Development of Hydrodynamic and Water Quality Models for the Lynnhaven River System

Mac Sisson  
*Virginia Institute of Marine Science*

Harry Wang  
*Virginia Institute of Marine Science*

Yuepeng Li  
*Virginia Institute of Marine Science*

Jian Shen  
*Virginia Institute of Marine Science*

Albert Y. Kuo  
*Virginia Institute of Marine Science*

*See next page for additional authors*

Follow this and additional works at: https://scholarworks.wm.edu/reports

Part of the [Marine Biology Commons](https://scholarworks.wm.edu/reports)

**Recommended Citation**

Authors
Mac Sisson, Harry Wang, Yuepeng Li, Jian Shen, Albert Y. Kuo, Wenping Gong, Mark J. Brush, and Kenneth Moore

This report is available at W&M ScholarWorks: https://scholarworks.wm.edu/reports/1071
Development of Hydrodynamic and Water Quality Models for the Lynnhaven River System

Mac Sisson, Harry Wang, Yuepeng Li, Jian Shen, Albert Kuo, Wenping Gong, Mark Brush, and Ken Moore

Final Report to the
U. S. Army Corps of Engineers, Fort Norfolk Office
and
The City of Virginia Beach

Special Report No. 408
In Applied Marine Science and Ocean Engineering

Virginia Institute of Marine Science
Department of Physical Sciences
Gloucester Point, Virginia 23062

November 2010
EXECUTIVE SUMMARY
"Development of Hydrodynamic and Water Quality Models for the Lynnhaven River System"

1. The Norfolk District of the US Army Corps of Engineers and the City of Virginia Beach are working together on a cost-shared basis for the feasibility study of the Lynnhaven River environmental restoration. In January 2005, these agencies contracted with the Virginia Institute of Marine Science (VIMS) for the development of hydrodynamic and water quality models for the Lynnhaven River System.

2. VIMS has performed a successful development of an integrated numerical modeling framework for the Lynnhaven River. This framework combines a high-resolution 3D hydrodynamic model (UNTRIM) that provides the required transport for a water quality model (CE-QUAL-ICM) that, in turn, provides intra-tidal predictions of 23 water quality state variables.

3. Prior to the inception of the project, all available historical Lynnhaven hydrodynamic and water quality data were amassed in a MicroSoft ACCESS database and analyzed for model calibration suitability and long-term trends. These data were collected from monitoring programs of the Virginia Department of Environmental Quality (VA-DEQ) and the Virginia Health Department, Shellfish Sanitation Division (VA-DSS), intensive surveys conducted by VIMS and Malcolm Pirnie Environmental Engineers, and tidal surveys conducted by the National Oceanic and Atmospheric Administration (NOAA).

4. A strategy of project-specific field surveys and laboratory experiments was devised based on which measurements would complement the existing historical data and be most useful to the model calibration and validation processes. These field surveys included the following:
   - a hydrodynamic survey of synoptic measurements of times series of surface elevations plus currents and salinities in all Lynnhaven branches and outside the Inlet
   - seasonal sediment flux measurements at the Inlet and in all branches to determine the spatial and seasonal variations of the fluxes from the water column to the sediment (and vice versa) of dissolved oxygen, ammonia, nitrate-nitrite, and phosphate
   - sediment flux measurements of dissolved oxygen, ammonia, nitrate and nitrite, and phosphate in the laboratory under controlled environments
   - critical shear stress measurements at multiple sites in the basin to determine the spatial and seasonal variations to the erodibility of bottom sediments
   - high-frequency time series measurements of chlorophyll-a, turbidity, Colored Dissolved Organic Matter (CDOM), and dissolved oxygen (DO) to evaluate water quality conditions with high temporal resolution
5. The hydrodynamic model was calibrated using historical datasets and NOAA tide predictions. The water quality model was calibrated using the 2006 dataset collected by the VA-DEQ.

(a) Calibration of the hydrodynamic model

Calibration of the hydrodynamic model for tides was performed by comparing model results with synoptic measurements at 5 locations spanning from Long Creek to Broad Bay to Linkhorn Bay, as well as by comparing the NOAA predicted tide ranges and phases to model results at two Western Branch stations (Bayville Creek and Buchanan Creek) and one Eastern Branch location (Brown Cove). Calibration for velocity was made by comparing model predictions with high-frequency measurements made in 2003 at two locations bounding Long Creek. Calibrations for both temperature and salinity were made throughout 2006 by comparing model predictions with observations made at the 16 Lynnhaven VA-DEQ stations monitored every other month.

(b) Calibration of the water quality model

Calibration of the water quality model was performed for 2006 by comparing model predictions with measurements taken every other month at the 16 Lynnhaven DEQ stations for the parameters of dissolved oxygen (DO), chlorophyll-a (chl-a), total Kjeldahl nitrogen (TKN), total phosphorus (TP), ammonium (NH4), nitrate-nitrite (NO3), and ortho phosphorus (PO4).

6. Validation of the hydrodynamic model was made by comparing the 2005 simulation results with observations collected in VIMS hydrodynamic surveys of that year. Validation of the water quality model used the two-year period 2004-2005 as the period of validation. No adjustments to the values of calibration parameters, which were set in the calibration process, were made in the validation process.

(a) Validation of the hydrodynamic model

Validation for water surface elevations was made by making a 30-day, high-frequency comparison of model predictions to observations at the Virginia Pilot’s Station just inside the Inlet and a 16-day, high-frequency comparison of predictions to observations at West Neck Creek, Upper Eastern Branch. Validation of velocities was made by comparing model predictions to 30-day measurements of velocity at representative locations in each branch as well as at surface, middle, and bottom layers of a station in the channel just outside of the Inlet. Validations for both temperature and salinity were made throughout 2004-2005 by comparing model predictions with observations made at the 16 Lynnhaven VA-DEQ stations monitored every other month.

(b) Validation of the water quality model

Validation of the water quality model was performed for 2004-2005 by comparing model predictions with measurements taken every other month at the 16 Lynnhaven VA-DEQ stations for the water quality variables of dissolved oxygen (DO), chlorophyll-a (chl-a), total
Kjeldahl nitrogen (TKN), total phosphorus (TP), ammonium (NH₄), nitrate-nitrite (NO₃), and ortho phosphorus (PO₄).

7. A sediment transport model utilizing the equilibrium critical shear stress defined at the interface between layers was incorporated into the modeling framework. This model was calibrated by comparing its predictions of total suspended solids (TSS) with observations at the 16 Lynnhaven DEQ stations during 2006 and validated by comparing the 2004-2005 model results with DEQ observations for those years. Additionally, the validation compared model predictions with TSS values derived from VIMS high-frequency measurements of turbidity at 3 locations in 2005.

8. The major findings of the study included degraded water clarity due to significant concentrations of suspended sediment and localized summertime dissolved oxygen problems in headland areas. VIMS is attempting to assess the impacts that these conditions have on the restoration effort by conducting sensitivity tests of the model to reductions in the sediment and nutrient loadings associated with these conditions.

9. The entire modeling framework has been calibrated and validated and has been prepared for its application in conducting scenario runs. The models thus become a management tool for environmental assessments of the effects of variations in nutrient and sediment loadings, and other mitigation practices, in the Lynnhaven River system.
# TABLE OF CONTENTS

EXECUTIVE SUMMARY FOR REPORT .................................................................................... i
TABLE OF CONTENTS .............................................................................................................. iv
LIST OF TABLES ......................................................................................................................... vi
LIST OF FIGURES ..................................................................................................................... viii

I. BACKGROUND .........................................................................................................................1

II. INTRODUCTION ......................................................................................................................4

III. NUMERICAL MODELING METHODOLOGY ....................................................................9

   III-1. Description of the numerical modeling framework.................................................9
   III-2. The UnTRIM hydrodynamic model ........................................................................10
      III-2-1. Description of UnTRIM ........................................................................... 10
      III-2-2. Formulation of UnTRIM governing equations ........................................ 12
   III-3. The CE-QUAL-ICM water quality model ........................................................... 14
      III-3-1. Linkage between UnTRIM and CE-QUAL-ICM .................................... 15
      III-3-2. Dissolved oxygen process ........................................................................ 17
      III-3-3. Model phytoplankton kinetics .................................................................. 20
      III-3-4. Benthic sediment process ........................................................................ 27
   III-4. Description of sediment transport model ............................................................... 28
   III-5. Description of watershed model for the Lynnhaven River Basin ......................... 31

IV. HISTORICAL DATA AND FIELD OBSERVATION PROGRAM ....................................34

   IV-1. Historical data .......................................................................................................... 34
   IV-2. Project-specific field measurements........................................................................ 38
      IV-2-1. VIMS hydrodynamic survey ................................................................. 38
      IV-2-2. Seasonal sediment flux measurements ..................................................... 43
      IV-2-3. Critical shear stress measurements ......................................................... 53
      IV-2-4. VIMS dataflow surveys ........................................................................... 57
      IV-2-5. VIMS high-frequency time series measurements .................................... 68

V. MODEL CALIBRATION ...................................................................................................... 81

   V-1. Calibration of the hydrodynamic model .................................................................... 81
      V-1-1. Boundary conditions .................................................................................. 81
      V-1-2. External loading ....................................................................................... 83
      V-1-3. Calibration for tidal elevation ...................................................................... 83
      V-1-4. Calibration for velocity ............................................................................. 87
      V-1-5. Calibration for salinity .............................................................................. 89
      V-1-6. Calibration for temperature ....................................................................... 92
   V-2. Calibration of the water quality model .................................................................... 94
      V-2-1. Boundary condition .................................................................................. 94
      V-2-2. External loading ....................................................................................... 94
V-2-3. Initial condition ........................................................................................................95
V-2-4. Estimation of parameters .........................................................................................95
V-2-5. Model calibration results .......................................................................................105
V-3. Calibration of the sediment transport model ...............................................................123

VI. MODEL VALIDATION .....................................................................................................126

VI-1. Validation of the hydrodynamic model .................................................................126
  VI-1-1. Validation for tidal elevation ..............................................................................127
  VI-1-2. Validation for velocity .......................................................................................128
  VI-1-3. Validation for salinity .........................................................................................133
  VI-1-4. Validation for temperature ...............................................................................133
VI-2. Validation of the water quality model .......................................................................140
  VI-2-1. Model validation results ....................................................................................140
VI-3. Validation of the sediment transport model ...............................................................169

VII. SENSITIVITY ANALYSIS ON BENTHIC MICROALGAE DYNAMICS .......................174

VII-1. Benthic Microalgae Model Formulation ................................................................174
  VII-1-1. Modeling biomass of BMA ..............................................................................174
VII-2. Nutrient Budgets in the Lynnhaven River .............................................................179
  VII-2-1. Annual nutrient budgets in the Lynnhaven River .............................................179
VII-3. Comparison of Nutrient Budgets between Shallow and Deep Water Systems ......189

VIII. DISCUSSION AND CONCLUSIONS .........................................................................192

IX. REFERENCES ...............................................................................................................195
LIST OF TABLES

Table III.1. Impervious percentages of Lynnhaven Basin landuse categories .........................33
Table IV.1. Lynnhaven monitoring and survey data collected, by parameter and agency ..........34
Table IV.2. Lynnhaven DEQ monitoring long-term trends ...........................................................37
Table IV.3. Precision and accuracy of YSI Data (model 6600) .......................................................58
Table IV.4. High frequency time series sensor deployment locations (navigational markers), dates (excluding gaps), and parameters ..........................................................70
Table IV.5. Coordinates of sensor locations for high frequency time series measurements ......71
Table IV.6. Multiple linear regression models for predicting light attenuation as a function of water quality parameters .........................................................................................78
Table V.1. UnTRIM Modeled Tide Predictions versus Tide Table Predictions in Lynnhaven River Eastern and Western Branches ................................................................................87
Table V.2. Model state variables in the eutrophication water quality model .....................96
Table V.3. Model state variables and fluxes in the benthic sediment flux model .......................96
Table V.4. Parameters related to algae in the water column ......................................................97
Table V.5. Parameters related to organic carbon in the water column ......................................98
Table V.6. Parameters related to nitrogen in the water column ................................................99
Table V.7. Parameters related to phosphorus in the water column ..........................................100
Table V.8. Parameters related to silica in the water column .....................................................100
Table V.9. Parameters related to chemical oxygen demand and dissolved oxygen in the water column ......................................................................................................................101
Table V.10. Parameters used in the sediment flux model ..............................................................101
Table V.11. Water quality parameters in CBP monitoring data .....................................................105
Table V.12. Statistical summary of errors derived by comparing predicted vs. observed values of DO, chl-a, TKN, and TP for all 16 Lynnhaven DEQ stations for year 2006 .................122
Table V.13. Statistical summary of errors derived by comparing predicted vs. observed values of NH₄, NOₓ, and DIP for all 16 Lynnhaven DEQ stations for year 2006 ........................122
Table VI.1. Statistical summary of errors derived by comparing predicted vs. observed values of DO, chl-a, TKN, and TP for all 16 Lynnhaven DEQ stations for years 2004-2005 ....166

Table VI.2. Statistical summary of errors derived by comparing predicted vs. observed values of NH₄, NOₓ, and DIP for all 16 Lynnhaven DEQ stations for years 2004-2005 ..........167
LIST OF FIGURES

Figure I.1. Location of the Lynnhaven River in the Chesapeake Bay .............................................1
Figure II.1. Physical features of the Lynnhaven River system ..........................................................4
Figure III.1. The integrated modeling approach used for the VIMS water quality model ..........9
Figure III.2. An example of an orthogonal grid .............................................................................15
Figure III.3. Sand percentage of the bottom sediment of the Lynnhaven River .........................28
Figure III.4. Average bottom shear stress obtained by one month of hydrodynamic simulation .30
Figure III.5. The 1079 catchment areas delineated by the URS watershed model superimposed on the UnTRIM model grid ....................................................................................32
Figure IV.1. Long-term average salinity based on Lynnhaven DEQ observations .......................35
Figure IV.2. Long-term average total phosphorus based on Lynnhaven DEQ observations ....36
Figure IV.3a. Long-term trend of observed dissolved oxygen at DEQ Station THA000.76 ......36
Figure IV.3b. Long-term trend of observed dissolved oxygen at DEQ Station BBY002.88 .....37
Figure IV.4. Instrument Locations for VIMS Hydrodynamic Survey ...........................................39
Figure IV.5. Tide at Inlet versus CBBT tide ..................................................................................39
Figure IV.6. ADP velocity outside Inlet ........................................................................................40
Figure IV.7. Western Branch velocity and salinity .........................................................................40
Figure IV.8. Eastern Branch velocity ............................................................................................41
Figure IV.9. Broad Bay velocity and salinity ...................................................................................41
Figure IV.10. Western Branch velocity and temperature ..............................................................42
Figure IV.11. Broad Bay velocity and temperature ........................................................................42
Figure IV.12. Location of core collection sites for sediment flux in the Lynnhaven River ....44
Figure IV.13. Experimental design for sediment flux experiments .............................................46
Figure IV.14. Chlorophyll-α concentrations measured in the top 1 and 3 cm of sediment at each site .........................................................................................................................................47
Figure IV.15. Typical time course for DO incubated in the dark and light.................................48

Figure IV.16. Net sediment-water fluxes of dissolved oxygen by site and date .........................49

Figure IV.17. Relationship of net sediment-water DO fluxes to water temperature in the dark (left) and sediment chlorophyll in the light (right) .................................................................49

Figure IV.18. As for Figure IV.16, but for fluxes of NH$_4^+$..........................................................50

Figure IV.19. As for Figure IV.16, but for fluxes of NO$_x^-$ (NO$_2^-$ + NO$_3^-$). ..........................50

Figure IV.20. As for Figure IV.16, but for fluxes of PO$_4^{3-}$..........................................................51

Figure IV.21. Relationship of net sediment-water nutrient and oxygen fluxes in the dark...........52

Figure IV.22. Relationship of computed BMA nutrient demand in the light vs. computed uptake in the light...........................................................................................................................52

Figure IV.23. Locations for 19 samples characterized for grain size prior to critical shear stress surveys ...........................................................................................................................................53

Figure IV.24. Percentage distributions of sand, silt, and clay for 19 sediment samples ..........54

Figure IV.25. Locations of erodibility core sites for all 3 critical shear stress surveys ..........55

Figure IV.26. Critical stress profiles for all twenty-four cores that were run from the three field erosion studies. X-axis is critical shear stress in Pascals, and Y-axis is eroded mass in kilograms per square meter............................................................................................................56

Figure IV.27. Lynnhaven River system DATAFLOW cruise tracks showing turbidity levels during the 5-24-05 cruise...........................................................................................................................................60

Figure IV.28. 2006 verification station YSI NTU (turbidity) vs. light attenuation (Kd) ..........61

Figure IV.29. Concentration-distance plots of turbidity along the A.) Western Branch, B.) Eastern Branch, and C.) Broad and Linkhorn Bays during July 2006.........................................................62

Figure IV.30. Spatially averaged turbidities (NTU) for the individual branch cruise track reaches for each monthly DATAFLOW cruise in 2006 ..............................................................................................63

Figure IV.31. 2005-2006 verification station YSI chlorophyll vs. extracted chlorophyll.........63

Figure IV.32. Concentration-distance plots of chlorophyll along the A.) Western Branch, B.) Eastern Branch, and C.) Broad and Linkhorn Bays during July 2006.........................................................65

Figure IV.33. Spatially averaged chlorophyll concentrations for the individual branch DATAFLOW cruise track reaches for each monthly cruise in 2006..........................................................66
Figure IV.34. Concentration-distance plots of dissolved oxygen along the A.) Western Branch, B.) Eastern Branch, and C.) Broad and Linkhorn Bays during July 2006

Figure IV.35. Spatially averaged surface dissolved oxygen concentrations for the individual branch DATAFLOW cruise track reaches for each monthly cruise in 2006

Figure IV.36. Locations of time series sensors

Figure IV.37. Sample calibration plot relating sensor output to measured water quality, in this case chlorophyll-a

Figure IV.38. Time series measurements from 2005 in Broad Bay

Figure IV.39. Time series of chlorophyll-a collected at shore-based sites by Lynnhaven River Now in 2005 compared to in situ fluorometer time series deployed mid-channel at navigational markers

Figure IV.40. Time series measurements of surface chlorophyll-a in 2006

Figure IV.41. Time series measurements of bottom water quality

Figure IV.42. Daily average values from the 2005-06 time series sensors plotted with daily irradiance from the Chesapeake Bay Virginia National Estuarine Research Reserve site on the York River (photosynthetically active radiation, PAR), average daily wind speed and total daily precipitation at the Norfolk International Airport (obtained from the NOAA National Climatic Data Center), and daily tide range at the NOAA CBBT tide station

Figure IV.43. Relationship between measured attenuation coefficient for light ($k_D$) and (a) chlorophyll-a, (b) turbidity, and (c) CDOM, and (d) confirmation of a multiple regression-based model for predicting $k_D$ as a function of these parameters

Figure IV.44. Calculation of potential SAV habitat in Broad Bay from (a) bathymetry and in situ time series sensors (red point). (b) Area of Broad Bay receiving greater than 20% of incident irradiance on average (white). (c) Long term average SAV cover in Broad Bay, 1992-2003, based on VIMS SAV monitoring program data

Figure IV.45. Experimental setup (light gradient box) for P-I measurements in 2007-08 and a typical result (blue circles) with a statistically-fit regression (red line)

Figure V.1 Locations of boundary condition specifications for Lynnhaven River models

Figure V.2. Correlation of CBBT wind speed with Creeds, VA surface elevation

Figure V.3. Constructed series of 2005 surface elevations used for upstream boundary

Figure V.4. Locations of NOAA tide stations monitored in the Lynnhaven in the late 1970s

Figure V.5. Comparison of modeled and measured $M_2$ amplitudes and phases in the Broad Bay/Linkhorn Bay Branch of the Lynnhaven
Figure V.6. Real-time comparisons of UnTRIM predictions and NOAA water surface observations ..........................................................86

Figure V.7. Locations of Lynnhaven Velocity ADCP Stations, October 2003.................................87

Figure V.8. East-west and north-south components of measured versus modeled velocity at Station V1 of Long Creek, Lynnhaven ............................................................88

Figure V.9. East-west and north-south components of measured versus modeled velocity at Station V2 of Long Creek, Lynnhaven ........................................................................88

Figure V.10. Locations of Lynnhaven DEQ stations used to compare measured and modeled salinity, temperature, and water quality parameters ................................................89

Figure V.11. UnTRIM modeled versus measured salinities at Western Branch DEQ stations for 2006 .............................................................................................................90

Figure V.12. UnTRIM modeled versus measured salinities at Eastern Branch DEQ stations for 2006 ...............................................................................................................91

Figure V.13. UnTRIM modeled versus measured salinities at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006 ................................................................................91

Figure V.14. UnTRIM modeled versus measured temperatures at Western Branch DEQ stations for 2006 .........................................................................................................92

Figure V.15. UnTRIM modeled versus measured temperatures at Eastern Branch DEQ stations for 2006 .........................................................................................................93

Figure V.16. UnTRIM modeled versus measured temperatures at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006 .................................................................93

Figure V.17. Locations of CBP Stations CB8.1 and CB8.1E to the northeast and northwest of Lynnhaven River model domain ...........................................................................95

Figure V.18. Grouping of Lynnhaven DEQ stations by branch as used in displaying CE-QUAL-ICM water quality model calibration results .........................................................106

Figure V.19. Predicted vs. observed dissolved oxygen at Western Branch DEQ stations for 2006 ......................................................................................................................107

Figure V.20. Predicted vs. observed chlorophyll-a at Western Branch DEQ stations for 2006 ..108

Figure V.21. Predicted vs. observed TKN at Western Branch DEQ stations for 2006 ..........108

Figure V.22. Predicted vs. observed ammonium at Western Branch DEQ stations for 2006 .....109

Figure V.23. Predicted vs. observed nitrate-nitrite at Western Branch DEQ stations for 2006 ..109
Figure V.24. Predicted vs. observed total phosphorus at Western Branch DEQ stations for 2006..............................................................................................................................................110
Figure V.25. Predicted vs. observed ortho phosphorus at Western Branch DEQ stations for 2006..............................................................................................................................................110
Figure V.26. Predicted vs. observed dissolved oxygen at Eastern Branch DEQ stations for 2006..............................................................................................................................................112
Figure V.27. Predicted vs. observed chlorophyll-a at Eastern Branch DEQ stations for 2006 ...112
Figure V.28. Predicted vs. observed TKN at Eastern Branch DEQ stations for 2006 .............113
Figure V.29. Predicted vs. observed ammonium at Eastern Branch DEQ stations for 2006 .....113
Figure V.30. Predicted vs. observed nitrate-nitrite at Eastern Branch DEQ stations for 2006 ...114
Figure V.31. Predicted vs. observed total phosphorus at Eastern Branch DEQ stations for 2006..............................................................................................................................................114
Figure V.32. Predicted vs. observed ortho phosphorus at Eastern Branch DEQ stations for 2006..............................................................................................................................................115
Figure V.33. Predicted vs. observed dissolved oxygen at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006 .................................................................................................................116
Figure V.34. Predicted vs. observed chlorophyll-a at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006..............................................................................................................................................116
Figure V.35. Predicted vs. observed TKN at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006..............................................................................................................................................117
Figure V.36. Predicted vs. observed ammonium at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006..............................................................................................................................................117
Figure V.37. Predicted vs. observed nitrate-nitrite at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006..............................................................................................................................................118
Figure V.38. Predicted vs. observed total phosphorus at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006 .................................................................................................................118
Figure V.39. Predicted vs. observed ortho phosphorus at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006 .................................................................................................................119
Figure V.40. Plots of 1:1 predicted vs. observed DO, chl-a, TKN, and TP at all 16 Lynnhaven DEQ stations for 2006 ..............................................................................................................................................121
Figure V.41. Plots of 1:1 predicted vs. observed NH₄, NOₓ, and DIP at all 16 Lynnhaven DEQ stations for 2006

Figure V.42. Predicted vs. observed TSS at Western Branch DEQ stations for 2006

Figure V.43. Predicted vs. observed TSS at Eastern Branch DEQ stations for 2006

Figure V.44. Predicted vs. observed TSS at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006

Figure VI.1. Locations of Lynnhaven observation stations (tide and velocity) in 2005

Figure VI.2. Modeled versus observed water elevations at the Virginia Pilot’s station (November 2005) and in West Neck Creek (September 2005)

Figure VI.3. Locations of Lynnhaven Velocity Stations, November 2005

Figure VI.4. East-west and north-south components of measured versus modeled velocity at surface, middle, and bottom layers outside Lynnhaven Inlet

Figure VI.5. Magnitude and direction of measured versus modeled velocity at mid-depth in the Western Branch

Figure VI.6. Magnitude and direction of measured versus modeled velocity at mid-depth in the Eastern Branch

Figure VI.7. Magnitude and direction of measured versus modeled velocity at mid-depth in Broad Bay

Figure VI.8. Grouping by branch of Lynnhaven DEQ stations used to compare measured and modeled salinities, temperatures, and CE-QUAL-ICM water quality parameters

Figure VI.9. UnTRIM modeled versus measured salinities at Western Branch DEQ stations for 2004

Figure VI.10. UnTRIM modeled versus measured salinities at Western Branch DEQ stations for 2005

Figure VI.11. UnTRIM modeled versus measured salinities at Eastern Branch DEQ stations for 2004

Figure VI.12. UnTRIM modeled versus measured salinities at Eastern Branch DEQ stations for 2005

Figure VI.13. UnTRIM modeled versus measured salinities at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004

Figure VI.14. UnTRIM modeled versus measured salinities at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005
Figure VI.15. UnTRIM modeled versus measured temperatures at Western Branch DEQ stations for 2004...........................................................................................................................137

Figure VI.16. UnTRIM modeled versus measured temperatures at Western Branch DEQ stations for 2005...........................................................................................................................137

Figure VI.17. UnTRIM modeled versus measured temperatures at Eastern Branch DEQ stations for 2004...........................................................................................................................138

Figure VI.18. UnTRIM modeled versus measured temperatures at Eastern Branch DEQ stations for 2005...........................................................................................................................138

Figure VI.19. UnTRIM modeled versus measured temperatures at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004.....................................................................................................139

Figure VI.20. UnTRIM modeled versus measured temperatures at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005.....................................................................................................139

Figure VI.21. Predicted vs. observed dissolved oxygen at Western Branch DEQ stations for 2004..............................................................................................................................................142

Figure VI.22. Predicted vs. observed dissolved oxygen at Western Branch DEQ stations for 2005..............................................................................................................................................142

Figure VI.23. Predicted vs. observed chlorophyll-a at Western Branch DEQ stations for 2004 143

Figure VI.24. Predicted vs. observed chlorophyll-a at Western Branch DEQ stations for 2005 143

Figure VI.25. Predicted TKN at Western Branch DEQ stations for 2004...................................144

Figure VI.26. Predicted vs. observed TKN at Western Branch DEQ stations for 2005..............144

Figure VI.27. Predicted ammonium at Western Branch DEQ stations for 2004.........................145

Figure VI.28. Predicted vs. observed ammonium at Western Branch DEQ stations for 2005....145

Figure VI.29. Predicted nitrate-nitrite at Western Branch DEQ stations for 2004......................146

Figure VI.30. Predicted vs. observed nitrate-nitrite at Western Branch DEQ stations for 2005.146

Figure VI.31. Predicted vs. observed total phosphorus at Western Branch DEQ stations for 2004..............................................................................................................................................147

Figure VI.32. Predicted vs. observed total phosphorus at Western Branch DEQ stations for 2005..............................................................................................................................................147

Figure VI.33. Predicted ortho phosphorus at Western Branch DEQ stations for 2004 ...............148
Figure VI.34. Predicted vs. observed ortho phosphorus at Western Branch DEQ stations for 2005..............................................................................................................................................148
Figure VI.35. Predicted vs. observed dissolved oxygen at Eastern Branch DEQ stations for 2004..............................................................................................................................................150
Figure VI.36. Predicted vs. observed dissolved oxygen at Eastern Branch DEQ stations for 2005..............................................................................................................................................150
Figure VI.37. Predicted vs. observed chlorophyll-a at Eastern Branch DEQ stations for 2004..151
Figure VI.38. Predicted vs. observed chlorophyll-a at Eastern Branch DEQ stations for 2005..151
Figure VI.39. Predicted TKN at Eastern Branch DEQ stations for 2004 .........................152
Figure VI.40. Predicted vs. observed TKN at Eastern Branch DEQ stations for 2005 ...............152
Figure VI.41. Predicted ammonium at Eastern Branch DEQ stations for 2004 .....................153
Figure VI.42. Predicted vs. observed ammonium at Eastern Branch DEQ stations for 2005 .....153
Figure VI.43. Predicted nitrate-nitrite at Eastern Branch DEQ stations for 2004 .....................154
Figure VI.44. Predicted vs. observed nitrate-nitrite at Eastern Branch DEQ stations for 2005 ..154
Figure VI.45. Predicted vs. observed total phosphorus at Eastern Branch DEQ stations for 2004.................................................................................................................155
Figure VI.46. Predicted vs. observed total phosphorus at Eastern Branch DEQ stations for 2005.................................................................155
Figure VI.47. Predicted ortho phosphorus at Eastern Branch DEQ stations for 2004 .............156
Figure VI.48. Predicted vs. observed ortho phosphorus at Eastern Branch DEQ stations for 2005..............................................................................................................................................156
Figure VI.49. Predicted vs. observed dissolved oxygen at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004 .................................................................................................................158
Figure VI.50. Predicted vs. observed dissolved oxygen at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005 ..............................................................................................................................................158
Figure VI.51. Predicted vs. observed chlorophyll-a at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004 ..............................................................................................................................................159
Figure VI.52. Predicted vs. observed chlorophyll-a at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005 ..............................................................................................................................................159
Figure VI.53. Predicted TKN at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004 .....160
Figure VI.54. Predicted vs. observed TKN at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005

Figure VI.55. Predicted ammonium at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004

Figure VI.56. Predicted vs. observed ammonium at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005

Figure VI.57. Predicted nitrate-nitrite at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004

Figure VI.58. Predicted vs. observed nitrate-nitrite at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005

Figure VI.59. Predicted vs. observed total phosphorus at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004

Figure VI.60. Predicted vs. observed total phosphorus at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005

Figure VI.61. Predicted ortho phosphorus at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004

Figure VI.62. Predicted vs. observed ortho phosphorus at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005

Figure VI.63. Plots of 1:1 predicted vs. observed DO, chl-a, TKN, and TP

Figure VI.64. Plots of 1:1 predicted vs. observed NH₄, NOₓ, and DIP

Figure VI.65. Station locations for high-frequency measurements of turbidity in 2005 in the Lynnhaven River system

Figure VI.66. Predicted TSS vs. TSS derived from high-frequency turbidity measurements at 3 locations in the Lynnhaven in 2005

Figure VI.67. Predicted vs. observed TSS at Western Branch DEQ stations for 2004

Figure VI.68. Predicted vs. observed TSS at Western Branch DEQ stations for 2005

Figure VI.69. Predicted vs. observed TSS at Eastern Branch DEQ stations for 2004

Figure VI.70. Predicted vs. observed TSS at Eastern Branch DEQ stations for 2005

Figure VI.71. Predicted vs. observed TSS at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004
Figure VI.72. Predicted vs. observed TSS at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005........................................................................................................................................173

Figure VII.1. Framework of benthic algae model..................................................................................................175

Figure VII.2. Annual Total Nitrogen budget (mg N m$^{-2}$ d$^{-1}$) for Lynnhaven River (values in parentheses indicate results without BMA)........................................................................................................182

Figure VII.3. Annual Total Phosphorus budget (mg P m$^{-2}$ d$^{-1}$) for Lynnhaven River (values in parentheses indicate results without BMA)........................................................................................................183

Figure VII.4. Annual Total Nitrogen budget (mg N m$^{-2}$ d$^{-1}$) in three branches of Lynnhaven River (WB: Western Branch, EB: Eastern Branch, BB: Broad Bay)..........................................................184

Figure VII.5. Annual Total Phosphorus budget (mg P m$^{-2}$ d$^{-1}$) in three branches of Lynnhaven River (WB: Western Branch, EB: Eastern Branch, BB: Broad Bay)..........................................................185

Figure VII.6. Monthly Total Nitrogen budget (mg N m$^{-2}$ d$^{-1}$) and Total Phosphorus budget (mg P m$^{-2}$ d$^{-1}$) in the water column for Lynnhaven River........................................................................................................187

Figure VII.7. Monthly Total Nitrogen budget (mg N m$^{-2}$ d$^{-1}$) and Total Phosphorus budget (mg P m$^{-2}$ d$^{-1}$) in sediment for Lynnhaven River........................................................................................................188

Figure VII.8. Monthly BMA uptake contribution to sediment flux nitrogen and phosphorus for Lynnhaven River............................................................................................................................190

Figure VII.9. The percent of total nitrogen and phosphorus input from land and atmosphere that is exported from a sample of estuaries and lakes as a function of mean residence time in the system.........................................................................................................................................191
CHAPTER I. BACKGROUND

The Lynnhaven River system, comprised of the Eastern, Western Branches, Broad Bay, and Linkhorn Bay, is a shallow-water coastal system located near the southeast corner of the Chesapeake Bay. It traverses a 64-square-mile watershed that spans most of the northern half of Virginia Beach with a land use that is 40% residential and 35% streets, commercial and office space, and military use, and it flows northerly and empties into the Chesapeake Bay about 10 miles east of Norfolk (see Figure I.1). Due to its narrow entrance and greater influence by the tide of the Bay than by river discharge, it is technically considered as a tidal inlet system. Like many Chesapeake Bay small coastal basins, the Lynnhaven River system was a highly productive ecosystem, supporting a large oyster population and various shallow water organisms. Clampitt et al. (1993) documented that 20 species of vertebrate, 39 invertebrate species, 76 plant species, and 19 types of rare natural communities of statewide significance are supported in the Lynnhaven. In the early twentieth century, Lynnhaven River was known for its abundant harvest of “oysters suitable for kings”. The Lynnhaven oyster population has since drastically diminished along with water quality degradations that include poor water clarity, recession of submerged aquatic vegetation areas, and high chlorophyll, suspended solids, and seasonally-low dissolved oxygen levels in headland regions of the branches.

Figure I.1. Location of the Lynnhaven River in the Chesapeake Bay
In May 1998, the Lynnhaven River Environmental Restoration Study was authorized by Resolution of the Committee on Transportation and Infrastructure of the U.S. House of Representatives. Congress appropriated funding in 2002 to initiate a reconnaissance analysis in support of this authority. The ensuing reconnaissance report, issued by the U.S. Army Corps of Engineers (2002), cited a number of problems in water quality deterioration, siltation, sedimentation, and habitat management in the Lynnhaven. The report stated that “the river has become increasingly stressed as the watershed has experienced a shift from a predominantly rural to a predominantly urban/suburban land use pattern”.

Over the past several decades, Lynnhaven River water quality has been degraded by increased volume and decreased quality of stormwater runoff. Non-point sources (NPS), such as storm drains, soil erosion, lawn fertilizer, street litter, estuarine sediments, animal wastes, and failing septic systems, have caused the most degradation. The reconnaissance report cites additional causes of Lynnhaven water quality degradation as including the loss of wetland buffers associated with shoreline hardening and erosion, degradation of riparian buffers near stormwater outfalls, increased siltation from land-based construction, and increased stormwater runoff due to more developments and roadways. Additional concerns regarding water quality in the Lynnhaven include water clarity and the levels of total suspended solids measured throughout the branches of the Lynnhaven as well as seasonally low dissolved oxygen and high fecal coliform levels measured in the upper Western and Eastern Branches, where the River’s flushing capacity diminishes.

Whereas decreased water quality can have severe ecological impact on both benthic and pelagic populations and species diversity, there are additional ecological impacts emerging in the Lynnhaven. These impacts affect:

1) the abundance of tidal wetlands caused by construction activities such as dredging, filling, bulkheading, and channelization,
2) the oyster resources caused by high fecal coliform levels, and
3) the submerged aquatic vegetation (SAV) habitats caused by high nutrient and sediment inputs and the ensuing poor water clarity.

Another noteworthy issue regarding environmental restoration of the Lynnhaven includes siltation in the upper reaches, which has increased over the past several decades, and which can decrease the flushing capability upstream by decreasing the tidal prism. Lastly, sediments with elevated levels of heavy metals or other toxicants, which could severely impact living resources, have been noted in several Lynnhaven reports.

In an evaluation of alternative, the U.S. Army Corps of Engineers reconnaissance report determined that the alternatives would result in net environmental benefits through ecosystem restoration, and recommended that this study continue into its next phase, a cost-shared feasibility study.
The agencies in charge of the present development efforts are the Norfolk District, U.S. Army Corps of Engineers (ACE), representing the Federal Government, and the City of Virginia Beach, acting as the Local Sponsor. These agencies signed a feasibility cost-sharing agreement and embarked on determining suitable and acceptable means for designing and implementing the environmental restoration of the Lynnhaven. During discussions with personnel from VIMS and URS Corporation of Virginia Beach, it was resolved that a fully comprehensive system, including spatially high-resolution numerical modeling and watershed loading estimation, was required in order to address the issues cited in the reconnaissance report and to provide the management option of a control strategy of attaining the required endpoints for environmental restoration.

In early 2005, the ACE (Norfolk District) and the City of Virginia Beach contracted with VIMS for the development of hydrodynamic and water quality models for the Lynnhaven River System receiving waters and with URS Corporation for the development of a watershed model to provide both freshwater flows and nutrient and sediment loadings from the Lynnhaven River Basin.
CHAPTER II. INTRODUCTION

The Lynnhaven River system is an extremely shallow waterbody with average depths of only 0.62 m, 0.75 m, and 2.16 m, respectively, is the Western, Eastern, and Broad Bay/Linkhorn Bay systems (Figure II.1). It is also characterized by a narrow Inlet opening and tidal flats, small islands, and branching shorelines in its branches. The shallow water portion of the coastal system (with water depths less than 2-3 meters) is ubiquitous along the edge of the shoreline and many coastal embayments. Its habitat supports a tremendous diversity of aquatic life, including plants, benthos, invertebrates, plankton, crabs, fish, and seabirds; in particular, it serves as the major fish spawning ground providing shelter and food sources. Therefore, the shallow water region (SWR) is a unique habitat and an integral part of the productivity of the Bay ecosystem.

The SWR is the buffer zone between aquatic and terrestrial landscapes. It has been shown that nonpoint sources of nutrient inputs, including groundwater and surface water runoff, that pass through this region contribute significantly to the overall eutrophication problem. Human activities in watersheds have caused major changes in water quality, resulting in increased loading of nutrients, organic matter, and sediment to the SWR (Fleischer, 1987; Frink, 1991; Hopkinson and Vallino, 1995). Industrial activities and agriculture generate a mixture of chemicals, including nutrients, some of which are inevitably discharged into aquatic ecosystems. As a result, the SWR, such as

![Image of the Lynnhaven River system with labeled physical features: Narrow opening at Inlet, Islands, Tidal flats, Branching shorelines.](image-url)

Figure II.1. Physical features of the Lynnhaven River system
coastal lagoons and embayments, has received large inputs of nutrients from watershed due to anthropogenic activities for many years. Therefore, the SWR is a highly productive environment. Nutrient loading usually arises from sources including: fertilizer runoff, groundwater, sewage discharges, and aquaculture (Balls, 1994). Accordingly, there are increasing interests and demands for further understanding of eutrophication processes in the SWR.

The characteristics of the SWR differ from those of deepwater regions. The water table is usually at or near the surface, and it is constantly under the influence of tide, wave, and climate changes, which leads to wetting and drying of tidal flats, larger variation of salinity and nutrients change, suspension of sediment, and runoff of nutrients released from the land. The shallowness permits wind and tide-driven mixing to occur through the water column over the entire year. In deeper estuaries, stratification may be significant due to the high bottom salinity and sediment concentration. In the SWR system, however, continuous mixing causes the salinity stratification to become almost vertically homogeneous. Meanwhile, vertically well-mixed conditions also resuspend sediment material, including the nutrients required for primary productivity, to the overlying water column. Thus, the potential for the primary productivity is increased. Shallowness also enables sunlight to penetrate to the bottom of the sediments, which creates favorable conditions for the benthic primary producers. Combining these two factors, the primary productivity usually is high in this shallow water system.

The dynamics of the SWR are very rich because of the input of mechanical energy (freshwater discharge, tide, and wind), solar radiation, and nutrients (nitrogen and phosphorus). These natural resources stimulate primary production in both the water column and the benthic zone of the SWR. In contrast to a pelagic system, the benthos of the SWR may provide an important source of nutrients because of both its shallowness and the vertical turbulence caused by wind and tidal agitation. The nutrient exchange across the sediment-water interface is an important pathway for nutrient cycles in the SWR. The evaluation of the exchange oxygen and nutrients flux is indispensable to identifying the effects of SWR estuaries or embayments (Reay et al., 1995; Sanders et al., 1997; Yin and Harrison, 2000). Therefore, benthic nutrient fluxes have long been recognized as being an important component of estuarine ecosystems due to their ability to significantly influence water quality (Nixon, 1981; Blackburn and Henriksen, 1983; Boynton and Kemp, 1985; Kemp et al., 1990; Rizzo and Christian, 1996).

Furthermore, benthic microalgae (BMA) influence several key estuarine biogeochemical processes. Through photosynthesis of BMA, the upper sediment is oxygenated. An increase in the sediment oxygenation can lead to an indirect influence on sediment biogeochemistry as anoxic microbial processes are pushed deeper (Sundbäck et al., 2000). Meanwhile, BMA also uptake nutrients to sustain their autotrophic processes (Rizzo, 1990; Rizzo et al., 1992; Sundbäck et al., 2000; Anderson et al., 2003). This has important implications for regenerated nutrients as the oxic state of sediments closely controls benthic nutrient regeneration (Boynton and Kemp, 1985; Rysgaard et al., 1994; Banta et al., 1995; Chapelle, 1995). Nutrient release from the sediments increases
dramatically during hypoxic and anoxic events (Sundby et al., 1992; Cowan and Boynton, 1996).

Overall, in shallow portions of estuaries, BMA photosynthesis and respiration are important components of the entire ecosystem. Several studies indicated that BMA production could account for up to 50% of the entire system primary production in shallow estuarine and coastal waters (van Raalte et al., 1976; Sullivan and Moncrieff, 1988; Sundbäck and Jönsson, 1988), and benthic respiration accounts for 25% of the organic matter respired in various environments (Nixon, 1981). Nutrient loading, resulting from human activities, can also have significant impacts on benthic photosynthesis and respiration. Nutrient enrichment has been demonstrated to increase BMA production and biomass in field experiments (van Raalte et al., 1976; Granéli and Sundbäck, 1985).

The lagoons and shallow water estuaries can be exploited for recreational purposes, and for economic activities such as oyster restoration, crab rearing, and fish farming. It is very difficult to forecast the behaviour of a shallow water ecosystem, a complex network of relationships between plants and animals within a given environment, because of its complexity. The trophic network of this ecosystem is based on primary production, nutrient loading, and the amount of solar free energy, which is converted into biomass by means of photosynthesis. Primary production varies in space and in time, and depends on three important factors: water temperature, solar energy, and nutrients such as nitrogen and phosphorus in the aquatic system.

At a qualitative level, the role of each of these three factors in the ecosystem is well understood and it is common knowledge that the primary production depends on the interaction between these factors. However, it is difficult to quantify how much each of these factors would affect the year-to-year biomass production, and the occurrence of an anoxic crisis caused by excessive primary production. An integrated modeling approach has been successfully applied in the Chesapeake Bay for investigating hypoxia and anoxia over the deep water region in the mainstem Bay and major tributaries (Cerco et al., 2002). The approach calls for a system of models including hydrodynamic, watershed, water quality, and sediment flux models to be setup and operated in the study domain. The hydrodynamic model results provide transport information for the water quality model. Meanwhile, results from the watershed model will provide the nutrient loadings from land. The rates of nutrient exchange between sediment and the overlying water column are calculated from the sediment flux model.

The concern about eutrophication in coastal areas has prompted a large number of field and modeling studies on the dynamics of these environments. A number of historical surveys for water quality data collection and modeling studies for the Lynnhaven River have been conducted by the Virginia Institute of Marine Science (VIMS), Virginia Department of Environmental Quality, Department of Shellfish and Sanitation, and Malcolm Pirnie Engineers, over the past three decades. Previous modeling efforts used a simplified tidally averaged hydrodynamic component. An initial water quality study of Buchanan Creek, a small tributary in the Western Branch of Lynnhaven, was done by Ho
et al. (1977a). Later, these researchers used both slack water surveys and intensive surveys to contrast the circulation in the Lynnhaven River System with that of nearby Little Creek Harbor (Ho et al., 1977b). Malcolm Pirnie Engineers (1980), in a report to the Norfolk District Army Corps of Engineers, described the conditions of Lynnhaven at that time, citing the expected problems as the watershed was further “built-out”. In response, Kuo et al. (1982) applied the inter-tidal tidal prism model to study the effects of stormwater impacts on the water quality of the Lynnhaven. Later, Park et al. (1995a; 1995b), in work for the Virginia Department of Environmental Quality’s (DEQ’s) Coastal Resources Management Program, analyzed numerous surveys from 1980 and 1994 and further refined the tidal prism model.

Early models of sediment-water nutrients fluxes were based on net heterotrophic sediments and showed fluxes as primarily net nutrient sources to the water column (DiToro and Fitzpatrick, 1993). Flux measurements were also commonly made in the dark since there was no light available at the sediment surface, and benthic metabolism was driven by the heterotrophic breakdown of particulate organic matter derived from the water column (Davies, 1975). Recently, the importance of productivity by BMA in euphotic sediments was demonstrated (Colijn and de Jonge, 1984; Rizzo and Wetzel, 1985; Sundbäck, 1986), and autotrophic benthic production was shown to have direct and indirect impacts on benthic nutrient fluxes (Andersen et al., 1984; Sundbäck and Granéli, 1988; Anderson et al., 2003; Tyler et al., 2003). These included the direct assimilation of nutrients by benthic primary producers, as well as influencing microbial metabolism through modification of sediment biogeochemistry, for example, oxygen penetration (Revsbech et al., 1980; Rueter et al., 1986; Lorenzen et al., 1998). Therefore, several mathematical models were developed that vertically integrated the effects of oxygen penetration on benthic microbial processes (Christensen et al., 1989; 1990; Blackburn, 1990).

There are many challenges to modeling efforts in shallow water regions, in general, but particularly for the Lynnhaven River for several reasons:

1) the narrow opening at the Inlet
2) extensive tidal flats just inside the Inlet
3) 150 miles of meandering shorelines throughout the Lynnhaven
4) islands within this system.

These factors primarily affect the hydrodynamic modeling efforts. A key modeling challenge for any water quality application is the determination of whether all the vital mechanisms are accounted for in the selection of state variables in the model formulation. The pioneering work done by Li (2006) has demonstrated quantitatively the important role played by BMA for the shallow-water Lynnhaven River system.

With the given basin geometry, initial condition, and loading information from the surrounding watershed as the boundary conditions, the model framework solves the mathematical equations governing the processes. The results are then calibrated and verified with the observation data. When properly tuned, the modeling framework
renders a holistic view of the system functions, can assess ‘what-if’ scenarios, and provides tremendous predictive capability to aid management decisions and scientific research. In a similar vein, there is an excellent opportunity to make use of the integrated modeling approach to study the shallow water processes in the coastal basins. The timing is particularly appropriate, given the new shallow water monitoring technologies with high spatial and temporal coverage that are emerging (http://mddnr.chesapeakebay.net/eyesonthebay/index.cfm). This study attempts to address these difficulties by performing an integrated modeling approach, which mimics the main features of the shallow estuary. With this model, it is possible to capture the main dynamic features of the systems at a reasonable computational cost.

In order to explore these dynamics, a BMA model has been developed and uniquely coupled to the water column model that provides an otherwise comprehensive description of physical processes and both the benthic and pelagic marine trophic systems. For the water column, the well-tested CE-QUAL-ICM model was used. The relative complexity of CE-QUAL-ICM allows consideration of the full range of potential influences that BMA may have on the marine ecosystem. More recently, a robust finite difference/finite volume model for three-dimensional flows, UnTRIM (Unstructured Tidal Residual Intertidal Mudflat), has been formulated and tested on an unstructured orthogonal grid (Casulli and Zanolli, 1998; Casulli and Walters, 2000). UnTRIM, which uses an unstructured grid to better resolve complicated coastlines in the shallow environment, was further developed using the finite volume method calculation to ensure conservation of mass for all the physical and chemical constituents. UnTRIM provides hydrodynamic information that is needed by the water quality model, such as surface water elevation, three-dimensional velocity field, vertical eddy diffusivity, and so on.

An introduction has herein been presented in Chapter II. Chapter III provides a description of the methodology utilized during the project, from the overall numerical modeling framework to the individual interactive models. Chapter IV describes field observation data, both historical data and project-specific field measurements. The calibrations of the hydrodynamic and water quality models are presented in Chapter V and their validations are presented in Chapter VI. Chapter VII describes a sensitivity analysis on benthic microalgae dynamics. Lastly, Chapter VIII provides a discussion and conclusions.
CHAPTER III. NUMERICAL MODELING METHODOLOGY

III-1. Description of Numerical Modeling Framework

Numerical modeling, in a broad sense, is a process of building a mathematical abstraction of an actual system. In the estuarine and coastal environmental context, the system consists of physical, chemical, and biological components that are interactive and feed back on one another. The VIMS numerical modeling framework, as shown in Figure III.1, involves an integrated approach that combines several different processes such as hydrodynamic, water quality, nutrient, sediment processes in order to fully address the environmental impact. Whereas the CE-QUAL-ICM water quality model is shown to be the central processing mechanism, it depends heavily upon the other models with which it interacts:

1) the UnTRIM hydrodynamic model for mass and volume transport,
2) the HSPF watershed model for freshwater discharge and nutrient loadings, and
3) the sediment model for sediment flux information.

Figure III.1. The integrated modeling approach used for the VIMS water quality model
III-2. The UnTRIM hydrodynamic model

The hydrodynamic model selected for use in the numerical modeling framework is vital in that it provides the transport information required by the water quality model. The VIMS selection of UnTRIM as the hydrodynamic model for this project was based on several key features that make UnTRIM ideally suited for application to the Lynnhaven:

1) UnTRIM’s use of an unstructured grid allows for a better fit of the meandering shorelines of the Lynnhaven branches
2) UnTRIM’s efficient wetting-and-drying algorithm affords accurate representation of the intra-tidal areas in the system
3) UnTRIM’s finite volume representation has the quality of conserving mass locally as well as globally
4) UnTRIM’s independence from the Courant-Friedrich-Levy (CFL) stability criterion allows for the use of a comparatively long timestep for calculations (several minutes) despite maintaining high spatial resolutions on the order of 10 meters

III-2-1. Description of UnTRIM

The hydrodynamic model UnTRIM (Unstructured Tidal, Residual, and Intertidal Mudflat) was developed by Professor Vincenzo Casulli (Trento University, Italy). UnTRIM is a semi-implicit finite difference (-volume) model based on the three-dimensional shallow water equations as well as on the three-dimensional transport equation for salt, heat, dissolved matter and suspended sediments. UnTRIM is governed by the equations of motion, the equation of continuity, and the transport equation.

UnTRIM is able to work on unstructured orthogonal grids (UOG). The modeling domain is covered by a grid consisting of a set of non-overlapping convex polygons, usually either triangles or quadrilaterals. The grid is said to be an unstructured orthogonal grid if within each polygon a point (hereafter called a center) can be identified in such a way that the segment joining the center of two adjacent polygons and the side shared by the two polygons, have a non-empty intersection and are orthogonal to each other (Casulli and Zanolli, 1998).

UnTRIM has been widely used (Li, 2002; Li et al., 2004; Wang et al., 2004; Luckenbach et al., 2005; Sisson et al., 2005; Shen et al., 2006). The governing equations of UnTRIM are solved using a semi-implicit, finite difference/finite volume numerical scheme based on the three-dimensional shallow water equations as well as on the three-dimensional transport equation. Quantities computed by the model include three-dimensional velocities, surface elevation, vertical viscosity and diffusivity, salinity, and temperature. Li (2006) performed numerous rigorous tests comparing the inlet dynamics predicted from UnTRIM with the classic analytical solutions of Keulegan (1967), King (1974), and DiLorenzo (1988) using ideal cases.
The numerical algorithms of UnTRIM (Casulli and Zanolli, 1998; Casulli, 1999; Casulli and Walters, 2000; Casulli and Zanolli, 2002) are relatively straightforward, and yet general and robust. The detailed model description can be found in the above references. Compared with an unstructured finite element model, UnTRIM has a number of interesting properties, such as global and local mass conservation, high-order numerical accuracy, and unconditional stability.

An unstructured orthogonal grid differs from the orthogonal grid, such as that used by other models like the Hydrodynamic Eutrophication Model in 3 Dimensions (HEM-3D) or the Princeton Ocean Model (POM). The orthogonal grid used by HEM-3D and POM consist of only four-sided structured polygons, but UnTRIM can use both three- or four-sided polygons. As with other models, the horizontal computational domain must be covered with a set of non-overlapping convex three- or four-sided polygons. Each side of the polygon is either a boundary line or a side of an adjacent polygon.

The highest numerical accuracy is obtained when a uniform grid, composed of equilateral triangles or uniform quadrilaterals (i.e., rectangles), is used. In these cases, the normal velocity on each face of each polygon is located at the center point of the face and the centers of two adjacent polygons are equally spaced from the common face. Consequently, the discretization error is small. An unstructured, nonuniform grid can be used with a somewhat larger discretization error (Casulli and Zanolli, 1998). The error would be amplified as the simulation time is long enough, which is common in water quality simulation. However, this error can be minimized when the polygon size and shape variations through the flow domain are properly arranged. So, in order to take full advantage of the new flexibilities of the unstructured grid, the grid size and shape should change gradually.

In the UnTRIM numerical scheme, the local volume conservation is assured by the finite volume formulation. At the same time, a finite volume method is used to discretize the free-surface two-dimensional equation at each polygon. In this fashion, local and global volume conservation is guaranteed. The transport equations are solved by using the sub-cycle upwind scheme, or using a higher resolution scheme -- flux limiter method (Casulli and Zanolli, 2005). Therefore, when the transport equations are calculated, mass is also conserved locally and globally because a finite volume form is used.

The Eulerian–Lagrangian method (ELM), also known as the semi-Lagrangian method (SL), is applied in the UnTRIM numerical scheme to solve the momentum equations. It allows one to achieve a very accurate discretization of the nonlinear advection terms (Staniforth and Temperton, 1991). The advection term is solved by the Lagrangian method, which can be computed independently at each time step by the method of characteristics applied to a fixed grid domain. ELM is especially efficient when applied to unstructured Cartesian grids (Casulli and Walters, 2000; Casulli and Zanolli, 2002; Cheng et al., 1993). When momentum equations are solved, ELM combines the advantages of the Eulerian method and the Lagrangian method, by merging the simplicity of a fixed Eulerian grid with the computational power of the Lagrangian method. The advantage of ELM is that the sharp front of velocity or concentration is easier to trace
since the system matrix becomes symmetric and diagonal (Casulli and Zanolli, 2002). Secondly, a large time step can be used, since the Courant number is not constrained by the small grid size (Casulli and Cattani, 1994; Casulli, 1999; Casulli and Walters, 2000; Casulli and Zanolli, 2002; Cheng and Casulli, 1996).

In applications to domains using the unstructured grid, there are two keys steps: approximation of the Lagrangian paths (characteristic streamlines) and interpolation at the departure point of the Lagrangian trajectory. The determination of the approximation of the characteristic streamline is solved using an integration method (Euler method) with a small time step shorter than the global time step. The method used by UnTRIM is called “Substepping” for the approximation of the backward trajectory (Casulli and Cattani, 1994; Casulli, 1999). In order to calculate the departure point, the bilinear interpolation is used by UnTRIM, which is sufficiently accurate.

The minimum grid size for a UnTRIM application can be as small as a few meters. However, due to its unconditional stability, UnTRIM can still use a very large timestep on the order of 10 minutes. Casulli and Cattani (1994) noted that the stability analysis of the semi-implicit finite difference method has been carried out in the case of barotropic and hydrostatic flow on a uniform rectangular grid. They assumed that the governing differential equations are linear, with constant coefficients, and are defined over an infinite horizontal domain. The analysis shows that the method is stable. Computational results of several test cases have indicated that no additional stability restrictions are required when a non-uniform unstructured mesh is used and when the hydrostatic assumption is removed. Thus, the stability of the present algorithm is independent of the celerity, wind stress, vertical viscosity, and bottom friction. It does depend on the discretization of the advection and horizontal viscosity terms. When an Eulerian-Lagrangian method is used for the explicit terms, a mild limitation on the time step depends on the horizontal viscosity coefficient and on the smallest polygon size. A further mild limitation on the time step is imposed in baroclinic flows because the baroclinic pressure term in the momentum equation has been discretized explicitly. This limitation is related to the internal wave speed that is typically smaller than the surface wave speed. This method becomes unconditionally stable for barotropic flows when the horizontal viscosity terms are neglected.

### III-2-2. Formulation of UnTRIM governing equations

The UnTRIM model was developed by Casulli (1999). Detailed descriptions of the numerical algorithms of the model can by found in Casulli and Zanolli (1998), Casulli (1999), and Casulli and Walters (2000). In Cartesian coordinates, the governing continuity and momentum equations for three-dimensional flows solved by the model are:

\[
\frac{\partial u}{\partial x} + \frac{\partial v}{\partial y} + \frac{\partial w}{\partial z} = 0
\]  

(III-1)
\[
\frac{\partial \eta}{\partial t} + \frac{\partial}{\partial x} \int_{z_h}^{z} \eta u dz + \frac{\partial}{\partial y} \int_{z_h}^{z} \eta v dz = 0 \quad \text{(III-2)}
\]

\[
\frac{D\mathbf{u}}{Dt} - fv = -\frac{\partial p_a}{\partial x} - g \frac{\partial \eta}{\partial x} - g \frac{\partial}{\partial z} \int_{z}^{\eta} \rho - \rho_0 d\xi - \frac{\partial q}{\partial x} (v_v \frac{\partial \mathbf{u}}{\partial z}) + \nu_h \left( \frac{\partial^2 \mathbf{u}}{\partial x^2} + \frac{\partial^2 \mathbf{u}}{\partial y^2} \right) \quad \text{(III-3)}
\]

\[
\frac{D\mathbf{v}}{Dt} + fu = -\frac{\partial p_a}{\partial y} - g \frac{\partial \eta}{\partial y} - g \frac{\partial}{\partial z} \int_{z}^{\eta} \rho - \rho_0 d\xi - \frac{\partial q}{\partial y} (v_v \frac{\partial \mathbf{v}}{\partial z}) + \nu_h \left( \frac{\partial^2 \mathbf{v}}{\partial x^2} + \frac{\partial^2 \mathbf{v}}{\partial y^2} \right) \quad \text{(III-4)}
\]

The transport equation for salt, temperature, and conservative solutes, \( C \), and an equation of state showing that the water density is a function of salinity and temperature are:

\[
\frac{\partial C}{\partial t} + \frac{\partial (uC)}{\partial x} + \frac{\partial (vC)}{\partial y} + \frac{\partial (wC)}{\partial z} = \frac{\partial}{\partial z} (K_v \frac{\partial C}{\partial z}) + \frac{\partial}{\partial x} \left( Kh \frac{\partial C}{\partial x} \right) + \frac{\partial}{\partial y} \left( Kh \frac{\partial C}{\partial y} \right) \quad \text{(III-5)}
\]

\[
\rho = \rho_0 \left[ 1 + \alpha s + \beta (T - T_0)^2 \right] \quad \text{(III-6)}
\]

where \((u, v, w)\) are \((x, y, z)\) velocity components, \( \eta \) is the free-surface elevation measured from a reference datum, \( v_v \) and \( v_h \) are vertical and horizontal eddy viscosities, \( \frac{D}{Dt} \) is the substantial derivative, \( \rho \) and \( \rho_0 \) are density and a reference density, \( p_a \) is atmospheric pressures, \( q \) is non-hydrostatic pressure component, \( f \) is the Coriolis parameter, \( C \) represents salinity, temperature, or other conservative solutes, \( K_v \) and \( K_h \) are the vertical and horizontal eddy diffusivities, \( s \) is salinity in practical salinity units (psu), \( T \) and \( T_0 \) are temperature and a reference temperature in °C, respectively, and constants \( \alpha = 7.8 \times 10^{-4} \) and \( \beta = 7 \times 10^{-6} \).

The surface wind stress components are computed using the quadratic relationships and the surface boundary conditions are:

\[
\nu_v \frac{\partial u}{\partial z} = \tau_{ux} = C_a \rho_a |u_a| u_a \quad \text{(III-7)}
\]

\[
\nu_v \frac{\partial v}{\partial z} = \tau_{vy} = C_a \rho_a |u_a| v_a \quad \text{(III-8)}
\]

where \(|u_a| = (u_a^2 + v_a^2)^{1/2}\), \( u_a \) and \( v_a \) are the horizontal components of wind velocity near the ocean surface, \( \rho_a \) is the air density, and \( C_a \) is the drag coefficient based on the following equation:

\[
C_a = (0.75 + 0.067 |u_a|) \times 10^{-3} \quad \text{(III-9)}
\]
The bottom stress is represented by the Manning’s friction relationship:

\[
\nu_y \frac{\partial \mathbf{u}}{\partial z} = \tau_{by} = \rho \frac{gh^2}{(\Delta z)^{0.5}} (u^2 + v^2)^{1/2} \mathbf{u}
\]  

(III-10)

\[
\nu_y \frac{\partial \mathbf{v}}{\partial z} = \tau_{by} = \rho \frac{gh^2}{(\Delta z)^{0.5}} (u^2 + v^2)^{1/2} \mathbf{v}
\]  

(III-11)

where \( n \) is the Manning parameter, \( u \) and \( v \) are bottom layer horizontal velocities, \( \Delta z \) is the bottom layer thickness, and \( \rho \) is the water density.

The model is a general three-dimensional model capable of simulating both 2-dimensional (vertical averaged) and 3-dimensional hydrodynamics and transport processes. The model uses a combined finite difference and finite volume scheme. Also, it uses an orthogonal, unstructured grid with mixed triangular and quadrilateral grid cells, which allows better fitting boundaries and local grid refinements to meet the needs of resolving spatial resolution in numerical modeling tasks. Figure III.2 shows an example of an orthogonal grid. The domain is covered by a set of non-overlapping convex polygons. Each side of a polygon is either a boundary line or a side of an adjacent polygon. The z-coordinate is used in the vertical. To relax the CFL condition, the Eulerian-Lagrangian transport scheme is used for treating the convective terms. A semi-implicit finite-difference method of solution was implemented in the model (Casulli, 1999). The terms that affect the numerical stability are treated implicitly, and the remaining terms are treated explicitly, which has proven to be computationally efficient (Cheng and Casulli, 2002). With the use of a Eulerian-Lagrangian transport scheme, the model is not restricted by the CFL condition. Therefore, very fine model grids can be used to represent the model domain without reducing computational efficiency.

III-3. The CE-QUAL-ICM Water Quality Model

The CE-QUAL-ICM water quality model was initially developed as one component of a model package employed to study eutrophication processes in Chesapeake Bay (US Army ERDC, 2000). ICM stands for "integrated compartment model," which is analogous to the finite volume numerical method. The model computes and reports concentrations, mass transport, kinetics transformations, and mass balances. This eutrophication model computes 22 state variables including multiple forms of algae, carbon, nitrogen, phosphorus, silica, and dissolved oxygen. One significant feature of ICM is a diagenetic sediment sub-model, which interactively predicts sediment-water oxygen and nutrient fluxes. Alternatively, these fluxes may be specified based on observations.

CE-QUAL-ICM has been applied to many sites, including Chesapeake Bay, Inland Bays of Delaware, New York Bight, Newark Bay, New York - New Jersey Harbors and

The foundation of CE-QUAL-ICM is the solution to the three-dimensional mass-conservation equation for a control volume based on the finite volume approach. Transport within the CE-QUAL-ICM (Cerco and Cole, 1995) is based on the integrated compartment method (or box model methodology). The present version of CE-QUAL-ICM transport is a loose extension of the original WASP code (Ambrose et al., 1986). The notion of utilizing the box model concept was retained in order to allow the coupling, via map files, of ICM with various hydrodynamic models. ICM represents "integrated compartment model," which is the finite volume numerical method. The model computes constituent concentrations resulting from transport and transformations in well-mixed cells that can be arranged in arbitrary triangular and quadrilateral configurations. Thus, the model employs an unstructured grid system, which is compatible with UnTRIM.

**III-3-1. Linkage between UnTRIM and CE-QUAL-ICM**

The foundation of CE-QUAL-ICM is the solution to the three-dimensional mass-conservation equation for a control volume based on the finite volume approach. For each volume and for each state variable, the governing equation that CE-QUAL-ICM solves is:
\[
\frac{\partial V_j C_j}{\partial t} = \sum_{k=1}^{n} Q_k C_k + \sum_{k=1}^{n} A_k D_k \frac{\partial C}{\partial x_k} + \sum_{k=1}^{n} S_j
\]  

(III-12)

where:

- \( V_j \) = volume of jth control volume (m\(^3\))
- \( C_j \) = concentration in jth control volume (mg m\(^{-3}\))
- \( t, x \) = temporal and spatial coordinates
- \( n \) = number of flow faces attached to jth control volume
- \( Q_k \) = volumetric flow across flow face k of jth control volume (m\(^3\) sec\(^{-1}\))
- \( C_k \) = concentration in flow across flow face k (mg m\(^{-3}\))
- \( A_k \) = area of flow face k (m\(^2\))
- \( D_k \) = diffusion coefficient at flow face k (m\(^2\) sec\(^{-1}\))
- \( S_j \) = external loads and kinetic sources and sinks in jth control volume (mg sec\(^{-1}\))

The above conservation-of-mass equation is solved in two steps. In the first step, an intermediate value is computed. The intermediate value includes the effects of change in cell volume, longitudinal and lateral transport, and external loading. This horizontal transport is solved using the UPWIND algorithm or the third-order-accurate non-uniform grid QUICKEST algorithm. In the second step, the effects of vertical transport and kinetic transformation are computed. The second-order implicit Crank-Nicolson scheme is used in the vertical direction. The linkage between UnTRIM and CE-QUAL-ICM focuses on the horizontal transport. The details of the horizontal transport methodology and the modifications required for a non-uniform and non-structured grid are presented below.

The original horizontal advection operator in CE-QUAL-ICM was designed to work with structured grid hydrodynamic models such as CH3D (Chapman and Cole, 1992). For a structured grid, grid information is described by rows and columns of cells combined with cell dimensions. The box lengths are directly calculated according to the relationship of rows and columns using a structured grid, and then are used to compute the UPWIND or QUICKEST transport multipliers. Due to prior successful applications of the UPWIND and QUICKEST transport algorithms in CE-QUAL-ICM (Dortch et al., 1991; Chapman and Cole, 1992), a similar approach was adopted for the non-structured version of CE-QUAL-ICM. The vertical transport computation utilizes the same solution, both for structured and unstructured grids.

An essential task of this study was the development of linkage software to provide geometric and hydrodynamic information transferring from UnTRIM output to the CE-QUAL-ICM code and to test the success of the linkage. The software development consisted of three basic parts:

a. Unstructured grid information used by the hydrodynamic model was transferred into CE-QUAL-ICM, including the number of polygons, faces, and the relationship between polygons and faces. The linkage software was developed to map the unstructured grid configuration and geometry information into several files that could be interpreted by the CE-QUAL-ICM code.
b. Hydrodynamic simulation results required for output and transferred into CE-
QUAL-ICM. A postprocessor code of UnTRIM was developed to output the
3-dimensional surface area of each polygon and volume of each polygon only
at the beginning of the simulation. The 3-dimensional velocity field, surface
water elevation information at each face and the center point of each polygon,
and vertical diffusivity were output at each time step.
c. CE-QUAL-ICM was modified to accept the UnTRIM linkage information,
especially in the input program and transport calculation.

The mapping of grid information between UnTRIM and CE-QUAL-ICM, and the
transfer of information between these two models, are described in more detail in Li

III-3-2. Dissolved oxygen process

(1) Effects of algae in water column on dissolved oxygen

Algae produce oxygen during photosynthesis and consume oxygen through respiration. The quantity produced during photosynthesis depends on the form of nitrogen taken up. Since oxygen is released in the reduction of nitrate (NO$_3$), more oxygen is produced, per unit of carbon fixed, when NO$_3$ is the algal nitrogen source than when ammonia NH$_4$ is the source. When NH$_4$ is the nitrogen source, one mole of oxygen is produced per mole carbon dioxide fixed. When NO$_3$ is the nitrogen source, 1.3 moles oxygen are produced per mole carbon dioxide fixed. The equation that describes the effect of algae photosynthesis on DO in the model is:

$$\frac{\Delta DO}{\Delta t} = \sum_x \left( (1.3 - 0.3 PN_x) P_x \right) AOCR \cdot B_x$$  \hspace{1cm} (III-13)

where:

PN$_x$ = algal group x preference for ammonium

P$_x$ = production rate of algal group x (day$^{-1}$)

AOCR = DO-to-carbon ratio in respiration (2.67 g O$_2$ per g C)

B$_x$ = algal biomass (g C m$^{-3}$)

As employed here, basal metabolism is the sum of all internal processes that decrease algal biomass. A portion of the metabolism is respiration and may be viewed as a reversal of production. In respiration, carbon and nutrients are returned to the environment accompanied by the consumption of DO. Respiration cannot proceed in the absence of DO. Basal metabolism cannot decrease in proportion to oxygen availability. Formulation of this process is described as:
\[
\frac{\delta \text{DO}}{\delta t} = \sum_x \left( -\frac{\text{DO}}{\text{KHR}_x + \text{DO}} \text{BM}_x \right) \text{AOCR} \cdot \text{B}_x \tag{III-14}
\]

where:

\( \text{KHR}_x = \) half-saturation constant of DO for algal DOC exudation (g O₂ m⁻³)

\( \text{BM}_x = \) basal metabolism rates for algal group \( x \) (day⁻¹)

(2) Effects of nitrification on dissolved oxygen

Nitrification is a process mediated by specialized groups of autotrophic bacteria that obtain energy through the oxidation of ammonia to nitrite and oxidation of nitrite to nitrate. A simplified expression for complete nitrification is:

\[
\text{NH}_4^+ + 2\text{O}_2 \rightarrow \text{NO}_3^- + \text{H}_2\text{O} + 2\text{H}^2+ \tag{III-15}
\]

The equation indicates that two moles of oxygen are required to nitrify one mole of ammonia into nitrate. The simplified equation is not strictly true, however. Cell synthesis by nitrifying bacteria is accomplished by the fixation of carbon dioxide so that less than two moles of oxygen are consumed per mole ammonium utilized (Wezernak and Gannon, 1968). In this study, nitrification is modeled as a function of available ammonium, dissolved oxygen, and temperature:

\[
\text{NT} = \frac{\text{DO}}{\text{KHONT} + \text{DO}} \frac{\text{NH}_4}{\text{KHNNT} + \text{NH}_4} \cdot f(T) \cdot \text{NTM} \tag{III-16}
\]

where:

\( \text{NT} = \) nitrification rate (gm N m⁻³ day⁻¹)

\( \text{NTM} = \) maximum nitrification rate at optimal temperature (gm N m⁻³ day⁻¹)

\( \text{KHONT} = \) half-saturation constant of DO required for nitrification (gm DO m⁻³)

\( \text{KHNNT} = \) half-saturation constant of \( \text{NH}_4 \) required for nitrification (gm N m⁻³)

Therefore, the effect of nitrification on DO is described as follows:

\[
\frac{\delta \text{DO}}{\delta t} = -\text{AONT} \cdot \text{NT} \tag{III-17}
\]

where:

\( \text{AONT} = \) mass DO consumed per mass ammonia nitrified (4.33 gm DO gm⁻¹ N)
(3) Effects of surface reaeration on dissolved oxygen

Reaeration occurs only in the model surface cells. The effect of reaeration is:

\[
\frac{\delta DO}{\delta t} = \frac{K_R}{\Delta z_s} (DO_s - DO)
\]  

(III-18)

where:

\(K_R\) = reaeration coefficient (m day \(^{-1}\))

\(\Delta z_s\) = model layer thickness (m)

\(DO_s\) = dissolved oxygen saturation concentration (gm DO m\(^{-3}\))

Saturation dissolved oxygen concentration \(DO_s\) is computed (Genet et al., 1974):

\[
DO_s = 14.5532 - 0.38217 \cdot T + 0.0054258 \cdot T^2
- \frac{S}{1.80655} \left(0.1665 - 5.866 \cdot 10^{-3} \cdot T + 9.796 \cdot 10^{-5} \cdot T^2\right)
\]

(III-19)

where:

\(S\) = salinity (ppt)

(4) Effects of Chemical Oxygen Demand on dissolved oxygen

In the present model, chemical oxygen demand represents the reduced materials that can be oxidized through inorganic means. The kinetic equation showing the effect of chemical oxygen demand is:

\[
\frac{\delta DO}{\delta t} = - \frac{DO}{K_{HOCOD} + DO} K_{COD} \cdot COD
\]

(III-20)

where:

\(COD\) = chemical oxygen demand concentrations (g O\(_2\)-equivalents m\(^{-3}\))

\(K_{HOCOD}\) = half-saturation constant of DO for oxidation of COD (g O\(_2\) m\(^{-3}\))

\(K_{COD}\) = oxidation rate of COD (day\(^{-1}\))

\[
K_{COD} = K_{CD} \cdot \exp(K_{TCOD}[T - TR_{COD}])
\]

(III-21)

where:

\(K_{CD}\) = oxidation rate of COD at reference temperature \(TR_{COD}\) (day\(^{-1}\))
KT_{COD} = \text{effect of temperature on oxidation of COD (°C}^{-1})
T = \text{water temperature (°C)}
TR_{COD} = \text{reference temperature for oxidation of COD (°C)}.

Overall, the internal sources and sinks of dissolved oxygen include algal photosynthesis and respiration, atmospheric reaeration (surface cells only), heterotrophic respiration, nitrification, and oxidation of COD. The complete kinetic equation showing sediment oxygen demand (bottom cells only) is:

\[
\frac{\delta \text{DO}}{\delta t} = \sum_x \left( (1.3 - 0.3 \cdot \text{PN}_x)P_x - \frac{\text{DO}}{\text{KHR}_x + \text{DO}} \cdot \text{BM}_x \right) \cdot \text{AOCR} \cdot B_x
\]
\[+ \lambda_1 \frac{K_R}{\Delta z} (DO_s - DO) - \frac{\text{DO}}{\text{KHO}_{\text{DOC}} + \text{DO}} \cdot \text{AOCR} \cdot K_{\text{DOC}} \cdot \text{DOC}
\]
\[\quad - \text{AONT} \cdot \text{NIT} - \frac{\text{DO}}{\text{KHO}_{\text{COD}} + \text{DO}} K_{\text{COD}} \cdot \text{COD} + \lambda_2 \frac{SOD}{\Delta z}
\]

(III-22)

III-3-3. Model Phytoplankton Kinetics

There are three functional groups for algae: cyanobacteria, diatoms, and green algae. This grouping is based upon the distinctive characteristics of each class and upon the significant roles these characteristics play in the ecosystem. Cyanobacteria are characterized by their bloom-forming characteristics in freshwater. They are characterized as having small settling velocity and are subject to low predation pressure. Diatoms are large phytoplankton that usually produces the spring bloom in the saline water. Settling velocity of diatoms is relatively large, so the diatoms settling into sediment may be a significant source of carbon for sediment oxygen demand. Diatoms are also distinguished by their requirement of silica as a nutrient. The green algae represent the mixture that characterizes blooming in saline waters during summer and autumn, and are subject to relatively high grazing pressure.

Equations governing the three algal groups are similar. Differences among groups are expressed through the magnitudes of parameters in the equations. Generic equations are presented below, except when group-specific relationships are required. Algal sources and sinks in the conservation equation include production, metabolism, predation, and settling. In the following equations, a subscript, \( x \), is used to denote three algal groups: \( c \) for cyanobacteria, \( d \) for diatoms, and \( g \) for green algae. The internal sources and sinks included are growth (production), basal metabolism (respiration and exudation), predation, and settling. The kinetic equations for algae are:

\[
\frac{\delta B_x}{\delta t} = \left( P_x - \text{BM}_x - \text{PR}_x \right) B_x - WS_x \frac{\delta B_x}{\delta z}
\]

(III-23)

where:
Bx = algal biomass, expressed as carbon (g C m\(^{-3}\))
Px = growth (production) rates of algae (day\(^{-1}\))
BMx = basal metabolism rates of algae (day\(^{-1}\))
PRx = predation rates of algae (day\(^{-1}\))
WSx = algal settling velocity (m day\(^{-1}\))
z = vertical coordinate

(1) Growth (Production)

Algal growth rate depends on nutrient availability, ambient light, and temperature. The effects of these processes are considered to be multiplicative as follows:

\[
P_x = PM_x \cdot f(N) \cdot f(I) \cdot f(T)
\]

(III-24)

where:

PMx = maximum production rate under optimal conditions (day\(^{-1}\))
f(N) = effect of sub-optimal nutrient
f(I) = effect of light intensity
f(T) = effect of temperature

(2) Effect of nutrient on growth

Liebig’s “law of the minimum” (Odum, 1971) is used, so that nutrient limitation is determined by the single most limiting nutrient:

\[
f(N) = \text{minimum}\left\{ \frac{\text{NH}_4 + \text{NO}_3}{\text{KHN}_x + \text{NH}_4 + \text{NO}_3}, \frac{\text{PO}_{4d}}{\text{KHP}_x + \text{PO}_{4d}}, \frac{\text{SAd}}{\text{KHS}_d + \text{SAd}} \right\}
\]

(III-25)

where:

\(\text{NH}_4, \text{NO}_3\) = ammonium and nitrate nitrogen concentrations, respectively (g N m\(^{-3}\))
PO\(_4d\) = dissolved phosphate concentration (g P m\(^{-3}\))
SAd = dissolved silica concentration (g Si m\(^{-3}\))
KHN\(_x\) = half-saturation constant for algal nitrogen uptake (g N m\(^{-3}\))
KHP\(_x\) = half-saturation constant for algal phosphorus uptake (g P m\(^{-3}\))
KHS\(_d\) = half-saturation constant for silica uptake by diatoms (g Si m\(^{-3}\))

(3) Effects of light on growth

The influence of light on phytoplankton production is represented by a chlorophyll-specific production equation (Jassby and Platt, 1976):
\[ P^B = P^m m \frac{I}{\sqrt{I + I_k^2}} \quad \text{(III-26)} \]

where:

- \( P^B \) = photosynthetic rate (g C g\(^{-1}\) Chl d\(^{-1}\))
- \( P^m m \) = maximum photosynthetic rate (g C g\(^{-1}\) Chl d\(^{-1}\))
- \( I \) = irradiance (E m\(^{-2}\) d\(^{-1}\))

Parameter \( I_k \) is defined as the irradiance at which the initial slope of the production vs. irradiance relationship intersects the value of \( P^m m \):

\[ I_k = \frac{P^m m}{\alpha} \quad \text{(III-27)} \]

where:

- \( \alpha \) = initial slope of production vs. irradiance relationship (g C g\(^{-1}\) Chl (E m\(^{-2}\))\(^{-1}\))

Chlorophyll-specific production rate is readily converted to carbon-specific growth rate, through division by the carbon-to-chlorophyll ratio:

\[ G = \frac{P^B}{CChl} \quad \text{(III-28)} \]

where:

- \( CChl \) = carbon-to-chlorophyll ratio (g C g\(^{-1}\) chlorophyll-a)

(4) Effect of temperature on growth

The effect of temperature on algal production is represented by a function similar to a Gaussian probability curve:

\[ f(T) = \exp\left(-KTG_{1x} [T - TM_x]^2\right) \quad \text{when} \; T \leq TM_x \]
\[ = \exp\left(-KTG_{2x} [TM_x - T]^2\right) \quad \text{when} \; T > TM_x \quad \text{(III-29)} \]

where:

- \( TM_x \) = optimal temperature for algal growth (°C)
- \( KTG_{1x} \) = effect of temperature below \( TM_x \) on algal growth (°C\(^{-2}\))
- \( KTG_{2x} \) = effect of temperature above \( TM_x \) on algal growth (°C\(^{-2}\))
(5) Constructing the photosynthesis vs. irradiance curve

A production versus irradiance relationship is constructed for each model cell at each time step. First, the maximum photosynthetic rate under ambient temperature and nutrient concentrations is determined:

$$P^Bm(N, T) = P^Bm * f(T) * f(N)$$  \hspace{1cm} (III-30)

where:

$$P^Bm(N, T) = \text{maximum photosynthetic rate under ambient temperature and nutrient concentrations (g C g}^{-1} \text{ Chl d}^{-1})$$

The single most limiting nutrient is employed in determining the nutrient limitation. Next, parameter $I_k$ is derived from Equation III-27. Finally, the production vs. irradiance relationship is constructed using $P^Bm(N, T)$ and $I_k$.

(6) Water surface irradiance

Irradiance at the water surface is evaluated at each model time step. Instantaneous irradiance is computed by fitting a sine function to daily total irradiance:

$$I_o = \frac{I_T \pi}{FD} \sin\left(\pi \frac{DSSR}{FD}\right)$$  \hspace{1cm} (III-31)

where:

$I_o = \text{irradiance at water surface (E m}^{-2} \text{ d}^{-1})$

$I_T = \text{daily total irradiance (E m}^{-1})$

$FD = \text{fractional daylength (0 < FD < 1)}$

$DSSR = \text{time since sunrise (d)}$

$I_o$ is evaluated only during the interval:

$$\frac{1 - FD}{2} \leq DSM \leq \frac{1 + FD}{2}$$  \hspace{1cm} (III-32)

where:

$DSM = \text{time since midnight (d)}$
Outside the specified interval, $I_0$ is set to zero.

Irradiance declines exponentially with depth below the surface. The diffuse attenuation coefficient, $K_e$, is computed as a function of background extinction and concentrations of chlorophyll-a and total suspended solids.

(7) The light attenuation model

The water quality model requires daily solar radiation intensity and fractional day length, in order to simulate the algal growth. The light attenuation model also requires input of the light attenuation coefficient. It is assumed that the light extinction coefficient consists of three parts: background extinction, the light extinction due to suspended solids, and light extinction due to algae:

$$K_e = a_1 + a_2 \cdot TSS + a_3 \cdot CHL$$  \hspace{1cm} (III-33)

where:

- $a_1$ = background attenuation (m$^{-1}$)
- $a_2$ = attenuation by inorganic suspended solids (m$^2$ g$^{-1}$)
- $a_3$ = attenuation by organic suspended solids (m$^2$ g$^{-1}$ CHL)
- TSS = total suspended solids concentration (g m$^{-3}$)
- CHL = chlorophyll-a concentration (mg CHL m$^{-3}$)

The “background” attenuation term included attenuation from both water and dissolved organic matter. Individual parameters were determined from Park et al. (1995b). The value for $a_1$ used in the model is 0.735 m$^{-1}$, $a_2$ is 0.018 m$^2$ g$^{-1}$, and $a_3$ is 0.06 m$^2$ mg$^{-1}$ CHL.

(8) Basal metabolism

Basal metabolism is commonly considered to be an exponentially increasing function of temperature:

$$BM_x = BMR_x \cdot \exp(KTB_x \cdot [T - TR_x])$$  \hspace{1cm} (III-34)

where:

- $BMR_x$ = metabolic rate at reference temperature $TR_x$ (day$^{-1}$)
- $KTB_x$ = effect of temperature on metabolism (C$^o$)$^{-1}$
- $TR_x$ = reference temperature for metabolism (C$^o$)

(9) Predation
The predation formulation is identical to basal metabolism. The difference in predation and basal metabolism lies in the distribution of the end products of these processes.

\[ PR_x = BPR_x \exp (KTB_x (T - TR_x)) \]  

(III-35)

where:

- \( BPR_x \) = predation rate at \( TR_x \) (day \(^{-1}\))
- \( KTB_x \) = effect of temperature on predation (C\(^{-1}\))
- \( TR_x \) = reference temperature for predation (C)

(10) Settling velocity

The algal settling rate employed in the model represents the total effect of all physiological and behavioral processes that result in the downward transport of phytoplankton. The settling rate employed, from 0.1 m d\(^{-1}\) to 0.2 m d\(^{-1}\), was used in the model to optimize the agreement between predicted and observed algae.

(11) Effect of algae on phosphorus

Model phosphorus state variables include total phosphate (dissolved, sorbed, and algal), dissolved organic phosphorus, labile particulate organic phosphorus, and refractory particulate organic phosphorus. The amount of phosphorus incorporated in algal biomass is quantified through a stoichiometric ratio. Thus, total phosphorus in the model is expressed:

\[ \text{TotP} = PO_{4d} + PO_{4p} + \sum_x Apc \cdot Bx + DOP + LPOP + RPOP \]  

(III-36)

where:

- \( \text{TotP} \) = total phosphorus (g P m\(^{-3}\))
- \( PO_{4d} \) = dissolved phosphate (g P m\(^{-3}\))
- \( PO_{4p} \) = particulate inorganic phosphate (g P m\(^{-3}\))
- \( Apc \) = algal phosphorus-to-carbon ratio (g P g\(^{-1}\) C)
- \( DOP \) = dissolved organic phosphorus (g P m\(^{-3}\))
- \( LPOP \) = labile particulate organic phosphorus (g P m\(^{-3}\))
- \( RPOP \) = refractory particulate organic phosphorus (g P m\(^{-3}\))

Algae take up dissolved phosphate during production and release dissolved phosphate and organic phosphorus through respiration. The fate of phosphorus released by respiration is determined by empirical distribution coefficients. The fate of algal phosphorus incorporated by zooplankton and lost through zooplankton mortality is determined by a second set of distribution parameters.
Effect of algae on nitrogen

Model nitrogen state variables include ammonium, nitrate + nitrite, dissolved organic nitrogen, labile particulate organic nitrogen, and refractory particulate organic nitrogen. The amount of nitrogen incorporated in algal biomass is quantified through a stoichiometric ratio. Thus, total nitrogen in the model is expressed:

$$\text{TotN} = \text{NH}_4 + \text{NO}_3 + \sum_x \text{Anc} \ast B_x + \text{DON} + \text{LPON} + \text{RPON}$$  \hspace{1cm} (III-37)

where:

- $\text{TotN}$ = total nitrogen (g N m$^{-3}$)
- $\text{NH}_4$ = ammonium (g N m$^{-3}$)
- $\text{NO}_3$ = nitrate + nitrite (g N m$^{-3}$)
- $\text{Anc}$ = algal nitrogen-to-carbon ratio (g N g$^{-1}$ C)
- $\text{DON}$ = dissolved organic nitrogen (g N m$^{-3}$)
- $\text{LPON}$ = labile particulate organic nitrogen (g N m$^{-3}$)
- $\text{RPON}$ = refractory particulate organic nitrogen (g N m$^{-3}$)

Algae take up ammonium and nitrate + nitrite during production and release ammonium and organic nitrogen through respiration. Nitrate + nitrite is internally reduced to ammonium before synthesis into biomass occurs (Parsons et al., 1984). Trace concentrations of ammonium inhibit nitrate reduction so that, in the presence of multiple nitrogenous nutrients, ammonium is utilized first. The “preference” of algae for ammonium is expressed by an empirical function (Thomann and Fitzpatrick, 1982):

$$\text{PN} = \frac{\text{NH}_4 \ast \text{NO}_3}{(\text{KHn} + \text{NH}_4) \ast (\text{KHn} + \text{NO}_3)}$$  \hspace{1cm} (III-38)

where:

- $\text{PN}$ = algal preference for ammonium uptake ($0 < \text{PN} < 1$)
- $\text{KHn}$ = half saturation concentration for algal nitrogen uptake (g N m$^{-3}$)

When nitrate + nitrite is absent, the preference for ammonium is unity. When ammonium is absent, the preference is zero.

Effect of algae on silica
The model incorporates two siliceous state variables: dissolved silica and particulate biogenic silica. The amount of silica incorporated in algal biomass is quantified through a stoichiometric ratio. Thus, total silica in the model is expressed:

\[
\text{TotSi} = D_{\text{sil}} + A_{\text{sc}} \times B_x + PBS \tag{III-39}
\]

where:

- \( \text{TotSi} \) = total silica (g Si m\(^{-3}\))
- \( D_{\text{sil}} \) = dissolved silica (g Si m\(^{-3}\))
- \( A_{\text{sc}} \) = algal silica-to-carbon ratio (g Si g\(^{-1}\) C)
- \( PBS \) = particulate biogenic silica (g Si m\(^{-3}\))

As with the other nutrients, the fate of algal silica released by metabolism and predation is represented by distribution coefficients.

### III-3-4. Benthic sediment process

Additionally, a benthic sediment process model developed by DiToro and Fitzpatrick (1993) was incorporated and coupled with CE-QUAL-ICM for the present model application. The model state variables, and resulting fluxes, include dissolved oxygen, ammonium, nitrate-nitrite, and phosphate and the parameters used in this sediment flux model are listed in the Table V.10 of Chapter V.

The sediments in this model are represented by two layers: the upper aerobic layer (Layer 1) and the lower anoxic layer (Layer 2). The sediment process model is coupled with the water column eutrophication model through depositional and sediment fluxes. First, the sediment model is driven by net settling of particulate organic matter from the overlying water column to the sediments (depositional flux). Then, the mineralization of particulate organic matter in the lower anoxic sediment layer produces soluble intermediates, which are quantified as diagenesis fluxes. The intermediates react in the upper oxic and lower anoxic layers, and portions are returned to the overlying water column as sediment fluxes. Computation of sediment fluxes requires mass-balance equations for ammonium, nitrate, phosphate, sulfide/methane, and available silica. Mass-balance equations are solved for these variables for both the upper and lower layers. Complete model documentation of the sediment flux model can be found in DiToro and Fitzpatrick (1993).

It should be noted that, due to the critical nature of impacts to Lynnhaven water clarity from total suspended solids (TSS), a decision was made to add to the project scope of work the development of a sediment transport model capable of fully simulating the processes of erosion, deposition, and sediment resuspension. This sediment transport model is described in the next section.
III-4. Description of the sediment transport model

The model utilized in this study is principally based on that of Sanford (2008). As the mud percentage of the bottom sediments in the Lynnhaven basin is larger than 10% in most parts of the Basin and the bottom sediments are mainly composed of silty clay, the formulae of cohesive sediment erosion and deposition were adopted, which are described in the following. The spatial distribution of the sand percentage, and the percentage of silt and clay in the bottom sediment was obtained by grain size analysis of the sediment samples in the basin (Figure III.3). It can be seen that in the inlet and the main channels of the Western and Eastern Branches, sand takes up most part of the sediment. Sand also dominates in the shallow area along the shoreline, mostly induced by shoreline erosion. For most of the area in the basin, sand percentage is less than 90%.

In this study, only silt and clay were simulated. To account for the sediment consolidation, the method of Sanford (2008) for adjusting the bottom critical shear stress was adopted. It assumes that there exists a vertical profile of the equilibrium critical shear stress through the sediment bed, and the actual critical shear stress adapts to the equilibrium one in a first-order time evolution manner.

\[
\frac{\partial \tau_c}{\partial t} = r_c (\tau_{ceq} - \tau_c) H(\tau_{ceq} - \tau_c) + r_s (\tau_{ceq} - \tau_c) H(\tau_c - \tau_{ceq})
\]  

(III-40)

Figure III.3. Sand percentage of the bottom sediment of the Lynnhaven River.
Where \( \tau_c \) is the instantaneous critical shear stress, \( \tau_{ceq} \) is the equilibrium critical shear stress, \( H \) is the Heaviside step function, defined such that \( H = 1 \) when its argument is \( \geq 0 \) and \( H = 0 \) otherwise. In Eq. (III-40), \( r_c \) is the first-order consolidation rate and \( r_s \) is the first-order swelling rate, which is much smaller than \( r_c \). In this study \( r_c \) was set as \( 0.01 \) per day and \( r_s = 0.01 r_c \), following Sanford (2008).

The erosion rate is
\[
E = M \left( \frac{\tau_b(t)}{\tau_c} - 1 \right) \quad \text{if} \ (\tau_b > \tau_c)
\]
\[
E = 0 \quad \text{if} \ (\tau_b \leq \tau_c)
\]

Where \( \tau_b \) is the bottom stress, \( M \) is an erosion rate parameter, which can be obtained from the observation data, like that in Baltimore Harbor, Maryland, USA (Lin et al., 2003). In this study it was adjusted until the model results agreed with measurements and, thus, the calibrated value of \( M \) is 0.0004 g/m²/s.

In this study, the equilibrium critical shear stress profile was set equal to the critical stress profile obtained by bottom sediment erodibility tests in Lynnhaven basin by Sanford and Suttles.

\[
\tau_{ceq} = 0.7006 m^{1.5309}
\]

Where \( \tau_{ceq} \) is the equilibrium critical shear stress defined at the interface between layers, \( m \) is the accumulated sediment mass (kg) within the layers above the interface. The equilibrium critical shear stress at the water-sediment interface was specified spatially varying. The spatial distribution of the water-sediment interface equilibrium critical shear stress was obtained by executing the hydrodynamic model for approximately one month to cover the spring-neap tidal variability, and averaging the modeled bottom stress for every cell. The result of equilibrium critical shear stress distribution at the water-sediment interface is shown in Figure III.4. It can be seen that the shear stress has good correlation with the sand percentage of the bottom sediment, the higher sand percentage, the larger of the shear stress. This is consistent with the findings of Molinaroli et al. (2007) that the sediment sorting was mostly controlled by the tidal hydrodynamics in the Lagoon of Venice, Italy. They obtained a good relationship between the sand percentage of the bottom sediment and the mean tidal velocity.

The equilibrium critical shear stress of water-sediment interface was assigned to the corresponding cells. From Figures III.3 and III.4, the equilibrium critical shear stress of the water-sediment interface for the areas with sand percentages less than 70% was mostly close to 0.03 Pa, which is consistent with the measurement data of Sanford and Suttles. Under the water-sediment interface, a total of 25 bed layers were defined. At each layer of the first 20 layers a sediment mass of 0.5 kg/m² was specified, whereas for
The last 5 layers sediment masses were given as 5.0, 25, 50, 75 and 100 kg/m$^2$, respectively. The equilibrium critical shear stress for each layer was specified as the larger of water-sediment interface one and that derived from Eq. (III-42).

At each time step, the bed layers were adjusted by adding or removing layers to account for the deposition or erosion in the bed based on Sanford (2008). With newly deposited sediment at first layer of the bottom, the critical shear stress at the water-sediment interface was decreased as demonstrated by Lin et al. (2003). When the sediment was eroded from the layer, the critical shear stress was increased as illustrated from Eq. (III-41). After the above adjustment, the critical shear stresses were relaxed to the equilibrium ones based on Eq. (III-40).

The deposition rate of cohesive sediment was calculated as

\[
D = \begin{cases} 
    w_s C_b \frac{\tau_{dc} - \tau_b}{\tau_{dc}} & \text{for } \tau_{dc} > \tau_b \\
    0 & \tau_{dc} \leq \tau_b 
\end{cases} \tag{III-43}
\]

Figure III.4. Average bottom shear stress obtained by one month of hydrodynamic simulation.
Where $\tau_{dc}$ is the critical shear stress for deposition, which was set as 0.03 Pa in this study. The existence of a critical shear stress for deposition is debatable, a value of 0.035 Pa has been utilized in Lin and Kuo’s (2003) study, and a continuous settling concept was adopted by Sanford (2008).

To account for the flocculation, the cohesive sediment’s settling velocity dependence on concentration was utilized, which was obtained by Kwon (2005) through measurement in the York River as follows:

$$w_s = 3.5 \times 10^{-5} C^{0.375}$$

(III-44)

where $w_s$ is in units of m/s and $C$ is in units of g m$^{-3}$.

The calibration of the Lynnhaven River sediment transport model is presented in Section V-3 and its validation is presented in Section VI-3.

### III-5. Description of the watershed model for the Lynnhaven River Basin

As VIMS has developed the hydrodynamic and water quality models for the Lynnhaven River receiving waters, URS Corporation of Virginia Beach has developed a watershed model for the Lynnhaven River Basin. The watershed model used by URS is HSPF (Hydrological Simulation Program – FORTRAN), version 12 (URS Technical Memorandum, Hydrologic Concepts and Parameter Development, 2006).

The goal of the watershed modeling effort is to provide the freshwater discharge and nutrient and sediment loadings from the watershed at high spatial and temporal resolutions. The Lynnhaven River Basin, consisting of 7 sub-basins, has been delineated into 1,079 catchments, ranging in size from approximately 40 acres, as shown in Figure III.5.

The landuse in the Lynnhaven Basin is 40% residential and 35% composed of streets, commercial and office space, and military use. In its watershed model development, URS selected a total of 23 land uses within the Lynnhaven River basin into which zoning codes could then be grouped. URS then assigned to each landuse a directly connected impervious percentage, as shown in Table III.1. Landuse was employed to develop effective impervious area percentages for the nearly 57,000 land parcels within the Lynnhaven Basin.
Figure III.5. The 1079 catchment areas delineated by the URS watershed model superimposed on the UnTRIM model grid.
For each of these catchments, the URS model simulates the following 9 constituents:
- biochemical oxygen demand (BOD)
- total dissolved solids (TDS)
- chemical oxygen demand (COD)
- nitrate – nitrite (NO₃)
- total Kjeldahl nitrogen (TKN)
- ammonia (NH₃)
- total phosphorus (TP)
- dissolved phosphorus (DP)
- total suspended sediments (TSS)

The URS model was calibrated for by comparing its predictions to monitoring data collected at 5 sites within and/or nearby the Lynnhaven basin (URS, 2007). The calibrated model was then used to provide multi-year datasets of its outputs of hourly nutrient loadings and freshwater discharge to the VIMS models.

Table III.1. Impervious percentages of Lynnhaven Basin Landuse Categories.

<table>
<thead>
<tr>
<th>Landuse No.</th>
<th>Landuse</th>
<th>Landuse Description</th>
<th>Impervious Percentage</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>AG</td>
<td>Agricultural</td>
<td>15%</td>
</tr>
<tr>
<td>2</td>
<td>SFL</td>
<td>Single Family Low Density</td>
<td>16%</td>
</tr>
<tr>
<td>3</td>
<td>SFM</td>
<td>Single Family Medium Density</td>
<td>21%</td>
</tr>
<tr>
<td>4</td>
<td>SFH</td>
<td>Single Family High Density</td>
<td>24%</td>
</tr>
<tr>
<td>5</td>
<td>MFM</td>
<td>Multi-Family Medium Density</td>
<td>37%</td>
</tr>
<tr>
<td>6</td>
<td>MFH</td>
<td>Multi-Family High Density</td>
<td>62%</td>
</tr>
<tr>
<td>7</td>
<td>PD</td>
<td>Planned Development</td>
<td>29%</td>
</tr>
<tr>
<td>8</td>
<td>O</td>
<td>Office</td>
<td>71%</td>
</tr>
<tr>
<td>9</td>
<td>NB</td>
<td>Neighborhood Business</td>
<td>39%</td>
</tr>
<tr>
<td>10</td>
<td>B</td>
<td>Business</td>
<td>73%</td>
</tr>
<tr>
<td>11</td>
<td>I</td>
<td>Industrial</td>
<td>45%</td>
</tr>
<tr>
<td>12</td>
<td>RT</td>
<td>Resort Tourist</td>
<td>71%</td>
</tr>
<tr>
<td>13</td>
<td>PK</td>
<td>Park</td>
<td>5%</td>
</tr>
<tr>
<td>14</td>
<td>GC</td>
<td>Golf Course</td>
<td>5%</td>
</tr>
<tr>
<td>15</td>
<td>OS</td>
<td>Open Space</td>
<td>0.5%</td>
</tr>
<tr>
<td>16</td>
<td>OF</td>
<td>Other facilities</td>
<td>8%</td>
</tr>
<tr>
<td>17</td>
<td>SC</td>
<td>School</td>
<td>47%</td>
</tr>
<tr>
<td>18</td>
<td>ST</td>
<td>Street</td>
<td>60%</td>
</tr>
<tr>
<td>19</td>
<td>CM</td>
<td>Cemetary</td>
<td>5%</td>
</tr>
<tr>
<td>20</td>
<td>CH</td>
<td>Church</td>
<td>47%</td>
</tr>
<tr>
<td>21</td>
<td>WT</td>
<td>Wetland</td>
<td>100%</td>
</tr>
<tr>
<td>22</td>
<td>BMP</td>
<td>Best Management Practice</td>
<td>100%</td>
</tr>
<tr>
<td>23</td>
<td>WAT</td>
<td>Water</td>
<td>100%</td>
</tr>
</tbody>
</table>
CHAPTER IV. HISTORICAL DATA AND FIELD OBSERVATION PROGRAM

IV-1. Historical Data

Historical monitoring and survey data collection in the Lynnhaven River have taken place since the late 1950s. Prior to the inception of this project, VIMS made a conscious effort to gather all available hydrodynamic and water quality data recorded from the Lynnhaven River system into a central database. The intended range of parameters included in the database span those needed for the calibration and validation of the hydrodynamic and water quality models. Specifically, these include hydrodynamic parameter data (tides, velocities, salinities, and temperatures) and water quality parameter data (dissolved oxygen, chlorophyll, nutrient concentrations, and sediment-related measurements).

Historical data for the Lynnhaven originated from 3 state agencies (Virginia Department of Environmental Quality [VA-DEQ], Virginia Department of Shellfish Sanitation [VA-DSS], and Virginia Institute of Marine Science [VIMS]), 1 federal agency (National Oceanic and Atmospheric Administration [NOAA]), and 1 environmental consulting company (Malcolm Pirnie Engineers). Whereas VIMS, NOAA, and Malcolm Pirnie conducted surveys of the Lynnhaven, most water quality parameter measurements have been provided by the ongoing monitoring programs of VA-DEQ (every other month, 1984 to present) and VA-DSS (monthly, 1986 to present). These data are summarized in Table IV.1.

Table IV.1. Lynnhaven monitoring and survey data collected, by parameter and agency.

<table>
<thead>
<tr>
<th>Sections</th>
<th>Parameter</th>
<th>Number of Observations by Agency</th>
<th>Total Observations</th>
</tr>
</thead>
<tbody>
<tr>
<td>II A</td>
<td>Tides</td>
<td>DEQ: 2924, DSS: 2269, VIMS: 511, M. PIRNIE: 200</td>
<td>5953</td>
</tr>
<tr>
<td>II B</td>
<td>Velocity</td>
<td></td>
<td></td>
</tr>
<tr>
<td>II C</td>
<td>Salinity</td>
<td>DEQ: 2648, DSS: 1275, VIMS: 475, M. PIRNIE: 200</td>
<td>4598</td>
</tr>
<tr>
<td>II D</td>
<td>Temperature</td>
<td>DEQ: 5208, DSS: -2, VIMS: 527, M. PIRNIE: 400</td>
<td>6135</td>
</tr>
<tr>
<td>II A A</td>
<td>Dissolved Oxygen</td>
<td>DEQ: 149, DSS: -2, VIMS: 511, M. PIRNIE: 200</td>
<td>860</td>
</tr>
<tr>
<td>II B B</td>
<td>Chlorophyll a</td>
<td>DEQ: 2133, DSS: 135, VIMS: 200</td>
<td>2468</td>
</tr>
<tr>
<td>II C C</td>
<td>BOD5</td>
<td>DEQ: 1863, DSS: -2, VIMS: -2, M. PIRNIE: -2</td>
<td>1863</td>
</tr>
<tr>
<td>II E E</td>
<td>TKN</td>
<td>DEQ: 2351, DSS: -2, VIMS: -2, M. PIRNIE: -2</td>
<td>2351</td>
</tr>
<tr>
<td>II F F</td>
<td>Ammonia</td>
<td>DEQ: 2645, DSS: -2, VIMS: -2, M. PIRNIE: -2</td>
<td>2645</td>
</tr>
<tr>
<td>II H H</td>
<td>Nitrate</td>
<td>DEQ: 1682, DSS: 459, VIMS: 200</td>
<td>2341</td>
</tr>
<tr>
<td>III I I</td>
<td>Total Phosphorus</td>
<td>DEQ: 1158, DSS: -2, VIMS: -2, M. PIRNIE: -2</td>
<td>1158</td>
</tr>
<tr>
<td>III L L</td>
<td>TSS</td>
<td>DEQ: 2072, DSS: 16, VIMS: 200</td>
<td>2288</td>
</tr>
<tr>
<td>III O O</td>
<td>Turbidity</td>
<td>DEQ: 1061, DSS: -2, VIMS: -2, M. PIRNIE: -2</td>
<td>1061</td>
</tr>
<tr>
<td>III P P</td>
<td>Secchi depths</td>
<td>DEQ: -1142, DSS: 459, VIMS: 200</td>
<td>1801</td>
</tr>
<tr>
<td>III Q Q</td>
<td>Fecal Coliform</td>
<td>DEQ: 1010, DSS: 17,725, VIMS: 459, M. PIRNIE: 200</td>
<td>19,394</td>
</tr>
</tbody>
</table>
Spatial plots of long-term averages of hydrodynamic and water quality parameters can often reveal important characteristics of a waterbody such as the Lynnhaven. It can be seen from the long-term averages for salinity at DEQ stations, shown in Figure IV.1, that much larger salinity gradients exist in the Western and Eastern Branches than in the Broad Bay / Linkhorn Bay Branch. This is because the freshwater inputs from the former branches are larger than that of the later. Spatial plots of water quality parameters can be used to highlight the spatial gradient of the water quality parameters as well as identify the regions of concerns, such as the DEQ stations at Thalia Creek and London Bridge, as shown in Figure IV.2. One of the major characteristics revealed was that the concentration of all water quality variables were higher at the upstream of each branch and decreased moving downstream toward the Inlet.

The availability of long-term monitoring data additionally allows for time series analysis and, in the case of long-term trend, a simple linear trend analysis was performed for all parameters. Examples of this include the long-term decrease of dissolved oxygen at the Thalia Creek Station shown in Figure IV.3a and the decrease of total organic carbon at the Broad Bay Station BBY002.88 shown in Figure IV.3b. Table IV.2 enumerates the long-term trends of all water quality parameters measured at each Lynnhaven DEQ station as either increasing (I) or decreasing (D).

![Figure IV.1. Long-term average salinity based on Lynnhaven DEQ observations.](image-url)
Figure IV.2. Long-term average total phosphorus based on Lynnhaven DEQ observations.

Figure IV.3a. Long-term trend of observed dissolved oxygen at DEQ Station THA000.76.
Figure IV.3b. Long-term trend of observed TOC at DEQ Station BBY002.88.

Table IV.2. Lynnhaven DEQ monitoring long-term trends.

<table>
<thead>
<tr>
<th>DEQ Station</th>
<th>THA000.76</th>
<th>WES002.85</th>
<th>WES001.68</th>
<th>WES000.85</th>
<th>LYN000.03</th>
<th>EBL000.01</th>
<th>EBL000.02</th>
<th>LEB001.79</th>
<th>LOB001.79</th>
<th>BBY002.88</th>
<th>LNK001.19</th>
<th>LNK002.77</th>
<th>LNC000.68</th>
<th>CRY000.59</th>
</tr>
</thead>
<tbody>
<tr>
<td>Parameter</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Salinity</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
</tr>
<tr>
<td>Temperature</td>
<td>D</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
</tr>
<tr>
<td>Dissolved Oxygen</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
</tr>
<tr>
<td>Chlorophyll a</td>
<td>D</td>
<td>D</td>
<td>I</td>
<td>I</td>
<td>D</td>
<td>I</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>I</td>
<td>I</td>
<td>D</td>
<td>D</td>
</tr>
<tr>
<td>BOD5</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
</tr>
<tr>
<td>TOC</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
</tr>
<tr>
<td>TKN</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
</tr>
<tr>
<td>Ammonia</td>
<td>Regression not reported - interference from detection limit change</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Nitrite</td>
<td>Regression not reported - interference from detection limit change</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Nitrate</td>
<td>Regression not reported - interference from detection limit change</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total Phosphorus</td>
<td>Regression not reported - interference from detection limit change</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ortho Phosphorus</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
</tr>
<tr>
<td>Dissolved Silica</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
</tr>
<tr>
<td>TSS</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
</tr>
<tr>
<td>VSS</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
</tr>
<tr>
<td>Volatile Solids</td>
<td>I</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>I</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
</tr>
<tr>
<td>Turbidity</td>
<td>D</td>
<td>D</td>
<td>I</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>I</td>
<td>D</td>
<td>D</td>
<td>D</td>
<td>D</td>
</tr>
</tbody>
</table>

“T” denotes a long-term increasing trend and “D” denotes a long-term decreasing trend
IV-2. Project-specific field measurements

For the data to be useful for the hydrodynamic and water quality modeling, project-specific measurements are required. There are five field data collections were designed and conducted during the course of the project. They are described in the following sections: (1) the hydrodynamic survey in Section IV-2-1, (2) seasonal sediment flux measurements in Section IV-2-2, (3) sediment critical shear stress measurements in Section IV-2-3, (4) High spatial resolution dataflow surveys in Section IV-2-4, and (5) high-frequency time series measurements in Section IV-2-5.

IV-2-1. VIMS hydrodynamic survey

A unique VIMS hydrodynamic survey was conducted from November 1, 2005 to December 1, 2005. The purpose of the survey was to obtain a synoptic dataset of tide and representative currents for validation of the hydrodynamic model. In order that the data can be analyzed using harmonic analysis, the survey was designed to be at the least on the order of 30 days (at least 697 hours).

There are multiple measurements that were conducted depending on the site characteristics. Instruments were deployed as follows: 1) a tide gauge recording water surface elevations at 6-minute intervals at the Virginia Pilot’s Station, 2) an Acoustic Doppler Current Profiler (ADCP) outside the Inlet recording current magnitude and direction at 20-minute intervals at each of ten 0.3-m intervals in the vertical, and 3) S4 current meters located at mid-depths in each of the three Lynnhaven Branches recording velocity speed and direction, temperature, and salinity at 30-minute intervals.

The instrument locations are shown in Figure IV.4. Tide measured at the Virginia Pilot’s Station (Inlet mouth) showed a 1-hour phase lag from that at the nearby Chesapeake Bay Bridge Tunnel (CBBT) primary station, as well as a drop in amplitude to 36 cm from 38 cm at CBBT (Figure IV.5). The ADCP profiler was used because the channel has a greater depth and potentially different velocities from surface to bottom. The ADP velocity measured results outside the Inlet, as shown in Figure IV.6, and indeed showed a 2-layer circulation with a slight residual in the ebb (north) direction at the surface and in the flood (south) direction at the bottom.

Within the branches, the single S4 current meters were deployed due to their shallow depth. The time series plots show maximum currents on the order of 30 cm/sec, 40 cm/sec, and 80 cm/sec, respectively, for the Western, Eastern, and Broad Bay Branches (Figures IV.7 - IV.9). The larger velocity measured in Broad Bay was because the location of deployment was near Long Creek, where the cross section is much narrower. Otherwise, the range of velocity was typical for the coastal bays in the Chesapeake Bay. Additionally, the impacts of both a heavy rainstorm on salinity (Figures IV.7 and IV.9) and a noteworthy cold front on water temperature (Figures IV.10 and IV.11) are readily observable in this shallow water system.
Figure IV.4. Instrument Locations for VIMS Hydrodynamic Survey.

Figure IV.5. Tide at Inlet versus CBBT tide.
Figure IV.6. ADP velocity outside Inlet.

Figure IV.7. Western Branch velocity and salinity.
Figure IV.8. Eastern Branch velocity.

Figure IV.9. Broad Bay velocity and salinity.
Figure IV.10. Western Branch velocity and temperature.

Figure IV.11. Broad Bay velocity and temperature.
IV-2-2. Seasonal sediment flux measurements

Due to the shallowness of the Lynnhaven River, the sediment and water column interact. Fluxes of dissolved oxygen and inorganic nutrients between sediments and the overlying water column were measured seasonally in four selected regions of the Lynnhaven River system: Western Branch, Eastern Branch, the Inlet, and Broad Bay (Figure IV.12). Sites were selected in nearshore, shallow regions of the Lynnhaven so samples would contain actively photosynthesizing benthic microalgae (BMA), which can dominate carbon production in shallow systems, in addition to the microbial community that dominates respiration at all depths. The averaged water depth at the collection sites was about 0.4 meter at mean low water with a tidal range of 1.0 meter. Within each embayment, four sediment cores were taken during each survey. A preliminary site selection and characterization study was conducted in March 2005, with flux studies occurring in April 2005 (14°C), July 2005 (26°C), November 2005 (15°C), and May 2006 (22°C).

In the field, four sediment cores (clear acrylic, 20-cm sediment depth, 20-cm overlying water, 13.3 cm diameter) were collected from each embayment with minimal disturbance. For each core, a second small core was collected for measurement of sediment bulk density, organic content, and BMA biomass measured as chlorophyll-α in the top 1 and 3 cm of sediment. Ambient water was collected at each site for use during core incubations.

All cores were placed in a temperature and light-controlled environmental chamber at VIMS, submerged (without lids) in large mesocosms with site-specific water (Figure IV.13), and gently bubbled with air overnight to allow cores to acclimate to the experimental chamber. Two “water blank” cores per site were filled with water only to serve as controls to correct for processes occurring in the water overlying each sediment core. Temperature in the chamber was set to the average field temperature to ensure comparability.

The following morning, cores were capped with clear lids fit with magnetic stir-bars to gently circulate the water within the cores, controlled by a central motor in each mesocosm (Figure IV.13). Each lid was equipped with two ports, one for sampling and a second to allow replacement water to flow in from a reservoir with site-specific water. Care was taken to exclude bubbles while capping the cores.

Cores were incubated following the general procedure of Anderson et al. (2003), beginning in the dark for 3-4 hours to measure fluxes associated with sediment respiration. Samples were collected hourly for determination of concentrations of dissolved oxygen (DO), ammonium (NH₄⁺), nitrate + nitrite (NOₓ⁻), and phosphate (PO₄³⁻). Following the last sampling, the lights in the environmental chamber were turned on to approximately saturating levels of irradiance for BMA (417-673 μE m⁻² s⁻¹) at the
Figure IV.12. Location of core collection sites for sediment flux in the Lynnhaven River. Four cores were collected in close proximity inside the Inlet.

water surface; 165-360 μE m⁻² s⁻¹ at the sediment surface). Cores were allowed to acclimate for 30 minutes after which DO and nutrients were again sampled hourly for 3-4 hours to measure fluxes associated with BMA photosynthesis.

DO and nutrients in each reservoir of replacement water were measured at the beginning, midpoint, and end of each experiment to allow for dilution correction of the water within each core.

Dissolved oxygen and temperature were measured with an Orion galvanic DO sensor. Samples for nutrients were filtered through 0.45 μm filters (Gelman Supor) and frozen (-15°C) until later analysis on a Lachat autoanalyzer. Samples for sediment chlorophyll-α and pheophytin concentrations were frozen until extraction with 100% acetone following the methods of Pinckney and Zingmark (1994) as modified by Pinckney and Lee (2008). Concentrations were analyzed on a Shimadzu UV-1601 spectrophotometer and calculated using the equations of Lorenzen (1967). Sediment organic content was determined as the percent weight loss following combustion at 500°C for 5 hours.
Flux rates in the light and dark were computed as the time rate of change (i.e., slope) of concentration, corrected for dilution by reservoir water. To determine fluxes attributable to the sediments only, the average slope from the two water blanks at each site was subtracted from the slope of each sediment core.

Results

BMA biomass as measured by sediment chlorophyll-a concentration was higher at the Broad Bay and Inlet sites than at the Eastern and Western Branch sites (Figure IV.14), but there were no consistent seasonal trends in biomass. Approximately half of the measured BMA biomass occurred in the upper 1 cm of sediment.

Typical time courses of DO during the incubations are shown in Figure IV.15. Linear slopes were fit to the results for DO and each nutrient species and used to compute the mean net fluxes shown in Figures IV.16 through IV.22 after correcting sediment cores for the water blanks.

Net fluxes of DO were into the sediments in the dark and out of the sediments in the light, confirming the dominance of microbial respiration at night and BMA photosynthesis during the day (Figure IV.16). With the exception of the Western Branch, daytime DO production exceeded nighttime DO consumption, in many cases by a large amount, suggesting these nearshore sites were net autotrophic due to BMA primary production which likely contributes a significant fraction of total carbon fixation in the Lynnhaven.

Dark DO fluxes at each site were directly related to water temperature, with warmer temperatures leading to higher rates of respiration (Figure IV.17). Dark fluxes were not related to sediment chlorophyll, nor were chlorophyll-normalized rates related to temperature, confirming that the majority of sediment respiration was due to the bacterial community. Dark fluxes were also independent of sediment organic content, which ranged from 0.3 to 4.3% at these sites.

Taken as a whole, DO fluxes in the light were generally related to BMA biomass measured as chlorophyll-a content (Figure IV.17). Rates were not correlated to organic content or water temperature, nor were chlorophyll-normalized rates correlated to temperature. BAM photosynthetic rates were high at most sites regardless of season (Figure IV.16).

Fluxes of NH$_4^+$ were highest in the warmer months, and generally out of the sediments in the dark and into the sediments or near zero in the light (Figure IV.18). NH$_4^+$ is the product of organic matter degradation by bacteria in the sediments, which was responsible for the dark release.
Figure IV.13. Experimental design for sediment flux experiments. Four mesocosms were filled with site water, four sediment cores with overlying water, and two cores with water only to serve as controls. Core water was mixed with a central magnetic stirrer, and hourly samples withdrawn from each core were replaced by site water held in reservoirs (“replacement water”).
Uptake by BMA in the light to support photosynthetic production was enough to greatly reduce, eliminate, or completely reverse this release (Figure IV.18). Fluxes of NO\textsubscript{x} were much lower than for NH\textsubscript{4}\textsuperscript{+} and mostly centered around zero (Figure IV.19; note different scales between Figures IV.18 and IV.19). The net uptake of NO\textsubscript{x} at the Eastern Branch and Inlet sites in November 2005 was likely due to denitification. Fluxes of PO\textsubscript{4}\textsuperscript{3-}, also a by-product of organic matter degradation by bacteria, were often small and highly variable with no consistent trends (Figure IV.20).

Since NH\textsubscript{4}\textsuperscript{+} and PO\textsubscript{4}\textsuperscript{3-} remineralization and subsequent release from sediments is the result of bacterial decomposition of organic matter, rates in the dark (in the absence of BMA production) should generally be correlated to dark DO consumption (i.e., respiration), although BMA have been shown to take up nutrients in the dark to support
subsequent daytime production. While the relationships contained scatter, dark nutrient releases were generally correlated to dark DO consumption and therefore water temperature (Figure IV.21). Scatter was likely the result of dark BMA uptake and coupled nitrification-denitification. To assess the potential for the former, the rates of nutrient uptake measured in the light were compared to computed BMA demand for nutrients based on DO production rates (Figure IV.16) and molar conversions for nitrogen (9:9:1 O$_2$:C:N, F. Parker unpublished data) and phosphorus (106:106:1 O$_2$:C:P, Redfield ratios). With one exception, computed BMA nutrient demand was always greater than measured uptake in the light, suggesting a large amount of BMA demand is satisfied by uptake at night (Figure IV.22).
Figure IV.16. Net sediment-water fluxes of dissolved oxygen by site and date. Positive values reflect a release to the water; negative values indicate uptake by the sediments. Error bars denote 1 standard deviation.

Figure IV.17. Relationship of net sediment-water DO fluxes to water temperature in the dark (left) and sediment chlorophyll in the light (right).
Figure IV.18. As for Figure IV.16, but for fluxes of $\text{NH}_4^+$.

Figure IV.19. As for Figure IV.16, but for fluxes of $\text{NO}_x^-$ ($\text{NO}_2^- + \text{NO}_3^-$).
Our results confirm the importance of BMA in the Lynnhaven River, as reported for other shallow nearshore systems (e.g., Anderson et al., 2003). While sediment-water fluxes for deeper estuaries are typified by uptake of DO and release of nutrients due to respiration and subsequent remineralization, BMA have the potential to completely reverse these heterotrophic fluxes during the day due to photosynthetic biomass production. The BMA-associated biomass and sediment flux rates determined in this study should serve as useful calibration data for eutrophication and water quality modeling efforts in the Lynnhaven.

Figure IV.20. As for Figure IV.16, but for fluxes of PO₄³⁻.
Figure IV.21. Relationship of net sediment-water nutrient and oxygen fluxes in the dark.

Figure IV.22. Relationship of computed BMA nutrient demand in the light vs. computed uptake in the light. Filled symbols in the plot on the left are for NH$_4^+$ only; open circles behind the points are for NH$_4^+$ + NO$_x^-$. 
IV-2-3. Sediment critical shear stress measurements

The calculation of sediment concentration in the CE-QUAL-ICM model has a critical dependence on the determination of critical shear stress, which varies spatially and seasonally in the Lynnhaven River. For this reason, a series of surveys were conducted to measure critical shear stress in each branch in different seasons.

An initial bottom sediment mapping survey of the Lynnhaven River Basin was carried out by VIMS to characterize spatial distributions of sediment grain size, water content, etc. Based on the results of this survey, four sites were selected to represent the different environments of the bay and to characterize spatial variability. These sites are located near the Inlet entrance, in the Lower Western and Eastern Branches, and in Broad Bay. These sites were visited 3 times between autumn 2003 and autumn 2004 to conduct erosion experiments. At least two of the erosion testing sites remained fixed as index sites for characterizing seasonal variability. The other two erosion testing sites were moved to increase spatial coverage, depending on the results of the sediment mapping survey.

The sediment was characterized at 19 locations, as shown in Figure IV.23. The results of this sediment characterization survey are shown in Figure IV.24. It is readily seen that the upstream silt and clay fractions give way to the sand fraction moving toward the Inlet in any of the 3 branches.

Figure IV.23. Locations for 19 samples characterized for grain size prior to critical shear stress surveys.
Figure IV.24. Percentage distributions of sand, silt, and clay for 19 sediment samples.
Erosion tests were carried out using an existing erosion testing system, called a microcosm system, operational at Horn Point Laboratory. This Microcosm system consists of 2 10-cm Gust Microcosms (Gust and Mueller, 1997), a Campbell Datalogger connected to a laptop computer, a Fluid Metering Inc. (FMI) positive displacement pump, 2 turbidimeters, and 2 Maxon precision motors. The Microcosms use a spinning disk with central suction to generate a controllable, nearly uniform shear stress (Gust and Mueller, 1997). The Campbell Datalogger controls the pump and motor and collects and stores data.

During erosion experiments, a sequence of increasing levels of shear stress is applied to the undisturbed cores. The effluent from each Microcosm is passed through a turbidimeter and time series of turbidity are measured. The effluent is collected, filtered and weighed to determine the actual mass eroded during each step, which is used to calibrate the turbidimeter. HPL and VIMS shared the filtering responsibilities, and VIMS carried out all filter analyses. Erosion rate is subsequently calculated as the product of pumping rate and suspended sediment concentration.

There were a total of 3 critical shear stress surveys conducted in May 2005, February 2006, and August 2006. It is important to measure at different times of the season because the sediment erodibility could be affected by the activity due to bio-turbation. The locations of the erodibility core sites for all 3 surveys are shown in Figure IV.25.

Figure IV.25. Locations of erodibility core sites for all 3 critical shear stress surveys.
Critical stress profiles for all twenty-four cores that were processed from the three field erosion studies are shown in Figure IV.26. X-axis is critical shear stress in Pascals, and Y-axis is eroded mass in kilograms per square meter. The plots of the cores are color-coded so that all cores from May 2005 are green, those from February 2006 are blue, and those from August 2006 are red.

The erosion data were analyzed using the erosion formulation of Sanford and Maa (2001). This erosion formulation uses a linear erosion rate expression with depth-varying critical stress to describe both unlimited and limited erosion, with erosion behavior depending on the rate of increase in critical stress relative to the rate of change of bottom shear stress. Results from this formulation are then incorporated into the sediment transport model to represent the real in situ sediment erosion rate.

Figure IV.26. Critical stress profiles for all twenty-four cores that were run from the three field erosion studies. X-axis is critical shear stress in Pascals, and Y-axis is eroded mass in kilograms per square meter.
IV-2-4. VIMS dataflow surveys

The development of new water quality standards for turbidity, chlorophyll, and dissolved oxygen, has placed new requirements on accurate measurements of the temporal and spatial variability of water quality constituents. Detailed ecosystem modeling also requires high density spatial measurement for model calibration and validation. Until recently our capacity to measure, monitor, and evaluate water quality constituents in detail over ecologically relevant regions and time scales was limited. However, there has been recent application in Virginia of a new state-of-the-art DATAFLOW Surface Water Quality Mapping System (www.VECOS.org) for high-speed, high-resolution mapping of surface water quality from small vessels capable of sampling shoal, littoral areas. Such a mapping system has been demonstrated to have practical application in the determination of attainment of water quality criteria constituents in shallow water designated use areas. Here we have implemented these new technologies to provide information over small spatial scales to assist in the monitoring of and modeling of light attenuation, chlorophyll concentrations, surface dissolved oxygen, and other water quality conditions in the Lynnhaven River system.

DATAFLOW Mapping System

DATAFLOW is a compact, self-contained surface water quality mapping system, suitable for use in a small boat operating at speeds of up to 25 KT. The system collects water through a pipe ("ram") deployed on the transom of the vessel, pumps it through an array of water quality sensors, and then discharges the water overboard. The entire system from intake ram tube to the return hose is shielded from light to negate any effect high-intensity surface light might have on phytoplankton in the flow-through water that is being sampled. A blackened sample chamber is also used to minimize any effect of light on measurements by the fluorescence probe.

The DATAFLOW mapping system collects a sample once every 2-4 seconds. The resulting distance between samples is therefore a function of vessel speed. An average speed of 25 knots results in one observation collected every 40-60 m.

The DATAFLOW system has a YSI (Yellow Springs Instruments, Inc.) 6600 sonde equipped with a flow-through chamber. The sensors include a Clark-type 6562 DO probe, a 6561 pH probe, a 6560 conductivity/temperature probe, a 6026 turbidity probe, and a 6025 chlorophyll probe. The sonde transmits data collected from the sensors directly to a 600 MHz embedded computer board contained in a waterproof Pelican case using a data acquisition system created with LabVIEW software (National Instruments Corporation, Austin, TX). Custom software written in the LabVIEW environment provides for data acquisition, display, control, and storage. Real-time graphs and indicators provide feedback to the operator in the field, ensuring quality data is being collected. All calibrations and maintenance on the YSI 6600 sondes are completed in accordance with the YSI, Inc. operating manual methods (YSI 6-series Environmental Monitoring Systems Manual; YSI, Inc. Yellow Springs, OH). Table IV.3 provides the precision, accuracy and minimum detection limits of the sensors.
<table>
<thead>
<tr>
<th>PARAMETER</th>
<th>UNITS</th>
<th>PRECISION</th>
<th>ACCURACY</th>
<th>MDL</th>
</tr>
</thead>
<tbody>
<tr>
<td>DO</td>
<td>% Saturation</td>
<td>0.1%</td>
<td>± 2%</td>
<td>0 %</td>
</tr>
<tr>
<td>DO</td>
<td>mg/L</td>
<td>0.01mg/L</td>
<td>0.2mg/L</td>
<td>0 mg/L</td>
</tr>
<tr>
<td>Salinity</td>
<td>ppt</td>
<td>0.01ppt</td>
<td>0.1ppt</td>
<td>0 ppt</td>
</tr>
<tr>
<td>Temperature</td>
<td>ºC</td>
<td>0.01ºC</td>
<td>±0.15ºC</td>
<td>-5ºC</td>
</tr>
<tr>
<td>pH</td>
<td>unit</td>
<td>0.01units</td>
<td>±0.2units</td>
<td>0 units</td>
</tr>
<tr>
<td>Turbidity</td>
<td>NTU</td>
<td>0.1NTU</td>
<td>2 NTU</td>
<td>0 NTU</td>
</tr>
<tr>
<td>Chlorophyll</td>
<td>µg/L Chl</td>
<td>0.1µg/L Chl</td>
<td>-</td>
<td>0 µg/L Chl</td>
</tr>
</tbody>
</table>

The DATAFLOW system was equipped with a Garmin GPSMAP 168 Sounder. This unit served several functions including chart plotting, position information, and depth. The unit was WAAS (Wide Area Augmentation System) enabled and provided a position accuracy of better than three meters 95 percent of the time. The NEMA 0183 data sentence containing all pertinent position and depth information was output to the SBC data acquisition system.

The DATAFLOW system utilized a SBC data acquisition system for data collection and storage. The system was based on 600 MHz single, embedded board computer designed to run on a Windows Intel platform. All data, including latitude and longitude, was collected simultaneously in one file, removing any errors associated with merging separate files into one.

**Calibration Sampling**

A total of eight calibration stations were sampled along the cruise tracks each month. Stations were selected to maximize the range of values that are seen along a track (e.g., when moving up a tributary with a salinity gradient, samples were taken to get a high, medium, and low salinity value). Extra sampling supplies were available to sample more stations under special conditions such as in areas of large blooms. At each station the boat was stopped and water samples were collected from the effluent tubing of the DATAFLOW System (sampling water depth of approximately 0.25 - 0.5 m) for total suspended solids (TSS), volatile suspended (VSS), chlorophyll-a, chlorophyll-b, dissolved inorganic nitrogen (DIN), dissolved inorganic phosphorus (DIP), total phosphorus (TP), particulate inorganic phosphorus (PIP), and dissolved oxygen (DO) for processing with the Winkler method. At these stations secchi depth and a vertical profile of photosynthetically available radiation (PAR) were also measured. Samples for TSS, VSS, DIN, DIP, and chlorophyll were collected in darkened bottles, which were rinsed three times with ambient water before filling. Samples for DIN, DIP, chlorophyll and pheophytin were immediately filtered into sterile Whirl-Pak™ bags upon collection. These were then packed on ice and returned to the laboratory where they were stored at -20ºC. Samples were then delivered to the VIMS Analytical Service Center, Gloucester Point, VA for further processing. Additionally, at each verification station light
attenuation was measured from in situ light profiles using EPA-approved LI-COR (LI-COR Biosciences, Lincoln, NEB) underwater quantum sensors.

**Quality Assurance and Quality Control**

The quality assurance procedures followed in this project were documented in "Work/Quality Assurance Project Plan for Spatially Intensive Water Quality Monitoring (For the Period: April 1, 2004 through June 30, 2004)". This plan was submitted and approved by EPA Chesapeake Bay Program and the Virginia Department of Environmental Quality, Richmond, Virginia.

All field data were recorded on specially prepared field data sheets. The initials of the person recording the data were recorded on each data sheet. The raw data sheets were reviewed for possible missing data values due to sample collection problems prior to data entry. These sheets were filed in the VIMS laboratory. A cruise logbook was also kept.

**Results**

Dataflow mapping cruises were undertaken approximately monthly from March 2005 through November 2005 and again March 2006 through November 2006. The archived data and visualized tracks of surface temperature, salinity, dissolved oxygen, turbidity, chlorophyll and pH are available at the website [www.VECOS.org](http://www.VECOS.org). Figure IV.27 shows the typical cruise tracks with the range of turbidities recorded during the May 24, 2005 cruise, and the reaches of the of the cruise tracks that are presented as examples in subsequent figures.

Regressions of calibration station sample measurements with simultaneous DATAFLOW measurements were used to develop Lynnhaven-specific calibration of the in vivo measurements. Figure IV.28 shows the regression of the DATAFLOW turbidity measurements to downwelling light attenuation ($K_d$) profiles for all calibration stations during 2006. Light attenuation was then used to calculate light at depth using the standard Lambert-Beer relationship,

\[ I_z = I_0 \exp \left[\frac{(-K_d)Z}{Z}\right] \]  

(IV-1)

where \(I_z\) is light at depth \(Z\), \(I_0\) is light at surface, and \(K_d\) is the light attenuation coefficient.
Figure IV.27. Lynnhaven River system DATAFLOW cruise tracks showing turbidity levels during the 5-24-05 cruise. Arrows indicate the reaches that are presented in subsequent graphs in this chapter.
Figures IV.29A, B, and C show representative concentration-distance plots of turbidities (NTU) for the individual branches of the Lynnhaven system. Using the 2006 Lynnhaven system NTU to light attenuation relationship (Figure IV.28) the turbidity (~6 NTU) that is equal to 22% of surface irradiance at 1m bottom depth is provided for a reference. Typically, 22% of surface irradiance is used as a standard by EPA and the Commonwealth of Virginia to define sufficient light available for SAV sustained growth.

All three systems had comparable turbidities near the inlet of the Lynnhaven. Turbidities in the Eastern and Western Branches increased precipitously with distance upstream during July (Figures IV.29A and IV.29B) and during most other months (data not shown). Levels in Broad and Linkhorn Bays were much lower than the other two branches (Figure IV.29C). Turbidities in parts of Linkhorn Bay were lower compared to Broad Bay.

Figure IV.30 shows the spatially averaged turbidity for each of the three individual branches of the Lynnhaven system for the eight cruises in 2006. Averaged turbidities were seasonally highest in September of 2006 and highest in the Eastern Branch. Averaged turbidities in Broad and Linkhorn Bays were lower during all months than the Eastern and Western Branches.
Figure IV.29. Concentration-distance plots of turbidity along the A.) Western Branch, B.) Eastern Branch, and C.) Broad and Linkhorn Bays during July 2006. Dotted red lines indicate turbidity levels where light at 1m depth is equal to 22% of surface irradiance (SAV light criteria).
Figure IV.30. Spatially averaged turbidities (NTU) for the individual branch cruise track reaches for each monthly DATAFLOW cruise in 2006.

Figure IV.31. 2005-2006 verification station YSI chlorophyll vs. extracted chlorophyll.
All in vivo DATAFLOW chlorophyll data have been converted to extracted chlorophyll values using the 2005 and 2006 Lynnhaven system YSI chlorophyll to extracted chlorophyll relationship developed from the calibration station data (Figure IV.31).

Figures IV.32A, B, and C show representative concentration-distance plots of chlorophyll for the individual branches of the Lynnhaven system for July 2006. All three branches have low comparable chlorophyll levels in the vicinity of the inlet. In July 2006 these levels were comparable to the summertime chlorophyll standards set by the Virginia DEQ for the James River (red line). Rapid increases in chlorophyll were observed with distance upstream for the Eastern and Western Branches. There was some increases in Broad and Linkhorn Bays but during July concentrations only reached approximately 15 µg/l.

Figure IV.33 shows the spatially averaged chlorophyll concentrations for each of the three individual branches of the Lynnhaven system for the eight cruises in 2006. These data indicate that the average chlorophyll concentrations in all branches of the system exceeded the water quality standards from approximately April through September. The Eastern Branch has the highest levels followed by the Western Branch and the Broad and Linkhorn Bays.

Figures IV.34A, B, and C show representative concentration-distance plots of dissolved oxygen for the individual branches of the Lynnhaven system for July 2006. All three branches recorded high, daytime, dissolved oxygen levels that varied little from the inlet region to the upper regions of the branches. In July 2006 these levels met the summertime dissolved oxygen standards set by the Virginia DEQ for the James River (red line) of 4.3 mg/l.

Figure IV.35 shows the spatially averaged surface dissolved oxygen concentrations for each of the three individual branches of the Lynnhaven system for the eight cruises in 2006. These data indicate that the average dissolved oxygen concentrations in all branches of the system met the standards throughout the year.

**Summary**

Water quality measurements using spatially intensive water quality mapping (DATAFLOW) for the Lynnhaven system demonstrated that Broad and Linkhorn Bays had distinctly better water quality that the Western and Eastern Branches. Water quality was generally best in all regions in the vicinity of Lynnhaven Inlet and rapidly deteriorated with distance upriver in both the Western and Eastern Branches. Turbidity levels in both the Western and Eastern Branches generally exceeded that required for SAV growth to 1m while levels appeared sufficient for SAV growth in both Broad and
Figure IV.32. Concentration-distance plots of chlorophyll along the A.) Western Branch, B.) Eastern Branch, and C.) Broad and Linkhorn Bays during July 2006. Dotted red lines indicate DEQ summer chlorophyll standards for the James River of 10 µg/l.
Figure IV.33. Spatially averaged chlorophyll concentrations for the individual branch DATAFLOW cruise track reaches for each monthly cruise in 2006. Red lines indicate the Va. DEQ chlorophyll standards of 12 µg/l for March 1 - May 31 and 10µ for July 1 – September 30.

Figure IV.34. Concentration-distance plots of dissolved oxygen along the A.) Western Branch, B.) Eastern Branch, and C.) Broad and Linkhorn Bays during July 2006. Dotted red lines indicate DEQ surface dissolved oxygen standards for the James River of 4.3 mg/l.
Linkhorn Bays. These measurements agreed with the current distributions of SAV that are currently only found in Broad and Linkhorn Bays.

Chlorophyll levels were above the numeric standards in most areas except for the region near Lynnhaven Inlet from April through September. Highest concentrations occurred during July and in the upper reaches of the Western and Eastern Branches where concentrations approached 40 µg/l during July 2006. Daytime surface dissolved oxygen concentrations were generally good and met the standards throughout the system. Nighttime concentrations were not measured, but concentrations could be expected to drop significantly in the upper reaches of the Western and Eastern Branches due to the high phytoplankton biomass and other factors.
IV-2-5. VIMS high-frequency time series measurements

High frequency water quality measurements were obtained for use in model calibration, assessing water quality, and understanding the Lynnhaven ecosystem from 2005 to 2008 with a network of in situ sensors (Figure IV.36). Self-cleaning, internally-logging WET Labs ECO fluorometers ([www.wetlabs.com/products/ecfcombo/fl.htm](www.wetlabs.com/products/ecfcombo/fl.htm)) were deployed approximately 0.5 m below the surface (MLLW) to measure phytoplankton biomass as chlorophyll-a (chl-a), turbidity expressed in nephelometric turbidity units (NTU), and water temperature. Since seagrass has traditionally been found in Broad Bay ([web.vims.edu/bio/sav](web.vims.edu/bio/sav)) and is highly dependent on adequate light penetration, an additional WET Labs fluorometer capable of measuring the concentration of chromophoric dissolved organic matter (CDOM) was also deployed in Broad Bay to enable measurement of all three parameters that affect light penetration (chl-a, NTU, CDOM) in that embayment. A self-cleaning, internally-logging Hydrolab DS-5X instrument ([www.hydrolab.com/products/hydrolabds5x.asp](www.hydrolab.com/products/hydrolabds5x.asp)) was deployed approximately 0.5 m above the bottom to measure temperature, salinity, and dissolved oxygen (DO) using optical sensor technology. This instrument was deployed in the Eastern Branch in 2005 and the Western Branch in 2006.

Monitoring began in 2005 with a single fluorometer and DS-5X in the Eastern Branch (moved from the lower to upper branch part way through the summer), and both types of WET Labs fluorometers in Broad Bay (Figure IV.36, Tables IV.4 and IV.5). In 2006 new equipment acquisitions allowed us to expand into the upper and lower Eastern and Western Branches. The DS-5X was moved to the upper Western Branch to assess a second location for low DO. To assess the potential for local phytoplankton bloom formation within the Lynnhaven as opposed to advection of blooms from the lower Chesapeake Bay, a final WET Labs fluorometer was deployed at the NOAA tide station on the Chesapeake Bay Bridge-Tunnel (CBBT) fishing pier.

All sensors recorded data at 30-minute intervals and were serviced as frequently as possible (approximately every two weeks). At each servicing, water samples were collected for determination of chlorophyll-a, total suspended solids (TSS – 2006 only), and CDOM concentrations for sensor calibration, and independent measurements of DO and salinity were made with a freshly calibrated Hydrolab to provide data for sensor confirmation. Chlorophyll samples were filtered onto 0.7 µm GF/F filters and frozen until extraction with a 45/45/9.9/0.1% acetone/DMSO/distilled water/diethylamine solution for 24 hours (Shoaf and Lium, 1976) followed by analysis on a model 10-AU Turner Designs fluorometer. TSS samples were filtered onto pre-weighed 0.7 µm GF/F filters and dried to constant weight at 50°C. CDOM samples were filtered through a 0.2 µm membrane filter and frozen until analysis of absorption on a Shimadzu UV-1601 scanning spectrophotometer (Gallegos and Neale, 2002; Gallegos et al., 2005).
Absorption at 440 nm (m⁻¹) was taken as the index of CDOM concentration. Chlorophyll and NTU data from nearby Dataflow calibration stations and long-term Virginia Department of Environmental Quality monitoring stations were also used to develop sensor calibration curves. A sample calibration curve is shown in Figure IV.37. In 2005, Lynnhaven River Now personnel collected shore-based chlorophyll samples (analyzed at VIMS) at two sites (Fig. 1) for comparison of nearshore concentrations to those measured at the mid-channel *in situ* sensors. All sensor data were quality controlled via visual inspection and through use of the independent DO and salinity data to remove obviously corrupted data due to sensor fouling and malfunction.

One of the key parameters in shallow aquatic systems is the vertical attenuation coefficient of irradiance, \( k_D \), which controls the amount of light available to support both water column and benthic primary production according to Beer’s Law:

\[
I_z = I_o e^{-k_D z}
\]  

(IV-2)
Table IV.4. Sensor deployment locations (navigational markers), dates (excluding gaps), and parameters$^{1}$.

<table>
<thead>
<tr>
<th>Location</th>
<th>Dates</th>
<th>Parameters</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>2005</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>E. Branch</td>
<td>G7</td>
<td>5/5-8/12 Chl, NTU, T (surface)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>7/7-7/27 T, S, DO (bottom)</td>
</tr>
<tr>
<td>G19</td>
<td>8/17-11/15 Chl, NTU, T (surface)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>7/14-10/22 T, S, DO (bottom)</td>
<td></td>
</tr>
<tr>
<td>Broad Bay</td>
<td>17</td>
<td>5/31-9/1 Chl, NTU, T, CDOM (surface)</td>
</tr>
<tr>
<td><strong>2006</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lower W. Branch</td>
<td>G19</td>
<td>4/14-11/8 Chl, NTU, T (surface)</td>
</tr>
<tr>
<td>Upper W. Branch</td>
<td>R32$^2$</td>
<td>2/16-11/9 Chl, NTU, T (surface)</td>
</tr>
<tr>
<td></td>
<td>R32</td>
<td>5/17-9/22 T, S, DO (bottom)</td>
</tr>
<tr>
<td>Lower E. Branch</td>
<td>G7$^3$</td>
<td>4/14-11/9 Chl, NTU, T (surface)</td>
</tr>
<tr>
<td>Upper E. Branch</td>
<td>G19</td>
<td>4/14-9/20 Chl, NTU (surface)</td>
</tr>
<tr>
<td>Broad Bay</td>
<td>17</td>
<td>2/16-8/24 Chl, NTU, T (surface)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>3/9-8/11 CDOM (surface)</td>
</tr>
<tr>
<td>CBBT$^4$</td>
<td>-</td>
<td>2/16-7/6 Chl, NTU (surface)</td>
</tr>
<tr>
<td><strong>2007-08$^5$</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lower W. Branch</td>
<td>see Fig 1</td>
<td>5/17/07-3/26/08 Chl, NTU (surface)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>6/20/07-7/3/08 T, S, DO (surface)</td>
</tr>
<tr>
<td>Upper W. Branch</td>
<td>see Fig 1</td>
<td>5/17/07-7/1/08 Chl, NTU (surface)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>9/13/07-6/19/08 T, S, DO (surface)</td>
</tr>
<tr>
<td>Lower E. Branch</td>
<td>see Fig 1</td>
<td>5/17/07-7/1/08 Chl, NTU (surface)</td>
</tr>
<tr>
<td>Upper E. Branch</td>
<td>see Fig 1</td>
<td>5/17/07-6/5/08 Chl, NTU (surface)</td>
</tr>
<tr>
<td>Broad Bay</td>
<td>see Fig 1</td>
<td>5/17/07-7/1/08 Chl, NTU (surface)</td>
</tr>
<tr>
<td>Linkhorn Bay</td>
<td>see Fig 1</td>
<td>5/17/07-7/1/08 Chl, NTU (surface)</td>
</tr>
</tbody>
</table>

$^1$ Parameter abbreviations are as follows: Water temperature (T), Salinity (S), Dissolved oxygen (DO), Chlorophyll-$a$ (Chl), Turbidity (NTU), Chromophoric dissolved organic matter (CDOM).

$^2$ Sensor moved from marker G25 to R32 on 2/23/06 to get farther up the branch.

$^3$ Sensor moved to marker G5 on 6/29/06 when G7 was hit by a vessel.

$^4$ NOAA tide station on the Chesapeake Bay Bridge-Tunnel.

$^5$ Several gaps in the record exist but were excluded due to limited space.
Table IV.5. Coordinates of sensor locations.

<table>
<thead>
<tr>
<th>Location</th>
<th>Latitude</th>
<th>Longitude</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>2005-06</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lower W. Branch</td>
<td>G19 36°53’17.69”N</td>
<td>76° 6’29.66”W</td>
</tr>
<tr>
<td>Upper W. Branch</td>
<td>R32 36°52’9.23”N</td>
<td>76° 6’37.71”W</td>
</tr>
<tr>
<td></td>
<td>G25 36°52’32.64”N</td>
<td>76° 6’33.96”W</td>
</tr>
<tr>
<td>Lower E. Branch</td>
<td>G7 36°53’0.43”N</td>
<td>76° 4’16.93”W</td>
</tr>
<tr>
<td></td>
<td>G5 36°53’15.61”N</td>
<td>76° 4’29.49”W</td>
</tr>
<tr>
<td>Upper E. Branch</td>
<td>G19 36°51’57.59”N</td>
<td>76° 4’14.19”W</td>
</tr>
<tr>
<td>Broad Bay</td>
<td>17 36°53’49.53”N</td>
<td>76° 2’3.07”W</td>
</tr>
<tr>
<td>CBBT</td>
<td>- 36°58’0.68”N</td>
<td>76° 6’49.17”W</td>
</tr>
<tr>
<td><strong>2007-08</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lower W. Branch</td>
<td>dock 36°53’12.11”N</td>
<td>76° 6’11.66”W</td>
</tr>
<tr>
<td>Upper W. Branch</td>
<td>dock 36°51’43.33”N</td>
<td>76° 6’48.74”W</td>
</tr>
<tr>
<td>Lower E. Branch</td>
<td>dock 36°52’45.18”N</td>
<td>76° 4’20.64”W</td>
</tr>
<tr>
<td>Upper E. Branch</td>
<td>dock 36°52’5.46”N</td>
<td>76° 4’22.50”W</td>
</tr>
<tr>
<td>Broad Bay</td>
<td>dock 36°53’53.55”N</td>
<td>76° 2’34.10”W</td>
</tr>
<tr>
<td>Linkhorn Bay</td>
<td>dock 36°52’25.44”N</td>
<td>76° 0’45.68”W</td>
</tr>
</tbody>
</table>

in which $I_o$ and $I_z$ are incident irradiance at the surface and irradiance at depth $z$, respectively. $k_D$ is controlled by the concentrations of chlorophyll-a, turbidity (as NTU or TSS), and CDOM in the water column. To develop a simple empirical model for predicting $k_D$ as a function of these water quality parameters, data for chlorophyll, NTU, TSS, and $k_D$ measured by the DATAFLOW group at their calibration stations were combined with CDOM concentrations measured as described above at the same stations (water provided by the DATAFLOW group after each cruise) to develop a multiple linear regression. This regression for $k_D$ was then combined with the in situ sensor time series data from Broad Bay to compute the amount of light reaching the bottom as this is a key index for survival of submerged aquatic vegetation (SAV) such as eelgrass (*Zostera marina*) which has historically been present in Broad Bay.

Finally, enough funds were saved throughout the project to make possible an extra sensor deployment over an annual cycle in 2007-08 (Tables IV.4-IV.5), combined with measurements of water column primary production and respiration to complement the sediment flux data of Brush and Anderson, make possible a total metabolic budget of the
Figure IV.37. Sample calibration plot relating sensor output to measured water quality, in this case chlorophyll-a.

system, and provide critical rate process data for model calibration. WET Labs sensors were deployed on private docks throughout the Lynnhaven (Figure IV.36) and serviced approximately monthly from spring through fall and bimonthly in the winter. During each servicing trip, calibration samples were collected for measurement of chlorophyll and dissolved inorganic nutrients (0.45 μm Supor filters), temperature, salinity, and $k_D$ were measured (using Hydrolab MS5, YSI 6600V2, and Li-Cor LI-1400 and LI-192SA instrumentation), and water samples were returned to VIMS for incubation at field temperatures in 60 mL bottles in a temperature-controlled light gradient box for determination of photosynthesis-irradiance (P-I) curves. Photosynthesis and respiration were measured as the rate of change in dissolved oxygen as measured with Hach HQ40d optical DO sensors. On three trips, sediment cores were collected at each site and incubated in the dark and at saturating irradiance to obtain data from the same annual cycle for comparison to the earlier sediment flux data of Brush and Anderson. Hydrolab and/or YSI sensors were deployed 0.5 m below the surface on selected trips to collect DO data every 30 minutes for computation of metabolism using the free water method for comparison to the incubation results. This annual cycle was recently completed and data are still being analyzed.
Results

Time series data displayed high frequency variations due to tidal and diel cycles, as well as longer-term, event scale and phytoplankton bloom dynamics on the order of 1-2 weeks (Figure IV.38). Shore-based samples had similar concentrations and patterns as the mid-channel, in situ sensors, suggesting the latter were reflective of the entire embayment within which they were located (Figure IV.39).

Chlorophyll-a from 2006 showed the expected increasing trend in phytoplankton biomass from the lower to the upper estuary, with highest values in the upper Western Branch (Figure IV.40). Lowest chlorophyll concentrations occurred in Broad Bay. Chlorophyll at all locations was higher than in the lower Chesapeake Bay as measured at the CBBT. A small February bloom at the CBBT also occurred inside the Lynnhaven. The spring phytoplankton bloom in the lower bay typically occurs in April. While none was detected at the CBBT, a late April bloom was detected throughout the Lynnhaven, as were frequent blooms throughout the season. These blooms were higher than at the CBBT, and often occurred at multiple stations. The data suggest that conditions within the Lynnhaven are favorable to bloom formation, and counter an alternative hypothesis that blooms are the result of advection of high chlorophyll water from the lower Chesapeake into the system.

Bottom water hypoxia occurred in both years in the upper branches of the Lynnhaven (Figure IV.41). Values were fairly constant around 5 mg L⁻¹ on average in the Eastern Branch, with lower values being limited to the early morning hours as part of the diel cycle. In contrast, large swings in DO appeared to occur in the Western Branch. However, the sensor at this site was repeatedly and heavily fouled throughout the sampling season and appeared to be located within a thick bottom layer of detritus and macroalgae which likely resulted in the low DO. The repeated, rapid declines in DO following each servicing of the sensor and erratic changes in salinity (sensor also fouled) support this conclusion. However, the long term hypoxia from late July through early August appears to have been a real phenomenon, although it is impossible to determine if this was a lower water column event or restricted to the bottom detrital layer at this site.

Phytoplankton blooms in the Lynnhaven as measured by chlorophyll-a concentration often coincided at multiple sites around the system (Figure IV.42). In many cases chlorophyll and turbidity showed similar dynamics suggesting they were driven by the same forces (e.g. rain or wind events), while in other cases they were inversely related to one another, suggesting limitation of photosynthesis by high turbidity. Rain events should lead to runoff which would deliver sediments (thereby increasing turbidity) and nutrients which could stimulate phytoplankton blooms, while wind events would mix bottom sediments and potentially benthic microalgal chlorophyll into the water column. Blooms in 2005 often followed rain events, although the pattern in 2006 was less clear, and it is likely that internal remineralization of nutrients is also a major driver of bloom dynamics in this system.
Figure IV.38. Time series measurements from 2005 in Broad Bay.

Figure IV.39. Time series of chlorophyll-a collected at shore-based sites by Lynnhaven River Now in 2005 compared to in situ fluorometer time series deployed mid-channel at navigational markers.
Dynamics of chlorophyll and DO were linked, presumably through photosynthetic oxygen production, even though DO was measured on the bottom. DO concentrations also appeared closely related to incident irradiance, more so than chlorophyll-a, suggesting the importance of benthic microalgal production and sediment respiration in this system. CDOM and salinity also appeared closely coupled to recent rain events in 2005.
Attenuation of light in the Lynnhaven was correlated to both chlorophyll and turbidity, with the latter having the stronger correlation (Figure IV.43a-b). Attenuation did not appear to have a strong correlation with CDOM in this system (Figure IV.43c). Three different multiple regression models for predicting $k_D$ were fit to the data (Table IV.6). The first two used all three attenuating substances, one using NTU for turbidity and the other using TSS, while the third used only chlorophyll and NTU. Model fit was better when turbidity was expressed in NTU units, and inclusion of CDOM did not improve model fit. The resulting regressions reproduced measured $k_D$ well (Figure IV.43d).
Figure IV.42. Daily average values from the 2005-06 time series sensors plotted with daily irradiance from the Chesapeake Bay Virginia National Estuarine Research Reserve site on the York River (photosynthetically active radiation, PAR), average daily wind speed and total daily precipitation at the Norfolk International Airport (obtained from the NOAA National Climatic Data Center), and daily tide range at the NOAA CBBT tide station. Vertical lines connect approximately co-occurring chlorophyll blooms. Most turbidity sensors were not factory calibrated to read higher than 25 NTU.
Figure IV.43. Relationship between measured attenuation coefficient for light ($k_D$) and (a) chlorophyll-a, (b) turbidity, and (c) CDOM, and (d) confirmation of a multiple regression-based model for predicting $k_D$ as a function of these parameters. See Table IV.6 for a definition of the three regressions that were tested.

Table IV.6. Multiple linear regression models for predicting light attenuation as a function of water quality parameters.

<table>
<thead>
<tr>
<th>Model</th>
<th>Equation</th>
<th>$r^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Regress1</td>
<td>$y = 0.71 + 0.022^{<em>} \text{Chl} + 0.089^{</em>} \text{NTU} - 0.032^{*} \text{CDOM}$</td>
<td>0.94</td>
</tr>
<tr>
<td>Regress2</td>
<td>$y = 0.98 + 0.075^{<em>} \text{Chl} - 0.0013^{</em>} \text{TSS} - 0.18^{*} \text{CDOM}$</td>
<td>0.76</td>
</tr>
<tr>
<td>Regress3</td>
<td>$y = 0.71 + 0.02^{<em>} \text{Chl} + 0.09^{</em>} \text{NTU}$</td>
<td>0.94</td>
</tr>
</tbody>
</table>
The resulting regression for $k_D$ (Regress3 in Table IV.6) was combined with the 2005 and 2006 times series data from Broad Bay to estimate the average $k_D$ in the system (1.57 m$^{-1}$). Using Beer’s Law, this value translates into a depth at which 20% of surface irradiance remains of 1.02 m. The 20% light level is generally the minimum light requirement for SAV survival in the polyhaline Chesapeake (Dennison et al., 1993; Kemp et al., 2004). Using the bathymetry from Wang et al.’s hydrodynamic-water quality model, only a thin area of bottom around the shoreline of Broad Bay receives enough light to support SAV, in marked agreement with the observed long-term SAV distribution as reported by VIMS (Figure IV.44). The shoreline along the northeast quadrant of Broad Bay which appears to have enough light but no SAV historically has in fact supported ephemeral *Ruppia maritima* beds, although sediments are likely too sandy for eelgrass.

While results from the 2007-08 time series and metabolic measurements are still being analyzed, a typical P-I curve is shown in Figure IV.45. Water column production increased rapidly from negative values in the dark (i.e., net respiration) and saturated at high light levels. Data will be used to develop a metabolic budget for the entire Lynnhaven system, assess its net metabolic balance, and assess water column vs. sediment dominance of metabolism.

Figure IV.44. Calculation of potential SAV habitat in Broad Bay from (a) bathymetry and *in situ* time series sensors (red point). (b) Area of Broad Bay receiving greater than 20% of incident irradiance on average (white). (c) Long term average SAV cover in Broad Bay, 1992-2003, based on VIMS SAV monitoring program data.
Figure IV.45. Experimental setup (light gradient box) for P-I measurements in 2007-08 and a typical result (blue circles) with a statistically-fit regression (red line). Photosynthesis is expressed as net community production (NCP). Irradiance is expressed as photosynthetically active radiation (PAR).
CHAPTER V. MODEL CALIBRATION

The hydrodynamic and water quality models applied to the Lynnhaven River system were developed using the framework outlined in Chapter III. The calibration is a process by which the performance parameters are constrained by comparing with the field measured observations. For example, the bottom friction parameters were adjusted during the calibration process. A calibration assures that the model will produce results that meet or exceed some defined criteria with a specified degree of confidence. The hydrodynamic model was calibrated with observed surface elevations and velocities using historical data and VIMS hydrodynamic survey data collected in November 2005. The water quality model was calibrated using the 2006 DEQ data and validated over the years 2004 and 2005, during which period both the freshwater discharge and the non-point source loading data were provided by the HSPF watershed model developed for the Lynnhaven by URS Corporation.

V-1 Calibration of the Hydrodynamic Model

The model calibration for the Lynnhaven River used NOAA historical tide data of the late 1970s, NOAA tide prediction data at locations in both the Eastern and Western branches, and short-term velocity measurements taken in the Broad Bay branch in 2003, providing an early view of the model’s ability to reproduce the system’s hydrodynamics. However, VIMS later decided to conduct a systematic, high-frequency hydrodynamic survey, measuring water elevations inside the inlet synoptically with representative currents and salinities in each branch as well as outside of the Inlet (see Section IV-2-A for a full description of the VIMS Lynnhaven hydrodynamic survey). With these data in hand, validation then consisted of a real-time simulation of the prototype condition for the period November 1 to November 30, 2005. The validation of the hydrodynamic model is described in Chapter VI.

V-1-1 Boundary conditions

For the application of the UnTRIM hydrodynamic model to the Lynnhaven, it was necessary to specify both downstream and upstream boundary conditions. The downstream boundary conditions consisted of specifications of time series of surface elevation and salinity along the row of grid cells at the northern extent of the model grid outside of the Inlet, as shown in Figure V.1. These data were measured at the NOAA facility at the nearby Chesapeake Bay Bridge Tunnel (CBBT), and the surface elevation boundary specification was adjusted for phase by comparing the CBBT record with that from the Kiptopeke primary NOAA station on the Eastern Shore.

Of the 3 Lynnhaven branches, only the Eastern Branch extends beyond the terminus of the watershed region discussed earlier in Section III-5. Therefore, specification of the upstream boundary condition of surface elevation was based on time series of surface elevations recorded at Creeds, VA (i.e., connecting to the southeastern end of the Eastern Branch).
However, the period of measurement of surface elevation at Creeds, VA (2006) differed from the period required for calibration. In the upstream areas of the Eastern Branch, the flow direction is controlled by wind direction as well as tide. For that reason, VIMS performed a correlation between time series of the 2006 CBBT high-frequency wind and the 2006 Creeds, VA surface elevations. The results of this correlation are shown in Figure V.2. Using a relationship based on this correlation, it was then possible to generate a water surface time series specification for the upstream boundary condition of the model at Creeds, VA. An example of the estimated upstream boundary condition is shown in Figure V.3.
Figure V.2. Correlation of CBBT wind speed with Creeds, VA surface elevation.

V-1-2 External loading

There are no USGS gauges recording freshwater inflow to any of the Lynnhaven branches. For this reason, the VIMS hydrodynamic model was entirely dependent upon the URS watershed model for its freshwater discharge inputs. As discussed in Section III-5, the URS model included hourly freshwater discharge values at each catchment site along with its non-point source loadings.

V-1-3 Calibration for tidal elevation

The astronomical tide accounts for about 80% of the energy of water surface fluctuations in the Lynnhaven River system. Therefore an accurate reproduction of the tidal wave propagation in the Lynnhaven River is of the utmost importance. Furthermore, once the model is calibrated with respect to astronomical tide, a minimum of additional adjustment is required for calibrations of surface elevation and current velocity.

Preliminary testing of the UnTRIM capability to simulate the propagation of tide was performed prior to the inception of the project, and a thorough search for historical tide data in the Lynnhaven led to a set of 6 stations spanning from outside the Inlet through
Broad Bay and lastly Linkhorn Bay. The locations of these stations are shown in Figure V.4. Measurements at these 6 stations occurred in the late 1970s, but they were synoptic! Tidal propagation in an estuary is controlled by river geometry and frictional dissipation of energy. With river geometry and average tidal range at the open boundary given, we used the distribution of tidal range as a function of distance along the Broad Bay/Linkhorn Bay to calibrate against the roughness height, the model parameter for bottom friction. Figure V.5 shows the comparison of both amplitudes and phase lags of modeled and measured values of the primary tidal constituent (i.e., M₂) at Stations T2 through T6.

The top panel of Figure V.5 shows that dampening of the M₂ tidal amplitude from approximately 0.35 m at the Inlet to approximately 0.18 m at the head of Linkhorn Bay. It can be seen in Figure V.5 that the modeled vs. measured comparison of amplitude is within 2 cm at all 6 stations.

The lower panel of Figure V.5 shows a tidal phase lag of approximately 2.5 hours moving from the Inlet to the head of Linkhorn Bay. The modeled vs. measured phase difference is within a few minutes at all 6 stations.

![Figure V.3. Constructed series of 2005 surface elevations used for upstream boundary.](image-url)
Figure V.4. Locations of NOAA tide stations monitored in the Lynnhaven in the late 1970s.

Figure V.5. Comparison of modeled and measured M2 amplitudes and phases in the Broad Bay/Linkhorn Bay Branch of the Lynnhaven.
Early efforts to calibrate the tides in the Broad Bay/Linkhorn Bay Branch using the CBBT 6-minute tides as an open boundary resulted in good comparisons between prediction of the UnTRIM model and the 1977 NOAA observed tides. Real-time comparisons at Stations T2 through T6 are shown in Figure V.6 below.

Figure V.6. Real-time comparisons of UnTRIM predictions and NOAA water surface observations.
Table V.1. UnTRIM Modeled Tide Predictions versus Tide Table Predictions in Lynnhaven River Eastern and Western Branches.

<table>
<thead>
<tr>
<th>Station</th>
<th>Tide Range (m)</th>
<th>High tide phase (minutes later than Inlet)</th>
<th>Low tide phase (minutes later than Inlet)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bayville Creek (Western Br.)</td>
<td>0.518</td>
<td>59</td>
<td>97</td>
</tr>
<tr>
<td>Model Results</td>
<td>0.518</td>
<td>60</td>
<td>99</td>
</tr>
<tr>
<td>Buchanan Creek (Western Br.)</td>
<td>0.579</td>
<td>69</td>
<td>105</td>
</tr>
<tr>
<td>Model Results</td>
<td>0.578</td>
<td>63</td>
<td>115</td>
</tr>
<tr>
<td>Brown Cove (Eastern Br.)</td>
<td>0.518</td>
<td>55</td>
<td>97</td>
</tr>
<tr>
<td>Model Results</td>
<td>0.554</td>
<td>45</td>
<td>78</td>
</tr>
</tbody>
</table>

Whereas no historical data could be found in either the Western or Eastern Branches, the published NOAA Tide Tables did provide predictions at 2 locations in the Western Branch (Bayville Creek and Buchanan Creek) and 1 location in the Eastern Branch (Brown Cove) for both tidal range and phase lag from the Inlet. These predictions were compared with results from the model when driven by average tidal range with no discharge or wind specifications, and are shown in Table V.1.

V-1-4 Calibration for velocity

In conjunction with early attempts to calibrate the model for tide, 2 locations were measured for velocity in October, 2003. ADCP instruments were deployed at 2 locations bounding the Long Creek portion of Broad Bay, as shown in Figure V.7 below.

Figure V.7. Locations of Lynnhaven Velocity ADCP Stations, October 2003.
These ADCP measurements were high-frequency (measurements every 60 seconds). Whereas the deployments were of short duration (less than 2 days), they were sufficient in length to confirm the predictive capability of the UnTRIM model for velocity. The comparisons of measured and modeled velocities are shown in Figures V.8 and V.9, respectively, for Stations V1 and V2.

Figure V.8. East-west and north-south components of measured versus modeled velocity at Station V1 of Long Creek, Lynnhaven.

Figure V.9. East-west and north-south components of measured versus modeled velocity at Station V2 of Long Creek, Lynnhaven.
**V-1-5 Calibration for salinity**

In an estuary, freshwater originating from inland river sources encounters the salt water coming from the ocean to produce the longitudinal salinity gradient. The baroclinic pressure gradient generated from the fresh water at the upstream of the estuary and the salt water at the downstream then serves as the major driving force for the gravitational circulation, in which the freshwater flows seaward while the salt water flows landward. When freshwater overlays salt water, the vertical profile of salinity exhibits stratification as a result of the density difference from surface to bottom. The turbulent mixing induced by forces such as tide, wind, surface waves, internal waves and internal current shear, on the other hand, tends to homogenize property gradients in the water column both in the vertical and the horizontal direction. This turbulent activity thus counter-acts the stratification produced by the buoyancy forces.

In order to calibrate salinity predicted by the UnTRIM hydrodynamic model, comparisons between measurements and model predictions were made at all 16 VA-DEQ stations monitored every other month in the Lynnhaven River throughout calendar year 2006. The locations of these stations are shown below in Figure V.10.

![Figure V.10. Locations of Lynnhaven DEQ stations used to compare measured and modeled salinity, temperature, and water quality parameters.](image-url)
Each estuary has its own shoreline, topography, hydrology, freshwater inputs, and turbulent mixing pattern; the salinity distributions are thus different from one another. By carefully examining the salinity pattern, the characteristics of the estuary can be revealed and classified. Salinity is also an excellent natural tracer due to its conservative property. All in all, salinity is an important parameter for estuarine hydrodynamics and thus is selected to assess the performance of the estuarine hydrodynamic model. In this study, salinity time series and spatial distributions are presented from prototype measurement and compared with the model simulation results.

Measured salinity data also included those made by the VIMS dataflow surveys during this period (please note that the dataflow coverage did not extend to all 16 stations). The modeled vs. measured salinities for 2006 are shown in Figures V.11 through V.13 for comparison at DEQ stations in the Western, Eastern, and Broad Bay /Linkhorn Bay Branches, respectively. It is noted that the model predictions shown in Figures V.11 through V.13 are represented by a gray band bounded by the minimum and maximum daily predictions of salinity at each specified Lynnhaven DEQ station.

Figure V.11. UnTRIM modeled versus measured salinities at Western Branch DEQ stations for 2006. Red asterisks denote DEQ measurements and red circles denote VIMS dataflow measurements.
Figure V.12. UnTRIM modeled versus measured salinities at Eastern Branch DEQ stations for 2006. Red asterisks denote DEQ measurements and red circles denote VIMS dataflow measurements.

Figure V.13. UnTRIM modeled versus measured salinities at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006. Red asterisks denote DEQ measurements and red circles denote VIMS dataflow measurements.
V-1-6 Calibration for temperature

The modeled vs. measured water temperatures for 2006 are shown in Figures V.14 through V.16 for comparison at DEQ stations in the Western, Eastern, and Broad Bay/Linkhorn Bay Branches, respectively.

Modeling of water temperatures is an essential part of the overall water quality modeling effort due to the critical role that temperature plays in the kinetics for all other state variables. As can be seen in Figures V.14 through V.16, water temperatures in the Lynnhaven show a wide seasonal variation from about 5 degrees Celsius in the winter to approximately 25 degrees Celsius in the summer.

Figures V.14 through V.16 show excellent agreement between predicted and observed water temperatures throughout the domain, with some small discrepancies at the most headland stations (e.g., 7-THA000.76 at the head of the Western Branch and 7-LKN002.77 in the upper Linkhorn Bay).

Figure V.14. UnTRIM modeled versus measured temperatures at Western Branch DEQ stations for 2006.
Figure V.15. UnTRIM modeled versus measured temperatures at Eastern Branch DEQ stations for 2006.

Figure V.16. UnTRIM modeled versus measured temperatures at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006.
V-2 Calibration of Water Quality Model

The overall objective of the model calibration is to tune the water quality model to the observed data utilizing a set of model coefficients and parameters that are consistent with field measurements and are within the general ranges of values accepted by the modeling community as reported in the literature.

The main steps involved in the calibration of the water quality model are: the appropriate boundary condition has to be chosen, the verified external nutrient loads have to be included, the correct initial condition has to be specified, and the suitable parameter values have to be estimated.

V-2-1 Boundary condition

As was done for the salinity calibration, the water quality monitoring data from Stations CB8.1 and CB8.1E of the Chesapeake Bay Program (CBP) were used for the water quality open boundary condition (Figure V.17). The monthly water quality parameters at both the surface and bottom are available from 1984 to present. Table V.11 shows the parameters measured.

The data from CBP Stations 8.1 and 8.1E are available semi-monthly during the period from spring to fall and monthly during the winter at both the surface and bottom. The middle layers were specified from the linear interpolation between the layers which were measured. The daily values were interpolated between the measured period either semi-monthly or monthly. The present water quality model is configured such that the freshwater discharge and nutrient loadings input are specified as lateral input. The open boundary condition for the hydrodynamic model was forced by the averaged measured tide of the NOAA tidal station at the Chesapeake Bay Bridge Tunnel.

V-2-2 External loading

There is no point source input into the Lynnhaven River. The nonpoint nutrient loadings from the watershed discharged to the Lynnhaven River were obtained from the watershed model developed by URS Corporation of Virginia Beach (see Chapter III, Section III-5). Nonpoint source loads enter the water quality model through specification of the loading at model grid cells adjacent to the land. The procedure involves mapping of the hydrodynamic model grid with watershed catchment areas adjacent to the receiving waters. These nonpoint source inputs are specified at the surface of the model cell at the location of discharge. The external nutrient loads also include the atmospheric loads that are generated by the watershed model and are specified at each surface cell of the model. The time increment for loading input from the watershed model is hourly.
V-2-3 Initial condition

For an initial simulation, an initial condition was specified as the long-term averaged data measured by DEQ, interpolated spatially. Within the Lynnhaven, the initial condition for each cell was specified through linear interpolation between two adjacent DEQ stations. Since only surface water data are available, the same value was specified for each layer vertically for those cells. Outside of the Lynnhaven, the initial condition was specified based on the linear interpolation between DEQ Station 7-LYN000.03 and CBP Station CB8.1. Upon attaining dynamic equilibrium, the values of all computed model cell output from prior model results were used to specify a suitable initial condition.

V-2-4 Estimation of parameters

Most of the parameters in the CE-QUAL-ICM water quality model were adopted from the default parameters for the Chesapeake Bay (Cerco and Cole, 1994). The parameters used in the water column of this study are listed in Tables V.4 to V.9. The modification of parameters depended on the comparison with measured data or unique features of the Lynnhaven. The remaining parameters used in the sediment flux are listed in Table V.10.
Table V.2. Model state variables in the eutrophication water quality model

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Symbol</th>
</tr>
</thead>
<tbody>
<tr>
<td>Temperature</td>
<td>T</td>
</tr>
<tr>
<td>Salinity</td>
<td>S</td>
</tr>
<tr>
<td>Total Suspended Solids</td>
<td>TSS</td>
</tr>
<tr>
<td>Cyanobacteria</td>
<td>Bc</td>
</tr>
<tr>
<td>Diatoms</td>
<td>Bd</td>
</tr>
<tr>
<td>Green Algae</td>
<td>Bg</td>
</tr>
<tr>
<td>Refractory Particulate Organic Carbon</td>
<td>RPOC</td>
</tr>
<tr>
<td>Labile Particulate Organic Carbon</td>
<td>LPOC</td>
</tr>
<tr>
<td>Dissolved Organic Carbon</td>
<td>DOC</td>
</tr>
<tr>
<td>Refractory Particulate Organic Nitrogen</td>
<td>RPON</td>
</tr>
<tr>
<td>Labile Particulate Organic Nitrogen</td>
<td>LPON</td>
</tr>
<tr>
<td>Dissolved Organic Nitrogen</td>
<td>DON</td>
</tr>
<tr>
<td>Ammonium Nitrogen</td>
<td>NH₄</td>
</tr>
<tr>
<td>Nitrate+nitrite Nitrogen</td>
<td>NO₃</td>
</tr>
<tr>
<td>Refractory Particulate Organic Phosphorus</td>
<td>RPOP</td>
</tr>
<tr>
<td>Labile Particulate Organic Phosphorus</td>
<td>LPOP</td>
</tr>
<tr>
<td>Dissolved Organic Phosphorus</td>
<td>DOP</td>
</tr>
<tr>
<td>Total Phosphate</td>
<td>PO₄t</td>
</tr>
<tr>
<td>Particulate Biogenic Silica</td>
<td>SU</td>
</tr>
<tr>
<td>Available Silica</td>
<td>SA</td>
</tr>
<tr>
<td>Chemical Oxygen Demand</td>
<td>COD</td>
</tr>
<tr>
<td>Dissolved Oxygen</td>
<td>DO</td>
</tr>
</tbody>
</table>

Table V.3. Model state variables and fluxes in the benthic sediment flux model

Parameters

- particulate organic carbon in Layer 2 (G₁, G₂, and G₃ classes)
- particulate organic nitrogen in Layer 2 (G₁, G₂, and G₃ classes)
- particulate organic phosphorus in Layer 2 (G₁, G₂, and G₃ classes)
- particulate biogenic silica in Layer 2
- sulfide (salt water) or methane (fresh water) in Layers 1 and 2
- ammonium nitrogen in Layers 1 and 2
- nitrate nitrogen in Layers 1 and 2
- phosphate phosphorus in Layers 1 and 2
- available silica in Layers 1 and 2
- ammonium nitrogen flux
- nitrate nitrogen flux
- phosphate flux
- silica flux
- sediment oxygen demand
- release of chemical oxygen demand
- sediment temperature
- benthic microalgae
Table V.4. Parameters related to algae in the water column

<table>
<thead>
<tr>
<th>parameter</th>
<th>description</th>
<th>value</th>
<th>units</th>
</tr>
</thead>
<tbody>
<tr>
<td>PMc</td>
<td>maximum growth rate of algae group 1</td>
<td>250</td>
<td>g C g(^{-1}) Chl d(^{-1})</td>
</tr>
<tr>
<td>PMd</td>
<td>maximum growth rate of algae group 2</td>
<td>300</td>
<td>g C g(^{-1}) Chl d(^{-1})</td>
</tr>
<tr>
<td>PMg</td>
<td>maximum growth rate of algae group 3</td>
<td>300</td>
<td>g C g(^{-1}) Chl d(^{-1})</td>
</tr>
<tr>
<td>KHNx</td>
<td>half-saturation constant of N uptake by algae</td>
<td>0.01</td>
<td>g N m(^{-3})</td>
</tr>
<tr>
<td>KHPx</td>
<td>half-saturation constant of P uptake by algae</td>
<td>0.001</td>
<td>g P m(^{-3})</td>
</tr>
<tr>
<td>KHS</td>
<td>half-saturation constant of Si uptake by diatoms</td>
<td>0.05</td>
<td>g Si m(^{-3})</td>
</tr>
<tr>
<td>KHRx</td>
<td>half-saturation constant of DO for algal excretion of DOC</td>
<td>0.5</td>
<td>g O(_2) m(^{-3})</td>
</tr>
<tr>
<td>(\alpha_c)</td>
<td>initial slope of production vs. irradiance relationship for algal group 1</td>
<td>8</td>
<td>g C g(^{-1}) Chl (E m(^{-2}))(^{-1})</td>
</tr>
<tr>
<td>(\alpha_d)</td>
<td>initial slope of production vs. irradiance relationship for algal group 2</td>
<td>8</td>
<td>g C g(^{-1}) Chl (E m(^{-2}))(^{-1})</td>
</tr>
<tr>
<td>(\alpha_g)</td>
<td>initial slope of production vs. irradiance relationship for algal group 3</td>
<td>8</td>
<td>g C g(^{-1}) Chl (E m(^{-2}))(^{-1})</td>
</tr>
<tr>
<td>a(_1)</td>
<td>background light attenuation coefficient</td>
<td>0.735</td>
<td>m(^{-1})</td>
</tr>
<tr>
<td>a(_2)</td>
<td>light attenuation coefficient due to total suspended solid</td>
<td>0.018</td>
<td>m(^{2}) per g TSS</td>
</tr>
<tr>
<td>a(_3)</td>
<td>light attenuation coefficient due to algae</td>
<td>0.06</td>
<td>m(^{2}) per mg CHL</td>
</tr>
<tr>
<td>CCHL(_x)</td>
<td>C-to-CHL ratio in algae</td>
<td>60.0</td>
<td>g C per g CHL</td>
</tr>
<tr>
<td>TMc</td>
<td>optimum T for algal group 1 growth</td>
<td>29.0</td>
<td>°C</td>
</tr>
<tr>
<td>TMd</td>
<td>optimum T for algal group 2 growth</td>
<td>16.0</td>
<td>°C</td>
</tr>
<tr>
<td>TMg</td>
<td>optimum T for algal group 3 growth</td>
<td>25.0</td>
<td>°C</td>
</tr>
<tr>
<td>KTG(_{1c})</td>
<td>effect of T below optimum T on algal Group 1 growth</td>
<td>0.006</td>
<td>°C(^{-2})</td>
</tr>
<tr>
<td>KTG(_{2c})</td>
<td>effect of T above optimum T on algal Group 1 growth</td>
<td>0.006</td>
<td>°C(^{-2})</td>
</tr>
<tr>
<td>KTG(_{1d})</td>
<td>effect of T below optimum T on algal Group 2 growth</td>
<td>0.004</td>
<td>°C(^{-2})</td>
</tr>
<tr>
<td>KTG(_{2d})</td>
<td>effect of T above optimum T on algal Group 2 growth</td>
<td>0.006</td>
<td>°C(^{-2})</td>
</tr>
<tr>
<td>KTG(_{1g})</td>
<td>effect of T below optimum T on algal Group 3 growth</td>
<td>0.012</td>
<td>°C(^{-2})</td>
</tr>
<tr>
<td>KTG(_{2g})</td>
<td>effect of T above optimum T on algal Group 3 growth</td>
<td>0.007</td>
<td>°C(^{-2})</td>
</tr>
<tr>
<td>BMR(_c)</td>
<td>basal metabolism rate of algae group 1 at reference T</td>
<td>0.02</td>
<td>day(^{-1})</td>
</tr>
<tr>
<td>BMR(_d)</td>
<td>basal metabolism rate of algae group 2 at reference T</td>
<td>0.04</td>
<td>day(^{-1})</td>
</tr>
<tr>
<td>BMR(_g)</td>
<td>basal metabolism rate of algae group 3 at reference T</td>
<td>0.02</td>
<td>day(^{-1})</td>
</tr>
<tr>
<td>PRR(_c)</td>
<td>predation rate of algae group 1 at reference T</td>
<td>0.02</td>
<td>day(^{-1})</td>
</tr>
<tr>
<td>PRR(_d)</td>
<td>predation rate of algae group 2 at reference T</td>
<td>0.15</td>
<td>day(^{-1})</td>
</tr>
<tr>
<td>PRR(_g)</td>
<td>predation rate of algae group 3 at reference T</td>
<td>0.25</td>
<td>day(^{-1})</td>
</tr>
<tr>
<td>KTBx</td>
<td>effect of T on basal metabolism of algae</td>
<td>0.069</td>
<td>°C(^{-1})</td>
</tr>
<tr>
<td>TR(_x)</td>
<td>reference T for basal metabolism of algae</td>
<td>20.0</td>
<td>°C</td>
</tr>
<tr>
<td>WS(_{c})</td>
<td>settling velocity for algal group 1</td>
<td>0.1</td>
<td>m day(^{-1})</td>
</tr>
</tbody>
</table>
Table V.4 (cont’d)

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Description</th>
<th>Value</th>
<th>Units</th>
</tr>
</thead>
<tbody>
<tr>
<td>$WS_d$</td>
<td>settling velocity for algal group 2</td>
<td>0.2</td>
<td>m day$^{-1}$</td>
</tr>
<tr>
<td>$WS_g$</td>
<td>settling velocity for algal group 3</td>
<td>0.1</td>
<td>m day$^{-1}$</td>
</tr>
</tbody>
</table>

Table V.5. Parameters related to organic carbon in the water column

<table>
<thead>
<tr>
<th>Parameters</th>
<th>Description</th>
<th>Value</th>
<th>Units</th>
</tr>
</thead>
<tbody>
<tr>
<td>FCRP</td>
<td>fraction of predated algal C produced as RPOC</td>
<td>0.20</td>
<td>none</td>
</tr>
<tr>
<td>FCLP</td>
<td>fraction of predated algal C produced as LPOC</td>
<td>0.65</td>
<td>none</td>
</tr>
<tr>
<td>FCDP</td>
<td>fraction of predated algal C produced as DOC</td>
<td>0.15</td>
<td>none</td>
</tr>
<tr>
<td>FCDx</td>
<td>fraction of metabolized C by algae produced as DOC</td>
<td>0.00</td>
<td>none</td>
</tr>
<tr>
<td>KHRx</td>
<td>half-saturation constant of DO for algal excretion of DOC</td>
<td>0.5</td>
<td>g O$_2$ m$^{-3}$</td>
</tr>
<tr>
<td>KHO$_{DOC}$</td>
<td>half-saturation constant of DO for oxic respiration of DOC</td>
<td>0.5</td>
<td>g O$_2$ m$^{-3}$</td>
</tr>
<tr>
<td>$K_{RC}$</td>
<td>minimum respiration rate of RPOC</td>
<td>0.005</td>
<td>day$^{-1}$</td>
</tr>
<tr>
<td>$K_{LC}$</td>
<td>minimum respiration rate of LPOC</td>
<td>0.075</td>
<td>day$^{-1}$</td>
</tr>
<tr>
<td>$K_{DC}$</td>
<td>minimum respiration rate of DOC</td>
<td>0.020</td>
<td>day$^{-1}$</td>
</tr>
<tr>
<td>$K_{Rcalg}$</td>
<td>constant relating respiration of RPOC to algal biomass</td>
<td>0.0</td>
<td>day$^{-1}$ per g C m$^{-3}$</td>
</tr>
<tr>
<td>$K_{Lcalg}$</td>
<td>constant relating respiration of LPOC to algal biomass</td>
<td>0.0</td>
<td>day$^{-1}$ per g C m$^{-3}$</td>
</tr>
<tr>
<td>$K_{Dcalg}$</td>
<td>constant relating respiration of DOC to algal biomass</td>
<td>0.0</td>
<td>day$^{-1}$ per g C m$^{-3}$</td>
</tr>
<tr>
<td>$KT_{HDR}$</td>
<td>effect of T on hydrolysis/mineralization of POM/DOM</td>
<td>0.069</td>
<td>°C$^{-1}$</td>
</tr>
<tr>
<td>$KT_{MNL}$</td>
<td>effect of T on hydrolysis/mineralization of POM/DOM</td>
<td>0.069</td>
<td>°C$^{-1}$</td>
</tr>
<tr>
<td>$TR_{HDR}$</td>
<td>reference T for hydrolysis of POM</td>
<td>20.0</td>
<td>°C</td>
</tr>
<tr>
<td>$TR_{MNL}$</td>
<td>reference T for mineralization of DOM</td>
<td>20.0</td>
<td>°C</td>
</tr>
<tr>
<td>KHNDN$_N$</td>
<td>half-saturation constant of NO$_{23}$ for denitrification</td>
<td>0.1</td>
<td>g N m$^{-3}$</td>
</tr>
<tr>
<td>AANOX</td>
<td>ratio of denitrification to oxic DOC respiration rate</td>
<td>0.5</td>
<td>none</td>
</tr>
<tr>
<td>Parameters</td>
<td>description</td>
<td>value</td>
<td>units</td>
</tr>
<tr>
<td>------------</td>
<td>-------------</td>
<td>-------</td>
<td>-------</td>
</tr>
<tr>
<td>FNRP</td>
<td>fraction of predated algal N produced as RPON</td>
<td>0.15</td>
<td>none</td>
</tr>
<tr>
<td>FNLP</td>
<td>fraction of predated algal N produced as LPON</td>
<td>0.25</td>
<td>none</td>
</tr>
<tr>
<td>FNDP</td>
<td>fraction of predated algal N produced as DON</td>
<td>0.20</td>
<td>none</td>
</tr>
<tr>
<td>FNIP</td>
<td>fraction of predated algal N produced as NH$_4$</td>
<td>0.40</td>
<td>none</td>
</tr>
<tr>
<td>FNR</td>
<td>fraction of metabolized algal N produced as RPON</td>
<td>0.05</td>
<td>none</td>
</tr>
<tr>
<td>FNL</td>
<td>fraction of metabolized algal N produced as LPON</td>
<td>0.20</td>
<td>none</td>
</tr>
<tr>
<td>FND</td>
<td>fraction of metabolized algal N produced as DON</td>
<td>0.20</td>
<td>none</td>
</tr>
<tr>
<td>FNI</td>
<td>fraction of metabolized algal N produced as NH$_4$</td>
<td>0.55</td>
<td>none</td>
</tr>
<tr>
<td>ANC$_{\text{min}}$</td>
<td>minimum N-to-C ratio in algae</td>
<td>0.135</td>
<td>g N per g C</td>
</tr>
<tr>
<td>ANC$_{\text{max}}$</td>
<td>maximum N-to-C ratio in algae</td>
<td>0.20</td>
<td>g N per g C</td>
</tr>
<tr>
<td>ANDC</td>
<td>mass of NO$_{23}$-N consumed per mass DOC oxidized</td>
<td>0.933</td>
<td>g N per g C</td>
</tr>
<tr>
<td>K$_{RN}$</td>
<td>minimum hydrolysis/mineralization rate of RPON</td>
<td>0.005</td>
<td>day$^{-1}$</td>
</tr>
<tr>
<td>K$_{LN}$</td>
<td>minimum hydrolysis/mineralization rate of LPON</td>
<td>0.075</td>
<td>day$^{-1}$</td>
</tr>
<tr>
<td>K$_{DN}$</td>
<td>minimum hydrolysis/mineralization rate of DON</td>
<td>0.015</td>
<td>day$^{-1}$</td>
</tr>
<tr>
<td>K$_{Rnalg}$</td>
<td>constant relating hydrolysis/mineralization of RPON to algal biomass</td>
<td>0.0</td>
<td>day$^{-1}$ per g N m$^{-3}$</td>
</tr>
<tr>
<td>K$_{Lnalg}$</td>
<td>constant relating hydrolysis/mineralization of LPON to algal biomass</td>
<td>0.0</td>
<td>day$^{-1}$ per g N m$^{-3}$</td>
</tr>
<tr>
<td>K$_{Dnalg}$</td>
<td>constant relating hydrolysis/mineralization of DON to algal biomass</td>
<td>0.0</td>
<td>day$^{-1}$ per g N m$^{-3}$</td>
</tr>
<tr>
<td>KHDO$_{\text{NIT}}$</td>
<td>half-saturation constant of DO for nitrification</td>
<td>1.0</td>
<td>g O$_2$ m$^{-3}$</td>
</tr>
<tr>
<td>KHN$_{\text{NIT}}$</td>
<td>half-saturation constant of NH$_4$ for nitrification</td>
<td>1.0</td>
<td>g N m$^{-3}$</td>
</tr>
<tr>
<td>NT$_M$</td>
<td>maximum nitrification at optimum T</td>
<td>0.007</td>
<td>day$^{-1}$</td>
</tr>
<tr>
<td>KT$_{NT1}$</td>
<td>effect of T below optimum T on nitrification rate</td>
<td>0.0045</td>
<td>°C$^{-2}$</td>
</tr>
<tr>
<td>KT$_{NT1}$</td>
<td>effect of T above optimum T on nitrification rate</td>
<td>0.0045</td>
<td>°C$^{-2}$</td>
</tr>
<tr>
<td>TM$_{NT}$</td>
<td>optimum T for nitrification rate</td>
<td>27.0</td>
<td>°C</td>
</tr>
</tbody>
</table>
Table V.7. Parameters related to phosphorus in the water column

<table>
<thead>
<tr>
<th>Parameter</th>
<th>description</th>
<th>value</th>
<th>units</th>
</tr>
</thead>
<tbody>
<tr>
<td>FPRP</td>
<td>fraction of predated algal P produced as RPOP</td>
<td>0.03</td>
<td>none</td>
</tr>
<tr>
<td>FPLP</td>
<td>fraction of predated algal P produced as LPOP</td>
<td>0.07</td>
<td>none</td>
</tr>
<tr>
<td>FPDP</td>
<td>fraction of predated algal P produced as DOP</td>
<td>0.40</td>
<td>none</td>
</tr>
<tr>
<td>FPIP</td>
<td>fraction of predated algal P produced as DIP</td>
<td>0.50</td>
<td>none</td>
</tr>
<tr>
<td>FPRx</td>
<td>fraction of metabolized P by algae produced as RPOP</td>
<td>0.0</td>
<td>none</td>
</tr>
<tr>
<td>FPLx</td>
<td>fraction of metabolized P by algae produced as LPOP</td>
<td>0.0</td>
<td>none</td>
</tr>
<tr>
<td>FPDx</td>
<td>fraction of metabolized P by algae produced DOP</td>
<td>0.25</td>
<td>none</td>
</tr>
<tr>
<td>FPIx</td>
<td>fraction of metabolized P by algae produced DOP</td>
<td>0.75</td>
<td>none</td>
</tr>
<tr>
<td>APCMIN</td>
<td>minimum P-to-C ratio in algae</td>
<td>0.0125</td>
<td>g P per g C</td>
</tr>
<tr>
<td>APCMAX</td>
<td>maximum P-to-C ratio in algae</td>
<td>0.0175</td>
<td>g P per g C</td>
</tr>
<tr>
<td>PO4DMAX</td>
<td>maximum PO4d beyond which APC = APCMAX</td>
<td>0.01</td>
<td>g P m⁻³</td>
</tr>
<tr>
<td>KRP</td>
<td>minimum hydrolysis/mineralization rate of RPOP</td>
<td>0.005</td>
<td>day⁻¹</td>
</tr>
<tr>
<td>KLP</td>
<td>minimum hydrolysis/mineralization rate of LPOP</td>
<td>0.075</td>
<td>day⁻¹</td>
</tr>
<tr>
<td>KDP</td>
<td>minimum hydrolysis/mineralization rate of DOP</td>
<td>0.1</td>
<td>day⁻¹</td>
</tr>
<tr>
<td>KRpalg</td>
<td>constant relating hydrolysis/mineralization of RPOP to algal biomass</td>
<td>0.0</td>
<td>day⁻¹ per g P m⁻³</td>
</tr>
<tr>
<td>KLpalg</td>
<td>constant relating hydrolysis/mineralization of LPOP to algal biomass</td>
<td>0.0</td>
<td>day⁻¹ per g P m⁻³</td>
</tr>
<tr>
<td>KDpalg</td>
<td>constant relating hydrolysis/mineralization of DOP to algal biomass</td>
<td>0.0</td>
<td>day⁻¹ per g P m⁻³</td>
</tr>
</tbody>
</table>

Table V.8. Parameters related to silica in the water column

<table>
<thead>
<tr>
<th>Parameter</th>
<th>description</th>
<th>value</th>
<th>units</th>
</tr>
</thead>
<tbody>
<tr>
<td>FSA</td>
<td>fraction of predated diatom Si as SA</td>
<td>0.0</td>
<td>none</td>
</tr>
<tr>
<td>ASCd</td>
<td>Si-to-C ratio in diatoms</td>
<td>0.5</td>
<td>g Si per g C</td>
</tr>
<tr>
<td>KSU</td>
<td>dissolution rate of SU at reference T</td>
<td>0.025</td>
<td>day⁻¹</td>
</tr>
<tr>
<td>KT SUA</td>
<td>effect of T on dissolution of SU</td>
<td>0.092</td>
<td>°C⁻¹</td>
</tr>
<tr>
<td>TR SUA</td>
<td>reference T for dissolution of SU</td>
<td>20.0</td>
<td>°C</td>
</tr>
</tbody>
</table>
Table V.9. Parameters related to chemical oxygen demand and dissolved oxygen in the water column

<table>
<thead>
<tr>
<th>Parameters</th>
<th>description</th>
<th>value</th>
<th>units</th>
</tr>
</thead>
<tbody>
<tr>
<td>$K_{HOCOD}$</td>
<td>half-saturation constant of DO for oxidation of COD</td>
<td>1.5</td>
<td>g O$_2$ m$^{-3}$</td>
</tr>
<tr>
<td>$K_{CD}$</td>
<td>oxidation rate of COD at reference temperature</td>
<td>20.0</td>
<td>day$^{-1}$</td>
</tr>
<tr>
<td>$K_{TCOD}$</td>
<td>effect of T on oxidation of COD</td>
<td>0.041</td>
<td>°C$^{-1}$</td>
</tr>
<tr>
<td>$T_{RCOD}$</td>
<td>reference T for oxidation of COD</td>
<td>20.0</td>
<td>°C</td>
</tr>
<tr>
<td>$K_{RDO}$</td>
<td>reaeration coefficient</td>
<td>2.4</td>
<td>m day$^{-1}$</td>
</tr>
<tr>
<td>AOCR</td>
<td>mass DO consumed per mass C resired by algae</td>
<td>2.67</td>
<td>g O$_2$ per g C</td>
</tr>
<tr>
<td>AONT</td>
<td>mass DO consumed per mass $NH_4$-N nitrified</td>
<td>4.33</td>
<td>g O$_2$ per g N</td>
</tr>
</tbody>
</table>

Table V.10. Parameters used in the sediment flux model

<table>
<thead>
<tr>
<th>parameter</th>
<th>description</th>
<th>value</th>
<th>units</th>
</tr>
</thead>
<tbody>
<tr>
<td>HSEDALL</td>
<td>depth of sediment</td>
<td>10</td>
<td>cm</td>
</tr>
<tr>
<td>DIFFT</td>
<td>heat diffusion coefficient between water column and sediment</td>
<td>0.0018</td>
<td>cm$^2$ sec$^{-1}$</td>
</tr>
<tr>
<td>SALTSW</td>
<td>salinity for dividing fresh and saltwater for SOD kinetics (sulfide in saltwater or methane in freshwater) and for PO$_4$ sorption coefficients</td>
<td>1.0</td>
<td>ppt</td>
</tr>
<tr>
<td>SALTND</td>
<td>salinity for dividing fresh or saltwater for nitrification/denitrification rates (larger values for freshwater)</td>
<td>1.0</td>
<td>ppt</td>
</tr>
<tr>
<td>FRPPH1(1)</td>
<td>fraction of POP in algal group No. 1 routed into G$_1$ class</td>
<td>0.65</td>
<td>none</td>
</tr>
<tr>
<td>FRPPH1(2)</td>
<td>fraction of POP in algal group No. 1 routed into G$_2$ class</td>
<td>0.255</td>
<td>none</td>
</tr>
<tr>
<td>FRPPH1(3)</td>
<td>fraction of POP in algal group No. 1 routed into G$_3$ class</td>
<td>0.095</td>
<td>none</td>
</tr>
<tr>
<td>FRPPH2(1)</td>
<td>fraction of POP in algal group No. 2 routed into G$_1$ class</td>
<td>0.65</td>
<td>none</td>
</tr>
<tr>
<td>FRPPH2(2)</td>
<td>fraction of POP in algal group No. 2 routed into G$_2$ class</td>
<td>0.255</td>
<td>none</td>
</tr>
<tr>
<td>FRPPH2(3)</td>
<td>fraction of POP in algal group No. 2 routed into G$_3$ class</td>
<td>0.095</td>
<td>none</td>
</tr>
<tr>
<td>FRPPH3(1)</td>
<td>fraction of POP in algal group No. 3 routed into G$_1$ class</td>
<td>0.65</td>
<td>none</td>
</tr>
<tr>
<td>Variable</td>
<td>Description</td>
<td>Value</td>
<td></td>
</tr>
<tr>
<td>------------</td>
<td>--------------------------------------------------</td>
<td>--------</td>
<td></td>
</tr>
<tr>
<td>FRPPH3(2)</td>
<td>fraction of POP in algal group No. 3 routed into G2 class</td>
<td>0.255</td>
<td></td>
</tr>
<tr>
<td>FRPPH3(3)</td>
<td>fraction of POP in algal group No. 3 routed into G3 class</td>
<td>0.095</td>
<td></td>
</tr>
<tr>
<td>FRNPH1(1)</td>
<td>fraction of PON in algal group No. 1 routed into G1 class</td>
<td>0.65</td>
<td></td>
</tr>
<tr>
<td>FRNPH1(2)</td>
<td>fraction of PON in algal group No. 1 routed into G2 class</td>
<td>0.28</td>
<td></td>
</tr>
<tr>
<td>FRNPH1(3)</td>
<td>fraction of PON in algal group No. 1 routed into G3 class</td>
<td>0.07</td>
<td></td>
</tr>
<tr>
<td>FRNPH2(1)</td>
<td>fraction of PON in algal group No. 2 routed into G1 class</td>
<td>0.65</td>
<td></td>
</tr>
<tr>
<td>FRNPH2(2)</td>
<td>fraction of PON in algal group No. 2 routed into G2 class</td>
<td>0.28</td>
<td></td>
</tr>
<tr>
<td>FRNPH2(3)</td>
<td>fraction of PON in algal group No. 2 routed into G3 class</td>
<td>0.07</td>
<td></td>
</tr>
<tr>
<td>FRNPH3(1)</td>
<td>fraction of PON in algal group No. 3 routed into G1 class</td>
<td>0.65</td>
<td></td>
</tr>
<tr>
<td>FRNPH3(2)</td>
<td>fraction of PON in algal group No. 3 routed into G2 class</td>
<td>0.28</td>
<td></td>
</tr>
<tr>
<td>FRNPH3(3)</td>
<td>fraction of PON in algal group No. 3 routed into G3 class</td>
<td>0.07</td>
<td></td>
</tr>
<tr>
<td>FRCPH1(1)</td>
<td>fraction of POC in algal group No. 1 routed into G1 class</td>
<td>0.65</td>
<td></td>
</tr>
<tr>
<td>FRCPH1(2)</td>
<td>fraction of POC in algal group No. 1 routed into G2 class</td>
<td>0.255</td>
<td></td>
</tr>
<tr>
<td>FRCPH1(3)</td>
<td>fraction of POC in algal group No. 1 routed into G3 class</td>
<td>0.095</td>
<td></td>
</tr>
<tr>
<td>FRCPH2(1)</td>
<td>fraction of POC in algal group No. 2 routed into G1 class</td>
<td>0.65</td>
<td></td>
</tr>
<tr>
<td>FRCPH2(2)</td>
<td>fraction of POC in algal group No. 2 routed into G2 class</td>
<td>0.255</td>
<td></td>
</tr>
<tr>
<td>FRCPH2(3)</td>
<td>fraction of POC in algal group No. 2 routed into G3 class</td>
<td>0.095</td>
<td></td>
</tr>
<tr>
<td>FRCPH3(1)</td>
<td>fraction of POC in algal group No. 3 routed into G1 class</td>
<td>0.65</td>
<td></td>
</tr>
<tr>
<td>FRCPH3(2)</td>
<td>fraction of POC in algal group No. 3 routed into G2 class</td>
<td>0.255</td>
<td></td>
</tr>
<tr>
<td>FRCPH3(3)</td>
<td>fraction of POC in algal group No. 3 routed into G3 class</td>
<td>0.095</td>
<td></td>
</tr>
<tr>
<td>KPDIAG(1)</td>
<td>reaction (decay) rates for G1 class POP at 20°C</td>
<td>0.035  day⁻¹</td>
<td></td>
</tr>
<tr>
<td>KPDIAG(2)</td>
<td>reaction (decay) rates for G2 class POP at 20°C</td>
<td>0.0018 day⁻¹</td>
<td></td>
</tr>
<tr>
<td>KPDIAG(3)</td>
<td>reaction (decay) rates for G3 class POP at 20°C</td>
<td>0.0    day⁻¹</td>
<td></td>
</tr>
<tr>
<td>DPTHTA(1)</td>
<td>constant for T adjustment for G1 class POP decay</td>
<td>1.10</td>
<td></td>
</tr>
<tr>
<td>DPTHTA(2)</td>
<td>constant for T adjustment for G2</td>
<td>none</td>
<td></td>
</tr>
<tr>
<td>Parameter</td>
<td>Value</td>
<td></td>
<td></td>
</tr>
<tr>
<td>--------------------</td>
<td>--------------------------------</td>
<td></td>
<td></td>
</tr>
<tr>
<td>POP decay</td>
<td>1.15 day⁻¹</td>
<td></td>
<td></td>
</tr>
<tr>
<td>KNDIA (1)</td>
<td>reaction (decay) rates for G₁</td>
<td></td>
<td></td>
</tr>
<tr>
<td>POP at 20°C</td>
<td>0.035 day⁻¹</td>
<td></td>
<td></td>
</tr>
<tr>
<td>KNDIA (2)</td>
<td>reaction (decay) rates for G₂</td>
<td></td>
<td></td>
</tr>
<tr>
<td>POP at 20°C</td>
<td>0.0018 day⁻¹</td>
<td></td>
<td></td>
</tr>
<tr>
<td>KNDIA (3)</td>
<td>reaction (decay) rates for G₃</td>
<td></td>
<td></td>
</tr>
<tr>
<td>POP at 20°C</td>
<td>0.0 day⁻¹</td>
<td></td>
<td></td>
</tr>
<tr>
<td>DNTHTA (1)</td>
<td>constant for T adjustment for</td>
<td></td>
<td></td>
</tr>
<tr>
<td>POP decay</td>
<td>G₁ class PON decay</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1.10 none</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>DNTHTA (2)</td>
<td>constant for T adjustment for</td>
<td></td>
<td></td>
</tr>
<tr>
<td>POP decay</td>
<td>G₂ class PON decay</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1.15 none</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>KCDIA (1)</td>
<td>reaction (decay) rates for G₁</td>
<td></td>
<td></td>
</tr>
<tr>
<td>POC at 20°C</td>
<td>0.035 (day⁻¹)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>KCDIA (2)</td>
<td>reaction (decay) rates for G₂</td>
<td></td>
<td></td>
</tr>
<tr>
<td>POC at 20°C</td>
<td>0.0018 (day⁻¹)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>KCDIA (3)</td>
<td>reaction (decay) rates for G₃</td>
<td></td>
<td></td>
</tr>
<tr>
<td>POC at 20°C</td>
<td>0.0 (day⁻¹)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>DCTHTA (1)</td>
<td>constant for T adjustment for</td>
<td></td>
<td></td>
</tr>
<tr>
<td>POP decay</td>
<td>G₁ class POC decay</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1.10 none</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>DCTHTA (2)</td>
<td>constant for T adjustment for</td>
<td></td>
<td></td>
</tr>
<tr>
<td>POP decay</td>
<td>G₂ class POC decay</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1.15 none</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>KSI</td>
<td>1ˢᵗ-order reaction (dissolution)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>PSi at 20°C</td>
<td>0.5 day⁻¹</td>
<td></td>
<td></td>
</tr>
<tr>
<td>THTAS (1)</td>
<td>constant for T adjustment for</td>
<td></td>
<td></td>
</tr>
<tr>
<td>PSi dissolution</td>
<td>1.1 day⁻¹</td>
<td></td>
<td></td>
</tr>
<tr>
<td>M1</td>
<td>solid concentrations in Layer 1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>0.5 kg l⁻¹</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>M2</td>
<td>solid concentrations in Layer 2</td>
<td></td>
<td></td>
</tr>
<tr>
<td>0.5 kg l⁻¹</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>THTAD (1)</td>
<td>constant for T adjustment for</td>
<td></td>
<td></td>
</tr>
<tr>
<td>diffusion coefficient for particle mixing</td>
<td>1.117 none</td>
<td></td>
<td></td>
</tr>
<tr>
<td>THTAD (2)</td>
<td>constant for T adjustment for</td>
<td></td>
<td></td>
</tr>
<tr>
<td>diffusion coefficient for dissolved phase</td>
<td>1.08 none</td>
<td></td>
<td></td>
</tr>
<tr>
<td>KAPPNH₄ (1)</td>
<td>optimum reaction velocity for</td>
<td></td>
<td></td>
</tr>
<tr>
<td>nitrification in Layer 1 for freshwater</td>
<td>0.20 m day⁻¹</td>
<td></td>
<td></td>
</tr>
<tr>
<td>KAPPNH₄ (2)</td>
<td>optimum reaction velocity for</td>
<td></td>
<td></td>
</tr>
<tr>
<td>nitrification in Layer 1 for saltwater</td>
<td>0.14 m day⁻¹</td>
<td></td>
<td></td>
</tr>
<tr>
<td>THTAN (1)</td>
<td>constant for T adjustment for</td>
<td></td>
<td></td>
</tr>
<tr>
<td>nitrification</td>
<td>1.08 none</td>
<td></td>
<td></td>
</tr>
<tr>
<td>KMNH₄</td>
<td>half-saturation constant of NH₄</td>
<td></td>
<td></td>
</tr>
<tr>
<td>for nitrification</td>
<td>1500.0 mg N m⁻³</td>
<td></td>
<td></td>
</tr>
<tr>
<td>KMNH₄O₂</td>
<td>half-saturation constant of DO</td>
<td></td>
<td></td>
</tr>
<tr>
<td>for nitrification</td>
<td>1.0 g O₂ m⁻³</td>
<td></td>
<td></td>
</tr>
<tr>
<td>PIENH₄</td>
<td>partition coefficient for NH₄ in</td>
<td></td>
<td></td>
</tr>
<tr>
<td>both layers</td>
<td>1.0 per kg l⁻¹</td>
<td></td>
<td></td>
</tr>
<tr>
<td>KAPPNO₃F</td>
<td>reaction velocity for denitrification</td>
<td></td>
<td></td>
</tr>
<tr>
<td>in Layer 1 at 20°C for freshwater</td>
<td>0.3 m day⁻¹</td>
<td></td>
<td></td>
</tr>
<tr>
<td>KAPPNO₃S</td>
<td>reaction velocity for denitrification</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Table V.10 (cont’d)

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value Details</th>
</tr>
</thead>
<tbody>
<tr>
<td>K2NO3 in Layer 1 at 20°C for saltwater</td>
<td>0.125 m day(^{-1})</td>
</tr>
<tr>
<td>reaction velocity for denitrification in Layer 2 at 20°C</td>
<td>0.25 m day(^{-1})</td>
</tr>
<tr>
<td>THTANO3 constant for T adjustment for denitrification</td>
<td>1.08 none</td>
</tr>
<tr>
<td>KAPPP1 reaction velocity for particulate H(_2)S oxidation in Layer 1 at 20°C</td>
<td>0.2 m day(^{-1})</td>
</tr>
<tr>
<td>PIE1S partition coefficient for H(_2)S in Layer 1</td>
<td>100.0 per kg l(^{-1})</td>
</tr>
<tr>
<td>PIE2S partition coefficient for H(_2)S in Layer 2</td>
<td>100.0 per kg l(^{-1})</td>
</tr>
<tr>
<td>THTAPD1 constant for T adjustment for both dissolved &amp; particulate H(_2)S oxidation</td>
<td>1.08 none</td>
</tr>
<tr>
<td>KMHSO2 constant to normalize H(_2)S oxidation rate for oxygen</td>
<td>4.0 g O(_2) m(^{-3})</td>
</tr>
<tr>
<td>CSISAT saturation concentration of Si in the pore water</td>
<td>40000.0 mg Si m(^{-3})</td>
</tr>
<tr>
<td>DPIE1SI incremental partition coefficient for Si in Layer 1</td>
<td>10.0 per kg l(^{-1})</td>
</tr>
<tr>
<td>PIE2SI critical DO concentration for Layer 1</td>
<td>100.0 per kg l(^{-1})</td>
</tr>
<tr>
<td>O2CRITSI critical DO concentration for Layer 1 incremental Si sorption</td>
<td>1.0 g O(_2) m(^{-3})</td>
</tr>
<tr>
<td>KMPSI half-saturation constant of PSi for Si dissolution</td>
<td>5 × 10(^{7}) mg Si m(^{-3})</td>
</tr>
<tr>
<td>JSIDETR detrital flux of PSi to account for PSi settling to the sediment that is not associated with algal flux of PSi</td>
<td>100.0 mg Si m(^{-2}) day(^{-1})</td>
</tr>
<tr>
<td>DPIE1PO4F* incremental partition coefficient for PO(_4) in Layer 1 for freshwater</td>
<td>3000.0 per kg l(^{-1})</td>
</tr>
<tr>
<td>DPIE1PO4S* incremental partition coefficient for PO(_4) in Layer 1 for saltwater</td>
<td>300.0 per kg l(^{-1})</td>
</tr>
<tr>
<td>PIE2PO4* partition coefficient for PO(_4) in Layer 2</td>
<td>100.0 per kg l(^{-1})</td>
</tr>
<tr>
<td>O2CRIT critical DO concentration for Layer 1 incremental PO(_4) sorption</td>
<td>2.0 g O(_2) m(^{-3})</td>
</tr>
<tr>
<td>KMO2DP half-saturation constant of DO for particle mixing</td>
<td>4.0 g O(_2) m(^{-3})</td>
</tr>
<tr>
<td>TEMPBEN temperature at which benthic stress accumulation is reset to zero</td>
<td>10.0 °C</td>
</tr>
<tr>
<td>KBENSTR 1(^{st})-order decay rate for benthic stress</td>
<td>0.03 day(^{-1})</td>
</tr>
<tr>
<td>KLBNTH ratio of bio-irrigation to bioturbation</td>
<td>0.0 none</td>
</tr>
<tr>
<td>DPMIN minimum diffusion coefficient for particle mixing</td>
<td>3 × 10(^{-6}) m(^{2}) day(^{-1})</td>
</tr>
<tr>
<td>KAPPCH4 reaction velocity for dissolved CH(_4) oxidation in Layer 1 at 20°C</td>
<td>0.2 m day(^{-1})</td>
</tr>
</tbody>
</table>
Table V.10 (con’t)

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>THTACH4</td>
<td>constant for T adjustment for dissolved CH₄ oxidation</td>
</tr>
<tr>
<td>VSED</td>
<td>net burial (sedimentation) rate</td>
</tr>
<tr>
<td>VPMIX</td>
<td>diffusion coefficient for particle mixing</td>
</tr>
<tr>
<td>VDMIX</td>
<td>diffusion coefficient in pore water</td>
</tr>
<tr>
<td>WSCNET</td>
<td>net settling velocity for algal group 1</td>
</tr>
<tr>
<td>WSDNET</td>
<td>net settling velocity for algal group 2</td>
</tr>
<tr>
<td>WSGNET</td>
<td>net settling velocity for algal group 3</td>
</tr>
</tbody>
</table>

Table V.11. Water quality parameters in CBP monitoring data

<table>
<thead>
<tr>
<th>Parameters</th>
<th>symbol</th>
<th>units</th>
</tr>
</thead>
<tbody>
<tr>
<td>temperature</td>
<td>T</td>
<td>degrees C</td>
</tr>
<tr>
<td>salinity</td>
<td>S</td>
<td>ppt</td>
</tr>
<tr>
<td>dissolved oxygen</td>
<td>DO</td>
<td>mg/l</td>
</tr>
<tr>
<td>chlorophyll-a</td>
<td>CHL</td>
<td>μg/l</td>
</tr>
<tr>
<td>total suspended solids</td>
<td>TSS</td>
<td>mg/l</td>
</tr>
<tr>
<td>secchi depth</td>
<td></td>
<td>m</td>
</tr>
<tr>
<td>particulate carbon</td>
<td>PC</td>
<td>mg/l</td>
</tr>
<tr>
<td>dissolved organic carbon</td>
<td>DOC</td>
<td>mg/l</td>
</tr>
<tr>
<td>particulate nitrogen</td>
<td>PN</td>
<td>mg/l</td>
</tr>
<tr>
<td>total dissolved nitrogen</td>
<td>TDN</td>
<td>mg/l</td>
</tr>
<tr>
<td>ammonium nitrogen</td>
<td>NH₄</td>
<td>mg/l</td>
</tr>
<tr>
<td>nitrate+nitrite nitrogen</td>
<td>NO₃</td>
<td>mg/l</td>
</tr>
<tr>
<td>particulate phosphorus</td>
<td>PP</td>
<td>mg/l</td>
</tr>
<tr>
<td>total dissolved phosphorus</td>
<td>TDP</td>
<td>mg/l</td>
</tr>
<tr>
<td>dissolved phosphate</td>
<td>PO₄d</td>
<td>mg/l</td>
</tr>
<tr>
<td>particulate inorganic phosphorus</td>
<td>PIP</td>
<td>mg/l</td>
</tr>
<tr>
<td>particulate biogenic silica</td>
<td>SU</td>
<td>mg/l</td>
</tr>
<tr>
<td>dissolved silica</td>
<td>SA</td>
<td>mg/l</td>
</tr>
</tbody>
</table>

V-2-5 Model Calibration Results

Calibration of the water quality model is shown by the comparison of time series plots of selected water quality parameters with DEQ observations at all 16 DEQ stations spanning the Lynnhaven River. The locations of the stations are shown in Figure V.18. To facilitate the comparison, stations of each Lynnhaven River branch are clustered in the figures comparing observed versus predicted values of each parameter for stations of that branch.
DEQ Measurement Stations in Lynnhaven River

Figure V.18. Grouping of Lynnhaven DEQ stations by branch, as used in displaying CE-QUAL-ICM water quality model calibration results.

For the calibration, comparisons at each station were made for the full calendar year 2006. These comparisons included the primary parameters of dissolved oxygen, chlorophyll-a, total Kjeldahl nitrogen (TKN), ammonium, nitrate-nitrite, total phosphorus, and ortho phosphorus. For validation of the model, these same comparisons were also made for the full calendar years 2004 and 2005 and are presented in Chapter VI.

The quantification of the model’s overall ability to reproduce the observed data at these stations, as measured by statistical analysis, is presented later in this section. For the analysis on each water quality state variable, the differences of predicted and observed values for all 16 Lynnhaven DEQ stations were included.
A. Western Branch DEQ stations calibration results

Water quality model calibration results for Western Branch DEQ stations for 2006 are shown in Figures V.19 through V.25. In all figures comparing modeled and measured water quality parameters, the model predictions are represented as a gray band bounded by daily minimum and maximum predictions.

Results for dissolved oxygen are shown in Figure V.19. As illustrated, the model reproduces the observed temporal distribution of dissolved oxygen reasonably well, with some discrepancy at the upstream Thalia Creek station, the only Western Branch DEQ station where DO values fall below 5 mg/l. Figure V.20 presents the predicted versus observed comparisons for chlorophyll-a, catching the trend for the downstream stations, but showing slight under-predictions. Figure V.21 shows that the model captures TKN values well for all Western Branch DEQ stations. The predictions of ammonium and nitrate-nitrite shown in Figures V.22 and V.23, respectively, have some large diurnal fluctuations, but observed values primarily fall within these ranges. Figures V.24 and V.25 show that both total phosphorus and ortho phosphorus measurements are captured reasonably well at all stations. An inspection of Figures V.19 through V.25 shows the gradual decrease of dissolved oxygen and increases of both chlorophyll-a and nutrients in moving from the Inlet upstream to Thalia Creek.

Figure V.19. Predicted vs. observed DO at Western Branch DEQ stations for 2006.
Figure V.20. Predicted vs. observed chlorophyll-a at Western Branch DEQ stations for 2006.

Figure V.21. Predicted vs. observed TKN at Western Branch DEQ stations for 2006.
Figure V.22. Predicted vs. observed ammonium at Western Branch DEQ stations for 2006.

Figure V.23. Predicted vs. observed nitrate-nitrite at Western Branch DEQ stations for 2006.
Figure V.24. Predicted vs. observed total phosphorus at Western Branch DEQ stations for 2006.

Figure V.25. Predicted vs. observed ortho phosphorus at Western Branch DEQ stations for 2006.
B. Eastern Branch DEQ stations calibration results

The calibration process was continued from the DEQ stations located in the Western Branch to the 6 DEQ stations located in the Eastern Branch. Initially, it was uncertain whether the model calibration coefficients and parameters would be the same in the Eastern Branch as in the Western Branch due to different characteristics. For example, the Eastern Branch is much longer than the Western Branch and includes a canal that was dredged and deepened to the headwater. Since nonpoint sources are the only source of pollutants, the increase in freshwater runoff to the Eastern Branch will have an accompanying increase in pollutant loads that will affect general property of algae growth rates, respiration rates, cell nutrient composition, and sediment characteristics.

At a meeting in June 2005 between representatives of the City of Virginia Beach, the Army Corps, URS, and VIMS, representatives from the City of Virginia Beach expressed a concern that the VIMS modeling domain did not extend to the West Neck Creek region. This region is at the head of the Eastern Branch and is known as the West Neck Creek - London Bridge Creek System, including the Canal No. 2. It was noted that many water quality issues were associated with conditions originating in this system. VIMS thus extended the model domain beyond London Bridge to include West Neck Creek.

After a series of runs comparing between model results and observed data, it became apparent that the new boundary upstream of West Neck Creek produced better results. Given the proper hydrodynamic results, without much change on the water quality parameters, the water quality model results were satisfactory. Water quality model calibration results for Eastern Branch DEQ stations for 2006 are shown in Figures V.26 through V.32. In all figures comparing modeled and measured water quality parameters, the model predictions are represented as a gray band bounded by daily minimum and maximum predictions.

Results for dissolved oxygen are shown in Figure V.26. As illustrated, the model reproduces the observed temporal distribution of dissolved oxygen reasonably well, with only a slight over-prediction at the upstream London Bridge (7-LOB011.79) and Canal No. 2 (7-XBO011.30) stations, where summertime DO measurements fall below 5 mg/l. Figure V.27 presents the predicted versus observed comparisons for chlorophyll-a, catching the trend for all stations, but there were a couple of outliers in the sparse observation data. Figure V.28 shows that the model captures the trend of measured TKN values well for all Eastern Branch DEQ stations. The predictions of ammonium and nitrate-nitrite shown in Figures V.29 and V.30, respectively, have some large diurnal fluctuations, but observed values primarily fall within these ranges. Figure V.31 shows that total phosphorus predictions match observations well overall, although these may slightly under-predict in summer at the mid-branch stations of 7-EBL02.54, 5BWNC010.02, and 7-XBO011.30. Ortho phosphorus measurements are captured reasonably well at all stations, as shown in Figure V.32. An inspection of Figures V.26 through V.32 shows gradual increases of both chlorophyll-a and nutrients in moving from the Inlet upstream to West Neck Creek (5BWNC010.02), and a slight decrease in dissolved oxygen is seen moving upstream in the Eastern Branch.
Figure V.26. Predicted vs. observed dissolved oxygen at Eastern Branch DEQ stations for 2006.

Figure V.27. Predicted vs. observed chlorophyll-a at Eastern Branch DEQ stations for 2006.
Figure V.28. Predicted vs. observed TKN at Eastern Branch DEQ stations for 2006.

Figure V.29. Predicted vs. observed ammonium at Eastern Branch DEQ stations for 2006.
Figure V.30. Predicted vs. observed nitrate-nitrite at Eastern Branch DEQ stations for 2006.

Figure V.31. Predicted vs. observed total phosphorus at Eastern Branch DEQ stations for 2006.
C. Broad Bay / Linkhorn Bay Branch DEQ stations calibration results

Water quality model calibration results for Broad Bay / Linkhorn Bay Branch DEQ stations for 2006 are shown in Figures V.33 through V.39. In all figures comparing modeled and measured water quality parameters, the model predictions are represented as a gray band bounded by daily minimum and maximum predictions.

Results for the comparison of modeled versus measured dissolved oxygen at Broad and Linkhorn Bay Branch DEQ stations are shown in Figure V.33. As illustrated, the model reproduces the observed temporal distribution of dissolved oxygen extremely well at all 5 DEQ stations in this branch. One may note that all modeled and observed values exceed 5 mg/l throughout the year. Figure V.34 shows reasonably good agreement overall between predicted and observed values for chlorophyll-a, but there may be some over-prediction at upstream stations 7-CRY000.59, 7-LNC000.68, and 7-LKN002.77 beyond Julian Day 280. Figure V.35 shows good agreement between modeled and measured TKN values at all Broad Bay and Linkhorn Bay DEQ stations. The predicted values of ammonium and nitrate-nitrite shown in Figures V.36 and V.37, respectively, match observed values quite well. Figures V.38 and V.39 show that total phosphorus and ortho phosphorus predictions match observations well at all 5 DEQ stations in this branch. An inspection of Figures V.33 through V.39 shows better water quality in this branch than in the Western and Eastern Branches. Finally, there is almost no spatial decrease in dissolved oxygen nor increase in either chlorophyll-a or nutrients in moving from the Inlet upstream to the head of Linkhorn Bay.
Figure V.33. Predicted vs. observed dissolved oxygen at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006.

Figure V.34. Predicted vs. observed chlorophyll-a at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006.
Figure V.35. Predicted vs. observed TKN at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006.

Figure V.36. Predicted vs. observed ammonium at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006.
Figure V.37. Predicted vs. observed nitrate-nitrite at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006.

Figure V.38. Predicted vs. observed total phosphorus at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006.
Summary Statistics of Water Quality Model Calibration Results

In the previous portion of this section, qualitative comparisons between model results and observed values were presented. Although the comparisons indicate that the CE-QUAL-ICM water quality model can reproduce the physical, chemical, and biological processes that affect the eutrophication process in the Lynnhaven River, a more specific measure of the model performance is desirable.

In order to provide a more quantifiable measure of the performance of the water quality model, a statistical analysis was applied to the predicted and observed data of the water quality calibration results.

For model predictions vs. observations of the water quality parameters compared at the surface layer for the year 2006, various error measurements serve to quantify the performance of the water quality model. Error measurements determined include:

1) Mean error – The mean error statistic is defined as:

\[ ME = \frac{\sum (O - P)}{n} \]
where: $ME = \text{mean error}$, $O = \text{observation}$, $P = \text{model predicted result}$, and $n = \text{number of observations}$. The mean error is a summary of the model tendency to overestimate or underestimate the data.

2) **Absolute Mean error** – The absolute mean error statistic is defined as:

$$AME = \frac{\sum |O - P|}{n}$$

where: $AME = \text{absolute mean error}$. The absolute mean error is a measure of the average discrepancy between observations and model results.

3) **Root-Mean-Square Error** – The root-mean-square error statistic is defined as:

$$RME = \sqrt{\frac{\sum (O - P)^2}{n}}$$

where: $RME = \text{root-mean-square error}$. The root-mean-square error is an alternate quantification of the average discrepancy between observations and model results.

4) **Relative Error** – The relative error statistic is defined as:

$$RE = \frac{\sum |O - P|}{\sum O}$$

where: $RE = \text{relative error}$. The relative error statistic normalizes absolute mean error by the magnitude of the observations.

Additionally, 1:1 plots of predicted results vs. observations show visually how well the model predictions compare with observations and whether the model shows a bias towards either over-prediction or under-prediction.

**A. Statistical Analysis of Dissolved Oxygen, Chlorophyll-a, TKN, and Total Phosphorus Results**

Statistical analysis of 7 key water quality parameters was performed by comparing predicted and observed results of each parameter for all of the 16 Lynnhaven DEQ stations combined. The every-other-month DEQ measurements taken during the 2006 year thus provided sample sizes of 90, 86, 90, and 90, respectively, for DO, chl-a, TKN, and TP predicted vs. observed comparisons at all Lynnhaven River DEQ stations. The 1:1 plots are shown in Figure V.40 for these 4 comparisons and their corresponding error measures are shown in Table V.12. Overall, predicted and observed DO values compare well. The median value for mean error is about 0.69 mg/l while the absolute mean error is 1.07 mg/l. The root-mean-square error for both surface and bottom DO is about 1.47
mg/l, whereas the relative error is around 13%. These statistics are comparable to other eutrophication model studies such as the Three-dimensional Eutrophication Model Study of the Chesapeake Bay (Cerco and Cole, 1994).

It was also worthwhile to point out that the absolute mean error and root-mean-square error of water quality parameters shown in Table V.12 are well within the range of natural variation in a given season of measurements when compared with available observations, for example, Figures V.19-V.21, V.24, V.26-V.28, V.31, V.33-V.35, and V.38.

Figure V.40. Plots of 1:1 predicted vs. observed DO, chl-a, TKN, and TP at all 16 Lynnhaven DEQ stations for 2006.
Table V.12. Statistical summary of errors derived by comparing predicted vs. observed surface values of DO, chl-a, TKN, and TP for year 2006.

<table>
<thead>
<tr>
<th>Surface Comparisons of Predicted vs. Observed Dissolved Oxygen, Chlorophyll-a, TKN, and Total Phosphorus</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
<tr>
<td></td>
</tr>
</tbody>
</table>

B. Statistical Analysis of Ammonia, Nitrate-Nitrite, and Dissolved Inorganic Phosphate

To quantify the comparison between predicted and observed values NH\textsubscript{4}, NO\textsubscript{x}, and DIP, determination of statistical errors and construction of 1:1 plots were performed for these parameters as well. Table V.13 below shows error values of each parameter for predicted vs. observed comparisons of all 16 Lynnhaven DEQ stations combined for 2006.

The nitrogen and phosphorus are major nutrients that can be used for photosynthesis. In particular, NH\textsubscript{4}, NO\textsubscript{x}, and dissolved phosphorus are species that can be uptaken directly by the phytoplankton. Therefore, they are important indicators for the environmental quality. Nitrogen’s concentration is usually higher than that of phosphorus. The 1:1 plots of predicted vs. observed comparisons of NH\textsubscript{4}, NO\textsubscript{x}, and DIP are shown in Figure V.41. The summary is shown in Table V.13. The absolute mean error and root-mean-square error of these water quality parameters show the differences between model predictions and observations are within the range of natural variation in a given season of measurements when compared with available observations, for example, as shown in Figures V.22-V.23, V.25, V.29-V.30, V.32, V.36-V.37, and V.39.

Table V.13. Statistical summary of errors derived by comparing predicted vs. observed values of NH\textsubscript{4}, NO\textsubscript{x}, and DIP for all 16 Lynnhaven DEQ stations combined for 2006.
V-3 Calibration of the Sediment Transport Model

The model was calibrated by adjusting the erosion coefficient \( M \) to make the modeled results agree with observation data. The TSS observation data of 2006 collected at the 16 Lynnhaven DEQ stations (locations shown earlier in Figure V.18) were used to calibrate the model. The comparisons between model predictions and observations for TSS are shown in Figures V.42 through V.44, respectively, for the Western, Eastern, and Broad Bay / Linkhorn Bay DEQ stations for calibration year 2006.

Validation of the sediment transport model, using the 2004 and 2005 DEQ data, is shown in Chapter VI, Section VI-3.
Figure V.42. Predicted vs. observed TSS at Western Branch DEQ stations for 2006.

Figure V.43. Predicted vs. observed TSS at Eastern Branch DEQ stations for 2006.
Figure V.44. Predicted vs. observed TSS at Broad Bay / Linkhorn Bay Branch DEQ stations for 2006.
CHAPTER VI. MODEL VALIDATION

The hydrodynamic and water quality models applied to the Lynnhaven River system were developed using the framework outlined in Chapter III. Chapter V describes how the models were calibrated based on 2006 intensive field measured data described. As part of quality control, the model validation is a process for independent checking that the modeling results meet specifications using a different dataset and that it fulfills its intended purpose.

The hydrodynamic model was validated using synoptic data collected in September and November 2005 and the water quality model for the years 2004 and 2005, during which period both the freshwater discharge and the non-point source loading data were provided by the HSPF watershed model in Lynnhaven River, developed by URS Corporation.

VI-1 Validation of the Hydrodynamic Model

It was critical to conduct a systematic, high-frequency hydrodynamic survey, measuring water elevations inside the inlet synoptically with representative currents and salinities in each branch as well as outside of the Inlet (see Section IV-2-A for a full description of the VIMS Lynnhaven hydrodynamic survey). With these data in hand, validation then

![Diagram of Lynnhaven observation stations](image)

Figure VI.1. Locations of Lynnhaven observation stations (tide and velocity) in 2005.
consisted of a real-time simulation of the prototype condition for periods in September and November, 2005, during which time high-frequency observations of tides, as well as representative high-frequency velocities and salinities in each branch, were available.

**VI-1-1 Validation for tidal elevation**

In September 2005, a tidal gauge was deployed for 2 weeks in the upper Eastern Branch at West Neck Creek (WNC). In November 2005, a 30-day deployment was made at the Virginia Pilot Station, just inside the Inlet. Locations of these 2 stations are shown in Figure VI.1.

These tidal observations in 2005 were compared to UnTRIM model results from a real-time simulation invoking both the freshwater discharge provided by URS and high frequency wind from the Chesapeake Bay Bridge Tunnel (CBBT) station. The comparison of UnTRIM modeled predictions with observations is shown in Figure VI.2.

![Figure VI.2. Modeled versus observed water elevations at the Virginia Pilot’s station (November 2005) and in West Neck Creek (September 2005).](image-url)
VI-1-2 Validation for velocity

For the VIMS hydrodynamic survey conducted in November 2005, the measurements of tidal velocity were made over a 30-day period using an ADP instrument outside the inlet and an S4 current meter at representative locations of each Lynnhaven branch. Locations of these instruments are shown in Figure VI.3 below.

The bottom-mounted ADP outside the Inlet measured velocities at 10 layers in the vertical at a frequency of every 20 minutes for the 30-day deployment. The S4 instruments deployed in each branch measured mid-depth velocity at 30-minute intervals over the deployment.

East-west and north-south component comparisons between observed and predicted currents outside the Inlet are shown in Figure VI.4. The modeled and observed velocity magnitude and direction comparisons are shown for the Western, Eastern, and Broad Bay branches, respectively, in Figures VI.5 through VI.7. In general, good agreement is shown between modeled and observed tidal velocities.
Figure VI.4. East-west and north-south components of measured versus modeled velocity at surface, middle, and bottom layers outside Lynnhaven Inlet.
Comparison of Velocity (Western Branch)

Figure VI.5. Magnitude and direction of measured versus modeled velocity at mid-depth in the Western Branch.
Comparison of Velocity (Eastern Branch)

Figure VI.6. Magnitude and direction of measured versus modeled velocity at mid-depth in the Eastern Branch.
Comparison of Velocity (Broad Bay)

Figure VI.7. Magnitude and direction of measured versus modeled velocity at mid-depth in Broad Bay.
VI-1-3 Validation for salinity

In order to validate salinity predicted by the UnTRIM hydrodynamic model, comparisons between measurements and model predictions were made at all 16 VA-DEQ stations monitored every other month in the Lynnhaven River throughout calendar years 2004 and 2005. Measured data also included those made by the VIMS dataflow surveys during this period (please note that the dataflow coverage did not extend to all 16 stations). The locations of these stations are shown below in Figure VI.8 and the modeled vs. measured salinities for 2004-2005 are shown in Figures VI.9-VI.10, VI.11-VI.12, and VI.13-VI.14, respectively, for the Western, Eastern, and Broad Bay/Linkhorn Bay Branches.

VI-1-4 Validation for temperature

The locations of these stations are shown in Figure VI.8 and the modeled vs. measured temperatures for 2004-2005 are shown in Figures VI.15-VI.16, VI.17-VI.18, and VI.19-VI.20, respectively, for the Western, Eastern, and Broad Bay/Linkhorn Bay Branches.

DEQ Measurement Stations in Lynnhaven River

Figure VI.8. Grouping by branch of Lynnhaven DEQ stations as used to compare measured and modeled salinity, temperature, and CE-QUAL-ICM water quality model validation results.
Figure VI.9. UnTRIM modeled versus measured salinities at Western Branch DEQ stations for 2004.

Figure VI.10. UnTRIM modeled versus measured salinities at Western Branch DEQ stations for 2005. Red asterisks denote DEQ measurements and red circles denote VIMS dataflow measurements.
Figure VI.11. UnTRIM modeled versus measured salinities at Eastern Branch DEQ stations for 2004.

Figure VI.12. UnTRIM modeled versus measured salinities at Eastern Branch DEQ stations for 2005. Red asterisks denote DEQ measurements and red circles denote VIMS dataflow measurements.
Figure VI.13. UnTRIM modeled versus measured salinities at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004.

Figure VI.14. UnTRIM modeled versus measured salinities at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005. Red asterisks denote DEQ measurements and red circles denote VIMS dataflow measurements.
Figure VI.15. UnTRIM modeled versus measured temperatures at Western Branch DEQ stations for 2004.

Figure VI.16. UnTRIM modeled versus measured temperatures at Western Branch DEQ stations for 2005.
Figure VI.17. UnTRIM modeled versus measured temperatures at Eastern Branch DEQ stations for 2004.

Figure VI.18. UnTRIM modeled versus measured temperatures at Eastern Branch DEQ stations for 2005.
Figure VI.19. UnTRIM modeled versus measured temperatures at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004.

Figure VI.20. UnTRIM modeled versus measured temperatures at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005.
VI-2 Validation of the Water Quality Model

The overall objective of the model validation procedure is to confirm the predictive capability of the CE-QUAL-ICM model by simulating an entirely different period than that selected for model calibration. Results of the calibration simulation (2006) are shown in Chapter V, Section V-2-5.

Because some parameters were not measured by DEQ in 2004 and in the first half of 2005 due to Virginia State budgetary restrictions that impacted the DEQ monitoring program, the full period of 2004-2005 was selected for model validation.

VI-2-1 Model Validation Results

Lynnhaven hydrologies in 2004 and 2005 differ from that in 2006. On an annual basis, the year of 2004 had higher freshwater input than 2005 and 2005, in turn, had higher input than 2006. In other words, the year 2006 had the lowest freshwater input among 2004, 2005, and 2006. As a result, the salinity of 2006 was the largest and that of 2004 was the smallest. This is part of a long-term trend of decreasing freshwater water input spanning from 2003 to 2008 noted from James River freshwater records.

On the seasonal basis, the year 2004 has a relatively dry winter/spring (from day 70 – 100) but a wet summer (from day 180- 210). On the contrary, the year 2005 had a wet winter/spring (from day 50- 75) and a dry summer/fall (from day 210 – 270). This pattern shift affects the seasonal variation of the water quality within the yearly cycle.

In terms of the annual temperature pattern, the year 2005 had the highest summer water temperature reaching 29.8 degrees Celsius in August, followed by 2006 and 2004. It does not, however, show a significant seasonal shift over the three years 2004-2006. Water quality variables are affected by both salinity and temperature and, thus, it is important to recognize that there are inter-annual, as well as seasonal, variations.

Given that the physical parameters varied from year to year, it is obvious that there will be ramifications on the water quality variables both in terms of their loading as well as the result of chemical kinetics. Validation of the water quality model took place by comparison of time series plots of selected water quality parameters with DEQ observations at the 16 locations shown earlier in Figure VI.8. As was done for the display of calibration results, stations of each Lynnhaven River branch are clustered in the figures comparing observed versus predicted values of each parameter for stations of that branch to facilitate the comparison.

Model simulation results at each station are shown for the full calendar years of 2004 and 2005 and include the primary water quality parameters of dissolved oxygen, chlorophyll-a, TKN, ammonia, nitrate-nitrite, total phosphorus, and ortho phosphorus. Due to the restrictions on monitoring in 2004 and early 2005, validation comparisons of TKN, ammonium, nitrate-nitrite, and ortho phosphorus are limited to the latter half of 2005.
A. Western Branch DEQ stations validation results

As described above, the hydrological conditions in 2004 and 2005 are quite different from those in 2006. After the calibration has been performed for the year of 2006, the validation provides an independent check of whether the modeling results can meet specifications using different hydrological datasets and fulfills its intended purpose.

Keep in mind, however, that between 2004 and 2005, the seasonal patterns are also different. The year of 2004 has a dry spring and wet summer whereas the year of 2005 has a wet spring, but a dry summer. Water quality model validation results for Western Branch DEQ stations for 2004 and 2005 are carried out with different salinity patterns and the reaction constants that are temperature-dependent. The results are shown in Figures VI.21 through VI.34. In all figures, the model predictions are represented as a gray band bounded by daily minimum and maximum.

Results for dissolved oxygen in 2004 and 2005, respectively, are shown in Figures VI.21 and VI.22. As illustrated, the model reproduces the observed temporal distribution of dissolved oxygen quite well. The seasonally low DO values (i.e., below 5 mg/l) measured throughout the Western Branch around Julian Day 200 of 2005 were well-captured by model predictions. Figures VI.23 and VI.24 present the predicted versus observed comparisons for chlorophyll-a, catching the trend for the downstream stations, but showing some isolated discrepancies at the upstream stations 7-WES002.58 and 7-THA000.76. Figures VI.25 and VI.26 show model predictions of TKN during 2004 and 2005 for all Western Branch DEQ stations. Observed TKN was only available in latter 2005, but showed good agreement with predictions over this period. The predictions of ammonium shown in Figures VI.27 and VI.28 for 2004 and 2005, respectively, have similar seasonal trends at all stations, and the available observed data from the latter part of 2005 match the predictions reasonably well at all Western Branch DEQ stations. Figures VI.29 and VI.30 show predictions of nitrate-nitrite for 2004 and 2005, respectively, and the available observation measurements of the latter part of 2005 are shown to match reasonably well. An inspection of Figures VI.31 through VI.34 shows that both total phosphorus and ortho-phosphorus measurements are captured reasonably well at all Western Branch DEQ stations.

As in the case of comparisons of observed vs. predicted parameter values for the model calibration (2006) shown in Chapter V, an inspection of Figures VI.21 through VI.34 shows the gradual decrease of dissolved oxygen and increases of both chlorophyll-a and nutrient levels in moving from the Inlet upstream to Thalia Creek. This is a spatial gradient pattern that is consistent with what was observed in the historical data. The shift on the spring and summer pattern basically reflects the difference of the hydrological year. The model does respond truthfully to the real environmental conditions.
Figure VI.21. Predicted vs. observed dissolved oxygen at Western Branch DEQ stations for 2004.

Figure VI.22. Predicted vs. observed dissolved oxygen at Western Branch DEQ stations for 2005.
Figure VI.23. Predicted vs. observed chlorophyll-a at Western Branch DEQ stations for 2004.

Figure VI.24. Predicted vs. observed chlorophyll-a at Western Branch DEQ stations for 2005.
Figure VI.25. Predicted TKN at Western Branch DEQ stations for 2004.

Figure VI.26. Predicted vs. observed TKN at Western Branch DEQ stations for 2005.
Figure VI.27. Predicted ammonium at Western Branch DEQ stations for 2004.

Figure VI.28. Predicted vs. observed ammonium at Western Branch DEQ stations for 2005.
Figure VI.29. Predicted nitrate-nitrite at Western Branch DEQ stations for 2004.

Figure VI.30. Predicted vs. observed nitrate-nitrite at Western Branch DEQ stations for 2005.
Figure VI.31. Predicted vs. observed total phosphorus at Western Branch DEQ stations for 2004.

Figure VI.32. Predicted vs. observed total phosphorus at Western Branch DEQ stations for 2005.
Figure VI.33. Predicted ortho-phosphorus at Western Branch DEQ stations for 2004.

Figure VI.34. Predicted vs. observed ortho-phosphorus at Western Branch DEQ stations for 2005.
B. Eastern Branch DEQ stations validation results

As mentioned, the hydrological condition in 2004 and 2005 are different from those of 2006; between 2004 and 2005, the seasonal patterns also shifted differently. The year 2004 has a dry spring and wet summer whereas the year 2005 has a wet spring, but a dry summer. These conditions applied in the Western Branch as well as in the Eastern Branch. Water quality model validation results for Eastern Branch DEQ stations for 2004 and 2005 are carried out with different salinity patterns and the reaction constants that are temperature-dependent. Water quality model validation results for Eastern Branch DEQ stations for 2004 and 2005 are shown in Figures VI.35 through VI.48. In all figures comparing modeled and measured water quality parameters, the model predictions are represented as a gray band bounded by daily minimum and maximum.

Results for dissolved oxygen in 2004 and 2005, respectively, are shown in Figures VI.35 and VI.36. As illustrated, the model reproduces the observed temporal distribution of dissolved oxygen reasonably well, with only occasional over-prediction at the upstream stations of London Bridge (7-LOB001.79) Canal No. 2 (7-XBO001.30), and West Neck Creek (5BWNC010.02), in the latter part of each year. Figures VI.37 and VI.38 present the predicted versus observed comparisons for chlorophyll-a, catching the trend for all stations, but there are a few out-liers in the sparse observation data. Figures VI.39 and VI.40 show reasonable predicted results for 2004 and 2005, respectively, with good agreement with measured TKN values in latter 2005 (Figure VI.40). Predicted values for 2004-2005 ammonium for the Eastern Branch stations are shown in Figures VI.41 and VI.42. Despite some large diurnal fluctuations, these results appear to be reasonable, and agree well with the DEQ measurements taken in latter 2005 shown in Figure VI.42. Figures VI.43 and VI.44 show the 2004-2005 model predictions for nitrate-nitrite. Measured values of nitrate-nitrite in latter 2005 all fall within the daily min-max prediction range. Figures VI.45 and VI.46 show that, whereas total phosphorus predictions have a large diurnal range in the Eastern Branch, all observation data fall within this range. Lastly, the 2004-2005 ortho phosphorus predictions shown in Figures VI.47 and VI.48 appear reasonable and match the observation data shown in Figure VI.48 for latter 2005.

As was the case for the 2006 calibration data for Eastern Branch DEQ stations shown in Chapter V, an overall inspection of Figures VI.35 through VI.48 shows gradual increases of both chlorophyll-a and nutrients in moving from the Inlet upstream to West Neck Creek (5BWNC010.02), and a slight decrease in the summer of dissolved oxygen as seen moving upstream in the Eastern Branch. Overall, the responses in the Eastern Branch are very similar to those in the Western Branch, except that, at the very upstream stations, we consistently observe that Thalia Creek in the Western Branch has slightly, but consistently, higher TKN, NH4, and chlorophyll values as compared to the stations at London Bridge, Canal No. 2, and West Neck Creek stations. That could contribute to a higher chance of forming localized low DO in the summer.
Figure VI.35. Predicted vs. observed dissolved oxygen at Eastern Branch DEQ stations for 2004.

Figure VI.36. Predicted vs. observed dissolved oxygen at Eastern Branch DEQ stations for 2005.
Figure VI.37. Predicted vs. observed chlorophyll-a at Eastern Branch DEQ stations for 2004.

Figure VI.38. Predicted vs. observed chlorophyll-a at Eastern Branch DEQ stations for 2005.
Figure VI.39. Predicted TKN at Eastern Branch DEQ stations for 2004.

Figure VI.40. Predicted vs. observed TKN at Eastern Branch DEQ stations for 2005.
Figure VI.41. Predicted ammonium at Eastern Branch DEQ stations for 2004.

Figure VI.42. Predicted vs. observed ammonium at Eastern Branch DEQ stations for 2005.
Figure VI.43. Predicted nitrate-nitrite at Eastern Branch DEQ stations for 2004.

Figure VI.44. Predicted vs. observed nitrate-nitrite at Eastern Branch DEQ stations for 2005.
Figure VI.45. Predicted vs. observed total phosphorus at Eastern Branch DEQ stations for 2004.

Figure VI.46. Predicted vs. observed total phosphorus at Eastern Branch DEQ stations for 2005.
Figure VI.47. Predicted ortho phosphorus at Eastern Branch DEQ stations for 2004.

Figure VI.48. Predicted vs. observed ortho phosphorus at Eastern Branch DEQ stations for 2005.
C. Broad Bay / Linkhorn Bay Branch DEQ stations validation results

In the past two sections, we have emphasized that the hydrological conditions in 2004 and 2005 are different from those in 2006. In addition, the year 2004 had a dry spring and wet summer whereas the year of 2005 had a wet spring, but a dry summer. These conditions apply in the Western and Eastern Branches, but do not seem to affect Broad Bay and Linkhorn Bay as much. This is likely because the freshwater inputs in the Broad Bay and Linkhorn Bay are less than those in Eastern and Western Branches and, therefore, the loading was not the single most important reason for the temporal and spatial variability.

Water quality model validation results for Broad Bay/Linkhorn Bay Branch DEQ stations for 2004 and 2005 are carried out with different salinity patterns and the reaction constants that are temperature-dependent. Water quality model validation results for Broad Bay / Linkhorn Bay Branch DEQ stations for 2004 and 2005 are shown in Figures VI.49 through VI.62. In all figures comparing modeled and measured water quality parameters, the model predictions are represented as a gray band bounded by daily minimum and maximum.

Validation results for the comparison of modeled versus measured dissolved oxygen in 2004 and 2005 at Broad and Linkhorn Bay Branch DEQ stations are shown, respectively, in Figures VI.49 and VI.50. As illustrated, the model reproduces the observed temporal distribution of dissolved oxygen extremely well at all 5 DEQ stations in this branch for both years. Figures VI.51 and VI.52 show reasonably good agreement overall between predicted and observed values for chlorophyll-a. Figures VI.53 and VI.54, respectively, show model predictions for 2004 and 2005 for TKN at all Broad Bay and Linkhorn Bay stations, and a good agreement between modeled and measured TKN values can be seen for latter-2005 in Figure VI.54. The 2004 and 2005 predicted values of ammonium are shown in Figures VI.55 and VI.56, respectively, and show good agreement with observations taken in the latter part of 2005 (Figure VI.56). Figures VI.57 and VI.58 show predictions of nitrate-nitrite by the model and match well with available nitrate-nitrite data from latter 2005 (Figure VI.58). Figures VI.59 and VI.60 show that total phosphorus predictions from the model agrees reasonably well with observations at all stations with a slight tendency to over-predict at upstream stations. The model predictions of ortho phosphorus for 2004 and 2005 shown in Figures VI.61 and VI.62 appear reasonable and match the observations available in late 2005 shown in Figure VI.62. Finally, inspection of Figures VI.49 through VI.62 shows that there is almost no spatial decrease in dissolved oxygen nor increase in chlorophyll-a in moving from the Inlet upstream to the head of Linkhorn Bay, similar to what was found for the 2006 calibration data presented in Chapter V. Overall, the Broad Bay and Linkhorn Bay have lower higher TKN, NH4, TP, and Chlorophyll values as compared to those in the Western and Eastern Branches. Hypoxic conditions in this branch are rare occurrences. On a parallel effort, however, there is evidence that the Mill Dam Creek on the southern shore of Broad Bay can occasionally discharge high concentrations of nitrogen and phosphorus into the system. That is beyond the scope of this study.
Figure VI.49. Predicted vs. observed dissolved oxygen at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004.

Figure VI.50. Predicted vs. observed dissolved oxygen at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005.
Figure VI.51. Predicted vs. observed chlorophyll-a at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004.

Figure VI.52. Predicted vs. observed chlorophyll-a at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005.
Figure VI.53. Predicted TKN at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004.

Figure VI.54. Predicted vs. observed TKN at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005.
Figure VI.55. Predicted ammonium at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004.

Figure VI.56. Predicted vs. observed ammonium at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005.
Figure VI.57. Predicted nitrate-nitrite at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004.

Figure VI.58. Predicted vs. observed nitrate-nitrite at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005.
Figure VI.59. Predicted vs. observed total phosphorus at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004.

Figure VI.60. Predicted vs. observed total phosphorus at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005.
Figure VI.61. Predicted ortho phosphorus at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004.

Figure VI.62. Predicted vs. observed ortho phosphorus at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005.
Summary Statistics of Water Quality Model Validation Results

In the previous portion of this section, qualitative comparisons between model results and observed values were presented. As in the case for the model calibration results shown in Chapter V, although the comparisons indicate that the CE-QUAL-ICM water quality model can reproduce the physical, chemical, and biological processes that affect the eutrophication process in the Lynnhaven River, a more specific measure of the model performance is desirable.

In order to provide a more quantifiable measure of the performance of the water quality model during the validation process, a statistical analysis is applied to the comparisons of predicted and observed data of the water quality validation results for 2004 and 2005. Error measurement parameters for these comparisons (i.e., mean error, absolute mean error, root-mean-square error, and relative error) are fully described in Chapter V, which shows the analysis of the performance of the model during calibration.

Additionally, 1:1 plots of predicted results vs. observations show visually how well the model predictions compare with observations and whether the model shows a bias towards either over-prediction or under-prediction.

A. Statistical Analysis of Dissolved Oxygen, Chlorophyll-a, TKN, and Total Phosphorus Results

Statistical analysis of 7 key water quality parameters was performed by comparing predicted and observed results of each parameter for all of the 16 Lynnhaven DEQ stations combined. The every-other-month DEQ measurements taken in 2004 and 2005 thus provided sample sizes of 185, 179, 45, and 18, respectively, for DO, chl-a, TKN, and TP predicted vs. observed comparisons at all Lynnhaven River DEQ stations. The error measures for these 4 comparisons are shown in Table VI.1 below and their corresponding 1:1 plots are shown in Figure VI.63. Overall, predicted and observed DO values compare well. The median value for mean error is about -0.07 mg/l while the absolute mean error is 1.10 mg/l. The root-mean-square error for both surface and bottom DO is about 1.44 mg/l, whereas the relative error is around 12%. It is noted that these statistics compare well with those for the 2006 calibration and that they are comparable to other eutrophication model studies such as the Three-dimensional Eutrophication Model Study of the Chesapeake Bay (Cerco and Cole, 1994).

It was also worthwhile to point out that the absolute mean error and root-mean-square error of water quality parameters shown in Table VI.1 are well within the range of natural variation in a given season of measurements when compared with available observations, for example, Figures VI.21-VI.26, VI.31-VI.32, VI.35-VI.40, VI.45-VI.46, VI.49-VI.54, and VI.59-VI.60.
Table VI.1. Statistical summary of errors derived by comparing predicted vs. observed surface values of DO, chl-a, TKN, and TP for years 2004 - 2005.

<table>
<thead>
<tr>
<th>Parameter:</th>
<th>DO</th>
<th>Chl-a</th>
<th>TKN</th>
<th>TP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sample size</td>
<td>185</td>
<td>179</td>
<td>45</td>
<td>18</td>
</tr>
<tr>
<td>Mean Error</td>
<td>-0.07</td>
<td>0.60</td>
<td>0.13</td>
<td>-0.04</td>
</tr>
<tr>
<td>Absolute Mean Error</td>
<td>1.10</td>
<td>5.17</td>
<td>0.26</td>
<td>0.05</td>
</tr>
<tr>
<td>RMS Error</td>
<td>1.44</td>
<td>10.38</td>
<td>0.30</td>
<td>0.06</td>
</tr>
<tr>
<td>Relative Error</td>
<td>0.12</td>
<td>0.36</td>
<td>0.18</td>
<td>0.49</td>
</tr>
<tr>
<td>Corr. Coeff. (r)</td>
<td>0.89</td>
<td>0.79</td>
<td>0.80</td>
<td>0.85</td>
</tr>
</tbody>
</table>

Figure VI.63. Plots of 1:1 predicted vs. observed DO, chl-a, TKN, and TP at all 16 Lynnhaven DEQ stations for 2004 - 2005.
B. Statistical Analysis of Ammonia, Nitrate-Nitrite, and Dissolved Inorganic Phosphate

To quantify the comparison between predicted and observed values NH₄, NOₓ, and DIP, determination of statistical errors and construction of 1:1 plots were performed for these parameters as well. Table VI.2 below shows error values of each parameter for predicted vs. observed comparisons of all 16 Lynnhaven DEQ stations combined for 2004 and 2005.

The nitrogen and phosphorus are major nutrients that can be used for photosynthesis. In particular, NH₄, NOₓ, and dissolved phosphorus are species that can be uptaken directly by the phytoplankton. Therefore, they are important indicator for the environmental quality. Nitrogen’s concentration is usually higher than phosphorus. The 1:1 plots of predicted vs. observed comparisons of NH₄, NOₓ, and DIP are shown in Figure VI.64. The absolute mean error and root-mean-square error of these water quality parameters show that the differences between model predictions and observations are within the range of natural variation in a given season of measurements when compared with available observation, for example, Figures VI.27-VI.30, VI.33-VI.34, VI.41-VI.44, VI.47-VI.48, VI.55-VI.58, and VI.61-VI.62.

Table VI.2. Statistical summary of errors derived by comparing predicted vs. observed values of NH₄, NOₓ, and DIP for all 16 Lynnhaven DEQ stations for 2004 - 2005.

<table>
<thead>
<tr>
<th>Parameter:</th>
<th>NH₄</th>
<th>NOₓ</th>
<th>DIP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sample Size</td>
<td>45</td>
<td>45</td>
<td>45</td>
</tr>
<tr>
<td>Mean Error</td>
<td>0.00</td>
<td>0.01</td>
<td>0.02</td>
</tr>
<tr>
<td>Absolute Mean Error</td>
<td>0.03</td>
<td>0.02</td>
<td>0.02</td>
</tr>
<tr>
<td>RMS Error</td>
<td>0.04</td>
<td>0.02</td>
<td>0.02</td>
</tr>
<tr>
<td>Relative Error</td>
<td>0.48</td>
<td>0.42</td>
<td>0.47</td>
</tr>
<tr>
<td>Corr. Coeff. (r)</td>
<td>0.22</td>
<td>0.58</td>
<td>0.70</td>
</tr>
</tbody>
</table>
Figure VI.64. Plots of 1:1 predicted vs. observed NH₄, NOₓ, and DIP TP at all 16 Lynnhaven DEQ stations for 2004 - 2005.
VI-3 Validation of the Sediment Transport Model

For validation of the sediment transport model, two observation datasets were utilized:

1) High-frequency, continuously measured turbidity time series data from 3 VIMS deployments in 2005 were used to validate the sediment transport model, based on a derived correlation between turbidity and TSS. Station locations for these 3 deployments are shown in Figure VI.65. Comparisons of the modeled TSS values and those derived from these high-frequency turbidity measurements are shown in Figure VI.66. Whereas the magnitudes of the modeled sediment concentration generally agreed with those derived from turbidity measurements, detailed variations did not completely match, probably due to the uncertainty between observed turbidity and TSS.

2) To confirm the model performance over the full spatial domain, predictions from model simulations for both 2004 and 2005 were used to compare to DEQ data at all 16 Lynnhaven stations. These comparisons are shown in Figures VI.67-VI.68, VI.69-VI.70, and VI.71-VI.72, respectively, for the Western, Eastern, and Broad Bay/Linkhorn Bay Branch DEQ stations of the Lynnhaven.

Inspection of Figures VI.67 through VI.72 shows that the model, in general, reproduced TSS concentrations at all stations reasonably well. It should be noted that no parameters were altered for the simulations of validation years 2004 and 2005.

Figure VI.65. Station locations for high-frequency measurements of turbidity in 2005 in the Lynnhaven River system.
Figure VI.66. Predicted TSS vs. TSS derived from high-frequency turbidity measurements at 3 locations in the Lynnhaven in 2005.
Figure VI.67. Predicted vs. observed TSS at Western Branch DEQ stations for 2004.

Figure VI.68. Predicted vs. observed TSS at Western Branch DEQ stations for 2005.
Figure VI.69. Predicted vs. observed TSS at Eastern Branch DEQ stations for 2004.

Figure VI.70. Predicted vs. observed TSS at Eastern Branch DEQ stations for 2005.
Julian Date 2004

Figure VI.71. Predicted vs. observed TSS at Broad Bay / Linkhorn Bay Branch DEQ stations for 2004.

Julian Date 2005

Figure VI.72. Predicted vs. observed TSS at Broad Bay / Linkhorn Bay Branch DEQ stations for 2005.
CHAPTER VII. SENSITIVITY ANALYSIS ON BENTHIC MICROALGAE DYNAMICS

The shallow water region (SWR) of coastal marine ecosystems, such as the Lynnhaven River system with depths less than 3-5 meters, encompasses the land-water margin and serves as the buffer zone for the transport of nutrients between land and water. When light can penetrate through the water column and reach the bottom, it triggers benthic microalgae (BMA) to perform photosynthesis, resulting in oxygen and nutrient benthic-pelagic exchange fluxes. BMA and their consumers are essential components of the Lynnhaven ecosystem; they uptake more nutrients and are more labile than vascular plants, and thus are clearly a source for fueling secondary primary production.

VII-1 Benthic Microalgae Model Formulation

The present model framework for benthic microalgae was inspired by the previous studies by Cerco and Seitzinger (1997) and Blackford (2002). The key variables determining the biomass of BMA are irradiance at the sediment surface, the self-shading of BMA, nutrients in the water column and sediment concentration, temperature, metabolism, and grazing rate. Figure VII.1 presents the conceptual diagram of the BMA model. BMA dynamics influence several biochemical processes: oxygen and nutrient fluxes between the water column and sediments, oxic layer thickness in the sediment, and the particulate organic material concentration in the sediment. All these processes have been built into the CE-QUAL-ICM model for its application to the Lynnhaven River system.

VII-1-1 Modeling biomass of BMA

BMA reside in a thin layer between the water column and sediments and its biomass is determined by the balance of production, respiration, and predation:

$$\frac{\delta B}{\delta t} = (P - BM - PR)B$$  \hspace{1cm} (VII-1)

where:
- \(B\) = BMA biomass, as carbon (gm C m\(^{-2}\))
- \(P\) = production rate (d\(^{-1}\))
- \(BM\) = basal metabolism (respiration) rate (d\(^{-1}\))
- \(PR\) = predation rate (d\(^{-1}\))

The production (growth) was determined by available light, nutrients, and ambient temperature:

$$P = P^m_b m^* f(I)^* f(N)^* f(T)$$  \hspace{1cm} (VII-2)

where:
- \(P^m_b\) = maximum production rate under optimal conditions (g C g\(^{-1}\) Chl d\(^{-1}\))
f(I) = effect of suboptimal light conditions
f(N) = effect of limited nutrient availability
f(T) = effect of temperature

Figure VII.1. Framework of benthic algae model.
Light effect
Available light for BMA photosynthesis is the key factor to control the biomass of BMA. For example, the BMA biomass variability in the Southeastern Kattegat is 70% controlled by light availability (Sundbäck, 1984). The effect of light on production is expressed as:

\[ f(I) = \frac{I}{\sqrt{I^2 + IK^2}} \]  \hspace{1cm} (VII-3)

where:
\[ I = \text{local irradiance} \]

As is done for phytoplankton, the parameter \( I_k \) is defined as the irradiance at which the initial slope of the production vs. irradiance relationship intersects the value of \( P_B^m \):

\[ IK = \frac{P_B^m}{\alpha} \]  \hspace{1cm} (VII-4)

where:
\[ \alpha = \text{initial slope of production vs. irradiance relationship (g C g^{-1} Chl (E m^{-2})^{-1})} \]

Local irradiance varies within the BMA layer due to BMA self-shading and extinction due to sediment solids:

\[ I = I_s e^{-Ksz} \]  \hspace{1cm} (VII-5)

where:
\[ I_s = \text{irradiance at surface of BMA layer (same as irradiance at bottom of water column)} \]
\[ K_s = \text{light attenuation within BMA layer due to BMA self-shading and sediment (m^{-1})} \]
\[ z = \text{local coordinate measured down from surface of algal layer} \]

Self-shading has been cited as an important factor influencing BMA (Cahoon and Cooke, 1992). Consequently, it is reasonable to separate \( K_s \) (light attenuation) into two terms; one is self-shading related to the BMA biomass, and the other is sediment solids extinction. The mean light within the BMA layer is represented as:

\[ I_{mean} = I_o e^{-K_{sed}} \frac{1 - e^{-K_{algae}B}}{K_{algae}B} \]  \hspace{1cm} (VII-6)

where:
\[ I_{mean} = \text{available light within BMA layer} \]
\[ I_o = \text{irradiance at the surface of sediment} \]
\[ K_{sed} = \text{attenuation due to sediment solid} \]
\[ K_{algae} = \text{attenuation due to benthic microalgae self-shading (m^2 g^{-1} C)} \]
\[ B = \text{benthic microalgae biomass (g C m^{-2})} \]
Equation (VII-6) mainly constrains unlimited growth of BMA. When the biomass of BMA becomes larger, the mean light within the BMA layer will be smaller. As a result, BMA growth will be limited. Irradiance at the surface of the BMA layer is calculated from the irradiance at the surface of the water column through the following equation:

\[ \text{Ke} = \text{a}_1 + \text{a}_2 \cdot TSS + \text{a}_3 \cdot \text{CHL} \]  

(VII-7)

where:

- \( \text{a}_1 \): background attenuation (m\(^{-1}\))
- \( \text{a}_2 \): attenuation by inorganic suspended solids (m\(^2\) g\(^{-1}\))
- \( \text{a}_3 \): attenuation by organic suspended solids (m\(^2\) gm\(^{-1}\) CHL)
- TSS: total suspended solids concentration (g m\(^{-3}\))
- CHL: chlorophyll-a concentration (mg CHL m\(^{-3}\))

**Nutrients**

The influence of nutrients on BMA production is represented by the Monod formulation:

\[ f(N) = \frac{N}{\text{Kh} + N} \]  

(VII-8)

where:

- \( N \): concentration of nutrient available for BMA uptake (g m m\(^{-2}\))
- \( \text{Kh} \): nutrient concentration at which algal uptake is halved (g m m\(^{-2}\))

There are two nutrient sources for BMA, one from the water column and the other from returned nutrients as they diffuse from the sediment into the overlying water column. A nutrient concentration available on an areal basis is calculated as follows:

\[ N = \text{Nflux} \cdot \Delta t + \text{Nwater} \cdot H_{\text{water}} \]  

(VII-9)

where:

- \( \text{Nflux} \): sediment nutrient release (g m\(^{-2}\) d\(^{-1}\))
- \( \Delta t \): discrete time step (day)
- \( \text{Nwater} \): nutrient concentration in overlying water (g m\(^{-2}\))
- \( H_{\text{water}} \): depth of bottom layer (m)

Two nutrients potentially limit BMA production: dissolved inorganic nitrogen and phosphorus. As in the case for phytoplankton, Liebig’s “law of the minimum” (Odum, 1971) is used. Therefore, nutrient limitation is determined by the most limiting nutrient. Based on the reported value, half-saturation constants were set as \( \text{K}_{hn} = 0.01 \text{ g N m}^{-2} \) for nitrogen and \( \text{K}_{hp} = 0.001 \text{ g P m}^{-2} \) for phosphorus (Cerco and Seitzinger, 1997). It is
assumed that silica is not a limiting factor in the present BMA model, even though benthic diatoms can uptake silica.

Temperature
Temperature is also shown to have a strong effect on production, respiration, and grazing rates. For example, temperature was recognized to account for up to 70% of the variability of microphytobenthic populations (Uthicke and Klumpp, 1998). The effect of temperature on algal production is represented by a function similar to a Gaussian probability curve:

\[
f(T) = \exp\left(- K_{T1}(T - T_M)^2\right) \quad \text{when } T \leq T_M \\
= \exp\left(- K_{T2}(T - T_M)^2\right) \quad \text{when } T > T_M
\]  

(VII-10)

where:

- \(T_M\) = optimal temperature for BMA growth (°C)
- \(K_{T1}\) = effect of temperature below \(T_M\) on BMA growth (°C\(^{-2}\))
- \(K_{T2}\) = effect of temperature above \(T_M\) on BMA growth (°C\(^{-2}\))

As a result, BMA production increases as a function of temperature until an optimum temperature is attained, and then decreases with temperature after an optimum temperature is reached.

Basal metabolism (Respiration)
Basal metabolism is commonly considered to be an exponentially increasing function of temperature:

\[
BM = BMR \times \exp[K_{TB}(T - TR)]
\]  

(VII-11)

where:

- \(BMR\) = metabolic rate at reference temperature \(TR\) (day\(^{-1}\))
- \(K_{TB}\) = effect of temperature on metabolism (°C\(^{-1}\))
- \(TR\) = reference temperature for metabolism (°C)

Predation
Predation is calculated by a relationship identical to that for respiration:

\[
PR = BPR \times \exp[K_{TB}(T - TR)]
\]  

(VII-12)

where:

- \(BPR\) = predation rate at \(TR\) (day\(^{-1}\))
- \(K_{TB}\) = effect of temperature on predation (°C\(^{-1}\))
- \(TR\) = reference temperature for predation (°C)

The rates of both metabolism and predation for BMA both increase with temperature. The differences lie in the parameter values, and their distribution.
VII-2 Nutrient Budgets in the Lynnhaven River

A nutrient budget provides a basis for assessing potential effects of system responses in the context of various sources and sinks. The purposes for constructing the nutrient budget were: (1) to present the nutrient pathway on an annual basis, especially under the scenarios of with and without the effects from BMA, (2) to evaluate the relative importance of the various sources and sinks of nitrogen and phosphorus during the seasonal cycle from the monthly nutrient budget, (3) to estimate recycling processes in order to allow estimates of turnover times and the relative importance of “new” versus “recycled” nutrients, and (4) to quantify nutrient export to the coastal ocean and losses from the sediment on an annual basis comparing with results from deep water systems (Nixon et al., 1996).

In order to quantify the nutrient budget in an estuary, both nutrient storage in sediments and nutrient exchange with the ocean and the atmosphere must be quantified. Nutrient storage in sediments is difficult to measure in the field due to large spatial and temporal gradients (Boynton et al., 1995). Nutrient exchange with the outside ocean is complicated by tidal currents, with large temporal and spatial gradients (Kjerfve and Proehl, 1979). Therefore, the nutrient budget calculation from a well-calibrated numerical model represents one of the most efficient and accurate ways to achieve the goal.

VII-2-1 Annual nutrient budget in Lynnhaven River system

This Lynnhaven hydrodynamic/water quality model comprises the estimate of major inputs, exports, storages, and recycling of TN and TP in the Lynnhaven River proper and its branches. There are two types of nutrient inputs into the system including nonpoint loading from watershed and atmospheric sources. Loss terms include burial of TN and TP in sediments in depositional portions of study areas, denitrification of N in sediments, and net exchanges of N and P at the mouth of the river. Since it is probably a small source as is the case in most nutrient-rich estuarine systems, nitrogen fixation is not evaluated (Howarth et al., 1988).

The conceptual model of the nutrient budget can be expressed as differential equations for TN and TP both in the water column and in the sediment based on Boynton et al. (1995). In the water column, the time rates of change of TN and TP vary with nonpoint, atmospheric and depositional fluxes, and oceanic sources:

\[
\frac{dTN_w}{dt} = TN_{\text{nonpoint}} + TN_{\text{atm}} - TN_{\text{dp}} + TN_{\text{flux}} + TN_{\text{ocean}} \quad (VII-13)
\]

\[
\frac{dTP_w}{dt} = TP_{\text{nonpoint}} + TP_{\text{atm}} - TN_{\text{dp}} + TN_{\text{flux}} + TN_{\text{ocean}} \quad (VII-14)
\]

In the sediment, the important processes impacting the time rates of changes of TN and TP include deposition, flux, burial, and denitrification:
\[
\begin{align*}
\frac{dTN_w}{dt} &= TN_{dp} - TN_{flux} - TN_{burial} - TN_{denitri} \\
\frac{dTP_w}{dt} &= TP_{dp} - TP_{flux} - TP_{burial}
\end{align*}
\]  
(VII-15)  
(VII-16)

where:
- \(TN_w, TP_w\) = total nitrogen, phosphorus in water column
- \(TN_s, TP_s\) = total nitrogen, phosphorus in sediment
- \(TN_{nonpoint}, TP_{nonpoint}\) = total nitrogen, phosphorus loading from nonpoint source
- \(TN_{atm}, TP_{atm}\) = total nitrogen, phosphorus loading from atmosphere
- \(TN_{dp}, TP_{dp}\) = total nitrogen, phosphorus deposition into sediment
- \(TN_{flux}, TP_{flux}\) = total nitrogen, phosphorus flux from sediment into water column
- \(TN_{ocean}, TP_{ocean}\) = net total nitrogen, phosphorus exchange with adjacent seaward system
- \(TN_{burial}, TP_{burial}\) = total nitrogen, phosphorus burial in deep sediment
- \(TN_{denitri}\) = total nitrogen, phosphorus denitrified in sediment

Annual nutrient budget in the mainstem of Lynnhaven River
The mean annual water quality budget in the Lynnhaven River was studied first. It was assumed that, on an annual basis, the nutrient species are in an equilibrium condition. Consequently, \(\frac{dTN_w}{dt}, \frac{dTP_w}{dt}, \frac{dTN_s}{dt}\) and \(\frac{dTP_s}{dt}\) are equal to zero by definition. The results of annual nutrient budgets are shown in Figure VII.2 and Figure VII.3 (values in parentheses denote results without BMA).

Annual TN and TP budgets, reported in units per square meter of surface area of Lynnhaven River, show the loading of nutrients from the watershed we calculated is slightly less than that for Chesapeake Bay (Boynton et al., 1995). Our loading for Lynnhaven River is 27.79 (mg N m\(^{-2}\) d\(^{-1}\)) for TN and 2.08 (mg P m\(^{-2}\) d\(^{-1}\)) for TP. In a previous comparison with Chesapeake Bay loading of nutrients from its watershed, TN loading was 36.01 (mg N m\(^{-2}\) d\(^{-1}\)) and TP loading was 2.67 (mg P m\(^{-2}\) d\(^{-1}\)). The ratio of the Lynnhaven River watershed (166 km\(^2\)) to the surface area of the receiving waters (18.1 km\(^2\)) is 9.2. This ratio for Chesapeake Bay is 14.4 (165,760 km\(^2\) watershed area, 11,542 km\(^2\) surface area of its receiving waters). The atmosphere deposition directly deposited through the surface of the river contributed only 9.5\% for TN and 4.4\% for TP. While direct atmospheric deposition represents a very small nutrient source compared to nonpoint sources from the watershed, the influence of atmospheric deposition on primary production may be larger. The reason for this is that a substantial fraction of TN and TP entering from watershed sources is in a form not directly available to phytoplankton, being either dissolved organic nutrient or a form of particulate material. However, virtually all of the nitrogen and phosphorus deposited from the atmosphere is immediately available for phytoplanktonic uptake.

180
Figure VII.2 and Figure VII.3 show the results of the water quality model simulation and indicate that, over lengthy time scales, benthic algae can influence most terms of the nutrient budget in the water column. The presence of BMA reduced the export of nutrients into Chesapeake Bay. There are two reasons: 1) a larger quantity of particulate nitrogen and phosphorus deposit into the sediment in the presence of BMA and 2) for nitrogen flux between the water column and sediment with BMA, the flux direction changed from traditional flux in that the BMA uptake dissolved nutrients both from the sediment and the water column, which causes the net dissolved nitrogen flux to occur from the water column into the sediment. For phosphorus flux between the water column and the sediment with BMA, the flux direction does not change, but less dissolved phosphorus is released from the sediment due to BMA uptake. The nutrients that are uptaken by BMA are stored in the sediment in winter and spring, and released from the sediment as dissolved nutrient in summer and autumn. Simulations indicate that larger quantities of dissolved nitrogen are incorporated into the sediments in the presence of benthic algae. Deposition of particulate nitrogen computed in the presence of benthic algae also increases. Enhanced deposition results from the stimulation of primary production in the water column by summer nutrients released in the presence of benthic algae.

The computed net annual flux of dissolved phosphorus is from the sediments to the water column, both with and without the effects of benthic algae (Figure VII.3). Annual average sediment release is diminished when algae are present, however, due to uptake during periods of benthic production. The simulation indicates that Lynnhaven River would export more phosphorus to the ocean in the absence of benthic algae.

Figure VII.2 and Figure VII.3 also indicate that benthic algae can influence burial and denitrification in sediment. For both particulate nitrogen and particulate phosphorus, computed deposition and burial is increased in the presence of benthic algae. As a result of the uptake by BMA and enhanced deposition, more nitrogen and phosphorus are buried into deep, unavailable sediments instead of being exported into Chesapeake Bay without benthic algae. The denitrification rate also increased due to BMA. In general, the annual averaged denitrification rates with BMA and without BMA are within the range 5 to 250 µmol N m⁻² h⁻¹ (1.68 mg N m⁻² d⁻¹ to 84 mg N m⁻² d⁻¹) reported for several estuarine systems (Andersen et al., 1984; Seitzinger, 1988; 1990; Rysgaard et al., 1993; 1995; Nowicki et al., 1997; Sundbäck et al., 2000). The highest denitrification rate, 98 mg N m⁻² d⁻¹, occurred in the late summer during the simulation including BMA. There are also several studies that show extremely high denitrification rates of approximately 500 to 1300 µmol N m⁻² h⁻¹ (168 to 437 mg N m⁻² d⁻¹) in some estuarine sediments (Seitzinger, 1988; 1990; Ogilvie et al., 1997; Dong et al., 2000).

Annual nutrient budget in the three tributaries of Lynnhaven River
In the Lynnhaven River, there are three major branches: Western Branch, Eastern Branch, and Broad Bay. Their dynamics are different. It is valuable to characterize the difference between these three branches. For example, which tributary receives the majority of the nutrient loading from the watershed? Which tributary exports the largest quantities of nutrients into Chesapeake Bay? Using the same methodology described
earlier, nutrient budgets in the three branches of Lynnhaven River were calculated (Figure VII.4 and Figure VII.5).

The results show that the Western and Eastern Branches receive significantly more nutrients than does Broad Bay. While the combined surface areas of the Western and Eastern Branches (11.1 km²) comprise only 61% of the entire system (18.1 km²), the percentage for nutrient loadings are 85% for TN and 83% for TP contributed from the watershed. The largest areal loadings of TN and TP are in Western Branch, which are almost 5 times and 4 times those in Broad Bay for TN and TP, respectively.

Figure VII.2. Annual Total Nitrogen budget (mg N m⁻² d⁻¹) for Lynnhaven River (Values in parentheses indicate results without BMA)
Figure VII.3. Annual Total Phosphorus budget (mg P m$^{-2}$ d$^{-1}$) for Lynnhaven River (Values in parentheses indicate results without BMA)
Figure VII.4. Annual Total Nitrogen budget (mg N m⁻² d⁻¹) in three branches of Lynnhaven River (WB: Western Branch, EB: Eastern Branch, BB: Broad Bay)
Figure VII.5. Annual Total Phosphorus budget (mg P m$^{-2}$ d$^{-1}$) in three branches of Lynnhaven River (WB: Western Branch, EB: Eastern Branch, BB: Broad Bay)
It is not surprising that most nutrients exported from the Lynnhaven into Chesapeake Bay are from the Western and Eastern Branches. With larger nutrient loadings, the Western and Eastern Branches contribute approximately 90% of TN and 89% of TP exported into Chesapeake Bay. The removal of nutrients via ocean exchange, as a percentage of TN input to the estuary, also varies between the three branches. The Western Branch exports 34% of its TN loading and 31% of its TP loading, the Eastern Branch exports 35% of its TN loading and 29% of its TP loading, and Broad Bay only exports 20% of its TN loading and 15% of its TP loading.

The difference appears to be due to different residence times for the three branches. From the results of an “age-of-water” investigation, we know that the residence time of either the Western or Eastern Branch is approximately 12 days for the mean flow condition, which is much smaller than that of Broad Bay, 72 days. Nixon (1996) showed that the net transport of nutrients through a system to the outside ocean is inversely correlated with the residence time of water in the system. With larger nutrient loading, the Western and Eastern Branches also show larger values of particulate nutrient deposition, dissolved nutrient flux, final burial into deep sediment, and denitrification rates than these values for Broad Bay.

VII-2-2 The monthly nutrient budget for the Lynnhaven River system

There are other time scales, such as seasonal time scales, that are important for the nutrient budget. The monthly nutrient budget was calculated using the formula presented above. For the water column, the monthly budget for the entire year is shown in Figure VII.6. It indicates that nonpoint sources account for most external loadings of nitrogen and phosphorus to the Lynnhaven River through the entire year. Atmospheric nitrogen and phosphorus loadings are almost constant throughout the year. From October through April, the sediment is the major sink of nitrogen from the water column. From May to September, sediments release remineralized nitrogen to the water column and function as a source. During July and August, sediment-released nitrogen is larger than the nonpoint source loading. From November through March, Lynnhaven River exports nitrogen to the Chesapeake Bay. During the rest of the year, nitrogen imports from the ocean are substantial. The monthly budget for phosphorus also reveals a similar pattern. From October through March, the sediment is the major sink. From April to September, sediments act as a source by releasing phosphorus to the water column. From October through February, the Lynnhaven River exports phosphorus to the Chesapeake Bay. Similar monthly patterns of the nutrient budget were found by Cerco and Seitzinger (1997) for their analysis of the Indian River-Rehoboth Bay system.

The sediment nutrient budget was also calculated (Figure VII.7). During winter and spring, sediments are net sinks of nutrients from the water column. Settling of nutrients in particulate form is one component of the nutrient budget during these months. In addition, BMA also uptake dissolved inorganic nutrients. Benthic fluxes of total dissolved nutrients are dominated by uptakes throughout the spring and winter. Benthic microalgae can assimilate a large proportion of the nitrogen and phosphorus and produce oxygen in the sediments (Ferguson et al., 2004).
Figure VII.6. Monthly Total Nitrogen budget (mg N m\(^{-2}\) d\(^{-1}\)) and Total Phosphorus budget (mg P m\(^{-2}\) d\(^{-1}\)) in the water column for Lynnhaven River (positive means entering the water column, and negative means leaving the water column)
Figure VII.7. Monthly Total Nitrogen budget (mg N m\(^{-2}\) d\(^{-1}\)) and Total Phosphorus budget (mg P m\(^{-2}\) d\(^{-1}\)) in sediment for Lynnhaven River (positive values indicate leaving sediment, negative values indicate entering sediment)
In April or May, the system undergoes a change as the sediment begins to release nutrients. It is possible that, in this condition, the extra pelagic production and resulting light extinction would decrease BMA production, leaving an unsustainable benthic respiratory requirement. Meanwhile, phytoplankton assimilate dissolved nutrients in the water column, which lowers the concentration of dissolved nutrients. The coupled effects cause sediments to release dissolved nutrients into the water column. Cerco and Seitzinger (1997) also indicated that this change is caused by phytoplankton shading out benthic algae and primary production in the water column exceeding production in the sediments. When temperatures become relatively high during this period, phytoplankton in the water column receive more light than BMA in the sediment. Mineralization of the organic matter in the sediments also increases with high temperature in summer, and dissolved inorganic nutrients are released from the sediment and support the primary production in the water column. Phytoplankton growth exceeds BMA, since light available to BMA decreases due to shading by phytoplankton. In summer and autumn, the sediments are a net source of nutrients to the water column.

In order to illustrate the influence of BMA uptake on sediment dissolved nutrient flux, Figure VII.8 shows the monthly dissolved nitrogen and phosphorus assimilated by BMA with the total and net nutrient fluxes. The total nutrient flux, without BMA uptake, indicates that sediment released both nitrogen and phosphorus over the entire year. The most intense period of release of nutrients from the sediment occurred in summer. The dissolved nutrients assimilated by BMA exceeded the released nutrients in winter and spring, while the released nutrients from the sediment dominated in summer and autumn. In summary, BMA could reverse the direction of nutrient sediment flux in early spring and late autumn.

VII-3 Comparison of Nutrient Budget between Shallow and Deep Water Systems

In deep estuaries, sediment-regenerated nutrients often account for the majority of the total nutrients regenerated. For example, the annual sediment releases of nitrogen and phosphorus ranged from 55% to 233% and 44% to 2140%, respectively, of their annual terrestrial plus atmospheric inputs. The most intense sediment nutrient flux from the sediment into the water column occurred in summer. In Lynnhaven River, however, the annual sediment flux of nitrogen is from the water column into the sediment. From monthly budget results, it is clear that the sediment still releases nitrogen in summer and fall as in deep estuaries, but the BMA in the sediment uptake nitrogen from the water column in winter and spring. The overall effect of annual sediment nitrogen flux is from the water column into the sediment. Meanwhile, the uptake effect of BMA also reduces the magnitude of the phosphorus flux from the sediment into the water column.

In most estuaries, nutrient loadings are dominated by freshwater inputs during spring. With abundant nutrients in the water column, phytoplankton usually bloom in spring, for example, in Chesapeake Bay (Kemp and Boynton, 1984; Malone et al., 1988). After phytoplankton decay and sink into the sediment, the recycling of nutrients from the sediments then supports further phytoplankton productivity in the summer (Kemp and Boynton, 1984; Rysgaard et al., 1995). It appears that nutrient cycling in these systems
Figure VII.8. Monthly BMA uptake contribution to sediment flux nitrogen and phosphorus for Lynnhaven River (Positive values indicate leaving sediment, negative values indicate entering sediment)
occurs over reasonably broad, seasonal time scales. In Chesapeake Bay, nutrients are removed from the water column during the spring phytoplankton bloom and are subsequently deposited in the sediments as detritus. A spring phytoplankton bloom has not been observed in the Lynnhaven River. However, the benthic algal bloom plays the role of the phytoplankton bloom in the deeper system. After BMA assimilates nutrients in winter and spring, nutrients stored in particulate form enter into the sediment. The microbial processes are responsible for nutrient regeneration in sediments, which are sensitive to temperature and oxygen conditions. In summer, nutrients are released from the sediment and support the water column primary production. Overall, mineralization of the organic matter stored in the sediments by BMA supports the summer maximum in the annual primary production.

Nixon et al. (1996) showed that the net transport of nutrients through estuaries to the continental shelf is inversely correlated with residence time of water in the system. Without BMA, the annual nutrient budget indicates that 70% of TN and 73% of TP, respectively, entering from land and atmosphere would be exported into Chesapeake Bay. These estimations of the efficiency of nitrogen and phosphorus transports through the Lynnhaven River fit well with the findings of Nixon et al. (1996), assuming that the residence time of water in the Lynnhaven River is 35 days (Figure VII.9). With the BMA, however, only 32% of TN and 26% of TP entering would be exported into Chesapeake Bay. This indicates that, as nutrients transported through the Lynnhaven River, more nutrients could be removed from the water column due to BMA uptake and subsequently through the buried and denitrified in sediments. This provides an alternative mechanism for the nutrient pathway in the shallow water system.

Figure VII.9. The percent of total nitrogen and phosphorus input from land and atmosphere that is exported from a sample of estuaries and lakes as a function of mean residence time in the system. Estuarine data marked as solid points; lake data marked as open circles (Nixon et al., 1996; modified); regression equations calculated by Nixon et al. (1996) (Blue dot shows results without BMA; red dot shows results with BMA)
CHAPTER VIII. DISCUSSION AND CONCLUSIONS

The Virginia Institute of Marine Science (VIMS) has successfully developed an integrated numerical modeling framework for the Lynnhaven system, a shallow water coastal bay in the City of Virginia Beach, Virginia. This framework combines a high-resolution 3D hydrodynamic model (UnTRIM) that provides the required transport for a water quality model (CE-QUAL-ICM) that, in turn, provides intra-tidal predictions of 23 water quality state variables. A suspended sediment transport model was also developed and incorporated into the modeling framework.

The hydrodynamic model UnTRIM is a state-of-the-art numerical model using an unstructured grid, which is able to follow complex shoreline geometry more closely than the traditional structured grid. This feature is particularly important for application to a shallow water body like the Lynnhaven system. The percent error in water volume due to any inaccuracy of the fitting of the model grid to the shoreline is amplified when the relative volume of deeper water decreases with decreasing overall depth. The UnTRIM model employs an Eulerian-Lagrangian approach and a semi-implicit numerical scheme to solve the momentum equation, thus eliminating the constraint of Courant’s condition and allowing a much larger time step (of the order of 10 minutes) in numerical computation. This is advantageous over the hydrodynamic model using the Eulerian approach, since the model needs to run for an extended period, normally longer than the annual cycle, to supply transport to the water quality model for evaluating seasonal variations in water quality conditions. The selection of CE-QUAL-ICM was based on its history of application to the Chesapeake Bay system. However, it was later deemed necessary to modify it by including the benthic microalgae for the application to the Lynnhaven system.

Prior to the inception of the model development, all available historical Lynnhaven hydrodynamic and water quality data were amassed in a Microsoft ACCESS database and analyzed for model calibration suitability and long-term trends. These data were collected from monitoring programs of the Virginia Department of Environmental Quality (VA-DEQ) and the Virginia Health Department, Shellfish Sanitation Division (VA-DSS), intensive surveys conducted by VIMS and Malcolm Pirnie Environmental Engineers, and tidal surveys conducted by the National Oceanic and Atmospheric Administration (NOAA).

A strategy of project-specific field surveys and laboratory experiments was devised based on which measurements would complement the existing historical data and be most useful to the model calibration and validation processes. These field surveys and experiments included the following:

- a hydrodynamic survey of synoptic measurements of times series of surface elevations plus currents and salinities in all Lynnhaven branches and outside the Inlet
- seasonal sediment flux measurements at the Inlet and in all branches to determine the spatial and seasonal variations of the fluxes from the water column to the sediment (and vice versa) of dissolved oxygen, ammonia, nitrate-nitrite, and phosphate

- sediment flux measurements of dissolved oxygen, ammonia, nitrate-nitrite and phosphate in the laboratory under controlled environments

- critical shear stress measurements at multiple sites in the basin to determine the spatial and seasonal variations to the erodibility of bottom sediments

- high-frequency time series measurements of chlorophyll-a, turbidity, Colored Dissolved Organic Matter (CDOM), and dissolved oxygen (DO) to evaluate water quality conditions with high temporal resolution

The analyses of sediment flux data of laboratory experiments clearly have indicated that benthic microalgae (BMA) play a significant role in the pelagic-benthic exchange process in the Lynnhaven system. The importance of the BMA process in shallow waters has been documented by other studies in various water bodies. Therefore, a microalgae model was developed based on the experimental data and literature formulations, and incorporated into the water quality model CE-QUAL-ICM. The BMA growth can reduce the rates, or even reverse the directions, of nutrient and oxygen exchanges between the water column and sediment, and significantly affect the nutrient budget of a water body. The photosynthesis of BMA would assimilate nutrients from the water column, store them in the sediment, and further bury them into deep sediment, or nitrify them in the case of nitrogen. Therefore, fewer nutrients would be exported out of the system. The VIMS model study indicated that 32% of total nitrogen and 26% of total phosphorus inputs into the Lynnhaven system were exported to the Chesapeake Bay.

The hydrodynamic portion of the integrated model was calibrated using historical datasets and NOAA tide predictions. The water quality portion of the model was calibrated using the 2006 data set collected by the VA-DEQ. The calibration parameters were adjusted, within their literature ranges, to achieve the best agreement between the model predictions and observation data.

Validation of the hydrodynamic model was made by comparing the 2005 simulation results with observations collected in VIMS hydrodynamic surveys of that year. Validation of the water quality model was conducted with a two-year model run simulating the water quality conditions of 2004-2005. The model predictions were compared with the monitoring data of VA-DEQ. Satisfactory agreements between the model predictions and field observations were achieved without altering any values of calibration parameters that were set in the calibration process.

The sediment transport model was developed utilizing the equilibrium critical shear stress defined at the interface between layers, and incorporated into the modeling framework. The values of some model parameters were derived from the critical shear stress.
measurements conducted specifically for the project, and the others were from literature reports. This model was calibrated by comparing its predictions of total suspended solids (TSS) with observations at the 16 Lynnhaven VA-DEQ stations during 2006 and validated by comparing the 2004-2005 model results with VA-DEQ observations for those years. Additionally, the validation compared model predictions with TSS values derived from VIMS high-frequency measurements of turbidity at 3 locations in 2005.

The model sensitivity analyses showed that 70% of total nitrogen and 73% of total phosphorus would have been exported if there were no BMA growth in the system. The CE-QUAL-ICM could not have successfully simulated the water quality conditions in the Lynnhaven system without the modification of including BMA. The BMA model developed by VIMS accurately predicted the oxygen and nutrient water-sediment flux measurements in the laboratory for various seasons and different locations. The addition of BMA model enabled the CE-QUAL-ICM to successfully simulate the water quality conditions in the Lynnhaven system.

There are two water quality problems identified through data analyses and model simulations. One is the degraded water clarity due to significant concentrations of suspended sediment. The other is the localized summertime low dissolved oxygen in headland areas. The modeling framework developed by VIMS is ready for its application in conducting scenario runs. The model should be used as a management tool to assess the effectiveness of alternative managing practices to mitigate these problems.
IX. REFERENCES


Cerco, C.F., Johnson, B., and Wang, H. (2002). Tributary refinements to the Chesapeake Bay model. ERDC TR-02-4, US Army Engineer Research and Development Center, Vicksburg, MS.


Keulegan, G. H. (1967). *Tidal Flow in Entrances Water-Level Fluctuations of Basins in Communications with Seas*. Technical Bulletin No. 14, Committee on Tidal Hydraulics, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.


