Delaware Department of Natural Resources
and Environmental Control

MURDERKILL RIVER WATERSHED TMDL MODEL DEVELOPMENT AND CALIBRATION

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Without all of their important contributions to the overall project, the models developed for the Murderkill River Watershed and presented in this report would not be possible.

SECTION 1

INTRODUCTION

The Murderkill River watershed is situated in the southeastern portion of Kent County in Delaware and includes several main tributaries (Double Run, Spring Creek, Browns Branch) and five lakes/ponds (McGinnis Pond, Andrews Lake, Killen Pond, Coursey Pond, McColley Pond). The river has tidal reaches from its mouth at Bowers Beach upstream for approximately 13 miles to locations just downstream from the pond/lake dams and near Barretts Chapel Road on Double Run (DNREC Station 206561). At Bowers Beach, the Murderkill River connects to Delaware Bay. The river is bounded by the St. Jones River watershed to the north and the Mispillion River watershed to the south. There are large tidal marshes surrounding the tidal portion of the river from Bowers Beach upstream to near Route 1. Figure 1 presents a study area map of the Murderkill River watershed. Historical water quality monitoring conducted by the Delaware Department of Natural Resources and Environmental Control (DNREC) has shown that waters in the tidal portions of the Murderkill River do not meet their designated uses because of low dissolved oxygen (DO) levels that are below State water quality standards. Based on these DO violations, DNREC has listed the tidal segments of the Murderkill River on the State's 303(d) list of impaired waters that requires the development of a Total Maximum Daily Load (TMDL) to bring the river into compliance with State water quality standards. In 2001, DNREC completed development of a water quality model of the Murderkill River and used it to propose TMDLs for sources of oxygen consuming compounds and nutrients in the watershed. This 2001 TMDL was amended by DNREC in 2005 (DNREC, 2005).

Since the development of the original Murderkill River Watershed TMDL in 2001, significant additional monitoring, modeling and related studies have been completed that have advanced the science and understanding of the water quality dynamics in the river. This effort has been coordinated through the activities of the Murderkill Study Group through the leadership of DNREC and the Kent County Department of Public Works (KCDPW). Members of this Study Group that have been involved in the new research and development include: DNREC; KCDPW; University of Delaware; United States Geological Survey (USGS); Delaware Geological Survey (DGS); University of Maryland; Stroud Water Research Center; Academy of Natural Science; and HDR|HydroQual. The purpose of these additional efforts was to refine the original TMDL based on the development of site-specific water quality standards for DO and nutrients in the tidal portion of the river only. The TMDL and associated allocations for the upstream watershed areas will remain the same as determined in the amended 2005 TMDL.

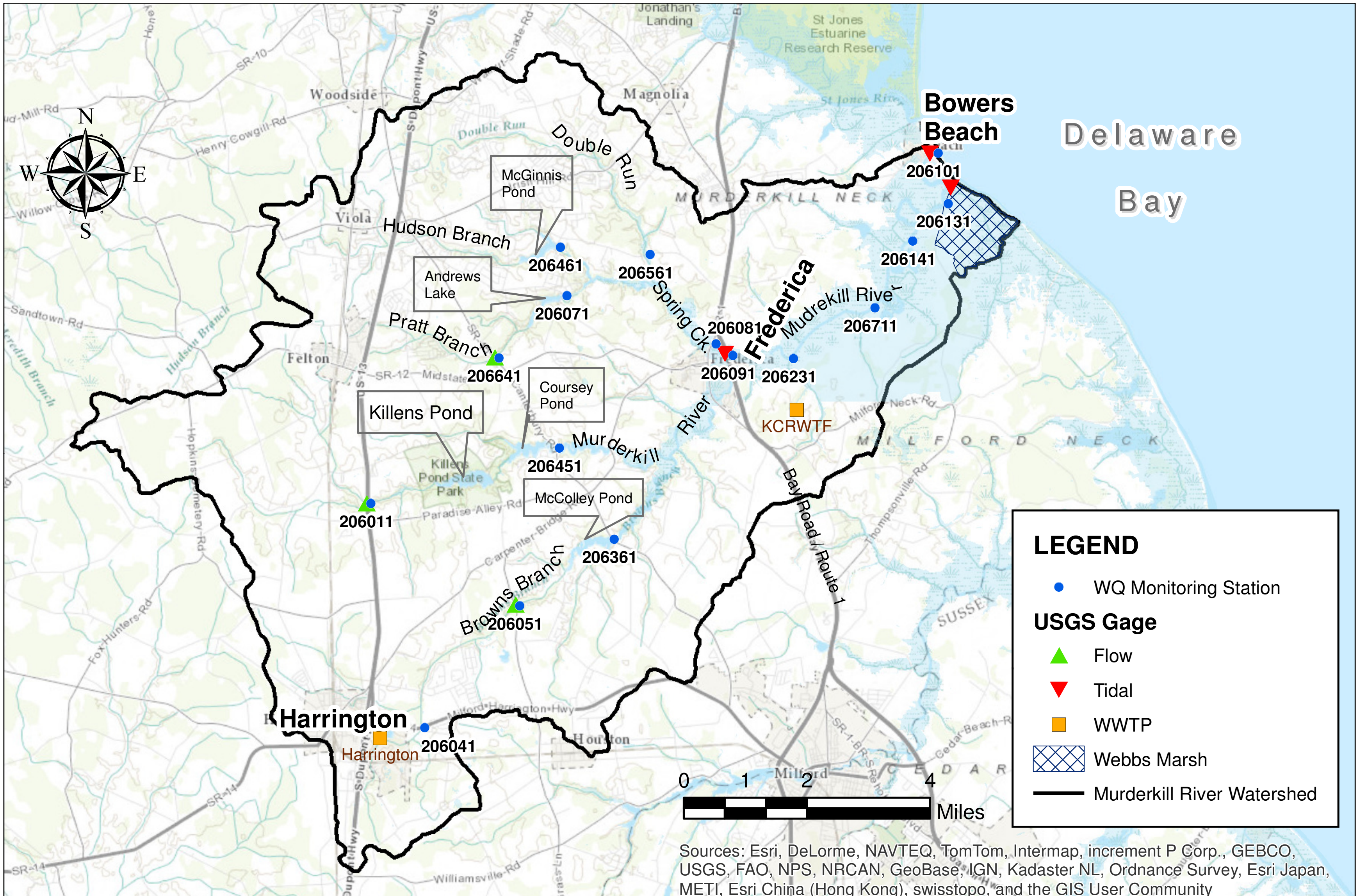

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Figure 1. Murderkill River Watershed Study Area

The principal goal of the collaborative Murderkill River Study was to plan and implement a comprehensive monitoring effort for quantifying the impact of tidal marshes and other natural resources on water quality in the tidal portions of the Murderkill River along with the development of improved watershed, hydrodynamic and water quality models. The following study specific monitoring efforts were identified and completed as part of the study.

- Water column primary production surveys were completed by Dr. Jonathan Sharp from the University of Delaware monthly from April 2007 to December 2008 (Sharp, 2011).
- Water, salt and nutrient balances in the Webb's Marsh were completed by Dr. William Ullman (University of Delaware) and Dr. Anthony Aufdenkampe (Stroud Water Research Center) during five surveys in 2007 and 2008 (Wong et al., 2009; Dzwonkowski et al., 2013; Ullman et al., 2013).
- Nutrient flux studies in the main river channel and tidal marsh were completed by Chesapeake Biogeochemical Associates in 2007 and 2008 (CBA, 2010).
- Characterization of the spatial and temporal inundation of the tidal marshes was completed in 2010 by Dr. Thomas E. McKenna from the University of Delaware (McKenna, 2013).
- Continuous tidal monitoring for salinity, temperature, DO, pH, water elevation and volume flux was completed by the USGS in the tidal Murderkill River near Frederica and at Bowers Beach.
- Vertical profiling of sediment cores in the tidal river was completed by the Academy of Natural Sciences in 2010 (Velinsky, et al., 2010).
- Installation of three stream gaging stations to monitor flow in the watershed by the USGS on the Murderkill River, Pratt Branch and Browns Branch.

In addition, increased sampling in the Murderkill River watershed by DNREC was also completed for this study along with the completion of long-term BOD studies on river samples and Kent County Regional Wastewater Treatment Facility (KCRWTF) effluent. The DNREC sampling frequency was increased to bi-weekly or monthly with the addition of a few additional monitoring locations. The data collected from these studies were used to develop improved watershed, hydrodynamic and water quality models that were eventually used to support site-specific criteria development for the tidal Murderkill River. Table 1 presents a summary of the data collected in the Murderkill River watershed and used in this modeling study.

Table 1. Summary of Available Monitoring Data (2007-2008 Period)		
Station Name	**Station Agency/Number**	**Available Data**
Murderkill River at Black Swamp Creek at Rte. 13	DNREC / 206011	Water Quality, LTBOD
Browns Branch at Milford-Harrington Hwy. (Rte. 14)	DNREC / 206041	Water Quality, LTBOD
Browns Branch at Killen Pond Rd. (Rd. 384)	DNREC / 206051	Water Quality, LTBOD
Pratt Branch at Canterbury Rd. (Rte. 15)	DNREC / 206641	Water Quality[1], LTBOD
Double Run at Barretts Chapel Rd.	DNREC / 206561	Water Quality[1], LTBOD
Andrews Lake at Andrews Lake Rd. (Rd. 380)	DNREC / 206071	Water Quality[1], LTBOD
McColley Pond at Canterbury Rd. (Rte. 15)	DNREC / 206361	Water Quality[1], LTBOD
Coursey Pond at Canterbury Rd. (Rte. 15)	DNREC / 206451	Water Quality[1], LTBOD
McGinnis Pond at McGinnis Pond Rd. (Rd. 378)	DNREC / 206461	Water Quality[1], LTBOD
Spring Creek at Frederica Rd. (Rte. 12)	DNREC / 206081	Water Quality[1], LTBOD, Algal Production
Murderkill River at Bay Rd. (Rte. 1/113)	DNREC / 206091	Water Quality[1], LTBOD, Algal Production
Murderkill River at KCRWTF Canal	DNREC / 206231	Water Quality[1], LTBOD, Algal Production
Murderkill River near Powerlines	DNREC / 206711	Water Quality[1], LTBOD, Algal Production
Murderkill River near Milford Neck Wildlife Area Levee	DNREC / 206141	Water Quality[1], LTBOD, Algal Production
Murderkill River at Webb Landing	DNREC / 206131	Water Quality[1], LTBOD, Algal Production
Murderkill River at Bowers Beach	DNREC / 206101	Water Quality[1], LTBOD, Algal Production
Murderkill River near Webb Landing	CBA / MK01	Sediment Fluxes[2]
Murderkill River near Powerlines	CBA / MK02	Sediment Fluxes[2]

Table 1. Summary of Available Monitoring Data (2007-2008 Period)		
Station Name	**Station Agency/Number**	**Available Data**
Murderkill River near KCRWTF Canal	CBA / MK03	Sediment Fluxes[2]
Murderkill River downstream from Bay Rd.	CBA / MK04	Sediment Fluxes[2]
Murderkill River near Felton	USGS / 01484000	Flow
Browns Branch near Harrington	USGS / 01484018	Flow
Pratt Branch near Felton	USGS / 01484050	Flow
Murderkill River at Bowers Beach	USGS / 01484085	Salinity, Temperature, Elevation, Flow/Velocity
Murderkill River at Frederica	USGS / 01484080	Salinity, Temperature, Elevation, Flow/Velocity
Webb's Marsh	USGS / 01484084	Salinity, Temperature, Elevation, Flow/Velocity

1 – Nutrients (N & P), chlorophyll-a, BOD_5, carbon, secchi depth, salinity, temperature and DO
2 – Sediment oxygen demand, ammonium/nitrate/phosphorus fluxes, denitrification

1.1 MURDERKILL RIVER MODELING FRAMEWORK

The Murderkill River modeling framework is comprised of three components: a watershed model; a hydrodynamic model; and a water quality model. The watershed model characterizes watershed processes in the watershed such as rainfall driven runoff and nonpoint source loadings including freshwater stream and lake/pond water quality interactions. The hydrodynamic model simulates the tidal movement of water due to tides and freshwater flow, density driven currents, and meteorology confined by a realistic representation of the systems bathymetry and also calculates salinity and temperature. The water quality model calculates nutrient mediated algal growth and death, DO, the various organic and inorganic forms of nitrogen, phosphorus, silica, and carbon (or BOD). In addition, the water quality model includes a sediment flux sub-model to calculate sediment oxygen demand (SOD) and sediment nutrient fluxes as a function of settling particulate organic matter (POM) and sediment diagenesis. Tidal marsh interactions were also included as loading functions based on the nutrient balance studies in Webb's Marsh.

The watershed model used in the study is the Hydrologic Simulation Program FORTRAN (HSPF) that is available with USEPA's multi-purpose BASINS package. It uses rainfall, temperature and solar radiation information, land use patterns, and land management practices to simulate the quantity and quality of runoff from multiple land use watersheds (e.g., urban, agricultural, forest). The model results provide runoff flow and nonpoint source loadings to the hydrodynamic and water quality models at the five main freshwater input locations to the tidal river (Double Run, McGinnis Pond, Andrews Lake, Coursey Pond and McColley Pond) and incrementally along the length of the tidal river downstream to near the Route 1 Bridge.

The hydrodynamic model used in the study is the three-dimensional, time-dependent, estuarine and coastal circulation model, Estuary and Coastal Ocean Model (ECOMSED), which has been successfully applied in numerous studies. The water quality model used in the study is a state-of-the-art eutrophication model Row Column Aesop (RCA), which is very similar to the WASP model, and is directly coupled with the hydrodynamic model, allowing computation of water quality within the tidal cycle. In addition, a sediment flux sub-model is also included in the water quality model to allow calculation of SOD and sediment nutrient fluxes in response to settled organic matter and its subsequent decay in the sediment. This coupled hydrodynamic/water quality model has been successfully applied in numerous studies including: St. Jones River, Blackbird Creek, Leipsic River, Smyrna River, Little River and Broadkill River (DE); Delaware River (NJ/PA/MD/DE); South Atlantic Bight (NY/NJ); Jamaica Bay (NY); Hudson-Raritan Estuary (NY/NJ); Long Island Sound (NY/CT); Chesapeake Bay (MD/DE); Massachusetts Bay and Boston Harbor (MA); Upper Mississippi River (MN); San Joaquin River (CA); Tar-Pamlico Estuary (NC); Escambia/Pensacola Bay, Fenholloway River and St. Andrews Bay (FL).

The watershed, hydrodynamic and water quality models were calibrated and validated with data collected by DNREC and USGS over the 2007-2008 monitoring period. The year 2007 was considered as the calibration and year 2008 as the validation with a consistent set of model parameters developed that best represented the observed data. These data include ADCP data, temperature, salinity and water quality (nitrogen, phosphorus, carbon, DO, chlorophyll-a) data throughout the Murderkill River watershed. The calibrated and validated watershed, hydrodynamic and water quality models resulted in a reasonable representation of both the complex mixing and circulation patterns observed in the study area and the observed nutrient, phytoplankton, organic carbon, and DO dynamics of the system. The linked watershed, hydrodynamic and water quality models were developed to support continued TMDL and site-specific criteria development in the Murderkill River watershed.

SECTION 2

WATERSHED MODEL (HSPF)

The watershed model used in the study is USEPA's Hydrologic Simulation Program FORTRAN (HSPF) model. The HSPF model uses meteorological conditions (precipitation, evapotranspiration, air temperature and solar radiation) and land use data to simulate flow, sediment transport, temperature variations, and water quality processes over the entire hydrologic cycle. The model can represent the processes that control watershed runoff quantity and quality and in-stream (river tributary and lake) flow and water quality dynamics.

The HSPF model was delineated into 28 sub-watersheds in the Murderkill River watershed and Figure 2 presents the final HSPF watershed model segmentation. Several factors were considered in the delineation of sub-watersheds including monitoring stations, location of lakes and tributary watersheds. Preliminary model segment delineation was performed based on the Digital Elevation Model (DEM) grid and National Elevation Dataset (NED) data. Further refinement of the model segmentation was then completed by using the location of the water quality stations, lakes/ponds and flow gages along with final watershed delineation as provided by USGS based on DEMs from 2007 Statewide Light Detection and Ranging (LiDAR) results. HSPF model inputs are then defined for each of the 28 sub-watersheds. In each model sub-watershed, multiple land use types and different model parameters can be applied. Stream geometry in the model is represented as a set of functional relationships to flow between variables, such as stream surface area, volume and velocity.

The model is calibrated for flow at three USGS flow gages and for water quality at eight DNREC water quality monitoring stations. The calibrated and validated HSPF model output (quantity and quality) is then transferred to the hydrodynamic and water quality models as boundary conditions and loadings to calculate water circulation and water quality in the tidal reaches of the Murderkill River. The HSPF model output is provided to the hydrodynamic and water quality models at the five main freshwater input locations to the tidal river (Double Run, McGinnis Pond, Andrews Lake, Coursey Pond and McColley Pond) and incrementally along the length of the tidal river downstream to near the Route 1 Bridge.

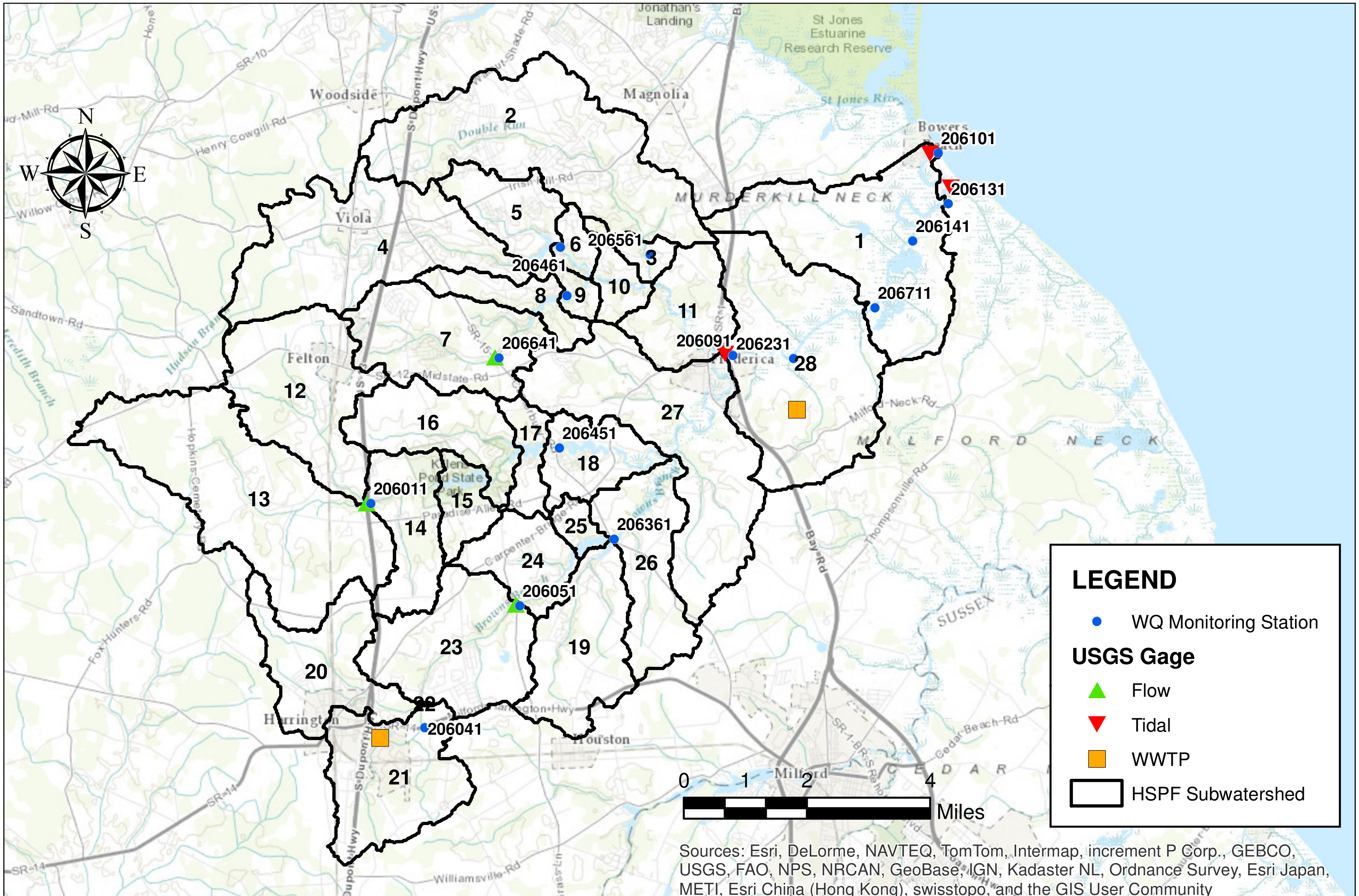

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Figure 2. Murderkill River Watershed HSPF Segmentation

2.1 WATERSHED CHARACTERISTICS

2.1.1 Land Use and Watershed Delineation

The land use information is an essential piece of the watershed model input. The land use information for the year 2007 was obtained from DNREC. As shown in Figure 3 and Table 2, the Murderkill River watershed is primarily non-urban (85%) with approximately 55% agricultural land use. The 2007 land use information included 47 categories which were regrouped into 9 categories for use in the HSPF model setup. Land use areas for 6 major sub-watersheds are presented in Figure 4. Table 3 presents the sub-watershed reach properties for length, elevation drop, downstream sub-watershed and whether a stream or lake. It should be noted that the HSPF model setup did not include Reach 1 (most downstream reach near Bowers Beach) because of instability issues and also because these marsh loads were assigned directly to the tidal water quality model based on the Webb's Marsh nutrient studies. Land use information for the HSPF model sub-watersheds are contained in Appendix 1.

Table 2. Murderkill River Watershed Land Use Summary

Land Use Type	Area (acres)	Area (mi^2)	% of Total
Agriculture	34,237	53.5	55.4%
Wetlands	8,949	14.0	14.5%
Residential	7,779	12.2	12.6%
Forest	7,077	11.1	11.4%
Urban	1,117	1.7	1.8%
Water	1,088	1.7	1.8%
Transitional	601	0.9	1.0%
Pasture	519	0.8	0.8%
Roadways	273	0.4	0.4%
CAFO	183	0.3	0.3%
Total	61,824	96.6	

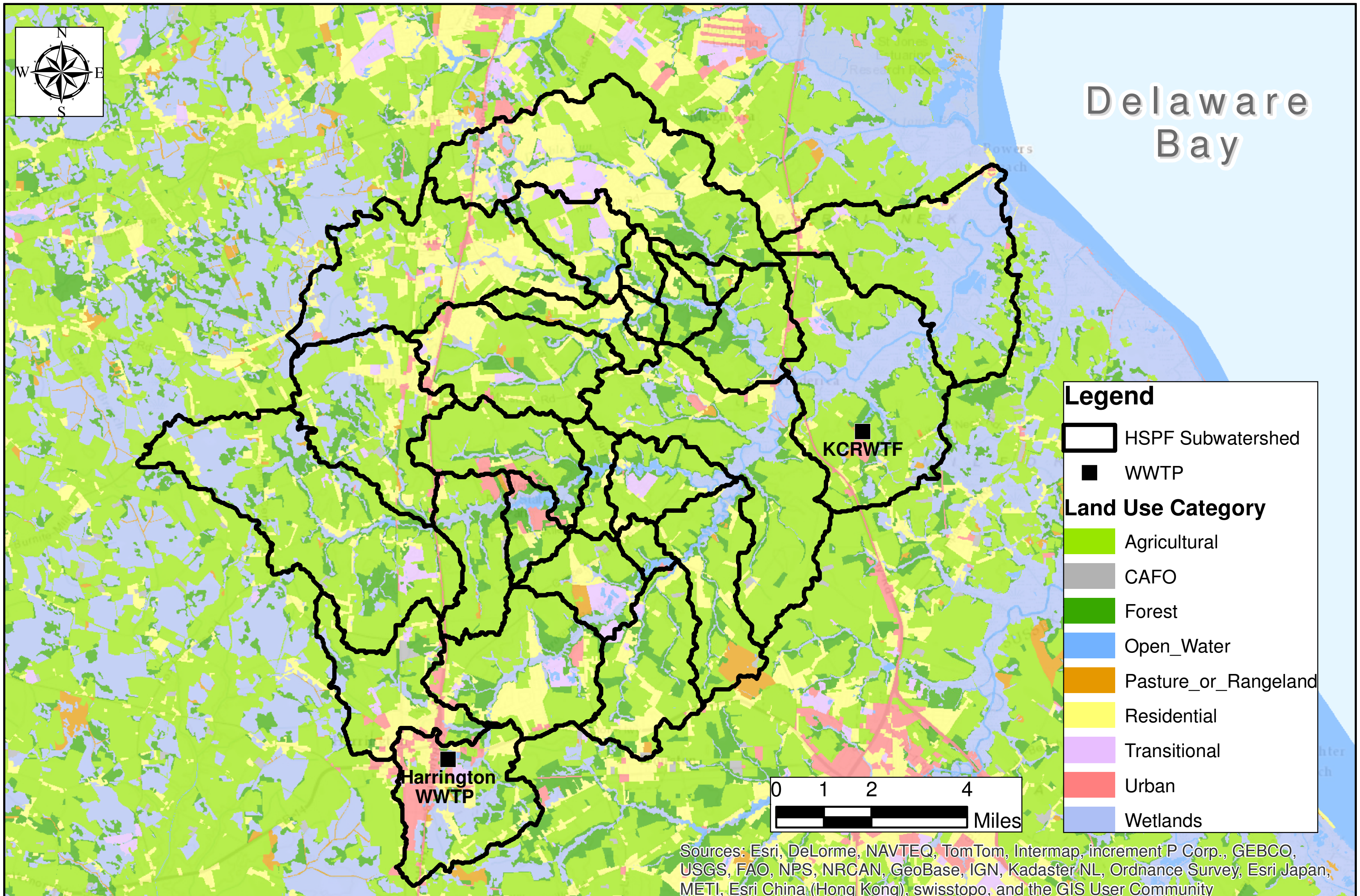

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Figure 3. Murderkill River Watershed 2007 Land Use

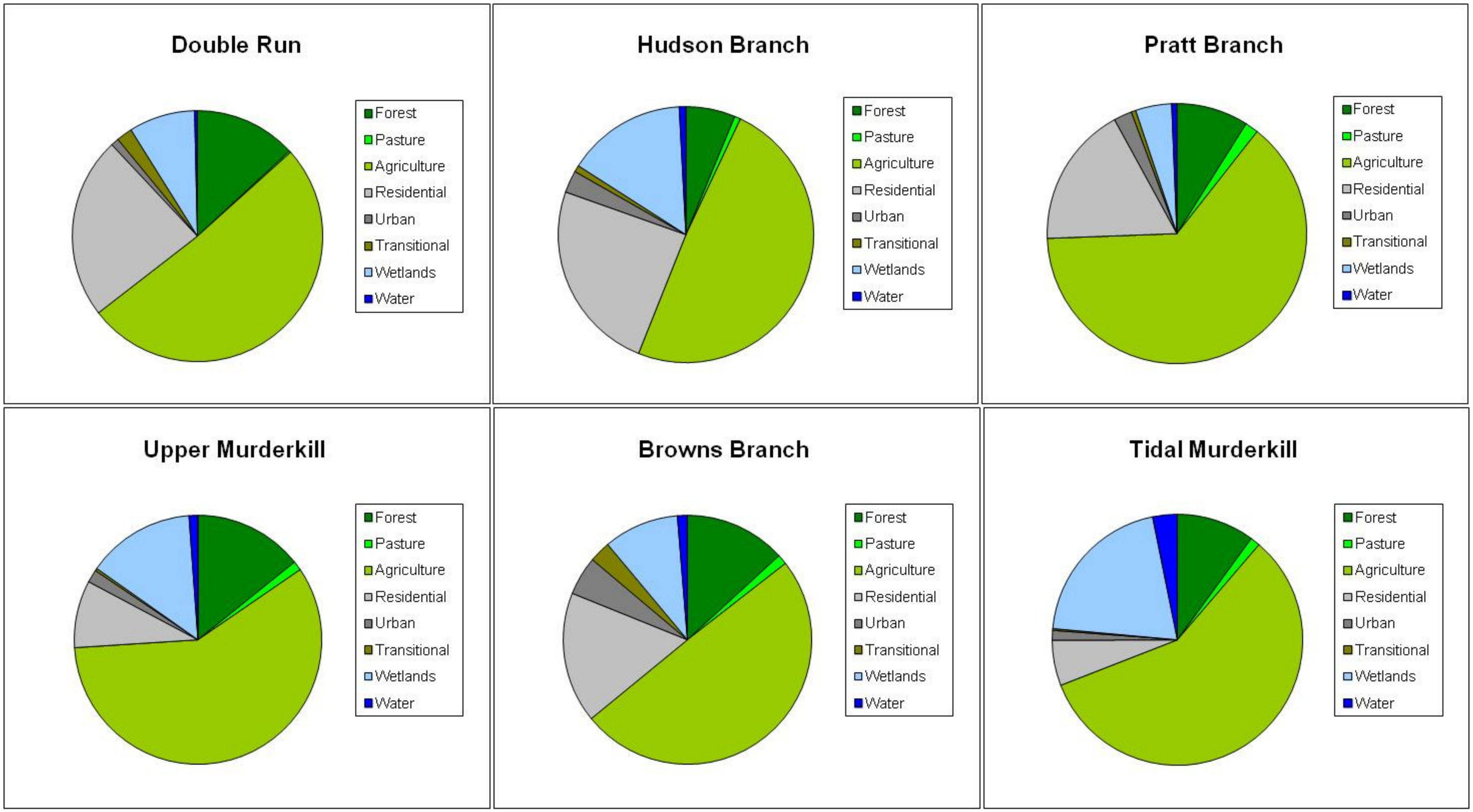

Figure 4. Murderkill River Watershed 2007 Land Use for Major Sub-watersheds

Table 3. HSPF Model Reach Properties				
Sub-watershed ID	**Length (miles)**	**Elevation Drop (ft)**	**Downstream Sub-watershed**	**Stream/Lake**
2	3.56	25.62	3	Stream
3	0.76	0.26	11	Stream
4	3.46	41.75	5	Stream
5	0.65	0.26	6	Lake
6	1.00	9.58	10	Stream
7	0.99	16.40	8	Stream
8	0.60	1.18	9	Lake
9	0.74	5.02	10	Stream
10	1.09	1.02	11	Stream
11	2.17	0.43	28	Stream
12	1.59	19.19	14	Stream
13	3.59	22.27	14	Stream
14	1.46	13.48	15	Stream
15	0.93	1.67	16	Lake
16	0.86	9.54	17	Stream
17	0.49	0.10	18	Lake
18	2.30	4.00	27	Stream
19	0.59	8.50	26	Stream
20	1.33	10.96	23	Stream
21	0.58	5.87	23	Stream
23	2.65	22.7	24	Stream
24	1.48	9.97	25	Stream
25	0.68	2.79	26	Lake
26	1.94	6.72	27	Stream
27	2.96	0.75	28	Stream
28	3.26	2.95	0	Stream

2.1.2 Rainfall and Runoff

Rainfall data is required as part of the HSPF model input and was obtained from a meteorological gage at the Delaware National Estuarine Research Reserve (DNERR) St. Jones Reserve in Dover, DE. The HSPF model requires hourly climatic data including hourly precipitation, evapotranspiration, dew point, photosynthetically active radiation (PAR), dew point and air temperature. Hourly meteorological data from the DNERR St. Jones Reserve station was used and was the closest station to the Murderkill River watershed. The annual total rainfall in 2007 was 40.0 inches and in 2008 was 40.6 inches, which is below the average annual rainfall total of about 45 inches.

Stream flow data are also required as part of the HSPF model input. There are three major tributaries in the Murderkill River watershed: Double Run; Spring Creek including Hudson Branch and Pratt Branch; and Browns Branch. Flow data are available at the following three freshwater USGS gages: Murderkill River near Felton (#01484000); Browns Branch near Harrington (#01484018); and Pratt Branch near Felton (#01484050). Figure 5 presents the rainfall and flow data for the calibration/validation period years of 2007-2008. A summary of the flow data is also presented in Table 4 for the 2007-2008 modeling period. These gaged flows were extrapolated to the entire Murderkill River watershed area using a drainage area ratio and resulted in an average freshwater flow during the monitored 2007-2008 period of 72 cfs (ranging from 14 to 940 cfs). The average runoff coefficient was 0.74 cfs/mi^2 and the summer average runoff coefficient was 0.43 cfs/mi^2. The watershed upstream from the Murderkill River gage appeared to produce less runoff than the other two gages (based on the runoff coefficients), particularly when compared to the Browns Branch gage that has a similar drainage area. This may be potentially due to the increased wetland areas upstream from the Murderkill River gage.

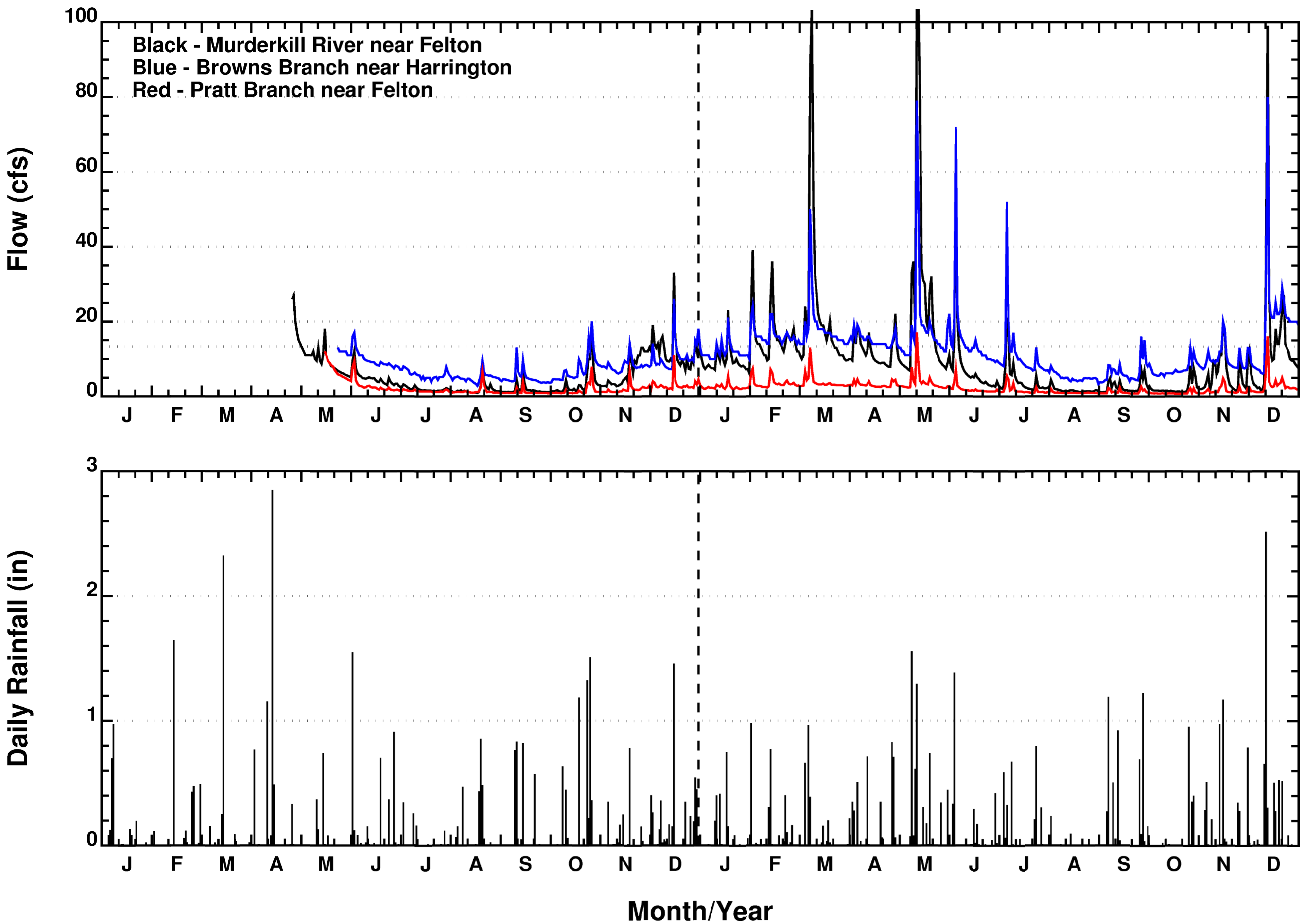

Figure 5. Murderkill River Watershed Gaged Flow and DNERR Rainfall (2007-2008)

Table 4. Watershed Freshwater Flow Summary				
Parameter	**Murderkill River**	**Browns Branch[3]**	**Pratt Branch**	**Watershed[2]**
Available Dates	4/27/2007-12/31/2008	5/25/2007-12/31/2008	5/17/2007-12/31/2008	
Drainage Area (mi^2)	12.9	12.5	2.9	97.7
Average (cfs)	8.7	10.2	2.3	74.5
Summer Average (cfs)[1]	3.4	7.2	1.6	44.3
Minimum (cfs)	0.9	2.1	0.7	16.5
Maximum (cfs)	212.0	79.3	17.0	942.4
Average Runoff (cfs/mi^2)	0.68	0.81	0.79	0.76
Summer Average Runoff (cfs/mi^2)	0.26	0.58	0.53	0.45

1 – Summer = June-September
2 – Gaged flow drainage area scaled to total watershed drainage area
3 – Browns Branch gage flow adjusted for average Harrington STP flow (0.43 MGD)

2.1.3 Watershed Stream Reach Geometry

The geometry information for all river reaches and lakes in the watershed is also required to setup the F-tables in the HSPF model. F-tables are used to define the geometry changes as a function of stream flow in the HSPF model and include information on depth, surface area and volume. Historical stream geometry, flow and velocity data were available at a number of the DNREC monitoring sites that were used to define stream geometry in the HSPF model. In addition, surface area maps and depth contours for the lakes were available from DNREC along with dam survey information collected as part of a flooding study completed by KCDPW (URS, 2009). This lake information was used to define the lake geometry in the HSPF model. Depth, width and velocity data from all the available stream sites were combined and regressions performed to develop relationships to flow. Figure 6 presents the regression results for geometry-flow relationships used to setup the HSPF F-tables for the stream reaches. Lake maps with total surface area, total volume and depth contours were available for McGinnis Pond, Andrews Lake, Killen Pond, Coursey Pond and McColley Pond. Weir information was also available from the dam survey that included weir top width, which were used to develop flow rating curves for the lakes.

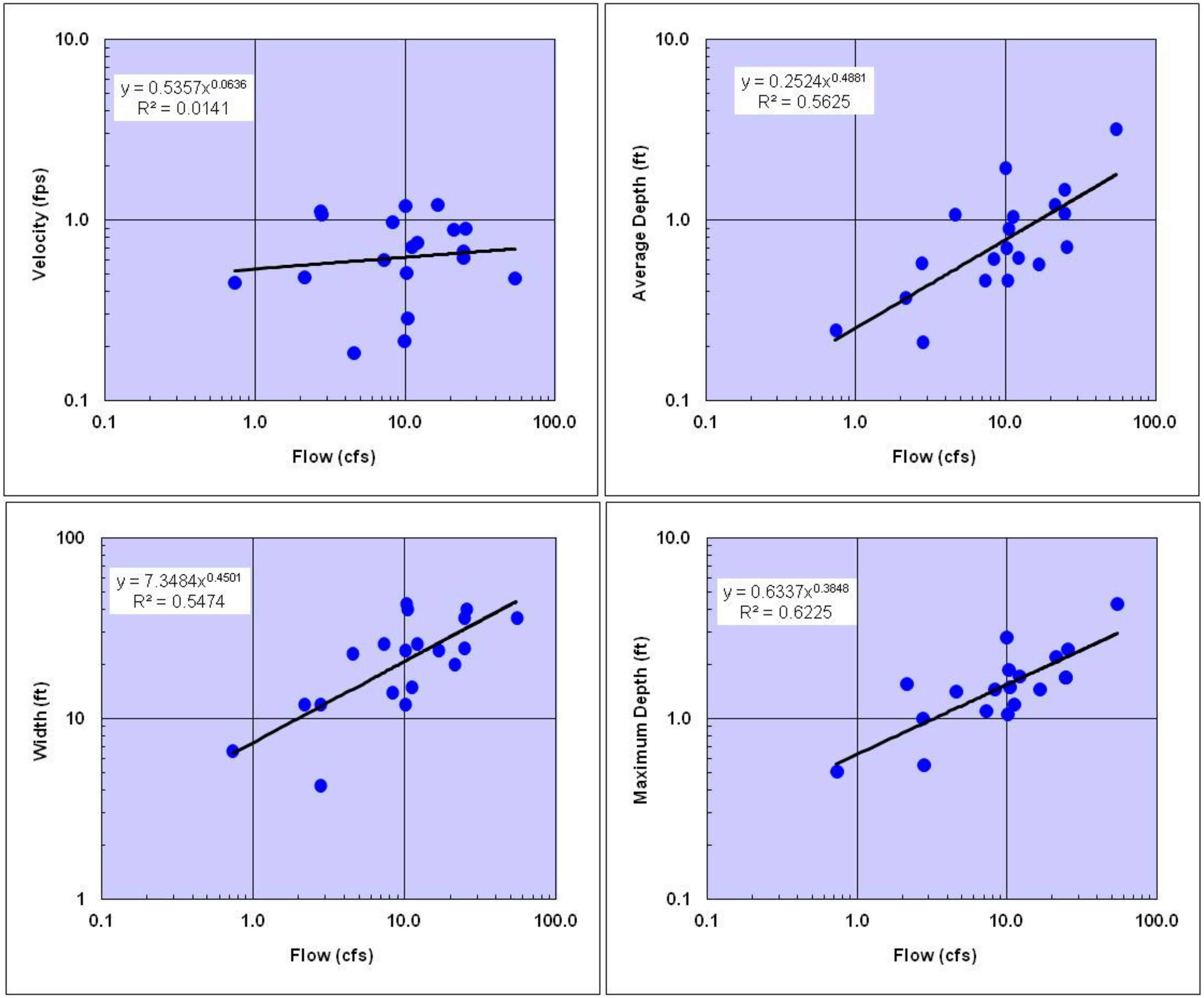

Figure 6. Stream Geometry vs. Flow Relationships used in HSPF F-Tables

2.1.4 Point Sources

In the Murderkill River watershed there were two active point sources during the 2007-2008 modeling period: KCRWTF and Harrington Sewage Treatment Plant (STP). There were two other point sources in the watershed (Canterbury Crossing and Southwood Acres Mobile Hone Park) that were eliminated before the 2007-2008 modeling period. The KCRWRF discharges into the tidal portion of the river and the Harrington STP discharges near the upstream end of Brown's Branch just east of the Town of Harrington. Given the locations of these two point sources, the KCRWRF flow and loads are assigned in the hydrodynamic/water quality model (ECOMSED/RCA) model and the Harrington STP flow and loads are assigned in the HSPF watershed model. Monthly flow, nutrient (nitrite plus nitrate (NO_2+NO_3), ammonia (NH_3), orthophosphate (PO_4)), DO, carbonaceous 5-day biochemical oxygen demand ($CBOD_5$), and total suspended solids (TSS) data are available for Harrington STP. The KCRWRF had similar effluent data available but it was available on a daily or weekly basis depending on the parameter. Table 5 presents the average flow and concentration data for each of the parameters from these point sources during the 2007-2008 modeling period. Figures 7 and 8 present the effluent data for the 2007-2008 modeling period. The total nitrogen (TN) and total phosphorus (TP) loads during the modeling period for the Harrington STP are 66.4 lb/d TN and 0.9 lb/d TP; and for the KCRWRF are 553.3 lb/d TN and 173.1 lb/d TP. Although the Harrington STP loads are less, this discharge is located in the headwaters of Browns Branch and given the high effluent NH_3 concentrations has a large impact on toxicity and DO levels in Browns Branch. Since the KCRWRF discharge is located in the tidal portion of the river with much greater rates of tidal mixing, the water quality impacts are less with current TP loads still being significant in the tidal river.

Septic systems are also nutrient sources in the watershed both through groundwater contributions but also directly to the streams from failing or improperly operated systems. Therefore, septic nutrient loads were assigned in the model based on the location of septic systems throughout the watershed (2005 data) as provided by DNREC. Figure 9 presents the septic system locations along with the HSPF model sub-watershed segmentation. For each sub-watershed segment, NO_2+NO_3 and PO_4 loads from septic tanks were estimated and assigned as point sources in the HSPF model as a constant source. Septic loads were computed for each sub-watershed by multiplying the number of septic systems by the average number of people served by each system, typical septic overcharge flow rate, failure rate and concentration. A unit septic system loading rate was calculated with the assumptions that the septic overcharge flow rate per capita is 70 gallons/day (Horsely & Whitten, 1996), the average number of persons served by a septic tank is 2.8, the NO_2+NO_3 and PO_4 concentrations reaching the stream from septic system overcharge is 5 mg/L and 1 mg/L, respectively. A final scale factor of 25% of the original calculated septic system load was determined during the calibration process, which may represent the percentage of failing septic systems. Table 6 presents a summary of the septic system loads by sub-watershed.

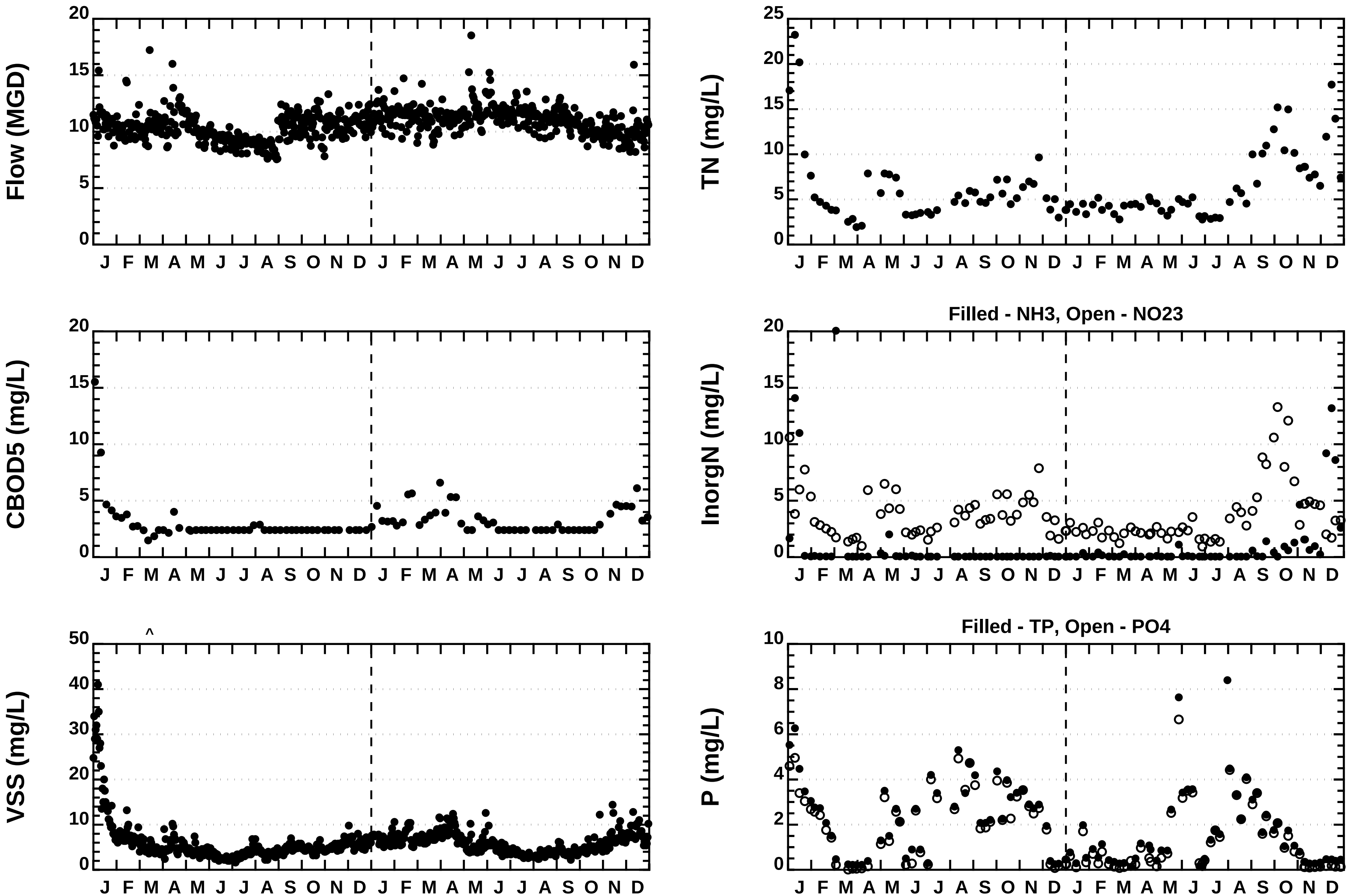

Figure 7. KCRWRF Effluent Data for 2007-2008

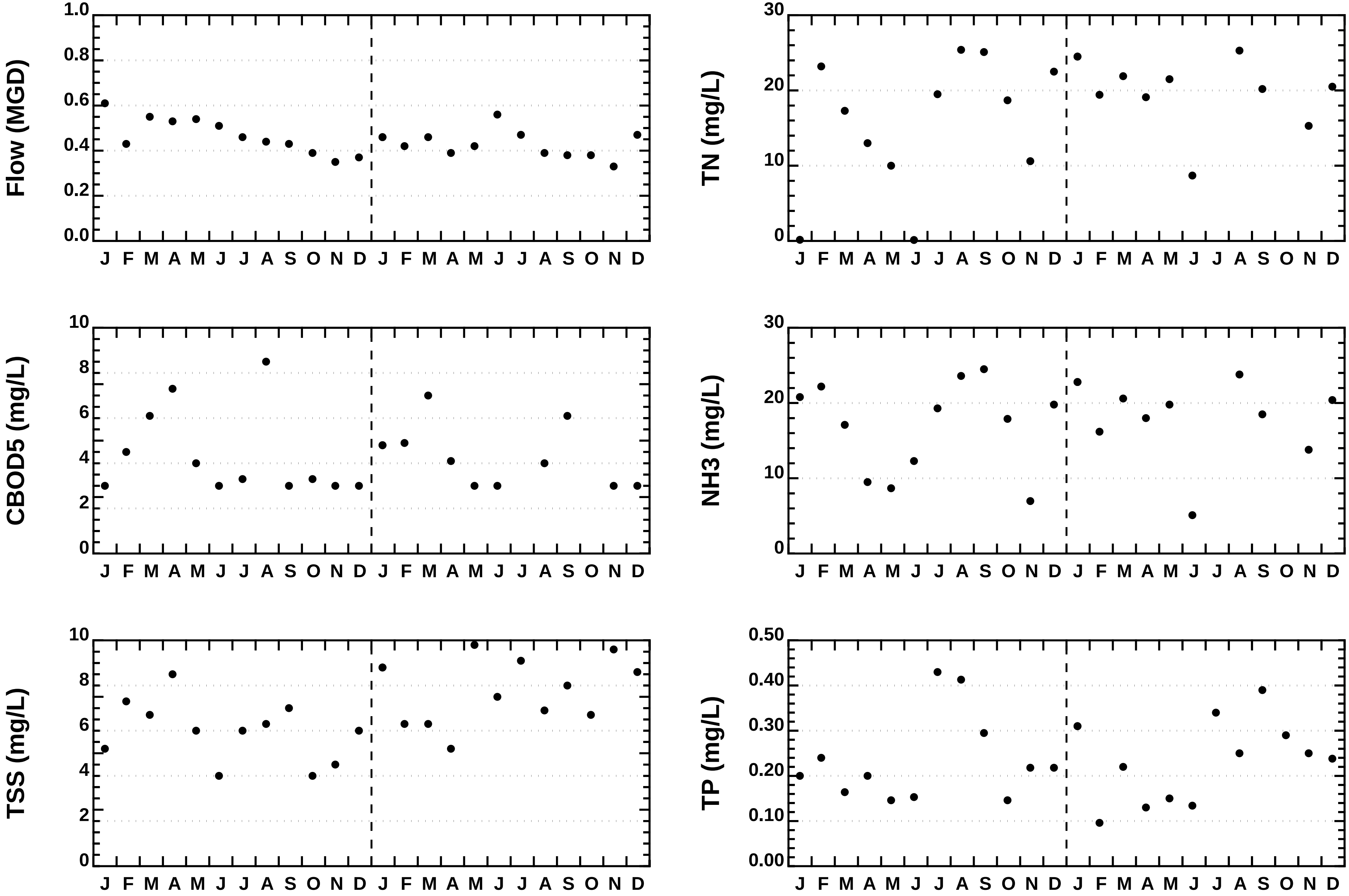

Figure 8. Harrington STP Effluent Data for 2007-2008

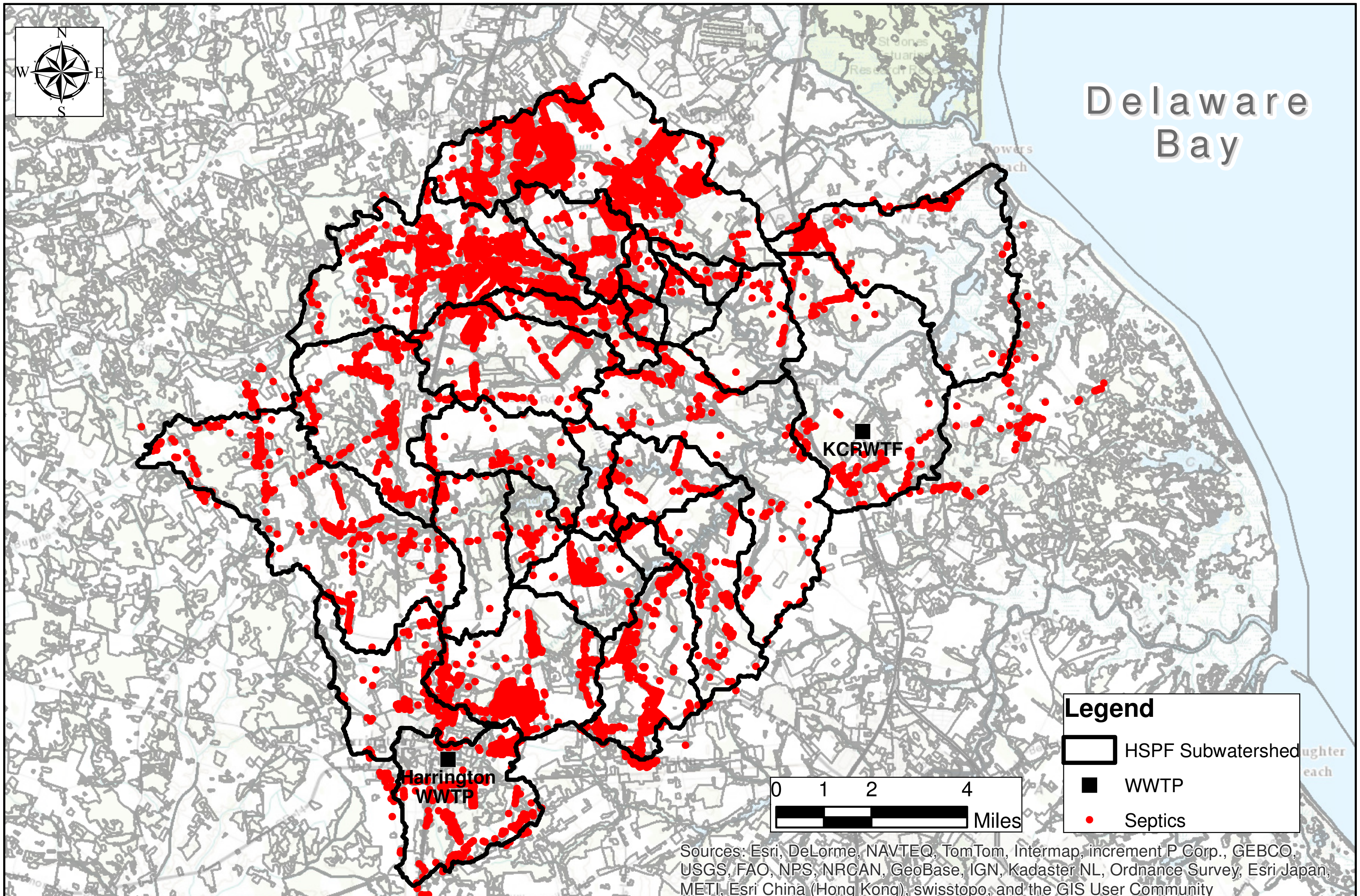

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Figure 9. Murderkill River Watershed Septic System Distribution

Table 5. Point Source Load Summary		
Parameter	**Harrington STP**	**KCRWRF**
Flow (MGD)	0.45	10.7
$CBOD_5$ (mg/L)	4.3	3.2
DO (mg/L)	n/a	8.4
TSS (mg/L)	6.9	6.2*
TN (mg/L)	17.7	6.2
NH_3 (mg/L)	17.6	1.0
NO_2+NO_3 (mg/L)	n/a	3.7
TP (mg/L)	0.24	1.94
PO_4 (mg/L)	n/a	1.66
* - VSS data		

Table 6. Septic System Load Summary				
Sub-watershed	**# of Septic Systems**	**Septic Flow (gpd)**	**Septic NO_2+NO_3 Load (lb/d)**	**Septic PO_4 Load (lb/d)**
1	195	38,220	0.40	0.08
2	1,752	343,392	3.59	0.72
3	32	6,272	0.07	0.01
4	1,135	222,460	2.32	0.47
5	241	47,236	0.49	0.10
6	77	15,092	0.16	0.03
7	282	55,272	0.58	0.12
8	224	43,904	0.46	0.09
9	48	9,408	0.10	0.02
10	32	6,272	0.07	0.01

Table 6. Septic System Load Summary

Sub-watershed	# of Septic Systems	Septic Flow (gpd)	Septic NO_2+NO_3 Load (lb/d)	Septic PO_4 Load (lb/d)
11	33	6,468	0.07	0.01
12	317	62,132	0.65	0.13
13	247	48,412	0.51	0.10
14	103	20,188	0.21	0.04
15	26	5,096	0.05	0.01
16	65	12,740	0.13	0.03
17	61	11,956	0.13	0.03
18	81	15,876	0.17	0.03
19	331	64,876	0.68	0.14
20	156	30,576	0.32	0.06
21	351	68,796	0.72	0.14
23	446	87,416	0.91	0.18
24	123	24,108	0.25	0.05
25	55	10,780	0.11	0.02
26	83	16,268	0.17	0.03
27	171	33,516	0.35	0.07
28	168	32,928	0.34	0.07
Total	**6,835**	**1,339,660**	**13.99**	**2.80**

2.1.5 Water Quality Input

The HSPF model was used to simulate the fluxes and storages of the water quality constituents on the surface and subsurface layer of the land segments, as well as kinetics in the non-tidal streams and lakes. The model results provided pollutant loadings from the nonpoint sources and upstream tributaries for the eutrophication model that simulated the tidal river and estuary.

The transport of sediment, carbon and nutrients associated with surface runoff was modeled by specifying accumulation rates, surface storage capacity and overland flow wash-off on pervious and impervious land surface. Contributions from subsurface flow and groundwater were taken into account by estimating land use specific concentrations and from available data in a headwaters monitoring station on Pratt Branch.

The in-stream processes simulated by the model include air-water heat exchanges for water temperature calculations, sediment deposition, simulation of dissolved oxygen as a result of atmospheric reaeration and decomposition of organic matter, reactions and balances of nitrogen and phosphorous forms, life cycle of phytoplankton, and bacterial decay in the water bodies.

The parameters that govern the water quality processes on land segments and in the streams were initialized using values developed in the Delaware Inland Bays Watershed Study. They were then adjusted to optimize the calibration of the model against the monitored data as described in the water quality calibration section.

2.2 WATERSHED MODEL CALIBRATION AND VALIDATION

2.2.1 Quantity (Flow)

An initial set of parameters for various hydrologic processes (such as deep percolation, infiltration, and interception) were developed from the HSPF modeling conducted by USGS for the Delaware Inland Bays watershed. F-tables that describe the cross-sectional data as well as the stage-discharge relationships in the model reaches were derived from available geometry data for the tributaries and lakes. The calibration process was performed according to the guidelines provided in the HSPF User's Guide (Bicknell et al., 2001).

Daily flow data available at the three USGS stations (Murderkill River, Browns Branch and Pratt Branch) were used for the flow calibration and validation. Parameters were first adjusted to match base flow volumes in the calibration/validation period, then the storm flow volumes and finally the characteristic shapes of the storm hydrographs. The USEPA BASINS Technical Note 6, Estimating Hydrology Parameters for NPSM/HSPF (USEPA, 2000) was also used to determine the ranges of different parameters and their influences on the model results. This document was referred to throughout the calibration/validation process. The HSPF model was run for the period January 2006 through December 2006 as a spin-up period to equilibrate soil-moisture conditions. The calibration period was January 2007 through December 2007, and the validation period was January 2008 to December 2008.

There are no generally accepted guidelines as to what constitutes an "acceptable" watershed model calibration/validation and is best conducted on a weight of evidence approach (Donigian, 2000, personal communication) that considers both graphical and statistical comparisons (Thomann, 1982). The approach used was to achieve the best fit to a series of metrics, while keeping model parameter values within accepted ranges. The metrics used for assessing the calibration were: total water balance in the calibration/validation periods; comparison of monitored and modeled hydrographs; and cross-plots of monitored and modeled flow.

Table 7 summarizes the total water balance within the calibration/validation periods along with the percent difference with respect to monitored flows at the USGS stations. Figure 10 presents the comparison of monitored and modeled hydrographs for the calibration and validation periods. Figure 11 presents cross-plots of daily average and monthly average monitored and modeled flow. Overall the HSPF model reproduces the observed hydrographs at the three USS gages fairly well (Figure 10) given the location of the meteorological data used for model inputs (i.e., 10 miles north of the USGS gages) and relatively low land surface slopes in the Murderkill River watershed. Percent differences between model and data for the long term flow volumes in 2007 and 2008 (Table 7) ranged from -24% to 7% with an overall percent difference of -10% for the June 2007 to December 2008 time period. Some observed peak flows were not reproduced in the model particularly at the Murderkill River gage, which appears to be influenced by an area of upstream wetlands that may affect runoff at this location. On a monthly average basis, the HSPF model reproduces the observed data well at the Browns Branch and Pratt Branch gages (Figure 11) but tends to underestimate the peak flows at the Murderkill River gage. In general, the HSPF model reproduces the observed flows in the watershed well and reasonably represents the hydrologic conditions in the Murderkill River watershed.

Table 7. HSPF Flow Calibration/Validation Summary

Period	HSPF Results (cfs)	USGS Data (cfs)	% Difference
Murderkill River near Felton (#01484000)			
6-12/2007	684.0	641.6	6.6
1-12/2008	1960.2	2575.3	-23.9
6/2007-12/2008	2644.2	3216.8	-17.8
Browns Branch near Harrington (#01484018)			
6-12/2007	1046.6	1020.3	2.6
1-12/2008	2674.0	3039.5	-12.0
6/2007-12/2008	3720.6	4059.8	-8.4
Pratt Branch near Felton (#01484050)			
6-12/2007	309.8	385.3	-19.6
1-12/2008	837.9	823.7	1.7
6/2007-12/2008	1147.7	1209.0	-5.1

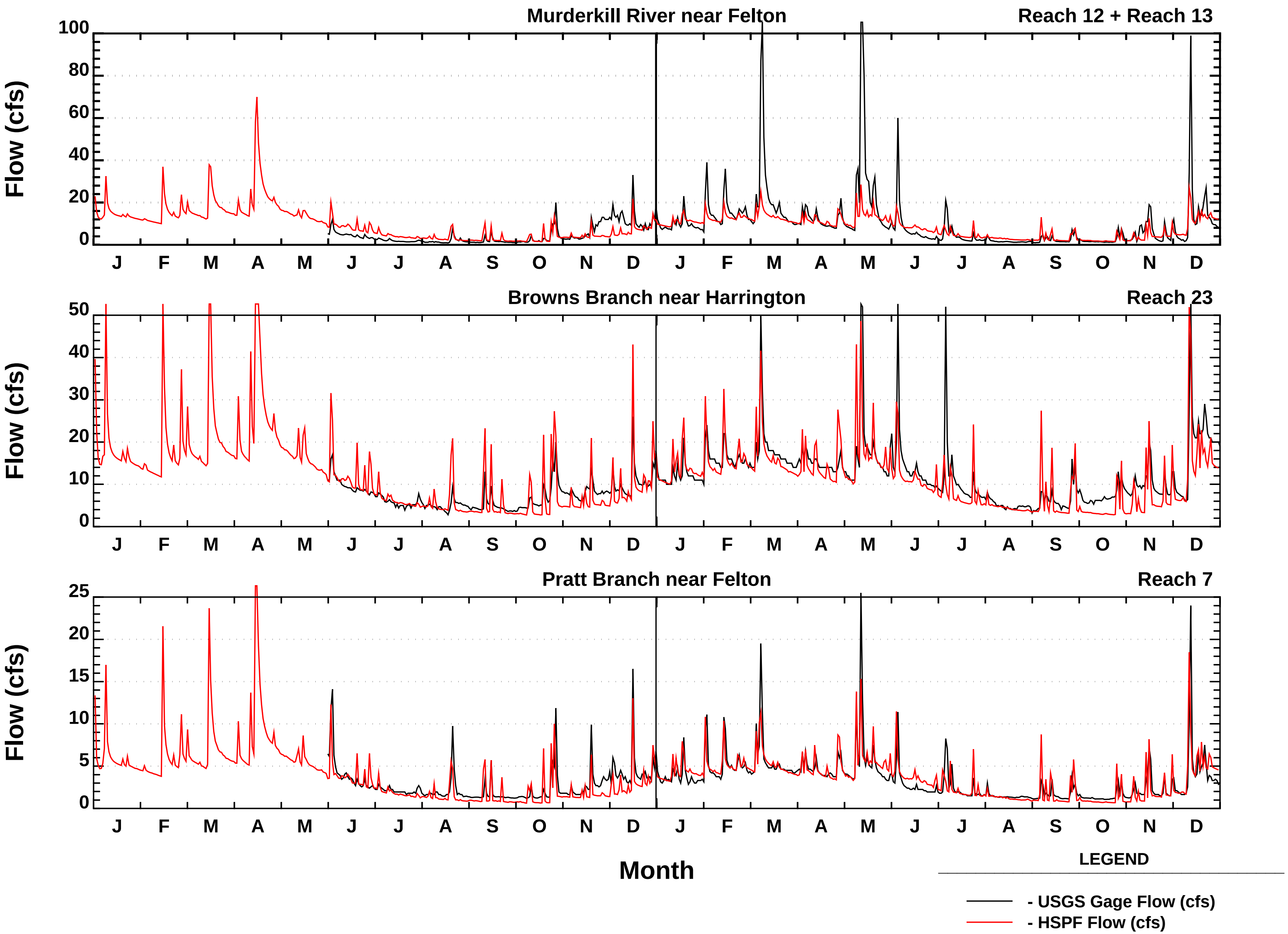

Figure 10. Murderkill River HSPF Flow Calibration/Validation (2007-2008)

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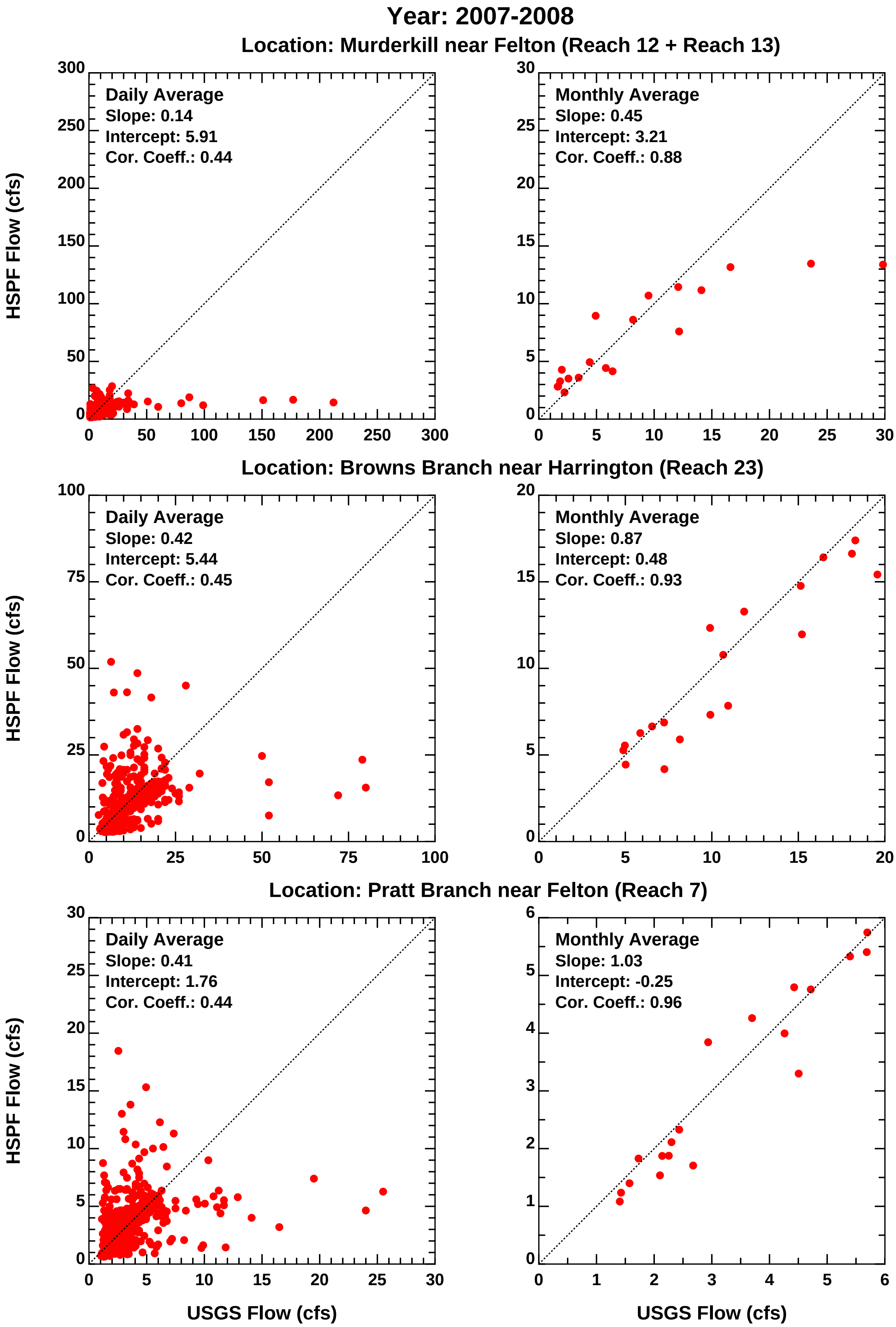

Figure 11. Daily and Monthly Averaged HSPF Flow Calibration (2007-2008)

2.2.2 Water Quality

After completing the model runoff calibration/validation, water quality simulations with the HSPF model were performed. An initial set of parameters for the various water quality processes related to TSS, DO, nutrients and biochemical oxygen demand (BOD) were developed from the HSPF model conducted by USGS for the Delaware Inland Bays watershed (USGS, 2003). Calibration/validation of the model required adjustment of various model parameters to reproduce observed water quality data and was iterated a number of times to produce a best fit with the data. Groundwater and interflow nutrient concentrations were based on data in Pratt Branch, Double Run and the Murderkill River near the headwaters of these tributaries and adjusted as part of the calibration process.

Figures 12 to 16 present the calibrated and validated model results compared to the observed data at five stations in the watershed that represent tributary locations that discharge to the tidal Murderkill River. The filled circles in these figures represent observed data and the solid lines represent model output. These locations are: Double Run at Barretts Chapel Road; McGinnis Pond; Andrews Lake; Coursey Pond; and McColley Pond. It should be noted that the Double Run monitoring location can be affected by the tides and is also located in a sluggish area affected by backwater effects. Therefore, HSPF model results at this location may not best represent the observed data but are shown for comparison mainly to the TN and TP data. The entire set of model calibration/validation figures are presented in Appendix 2. The parameters presented in these figures include TN, NO_2+NO_3, NH_3, TP, PO_4, chlorophyll-a (chl-a) and DO.

Overall the observed nutrient levels (TN, NO_2+NO_3, NH_3, TP, PO_4) are fairly well reproduced by the model with some over and under estimation at the various stations. The HSPF model was setup with one set of land use specific water quality parameters and groundwater/interflow concentrations that are used for all model sub-watersheds. This approach does not account for potential variations of these parameters by sub-watershed and may contribute to the variation in the model results (i.e., over and under estimation). At the McColley Pond monitoring location on Browns Branch, the upstream Harrington STP loads assigned and the watershed wide stream nitrification rate used resulted in the model under estimating observed NO_2+NO_3 levels but over estimating observed NH_3 levels but roughly equal amounts such that the sum of these two parameters (dissolved inorganic nitrogen) is well represented in the model. The model also reproduces the observed chl-a levels well over an annual cycle with the peak summer levels well reproduced. At most locations, DO levels are also reproduced well except at Double Run (affected by tides) and McGinnis Pond where the observed data is relatively constant over the year while the model calculates a typical seasonal DO pattern (i.e., lower DO during the warmer summer months). In general, the HSPF model is well calibrated and validated to the observed data and reflects the water quality dynamics in the watershed and the loadings to the tidal Murderkill River.

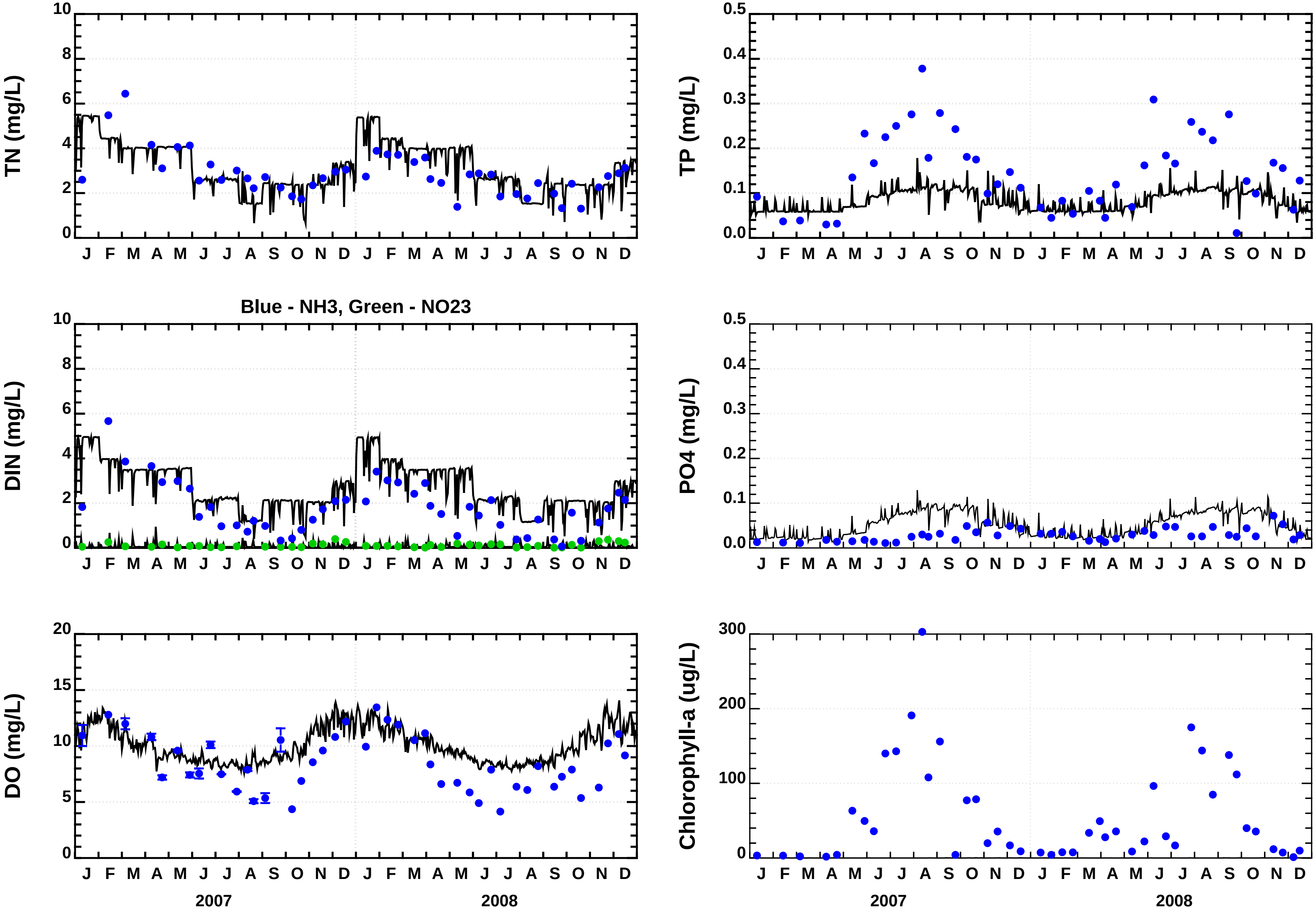

Figure 12. Water Quality Model Calibration at Station 206561, Double Run at Barratts Chapel Rd. (Rd. 371)

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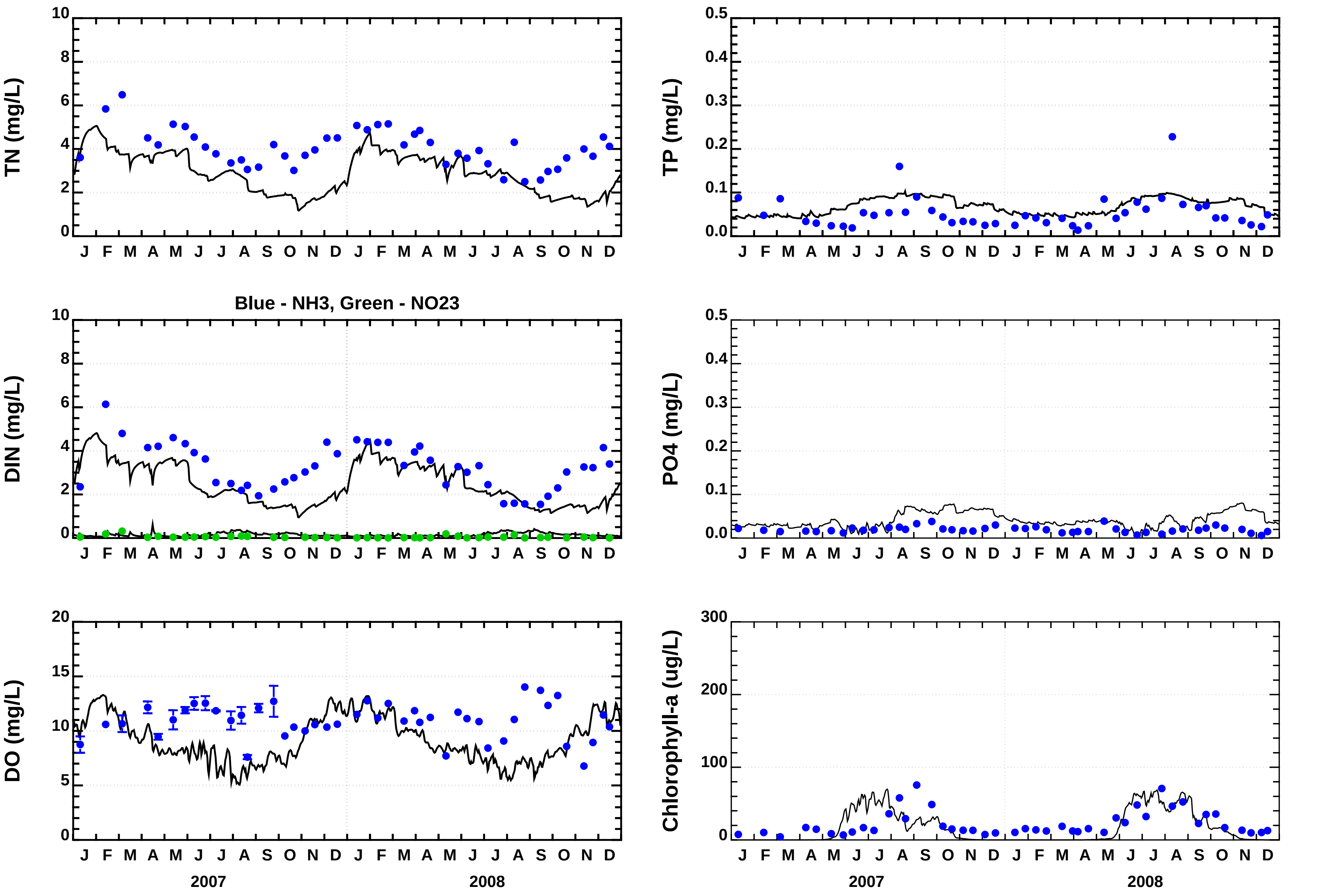

Figure 13. Water Quality Model Calibration at Station 206461, McGinnis Pond at McGinnis Pond Rd. (Rd. 378)

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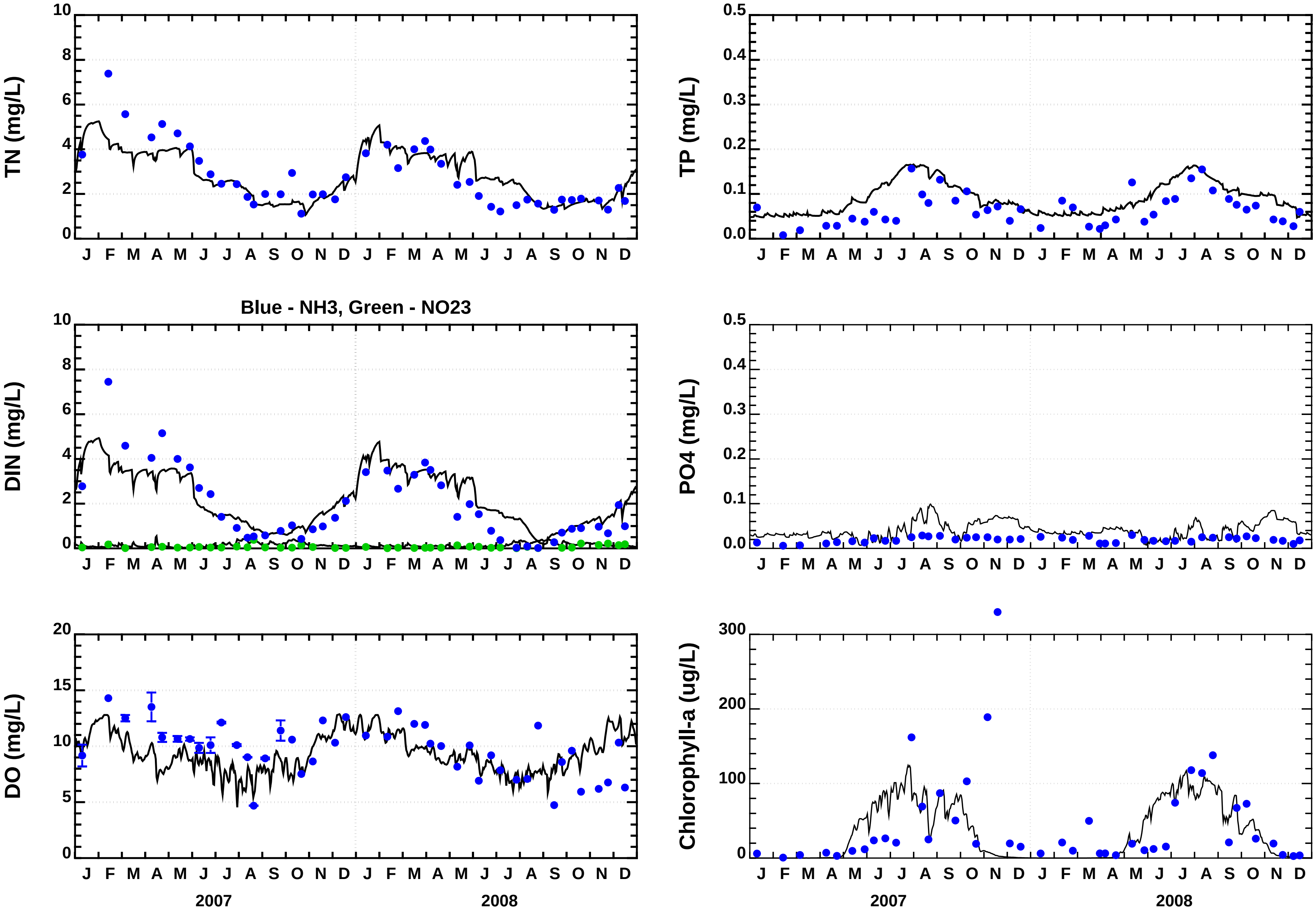

Figure 14. Water Quality Model Calibration at Station 206071, Andrews Lake at Rd. 380 Brdg.

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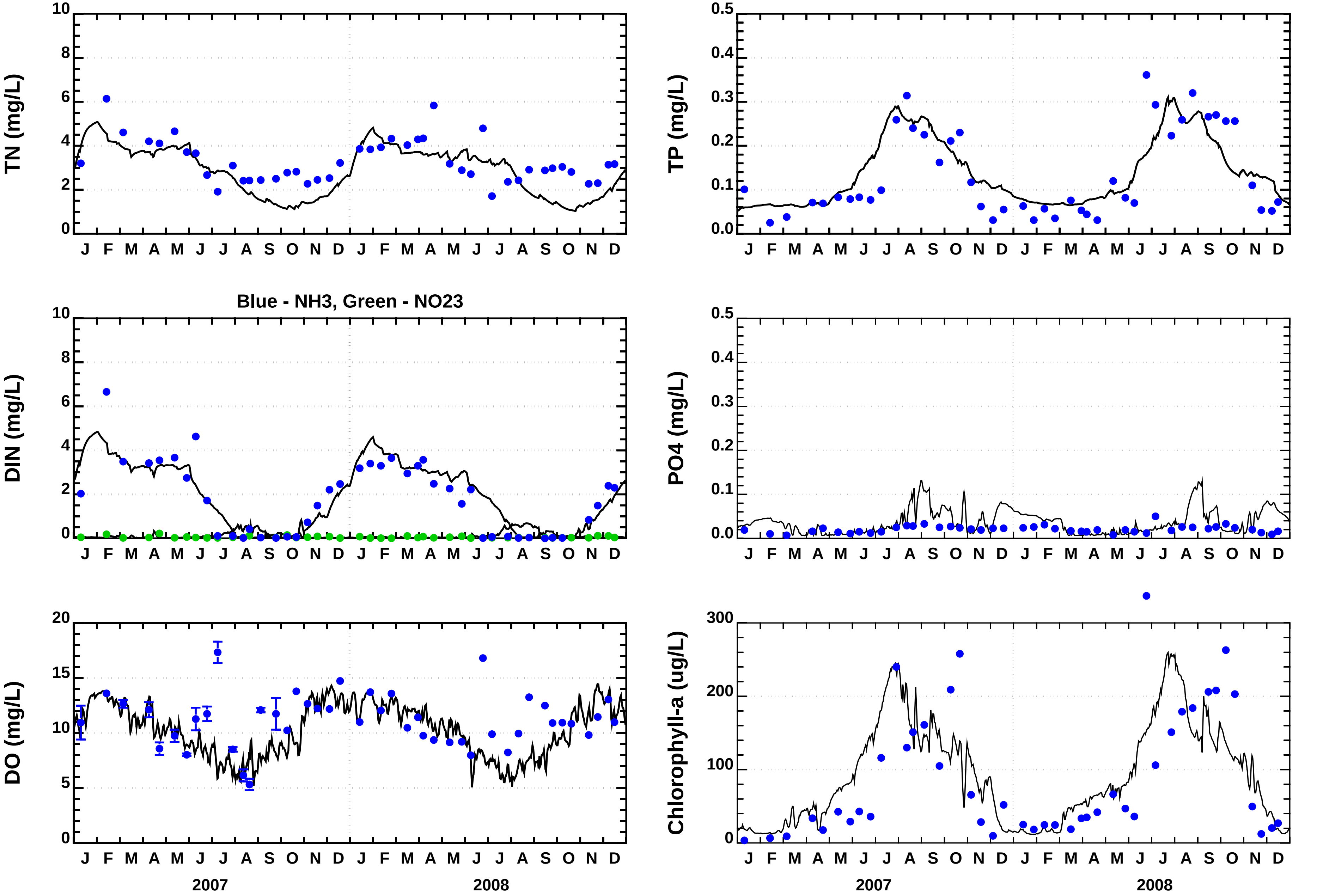

Figure 15. Water Quality Model Calibration at Station 206451, Coursey Pond at Canterbury Rd. (Rt. 15) at Rd. 388 Bridge

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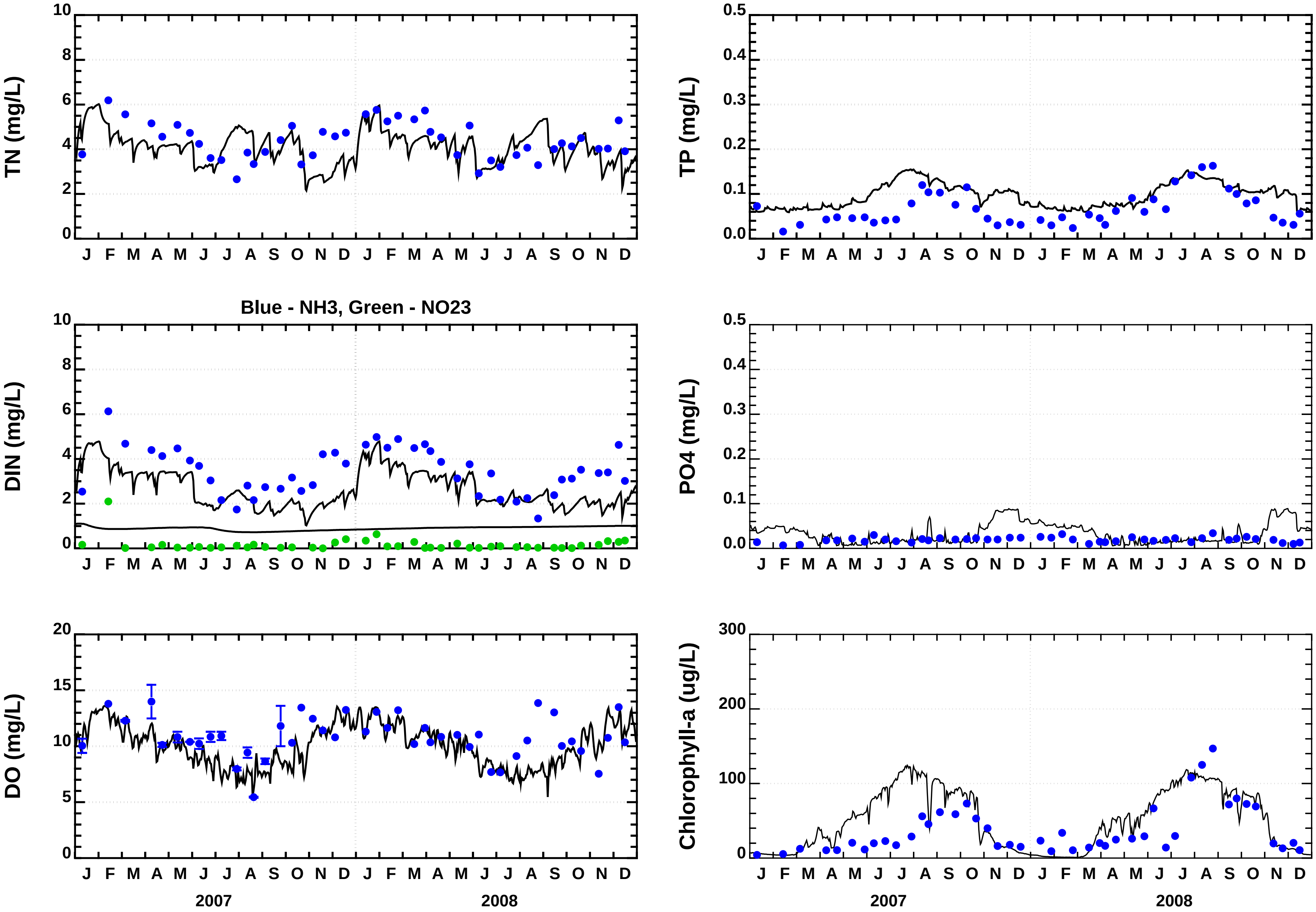

Figure 16. Water Quality Model Calibration at Station 206361, McColley Pond at Canterbury Rd. (Rt. 15) near Spillway

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Watershed TN and TP annual and unit area loadings from the HSPF model calibration and are presented in Table 8.

Table 8. HSPF Model TN and TP Watershed Loads					
Sub-Watershed	Area (mi^2)	TN		TP	
		(lb/d)	(lb/d/mi^2)	(lb/d)	(lb/d/mi^2)
Double Run	8.65	144.9	16.8	2.46	0.28
McGinnis Pond	8.06	122.9	15.2	1.96	0.24
Andrews Lake	1.40	90.4	64.6	1.79	1.28
McColley Pond	11.98	311.9	26.0	4.27	0.36
Coursey Pond	12.40	266.0	21.5	5.89	0.47
Tidal River	34.53	453.3	13.1	7.83	0.23
Total	77.02	1389.4	18.0	24.21	0.31

SECTION 3

HYDRODYNAMIC MODEL

The transport and mixing of point and nonpoint source loads introduced to the tidal Murderkill River are controlled by estuarine circulation in the system. At the same time, turbulent mixing created by tides, freshwater inflow and surface wind stress leads to horizontal and vertical dispersion in the water body. Coupled with turbulent mixing are heat exchange processes between the water column and the atmosphere. The complexity of the physical processes governing the evolution of an introduced constituent, such as nutrients, suggests the use of sophisticated hydrodynamic models. For this study, HDR|HydroQual's three-dimensional, time-dependent, estuarine and coastal circulation model (ECOMSED) was applied to compute current, temperature and salinity distributions in the tidal Murderkill River. The SED component of the ECOMSED model represents the cohesive and non-cohesive sediment transport model that is coupled to the hydrodynamic model (ECOM). In this application the sediment transport model was not used.

The ECOMSED model developed by Blumberg and Mellor (1980 and 1987) incorporates the Mellor and Yamada (1982) level 2-1/2 turbulent closure model to provide a realistic representation of vertical mixing. Horizontal diffusion is calculated with a Smagorinsky (1963) eddy parameterization to give a greater horizontal mixing coefficient near strong horizontal gradients. A system of curvilinear coordinates is used in the horizontal direction, which allows for a smooth and accurate representation of variable shoreline geometry. In the vertical scale, the model uses a transformed coordinate system known as the σ-coordinate transformation to permit better representation of bottom topography. Water surface elevation, three-dimensional water velocity, temperature, salinity, and water turbulence are calculated in response to weather conditions (wind and incident solar radiation), freshwater inflows and tides, and temperature and salinity at the open boundaries. A more detailed discussion of ECOMSED is presented in Appendix 3 along with calibration/validation constants and parameters.

The ECOMSED model has been extensively used by over 1,000 research groups around the world for simulations of various hydrodynamic systems; among them are works by Blumberg et al. (2004) on the Hudson estuary, Blumberg and Kim (2000) on St. Andrew Bay, Blumberg et al. (1999) on the New York Harbor, Ahsan and Blumberg (1999) on Onondaga Lake, NY, Mellor and Ezer (1991) on the Gulf Stream Region, Blumberg and Goodrich (1990) on Chesapeake Bay, Blumberg and Mellor (1985) on the Gulf of Mexico. In all these studies, model skill was assessed via extensive comparisons with data involving water surface elevations, currents, temperature, salinity, and pathogens. A confidence has been established that the predominant physics are realistically reproduced by the model. The model is also actively being applied in an operational forecasting mode for the Great Lakes, the Gulf of Mexico, East Coast of the United States and in Norwegian coastal waters.

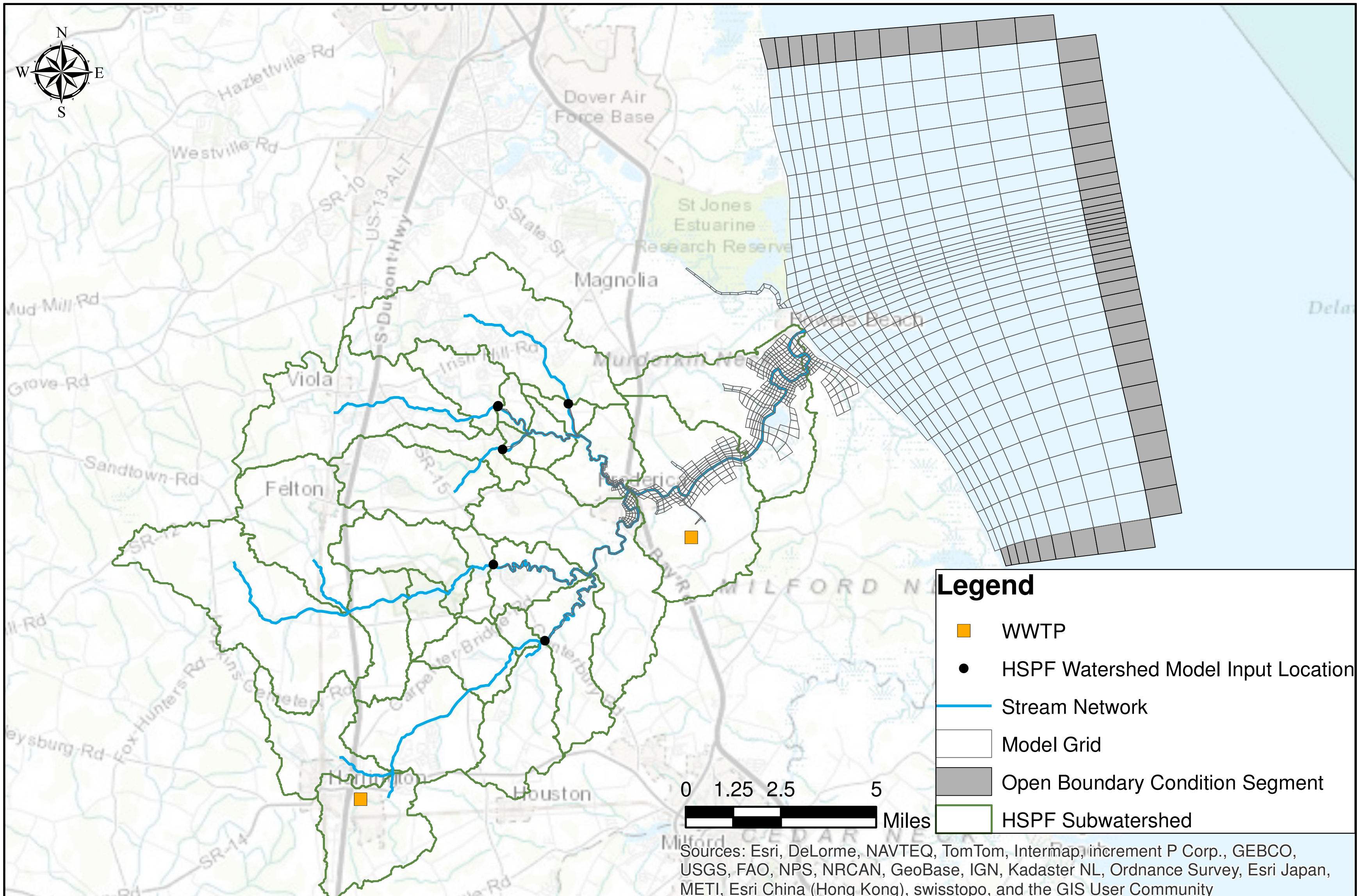
3.1 PROJECT SPECIFIC STUDY DATA

Three project specific studies were completed to support the Murderkill River Study that provided data to support the hydrodynamic model setup and calibration/validation efforts. These data included: the USGS continuous monitoring at Bowers Beach (#01484085) and Frederica (#01484080) for salinity, temperature, water elevation and tidal volume flux; tidal river bathymetric data; and the LiDAR mapping of the lower Murderkill River tidal marsh that provided estimates of the marsh area and volume at different tidal inundation levels (McKenna, 2013). The USGS continuous data was used for model calibration/validation (salinity, temperature, water elevation and tidal volume flux) but also to resolve the tidal volume that enters the tidal marshes based on the tidal volume difference between the gages at Bowers Beach and Frederica. In addition, the LiDAR mapping also provided an estimate of the salt mash volume at different tidal inundation levels that was also used to check the on the tidal volumes calculated with the model. The bathymetric data was used to setup the model grid. These data and there uses are discussed in more detail in the following sections.

3.2 MODEL CONFIGURATION

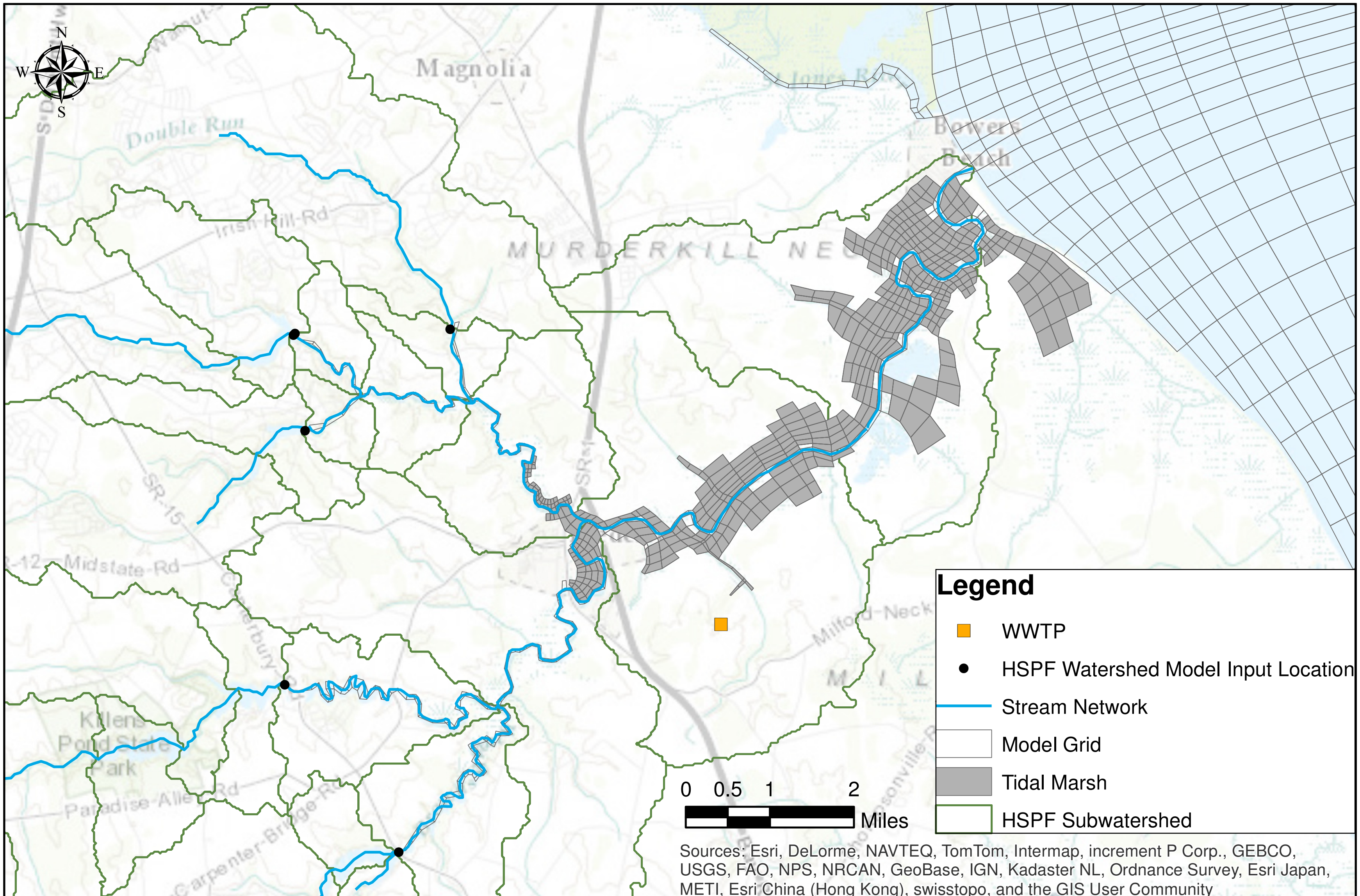
An orthogonal, curvilinear modeling grid system has been designed in order to discretize the tidal reaches of the lower portion of the Murderkill River and nearshore Delaware Bay (Figures 17a and 17b). The model open boundary extends approximately 4-7 miles into Delaware Bay from the shoreline. The grid system consists of an 89 x 63 segment model grid in the horizontal plane with 6 equally spaced σ-levels in the vertical plane (i.e., 5 vertical segments). In the transformed σ-coordinate system, the model has an equal number of vertical segments in all the computational grid boxes. It should be noted that the curvilinear grid system allows for much finer grid resolution near areas of interest and, therefore, allows the design of an efficient and computationally time-effective modeling framework.

Segmentation of the hydrodynamic model resulted in an 89 x 63 x 5 model computational grid (28,035 model segments) that consists of 6,445 active water segments and 2,535 active tidal marsh segments. Active model segments are those that are used for model circulation calculations as opposed to inactive model segments that represent land or non-water features of the watershed. The model segments range in size from about 85 feet by 100 feet square in the upper reaches of the river to about 2,760 feet by 4,150 feet square out in Delaware Bay. This model grid developed for the hydrodynamic model is also used for the water quality model. Also, the bottom of the model grid is a no flow boundary. Figures 17a and 17b presents the model segmentation used for both the hydrodynamic and water quality models. The model segments were developed in the tidal Murderkill River and extended into the Delaware Bay approximately 4-7 miles from the shoreline and 11 miles in the upstream/downstream direction in the bay (centered on the Murderkill River). The extension of the model grid into the bay is aimed at minimizing the bay boundary condition effects on the internal model calculations.



HDR | HydroQual

Figure 17a. Murderkill River ECOMSED/RCA Model Grid



HDR | HydroQual

Figure 17b. Murderkill River ECOMSED/RCA Model Grid

The tidal marsh segments in the model grid wet and dry depending on the tidal stage and interact with the main channel of the river through openings assigned in the model grid (20) to allow communication between the marsh and main channel model segments. When the marsh segments are wet they are modeled as any other active water segment in the model grid. These tidal marsh segments do not receive inputs from the watershed model (flows) and only interact with the main channel of the river through tidal filling and emptying over the tidal cycle. The importance of including the tidal marsh segments in the hydrodynamic model is so that the tidal volume in the river and subsequent impacts on the timing and magnitude of the tidal flows are reproduced. Previous HDR|HydroQual modeling efforts in other Delaware tidal rivers has highlighted the importance of the tidal volume that is stored in the tidal marshes during the tidal cycle.

There are there are three major types of boundary condition inputs used in the hydrodynamic model setup as presented below. The location of these boundary conditions is presented in Figure 17a with the open or tidal boundary conditions also varying with depth.

- River and incremental inflow boundary conditions: these are assigned as time-variable flow and temperature inputs based on output from the HSPF watershed model.
- Open or tidal boundary conditions: these are assigned as time-variable elevation, salinity and temperature inputs based on measured data (NOAA, DRBC Boat Run).
- Meteorological inputs or boundary conditions: these are assigned as time-variable and spatially constant based on measured data (DNERR).

Bathymetry data for the tidal river and bay were obtained from NOAA GEODAS CDs (NOAA, 1998) and also USGS bathymetry data collected as part of the project field efforts in the tidal river. Figure 18 presents the USGS bathymetry data collected in the tidal Murderkill River at mean sea level (MSL), which included 9 cross-sections (green) and a continuous track (blue) along the centerline of the river from the mouth to near Frederica. The model bathymetry assigned for the segmentation in the reaches upstream of Frederica was based on past modeling efforts and data collected in this area of the river. The red circles and lines in Figure 18 represent the model depths assigned and were developed to represent laterally averaged depths because the main channel segmentation of the river was one segment wide. Segmentation of the marsh area was based on USGS topographic maps, aerial photographs and LiDAR data collected as part of the project (McKenna, 2013).

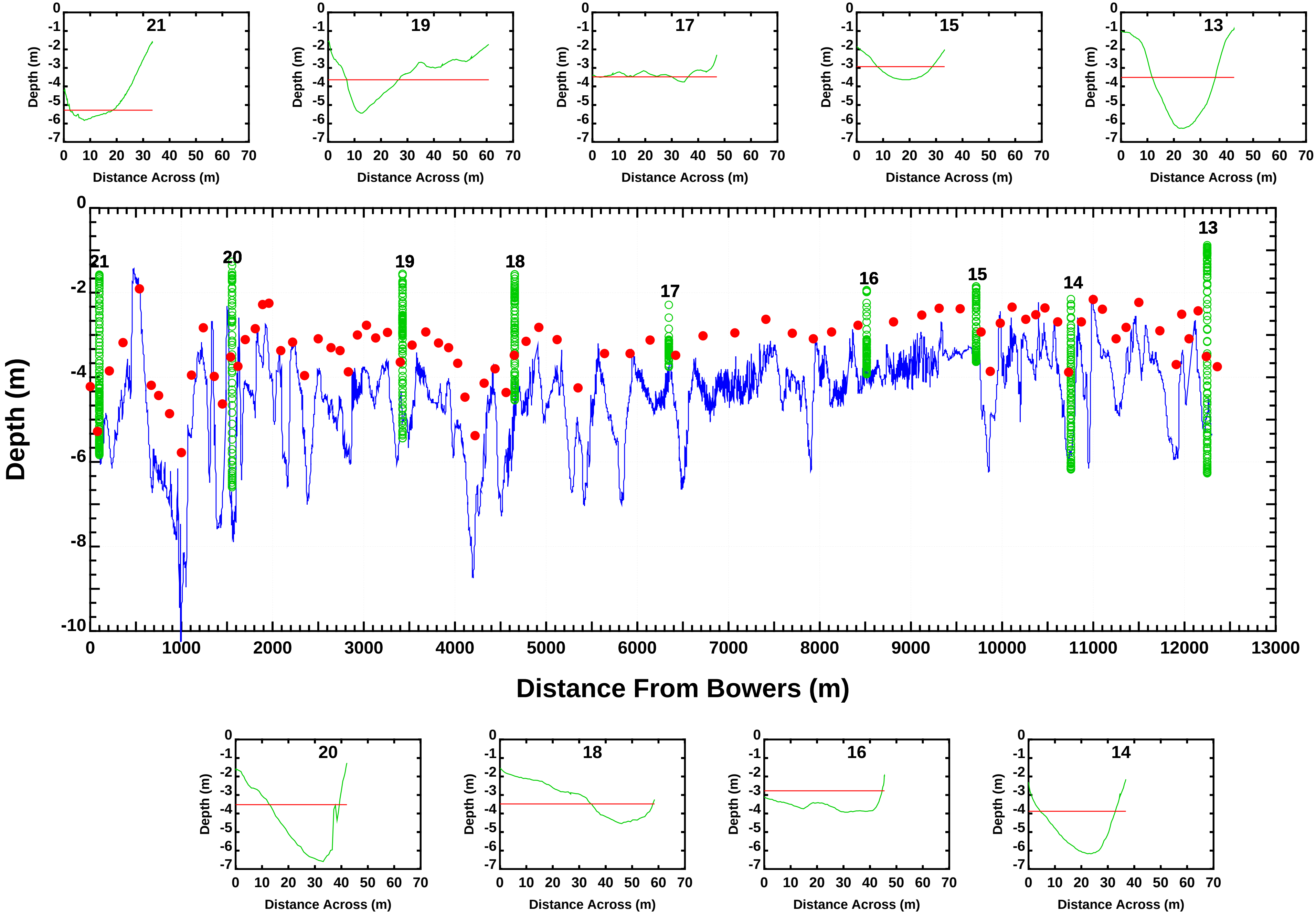


Figure 18. Murderkill River Measured Bathymetry and Model Assigned Depths (MSL)

(Green - Measured Cross-Sections, Blue - Measured Longitudinal Transect, Red - Model Depth Input)

3.3 MODEL BOUNDARY FORCING FUNCTIONS

The following types of forcing function data are needed for input at the model boundaries: watershed freshwater flow and temperature; downstream bay tidal water elevations, salinity and temperature; and meteorological inputs. Figure 19 presents the upstream (black circles) and downstream boundary condition (grey segments) locations and monitoring locations used to develop the model boundary condition inputs. The daily average freshwater flow and temperature inputs during the calibration/validation period are provided from the HSPF watershed model at the five main upstream tributary boundaries (Double Run; McGinnis Pond; Andrews Lake; Coursey Pond; and McColley Pond) along with incremental flow delivered to the tidal reaches of the river downstream from these freshwater boundaries. Incremental flows from the HSPF model are assigned in the hydrodynamic model from HSPF subwatershed segments 3, 6, 9, 10, 11, 18, 26, 27 and 28. In addition, the KCRWTF effluent flow and temperature are assigned in the hydrodynamic model based on observed effluent data.

Tidal water level (MSL) and temperature data in Delaware Bay were obtained from NOAA at the Ship John Shoal (SJSN4 – 8537121) and Brandywine Shoal Light (BRND1 – 8555889) stations and were applied at the downstream tidal bay boundaries during the calibration/validation period. The data were linearly interpolated onto the model grid tidal boundary segments for assigning model inputs. In addition to the tidal water surface elevation boundary conditions, the model was also forced with salinity data at these downstream tidal boundary locations. Salinity data from the USGS continuous monitoring station at Bowers Beach were used to assign the tidal salinity boundary conditions as follows: northern boundary used Bowers Beach salinity minus 5 ppt; southern boundary used Bowers Beach salinity plus 5 ppt; eastern boundary was linearly interpolated between the northern and southern boundary inputs. If the data indicated top to bottom salinity and temperature differences, the data were linearly interpolated from top to bottom for input to the five vertical model segments. Figure 20 presents the water level, salinity and temperature inputs used for the 2007-2008 modeling period.

Finally, the hydrodynamic model requires the input of wind speed and direction, air temperature, relative humidity, atmospheric pressure and solar radiation. Hourly data were available from the DNERR St. Jones River meteorological station located in central Kent County about 10 miles north of the river. These data are the same as that used for the HSPF watershed modeling. Figure 21 presents the meteorological data used for the 2007-2008 modeling period.

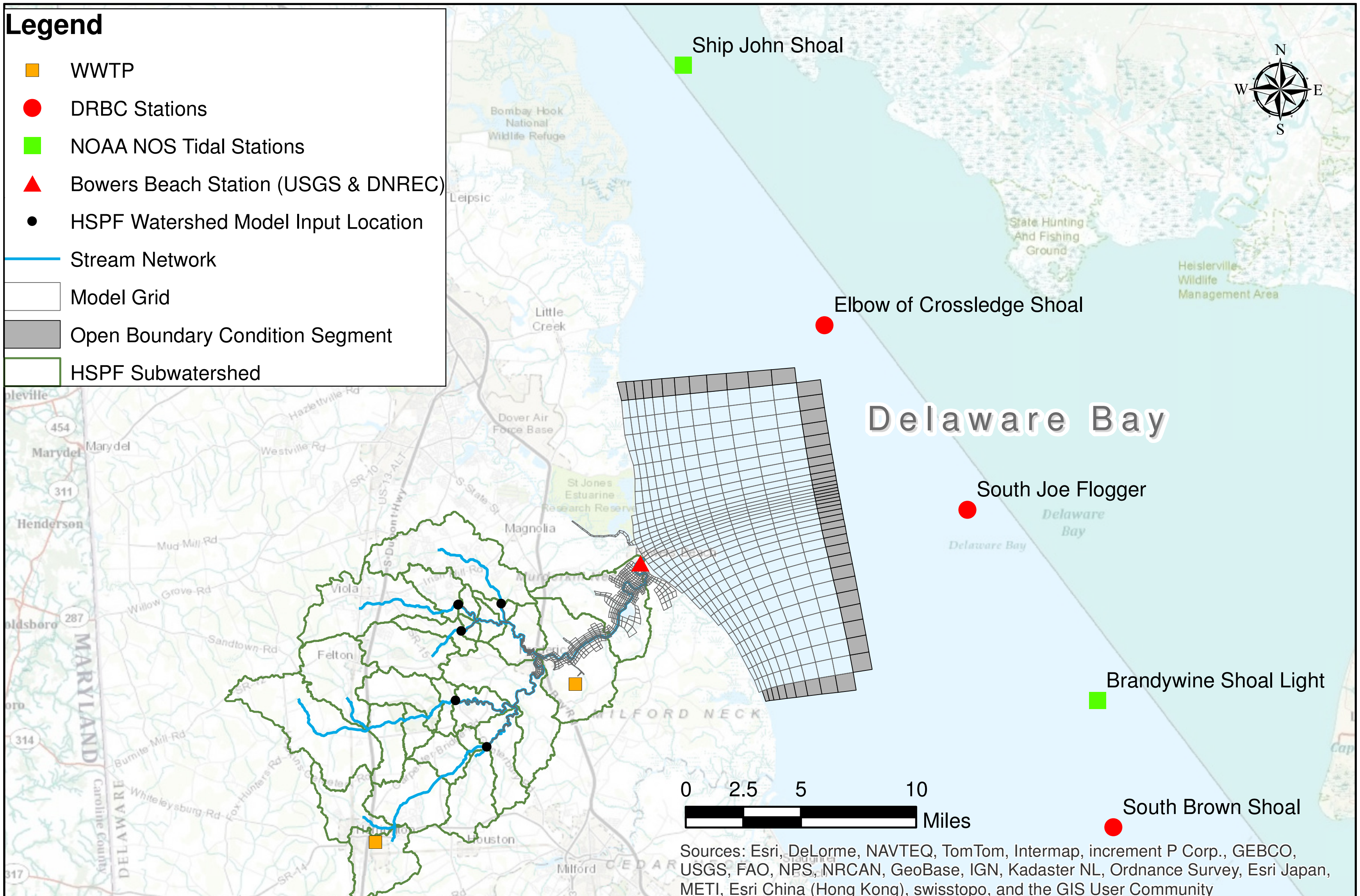


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Figure 19 ECOMSED Model Boundary Condition Locations and Delaware Bay Monitoring Locations

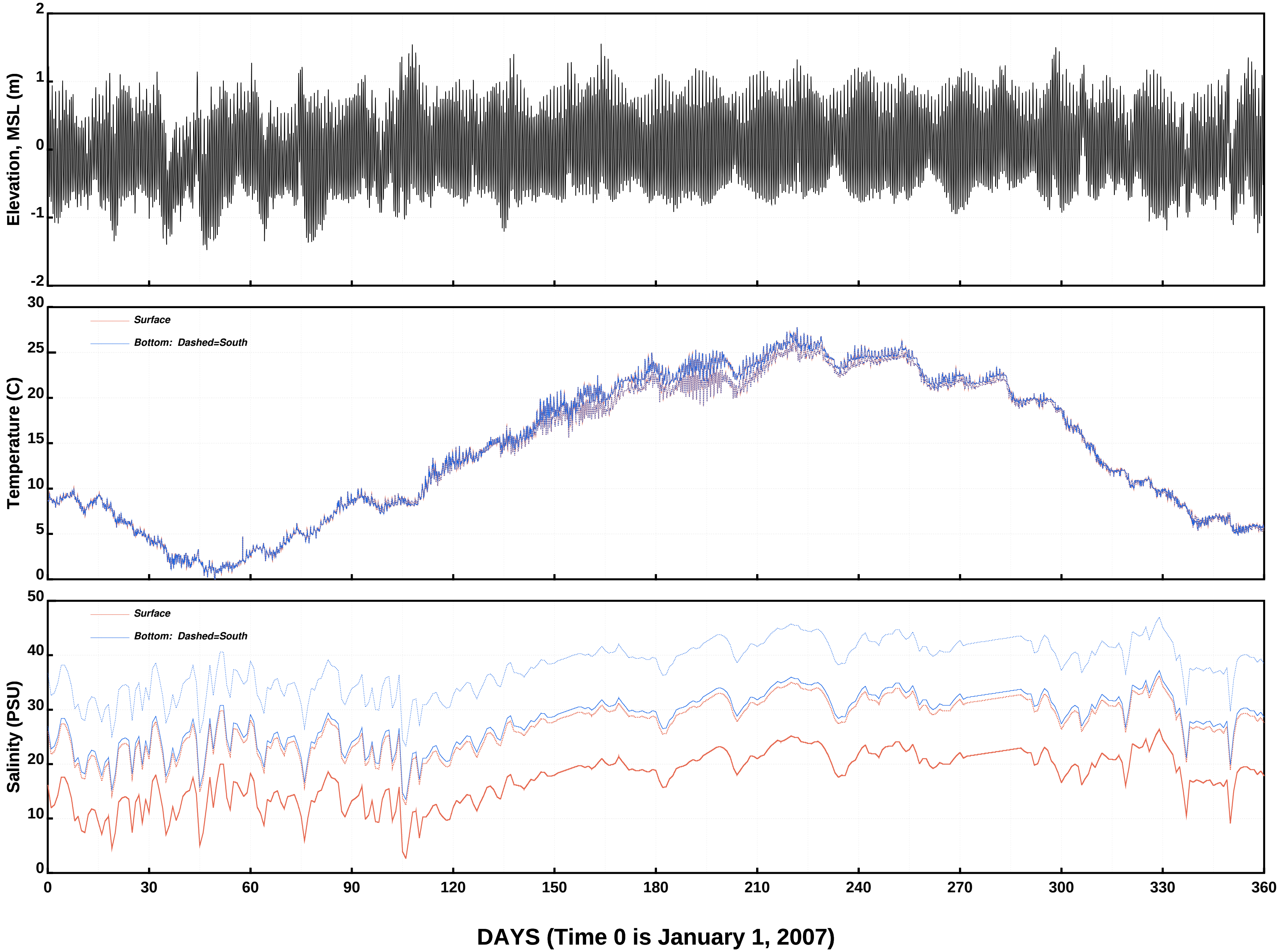

Figure 20a. ECOMSED Tidal Model Boundary Condition Input

RUN031: 010708*v18.thindams,defaultBFRIC,HORCONx0.1, Bowers salt,New100% flow,0 minAtemp .

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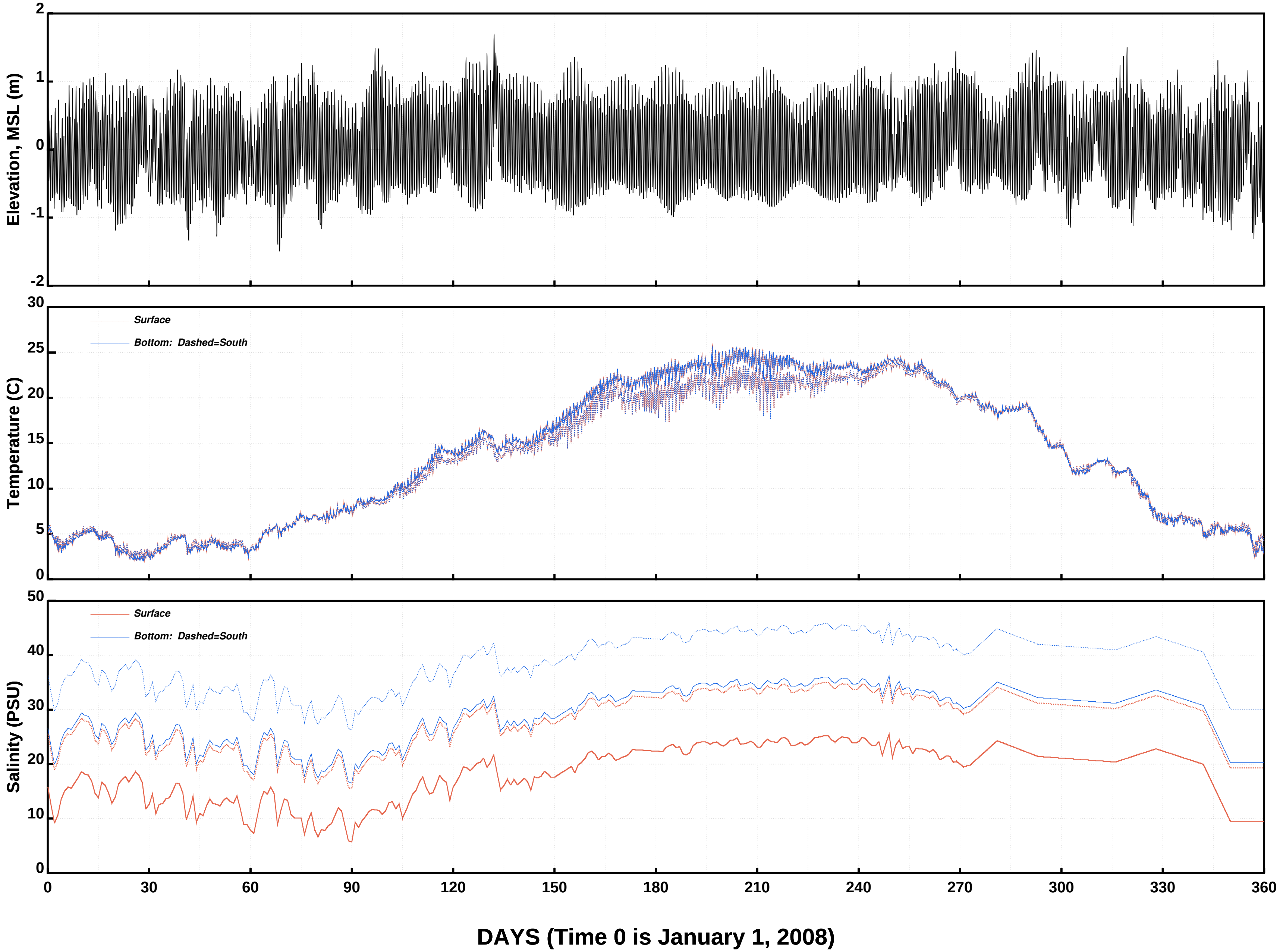

Figure 20b. ECOMSED Tidal Model Boundary Condition Input

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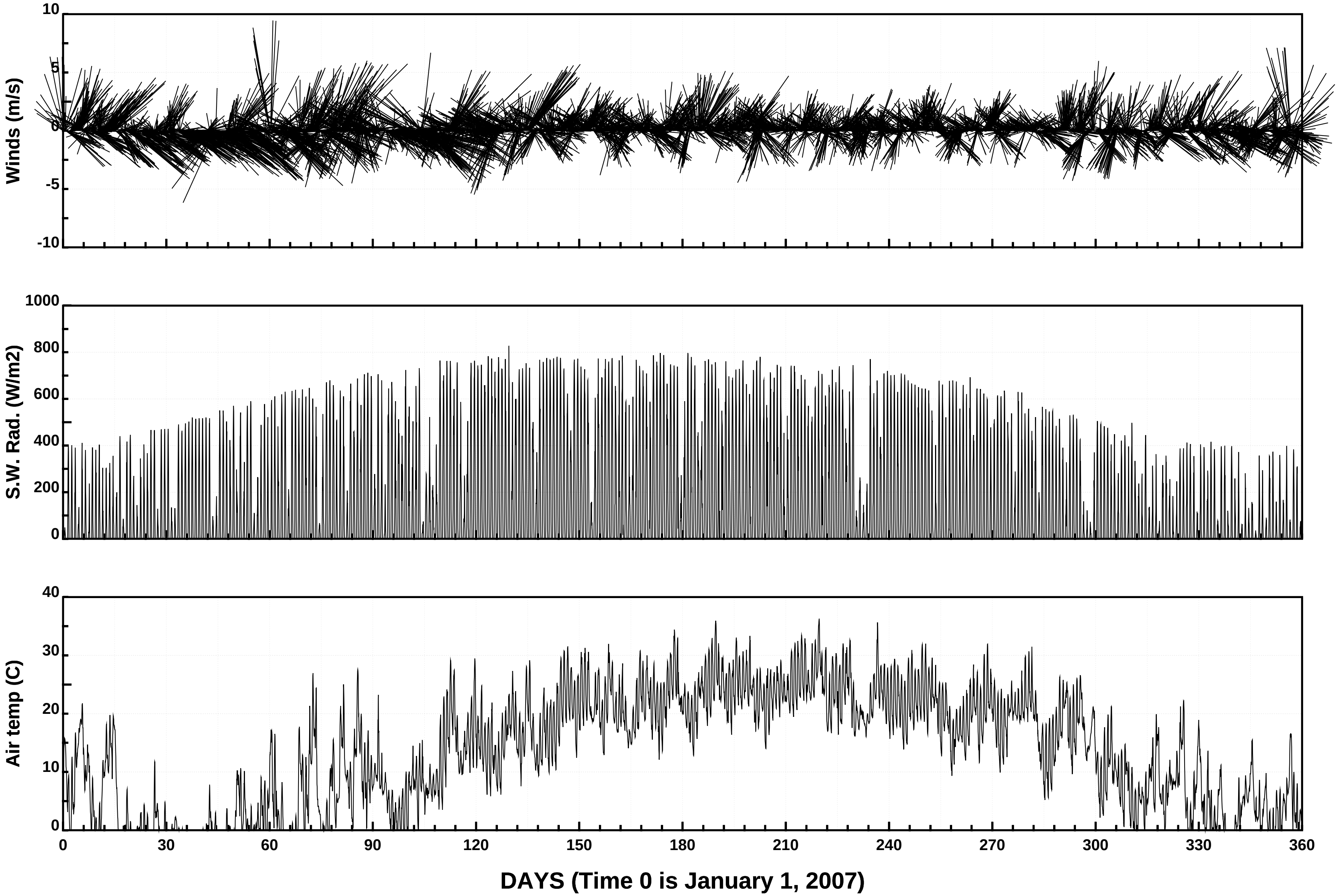

Figure 21a. ECOMSED Model Meteorological Input

RUN031: 010708*v18.thindams,defaultBFRIC,HORCONx0.1, Bowers salt,New100% flow,0 minAtemp .

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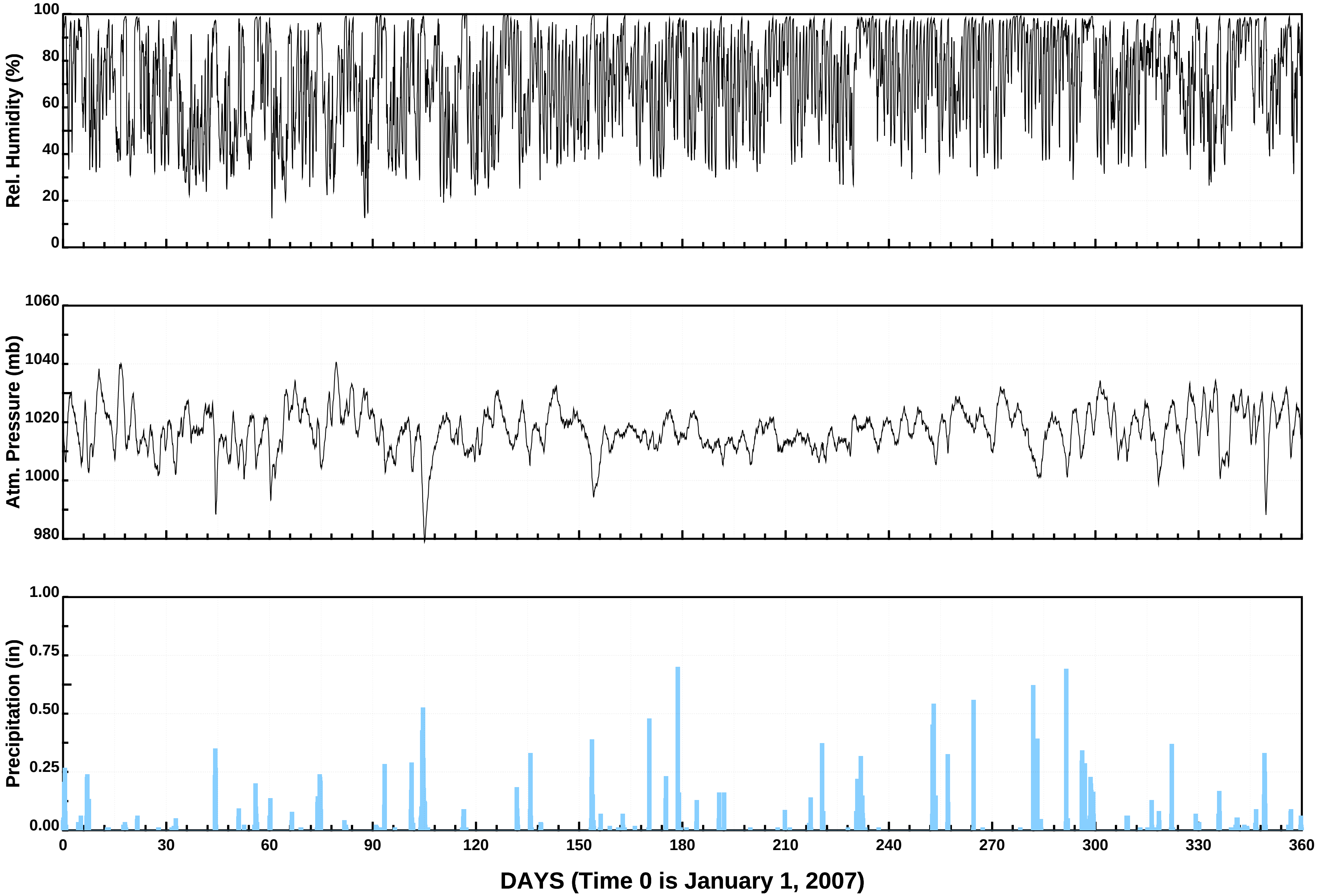

Figure 21b. ECOMSED Model Meteorological Input

RUN031: 010708*v18.thindams,defaultBFRIC,HORCONx0.1, Bowers salt,New100% flow,0 minAtemp .

DATE: 7/10/2013 TIME: 10:53:13

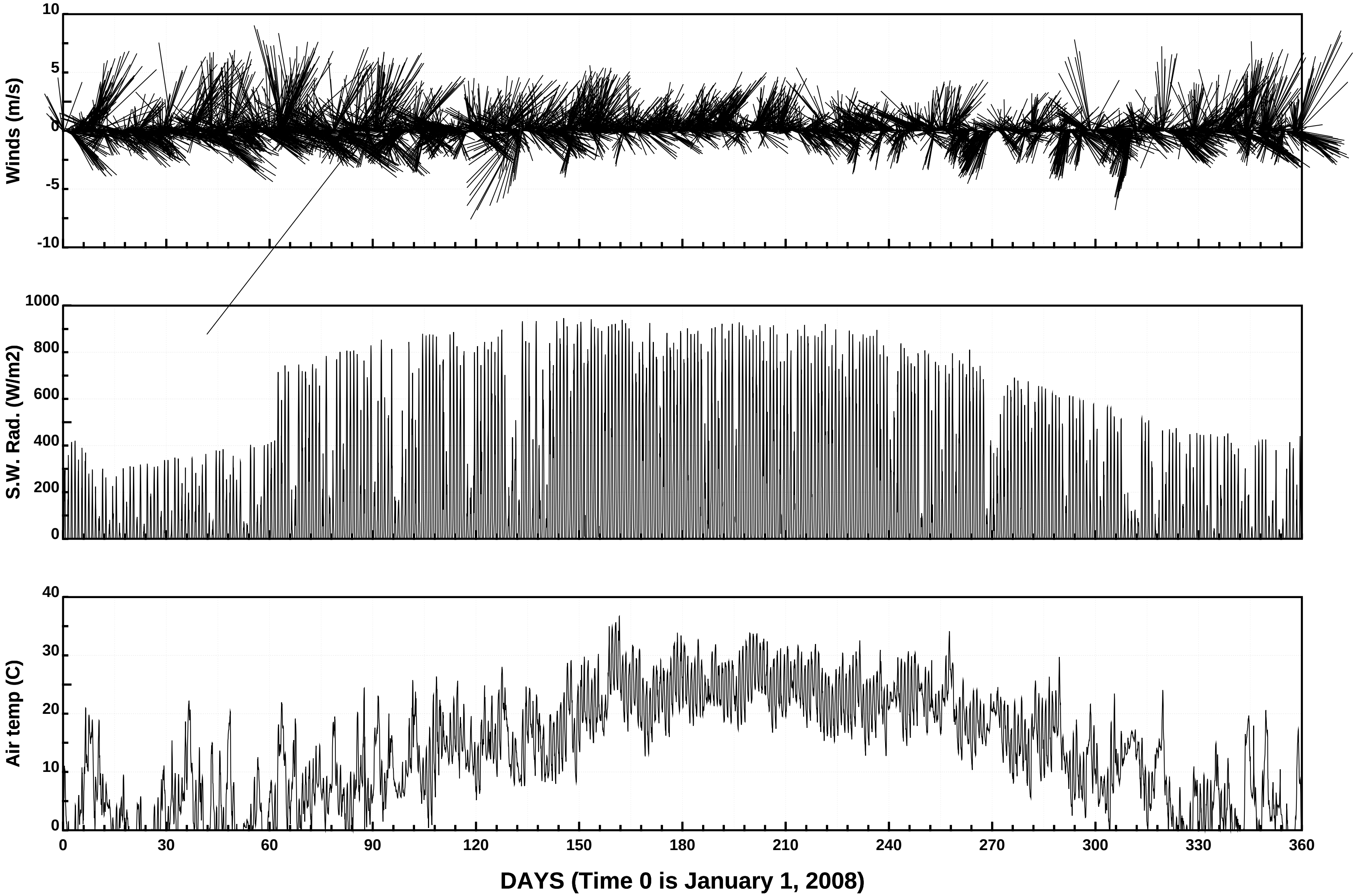

Figure 21c. ECOMSED Model Meteorological Input

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DATE: 7/10/2013 TIME: 10:53:11

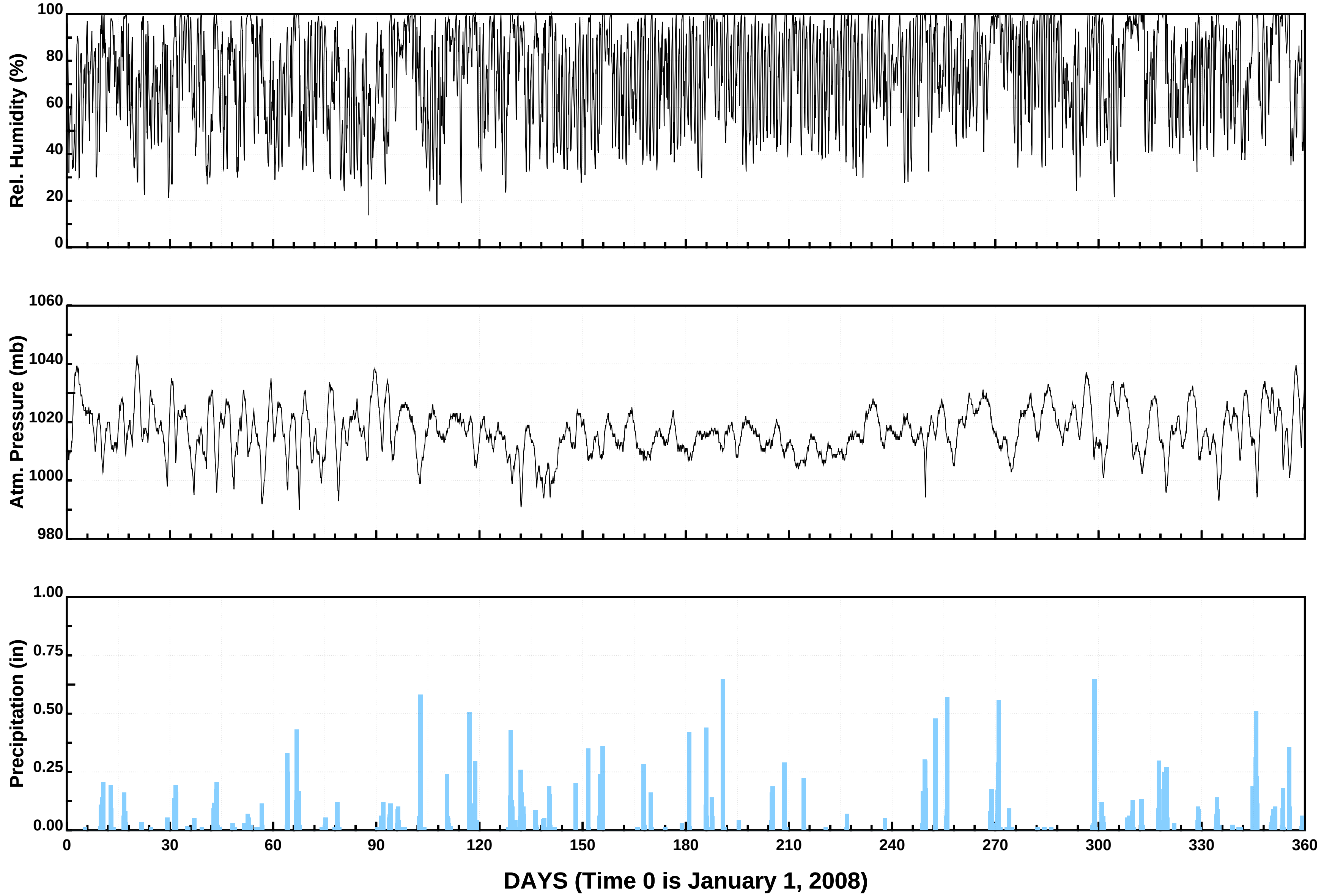

Figure 21d. ECOMSED Model Meteorological Input

08RUN003: 010708*v18.thindams,defaultBFRIC,HORCONx0.1, Bowers salt, 100% flow,0 minAtemp .

DATE: 7/10/2013 TIME: 10:53:11

3.4 MODEL CALIBRATION AND VALIDATION

The model calibration/validation period was the same as that used for the HSPF water model (January 2007 through December 2008) and provided an opportunity to model both long-term and short-term variability in the river. The ability of the hydrodynamic model to simulate advective and dispersive processes in the tidal Murderkill River was assessed by comparing model output and observed data for salinity, temperature, water elevation and tidal volume fluxes. Grab salinity and temperature data were available from DNREC monitoring at 8 stations in the tidal part of the river during 2007-2008 along with continuous salinity and temperature from USGS monitoring at 2 locations (Frederica and Bowers Beach). These data were used for model calibration/validation with the figures presented on a monthly basis in Appendix 4. The hydrodynamic model was also calibrated/validated to observed water elevations and tidal volume fluxes that were available from the two USGS continuous monitoring stations at Frederica and Bowers Beach. The model calibration/validation figures for the continuous data are presented in Appendices 5 and 6 for water elevation and tidal volume fluxes, respectively.

All of the water motions induced by small-scale processes not directly resolved by the model grid (sub-grid scale) are parameterized in terms of horizontal and vertical mixing processes and can be adjusted during the calibration/validation process. Therefore, horizontal and vertical mixing coefficients were adjusted to properly represent the mixing processes based on reproducing the observed salinity, temperature, water level and tidal volume flux data in the tidal Murderkill River. The horizontal mixing coefficients used in the calibration were adjusted using a non-dimensional parameter (HORCON, a unitless model coefficient), which was set as 0.001 in the main river channel and 0.01 in all other locations. For the vertical mixing coefficients, the background mixing values (UMOL) used in the calibration/validation were assigned 10^{-5} m^2/s in the model domain. A minimum bottom friction coefficient (BFRIC) of 0.0025 (non-dimensional) and bottom roughness coefficient (Z0B) of 0.001 meters were used throughout the model domain. All of these assigned values are within acceptable literature ranges (Blumberg and Kim, 2000; Beletsky and Schwab, 2001).

The model comparison to observed salinity and temperature data (both DNREC grabs and USGS continuous) is excellent over the 2-year calibration/validation period (Appendix 4). The model reproduces the seasonal patterns very well (lower salinity and temperature in winter/spring, higher salinity and temperature in summer/fall). Comparison of the model output to the continuous data is also very good with the model reproducing intra-tidal features of the observed salinity and temperature variations. The very good model comparison to the salinity and temperature data indicates that the advective and dispersive properties in the tidal Murderkill River are well represented in the hydrodynamic model.

The hydrodynamic model water elevations are compared to the continuous USGS data at Frederica and Bowers Beach (Appendix 5) and result in a good level of model calibration/validation. The model comparison to the data at Bowers Beach is excellent, which is expected since this location is close to the tidal boundary locations that are driven with NOAA tidal elevation observations. At Frederica, the model slightly over-predicts the tidal range during some time periods that is probably due to limited bathymetric information in the upstream reaches of the tidal Murderkill River (i.e., above Frederica) and also due to the complicated tidal interaction between the marshes and the main channel of the river. Overall though, the model reproduces water elevations in the river well. It should be noted that during January and February 2007, there appears to be an issue with the observed water elevation data at Bowers Beach such that the observed data is about 0.5 meters lower than the model output which is driven by NOAA observations at the tidal boundary locations. This issue may be due to a datum conversion for these first 2 months of 2007.

Finally, the hydrodynamic model output was compared to the continuous tidal volume fluxes measured by the USGS at Frederica and Bowers Beach to assess whether the correct water volume is moving into (negative fluxes) and out of (positive fluxes) the river and coupled marsh system (Appendix 6). In these model-data comparison figures, it should be noted that the tidal volume flux scale changes for the two sites. Overall, the model comparison to the data is very good with the model slightly over-predicting the tidal volume flux peaks during some time periods. Again, this is probably due to limited bathymetric information in the upstream reaches of the tidal Murderkill River (i.e., above Frederica) and also due to the complicated tidal interaction between the marshes and the main channel of the river. Overall though the model reproduces the change in the tidal volume flux peaks between Bowers Beach and Frederica well (7,500 cfs at Bowers Beach to 2,500-5,000 cfs at Frederica), which suggests that the correct marsh volume is assigned in the model.

Figures 22 to 25 present cross-plots of hourly model and data for salinity, temperature, water elevation and tidal volume flux at the continuous USGS monitoring locations near Bowers Beach and Frederica. Overall the model-data comparisons follow the line of perfect agreement (1:1 line) with some expected scatter about this 1:1 line. The comparisons for temperature are very good at both locations as are the comparisons for water elevation at Bowers Beach. There is more scatter about the 1:1 lines for salinity and tidal volume flux at both locations, and water elevation at Frederica. The increased scatter for water elevations (Frederica) and tidal volume fluxes at both locations is most likely due to the estimates of marsh area and volume, particularly at locations upstream from Frederica. In general, the model does reproduce the magnitudes of the water elevations and tidal volume fluxes with differences also a function of the timing of tidal wave in the model and observed.

USGS at Frederica 2007

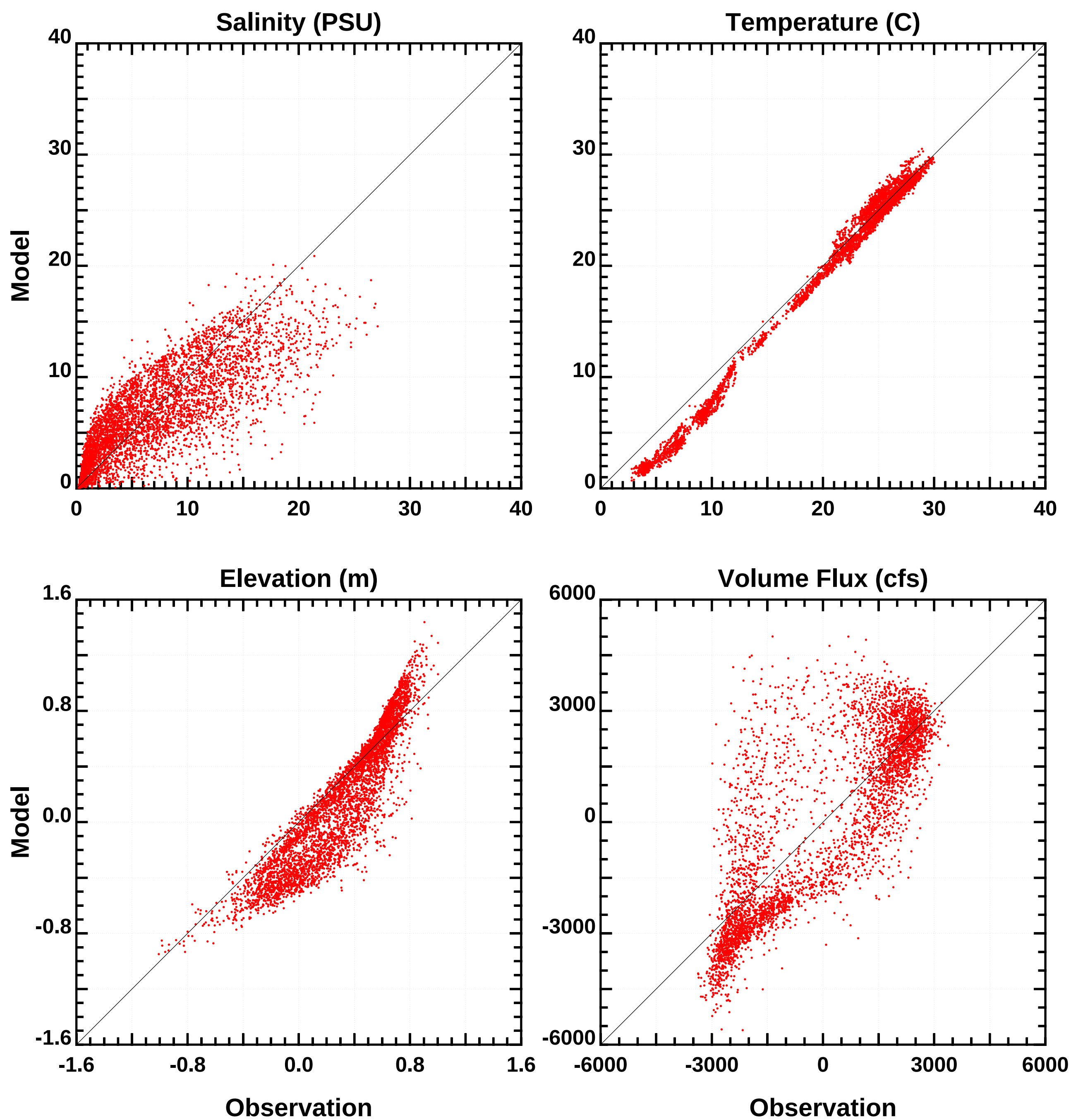

Figure 22. Hydrodynamic Model Cross-plot Comparisons to USGS Continuous Data (Hourly)

DNREC Study
/birch1/kcdw0014/HYDRO/PLOTS/TANDS/crossPlots.gdp

RUN031: 010708*v18.thindams,defaultBFRIC,HORCONx0.1, BowersDailyAvesalt-stratified-3,New100% flowp .

DATE: 7/09/2013 TIME: 18:13: 6

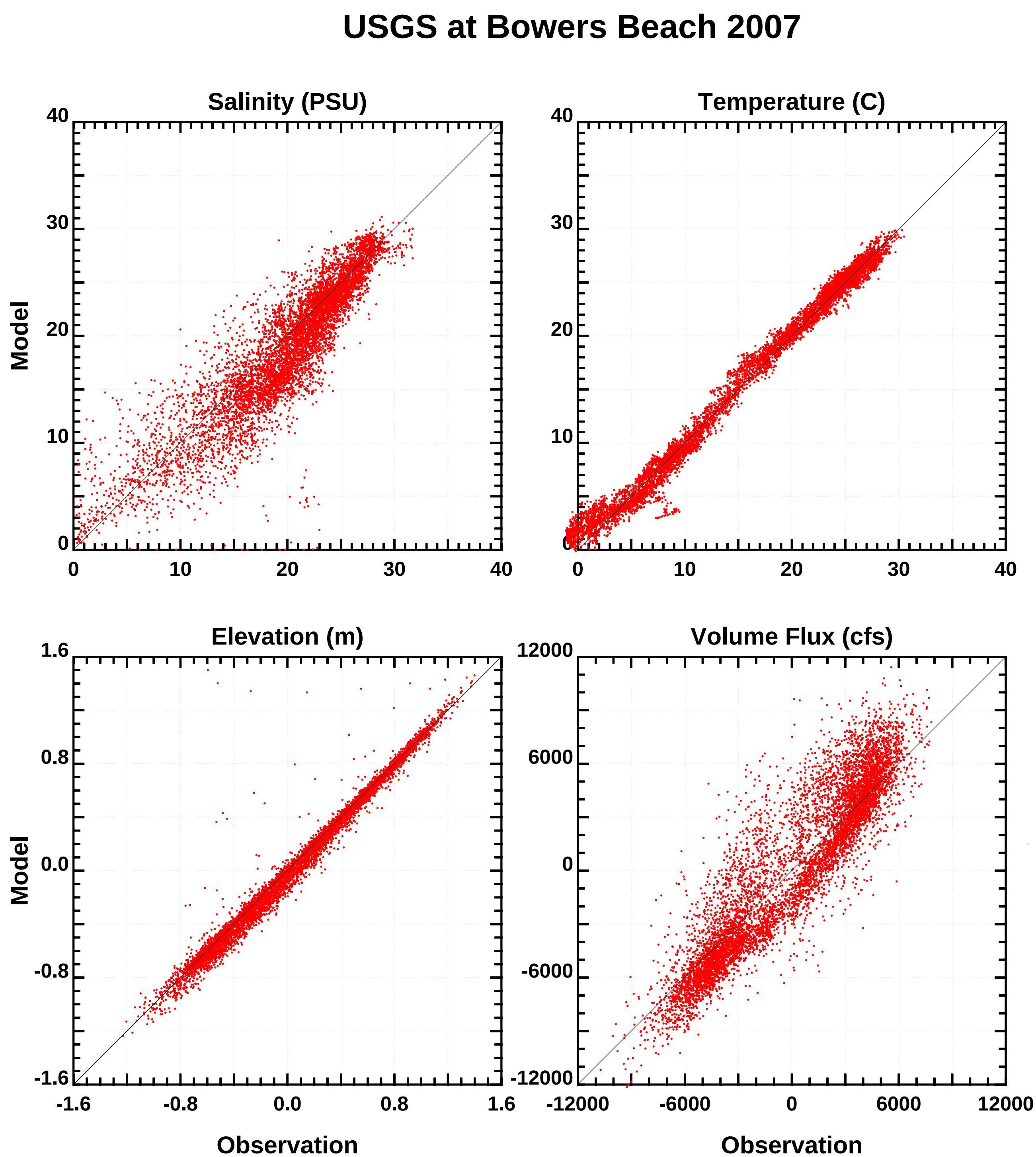

Figure 23. Hydrodynamic Model Cross-plot Comparisons to USGS Continuous Data (Hourly)

DNREC Study
/birch1/kcdw0014/HYDRO/PLOTS/TANDS/crossPlots.gdp

RUN031: 010708*v18.thindams,defaultBFRIC,HORCONx0.1, BowersDailyAvesalt-stratified-3,New100% flowp .

DATE: 7/09/2013 TIME: 18:13: 6

USGS at Frederica 2008

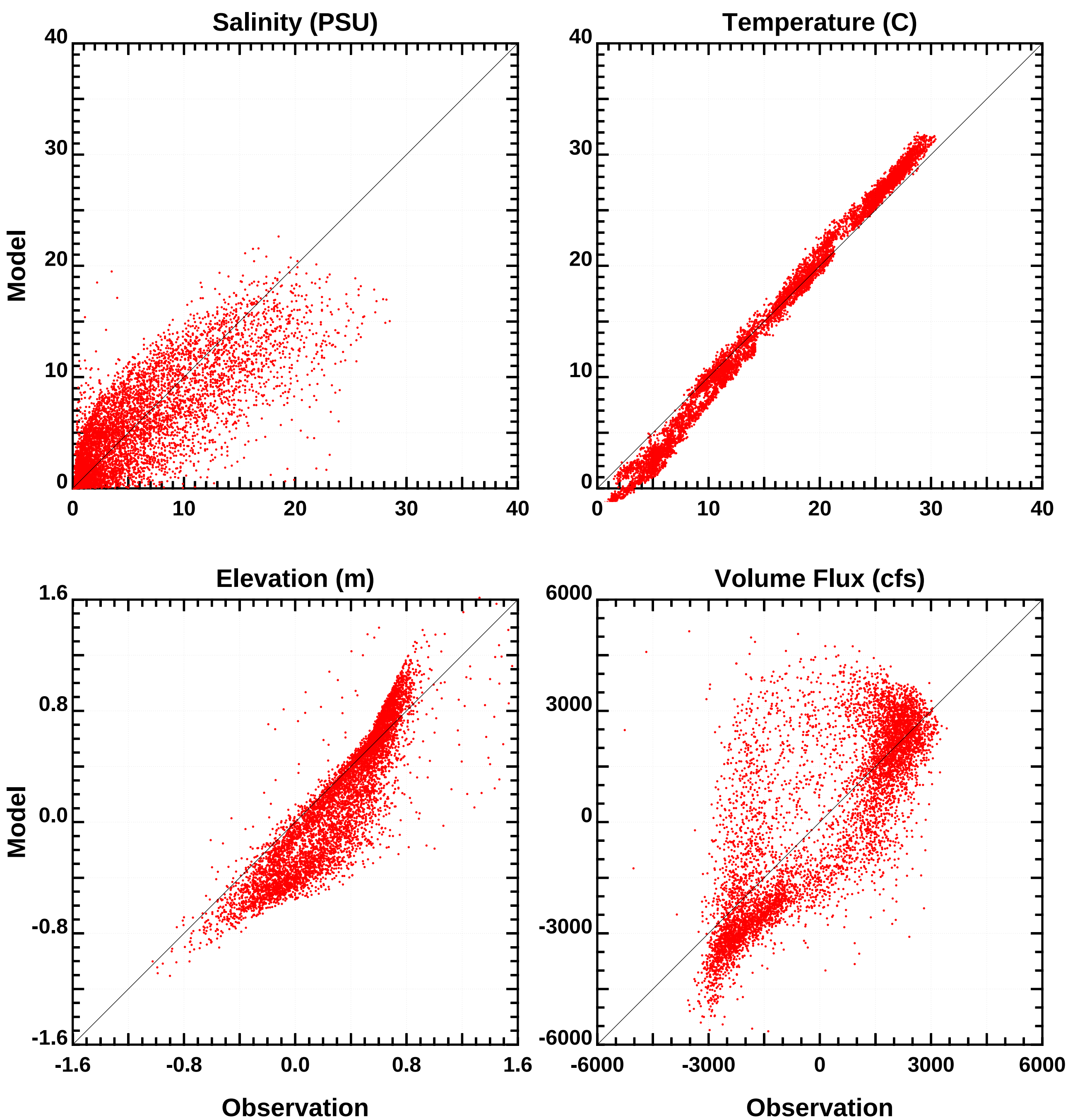

Figure 24. Hydrodynamic Model Cross-plot Comparisons to USGS Continuous Data (Hourly)

DNREC Study
/birch1/kcdw0014/HYDRO/PLOTS/TANDS/crossPlots.gdp

08RUN003: 010708*v18.thindams,defaultBFRIC,HORCONx0.1, BowersDailyAvesalt-stratified-3,New100% flows .

DATE: 7/09/2013 TIME: 18:13:23

USGS at Bowers Beach 2008

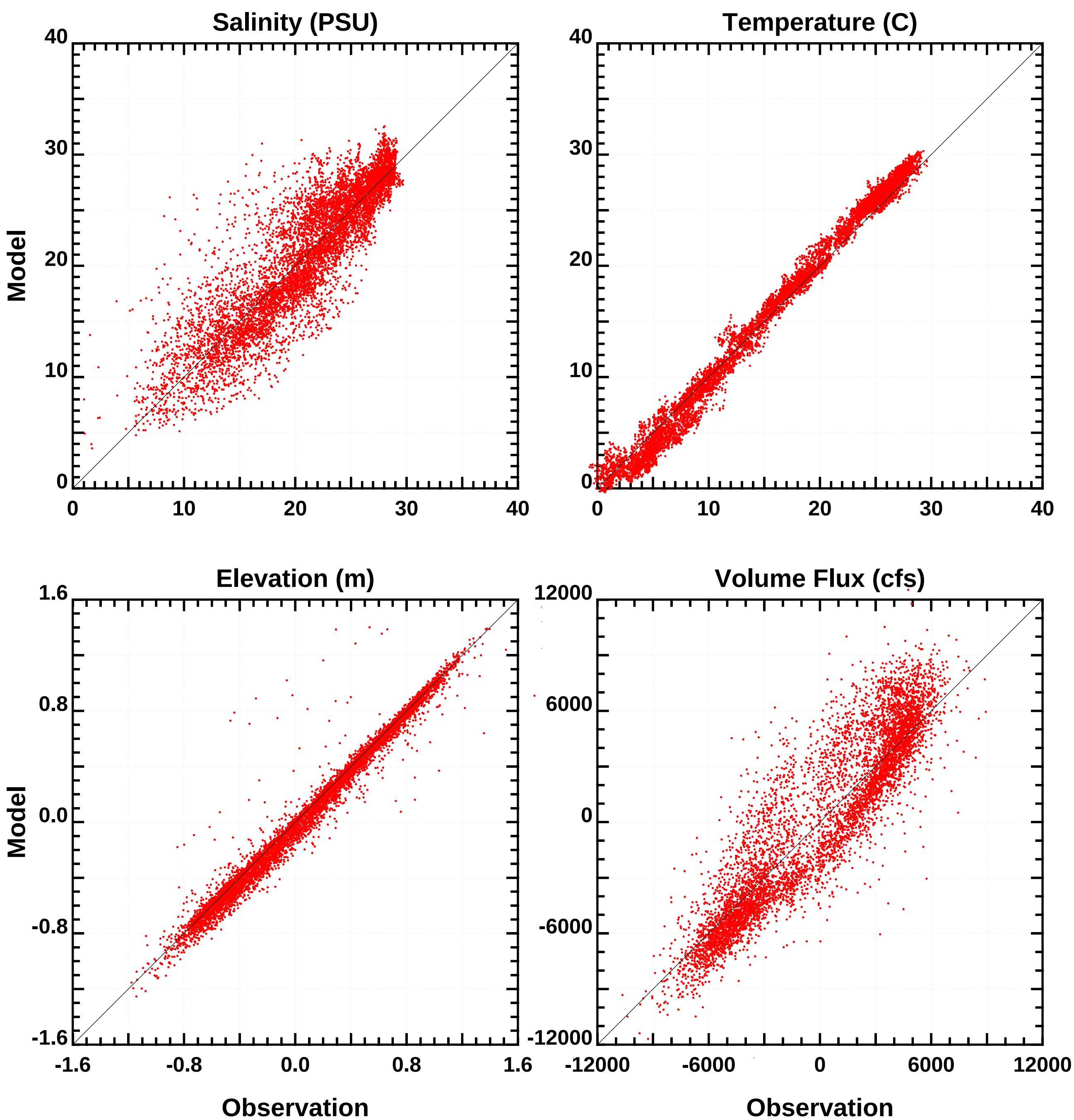

Figure 25. Hydrodynamic Model Cross-plot Comparisons to USGS Continuous Data (Hourly)

DNREC Study
/birch1/kcdw0014/HYDRO/PLOTS/TANDS/crossPlots.gdp

08RUN003: 010708*v18.thindams,defaultBFRIC,HORCONx0.1, BowersDailyAvesalt-stratified-3,New100% flows .

DATE: 7/09/2013 TIME: 18:13:23

SECTION 4

WATER QUALITY MODEL

The water quality model used in the study is a state-of-the-art eutrophication model. The water quality model is directly coupled with the hydrodynamic model and computation of water quality parameters also occurs within the tidal cycle. In addition, a fully coupled sediment flux submodel is also included in the water quality model to allow calculation of sediment oxygen demand (SOD) and sediment nutrient fluxes in response to settled organic matter and its subsequent decay in the sediment. The coupled water quality/hydrodynamic model has been successfully applied in numerous studies, including those of the Hudson-Raritan Estuary (NY/NJ), Long Island Sound (NY/CT), Chesapeake Bay (MD/DE), Massachusetts Bay and Boston Harbor (MA), Jamaica Bay (NY), Tar-Pamlico Estuary (NC), and the Upper Mississippi River (MN).

4.1 WATER QUALITY MODEL FRAMEWORK

The eutrophication model framework employed for the tidal Murderkill River was developed during the Long Island Sound (HydroQual, 1991) and Chesapeake Bay (HydroQual, 1987, 1989) modeling studies. Modeling of the exchanges of nutrients and oxygen between the water column and sediment was accomplished through the coupled sediment nutrient and oxygen flux model. The estuarine transport processes for the system were obtained from the hydrodynamic model (ECOMSED), which is described in the previous section. The nonpoint source loading input was calculated from the watershed model (HSPF). A detailed description of the eutrophication model is included in Appendix 7. The remainder of this section contains a general description of the eutrophication model framework employed in this study.

The eutrophication model includes the modeling of one phytoplankton group (although winter, summer, and fall groups are available), DO, and the various organic and inorganic forms of nitrogen, phosphorus, silica and carbon. The diagram presented in Figure 26 presents the various general kinetic pathways involved in the modeling framework. A brief description of the 26 state variables and their various kinetic pathways is presented below. The calibration/validation constants developed during the Murderkill River modeling study are presented along with a detailed discussion of the water quality kinetics in Appendix 7.

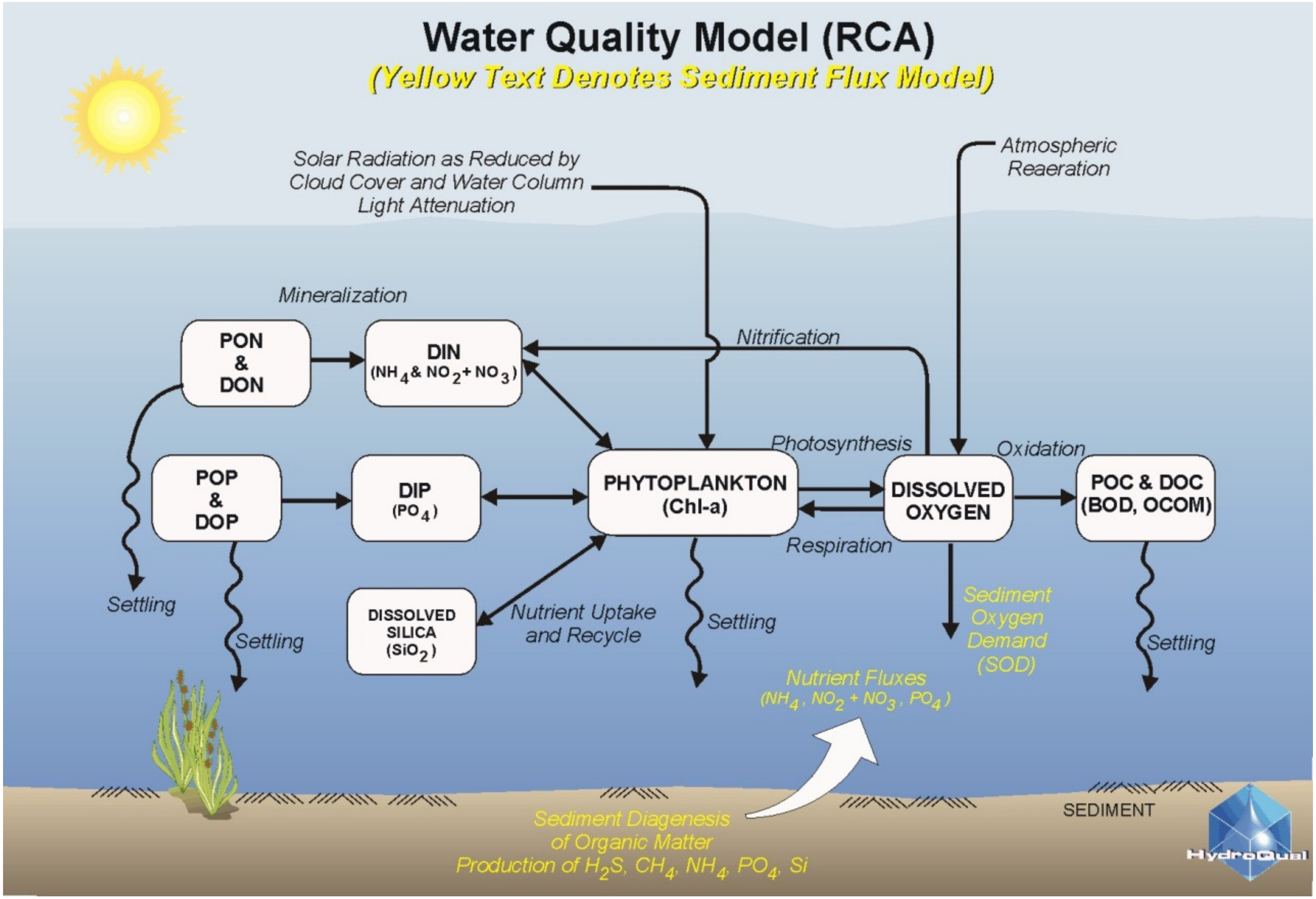

Figure 26. RCA Water Quality Model Eutrophication Kinetics Diagram

4.1.1 Phytoplankton

The eutrophication model includes three algal groups, a winter, summer, and fall population, represented by algal carbon in the model framework. The basic kinetics affecting phytoplankton growth and death are identical for the three groups, with a distinction in the assigned growth kinetic constants for each group. Phytoplankton growth is dependent upon temperature, ambient light and nutrient levels, which modify the maximum growth rate to ambient conditions. The growth rates of the three algal groups are controlled through the use of temperature optimums that maximize growth at a certain temperature and decrease growth above and below this temperature. In this manner, growth of winter, summer, and fall algal groups can peak at different times of the year or within different temperature regimes. In the Murderkill River model, only one phytoplankton group was used because the available algal data did not show multiple seasonal peaks. Ambient surface light conditions are input externally and decrease with depth as a function of light extinction coefficients calculated from measured secchi depths and vertical photosynthetically active radiation (PAR) data. The surface light conditions are based upon ambient measurements of solar radiation measured within the day. Algal growth is further decreased when the ambient nutrients (phosphorus, nitrogen and silica) approach their respective limiting concentrations. Nutrient limitation factors, defined in Appendix 7, are calculated for phosphorus, nitrogen and silica, with the minimum factor chosen to adjust the growth rate. The ambient growth rate, which is adjusted for temperature, light and nutrient limitations, is then used to determine the oxygen produced through photosynthesis during growth.

The loss of biomass from the water column through respiration, zooplankton grazing and settling is identical between the three algal groups. The respiration formulation for each algal group uses a variable respiration rate, which is a function of the ambient growth rate plus a minimum basal rate. During respiration, DO is consumed and nutrients are recycled to the phosphorus, nitrogen and silica systems. Zooplankton grazing is accounted for through a temperature-dependent decay rate and recycles nutrients and carbon. Algal settling to the sediment is a temperature-dependent process that increases as the nutrient limitation factor decreases (nutrient-stressed settling).

4.1.2 Phosphorus

Particulate and dissolved organic phosphorus forms are included in the model, with further distinctions based upon reactivity. These reactivity distinctions, in turn, are based upon relative decay rates for the organics. A labile fraction describes organic material that decays on a time scale of several weeks to a month or two, while a refractory fraction accounts for decay processes lasting months to a year. The labile fractions decay primarily in the water column or else rapidly in the sediments; the refractory components mainly decompose in the sediments. The inorganic form of phosphorus, orthophosphate (PO_4), is also modeled, for a total of five state variables for phosphorus (Table 9).

Table 9. Phosphorus State Variables	
Refractory Particulate Organic Phosphorus	(RPOP)
Labile Particulate Organic Phosphorus	(LPOP)
Refractory Dissolved Organic Phosphorus	(RDOP)
Labile Dissolved Organic Phosphorus	(LDOP)
Orthophosphate	(PO_4)

Particulate organic phosphorus, whether refractory or labile, decomposes to dissolved organic phosphorus through hydrolysis, which is a temperature- and bacterial biomass-mediated reaction. The size of the bacterial population involved in decomposing organic compounds in the water column affects the rate at which this process occurs. Because bacterial biomass is not directly modeled, algal biomass is used as a surrogate tracking variable for computational purposes. The particulate fraction of organic phosphorus settles within the water column at a temperature-dependent rate and is deposited to the sediment where it is further decomposed through anaerobic processes. The dissolved form of organic phosphorus further decomposes through mineralization into the inorganic form of phosphorus (PO_4), which is affected by the same factors controlling hydrolysis. Inorganic phosphorus, PO_4, is lost through its utilization by algae as a nutrient essential for growth and is supplied from or lost to the sediment through sediment fluxes. All forms of phosphorus, organic and inorganic, are supplied as a consequence of algal respiration and zooplankton grazing, which is termed algal nutrient recycling. Inputs of organic and inorganic phosphorus from the atmosphere, boundaries, tributaries, nonpoint sources and point sources are also accounted for in the modeling framework.

4.1.3 Nitrogen

Organic nitrogen is divided into the same four components or state variables as organic phosphorus. The addition of two inorganic forms of nitrogen, ammonia (NH_3) and nitrite plus nitrate nitrogen (NO_3) produce a total of six state variables for nitrogen (Table 10).

The particulate and dissolved forms of nitrogen decompose through the same reaction pathways as phosphorus, with the particulate fractions settling to the sediment. The dissolved organic forms mineralize to ammonia, which is subsequently nitrified to nitrite and nitrate via a reaction in which DO is consumed. Nitrification is an aerobic reaction; therefore, the reaction decreases as DO concentrations decrease below a certain value. The nitrification reaction is, therefore, dependent upon water column DO concentrations as well as temperature. The denitrification of nitrate to nitrogen gas is an anaerobic reaction that varies with temperature.

Ammonia and nitrite plus nitrate are utilized by algae as nutrients for growth with ammonia being the preferred nutrient. A preference scheme for determining ammonia or nitrite plus nitrate preference at varying concentrations is presented in Appendix 7. Algal nutrient recycling replenishes the four organic forms of nitrogen and ammonia during algal respiration and zooplankton grazing. Sediment fluxes of ammonia and nitrate are either a source of or sink for these nutrients in the water column. External inputs of all forms of nitrogen are also accounted for within the model.

Table 10. Nitrogen State Variables	
Refractory Particulate Organic Nitrogen	(RPON)
Labile Particulate Organic Nitrogen	(LPON)
Refractory Dissolved Organic Nitrogen	(RDON)
Labile Dissolved Organic Nitrogen	(LDON)
Ammonia Nitrogen	(NH_3)
Nitrite plus Nitrate Nitrogen	(NO_3)

4.1.4 Carbon

Organic carbon is divided into the same groups as organic nitrogen and phosphorus, with three additional state variables. Highly reactive dissolved organic material, such as carbonaceous inputs associated with sewage treatment plants or combined sewer outfalls which decay on a time scale of days to a week or two, is classified as reactive dissolved organic carbon. Reactive particulate organic carbon is assumed to have a very high setting rate that is a function of REPOC itself representing flocculation. Excretion of dissolved organic carbon by phytoplankton during photosynthesis is included as the sixth state variable, algal exudate. Algal exudate decays on a time scale similar to that for reactive dissolved organic carbon. The seven state variables described for carbon are shown in Table 11.

The particulate and dissolved forms of carbon decompose through the same reaction pathways as phosphorus and nitrogen, with the particulate fractions settling to the sediment. The dissolved forms of carbon oxidize to carbon dioxide, using DO during the process. Oxidation of dissolved organic carbon is aerobic and, therefore, reduced at low water column DO concentrations. The oxidation process is also modified by temperature and bacterial biomass levels, which are indirectly represented by algal biomass. Algal recycling due to zooplankton grazing is a source of both refractory and labile particulate and dissolved organic carbon. External inputs of organic carbon are also included in the modeling framework.

Table 11. Carbon State Variables	
Refractory Particulate Organic Carbon	(RPOC)
Labile Particulate Organic Carbon	(LPOC)
Refractory Dissolved Organic Carbon	(RDOC)
Labile Dissolved Organic Carbon	(LDOC)
Reactive Dissolved Organic Carbon	(REDOC)
Reactive Particulate Organic Carbon	(REPOC)
Algal Exudate Dissolved Organic Carbon	(EXDOC)

4.1.5 Silica

Two silica forms are included in the model: particulate biogenic silica, which is unavailable for algal growth, and silica, which is available for algal growth (primarily for diatoms). Particulate biogenic silica is mineralized to available silica at a temperature- and bacterial biomass-dependent rate and can also settle to the sediment. Available silica is utilized as a nutrient during algal growth and can interact with the sediment through silica fluxes. Algal recycling supplies the particulate biogenic silica system through algal respiration and zooplankton grazing. The two state variables for silica are shown in Table 12.

Table 12. Silica State Variables	
Biogenic Silica – Unavailable	(BSI)
Silica – Available	(SI)

4.1.6 Dissolved Oxygen

Levels of DO are affected by the nitrification of ammonia, denitrification of nitrate, oxidation of dissolved organic carbon or biochemical oxygen demand (BOD), algal oxygen production and respiration, sediment oxygen demand (SOD) and atmospheric reaeration. The SOD is calculated via the coupled sediment flux submodel. DO saturation is computed from water column temperature and salinity obtained from the hydrodynamic model. The effects of algal photosynthesis and respiration on DO are briefly described in the previous phytoplankton section.

4.1.7 Sediment Flux Submodel

A sediment flux submodel is incorporated into the eutrophication model. The sediment receives fluxes of particulate organic carbon (POC), particulate organic nitrogen (PON), and particulate organic phosphorus (POP), which are collectively referred to as particulate organic matter (POM). The water column model state-variables that are deposited to the sediment include: detrital algae, labile and refractory POC, labile and refractory PON, labile and refractory POP. The fluxes of these variables make up the incoming sources of particulate organic matter to the sediment. Mineralization, which is termed diagenesis, produces soluble end products. These products can react in the aerobic and anaerobic layers of the sediment. The difference between the resulting aerobic dissolved concentration and the overlying water concentration determines the flux to or from the sediment. The magnitude of the flux is determined by the surface mass transfer coefficient.

It is important to model an annual cycle with the sediment submodel because of the storage capacity of the sediments and the subsequent effects on nutrient fluxes and SOD. Organic matter (nitrogen, phosphorus and carbon) deposited in the sediments during the winter and spring undergoes slower decay pathways due to the cooler temperatures. When the temperature increases during the summer months, stored organic matter decays at a faster rate, which results in different nutrient fluxes and SOD. Without the modeling of an annual cycle (i.e., modeling of summertime alone), summer nutrient fluxes and SOD would be under-computed because stored organic matter would not be included. Compared with the water column, the sediment takes a longer time to reach steady state. It is important to perform iterations of the sediment model runs so that the sediment concentrations reach a quasi-steady state with the water column.

4.2 PROJECT SPECIFIC STUDY DATA

Five project specific studies were completed to support the Murderkill River Study that provided data to support the water quality model setup and calibration/validation efforts. These data included: the USGS continuous monitoring at Bowers Beach (#01484085) and Frederica (#01484080) for DO that were used for model calibration/validation; long term BOD (LTBOD) studies completed on river samples (DNREC) and KCRWTF effluent (KCDPW); algal production studies completed at seven tidal river stations (Sharp, 2011); tidal marsh nutrient, carbon and DO deficit load studies completed in Webb's Marsh (Wong et al., 2009; Dzwonkowski et al., 2013; Ullman et al., 2013); and sediment flux studies completed on cores in the tidal river (CBA, 2010).

The LTBOD studies were used to develop a relationship between the more frequently measured BOD_5 parameter and ultimate BOD (BODu), which is used for model calibration and conversion of model inputs to carbon units that are needed for model setup. This is required for both river and KCRWTF effluent samples. In addition, the LTBOD studies provide an estimate of the BOD oxidation rate, which is used to assign this constant in the model. In general, the LTBOD tests involve incubation of water or effluent samples in a BOD bottle (similar to a BOD_5 test) with a longer incubation period (about 60 days) so that an ultimate oxygen demand of the sample can be determined. In addition, samples are taken over the incubation period for the measurement of NO_2+NO_3 to determine if any build-up is occurring that represents nitrification. LTBOD tests were completed at 16 river stations (5 freshwater sites, 4 lake/pond sites, 7 tidal sites) during 6 sampling events in 2007-2008. For the KCRWTF effluent, LTBOD tests were completed 15 times on a roughly monthly basis in 2007-2008. Appendix 8 presents the LTBOD data and analysis results for the river samples and Appendix 9 presents the LTBOD for the KCRWTF effluent samples. Regression analyses are completed on the BOD data to determine the BODu and oxidation rate (Kd) using the following equation:

$$BOD_t = BOD_u\left(1 - e^{-K_d \times time}\right)$$

where: BODt – measured BOD at time t (mg/L);

BODu – ultimate BOD (mg/L);

Kd – BOD oxidation rate (1/day); and

time – time of measurement (day).

Table 13 presents the results from the LTBOD studies on both the river and effluent samples during the 2007-2008 modeling period. Based on these results, a BOD oxidation rate of 0.05/day was used that is also used to convert model calculated organic carbon to BOD_5. For the KCRWTF effluent, a BODu/$CBOD_5$ ratio of 4.0 was used to convert the daily $CBOD_5$ data to BODu and carbon units for assigning the LDOC load in the model. In addition, a LPOC load was assigned for the KCRWTF based on measured VSS data and a POC:VSS ratio of 0.4.

The algal production data collected by the University of Delaware (Sharp, 2011) was used for adjusting the model phytoplankton growth rate during calibration/validation so that model calculated ambient growth rates compared favorably to the algal production data. Algal production tests were completed at seven tidal river stations from Bowers Beach to Frederica on 22 dates in the 2007-2008 modeling period. Primary production was estimated through incubation of samples over a 24-hour period at varying light levels to simulate different depths in the water column from the surface down to the 1.5% light depth. The maximum algal production (Pmax) results in mgC/L/d (typically occurring at the surface) were compared to model calculated surface values as part of the calibration/validation process. From these model-data comparisons, the final maximum algal growth rate of 2.5/day was assigned. A summary of the algal production data collected in 2007-2008 is presented in Table 14. The variation in the Pmax data is related to the time of the year (i.e., lower values in the winter with higher values in the spring/summer/fall). Details of the algal production data are presented in Algal Production Study report (Sharp, 2011).

Table 13. Summary of LTBOD Study Results					
Station	**$CBOD_5$ (mg/L)**	**$CBOD_{20}$ (mg/L)**	**BODu (mg/L)**	**Kd (1/d)**	**BODu/ $CBOD_5$ Ratio**
Non-Tidal River					
206011	2.4	2.4-6.2	3.0-7.0	0.033-0.080	1.2-2.9
206041	2.4-2.7	2.4-10.9	7.7-56.8	0.029-0.088	3.2-16.2
206051	2.4	2.4-4.5	3.4-10.3	0.069-0.135	1.4-4.3
206641	2.4	2.4-4.5	2.4-5.7	0.048-0.068	1.0-2.4
206561	2.4-5.2	2.4-13.2	4.1-21.2	0.052-0.081	1.7-6.6
Lakes/Ponds					
206071	2.4-5.6	3.6-14.6	5.3-27.0	0.054-0.066	2.2-6.8
206361	2.4-3.3	3.7-11.1	6.7-19.8	0.062-0.090	2.8-7.6
206451	2.4-5.4	4.0-14.2	5.9-49.4	0.041-0.074	2.5-9.1
206461	2.4-7.0	2.4-16.6	5.6-25.6	0.049-0.112	2.3-8.2
Tidal River					
206081	2.4-4.9	2.9-12.1	6.9-20.3	0.043-0.067	2.9-6.7
206091	2.4-3.0	2.7-11.8	7.5-19.1	0.054-0.074	3.1-6.5
206231	2.2-2.4	3.4-6.5	5.2-12.5	0.046-0.116	2.3-5.2
206711	2.4	3.1-8.6	7.6-12.1	0.024-0.070	3.2-5.1
206141	2.4	3.5-7.1	6.9-15.9	0.029-0.075	2.9-6.6
206131	2.4-4.8	4.1-11.0	4.8-16.7	0.034-0.065	1.5-7.0
206101	2.4-5.1	2.4-11.6	7.0-14.2	0.038-0.072	2.2-5.9
KCRWTF					
Effluent	2.4-9.0	n/a	6.7-20.1	0.042-0.077	2.1-4.6

Table 14. Algal Production Data Summary	
Station	**Pmax (mgC/L/d)**
Bowers Beach (#206101)	0.12-1.80
Webbs Landing (#206131)	0.01-1.88
Milford Neck (#206141)	0.05-1.08
Power Lines (#206711)	0.01-1.62
KCRWTF Canal (#206231)	0.03-1.59
Bay Road (#206091)	0.03-2.47
Frederica Road (#206081)	0.04-2.24
Frederica Road-dup (#206081)	0.04-2.49

The tidal marsh studies completed in Webb's Marsh (Wong et al., 2009; Dzwonkowski et al., 2013; Ullman et al., 2013) provided estimates of nutrient, carbon and DO deficit loads that were then extrapolated to the rest of the Murderkill River tidal marsh area for use in assigning these loads in the model. The DO deficit load represents the difference between the ebb DO levels and the flood DO levels. Typically, the flood DO levels (incoming) are greater than the ebb DO levels (outgoing) due to oxygen consumption in the marsh and this process was reflected in the model input setup. The tidal marsh studies completed involved measurement of the various forms of organic and inorganic nitrogen and phosphorus, organic carbon, chl-a, silica and suspended solids. For the organic forms both particulate and dissolved fractions were measured. The data were collected near the mouth of Webb's Marsh with the Murderkill River approximately every hour over roughly two to three complete tidal cycles in July 2007, October 2007, April 2008, May 2008 and August 2008. In addition, the USGS had a continuous gage located at the same location that recorded salinity, temperature, DO, water elevation and tidal volume flux over the 2007-2008 modeling period. Through analysis of both the water quality data and USGS tidal data at the mouth of the marsh, nutrient, carbon and DO deficit loads were calculated for each monitoring period. These loads were used to define the tidal marsh loads in the model by normalizing them with the active Webb's Marsh area (157 acres) and extrapolating to the rest of the Murderkill River tidal marsh area based on the LiDAR study (McKenna, 2013) that developed tidal marsh areas by zones in the river (Figure 27). The marsh nutrient and POC loads were assigned based on the average loads from each survey except May 2008, which was affected by a large coastal storm that flooded the entire Delaware Bay marsh systems and affected the monitoring. For the DOC and DO deficit loads, monthly variable loads were assigned. Details of the load calculations are presented in the Webb's Marsh Study reports (Wong et al., 2009; Dzwonkowski et al., 2013; Ullman et al., 2013). Table 15 presents a summary of the areal marsh loads assigned in the model.

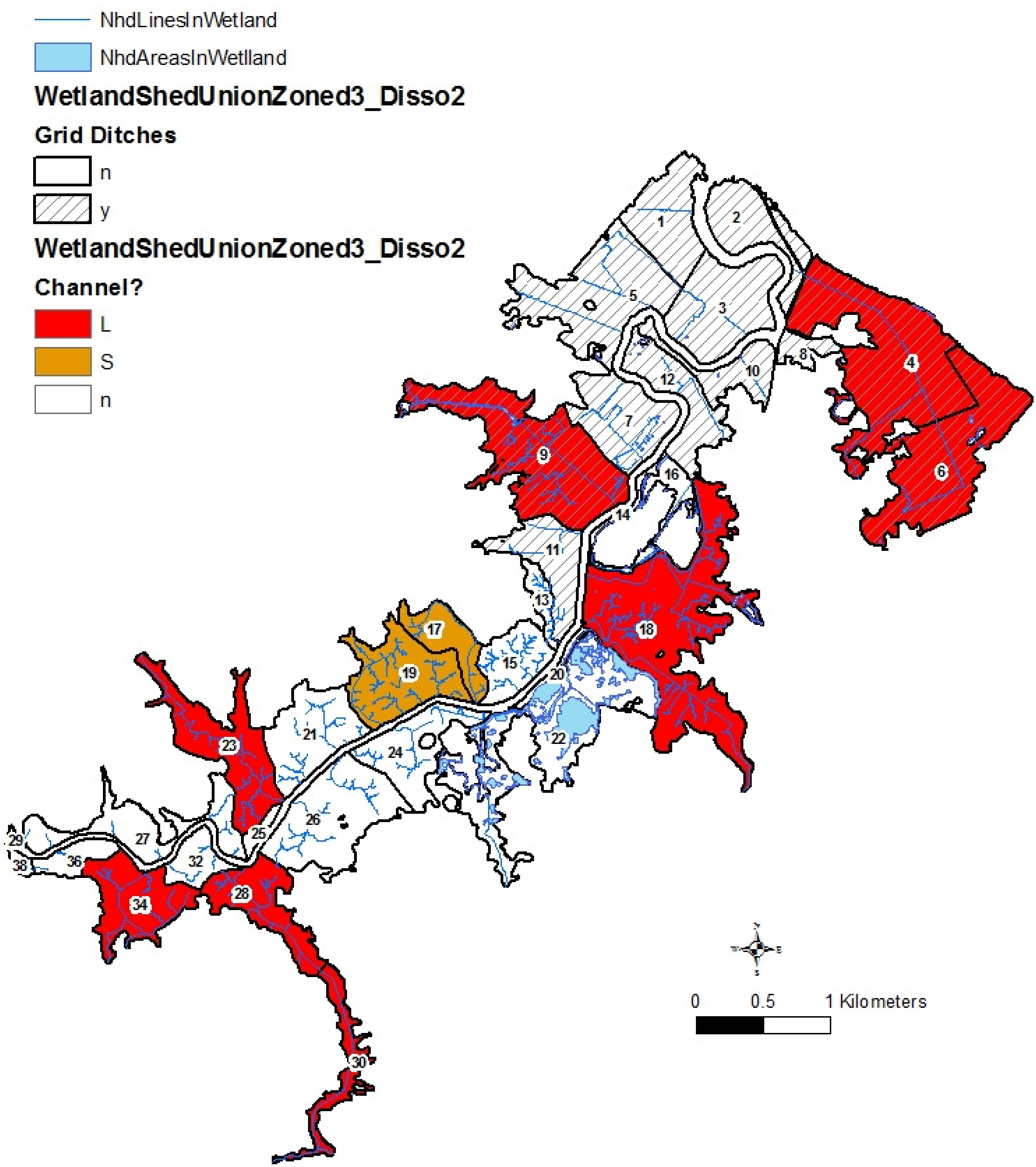

Figure 27. Murderkill River Tidal Marsh Zones based on LiDAR Data

(Ref: McKenna, 2013)

Table 15. Summary of Tidal Marsh Loads ($g/m^2/yr$)						
Parameter	**July 2007**	**October 2007**	**April 2008**	**May 2008**	**August 2008**	**Average**
DOC*	11.38	2.92	5.29	225.22	8.50	7.02
POC	22.07	2.06	-26.09	12.84	4.73	0.69
TOC	33.44	4.98	-20.80	238.06	13.23	7.71
NH_4	1.19	0.33	0.86	2.12	-0.31	0.52
NO_3	0.40	-0.05	-2.33	-4.70	-0.19	-0.54
DON	4.15	0.07	2.14	16.95	1.99	2.09
PON	-0.12	0.18	-4.35	-1.04	0.70	-0.90
TON	4.03	0.24	-2.21	15.92	2.69	1.19
PO_4	0.57	0.51	0.09	0.13	1.02	0.55
DOP	-0.04	-0.07	0.05	0.98	0.51	0.11
PP	0.08	0.03	n/a	0.58	-0.02	0.03
TOP	0.04	-0.03	0.05	1.55	0.49	0.15
DO Deficit**	n/a	n/a	n/a	n/a	n/a	n/a

* - DOC loads were assigned on a monthly basis and ranged from 0.52-71.61 $g/m^2/yr$.
** - DO deficit loads were assigned on a monthly basis and ranged from -72.48-1.30 $g/m^2/yr$.

The sediment flux studies completed in the tidal Murderkill River (CBA, 2010) provided estimates of sediment oxygen demand (SOD) and nutrient fluxes at four locations in the main stem of the river (July 2007) and six tidal marsh sites (July 2007 and April 2008). SOD and nutrient fluxes were determined from the collection of sediment cores that were incubated for approximately 5 hours with the measurement of DO and nutrients over time completed. From these data, regressions are completed to determine the sediment areal uptake or production rate of oxygen, NH_3, NO_2+NO_3, PO_4, N_2 gas and silica. The river data are used for calibration/validation of the sediment flux model and the marsh data used for an initial starting estimate of the tidal marsh denitrification rate. The final tidal marsh denitrification rate was determined from model calibration/validation and was assigned at 15 $gN/m^2/yr$. Table 16 presents a summary of the sediment flux data used for model calibration/validation. Details of the sediment flux measurements are presented in Sediment Flux Study report (CBA, 2010).

Table 16. Summary of River Sediment Flux Studies						
Station	**Date**	**SOD ($g/m^2/d$)**	**NH_4 ($mg/m^2/d$)**	**NO_2+NO_3 ($mg/m^2/d$)**	**N_2 ($mg/m^2/d$)**	**PO_4 ($mg/m^2/d$)**
Webb Landing (1)	July 2007	0.30	12.6	-9.0	22.8	1.8
Powerlines (2)	July 2007	0.86	0.7	-29.4	57.4	-6.1
KCRWTF Canal (3)	July 2007	1.20	179.3	-80.2	136.5	9.5
Frederica (4)	July 2007	1.05	63.8	-22.7	63.3	-10.5

These data and their uses in model calibration/validation are discussed in more detail in the following sections.

4.3 WATER QUALITY MODEL INPUTS

Model calibration and validation to the 2007-2008 modeling period required the time-variable input of boundary condition data for all of the 26 state variables. Hydrodynamic transport processes are directly coupled with the water quality model, so that water quality computations occur within the tidal cycle. The circulation obtained from the hydrodynamic model is averaged over a 1-hour period before transfer to the water quality model. The tidal circulation information passed to the water quality model includes: flows; dispersive fluxes; volumes; water elevations; depths; salinity; and temperature. Boundary condition locations are the same as for the hydrodynamic model (i.e., upstream freshwater tributary boundaries and downstream tidal bay boundaries). Boundary conditions were setup based on available data in the estuary including data from: DNREC monitoring data; and DRBC Boat Run Monitoring Data. Details of the water quality model boundary conditions, loads, constants, and other inputs are discussed in detail next. The water quality model calibration and validation section will present a summary of the model calibration and validation, highlighting important parameters and some key modeling issues.

4.3.1 Solar Radiation and Light Extinction

The ambient light level is a major factor controlling the growth of phytoplankton in an aquatic environment; therefore, it must be accurately represented in any modeling analysis. Ambient light levels can be determined directly from measurements of solar radiation near the water surface, or indirectly, from empirical relationships relating cloud cover to solar radiation. In the Murderkill River watershed study, ambient light levels were obtained from direct measurement of solar radiation within the day during the entire calibration/validation period at the DNERR St. Jones River meteorological station located in central Kent County about 10 miles north of the river.

Figure 21 presents the daily average solar radiation for 2007 and 2008. The data suggest that the solar radiation levels for the model validation period (2008) were higher than that for the model validation period (2007) and generally show a decreasing trend in the fall and winter seasons.

Another important factor controlling the growth of phytoplankton is the surface light attenuation with depth (light extinction). Available ambient light decreases with depth due to turbidity, which can be caused by suspended solids, color, and also phytoplankton. The contribution to light extinction by phytoplankton is termed "algal self shading." Light extinction coefficients can be determined directly, through the analysis of PAR measurements with depth, or indirectly, using relationships between light extinction and secchi depth measurements.

The light extinction coefficients (Ke) in the tidal Murderkill River were calculated from vertical PAR data collected by the University of Delaware during the algal production studies in the river during 2007 and 2008 (Sharp, 2011). These calculated light extinction coefficients reflect the total light extinction in the water column and, therefore, were corrected for algal self shading to a base light extinction coefficient for model input. The algal self shading correction was calculated as 0.017*chl-a. Although the calculated light extinction coefficients were not available during every month of the 2-year modeling period, the available data were used to interpolate to months when data was not available. In the tidal river, the base light extinction coefficients ranged from 1.5-6.5/m and averaged 4.2/m over the 2-year modeling period. In Delaware Bay, light extinction coefficients were calculated from available secchi depth (SD) data using the following equation: total Ke = 1.4/SD and corrected for algal self shading. A constant base light extinction coefficient of 0.71/m was used in Delaware Bay. These calculated light extinction coefficients include the effects of both non-algal sources (suspended solids, color, etc.) and algal self-shading.

4.3.2 Initial Conditions

The initial conditions for each state variable for the water quality model calibration in the year 2007 were set up using DNREC water quality data in the Murderkill River and DRBC Boat Run data in the bay. The initial conditions for the sediment model, including sediment temperature, particulate organic matter (PON, POP, POC) in three reactivity classes (G classes), inorganic nutrients (PO_4, NO_2+NO_3, NH_4, and Silica), methane, sulfate, and hydrogen sulfide in sediment layers 1 and 2, and benthic stress, were set up with reasonable initial estimates. Sediment initial conditions have a much larger effect on model results than the water column initial conditions because the sediment takes a much longer time to reach steady state. In order to ensure that the sediment reaches steady state with the water column, the water quality model (coupled with the sediment flux submodel) was recycled for at least 5 years to obtain approximate equilibrium conditions in the sediment before proceeding with the model calibration/validation. The initial conditions for the validation model used the model calibration results.

4.3.3 Boundary Conditions

One key issue in developing the model for the Murderkill River is the downstream tidal boundary condition in Delaware Bay. A boundary unaffected by internal sources is necessary to minimize boundary condition effects on the internal model calculations. In order to achieve this goal, the model segments were extended into Delaware Bay as discussed in Section 3.1 so that the internal water quality calculations do not affect the tidal bay boundary conditions. The data used for developing the water quality tidal bay boundary conditions are from the DRBC Boat Run database. Monitoring data from three stations were used to setup the tidal bay boundary conditions: Elbow of Crossledge Shoal, RM 22.75 (#91026); South Joe Flogger Shoal, RM 16.5 (#91028); and South Brown Shoal, RM 6.5 (#91030). Figures 28-30 presents the Delaware Bay boundary conditions at the north, east and south boundary locations used for the Murderkill River model (black circles are the Boat Run data and black lines are the model input). The green circles in this figure are the DNREC data at Bowers Beach. Because there was a significant difference between the data collected in the middle of the bay (Boat Run data) and that near the shoreline as indicated by the Bowers Beach data, the boundary condition segments that are perpendicular to the shoreline were linearly interpolated from the Bowers Beach data to the Boat Run data. It should also be noted that the DNREC ammonia nitrogen data in late 2007 and 2008 were considered unreliable and not used due to laboratory interference at salinity levels greater than 5 ppt. The upstream river boundary conditions were based on output from the HSPF watershed model.

Because there were limited data concerning the percentage of dissolved versus particulate forms of organic nitrogen, phosphorus and carbon, these fractions were determined from available data and prior modeling efforts. The fractions used for model input setup were 65% dissolved and 35% particulate for river boundary condition organic nitrogen, phosphorus and carbon. For the tidal bay boundary conditions, dissolved and particulate organic carbon data were available and the 65% dissolved and 35% particulate fractions were used for organic nitrogen and phosphorus. The split between labile and refractory organic matter was based on past modeling efforts and was 10% refractory and 90% labile for the river boundary conditions. For the tidal bay boundary conditions, 75% refractory and 25% labile fractions were used for the organic carbon and 10% refractory and 90% labile fractions were used for the organic nitrogen and phosphorus.

4.3.4 Point Source and Nonpoint Source Loads

The only point source load in the tidal Murderkill River is that from the KCRWTF discharge that enters the river about 1.3 miles downstream from Route 1 Bridge (Bay Road) near Frederica. The KCRWTF discharge data were discussed in Section 2.1.4 and presented in Figure 7.

Nonpoint source loads to the tidal Murderkill River are generated from the HSPF watershed model and are assigned at the five major tributary inputs and as incremental inputs along the tidal reaches of the Murderkill River. The incremental loads from the HSPF model are assigned in the water quality model from HSPF subwatershed segments 3, 6, 9, 10, 11, 18, 26, 27 and 28. Table 8 presents a summary of the watershed nutrient loads.

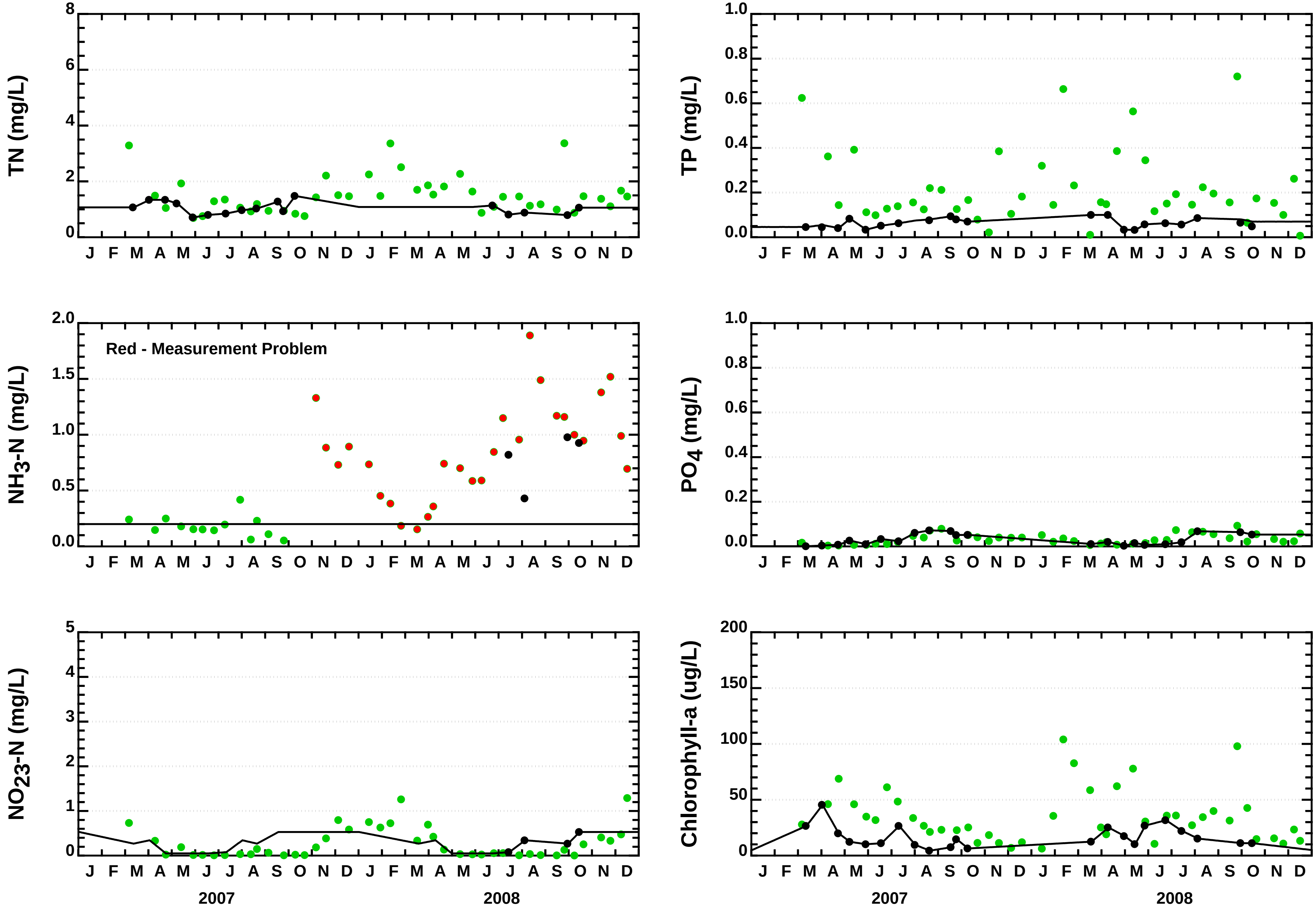

Figure 28a. Bay Boundary Condition Data & Model Input: Elbow of Crossledge Shoal, 91026 (RM 22.75)
(Black - DRBC Boat Run, Green - DNREC Bowers Beach)

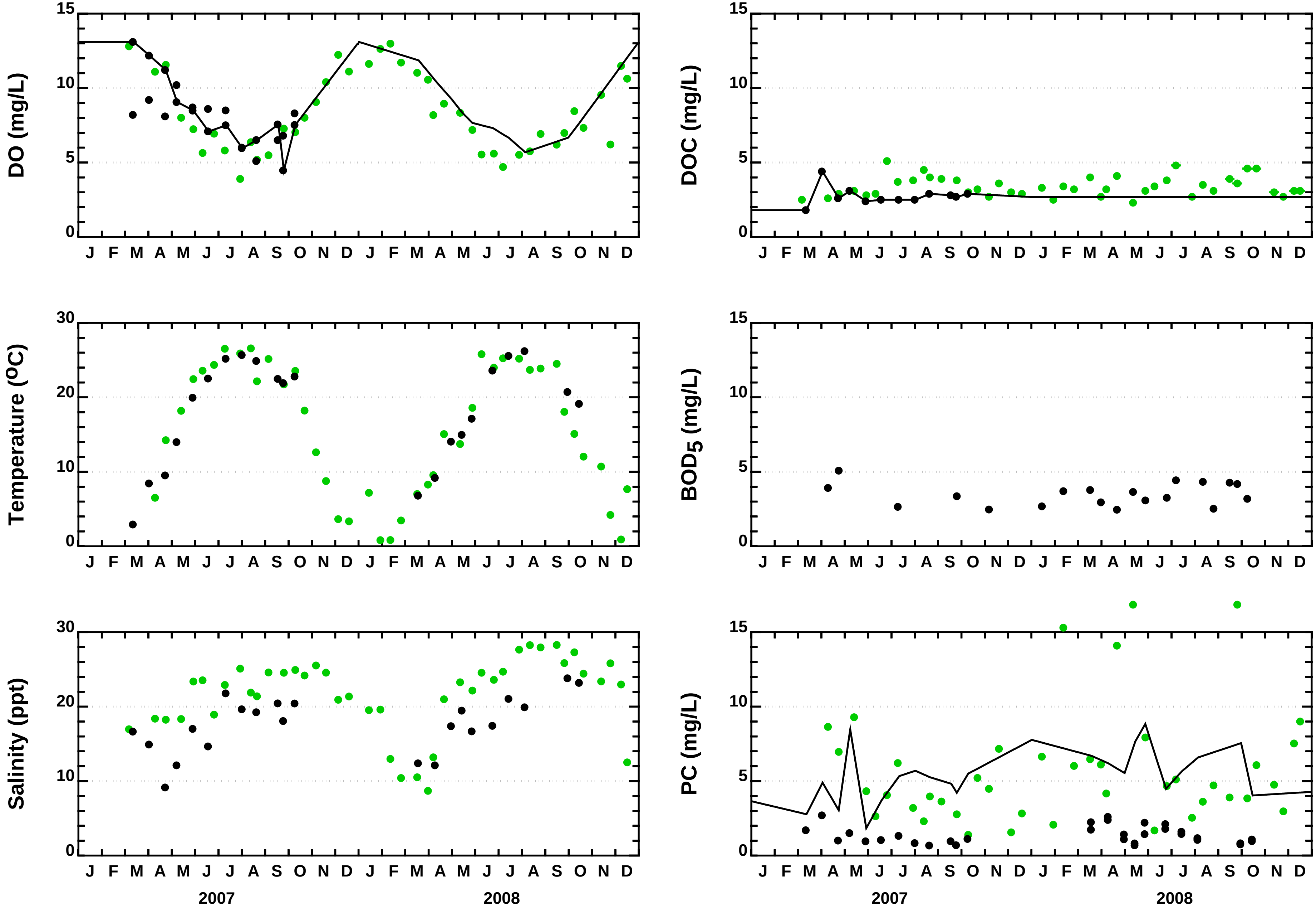

Figure 28b. Bay Boundary Condition Data & Model Input: Elbow of Crossledge Shoal, 91026 (RM 22.75)
(Black - DRBC Boat Run, Green - DNREC Bowers Beach)

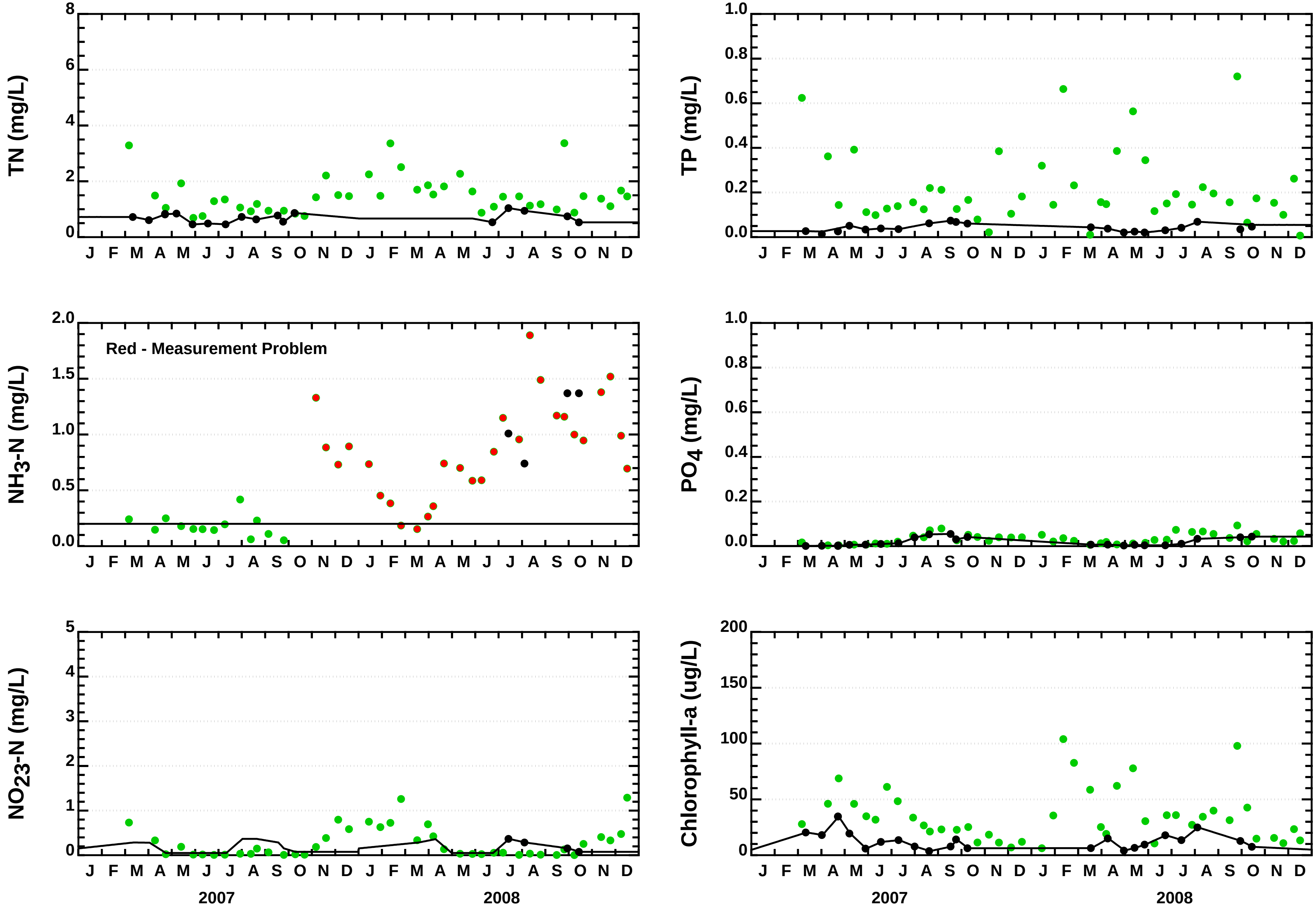

Figure 29a. Bay Boundary Condition Data & Model Input: South Joe Flogger Shoal, 91028 (RM 16.5)
(Black - DRBC Boat Run, Green - DNREC Bowers Beach)

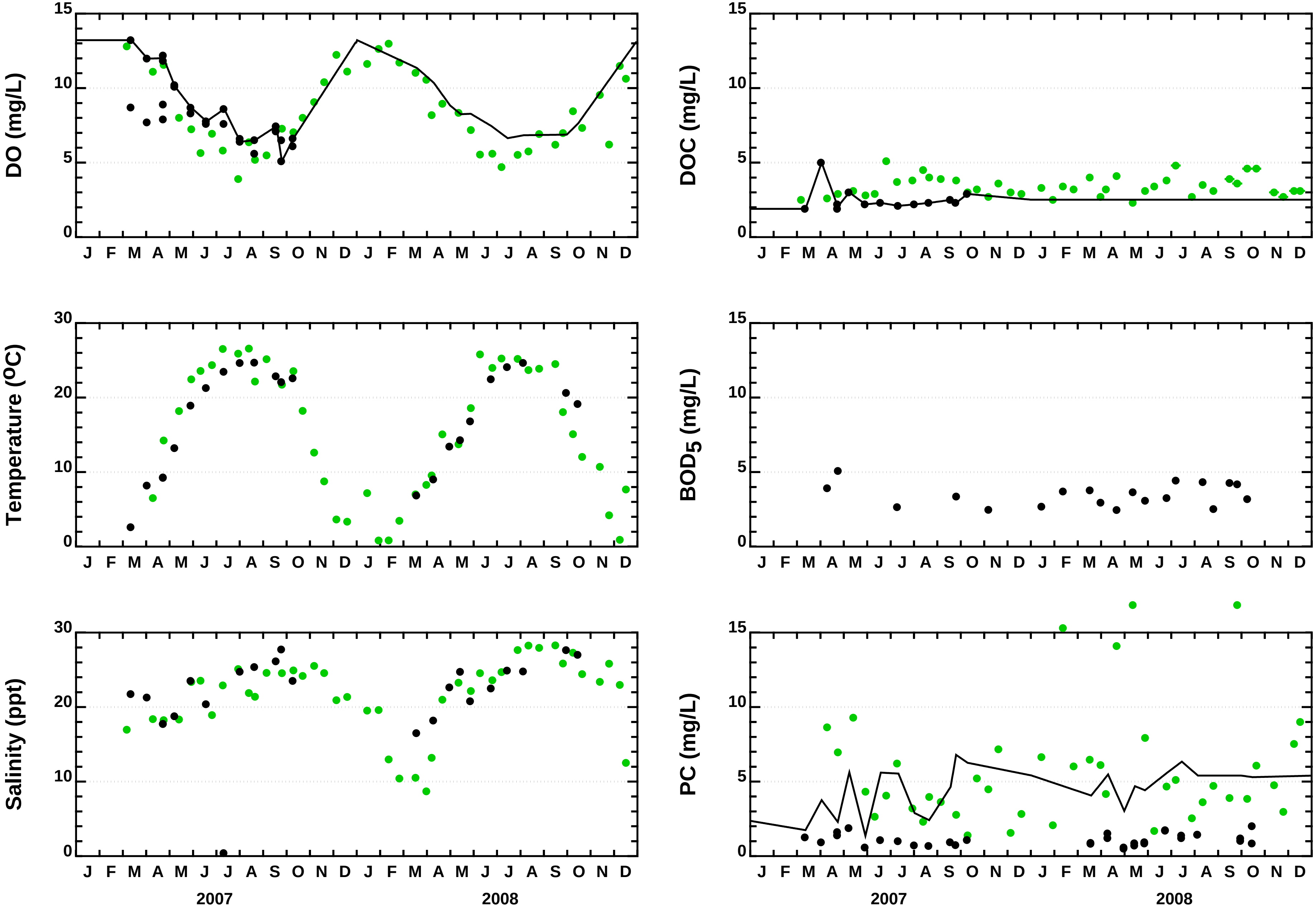

Figure 29b. Bay Boundary Condition Data & Model Input: South Joe Flogger Shoal, 91028 (RM 16.5)
(Black - DRBC Boat Run, Green - DNREC Bowers Beach)

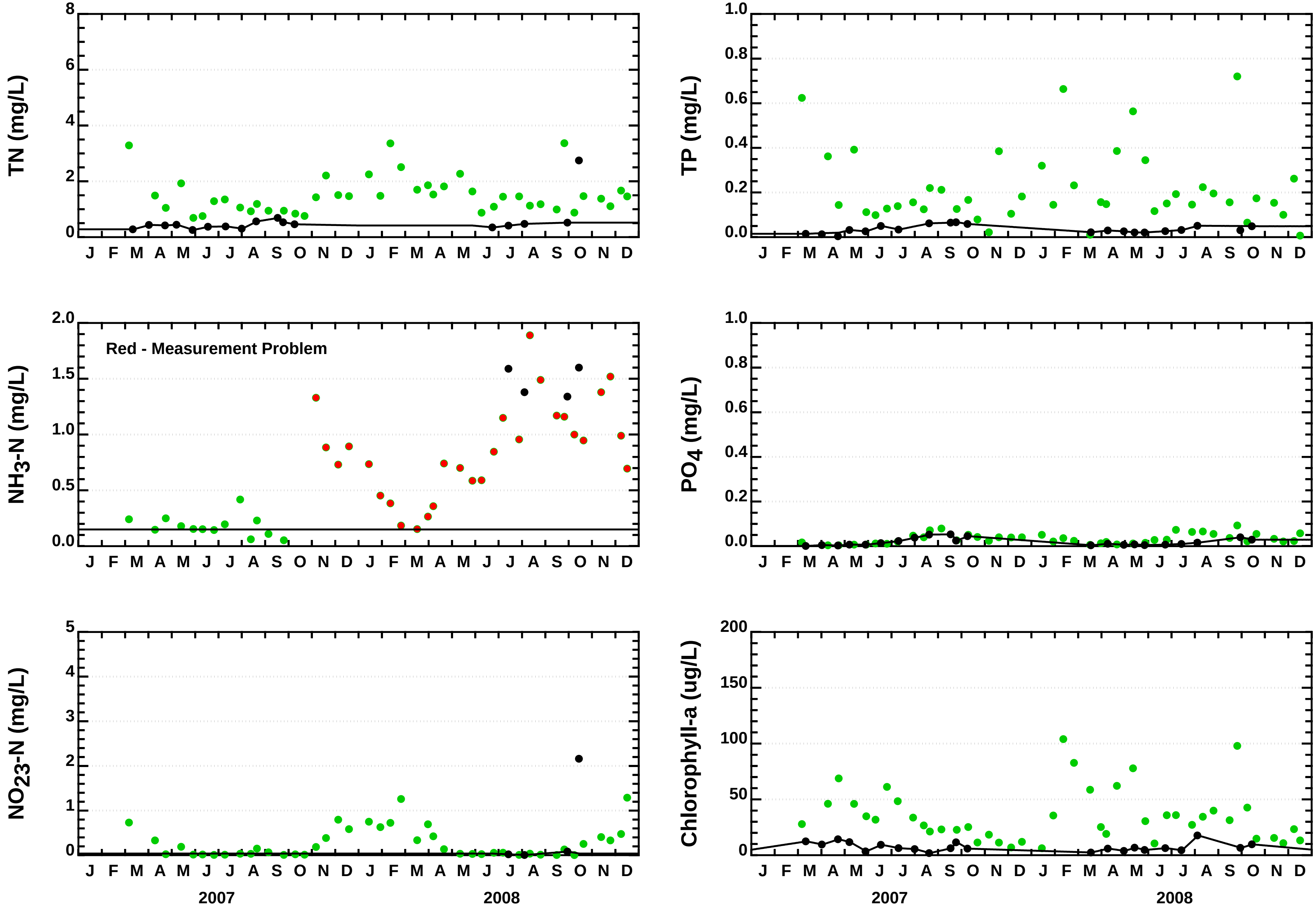

Figure 30a. Bay Boundary Condition Data & Model Input: South Brown Shoal, 91030 (RM 6.5)

(Black - DRBC Boat Run, Green - DNREC Bowers Beach)

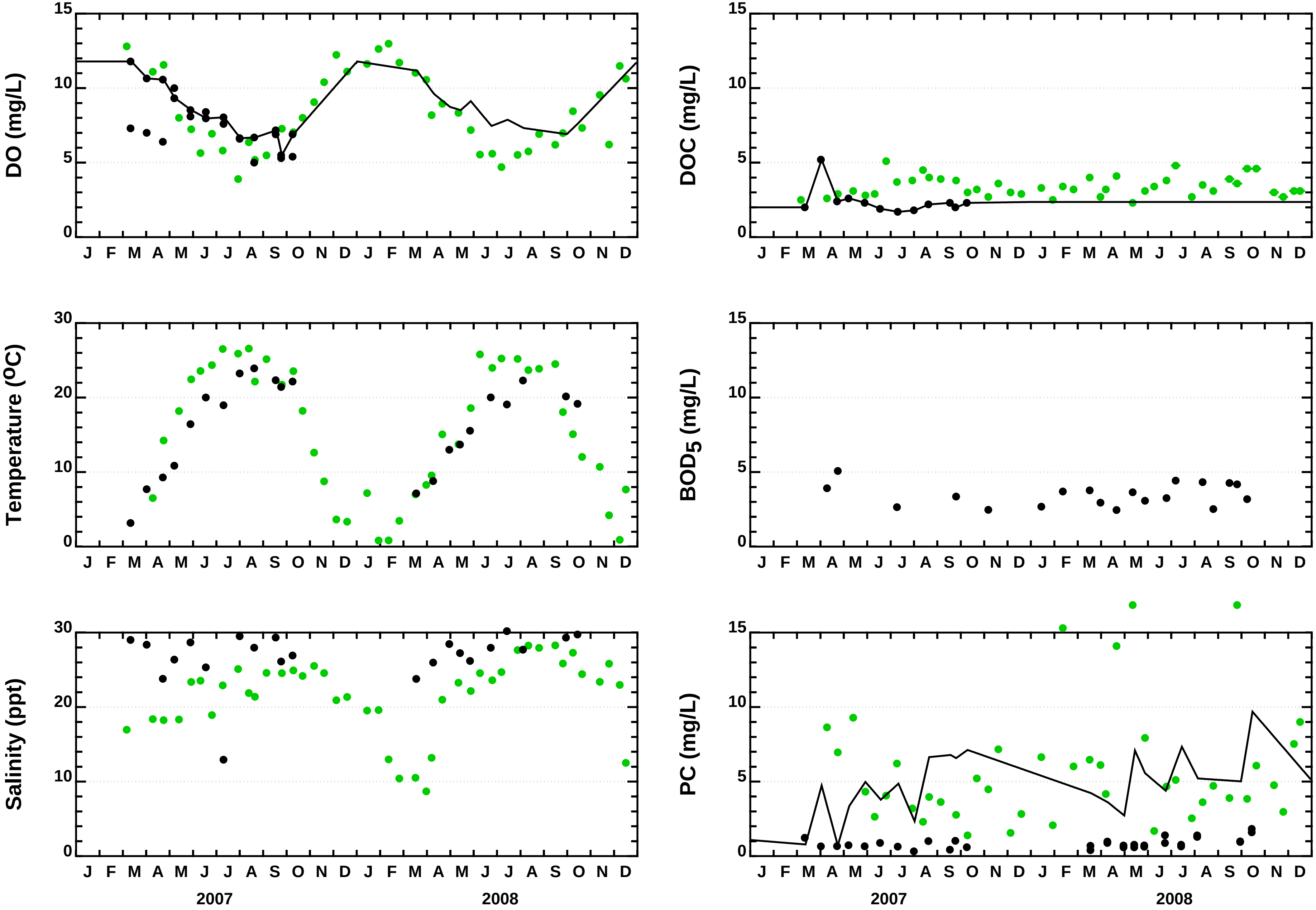

Figure 30b. Bay Boundary Condition Data & Model Input: South Brown Shoal, 91030 (RM 6.5)
(Black - DRBC Boat Run, Green - DNREC Bowers Beach)

4.3.5 Atmospheric Reaeration, Sediment Net Deposition and Settling

Atmospheric reaeration is one of the two major sources supplying DO in many estuarine water bodies; the other source is phytoplankton oxygen production. In estuarine systems, atmospheric reaeration is usually accounted for through the specification of a constant oxygen transfer coefficient (K_L) in units of length/time (m/d) or based on a velocity shear formulation. In order to allow atmospheric reaeration to be a function of river shear velocity and to account for low atmospheric oxygen transfer during slack tide conditions, a velocity shear formulation was used along with the use of a minimum oxygen transfer coefficient. The minimum oxygen transfer coefficient used was 0.3 meters/day (about 1 foot/day), which represents a low wind driven oxygen transfer coefficient. The velocity shear formulation used is:

$$KL_{vel} = \left[\frac{D \times Velocity_1}{Depth_1}\right]^{1/2}$$

where: KL_{vel} is velocity based oxygen transfer coefficient (m/d);

D is oxygen diffusivity (0.00017 m^2/d);

Velocity is average surface velocity (m/d); and

Depth is average surface depth (m).

The model input also requires settling and net sediment deposition rates for the algal groups and non-living particulate organic matter. The net sediment deposition rate of particles is a removal mechanism from the water column to the sediment bed and ultimately is incorporated into the sediment. To account for the high energy and velocity area near the mouth of the river and also to better reproduce observed SOD near the mouth, lower net sediment deposition rates (VSnet) were assigned near mouth as follows: 0.075 m/d from mouth upstream 0.5 miles; 0.15 m/d from 0.5 miles upstream to Webb Landing; and 0.3 m/d from Webb Landing upstream in the tidal river. In Delaware Bay, the net sediment deposition rate was 0.3 m/d. The water column settling rate was 0.3 m/d everywhere.

4.3.6 Constants

There are 157 constants required for the water quality model. Because each of the algal groups has 27 constants and only one algal group was modeled, the actual number of constants needed to develop the model was 103. All of the 103 constants used in the water quality model are tabulated in Appendix 7 along with a detailed discussion of the model theory. Table 17 presents a few of the more important constants used in the final calibration and validation. Many constants were first developed based on the St. Jones River TMDL modeling completed for DNREC (HydroQual, 2005) due to its close proximity, refined from site-specific studies as part of this project, obtained from former modeling studies or finalized through the model calibration/validation process. In addition, model constants were also based on guidance in the

Rates, Constants, and Kinetics Formulations in Surface Water Quality Modeling publication (EPA, 1985).

Table 17. Summary of Eutrophication Model Constants		
Parameter	**Value at 20°C**	**Source**
PHYTOPLANKTON		
Growth Rate (1/d)	2.5	Algal Production Study
Respiration Rate (1/d)	$0.3*\mu_a+0.05$	Previous Modeling Studies
Zooplankton Grazing Rate (1/d)	0.025	Previous Modeling Studies
Carbon/Chlorophyll Ratio	40.0	Previous Modeling Studies
Nitrogen/Carbon Ratio	0.100-0.176 (variable)	Previous Modeling Studies
Phosphorus/Carbon Ratio	0.011-0.025 (variable)	Previous Modeling Studies
Silica/Carbon Ratio	0.057-0.133 (variable)	Previous Modeling Studies
NITROGEN		
Hydrolysis Rate (1/d) (Particulate to Dissolved Organic)	0.01 – labile 0.05 – refractory	Previous Modeling Studies
Mineralization Rate (1/d) (Dissolved to Ammonia)	0.01 – labile 0.05 – refractory	Previous Modeling Studies
Nitrification Rate (1/d)	0.05	LTBOD Studies
PHOSPHORUS		
Hydrolysis Rate (1/d) (Particulate to Dissolved Organic)	0.01 – labile 0.05 – refractory	Previous Modeling Studies
Mineralization Rate (1/d) (Dissolved to Orthophosphate)	0.01 – labile 0.05 – refractory	Previous Modeling Studies
CARBON		
Hydrolysis Rate (1/d) (Particulate-to-Dissolved Organic)	0.01 – labile 0.05 – refractory	Previous Modeling Studies
Oxidation Rate (1/d)	0.05	LTBOD Studies
OXYGEN		
Oxygen to Carbon Ratio (O_2:C)	3.5	Model Calibration

4.3.7 Sediment Submodel

The sediment sub-model requires input of initial conditions and various coefficients. The sediment initial conditions, as described previously, were set up with best estimates and cycled to an equilibrium condition with the water column. As mentioned before, it is important to iterate the sediment runs because the sediment concentrations take a long time to reach steady state. Appendix 7 tabulates the sediment model coefficients used in the model.

4.4 WATER QUALITY MODEL CALIBRATION AND VALIDATION RESULTS

The water quality model calibration period covered the year 2007 and the validation period covered the year 2008. The calibration and validation of the model used the same model parameters and constants. The initial conditions for the validation model run (1/1/2008) were setup from the results at the end of the calibration model run (12/31/2007), and each modeling period had its own solar radiation, light extinction coefficients, and loads developed based on available data and the HSPF watershed model results. The water quality model calibration and validation compare model output and observed data collected by DNREC at eight stations in the tidal Murderkill River for DO, organic nitrogen (OrgN), NH_3, NO_2+NO_3, organic phosphorus (OrgP), PO_4, BOD_5 and chlorophyll-a (chl-a). These eight monitoring stations in the tidal Murderkill River are located at: Bowers Beach Wharf (#206101); Webb Landing (#206131); Milford Neck Wildlife Area Levee (#206141); Powerlines (#206711); Confluence of KCRWTF Canal (#206231); Bay Road, Route 1/113 (#206091); Spring Creek at Route 12 Bridge (#206081); and Double Run at Barretts Chapel Road (#206561). It should be noted that the Double Run monitoring location is at the boundary between freshwater and tidal affects, and tidal model results may not best represent the data collected at this location. In addition, continuous DO data was available for calibration/validation at Bowers Beach and near Frederica from the USGS gages.

Calibration/validation of any water quality model requires the adjustment of certain parameters for site-specific conditions based on comparisons between observed data and model output. In the case of the Murderkill River Study, there were a number of project specific studies that helped define model constants and the tidal marsh loads. The method of calibrating the model begins with the selection of a set of parameters (kinetic constants) based on other modeling studies. For the Murderkill River model, many of the initial parameters were based on the calibrated St. Jones River water quality model and prior modeling experience on other similar systems. The remainder of the calibration phase involves the adjustment of key parameters, both individually and in conjunction with other parameters, to obtain a reasonable representation of the water quality kinetics observed in the system. Adjustment of the model parameters is tightly constrained by typical ranges as determined from the literature (EPA, 1985) and other modeling studies.

During the Murderkill River water quality model calibration/validation, over 50 model runs were completed to arrive at the final calibration/validation. Based on the nature of the calibration process, this involved many sensitivities to key model parameters. These sensitivities differ from those involving variation of individual parameters with subsequent observation of the effect on model results; instead, they involve the simultaneous adjustment of linked parameters within typical ranges to assess their effects. For instance, model sensitivities were completed involving phytoplankton growth and respiration rates to assess the effects on calculated phytoplankton (chl-a), DO and nutrient levels with the set of parameters that best reproduced the observed data chosen for the final calibration/validation. Therefore, the method of calibrating the Murderkill River water quality model included many iterations involving the adjustment of individual parameters and also many sensitivities to coupled parameters. Another very important aspect of the water quality model calibration is the specification of boundary conditions, meteorological conditions (wind and solar radiation), and most importantly, a hydrodynamic transport pattern that reasonably reproduces the observed conditions during the calibration/validation period. For the calibration of the sediment submodel, an iteration of model runs was important for the sediment concentrations to reach steady state prior to initiating calibration/validation efforts. The next sections present the results of the calibrated and validated water quality model.

4.4.1 Water Column – Nutrients & Chl-a

The water quality model calibration/validation is presented in Figures 31-38 for TN, NH_3, NO_2+NO_3, TP, PO_4 and chl-a. These figures present the daily average model output as the solid line with the shaded region representing the model output variation over the day. DNREC water quality data are presented as the filled circles. It should be noted for the NH_3 data in these figures that data in late 2007 and 2008 were considered unreliable and not used due to laboratory interference at salinity levels greater than 5 ppt (the data not used are presented in red).

The calibrated/validated water quality model reproduces the observed nitrogen data (TN, NH_3 and NO_2+NO_3) very well with respect to the seasonal and spatial variation in the tidal river. The model captures the higher nitrogen levels during the winter/spring season when watershed runoff is greatest and the lower levels observed during the summer/fall period when watershed loads are less and nutrient uptake/loss is greatest in the tidal river and surrounding tidal marshes. In addition, the overall increase in nitrogen levels from the mouth in upstream direction is also reproduced by the model.

The calibrated/validated water quality model also captures the seasonal and spatial variation in the observed phosphorus data (TP and PO_4). Greater levels of phosphorus are observed and reproduced by the model during the summer/fall period due to the lower freshwater water flow entering the tidal river coupled with the KCRWTF phosphorus load near the middle of the river (#206231). The model does slightly over-calculate the phosphorus data in the middle of the river, which may be due to phosphorus adsorption to TSS that is currently not included in the model framework. Annual average TSS levels in the river ranged from 50-103 mg/L in 2007 and from 55-132 mg/L in 2008.

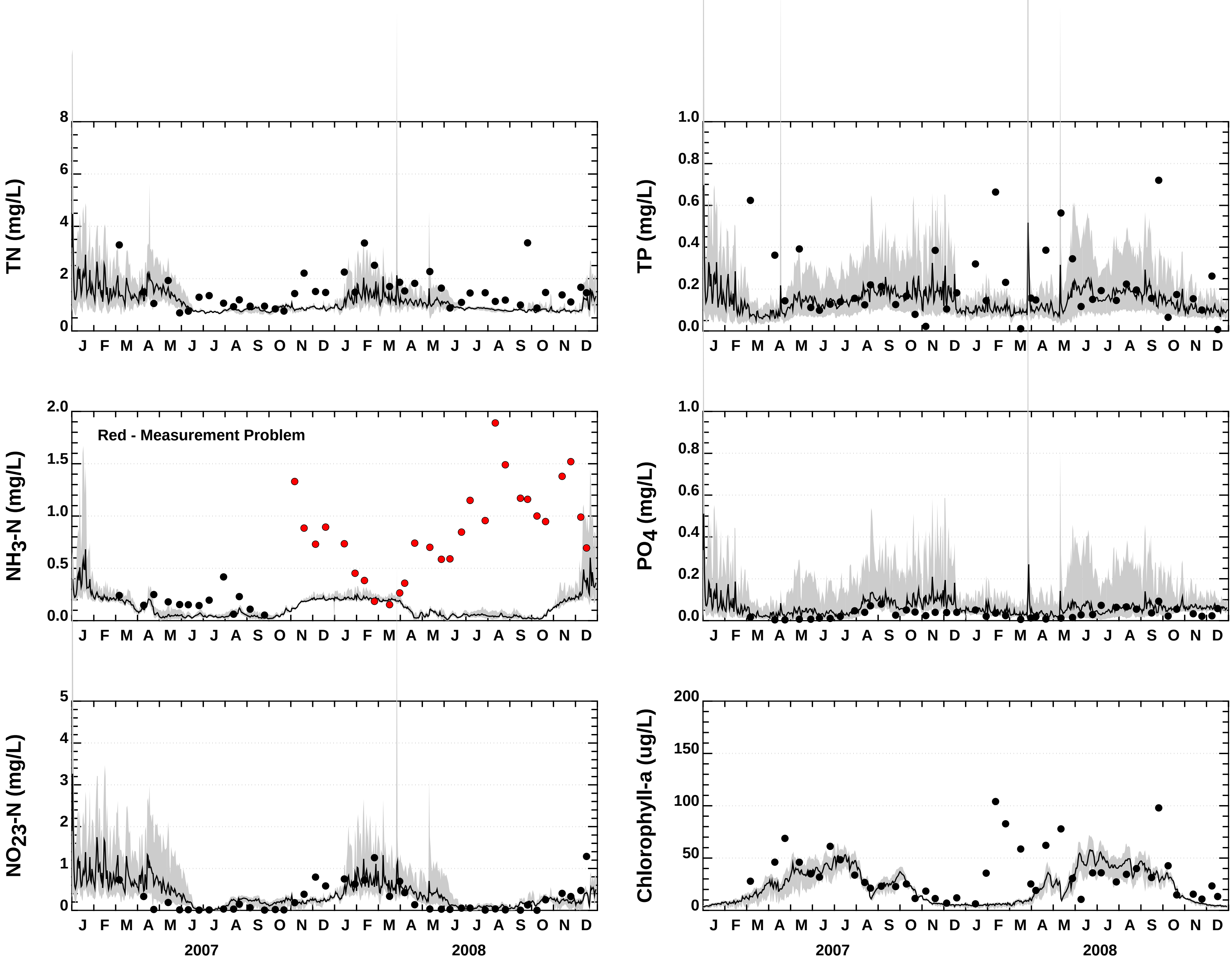

Figure 31. Water Quality Model Calibration/Validation (2007-2008)
Station 206101, Murderkill River at Bowers Beach Warf (Mouth), (74,41)

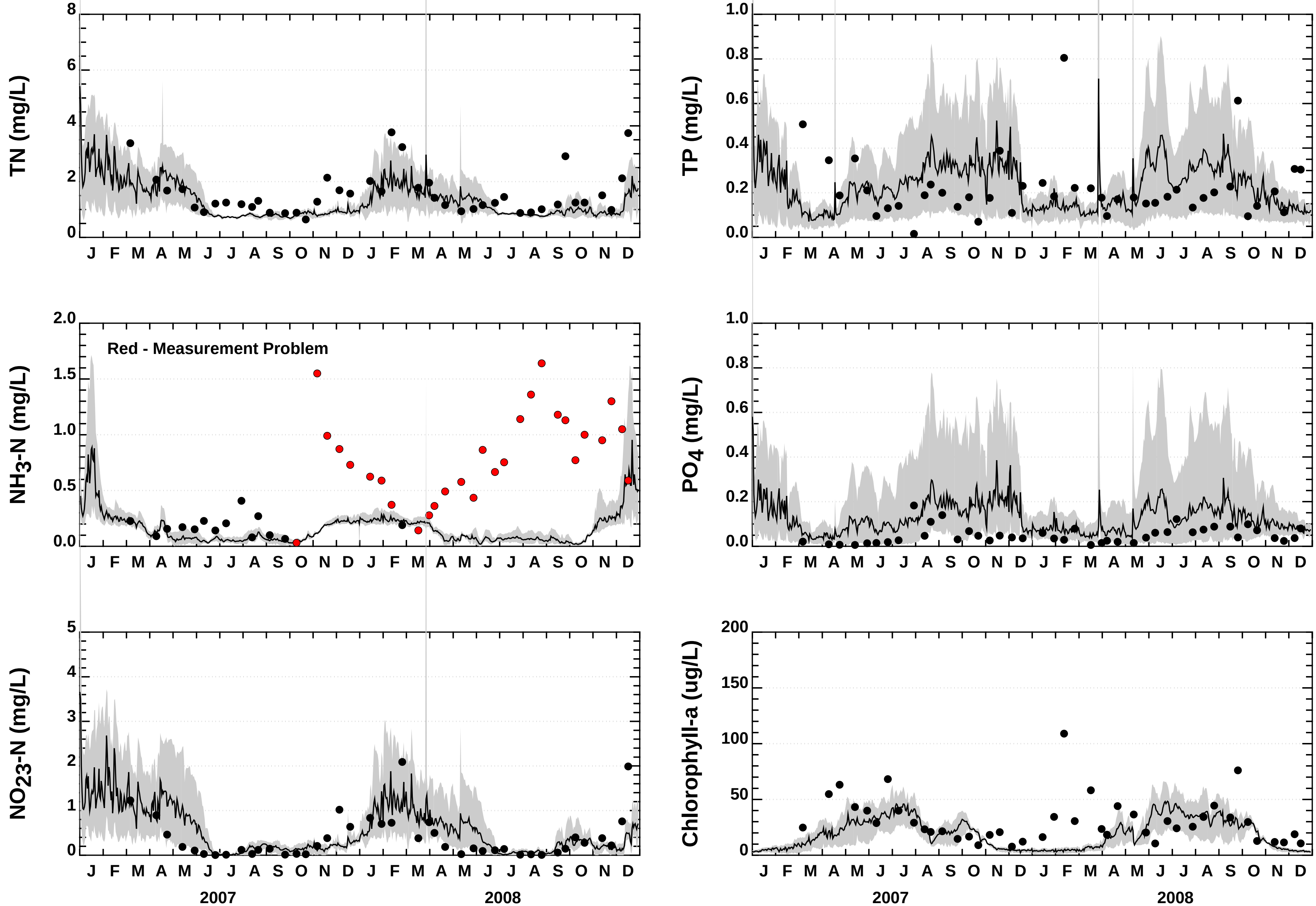

Figure 32. Water Quality Model Calibration/Validation (2007-2008)
Station 206131, Murderkill River at Webb Landing (1.25 Miles from Mouth), (67,32)

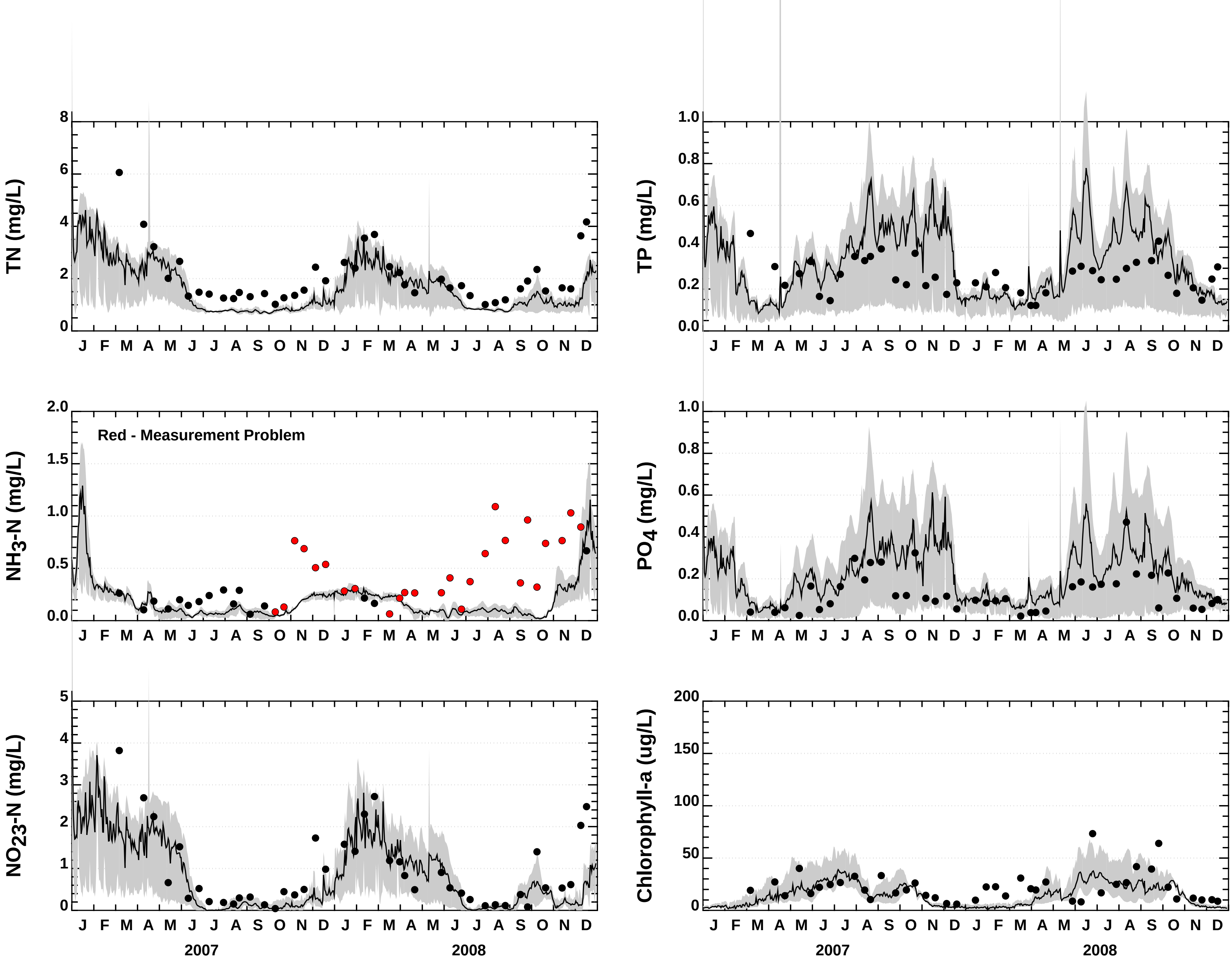

Figure 33. Water Quality Model Calibration/Validation (2007-2008)
Station 206141, Murderkill River near Levee at Milford Neck Wildlife, (54,32)

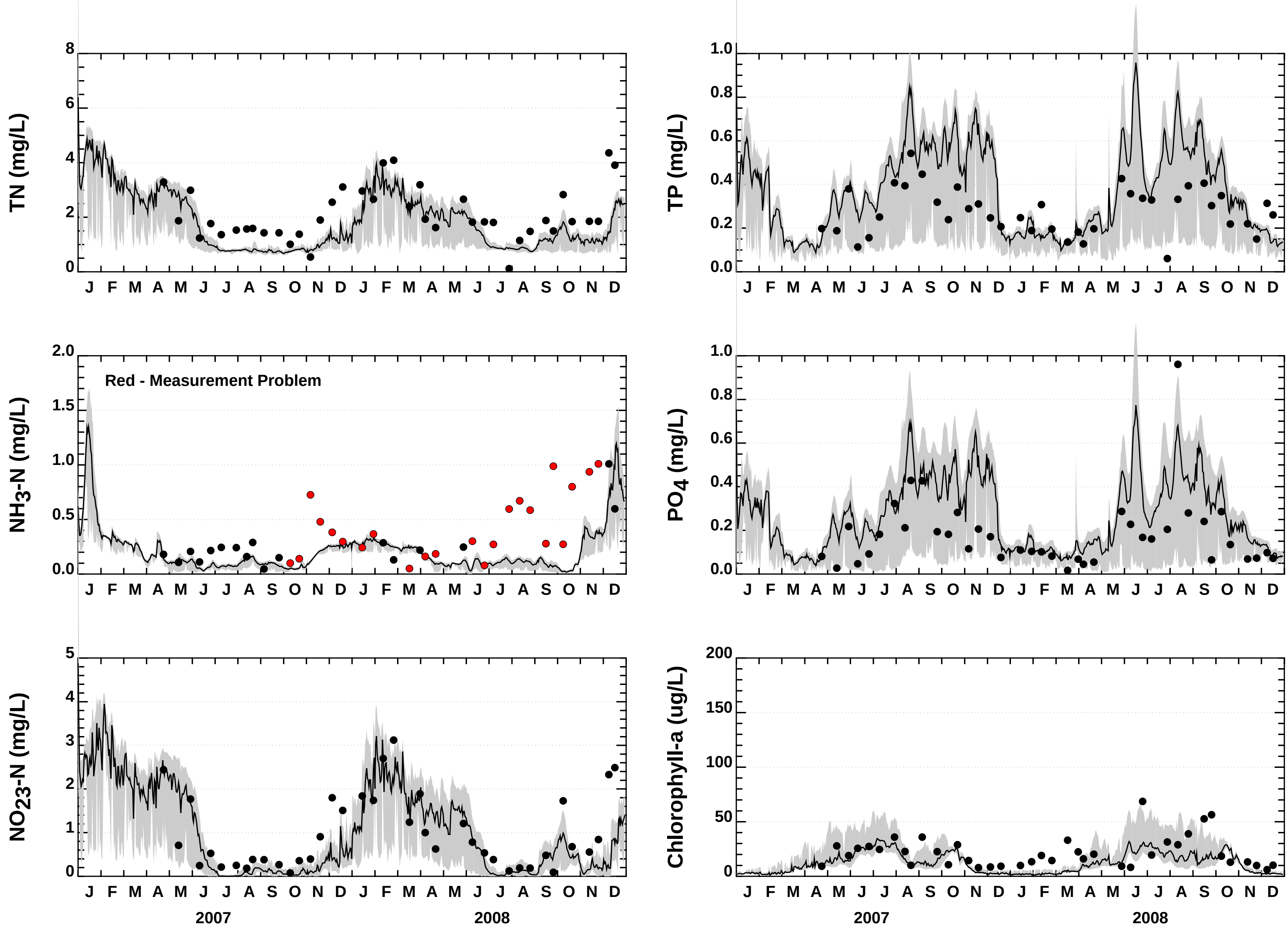

Figure 34. Water Quality Model Calibration/Validation (2007-2008)

Station 206711, Murderkill River near Power Lines (4.45 River Miles), (47,32)

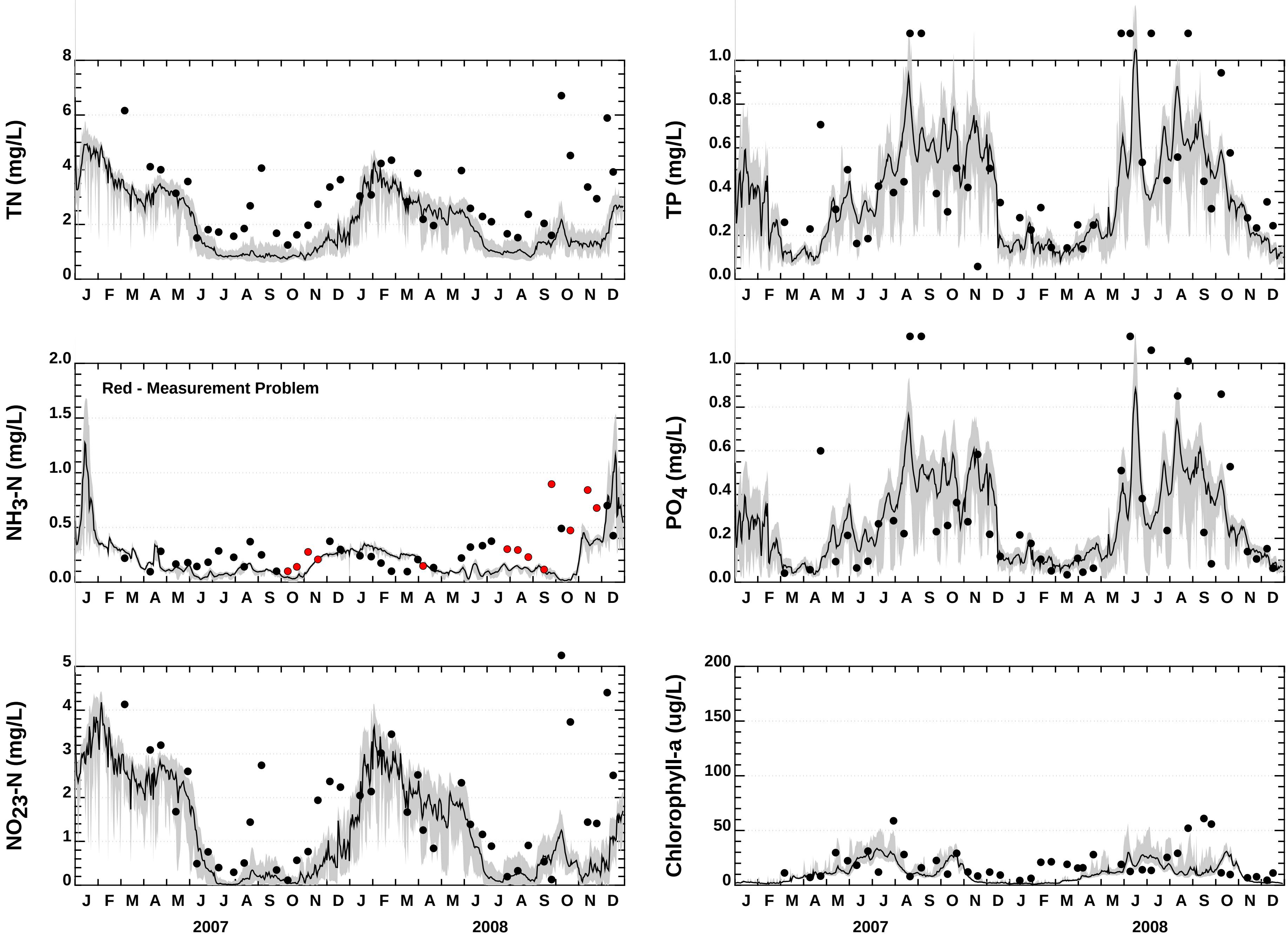

Figure 35. Water Quality Model Calibration/Validation (2007-2008)
Station 206231, Murderkill River at Confluence of Kent County WWTF, (36,33)

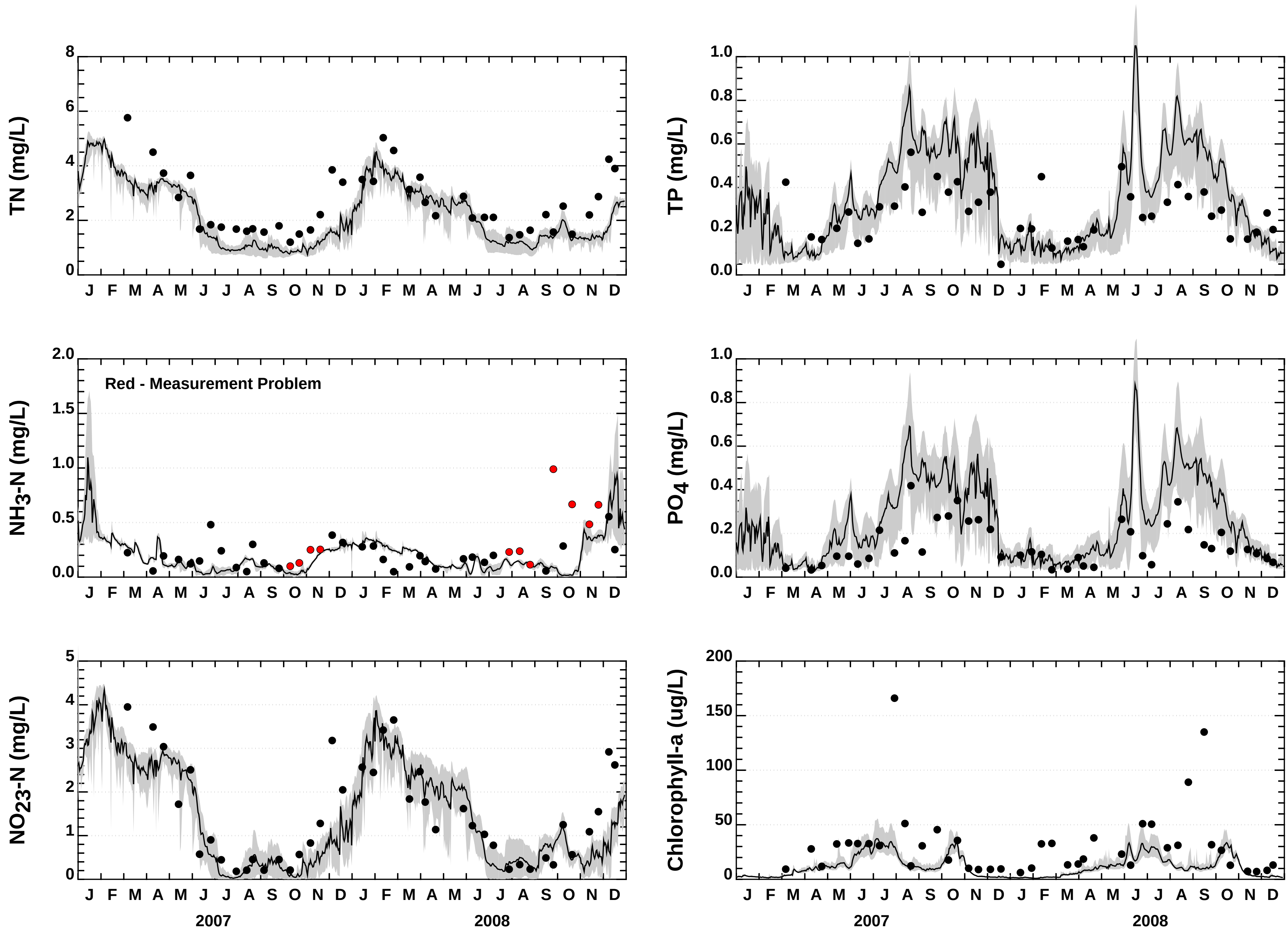

Figure 36. Water Quality Model Calibration/Validation (2007-2008)
Station 206091, Murderkill River at Bay Road (Rt 1/113), (27,39)

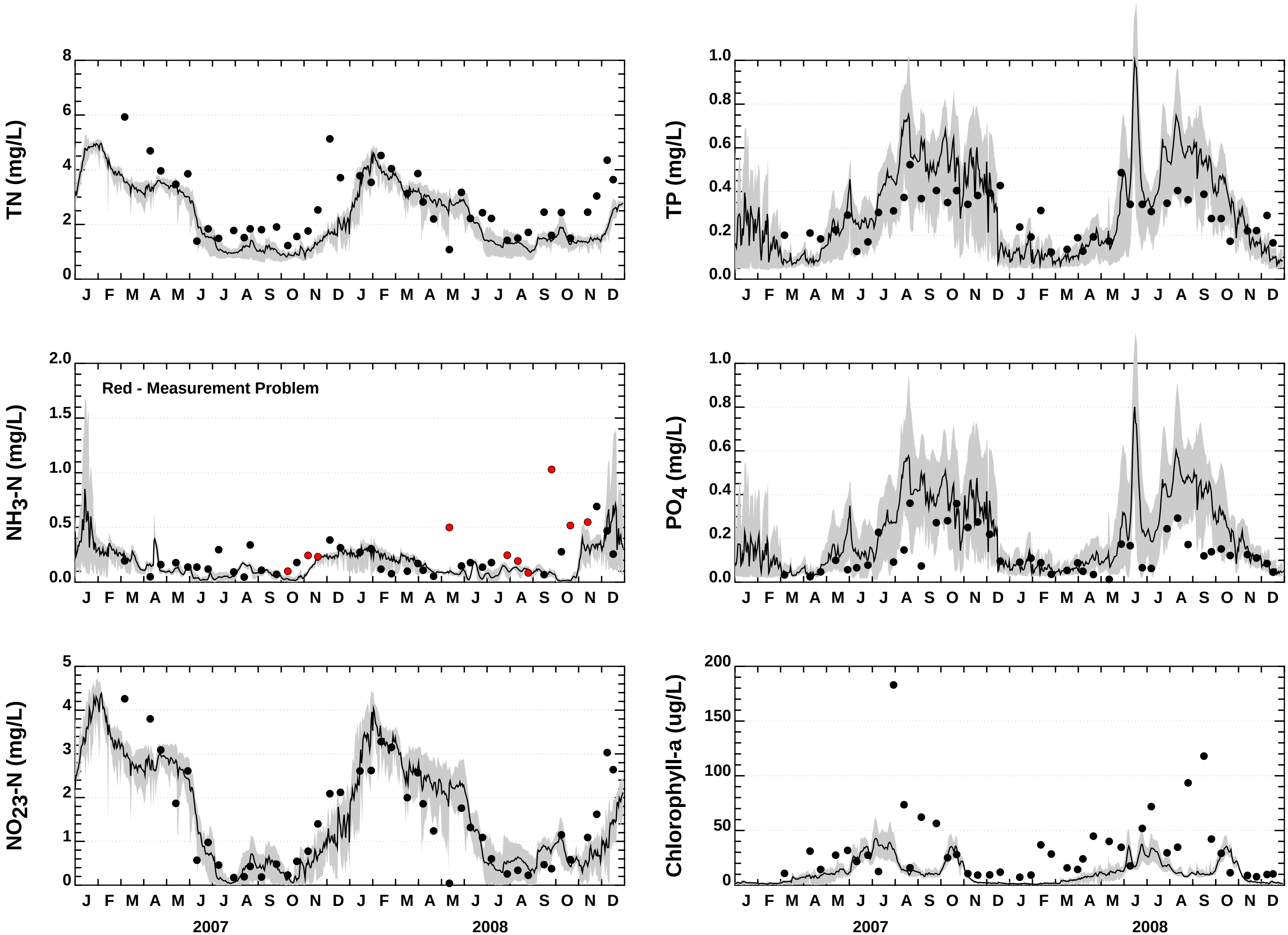

Figure 37. Water Quality Model Calibration/Validation (2007-2008)
Station 206081, Spring Creek at Rt. 12 Bridge at Frederica, (27,45)

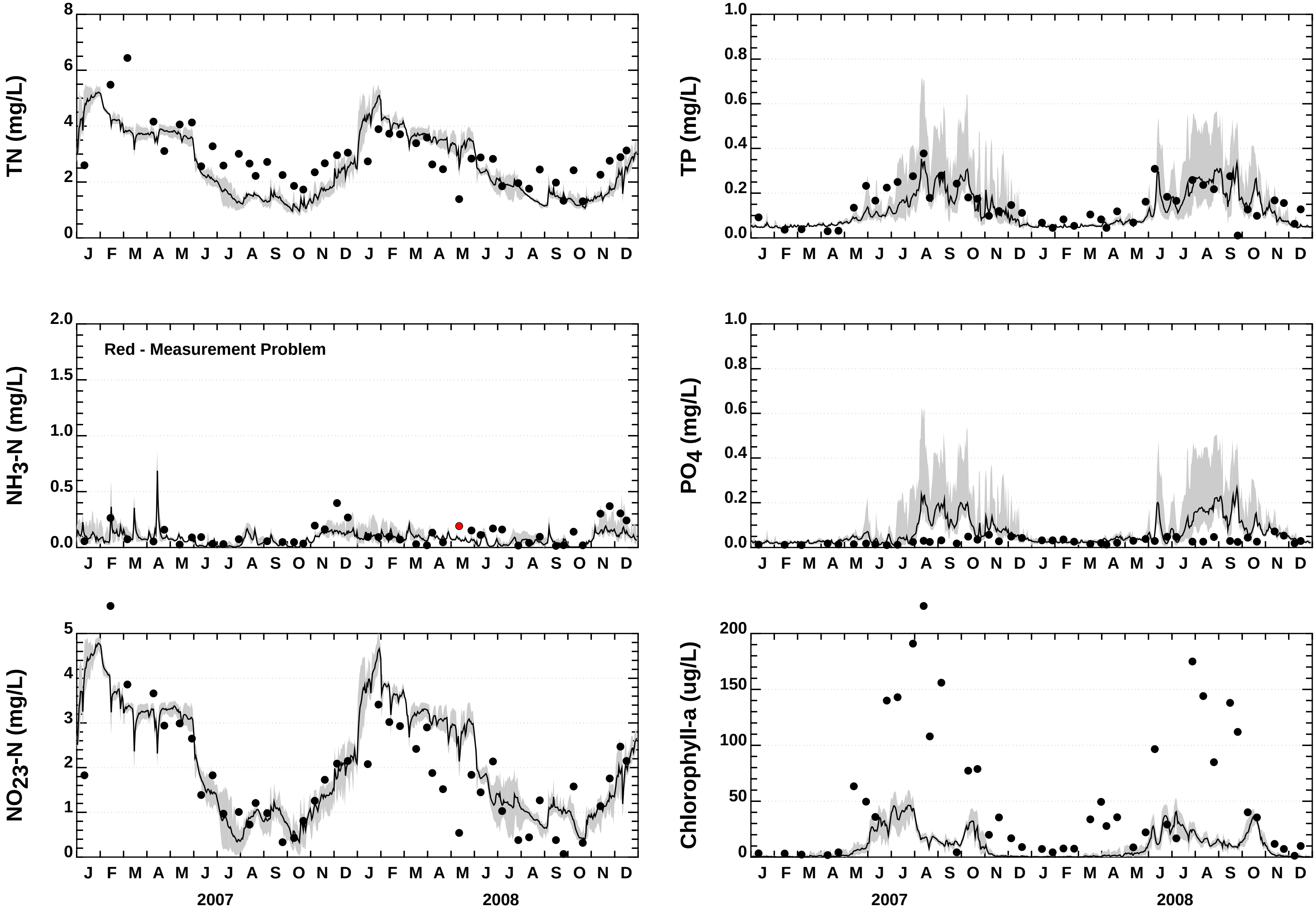

Figure 38. Water Quality Model Calibration/Validation (2007-2008)
Station 206561, Double Run at Barratts Chapel Rd. (Rd. 371), (11,32)

The calibrated/validated water quality model captures the algal growth in the river as represented by the chl-a data. The model reproduces the seasonal and spatial patterns of chl-a levels well with higher levels occurring during the spring/summer/fall period. Higher chl-a levels are also observed and reproduced by the model near the mouth and upstream as opposed to the middle of the river. This pattern of chl-a levels is a function of the clearer water near the mouth (bay influenced) and upstream that provides more light for algal growth.

Table 18 presents annual average results for observed and modeled TN and TP, and summer average (May-September) chl-a at the eight tidal river stations. On an annual average basis at all of the stations: the model under-calculates the TN levels by about 0.4 mg/L; and the model under-calculates the TP levels by about 0.004 mg/L. From the KCRWTF canal to the mouth of the river, the model under-calculates the chl-a levels by about 6 μg/L and by a greater difference from Bay Road and upstream. The chl-a data does include many measurements of very high chl-a levels (greater than 50 μg/L), particularly upstream, which the model does not capture that increase the data annual averages. If measured chl-a levels greater than 50 μg/L are excluded, the model slightly over-calculates the chl-a levels by about 0.2 μg/L (KCRWTF canal to the mouth of the river) and under-calculates the chl-a levels by about 9 μg/L from Bay Road and upstream. Typically, eutrophication models do not capture algal blooms very well that occur on a small spatial scale or are caused by site-specific characteristics of the water body or loading sources not represented in the model. In addition, limited information was available to assign light extinction coefficients in the model (i.e., roughly monthly data) and any light attenuation changes that occur on a time-scale less than a month (e.g., intra-tidal) that could cause algal blooms are not reflected in the model, especially since algal growth in the model is controlled by the available light. Further discussion of how well algal growth is represented in the model is discussed as part of the model calibration/validation to the algal production data.

Table 18. Summary of Model-Data Comparisons for TN, TP & Chl-a				
Station	Data		Model	
	2007	2008	2007	2008
TN (mg/L) – Annual Average				
Bowers Beach	1.32	1.66	1.16	1.01
Webbs Landing	1.41	1.69	1.40	1.24
Milford Neck	2.03	2.11	1.75	1.57
Power Lines	1.80	2.34	1.92	1.79
KCRWTF Canal	2.76	3.18	2.11	2.03
Bay Road	2.52	2.73	2.24	2.20
Rte. 12 Bridge	2.70	2.71	2.34	2.31
Double Run	3.14	2.59	2.68	2.61
TP (mg/L) – Annual Average				
Bowers Beach	0.199	0.235	0.154	0.137
Webbs Landing	0.206	0.230	0.250	0.213
Milford Neck	0.281	0.246	0.362	0.301
Power Lines	0.299	0.263	0.404	0.342
KCRWTF Canal	0.569	0.582	0.411	0.354
Bay Road	0.303	0.265	0.370	0.330
Rte. 12 Bridge	0.316	0.262	0.327	0.299
Double Run	0.163	0.135	0.118	0.122
Chl-a (µg/L) – Summer Average (May-September)				
Bowers Beach	35.0	42.2	35.4	39.4
Webbs Landing	33.0	33.7	30.0	33.0
Milford Neck	24.2	33.8	23.1	24.5
Power Lines	25.2	34.9	19.4	19.6
KCRWTF Canal	24.7	31.4	17.6	16.3
Bay Road	46.8	50.4	18.8	16.3
Rte. 12 Bridge	51.2	53.4	20.1	16.8
Double Run	119.4	82.7	20.8	16.0

The algal production data (Sharp, 2011) was used for calibration/validation of the model assigned maximum algal growth rate. Comparisons of the model calculated and measured algal production in the river is presented in Figures 39-46. The solid line represents the model calculated daily average algal production at the surface and is used for comparison to the measured maximum algal production (red circles) with the shaded region representing the model algal production variation over the day. At all stations where data was available, the model reproduces the algal production data very well both seasonally and spatially in the river. The good model comparison to the algal production data from Bay Road upstream and the general under-prediction of chl-a levels in this same area may indicate that different algal populations may exist in the upstream freshwater reaches as compared to the lower saline reaches. These different algal populations could have a different carbon/chl-a (C:Chl-a) ratio that affects calculation of the chl-a levels in the model since algae are presented as carbon in the model. For example, if we used a C:Chl-a ratio of 20 in the upstream reaches instead of 40, the model calculated chl-a in this upstream reach would double.

4.4.2 Water Column – DO & Carbon (BOD_5)

The water quality model calibration/validation is presented in Figures 47-54 for DO, temperature, salinity, DOC, BOD_5 and total organic carbon (TOC). The temperature and salinity model-data comparisons are just a repeat of that presented in the hydrodynamic model calibration/validation section and will not be discussed here. These figures present the daily average model output as the solid line with the shaded region representing the model output variation over the day. DNREC water quality data are presented as the filled circles.

Overall, the model reproduces the observed DNREC grab sample DO data very well at most locations and captures the seasonal and spatial variations observed. At a few of the upstream stations (i.e., Frederica at the Route 12 Bridge and Bay Road), the model tends to under-calculate the observed DO levels during the summer. Also during the fall/winter of 2008 but not in 2007, the model under-calculates the observed DO levels which may be due to a site-specific event that occurred that is not represented in the model.

The model also reproduces the observed DOC and BOD_5 data very well at all of the DNREC stations. Near the mouth of the river (Bowers Beach and Webbs Landing), the model over-calculates the TOC data by about 2-3 mg/L but the model-data comparison improves in the upstream direction. Although the model calculated TOC is greater than observed, the model calculation of BOD_5 is very good which is an important component of the DO balance.

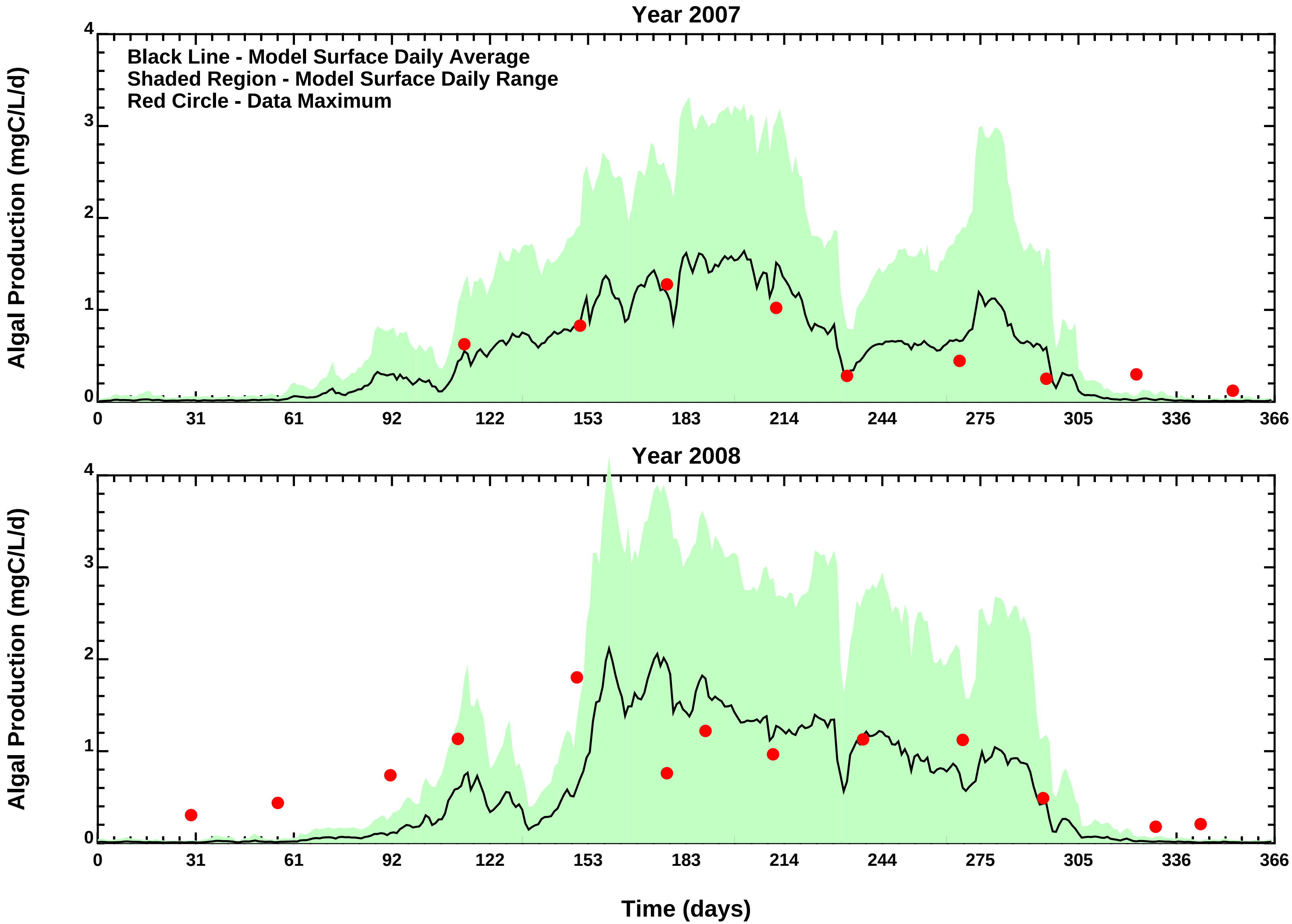

Figure 39. Murderkill River Model-Data Algal Production Comparison
(Bowers Beach Wharf - #206101)

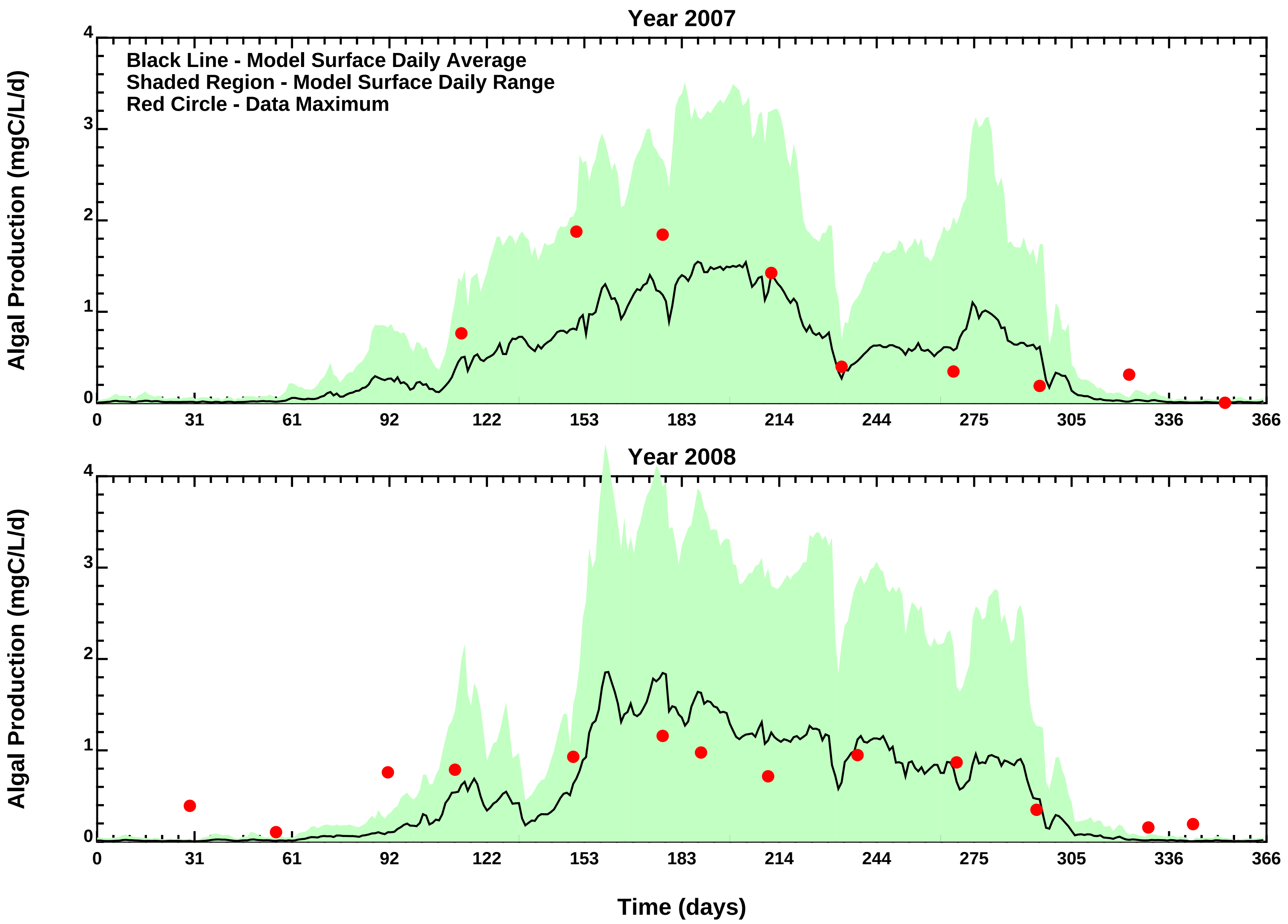

Figure 40. Murderkill River Model-Data Algal Production Comparison
(Webbs Landing - #206131)

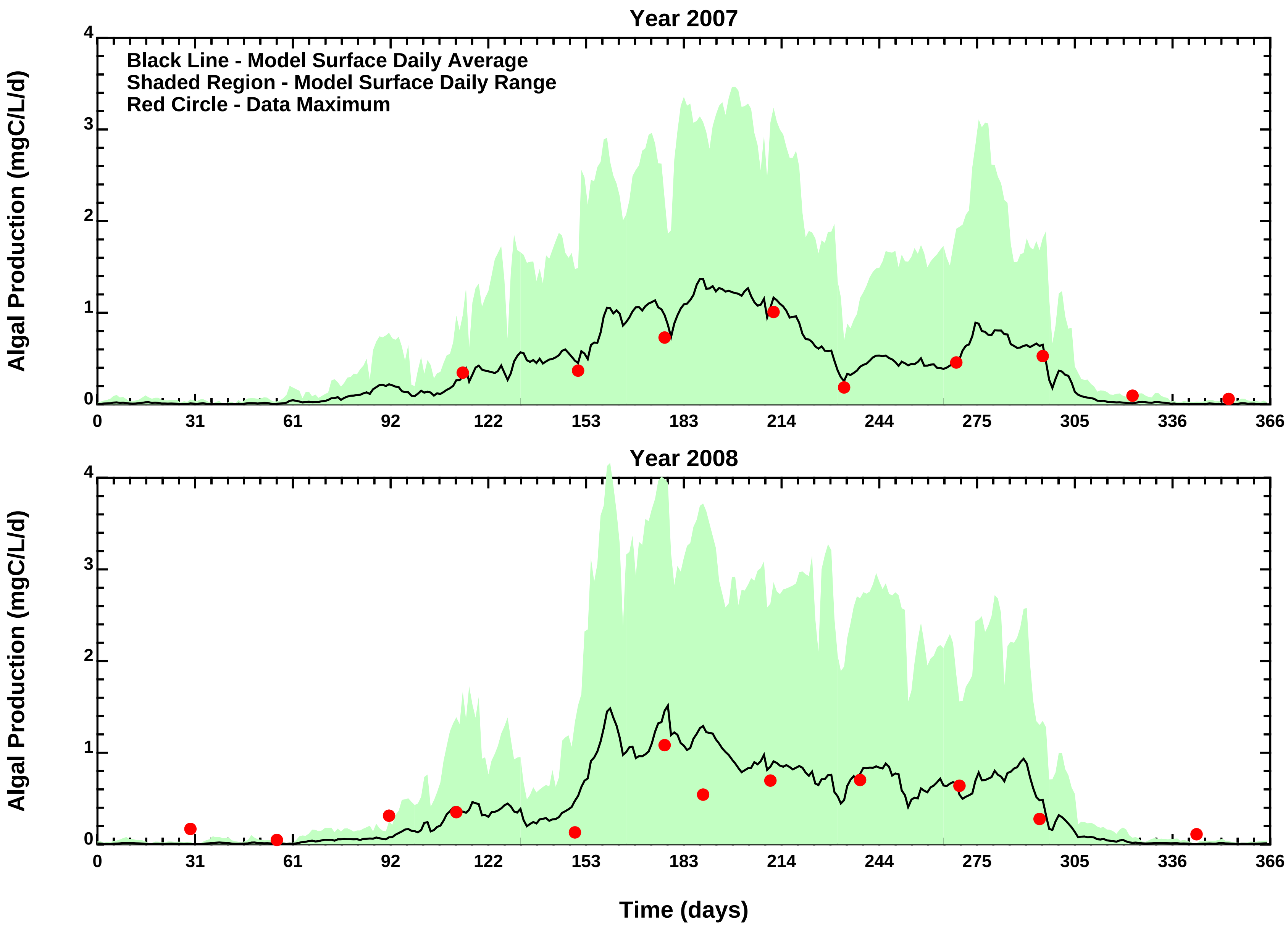

Figure 41. Murderkill River Model-Data Algal Production Comparison
(Milford Neck Wildlife Area Levee - #206141)

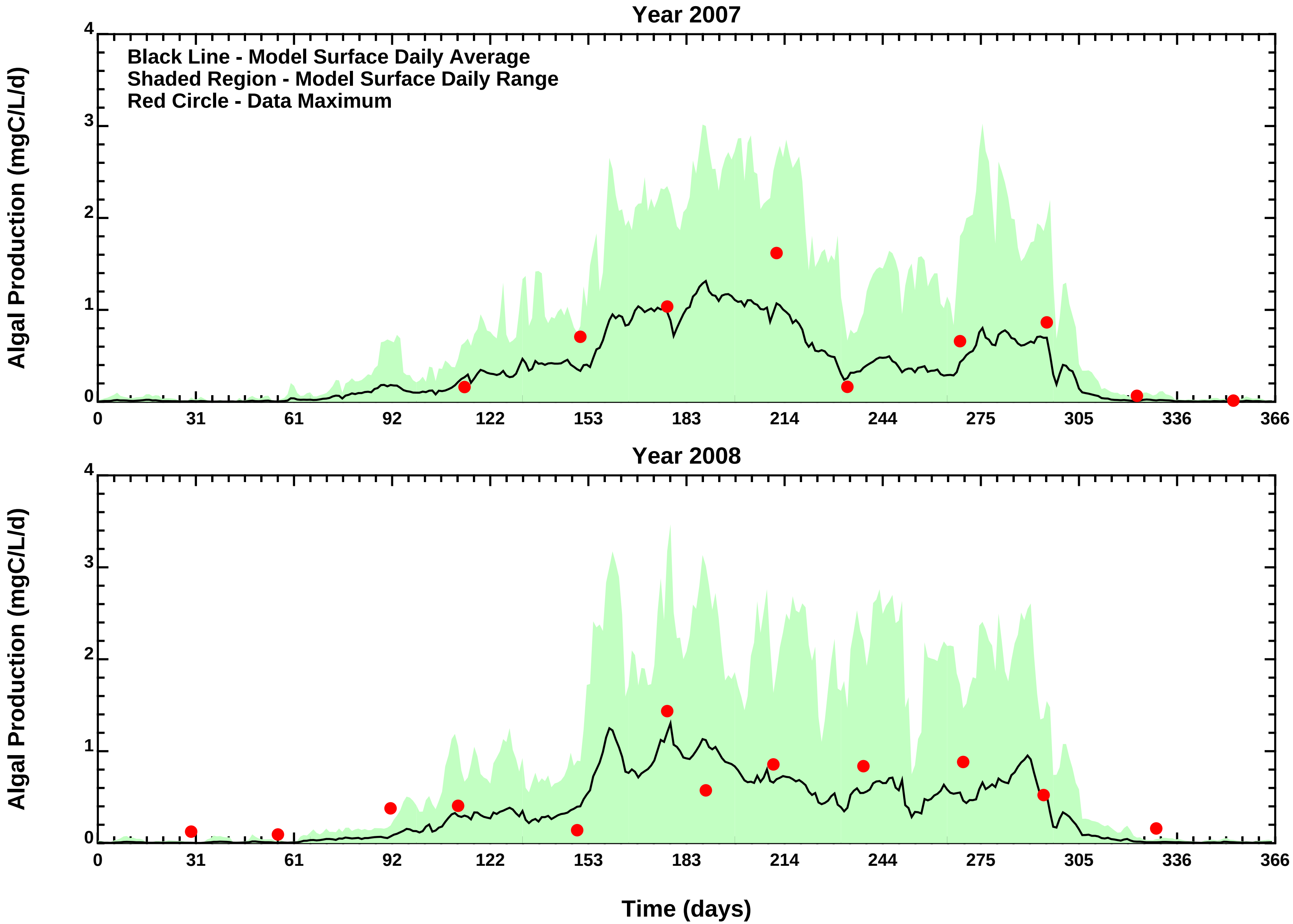

Figure 42. Murderkill River Model-Data Algal Production Comparison
(Powerlines - #206711)

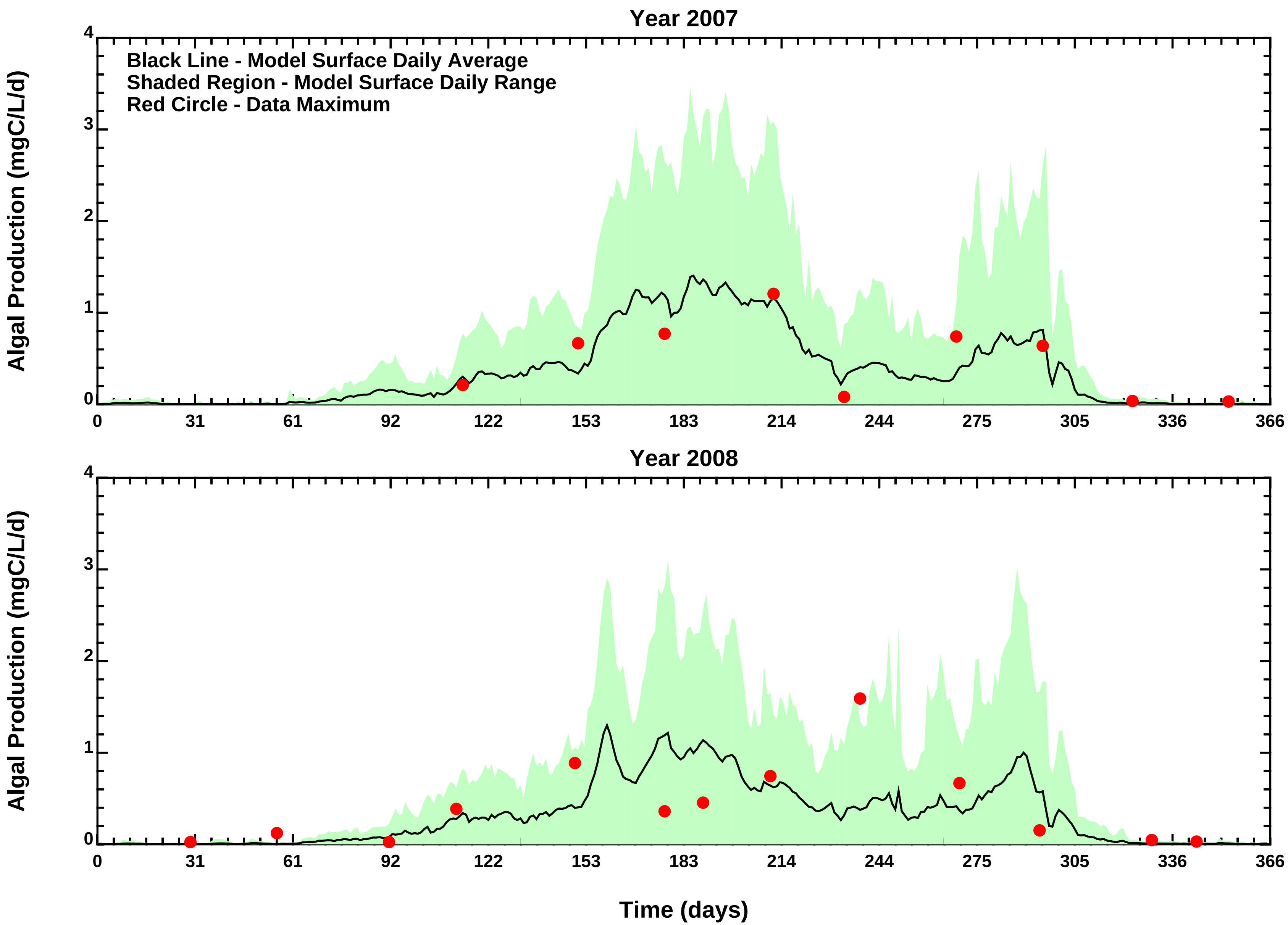

Figure 43. Murderkill River Model-Data Algal Production Comparison
(KCRWTF Canal - #206231)

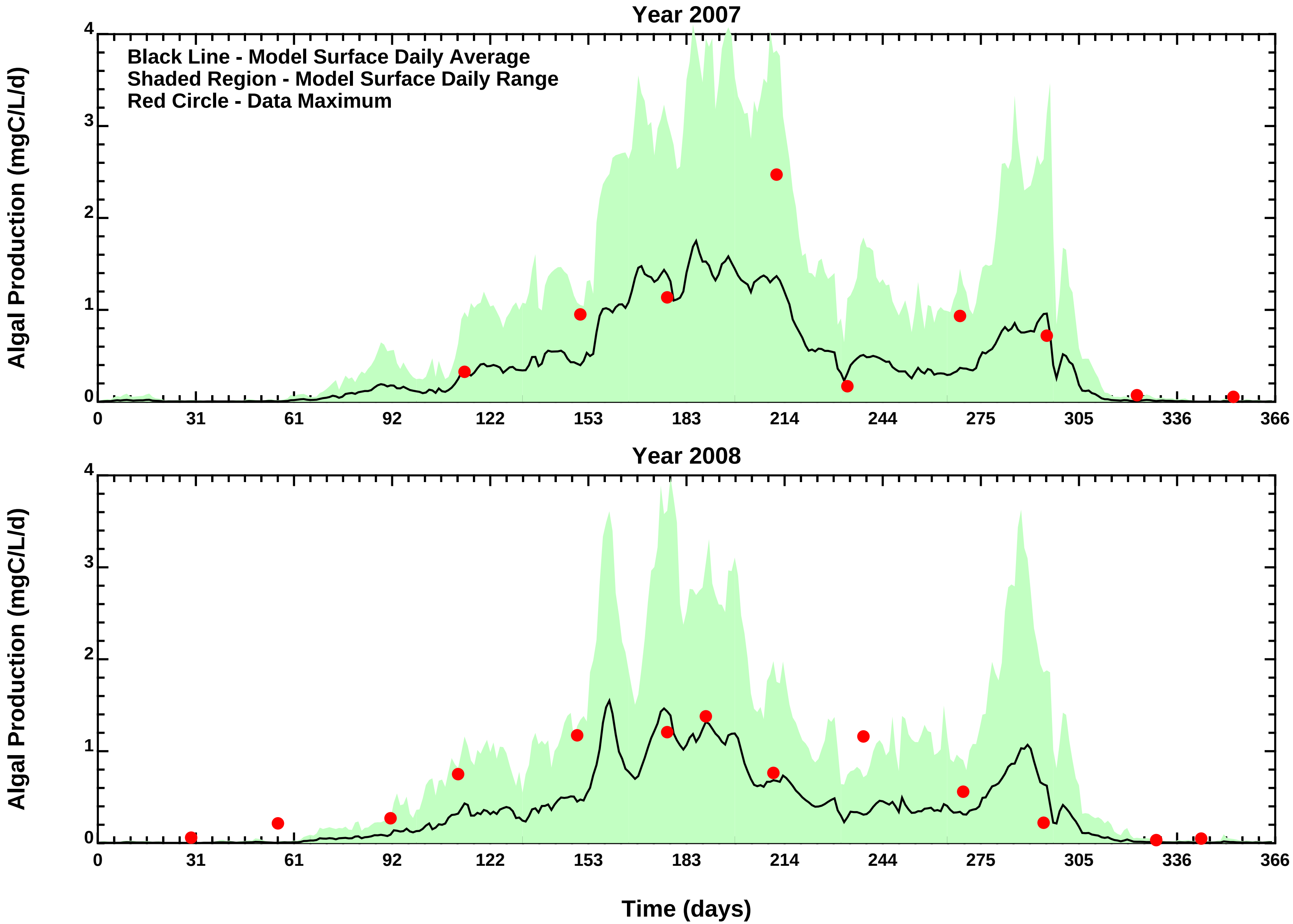

Figure 44. Murderkill River Model-Data Algal Production Comparison
(Bay Road, Route 1/113 - #206091)

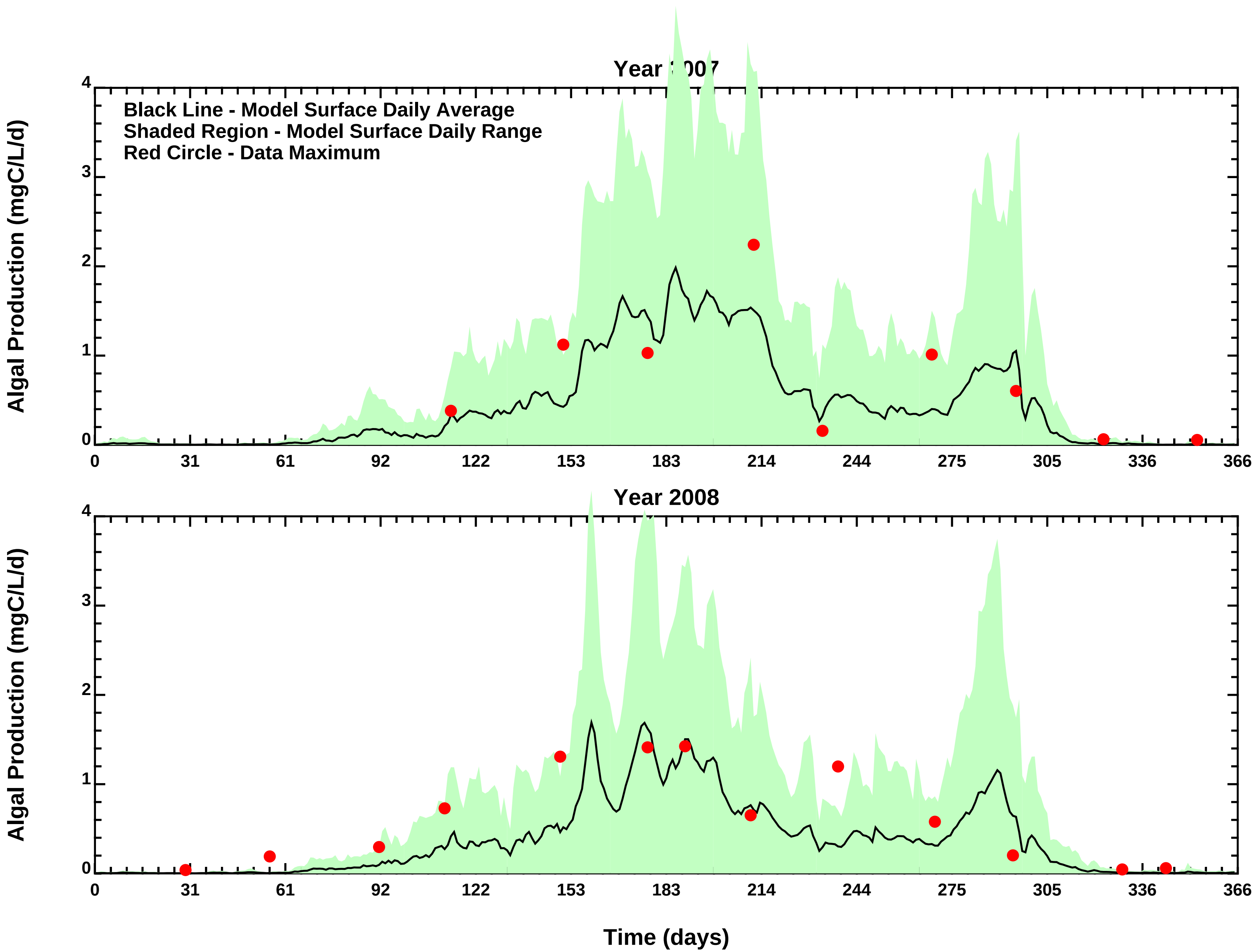

Figure 45. Murderkill River Model-Data Algal Production Comparison
(Spring Creek ar Route 12 Bridge - #206081)

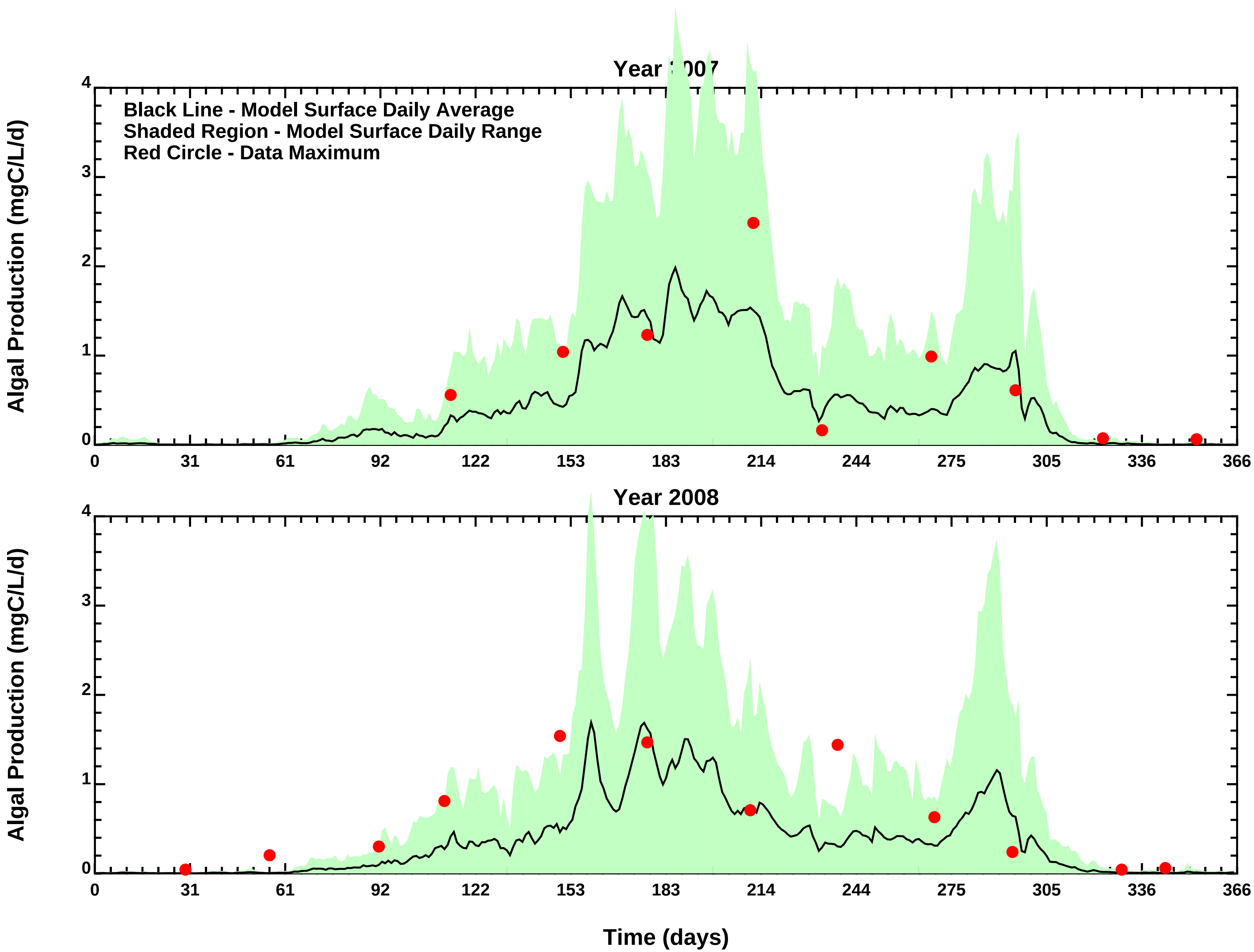

Figure 46. Murderkill River Model-Data Algal Production Comparison
(Spring Creek ar Route 12 Bridge - #206081-dup)

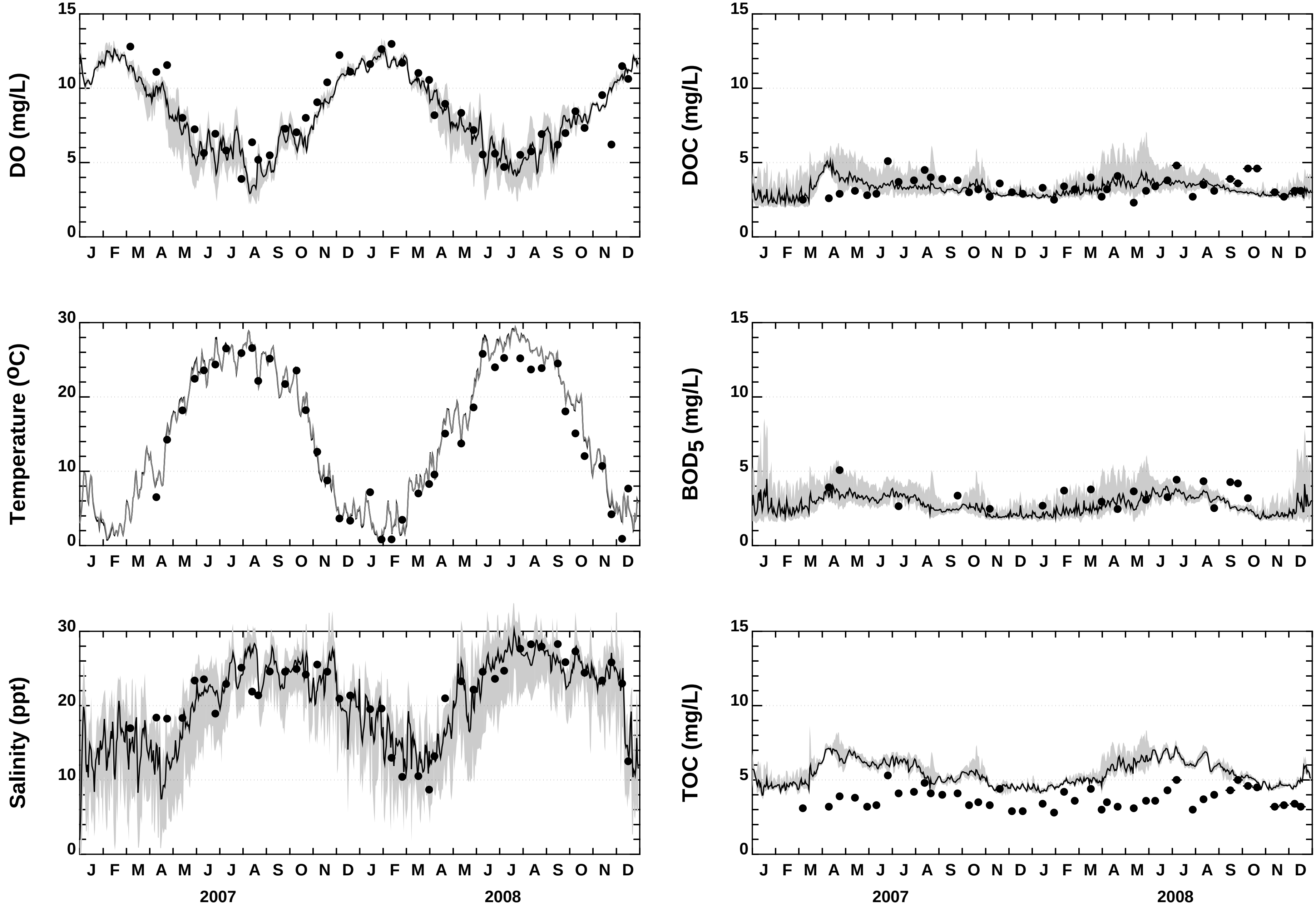

Figure 47. Water Quality Model Calibration/Validation (2007-2008)

Station 206101, Murderkill River at Bowers Beach Warf (Mouth)

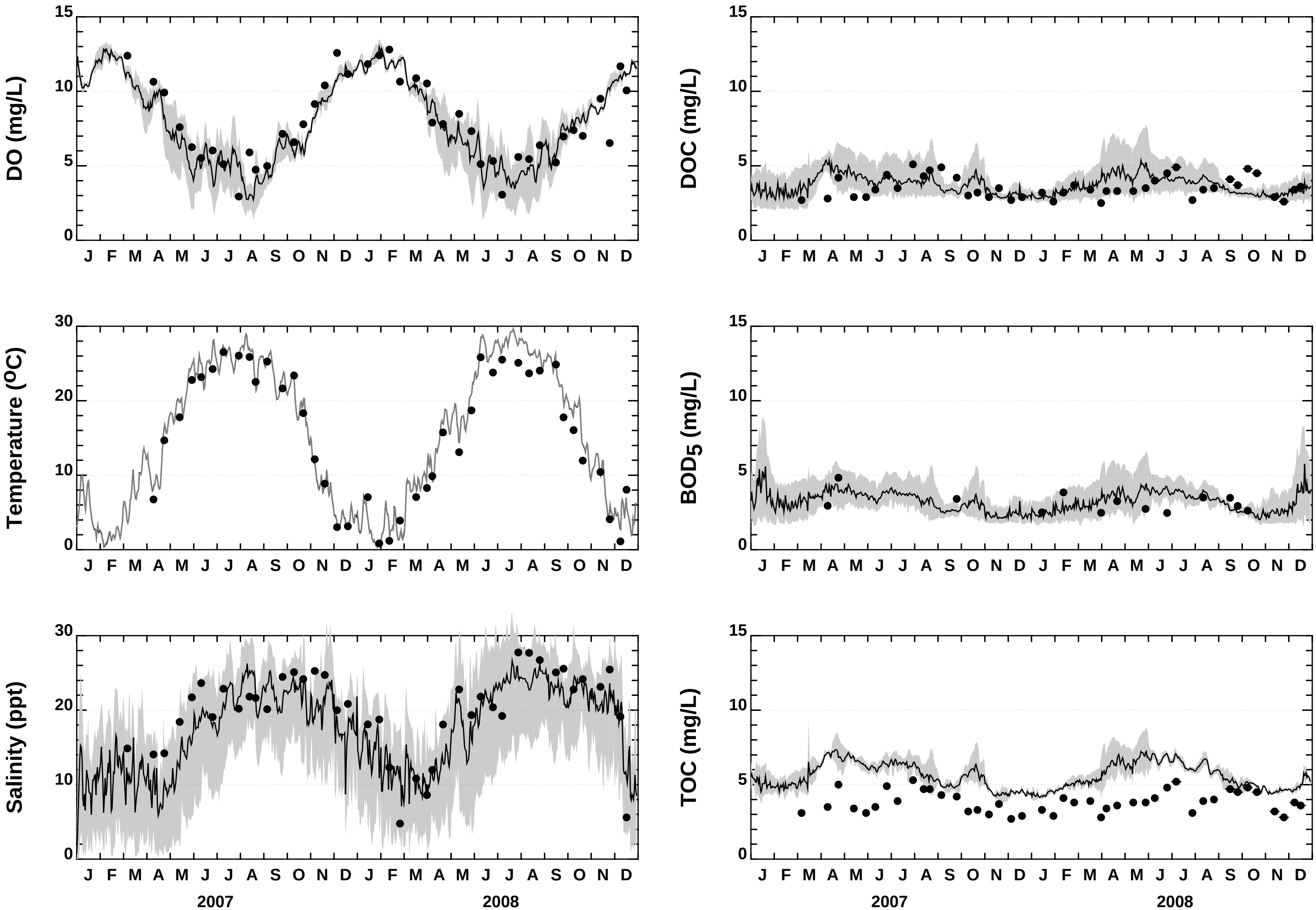

Figure 48. Water Quality Model Calibration/Validation (2007-2008)
Station 206131, Murderkill River at Webb Landing (1.25 Miles from Mouth)

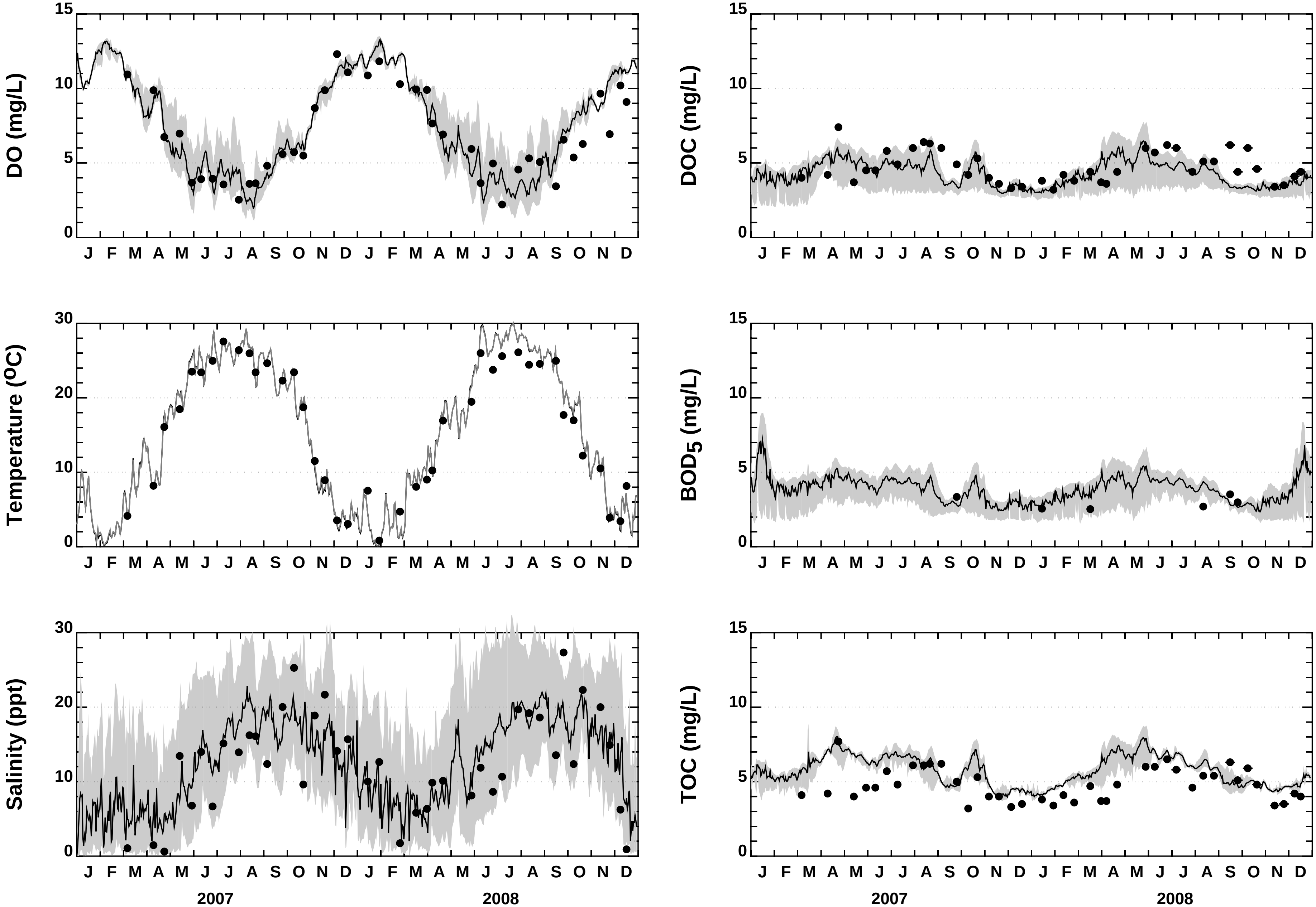

Figure 49. Water Quality Model Calibration/Validation (2007-2008)
Station 206141, Murderkill River near Levee at Milford Neck Wildlife

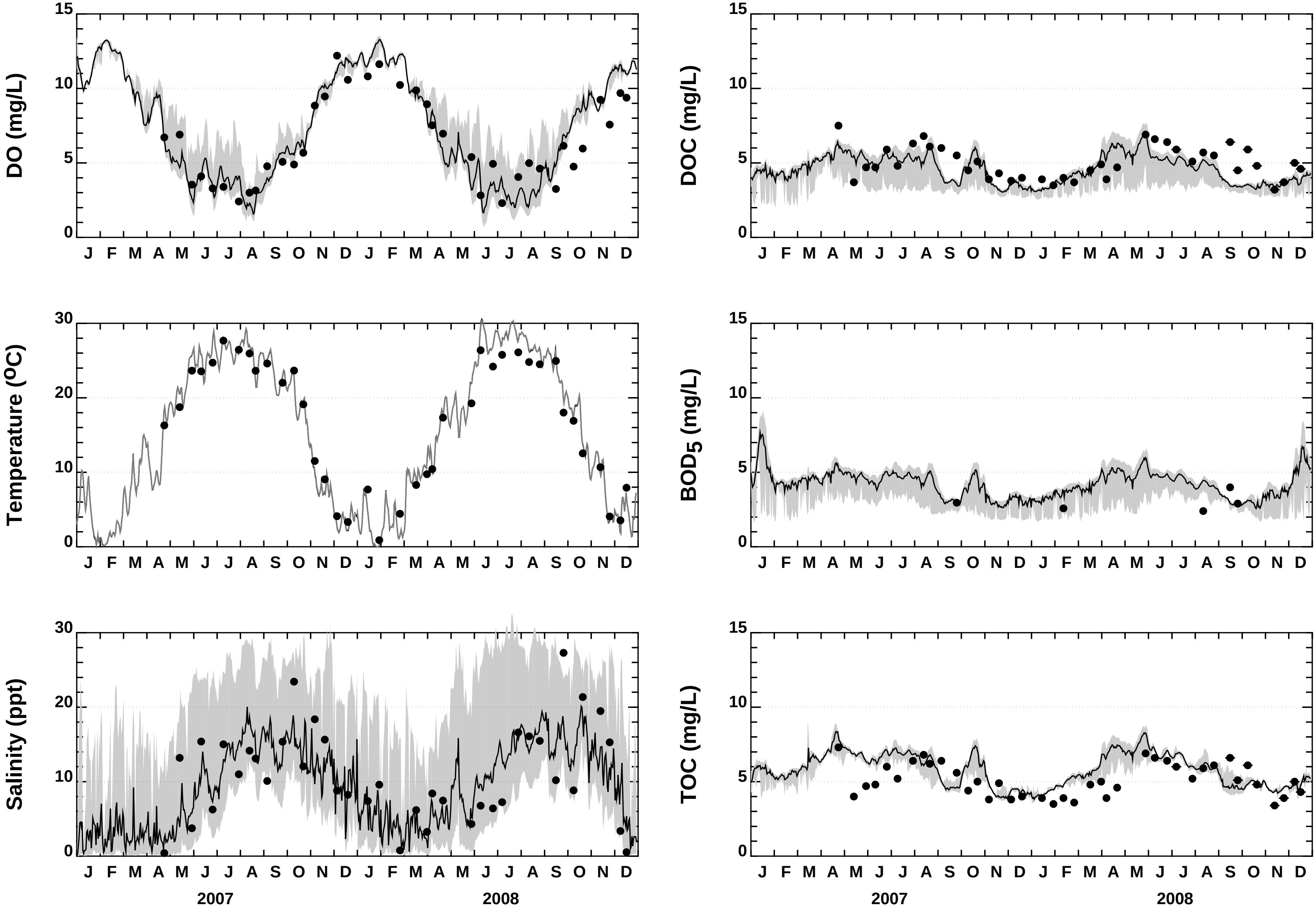

Figure 50. Water Quality Model Calibration/Validation (2007-2008)

Station 206711, Murderkill River near Power Lines (4.45 River Miles)

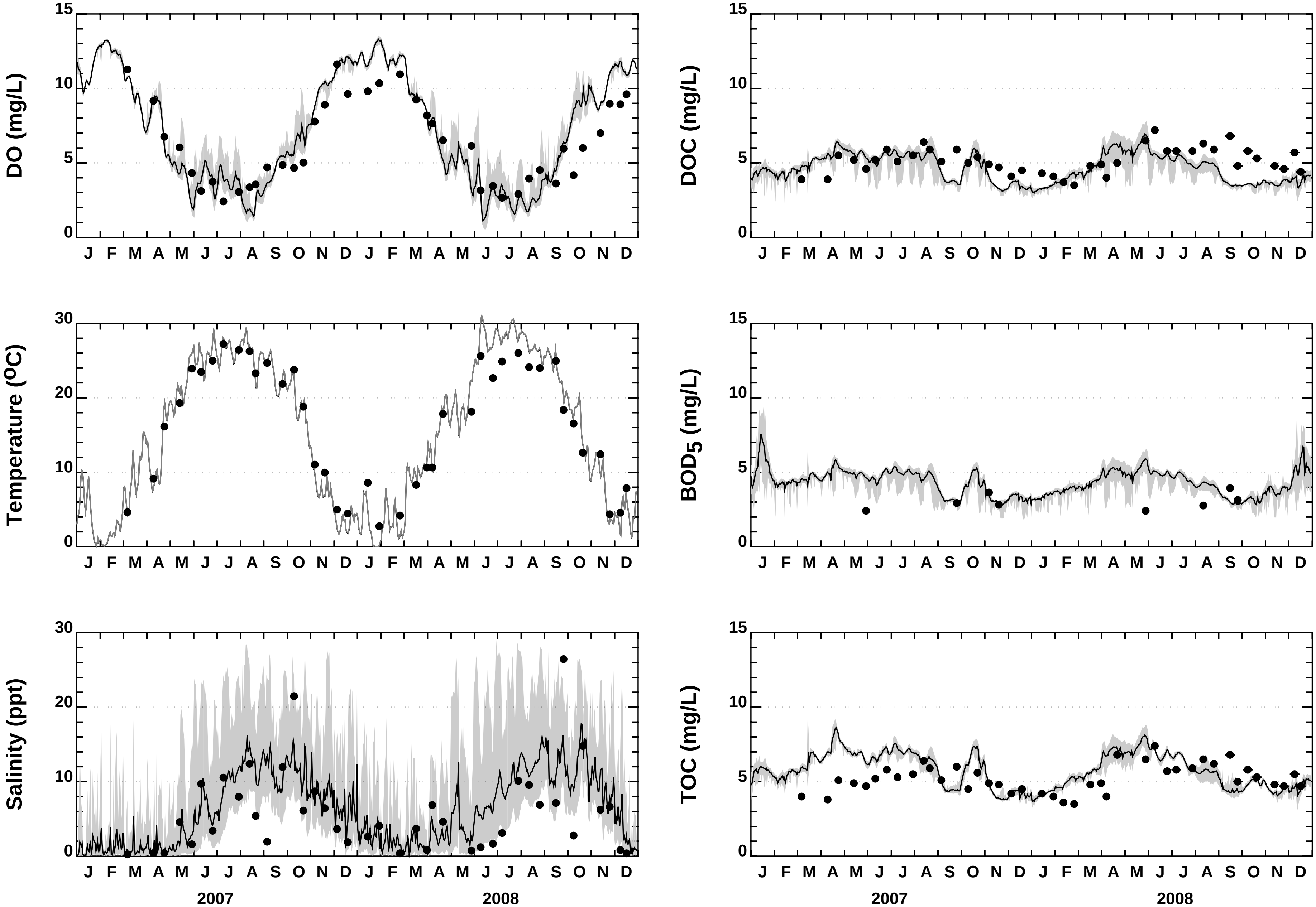

Figure 51. Water Quality Model Calibration/Validation (2007-2008)

Station 206231, Murderkill River at Confluence of Kent County WWTF

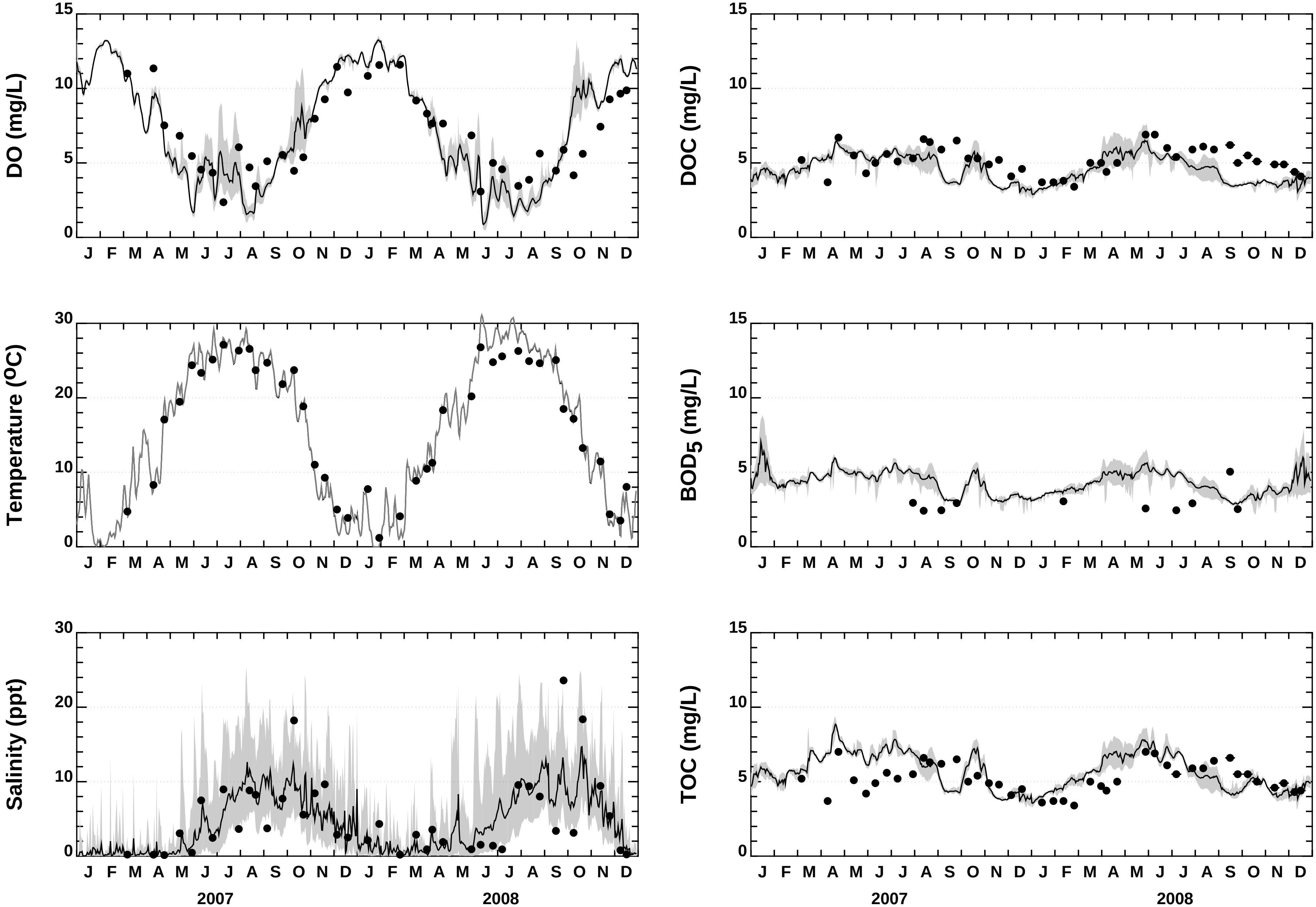

Figure 52. Water Quality Model Calibration/Validation (2007-2008)

Station 206091, Murderkill River at Bay Road (Rt 1/113)

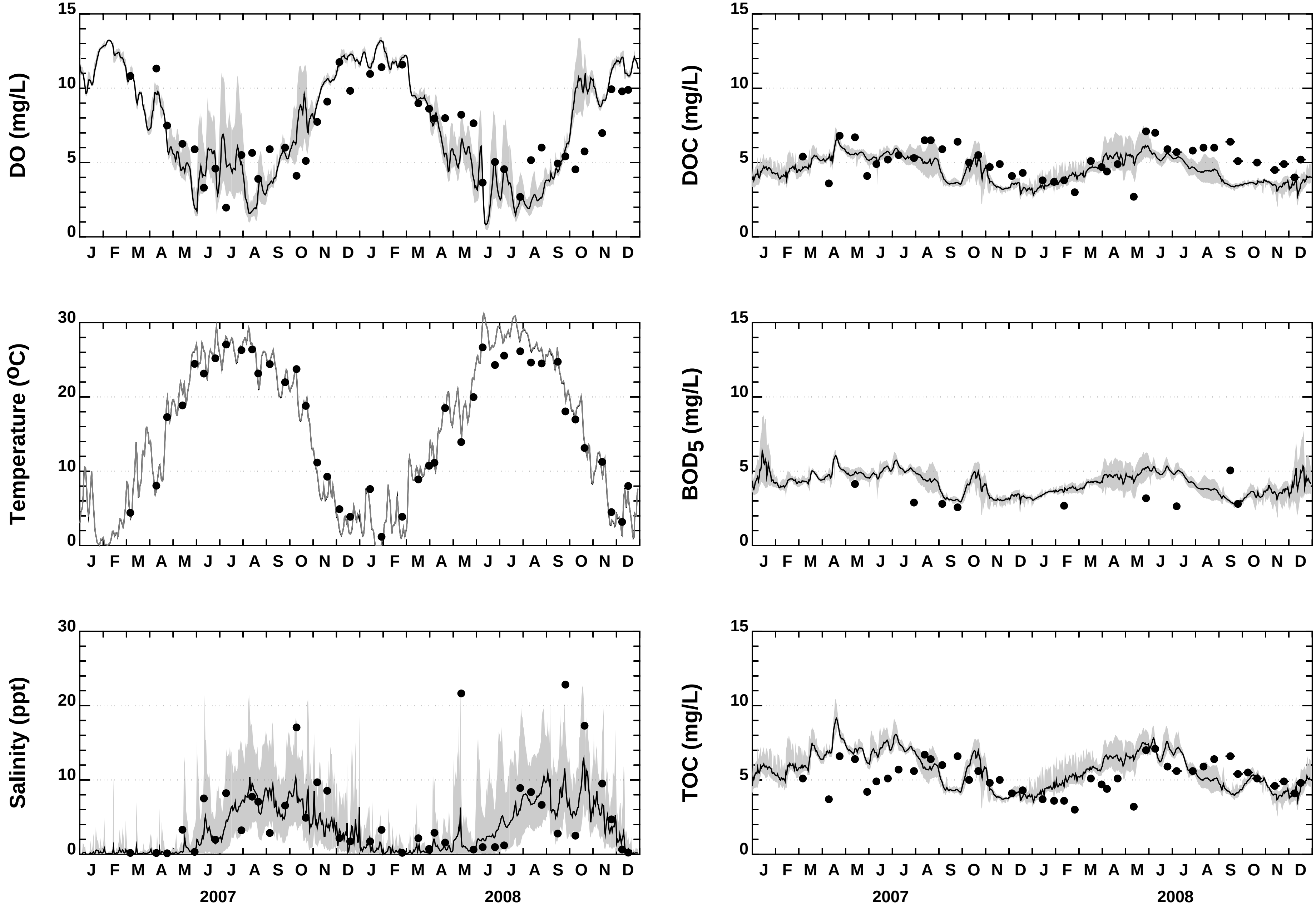

Figure 53. Water Quality Model Calibration/Validation (2007-2008)

Station 206081, Spring Creek at Rt. 12 Bridge at Frederica

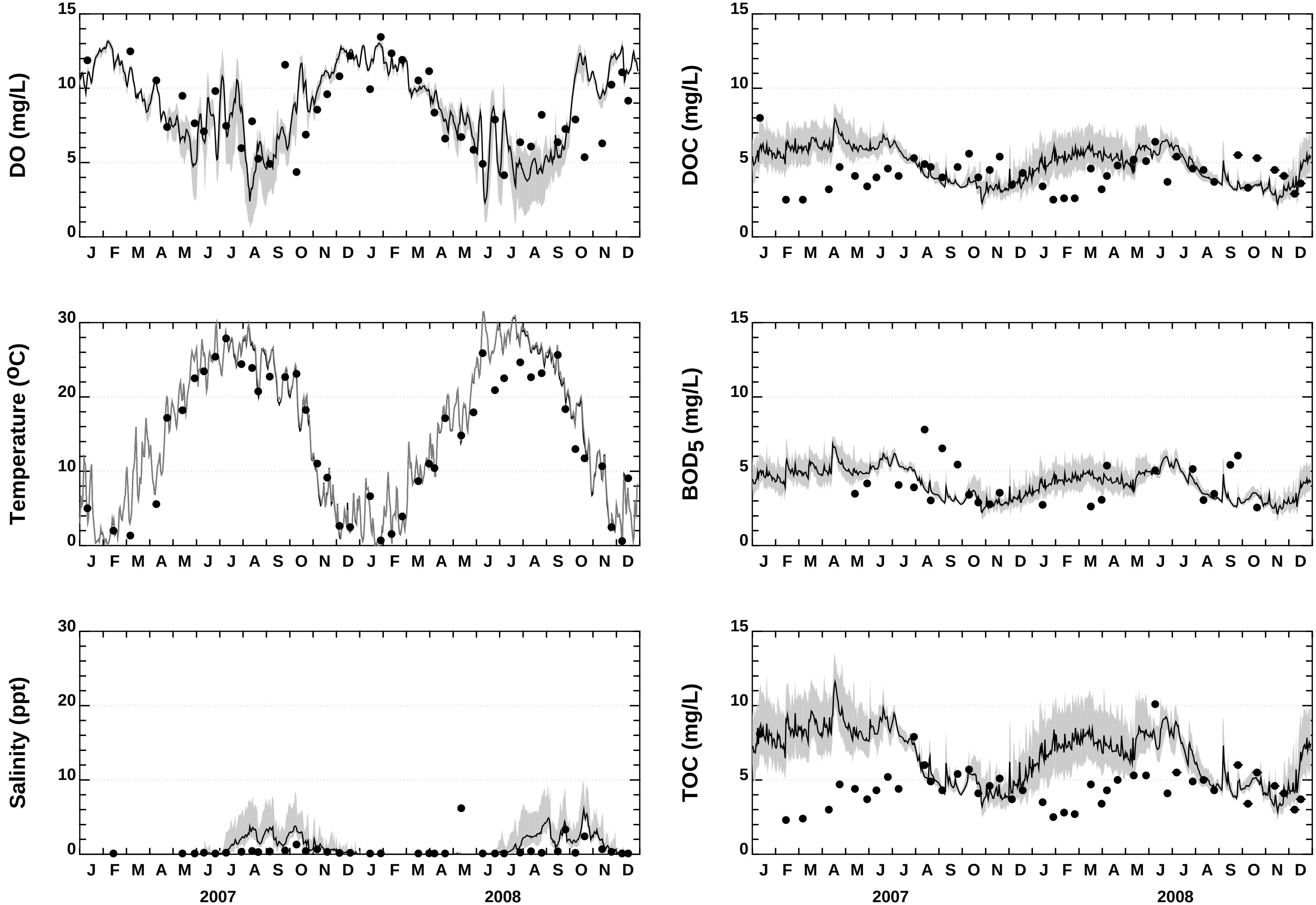

Figure 54. Water Quality Model Calibration/Validation (2007-2008)
Station 206561, Double Run at Barratts Chapel Rd. (Rd. 371)

The model calculated DO was also compared to the USGS continuous DO data collected at Bowers Beach and Frederica (Bay Road) in Figures 55 and 56. The continuous DO data are presents as a daily average (blue circle) and range (vertical blue lines) along with the DNREC grab sample DO data (red circles). Daily average model output is presented as the solid black line with the daily variation over the day represented by the shaded region. At the Bowers Beach station, the model compares very well with the observed daily average data (USGS and DNREC) but under-calculates the daily range during certain periods of the year. The calculated DO at the Frederica station is less than observed during certain time periods with the daily range under-calculated. This location in the river is complex due to the merging of the upper Murderkill River and Spring Creek from both a circulation and salt/fresh marsh interaction perspective. The under-calculation of DO in this area will provide a level of conservatism to the analysis in that calculated DO levels under future scenarios will be lower than may be observed. Table 19 presents the observed and modeled summer average (May-September) DO levels at the DNREC and USGS continuous stations in the river. Overall, the model under-calculates the observed DNREC and USGS summer average DO levels by roughly 0.6 mg/L, the DNREC minimum DO levels by about 1.9 mg/L, and the USGS minimum DO levels by about 0.2 mg/L. Because the USGS continuous data are a better representation of minimum DO levels in the river, an overall model bias was calculated based on the DNREC and USGS average DO levels, and USGS minimum DO levels. Overall, the model under-calculates the observed DO by 0.5 mg/L at all of the tidal river monitoring locations (Bowers Beach to Frederica). When just considering the critical middle part of the river (Rte. 12 Bridge to Power Lines stations), the model under-calculates the observed DO by 0.8 mg/L.

Table 19. Summary of Model-Data Summer Average DO Comparisons

Station	Data		Model	
	2007	2008	2007	2008
Bowers Beach	6.2/3.9	6.3/4.7	5.8/2.2	6.2/2.5
Bowers Beach (continuous)	6.7/2.0	6.5/2.1	5.8/2.2	6.2/2.5
Webbs Landing	5.6/2.9	5.9/3.1	5.1/1.4	5.5/1.4
Milford Neck	4.2/2.5	4.6/2.2	4.4/1.2	4.6/0.9
Power Lines	4.0/2.4	4.3/2.3	3.9/1.1	4.0/0.7
KCRWTF Canal	3.9/2.4	4.0/2.7	3.7/1.1	3.5/0.5
Bay Road	4.8/2.4	4.8/3.1	3.9/1.0	3.5/0.5
Frederica (continuous)	4.8/1.3	5.0/1.3	3.9/1.0	3.5/0.5
Rte. 12 Bridge	4.9/2.0	5.3/2.7	4.2/1.0	3.7/0.5
Double Run	7.7/4.9	6.4/4.2	6.6/1.3	5.7/1.0

4.0/2.4 – Average/minimum model output

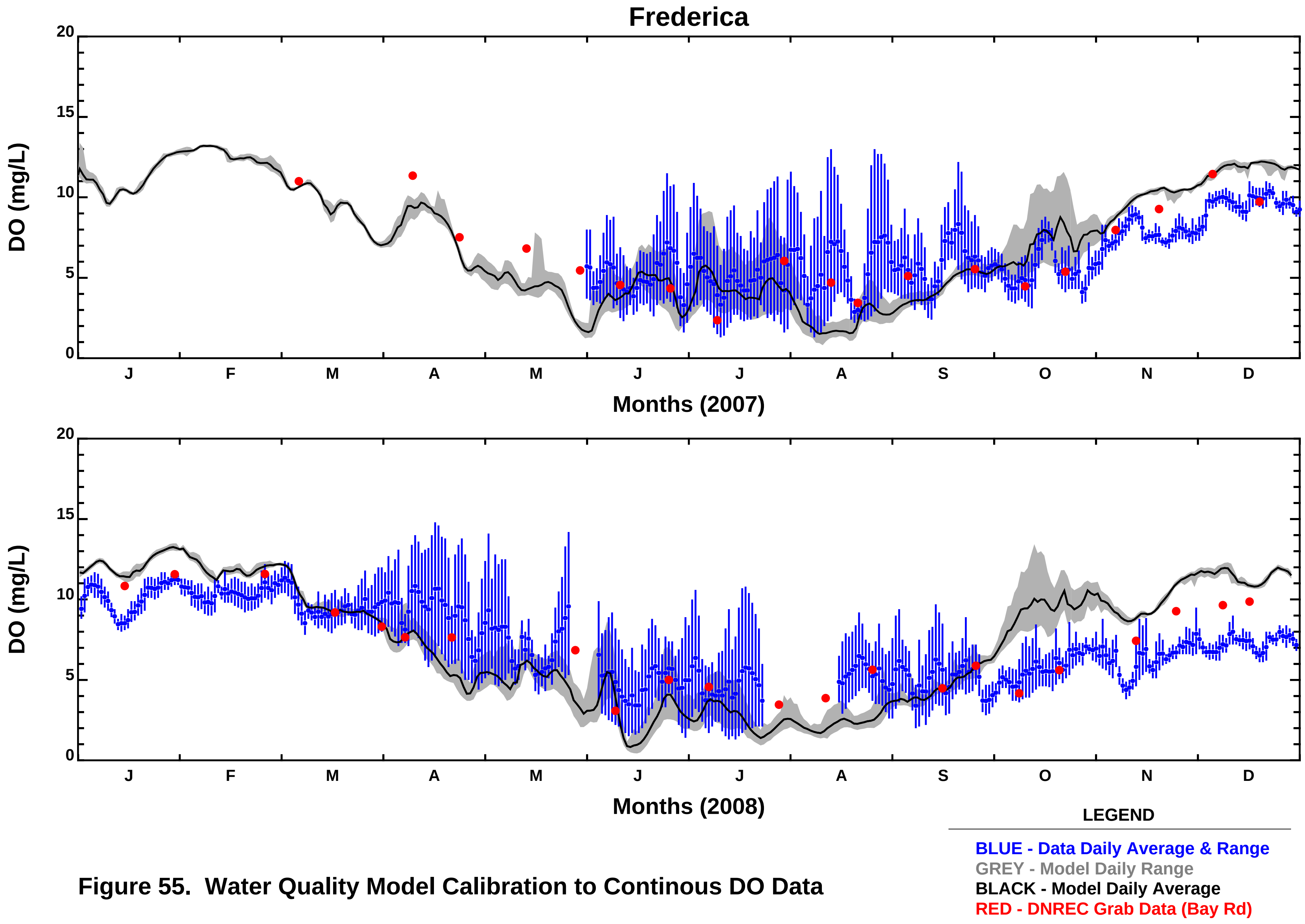

Figure 55. Water Quality Model Calibration to Continous DO Data

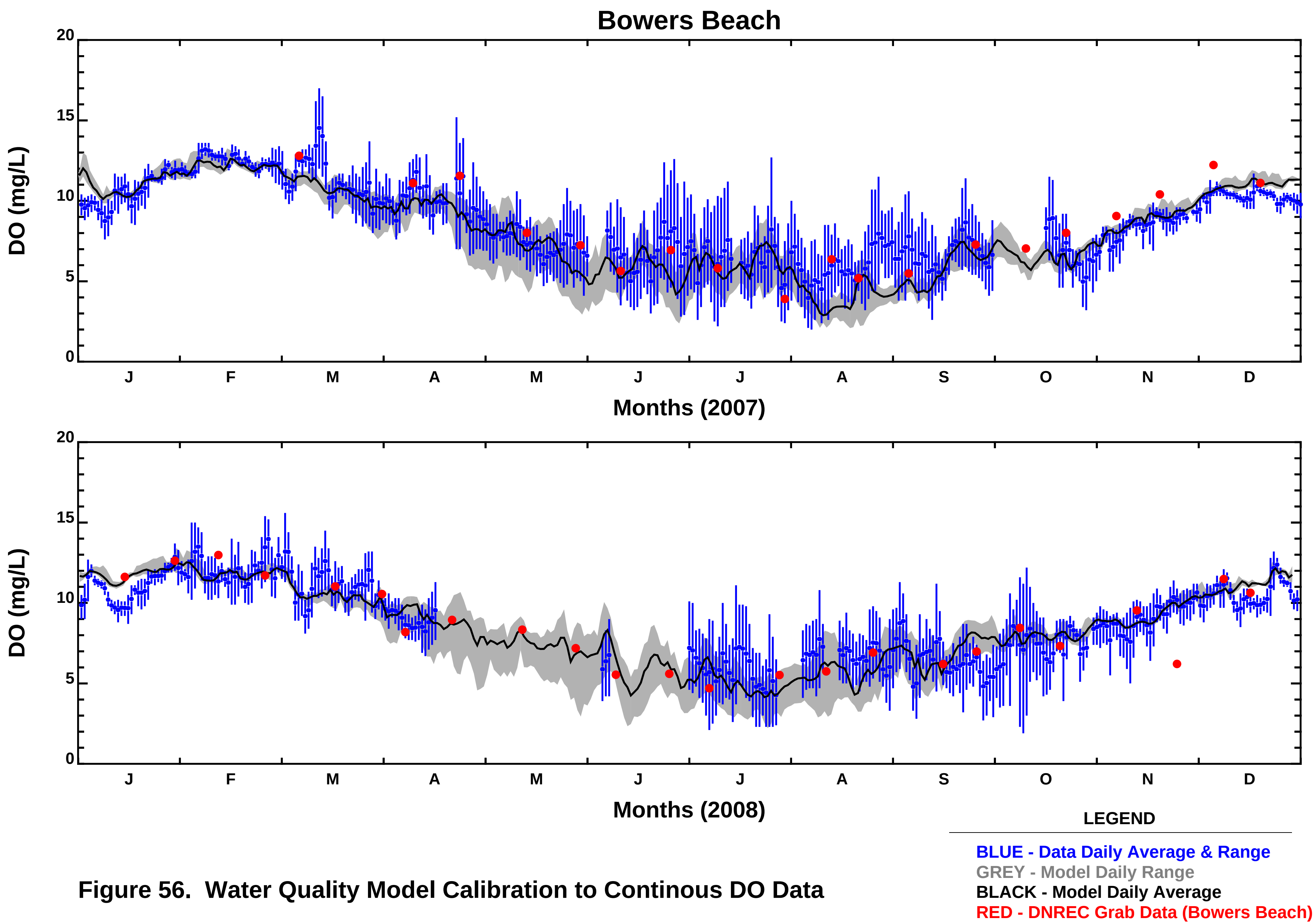

Figure 56. Water Quality Model Calibration to Continous DO Data

4.4.3 Sediment

The sediment flux submodel output was compared to the available SOD and sediment nutrient flux data collected at four stations in the tidal Murderkill River in July 2007. Figures 57 to 60 presents the model output for 2007 and 2008 as compared to measured sediment fluxes of NH_4, NO_3, N_2 (denitrification), PO_4 and SOD. The 2008 model output was also included because the results were similar to 2007. Although there was only one sampling event in the July 2007, the model output compares fairly well to the observations. There are times when the model is over-calculating and under-calculating the observations but overall represents the observations at all stations fairly well. The need to include the sediment flux submodel, even though there was limited data for calibration, was to allow the sediment nutrient fluxes and SOD to be calculated as a function of settling POM and watershed loadings. In addition, sediment characteristics measured during the sediment studies were used to assign deposition rates in the model so that the model output matched the spatial patterns observed. Table 20 presents the calculated SOD and sediment nutrient flux rates calculated at the DNREC monitoring stations. The NO_3 flux is generally into the sediment due to the higher NO_3 concentrations in the water column as compared to sediment.

Table 20. Summary of Model Calculated SOD & Nutrient Fluxes (May-September Average)

	SOD ($gO_2/m^2/d$)	NH_4 Flux* ($mgN/m^2/d$)	NO_3 Flux* ($mgN/m^2/d$)	PO_4 Flux* ($mgP/m^2/d$)
Bowers Beach	1.25/1.03	21.9/15.7	-2.4/2.0	4.1/4.9
Webbs Landing	1.18/1.17	31.7/30.8	-4.7/-0.4	3.7/5.9
Milford Neck	1.79/1.70	65.3/65.0	-8.0/-6.3	8.8/10.5
Power Lines	1.43/1.36	52.0/51.7	-14.9/-14.3	7.3/8.0
KCRWTF Canal	1.16/1.11	44.8/43.9	-24.3/-25.6	7.3/7.3
Bay Road	1.03/1.06	45.3/45.7	-33.8/-36.4	7.2/8.0
Rte. 12 Bridge	1.01/1.04	45.9/46.7	-40.7/-44.7	6.8/7.2
Double Run	1.02/0.99	41.3/38.2	-66.8/-75.6	7.2/5.1

1.43/1.36 – 2007/2008 model output
* - Positive is out of sediment, negative is into sediment

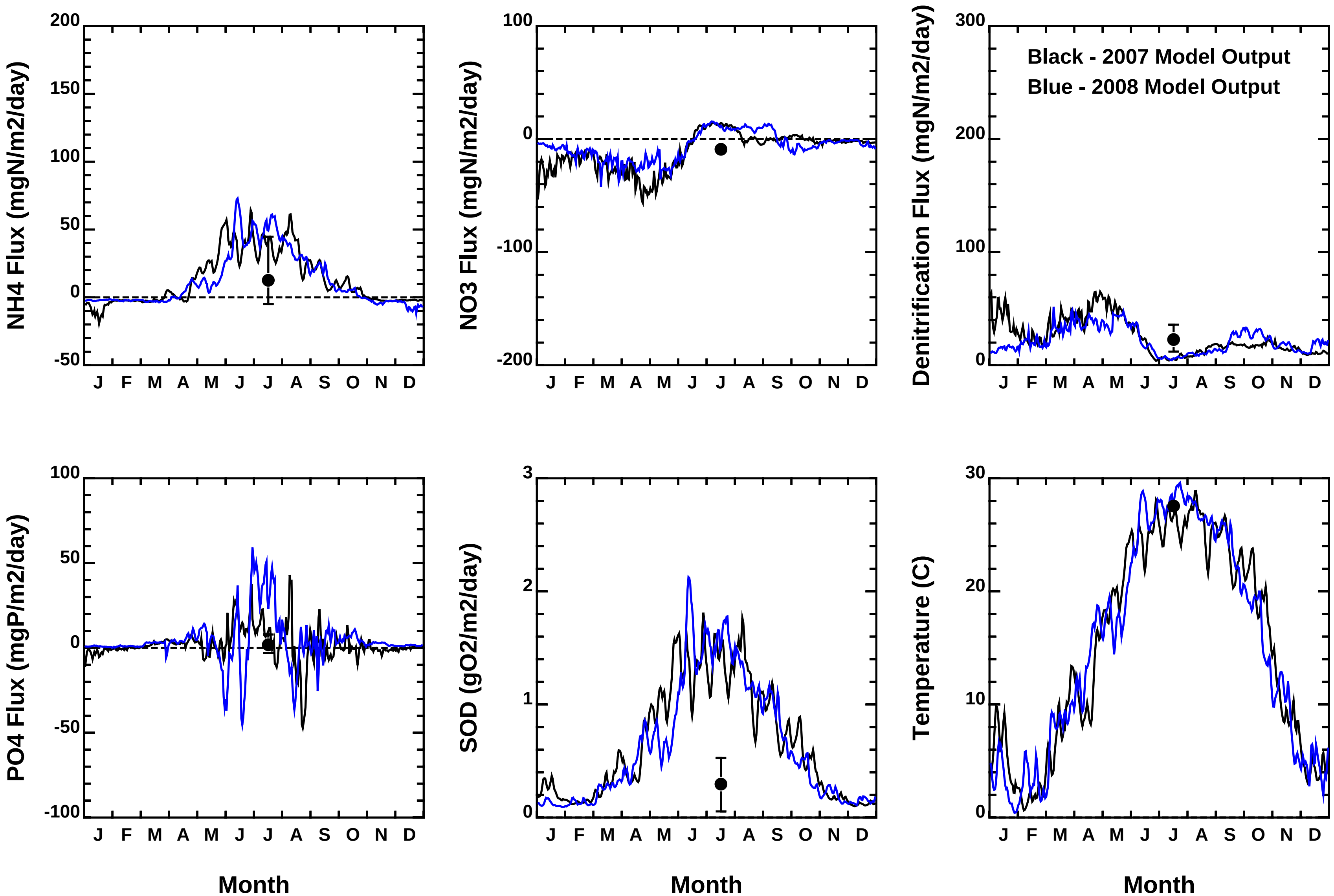

Figure 57. Murderkill River Sediment Model Calibration/Validation (2007-2008)
(Webbs Landing - #206131)

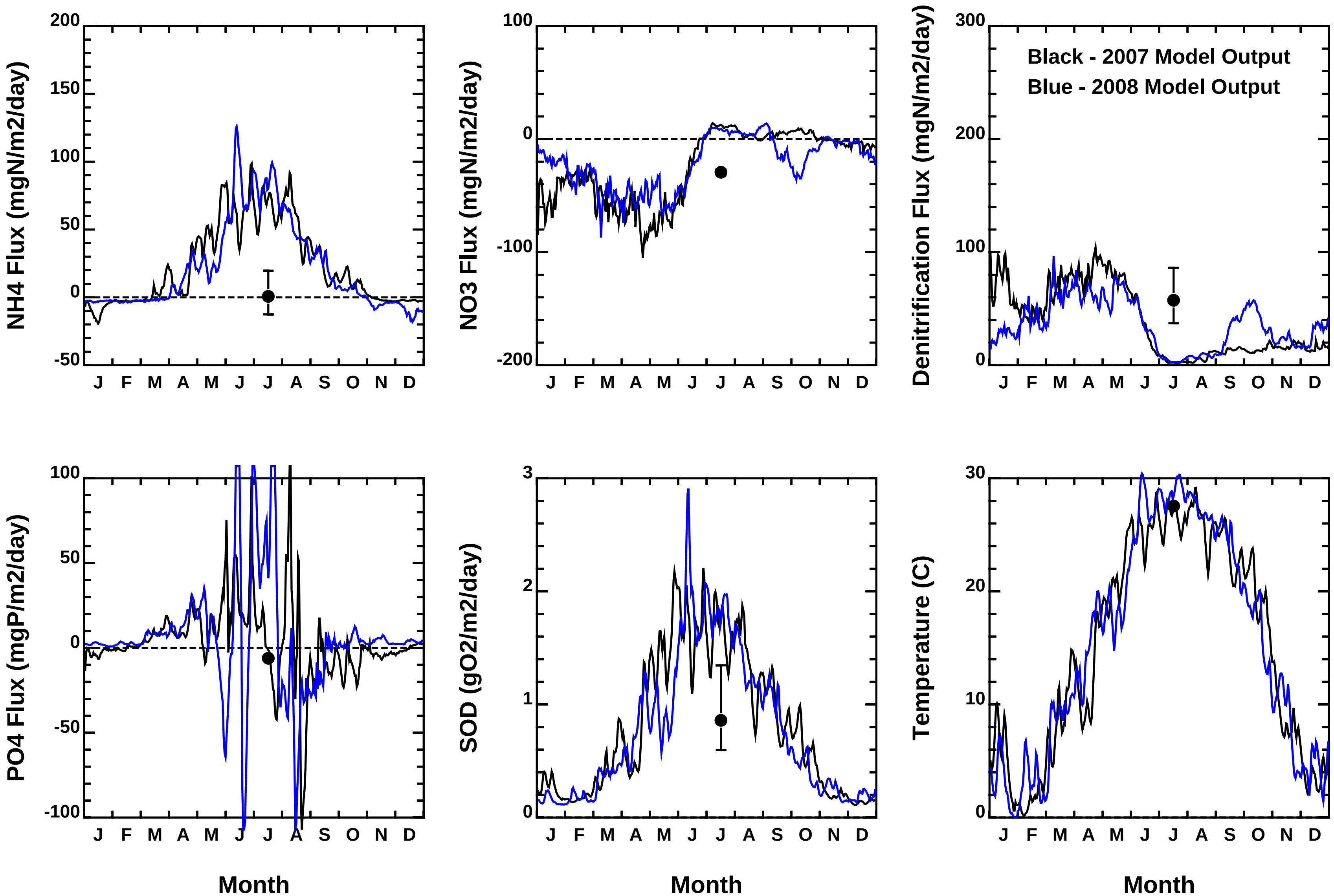

Figure 58. Murderkill River Sediment Model Calibration/Validation (2007-2008)
(Power Lines - #206711)

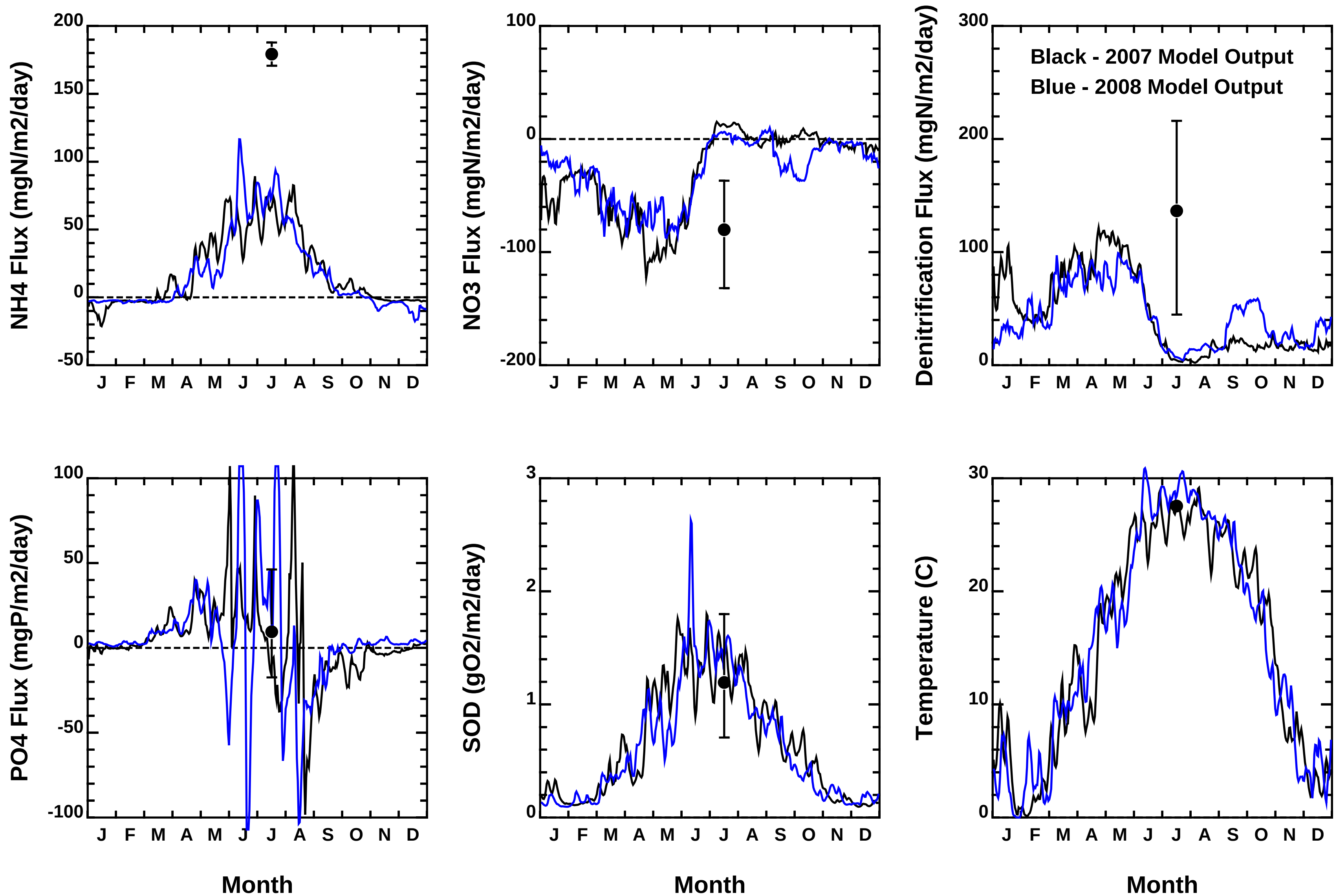

Figure 59. Murderkill River Sediment Model Calibration/Validation (2007-2008)
(KCRWTF Canal - #206231)

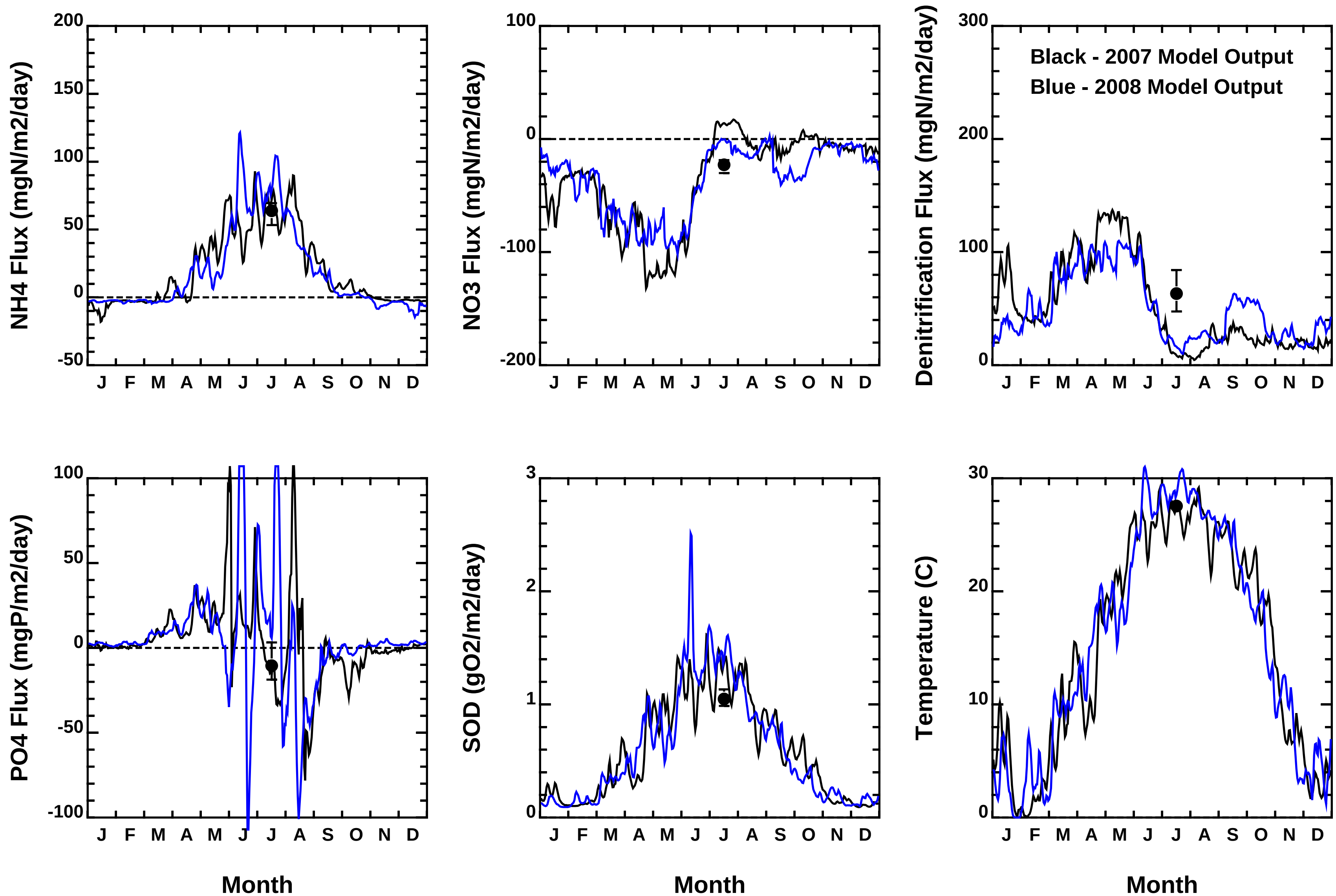

Figure 60. Murderkill River Sediment Model Calibration/Validation (2007-2008)
(Bay Road, Route 1/113 - #206091)

4.5 WATER QUALITY MODEL ERROR ANALYSIS

In order to quantitatively assess the level of model calibration and validation, error analyses were completed. Typically, model calibration and validation are completed based on a weight of evidence approach. That is, model comparison to observed data is completed in a qualitative manner by the modeler to achieve a best fit of the model to data at all monitoring stations for the parameters being analyzed. Although this method balances model comparison to data with the modeler's understanding of the physical, chemical and biological characteristics of the system it does not provide a quantitative measure of the goodness of fit. Ultimately, the goal of model calibration and validation is "not to curve fit model to data, but to describe the behavior of the data with a modeling framework of the principal mechanisms relevant to the problem" (Thomann, 1982).

There are number of measures that can be used to quantitatively assess model goodness of fit. Many of these measures are described in detail along with a good discussion of overall model verification assessments in a number of journal papers (Thomann, 1982; Ambrose and Roesch, 1982; Reckhow, et al., 1990). Not all measures are suitable for the error analysis due to inherent benefits and disadvantages in the measure that depend on the magnitude of the parameter under consideration. The following measures were selected for the Murderkill River error analysis.

- Model Bias: $= \bar{Y} - \bar{X}$
- Relative Model Bias: $= \dfrac{\bar{Y} - \bar{X}}{\bar{X}}$
- Mean Absolute Error: $= \dfrac{1}{n}\sum_{i=1}^{n}\left|Y_i - X_i\right|$
- Median Relative Error: $= median\left[\dfrac{(Y_i - X_i)}{X_i}\right]$

where: Y = model, X = data, $\bar{Y}$ = average of Y, $\bar{X}$ = average of X.

Table 21 presents the results of the error analyses at monitoring stations in the tidal Murderkill River for TN, NO_2+NO_3, TP, PO_4, chl-a and DO. The tidal stations were selected because they ultimately integrate all of the point and nonpoint source loads entering the river from the watershed and the effects of the downstream interaction with Delaware Bay. This table presents the number of observations, data and model averages, model bias, relative model bias, mean absolute error and median relative error. The chl-a data does include many measurements of very high chl-a levels (greater than 50 μg/L) that may represent patches of high algal biomass. In this error analysis, chl-a data greater than 50 μg/L were not used. In general, the relative measures are very sensitive to the magnitude of the parameter (particularly at low values). For instance, at low NO_2+NO_3 levels near the mouth of the river (<0.4 mg/L) the average model bias is 0.05 mg/L (good relative model bias of 18%) but the median relative error is high (85%). Therefore, it is important to also consider the other measures such as model bias (difference between model and data averages and mean absolute error). Figure 61 presents a summary of the median relative error (absolute) for TN, NO_2+NO_3, TP, PO_4, chl-a and DO for the seven tidal stations. Excluding some of the larger errors near the mouth of the river, the average median relative error is 31% for TN, 20% for NO_2+NO_3, 16% for TP, 58% for PO_4, 40% for chl-a, and 1% for DO. The error analysis does indicate that the model under-estimates the observed TN, NO_2+NO_3, and chl-a data; slightly over-estimates the TP and PO_4 data; and does a very good job of estimating the observed DO data. The average median relative errors calculated for the Murderkill River water quality model are within typical ranges for other eutrophication modeling studies.

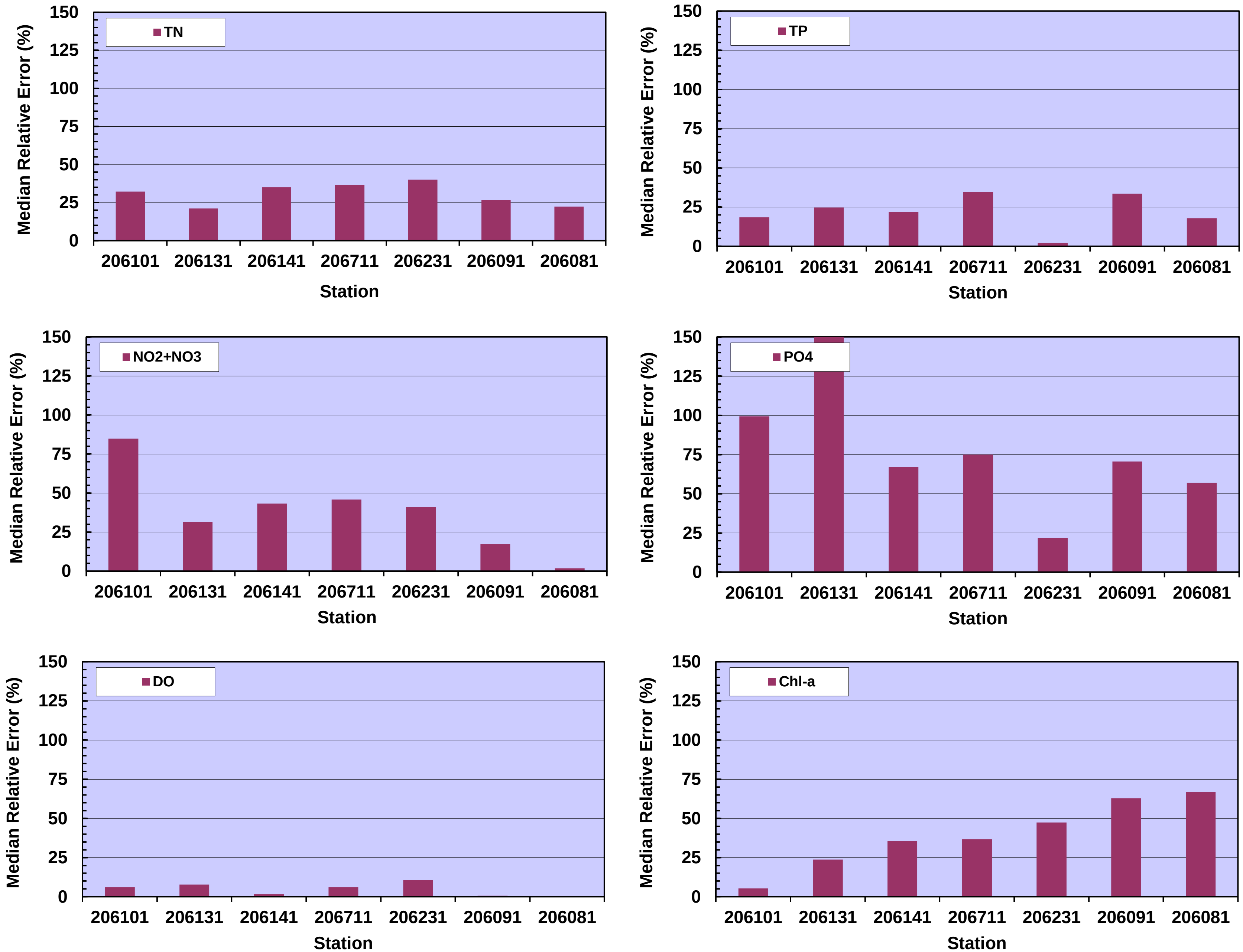

Figure 61. Murderkill River Model Error Analysis Summary

Table 21. Murderkill River Water Quality Error Analysis Results

Station	# of Observations	Data Average	Model Average	Model Bias	Relative Model Bias (%)	Mean Absolute Error	Median Relative Error (%)
TN (mg/L)							
206101	43	1.51	1.01	-0.51	-33.5	0.59	-32.2
206131	43	1.57	1.18	-0.38	-24.4	0.53	-21.1
206141	42	2.08	1.43	-0.65	-31.2	0.69	-35.1
206711	40	2.11	1.55	-0.56	-26.6	0.71	-36.6
206231	42	2.99	1.82	-1.17	-39.2	1.22	-40.0
206091	42	2.63	1.97	-0.67	-25.4	0.75	-26.7
206081	43	2.71	2.08	-0.63	-23.3	0.79	-22.4
Average		**2.23**	**1.57**	**-0.65**	**-29.1**	**0.75**	**-30.6**
NO_2+NO_3 (mg/L)							
206101	43	0.27	0.32	0.05	17.5	0.23	84.8
206131	43	0.39	0.45	0.06	16.4	0.32	31.5
206141	42	0.94	0.65	-0.28	-30.2	0.44	-43.3
206711	40	0.98	0.76	-0.22	-22.4	0.46	-45.7
206231	42	1.74	1.04	-0.70	-40.3	0.90	-40.9
206091	42	1.47	1.23	-0.24	-16.5	0.51	-17.4
206081	43	1.45	1.38	-0.07	-4.7	0.50	-1.7
Average		**1.03**	**0.83**	**-0.20**	**-11.5**	**0.48**	**-4.7**

Table 21. Murderkill River Water Quality Error Analysis Results							
Station	**# of Observations**	**Data Average**	**Model Average**	**Model Bias**	**Relative Model Bias (%)**	**Mean Absolute Error**	**Median Relative Error (%)**
TP (mg/L)							
206101	43	0.219	0.137	-0.082	-37.3	0.124	-18.6
206131	43	0.219	0.227	0.008	3.5	0.139	24.8
206141	42	0.262	0.329	0.067	25.7	0.127	21.9
206711	40	0.278	0.399	0.121	43.5	0.148	34.5
206231	42	0.576	0.408	-0.168	-29.2	0.317	-2.2
206091	42	0.283	0.381	0.099	35.0	0.157	33.6
206081	43	0.286	0.340	0.054	18.9	0.124	17.9
Average		**0.303**	**0.317**	**0.014**	**8.6**	**0.162**	**16.0**
PO_4 (mg/L)							
206101	43	0.034	0.056	0.022	65.6	0.029	99.5
206131	43	0.050	0.121	0.070	139.5	0.075	170.7
206141	42	0.134	0.213	0.080	59.6	0.085	67.1
206711	40	0.179	0.285	0.106	59.0	0.122	75.0
206231	42	0.413	0.300	-0.113	-27.3	0.254	21.9
206091	42	0.148	0.278	0.129	87.3	0.132	70.6
206081	43	0.133	0.233	0.101	75.8	0.111	57.1
Average		**0.156**	**0.212**	**0.056**	**65.6**	**0.115**	**80.3**

Table 21. Murderkill River Water Quality Error Analysis Results							
Station	**# of Observations**	**Data Average**	**Model Average**	**Model Bias**	**Relative Model Bias (%)**	**Mean Absolute Error**	**Median Relative Error (%)**
Chl-a (µg/L)							
206101	35	25.7	25.8	0.1	0.5	9.4	-5.3
206131	37	24.2	21.6	-2.6	-10.7	10.6	-23.8
206141	40	19.9	15.8	-4.0	-20.3	9.3	-35.6
206711	37	18.8	13.5	-5.3	-28.1	9.3	-36.8
206231	38	15.6	11.9	-3.7	-23.5	9.4	-47.4
206091	36	21.1	11.6	-9.5	-44.9	11.9	-62.9
206081	35	21.3	12.0	-9.2	-43.5	13.8	-66.8
Average		**20.9**	**16.0**	**-4.9**	**-24.4**	**10.5**	**-39.8**
DO (mg/L)							
206101	43	8.35	8.01	-0.35	-4.2	1.01	-6.1
206131	43	7.97	7.63	-0.34	-4.2	1.09	-7.7
206141	41	6.82	7.09	0.27	4.0	1.09	1.8
206711	39	6.39	6.68	0.29	4.6	1.25	6.1
206231	41	6.29	6.72	0.43	6.9	1.33	10.7
206091	41	6.88	6.85	-0.03	-0.5	1.54	-0.6
206081	42	7.00	7.07	0.07	1.0	1.67	0.0
Average		**7.10**	**7.15**	**0.05**	**1.1**	**1.28**	**0.6**

Another way to view model-data comparisons is through probability distributions of model output and observed data. Probability distributions are useful for presenting the mean and variation of a data set, and also provide a means for determining compliance (percent exceedance) from a given value (e.g., a water quality standard). The method for developing the distribution is to rank the data set from lowest to highest, calculate a percentage for each point (i/n-1) and to plot the transformed data on a probability scale, which implies a normal or log-normal distribution. The x-scale represents the percentage of data that are less than corresponding y-scale value (% less than or equal to) or conversely, 100 minus this percentage represents the percentage of the data exceeding the y-scale value. Figures 62-68 present the probability distributions at the seven tidal stations in the Murderkill River for TN, NO_2+NO_3, TP, PO_4, chl-a and DO. In these figures the model is represented as the solid line for the 2-year modeling period and the data as the filled circles. In general, the model output for these parameters captures the overall variability observed in the data but depending on the location can either over- or under-estimate the median observed data. Since there are sufficient nutrient levels in the river (i.e., much greater than algal growth limiting levels), the under-estimation of nitrogen levels and over-estimation of phosphorus levels should not have a significant impact on the calculated chl-a levels. The model under-estimation of chl-a levels is probably more driven by the assigned light regime in the model, which was based on limited secchi depth and PAR calculated light extinction coefficients. The main focus of the Murderkill River water quality modeling effort was to assess the factors influencing DO levels and for assisting in the development of site-specific DO criteria for the river. In this respect, the model estimation of observed DO levels in the river is very good.

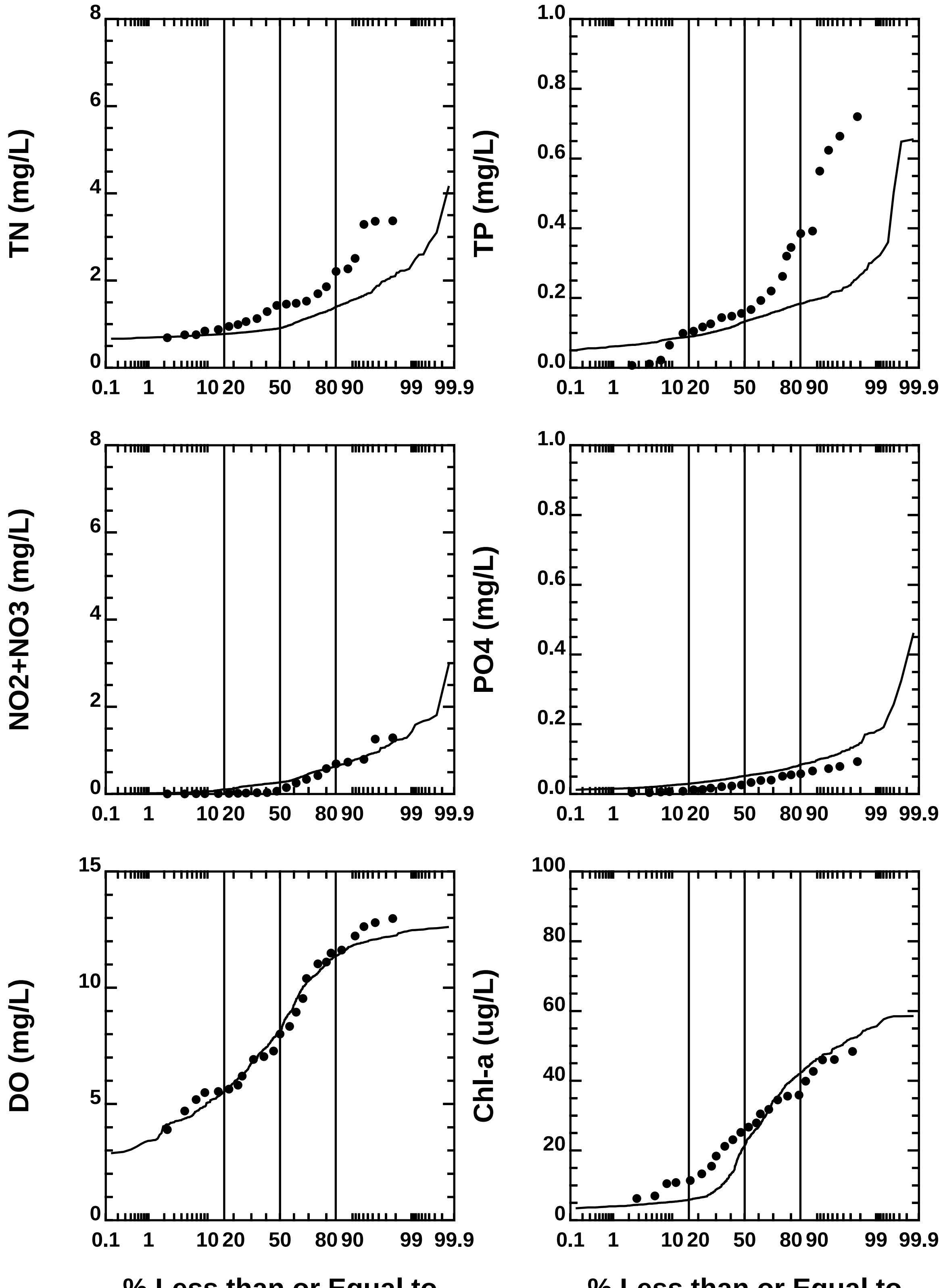

Figure 62. Water Quality Model-Data Probability Distributions
(Bowers Beach - 206101)

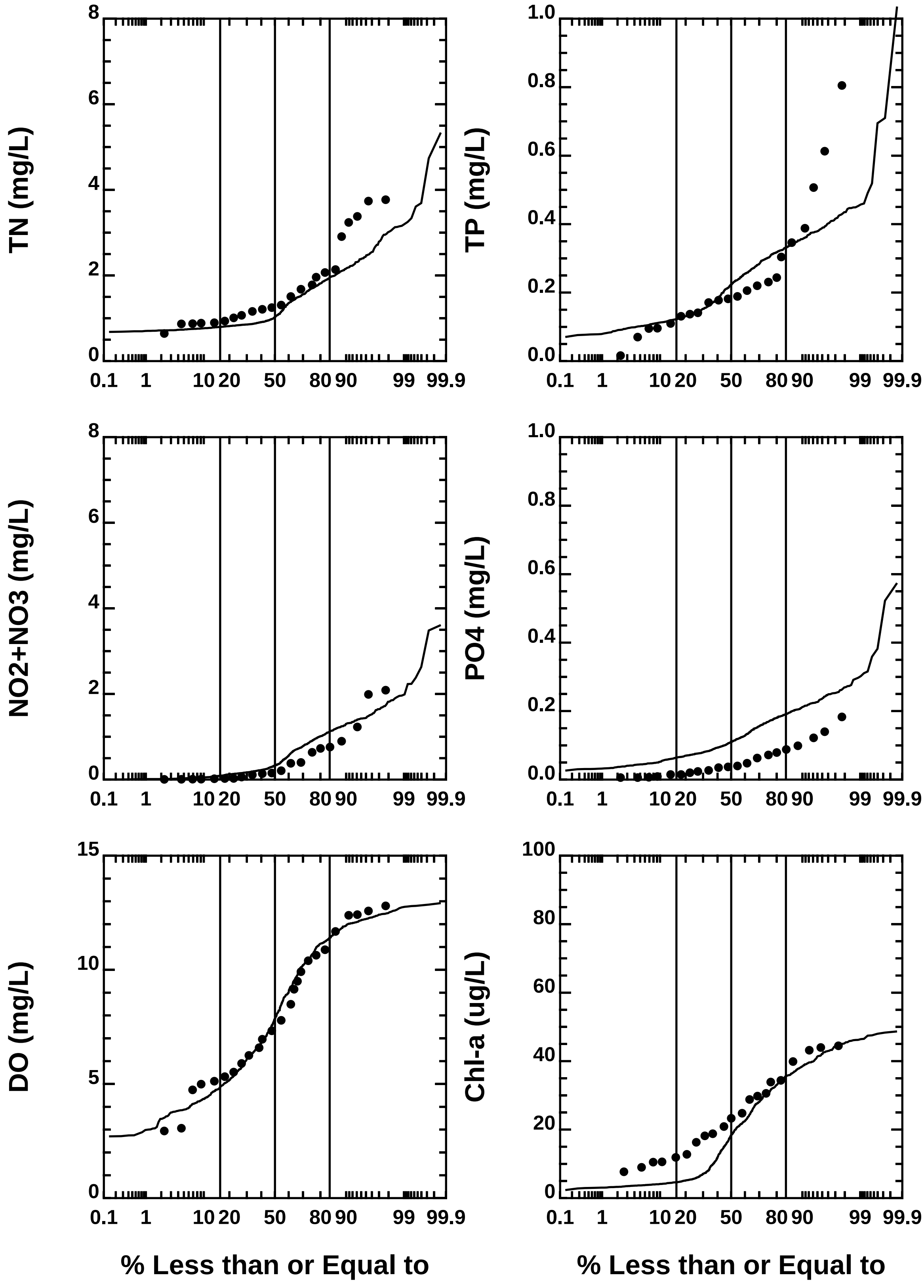

Figure 63. Water Quality Model-Data Probability Distributions
(Webbs Landing - 206131)

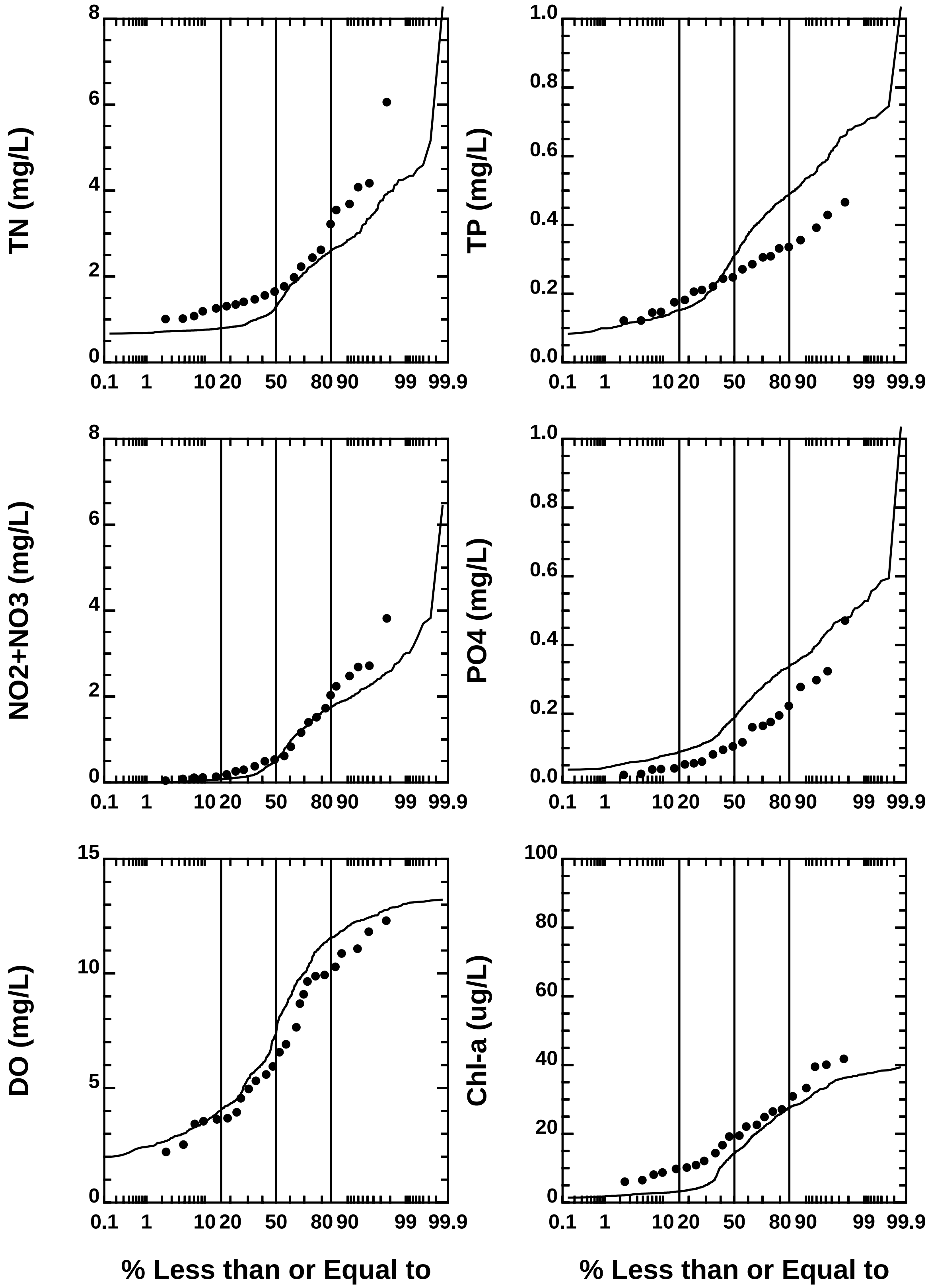

Figure 64. Water Quality Model-Data Probability Distributions
(Milford Neck Levee - 206141)

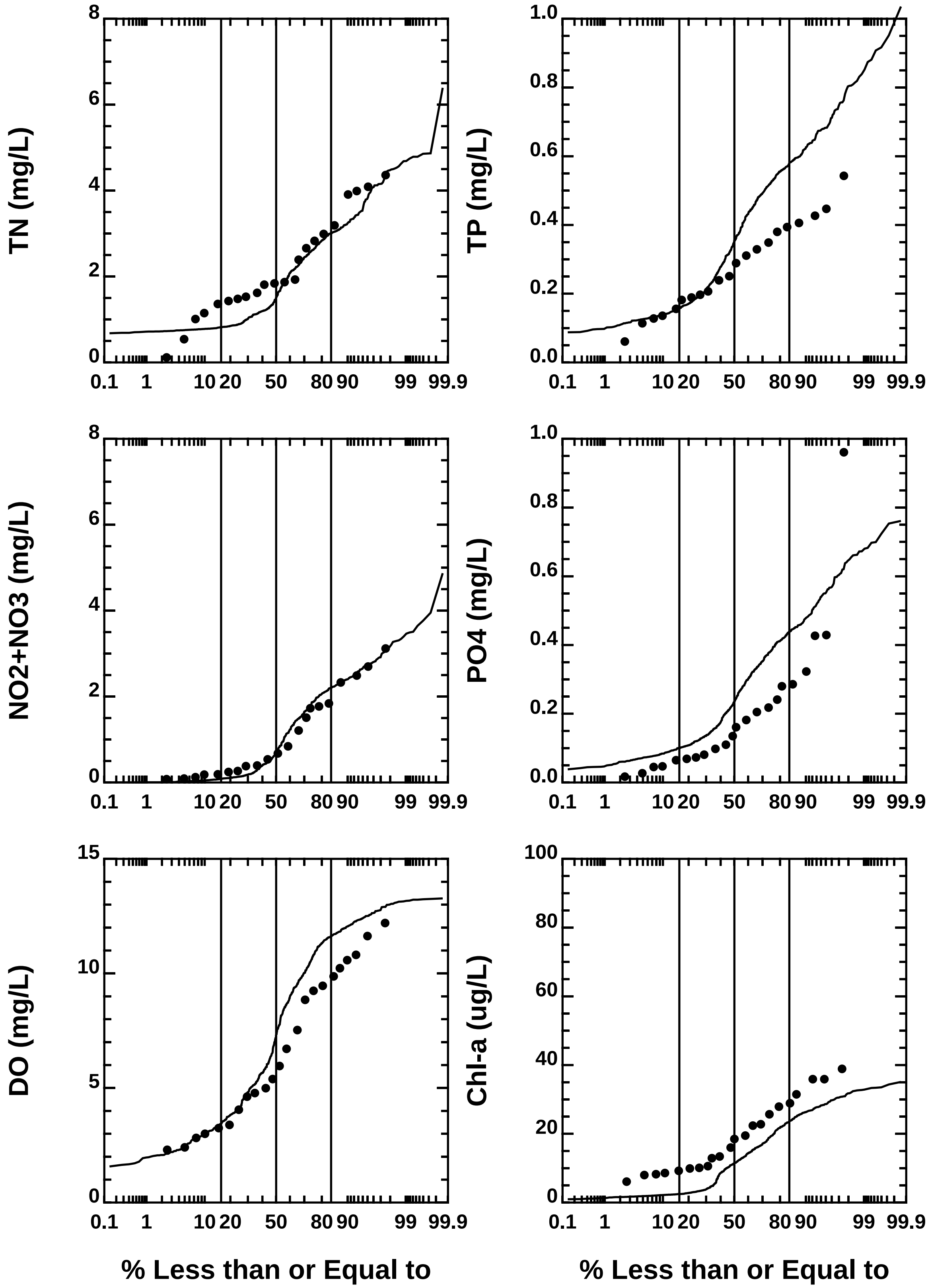

Figure 65. Water Quality Model-Data Probability Distributions
(Power Lines - 206711)

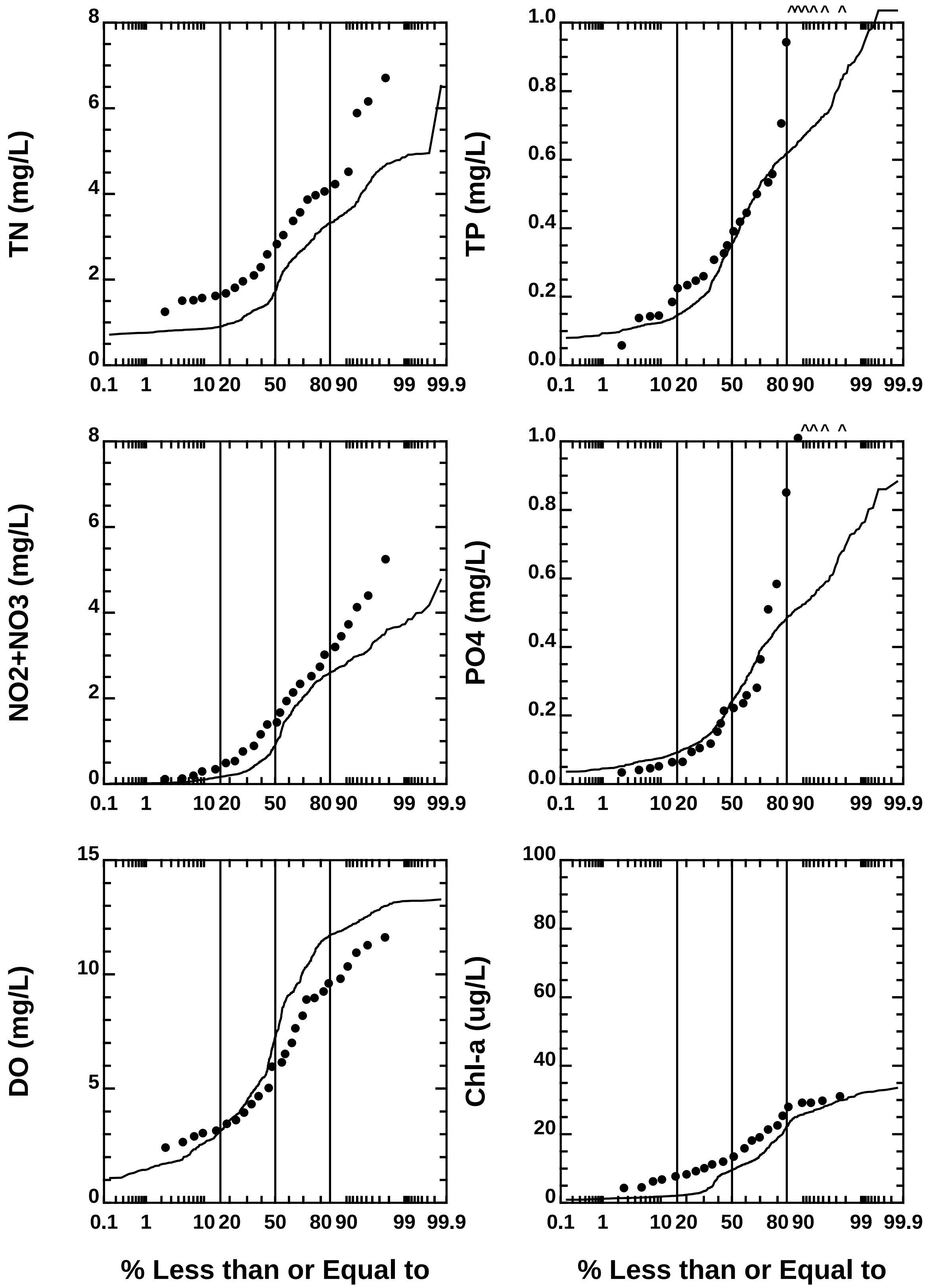

Figure 66. Water Quality Model-Data Probability Distributions
(KCRWTF Canal - 206231)

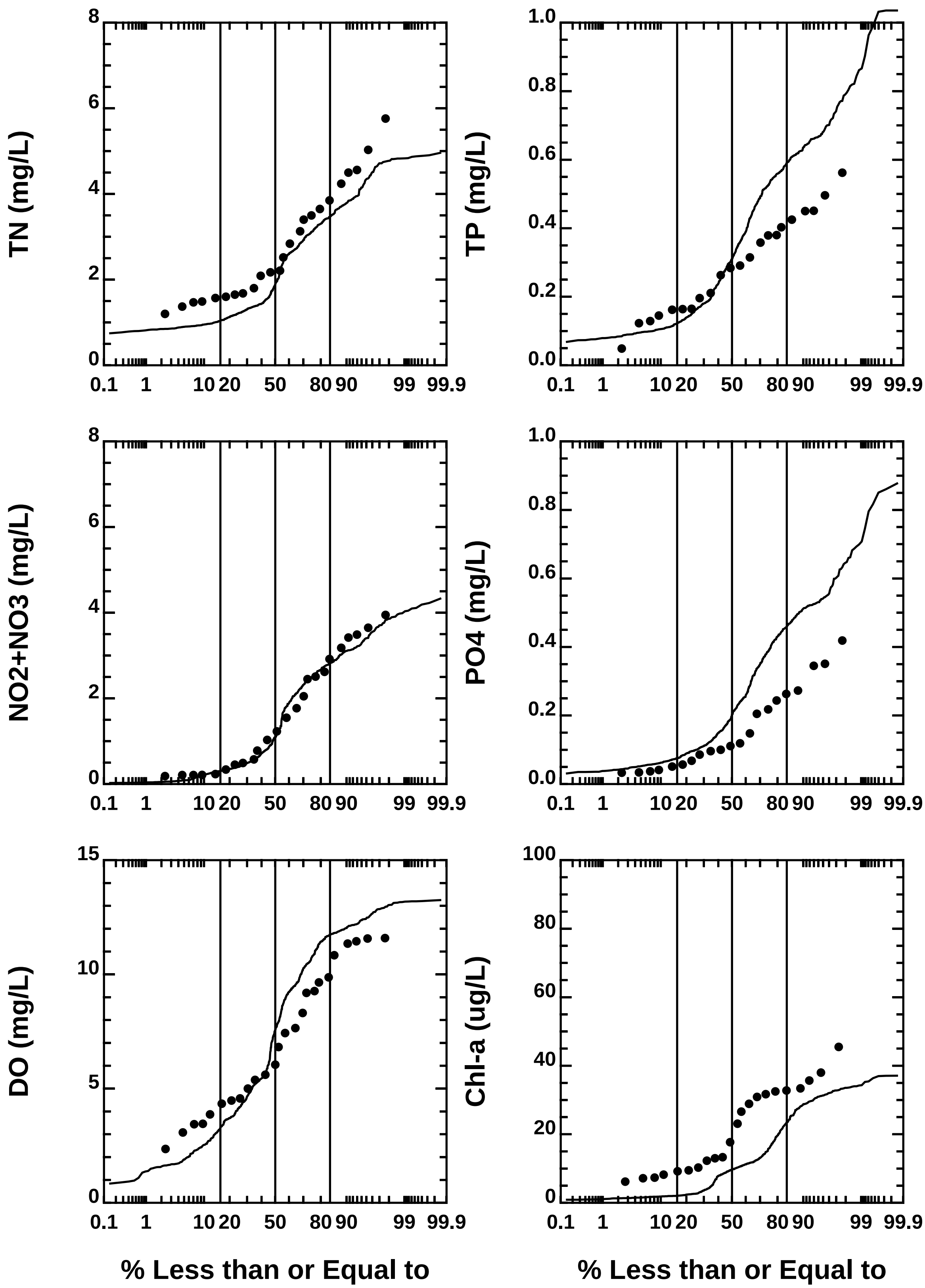

Figure 67. Water Quality Model-Data Probability Distributions
(Bay Road (Rte. 1) - 206091)

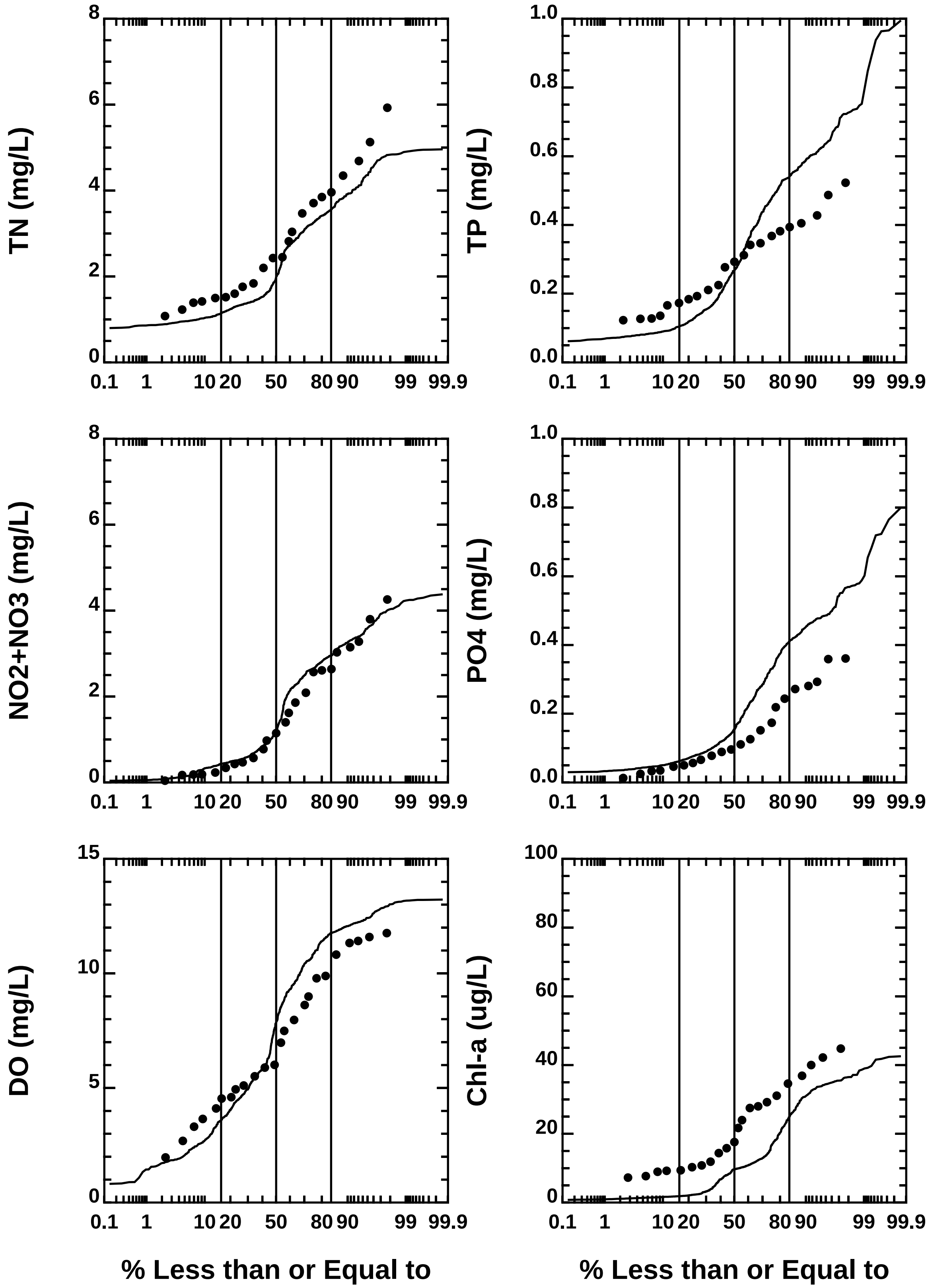

Figure 68. Water Quality Model-Data Probability Distributions
(Frederica (Rte. 12) - 206081)

SECTION 5

CONCLUSIONS

A coupled set of numerical models were developed for the Murderkill River watershed to investigate the factors influencing the low DO levels in the tidal reach of the Murderkill River, assist in setting site-specific nutrient and DO criteria, and in setting total maximum daily loads (TMDLs) in the watershed. The watershed model (HSPF) was used to represent the rainfall driven runoff processes in the watershed; the hydrodynamic model (ECOMSED) was used to represent the tidally driven circulation in the tidal river; and the eutrophication model (RCA) was used to represent the water quality interactions between nutrients, carbon or BOD, algae (phytoplankton) and DO. Development of these models was supported by an extensive monitoring program both in the watershed and tidal river along with special project-specific studies to further study important processes occurring in the watershed. These project-specific studies included:

- Water column algal primary production studies (Sharp, 2011);
- Water, salt and nutrient studies in Webb's Marsh (Wong et al., 2009; Dzwonkowski et al., 2013; Ullman et al., 2013);
- Nutrient sediment flux studies (CBA, 2010);
- Tidal marsh inundation studies (McKenna, 2013);
- LTBOD studies in the river and KCRWTF effluent;
- Continuous tidal monitoring near Frederica and at Bowers Beach by the USGS and expanded DNREC monitoring in the watershed;
- Installation of three stream gaging stations in the watershed by the USGS; and
- Vertical profiling of sediment cores (Velinsky, et al., 2010).

The data available to develop the Murderkill River models was very extensive and not typically available for model development. In this respect, the data collected and the models developed for the Murderkill River provided very valuable information on the factors influencing the interactions between nutrients and DO. Results from these studies and the modeling should be useful in other tidal river systems dominated by tidal marshes around Delaware Bay.

Based on the results from the modeling studies, the Murderkill River models (HSPF, ECOMSED and RCA) are well calibrated/validated to the observed data collected during the 2007-2008 modeling period. The models represent the important features and interactions that control the nutrient and DO dynamics in the tidal river from the upstream watershed to the extensive downstream tidal marshes and connection with Delaware Bay. Given the successful calibration/

validation of the model, they are appropriate to use in supporting site-specific nutrient and DO criteria and in developing TMDLs for the Murderkill River watershed.

SECTION 6

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