Appendix D

EFDC Receiving Water Modeling Report

Receiving Water Model Configuration and Evaluation for the San Diego Bay Toxic Pollutants TMDLs
FINAL

Prepared for:
U.S. Environmental Protection Agency, Region 9
San Diego Regional Water Quality Control Board

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1. Introduction

Water quality modeling can be used to establish the quantitative understanding necessary to develop scientifically justifiable Total Maximum Daily Loads (TMDLs) for a waterbody. A water quality model that is customized for a specific waterbody can simulate the major physical, chemical, and biological processes that occur in the system, and thus provide quantitative relationships between the water quality response and external forcing functions. This report summarizes the development and calibration of the hydrodynamic and sediment transport model components of a coupled hydrodynamic and water quality modeling system under development to support TMDLs in San Diego Bay. A customized modeling framework was developed to support toxic pollutant TMDLs for five shoreline areas of San Diego Bay.

Five San Diego Bay shoreline segments are impaired by toxic metals and organic pollutants including zinc, polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), chlordane, and lindane. The five shorelines include: 1) the mouth of Paleta Creek, 2) the mouth of Chollas Creek, 3) the mouth of Switzer Creek, 4) the vicinity of B Street and Broadway Piers, and 5) Downtown Anchorage. A TMDL report addressing sediment toxicity and benthic community impairments caused by PCBs, PAHs, chlordane, and lindane has been completed for the mouths of Paleta, Chollas, and Switzer Creeks. Note that the lindane impairment is for Switzer Creek only. This modeling report addresses modeling results for those TMDLs as well as modeling results that will be used in developing TMDLs for sediment toxicity and benthic community impairments caused by zinc, PCBs, PAHs, and chlordane at the mouths of the B Street/Broadway Piers and Downtown Anchorage watersheds. Note that the San Diego Regional Water Quality Control Board (San Diego Water Board) will write the TMDL report for the San Diego Bay shorelines at B Street/Broadway Piers and Downtown Anchorage at a later date.

Toxic pollutant loads at each of the five impaired shorelines were calculated by using a watershed model to simulate overland runoff from the five watersheds and linking the watershed output to a receiving water model to simulate pollutant loads in the bay. This document discusses the receiving water model and its results. The watershed model is discussed in two separate documents titled *Monitoring and Modeling of Chollas, Paleta, and Switzer Creeks* (Schiff and Carter 2007) and *Watershed Modeling for Simulation of Loadings to B Street/Broadway Pier and Downtown Anchorage, San Diego Bay – Draft* (Tetra Tech 2008).

The modeling framework used in this study can be divided into two major components that represent the processes essential for accurately modeling hydrology, hydrodynamics, and water quality in the San Diego Bay watershed. The first component of the modeling system is a series of watershed models developed to predict pollutant loadings for each of the three watersheds addressed in the TMDL report (Paleta, Chollas, and Switzer Creeks) and the two watersheds that will be addressed in the later TMDL report (B Street/ Broadway Piers and Downtown Anchorage). The second component is a receiving water model of the mouths of the watersheds (estuaries) and

San Diego Bay used to simulate water circulation and pollutant transport in the tidally-influenced waterbodies. The Loading System Program in C++ (LSPC) was selected to simulate the watershed loadings (Shen et al. 2004; USEPA 2003a). The mouths of the five watersheds (estuaries) and the bay were represented by the Environmental Fluid Dynamics Code (EFDC) (Hamrick 1992). The EFDC hydrodynamic model incorporates flow and loading from the watershed models (see watershed modeling reports) and subsequently determines their impact on the five impaired shorelines as the pollutants are transported through the bay.

EFDC, as well as LSPC, are components of the United States Environmental Protection Agency's (USEPA) TMDL Modeling Toolbox (Toolbox), which has been developed through a joint effort between USEPA and Tetra Tech, Inc. (USEPA 2003b). The Toolbox is a collection of models, modeling tools, and databases that have been utilized over the past decade in the determination of TMDLs for impaired waters. It takes these proven technologies and provides the capability to more readily apply the models, analyze the results, and integrate watershed and detailed hydrodynamic and water quality receiving water applications. The Toolbox provides exchange of information between the models through common databases; therefore, the results from the LSPC model were easily incorporated into the EFDC water quality model.

The EFDC model simulates the tidally-influenced waterbodies, including the mouths of the five watersheds (Figure 1-1), using a multi-dimensional grid. The EFDC receiving water model was linked to the LSPC watershed model to incorporate watershed loads from each of the subwatersheds draining to each of the watershed mouths. This modeling report is intended to accompany the TMDL report and provide a more detailed discussion on the EFDC model used for the TMDL analyses, including model configuration, calibration and validation, and assumptions.

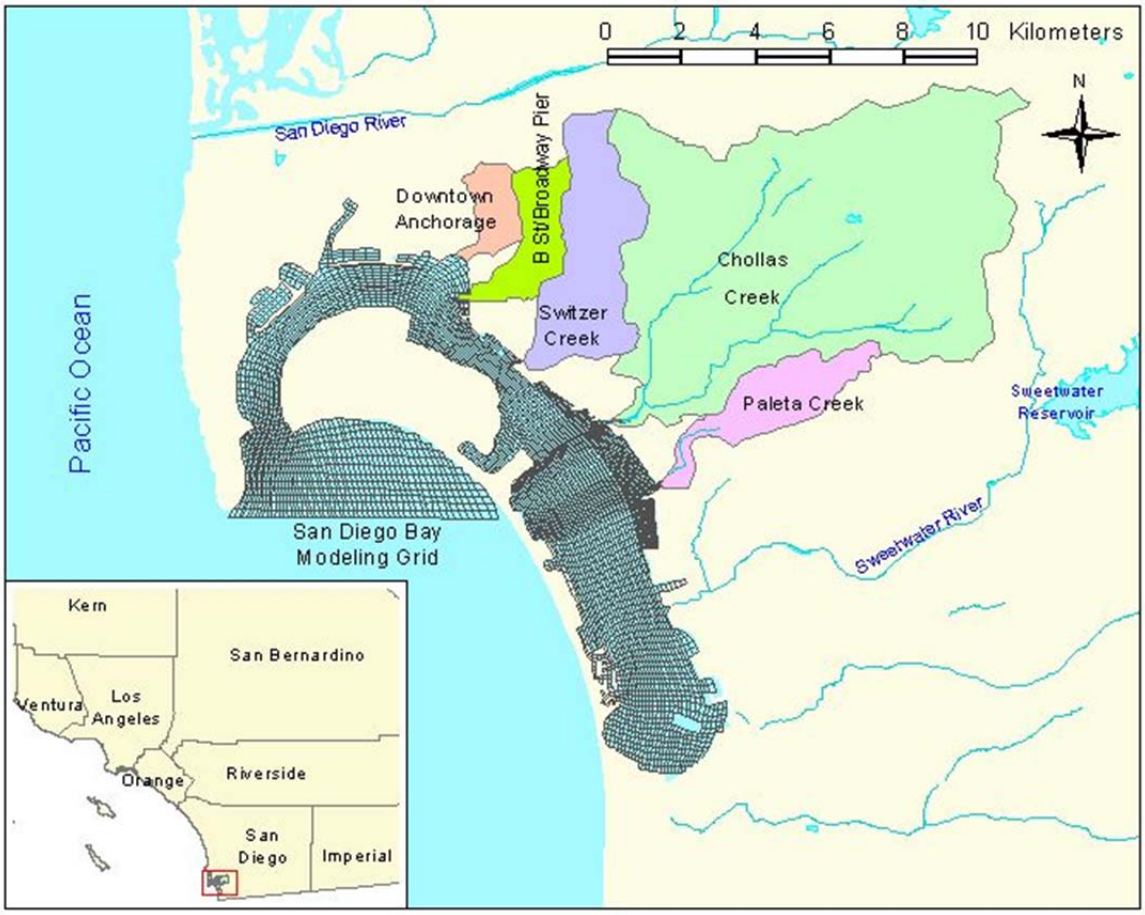

Figure 1-1. Model domain for San Diego Bay (note that the grid shows the areas modeled by EFDC).

2. Modeling Approach

An appropriate modeling framework was selected to support development of the toxic metals and organic pollutants (zinc, PCBs, PAHs, chlordane, and lindane) TMDLs for five shoreline segments in San Diego Bay. Analytical requirements were first defined. Based on these requirements, multiple potential approaches were then identified and evaluated. The approaches considered include various numerical models and combinations of models applied at different spatial scales and with various predictive capabilities. The following discussion summarizes the key considerations, approaches identified, and the final approach selected to meet the project objectives.

2.1 Analytical Requirements

Toxic pollutant TMDL development for the shoreline segments in San Diego Bay requires that a number of key analytical considerations be addressed:

- Influence of hydrodynamics, including tidal impacts and interactions between fresh water and tidal water during and after storm events.
- Representation of sediment and adsorptive contaminant transport. Ideally multiple classes of sediment size and species of toxics need to be represented in a single analytical framework to efficiently address the different settling velocity of different classes of sediment as well as their different capability of adsorbing toxics.
- Representation of all contributing sources (legacy and active)
  - Legacy sources include contaminated sediment in the bay.
  - Active sources include storm-induced watershed contributions and other shoreline activities.
- Consideration of variable meteorological and hydrologic regimes

2.2 Modeling Options

Numerical modeling was identified to be the most appropriate approach to meet the analytical requirements identified above. A range of numerical models are available, but only selected models are capable of directly addressing the analytical requirements. Models that are capable of meeting the project needs fall into the receiving water model category. Three general groupings within this category are (1) hydrodynamic models, (2) water quality models, and (3) combined hydrodynamic and water quality models.

Hydrodynamic models or models with hydrodynamic components are capable of predicting advective and diffusive transport and water column-sediment bed interface stress. Hydrodynamic representation is necessary to accurately simulate the movement of water as well as the deposition and erosion of sediment and sediment-borne contaminants in a tidally-influenced system.

Water quality models generally focus on representation of contaminant transport. Adsorptive contaminant transport is the movement of adsorptive contaminants in both the water column and the sediment bed. Water column processes include advective and diffusive transport of both the dissolved and particulate (sediment adsorbed) phases as well as settling of the particulate phase. Transport within the bed should include pore water diffusion of the dissolved phase and mixing and burial of the particulate phase. Exchange processes across the water column bed interface should include deposition and erosion of the particulate phase contaminant with associated entrainment and expulsion of pore water and dissolved phases contaminants, and porewater-water column diffusion of dissolved phase contaminants. For most applications, including TMDL development, equilibrium partitioning is assumed to be an acceptable method of defining the phase distribution. Although more sophisticated representation of the kinetic process of adsorption and desorption is potentially more accurate, it is generally impractical due to the absence of sufficient field and laboratory data.

A number of models that simulate both hydrodynamics and water quality are available. Some of these models also simulate sediment transport, which is critical to accurately simulate the transport, settling, deposition, and erosion of sediment. Ideally a sediment transport model should be able to simulate multiple sediment (size) classes that have both cohesive and noncohesive properties. This enables adsorptive contaminants and their variable sorptive properties to be most accurately represented. Sediment transport simulation should also include settling and representation of sediment bed exchange characteristics. The sediment bed mass should be conserved, and variability throughout the depth of bed sediment should be represented.

Three potential modeling approaches to address TMDL development in San Diego Bay were identified and evaluated. These approaches and their corresponding advantages and disadvantages are described below.

2.2.1 Approach 1: Coarse Grid WASP Models

This approach involves developing coarse grid WASP models for each of the five impaired areas. WASP is USEPA's Water Quality Analysis Simulation Program. The WASP-TOXI module simulates toxic contaminants. The approach assumes that toxic pollution contributed by the incoming creeks is highly localized and does not extend well into the bay. Therefore isolated WASP models can be developed for each of these impaired areas. The WASP models would represent water quality at the mouths of each incoming creek. A very coarse spatial resolution (i.e., coarse grid) would be used such that each model only contains several linked boxes.

Advantages of this approach include:

- WASP is a widely accepted, public domain model supported by EPA. Therefore it has been well-tested and frequently used for modeling toxics fate and transport.
- Model run-times for coarse grid WASP models are very short. Thus, the long-term impact of load reductions (e.g., from the watershed) on sediment toxicity can be readily evaluated.

Limitations of this approach include:

- WASP is not capable of simulating hydrodynamics, therefore mass transport of both sediment and toxics cannot be accurately simulated. This is a concern due to the complex interactions that occur in the presence of storm contributions and tidal impacts.
- During storm periods the sediment and toxics from the creeks are transported to a much larger area than the mouths of the creeks. Assuming that only localized impacts occur around the mouths may introduce significant boundary condition errors.
- Using a coarse grid may induce significant spurious mixing (also referred to as numerical diffusion) and potentially cause unrealistic model predictions.
- WASP's sediment transport module relies only on static secondary transport parameters such as deposition and resuspension rates. Ideally these should change with environmental conditions.
- WASP only simulates up to three toxic species. For locations where more than three toxics are listed, such as at Switzer Creek, two separate models are necessary. This makes the modeling process cumbersome.
- WASP only simulates four layers of bed sediment. Higher resolution may be necessary to accurately represent a thick sediment bed in some areas.

2.2.2 Approach 2: Fine Grid WASP Models Linked to CH3D

The second approach involves developing fine grid (high resolution) WASP models for each of the impaired areas. These models would cover larger areas than the mouths to more accurately account for stormwater-open boundary interactions. To address the transport of sediment and toxics, the CH3D model developed by the Navy would be used as the hydrodynamic model. CH3D is the Curvilinear-grid Hydrodynamics model in 3 Dimensions, a hydrodynamic model originally developed by Dr. Y. Peter Sheng and used by various agencies. This approach would thus include one hydrodynamic model and five independent toxic models. External linkage would enable simulation of fate and transport of toxics in the water column.

Advantages of this approach include:

- WASP is a widely accepted, public domain model supported by EPA. Therefore it has been well-tested and frequently used for modeling toxics fate and transport.
- WASP models, even with fine grid resolution, can run relatively fast. Thus it is possible to evaluate long-term scenarios (assuming the local hydrodynamic information can be generated equally efficiently).
- More accurate mass transport simulation in WASP is possible due to linkage with an external hydrodynamic model (CH3D).

Limitations of this approach include:

- Externally linking the CH3D model with fine grid WASP models is a cumbersome process. Significant effort would be required to develop a linkage interface and address model instability resulting from model linkage.
- Externally linking CH3D and WASP requires storing hydrodynamic information in an external file for each of the WASP models. This poses a problem when long-term simulations are implemented because very large external hydrodynamic output files will be generated resulting in storage problems or significantly slow run times.
- The Navy's CH3D model covers all of San Diego Bay. Therefore the simulation time is very long, particularly for long time periods. An extended time period is required to drive the sediment transport and toxic modeling of the five local areas for TMDL development.
- CH3D and WASP use different numerical schemes. Inconsistent model predictions can result.
- WASP's sediment transport module relies only on static secondary transport parameters such as deposition and resuspension rates. Ideally these should change with environmental conditions. While CH3D predictions may help to overcome this limitation, significant effort would be necessary to develop the corresponding program.
- WASP only simulates up to three toxic species. For locations where more than three toxics are listed, two separate models are necessary.
- WASP only simulates four layers of bed sediment. Higher resolution may be necessary to accurately represent a thick sediment bed in some areas.

2.2.3 Approach 3: EFDC Model Based on CH3D Model Grid

The third approach involves developing a hydrodynamic model, using the EPA's Environmental Fluids Dynamic Code (EFDC), for the entire San Diego Bay. The model grid would be built based on the Navy's existing CH3D model grid. Multiple local EFDC models would also be developed to simulate hydrodynamics, sediment transport, and

toxics fate and transport in the five impaired areas. The bay-wide hydrodynamic model would be used to provide boundary conditions to the local models.

Advantages of this approach include:

- EFDC is a widely accepted, public domain model supported by EPA. Therefore it has been well-tested and frequently used for modeling toxics fate and transport.
- This approach would maximize use of the Navy's CH3D model without requiring significant effort to develop linkage programs.
- EFDC provides an integrated modeling framework allowing hydrodynamics, sediment transport, and toxics fate and transport to be simulated in one holistic system. Therefore, external linkages among the models are not necessary.
- Developing local models in addition to the entire bay model enables load reduction scenarios to be run significantly more efficiently for impaired areas (without running the entire bay model each time).
- Using the same modeling framework for the entire bay and the local areas ensures consistency between the applications.
- EFDC is capable of simulating any number of sediment classes and toxic species.
- EFDC is capable of simulating any number of sediment bed layers, thus it has the flexibility to represent a thick sediment bed, if necessary.
- Sediment transport is directly related to hydrodynamics in the model.

Limitations of this approach include:

- This approach is more computationally intensive than the first approach (although it is more efficient than the second approach) and can therefore result in long model run times.

2.3 Selected Approach

After considering the advantages and disadvantages associated with each of the approaches, the third approach (EFDC Model Based on CH3D Model Grid) was selected. This approach provides greater predictive capabilities and an associated anticipated higher level of accuracy. It offers a fully-integrated modeling system which can be more readily and efficiently applied and managed. The approach also provides the flexibility of directly simulating sediment transport, which is critical to accurate representation of toxic transport in the system. The approach will also maximize use of data to be collected in the future supporting the TMDL implementation process.

EFDC is a general purpose modeling package for simulating one-, two-, and three-dimensional flow, transport, and bio-geochemical processes in surface water systems including rivers, lakes, estuaries, reservoirs, wetlands, and coastal regions. The EFDC model was originally developed at the Virginia Institute of Marine Science for estuarine and coastal applications. This model is now being supported by USEPA and has been used extensively to support TMDL development throughout the country. In addition to hydrodynamic, salinity, and temperature transport simulation capabilities, EFDC is capable of simulating cohesive and non-cohesive sediment transport, near field and far field discharge dilution from multiple sources, eutrophication processes, the transport and fate of toxic contaminants in the water and sediment phases, and the transport and fate of various life stages of finfish and shellfish. The EFDC model has been extensively tested, documented, and applied to environmental studies world-wide by universities, governmental agencies, and environmental consulting firms.

The structure of the EFDC model includes four major modules: (1) a hydrodynamic sub-model, (2) a water quality sub-model, (3) a sediment transport sub-model, and (4) a toxics sub-model. The modeling effort for San Diego Bay included the hydrodynamic, sediment transport, and toxic sub-models.

3. Observational Data for Model Configuration and Calibration

Observational data for the hydrodynamic model falls within two general classes: data used for model configuration and data used for model calibration. Model configuration data includes the water body shoreline, bathymetry, data used for specifying hydrodynamic and salinity and temperature boundary conditions, atmospheric wind and thermal forcing, and inflows. Calibration data includes observations of hydrodynamic variables predicted by the modeling including water surface elevation, and salinity.

Table 3-1 summarizes the observational data used for model configuration and calibration. Data listed in Table 3-1 and used for the hydrodynamic model configuration and calibration are discussed later in this report. The available data being used for calibration are limited to the 2001 data for one tide gauge and 14 salinity track-line data sets collected in February 2001.

Table 3-1. Data used for model configuration and calibration

Data Type	Use	Source
Shoreline	Model Grid Generation	Navy CH3D Model Grid
Bathymetry	San Diego Bay Model Bathymetry Configuration	Navy CH3D Model Grid
Bathymetry	Local Model Bathymetry Configuration	Navy CH3D Model Grid
Wind Speed and Direction Records	Wind Forcing	National Climatic Data Center (NCDC) data
Atmospheric Temperature, Relative Humidity, Solar Radiation and Cloud Cover Records	Atmospheric Thermal Forcing	NCDC data
Salinity and Temperature Monitoring Data	Boundary condition forcing	Scripps Oceanography Station #95

4. Model Development

The selected modeling approach required development of a large-scale EFDC hydrodynamic model for San Diego Bay and five separate small-scale EFDC sediment and toxic simulation models for the impaired shoreline areas in the bay. This section describes key elements of the models.

4.1 Hydrodynamic Model for San Diego Bay

A hydrodynamic model was developed to simulate water circulation patterns in San Diego Bay. It was important to develop this model in order to provide accurate boundary conditions for the five local models representing the impaired areas. As noted above, the EFDC model was implemented.

Configuration of the EFDC model for San Diego Bay involved identifying and processing bathymetric data, developing model grids, defining boundary and initial conditions, and creating a linkage with the existing LSPC watershed model using lateral inputs. Boundary conditions are fixed conditions applied to the modeling system to drive the hydrodynamic simulation. Three types of boundary conditions were applied to the hydrodynamic model: open ocean, lateral flux (representing watershed contributions), and meteorological.

4.1.1 Grid Generation

The EFDC modeling domain for San Diego Bay includes the entire bay as well as a portion of the ocean just outside the mouth of the bay. The model grid was generated based on the CH3D grid provided by the Navy, with minor refinements in the B-street area. The final grid is comprised of 5,796 computational cells (Figure 4-1).

This set of grids, based on the Navy's original CH3D model grid, provided a high resolution representation of the entire San Diego Bay. The average resolution of the grid is approximately 100 meters, with finer resolution at the mouths of Paleta, Chollas, and Switzer creeks in order to resolve the near-shore feature at the areas of concern. The model was configured as a three-dimensional model, with 4 layers along the vertical axis to resolve vertical variability. Since water in San Diego Bay is generally not significantly stratified, a 4-layer representation was considered appropriate. The maximum and minimum cell widths in the grid are 250 meters and 18.1 meters, respectively. The maximum and minimum cell lengths are 362 meters and 18.8 meters, respectively. And cell depths range from 2.2 to 20.1 meters.

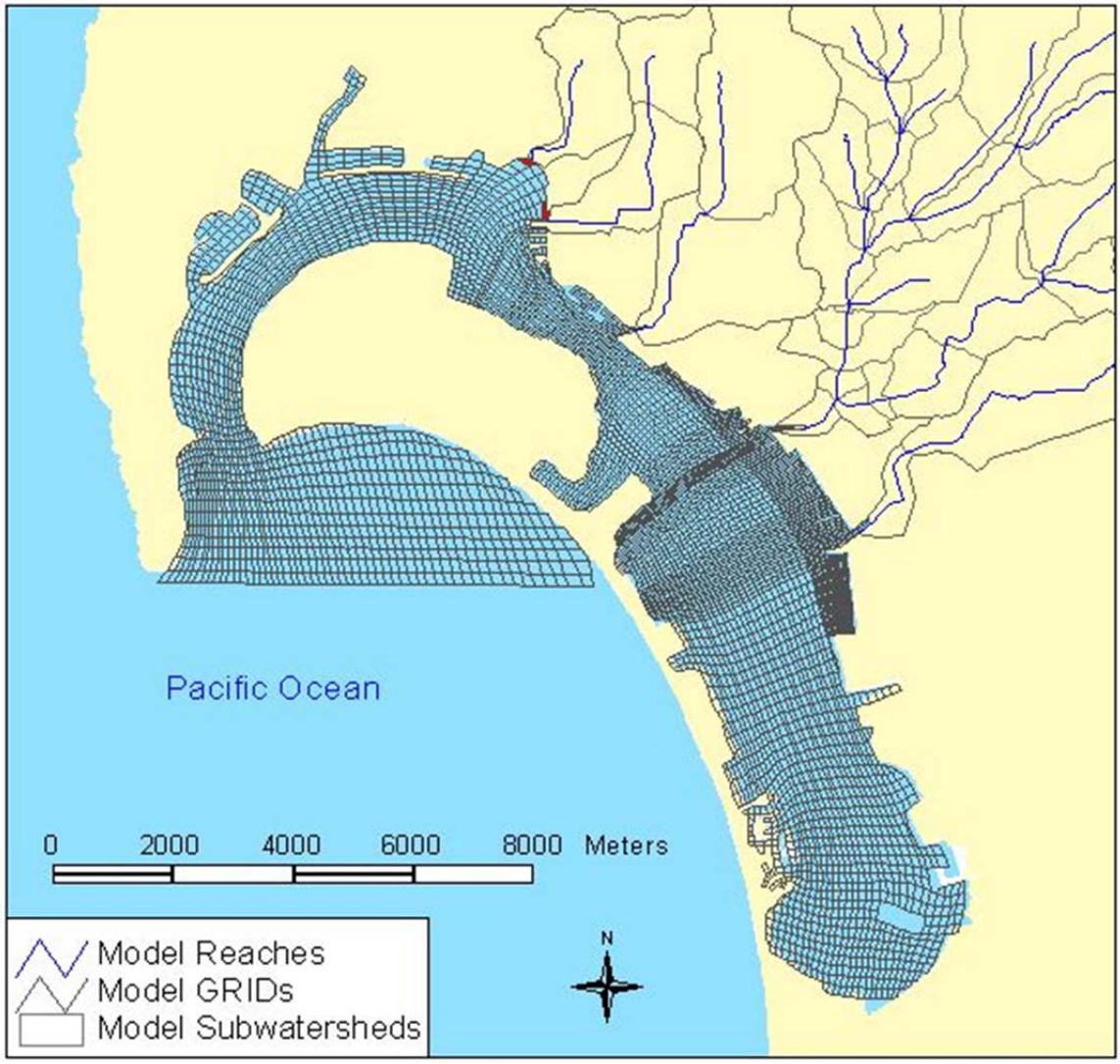

Figure 4-1. Computational grid for San Diego Bay hydrodynamic model.

4.1.2 Boundary Conditions

4.1.2.1 Open Ocean Boundary Conditions

The mouth of San Diego Bay opens to the Pacific Ocean, therefore, the model requires representation of an open ocean boundary. This boundary was represented by the 54 cells in the grid that extended farthest into the ocean. These cells were assigned time-variable water levels, temperature, and salinity.

Real-time hourly water level data were available from the National Oceanic and Atmospheric Administration Center for Operational Oceanographic Products and Services (NOAA-COOPS) for station #9410230, located in La Jolla, California. Data for this station were processed and an EFDC-compatible tidal time series dataset was created and applied to all the cells at the open boundary.

Two Scripps Institution of Oceanography stations with continuous surface temperature observations were used to obtain temperature for the open ocean boundary. The closest station to San Diego Bay is station #091 and is located 8.5 miles west of Point Loma; however, this station is a seasonal buoy and is operated from approximately February to August of each year. Station #095 is located 3.8 miles west of La Jolla and operates year-round. Temperature data from these two stations were compared for 180 overlapping days in 2001. The comparison resulted in good correlation between the two stations with an $R^2 = 0.92$. Therefore, temperature data from La Jolla (station #095) were selected to build the time series at the open ocean boundary on a daily basis for the simulation period. The time series was first built for the calibration period (discussed later as February to March, 2001) and then extended to the remainder of 2001.

A station operated by the Port of San Diego and located in San Diego Bay provided salinity data for the open ocean boundary. Continuous salinity observations were available from March 7, 2001 to December 13, 2001 and January 13, 2002 to February 7, 2002. The January to February 2002 data were used to fill the data gaps in for 2001.

4.1.2.2 Lateral Flux Boundary Conditions

The lateral flux boundary conditions include the inflow of water and associated temperature and salinity from the five subwatersheds draining into the five impaired areas in this study, including the mouths of the Downtown Anchorage, B Street/Broadway Piers, Switzer Creek, Chollas Creek, and Paleta Creek watersheds (see Figure 1-1). It was assumed that other creeks draining into the bay have no significant impact on the overall hydrodynamics in the bay. This assumption was adopted in the previous Navy modeling study (Chadwick et al. 2008) and is justified by the fact that in the San Diego Bay tidal flows dominate the incoming watershed flows. The locations of these inputs to the modeling grid were determined by overlaying the watershed boundaries (and stream coverage) with the model grid and identifying the corresponding grid cells.

Model simulation results from LSPC were available for flow; however, available monitoring data were used to represent temperature and salinity. Continuous surface temperature observations from NOAA station #9410170, located near G Street in the San Diego Bay watershed, were used to specify the temperature for the watershed inflows. Although the temperature of bay water can be different from the incoming tributary flows, temperature measurements for the incoming streams were not available. Since watershed flows account for a negligible portion of the total flow balance in the bay, the uncertainty associated with the inflow temperature values has a minimal impact on the model results. Salinity data for the inflows were also not available and were thus set to zero. This is also expected to have a negligible impact on the model results because the inflows account for such a small portion of the volume of the bay.

4.1.2.3 Meteorological Boundary Conditions

The meteorological boundary conditions are represented by time-variable solar radiation, wind speed and direction, air temperature, atmospheric pressure, relative humidity, and cloud cover. Five airway stations in close proximity to San Diego Bay were evaluated for potential inclusion in the model. The stations were evaluated based on their proximity to the modeling domain, period of record, parameters measured, and completeness of data. Data for 1990 to 2004 were obtained from the National Climatic Data Center (NCDC).

Results of the evaluation indicated that the Lindbergh Field Airway Station in San Diego was the most appropriate weather station (Figure 4-2) and it was used to create the meteorological file. This station had data for most of the required parameters, provided the most complete temporal data record, and is located in close proximity to San Diego Bay. Data for dry and wet bulb temperature, dew point temperature, relative humidity, wind speed, wind direction, sea level pressure, and sky conditions for 2001 were obtained for the Lindbergh Field station. Sky condition was converted to “percent cloud cover” and solar radiation was estimated by calculating the clear sky solar radiation using latitude and longitude and adjusting the values based on the estimated cloud cover.

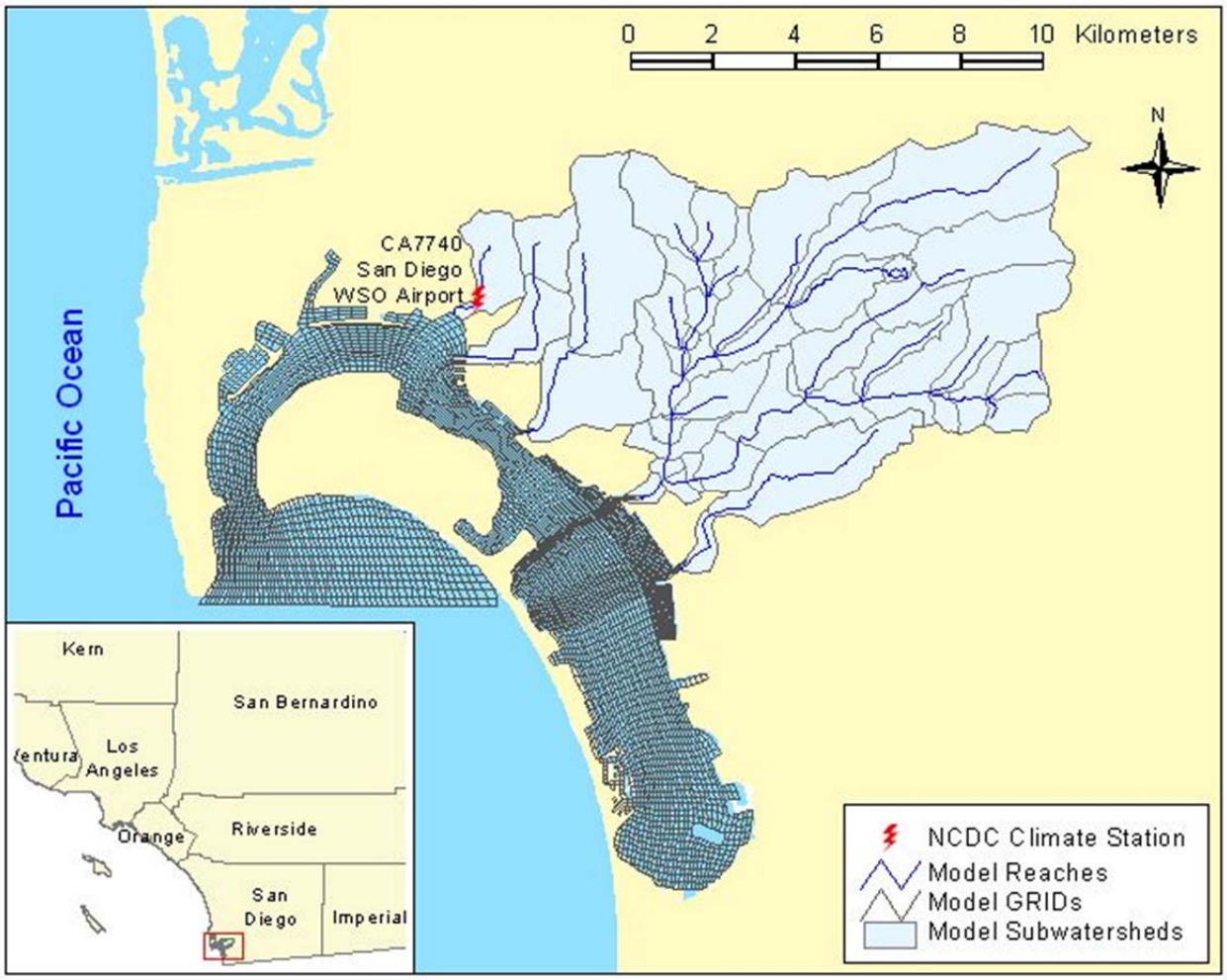

Figure 4-2. Location of the Lindbergh Field Airway Station in San Diego.

4.1.3 Initial Conditions

In hydrodynamic modeling, initial conditions provide a starting point for the model to progress through time. Initial temperature, salinity, flow velocity, and water depth values were specified for the entire domain of the model. Chadwick et al. (2008) reported that San Diego Bay can sufficiently dampen out the impact of the initial condition in about 48 hours; therefore, it is reasonable to specify the initial conditions roughly based on data or professional judgment, and let the model “spin-up” to remove any impacts.

A uniform temperature of 15ºC and a salinity of 33 ppt were included as initial conditions throughout the water column. This temperature was verified using data from Scripps Institution of Oceanography stations #091 and #095 and was determined reasonable considering that the models began in early February. The initial water velocity was set to 0.0 meters per second (m/s), and the initial water surface elevation was 0.0 meters above mean sea level.

4.2 Sediment Transport and Toxics Models for Impaired Areas

Instead of developing a bay-wide sediment transport and toxics modeling system based on the hydrodynamic model, individual sediment transport and toxic models were developed for the impaired areas. This was done to focus on the depositional zones at the mouths of the creeks and to reduce computational time. Four separate models were constructed. Paleta, Chollas, and Switzer Creek mouths each had their own model and the Downtown Anchorage and B Street/Broadway Pier areas were covered in a single modeling domain because of their proximity to each other.

Sediment and contaminant transport formulations in the EFDC model are documented by Tetra Tech (2007). Both fine, cohesive sediment and noncohesive sand are simulated within EFDC. Particulate organic material is assumed to be associated with the fine sediment class. Two-phase equilibrium partitioning is used to represent adsorption of the metals and organics to the different sediment classes.

The EFDC model simulates the transport and fate in both the water column and sediment bed. Water column transport includes advection, diffusion, and settling for sediment and sediment adsorbed contaminants. The sediment bed is represented using multiple layers with internal transport of contaminants by pore water advection and diffusion. Sediment and water is exchanged between the water column and bed by deposition and erosion, with corresponding exchange of adsorbed and dissolved contaminants. Dissolved phase contaminants are also exchanged by diffusion between bed pore water and the overlying water column.

The following sections describe key aspects of model development and application.

4.2.1 Grid Generation

The computational grids of the four local models were developed based on the bathymetry of the Navy's CH3D grid (Figure 4-3). For each model, the computational domain was constructed to be significantly larger than the impaired area at the mouth of each inflowing tributary. This ensures that the open boundary for each model is located far enough away from the freshwater inflows to avoid potential boundary errors during storm events. The grids were tested to ensure that during storm events sediment concentrations were low at cells close to the boundaries even though they were very high at the tributary mouths. Model grids were also constructed to align with the bay-wide hydrodynamic model for a seamless linkage. Figures 4-4 through 4-7 show the computational grids in relation to the whole bay grid.

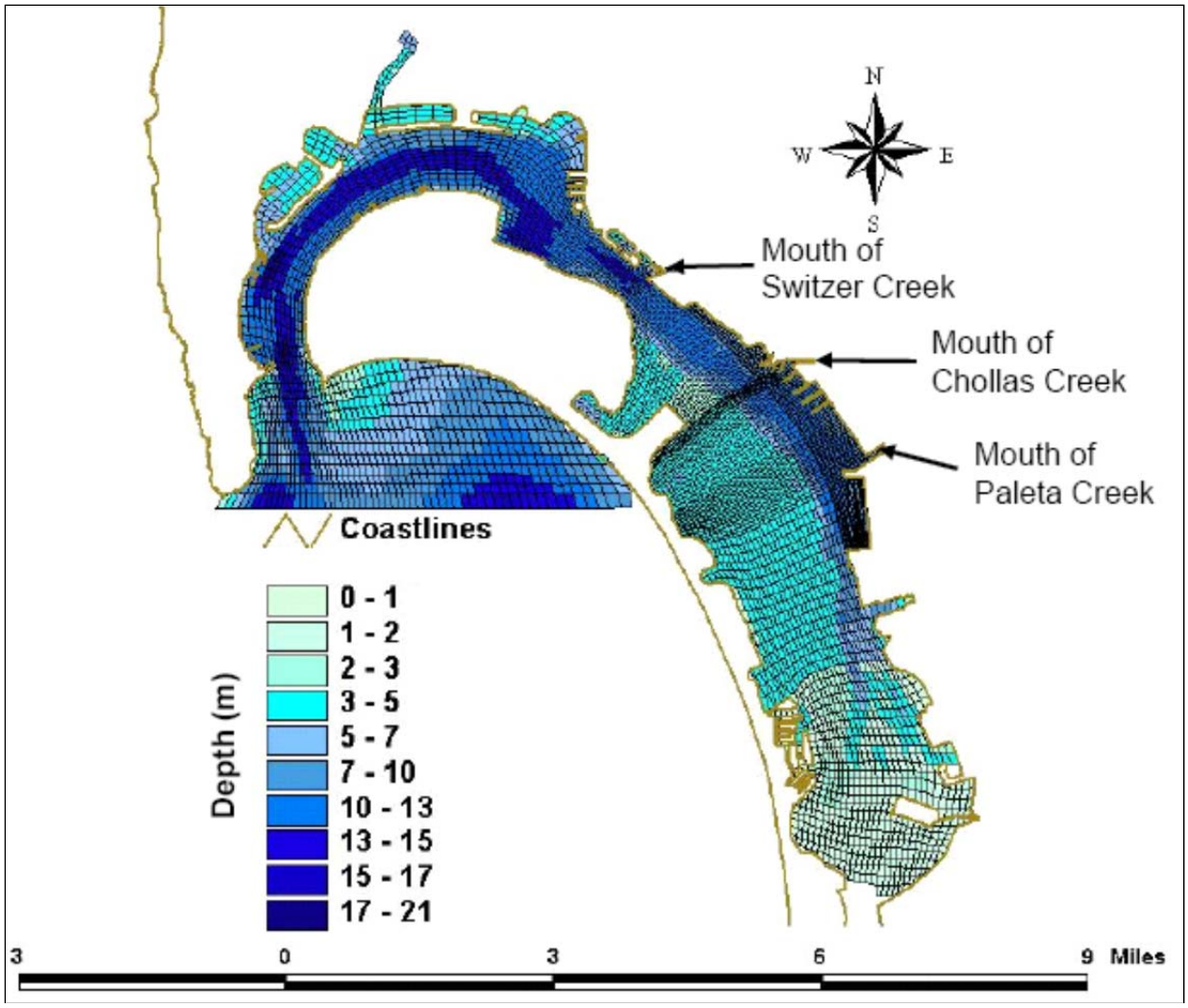

Figure 4-3. Bathymetry of the San Diego Bay.

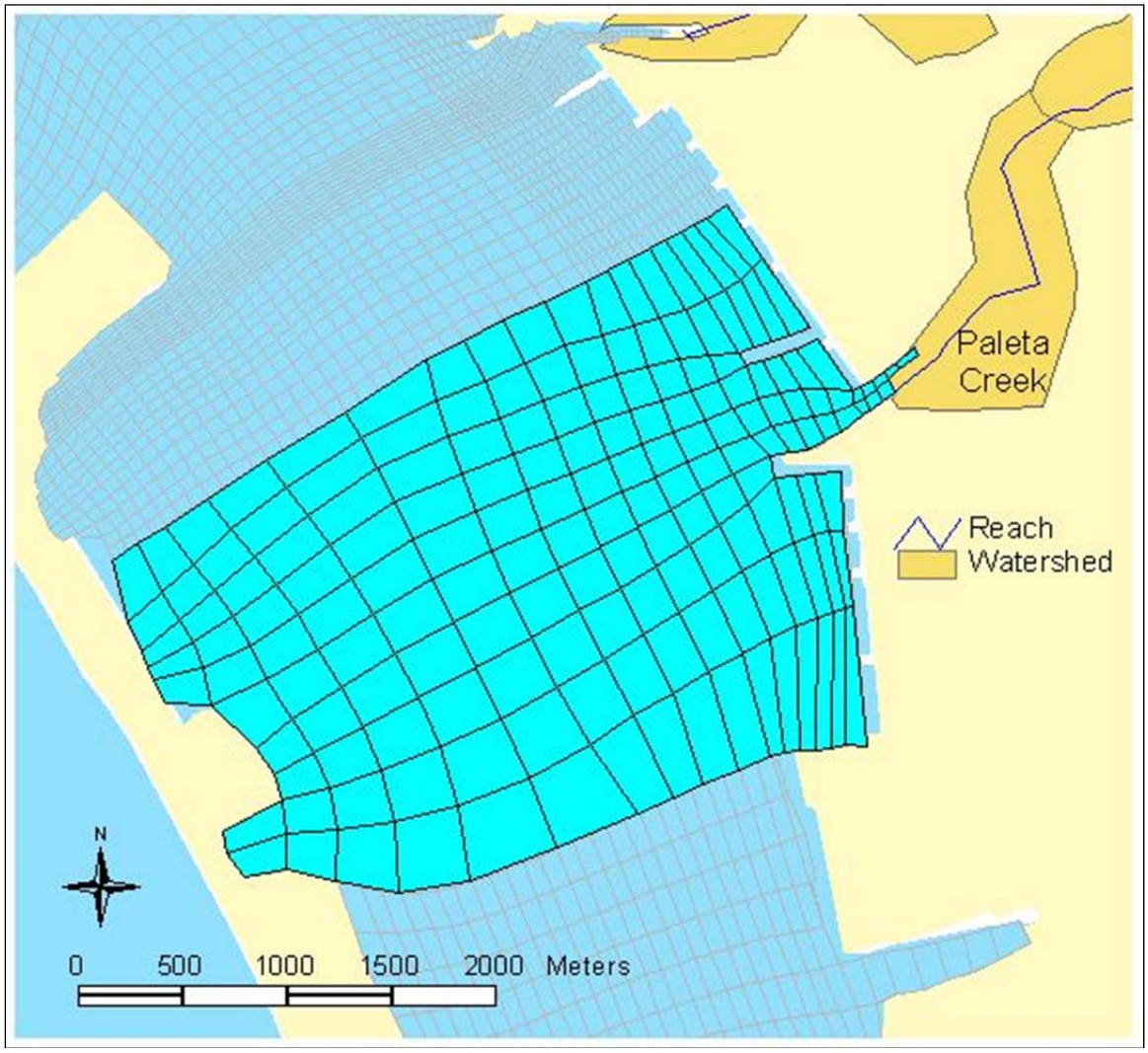

Figure 4-4. EFDC grid for the mouth of Paleta Creek.

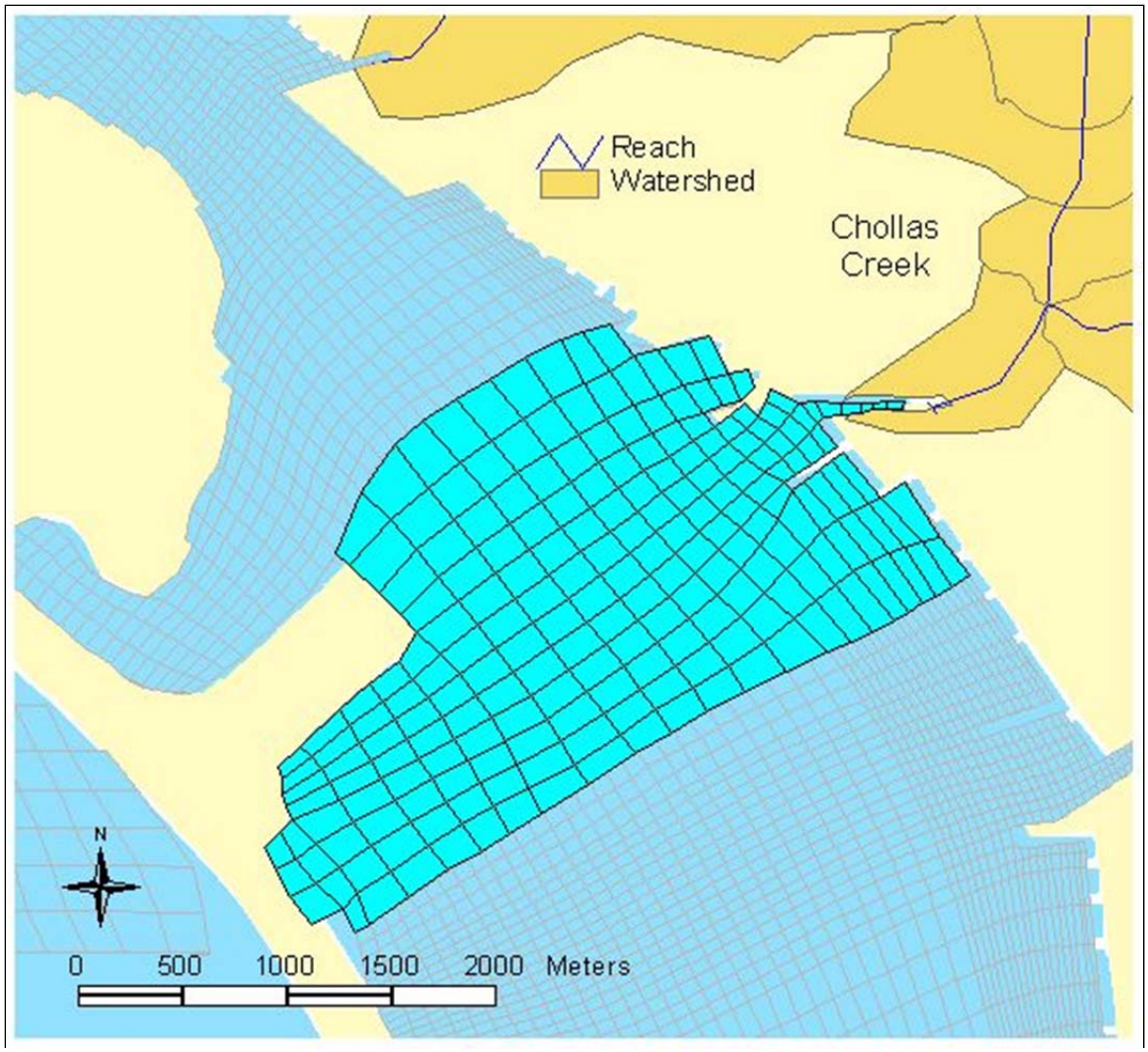

Figure 4-5. EFDC grid for the mouth of Chollas Creek.

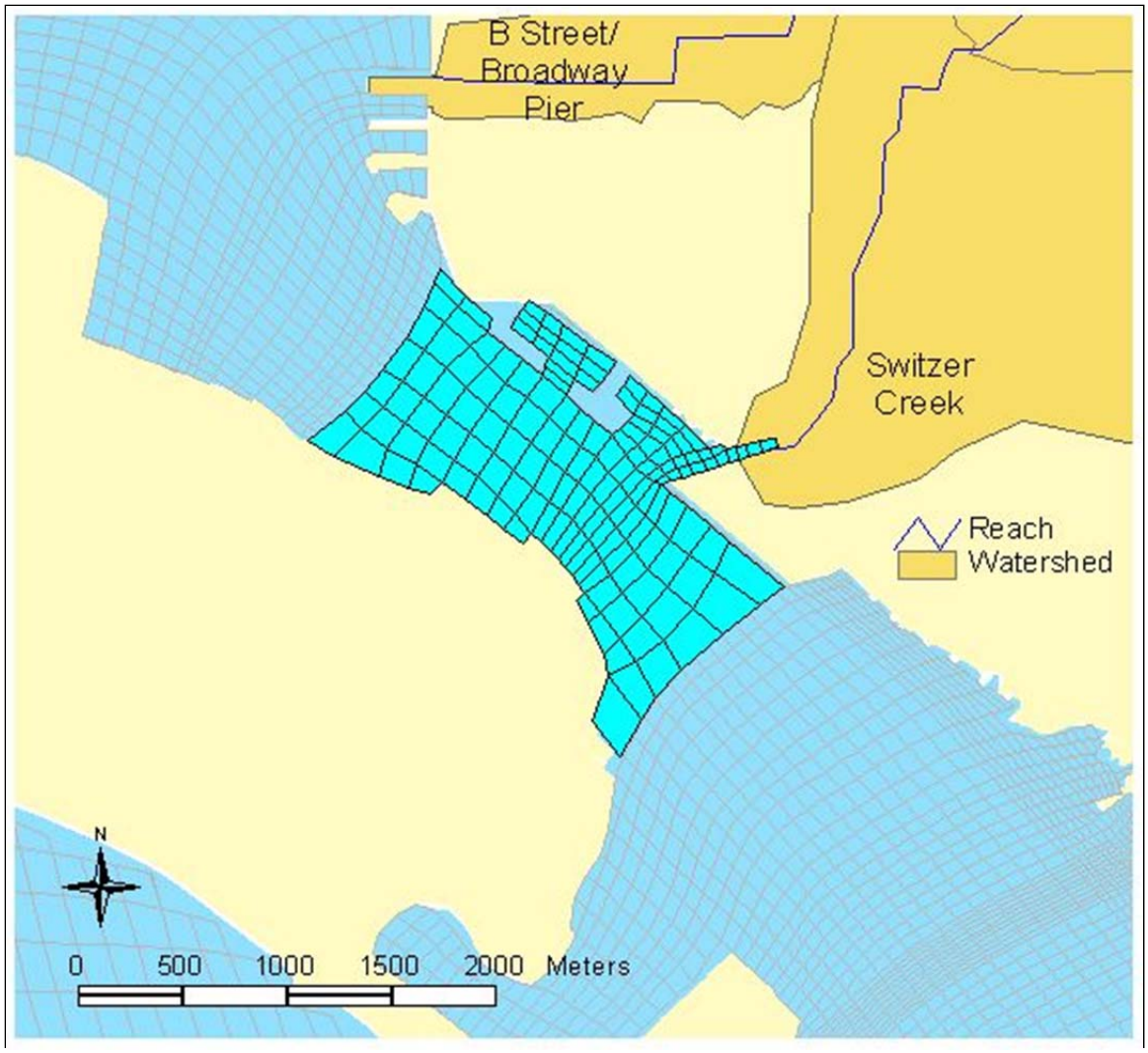

Figure 4-6. EFDC grid for the mouth of Switzer Creek.

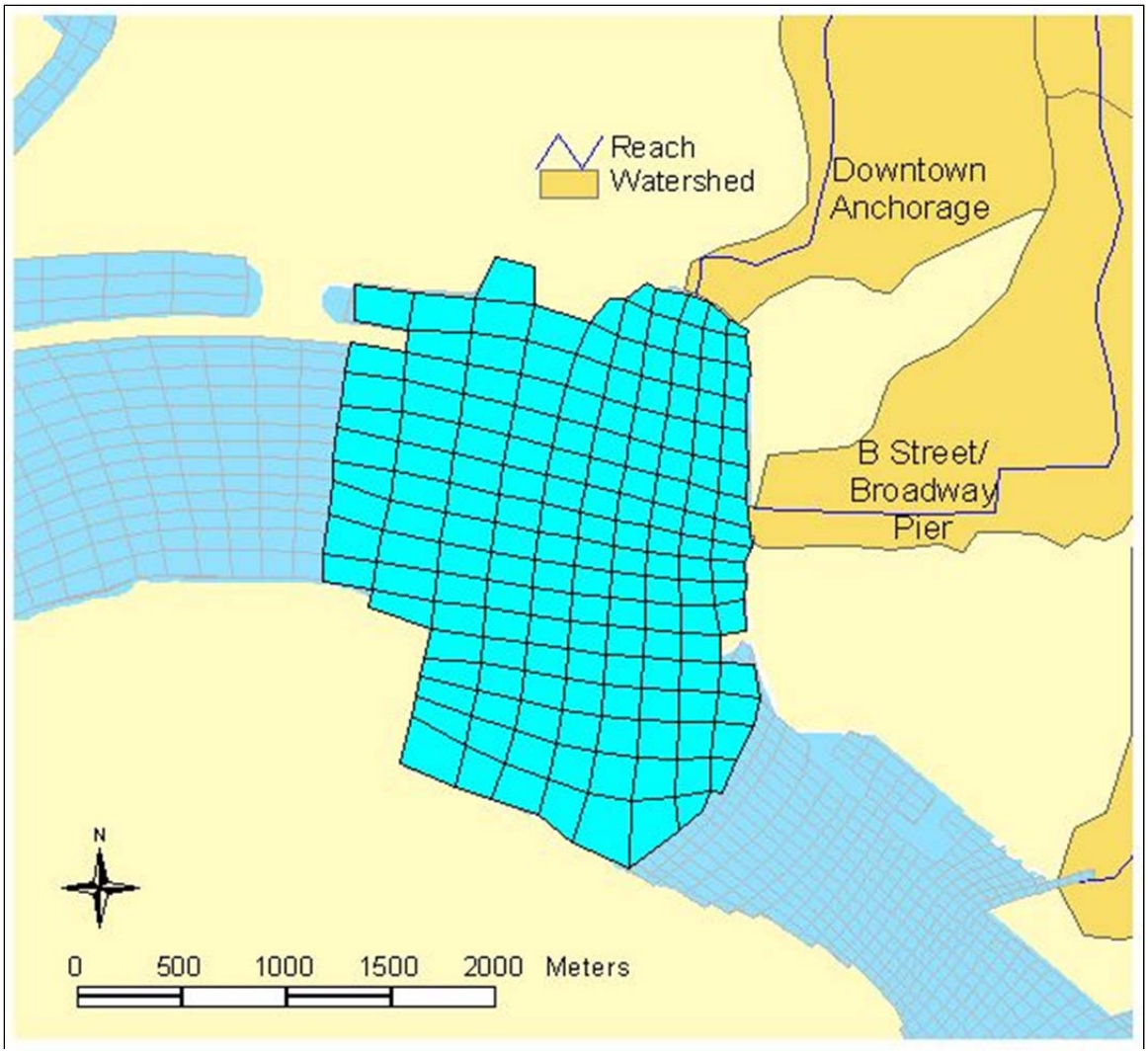

Figure 4-7. EFDC grid for the Downtown Anchorage and B Street/Broadway Pier areas.

4.2.2 State Variables

Each of the sediment transport models was configured to simulate two cohesive sediment classes: clay (with a diameter < 3.9 micrometers) and silt (with a diameter > 3.9 micrometers and < 63 micrometers); and one non-cohesive sediment class: sand (with a diameter > 63 micrometers). The sediment bed was configured to have a maximum of six layers, with a maximum layer thickness of 20 centimeters. This allowed the model to represent up to 1.2 meters of active bed, which was deemed sufficient for representing the bed dynamics in San Diego Bay.

The toxic models were each configured to simulate the contaminants identified by the San Diego Water Board to address sediment impairments. The Paleta and Chollas Creek models were configured to simulate total PAHs (TPAH), total PCBs (TPCB), and total chlordane (TCHLOR). The Switzer Creek model was configured to simulate TPAH, TPCB, TCHLOR, and total lindane. The Downtown Anchorage and B-street model was

configured to simulate TPAH, TPCB, TCHLOR, and total zinc. The transport of these contaminants is simulated in association with the sediment transport model, because they tend to adsorb to suspended solids, settle into the bed, and re-enter the water column due to resuspension of bed sediment.

4.2.3 Boundary Conditions

4.2.3.1 Open Ocean Boundary Conditions

The east and west sides of each of the model grids are open to tidal water influence while the north and south boundaries are constrained by land. The east and west open cells were configured as open boundaries. They were assigned time-variable water levels, temperature, and salinity values based on predicted water levels, temperature, and salinity from the calibrated bay-wide hydrodynamic model. Sediment concentrations at these boundary cells were determined through the iterative calibration processes (which is described in the calibration section).

Toxics concentrations at these boundary cells were derived from historical data. Katz (1998) reported measured water column TPAH concentrations at 13 stations during two sampling events during July and November 1997. Due to the sparseness of data, no apparent spatial pattern was identified for the data at the 13 stations; therefore, it was not attempted to apply spatial variable data to each local model. Instead, the mean of all these sampling values (81.9 ng/L) was applied to represent constant boundary conditions at each of the local toxic models to represent the concentration in the boundary cells. No water column data were available for the other contaminants, therefore, they were derived using TPAH concentrations and the ratio of concentrations between TPAH and each contaminant in the bed. Sediment bed contaminant data were summarized in SCCWRP and SPAWAR (2005). Based on these data, an average ratio between TPCB and TPAH was calculated as 0.028. Similarly, the ratio between TCHLOR and TPAH was calculated as 0.002, and the ration between zinc and TPAH was 77.

For the Switzer Creek model, the lindane boundary condition also needed to be specified. Anderson et al. (2004) noted a number of lindane measurements in the sediment bed, however all of the samples were found to be below detection limits. For these non-detect samples, the concentration of lindane was assumed to be half of the detection limit. This resulted in a bed sediment lindane concentration of 0.5 ng/g. Based on this lindane concentration and the TCHLOR concentration in the same sample, the ratio between lindane and TCHLOR was calculated to be 0.088.

Boundary concentrations for all five toxic contaminants are summarized in Table 4-1. It was assumed that the boundary conditions represent the background condition in the bay. They were set to be constant over time and uniform in space.

Table 4-1. Toxic boundary conditions for the local models

	TPAH(ng/L)	TPCB(ng/L)	TCHLOR(ng/L)	Zinc (µg/L)	Lindane (ng/L)
Paleta Model	81.9	2.29	0.16	6.3	0.014
Chollas Model	81.9	2.29	0.16	6.3	0.014
Switzer Model	81.9	2.29	0.16	6.3	0.014
Downtown/B-Street Model	81.9	2.29	0.16	6.3	0.014

4.2.3.2 Lateral Boundary Conditions

The watershed loading results used to formulate lateral boundary conditions for the EFDC models were based on a combination of LSPC model predicted flow and TSS concentrations and assumptions based on mean organics and TSS concentrations observed within Chollas, Paleta, and Switzer Creeks, as reported in the Watershed Modeling for Simulation of Loadings to San Diego Bay (Watershed Modeling Report), prepared by Tetra Tech, Inc. (2008). The watershed loading from each of the inflowing tributaries was assigned as a lateral boundary condition for the corresponding models. Specifically, flow, suspended solids, and toxic concentrations for the watershed inflows were extracted from the LSPC model output data files and formatted for EFDC model compatibility. The flows and toxic contaminant concentrations were directly extracted; however, additional effort was required to divide the TSS concentration among the three modeled sediment classes in EFDC (i.e., clay, silt, and sand). Chadwick et al. (2008) reported sediment ratios for Paleta Creek, Chollas Creek, and Switzer Creek. These ratios were used as the basis for TSS division. Sediment ratios were not available for B-Street and Downtown Anchorage watershed inflows, therefore the division was based on the composition of bed sediments in these areas. This was measured by the CRG Marine Laboratory in 2003. Table 4-2 summarizes the ratios used. These ratios were assigned as constant values for all storms though in reality the ratios may differ from one storm to the next. Insufficient data were available to accurately model time-variable ratios with the watershed models.

Table 4-2. Fraction of each sediment size class in the watershed inflows

	Clay (%)	Silt (%)	Sand (%)
Paleta Creek	18	35	47
Chollas Creek	26	43	31
Switzer Creek	10	43	47
B-Street Inflow	15	49	36
Downtown Inflow	15	54	31

4.2.4 Initial Conditions

For each of the four models a uniform temperature of 15ºC and a salinity of 33 ppt were included as the initial conditions throughout the water column. The initial velocity was set to 0.0 m/s, and the water surface elevation was set to 0.0 meters above mean low sea level.

The initial sediment concentration in the water column was set to 1.0 mg/L for each of the three classes. For toxics, the water column concentrations were set to be the same as the background concentrations, as indicated in Table 4-1.

Initial bed sediment compositions for the Paleta, Chollas, and Downtown/B-Street models were specified based on data reported in SCCWRP and SPAWAR (2005). These data were collected in 2001, which corresponds to the modeling period selected. Since no data were available to set the initial bed composition at the Switzer Creek mouth for 2001, the data for 2003 collected by the CRG Marine Lab were used. Initial bed toxic concentrations for the Paleta and Chollas models were specified based on data reported in SCCWRP and SPAWAR (2005). Initial conditions for the Switzer and Downtown/B-Street models were specified based on data reported in Anderson et al. (2004, 2005). Since data were available at multiple locations at the mouths of the creeks, the initial bed toxic conditions were specified on a spatially-variable basis. Where data were available, the values were directly applied. Where data were not available, conditions were set using the minimum values of the data available at the hot spots. The initial bed condition for these cells does not have a significant impact on the simulation results, because these locations are generally outside the area of the incoming tributary mouths and are generally deep. Therefore, resuspension is not expected to occur and contribute significantly to re-distribution of toxics among cells.

5. Model Calibration and Validation

This section of the report discusses the calibration and validation of the hydrodynamic model, sediment transport model, and toxic model. Calibration refers to the adjustment or fine-tuning of modeling parameters to reproduce observations. After the model was configured, model calibration and validation were performed. This was a two-phase process, with hydrodynamic model calibration and validation completed before evaluating the performance of sediment transport and toxic modeling.

5.1 Hydrodynamic Model Calibration and Validation

The hydrodynamic model of the bay was calibrated using observed surface elevation data from the bay. Subsequently, model validation was performed to test the model's capability to represent local freshwater-salt water interactions at the mouths of incoming creeks, without further adjustment of parameters.

5.1.1 Surface Water Elevation

February 6, 2001 through March 6, 2001 was selected as the simulation/ calibration time period. This period covers the Navy data collection event around Paleta and Chollas Creeks on February 13, 2001, and extends 7 days earlier to allow the model to stabilize before receiving watershed storm-driven inputs. To validate model performance, the simulated period was extended to cover all of 2001 after the calibration was completed.

Model-computed water surface elevations were compared with hourly real-time data from NOAA-COOPS station #9410170, located near G Street. The primary parameters subject to adjustment for the hydrodynamic calibration were bottom roughness height and bathymetry. Since the bathymetry and grid for the current model is based on the previously-calibrated CH3D model developed by the Navy, it was not anticipated that significant calibration effort would be necessary. Only a few iterations were required to allow the model to achieve stable solution, with a bottom roughness height of 0.01 meter. The simulated water surface elevation for the calibration period is plotted against the observed data (Figure A-1 in Appendix A). As shown, the simulated elevation matches the observed elevation very well, indicating a reasonable representation of tidally-influenced water movement in the bay.

To validate the performance of the hydrodynamic model, the calibrated model was extended in time to cover the remaining period of 2001 (i.e., March to December, 2001). No additional parameter adjustment was required for this simulation, and the resulting surface elevation was again plotted against the observed data at the same location for three additional, randomly chosen months. These comparisons are shown in Figures A-2 through A-4 in Appendix A.

The simulated elevation matches the observed data very well (see Figures A-3 and A-4 in Appendix A), which suggests that calibration of the hydrodynamic model is reliable for periods beyond the calibration period.

5.1.2 Freshwater-Saltwater Interaction at the Watershed Mouths

In addition to the calibration and validation for water surface elevation, model performance was also evaluated by checking the ability of the model to predict local freshwater-saltwater interaction at the mouths of the watersheds. In February 2001, the Navy conducted a survey before and during a storm event, and multiple trackline data were collected at the mouths of Chollas and Paleta Creeks and described in Chadwick et al. (2008). Figure 5-1 presents the locations of the trackline data collection sites. Since the trackline data were collected at different times and locations along the tracklines, the data are not suitable for conducting a time series type of model evaluation. Therefore, an approach that compares general statistics for model-simulated results against those for observed data along the tracklines was adopted. For this effort, five statistical measures were used to compare the model results against the data: minimum, 25th percentile, median, 75th percentile, and maximum.

At the Chollas Creek mouth, seven tracklines were surveyed for seven different time intervals. The first trackline was sampled on February 12, 2001, which was several hours before a storm started. The other six tracklines were sampled during and after the storm event. To implement the data-model comparison, the model results at the cells along the tracklines were extracted for the corresponding time interval of each trackline. Then, corresponding statistics were calculated and compared with those for the data. Figures B-1 through B-6 in Appendix B show the comparisons. Note in these figures, the “index” on the horizontal axis represents the five statistics, from 1 to 5 (representing minimum, 25th percentile, median, 75th percentile, and maximum). As shown, the model-simulated salinity at the mouth of Chollas Creek matches the observed temporal-spatial distribution well.

A similar comparison was made for the mouth of Paleta Creek at seven tracklines (Figures B-7 through B-13 in Appendix B). The model results match the data at these temporal-spatial locations as well. These results again suggest that the hydrodynamic model is well calibrated, and is capable of representing the interaction between freshwater and tidal water at the mouths of the watersheds. Although there are some deviations between the observed data and model results, these discrepancies can largely be explained by the fact that the hydrodynamic model is driven by inputs from the LSPC watershed model, which is not exactly the same as the real values at the inflow locations. Therefore, the simulated salinity plume is not expected to be exactly the same as the observed.

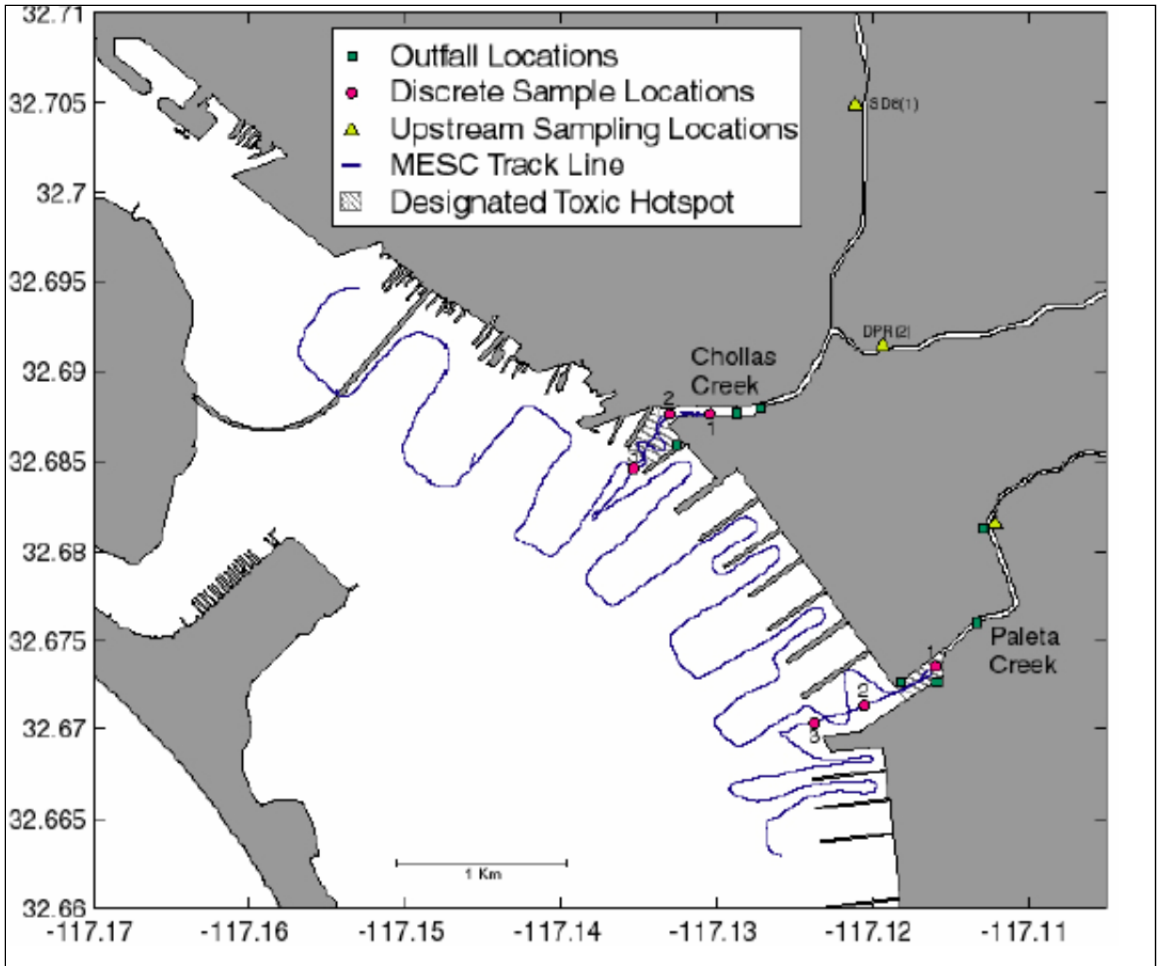

Figure 5-1. Location of the trackline data collection sites (the solid blue line).

5.2 Sediment Transport Model Calibration

The sediment transport models were calibrated using the trackline data at the mouths of Paleta and Chollas Creeks collected in February 2001. The model was run from the same start date as for the bay-wide hydrodynamic model, i.e., February 6, 2001. It was run for 12 days to cover the period of data collection. The trackline data collection was conducted for the same event as for the salinity data used in the hydrodynamic calibration. The same data-model comparison approach used for the hydrodynamic calibration was adopted to evaluate the sediment transport model.

Model calibration was conducted by comparing the simulated TSS plume versus the observed TSS plume and adjusting related parameters until a reasonable match between the model results and data were achieved. The primary parameters governing sediment transport include settling velocity, critical bottom shear stress of deposition, and critical bottom shear stress of resuspension. In addition, the open boundary sediment concentration was adjusted during the calibration process to obtain reasonable estimates of the background conditions.

For this study, the settling velocities for clay and silt were set to be the same as in Chadwick et al. (2008): 0.048 m/day for clay and 9.5 m/day for silt. The settling velocity for sand is internally calculated based on the assumed median size of 75 micrometers. This is consistent with the value used in Chadwick et al. (2008).

There were no measured critical shear stress data for both deposition and resuspension or reference values from previous studies available for San Diego Bay. Hwang and Mehta (1989) reported that the critical shear stress for cohesive sediment resuspension varied from 0.125 N/m2 to 0.525 N/m2. Ji et al (2002) also verified that values within this range worked well in their study of cohesive sediment transport modeling. For this study, a value of 0.125 N/m2 was used. The model generated reasonable results (within a range found in the literature). The critical bottom shear stress for deposition was set to the value of 0.1 N/m2 based on the rate applied in the previous study (Ji et al. 2002).

The boundary concentration for each of the three sediment classes was initially set to 1.0 mg/L. It was refined iteratively through the calibration process until the model reproduced reasonable results in comparison with the data. The final values were 0.001 mg/L, which indicates that the background water in the San Diego Bay is very clear and almost free of suspended solids. This value is justifiable because data show that before storms the TSS concentration in the mouth area of Chollas and Paleta Creeks is lower than 0.002 mg/L.

The simulated TSS results were extracted and compared with the observed trackline data again in terms of the minimum, 25th percentile, median, 75th percentile, and maximum for each trackline. Figures C-1 through C-7 in Appendix C compare the model results and data for the mouth of Chollas Creek, while Figures C-8 through C-14 in Appendix C compare model results and data for the mouth of Paleta Creek.

In general, the model results reproduce the observations. Timing associated with the sediment plume's entrance and dissipation during and after the storm, is accurately represented. The model predicts the range of observed TSS concentrations well, i.e., the minimum and maximum, and reasonably simulates the 25th percentile and median values. Prediction of the 75th percentile level is not as accurate as for the other percentiles.

Discrepancies between the model predictions and data may be caused by several factors. First, storm inputs to the bay grid were generated by the LSPC watershed model rather than actual data. Any discrepancy between the watershed predictions and actual data would propagate into the bay. Second, partitioning of sediment into sand, silt, and clay was based on a fixed ratio while in reality this ratio may vary over the course of a storm. Third, the model simplifies sediment representation using three discrete size classes and characterizes their behavior based on the representative class diameter. In reality, sediment exhibits much less homogeneity and more variable behavior.

5.3 Toxic Model Calibration

The performance of toxic models is affected by the following factors:

- *Hydrodynamics.* Accurate simulation of toxics transport relies heavily on a reasonable representation of water circulation in the modeling domain.
- *Sediment transport.* Toxics, including organics and metals, have the tendency of adsorbing to sediment and being transported with sediment through water column advection/dispersion, settling/deposition, and resuspension.
- *Adsorption parameters.* Adsorption and desorption of toxics to sediment particles are simulated using the assumption of local equilibrium. This assumes that adsorption and desorption occur relatively fast and therefore equilibrium can be achieved in a short period of time - from a few hours to a day (Chapra 1997; Ji et al. 2002; Hayter 2006; Tetra Tech 2007). Although this assumption is sometimes brought into question, it is widely applied to TMDL development and is the most practical approach to toxics modeling. A more detailed representation of adsorption and desorption kinetic process may be desirable; however, insufficient data are frequently available (as is the case with San Diego Bay). The partitioning coefficient of each toxic contaminant needs to be specified to implement the local equilibrium assumption in toxic modeling. The partitioning coefficient characterizes the tendency of the contaminant to adsorb to specific classes of sediment.
- *Chemical process in the water column and sediment bed.* Toxics can undergo various chemical transformations and be affected by loss processes in the water column and sediment. For conservative purposes associated with TMDL development, the chemical loss terms have been assumed to be 0.0 for this modeling effort.

The first two items noted above were addressed through the hydrodynamic and sediment transport model development and calibration process. The fourth item is a simple conservative assumption. To address the third item, which entails setting the partitioning coefficients, site-specific data were used. The partitioning coefficients in the sediment bed were derived based on data reported in Chadwick et al. (1999). Data were collected to characterize the sediment toxicity in several locations near the Naval Station (Chadwick et al. 1999). The data included particulate concentrations of TPAH, TPCB, and zinc in the bed, porewater concentrations of these contaminants, and corresponding sediment size classification data. Based on these data, the partitioning coefficients of TPAH, TPCB, and zinc to clay, silt, and sand were estimated (Table 5-1).

Table 5-1. Estimated bed partitioning coefficients used in the TMDL models

	Clay (L/mg)	Silt (L/mg)	Sand (L/mg)
TPAH	0.0011	0.0011	0.0001
TPCB	0.0019	0.0012	0.0001
Zinc	0.024	0.024	0.01

Since no data were available for chlordane or lindane to derive partitioning coefficients, they were assumed to be the same as PCBs. The reason for choosing PCBs over PAHs is because the partitioning coefficients estimated from data appear to be higher for PCBs than for PAHs. To provide an added level of conservatism for TMDL development, the higher partitioning coefficients were adopted for the contaminants with unknown values.

Water column partitioning coefficients were derived based on sampled water column data reported in Chadwick et al. (2008). Organic contaminants (i.e., TPAH and chlordane) were sampled for the storm event in April 2007 for Chollas Creek. This sampling effort, which was performed for both the South and North Chollas branches, provided particulate TPAH and chlordane concentrations for the event. Data for North Chollas show that the majority of TPAH was adsorbed to fine sediments (clay and silt). For South Chollas, however, TPAH was found to show stronger adsorption to sand than to fine sediments. This phenomenon indicates the complexity and variability of toxic-sediment interaction and suggests there can be significant uncertainty in the numerical representation of the adsorption/desorption of a contaminant to a sediment class using a constant coefficient.

Higher partitioning coefficients were generally selected for the water column in order to be more conservative and protective for TMDL development. Therefore, the adsorbed TPAH concentration on sand for South Chollas was integrated with the adsorbed TPAH concentration on clay and silt for North Chollas to determine the partitioning coefficients. The resulting partitioning coefficients account for the high adsorptive characteristics indicated by the data for all the sediment classes. For chlordane, only the data for North Chollas were used since both the South and North Chollas data showed a similar trend. The majority of the chlordane was adsorbed to fine sediments. The water column partitioning coefficients were derived based on these data (Table 5-2).

Zinc was measured for all the storm events reported in Chadwick et al. (2008), therefore all zinc data were used to derive partitioning coefficients. The derived partitioning coefficients were all significantly lower than those for the bed sediment, although in reality water column partition coefficients are generally higher than those for the bed. Again, to be conservative, the partition coefficients for zinc in the water column were set to be the maximum of the derived bed and water column values. The final values used in the model are presented in Table 5-2. Note that for PCBs and lindane the values for chlordane were adopted.

Table 5-2. Estimated water column partitioning coefficients used in the TMDL models

	Clay (L/mg)	Silt (L/mg)	Sand (L/mg)
TPAH	0.0543	0.0593	0.0223
TCHLOR	0.0033	0.0033	0.0001
Zinc	0.024	0.024	0.01
PCB	0.0033	0.0033	0.0001
Lindane	0.0033	0.0033	0.0001

The above parameter values were applied to all four toxic models. Since the major parameters were derived based on field data, minimal parameter adjustment was necessary for the toxic models.

To evaluate the performance of the models, the simulated toxic concentrations (TPAH, TPCB, and chlordane) in the bed sediment were compared to data collected from July 2001 to October 2002 at four sites near the mouths of Paleta Creek and Chollas Creek. The locations of the sampling sites are presented in the Phase I sediment assessment report titled *Sediment Assessment Study for the Mouths of Chollas and Paleta Creek, San Diego* (SCCWRP and SPAWAR 2005). Sediment bed concentration data were found to vary significantly even for cores collected from locations in proximity to one another. This highly localized nature makes it extremely difficult for a model to precisely match time series observed data. Therefore, the calibration approach compared the range of observed and simulated concentrations. Specifically, the simulated maximum and minimum bed concentrations during the simulation period were compared with those of the observed data. The toxic model calibration results are presented in Figures D-1 through D-8 in Appendix D. The model generally matches the observed range of toxicity in the bed sediment well, although the data show greater range than the model results. This is because the data in bed sediment are localized and can vary significantly between adjacent locations. Even though the data were collected at the same station (i.e., C10) it is highly likely that the exact coordinate of the core varies slightly between different sampling events. This introduces significant variability in the data. In contrast, the model results represent the spatially averaged concentration in the corresponding grid, which represents much less heterogeneity than the field data; therefore, the range of model results is expected to have less variability. In addition, the current model is configured with only watershed sources and the benthic legacy sources, while in reality other unquantifiable sources might have contributed to the change in benthic toxicity. This can also cause higher variability in data then in the model results. Additional uncertainty might have been related to the uncertainty in the LSPC watershed model and the simplified representation of sediment components that were discussed earlier in the report (see Section 5.2). Considering all the sources of uncertainty, the model results are considered to be reasonable.

6. Model Application for Baseline Analysis

6.1 Determination of Baseline Conditions

After the hydrodynamic, sediment transport, and toxic models were calibrated and evaluated, they were used to conduct a set of baseline analyses to help understand the response of bed sediment toxicity to watershed loading. The first step in configuring the baseline models was to evaluate the loading distribution over an extended recent period predicted by the LSPC models for each watershed (2001 to 2006). The annual toxic loading was summarized by hydrologic year (Table 6-1). The hydrologic year from October 2004 to September 2005 had the highest flow rate as well as TSS and toxics loading; therefore, this year was used as the baseline condition as it represented the worst case scenarios in terms of watershed loading.

Table 6-1. Annual loading comparison at the mouths of the five watersheds for 2001-2006

19001	***Chollas Creek***				
Hydrologic Year	**October 2001-September 2002**	**October 2002-September 2003**	**October 2003-September 2004**	**October 2004-September 2005**	**October 2005-September 2006**
Flow (m3)	681,238	4,859,085	2,101,106	15,896,657	3,032,667
TSS (kg)	62,300	1,555,357	318,297	20,846,674	519,669
Cu (g)	9,565	253,311	51,677	2,951,606	84,123
Pb (g)	6,895	208,934	37,803	2,045,280	60,863
Zn (g)	81,565	2,076,595	437,861	19,122,090	709,279
PAH (mg)	443,537	11,073,242	2,266,094	148,416,293	3,699,743
PCB (mg)	269	6,719	1,375	90,054	2,245
Chlordane (mg)	11,915	297,469	60,876	3,987,021	99,389
Lindane (mg)	269	6,719	1,375	90,054	2,245

19040	***Switzer Creek***				
Hydrologic Year	**October 2001-September 2002**	**October 2002-September 2003**	**October 2003-September 2004**	**October 2004-September 2005**	**October 2005-September 2006**
Flow (m3)	265,715	1,370,857	624,526	4,647,085	863,561
TSS (kg)	27,489	483,422	83,035	8,713,469	117,226
Cu (g)	6,819	128,042	21,617	2,047,204	30,944
Pb (g)	5,253	110,465	17,255	1,395,397	24,532
Zn (g)	66,348	1,165,674	205,186	13,812,521	292,428
PAH (mg)	40,311	708,923	121,769	12,778,005	171,908
PCB (mg)	38	662	114	11,926	160
Chlordane (mg)	3,557	62,555	10,745	1,127,527	15,169
Lindane (mg)	38	662	114	11,926	160

19042	*Paleta Creek*				
Hydrologic Year	**October 2001-September 2002**	**October 2002-September 2003**	**October 2003-September 2004**	**October 2004-September 2005**	**October 2005-September 2006**
Flow (m3)	180,227	650,505	315,446	2,622,822	463,797
TSS (kg)	16,736	151,033	39,163	7,697,295	56,125
Cu (g)	3,415	33,922	8,369	1,752,745	12,154
Pb (g)	2,202	22,520	5,376	814,764	7,809
Zn (g)	25,467	247,889	62,392	6,674,765	89,981
PAH (mg)	85,826	774,533	200,835	39,473,546	287,823
PCB (mg)	50	455	118	23,171	169
Chlordane (mg)	4,080	36,817	9,547	1,876,361	13,682
Lindane (mg)	50	455	118	23,171	169

19044	*B Street/Broadway Pier*				
Hydrologic Year	**October 2001-September 2002**	**October 2002-September 2003**	**October 2003-September 2004**	**October 2004-September 2005**	**October 2005-September 2006**
Flow (m3)	158,825	619,009	306,167	2,148,606	336,770
TSS (kg)	18,435	159,339	43,063	4,866,390	44,910
Cu (g)	4,224	40,203	10,058	904,042	10,490
Pb (g)	3,422	33,692	8,127	649,293	8,504
Zn (g)	45,821	418,672	109,150	7,371,398	113,845
PAH (mg)	27,034	233,666	63,151	7,136,395	65,859
PCB (mg)	25	218	59	6,661	61
Chlordane (mg)	2,385	20,619	5,572	629,713	5,811
Lindane (mg)	25	218	59	6,661	61

19046	*Downtown Anchorage*				
Hydrologic Year	**October 2001-September 2002**	**October 2002-September 2003**	**October 2003-September 2004**	**October 2004-September 2005**	**October 2005-September 2006**
Flow (m3)	87,693	301,341	153,426	1,111,244	179,217
TSS (kg)	7,501	66,759	17,432	2,944,081	18,146
Cu (g)	2,293	22,807	5,517	810,949	5,641
Pb (g)	1,870	19,241	4,508	505,964	4,596
Zn (g)	20,063	193,576	48,123	4,597,110	49,413
PAH (mg)	11,000	97,900	25,564	4,317,395	26,610
PCB (mg)	10	91	24	4,030	25
Chlordane (mg)	971	8,639	2,256	380,966	2,348
Lindane (mg)	10	91	24	4,030	25

6.2 Configuration of Baseline Models

Baseline models were configured based on the calibration model for the five shoreline areas of concern, but the flows and loading conditions were replaced based on the October 2004 to September 2005 LSPC model results. The model simulation period was set for one year (October 1, 2004, to September 30, 2005). All parameters were set to be the same as the calibration model. For multiple year simulations, the results at the end of the preceding year were saved as initial conditions for the simulation of the next year.

6.3 Water Column Model Results at the Outer Boundary of the Creek Mouths

During storm periods, the water column sediment concentration at the outer boundary of the watershed mouths can be impacted by stormwater, which prevents an accurate specification of the open boundary condition in a coarse grid WASP model (see Section 2.2). The simulated fine sediment concentrations at the outer boundary of the Paleta Creek mouth are used as an example to illustrate this condition (Figure 6-1). The fine sediment concentration at the outer boundary of the Paleta Creek mouth can be very high during storm events, reaching values close to 1,000 mg/L. If a coarse grid WASP model was configured to simulate the sediment transport for this area, the prescribed boundary condition must reflect the impact from stormwater, which is not available in a WASP model framework since this information can only be obtained from a sediment transport model driven by a hydrodynamic model. This limitation would be a serious limitation of a coarse grid WASP approach, which would be unreliable as a modeling framework for developing TMDLs at the creek mouths.

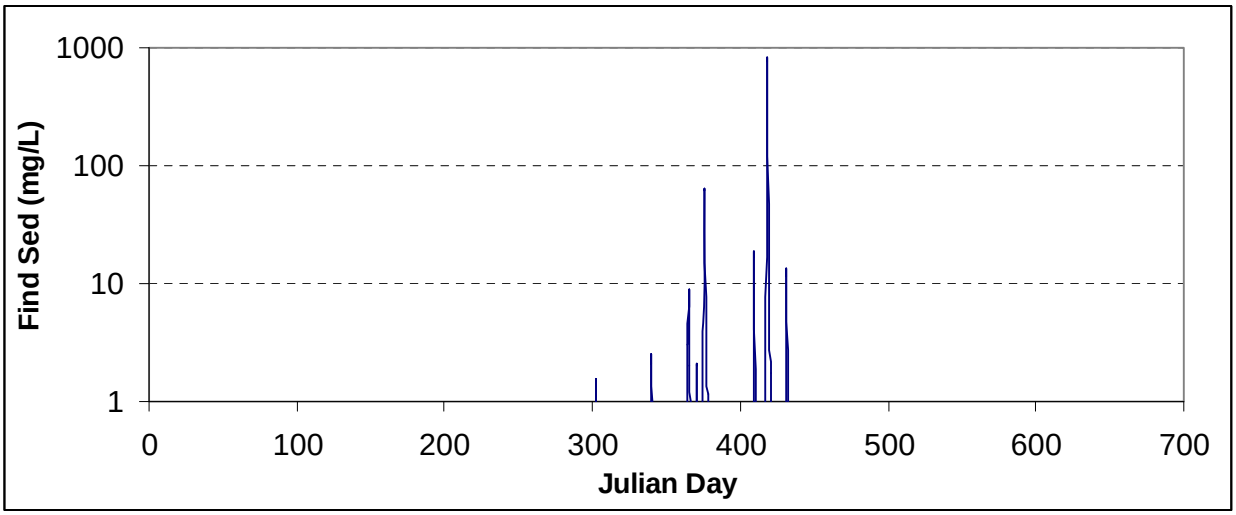

Figure 6-1. EFDC simulated fine sediment concentrations at the outer boundary of the Paleta Creek mouth.

6.4 Spatial Variability in Sediment Bed Model Results

The spatial variability in model results in the sediment bed is important, and directly dictates how the TMDLs focusing on sediment bed toxicity will be developed. The impaired areas need to be restored in a TMDL, thus the maximum concentration across the grids in the impaired areas should be maintained under numeric targets. The spatial variability in model results is illustrated using modeling results at the mouth of Paleta Creek.

Figure 6-2 shows the time series of surface bed layer TPAH results at three randomly selected sites in the Paleta model. The initial TPAH concentration at site 1 is very high and gradually decreases until approximately Julian day 364. The concentration shows a sudden drop because of fast deposition of relatively cleaner sediment at this location. At site 2, the initial TPAH concentration is much lower than that at site 1 and the concentration, instead of decreasing, increases with time. Site 3 has initial TPAH concentrations between those of site 1 and site 2. The concentrations at site 3 also tend to increase with time. Temporal variability of bed sediment toxicity show different patterns at different locations. In the context of the TMDLs, it is clear that during the early period of the simulation, site 1 serves as the driving location since its concentration is the highest among the three sites. However, with the decrease in concentration at site 1 and increase in concentration at site 3, site 3 then turns into the driving location after a certain period.

This phenomenon also exists at other modeled local areas. It suggests that to develop a reliable TMDL, it is necessary to reasonably represent the spatial variability through simulating the interactions between hydrodynamics, sediment transport, and toxic contaminants. Particularly, a full representation of hydrodynamics at these areas of concern is of critical importance since it directly controls the transport of sediment and toxics in the areas. If the transport of toxics and sediment is not reasonably simulated, significant errors might be introduced into the resultant sediment toxicity pattern, hence impair the reliability of the developed TMDL.

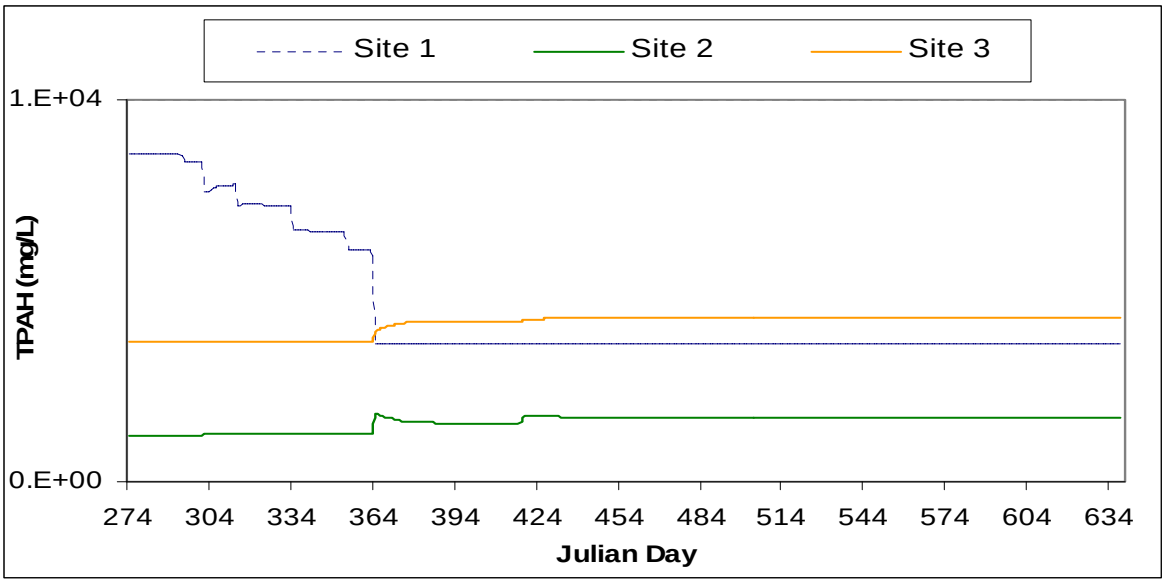

Figure 6-2. Time series of surface bed layer TPAH results at sites 1, 2, and 3 in the Paleta model.

6.5 Sensitivity to Watershed Loading Level

To evaluate the sensitivity of the bed toxicity to the uncertainty in the watershed loading, a set of sensitivity runs were conducted. The sensitivity models were configured based on the baseline model except that the toxic contaminant boundary conditions were reset with the values based on the maximum values of available data. Since only TPAH had a significant difference between the maximum and mean value in the boundary condition, only results for TPAH are shown to analyze the sensitivity of the boundary condition (See the Watershed Modeling report. Note that for lindane and TPCB, concentrations were based on detection limits and therefore no variability of concentrations is provided). The sediment bed TPAH concentration is compared between the baseline and sensitivity run at the grids two cells downstream of the inflowing cell for all five impaired shoreline areas. The results are presented in Figures E-1 through E-5 in Appendix E.

The results show that the bed sediment concentration of TPAH has a certain level of sensitivity to the change in watershed loading. The relative sensitivity depends on the absolute magnitude of the concentration. The relative sensitivity appears to be smaller for locations with higher absolute concentration than at those locations with lower absolute concentration. Overall, the difference caused by using the mean value from available data is not significant except for Switzer Creek, where the difference is significant due mostly to the large variability of TPAH concentrations from the watershed (see the Watershed Modeling Report). In addition to the absolute concentration, it appears that the location and bathymetry might also contribute to the sensitivity. It is expected that at cells far from the inflowing cells, the sensitivity would be very small.

6.6 Temporal Response in Sediment Bed Toxicity

The baseline models were run for three years to gain preliminary knowledge of the temporal response of sediment bed toxicity to the baseline watershed loading. The simulated maximum surface bed concentrations for the corresponding toxic contaminants for three example areas (Paleta Creek, Switzer Creek, and Downtown Anchorage) are presented in Figures F-1 through F-10 in Appendix F.

Note the surface bed is defined as the top two layers, which consist of the fixed thickness layer of 20 centimeters below the top layer, and the top layer with variable thickness. Both the top layer and the layer below are included because the thickness of the top layer varies significantly (from 0 to 20 centimeters) over time at locations near the inflow cells. Therefore, even though the sediment toxicity might reach the target at this layer, it is not guaranteed that sufficient depth of compliance is achieved.

The time variable sediment bed concentrations of TPAH, TPCB, and chlordane at the mouth of Paleta Creek are shown in Figures F-1 through F-3. The results show that under the baseline loading level, the maximum TPAH concentration in the impaired area tends to reduce over time because the inflowing stormwater contains sediment of lower TPAH concentration. TPCB concentrations also reduce over time; however, the slope of decrease appears to be more level. Chlordane shows a significant trend of increase with time because the incoming stormwater has high concentrations of chlordane under the baseline condition. In all cases, the bed toxicity shows detectable response to watershed loading.

The time variable sediment bed concentrations of TPAH, TPCB, chlordane, and lindane at the mouth of Switzer Creek are shown in Figures F-4 to F-7. The results show that under baseline loading conditions, the TPAH and TPCB concentrations decrease sharply over time during the first three years. The decrease of TPAH and TPCB shows a step-shape pattern, which is caused by the addition of bed layer at the time when sufficient sediment is deposited to the bed. When sufficient sediment is deposited at a specific grid location, one more layer will be added to the bed, which has a different toxicity level than the layer below it. This can result in a shift of cells of maximum concentration from one cell to the other, thus the time series of the maximum toxicity level would show a step-shape pattern. Unlike TPAH and TPCB, the bed chlordane level shows a trend of increase due to higher incoming concentrations. The lindane concentrations seem pretty stable over the three years of simulation, indicating that the incoming loading has minimum impact on the bed toxicity of lindane.

An additional set of results were generated for the Downtown Anchorage area (Figures F-8 to F-10). The TPAH concentration in the bed sediment shows a slight upward trend during the first three years, while TPCB tends to decrease with time. As with Paleta Creek and Switzer Creek, the chlordane levels tend to significantly increase over time due to the high concentration inflows.

6.7 TMDL Development Strategy

As shown in the temporal response examples in Section 6.6, the sediment toxicity in the impaired shoreline areas tend to have a detectable relationship to the watershed loading. When the watershed concentration is higher than the existing concentration, the incoming sediment load causes an increase of bed toxicity with time. If the watershed concentration is the same as the existing concentration in the bed, it is expected that the condition in the bed will not become worse. This leads to an inference that if under existing condition the sediment bed meets numeric targets, and the watershed inflow concentration can be maintained below or at the numeric target, then the bed will continue to meet the target. For these conditions, there is no need to further conduct model runs to determine TMDLs for the contaminants at the shoreline areas where the numeric target is met under existing conditions. Chlordane is an example. Under existing conditions, chlordane has very low concentrations in the bed at Downtown Anchorage, therefore, the TMDL allocation for watershed loading can include a specification of a suspended solids concentration equal to the numeric target for bed sediment concentration, and the bed sediment toxicity will not further worsen.

For contaminants that are higher than the numeric targets, TMDLs need to be developed through reducing the watershed loading. However, as shown in the temporal response result, the response of bed toxicity to watershed loading might take a long time. Although, theoretically any loading level lower than the numeric target would eventually result in meeting numeric targets. The time of reaching the numeric target would vary at different watershed loading levels. Using the model to estimate how long it would take for the bed to recover and meet the numeric targets is the objective of implementation scenario analysis that could be performed in the future. For TMDL development, an approach is recommended that involves re-initializing the sediment toxicity condition with various loading levels. This approach involves the following scenarios:

> Scenario 1: set the initial bed sediment toxic concentrations at numeric targets for all the local models, then run the model with existing watershed loading as in the baseline analysis. The model will be run for three years to check the response of sediment toxicity. If for a given contaminant, the sediment toxic concentration does not increase over time with the existing watershed loading, then the existing watershed loading will be the basis for the TMDL, requiring no load reduction. In this case, sediment dredging would be required. And, if the sediment toxicity increase over time with existing loading, then watershed loading reduction is needed, leading to the second scenario.
>
> Scenario 2: set the initial sediment toxic concentration as the same as for Scenario 1, but reduce the watershed loading to the numeric target values. The model will be run for three years, and time series of sediment toxicity level will be evaluated against numeric target values. If the predicted sediment loading is below the numeric target, then this suggests that the watershed loading can be increased. This would leads to the third set of scenarios.

Scenario 3: this scenario would likely be iterative. Based on scenario 2, the watershed suspended solids organics concentrations will be increased to levels between existing and numeric target values for bed sediments. The exact level will be determined through an iterative process if time allows.

7. REFERENCES

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Appendix A

Calibration and Validation Plots for the EFDC Hydrodynamic Model

Figure A-1. Calibration results for the San Diego Bay hydrodynamic model. 1
Figure A-2. Extended comparison (validation) of the San Diego Bay hydrodynamic model results and data (June 2001). 1
Figure A-3. Extended comparison (validation) of the San Diego Bay hydrodynamic model results and data (September 2001). 2
Figure A-4. Extended comparison (validation) of San Diego Bay hydrodynamic model results and data (December 2001). 2

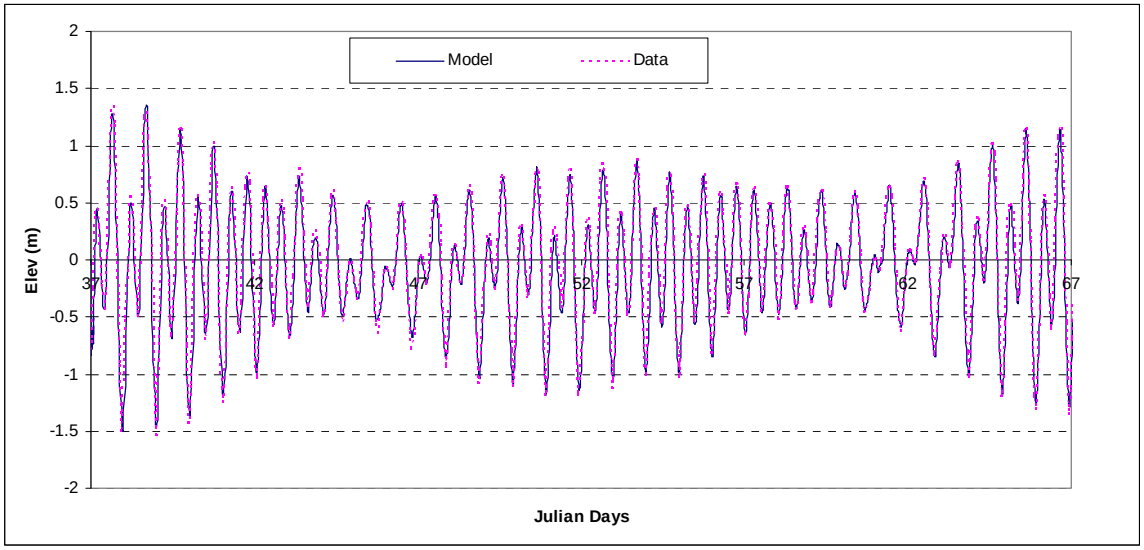

Figure A-1. Calibration results for the San Diego Bay hydrodynamic model.

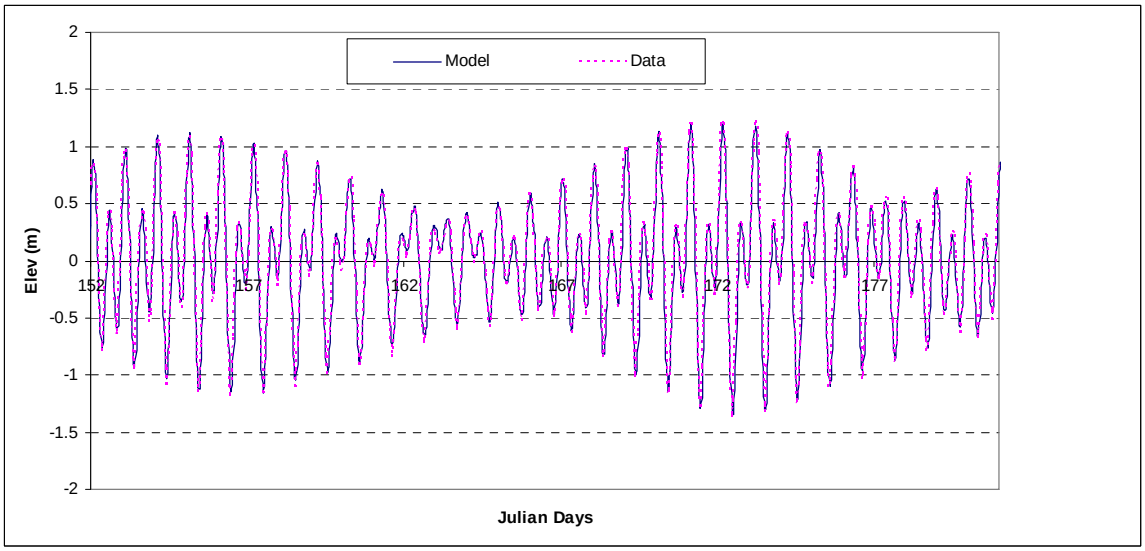

Figure A-2. Extended comparison (validation) of the San Diego Bay hydrodynamic model results and data (June 2001).

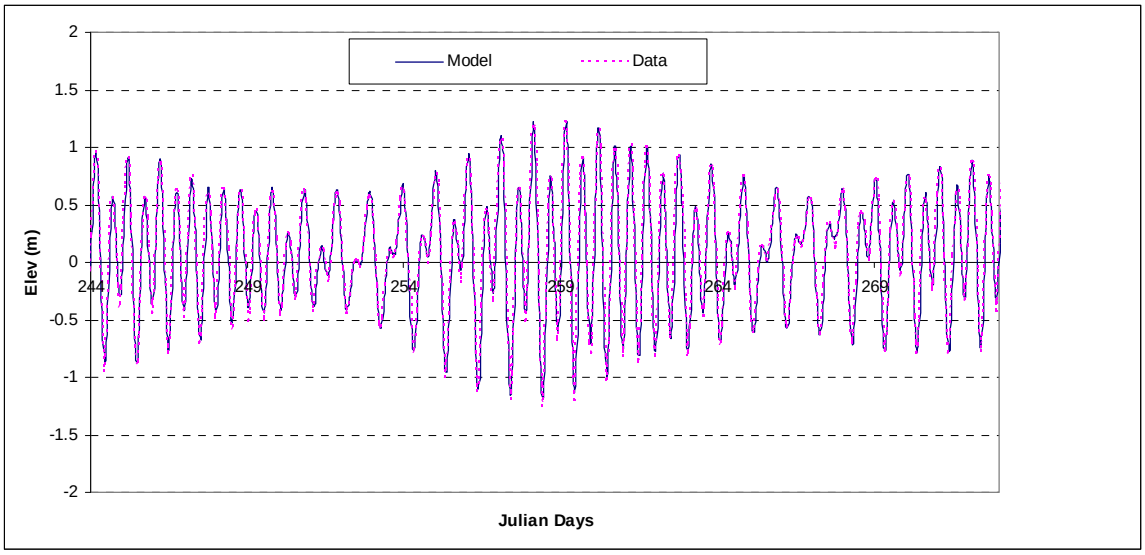

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Figure A-3. Extended comparison (validation) of the San Diego Bay hydrodynamic model results and data (September 2001).

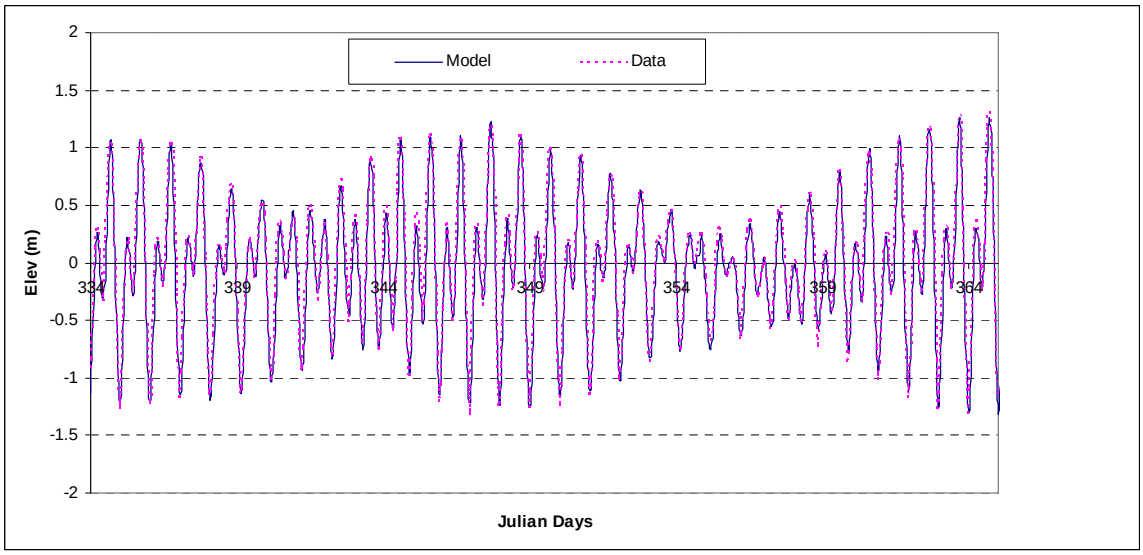

Figure A-4. Extended comparison (validation) of San Diego Bay hydrodynamic model results and data (December 2001).

Appendix B

Plots Showing Freshwater – Saltwater Interactions at the Mouths of the Five Watersheds

Figure B-1. Comparison of simulated salinity and observed salinity along trackline 1, Chollas Creek. 2
Figure B-2. Comparison of simulated salinity and observed salinity along trackline 3, Chollas Creek. 2
Figure B-3. Comparison of simulated salinity and observed salinity along trackline 4, Chollas Creek. 3
Figure B-4. Comparison of simulated salinity and observed salinity along trackline 5, Chollas Creek. 3
Figure B-5. Comparison of simulated salinity and observed salinity along trackline 6, Chollas Creek. 4
Figure B-6. Comparison of simulated salinity and observed salinity along trackline 7, Chollas Creek. 4
Figure B-7. Comparison of simulated salinity and observed salinity along trackline 1, Paleta Creek. 5
Figure B-8. Comparison of simulated salinity and observed salinity along trackline 2, Paleta Creek. 5
Figure B-9. Comparison of simulated salinity and observed salinity along trackline 3, Paleta Creek. 6
Figure B-10. Comparison of simulated salinity and observed salinity along trackline 4, Paleta Creek. 6
Figure B-11. Comparison of simulated salinity and observed salinity along trackline 5, Paleta Creek. 7
Figure B-12. Comparison of simulated salinity and observed salinity along trackline 6, Paleta Creek. 7
Figure B-13. Comparison of simulated salinity and observed salinity along trackline 7, Paleta Creek. 8

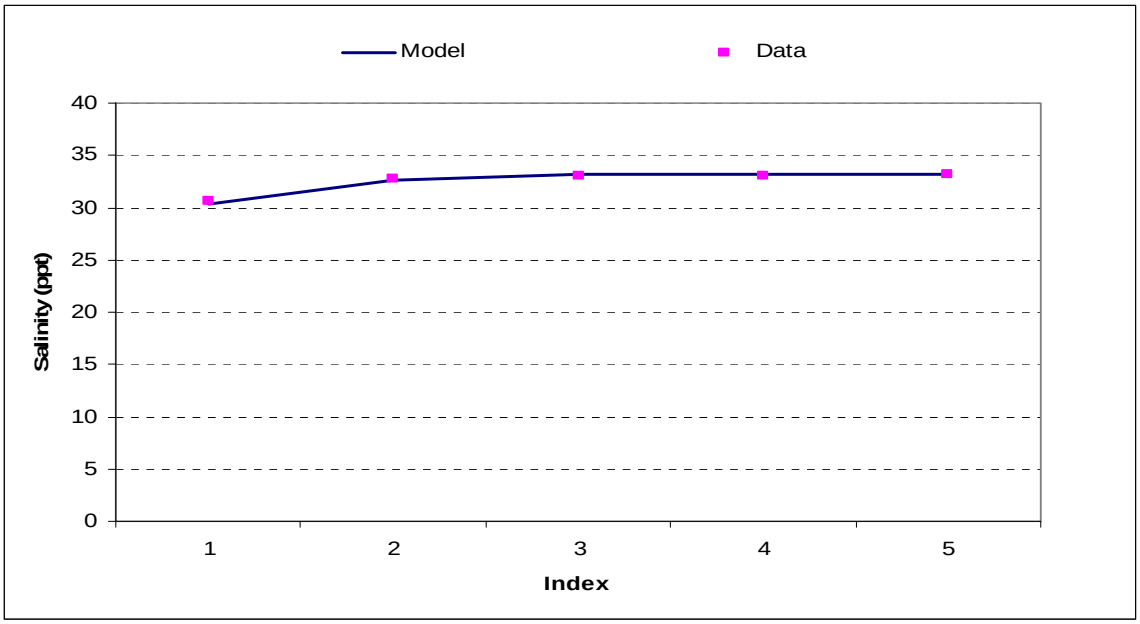

Figure B-1. Comparison of simulated salinity and observed salinity along trackline 1, Chollas Creek.

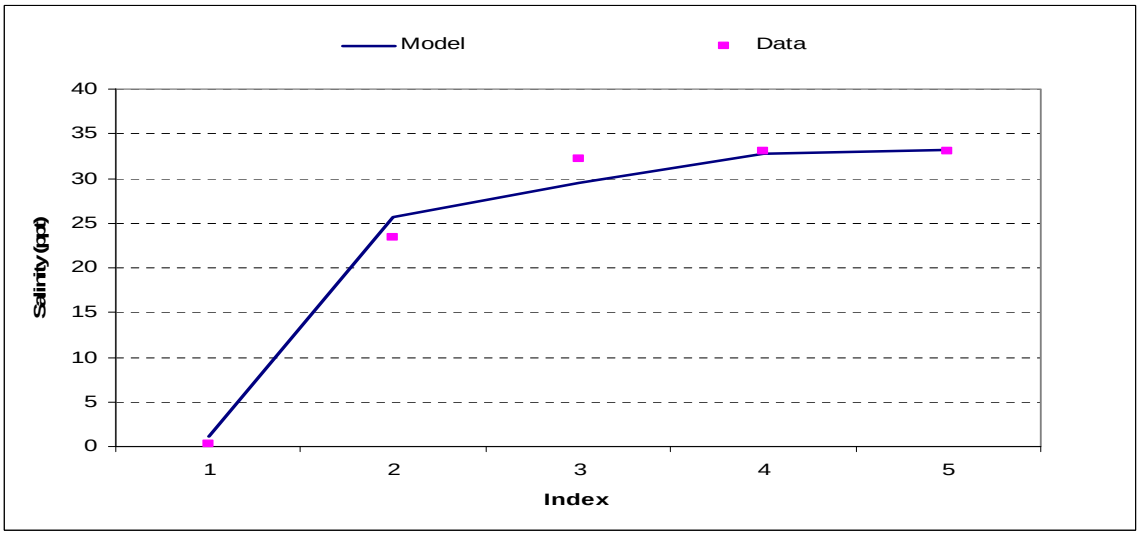

Figure B-2. Comparison of simulated salinity and observed salinity along trackline 3, Chollas Creek.

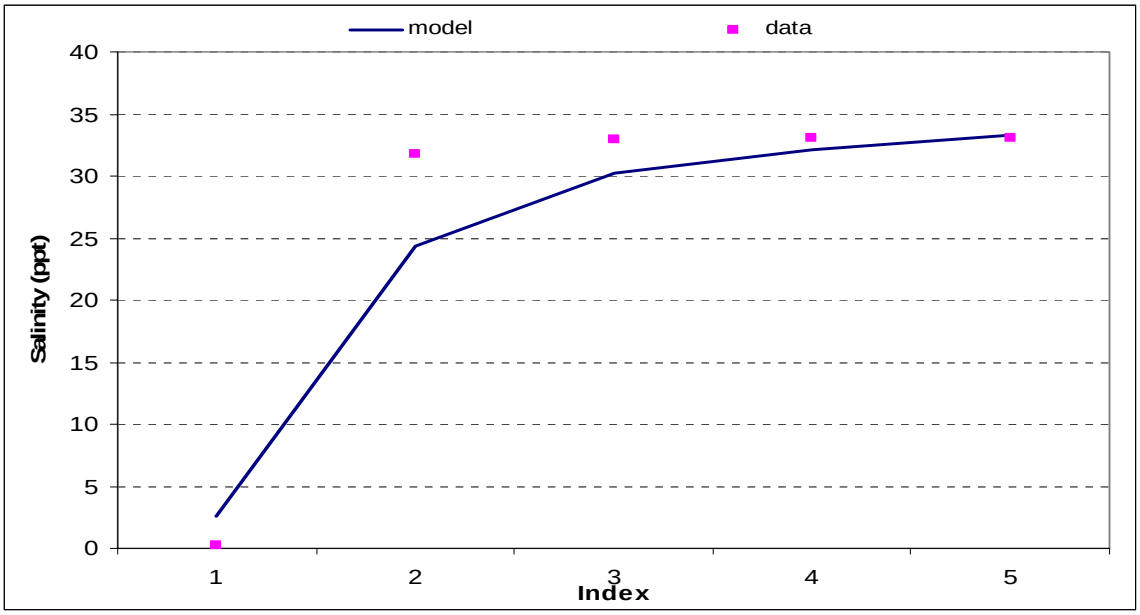

Figure B-3. Comparison of simulated salinity and observed salinity along trackline 4, Chollas Creek.

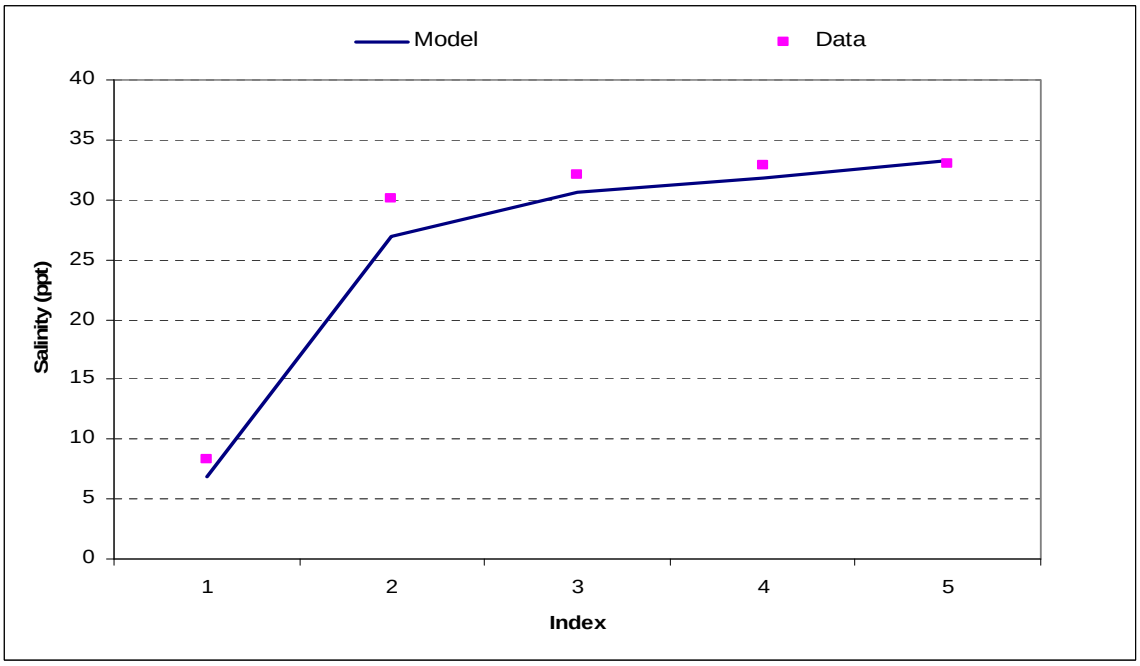

Figure B-4. Comparison of simulated salinity and observed salinity along trackline 5, Chollas Creek.

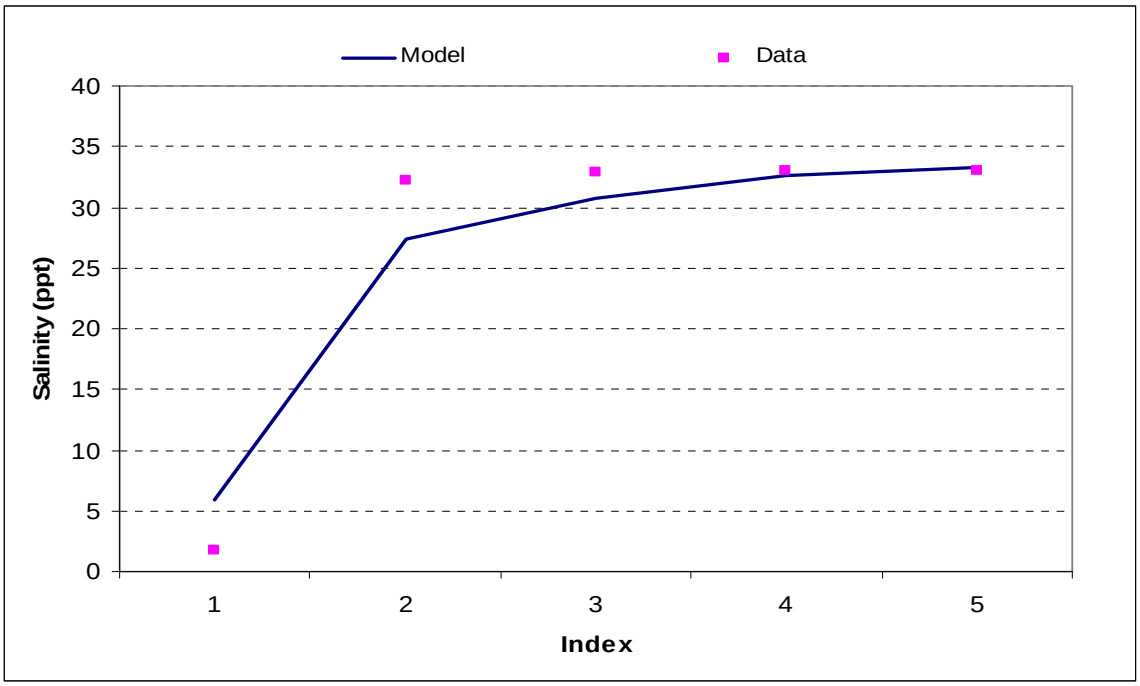

Figure B-5. Comparison of simulated salinity and observed salinity along trackline 6, Chollas Creek.

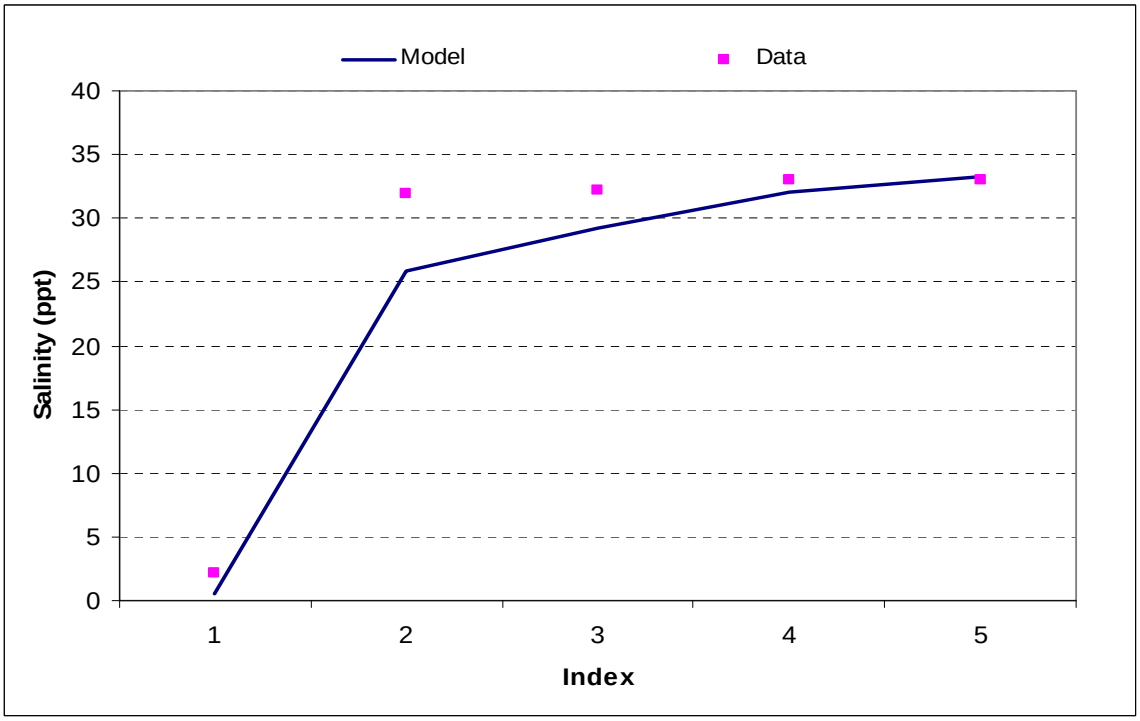

Figure B-6. Comparison of simulated salinity and observed salinity along trackline 7, Chollas Creek.

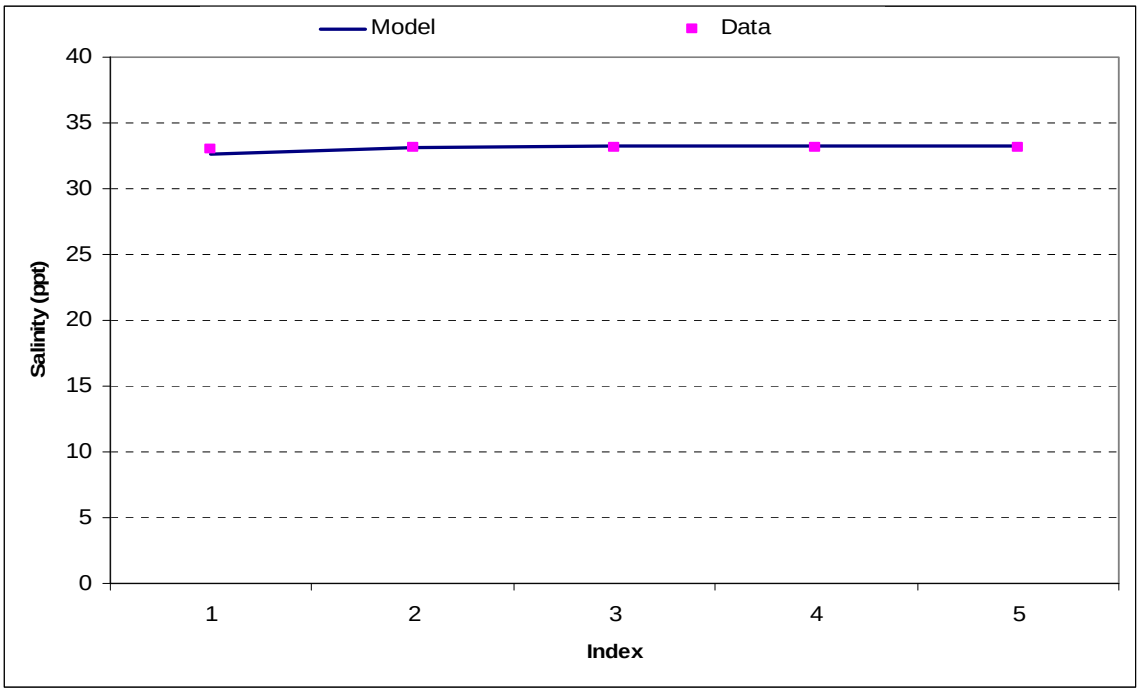

Figure B-7. Comparison of simulated salinity and observed salinity along trackline 1, Paleta Creek.

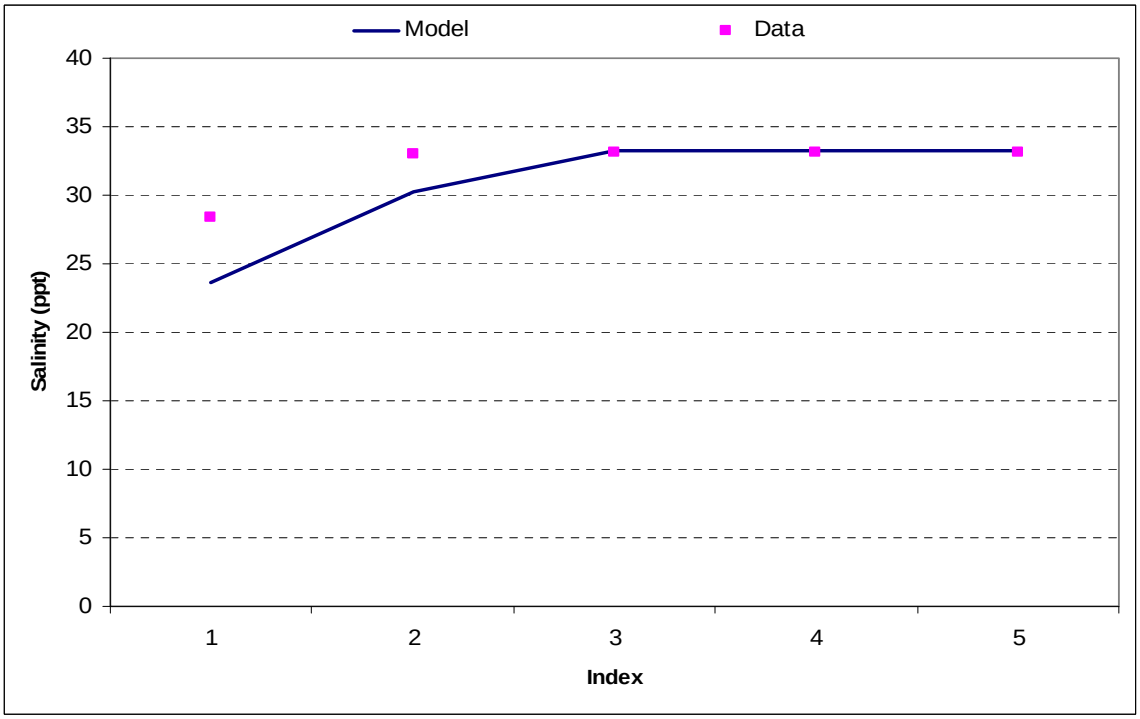

Figure B-8. Comparison of simulated salinity and observed salinity along trackline 2, Paleta Creek.

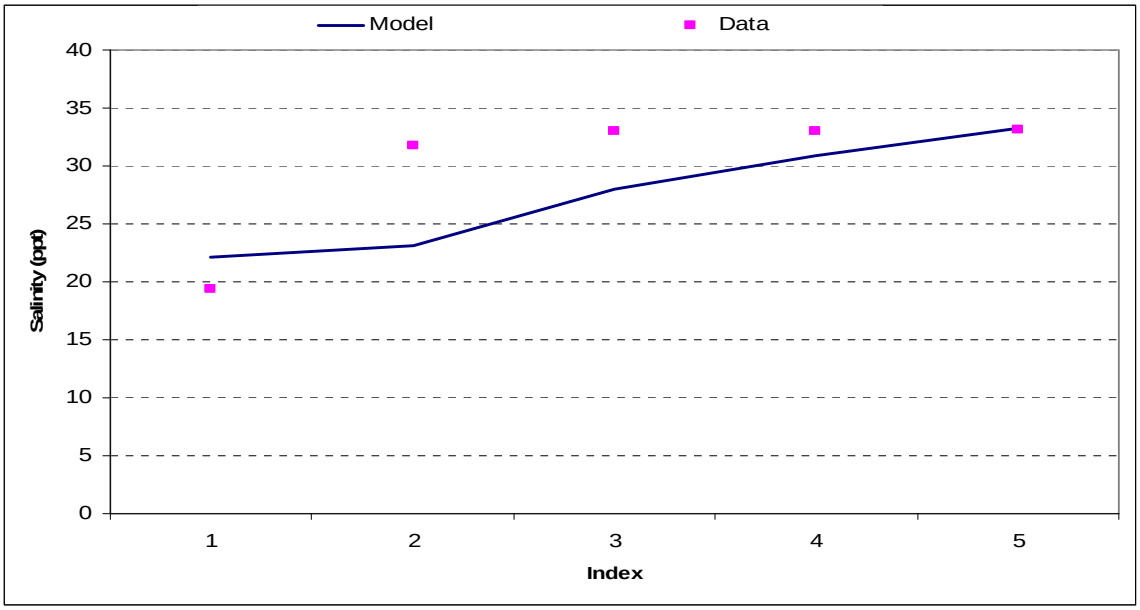

Figure B-9. Comparison of simulated salinity and observed salinity along trackline 3, Paleta Creek.

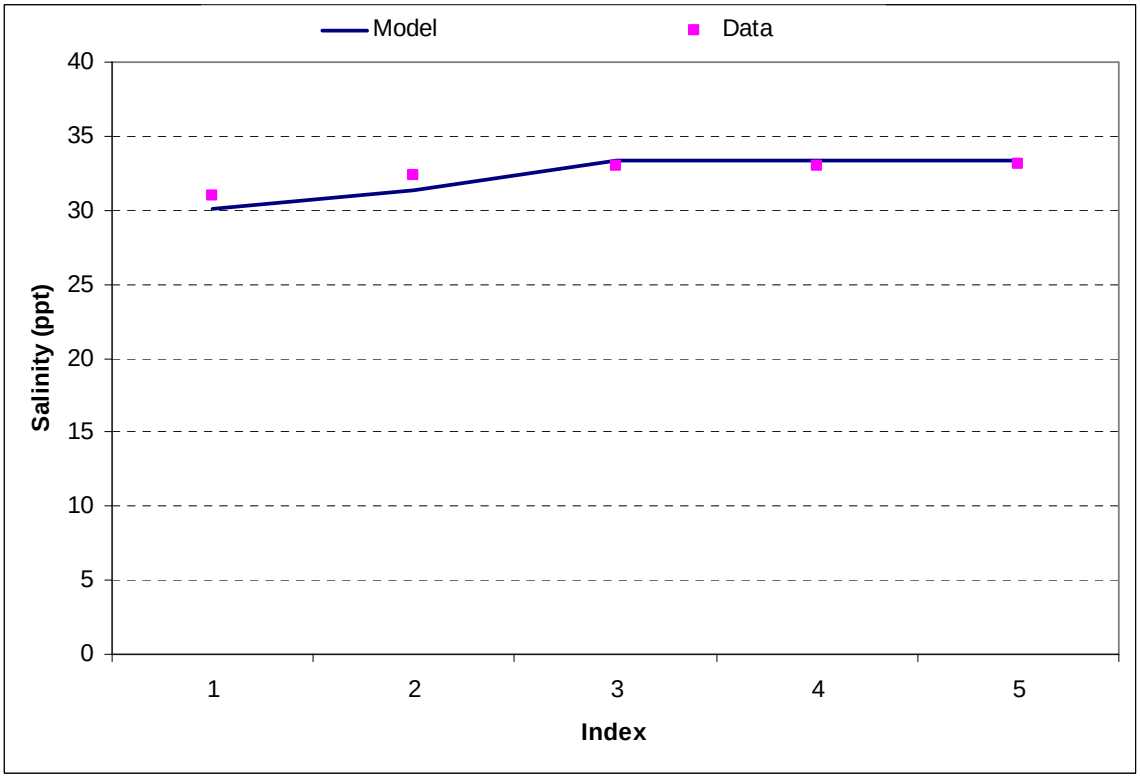

Figure B-10. Comparison of simulated salinity and observed salinity along trackline 4, Paleta Creek.

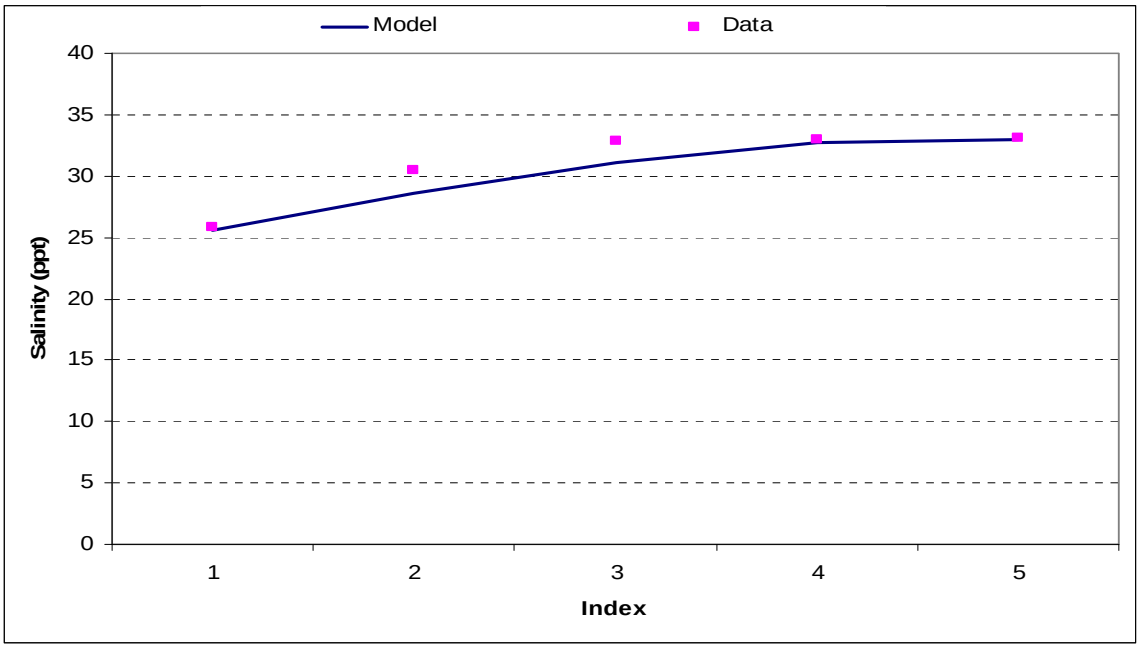

Figure B-11. Comparison of simulated salinity and observed salinity along trackline 5, Paleta Creek.

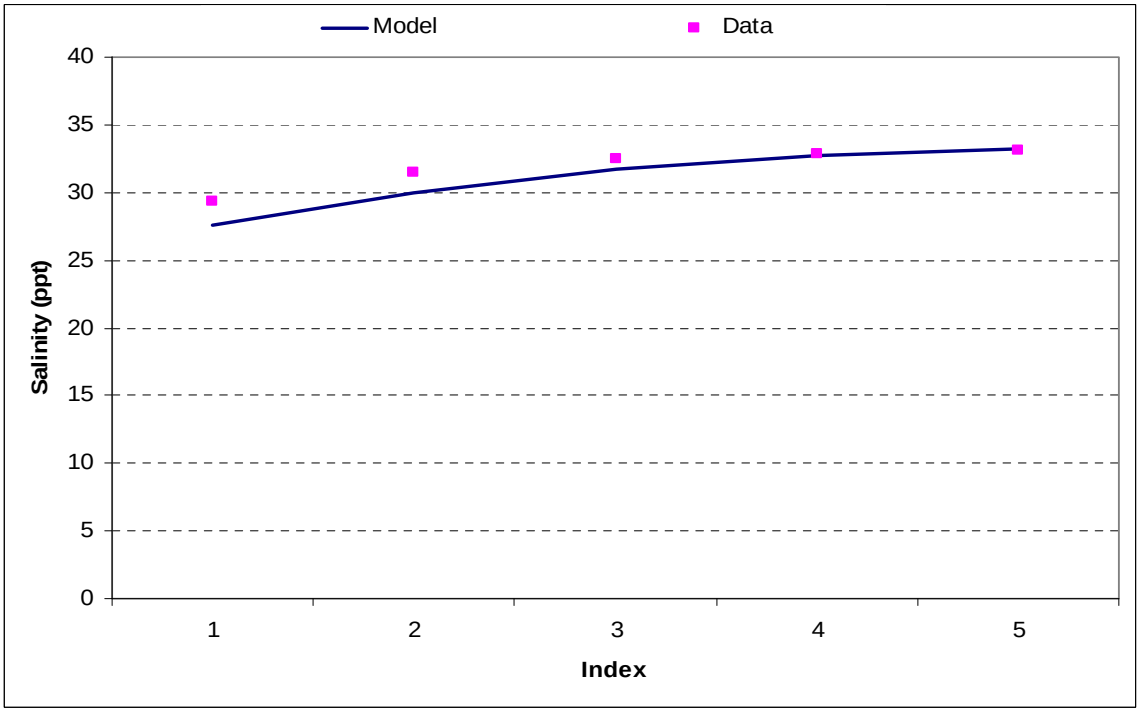

Figure B-12. Comparison of simulated salinity and observed salinity along trackline 6, Paleta Creek.

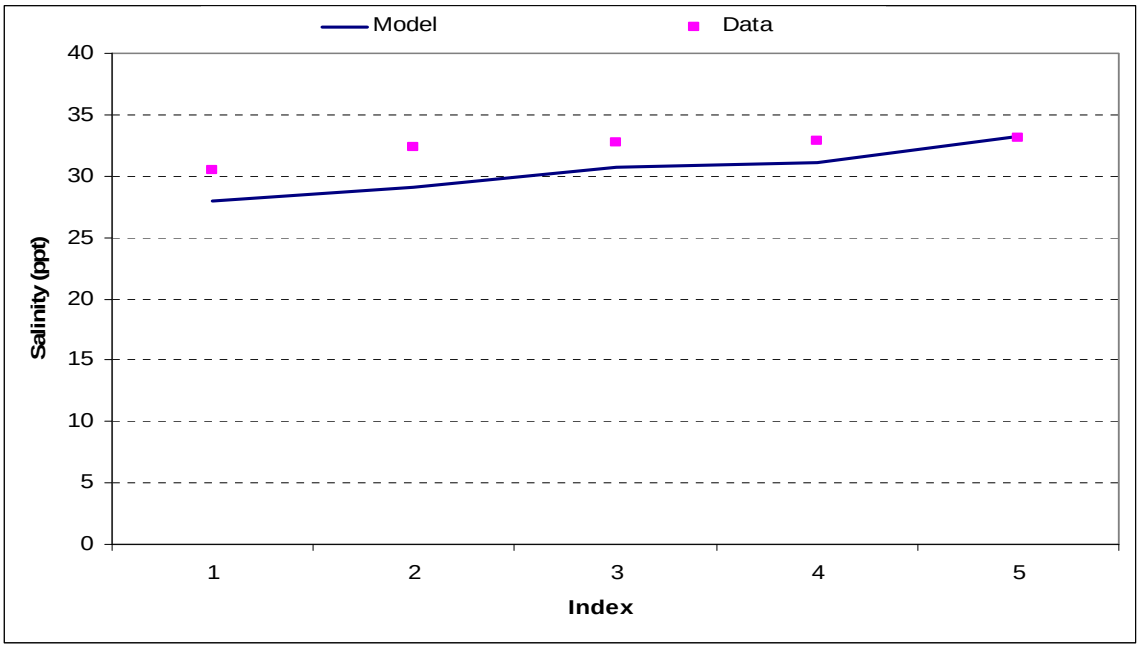

Figure B-13. Comparison of simulated salinity and observed salinity along trackline 7, Paleta Creek.

Appendix C

Calibration Plots for the EFDC Sediment Transport Model at the Mouths of Chollas and Paleta Creeks

Figure C-1. Comparison of simulated and observed TSS concentration at the mouth of Chollas Creek, Trackline 1. 1
Figure C-2. Comparison of simulated and observed TSS concentrations at the mouth of Chollas Creek, Trackline 3. 1
Figure C-3. Comparison of simulated and observed TSS concentration at the mouth of Chollas Creek, Trackline 4. 2
Figure C-4. Comparison of simulated and observed TSS concentration at the mouth of Chollas Creek, Trackline 5. 2
Figure C-6 Comparison of simulated and observed TSS concentration at the mouth of Chollas Creek, Trackline 6. 3
Figure C-7. Comparison of simulated and observed TSS concentration at the mouth of Chollas Creek, Trackline 7. 3
Figure C-8 Comparison of simulated and observed TSS concentration at the mouth of Paleta Creek, Trackline 1. 4
Figure C-9 Comparison of simulated and observed TSS concentration at the mouth of Paleta Creek, Trackline 2. 4
Figure C-10 Comparison of simulated and observed TSS concentration at the mouth of Paleta Creek, Trackline 3. 5
Figure C-11 Comparison of simulated and observed TSS concentration at the mouth of Paleta Creek, Trackline 4. 5
Figure C-12. Comparison of simulated and observed TSS concentration at the mouth of Paleta Creek, Trackline 5. 6
Figure C-13. Comparison of simulated and observed TSS concentration at the mouth of Paleta Creek, Trackline 6. 6
Figure C-14. Comparison of simulated and observed TSS concentration at the mouth of Paleta Creek, Trackline 7. 7

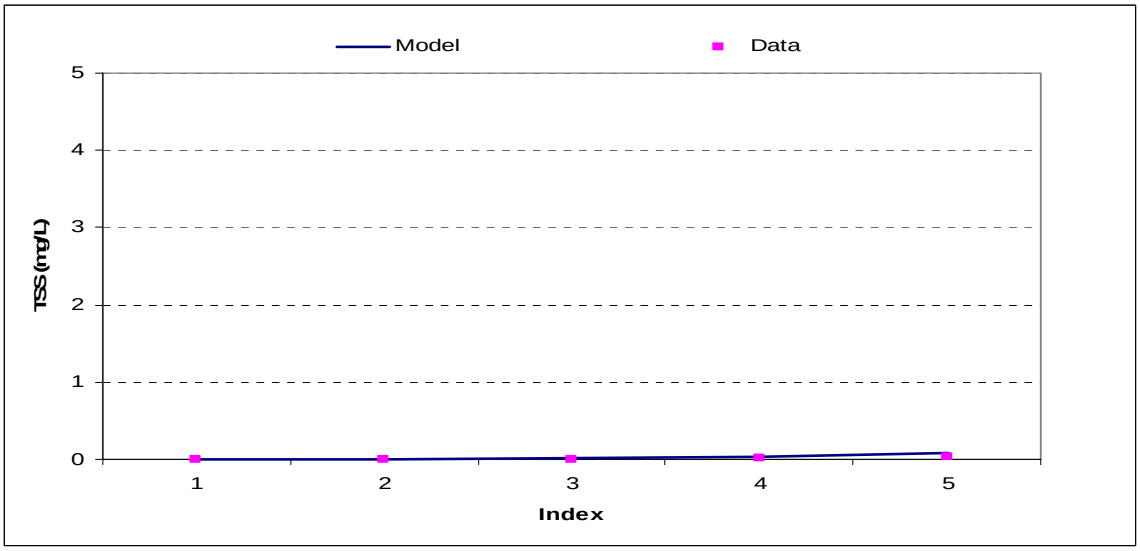

Figure C-1. Comparison of simulated and observed TSS concentration at the mouth of Chollas Creek, Trackline 1.

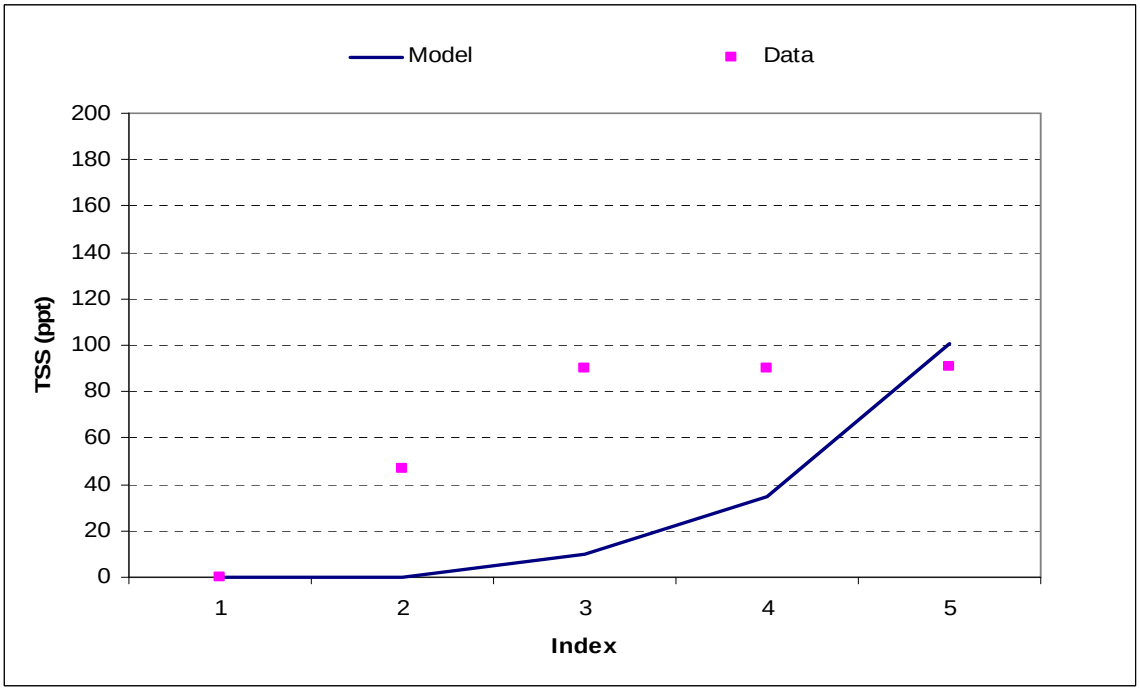

Figure C-2. Comparison of simulated and observed TSS concentrations at the mouth of Chollas Creek, Trackline 3.

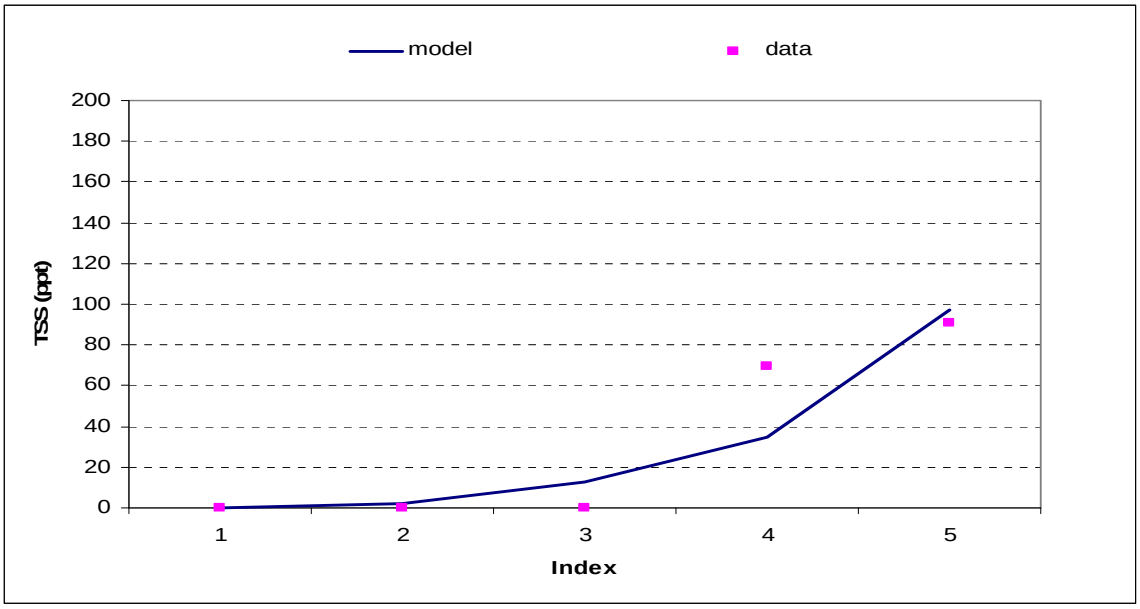

Figure C-3. Comparison of simulated and observed TSS concentration at the mouth of Chollas Creek, Trackline 4.

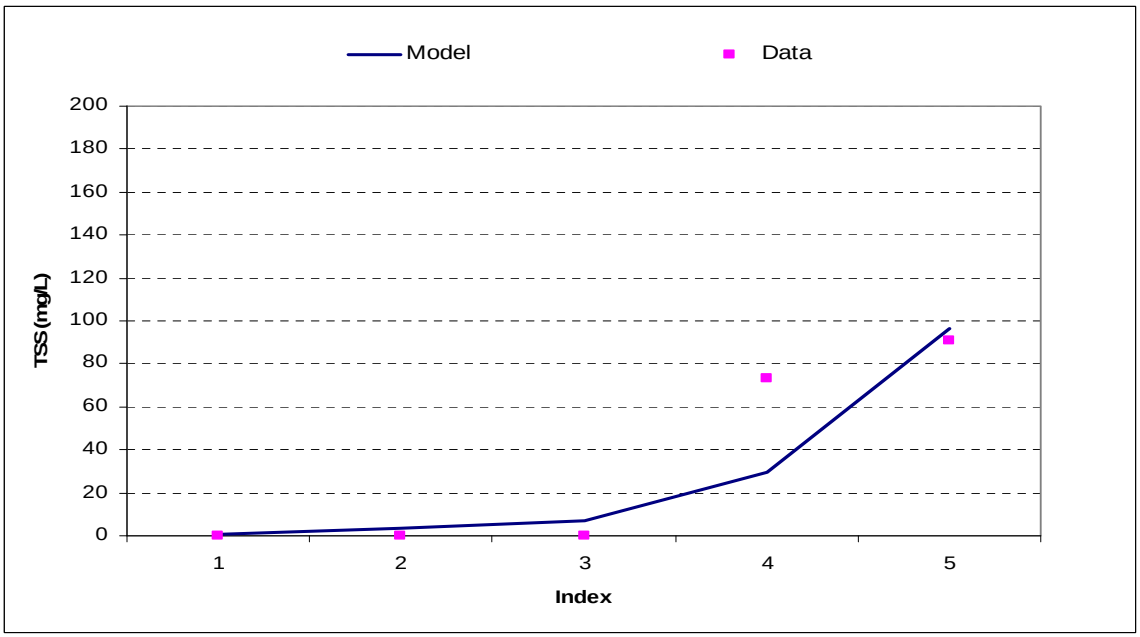

Figure C-4. Comparison of simulated and observed TSS concentration at the mouth of Chollas Creek, Trackline 5.

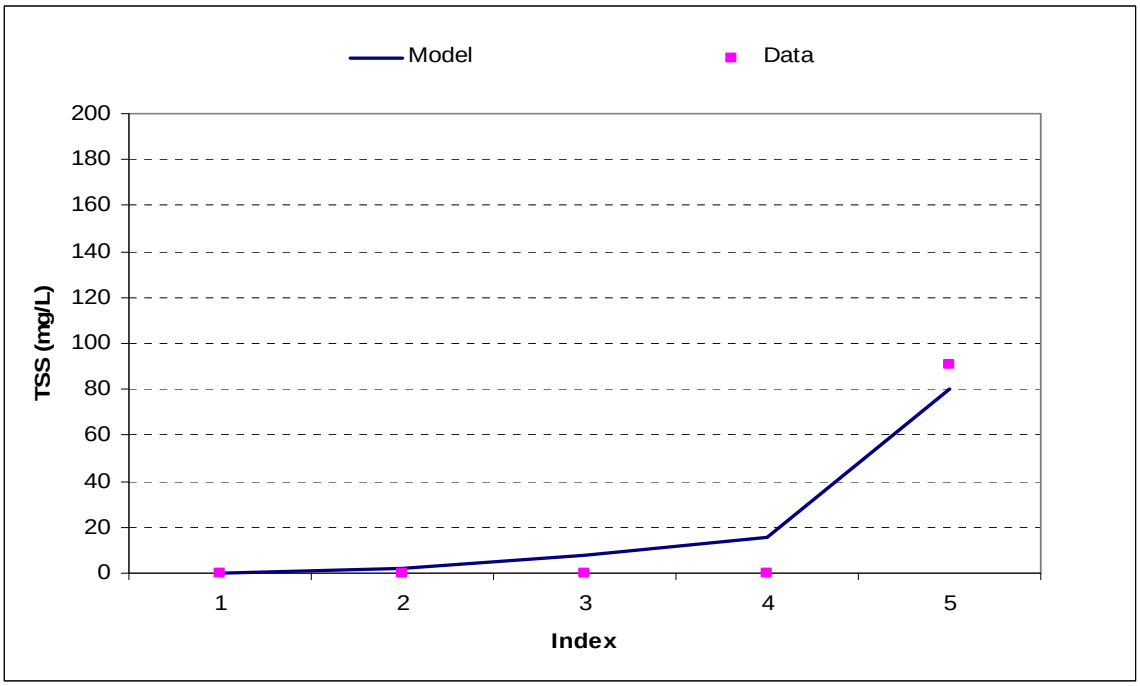

Figure C-6 Comparison of simulated and observed TSS concentration at the mouth of Chollas Creek, Trackline 6.

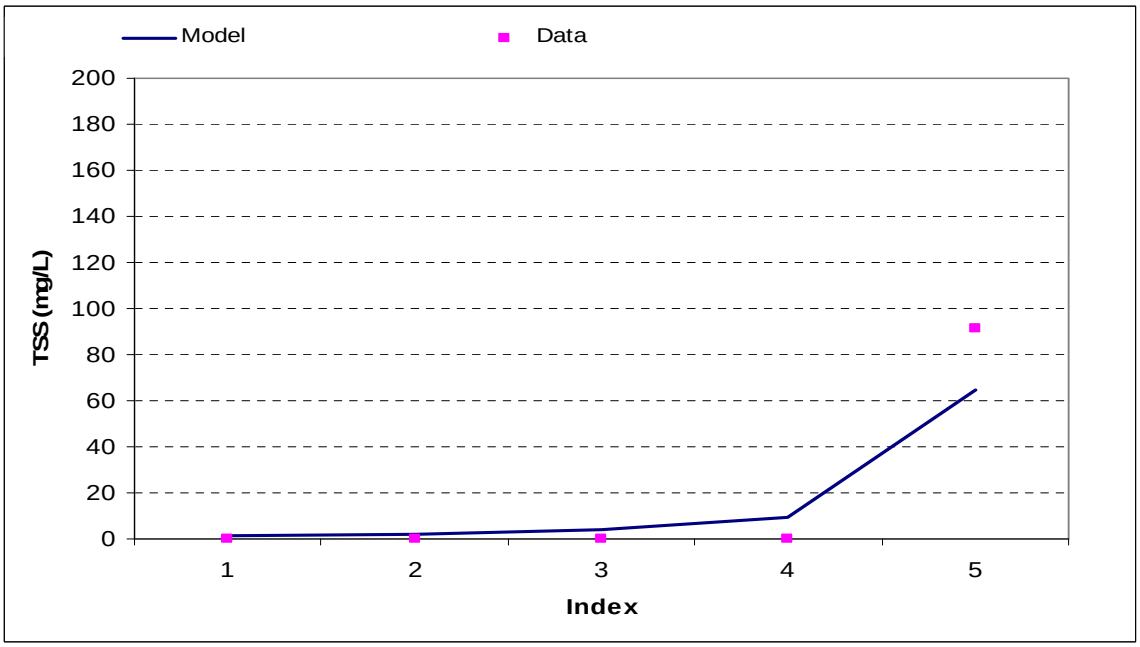

Figure C-7. Comparison of simulated and observed TSS concentration at the mouth of Chollas Creek, Trackline 7.

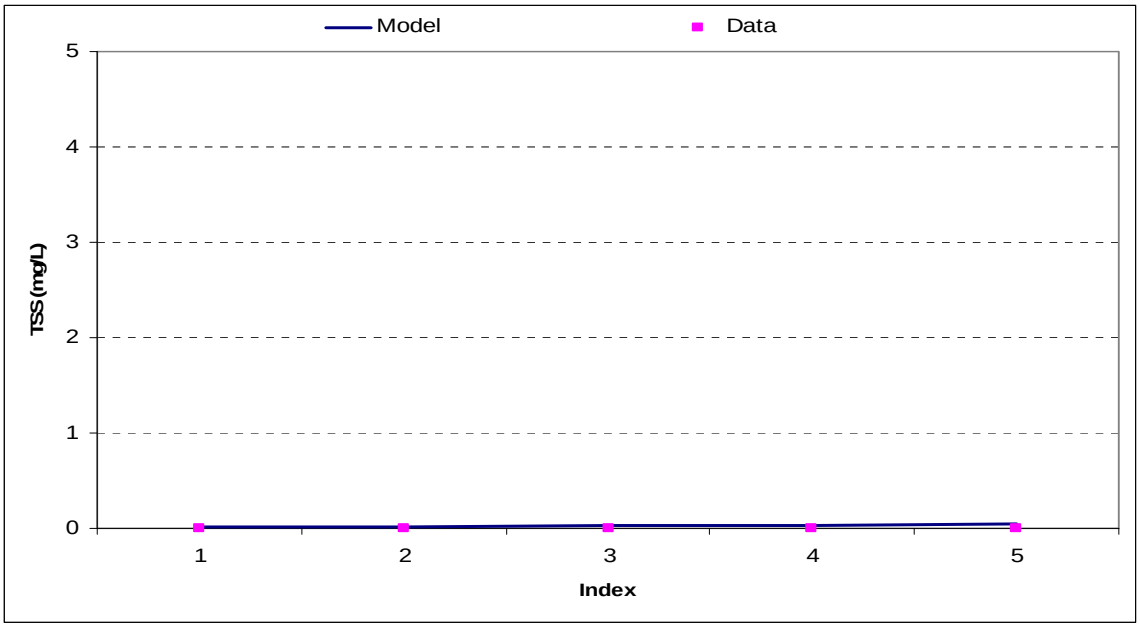

Figure C-8 Comparison of simulated and observed TSS concentration at the mouth of Paleta Creek, Trackline 1.

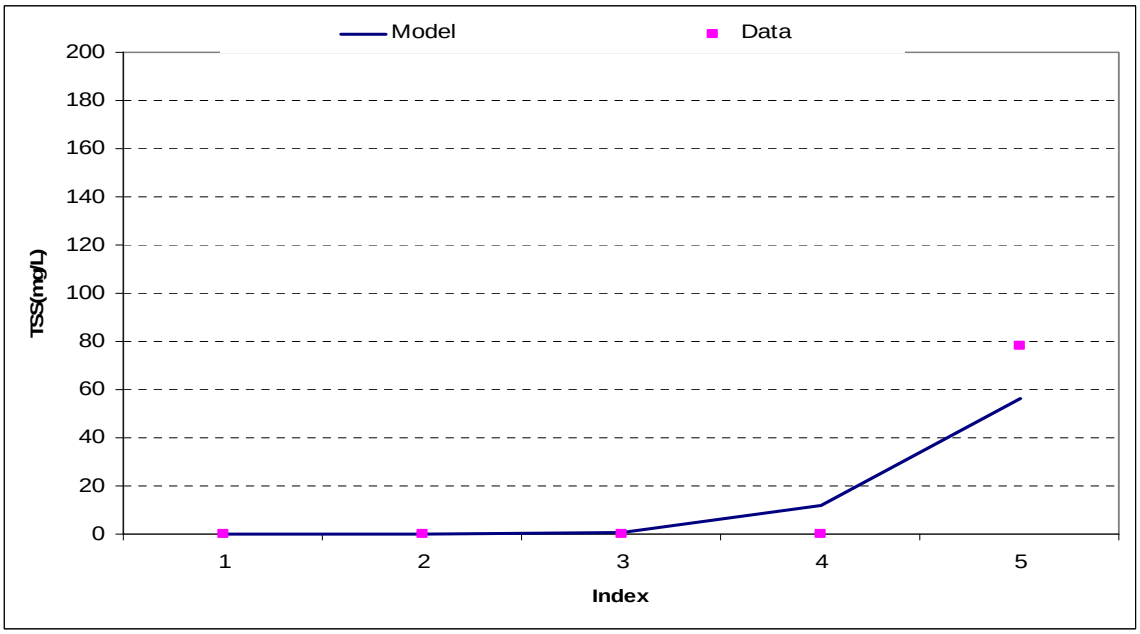

Figure C-9 Comparison of simulated and observed TSS concentration at the mouth of Paleta Creek, Trackline 2.

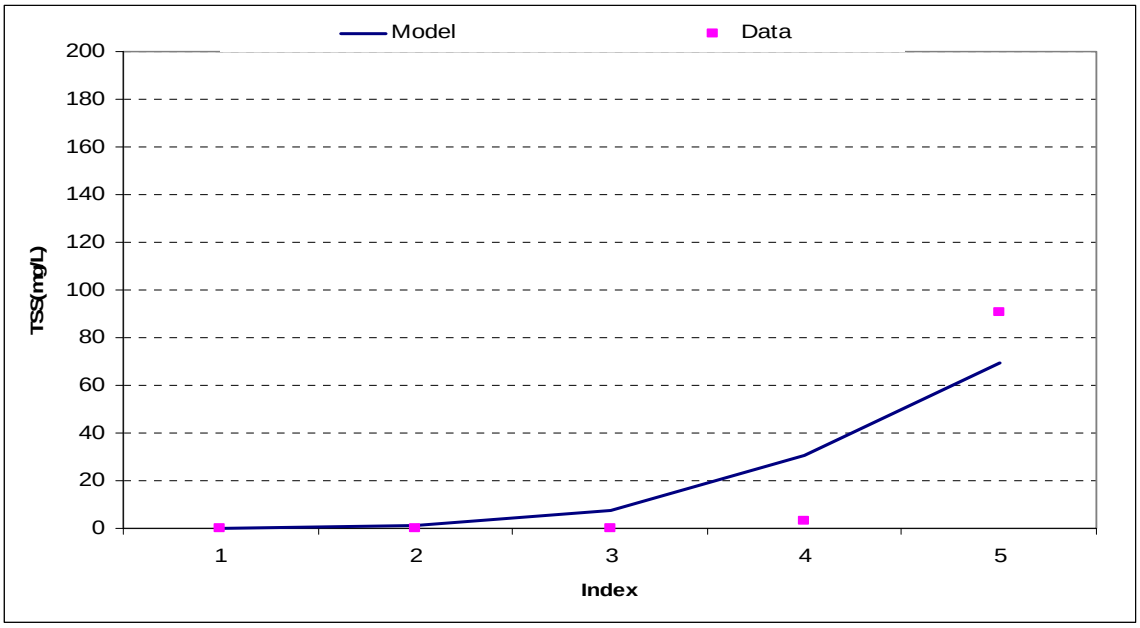

Figure C-10 Comparison of simulated and observed TSS concentration at the mouth of Paleta Creek, Trackline 3.

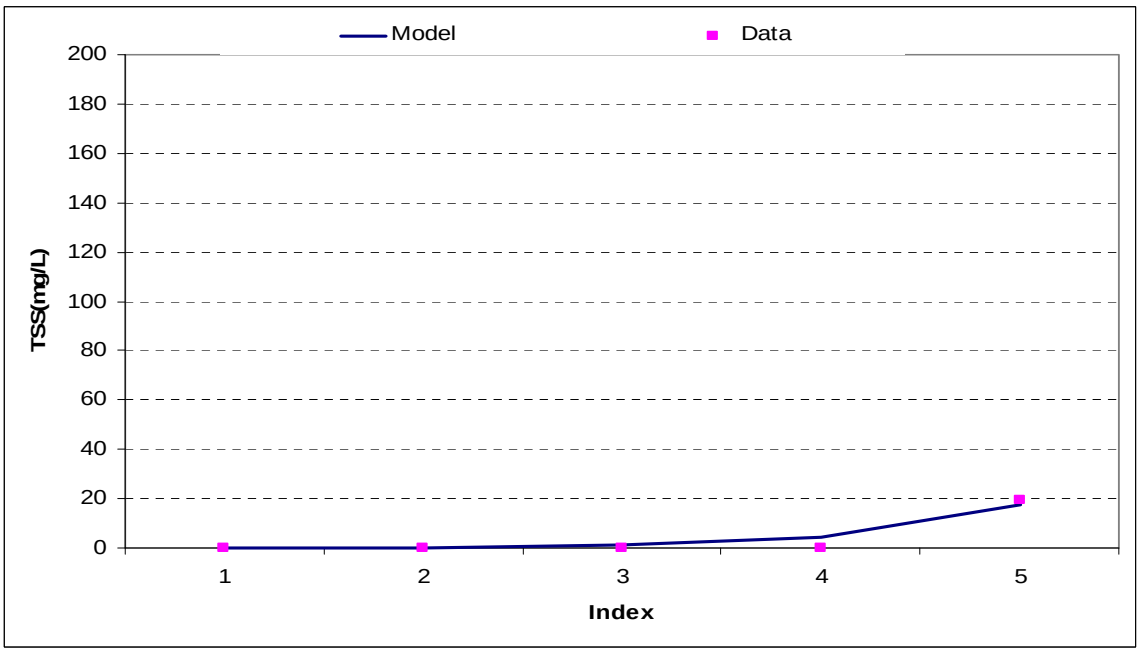

Figure C-11 Comparison of simulated and observed TSS concentration at the mouth of Paleta Creek, Trackline 4.

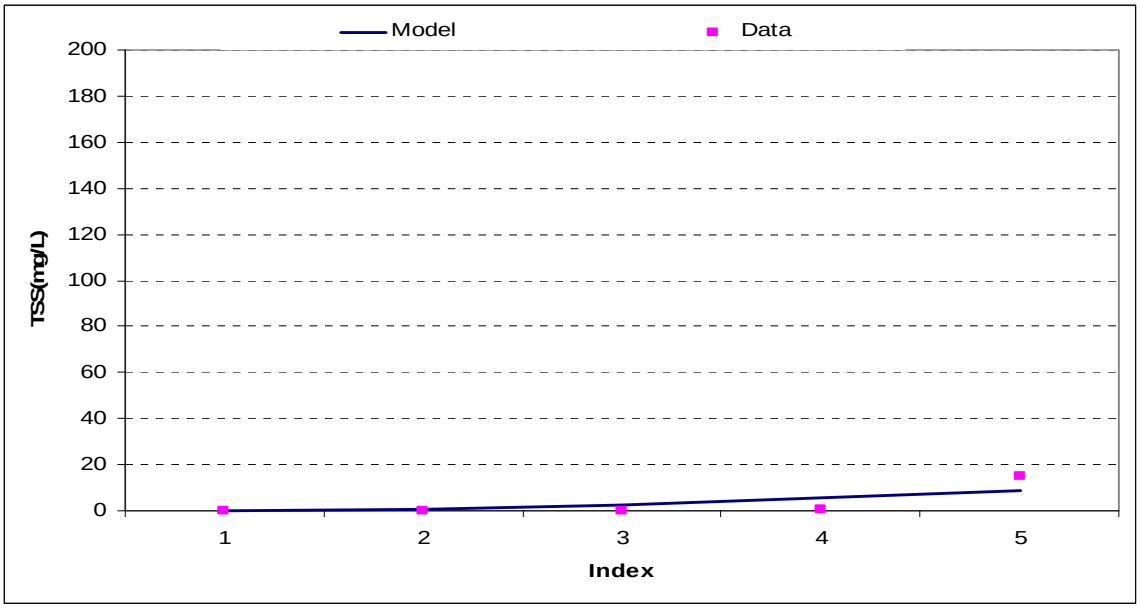

Figure C-12. Comparison of simulated and observed TSS concentration at the mouth of Paleta Creek, Trackline 5.

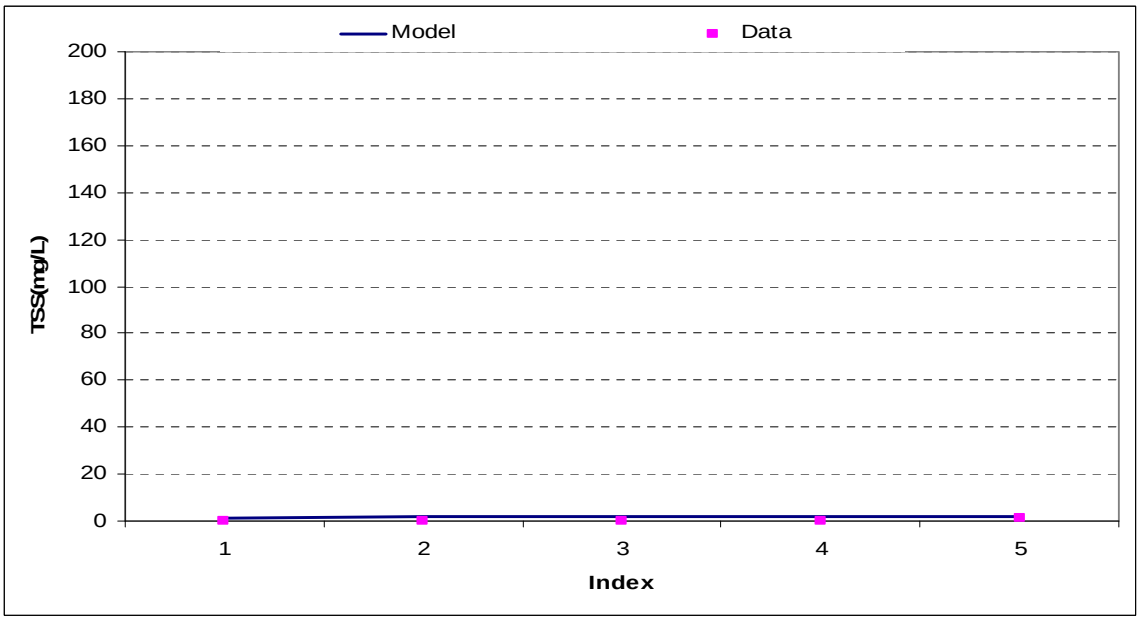

Figure C-13. Comparison of simulated and observed TSS concentration at the mouth of Paleta Creek, Trackline 6.

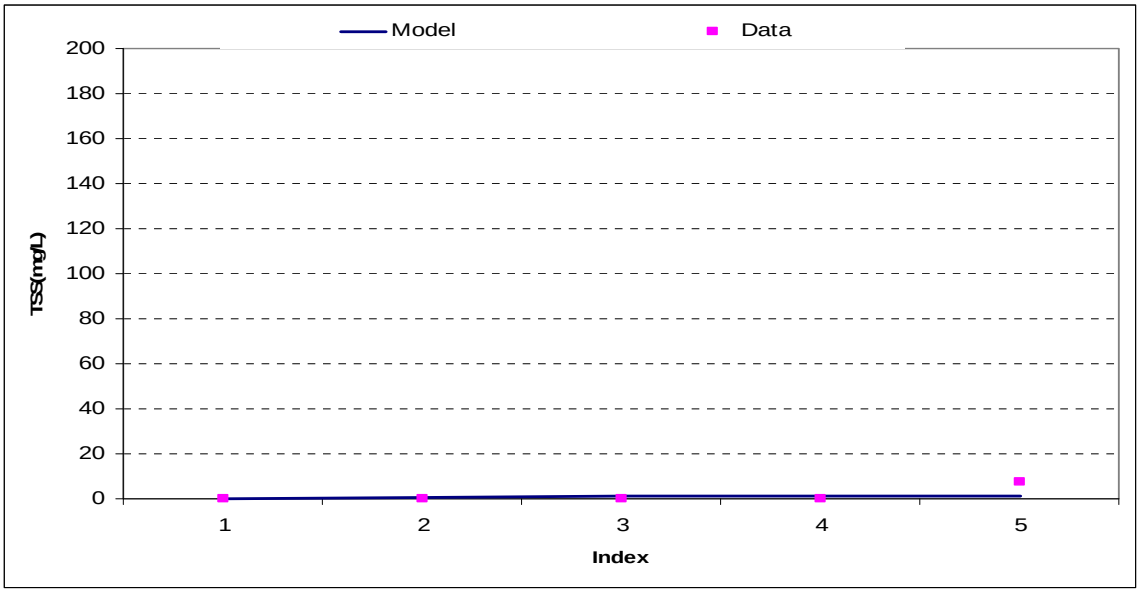

Figure C-14. Comparison of simulated and observed TSS concentration at the mouth of Paleta Creek, Trackline 7.

Appendix D

Calibration Plots for the EFDC Toxics Model at the Mouths of Chollas and Paleta Creeks

Figure D-1. Comparison of minimum simulated and observed toxics concentrations at the mouth of Chollas Creek, Station C14. 1
Figure D-2. Comparison of maximum simulated and observed toxics concentrations at the mouth of Chollas Creek, Station C14. 1
Figure D-3. Comparison of minimum simulated and observed toxics concentrations at the mouth of Chollas Creek, Station C10. 2
Figure D-4. Comparison of maximum simulated and observed toxics concentrations at the mouth of Chollas Creek, Station C10. 2
Figure D-5. Comparison of minimum simulated and observed toxics concentrations at the mouth of Paleta Creek, Station P11. 3
Figure D-6. Comparison of maximum simulated and observed toxics concentrations at the mouth of Paleta Creek, Station P11. 3
Figure D-7. Comparison of minimum simulated and observed toxics concentrations at the mouth of Paleta Creek, Station P17. 4
Figure D-8. Comparison of maximum simulated and observed toxics concentrations at the mouth of Paleta Creek, Station P17. 4

Please note that on the X-axis of the following figures, 1=TPAH, 2=TPCB, and 3=chlordane.

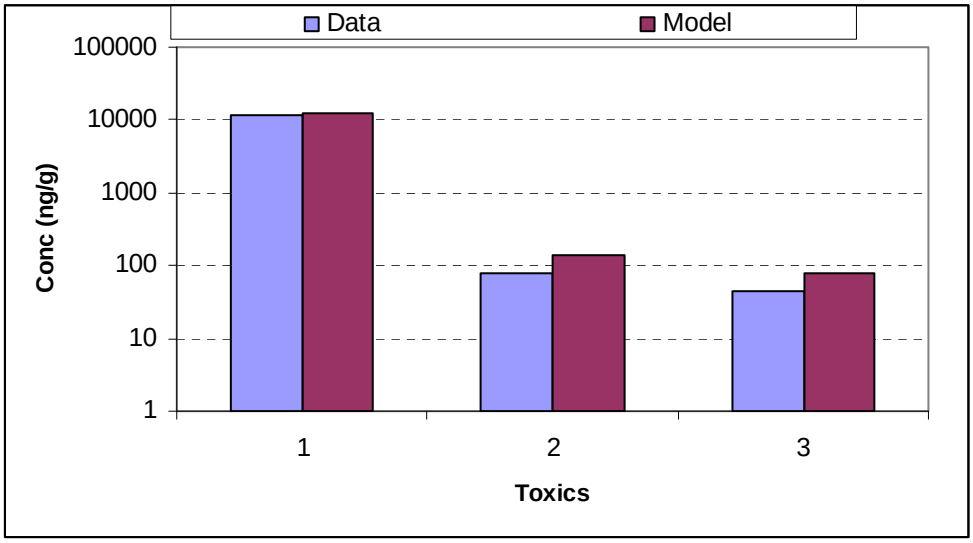

Figure D-1. Comparison of minimum simulated and observed toxics concentrations at the mouth of Chollas Creek, Station C14.

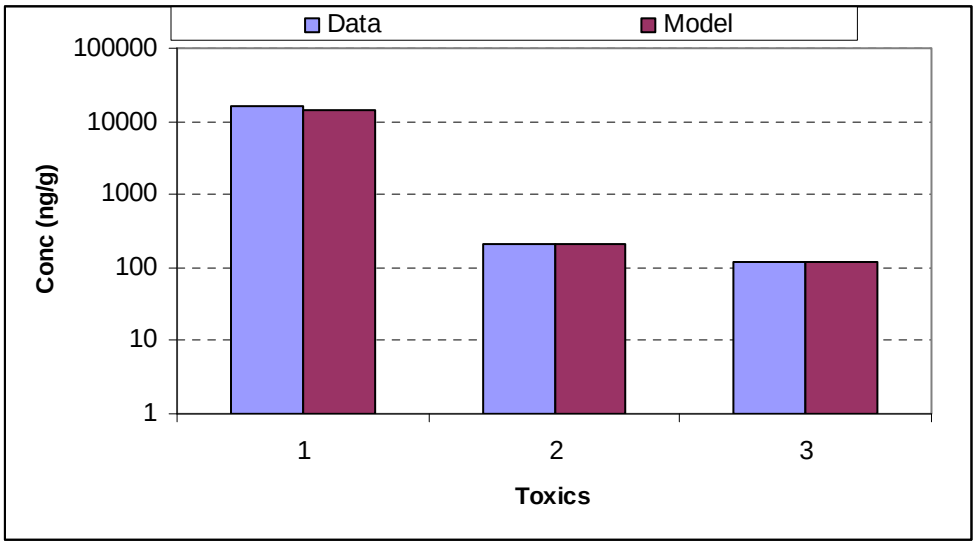

Figure D-2. Comparison of maximum simulated and observed toxics concentrations at the mouth of Chollas Creek, Station C14.

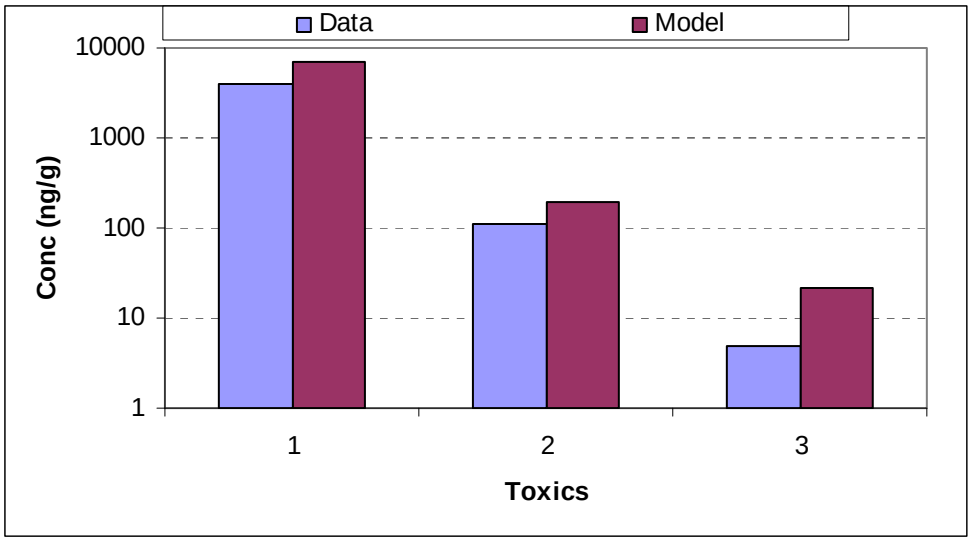

Figure D-3. Comparison of minimum simulated and observed toxics concentrations at the mouth of Chollas Creek, Station C10.

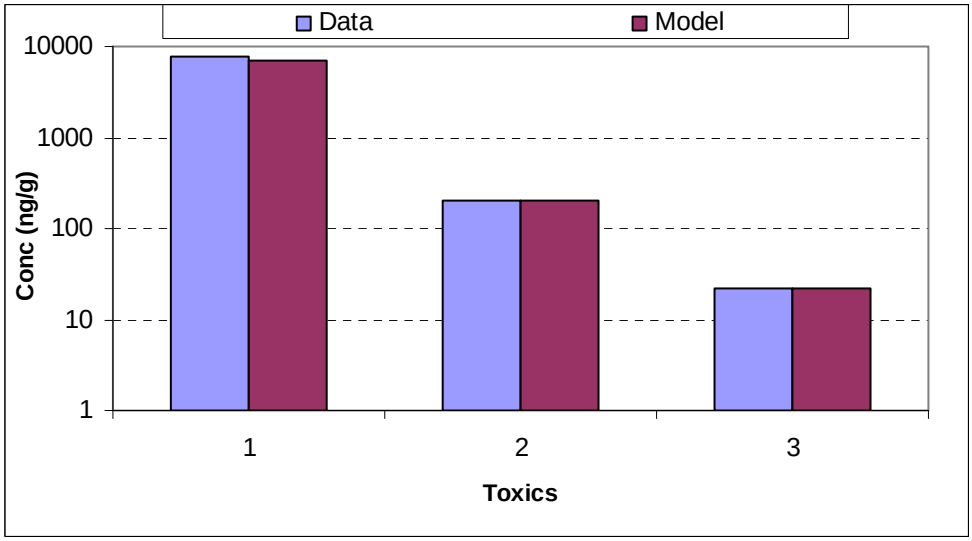

Figure D-4. Comparison of maximum simulated and observed toxics concentrations at the mouth of Chollas Creek, Station C10.

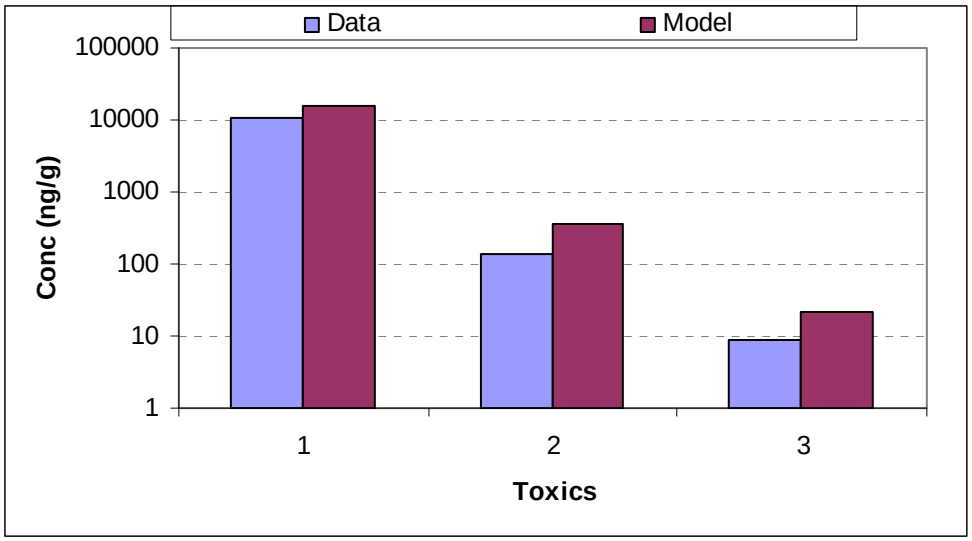

Figure D-5. Comparison of minimum simulated and observed toxics concentrations at the mouth of Paleta Creek, Station P11.

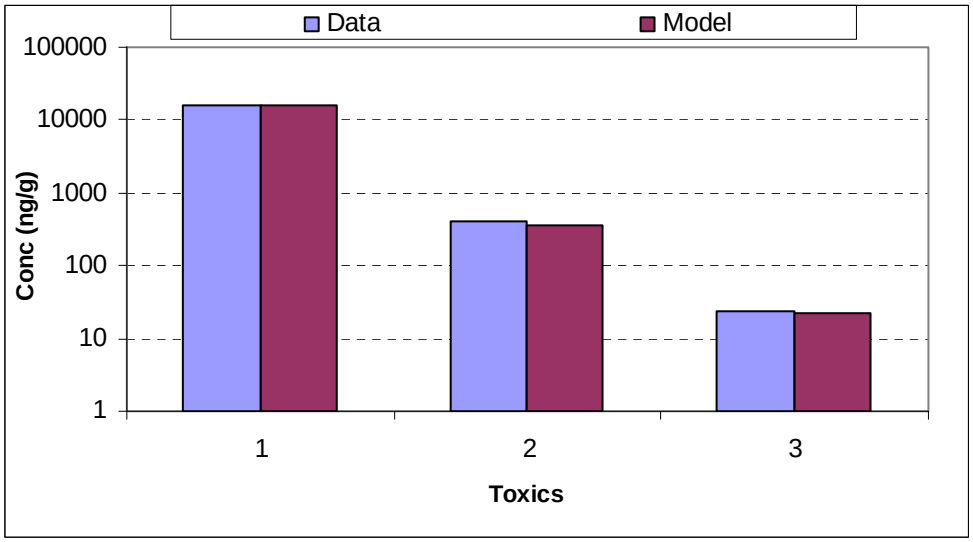

Figure D-6. Comparison of maximum simulated and observed toxics concentrations at the mouth of Paleta Creek, Station P11.

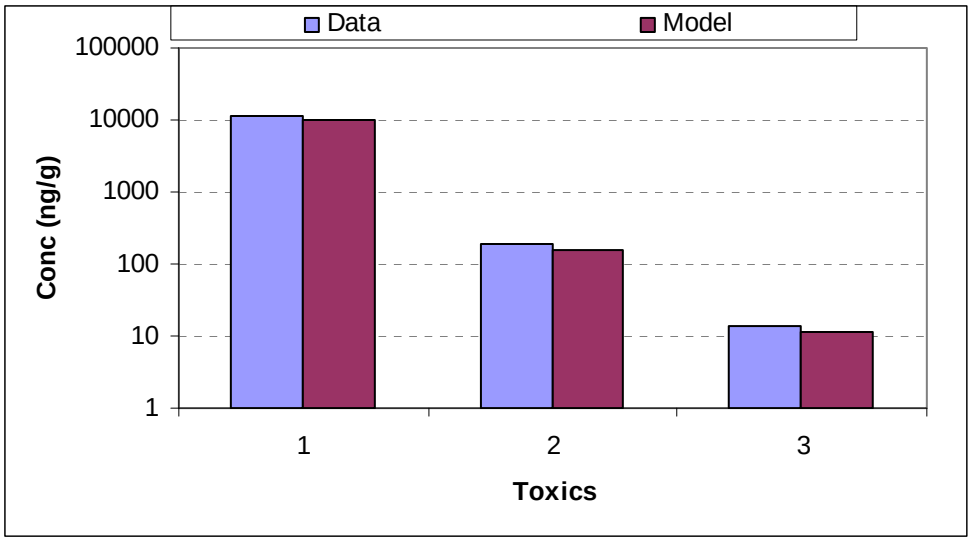

Figure D-7. Comparison of minimum simulated and observed toxics concentrations at the mouth of Paleta Creek, Station P17.

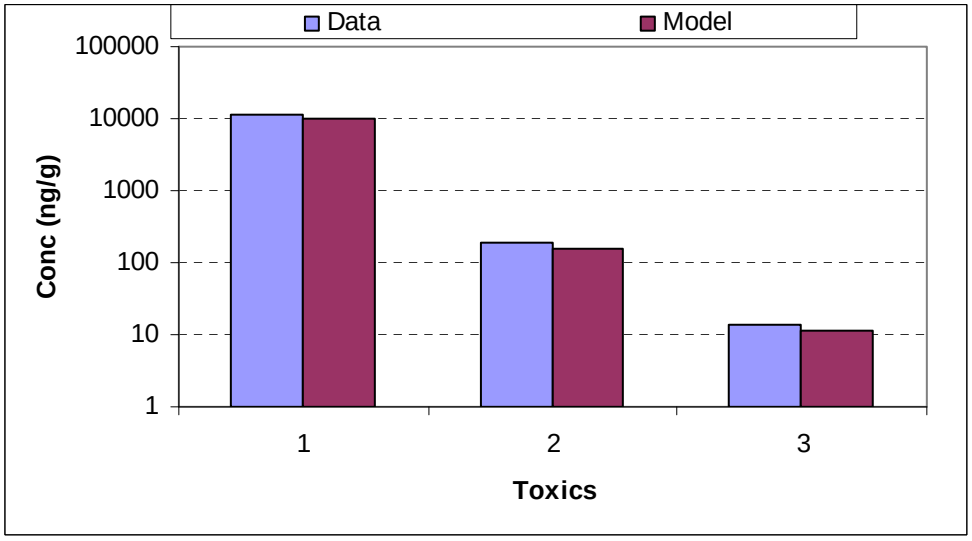

Figure D-8. Comparison of maximum simulated and observed toxics concentrations at the mouth of Paleta Creek, Station P17.

Appendix E

Sensitivity Plots for the EFDC Toxics Model at the Impaired Shoreline Areas

Figure E-1. Comparison of TPAH concentrations at the Paleta Creek mouth for the baseline and sensitivity runs. 1
Figure E-2. Comparison of TPAH concentrations at the Chollas Creek mouth for the baseline and sensitivity runs. 1
Figure E-3. Comparison of TPAH concentrations at the Switzer Creek mouth for the baseline and sensitivity runs. 2
Figure E-4. Comparison of TPAH concentrations at the B-Street mouth for baseline and sensitivity runs. 2
Figure E-5. Comparison of TPAH concentrations at the Downtown Anchorage mouth for the baseline and sensitivity runs. 3

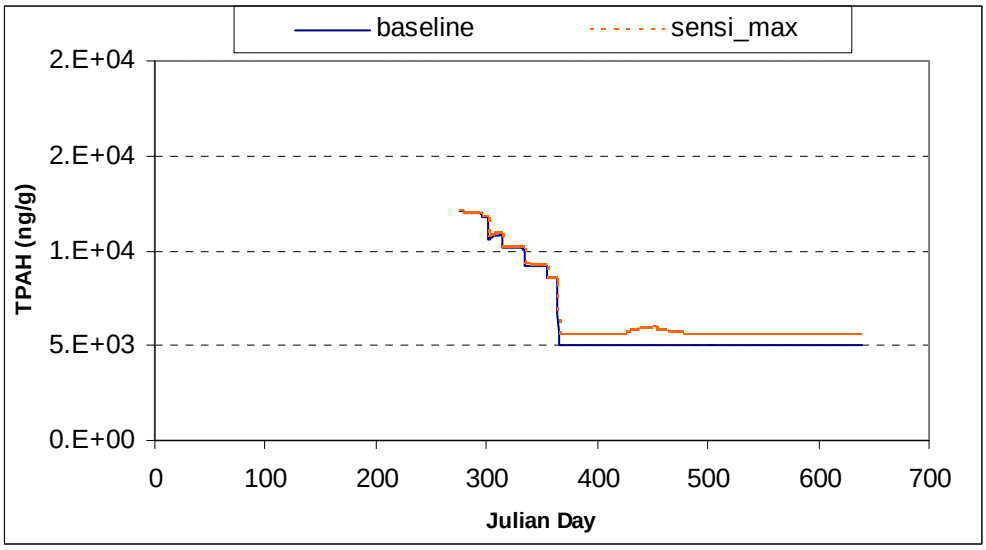

Figure E-1. Comparison of TPAH concentrations at the Paleta Creek mouth for the baseline and sensitivity runs.

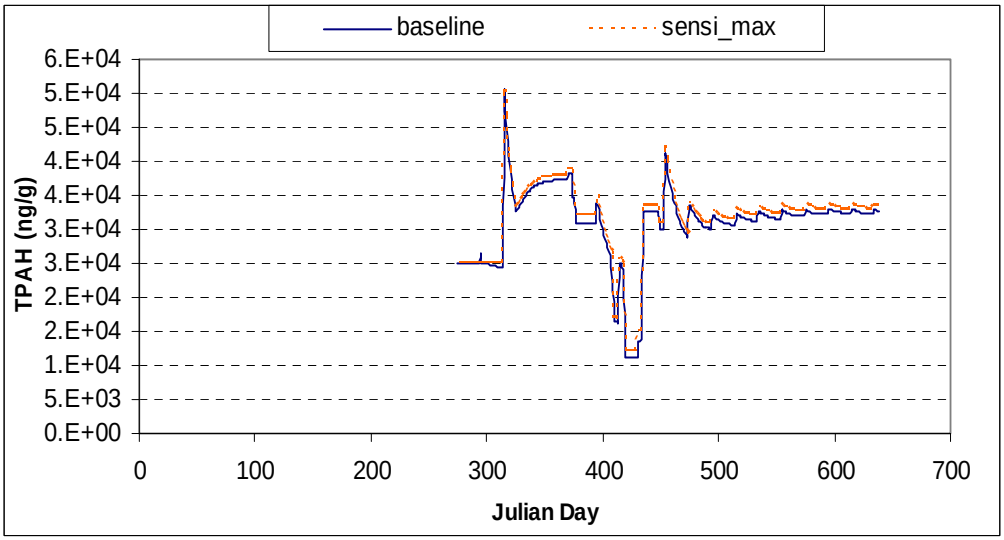

Figure E-2. Comparison of TPAH concentrations at the Chollas Creek mouth for the baseline and sensitivity runs.

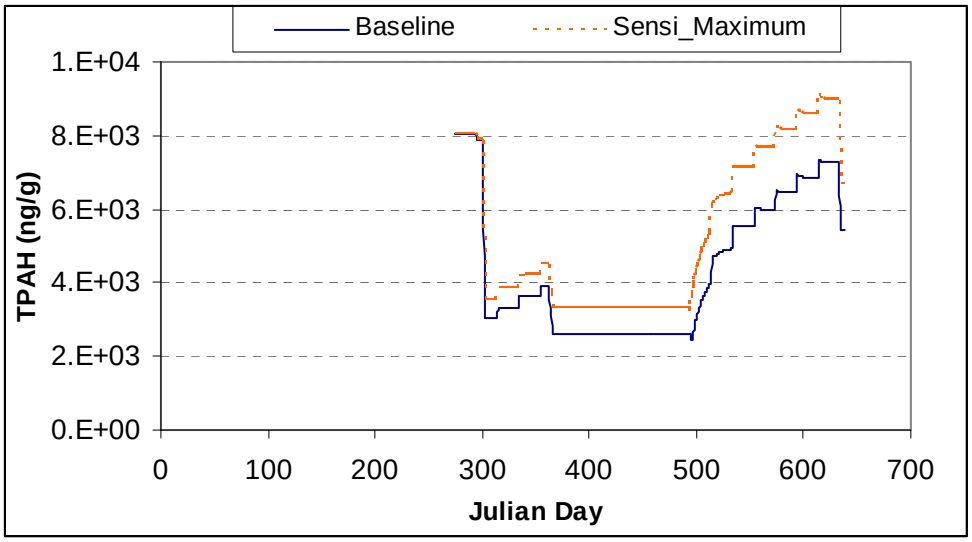

Figure E-3. Comparison of TPAH concentrations at the Switzer Creek mouth for the baseline and sensitivity runs.

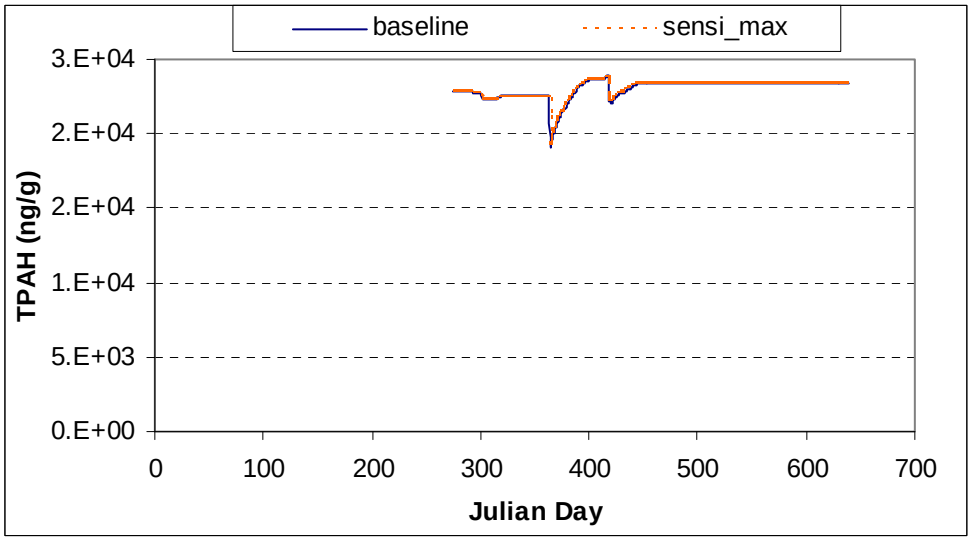

Figure E-4. Comparison of TPAH concentrations at the B-Street mouth for baseline and sensitivity runs.

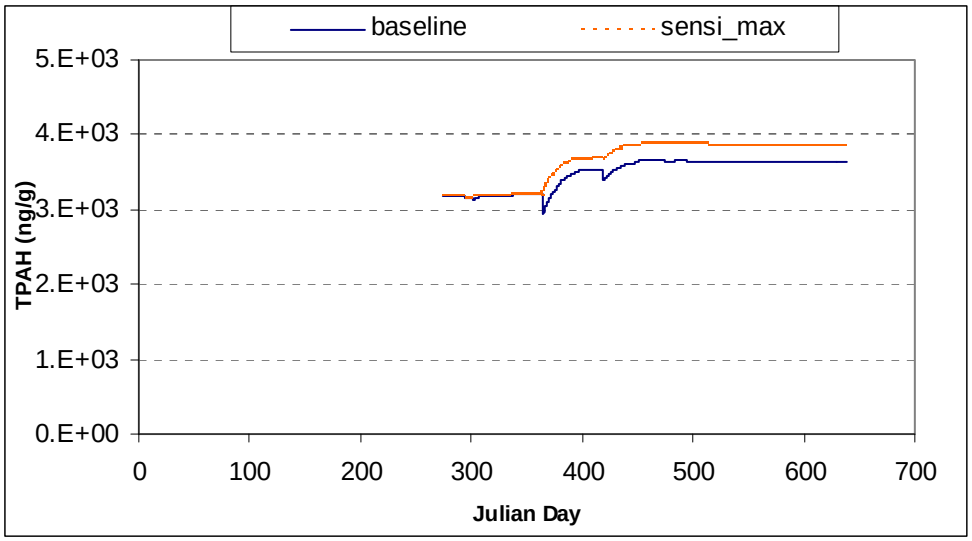

Figure E-5. Comparison of TPAH concentrations at the Downtown Anchorage mouth for the baseline and sensitivity runs.

Appendix F

Time Variable Sediment Toxicity Response to Baseline Loading Plots for the EFDC Toxics Model at the Impaired Shoreline Areas

Figure F-1. Bed TPAH concentrations at the Paleta Creek mouth for the baseline loading condition. 1
Figure F-2. Bed PCB concentrations at the Paleta Creek mouth for the baseline loading condition. 1
Figure F-3. Bed chlordane concentrations at the Paleta Creek mouth for the baseline loading condition. 2
Figure F-4. Bed TPAH concentrations at the Switzer Creek mouth for the baseline loading condition. 2
Figure F-5. Bed TPCB concentrations at the Switzer Creek mouth for the baseline loading condition. 3
Figure F-6. Bed chlordane concentrations at the Switzer Creek mouth for the baseline loading condition. 3
Figure F-7. Bed lindane concentrations at the Switzer Creek mouth for the baseline loading condition. 4
Figure F-8. Bed TPAH concentrations at the Downtown Anchorage mouth for the baseline loading condition. 4
Figure F-9. Bed TPCB concentrations at the Downtown Anchorage mouth for the baseline loading condition. 5
Figure F-10. Bed chlordane concentrations at the Downtown Anchorage mouth for the baseline loading condition. 5

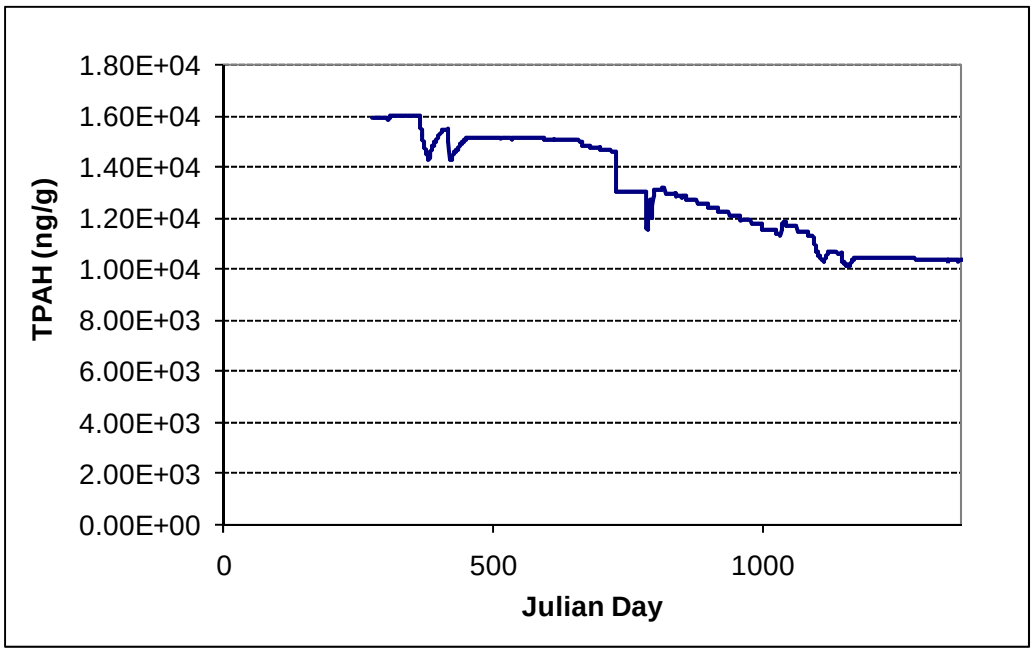

Figure F-1. Bed TPAH concentrations at the Paleta Creek mouth for the baseline loading condition.

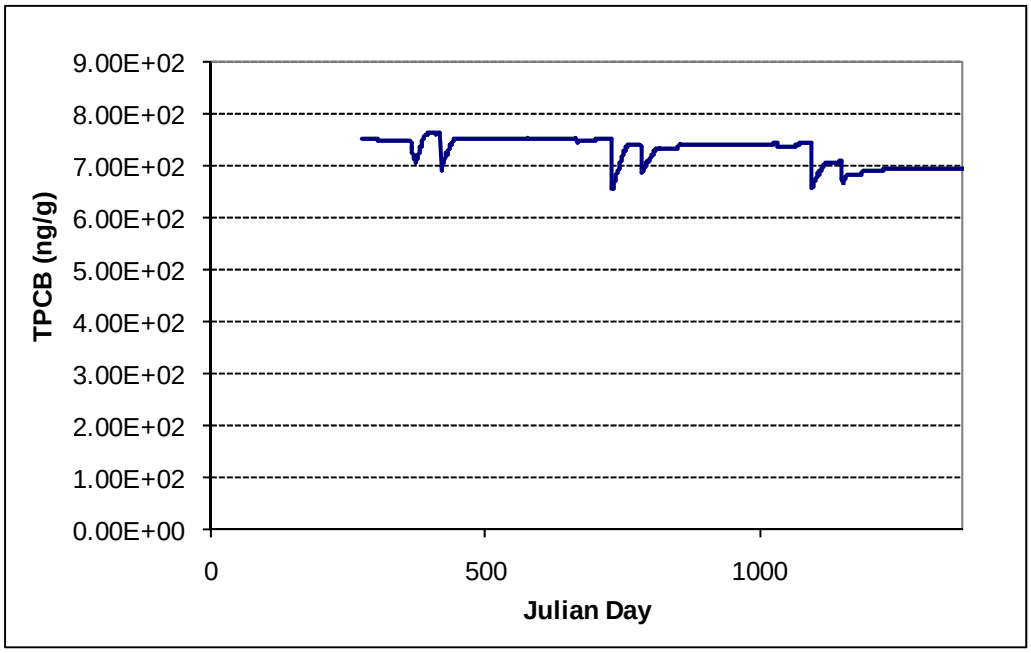

Figure F-2. Bed PCB concentrations at the Paleta Creek mouth for the baseline loading condition.

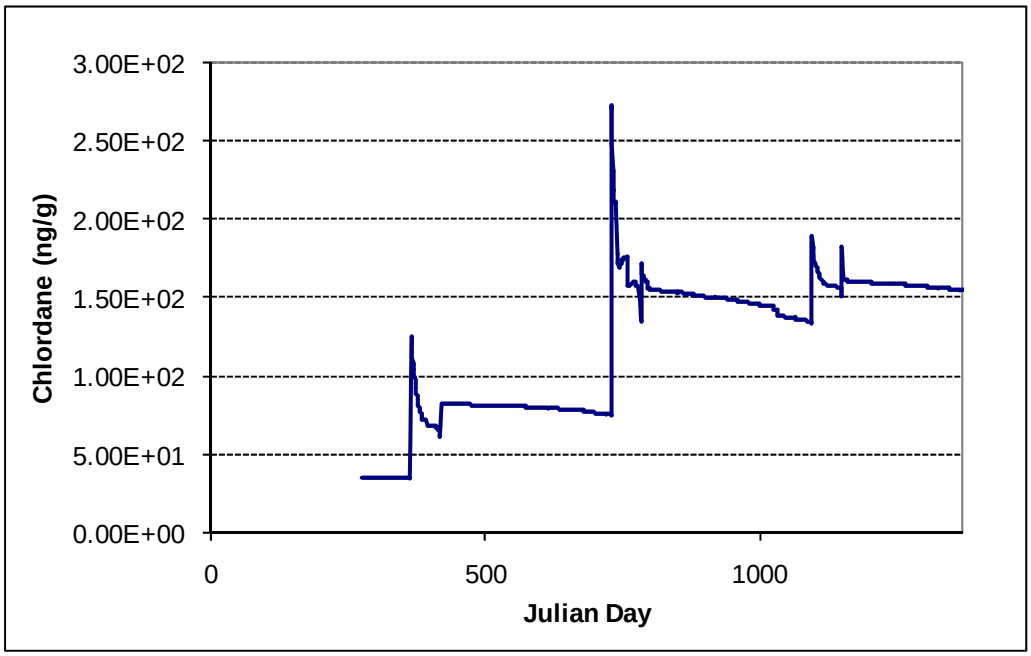

Figure F-3. Bed chlordane concentrations at the Paleta Creek mouth for the baseline loading condition.

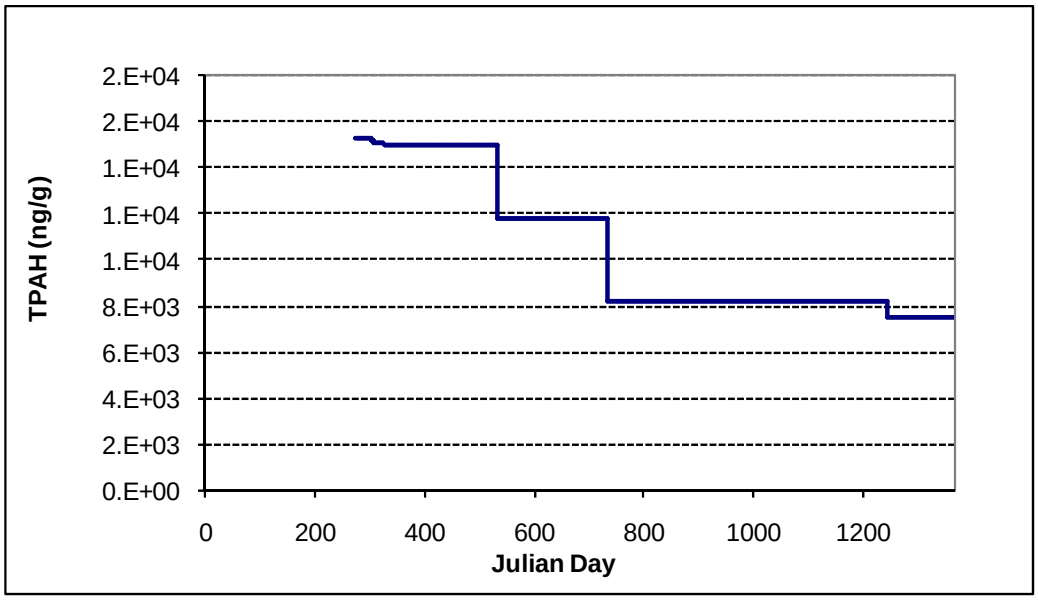

Figure F-4. Bed TPAH concentrations at the Switzer Creek mouth for the baseline loading condition.

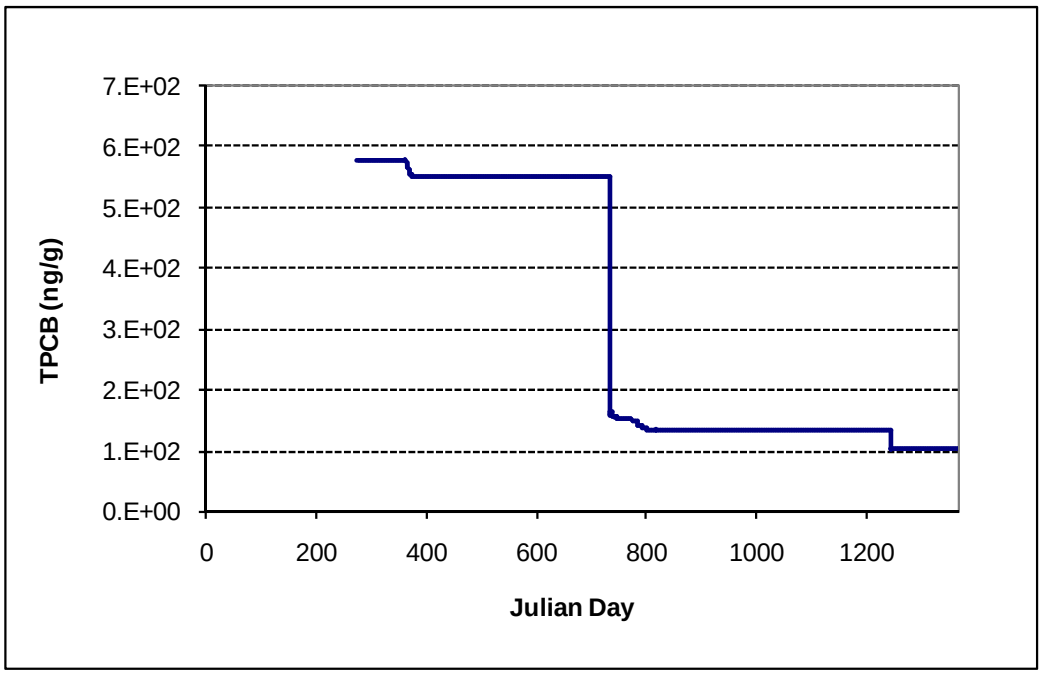

Figure F-5. Bed TPCB concentrations at the Switzer Creek mouth for the baseline loading condition.

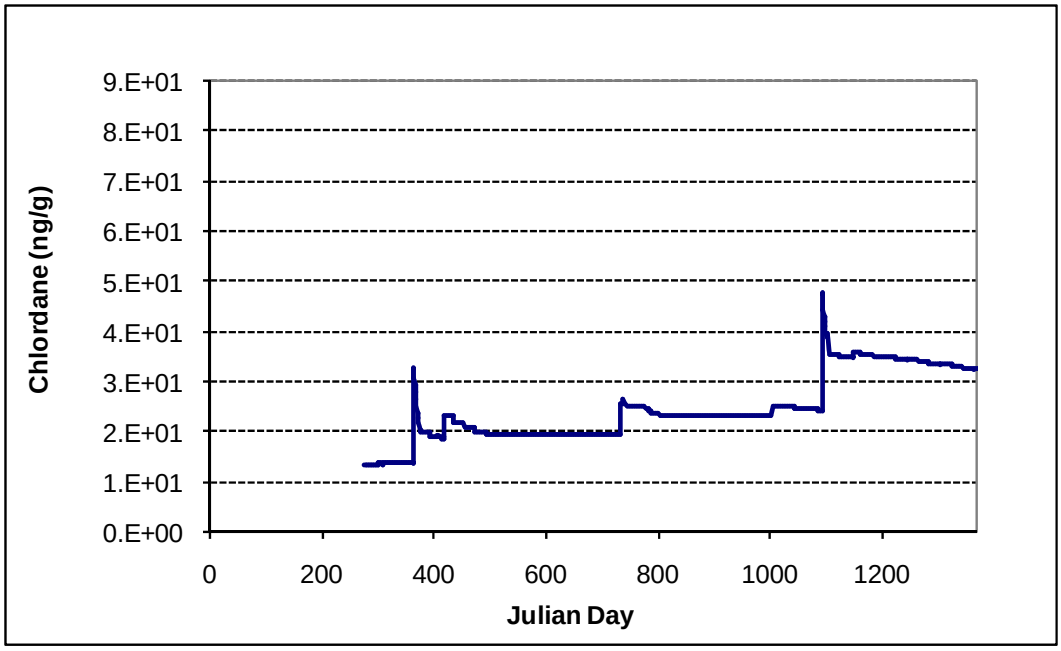

Figure F-6. Bed chlordane concentrations at the Switzer Creek mouth for the baseline loading condition.

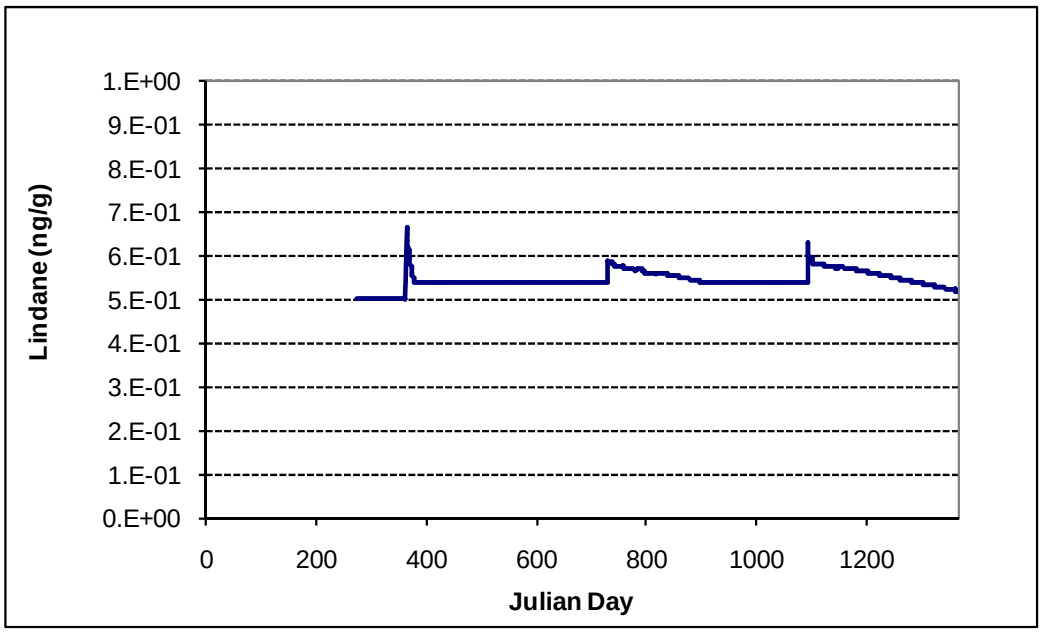

Figure F-7. Bed lindane concentrations at the Switzer Creek mouth for the baseline loading condition.

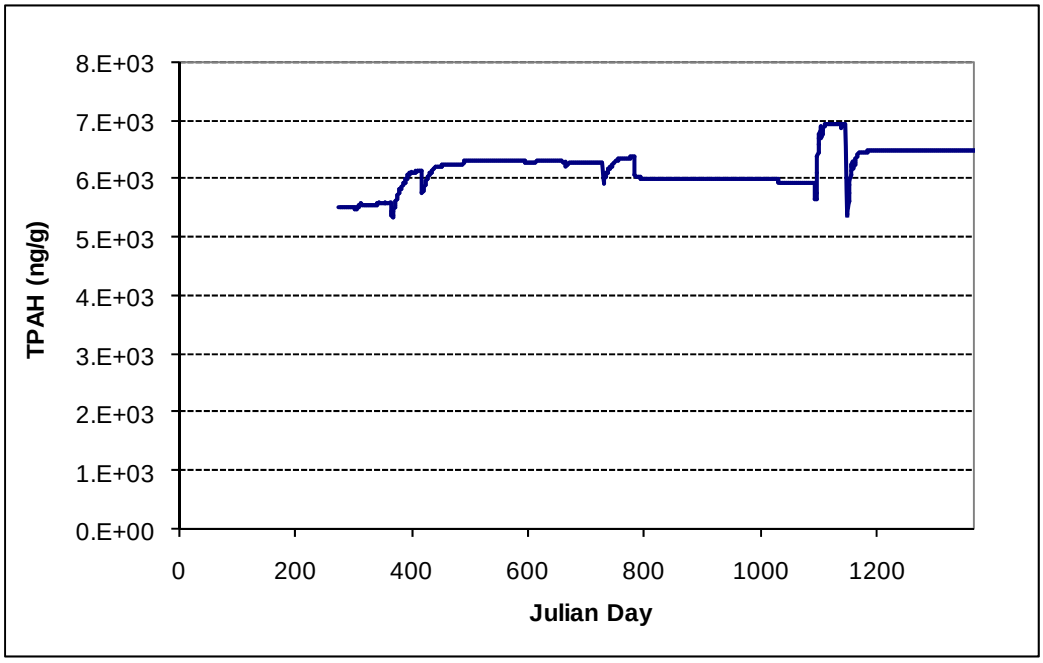

Figure F-8. Bed TPAH concentrations at the Downtown Anchorage mouth for the baseline loading condition.

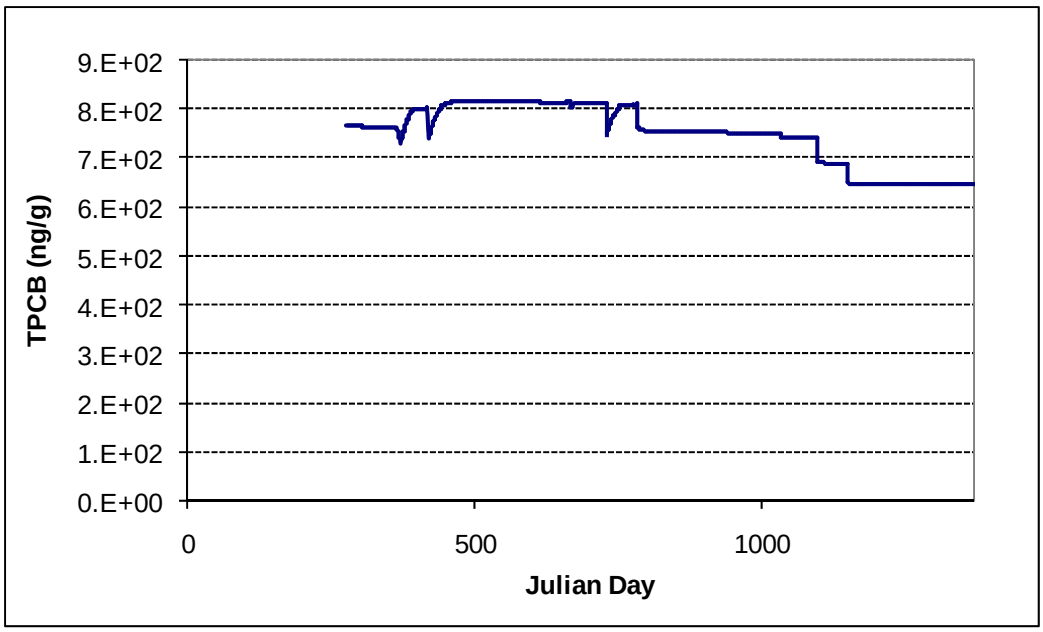

Figure F-9. Bed TPCB concentrations at the Downtown Anchorage mouth for the baseline loading condition.

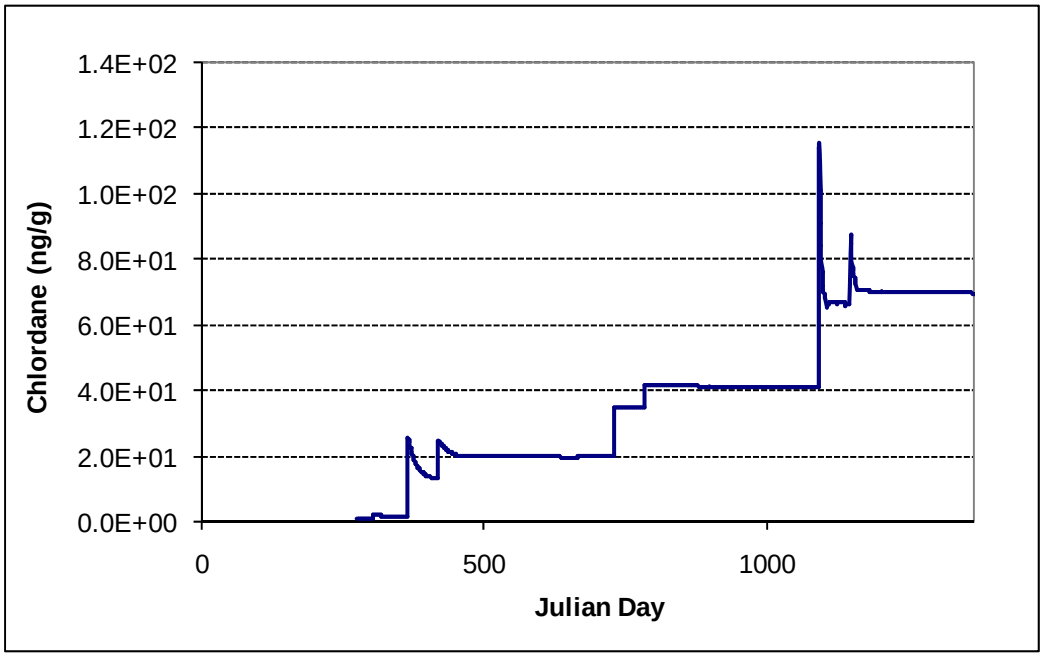

Figure F-10. Bed chlordane concentrations at the Downtown Anchorage mouth for the baseline loading condition.