downward routing of the sediment that is produced upslope while the downslope component takes into account the flow path length (modified by a weighting factor) that a particle has to travel to arrive at the closest target or sink. The equation for D_{up} is:

$$D_{up} = \bar{W}\bar{S}\sqrt{A}$$

where *W* is the average weighting factor of the upslope contributing area, *S* is the average slope gradient of the upslope contributing area (m/m) and A is the upslope contributing area (m^2). D_{dn} can be expressed as:

$$D_{dn} = \sum_i d_i W_i S_i$$

where d_i is the length of the flow path along the i^{th} cell according to the steepest downslope direction (m), W_i and S_i are the weighting factor and the slope gradient of the i^{th} cell, respectively. To calculate IC as expressed in Cavalli et al. (2013), a free, open source, and stand-alone application known as SedInConnect was used (Crema and Cavalli, 2018). SedInConnect requires the input of a digital terrain model (DTM), target features, and a weighting factor.

To calculate the SDR, a study area was delineated by the Bay Area watersheds fed by the local tributaries. A 30 m buffer around the provided watersheds was used to capture the sediment delivery at the Bay edge. A 10-meter resolution DEM produced through the National Elevation Dataset (USGS) in 2011 was used as input for this tool. The 10-meter resolution DEM also matched the resolution of the land cover dataset used to derive the C-Factor, discussed later. The Pit Remove tool from the Tau-DEM toolbox (Tarboton, 2005) was used to create a filled DTM without sinks in order to process SedInConnect.

The IC outputs are dictated by the flow path distance to input targets. For the purposes of our analysis, targets were defined as Roads, Streams, Reservoirs, and the Bay Edge. Roads originated from the United States Census TIGER dataset and were refined to only primary roads, secondary roads, local roads/streets, ramps, and service drives (feature classification codes: S1100, S1200, S1400, S1630, S1640). The selected road polylines were buffered 5 meters to form their respective polygonal shape. Streams were derived from the 10-meter filled DTM by process of a D8 flow direction, and setting an accumulation threshold of 20,000 cells. The resulting stream raster was converted to a polyline and buffered 5 meters using a dissolved output. Reservoirs were from the Bay Area Aquatic Resources Inventory (BAARI) dataset.

The weighting factor is related to the impedance to the water and sediment flux. A cover-management factor (C-Factor) was used as a representative assessment of the weighting factor. The C-Factor is often developed as a weighted average of the soil loss rate over the year, which itself is a function of prior land use, canopy cover, surface cover, surface roughness, and soil moisture. C-Factor values were estimated from the 2011 National Land Cover Dataset (Homer et al., 2011) using the crosswalk derived from Woznicki et al. (2020).

Index of Connectivity (IC) generated by the SedInConnect tool was used to calculate a SDR for clay, silt, and sand layers. The individual SDR layers were calculated with the ArcGIS Raster Calculator tool using the below equation proposed by Vigiak et al. (2014):

$$SDR_{IC} = \frac{SDR_{max}}{1 + exp(\frac{IC_0 - IC}{k})}$$

SDRmax is the maximum sediment delivery ratio, IC_0 and k are two calibration parameters. The maximum SDR is inversely related to particle size of sediment. Based on Vigiak et al. (2012), IC_0 was set to 0.5, k to 2, and SDRmax estimated by the equation below:

$$SDR_{max} = 0.92 - 0.00093 \times d$$

where d is the soil particle diameter (µm). A weighted average of percent clay, silt, and sand found near the surface (30 cm bottom depth) was developed using the Soil Data Development Toolbox (Peaslee 2018) accessing the region’s GSSURGO Database (Soil Survey Staff, 2019).

The SDR, driven by the IC (Figure 7), has its largest values nearest the input targets and along steep slopes. Roads, being sediment sinks, result in high SDR in urban settings, as well as areas along mountainous streams. Low values are found in flat areas and within urban areas beyond the initial edge of sinks. SDR rasters were created for clay (Figure 8), silt, and sand. Their average SDRs across the study area were 0.04, 0.04, and 0.03, respectively. The maximum value for clay was 0.56 while the maximum value for sand was 0.49. The low SDR values reflect the fact that much of the sediment delivered downstream does not reach the stream or channel, but settled in the landscape and became sediment sources.

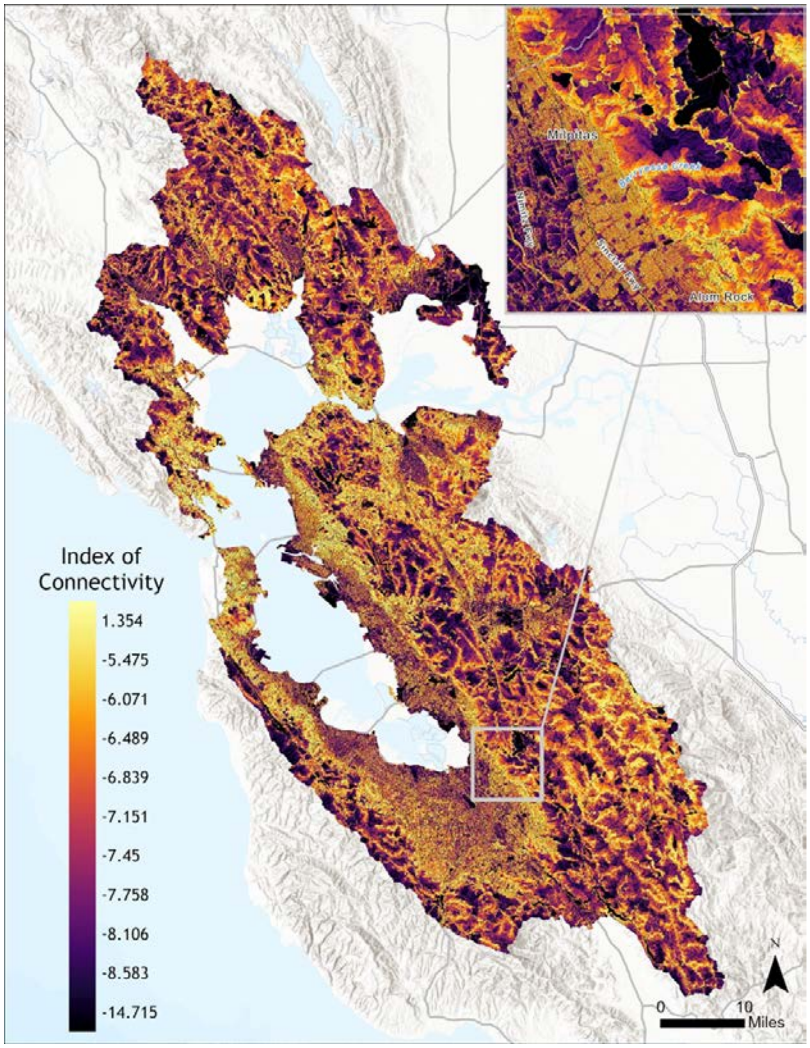

Figure 7. The spatial distribution of the index of connectivity.

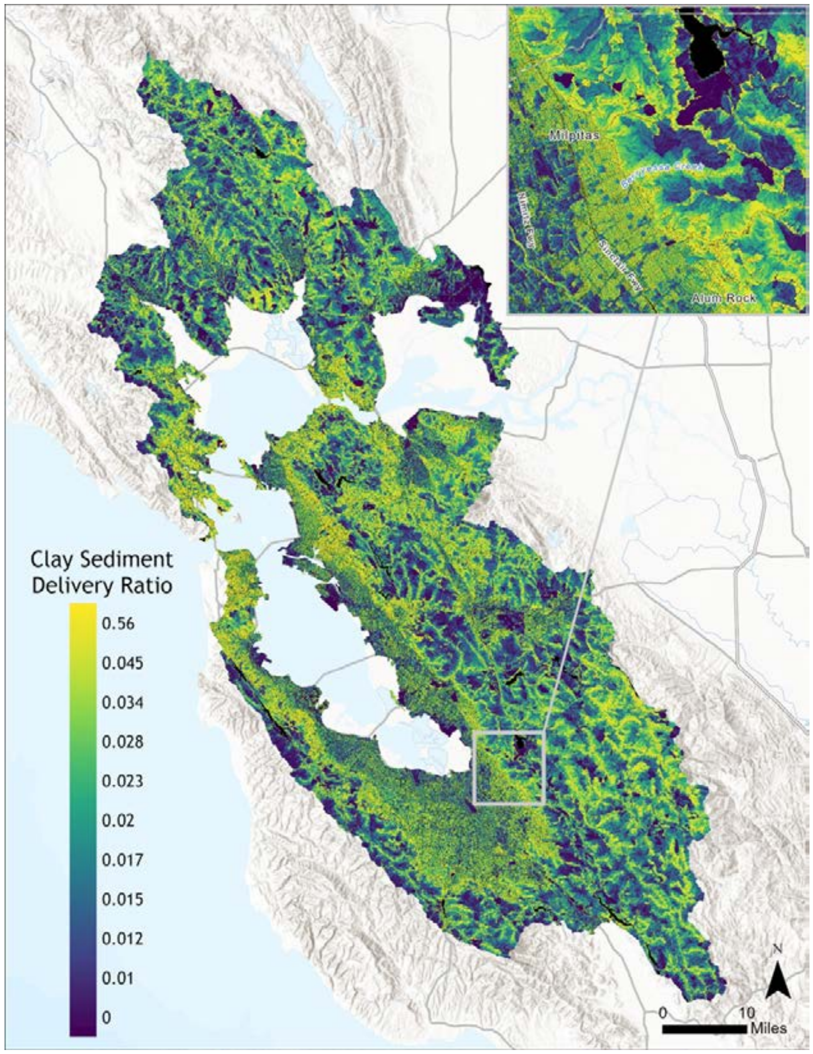

Figure 8. The spatial distribution of sediment delivery ratio for clay.

Table 6 summarizes the area-weighted SDR for different land uses and slopes. The SDR values for natural landscapes are between 0.02 to 0.03, with slightly higher values at medium and high slope classes. The SDR of agricultural land is higher than natural landscapes at low slope class. Urban land uses have higher SDR values than natural landscapes since the roads were considered as sediment sinks similar to channels. Thus a high density of roads and the areas close to roads lead to a high possibility for sediment to be transported into stormwater pipes and then to the Bay or channels. Transportation land use has the highest SDR values, then followed by mixed urban land use and residential area. The estimated SDR patterns for different land surface and topographic features were used to guide the calibration of KSER and JSER (two major calibration factors for upland sediment transport processes) at the HRU level.

Table 6. Area-weighted SDR value for different land use and land cover and slope classes.

Land Use Land Cover	Slope Classes		
	Low	**Medium**	**High**
Agriculture	0.03	NA	NA
Grass and Barren	0.02	0.02	0.03
Shrubland	0.02	0.03	0.03
Forest	NA	0.02	0.02
Mixed Urban	0.05		
New Commercial	0.03		
New Industrial	0.02		
New Residential	0.04		
Old Commercial	0.03		
Old Industrial	0.03		
Old Residential	0.04		
Transportation	0.09		

3.3.3 Sediment Yield Estimation

Sediment yield from different HRUs is the combined result of soil erosion and sediment delivery within each subwatershed. Previous studies of local sediment yields and event-mean concentrations by land use type were used to guide the calibration for the ranks and ranges of variability of sediment yield from different land surfaces. The goal of model calibration is to parameterize sediment properties that capture the relative range of variability among sources observed in literature.

Table 7 shows the comparison between modeled mean sediment yields from different land uses and the values from previous studies. Agricultural land and industrial areas are the two highest sediment yield land uses. The mean sediment yield from different land uses from the WDM follows a similar rank of previous studies as well as the quantities of sediment yield. Figure 9 shows the variation of sediment yield for different land uses. The large ranges in the sediment yield values from different land uses indicate that the land use is not the only control factor for sediment yield. Other factors, such as soil texture, geological structure, and topography also have large impacts on sediment yield. The modeled ranges for industrial and mixed urban land uses do not cover the value cited from reference sources. This is because highly heterogeneous land use classes that are unlikely to be characterized by a single number or range.

Table 7. Modeled suspended sediment yield (SSY) from different land uses.

Land Use	SSY from previous studies (metric tonnes/km2/yr)	Modeled results
	Mean	**Mean [Min, Max]**
Natural	72[1]	274 [1, 706]
Agricultural	2461[1]	1787 [84, 3271]
Industrial	1836[1]	646 [98, 1768]
Commercial	112[2]	189 [57, 369]
Low Density Urban	450[1]	309 [53, 1768]
High Density Urban	996[1]	309 [53, 1768]
Mixed Urban Use	36[3]	272 [120, 553]

1. Donigian et al., 2003
2. SCVURPPP, 2020
3. Lewicki et al., 2009

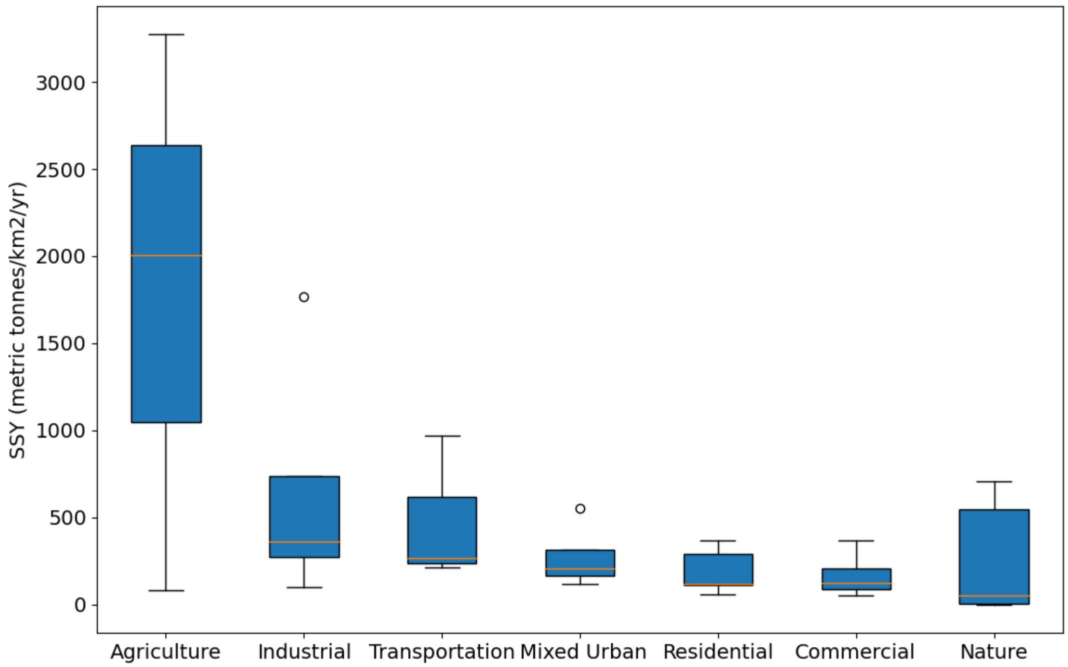

Figure 9. Boxplot of the modeled SSY from different land uses.

3.4 Instream Sediment Load Calibration

3.4.1 Calibration Watersheds

Instream sediment load simulation results were compared with the suspended sediment load monitoring records at five USGS gauges where multi-year flow and suspended sediment records are available. Table 8 summarizes the five USGS gauges at calibrated watersheds. Four out of five calibration watersheds have more than ten years of available data. The Corte Madera watershed has four years of data and is the only USGS site that has recent multi-year sediment data available in the North Bay. As shown in Figure 10, five calibrated watersheds cover a considerable area of East and South Bay, but lack the spatial coverage of San Mateo County, Contra Costa County, and the North Bay in general. The total area of the five watersheds is 3080 km^2, 37.6% of all local watersheds. Figure 11 shows the HRU distributions for all local watersheds and the five calibration watersheds. The HRU distribution from five calibration watersheds is similar to the HRU distribution of all local watersheds, indicating the calibration watersheds are good representations of local watersheds at the HRU level. The size of the five calibration watersheds ranges from 18 square miles to over 600 square miles, capturing the variation in the size of local watersheds.

Table 8. The summary table of calibration watersheds.

Site ID	Station Name	Lat	Lon	DA (mi^2)	Sediment Data Available Period (Water Year)
11169025	GUADALUPE R ABV HWY 101 A SAN JOSE CA	37.374	-121.933	160	2003-2020
11172175	COYOTE C AB HWY 237 A MILPITAS CA	37.422	-121.927	319	2004-2013
11179000	ALAMEDA CREEK NILES	37.587	-121.961	633	2000-2020
11181040	SAN LORENZO C A SAN LORENZO CA	37.684	-122.14	44.6	2009-2020
11460000	CORTE MADERA C A ROSS CA	37.963	-122.557	18.1	2010-2013

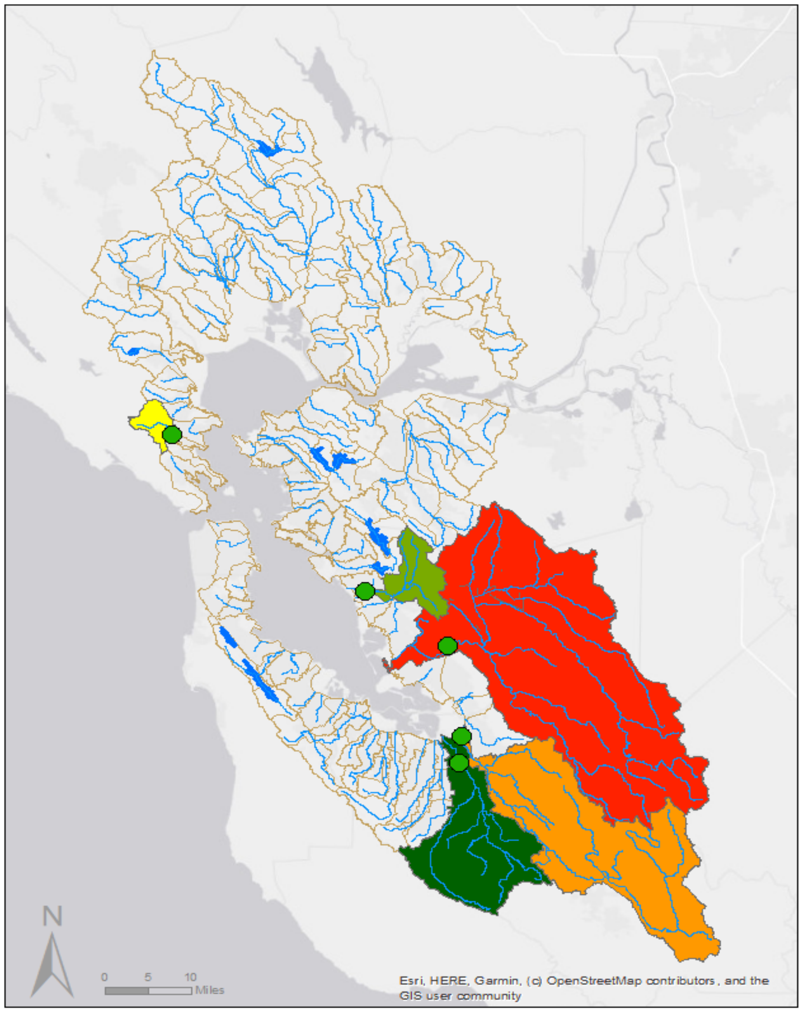

Figure 10. Map of calibration watersheds (colored). The green dots are the USGS sediment monitoring sites.

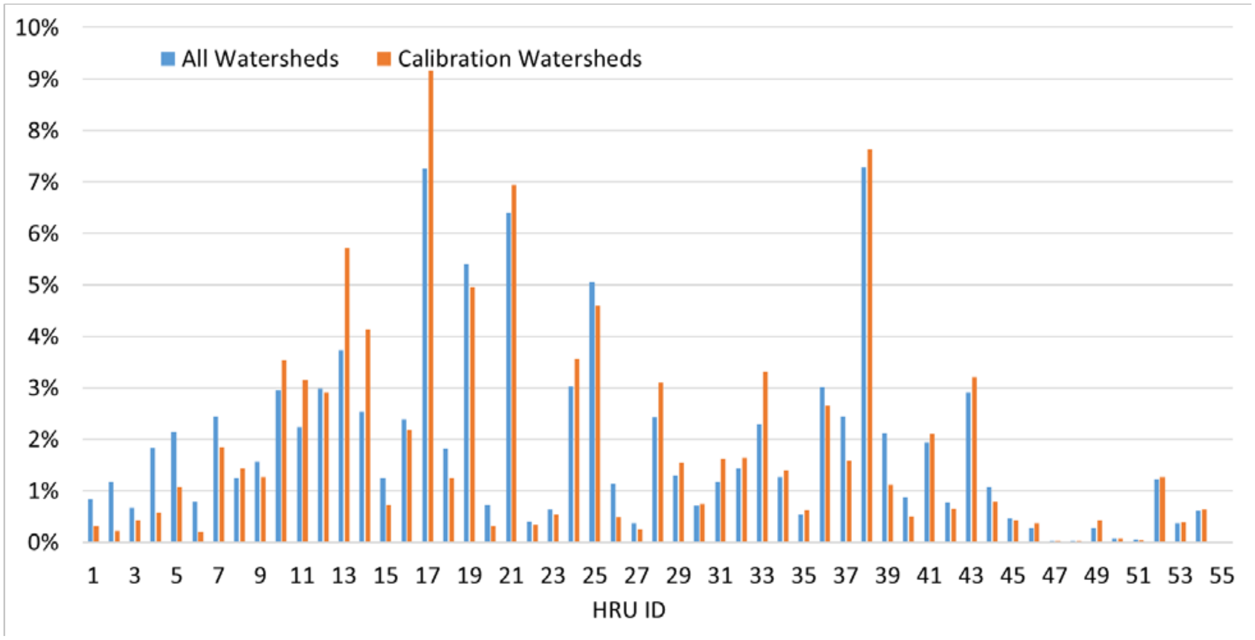

Figure 11. HRU distribution of all local watersheds and calibration watersheds. Y-axis is the percentage of total area.

3.4.2 Suspended Sediment Concentration (SSC) and Suspended Sediment Load (SSL)

Total suspended solids (TSS) and suspended sediment concentration (SSC) are predominantly used to quantify concentrations of suspended solid-phase material in surface waters. The SSC analytical method measures all sediment and the mass of the entire water-sediment mixture. Additionally, the percentage of sand-size and finer material can be determined as part of the SSC method, but not as part of the TSS method. The five calibration watersheds all have suspended sediment data available. Both the daily suspended sediment concentration (SSC) and daily suspended sediment load (SSL) records are available from the NWIS portal (https://nwis.waterdata.usgs.gov/usa/nwis/qwdata). The daily SSC values are cross-sectional average values or derived from point values at different depths. Ideally, the daily SSL would be the time integral of instantaneous water discharge multiplied by the instantaneous, cross-sectionally averaged sediment concentration (Edwards et al., 1999). However, the monitoring data does not support the integral calculation at a high frequency. GCLAS (Koltun et al., 2006) is used by the USGS to interpolate and extrapolate SSC measurement in discrete samples and convert it to daily SSL. GCLAS is essentially an empirical model, where curves are fit through datasets and then interpolation of that curve is used to obtain results for time periods when data do not exist. The daily SSL values were derived in different ways depending on available input data. For example, the cross-sectional SSC data and high frequency turbidity data were collected at the site (11179000) and the GCLAS was used to do regressions on the turbidity record to convert it from Formazin Nephelometric Unit (FNU) to sediment concentration. The turbidity records were used to assist with timing and magnitude estimation of daily USGS SSL. For sites that do not have high frequency turbidity data, high frequency flow records were used to aid the SSC interpolation and extrapolation.

The daily SSC records from USGS sites represent instantaneous cross-sectional or depth average values at sampling dates. USGS Daily SSL data derived from instantaneous SSC sampling data and other monitoring data, such as turbidity and flow, is a more appropriate estimate for the sediment loadings than use single or a few data points to extrapolate the daily sediment load. As shown in Table 9, the USGS SSL values from five calibration watersheds were compared to the values of the products of USGS daily SSC and daily discharge. Figure 12 shows the comparison between the SSL data and the product of SSC and discharge at the Alameda Creek at Niles USGS site (11179000). For different years, the sediment load values from the product of SSC and discharge vary from 46% to 77% of the SSL values published by USGS. This difference can be found in all five calibration sites. The annual average suspended sediment load values, derived directly from the daily SSC and discharge data, are only 56%-78% of the SSL values published by USGS at the five sites. The goal of the sediment modeling is to better estimate the sediment load from local tributaries to the Bay. Thus, the sediment load simulation was compared against the USGS SSL data to calibrate the modeled SSL. The USGS daily SSC records from the calibration sites were used to verify the ranges of SSC values at different watersheds. Model calibration is primarily to estimates of SSL, which omits bedload. The contaminants associated with bedload was considered negligible comparing to the suspended sediment load. If the bedload was a desired output, it can be estimated using the bed particle distribution and the flow simulation results from tributaries.

Table 9. The comparison of USGS daily SSL records and the daily SSL derived from USGS daily SSC and flow data at five calibration sites.

Site ID	Station Name	USGS SSL (metric tonnes/yr)	USGS SSC × USGS Discharge (metric tonnes/yr)	Ratio	Sediment Data Available Period (Water Years)
11169025	GUADALUPE R ABV HWY 101 A SAN JOSE CA	8963	6578	73%	2003-2020
11172175	COYOTE C AB HWY 237 A MILPITAS CA	5695	4456	78%	2004-2013
11179000	ALAMEDA CREEK NILES	165462	104965	63%	2000-2020
11181040	SAN LORENZO C A	26008	14593	56%	2009-2020

Site ID	Station Name	USGS SSL (metric tonnes/yr)	USGS SSC × USGS Discharge (metric tonnes/yr)	Ratio	Sediment Data Available Period (Water Years)
	SAN LORENZO CA				
11460000	CORTE MADERA C A ROSS CA	6966	4123	59%	2010-2013

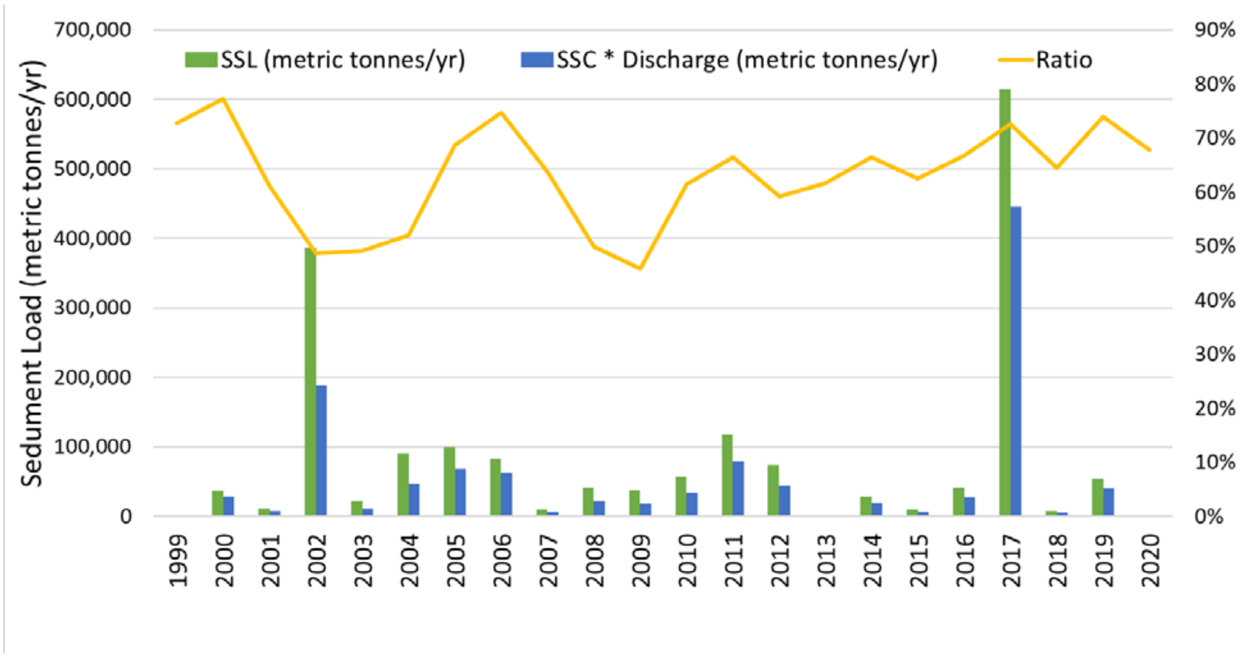

Figure 12. Annual SSL comparison between USGS SSL records and the SSL derived from USGS SSC and discharge records at the Alameda Creek at Niles USGS site (11179000).

3.4.3 Calibration Results

Transport of sand in the WDM is calculated at every model time-step (hourly, in this study) for each reach as a power function of average velocity (transport capacity). Based on the inflow and bed storage of sand compared to the transport capacity, sand scour or deposition occurs. Scour, deposition, and transport of the silt and clay in the WDM is based on user-defined critical shear stresses. Critical shear stresses are generally determined for each reach based on the examination of simulated shear stresses against streamflow. The WDM simulates scour or deposition at a user-defined erodibility and settling rate by comparing the shear stress at a time-step to the critical shear stress for scour and deposition. The WDM was parametrized for instream sediment simulation consistent with the methodology in BASINS Technical Note 8 (USEPA, 2000). Silt was generally set to deposit below the 20th percentile and scour above the

90th percentile shear stress, while clay was set to deposit below the 15th percentile and scour above the 85th percentile. The EPA recommended percentiles were used as starting points. The critical shear stress was unique for each stream segment based on the percentiles recommendations and further calibration for the in-stream sediment processes.

The distribution of simulated shear stress against streamflow for Alameda Creek near Nile (11179000) is shown in Figure 13. The percentile-based approach is used to adjust for these uncertainties in hydraulic characteristics to ensure that the model simulates scour for high flows, deposition for low flows, and transport for moderate flows. The shear stress value is simulated at a reach-by-reach basis, as shown in Figure 14. Figure 14 shows the 90th percentile value of simulated shear stress at different locations of channel segments. The higher the shear stress, the larger possibility of scouring occurs within the channel segment. The simulated shear stress values are consistent with the observation that erosional channel segments are located mainly at steep areas and depositional channel segments are located at flat regions.

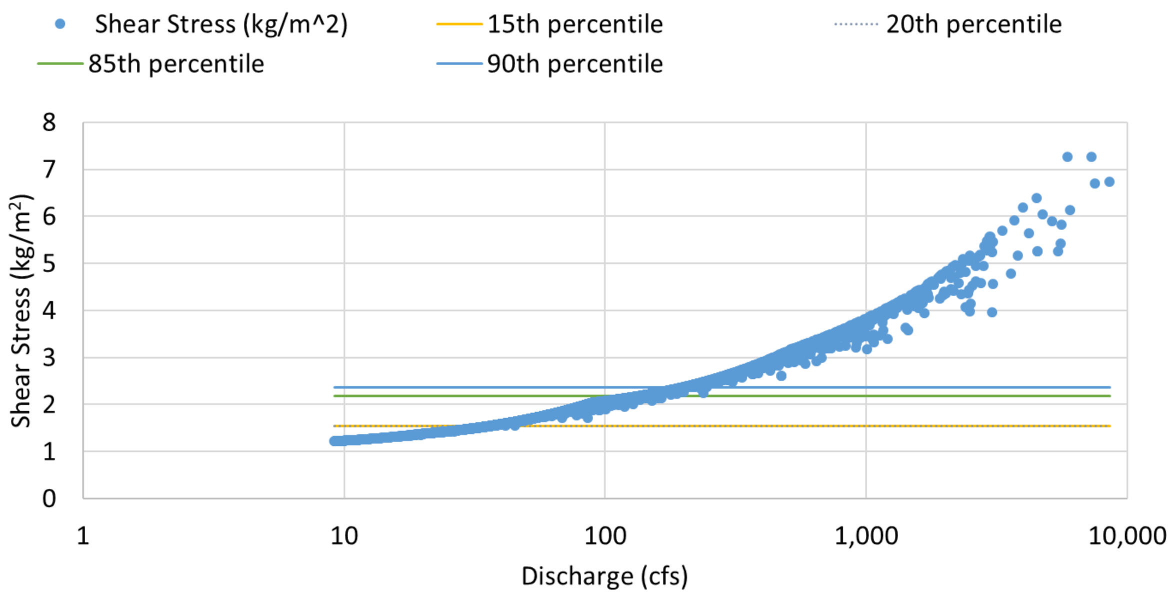

Figure 13. Scatter plot of modeled discharge rate and shear stress at Site 11179000.

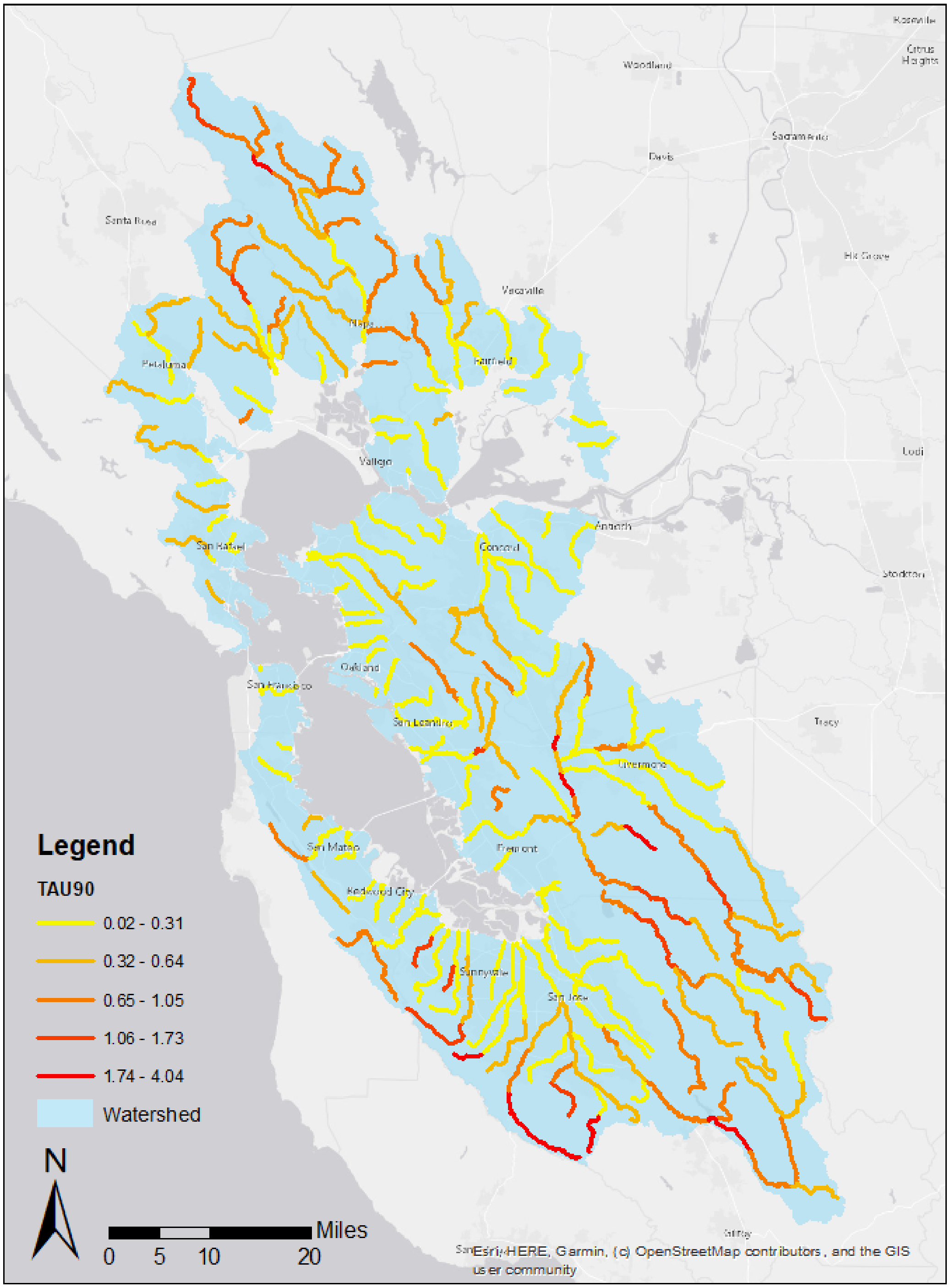

Figure 14. The modeled 90th percentile shear stress (kg/m^2) of channel segments.

Even though the sediment simulation was at the hourly time step and the WDM model could be calibrated for specific events, the goal of model calibration was to have a reasonable estimation of the average annual SSL. The model calibration is aimed at quantifying the annual average SSL not each single event for the past 26 years. Performance of the model for SSL was evaluated using the relative error of average annual SSL. Model performance at the five calibration watersheds is summarized in Table 10. The relative error in flow for the calibration watersheds ranges from -7% to 7%. The NSEs for monthly flow at the calibration watersheds were all above 0.5. Four out of five calibration watersheds are above 0.75. The relative errors in total sediment loads for the calibration watersheds were between -7% and 10%. The model generally simulated the trends in observed concentrations well but was unable to simulate some extremely high concentrations (Figure 15 and Appendix A). Normalized annual SSL was calculated for both monitoring records and modeled results at the five calibration watersheds. Figure 16 shows the comparison of the ratio of annual SSL (the sum of five watersheds) to the multi-year annual average SSL. The modeled annual SSL shows consistent inter-annual variability of sediment load with the monitored data. The variation of annual sediment load of the five watersheds derived from monitoring records is from 4% to 750% of annual average SSL, and the range for modeling data is 7% to 610%.

A closer examination of the observed data shows that some of the highest concentrations are not associated with extreme storm events, which implies the high sediment concentration may be driven by other reasons such as bank failure, reservoir release, or other factors not included in the model. The monitoring data of those situations were analyzed and sediment load from those situations was about 13% of average annual sediment load. Those types of events are complicated processes and are beyond the scope of the WDM model.

Table 10. Model performance at the five calibration sites.

Site No.	Site Name	Hydro-NSE	Hydro-RE	Sed-RE
11172175	COYOTE C AB HWY 237 A MILPITAS CA	0.87	6%	5%
11169025	GUADALUPE R ABV HWY 101 A SAN JOSE CA	0.57	1%	3%
11179000	ALAMEDA C NR NILES CA	0.79	-2%	-7%
11181040	SAN LORENZO C A SAN LORENZO CA	0.92	7%	4%
11460000	CORTE MADERA C A ROSS CA	0.91	4%	10%

* Hydro-NSE is the Nash-Sutcliffe efficiency of flow simulation, Hydro-RE is the relative error of flow volume, Sed-RE is the relative error of SSL.

** Color code: Green -> Very Good, Yellow -> Good, Pink -> Fair.

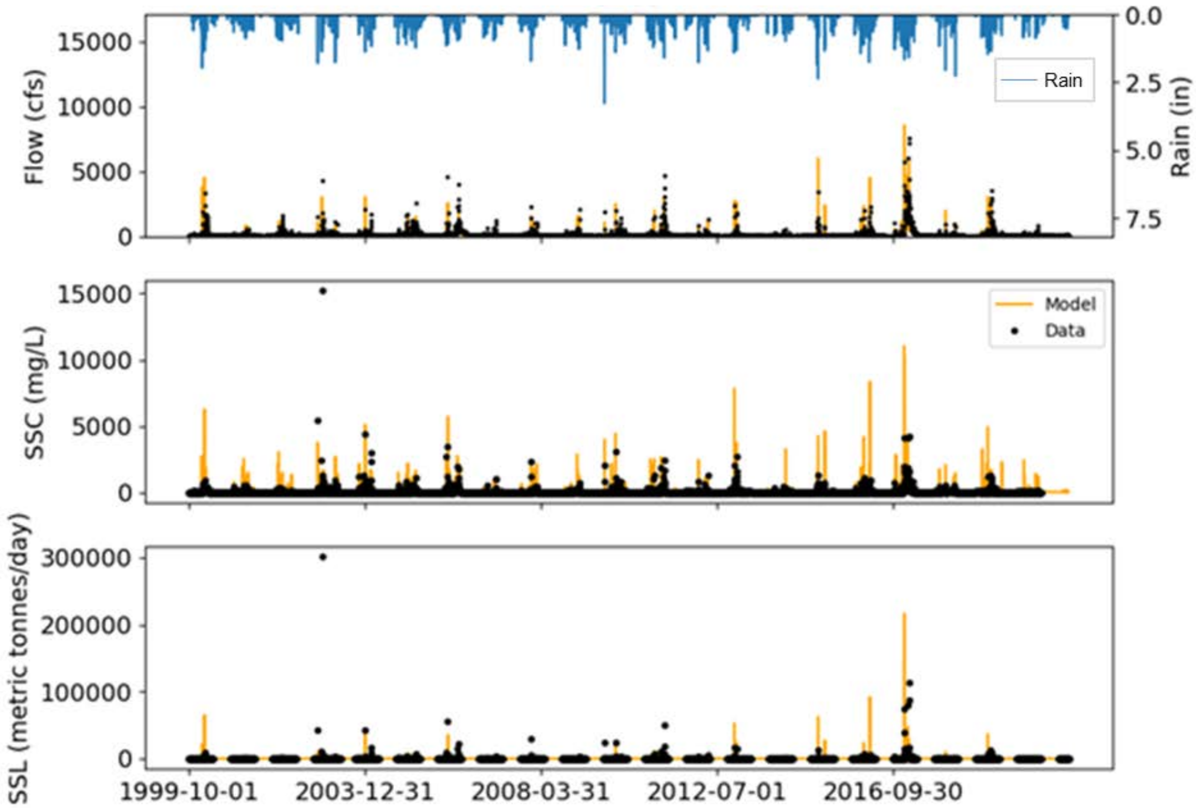

Figure 15. Time series of flow, SSC, and SSL at calibration site 11179000.

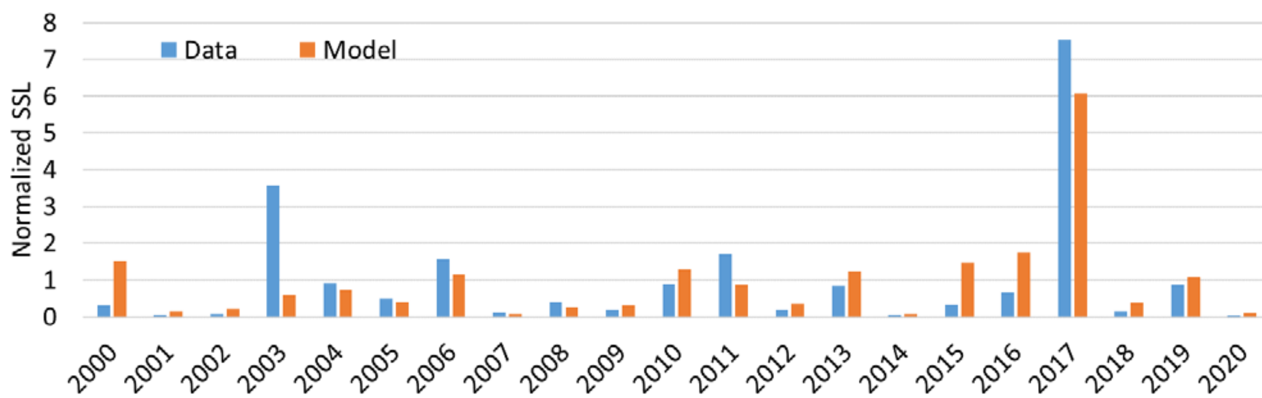

Figure 16. The ratio of annual SSL (the sum of five watersheds) to the multi-year annual average SSL for monitored and modeled SSL.

Monitored and modeled SSC values were compared at the five watersheds. Figure 17 shows the scatter plot between discharge rate and SSC at Coyote Creek watershed (Site ID: 11172175) as an example. The power of the derived relationships between discharge and SSC pairs from monitoring data and modeling results are close to each other and the range of the modeled SSC values are similar to the monitored SSC values. Both of these results indicate a good model representation of the sediment scour and settling processes in the WDM. The SSC scatter plots of all calibration watersheds can be found in Appendix B. Figure 18 shows the boxplot of monitored and modeled SSC (equal and larger than the 90th percentile monitored SSC values) at the five calibrated watersheds. The blue boxes are the monitored SSC values at the five watersheds and the purple ones are modeled. The median, maximum, minimum, and the first and third quartiles of monitored and modeled SSC are comparable at the five calibrated watersheds.

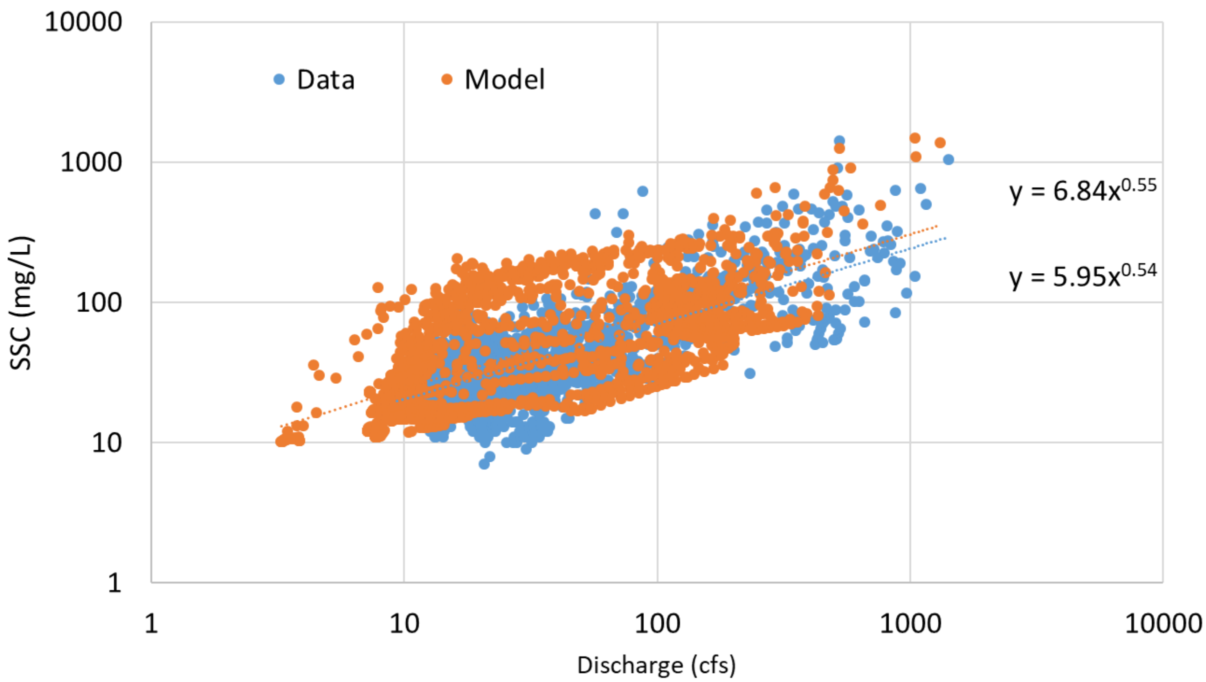

Figure 17. Scatter plot of modeled and monitored SSC at Site 11172175.

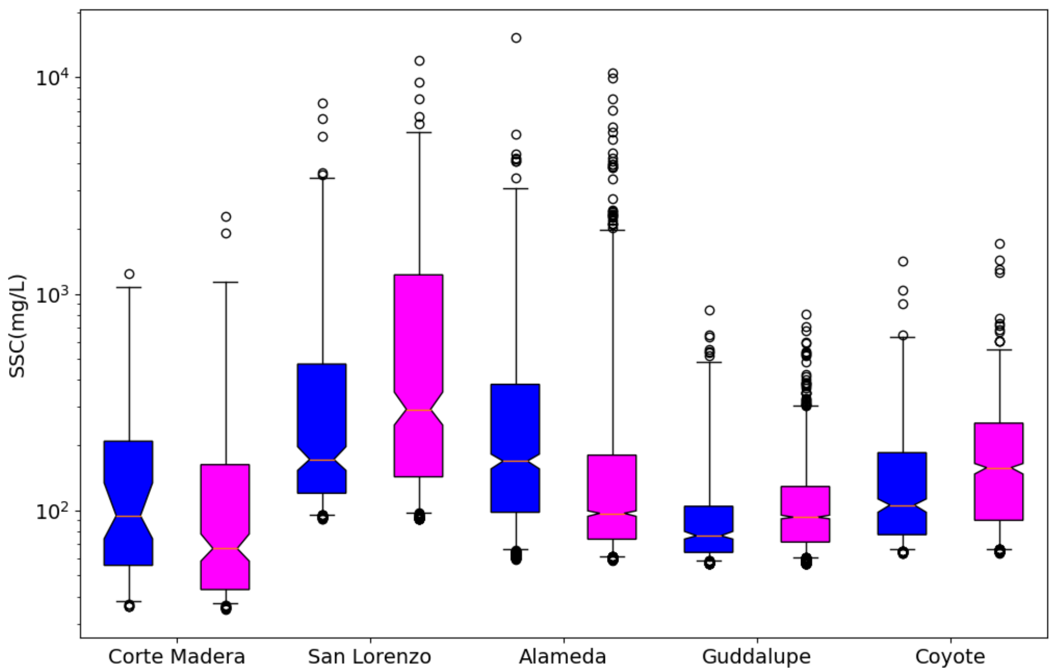

Figure 18. Boxplot of monitored (blue) and modeled (purple) SSC at five calibration watersheds.

Sediment of different size classes go through different upland erosion and in-stream transport processes. The sediment transport processes were grouped into three sediment classes (sand, silt, and clay) in the WDM. The settling velocity of sediment and the critical shear stresses varied by different size classes. Thus, the particle size distribution (PSD) is another variable that could be used to verify the robustness of the physical processes representation of the WDM. At the five calibration USGS sites, sediment PSD samples were available. Average sediment PSD (percentage of sand, silt and clay) was calculated from PSD samples from each of the USGS sites and compared to the modeled average PSD within the reach segments for the calibration sites. Figure 19 shows the comparison of monitored and modeled PSD at five calibration sites. The modeled and monitored average PSD at five calibration sites are similar, which indicates a good estimate of soil particle distribution upland plus a reasonable representation of in-stream sediment transport processes in the WDM. The stability of modeled sediment bed composition was checked during the calibration to assure the reasonable representation of in-stream processes.

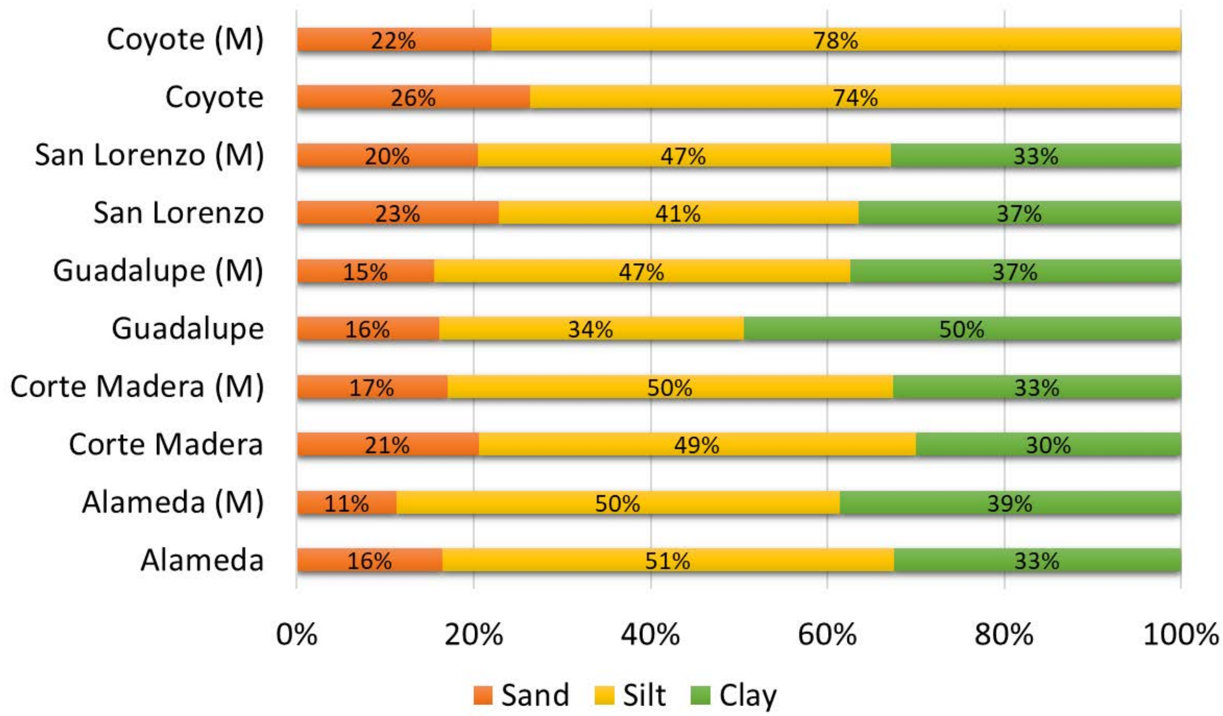

Figure 19. The monitored and modeled (with 'M' in parentheses) PSD at five calibration sites. Silt and clay were grouped into silt at the Coyote site due to no finer classification monitoring data available.

As shown in Figure 10, the calibration watersheds were mainly located in South and East Bay. Sonoma, Napa, and Contra Costa counties lack recent long-term sediment monitoring data for model calibration. Some sediment sites within the three counties have sediment records from decades ago (for instance, Wildcat Creek has records from 1970s and 1980s, as shown in Figure 20) and sampling records from a few storm events of recent years (Wildcat Creek), or sampling records from one wet season (Sonoma Creek and Napa River). Due to the scarcity of sediment samples from monitoring, thorough sediment calibration cannot be conducted at these watersheds. To verify the model performance at these sites, the modeled and monitored discharge-SSC relationships, as an indication of erosional power of flow, were compared. Figures 20 to 22 show the scatter plots for Wildcat Creek, Napa River and Sonoma Creek, respectively. The scatter plot of Wildcat Creek shows very good correlation between the modeled and monitored discharge-SSL relationships. The monitored discharge-SSC relationships from the Sonoma Creek and Napa River are both located at lower regions (smaller SSC value with same discharge rate) than modeled relationships. The recent sediment sampling at the two streams are from the wet season of 2018, which was a dry year. The fitted power from monitoring data is smaller than the fitted power from modeling results. The Napa and Sonoma sites only have one year of monitoring data, which is too limited to conduct thorough sediment model calibration. The average annual sediment supply estimate from the Napa and Sonoma Sediment TMDLs (Low et al., 2008; Napolitano et al., 2009) are 106,503 metric tonnes per year and 269,000 metric tonnes per year, respectively. The model estimated annual average suspended sediment load as 77,835 metric tonnes per year and 253,532 metric tonnes per year, which are comparable to the previous estimates. A few more years' monitoring data during different hydrologic conditions are desired to have a robust model calibration at the two major North Bay streams. As shown in Figure 20, the relationship between SSL and discharge is different from 1997 to 1980 from the relationship from 2004 to 2011. Several possible factors could result in the differences, such as changes in land use (thus sediment sources), channel geometry, and channel bank erosion potential. The development of the WDM could be used to explore the impacts on the discharge-sediment relation from the changes of land use, climate, channel geometry and other aspects than assuming a fixed discharge-sediment relation for decades.

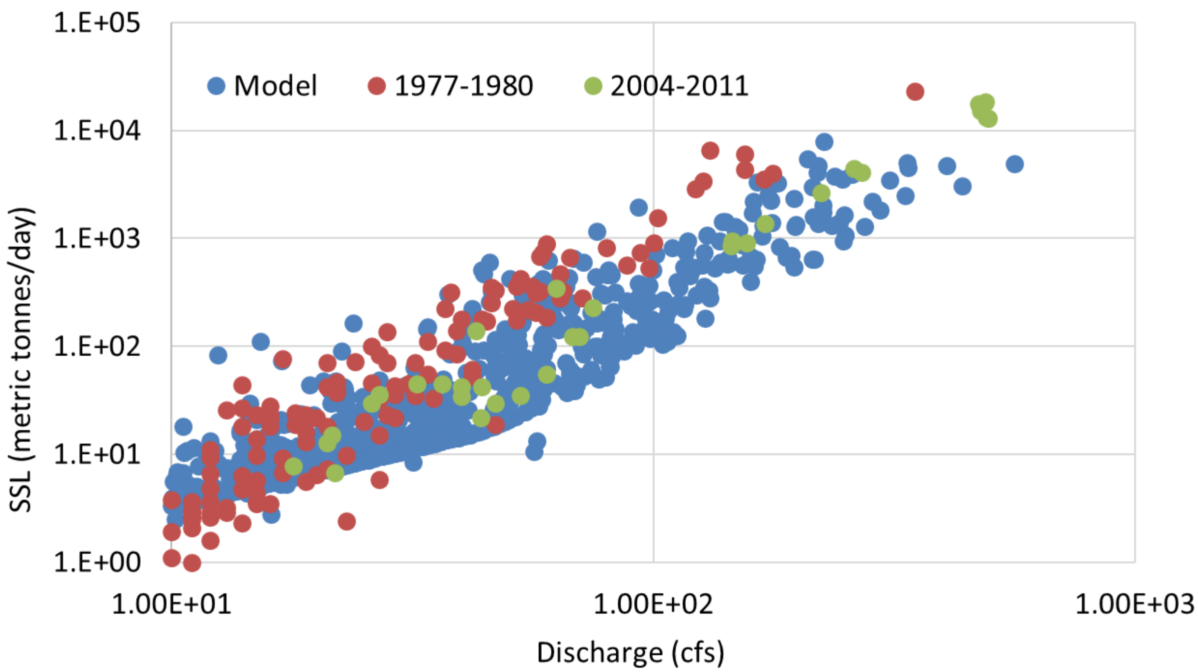

Figure 20. Discharge-SSL scatter plot of the Wildcat Creek. Blue dots are modeled suspended sediment load against modeled discharge (1995-2020). Red and green dots are the monitored data at the listed period.

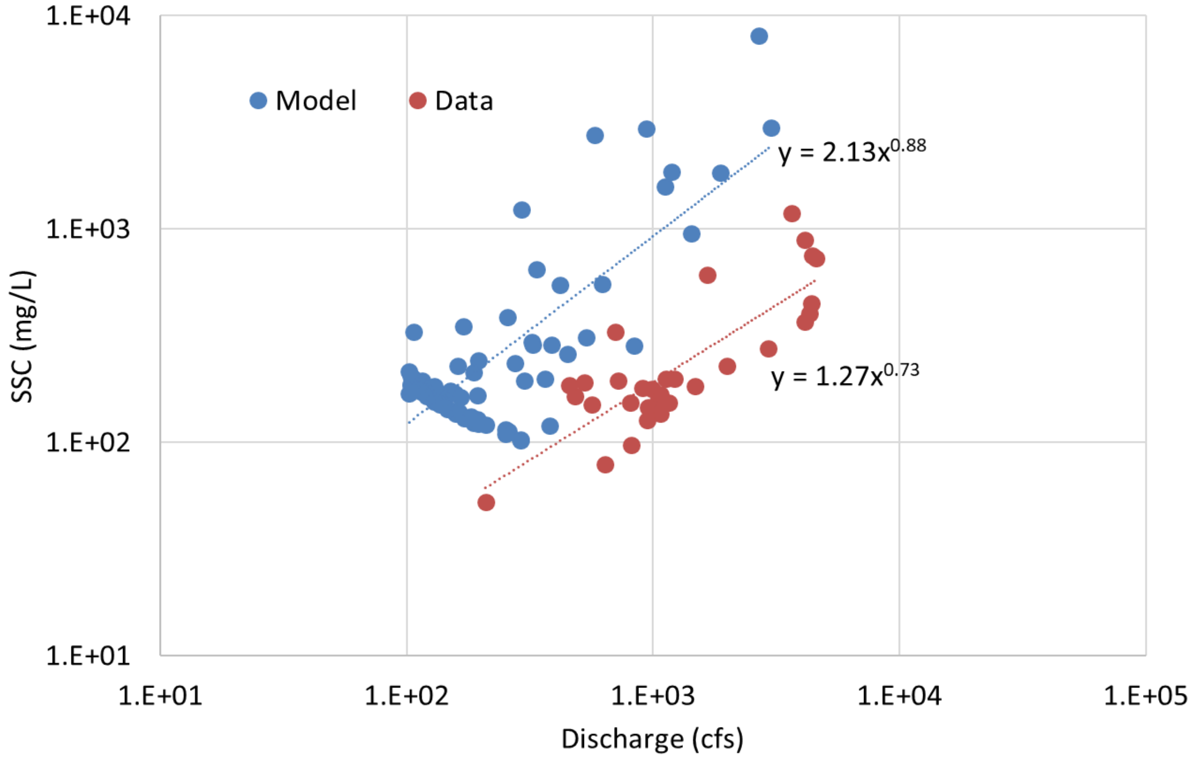

Figure 21. Discharge-SSC scatter plot for the Napa River. Blue dots are modeled and red dots are monitored suspended sediment concentration against modeled discharge (Water Year 2018).

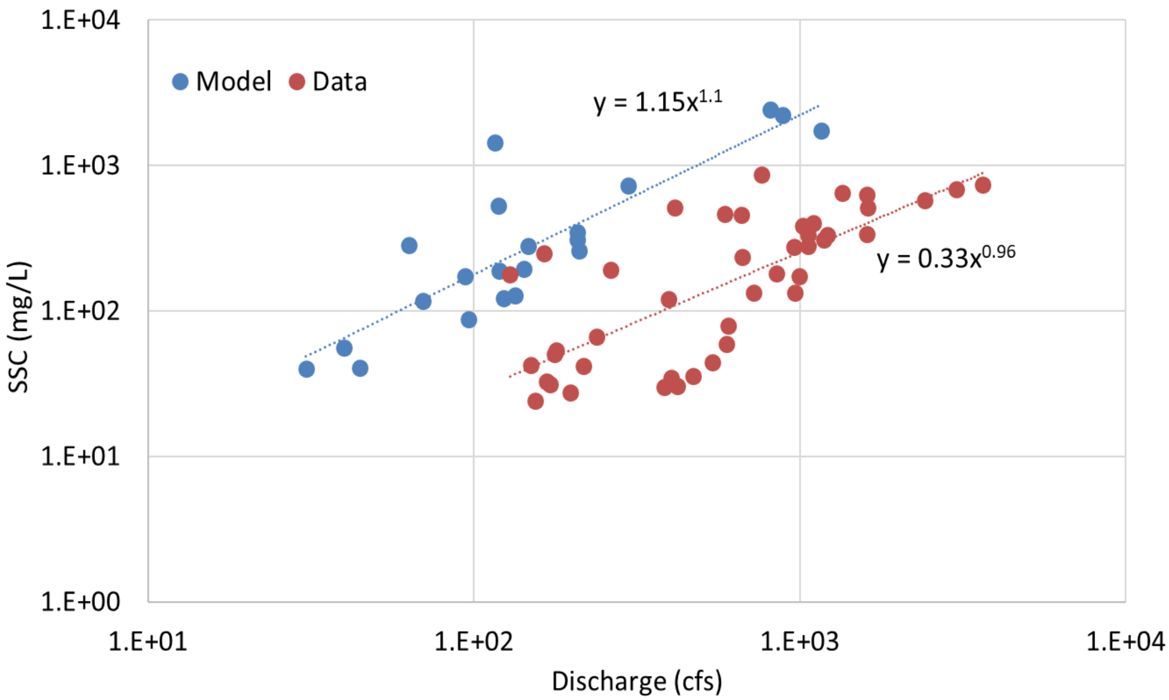

Figure 22. Discharge-SSC scatter plot for Sonoma Creek. Blue dots are modeled and red dots are monitored suspended sediment concentration against modeled discharge (Water Year 2018).

In general, the current WDM sediment module has a solid simulation performance on the total sediment load from local tributaries to the Bay. The model has good performance at calibrated watersheds during the calibration period. The model also shows consistently good performance in interannual sediment load variation, discharge-SSC relationships, and particle size distribution at different locations. For uncalibrated watersheds, the model still shows reasonable representations of the physical processes.

4. Sediment Load Estimation and Uncertainties

4.1 Regional Sediment Load

Modeled suspended sediment load for the region for 86 separately modeled local tributaries is summarized in Table 11. The daily total suspended sediment results from the WDM model were aggregated to yearly records. The estimated 26-year total suspended sediment load from local tributaries to the Bay (1995 to 2020) was 33.64 Mt. The estimated annual average total sediment supply from local tributaries to the Bay for this time period was 1.29 Mt with a standard deviation of 1.06 Mt. Figure 23 shows the average annual SSL of each local watershed. The average annual sediment load from local tributaries was estimated to be 1.4 Mt by McKee et al. (2013) using long-term monitoring sediment data from local watersheds. The modeled average annual SSL from local tributaries is close to the average annual SSL derived from monitoring data. The SSL from local tributaries to different embayments is summarized in Figure 24. San Pablo Bay had the largest sediment supply, 697,003 metric tonnes per year, 56% of the total SSL and 39% of stormwater runoff from local tributaries. Suisun Bay received the second highest, about 16% of the total load. Central Bay and South Bay each received about 12% of total suspended sediment load, and Lower South Bay 4% of the total suspended sediment load with the second highest stormwater runoff contribution (19%).

Table 11. Annual modeled suspended sediment load from local tributaries to the SF Bay. Mm^3 is million cubic meters, Mt is million metric tonnes.

Year	**Total flow (Mm^3)**	**Total suspended sediment (Mt)**	**Year**	**Total flow (Mm^3)**	**Total suspended sediment (Mt)**
1995	2791.58	1.60	2008	1018.42	1.14
1996	2172.02	1.46	2009	850.65	0.30
1997	2213.37	2.44	2010	1376.37	1.34
1998	3387.32	4.27	2011	1728.49	1.02
1999	1220.22	0.98	2012	722.11	0.48
2000	1550.97	1.29	2013	1046.04	0.81
2001	721.23	0.20	2014	501.23	0.31
2002	1244.98	0.89	2015	1258.43	1.19
2003	1468.02	0.98	2016	1365.85	0.96
2004	1233.87	1.57	2017	3317.99	3.59

2005	1473.76	0.48	2018	708.08	0.46
2006	2471.68	3.29	2019	2130.84	2.27
2007	517.55	0.19	2020	520.18	0.13
26-year Total	39011.27	33.64	26-year Annual Average	1500.43	1.29

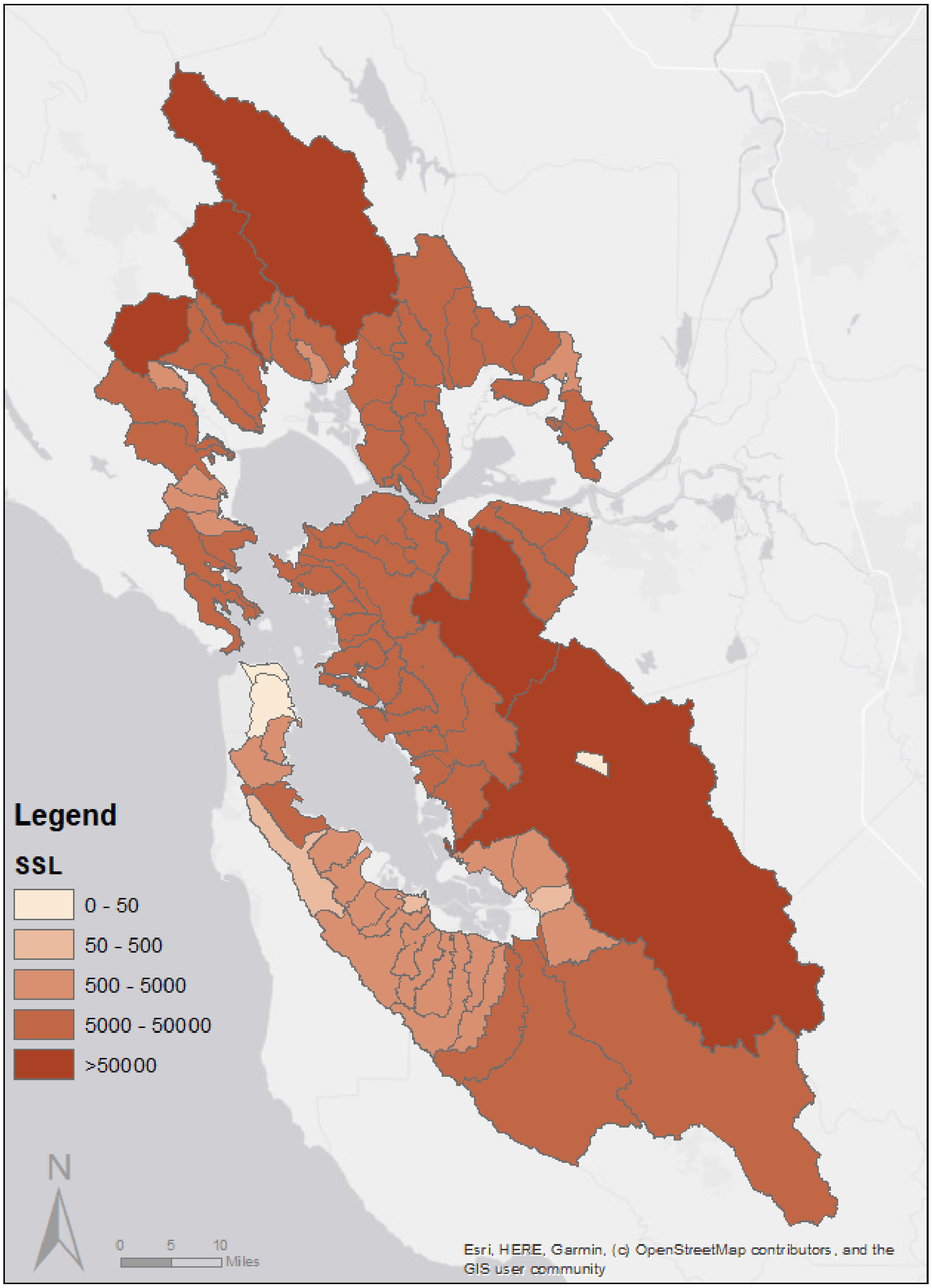

Figure 23. Average modeled annual SSL (metric tonnes) for local watersheds (San Francisco Watersheds not included due to the combined stormwater sewer system).

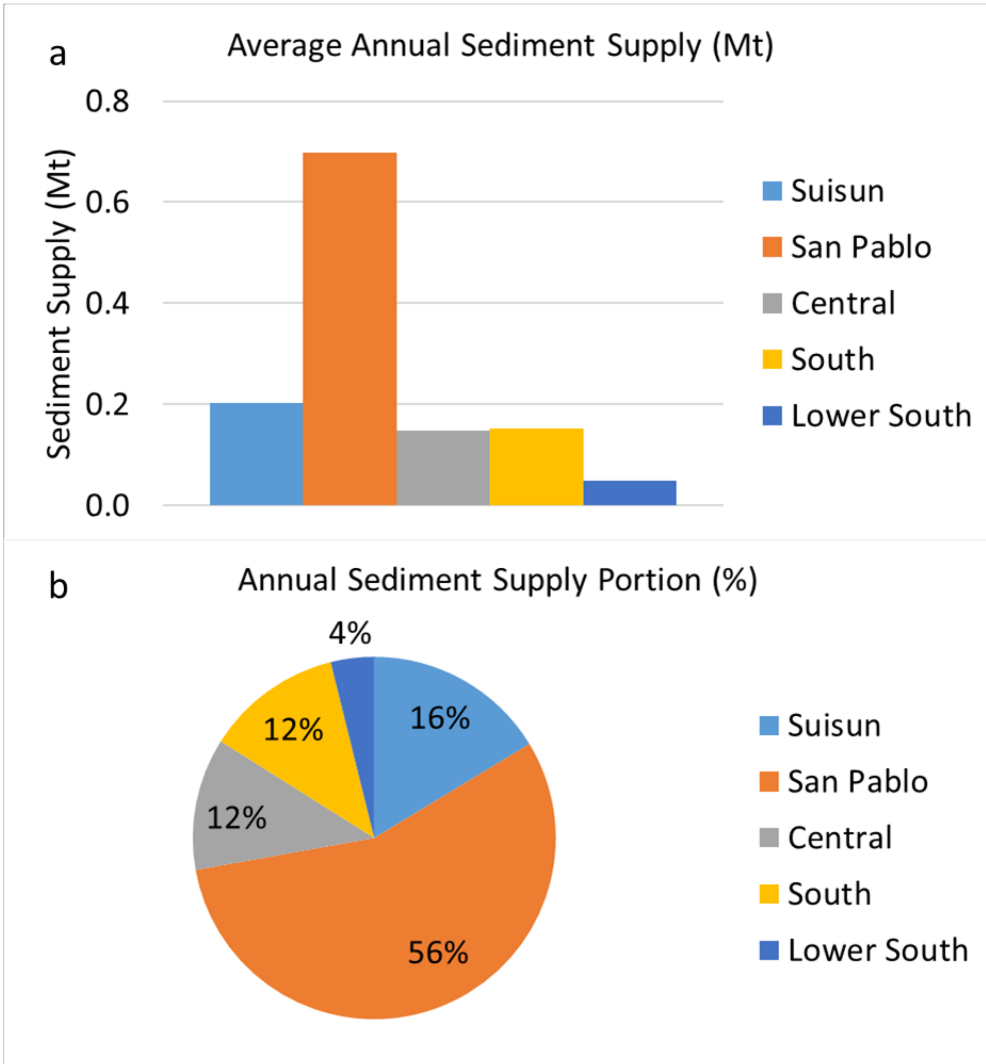

Figure 24. a) Average annual sediment supply (million metric tonnes) to embayment; b) Portion of annual sediment supply to each embayment.

Part of the high variability in SSL estimates generated by the WDM is due to the considerable temporal and spatial variability in rainfall across the Bay area. The average annual precipitation ranges from ~300 mm to more than 1,700 mm. To further exemplify this with an example from a single location, the 20-year average annual rainfall for Berkeley was 608.6 mm, with a standard deviation of 225.8 mm; about 37% of the annual rainfall. The maximum annual rainfall was five-fold greater than the minimum annual rainfall. The variation in rainfall results in larger variation in surface runoff and sediment erosion processes than would occur in flatter areas or those areas with a more temperate climate. This runoff variability is further exemplified by runoff variation at the five sediment calibration USGS gauging locations (Table 12). The ratio of maximum to minimum annual peak flow rate ranges from 10.8 to 35.2, and the ratio for flow volume ranges from 2.1 to 21.6. The coefficient of variation (CV) of annual peak flow rate ranges from 0.6 to 0.85, and the CV of annual flow volume ranges from 0.37 to 1.02. The considerable variation in rainfall and in-channel hydraulic conditions result in huge year to year variation in SSL and thus the sediment load from local tributaries to the Bay. The CV of SSL at the five USGS gauges ranges from 0.34 to 1.98, and the ratio between maximum to minimum SSL ranges from 2.68 to 368.51. With such a huge variability of sediment loadings from local tributaries at a yearly scale, long-term (20 years +) data or simulations are required for average annual loadings estimation (Inman and Jenkins, 1999; McKee et al., 2013).

Table 12. The monitored maximum-minimum ratio and coefficient of variation (CV) of flow volume, peak flow rate, and SSL of five USGS sites.

Site ID	Station Name	Water years with records	Flow volume Max/Min	Flow volume CV	Peak flow rate Max/Min	Peak flow rate CV	SSL Max/Min	SSL CV
11169025	GUADALUPE R ABV HWY 101 A SAN JOSE CA	19	13.67	0.89	18.69	0.63	72.95	1.84
11172175	COYOTE C AB HWY 237 A MILPITAS CA	11	4.53	0.57	10.76	0.6	13.35	0.79
11179000	ALAMEDA C NR NILES CA	22	21.64	1.02	35.21	0.69	368.51	1.83
11181040	SAN LORENZO C A SAN LORENZO CA	13	14.65	0.86	19.72	0.76	335.52	1.98
11460000	CORTE MADERA C A ROSS CA	5	2.14	0.37	27.35	0.85	2.68	0.37

The standard deviation of the WDM-simulated annual average SSL from local tributaries to the Bay for this time period was 82% of the annual average SSL. The maximum annual SSL for the region was 4.27 Mt in 1998, about 33 times of the annual SSL in 2020 (0.13 Mt), illustrating the influence of climatic variability on sediment erosion and transport. These minimum and maximum SSL and the overall variation compare closely to those previously reported for the Bay Area (min: 0.08; max: 4.27; var: 53: McKee et al., 2013). Since the WDM does incorporate real precipitation data and trends over time, it is possible to use the WDM to explore trends in resulting flow in relation to differing land uses, soil types, and slopes, as well as changes in sediment production through changing climatic and land-use related anthropogenic factors. Based on the model estimates, the average annual sediment supply anomalies (the deviation of annual sediment load from the long-term (1995-2020) average) were calculated from 1995 to 2020 (Figure 25). A Mann-Kendall monotonic trend analysis did not identify a significant trend in the anomalies of average annual sediment load. The Mann-Kendall test is a non-parametric test which can be used to statistically assess if there is a monotonic upward or downward trend of the variable of interest over time. Figure 25 shows model estimated long-term sediment load driven by dynamic weather patterns. The dynamic rainfall driven modeling SSL results show the large inter-annual variation of the sediment load due to the change of rainfall erosivity and suggest long-term data (i.e., decadal scale) should be used to quantify annual average sediment load. The trend analysis is sensitive to the starting and ending point of the analysis window, as well as uncertainties in the model. While the 26 years of model estimation provides some trend information for the regional sediment load, more factors (e.g., land use change, model sensitivity, uncertainties) should be taken into account and a carefully designed trend modeling study is recommended for a thorough trend analysis. Sediment load data from a single year or a few years may not show the big picture of the sediment dynamics of a region and should not be used to represent the annual average sediment load. The goal of a thorough trend analysis is to help understand how the sediment load would change given the predictable changing factors (such as climate, land use, management actions), so that the analysis can support the design and implementation of management actions.

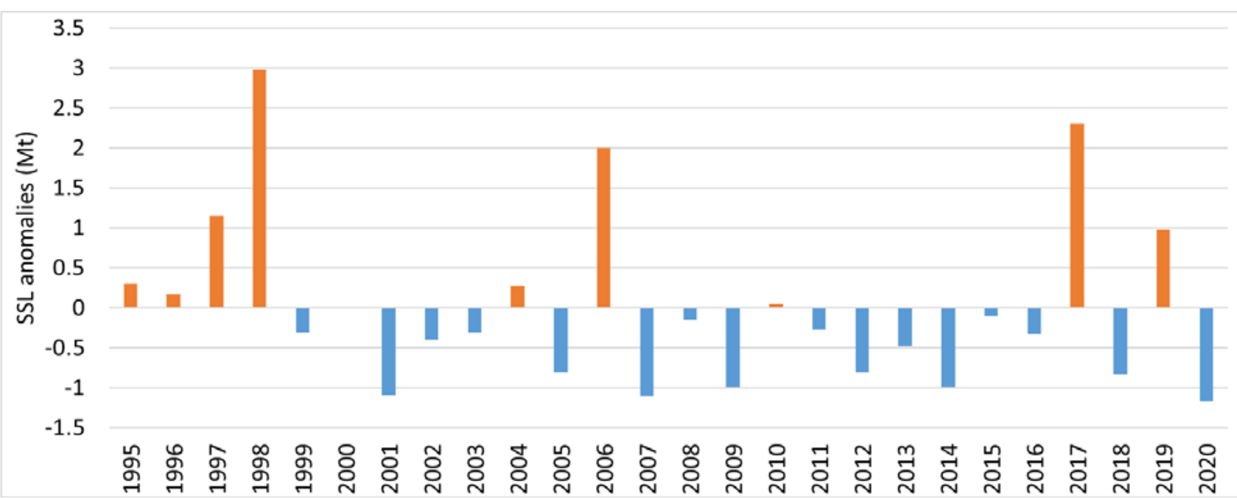

Figure 25. Model estimated annual local tributaries suspended sediment load anomalies (million metric tonnes) from average annual sediment load (1995-2020).

4.2 Model Uncertainties

The Bay tributary stream flow, storm runoff volume, and sediment load estimations were based on the WDM simulation results. There are many factors that contribute to modeling uncertainties. Although the dynamic model has been calibrated against monitoring data at several locations around the region, the estimates are uncertain due to data quality and availability, model process representation, and parameterization.

Sediment calibration in Phase 2 is generally more uncertain than the hydrologic calibration in Phase 1 because it is difficult to simulate varied and localized sediment processes, and because there is a lack of sufficient sediment data to accurately calibrate the model. For watersheds with calibration data available, the WDM has a relative error of model-estimated SSL ranging from -7% to 10%. The sediment load estimation of ungauged watersheds is based on the erosion process of the HRU calibrated at other watersheds and the hydraulic simulation based on the channel geometry. The lack of long-term continuous sediment monitoring data remains a local data weakness that also affects modeling uncertainty. Currently, sediment data in the Bay Area are scarce. The Bay Area has large spatial heterogeneity in climate, land use, soil, geomorphology, and geological conditions. The model was calibrated using data from five tributaries that collectively add to 3,080 km^2, or 37.6% of the total area of the model simulation area (8,191 km^2). Of interest, there is little calibration data available for Sonoma Creek, Napa River, and Walnut Creek, three large watersheds that together are estimated to supply about 22% of the suspended sediment to the Bay. The limited discharge and sediment monitoring sites that were used do not capture the spatial variances of the Bay area, leaving uncertainties in sediment load estimation at the ungauged watersheds. Monitoring data over a larger spatial extent can help reduce the uncertainties of model estimated sediment load at ungauged watersheds.

The accuracy of the USGS depth-discharge rating curve and reach geometry for a specific reach will be a critical factor in adequately representing the hydraulic radius and subsequent shear values, as a function of the stage, or depth of flow. Improper extension of the rating curve and reach geometry can lead to erroneous shear and scour conditions during high flow events and have major impacts on the model simulations for those events. The channel geometry and rating curve data are limited for the Bay Area, which also contributes to modeling uncertainties when using the channel geometry settings which are derived from empirical relationships. The impacts of tides on the sediment in-stream transport processes are out of WDM’s scope but have a large impact on the sediment dynamics of tidal channel segments.

Land use data are crucial for hydrologic, sediment, and contaminant simulation. The Phase 2 model uses ABAG 2005 as major urban land use sources, which is outdated and brings uncertainties in urban HRU accuracy of the WDM. Change in land use is quite important in some watersheds and the outdated land use information was found during the RAA modeling processes. The MTC is working on producing a new land use layer with recently acquired data. The new land use data is expected to correct some land use issues (such as wrong classification, outdated data) found in the ABAG 2005 data. Using the latest land use data will reduce the modeling uncertainties.

Bank erosion is lumped into the channel scouring process as a function of hydraulic forces applied to the streambed and channel sides. Hydraulic forces alone cannot explain the dynamics of bank erosion. Bank erosion is also a result of streambank erodibility and stability. Bank erosion monitoring could help to reduce the uncertainties.

Human activities can affect the sediment budget and transport. Some flood control channel segments are highly managed by dredging (Schoellhamer et al., 2018). Some channel segments are not in equilibrium and may be actively changing their cross-section geometry (e.g., Bigelow et al., 2008). All these aspects are not represented in the current WDM. The absence of dredging activities is likely to have a small influence on the estimation of sediment load since the simulation period is less than 30 years and the geometry of channels may not change a lot due to dredging. Bigelow et al. (2008) also suggests that channel evolution is not a large part of the overall sediment budget for the Bay area anymore. Other human impacts such as dams and constructed channels are likely to have a much larger effect on sediment dynamics, but most of these landscape modifications were made prior to the current timeframe of the WDM. More focused monitoring and modeling studies would be needed to quantify the impacts on sediment dynamics from human activities, if needed.

Erosion and sediment transport processes are very dynamic, subject to huge temporal variation caused by the temporally variable rainfall-runoff processes as well as other processes like land surface changes, including landslides and debris flows (East et al., 2018), which are usually not as well represented in monitoring data. The sediment load derived from the instantaneous samples of sediment thus brings uncertainties in load estimation from monitoring records and uncertainties into model calibration and validation. The current sediment model cannot simulate the dam releases, landslides, and debris flows caused by extreme events. Landslides are a large source of the long-term sediment supply process in the Bay Area (Collins et al., 2018). The Bay area has active tectonic uplift. Where uplift occurs, active dry ravel, which is the downslope transport of solids by gravity independent of overland flow, can be an important source of solids loads to streams. With limited monitoring records, the sediment loads due to landslides and debris flows could be causing additional uncertainties in sediment load estimation. These stochastic events, though not explicitly represented in the model, can be represented as an annual average sediment load at the regional scale, by summarizing monitoring data of such events and calculating the annual average for the whole region.

5. Summary and Future Recommendations

The WDM dynamically simulates rainfall-runoff and sediment generation processes for both coarse and fine sediment classes at an hourly time scale for water years 1995-2020, taking into account different geological characteristics, soil types, land use, land covers, and slopes thus providing a superior method to those used previously for spatially and temporally extrapolating the available limited calibration datasets. This dynamic watershed sediment model is now available for the Bay Area to estimate the total sediment and specific sediment classes (sand, silt, clay) load for the whole region and for specific watersheds. It can estimate the sediment yield from different land uses and simulate sediment dynamics within channels. Aided by monitoring data, this numerical model can be used to better understand the sediment transport processes and budgets of San Francisco Bay. This sediment modeling tool can be used to refine watershed sediment management and serves as a solid basis for sediment-associated contaminant load modeling in the next phase of the WDM and will be used as watershed boundary condition inputs to the in-Bay PCB fate model being developed by the PCBWG.

The Sediment for Survival report (Dusterhoff et al., 2021) analyzed future sediment supply from local tributaries using a combination of existing flow and sediment data and future climate scenarios (both a wetter and a drier future). The analysis used a number of “focus tributaries” around the Bay Area that had reasonable data records, and then extended the findings to the other tributaries. The study also applied climate change projections to a fixed rating curve that assumes no change in the sediment rating in relation to changing climate, vegetation, land use or sediment erosion patterns. The sensitivity analysis of the sediment load with the changes on rating curves did show high variability, which implies large uncertainties of future sediment load predictions. All these challenges can be addressed by further development of the WDM. This modeling platform could be used to simulate future scenarios in a dynamic manner. The future sediment supply dynamics and the range of likely changes to average annual watershed sediment load could be estimated based on downscaled climate model outputs and the range of likely changes to the frequency of extreme storm events combined with scenarios of the future land use distributions and management activities. The results can be used as a starting point for hindcasting past sediment supply and forecasting the future sediment supply to determine the relative change over previous and future decades.

Contaminant load calibration will be the next phase of the model development. The goal of contaminant load calibration is to obtain agreement of simulated and observed concentrations within acceptance criteria with physically realistic parameters, then provide reasonable load estimates from local tributaries to the Bay. The contaminant loading from different land uses should be consistent with the expected ranges based on the literature, conceptual models, and field observations. Management actions, such as GSI removal efficiency, can be reasonably represented in the modeling structure. With the completion of a specific contaminant load model, the WDM could be used to conduct both trend analysis for long-term contaminant load projections and to test management scenarios to support management decisions. The sediment load estimation from the WDM was calibrated using flow and SSC data. SSC values were derived from the dry weight of sediment in water samples, which may not be suitable for quantifying the loads of contaminants attached to volatile organic solids.

Some major data gaps, the level of complexity of sediment dynamics, and the lack of understanding of the details of related physical processes were major sources of uncertainties of the modeling efforts. The WDM is a work in progress. The data used and assumptions and categorizations described in this document are subject to change as new data are obtained and calibration efforts are required. For the following phases and future model development, the regional watershed model could be enhanced in several ways in the future.

A few recommendations are proposed to reduce the uncertainties of sediment load estimation and to better understand sediment dynamics at the SF Bay area:

1. Verifying and improving the accuracy of land use and vegetation cover inputs to the model (the current model impervious HRU area is relatively low compare to previous RAA model at the same region) and to include temporal change over the model calibration period (currently 1995-2020). These parameters have a large influence on the production of flow, sediment and contaminants from the landscape and is important on quantifying the contributions of different HRUs. This is also important for contaminant (PCBs, Hg, CECs) load estimation.
2. Gathering monitoring data at urban regions (impervious surfaces) for sediment accumulation, washoff rate, and removal rate (such as street sweeping) to better parameterize the sediment transport process at impervious surfaces. These monitoring data can improve sediment associated contaminant load estimation.
3. Monitoring processes such as bank erosion, landslides, debris flows at large spatial scale, identifying these events spatially, and qualitatively or quantitatively describing the magnitude of those processes. Doing this can help quantify the sediment supply from these events which the model does not simulate currently. It can also help parameterize and verify the bank erosion module of the sediment model.
4. Conducting monitoring to understand some key modeling parameters, such as sediment particle size distribution at land surfaces and at channel beds, channel geometries, and cross-sectional area.
5. Completing sediment gauging in the watersheds of Sonoma Creek, Napa River, and Walnut Creek, three large watersheds that produce a lot of sediment but that have no or limited recent data to support either load estimates for specific years or model calibration.
6. Evaluating the impacts of wildfire through some land processes such as soil sealing and removal of vegetative cover. More post-fire monitoring is recommended for regions that are prone to wildfire events to help the WDM better represent the wildfire impacts on sediment dynamics. Wildfire can have large impacts on sediment dynamics, especially in the Napa River and Sonoma Creek watersheds.

7. Conducting a thorough model sensitivity and uncertainty analysis on different factors to prioritize the key factors that affect the modeling results most and to quantify the modeling uncertainties by cross-validation and sensitivity analysis.
8. Extending the model for more years (backwards to capture more of the existing sediment data from the 1970s and 1980s or forwards if coupled with more sediment gauging in the Napa, Sonoma, and Walnut Creek watersheds) and using it for trend analysis or future predictions in relation to land use and climate change.
9. Conducting thorough trend analysis based on the results model sensitivity analysis and uncertainty analysis. The aim of the trend analysis is to explore the sediment load responses given different possible future changes (e.g., climate, land use, management actions) and to support management actions.
10. Utilizing multiple monitoring resources such as remote sensing images and high frequency turbidity sensors to increase the spatial and temporal coverages of monitoring data.
11. Recalibrating the sediment model every few years to keep the model representations up to date with the changes of erosion and sediment transport processes.
12. Adding monitoring activities that are designed for different pathways. The main purpose of the sediment simulation is to support estimation of contaminant loads (sediment and pollutant) by different pathways. A key factor is representing the right balance between pathways – e.g., between upland erosion and channel erosion. Direct evidence on the fraction of sediment load that is derived from source areas in recent contact with the atmosphere (e.g., tillage, roadway solids, and upland sheet and rill erosion) and those that have not (e.g., bank failure, gully formation, mass wasting) can be obtained through atmospheric radionuclide isotope analysis (Pb, Be, Cs).

In general, this report serves as a reference documentation of Phase 2 of the WDM development. The sediment model setup and calibration details are documented for audiences who are interested in using the regional watershed model in future. Future model enhancement directions are also listed in this document as a guidance for the future phases of model development.

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Appendix A. Modeled SSL Time Series at Calibrated Watersheds

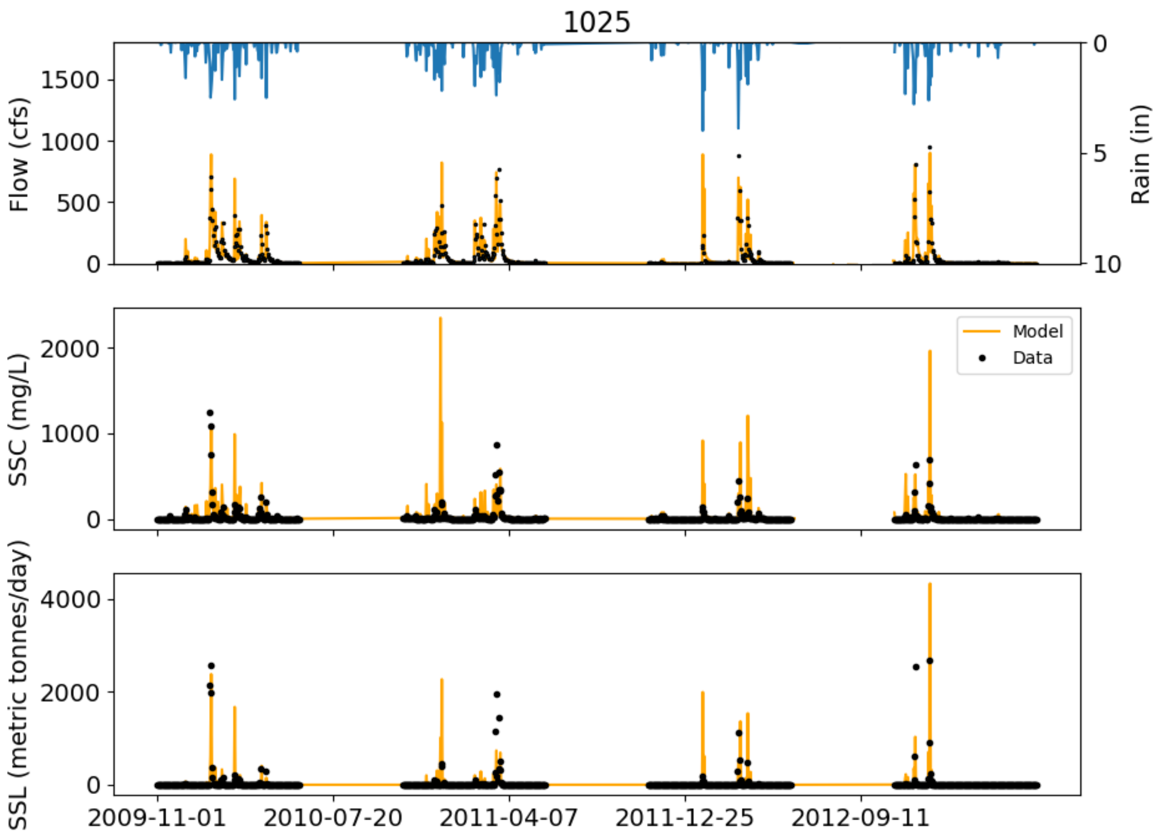

Fig A1. Monitored vs modeled flow, SSC, SSL at Corte Madera watershed

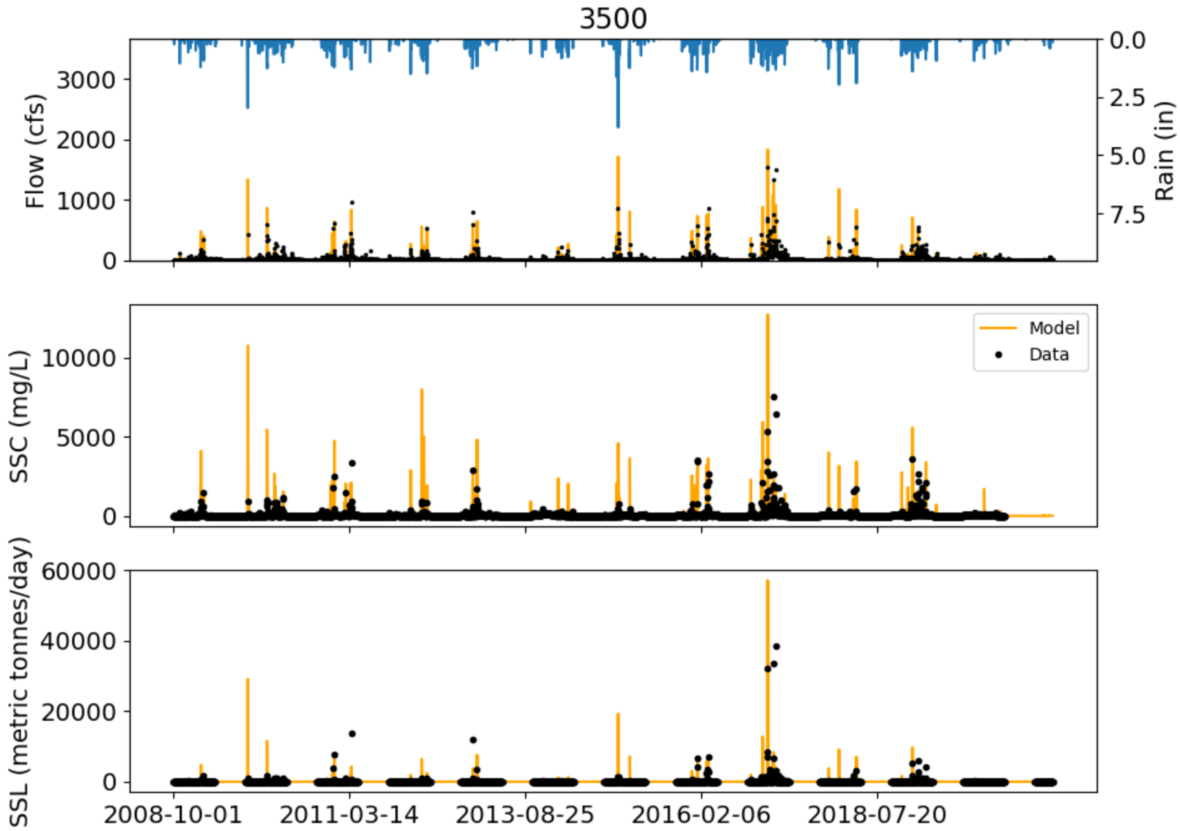

Fig A2. Monitored vs modeled flow, SSC, SSL at San Lorenzo watershed

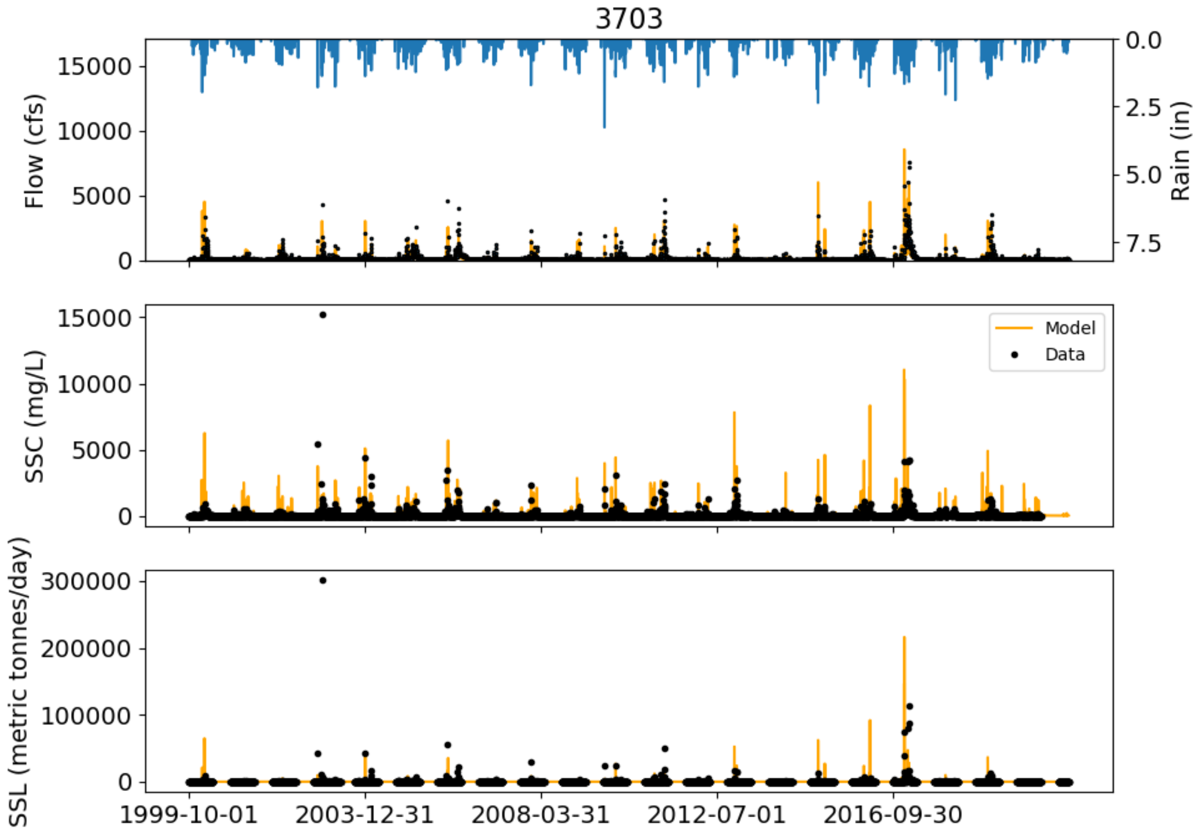

Fig A3. Monitored vs modeled flow, SSC, SSL at Alameda Creek watershed

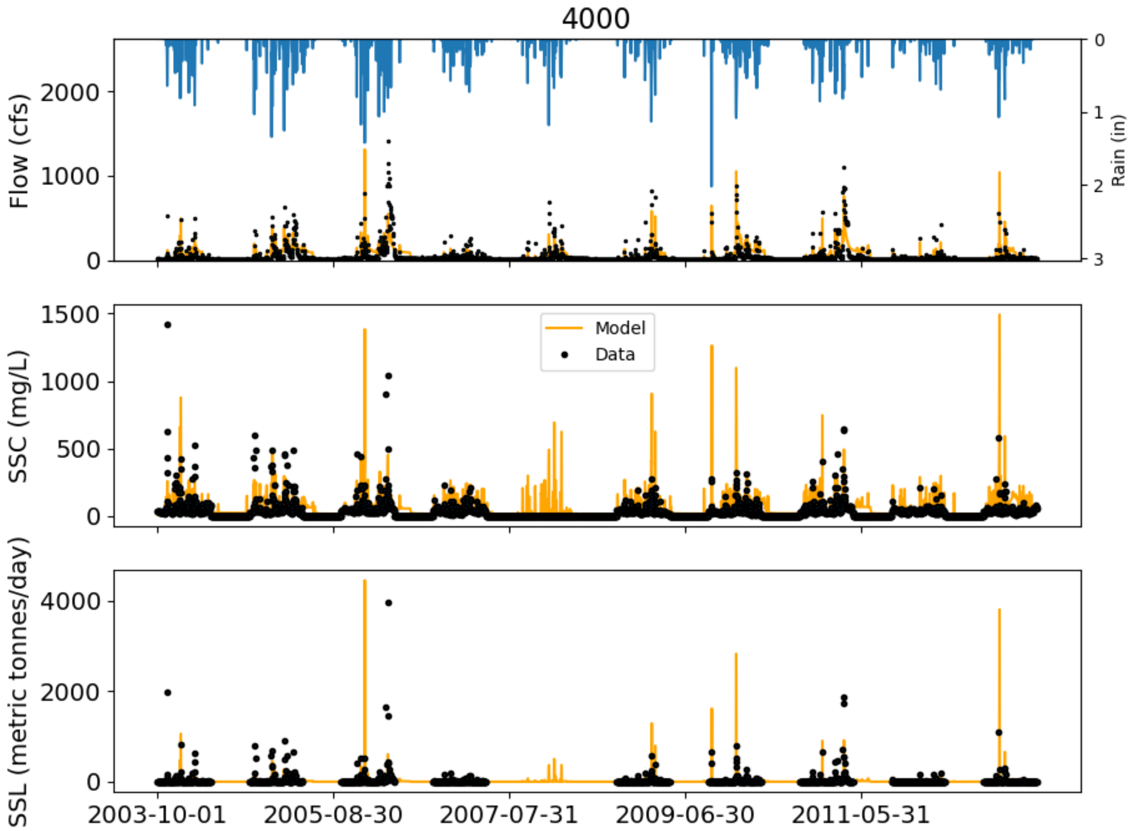

Fig A4. Monitored vs modeled flow, SSC, SSL at Coyote Creek watershed

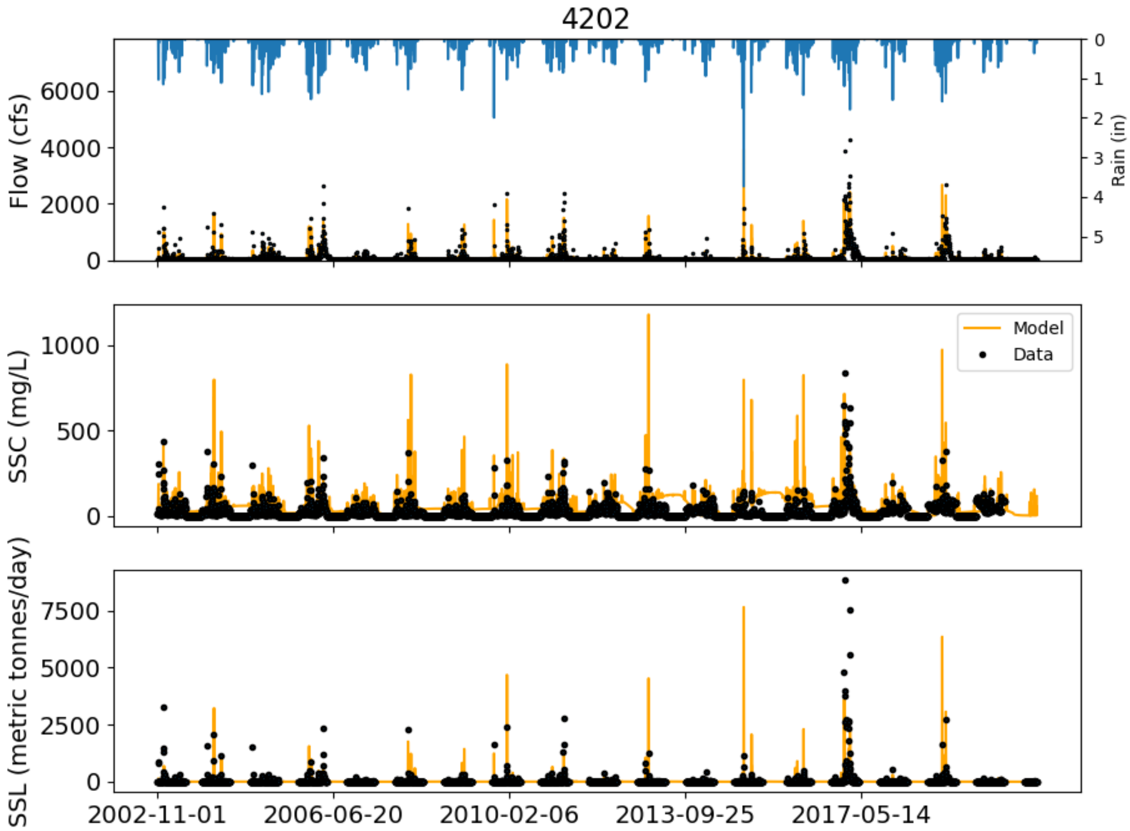

Fig A5. Monitored vs modeled flow, SSC, SSL at Guadalupe River watershed

Appendix B. Modeled and Monitored SSC Scatter Plots at Calibrated Watersheds

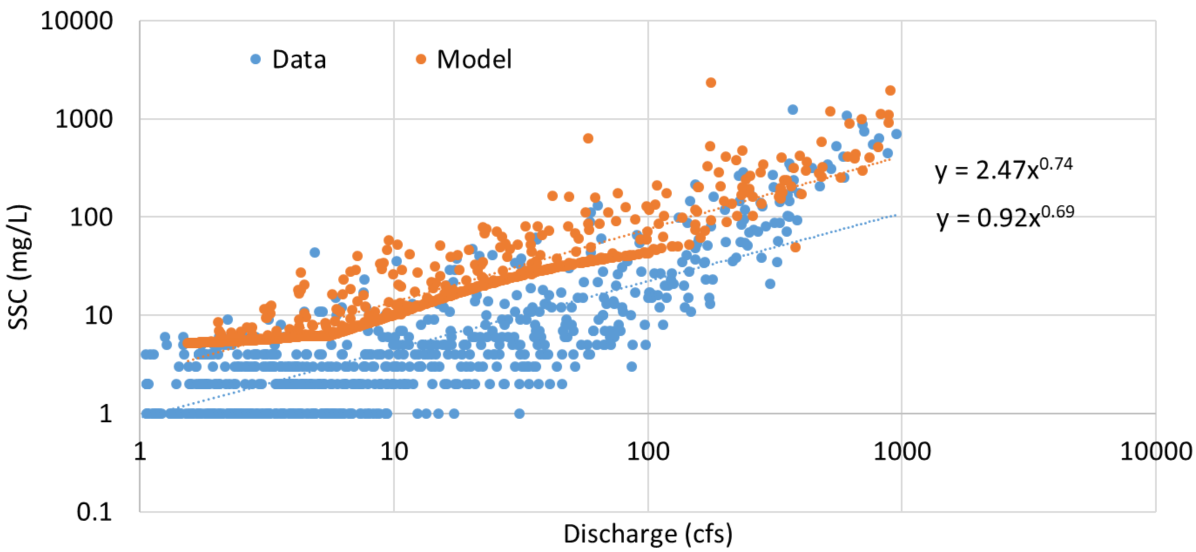

Fig B1. Scatter plot of daily discharge rate and suspended sediment concentration (SSC) at Corte Madera watershed

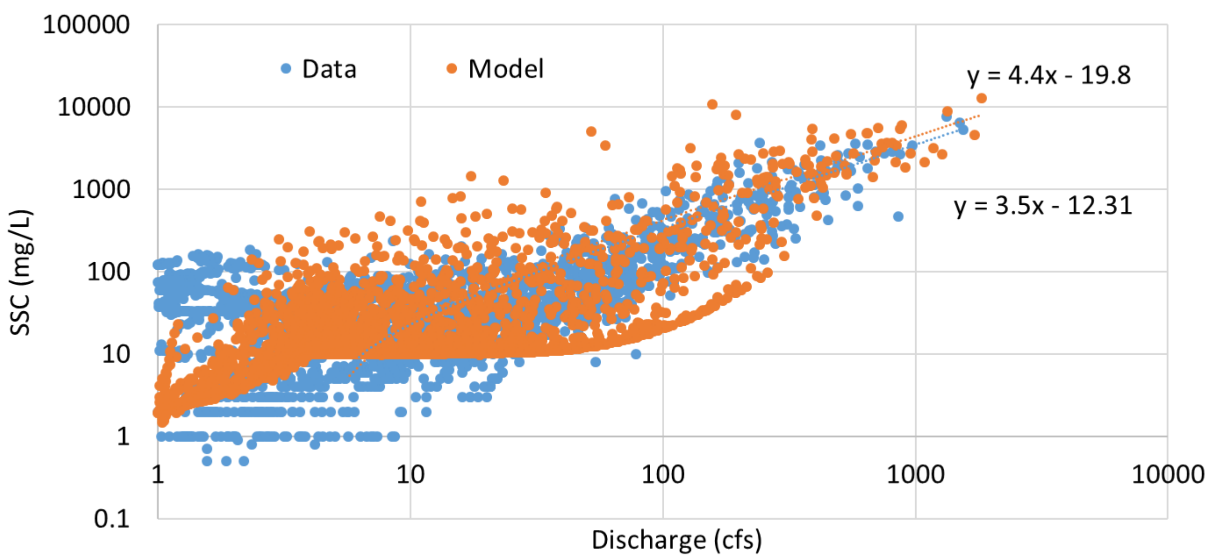

Fig B2. Scatter plot of daily discharge rate and suspended sediment concentration (SSC) at San Lorenzo watershed

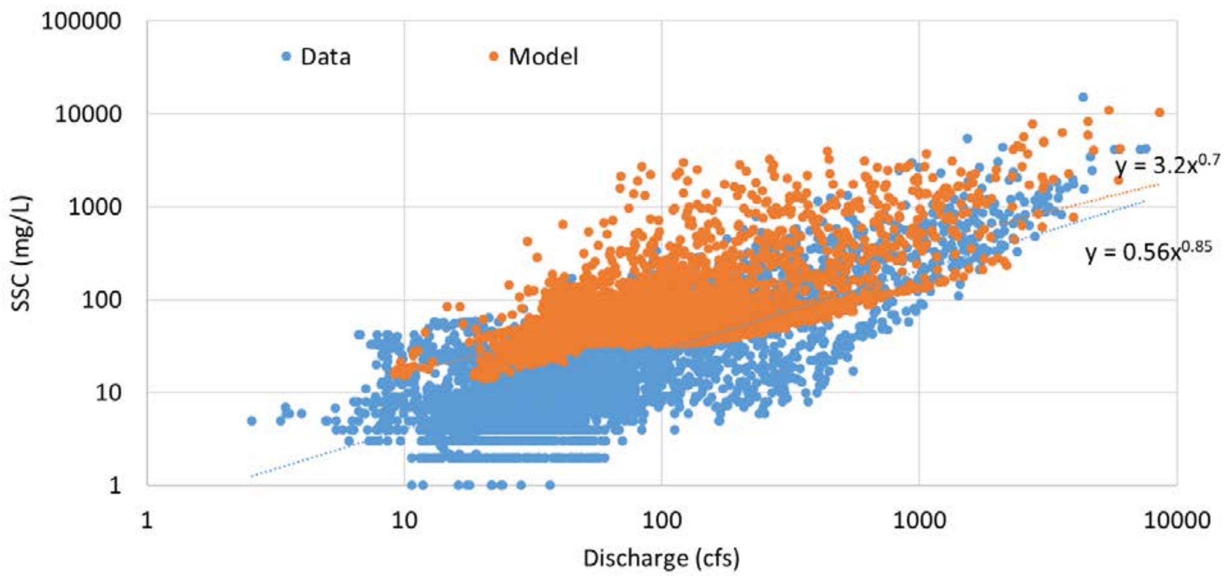

Fig B3. Scatter plot of daily discharge rate and suspended sediment concentration (SSC) at Alameda Creek watershed

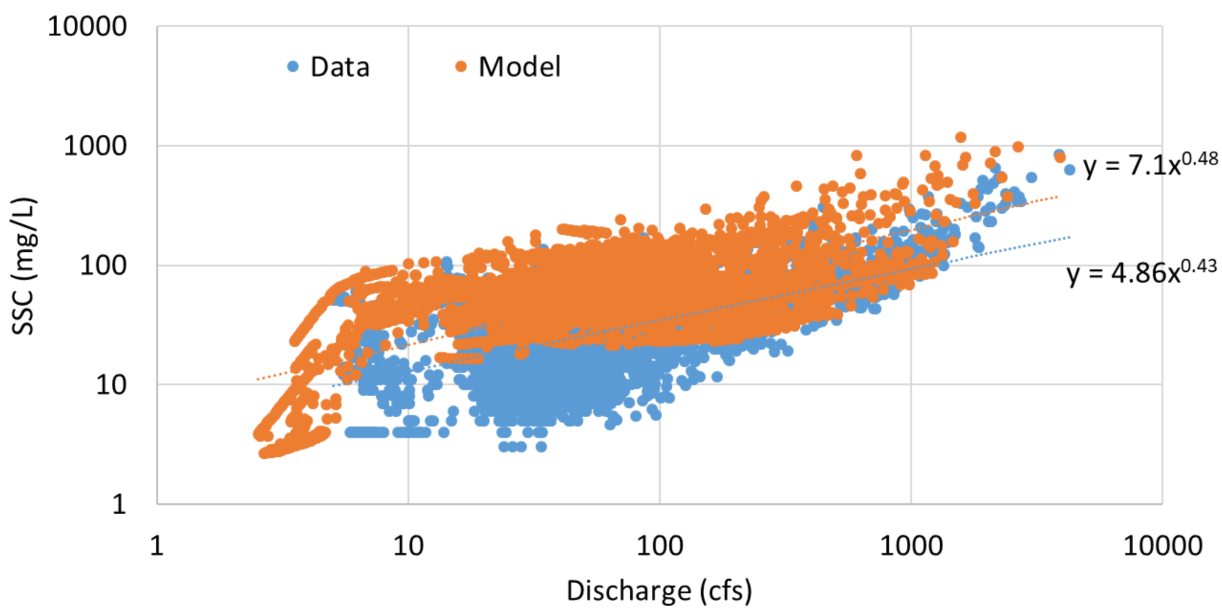

Fig B4. Scatter plot of daily discharge rate and suspended sediment concentration (SSC) at Guadalupe River watershed

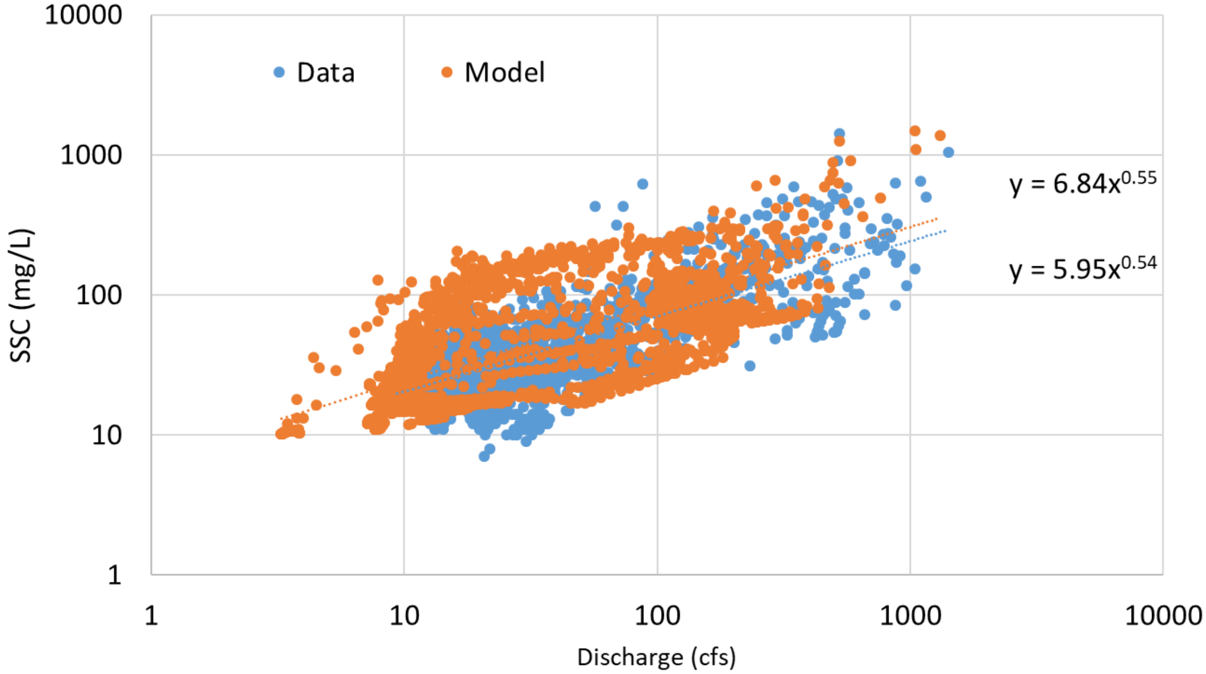

Fig B5. Scatter plot of daily discharge rate and suspended sediment concentration (SSC) at Coyote Creek watershed