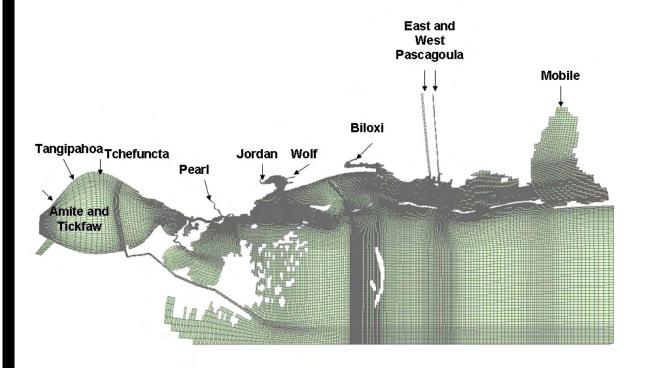


Application of a Water Quality Model to Mississippi Sound to Evaluate Impacts of Freshwater Diversions

Mark S. Dortch, Mansour Zakikhani, Mark R. Noel, and Sung-Chan Kim

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Abstract: This report describes the development and application of a water quality model to the Mississippi Sound region to address the impacts of various freshwater diversion alternatives. The CH3D-Sigma (sigma level vertical coordinates) model code was the hydrodynamic model that was used to provide transport fluxes for the CE-QUAL-ICM water quality model. The model domain also included Mobile Bay, the Mississippi coastal bays, Lakes Pontchartrain and Borgne, Biloxi Marsh, and part of Breton Sound. The three-dimensional model had five sigma coordinate vertical layers. The model included 15 water quality variables including temperature, salinity, inorganic and total suspended solids, dissolved oxygen, dissolved and particulate organic carbon, various forms of inorganic and organic nitrogen and phosphorus, phytoplankton biomass, chlorophyll a, and underwater light extinction. The model was calibrated for the period April through September 1998. Three diversions were simulated, diversion of freshwater flow from the Mississippi River at Bonnet Carre' spillway and into Lake Borgne near Violet, LA, and diversion of all of the Escatawpa River flow into Grand Bay. Summer average salinity was decreased along the western portions of Mississippi Sound by as much as 11 parts per thousand for the Bonnet Carre' diversion. For the Violet diversion, summer average salinity reductions were as great as 6 to 8 parts per thousand in western Mississippi Sound. The Escatawpa River diversion had little effect on Mississippi Sound. The Mississippi River diversion will also result in higher concentrations of nutrients, TSS, phytoplankton, and TOC, and greater light extinction, thus, less light reaching the bottom.

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Preface

This study was conducted by the U.S. Army Engineer Research and Development Center (ERDC) at the request of the U.S. Army Engineer District, Mobile. The study was conducted to help assess impacts of various freshwater diversion alternatives as part of the U.S. Army Corps of Engineers (USACE) Mississippi Coastal Improvements Program (MSCIP). Dr. Susan Rees was the Mobile District's point of contact (POC) for ERDC on this study.

Drs. Mark S. Dortch and Mansour Zakikhani conducted this study, with assistance from Mark Noel and Dr. Sung-Chan Kim, all of the Water Quality and Contaminant Modeling Branch (WQCMB), Environmental Processes and Engineering Division (EPED), of the Environmental Laboratory (EL), ERDC, Vicksburg, MS. Dortch and Zakikhani prepared this report. Dortch served as the model study POC, and Dr. Barbara Kleiss of the Wetlands and Coastal Ecology Branch, Ecosystem Evaluation and Engineering Division, EL, ERDC, served as the ERDC POC for MSCIP. This work was conducted under the general supervision of Dr. Barry Bunch, Chief, WQCMB, and Dr. Richard E. Price, Chief, EPED. Modeling suggestions by Dr. Carl Cerco of WQCMB during the course of the study are acknowledged and appreciated. Dr. Beth Fleming was Director of EL.

COL Richard B. Jenkins was Commander and Executive Director of ERDC. Dr. James R. Houston was Director.

1 Introduction

Background

In response to major damages on the Mississippi coast caused by Hurricane Katrina, Congress directed the U.S. Army Corps of Engineers (USACE) to conduct the Mississippi Sound Coastal Improvements Program (MSCIP), which is an analysis and design for comprehensive improvements, or modifications to existing improvements, in the coastal areas of Mississippi in the interests of (1) hurricane storm damage reduction, (2) prevention of saltwater intrusion, (3) preservation of fish and wildlife, (4) prevention of erosion, and (5) other related water resource purposes. Several measures are under consideration for restoring resources along the coast including construction of dunes, seawalls, and levees onshore; development of surge mitigation measures; wetland and ecosystem restoration; barrier island and beach restoration; and freshwater diversion.

Staff of the U.S. Army Engineer District, Mobile, met with staff of the U.S. Army Engineer Research and Development Center (ERDC) and Louisiana State University (LSU) on 5–6 June 2006 to discuss the MSCIP work and methods to aid in evaluating improvement alternatives. The discussion focused primarily on the use of mathematical models to help evaluate the effects of barrier island restoration and freshwater diversion. This report describes the development and application of a water quality model (WQM) of the Mississippi Sound region to address the impacts of various freshwater diversion alternatives.

Freshwater diversions will not only lower salinity but can also increase nutrient and suspended solids concentrations due to anthropogenic loadings into major rivers, such as the Mississippi River, that may be used for such diversions. Higher nutrient concentrations, primarily nitrogen, can fuel larger phytoplankton blooms. Increased suspended solids and phytoplankton biomass can reduce the amount of light available for submerged aquatic vegetation (SAV), which is important as habitat for living resources. Increased eutrophication, concomitant with more turbid water and elevated algal concentrations, is generally considered undesirable for environmental quality. There is also the possibility of lower dissolved oxygen during periods when the water column may stratify. Thus, any

considerations for freshwater diversions should include an analysis of the impacts on water quality.

Objective and scope

The objective of this study was to develop and apply a WQM of the Mississippi Sound and surrounding region to provide key information for evaluating coastal restoration and improvement alternatives. The results presented in this report focus on the water quality conditions that could be imposed with three freshwater diversion alternatives that are being considered.

The scope of this study was limited to providing enough information to allow for screening of alternatives and evaluating the sensitivity of the system to diversions, not to provide refined forecasts of future water quality conditions with diversions. The model will require more attention to provide refined forecasts, and such effort may be warranted if preferred alternatives progress to a more in-depth level of analysis and plan formulation.

Approach

ERDC previously developed an indirectly coupled, three-dimensional (3-D) hydrodynamic and WQM of a portion of Mississippi Sound for the Mobile District to aid in evaluating dredging and dredge material disposal impacts resulting from channel widening and/or deepening for the Port of Gulfport, Mississippi (Bunch et al. 2005). Dredging and disposal activities have the possibility of impacting localized transport of pollutants, increasing sediment oxygen demand (SOD) in locations above and adjacent to disposal sites, releasing nutrients to the water column, and altering salinity levels in the study area. The CH3D-Sigma (sigma level vertical coordinates) model code was used for the hydrodynamic model (Chapman et al. 1996), and the CE-QUAL-ICM (ICM) model code was used for water quality. ICM was first developed for Chesapeake Bay (Cerco and Cole 1993) but has subsequently been used for many diverse systems. A more recent version of ICM (Cerco et al. 2004) was used for the present study. A detailed description of this version of ICM is described in the report by Cerco et al. (2004).

The two models are indirectly coupled such that the hydrodynamic model (HM) is executed first and output is saved and read into and used by the WQM when it is executed. Thus, there is no feedback from water quality to

hydrodynamics, which is not necessary since water quality (other than salinity and temperature) has no effect on circulation in the marine environment, and because salinity and temperature, which do affect circulation, are included in the HM simulation. The HM variables that are processed, saved, and read into the WQM include computational cell volumes, volumetric flows across cell faces, and vertical eddy diffusivities for each hydrodynamic update interval. Hydrodynamic information was processed within the HM to produce hourly averages for flows and diffusivities that were output for the WQM hourly updates. Cell volumes at the beginning of each update interval are output to check volumetric continuity of the flow field. Hourly updates have been found to accurately preserve all residual transport information.

Although the model of Bunch et al. (2005) focused on the Port of Gulfport, the domain extended from east of Mobile Bay (including Mobile Bay), south to the Chandeleur Islands, and west past the mouth of the Pearl River into eastern Lake Borgne. The Gulfport model domain and grid were expanded for the present study to more fully meet the needs of the MSCIP. The extension was southward beyond Chandeleur Islands and further west to include all of Lake Borgne and Lake Pontchartrain, the Inner Harbor Navigation Channel of New Orleans, and the Mississippi River Gulf Outlet channel. The revised grid, shown in Figure 1-1, consists of 172×405 rows and columns and 40,406 active computational cells in planform. Five vertical sigma layers were used resulting in a total of 202,030 active cells.

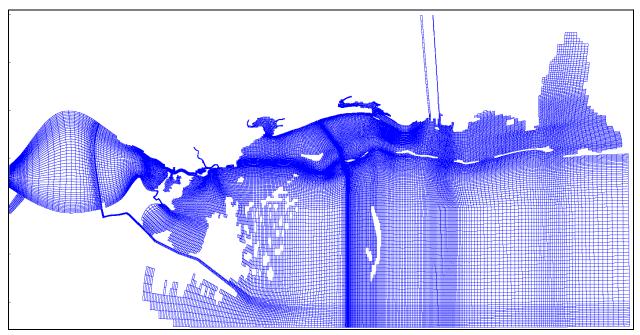


Figure 1-1. The model domain and computational grid.

Additional inputs of freshwater flows and tributary nutrient loads were included as required after extending the model domain. Model calibration was conducted again using the spring—summer 1998 conditions and observations, as was used for the Gulfport study. The WQM was executed for 1 April—30 September 1998 conditions when running baseline and scenario alternatives. The HM was run for the same period, except the month of March was also included to improve model spin-up.

Given the limited data availability and the limited scope of this study with the associated tight schedule, some of the more comprehensive WQM routines were not activated, such as the sediment diagenesis, benthic algae, and SAV routines. For these routines to provide added value, it would be necessary to simulate a multiyear period. Typically five or more years of simulation are required for bottom sediments to re-equilibrate for changing nutrient loads. Given the size of this grid, this would have required significant super-computing requirements and a longer study time with greater funding. Additionally, much more observational data would have been required for model calibration and validation than was available. The results of the WQM should still be representative of future alternative conditions given the mostly unstratified conditions of the Mississippi Sound, which diminishes the importance of sediment nutrient processes and their influence on the water column. The model also provides underwater light attenuation that can be used to infer impacts on SAV, which is of interest for habitat. If other model compartments are needed in a future study, they can be activated at that time.

2 Model Description

General overview

ICM was designed to be a flexible, widely applicable, state-of-the-art eutrophication model. Initial application was to Chesapeake Bay (Cerco and Cole 1993). Since the initial Chesapeake Bay study, the ICM model code has been generalized with various revisions and improvements. Subsequent additional applications of ICM included the Delaware Inland Bays (Cerco et al. 1994), Newark Bay (Cerco and Bunch 1997), the San Juan Bay Estuary (Bunch et al. 2000; Cerco et al. 2003), Florida Bay (Cerco et al. 2000), St. Johns River (Tillman et al. 2004), Pascagoula River Harbor (Bunch et al. 2003), Lake Washington (Cerco et al. 2004), the Port of Los Angeles (ongoing study), and other sites. Each model application employed a different combination of model features, and some applications required the addition of new capabilities to more fully capture the system dynamics.

General features of the model include the following:

- Model is operational in one-, two-, or three-dimensional (1-D, 2-D, or 3-D) configurations.
- User can include as many as 32 state variables.
- Sediment-water oxygen demand and nutrient fluxes may be computed in a predictive sub-model or specified as a flux.
- State variable may be individually activated or deactivated.
- User can conduct internal averaging of model output over arbitrary intervals.
- Model allows computation and reporting of concentrations, mass transport, kinetics transformations, and mass balances.
- Debugging aids include ability to activate and deactivate model features, diagnostic output, volumetric and mass balances.
- Model operates on a variety of computer platforms and is coded in ANSI Standard FORTRAN F77.

ICM is limited by not computing the hydrodynamics of the modeled system. Hydrodynamic variables (i.e., flows, vertical turbulent diffusion coefficients, and volumes) must be specified externally and read into the model. Hydrodynamics may be specified in binary or ASCII format and are usually obtained from a hydrodynamic model such as the CH3D model.

Conservation of mass equation

The foundation of ICM is the solution to the 3-D mass-conservation equation for a control volume. Control volumes correspond to cells on the model grid. ICM solves, for each volume and for each state variable, the equation

$$\frac{\delta V_j C_j}{\delta t} = \sum_{k=1}^n Q_k C_k + \sum_{k=1}^n A_k D_k \frac{\delta C}{\delta x_l} + \sum S_j$$
 (1-1)

in which:

 V_i = volume of jth control volume (m³)

 C_j = concentration in jth control volume (g m⁻³)

t, x = temporal and spatial coordinates

n = number of flow faces attached to jth control volume

 Q_k = volumetric flow across flow face k of jth control volume (m³ s⁻¹)

 C_k = concentration in flow across face k (g m⁻³)

 A_k = area of flow face k (m²)

 D_k = diffusion coefficient at flow face k (m² s⁻¹)

 S_j = external loads and kinetic sources and sinks in jth control volume (g s⁻¹).

Solution of Equation 1-1 on a digital computer requires discretization of the continuous derivatives and specification of boundary conditions, initial conditions, and parameter values. The equation is solved explicitly using an option of upwind differencing or the QUICKEST algorithm (Leonard 1979) to represent C_k . The QUICKEST algorithm reduces unwanted numerical diffusion and was used in this study. The time-step, determined by stability requirements, is usually on the order of 5–15 min. depending on the size of computational cells and magnitude of flow rates within the grid.

State variables

For this study, 14 state variables were activated and are listed in Table 2-1. From the variables computed by the model as shown in Table 2-1, information was derived for other water quality constituents, including total suspended solids, chlorophyll a, total organic carbon, total Kjeldahl nitrogen, underwater light extinction, and Secchi depth.

Table 2-1. Water quality model state variables for Mississippi Sound model.

Temperature	Salinity
Fixed Solids (inorganic suspended solids)	Dissolved Oxygen (DO)
Dissolved Organic Carbon (DOC)	Particulate Organic Carbon (POC)
Ammonium (NH ₄) Nitrogen	Nitrate + Nitrite Nitrogen (NO₃)
Dissolved Organic Nitrogen (DON)	Particulate Organic Nitrogen (PON)
Total Phosphate or Total Inorganic Phosphorous (TIP)	Dissolved Organic Phosphorus (DOP)
Particulate Organic Phosphorus (POP)	Phytoplankton Carbon

A limited number of variables and only one algal group were activated due to the limited amount of observed data needed for model calibration. Particulate organic components were lumped into a single labile compartment for each major nutrient. Inorganic suspended solids were included in addition to organic solids and phytoplankton due to the interest in changes in the light climate or light extinction resulting from the introduction of additional freshwater that can result in elevated suspended solids concentrations.

3 Model Input Data

The WQM requires loadings and boundary conditions, initial conditions, and model parameters including various process rate coefficients. The WQM inputs are described within this chapter.

Loadings and boundary concentrations

Loadings for all sources of nutrients and sediment must be specified for the model. These include loadings from inflowing rivers, atmospheric deposition, and other sources, such as local distributed runoff and major point source (e.g., wastewater discharge) loadings. Of these, the primary inputs are from rivers and the atmosphere, thus, loadings from local distributed runoff and point source wastewater discharges, including storm water drains, were not included in the model for this study. Methods for deriving estimates for riverine and atmospheric loadings are discussed in this section.

Additionally, boundary concentrations must be specified along all boundary flow faces for all water quality state variables, unless a variable is treated as a point source load instead as done for some rivers as explained below. Open water (ocean) boundary concentrations must be specified for all state variables. Boundary concentration data for rivers and ocean are also discussed and presented within this section.

River boundary concentrations and loadings

River loadings can be specified either of two ways in the model. One way is to specify the concentrations at the model boundaries where the river flows enter. This approach requires that the boundary has flows from the hydrodynamic model. The product of flow and concentration is load (mass/time). The other approach is to specify the loading as a point source load (kg/day) for the model cell where the river enters. This approach does not require a flow from the hydrodynamic model at the boundary although flows are provided for all major inflowing rivers in this study. River boundary concentrations should be set to zero for all state variables that are treated as point source loads. Both methods were used for this model study for nutrients. Some water quality variables are best specified as a concentration at the river inflow boundary, such as temperature, dissolved oxygen, and salinity. Salinity was set to 0.0 part per thousand (ppt) for all

river inflow boundaries. Other water quality variables, such as nutrients, are best specified as loads if data are available to determine loads. The boundary conditions for each river inflow are explained below. The rivers included in the model are shown in Figure 3-1.

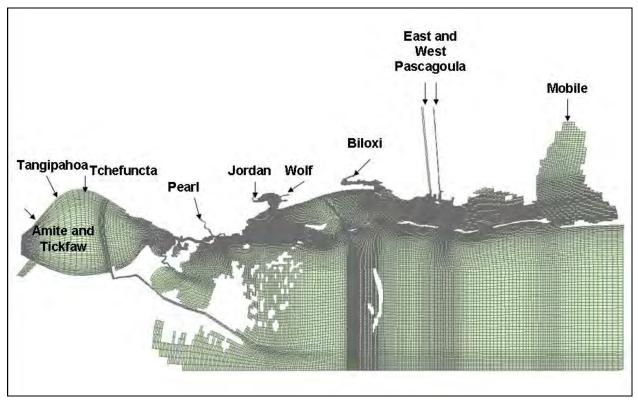


Figure 3-1. Locations of rivers included in the model.

Jordan River

Water quality concentrations for the Jordan River were based on observations and assumptions. Concentrations were applied to the river inflow discharge at the boundary, thus resulting in a loading. Water quality samples in the Jordan River were collected and analyzed monthly and sometimes every two weeks during 1998. These data were obtained from the Mississippi Department of Environmental Quality (MDEQ). Data near Kiln, MS, was used, which is near the mouth but upstream of tidal influence most of the time. Of the WQM state variables, data were available for temperature, dissolved oxygen (DO), ammonium nitrogen (NH₄), and nitrite + nitrate nitrogen (NO₃). Other water quality data that were available and related to model variables included total organic carbon (TOC), total Kjeldahl nitrogen (TKN), total phosphorus (TP), orthophosphate phosphorus (PO₄), and total suspended solids (TSS).

All variables were not available for all sampling dates, thus missing data were interpolated or assumed constant between dates. Assumptions were made to fill data gaps for state variables that were not available. Organic nutrients and organic carbon loads from the river are assumed to be comprised of mostly dissolved forms. Thus, TOC was assumed to be comprised of solely dissolved organic carbon (DOC), and particulate organic carbon (POC) was assumed to be zero. Similarly, total organic nitrogen (TON) was assumed to be dissolved organic nitrogen (DON), and particulate organic nitrogen (PON) was assumed to be zero. TON was computed by subtracting NH₄ from TKN. PO₄ was assumed to represent total inorganic phosphorus (TIP), the model state variable. There were only a few PO₄ values for 1998, but data from 1998 and other years indicated that TP was approximately equal to PO₄, or there was little or no organic phosphorus in the stream. Thus, TIP values were assigned the TP values, and zero concentrations were set for dissolved and particulate organic phosphorus (DOP and POP). Inorganic suspended solids (ISS), a model state variable, was assumed to equal TSS; thus, there was little or no organic suspended solids, which is consistent with the other assumptions for organic carbon and nutrients. The chlorophyll a values were obtained from some spot measurements collected near the river mouth taken by MDEQ. Jordan River concentrations used for the model inflow boundary at that location are shown in Table 3-1.

Table 3-1. Jordan River boundary concentrations, mg/L except where noted.

	Temp	Salinity	Chl a											
Date	deg C	ppt	µg/L	DOC	POC	NH4	ΝОз	DON	PON	TIP	DOP	POP	DO	ISS
4/1/1998	18.7	0.0	7.0	12.0	0.0	0.24	0.90	0.53	0.0	0.005	0.0	0.0	8.3	7.0
4/15/1998	21.4	0.0	7.0	8.0	0.0	0.13	0.06	0.53	0.0	0.085	0.0	0.0	7.2	7.0
4/30/1998	23.6	0.0	7.0	6.0	0.0	0.10	0.06	0.53	0.0	0.060	0.0	0.0	8.1	7.0
5/14/1998	26.2	0.0	7.0	4.0	0.0	0.10	0.02	0.45	0.0	0.020	0.0	0.0	7.9	6.0
5/30/1998	30.5	0.0	7.0	5.0	0.0	0.10	0.02	0.36	0.0	0.025	0.0	0.0	6.3	6.0
6/30/1998	31.5	0.0	6.6	4.0	0.0	0.11	0.02	0.36	0.0	0.075	0.0	0.0	5.8	6.0
7/25/1998	29.8	0.0	7.0	3.0	0.0	0.13	0.02	0.16	0.0	0.025	0.0	0.0	5.0	6.0
8/26/1998	31.1	0.0	7.1	8.0	0.0	0.24	0.03	0.23	0.0	0.015	0.0	0.0	6.2	6.0
9/21/1998	25.6	0.0	7.0	23.0	0.0	0.10	0.08	0.65	0.0	0.095	0.0	0.0	6.2	2.0

Wolf River

Data availability for the Wolf River was similar to that for the Jordan River and data were handled in a similar manner, except that TP values were greater than TIP values, yielding some values for total organic phosphorus (TOP). It was assumed that TOP was DOP, and thus POP values were set

to 0. Wolf River concentrations used for the model inflow boundary at that location are shown in Table 3-2.

	Temp	Salinity	Chl a											
Date	deg C	ppt	µg/L	DOC	POC	NH4	ΝОз	DON	PON	TIP	DOP	POP	DO	ISS
4/1/1998	18.7	0.0	7.0	12.0	0.0	0.24	0.90	0.10	0.0	0.005	0.072	0.0	8.3	6.0
4/15/1998	21.4	0.0	7.0	8.0	0.0	0.27	0.03	0.07	0.0	0.015	0.062	0.0	9.6	6.0
4/30/1998	23.6	0.0	7.0	6.0	0.0	0.10	0.06	0.24	0.0	0.060	0.017	0.0	8.1	6.0
5/14/1998	26.2	0.0	7.0	4.0	0.0	0.10	0.02	0.24	0.0	0.020	0.057	0.0	7.9	6.0
5/30/1998	30.5	0.0	7.0	5.0	0.0	0.10	0.02	0.24	0.0	0.065	0.012	0.0	6.3	6.0
6/30/1998	30.2	0.0	7.0	1.0	0.0	0.11	0.05	0.23	0.0	0.035	0.042	0.0	6.2	6.0
7/25/1998	30.5	0.0	7.0	3.0	0.0	0.13	0.02	0.21	0.0	0.025	0.052	0.0	6.0	6.0
8/26/1998	32.6	0.0	7.0	8.0	0.0	0.21	0.04	0.13	0.0	0.030	0.047	0.0	7.2	6.0
9/21/1998	24.1	0.0	7.0	10.0	0.0	0.10	0.06	0.24	0.0	0.020	0.057	0.0	5.8	6.0

Table 3-2. Wolf River boundary concentrations, mg/L except where noted.

Biloxi River

Data availability for the Biloxi River was not as good as for the Wolf and Jordan Rivers, but the data were handled in a similar manner. Biloxi River concentrations used for the model inflow boundary at that location are shown in Table 3-3.

Date	Temp deg C	Salinity ppt	Chl a µg/L	DOC	POC	NH4	NOз	DON	PON	TIP	DOP	POP	DO	ISS
4/1/1998	20.3	0.0	7.0	6.0	0.0	0.15	0.11	0.41	0.0	0.065	0.008	0.0	8.1	25.0
4/15/1998	21.3	0.0	10.0	8.0	0.0	0.10	0.02	0.46	0.0	0.065	0.008	0.0	7.9	25.0
4/30/1998	23.8	0.0	7.0	5.0	0.0	0.10	0.03	0.46	0.0	0.03	0.043	0.0	7.5	25.0
5/14/1998	26.2	0.0	7.0	4.0	0.0	0.10	0.02	0.46	0.0	0.02	0.053	0.0	7.9	25.0
5/30/1998	31.0	0.0	7.0	5.0	0.0	0.18	0.02	0.38	0.0	0.08	0.0	0.0	7.0	25.0
6/30/1998	28.6	0.0	7.0	5.0	0.0	0.25	0.02	0.31	0.0	0.05	0.023	0.0	6.2	25.0
7/25/1998	33.1	0.0	6.0	5.0	0.0	0.15	0.02	0.41	0.0	0.03	0.043	0.0	7.6	25.0
8/26/1998	32.5	0.0	6.0	8.0	0.0	0.15	0.02	0.41	0.0	0.035	0.038	0.0	7.8	25.0
9/21/1998	26.6	0.0	7.0	15.0	0.0	0.57	0.04	0.00	0.0	0.095	0.0	0.0	5.3	25.0

Table 3-3. Biloxi River boundary concentrations, mg/L except where noted.

Pearl River

No data were found for the Pearl River for 1998, but there were data for latter years. Data for the Pearl River at Highway 90 near Pearlington, MS, were used. Since there were no data for 1998, data from the latter years were averaged to provide constant concentrations for TOC, NH₄, TKN,

NO₃, TP, and ISS. Assumptions similar to those for the Jordan River were used. Values from the Jordan River for temperature, DO, TIP, and chlorophyll a were used. Data for TKN and TP were averaged yielding 0.581 and 0.112 mg/L, respectively, and they were used to compute DON and DOP values. The resulting Pearl River concentrations that could be used for the model inflow boundary at that location are shown in Table 3-4. However, as described below for the Lake Pontchartrain loadings, the Pearl River loads were calculated and used instead of concentrations for N and P components.

Date	Temp deg C	Salinity ppt	Chl a µg/L	DOC	POC	NH4	ΝОз	DON	PON	TIP	DOP	POP	DO	ISS
4/1/1998	18.7	0.0	7.0	4.0	0.0	0.23	0.07	0.35	0.0	0.005	0.107	0.0	8.3	15.5
4/15/1998	21.4	0.0	7.0	4.0	0.0	0.23	0.07	0.35	0.0	0.085	0.027	0.0	7.2	15.5
4/30/1998	23.6	0.0	7.0	4.0	0.0	0.23	0.07	0.35	0.0	0.060	0.052	0.0	8.1	15.5
5/14/1998	26.2	0.0	7.0	4.0	0.0	0.23	0.07	0.35	0.0	0.020	0.092	0.0	7.9	15.5
5/30/1998	30.5	0.0	7.0	4.0	0.0	0.23	0.07	0.35	0.0	0.025	0.087	0.0	6.3	15.5
6/30/1998	31.5	0.0	6.6	4.0	0.0	0.23	0.07	0.35	0.0	0.075	0.037	0.0	5.8	15.5
7/25/1998	29.8	0.0	7.0	4.0	0.0	0.23	0.07	0.35	0.0	0.025	0.087	0.0	5.0	15.5
8/26/1998	31.1	0.0	7.1	4.0	0.0	0.23	0.07	0.35	0.0	0.015	0.097	0.0	6.2	15.5
9/21/1998	25.6	0.0	7.0	4.0	0.0	0.23	0.07	0.35	0.0	0.095	0.017	0.0	6.2	15.5

Table 3-4. Pearl River boundary concentrations, mg/L except where noted.

West and East Pascagoula Rivers

A limited amount of data was found for the Pascagoula River for 1998. Even less data were found for the Escatawpa River, referred to herein as the East Pascagoula River because of the joining of the Escatawpa River with eastern channels of the Pascagoula River. The main channel of the Pascagoula River is referred to as the West Pascagoula River in this study. Water quality data for the Pascagoula River at Highway 90 near Gautier, MS, were used for both the east and west branches. Since data for 1998 were limited, values were averaged to provide constant concentrations for TOC, NH₄, TKN, NO₃, TP, and ISS. Assumptions similar to those for the Jordan River were used. Values from the Biloxi River for temperature, DO, TIP, and chlorophyll a were used. Data for TKN and TP were averaged yielding 0.46 and 0.082 mg/L, respectively, and they were used to compute DON and DOP values. West and East Pascagoula River concentrations used for the model inflow boundary at that location are shown in Table 3-5.

Date	Temp deg C	Salinity ppt	Chl a µg/L	DOC	POC	NH4	NO ₃	DON	PON	TIP	DOP	POP	DO	ISS
4/1/1998	20.3	0.0	7.0	7.0	0.0	0.14	0.09	0.32	0.0	0.065	0.017	0.0	8.1	24.0
4/15/1998	21.3	0.0	10.0	7.0	0.0	0.14	0.09	0.32	0.0	0.065	0.017	0.0	7.9	24.0
4/30/1998	23.8	0.0	7.0	7.0	0.0	0.14	0.09	0.32	0.0	0.03	0.052	0.0	7.5	24.0
5/14/1998	26.2	0.0	7.0	7.0	0.0	0.14	0.09	0.32	0.0	0.02	0.062	0.0	7.9	24.0
5/30/1998	31.0	0.0	7.0	7.0	0.0	0.14	0.09	0.32	0.0	0.08	0.002	0.0	7.0	24.0
6/30/1998	28.6	0.0	7.0	7.0	0.0	0.14	0.09	0.32	0.0	0.05	0.032	0.0	6.2	24.0
7/25/1998	33.1	0.0	6.0	7.0	0.0	0.14	0.09	0.32	0.0	0.03	0.052	0.0	7.6	24.0
8/26/1998	32.5	0.0	6.0	7.0	0.0	0.14	0.09	0.32	0.0	0.035	0.047	0.0	7.8	24.0
9/21/1998	26.6	0.0	7.0	7.0	0.0	0.14	0.09	0.32	0.0	0.095	0.000	0.0	5.3	24.0

Table 3-5. West and East Pascagoula River boundary concentrations, mg/L except where noted.

Loadings for rivers entering Lake Pontchartrain

Four major tributaries that flow into Lake Pontchartrain, the Amite, Tickfaw, Tangipohoa, and Tchefuncta Rivers, are included in the hydrodynamic model as freshwater inflows. Flows from the Amite and Tickfaw Rivers were combined in the hydrodynamic model. The nutrient and sediment loadings from all these rivers were handled as constant point sources in the WQM due to lack of sufficient data to specify concentrations at the inflow boundaries. Loadings from New Orleans along the south shore of the lake were not included in the present model but could be easily added later. The river nutrient and sediment loadings had to be estimated from various literature sources.

Turner et al. (2002) estimated total nitrogen annual loadings (million kg, Mkg, N) into Lake Pontchartrain of 7.8 Mkg from watershed, 1.3 Mkg from atmospheric, 1.0 Mkg from urban New Orleans, and 0.5 to 0.9 Mkg from leakage through the Bonnet Carre' flood control structure. Penland et al. (2002) present a graphical nutrient balance for Lake Pontchartrain that is shown in Figure 3-2. Information from this figure was approximated and is tabulated in Table 3-6.

It is suspected that the values in Table 3-6 do not include loadings from the Pearl River. Loadings for the Pearl River are believed to be included in the watershed loading estimate of Turner et al. (2002), thus possibly explaining the reason the TN loadings are higher than those in Table 3-6.

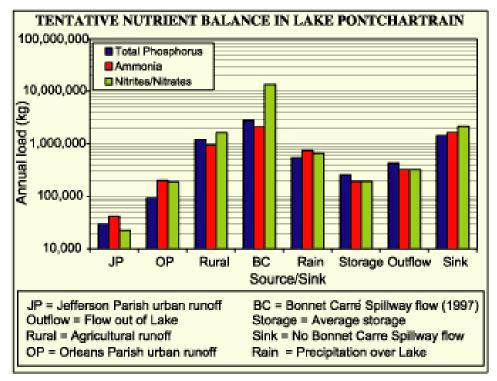


Figure 3-2. Estimated nutrient balance for Lake Pontchartrain (from Penland et al. 2002).

Table 3-6. Nutrient loading estimates for Lake Pontchartrain, Mkg/yr.

Source	NO ₂ +NO ₃	NH ₄	TP
Watershed	1.7	1.0	1.3
New Orleans	0.22	0.24	0.13
Atmospheric (rain)	0.7	0.8	0.58

Table 3-7 shows from two different sources the annual average discharge rate for rivers draining into the Lake Pontchartrain basin. Flow rates from the two information sources are of similar magnitude. It is evident that the Pearl River is a major contributor of flow and material.

Table 3-7. Annual average discharge rates for rivers draining into Lake Pontchartrain basin.

River	Flow rate, cms, Penland et al. (2002)	Flow rate, cms, Dunn (1996)
Tchefuncta	4.7	NA
Tickfaw	9.8	NA
Tangipahoa	33.8	37.9
Amite	60.8	71.6
Pearl	286.5	351.2

NA = not available.

Dunn (1996) provided an analysis of N and P loadings from streams entering the Gulf of Mexico. Annual average loadings relevant to this present study from this report are shown in Table 3-8. USACE (1990) compiled historical water quality data for the Mississippi River, Lake Pontchartrain, and rivers entering the Lake Pontchartrain basin and computed mean concentrations for various water quality constituents. Mean concentrations for constituents at locations of interest in this present study are shown in Table 3-9.

Table 3-8. Annual average N and P loadings for rivers draining into the Gulf of Mexico, from Dunn (1996).

River	TN loading, Mkg/yr	TP loading, Mkg/yr
Mississippi	1,280.00	110.00
Tangipahoa	1.67	0.18
Amite	4.35	0.40
Pearl	12.80	1.52
Pascagoula	9.38	0.80
Tombigbee	27.50	3.54
Alabama	22.20	2.57

Table 3-9. Historical average concentrations (mg/L) of NO₃, TP, and TOC for rivers draining into the Lake Pontchartrain basin, from USACE (1990).

River	NO ₃ ¹	TP	TOC
Tchefuncta	0.25	0.10	4.4
Tangipahoa	0.28	0.12	3.1
Pearl	0.16	0.11	4.0

¹ Nitrate + nitrite.

The above information and various assumptions similar to those already stated were used to estimate the loadings into Lake Pontchartrain and to adjust the loadings for the Pearl River as described below.

Annual average nitrogen loadings for the Amite River were estimated as follows. The watershed loading values in Table 3-6 for NH $_4$ and NO $_3$ were subtracted from the Turner et al. (2002) estimate of annual TN watershed load into Lake Pontchartrain of 7.8 Mkg to compute the TON annual load of 5.1 Mkg. It was assumed that TON was comprised totally of DON as with the tributaries in Mississippi. The ratios NO $_3$ /TN, NH $_4$ /TN, and DON/TN were computed using the values in Table 3-6 for NH $_4$ and NO $_3$, the computed TON value, and the 7.8-Mkg value for TN. Each ratio was multiplied by the TN loading of 4.35 Mkg from Table 3-8 to obtain Amite

River loadings for NH₄, NO₃, and DON. These values were converted to kg/day.

The annual average phosphorus loadings for the Amite River were estimated in a manner similar to that for nitrogen. No information was found for annual loadings of TIP and TOP for the Lake Pontchartrain basin. Thus, it was assumed that the Pearl River ratio of TIP/TP concentrations (0.40) is applicable to the other rivers entering the basin. Using this ratio and the TP annual loading of 0.40 Mkg from Table 3-8, the annual Amite River loading for TIP was computed. TOP loading was taken as the difference in TP and TIP loadings, and DOP was assumed to comprise all of TOP. The values were converted to kg/day.

The annual average nutrient loadings for the Tangipahoa River were estimated in the same manner as those for the Amite River. The only difference was that the TN and TP loadings in Table 3-8 for the Tangipahoa River were used for the last step. The computed N and P loading values are shown in Table 3-10.

Table 3-10. Nutrient loadings for rivers flowing into Lake Pontchartrain and the Pearl River,
kg/day.

River	NO ₃ ¹	NH ₄	DON	TIP	DOP
Amite/Tickfaw	3,021	1,777	9,063	512	762
Tangipahoa	997	587	2992	198	295
Tchefuncta	99	58	296	17	25
Pearl	7,470	10,944	16,654	1,673	2,491

¹ Nitrate + nitrite.

The annual average nutrient loadings for the Tickfaw River were estimated in nearly the same manner as those for the Amite and Tangipahoa Rivers except that the TN and TP loadings had to be estimated. The average NO_3 concentrations for the Tchefuncta and Tangipahoa Rivers in Table 3-9 were averaged to estimate an average NO_3 concentration of 0.27 mg/L for the Tickfaw River. This value was used to estimate the NO_3 load by multiplying the concentration by the flow from Table 3-7. The NO_3 load was multiplied by the TN/NO_3 ratio derived in the same manner as explained previously for the Amite River to obtain an estimated TN annual load of 0.38 Mkg for the Tickfaw River. Similarly, the Amite River TN annual load of 2.33 Mkg was computed using the same approach (i.e., concentration of 0.27 mg/L and average annual flow). The ratio of 0.38/2.33 was used to convert the previously computed Amite River loads to Tickfaw River loads.

The annual average TP loading for the Tickfaw River was estimated in the same manner as NO_3 . The average TP concentrations in Table 3-9 for the Tchefuncta and Tangipahoa Rivers were averaged to estimate an average TP concentration of 0.11 mg/L for the Tickfaw River. This value was multiplied by the flow from Table 3-7, yielding an annual TP load of 0.034 Mkg. This value was multiplied by the TIP/TP (0.40) ratio discussed earlier to get the TIP load. TOP loading was taken as the difference in TP and TIP loadings, and DOP was assumed to comprise all of TOP. The values were converted to kg/day. The Tickfaw and Amite River loadings were combined in the model, with the combined loads shown in Table 3-10, and applied at the location of the Tickfaw River.

The annual average nutrient loadings for the Tchefuncta River were estimated in a manner similar to that of the Tickfaw River. The average concentrations in Table 3-9 were multiplied by the annual average flow to yield annual average loadings for NO₃ and TP of 0.036 and 0.015 Mkg, respectively. These values were multiplied by the previously established ratios (for N, based on values in Table 3-6 and TN watershed loading for Lake Pontchartrain of 7.8 Mkg; for P, based on Pearl River TIP/TP concentration ratio) to obtain the loadings shown in Table 3-10.

Constituent concentrations for non-loaded variables had to be set for the rivers flowing into Lake Pontchartrain. Due to lack of data, temperature, DO, and chlorophyll a were set equal to those values used for the Pearl River. TOC concentrations of Table 3-9 are about the same as the Pearl River DOC concentration in Table 3-4, thus, DOC was set to 4.0 mg/L, same as for the Pearl River. Using a value of 4 mg/L for DOC and the flows in Table 3-7 results in a DOC loading that is approximately half the value reported by Argyrou et al. (1997) for DOC river loadings into Lake Pontchartrain (if Pearl River is excluded), and about twice their value if the Pearl River is included in the Dortch et al. (this report) calculation. It is not known whether Argyrou et al. (1997) included the Pearl River in their estimates, but it is suspected that they did not. No information was found on suspended sediment loadings into Lake Pontchartrain. Battelle (2005) did report an average measured suspended sediment concentration of about 15 mg/L for Lake Maurepas, which included measurements in the Amite River and other tributaries emptying into the lake. Since this value was close to the value for the Pearl River (Table 3-4), the Pearl River value of 15.5 mg/L was used for all rivers flowing into Lake Pontchartrain. The concentrations of non-loaded water quality variables for rivers flowing into Lake Pontchartrain are shown in Table 3-11.

Chl a Temp Salinity Date deg C ppt µg/L DOC POC NH₄ ΝОз DON PON TIP DOP POP DO ISS 4/1/1998 18.7 0.0 4.0 0.0 0.00 0.00 0.00 0.0 0.000 0.000 0.0 8.3 15.5 7.0 0.0 7.0 4.0 0.0 0.000 7.2 4/15/1998 21.4 0.00 0.00 0.00 0.0 0.000 0.0 15.5 0.0 4.0 4/30/1998 23.6 7.0 0.0 0.00 0.00 0.00 0.0 0.000 0.000 0.0 8.1 15.5 5/14/1998 26.2 0.0 7.0 4.0 0.0 0.00 0.00 0.00 0.0 0.000 0.000 0.0 7.9 15.5 30.5 0.0 7.0 4.0 0.0 0.00 0.00 0.0 0.000 6.3 5/30/1998 0.00 0.000 0.0 15.5 6/30/1998 6.6 4.0 0.000 5.8 31.5 0.0 0.0 0.00 0.00 0.00 0.0 0.000 0.0 15.5 0.0 7.0 4.0 0.0 5.0 7/25/1998 29.8 0.0 0.00 0.00 0.00 0.000 0.000 0.0 15.5 7.1 8/26/1998 31.1 0.0 4.0 0.0 0.00 0.00 0.00 0.0 0.000 0.000 0.0 6.2 15.5 7.0 6.2 0.0 4.0 0.0 0.00 0.0 0.000 9/21/1998 25.6 0.00 0.00 0.000 0.0 15.5

Table 3-11. Water quality boundary concentrations for rivers flowing into Lake Pontchartrain and Pearl River, mg/L except where noted.

Revised Pearl River loadings

When the nitrogen component concentrations for the Pearl River (Table 3-4) were multiplied by the average flow for the Pearl River (Table 3-7), the TN loadings were about half those in Table 3-8, even after adjusting the NO₃ concentration to the value shown in Table 3-9. Similarly, the computed TP loadings were about 2/3 those shown in Table 3-8 when using the concentrations of Table 3-4 and flow of Table 3-7. Thus, it was decided to use loads based upon TN and TP loads in Table 3-8 (12.8) and 1.52 Mkg/yr, respectively) rather than actual flows input to CH3D and concentrations of Table 3-4. The annual average nutrient loadings for the Pearl River were estimated in the same manner as those for the Amite River except for using the nitrogen ratio calculated with data from the Pearl River. The nitrogen component ratios were based upon the Pearl River NH₄ and DON values shown in Table 3-4 and the NO₃ value shown in Table 3-9. The computed N and P loadings for the Pearl River are shown in Table 3-10. The revised water quality boundary concentrations for Pearl River flows are shown in Table 3-11.

Mobile River

The Alabama and Tombigbee Rivers join about 45 miles upstream of Mobile to form the Mobile River. The combined annual average discharge of the Mobile River is approximately three times that of either the Pascagoula or Pearl Rivers (Dunn 1996). Considerable summary information was found in the literature regarding Alabama and Tombigbee River nutrient concentrations and loads. Since 1998 was a relatively dry year, mean concentrations (as described below) were used with model inflow

rates to produce loadings rather than using annual average loadings that were reported in the literature.

Although data were not found for 1998, McPherson et al. (2003) reported summary statistics for concentrations of nutrients and suspended sediment for January 1999 through December 2001 for the Alabama River at Claiborne, AL, and for the Tombigbee River below Coffeeville Lock and Dam. Both of these stations are the lowermost of all observation stations on these two rivers. Subsets of the summary statistics for these two stations are shown in Tables 3-12 and 3-13 where minimum (Min), maximum (Max), mean, 95% upper confidence limit (95%), and 5% lower confidence limit (5%) are provided for each constituent. The spread in the data is relatively small except for TSS, which is highly correlated with river flow rate, thus, the mean concentrations were used.

Mean N concentrations in Tables 3-12 and 3-13 were used to compute concentrations for the missing N components. Dissolved ammonia was assumed to be equal to total NH₄, and dissolved ammonia plus organic nitrogen less dissolved ammonia equals DON. Thus, the difference in total ammonia plus organic N and dissolved ammonia plus organic N equals PON. When summing NH₄, NO₂+NO₃, DON, and PON, to get TN, the summed values were slightly less than the TN values reported in Tables 3-12 and 3-13. The NO₂+NO₃ concentrations were increased a little to yield summed TN values that were equal to the reported TN values.

Likewise, the mean P concentrations in Tables 3-12 and 3-13 were used to compute concentrations for the missing P components. The dissolved P less the dissolved phosphate equals DOP. TP less the dissolved P equals total particulate phosphorus (TPP). There were no data to estimate the fractions of particulate inorganic and organic P comprising TPP. A common assumption used in applying ICM is that the ratio of POP to POC concentrations is about 0.025¹. The POC concentration was multiplied by 0.025 to estimate POP. The estimated POP concentration was subtracted from TPP to obtain an estimated particulate inorganic phosphorus (PIP) concentration. The dissolved phosphate concentration was assumed to equal the dissolved inorganic phosphorus (DIP) concentration. DIP and PIP were summed to yield TIP.

¹ Cerco, C. F. 2007. Personal communication, U.S. Army Engineer Research and Development Center, Vicksburg, MS.

Table 3-12. Summary statistics data of nutrient and sediment concentrations for the Alabama River, mg/L.

Constituent	N¹	Max	Min	Mean	95%	5%
Dissolved ammonia as N	34	0.09	<.02	0.026	0.08	<.02
Dissolved ammonia plus organic N as N	31	0.34	0.16	0.225	0.334	0.166
Total ammonia plus organic N as N	34	0.62	0.28	0.432	0.605	0.28
NO ₂ +NO ₃ as N	34	0.29	<.05	0.13	0.25	<.05
TN	29	0.87	0.4	0.574	0.825	0.435
Dissolved P as P	31	0.05	0.01	0.027	0.049	0.012
Dissolved phosphate as P	34	0.04	<.01	0.018	0.04	<.01
TP	34	0.173	0.051	0.077	0.146	0.052
DOC	31	7	3.3	4.41	6.16	3.3
POC	24	1.3	0.4	0.675	1.3	0.4
TSS	34	167	2	28.9	132	5

¹ Number of samples.

Table 3-13. Summary statistics data of nutrient and sediment concentrations for the Tombigbee River, mg/L.

Constituent	N	Max	Min	Mean	95%	5%
Dissolved ammonia as N	34	0.11	<.02	0.032	0.06	<.02
Dissolved ammonia plus organic N as N	31	0.66	0.16	0.256	0.468	0.166
Total ammonia plus organic N as N	34	3.6	0.25	0.548	1.725	0.303
NO ₂ +NO ₃ as N	34	0.83	<.05	0.259	0.56	<.05
TN	31	3.8	0.4	0.838	2.48	0.424
Dissolved P as P	31	0.042	0.008	0.023	0.041	0.008
Dissolved phosphate as P	34	0.03	<.01	0.017	0.03	<.01
TP	34	0.38	0.04	0.1	0.282	0.051
DOC	31	6.7	2.7	4.7	6.46	3
POC	24	3.3	0.2	0.796	2.85	0.225
TSS	33	507	5	69.9	368	8.5

ISS was assumed to be equivalent to TSS, which appears to be reasonable given the small concentrations of particulate organic constituents. The various mean estimated nutrient and sediment concentrations (as

intended for model use) for the Alabama and Tombigbee Rivers are provided in Table 3-14.

Table 3-14. Mean estimated nutrient and sediment concentrations used by the model, mg/L.

Constituent	Alabama River	Tombigbee River	Mobile River ¹
NH ₄	0.026	0.032	0.029
DON	0.199	0.224	0.212
PON	0.207	0.29	0.249
NO ₂ +NO ₃	0.14	0.29	0.215
TIP	0.051	0.074	0.063
DOP	0.009	0.006	0.008
POP	0.017	0.02	0.019
DOC	4.41	4.7	4.56
POC	0.675	0.796	0.736
ISS	28.9	69.9	49.4

¹ Average of Alabama and Tombigbee River values.

Nutrient and sediment concentrations for the Alabama and Tombigbee Rivers were averaged to yield the Mobile River concentrations (Table 3-14) since the mean annual flows of the two rivers are about the same (25 versus 23 million acre-ft/yr, Dunn 1996). The values in Table 3-3 for the Biloxi River for temperature, DO, and chlorophyll a were used for the Mobile River, and nutrient and sediment concentrations for the Mobile River (Table 3-14) were held constant resulting in the Mobile River boundary concentration data of Table 3-15.

Table 3-15. Mobile River boundary concentrations, mg/L except where noted.

	Temp	Salinity	Chl a											
Date	deg C	ppt	µg/L	DOC	POC	NH4	NO₃	DON	PON	TIP	DOP	POP	DO	ISS
4/1/1998	20.3	0.0	7.0	4.56	0.74	0.03	0.22	0.21	0.25	0.063	0.008	0.019	8.1	49.4
4/15/1998	21.3	0.0	10.0	4.56	0.74	0.03	0.22	0.21	0.25	0.063	0.008	0.019	7.9	49.4
4/30/1998	23.8	0.0	7.0	4.56	0.74	0.03	0.22	0.21	0.25	0.063	0.008	0.019	7.5	49.4
5/14/1998	26.2	0.0	7.0	4.56	0.74	0.03	0.22	0.21	0.25	0.063	0.008	0.019	7.9	49.4
5/30/1998	31.0	0.0	7.0	4.56	0.74	0.03	0.22	0.21	0.25	0.063	0.008	0.019	7.0	49.4
6/30/1998	28.6	0.0	7.0	4.56	0.74	0.03	0.22	0.21	0.25	0.063	0.008	0.019	6.2	49.4
7/25/1998	33.1	0.0	6.0	4.56	0.74	0.03	0.22	0.21	0.25	0.063	0.008	0.019	7.6	49.4
8/26/1998	32.5	0.0	6.0	4.56	0.74	0.03	0.22	0.21	0.25	0.063	0.008	0.019	7.8	49.4
9/21/1998	26.6	0.0	7.0	4.56	0.74	0.03	0.22	0.21	0.25	0.063	0.008	0.019	5.3	49.4

Mississippi River

Water quality concentrations were required for the Mississippi River since it is being considered as a source for freshwater diversions. Given that the diversion flows are specified and are much smaller than the flow in the river, the use of concentrations was appropriate rather than loading information for the entire river.

Water quality summary data from the U.S. Geological Survey (USGS) National Stream Quality Accounting Network (NASQAN) for 1996–2005 at the St. Francisville, LA, station were retrieved from the Internet (http://water.usgs.gov/nasqan/data/statsum/st.francis.html). These data included summary statistics for many water quality variables, including total suspended sediment, various forms of nitrogen and phosphorus, DOC, POC, and chlorophyll a. However, the St. Francisville station is upstream of where the river is most likely to be diverted, and concentrations for nutrients and TSS are lower than values reported downstream. Water quality data from the lower Mississippi River for the 1970s and 1980s are summarized by USACE (1990). These data include locations near New Orleans, which is in the vicinity of the river where diversions might occur. Additionally, water quality concentrations taken between 1988 and 1994 from the Mississippi River near the Caernarvon diversion, which is near New Orleans, are reported by Lane et al. (1999). The data reported by Lane et al. (1999) were used to establish concentrations for NO₂+NO₃-N, NH₄-N, TKN, TN, TP, and TSS of 1.4, 0.1, 1.0, 2.5, 0.26, and 200 mg/L, respectively. TON concentration of 0.9 mg/L was computed from the difference in TKN and NH₄-N. Mean concentrations for NO₂+NO₃-N and TP reported by USACE (1990) agree with the values from Lane et al. (1999). Mean concentration for TOC of 6.9 mg/L and mean percent DO saturation of 85% (USACE 1990) were used. Monthly average temperatures of the Mississippi River for the period March 2004 through February 2005 (Battelle 2005) were used to establish water temperatures. Water temperatures were used to compute saturated DO concentrations.

The data from the NASQAN station are more comprehensive, thus, these data were used to fill data gaps. The NASQAN data were used to compute ratios for DON/TON, TIP/TP, DOP/TP, and DOC/TOC, which were 0.41, 0.29, 0.32, and 0.58, respectively. The appropriate ratios were multiplied by the TON and TP concentrations from Lane et al. (1999) and the TOC concentration of 6.9 mg/L to obtain concentrations for DON, TIP, DOP, and DOC. Other required nutrient forms were obtained by differences, e.g., PON = TON - DON. The results of these computations are available

in Table 3-16. The mean chlorophyll a value of 3.5 mg/L reported at the NASQAN site was used. Salinity was set to 0.0, same as for the other rivers. The Mississippi River water quality concentrations used in the model for diverted Mississippi River water are shown in Table 3-16.

	Temp	Salinity	Chl a											
Date	deg C	_	µg/L	DOC	POC	NH4	NOз	DON	PON	TIP	DOP	POP	DO	ISS
4/1/1998	14.5	0.0	3.5	4.00	2.90	0.10	1.40	0.41	0.59	0.075	0.083	0.102	8.6	200.0
4/15/1998	16.8	0.0	3.5	4.00	2.90	0.10	1.40	0.41	0.59	0.075	0.083	0.102	8.2	200.0
4/30/1998	20.1	0.0	3.5	4.00	2.90	0.10	1.40	0.41	0.59	0.075	0.083	0.102	7.7	200.0
5/14/1998	23.4	0.0	3.5	4.00	2.90	0.10	1.40	0.41	0.59	0.075	0.083	0.102	7.2	200.0
5/30/1998	25.0	0.0	3.5	4.00	2.90	0.10	1.40	0.41	0.59	0.075	0.083	0.102	7.0	200.0
6/30/1998	28.0	0.0	3.5	4.00	2.90	0.10	1.40	0.41	0.59	0.075	0.083	0.102	6.6	200.0
7/25/1998	29.5	0.0	3.5	4.00	2.90	0.10	1.40	0.41	0.59	0.075	0.083	0.102	6.4	200.0
8/26/1998	28.0	0.0	3.5	4.00	2.90	0.10	1.40	0.41	0.59	0.075	0.083	0.102	6.6	200.0
9/21/1998	25.0	0.0	3.5	4.00	2.90	0.10	1.40	0.41	0.59	0.075	0.083	0.102	7.0	200.0

Table 3-16. Mississippi River concentrations used for diversions, mg/L except where noted.

Atmospheric loadings

Goolsby et al. (1999) summarizes a literature review of atmospheric loading of nitrogen to the Gulf of Mexico. They report that wet deposition of N is an order of magnitude or higher than dry deposition, thus, dry deposition can be ignored. Goolsby et al. (1999) report that the average wet deposition of inorganic N along the U.S. gulf coast is typically on the order of 3 to 4 kg/ha/yr with NO₃ accounting for about 60% of total N deposited. Using a value of 3.5 kg/ha/yr results in 2.1 and 1.4 kg/ha/yr for NO₃ and NH₄, respectively. If these loadings are applied to the surface of Lake Pontchartrain with an area of 1,632 km² (Penland et al. 2002), a loading of 0.34 and 0.23 Mkg/yr for NO₃ and NH₄, respectively, are calculated, and these values are less than half the values reported in Table 3-6. Goolsby et al. (1999) report that N loadings approach 7 kg/ha/yr for Gulf of Mexico waters near southern Louisiana. They also state that values are higher closer to shore than out in the gulf, with values of about 5.5 kg/ha/yr. Thus, a value of 5.5 kg/ha/yr was used for atmospheric TN loading in the model, which was entered as 3.3 and 2.2 kg/ha/yr NO₃ and NH₄, respectively. This TN loading from the atmosphere distributed over the model domain of 17,280 km² is about one-fourth as large as the TN loadings from the tributaries entering the model domain.

To estimate the appropriate TP atmospheric loading, the value 0.58 Mkg/yr for TP in Table 3-6 was scaled by 0.6, which is the ratio of

the 5.5 kg/ha/yr TN loading and the Table 3-6 TN loading (after converting units). This resulted in a TIP atmospheric loading of 2.1 kg/ha/yr.

Ocean boundary concentrations

The U.S. Environmental Protection Agency (USEPA) collected water quality data offshore in Mississippi Sound during July 2002 at four stations (MS1, MS2, MS3, and MS4), which are shown in Figure 3-3. These data along with various assumptions were used to estimate the outer gulf boundary conditions for water quality.

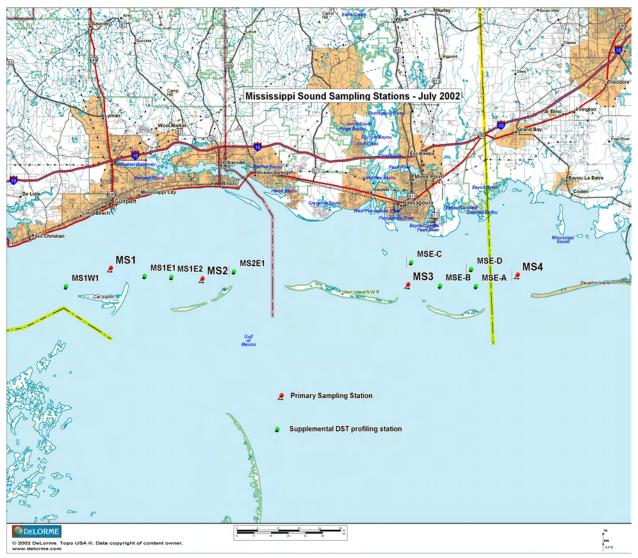


Figure 3-3. USEPA sampling stations for July 2002.

Variations in water quality variables over the depth were assumed to be small along the outer gulf boundary, thus, a constant value for each variable was assigned for all vertical layers of the model along the boundary.

Temperature and DO were varied over time along the outer gulf boundaries, but other constituents were held constant over time. There are two outer gulf boundaries, one along the south boundary running east—west outside the barrier islands, and one running north—south from the southern ocean boundary to the shore east of Mobile Bay. The same water quality boundary concentrations were used for both boundaries and are shown in Table 3-17.

	Temp	Salinity	Chl a											
Date	deg C	ppt	µg/L	DOC	POC	NH4	NO ₃	DON	PON	TIP	DOP	POP	DO	ISS
4/1/1998	18.7	*	1.0	1.0	0.0	0.05	0.05	0.09	0.0	0.025	0.0	0.0	9.3	2.0
4/15/1998	20.1	*	1.0	1.0	0.0	0.05	0.05	0.09	0.0	0.025	0.0	0.0	9.1	2.0
4/30/1998	23.6	*	1.0	1.0	0.0	0.05	0.05	0.09	0.0	0.025	0.0	0.0	8.5	2.0
5/14/1998	28.9	*	1.0	1.0	0.0	0.05	0.05	0.09	0.0	0.025	0.0	0.0	7.7	2.0
5/30/1998	26.2	*	1.0	1.0	0.0	0.05	0.05	0.09	0.0	0.025	0.0	0.0	8.1	2.0
6/30/1998	30.4	*	1.0	1.0	0.0	0.05	0.05	0.09	0.0	0.025	0.0	0.0	7.5	2.0
7/25/1998	33.6	*	1.0	1.0	0.0	0.05	0.05	0.09	0.0	0.025	0.0	0.0	7.1	2.0
8/26/1998	31.0	*	1.0	1.0	0.0	0.05	0.05	0.09	0.0	0.025	0.0	0.0	7.4	2.0
9/21/1998	29.2	*	1.0	1.0	0.0	0.05	0.05	0.09	0.0	0.025	0.0	0.0	7.7	2.0

Table 3-17. Outer gulf boundary concentrations, mg/L except where noted.

The same temperatures as used for the outer gulf boundaries in the Gulfport Harbor model study (Bunch et al. 2005) were used in this study for those boundaries. Salinity values along the two boundaries were the same as those used for the hydrodynamic model, which were varied spatially with salinity increasing in towards the southeast corner of the grid. Salinity along the ocean boundaries was held constant over time. Algal chlorophyll a concentrations are typically on the order of 1.0 μg/L in the open ocean or sea away from the shore, so this value was assumed for the outer gulf boundaries. The TOC measured at the USEPA stations was 1.0 mg/L, so this was the value assumed for DOC with POC set to zero along the boundaries. No values were measured at the USEPA stations above the detection limit of 0.05 mg/L N for NO₂+NO₃-N and NH₄-N. Therefore, the open gulf boundary values for these two water quality variables were set to 0.05 mg/L. Measured values for TKN at the USEPA stations averaged 0.62 and 0.14 mg/L N for the western (MS1 and MS2) and eastern (MS3 and MS4) stations, respectively. The western stations are more heavily influenced by terrestrial loadings from tributaries entering in that region, whereas the eastern stations are more representative of conditions in the open gulf. Thus, a value of 0.14 mg/L for TKN was used

^{*} Set to the values used for the outer boundaries in the hydrodynamic model.

to estimate TON, or DON since PON was assumed to be zero along the outer gulf boundaries. With an NH₄-N concentration of 0.05 mg/L and TKN concentration of 0.14 mg/L, the calculated DON was 0.09 mg/L, which was applied along the outer gulf boundaries. TP and total dissolved phosphorus (TDP) were measured at the USEPA stations, but all values were below the detection limit of 0.025 mg/L P except for TP values that averaged 0.033 mg/L at the western stations. Thus, it was assumed that the phosphorus along the outer gulf boundaries was TIP with a value of 0.025 mg/L. The DO concentrations for the outer gulf boundaries were set equal to the computed DO saturation based on water temperature. TSS was measured at the USEPA stations, but with the exception of one value of 8.0 mg/L at MS1, the other values were close to or below the detection limit of 4.0 mg/L. Thus, the outer gulf boundary concentration for ISS was set to half the USEPA station values, or 2.0 mg/L, since very little TSS would be expected this far out.

Initial conditions

Initial conditions for water quality constituents were first set equal to those used for the Gulfport Harbor model study (Bunch et al. 2005), which were based upon observed data from MDEQ. These initial concentrations for the water column, which are shown in Table 3-18, were specified as uniform throughout the grid, i.e., same values for all cells in all layers. To provide more realistic, spatially varied, initial conditions, ICM was run for one month using the uniform initial conditions discussed above. Hydrodynamics and water quality boundary conditions for April 1998 were used for this run. Water quality concentrations at the end of the month for all computational cells were saved to a file and were used as the initial conditions for a second month-long run, again using April 1998 hydrodynamics and water quality boundary conditions. The-end-of-month concentrations were again saved for all cells and used as initial conditions for a third month-long run with the same hydrodynamics and water quality boundary conditions. Thus, three one-month-long runs were used to spin-up the initial conditions for water quality that were used for all subsequent model calibration and scenario runs.

Constituent	Calibration Value
Temperature (°C)	22.2
Salinity (ppt)	23.0
Dissolved Organic Carbon (g/m³)	8.0
Dissolved Oxygen (g/m³)	8.1

0.12

0.02

0.16

Table 3-18. Uniform initial conditions.

Other inputs

Ammonium (g/m3)

TIP (g/m^3)

Nitrate-Nitrite (g/m³)

The ICM model requires various kinetic rate coefficients and other parameters to simulate water quality processes. All model parameters are described by Cerco et al. (2004) or in the draft user manual that was developed as a part of that study. Model parameters that were used for the final model calibration in the present study are presented in the Chapter 4 on Model Calibration.

Additionally, meteorological data are required for predicting temperature and photosynthetically active radiation (PAR), which affects plant growth. The model uses daily solar radiation incident on the water surface, equilibrium temperatures, and heat exchange coefficients (Edinger et al. 1974) to predict water temperature. These three variables are computed from a pre-processor program using meteorological data consisting of air temperature, dew point temperature, wind speed, and percentage cloud cover. If measured solar radiation is available, then measured values can be used rather than computed values. Daily solar radiation for temperature predictions is converted in the model to PAR for use in plant growth routines. Solar radiation and PAR are attenuated over the water depth as affected by water quality properties (i.e., suspended solids and algal concentrations). Meteorological data for 1998 from the airport in Mobile, AL, were used in this study. These meteorological data were obtained from the Air Force Combat Climatologic Center (https://notus2.afccc.af.mil/SCISPublic/).

4 Model Calibration

Background

A partial model calibration was performed due to the limited scope of this study. Hydrodynamics from CH3D were supplied to the WQM for March through September 1998 conditions. The WQM was executed for the period April through September 1998 for calibration.

Model calibration proceeded by making a limited number of runs with various adjustments to model kinetic coefficients and parameters. The primary parameters that were varied in the calibration simulations were particulate organic nitrogen and phosphorus hydrolysis rates, the dissolved organic nitrogen and phosphorus mineralization rates, the maximum nitrification rate, the suspended solids and algal settling rates, fractions of algal recycling and proportioning to various organic pools, carbon to chlorophyll a ratio, algal half-saturation constants for nutrient uptake, maximum photosynthesis rate for algal growth, and first-order algal predation rate. The calibration was particularly sensitive to the mineralization and nitrification rates.

Only algal group 3 was activated, and mortality as related to salinity was turned off. The intent was to specify SOD flux of 1.0 g/m²/day (at 20 °C), but this was not possible under the limited study scope and time constraints since model coding modifications were required. This SOD value would have been consistent with measurements taken by USEPA at MS1 to MS4 (see Figure 3-3) in August 2002 and corrected for temperature. Nitrate losses due to sediment denitrification were not specified as a benthic flux as intended, but they should also be considered in any future modeling of this site. All of the other parameter values that were used in the final model calibration are listed in Table 4-1.

Model results were compared to observed data obtained from MDEQ for various stations throughout the Mississippi Sound. The locations of the observation stations are shown in Figure 4-1. Observations were not available for all variables at all stations. Additionally, stations were added that did not have observational data to provide information on model output. Observational data from 1998 were supplemented with data from 2002 at stations MS1 to MS4 (Figure 3-3) collected by USEPA.

Table 4-1. Parameters in kinetics equations for Mississippi Sound.

Symbol	Definition	Value	Units
AANOX	Ratio of anoxic to oxic respiration	0.5	0 ≤ AANOX ≤ 1
ANC	Nitrogen-to-carbon ratio of algae	0.175	g N g-1 C
AOCR	Dissolved oxygen-to-carbon ratio in respiration	2.67	g O ₂ g ⁻¹ C
AONT	Mass dissolved oxygen consumed per mass ammonium nitrified	4.33	g O ₂ g ⁻¹ N
APC	Algal phosphorus-to-carbon ratio	0.0175	g P g-1 C
Areaer	Empirical constant in reaeration equation	0.1	
BM	Basal metabolic rate of algae at reference temperature Tr	0.03	d ⁻¹
BPR	Base predation rate	0.22	d ⁻¹
CChI	Carbon-to-chlorophyll a ratio	100	g C g ⁻¹ chl
FCD	Fraction of dissolved organic carbon produced by algal metabolism	0.0	0 ≤ FCD ≤ 1
FCDP	Fraction of dissolved organic carbon produced by predation	0.25	0 ≤ FCDP ≤ 1
FCL	Fraction of labile particulate carbon produced by algal metabolism	0.0	0 ≤ FCL ≤ 1
FCLP	Fraction of labile particulate carbon produced by predation	0.75	0 ≤ FCLP ≤ 1
FCR	Fraction of refractory particulate carbon produced by algal metabolism	Not used	0 ≤ FCR ≤ 1
FCRP	Fraction of refractory particulate carbon produced by predation	Not used	0 ≤ FCRP ≤ 1
FNI	Fraction of inorganic nitrogen produced by algal metabolism	0.6	0 ≤ FNI <u>≤</u> 1
FNIP	Fraction of inorganic nitrogen produced by predation	0.4	0 ≤ FNIP ≤ 1
FND	Fraction of dissolved organic nitrogen produced by algal metabolism	0.3	0 ≤ FND ≤ 1
FNDP	Fraction of dissolved organic nitrogen produced by predation	0.3	0 ≤ FNDP ≤ 1
FNL	Fraction of labile particulate nitrogen produced by algal metabolism	0.1	0 ≤ FNL ≤ 1
FNLP	Fraction of labile particulate nitrogen produced by predation	0.3	0 ≤ FNLP ≤ 1
FNR	Fraction of refractory particulate nitrogen produced by algal metabolism	Not used	0 ≤ FNR ≤ 1
FNRP	Fraction of refractory particulate nitrogen produced by predation	Not used	0 ≤ FNRP ≤ 1

Symbol	Definition	Value	Units
FPD	Fraction of dissolved organic phosphorus produced by algal metabolism	0.3	0 ≤ FPD ≤ 1
FPDP	Fraction of dissolved organic phosphorus produced by predation	0.3	0 ≤ FPDP ≤ 1
FPI	Fraction of dissolved inorganic phosphorus produced by algal metabolism	0.6	0 ≤ FPI ≤ 1
FPIP	Fraction of dissolved inorganic phosphorus produced by predation	0.5	0 <u><</u> FPIP <u><</u> 1
FPL	Fraction of labile particulate phosphorus produced by algal metabolism	0.1	0 <u><</u> FPL <u><</u> 1
FPLP	Fraction of labile particulate phosphorus produced by predation	0.2	0 ≤ FPLP ≤ 1
FPR	Fraction of refractory particulate phosphorus produced by algal metabolism	Not used	0 ≤ FPR ≤ 1
FPRP	Fraction of refractory particulate phosphorus produced by predation	Not used	0 ≤ FPRP ≤ 1
Kcod	Oxidation rate of chemical oxygen demand	20.0	d-1
Khso	DO concentration at which SOD is halved	2.0	g m ⁻³
KDC	Dissolved organic carbon respiration rate	0.01	d-1
KND	Dissolved organic nitrogen mineralization rate	0.012	d-1
KDP	Dissolved organic phosphorus mineralization rate	0.12	d ⁻¹
Kdpalg	Constant that relates mineralization rate to algal biomass	0.4	m ³ g ⁻¹ C d ⁻¹
KHn	Half-saturation concentration for nitrogen uptake by algae	0.05	g N m ⁻³
KHndn	Half-saturation concentration of nitrate required for denitrification	0.1	g N m ⁻³
KHnnt	Half-saturation concentration of NH ₄ required for nitrification	1.0	g N m ⁻³
KHocod	Half-saturation concentration of dissolved oxygen required for exertion of COD	0.5	g O ₂ m ⁻³
KHodoc	Half-saturation concentration of dissolved oxygen required for oxic respiration	0.5	g O ₂ m ⁻³
KHont	Half-saturation concentration of dissolved oxygen required for nitrification	3.0	g O ₂ m ⁻³
КНр	Half-saturation concentration for phosphorus uptake by algae	0.005	g P m ⁻³
KLC	Labile particulate organic carbon dissolution rate	0.02	d ⁻¹
KLN	Labile particulate organic nitrogen hydrolysis rate	0.08	d-1

Symbol	Definition	Value	Units
KLP	Labile particulate organic phosphorus hydrolysis rate	0.1	d-1
KRC	Refractory particulate organic carbon dissolution rate	Not used	d-1
KRN	Refractory particulate organic nitrogen hydrolysis rate	Not used	d-1
KRP	Refractory particulate organic phosphorus hydrolysis rate	Not used	d-1
KTb	Effect of temperature on basal metabolism of algae	0.032	°C-1
KTcod	Effect of temperature on exertion of chemical oxygen demand	0.041	d-1
KTg1	Effect of temperature below Tm on growth of algae	0.0035	°C-2
KTg2	Effect of temperature above Tm on growth of algae	0.01	°C-2
KThdr	Effect of temperature on hydrolysis rates	0.069	°C-1
KTmnl	Effect of temperature on mineralization rates	0.069	oC-1
KTnt1	Effect of temperature below Tmnt on nitrification	0.003	°C-2
KTnt2	Effect of temperature above Tmnt on nitrification	0.003	°C-2
MTC	First-order mass transfer coefficient for nitrate flux into sediment	0.1	m d ⁻¹
NTmax	Maximum nitrification rate at optimal temperature	0.04	g N m ⁻³ d ⁻¹
Pm	Maximum photosynthetic rate	250	g C g ⁻¹ Chl d ⁻¹
PRSP	Fraction of production consumed in algal photorespiration	0.25	fraction
Topt	Optimal temperature for growth of algae	25	°C
Tmnt	Optimal temperature for nitrification	30	°C
Tr	Reference temperature for metabolism	20	°C
Trhdr	Reference temperature for hydrolysis	20	°C
Trmnl	Reference temperature for mineralization	20	°C
Wa	Algal settling rate	0.05	m d ⁻¹
WI	Settling velocity of labile particles	0.05	m d ⁻¹
Wr	Settling velocity of refractory particles	Not used	m d ⁻¹
Wiss	Settling velocity of fixed solids	0.1	m d ⁻¹
α	Initial slope of production vs. Irradiance relationship	8.0	g C g ⁻¹ Chl (E m ⁻²) ⁻¹

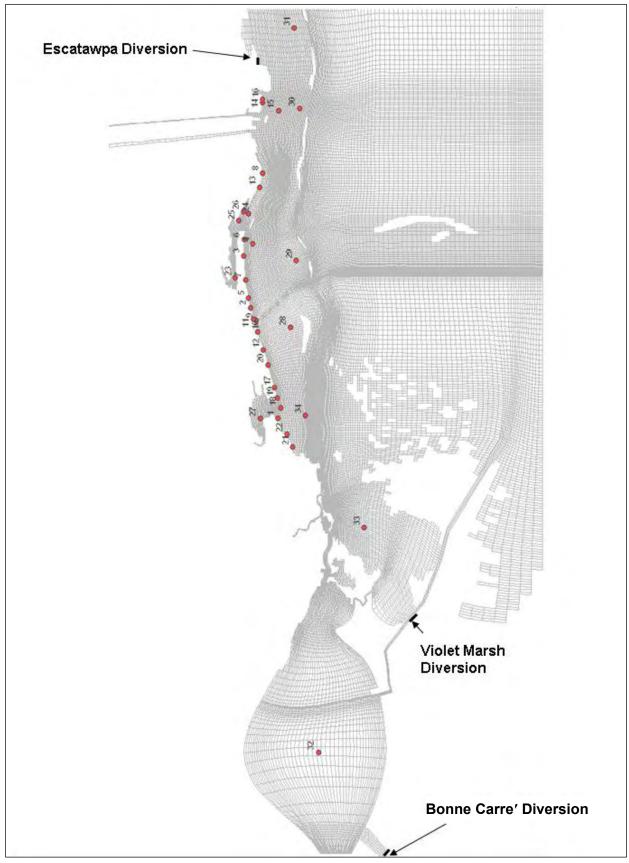


Figure 4-1. Observation station locations.

These data were included to evaluate the model results away from the shore and to provide information on variables not included in the MDEQ data, such as chlorophyll a. Although the USEPA data are for a different year than the study year, they at least provide a reality check of the model.

No data were available for light attenuation at the MDEQ stations, but this variable is plotted for future reference should any such data be taken later. Secchi disk depth observations are available, and the computed values of this quantity were estimated from model light extension according to the relation

$$SD = \frac{1.3}{\lambda} \tag{4-1}$$

where SD is Secchi disk depth (m), and λ is light extension (m⁻¹). Additionally, there were no chlorophyll a data in 1998 for the sound, but model results are plotted to provide a reference for scenario results and to compare with a few values collected at the MS1 to MS4 stations by USEPA in 2002. Stations 32, 33, and 34 were added to provide information on effects of diversions in areas closer to the diversions, such as in Lake Pontchartrain, in Lake Borgne adjacent to the Violet diversion location, and in the western area of the Mississippi Sound.

As noted previously, only a partial model calibration was performed due to the limited scope of this study. Therefore, the calibration is not as good as usually achieved with this model. Additionally, this system is quite large and complex, which complicated identifying and quantifying all the loadings. The model presently contains tributary and atmospheric loadings. However, there are other loadings, such as combined storm water outlets, wastewater discharges, and local runoff that are not accounted for in the model. Including these additional loadings would require a substantial amount of additional work and time. Additionally, there is considerable uncertainty in the loadings that were provided in the model due to the lack of data as discussed in Chapter 3. There was no attention give to calibrating the model for the back bays, which can be sensitive to localized loadings. With more time, it would be possible to focus more on the back bays and to add other observed data in the gulf, such as data from state and federal agencies in Alabama and Louisiana and possibly USEPA and NOAA (National Oceanic and Atmospheric Administration). Having additional data could help improve calibration.

Even though the calibration is not as complete as usually performed, the results are considered sufficient to meet the study objectives. The model can be used to make relative comparisons of water quality for diversion alternatives contrasted against baseline existing conditions, which is useful for evaluating the sensitivity of the system to freshwater diversions.

The model initial conditions were spun up one time during the first calibration run as explained in Chapter 3. These initial conditions were used for all subsequent runs including calibration and scenario runs. Ideally, the model's initial conditions should be spun-up for each new scenario run whenever anything is changed in the model including calibration parameters and modified freshwater flows and loads. The additional spin-up runs were not conducted due to the need to meet the study schedule constraint. It can require a month or longer for the initial conditions to flush out, so some of the model results early in the scenario simulations may not be as accurate as later in the simulation due to inappropriate specification of initial conditions.

Results

The final calibration plots for all stations and all variables are shown in Figure 4-2. Plots are grouped by water quality constituent with each of 12 constituents plotted for each of the 34 stations with surface and bottom layer results presented in each plot. There is no vertical stratification computed by the model (HM or WQM), so surface and bottom lines fall on top of each other in the plots. The 12 constituents in the order presented include chlorophyll a, DO, light extinction, ammonium, NO₂+NO₃, salinity, Secchi depth, TKN, TOC, TP, TSS, and temperature. As noted above, not all stations have observed data, and observed data for 1998 do not exist for all constituents for stations that do have observed data.

The agreement of the model with observed data is good for some constituents (such as temperature, TOC, and Secchi depth), fair for some (TP, DO, NO₃, NH₄, and TKN), and not as good as hoped for others, such as salinity and TSS. Each water quality constituent is discussed below in the order it appears in Figure 4-2.

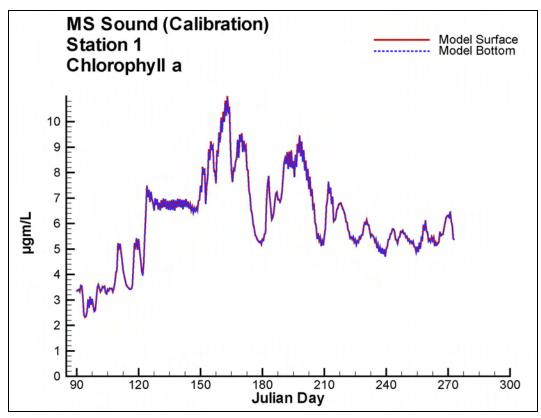


Figure 4-2. Time series of computed and observed water quality constituents following calibration plotted for 34 stations. (Click here for complete collection of time series plots.)

Chlorophyll a

Although the system is nitrogen-limited, there is sufficient inorganic nitrogen to produce algal concentrations as high as 12 μ g/L. Chlorophyll a observed in 2002 ranged between 1.4 and 7.1 μ g/L and averaged 4.1 μ g/L on the surface at stations MS1 to MS4. The chlorophyll a at stations 28 to 31, which correspond to the four MS stations, ranged between about 1 and 6 μ g/L with the trend approximating about 4 μ g/L (Figure 4-2). Higher concentrations are expected nearshore, which the model demonstrates. In general, model chlorophyll a values look very reasonable.

D₀

Model DO values generally look good early during the simulation, approximating saturation. Computed DO is not as good in late summer probably due to the fact that there were no SOD data included in the simulation. DO concentrations can probably be improved significantly by activating SOD. Activating SOD in this version of the ICM model code would have required coding corrections and modifications and time to implement. To meet the short study time constraints, the coding changes were not implemented.

Determining and introducing any omitted loadings, such as storm water outlets and wastewater discharges, should also improve the calibration for DO, especially in the back bays.

The phytoplankton concentrations are probably not elevated enough to cause very significant diurnal fluctuations in DO. However, it is possible that benthic algae and macrophytes could be impacting DO, especially in the shallow areas such as the back bays. The ICM model code contains benthic algae and SAV, but they were not activated in this study due to the limited scope of study.

Some of the observed low DO concentrations are near-bottom measurements, indicating there is some vertical stratification in DO. The model is not capturing any vertical stratification of DO. In fact, none of the water quality constituents exhibit any vertical stratification in the model. It is possible that there are periods of slight stratification that the model is not reproducing. Any future work should ensure that the HM is accurately calibrated to reproduce any observed salinity stratification, even for brief periods. The WQM can not reproduce vertical stratification unless the HM produces it since the vertical eddy diffusivity coefficients computed by the HM are used in the WQM. Even with vertical stratification, SOD would have to be activated in the WQM to produce much difference between surface and bottom DO. It should be noted that other constituents may be of more interest than DO for freshwater diversions since DO is generally near saturation except intermittently for nearshore and in back bays.

Light extinction

Although there were no light extinction data for 1998, there were some such data collected at the MS stations in July 2002. Those measured values ranged between 0.56 and 1.77 m⁻¹ with an average of 0.95 m⁻¹ for the four stations. The model values at these stations were generally around 0.7 m⁻¹ during July 1998. Model light extinction fluctuated around 1 m⁻¹ for most stations with most maximum values of about 1.3 m⁻¹ with the exception of several stations with a few values up to about 3 m⁻¹. The need for additional model spin-up following calibration changes is obvious in viewing the light extinction results, although light extinction should increase during the algal growing season as the model indicates.

Ammonium

Ammonium nitrogen generally ranges between about 0.1 and 0.2 mg/L for both the model and the data. The model does a fairly good job of reproducing much of the data. There are periods of relatively high observed NH₄ that the model does not capture, which could be due to external loadings that are not included or sediment release in back bays under lower DO conditions that are not simulated in this model. Ammonium concentrations are less offshore according to the model and as would be expected in the field. There is one curious result at station 33 (in Lake Borgne) that deserves more attention since the ammonium increases steadily during the summer to rather high values of 0.6 mg/L. This could be due to atmospheric deposition coupled with limited flushing in this area. Ammonium concentrations observed at the four MS (MS1 to MS4) stations in 2002 were all at the detection limit of 0.05 mg/L, whereas, the model indicates values of slightly higher than 0.05 at these stations during mid to late summer 1998.

Nitrate

Nitrate results include nitrite for both the model and observations. Much of the observed data is at the detection limit of 0.02 mg/L. The model results also approximate this value for much of the time, but model results are very spiky with a few nitrate concentrations up to 1.0 mg/L and some outliers up to as much as 3 mg/L. It is not clear that these spikes are real because the observed data are too sparse. The model results look fairly good when compared with the data in the time series plots of Figure 4-2, only the model shows more spikes than the data. Sediment denitrification flux was not activated in the model due to the short study scope and time constraints. Activating this feature could decrease nitrate concentrations substantially, especially where the highest values are closer to shore and in the back bays. Actual nitrate concentrations in July 2002 were at the detection limit of 0.05 mg/L at the MS stations offshore. The model results at these stations were also at or below 0.05 mg/L during summer 1998. Further calibration for nitrate and ammonium is expected to improve model results for both.

Salinity

Salinity results were the most troubling aspect of the model calibration. Salinity is usually one of the easier variables to calibrate since there is actually nothing in the WQM that can be used to calibrate salinity, other

than the horizontal eddy diffusion coefficient. This coefficient was not varied during calibration but was set to 2.0 m²/sec, which is the order of magnitude of values that should be used in the coastal zone. In general, the model underpredicts salinity during the summer. The deviation from observed salinity increases as the summer progresses. It is hypothesized that this deviation could be due partially to evaporation, which was not included in the model simulations. Relatively dry conditions were experienced during summer 1998. As the summer progresses, salinity could increase due to evaporation. The spring-summer average evaporation rate is approximately 0.4 cm/day (Bell 2004), which can result in a substantial water loss over 180 days (approximately 0.7 m if no rainfall). This is a lot of water loss when the water is relatively shallow (on the order of several meters) and more isolated as near the shore and in the back bays. The WQM contains a routine for handling evaporation, but evaporation must also be included in the HM, which it was not, in order to use the routine. Any future modeling of this system should include evaporation in both the HM and WQM.

Another possible explanation for the underprediction of salinity is saltwater intrusion into wetland areas in the back bays or along the coast. The coastal boundary of the model is a solid wall, whereas, in the field, this boundary is somewhat porous due to wetlands that absorb intruding water during flood tide. Such areas can serve as a storage area, thus allowing more saline water to intrude further inland causing saltier conditions along the coast. It is possible to add wetland/marsh storage areas to the model grid, but this was beyond the scope of this study. Development of initial conditions through model spin-up may also improve salinity predictions early in the simulation. It is obvious from some plots that initial conditions are not where they should be, thus impacting results for a month or longer.

The HM and WQM salinity along the outer boundaries was varied spatially with salinity increasing from west to east and north to south. However, the salinity on the boundaries was held constant over time. Salinity may vary with time along the boundaries for areas where the salinity is less than ocean salinity. Such temporal variations could influence values computed nearshore. There is not enough measured data to provide a time-varying salinity boundary condition along the outer boundaries, so it is doubtful that any improvements to the model can be made in this regard. The time-invariant salinity gradients along the outer boundaries should be investigated too since this could also affect the accuracy of computed salinities.

There are two other model improvements that could correct the underprediction of salinity. One is the addition of temporally and spatially varying horizontal eddy diffusivity in both the HM and WQM where the diffusivity is calculated based on local depth and flow conditions. Some sensitivity tests indicated that increasing the eddy diffusivity from 2 to $10 \text{ m}^2/\text{sec}$ increased salinity in the summer at the observation stations. It would be better to compute horizontal eddy diffusivity based on turbulence considerations rather than setting it to a constant value. Another potential improvement is to add surface wave radiation stresses due to wind-generated waves. With the wind blowing towards shore during most of the summer, wind-generated waves can contribute to the onshore current, which could bring saltwater closer to shore to be mixed with freshwater. It is difficult to assess the importance of this contribution without actually adding the feature to the model and testing it.

As a final note, the vertical salinity and temperature stratification was not calibrated in the HM. Calibration for proper stratification could also affect the intrusion of saltwater, thus affecting the salinity calibration overall.

Secchi depth

The model does a good job of matching the observed Secchi depth data (Figure 4-2). Both the model and the data generally have Secchi depths of about 1.0 m. Model values are greater offshore as would be expected if data were available to show this. The need for model spin-up to reach proper initial conditions is obvious in these plots. There are a couple of back bay stations where the model predicts higher than observed Secchi depths, illustrating the need to place more emphasis on the back bays in any subsequent modeling of this system.

TKN

Total Kjeldahl nitrogen (TKN) is the sum of ammonium nitrogen and total organic nitrogen. TKN varies widely in the model and in the observations since it is related to organic matter, which is related to algal production, that also varies widely. Observed TKN concentrations were as high as 2.5 mg/L, but mostly peaks were around 1.0 mg/L, whereas the model exhibits similar peaks. The comparison of model and observed TKN looks fairly good although both the data and the model are quite spiky. The model shows that TKN decreases offshore, as it should. Observed offshore values at the MS1 to MS4 stations ranged between 0.1 and 0.6 mg/L in

July 2002, whereas the model predicts about 0.2 mg/L at these stations during summer 1998.

TOC

Although there are sparse TOC measurements available, the model is in general agreement with what data were available. Most TOC concentrations were less than 10 mg/L and tended to range between 5 and 10 mg/L for the model and observations. The model shows that TOC decreases farther from shore as would be expected. TOC concentrations are directly related to phytoplankton and chlorophyll a concentrations.

TP

Total phosphorous generally ranged between about 0.05 and 0.1 mg/L. The model does a fair job of reproducing TP concentrations. There are some curious spikes in observed TP, such as at stations 17 and 20, where concentrations jump by an order of magnitude from about 0.1 to 1.0 mg/L. It is not evident what is causing such jumps in the field data unless there were problems in analysis or perhaps some samples were collected near the bottom or in an area where sediment rich in TP had been resuspended. TP is not a crucial water quality variable for assessing the future health of the system since it is not the limiting nutrient in most cases.

TSS

It is not clear why TSS predictions are in so poor agreement with observations. Many of the observations are considerably higher than TSS concentrations prescribed for the tributary inflows. It is possible that benthic resuspension of fine sediment particles occurs in this system. More study would be required to determine the causes of the high TSS concentrations before such concentrations can be properly computed by the model. Also, it should be recalled (Chapter 3) that TSS concentrations were held constant for inflowing rivers due to lack of information, whereas TSS concentration usually varies with river discharge. This simplification could certainly affect model predictions.

TSS is of much interest since freshwater diversions are likely to have elevated TSS, and TSS affects light extinction, which affects SAV and habitat. The model can be used to assess relative changes in TSS when comparing diversion scenarios against base conditions, but it should be recognized

that actual future TSS concentrations with any new management strategy could be higher than the concentrations indicated by the model.

Temperature

The model does a good job of reproducing temperature, as would be expected. There is nothing to directly calibrate in the model for temperature. Observed meteorological data are used to compute equilibrium temperatures, heat exchange coefficients, and short wave solar radiation, if solar radiation is not measured directly. Extinction of solar radiation is based on light extinction, which is computed based on a background attenuation plus attenuation as related to concentrations of phytoplankton (chlorophyll a), ISS, and DOC. Temperature is not of concern for the future health of this system as impacted by diversions. However, temperature is required to modulate various processes, including the dynamics of the biological variables, such as phytoplankton, which affect other water quality variables.

5 Scenario Results

Background

The WQM was applied for three alternative scenarios: (1) diversion of freshwater flow from the Mississippi River at Bonnet Carre' spillway, (2) diversion of freshwater flow from the Mississippi River near Violet, LA, and (3) diversion of all of the Escatawpa River flow into Grand Bay. The locations of where the three diversions are introduced are shown in Figure 4-1. The Bonnet Carre' diversion varied by month and is shown in Figure 5-1. The Violet diversion was a constant flow of 212.4 cms (7,500 cubic feet per second, cfs). The Escatawpa diversion is the flow that occurred in the Escatawpa River during 1998, and those values were varied daily in the model as shown in Figure 5-2. The WQM was applied for the period April-September 1998 using the same inputs as the final calibration run except for different hydrodynamics and different boundary conditions for the diverted flow and associated concentrations of the flow. The HM was run with the same conditions as used for the base conditions that were used in the WQM calibrations for 1998 except that the additional freshwater flows were introduced. A separate HM run was made for each of the three diversions. The water quality concentrations that were characteristic of Mississippi River, which were developed as discussed in Chapter 3, were associated with the first two freshwater diversion flows when executing the WQM. The water quality concentrations of the Pascagoula River were used for the Escatawpa diversion.

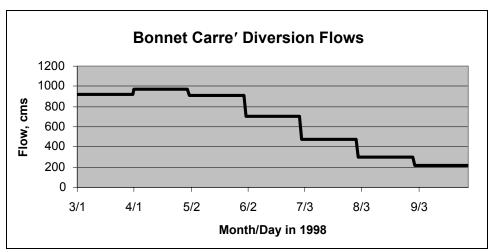


Figure 5-1. Bonnet Carre' diversion flows.

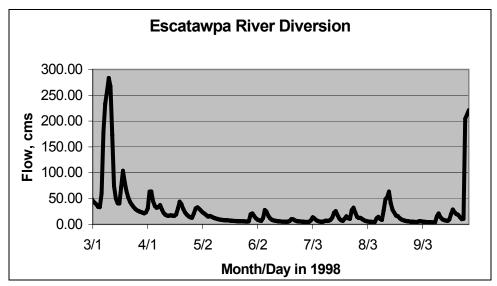


Figure 5-2. Escatawpa River diversion flows.

The results of the three scenarios are plotted together with the calibration results in the same fashion as shown in Figure 4-2 (i.e., grouped by water quality constituent with each station shown separately as a time series). Surface concentrations for each variable and each station are provided in Figure 5-3 for base (which is the same as the calibration as shown in Figure 4-2), Bonnet Carre' diversion, Violet diversion, and Escatawpa River diversion. The results are plotted together for comparison. The results in Figure 5-3 are discussed below in the sections referred to as Time Series Plots.

Results for the four scenarios (i.e., base, Bonnet Carre' diversion, Violet diversion (referred to as Violet Marsh in figures), and Escatawpa River diversion) were post-processed to produce summer average (July—September) surface concentrations for 1998. The summer average results were computed for salinity, chlorophyll a, light extinction, and TSS and are plotted as color contours in planform throughout the model domain. Results for the four water quality constituents are presented in Figures 5-4 through 5-7 with each figure containing the scenario results grouped together for a particular constituent. The results in Figures 5-4 through 5-7 are discussed below in the sections referred to as Summer Average Concentration Contours.

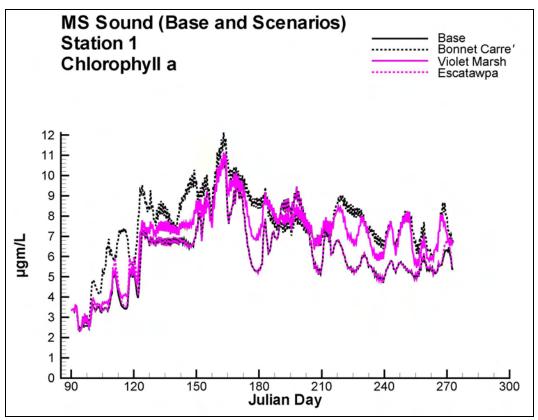


Figure 5-3. Time series of water quality concentrations for all stations and for four scenarios. (Click here for complete collection of time series plots.)

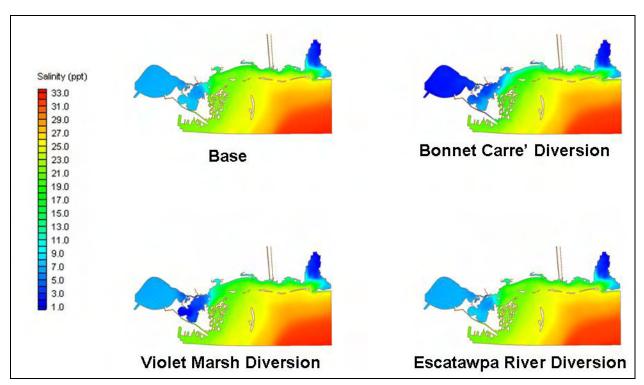


Figure 5-4. Summer average surface concentration contours for salinity for four conditions.

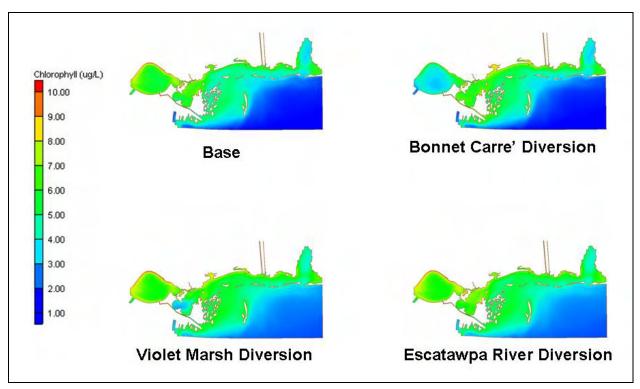


Figure 5-5. Summer average surface concentration contours for chlorophyll a for four conditions.

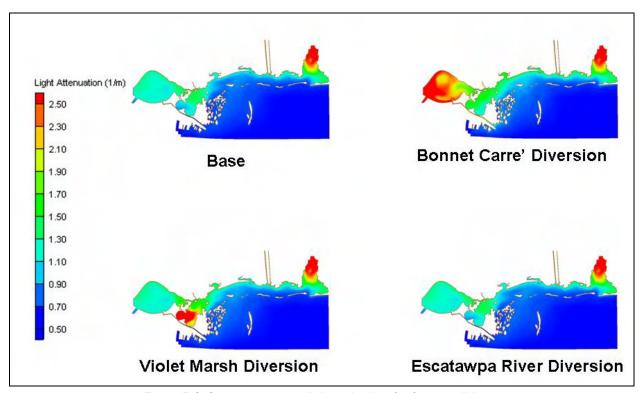


Figure 5-6. Summer average light extinction for four conditions.

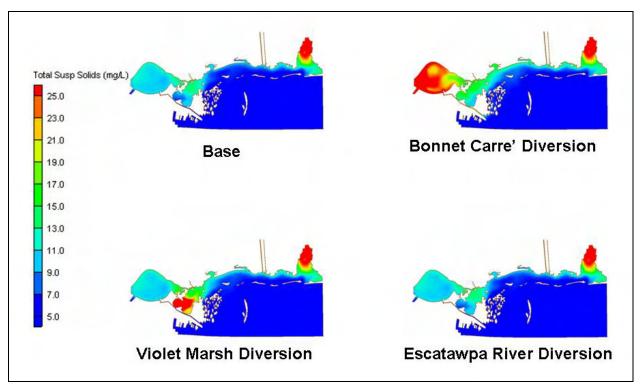


Figure 5-7. Summer average surface concentration contours for TSS for four conditions.

The differences in summer average surface concentrations and light extinction of each diversion scenario from base, i.e., summer average base minus summer average diversion, were determined. These differences were also plotted as color contours throughout the domain and are presented Figures 5-8 to 5-15 and discussed in the sections on Summer Average Concentration Contours.

Bonnet Carre' diversion

Time series plots

The time series plots in Figure 5-3 overwhelmingly show a common trend for nearly all stations. This trend is that the Bonnet Carre' diversion decreases salinity, increases concentrations for nutrients, chlorophyll a, and TSS, and increases light extinction above values computed with base conditions and the other diversions. Salinity is substantially lower for this scenario, as would be expected with the rather high diversion flows. Chlorophyll a is higher at all stations for this scenario, compared with the others, except in Lake Pontchartrain (station 32 in Figure 4-1) due to the high TSS concentrations that are limiting light for algal growth. Likewise, TOC is higher at all stations, due to higher phytoplankton concentrations, except in Lakes Pontchartrain and Borgne (station 33). There were small differences in DO (generally less than 2 mg/L) for the four scenarios, and

there were essentially no differences in temperature. Diversion tended to increase DO due to increased primary productivity. It is likely that increased productivity could lower DO in some areas due to increased depositional flux of detritus, which exerts a SOD. The sediment diagenesis sub-model, which computes SOD with the influence of detritus deposition, was not activated in this study. The benefit of activating this sub-model is that it should provide better insight into how DO would be impacted by diversion.

Nitrogen species concentrations are higher for the Bonnet Carre' diversion than for the other scenarios for all stations except station 33 (Lake Borgne). Ammonium is high in Lake Pontchartrain as a result of nitrification of organic N loads associated with the diversion. The NH $_4$ can be nitrified to NO $_3$ or rapidly taken up by phytoplankton once TSS settles out, which could explain some of the decrease in NH $_4$ as water moves from Lake Pontchartrain to Lake Borgne. TP is increased at all stations for this scenario compared with the others, except for early in the simulation at station 33.

Light extinction is increased substantially compared with base conditions. Light extinction in Lake Pontchartrain (station 32) increases greatly from about 1.0 to between 2 to 4 m⁻¹. The Violet diversion has a greater impact on light extinction in Lake Borgne than does the Bonnet Carre' diversion due to the proximity of TSS loadings for each scenario. TSS is higher for the Bonnet Carre' diversion than for base conditions at all stations and is considerably higher in Lake Pontchartrain.

Summer average concentration contours

The effects of the Bonnet Carre' diversion are very apparent in the western portion the domain, whereas, in other parts of the domain, the changes are not as obvious unless one looks closely at the nearshore conditions (see Figures 5-4 through 5-7). It is interesting how the diversion tends to affect the water quality along the shore of Mississippi Sound, where the influence beyond the barrier islands cannot be detected from the plots, except for possibly chlorophyll a. The diversion has a fairly significant influence all along the coast from Lake Borgne to Mobile Bay. There is also an influence within the Mississippi River Gulf Outlet and where it empties north of Breton Sound.

The differences from base in surface summer average concentrations and light extinction of the Bonnet Carre' diversion are shown in Figures 5-8 through 5-11. For each plot, the difference was obtained by subtracting the diversion values from the base condition values. Thus, if the concentration or light extinction difference is negative, it means that the diversion resulted in an increased concentration or light extinction, whereas a positive difference means the value decreased.

Figure 5-8 indicates that the diversion resulted in a decrease in salinity along and near the coast and in Lake Pontchartrain ranging up to roughly 11 ppt. Figure 5-9 indicates that chlorophyll a was increased (red) in some areas, such as along the Mississippi coast while it decreased (blue) in Lake Pontchartrain due to light limitation. Green indicates no change. It should be recognized that concentrations may be greater and less than the limits of the plotting range, i.e., deep red can indicate concentrations increased with the diversion more than 2.7 $\mu g/L$. The range was set such that variations could be detected throughout the domain. Figure 5-10 shows changes in light extinction. Light extinction is increased dramatically within Lake Pontchartrain and slightly in Lake Borgne and along the Mississippi coast. Figure 5-11 shows differences in TSS. TSS results are

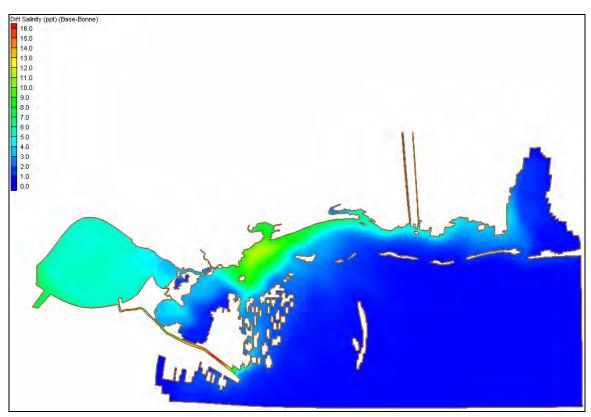


Figure 5-8. Difference in salinity (ppt) from base condition resulting from Bonnet Carre' diversion (base minus diversion).

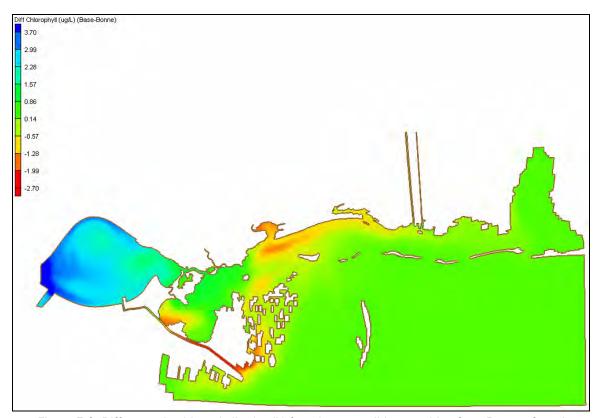


Figure 5-9. Difference in chlorophyll a ($\mu g/L$) from base condition resulting from Bonnet Carre' diversion (base minus diversion).

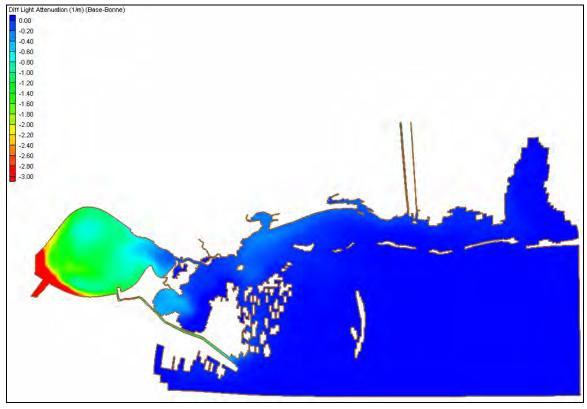


Figure 5-10. Difference in light extinction (m⁻¹) from base condition resulting from Bonnet Carre' diversion (base minus diversion).

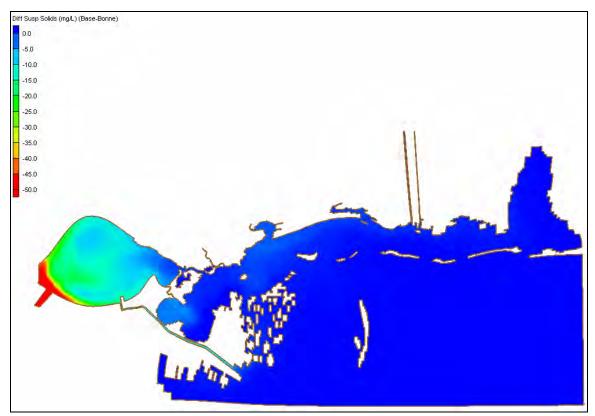


Figure 5-11. Difference in TSS (mg/L) from base condition resulting from Bonnet Carre' diversion (base minus diversion).

similar to those of light extinction where TSS is increased dramatically in Lake Pontchartrain and slightly in Lake Borgne and along the Mississippi coast. Values increased more than 50~mg/L near the diversion entrance, but 50~mg/L was selected as the cutoff so that slighter changes could be detected elsewhere.

Violet diversion

Time series plots

The results for the Violet diversion are very similar to those of the Bonnet Carre', except there is less departure from the base results as there is with the Bonnet Carre' diversion (Figure 5-3). The Violet diversion results in decreases in salinity, increases in concentrations for nutrients, chlorophyll a, and TSS, and increases in light extinction above values computed with base conditions, but less than those of the Bonnet Carre' diversion. Exceptions to these trends occur in the area of Lakes Pontchartrain and Borgne. In Lake Pontchartrain, there is little or no difference between this scenario and the base condition. In Lake Borgne, organic matter is less for this scenario than base conditions due to the higher light extinction limiting algal growth.

Summer average concentration contours

Similar to the Bonnet Carre' diversion, the effects of the Violet diversion are very apparent in the western portion the domain, whereas, in other parts of the domain, the changes are not as obvious unless one looks closely at the nearshore conditions (Figures 5-4 to 5-7). The diversion tends to affect the water quality along the shore of Mississippi Sound, whereas the influence beyond the barrier islands cannot be detected from the plots, except for possibly chlorophyll a. This diversion elevates chlorophyll a near the Chandeleur Islands, more so than the Bonnet Carre' diversion. The diversion influences water quality mostly along the western portions of the sound from Lake Borgne to Bay St. Louis, but not as much as the Bonnet Carre' diversion.

The differences in Violet diversion from base for the summer surface average of the four water quality constituents are shown in Figures 5-12 to 5-15. For each plot, the difference was obtained by subtracting the diversion values from the base condition values. Thus, as stated before, if the concentration or light extinction difference is negative, it means that the diversion resulted in an increased concentration or light extinction, whereas a positive difference means it decreased.

Figure 5-12 indicates that the diversion resulted in a decrease in salinity along and near the coast and in Lake Borgne ranging from about 4 to 7 ppt. The range of change is similar to that of the Bonnet Carre' diversion, but the change does not occur in as large an area. Figure 5-13 indicates that chlorophyll a was increased (yellow) along the Mississippi coast while it decreased (blue) in Lake Borgne due to light limitation. Green indicates no change. The increase in chlorophyll a along the Mississippi coast is not as great as for the Bonnet Carre' diversion and does not occur in as large an area. Figure 5-14 shows changes in light extinction for Violet diversion. Light extinction is increased dramatically within Lake Borgne and slightly along the western Mississippi coast. Figure 5-15 shows differences in TSS. TSS results are similar to those of light extinction. Values increased more than 50 mg/L near the diversion entrance, but 50 mg/L was selected as the cutoff so that slighter changes could be detected elsewhere.

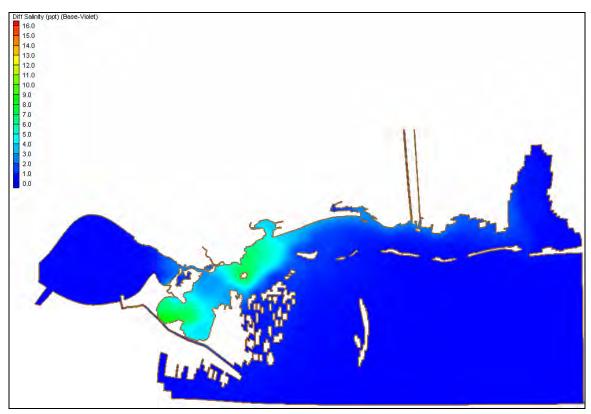


Figure 5-12. Difference in salinity (ppt) from base condition resulting from Violet diversion (base minus diversion).



Figure 5-13. Difference in chlorophyll a $(\mu g/L)$ from base condition resulting from Violet diversion (base minus diversion).

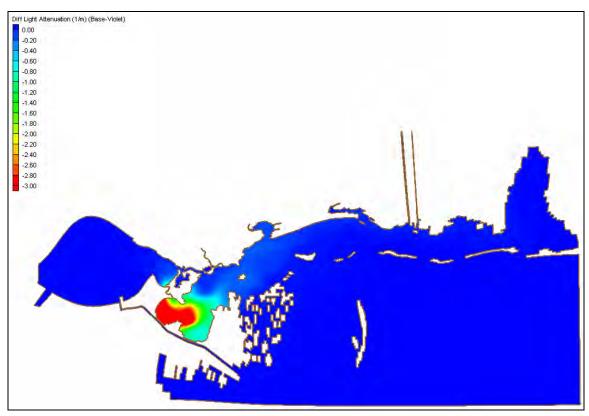


Figure 5-14. Difference in light extinction (m⁻¹) from base condition resulting from Violet diversion (base minus diversion).

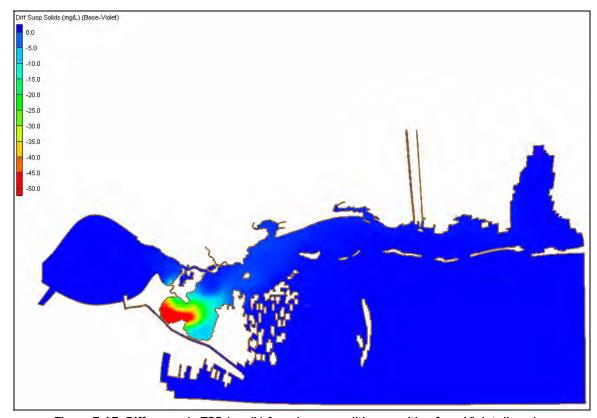


Figure 5-15. Difference in TSS (mg/L) from base condition resulting from Violet diversion (base minus diversion).

Escatawpa River diversion

Time series plots

The results for the Escatawpa River diversion are very similar to those of the base condition. In Figure 5-3, it is difficult to detect a difference between base results and this diversion. The solid black line of the base results is covered up by the pink dotted line representing the Escatawpa diversion. The Escatawpa River flows, and thus the diversion flows, for April—September are quite low (Figure 5-2); thus, it is reasonable that this scenario would have little effect outside of Grand Bay.

An additional plotting station was added within Grand Bay in the latter stages of the study. Grand Bay is adjacent to Mobile Bay. This additional station was added to see if more pronounced changes could be detected within Grand Bay. Figure 5-16 shows the location of the additional station, identified as station 35. Times series of salinity, chlorophyll a, light extinction, and TSS are plotted in Figures 5-17 through 5-20 for all four scenarios at station 35. As can be seen in these figures, the Escatawpa diversion has little effect at this station inside Grand Bay. The summer average plots indicate more significant changes nearshore in Grand Bay as discussed in the next section.

Summer average concentration contours

The summer average concentration contours also show that there is little, if any, difference in the results for the Escatawpa diversion and base conditions (Figures 5-4 through 5-7). The differences in Escatawpa diversion from base for the summer surface average of the four water quality constituents are shown in Figures 5-21 through 5-24. Different scales were used for these figures than those used for Figures 5-8 through 5-15 so that changes can be discerned. All four figures show that the changes in water quality are limited to Grand Bay and the Mississippi shoreline west of Grand Bay. It is interesting how this diversion has some impact within Bay St. Louis although the impact is small. Salinity is decreased a few parts per thousand in Grand Bay and westward along the coast. Chlorophyll a is increased slightly (less than 1 μ g/L), as well as TSS (about 1 μ g/L) in the same areas. Little to no change in light extinction occurred.

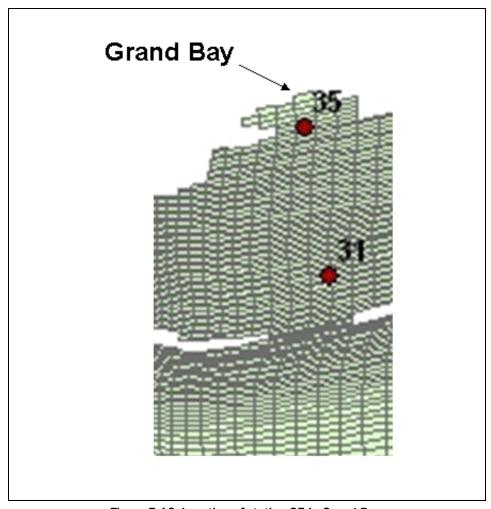


Figure 5-16. Location of station 35 in Grand Bay.

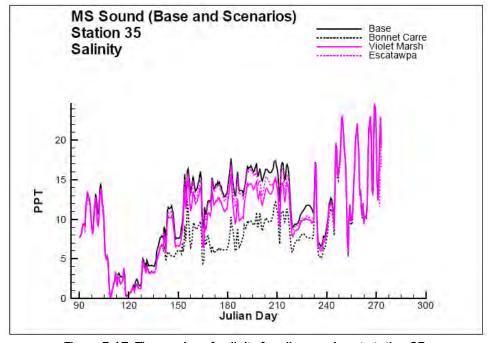


Figure 5-17. Time series of salinity for all scenarios at station 35.

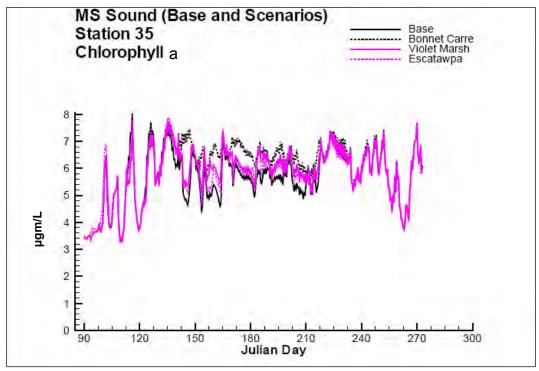


Figure 5-18. Time series of chlorophyll a for all scenarios at station 35.

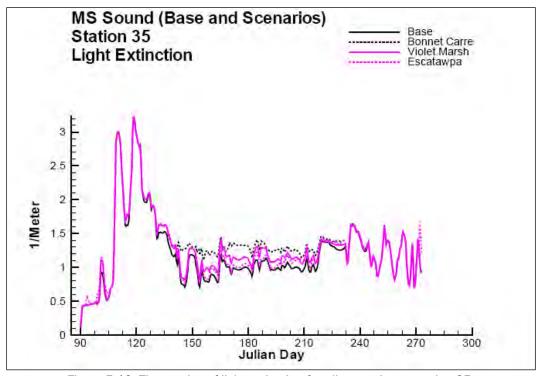


Figure 5-19. Time series of light extinction for all scenarios at station 35.

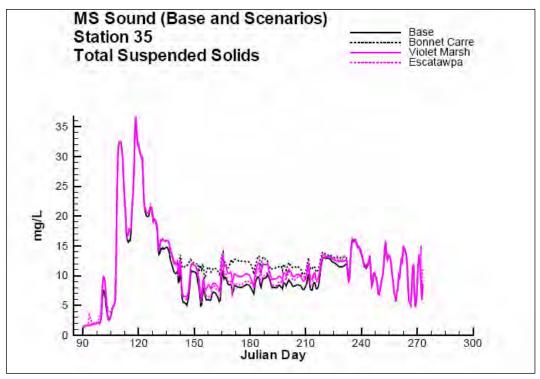


Figure 5-20. Time series of TSS for all scenarios at station 35.

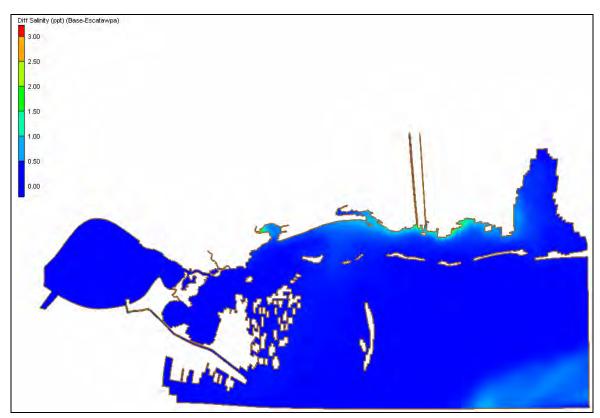


Figure 5-21. Difference in salinity (ppt) from base condition resulting from Escatawpa River diversion (base minus diversion).



Figure 5-22. Difference in chlorophyll a (μ g/L) from base condition resulting from Escatawpa River diversion (base minus diversion).

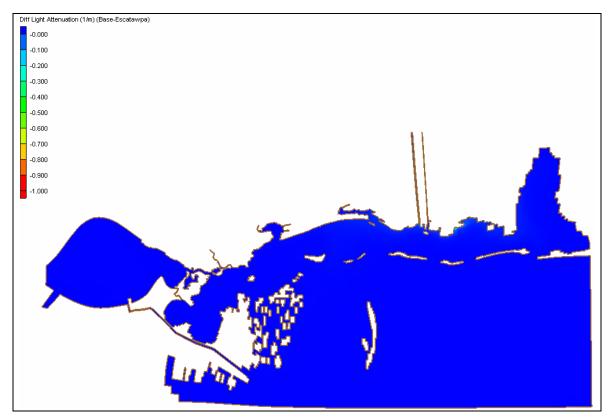


Figure 5-23. Difference in light extinction (m⁻¹) from base condition resulting from Escatawpa River diversion (base minus diversion).

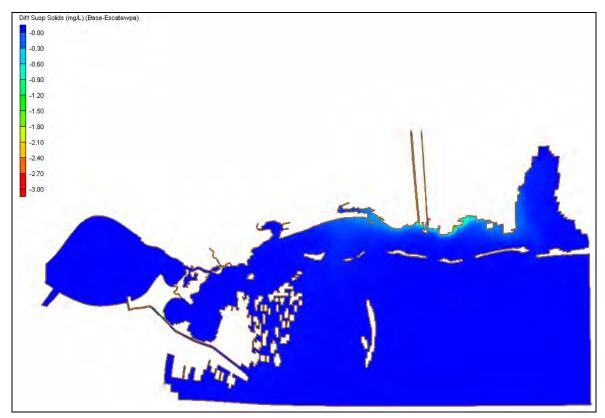


Figure 5-24. Difference in TSS (mg/L) from base condition resulting from Escatawpa River diversion (base minus diversion).

6 Conclusions

This model study indicates that freshwater diversions from the Mississippi River through either Bonnet Carre' Spillway or Violet will result in substantial changes in water quality for the magnitude of flows examined. Summer average salinity was decreased along the western portions of Mississippi Sound by as much as 11 ppt for the Bonnet Carre' diversion. For the Violet diversion, summer average salinity reductions were as great as 6 to 8 ppt in western Mississippi Sound. The effects of freshwater diversions are expected to be felt throughout much of the western extent of the Mississippi coast even for relatively modest diversions (7,500 cfs) introduced on the edges of the system, such as near Violet. The changes in Mississippi Sound water quality resulting from these diversions will include lower salinity, higher concentrations of nutrients, TSS, phytoplankton, and TOC, and greater light extinction, thus, less light reaching the bottom. The latter change could impact SAV densities.

Figures 5-8 through 5-15 and Figures 5-21 through 5-24 show the amount of change relative to the existing base conditions for the three diversions. The change is quite dramatic in some areas for the Mississippi River diversions. However, it is emphasized that the amount of water diverted can make a great difference. The amount of change for each diversion is directly proportional the amount of water diverted. Thus, the Bonnet Carre' diversion had a greater effect than the Violet diversion since the flows were substantially greater for Bonnet Carre'. Bonnet Carre' diversion flows ranged from 200 to 1,000 cms, whereas those for Violet were 214 cms. Similarly, the Violet diversion had a much greater impact than the Escatawpa River diversion for the same reason. The flows of the Escatawpa River were so low during April—September 1998 (averaged 18 cms) that this diversion had little impact except within Grand Bay, where changes were relatively small and mostly confined nearshore.

As with many model studies, results presented here should be treated as relative, rather than absolute forecasts. Thus, the water quality for diversions should be compared relative to the base conditions, rather than used as refined forecasts of future concentrations. This is particularly true for salinity and TSS since these two constituents of interest presented calibration challenges. A more detailed analysis with additional calibration work (potentially including model modifications) is expected to improve the accuracy of salinity predictions. Improving the accuracy of TSS predictions

is more problematic given the paucity of data and lack of full understanding of processes affecting TSS in this system. Model enhancements and more detailed study would be required to refine the accuracy of the water quality model for forecasting absolute water quality conditions with diversions. Such refinement of the model should be considered if the MSCIP proceeds with more definitive plans for diversions.

Results from the water quality model can still be used to estimate relative changes in habitat for living resources of interest. The best approach with the present model is to delineate the areas that exhibit the water quality conditions required for acceptable habitat using model output for base conditions. Model output for alternative diversions can then be used to delineate areas of acceptable habitat with diversion. The percentage change in area with acceptable habitat can then be determined. The percentage change in quality habitat is a metric that is consistent with the relative change in water quality provided by the model. If the model accuracy is improved further to an acceptable level, it will then be possible to delineate the specific location of quality habitat areas based on absolute water quality output from the model.

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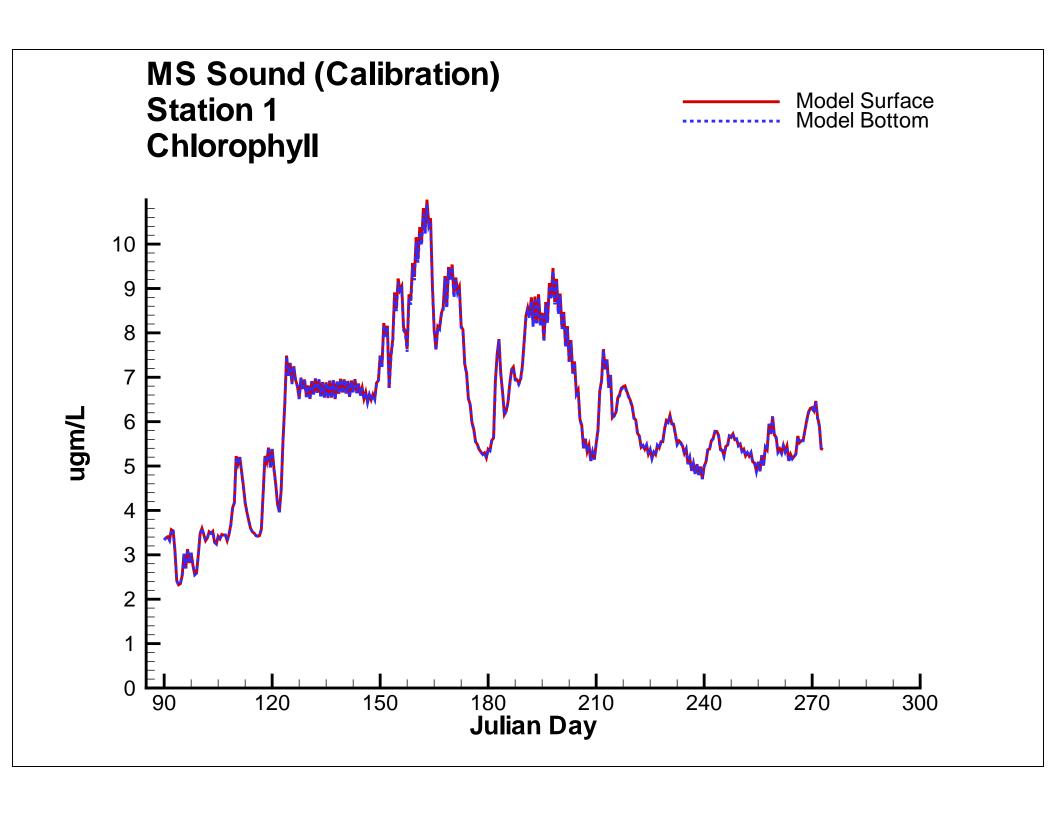
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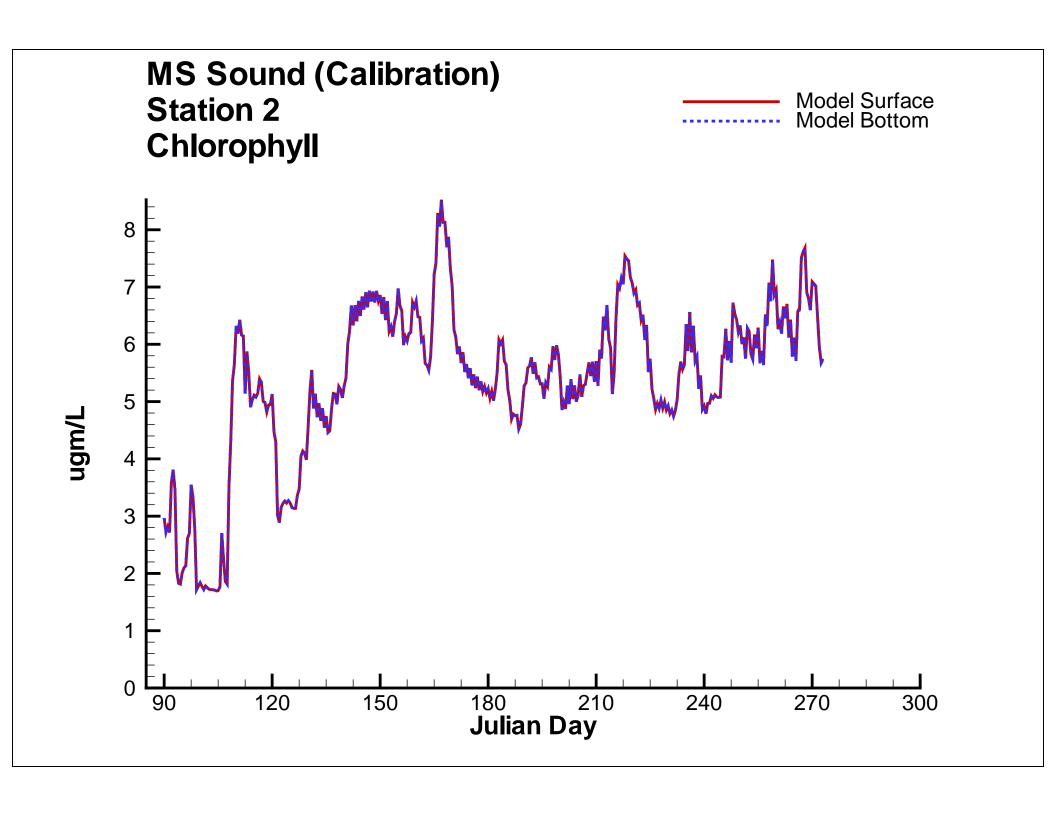
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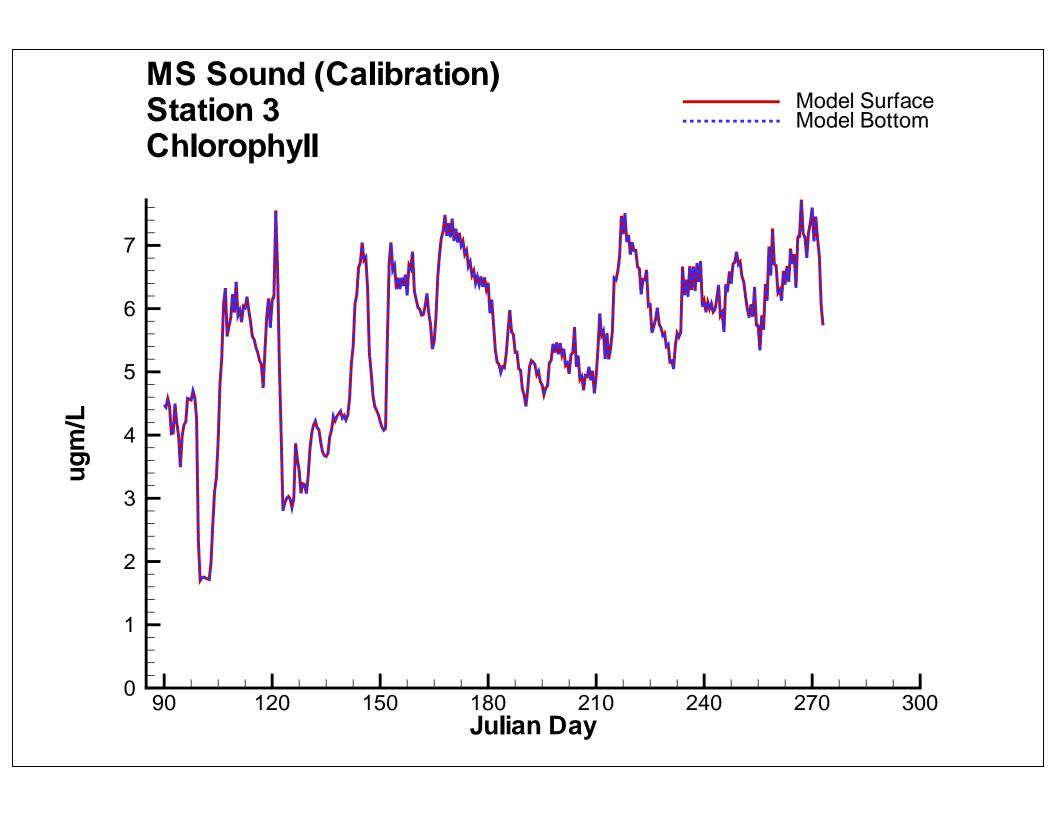
14. ABSTRACT

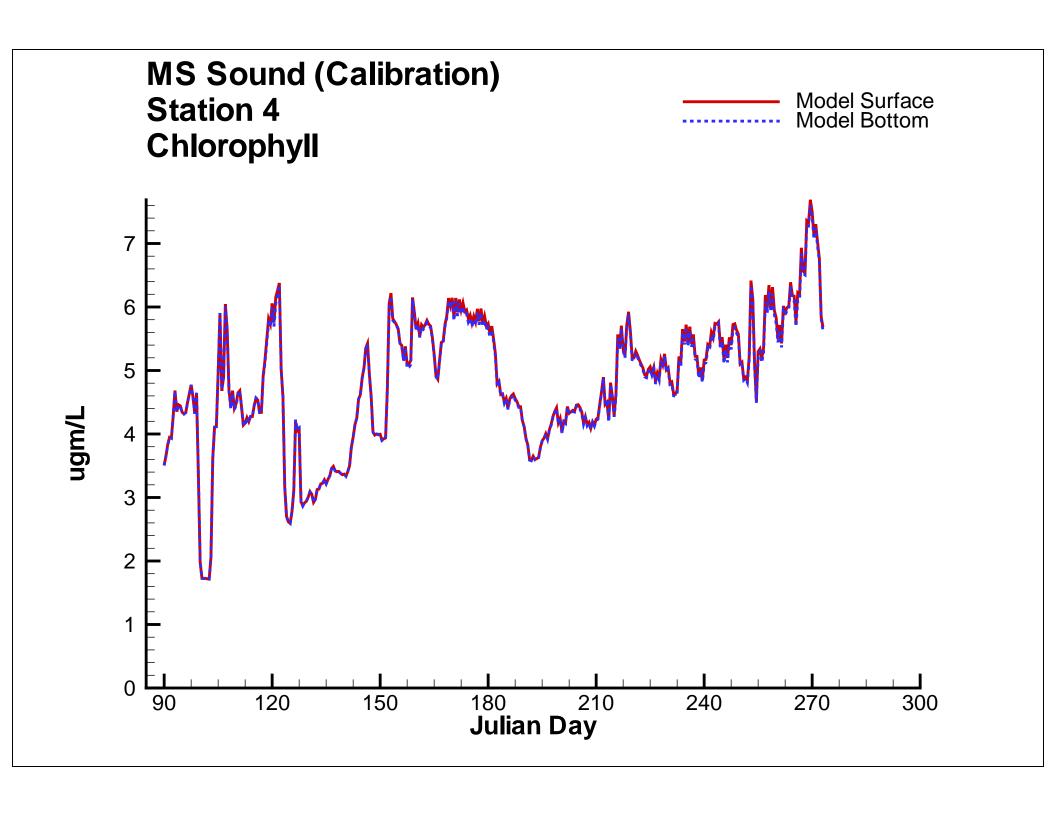
This report describes the development and application of a water quality model to the Mississippi Sound region to address the impacts of various freshwater diversion alternatives. The CH3D-Sigma (sigma level vertical coordinates) model code was the hydrodynamic model that was used to provide transport fluxes for the CE-QUAL-ICM water quality model. The model domain also included Mobile Bay, the Mississippi coastal bays, Lakes Pontchartrain and Borgne, Biloxi Marsh, and part of Breton Sound. The three-dimensional model had five sigma coordinate vertical layers. The model included 15 water quality variables including temperature, salinity, inorganic and total suspended solids, dissolved oxygen, dissolved and particulate organic carbon, various forms of inorganic and organic nitrogen and phosphorus, phytoplankton biomass, chlorophyll a, and underwater light extinction. The model was calibrated for the period April through September 1998. Three diversions were simulated, diversion of freshwater flow from the Mississippi River at Bonnet Carre' spillway and into Lake Borgne near Violet, LA, and diversion of all of the Escatawpa River flow into Grand Bay. Summer average salinity was decreased along the western portions of Mississippi Sound by as much as 11 parts per thousand for the Bonnet Carre' diversion. For the Violet diversion, summer average salinity reductions were as great as 6 to 8 parts per thousand in western Mississippi Sound. The Escatawpa River diversion had little effect on Mississippi Sound. The Mississippi River diversion will also result in higher concentrations of nutrients, TSS, phytoplankton, and TOC, and greater light extinction, thus, less light reaching the bottom.

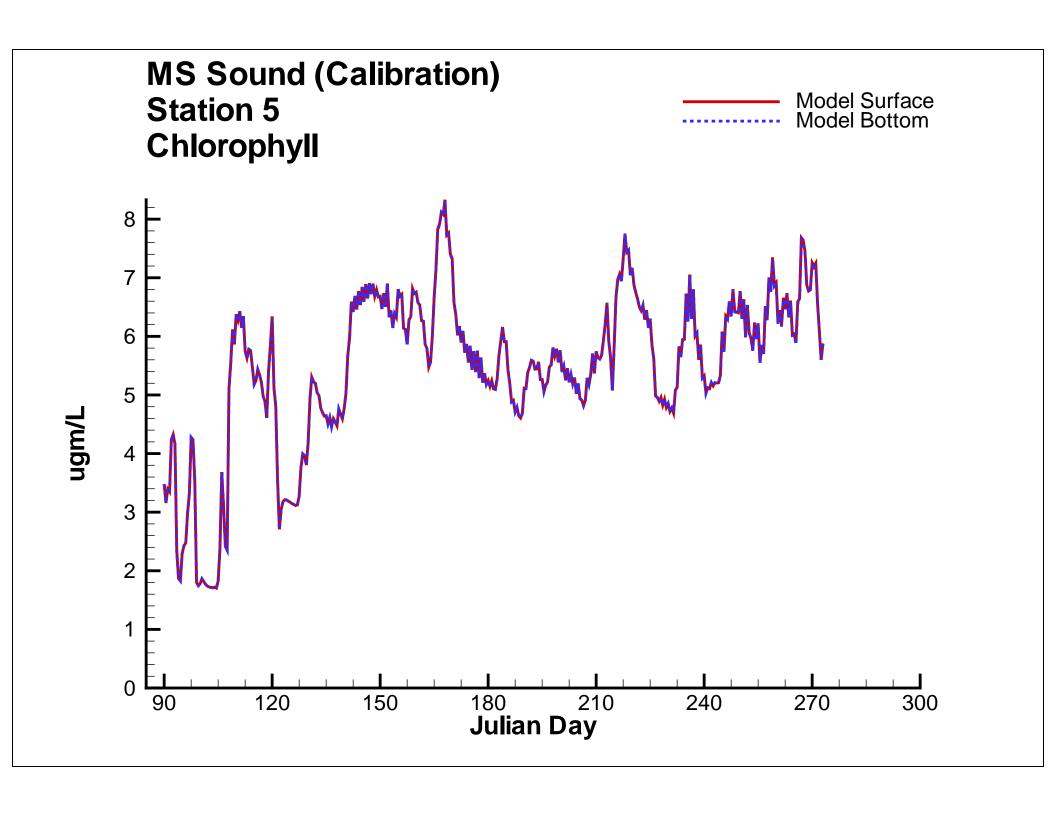
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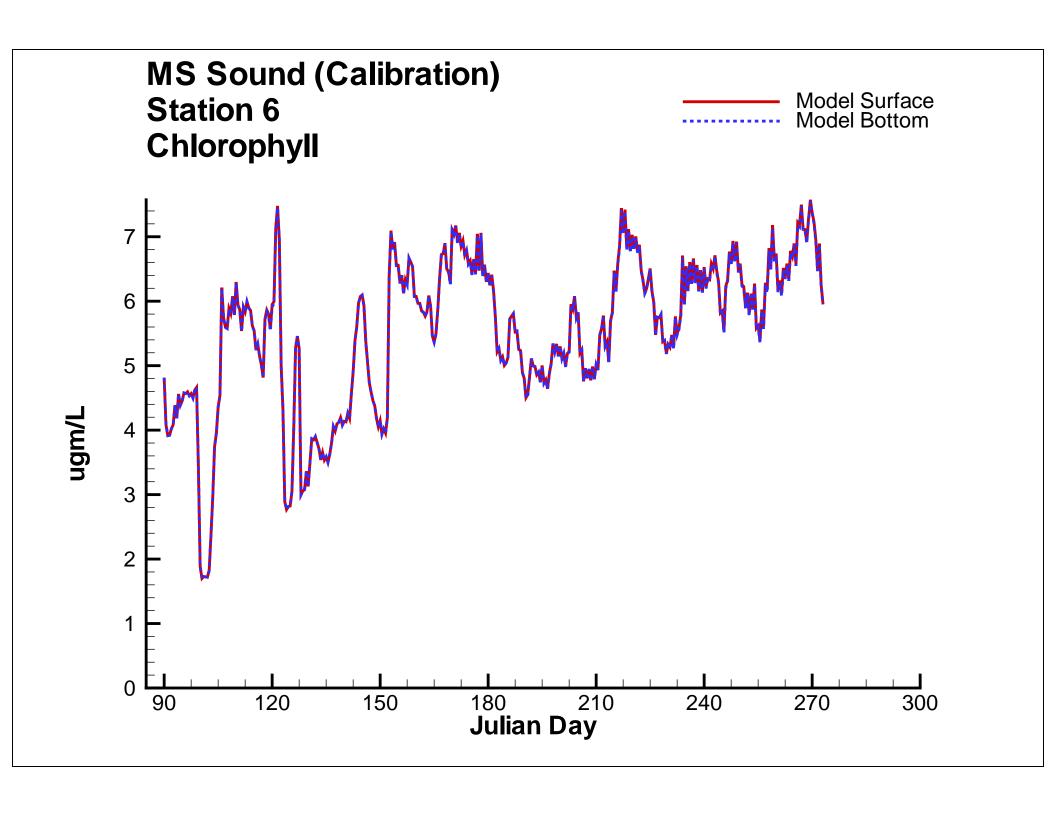


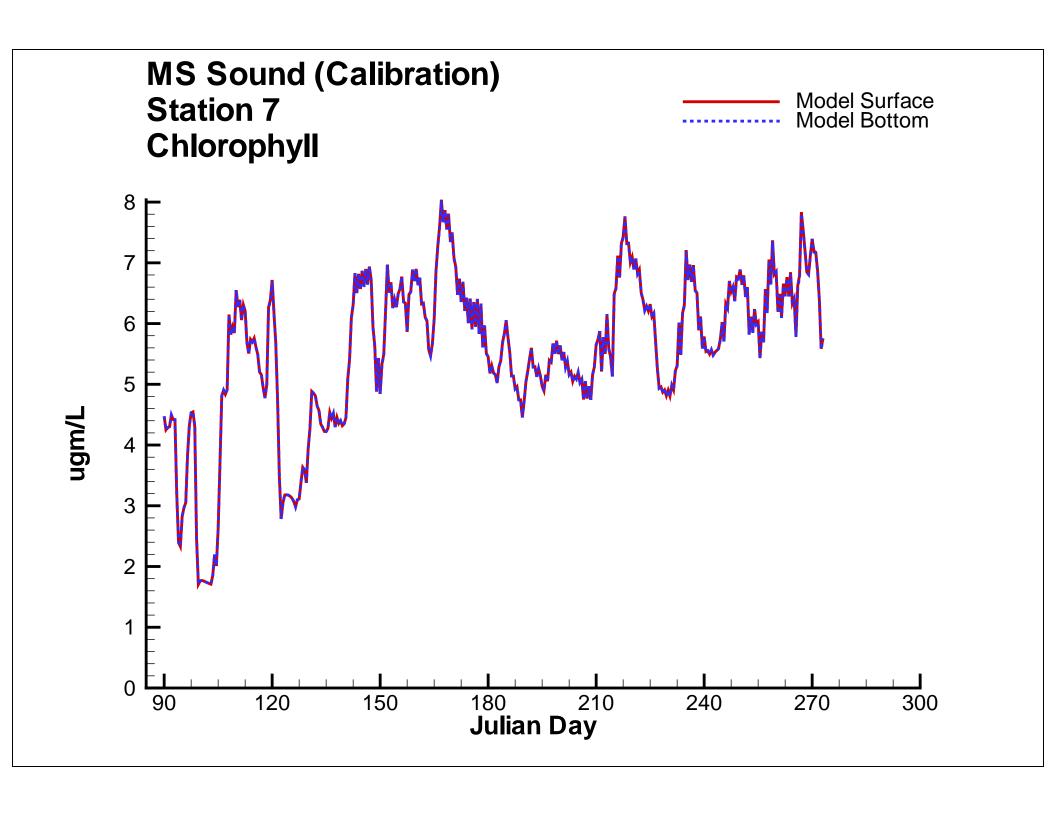


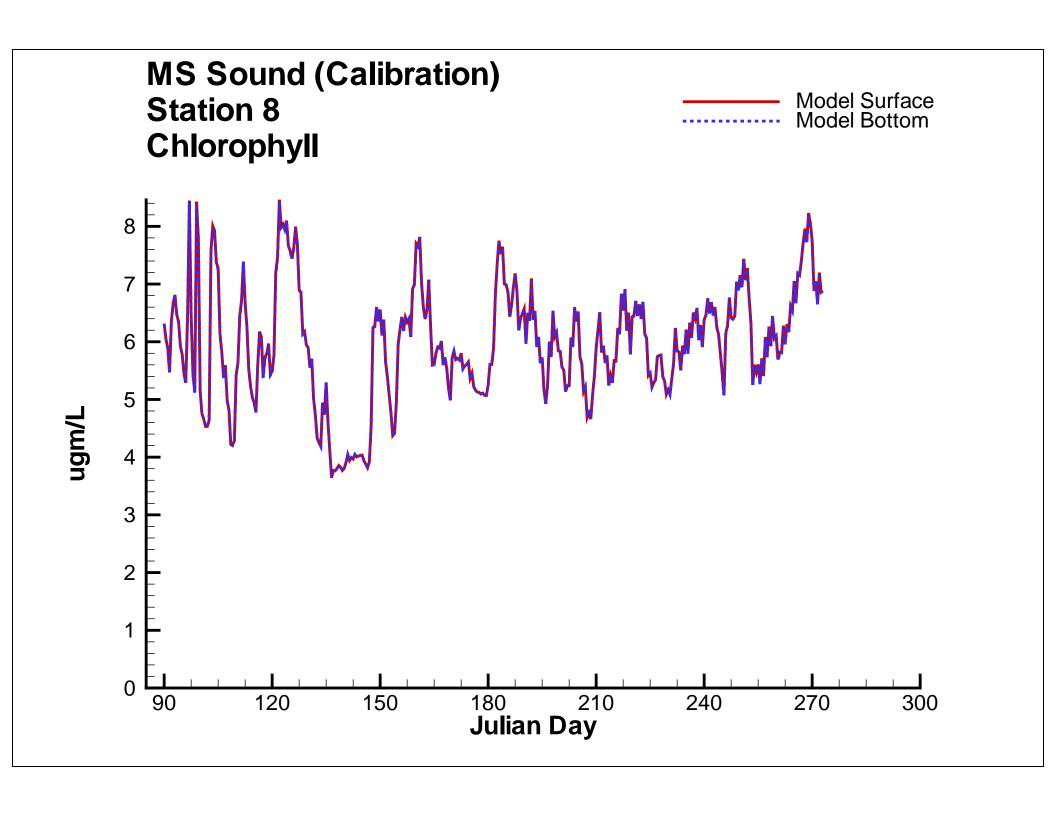


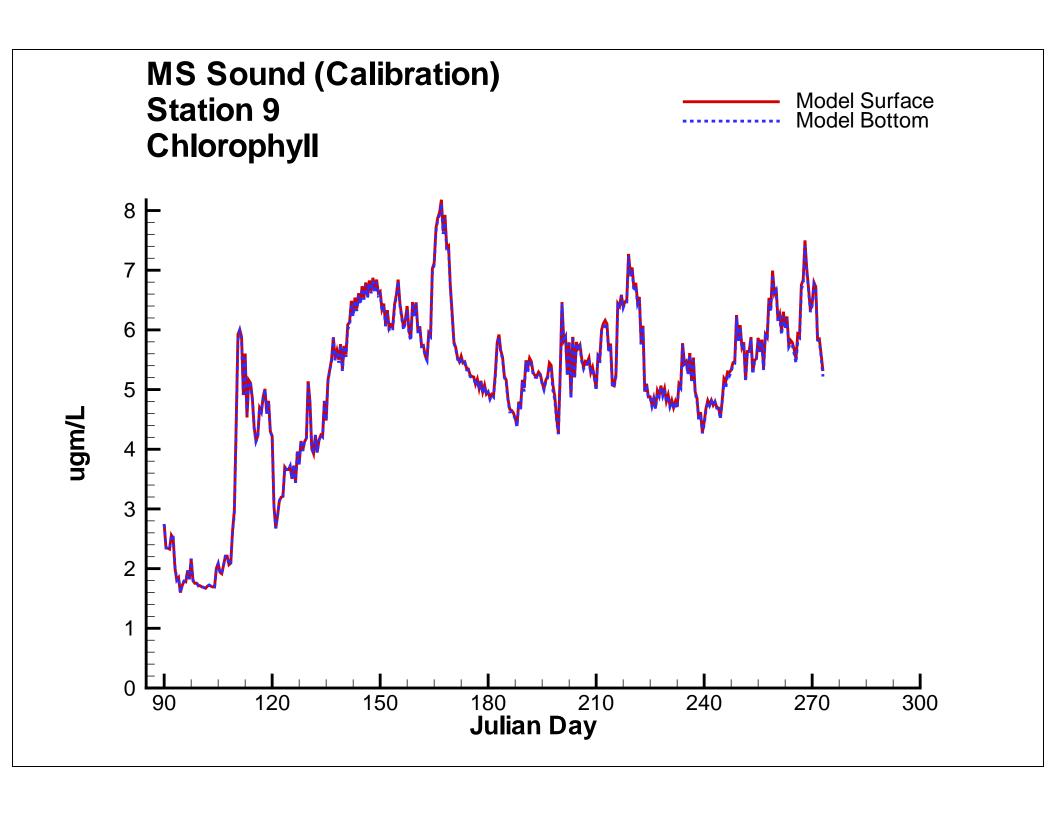


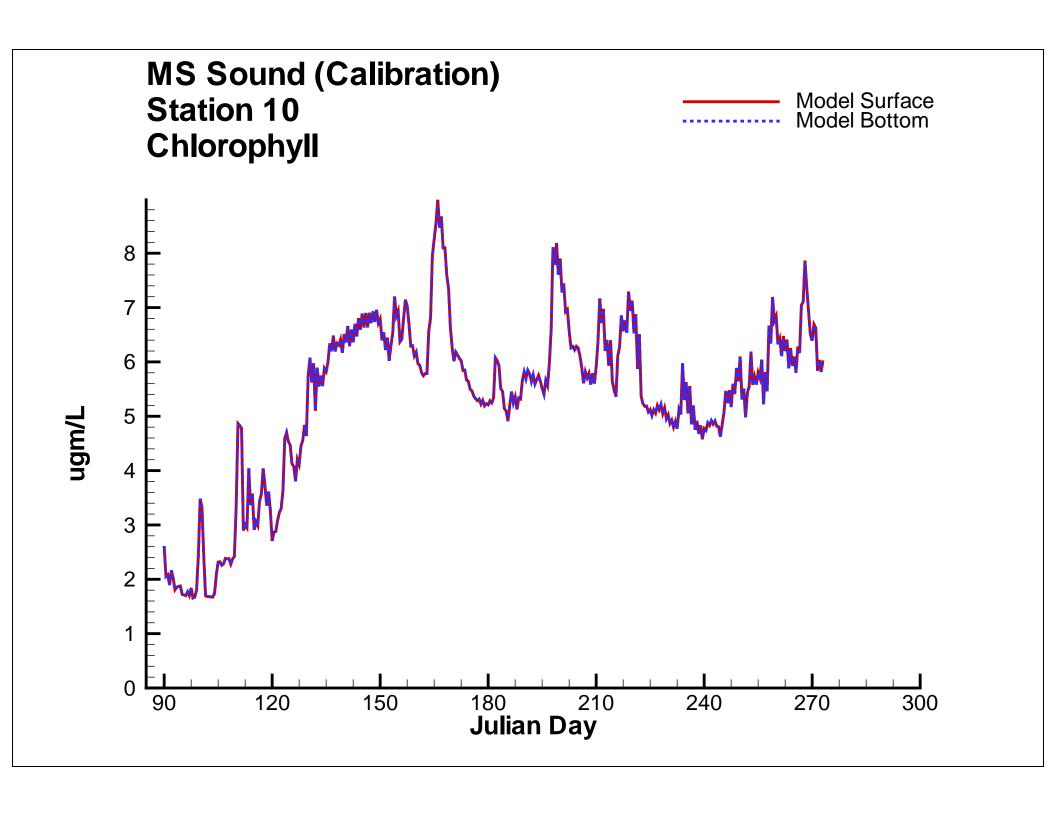


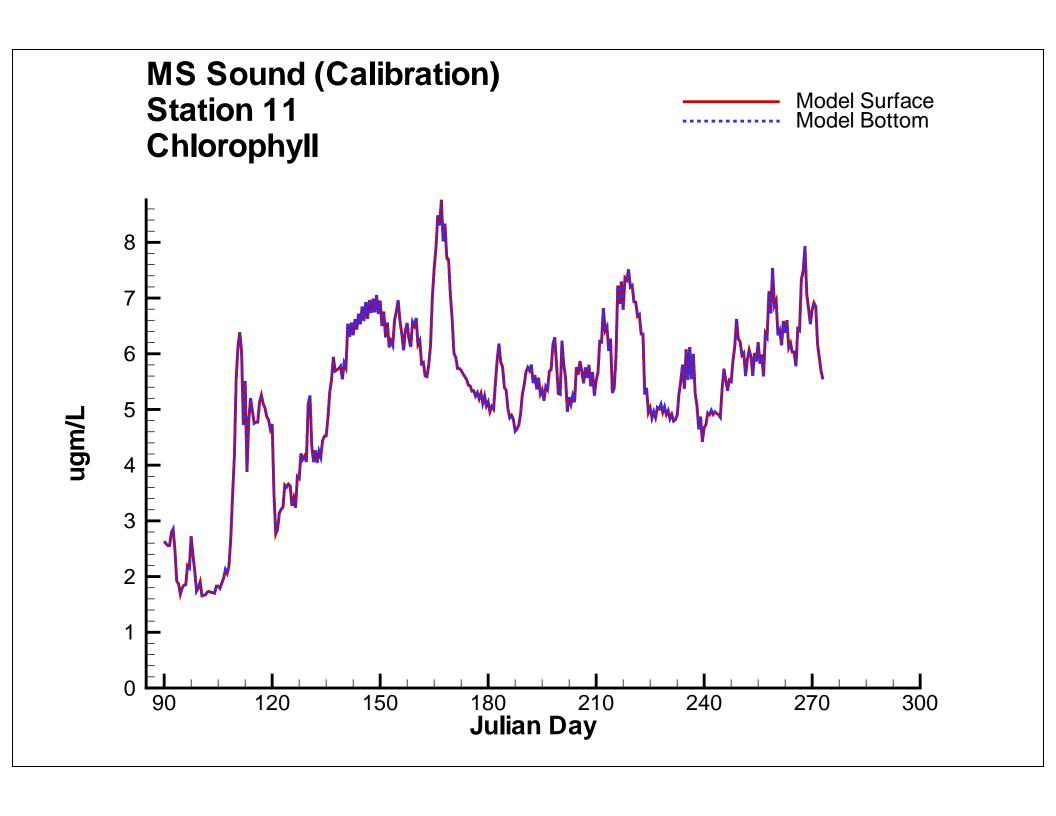


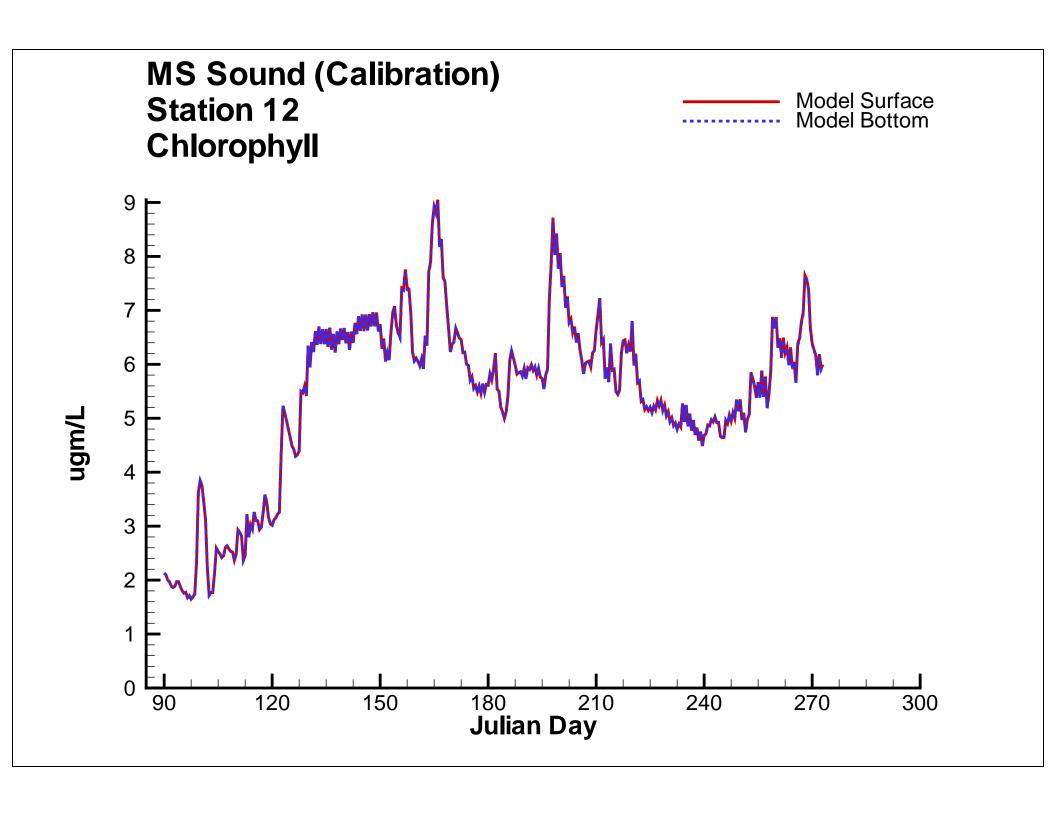


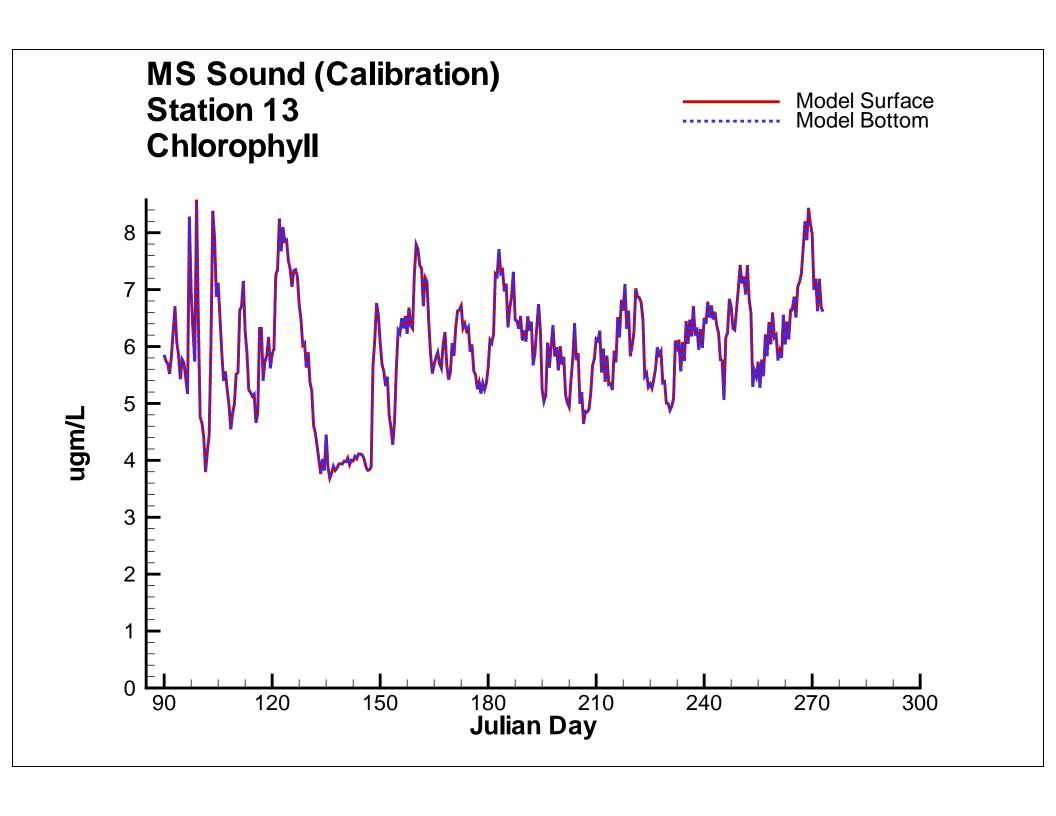


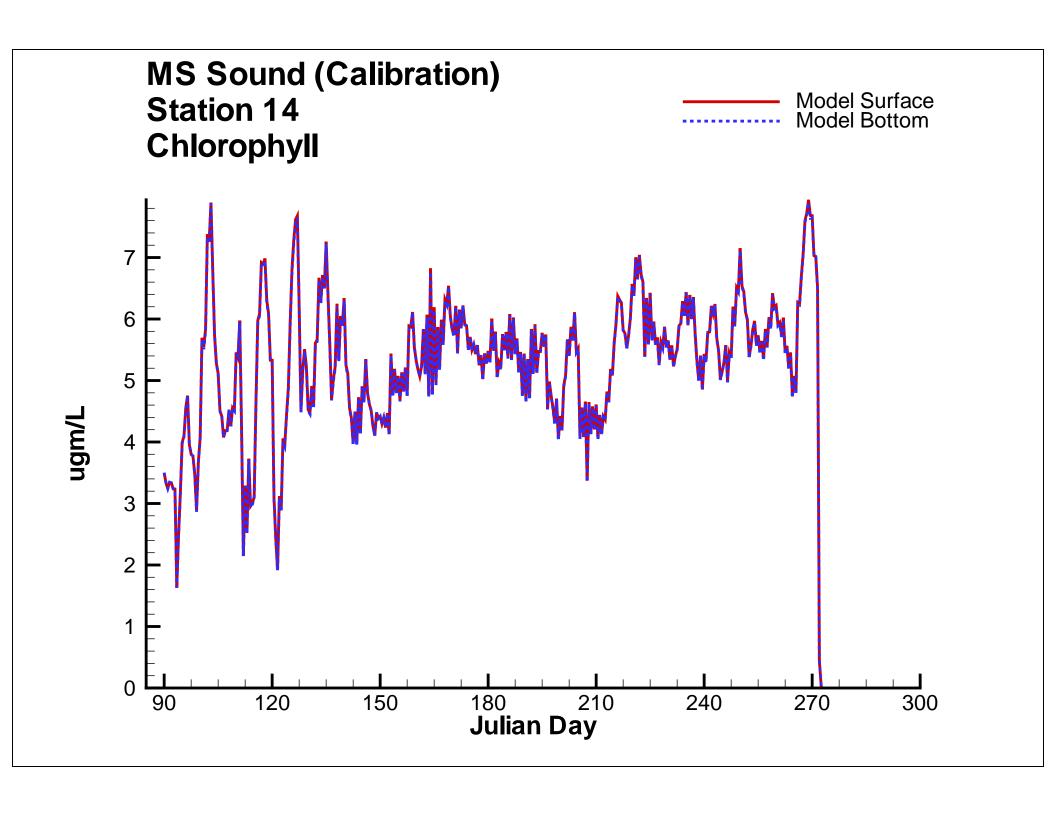


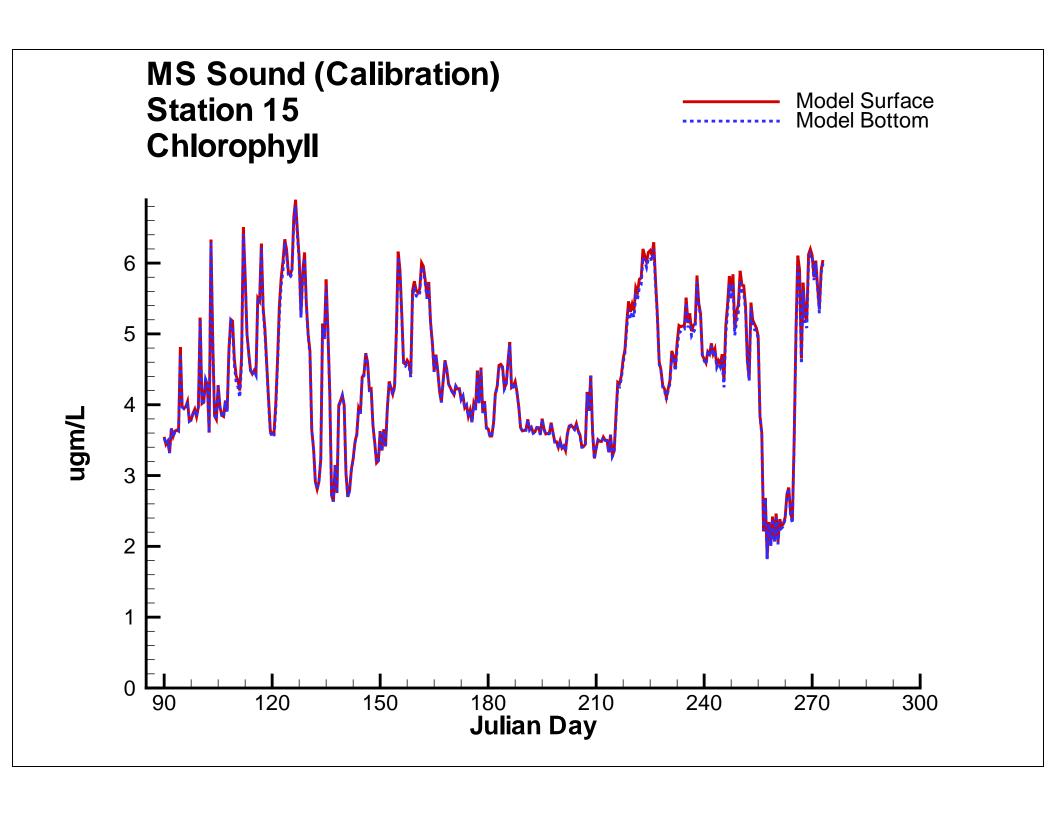


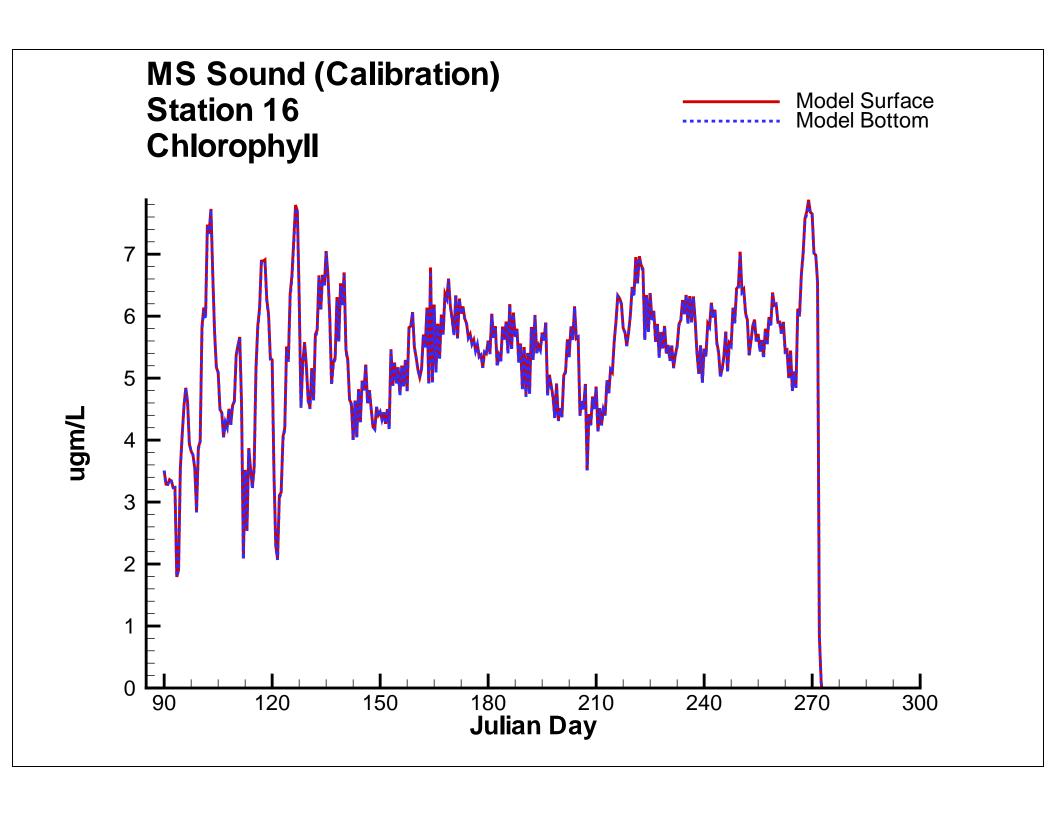


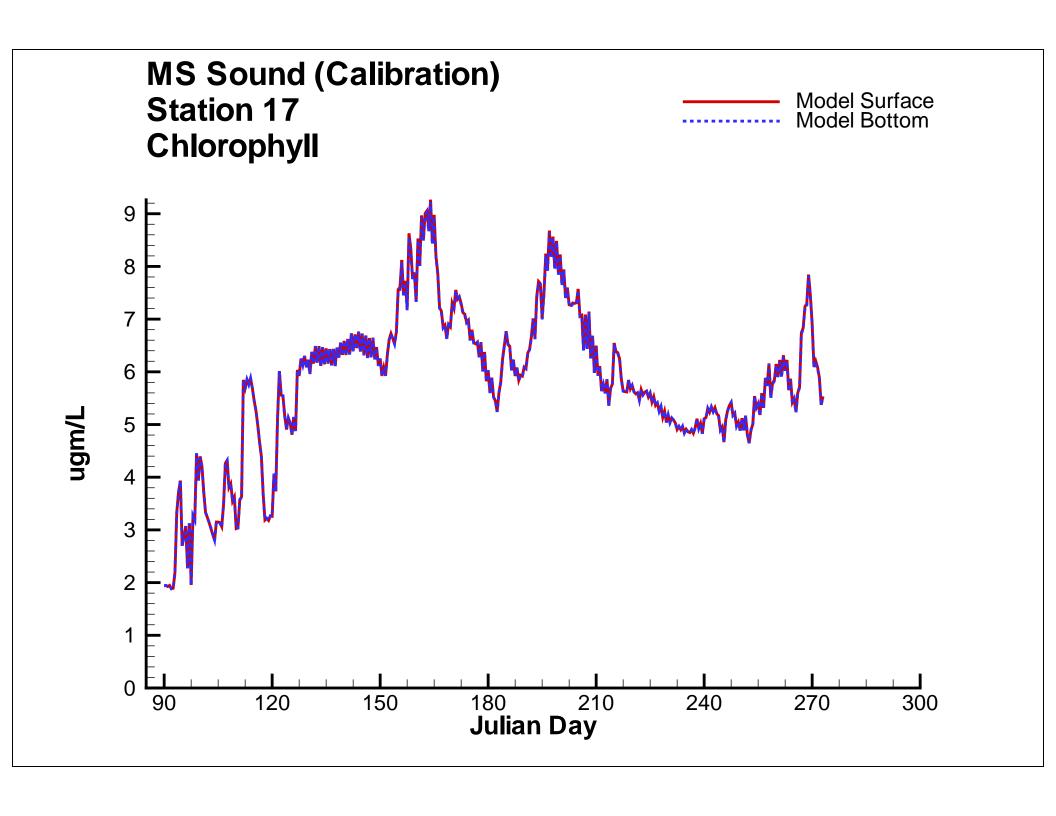


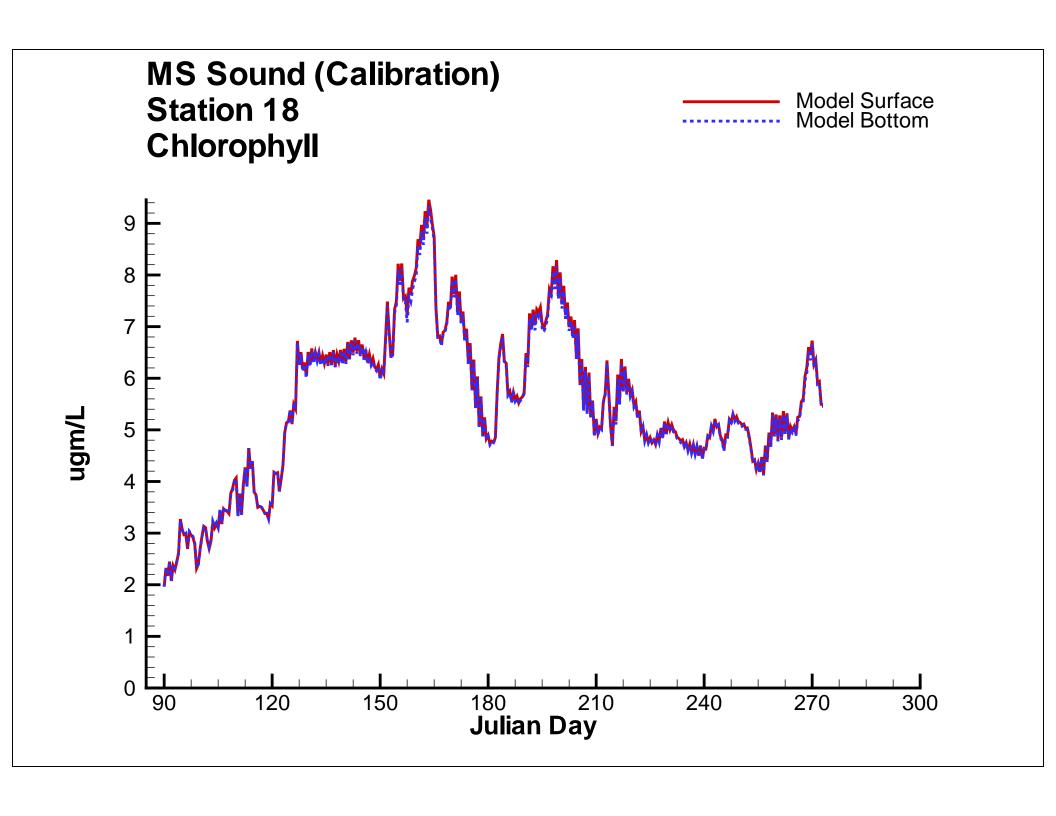


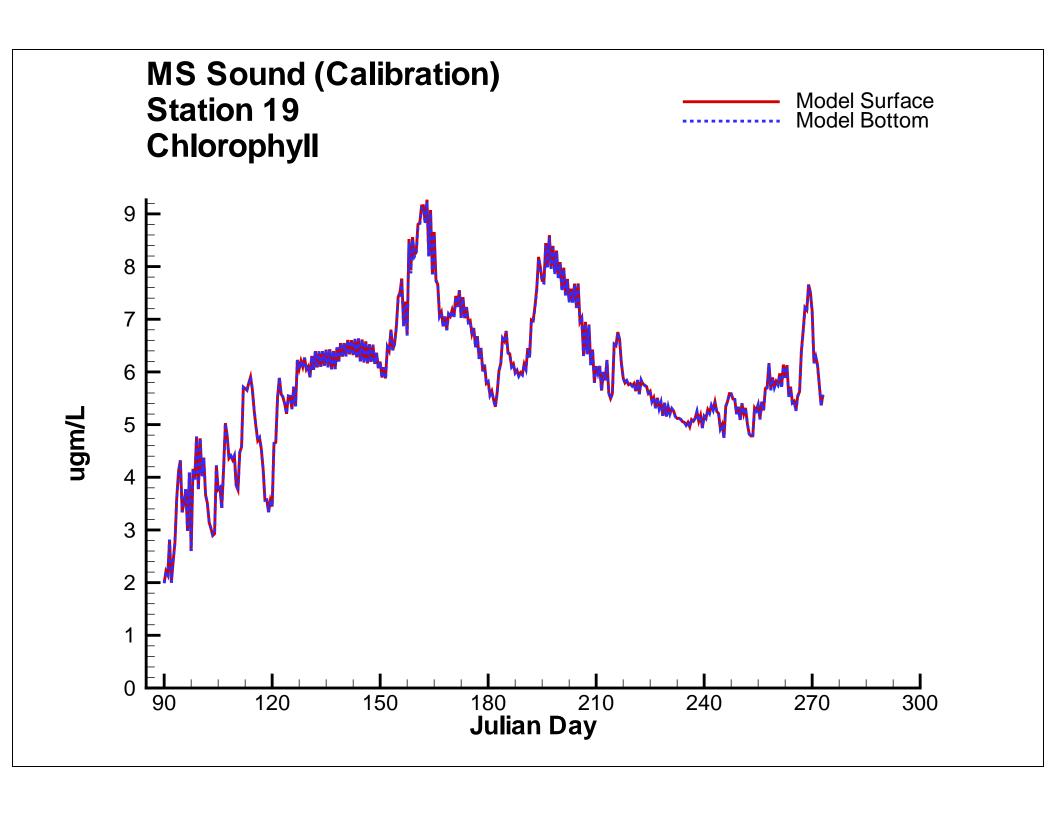


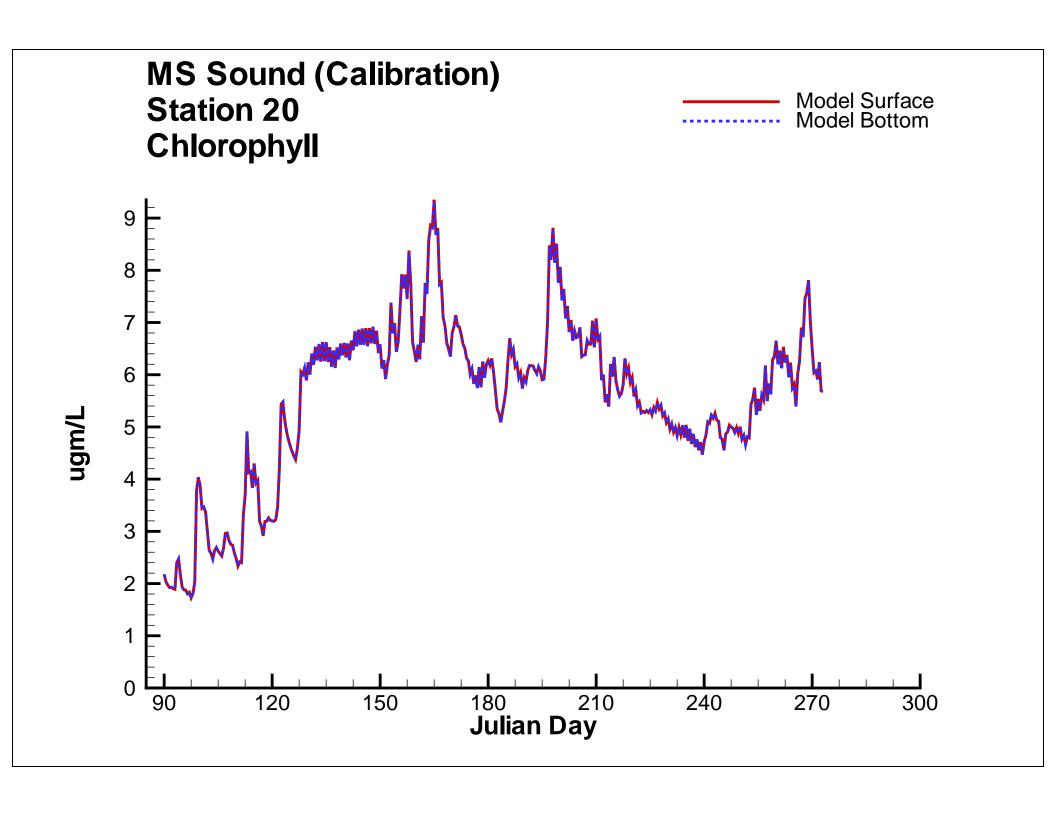


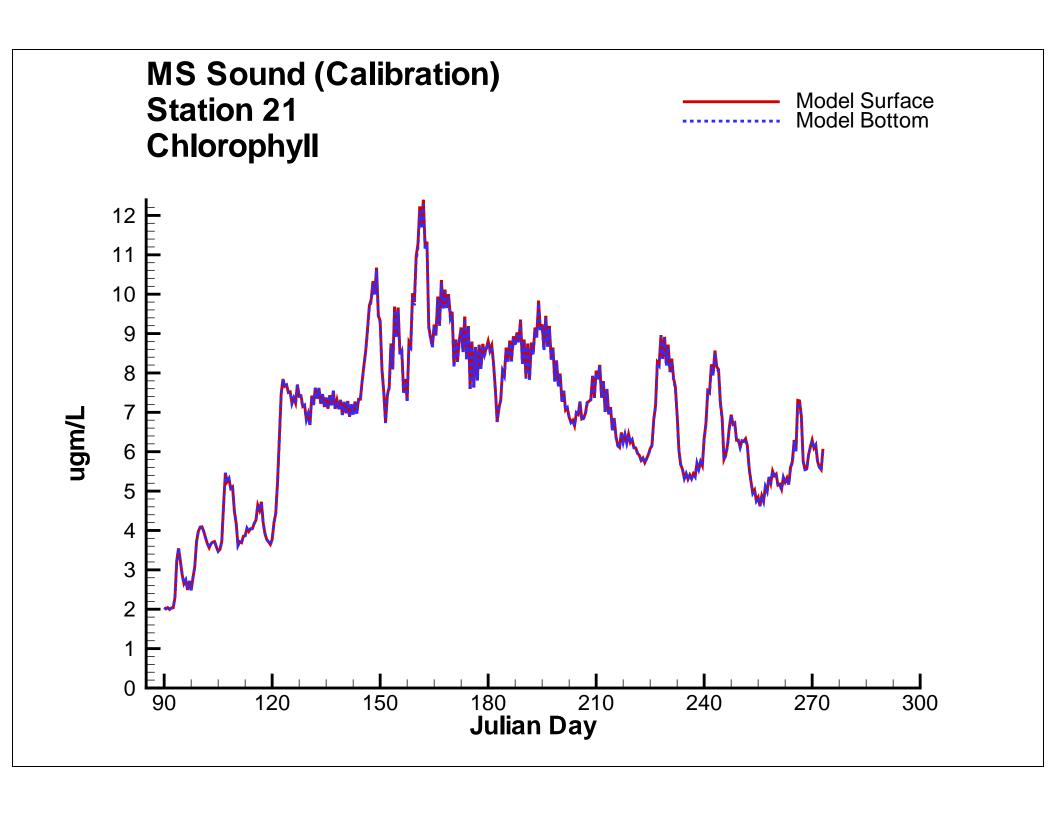


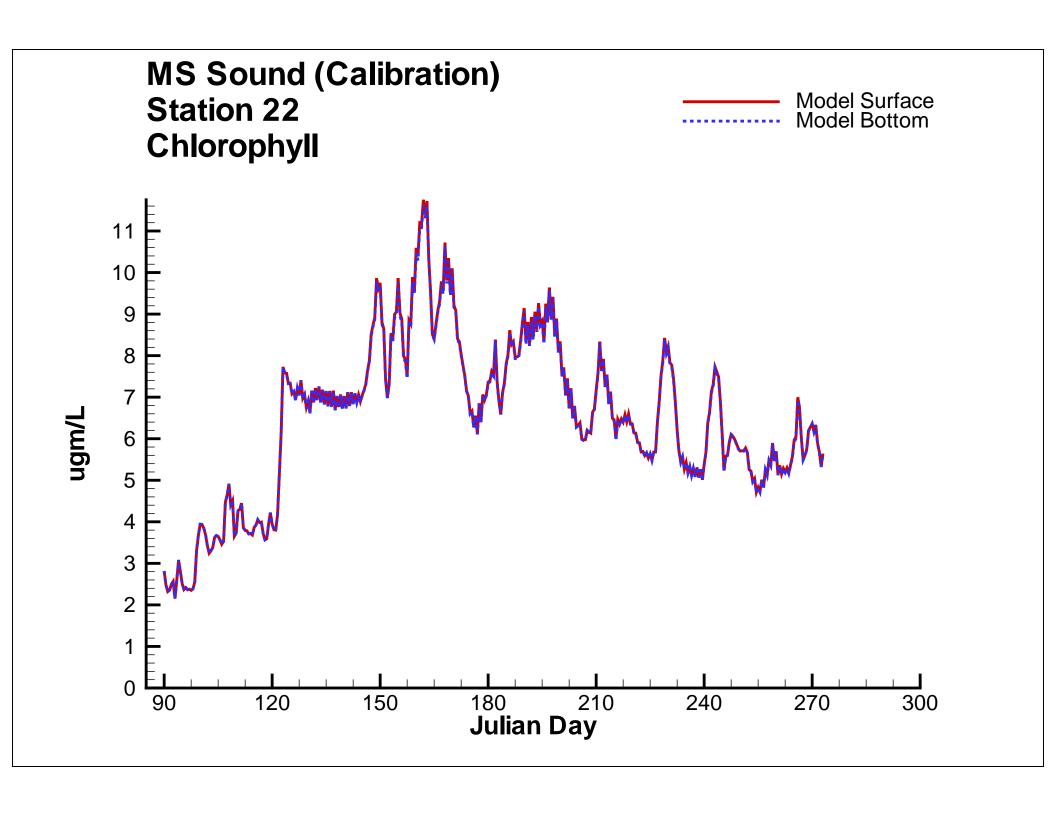


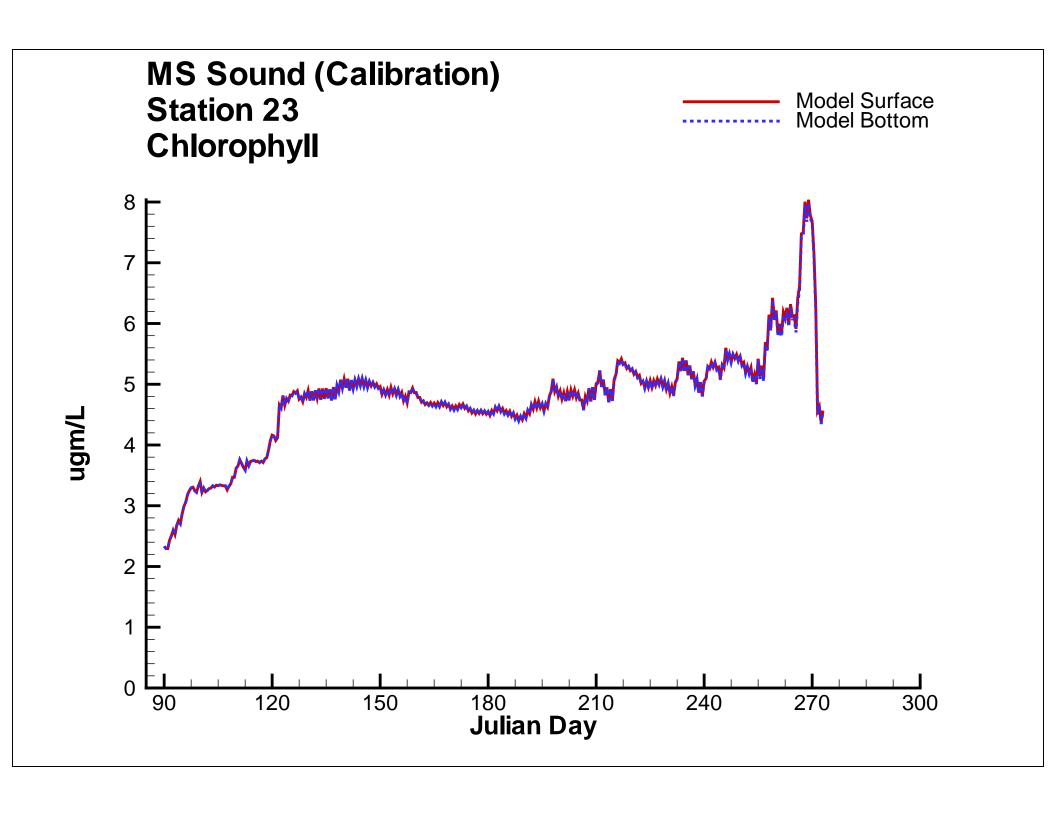


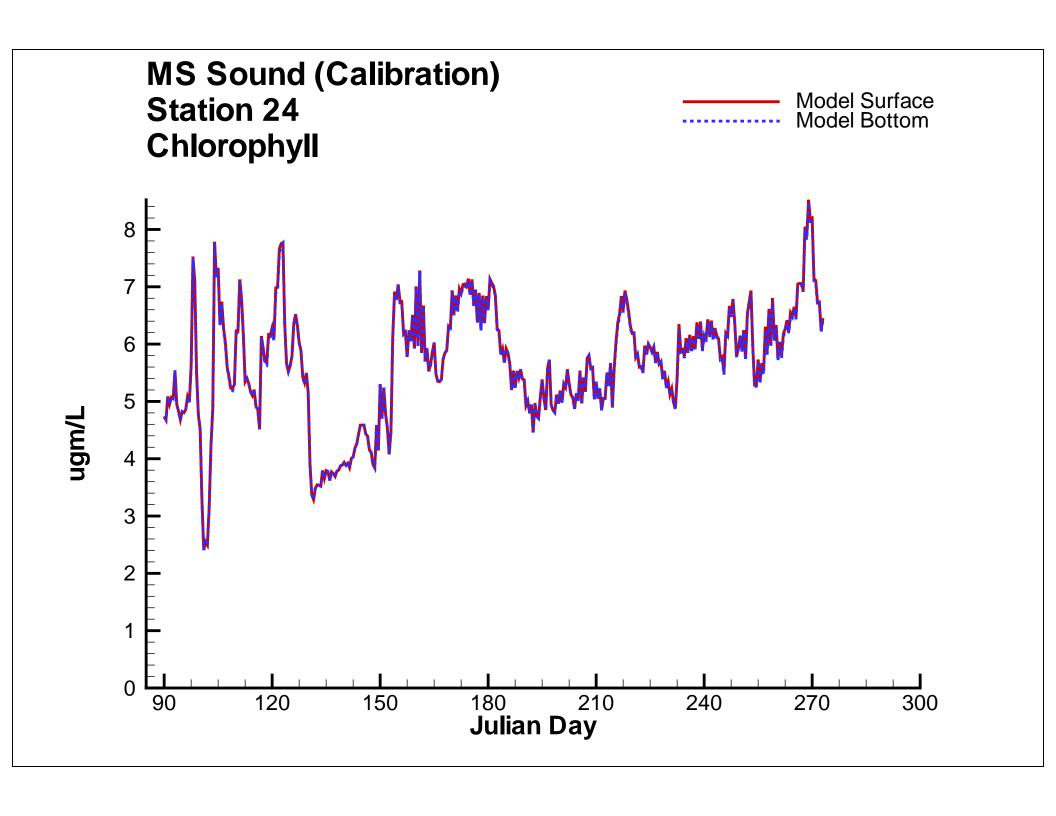


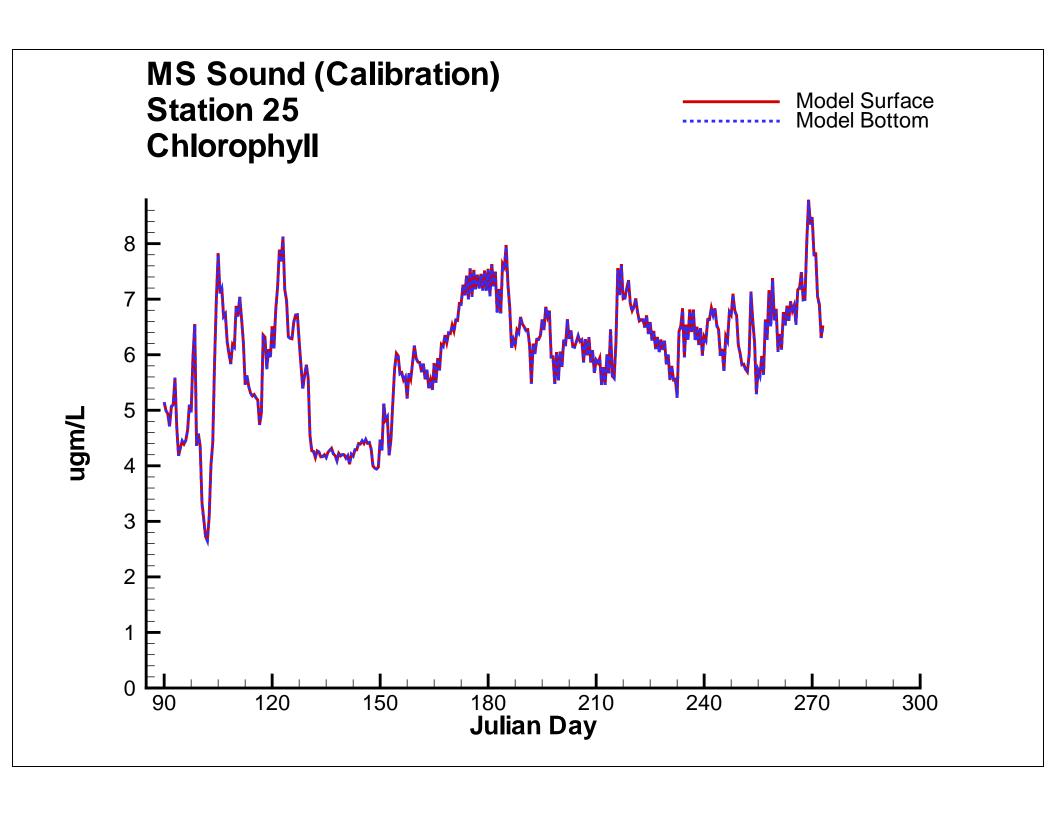


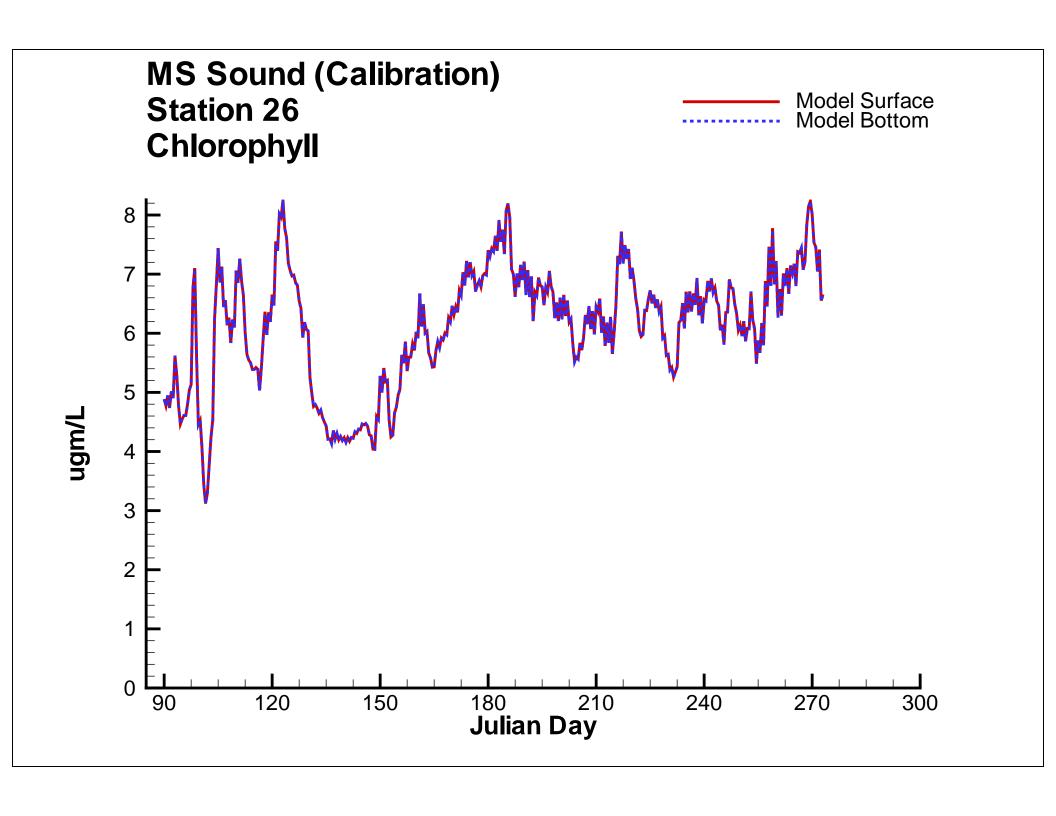


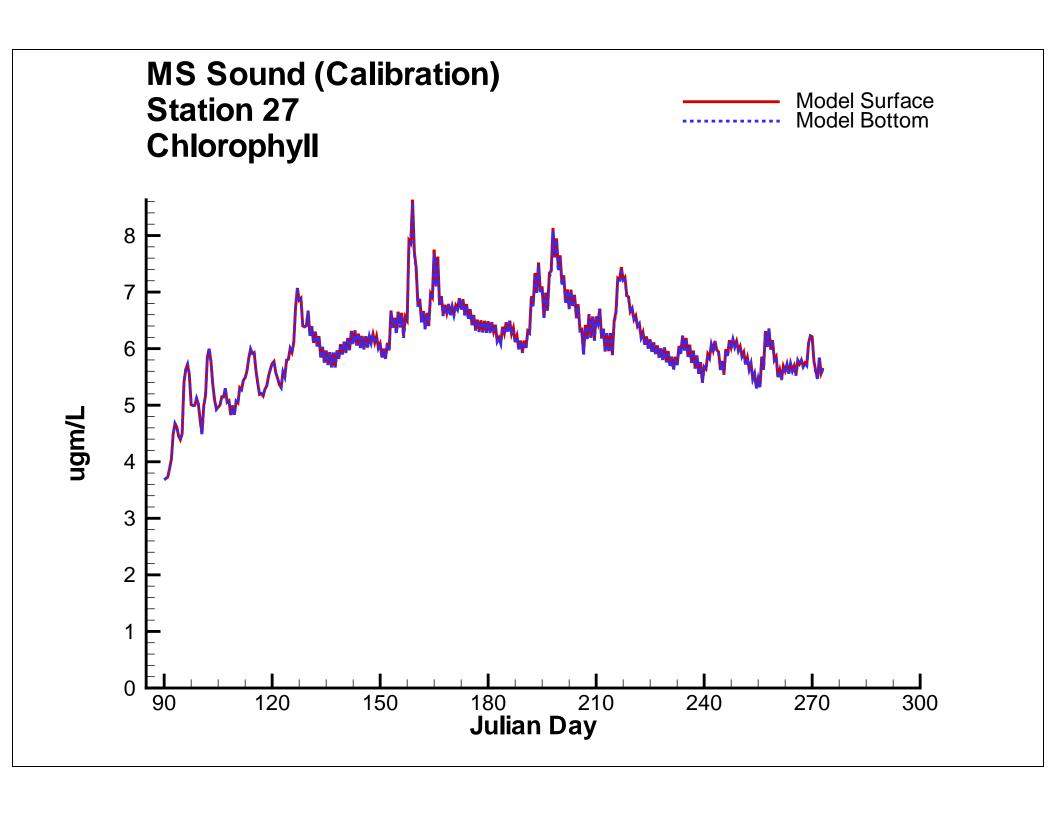


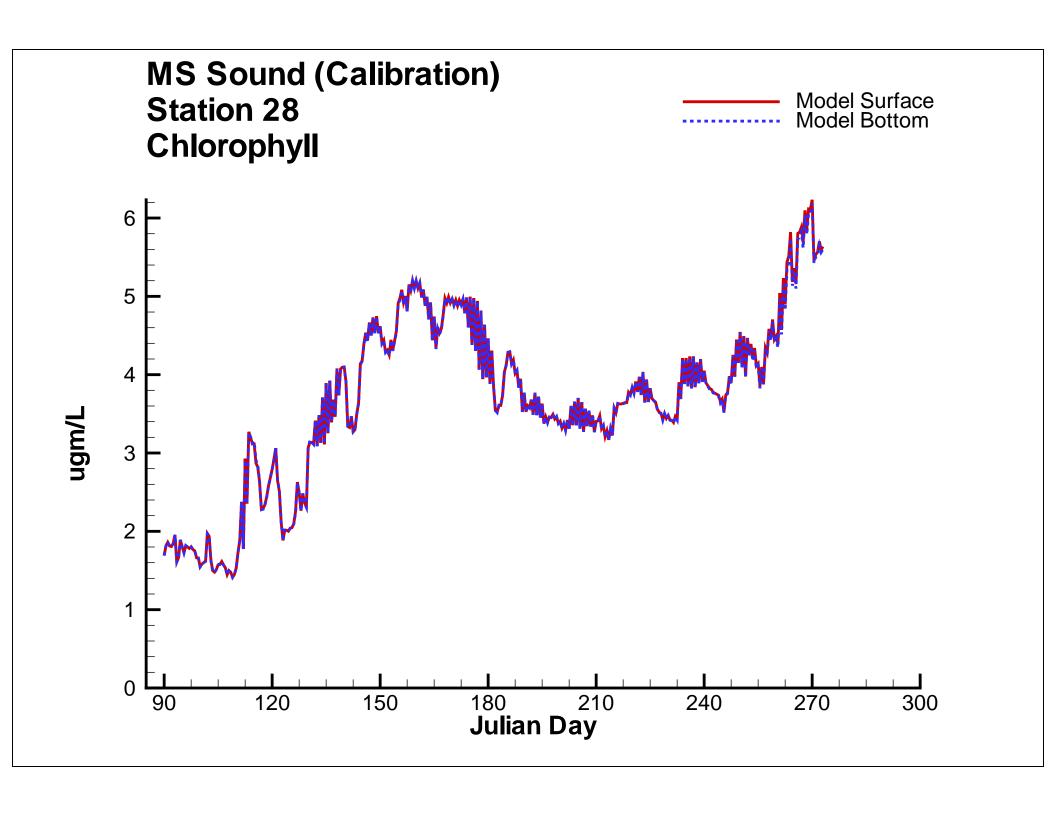


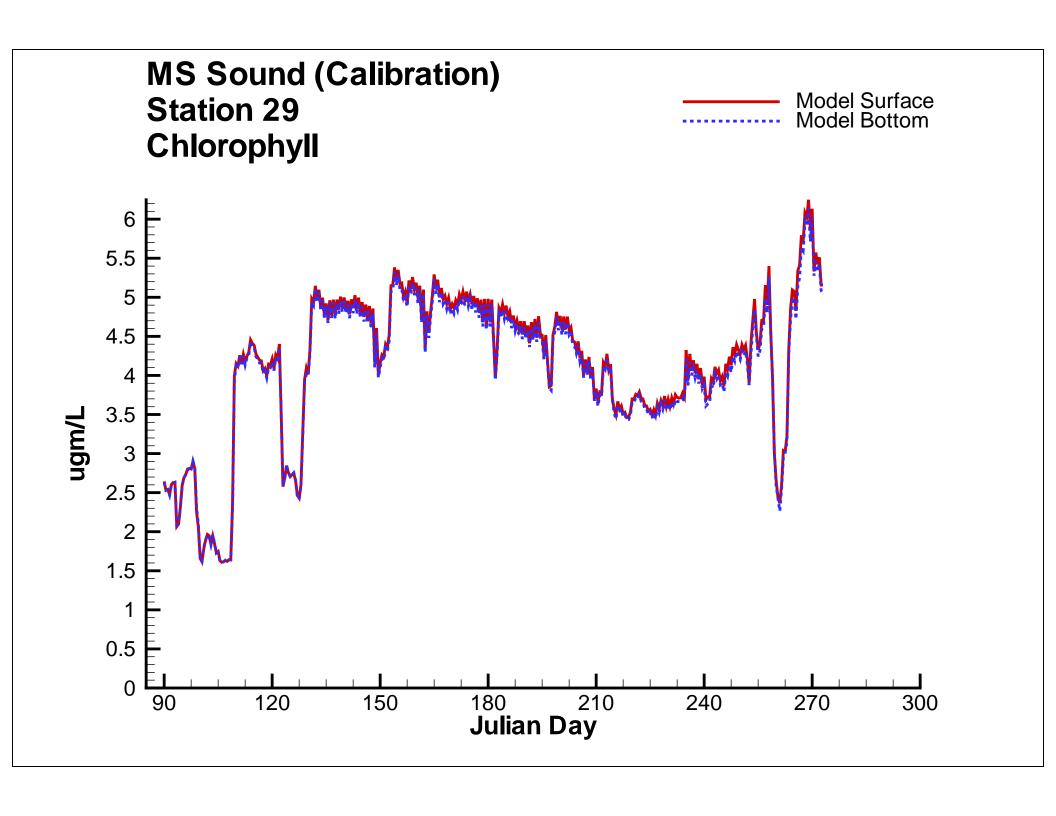


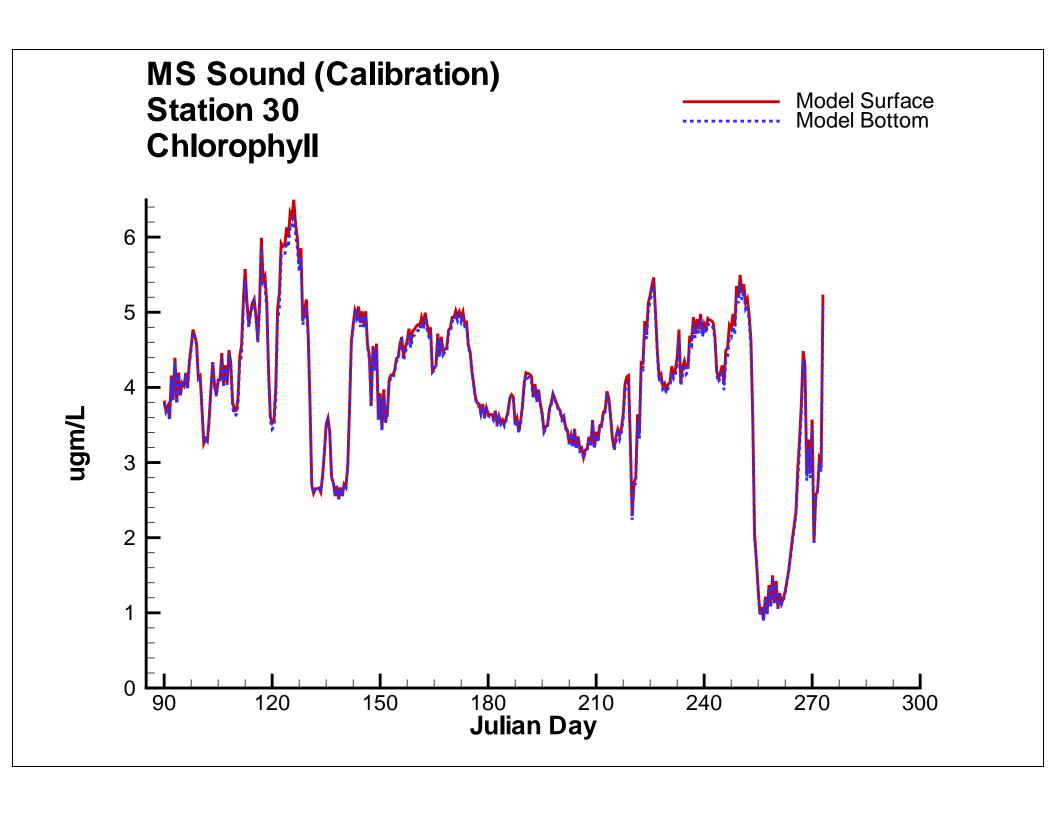


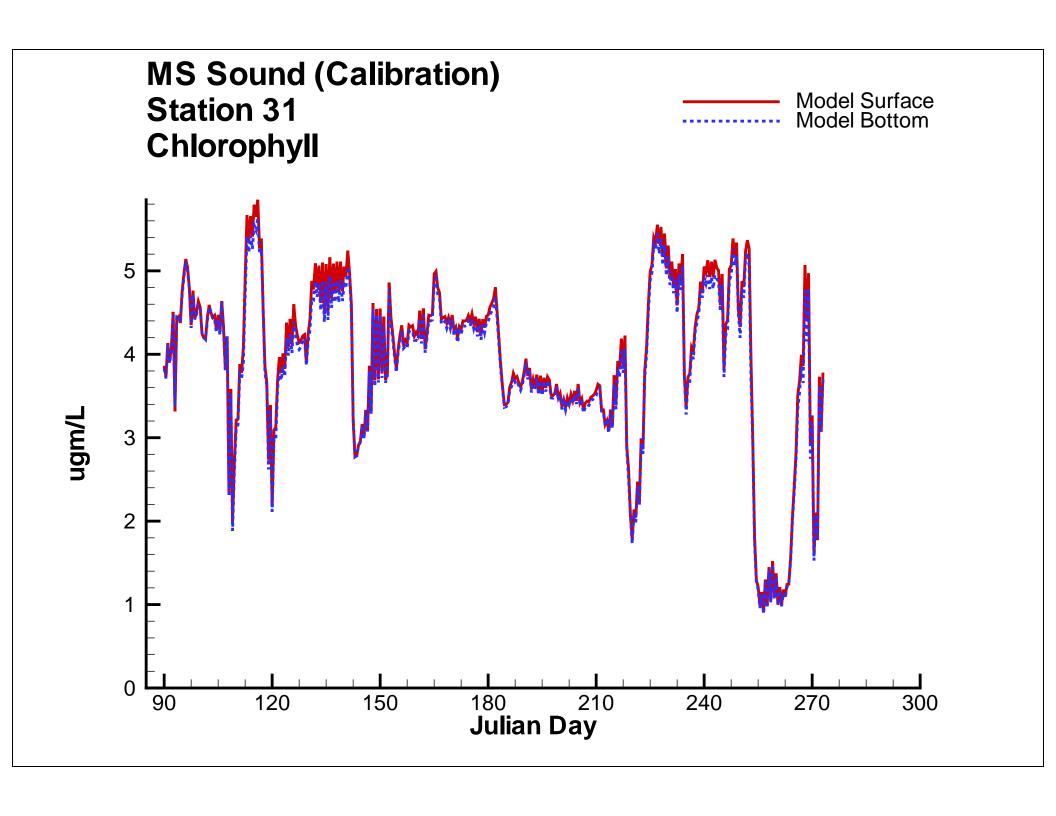


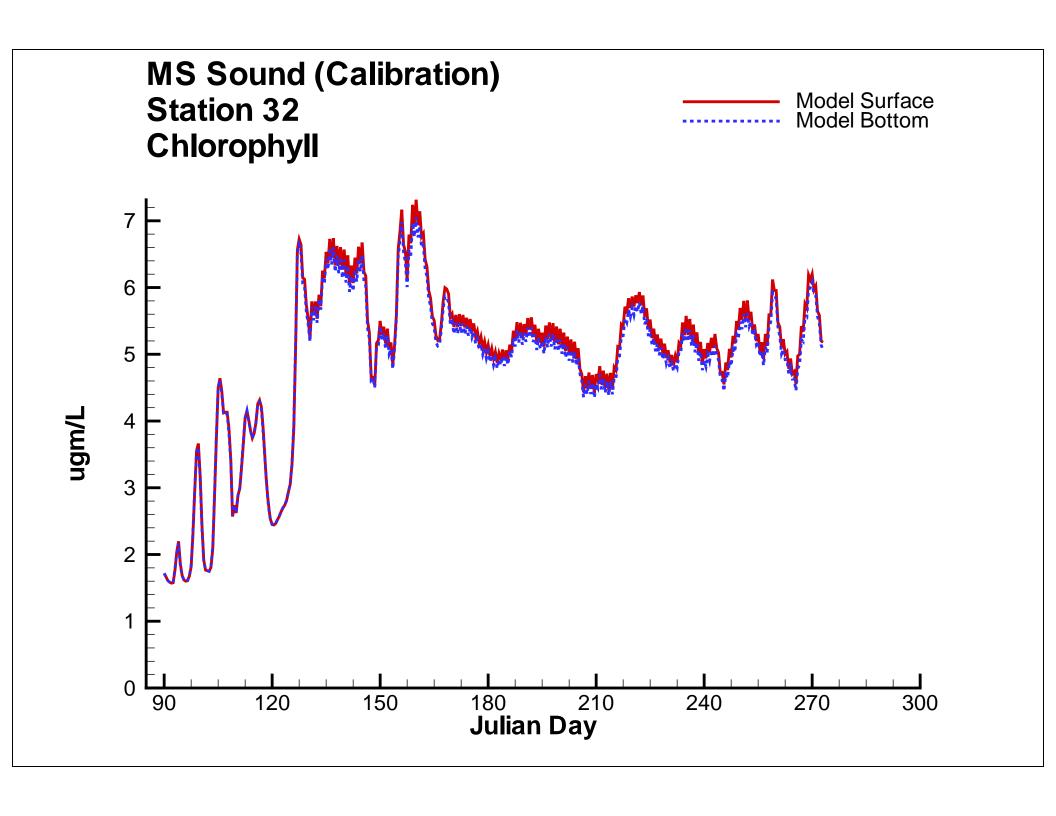


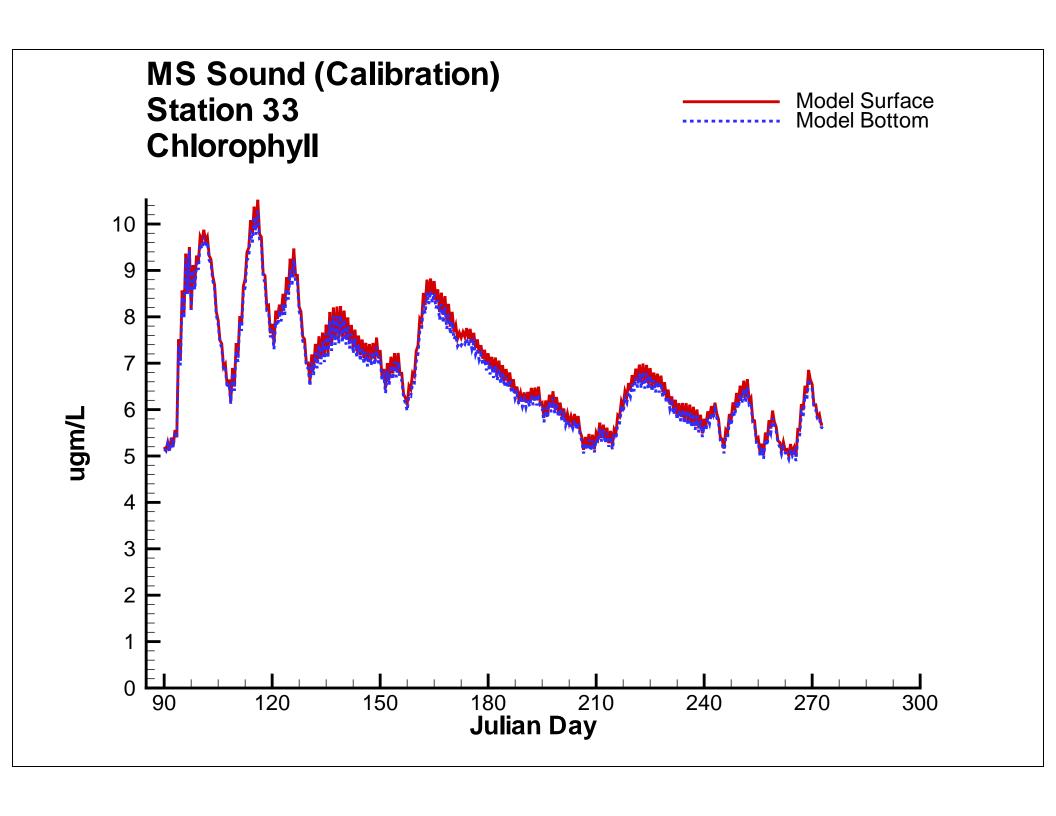


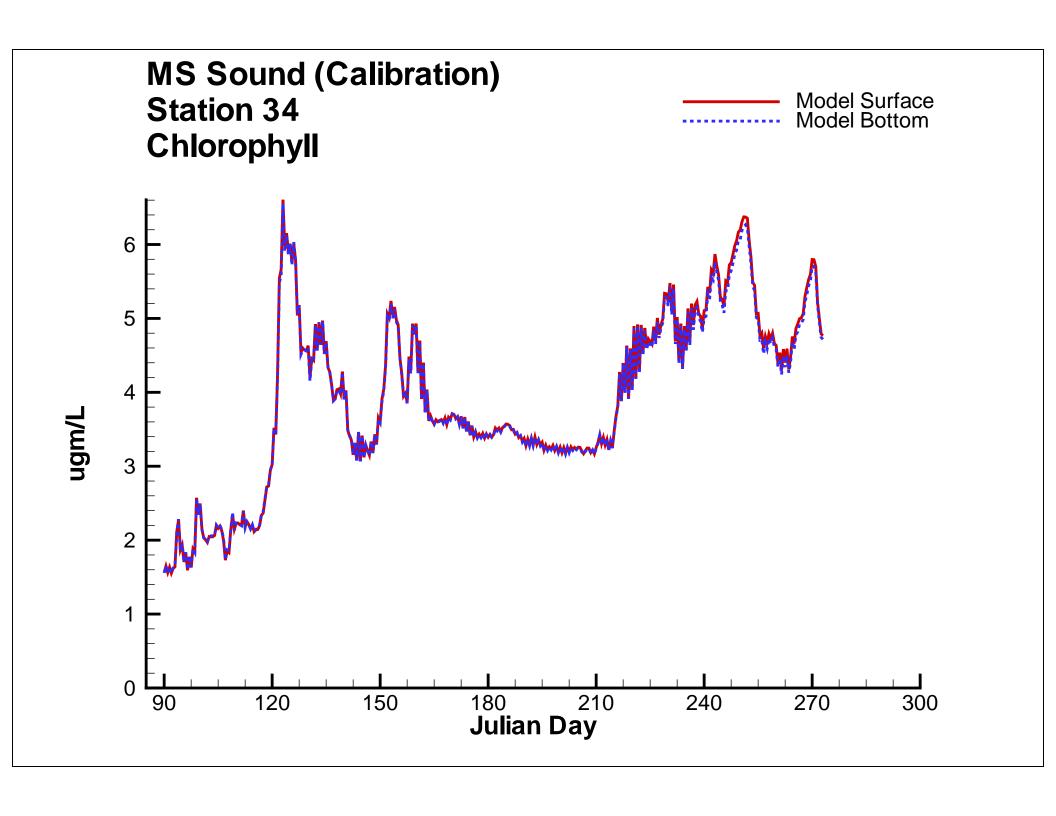


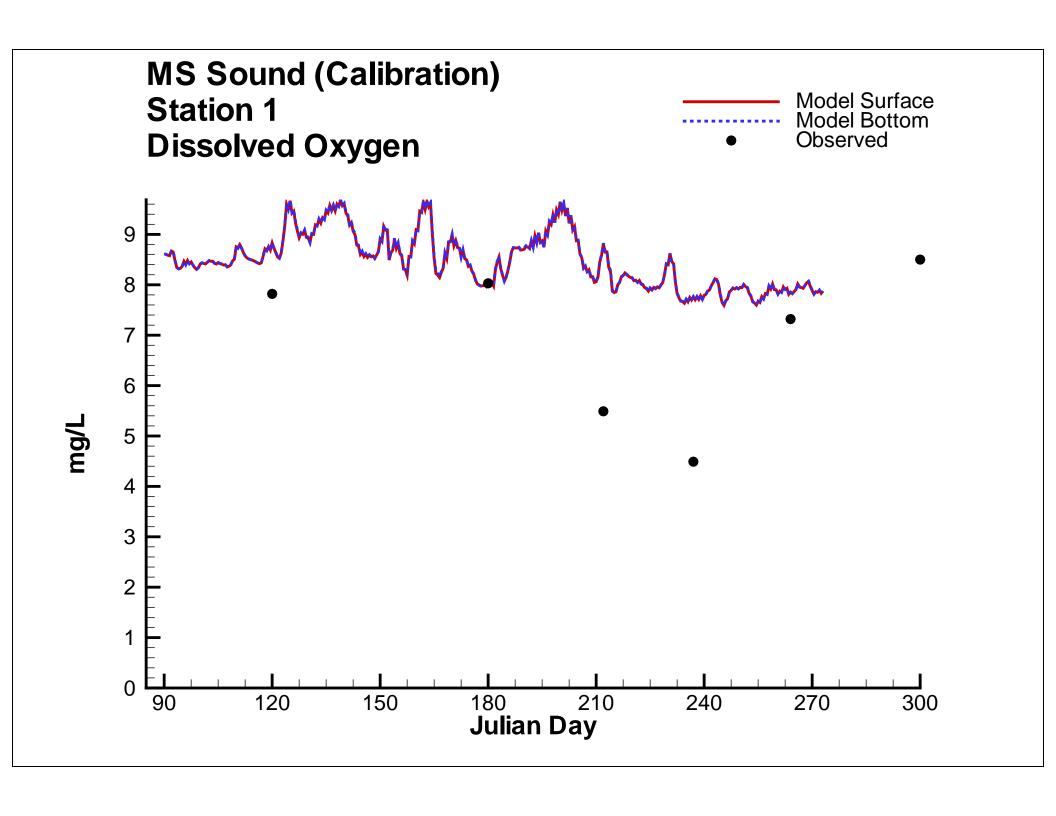


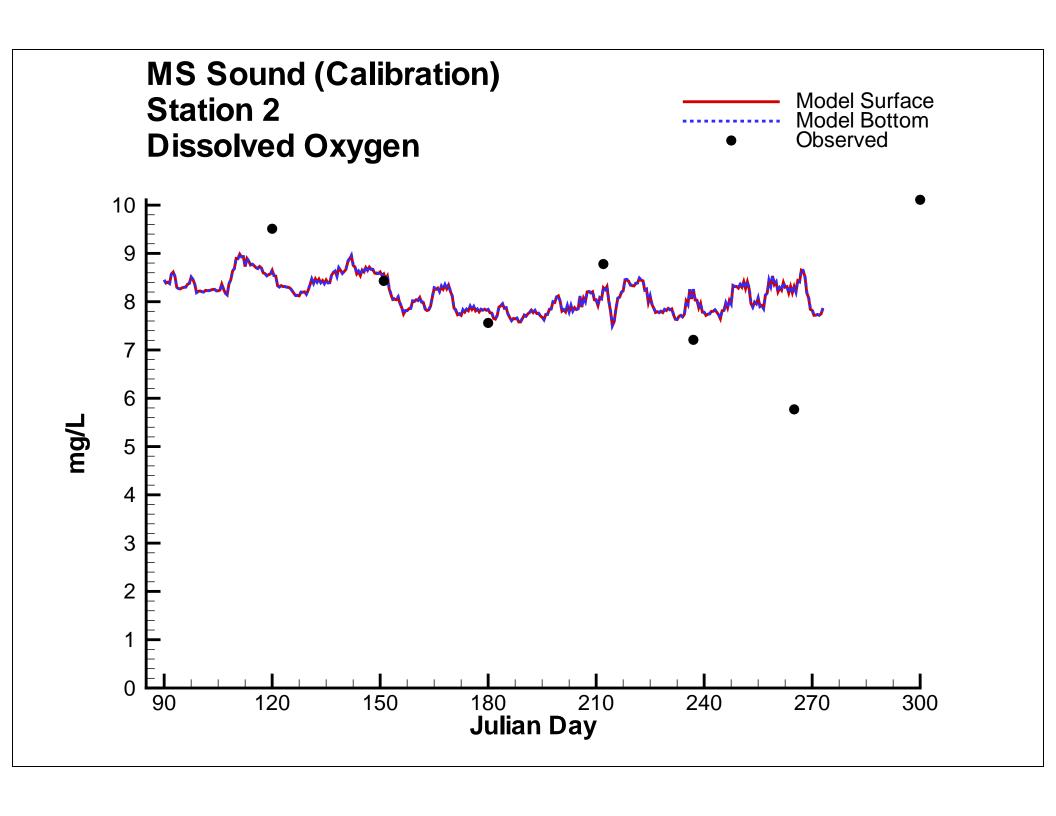


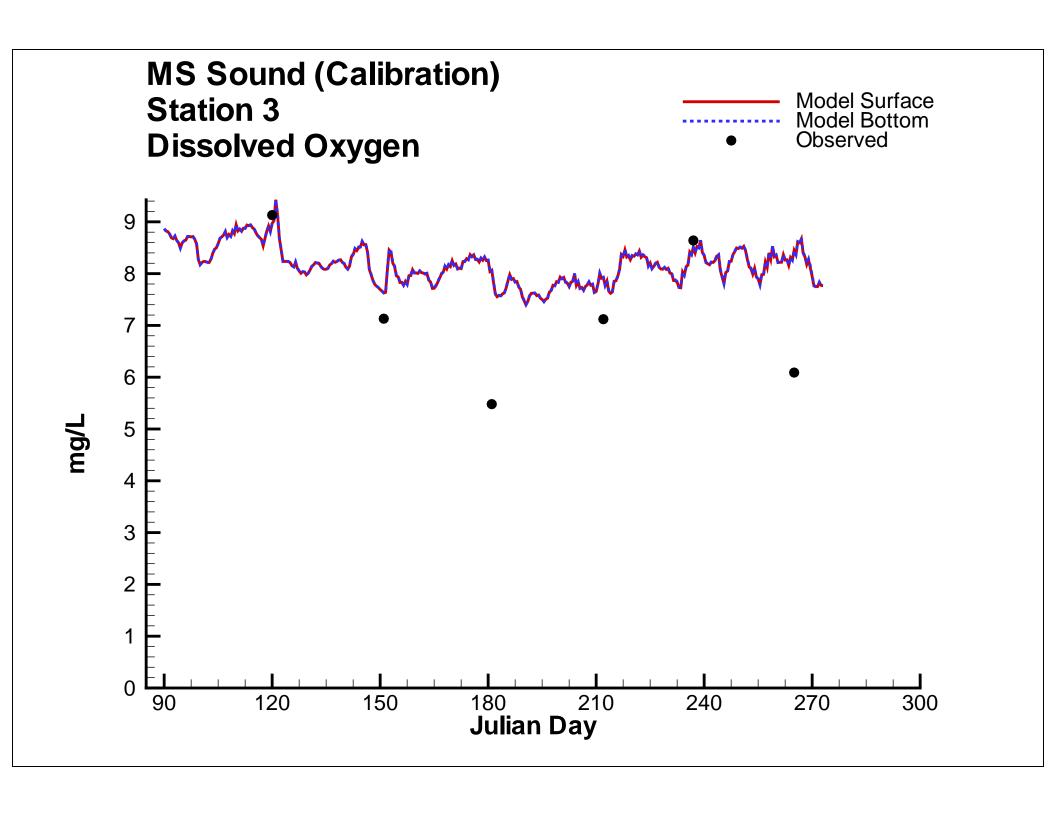


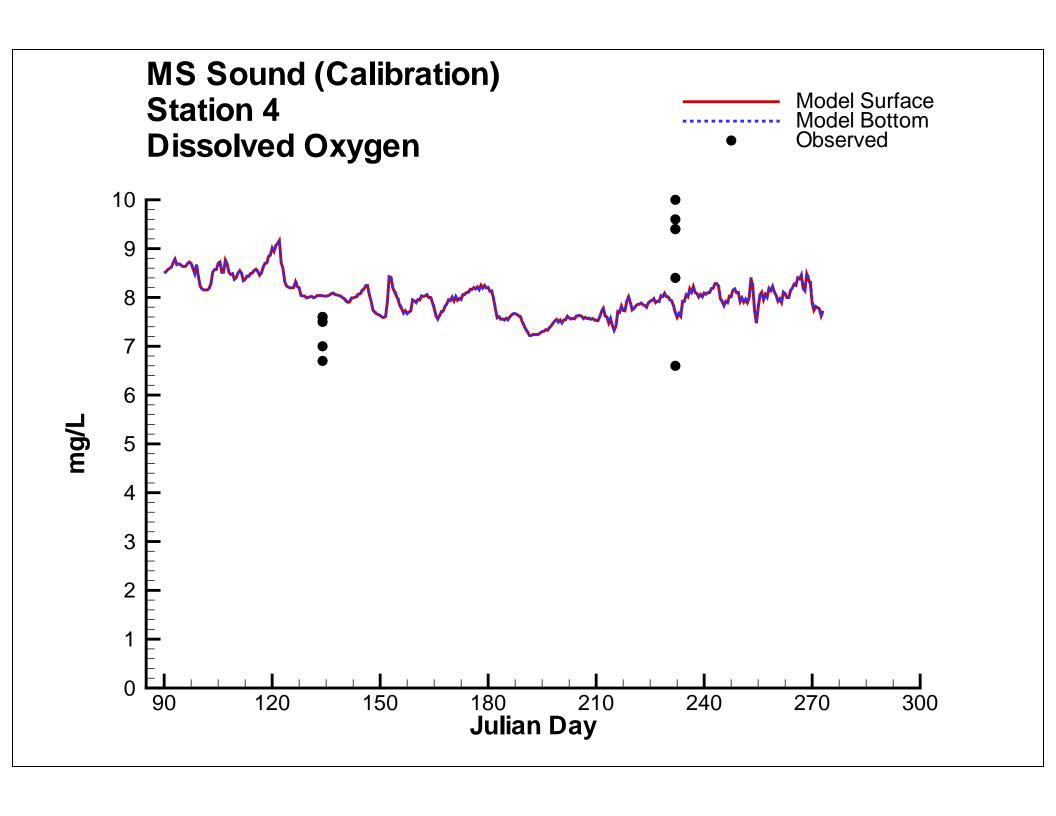


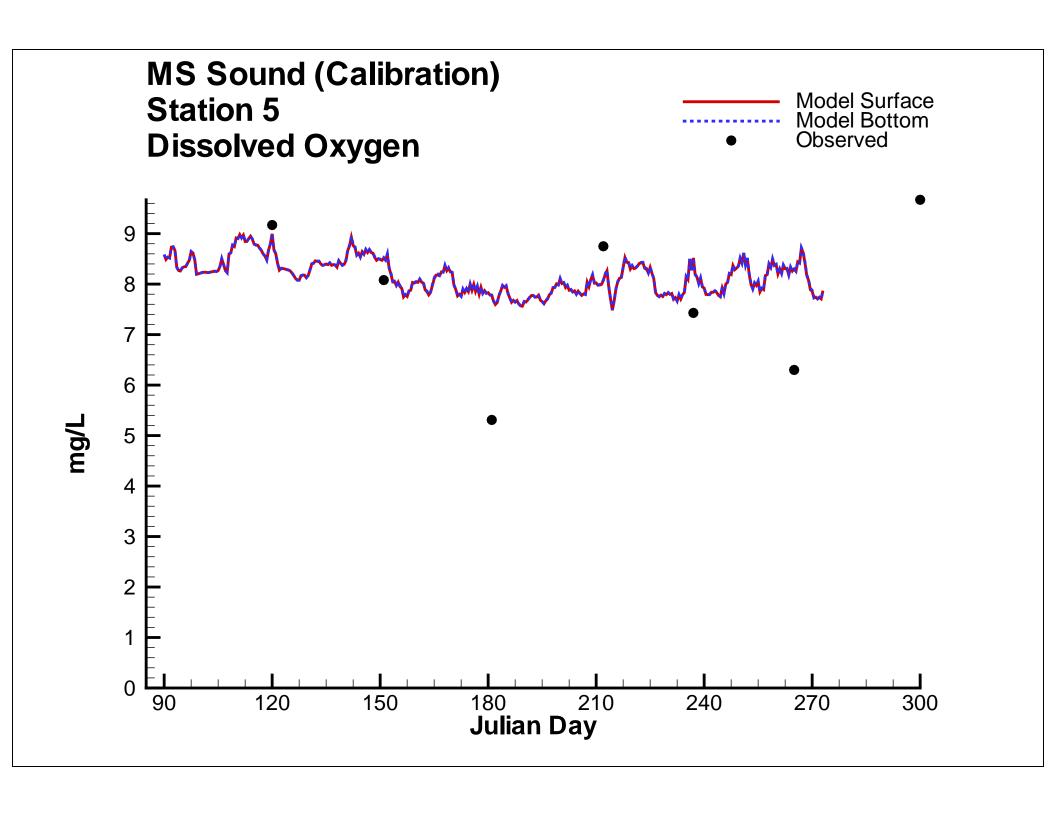


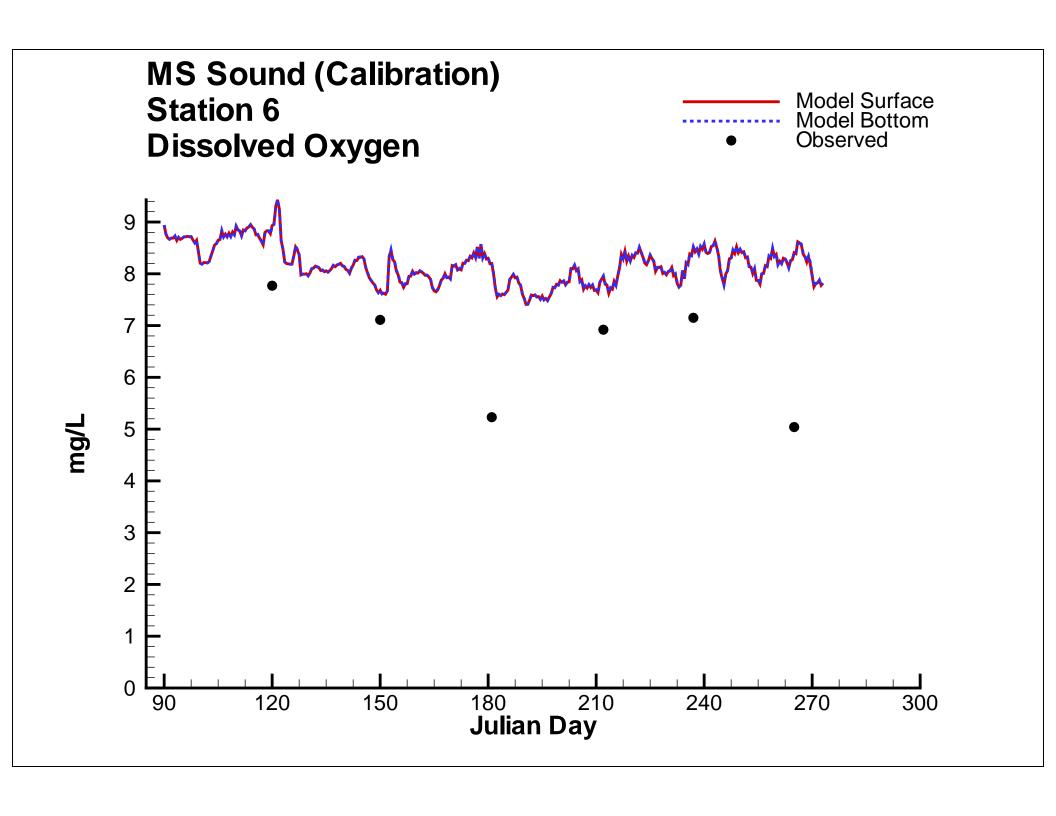


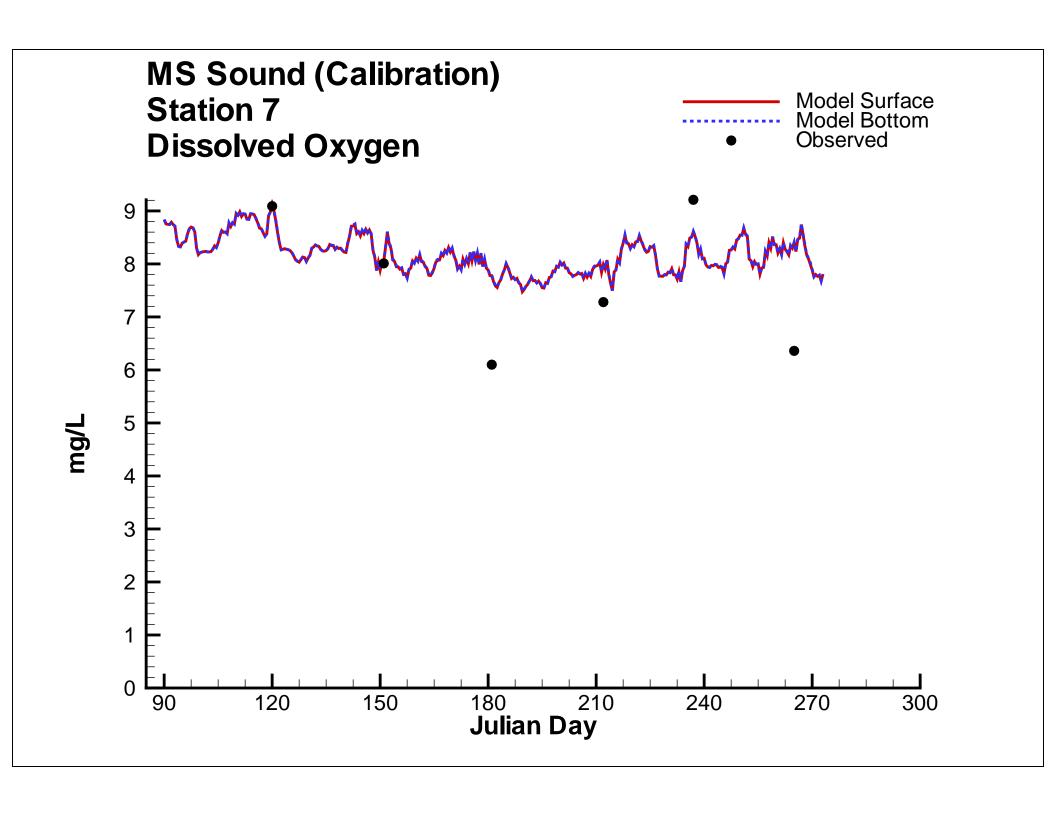


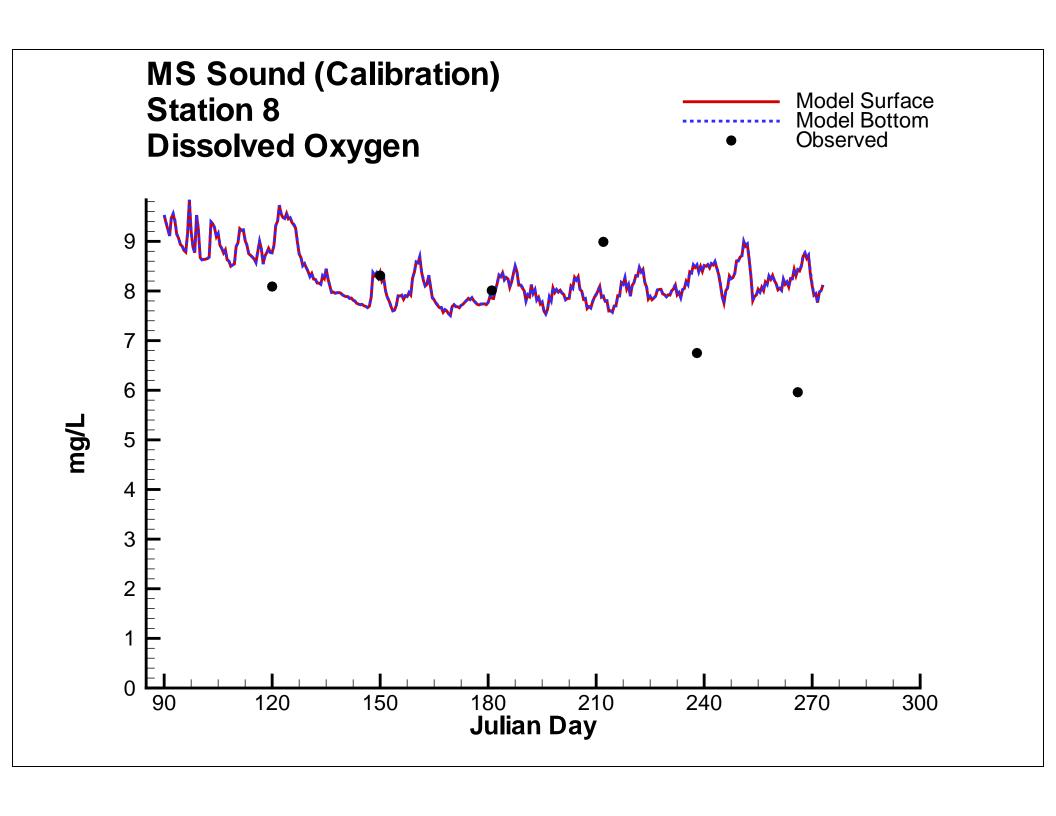


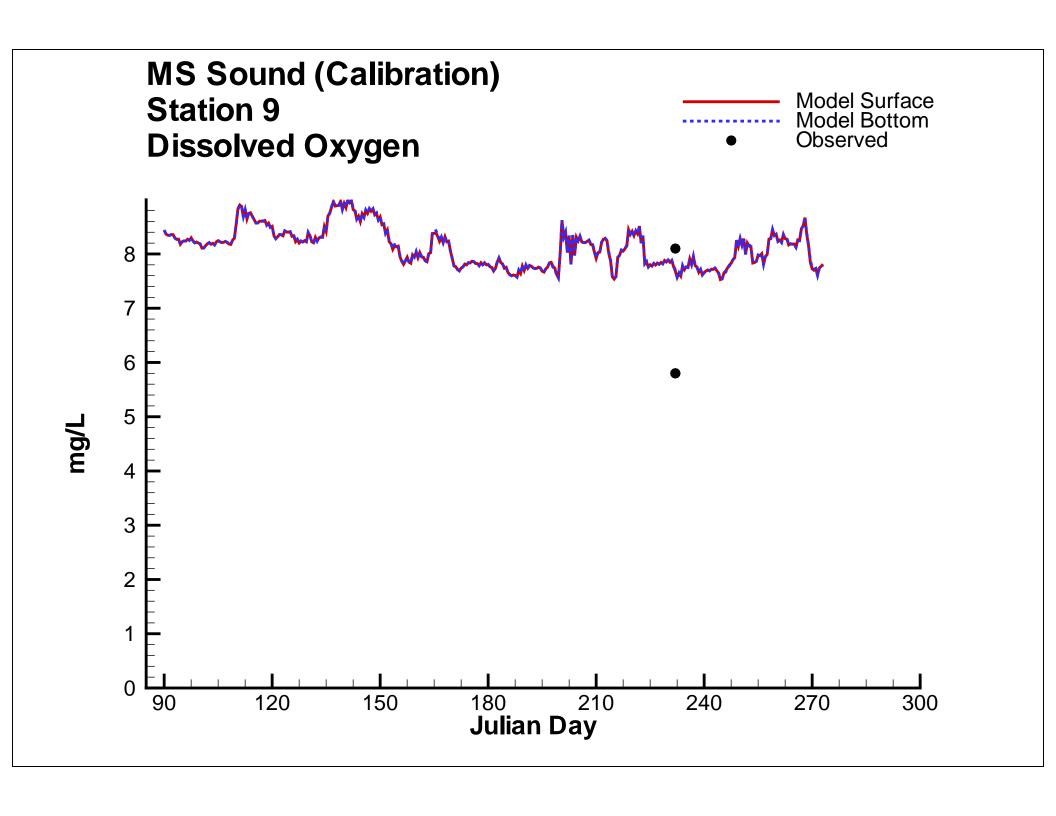


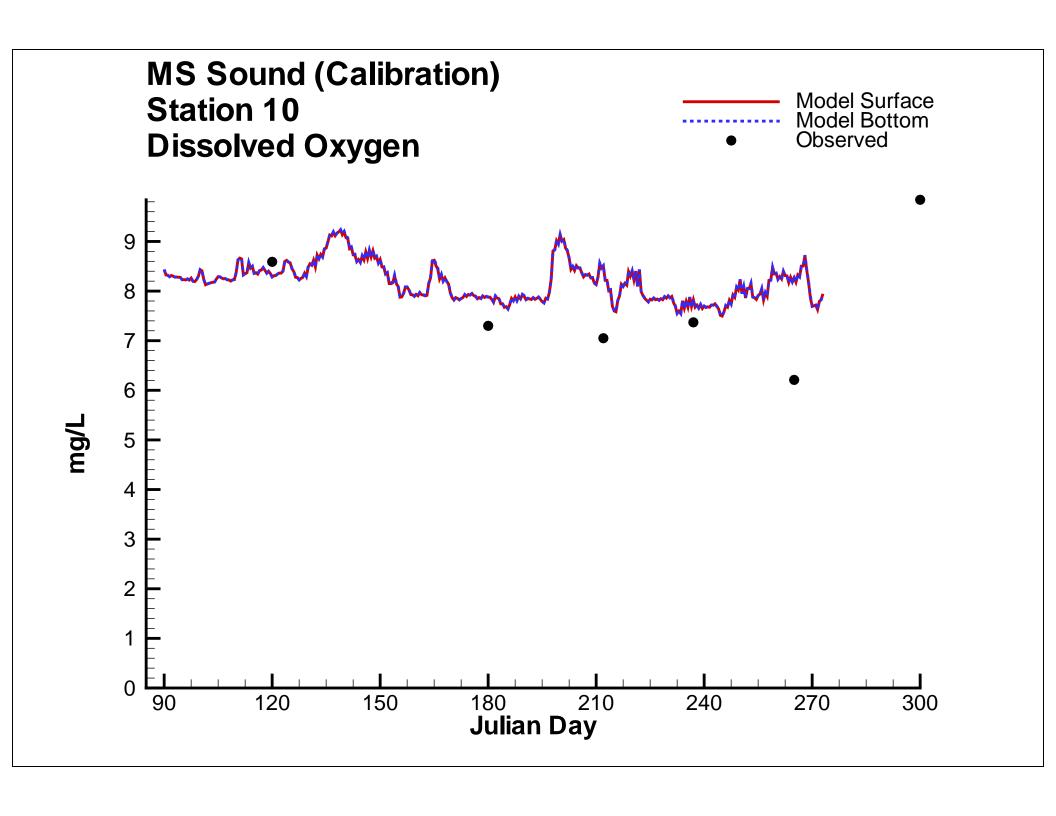


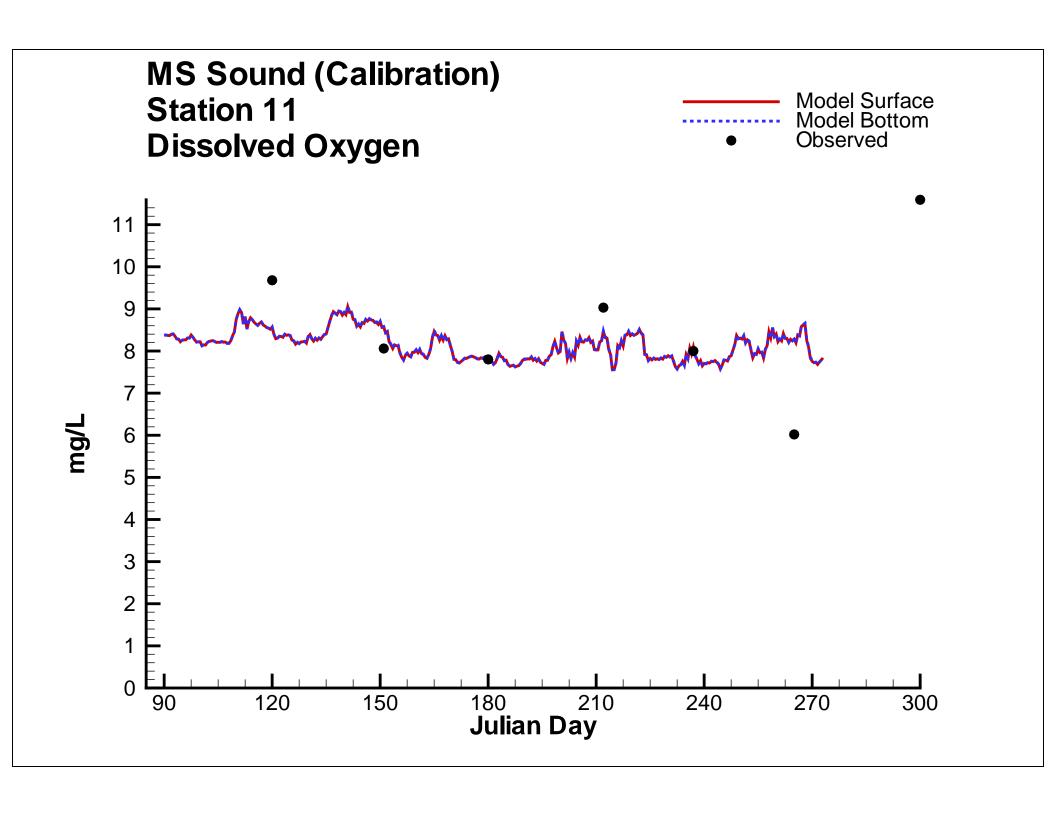


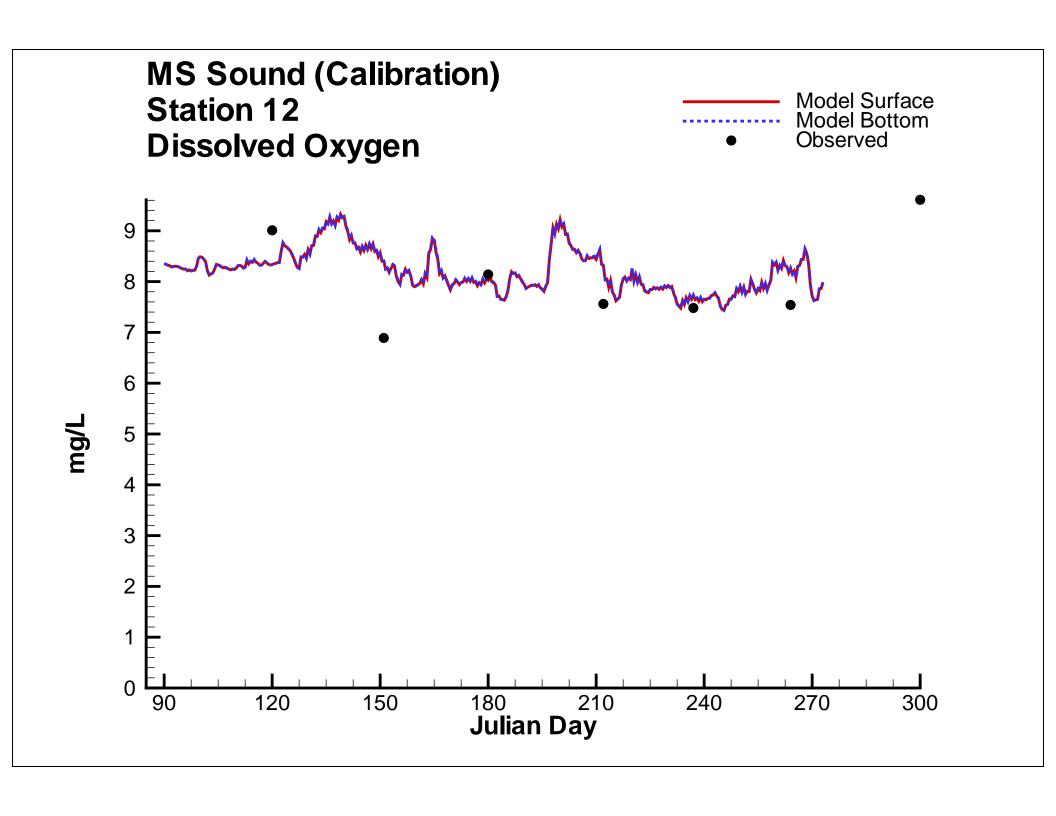


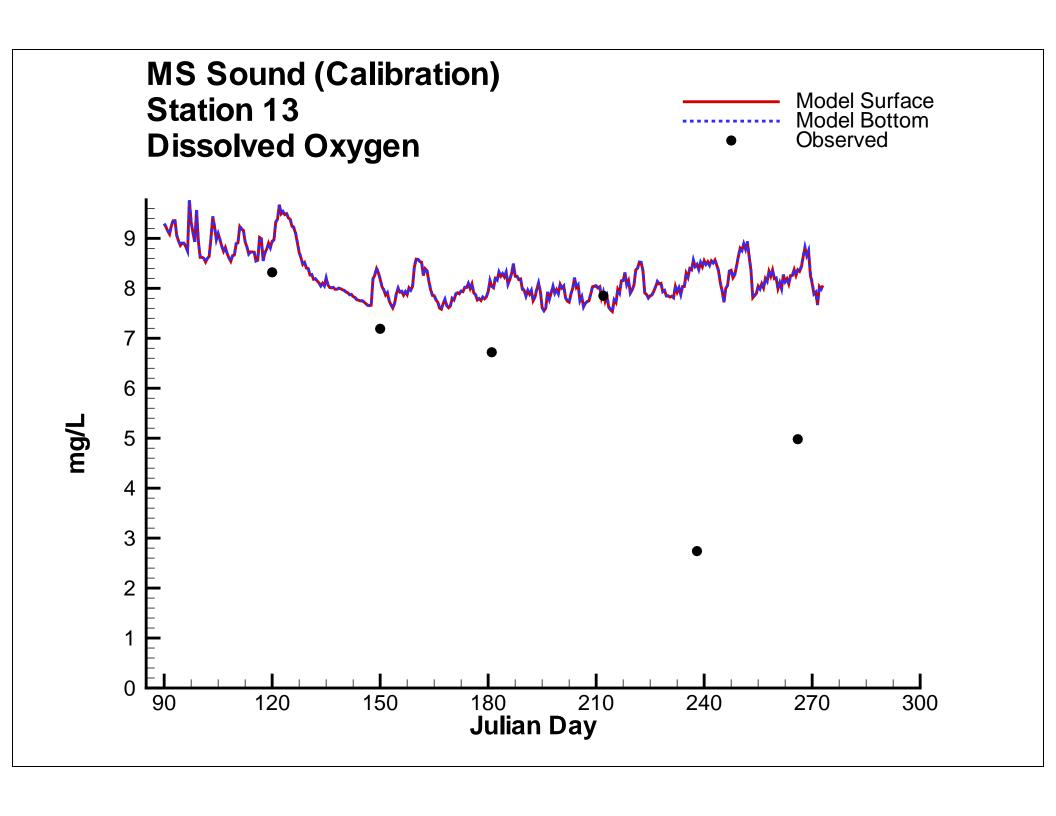


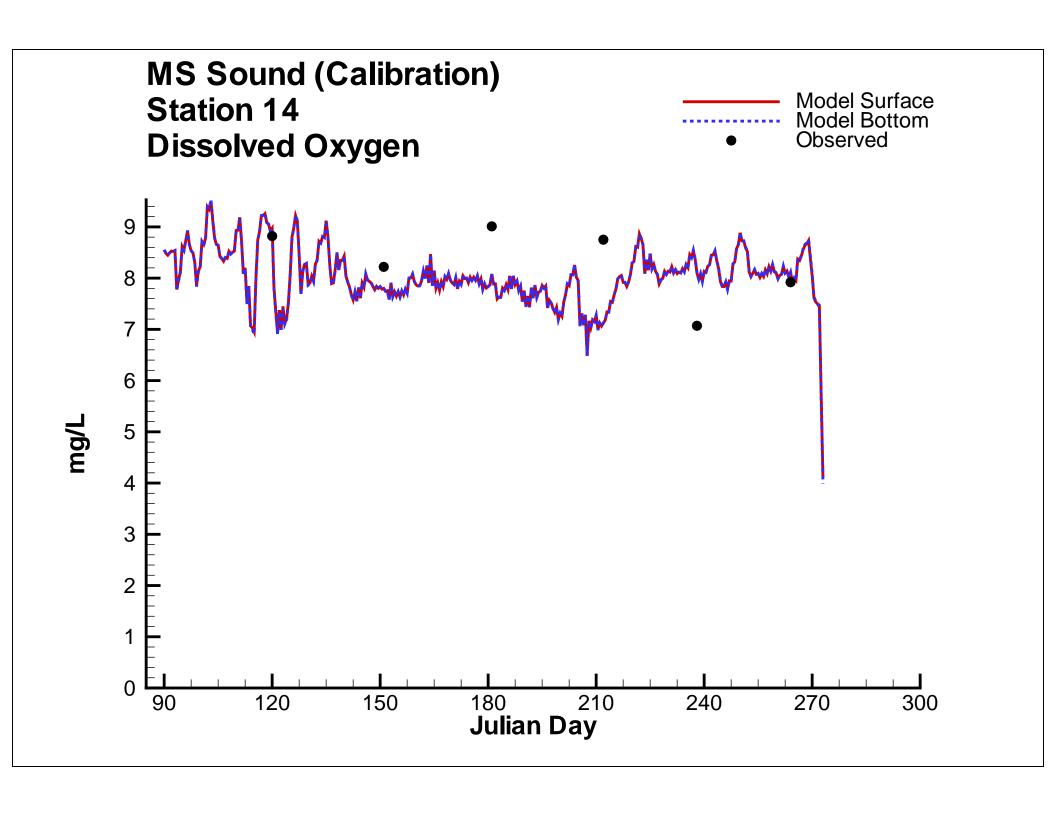


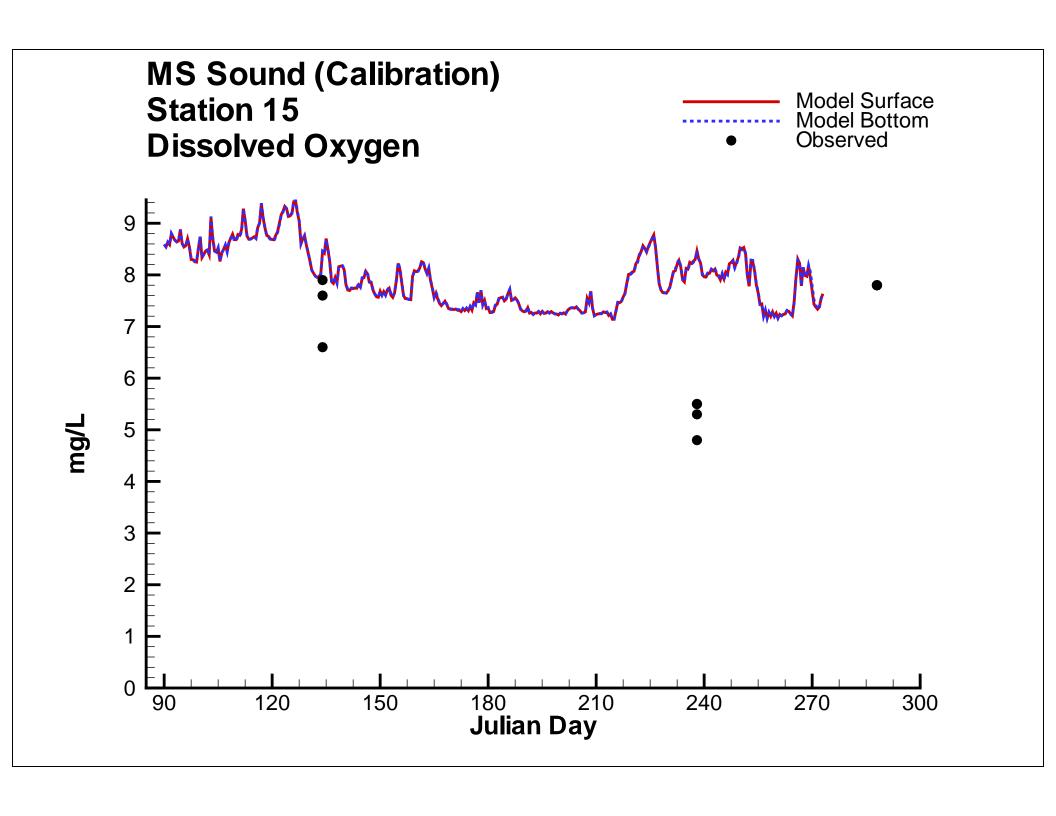


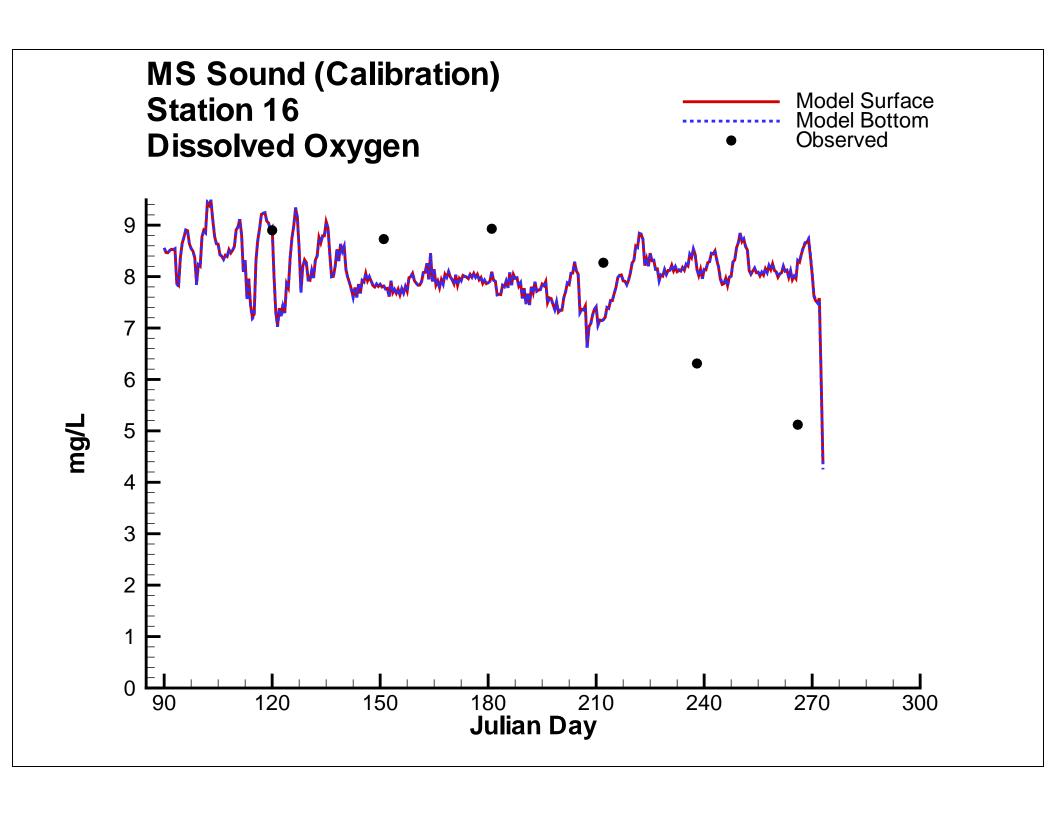


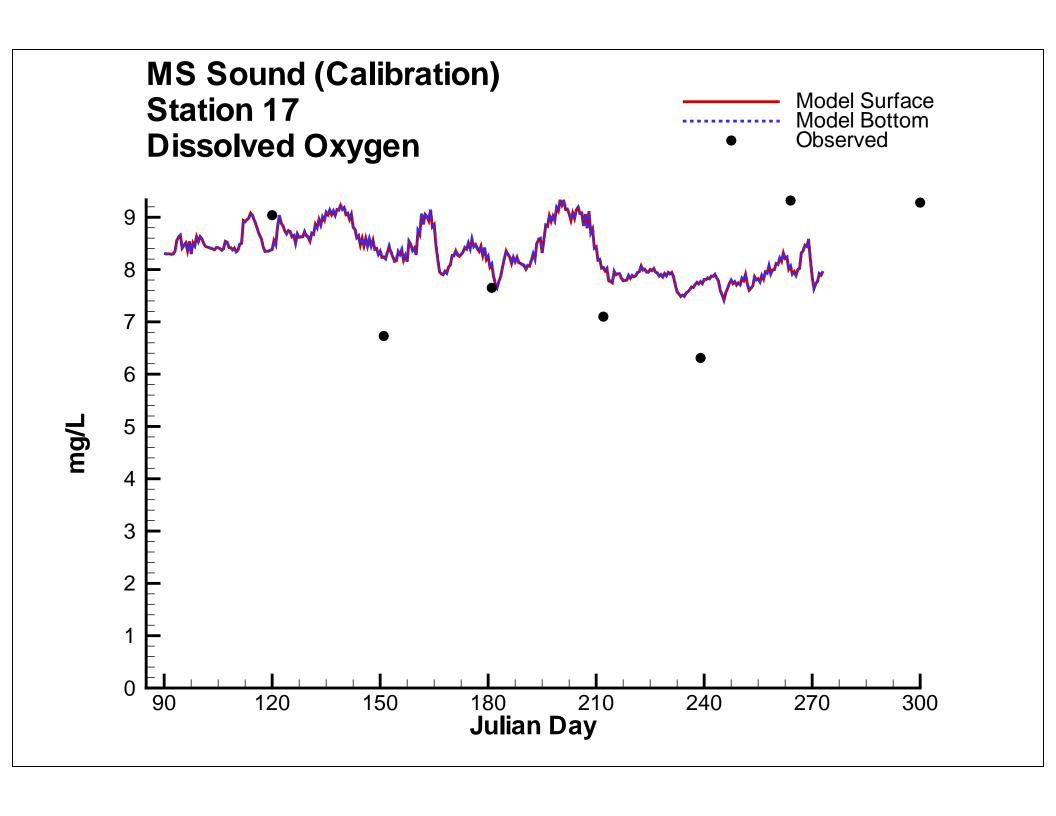


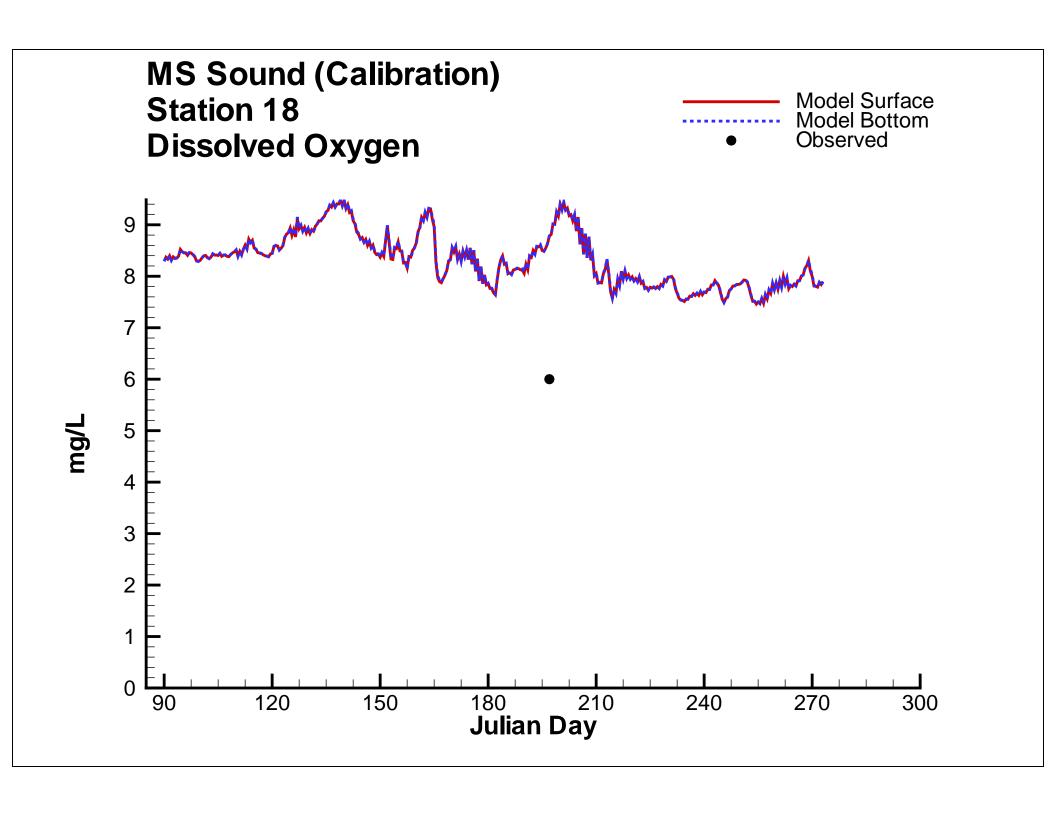


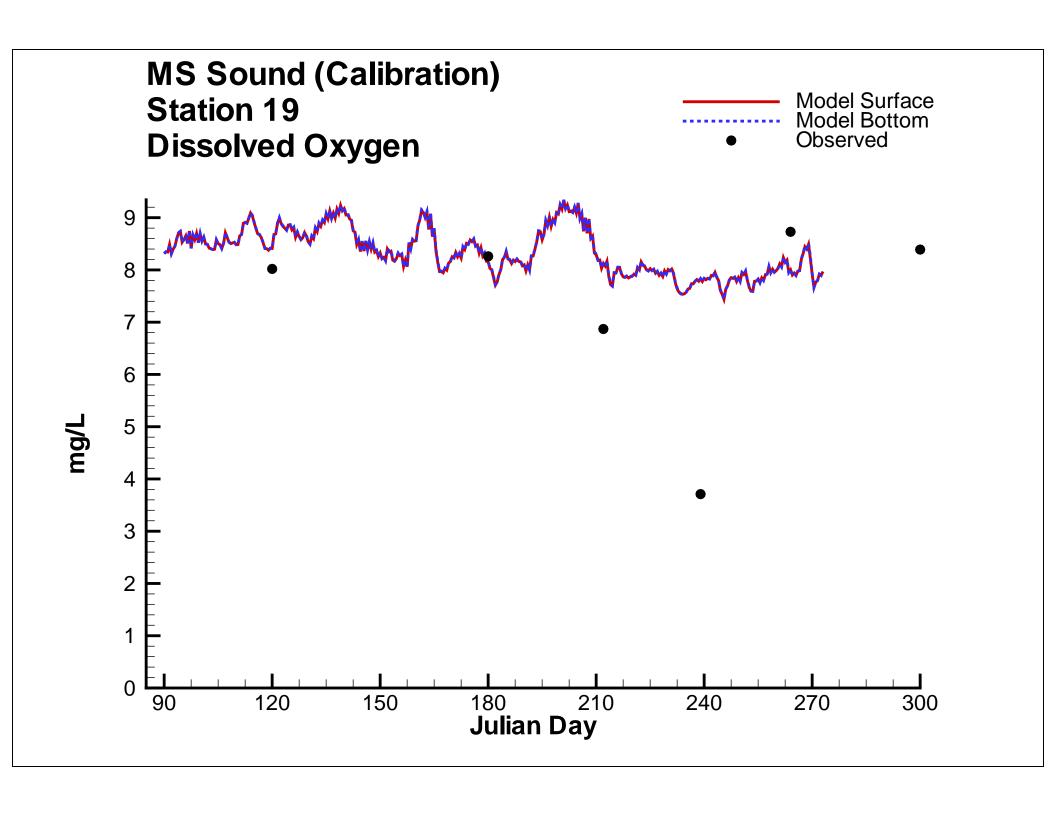


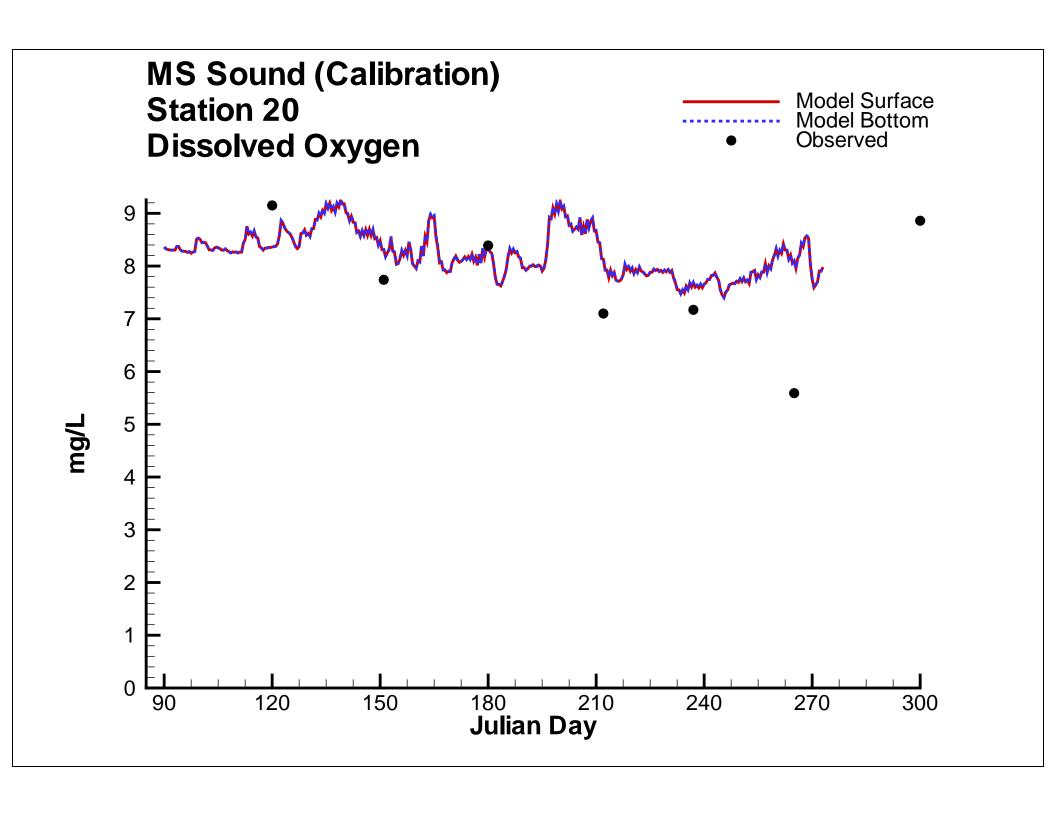


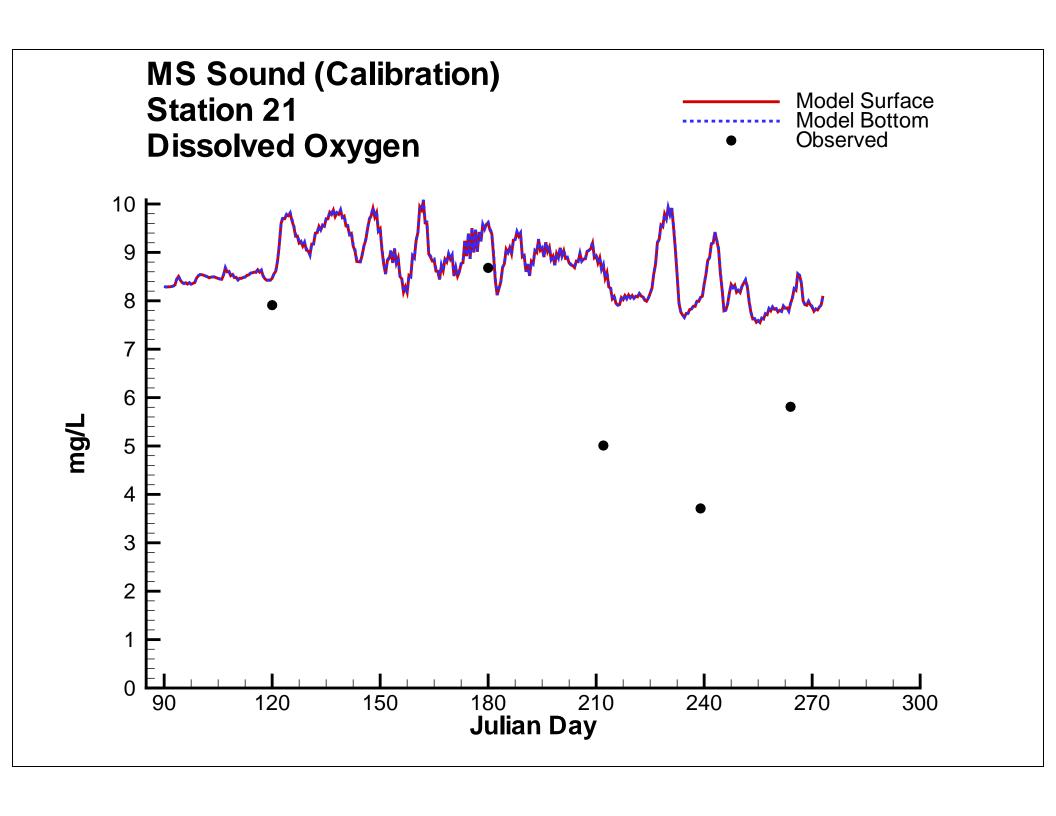


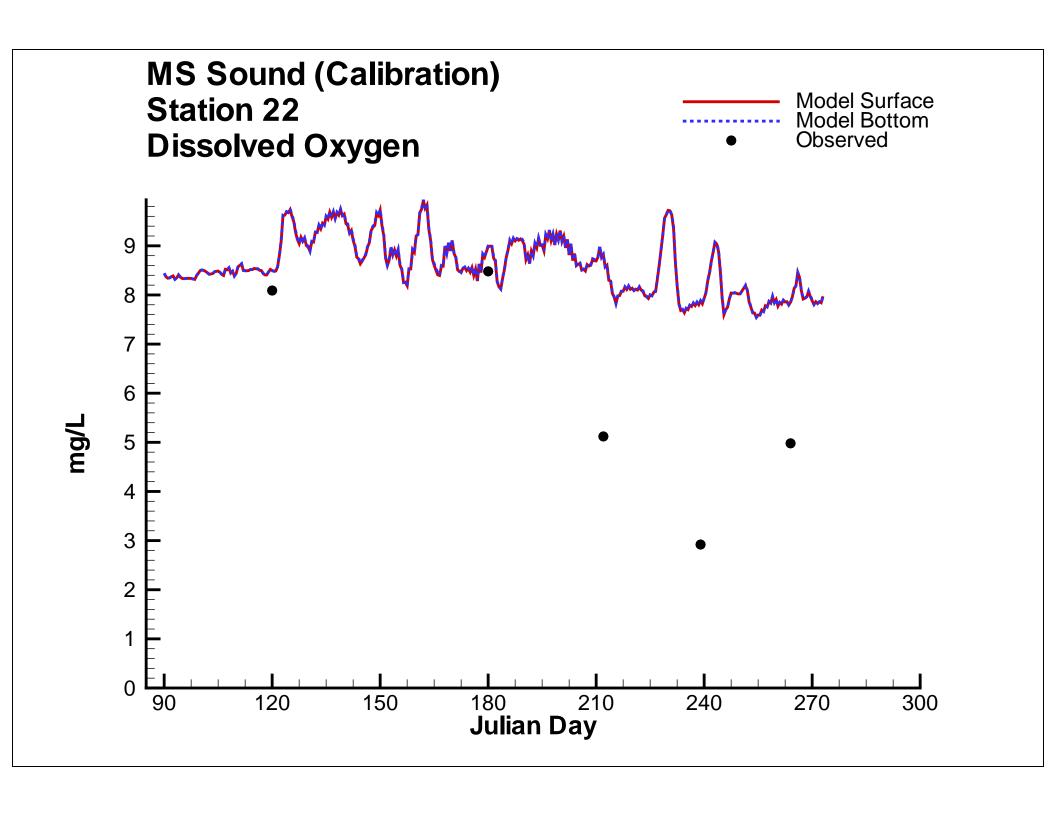


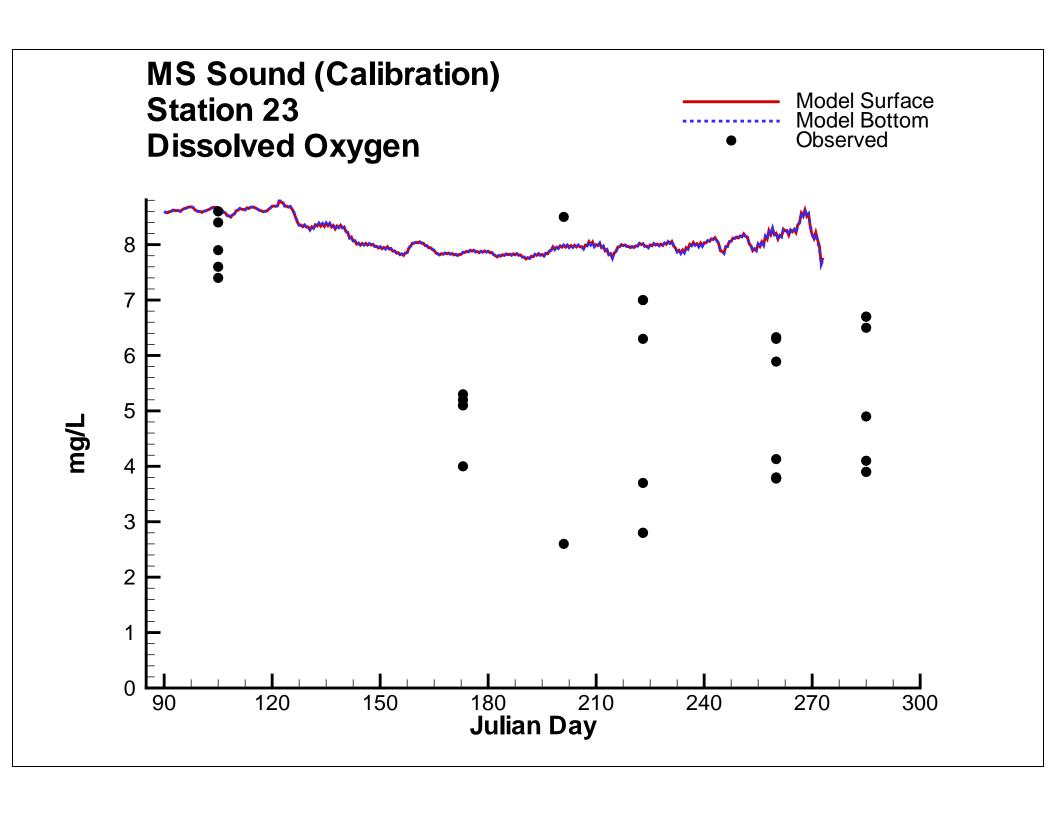


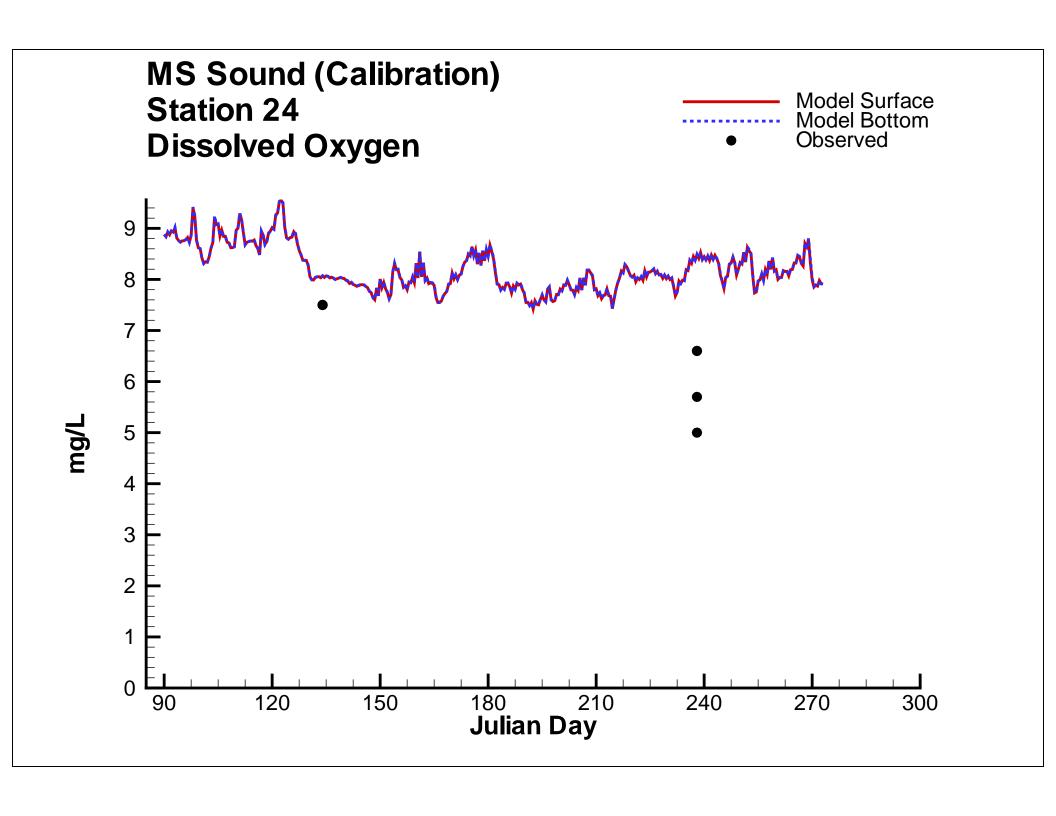


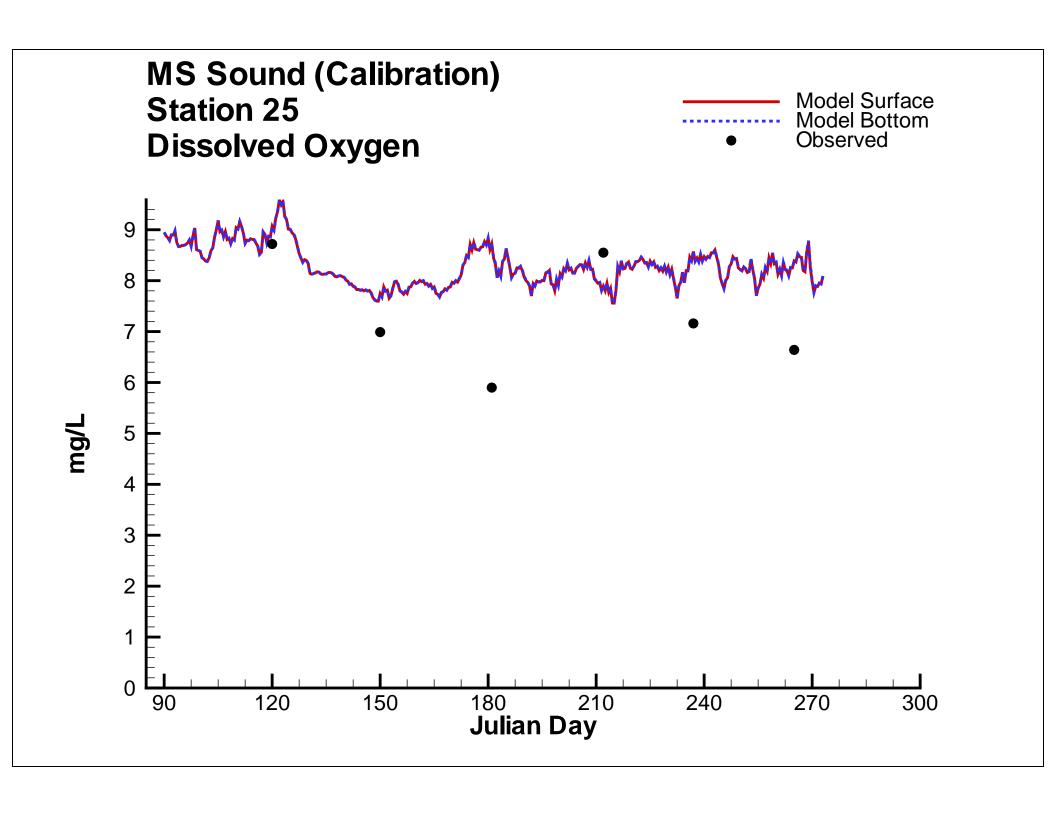


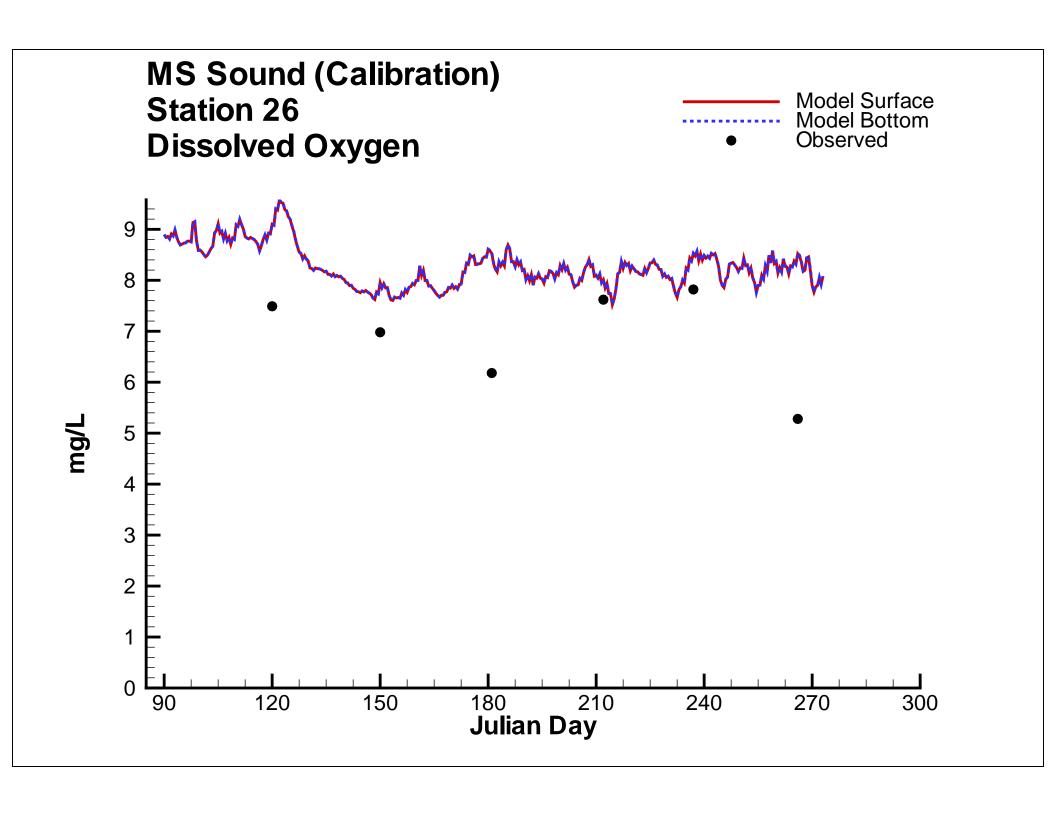


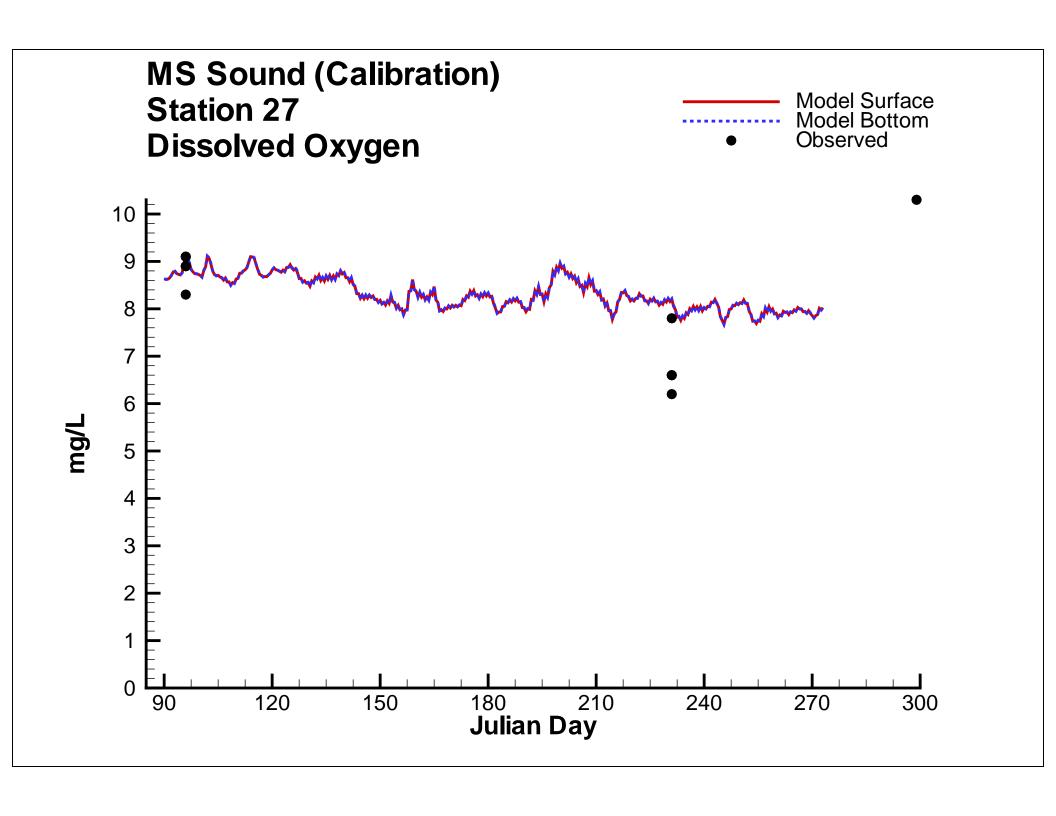


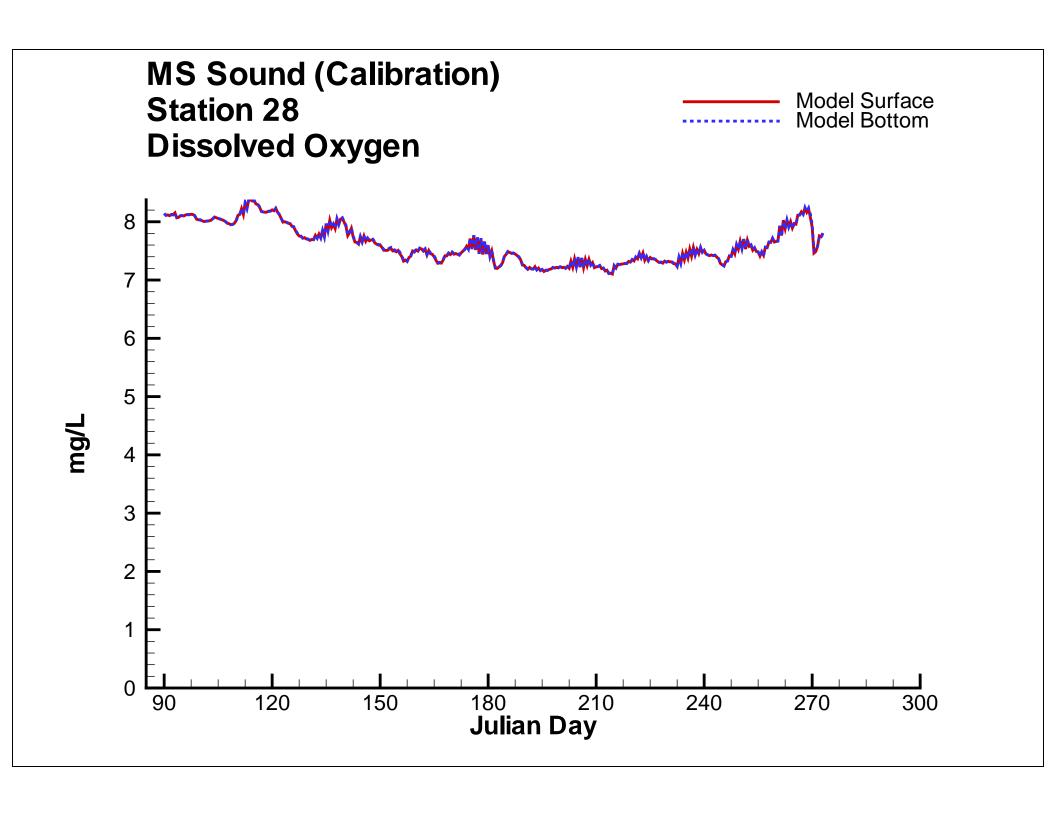


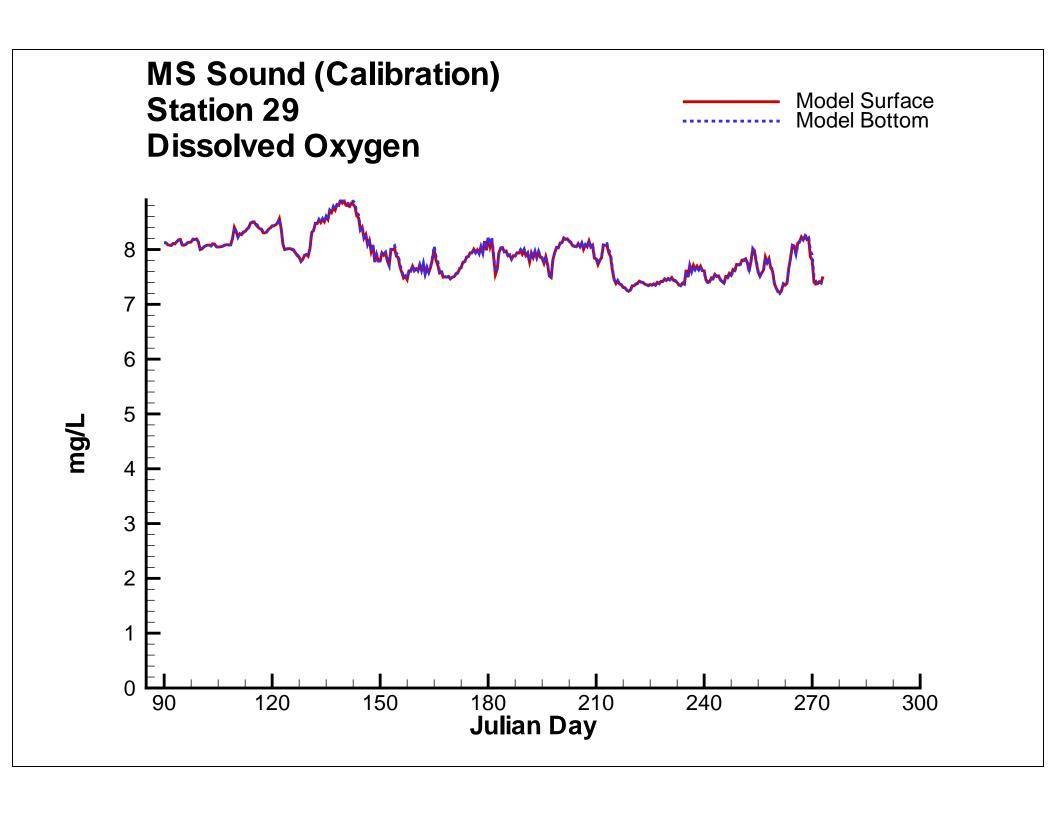


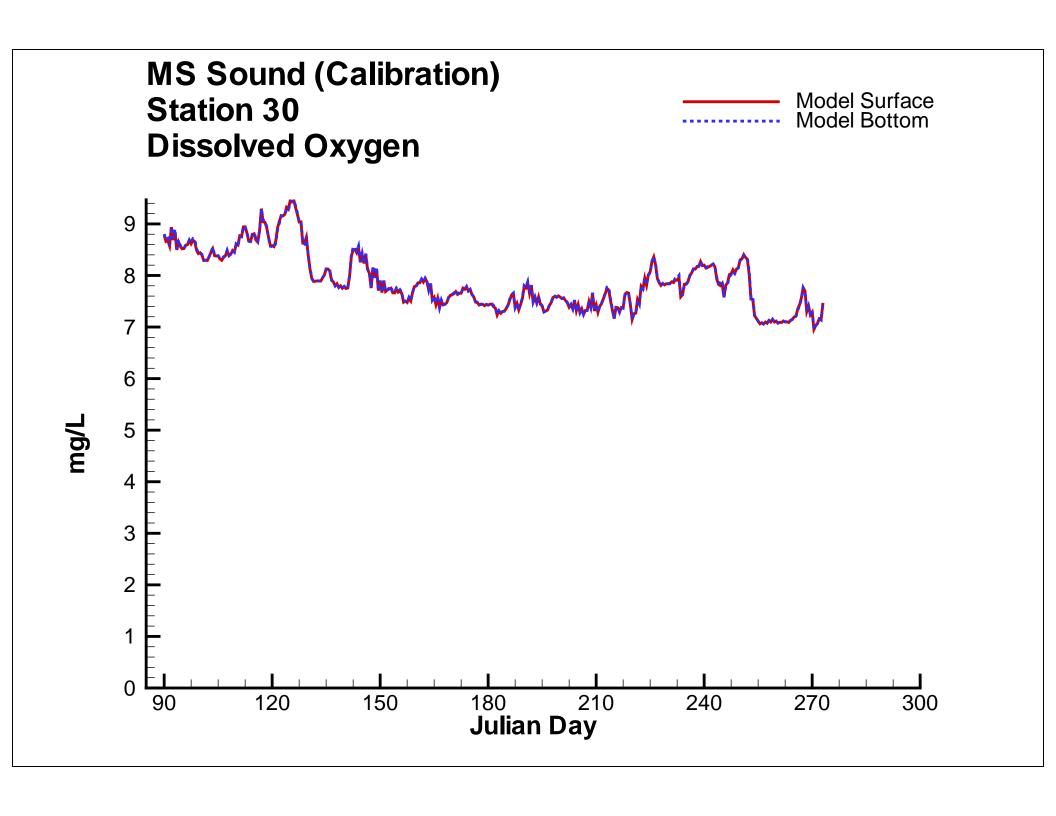


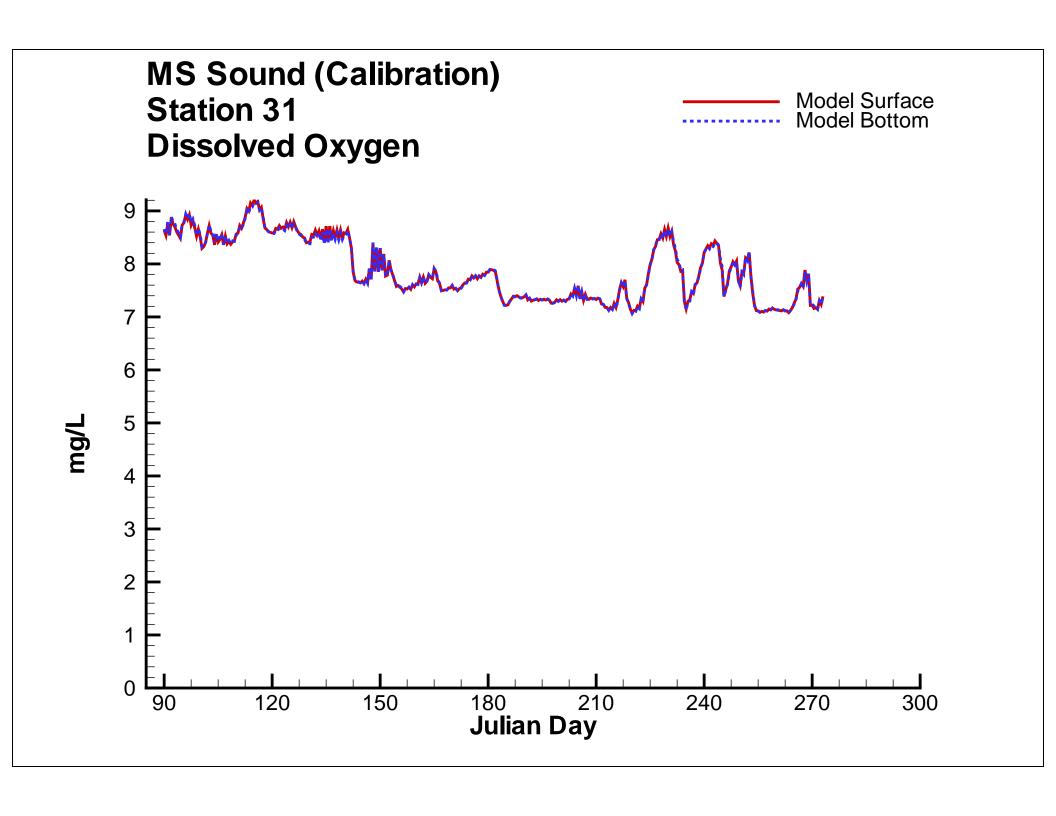


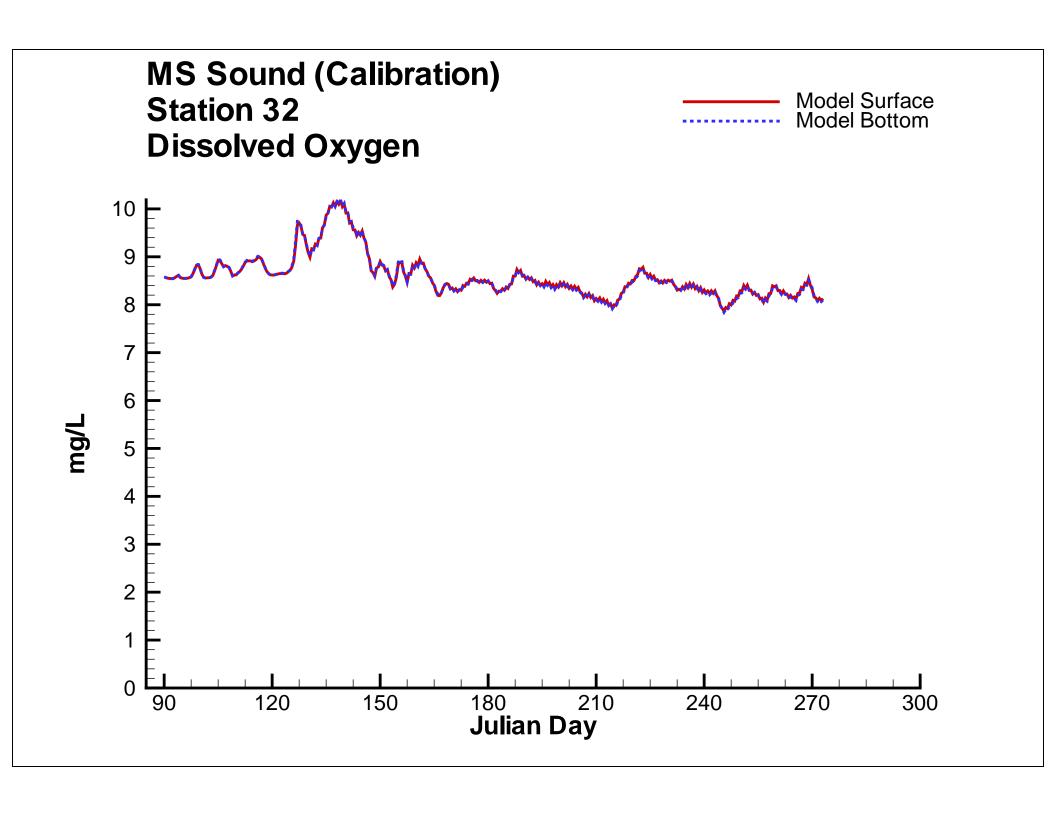


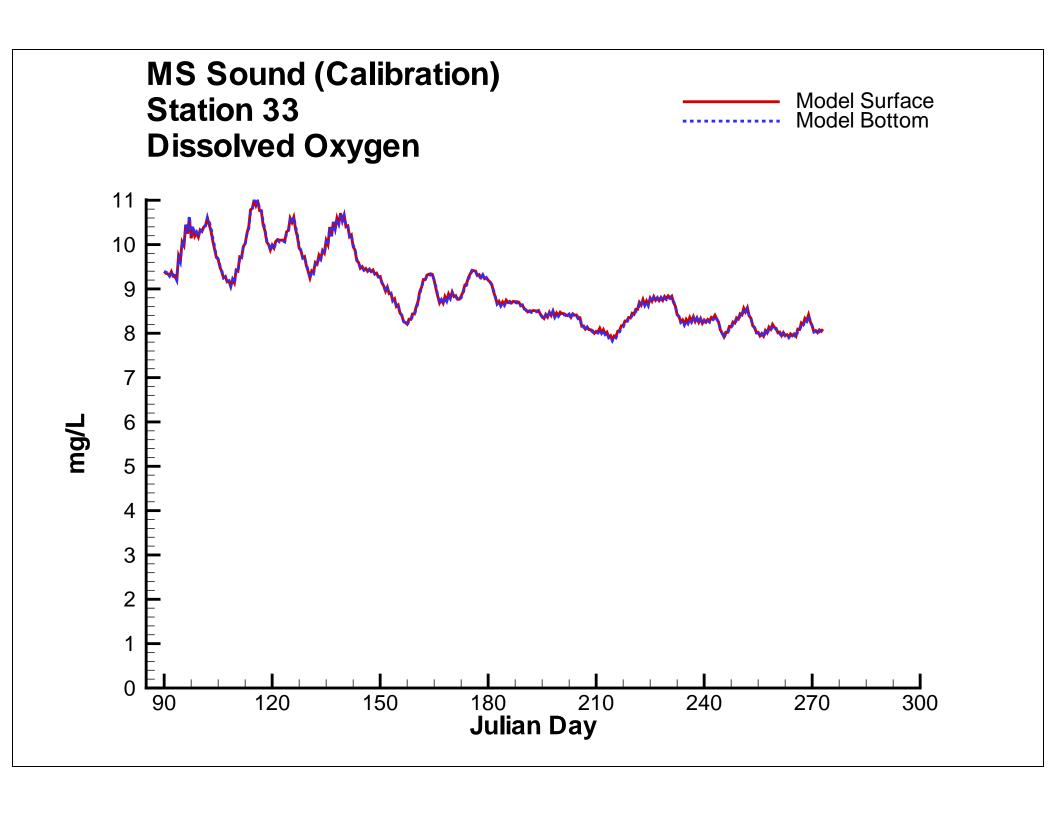


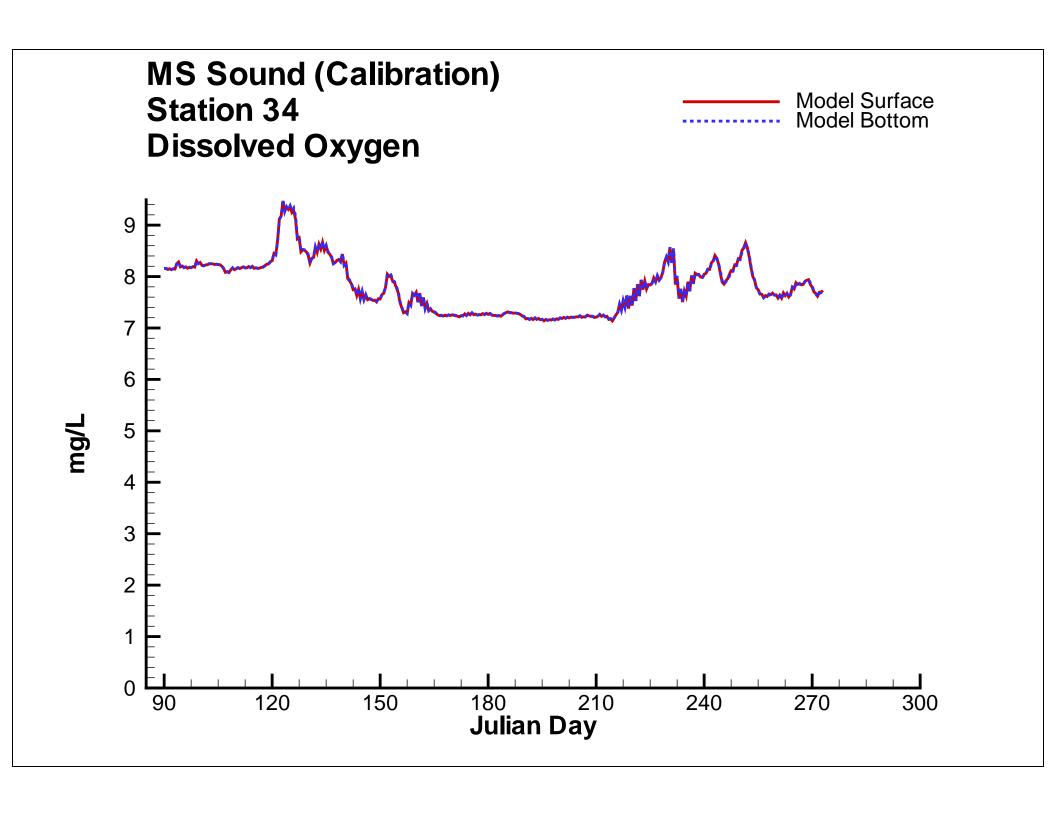


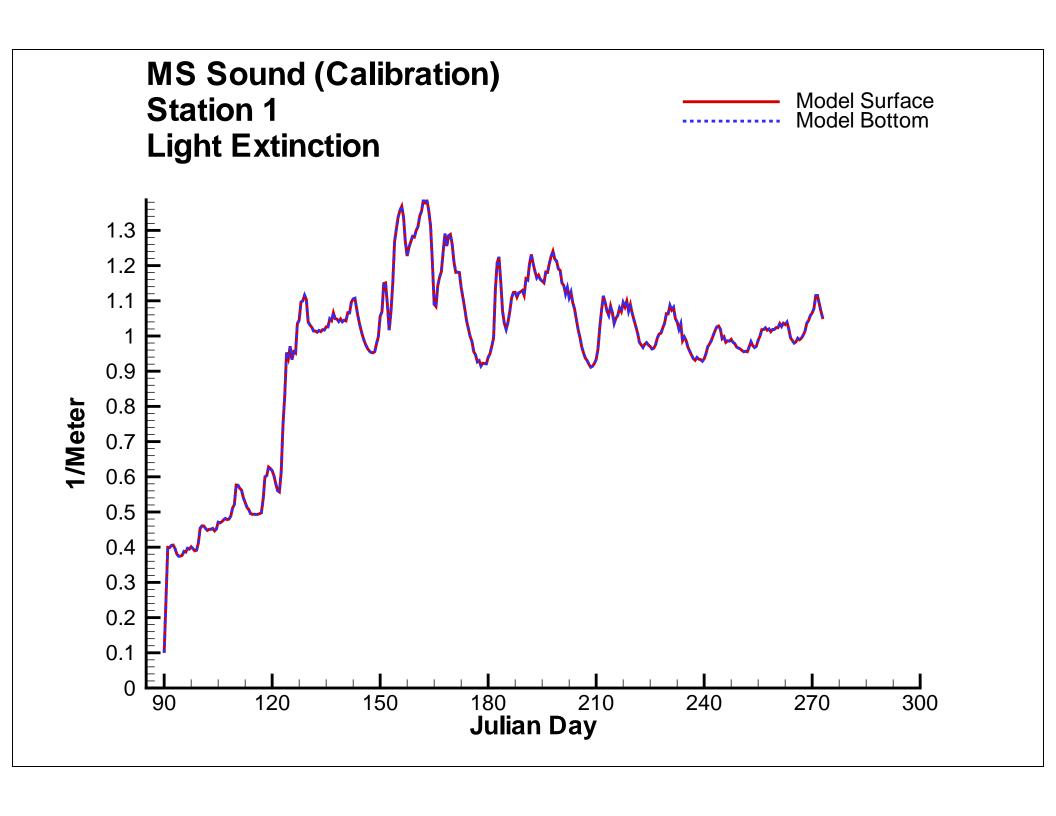


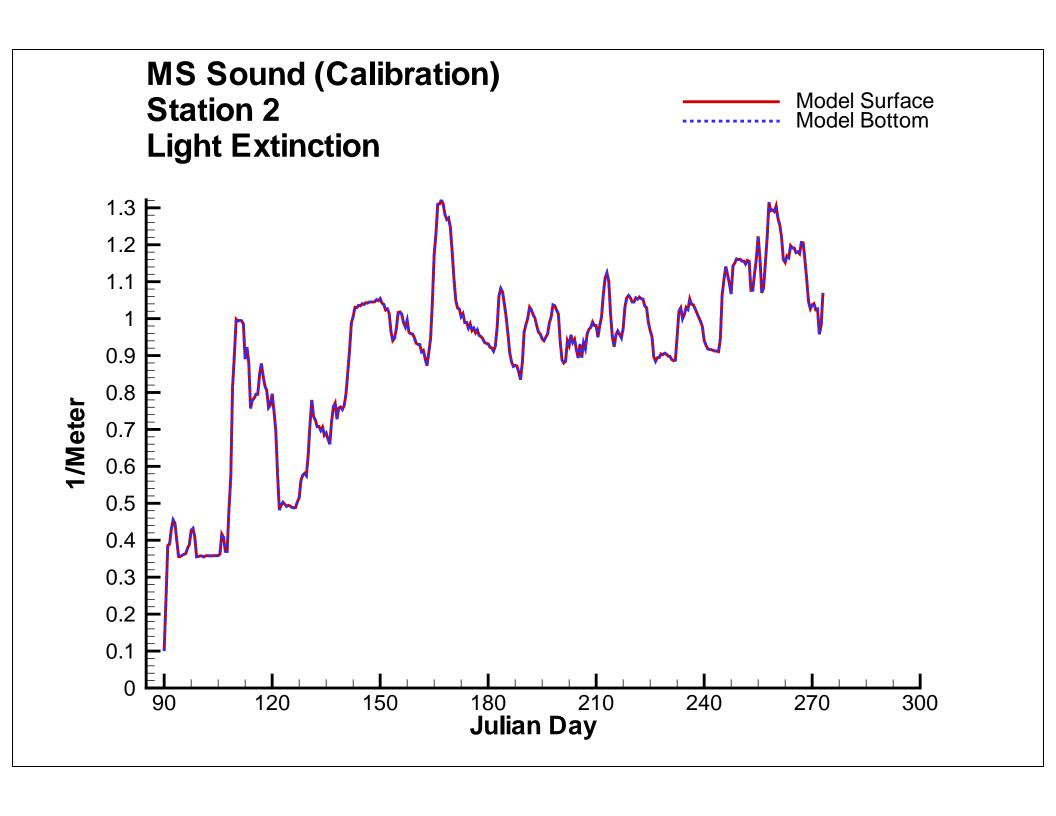


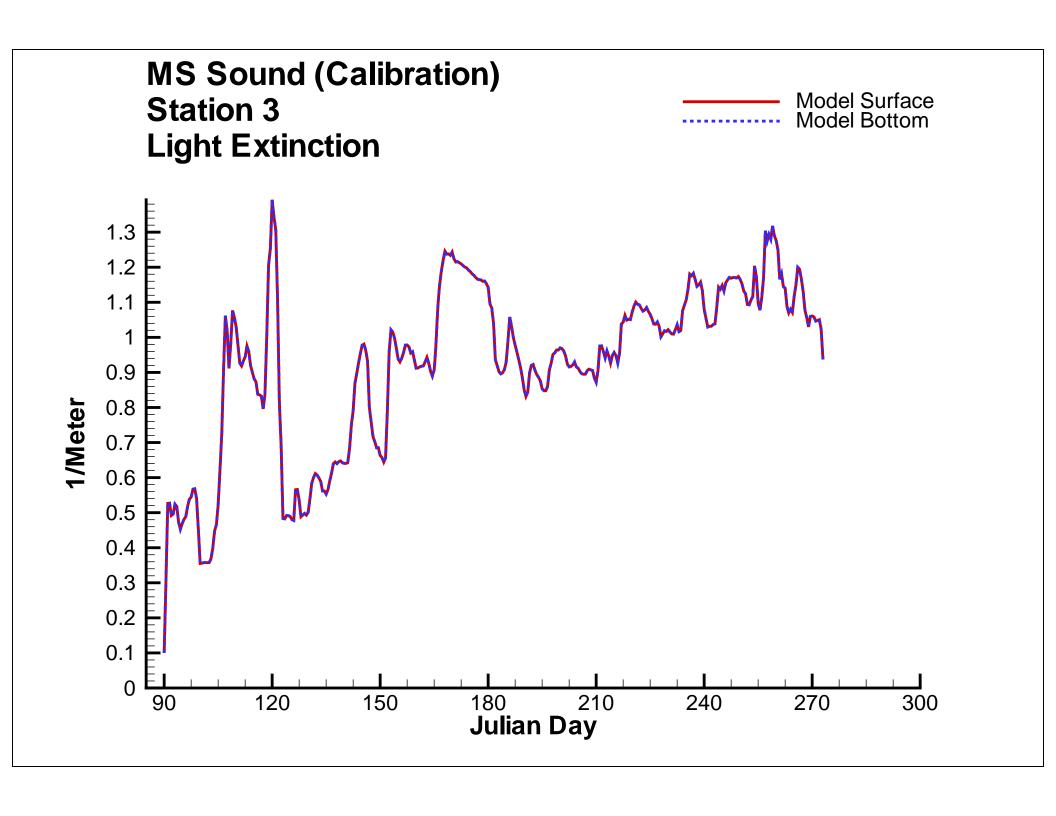


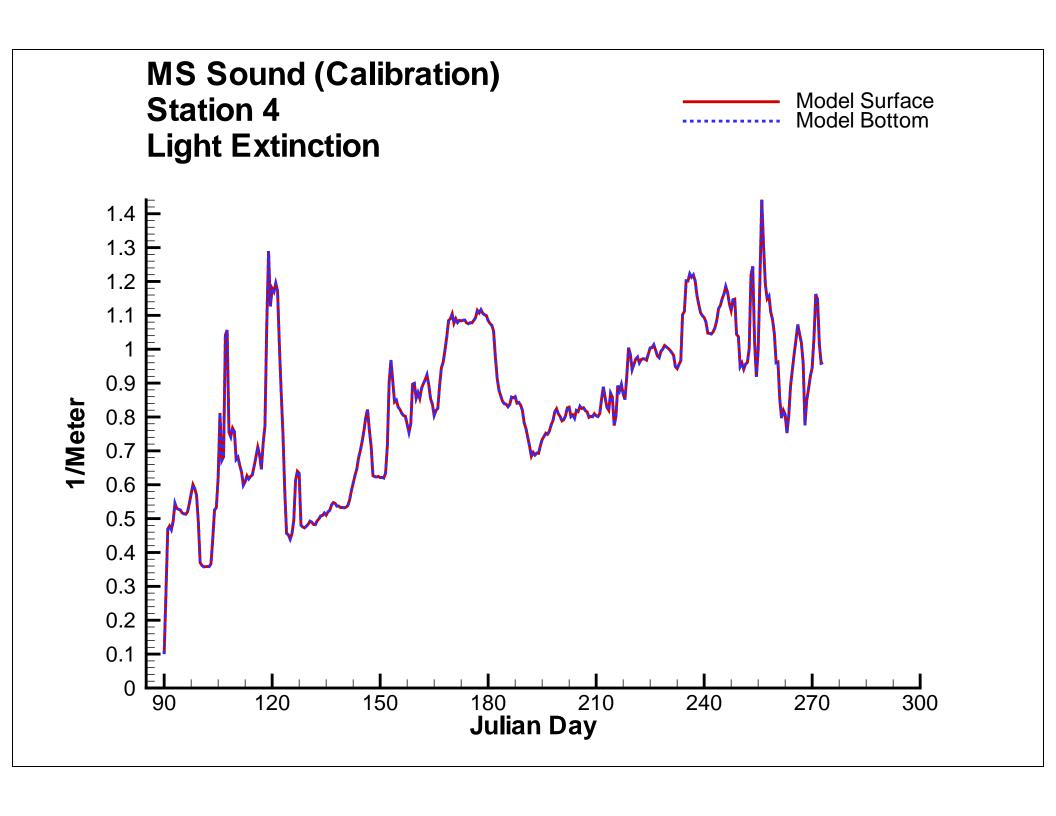


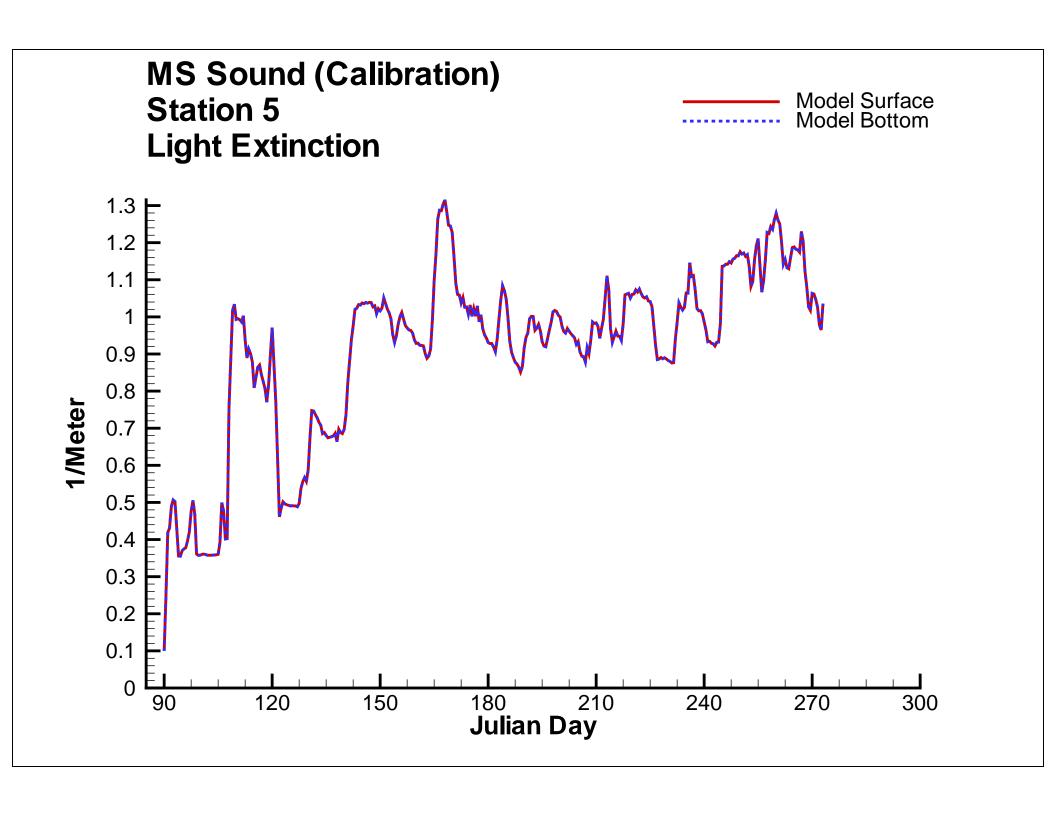


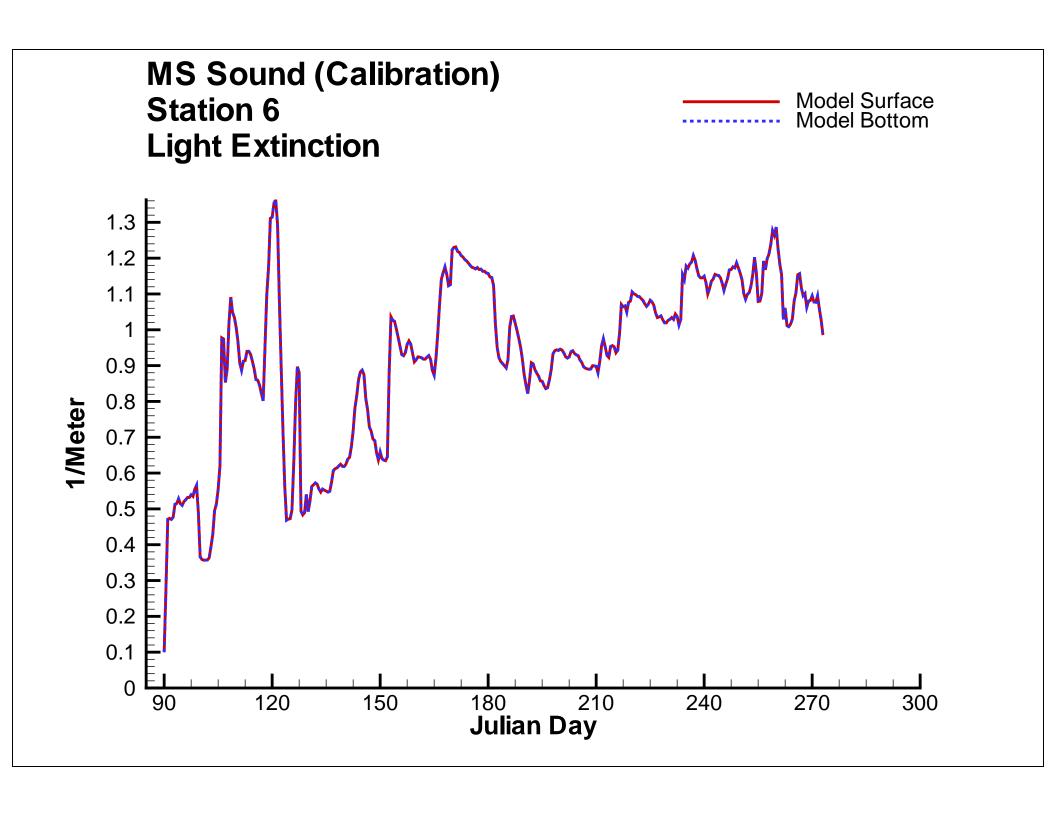


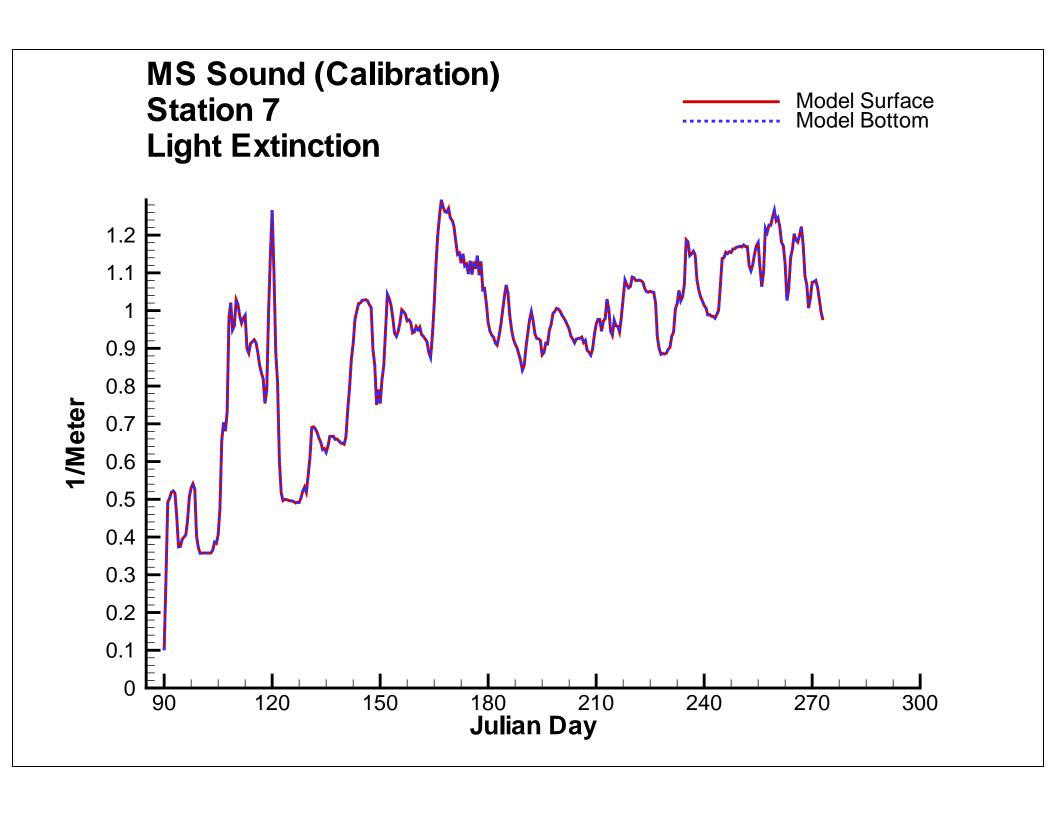


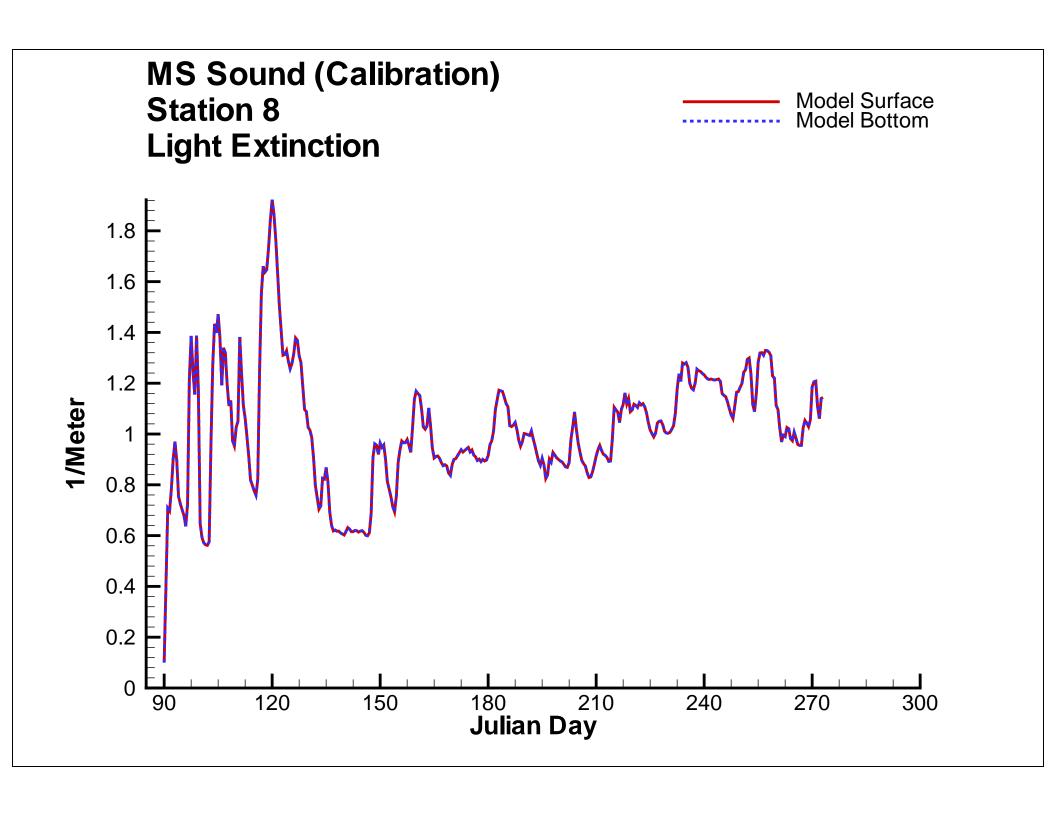


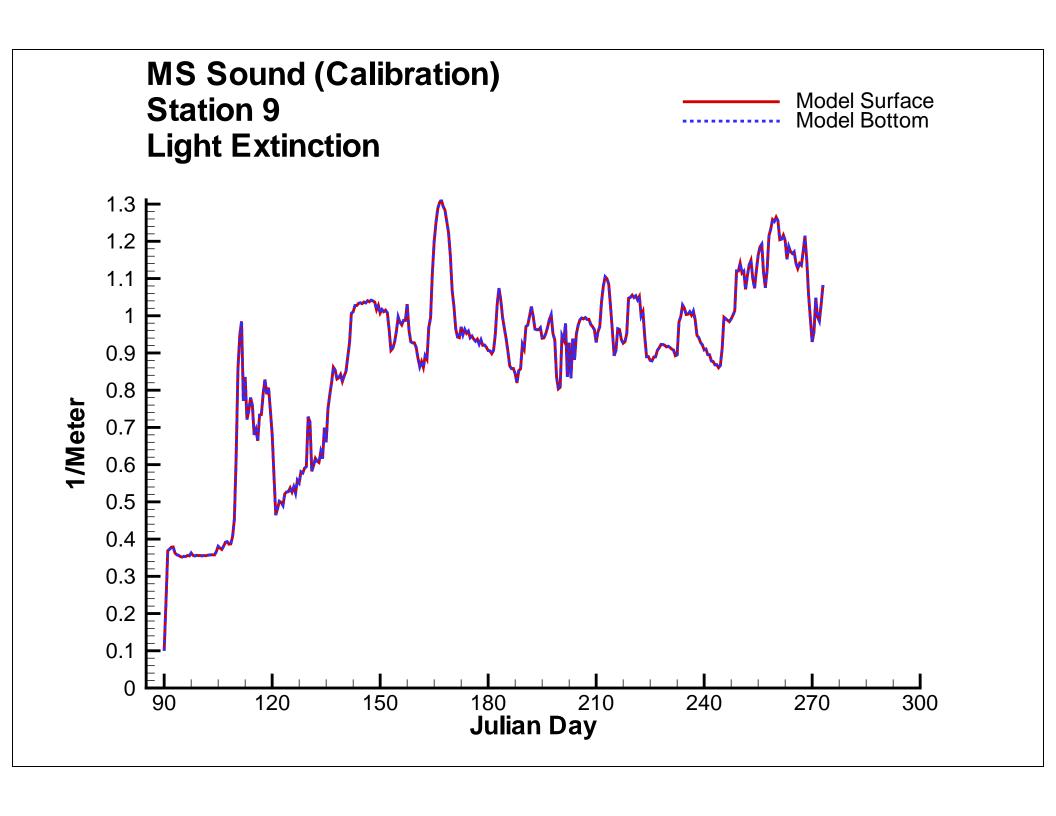


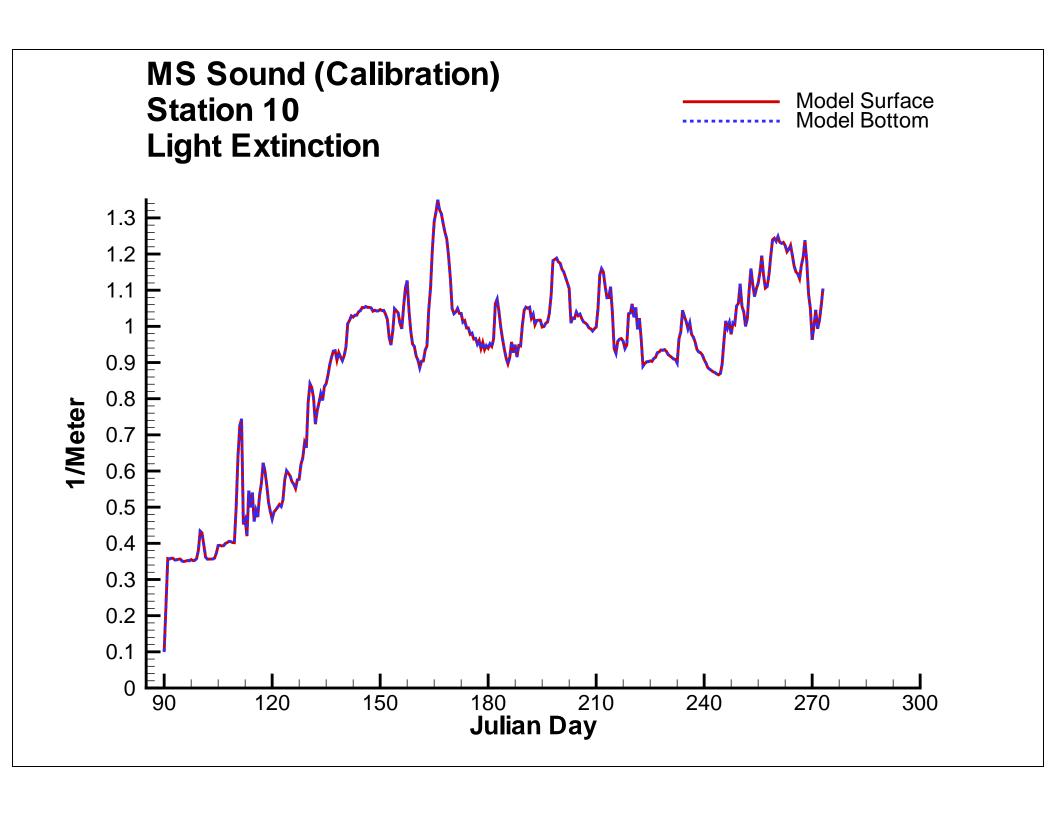


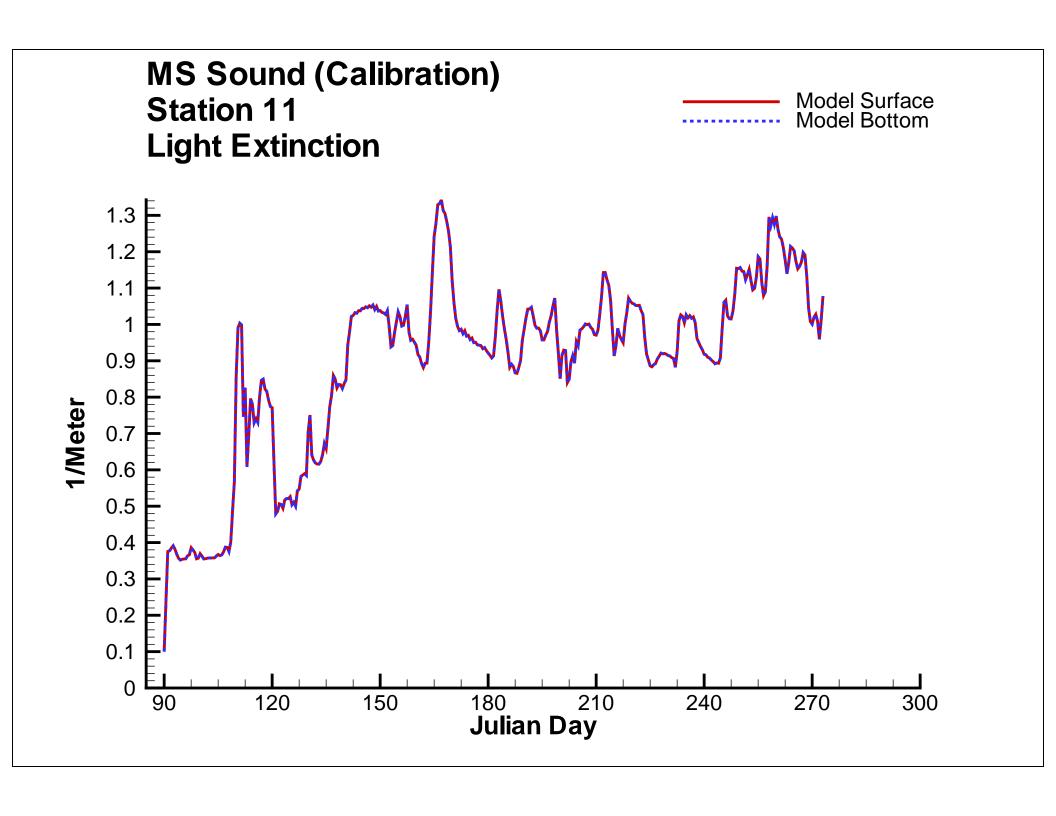


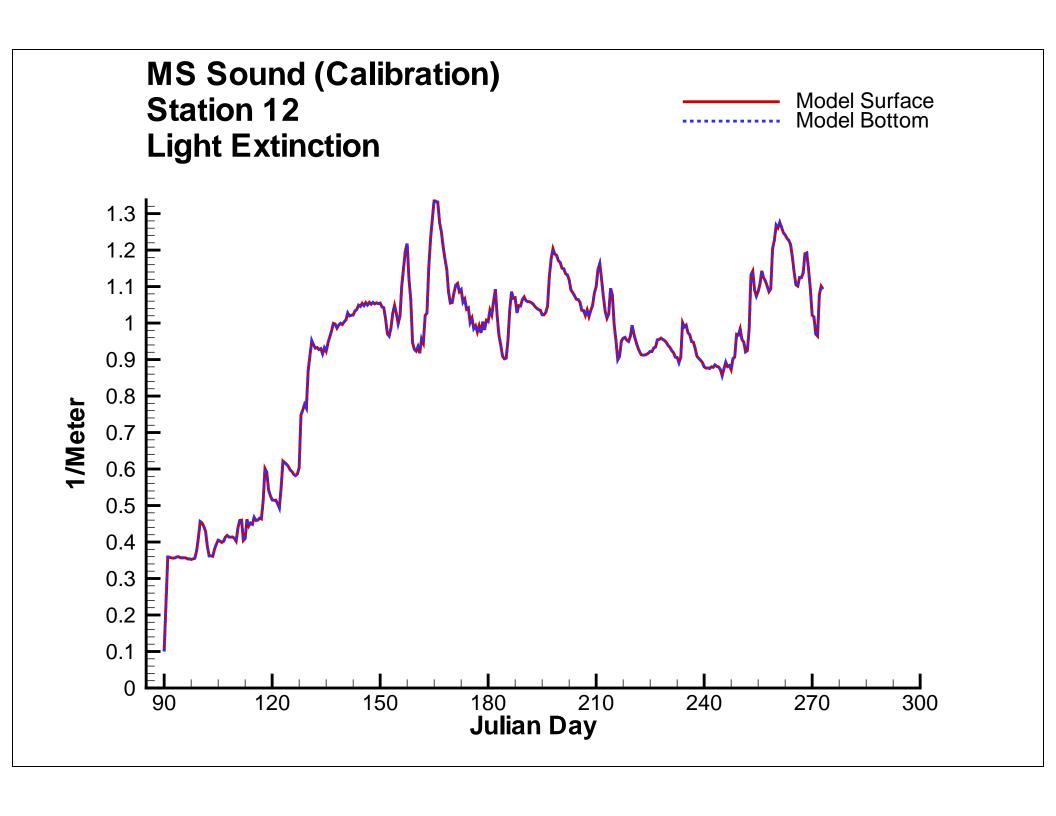


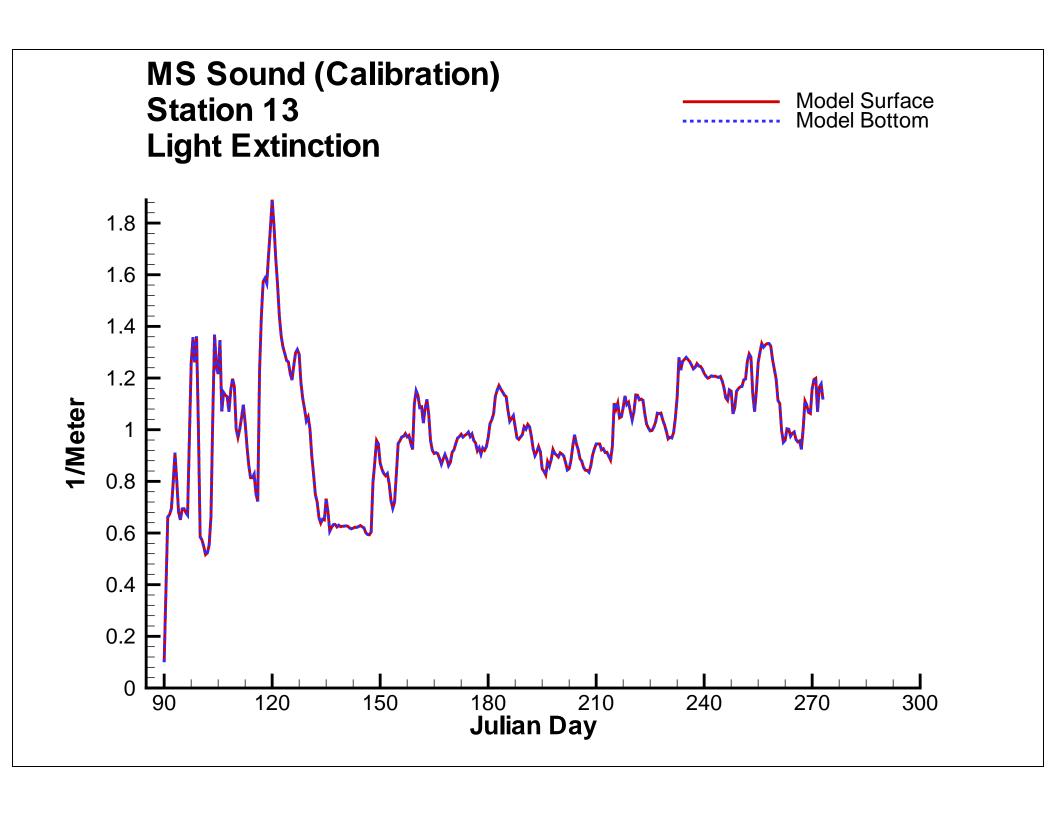


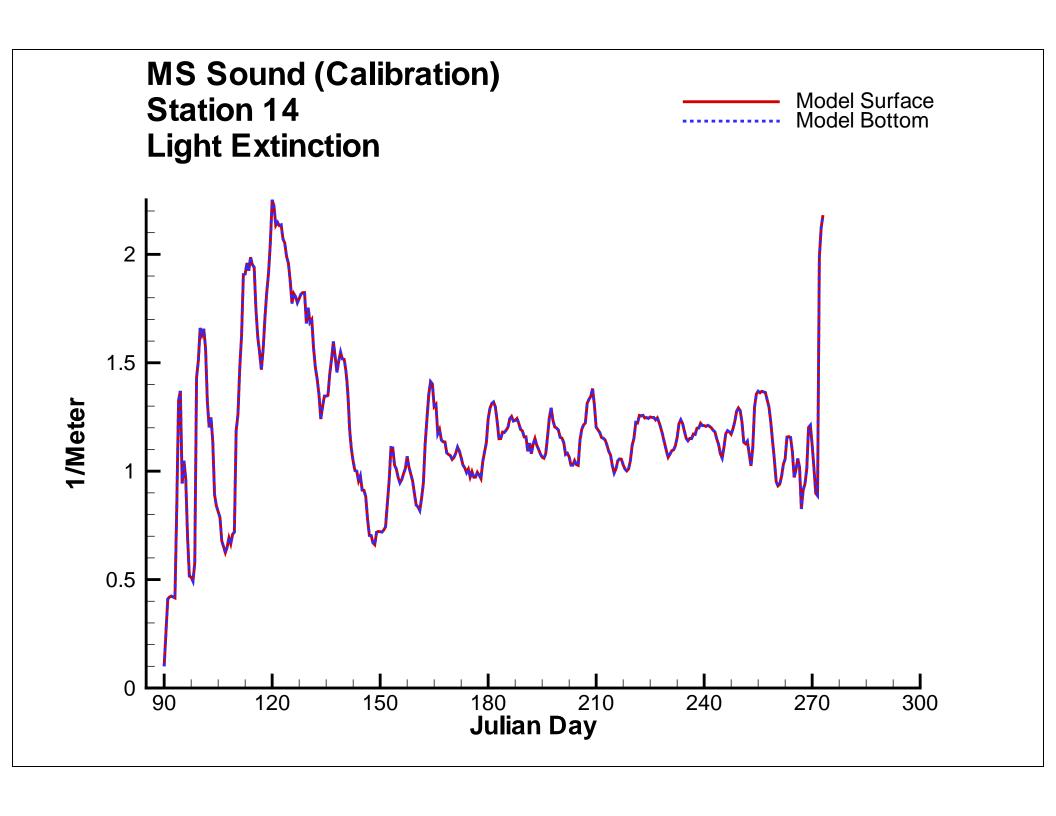


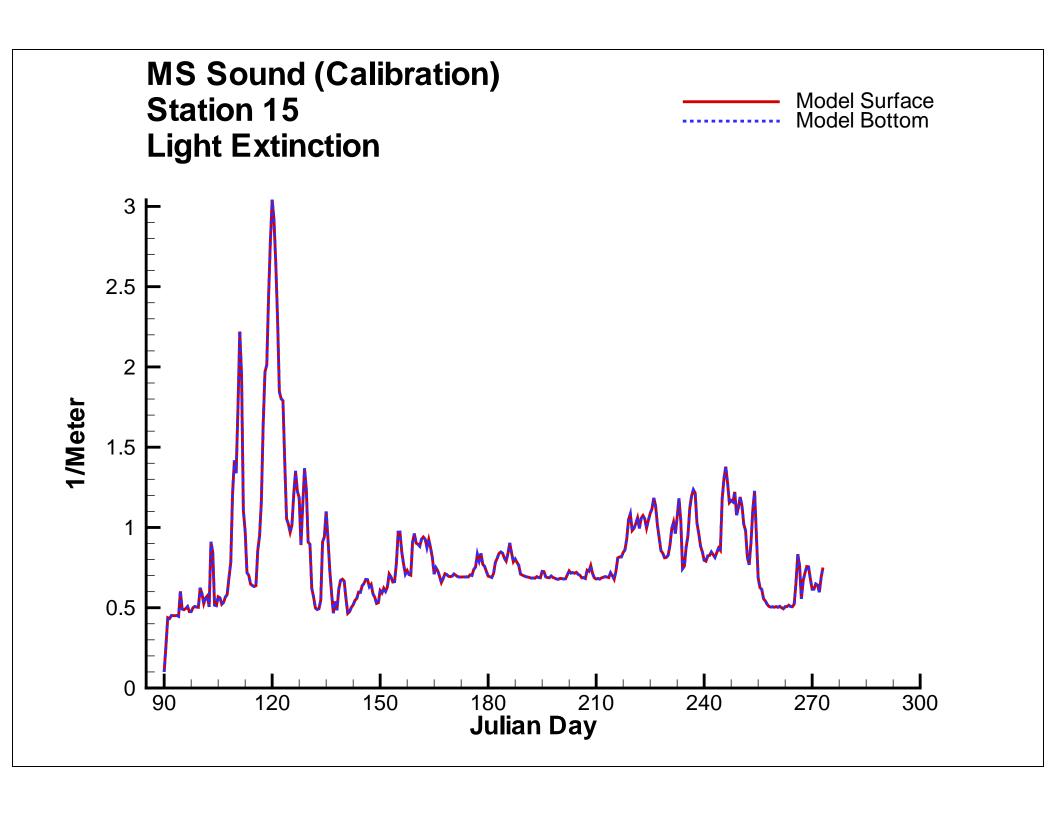


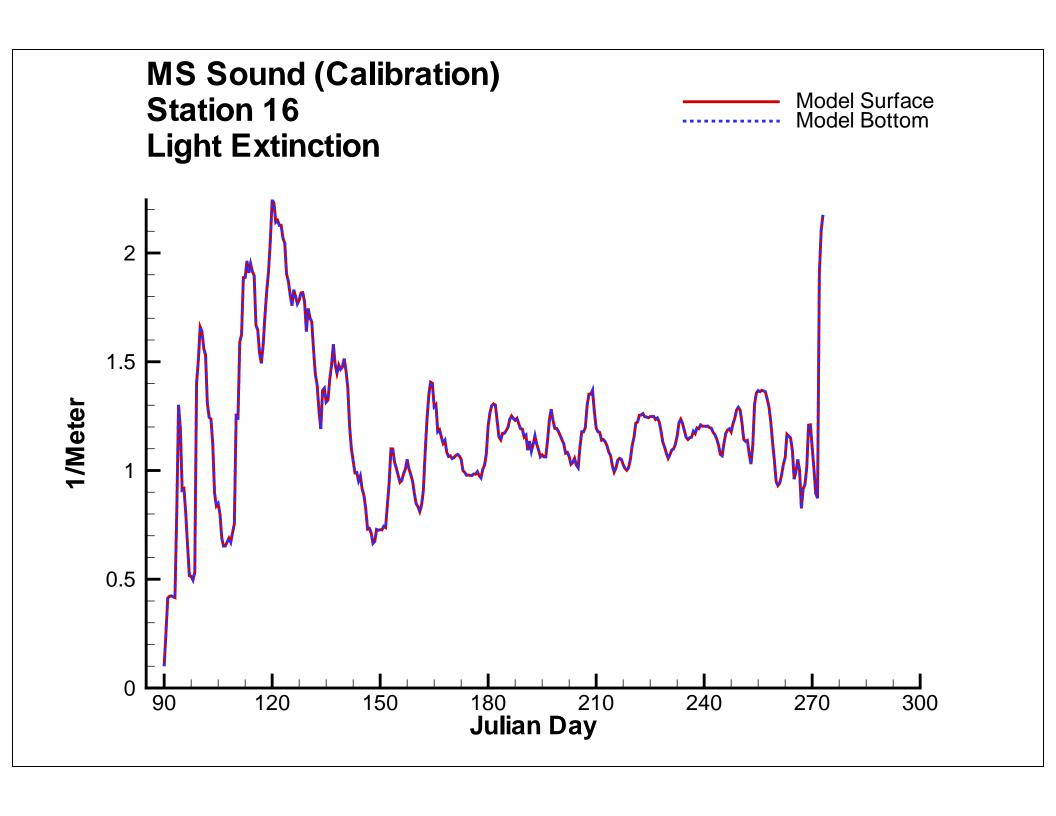


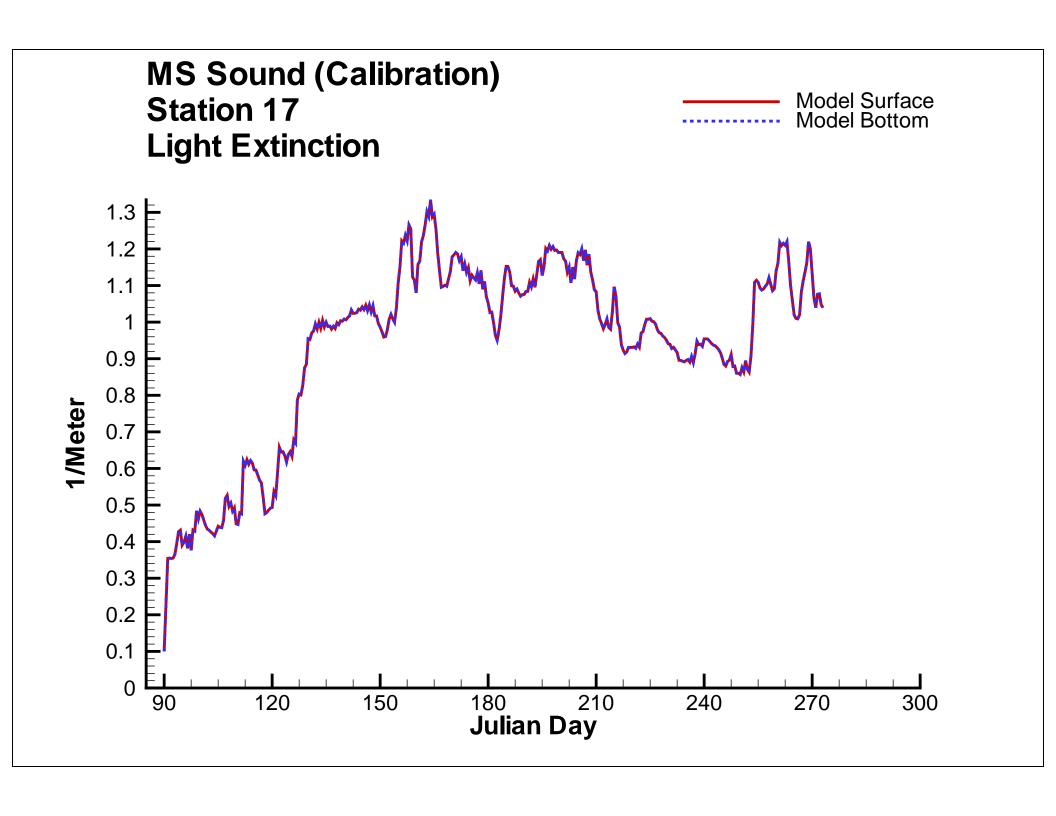


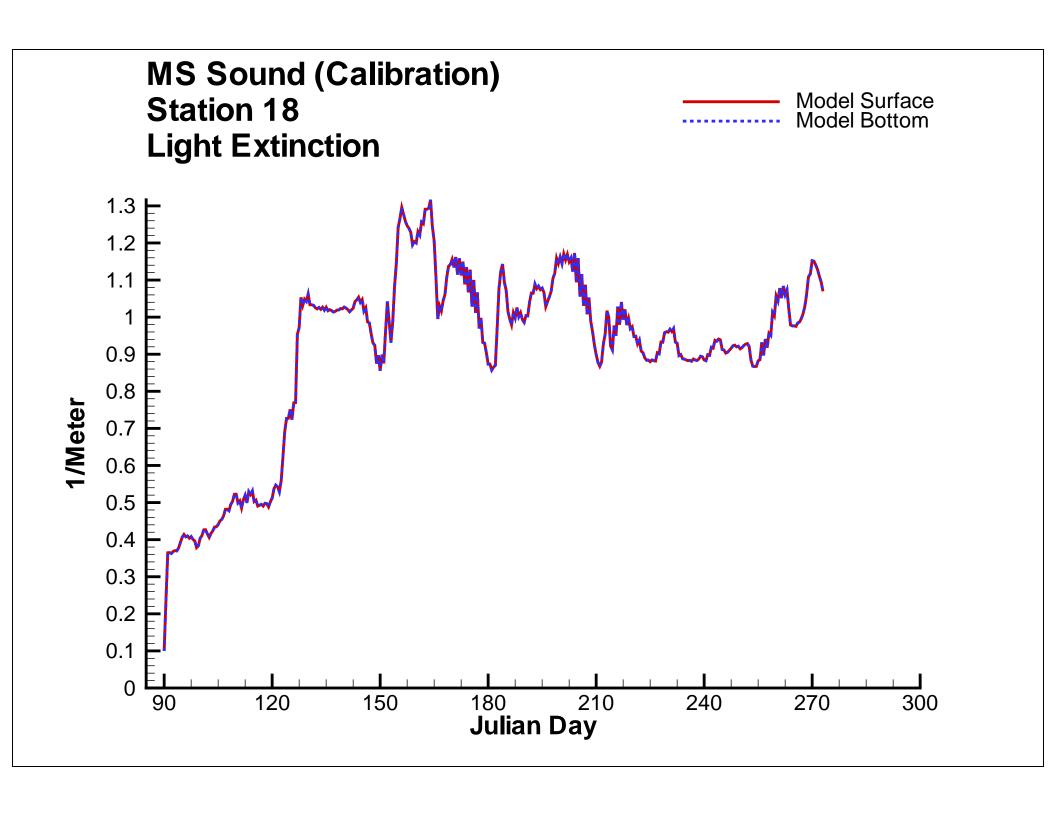


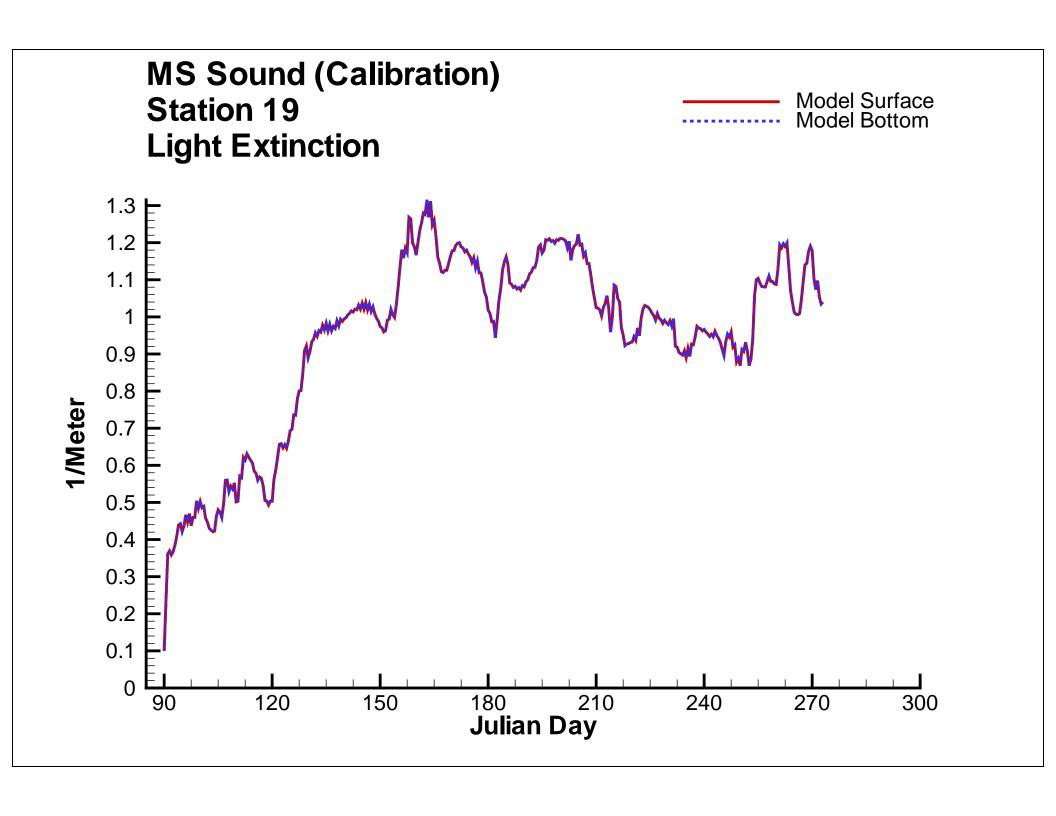


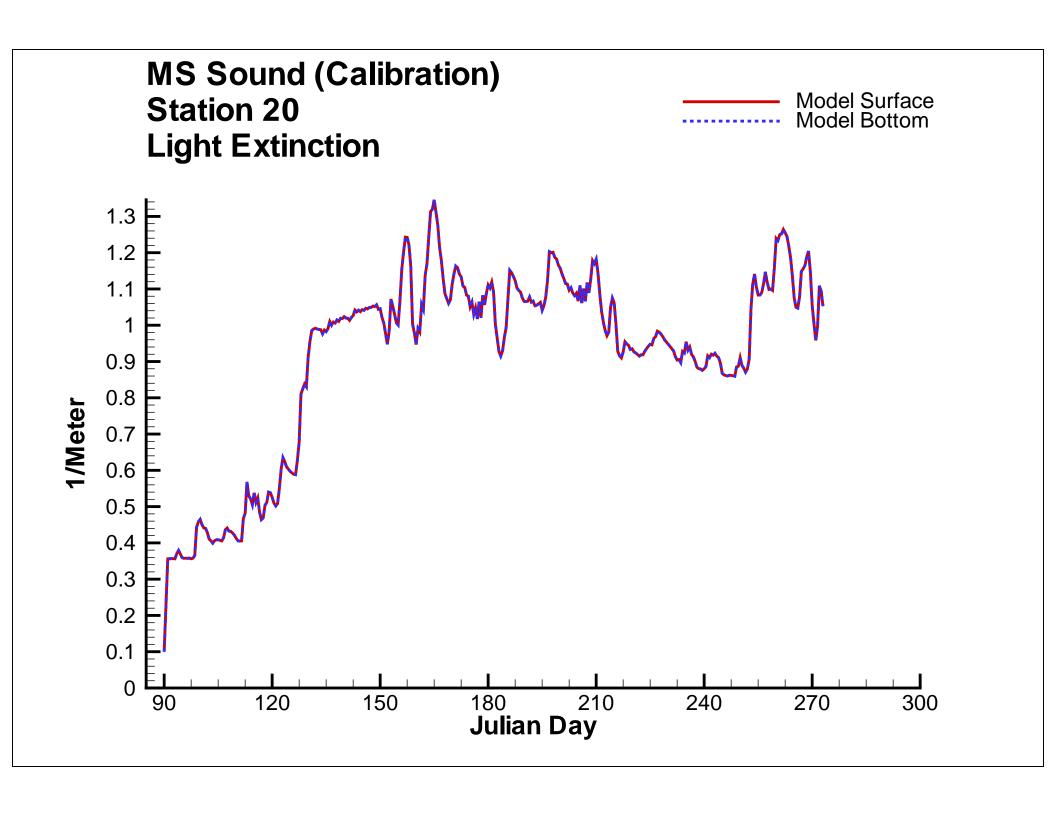


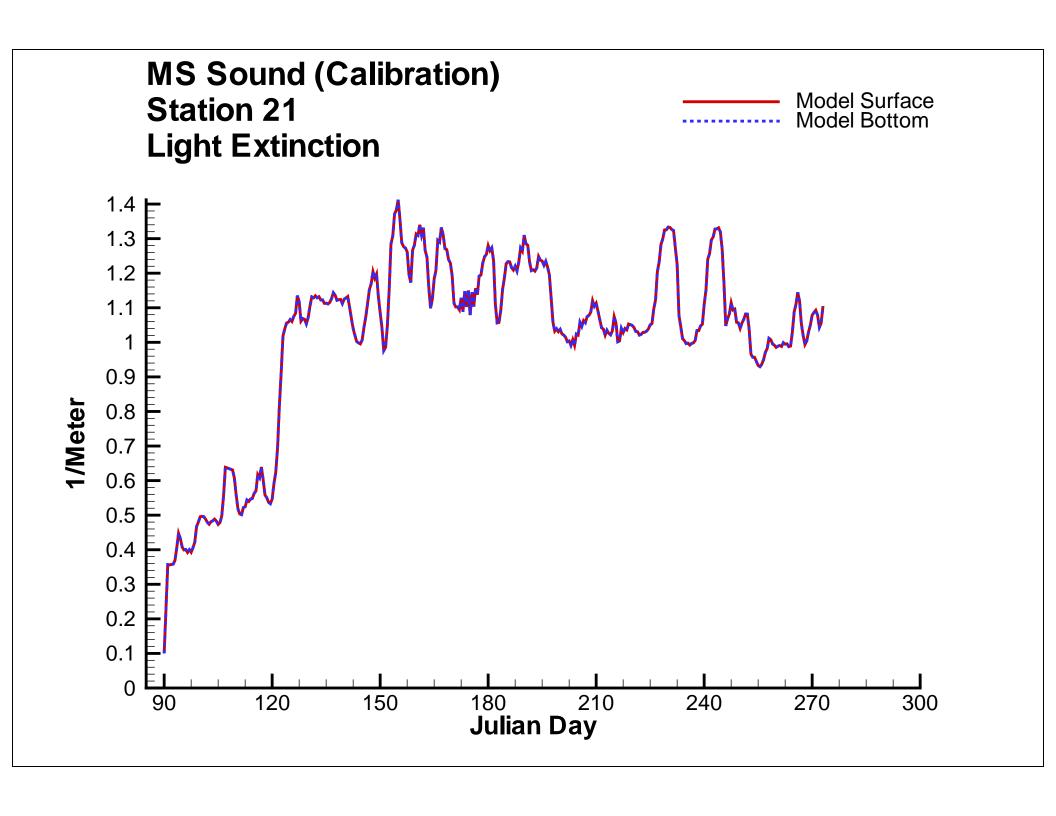


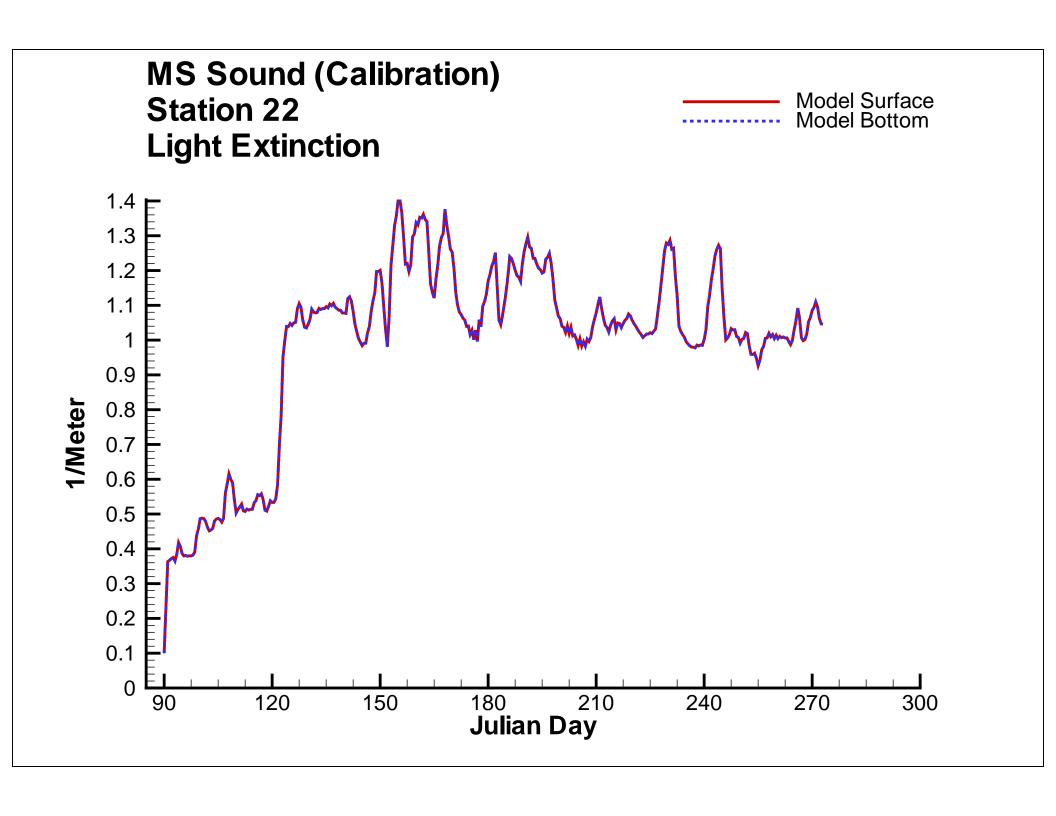


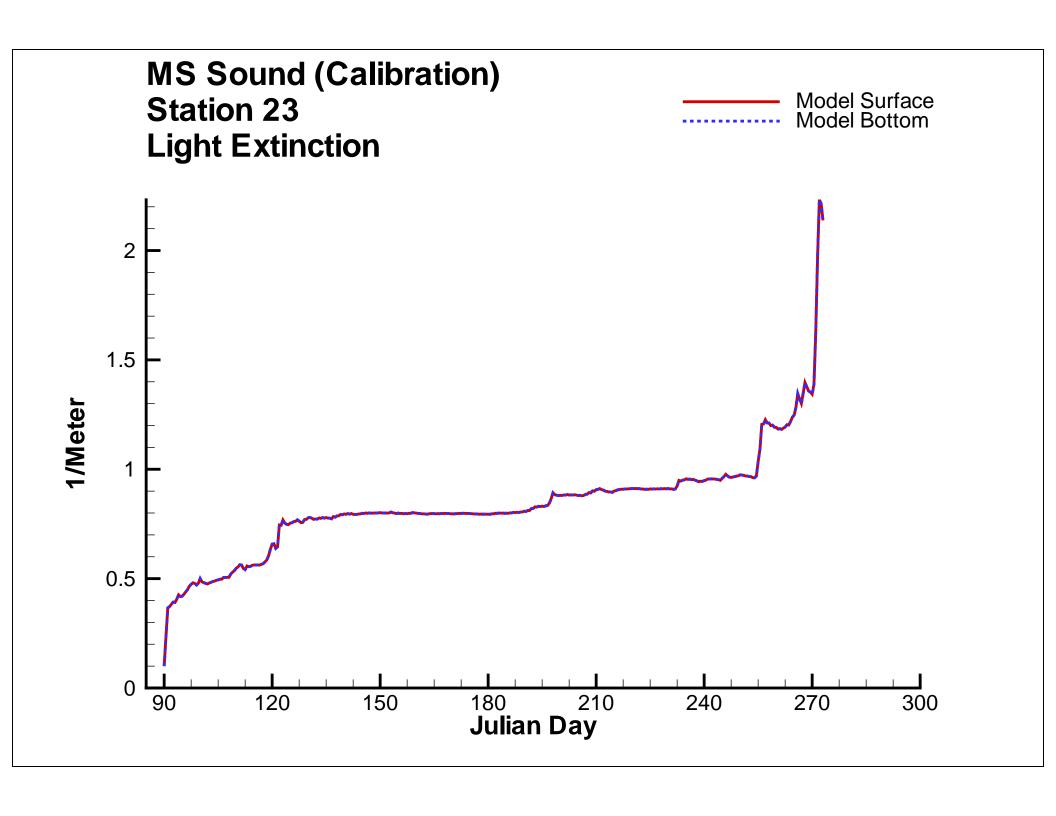


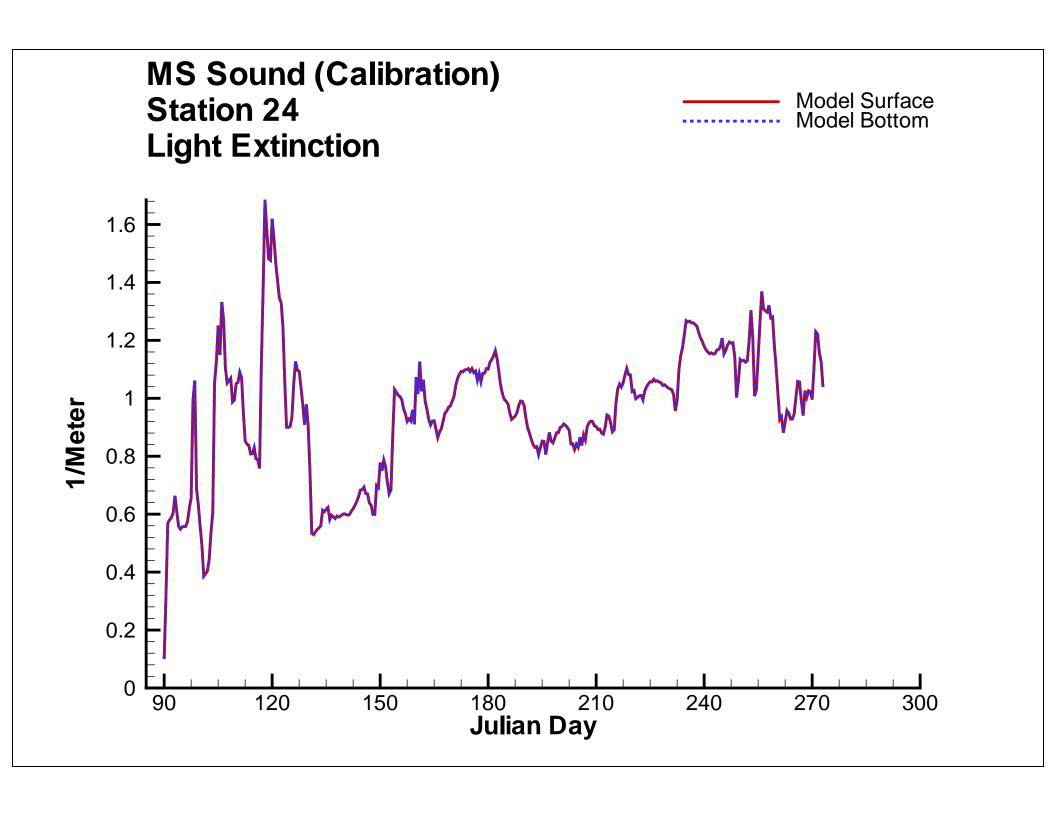


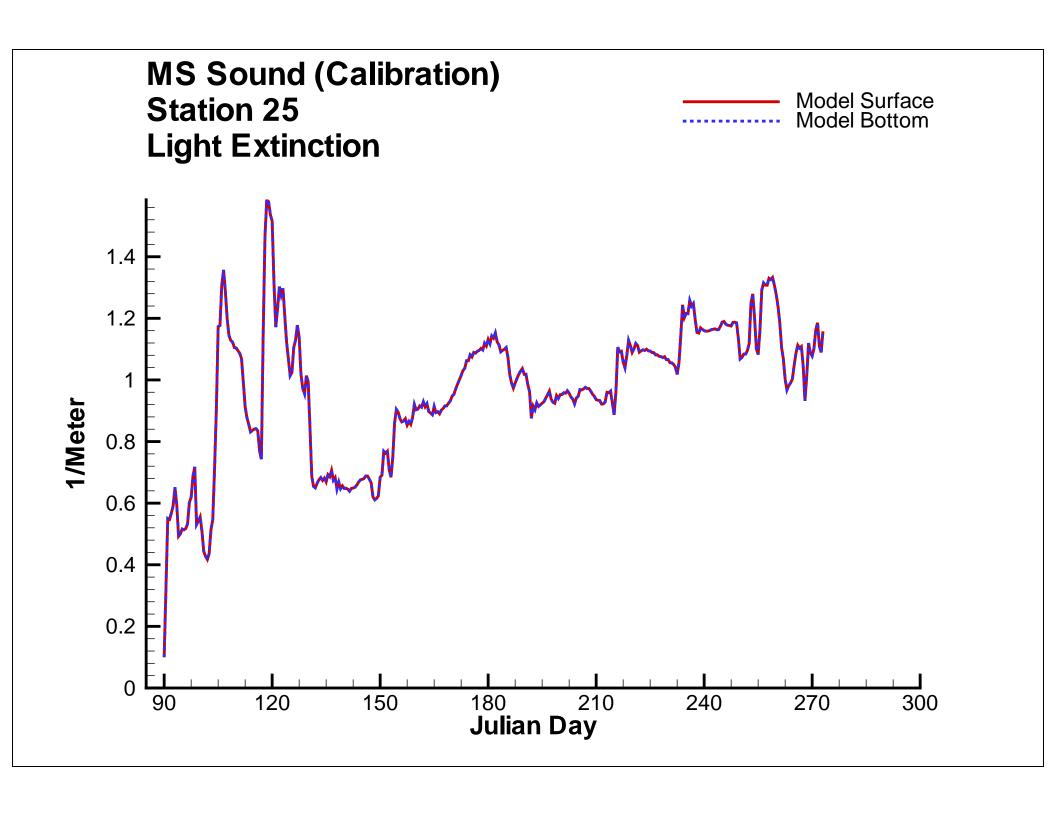


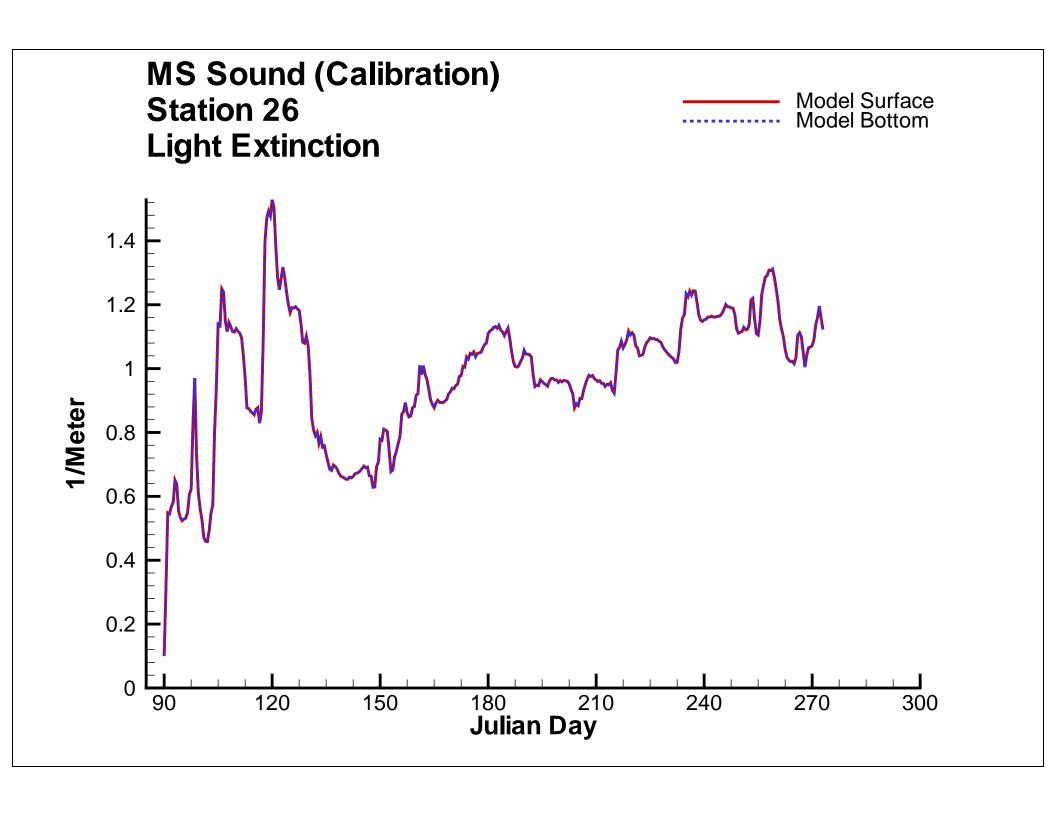


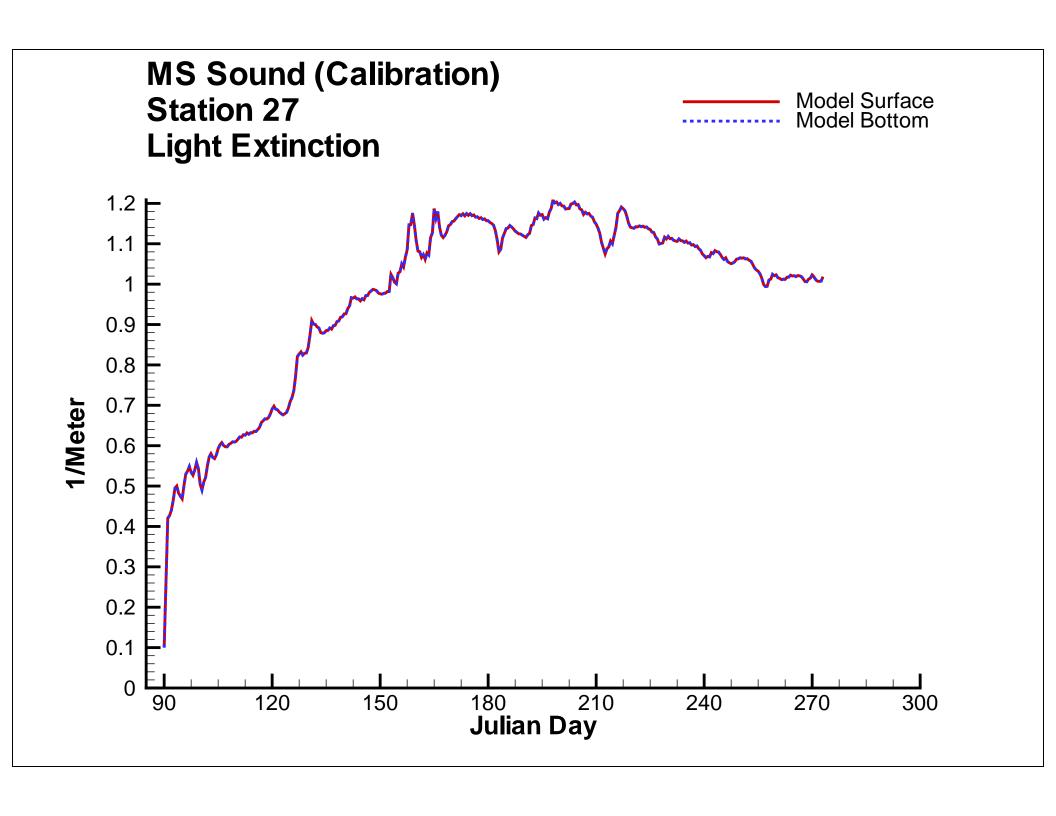


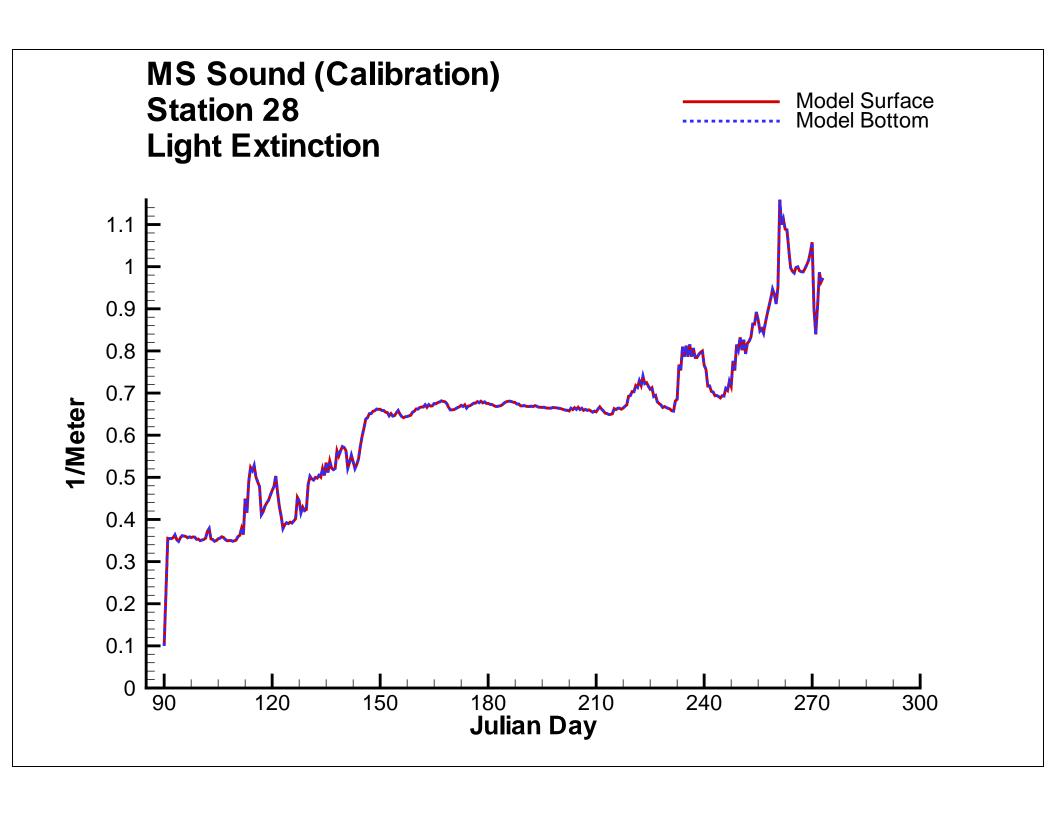


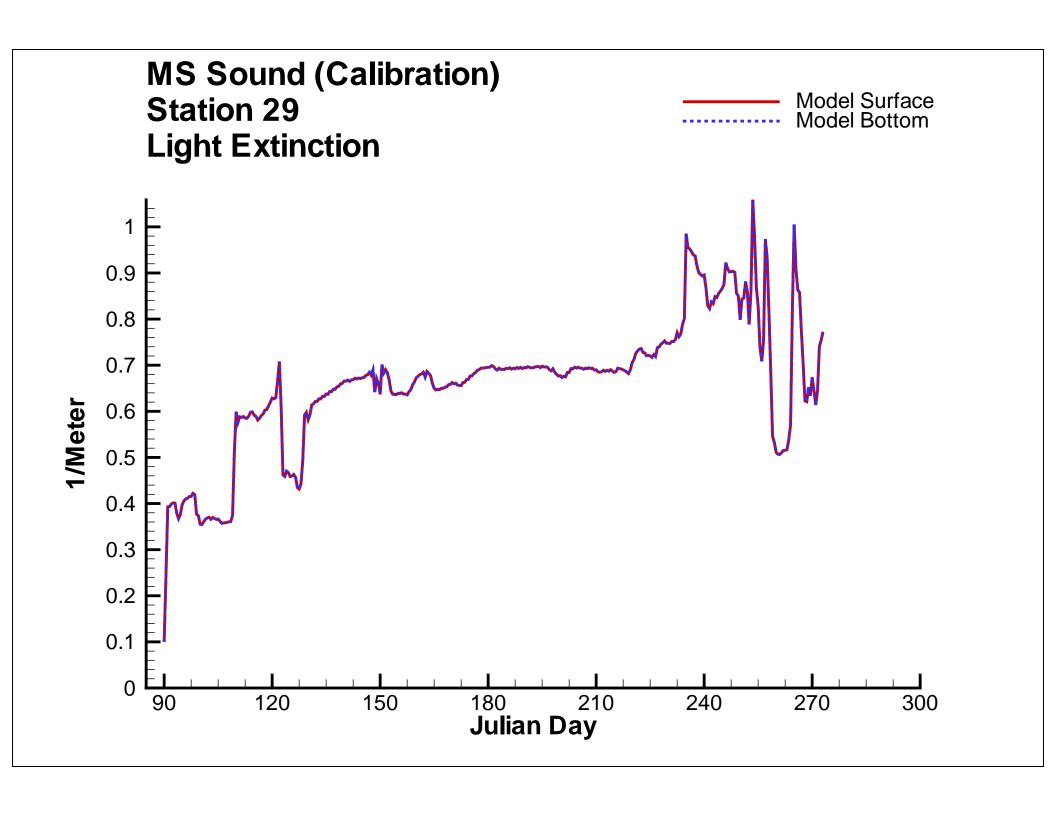


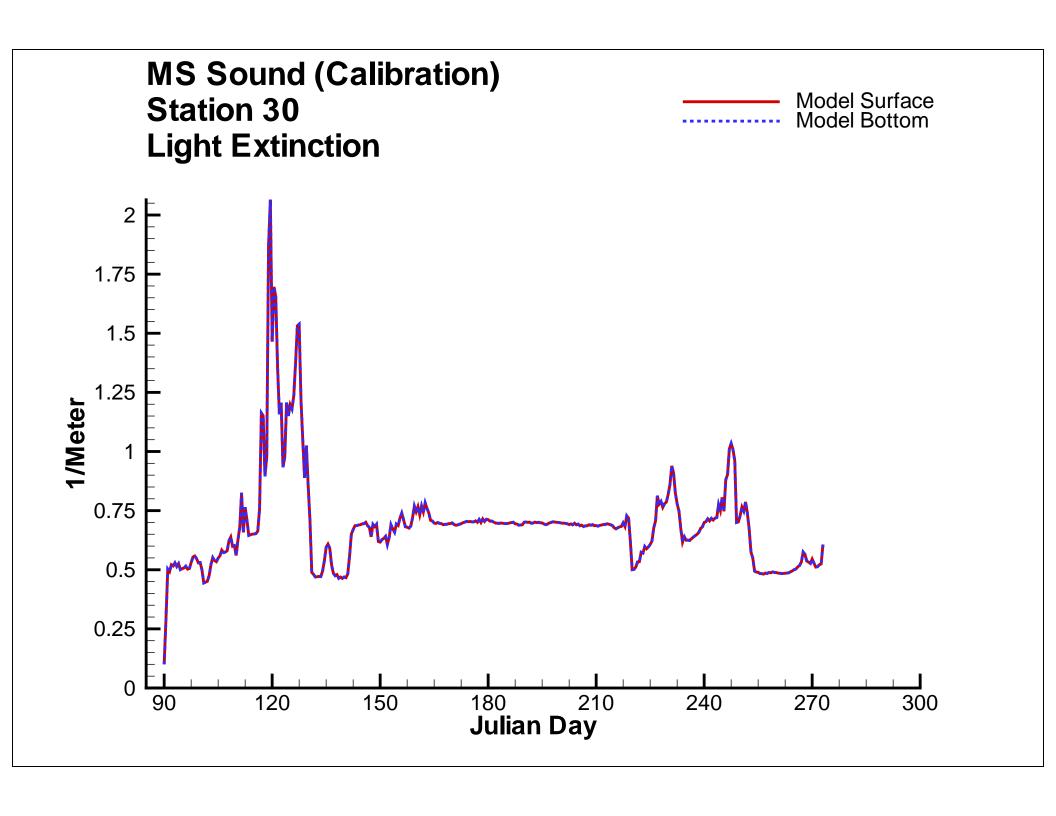


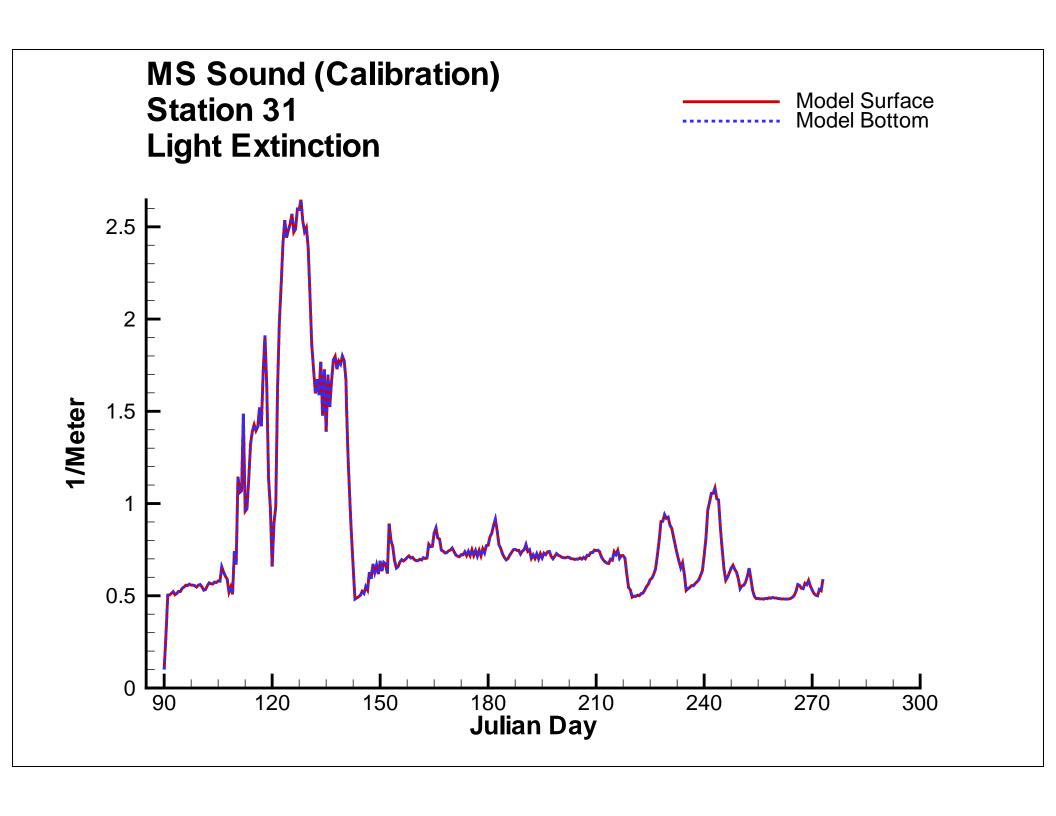


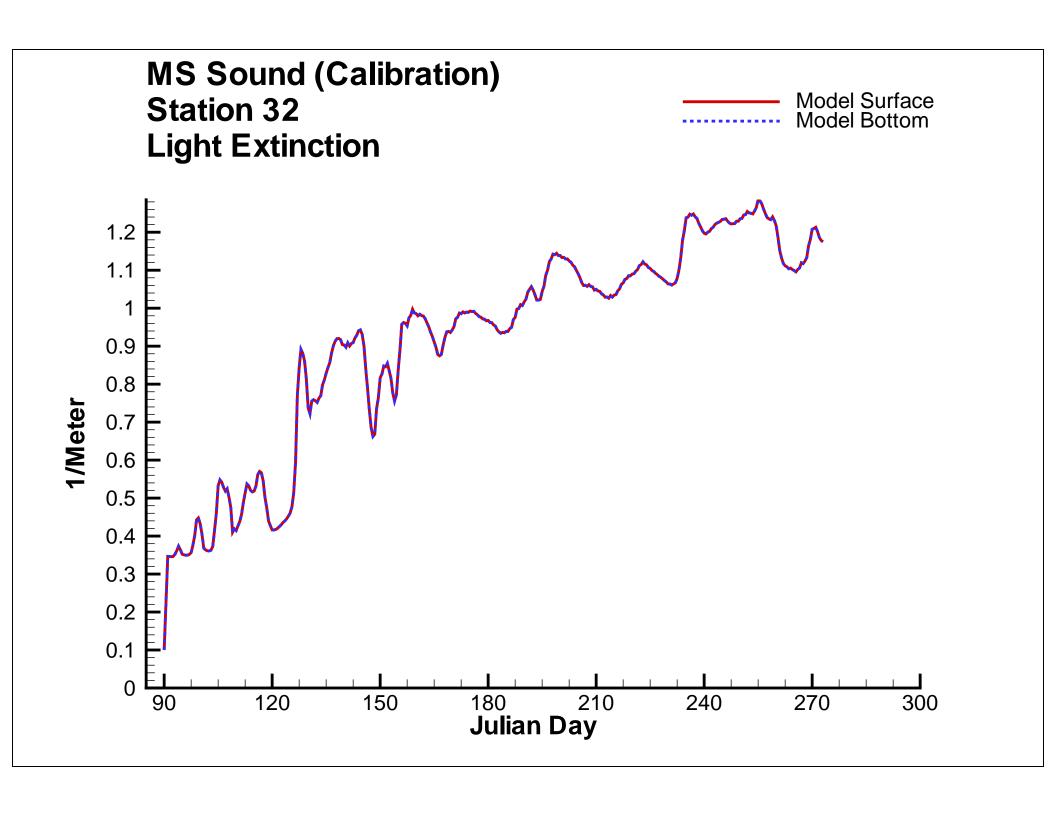


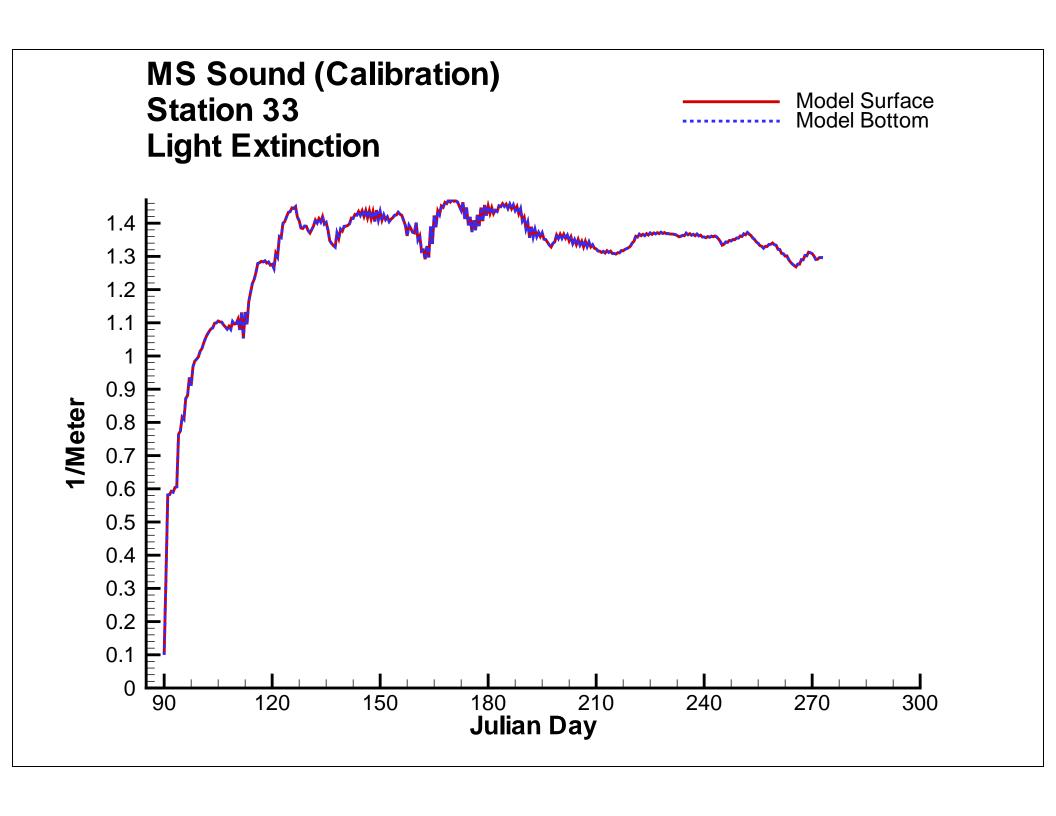


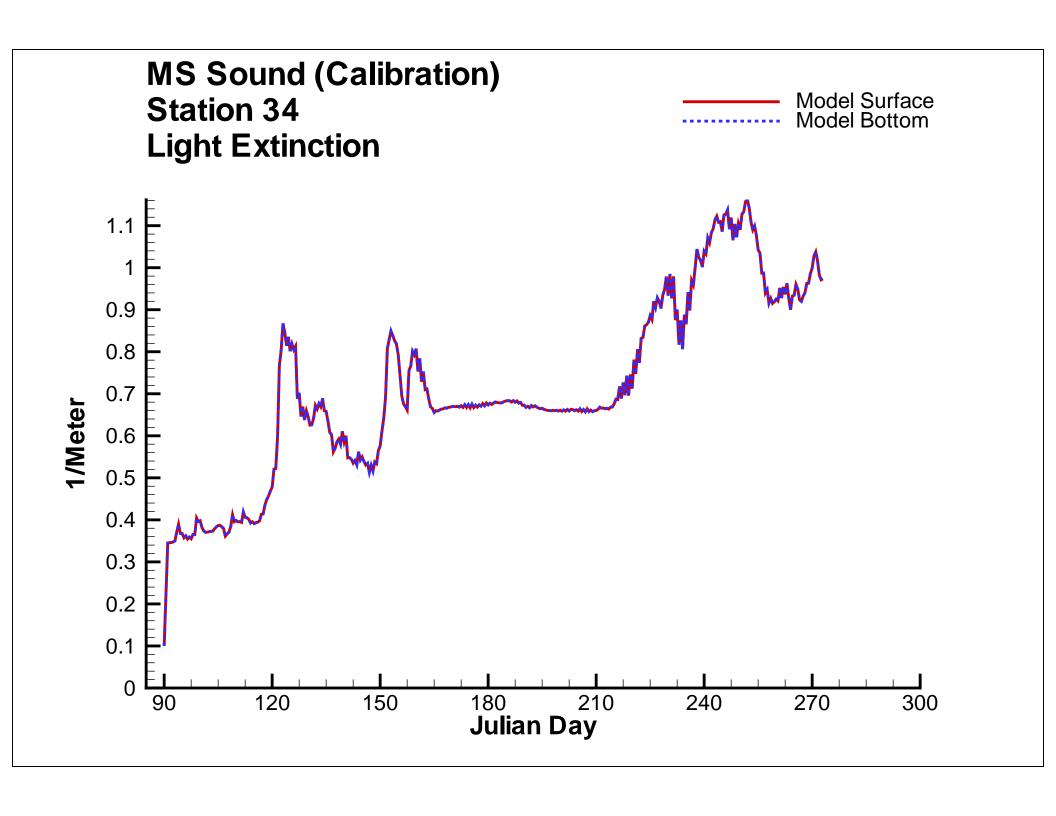


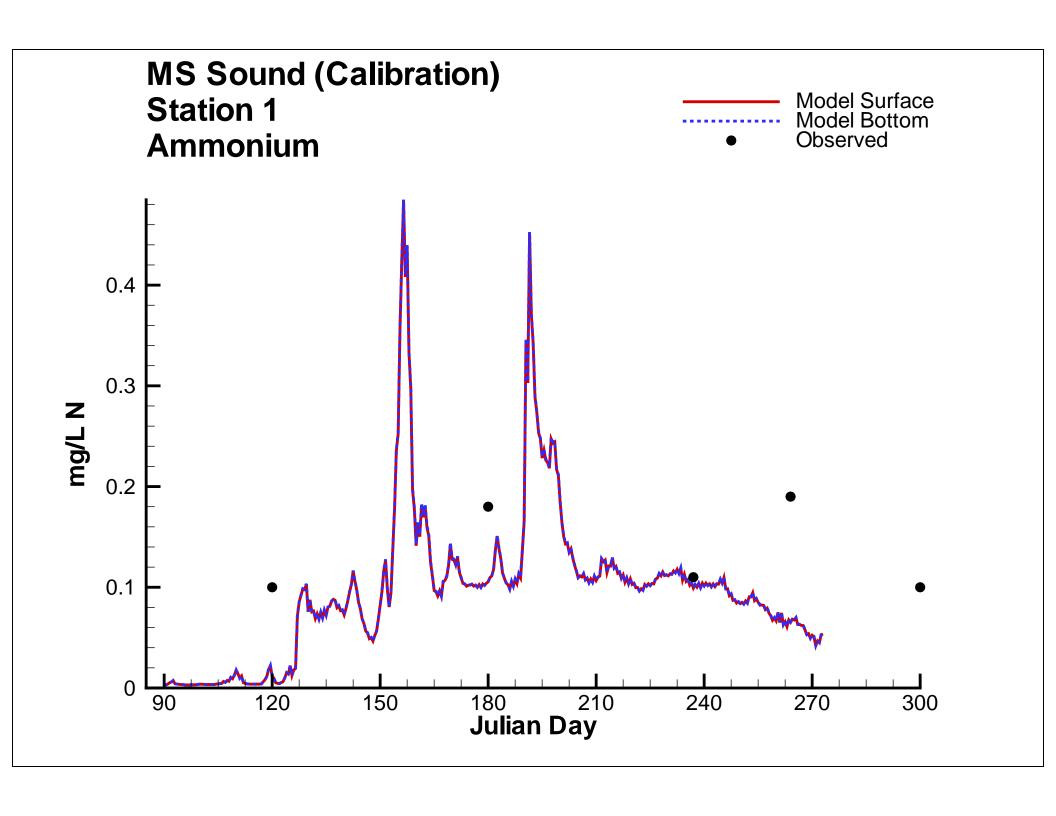


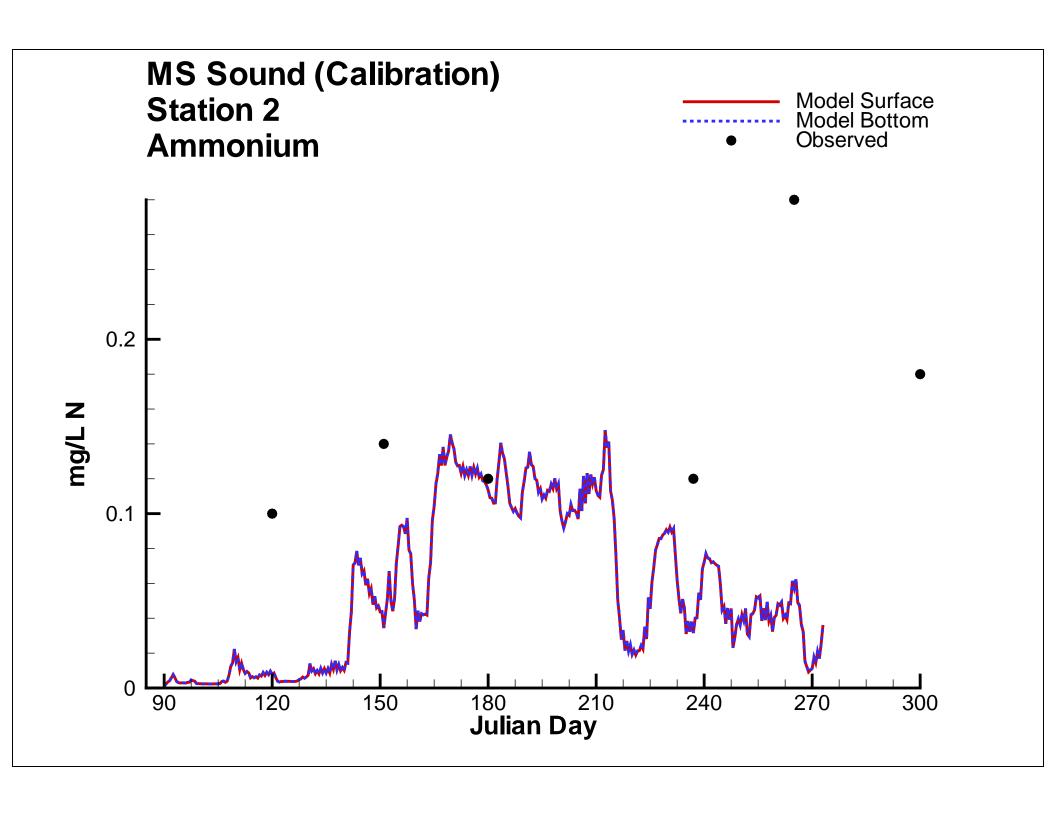


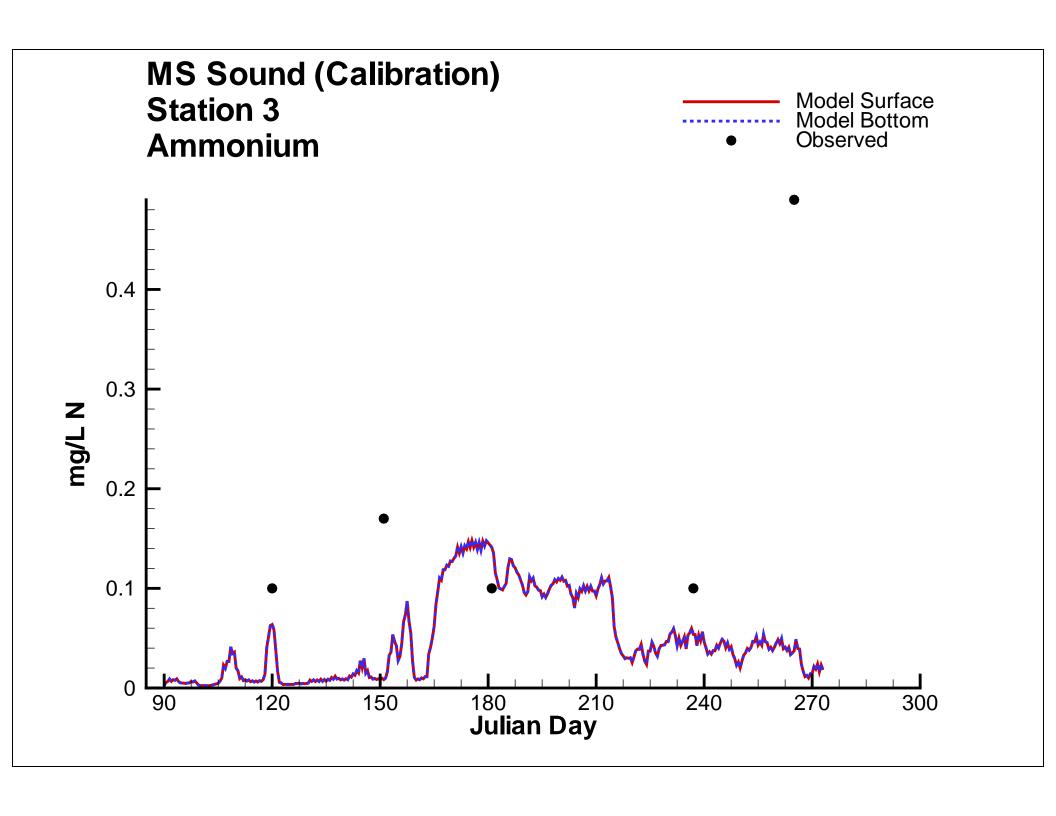


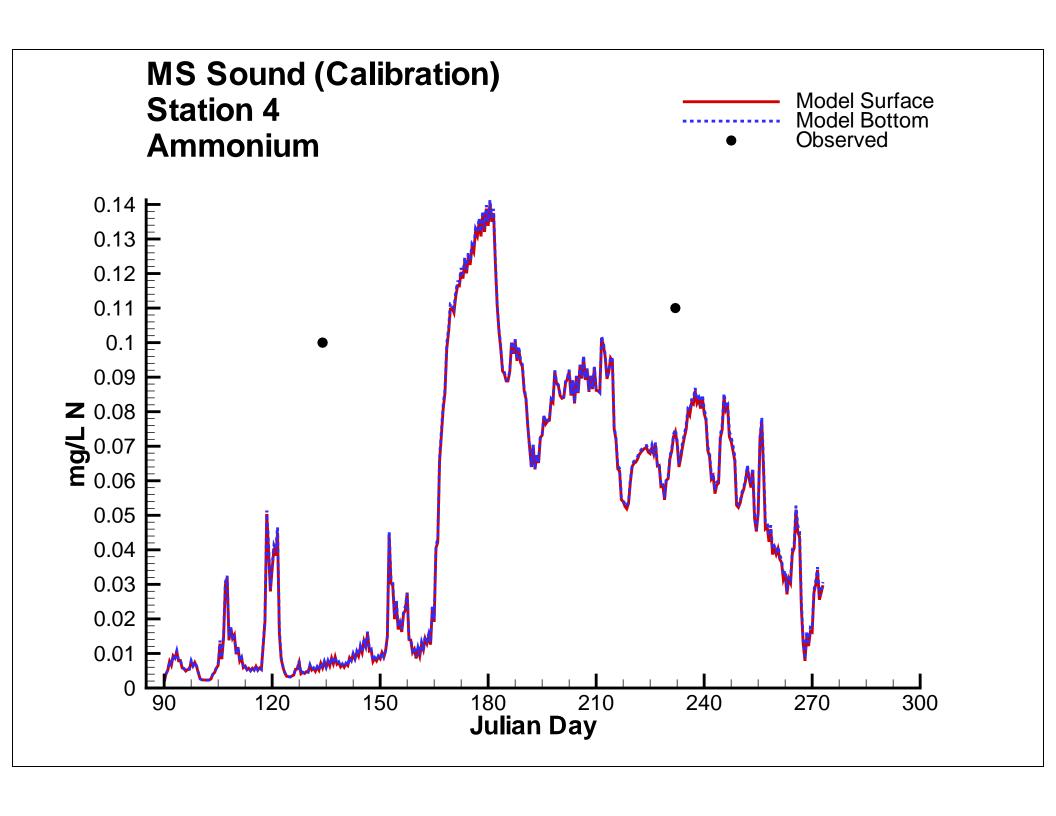


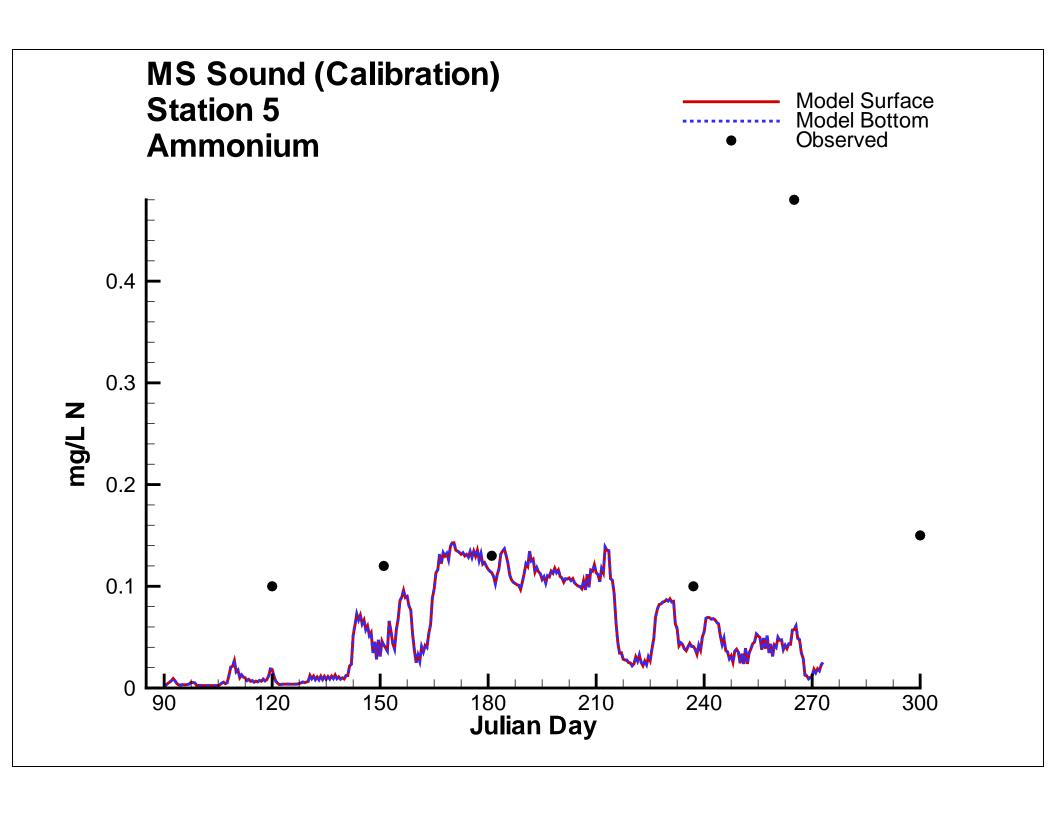


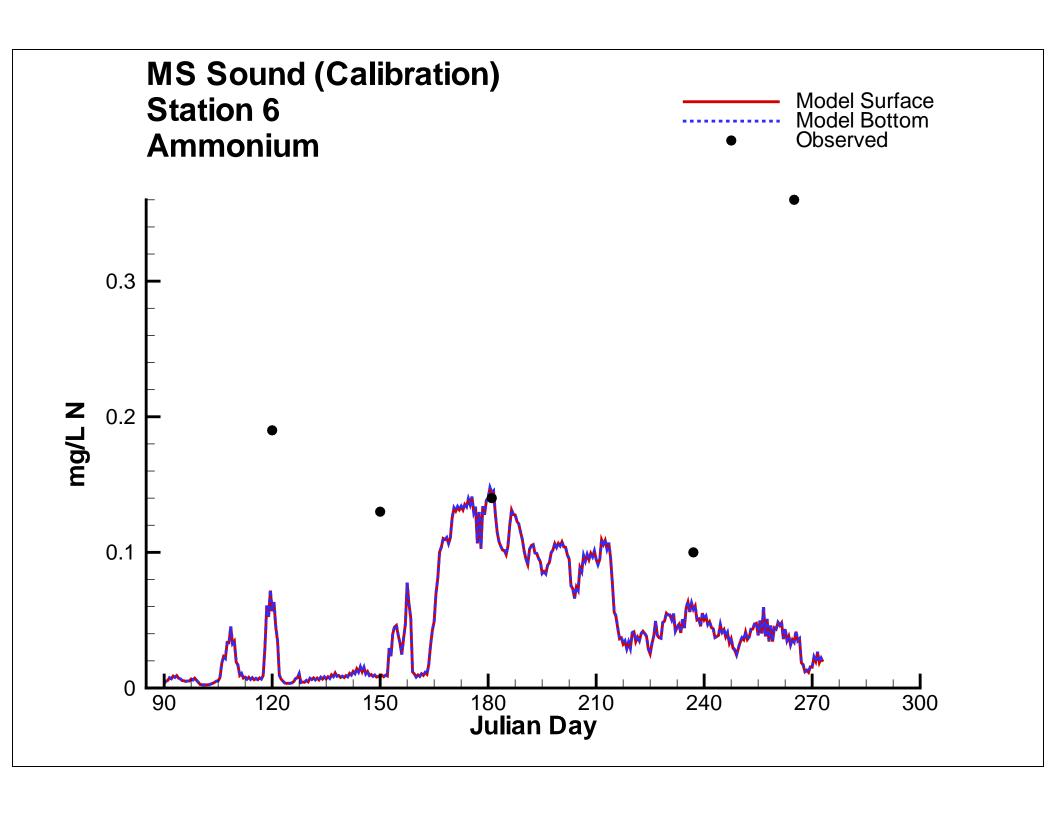


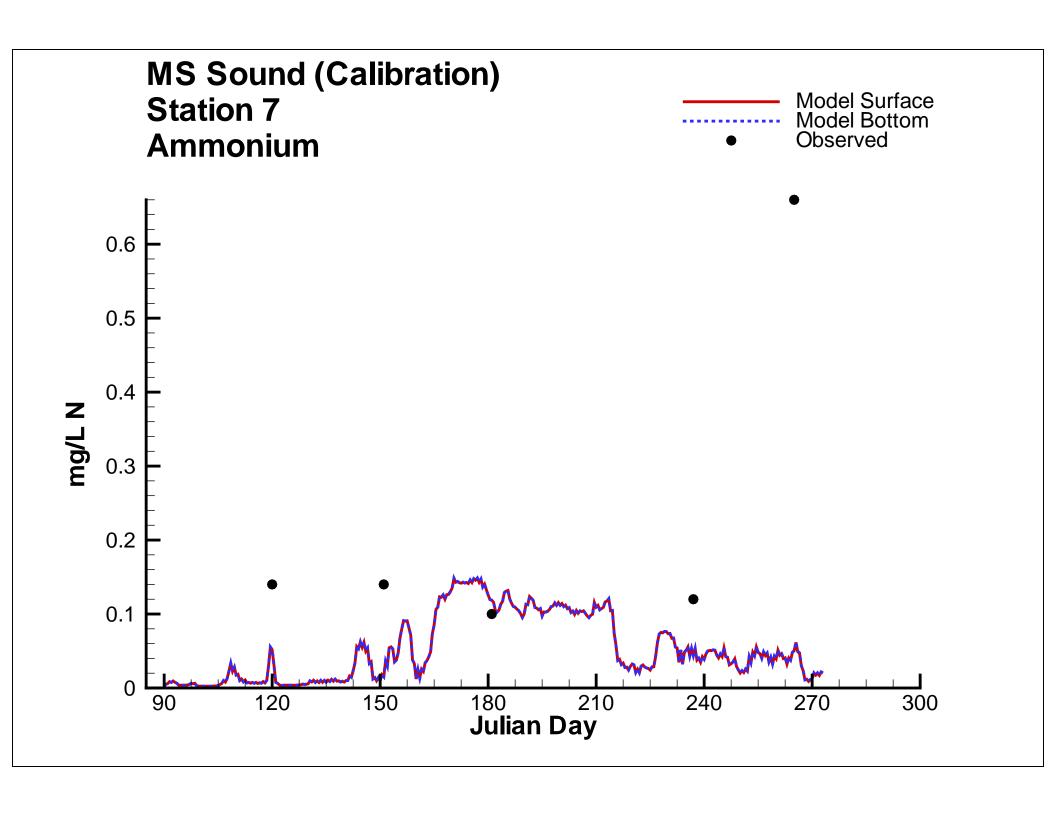


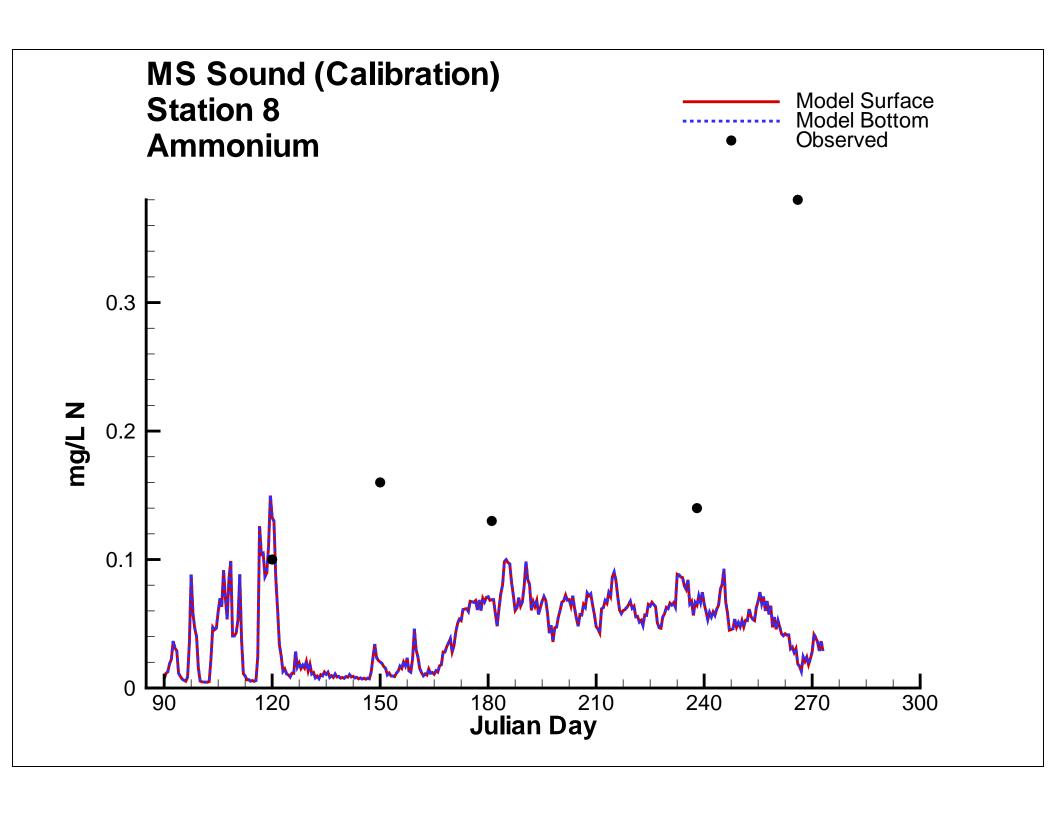


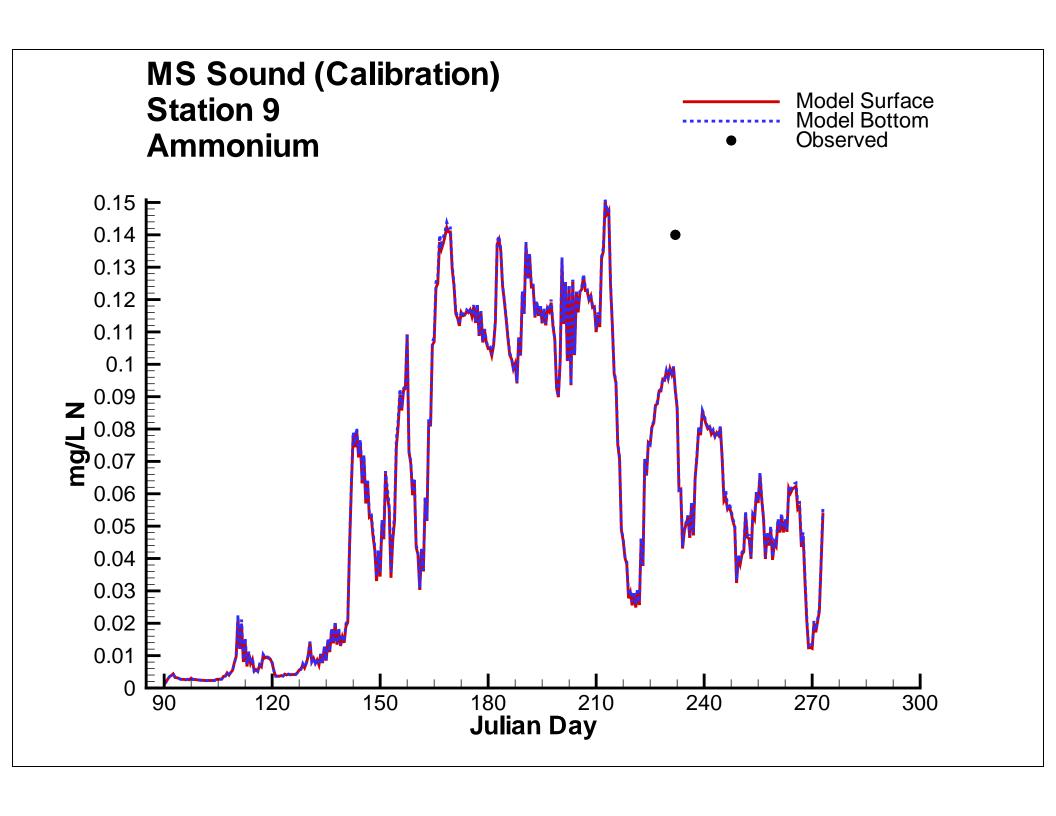


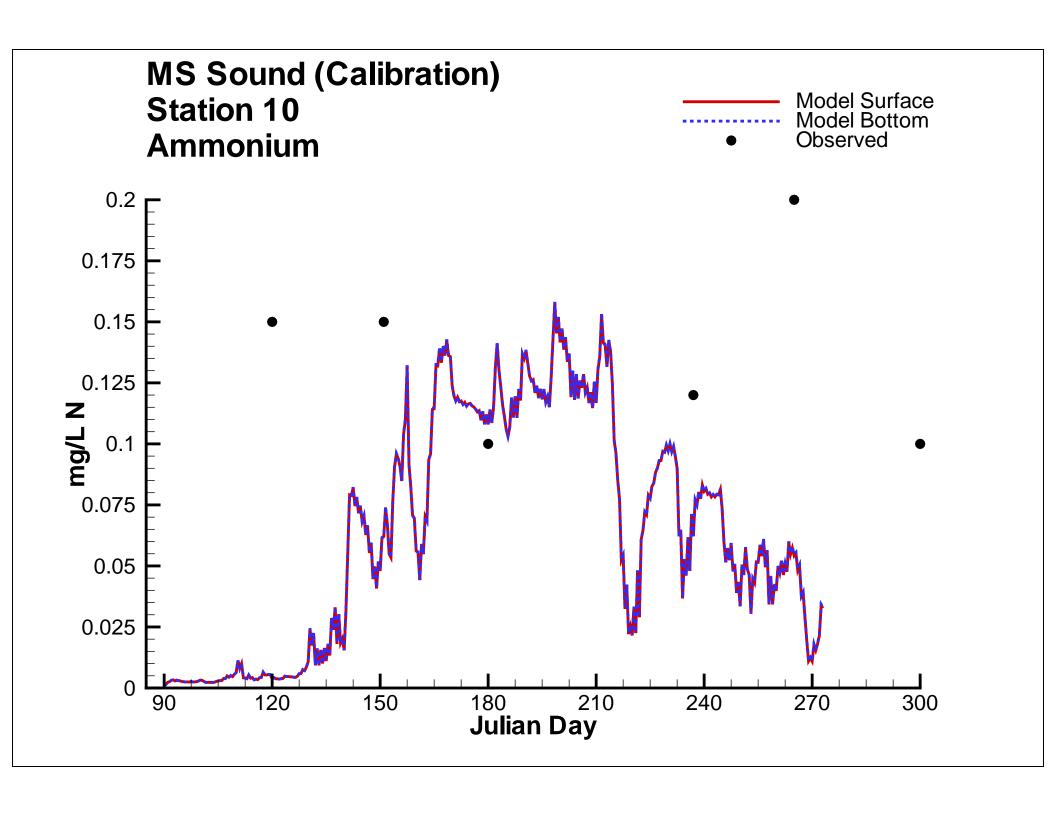


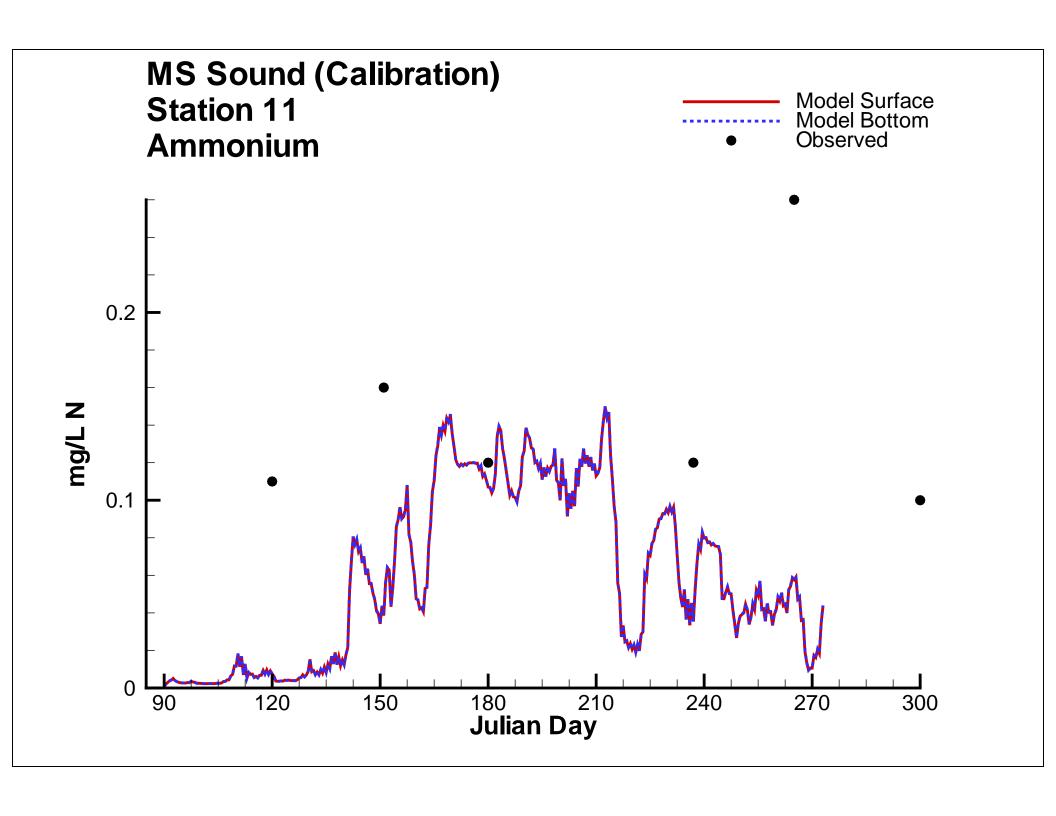


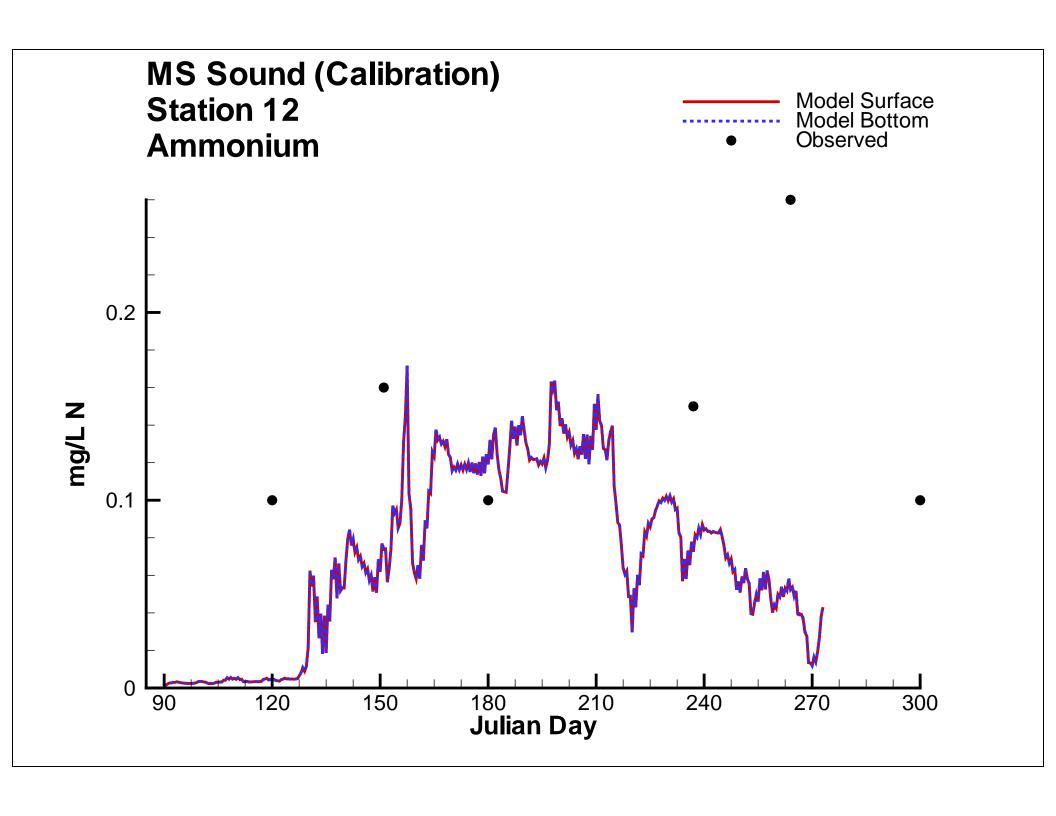


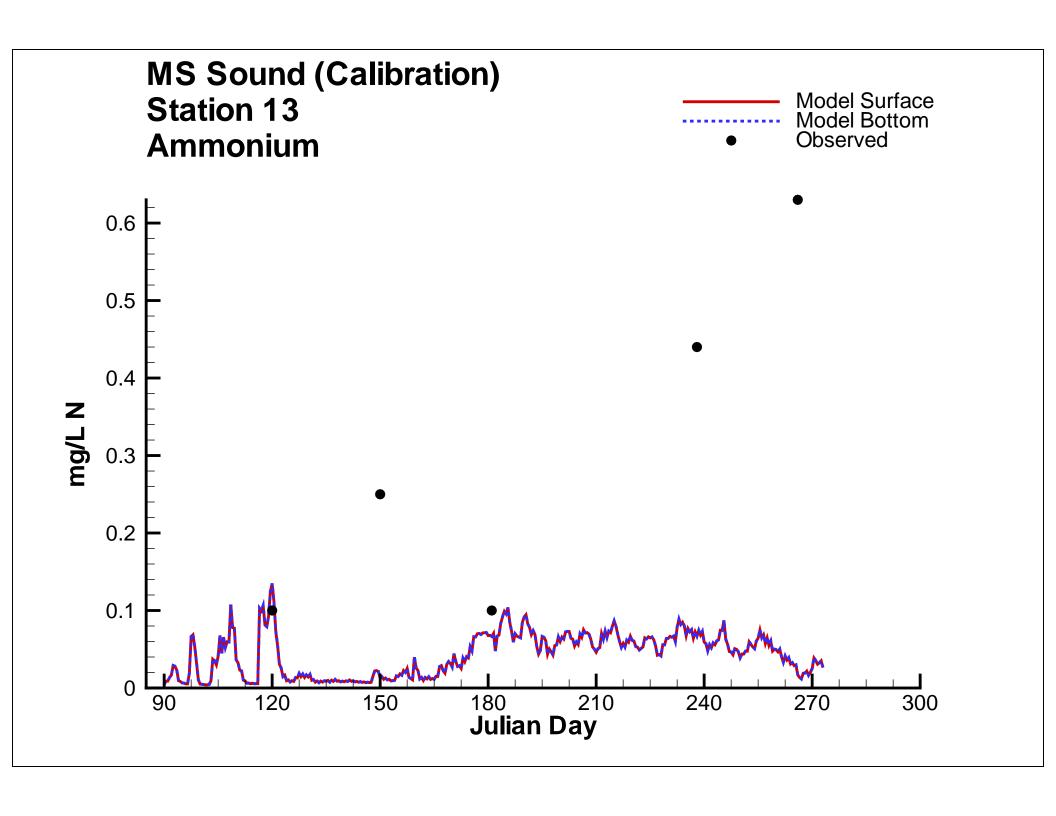


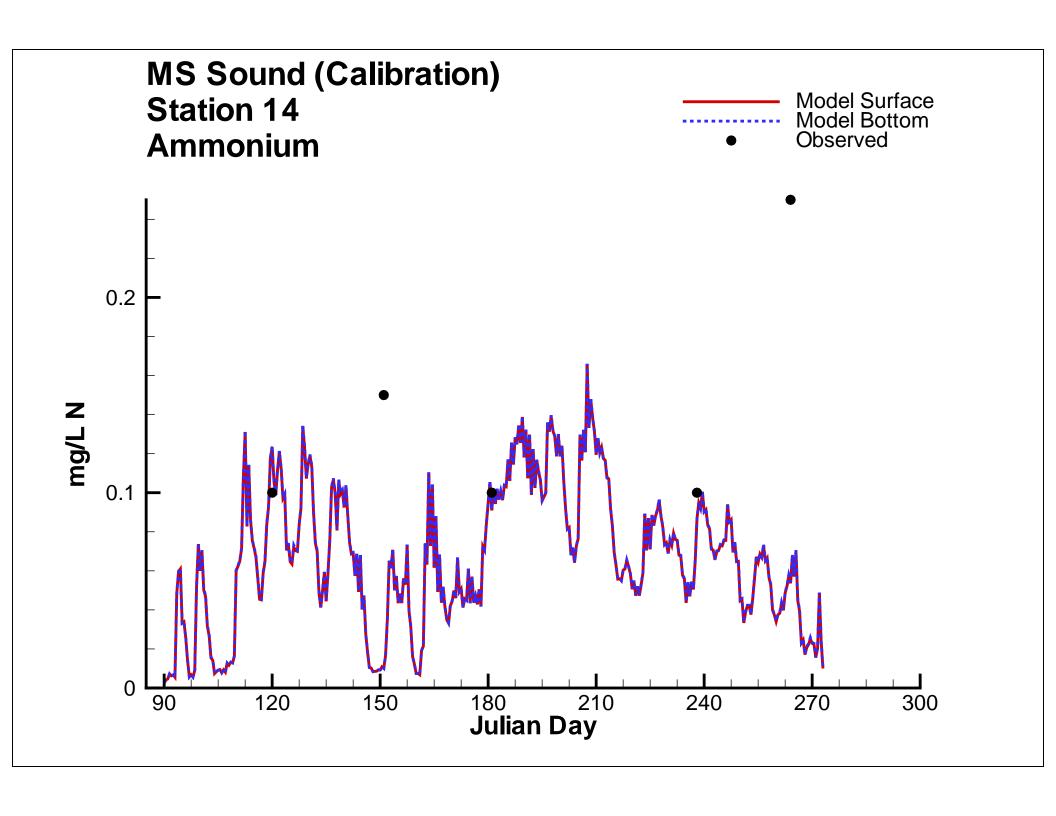


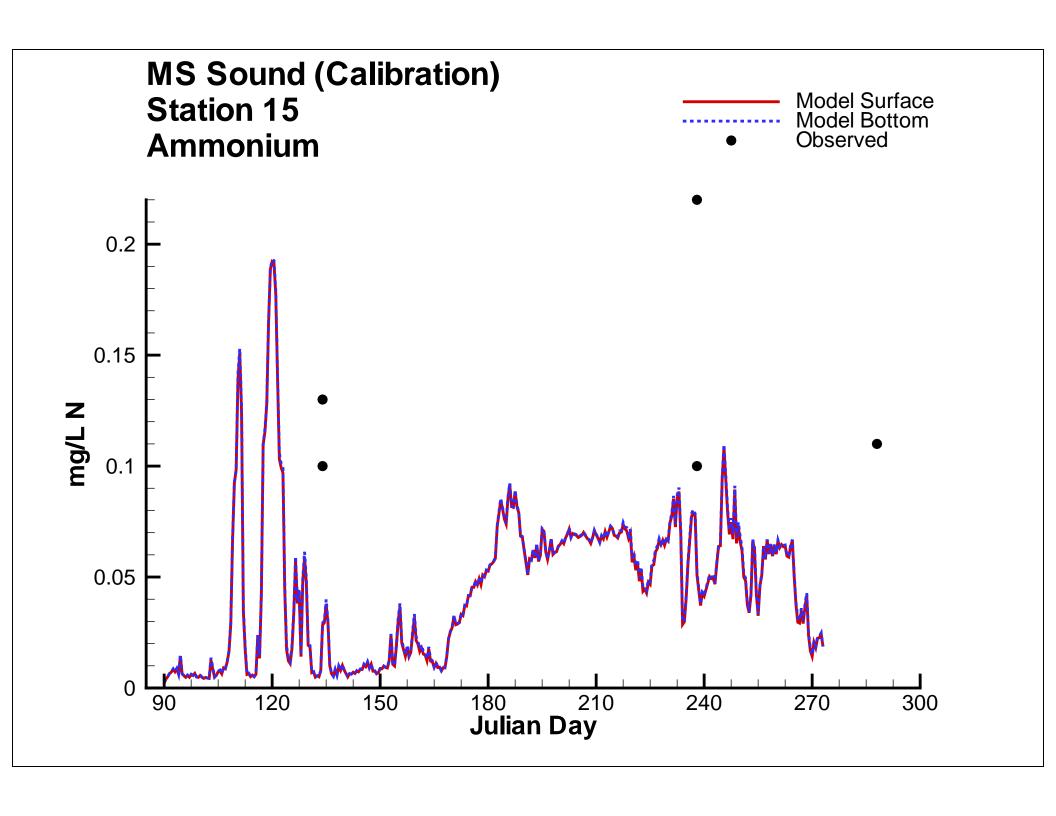


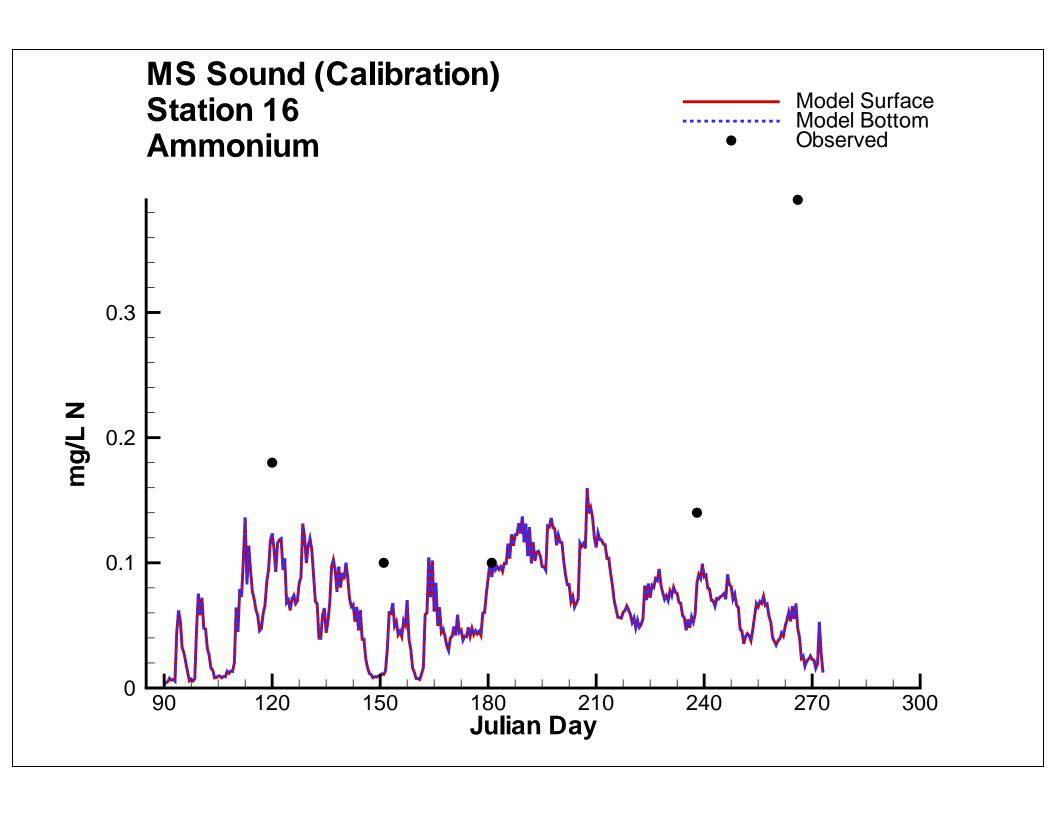


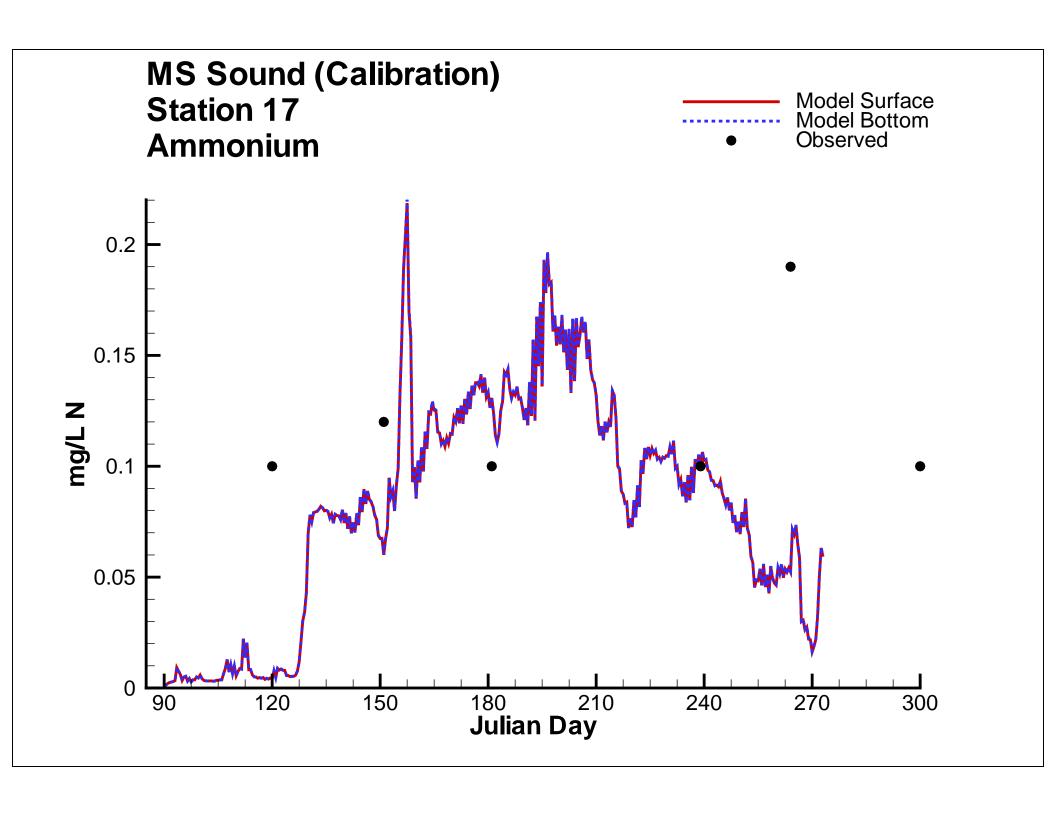


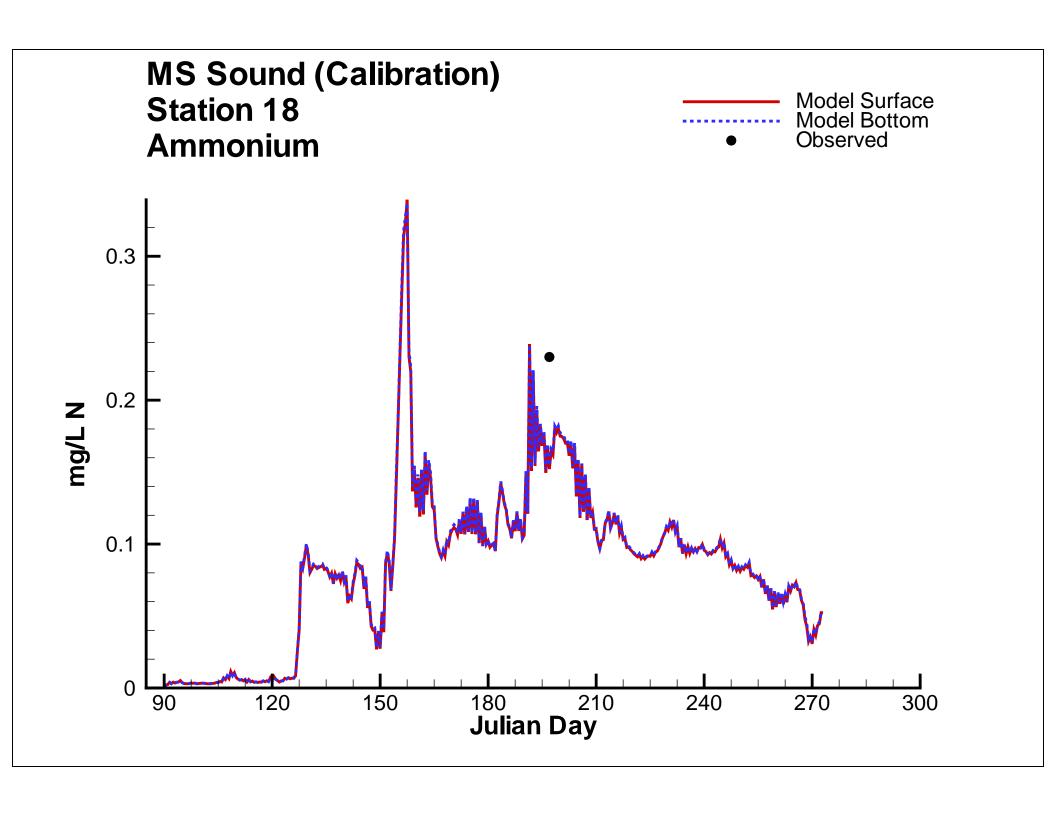


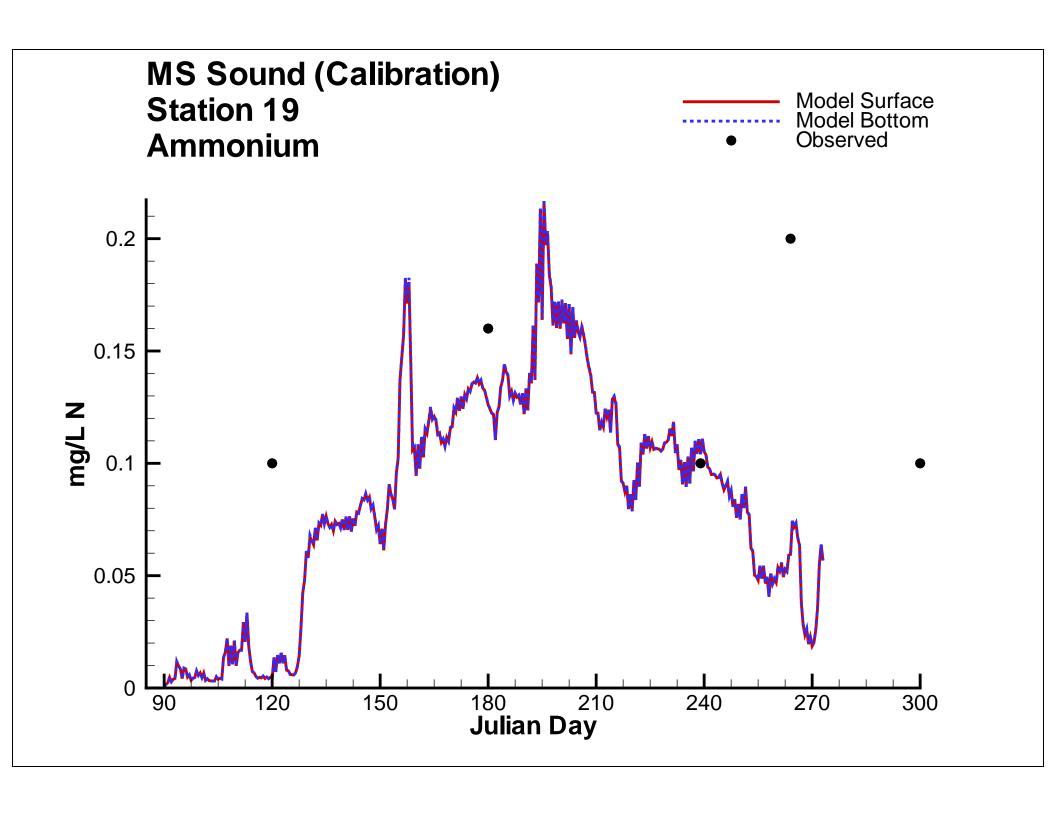


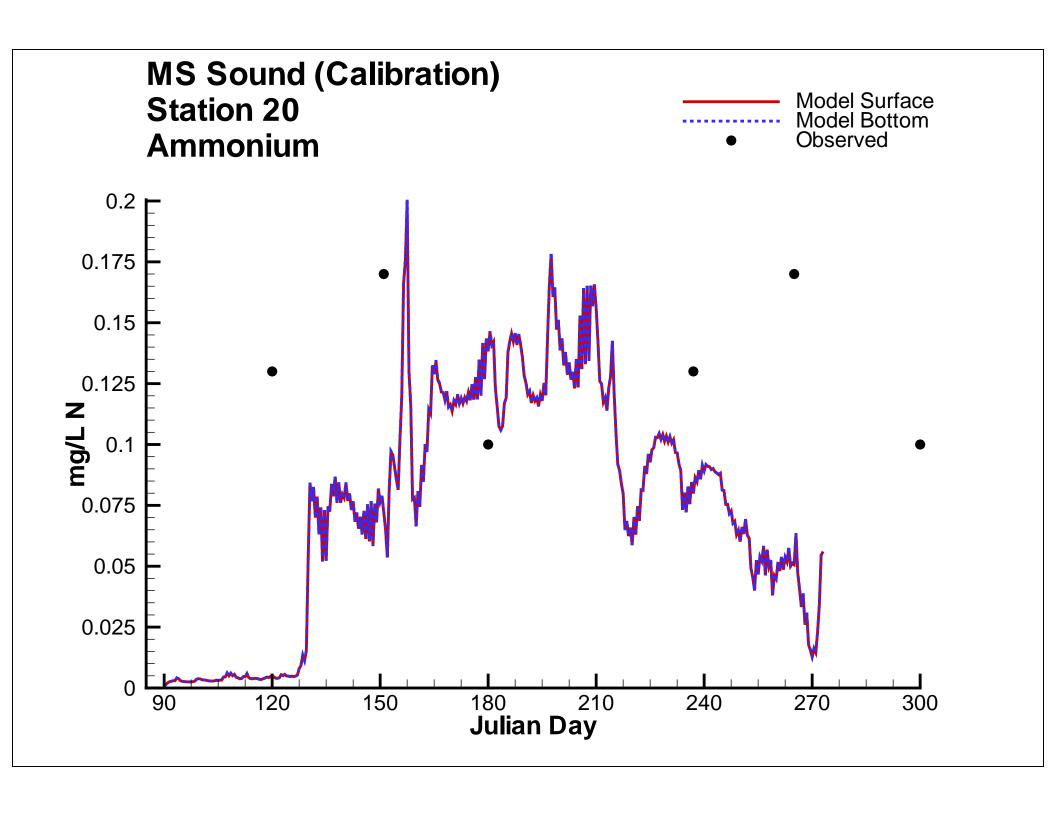


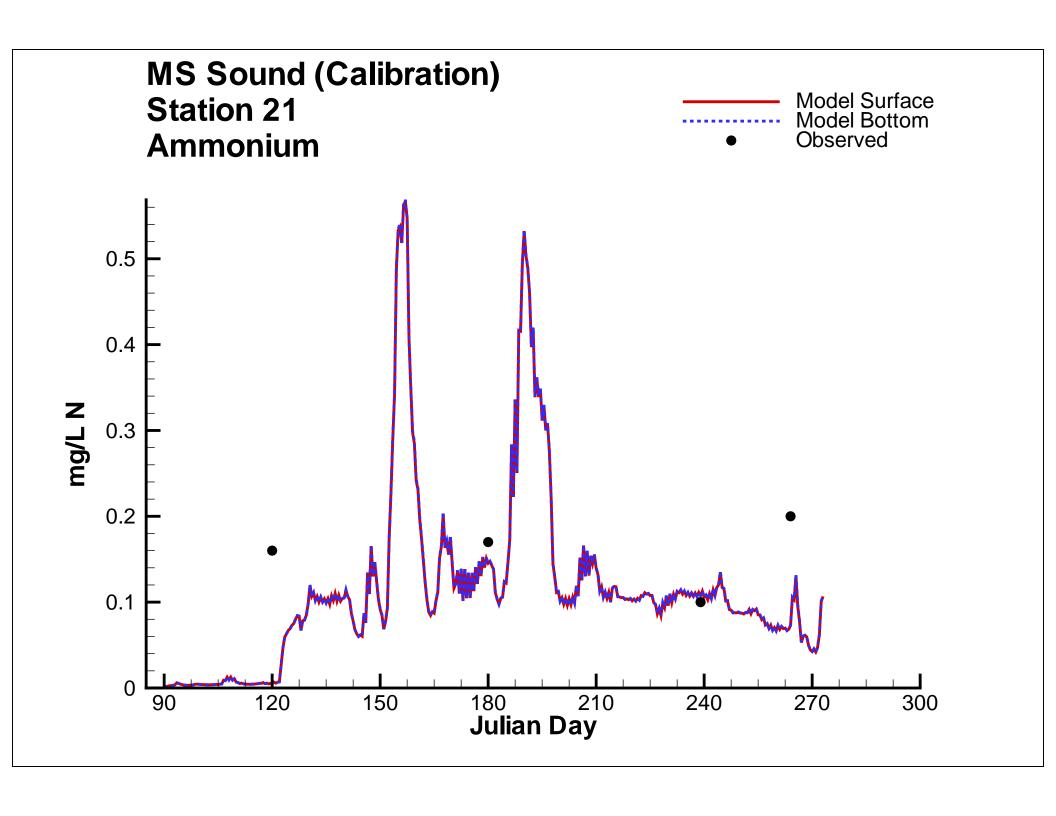


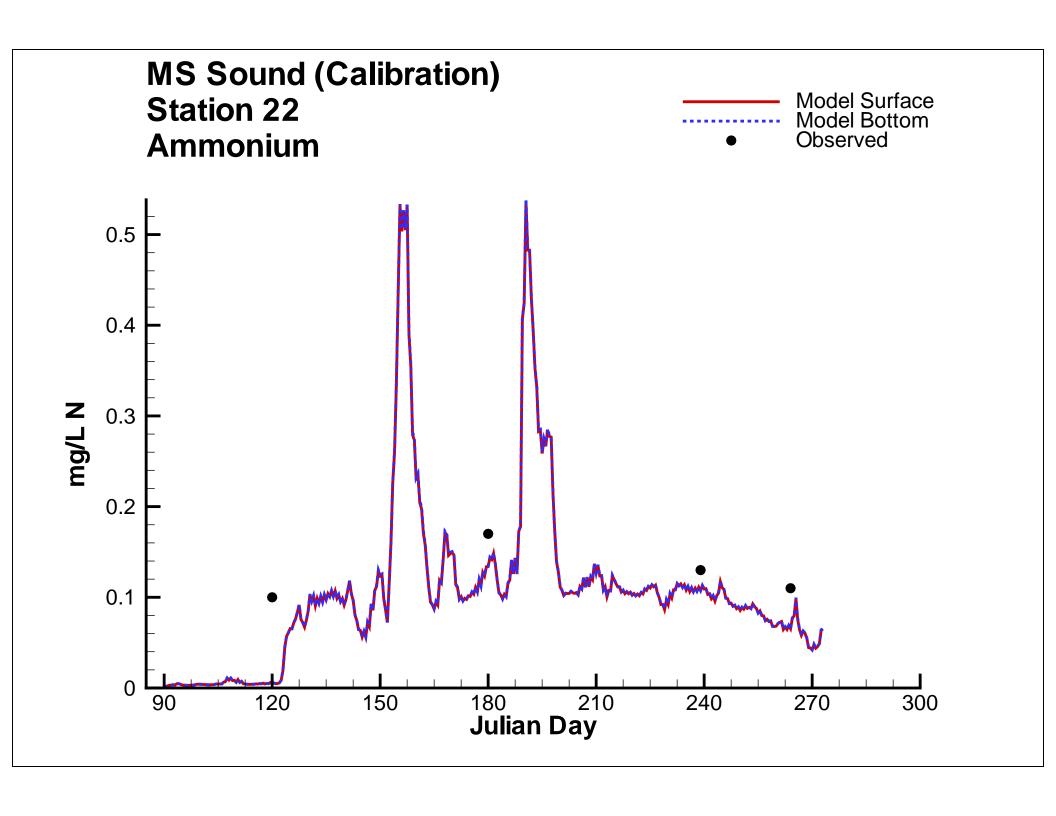


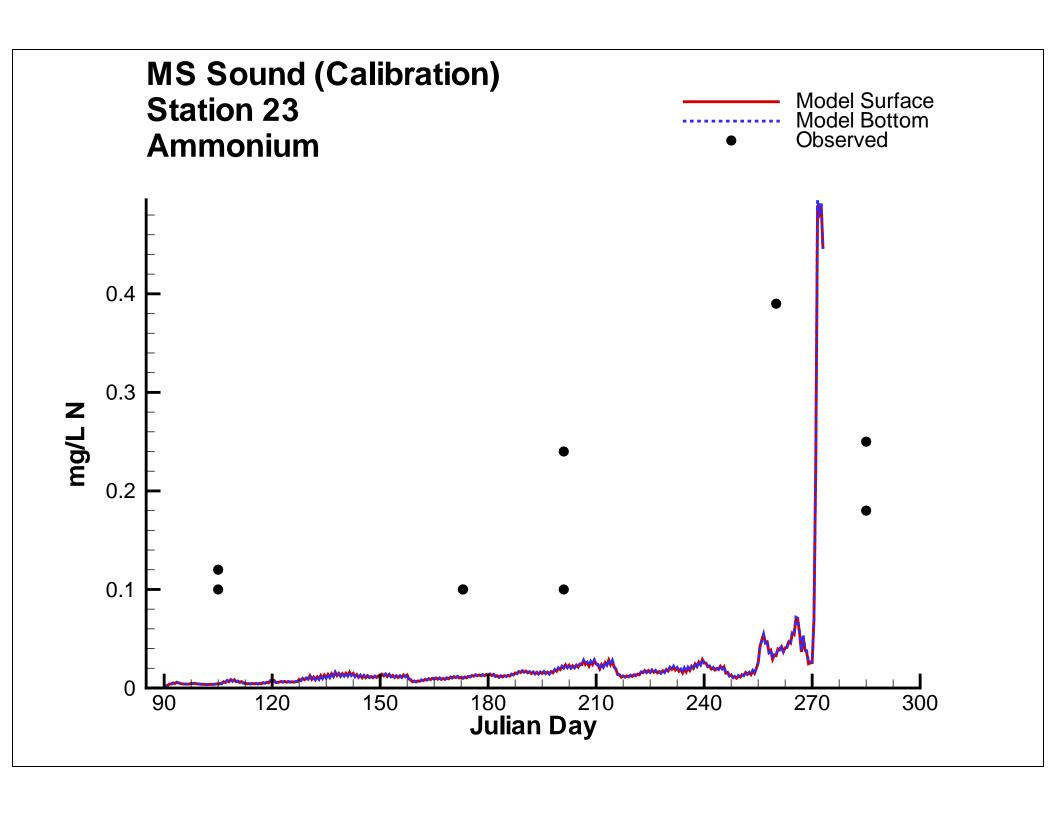


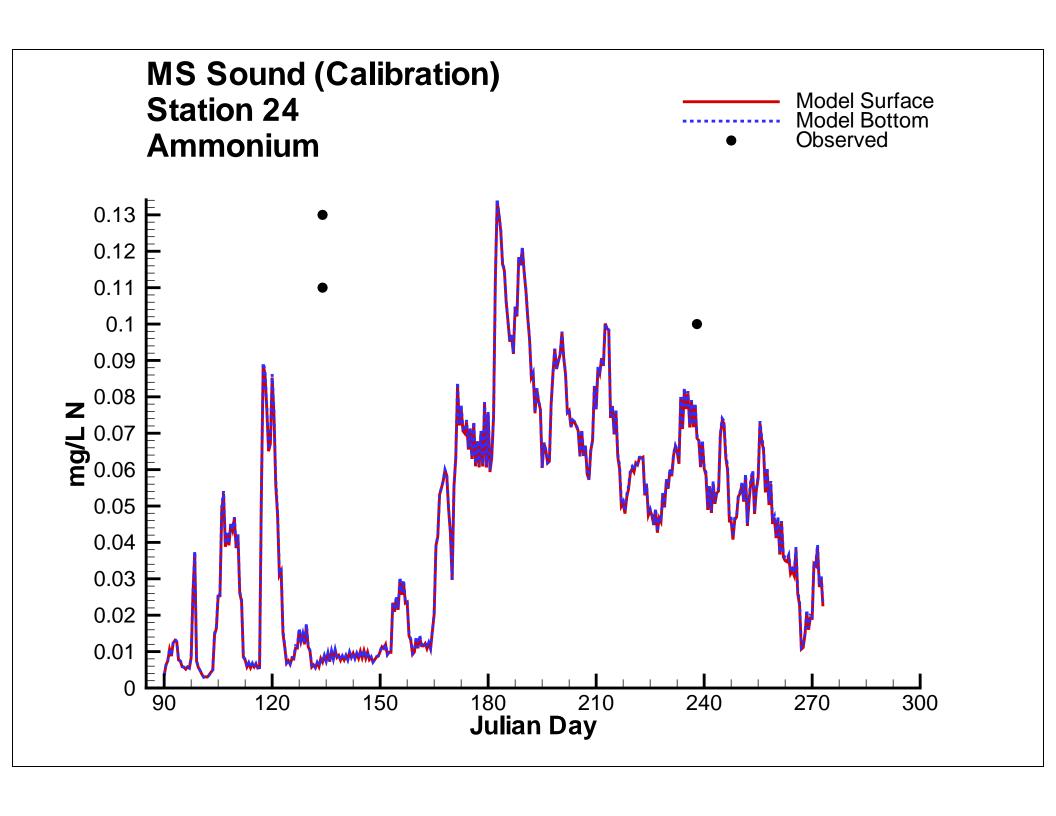


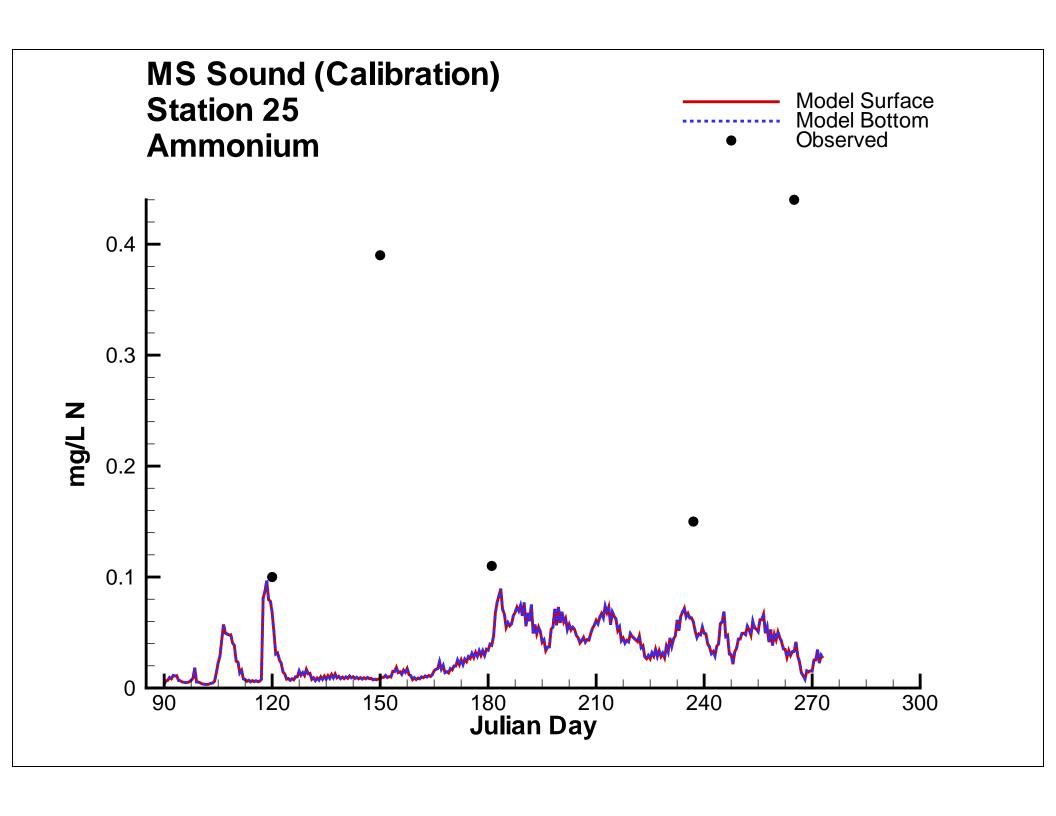


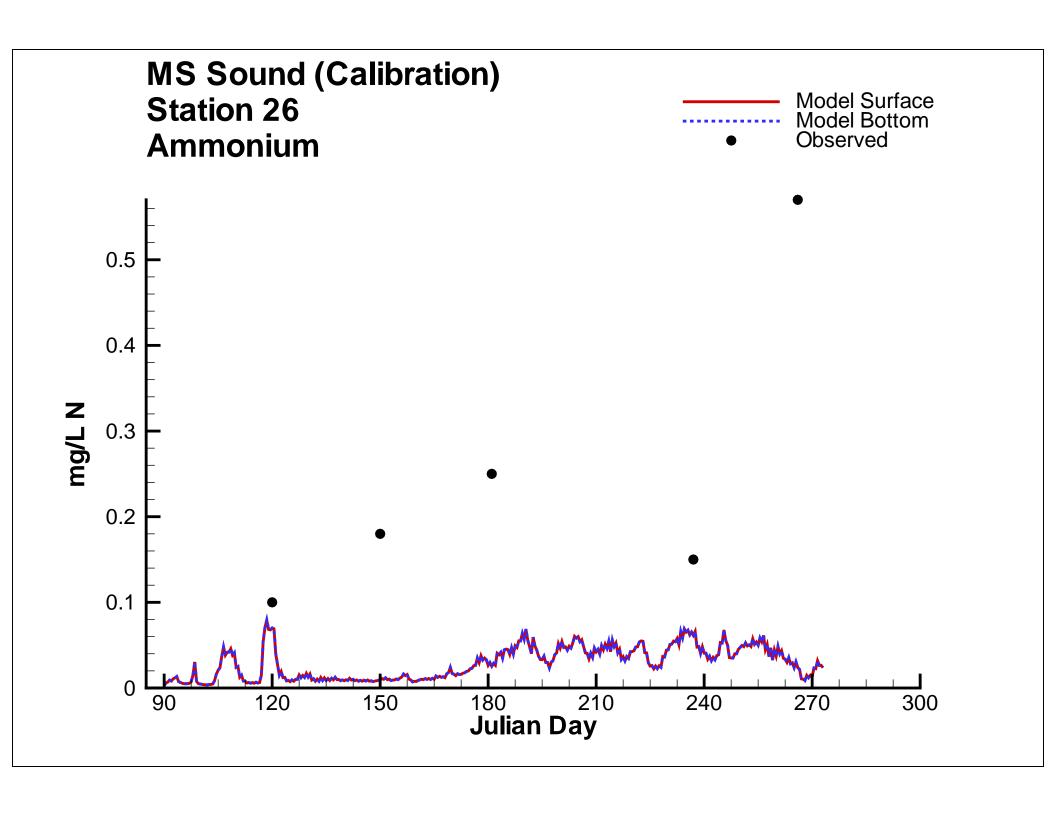


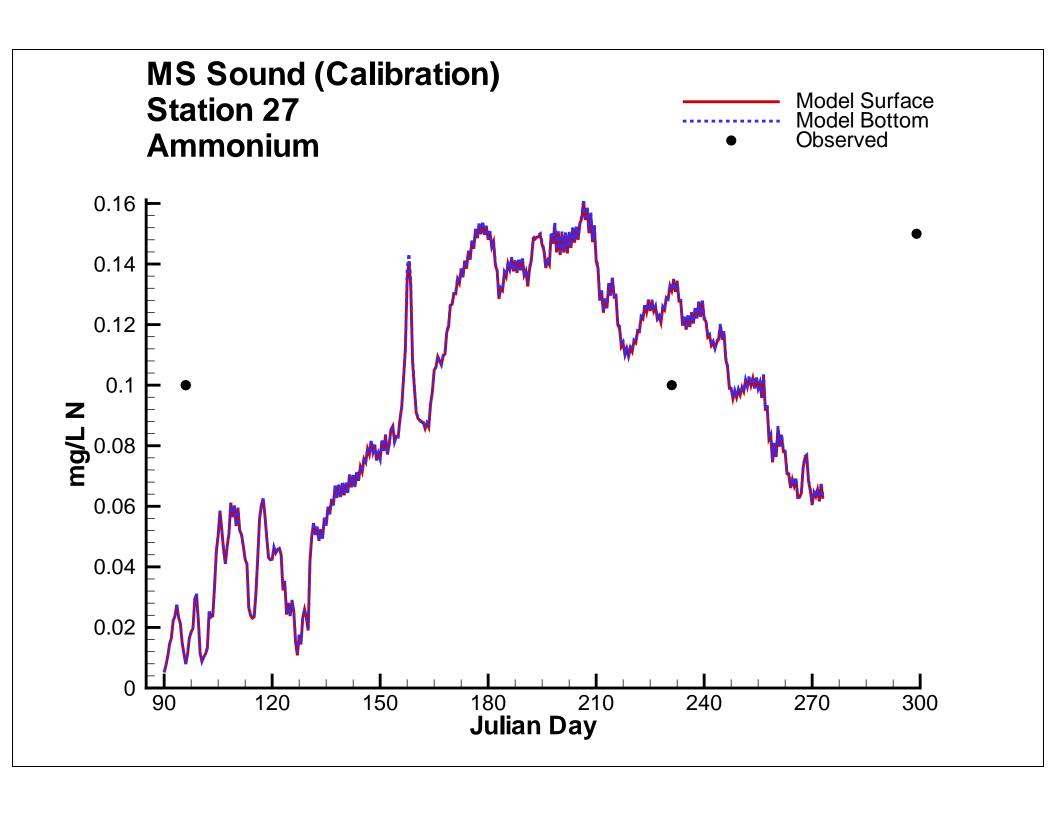


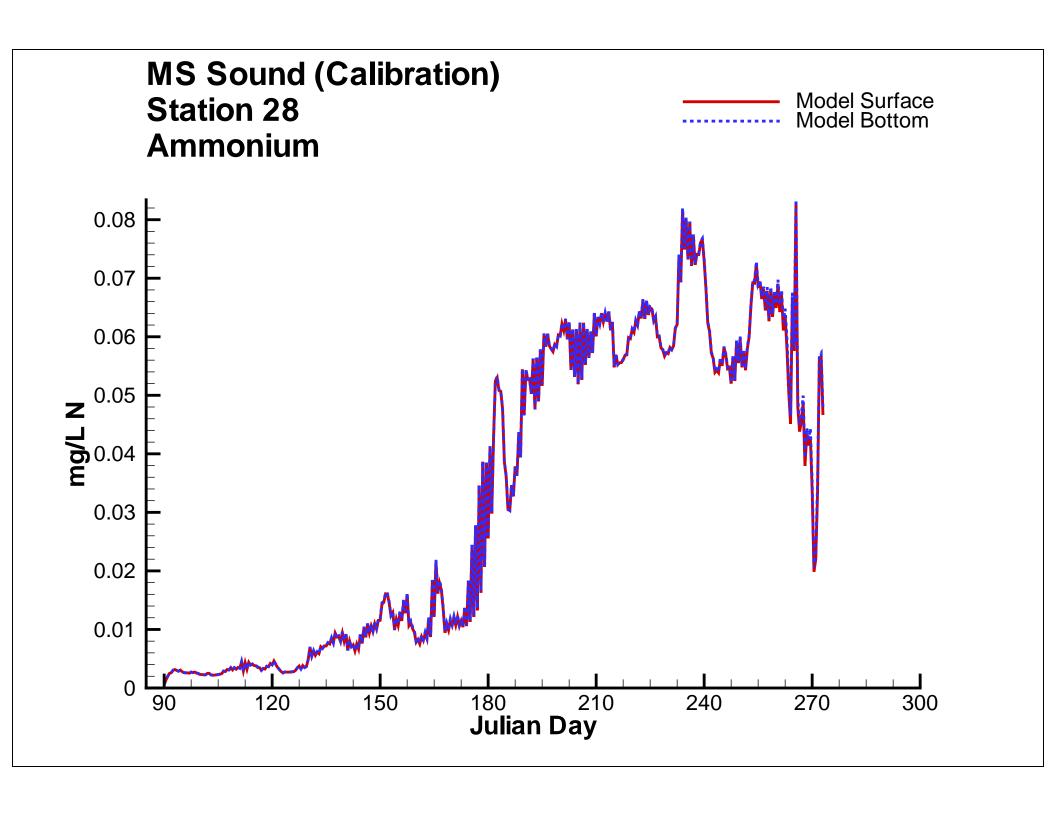


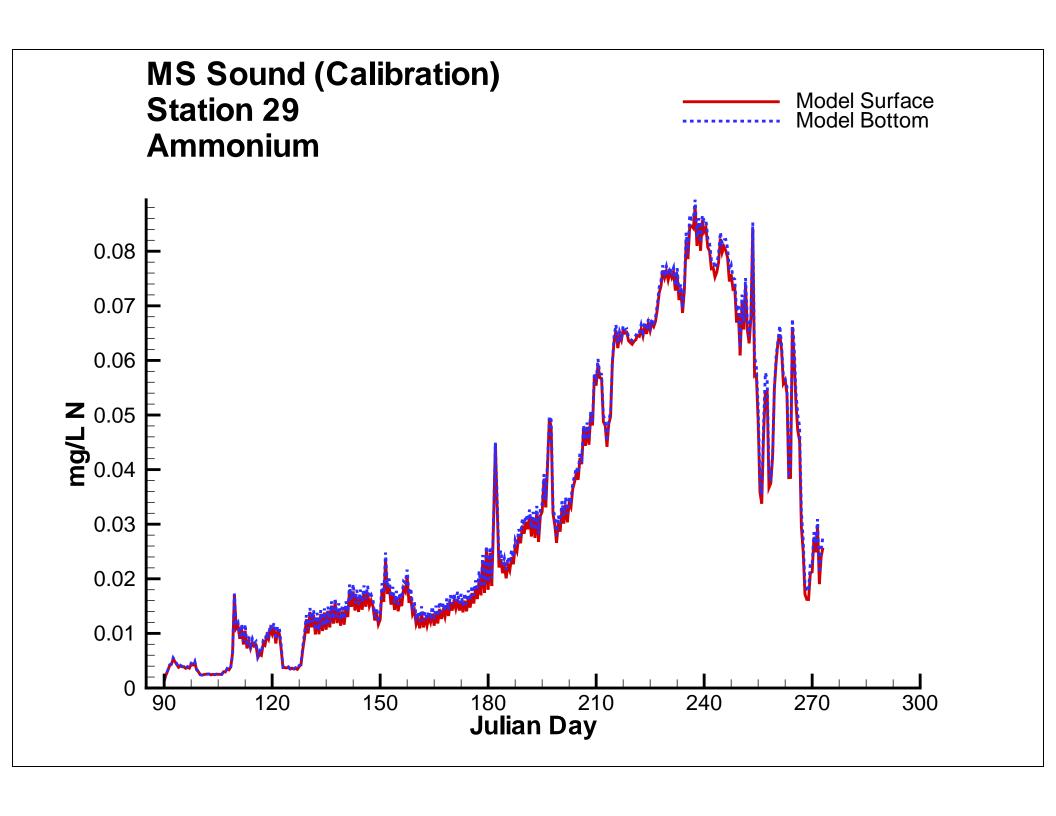


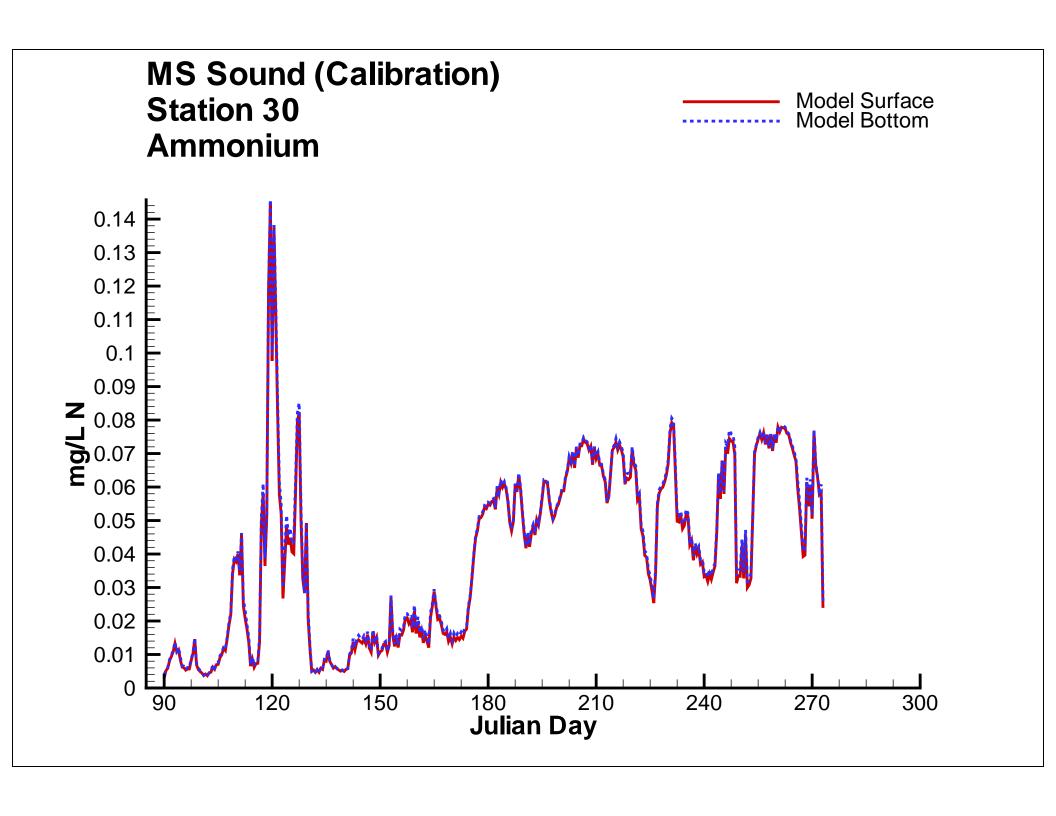


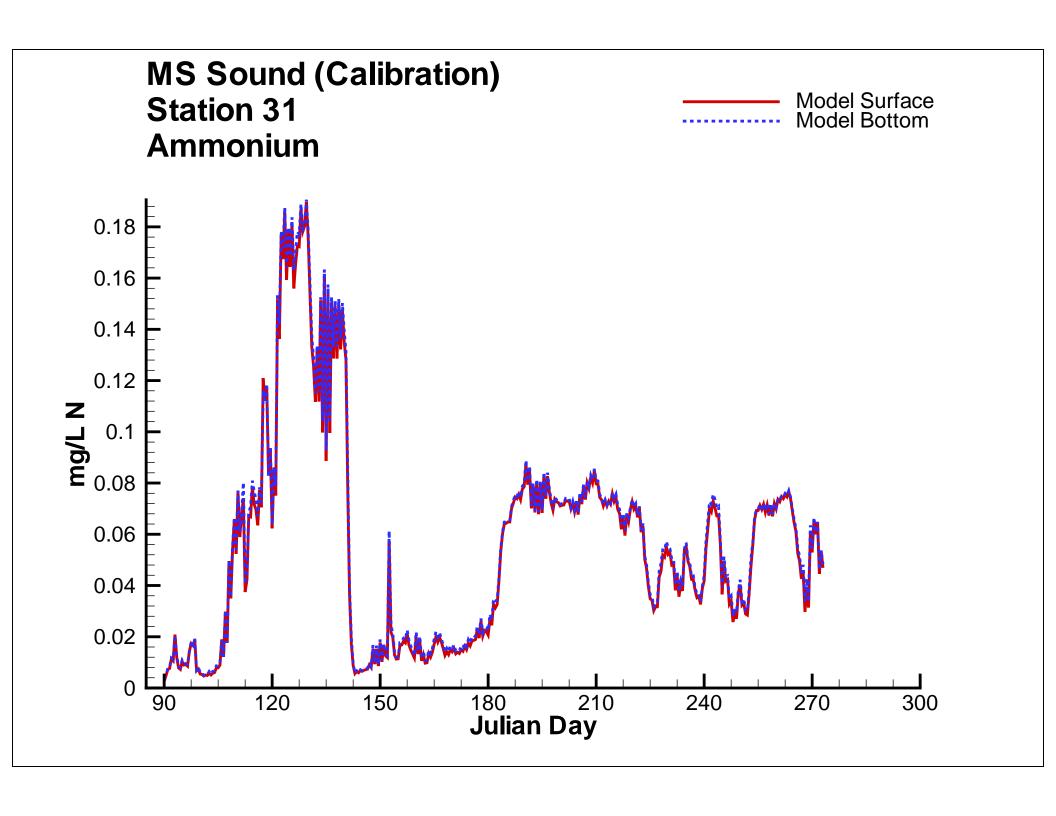


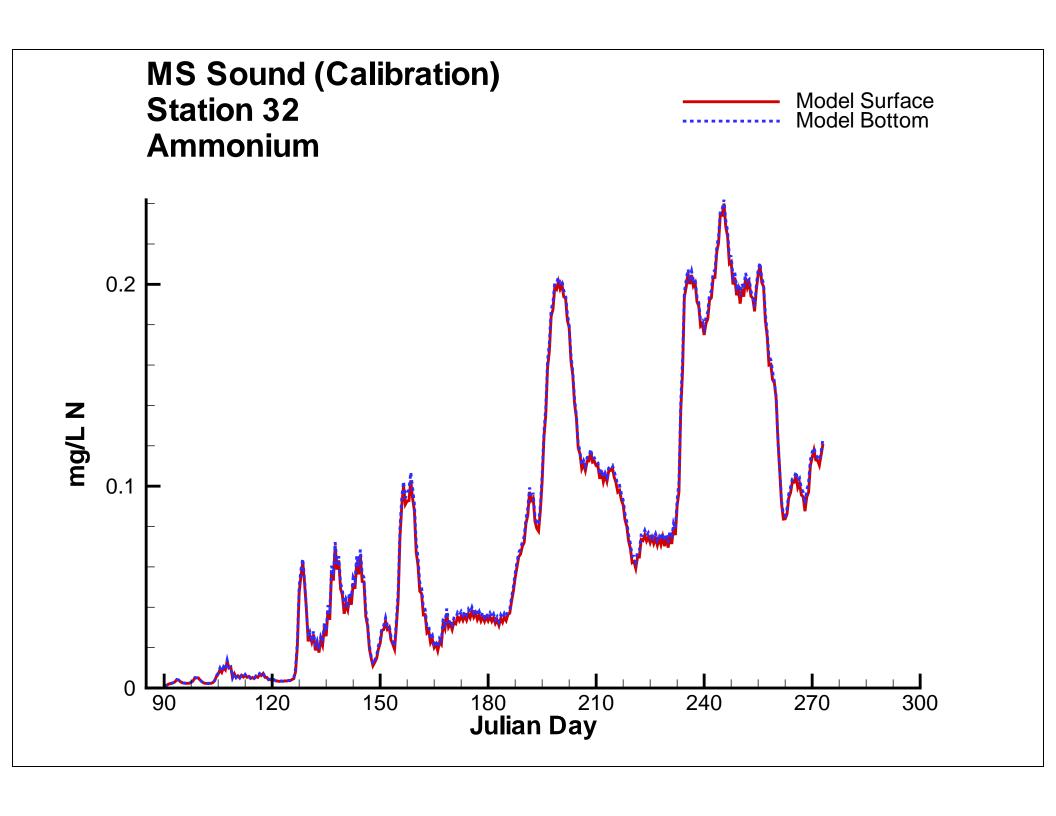


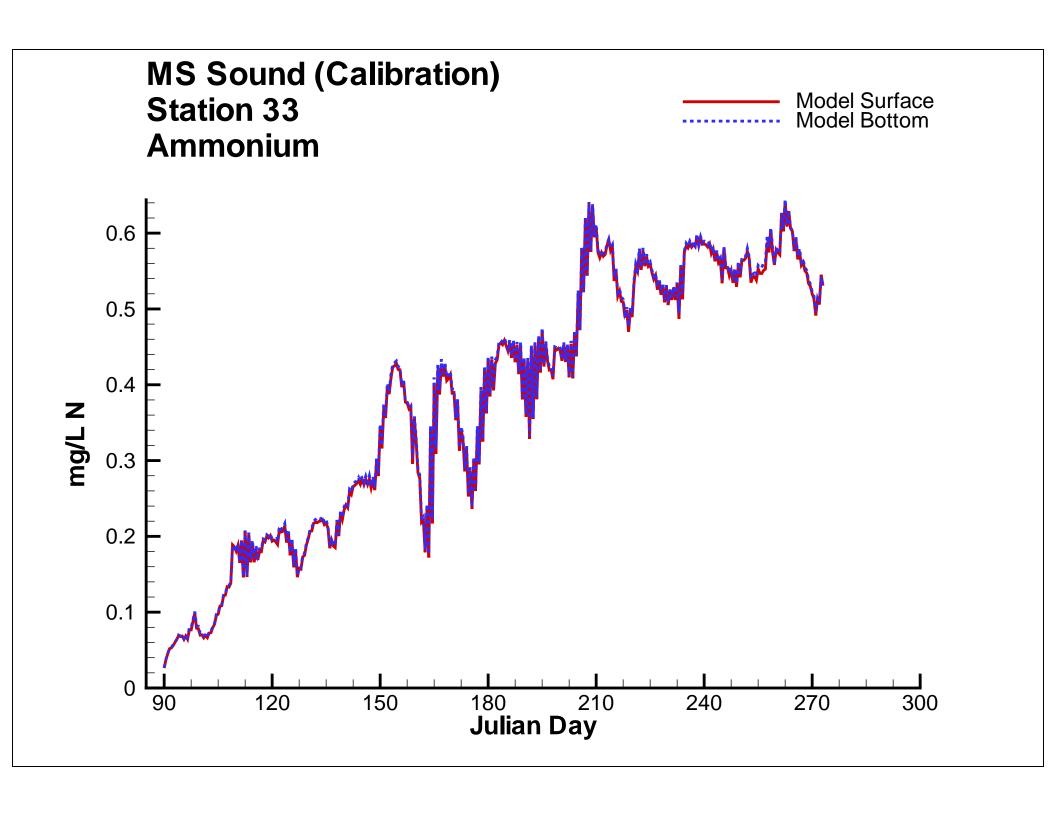


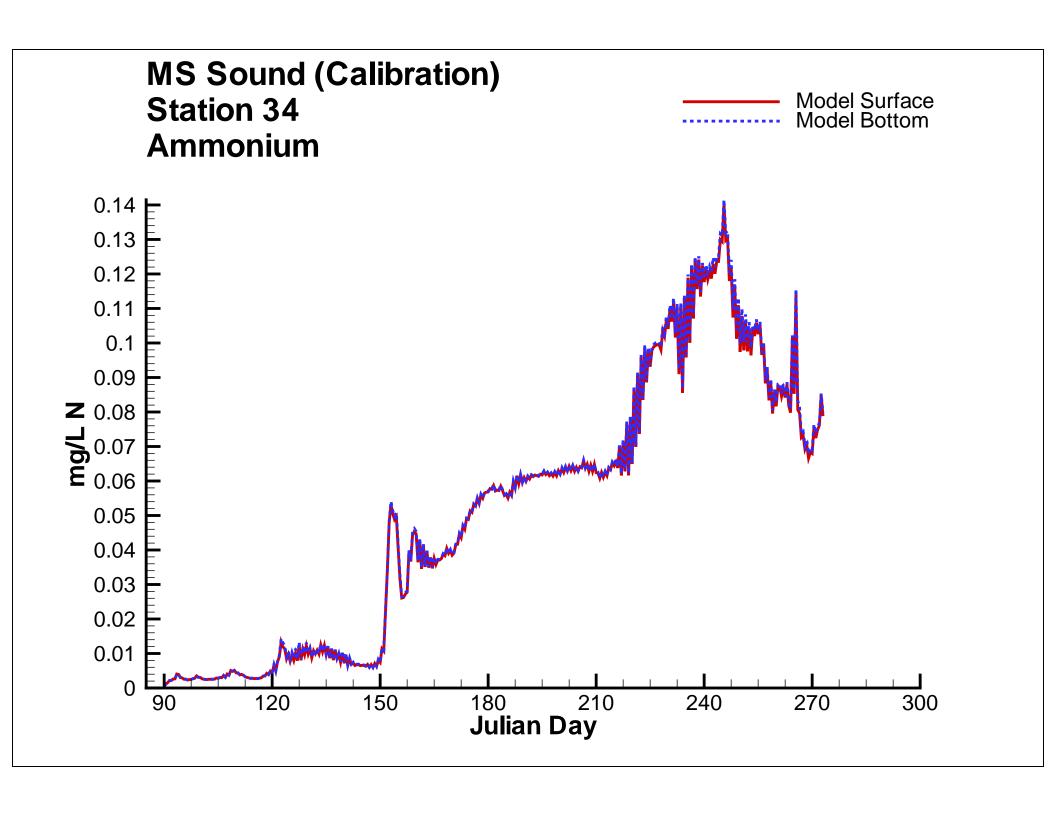


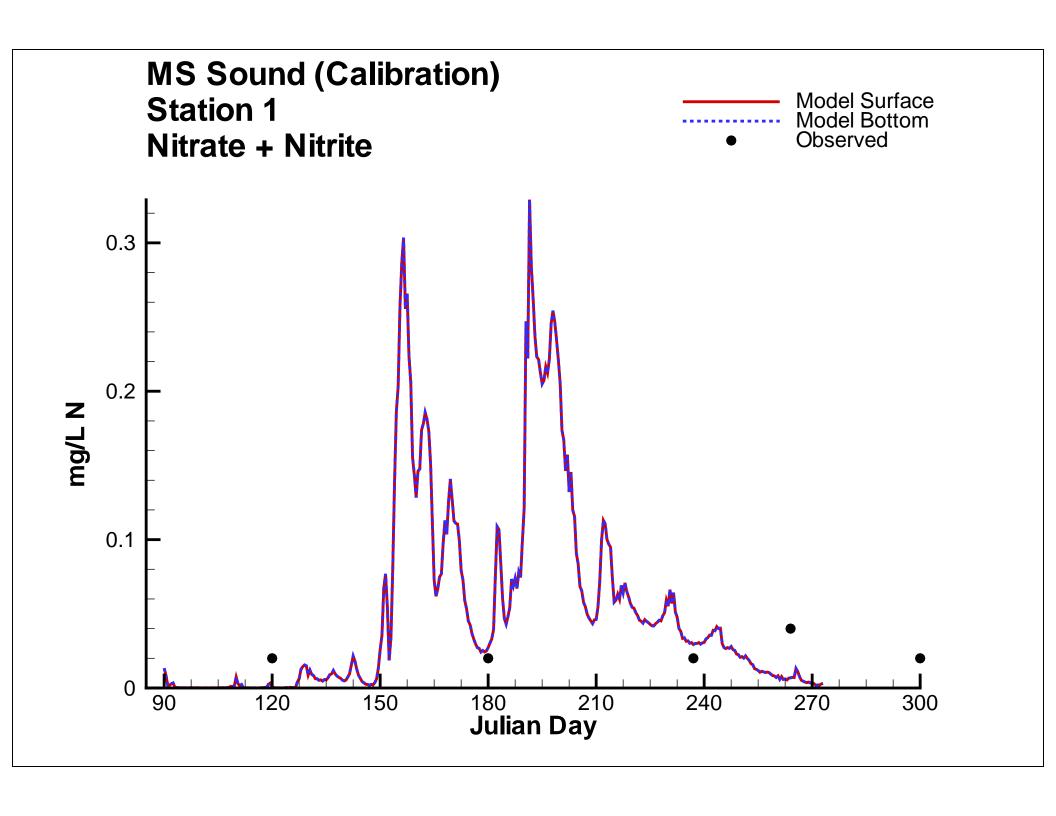


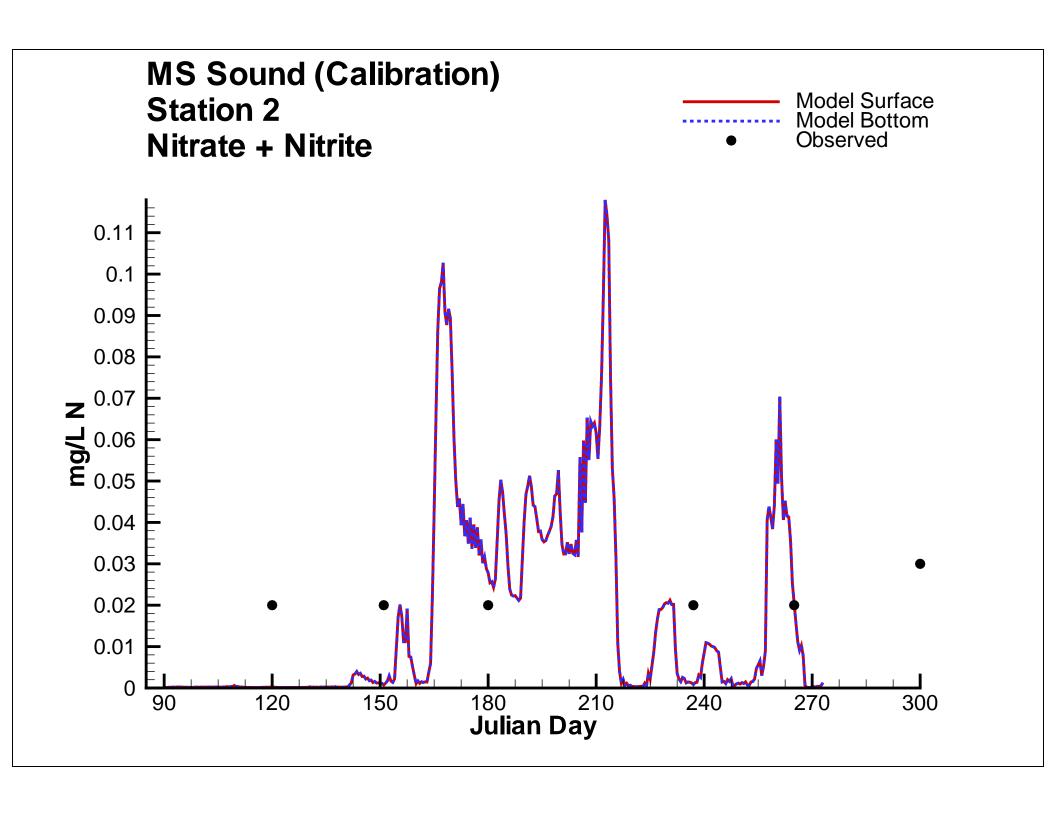


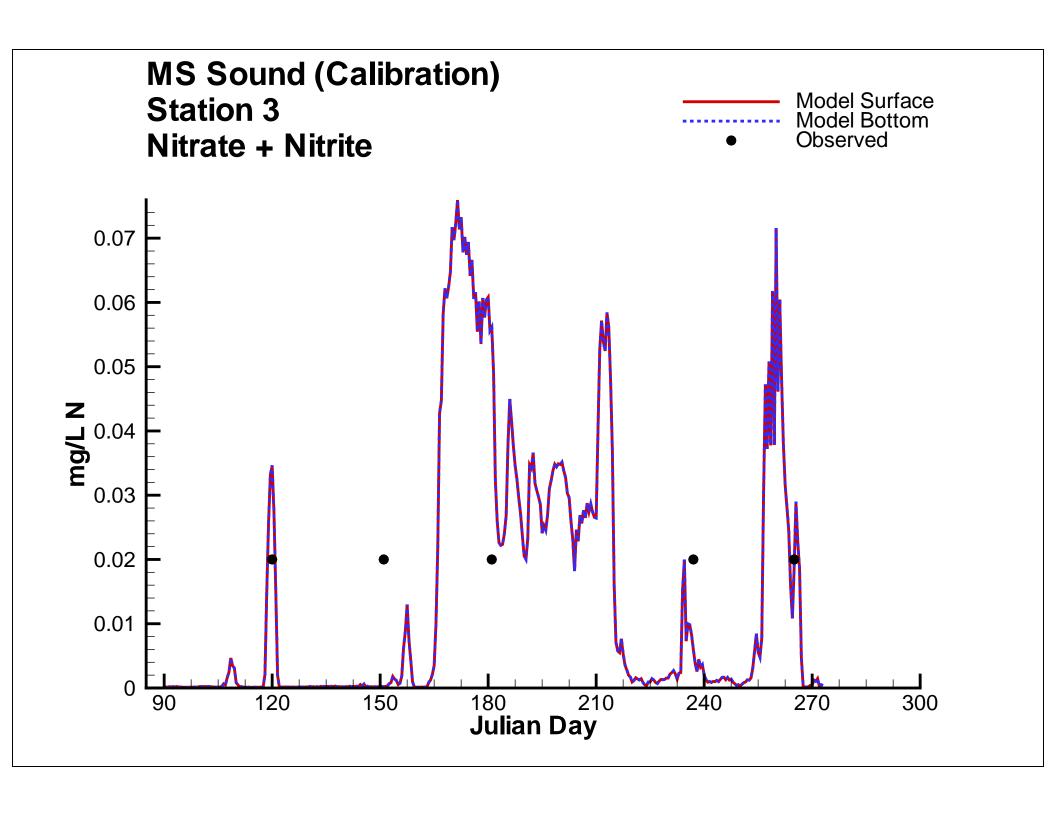


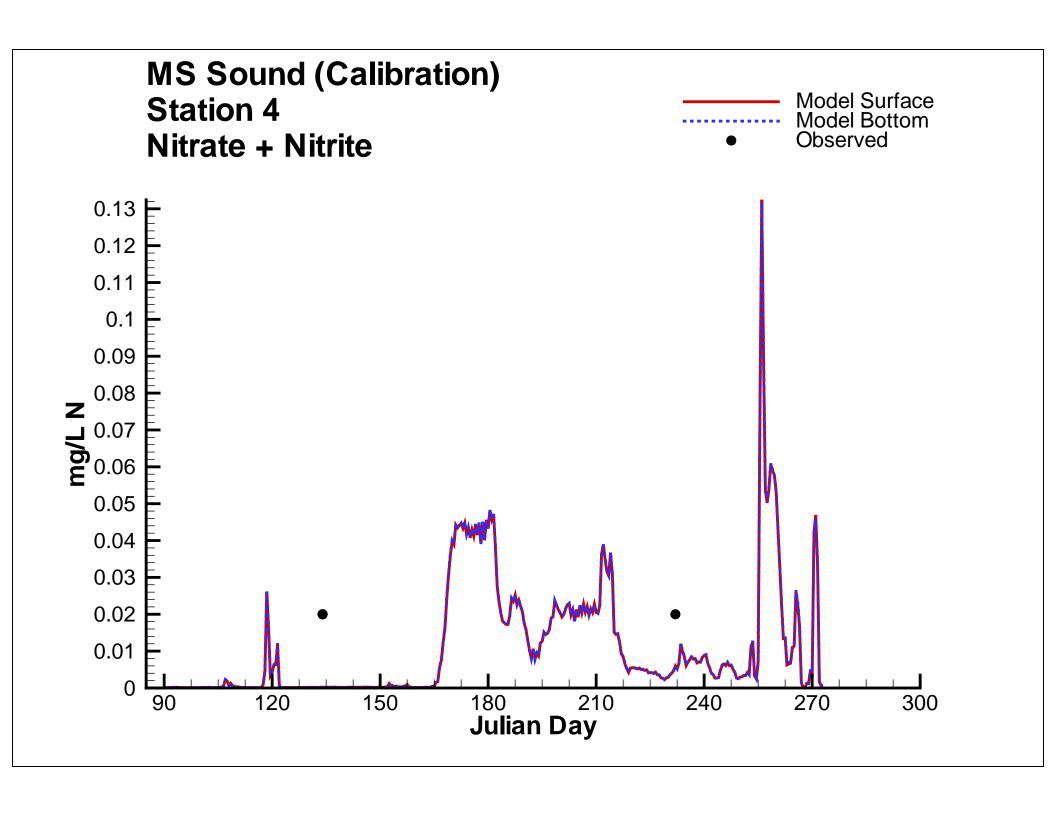


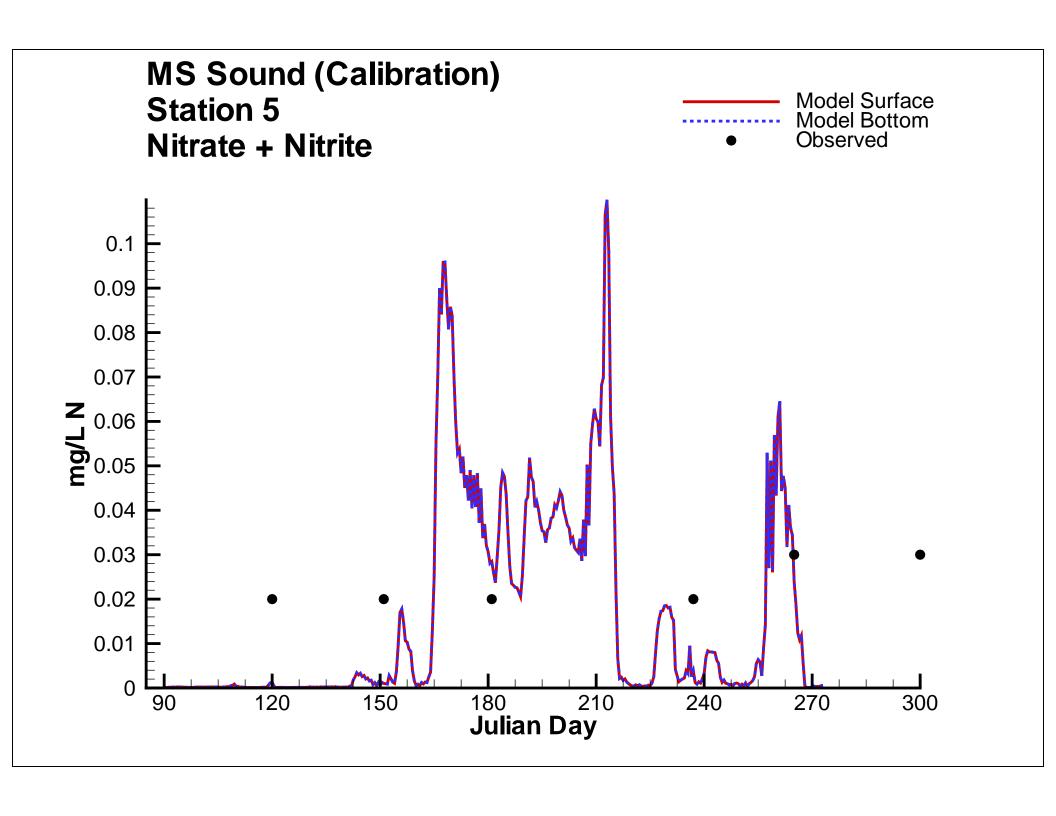


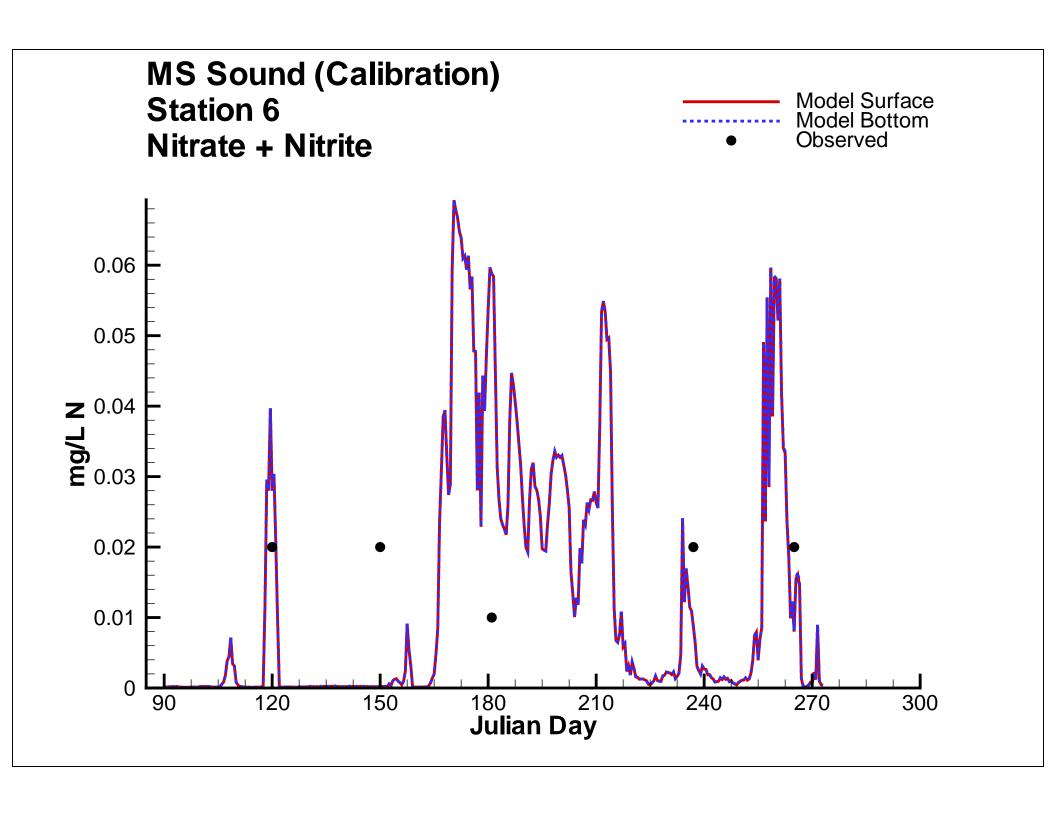


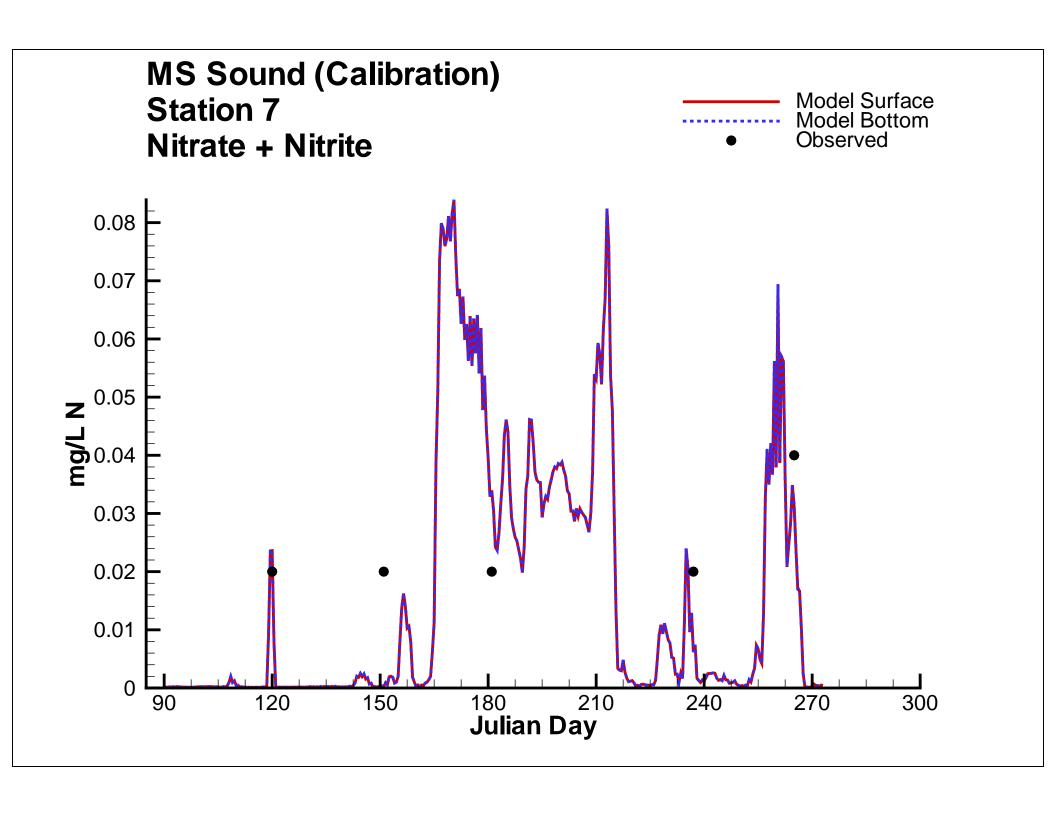


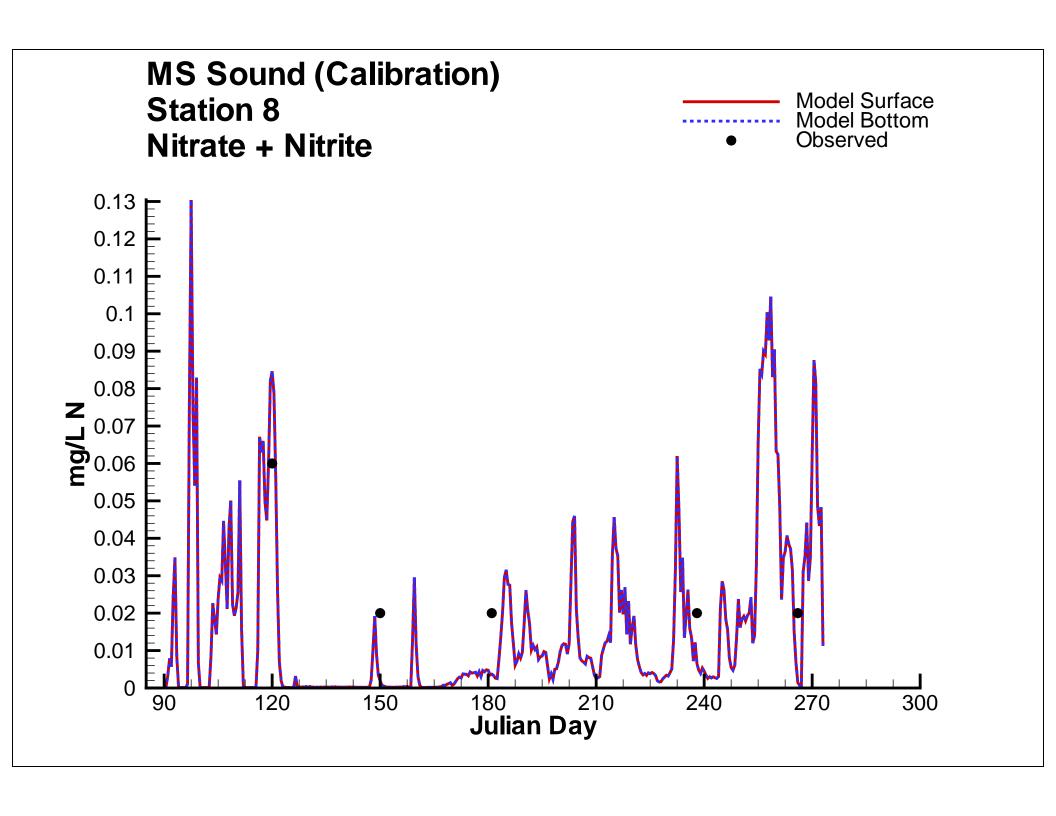


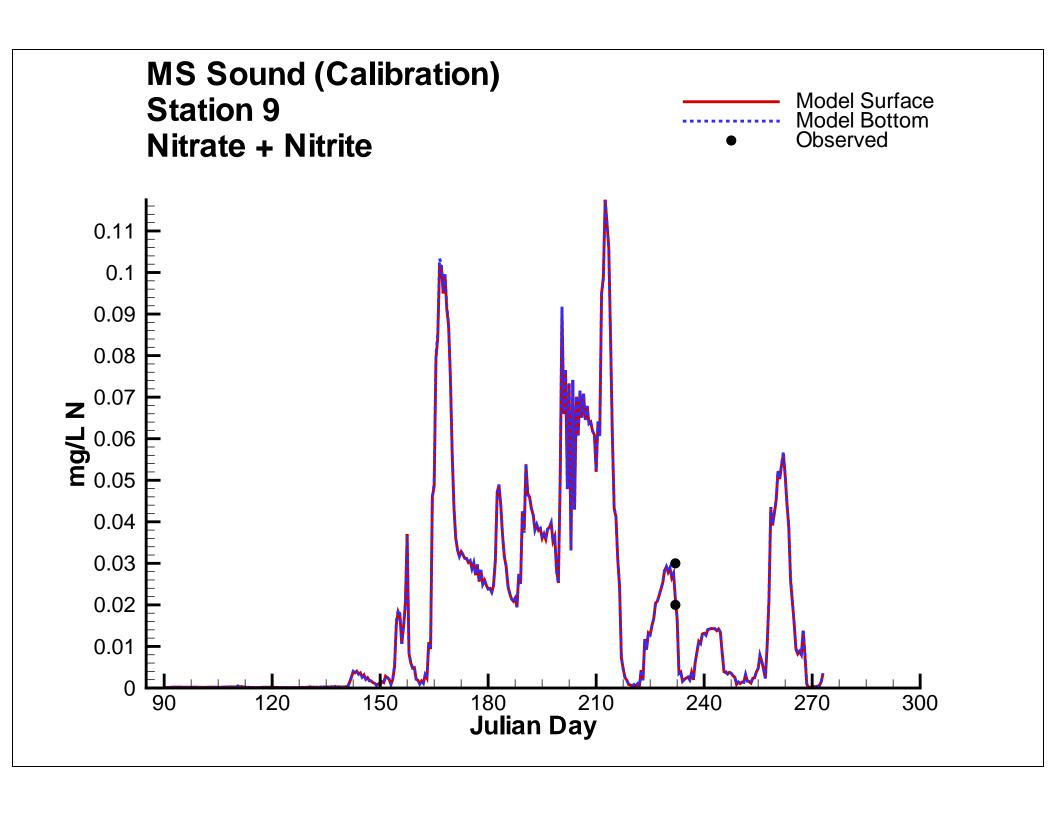


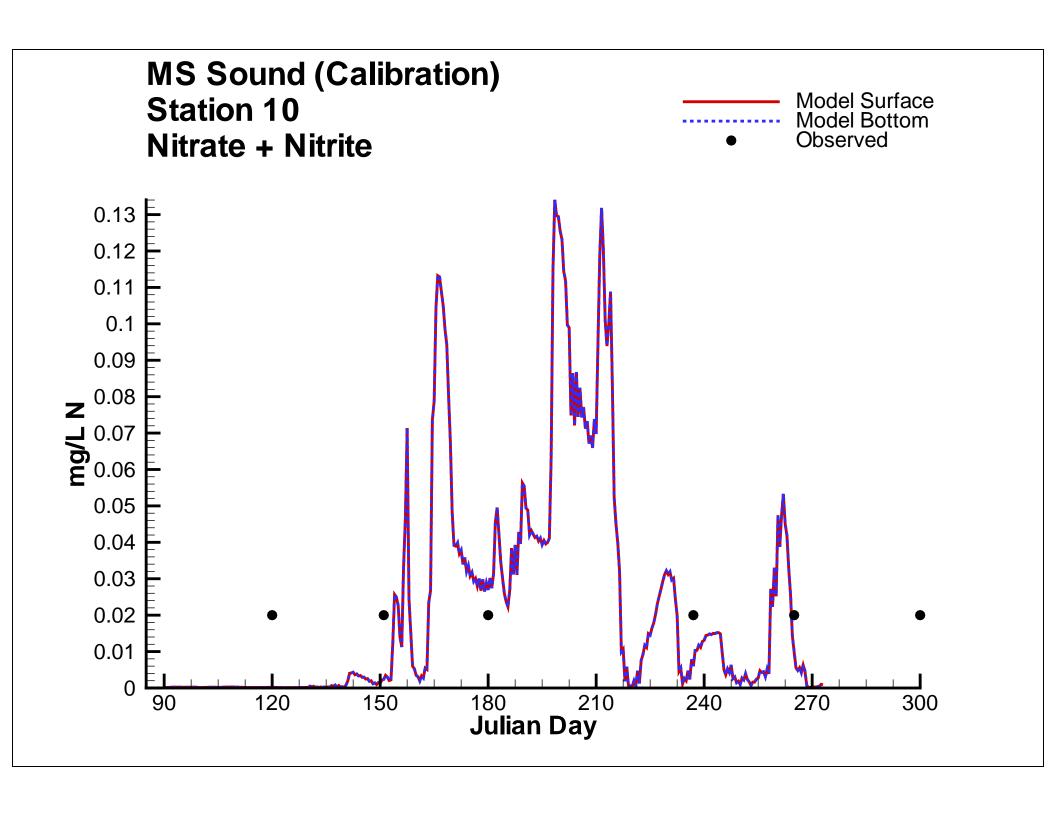


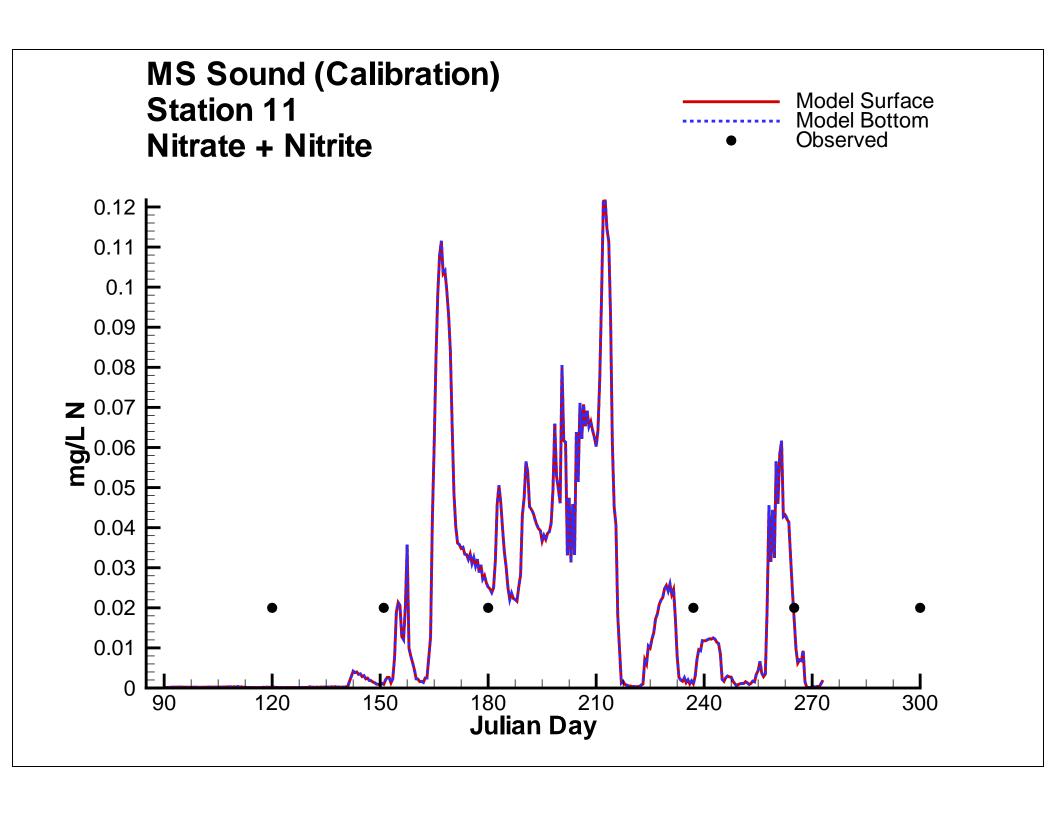


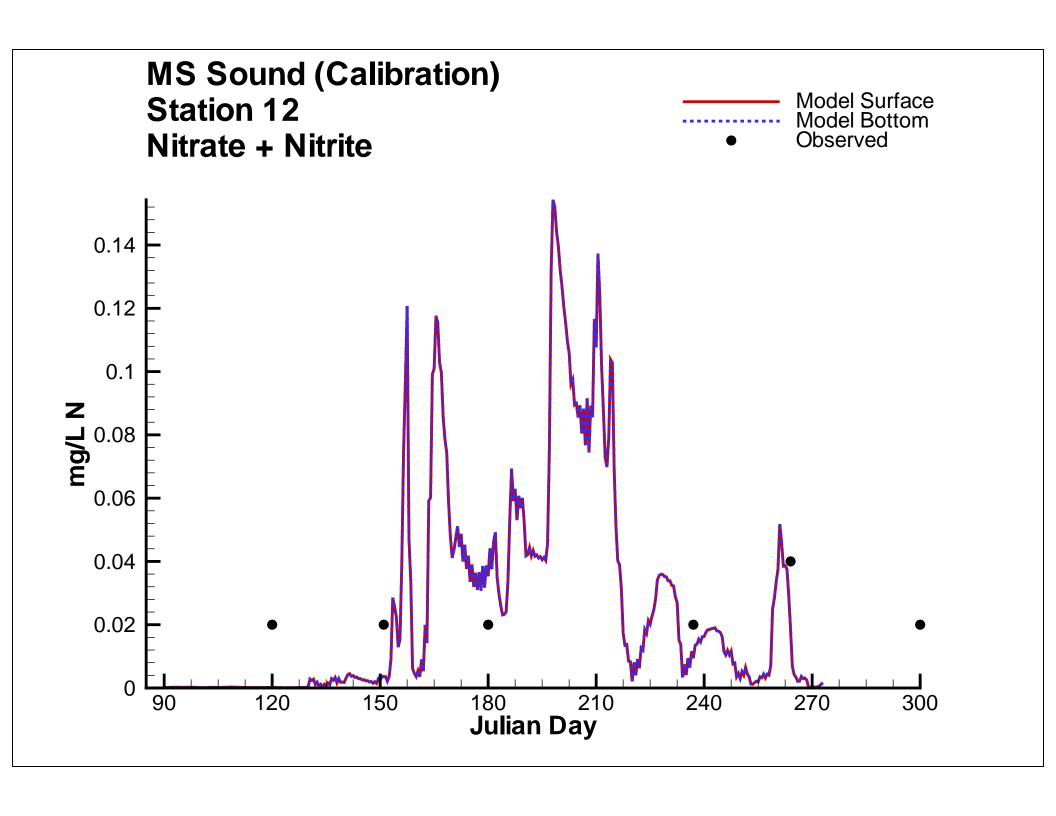


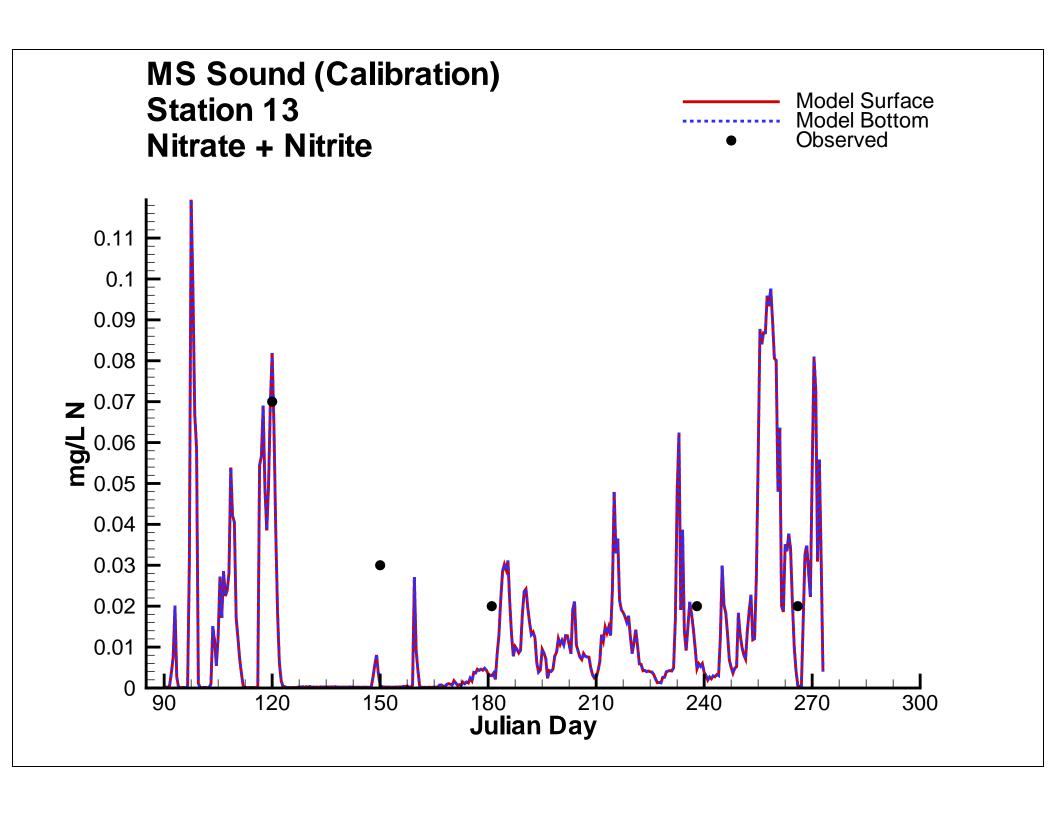


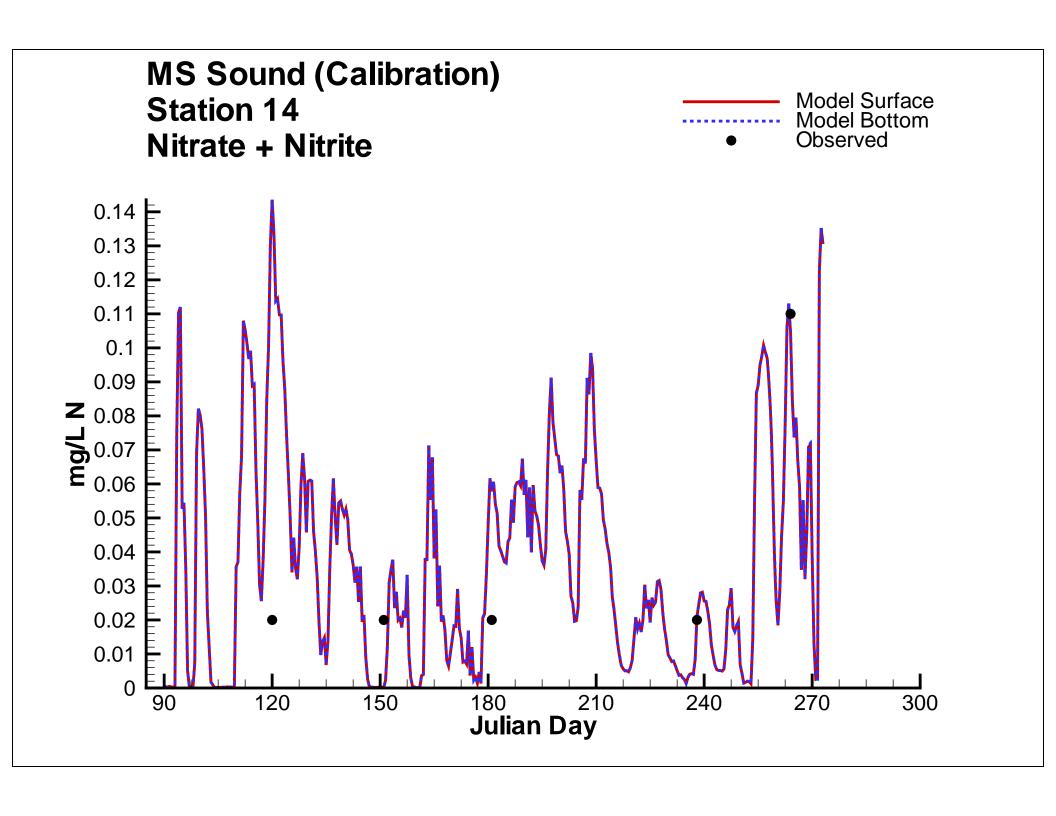


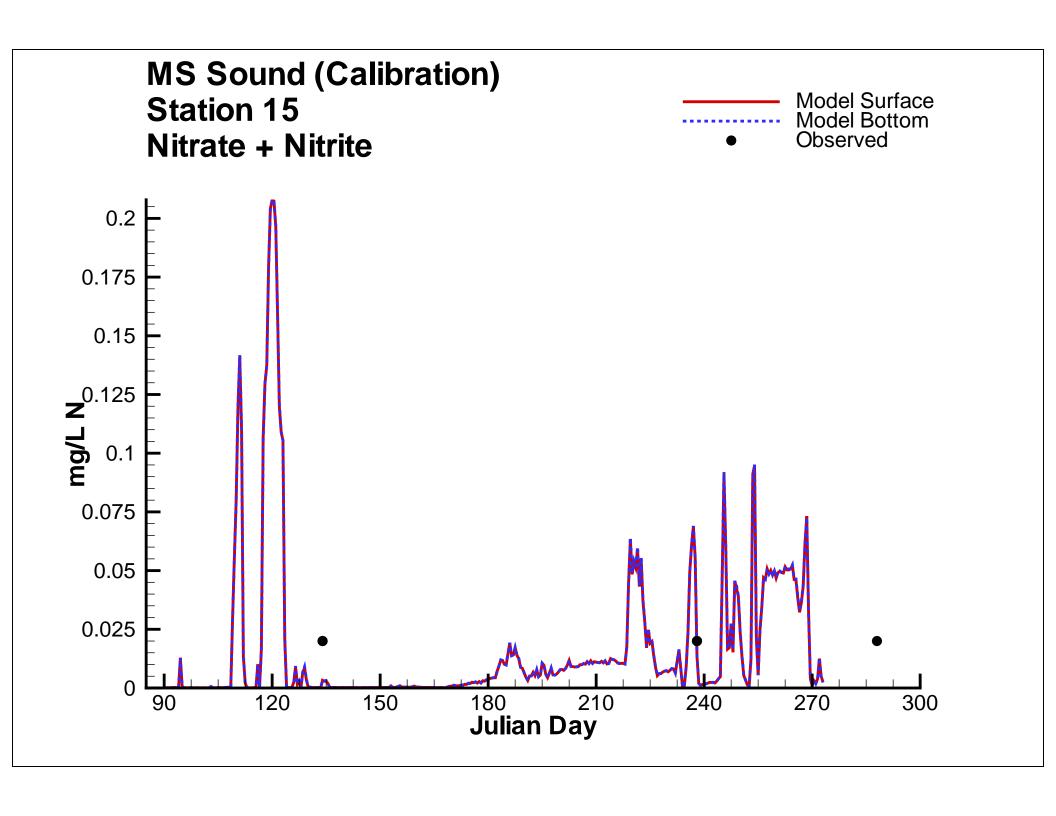


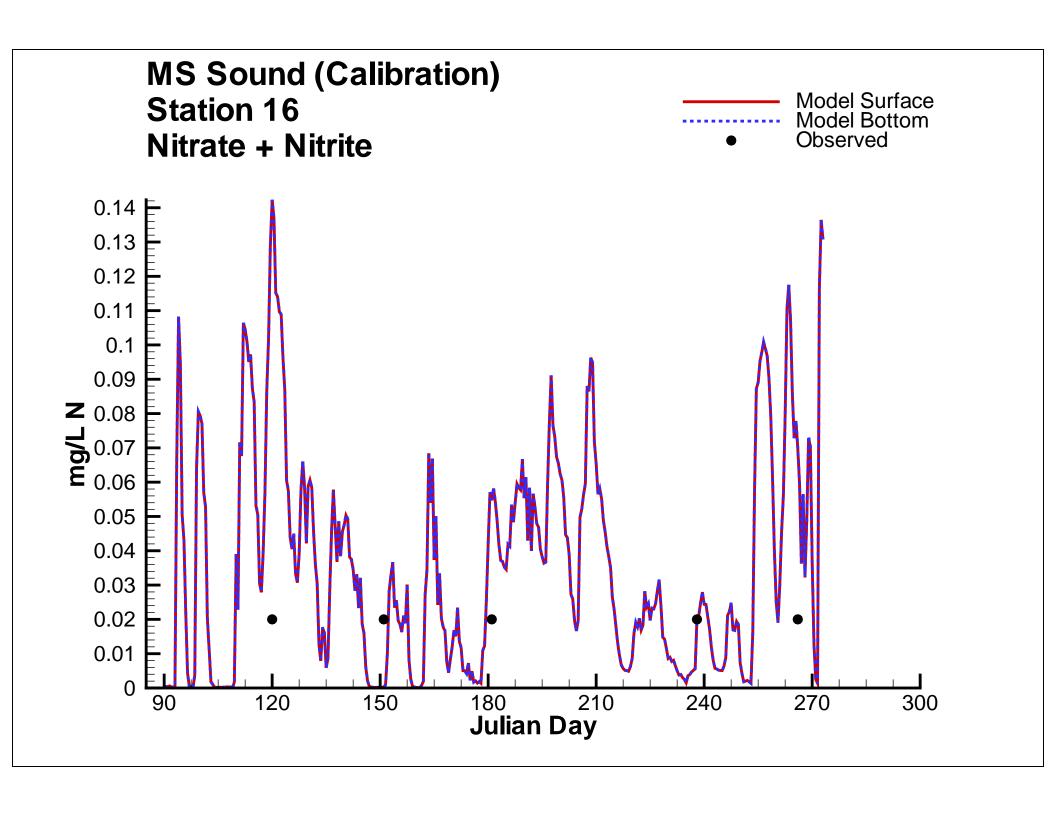


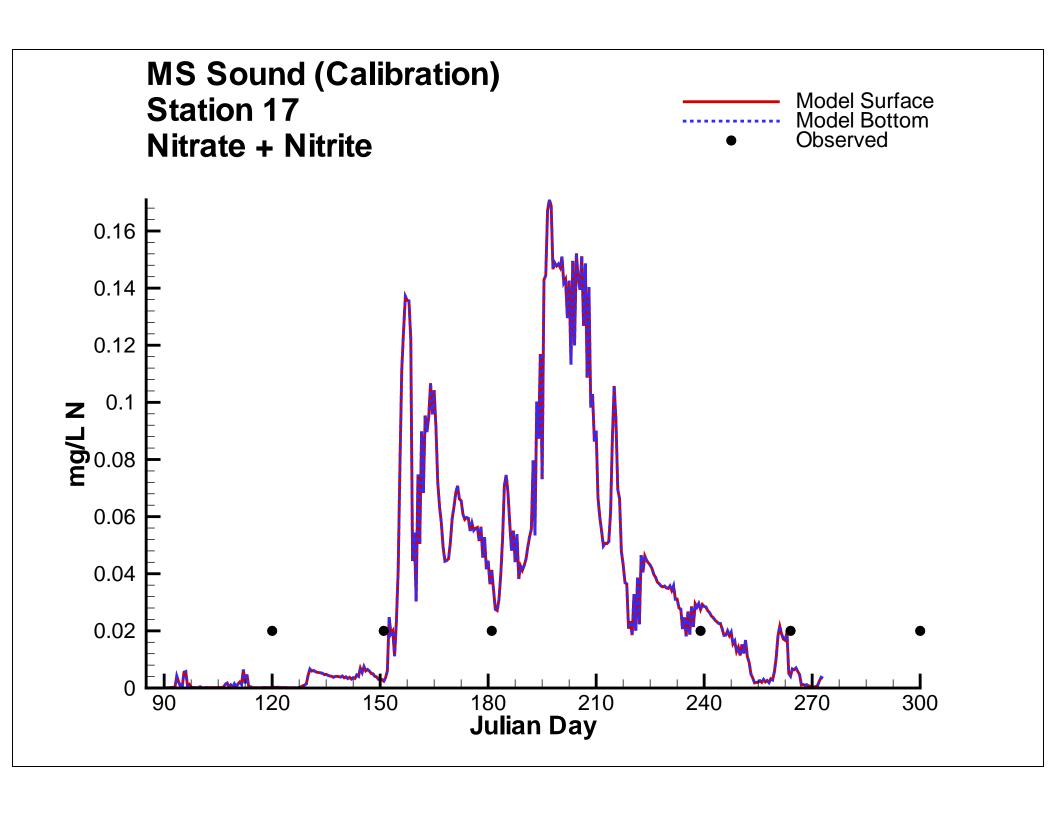


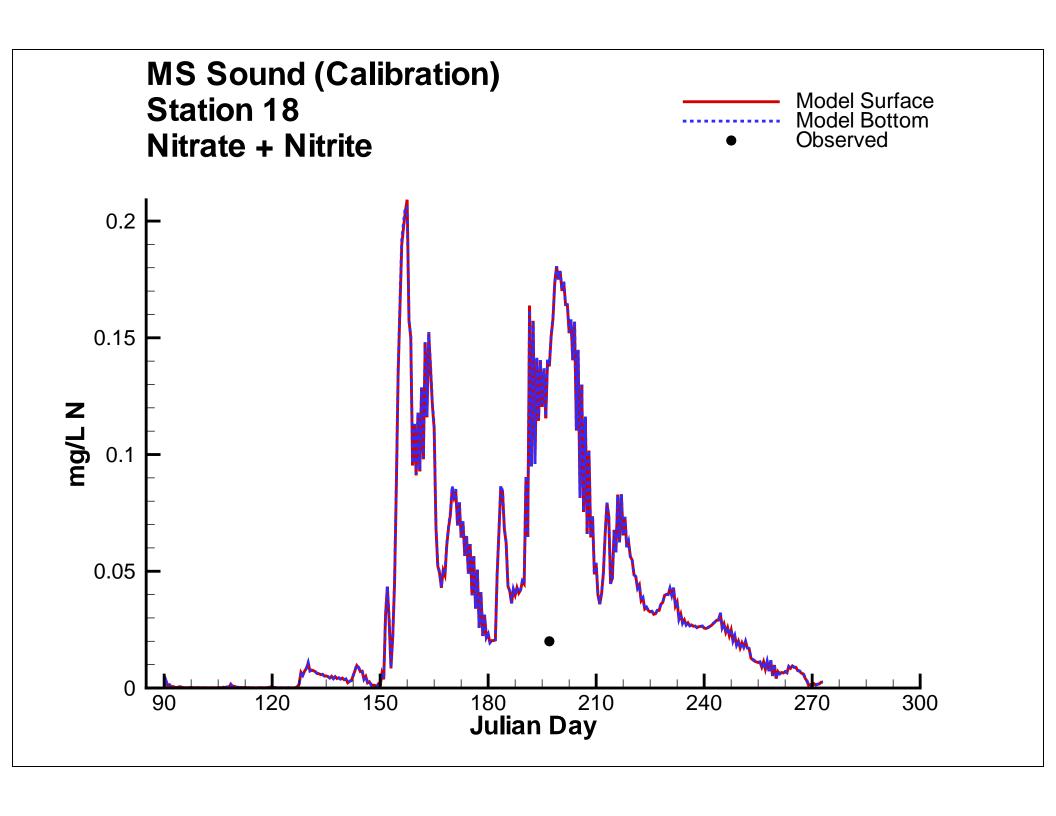


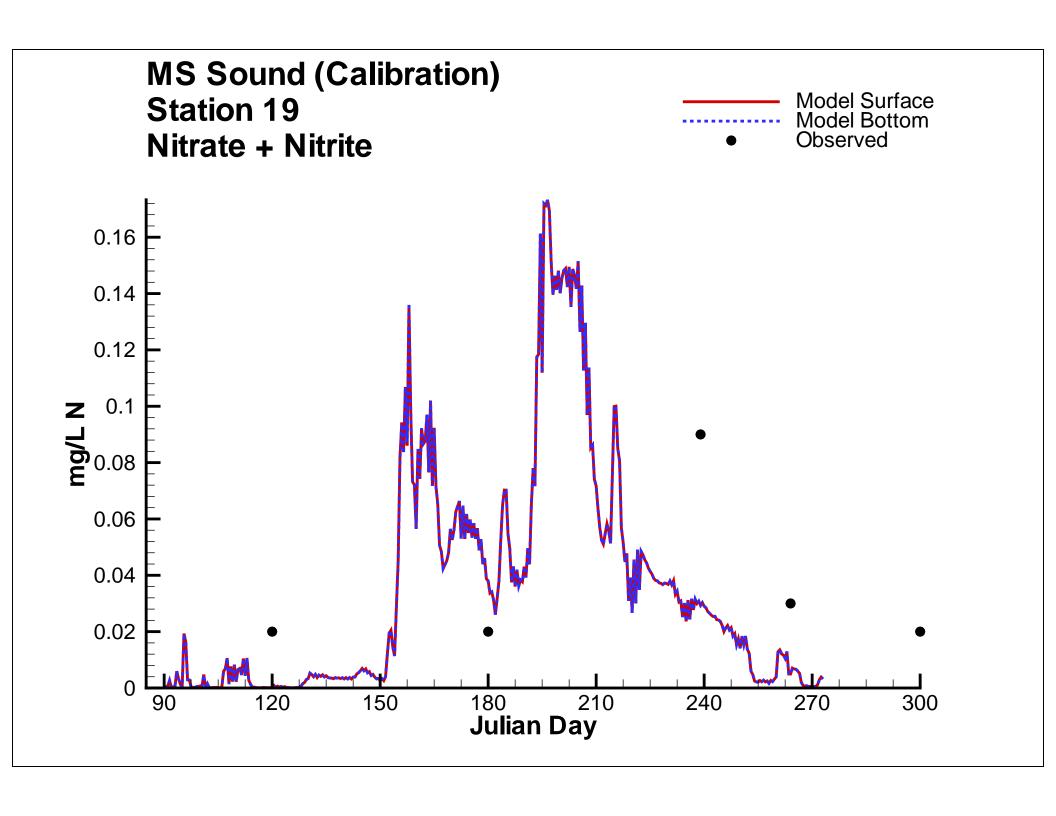


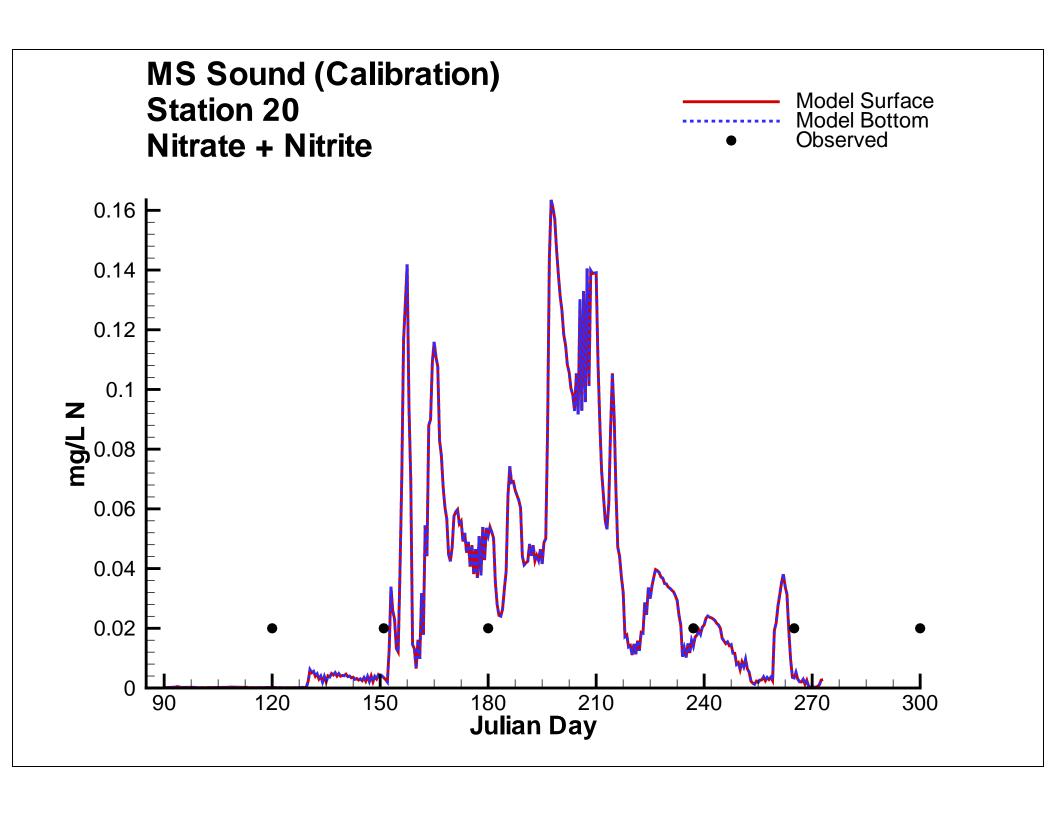


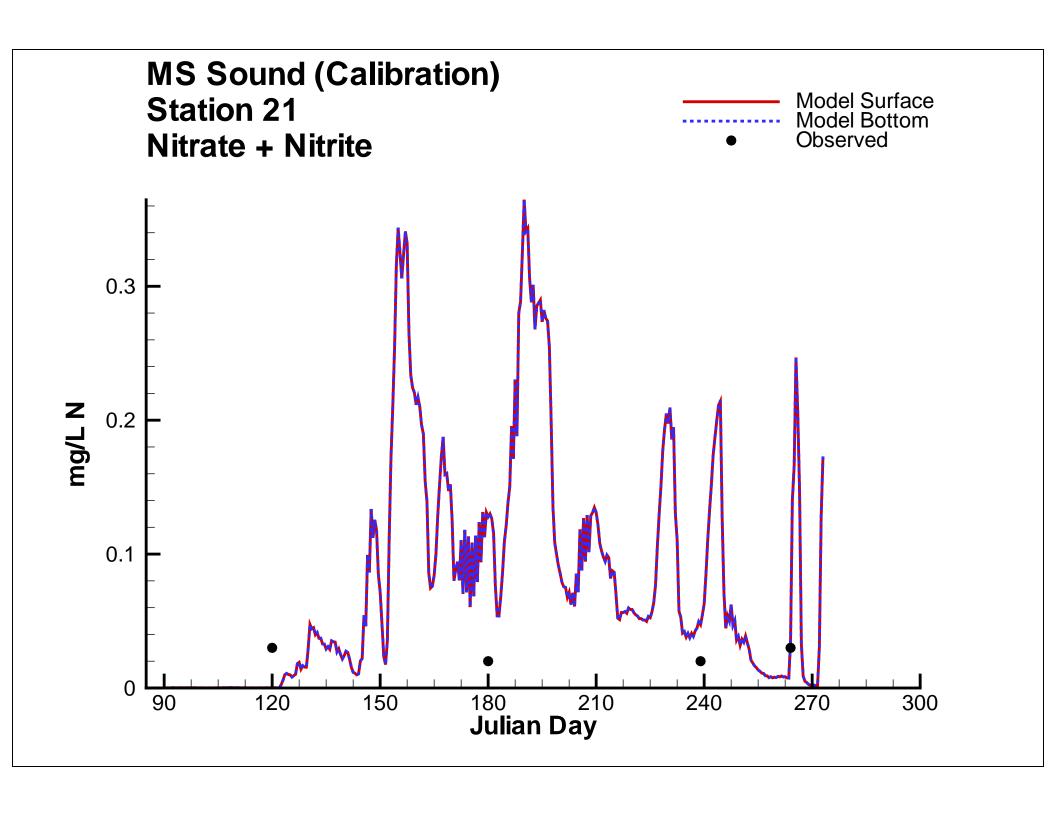


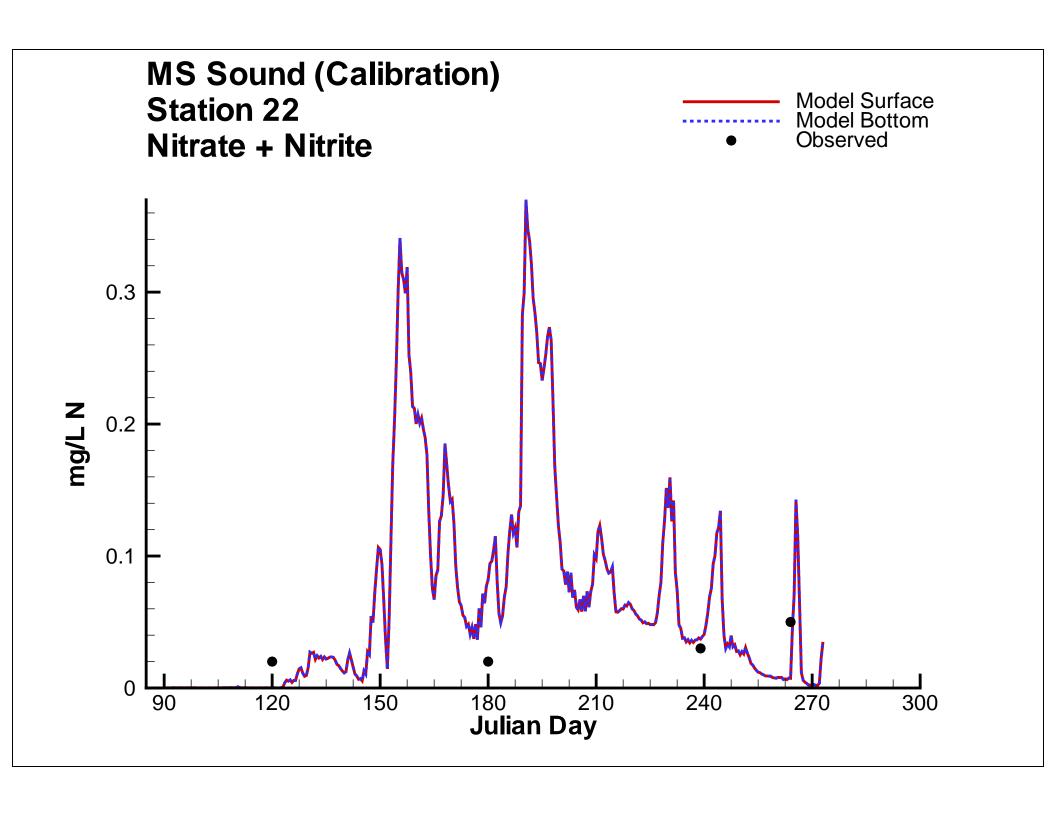


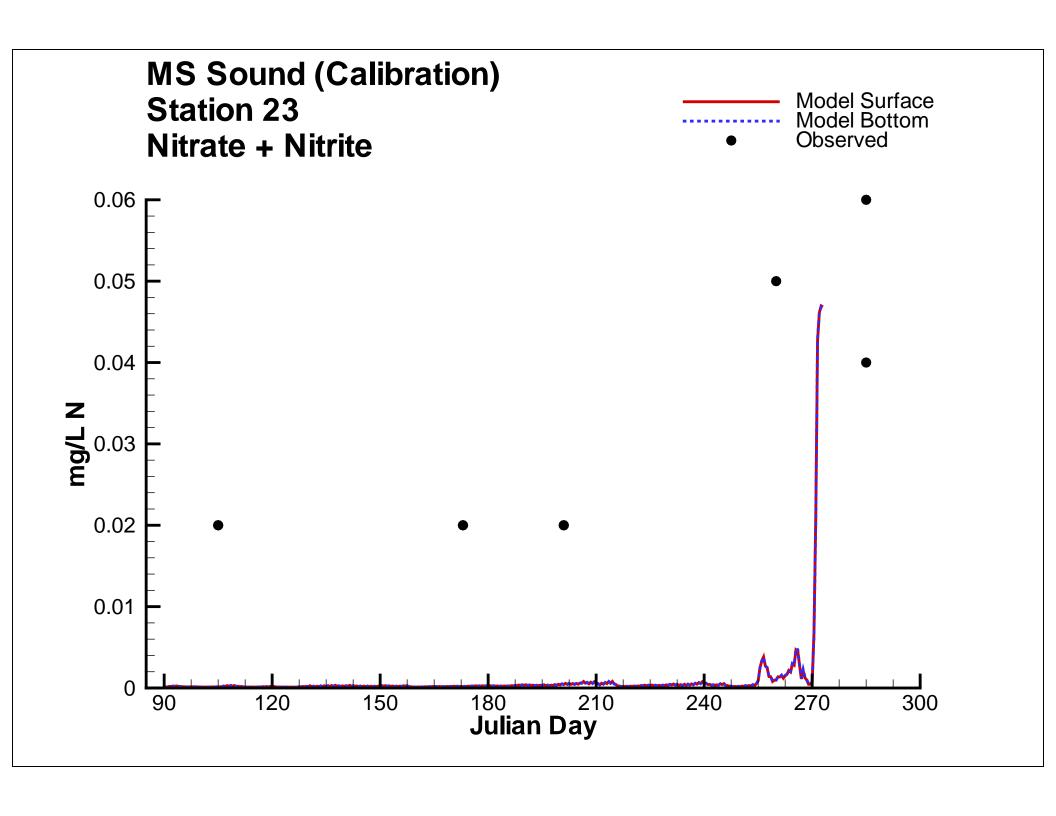


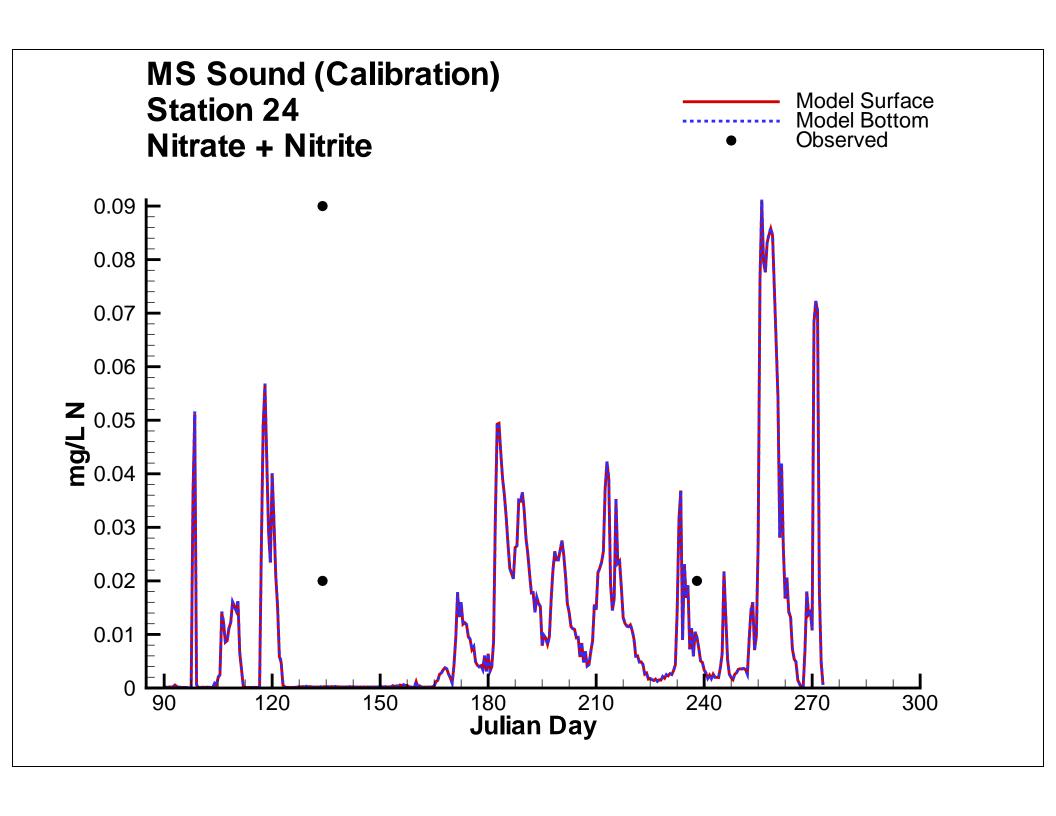


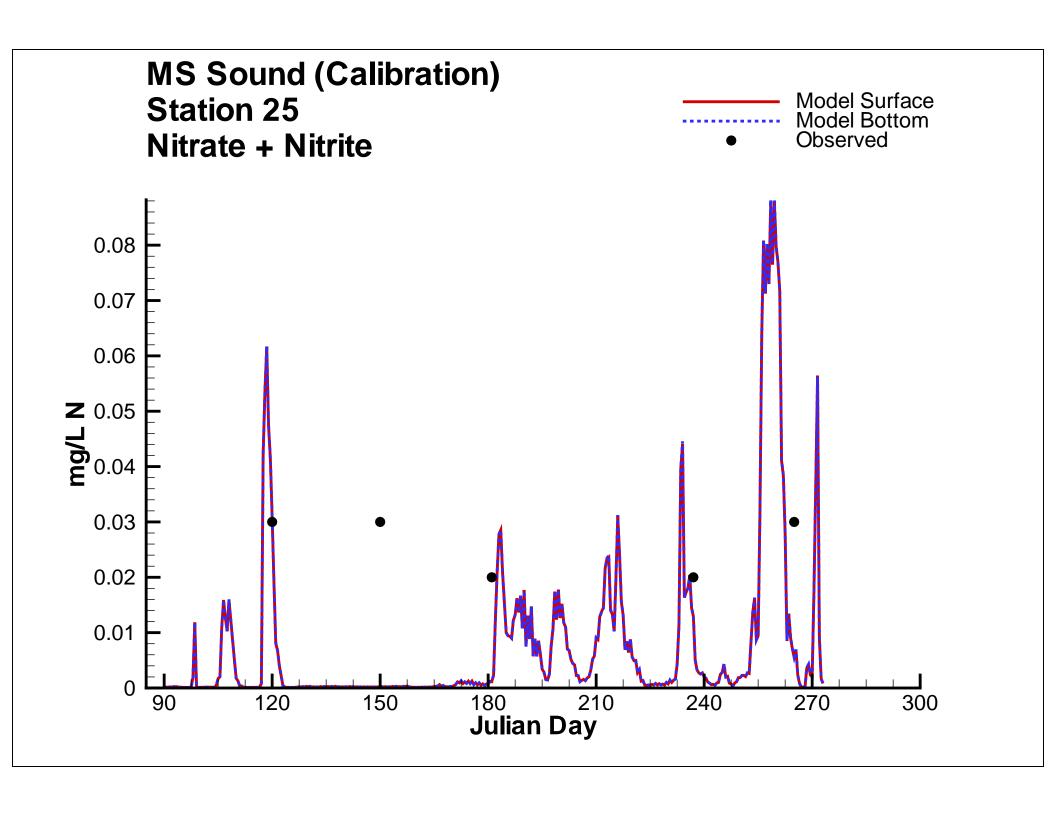


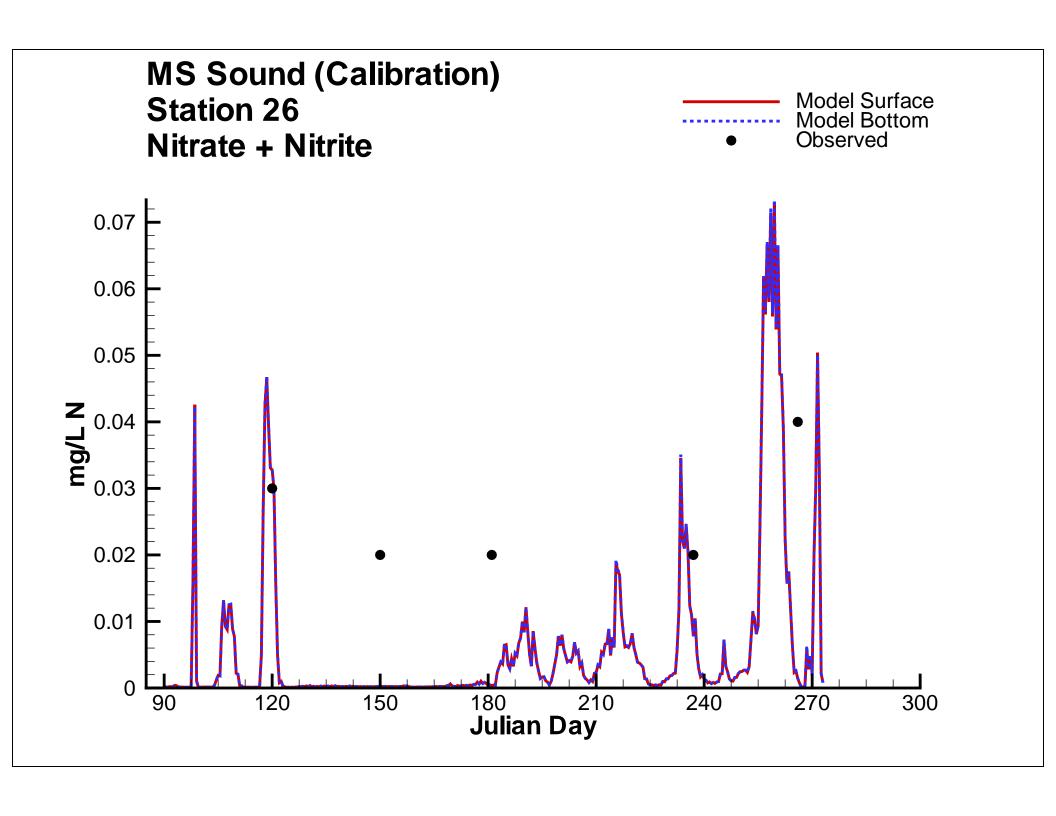


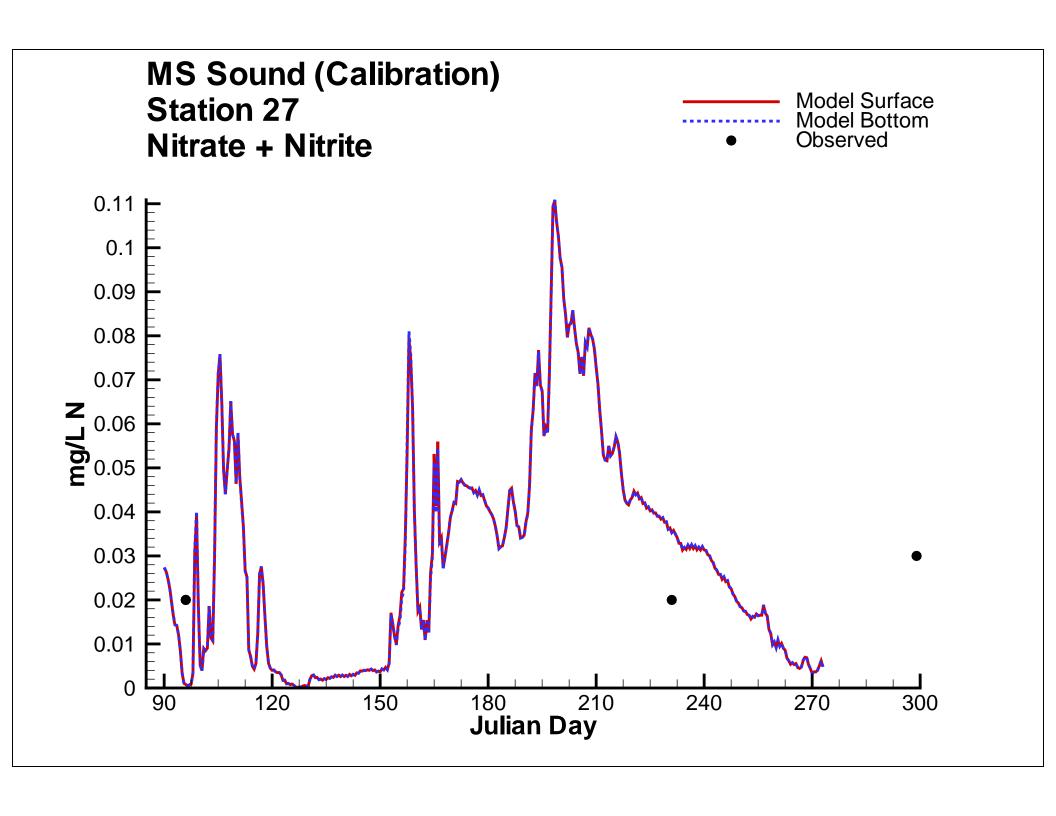


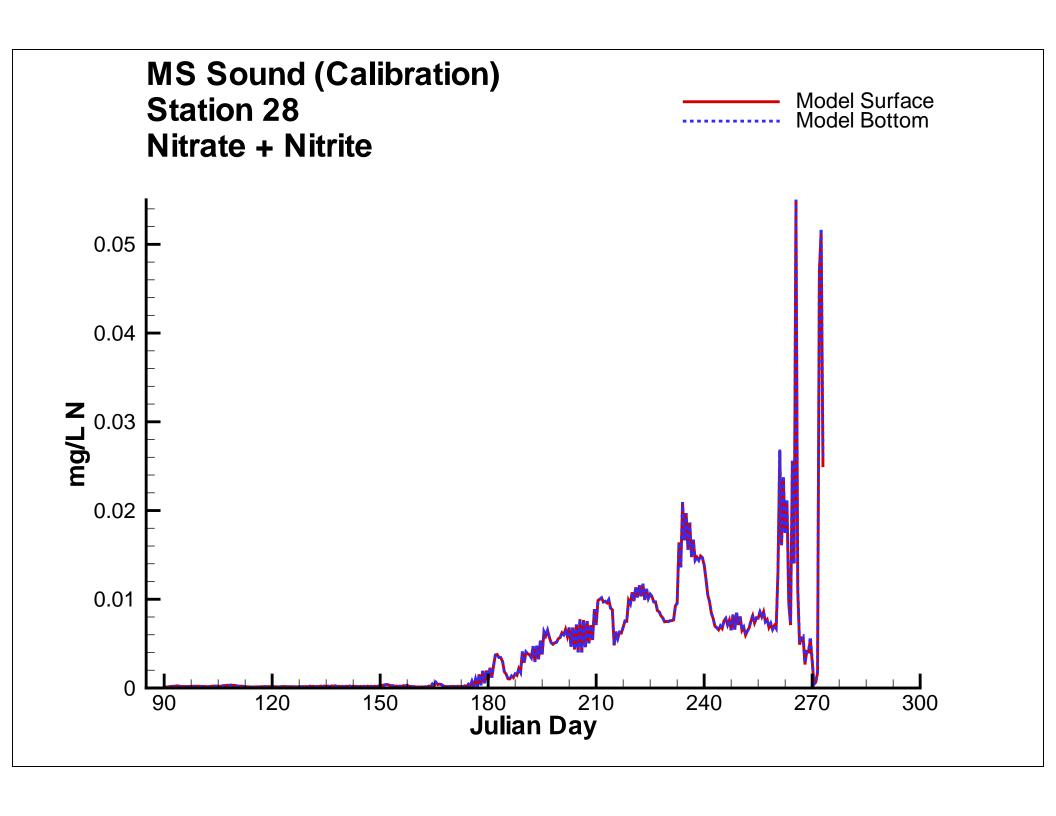


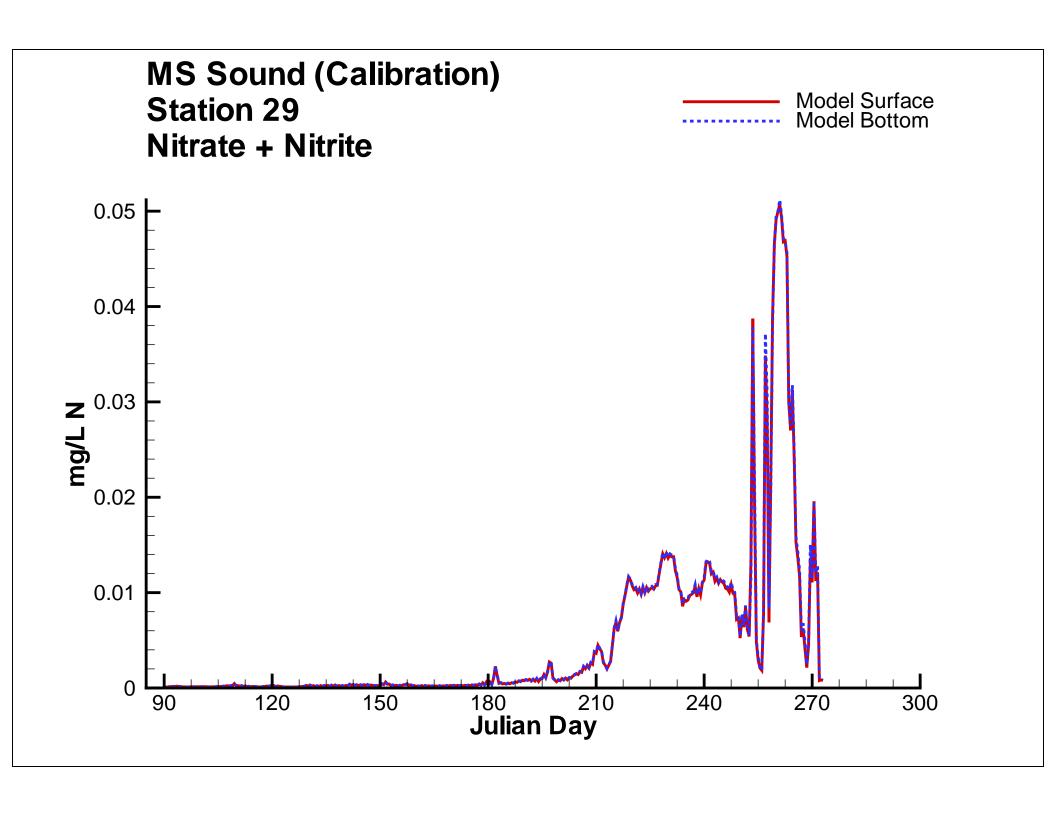


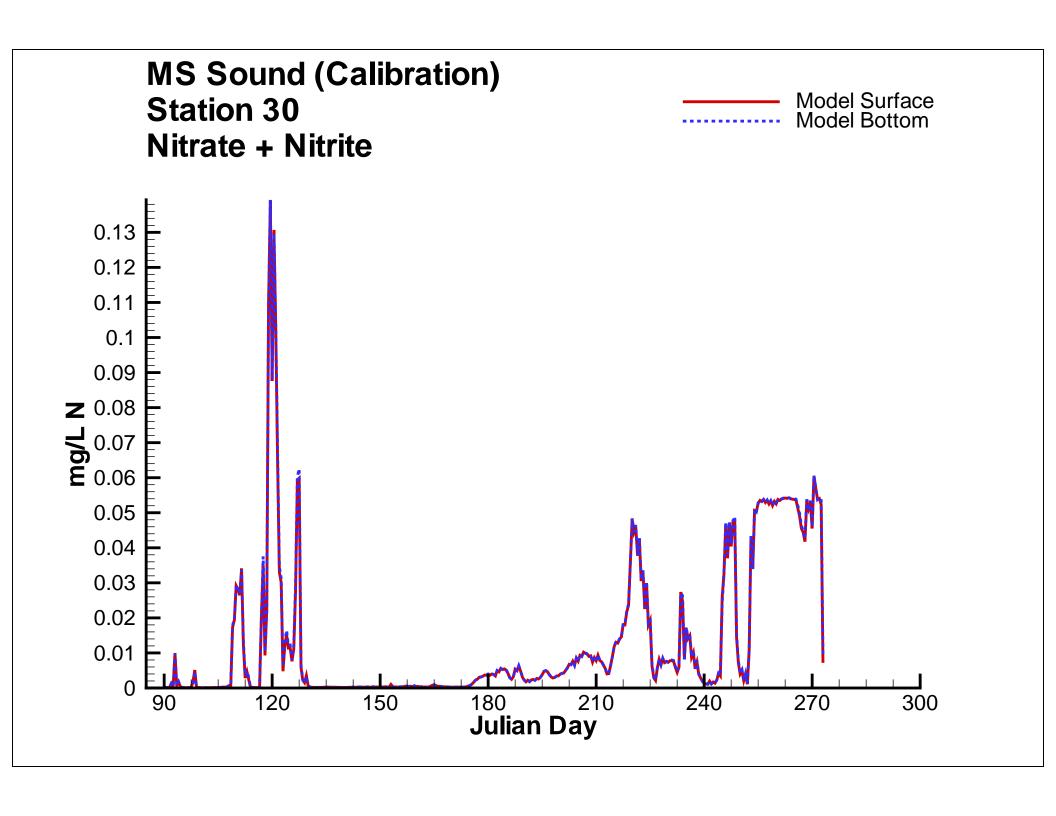


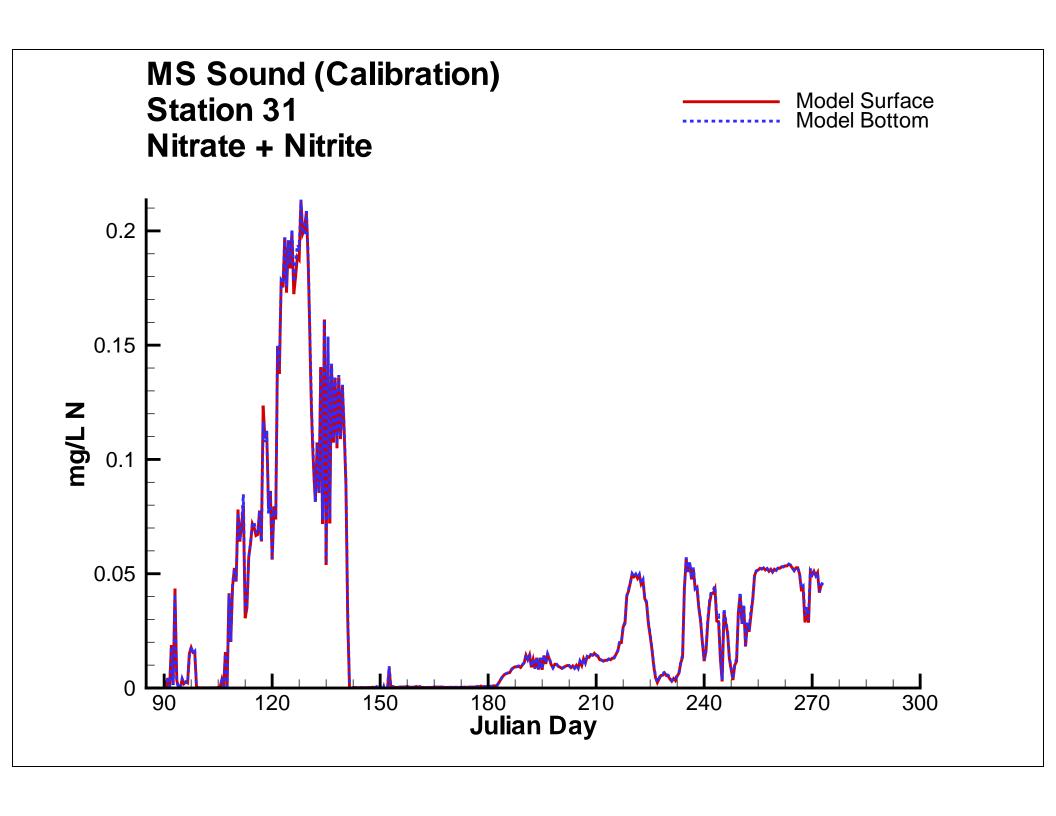


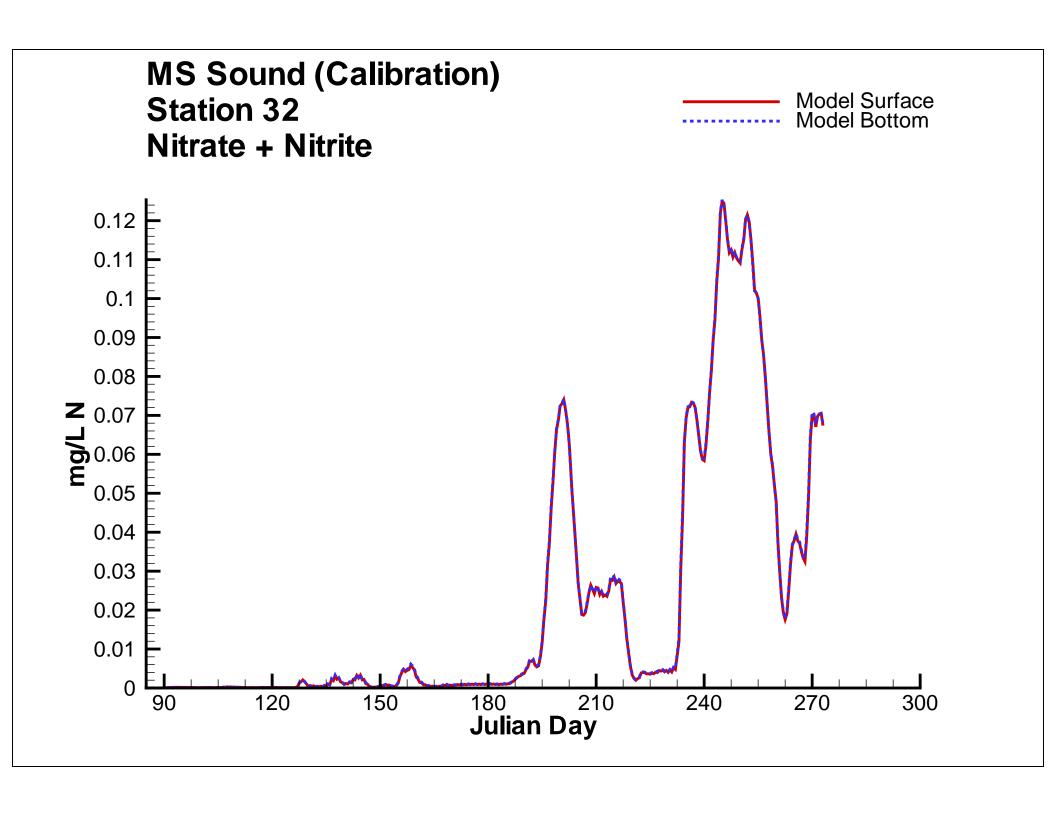


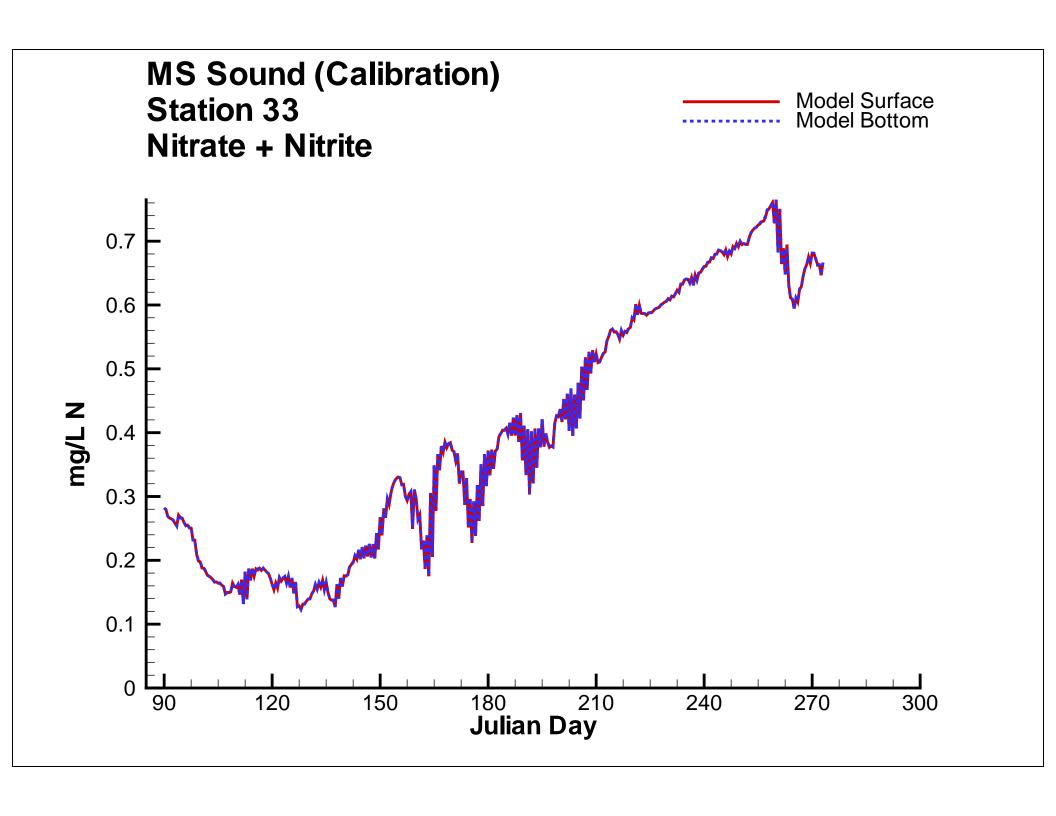


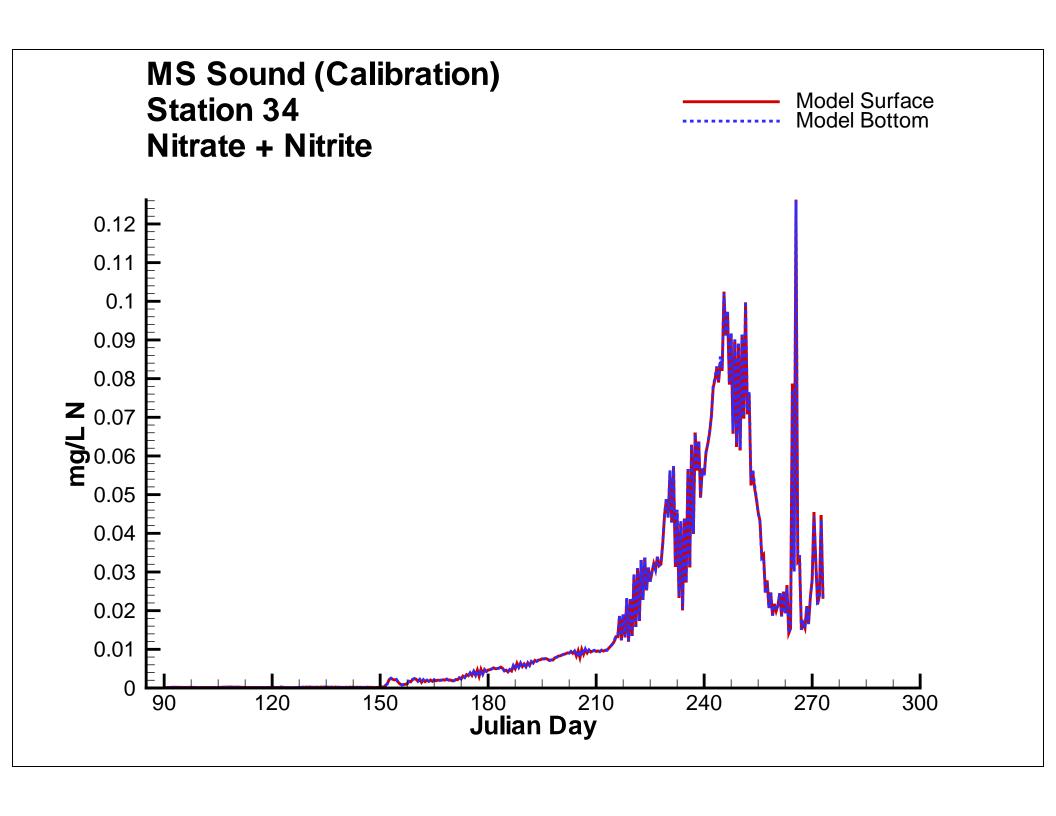


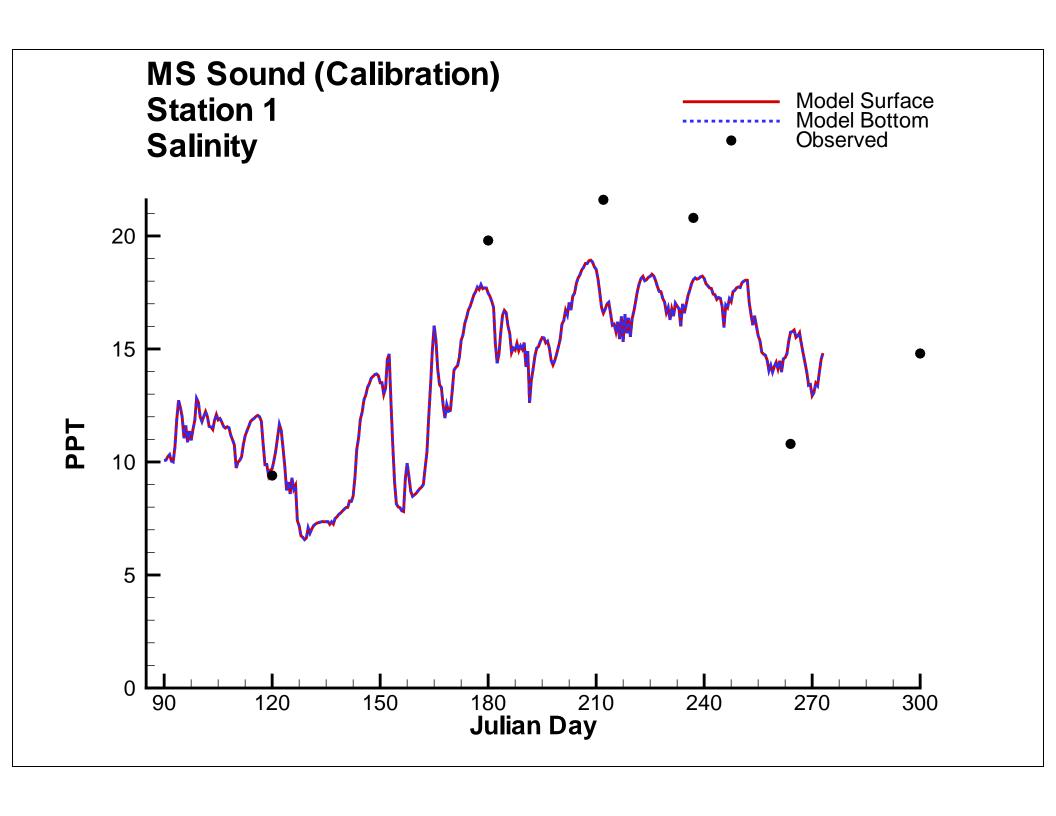


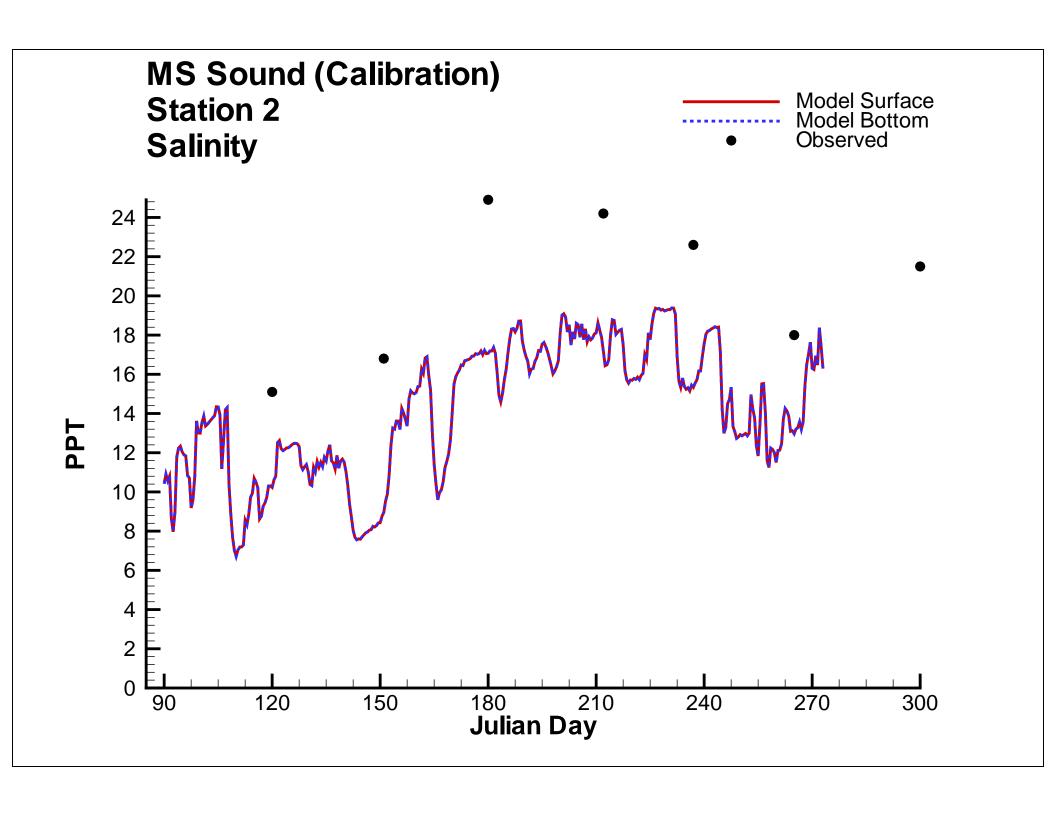


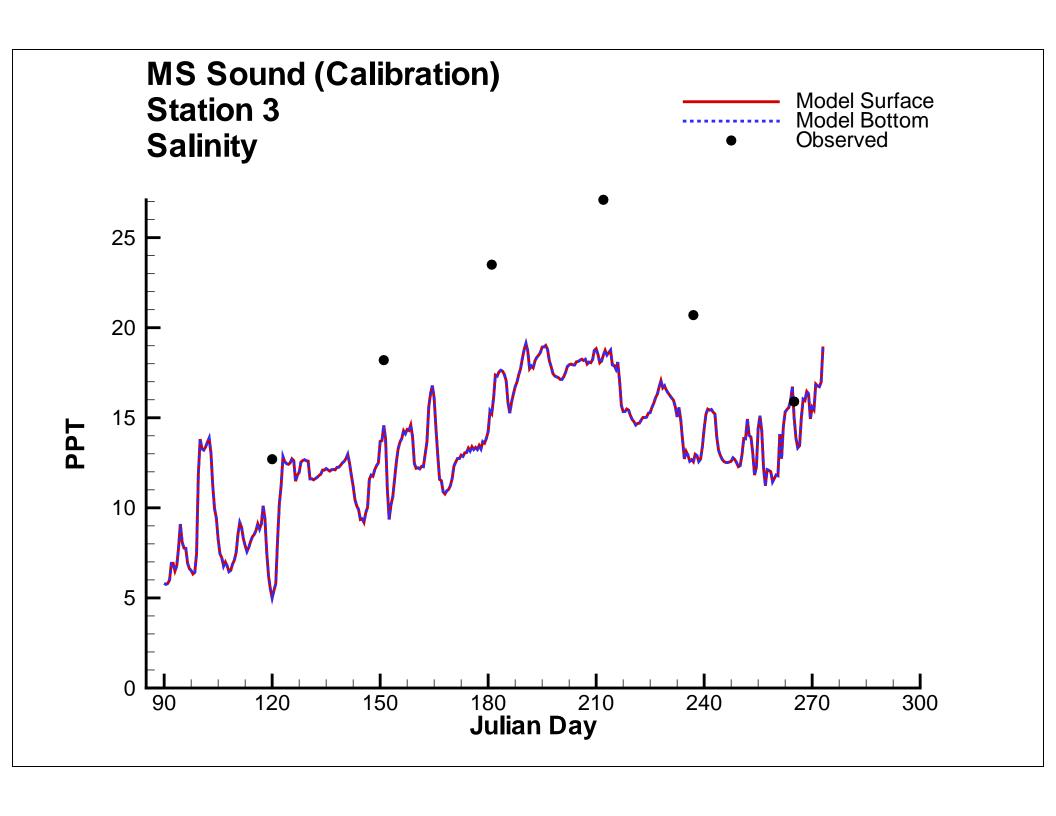


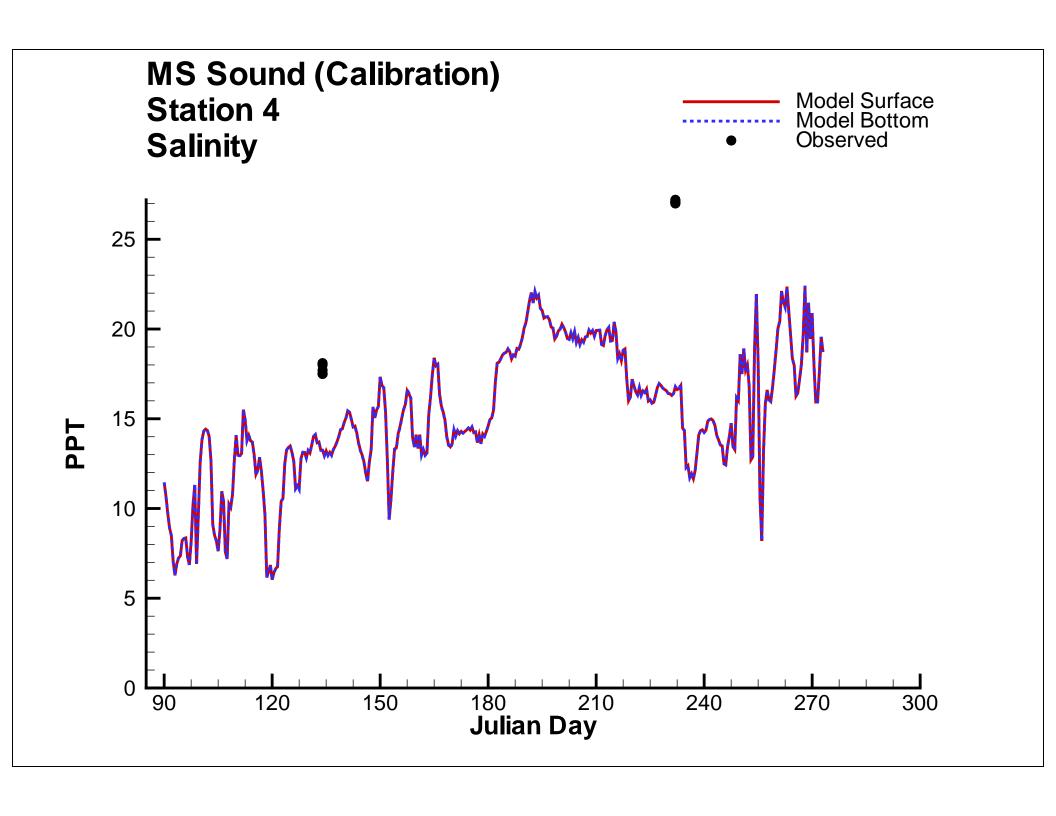


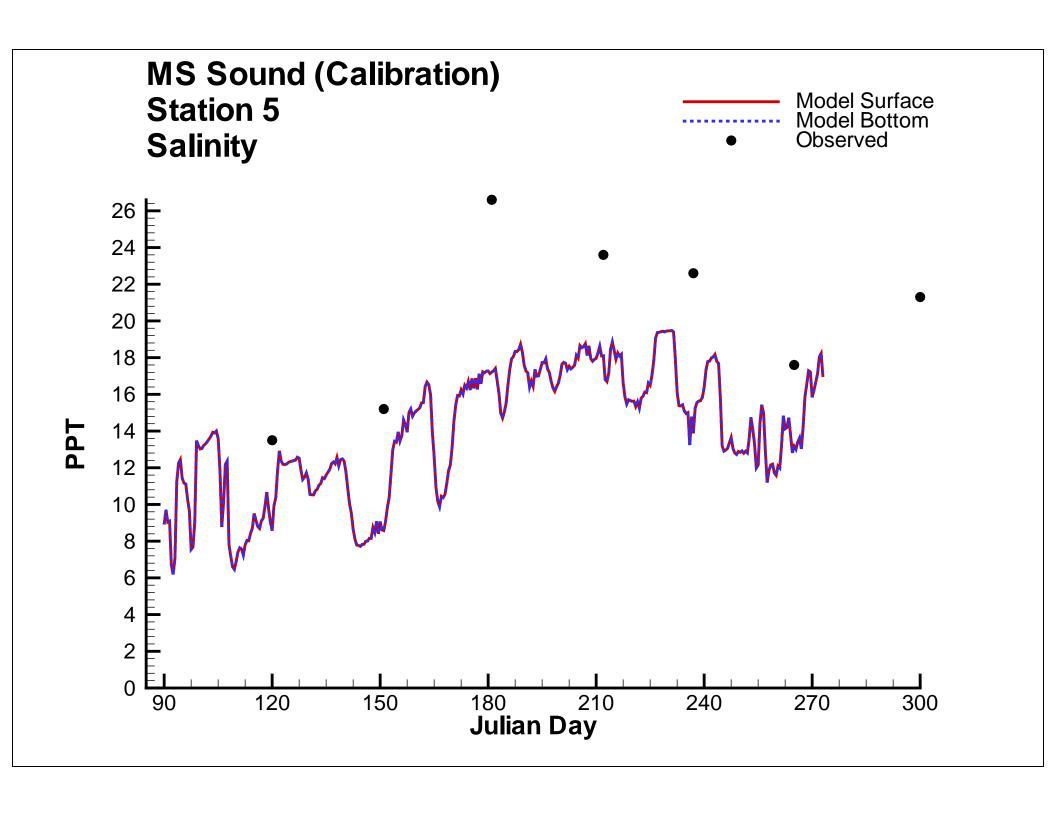


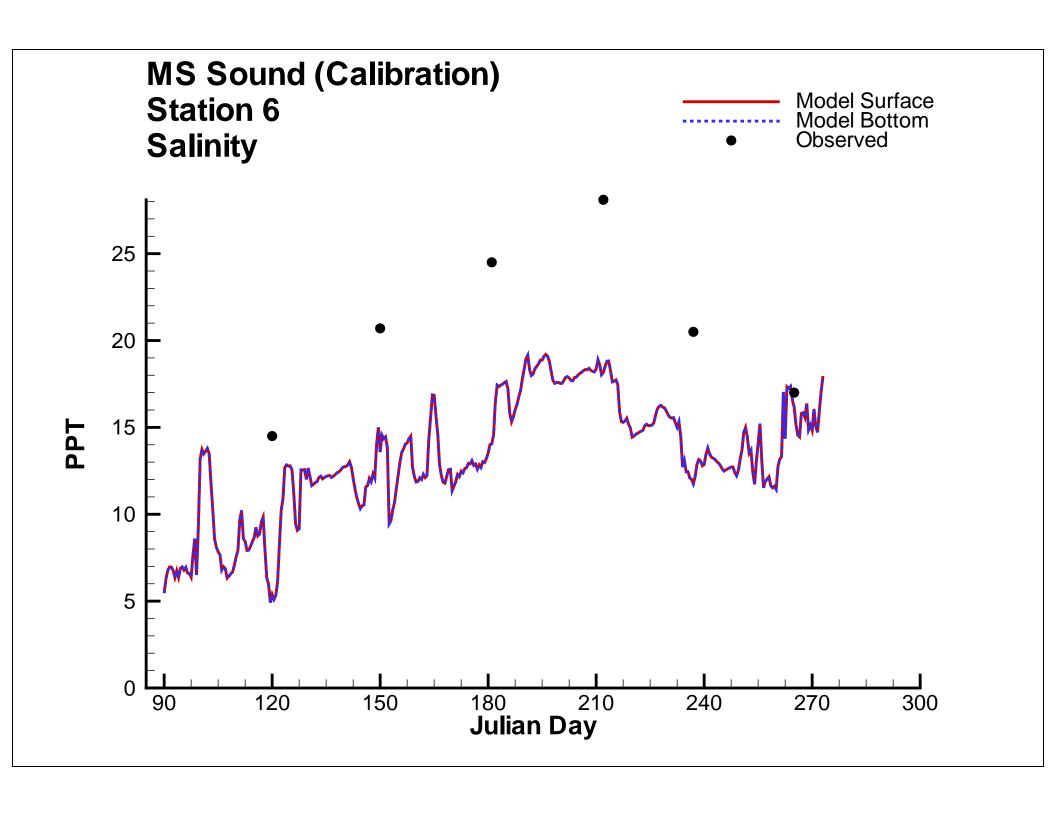


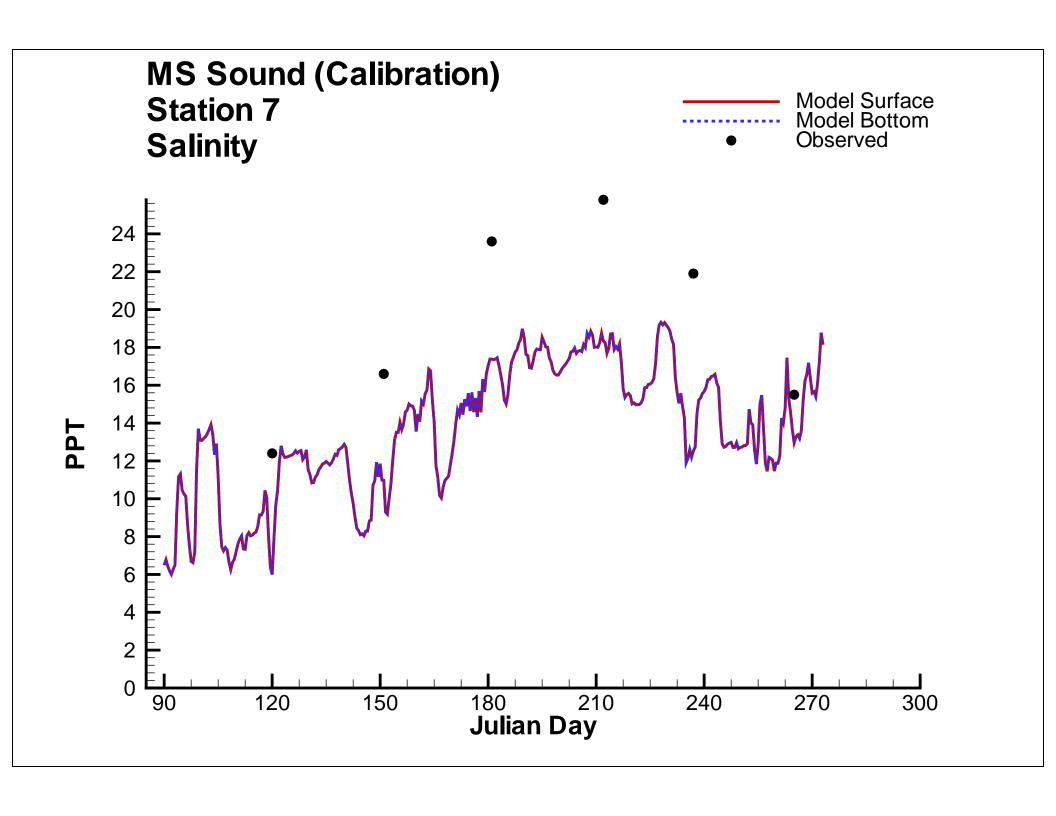


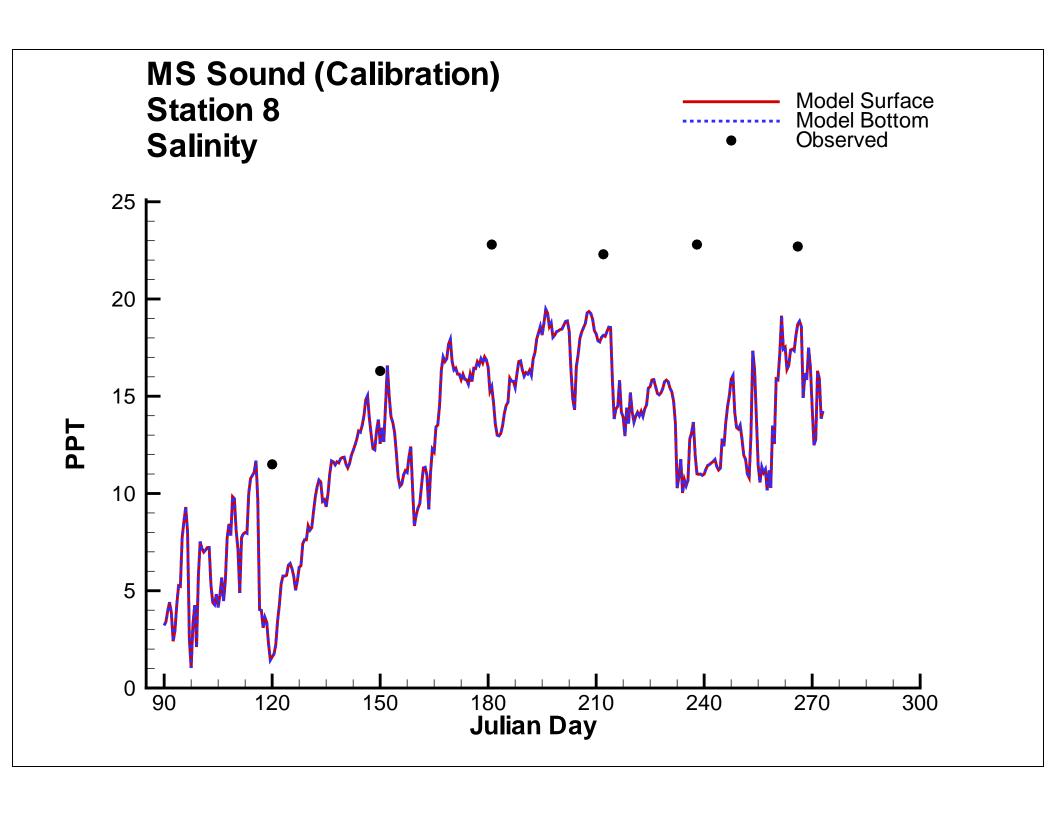


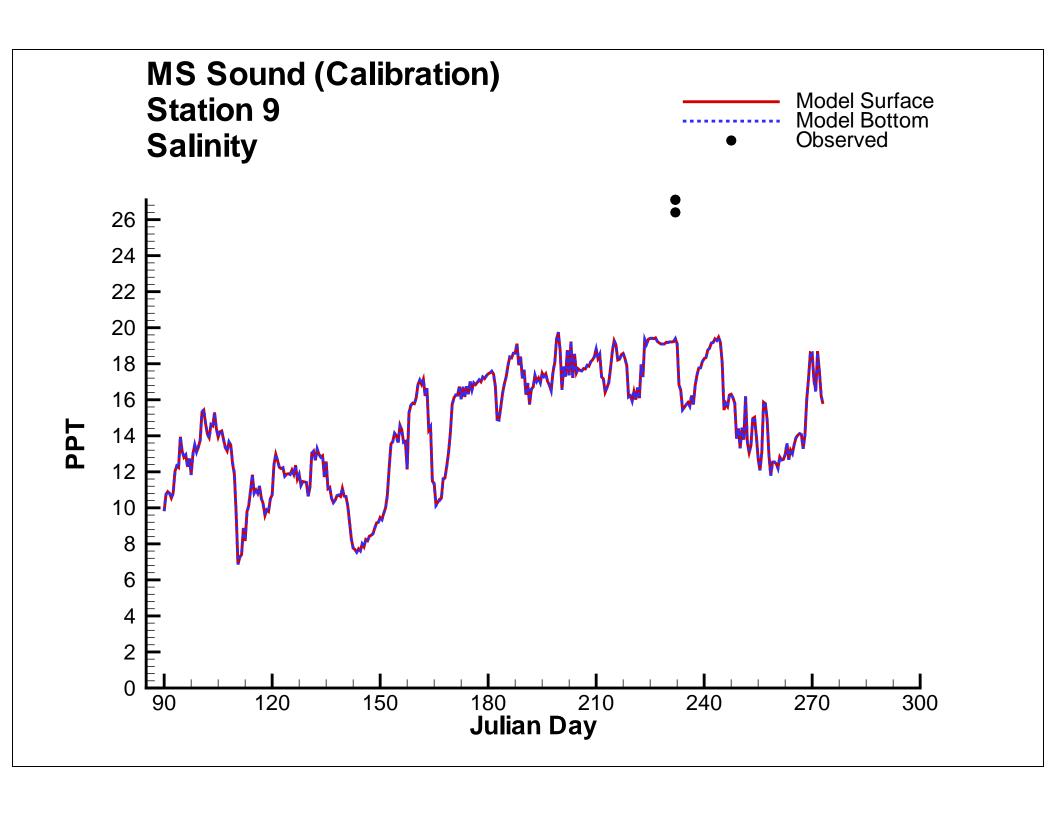


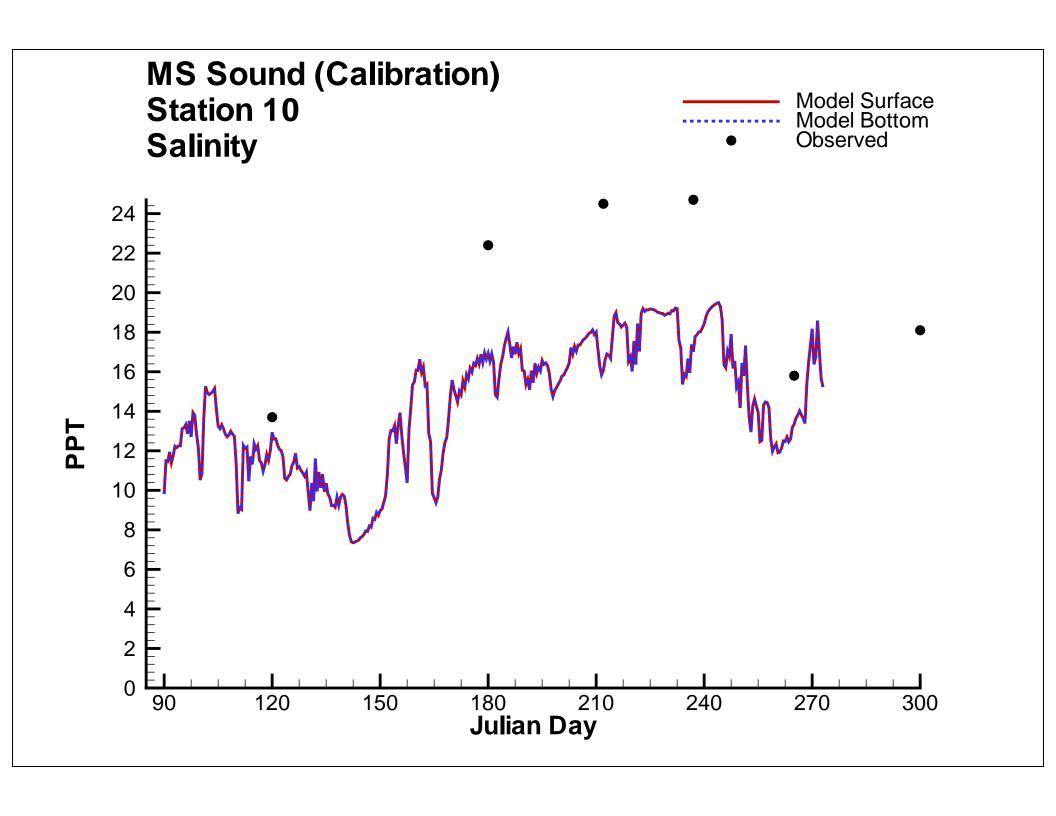


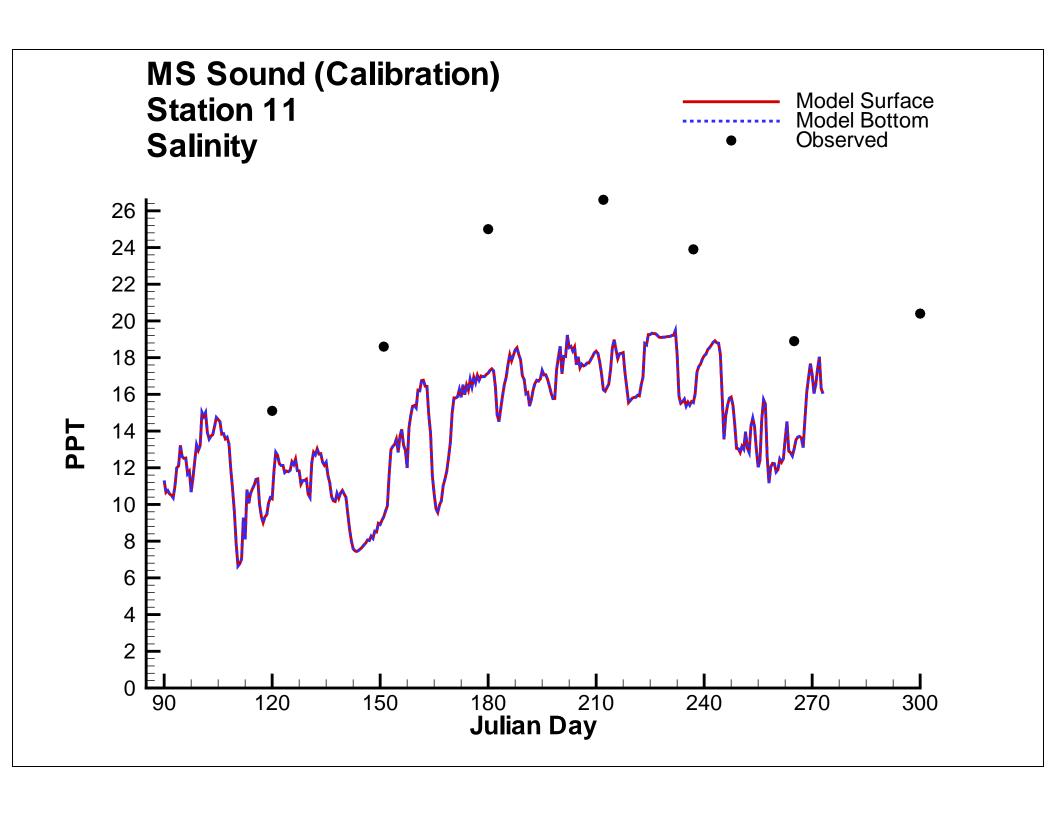


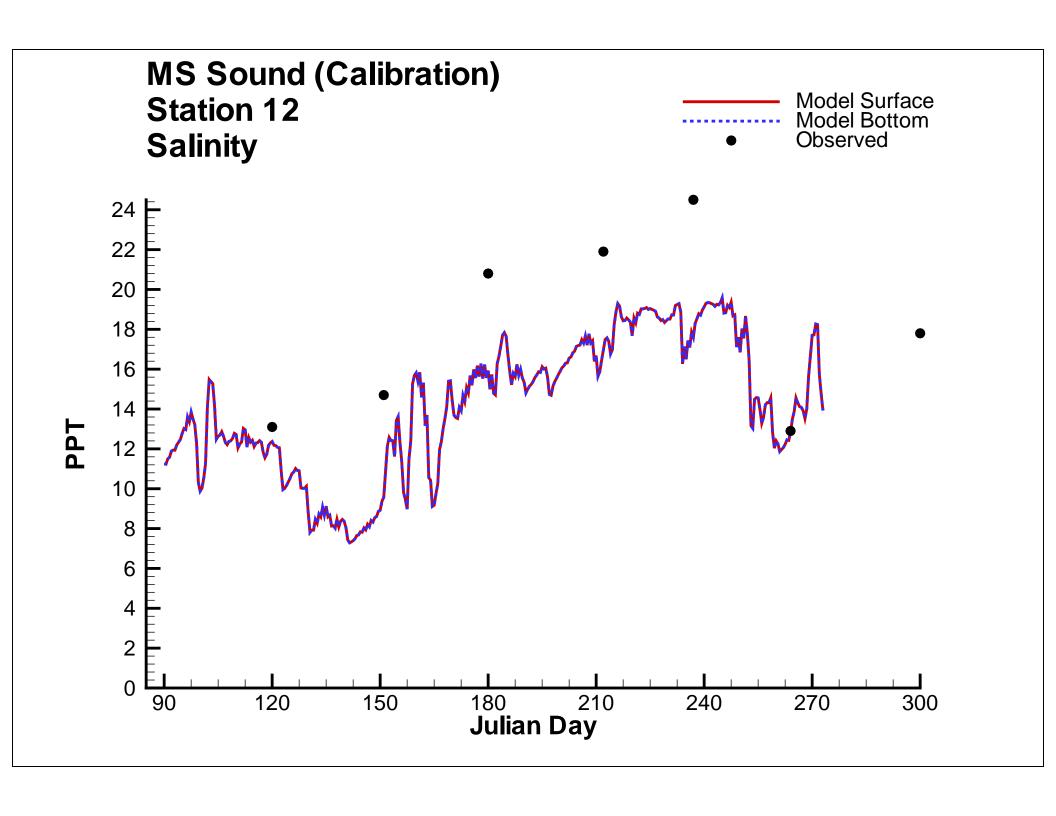


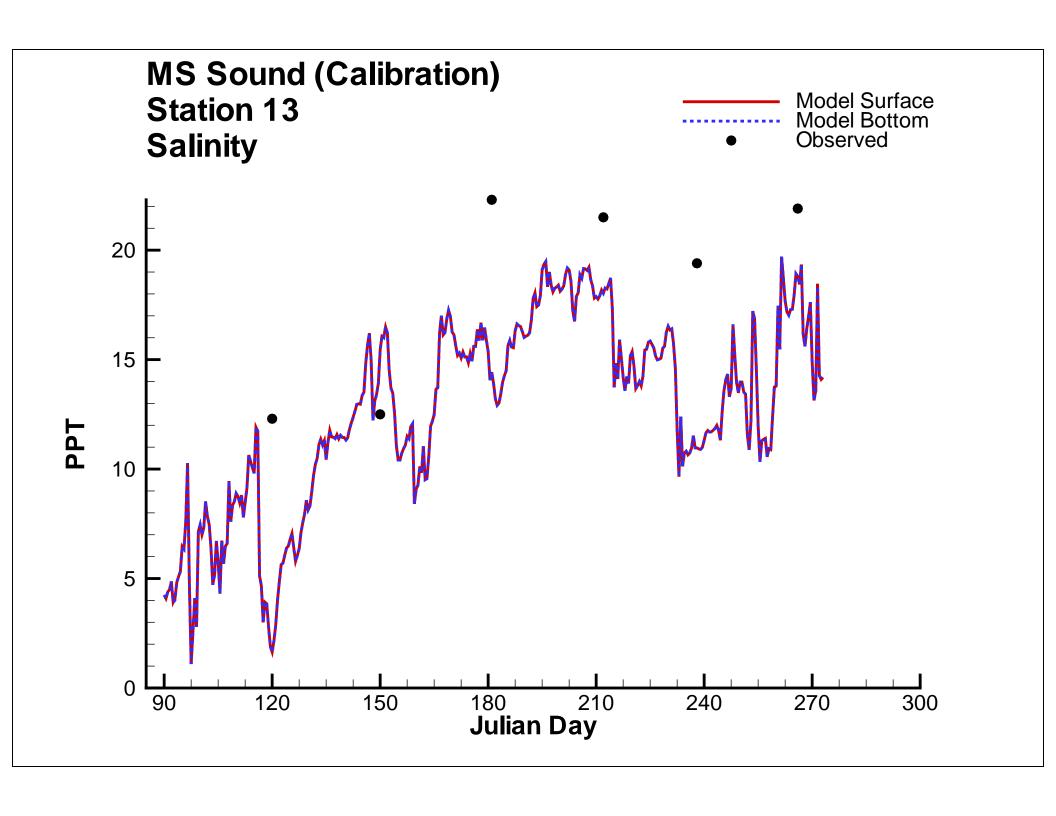


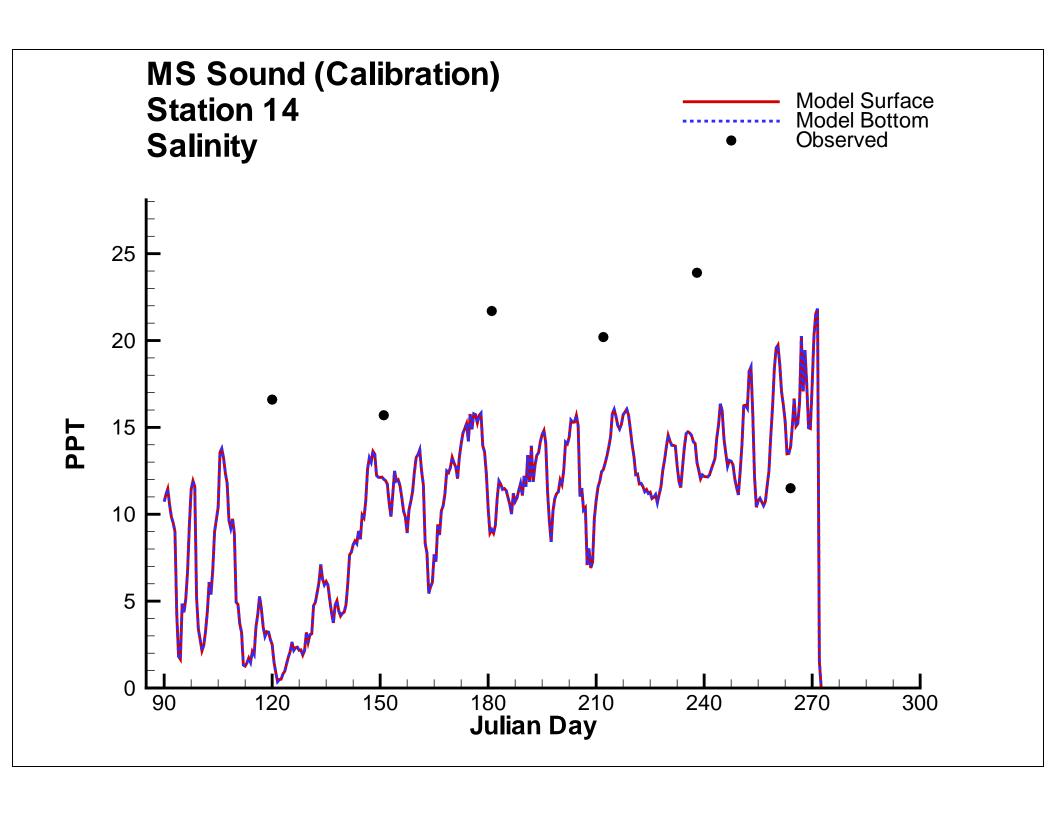


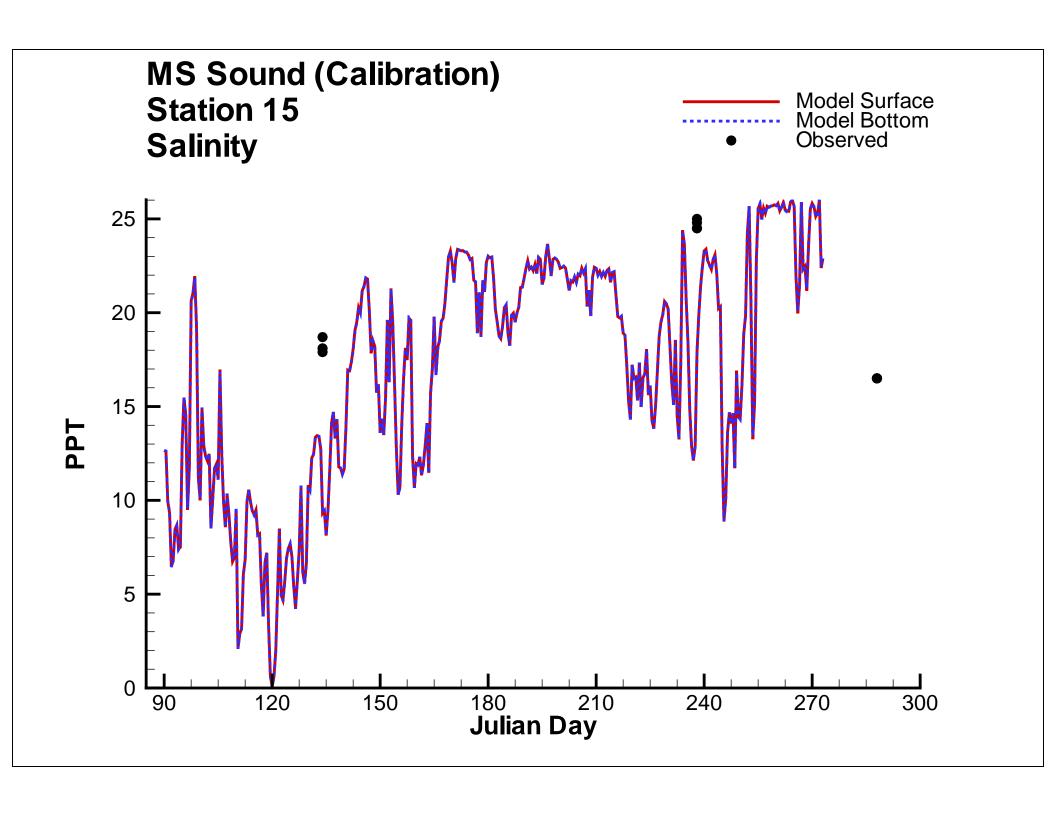


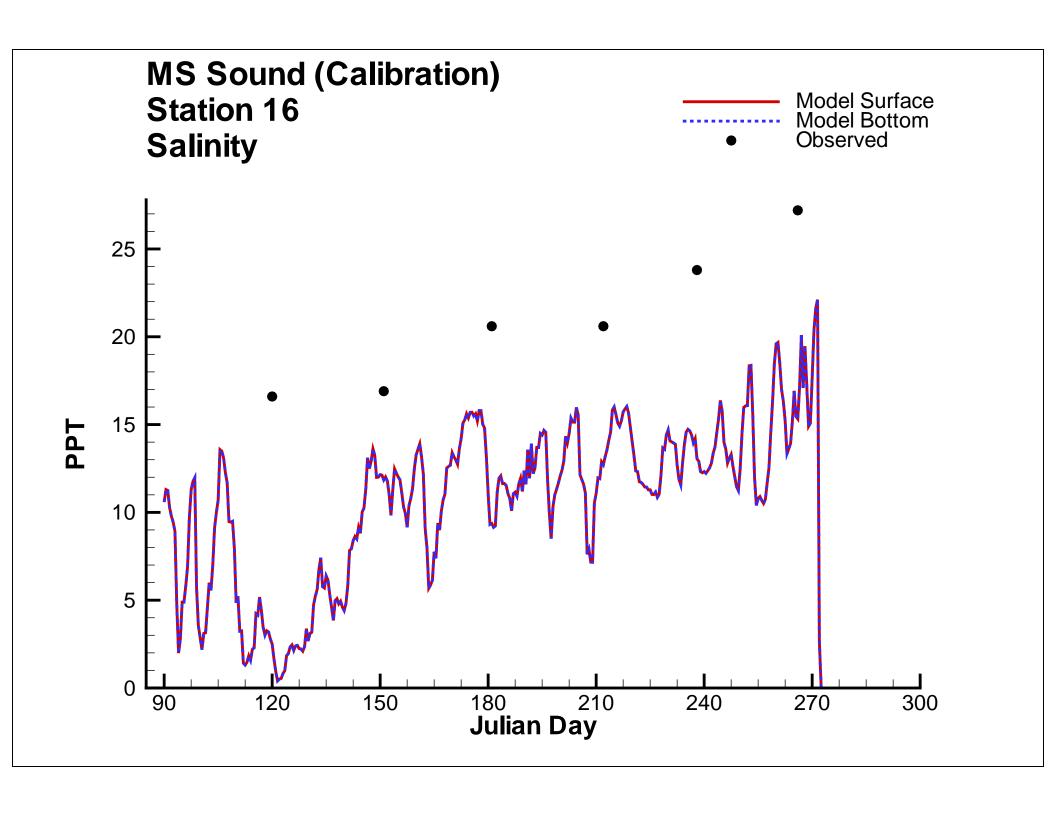


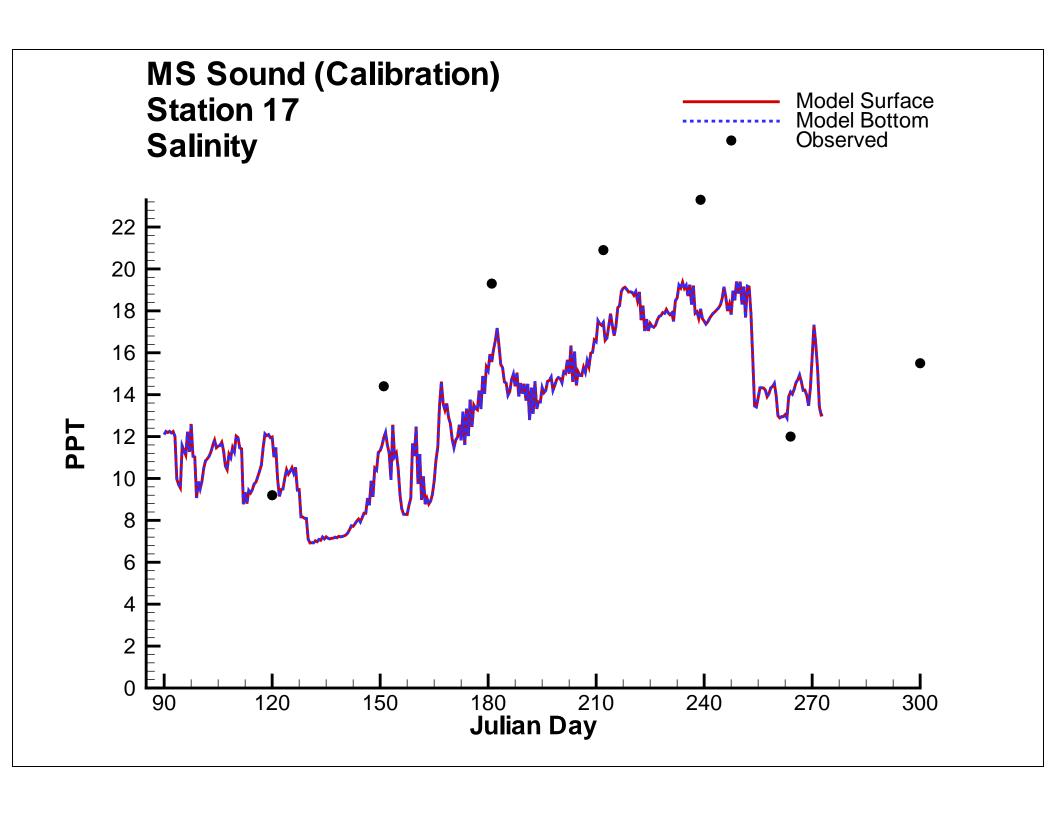


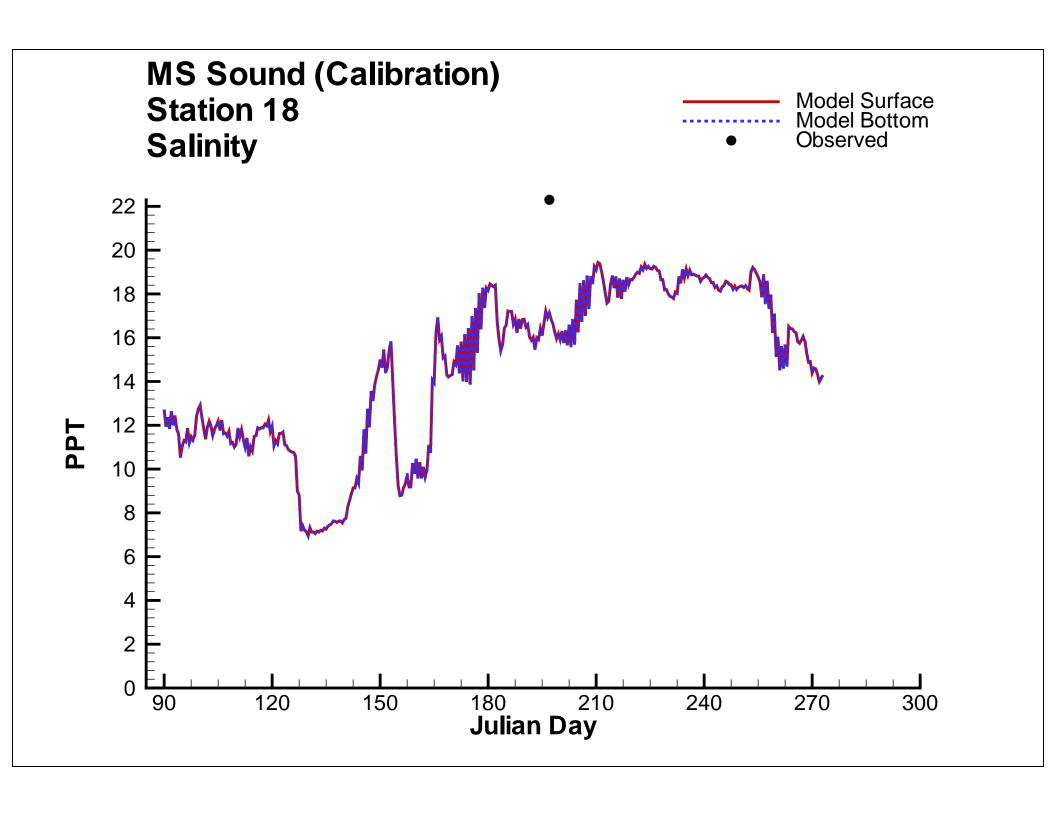


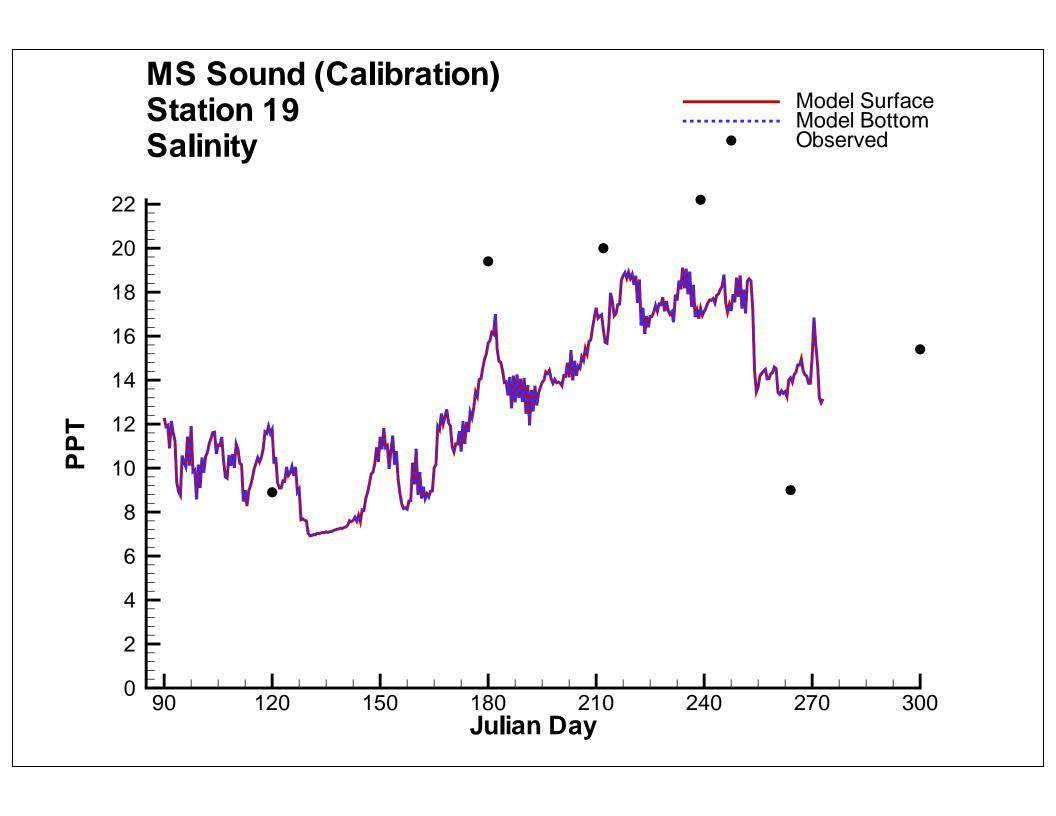


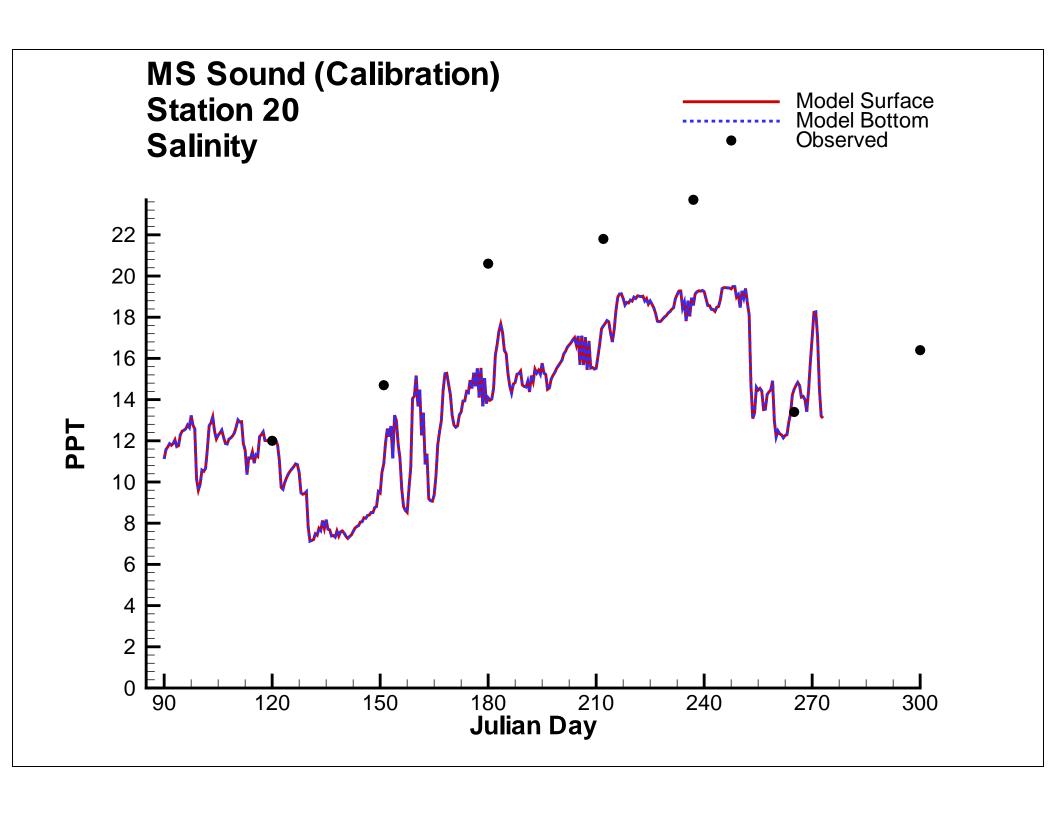


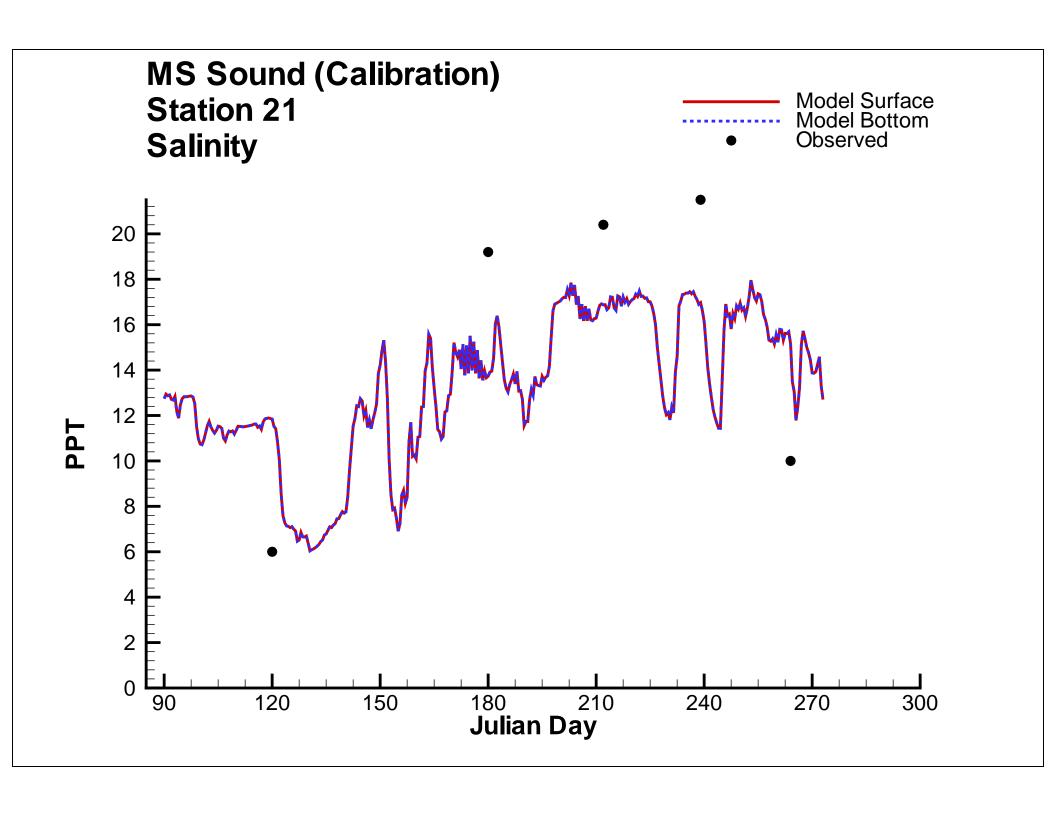


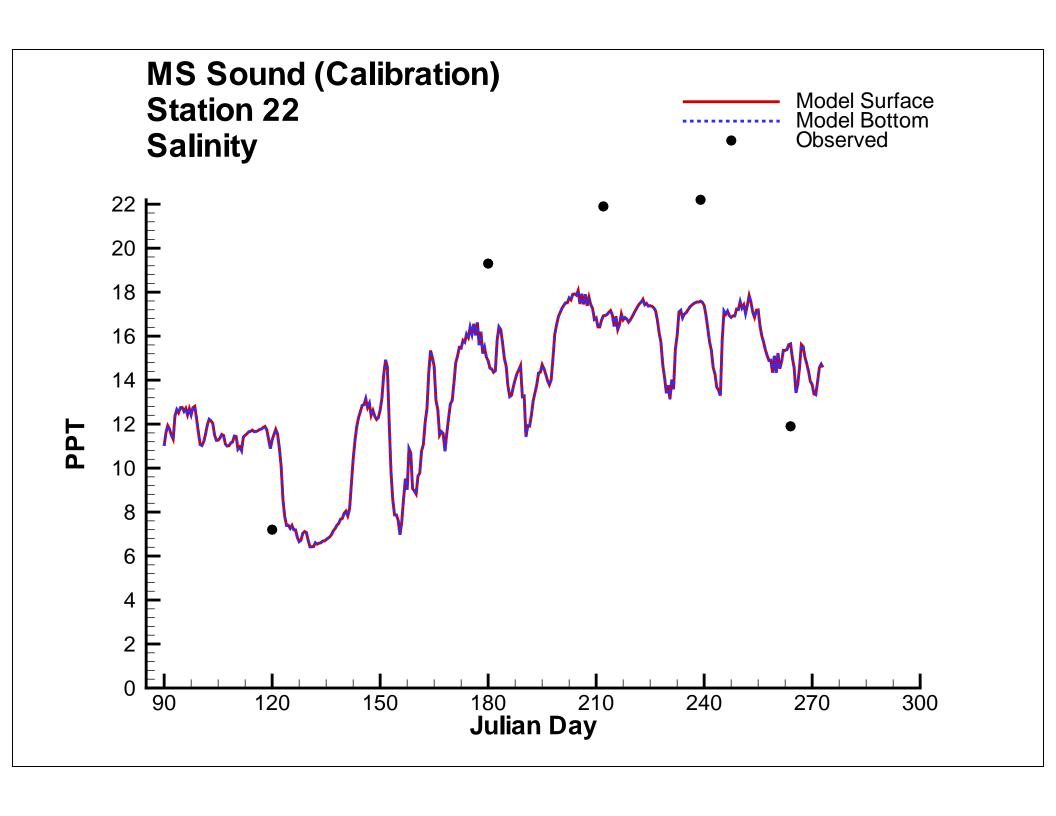


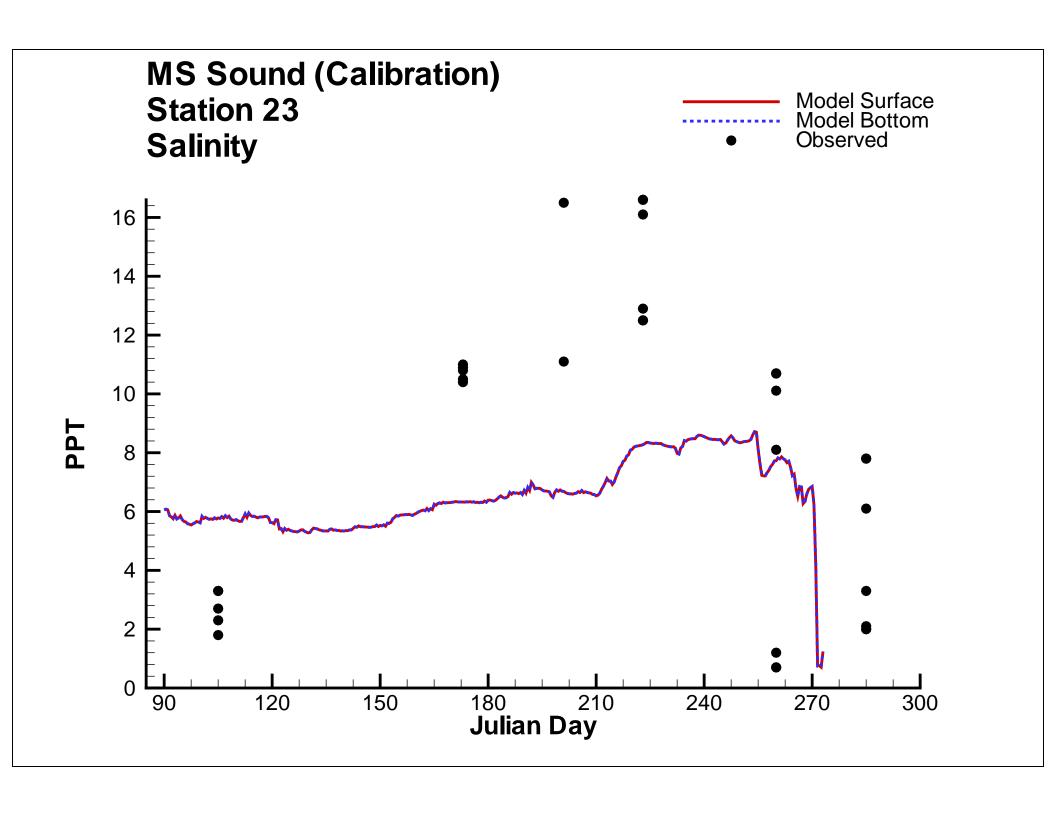


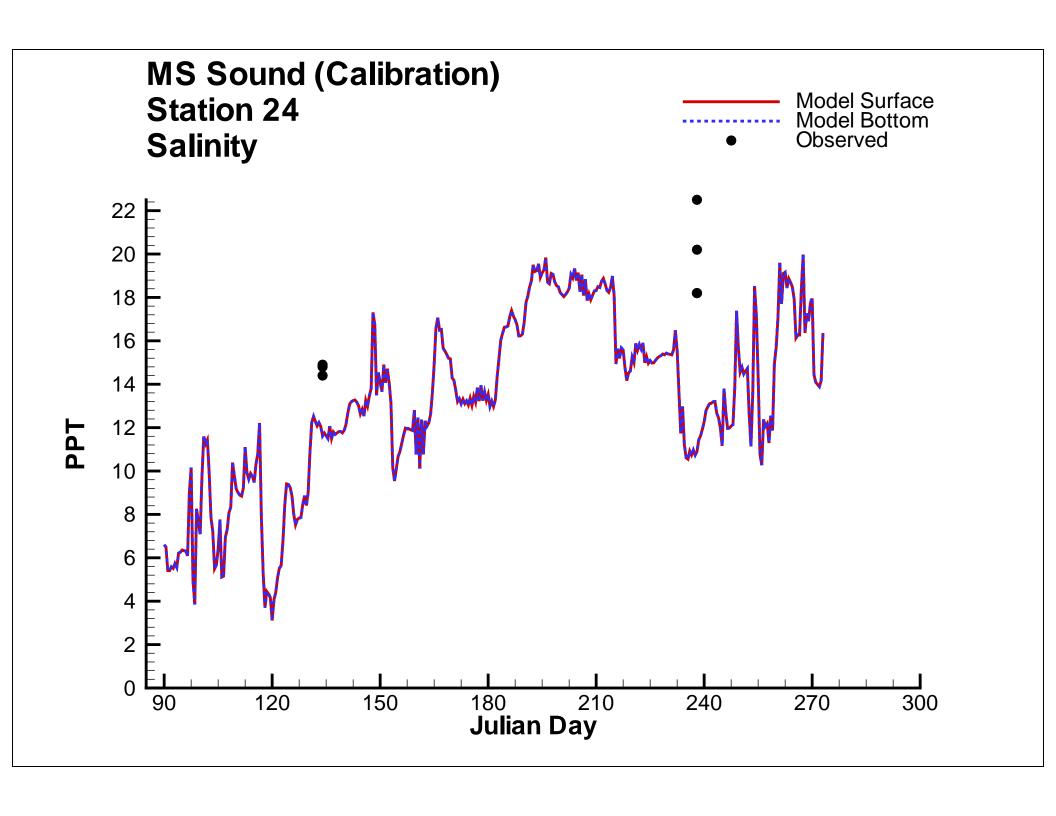


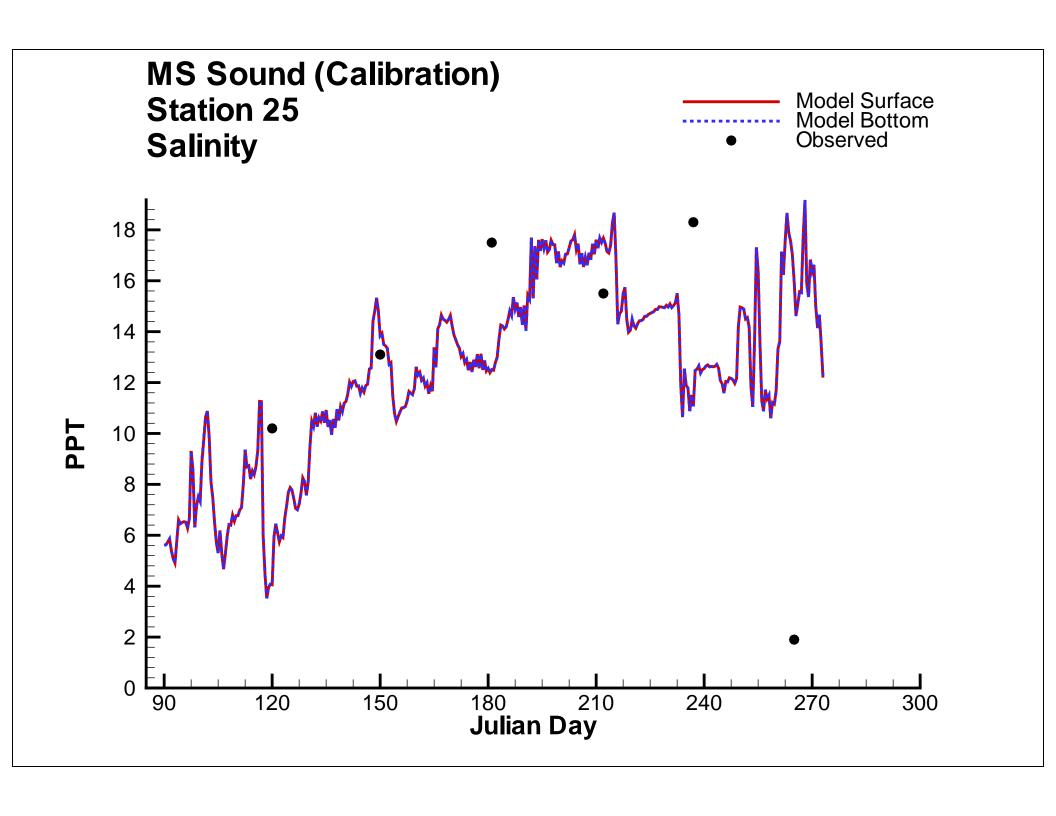


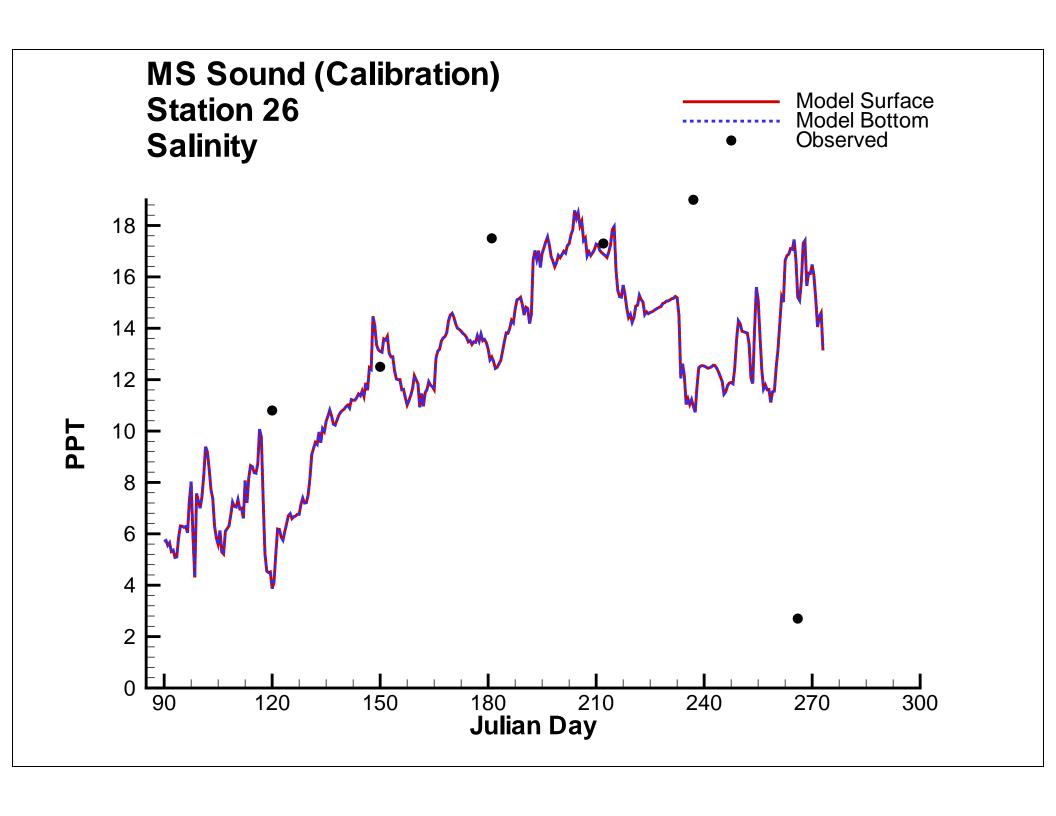


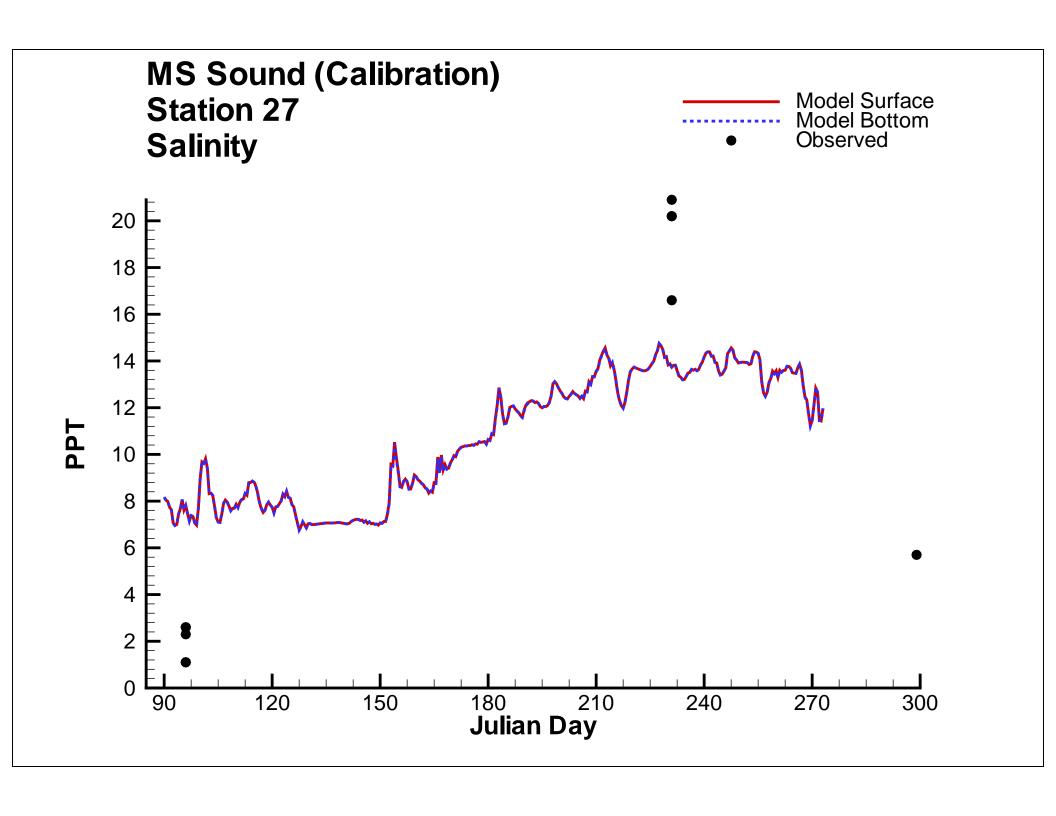


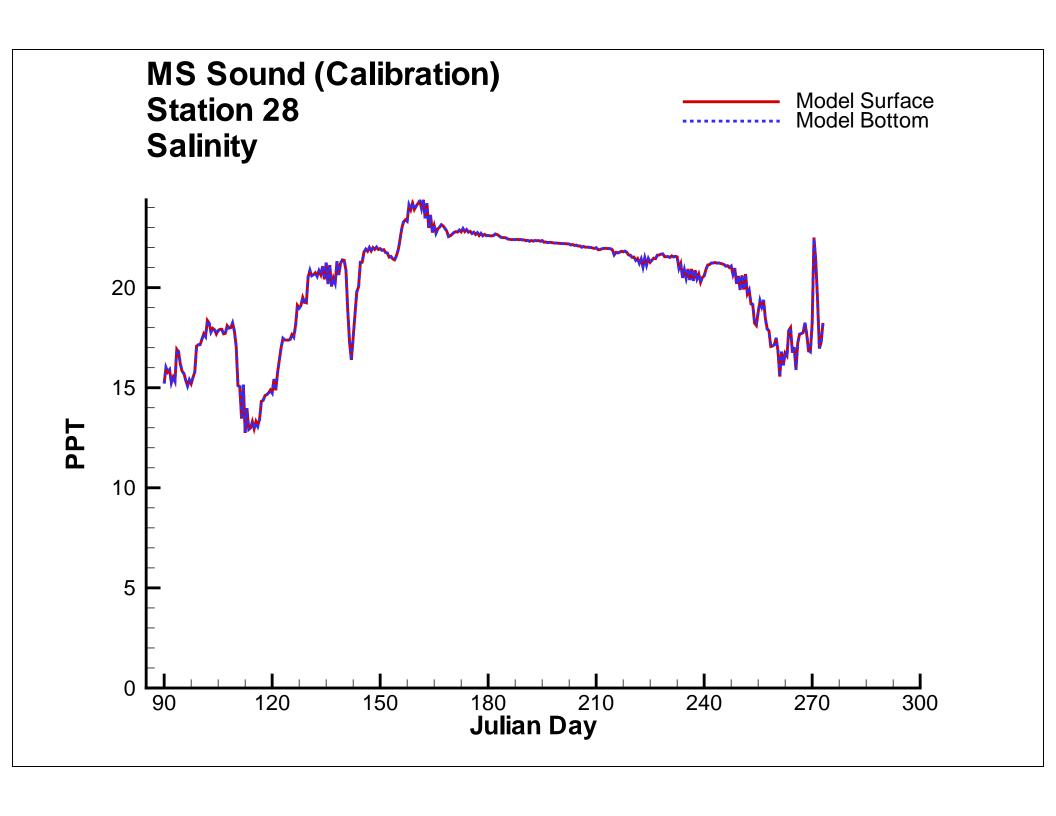


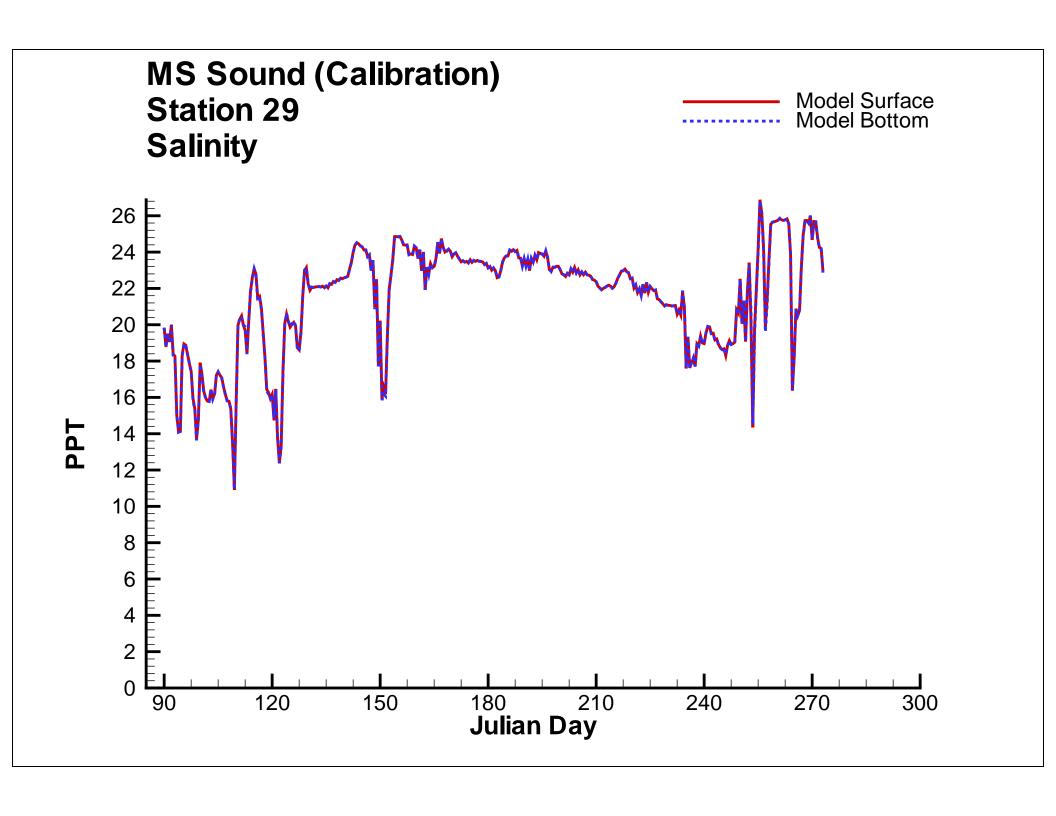


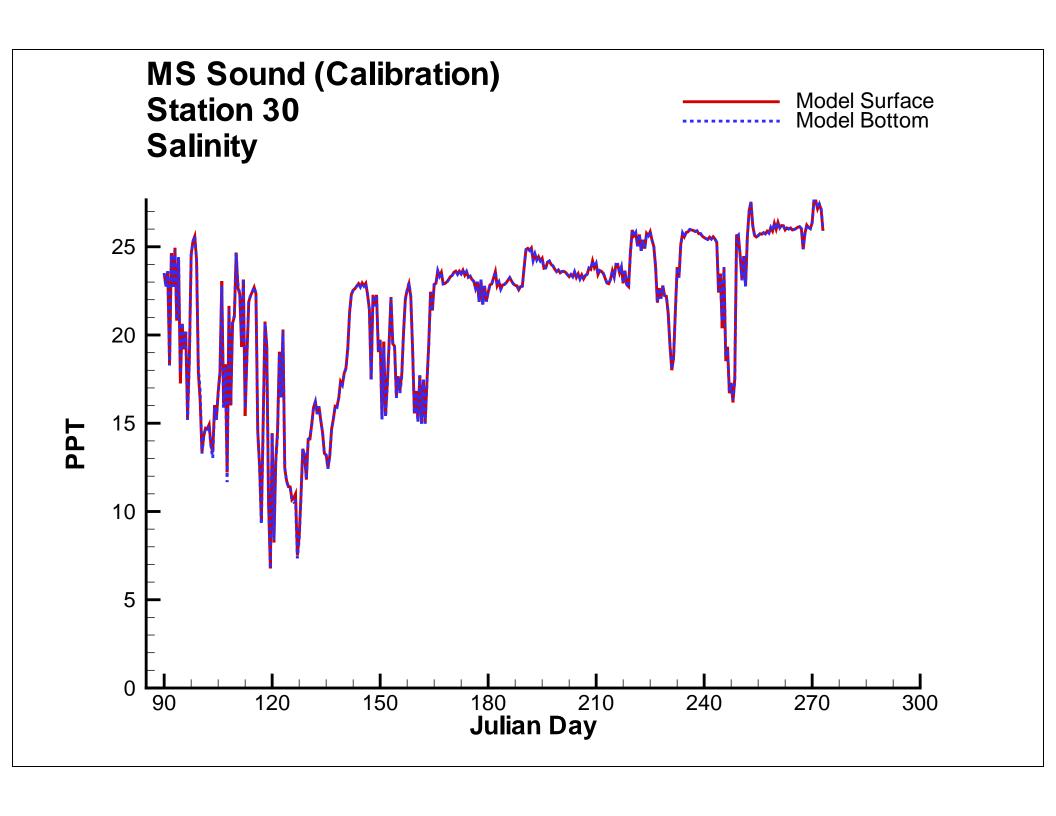


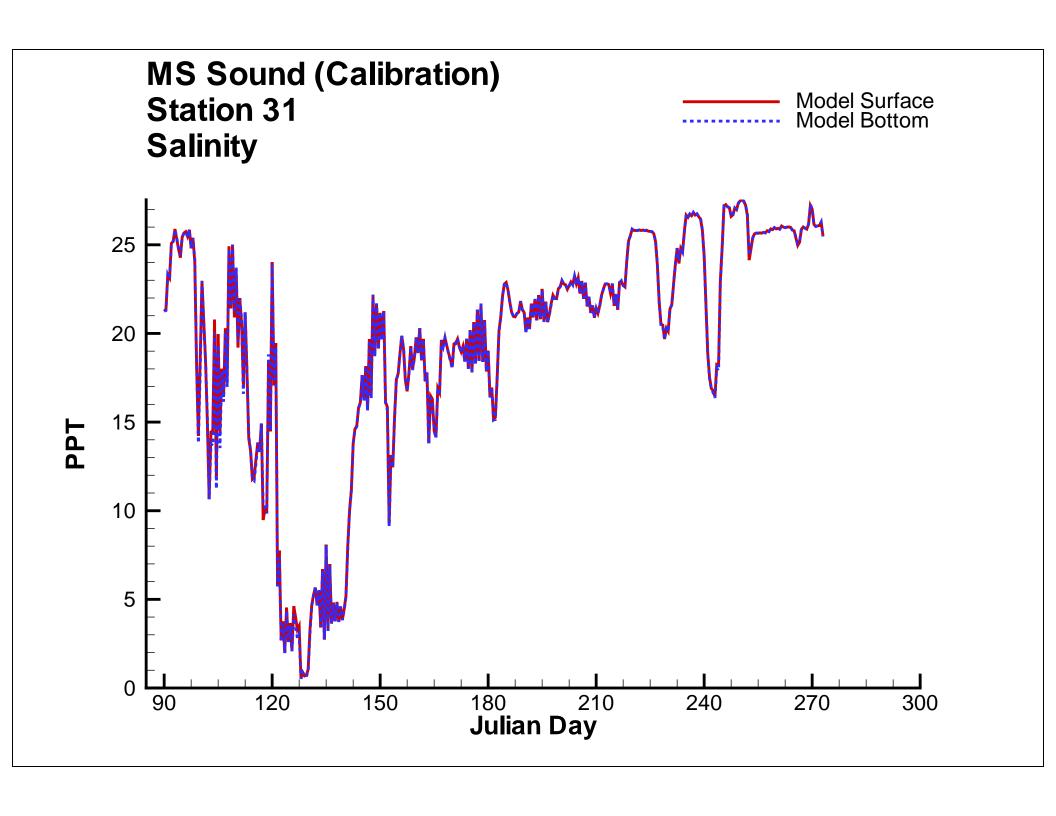


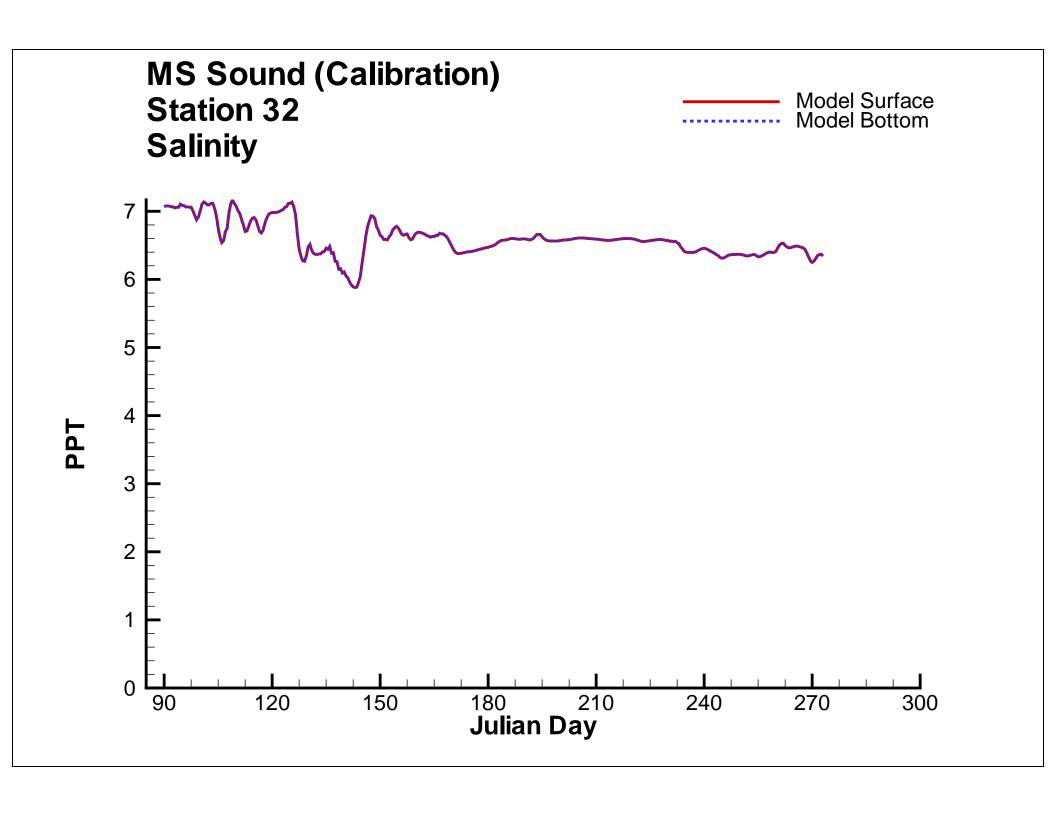


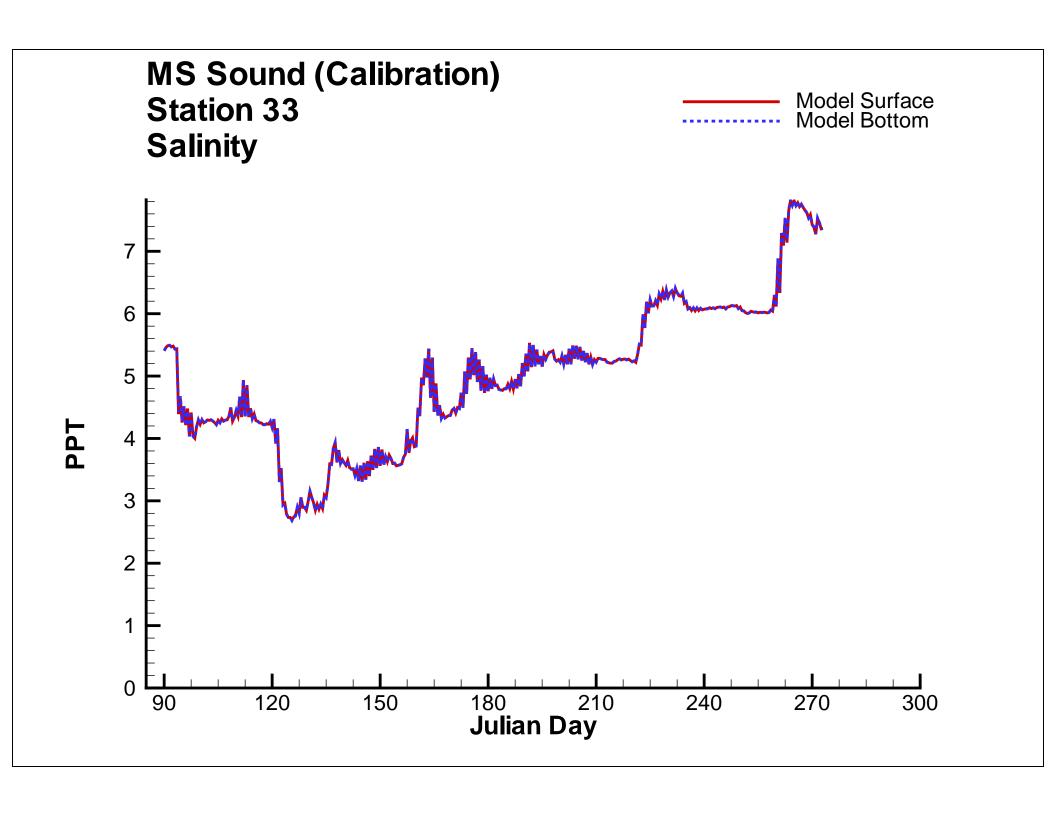


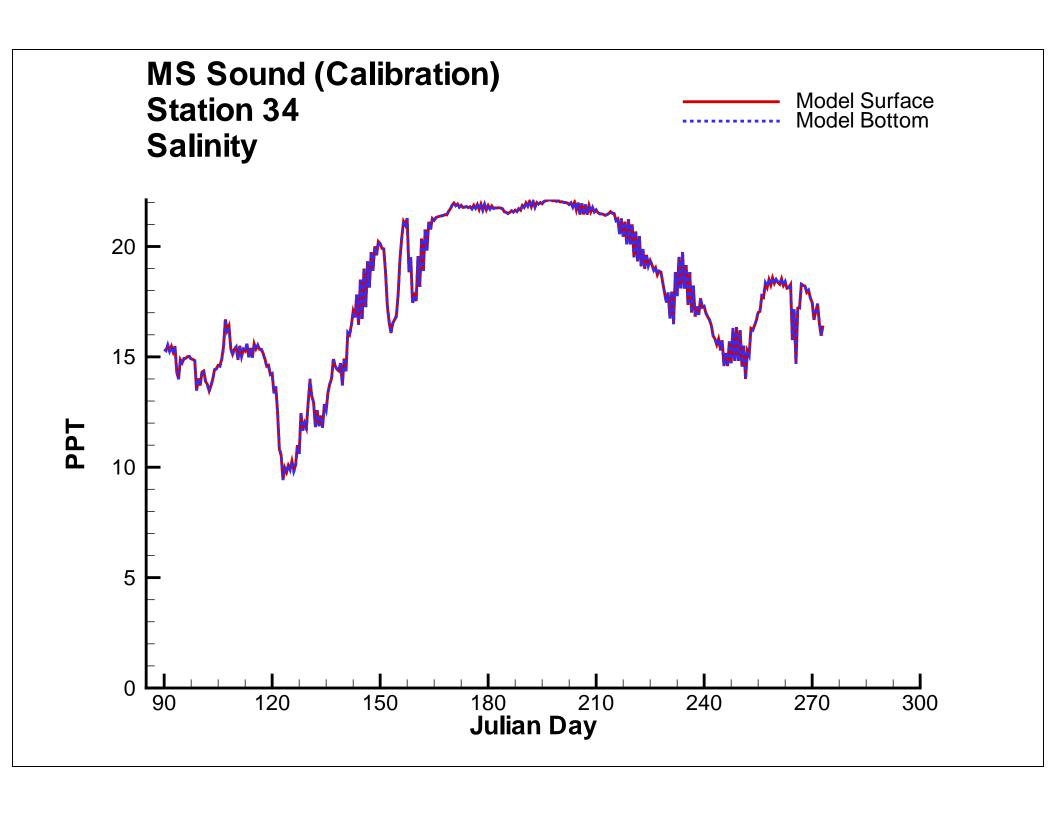


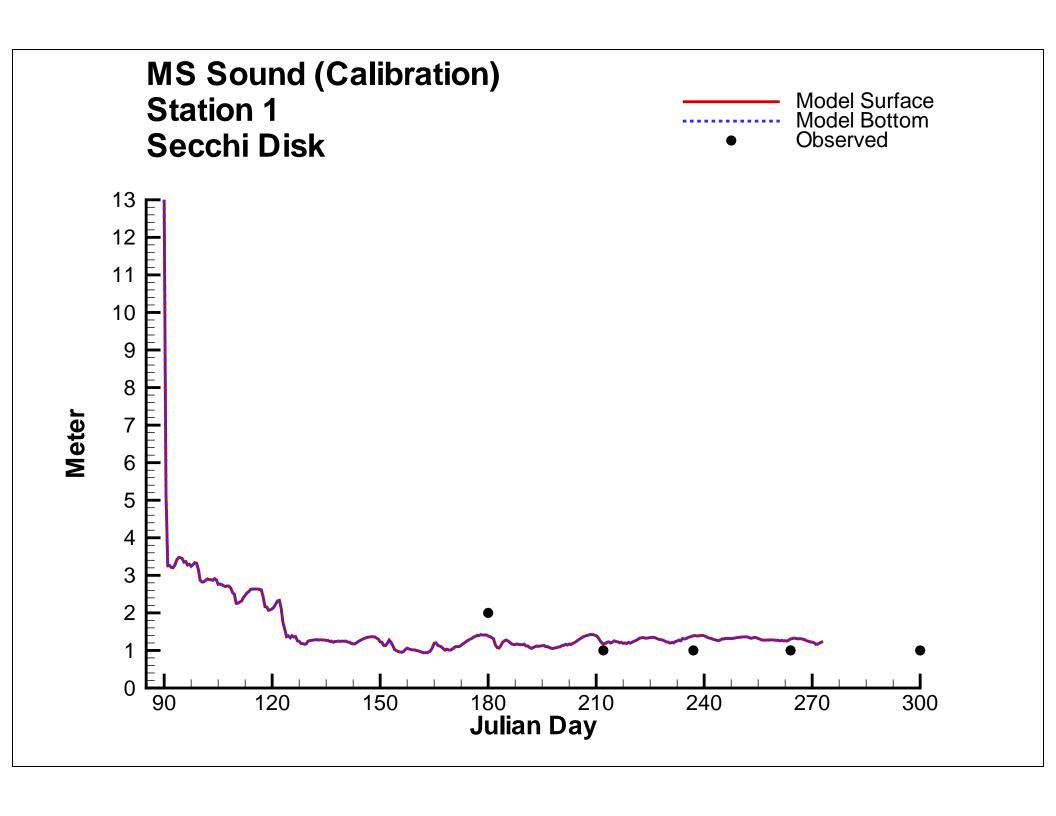


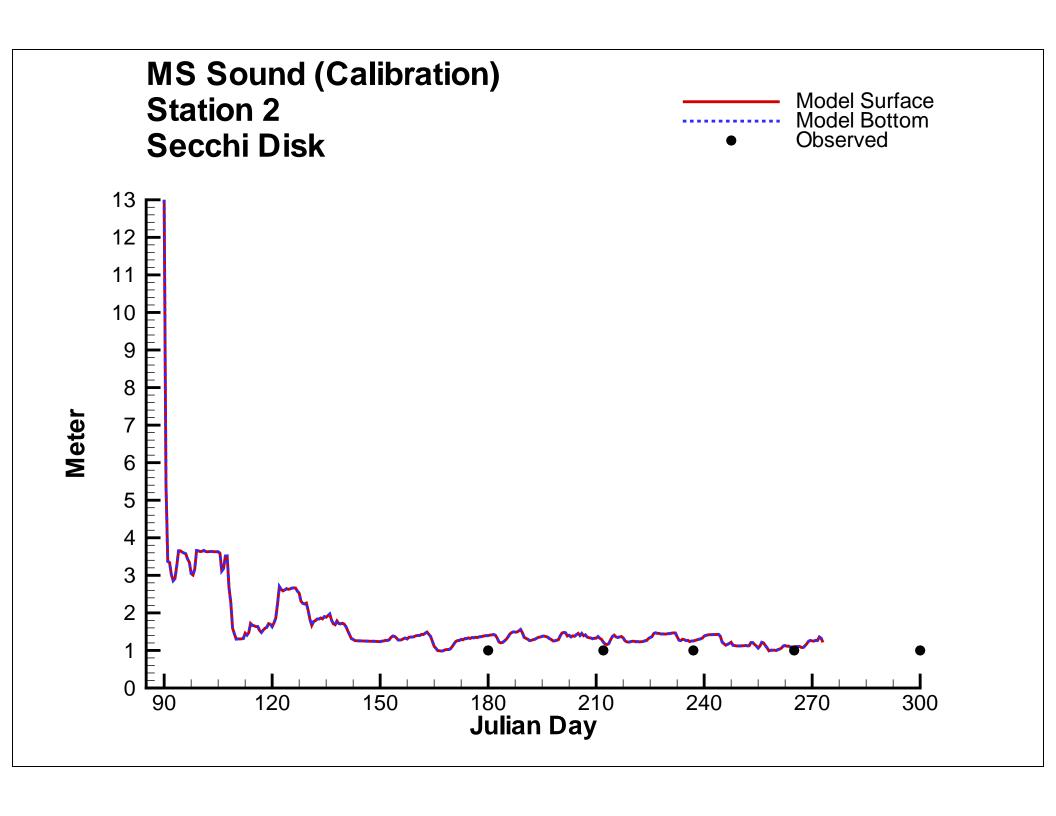


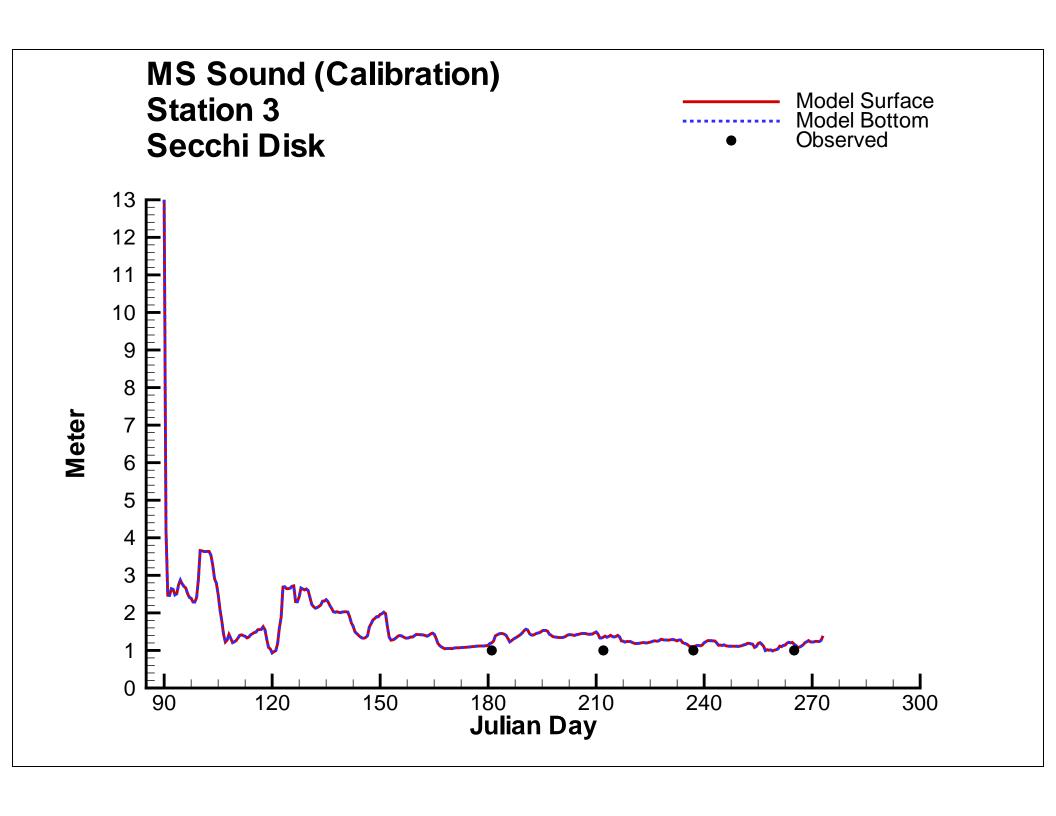


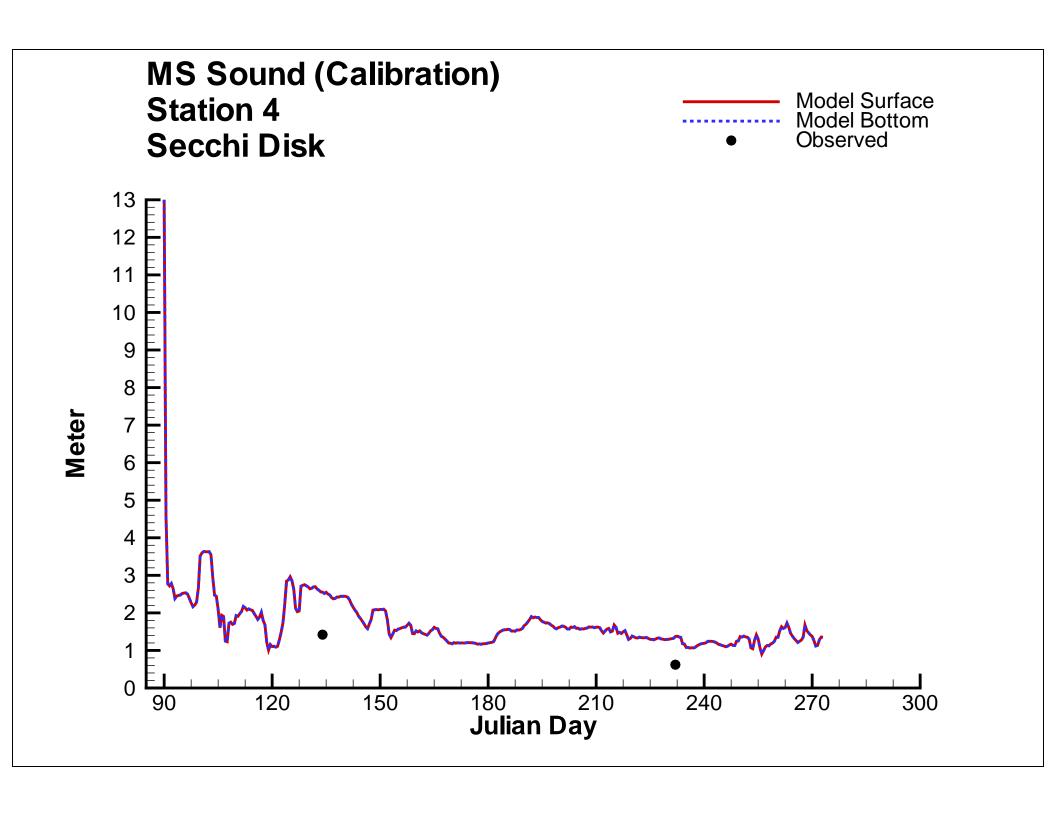


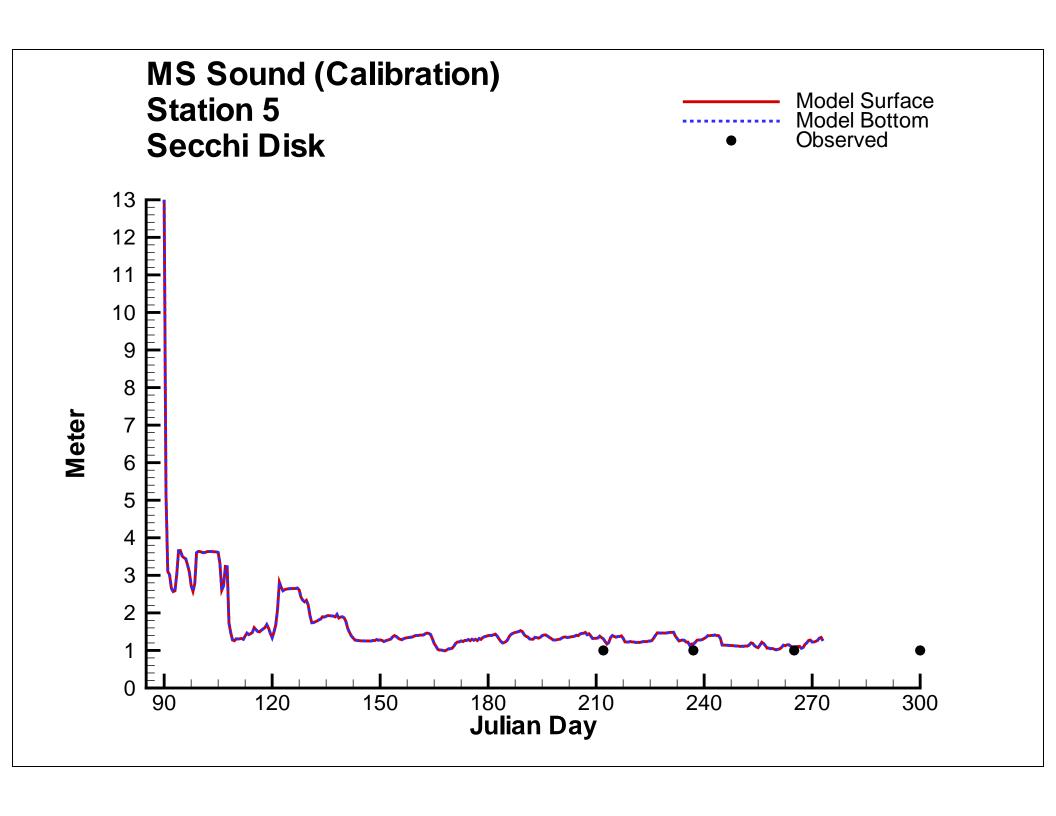


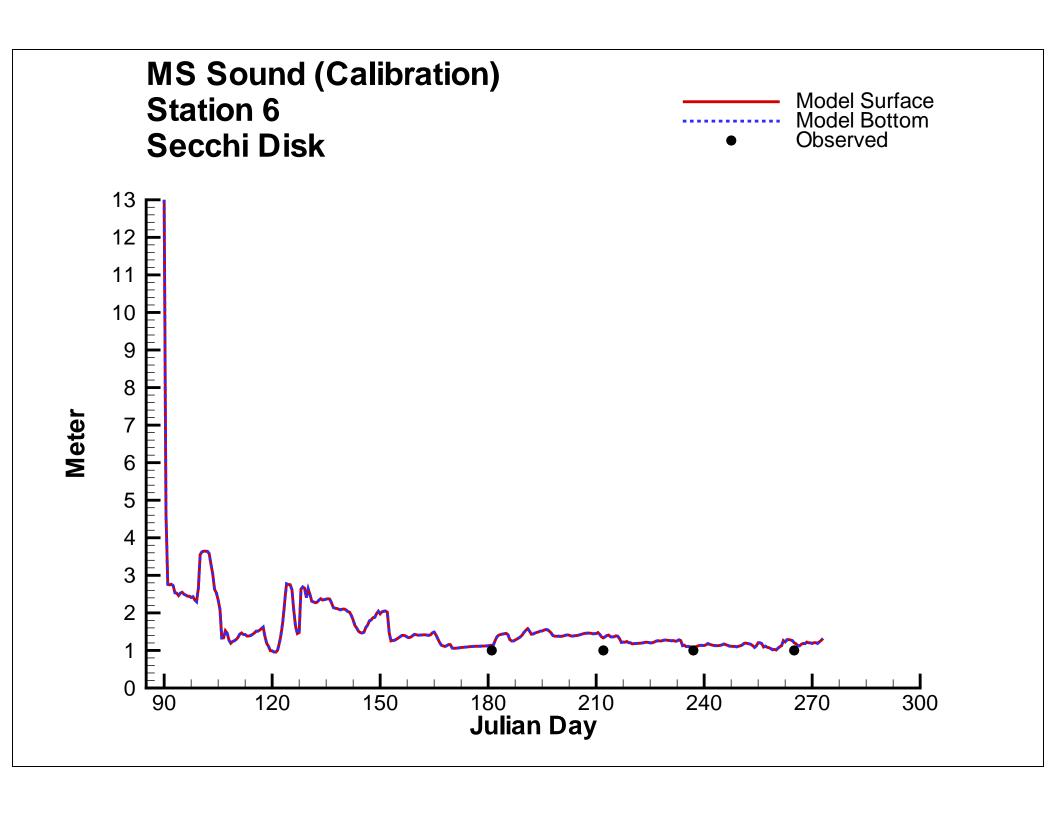


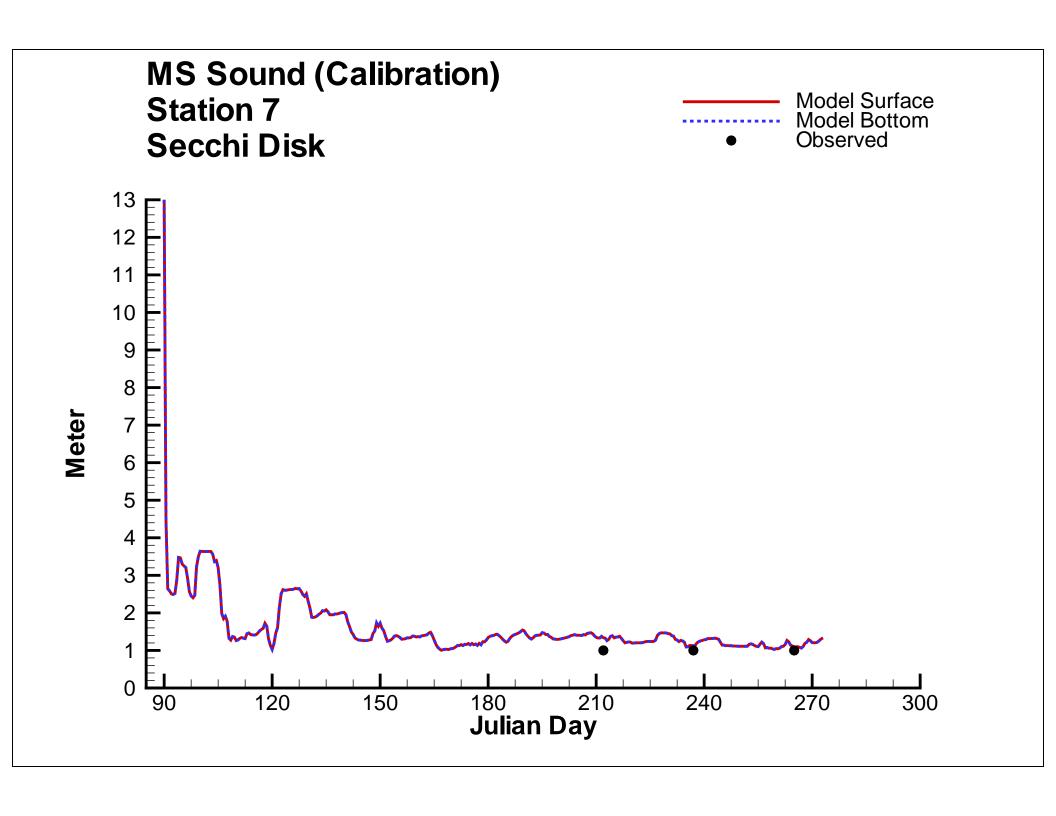


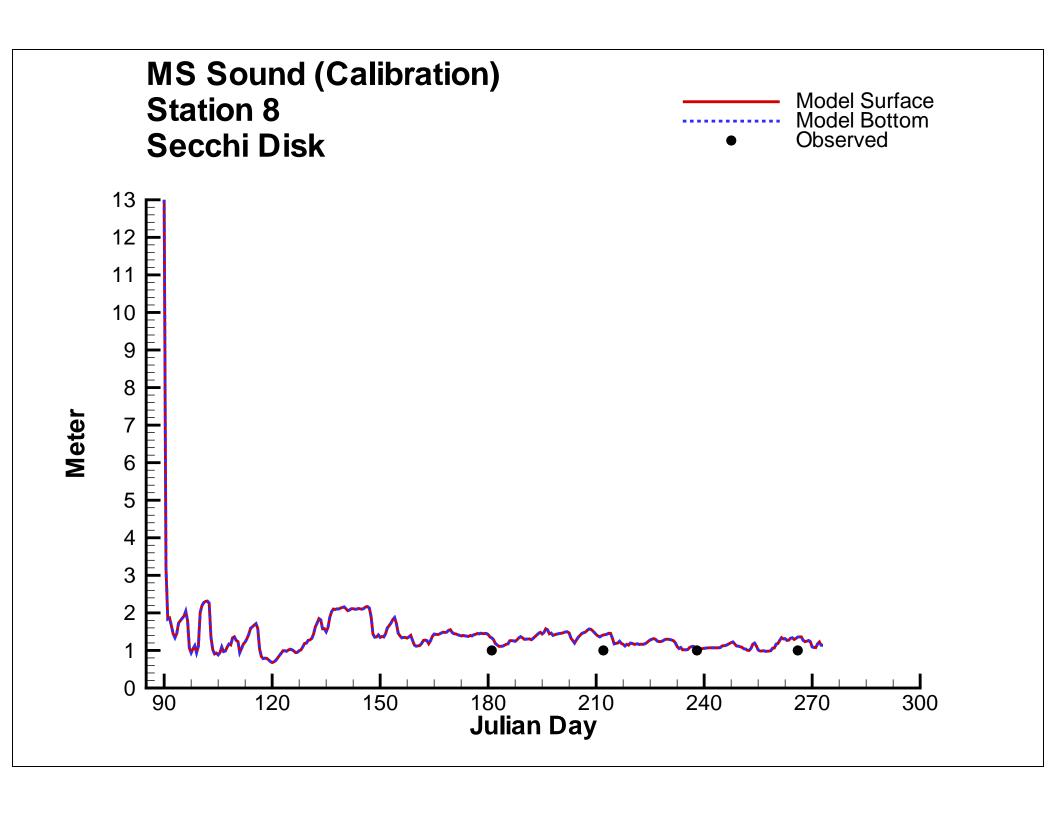


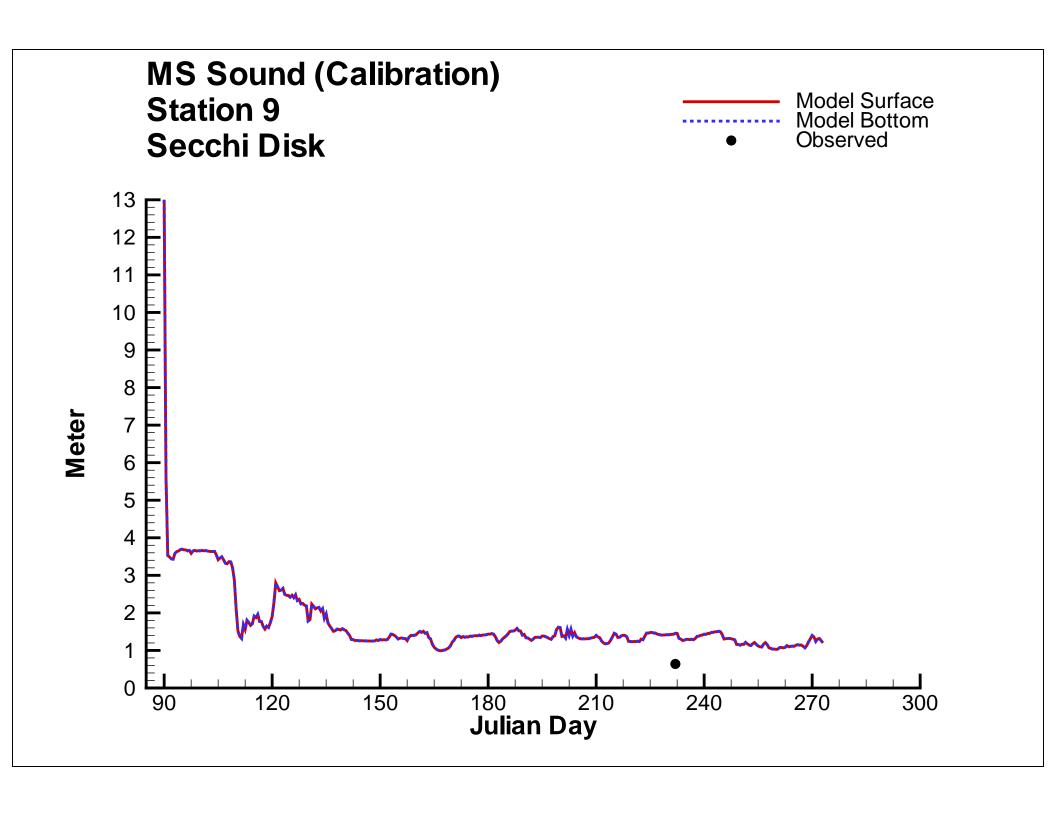


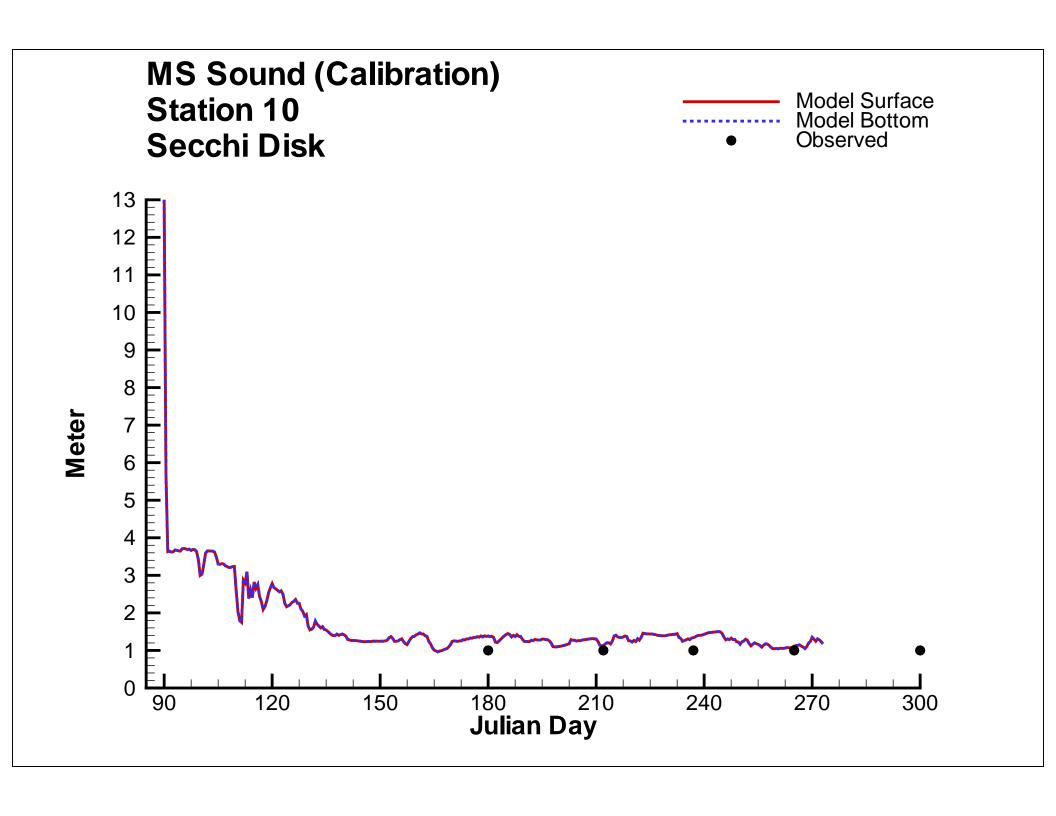


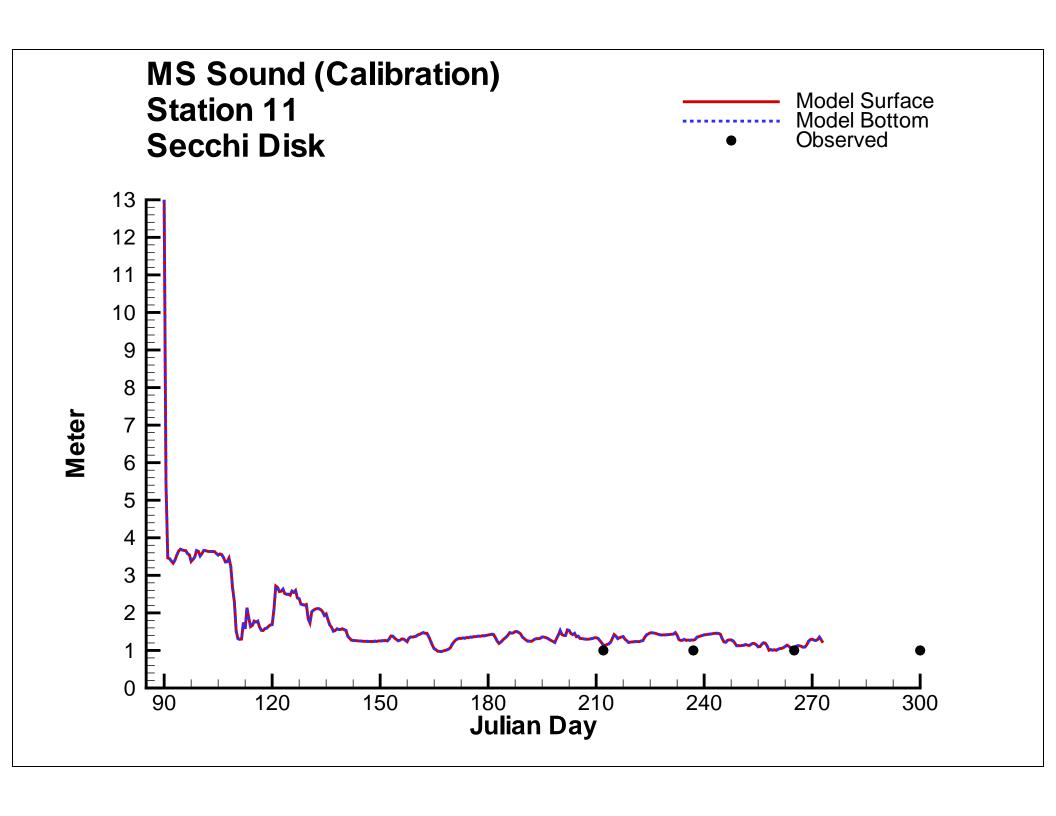


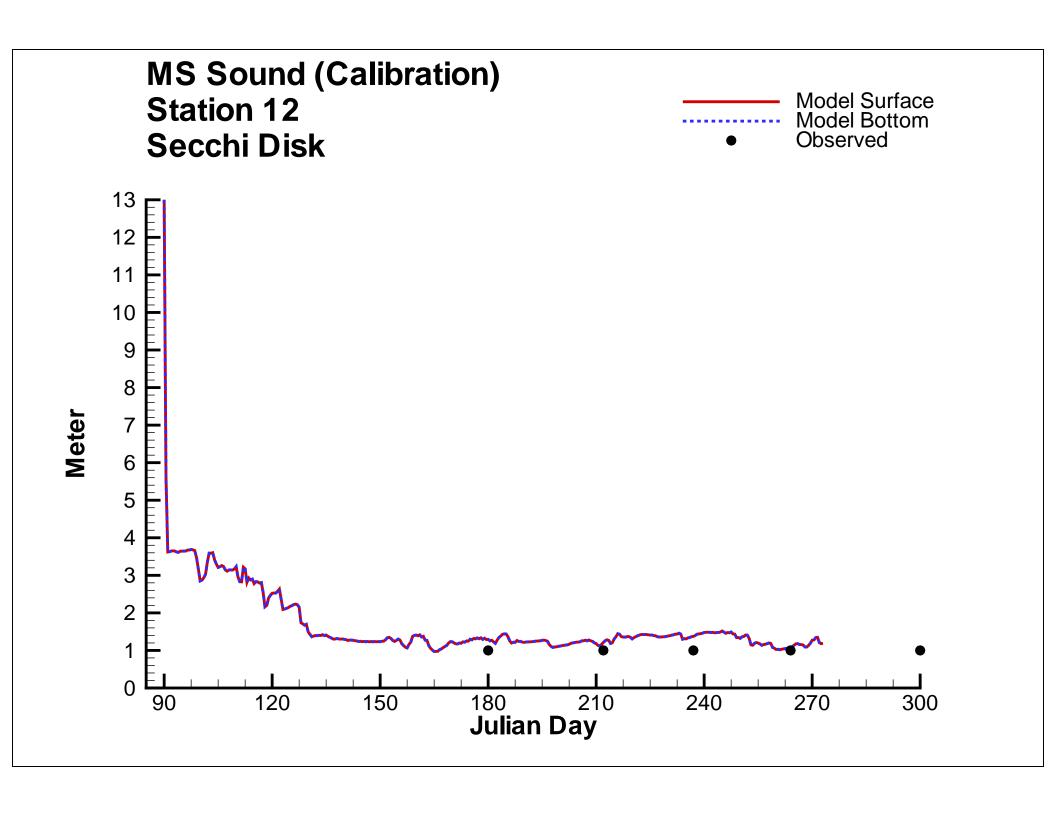


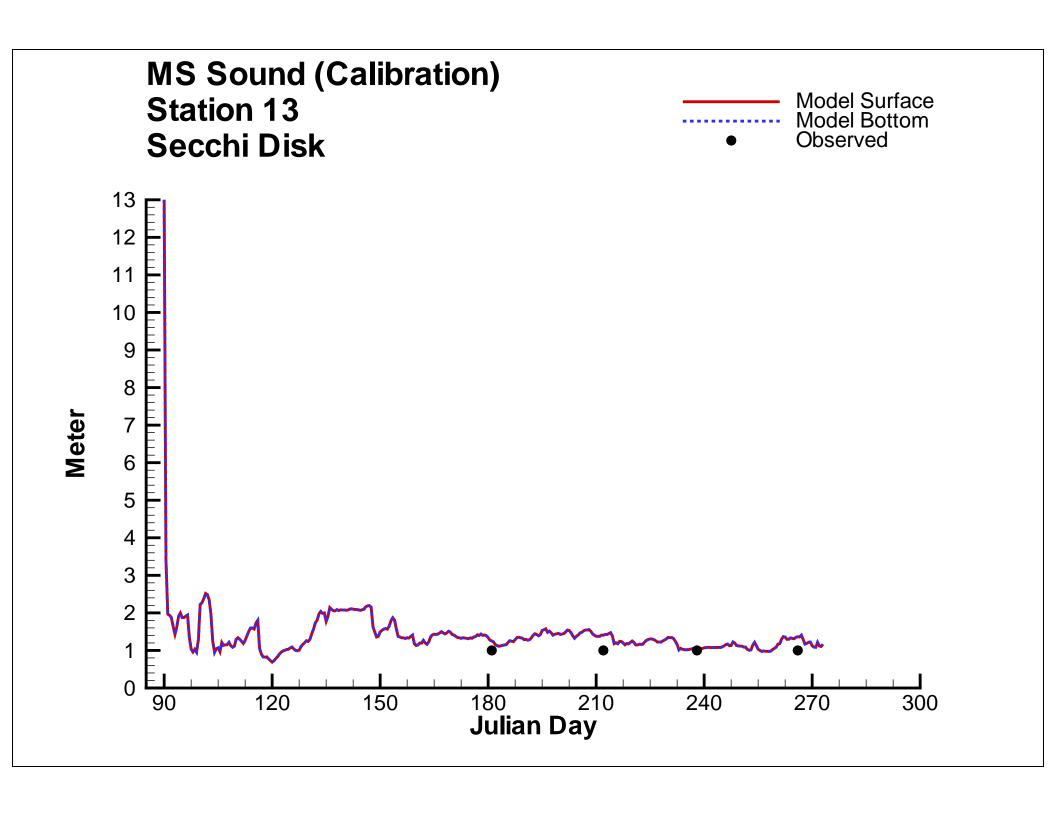


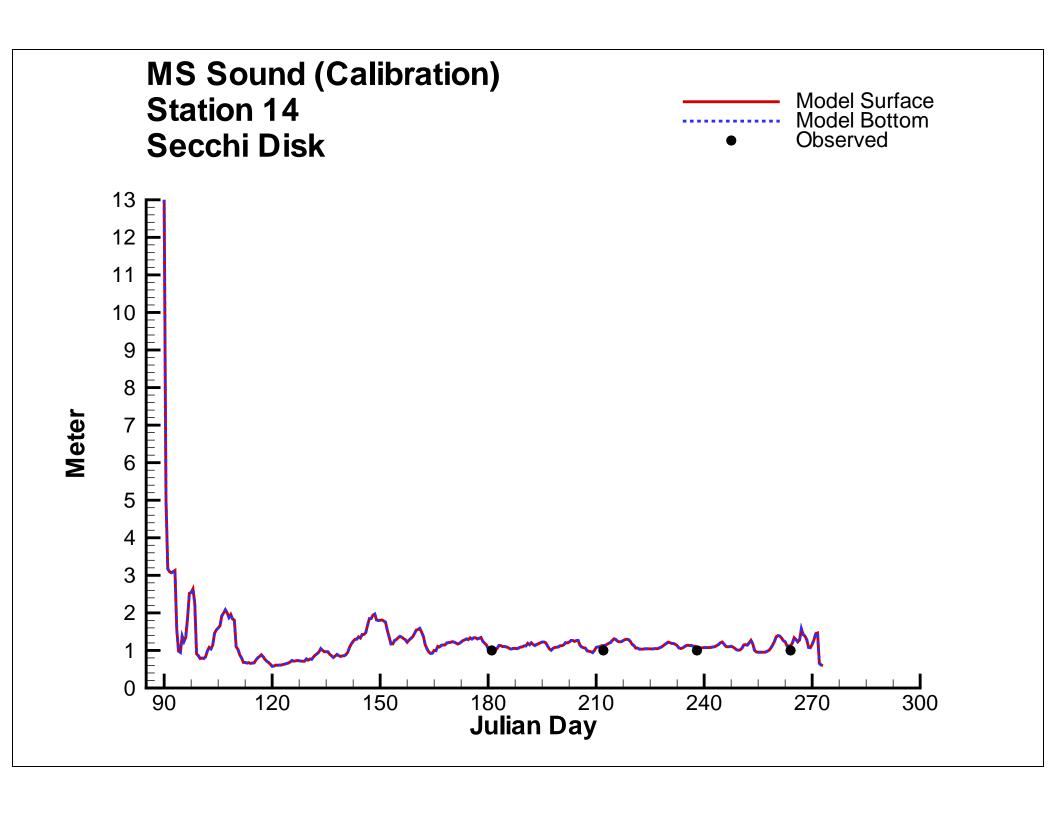


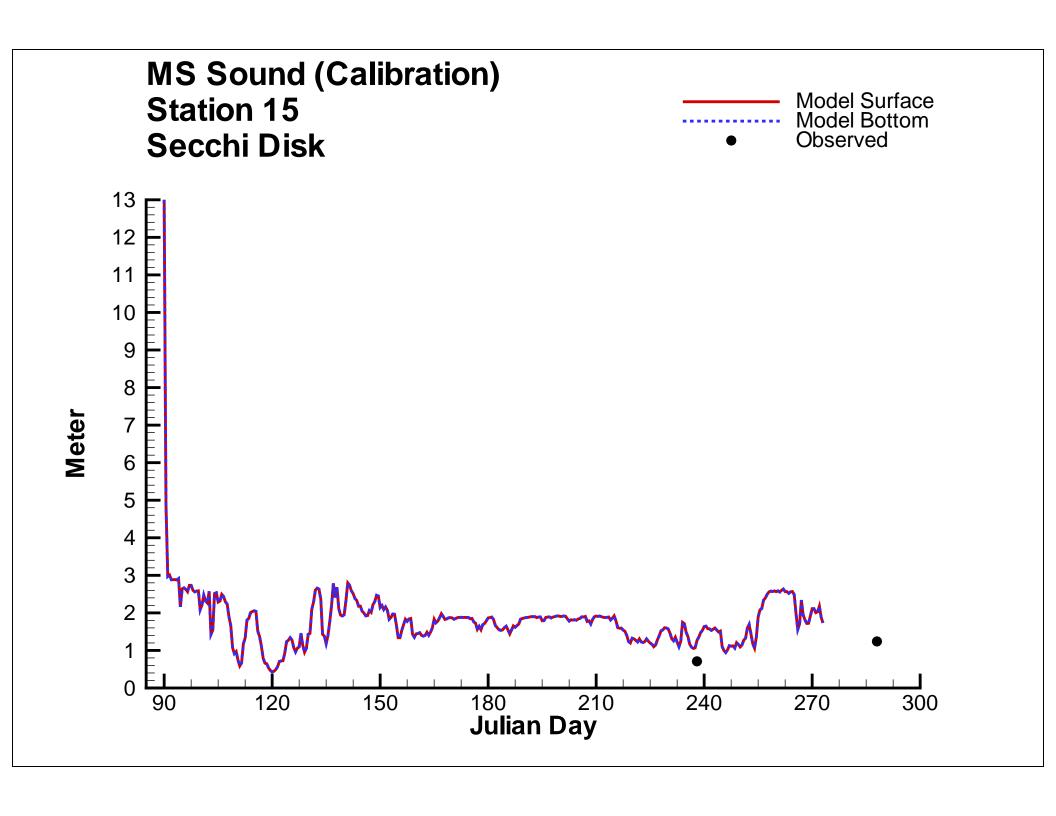


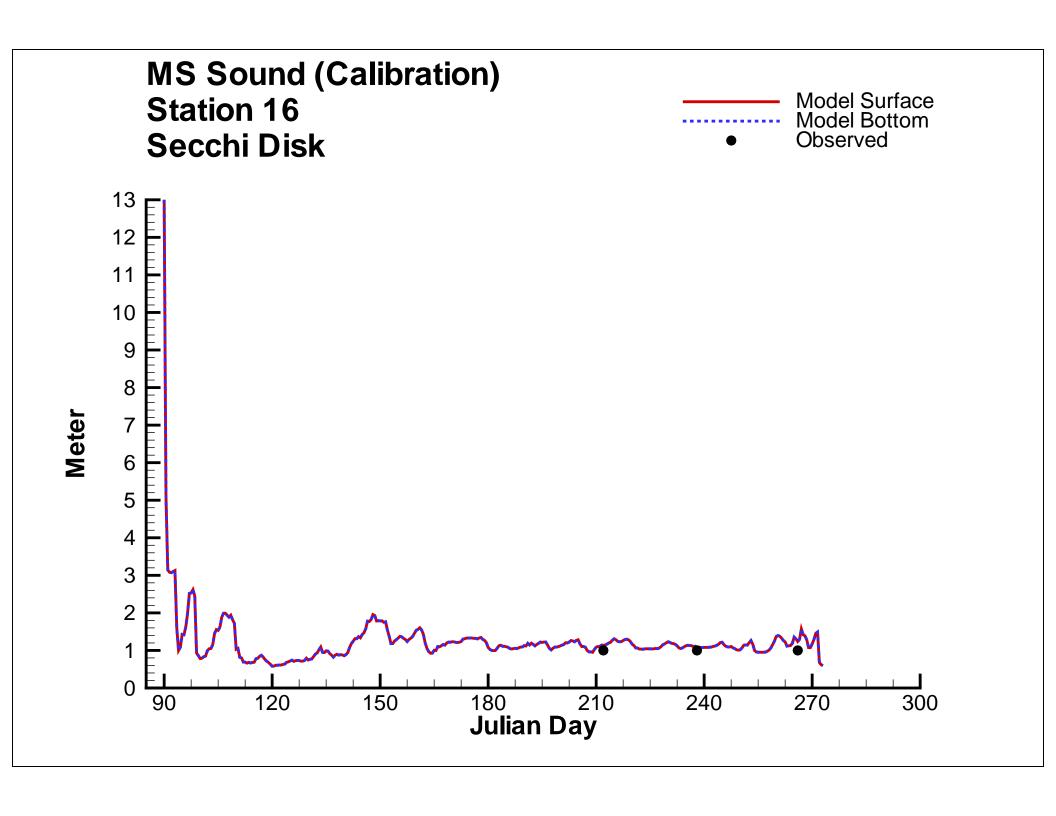


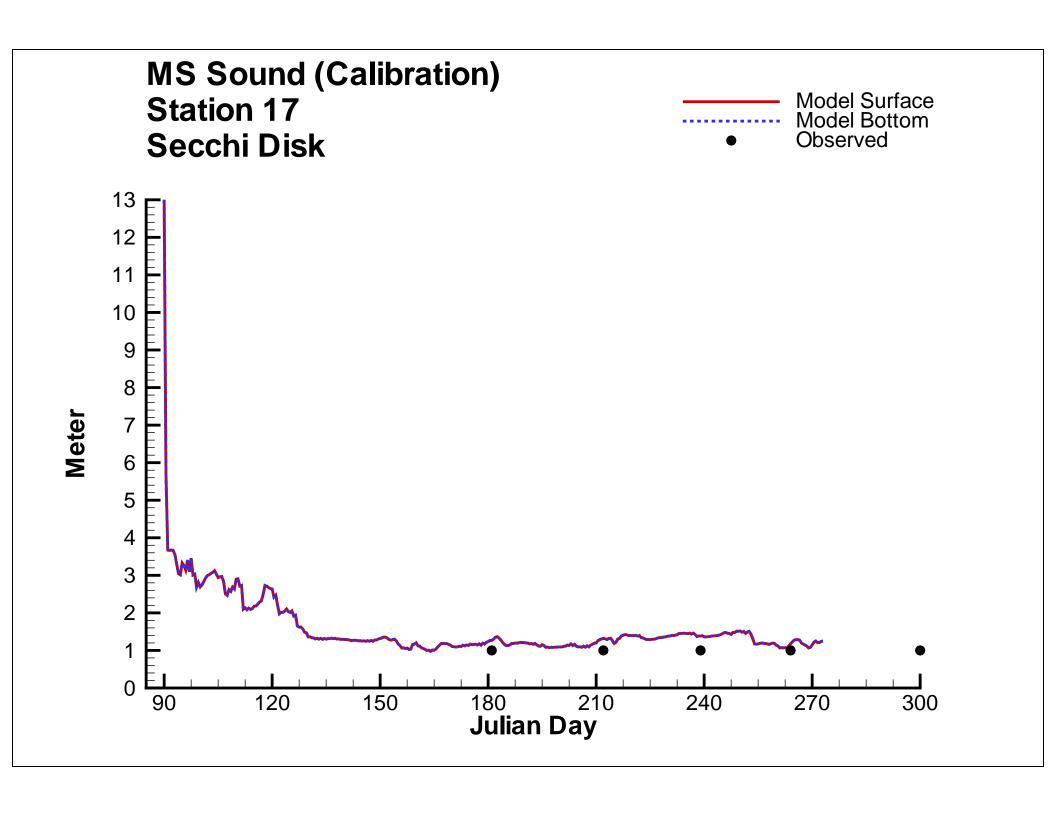


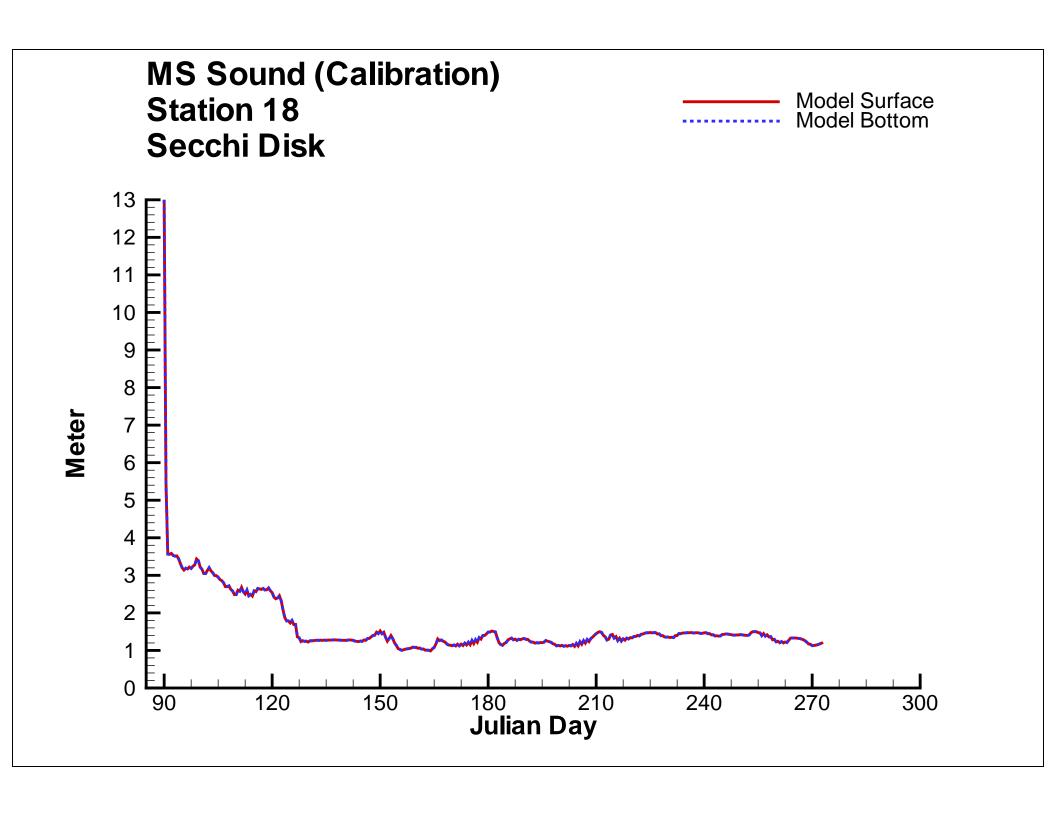


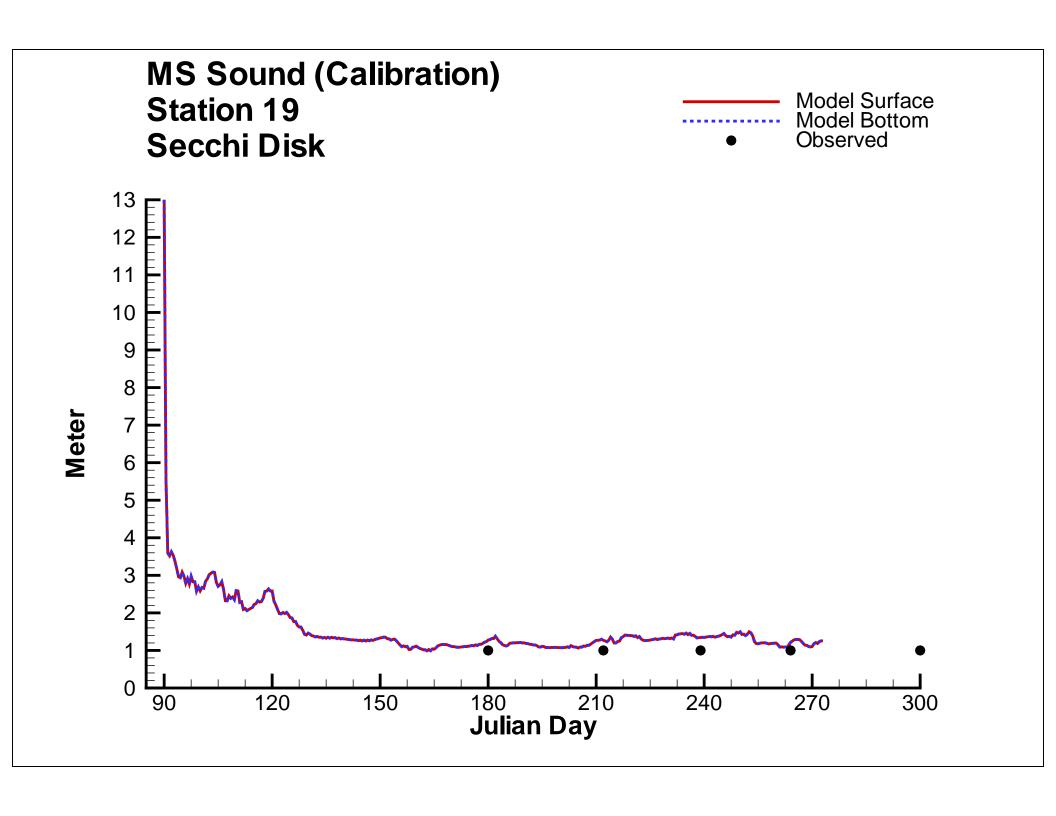


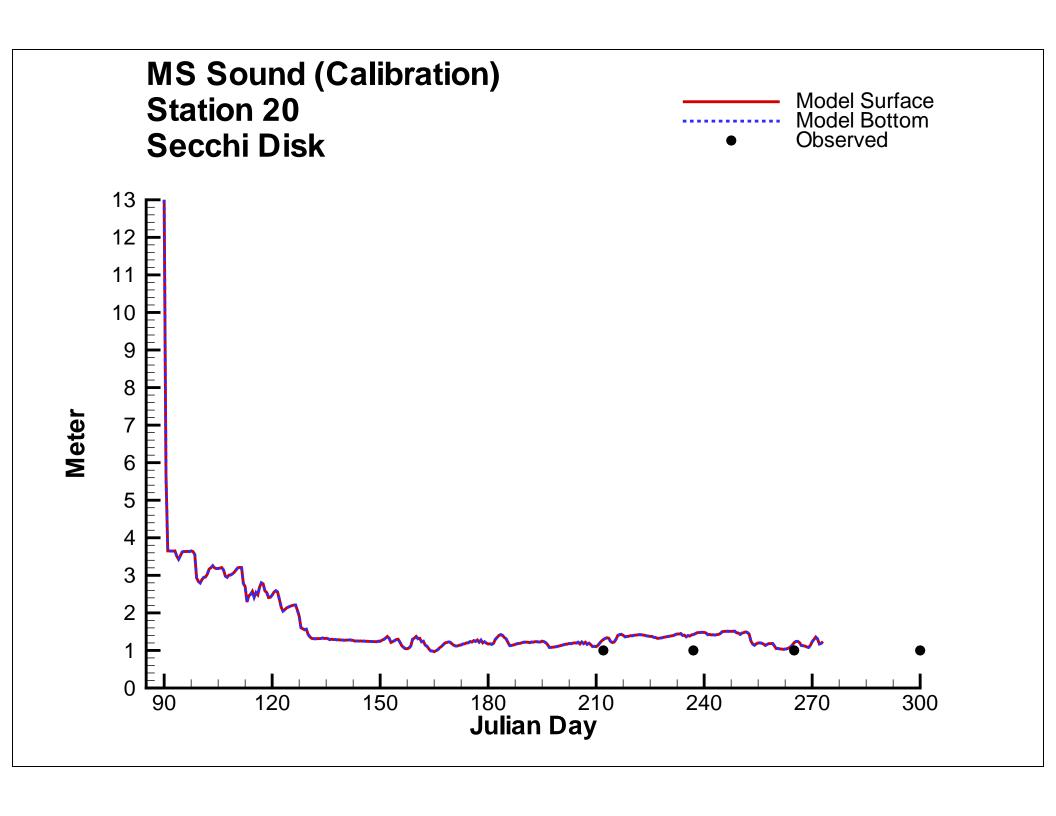


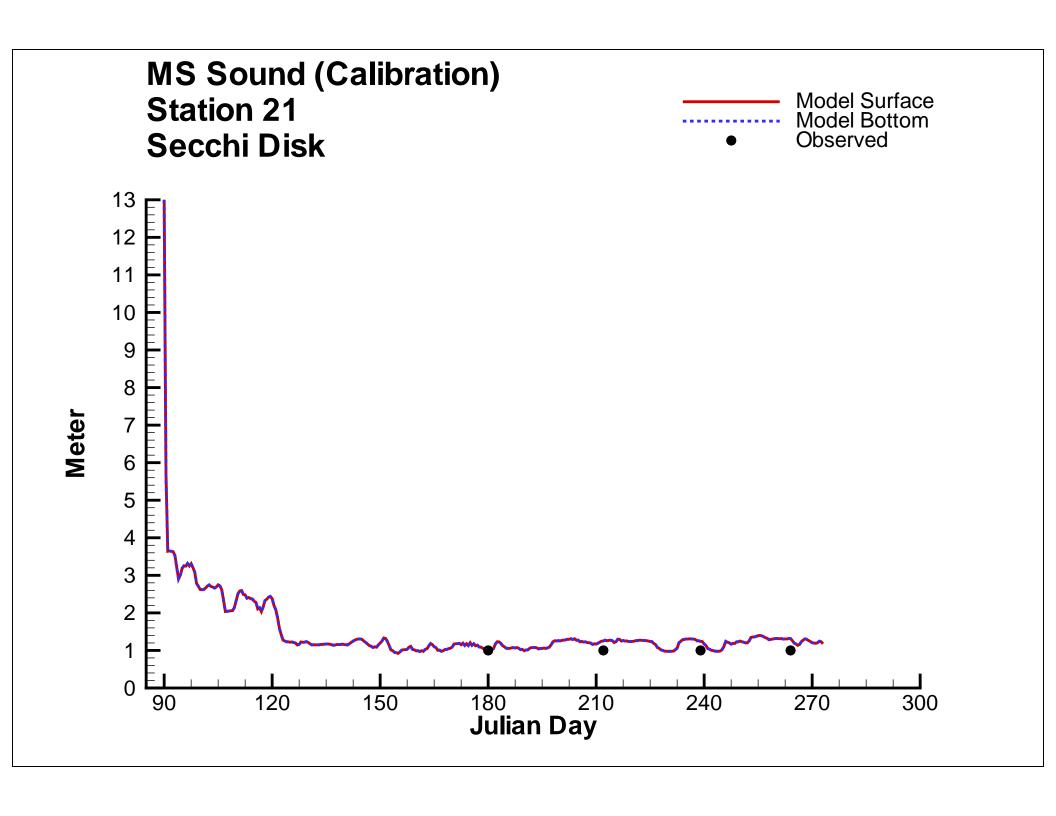


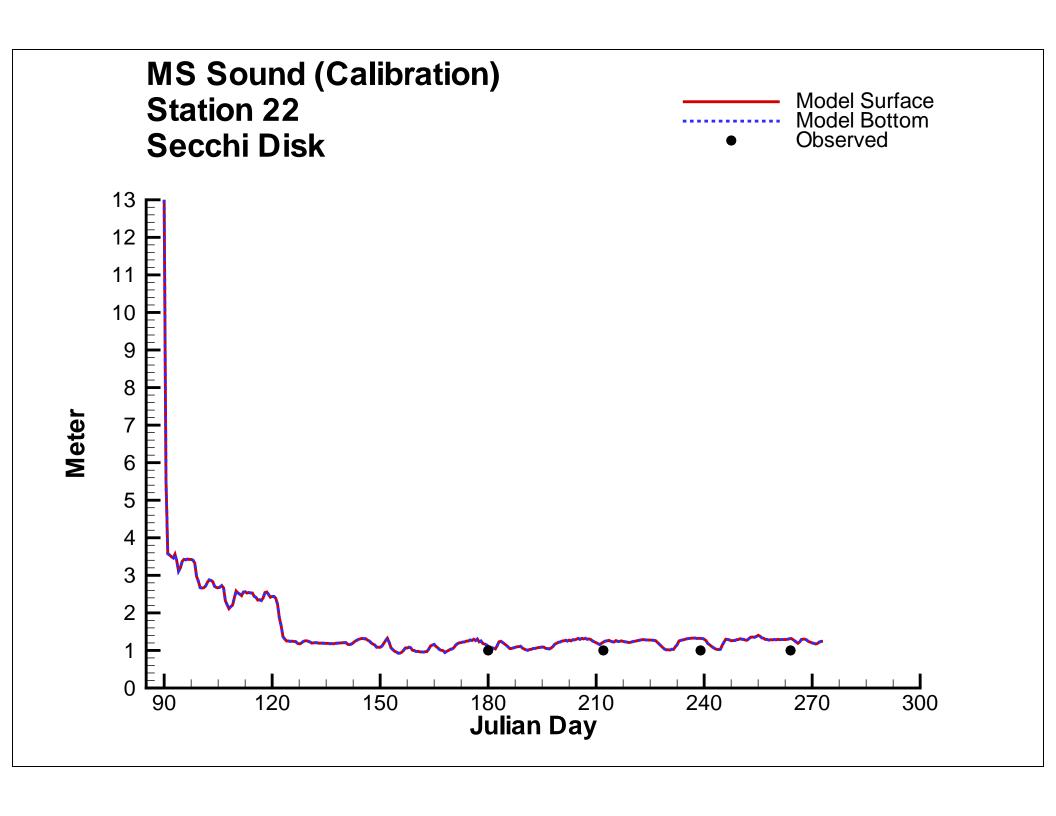


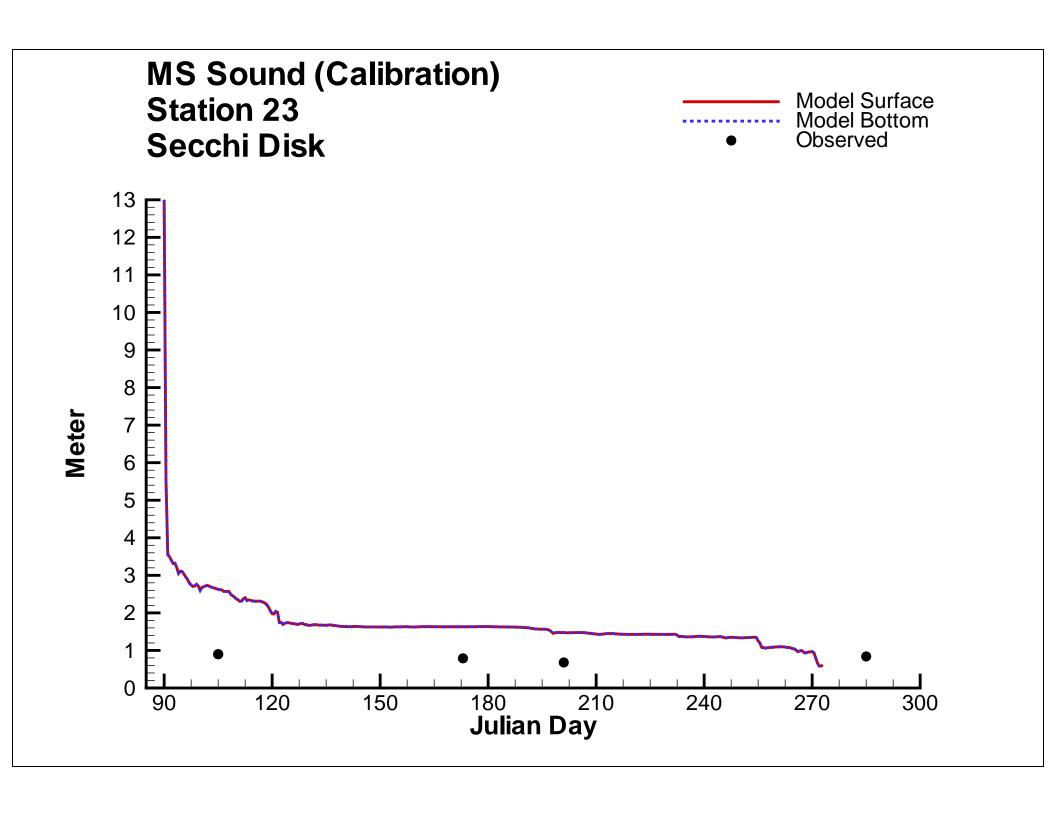


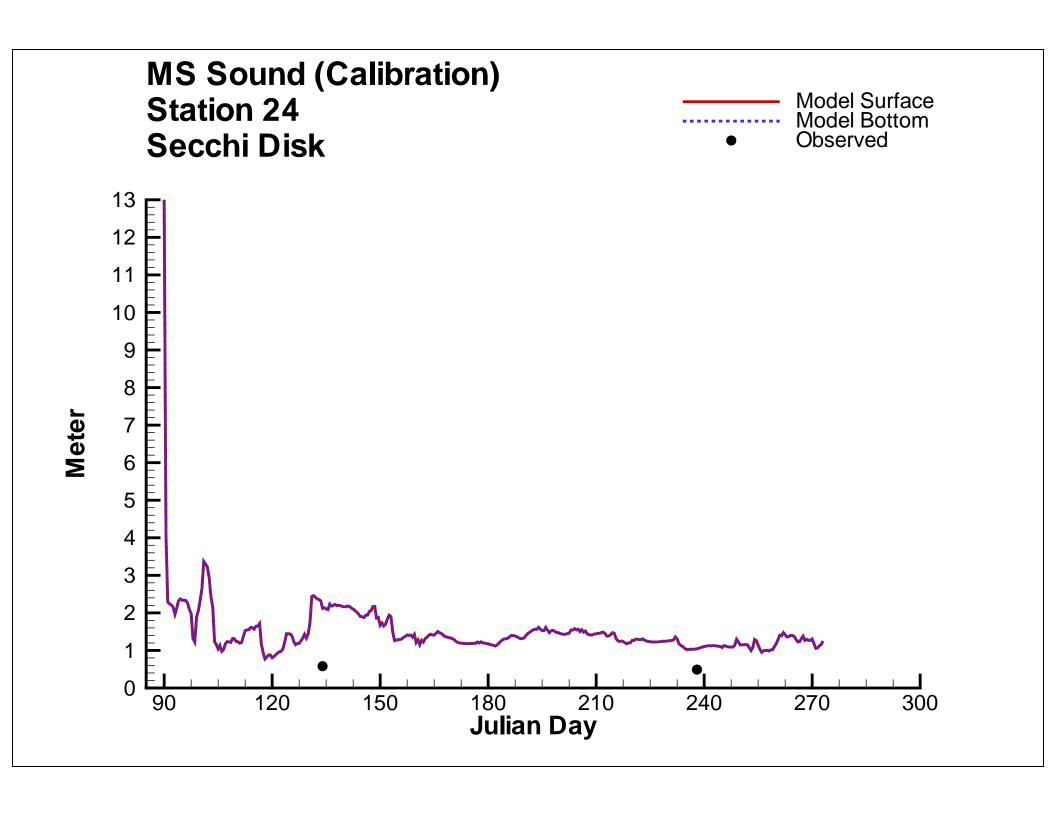


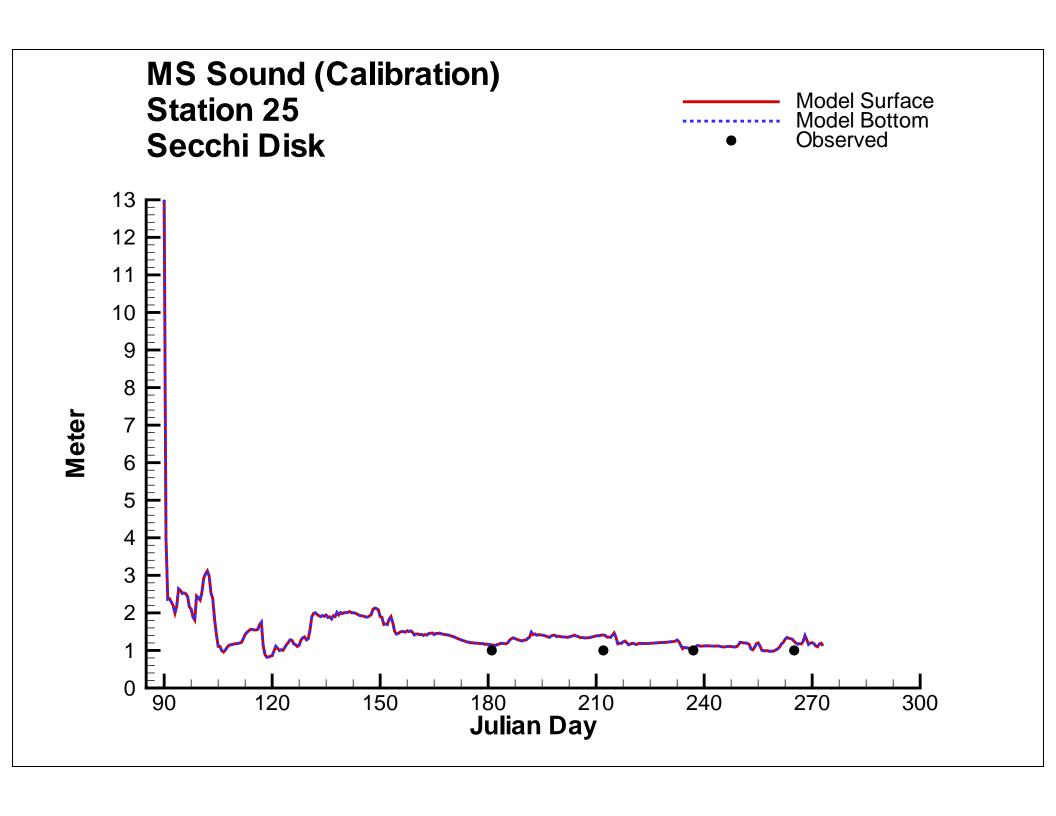


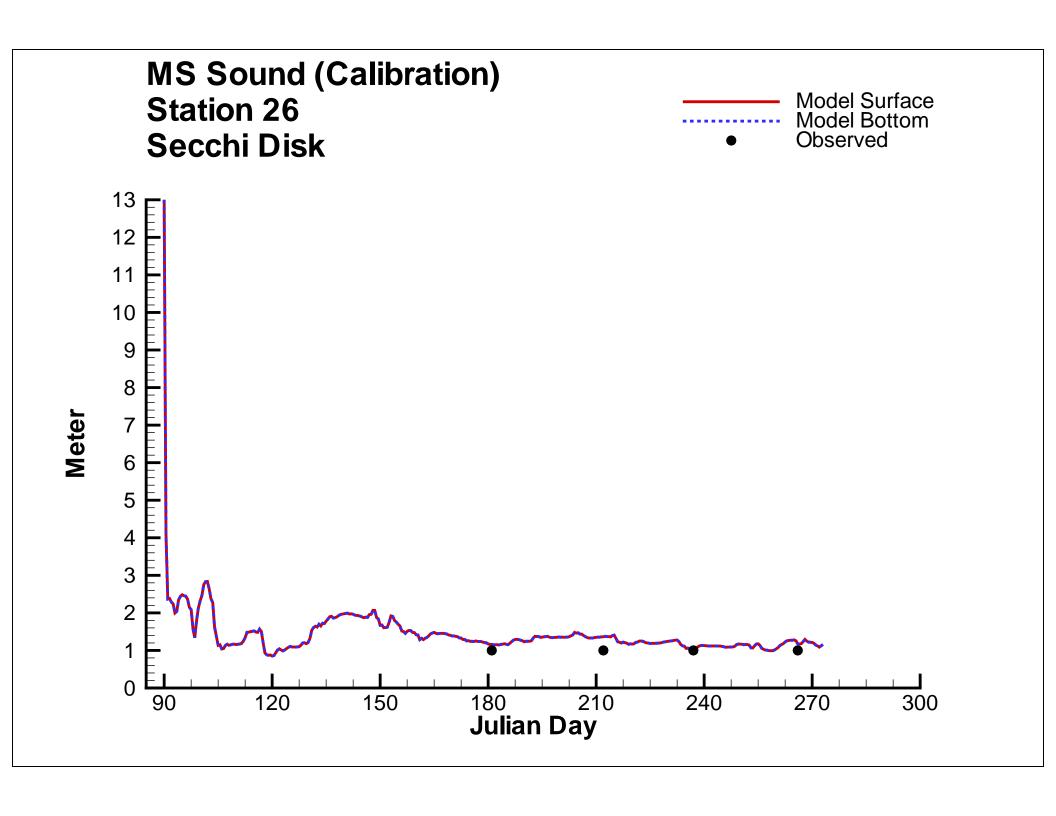


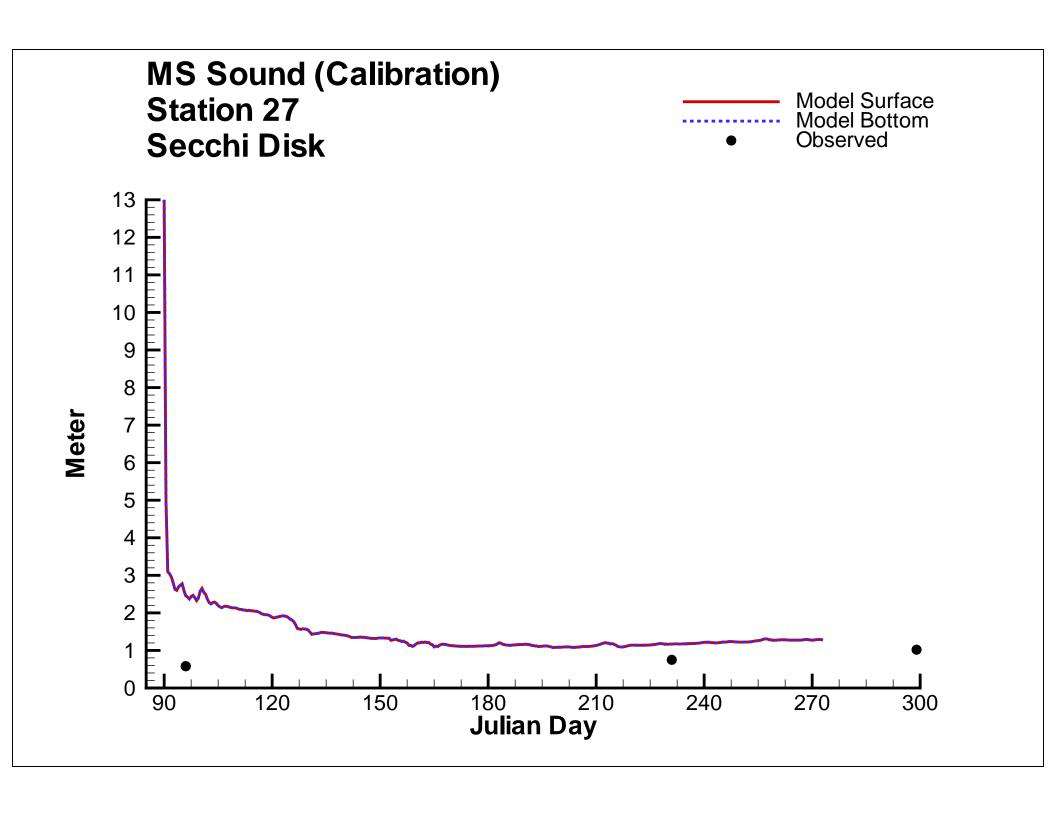


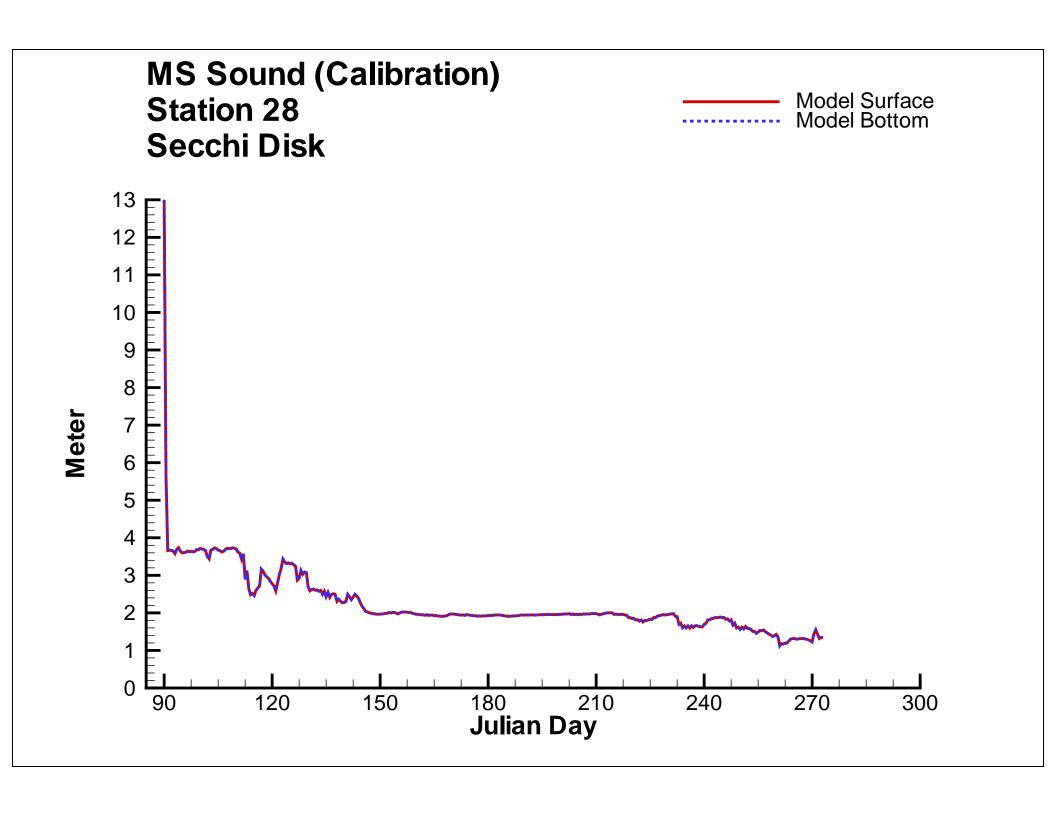


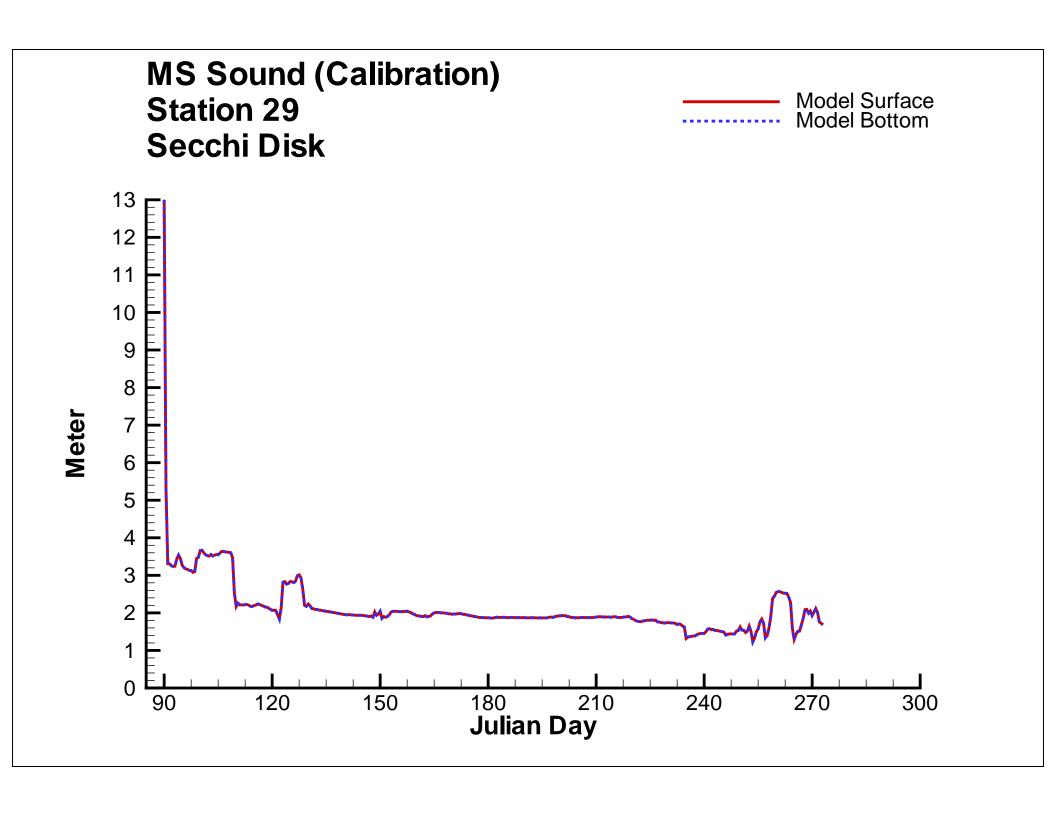


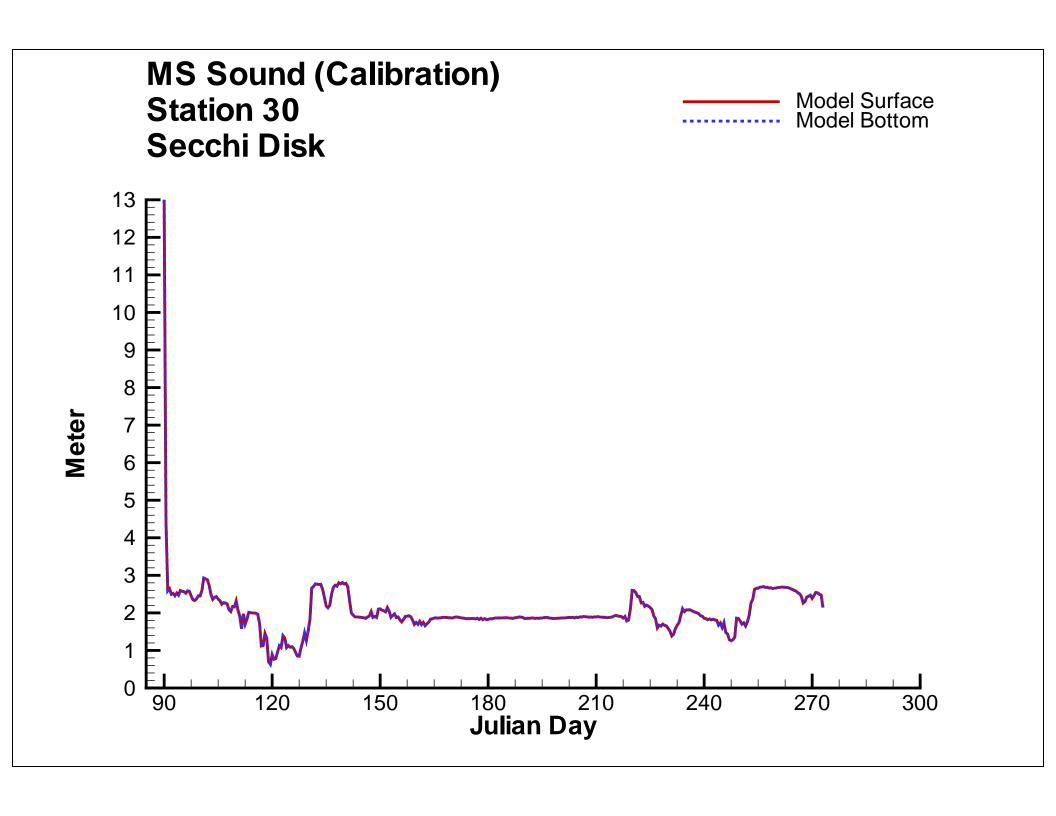


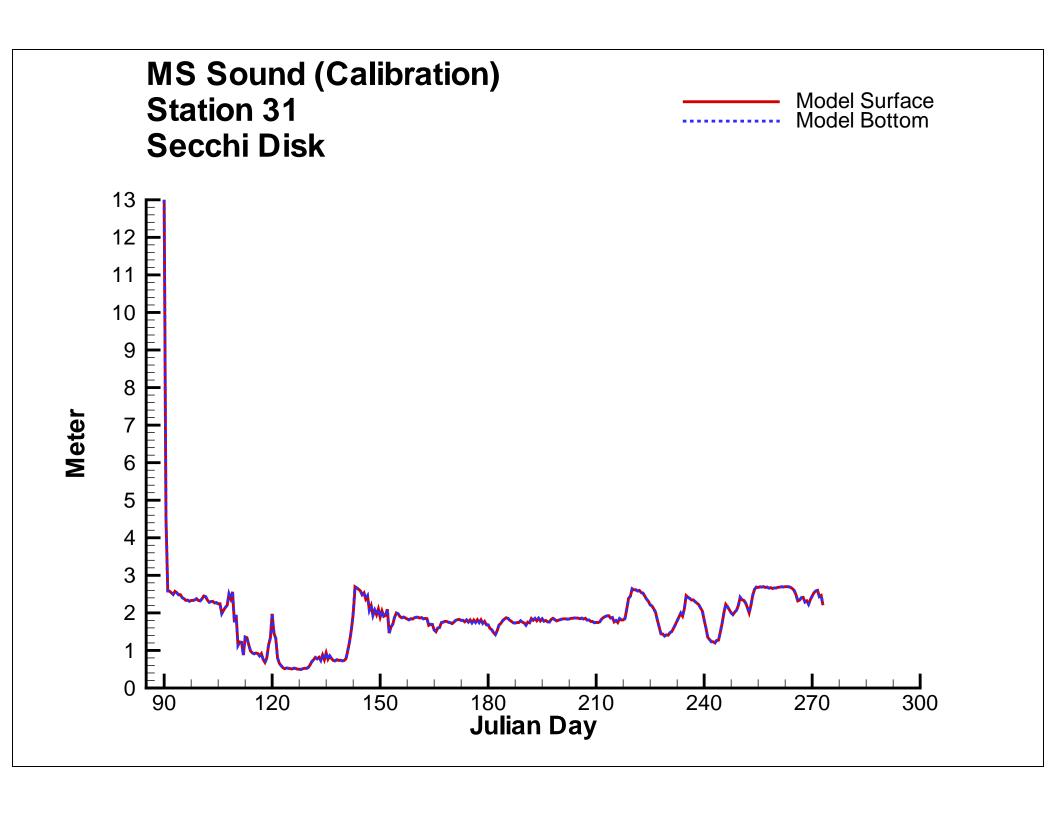


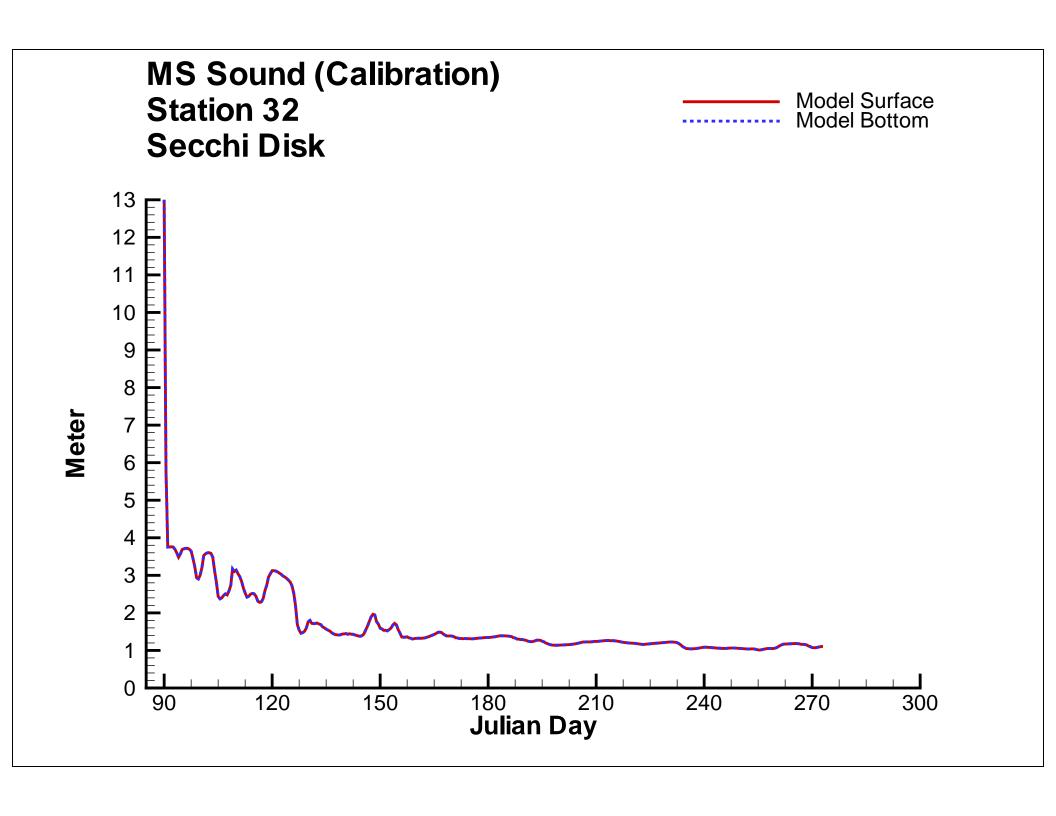


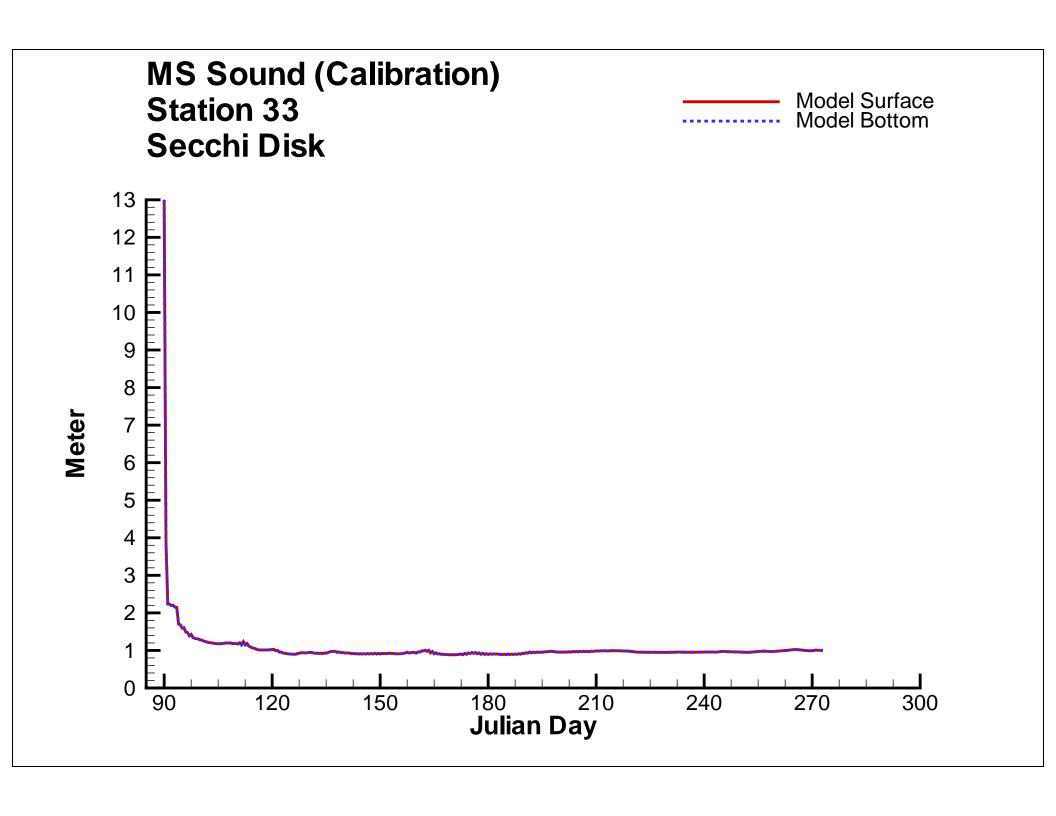


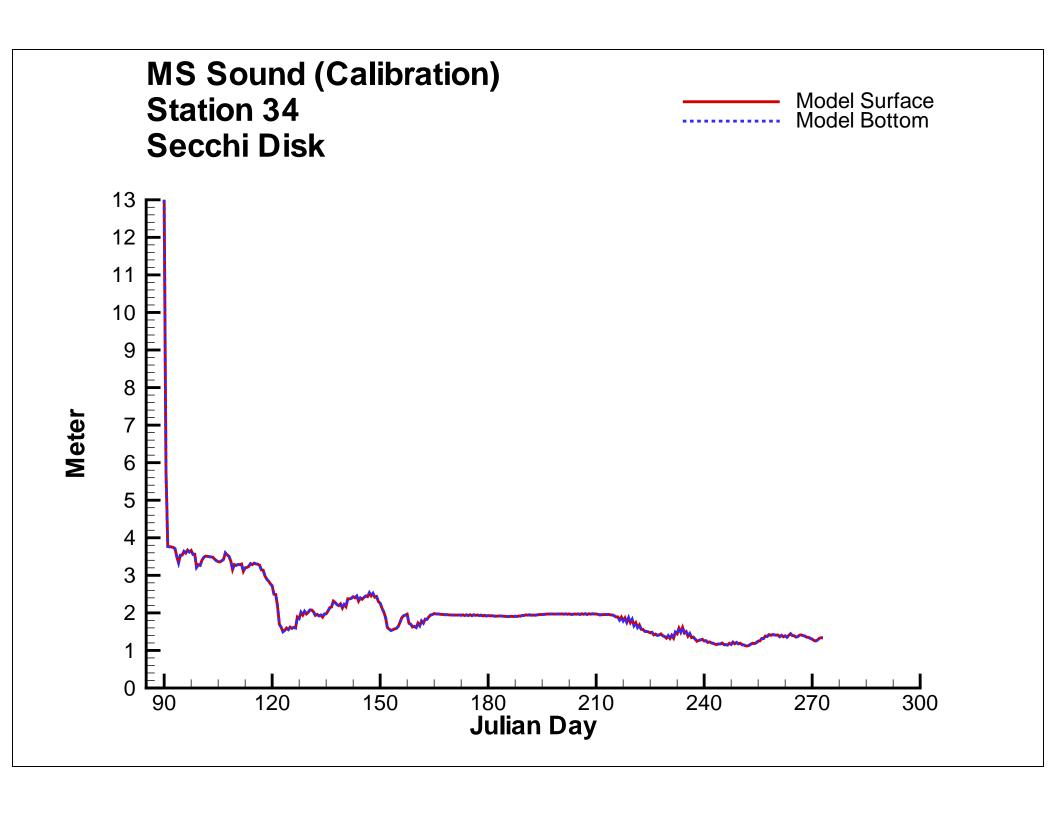


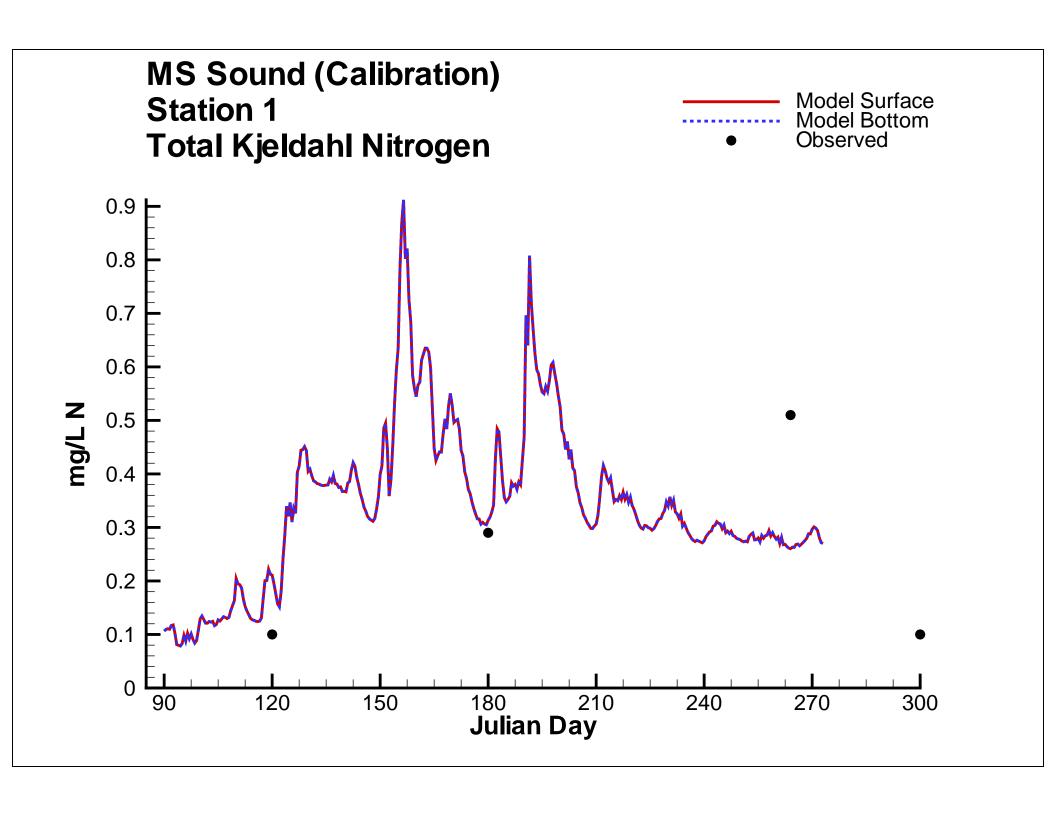


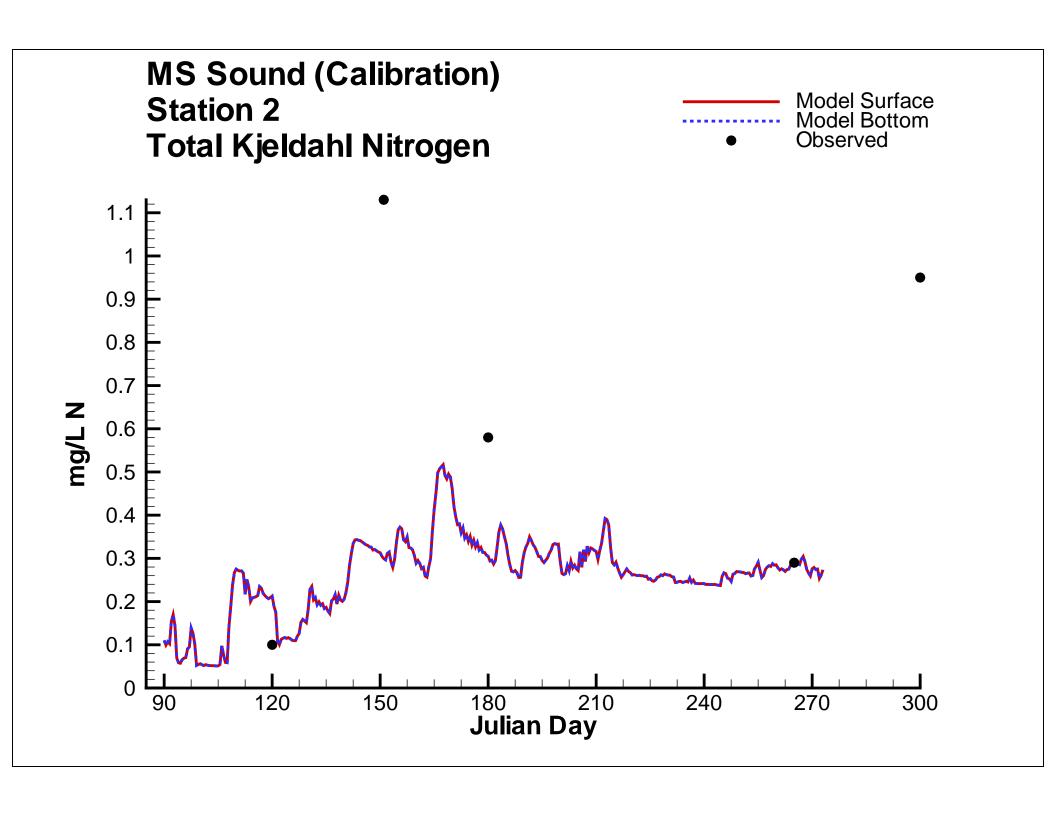


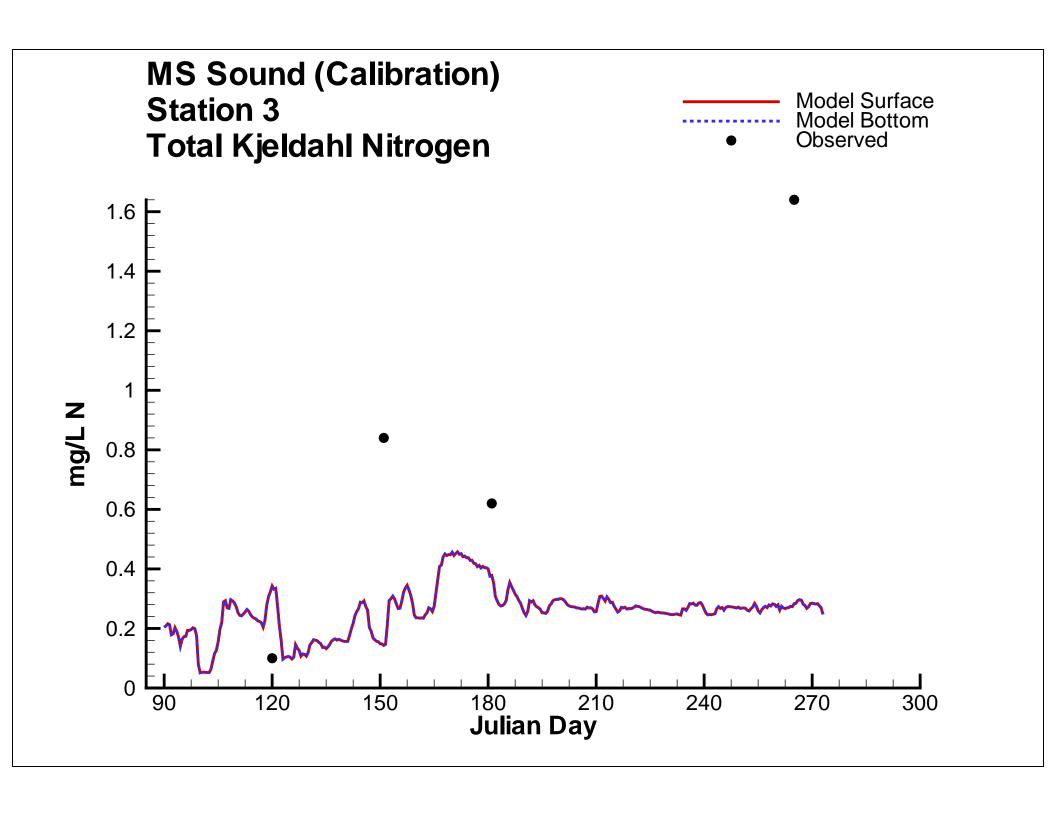


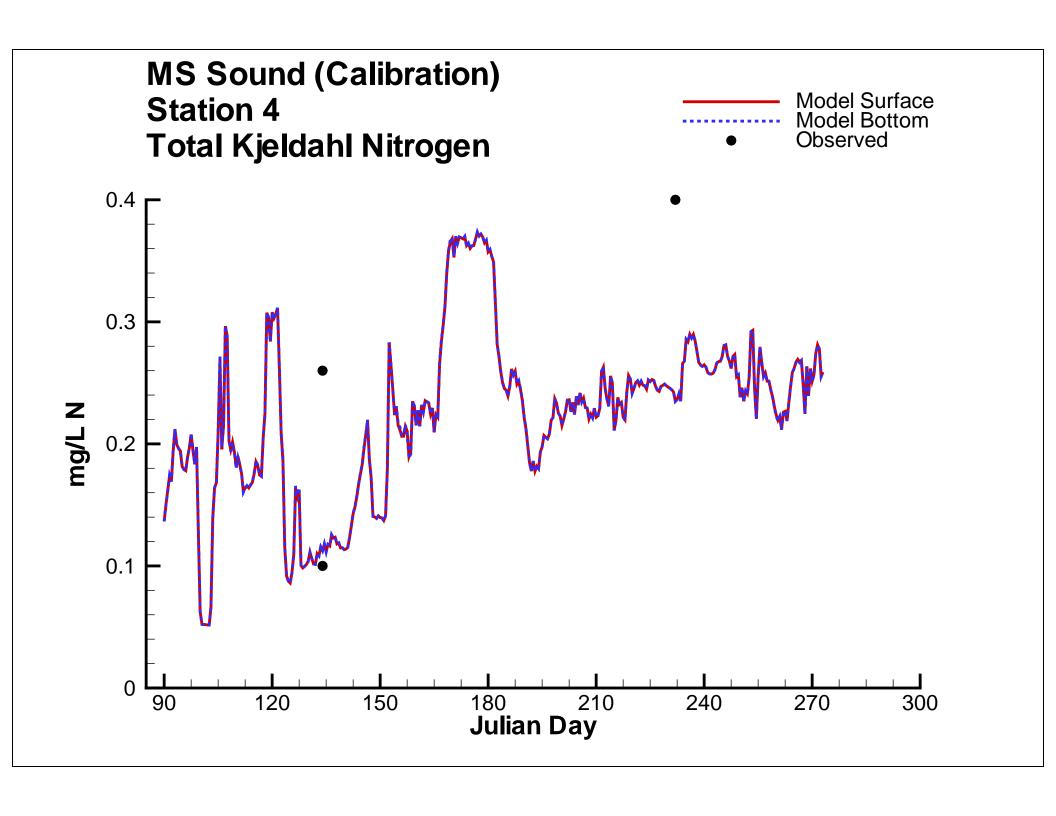


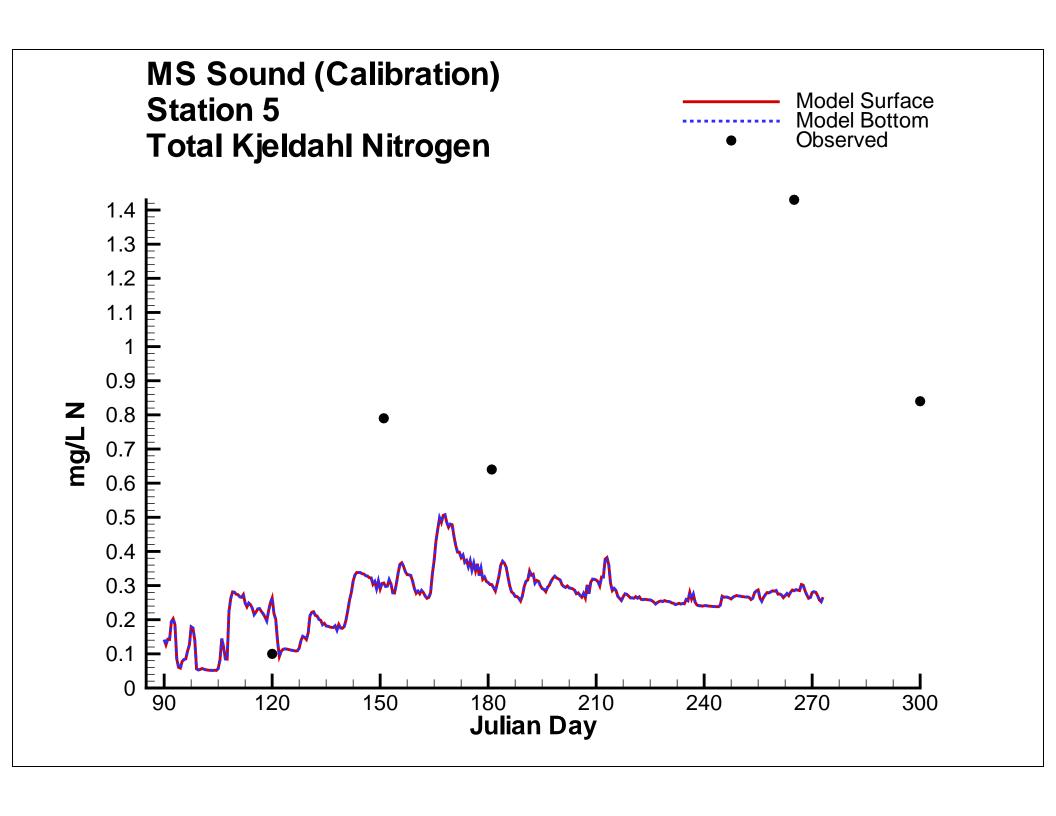


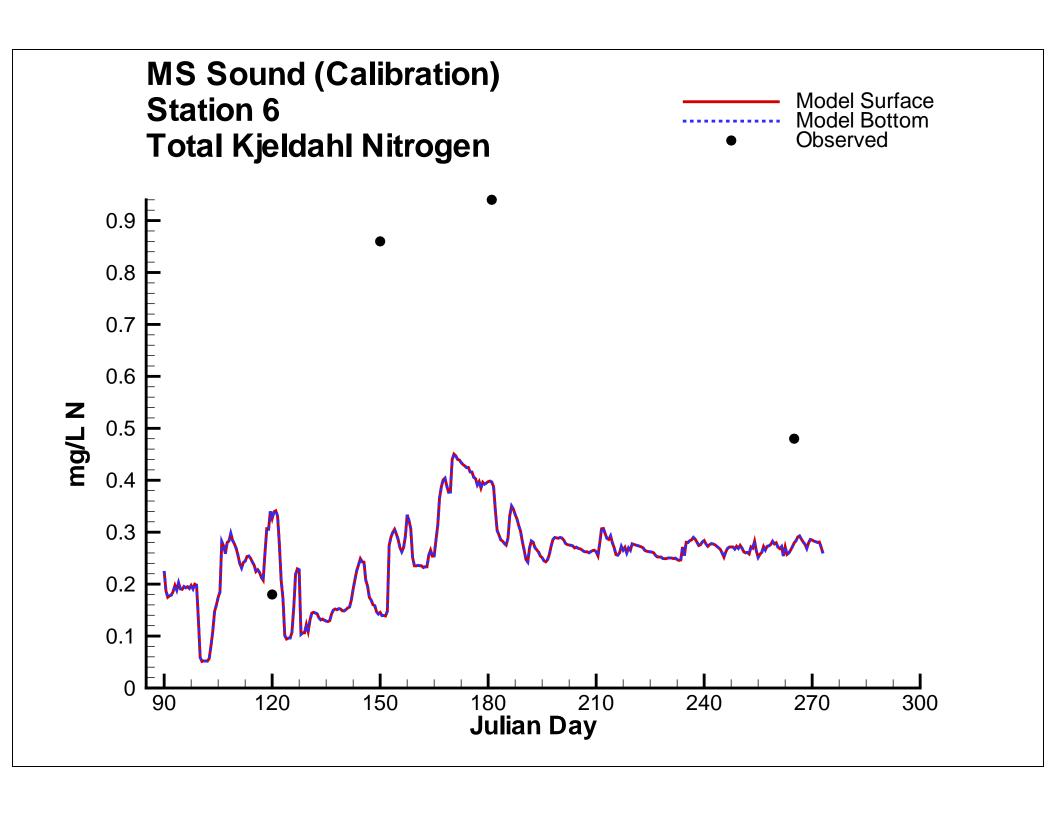


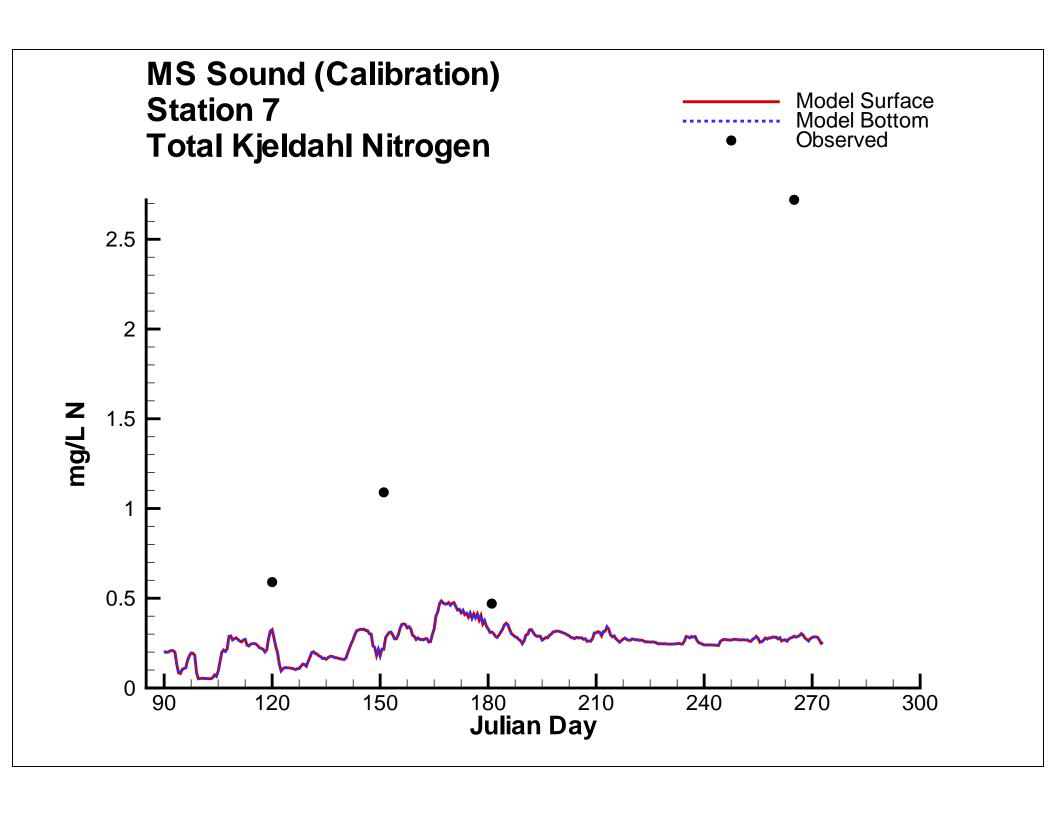


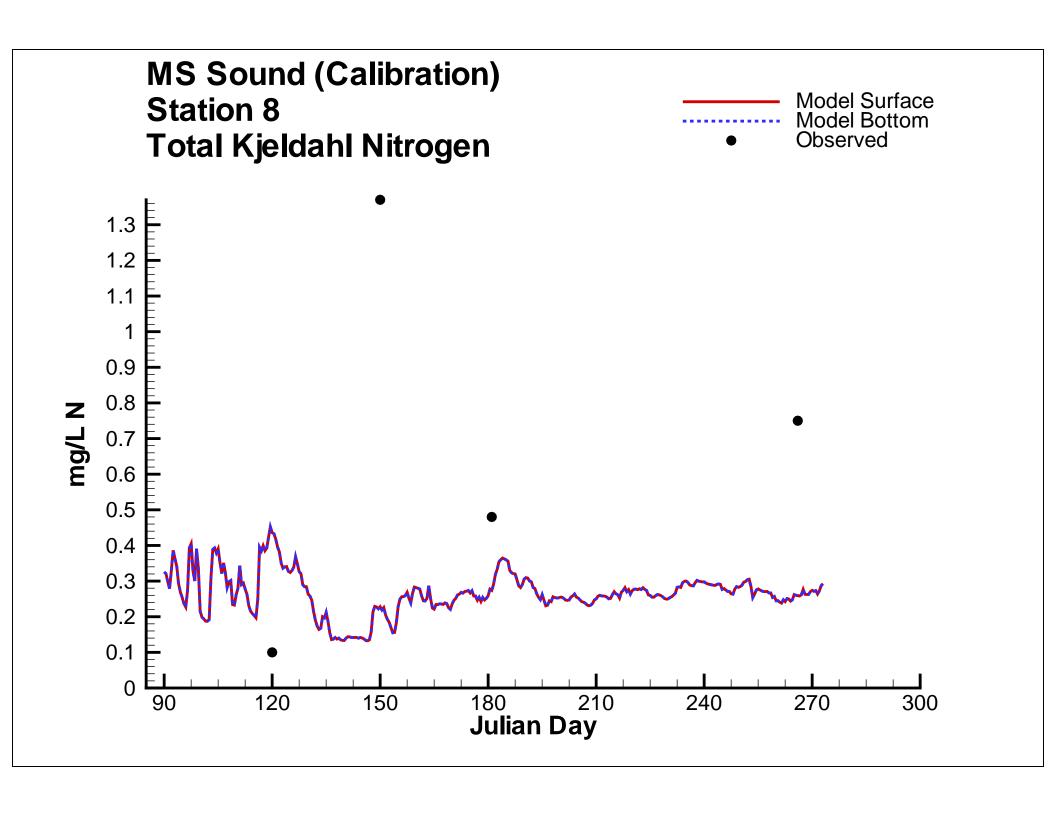


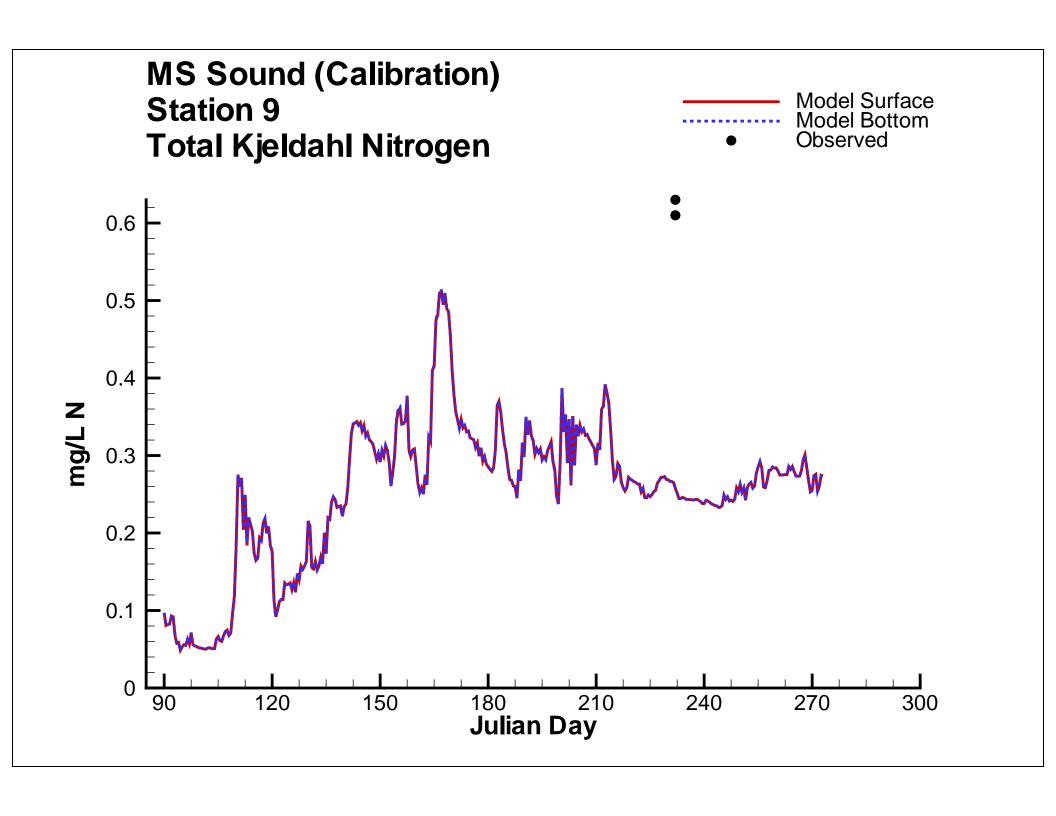


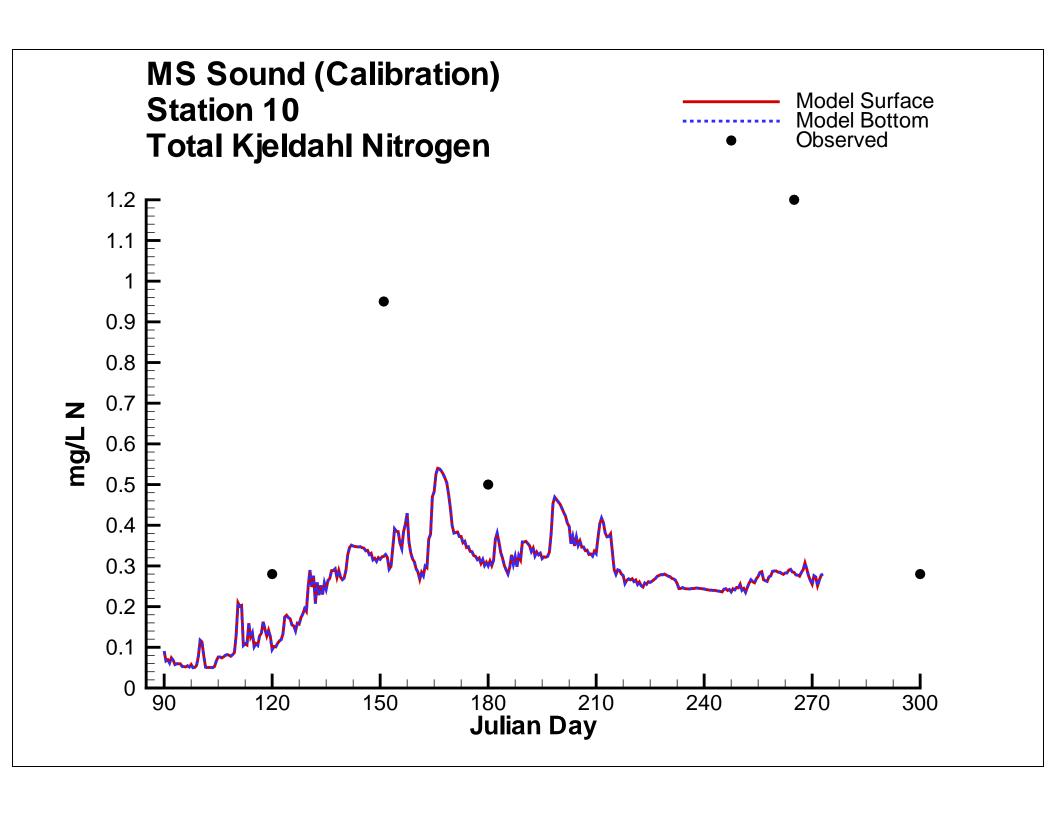


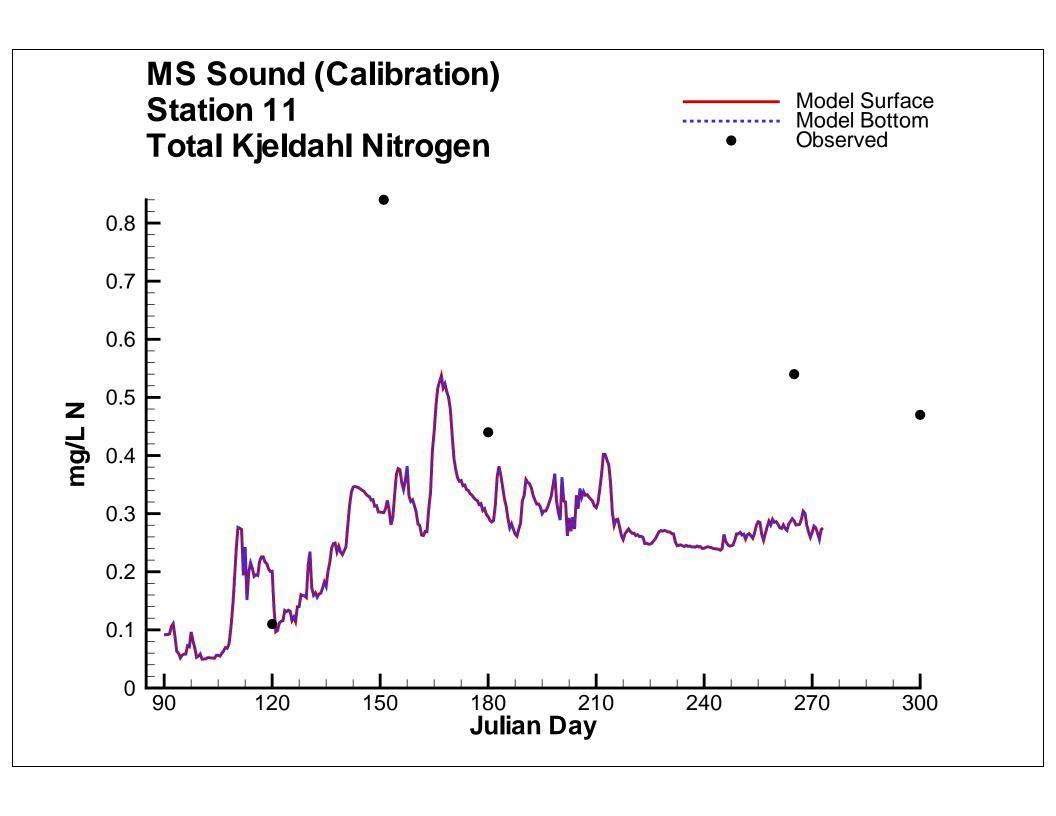


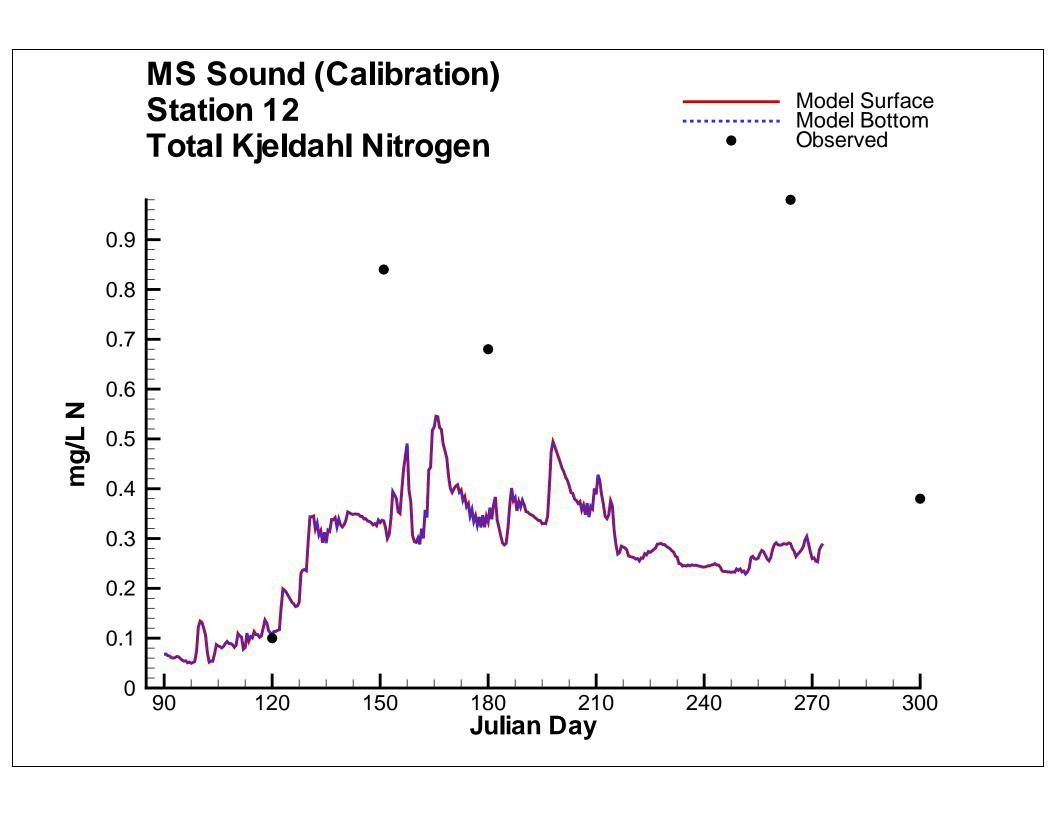


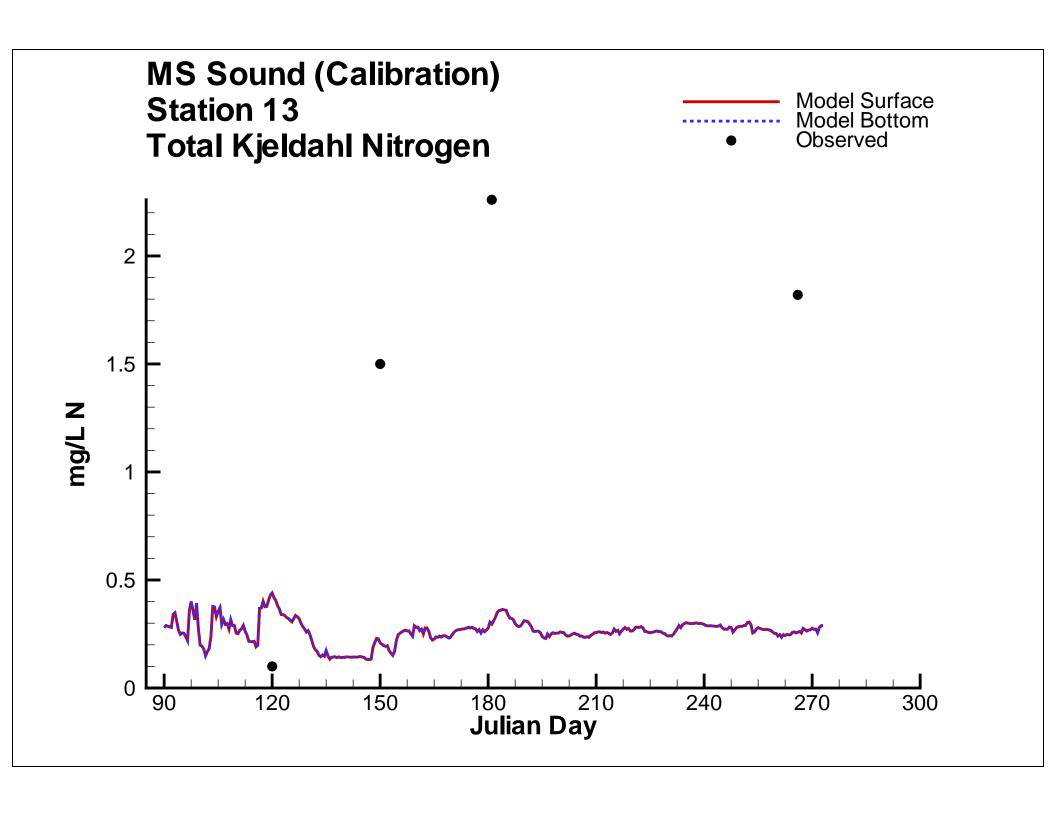


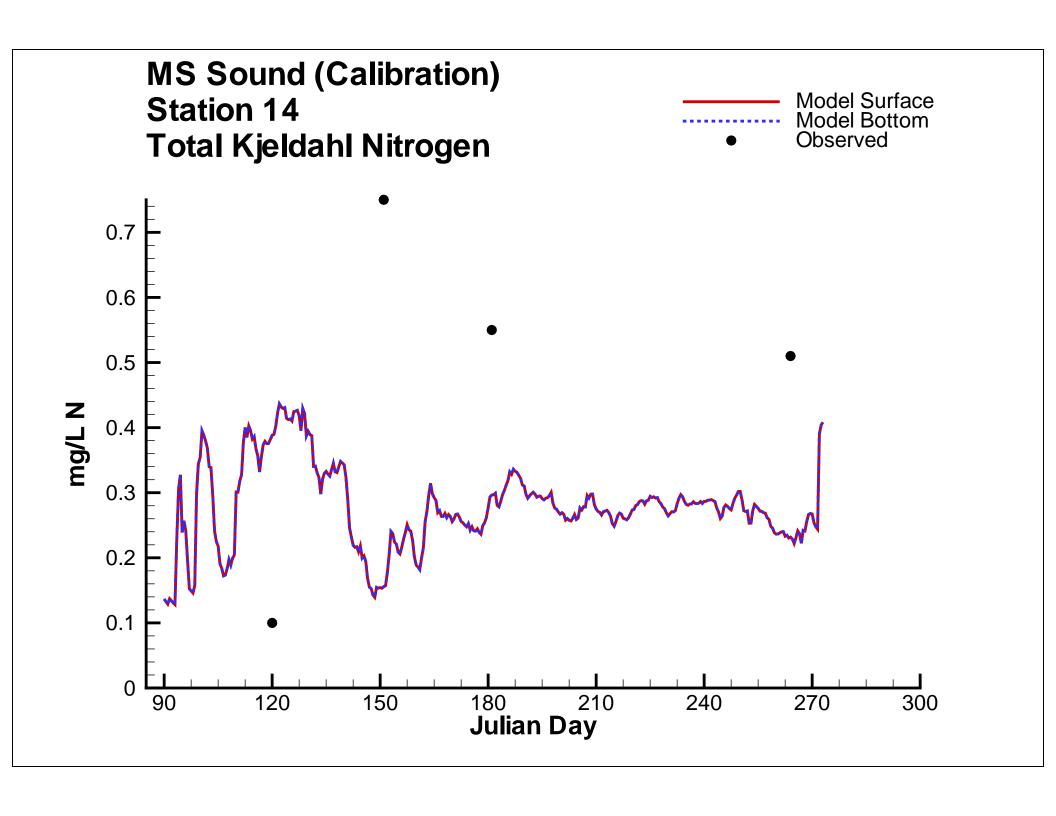


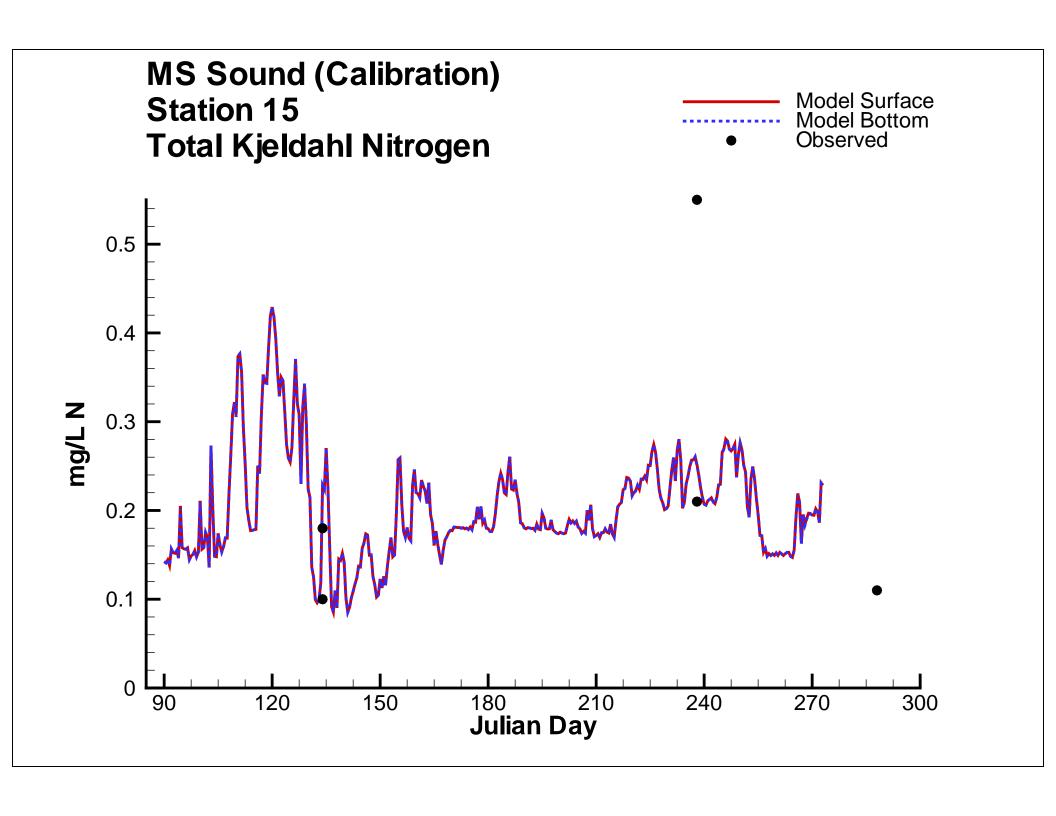


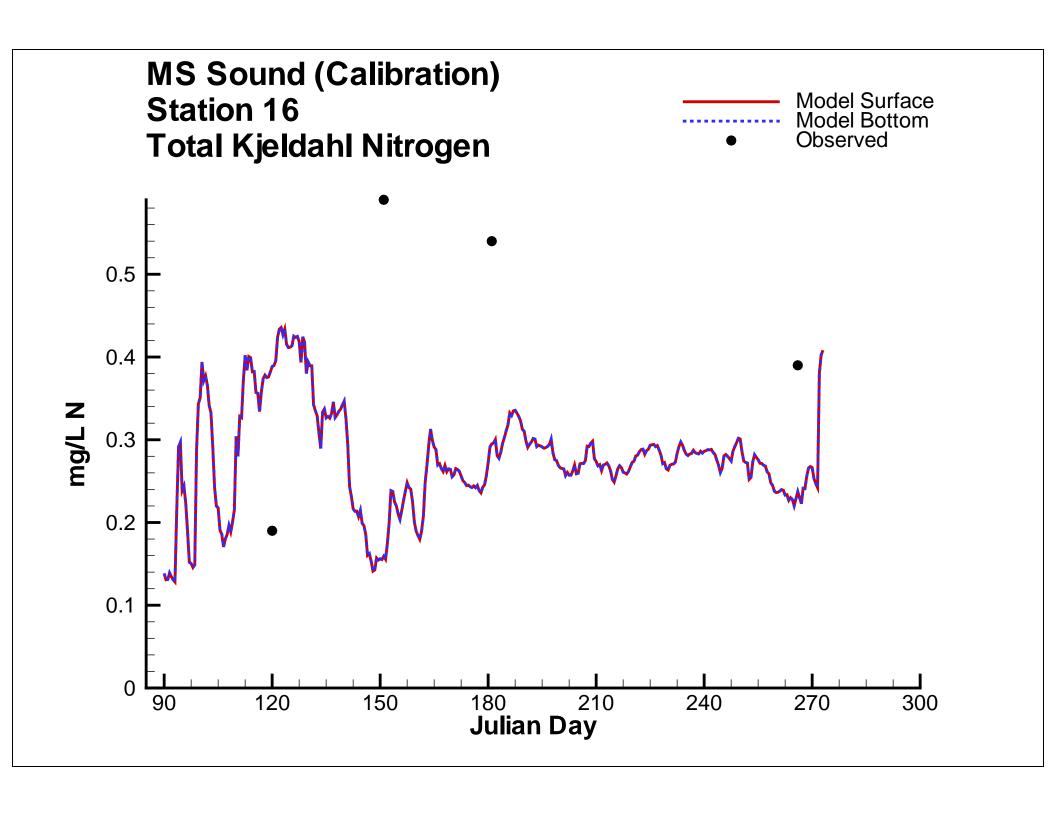


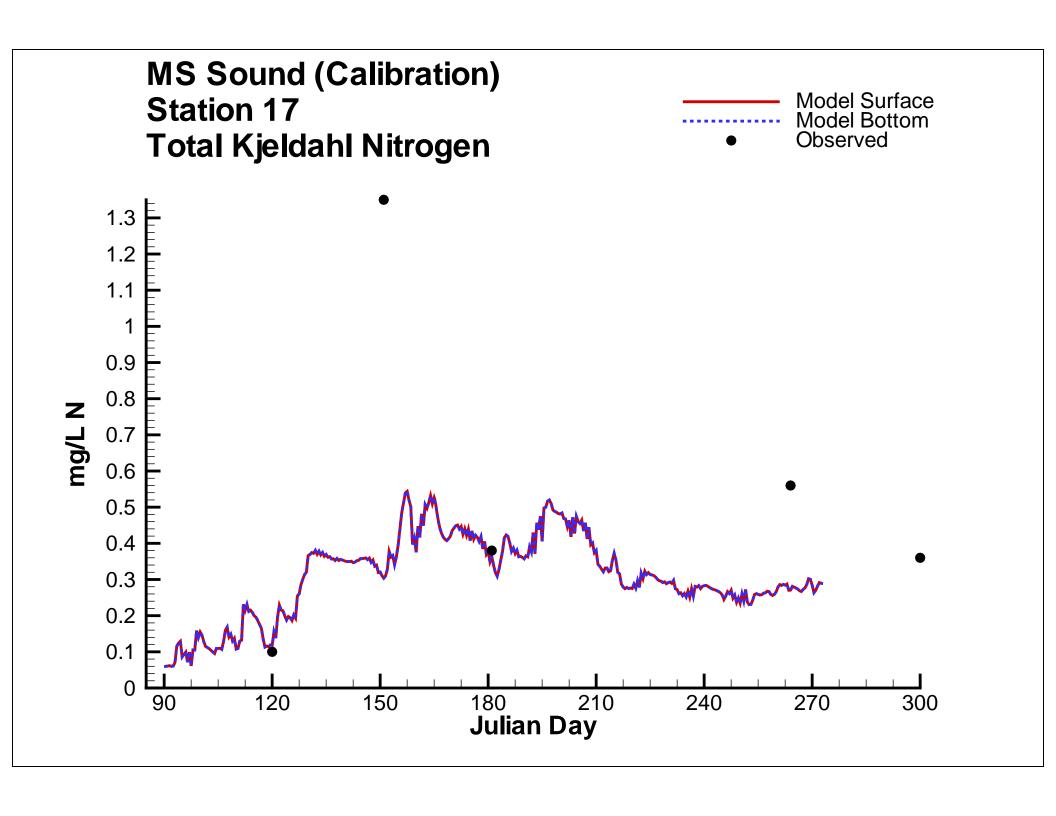


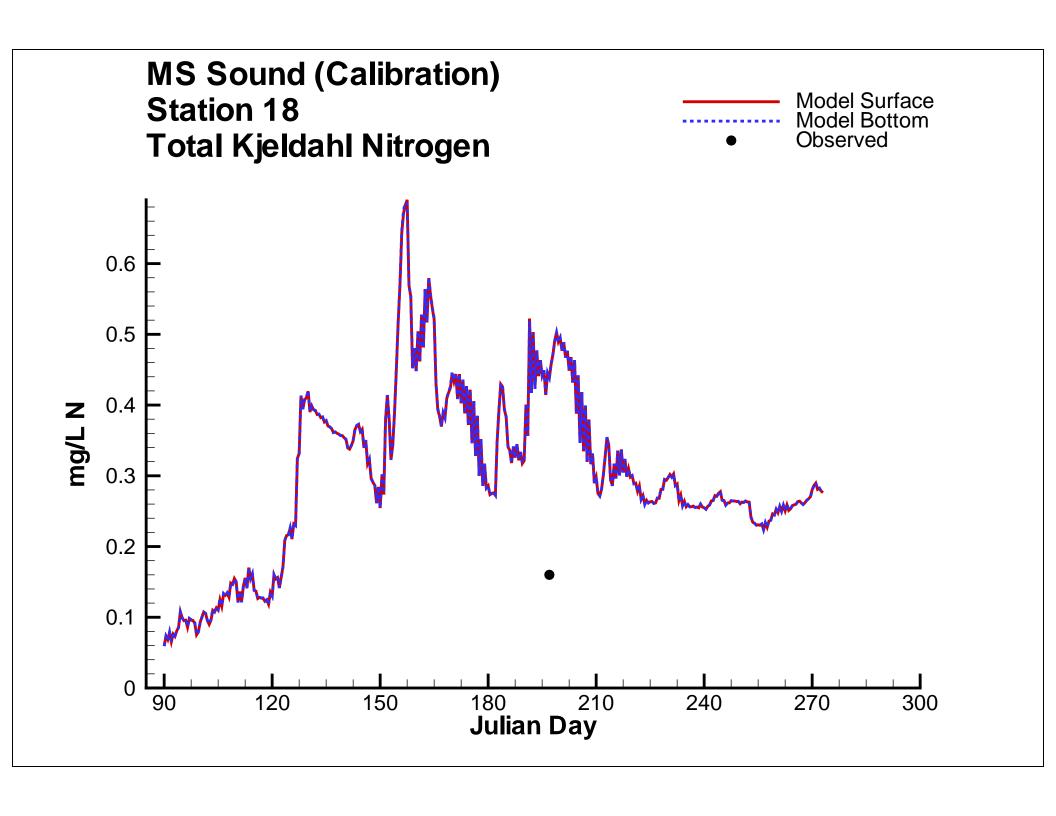


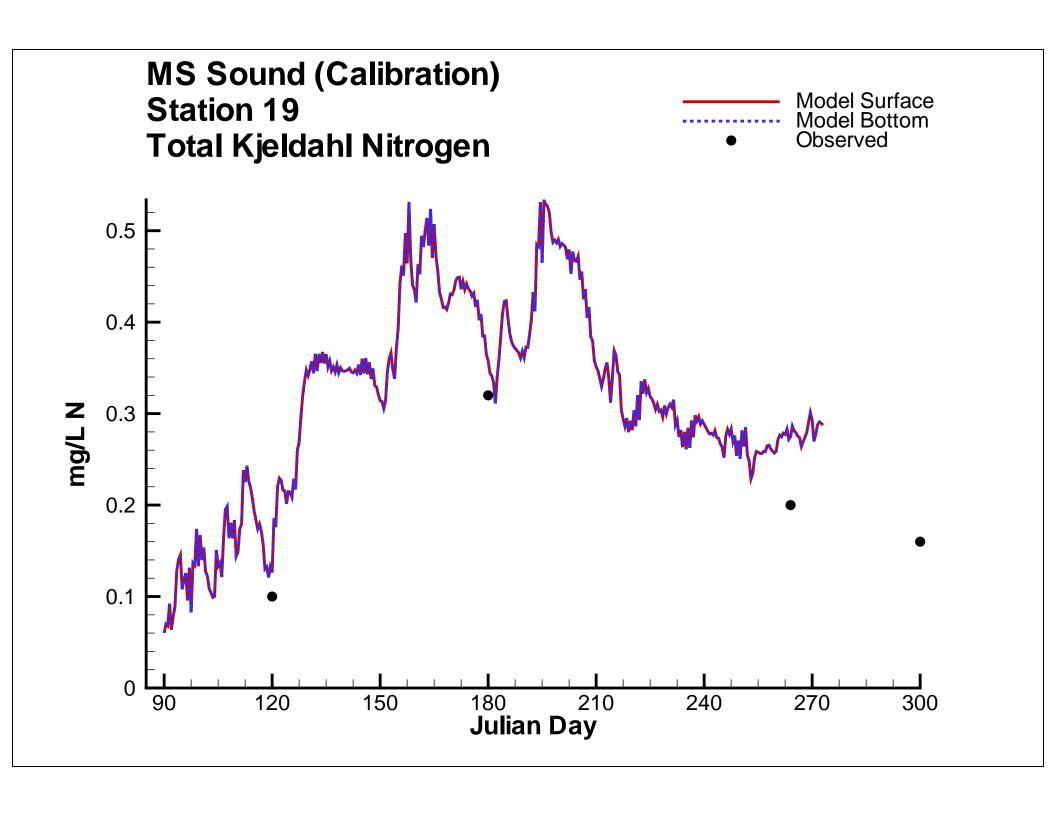


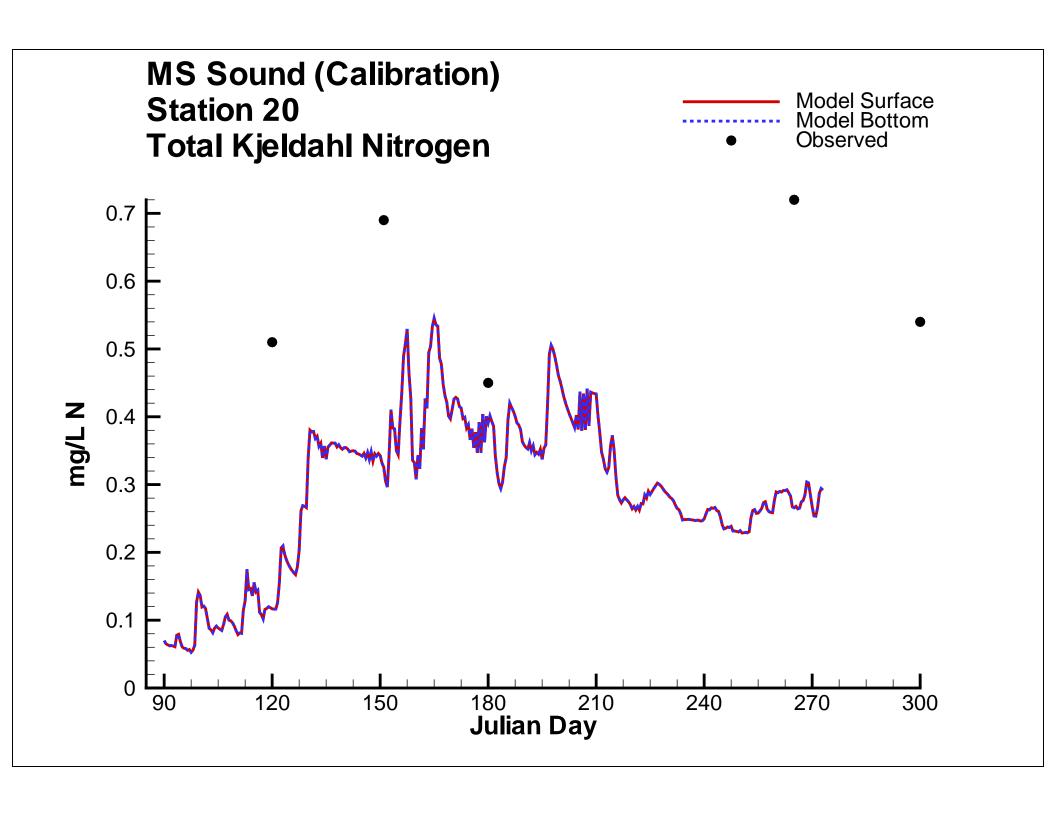


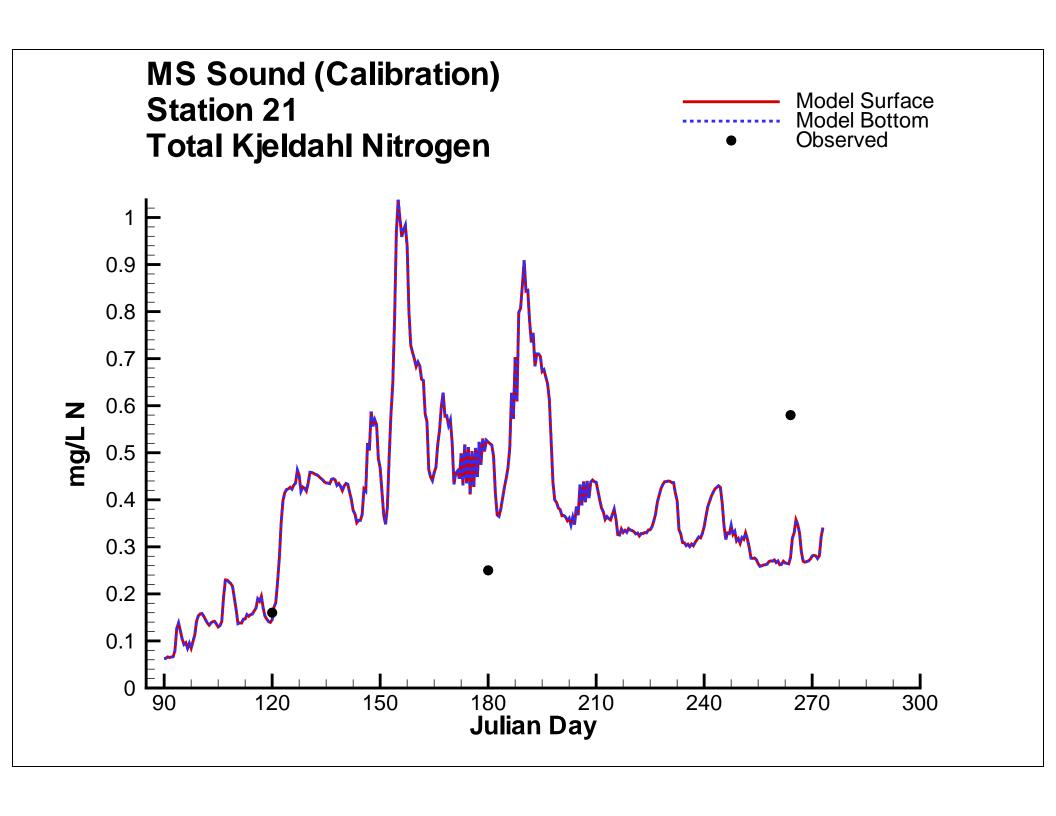


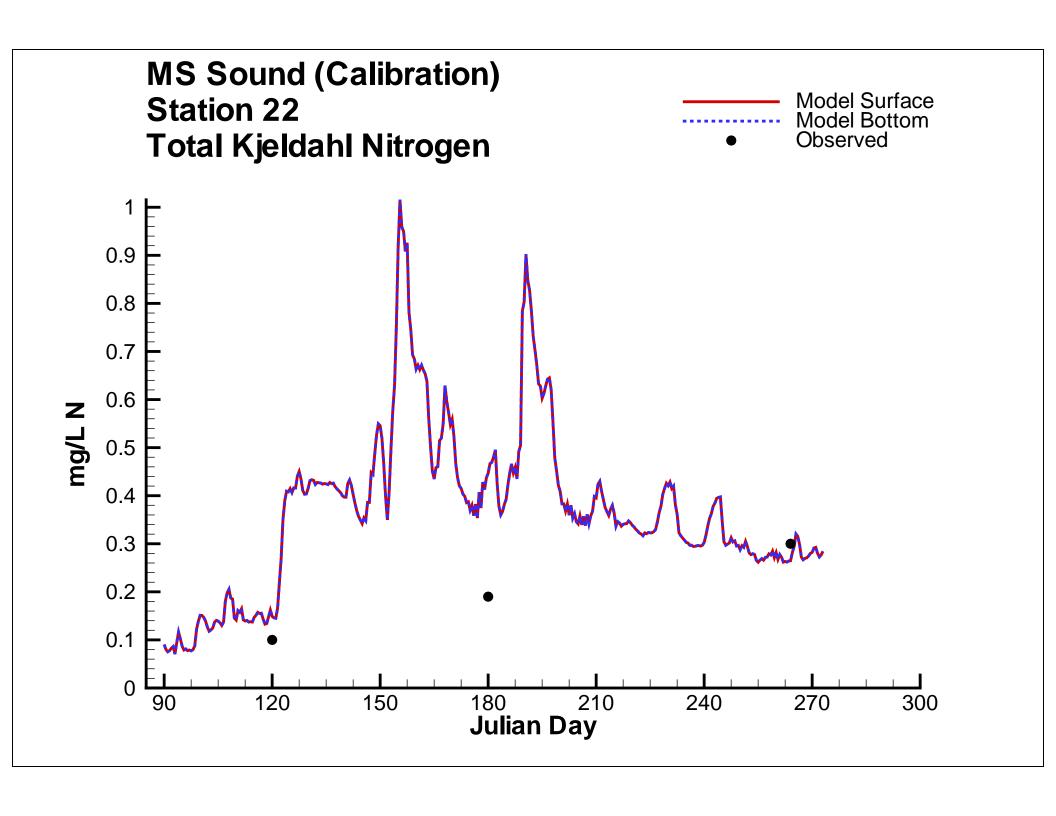


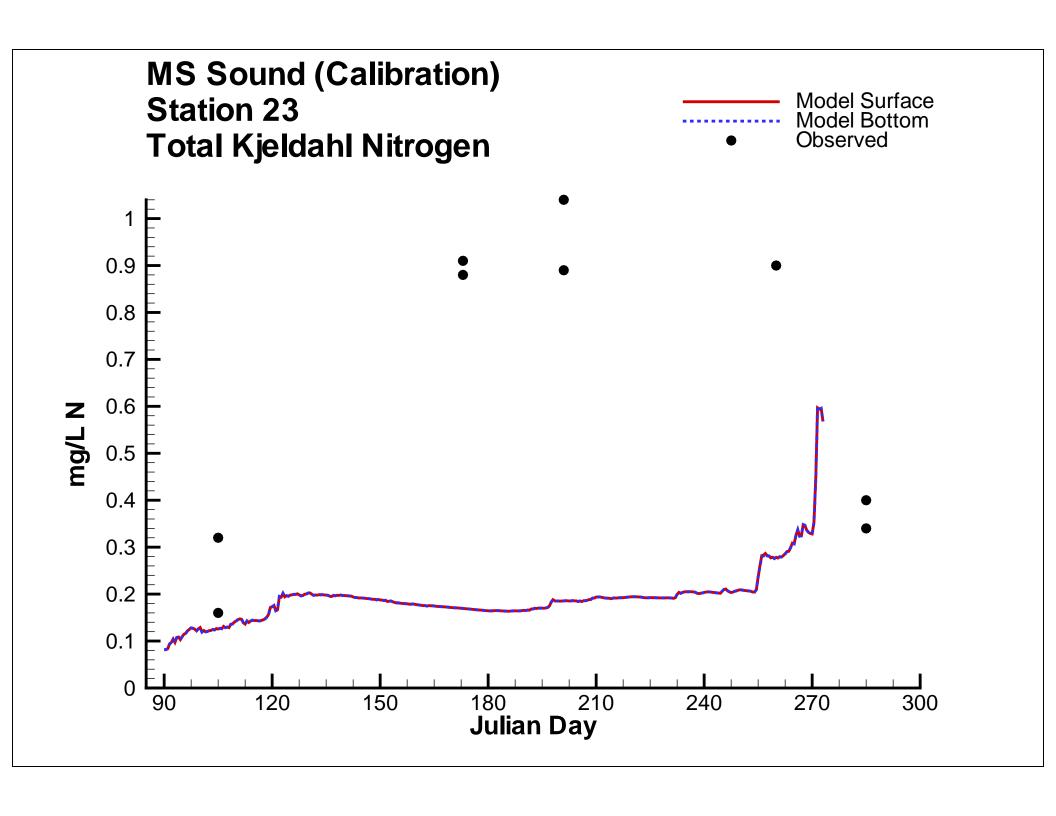


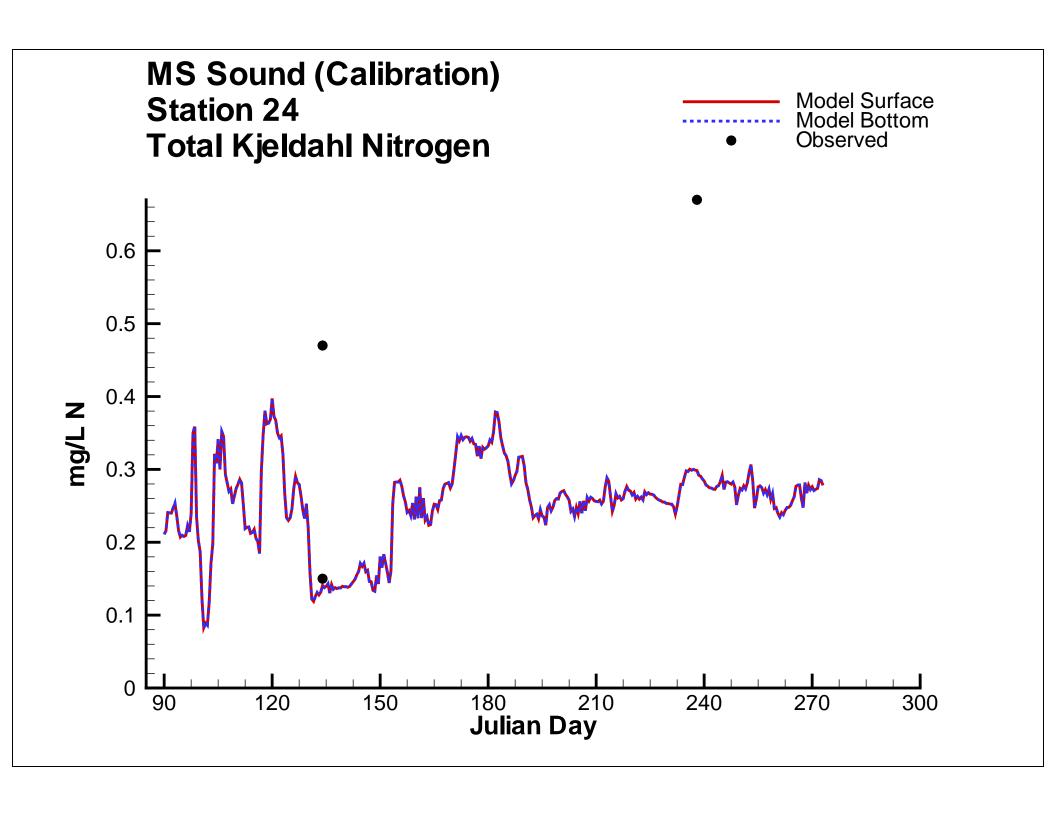


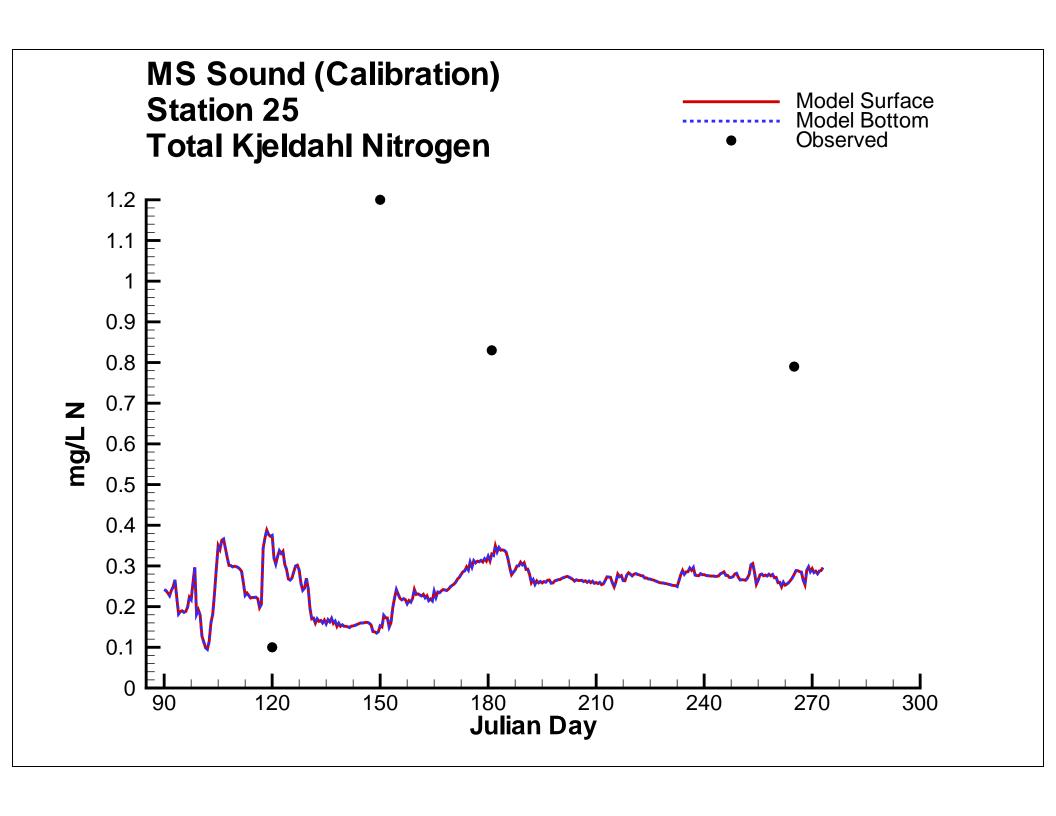


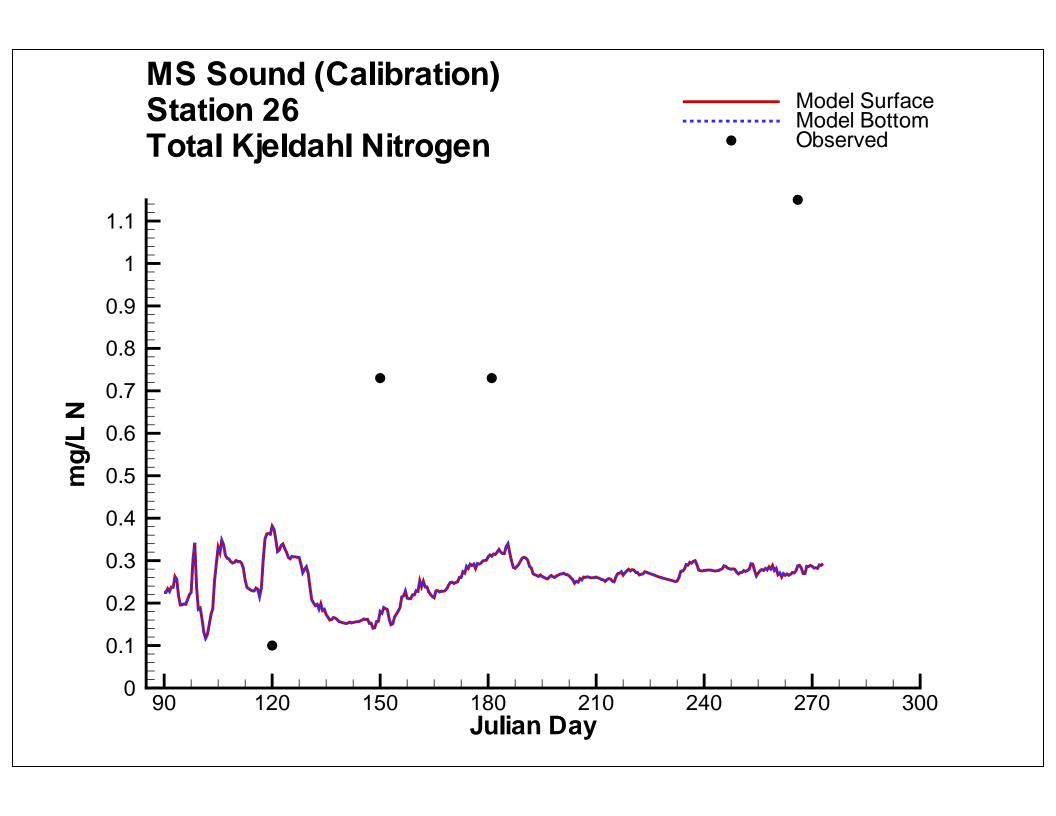


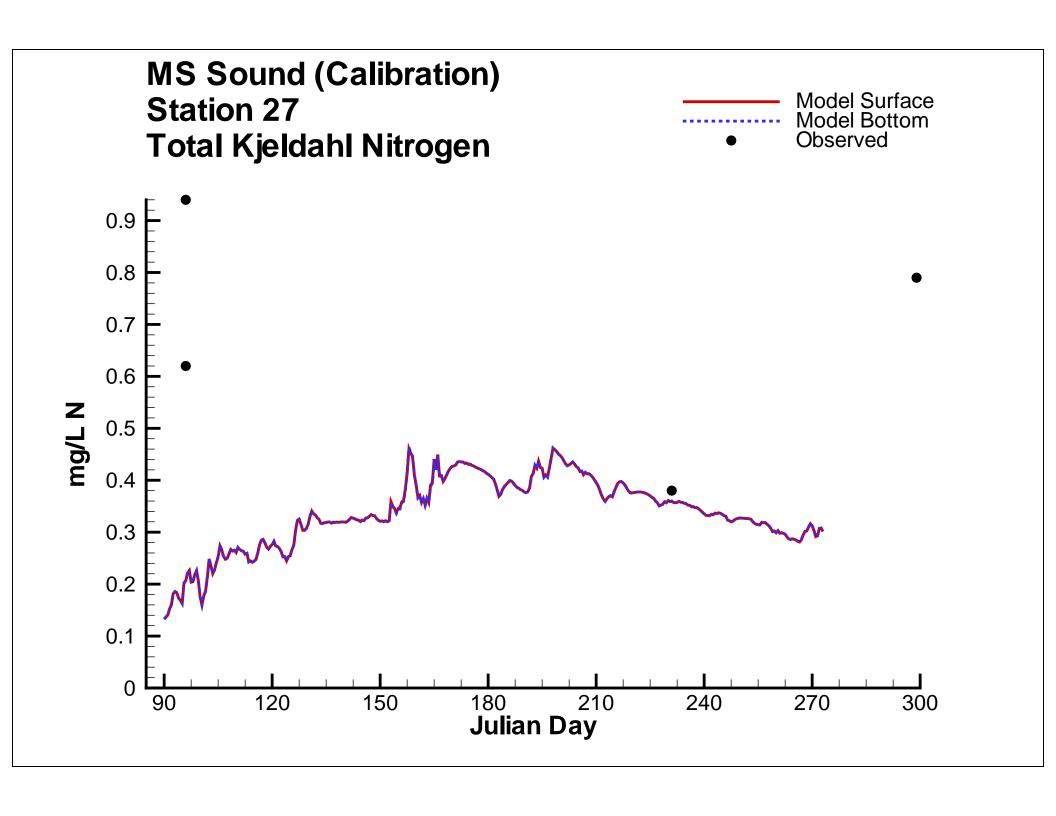


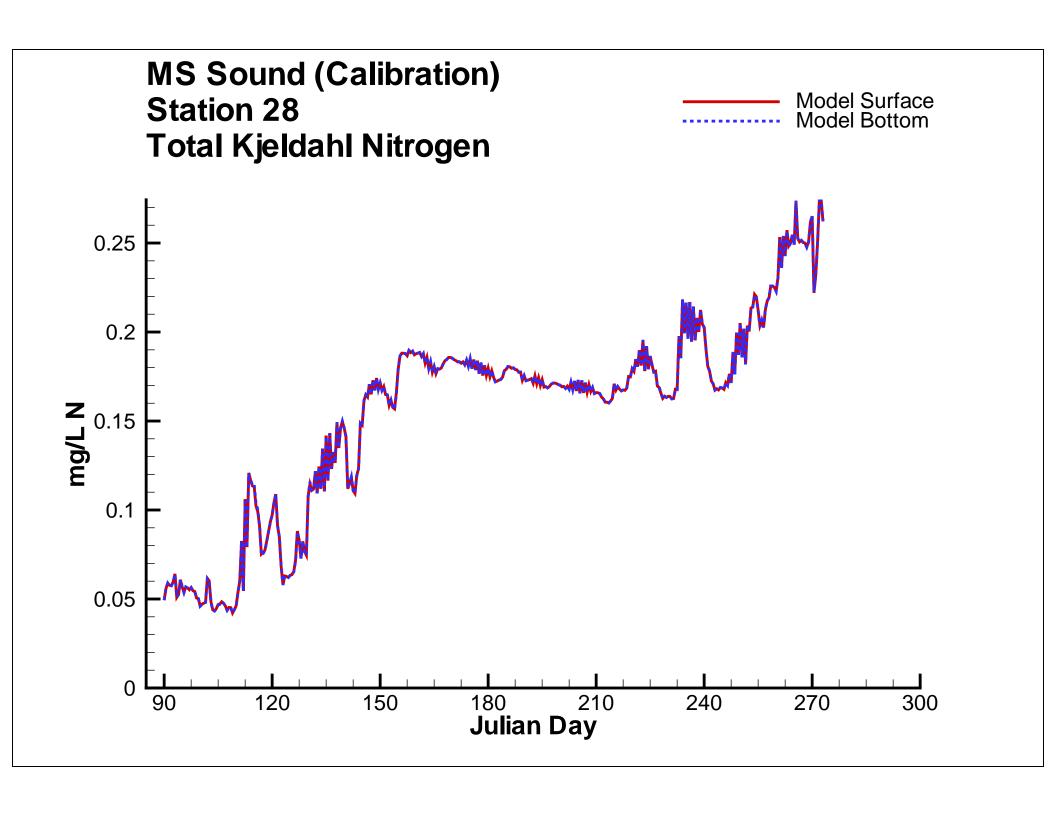


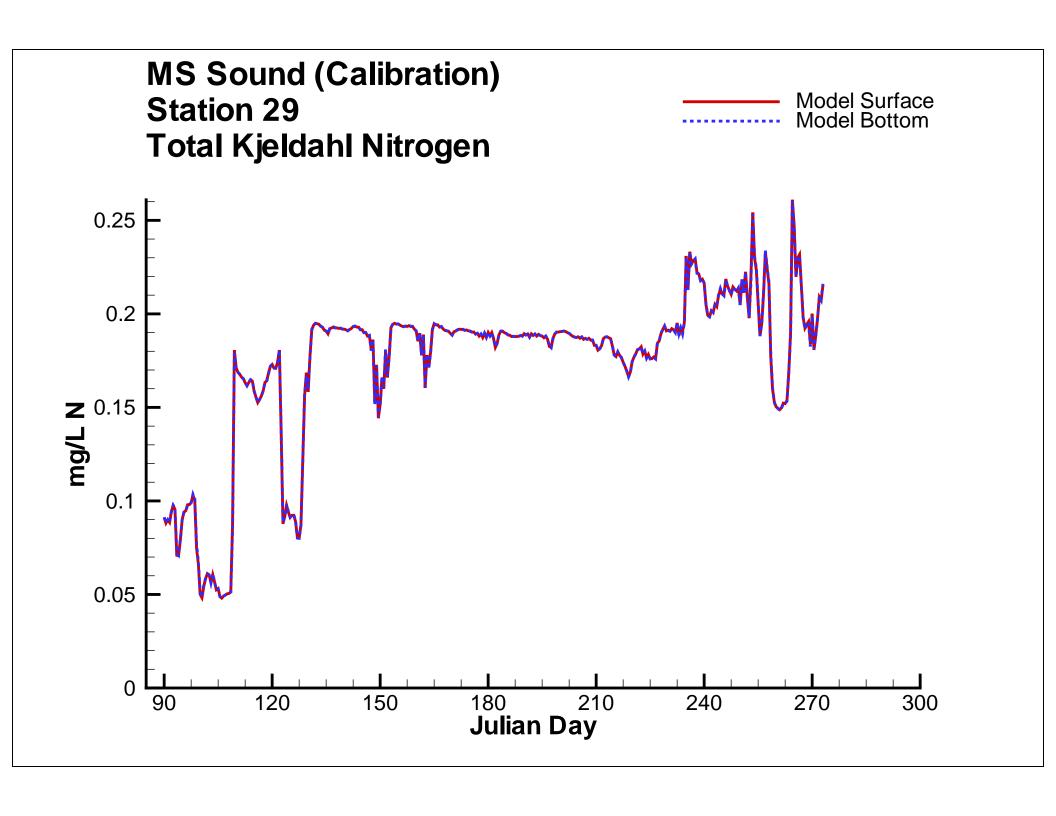


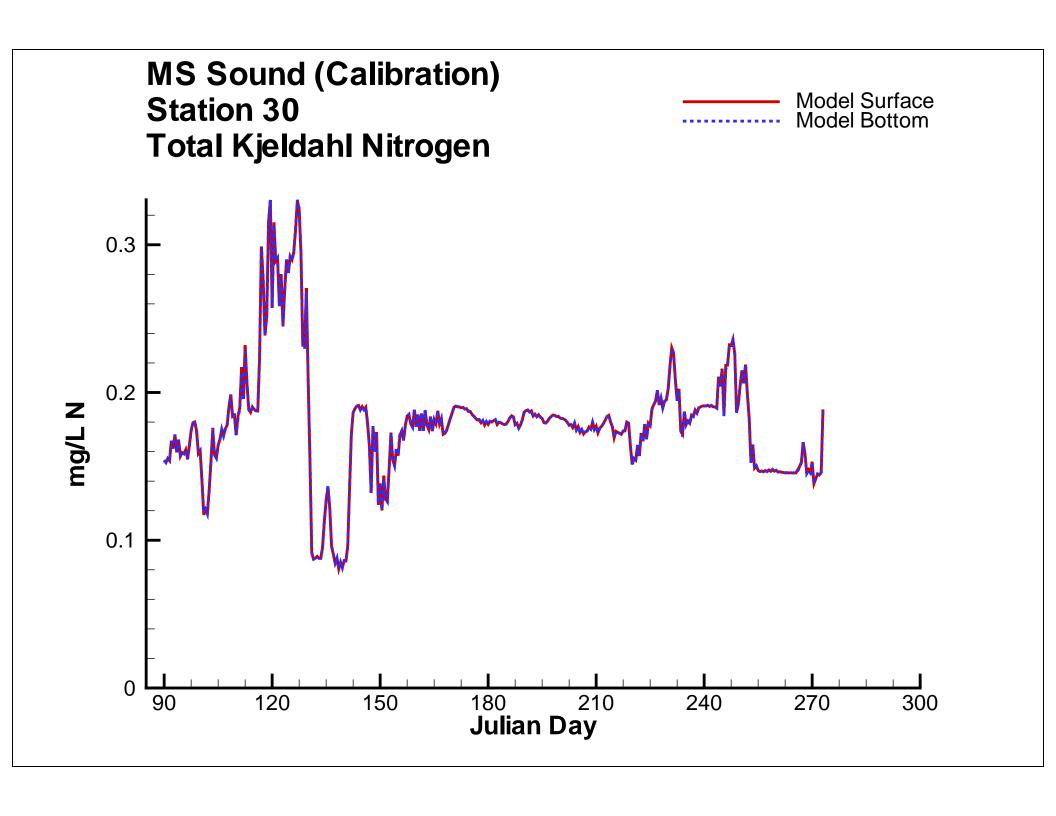


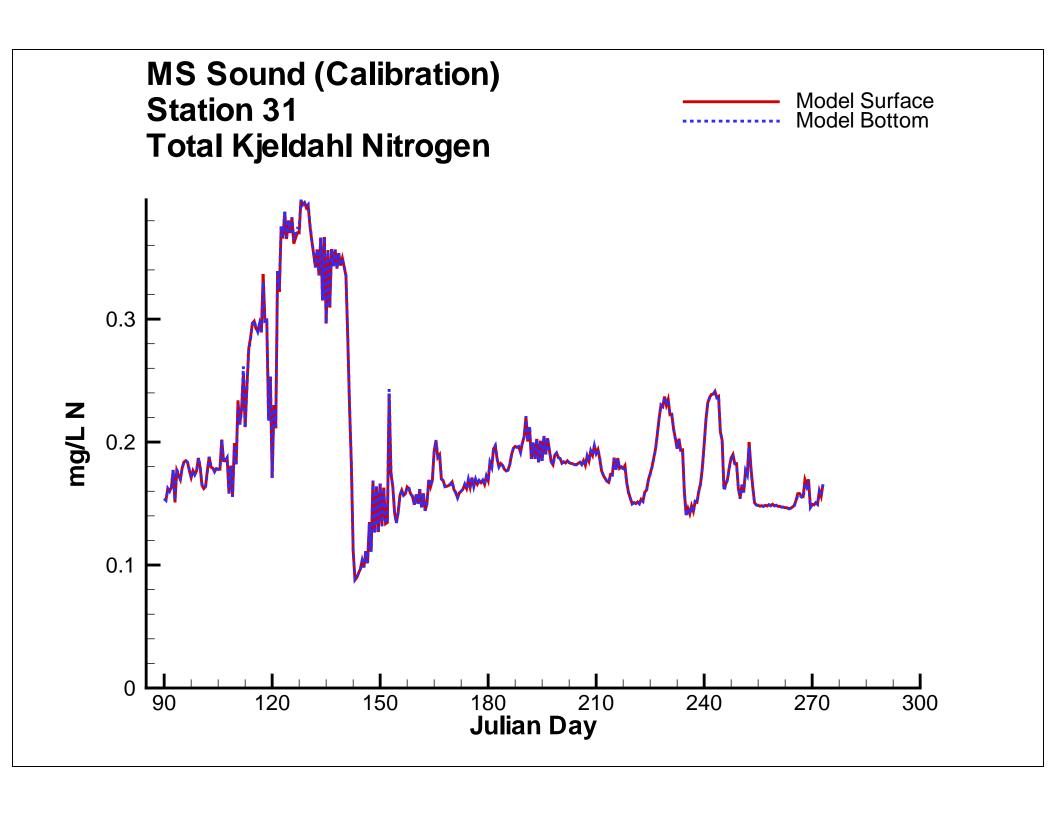


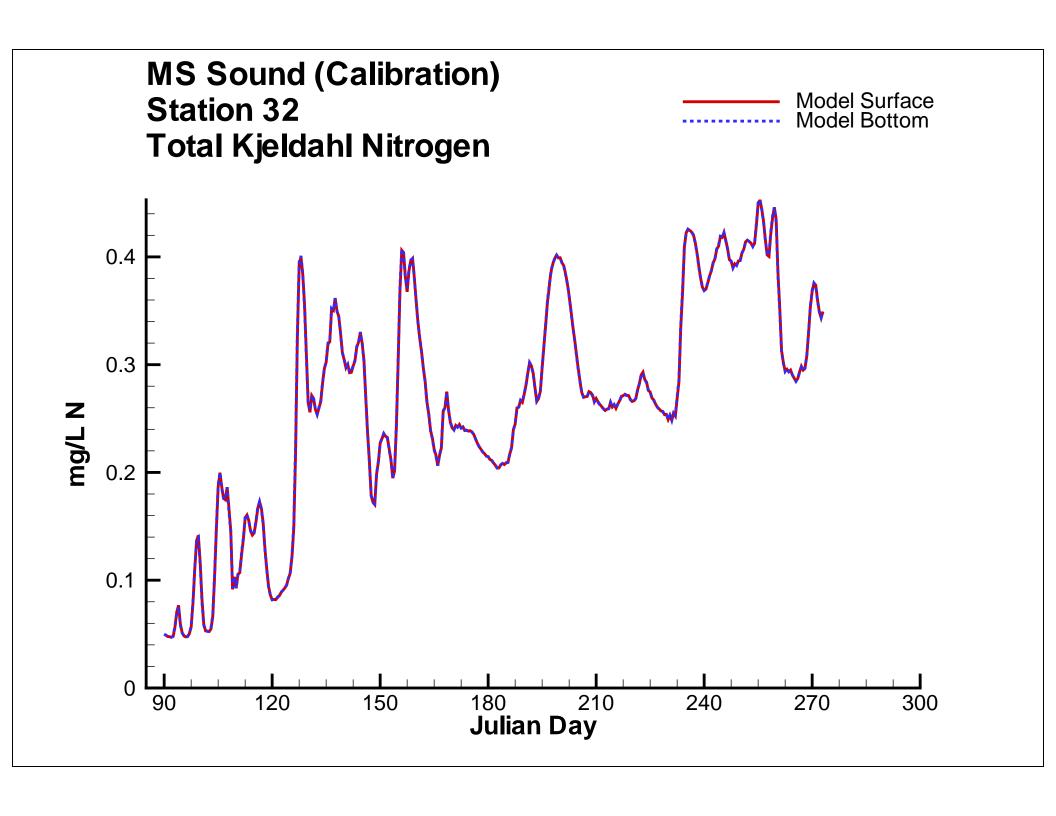


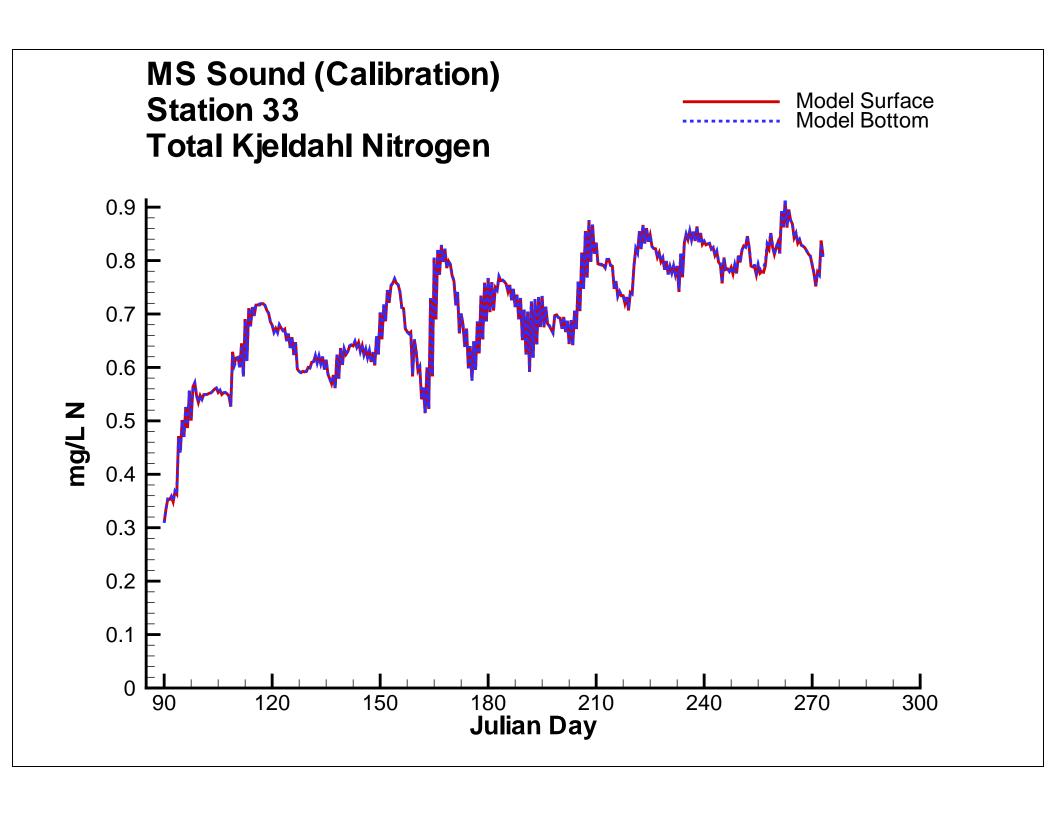


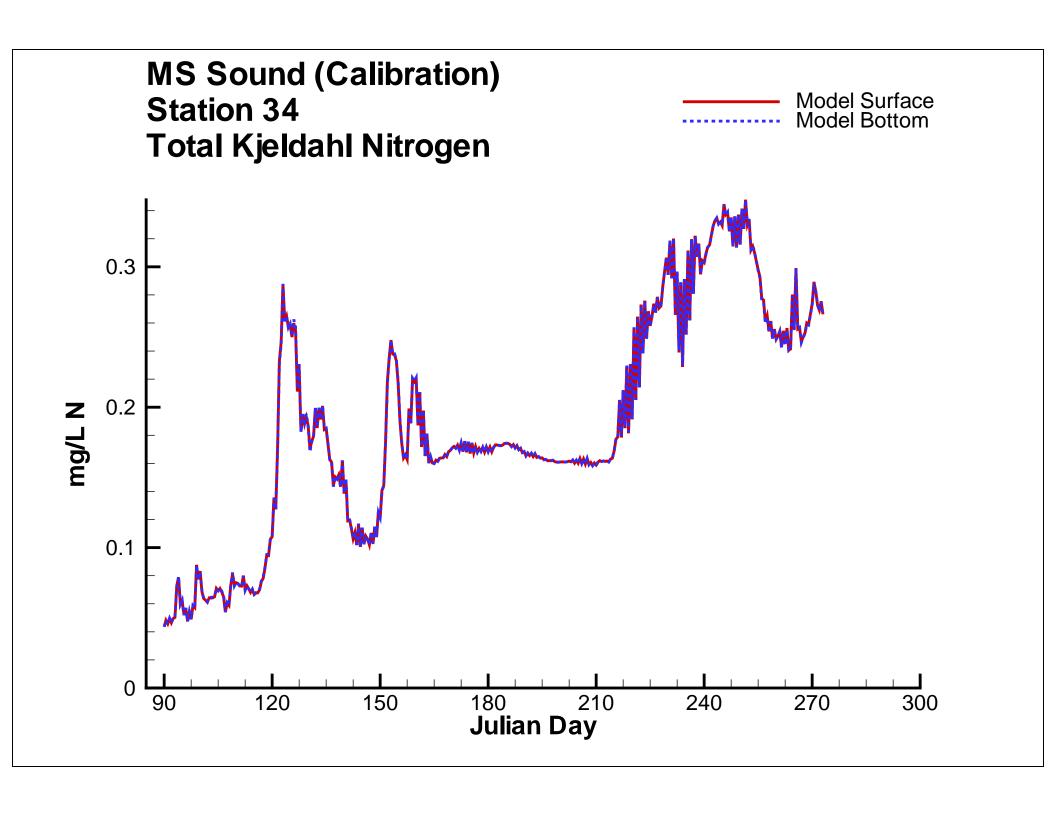


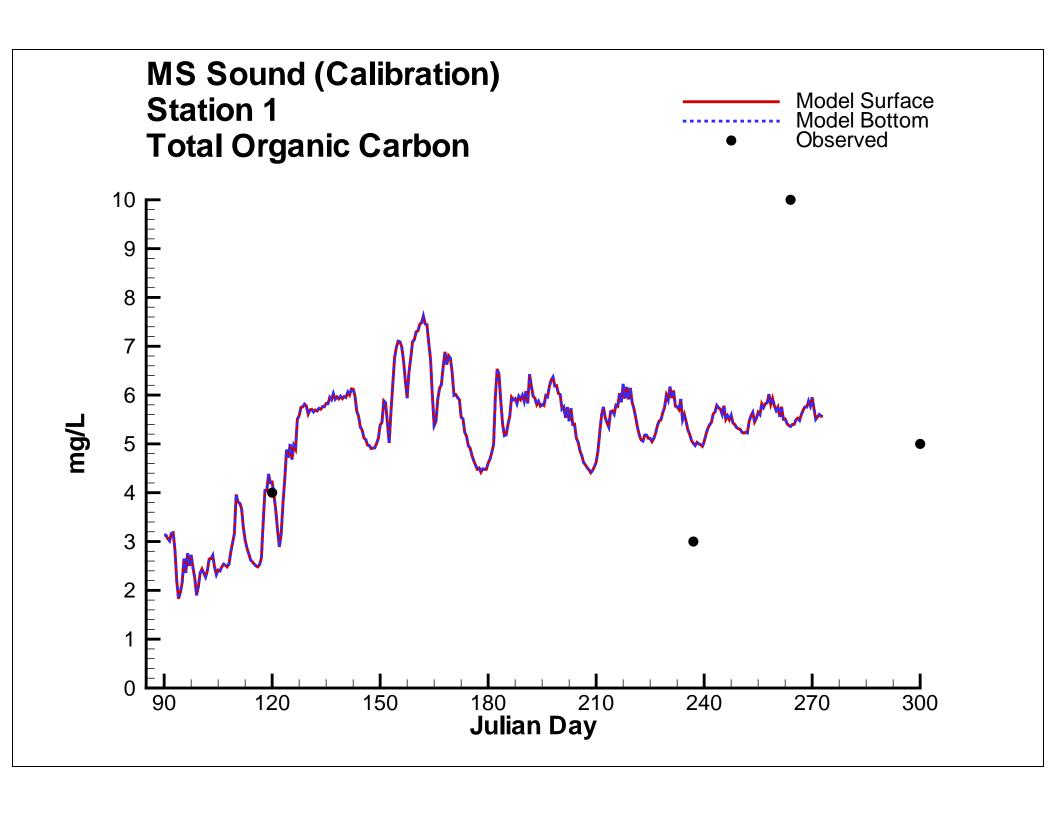


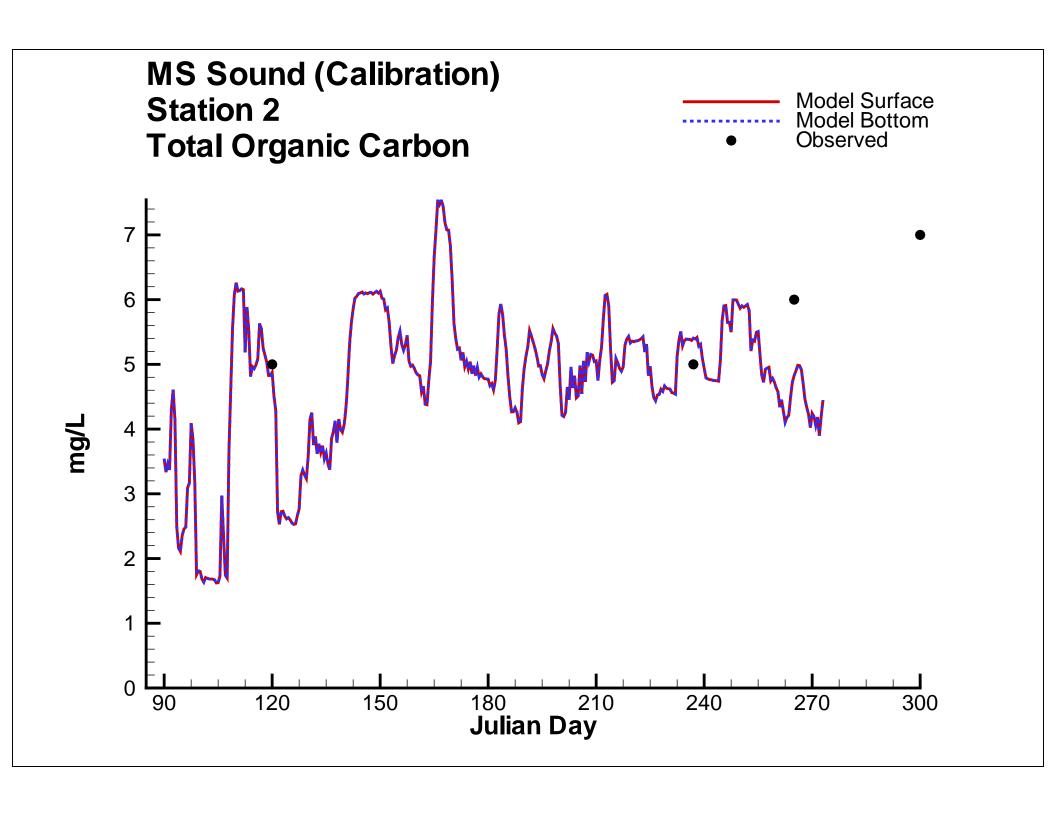


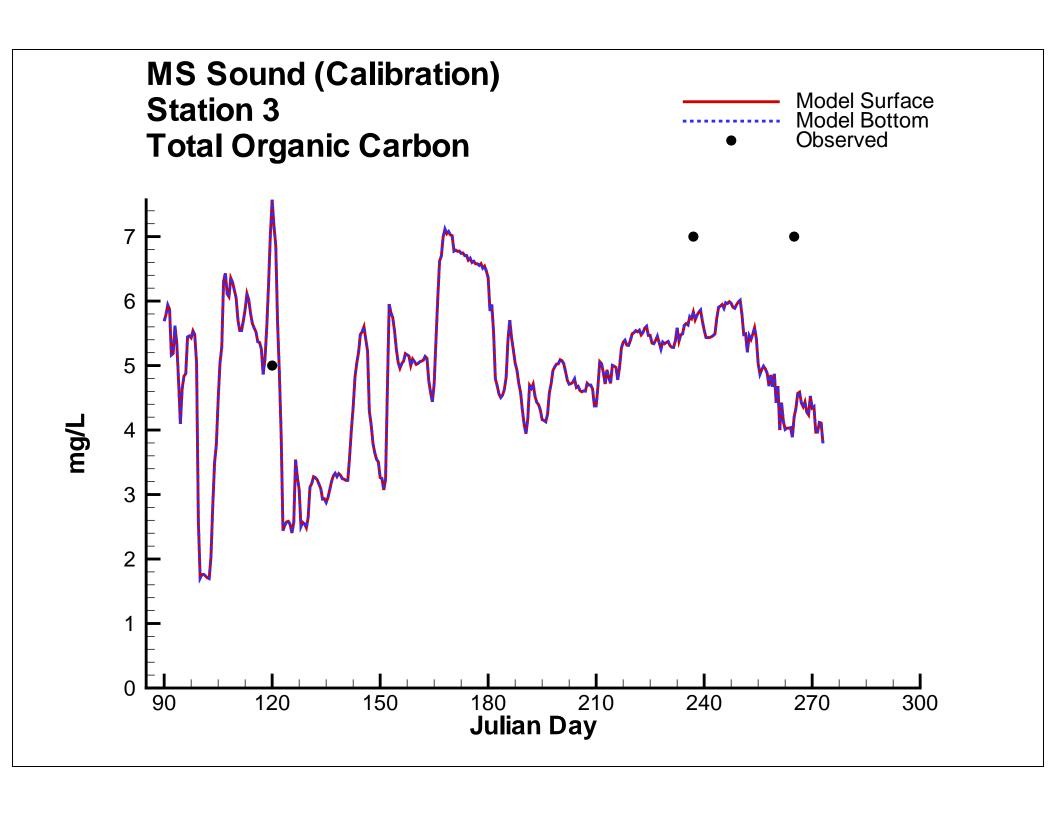


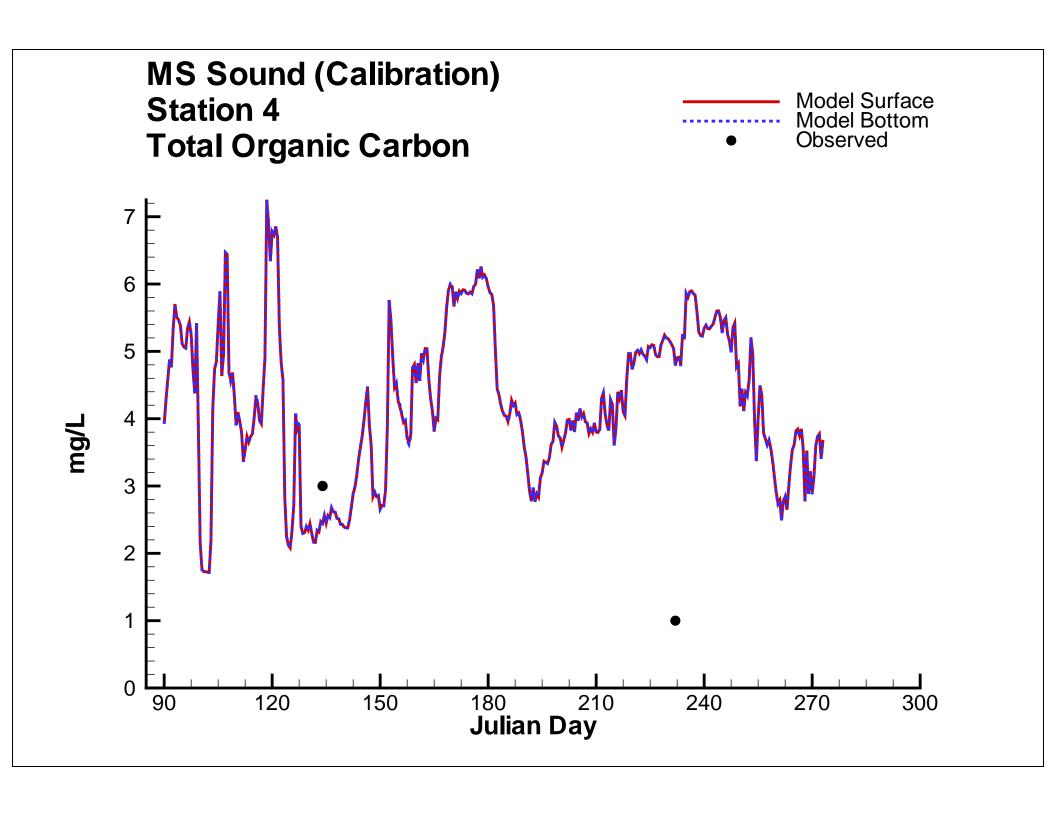


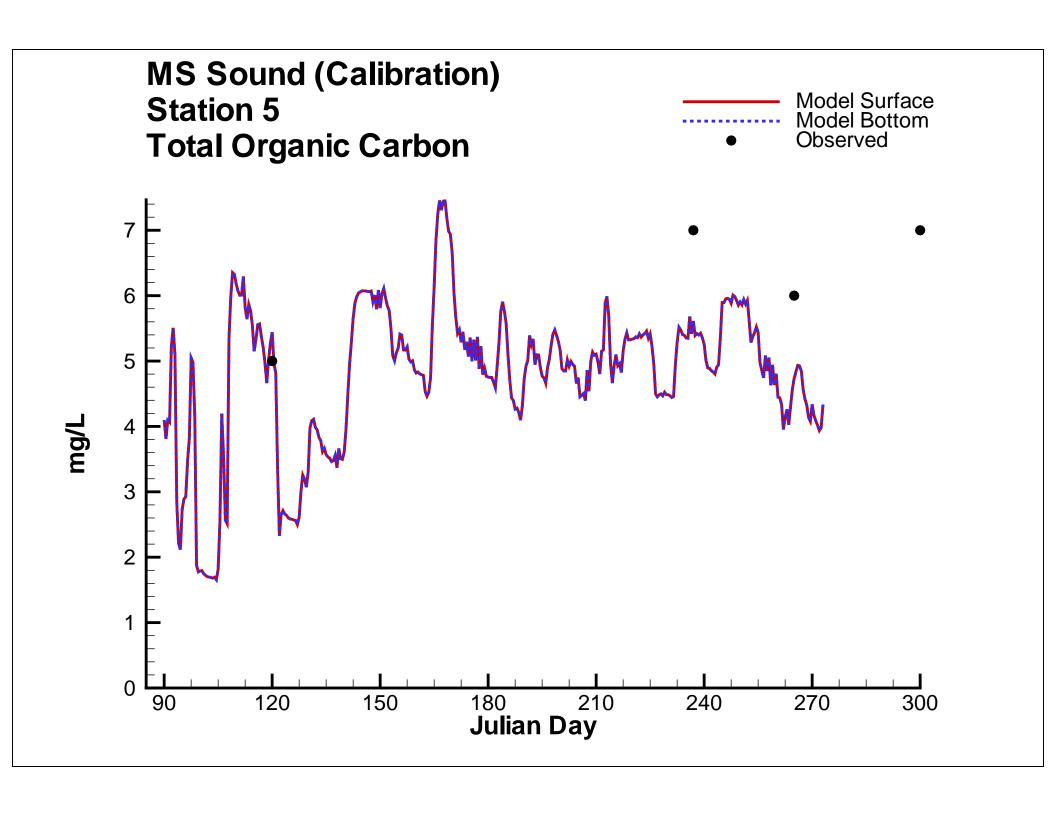


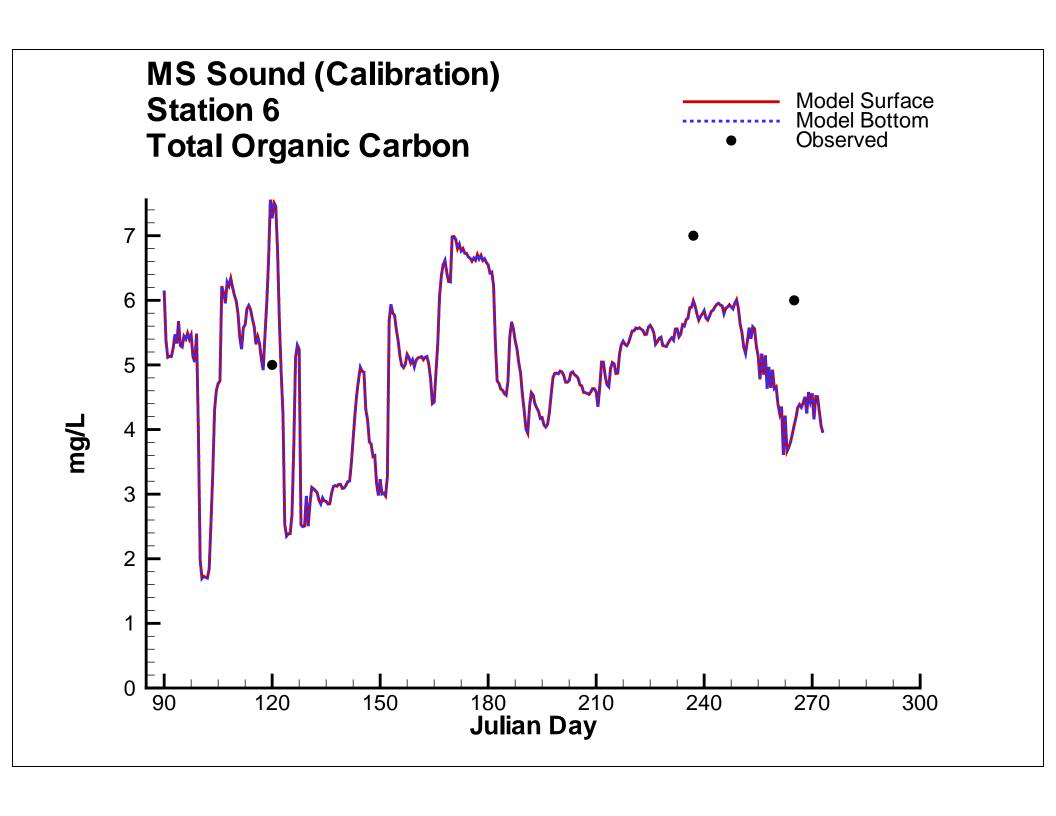


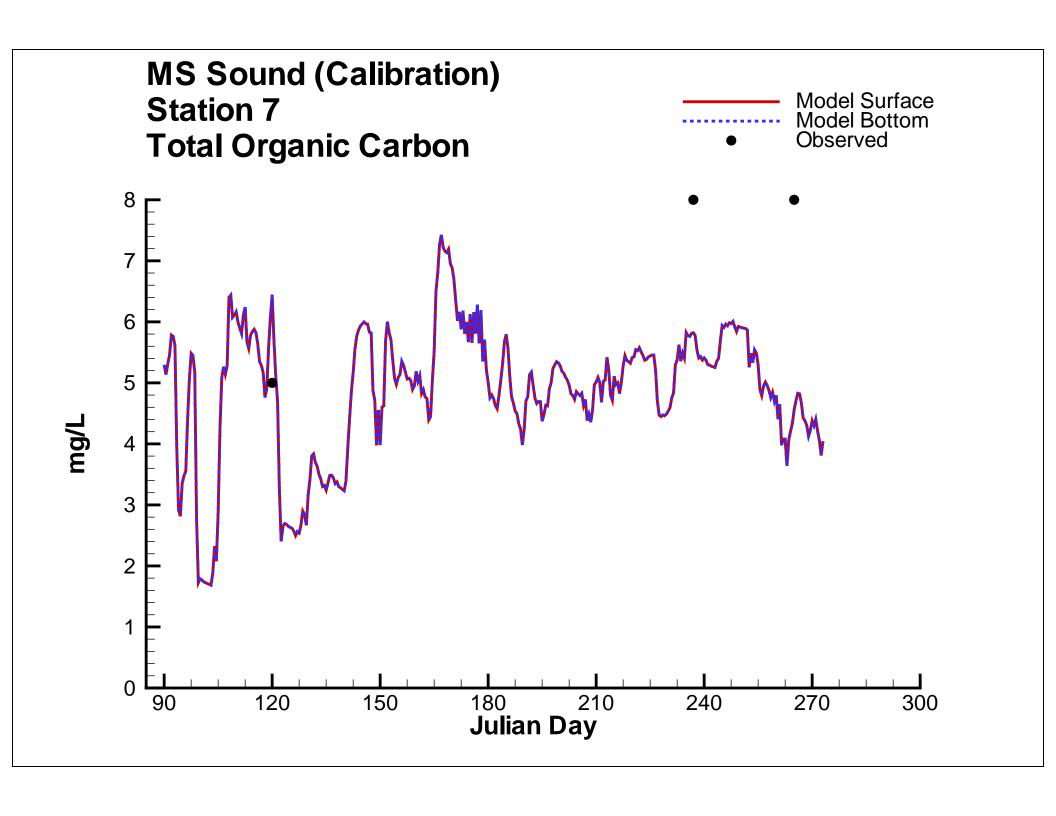


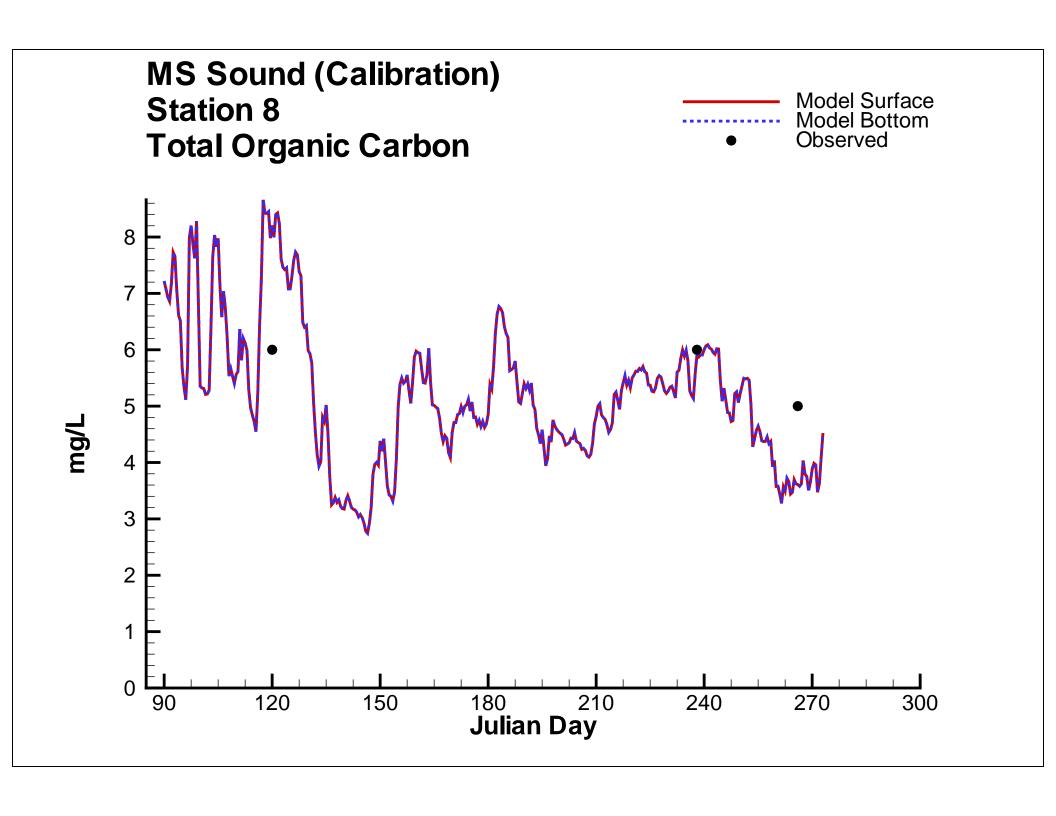


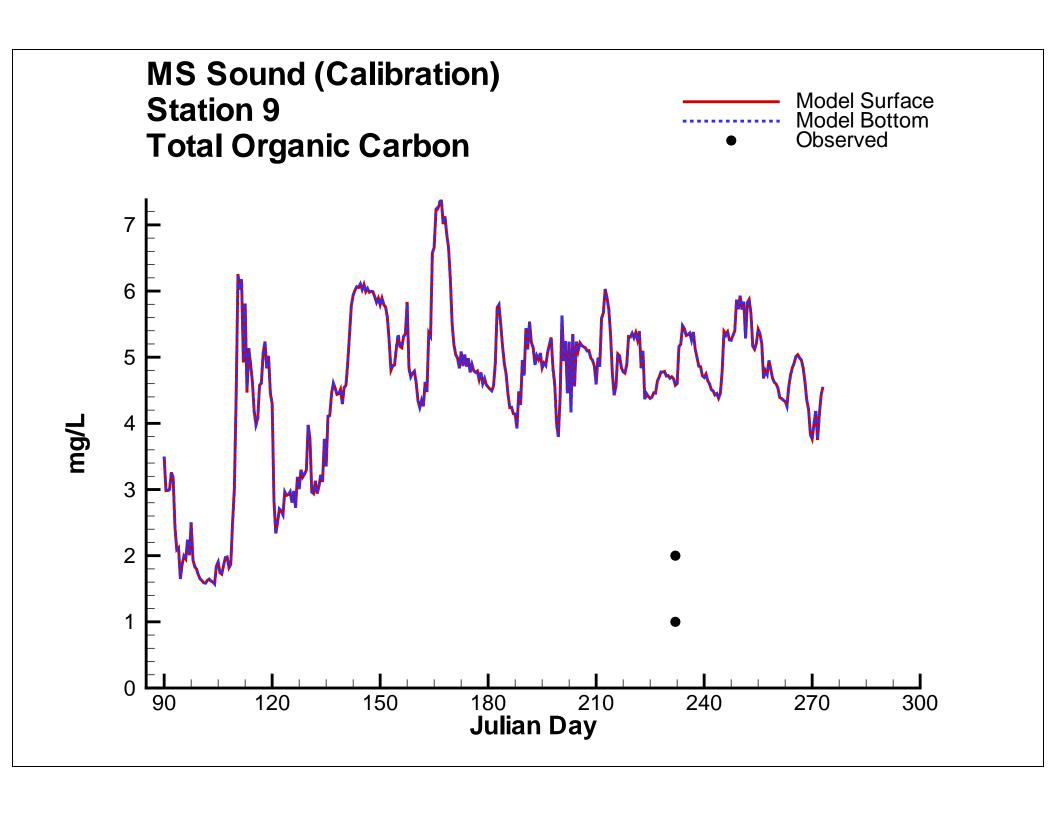


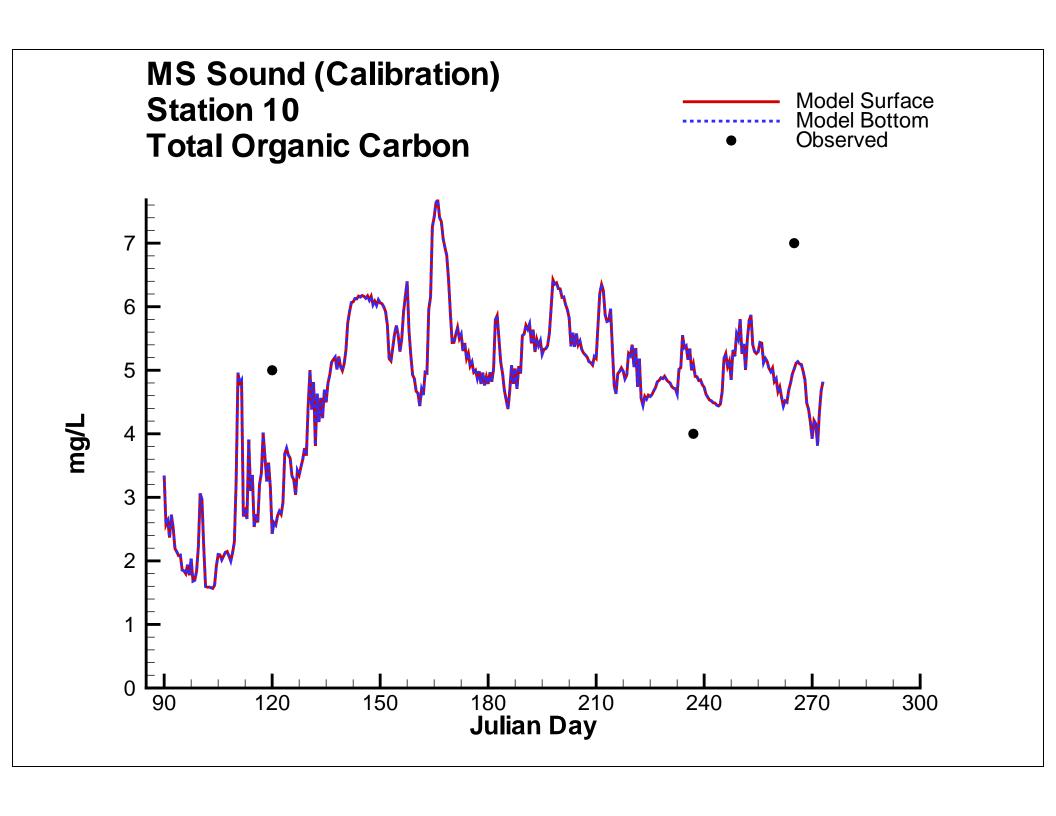


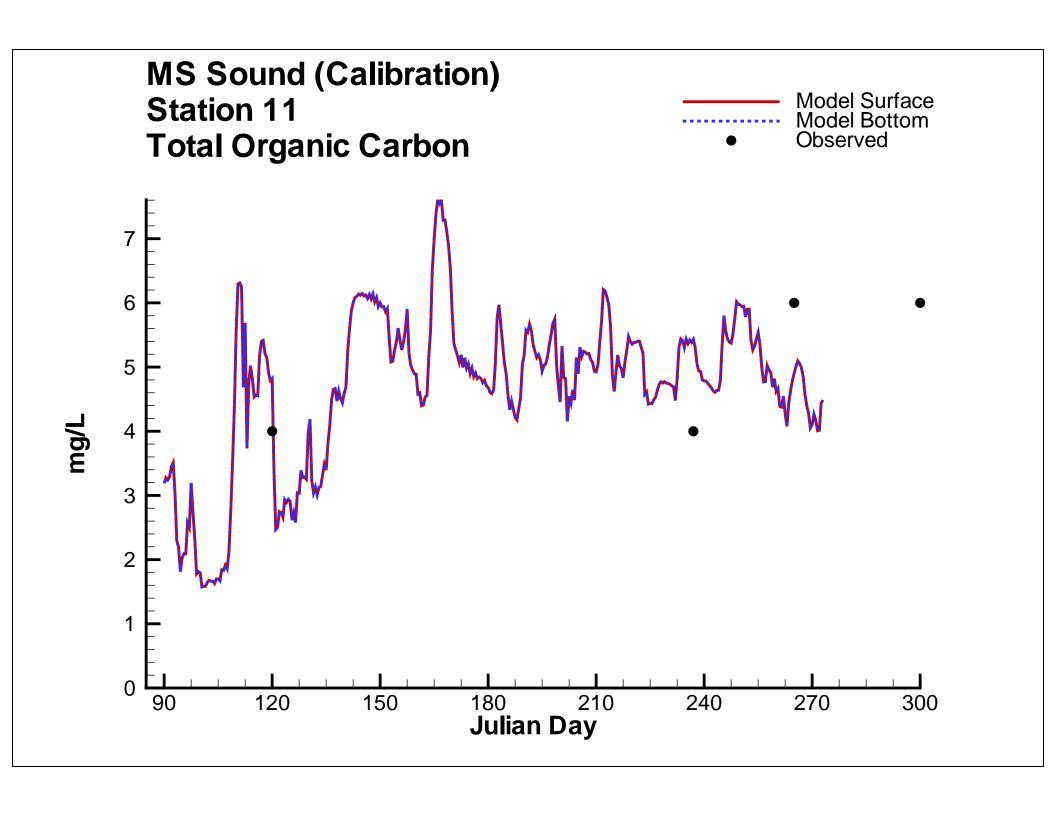


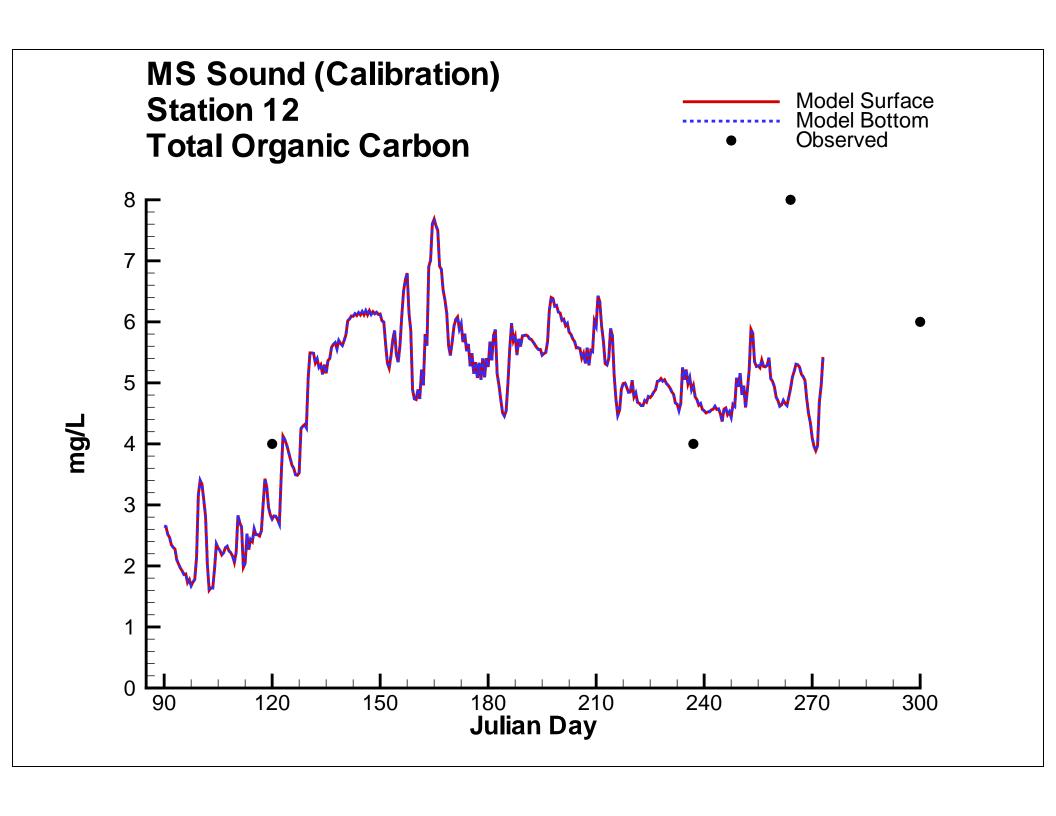


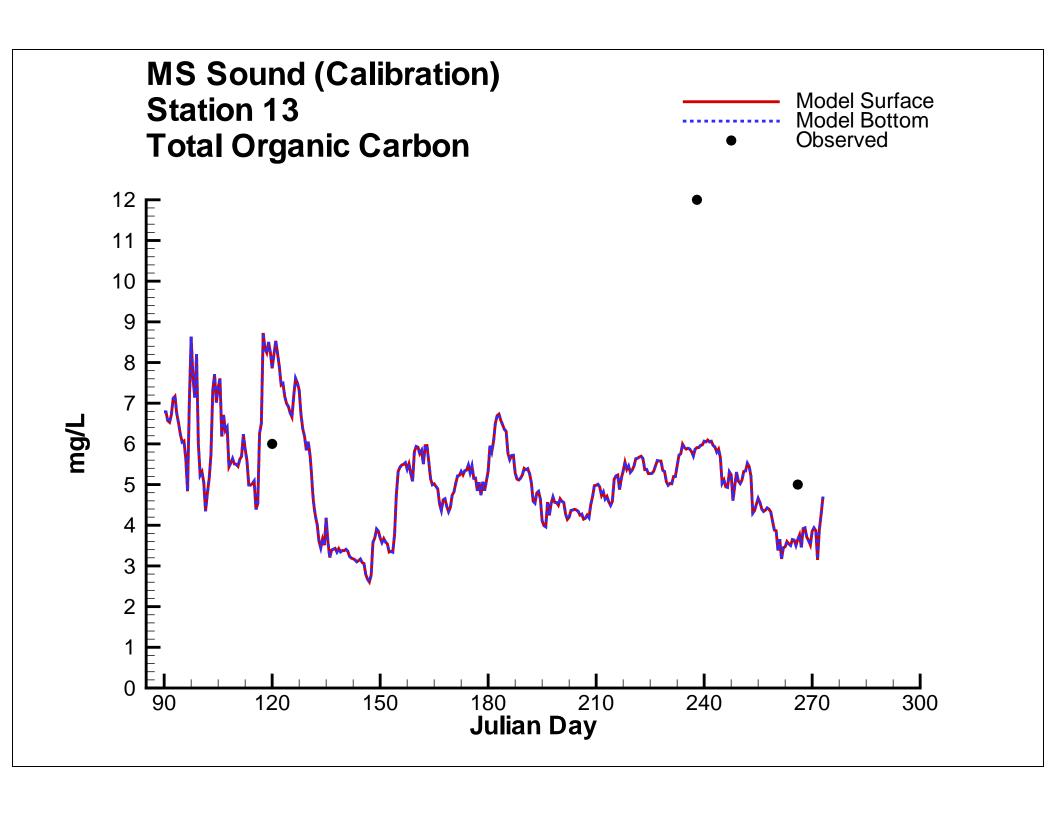


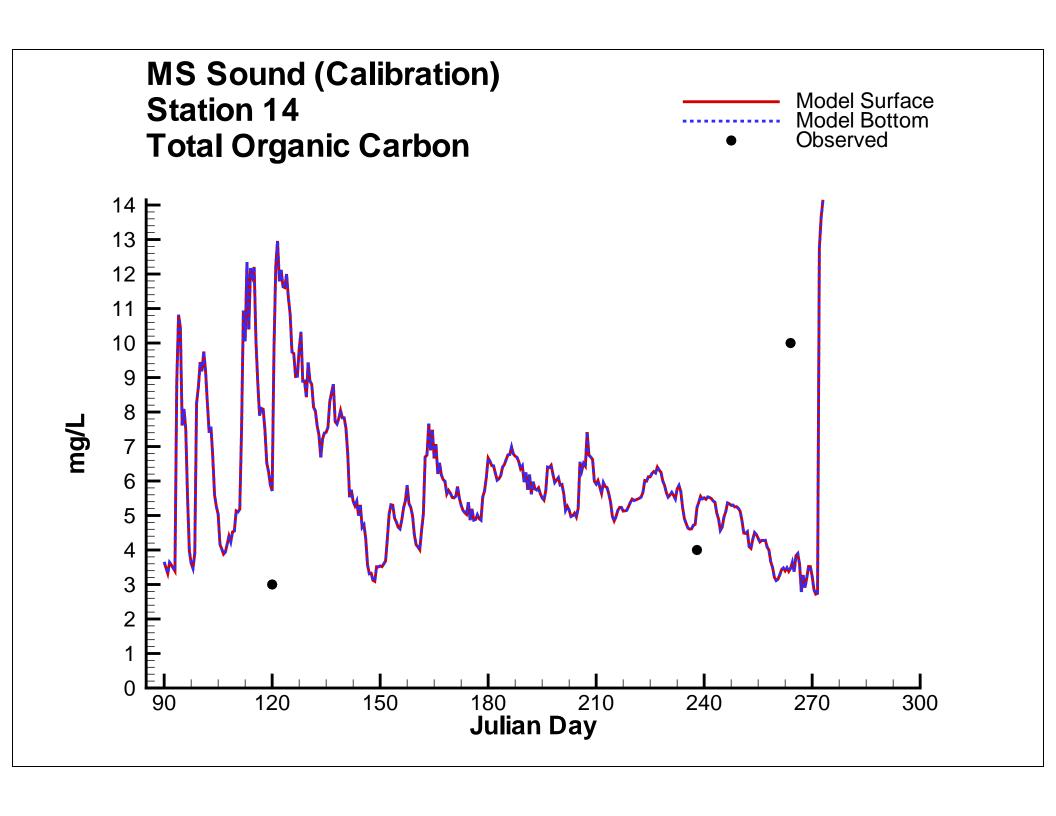


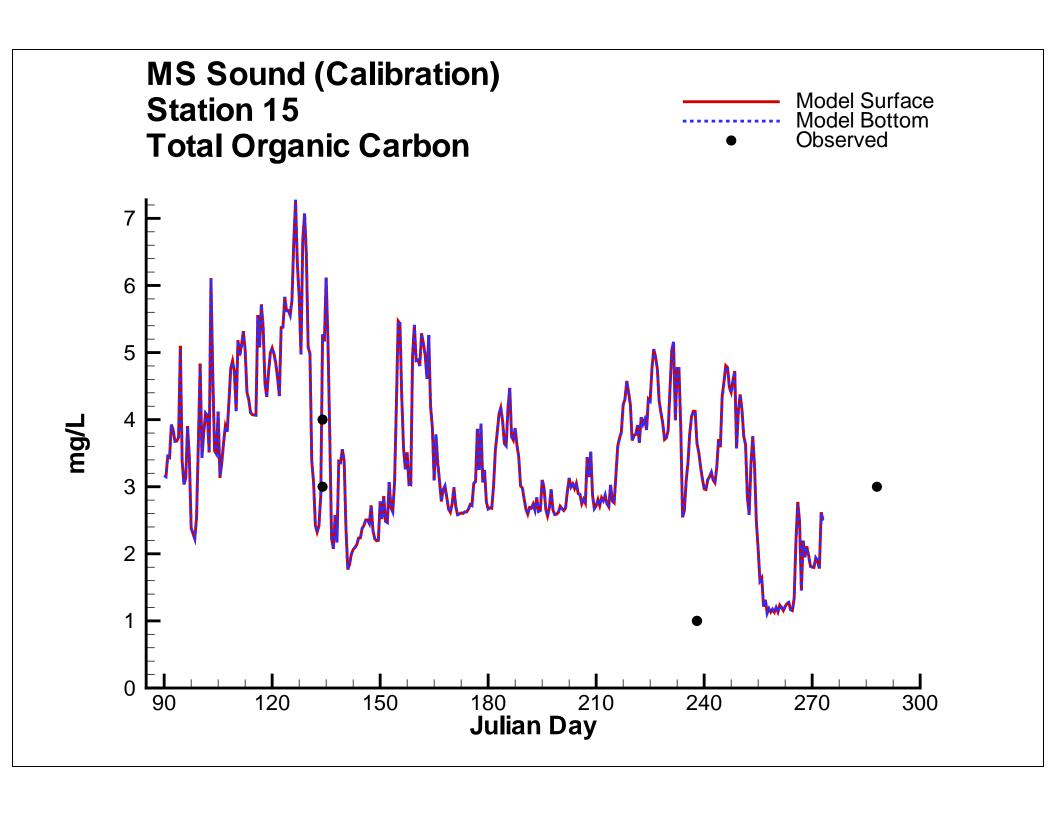


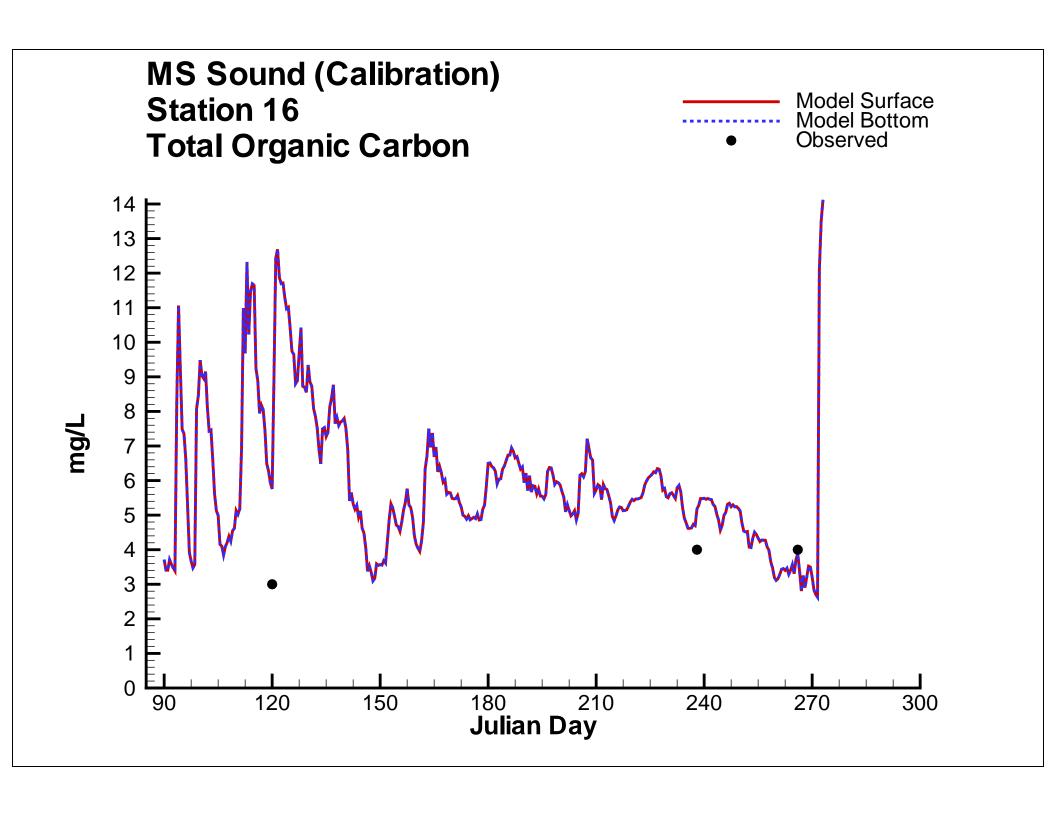


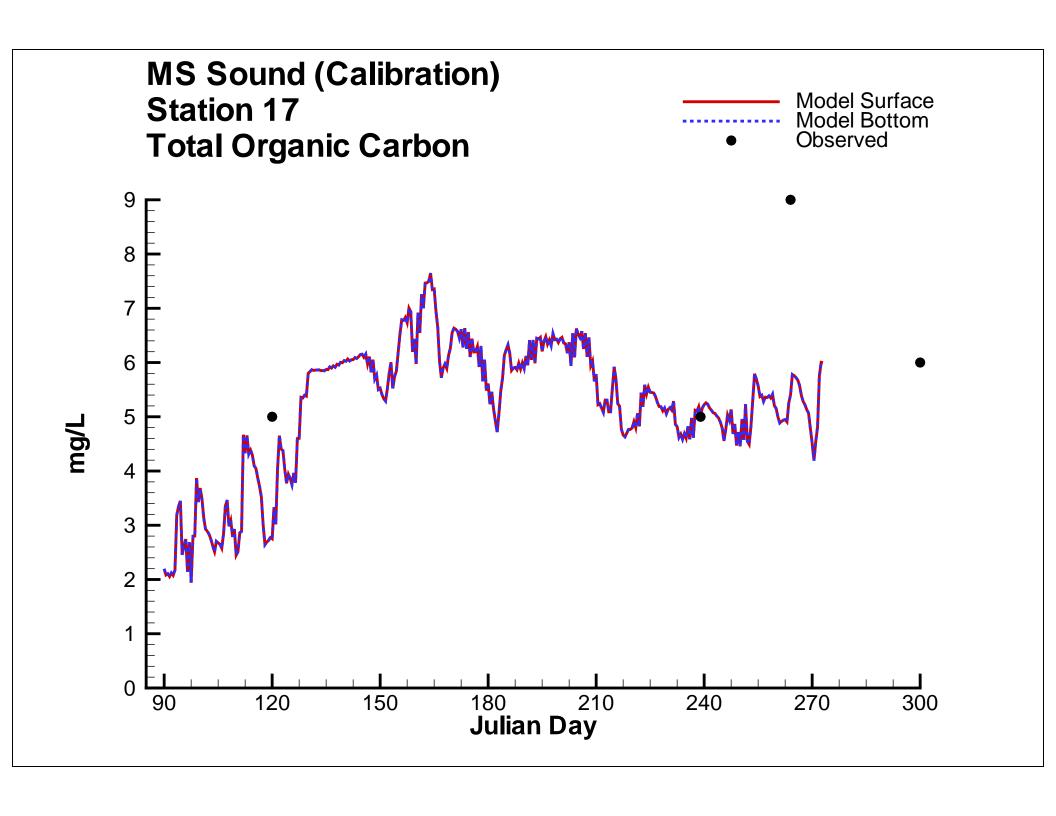


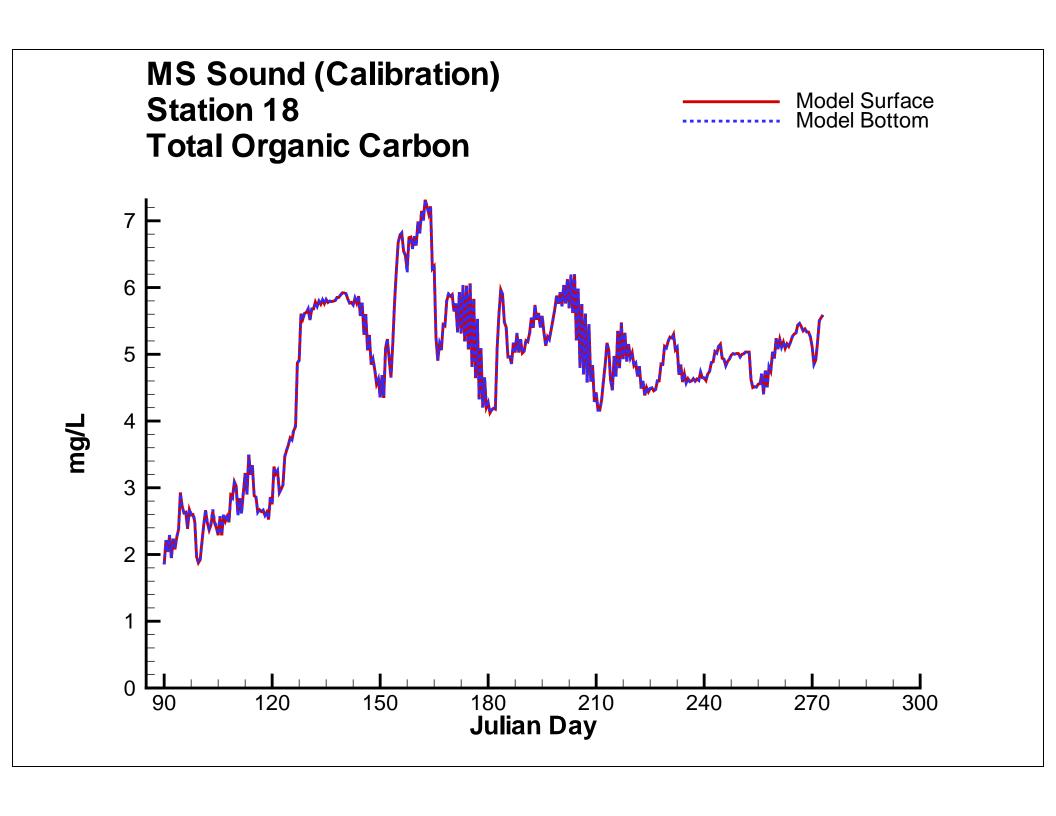


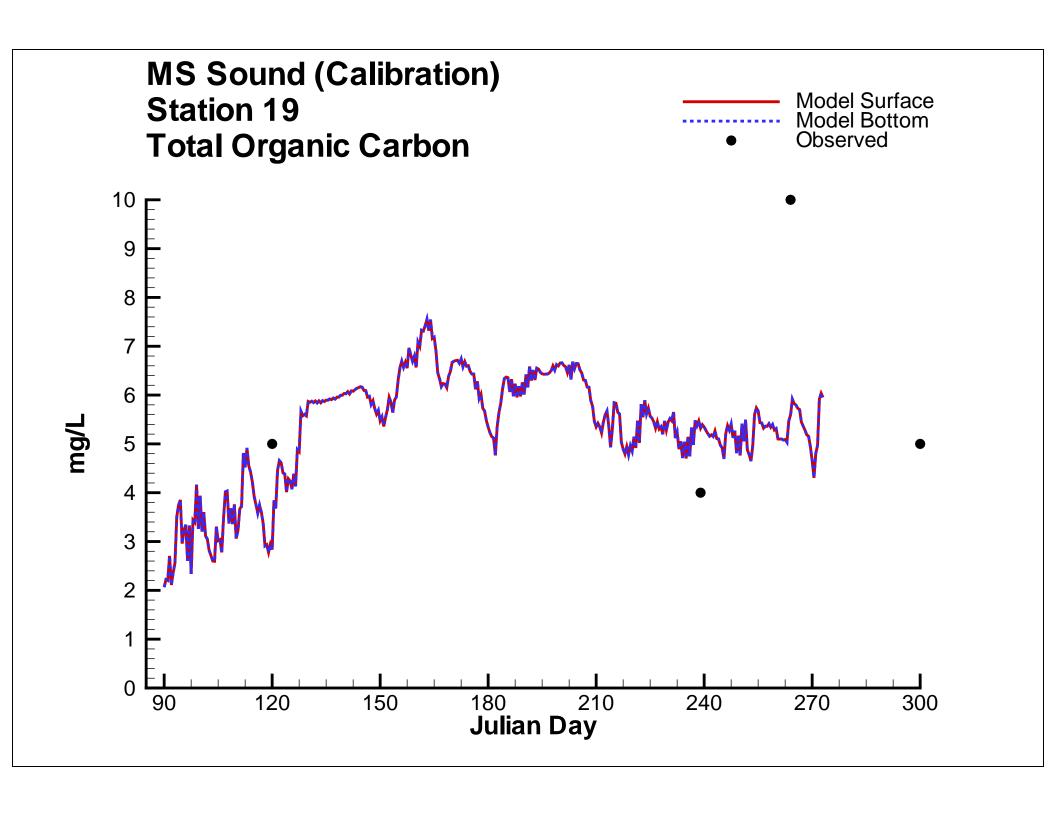


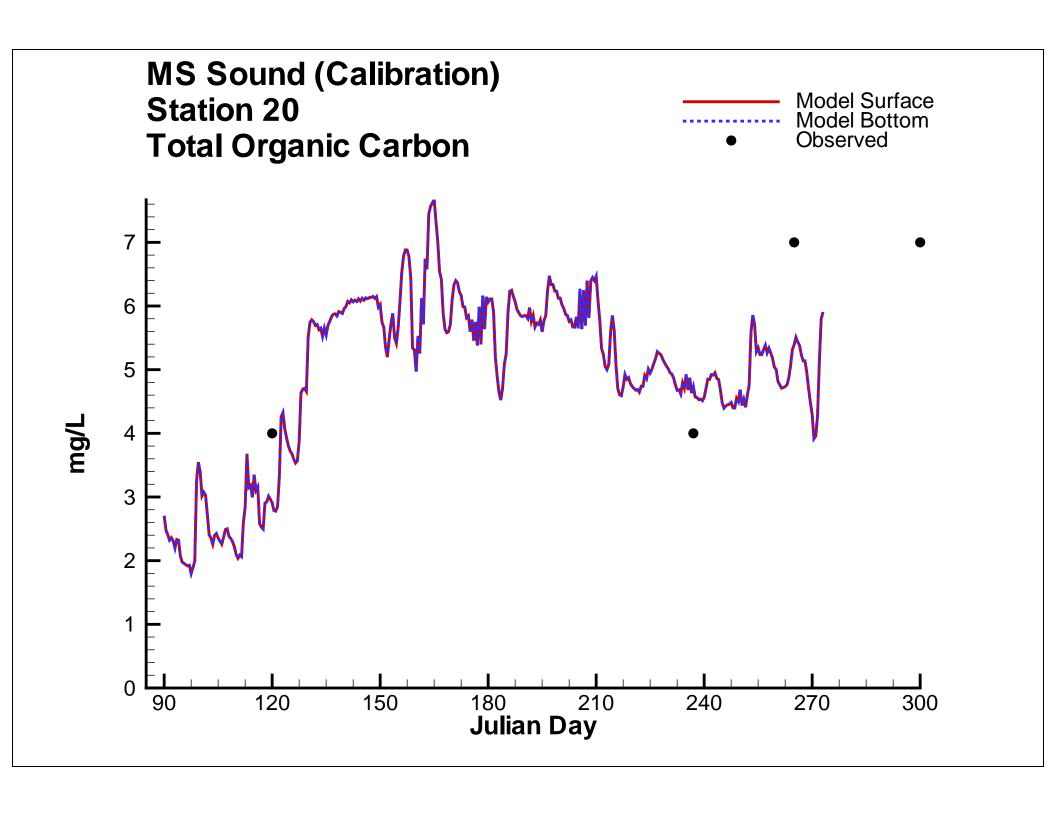


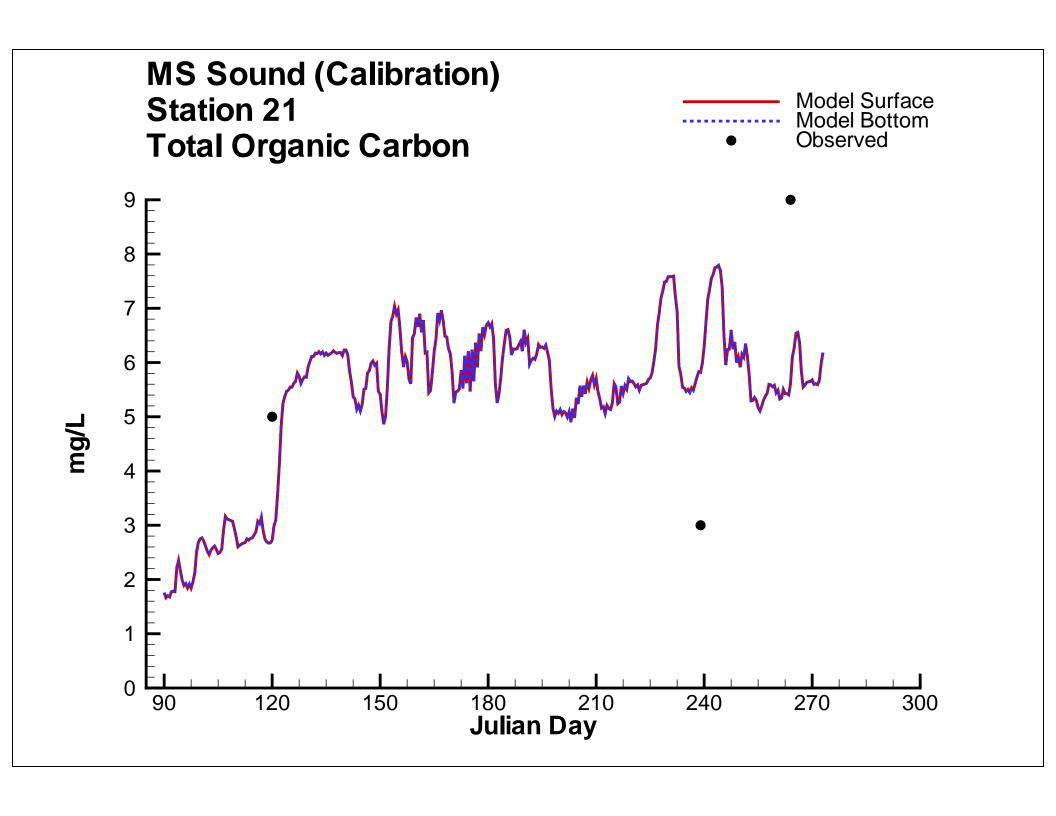


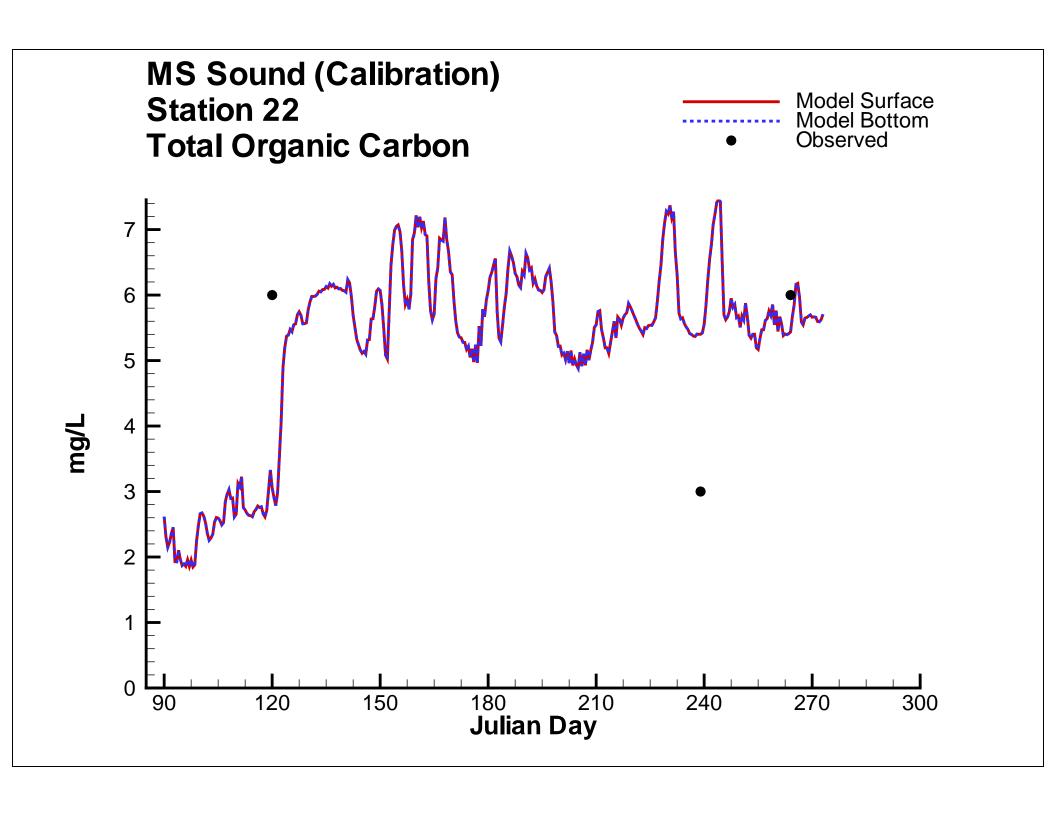


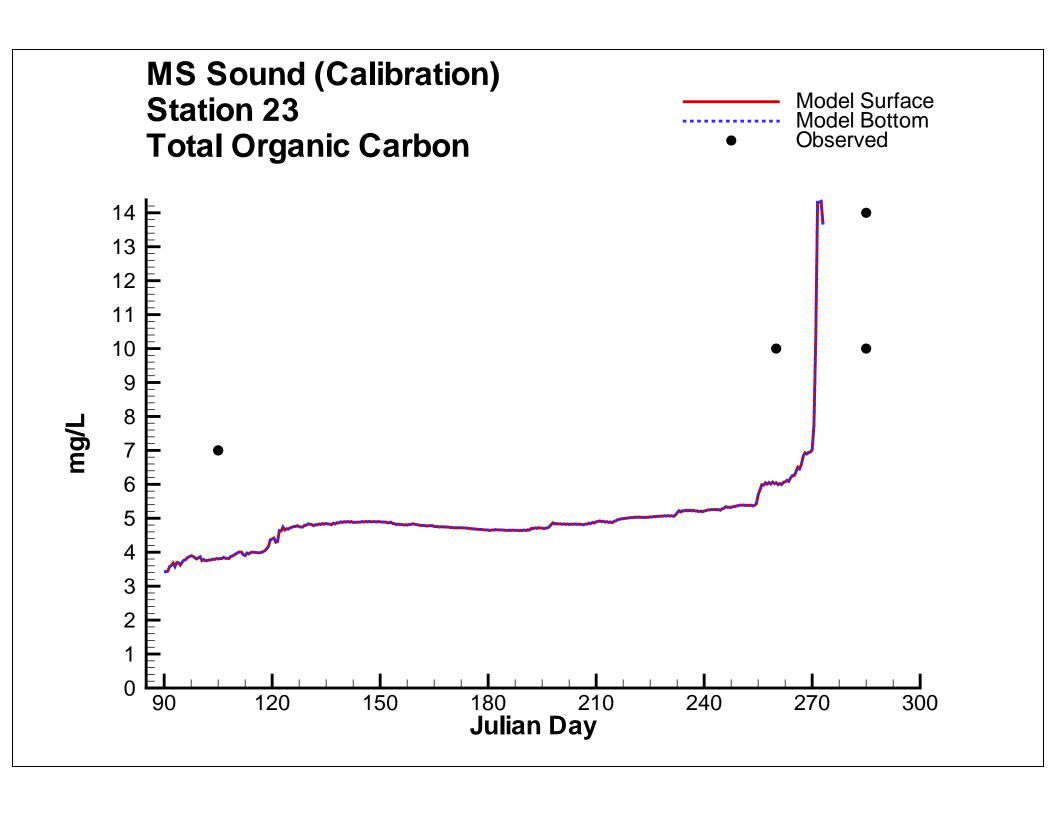


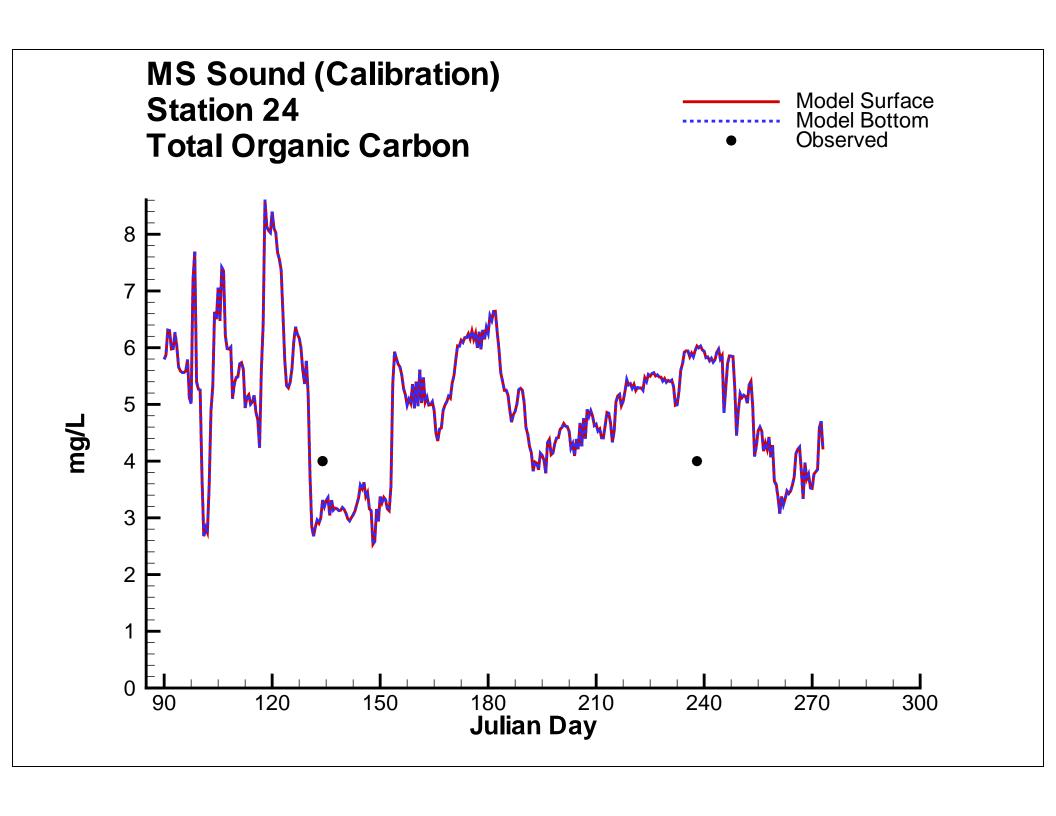


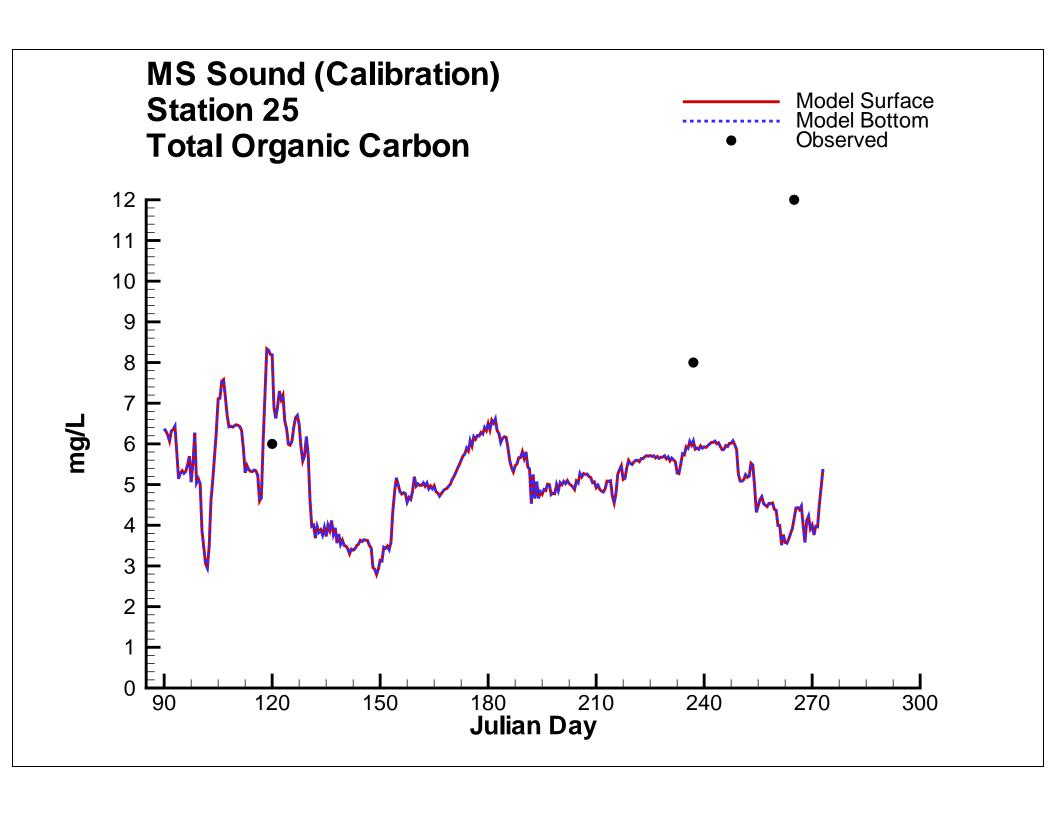


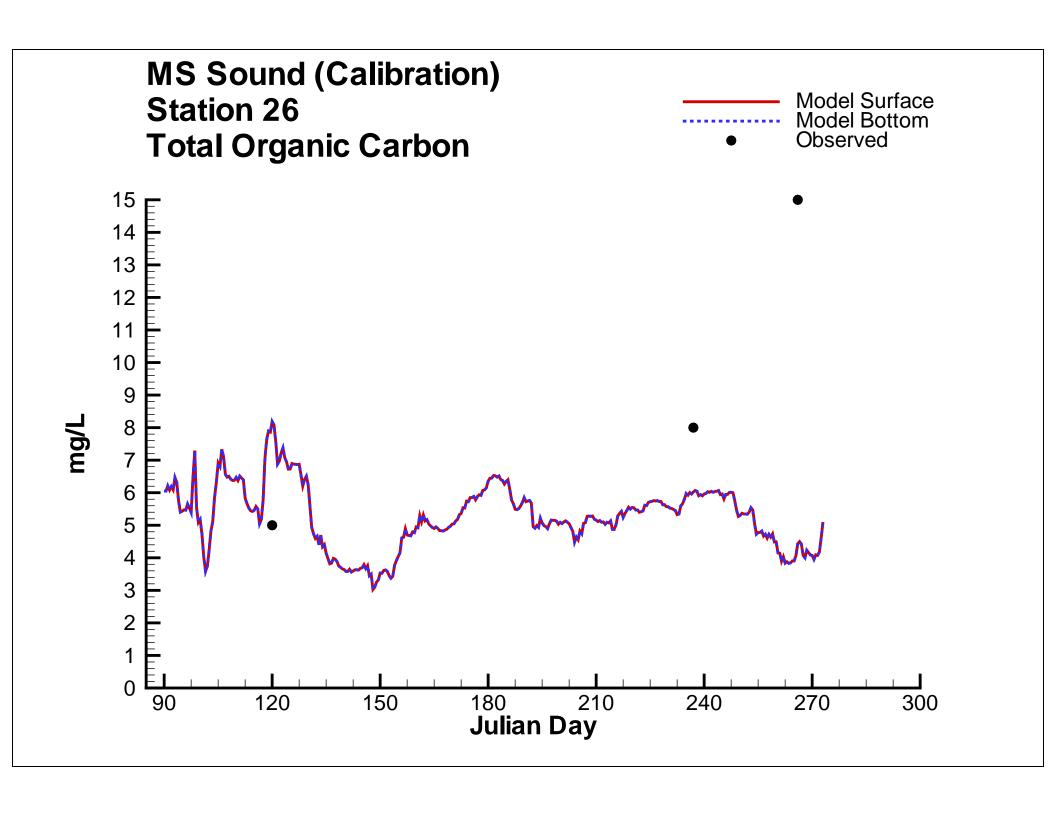


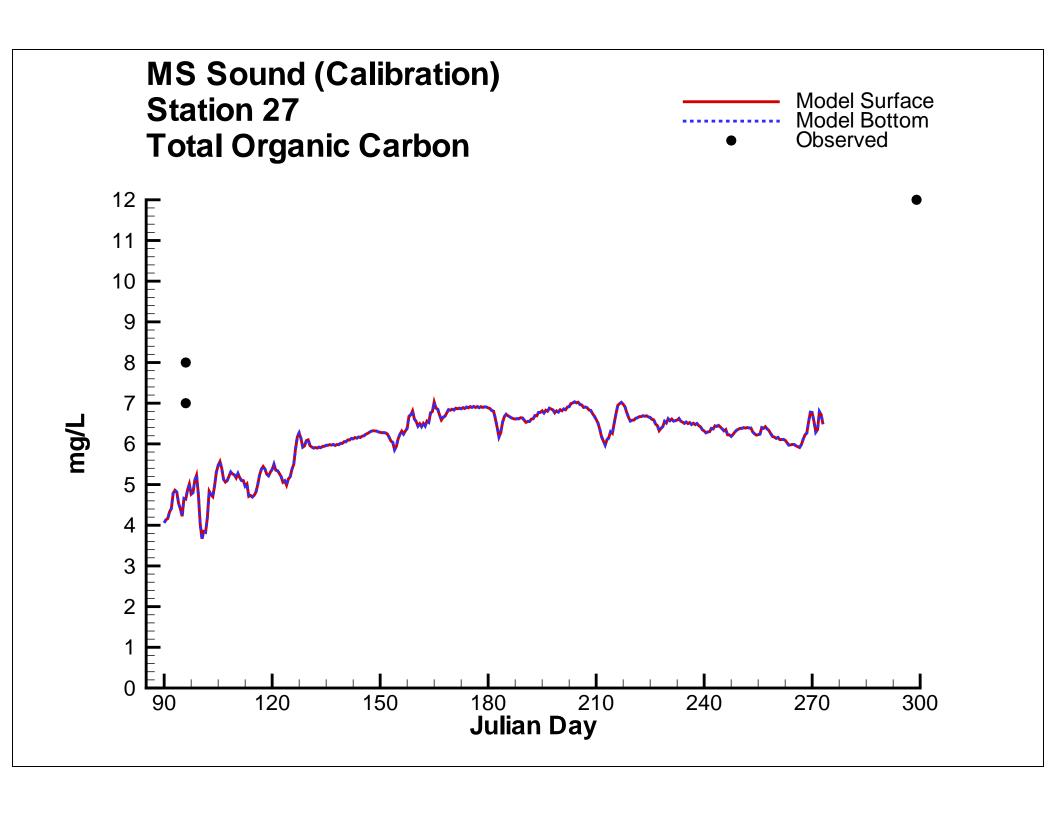


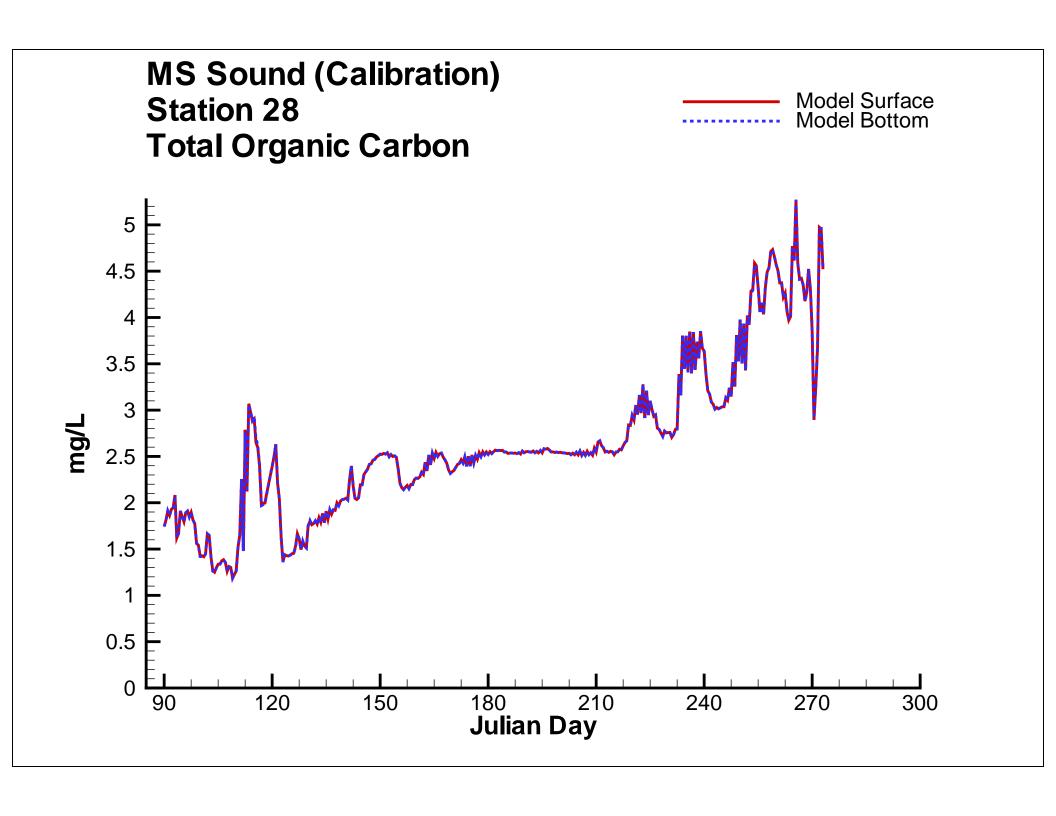


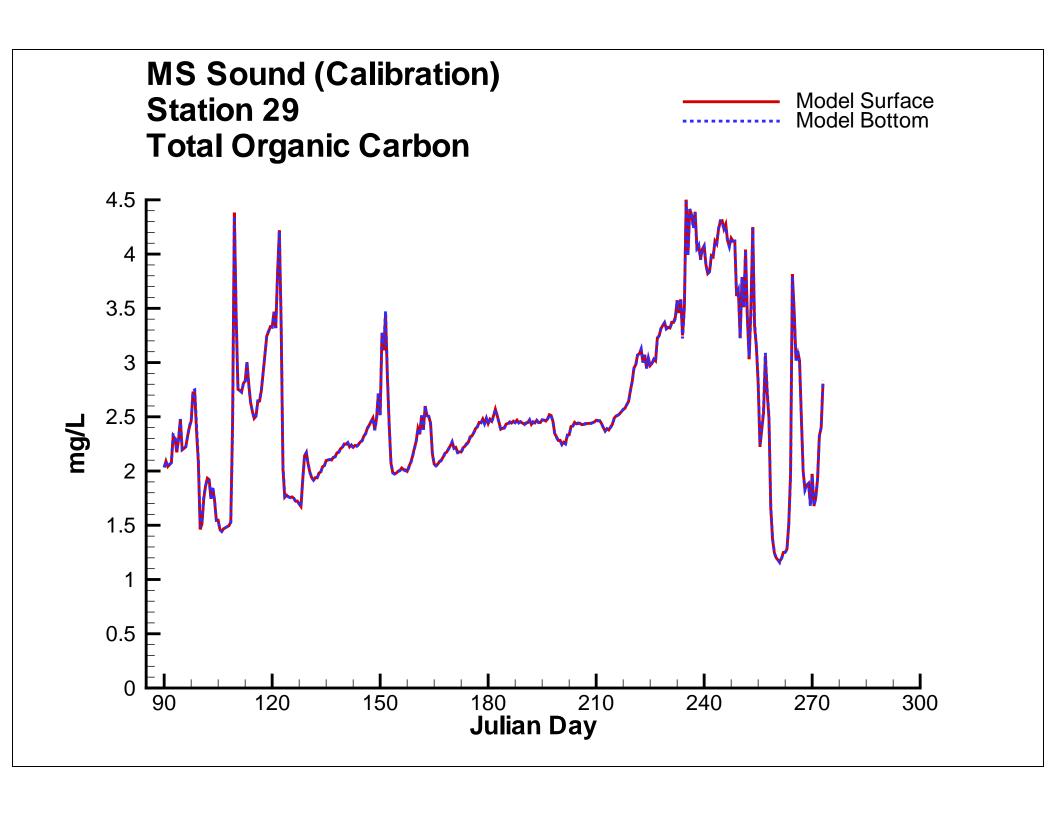


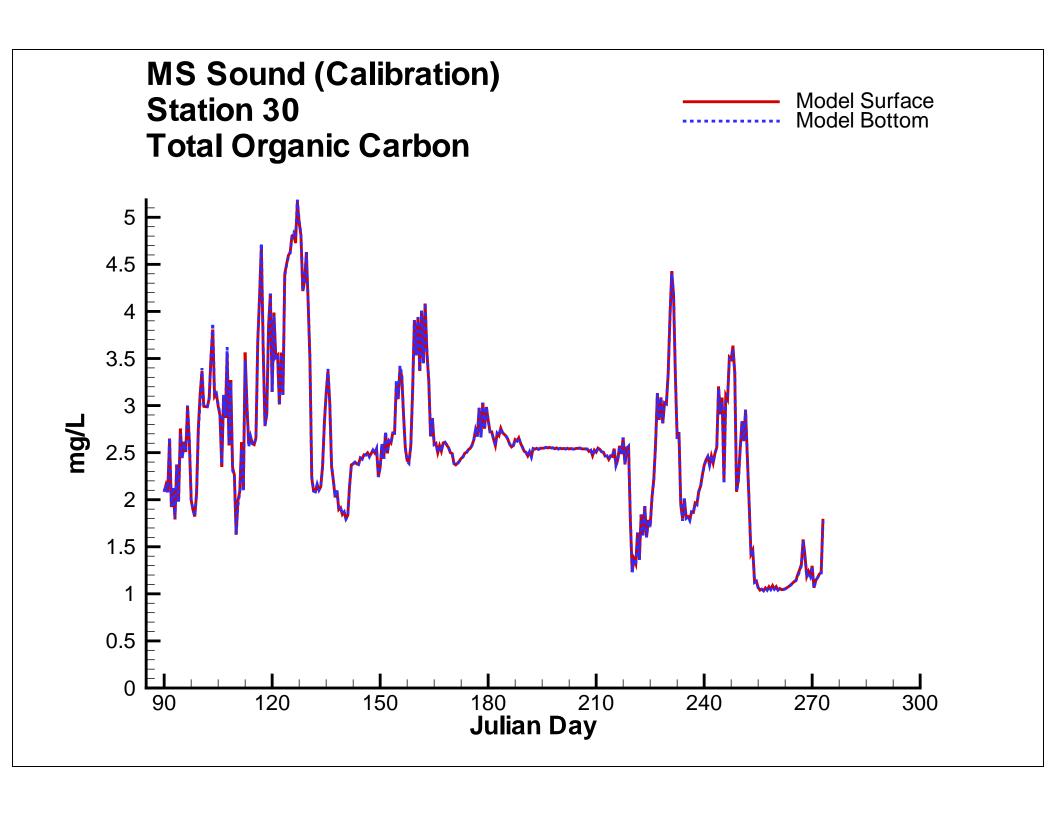


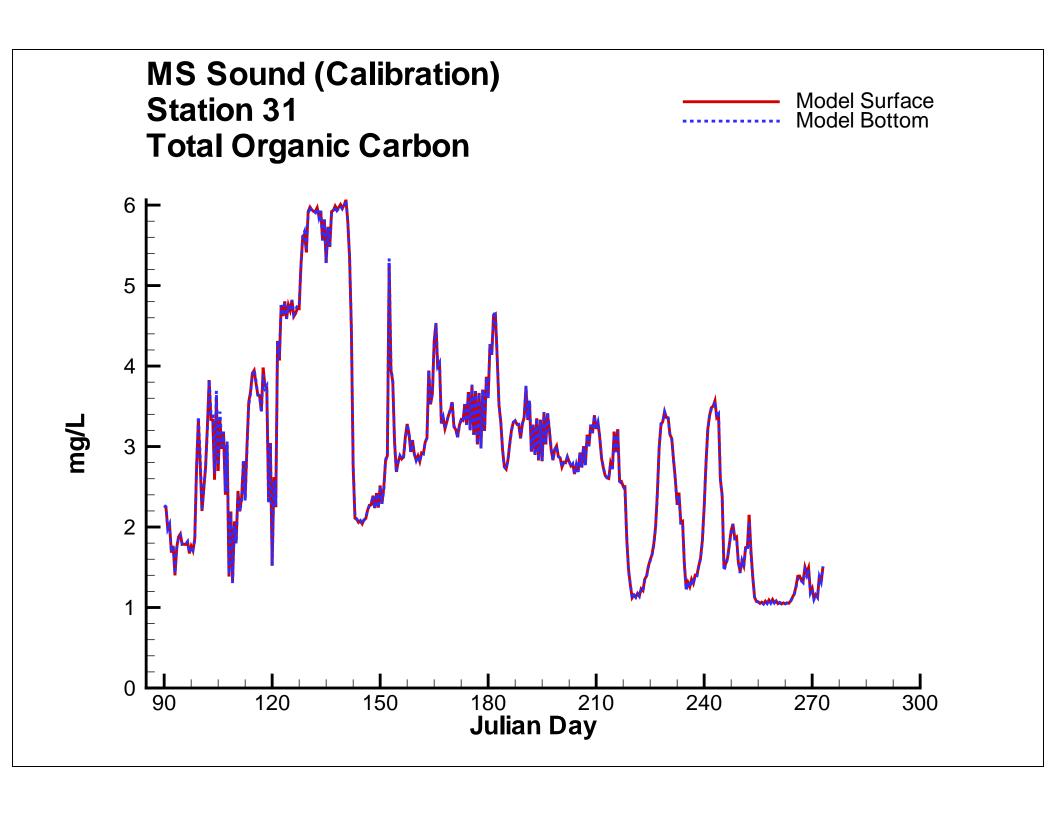


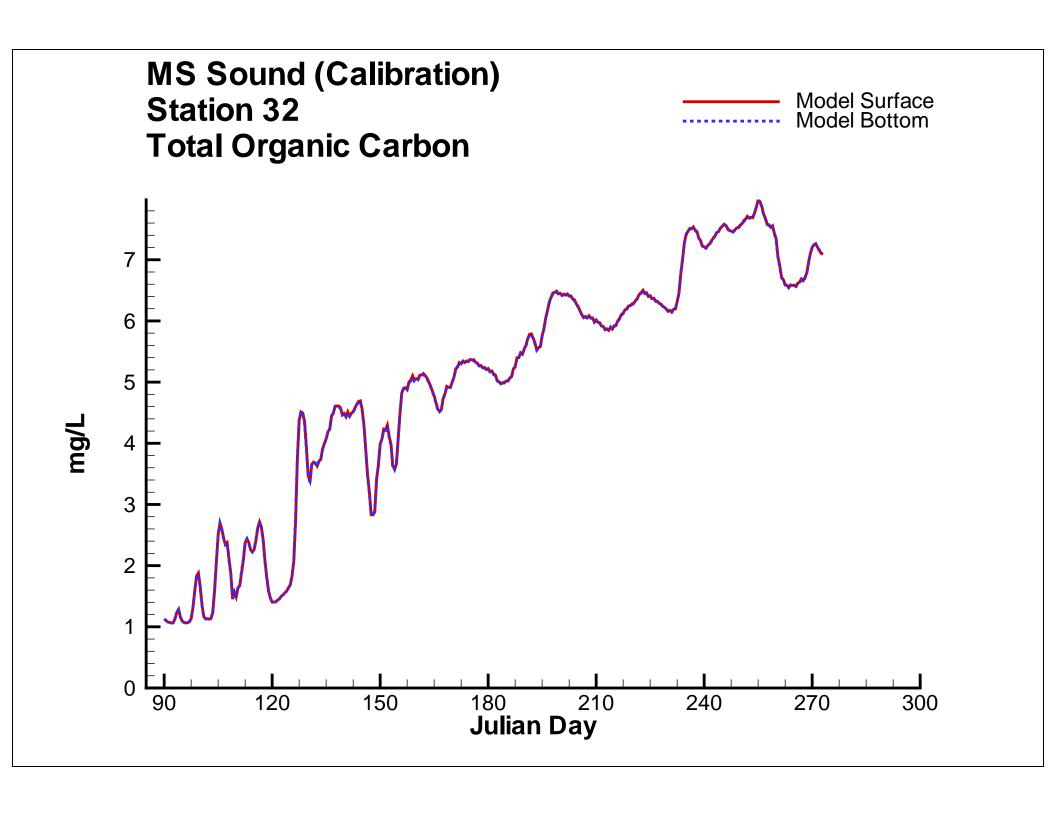


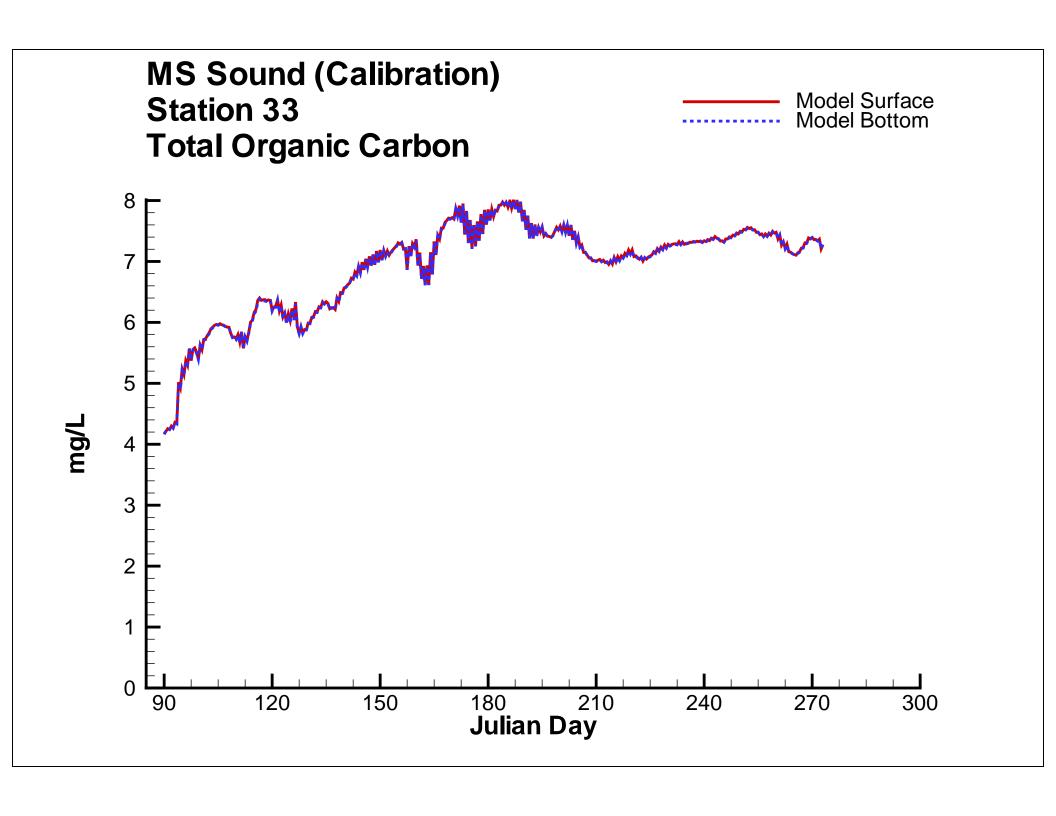


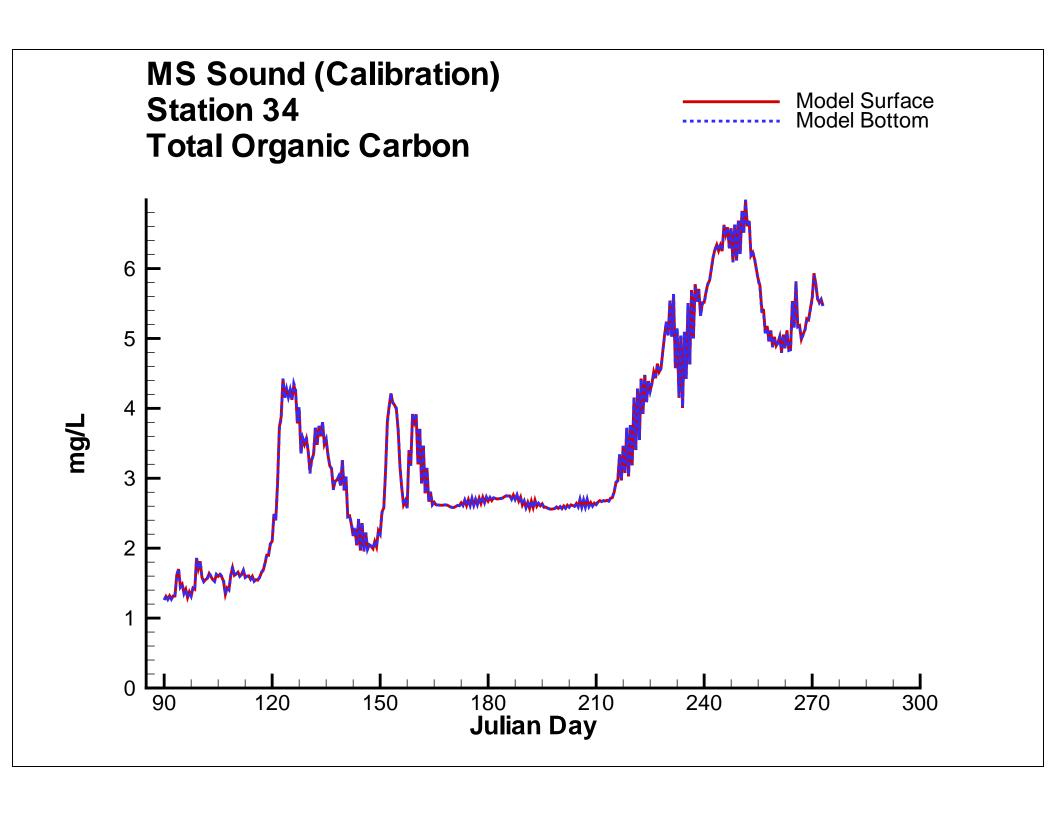


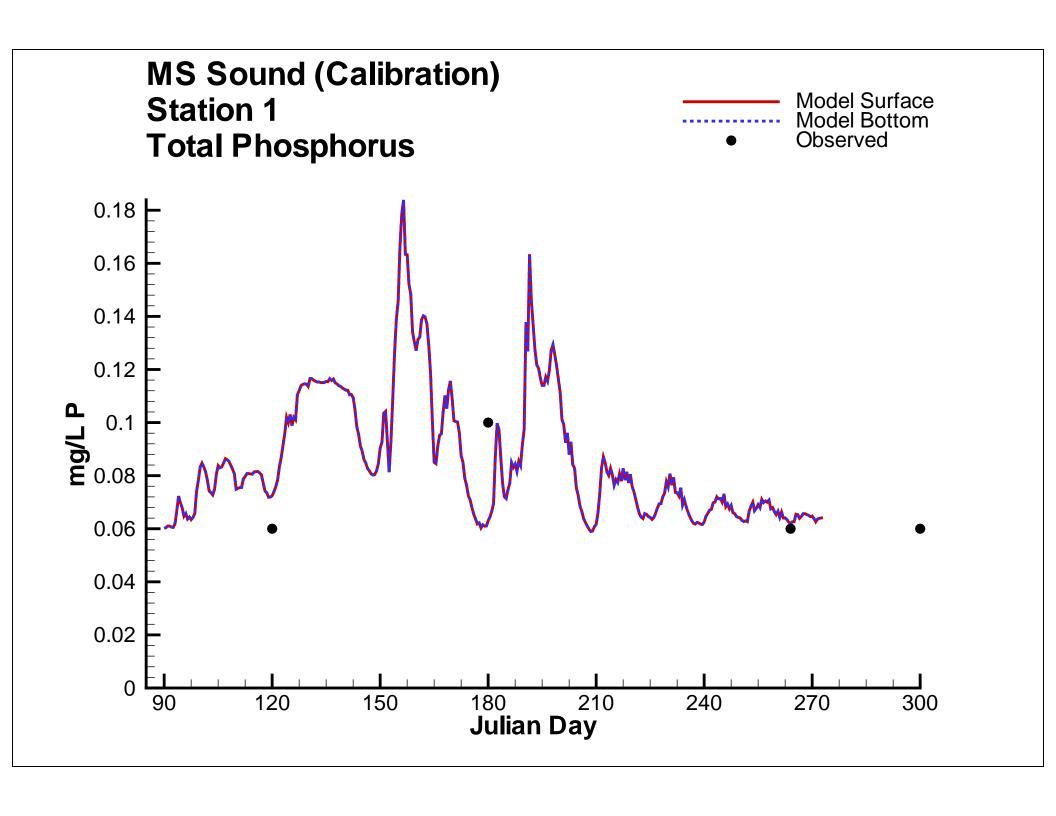


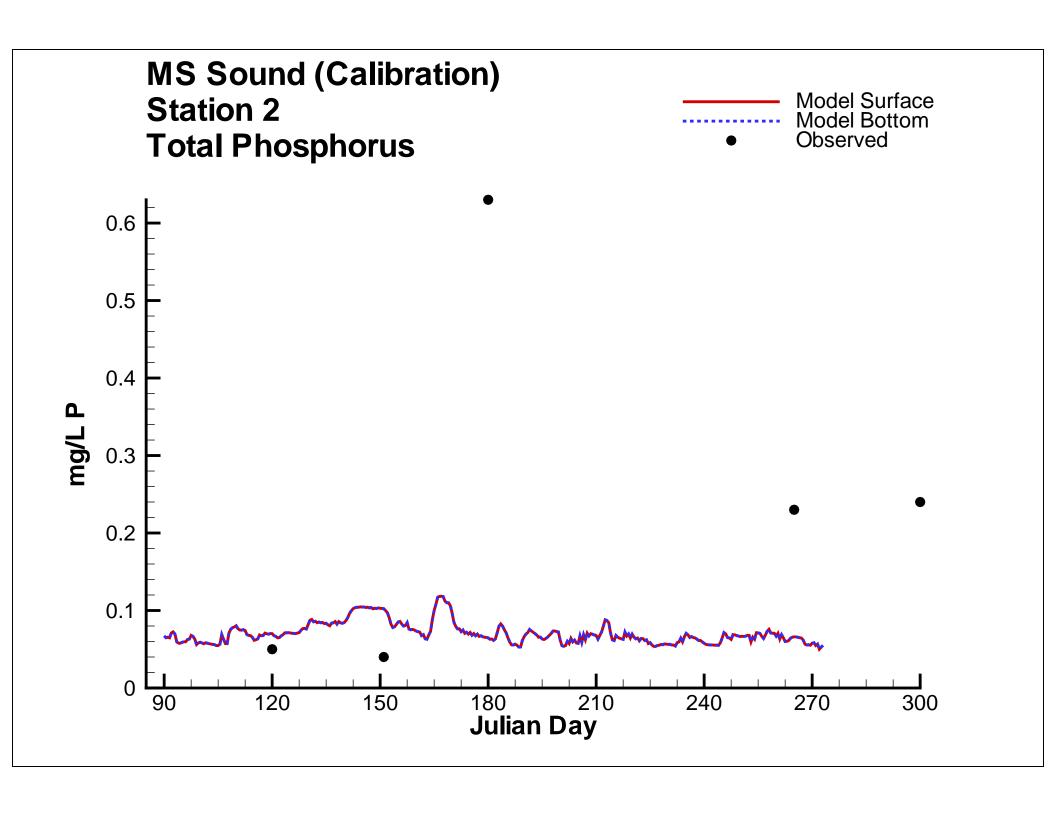


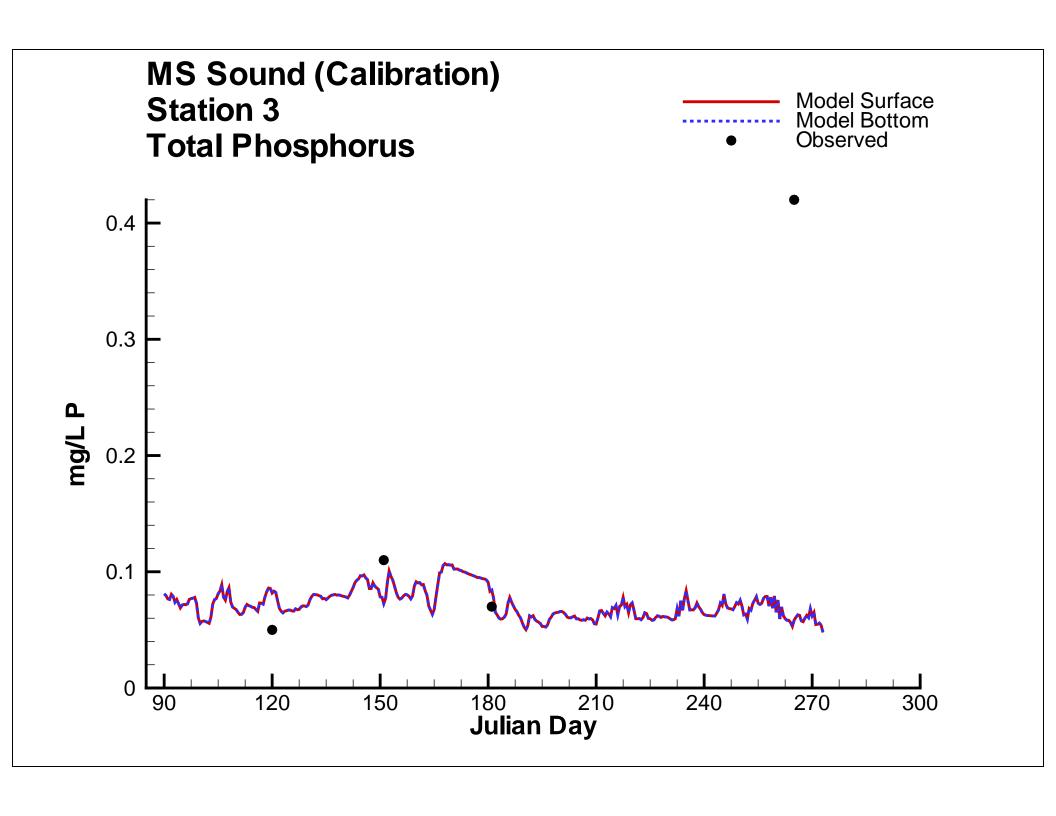


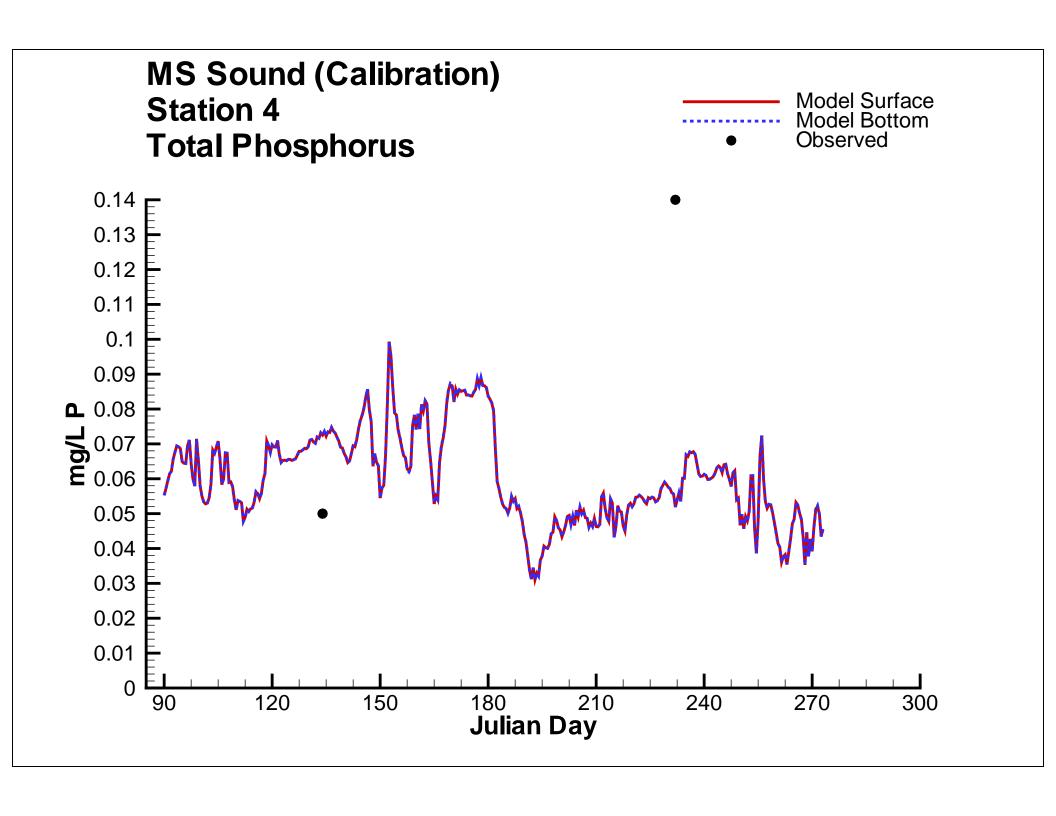


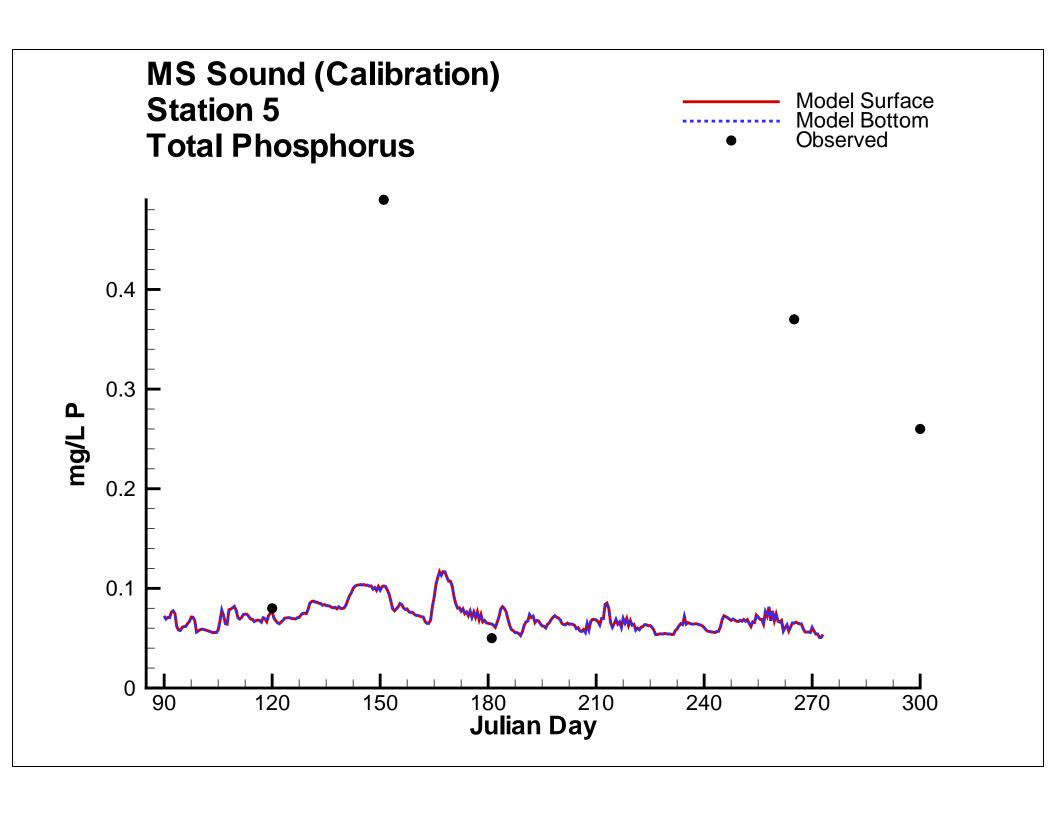


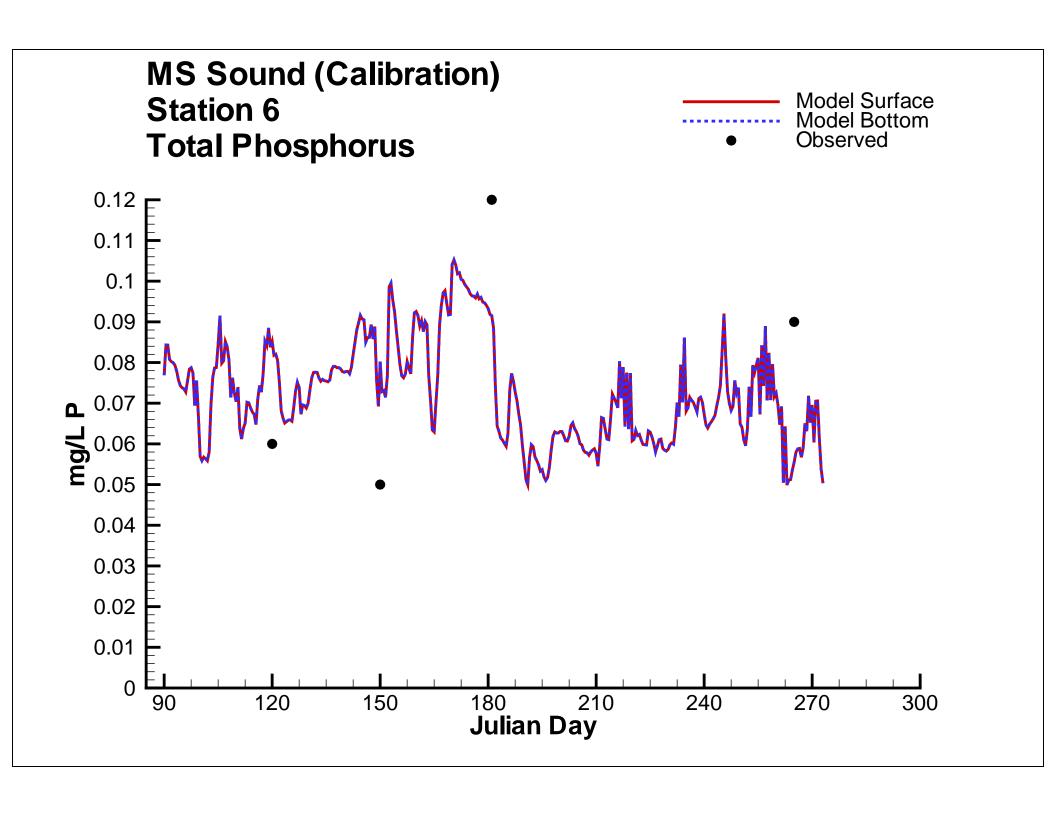


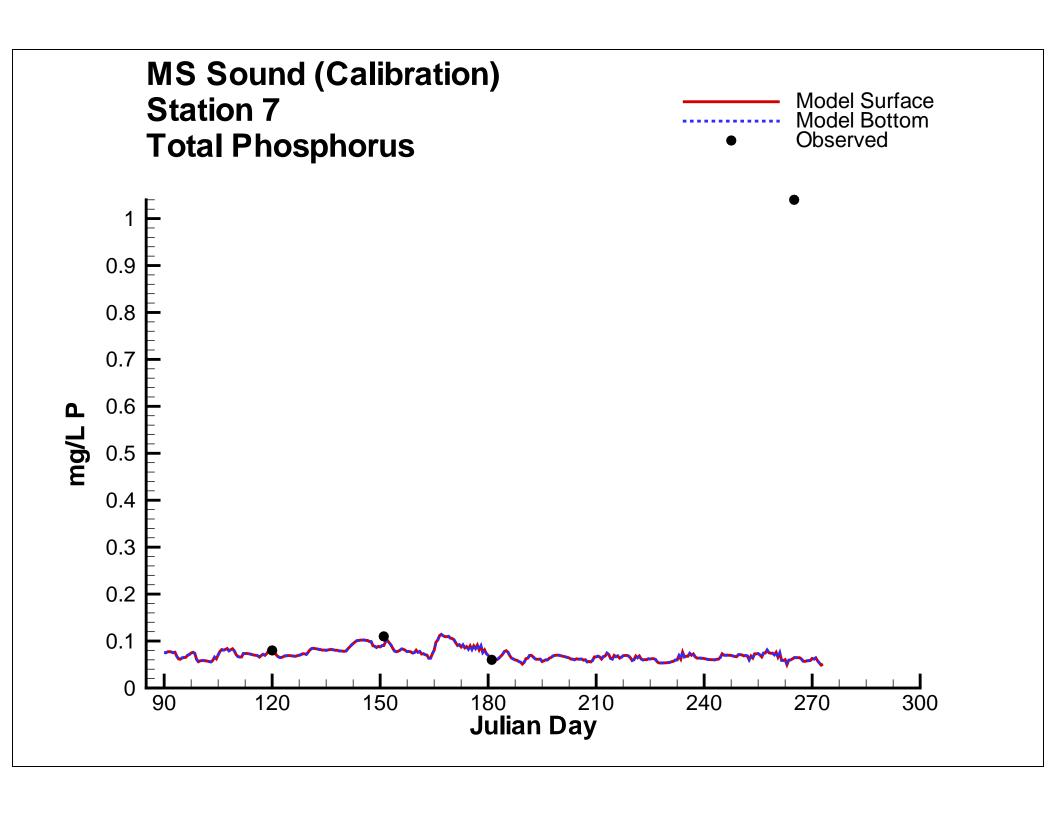


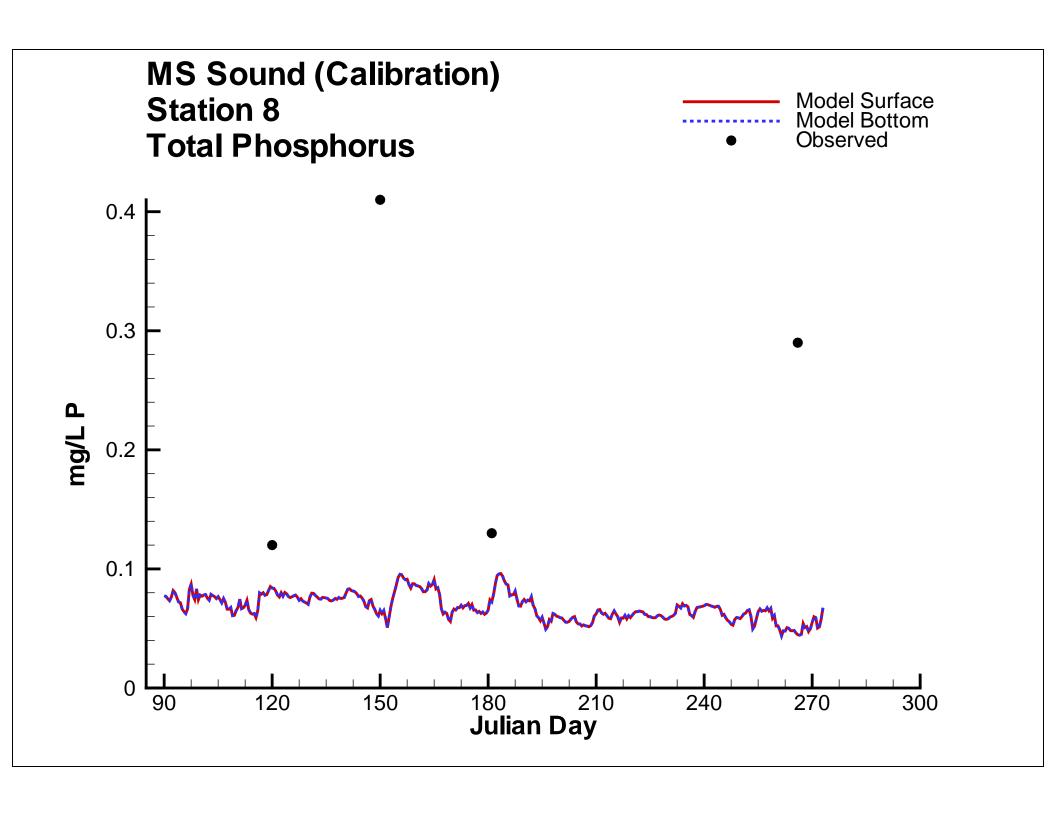


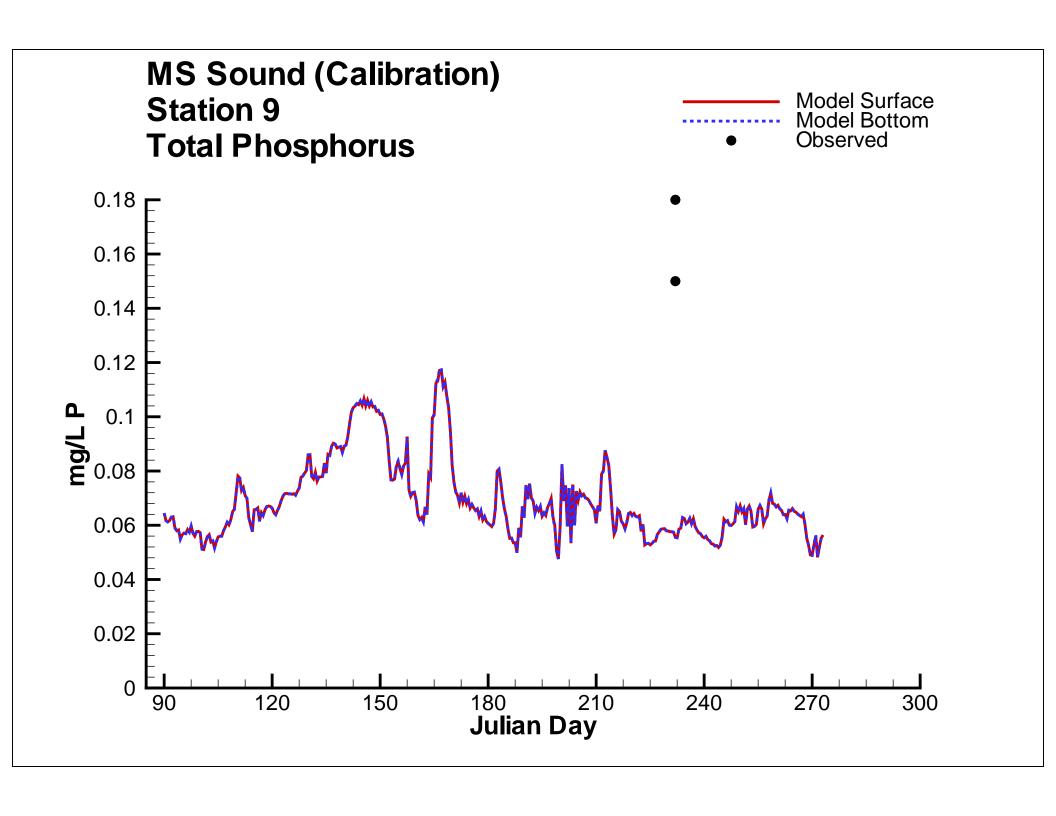


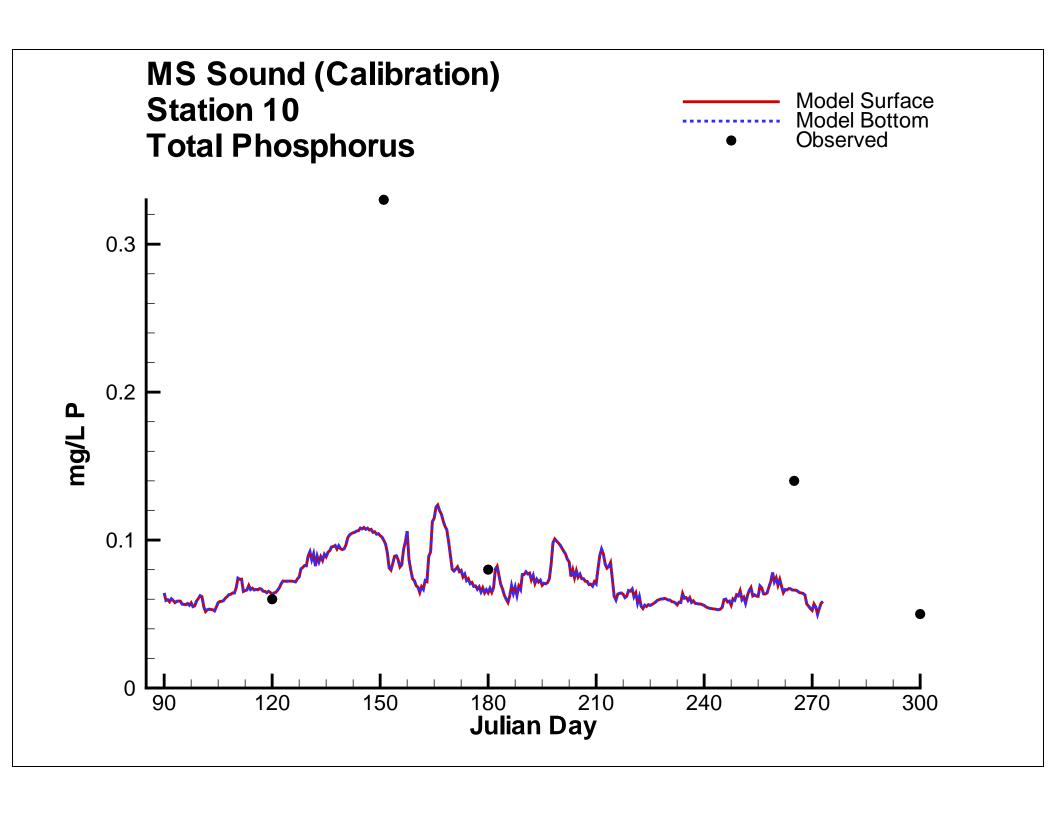


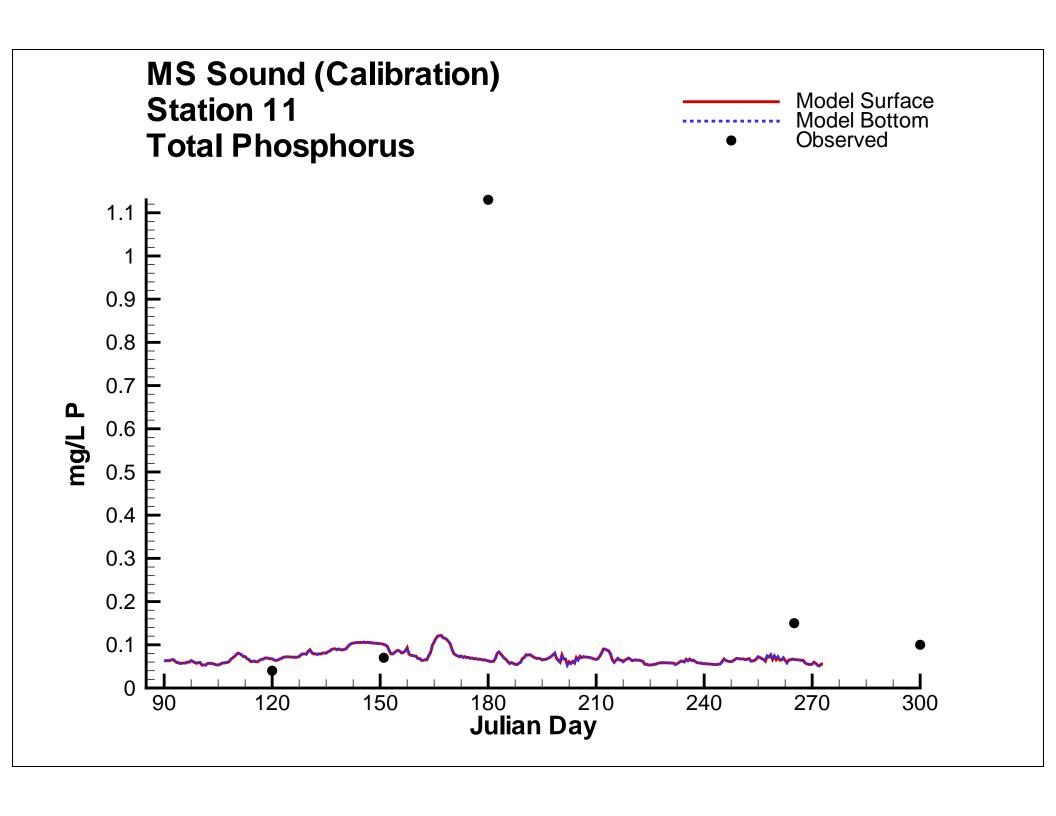


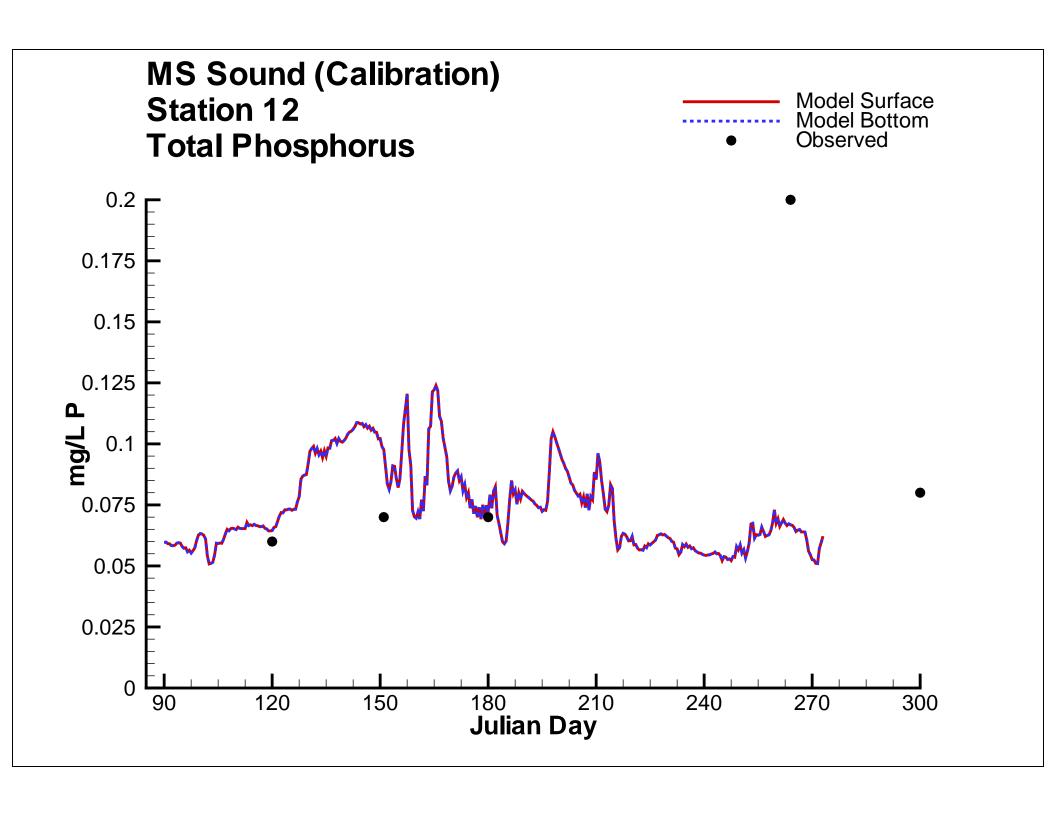


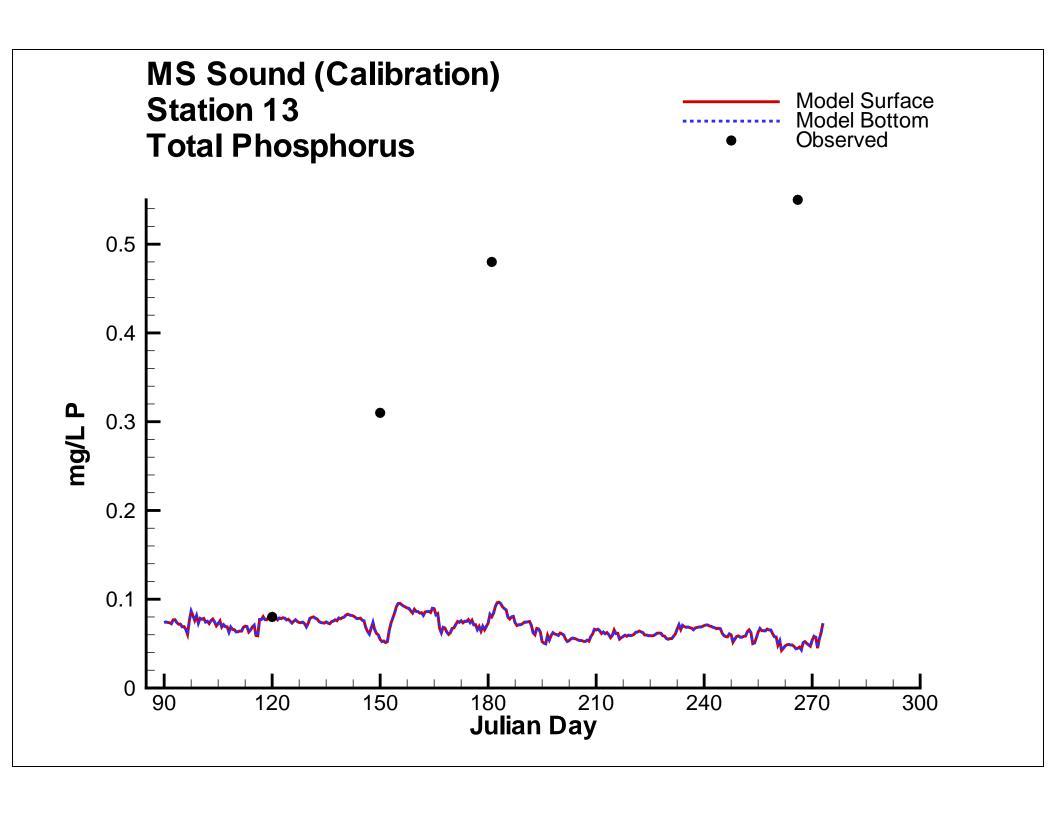


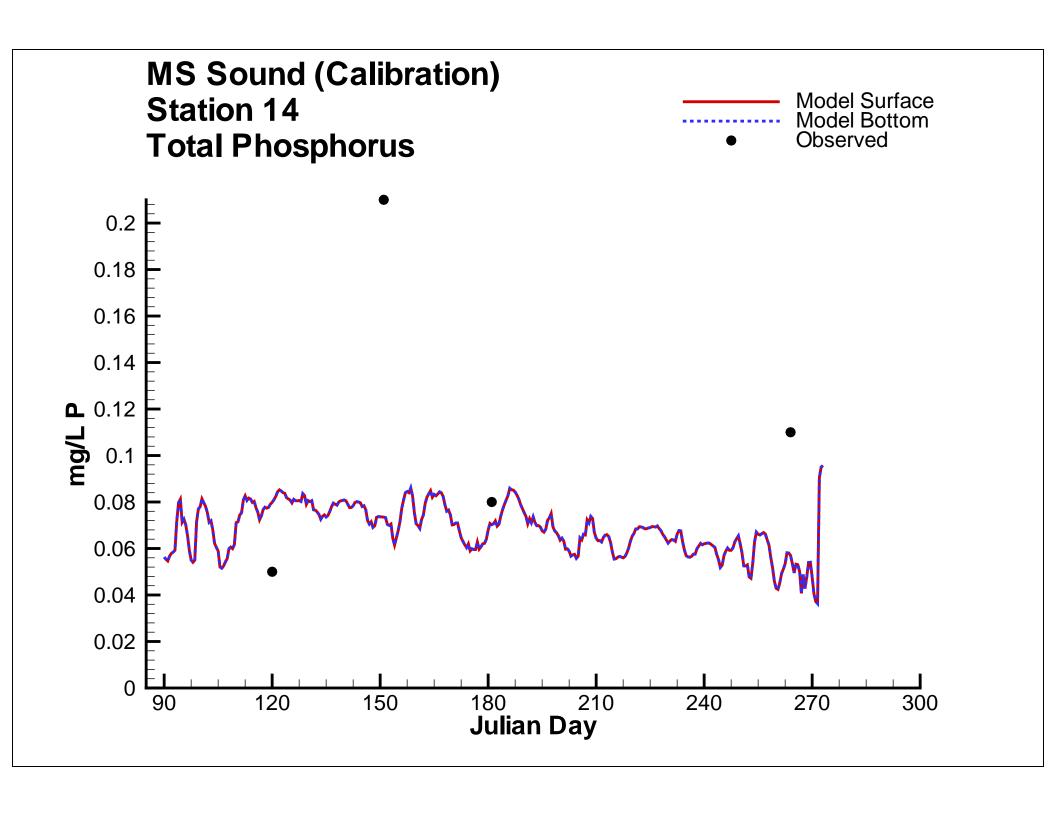


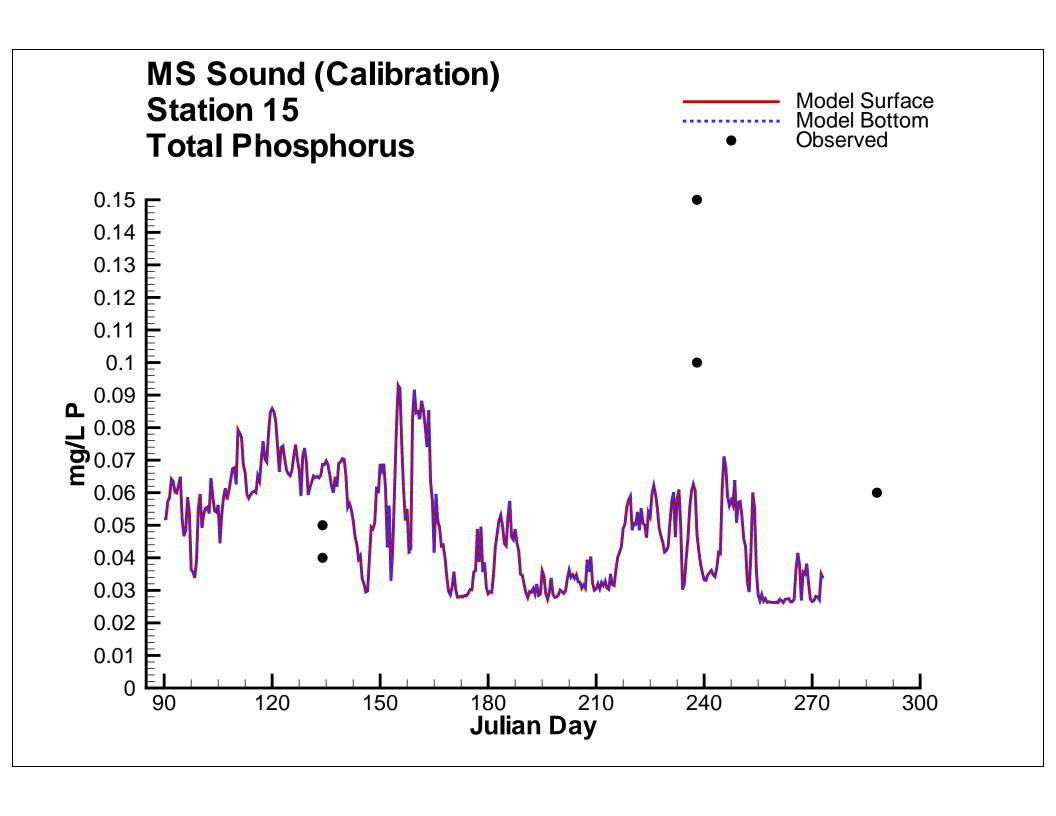


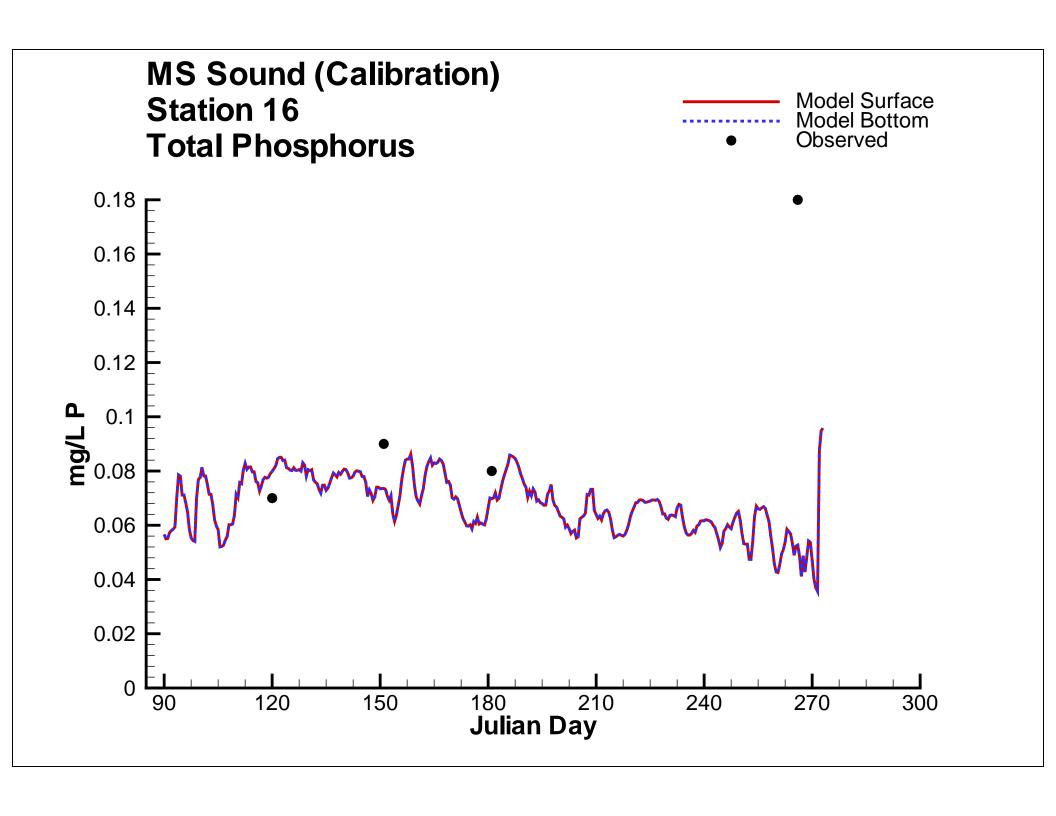


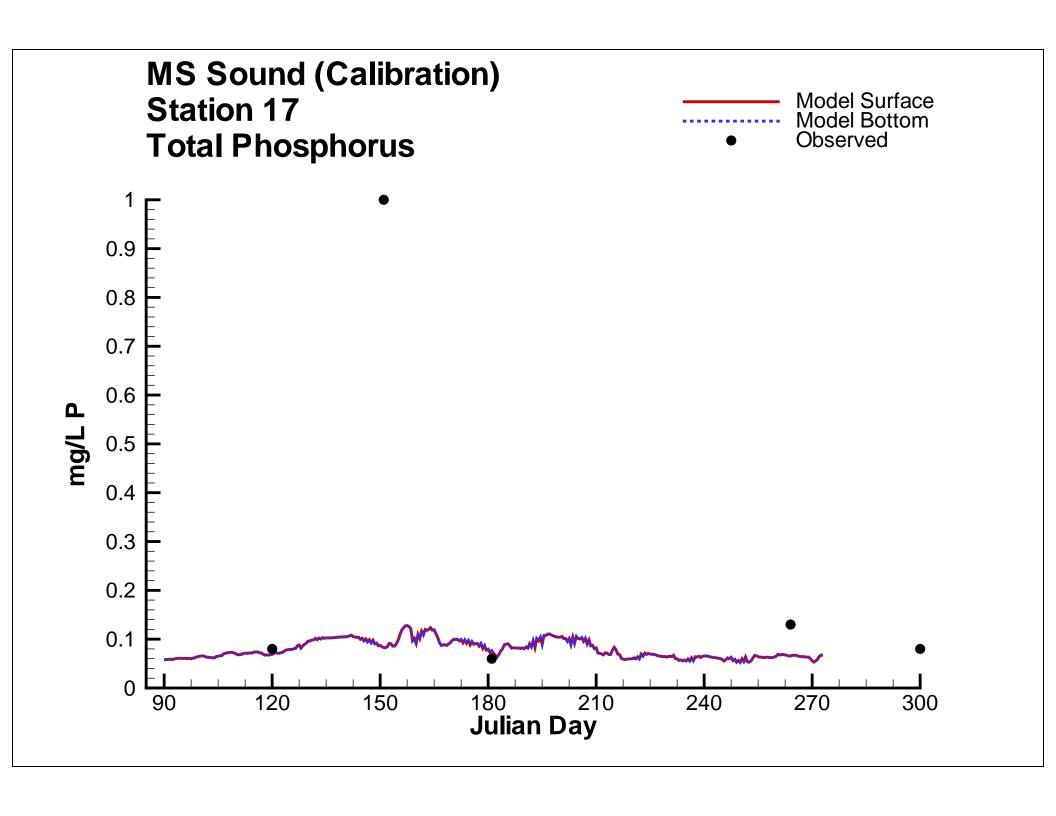


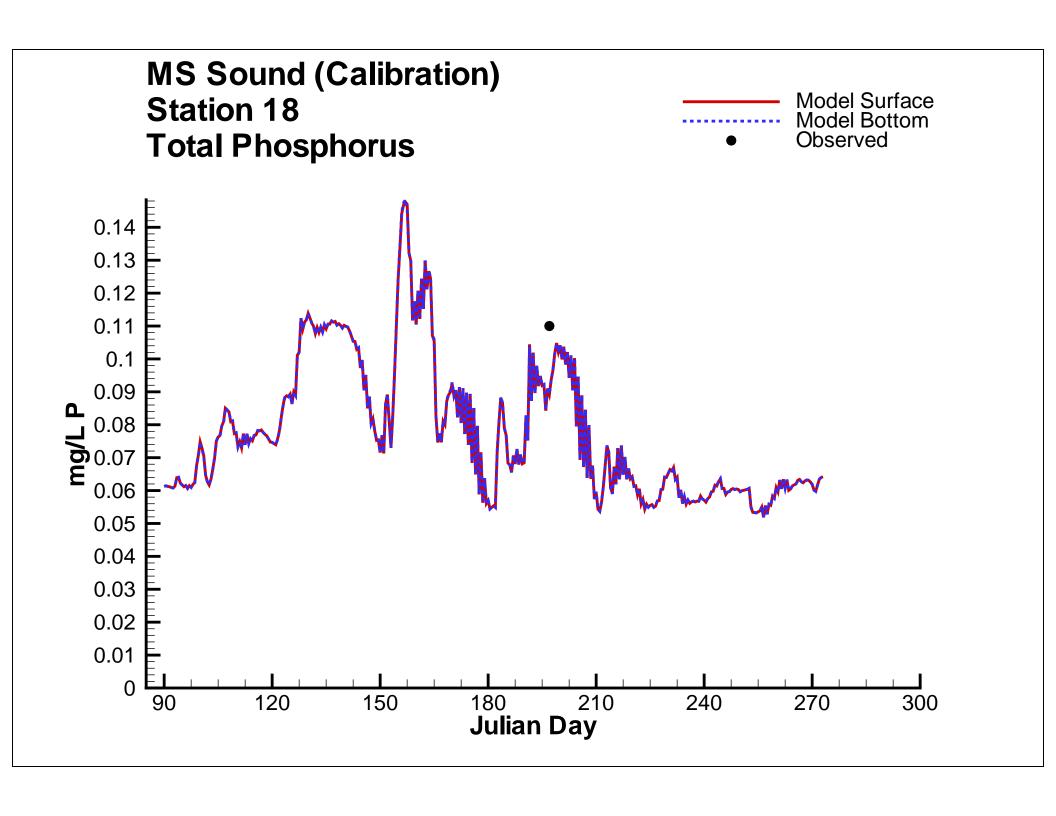


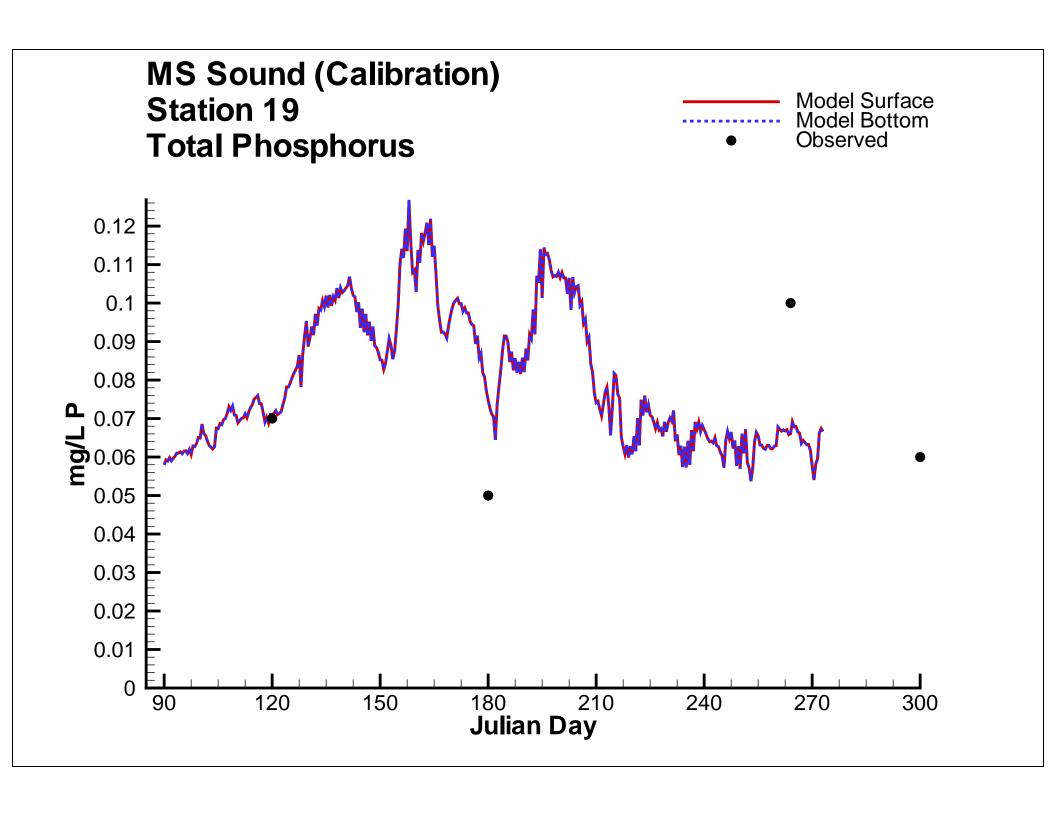


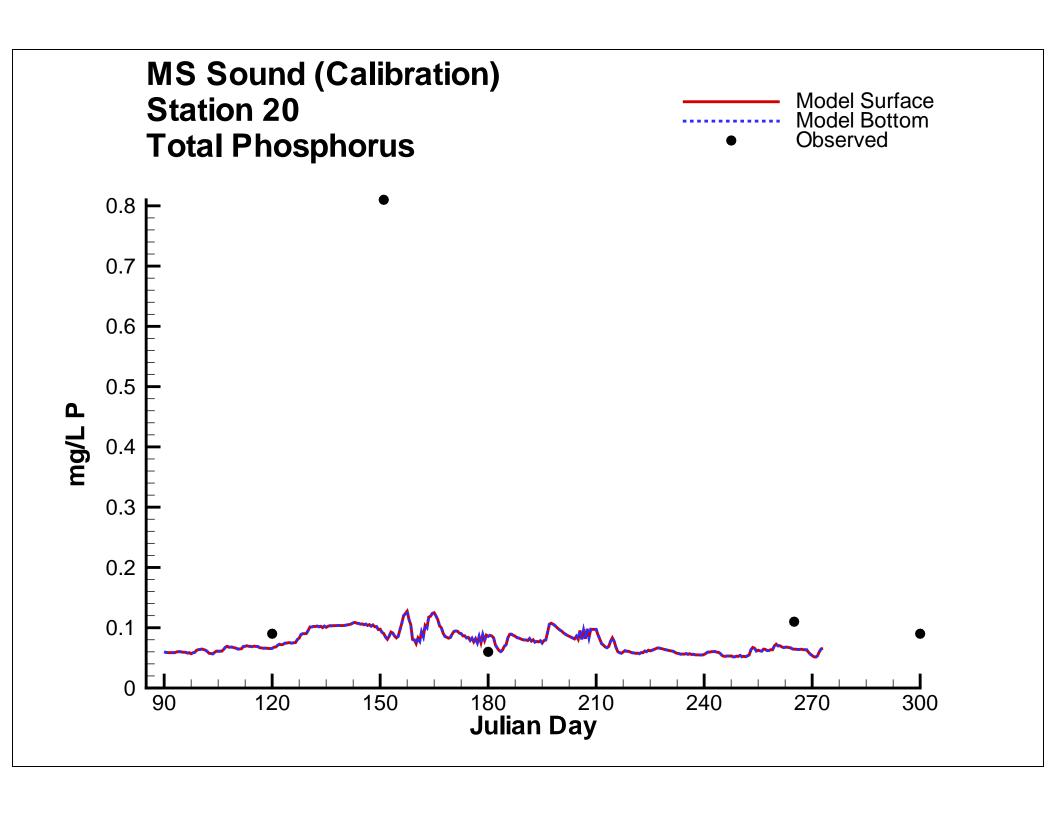


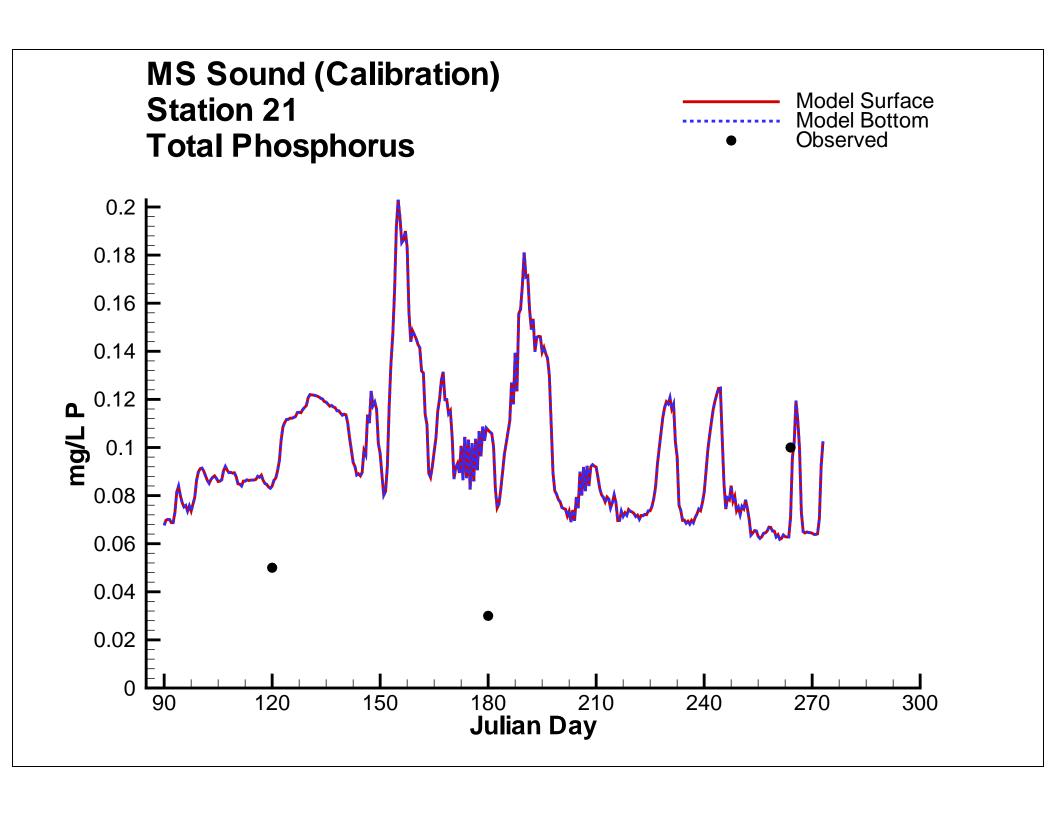


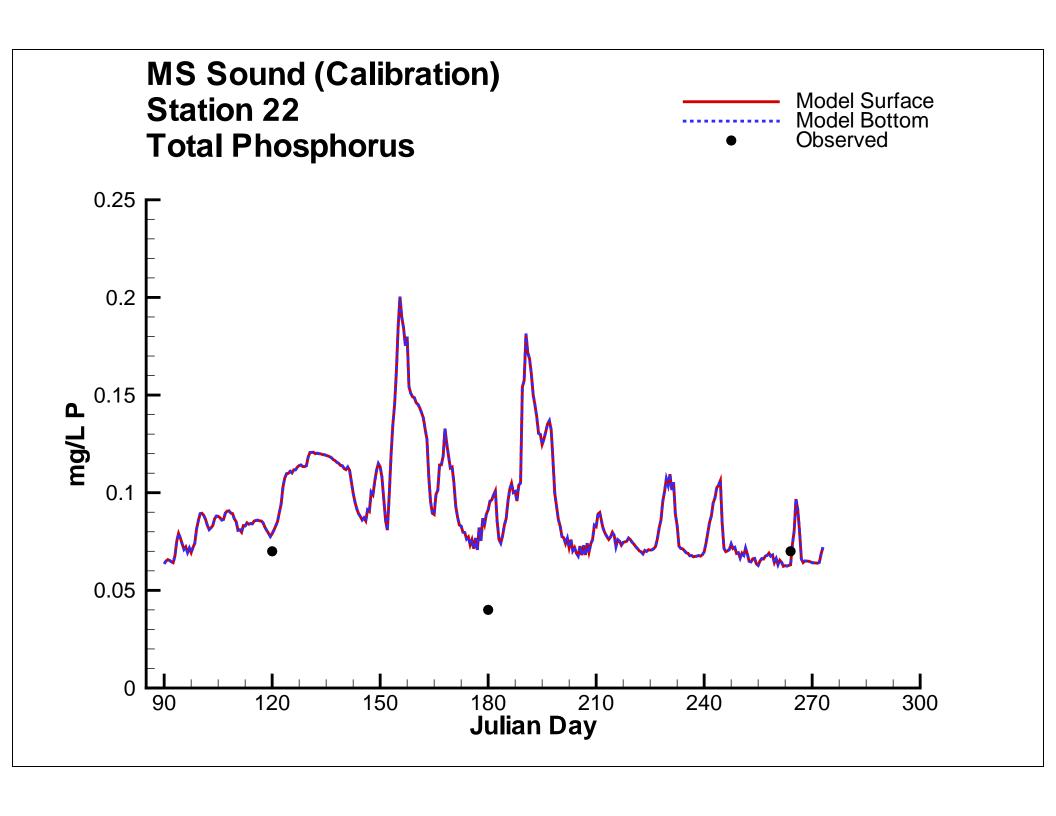


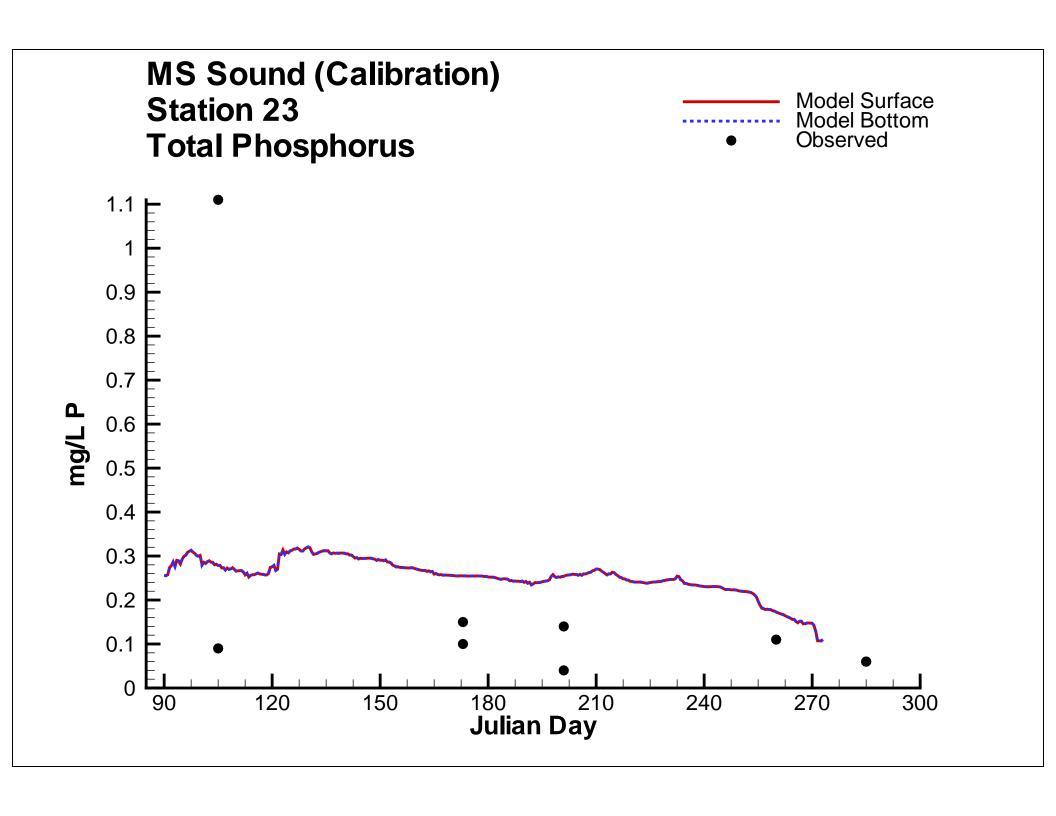


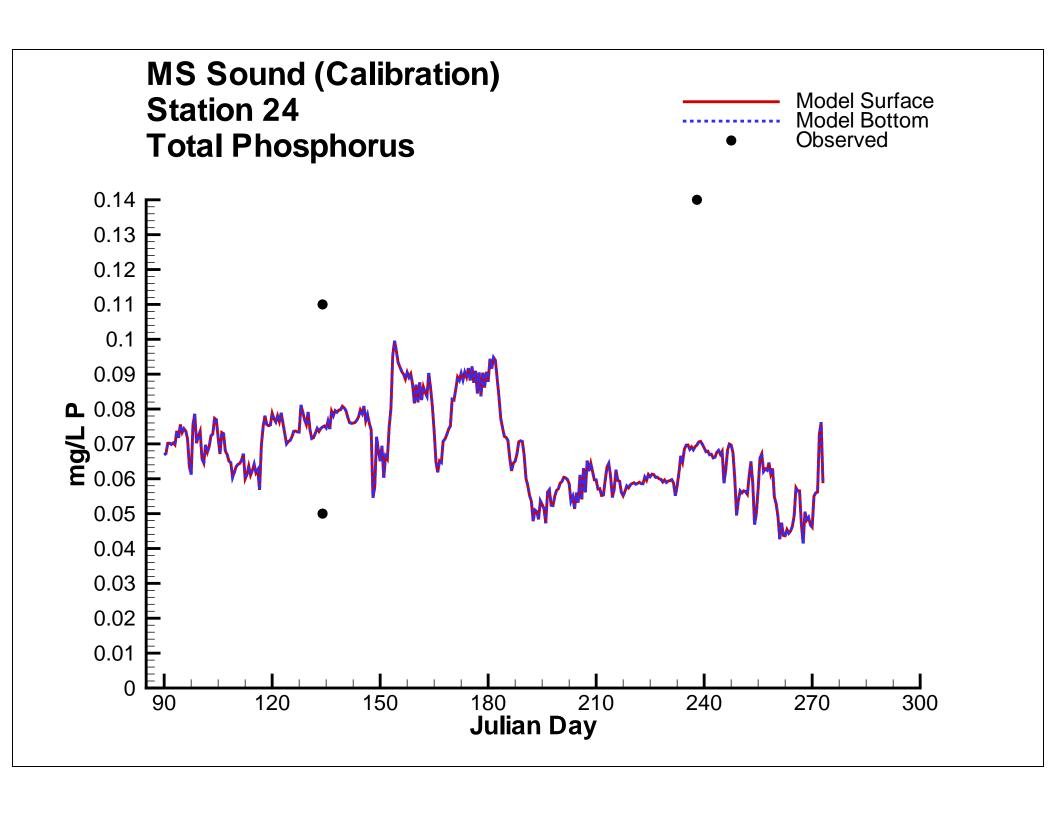


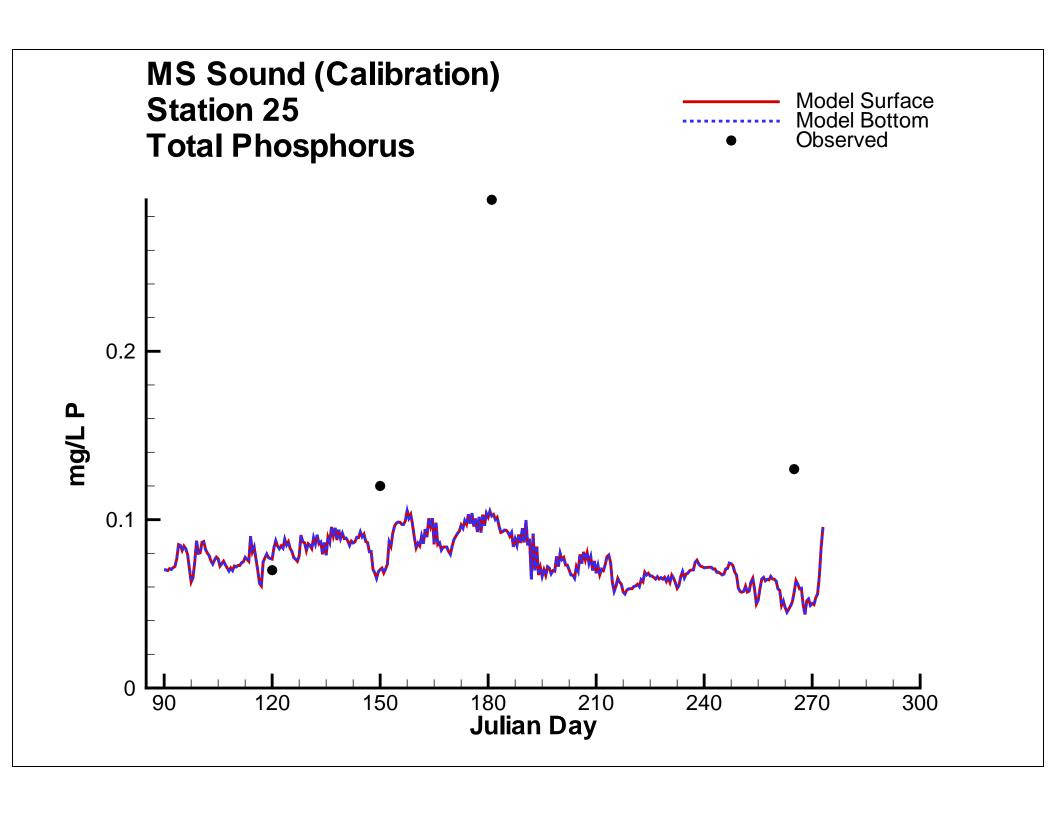


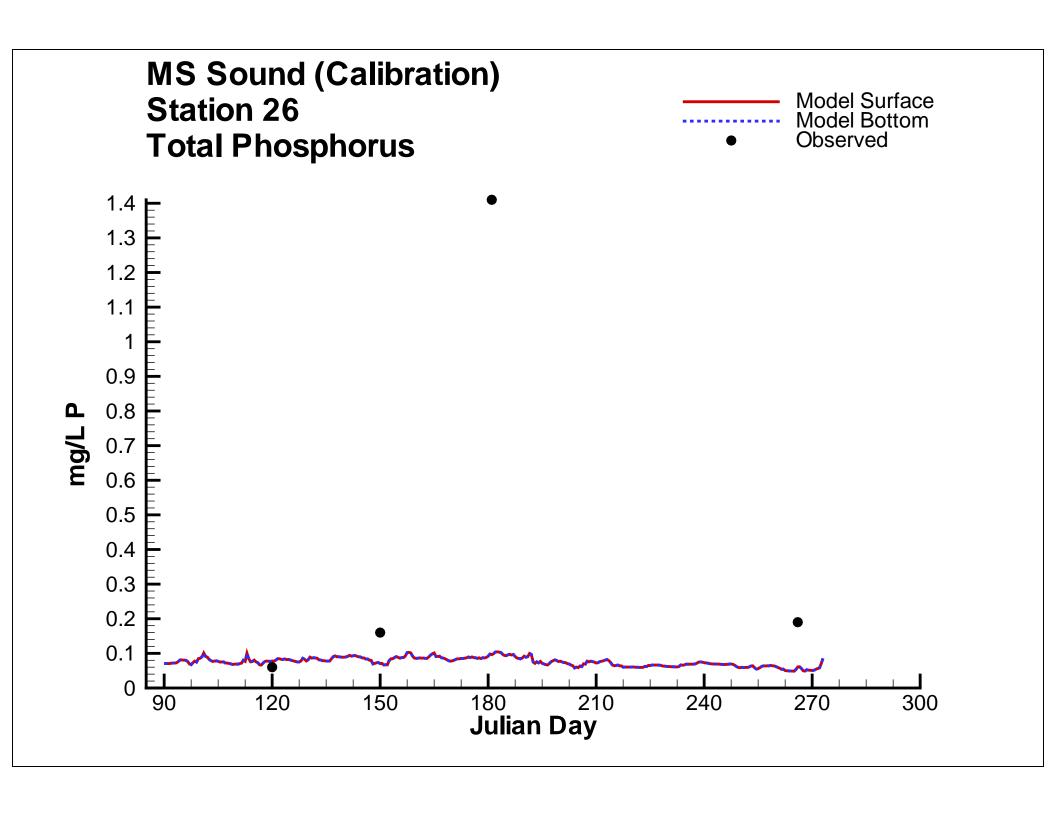


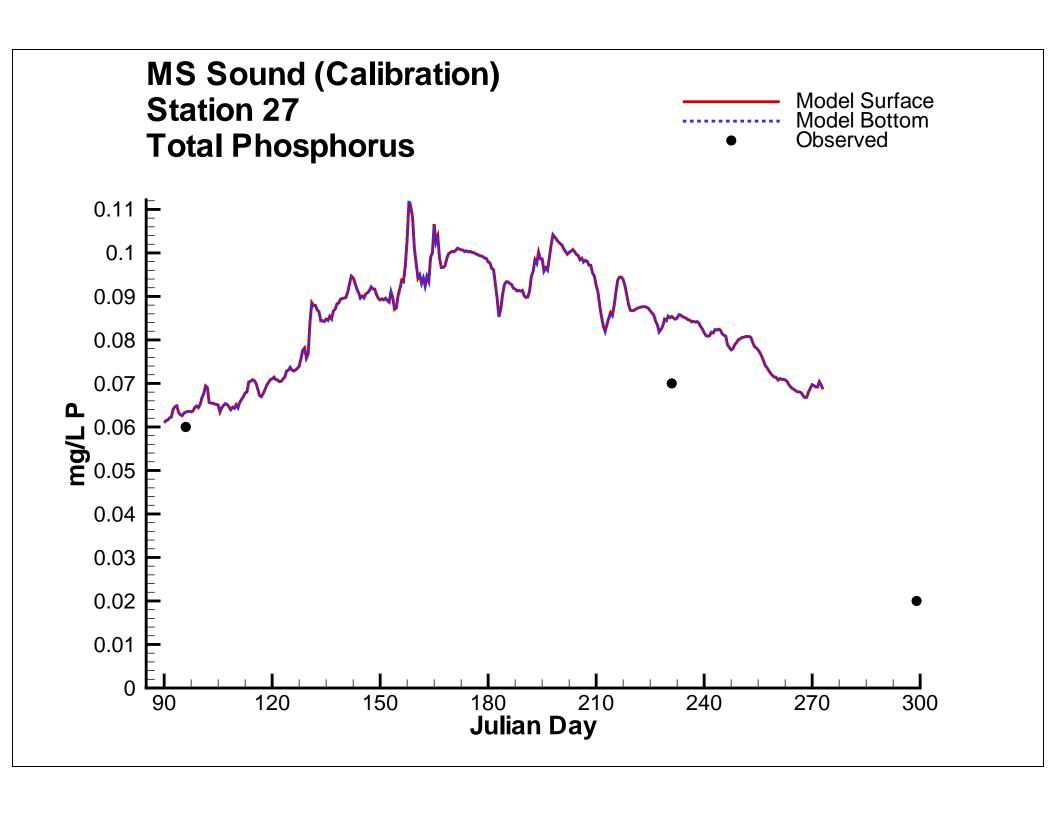


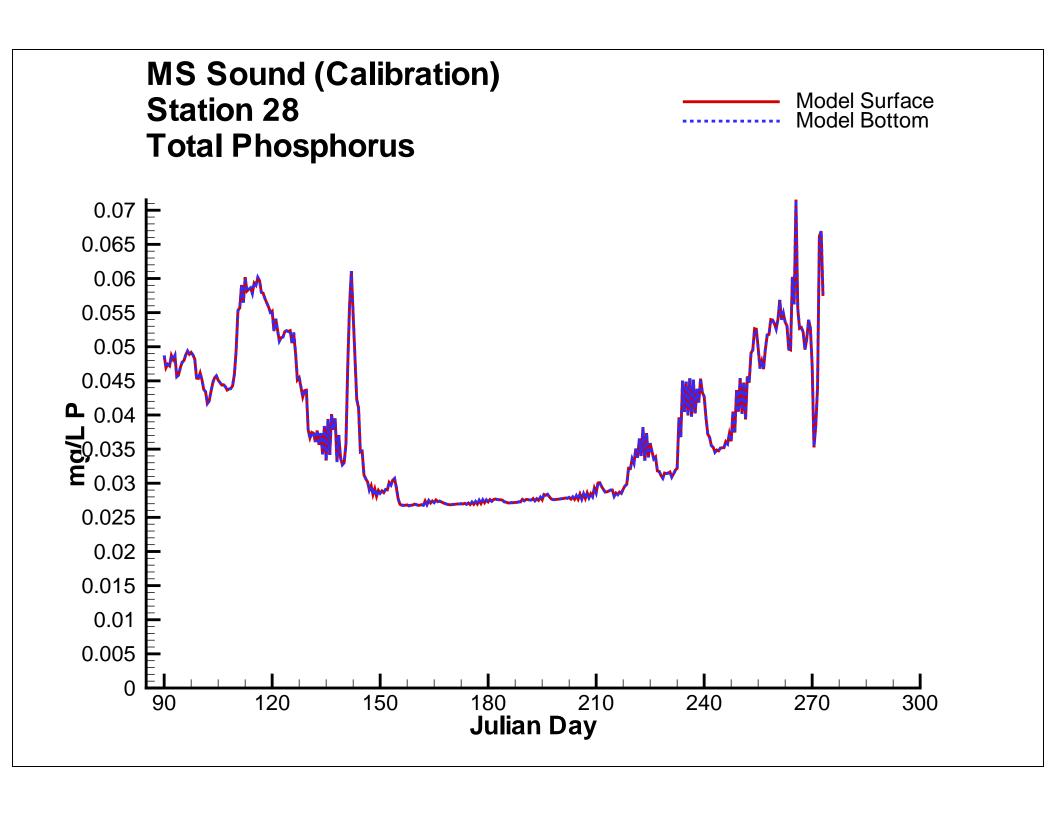


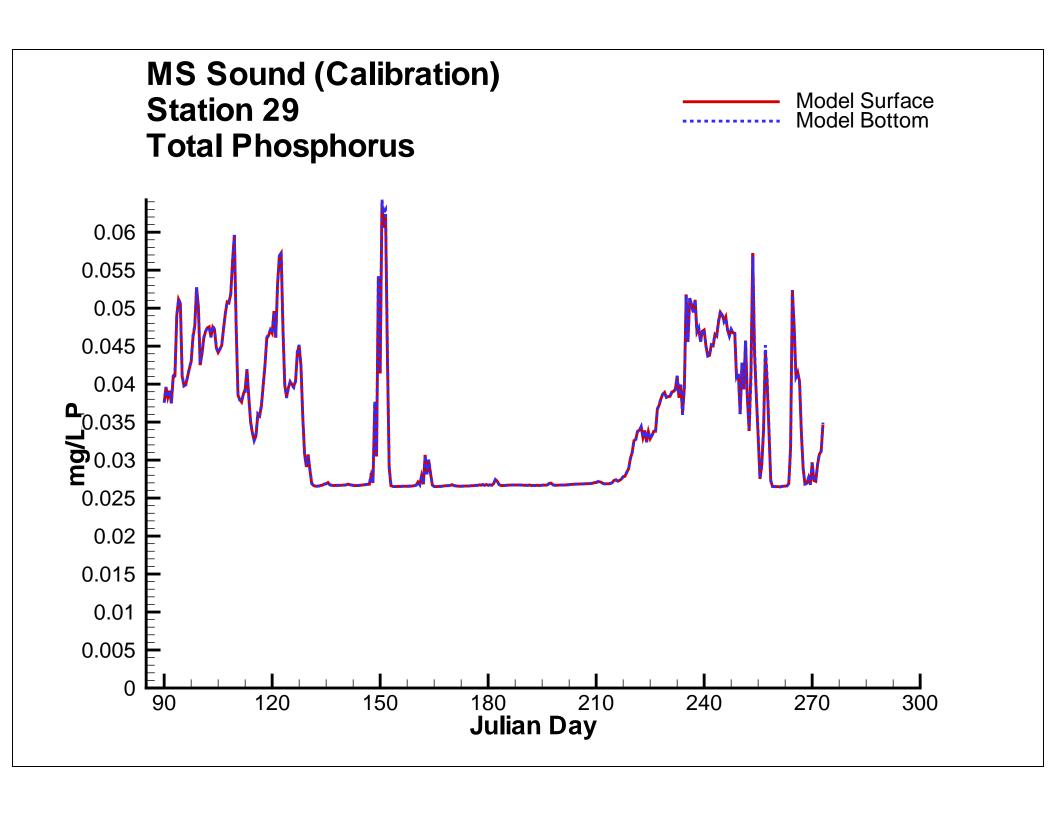


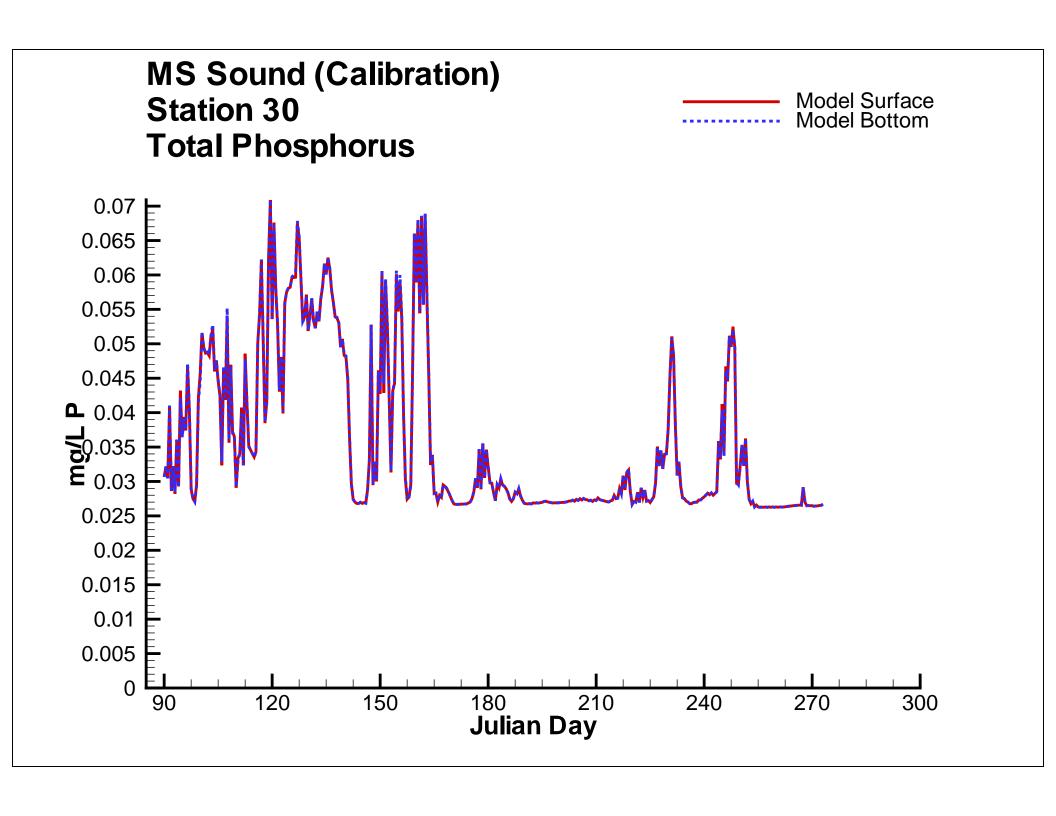


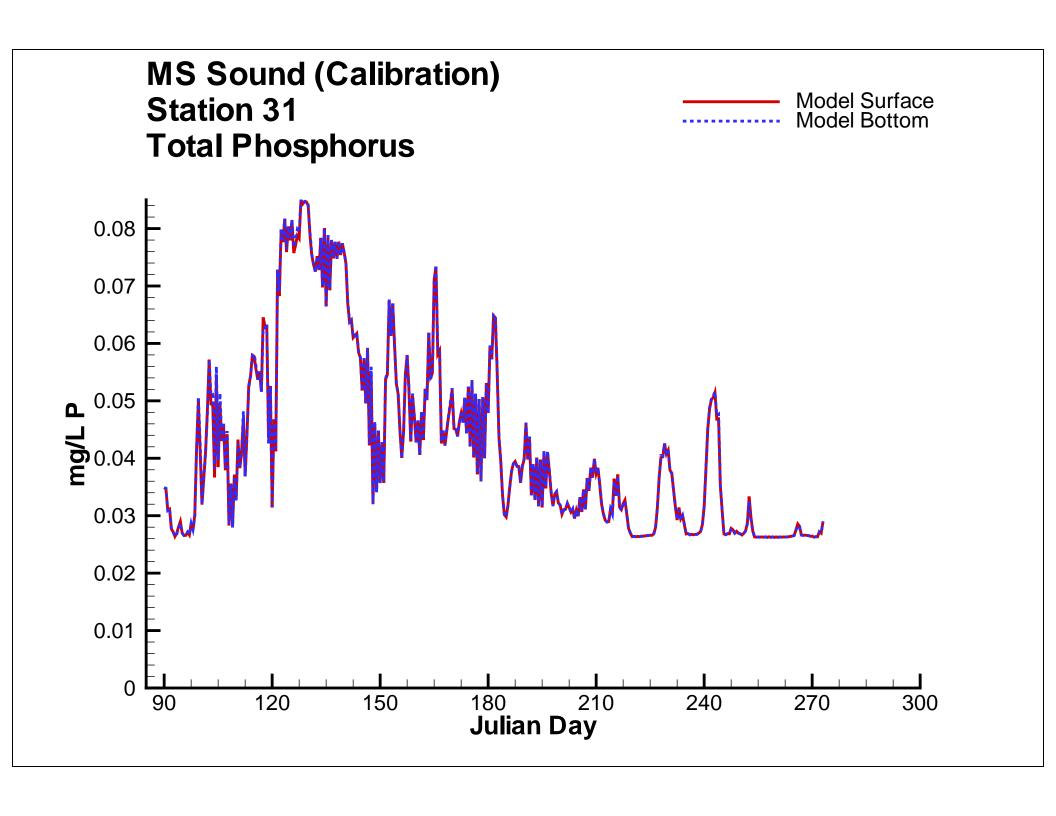


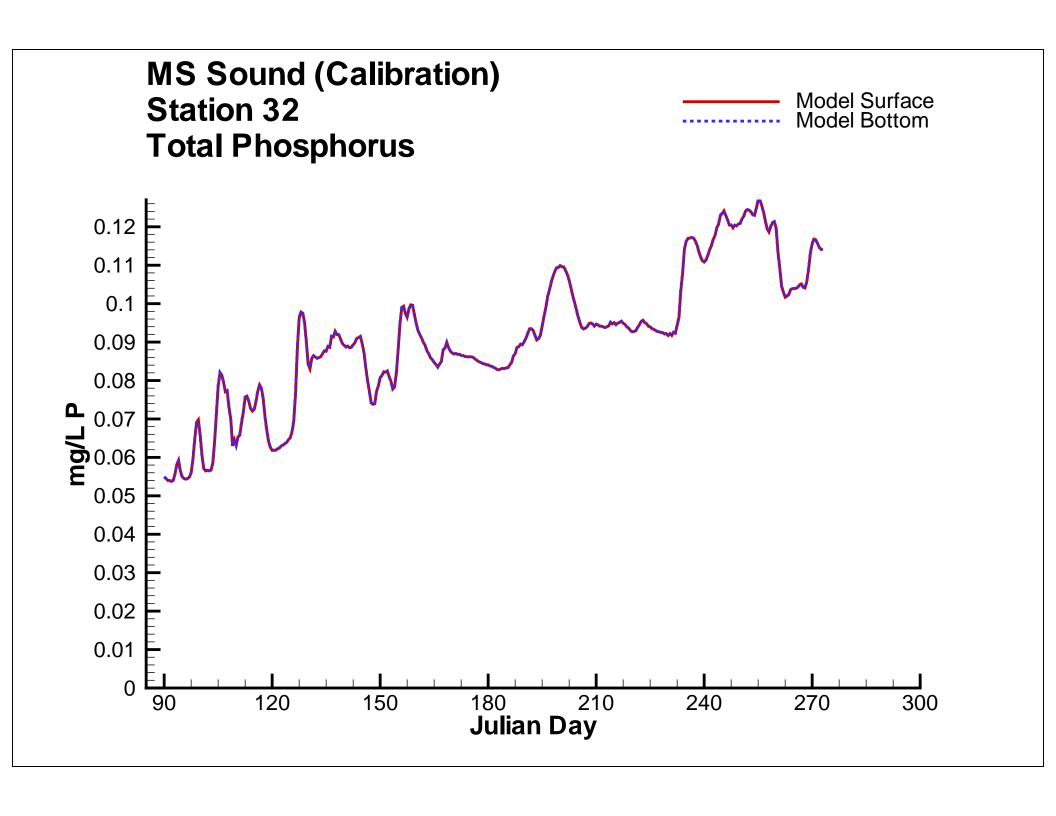


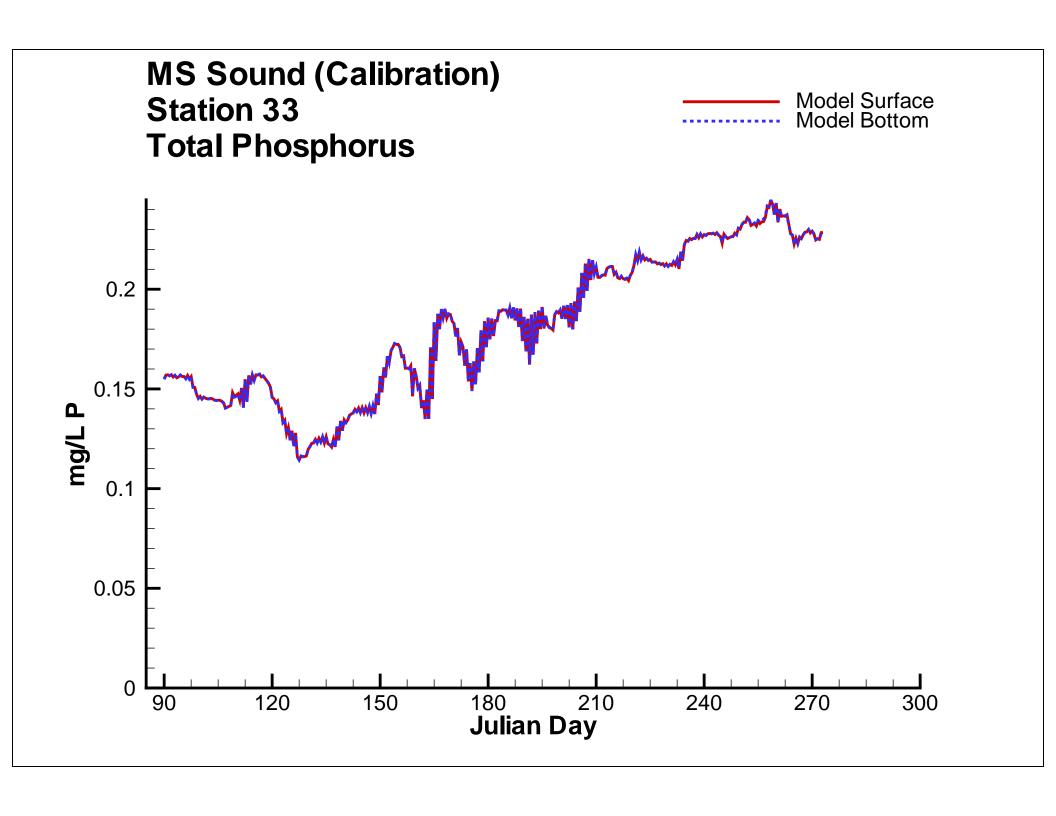


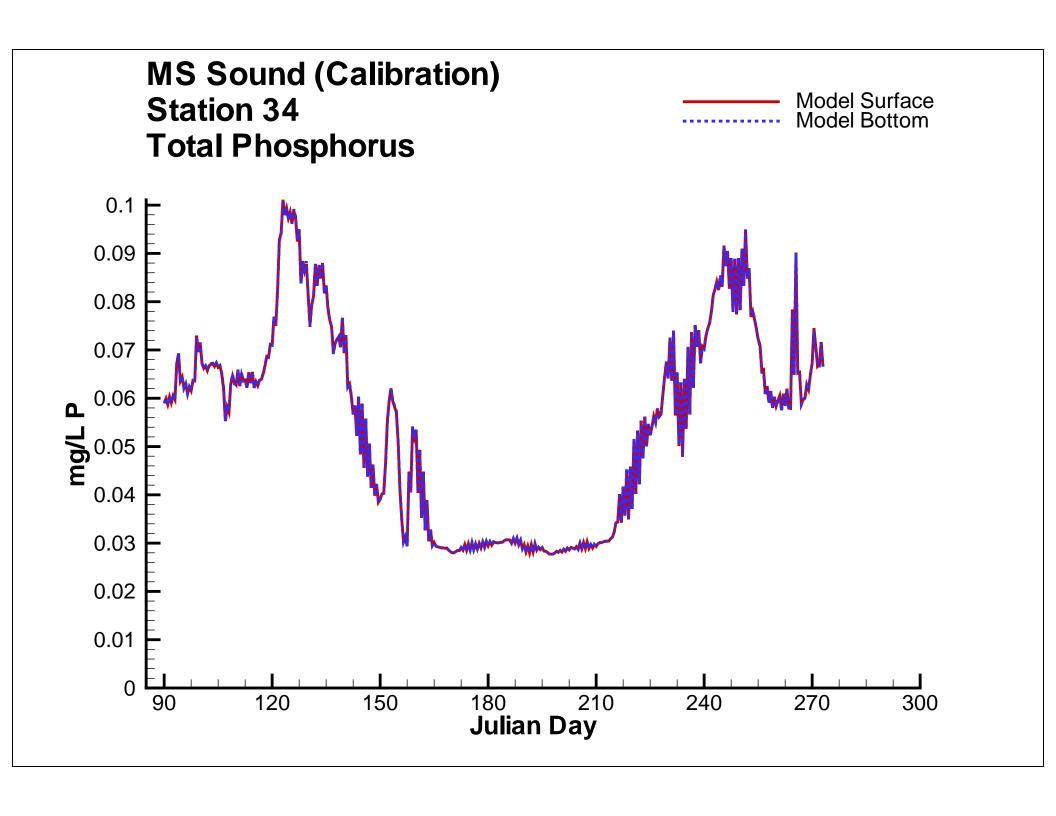


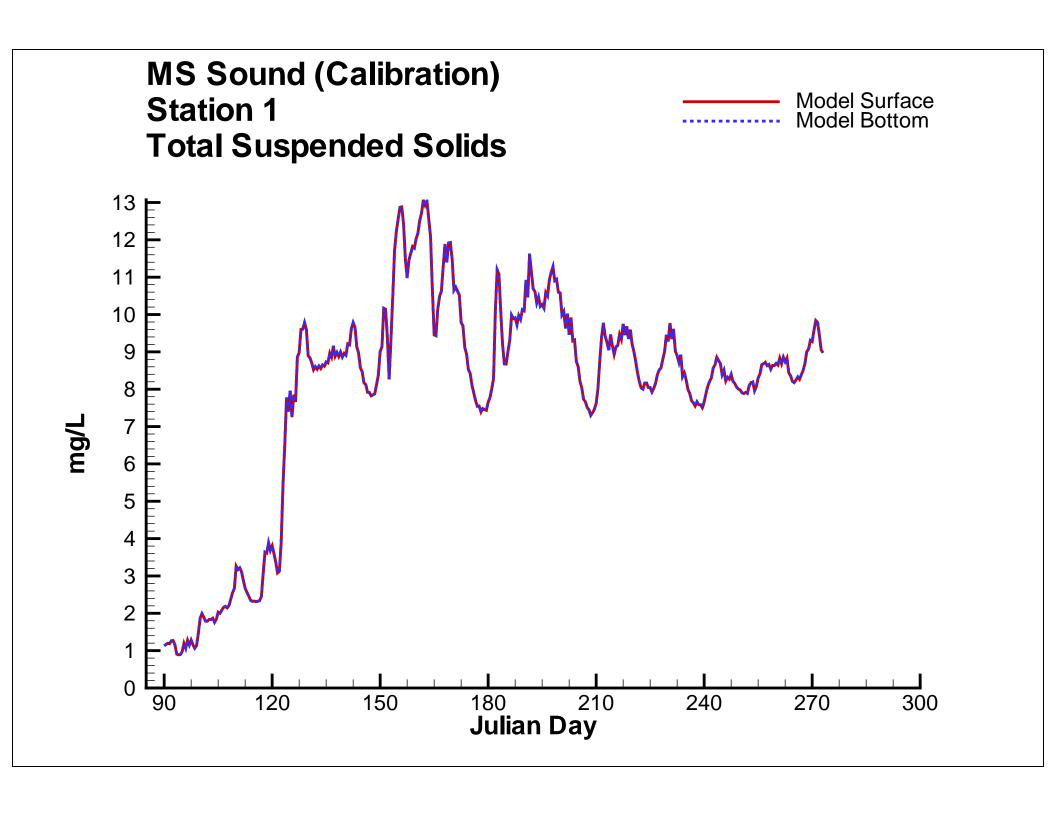


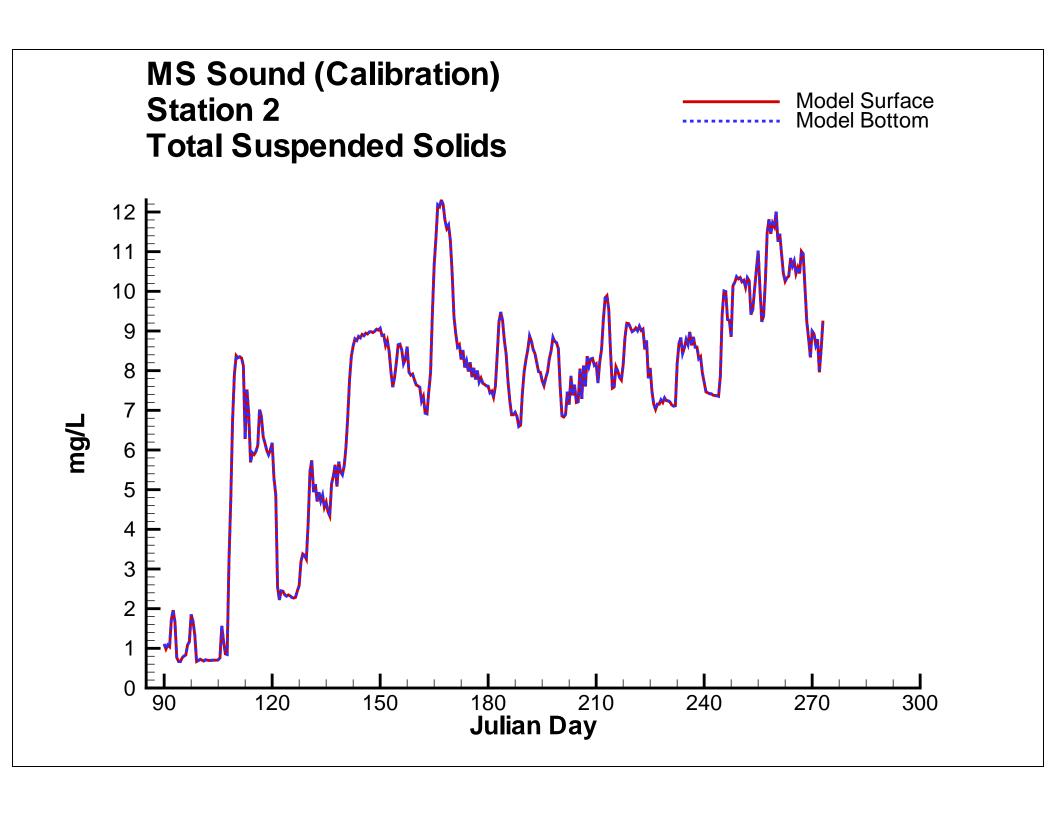


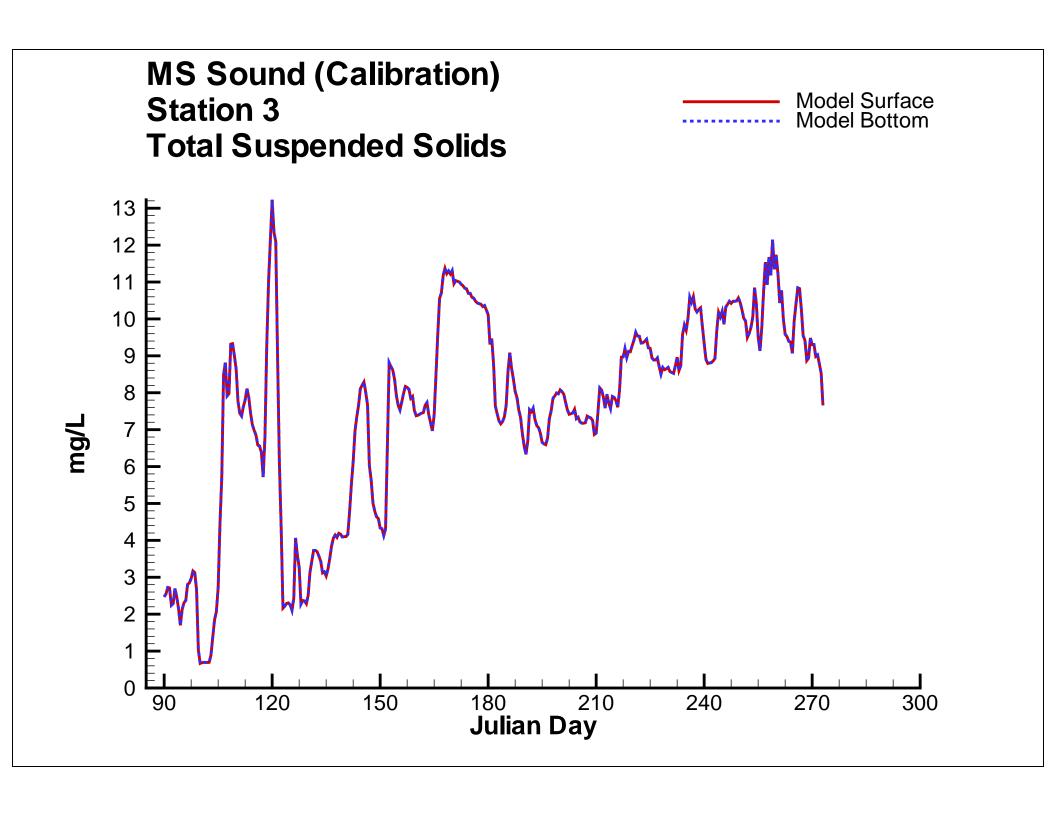


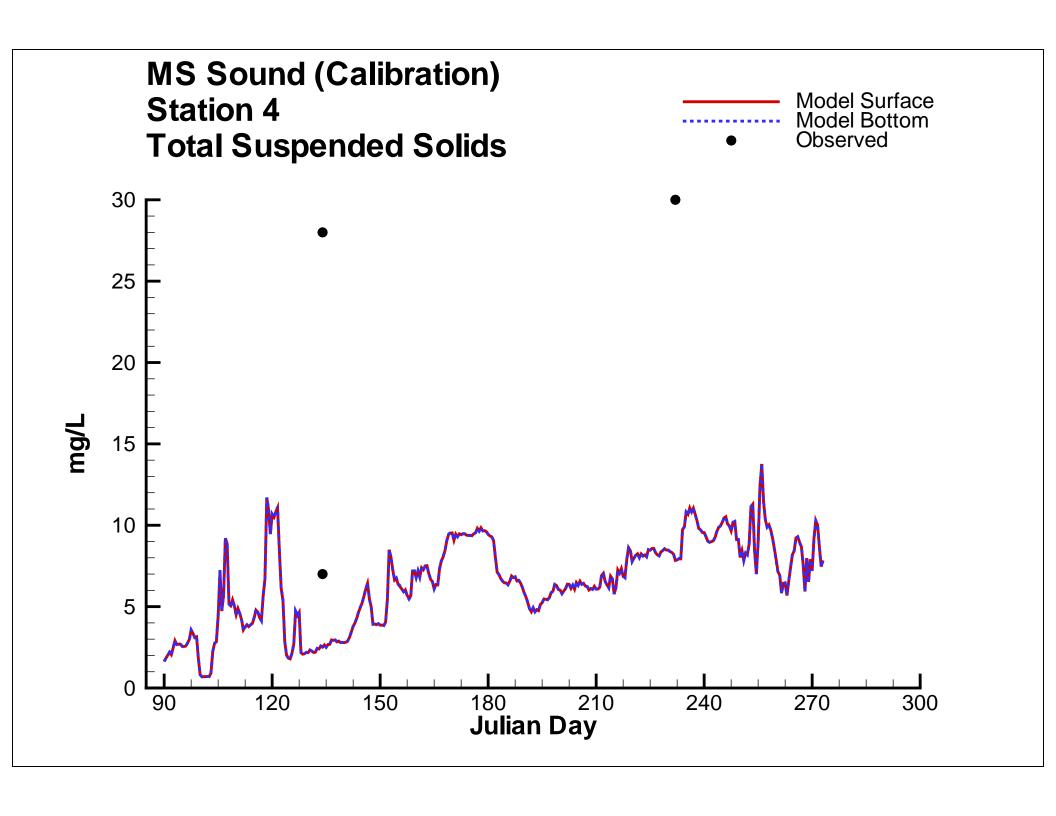


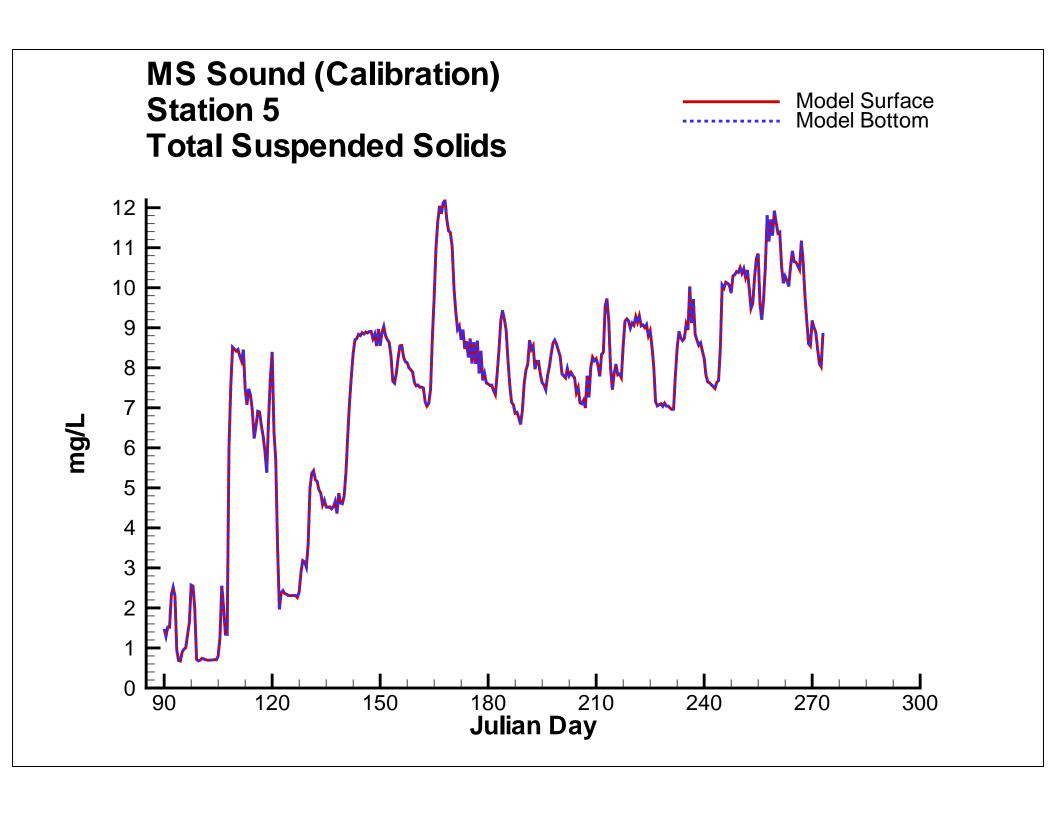


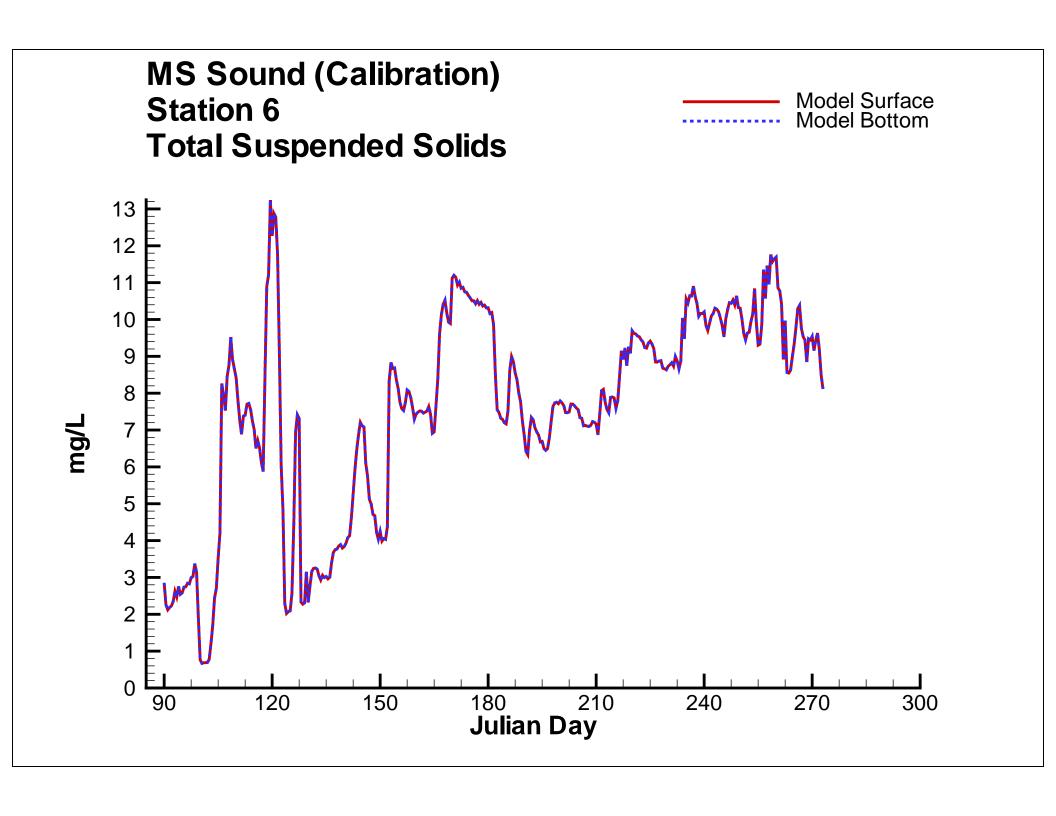


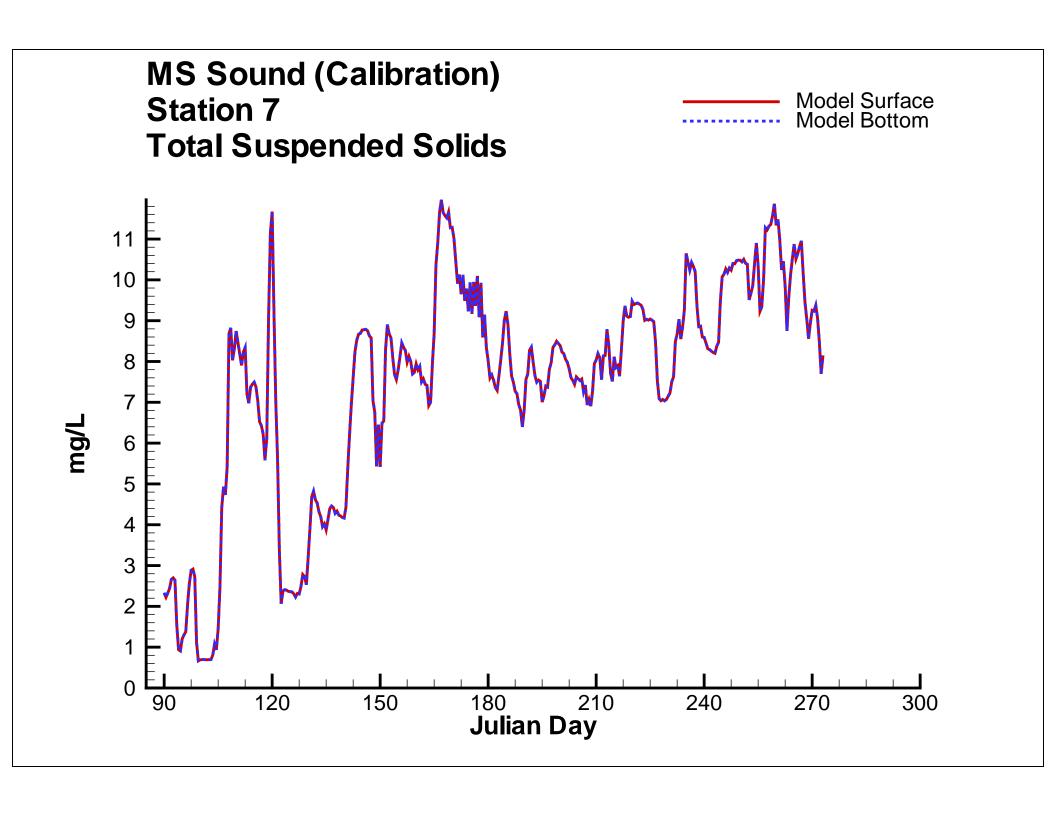


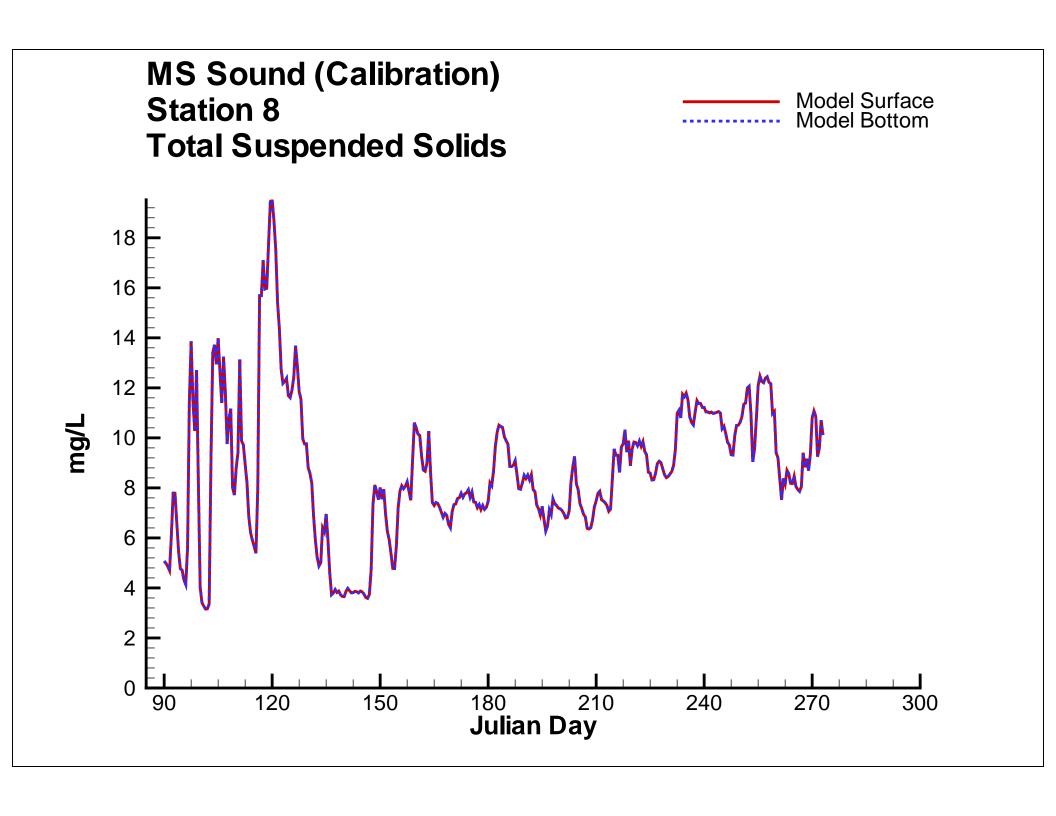


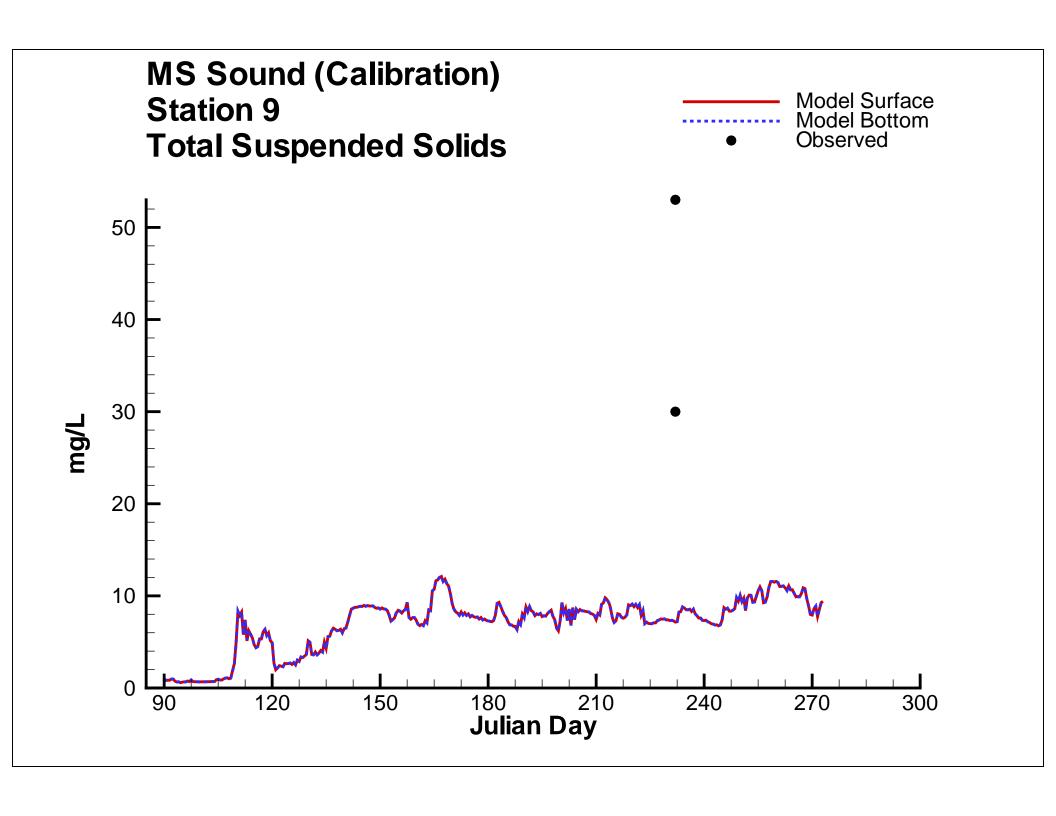


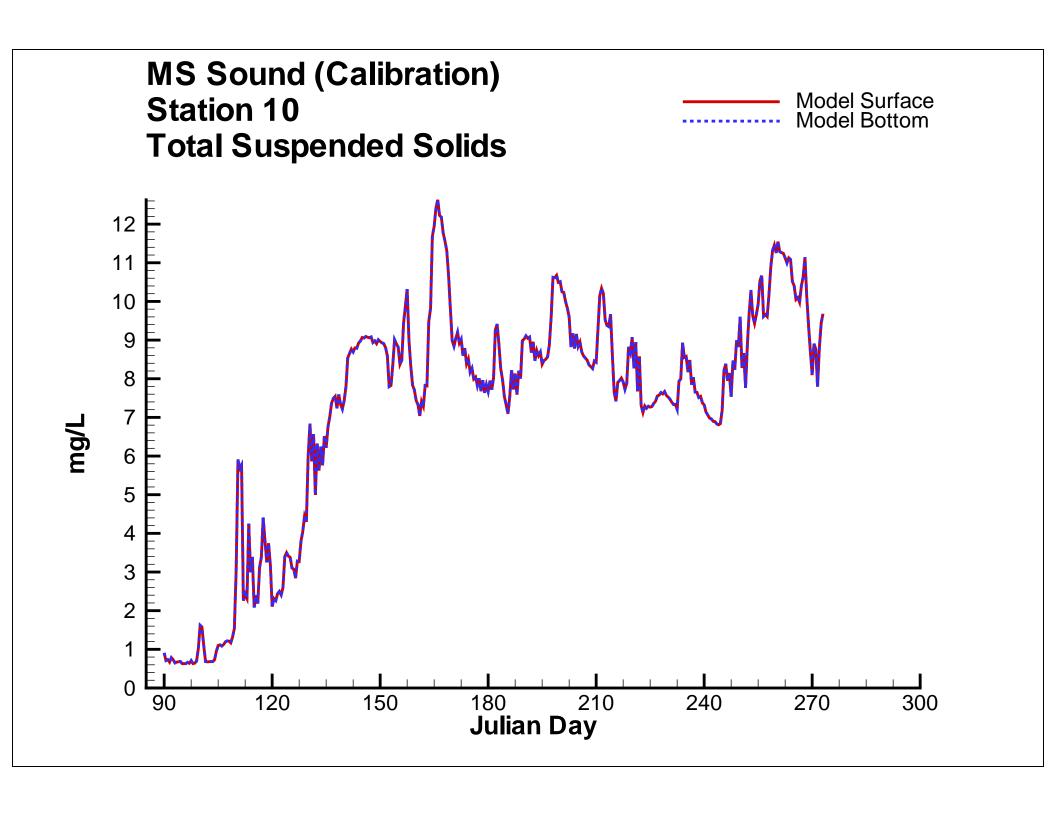


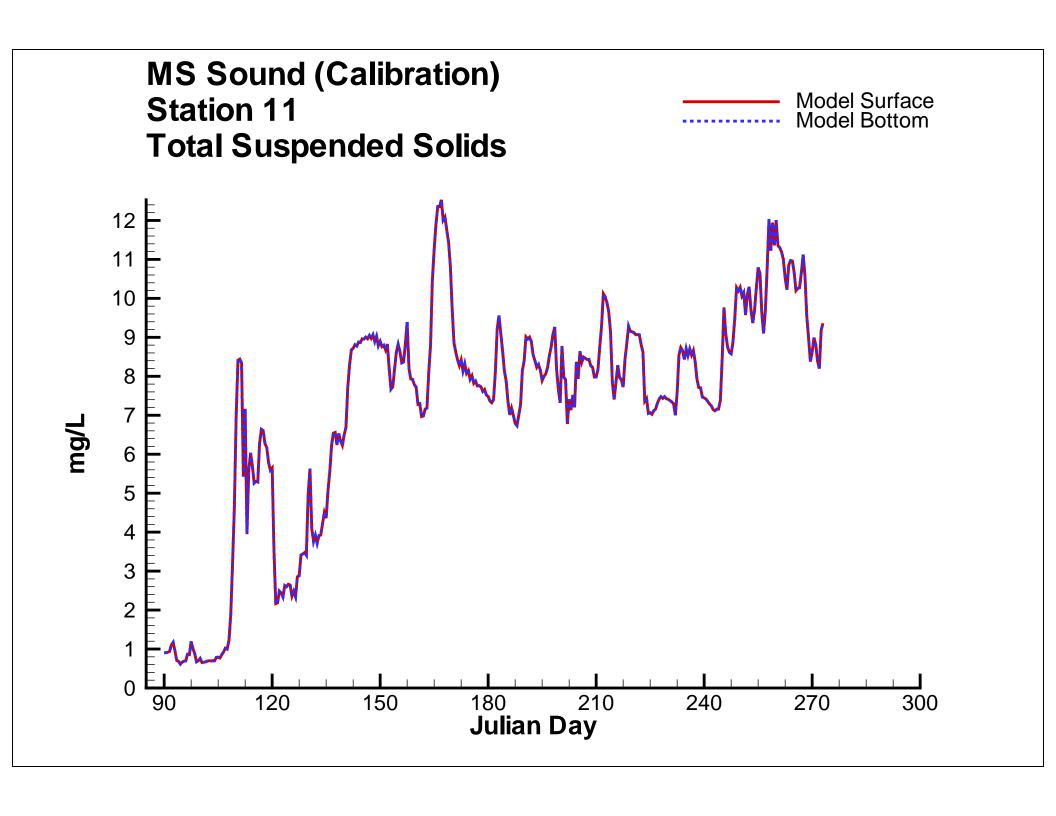


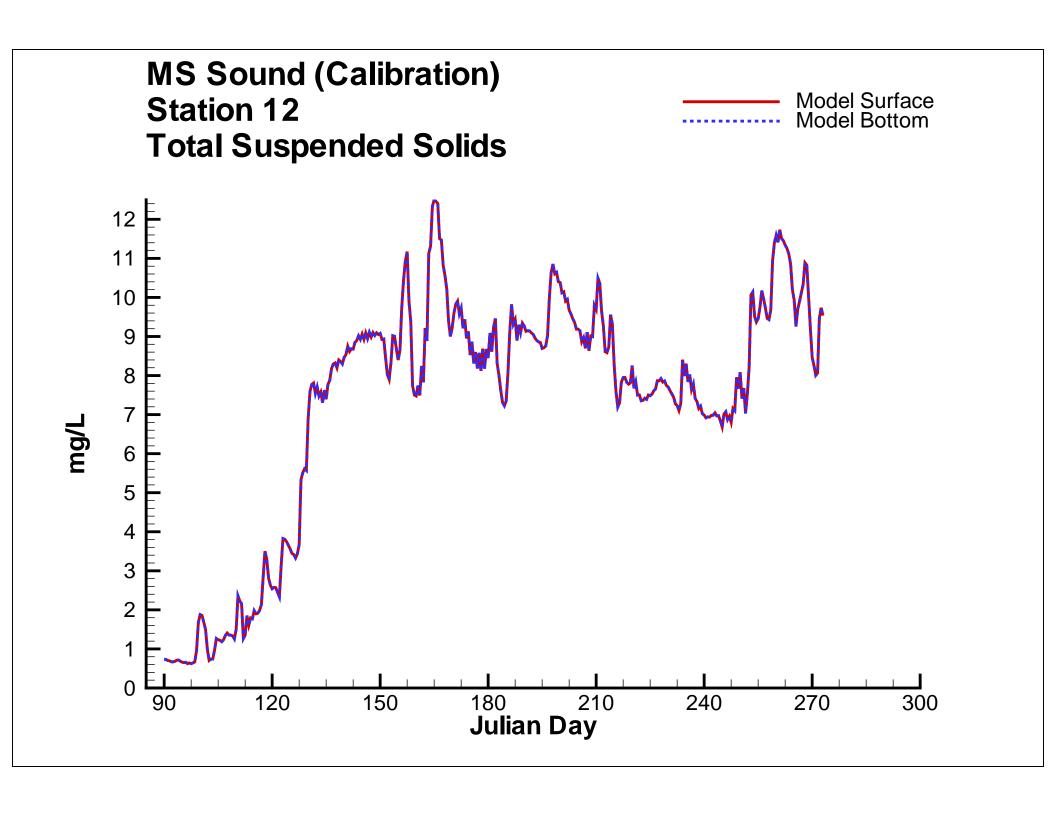


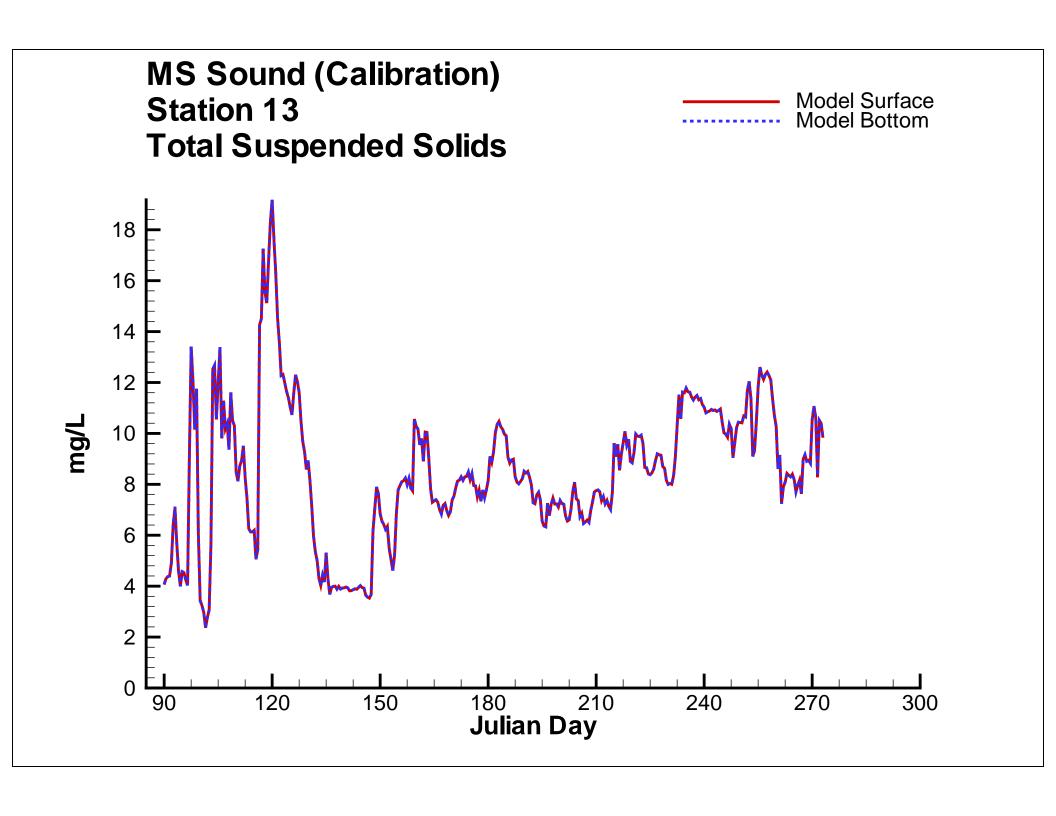


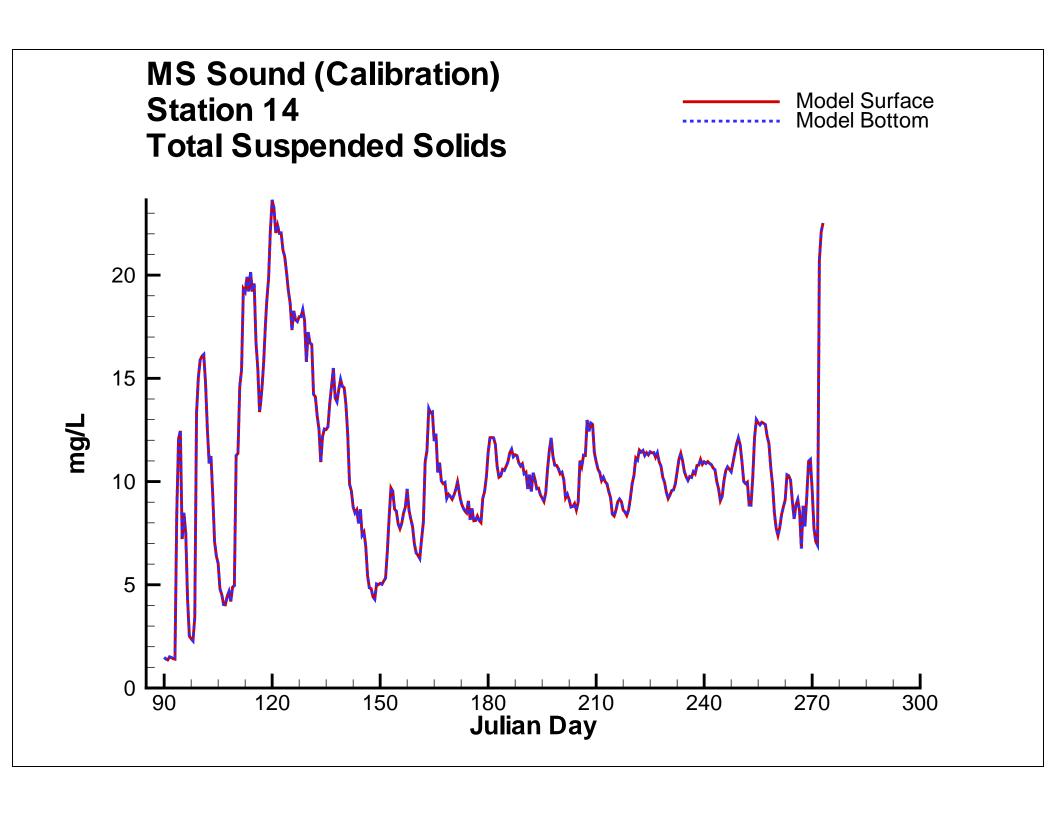


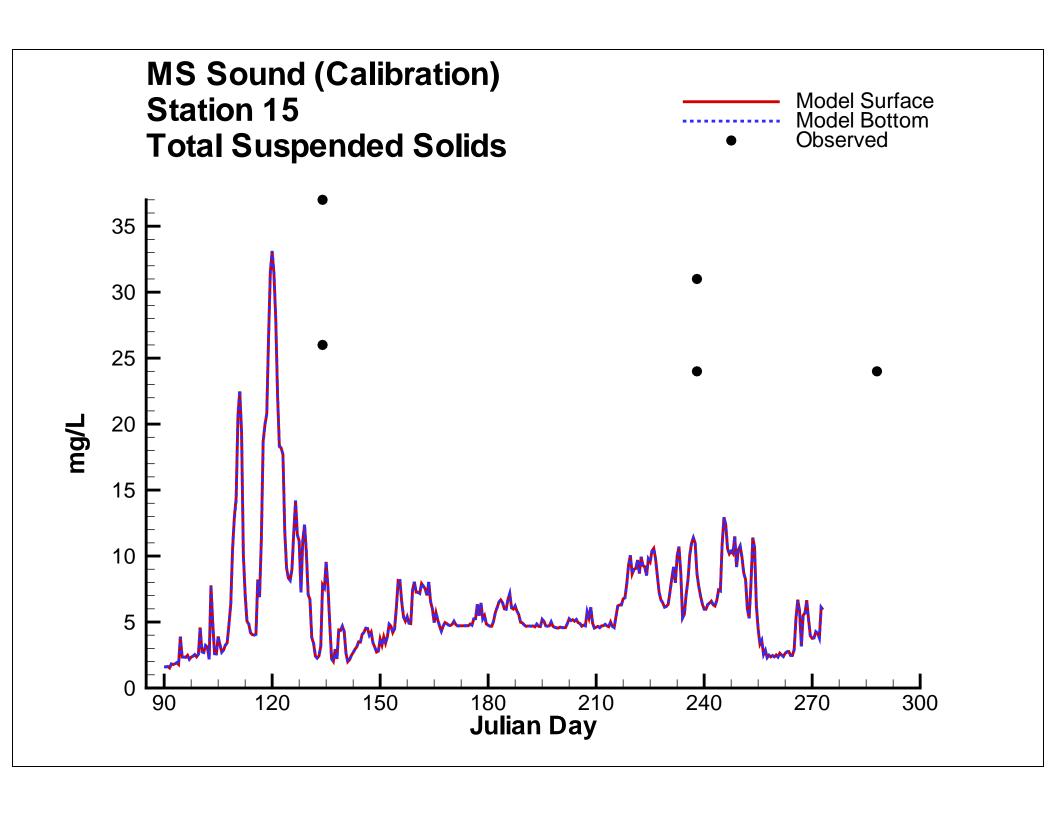


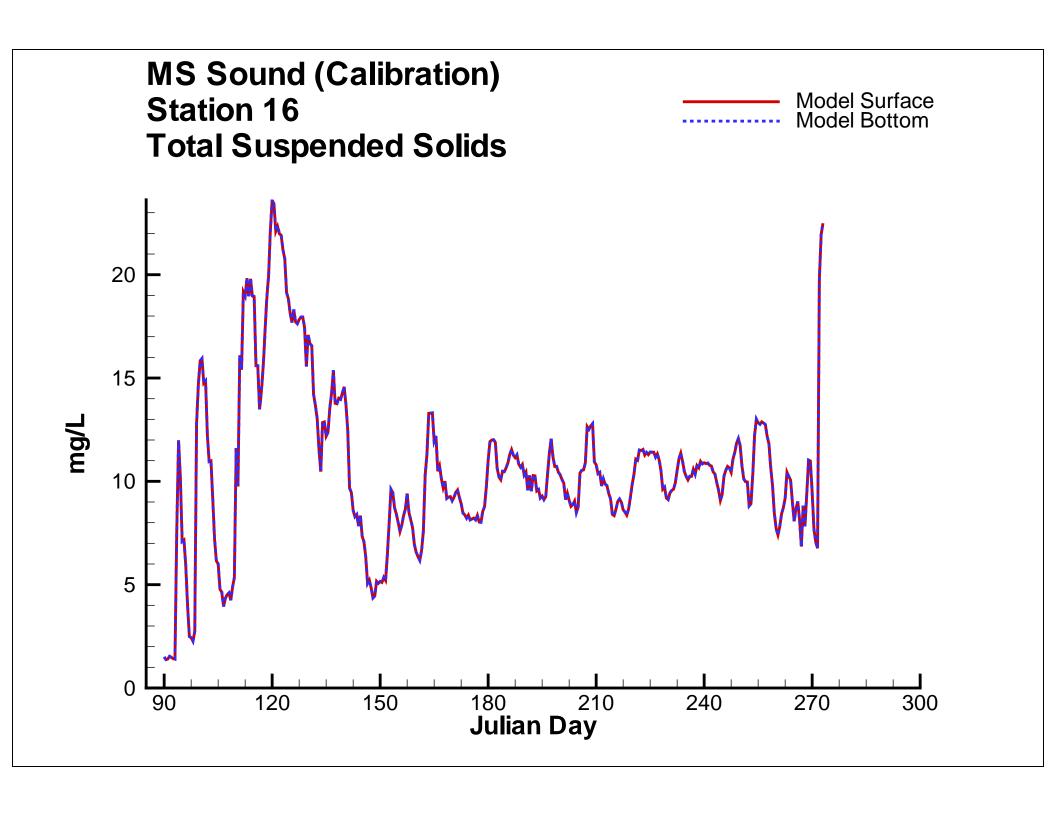


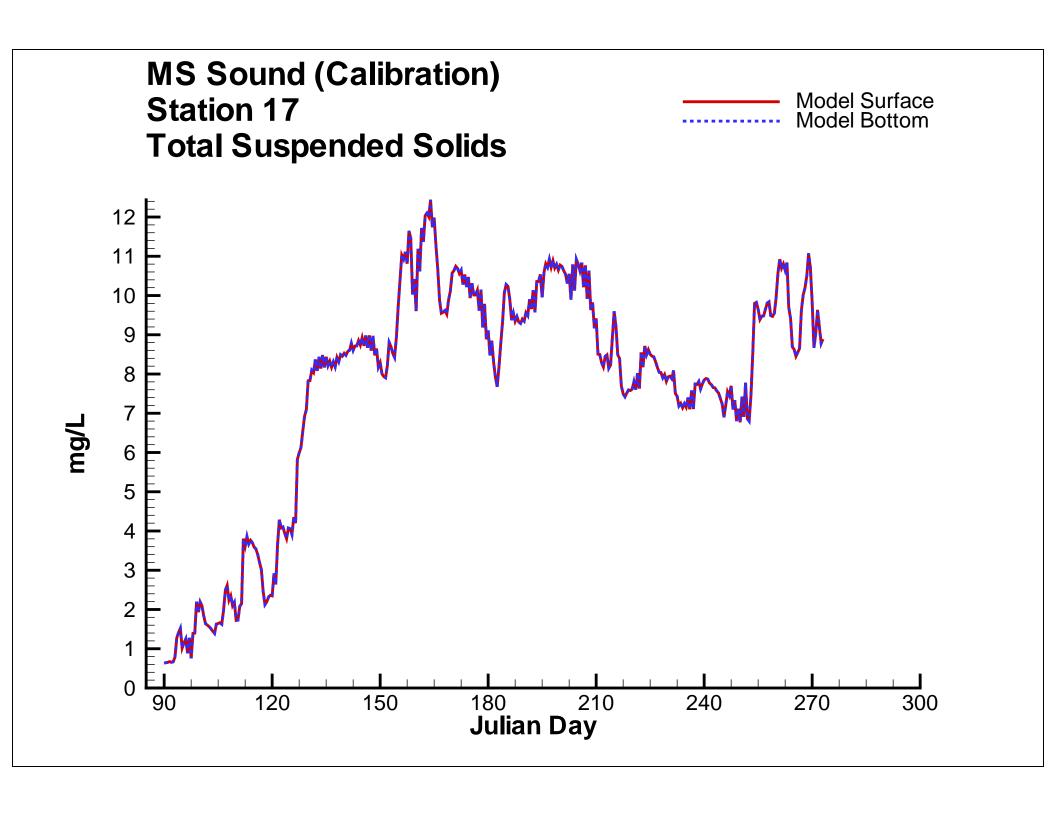


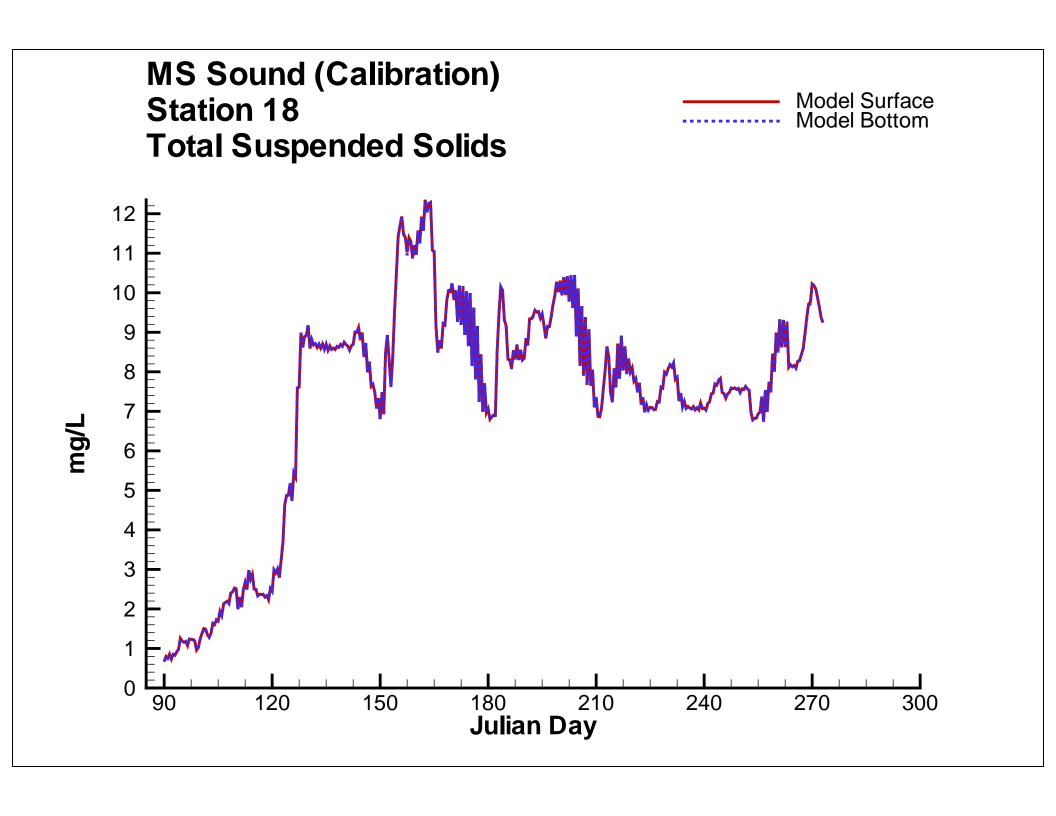


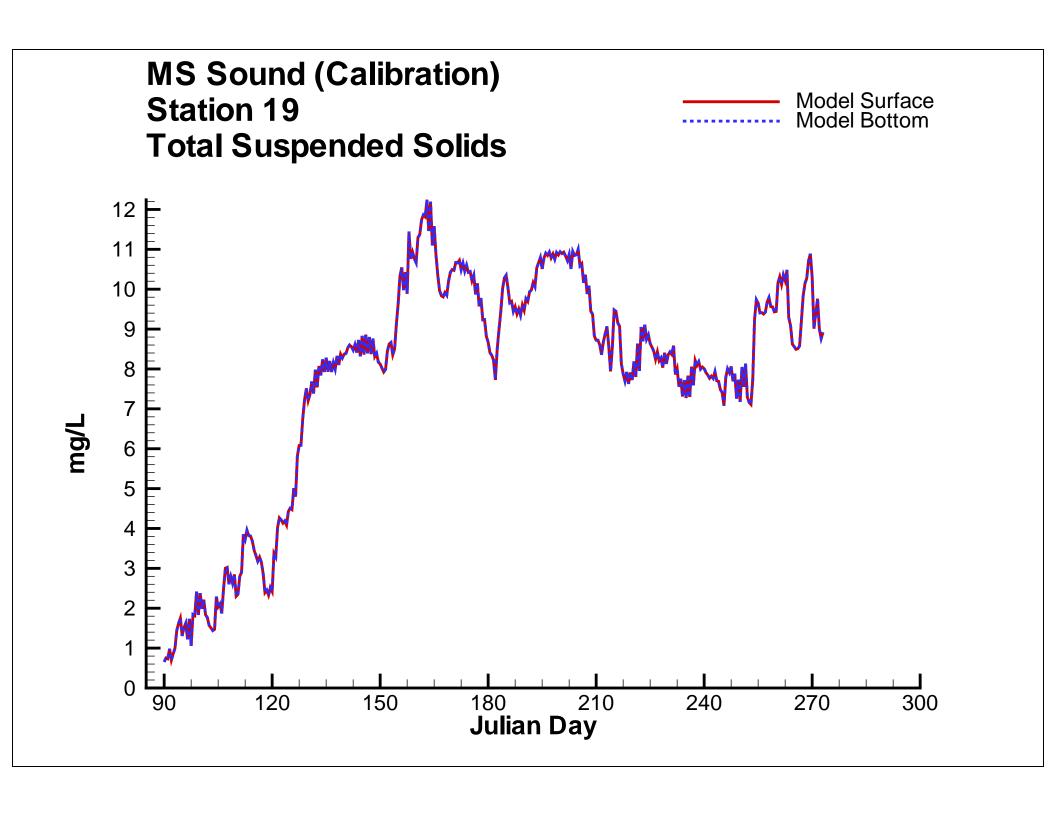


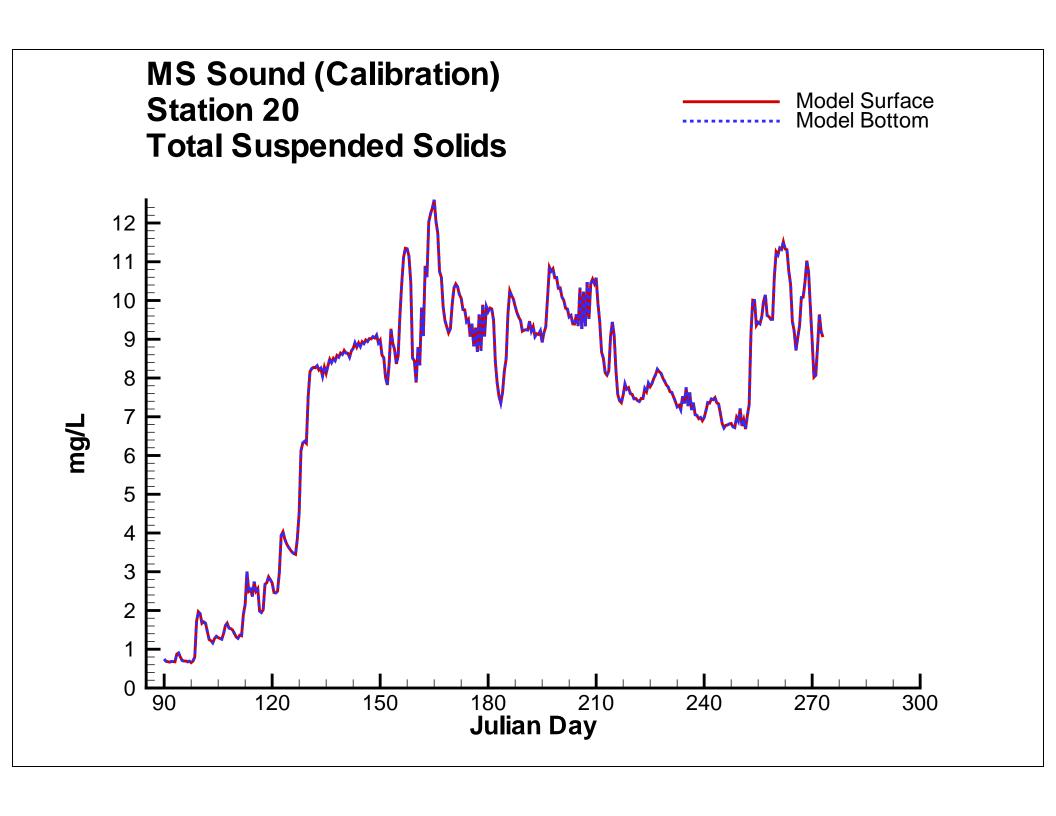


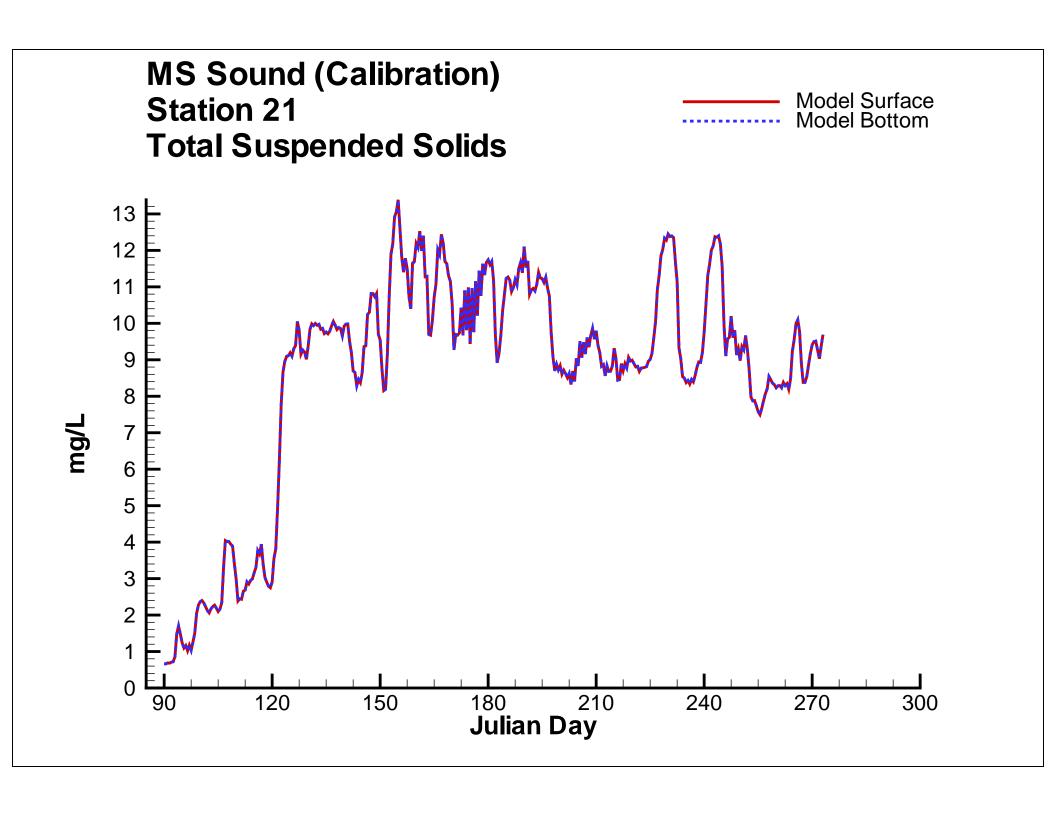


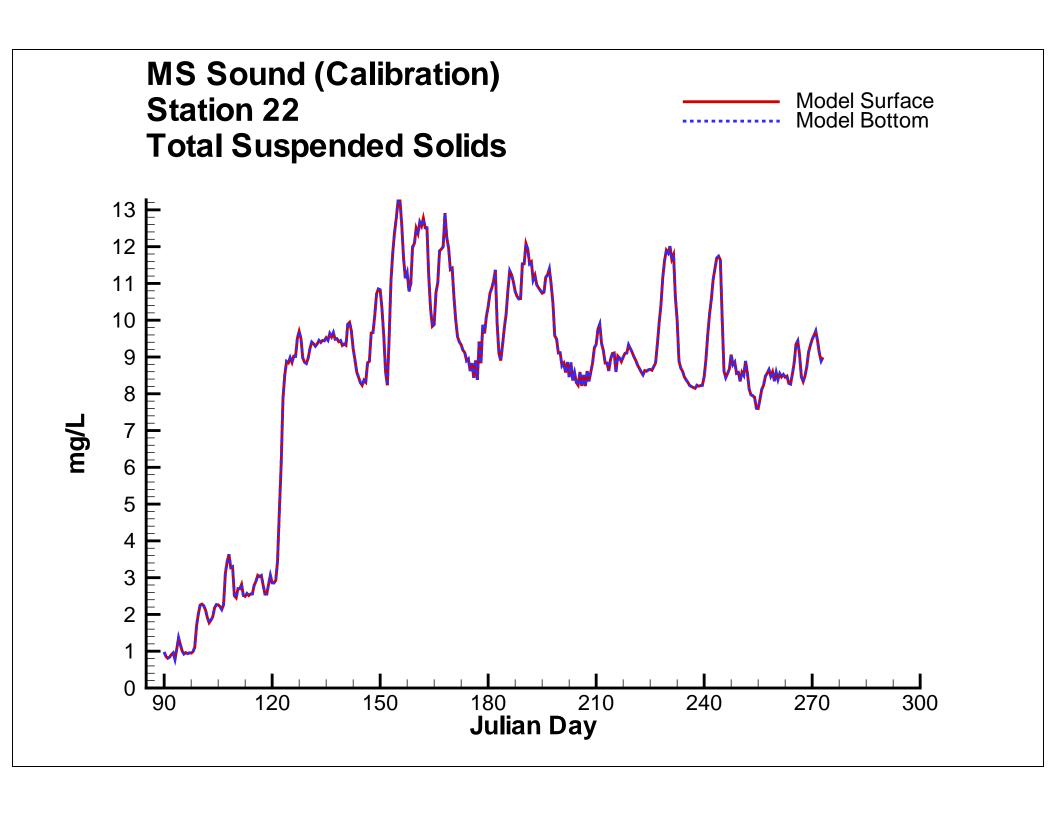


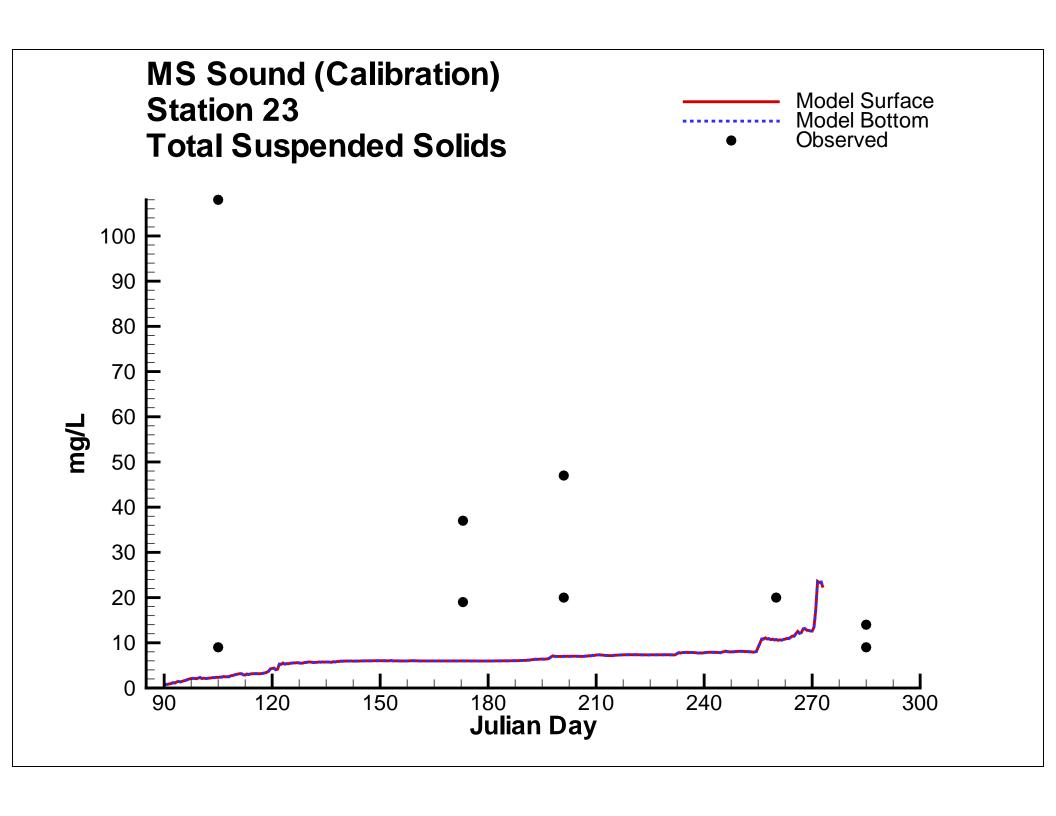


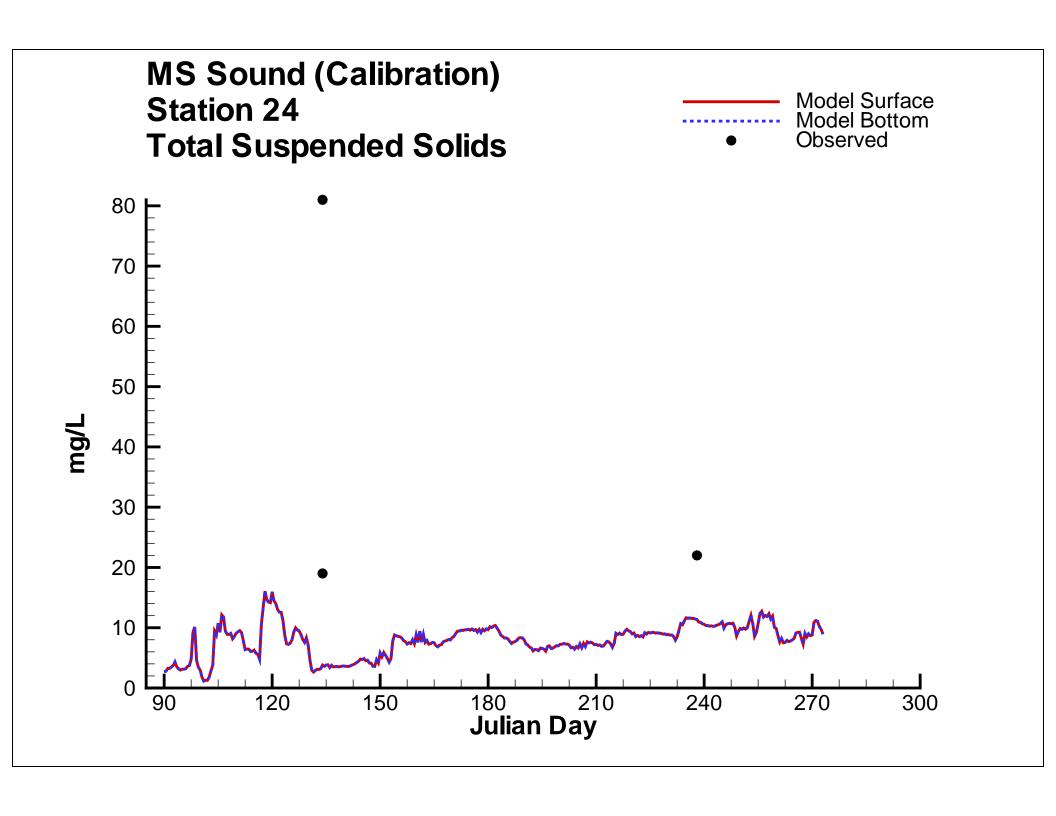


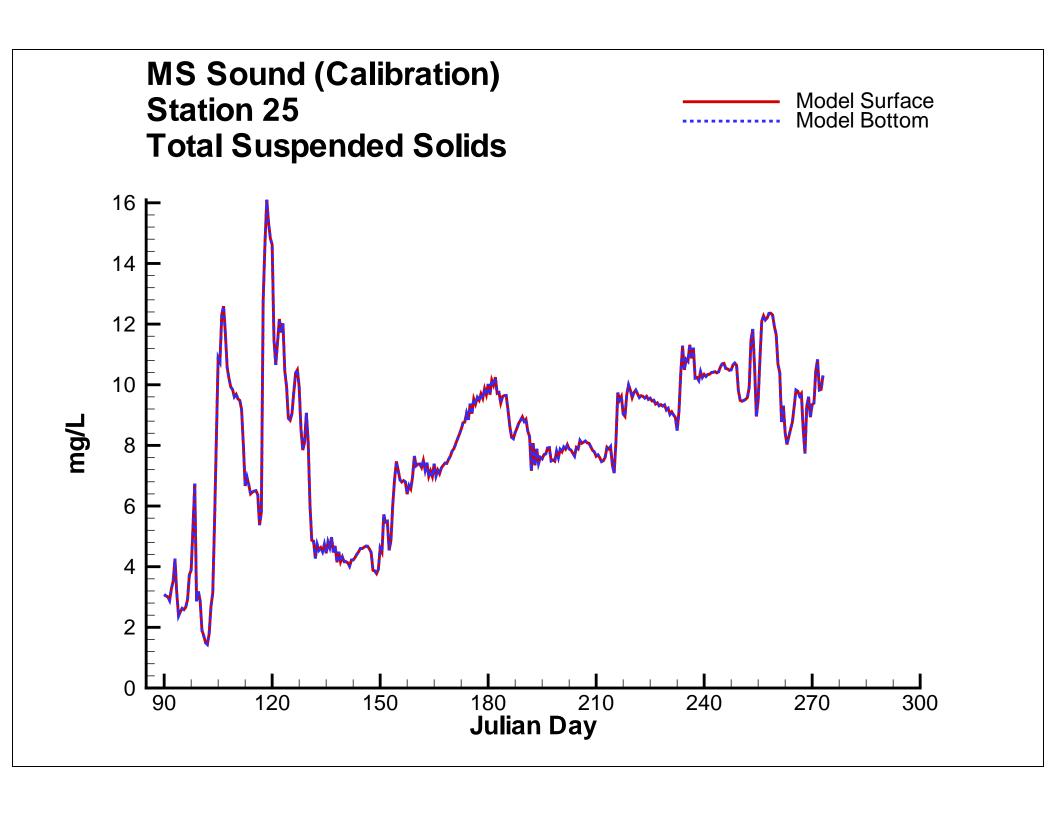


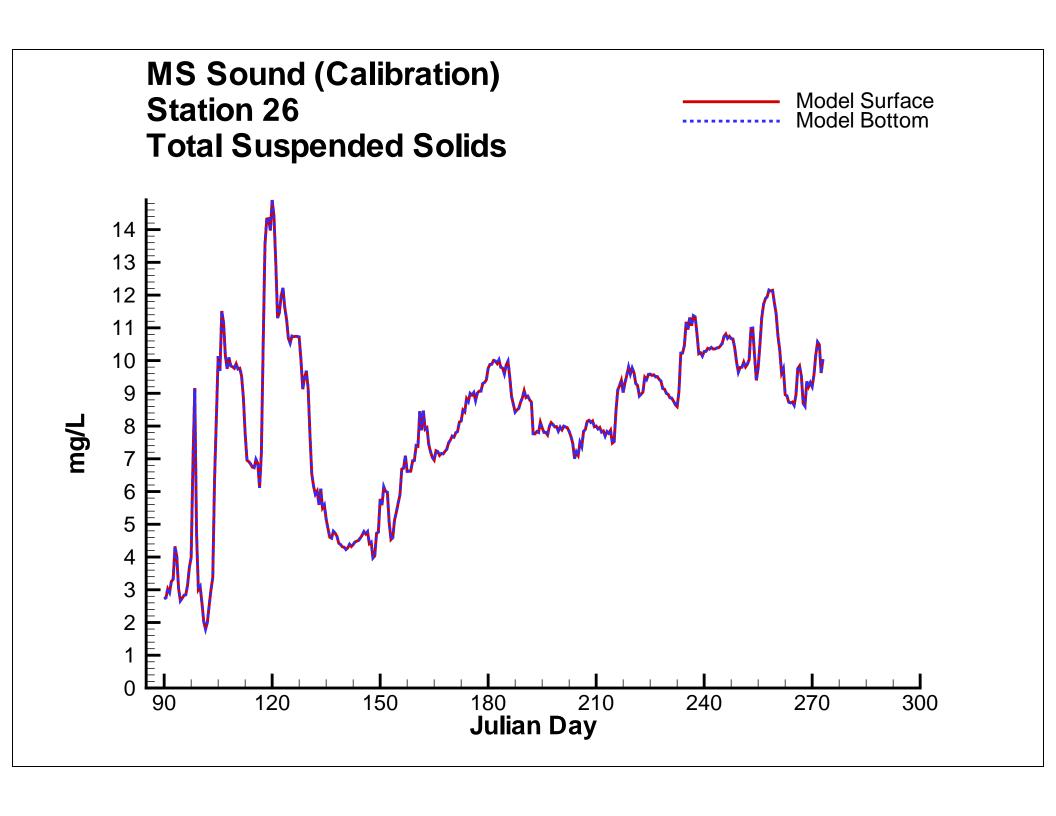


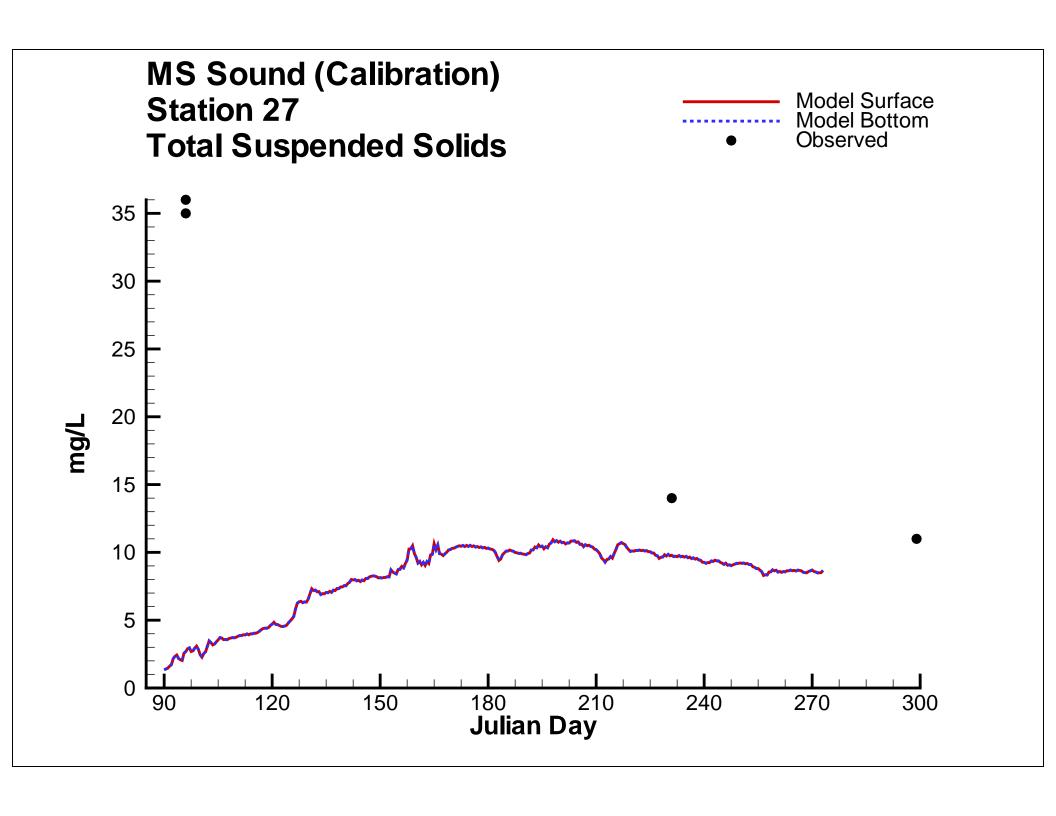


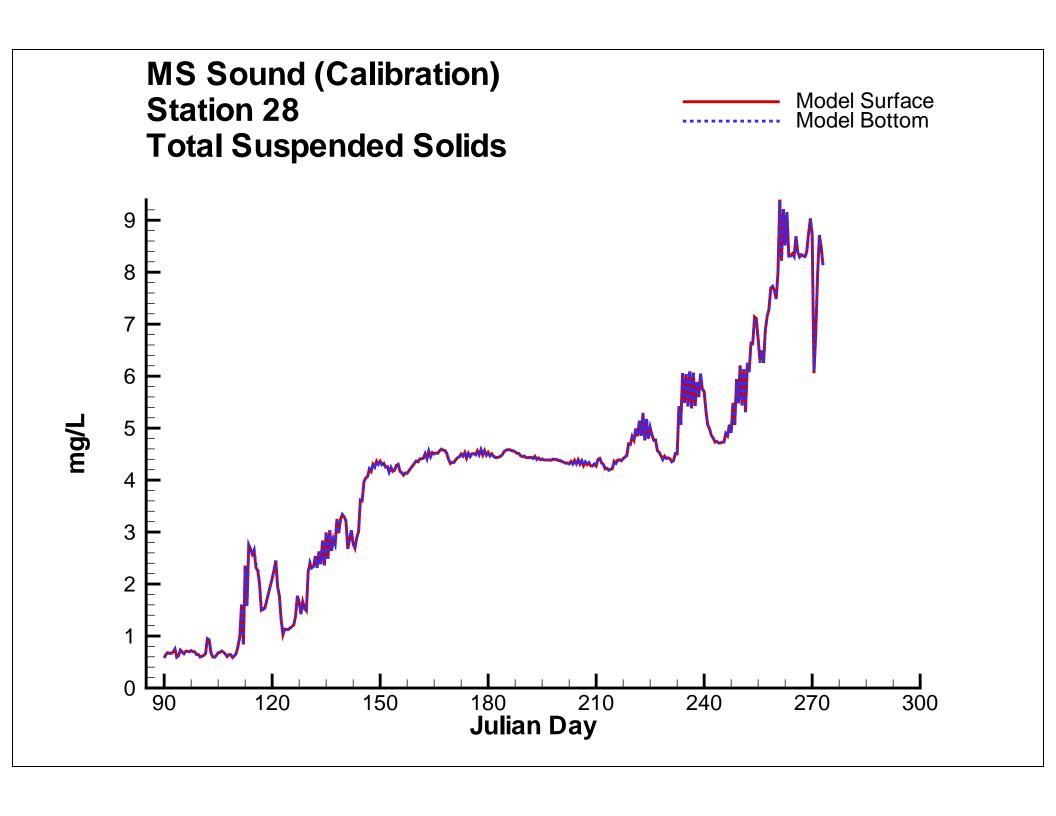


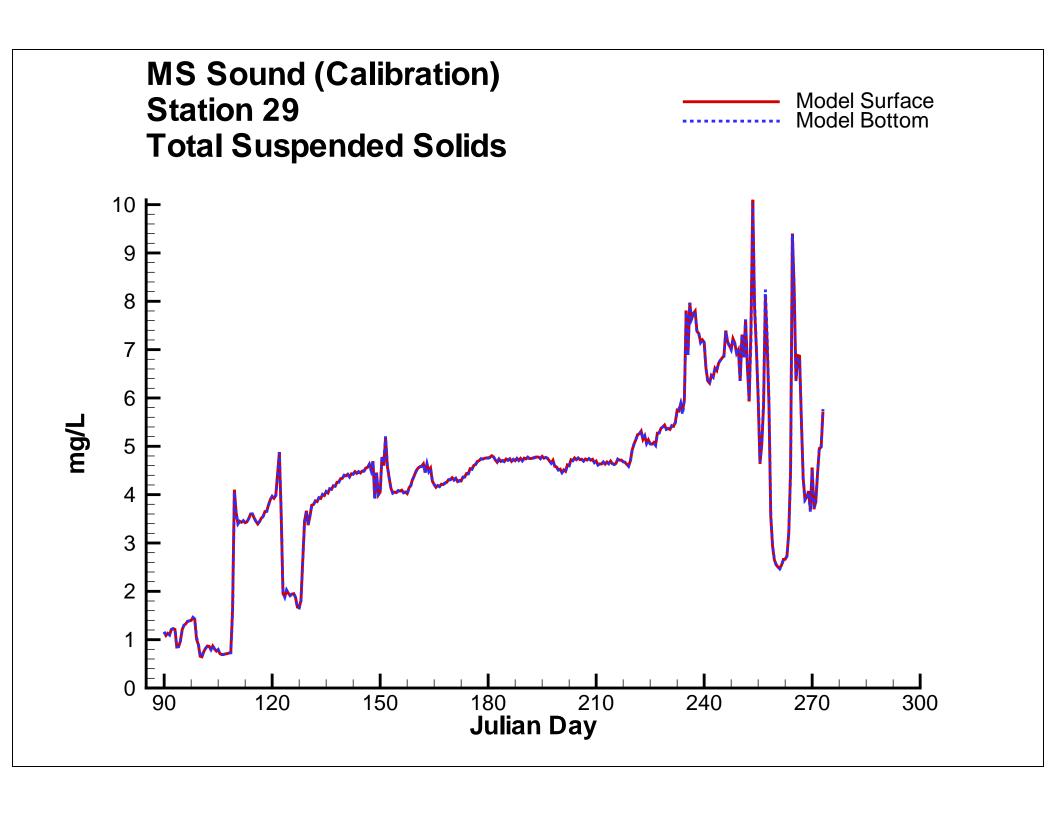


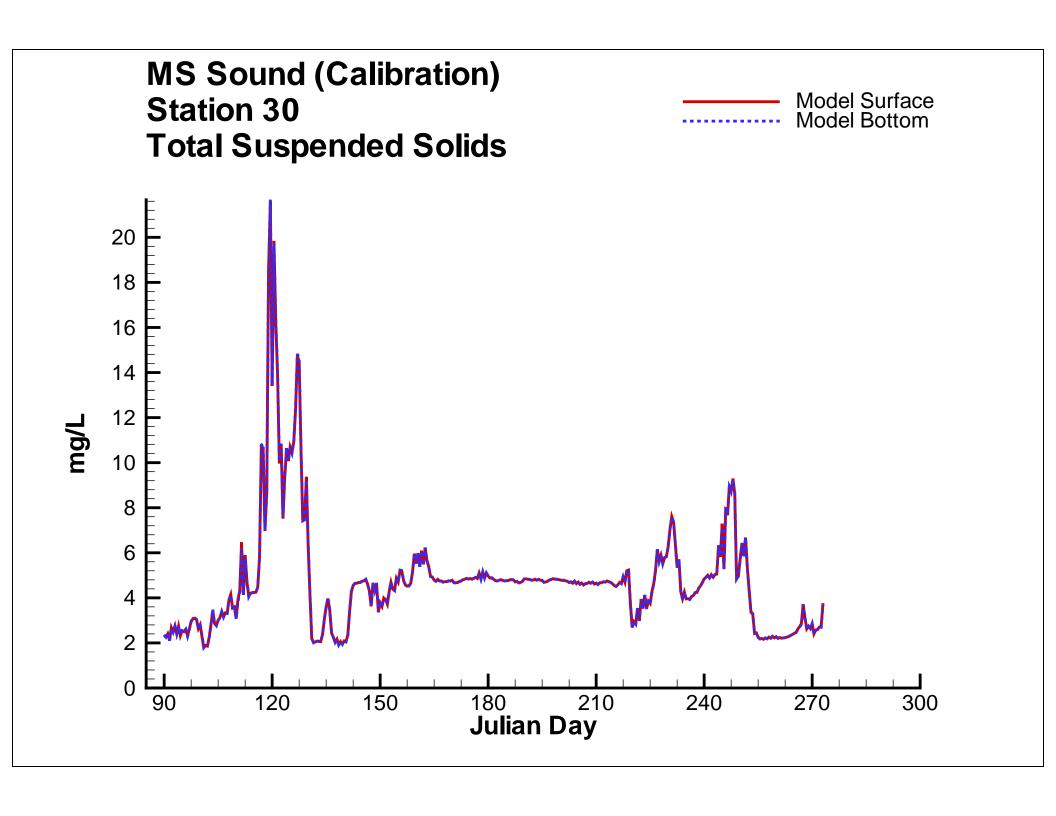


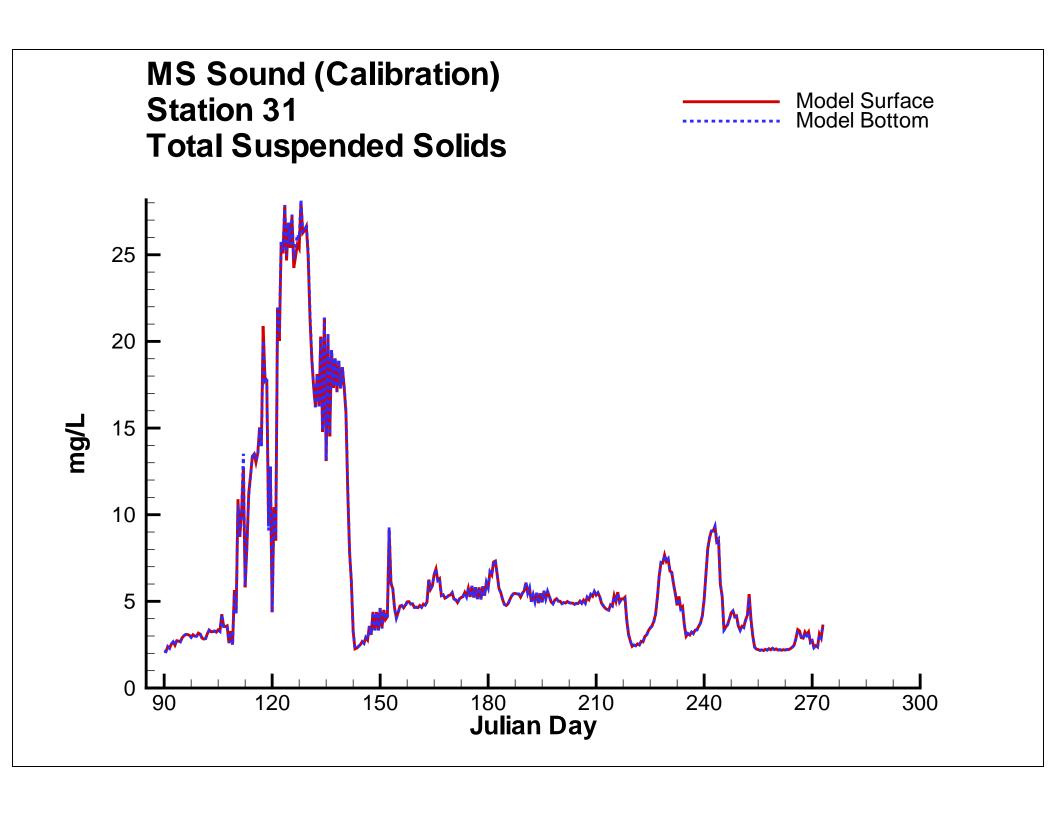


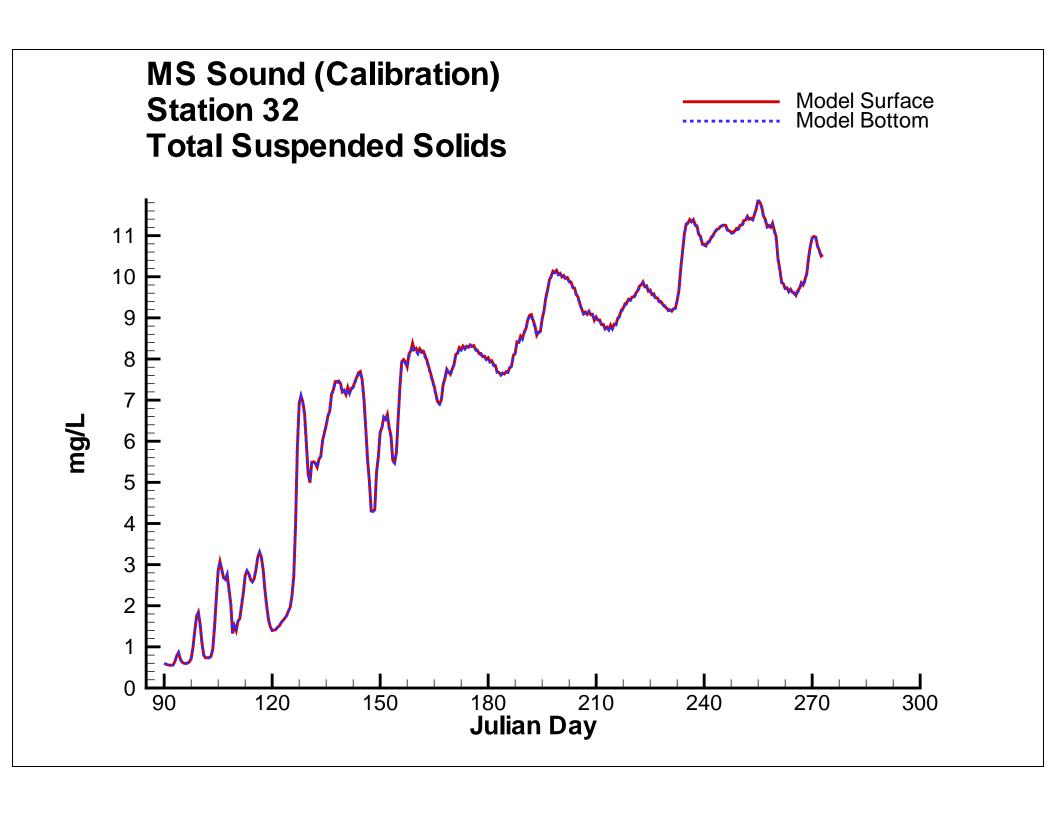


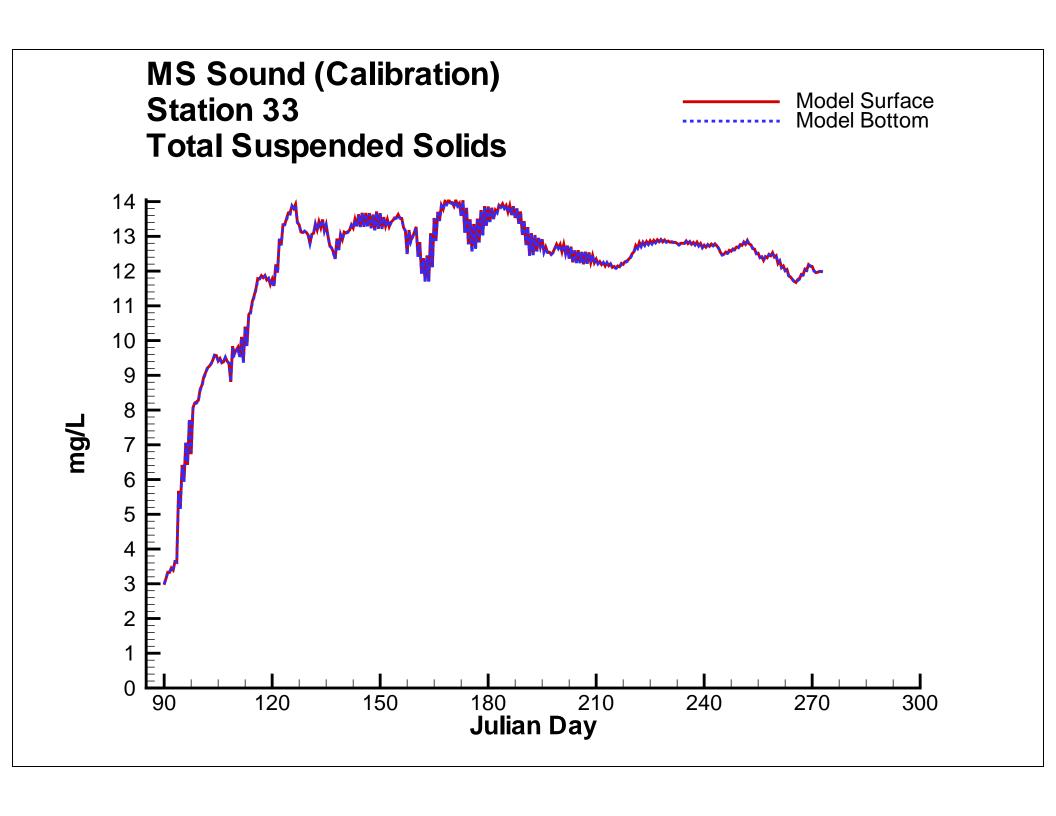


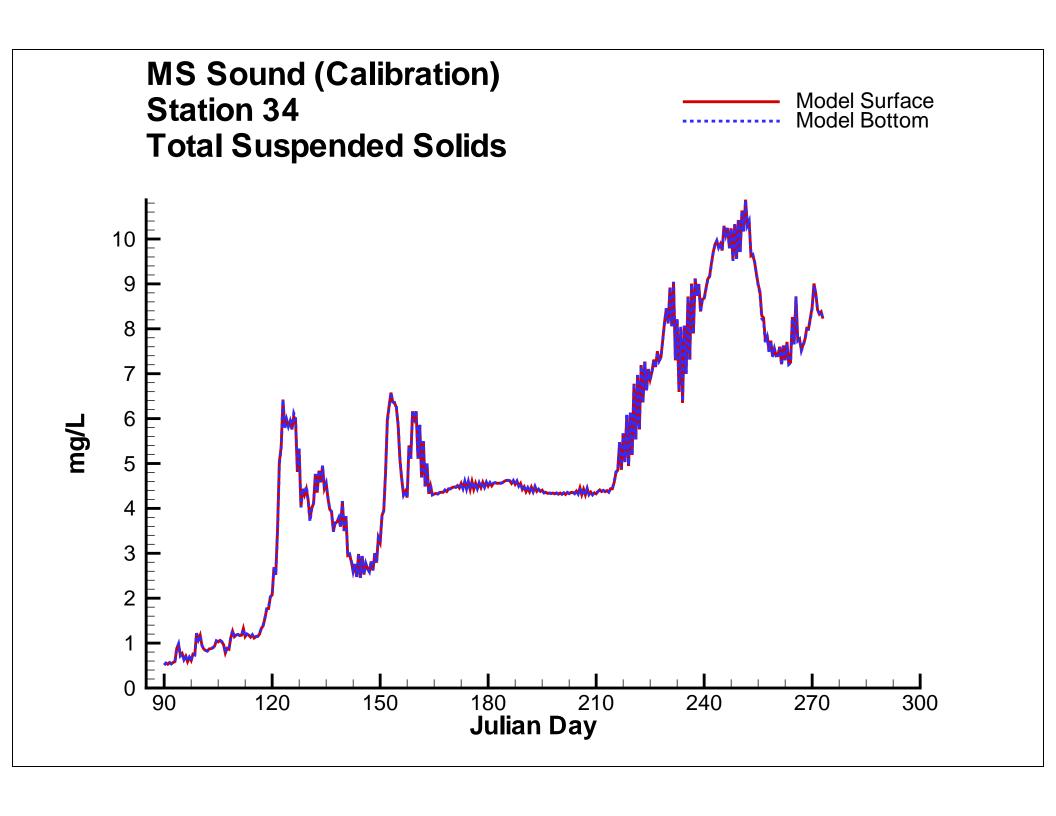


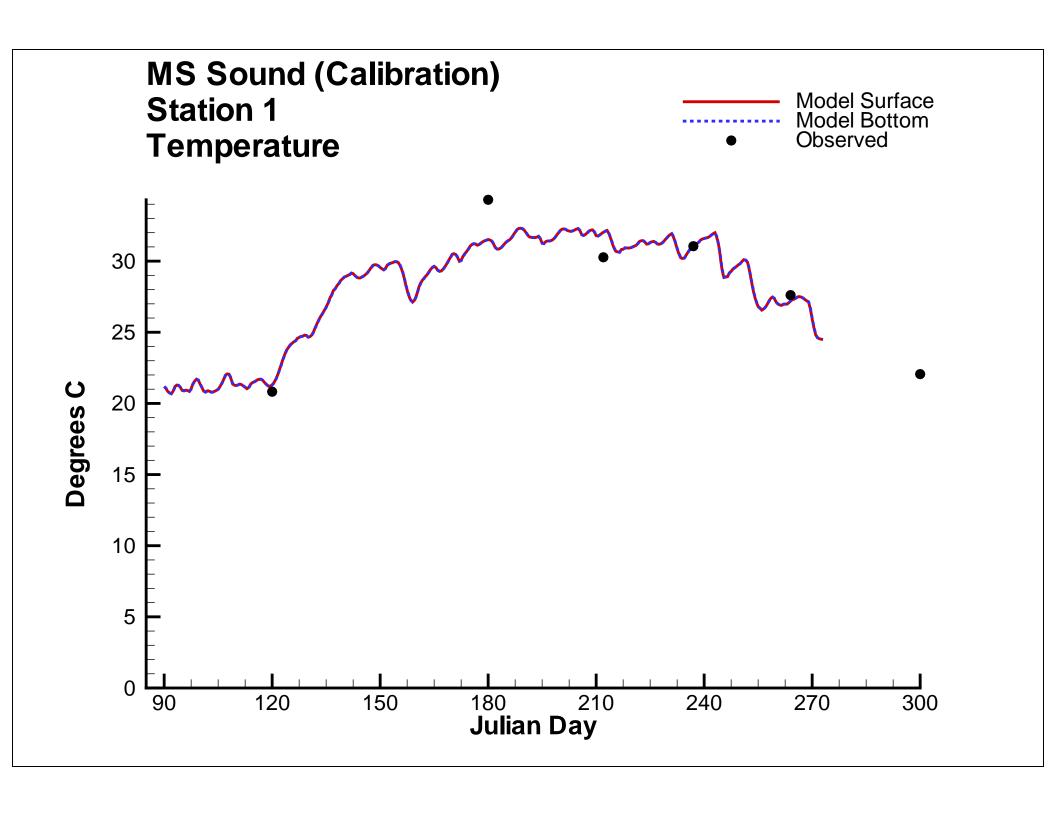


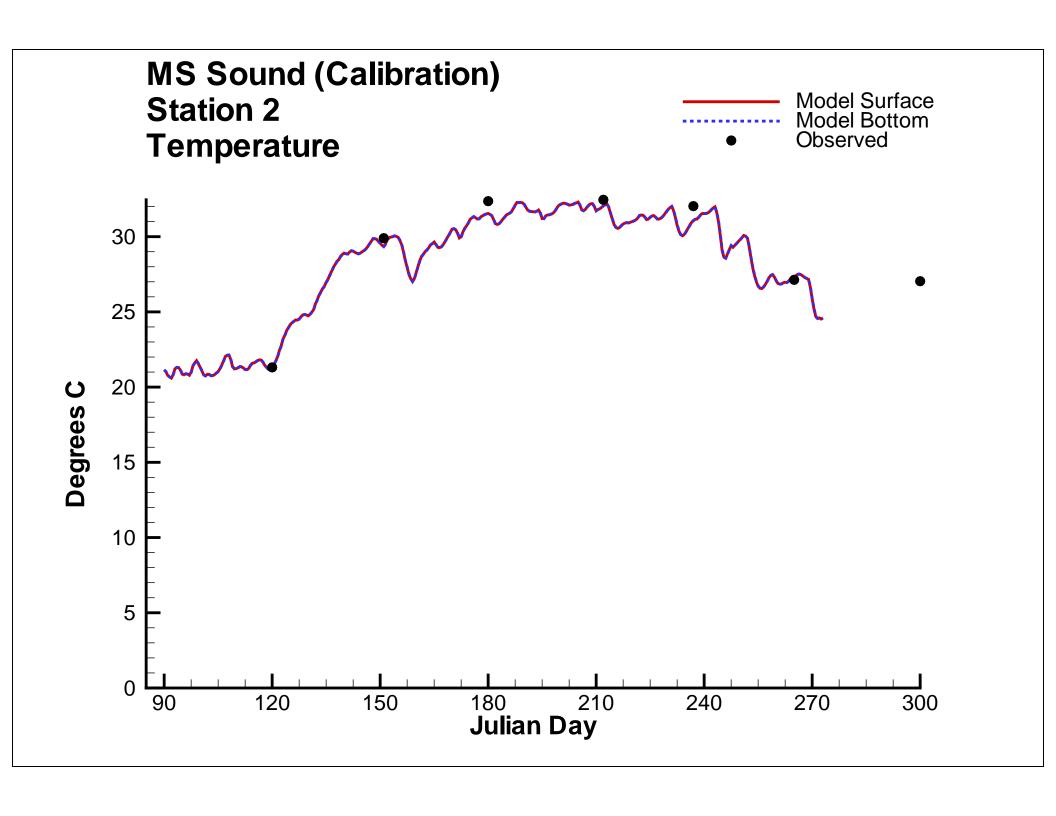


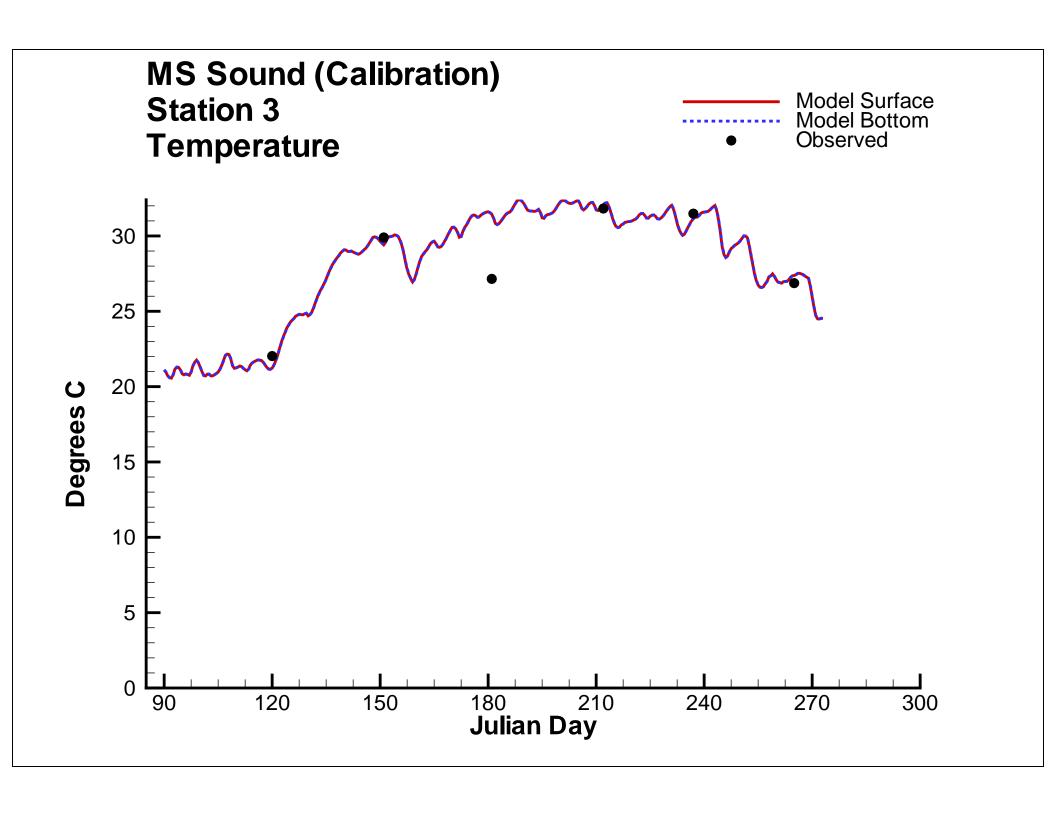


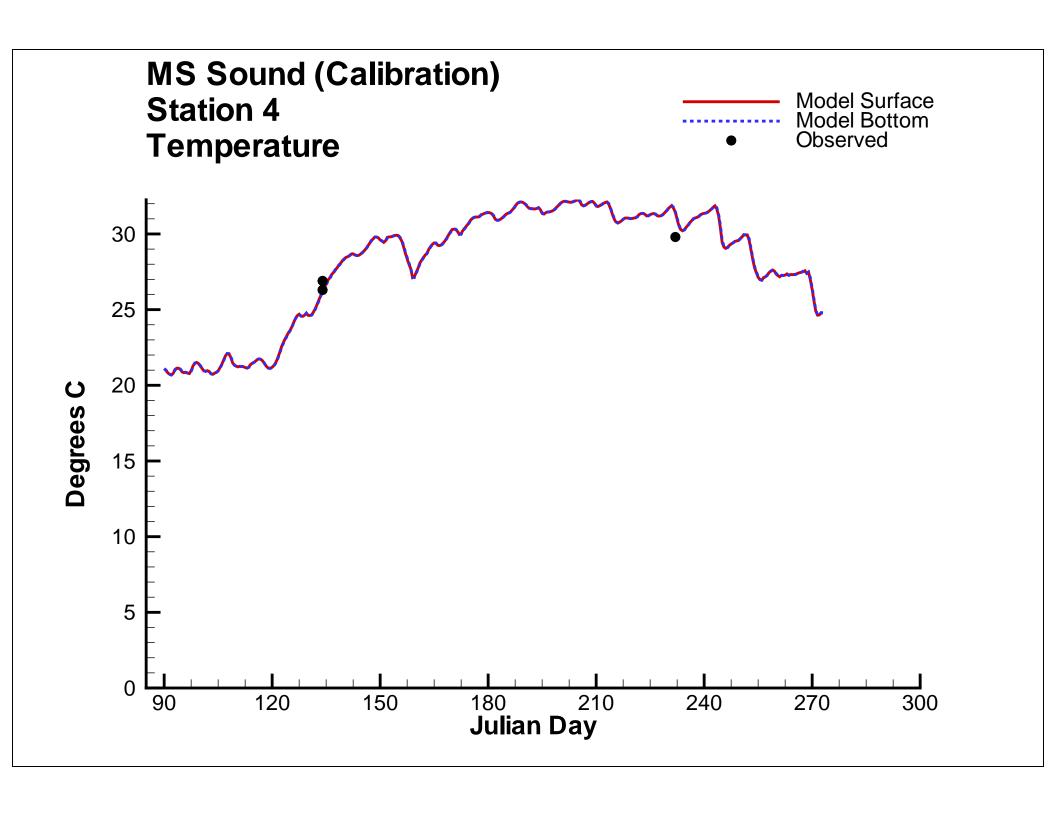


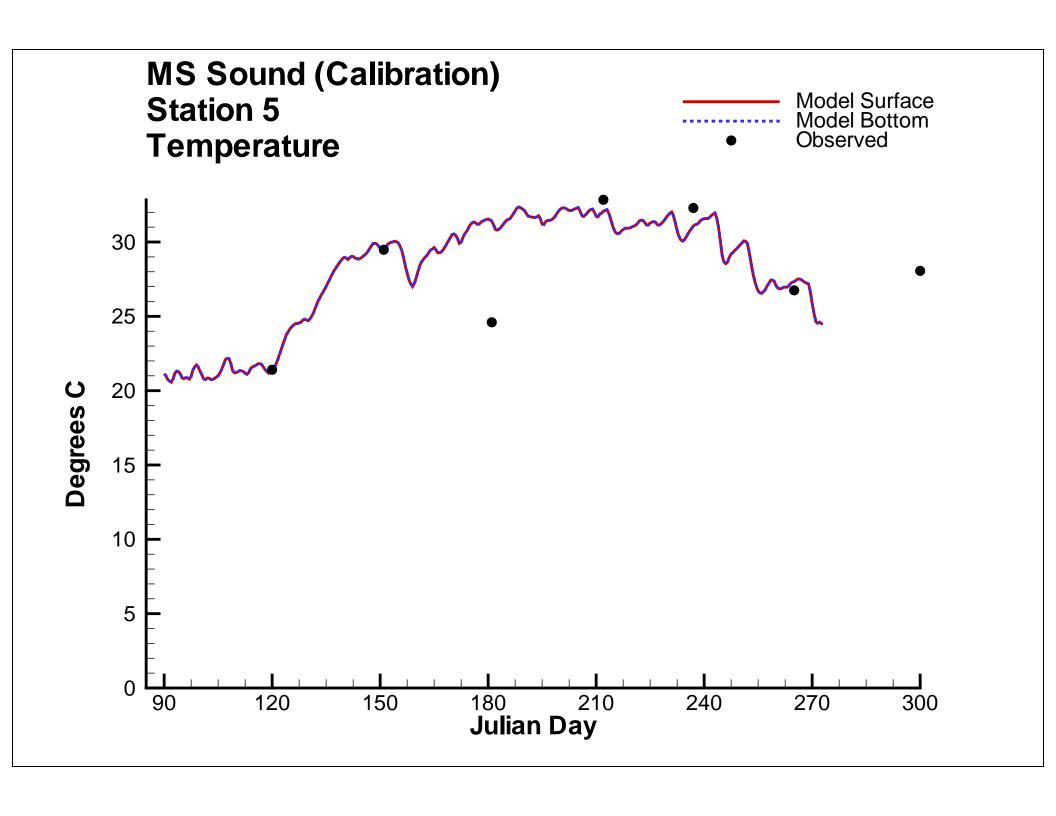


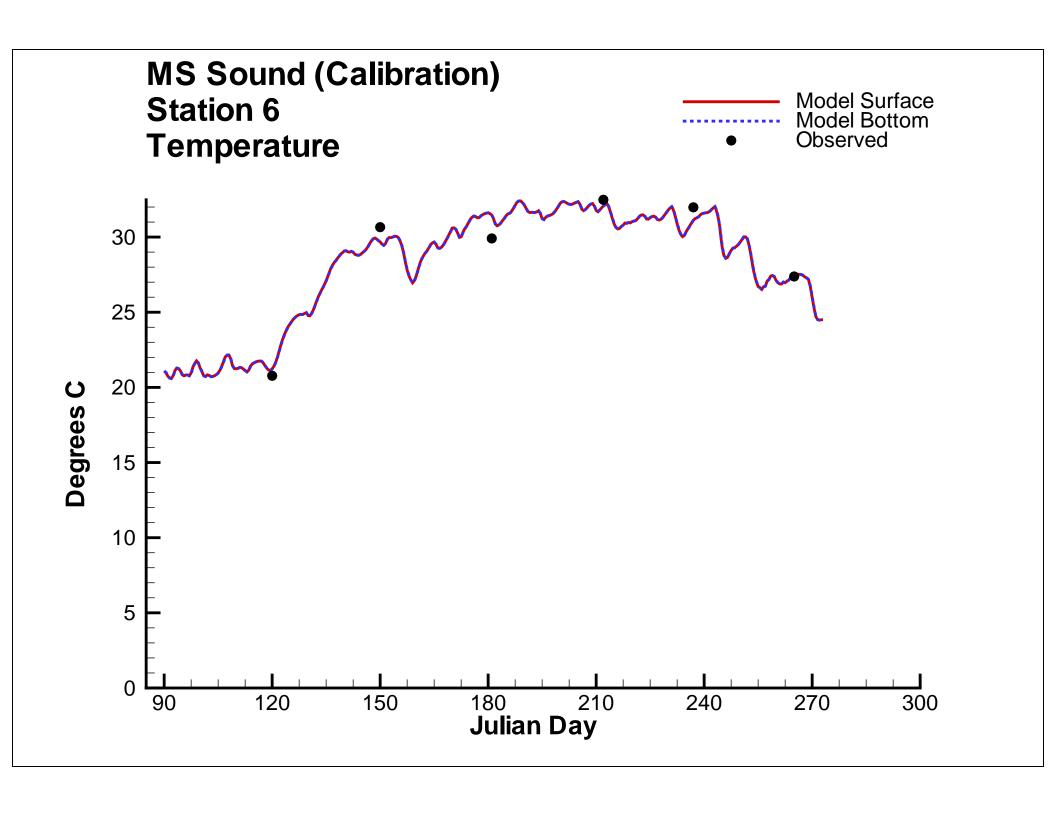


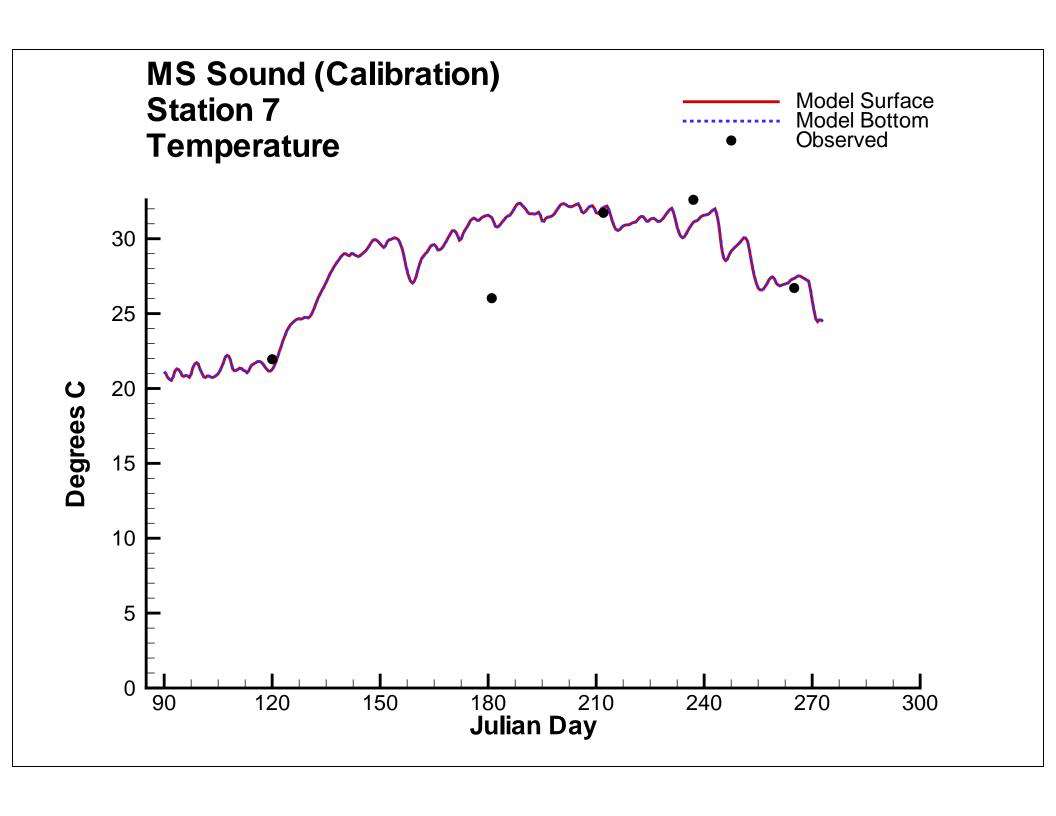


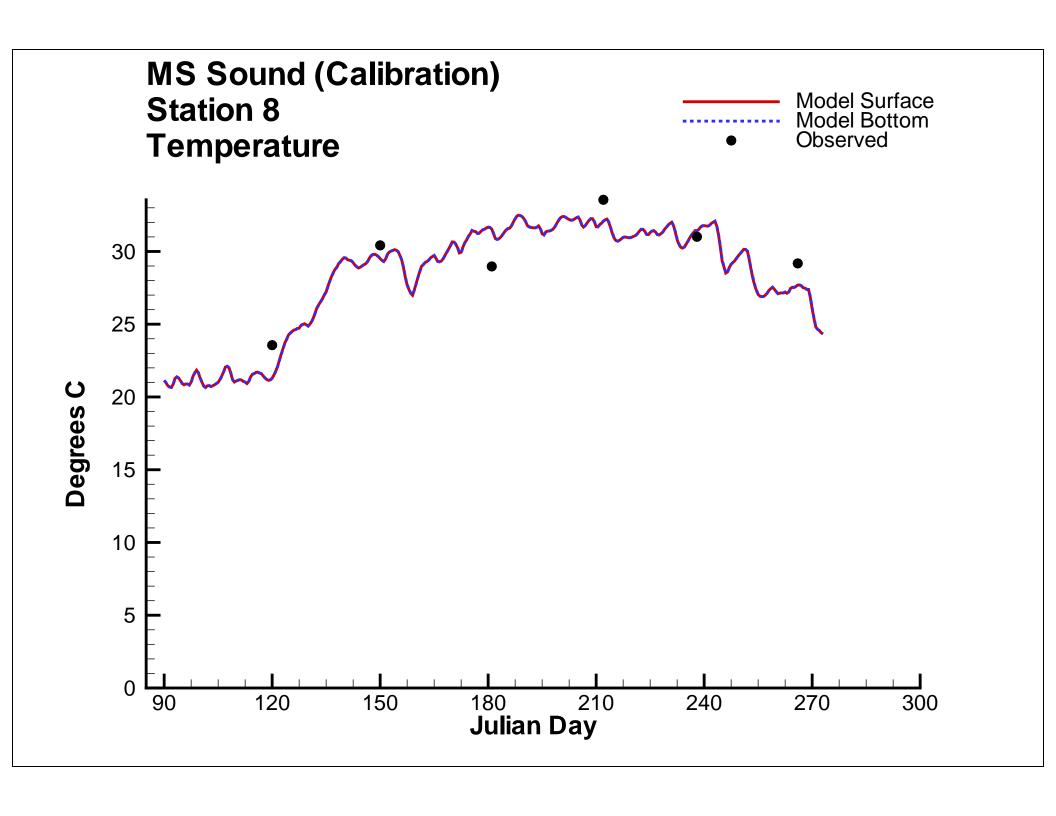


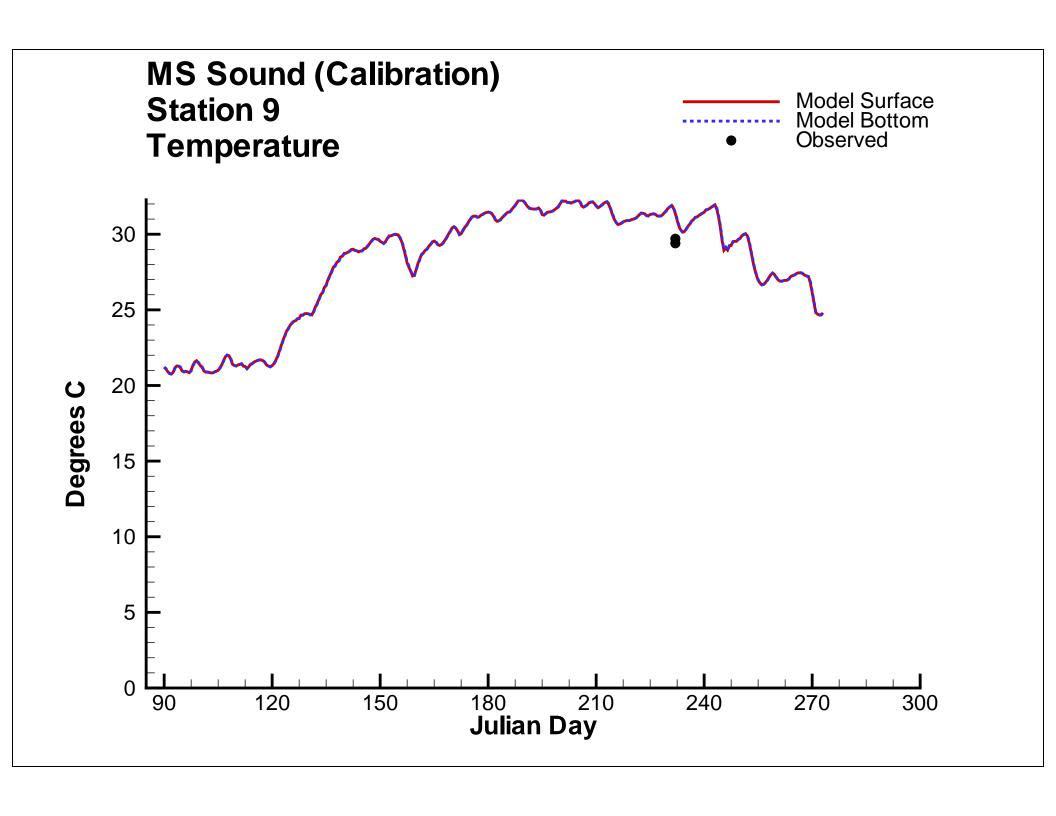


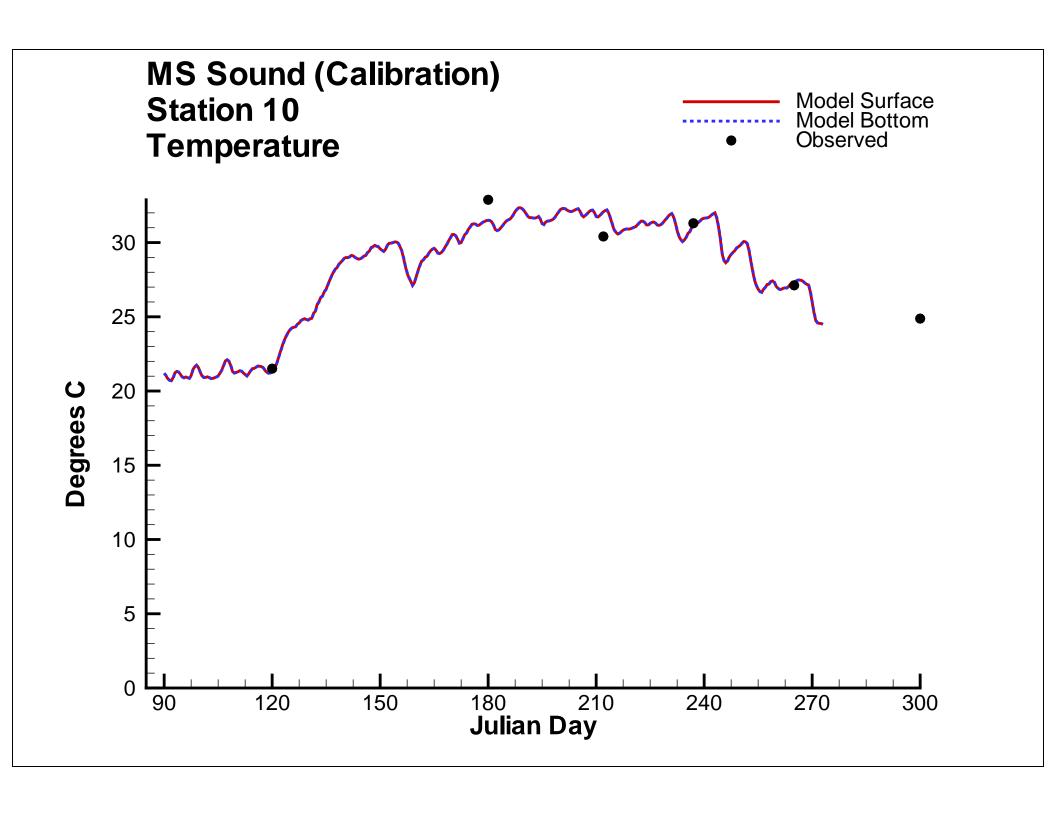


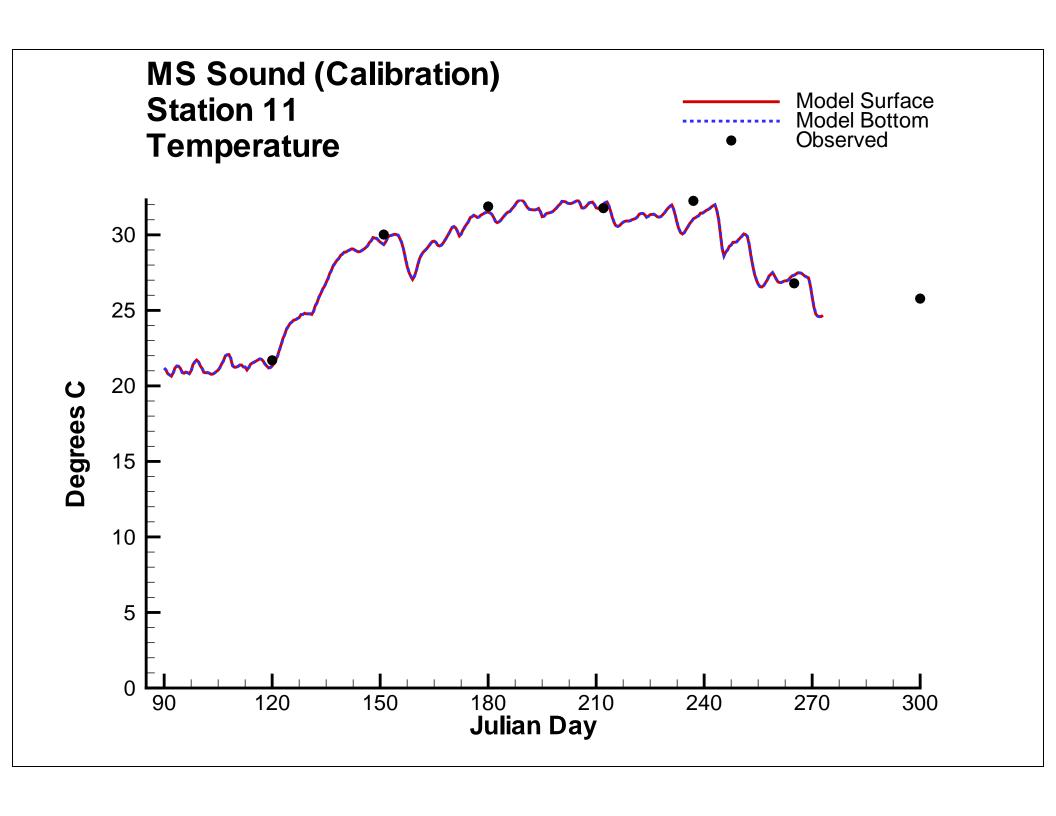


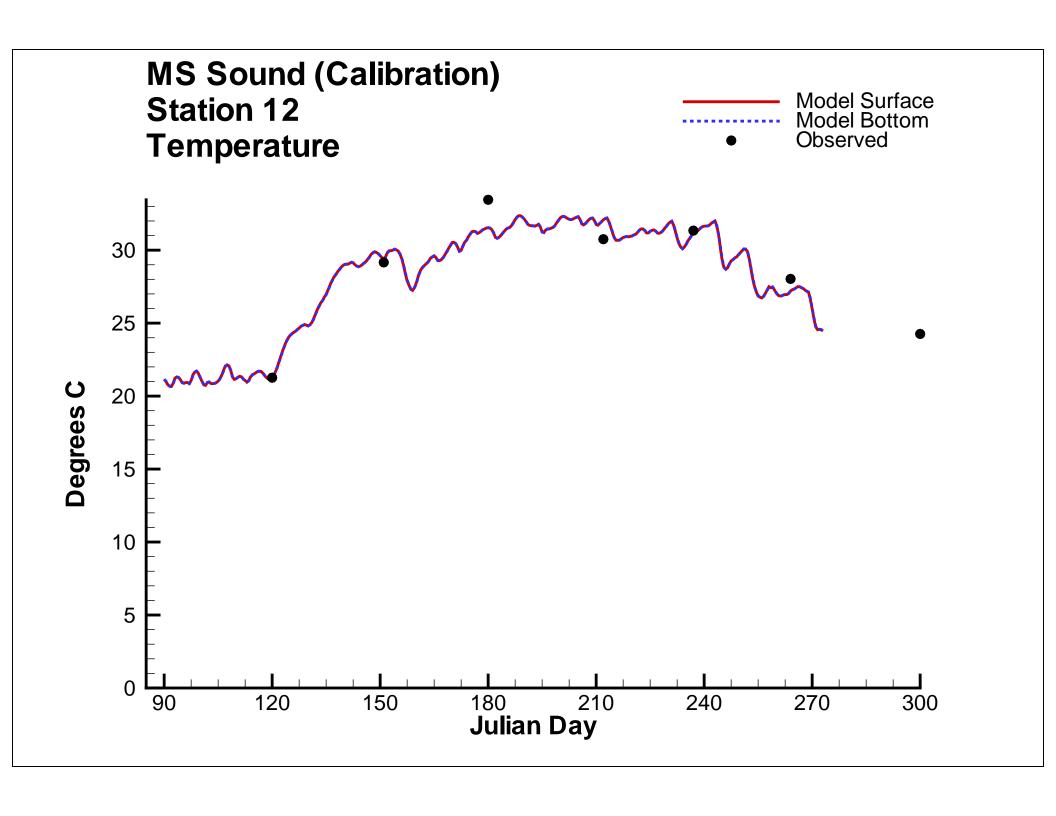


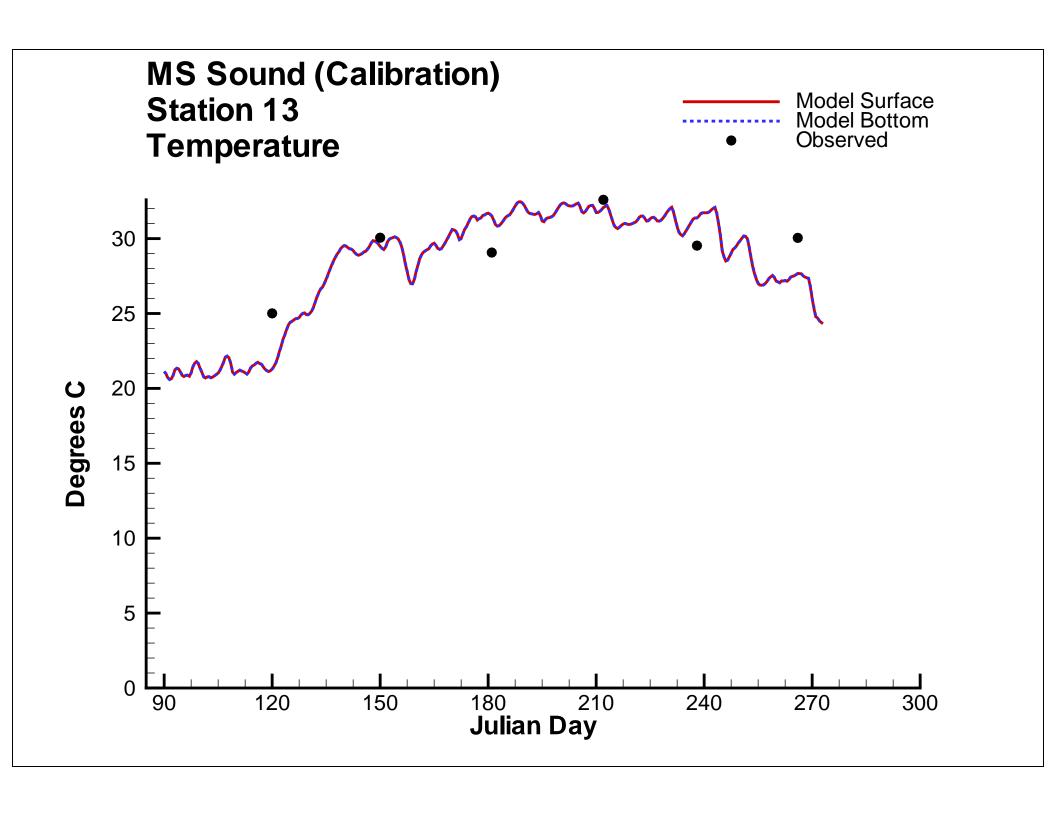


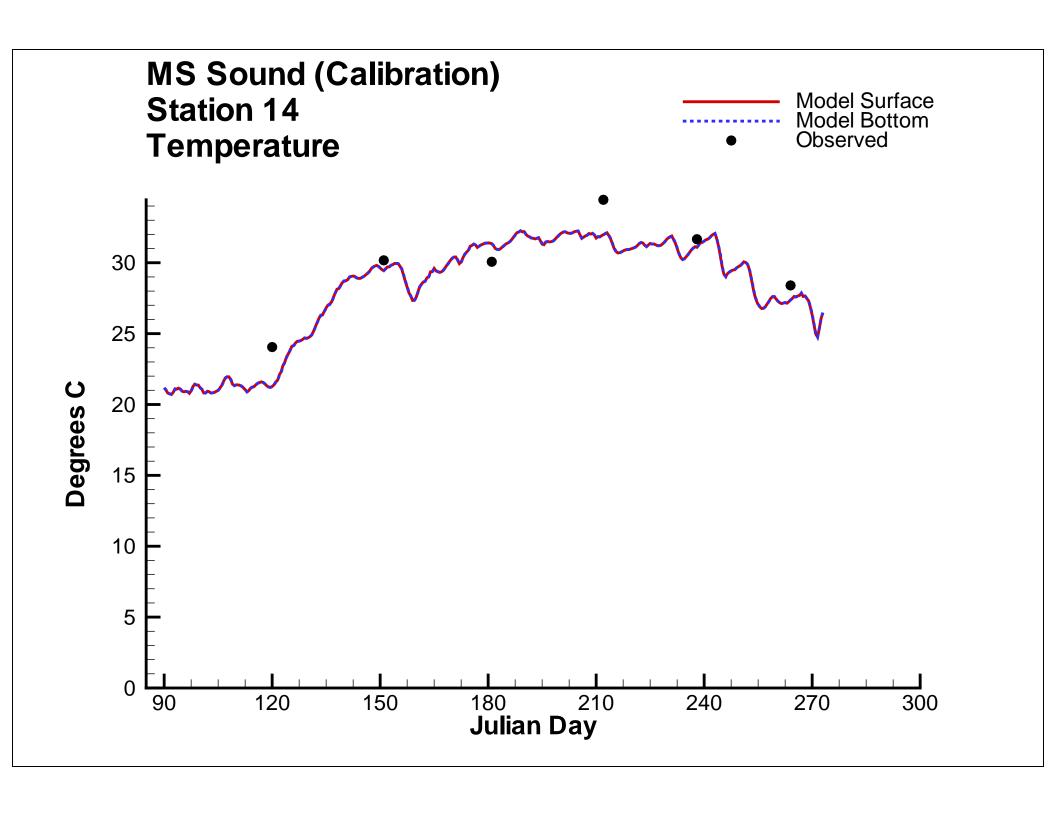


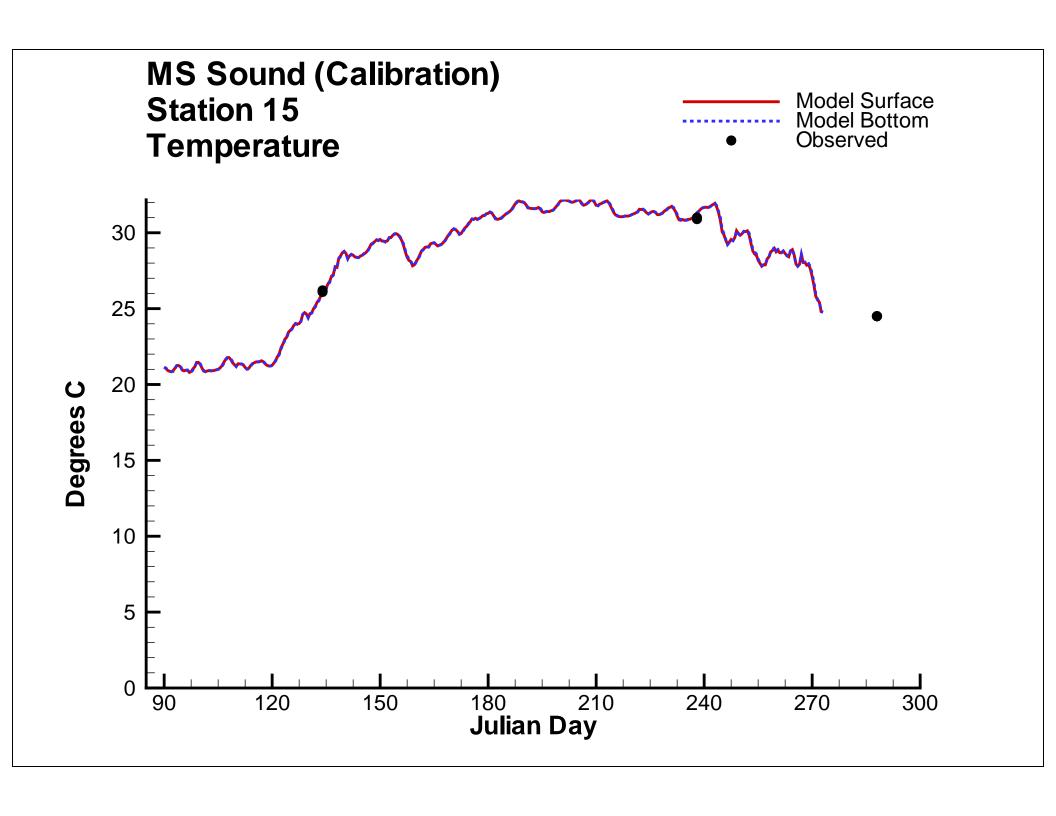


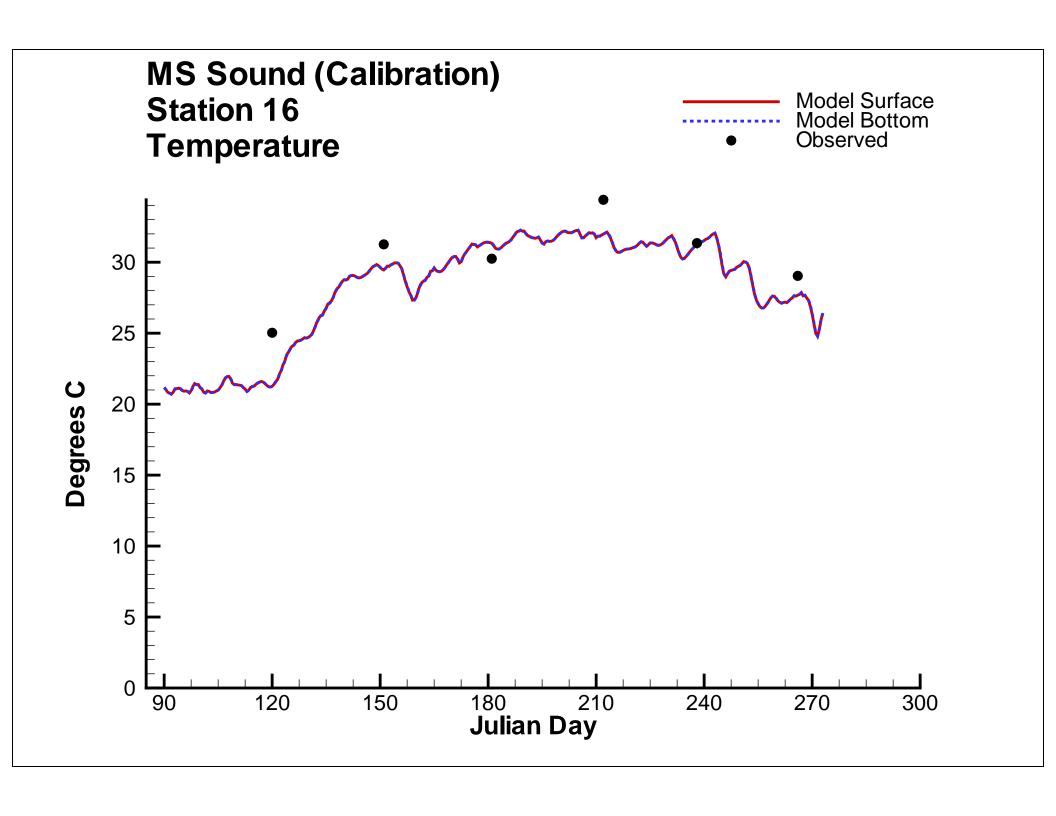


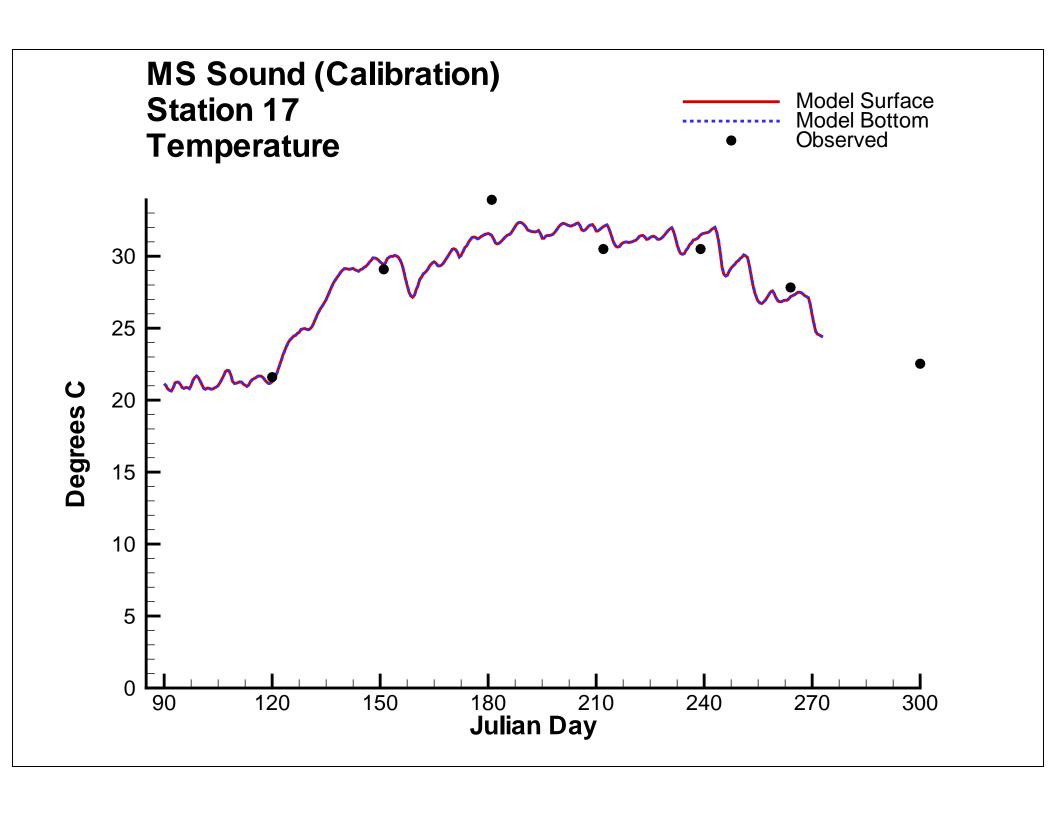


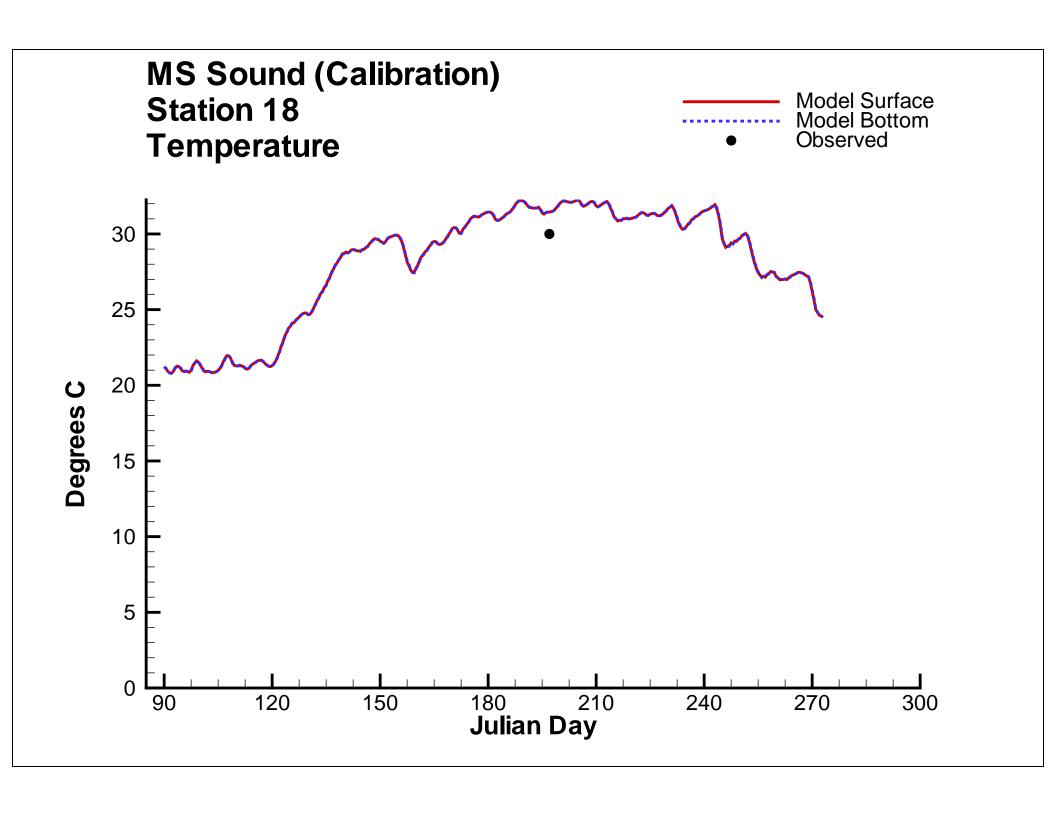


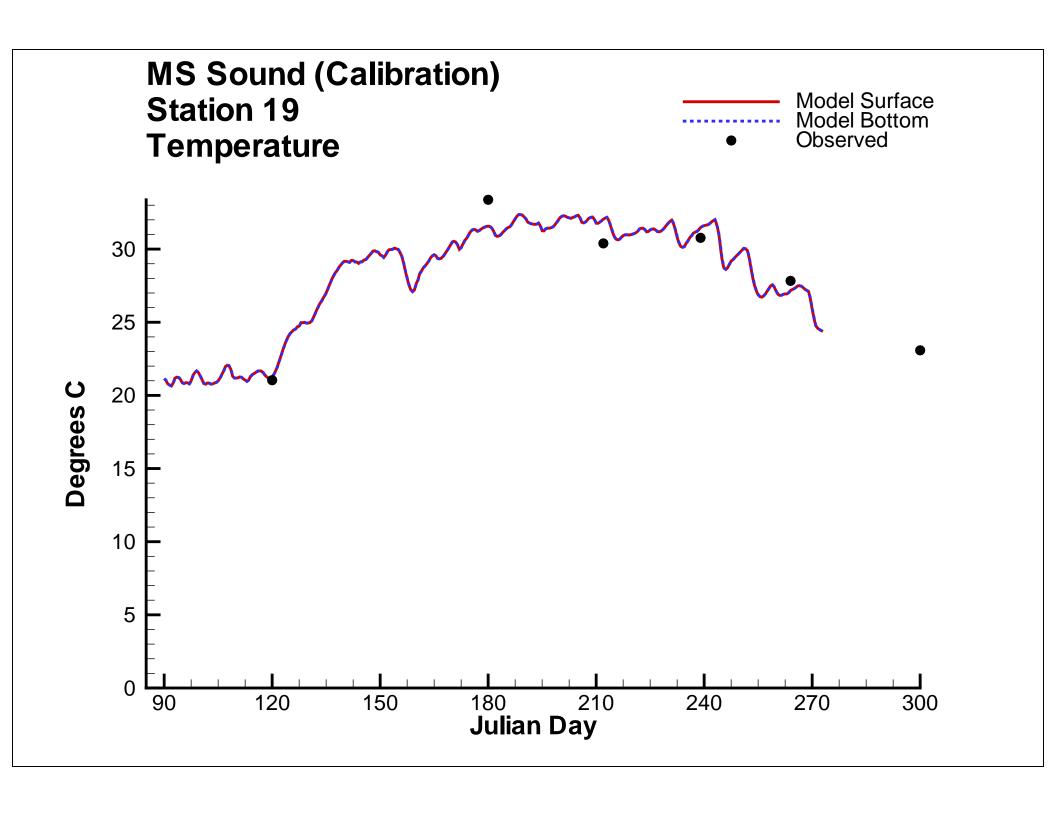


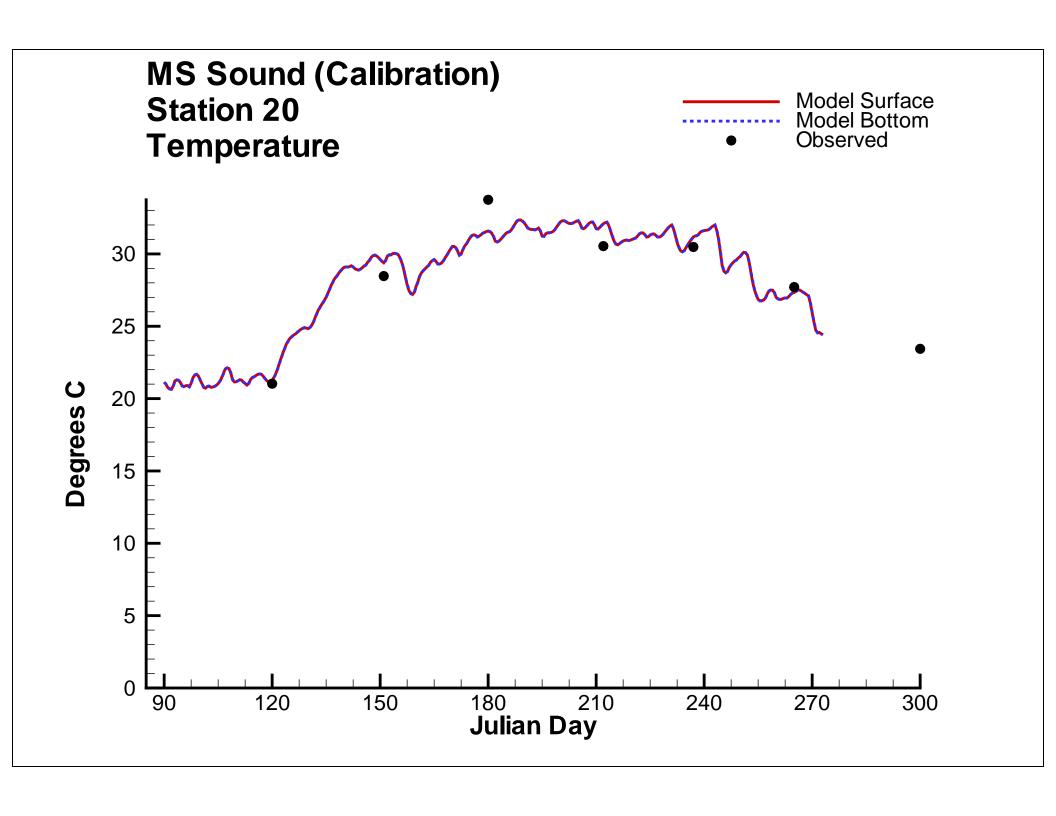


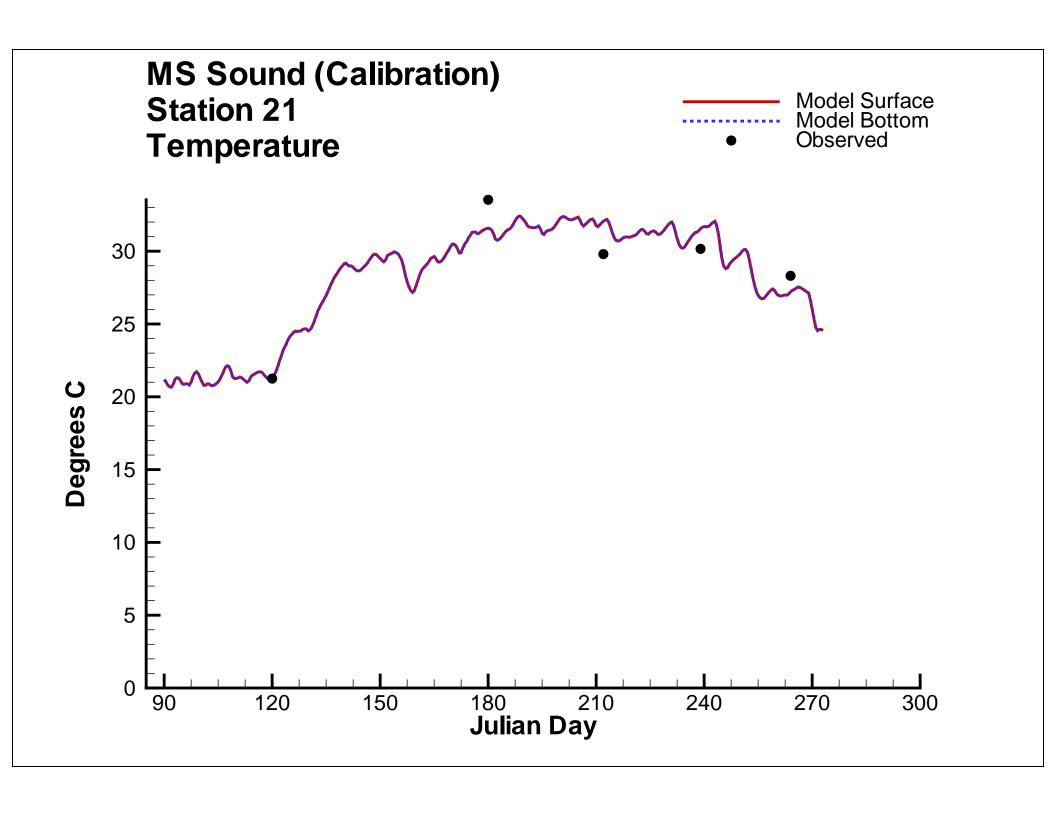


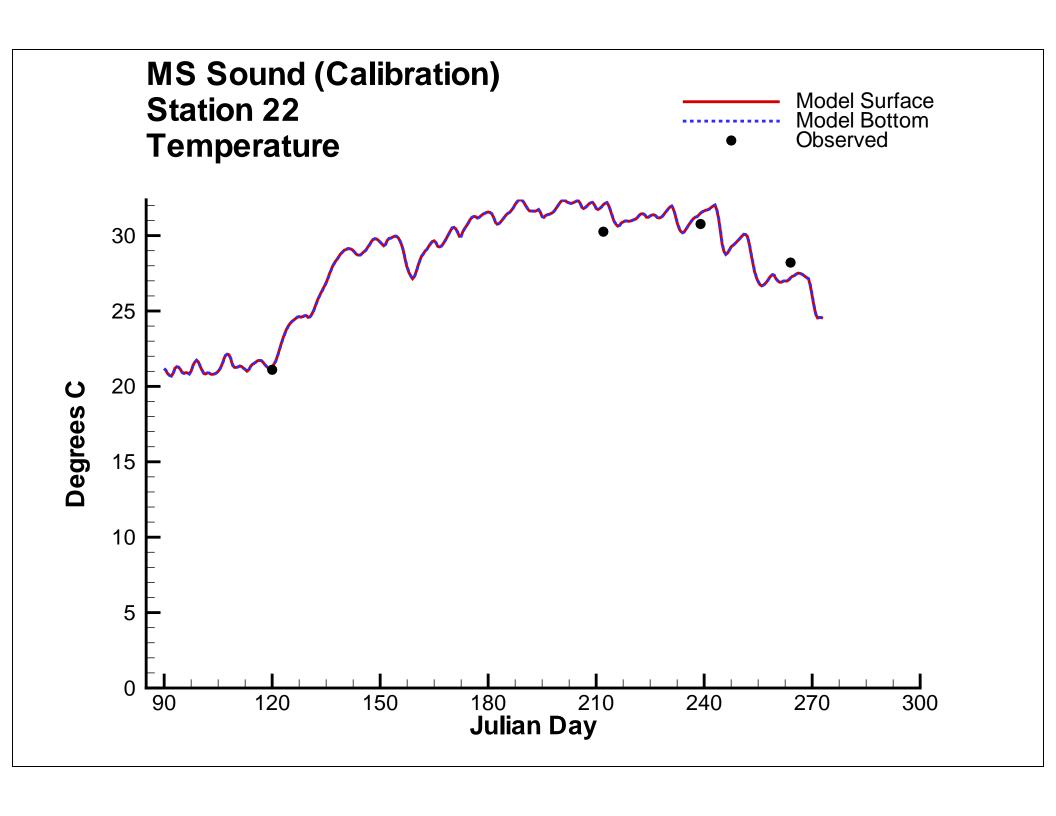


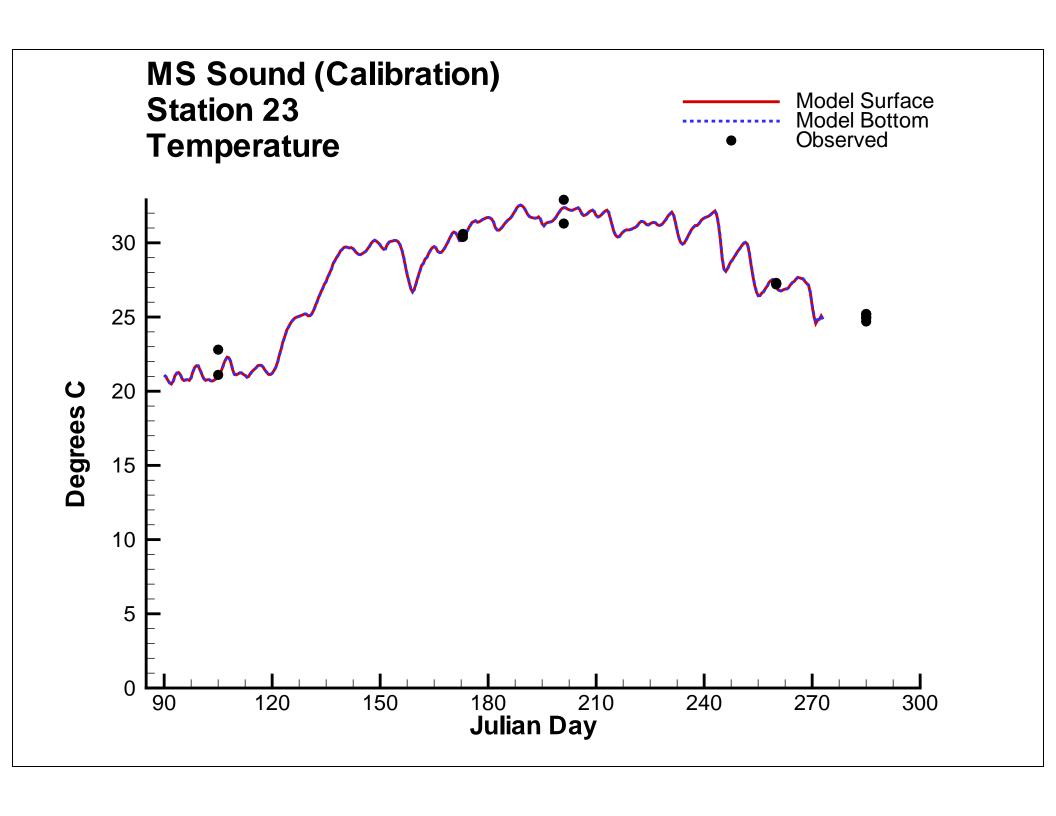


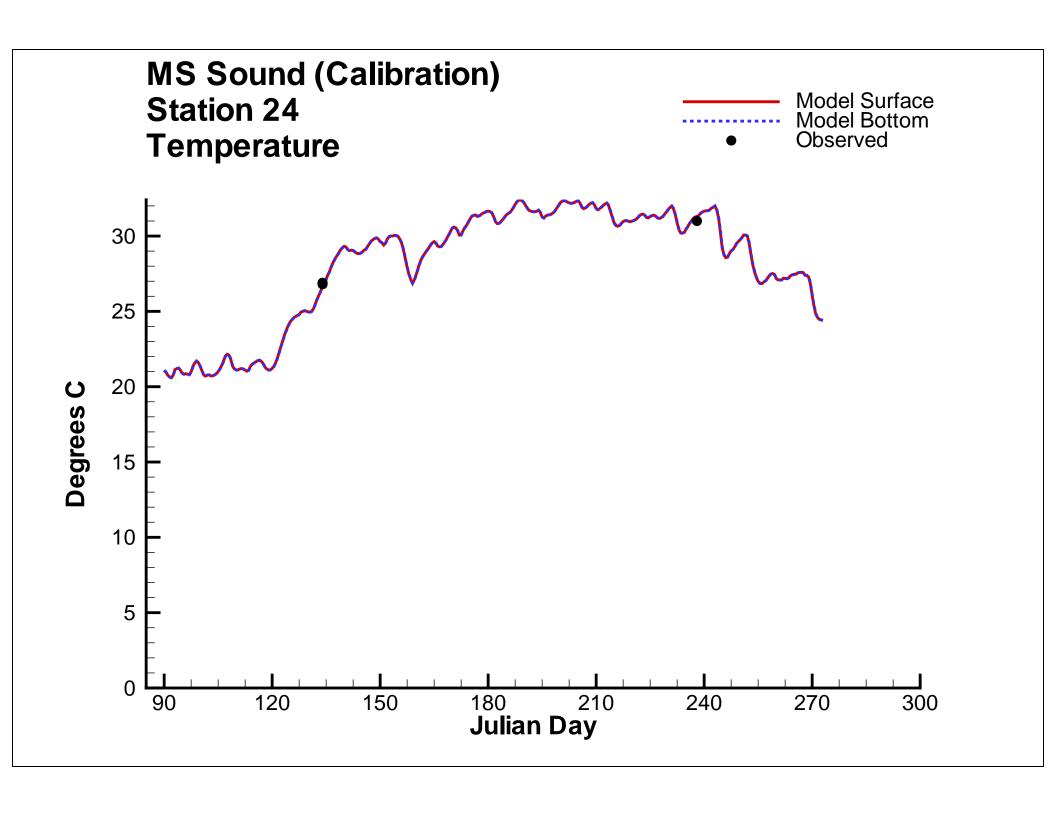


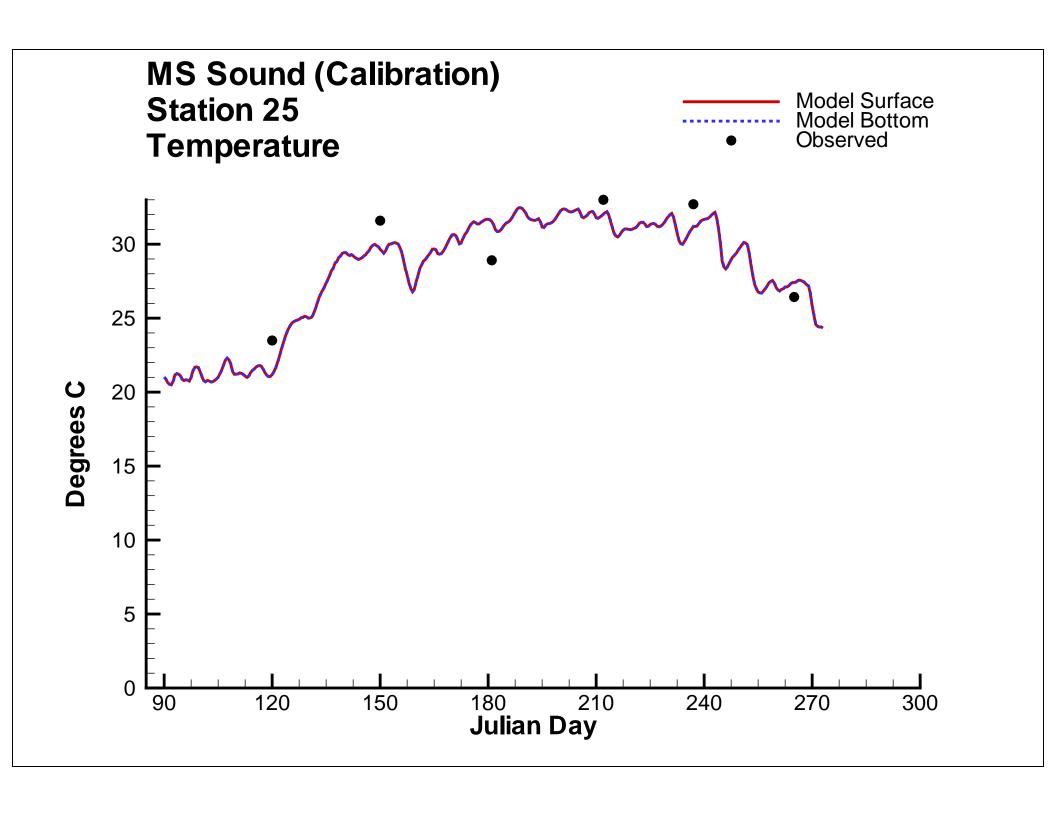


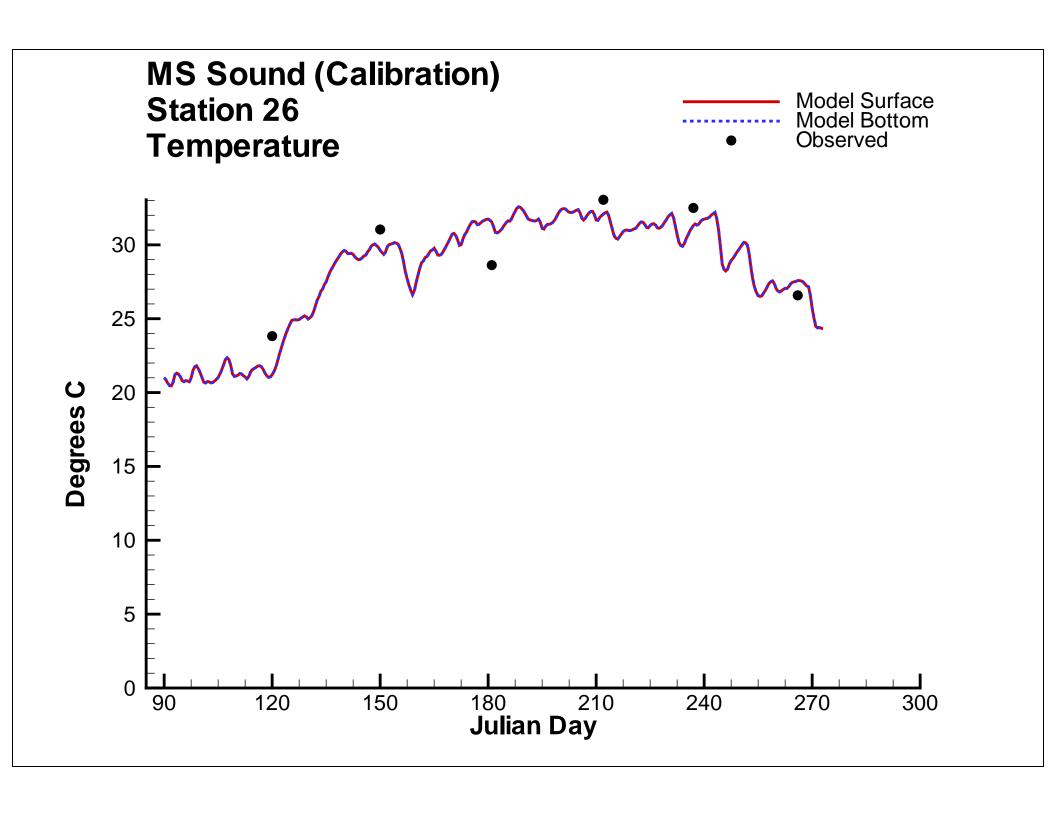


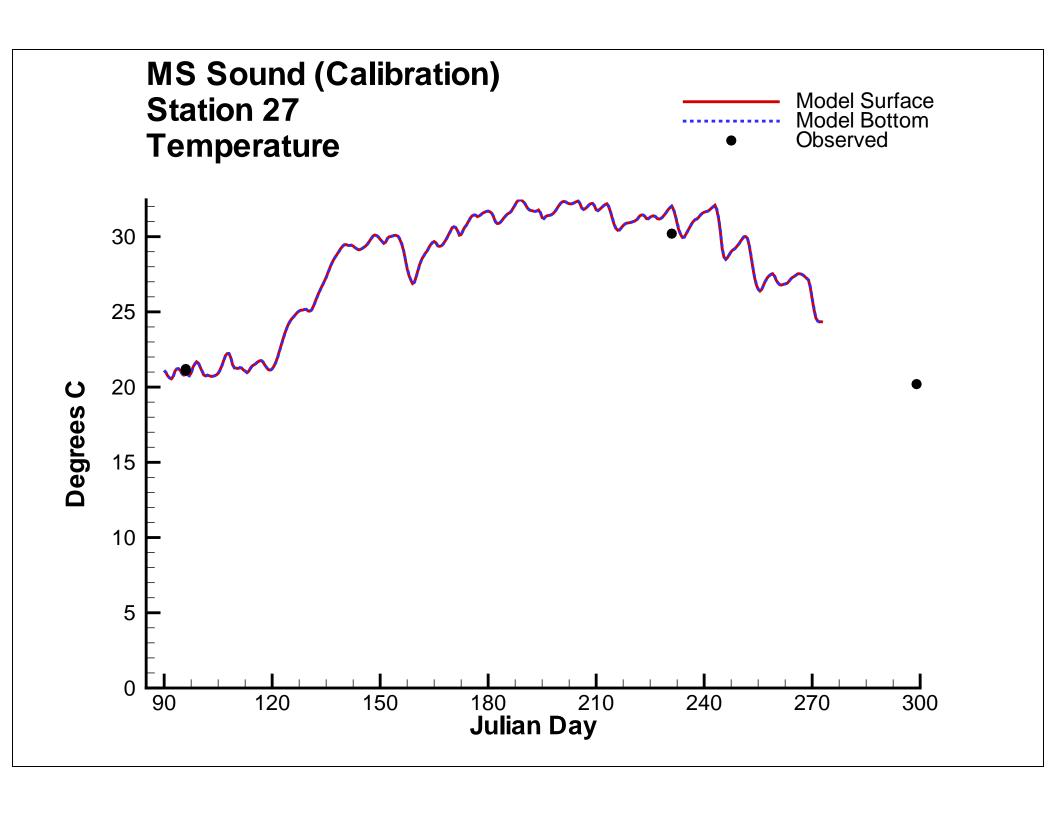


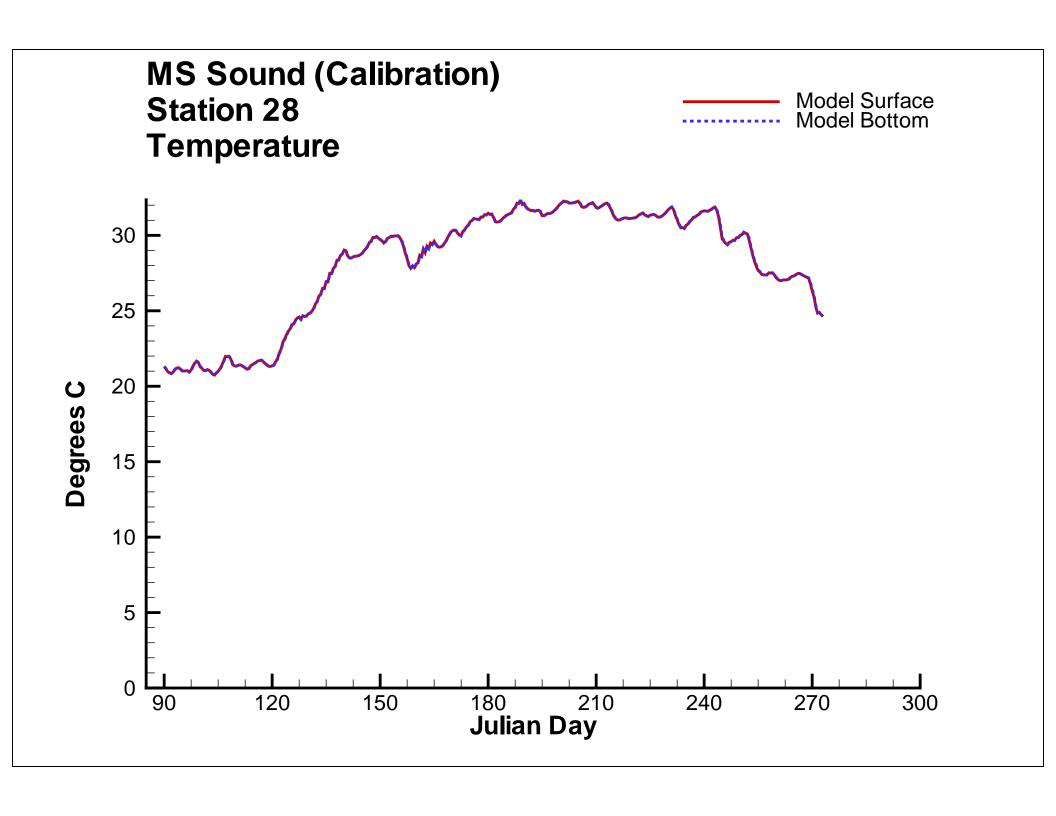


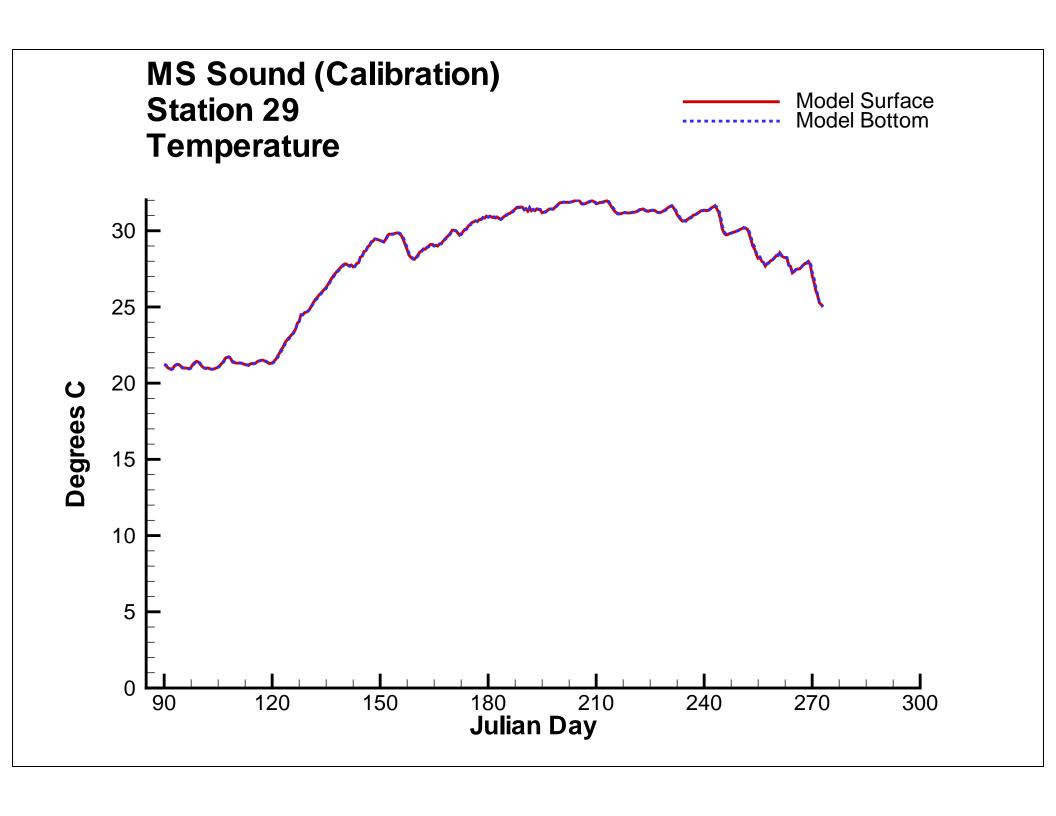


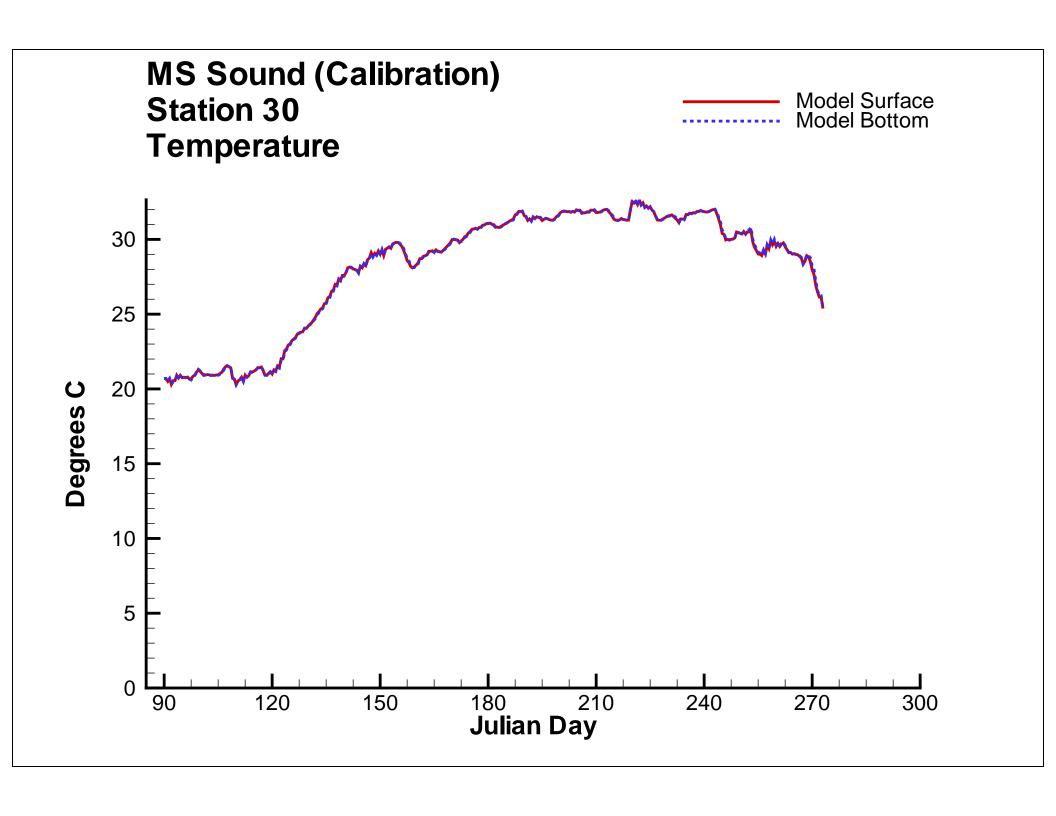


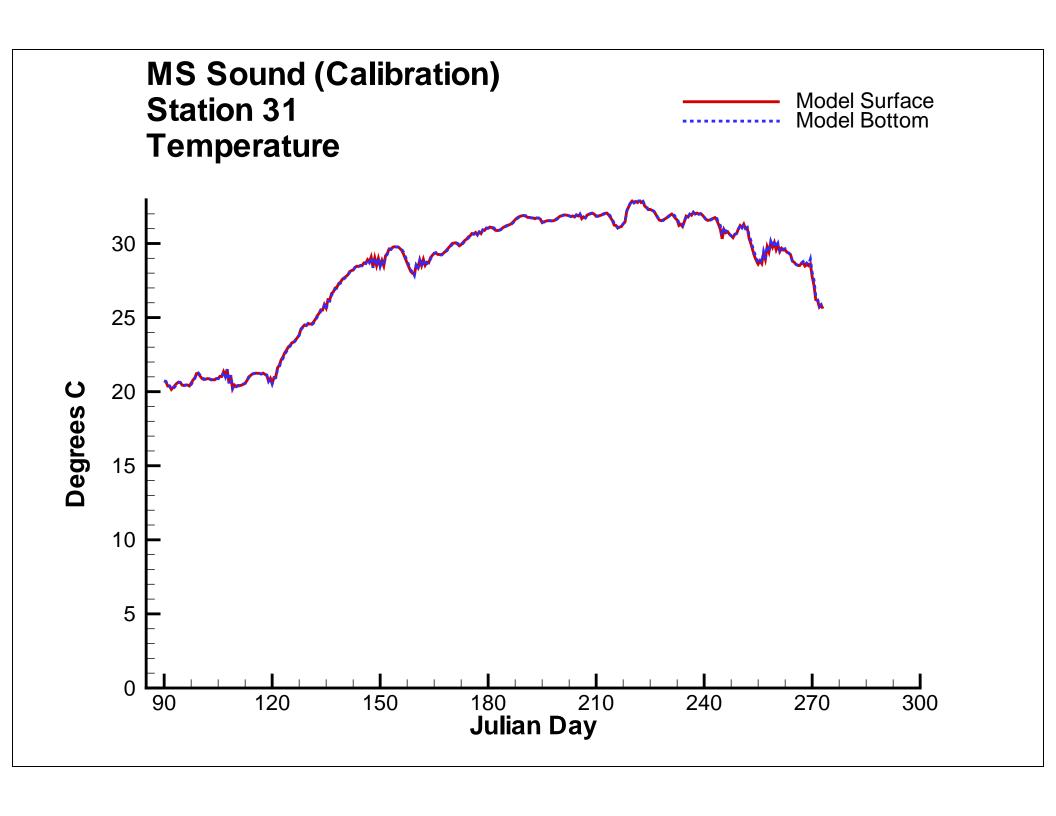


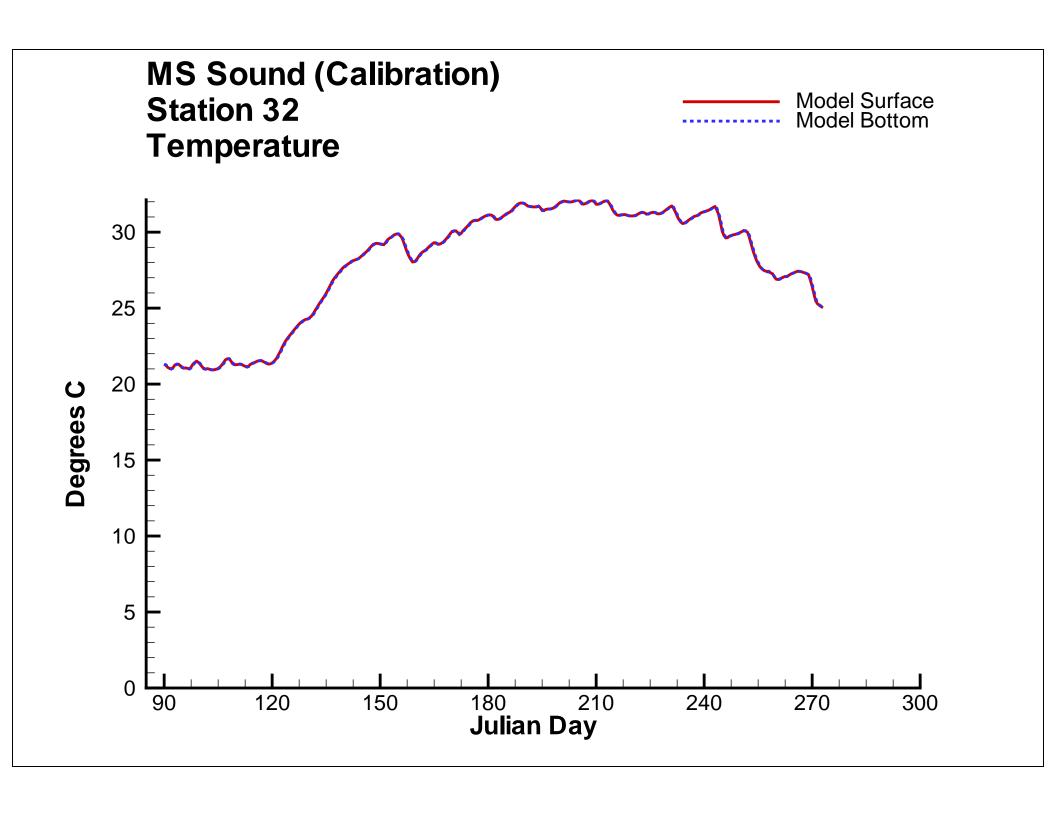


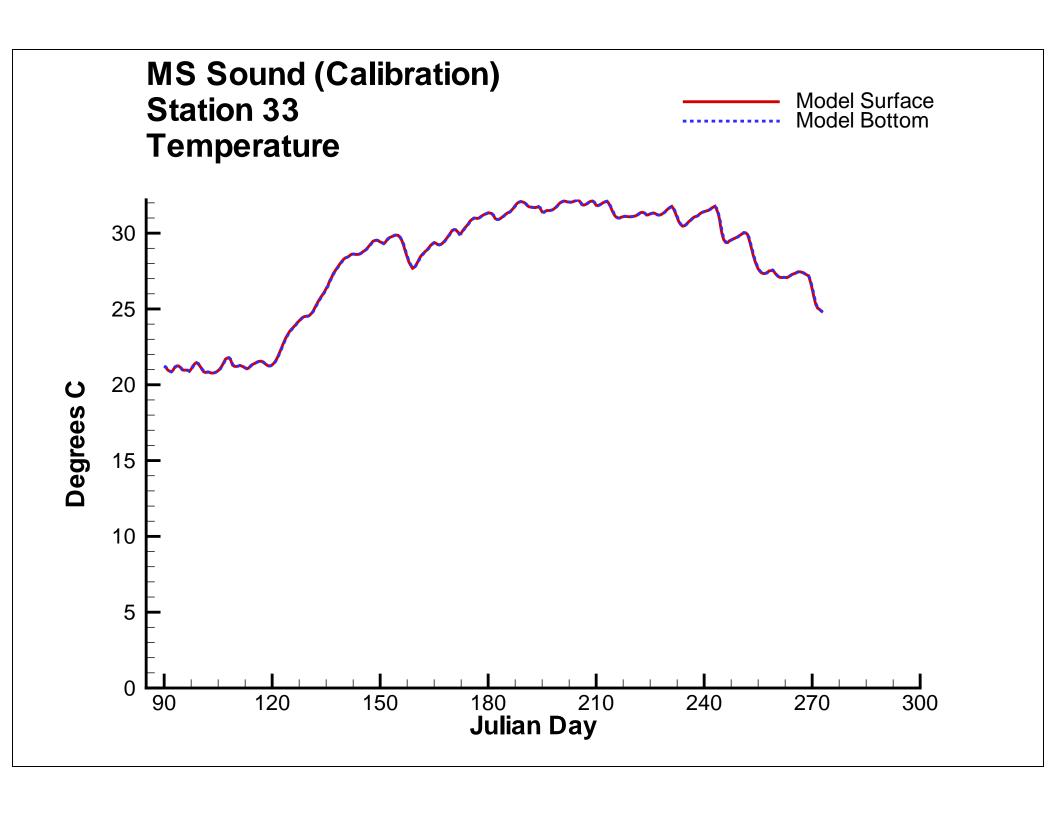


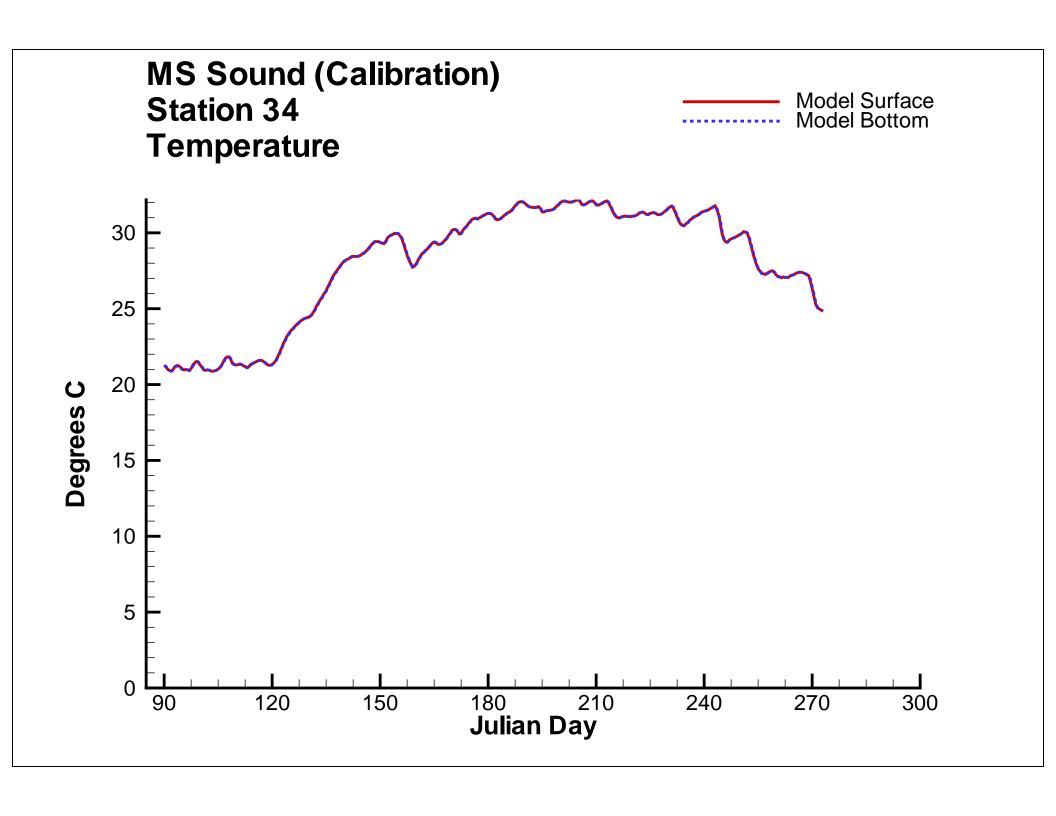




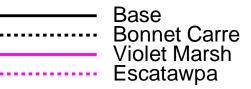


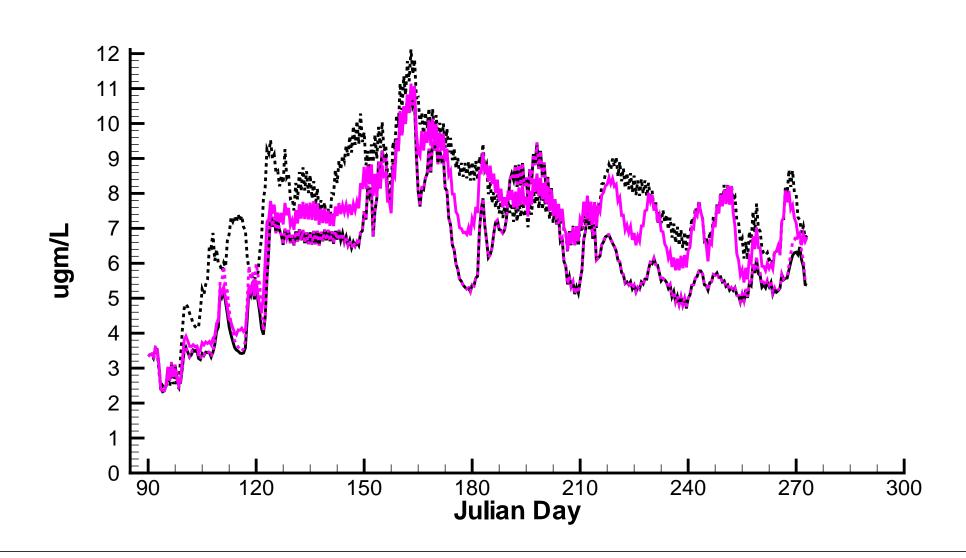




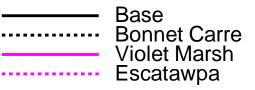


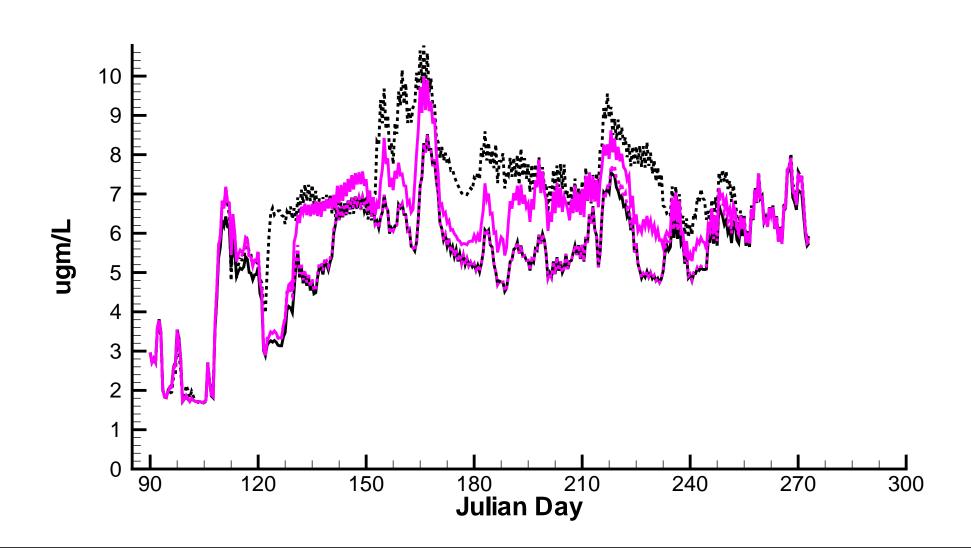






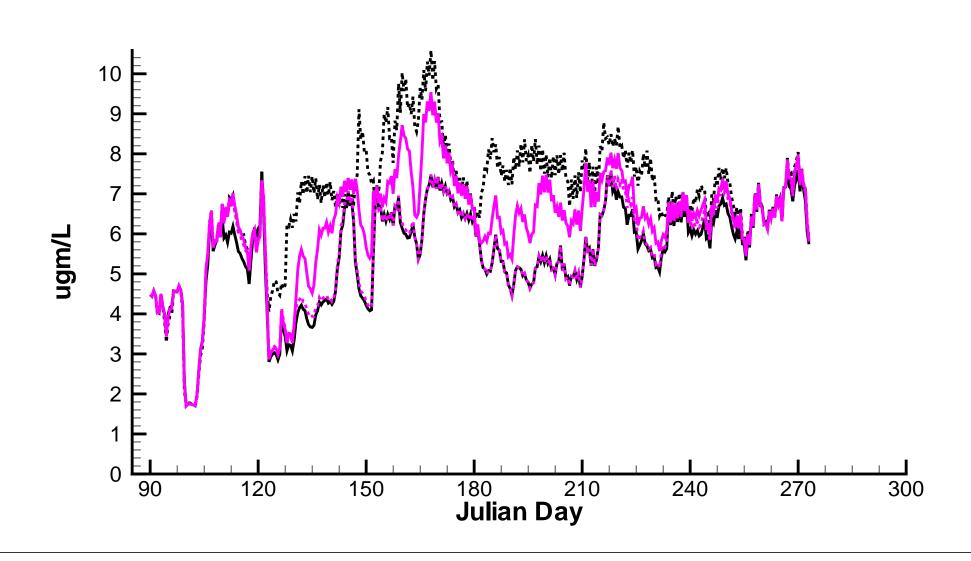






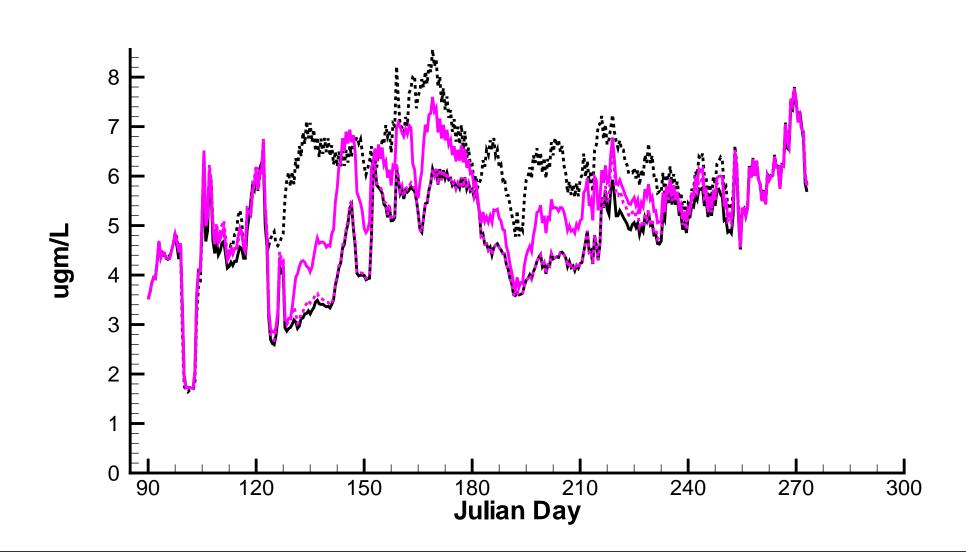






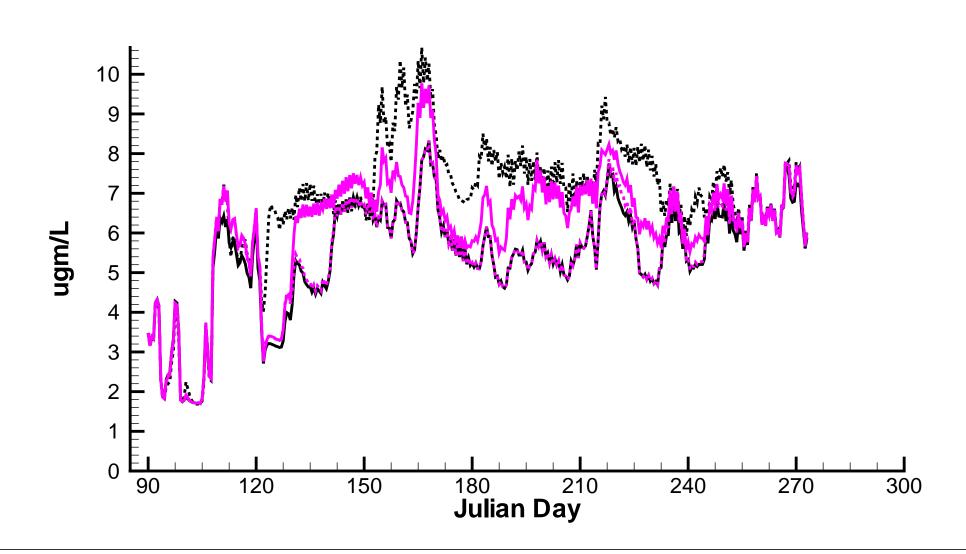






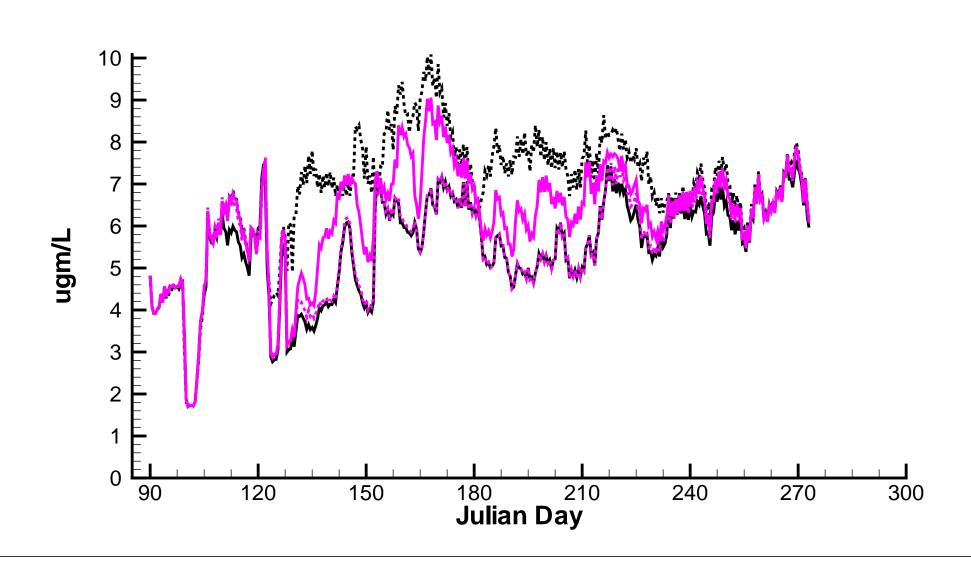






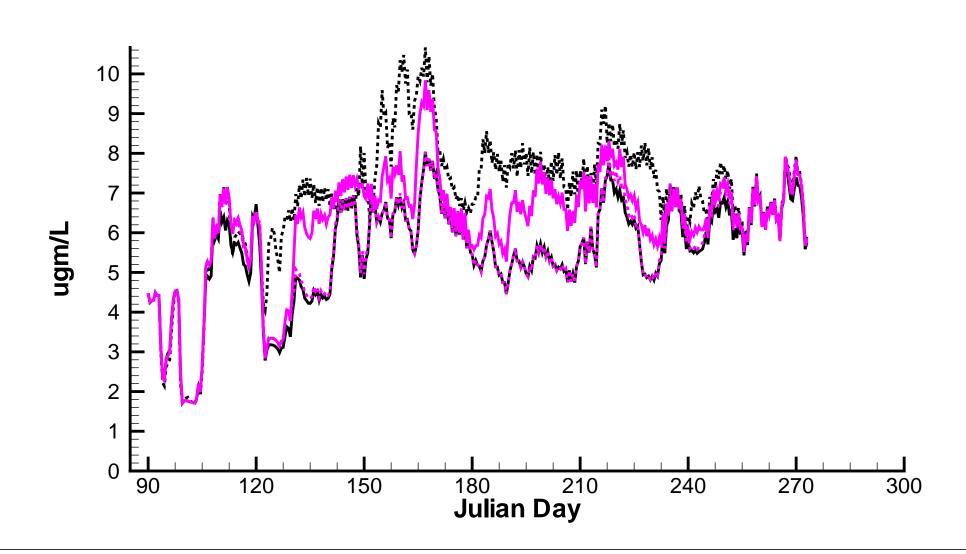


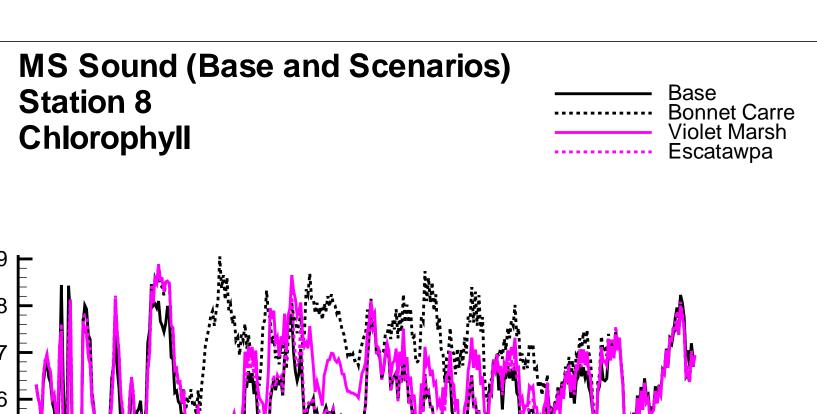


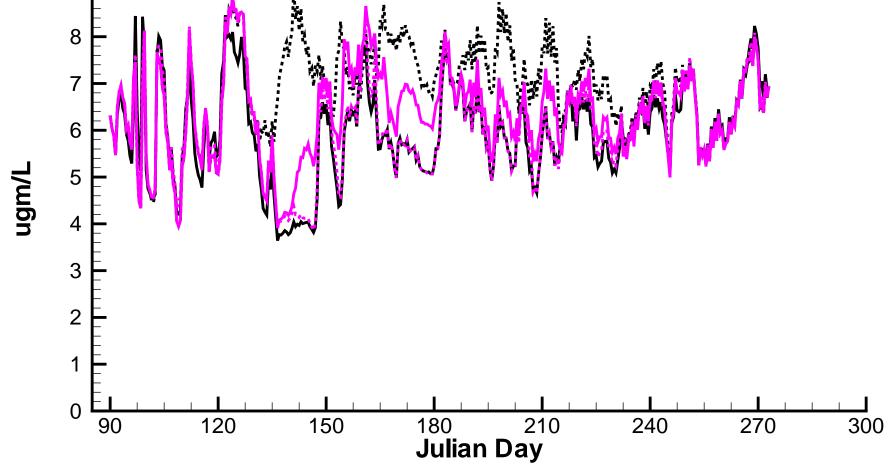






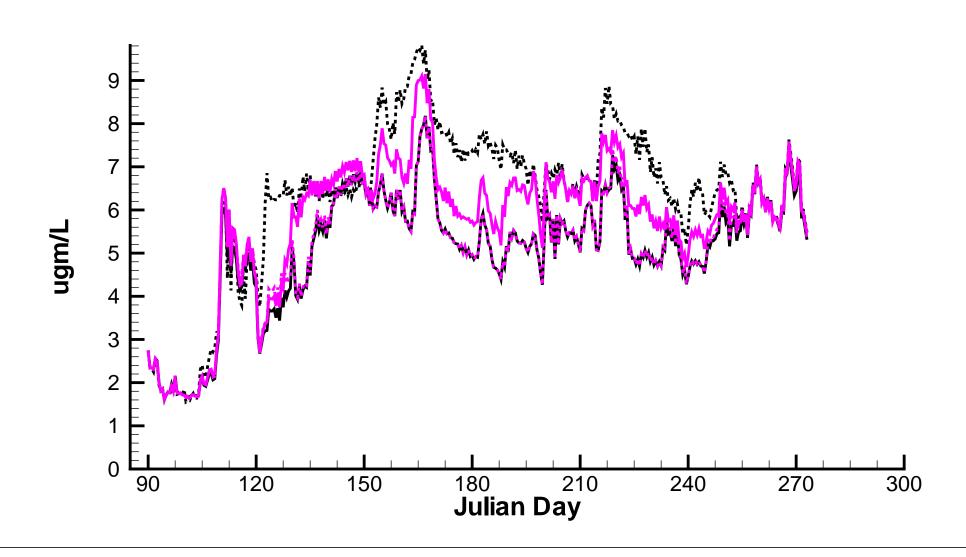






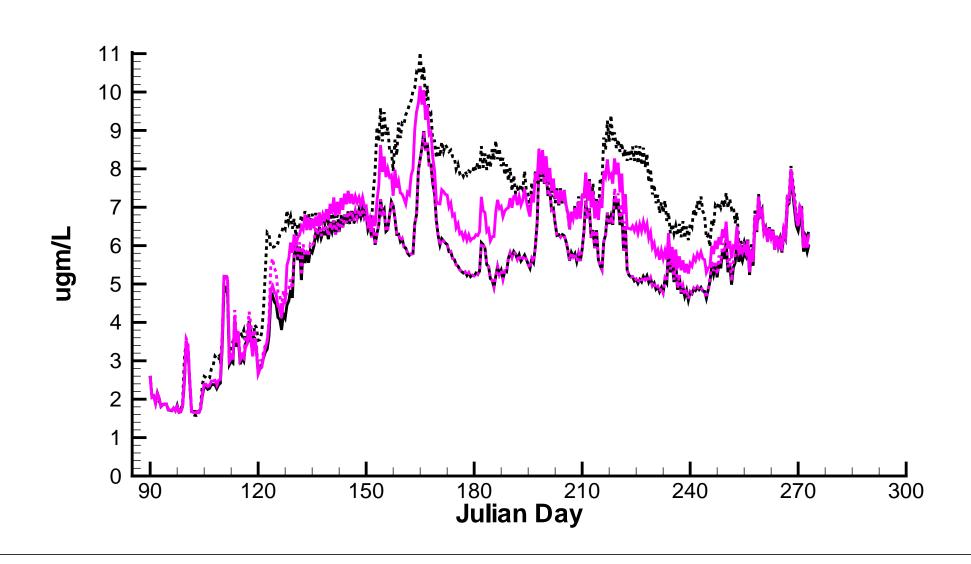






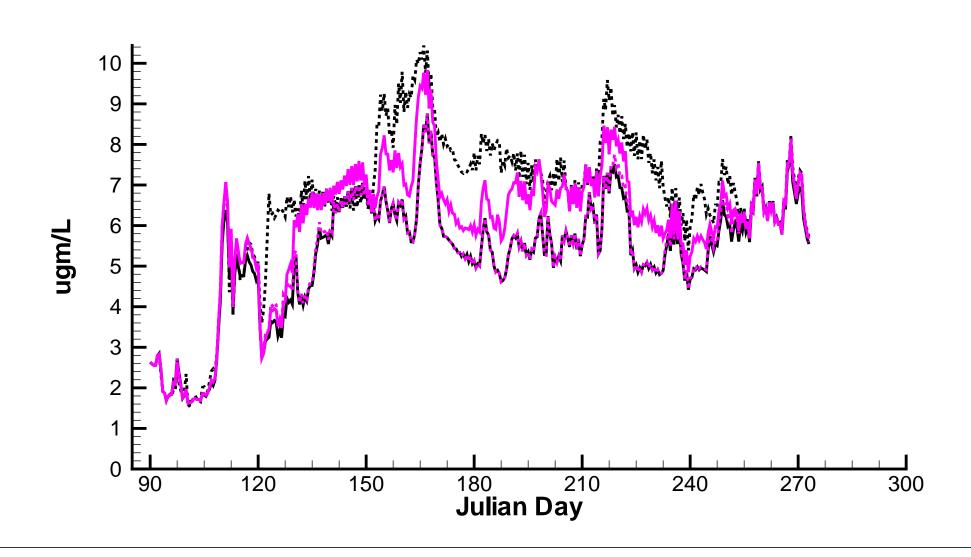




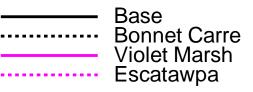


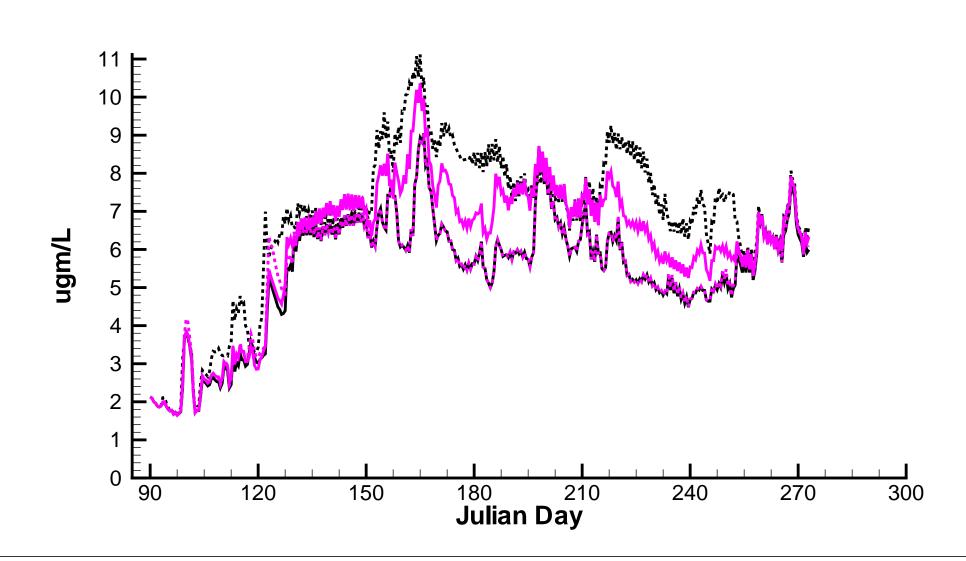






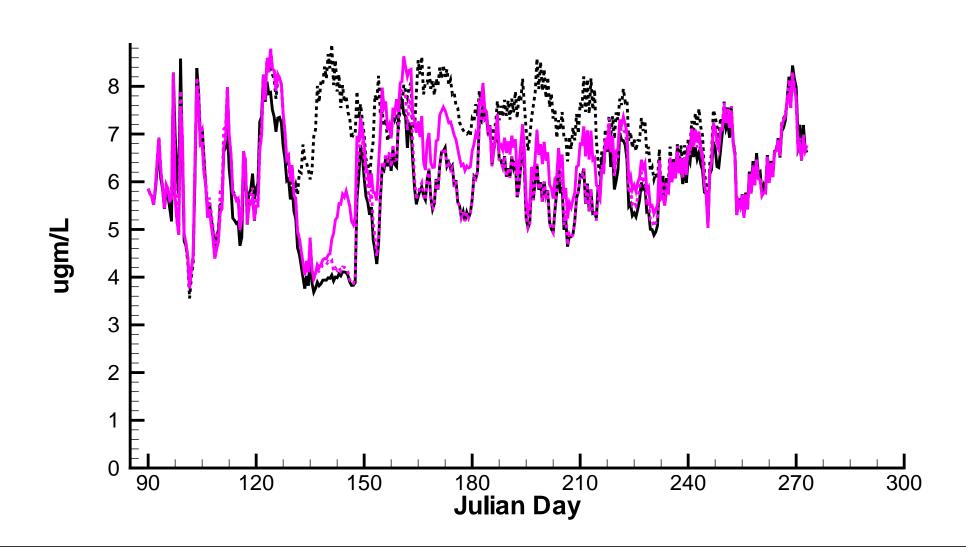






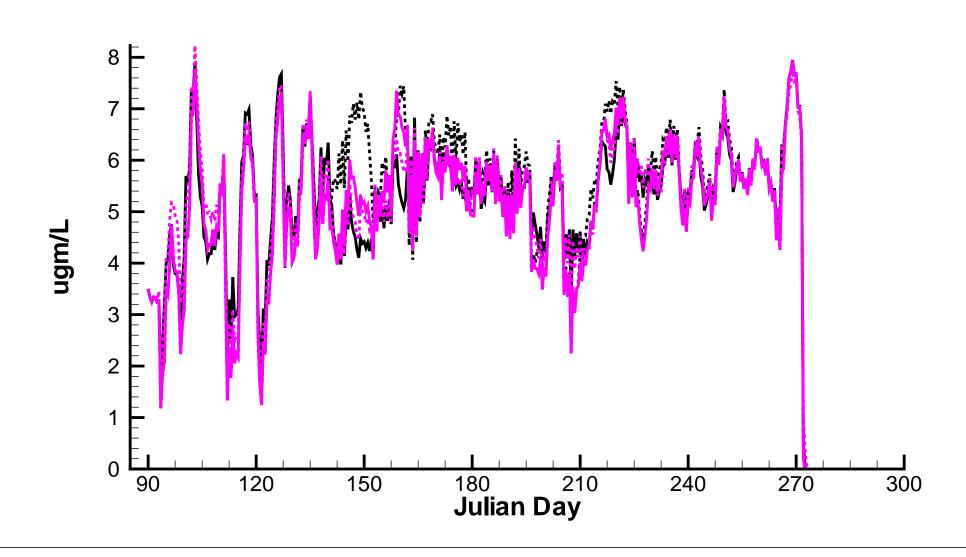


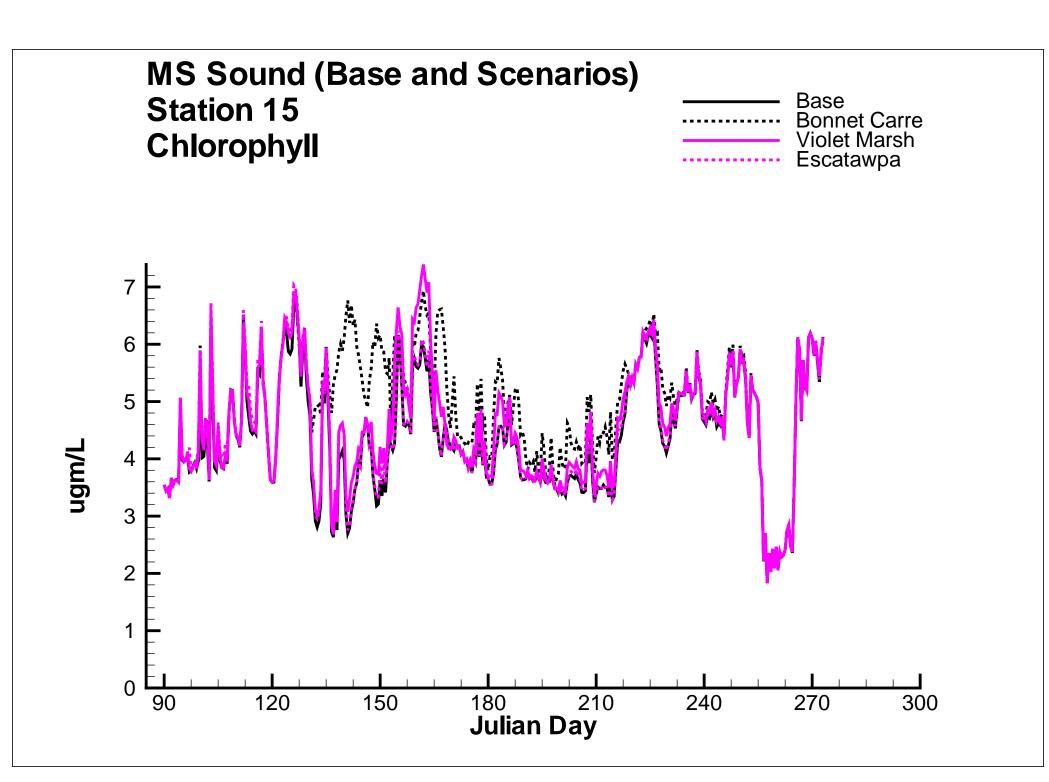




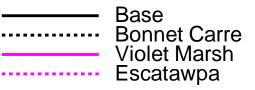


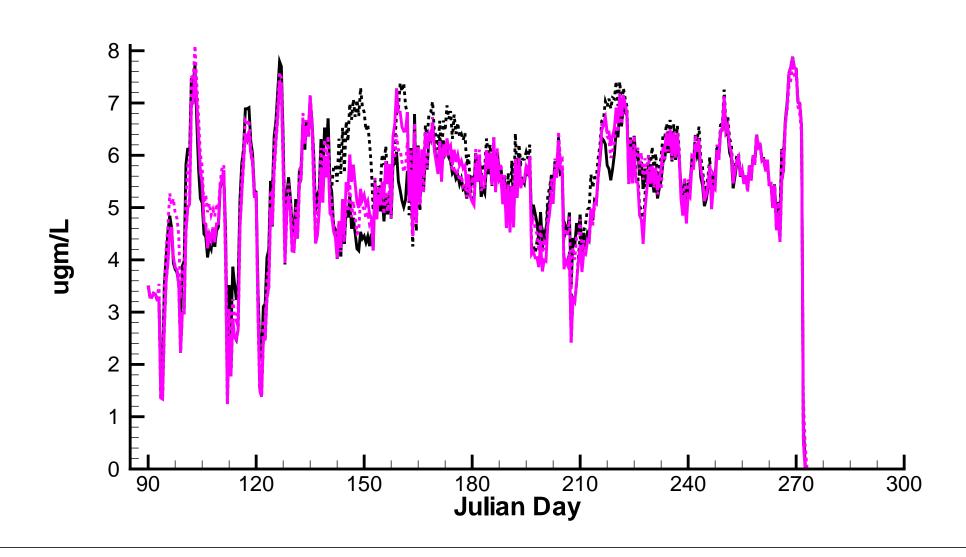






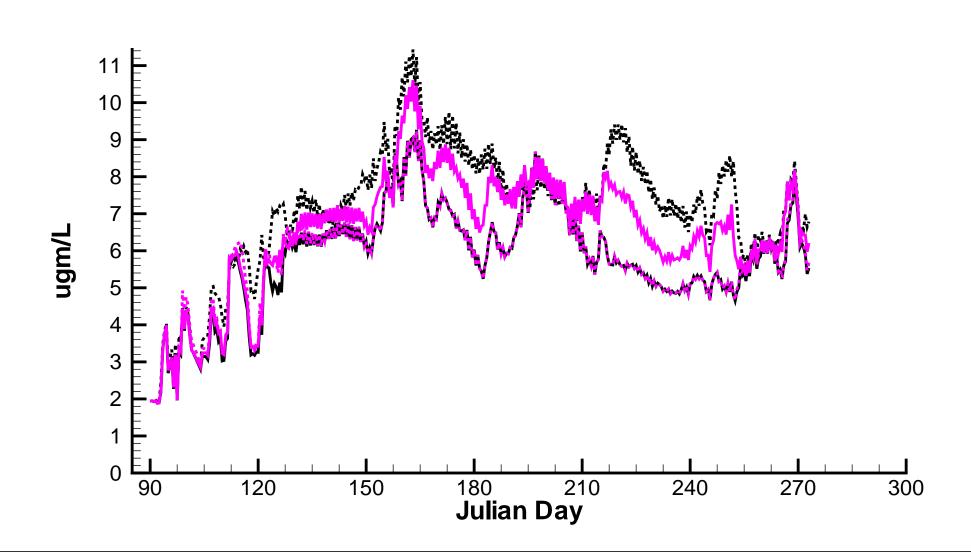






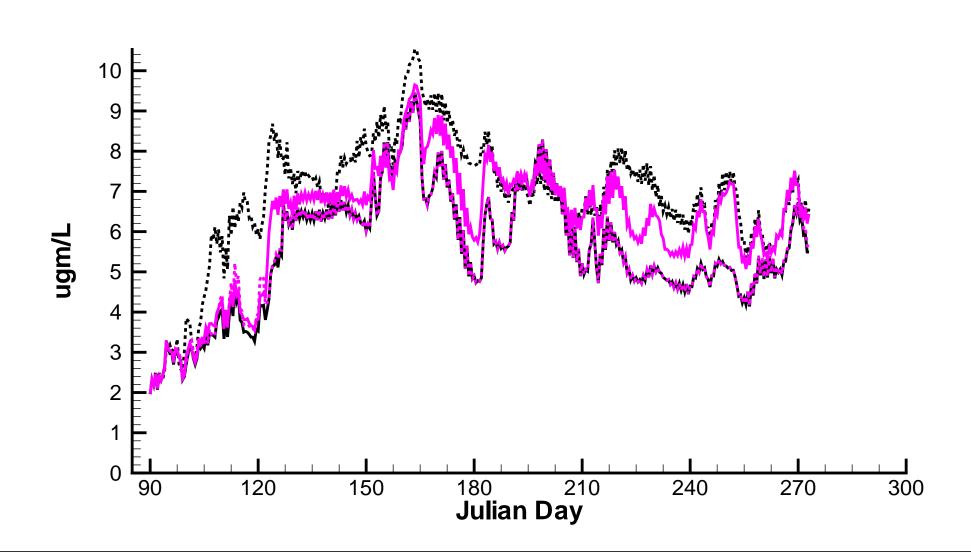




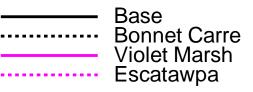


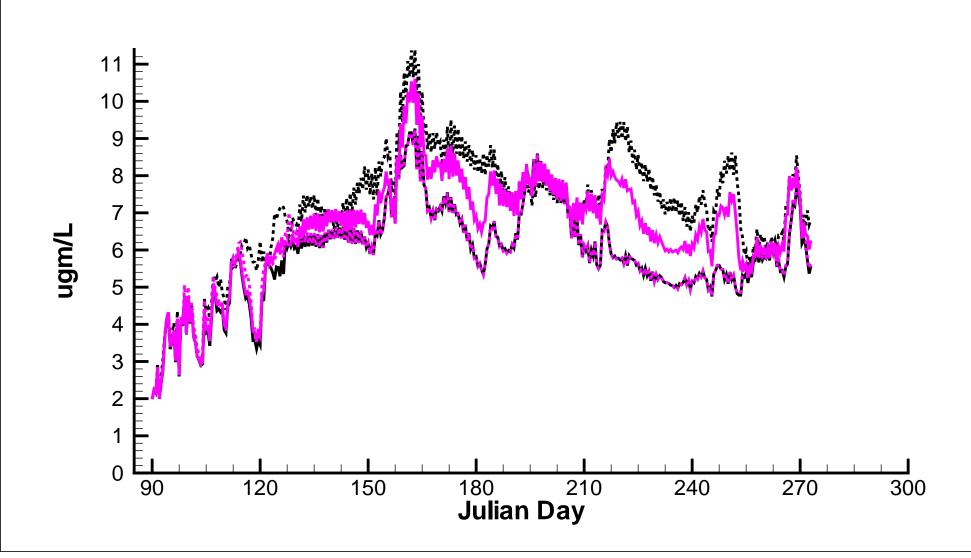




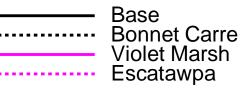


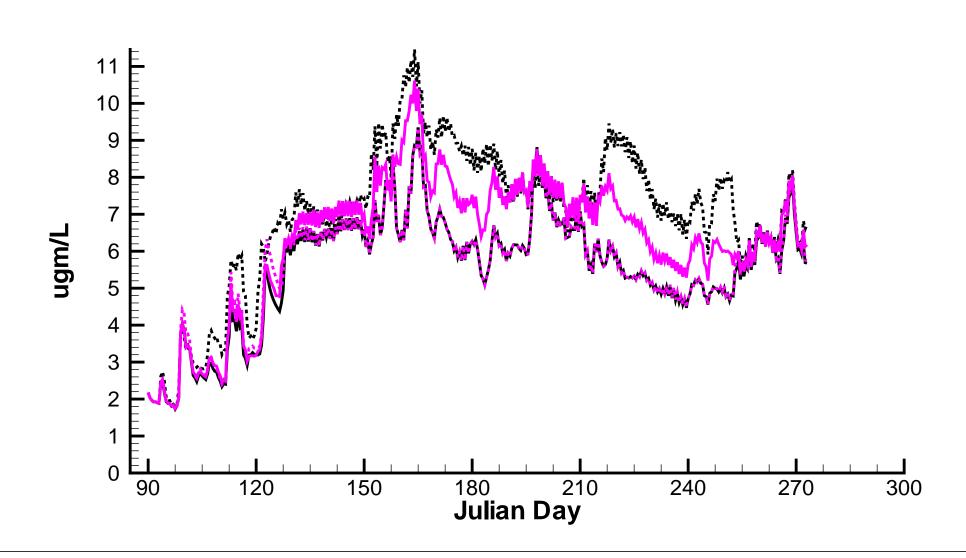






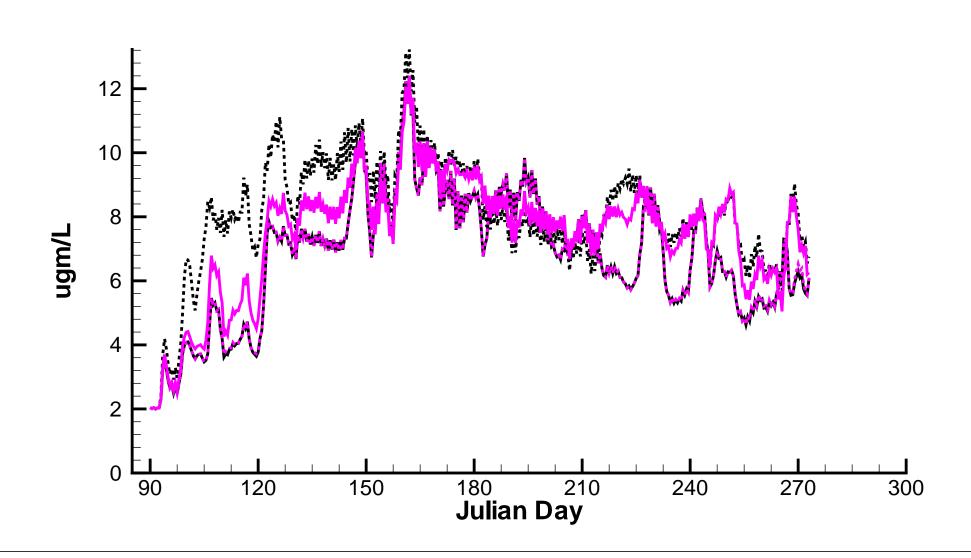




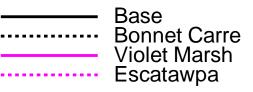


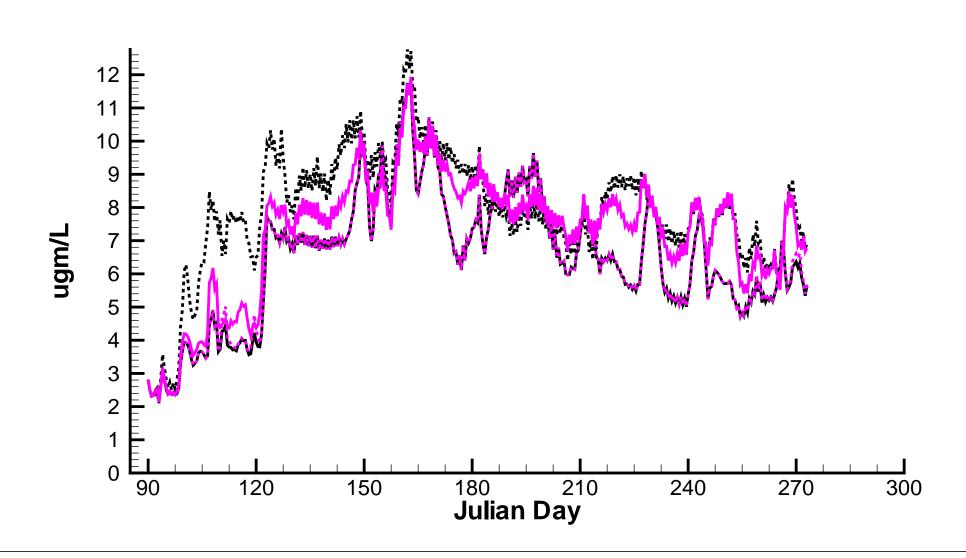


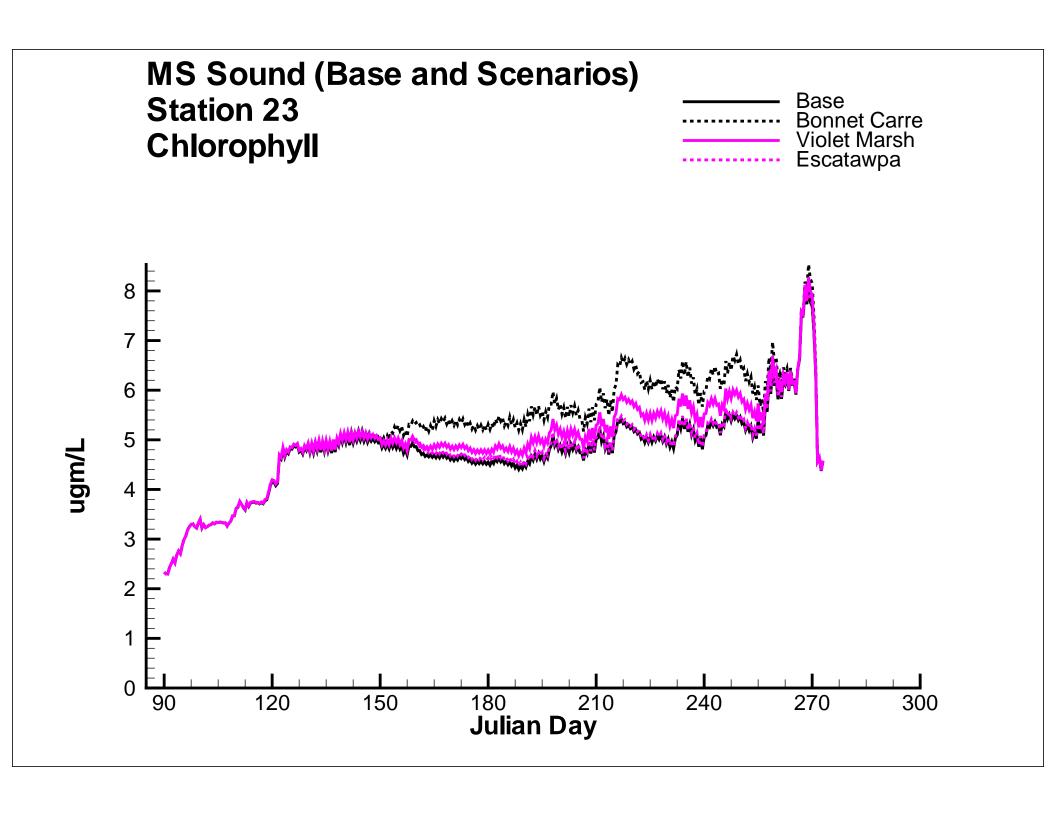






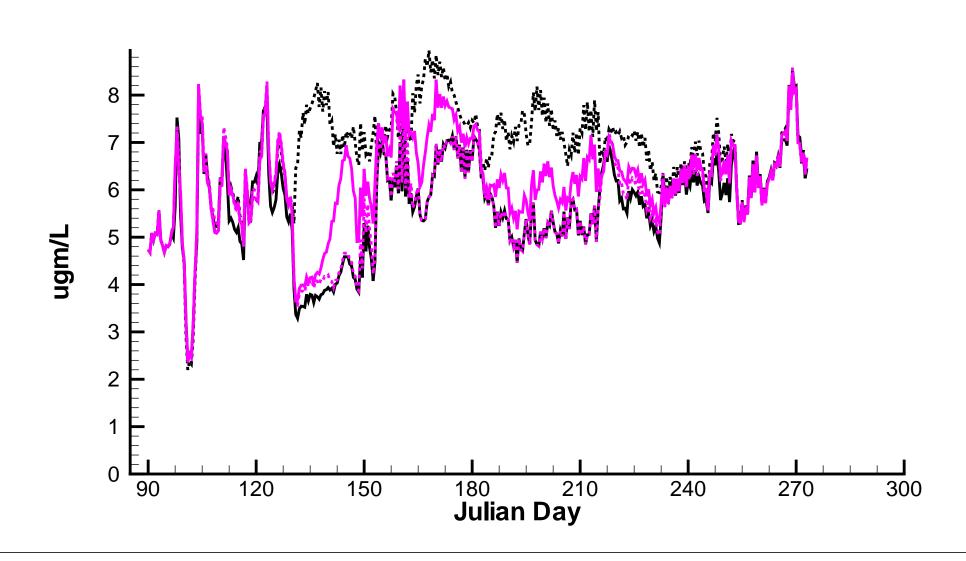






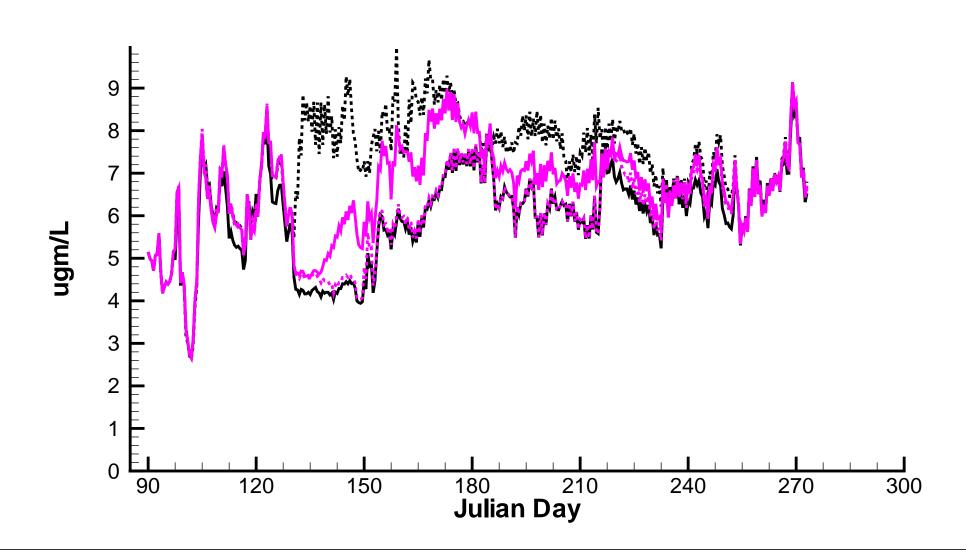




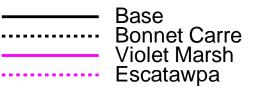


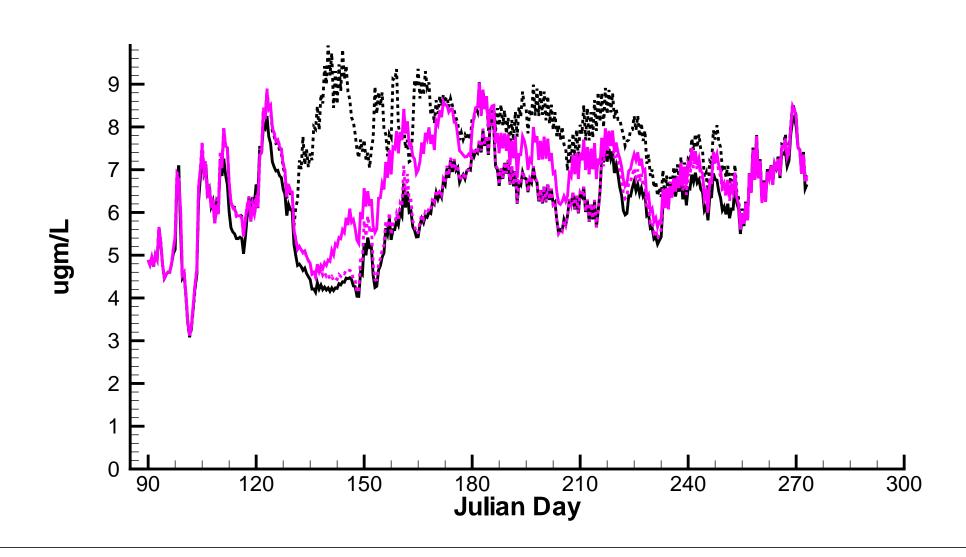




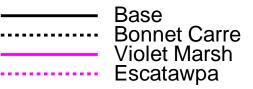


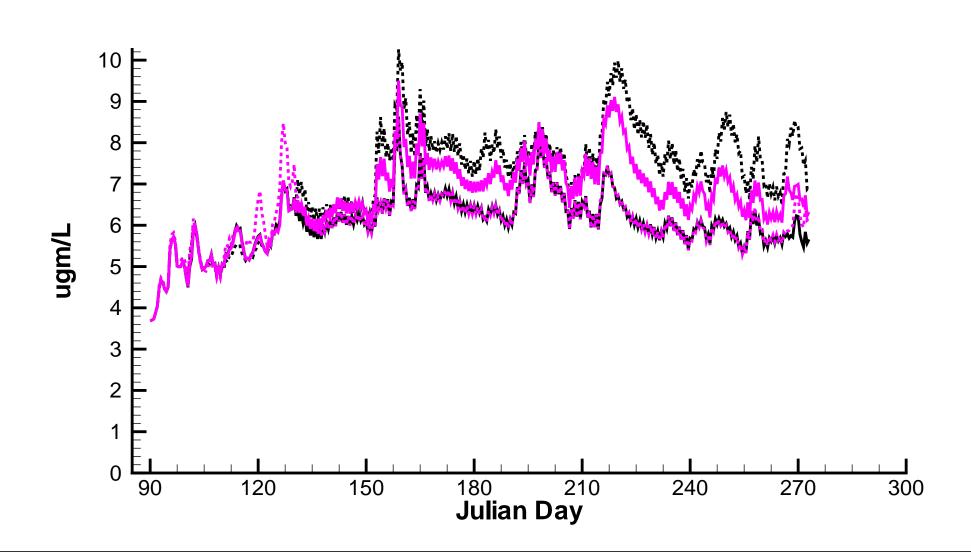


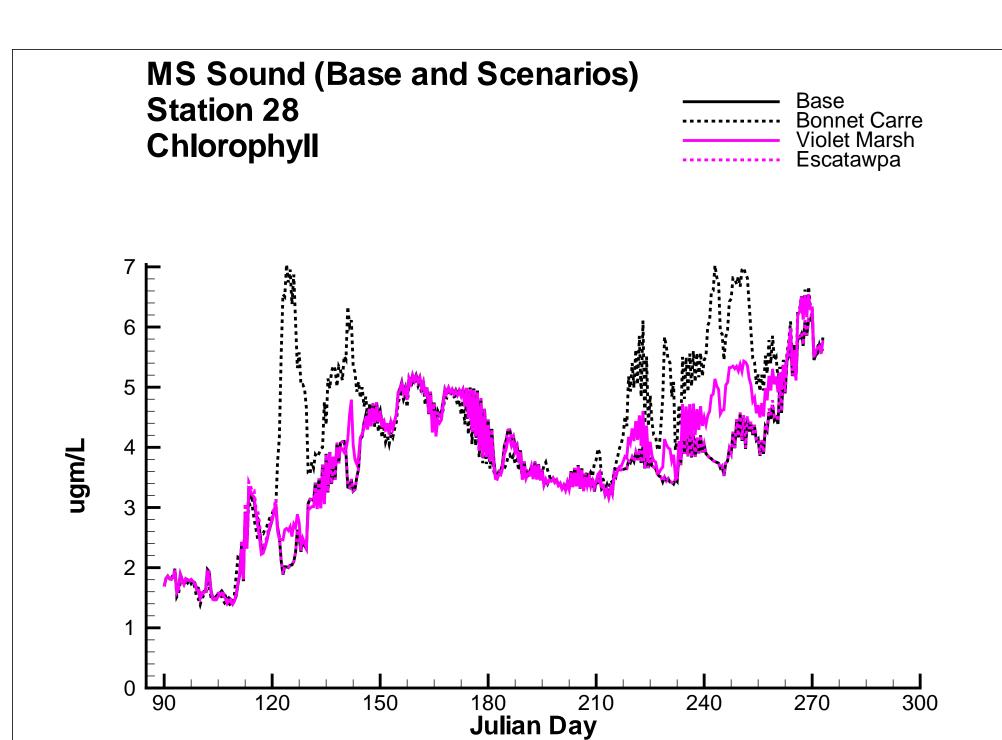


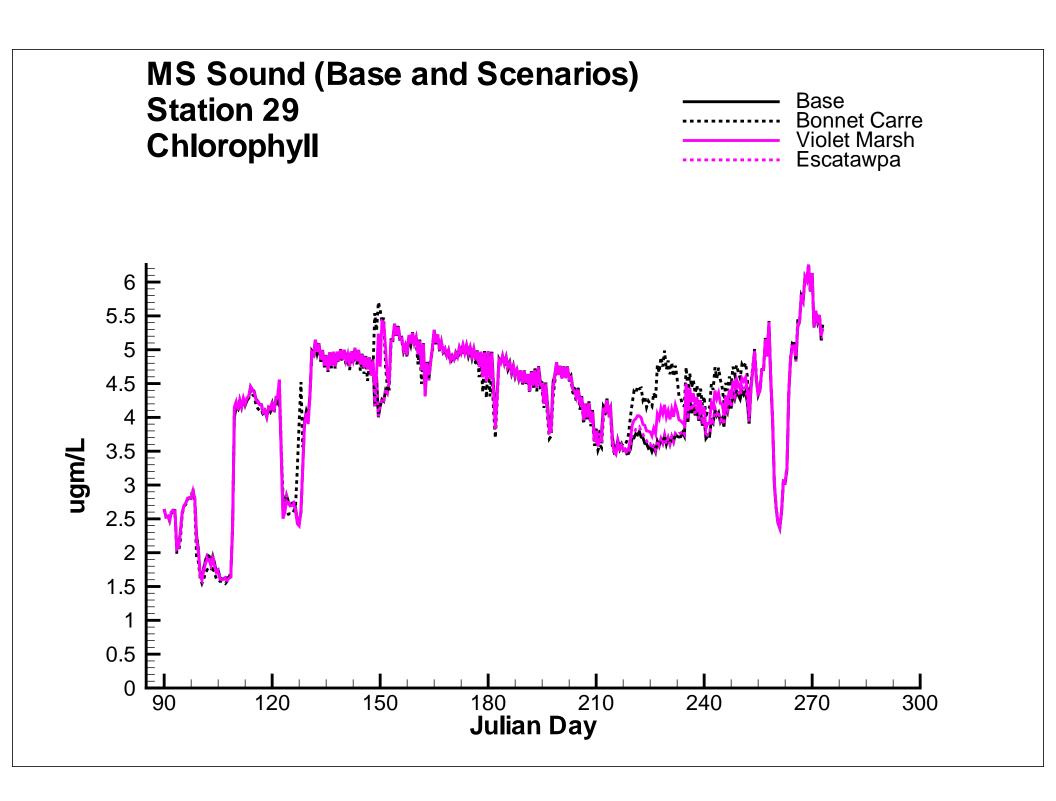


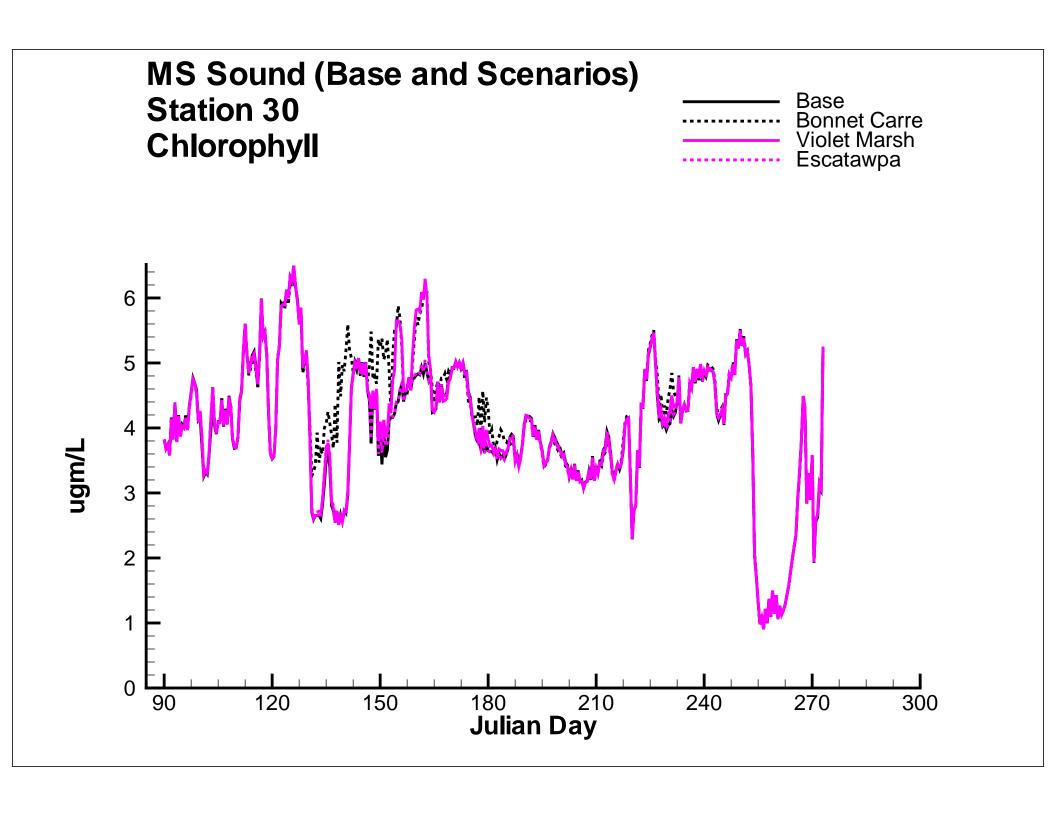


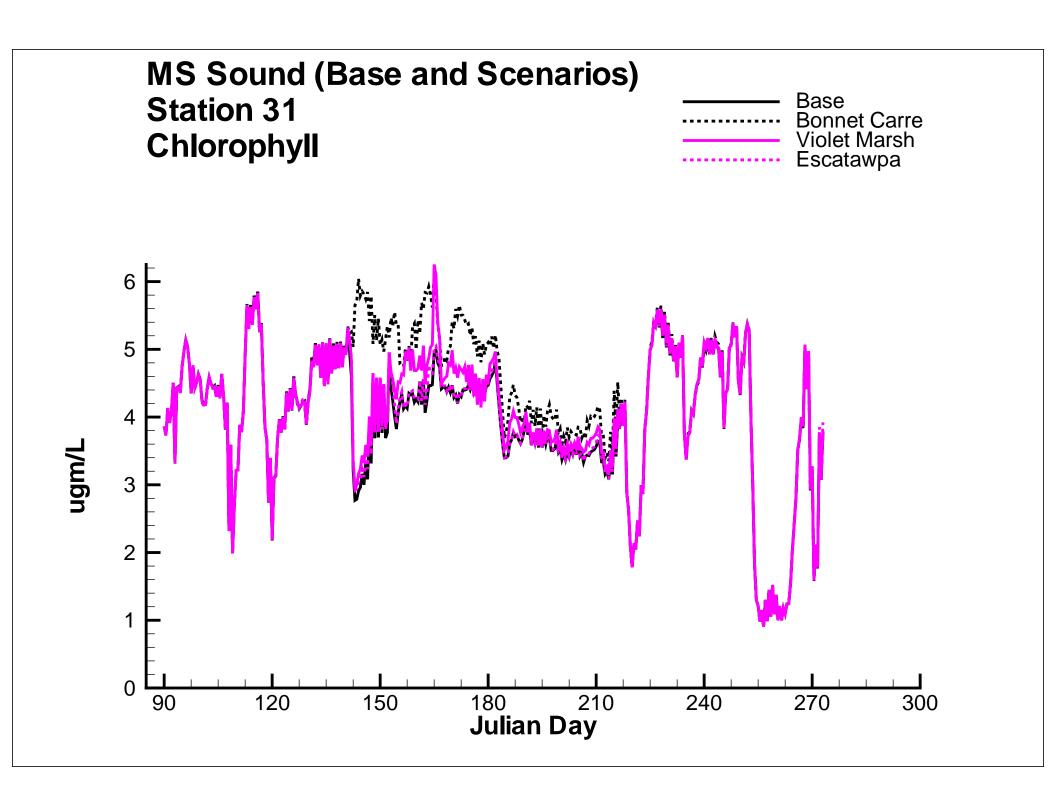


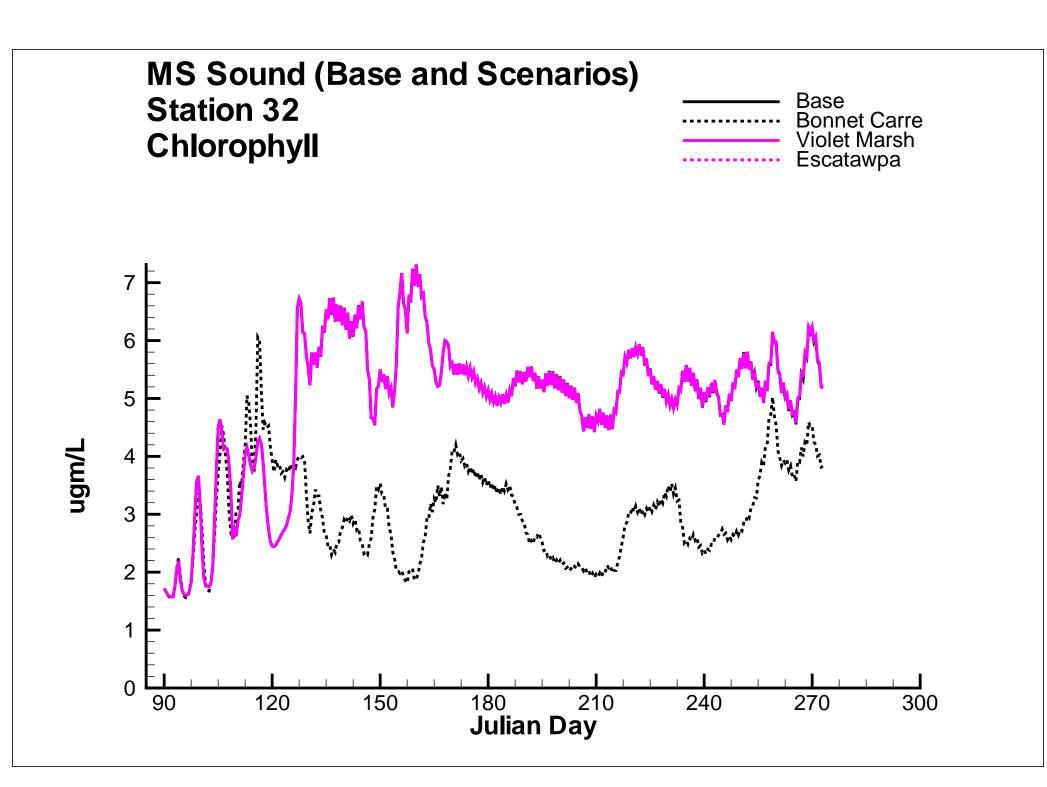






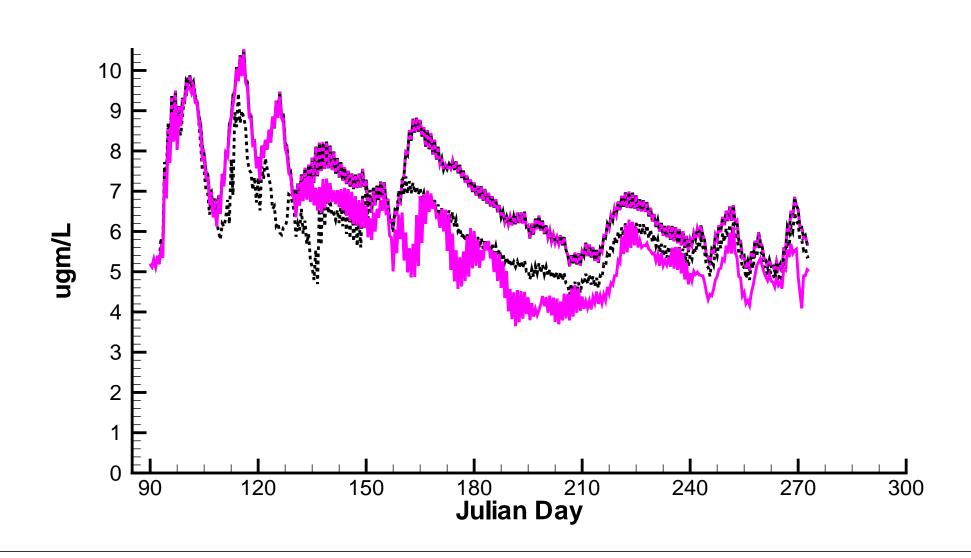




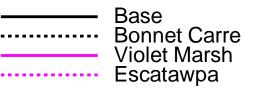


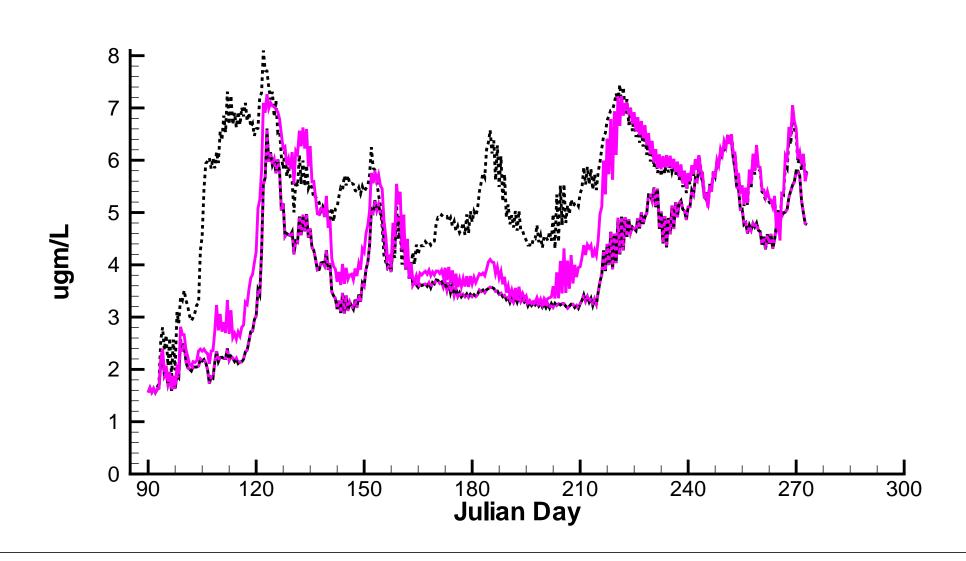


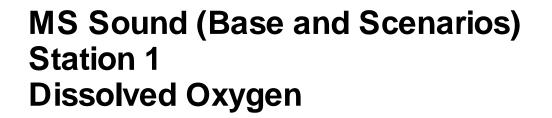




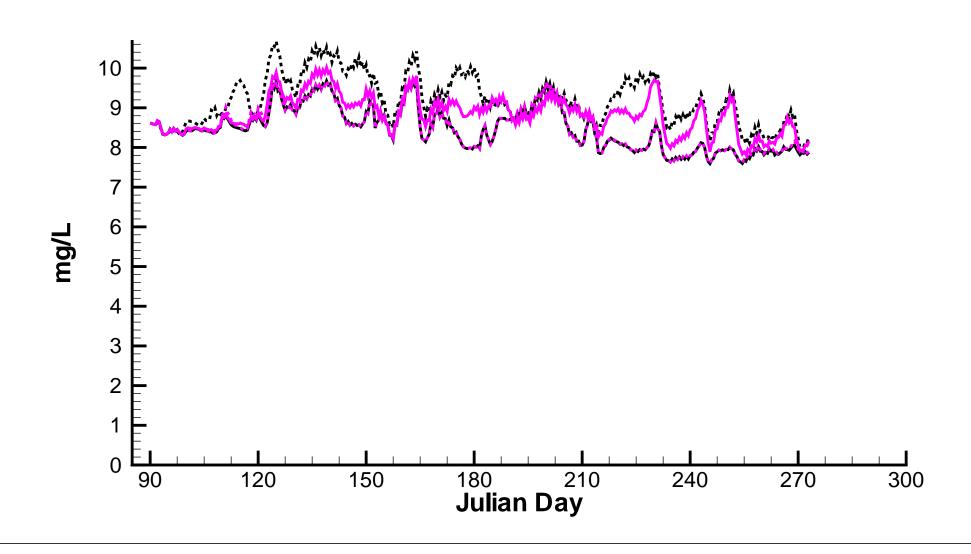


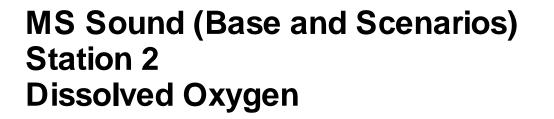




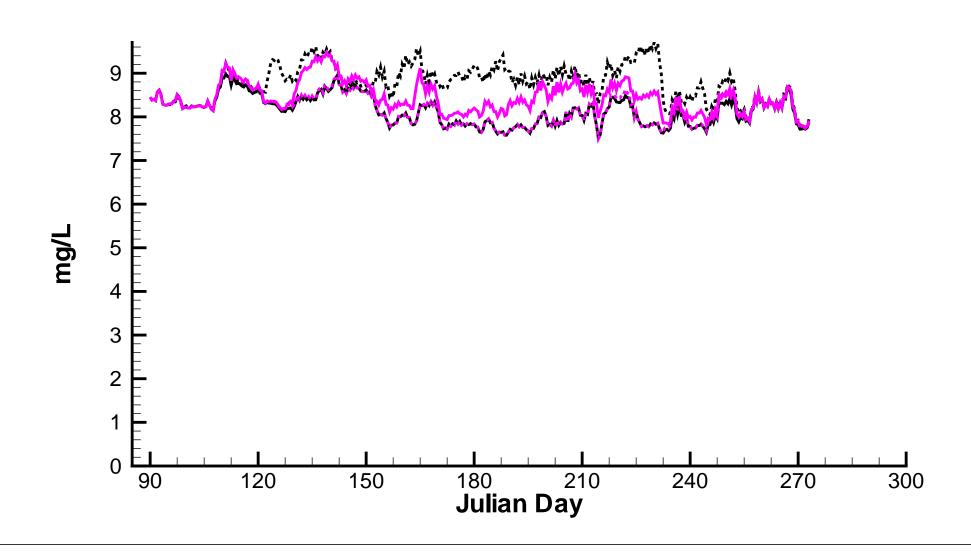


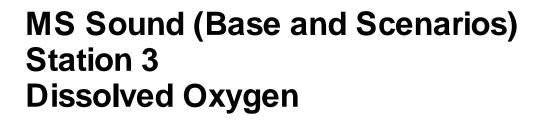




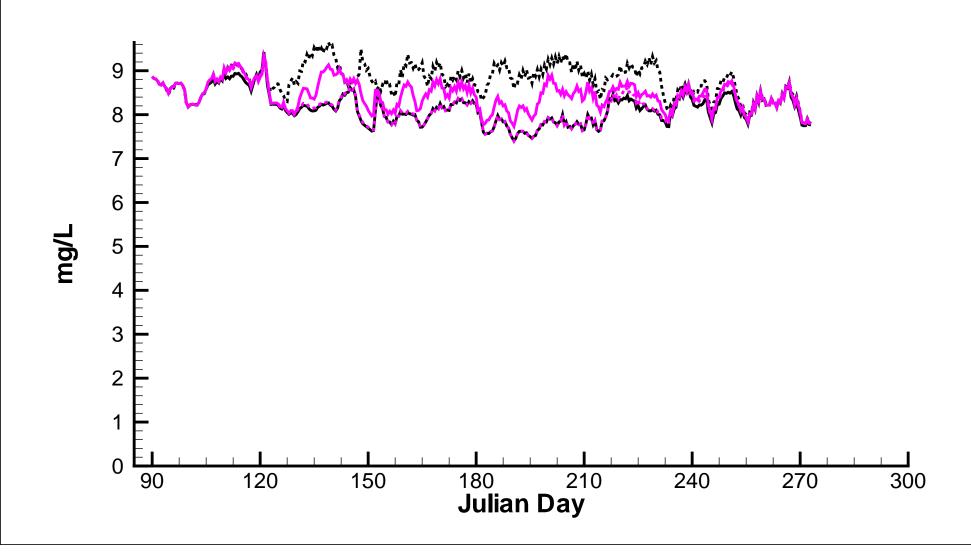


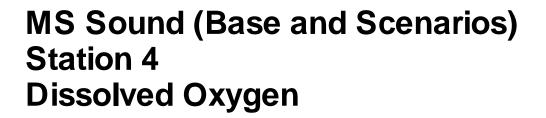




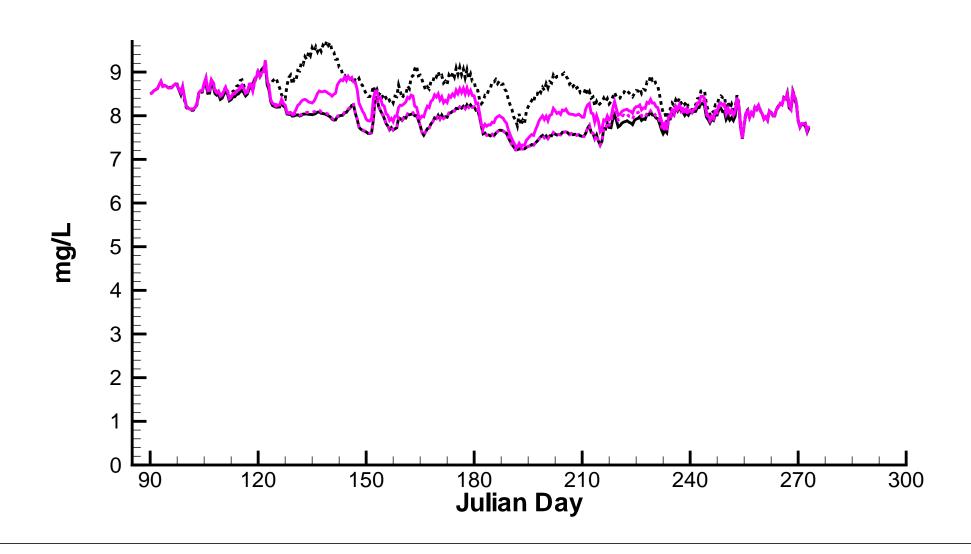


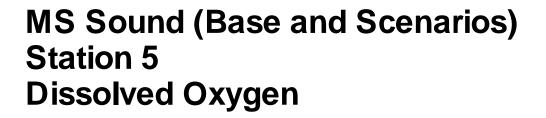




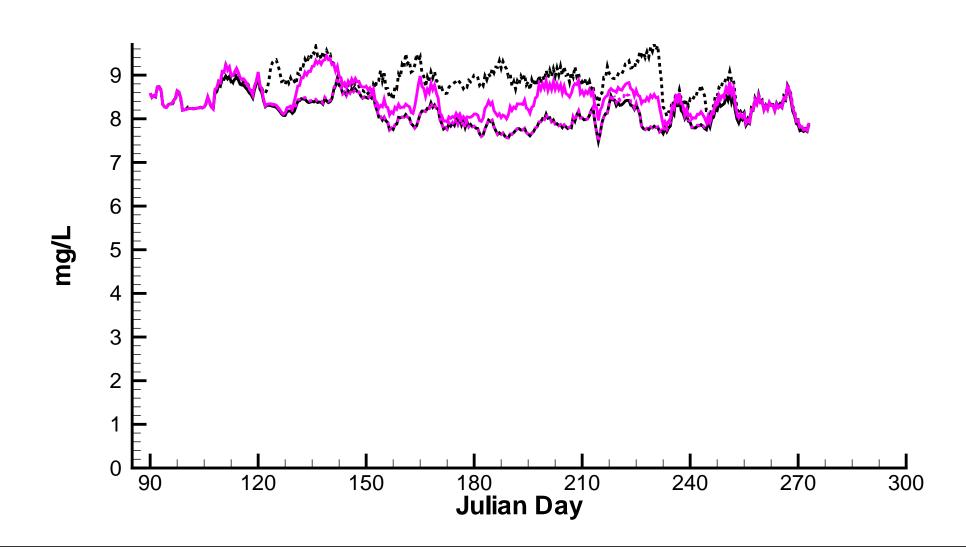


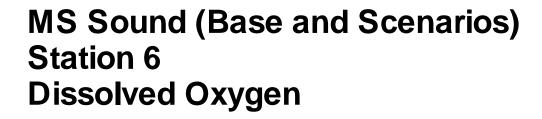




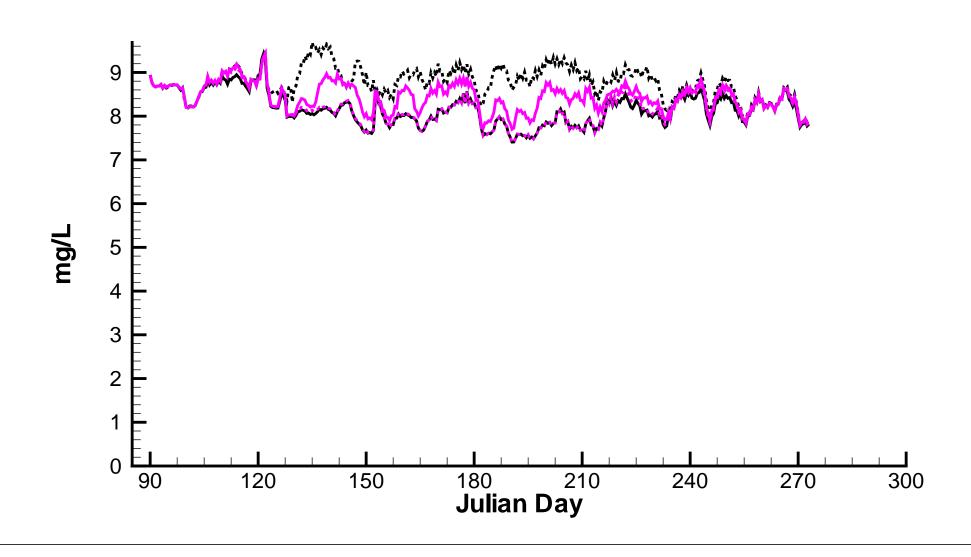


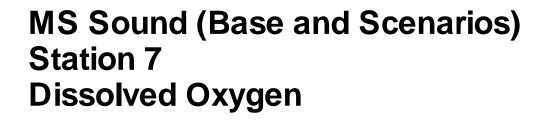




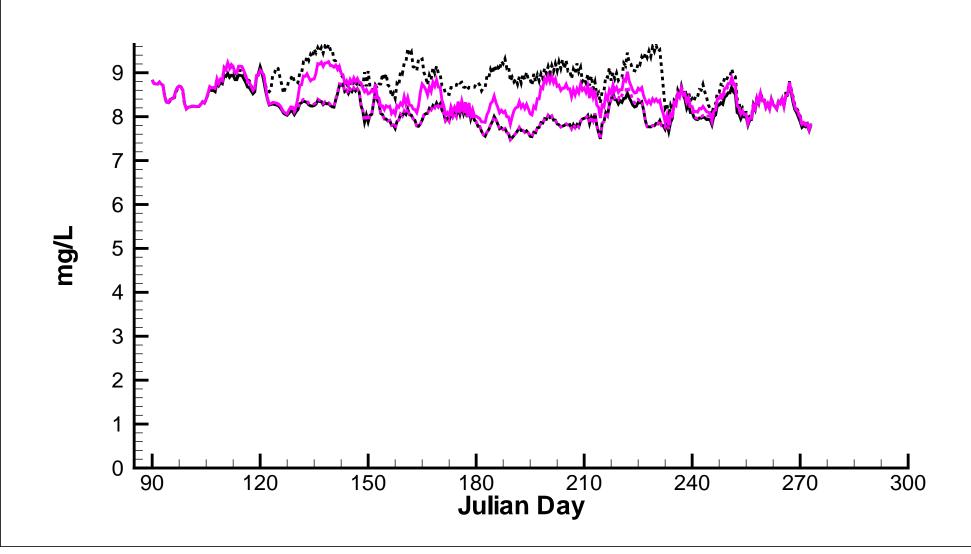


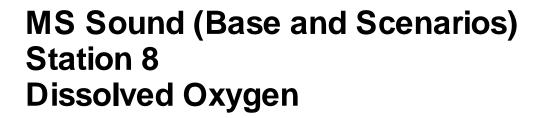




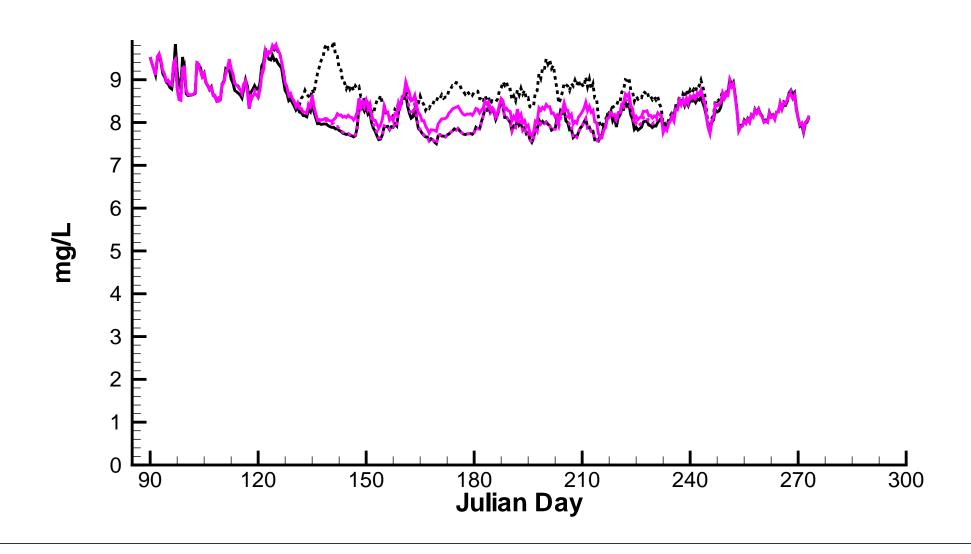


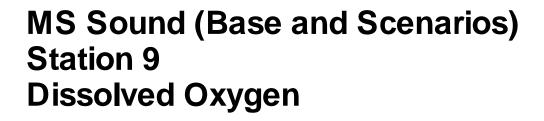




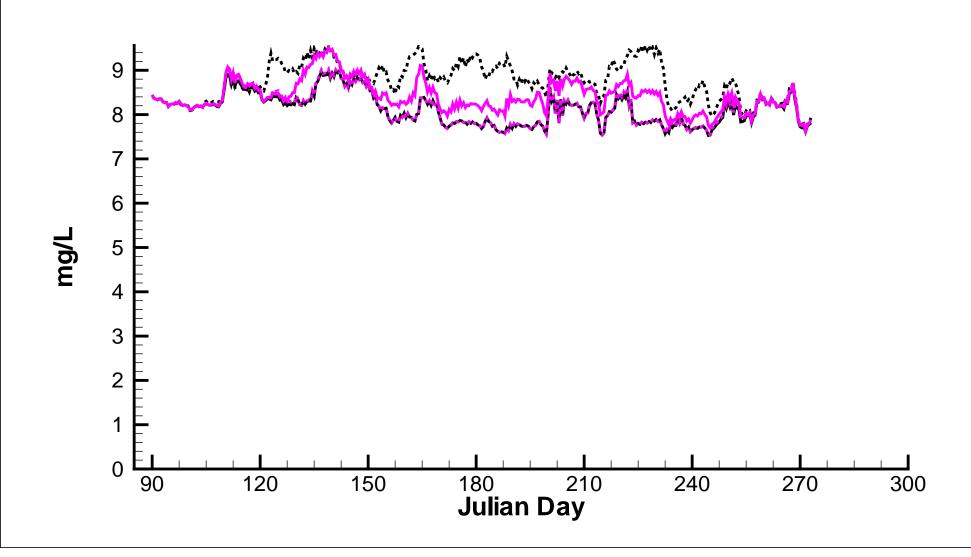


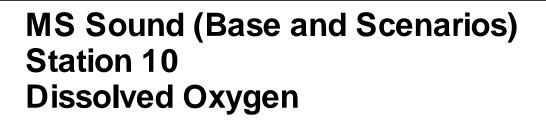




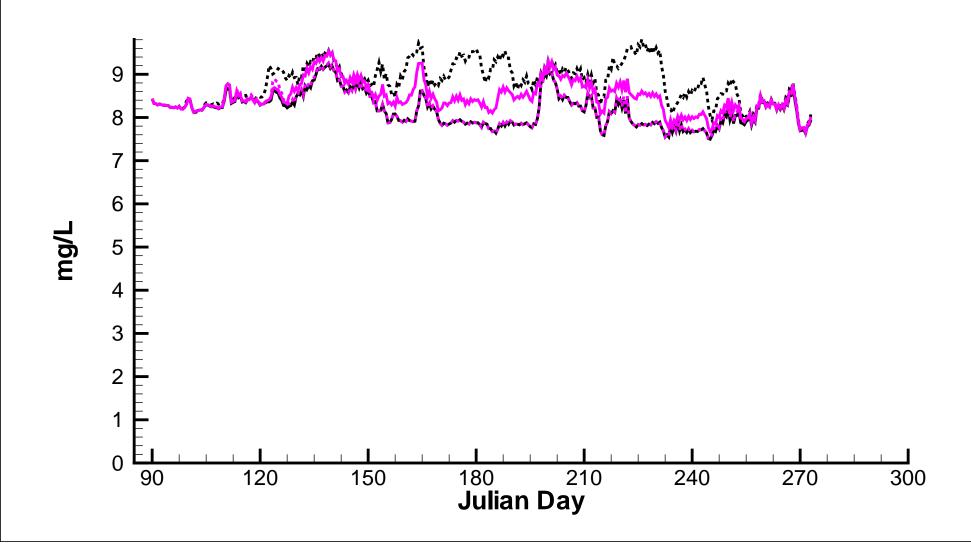


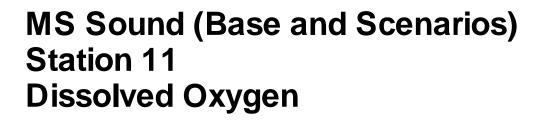




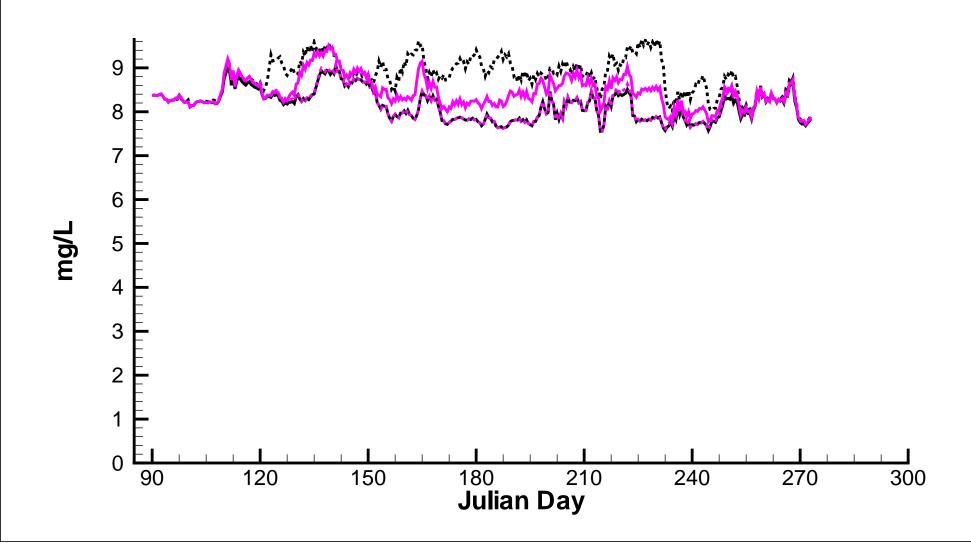


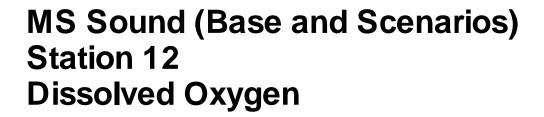




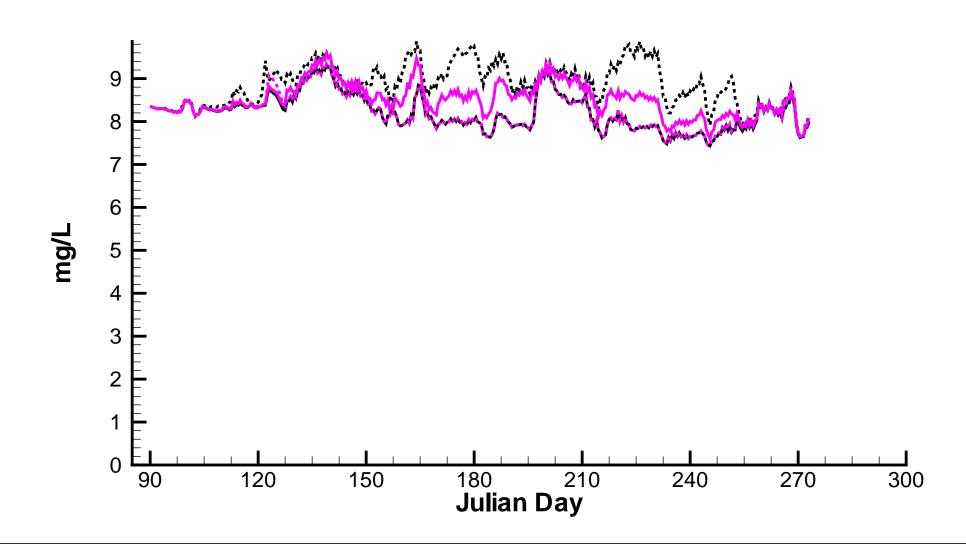


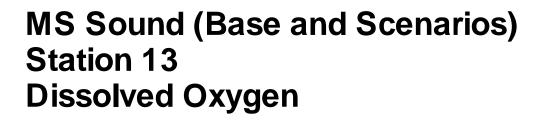




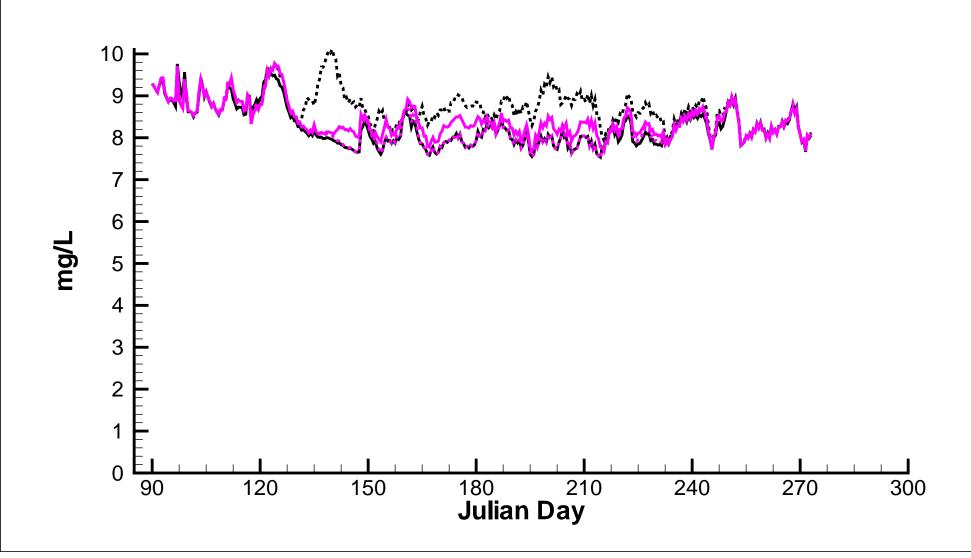


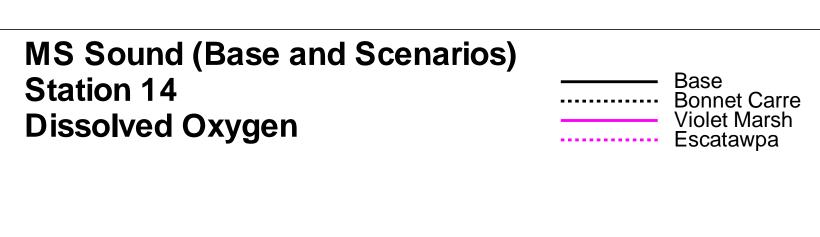


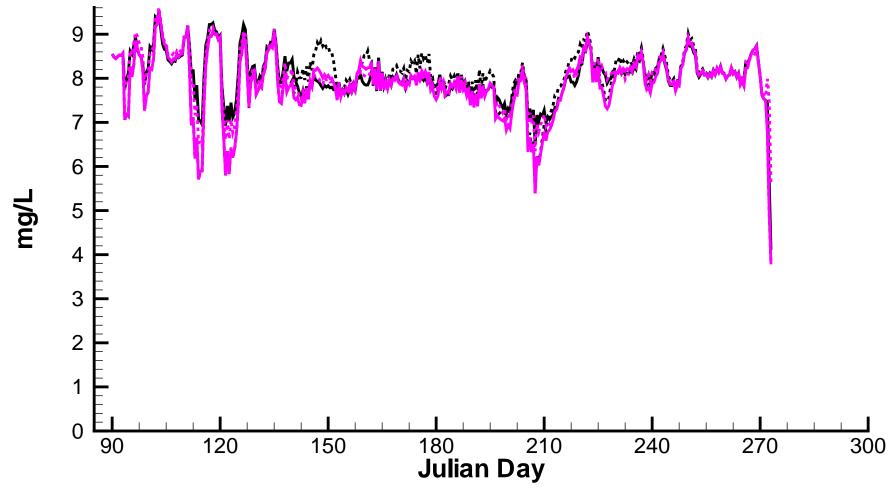


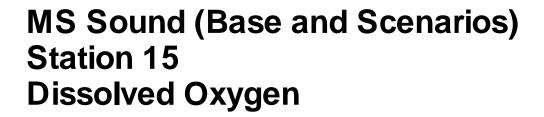




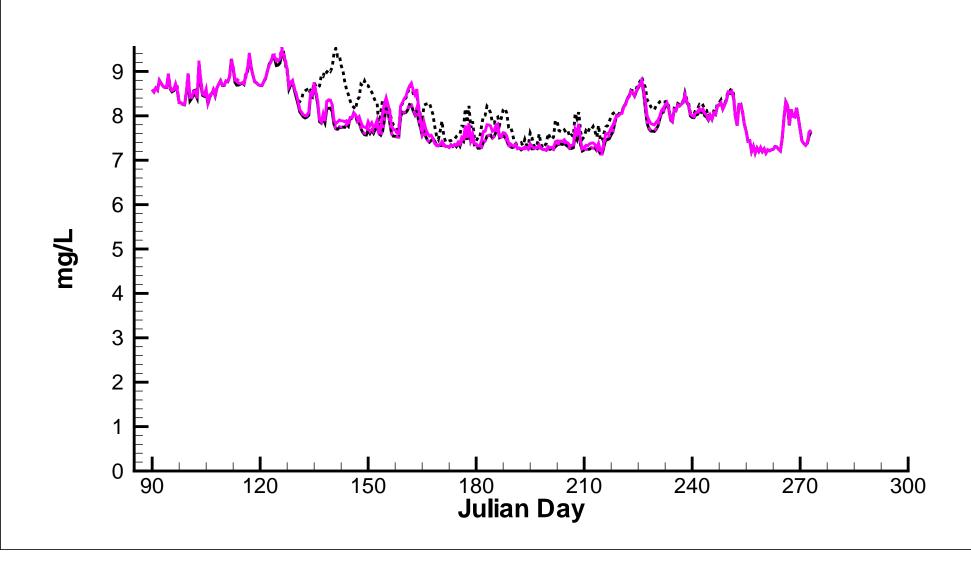


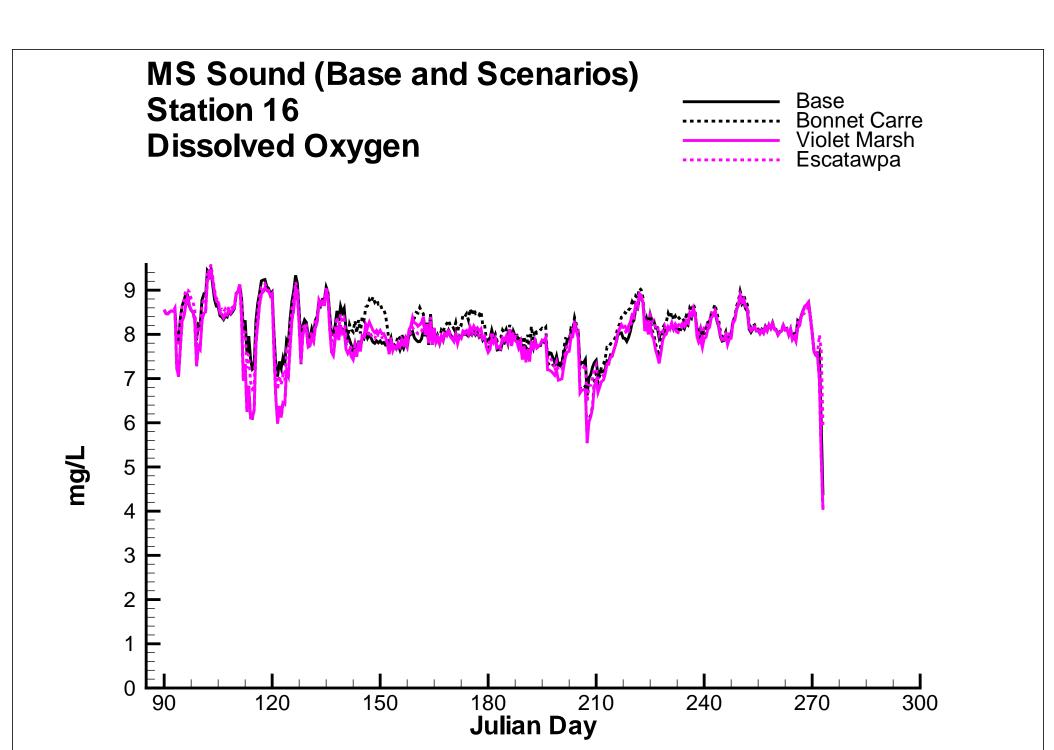


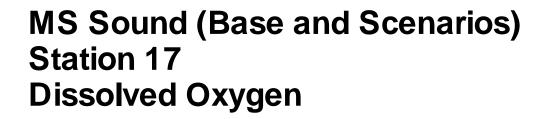




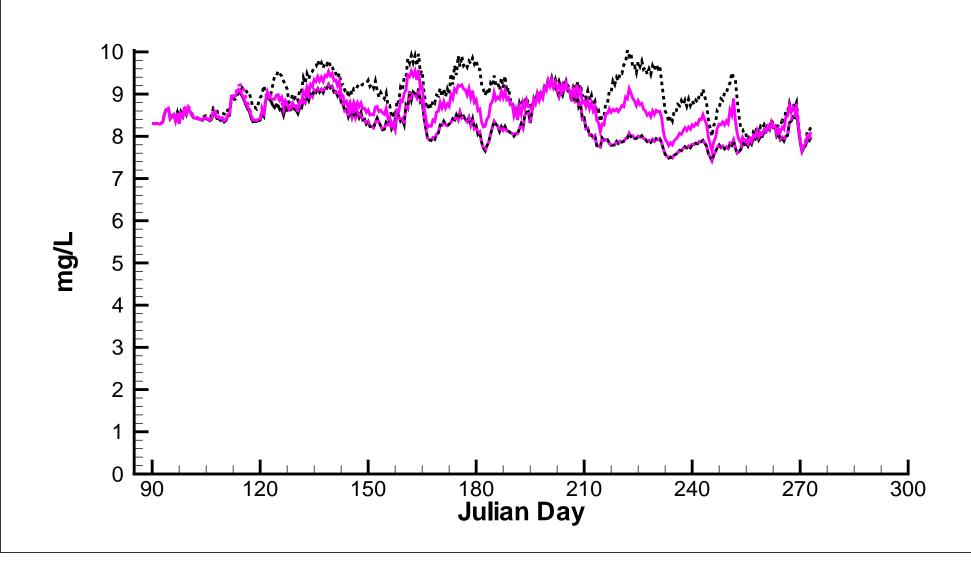


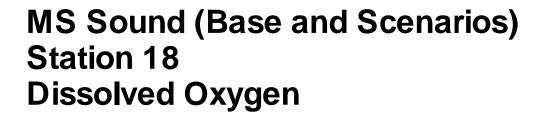


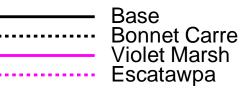


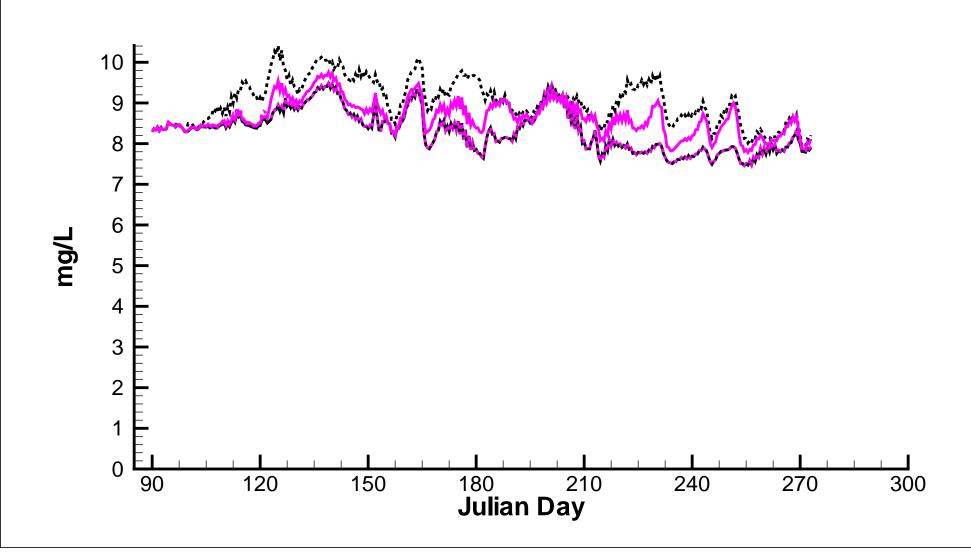


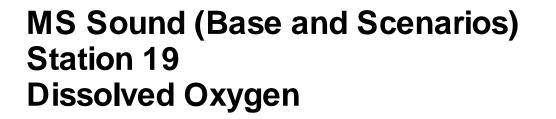


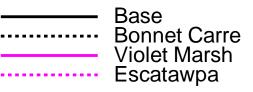


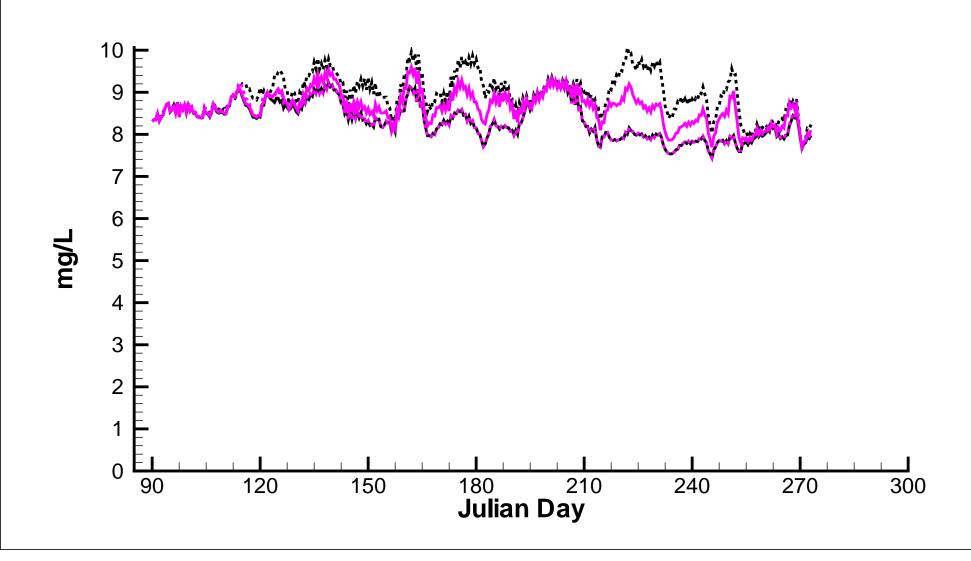


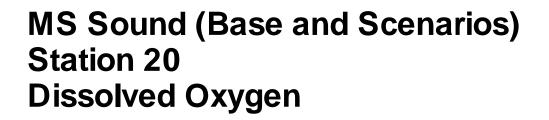


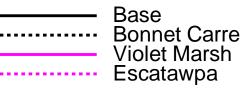


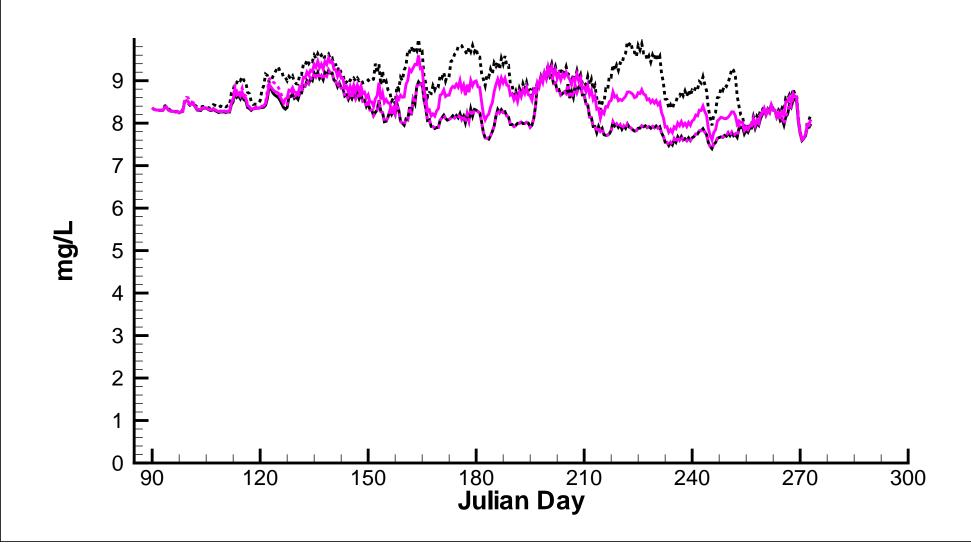


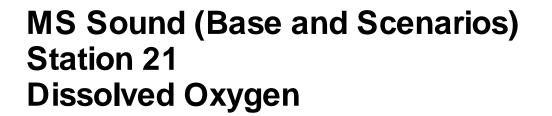




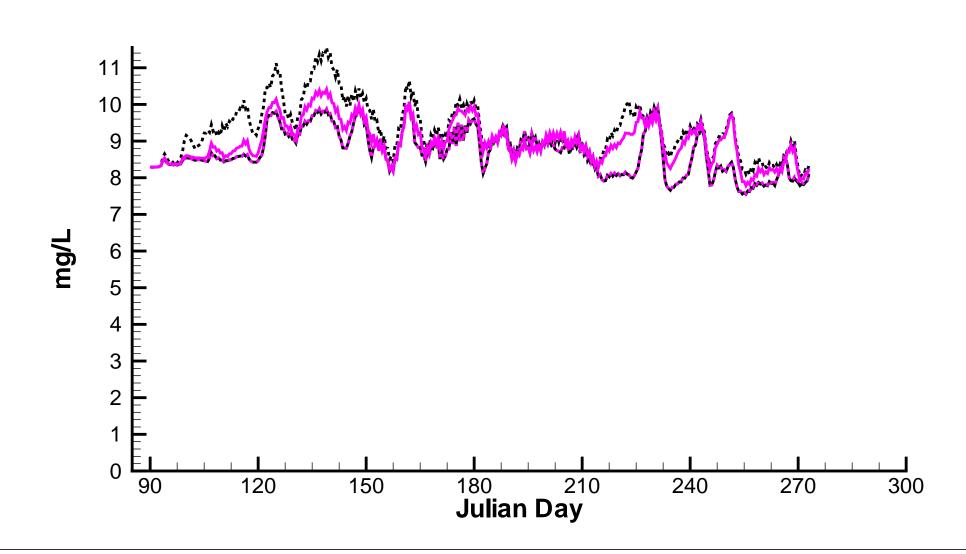


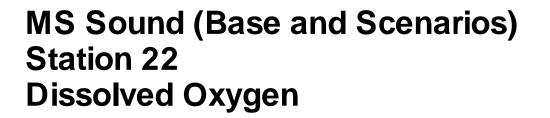




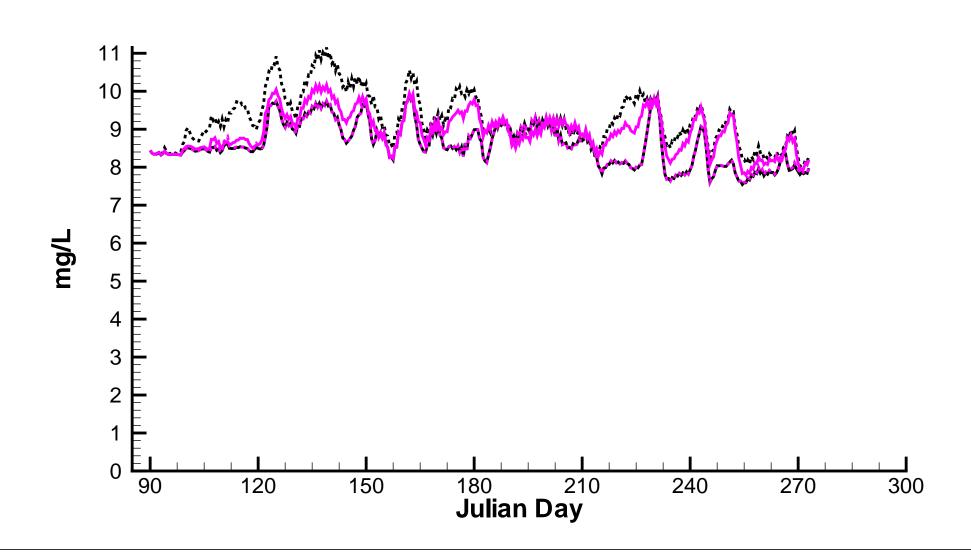


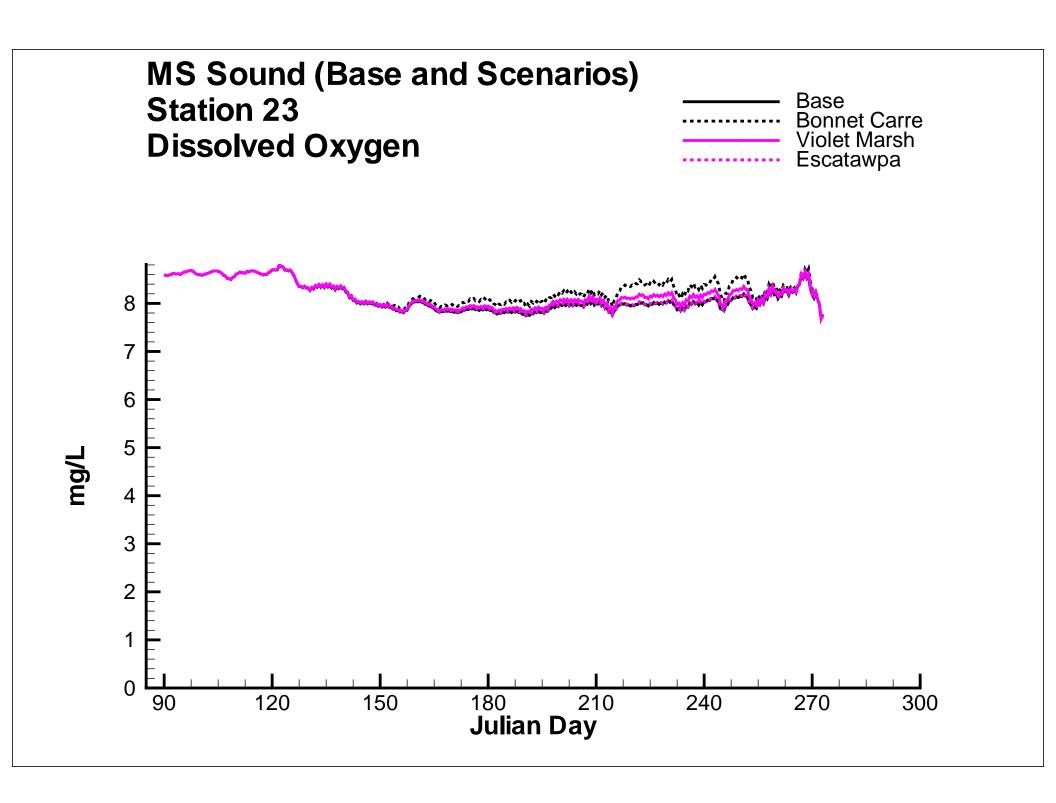


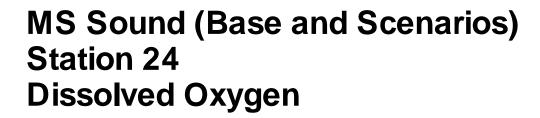




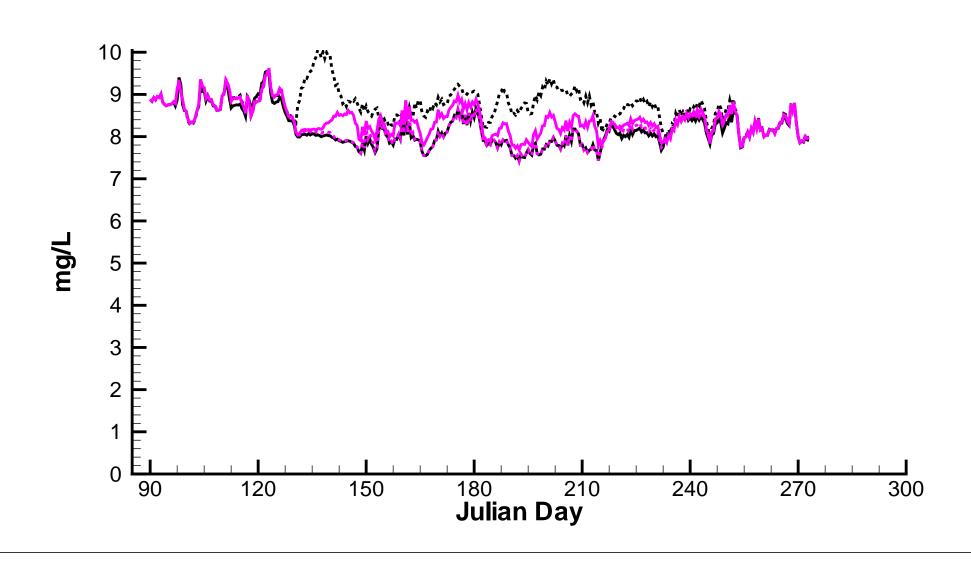


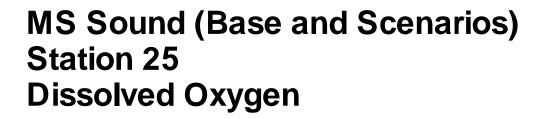




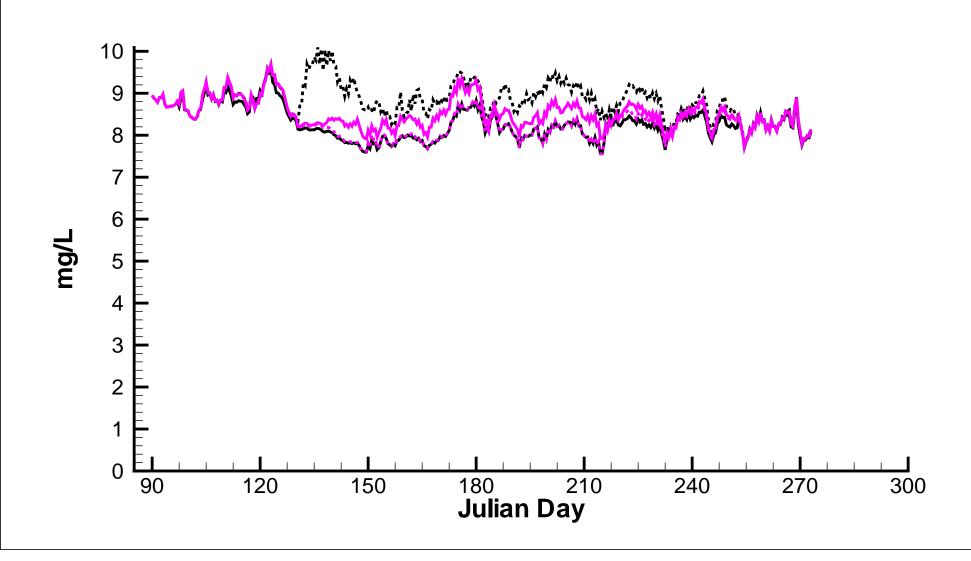


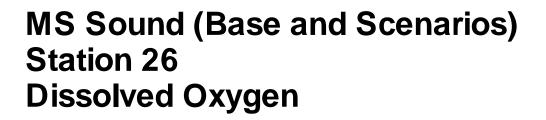




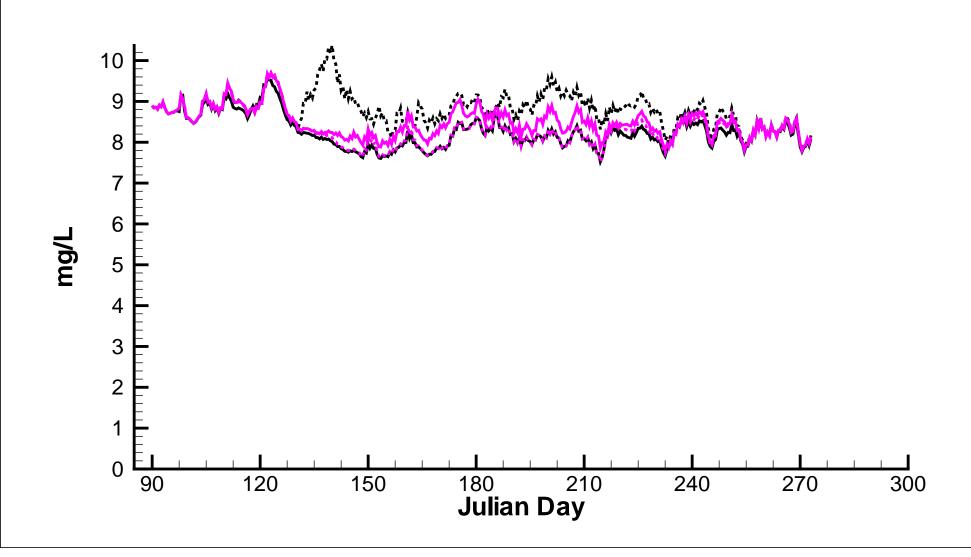


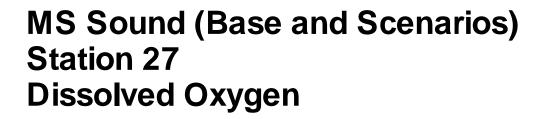




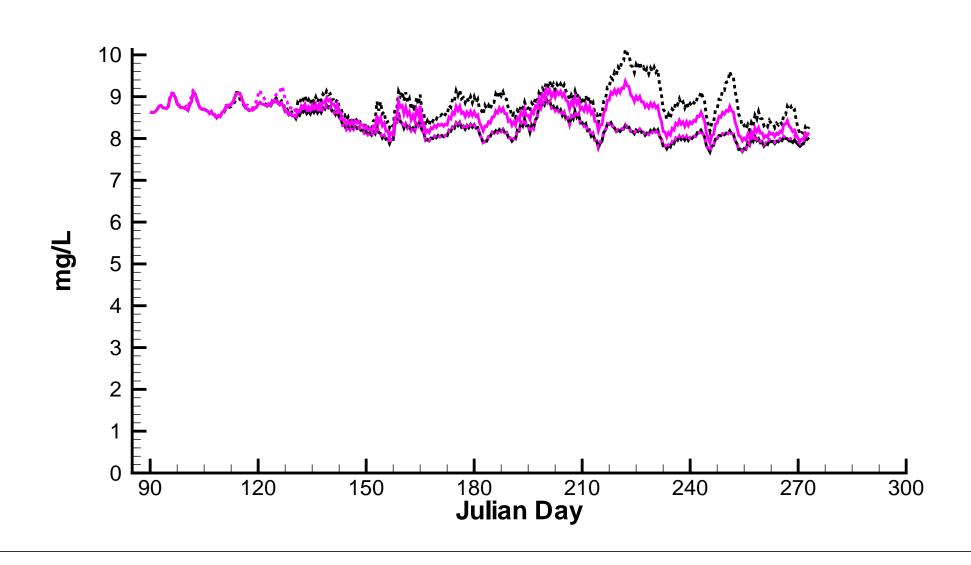


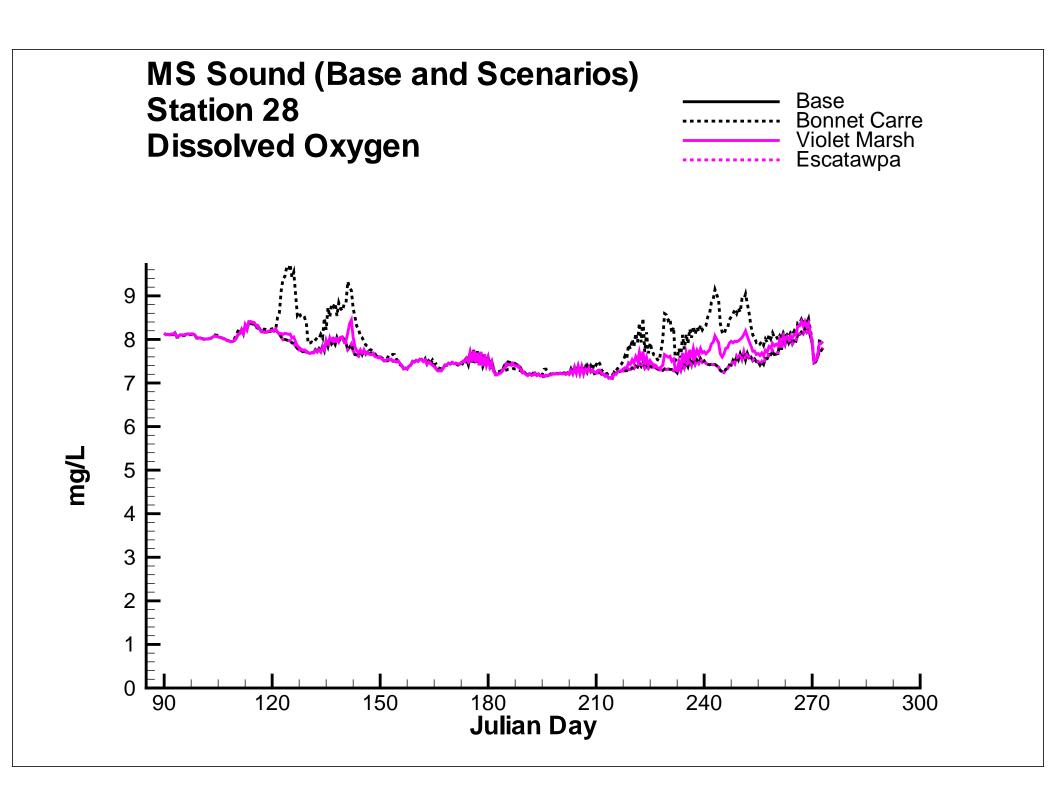


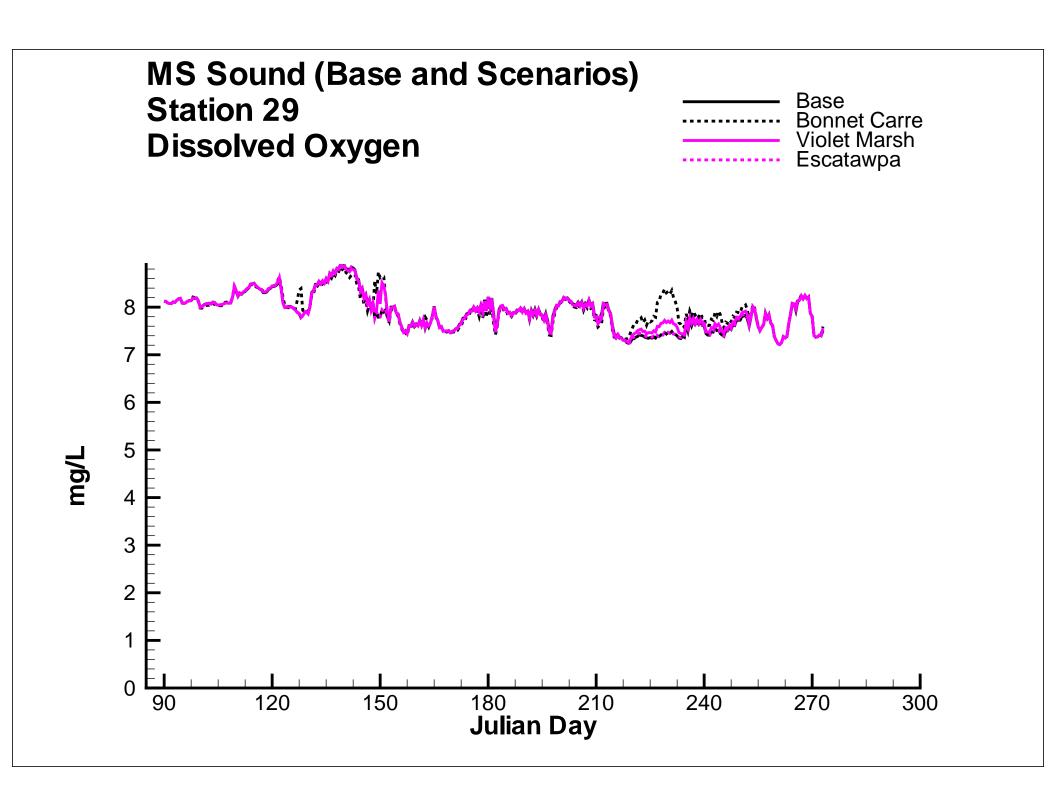


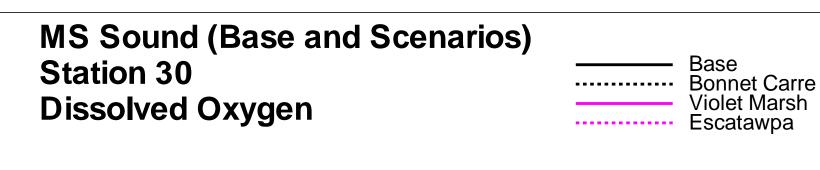


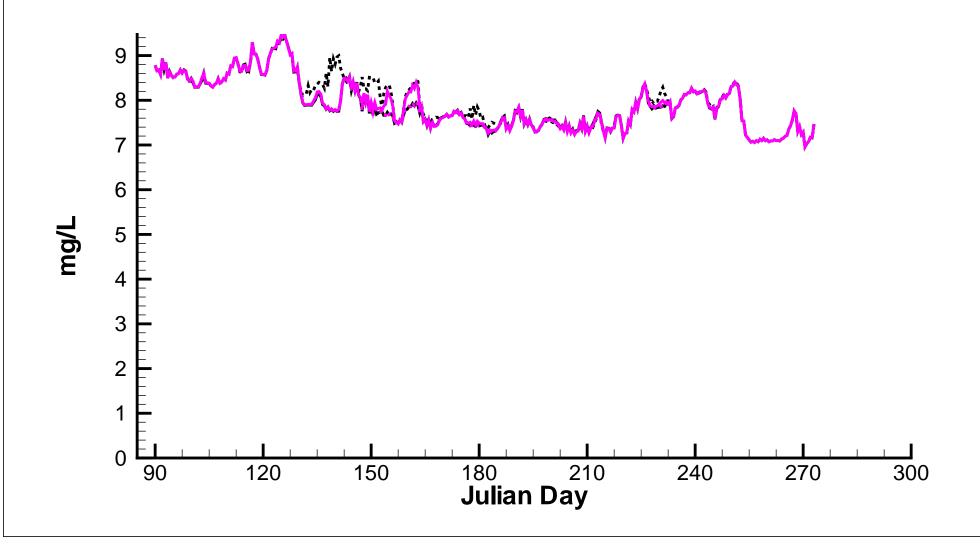


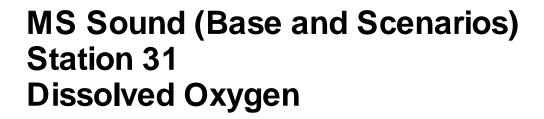


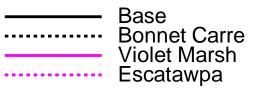


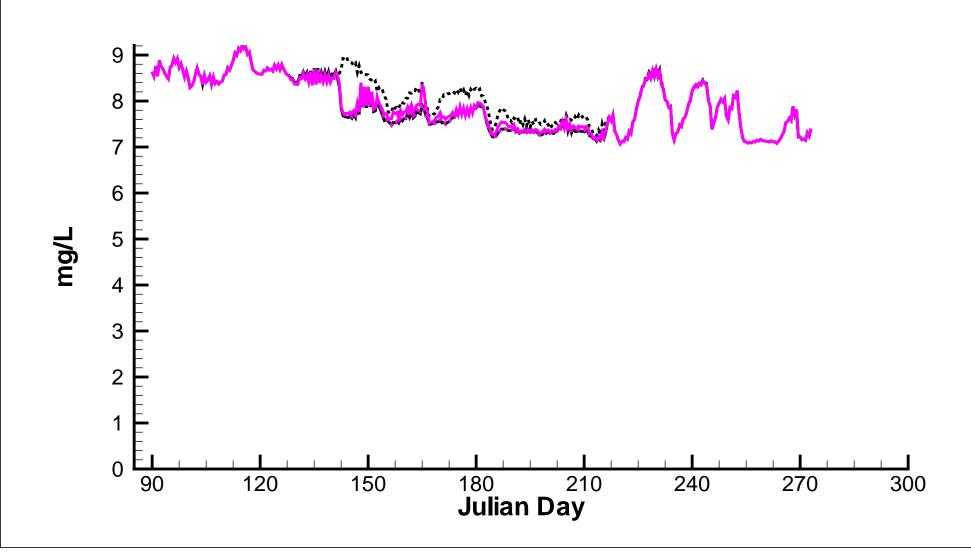


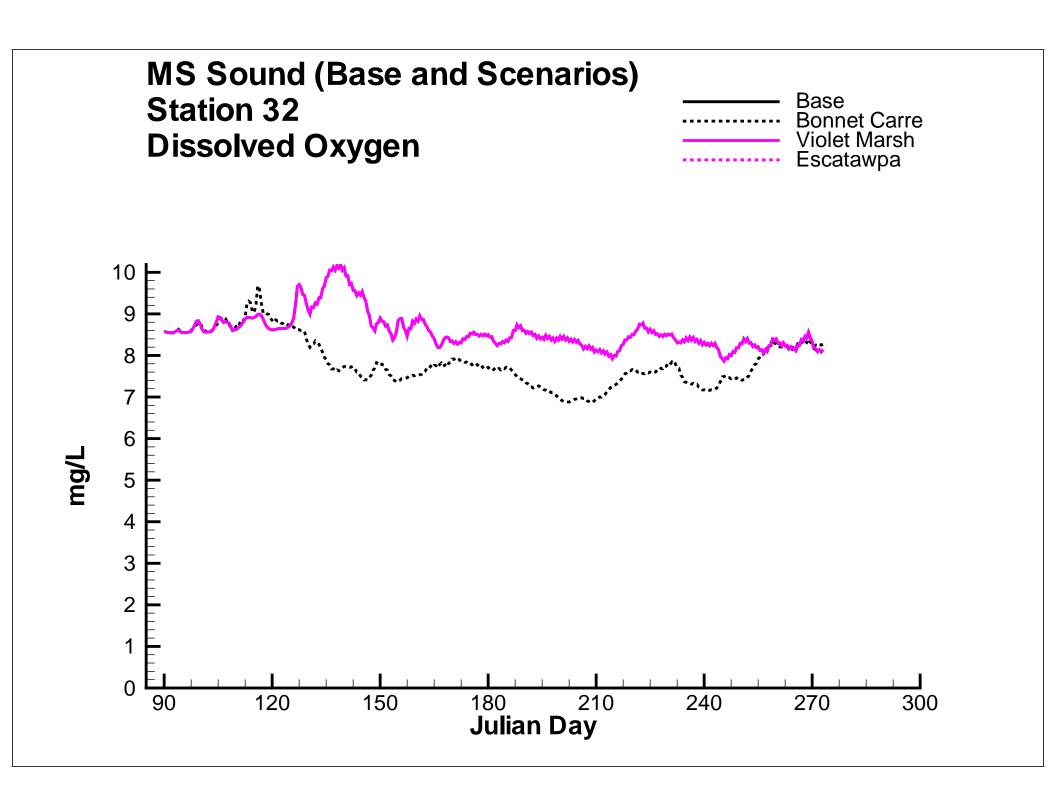


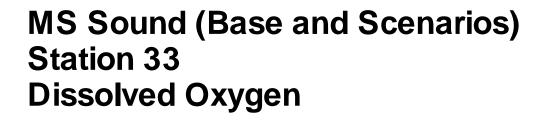




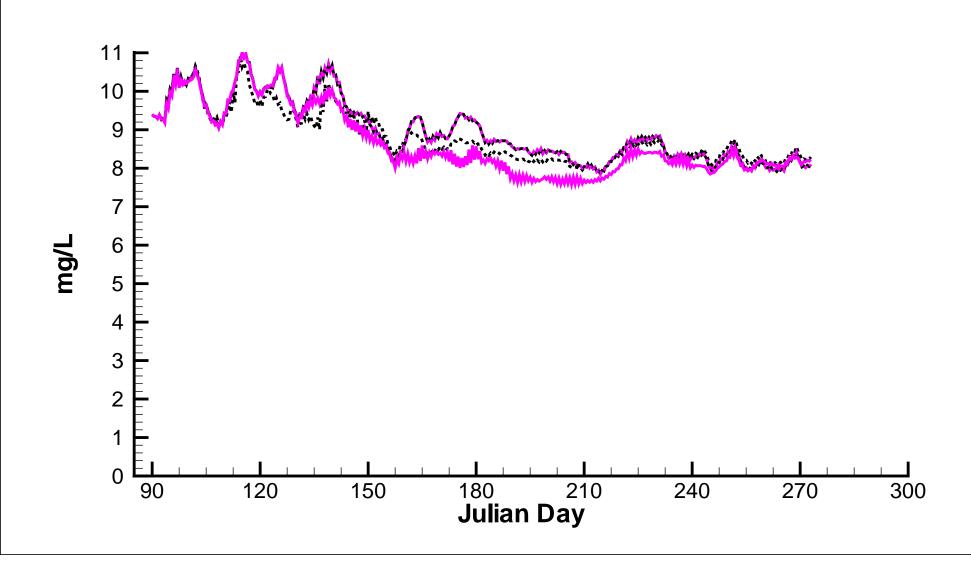


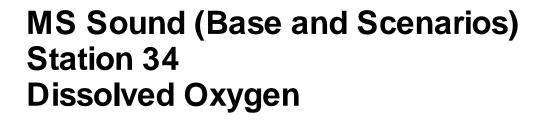




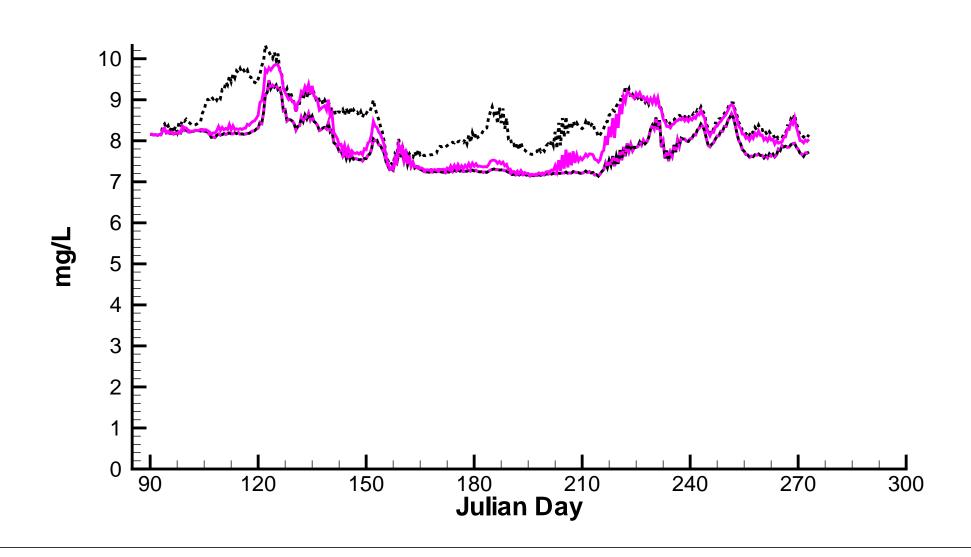




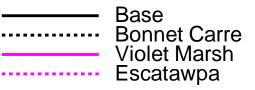


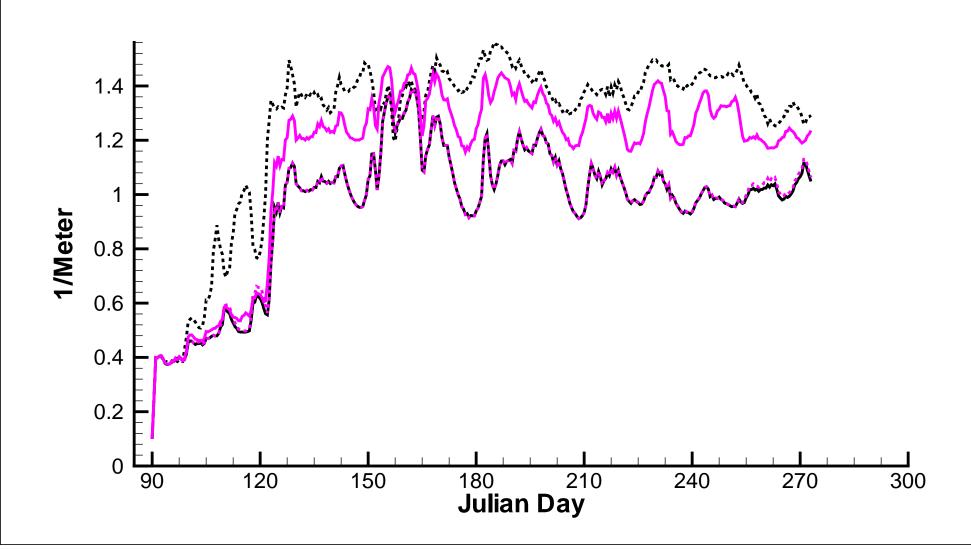






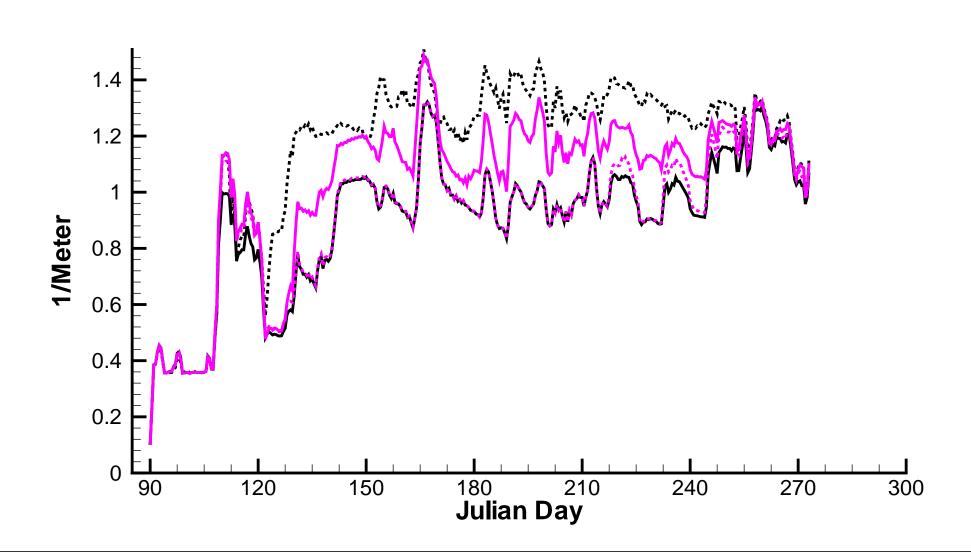


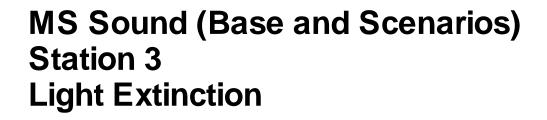




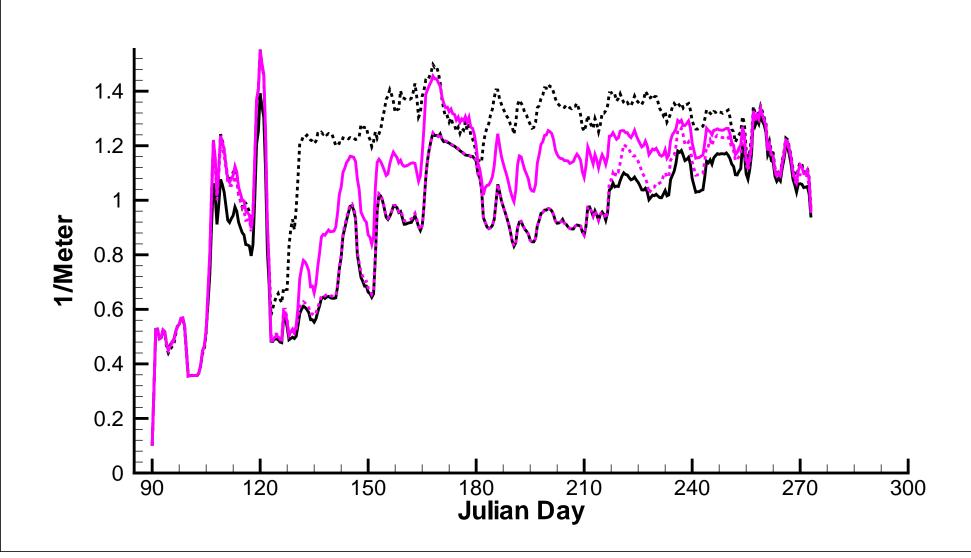






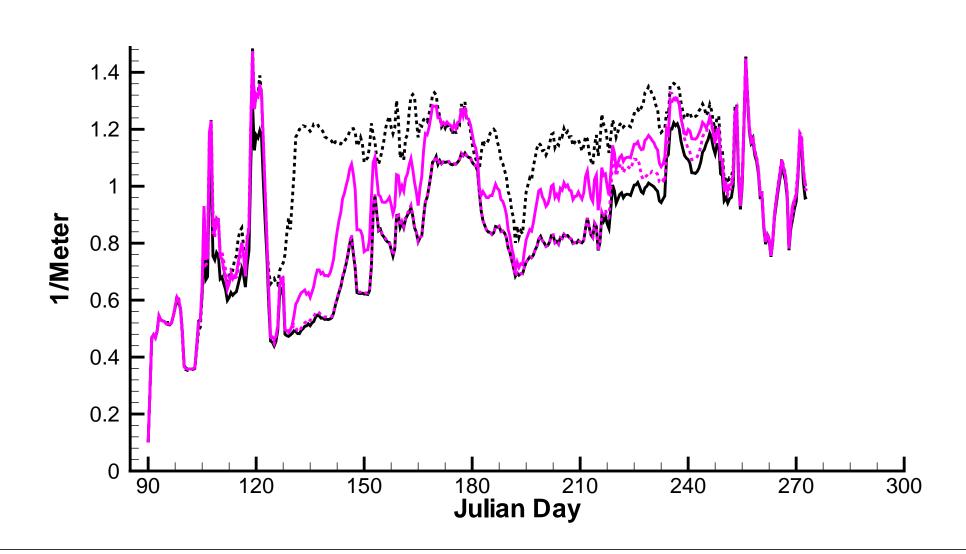






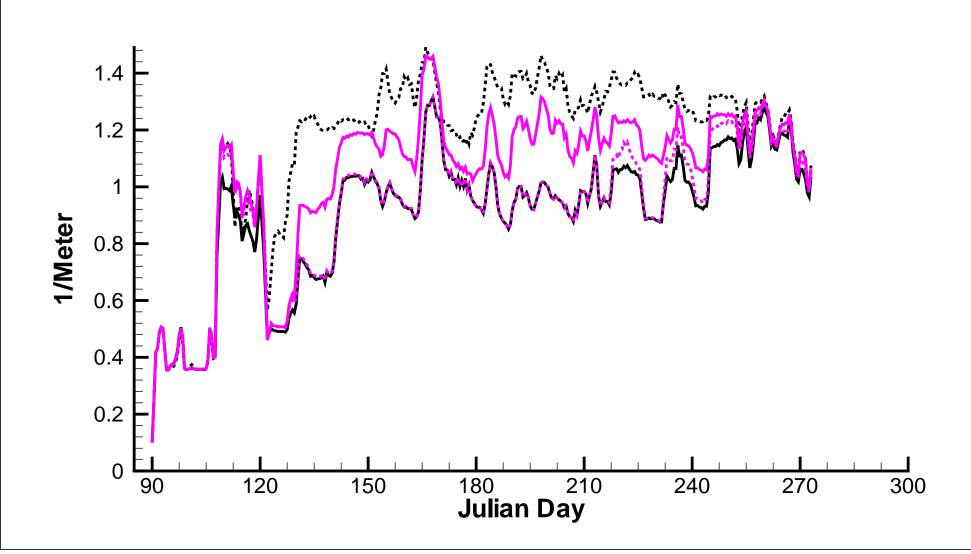


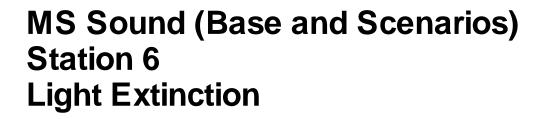


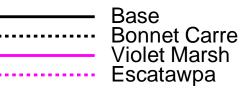


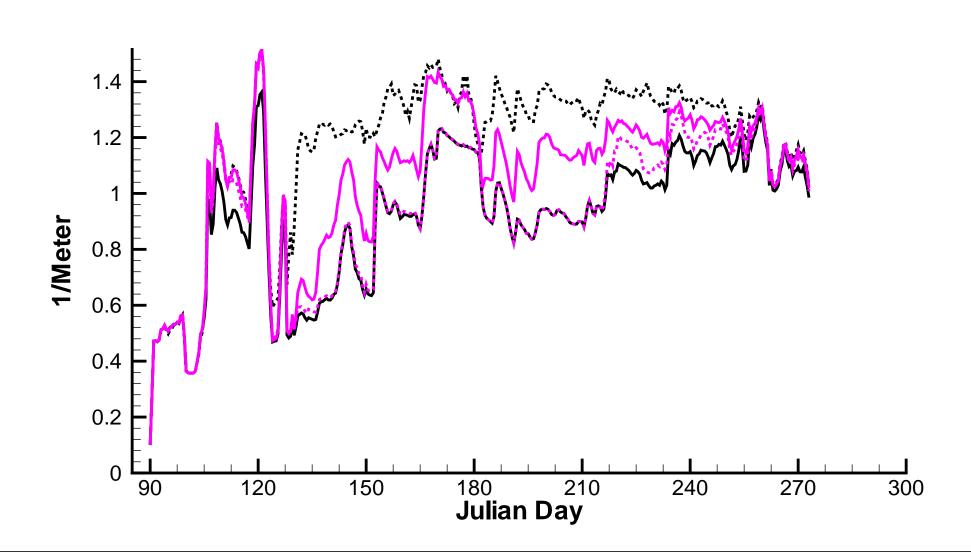






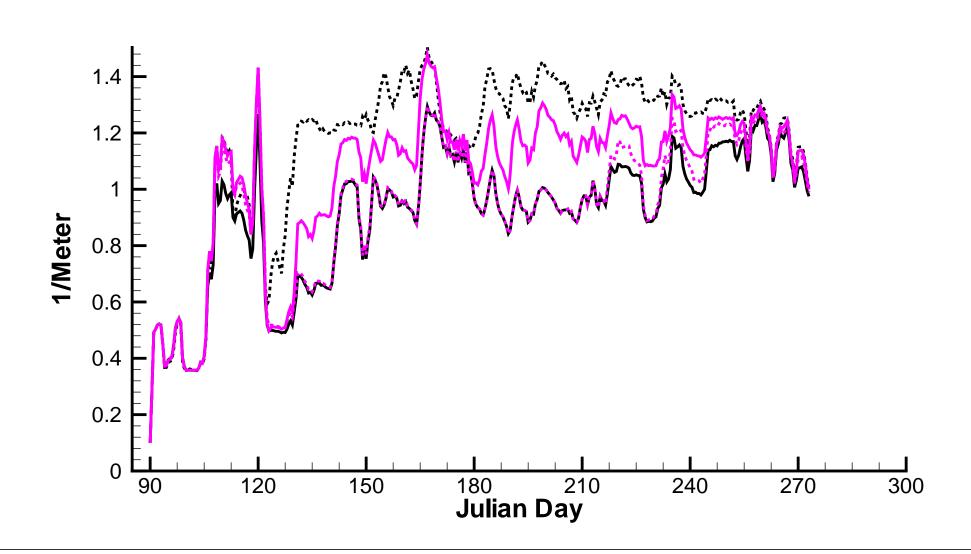






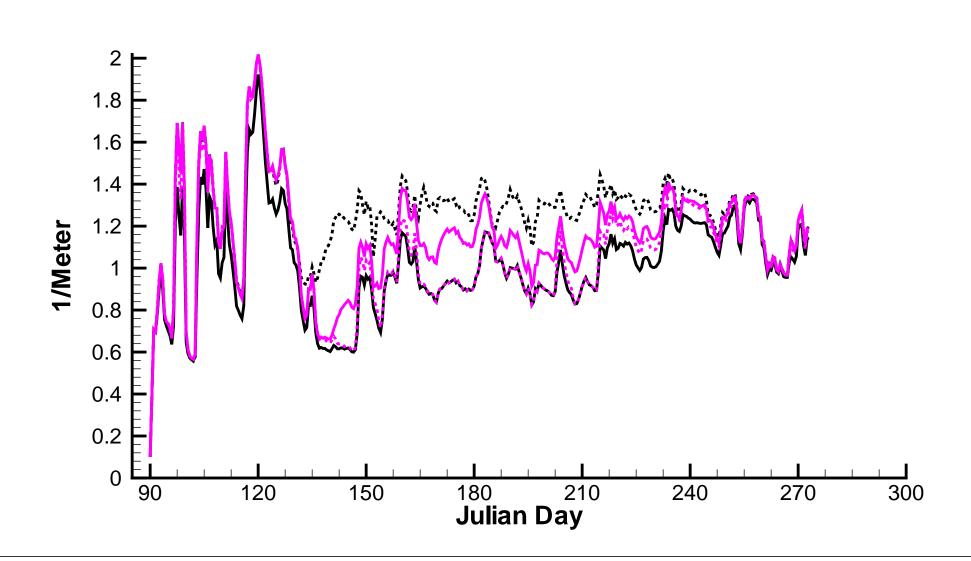






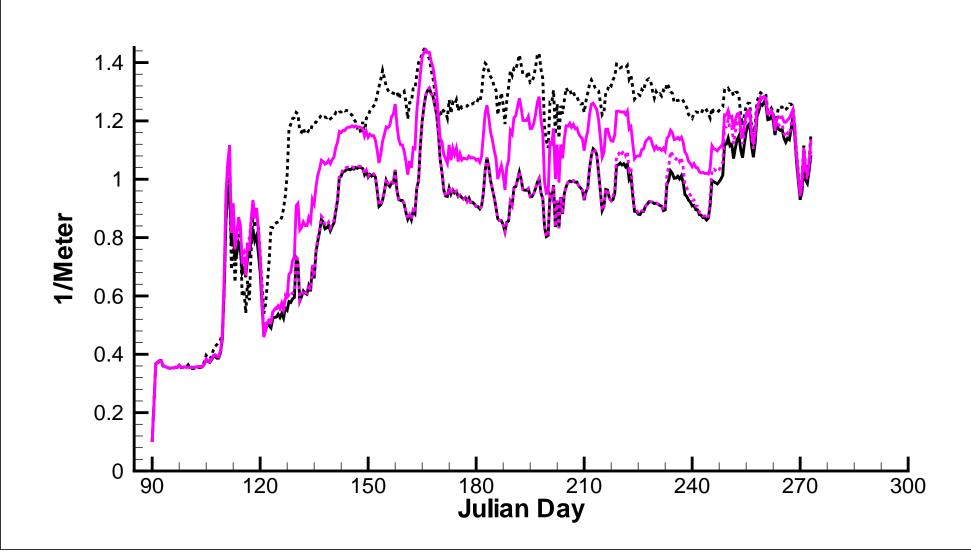


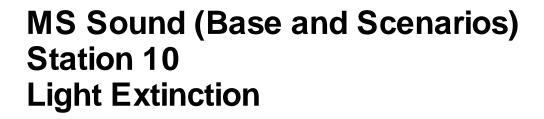


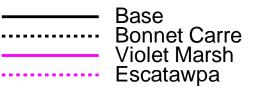


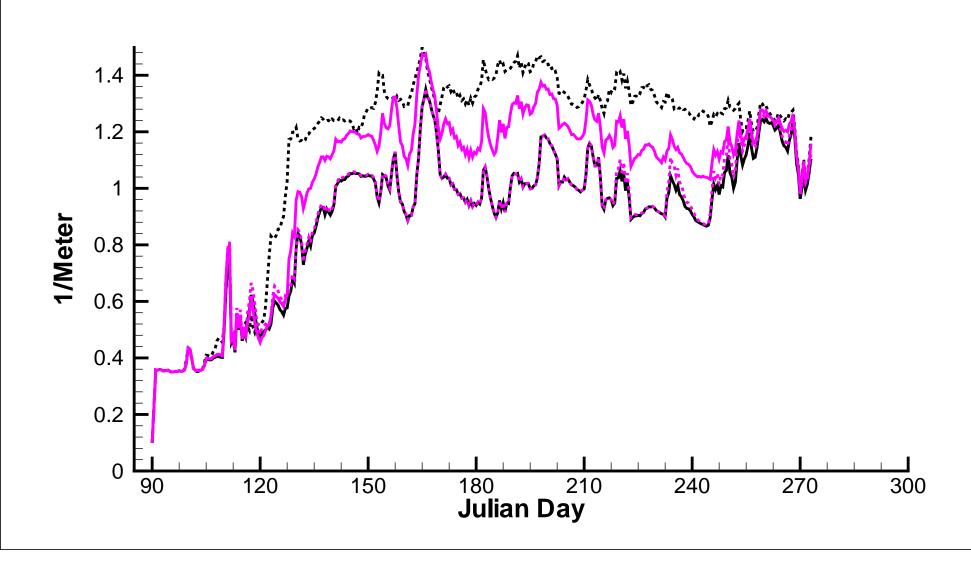






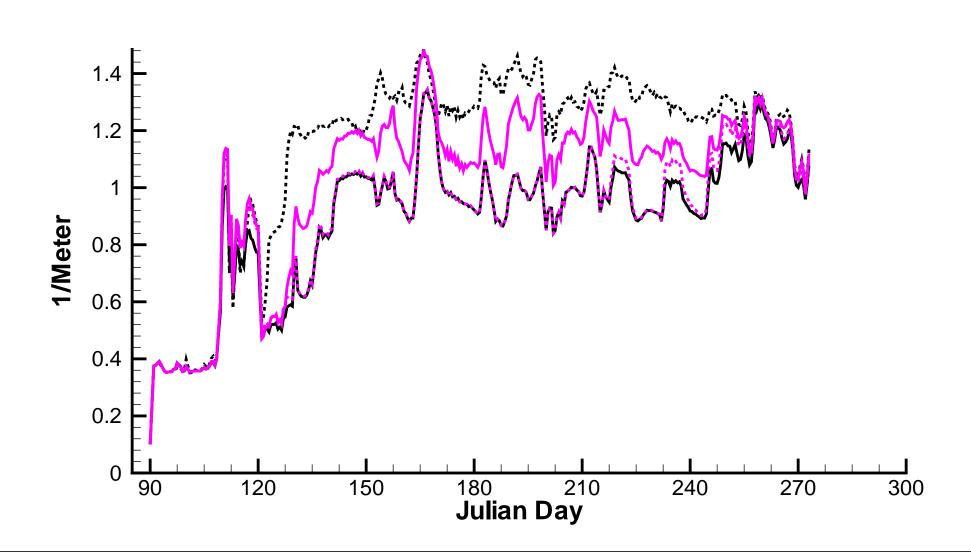






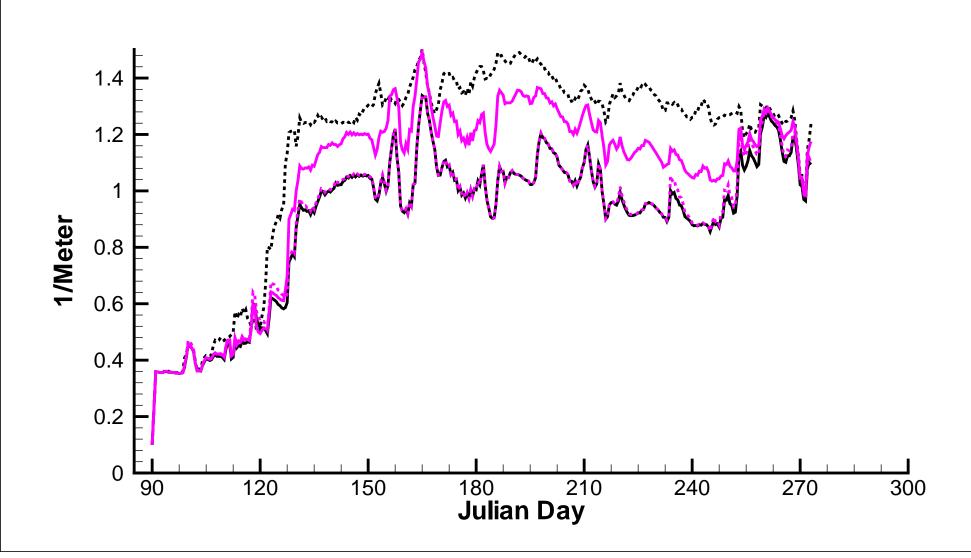






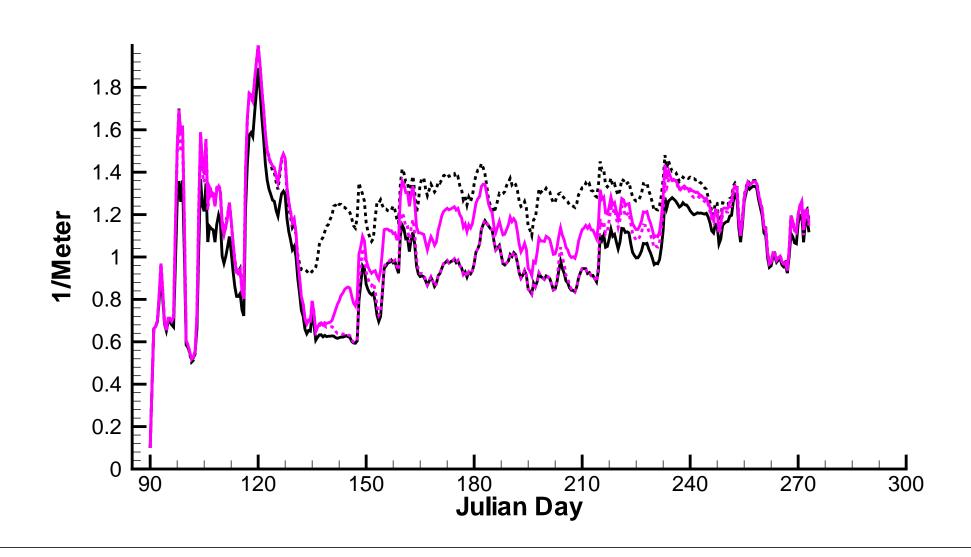






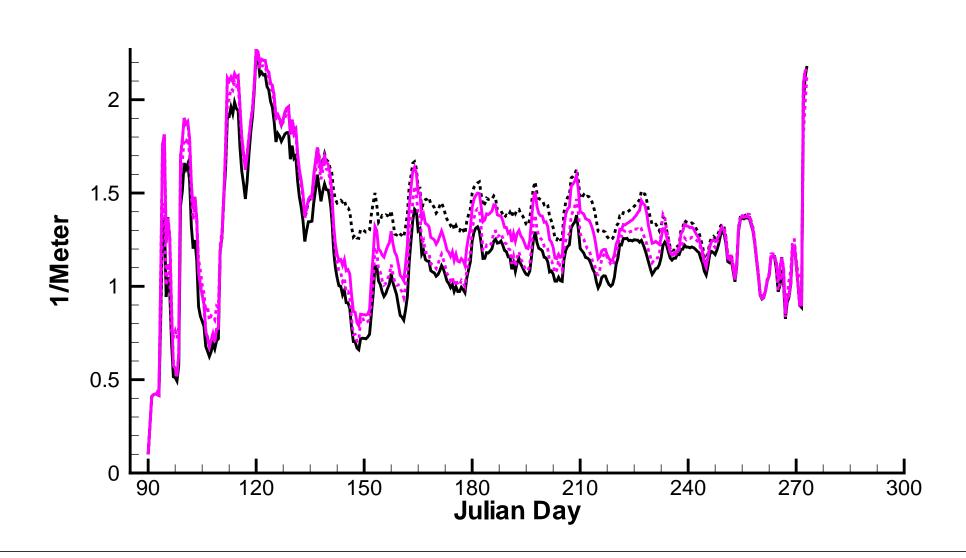






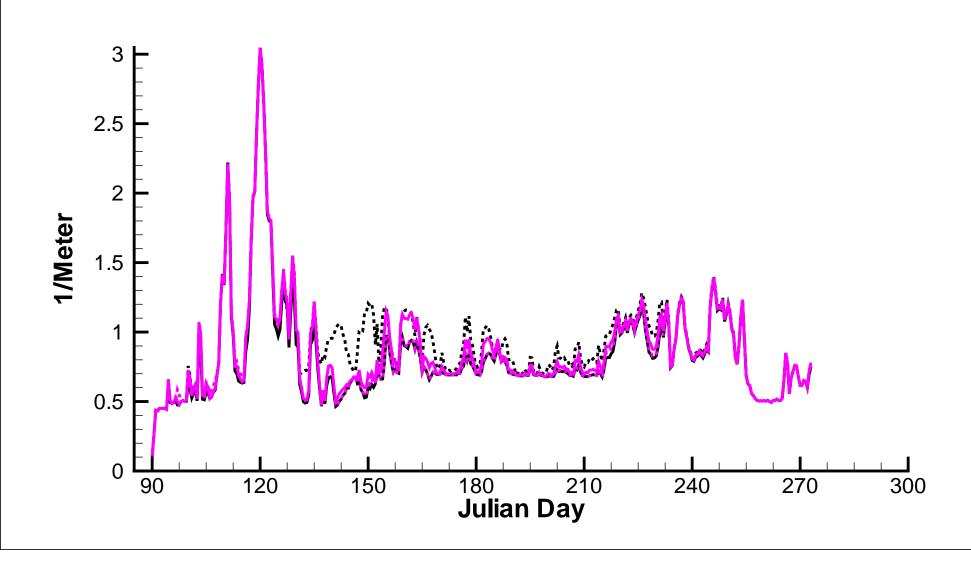






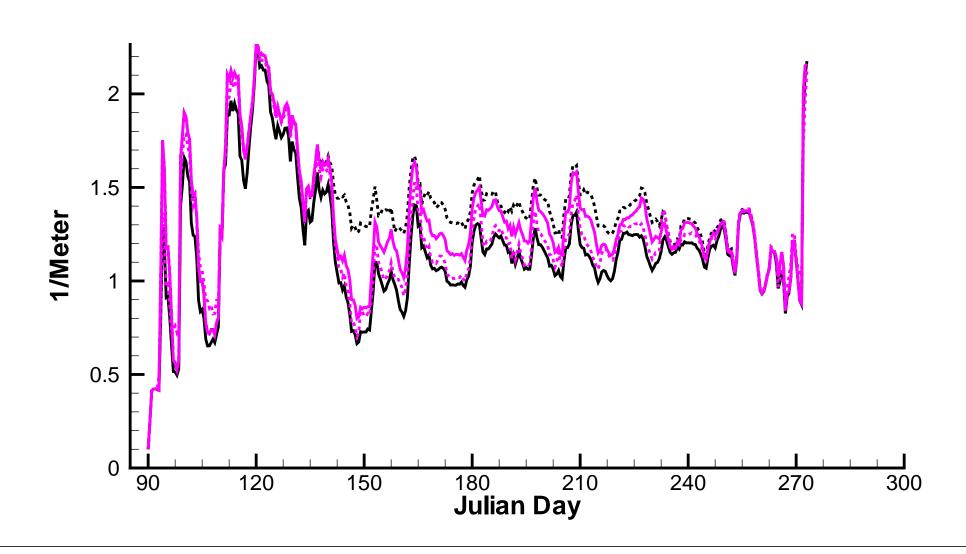






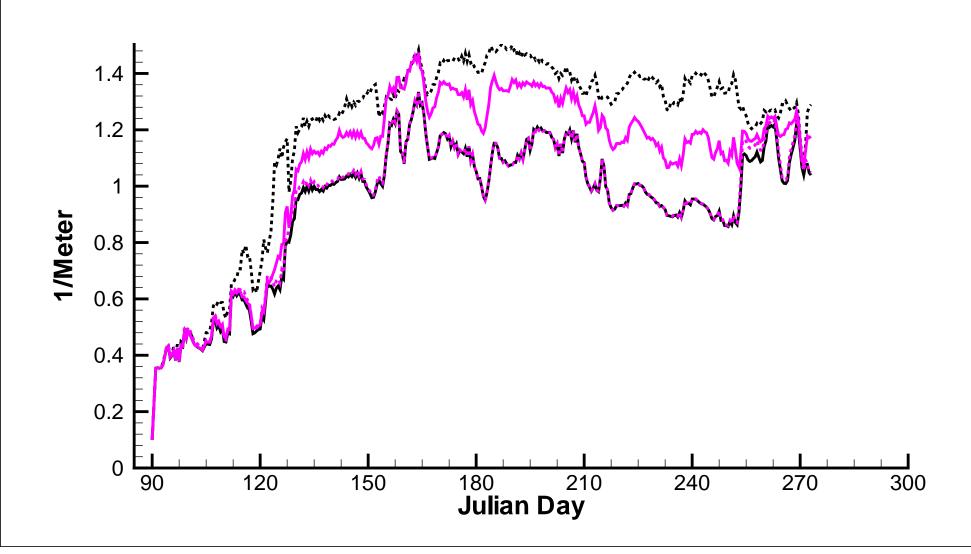






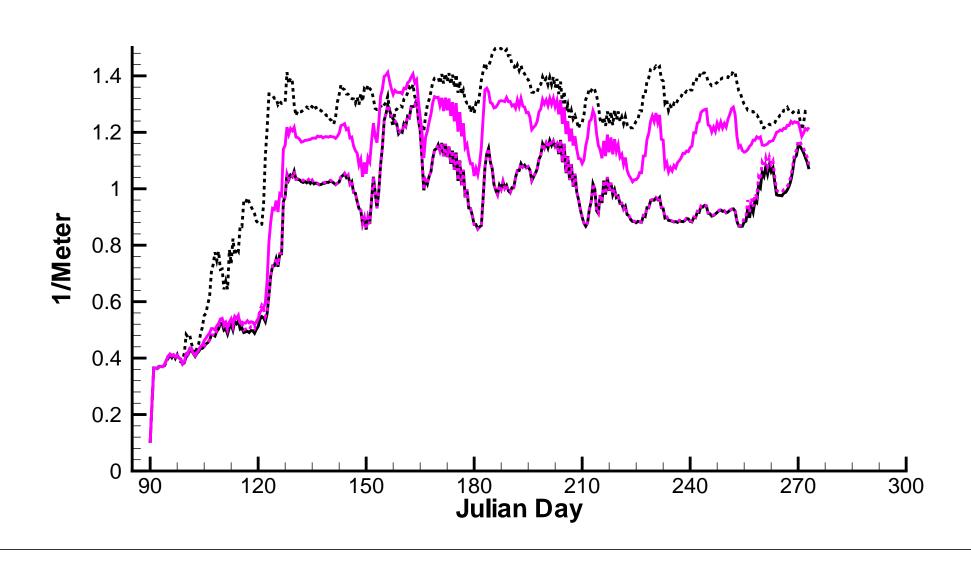






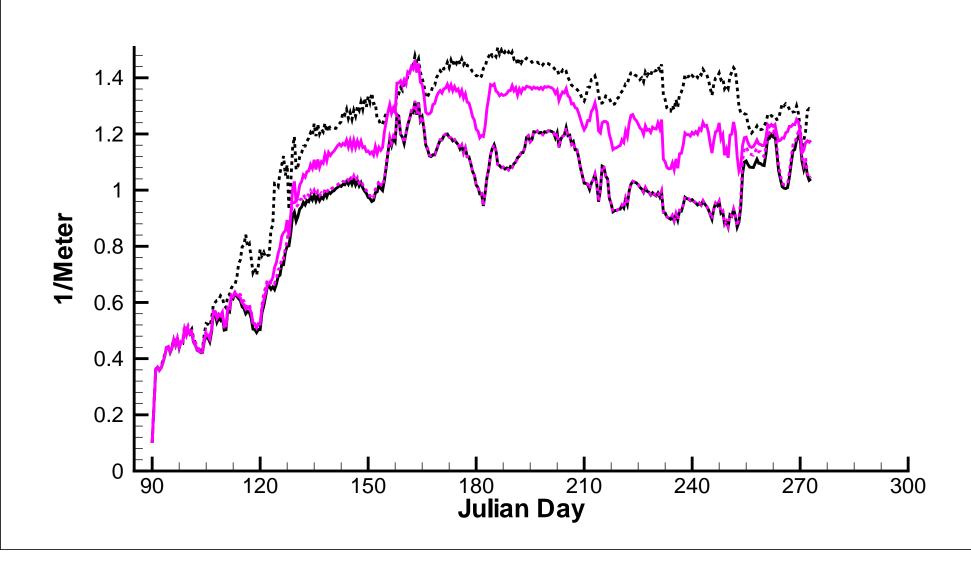






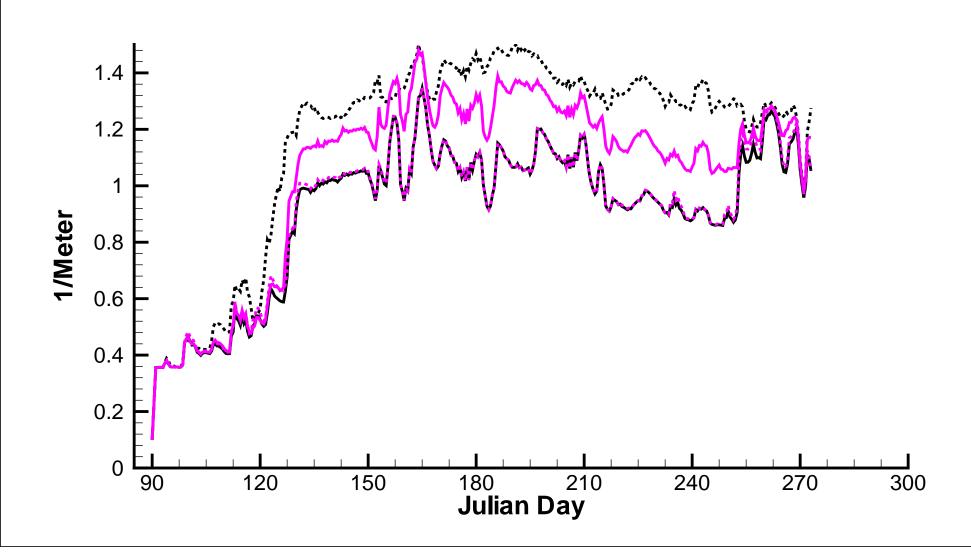






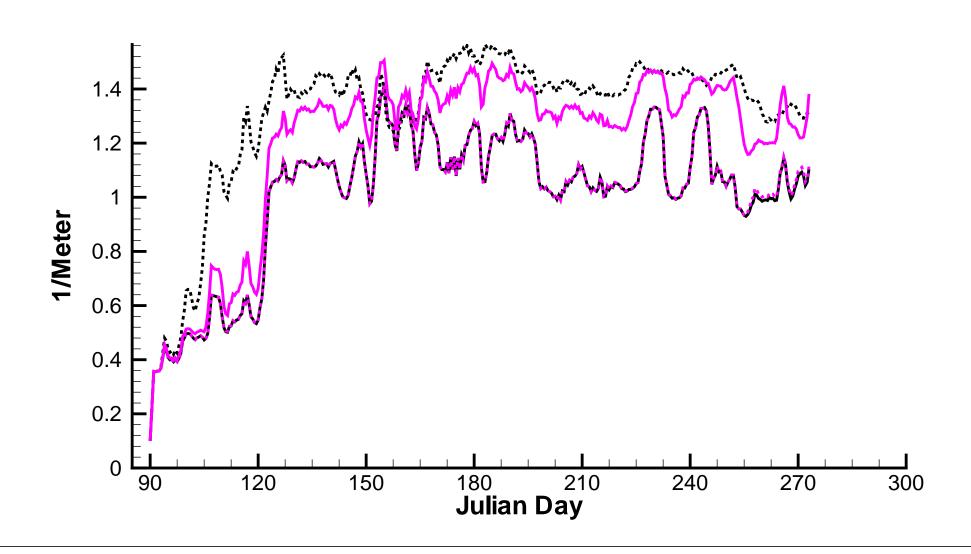


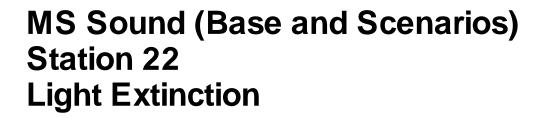




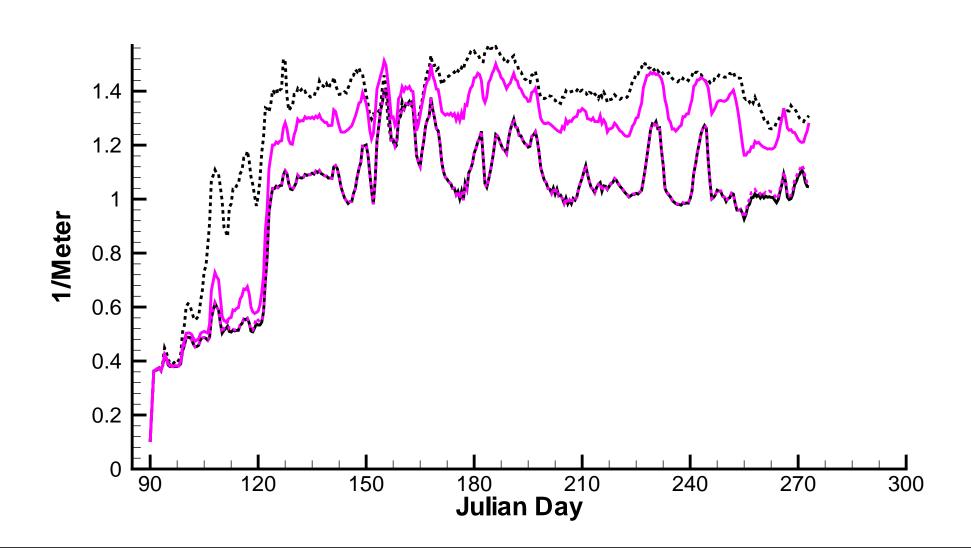


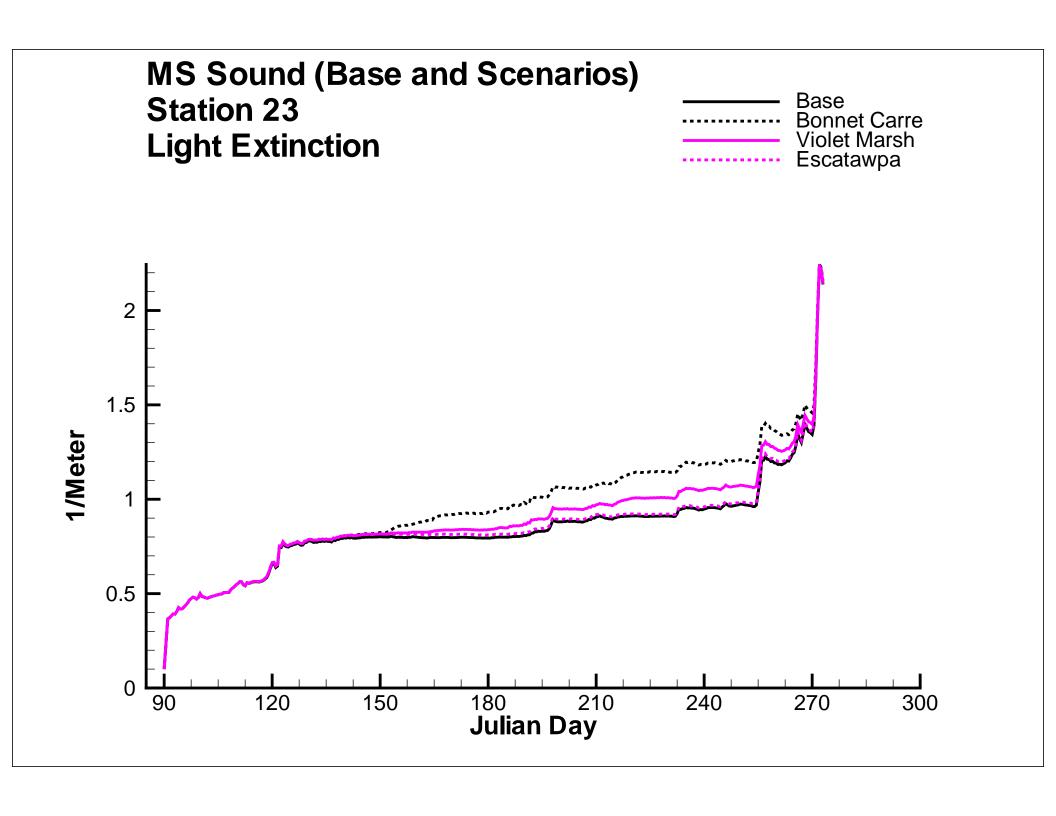






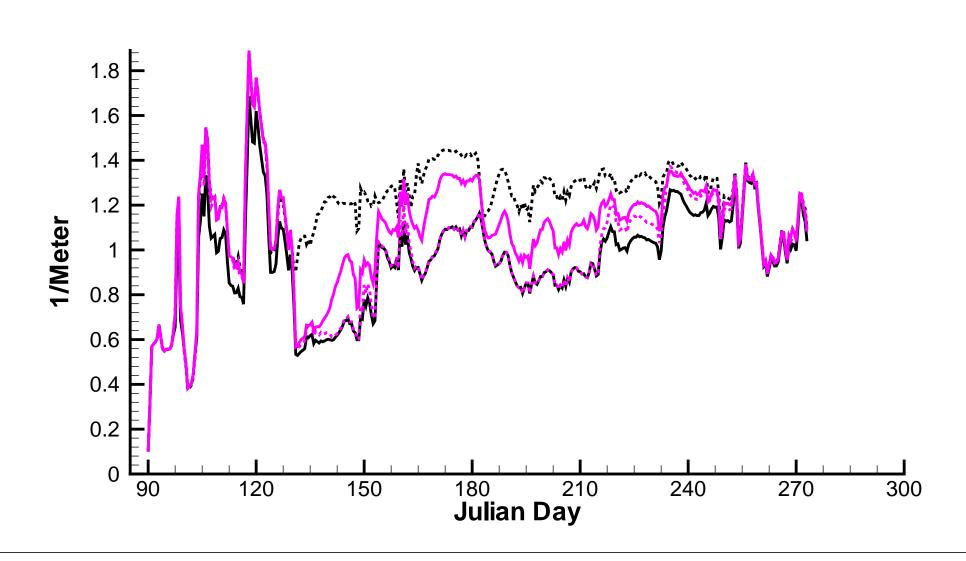






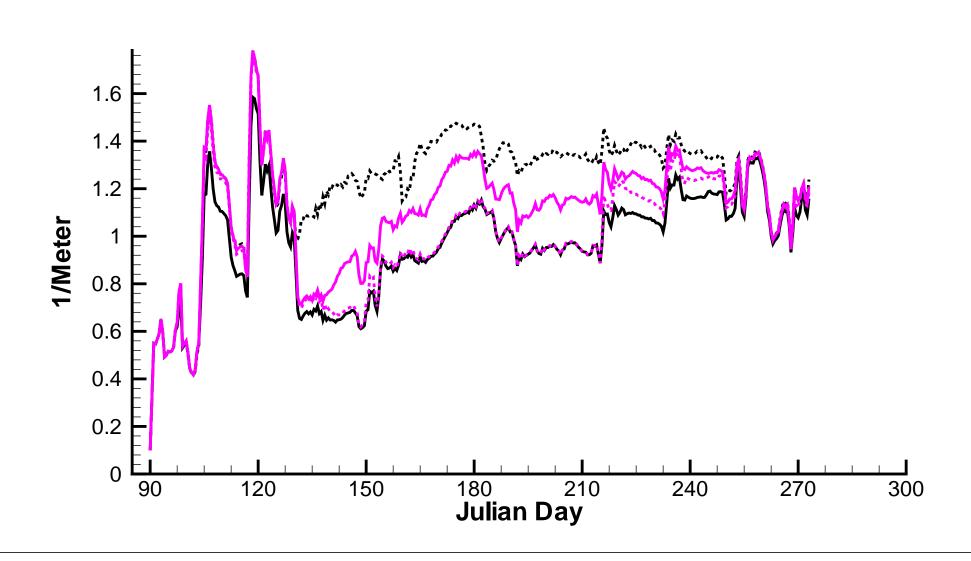






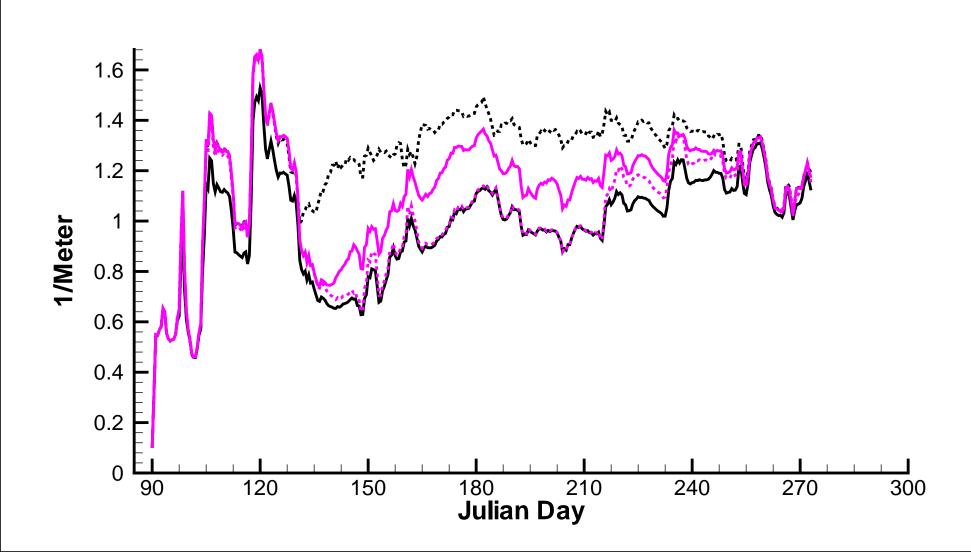






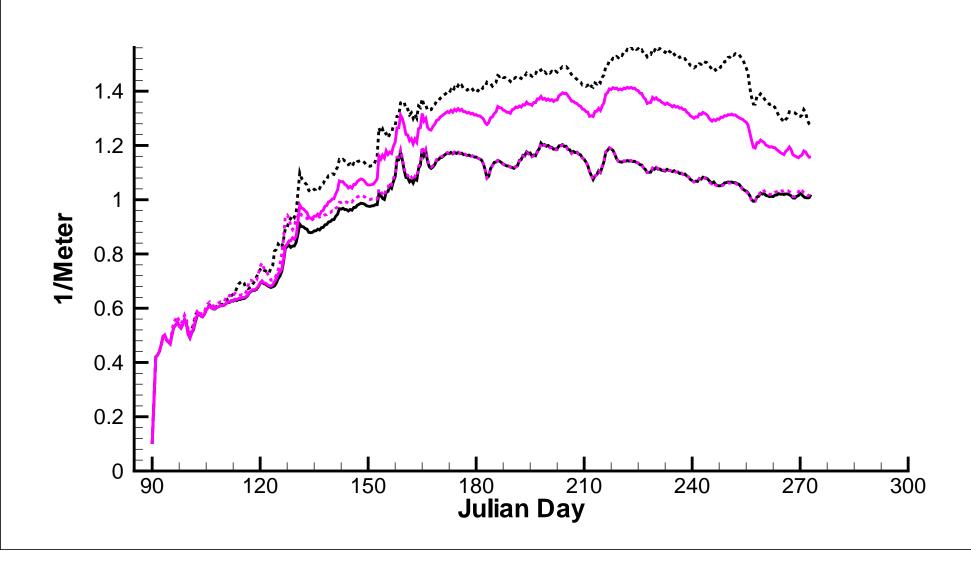


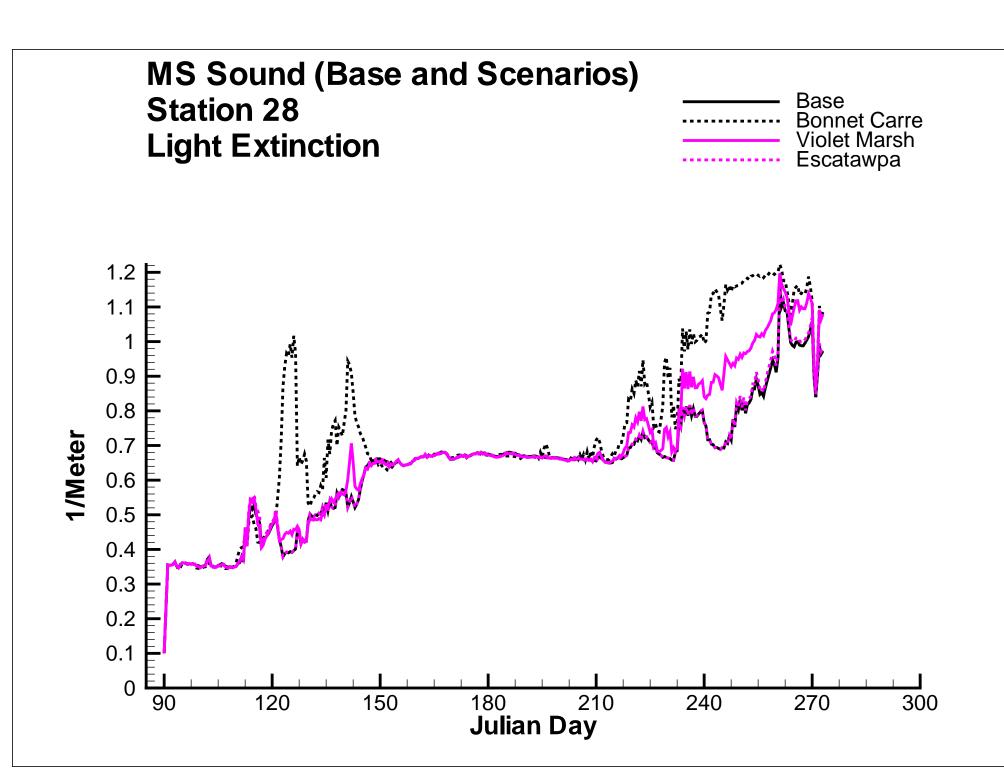


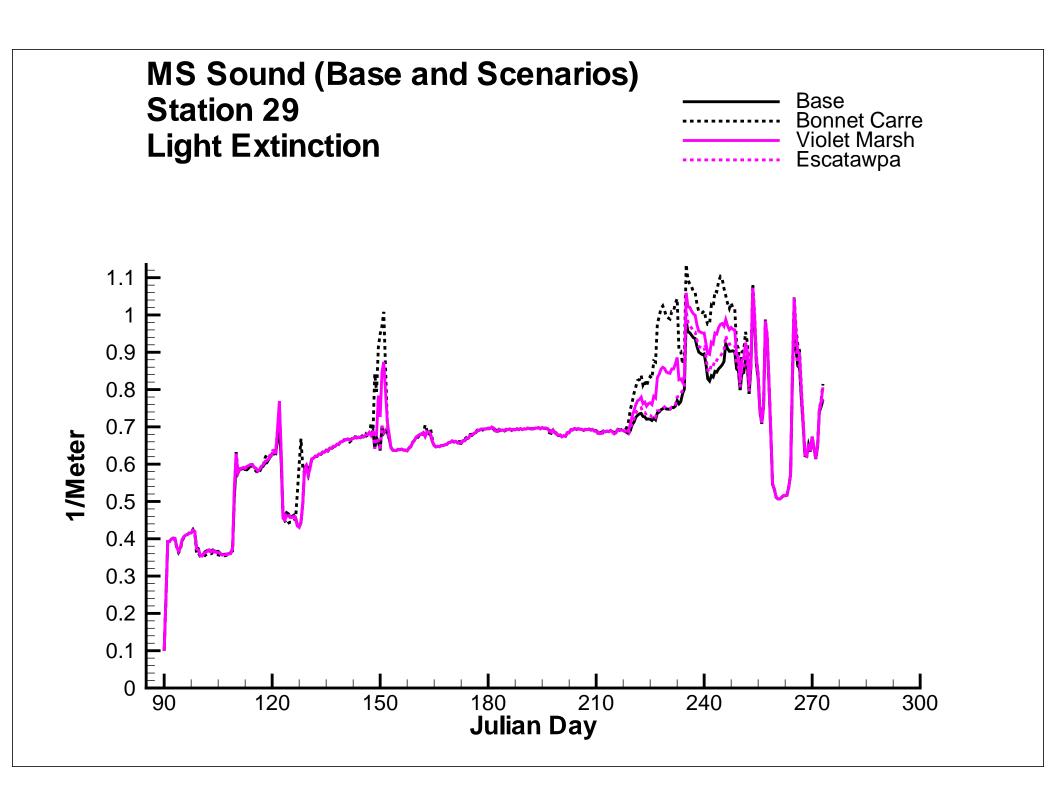


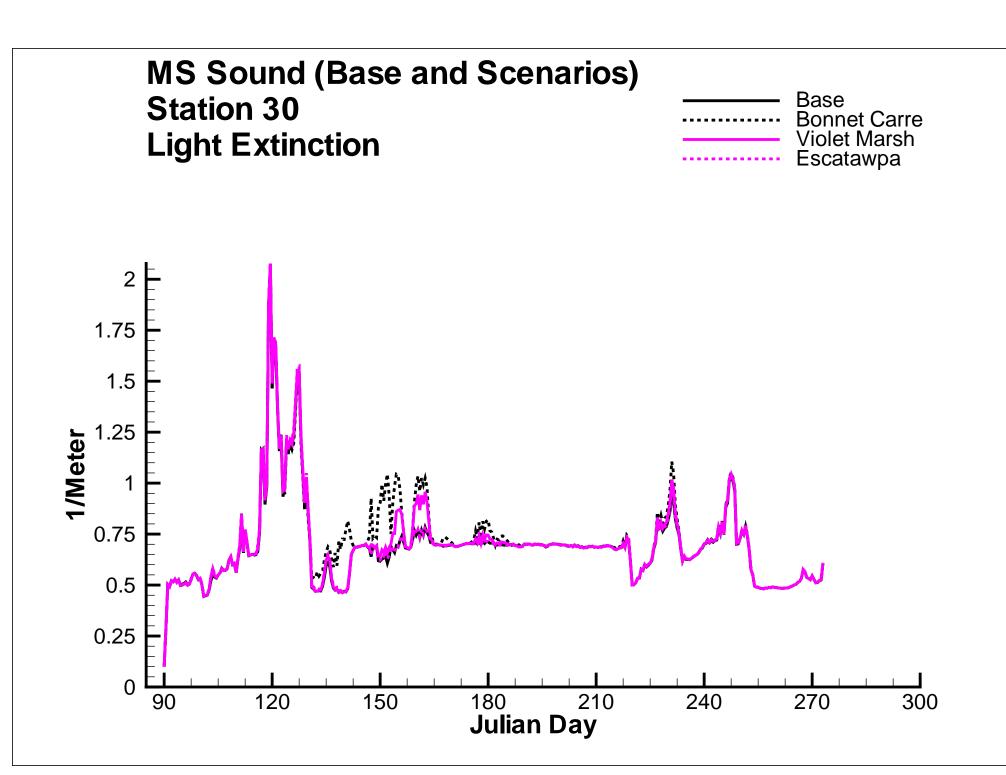






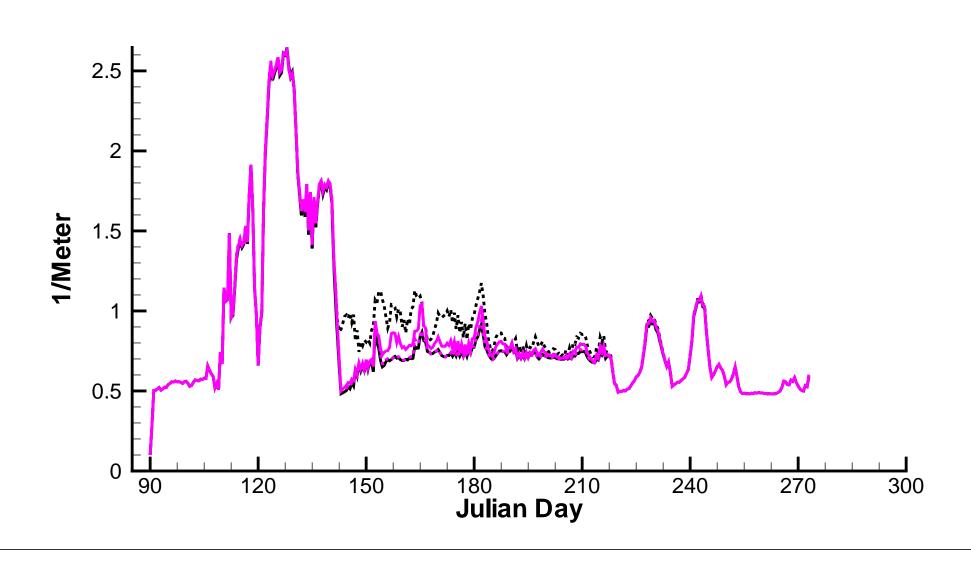


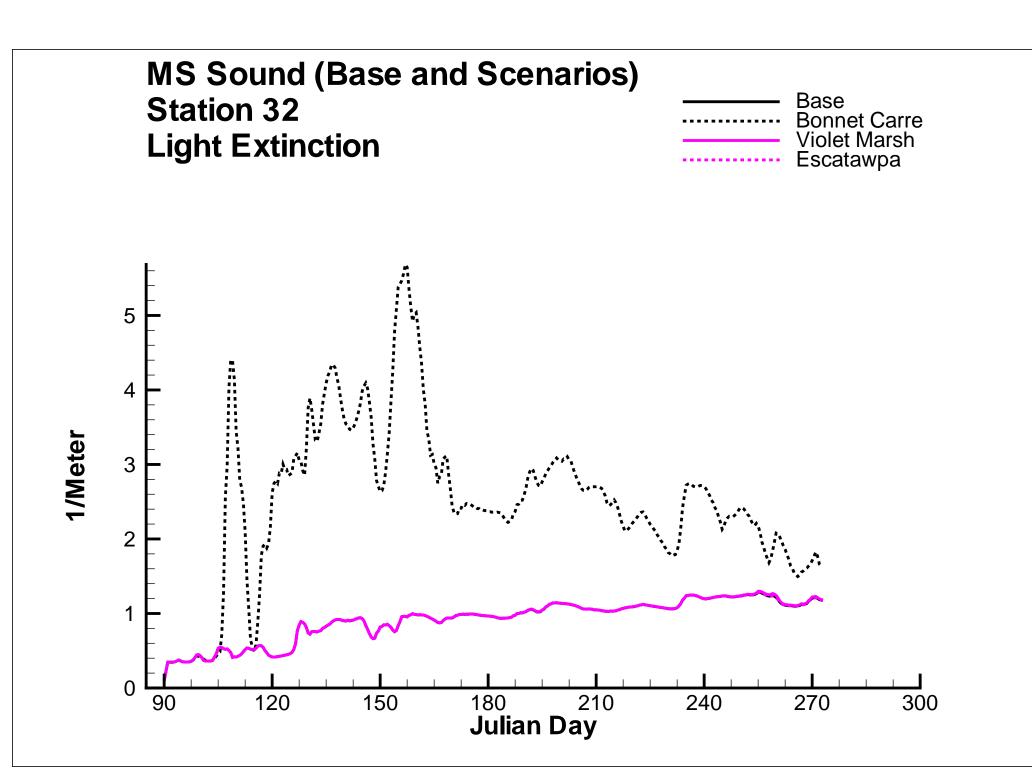




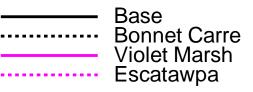


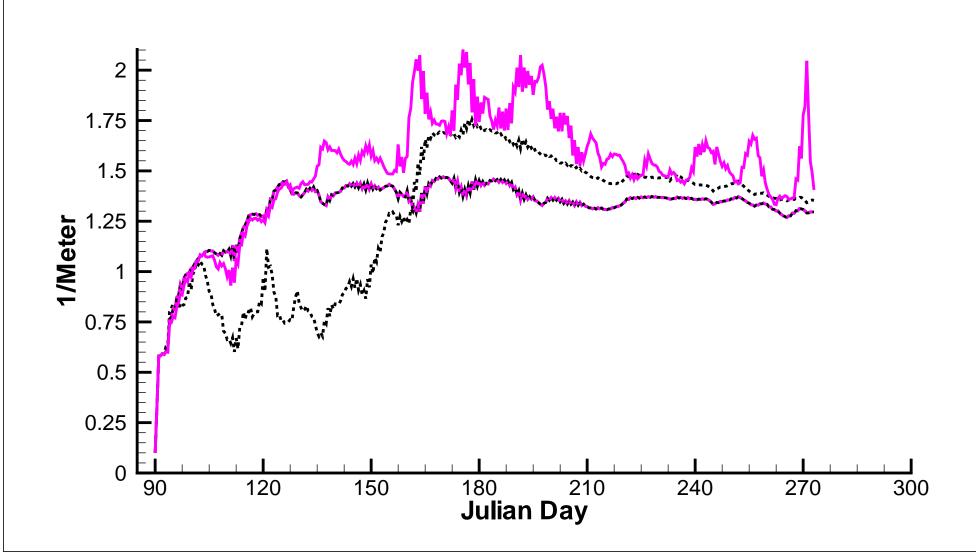




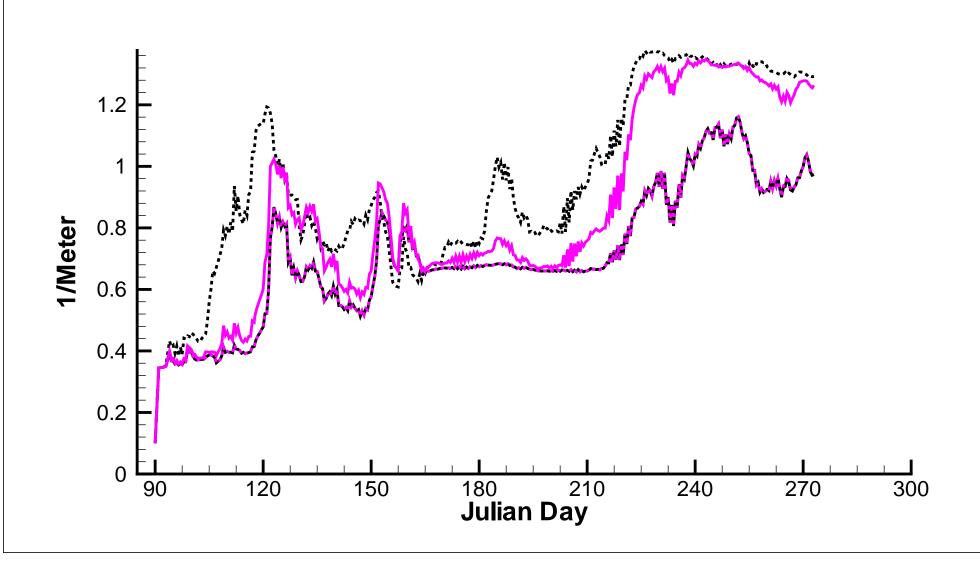




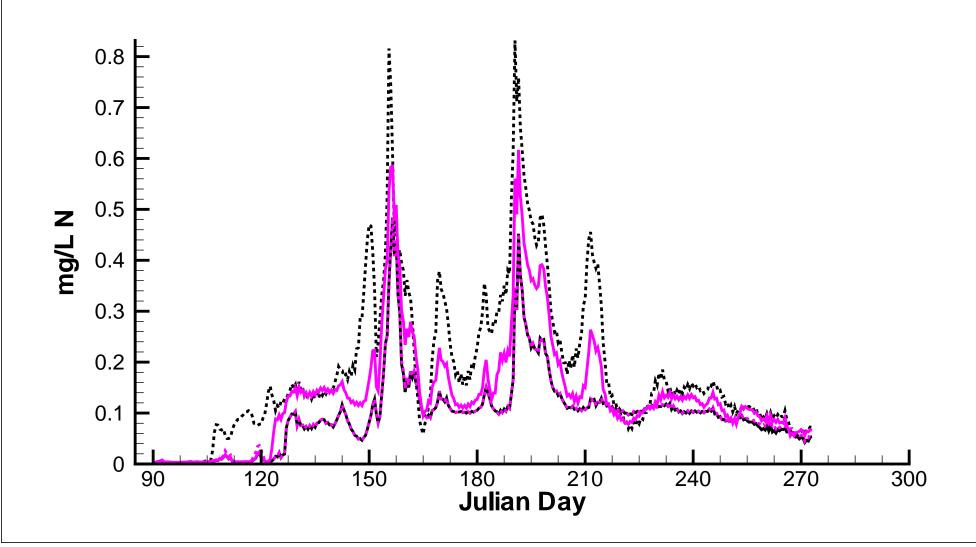




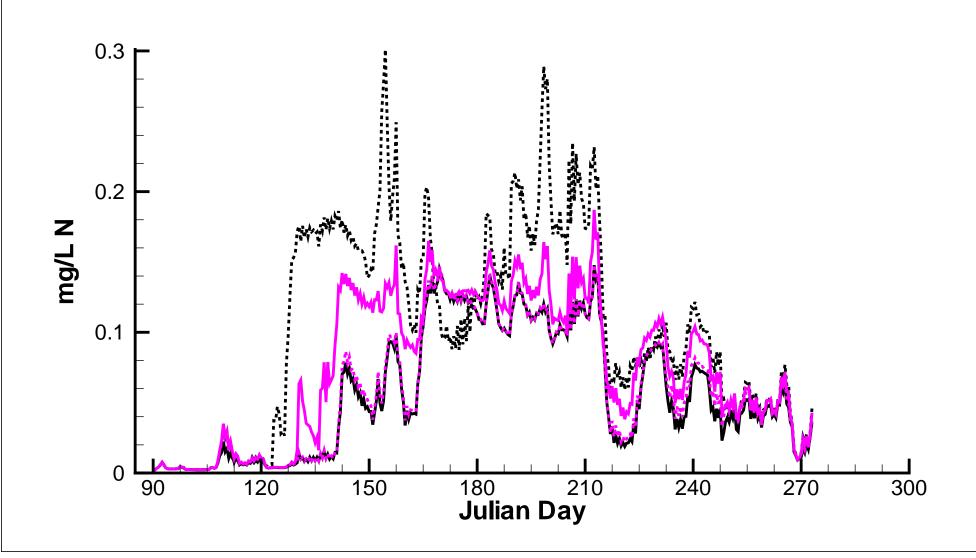




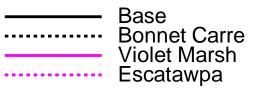


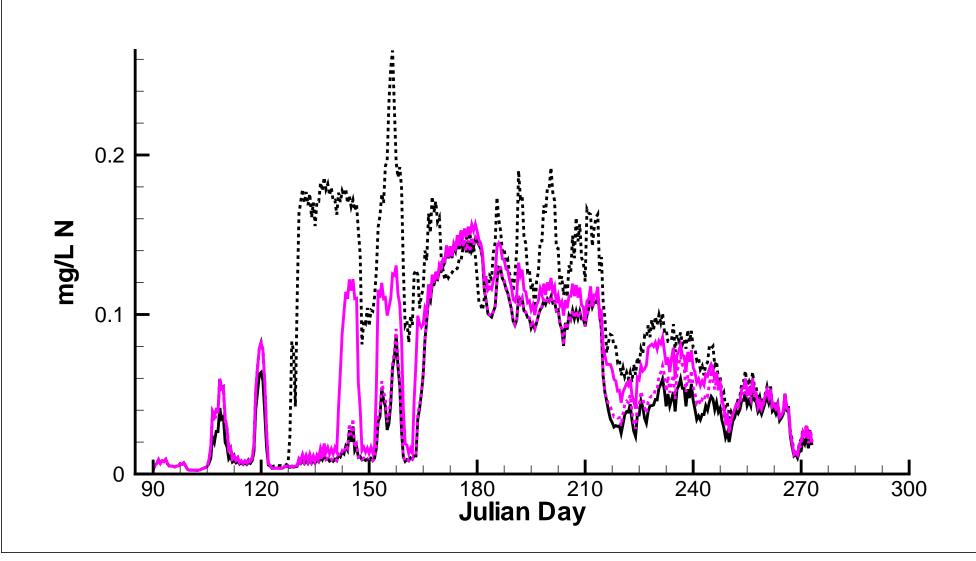






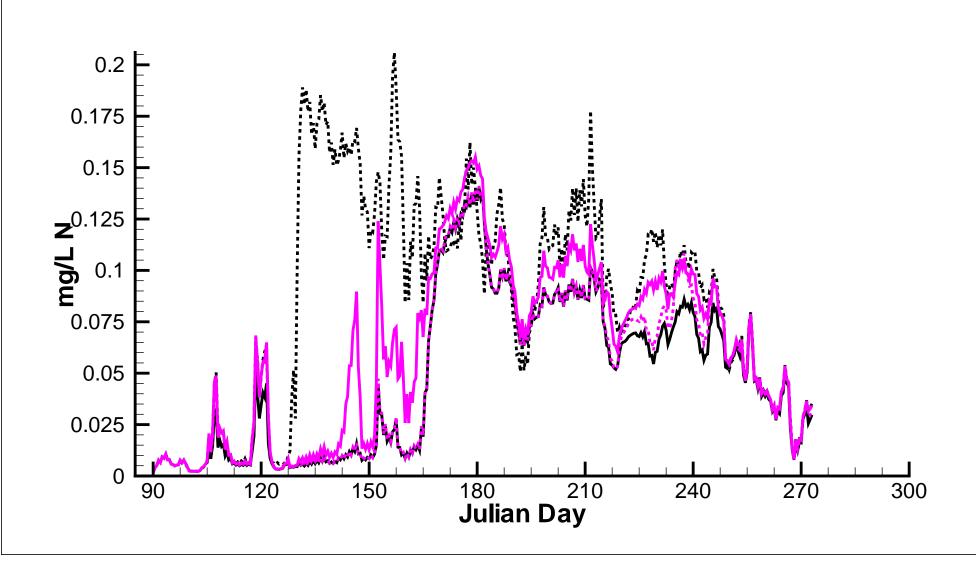


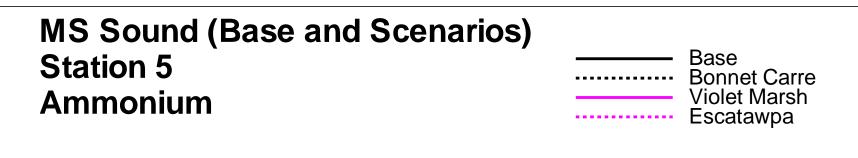


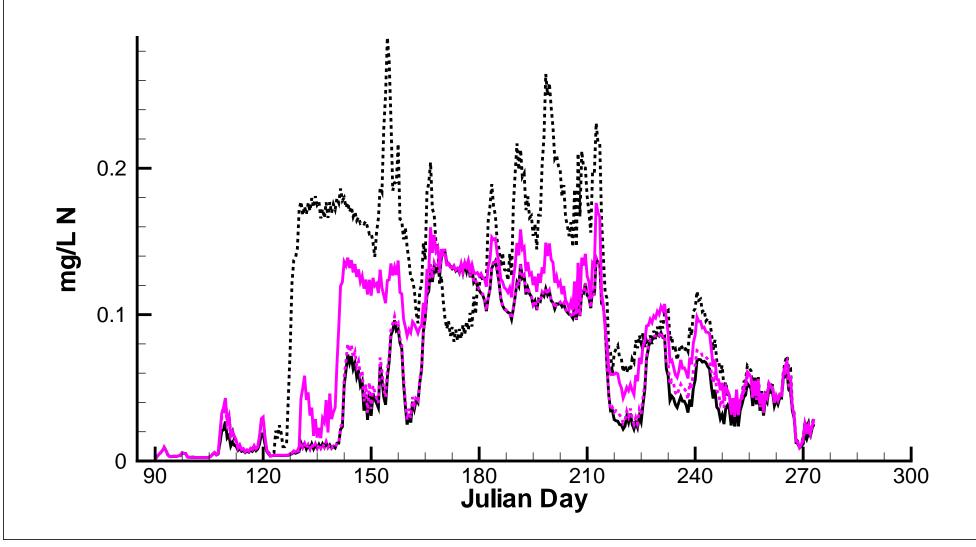




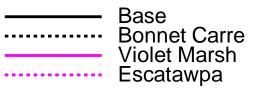


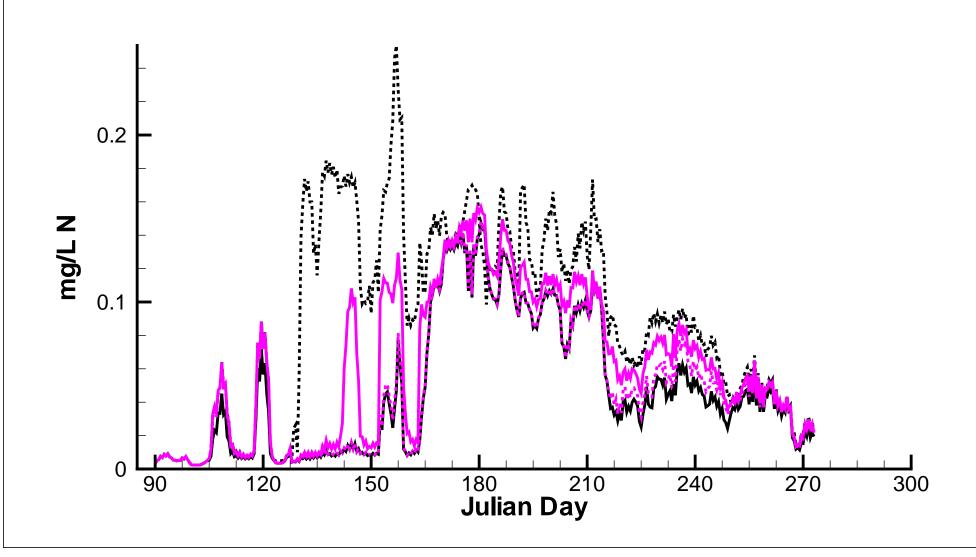






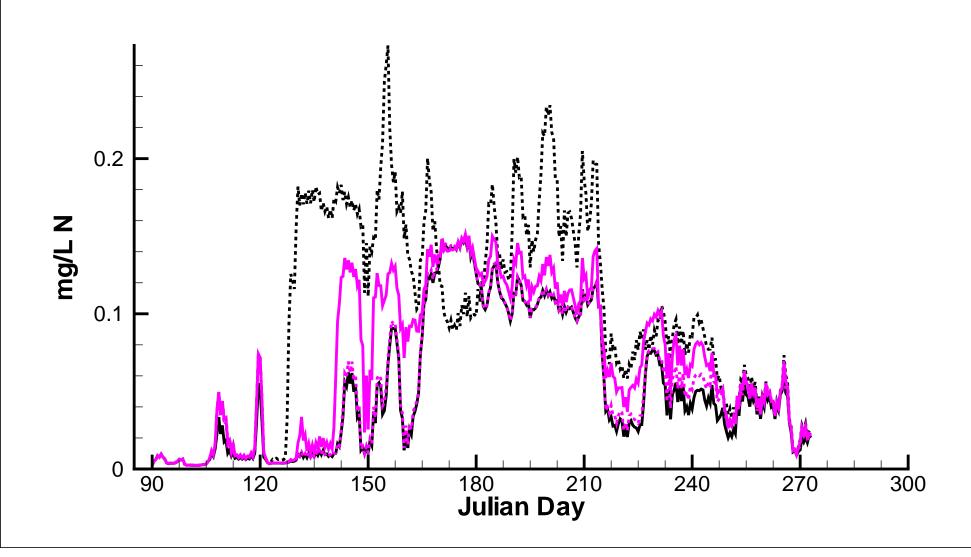






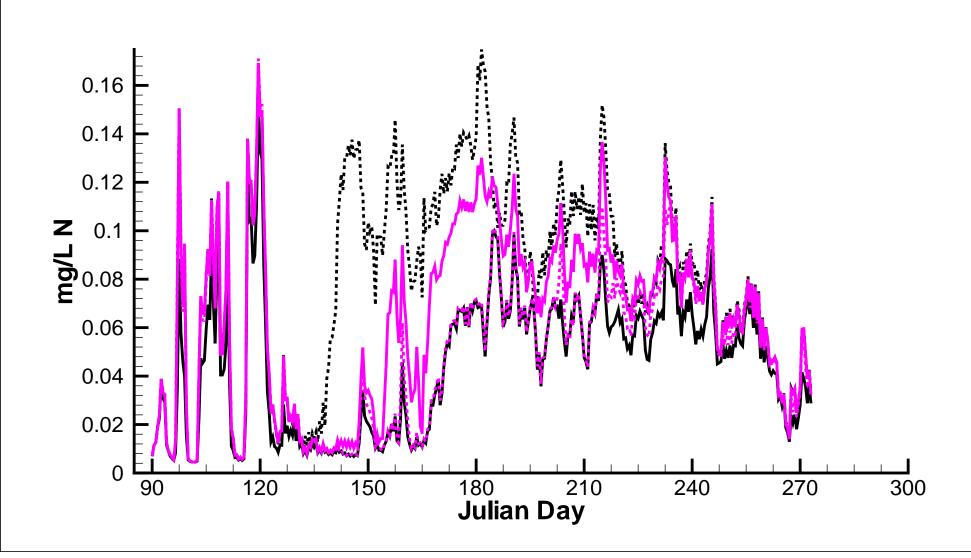




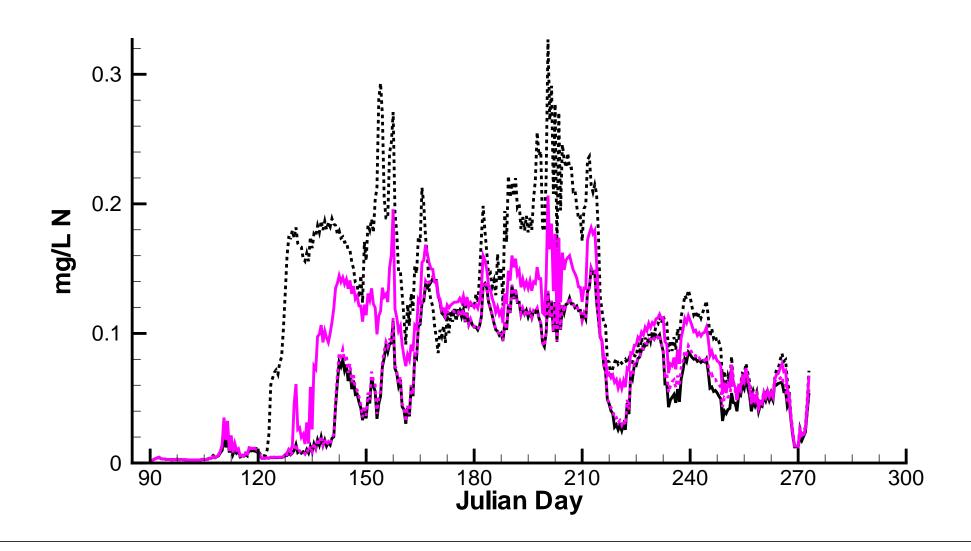


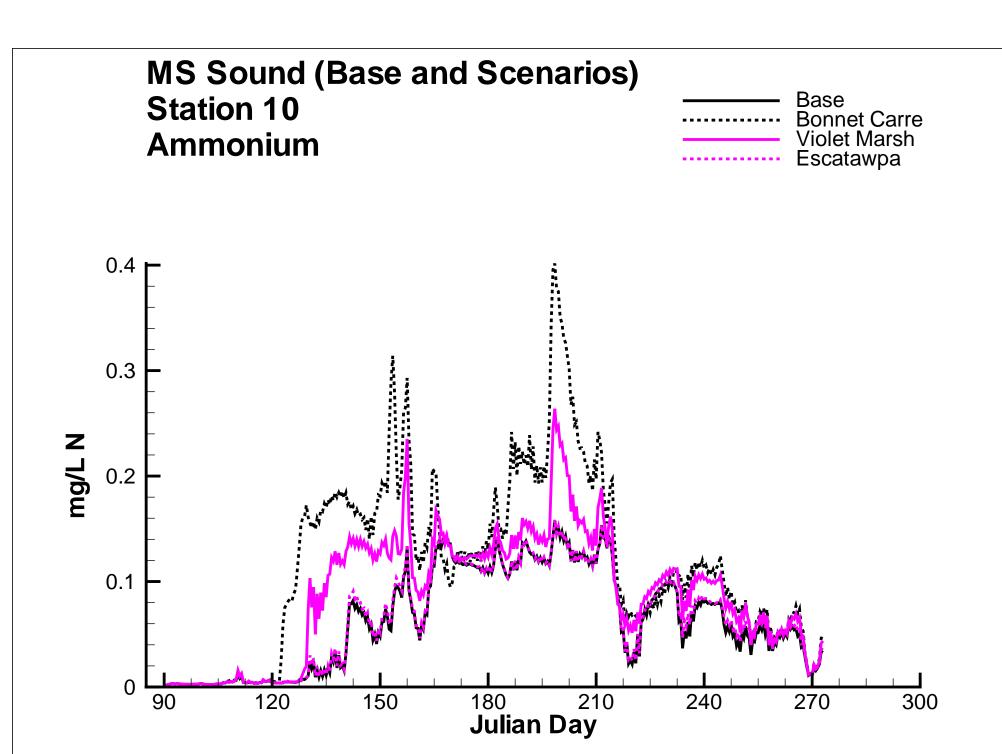




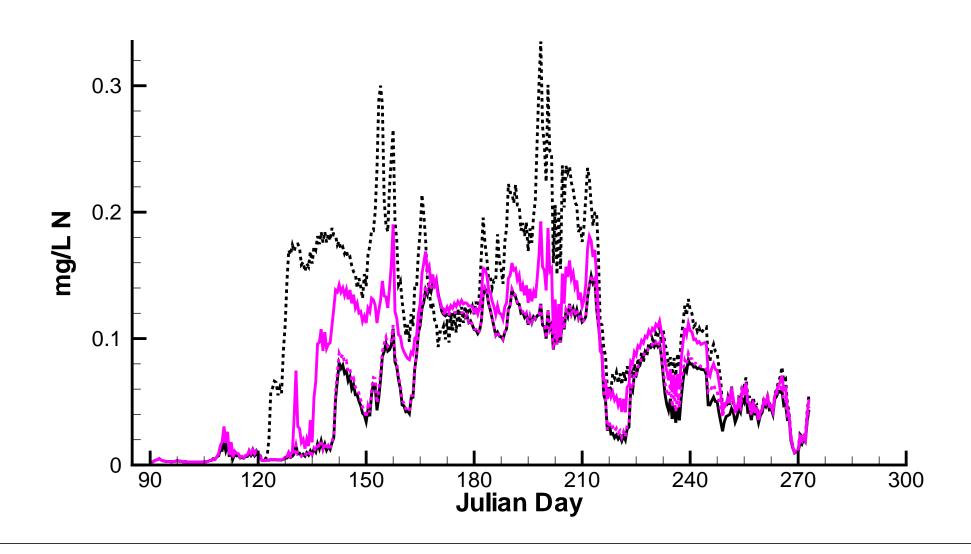




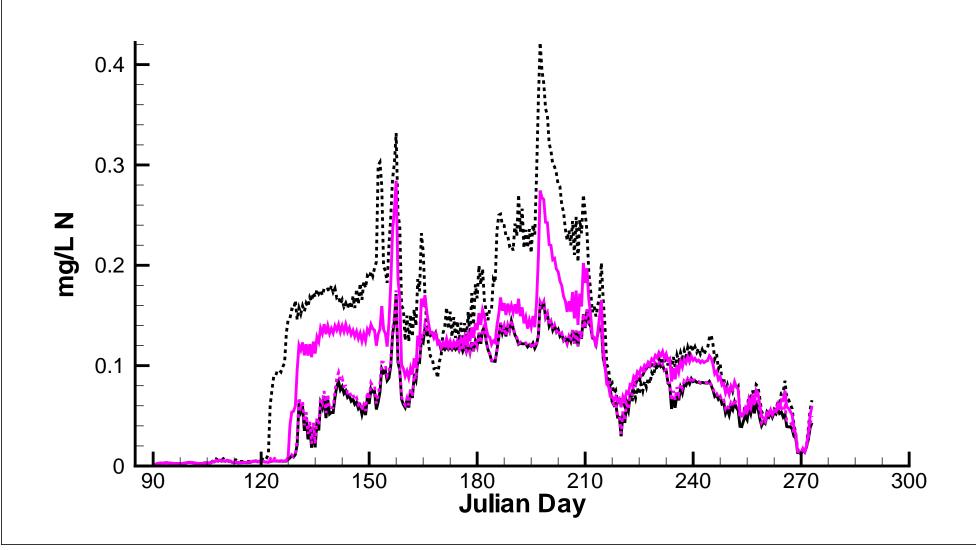






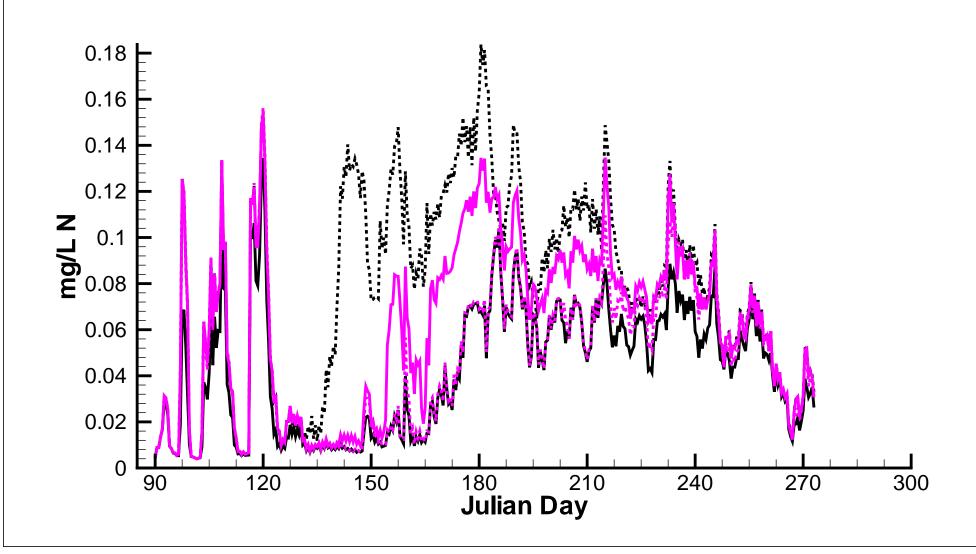






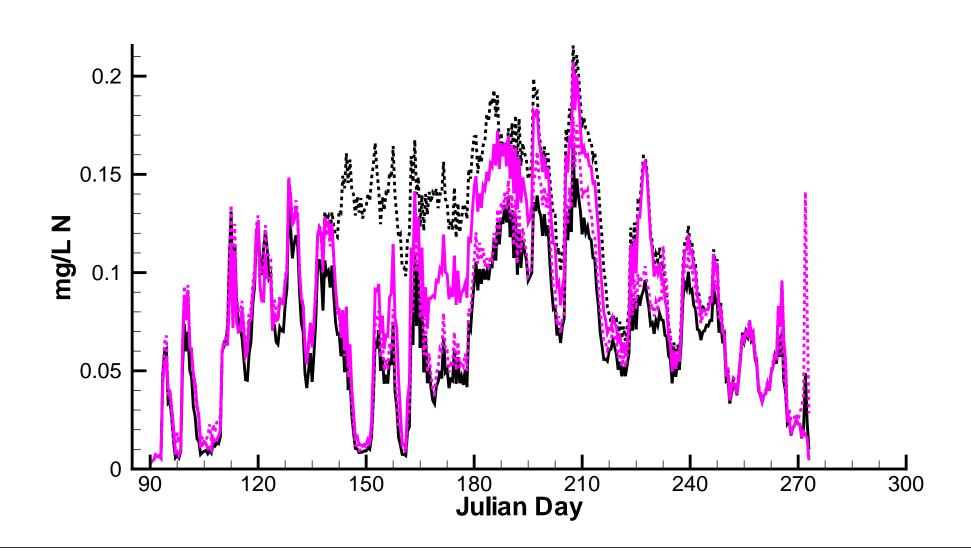






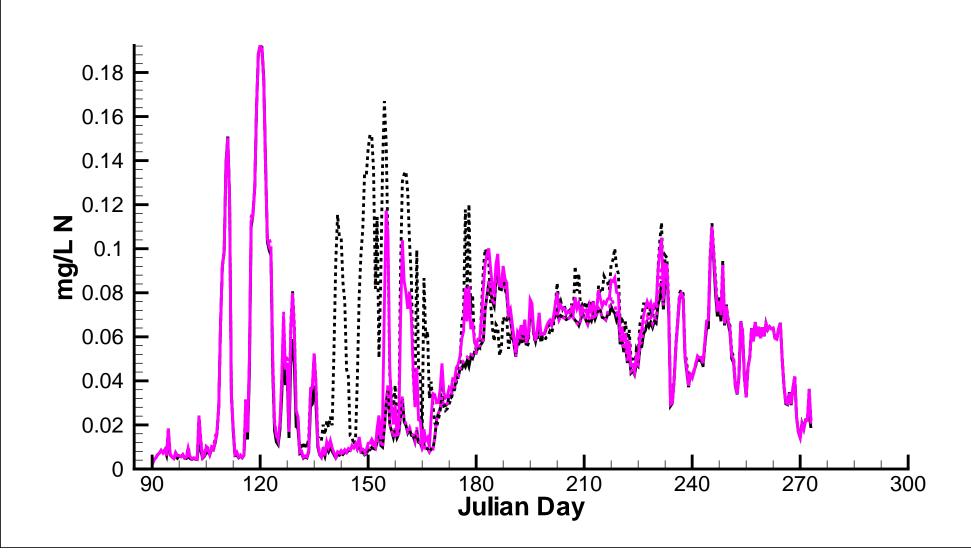






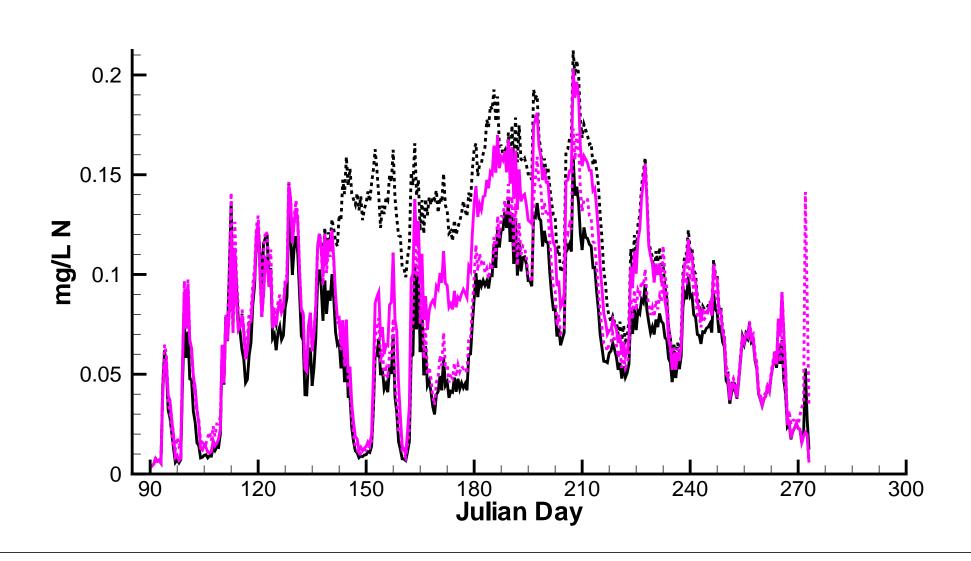






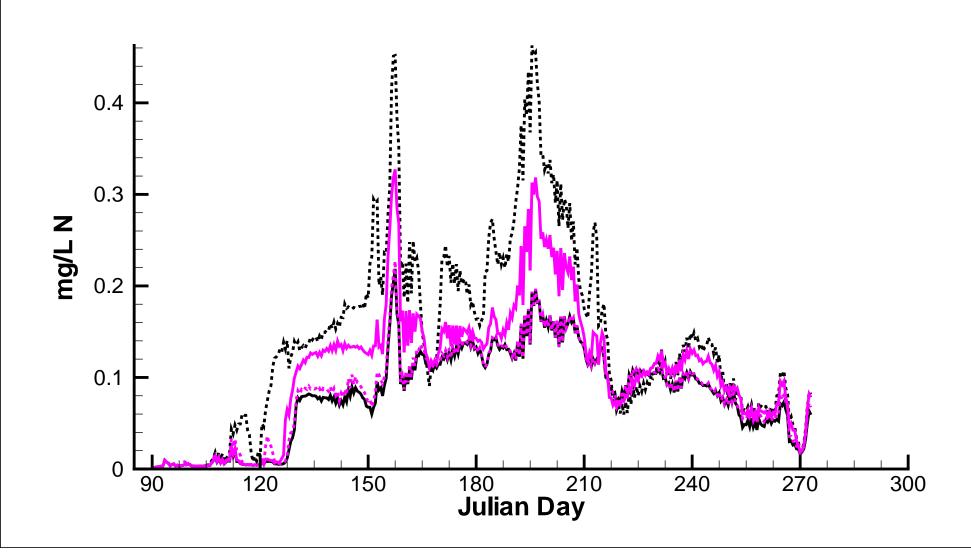


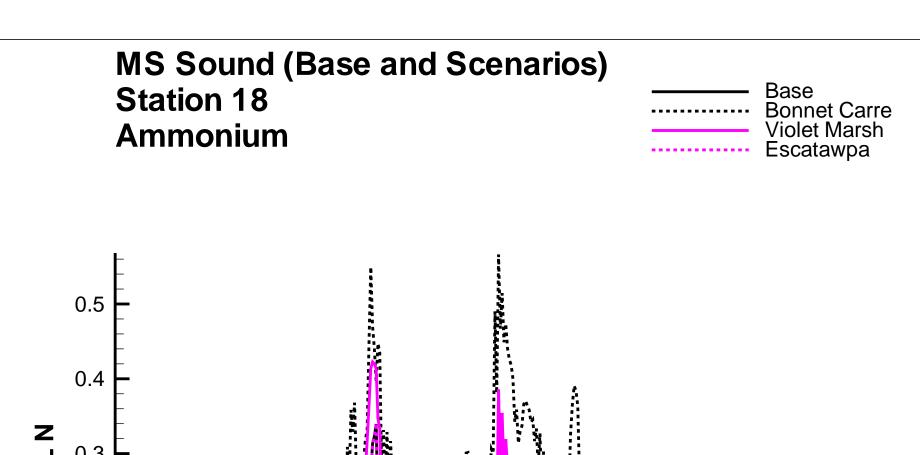


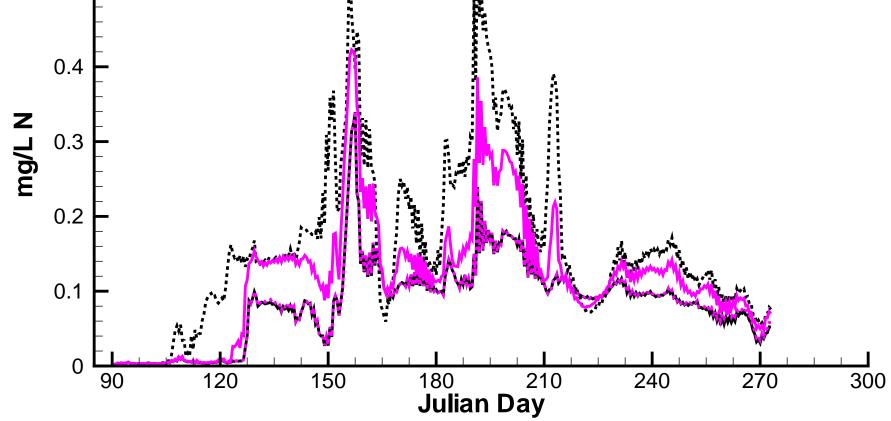




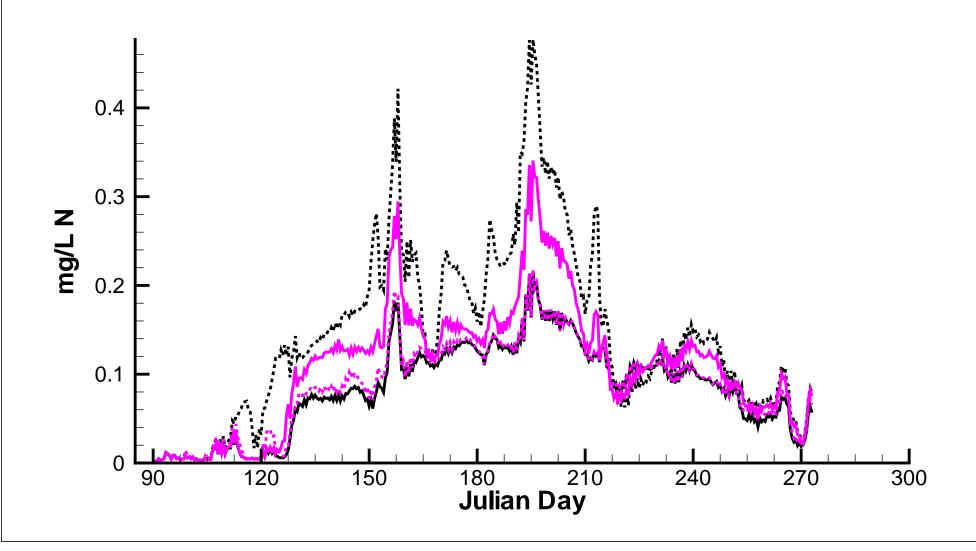




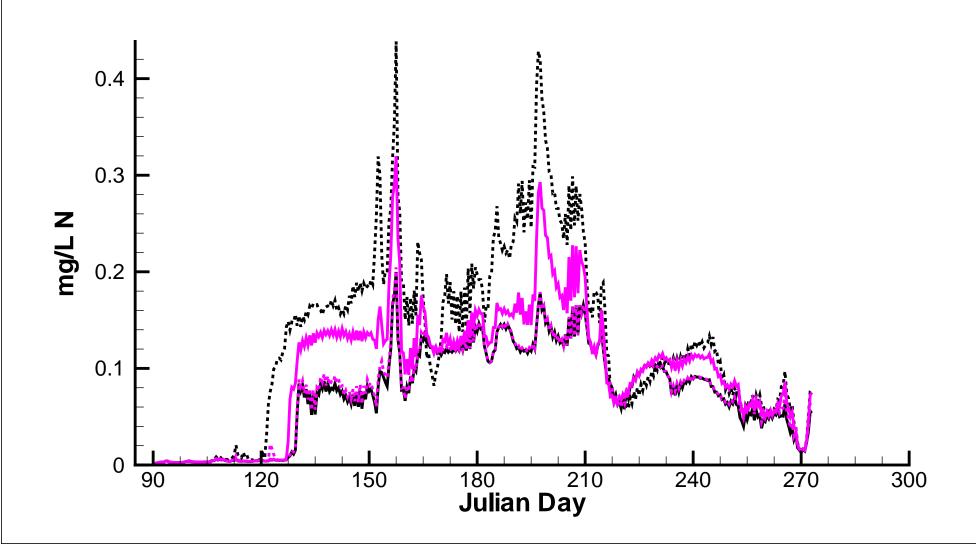






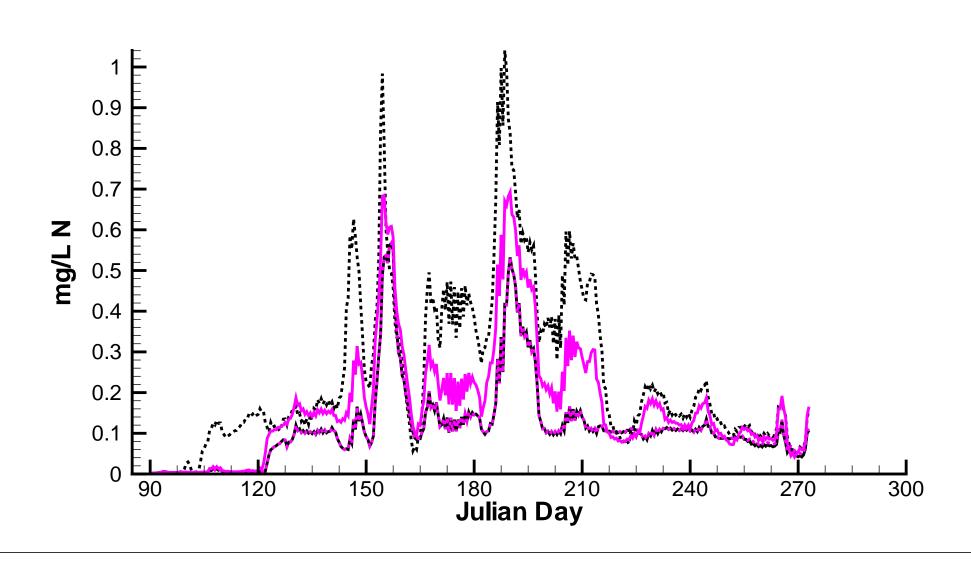






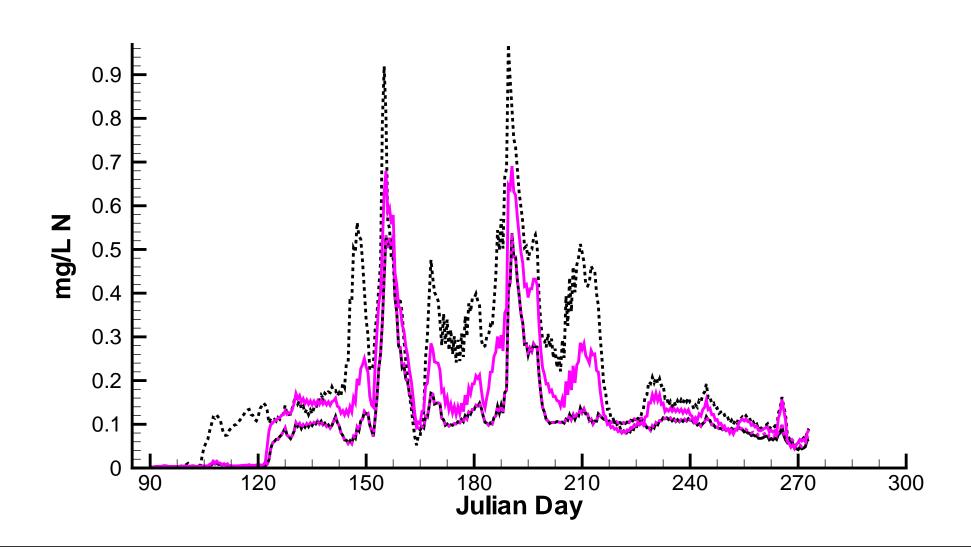


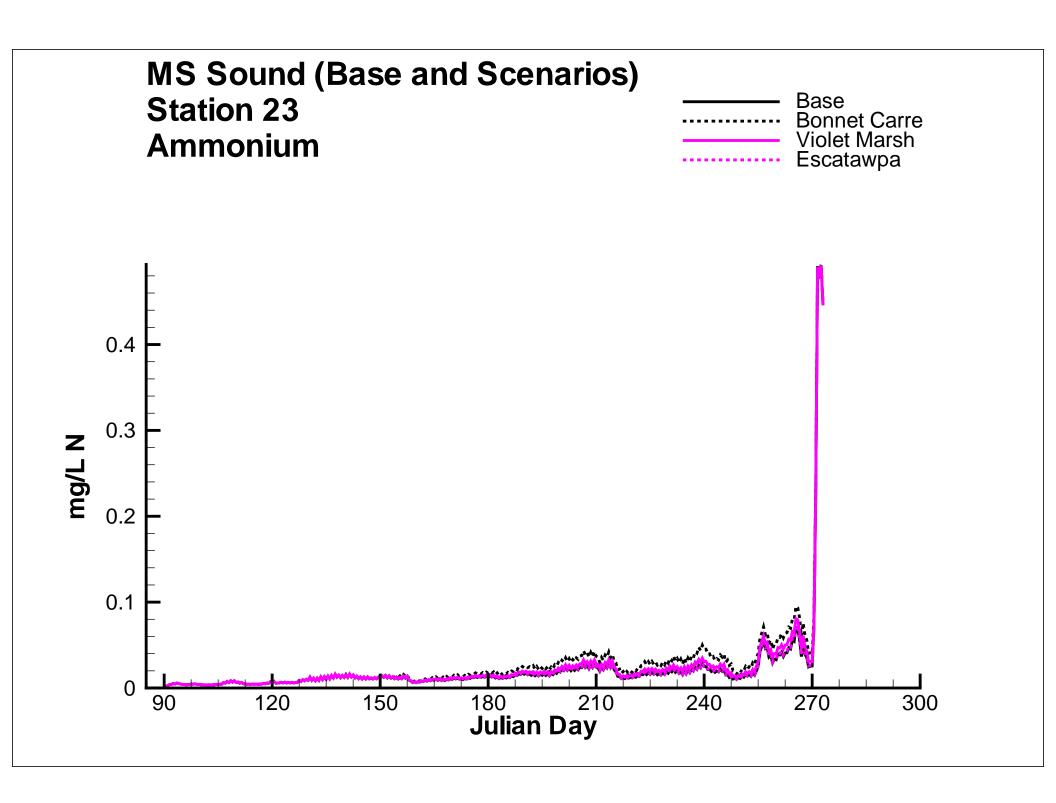






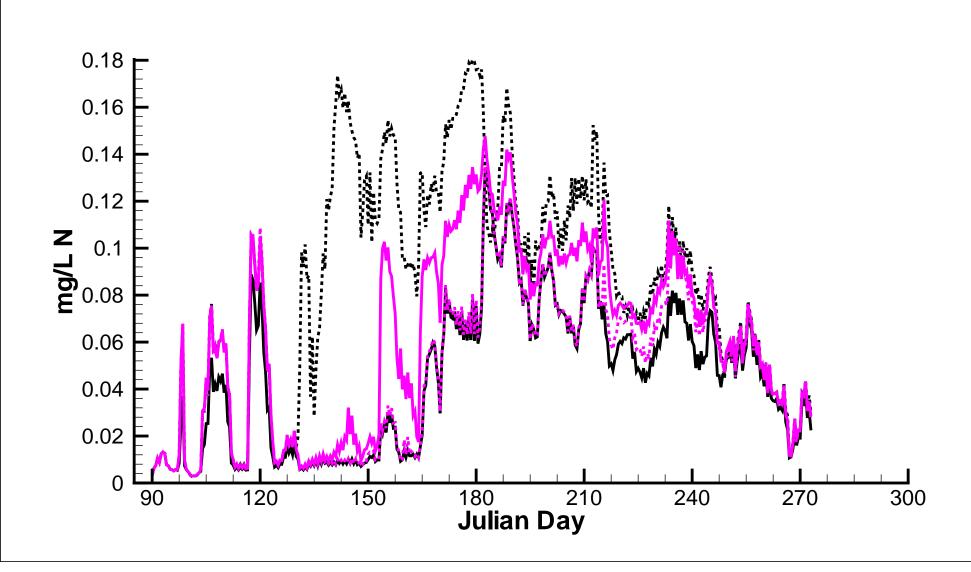




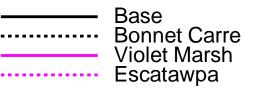


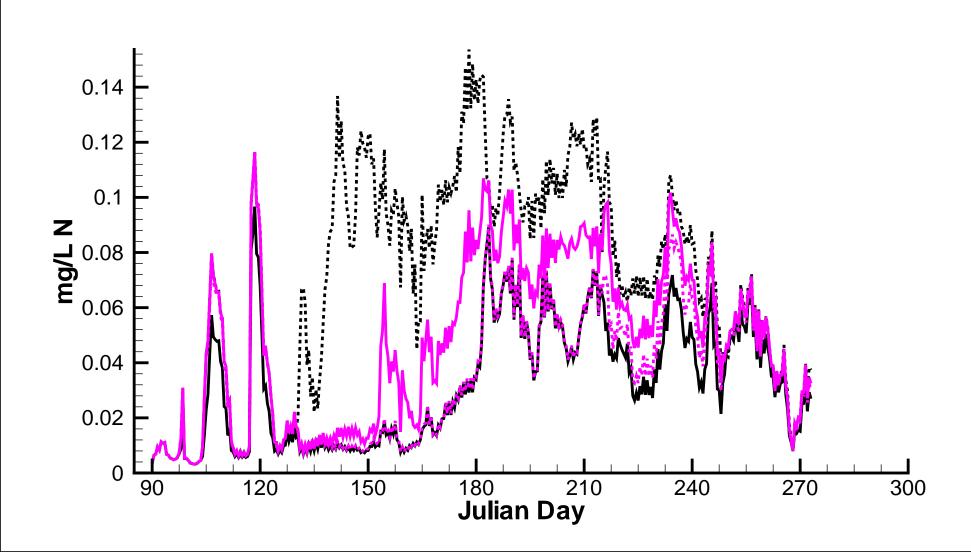






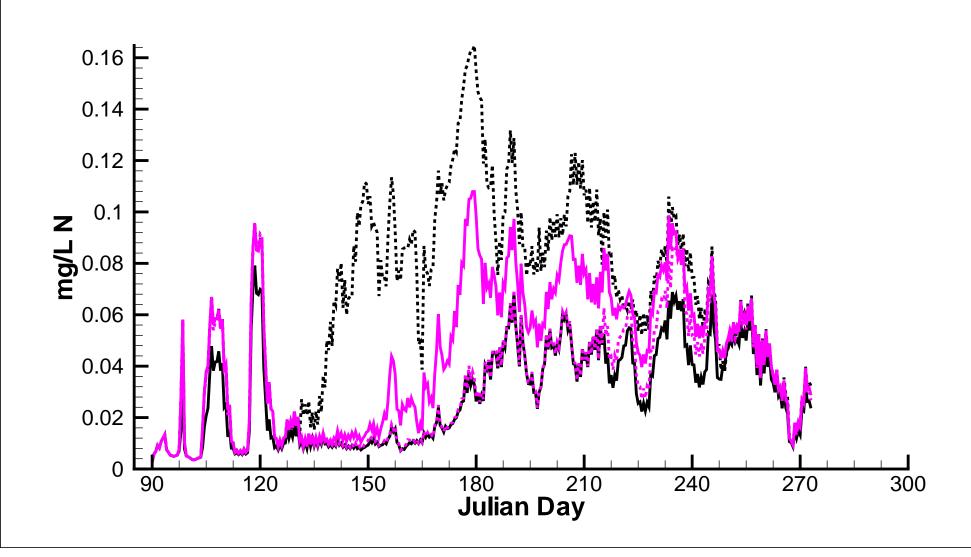






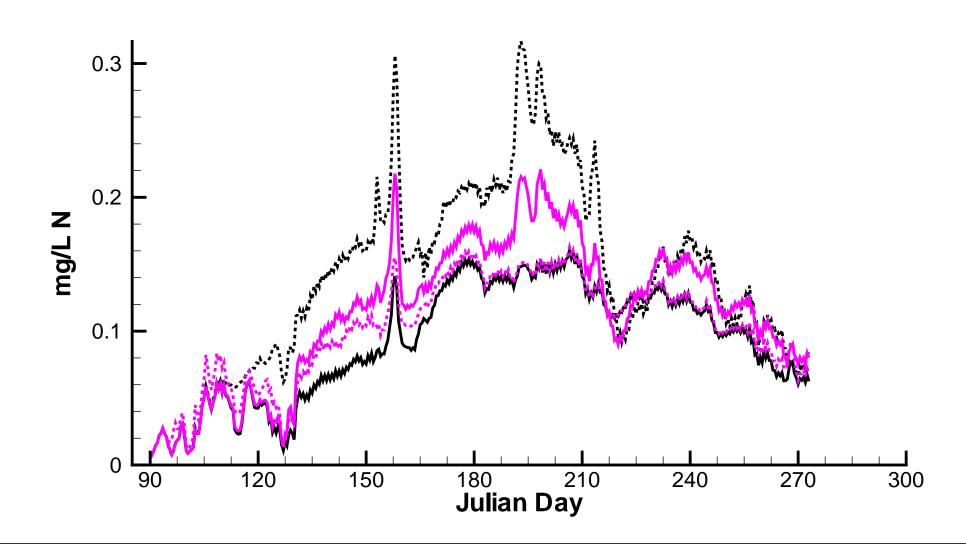




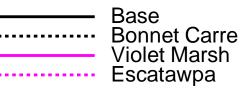


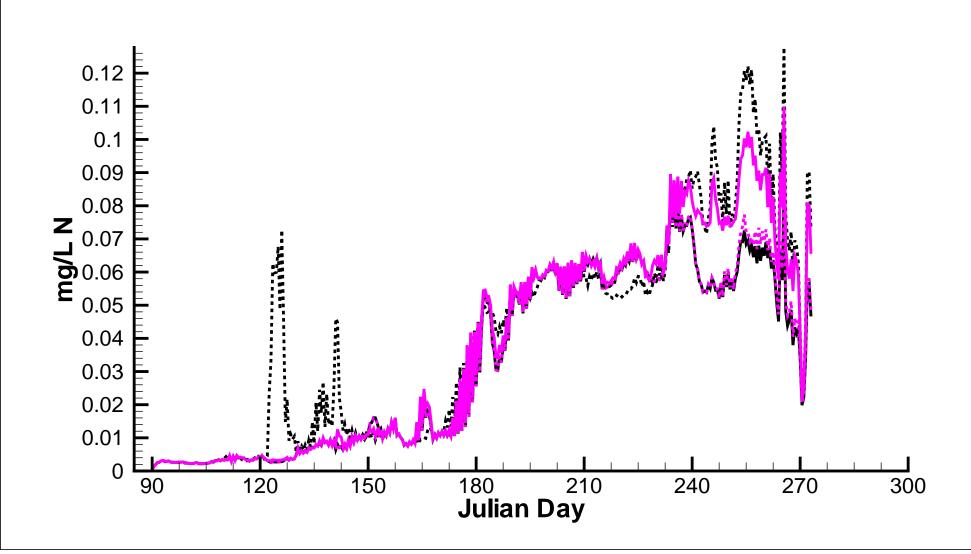


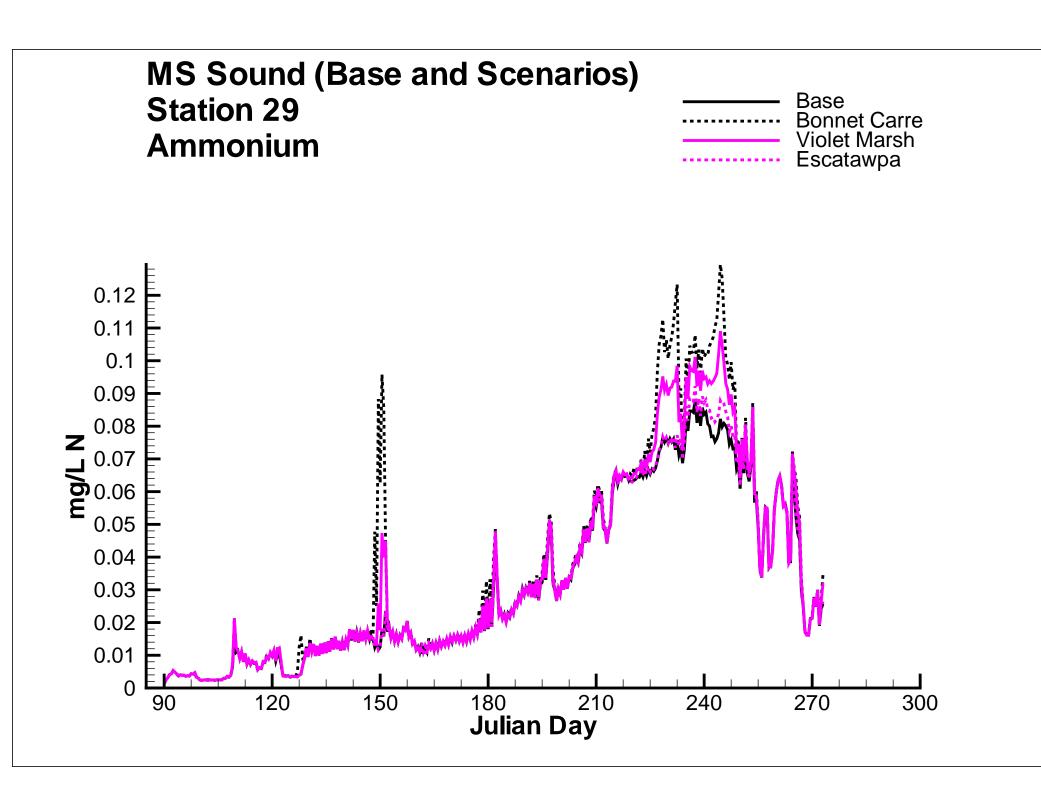






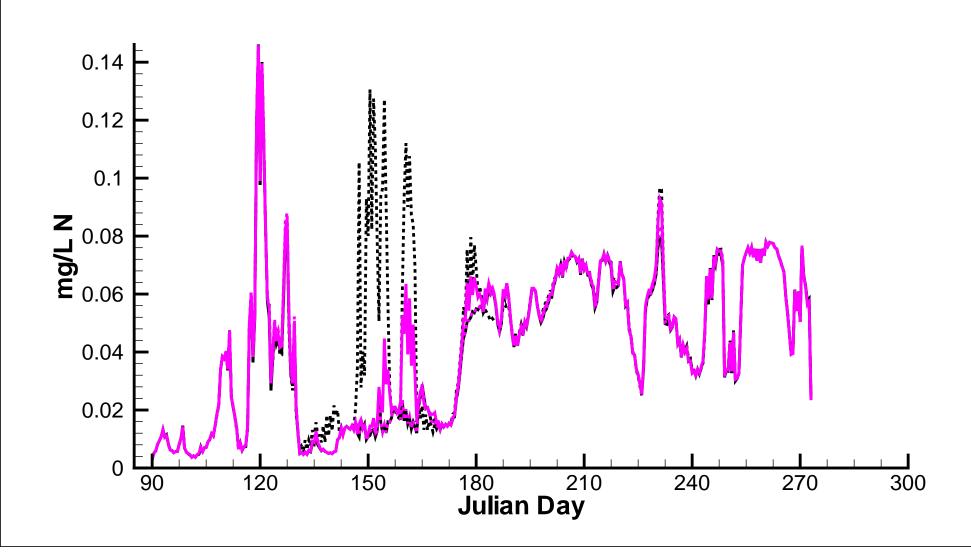






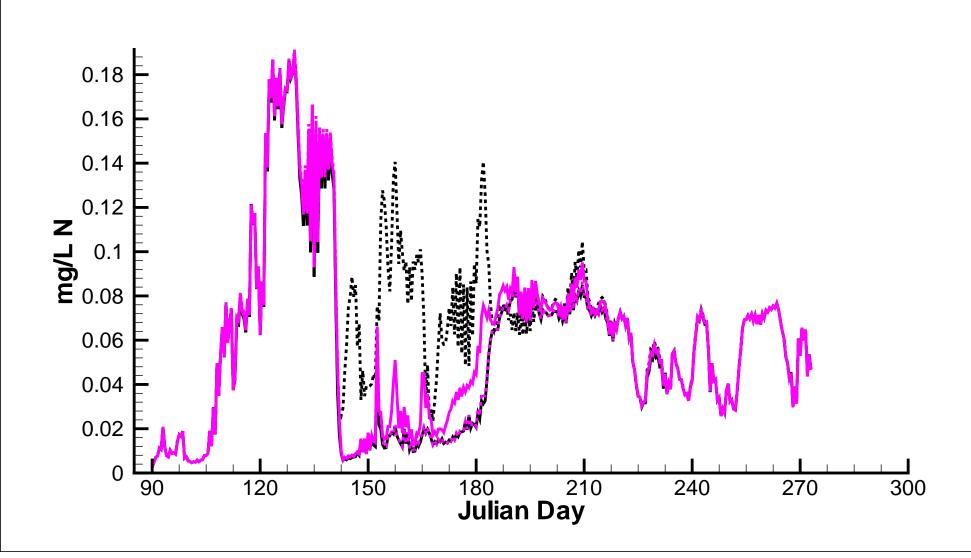








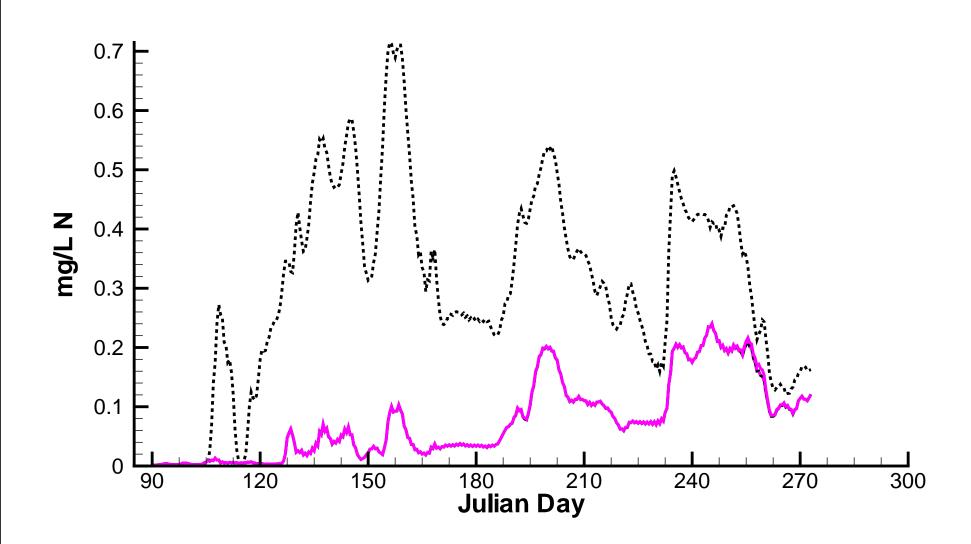






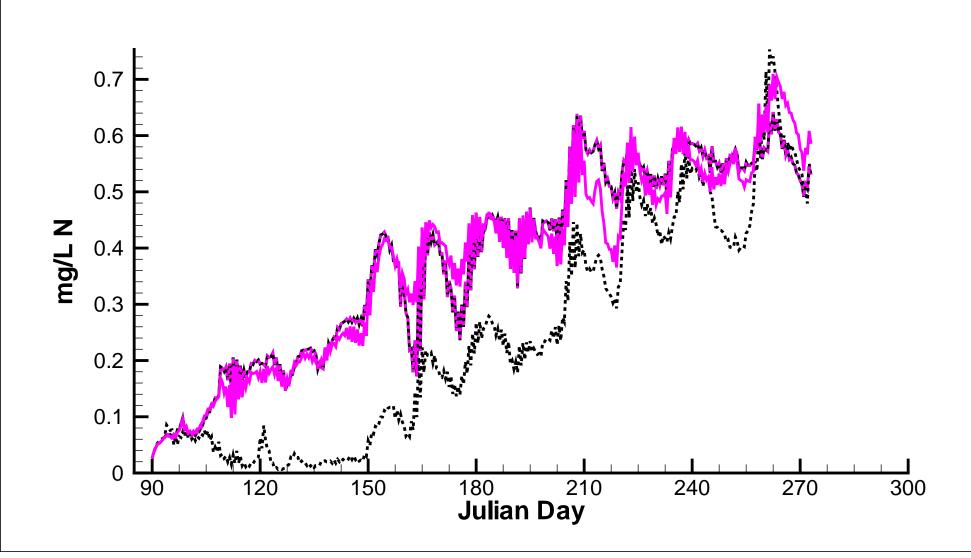
Base Bonnet Carre Violet Marsh

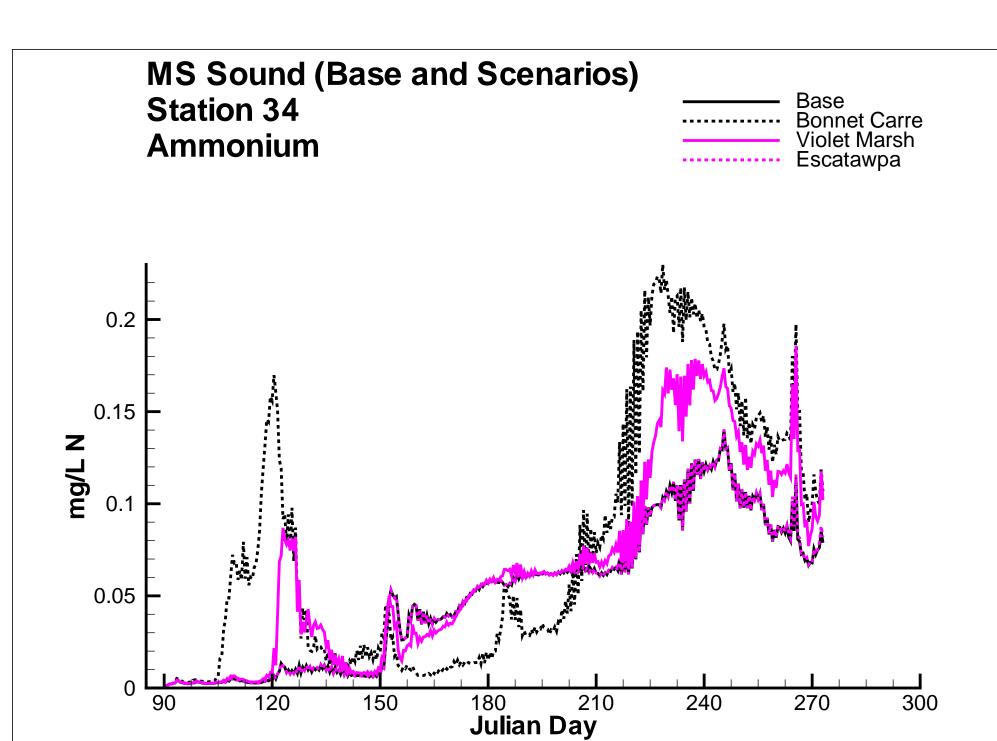
Escatawpa



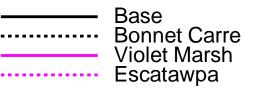


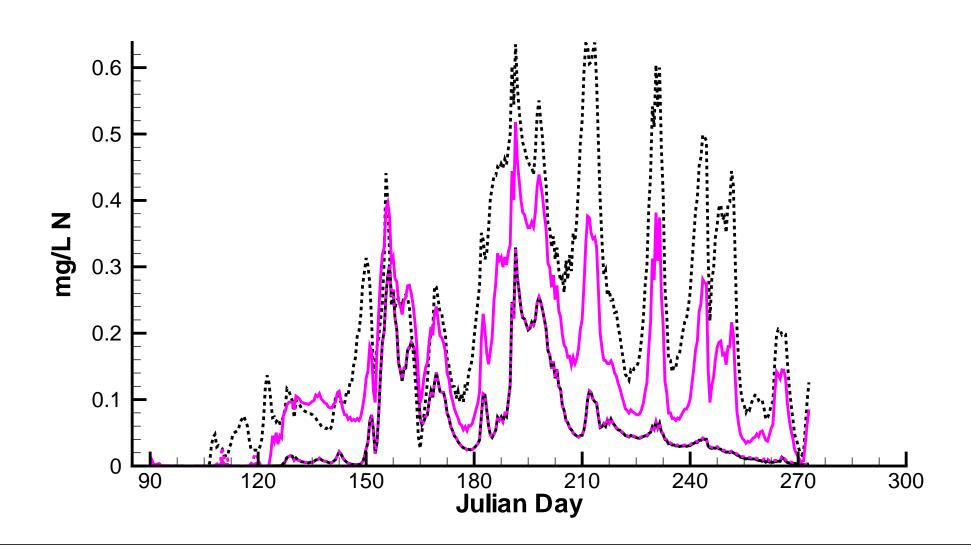




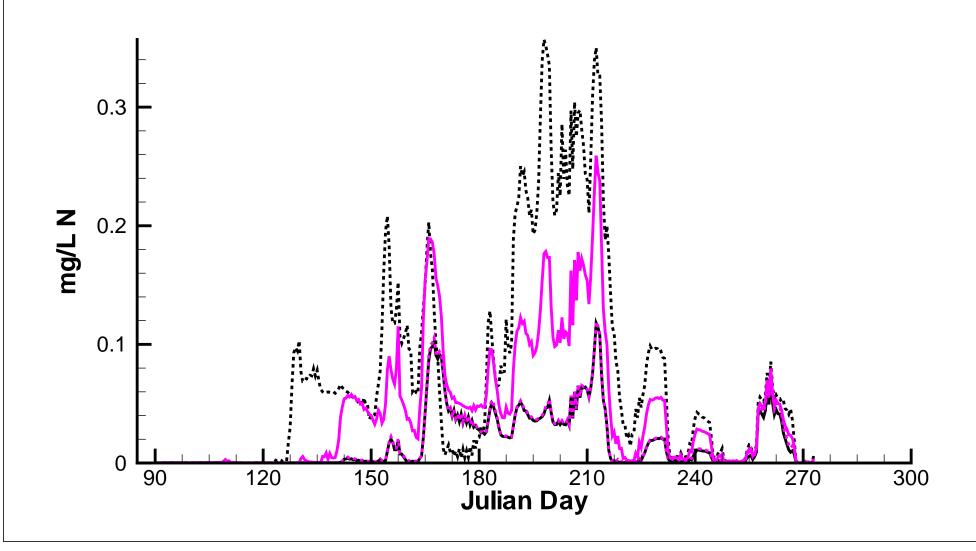


MS Sound (Base and Scenarios)
Station 1
Nitrate + Nitrite



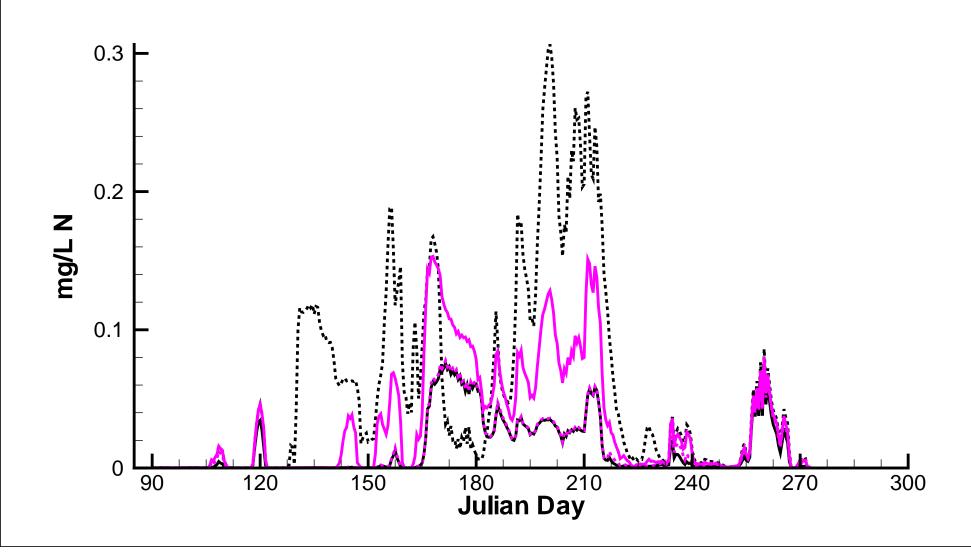






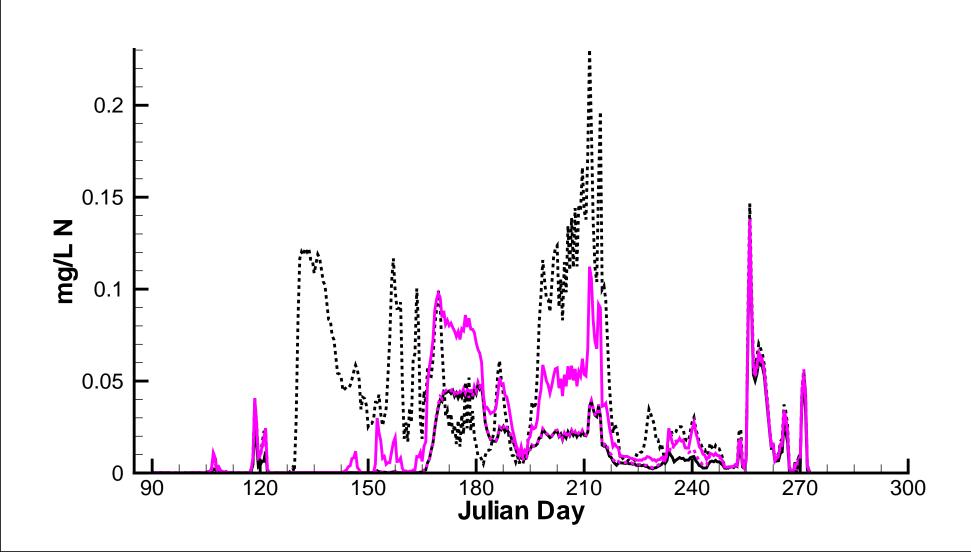




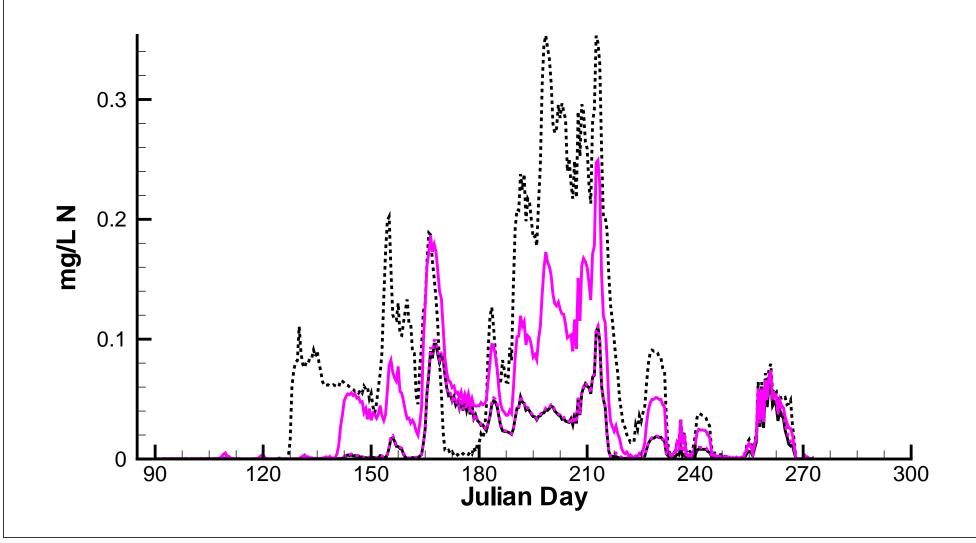




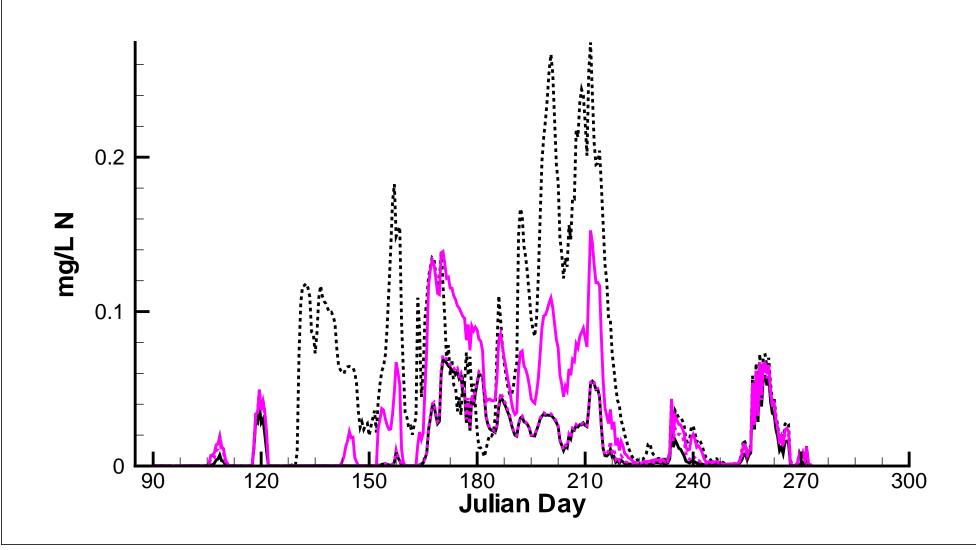










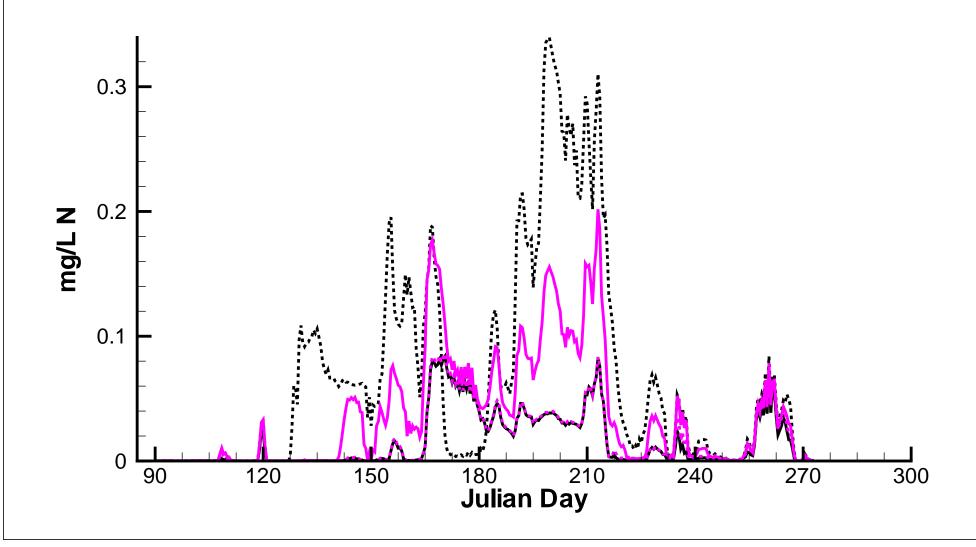




Base

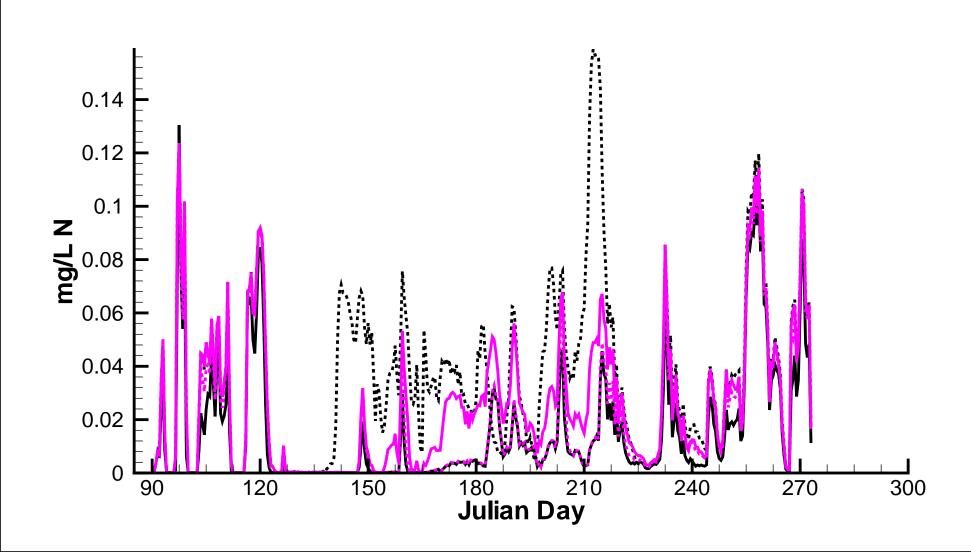
Bonnet Carre Violet Marsh

Escatawpa



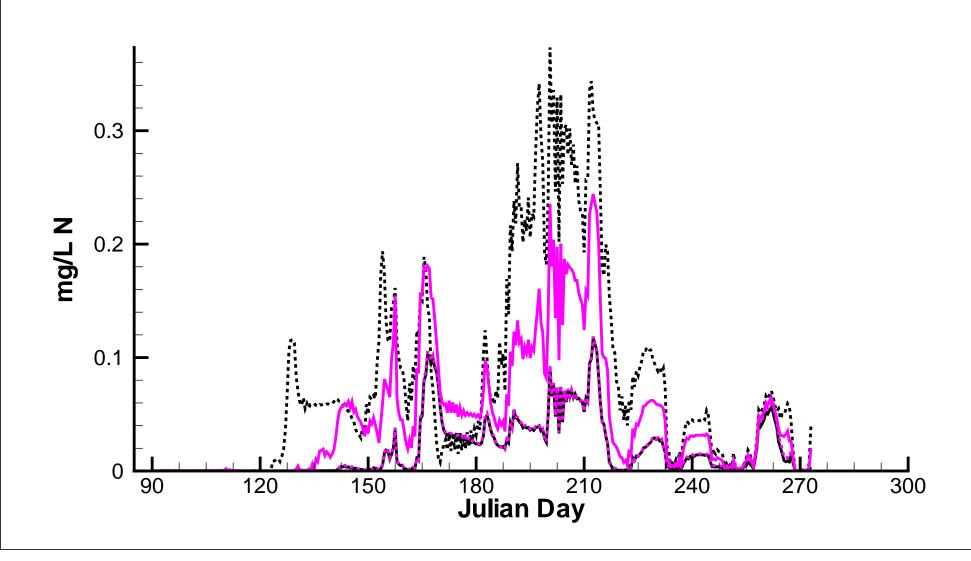
MS Sound (Base and Scenarios)
Station 8
Nitrate + Nitrite





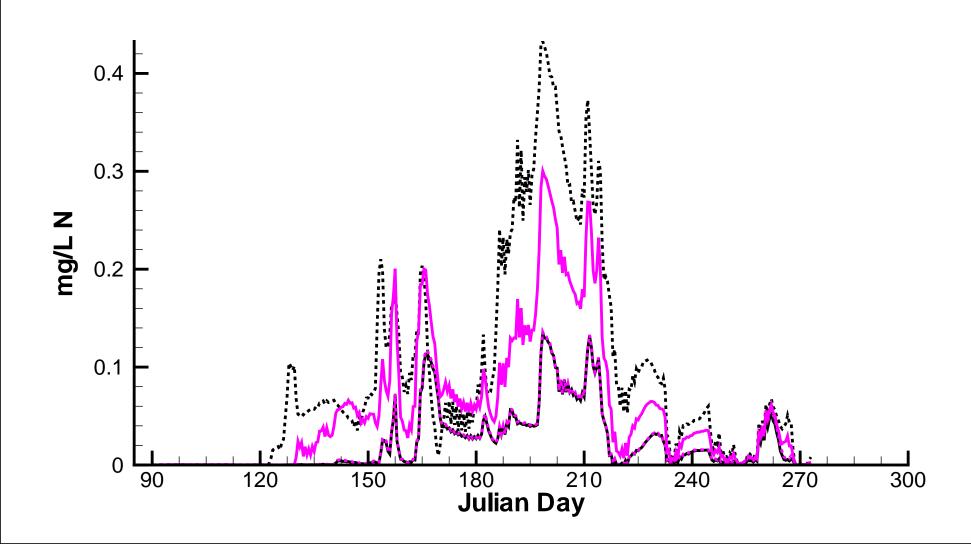


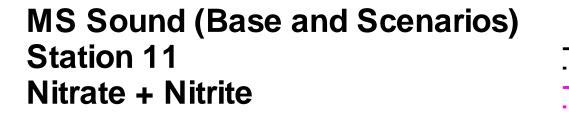




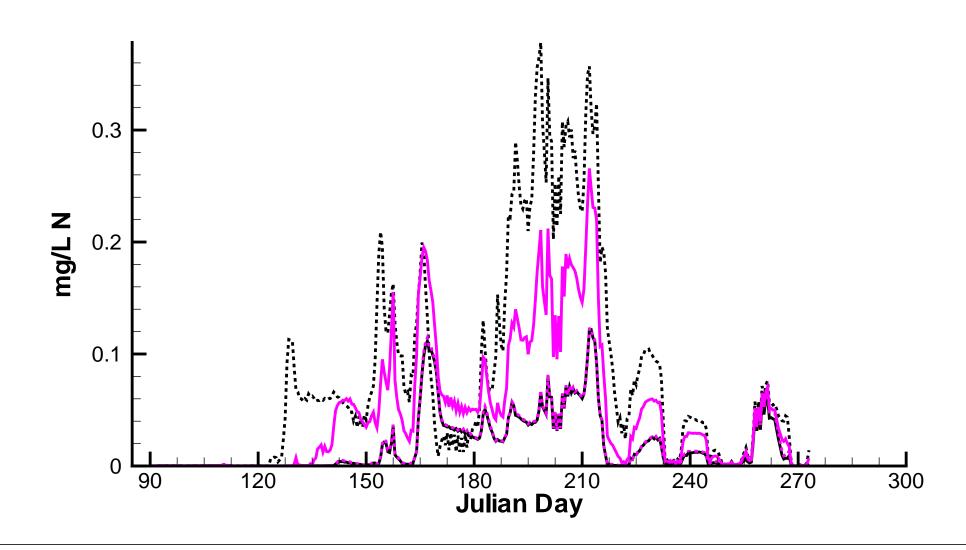






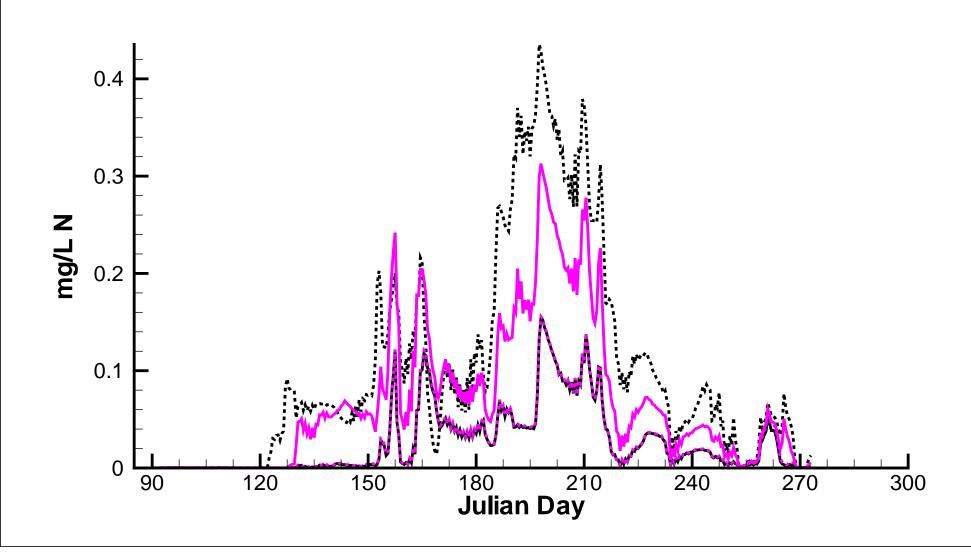






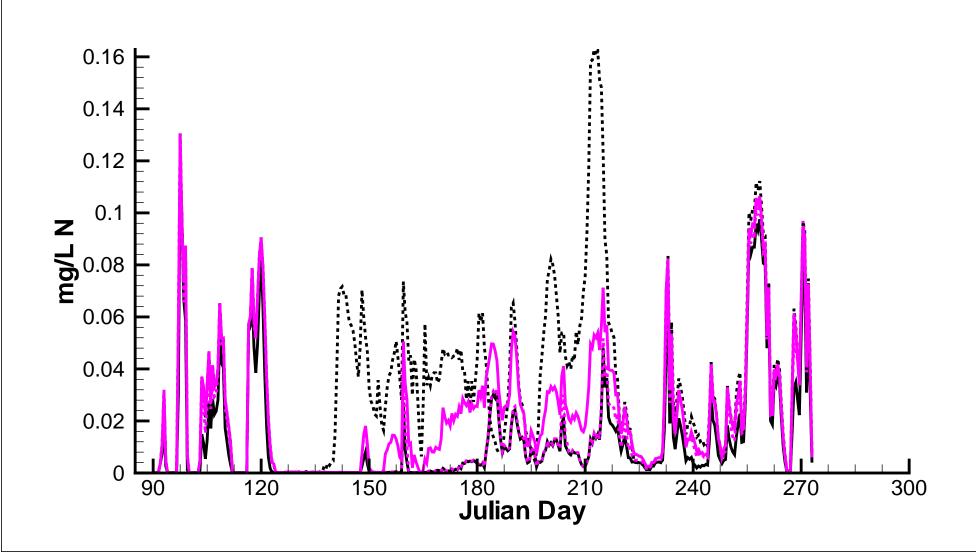






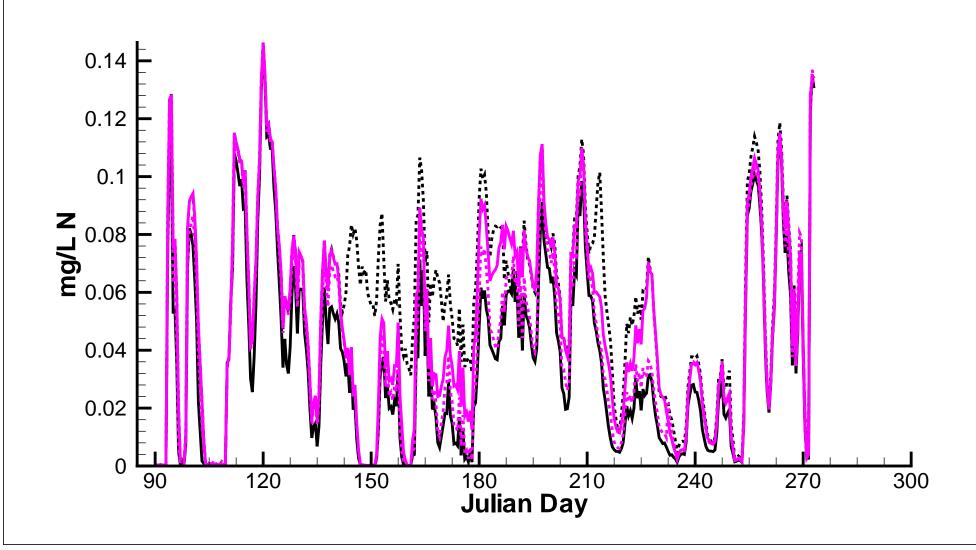
MS Sound (Base and Scenarios)
Station 13
Nitrate + Nitrite





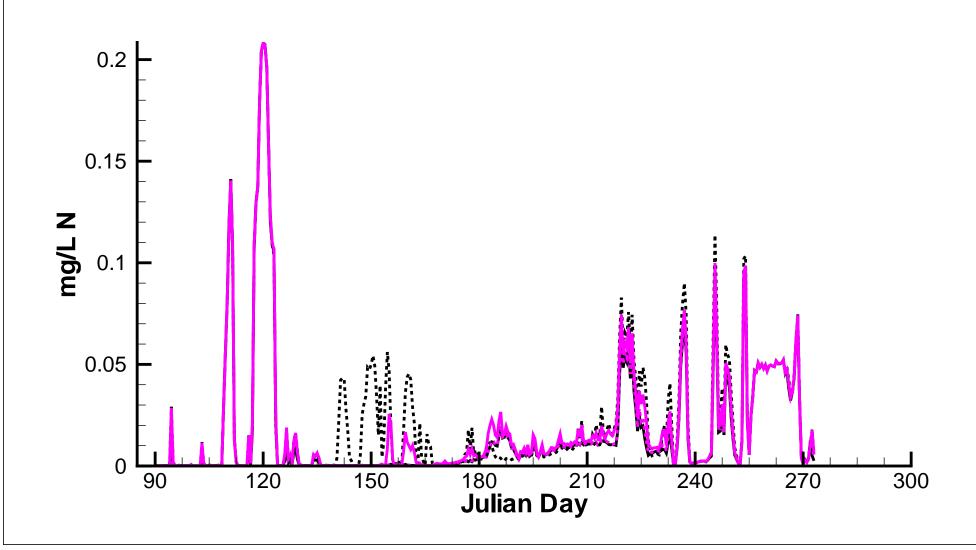
MS Sound (Base and Scenarios)
Station 14
Nitrate + Nitrite





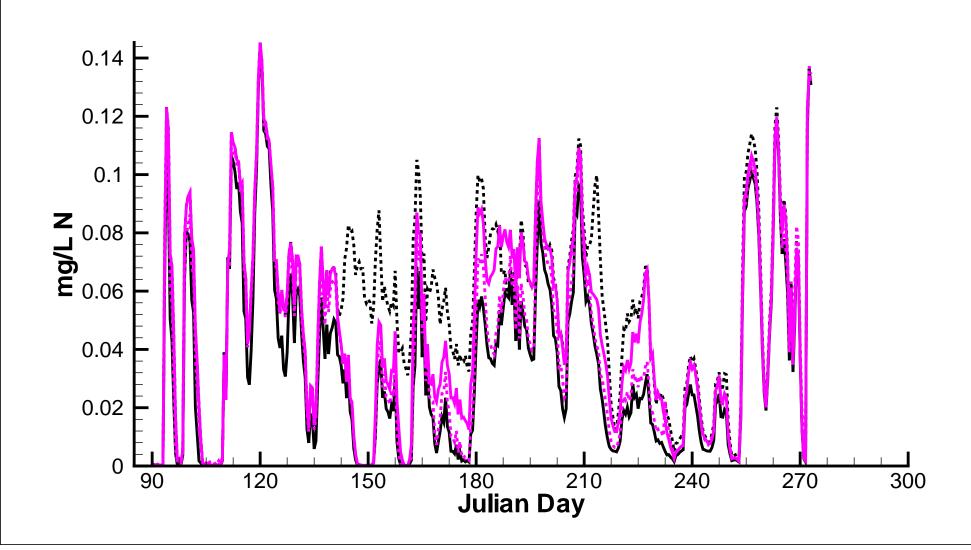






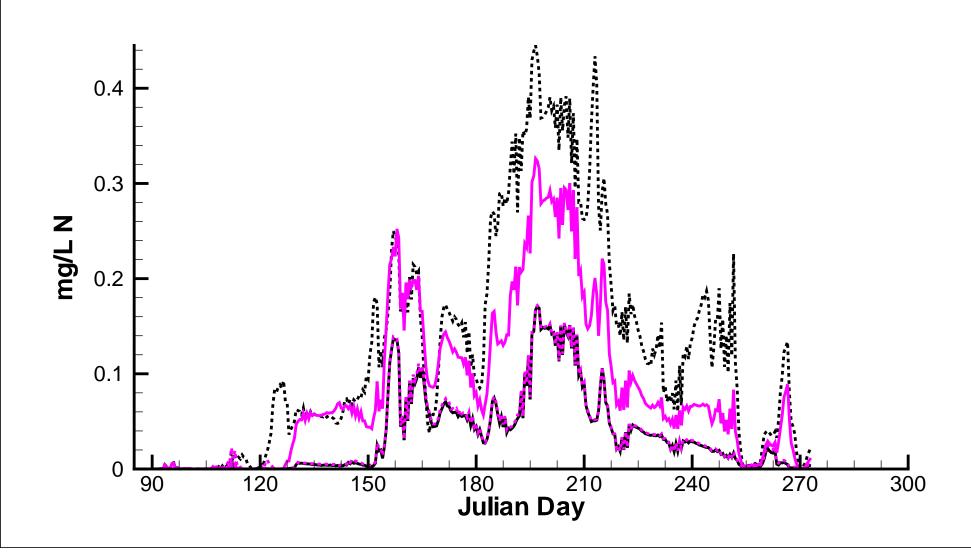






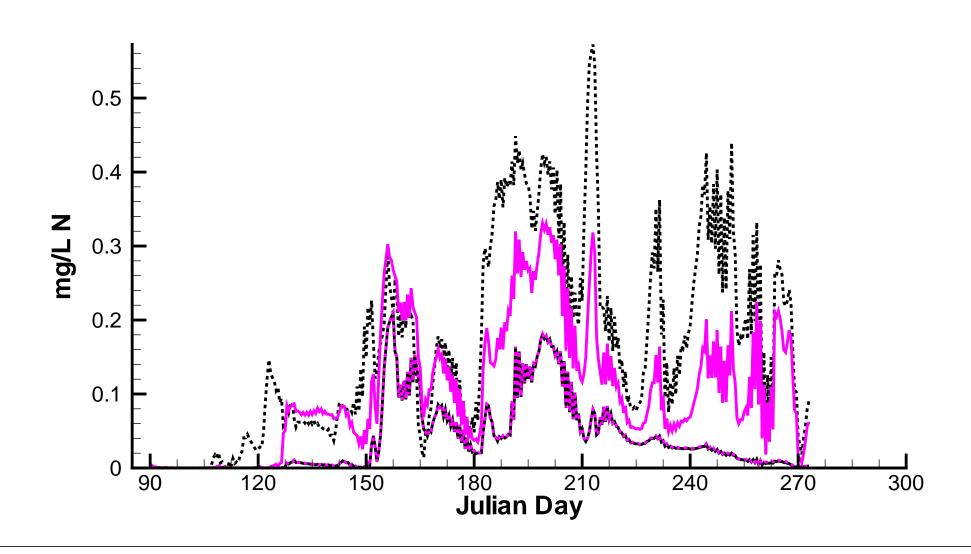


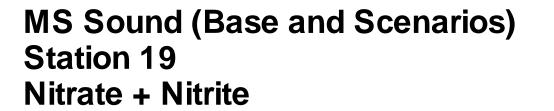




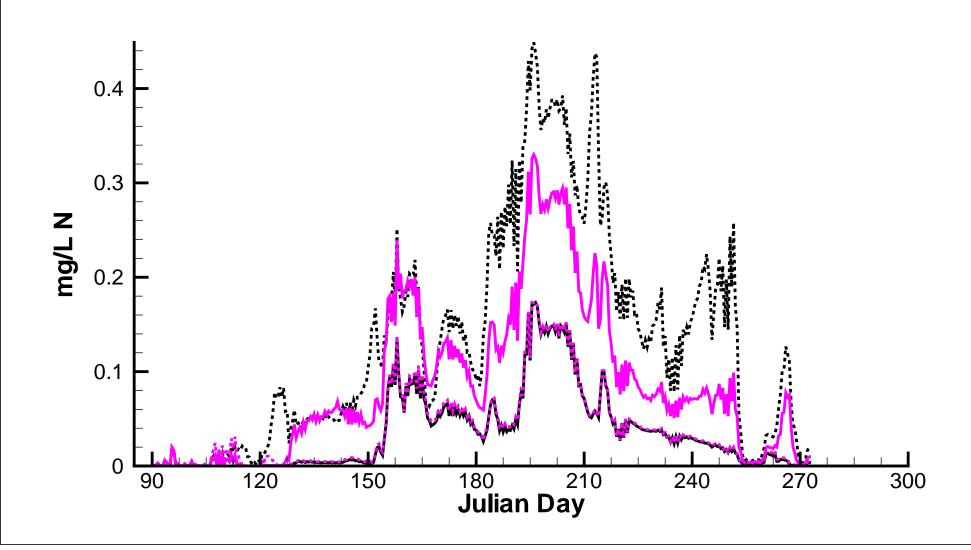
MS Sound (Base and Scenarios)
Station 18
Nitrate + Nitrite





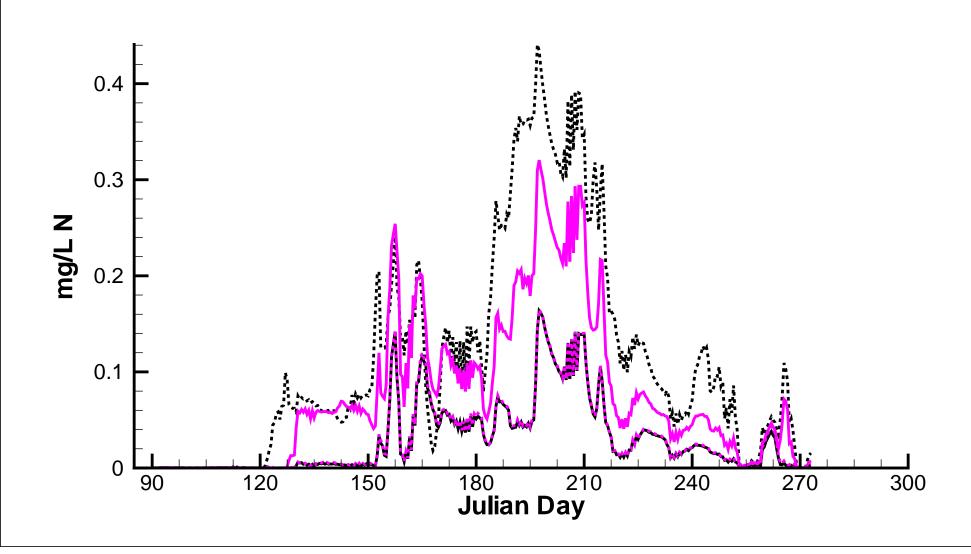




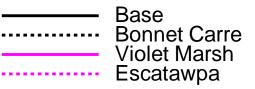


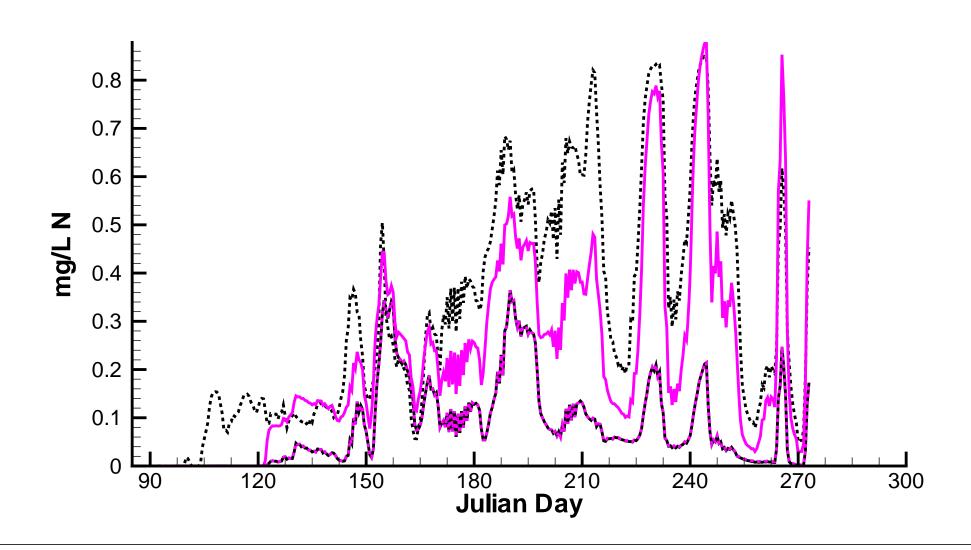
MS Sound (Base and Scenarios)
Station 20
Nitrate + Nitrite





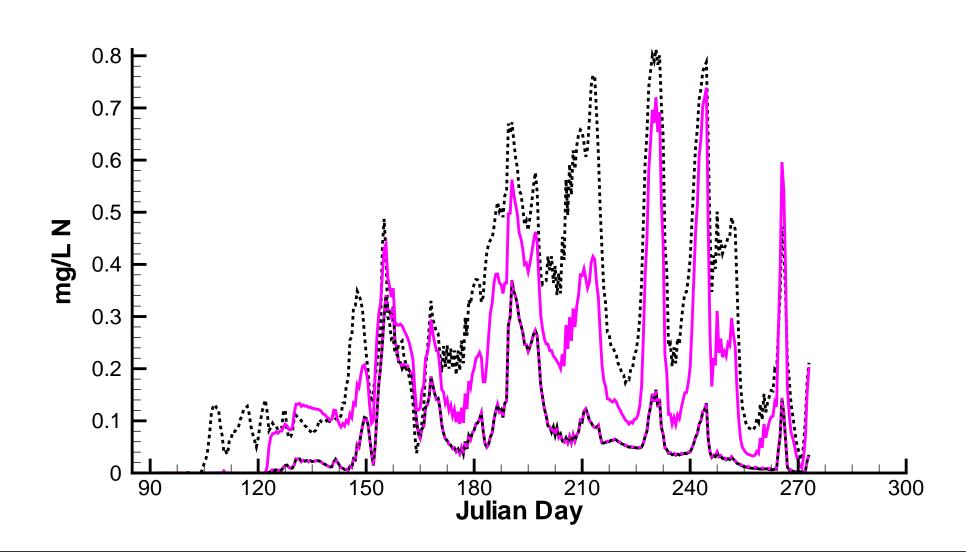
MS Sound (Base and Scenarios)
Station 21
Nitrate + Nitrite

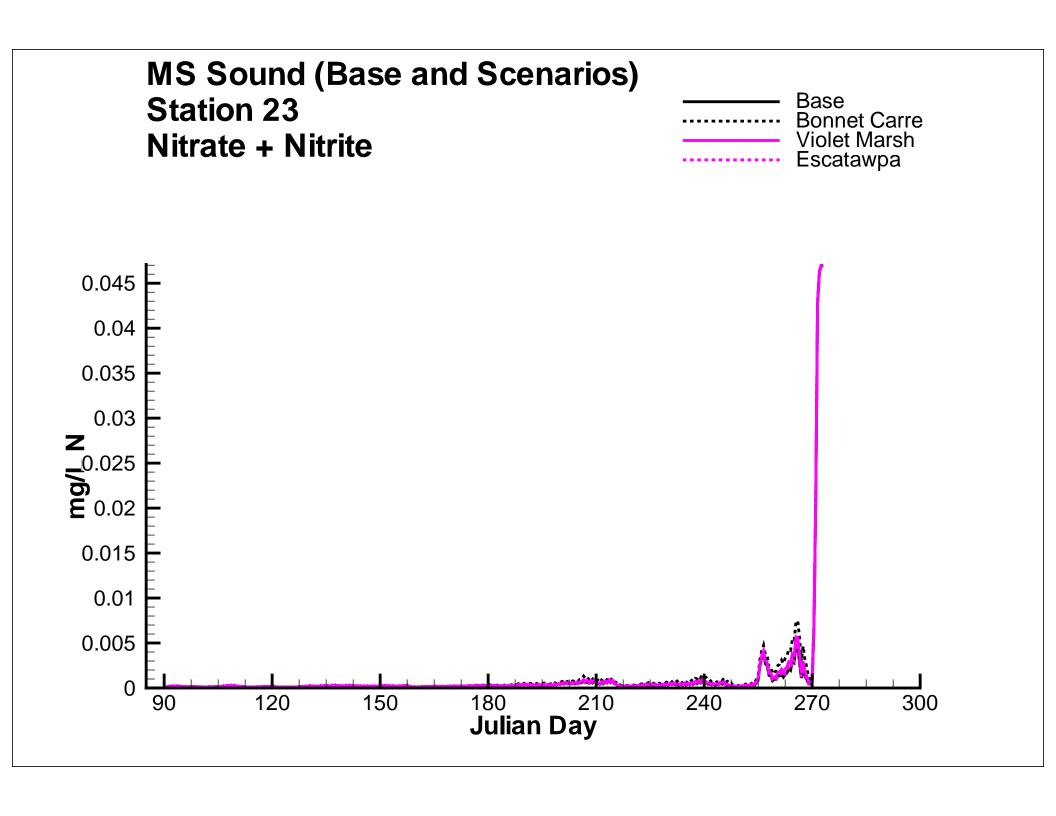






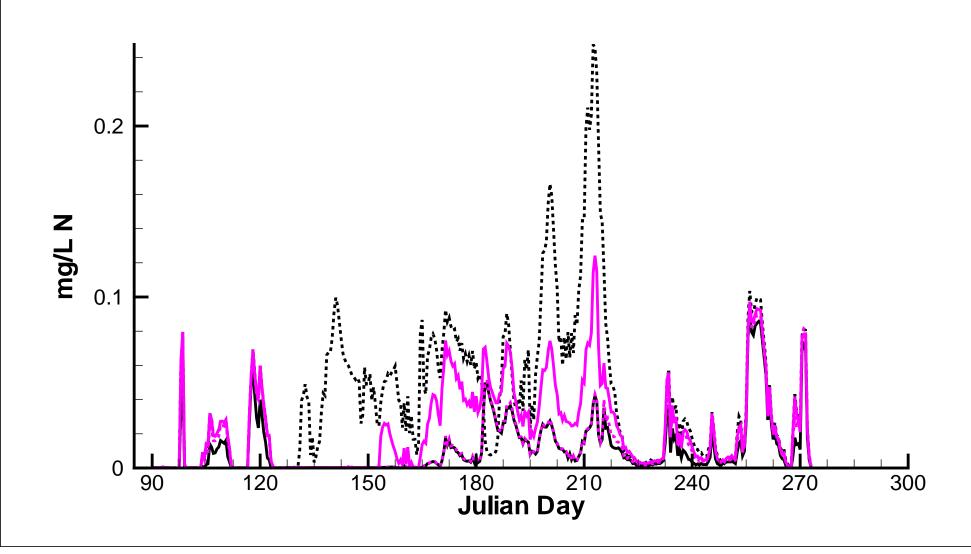




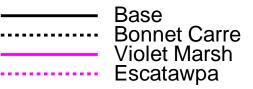


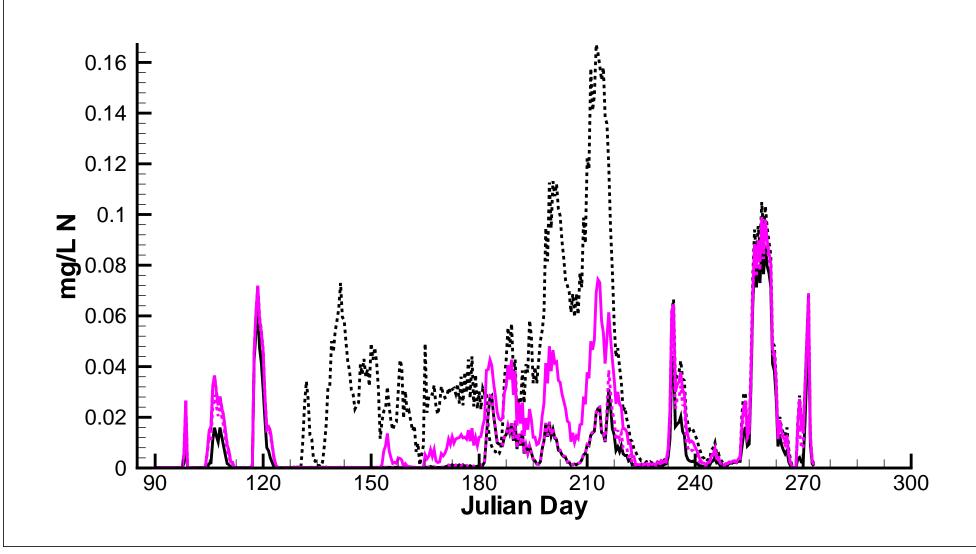




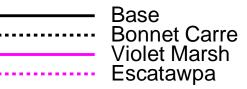


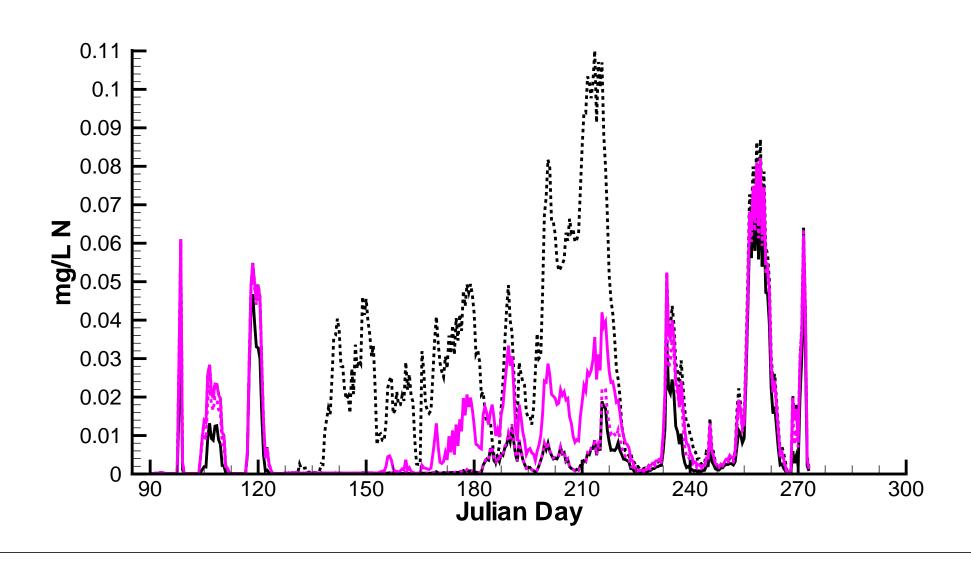






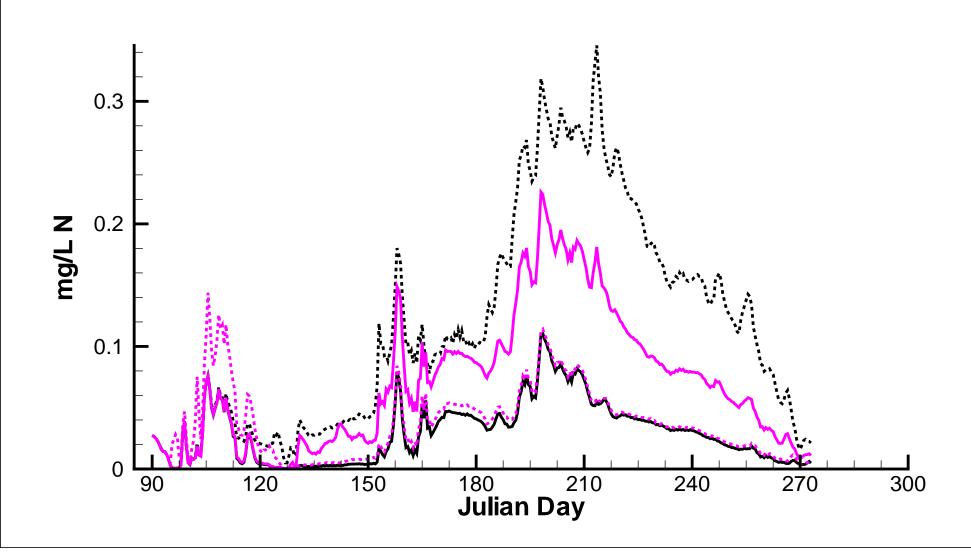
MS Sound (Base and Scenarios)
Station 26
Nitrate + Nitrite

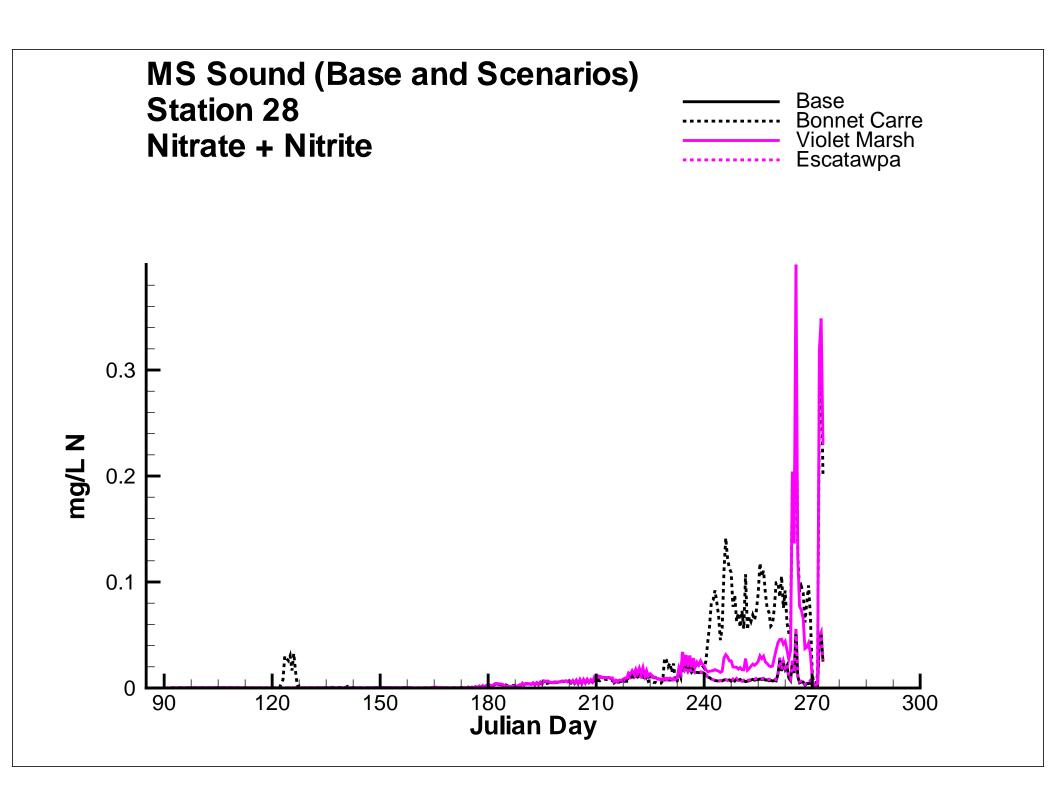


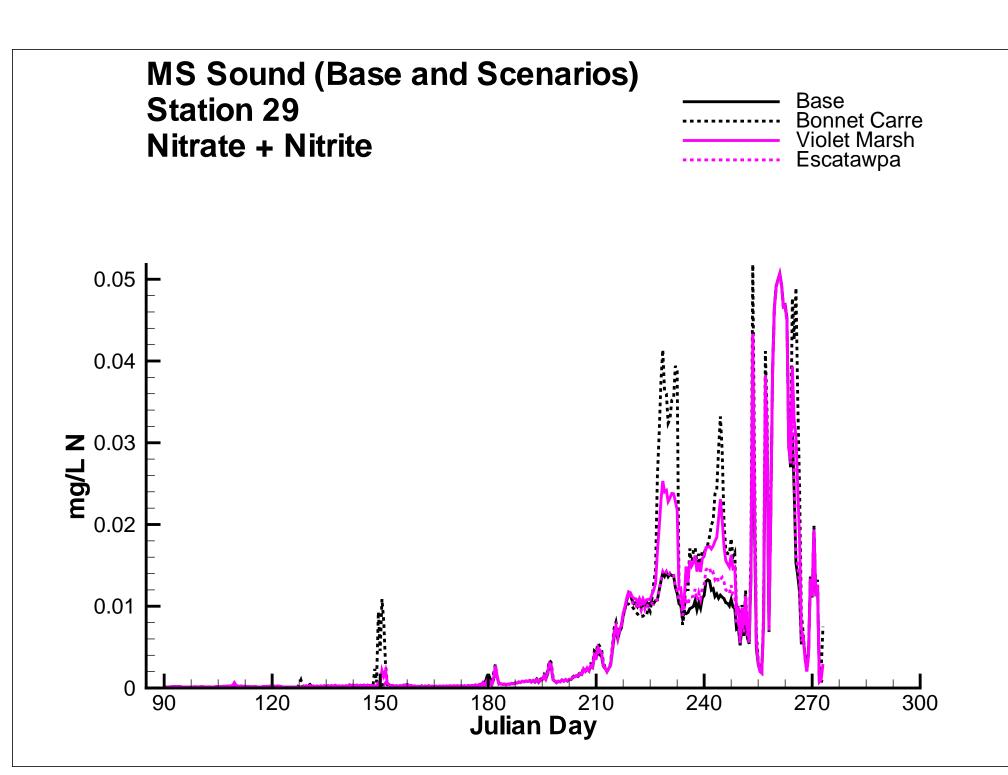






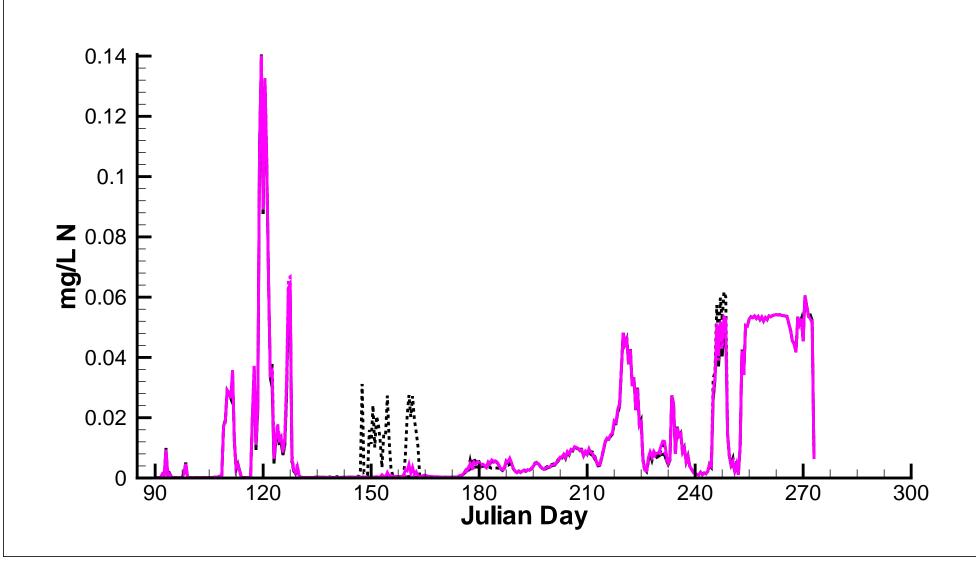






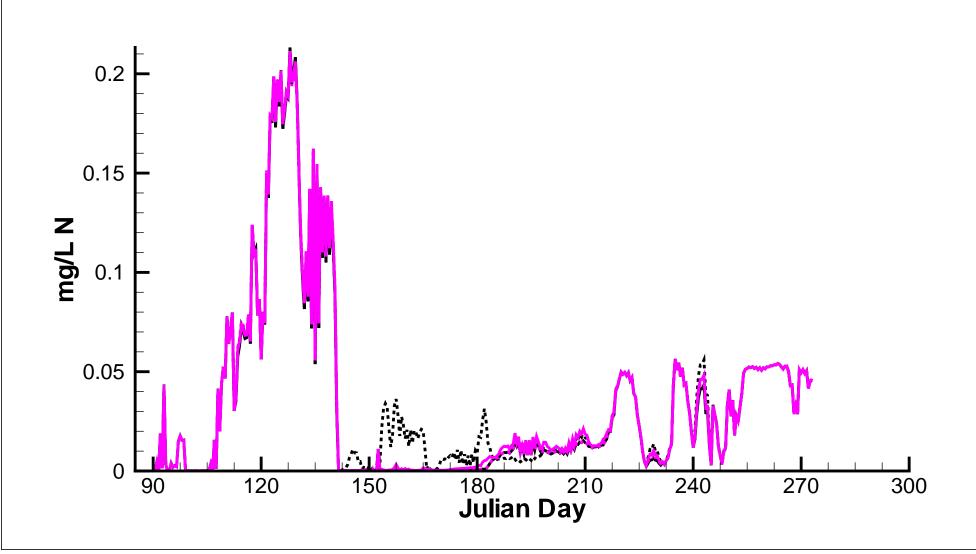






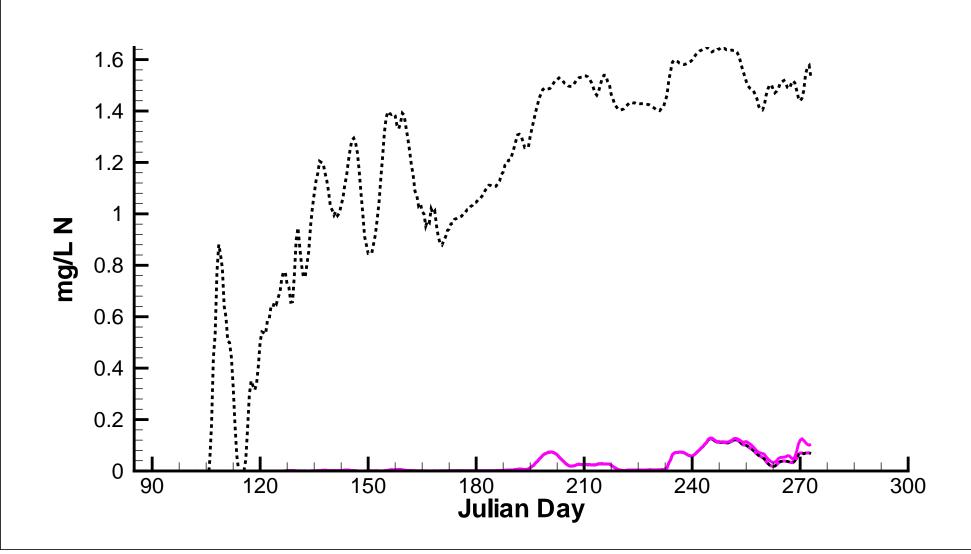
MS Sound (Base and Scenarios)
Station 31
Nitrate + Nitrite





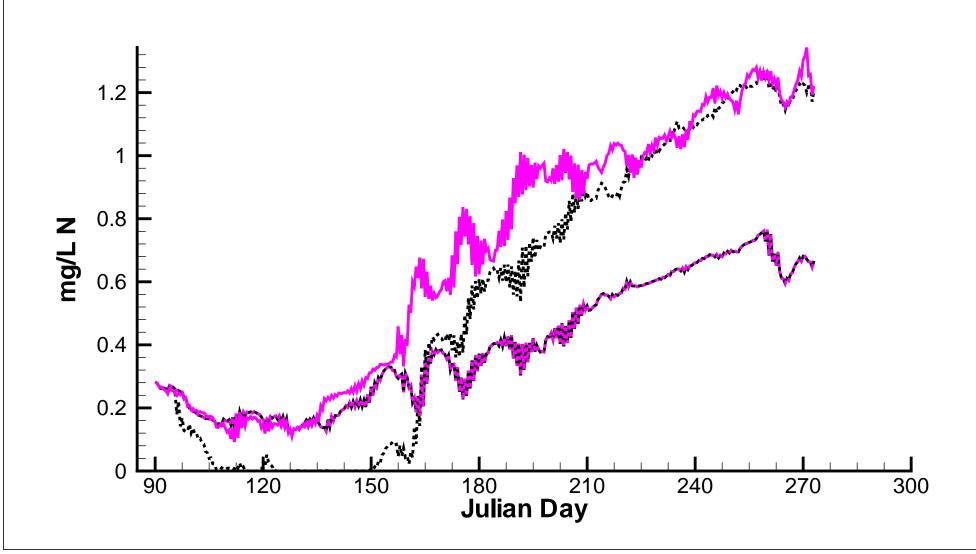


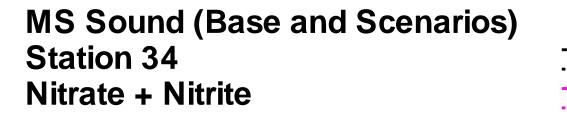


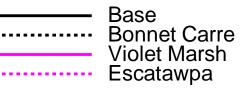


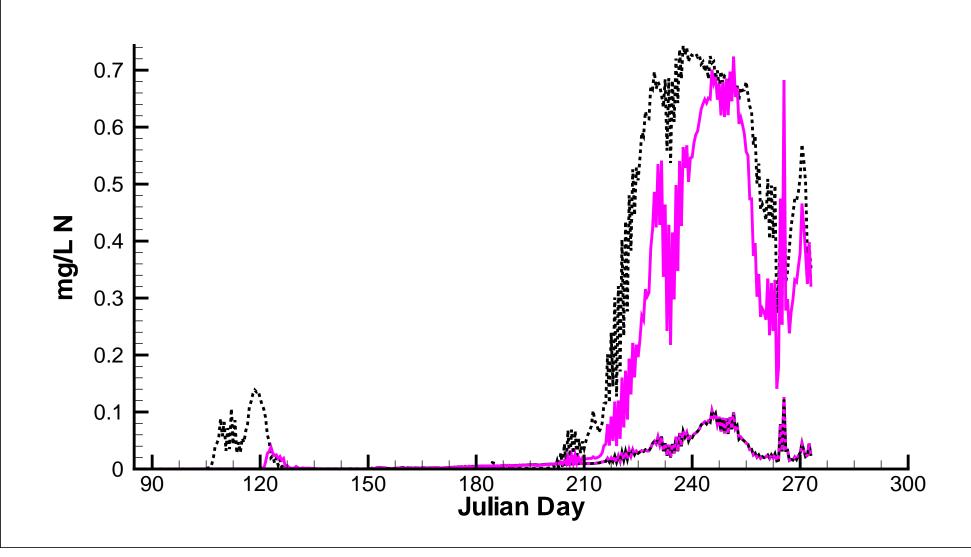






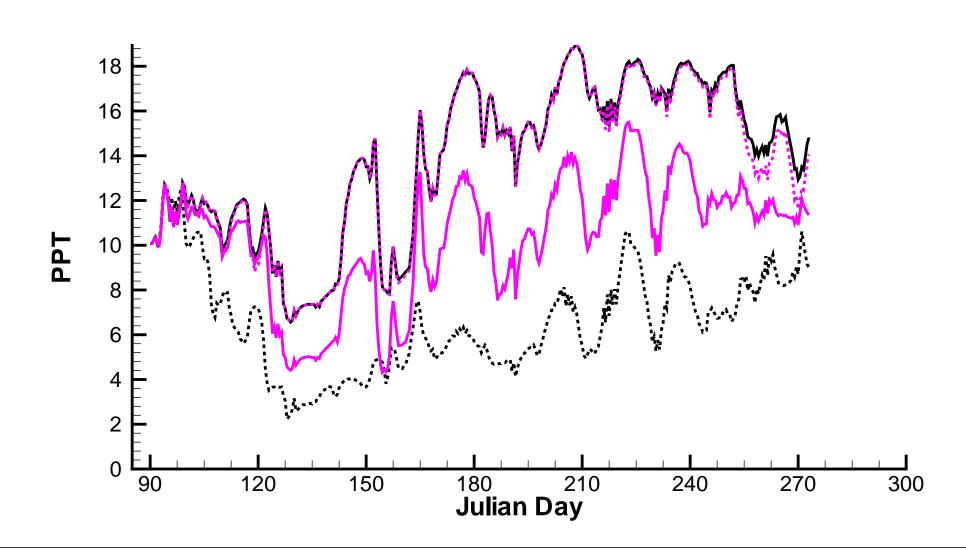




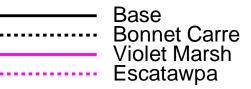


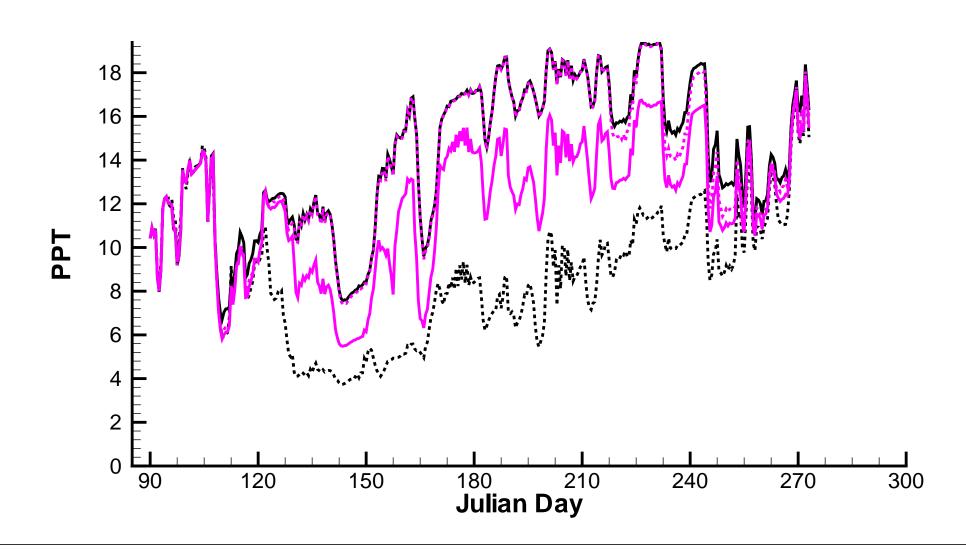






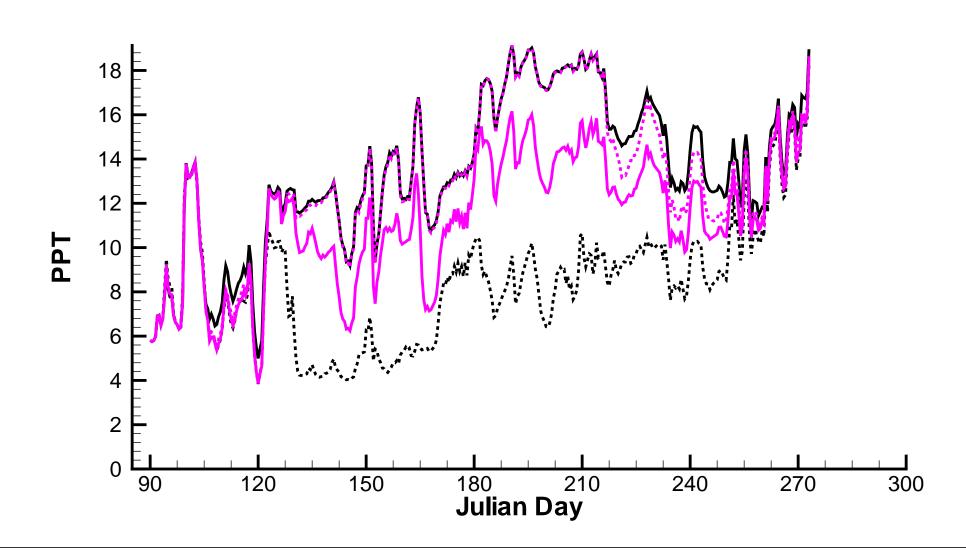










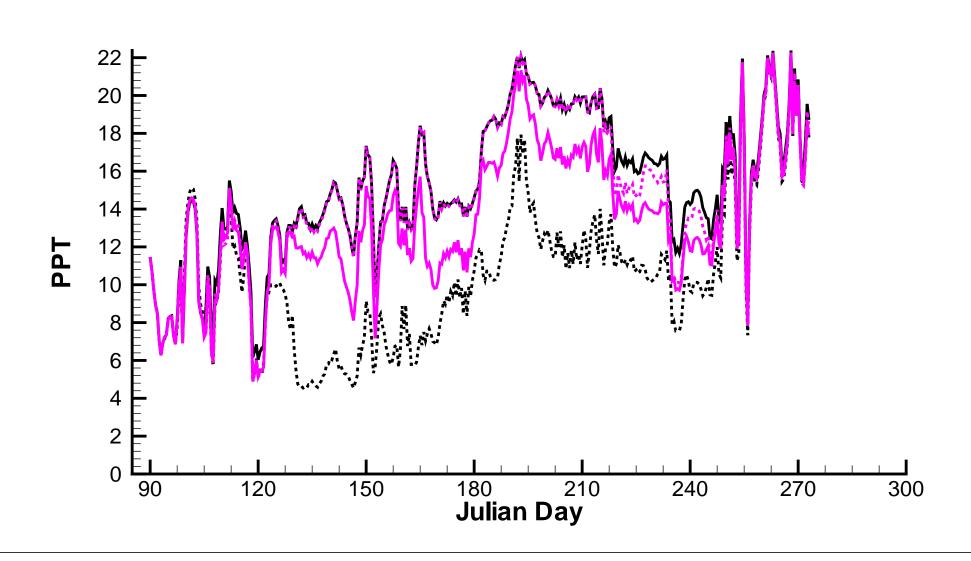




Base

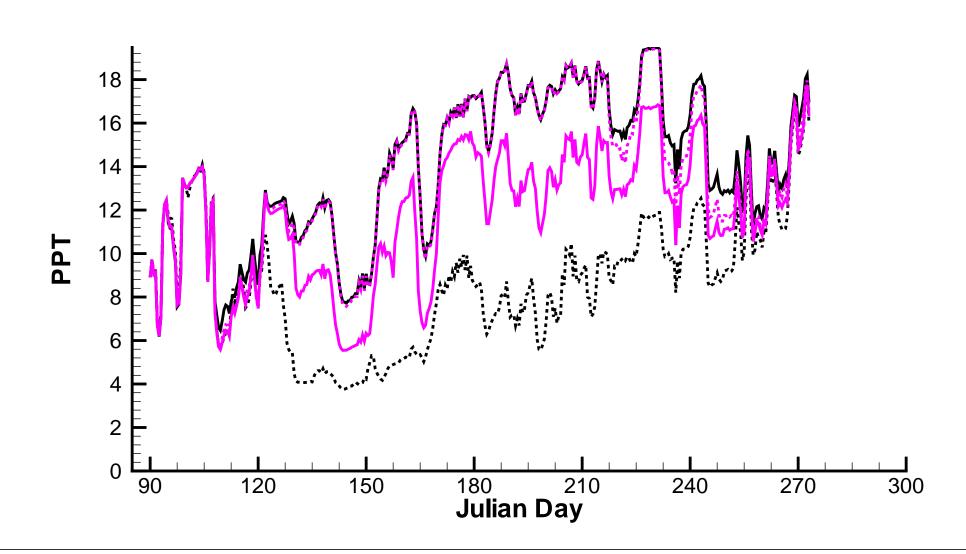
Bonnet Carre Violet Marsh

Escatawpa

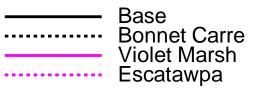


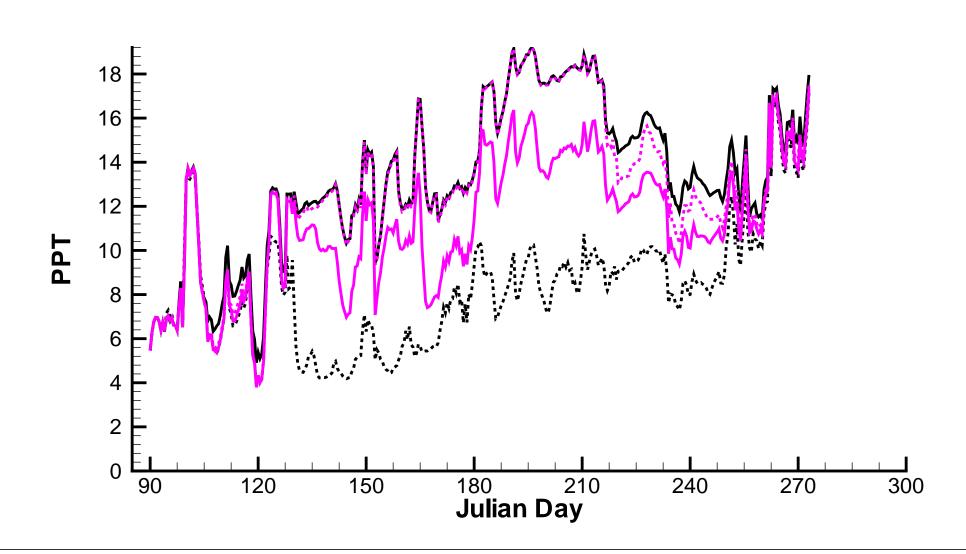






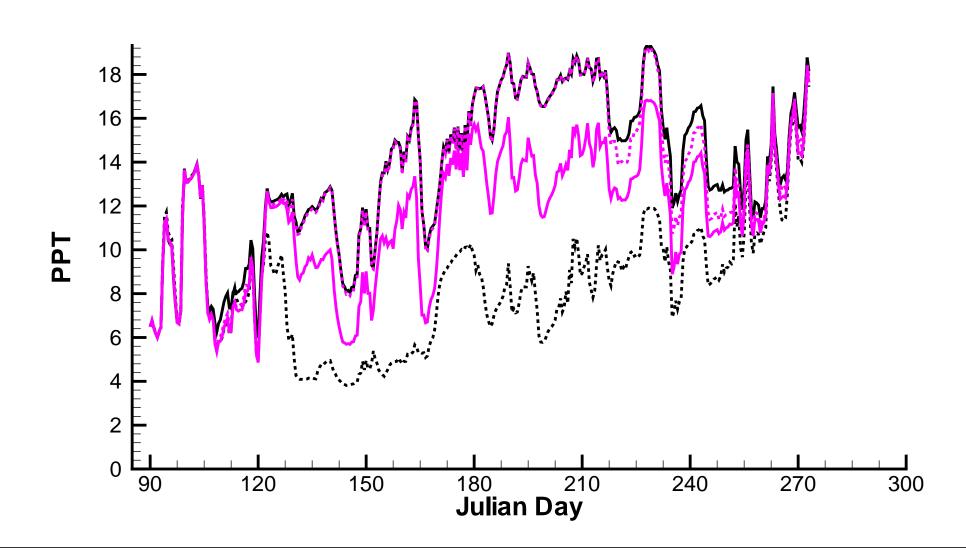






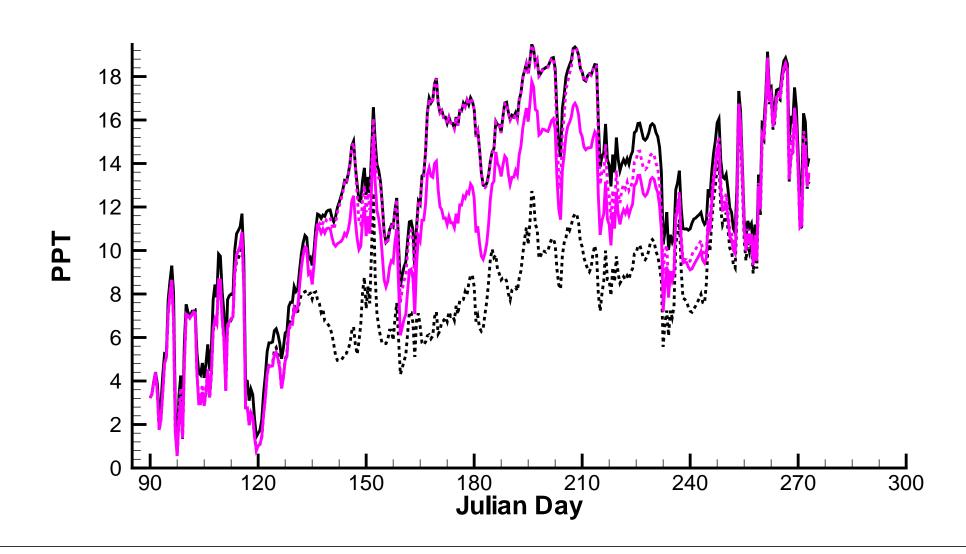




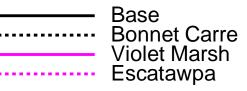


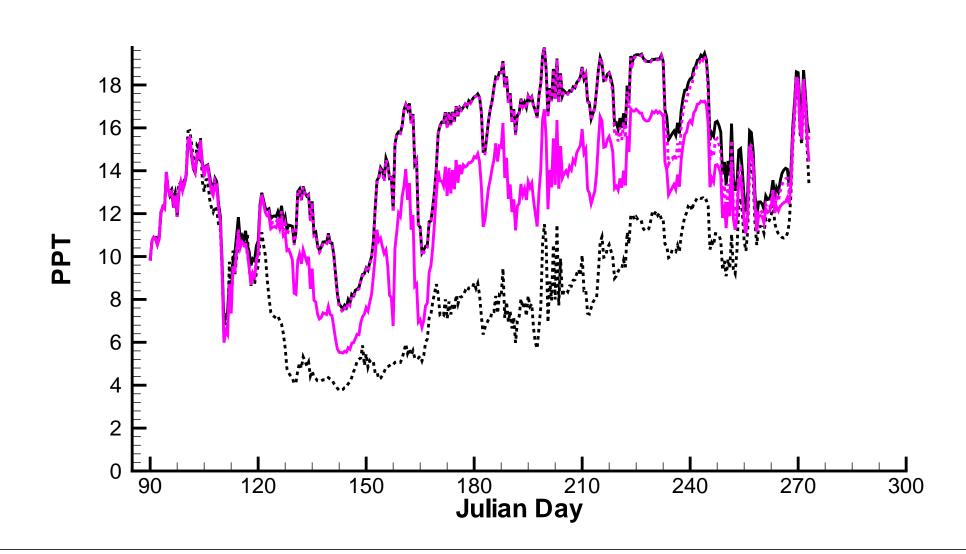




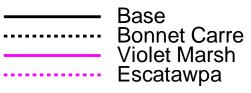


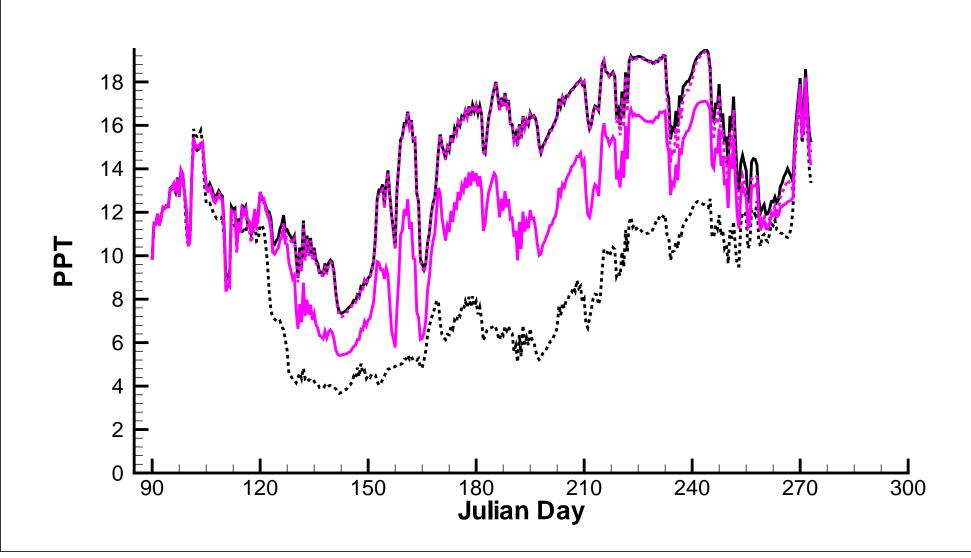






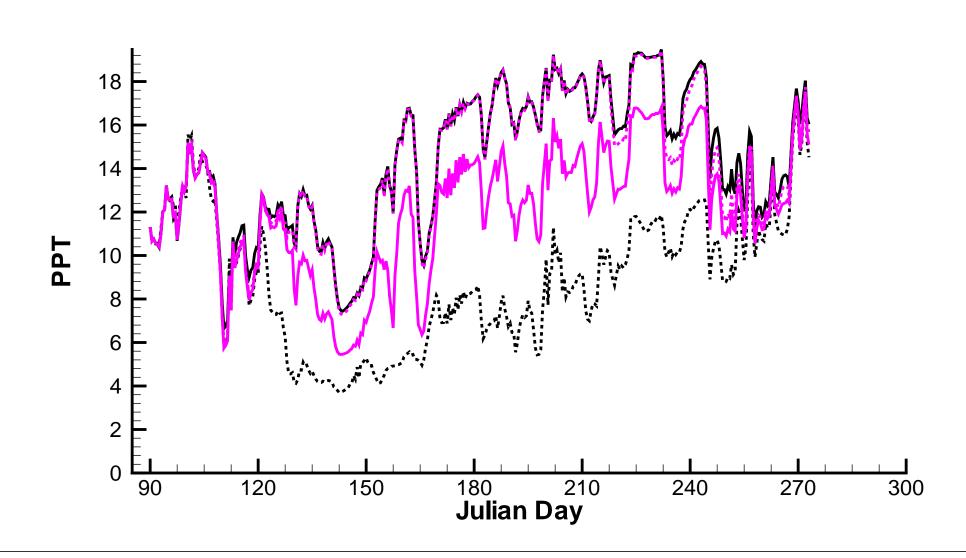


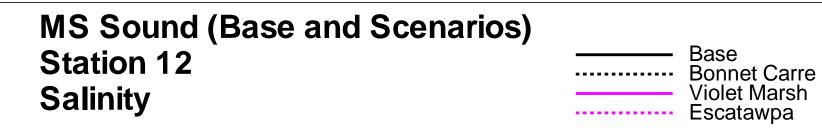


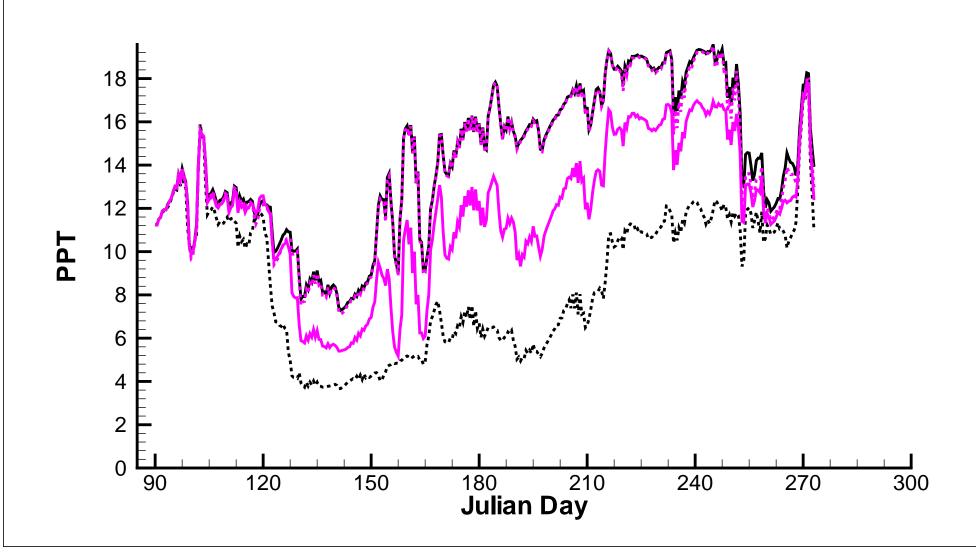






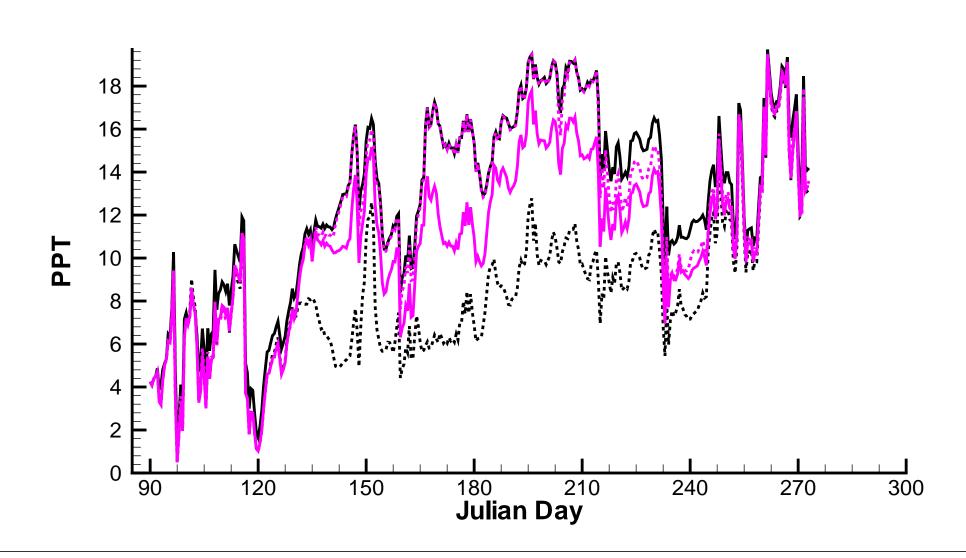






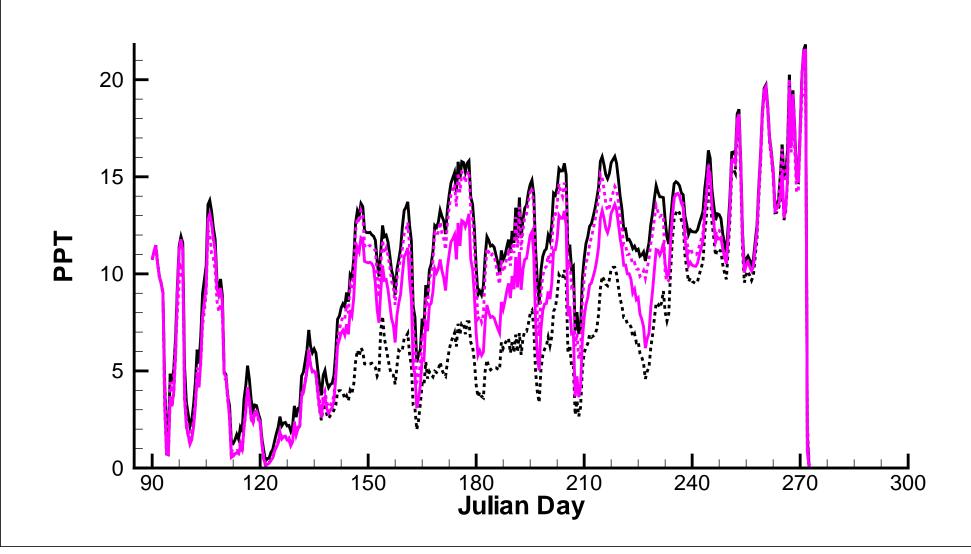


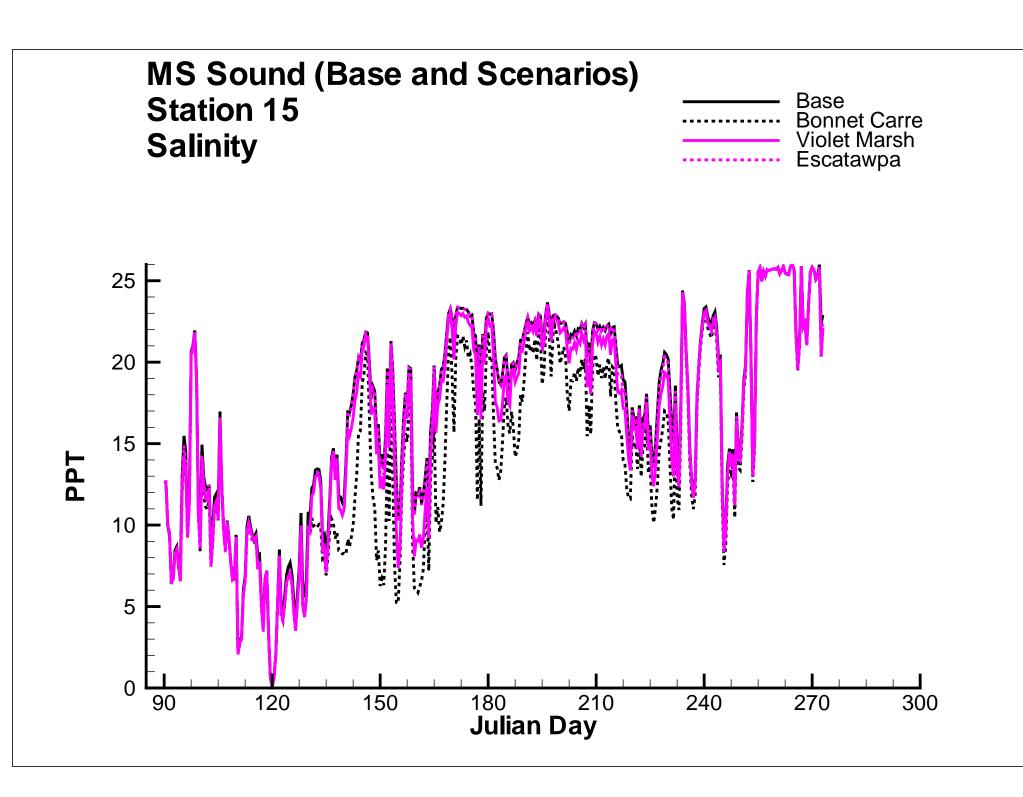






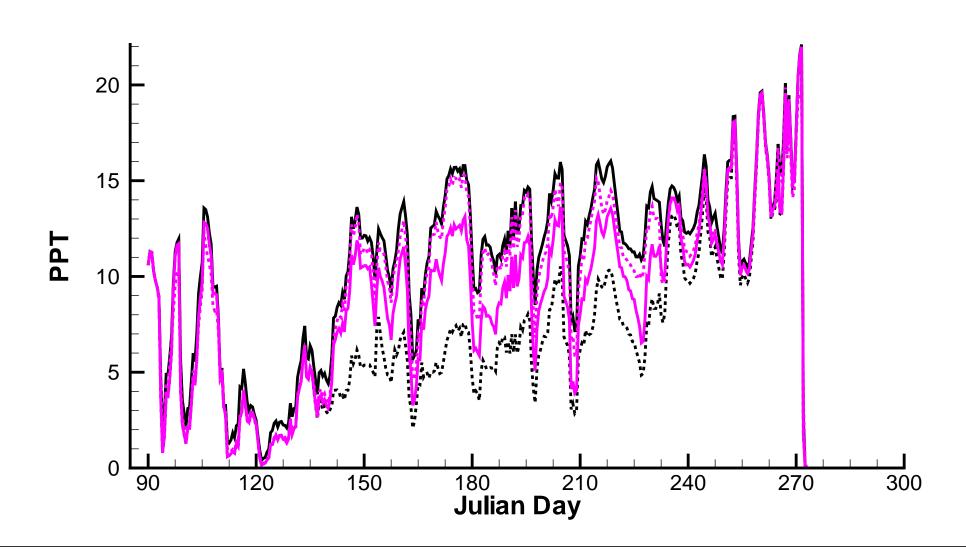






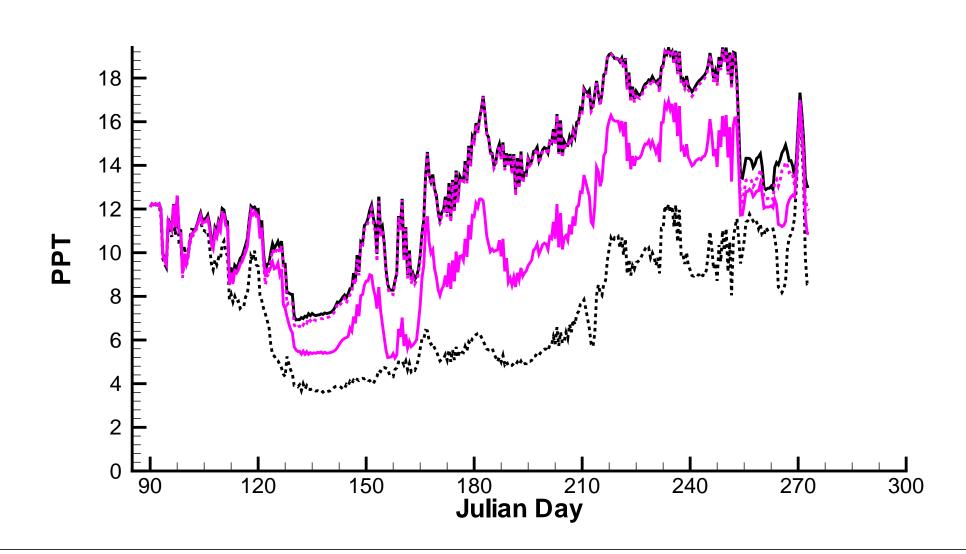






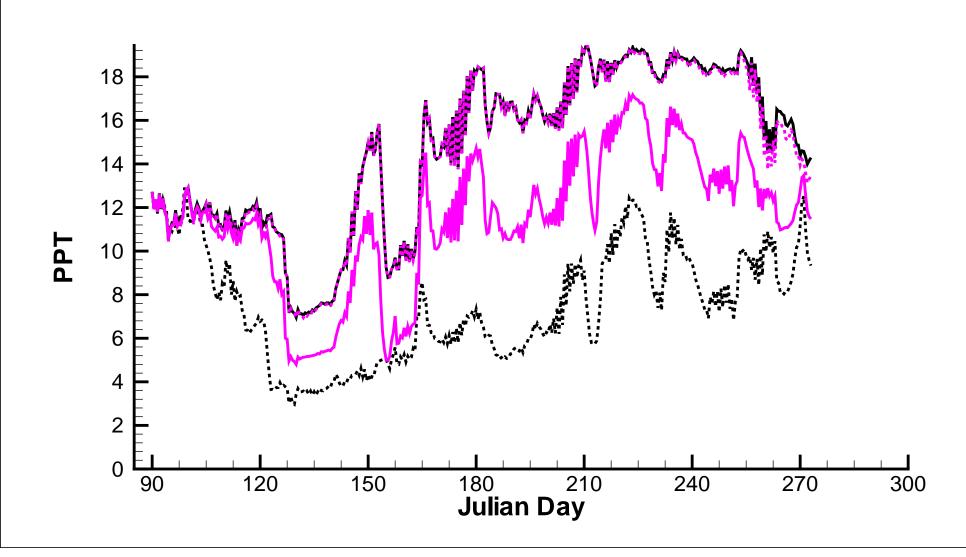










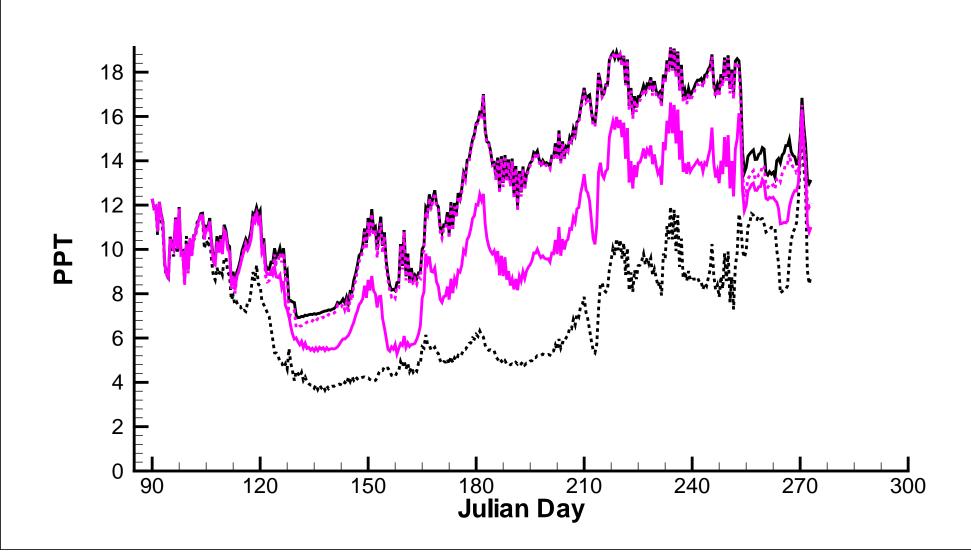




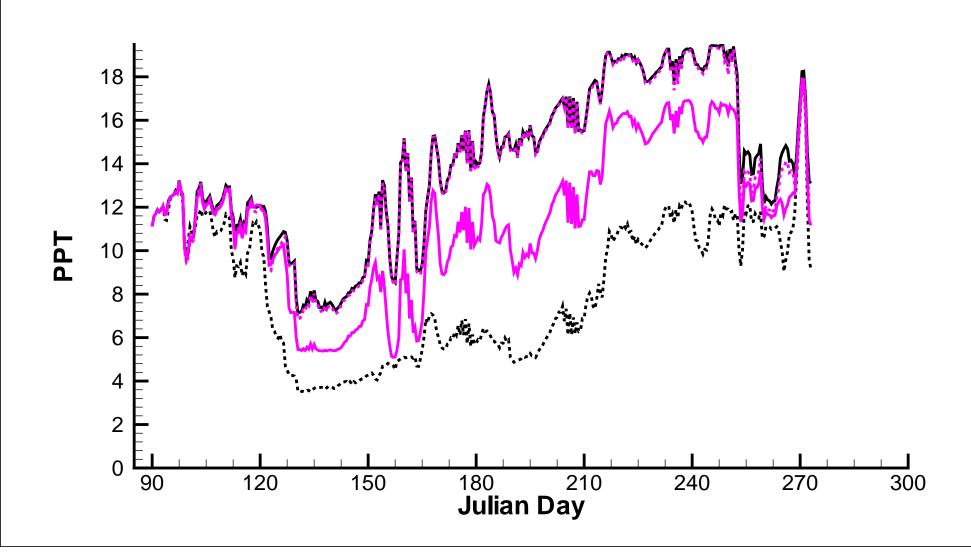
Base

Bonnet Carre Violet Marsh

Escatawpa

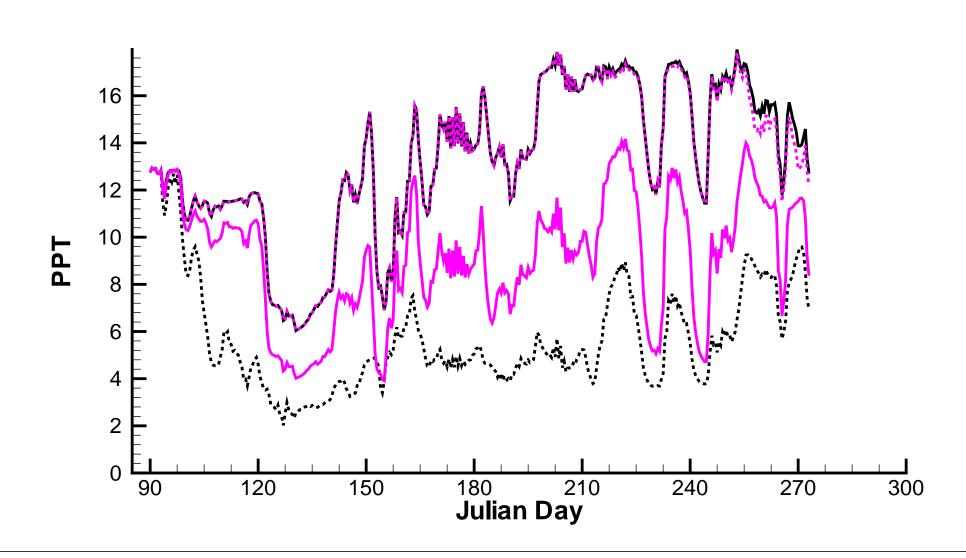






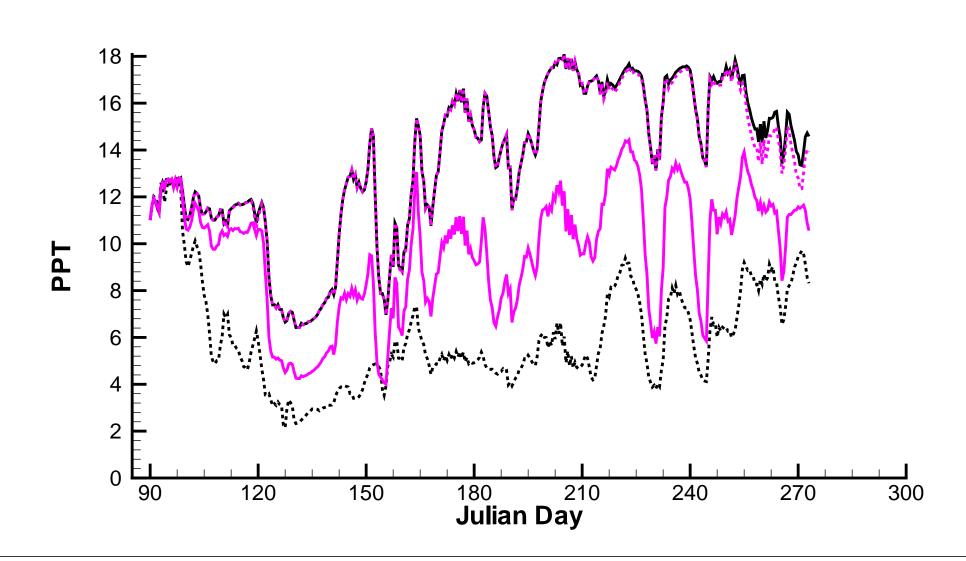


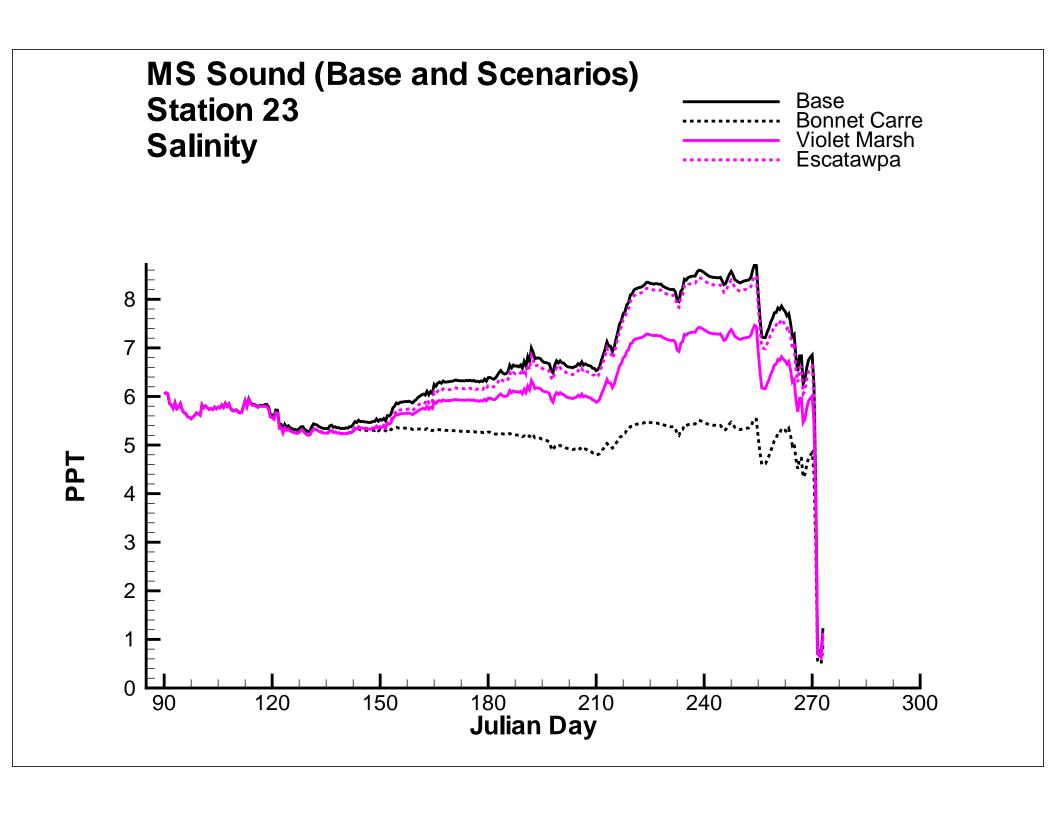






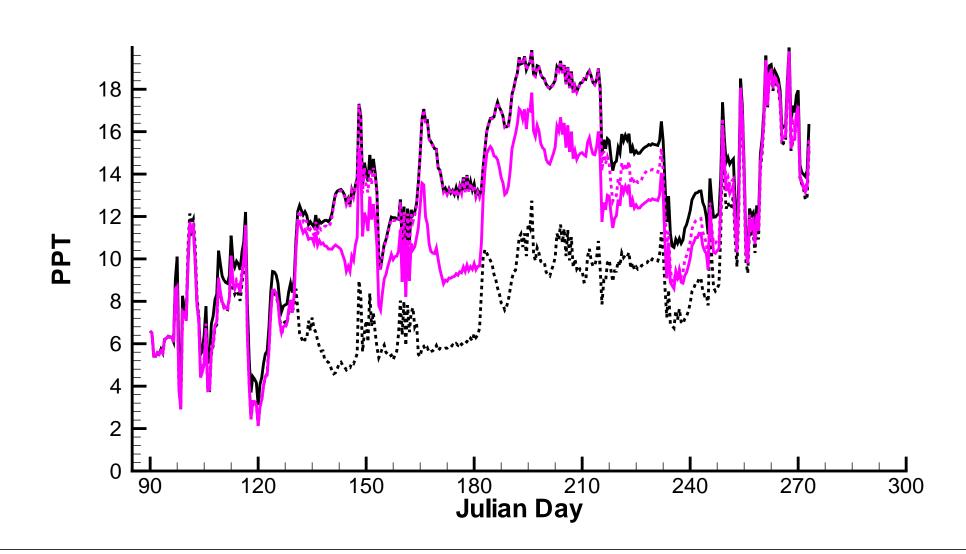


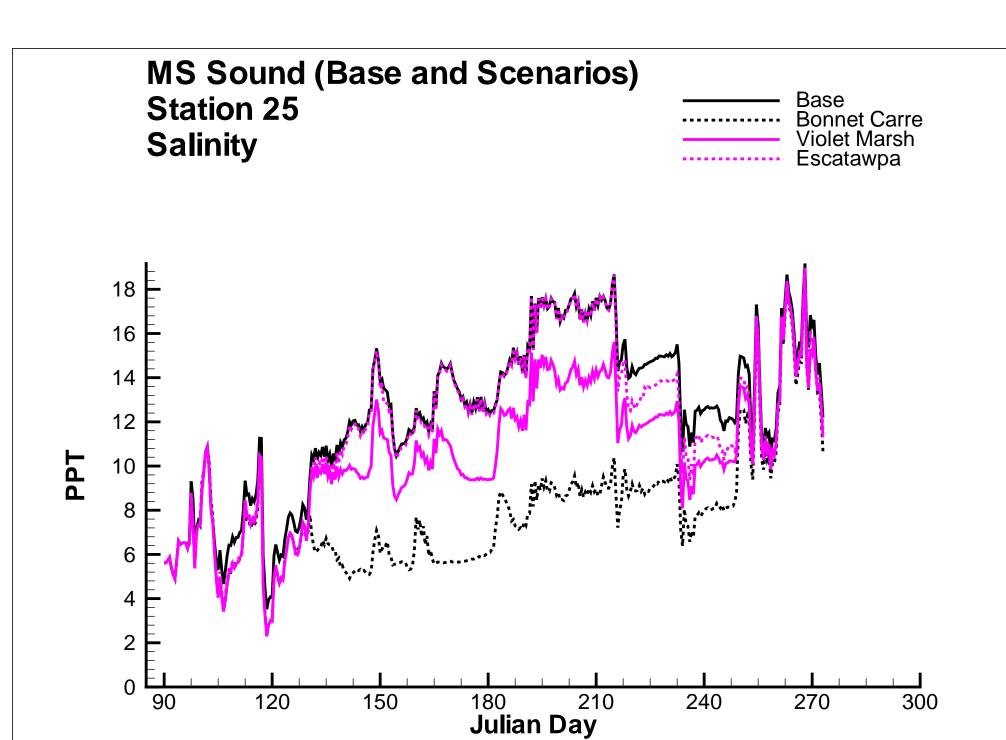


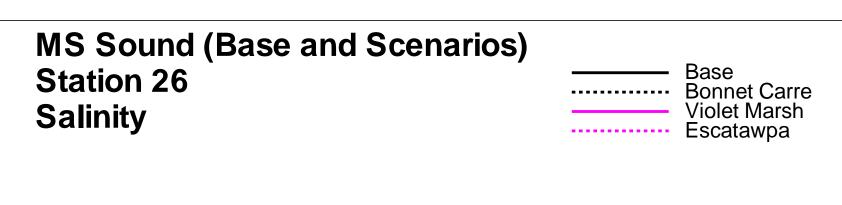


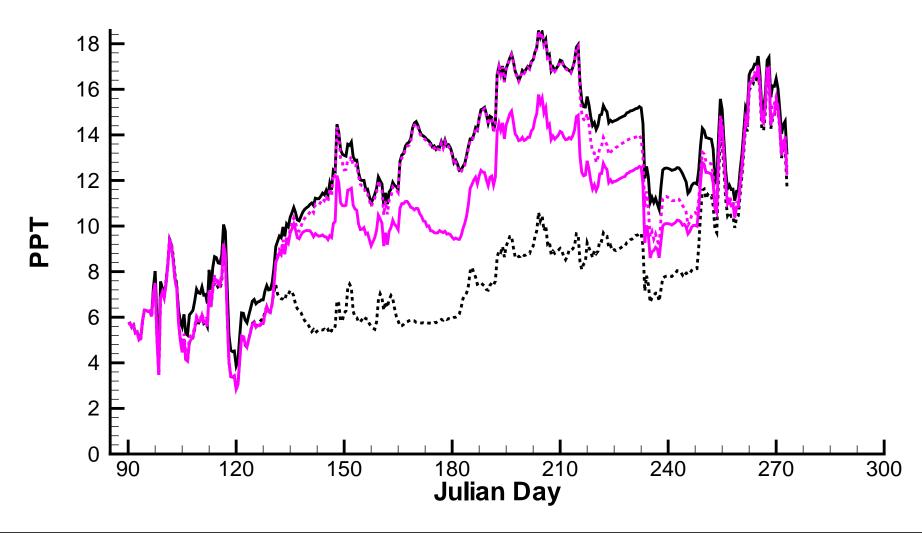




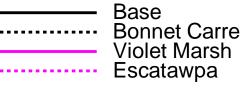


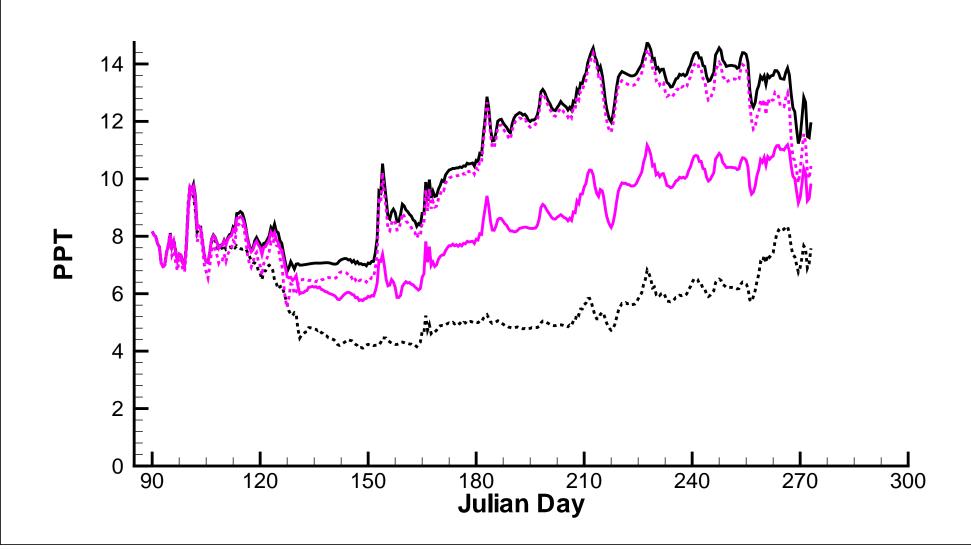


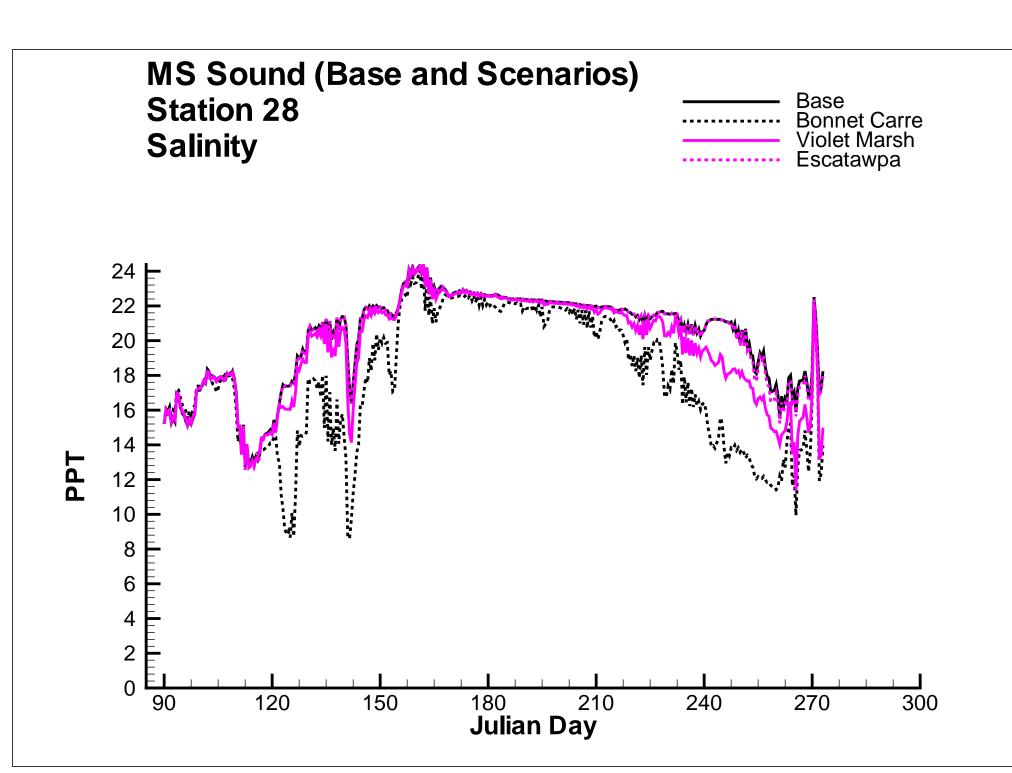


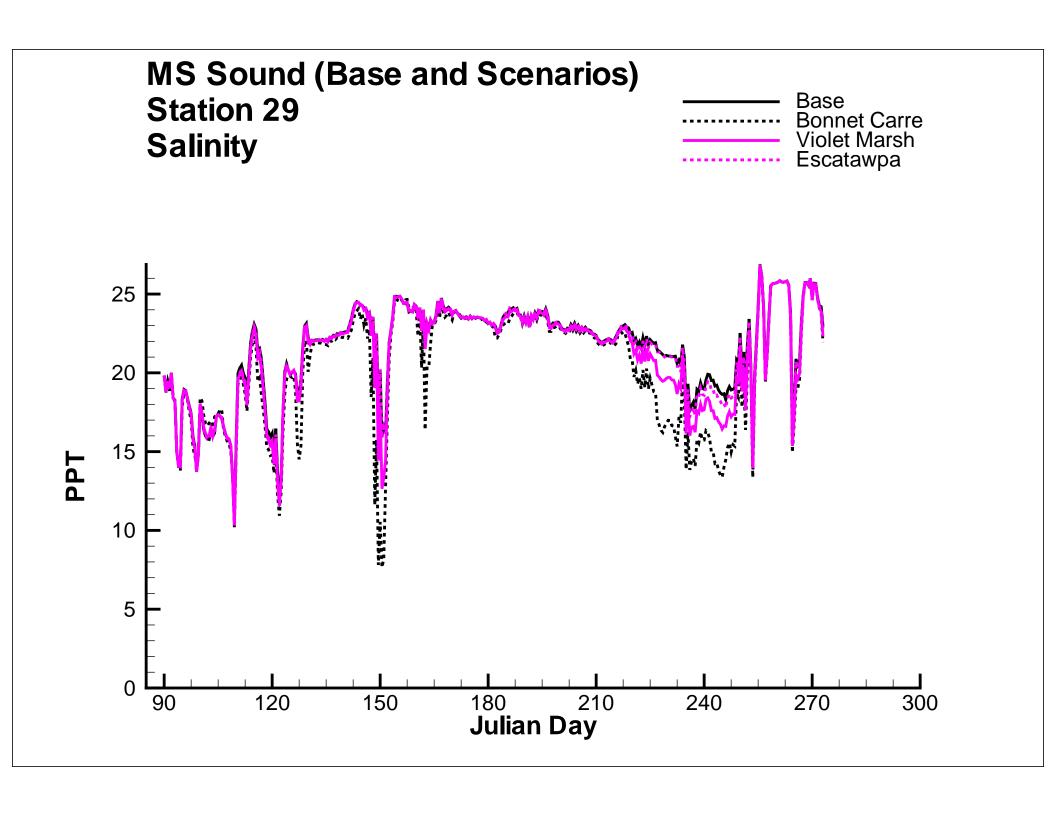


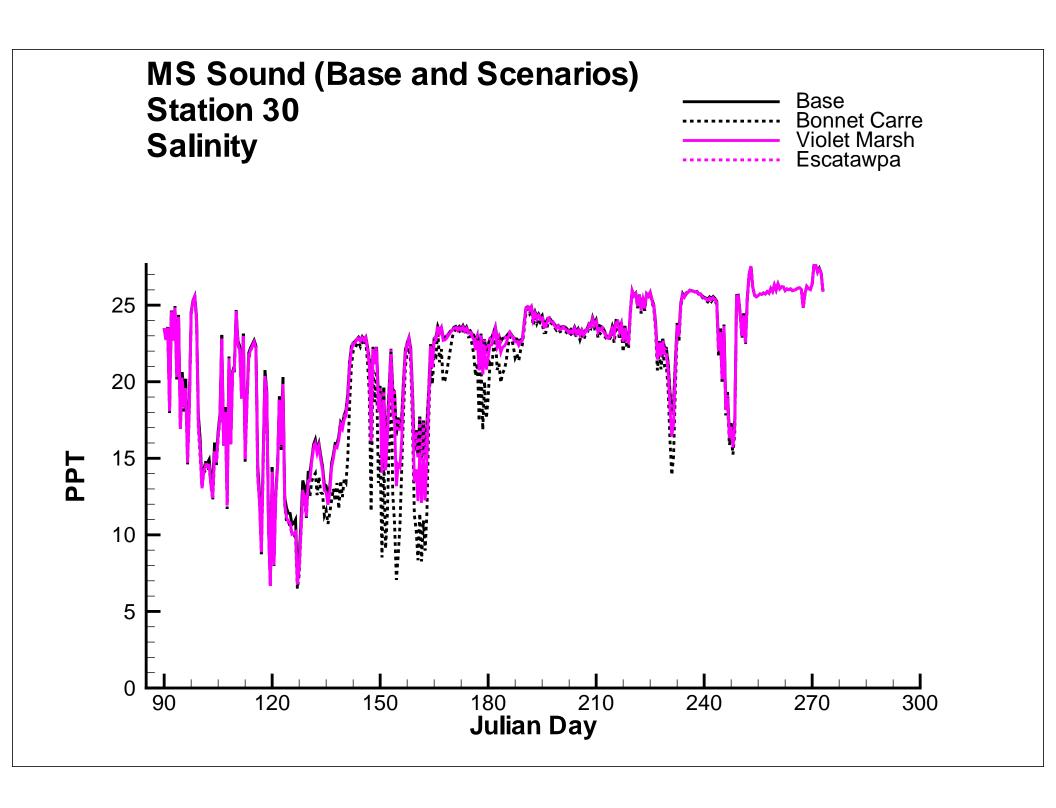






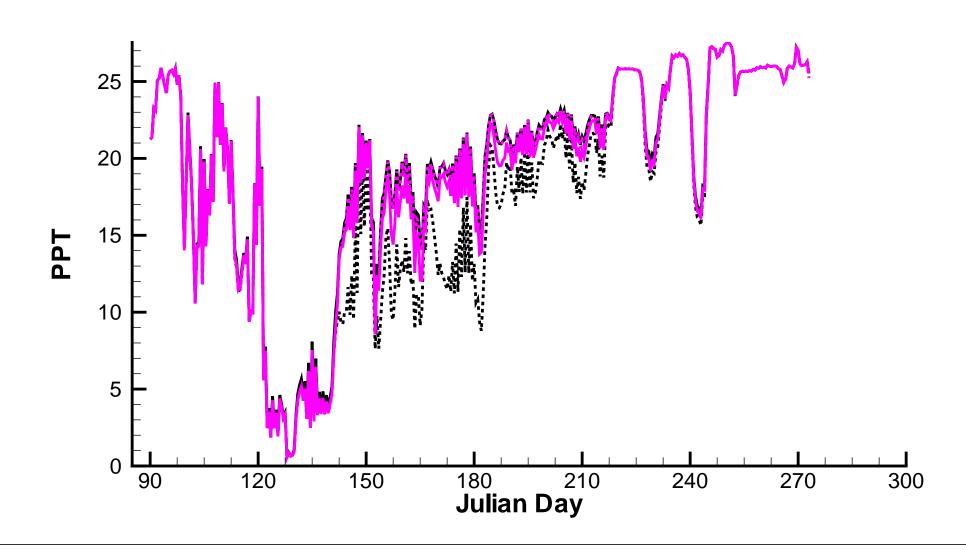


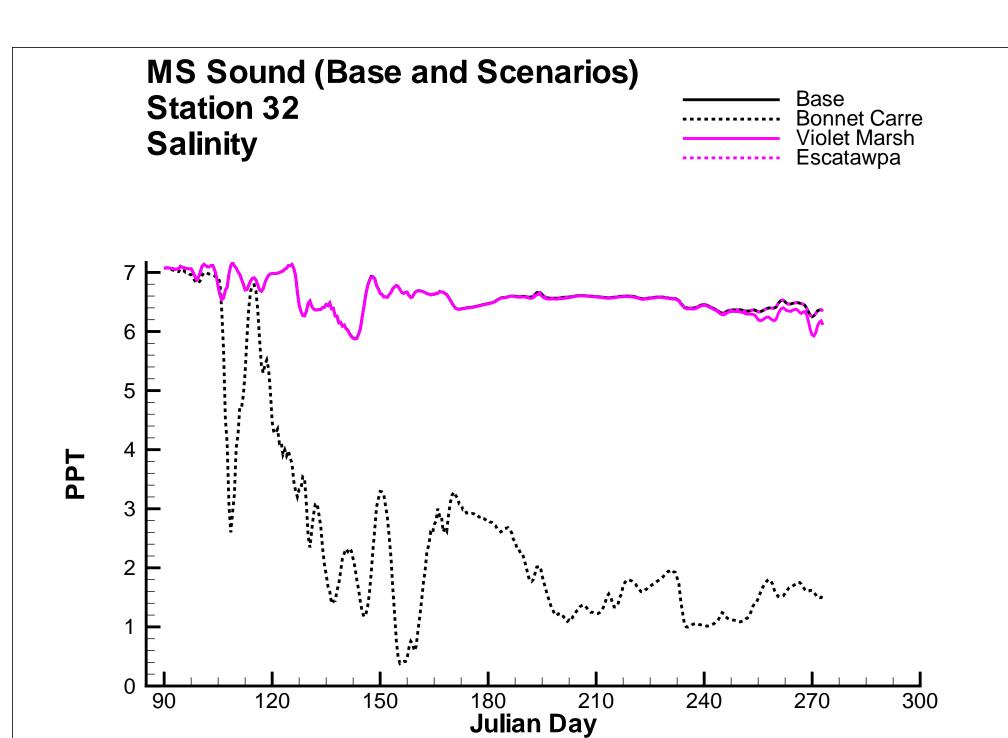


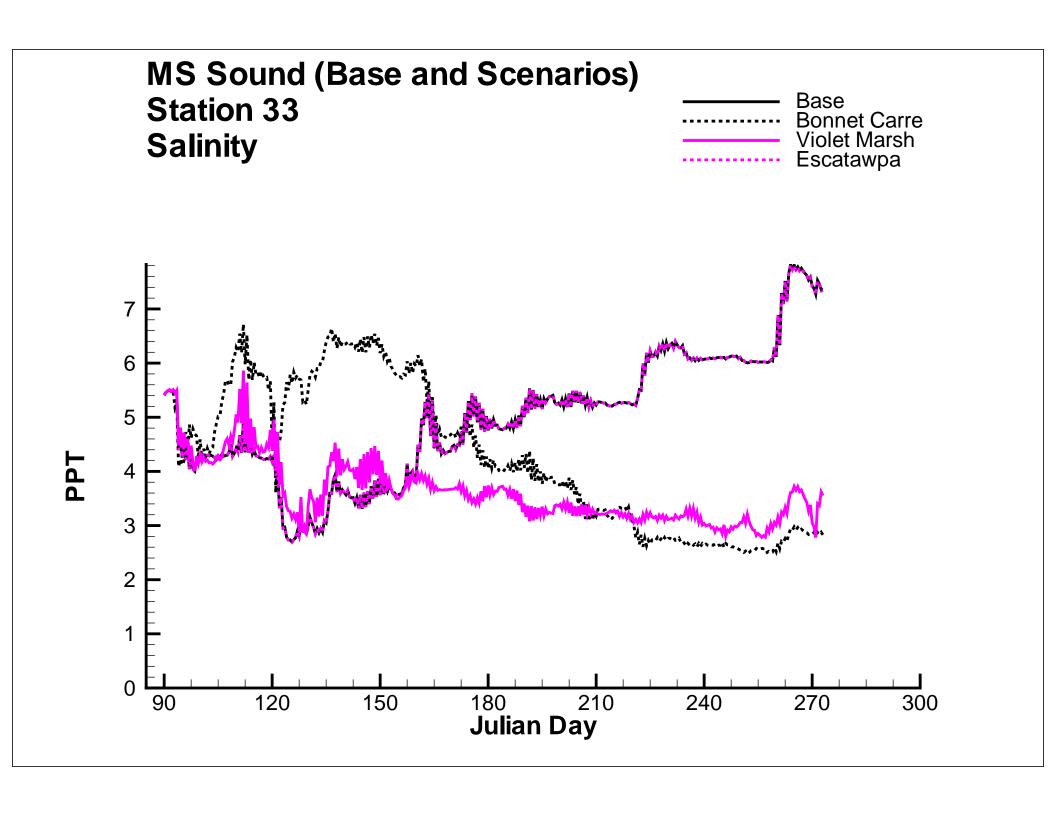




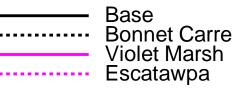


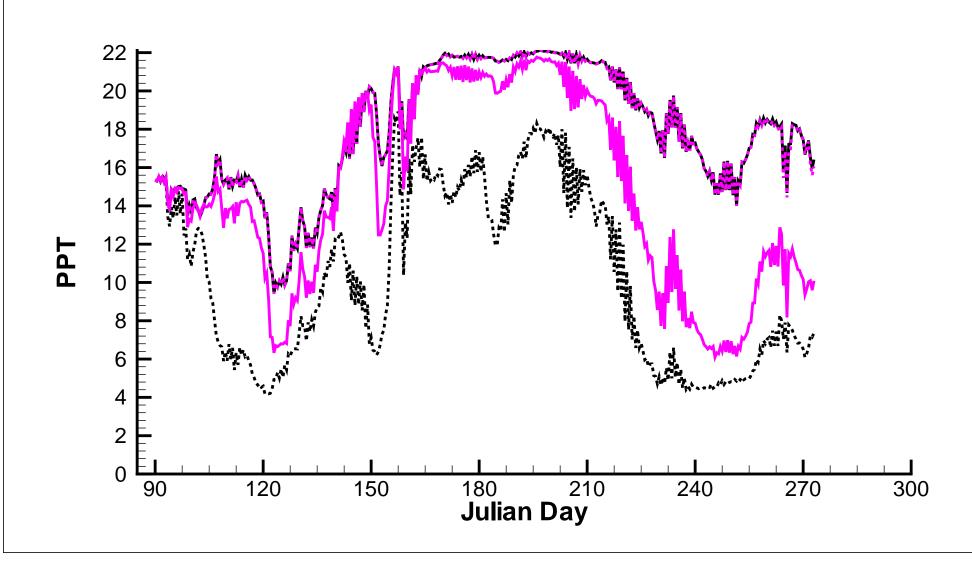


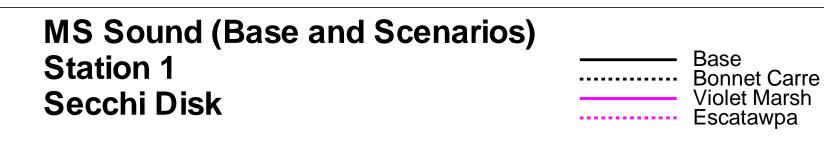


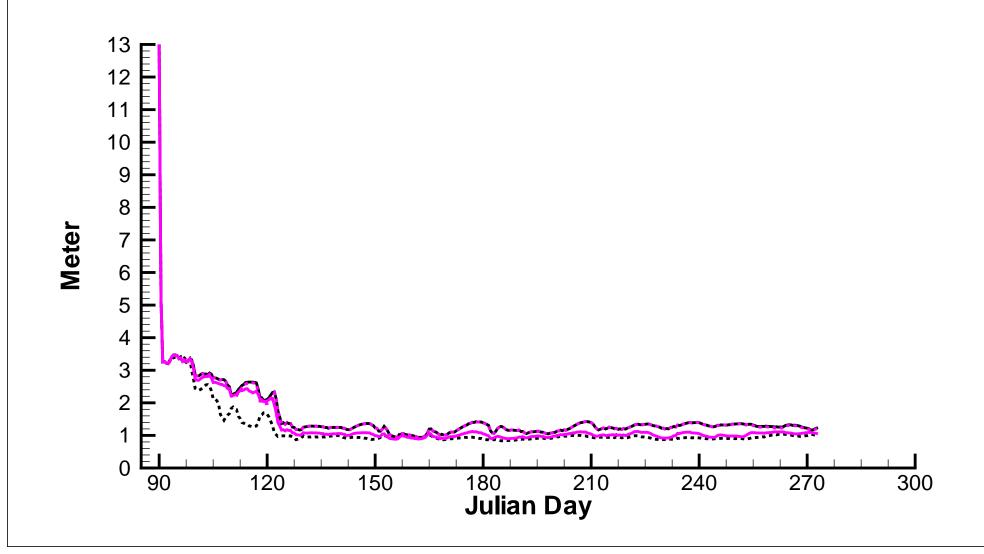


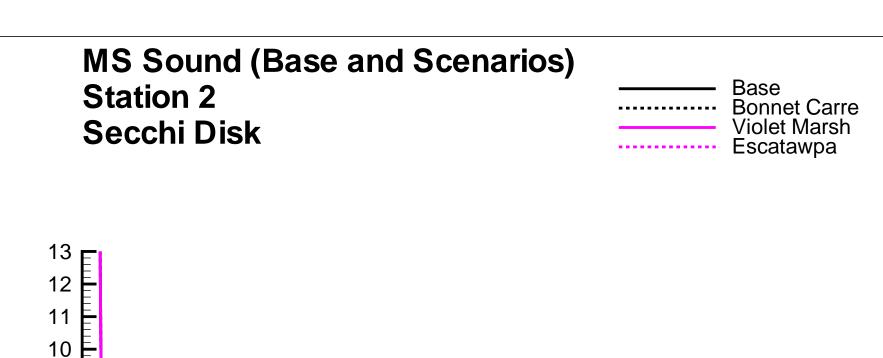


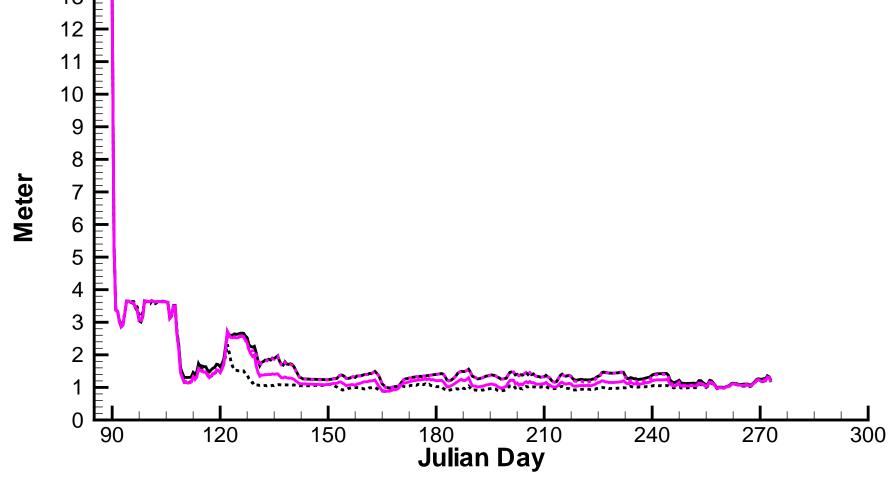


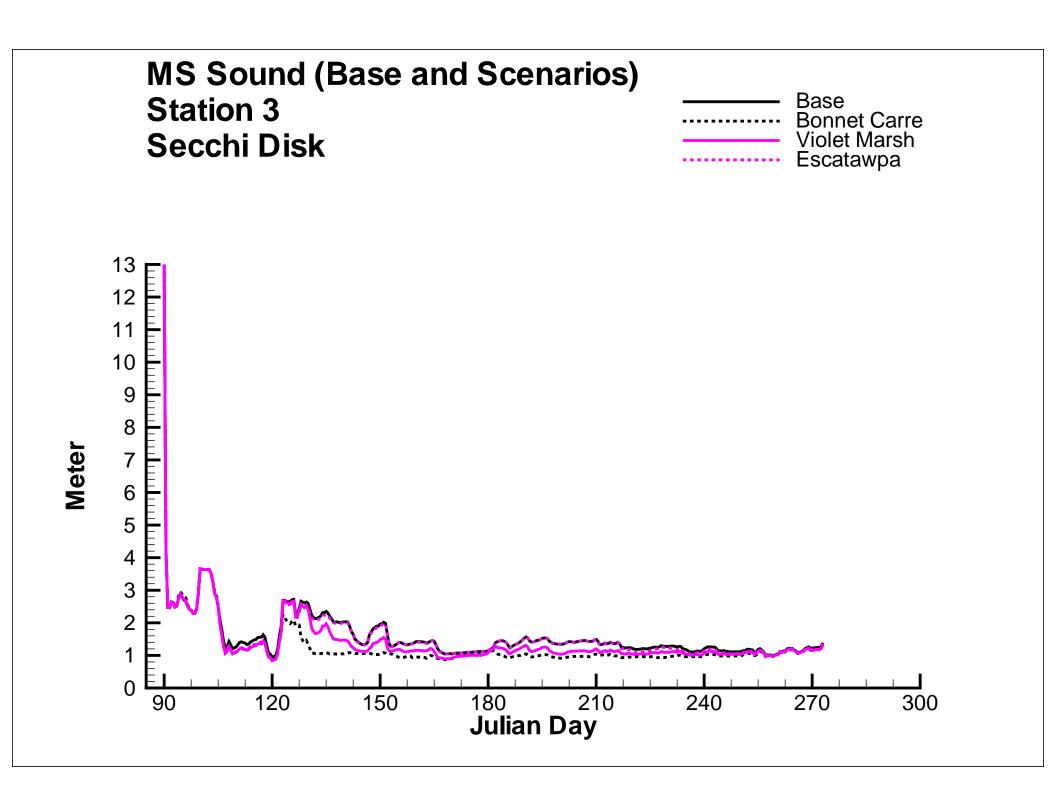


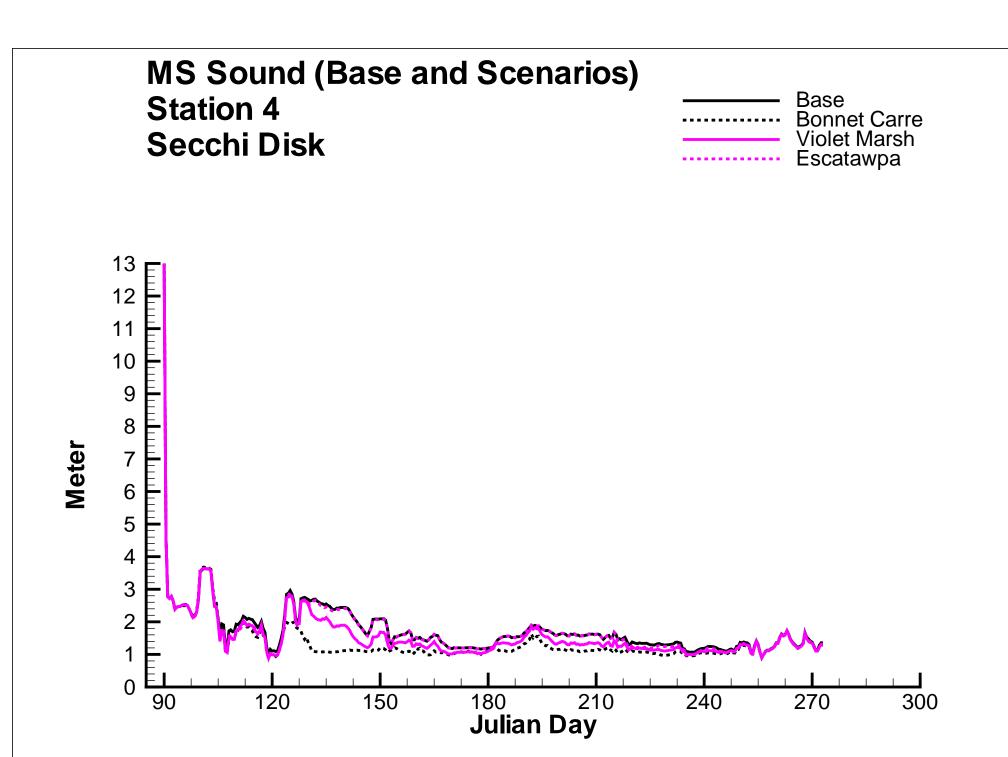


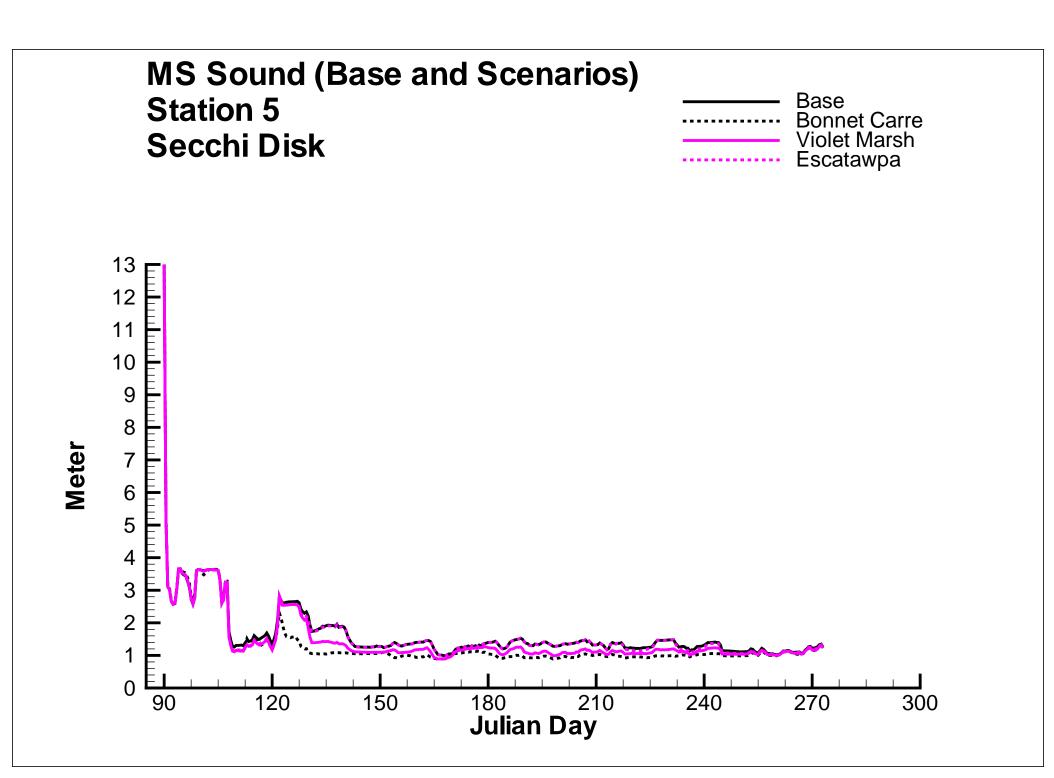


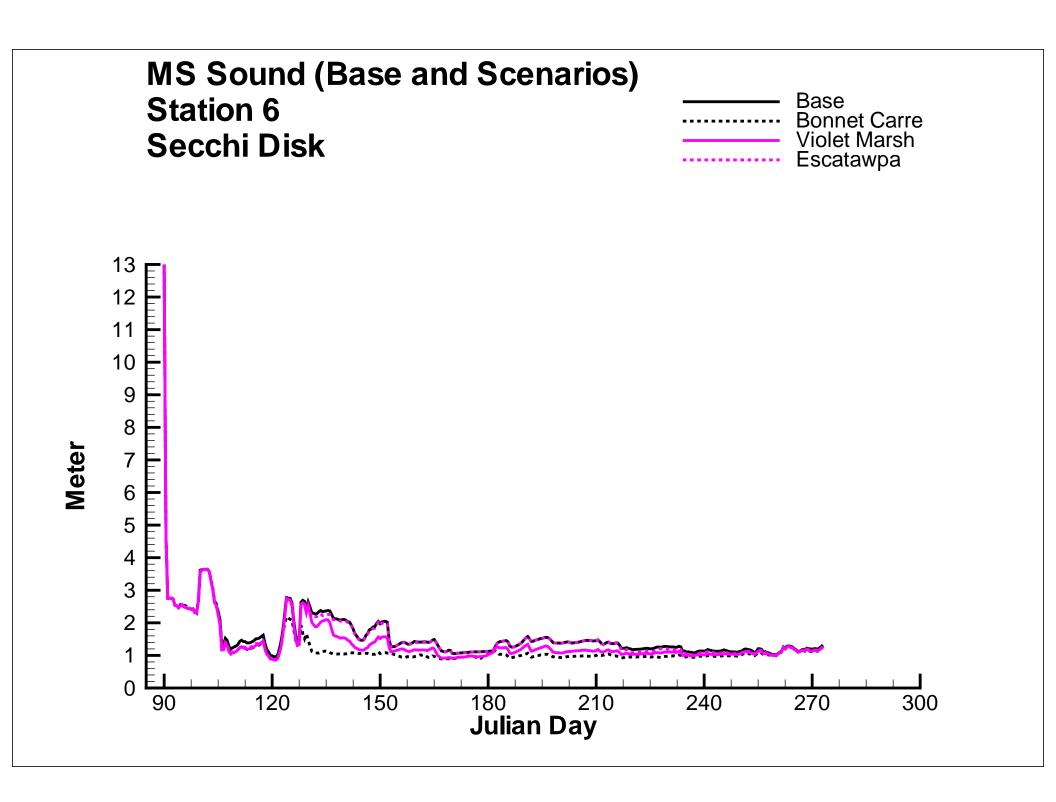


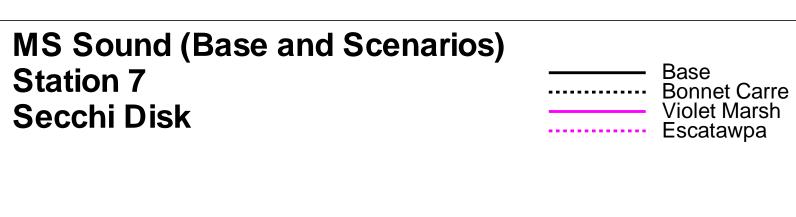


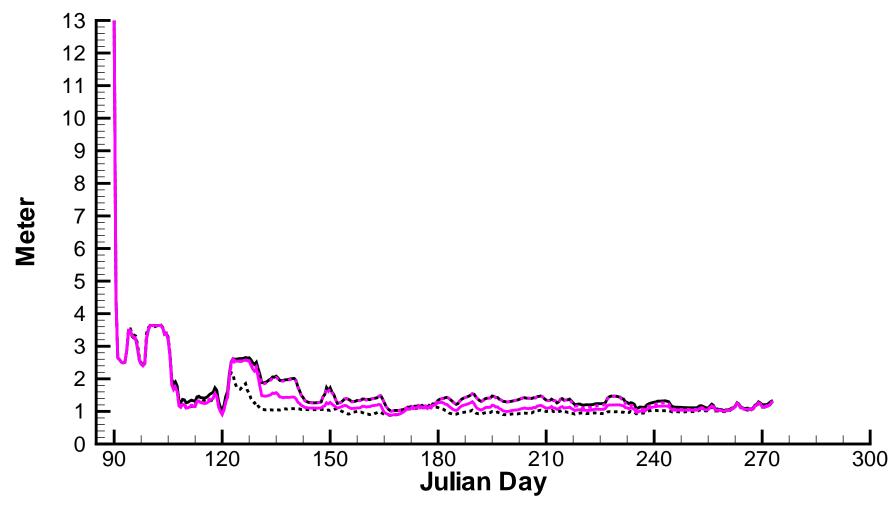


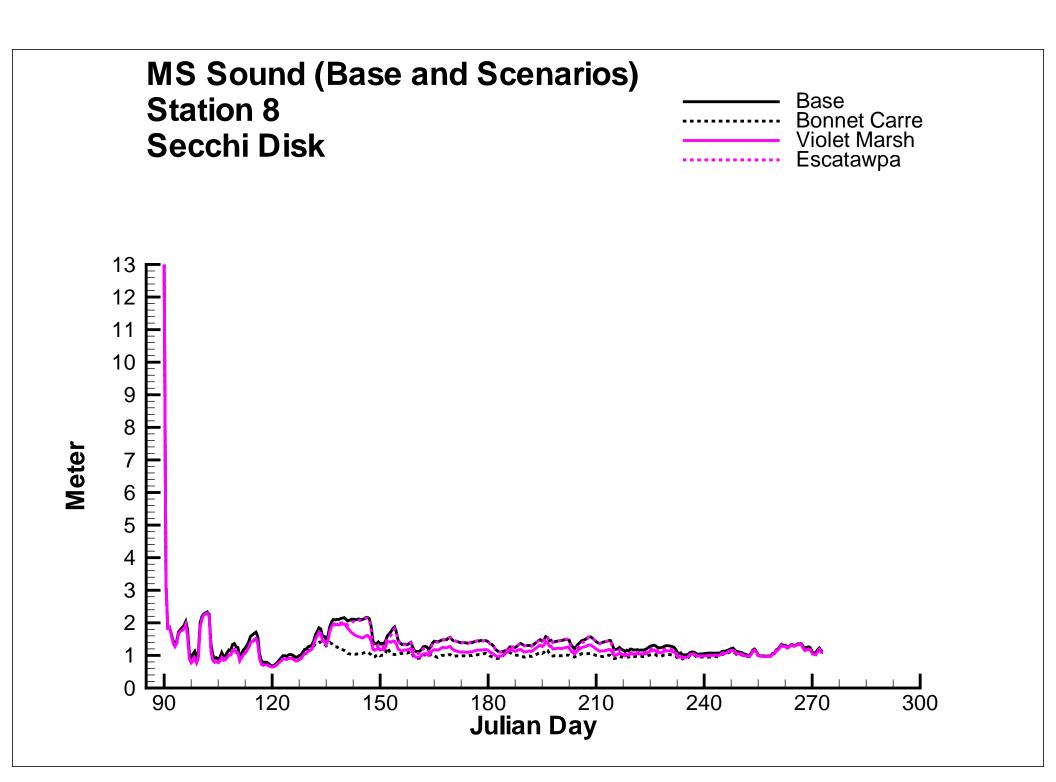


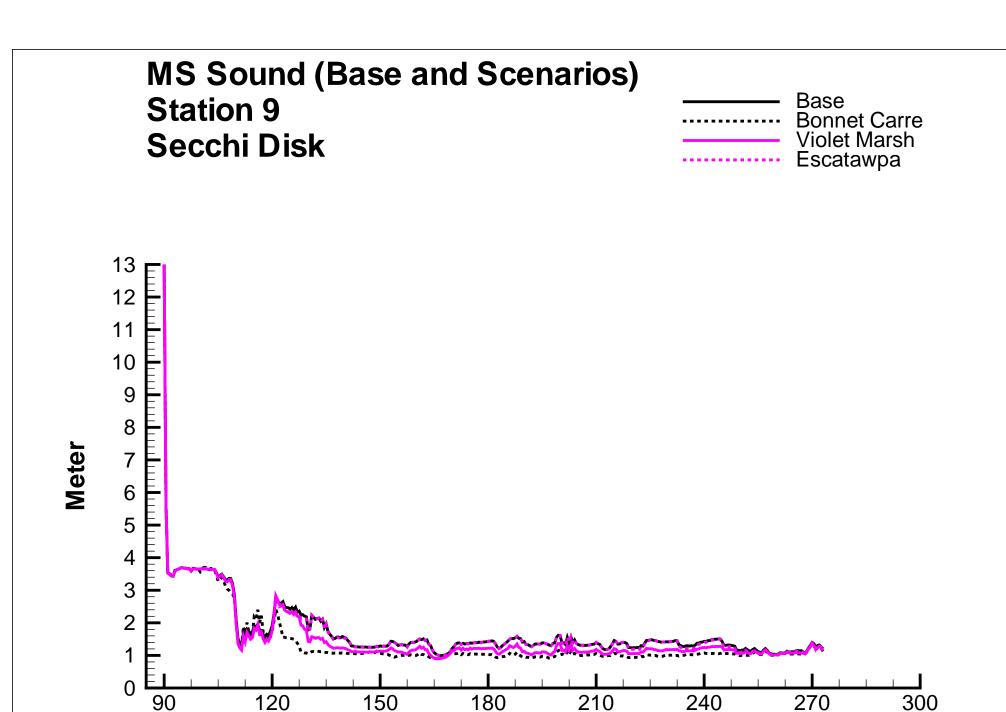




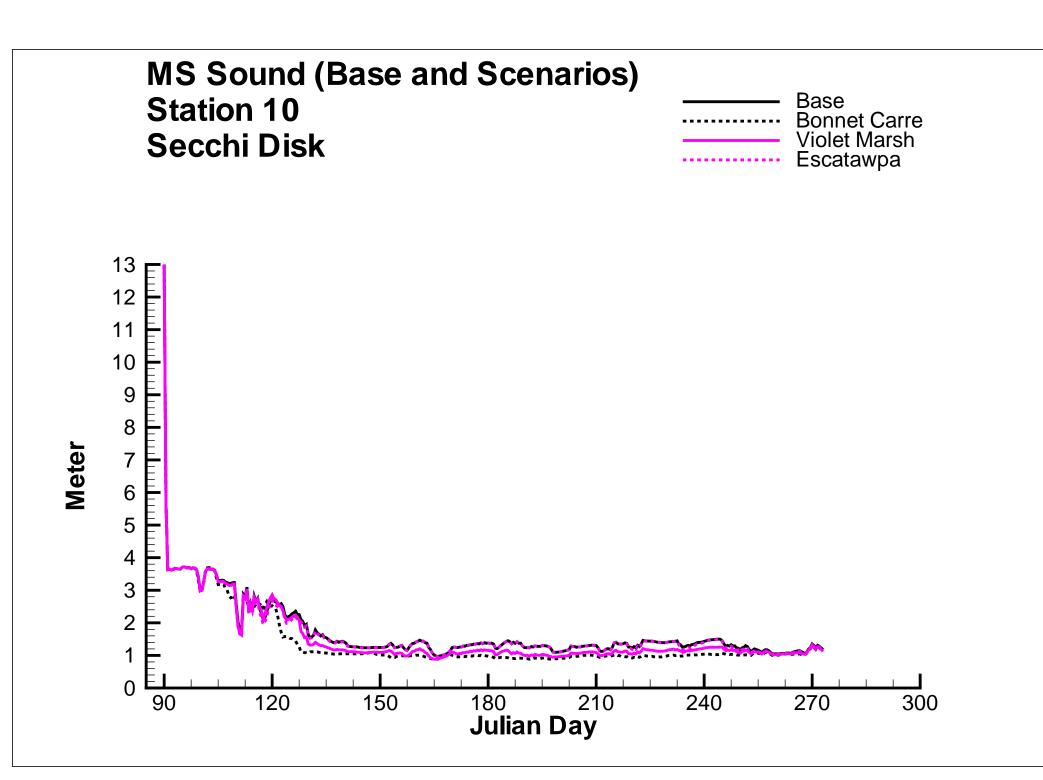






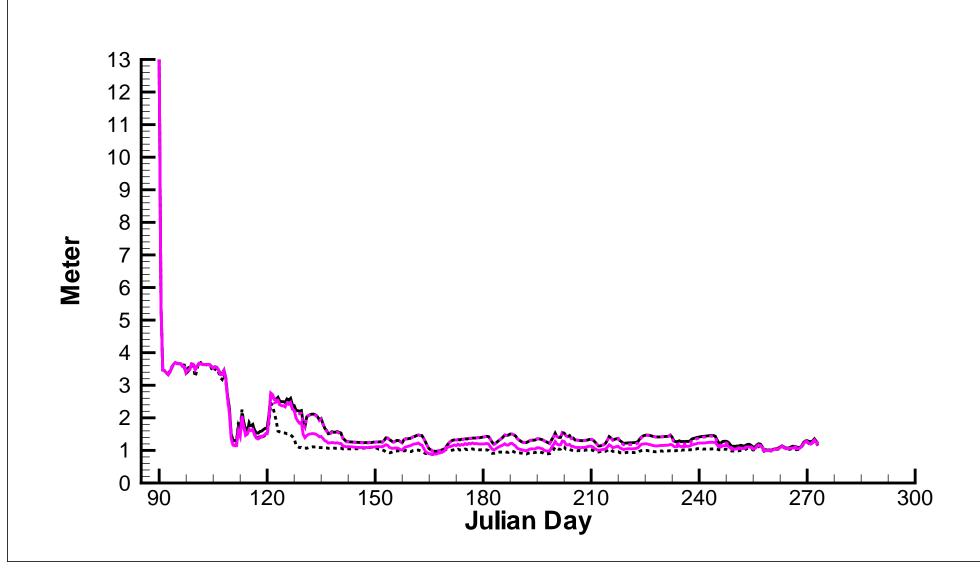


Julian Day



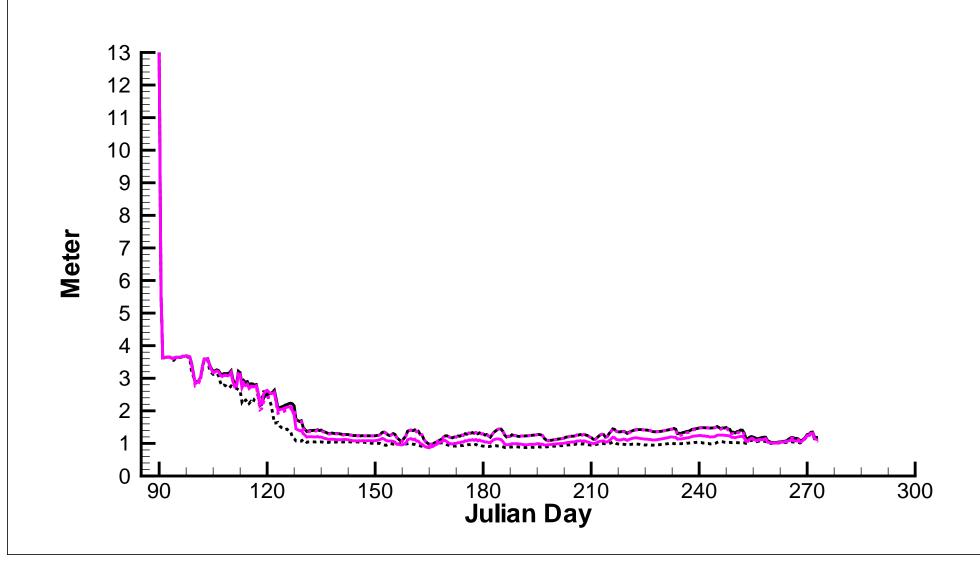


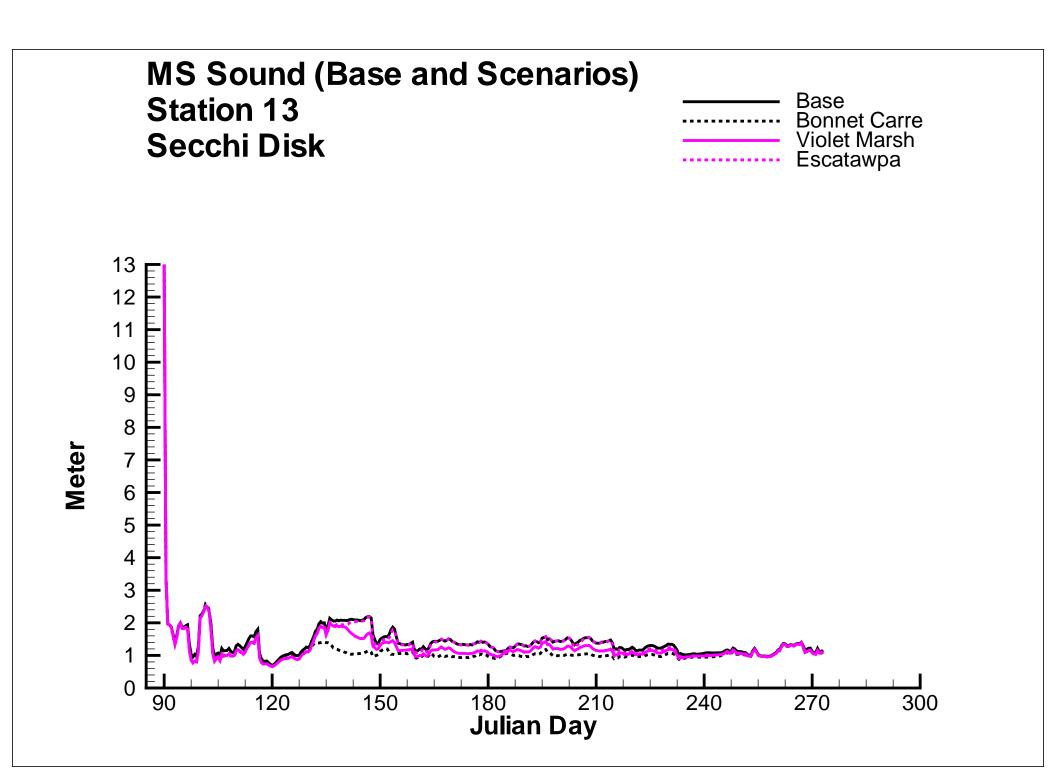


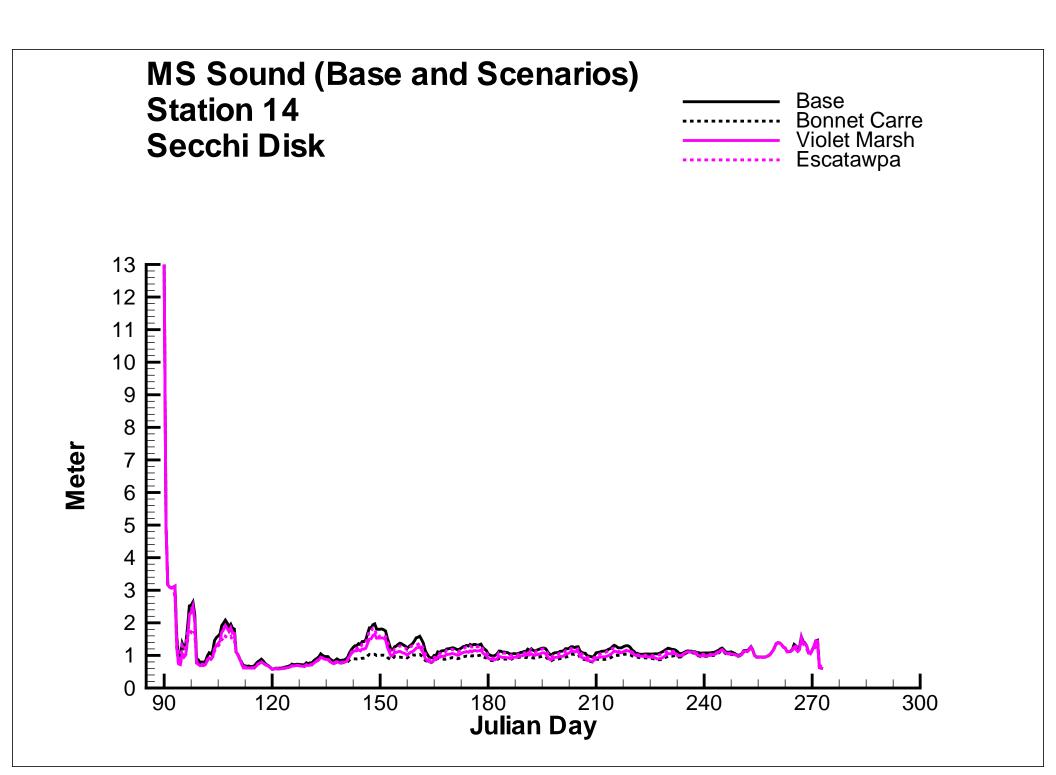






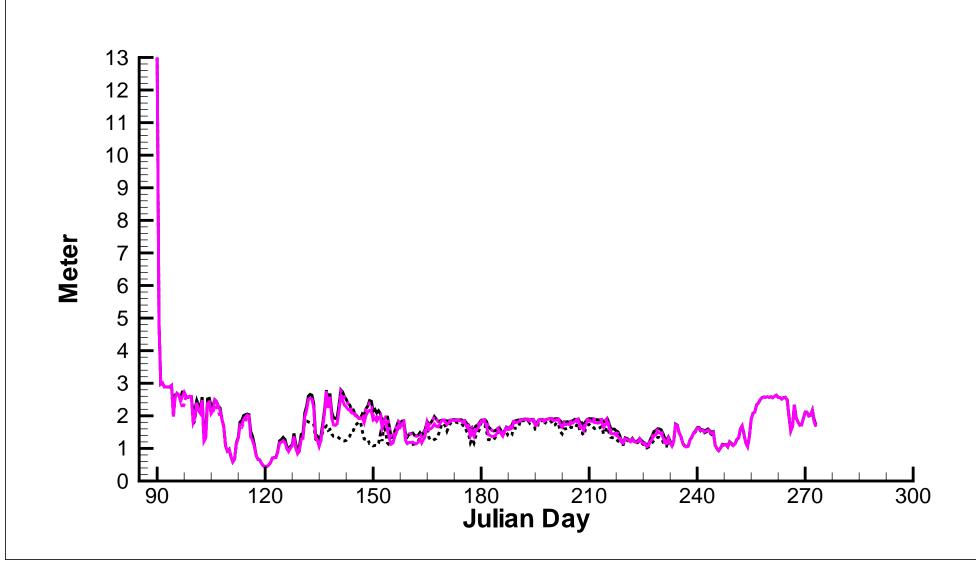


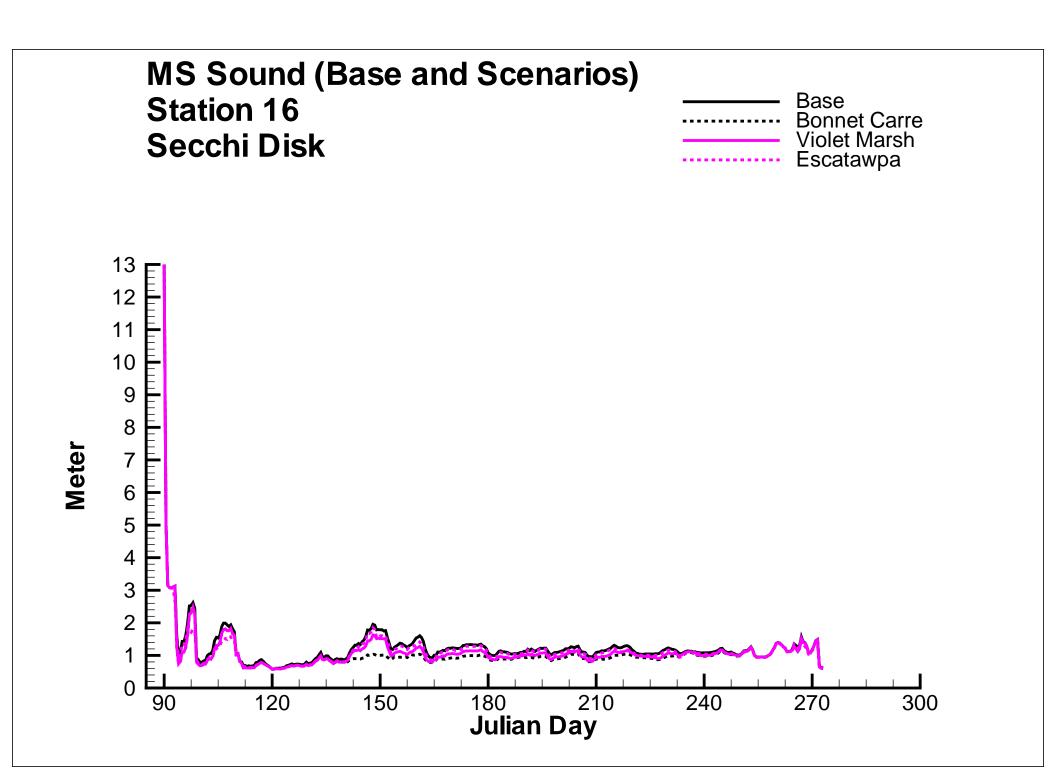


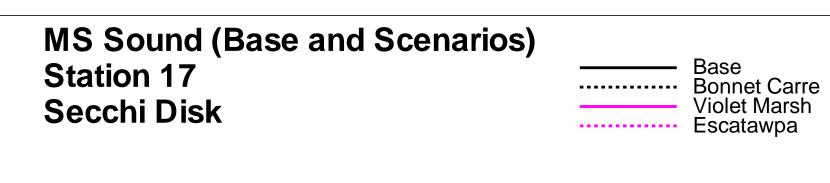


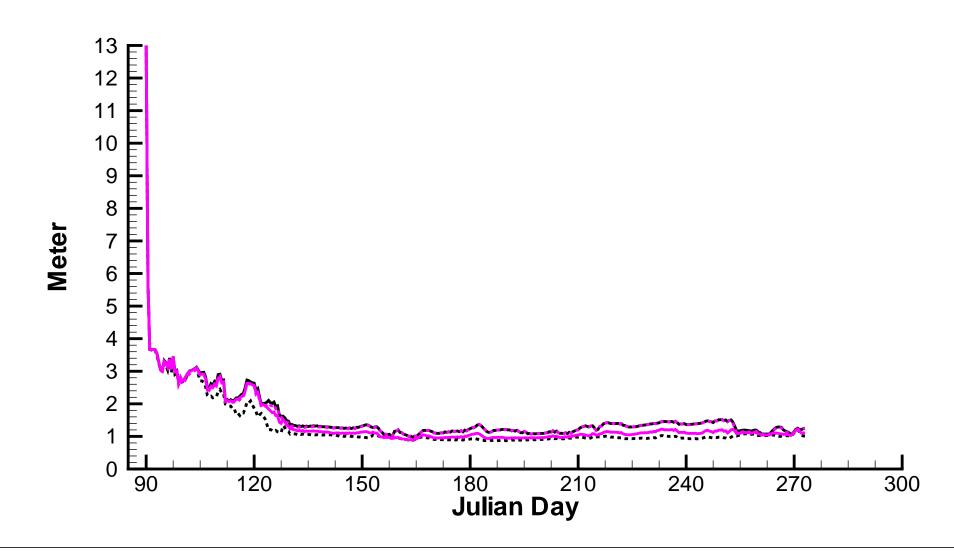






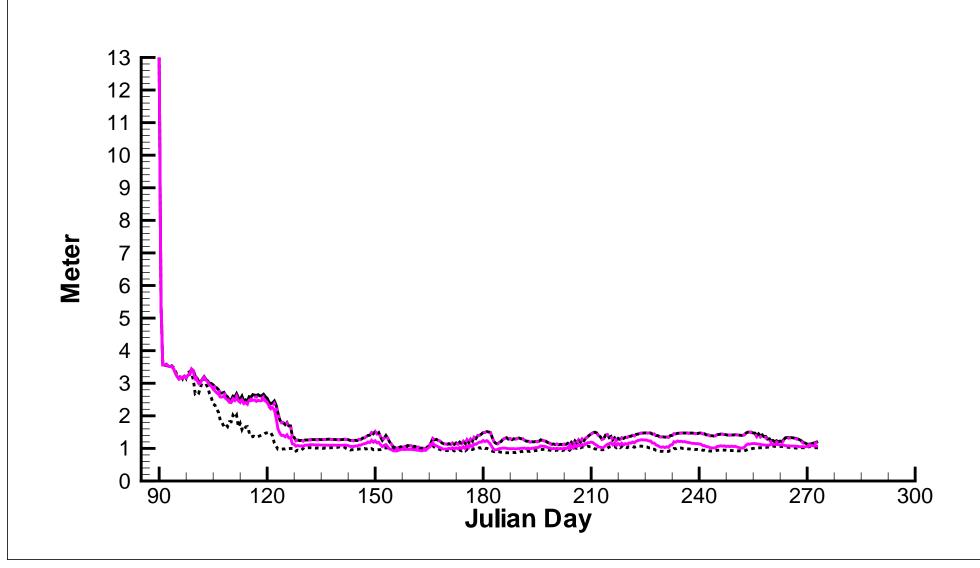


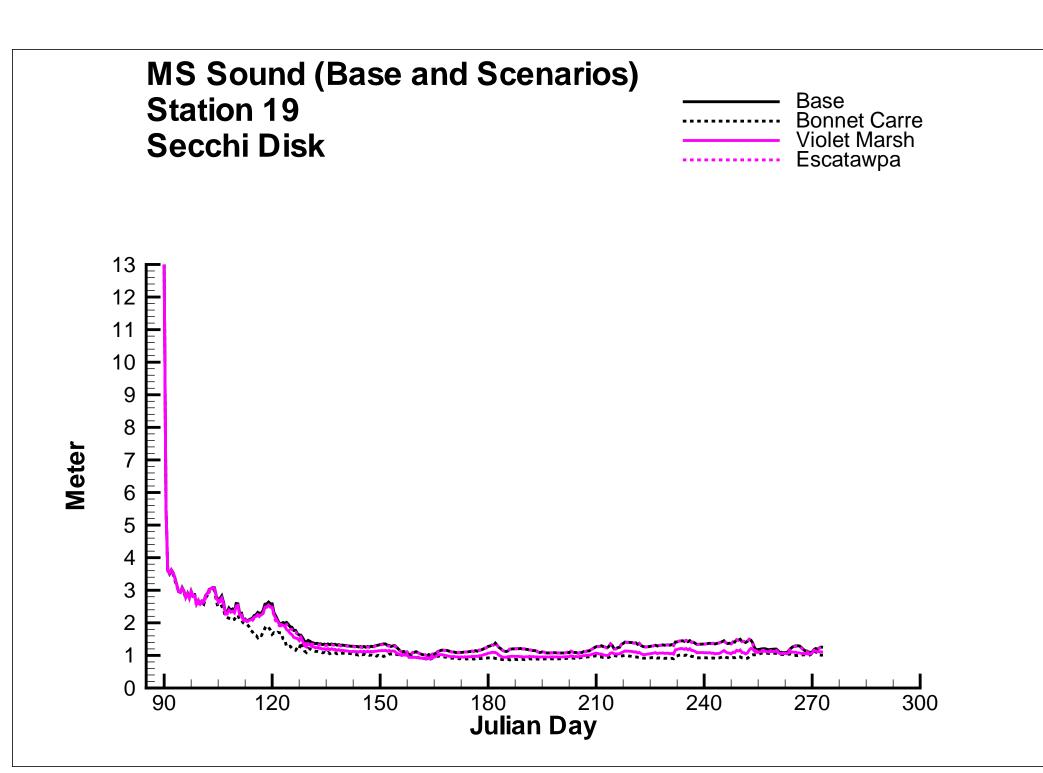


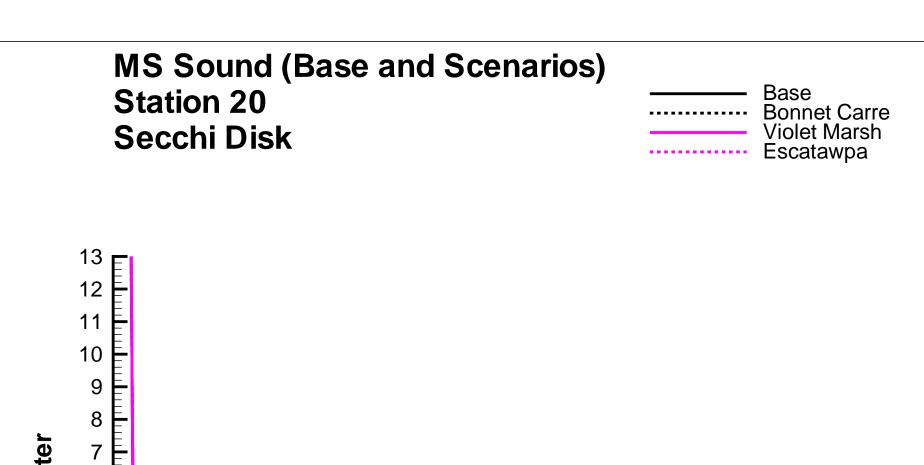


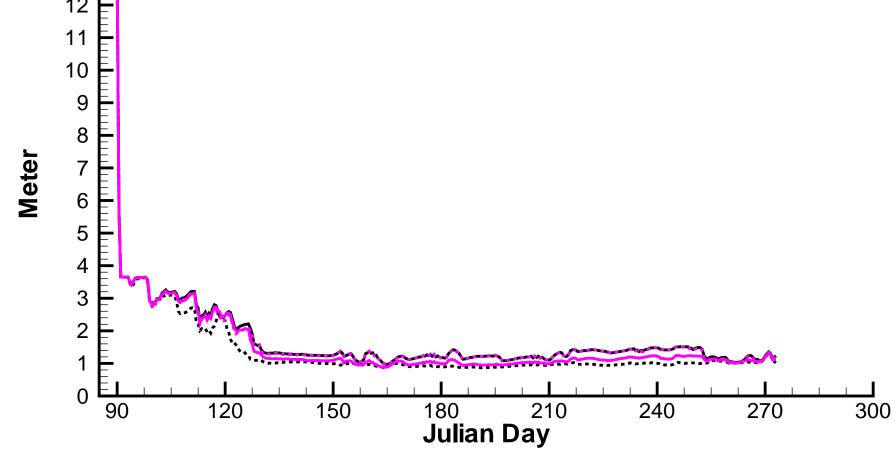


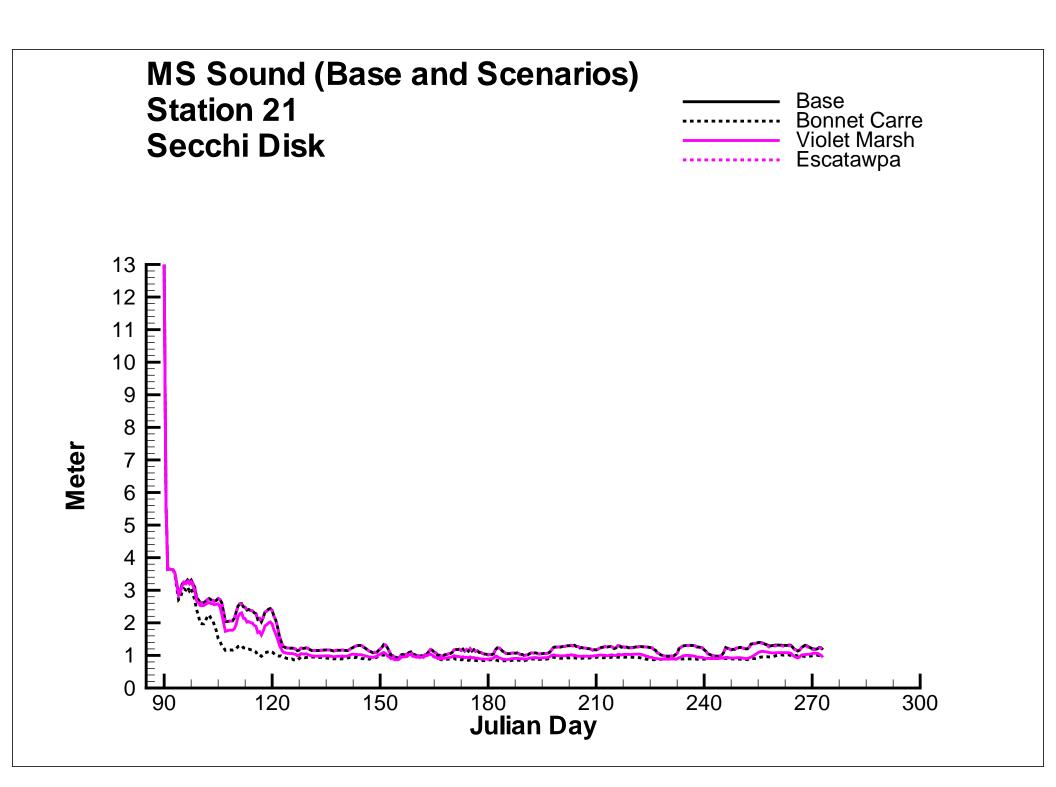


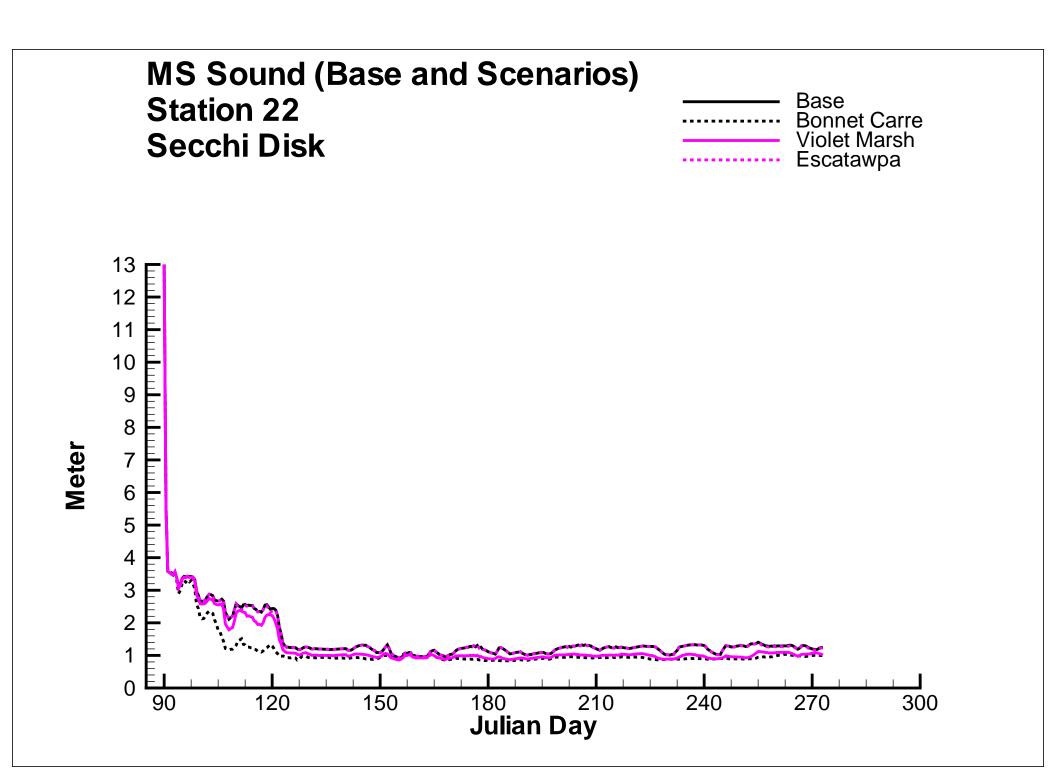


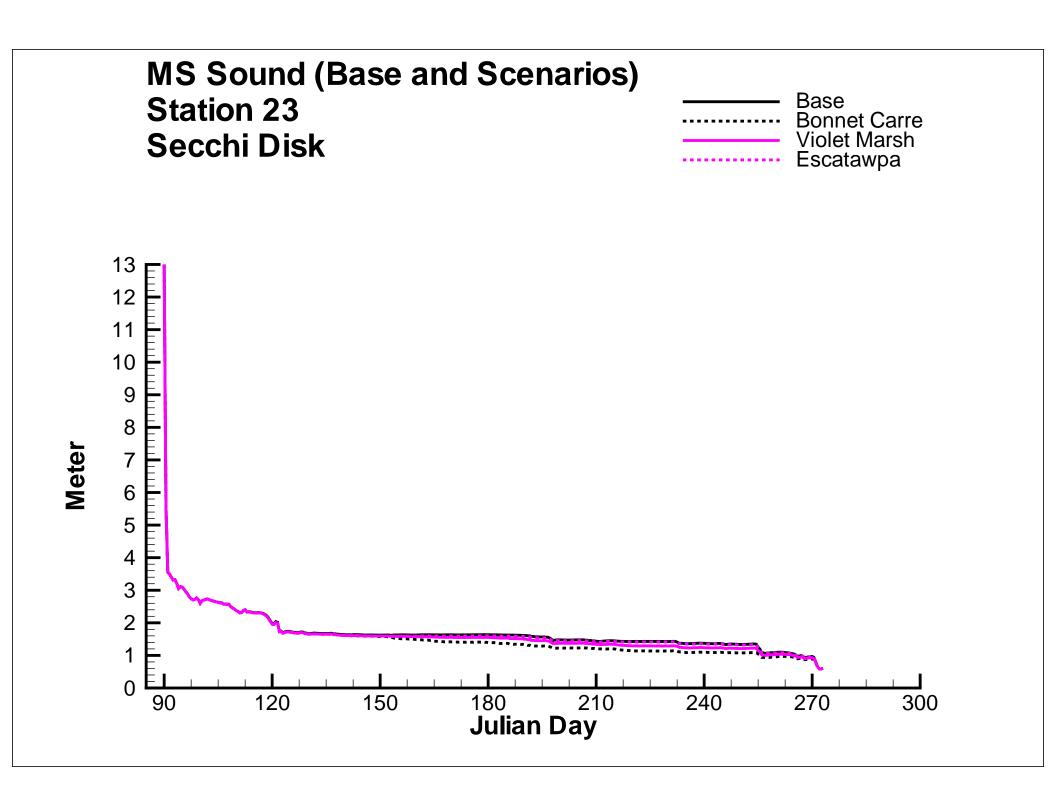


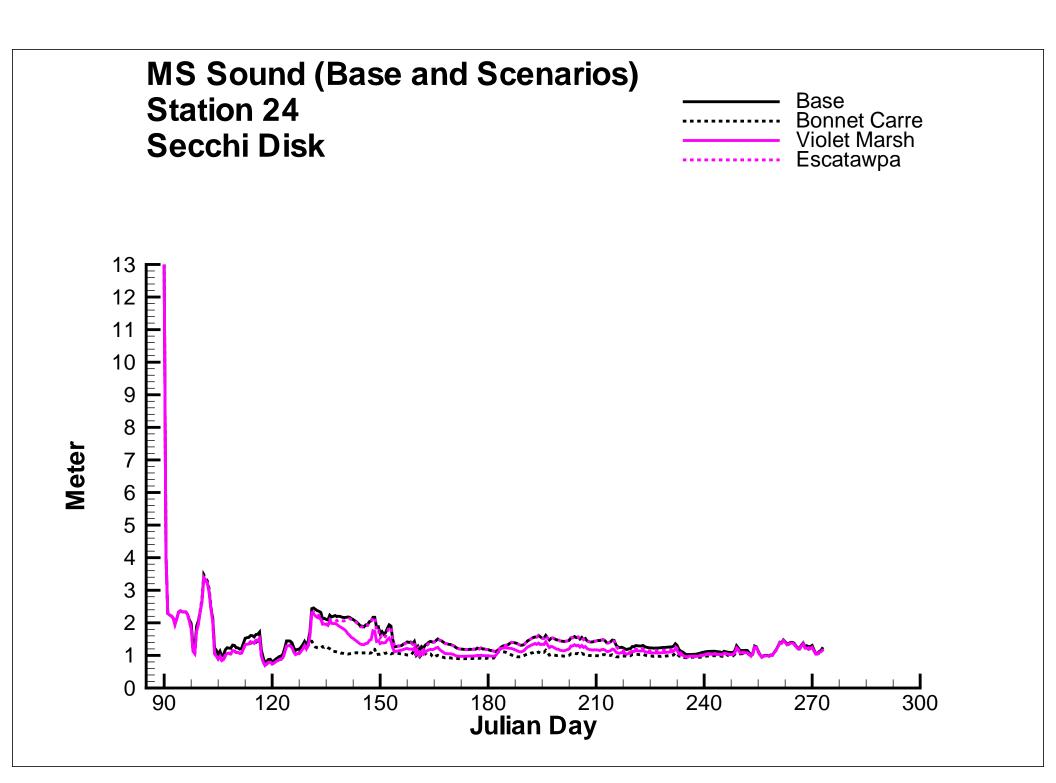


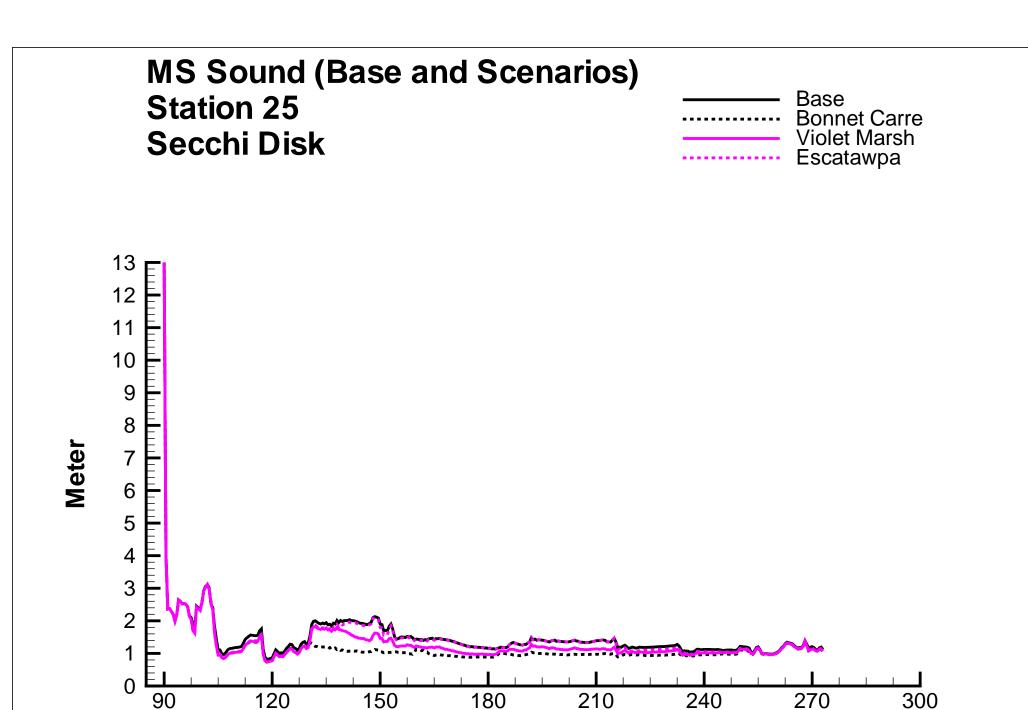




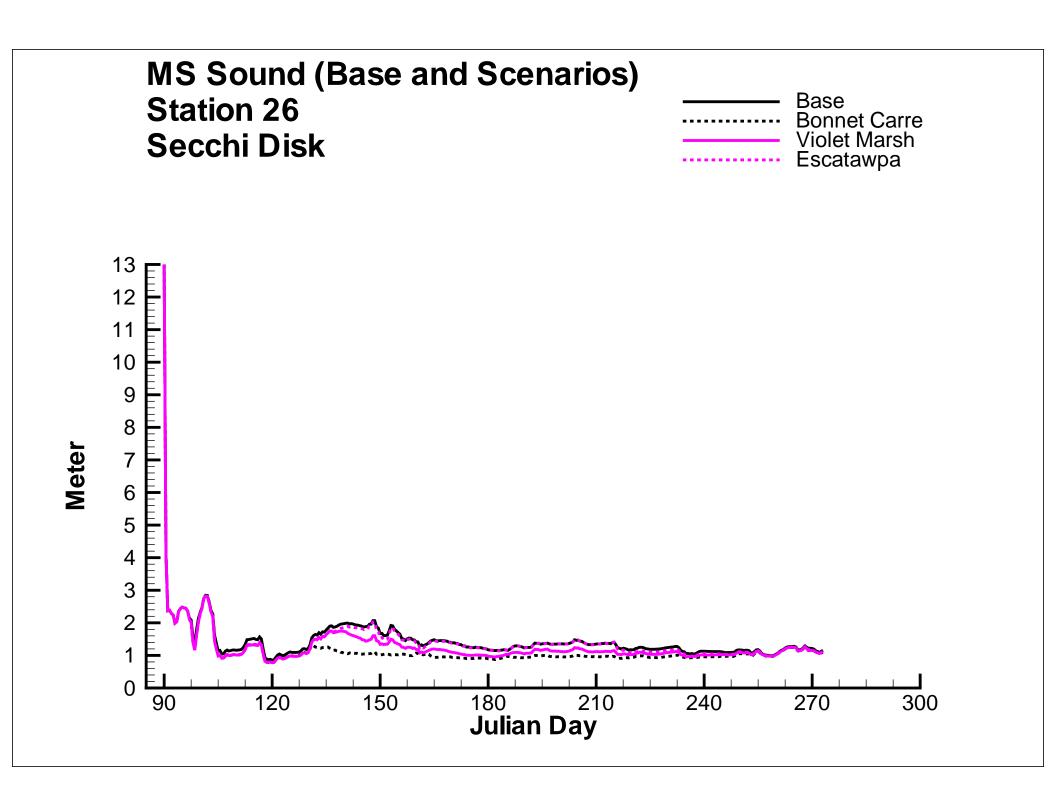


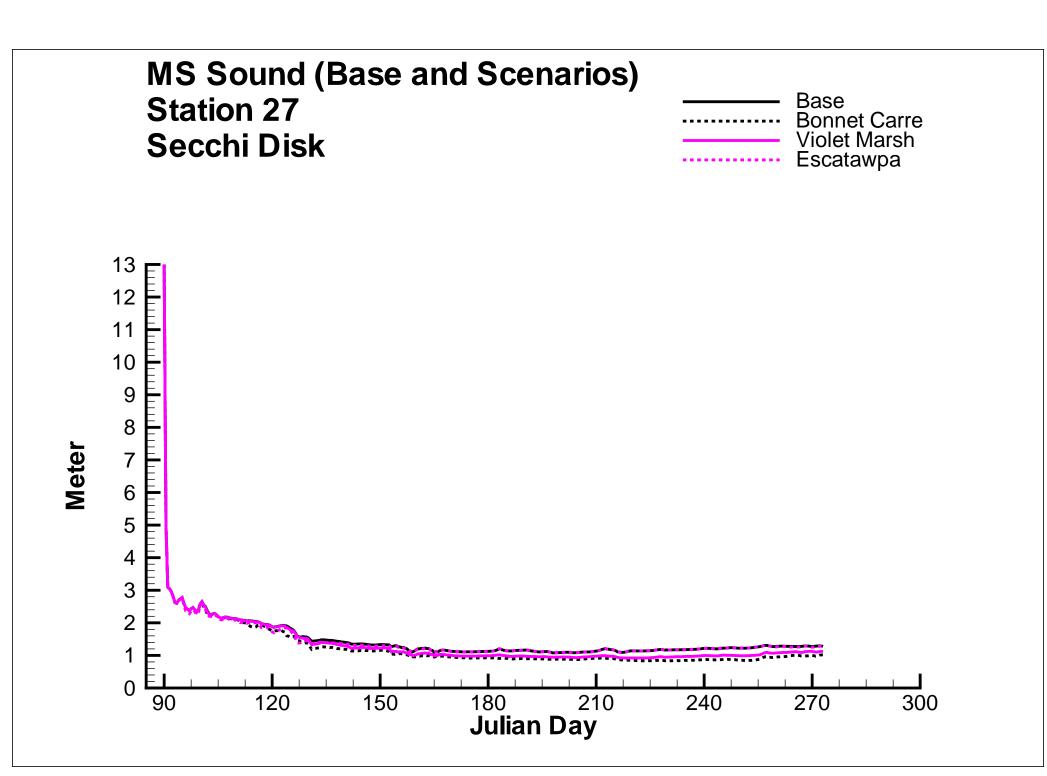


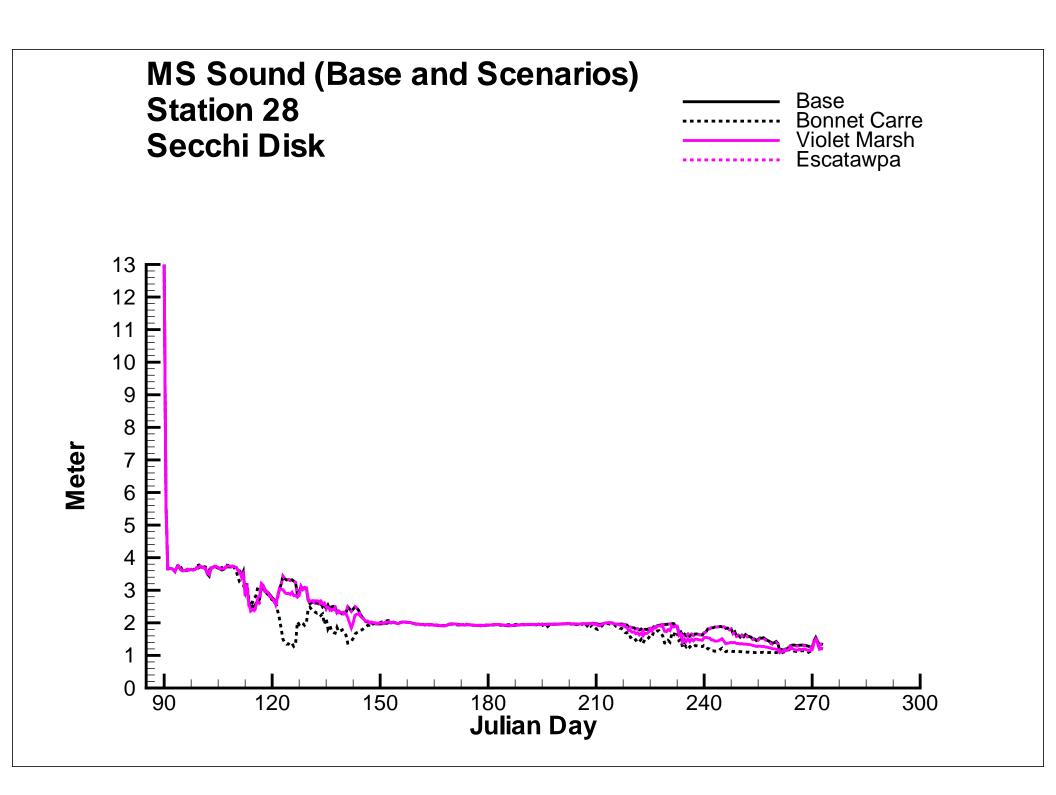


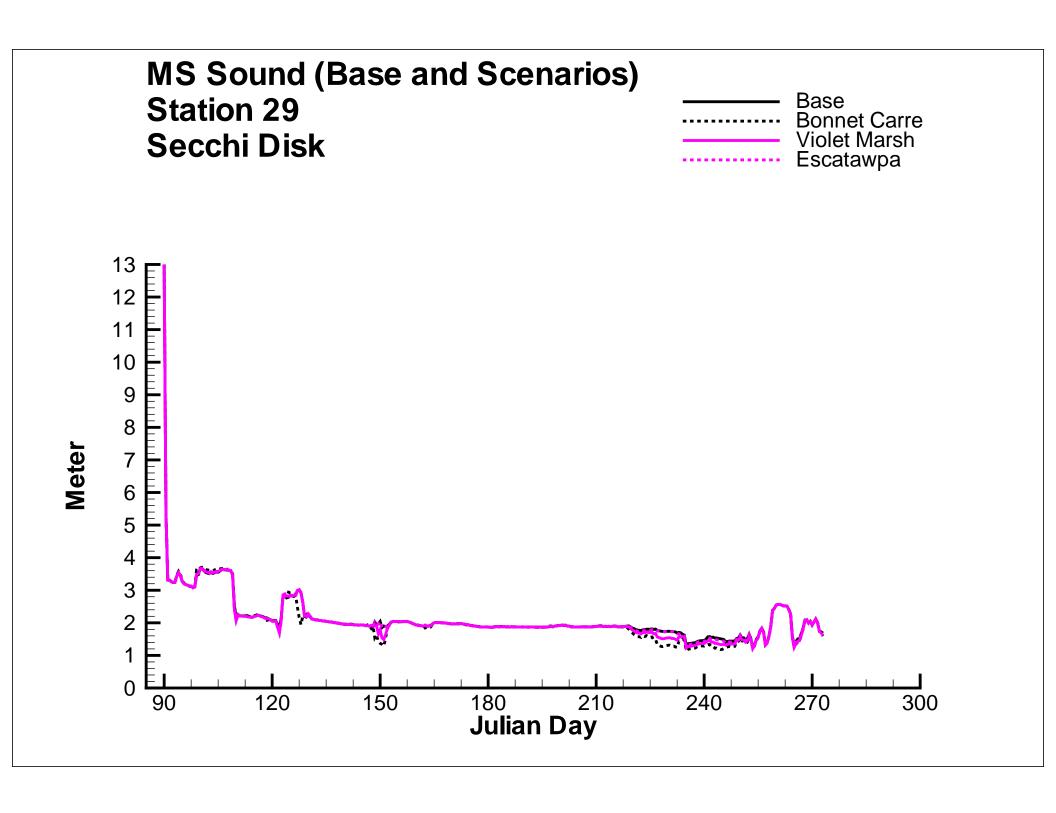


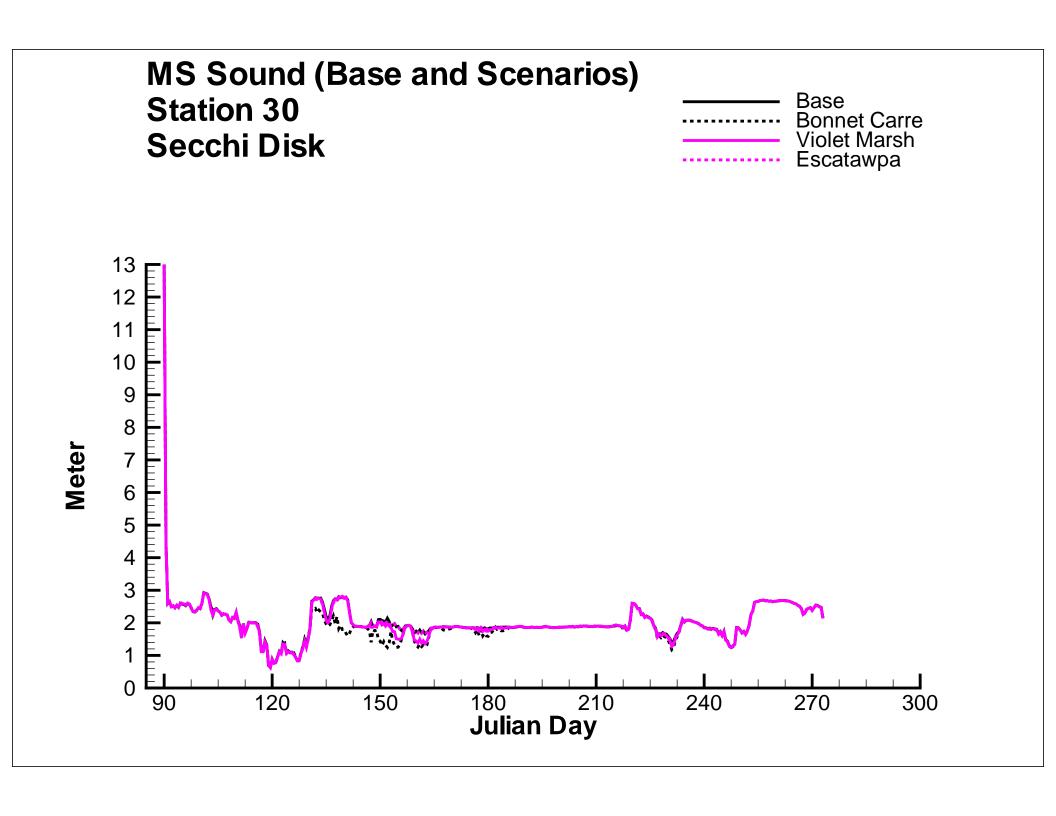
Julian Day

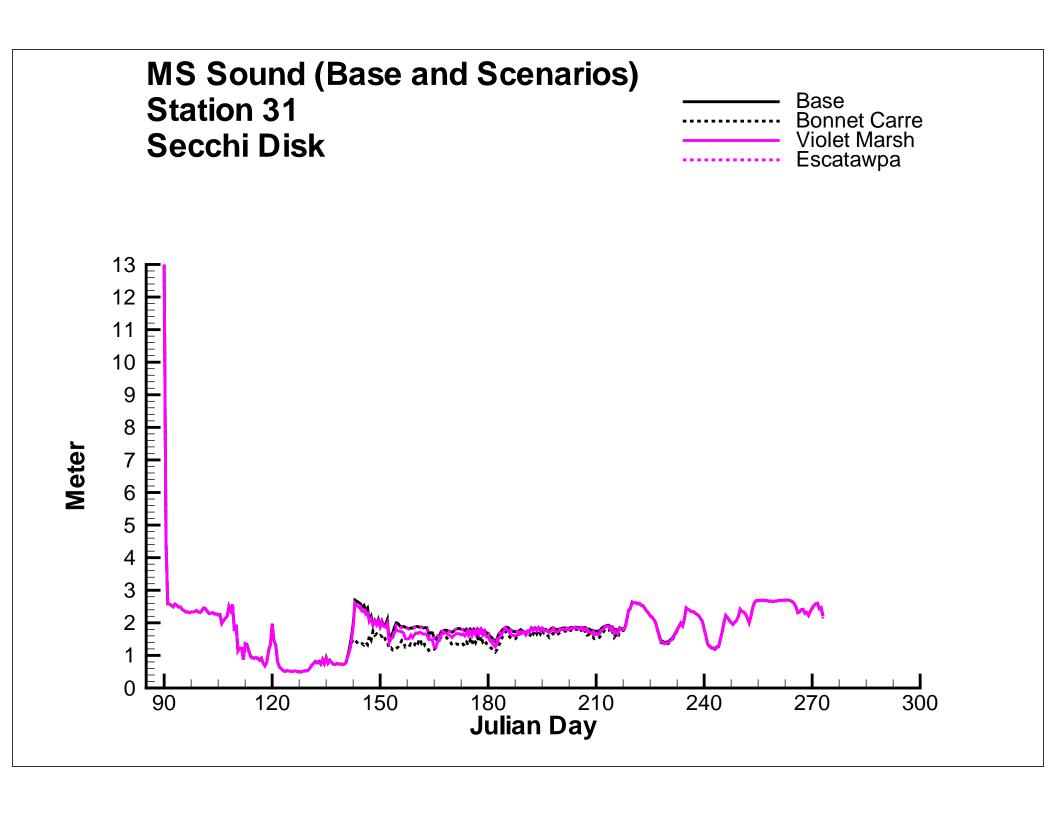


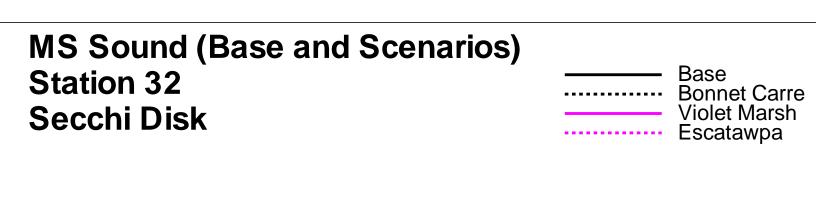


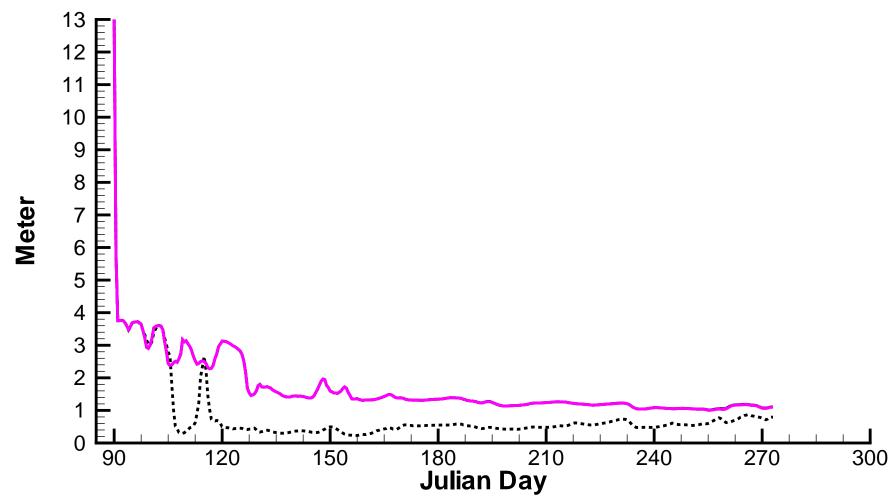


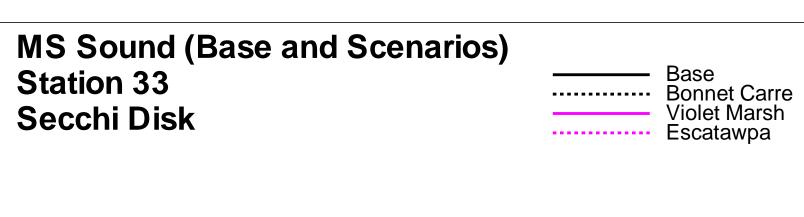


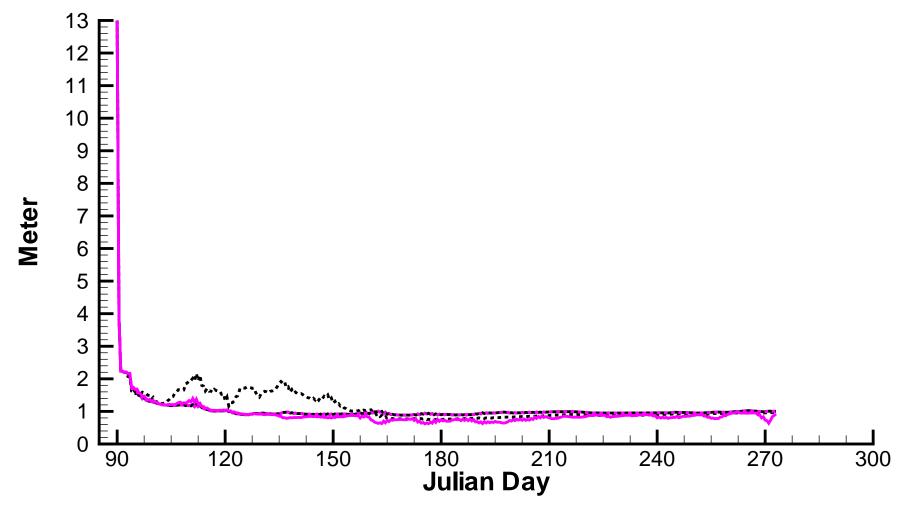


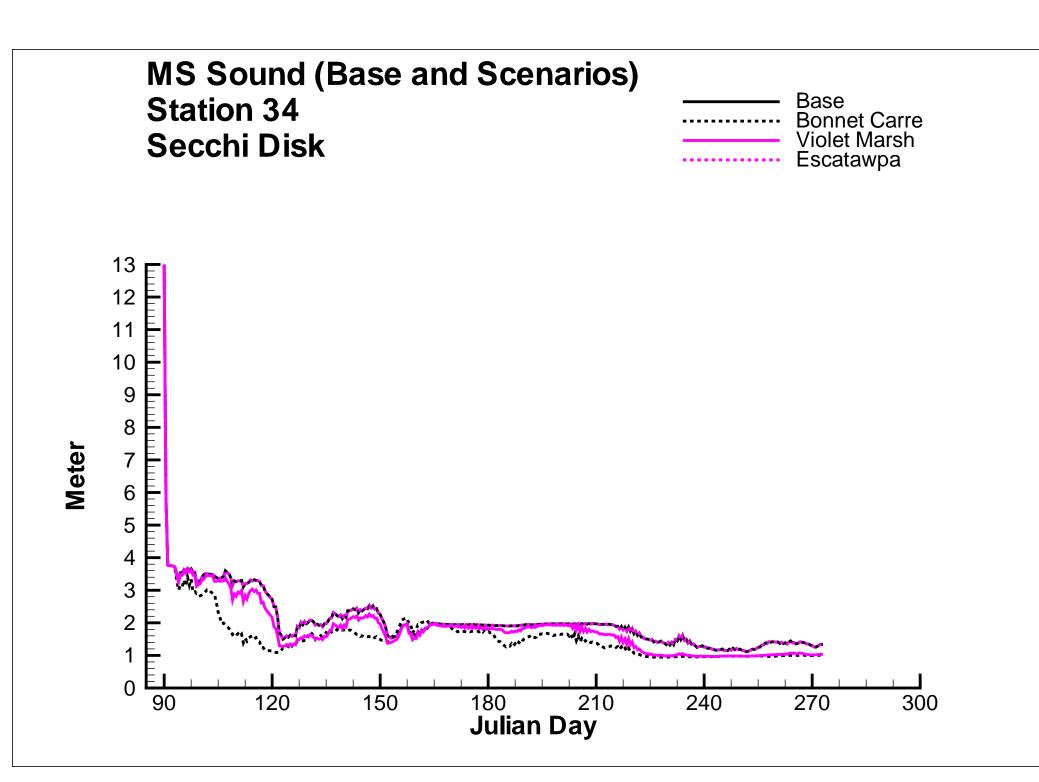






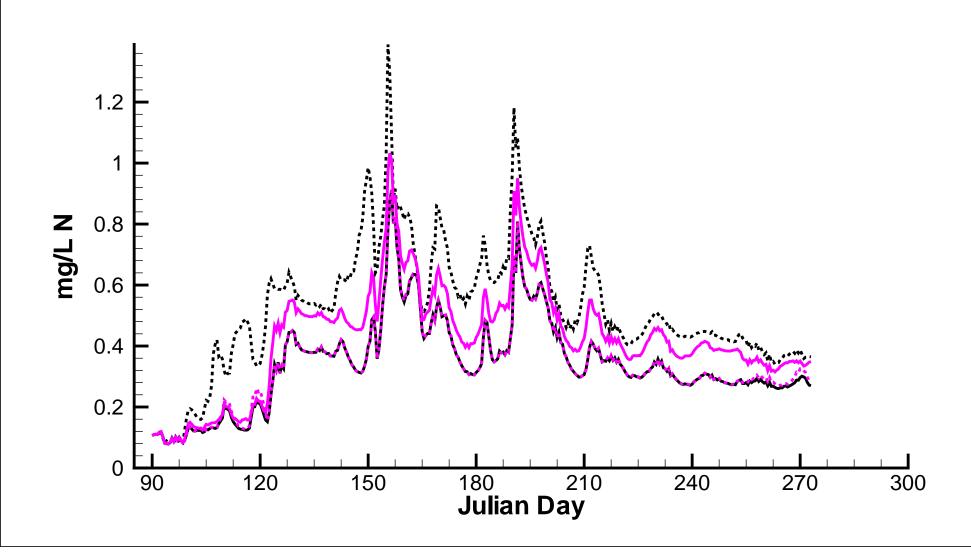


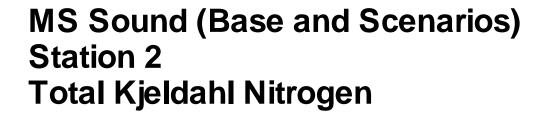




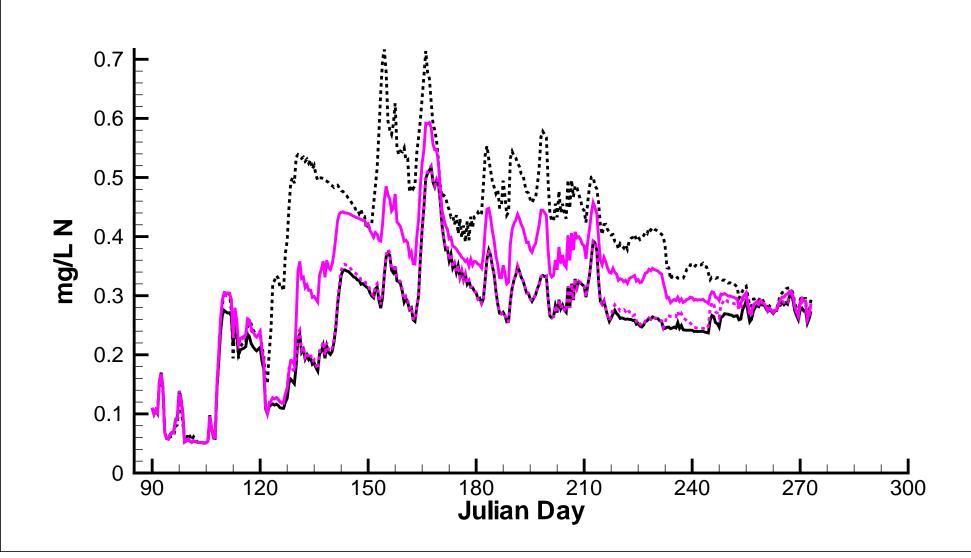


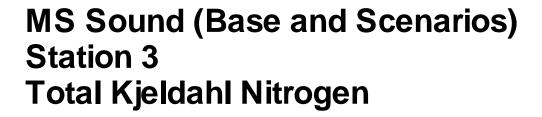




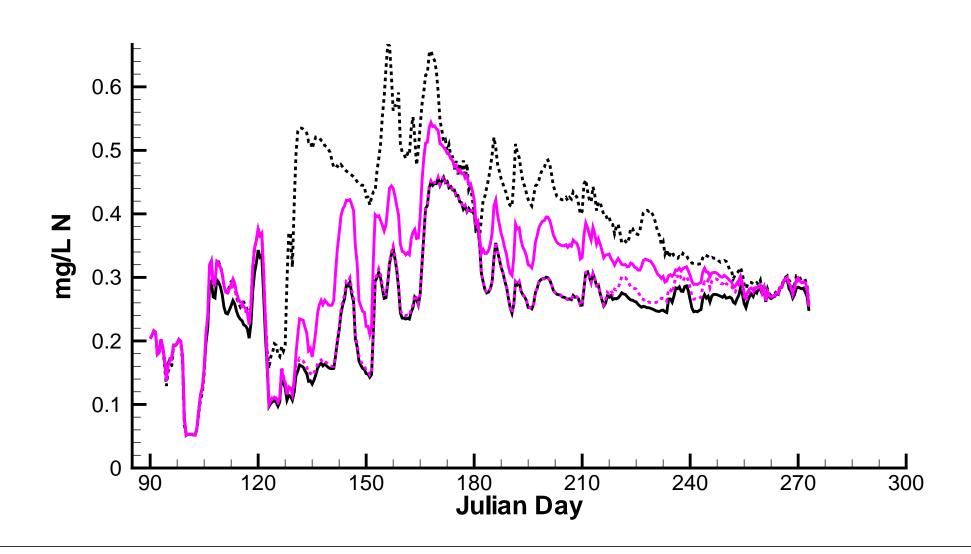


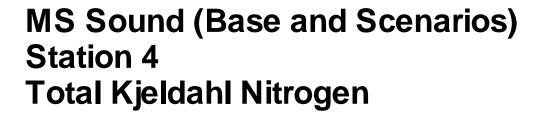




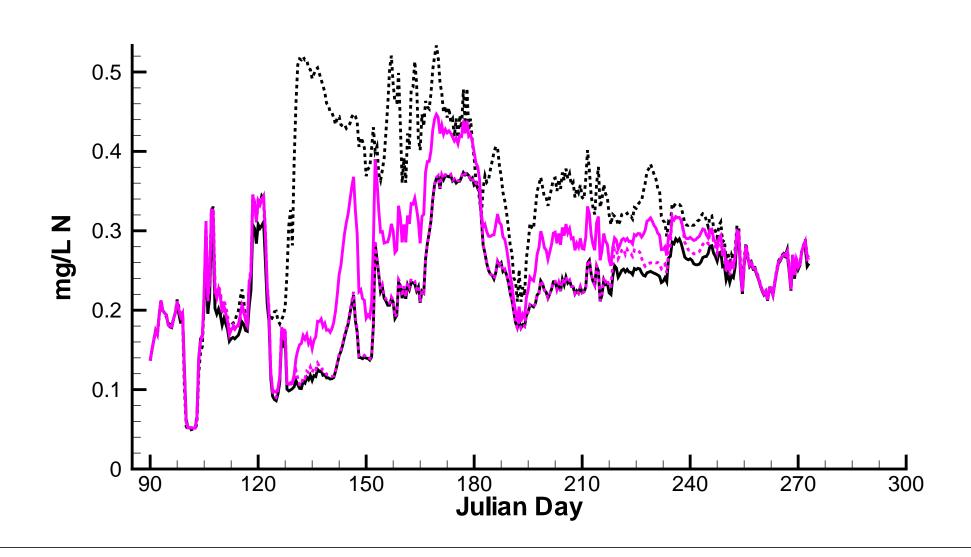






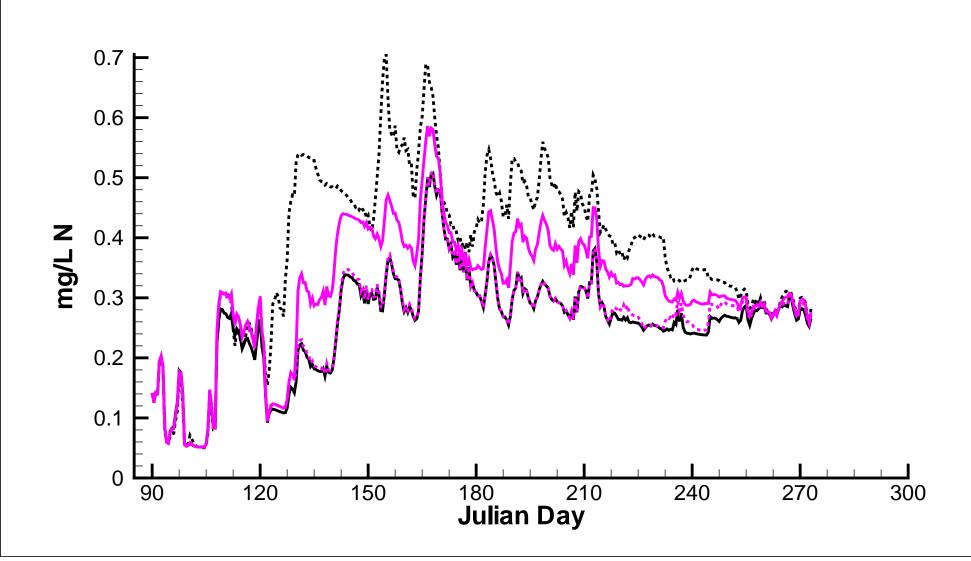


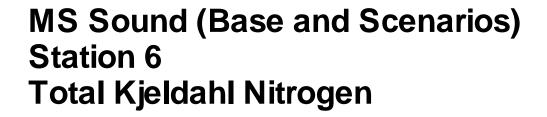




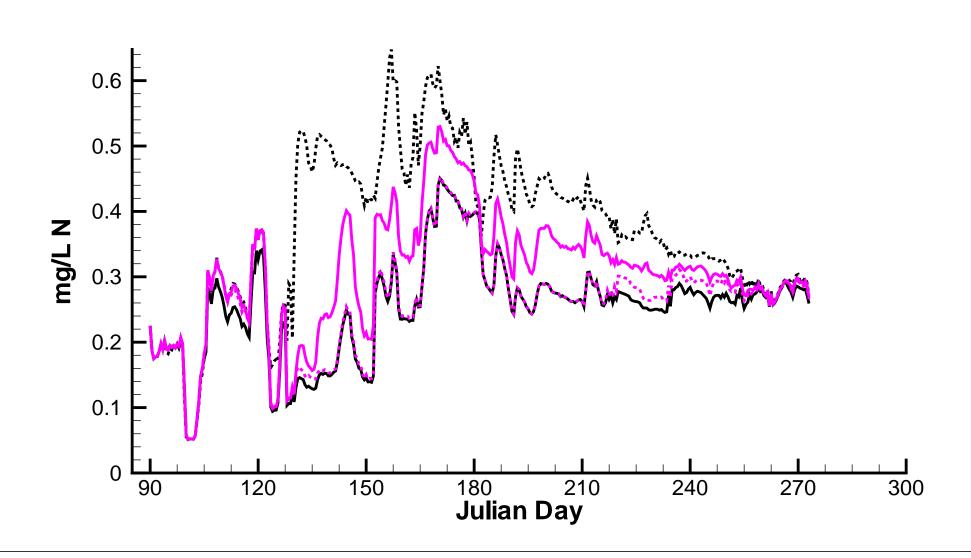


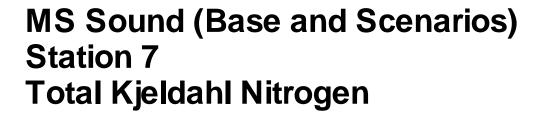


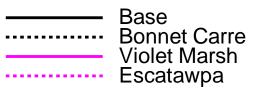


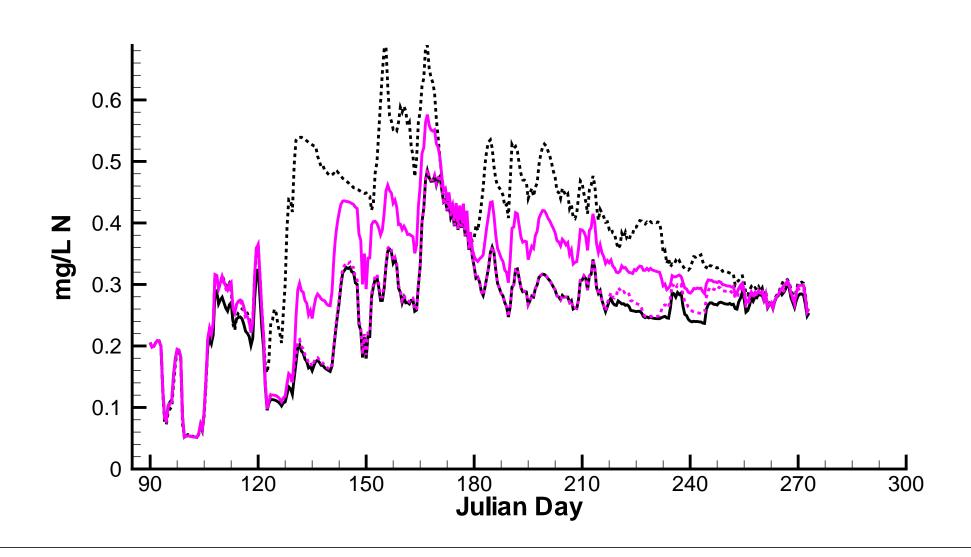






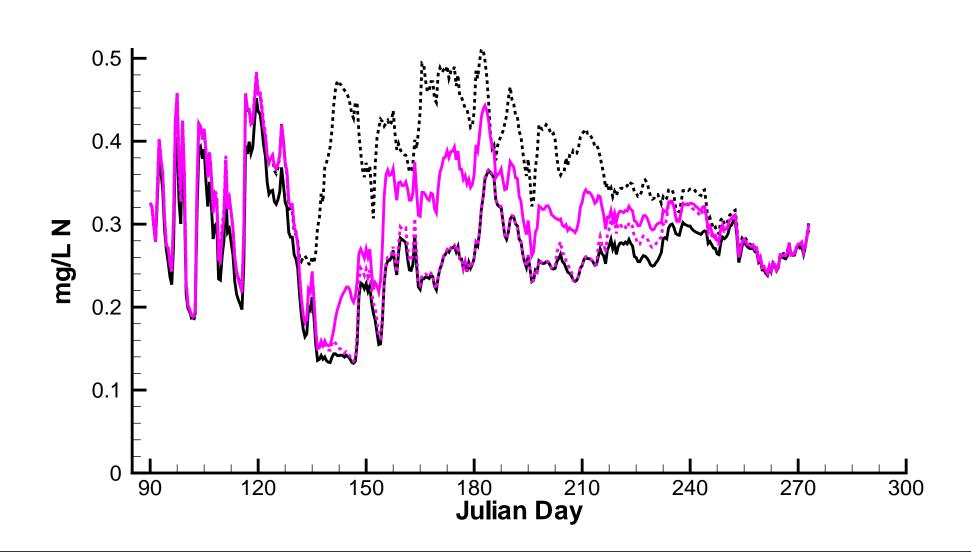






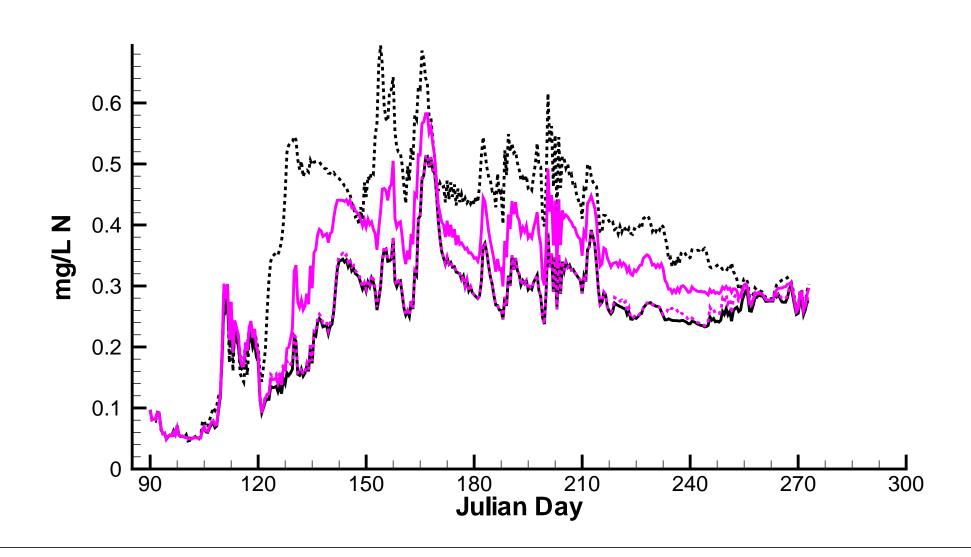






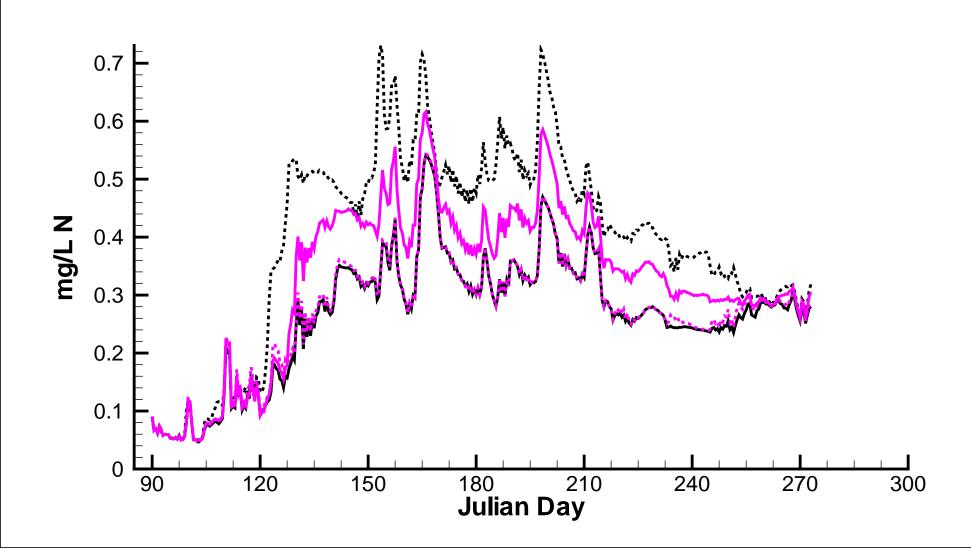






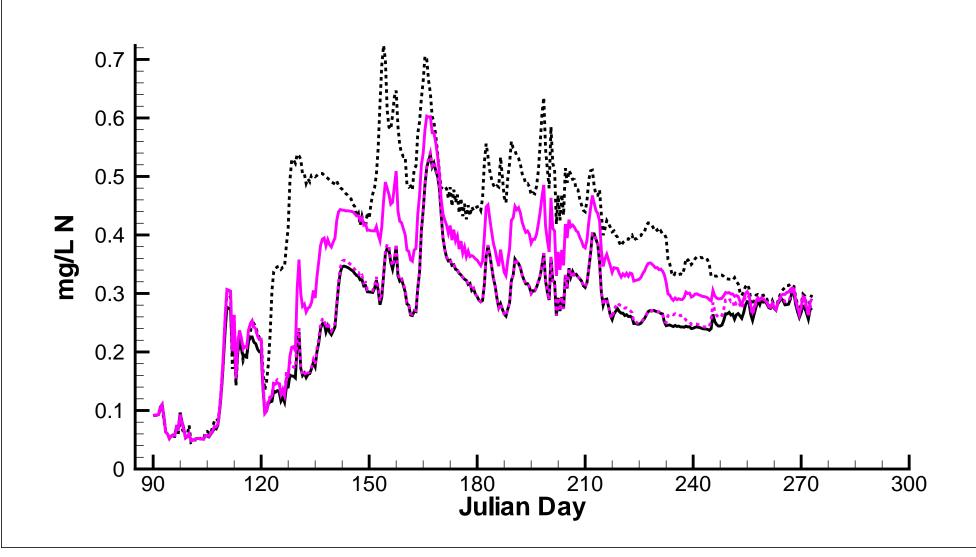






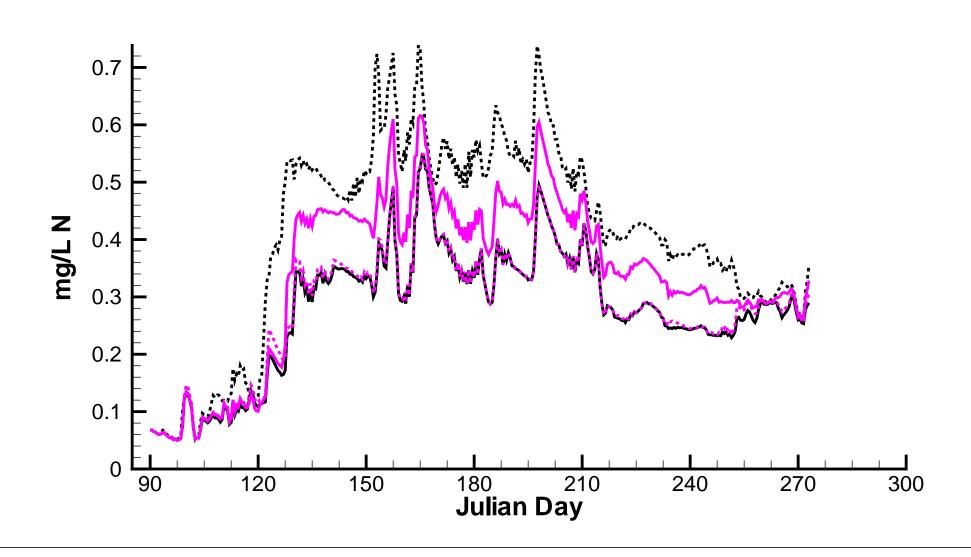






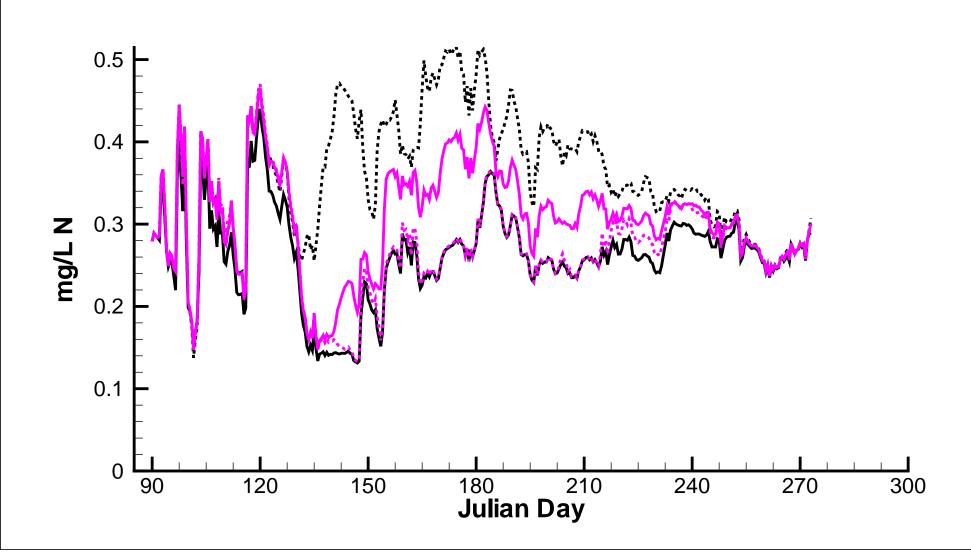






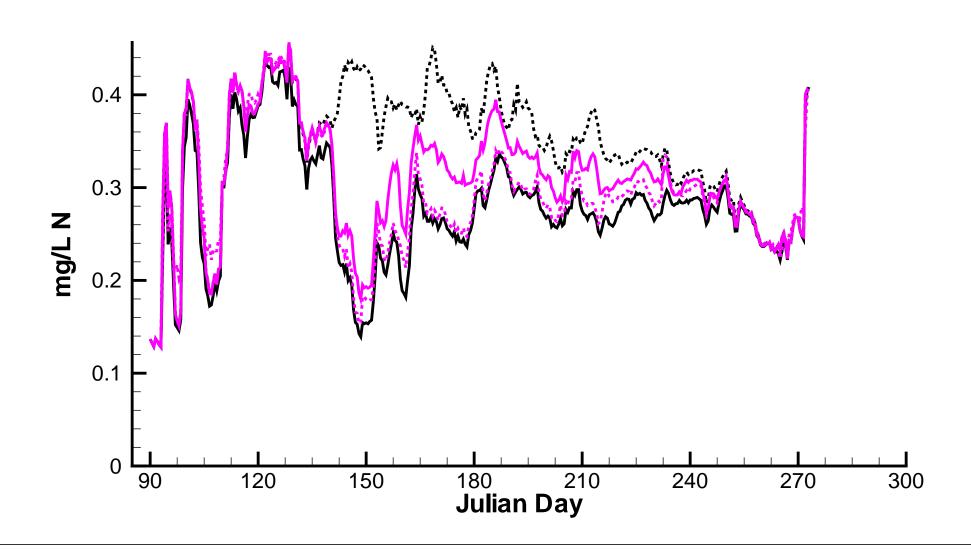


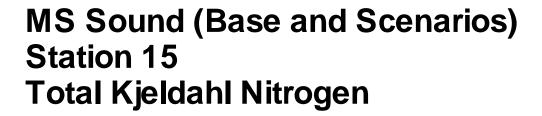




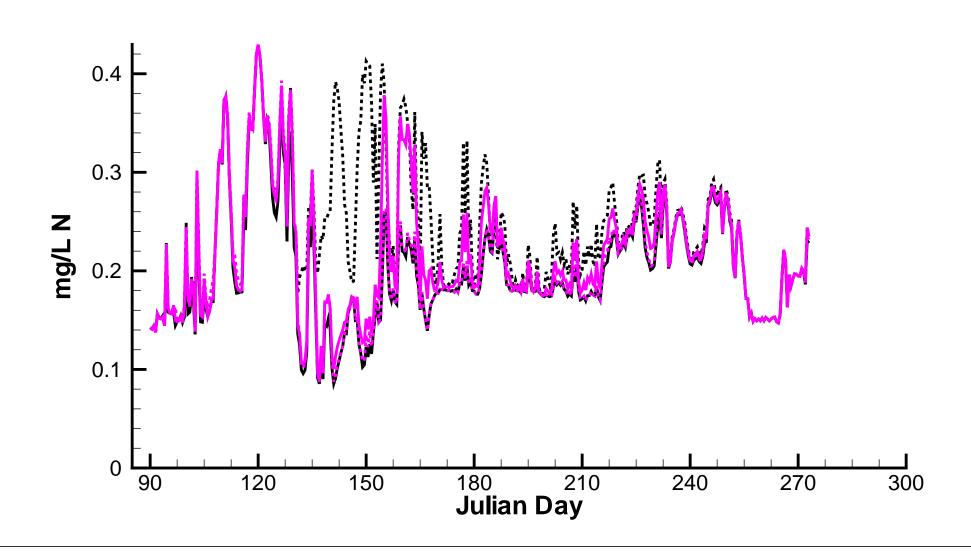




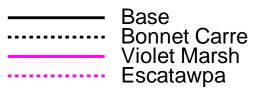


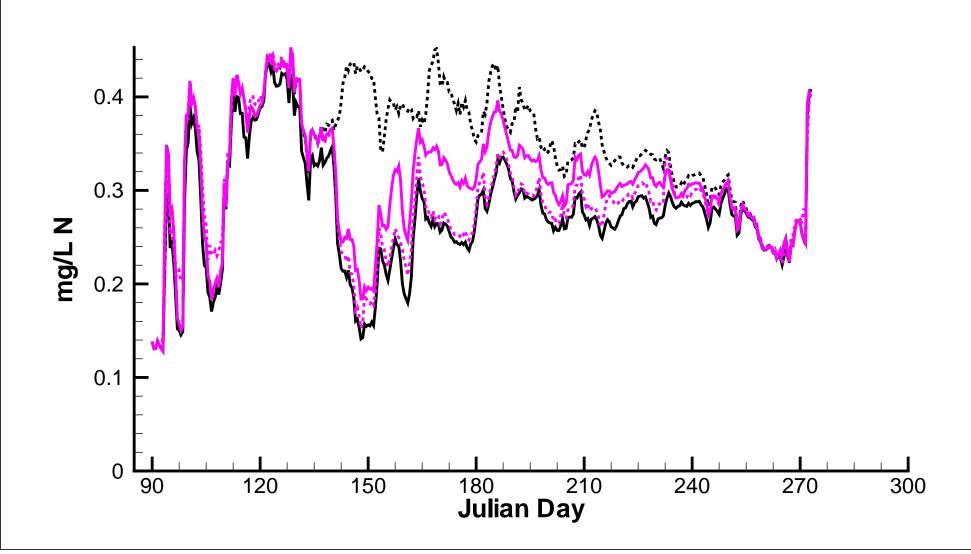


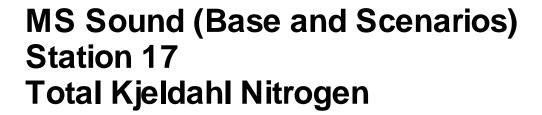




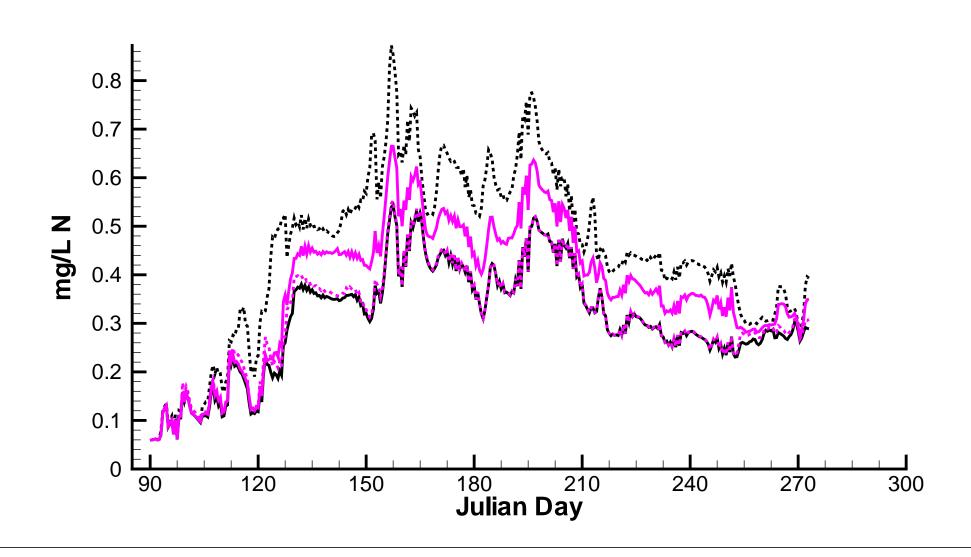






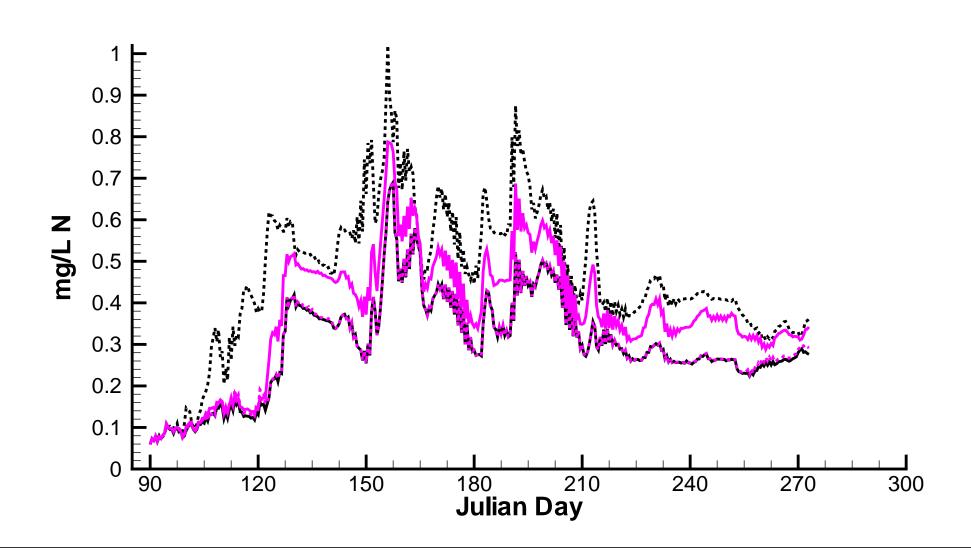


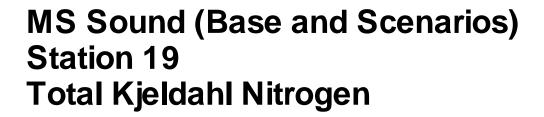




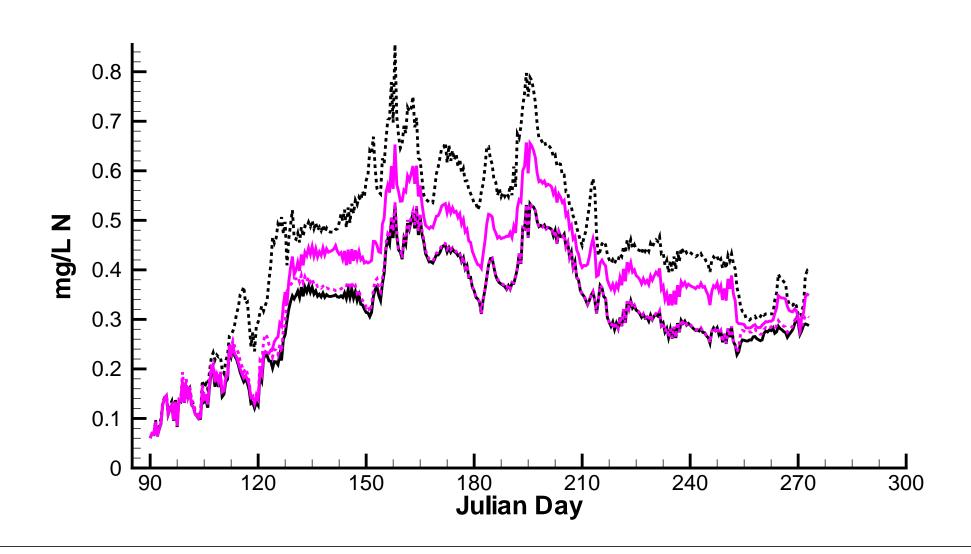


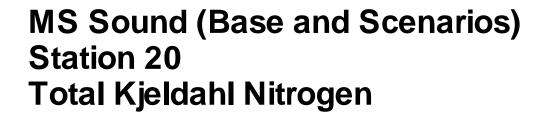




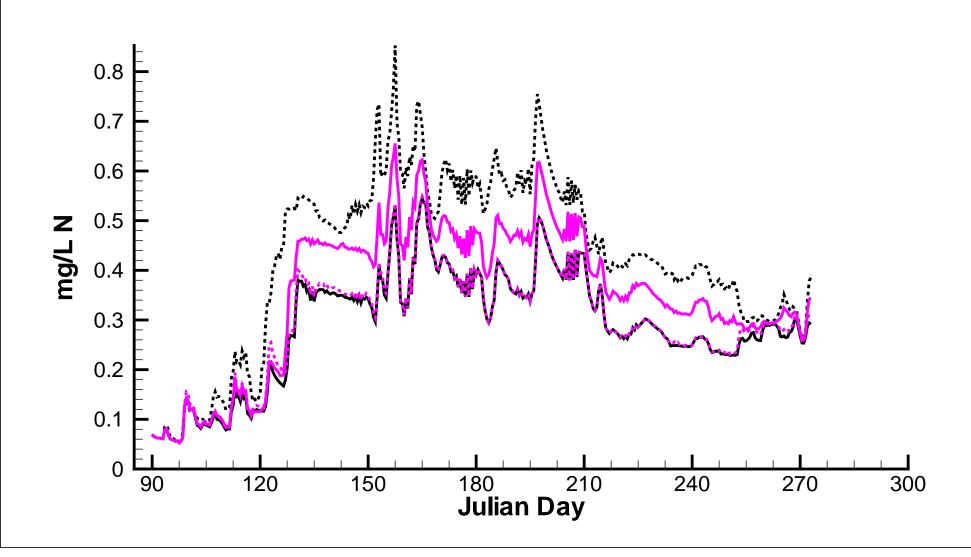


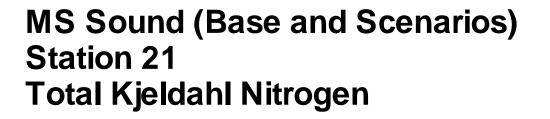




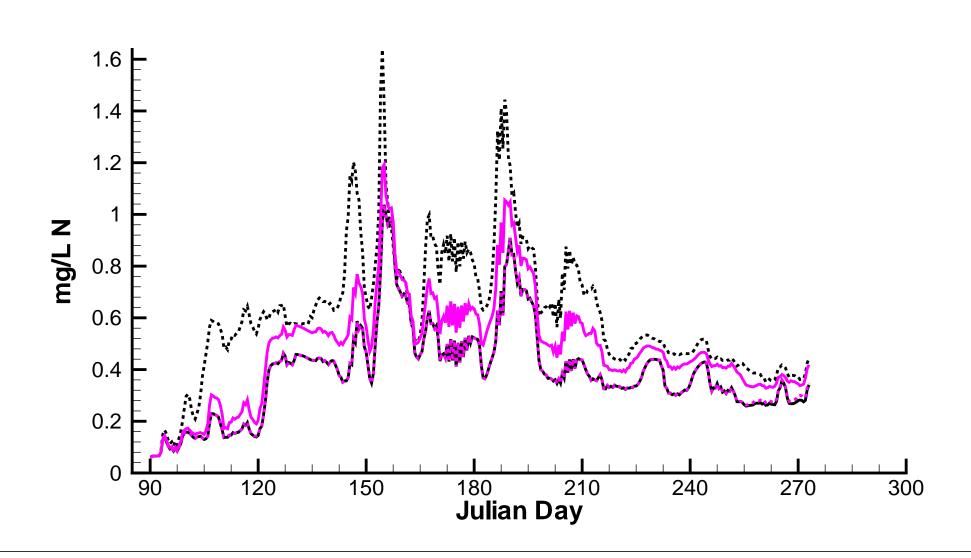






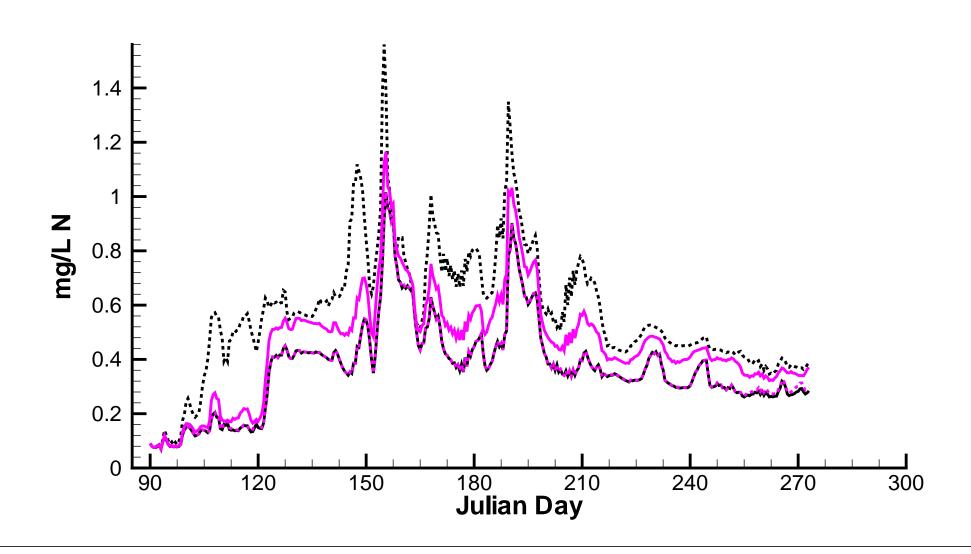


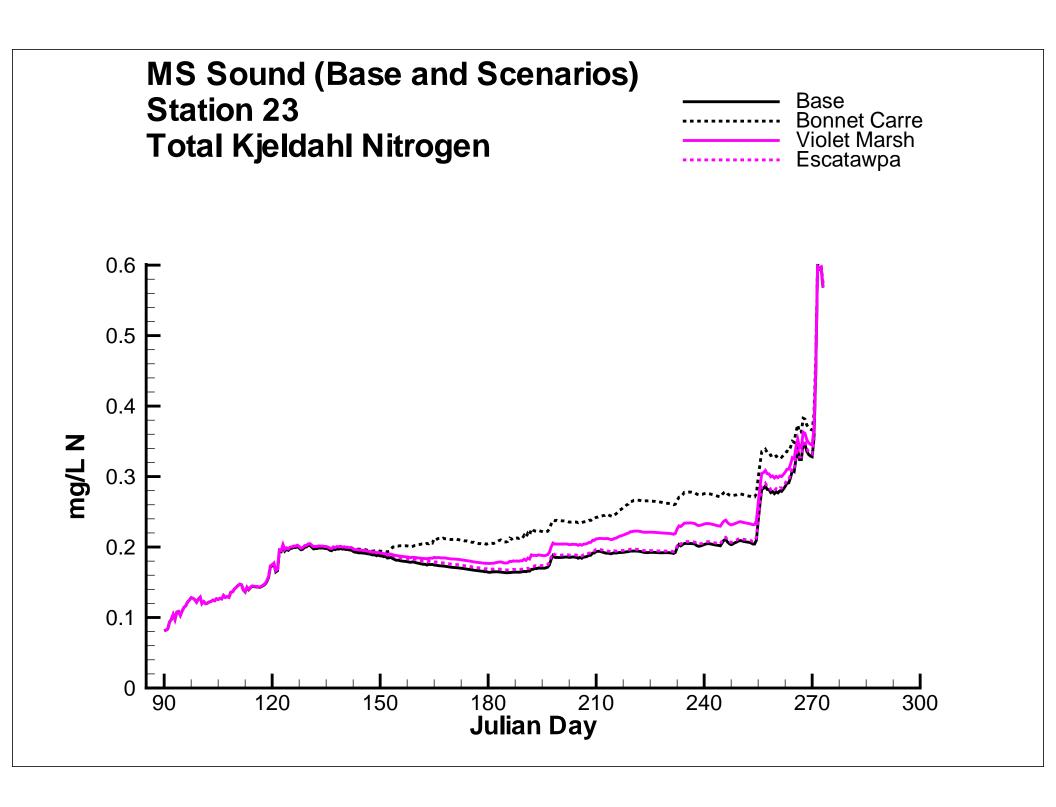






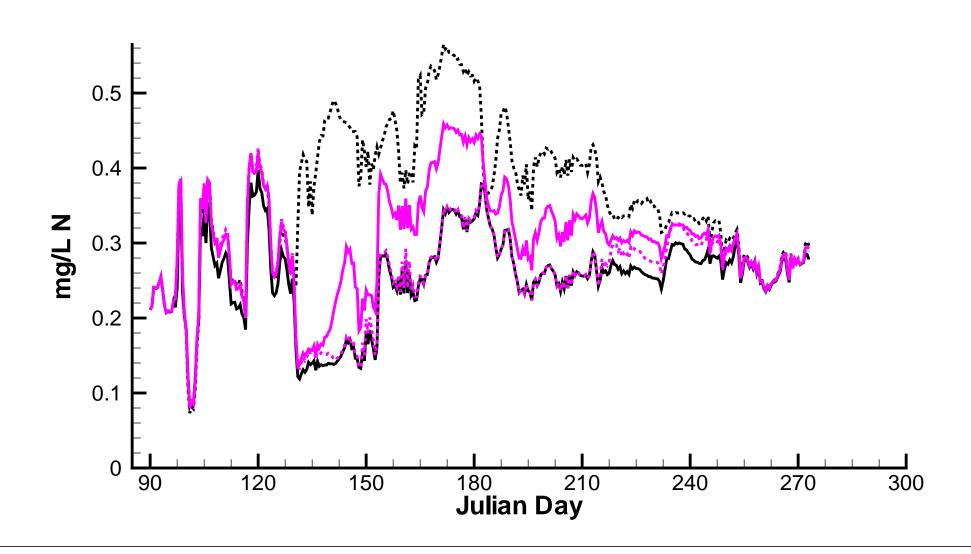




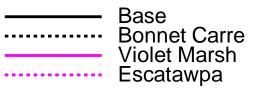


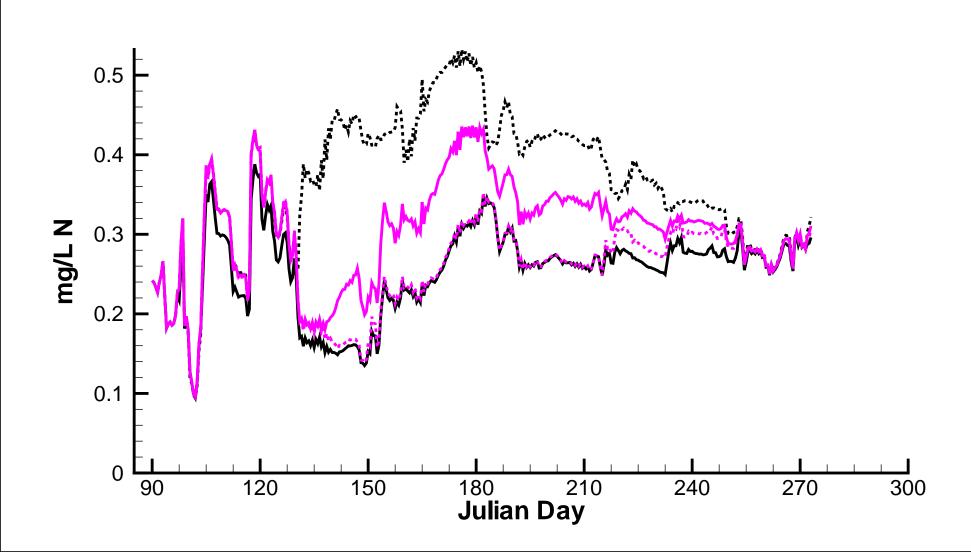




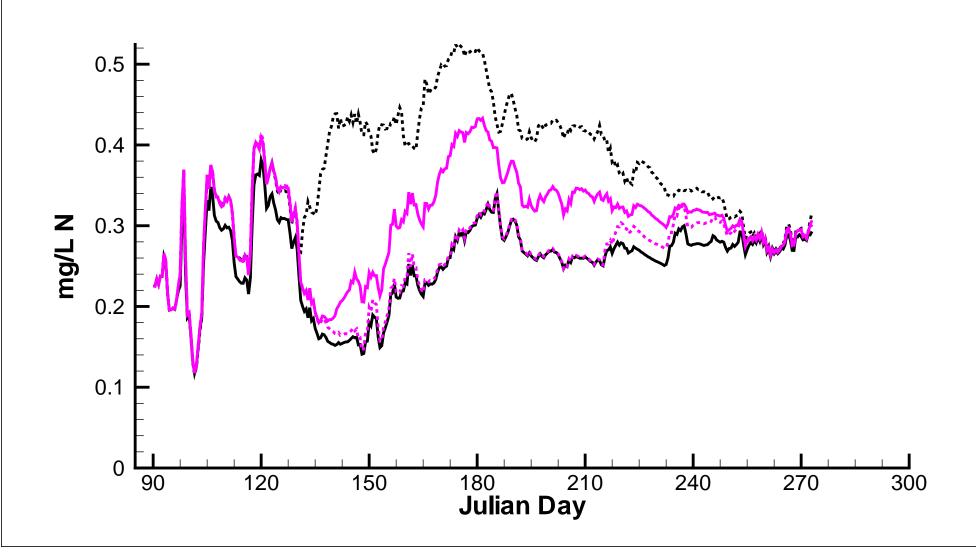


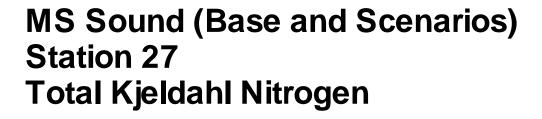




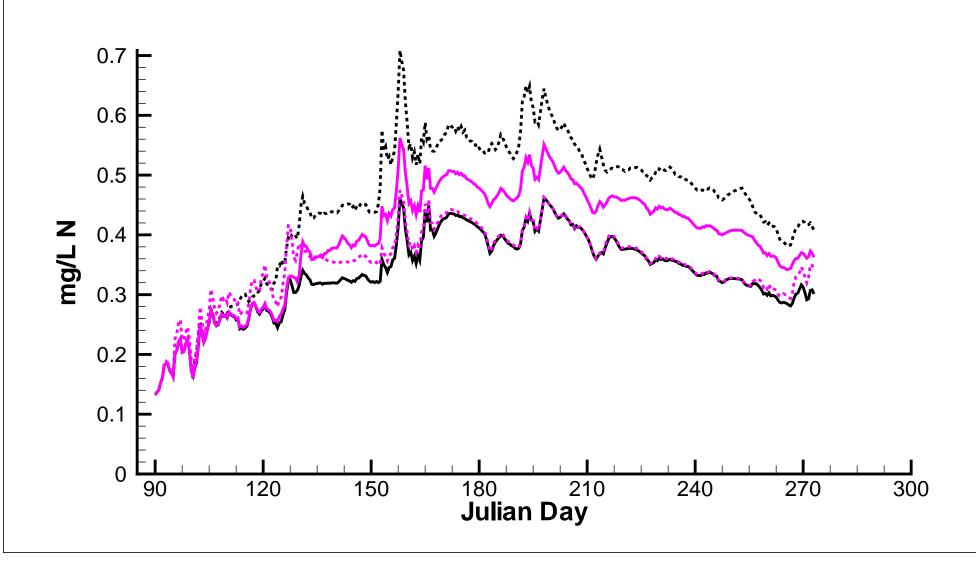


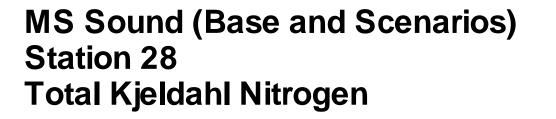




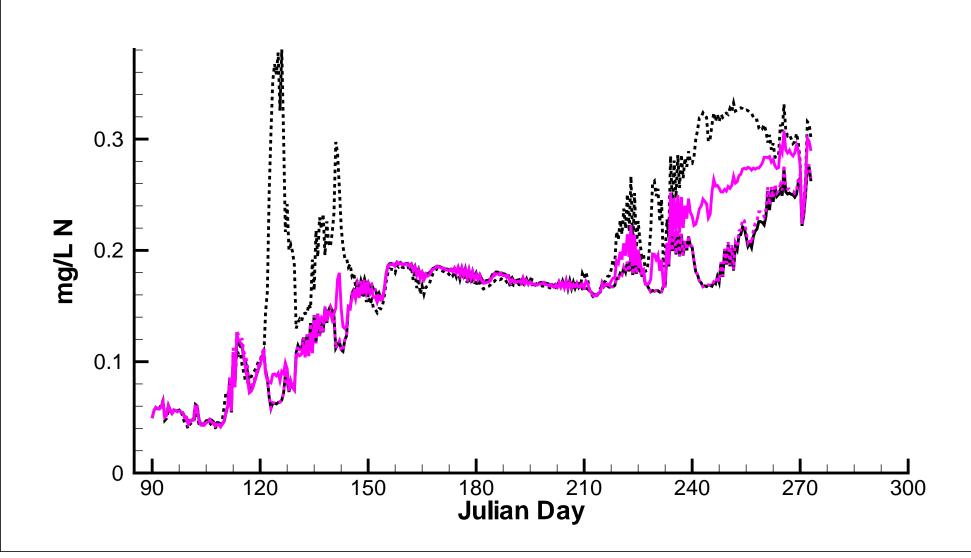


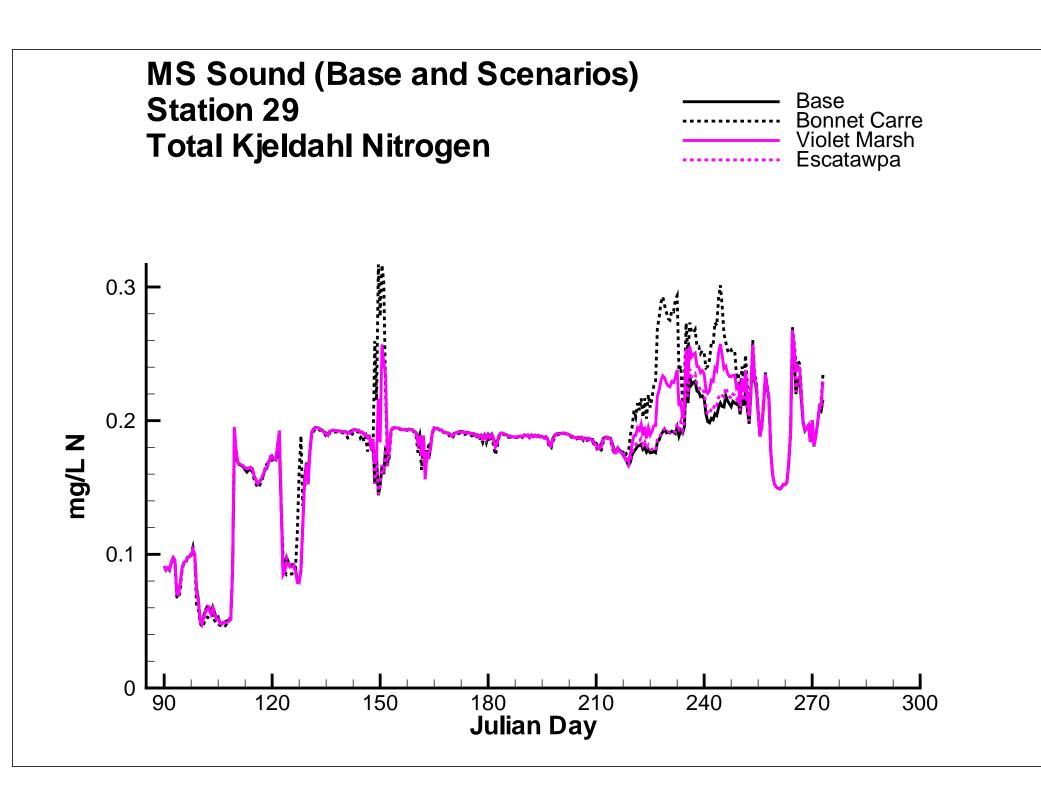


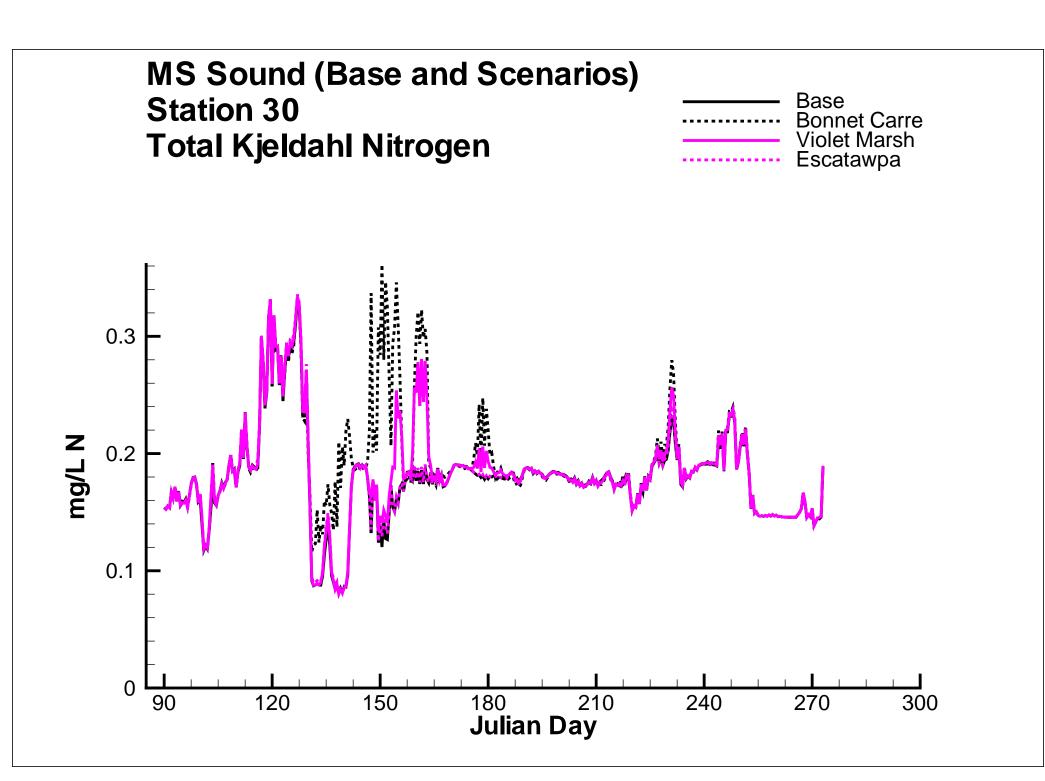


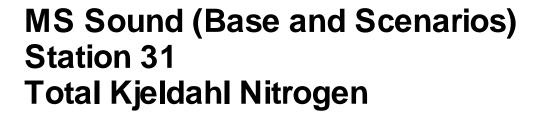




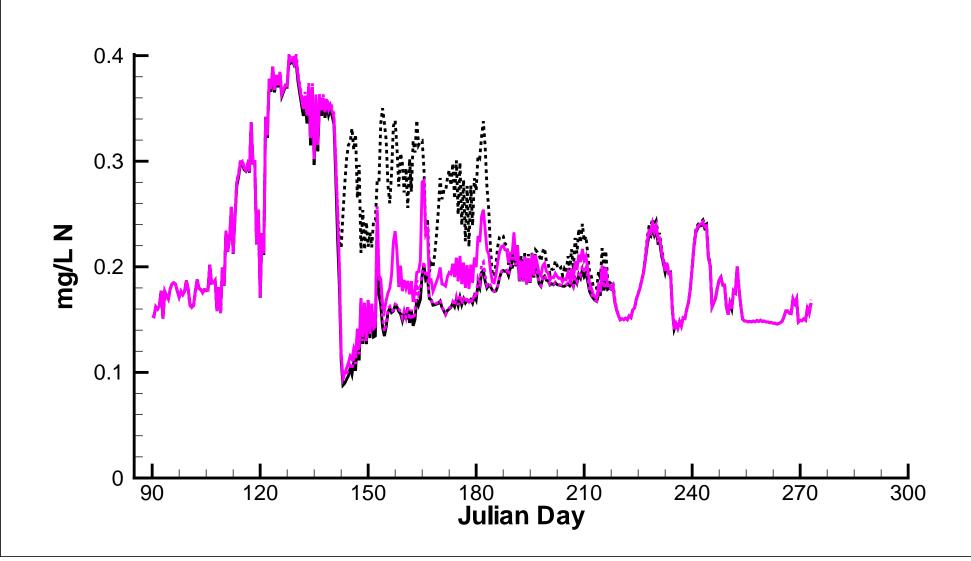


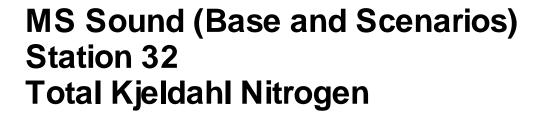


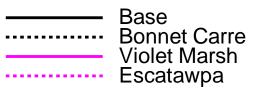


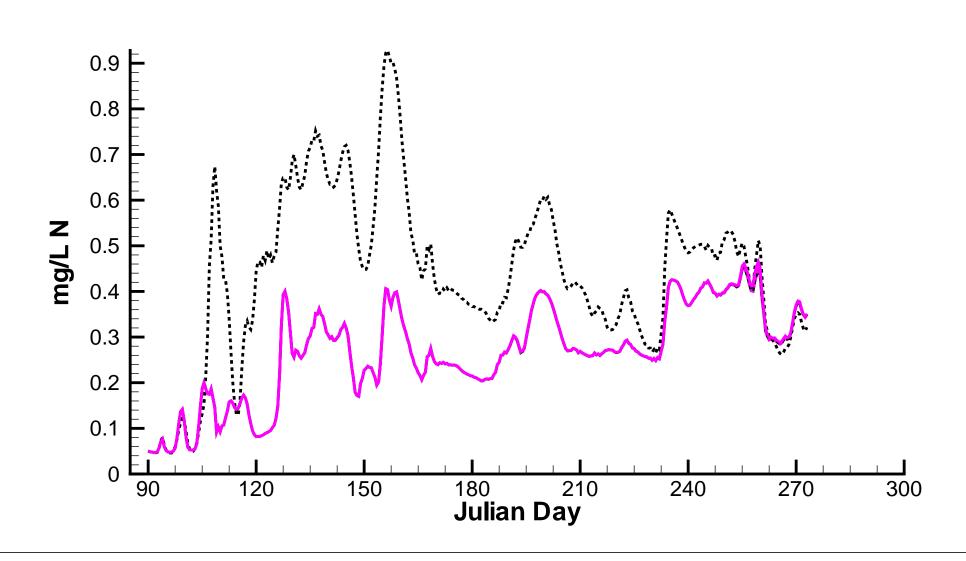






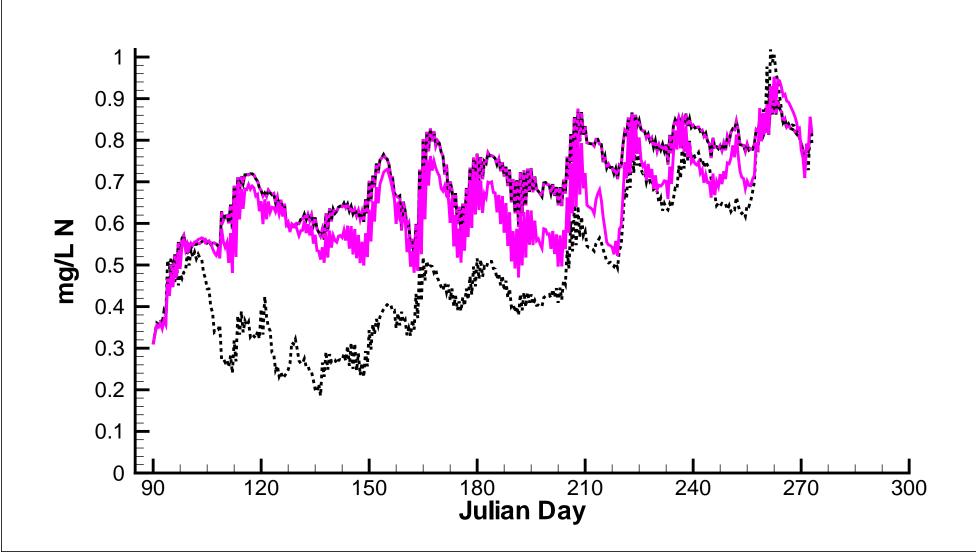


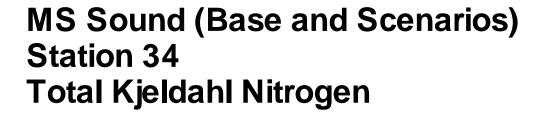




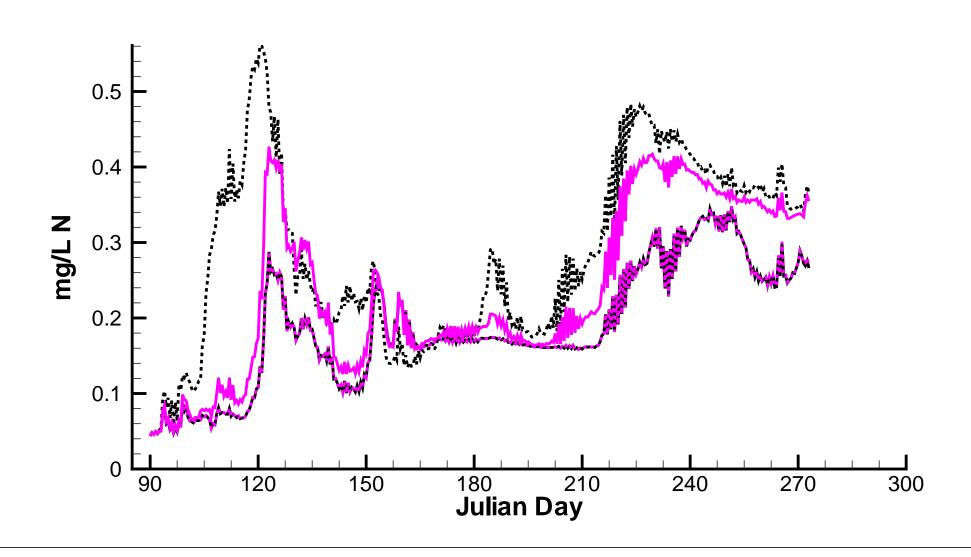


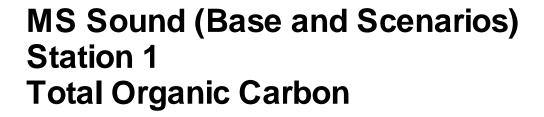




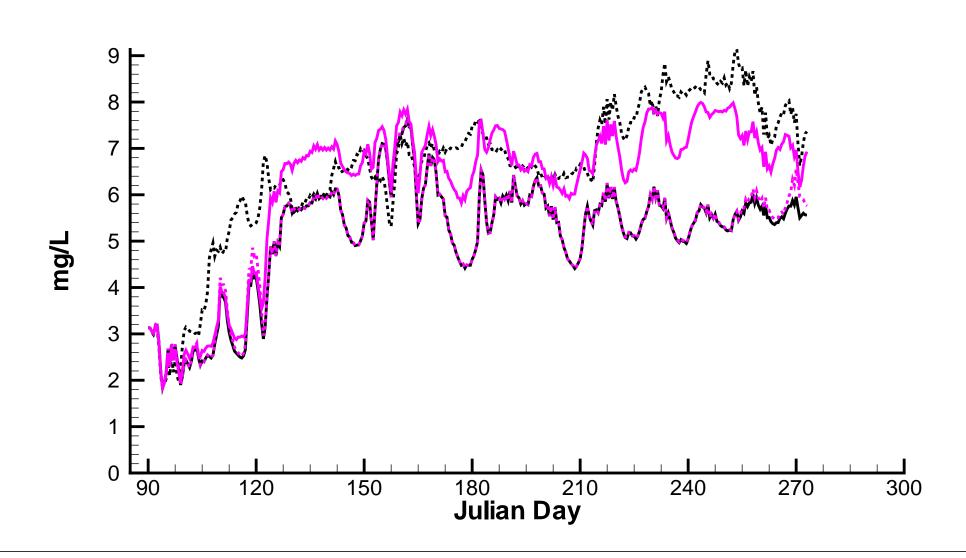


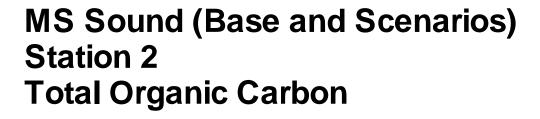




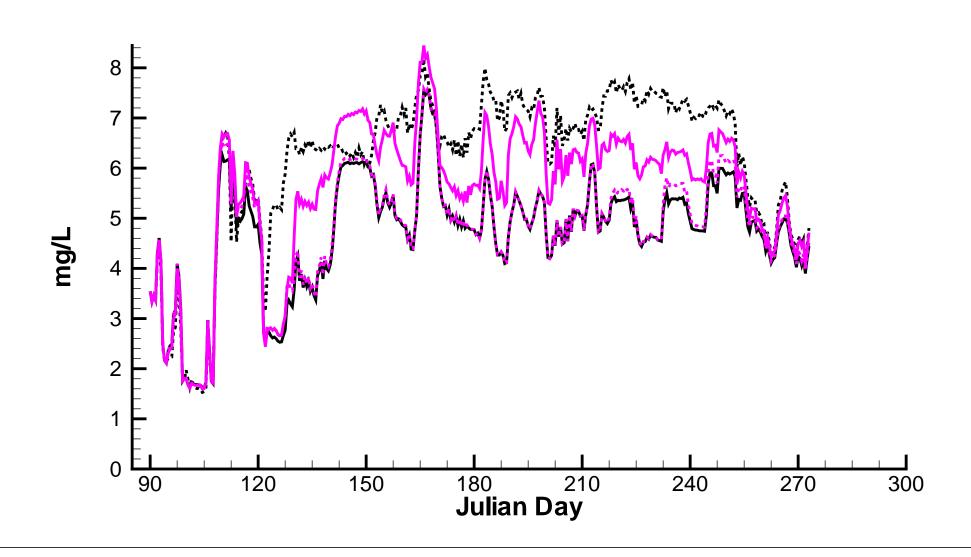


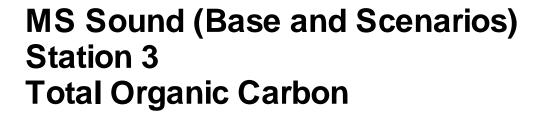




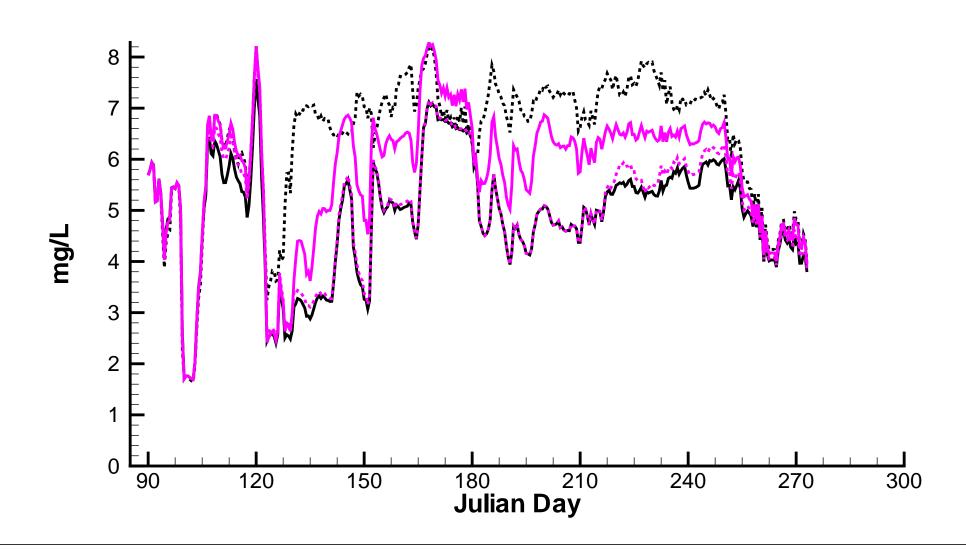


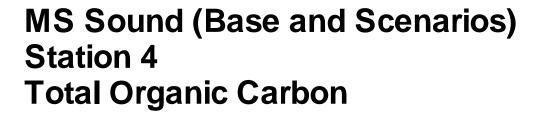




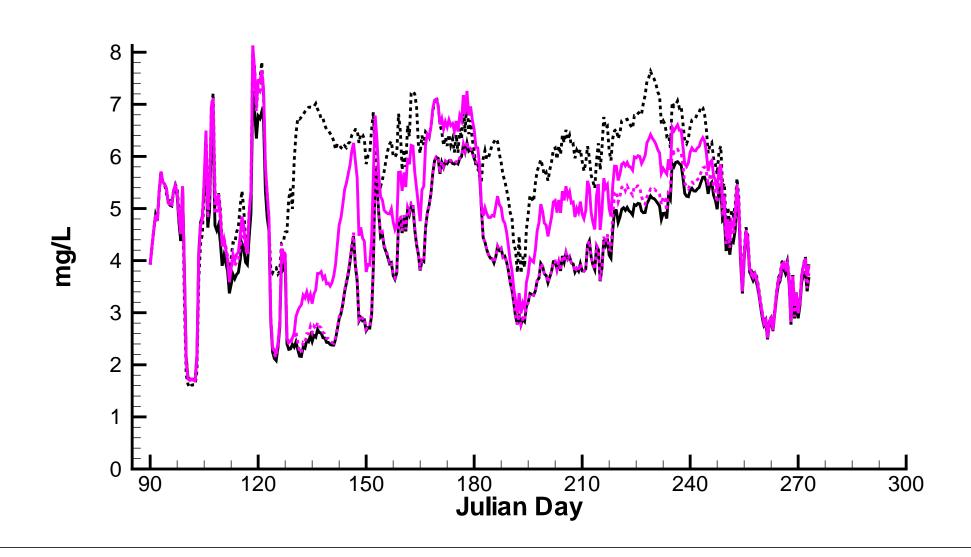


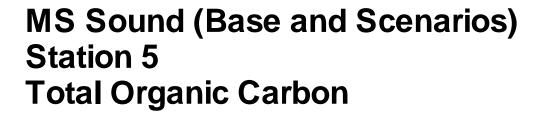




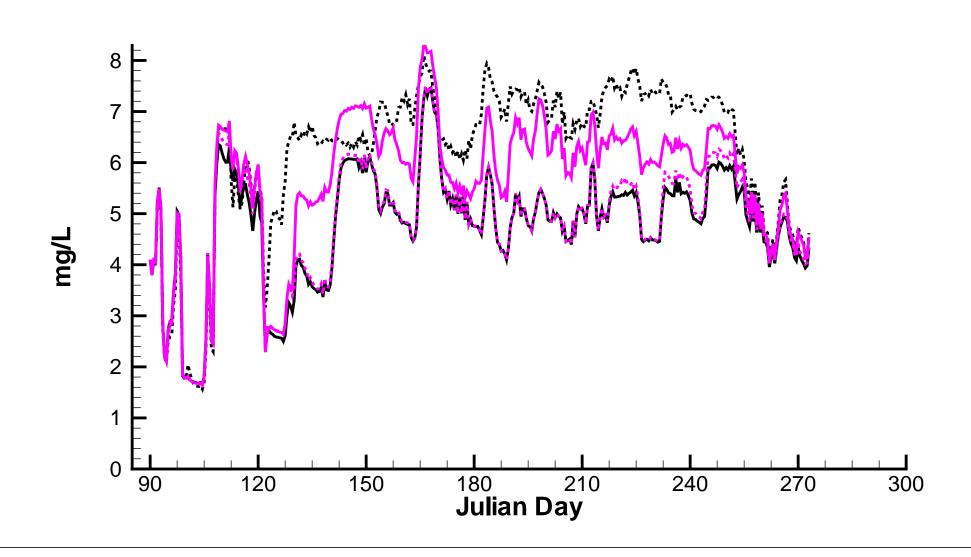


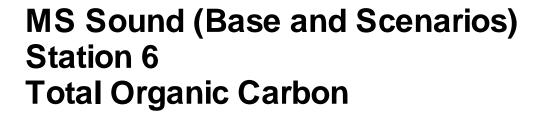




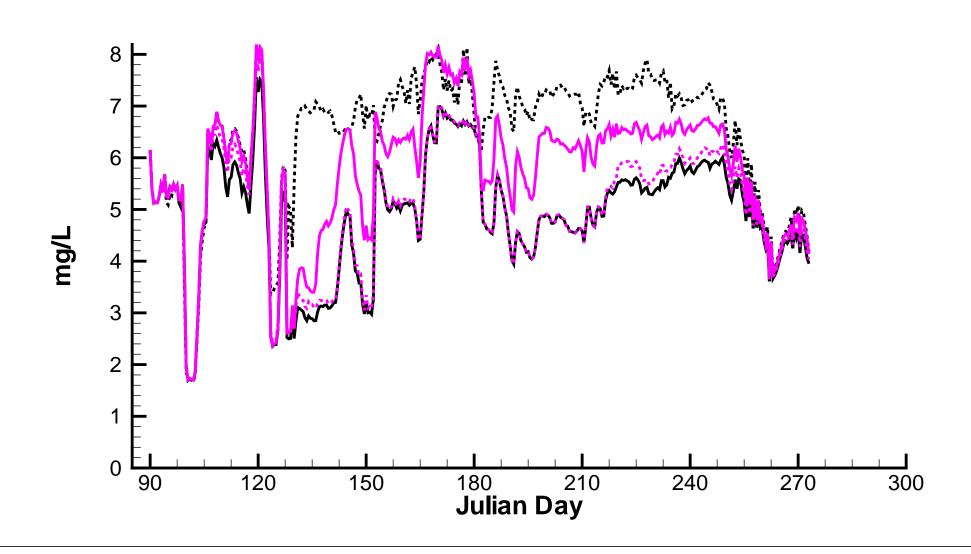


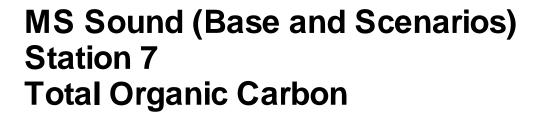




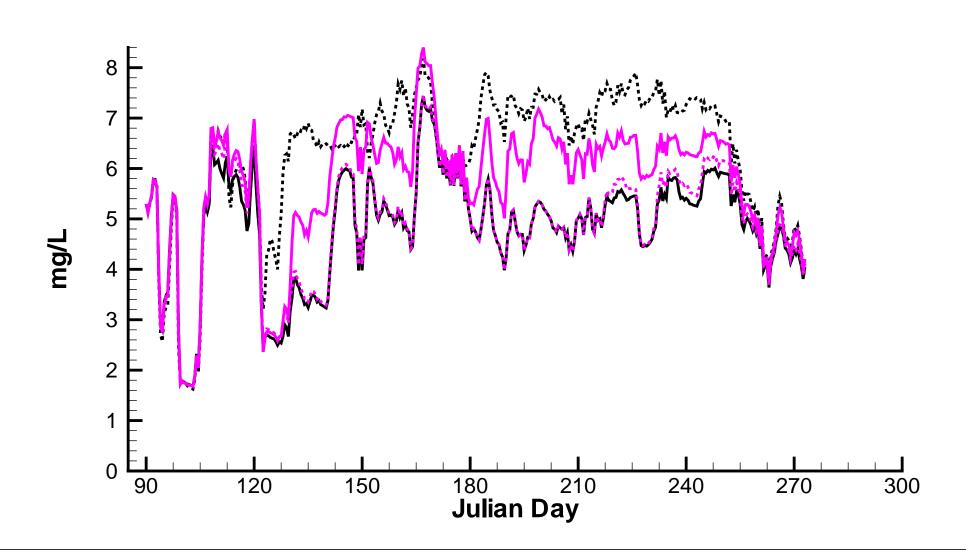


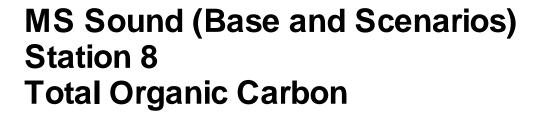


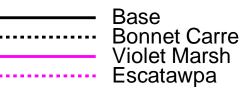


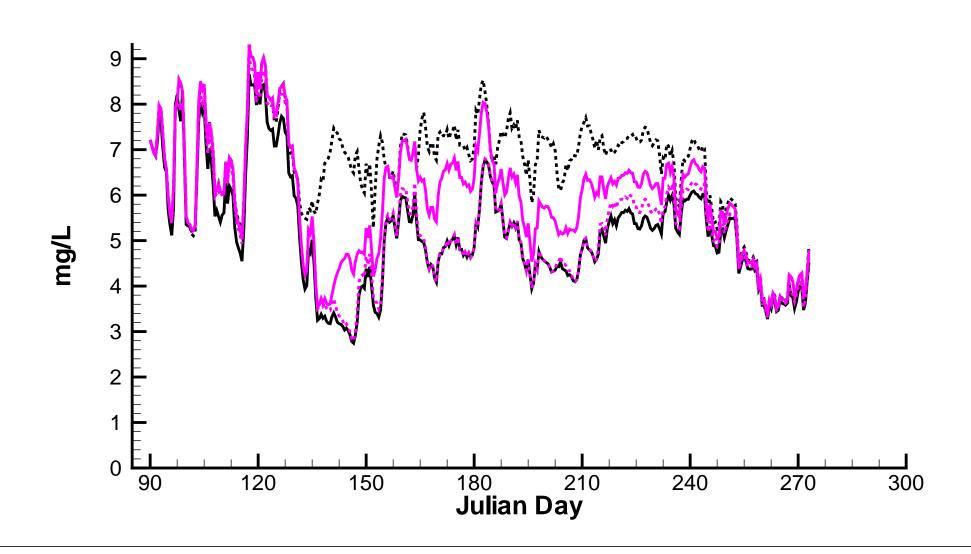


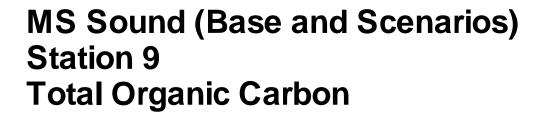


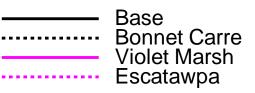


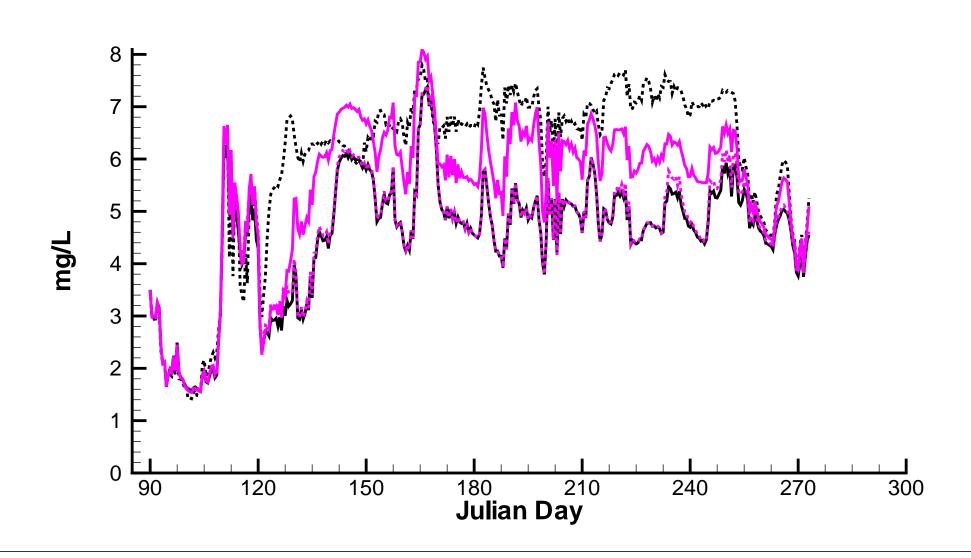


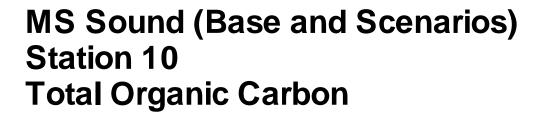




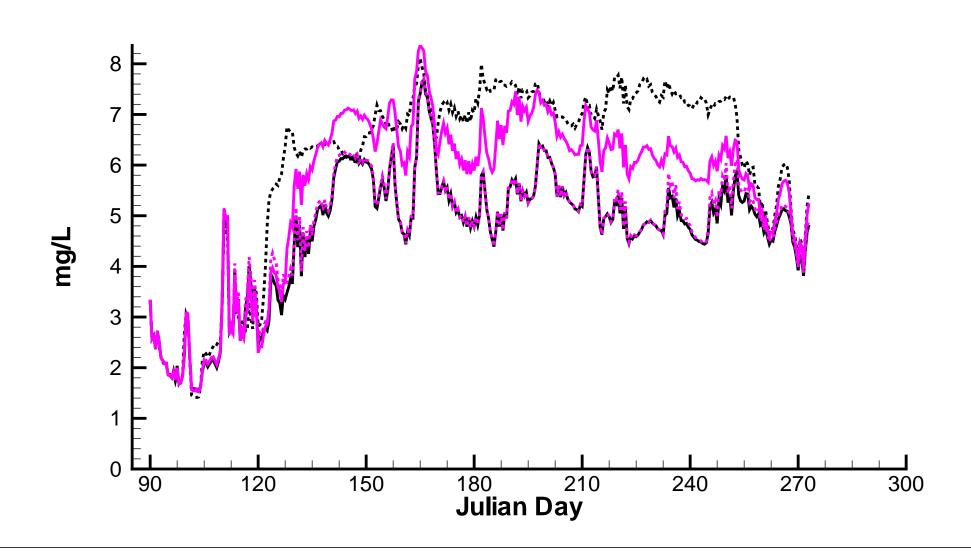


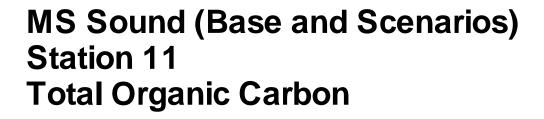




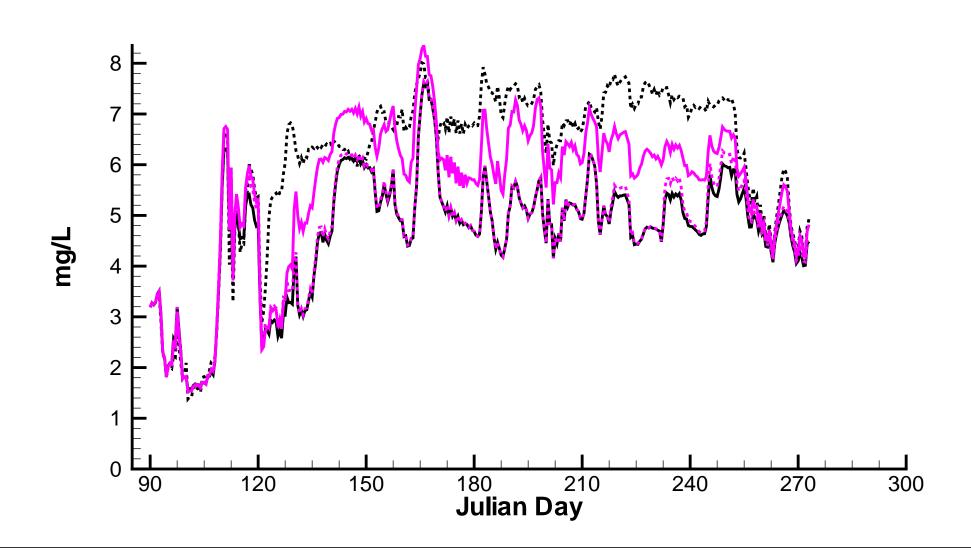


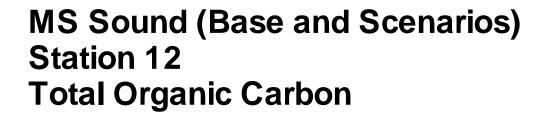




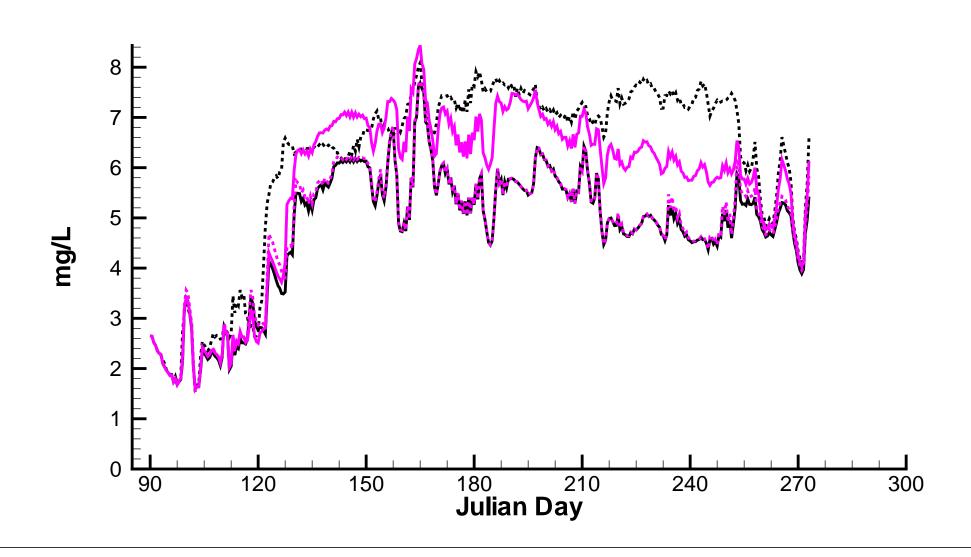


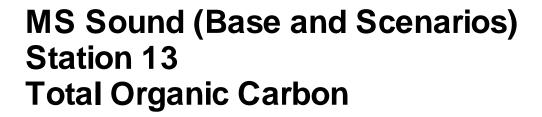


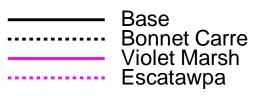


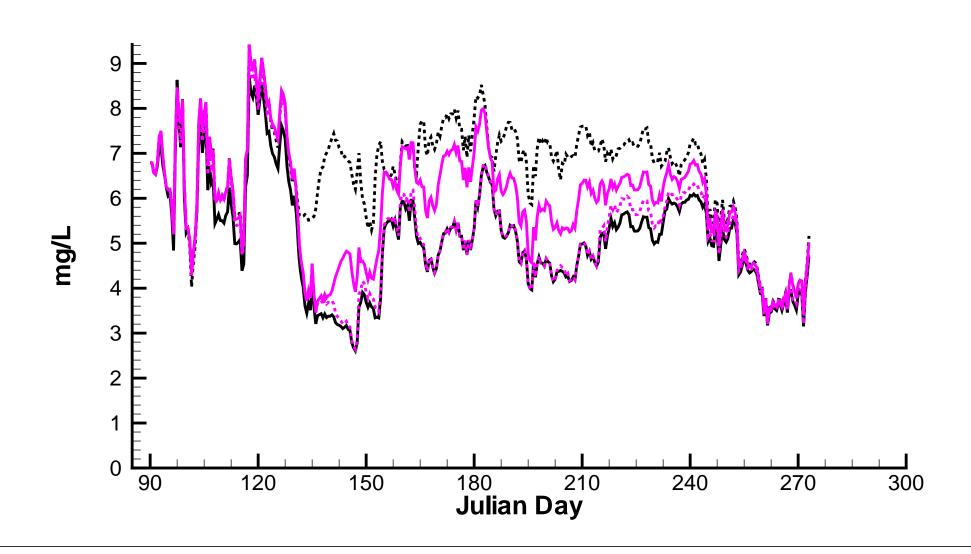


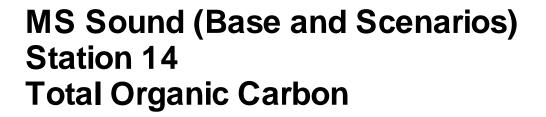




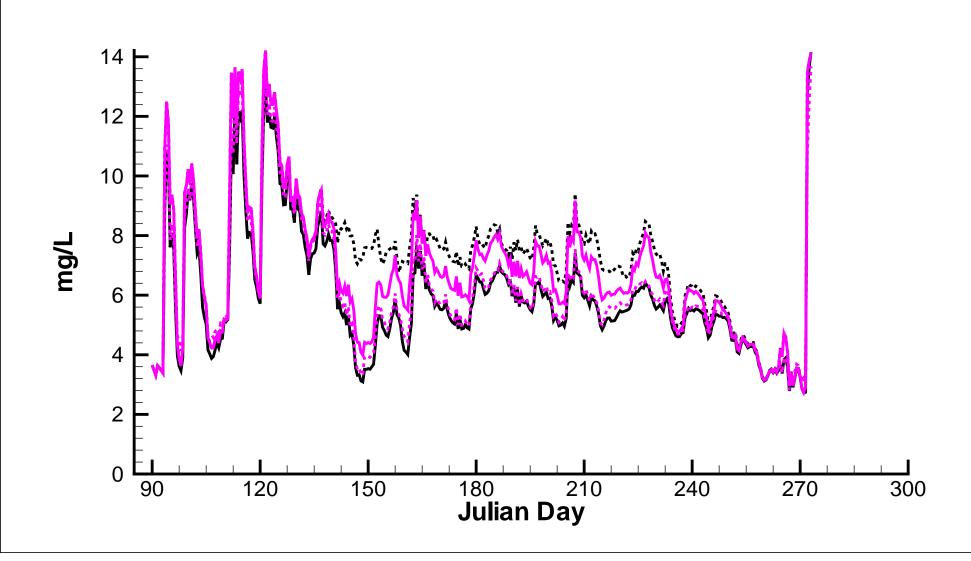


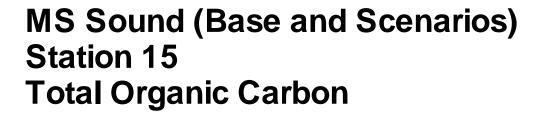


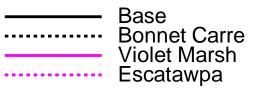


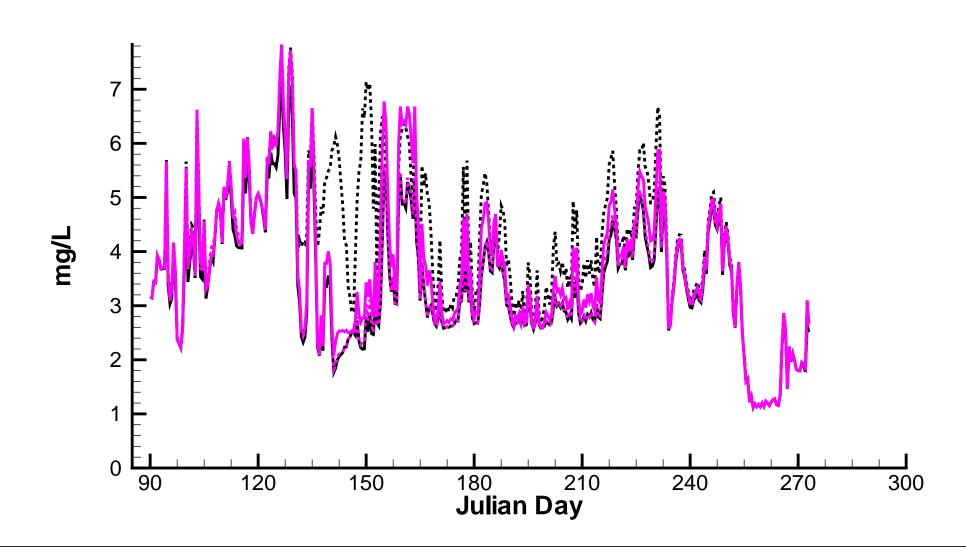


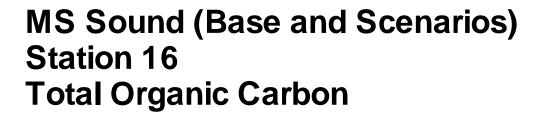




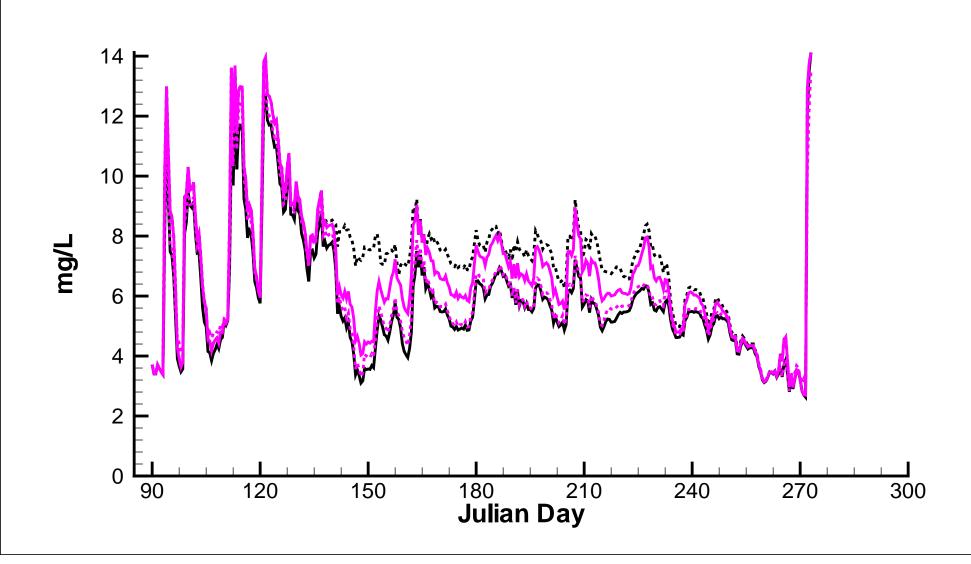


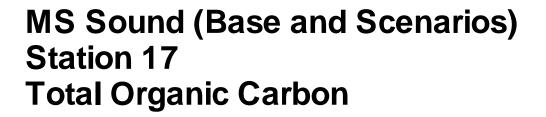




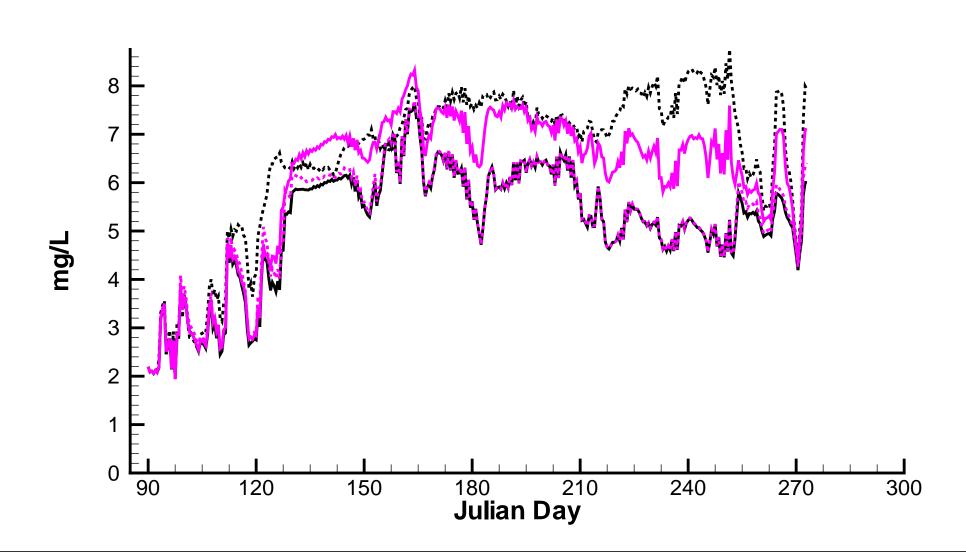


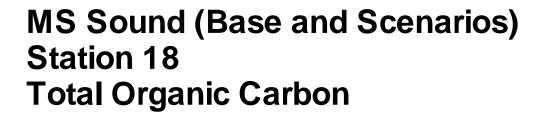




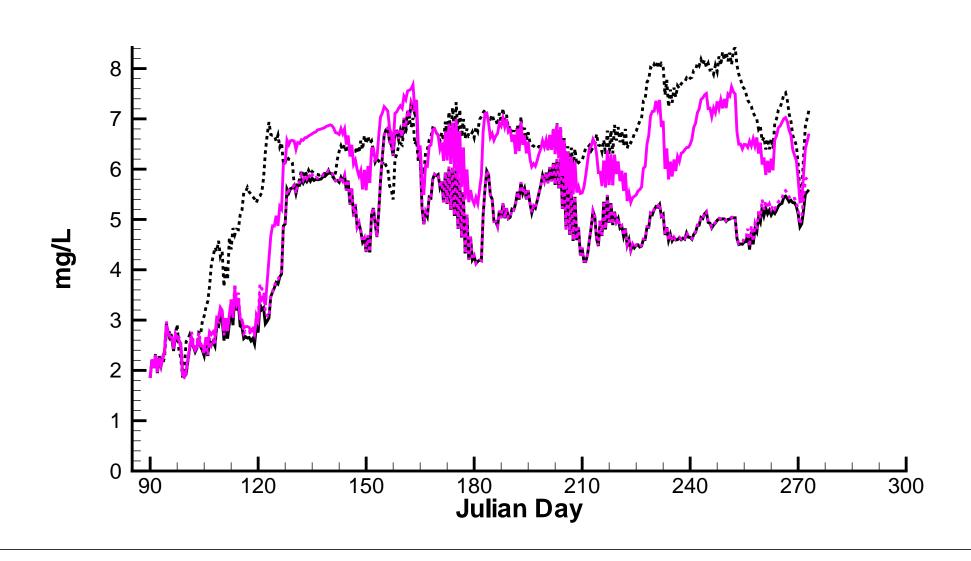


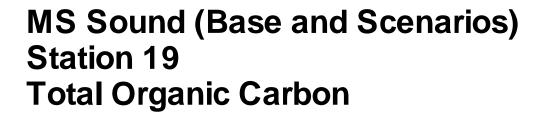




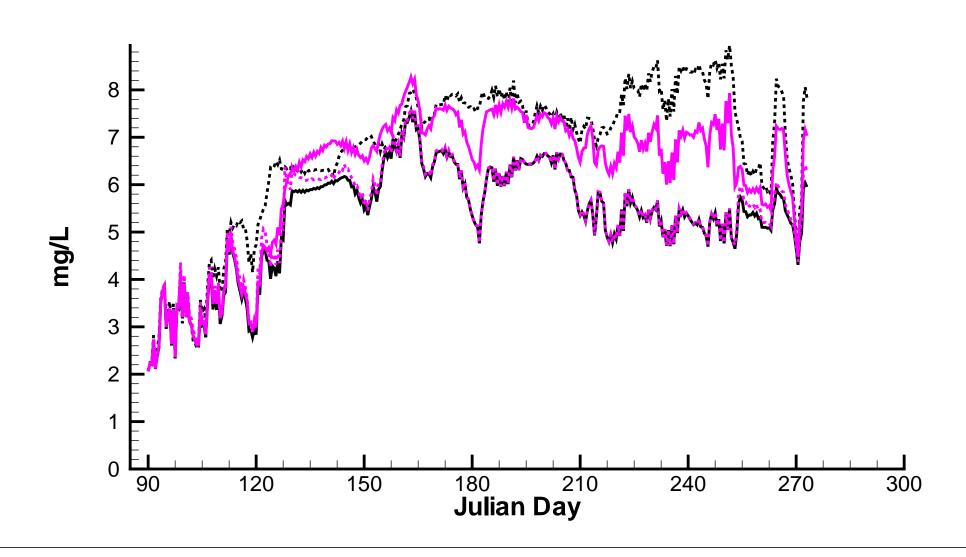


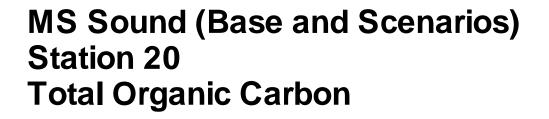


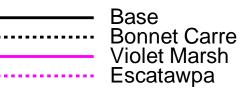


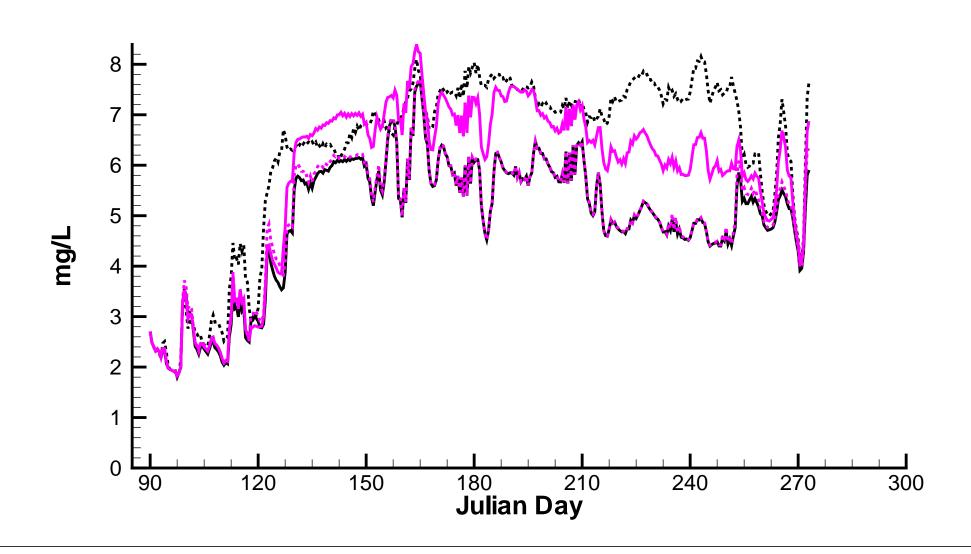


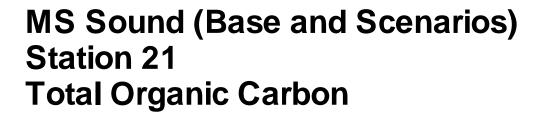




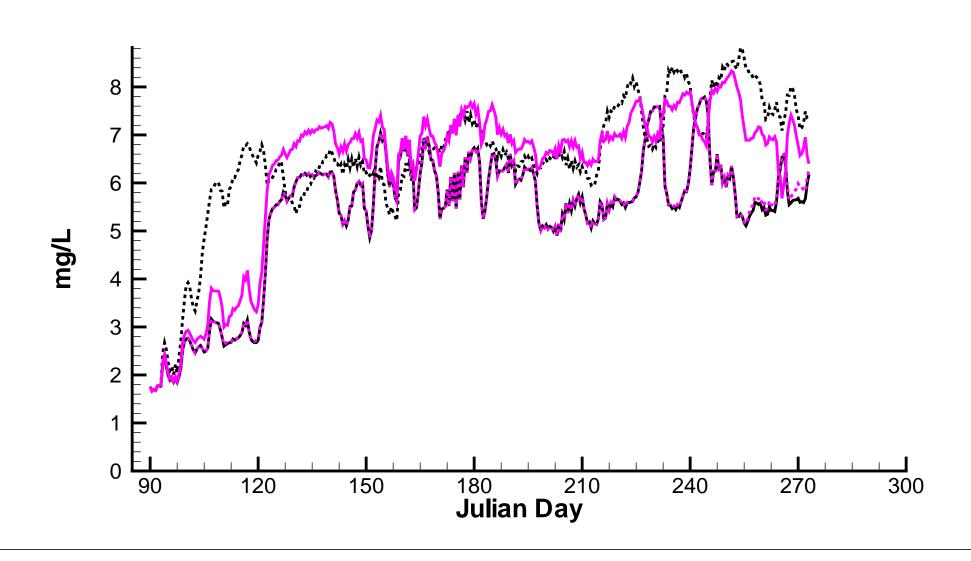


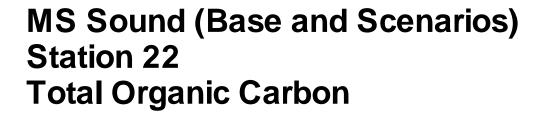




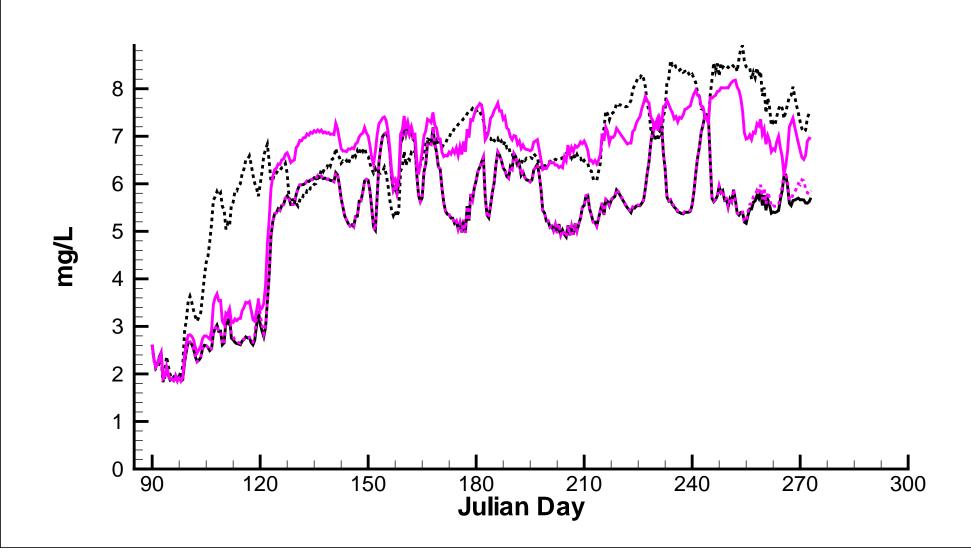


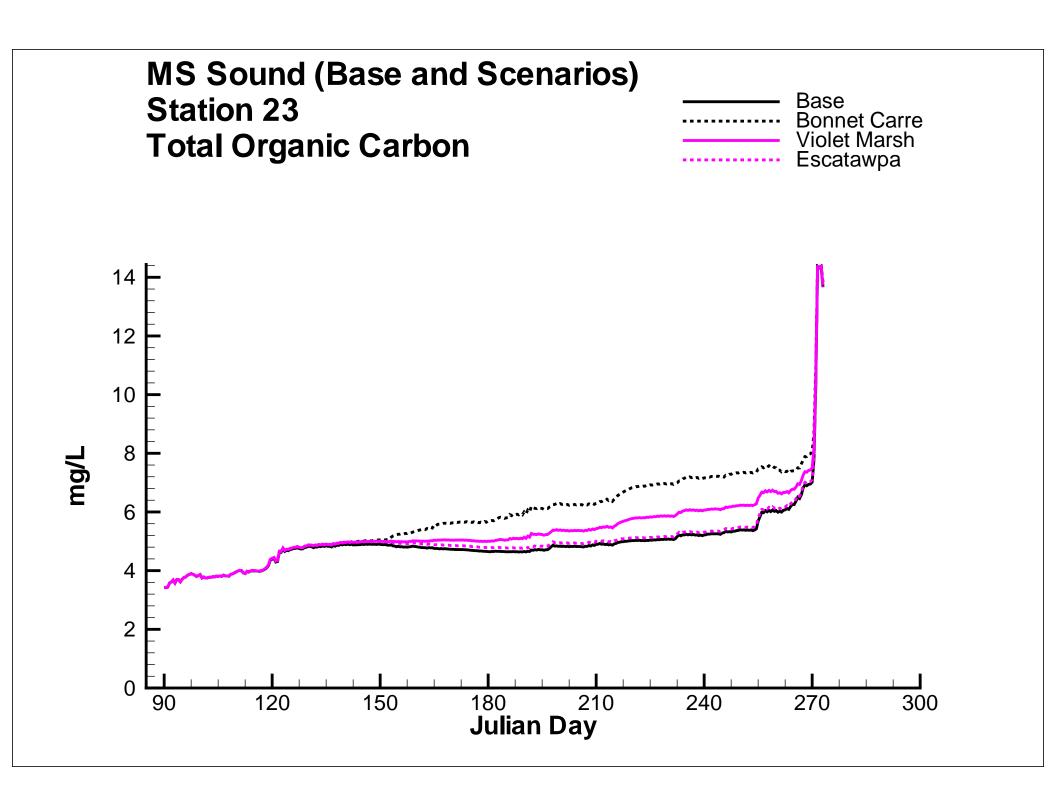


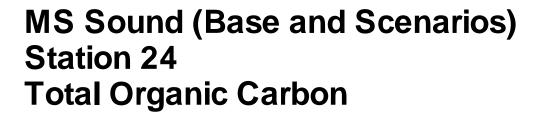




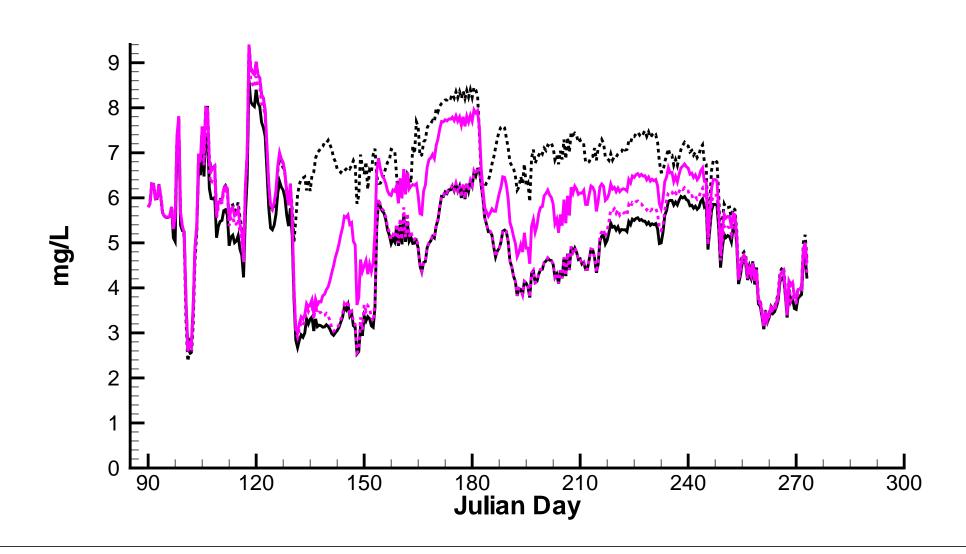


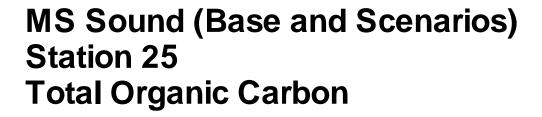




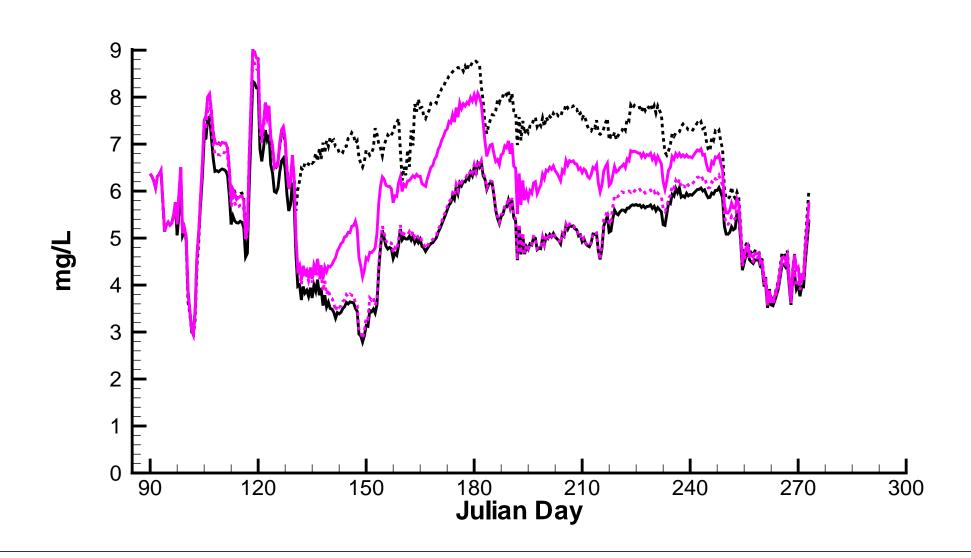


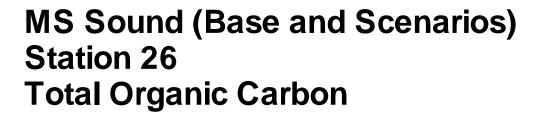




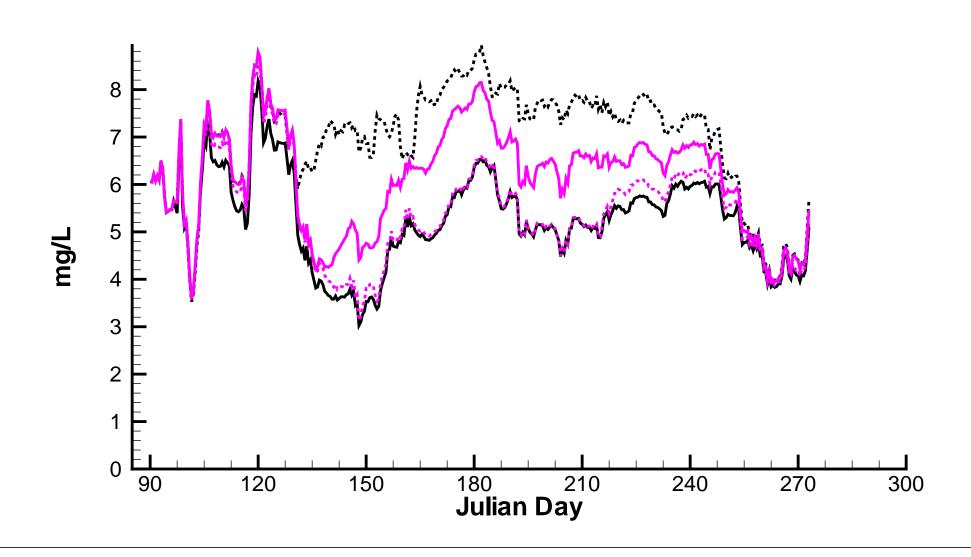


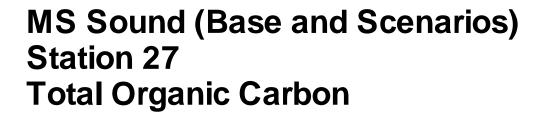




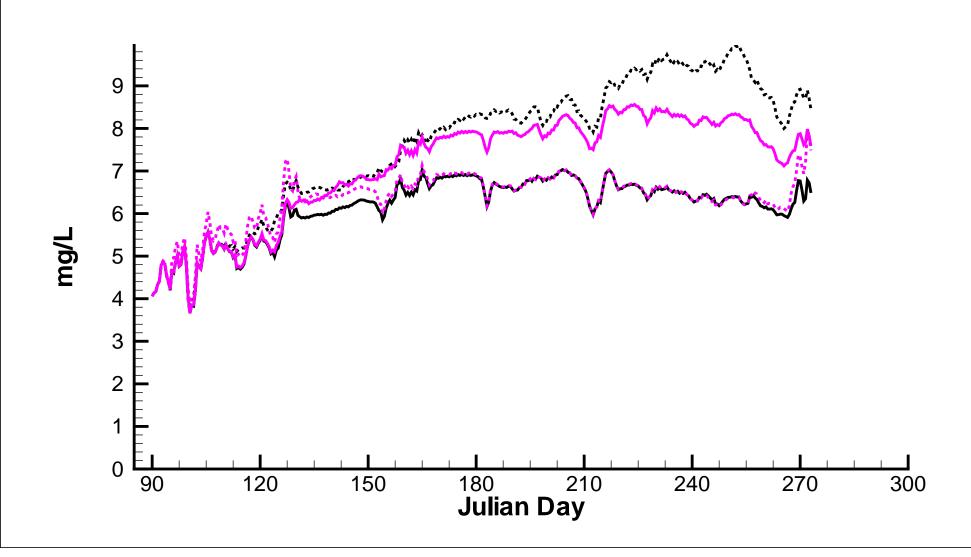


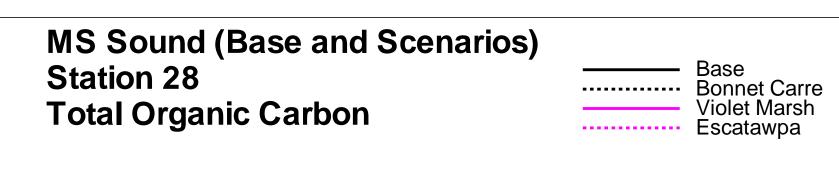


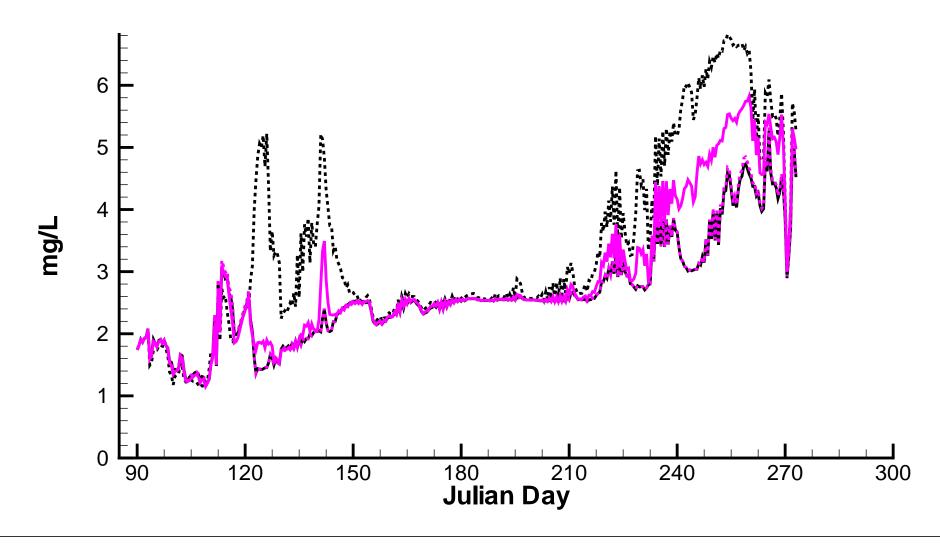


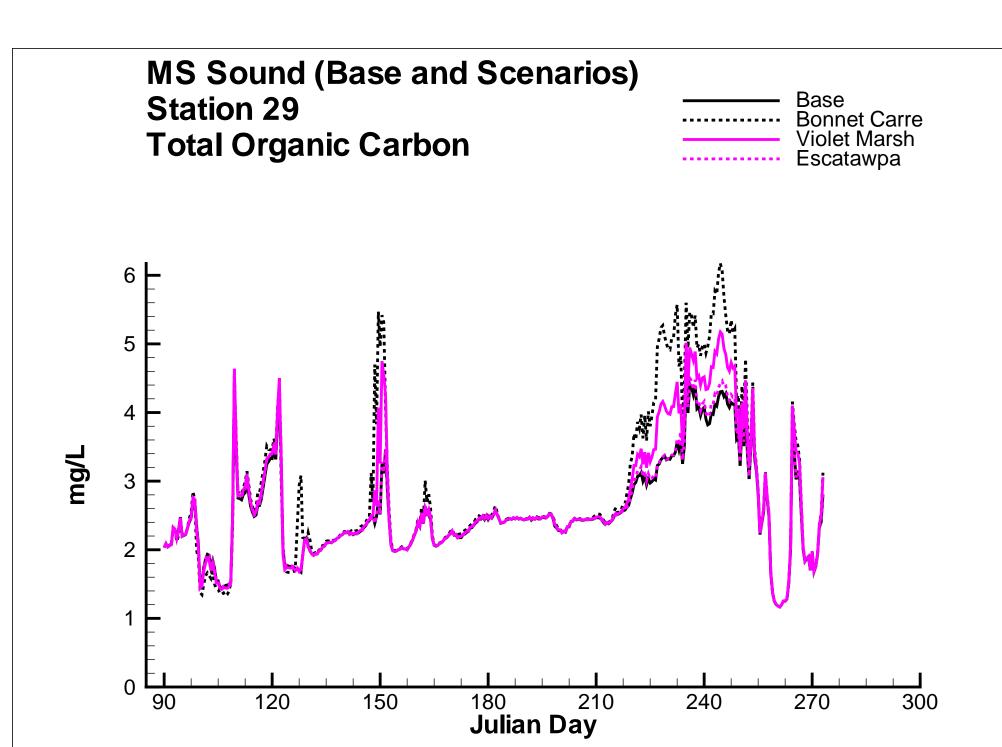


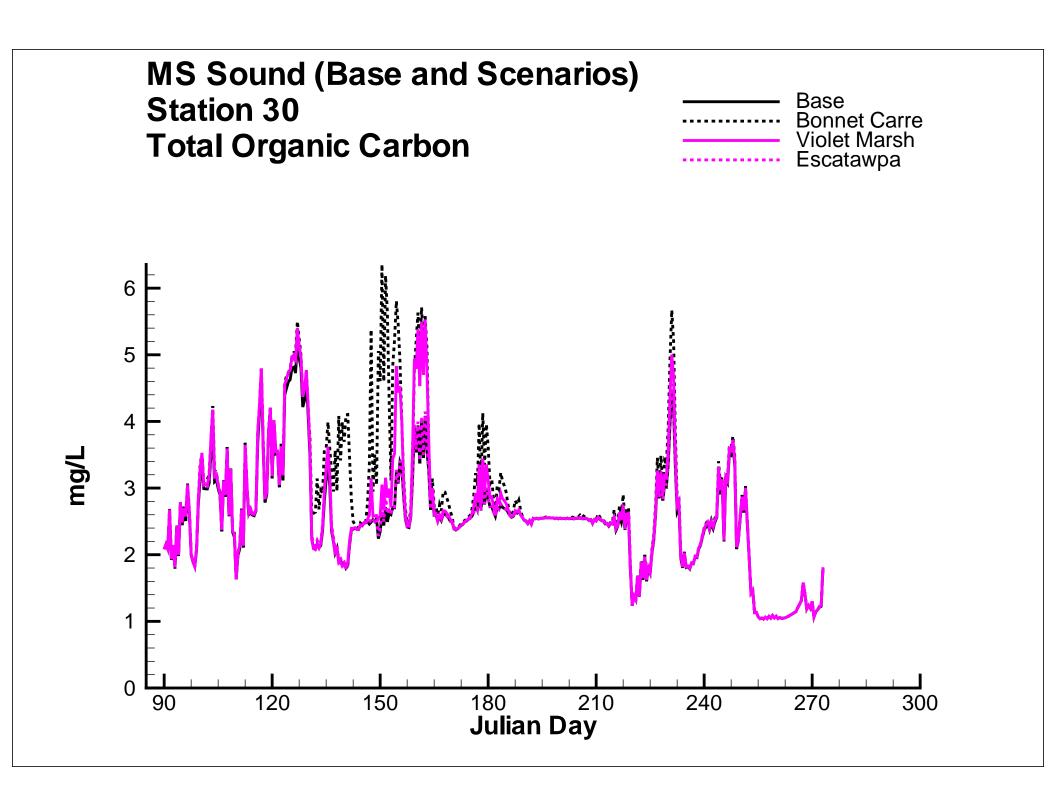


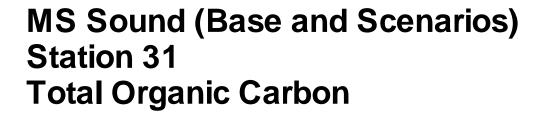




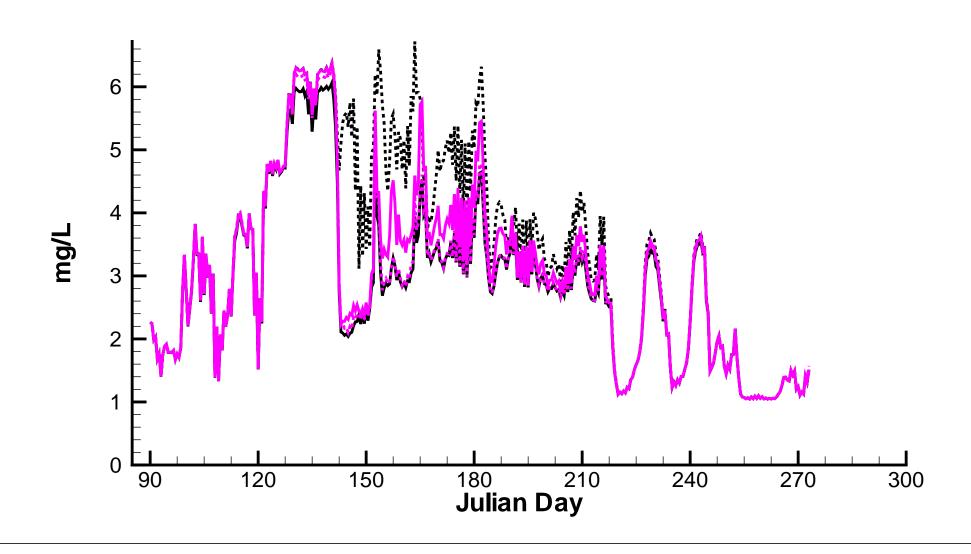


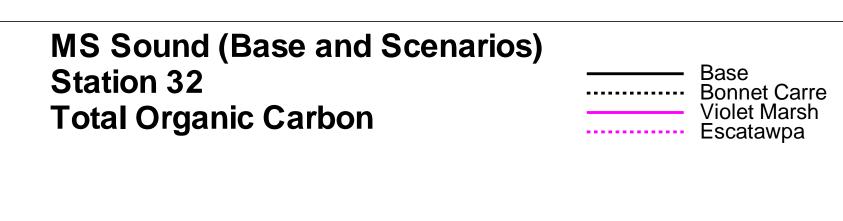


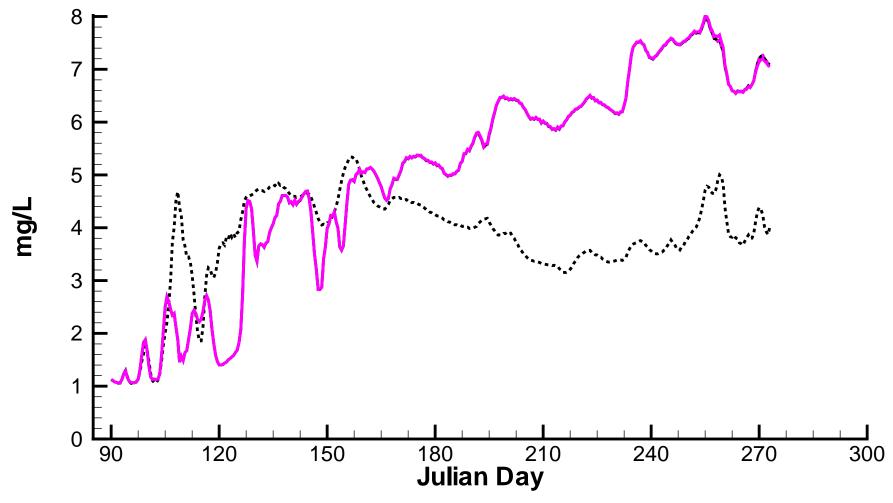


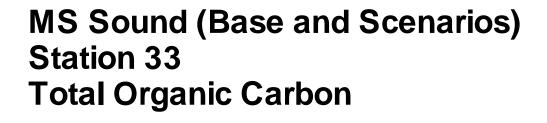


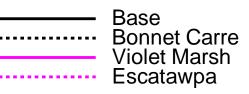


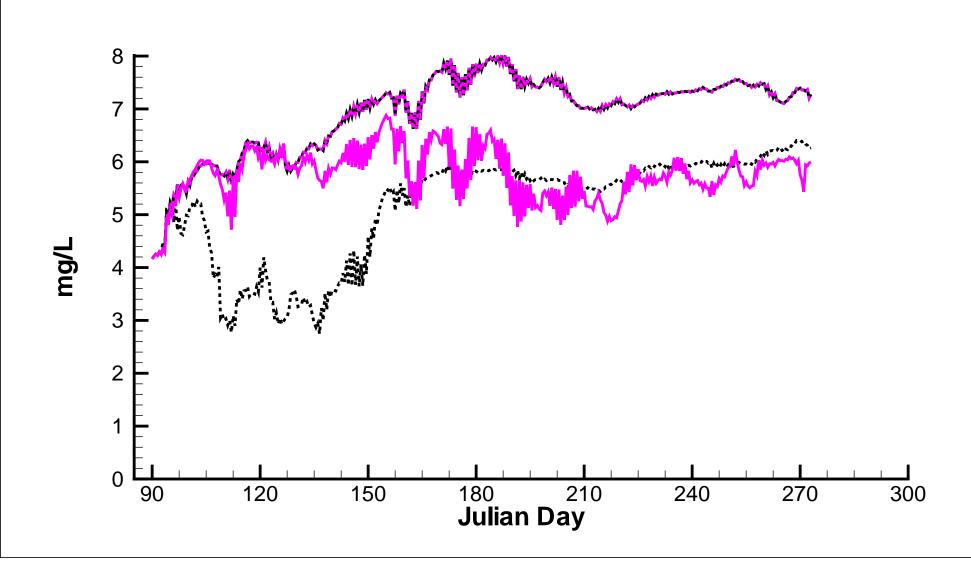


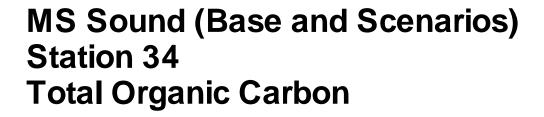




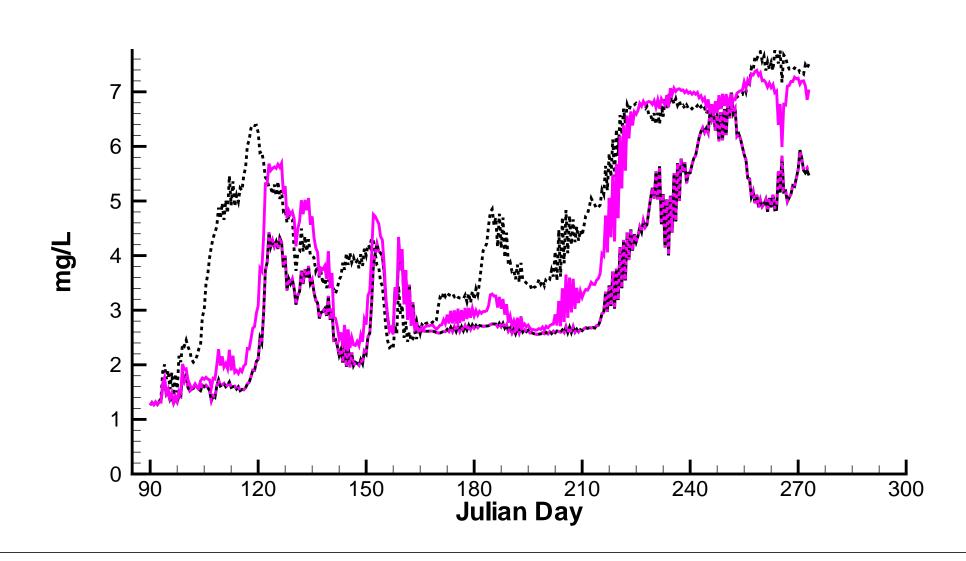


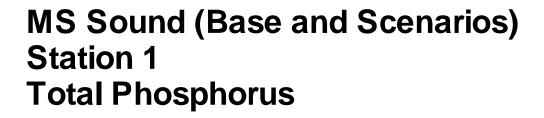




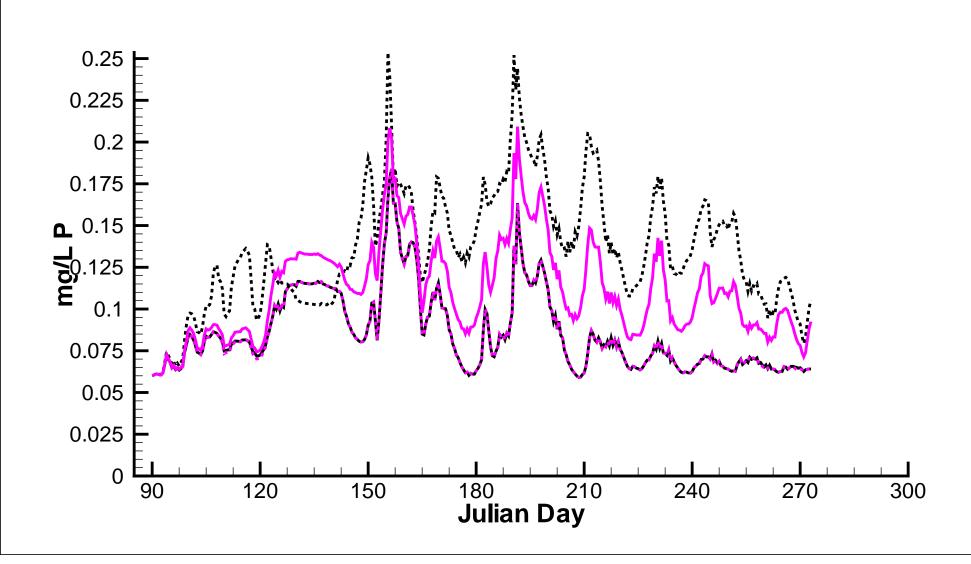


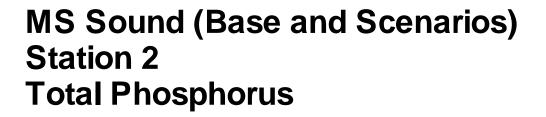




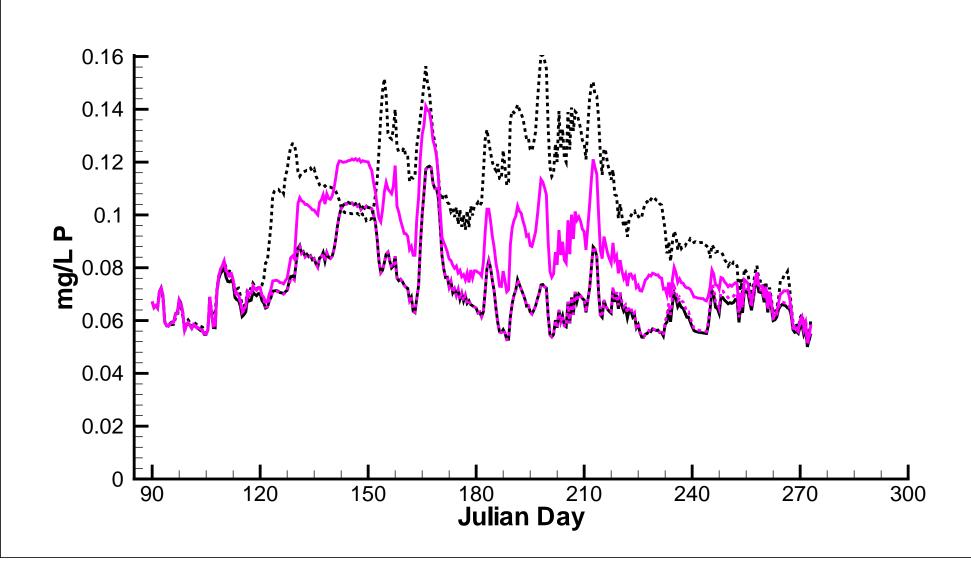


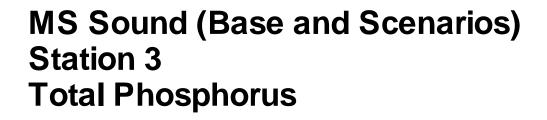




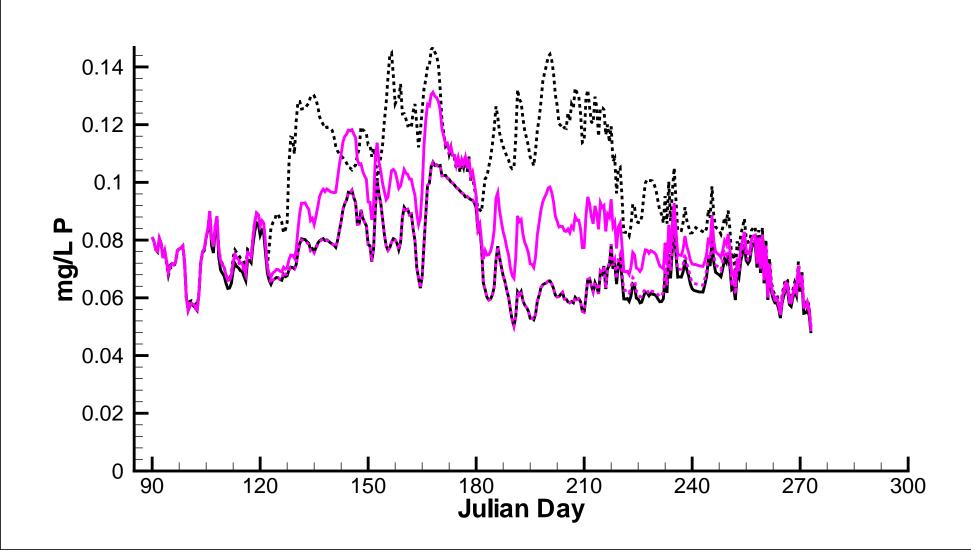


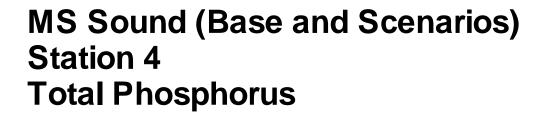




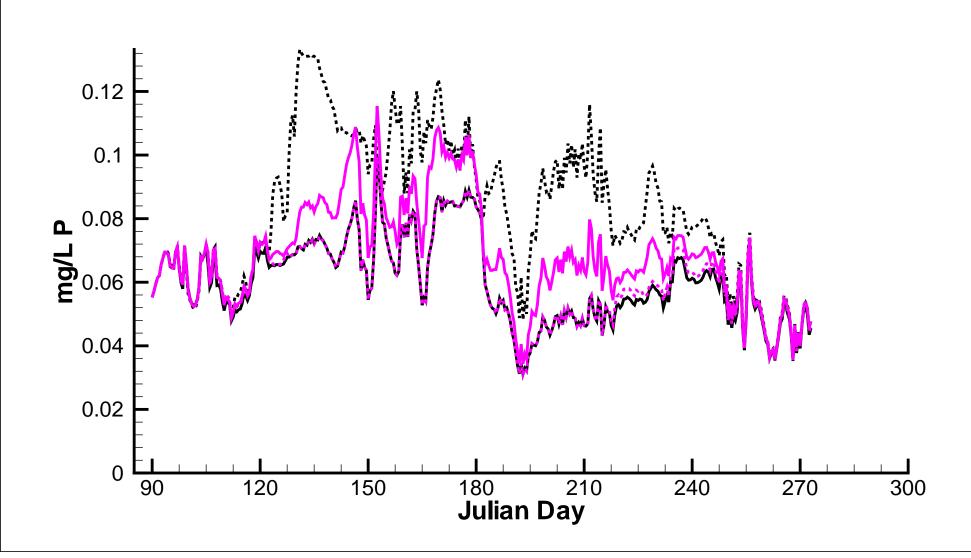


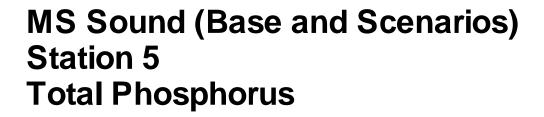




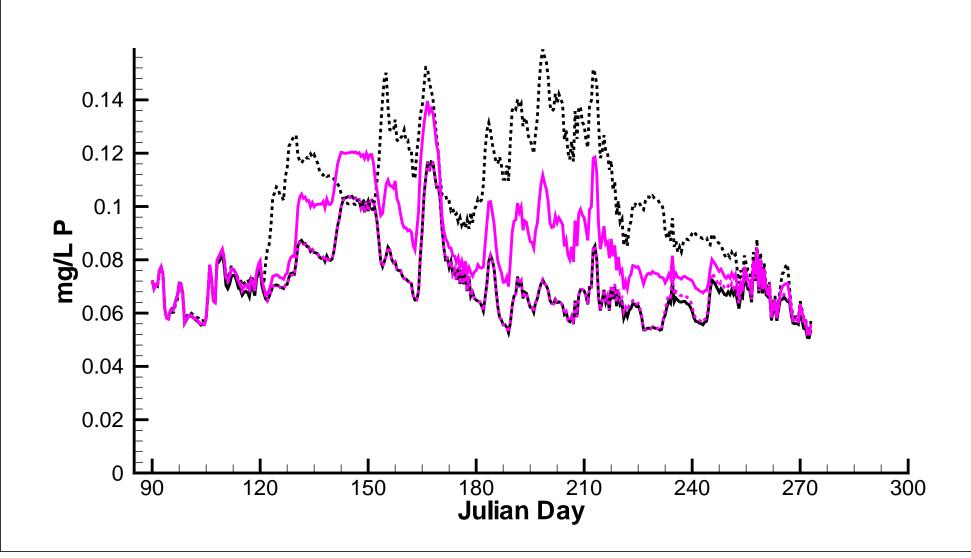


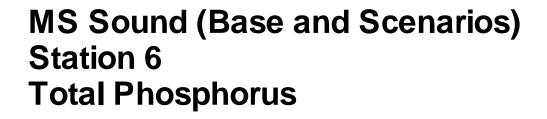




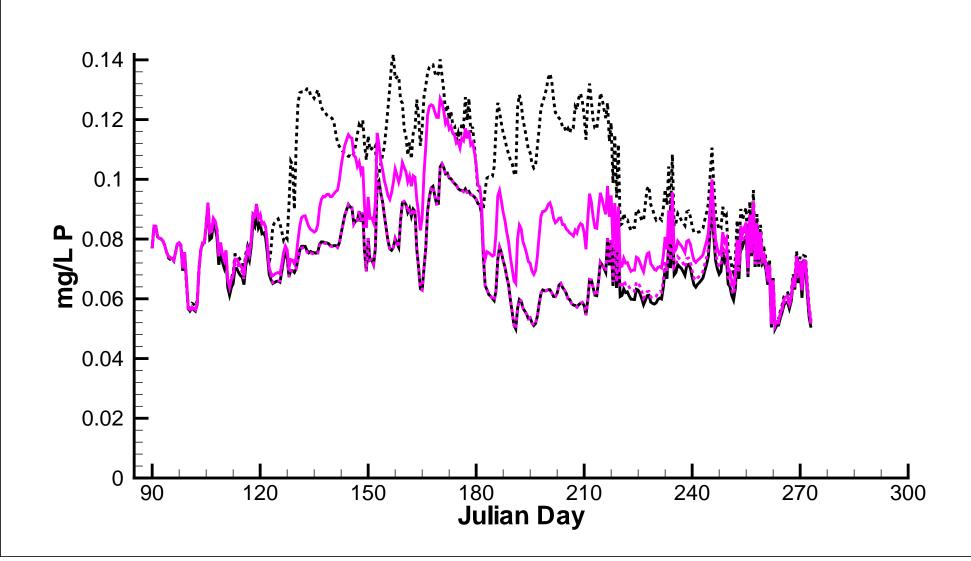


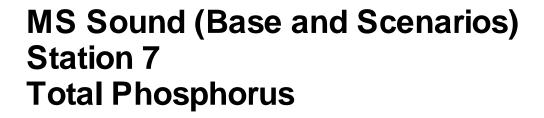




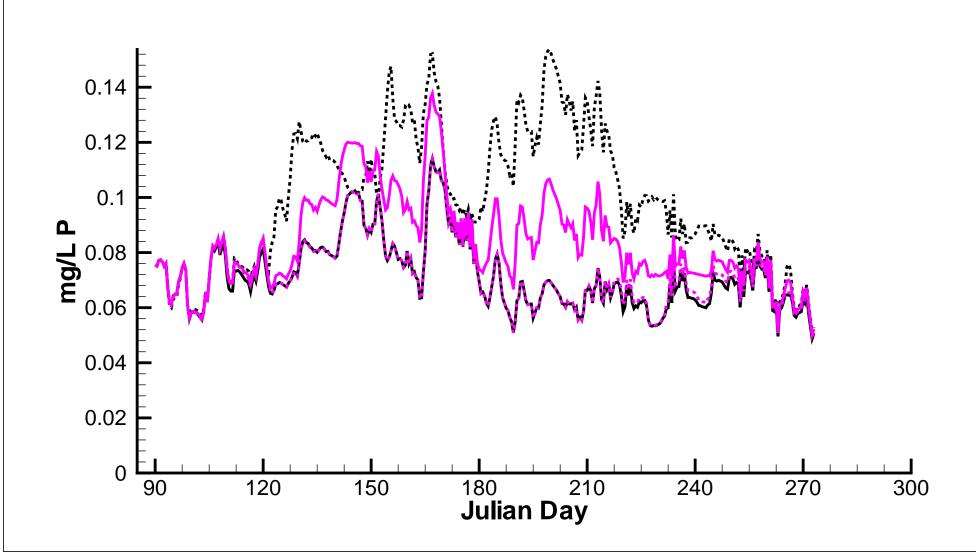


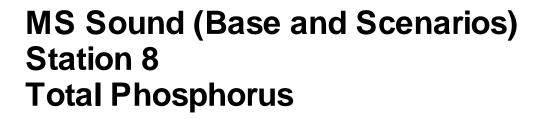


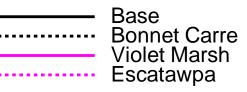


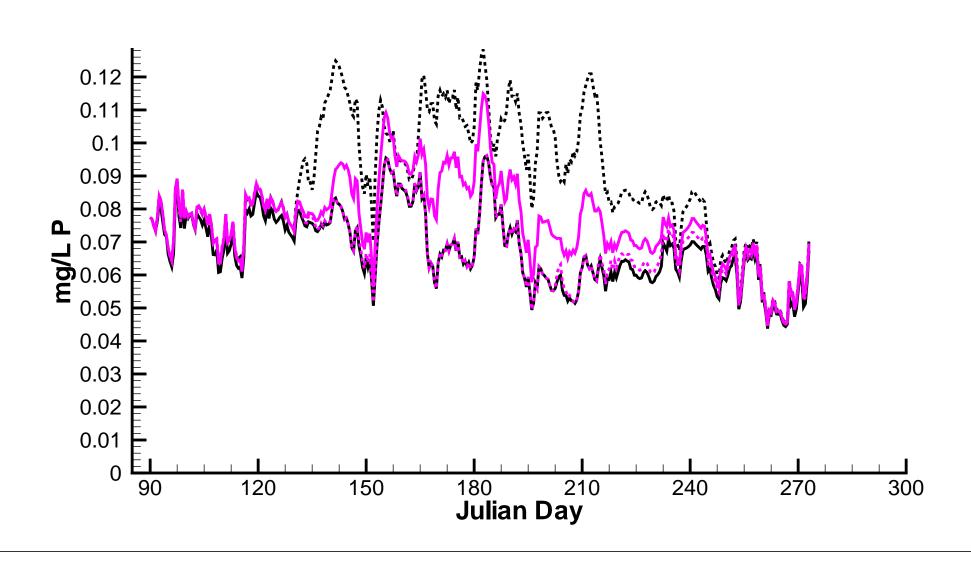


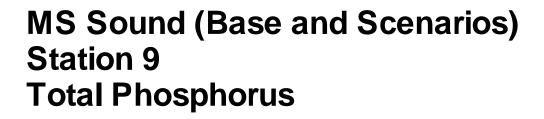




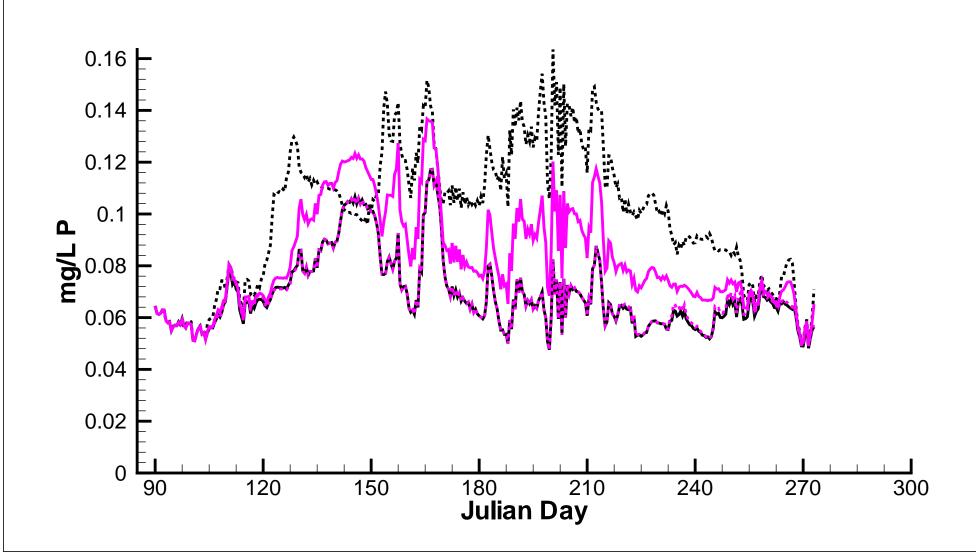


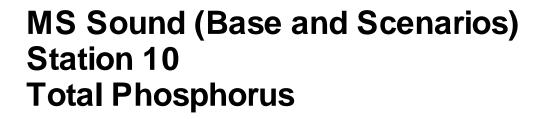




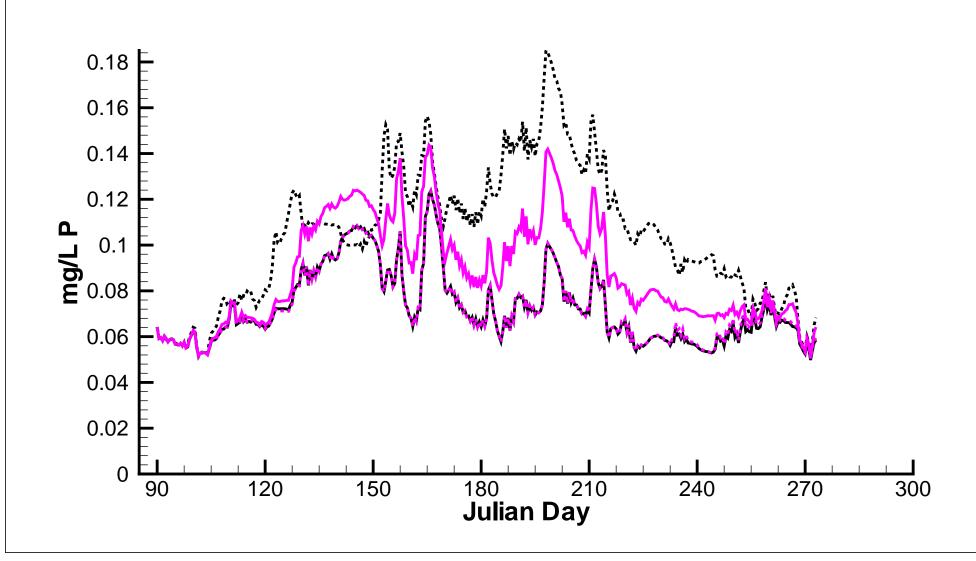


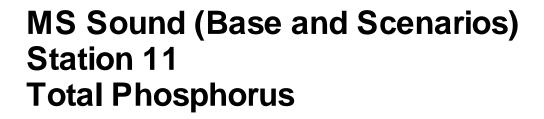




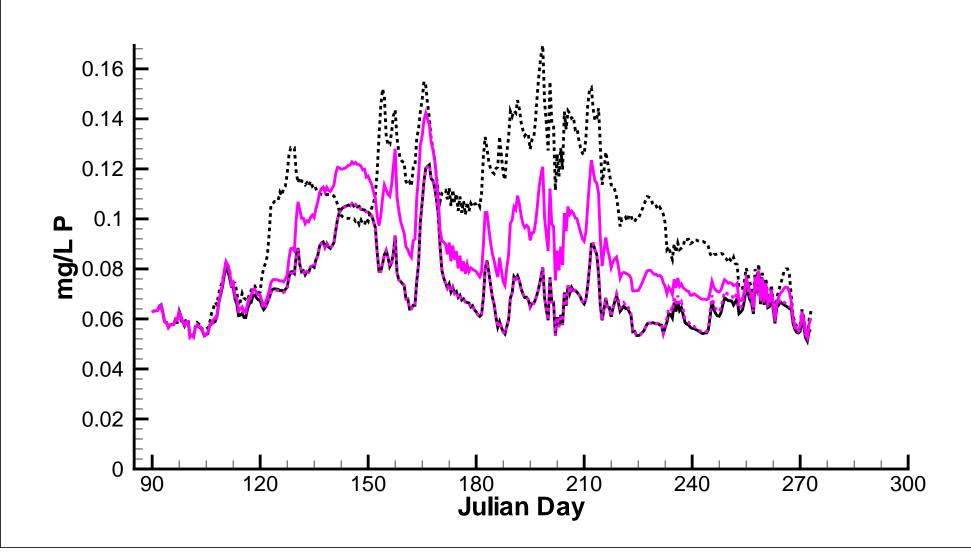


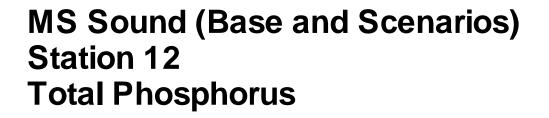




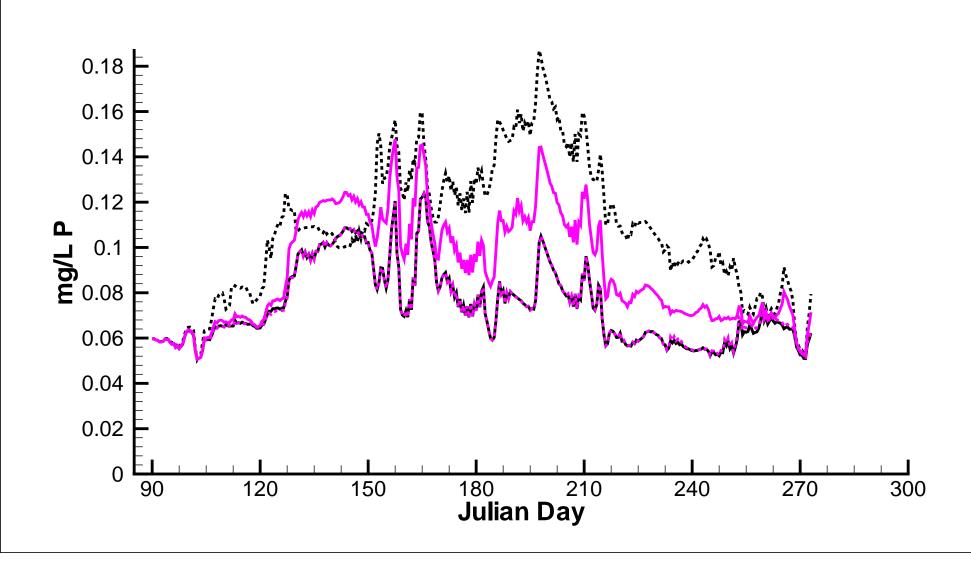


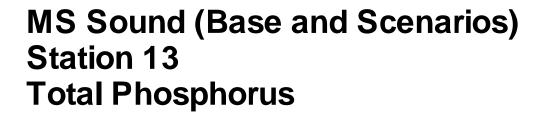




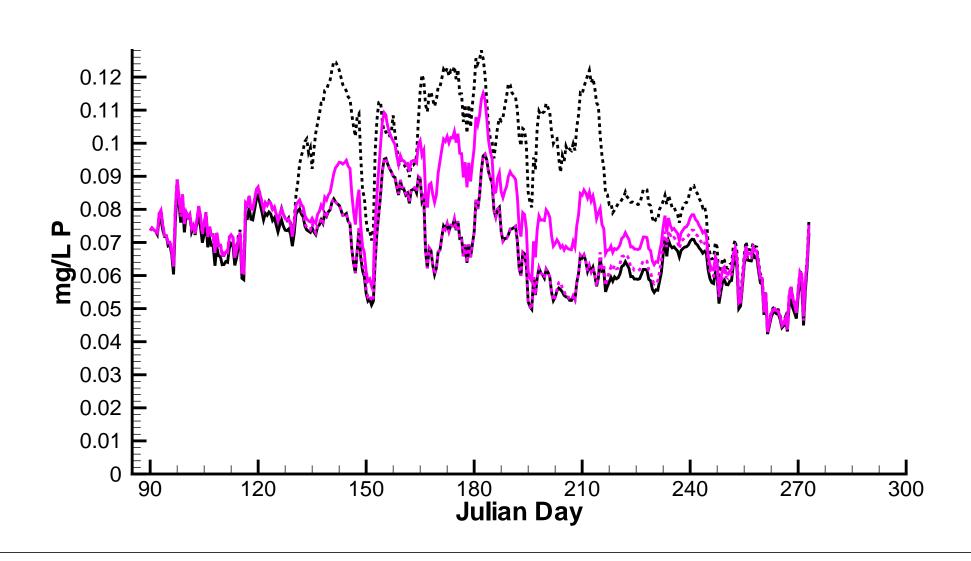


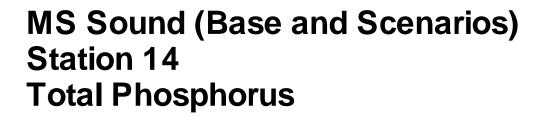




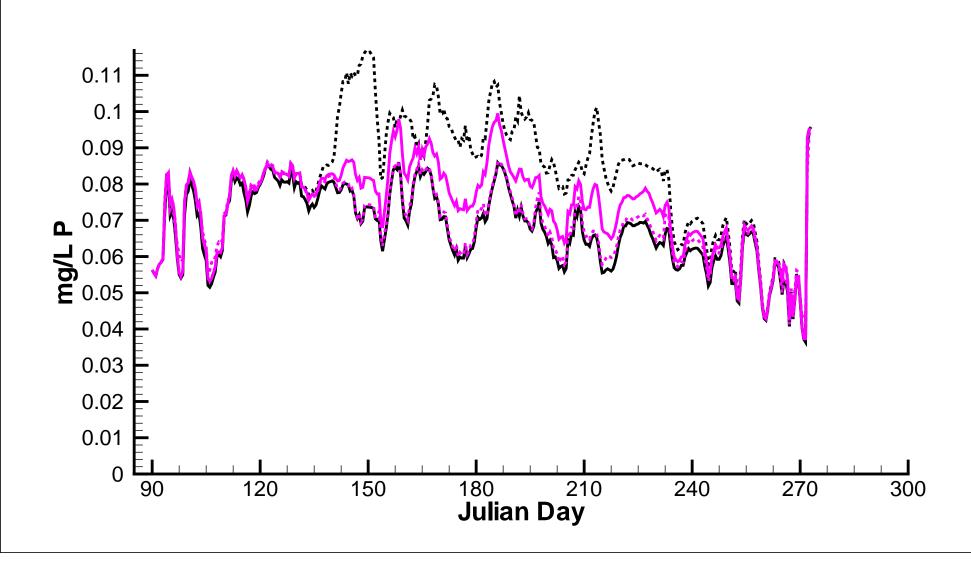


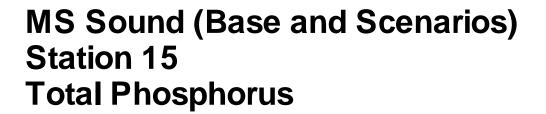




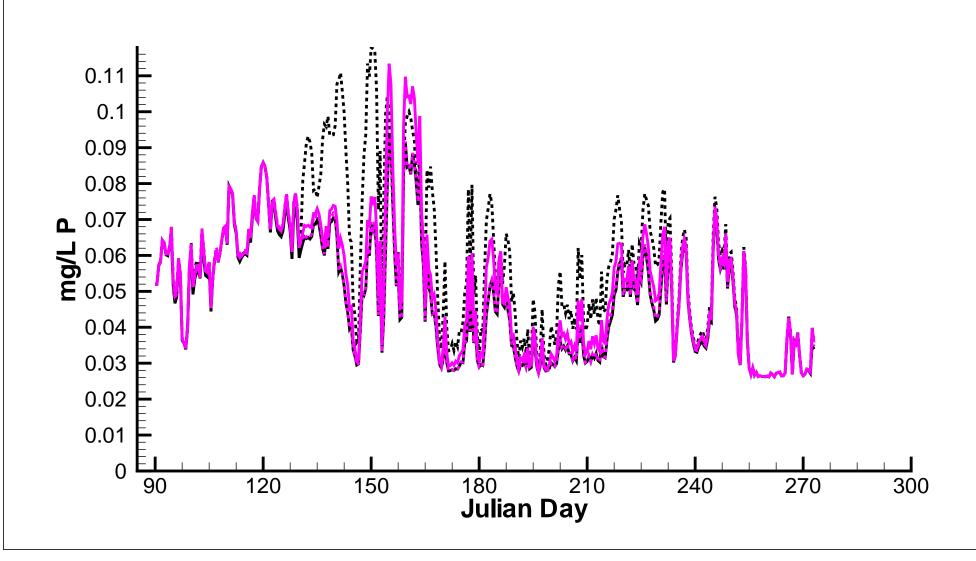


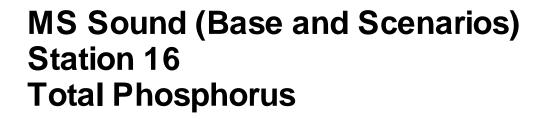




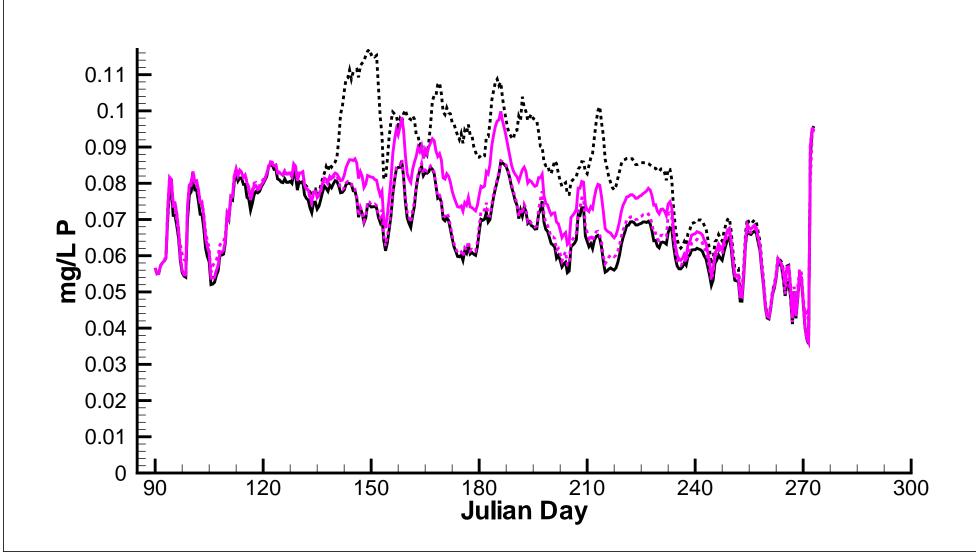






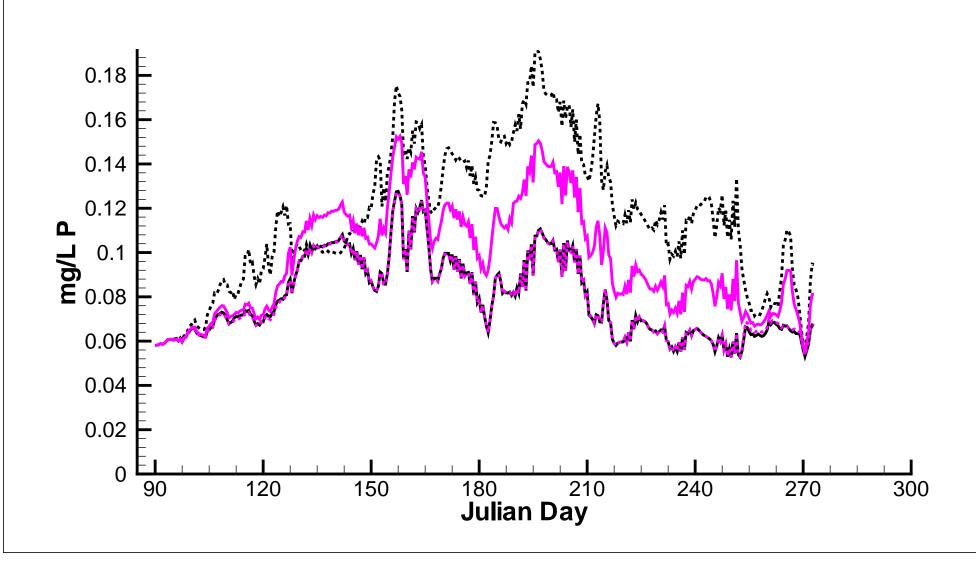


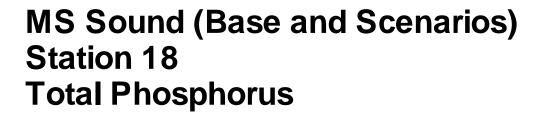




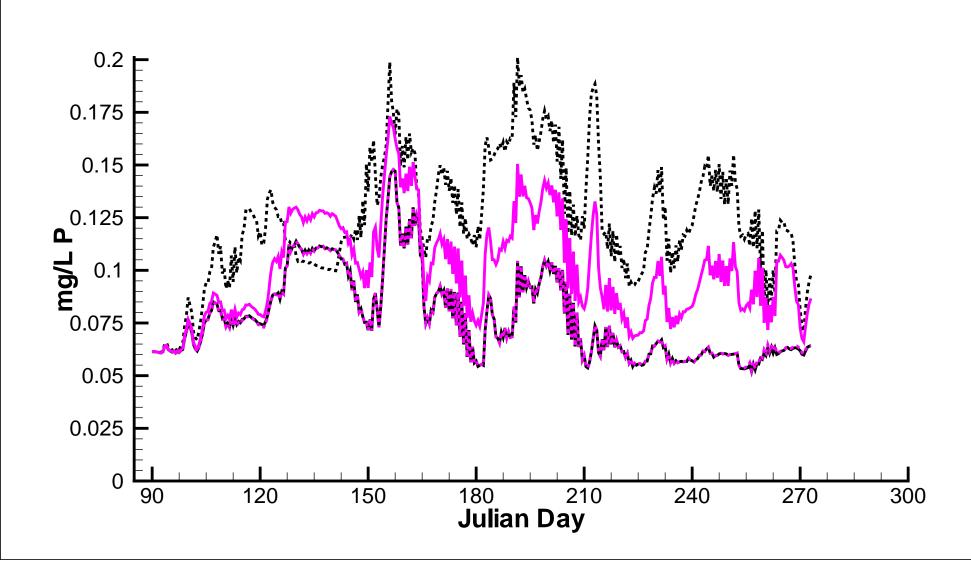


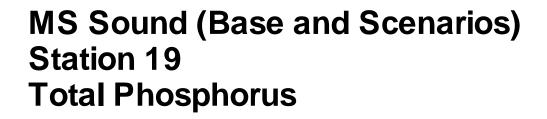




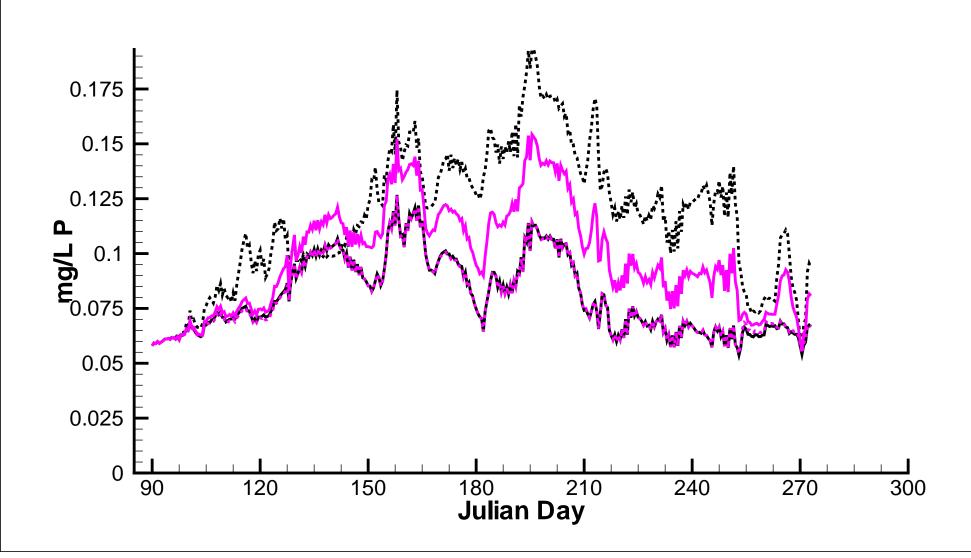


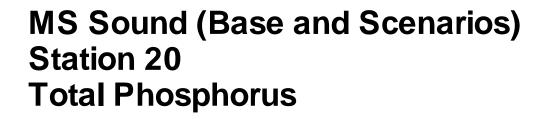




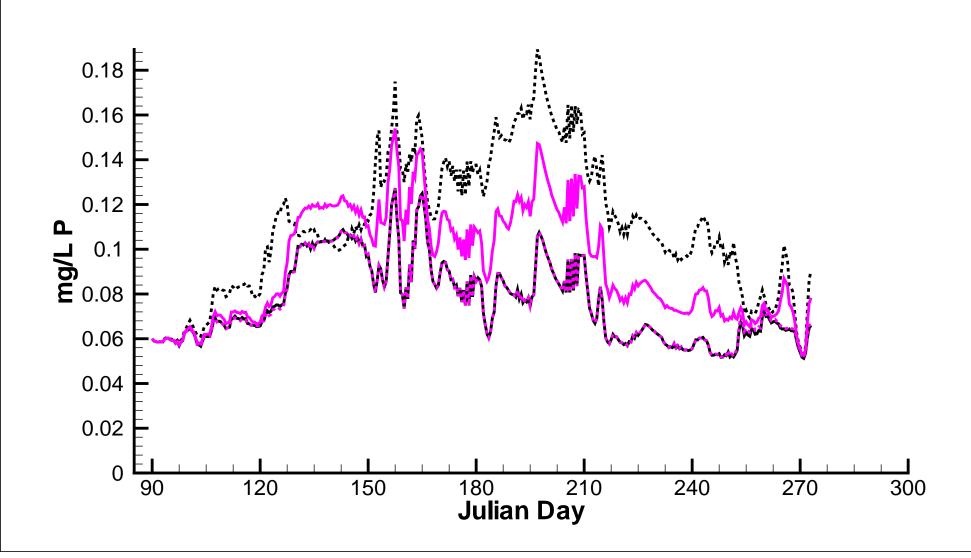


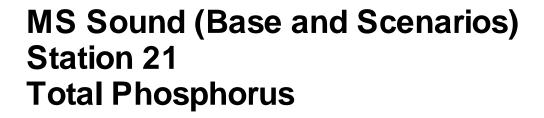


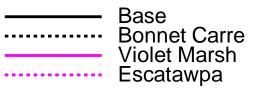


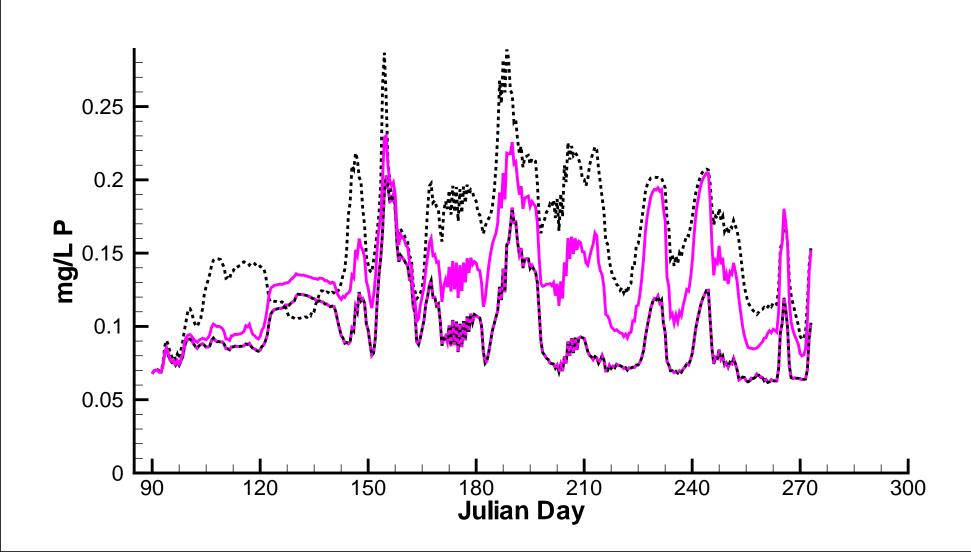


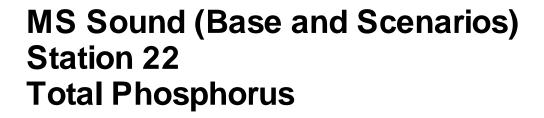




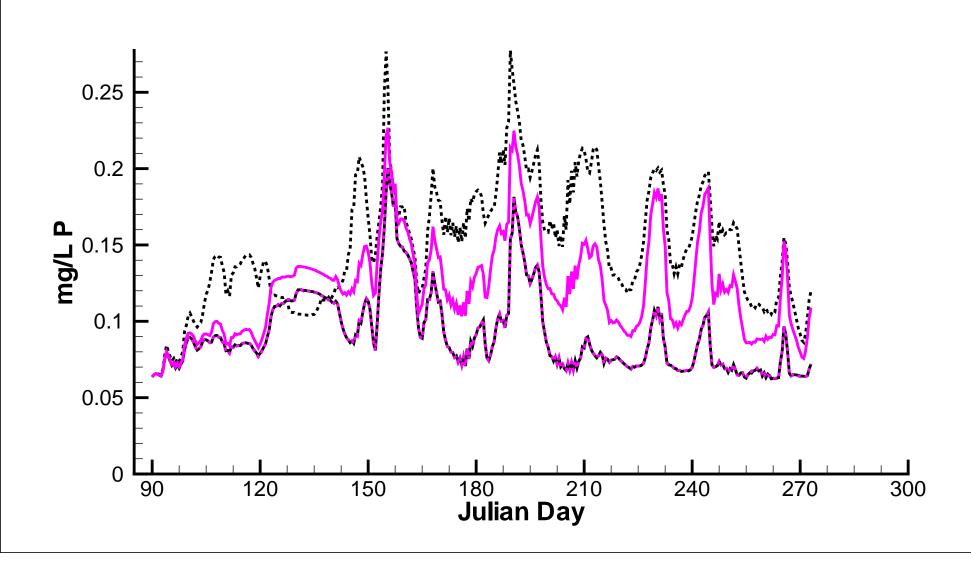


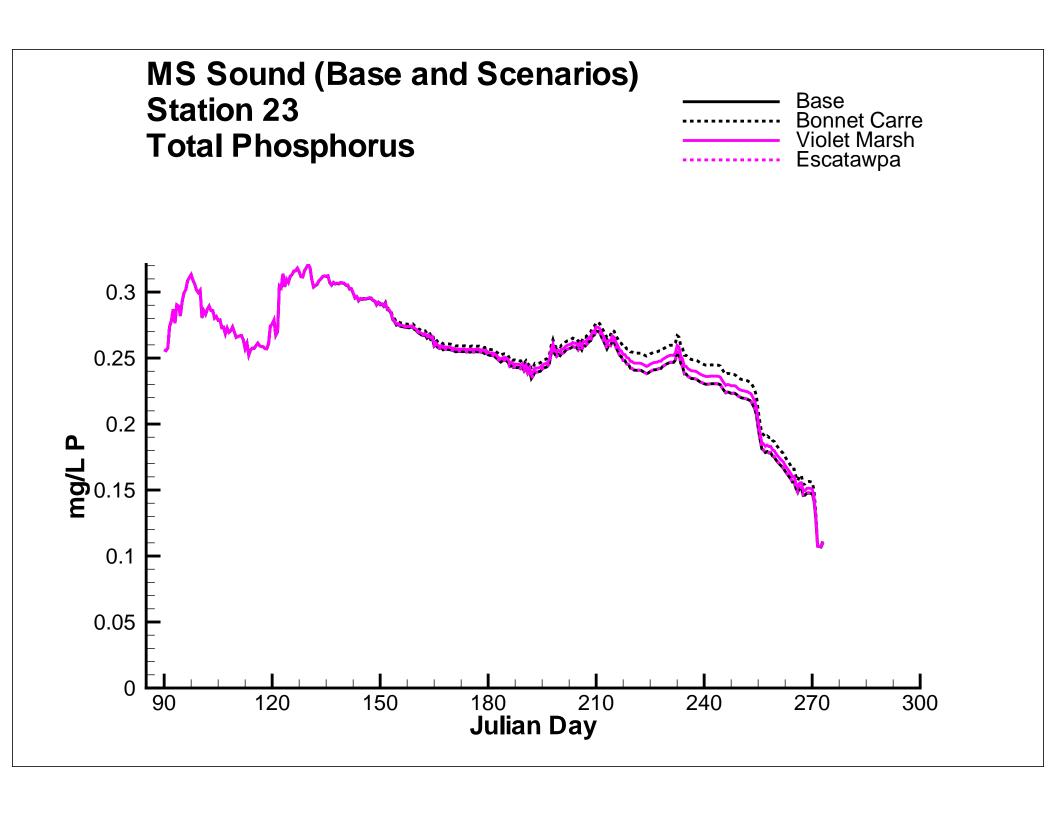


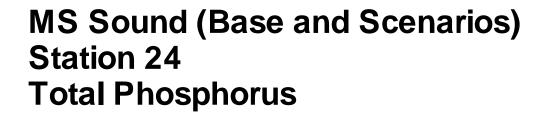




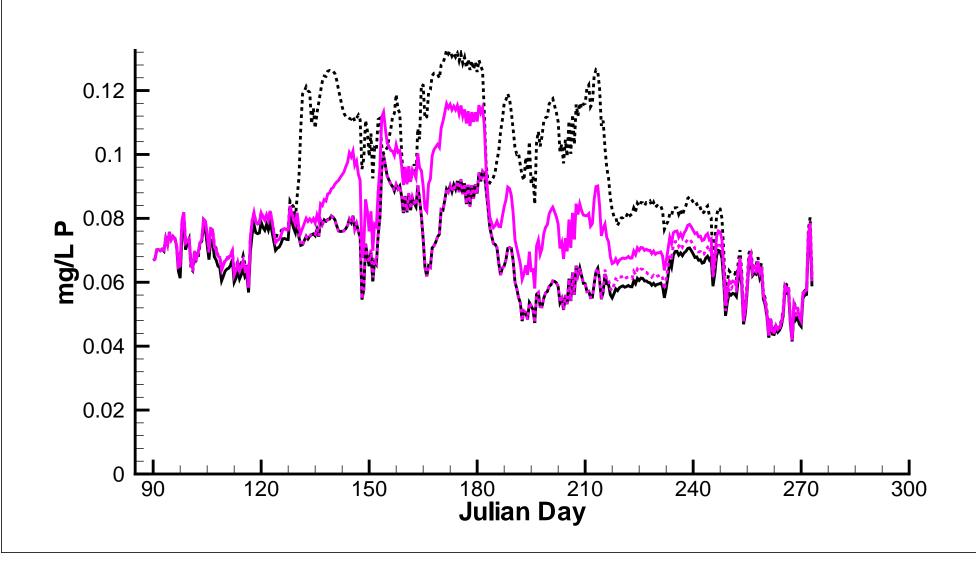


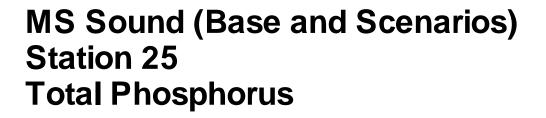




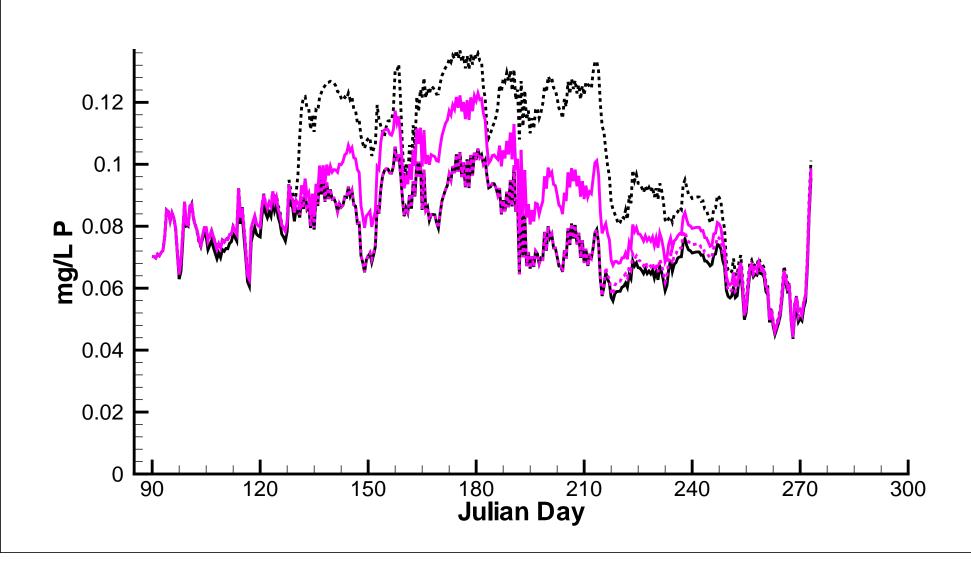


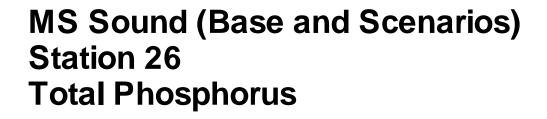




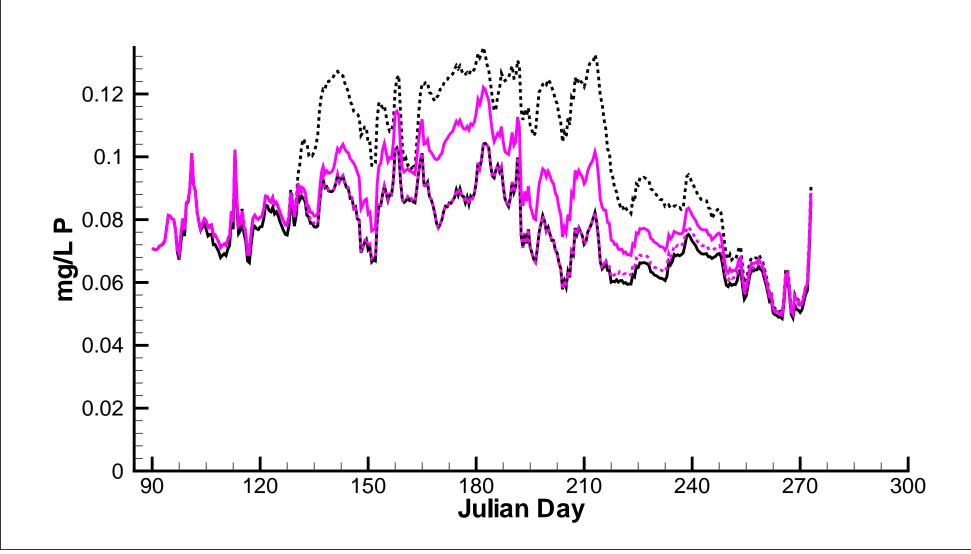






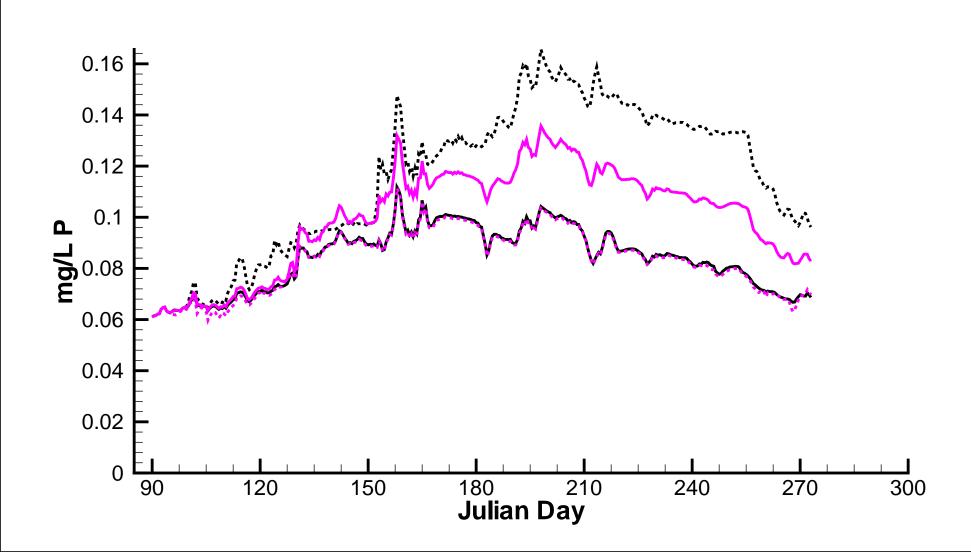


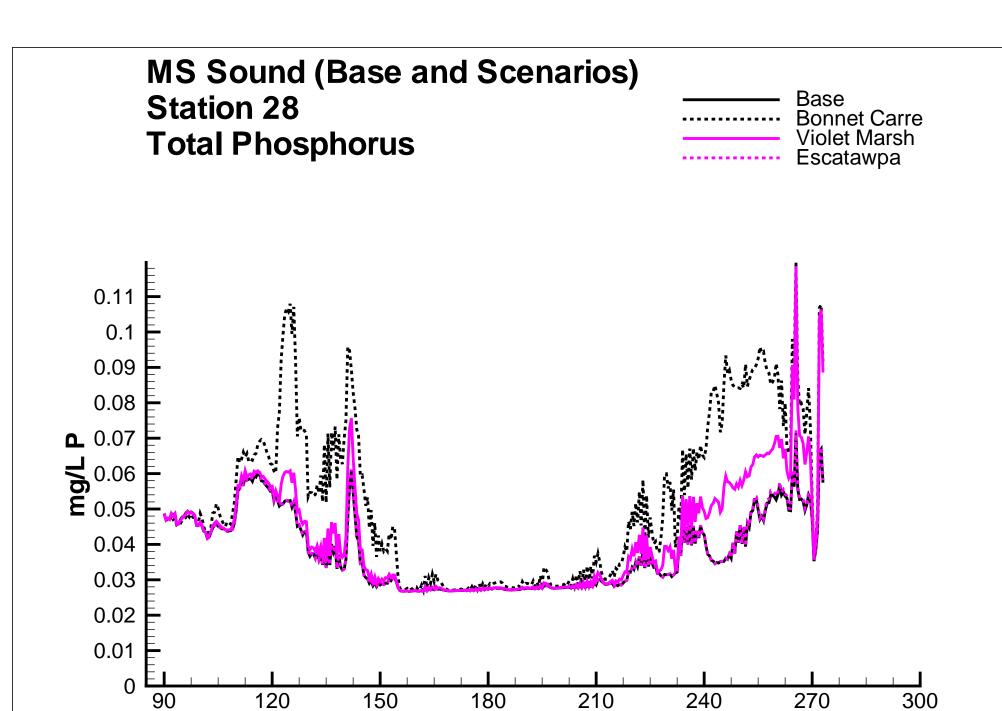




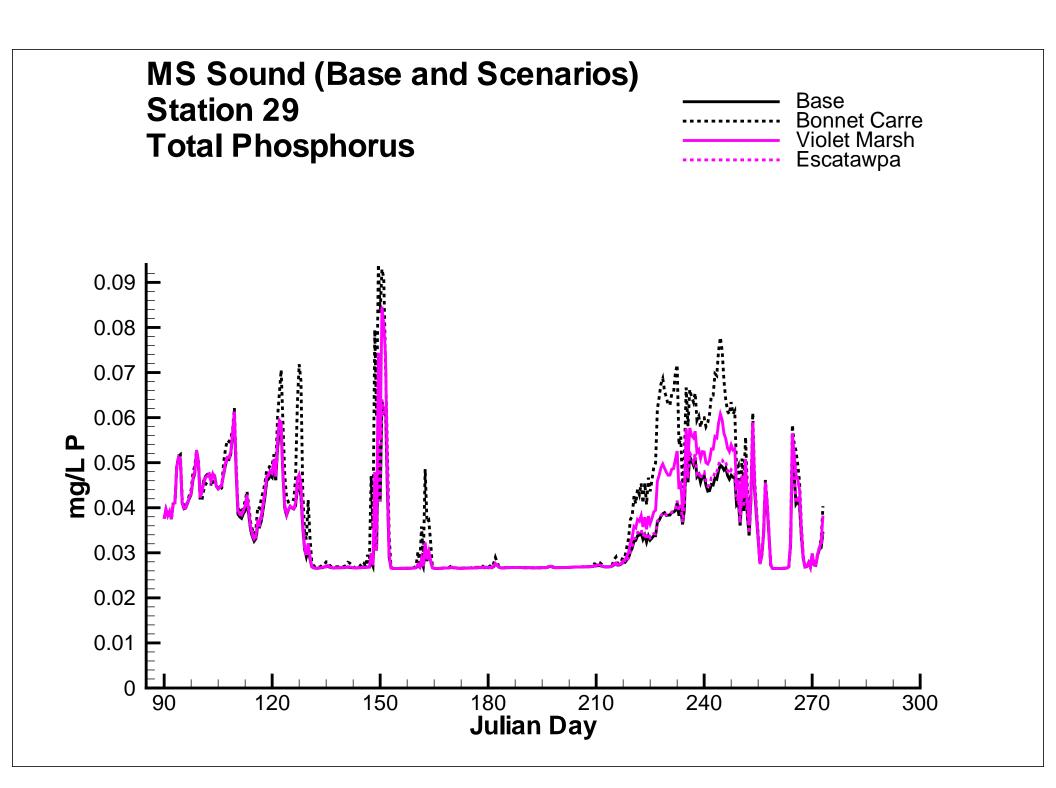


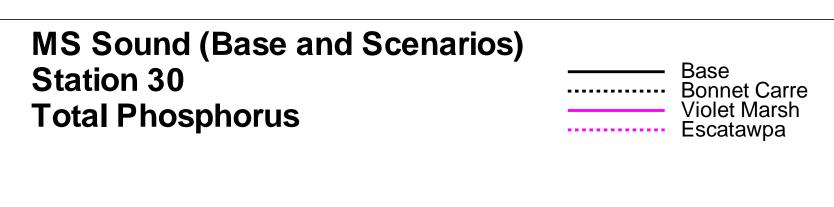


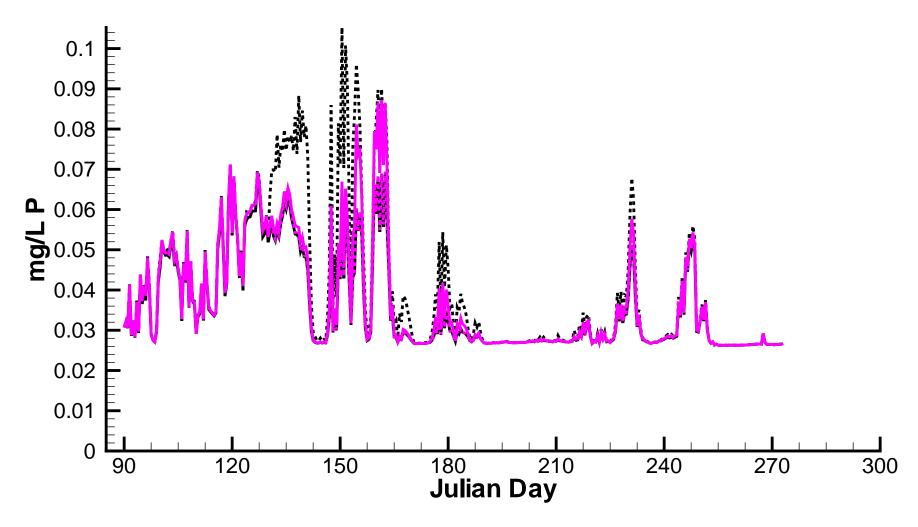


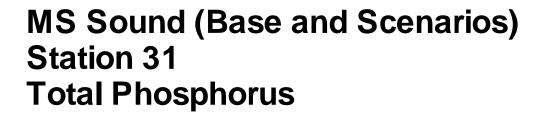


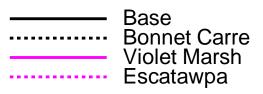
Julian Day

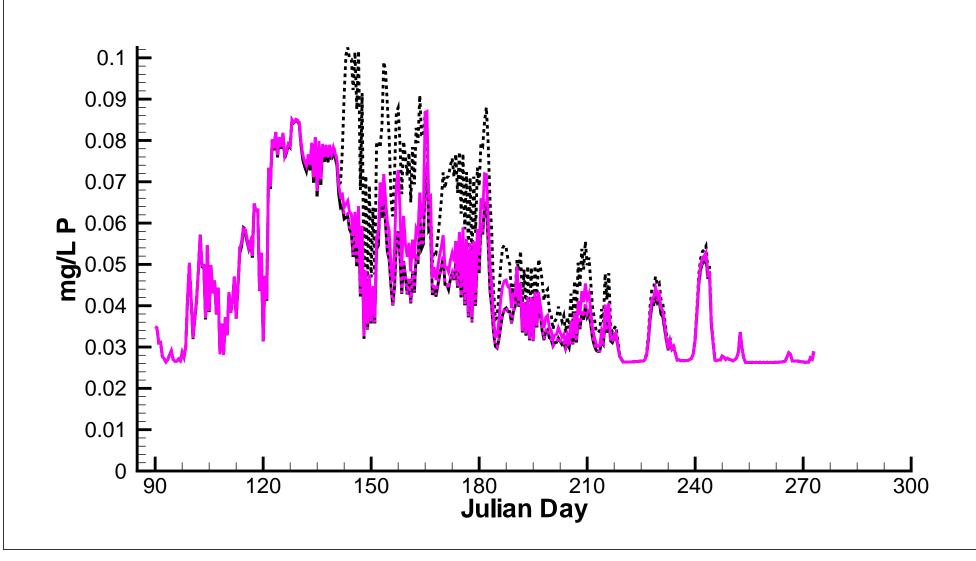


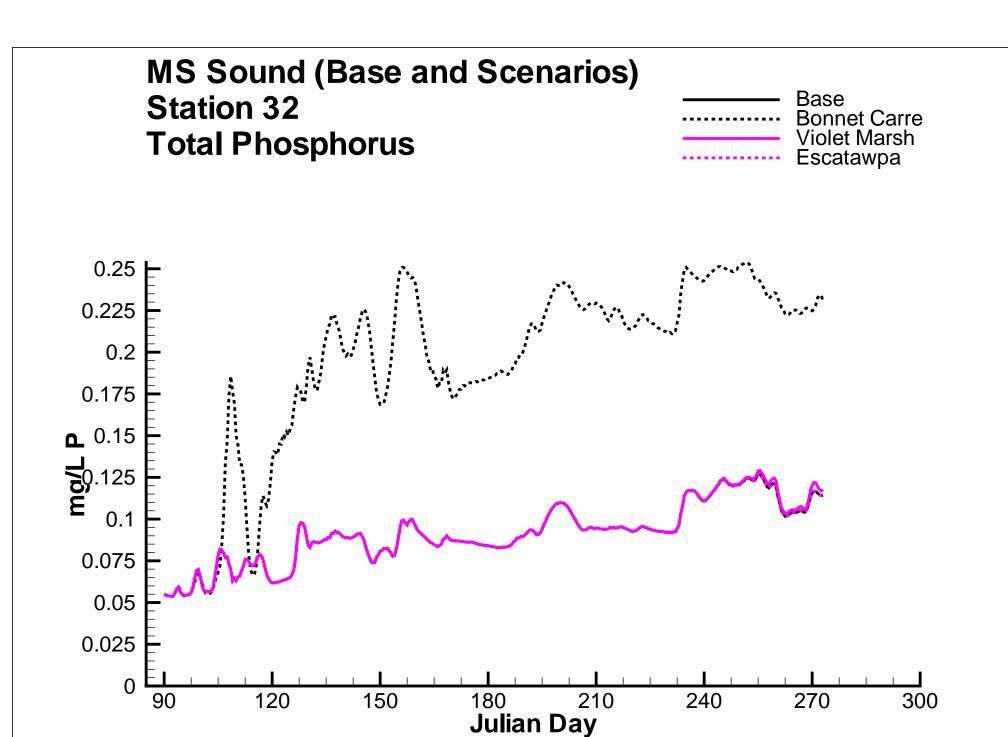


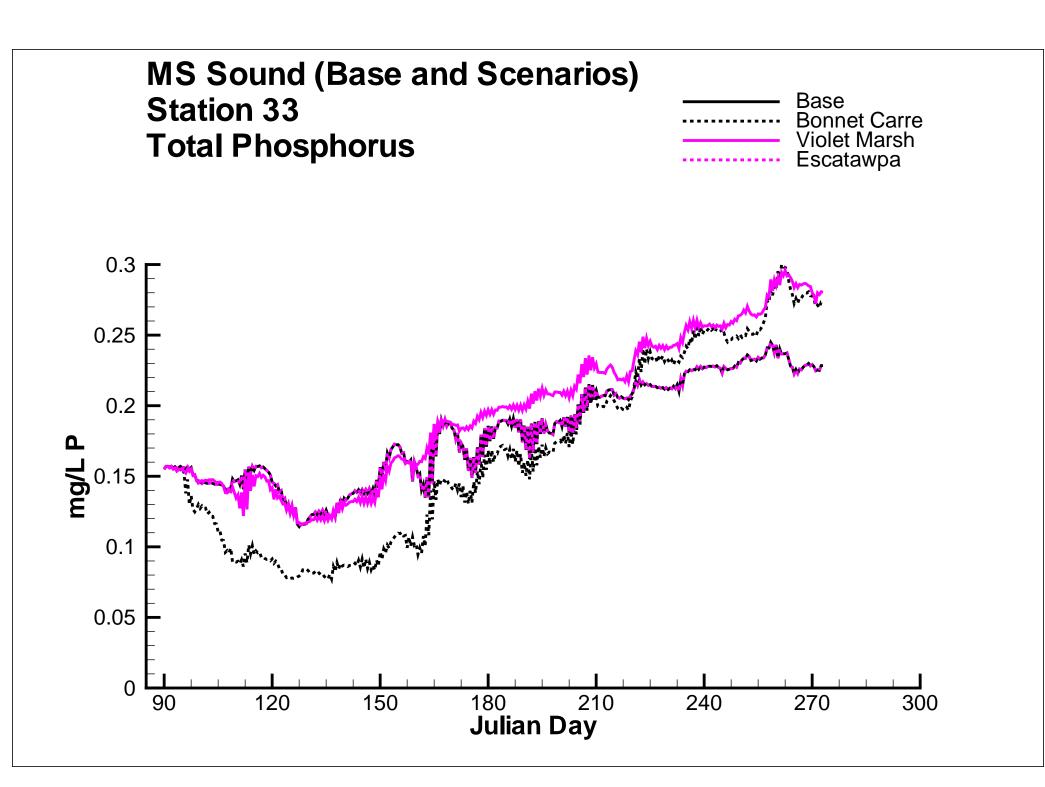




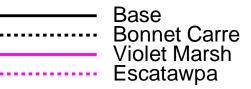


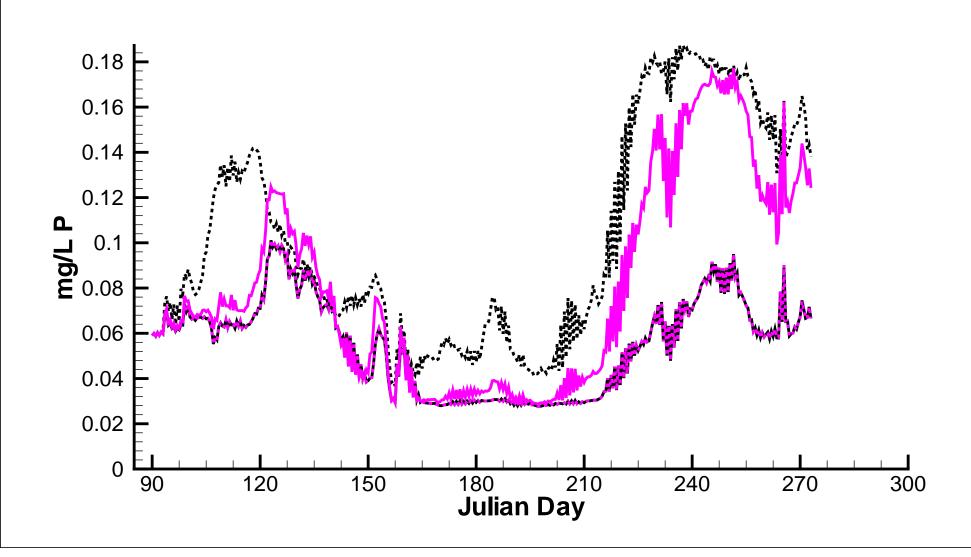






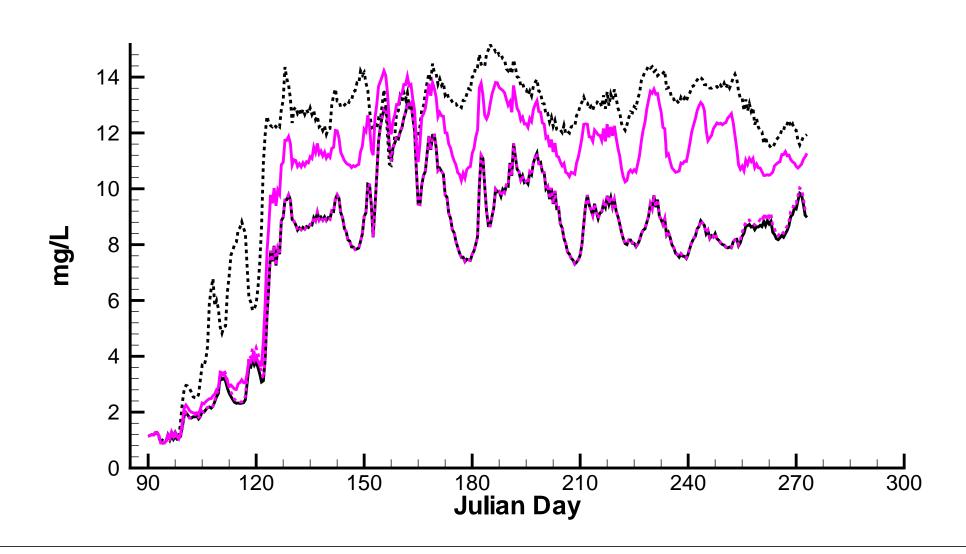




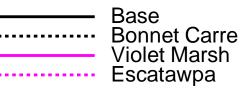


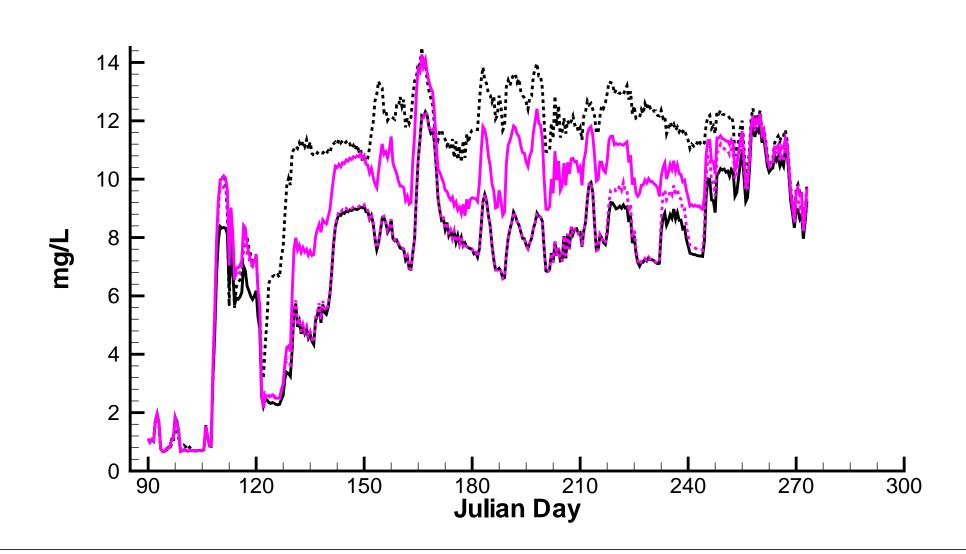




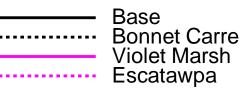


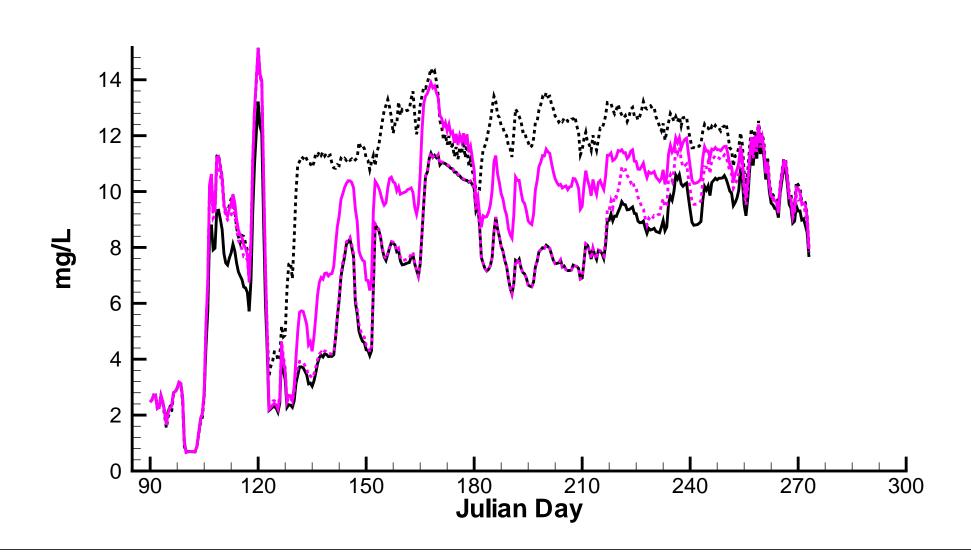


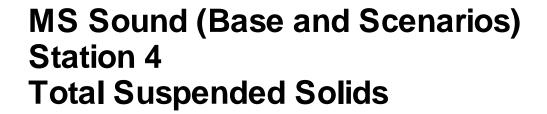




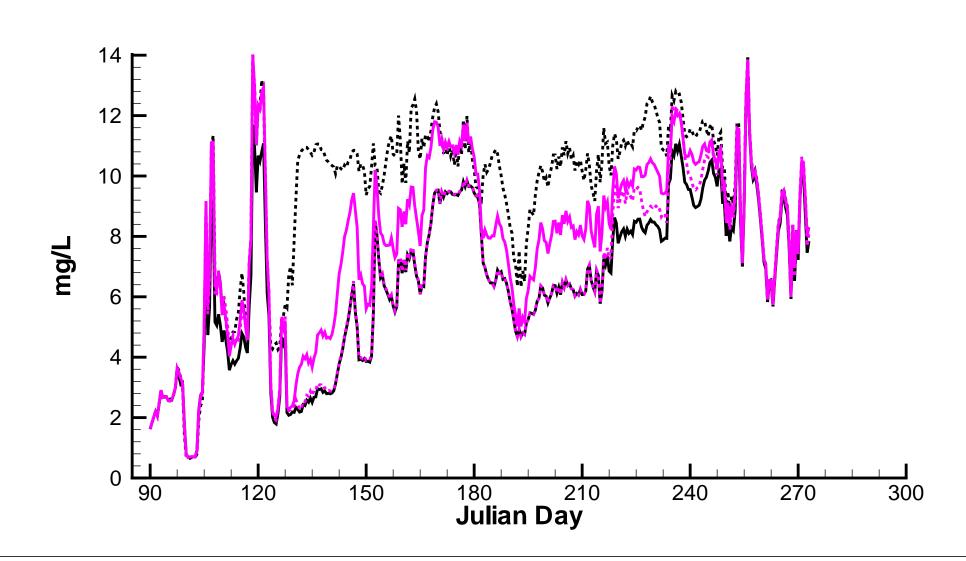


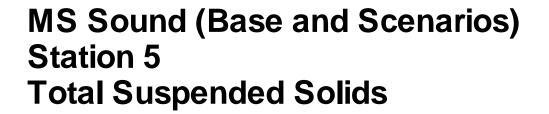


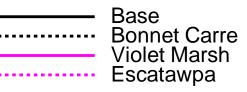


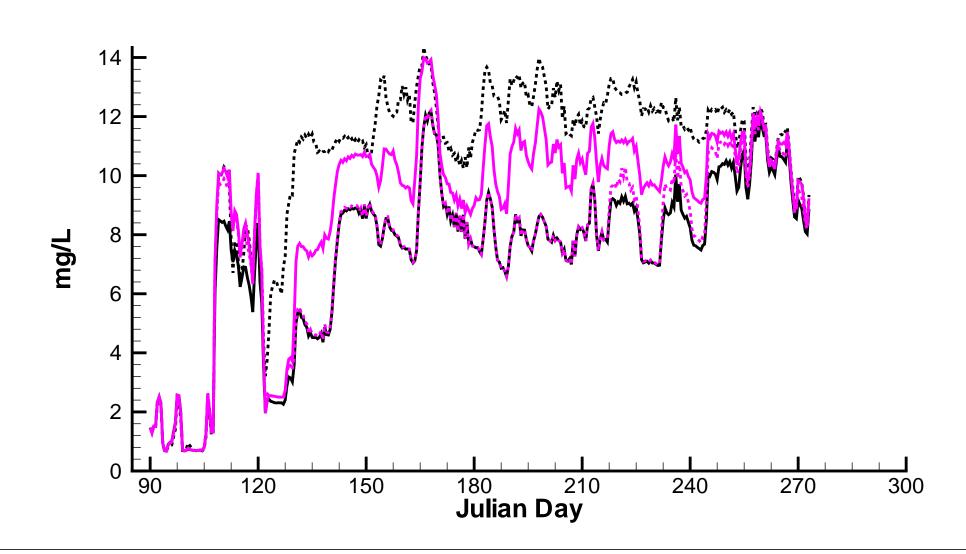






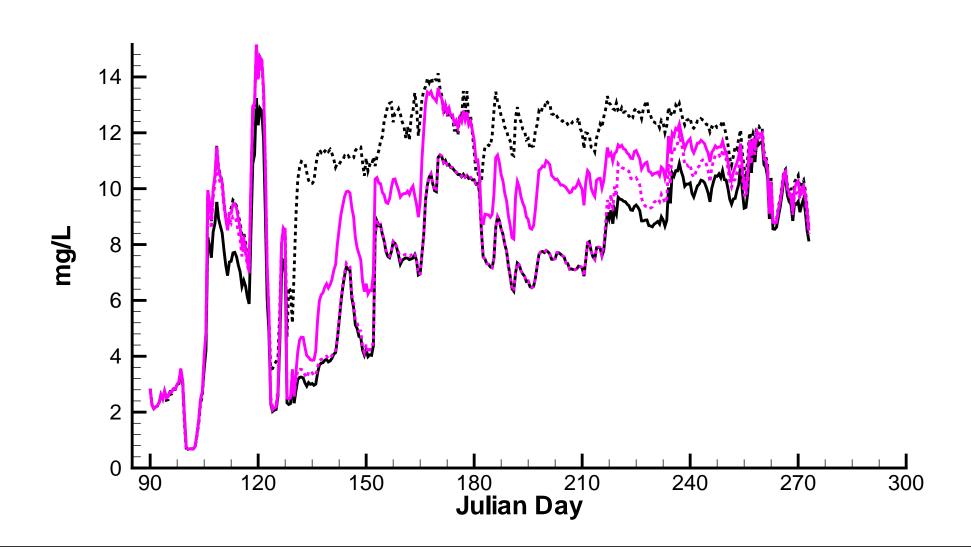






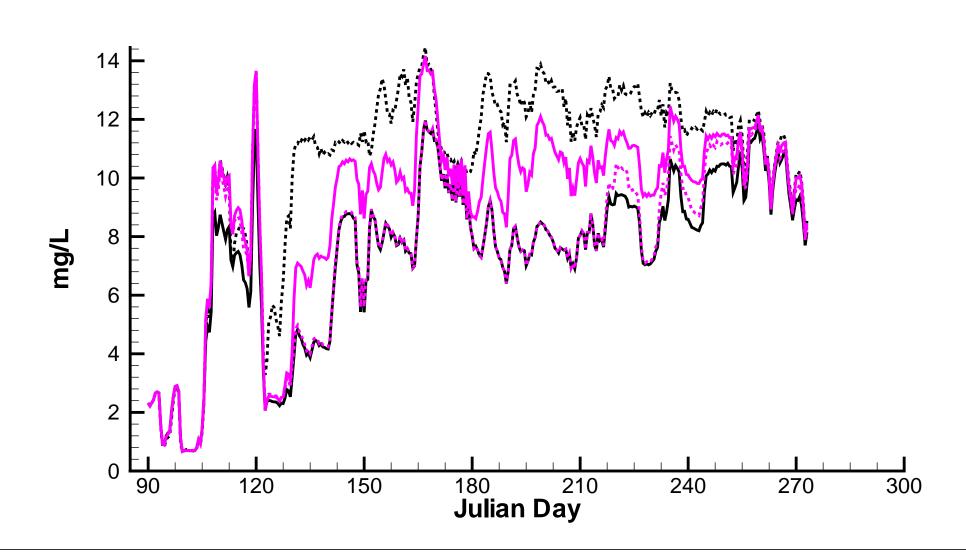


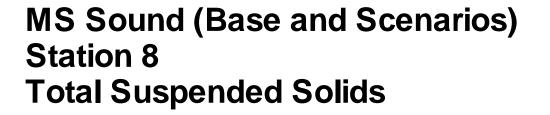




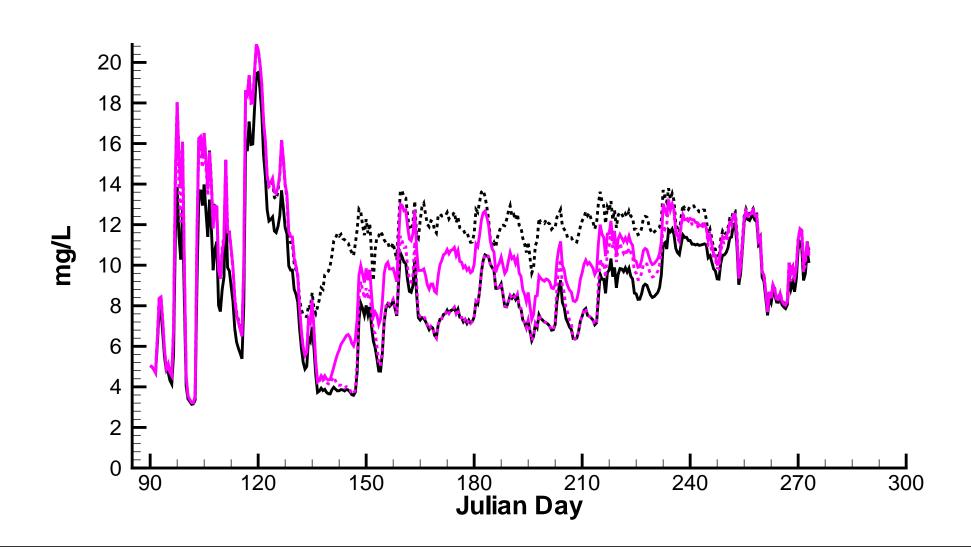


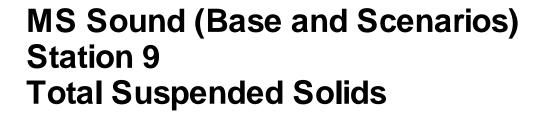




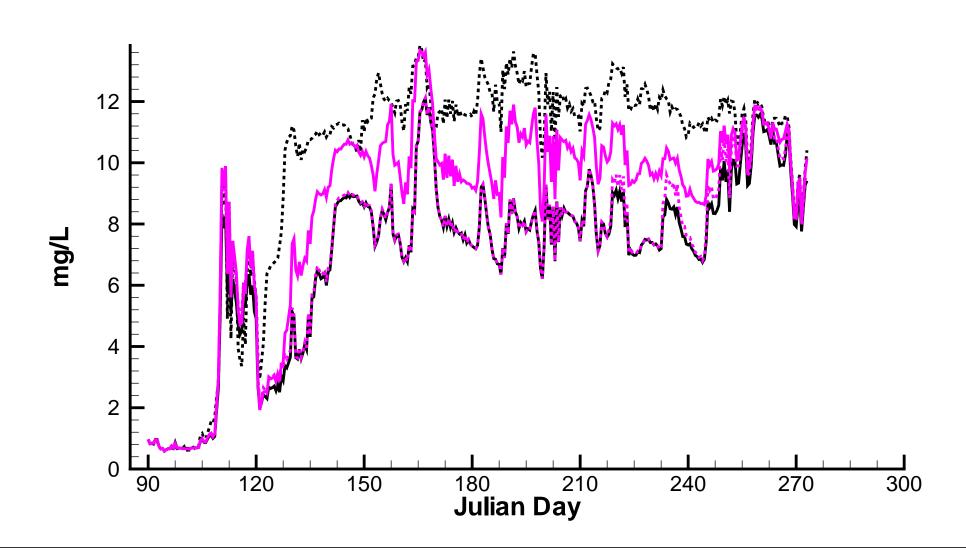


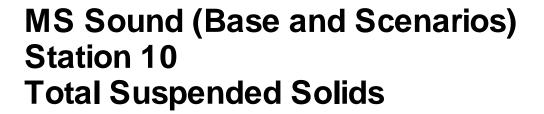




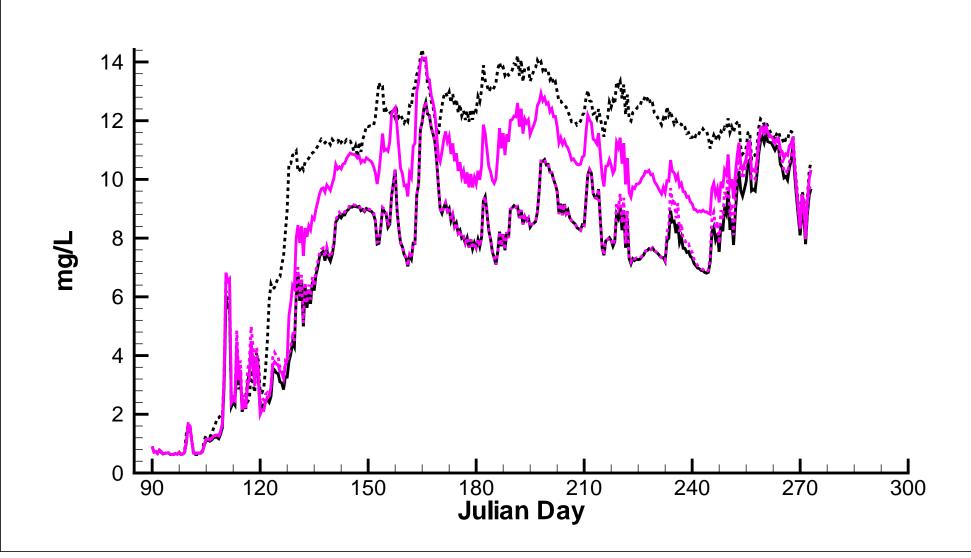


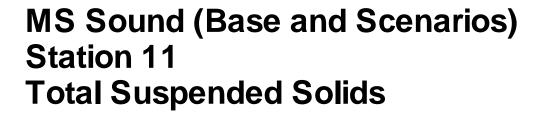


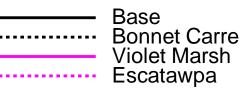


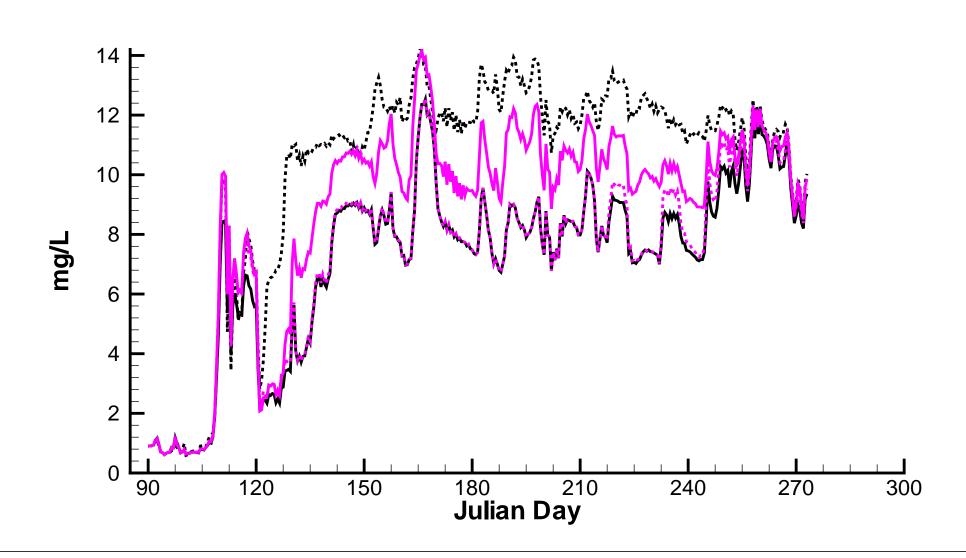






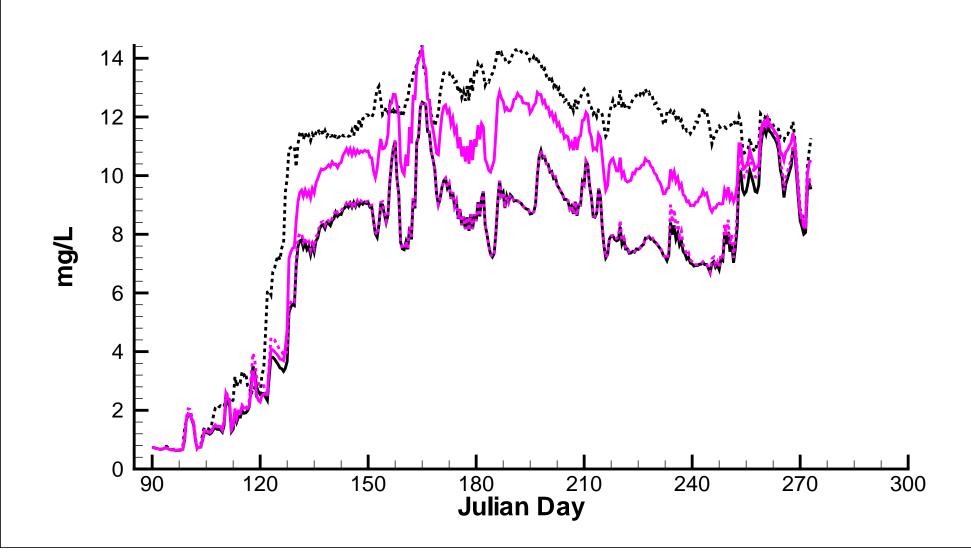


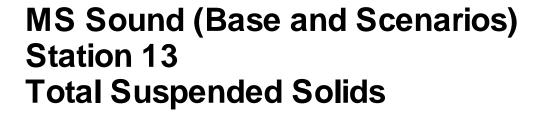




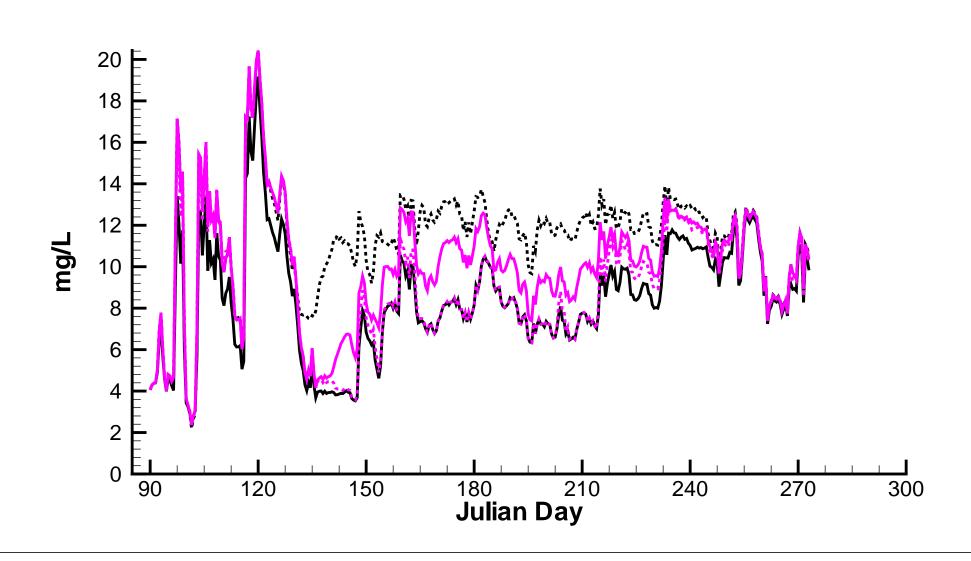


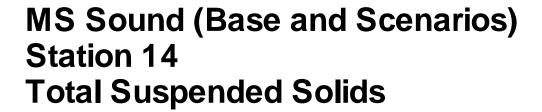


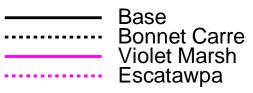


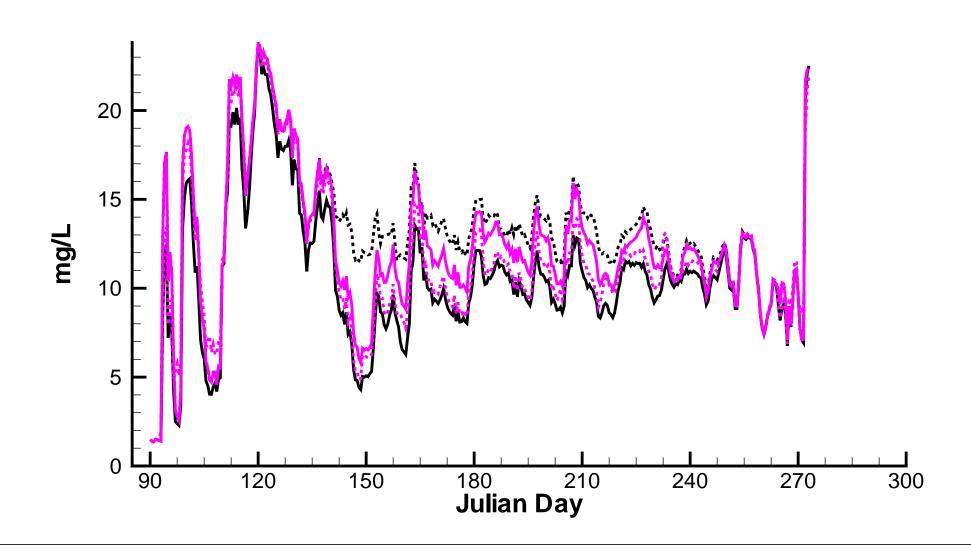


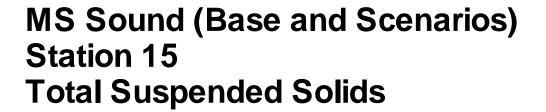




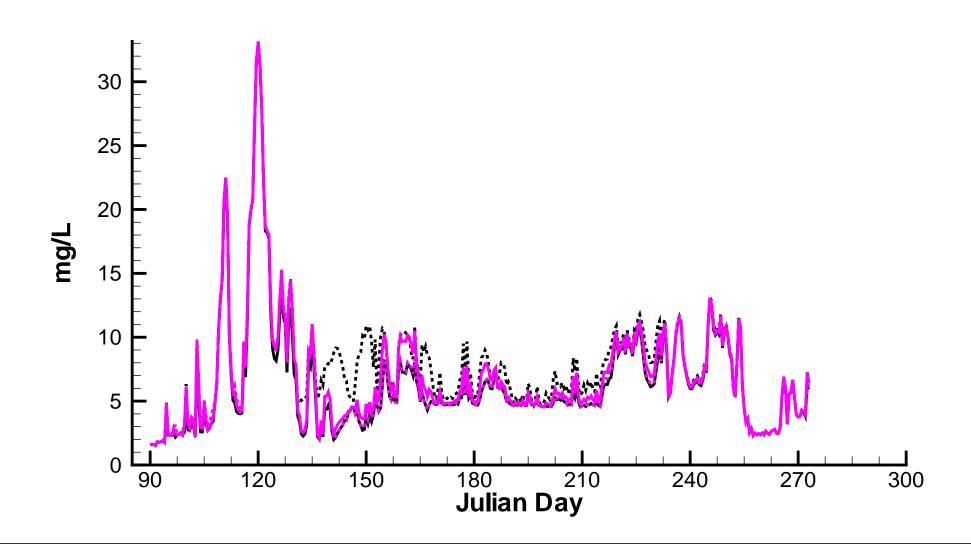


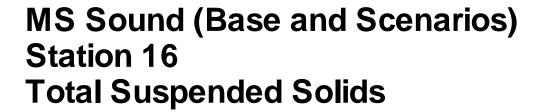




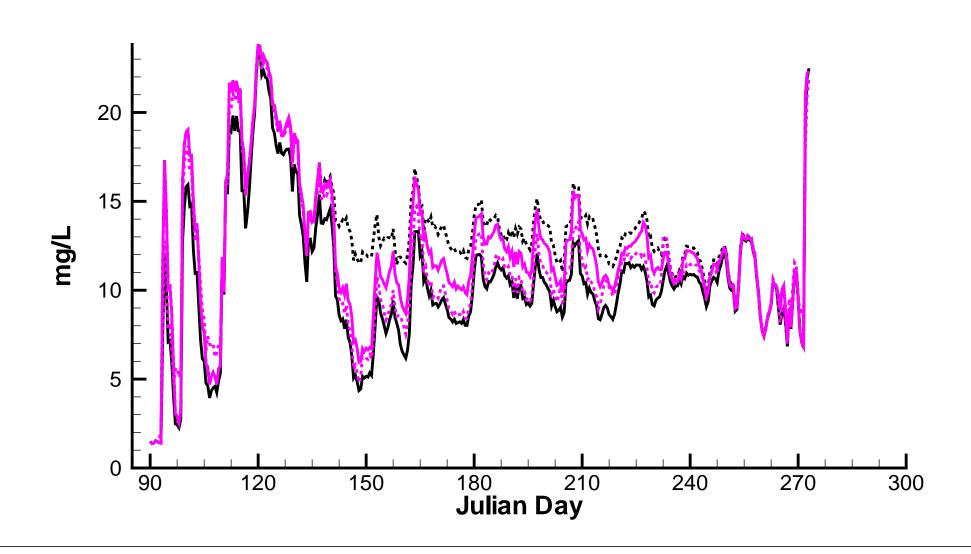


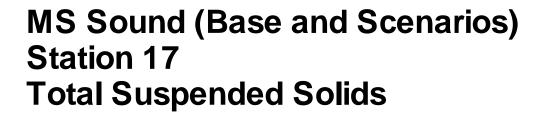


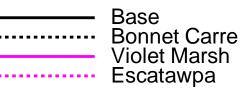


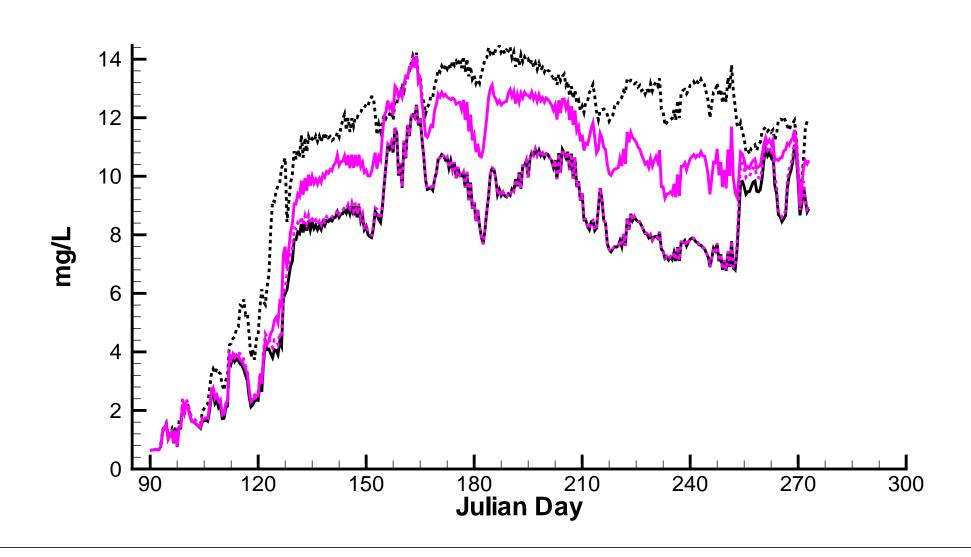


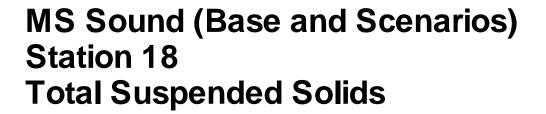


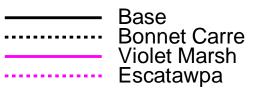


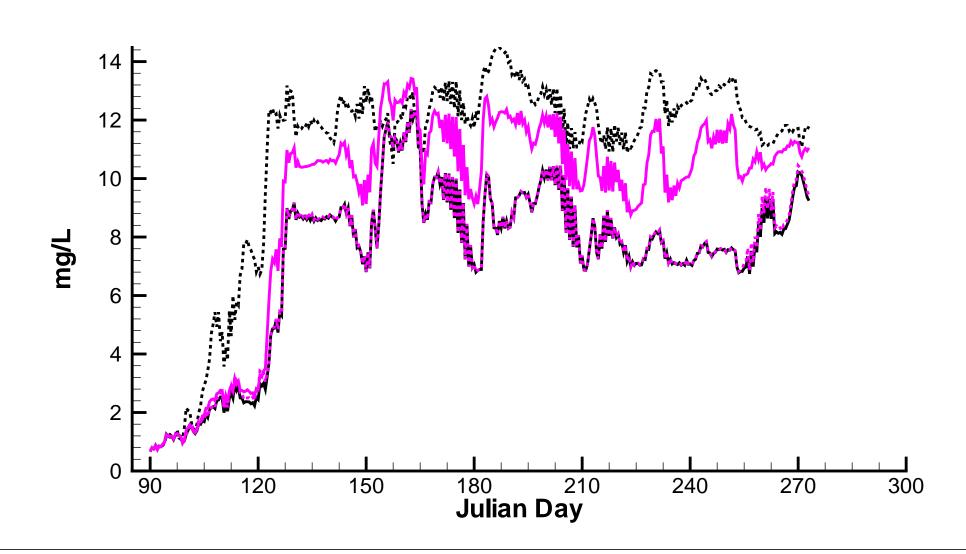


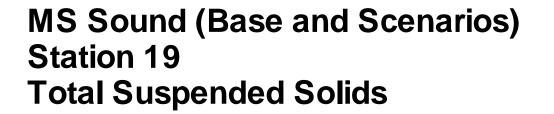


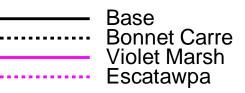


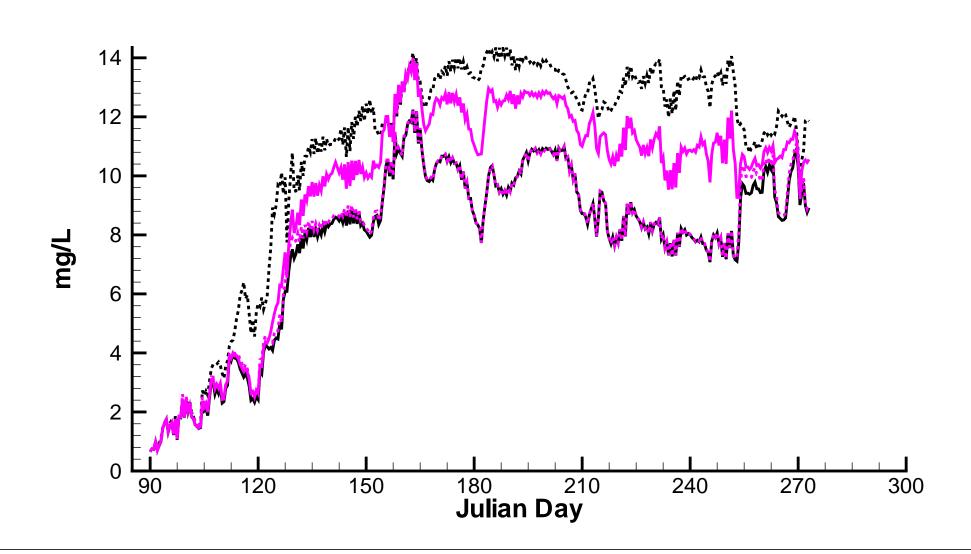




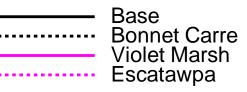


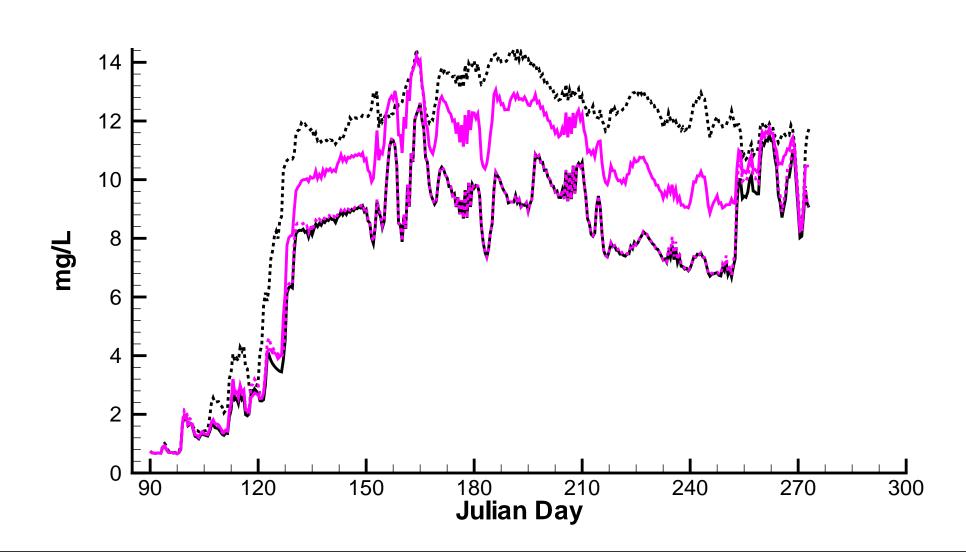






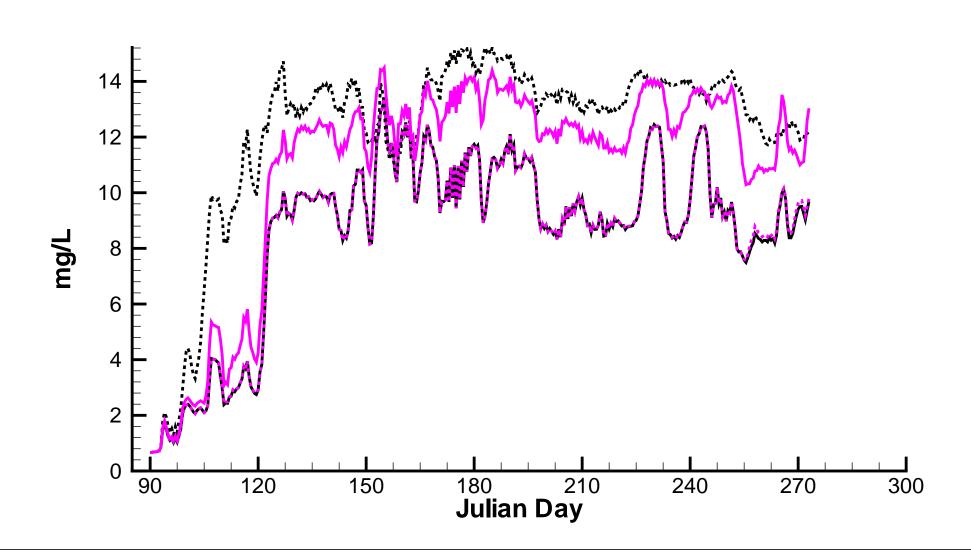






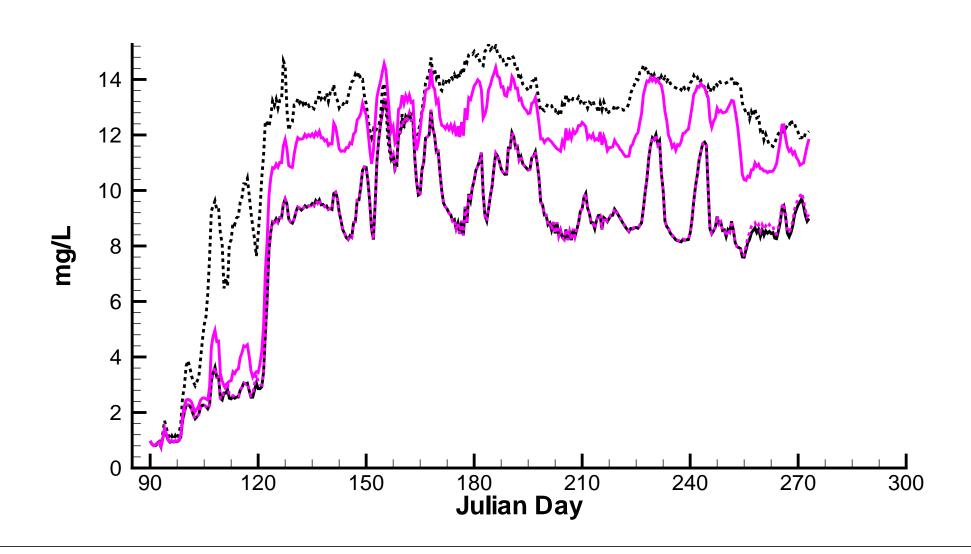
MS Sound (Base and Scenarios)
Station 21
Total Suspended Solids

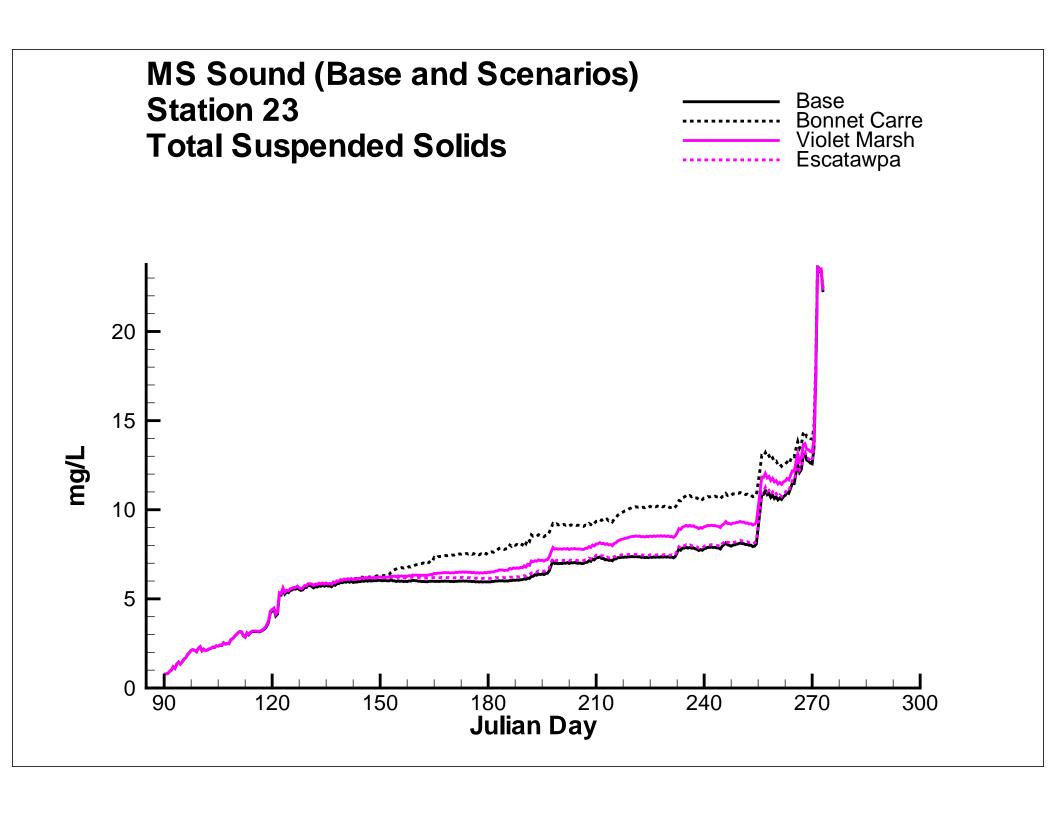


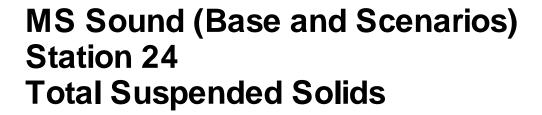


MS Sound (Base and Scenarios)
Station 22
Total Suspended Solids

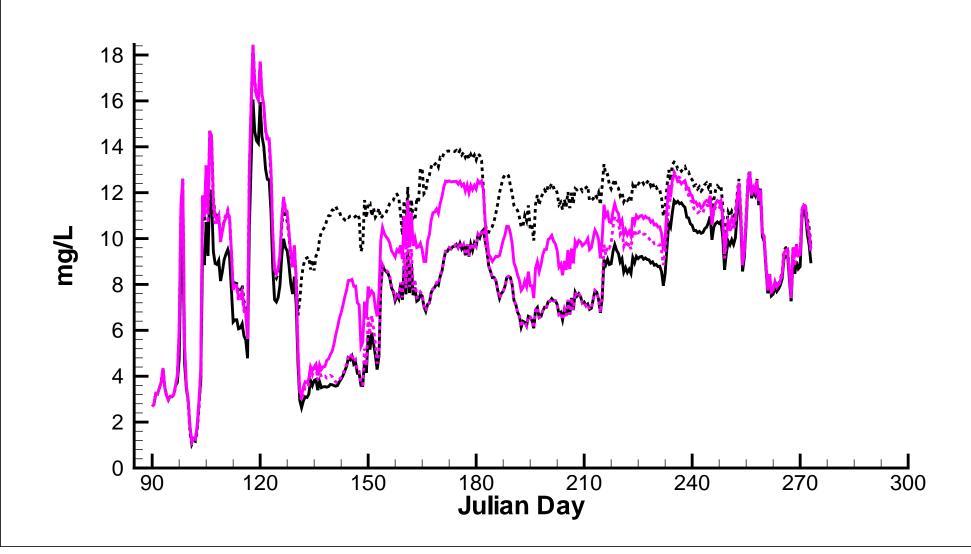


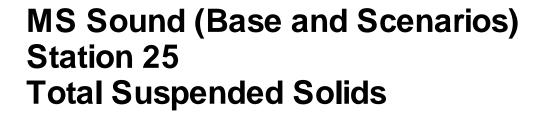


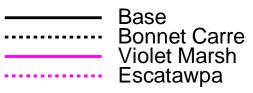


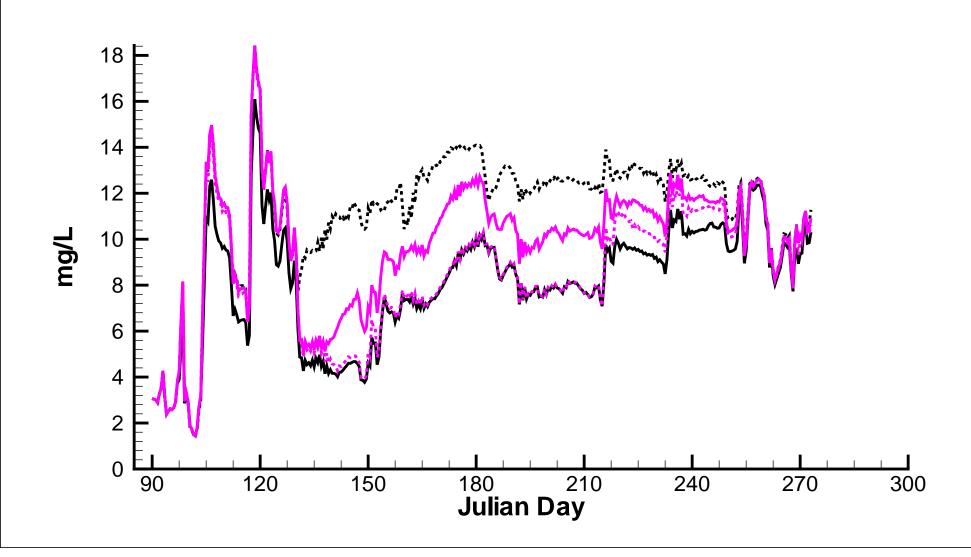






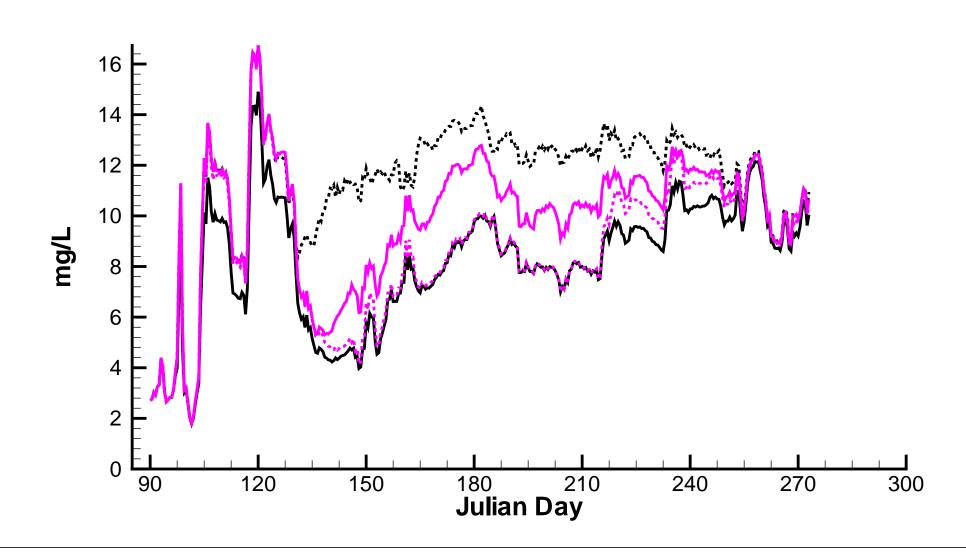






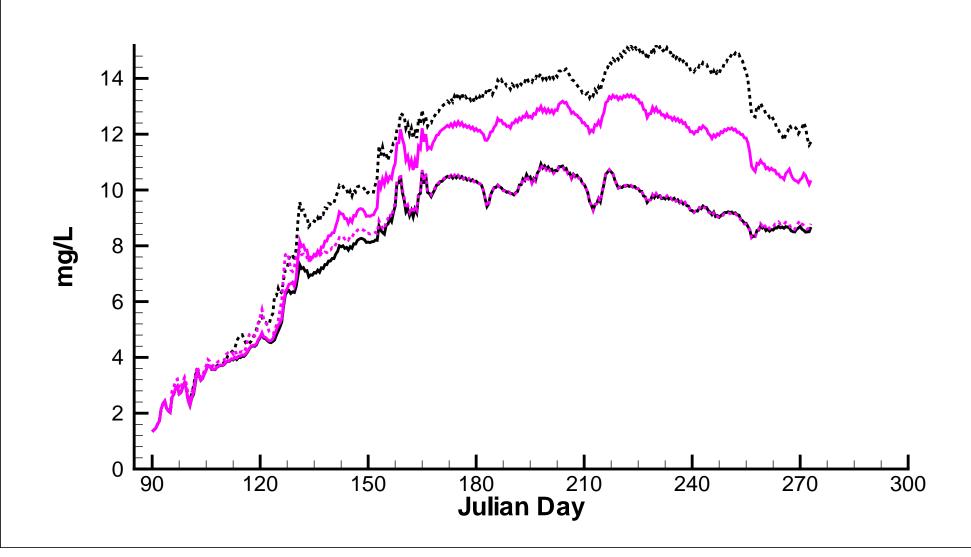




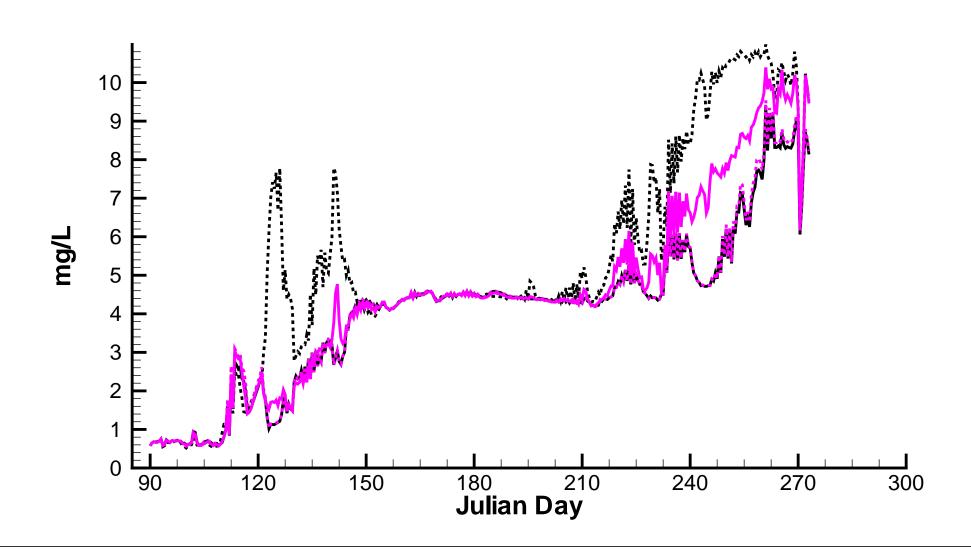


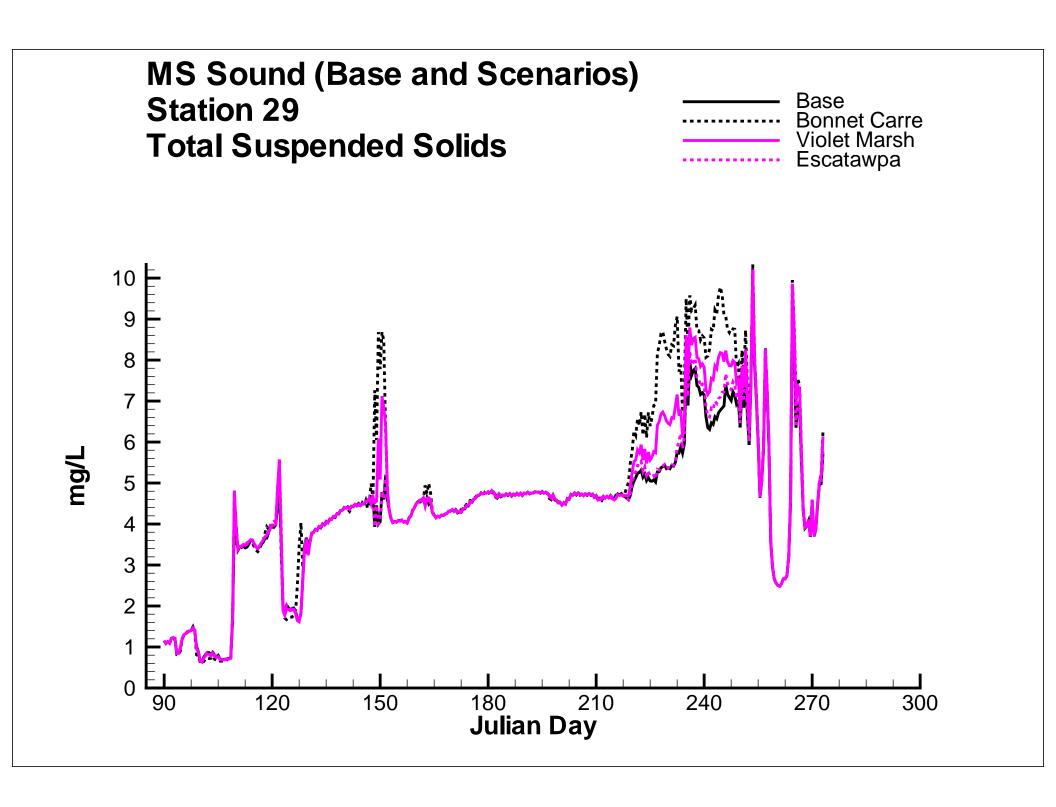


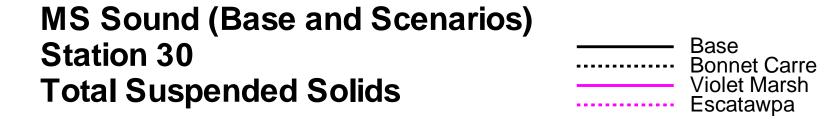


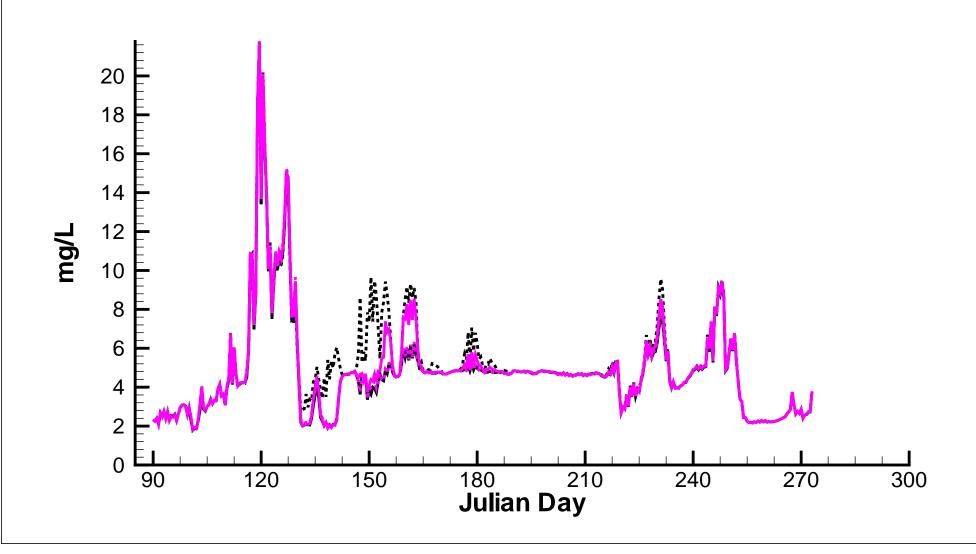


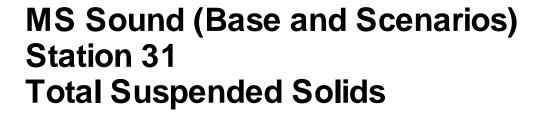




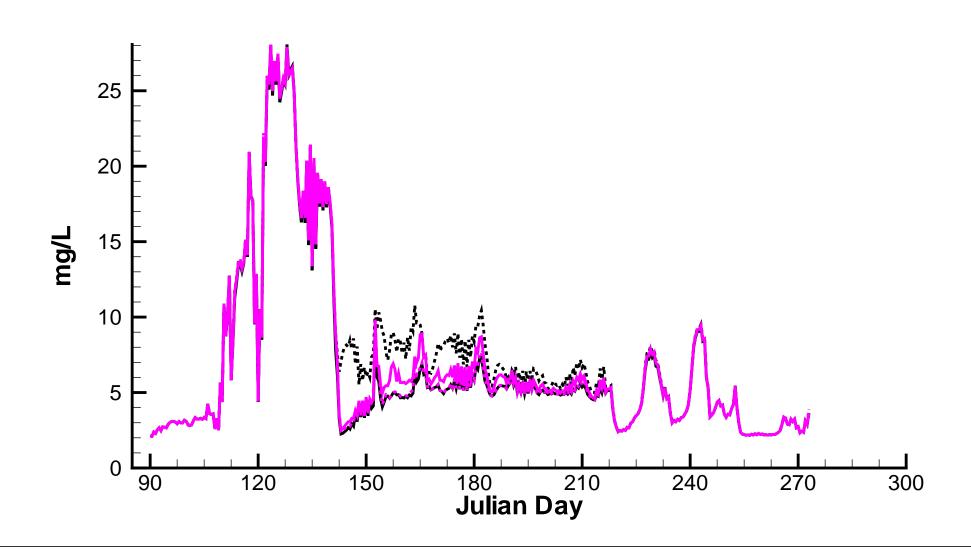














Base

240

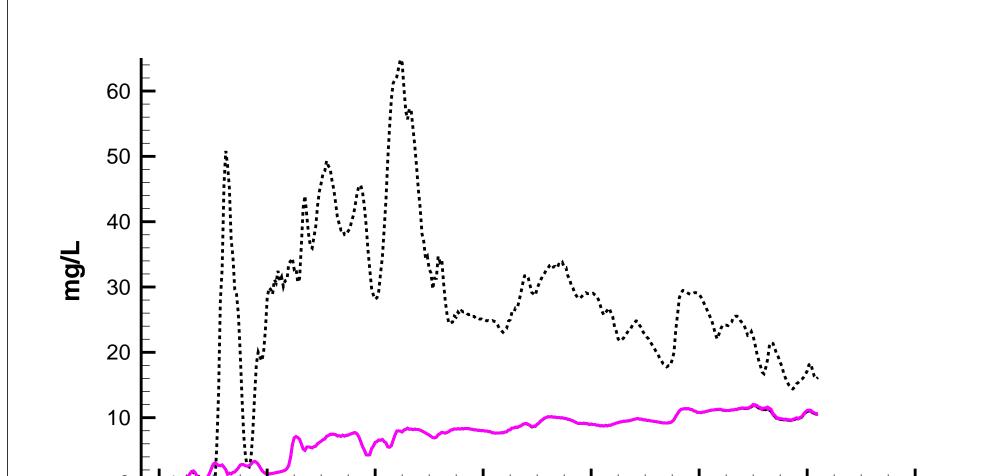
270

210

Bonnet Carre Violet Marsh

300

Escatawpa



180

Julian Day

150

120

