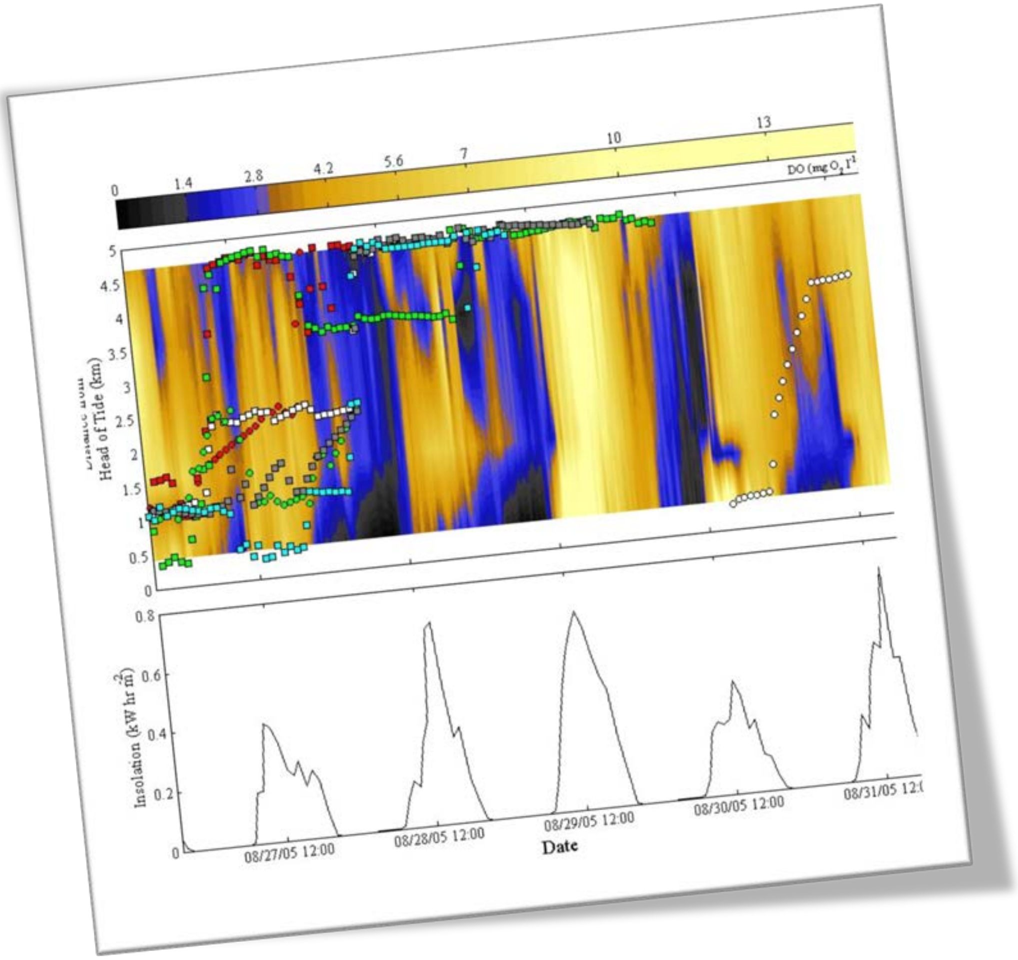
TMDL Model and Data Evaluation for Delaware’s Inland Bays:

Modeling Diel-cycling Hypoxia in Delaware’s Inland Bays



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EXECUTIVE SUMMARY

In March 2004, Entrix, INC. and J.E. Edinger Associates submitted their final report describing the Total Daily Maximum Load model currently used for Delaware’s Inland Bays (DIB). The model is a fully coupled 1-D watershed and 3-D hydrodynamic-water quality model called the Generalized Environmental Modeling Surface Water System (GEMSS). That model was primarily used to calculate water quality constituents such as nitrogen, phosphorus (particulate/dissolved, inorganic/organic) and dissolved oxygen (DO). The model was calibrated using data collected from 1998-2000. These three years represented the most data-rich period with respect to nutrient-loading on record. Since 2000, there has been a significant increase in both understanding and data collection in Delaware’s Inland Bays. In particular, the University of Delaware and Delaware’s Department of Natural Resources and Environmental Control have collected continuous data records for DO over many years and at many locations. DO is arguably the greatest potential water quality threat to Delaware’s Inland Bays, with multiple fish kills attributable to hypoxia occurring most years. Substantial research efforts have also demonstrated reduced growth rates and behavioral avoidance of hypoxia by juvenile estuary dependent fishes that rely on the Bays for Essential Fish Habitat. For this reason, the Center for the Inland Bays requested an independent assessment of the DO calculation in the GEMSS model. This report focuses on DO, but improvements in the understanding of nutrient loading and biogeochemical cycling will also be necessary to improve model formulations. The conclusions of this report are that GEMSS is not effective at simulating DO (especially in Indian River and tributaries). The calibration and validation datasets from 1998-2000 include few to no substantive continuous DO records. Assessing model performance in relation to diel-cycling hypoxia is exceedingly difficult, and that was not the original intent of that modeling effort. DO data from 2001-present contains DO fluctuations from 0% to 200% saturation in the headwaters of major creeks/tributaries and the model output contains no such fluctuations. The explanation for this is either: (1) diel-cycling hypoxia only became a significant feature of the water quality in Delaware’s Inland Bays in 2001 or (2) the monitoring program only became robust enough to detect diel-cycling hypoxia in 2001. In either case, the proliferation of data and understanding during the time period strongly argues for re-visiting the modeling framework for the Bays. Specific recommendations for future accurate simulations of diel-cycling hypoxia include:

- In this shallow estuary, benthic pelagic coupling between the water column and sediments is potentially a large source of oxygen demand. The current model uses fluxes measured from 1992-1993 and 2001. There is no mechanistic sediment flux model associated with this modeling effort. More recent flux measurements, a sediment flux model, and explicit incorporation of benthic algae will almost certainly be necessary to complete nutrient budgets. There is also potential for the sediment to play a role in time lags between the implementation of the Pollution Control Strategy and response in the estuary that cannot be explored in the current modeling framework.
- Incorporation of primary production and respiration rates into model calibration. Seasonal respiration appears well calibrated, but daily respiration rates are clearly not large enough to generate hypoxia in the early to late morning.
- Increased spatial resolution particularly in tidal headwaters where recent fish tagging evidence has highlighted potential fish exposure mechanisms reliant on spatial gradients in DO

- Incorporation of multiple meteorological records that were unavailable or offline during the calibration years (1998-2000) made available by the Delaware Environmental Observing System
- Secondary Recommendations:
    - Evaluation of sampling protocols for mobile dinoflagellate species that may require vertical profiles to accurately characterize vertically integrated water column primary production and respiration
    - Re-assessment of nutrient loading to include potential groundwater discharge being explored by many DIB researchers since 2000

INTRODUCTION

Hypoxia and low dissolved oxygen (DO) are terms used interchangeably to describe a DO condition that harms aquatic fauna (Diaz and Rosenberg 1995). This condition is commonly associated with eutrophication, a process whereby increased loading of organic matter and nutrients into aquatic ecosystems results in excessive algal growth (Nixon 1995, Valiela et al. 1997, Bricker et al. 1999). Respiration of living algae and the bacteria that decompose dead algae and other labile organic matter can reduce DO enough to impact fisheries (Breitburg 2002, Diaz et al. 2004, Breitburg et al. 2009) and in some cases disrupt or eliminate entire faunal communities (Diaz and Rosenberg 1995).

Eutrophication is a natural process (Nixon 1995) and low/no oxygen zones are a permanent natural feature of ocean basins (ocean minimum zones) and inland seas (e.g. the anoxic zone of the Black Sea) (Diaz and Rosenberg 1995). However, severe hypoxia (0.1 to 2.0 mg O_2 l^{-1}) and anoxia (< 0.1 mg O_2 l^{-1}) that now occur in coastal semi-enclosed seas (e.g. Baltic-Kattegat Sea - in Europe) and estuaries (e.g. Chesapeake Bay, Neuse River, Waquoit Bay, Delaware's Inland Bays - all in the USA) are relatively recent developments which have accelerated rapidly over the past 50 years in conjunction with human population growth (Diaz and Rosenberg 1995, Diaz 2001, Diaz et al. 2004, Diaz and Rosenberg 2008).

Coastal hypoxia is a warm season phenomenon and occurs in three temporal categories (seasonal, episodic, and diel-cycling) based on event duration (months to weeks, weeks to days, and hours to minutes, respectively). Duration is closely linked to the occurrence of a pycnocline, the presence and strength of which is heavily influenced by water column stability and depth. Seasonal hypoxia occurs beneath a strong pycnocline in bottom waters of relatively stable, deep systems (> 10 m) such as the Baltic-Kattegat Sea and the mainstem of Chesapeake Bay (Officer et al. 1984, Baden et al. 1990). Severe hypoxia develops during spring, may deteriorate further to anoxia, and can persist continuously for months, until fall. Episodic hypoxia occurs beneath a relatively weak pycnocline in bottom waters of relatively unstable, deep or shallow systems such as the lower York River (a large sub-estuary of Chesapeake Bay) or the Pamlico River (Albemarle-Pamlico Sound Complex, USA) in which severe hypoxia events can persist for weeks to days (Pihl et al. 1991, Stanley and Nixon 1992). The duration and intensity of seasonal and episodic hypoxia are heavily influenced by physical factors such as water temperature, freshwater inflow, tidal mixing and wind events (Haas 1977, Taft et al. 1980, Pihl et al. 1991).

Diel-cycling hypoxia occurs within the photic zone and extends to the bottom in shallow systems and above the pycnocline in deeper systems. It is driven primarily by the 24 h daytime-nighttime cycle of DO production and respiration by living algae (Kemp and Boynton 1980, D'Avanzo and Kremer 1994, Tyler et al. 2009). DO fluctuation is generally predictable, with lowest concentrations usually occurring around dawn following nighttime respiration and highest concentrations in the late afternoon following daytime productivity (D'Avanzo and Kremer 1994, Beck and Bruland 2000, Tyler et al. 2009). In environments with high algal density, over a single diel cycle DO can range from anoxia to highly supersaturated ($\geq$ 15 mg O_2 l^{-1}) (D'Avanzo and Kremer 1994, Tyler et al. 2009).

The occurrence of diel-cycling hypoxia has been recognized for several decades (Nixon and Oviatt 1973, Hackney et al. 1976, Kemp and Boynton 1980) and is geographically widespread. Some systems where varying degrees of diel-cycling hypoxia have been observed include Bojorquez Lagoon, Mexico (Reyes and Merino 1991); lagoons in Venice, Italy (Sorokin et al. 1996); Waquoit Bay, Massachusetts (D'Avanzo and Kremer 1994); Elkhorn Slough, California (Beck and Bruland 2000); and the Intracoastal Waterway of New Hanover County, North Carolina (Hubertz and Cahoon 1999). However, Tyler, Brady, and Targett (2009) have shown that diel-cycling hypoxia is particularly problematic in Delaware’s Inland Bays where the severity, duration and frequency of hypoxia is significantly augmented compared with the aforementioned locations.

Within a given estuary, the amplitude of diel-cycling DO fluctuation is primarily controlled by nutrient loading sources and proximity to oceanic exchange (Boynton et al. 1996). Tributaries have intrinsic characteristics that favor productivity rates which can result in high algal density thereby making these waters particularly susceptible to diel-cycling hypoxia. Generally, these characteristics include high nutrient loading, low flushing, high temperature, and protection from wind driven mixing. Deviations in DO dynamics from the general pattern of near dawn minimums and late afternoon maximums, as well as extreme events have been attributed to environmental forcing functions such as temperature, insolation, precipitation, tide, freshwater discharge, and wind (Beck and Bruland 2000, Tyler et al. 2009), and the DO dynamics in contiguous waters (Gallegos et al. 1992, Tyler et al. 2009). The worst hypoxic events have often occurred in association with early morning low tide (Edwards et al. 2004), heavy cloud cover (D'Avanzo and Kremer 1994, D'Avanzo et al. 1996, Tyler et al. 2009), and calm wind conditions - usually following a rain event (Gallegos et al. 1992, Lapointe and Matzie 1996).

Tributaries are often more susceptible to diel-cycling hypoxia because of their location (i.e. close to watershed sources of nutrients and far from sources of flushing) and hydrodynamic characteristics (i.e. poorly flushed). These intra-estuarine areas of low flushing, salinity, and DO as well as high productivity are often simultaneously areas of high fish abundance and densities (Meng et al. 2001, Ross 2003, Yamashita et al. 2003, Meng et al. 2004, Tyler and Targett 2007). There are many life-history processes that contribute to this seemingly counterintuitive overlap between anthropogenically impacted areas and nursery habitat. Larval ingress by means of selective tidal stream transport, bottom layer residual inflow, event-scale inflow, or some combination of these mechanisms all deposit larvae as far into the estuary as possible to ensure estuarine retention (Hare et al. 2006). Additionally, long term tagging (e.g. seasonal, Sackett et al. 2007) and stable isotope (Thorrold et al. 2001) studies have highlighted the role of philopatry, or natal homing, in estuary-dependent fishes. The larval and adult behaviors of estuary-dependent fishes serve to place juveniles in areas with higher prey resources, optimal temperatures for growth and that may be associated with lower predation (Paterson and Whitfield 2000, Manderson et al. 2004). In Delaware’s Inland Bays in particular, Michels (1986-2002), Clark (2001), and Tyler and Targett (2007) have found many estuary-dependent species to be relatively more abundant in the headwaters of tidal tributaries.

One powerful tool utilized by resource managers to understand hypoxia are water quality models (WQMs). They enable researchers to understand the response of a body of water to increased nutrient discharges and resulting eutrophication, while providing a predictive tool for investigating further loading or remediation efforts (Reckhow and Chapra 1999, Di Toro 2001).

Much effort has been invested in building models to simulate O_2 dynamics on several time and space scales in both sediments and the water-column. In general, model structures range from relatively simple statistical models and empirical relationships (Turner et al. 2006, Murphy et al. 2011) to more complex process simulations that focus either on sediment processes or on coupled biophysical dynamics (Boudreau 1991, Luff and Moll 2004, Fennel et al. 2013). Simple models are often focused on evaluating the major drivers of O_2 depletion and how management actions might alleviate low-O_2 zones, while coupled biophysical models are used to understand ecosystem interactions and feedbacks, where O_2 is one of many biogeochemically-linked model variables (Peña et al. 2009). Coupled biophysical models, however, are built at a variety of scales and complexities. 3-D hydrodynamic models may be built at meso (~1 km) horizontal scales with finer (1-10 m) vertical resolution and are computed at hourly to sub-hourly time scales, solving for salinity, temperature, elevation, three velocity components, and vertical mixing or more advanced turbulence schemes (Shchepetkin and McWilliams 2005). The hydrodynamics may be directly or indirectly coupled to multi-compartment biogeochemical models that include phytoplankton and zooplankton, particulate and dissolved inorganic and organic nutrient pools (N, P, Si, C), and O_2 (Zheng et al. 2004, Fennel et al. 2006, Xu and Hood 2006, Lin et al. 2007). Such models can include sediment parameterizations (Eiola et al. 2009), or be coupled to relatively simple sediment process models (Soetaert et al. 2000, Fennel et al. 2006, Sohma et al. 2008), or to more advanced models that compute diagenesis and remineralization of carbon, nitrogen, phosphorus, silica, sulfur and O_2 (Luff and Moll 2004, Brady et al. 2013, Testa et al. 2013).

Biogeochemical and ecosystem models are useful tools that can integrate the varied physical, biological, and anthropogenic drivers of ecosystem variability and associated internal interactions. By their nature, these models integrate our understanding of ecosystem processes, and they provide a number of unique capabilities, including (1) simulation of biological activity in the context of the physical regime, (2) coupling hydrologic and anthropogenic processes on land to compute watershed impacts on the receiving water, (3) providing metrics of uncertainty in predictions, as well quantitative goals for management (e.g., Total Maximum Daily Load), and (4) providing new quantitative insights into ecological processes that are otherwise difficult to understand. This latter point emphasizes that models are useful tools to understand ecosystem processes in ways that are impractical with observational or experimental studies; thus they provide an additional approach to improving our knowledge about coastal processes. The purpose of this report was to evaluate the dissolved oxygen calculations of a 1-D hydrologic model coupled to a 3-D linked hydrodynamic-water quality model called the Generalized Environmental Modeling System for Surfacewaters (GEMSS).

TMDL Model Implementations for Delaware's Inland Bays

The first state-of-the-art water quality modeling program implemented in Delaware's Inland Bays used a calibration database that included data from DNREC, USGS, US Army Corps of Engineers, University of Delaware researchers, citizen monitors and others (Cerco et al. 1994). The model was calibrated using data from 1988-1990 and is largely based on the same model used to manage water quality in Chesapeake Bay (CE-QUAL-W2) (Cerco and Noel 2005). The TMDL was established in 1998. This model included a mechanistic sediment flux model and even included a benthic algal model due to the shallow nature of the bays (Cerco and Seitzinger 1997). The second TMDL analysis used the aforementioned GEMSS model to establish the approved TMDL in 2004 (Entrix and JEEAI 2004). That model used a calibration dataset that spanned 1998-2000. This TMDL like most in marine waters, required that the daily average concentration for dissolved oxygen be above 5.5 mg O_2 L^{-1}. Many stations in Delaware's Inland Bays do not meet this metric in the early to late morning. It should be noted however, that there are many days where the average daily DO concentration can be above this level simply due to DO supersaturation in the afternoon period. Although the TMDL was set in 2004, the last water quality dataset used to establish the TMDL was 14 years ago (2000). Ultimately, the latest TMDL includes a systematic elimination of point source loading and an 85% and 65% reduction in nitrogen and phosphorus, respectively.

METHODS AND MATERIALS

Logistics of Obtaining the Model, Model Output, and Calibration Dataset

Although GEMSS output was used to determine the current TMDL for Delaware's Inland Bays by Delaware's DNREC, it has not been used often since the 2004 Final Report submitted by Entrix/Edinger (because no major update of the model was necessary). As a result, DNREC did not have the model output readily available. Inquiries to DNREC for the purposes of obtaining model output began in June 2012. Delaware DNREC indicated that it did not have archived model output and requested that we contact Entrix Engineering (the consulting firm that developed the model). Entrix/Edinger had since been subsumed into Environmental Resources Management (ERM). Mike Fichera, PE of ERM indicated that they also did not have archived model output. Further investigation by Delaware DNREC found an application CD of the GEMSS Delaware Inland Bays Application. This required re-running simulations from the application CD. This is unfortunately a fairly common fate for model output and a potentially important lesson for future TMDL implementations. Model output should be stored, archived, and made available in a common model output format (e.g., NetCDF).

GEMSS Model Formulation

The GEMSS model is built to easily add on modules to increase model functionality while avoiding too much complexity when not needed. The core of the model is the hydrodynamic and transport model (HDM). It uses Generalized Longitudinal-Lateral Vertical Hydrodynamic and Transport (GLLVHT), a state-of-the-art three dimensional numerical model that computes time varying velocities, water surface elevations, and water quality constituent concentrations in rivers, lakes, reservoirs, estuaries, and coastal water bodies (Edinger and Kolluru 1999, Kolluru 1999). The water quality model that is coupled to the HDM can vary according to the number of biogeochemical processes that are necessary to capture nutrient and oxygen dynamics in an estuary and the amount of data available. In the case of Delaware's Inland Bays, Edinger/Entrix made the case that the limited data available argued for the use of the Water Quality dissolved Particulate Model (WQDPM; Figure 1). This is an intermediate complexity model that separates nutrients (i.e., nitrogen and phosphorus) in organic/inorganic dissolved/particulate forms. The DO calculation is a function of phytoplankton respiration, oxidation of BOD, atmospheric input, and sediment oxygen demand (SOD). It should be noted, that in this formulation SOD is an empirical formulation and there is no sediment flux model.

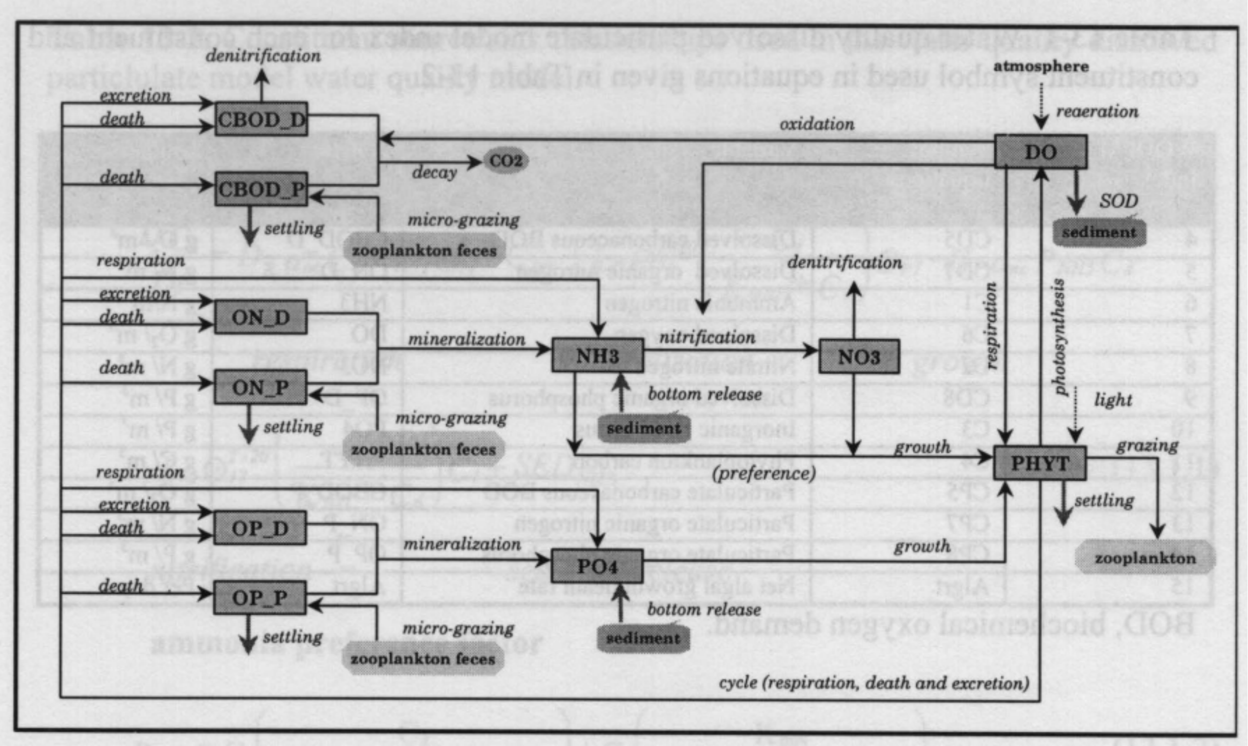


FIGURE 1. DIAGRAM OF THE WATER QUALITY DISSOLVED PARTICULATE MODEL (WQDPM). THE FIGURE IS TAKEN FROM EDINGER 2002 WATER BODY HYDRODYNAMIC AND WATER QUALITY MODELING

Metrics for Model Evaluation

The purpose of this analysis was to examine model-data comparisons of DO. In order to accurately predict DO in Delaware's Inland Bays, transport must also be accurately represented. Specifically, horizontal diffusion along the axes of tributaries regulates the influence of DO-rich ocean water from Indian River Inlet on DO. Additionally, transport controls the timing of low tide and therefore light penetration to the bottom (Tyler et al. 2009). Before showing model data comparisons, an introduction to the data used to calibrate the model is necessary to identify locations used for validation (Figures 2-8 show the locations of tidal stations within Delaware's Inland Bays (non-tidal stations were beyond the scope of this report)). HDM appears to accurately simulate tide height and temperature (Figure 9 for Tide Height Comparison with Rosedale Beach Tide Gage and Figure 10 for an example fit to Temperature).

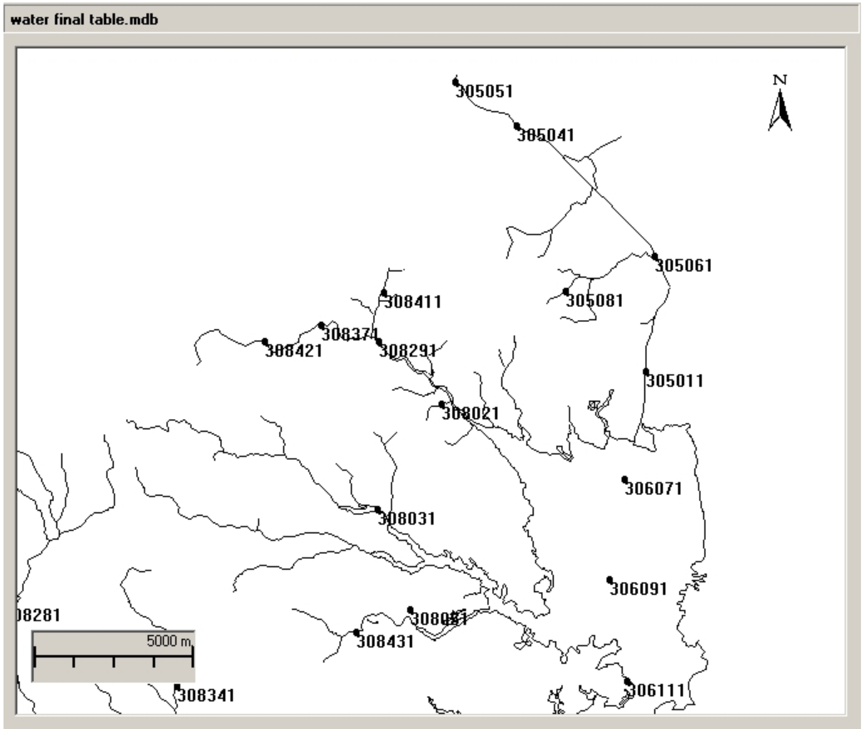


FIGURE 2. STORET LOCATIONS IN REHOBOTH BAY

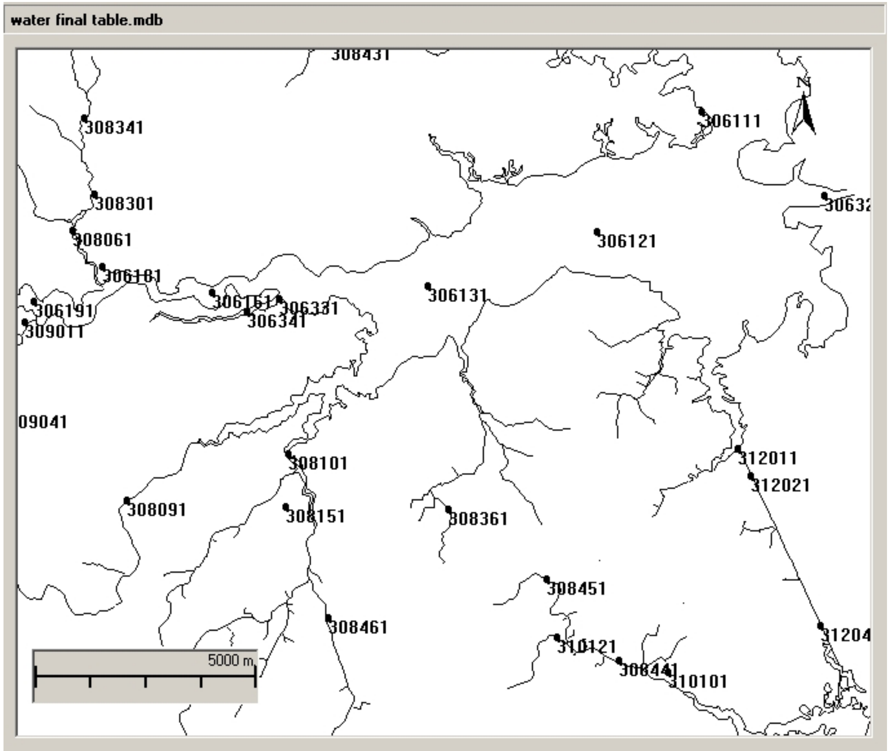


FIGURE 3. STORET LOCATIONS IN INDIAN RIVER

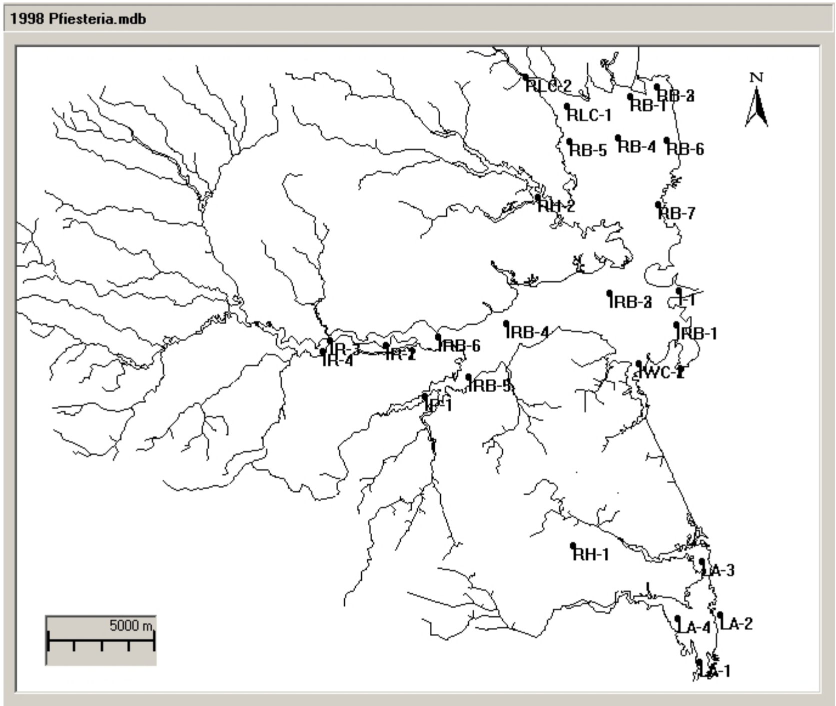


FIGURE 4. PFIESTERIA SAMPLING SITES FROM 1998-1999

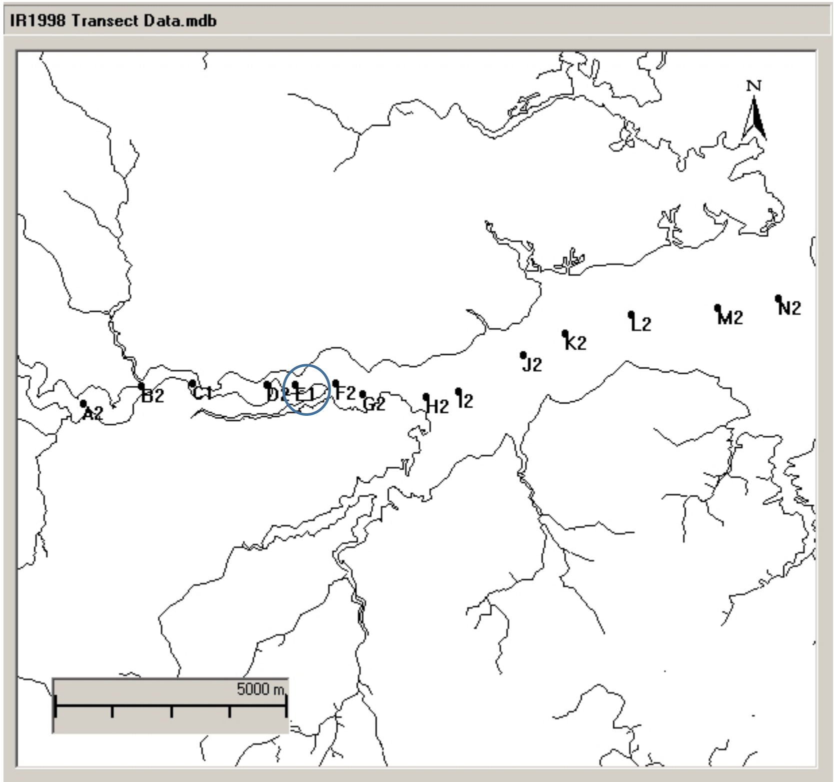


FIGURE 5. CONECTIV (INDIAN RIVER POWERPLANT) STATIONS ALONG THE AXIS OF INDIAN RIVER

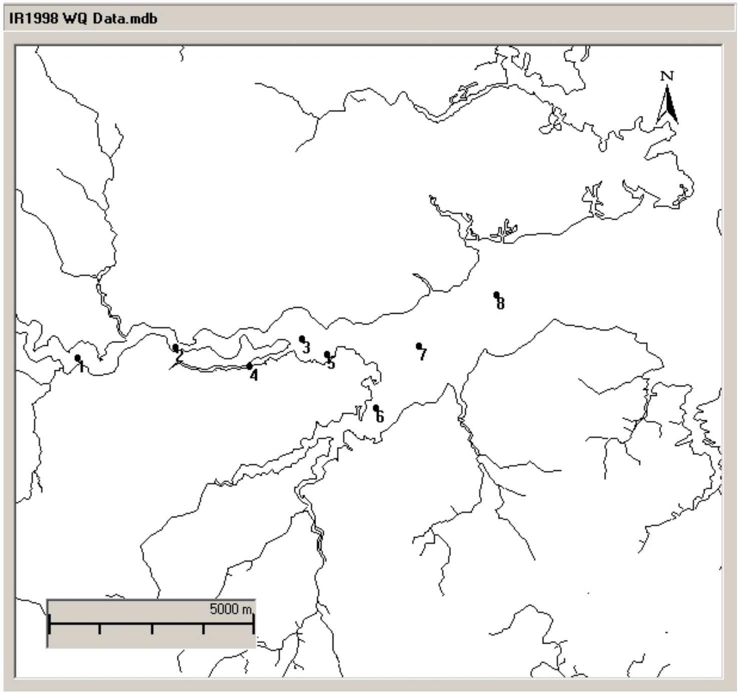


FIGURE 6. CONECTIV (INDIAN RIVER POWERPLANT) WATER QUALITY STATIONS

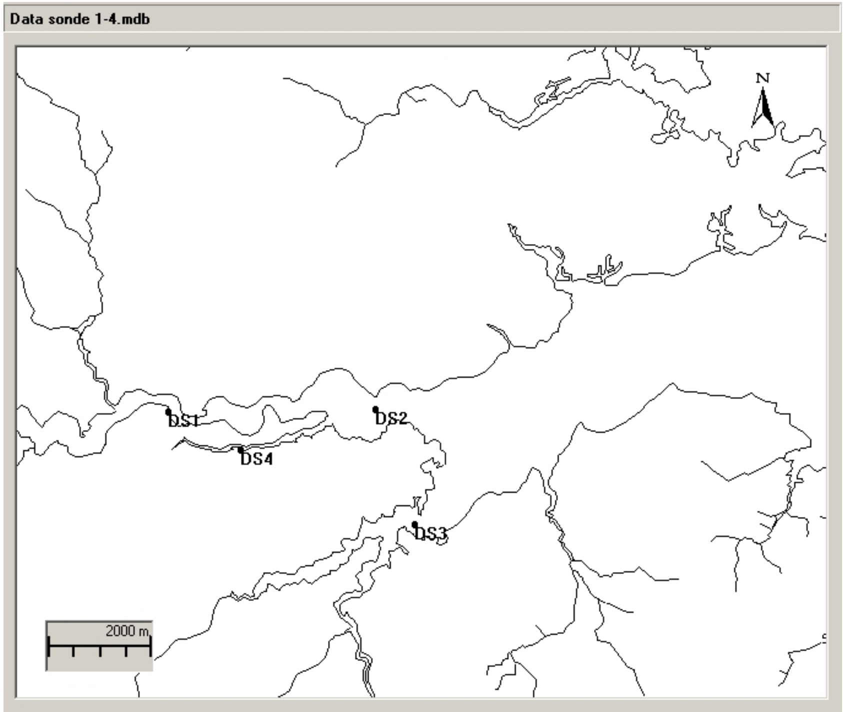


FIGURE 7. CONECTIV (INDIAN RIVER POWERPLANT) DATASONDE STATIONS 1998-1999

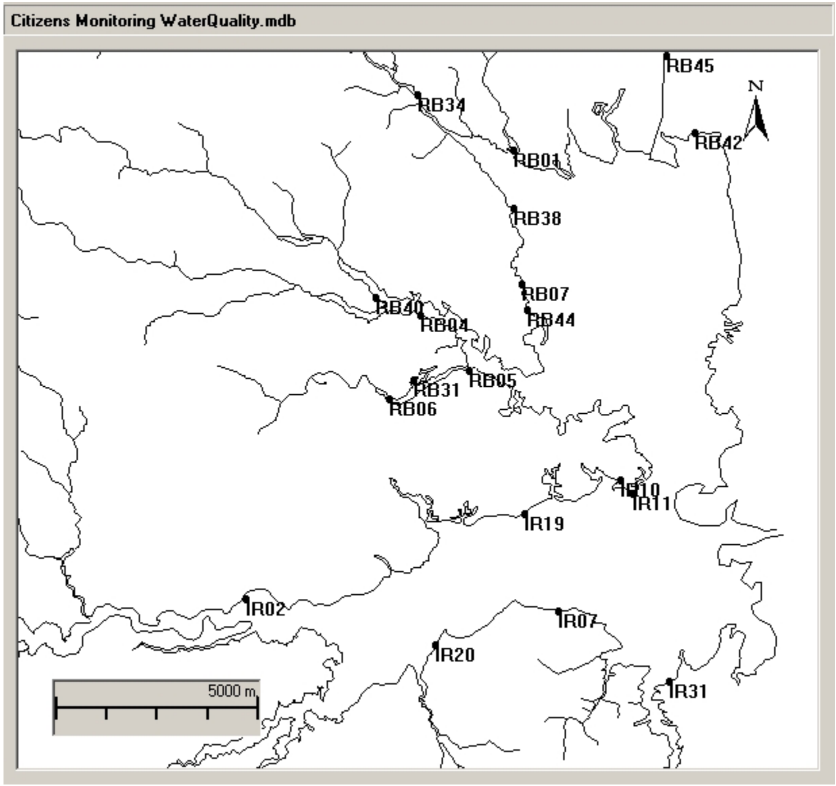
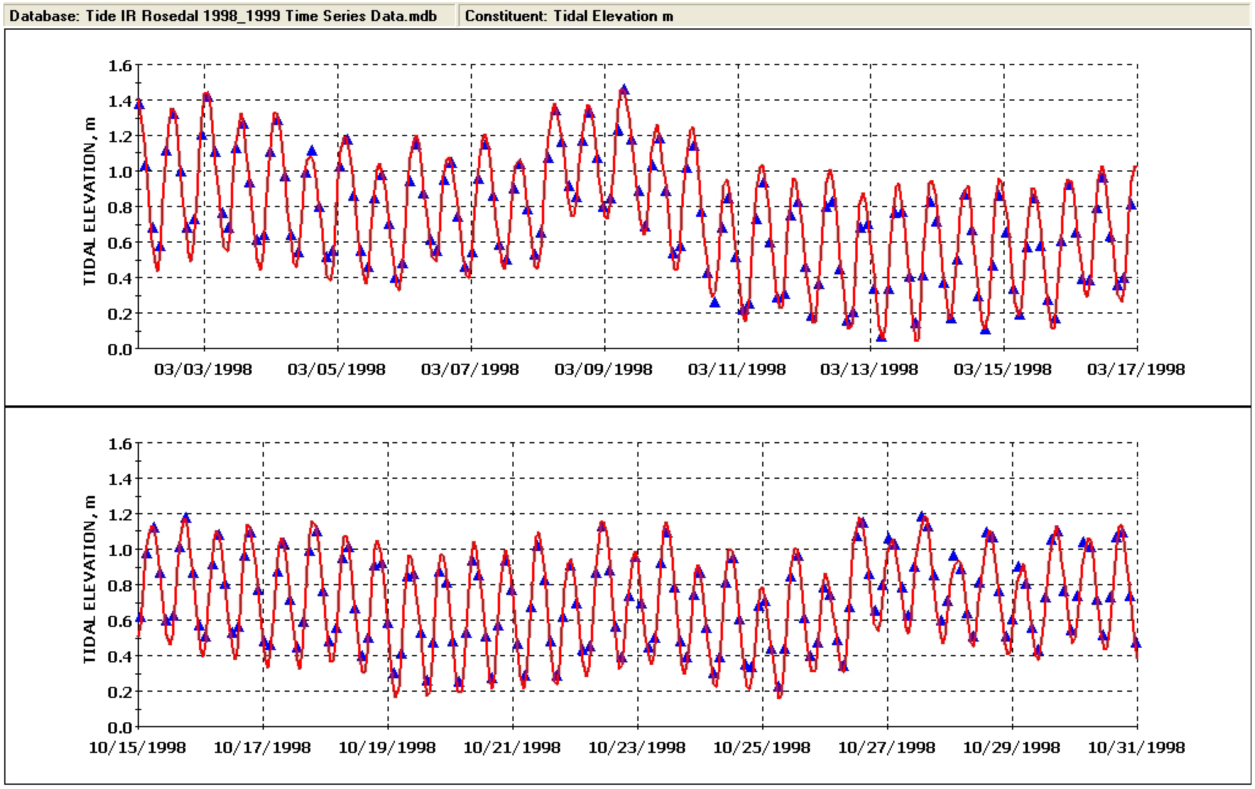


FIGURE 8. CITIZEN MONITORING NETWORK OF NUTRIENT SAMPLES



Database: Tide IR Rosedal 1998_1999 Time Series Data.mdb

Constituent: Tidal Elevation m

Figure 9. Comparison of model predictions of tide height at Rosedale Beach for a two week period in the beginning and end of 1998

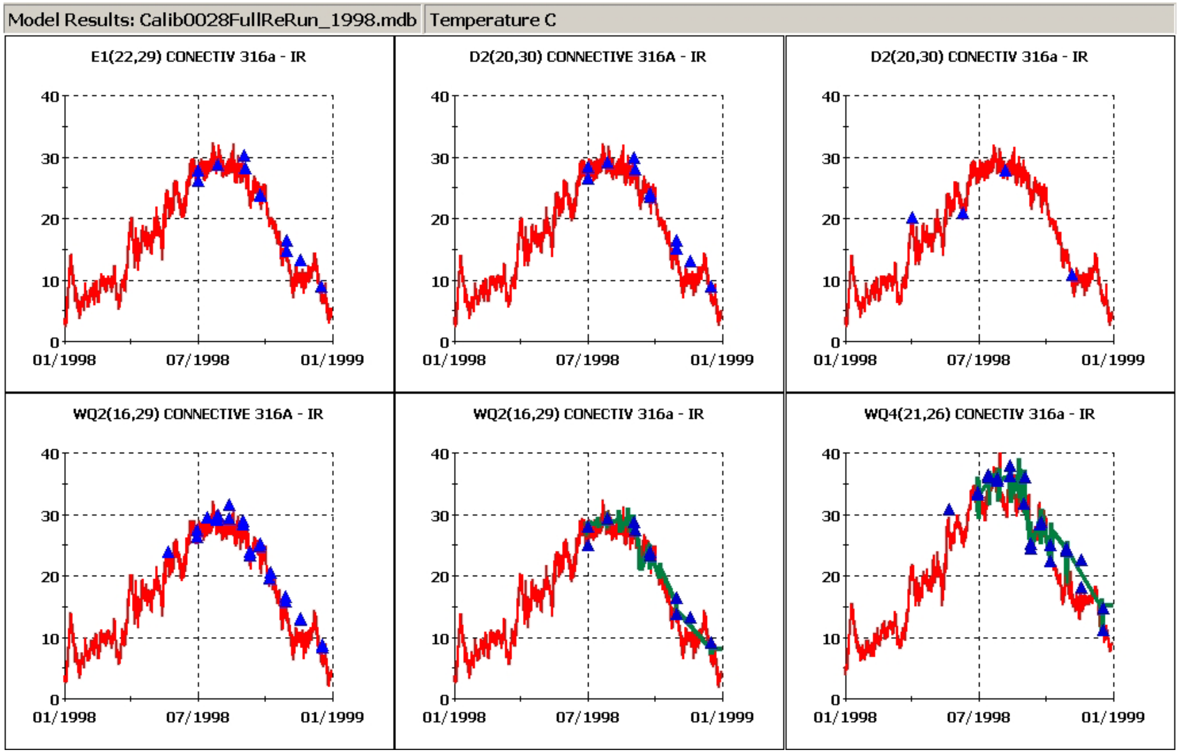


FIGURE 10. TEMPERATURE PREDICTIONS ALONG THE AXIS OF INDIAN RIVER BAY. GENERALLY, THE MODEL PREDICTS TEMPERATURE WELL BOTH TEMPORALLY AND SPATIALLY

For the purposes of estimating the influence of Indian River Inlet on ocean-estuary exchange, GEMSS did its due diligence and compared volumetric flow rates into Delaware's Inland Bays (Figure 11). Although the model adequately reproduces the observed volumetric flow rate, the depth of Indian River Inlet has almost certainly deepened since 1990 due to scouring. It would be interesting to simulate the effect of increasing the depth of Indian River inlet on DO and salinity exchange with the Atlantic Ocean with updated TMDL modeling.

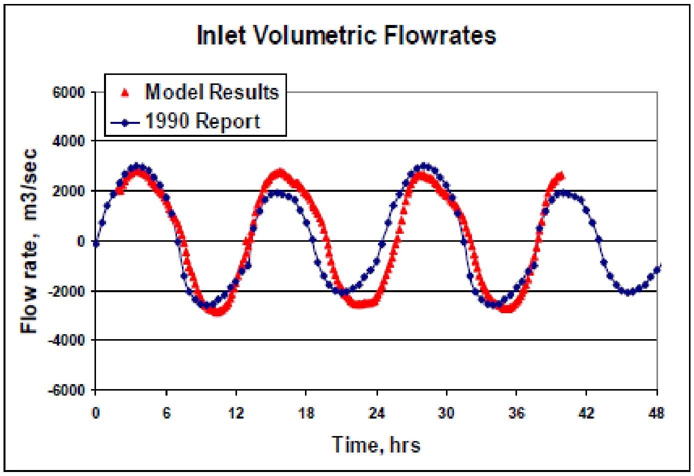


FIGURE 11. COMPARISON OF VOLUMETRIC FLOW RATE FROM GEMSS AND A 1990 REPORT.

Salinity was not surprisingly more difficult to simulate for GEMSS than temperature and salinity. For the purposes of further analysis in this report, it should be noted that 1998 was a very wet year, 1999 was relatively dry, and 2000-2001 were ‘normal’ (Figure 12) as measured at the Millsboro Dam gaging station.

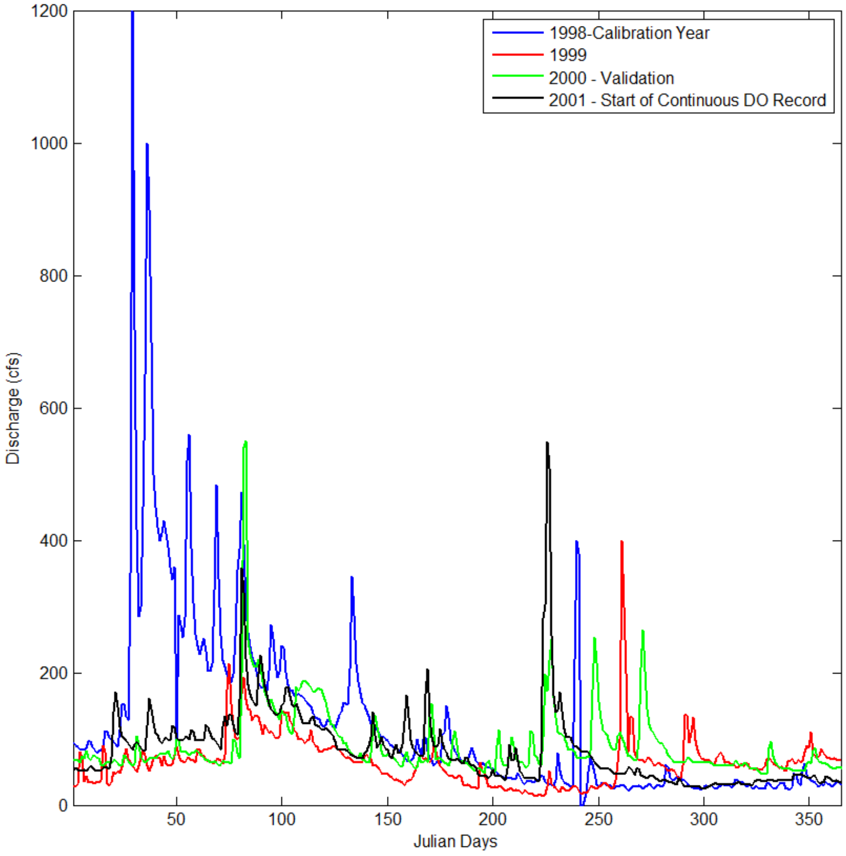
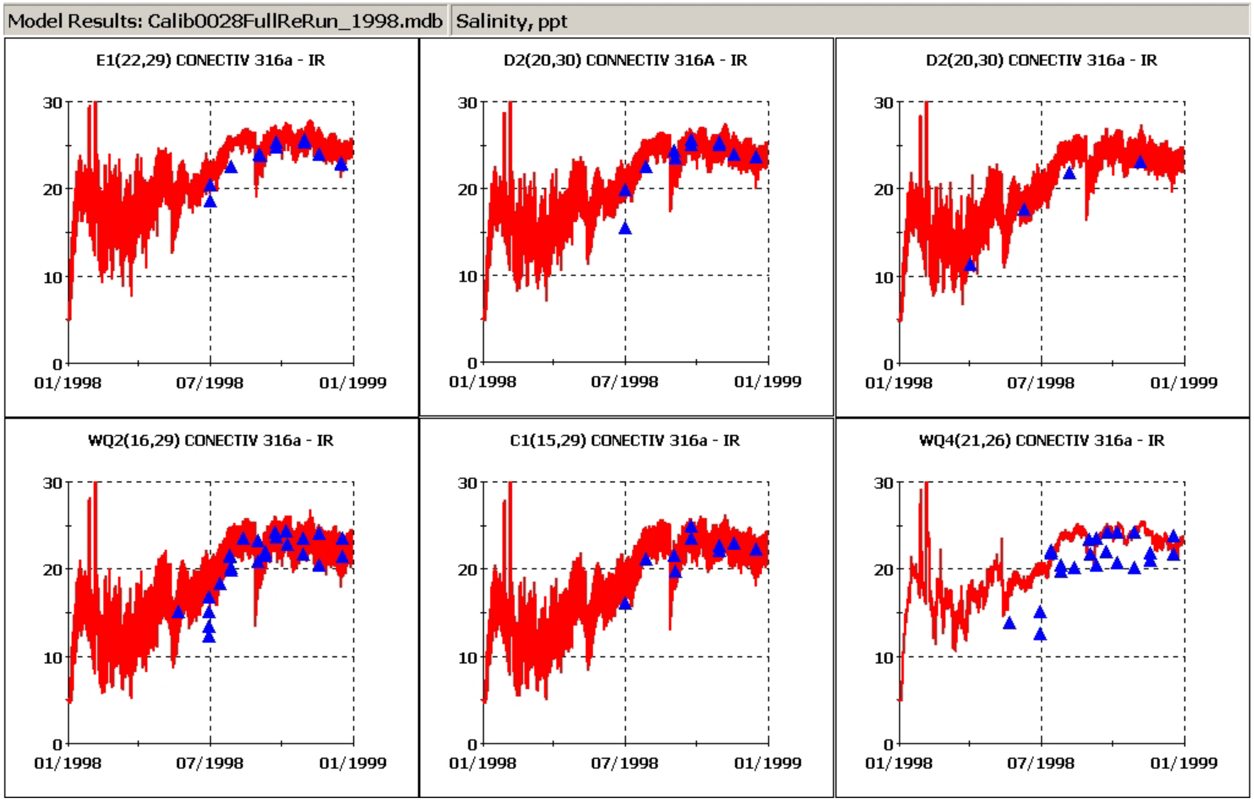


FIGURE 12: DISCHARGE FROM THE MILLSBORO DAM USGS GAGING STATION FOR 1998-2000 (GEMSS SIMULATION YEARS) AND 2001 (START OF CONTINUOUS MONITORING FOR DO). NOTICE THAT 2000 AND 2001 ARE SIMILAR YEARS FOR FLOW

As a result, during 1998, GEMSS predicts large swings in salinity during the early part of the year and tends to overpredict salinity in the later part of the year (Figure 13). Figure 13 shows multiple stations along the axis of Indian River during 1998, 1999, and 2000. Overprediction during 1998-1999 may be an indication of too much horizontal diffusion and consequently, an indication that DO may not go hypoxic in model calculations due to overflushing of tributaries. There are many fewer observations in 2000 when Conectiv and Pfiesteria monitoring ceased.


Model Results: Calib0028FullReRun_1998.mdb
Salinity, ppt
E1(22,29) CONECTIV 316a - IR
D2(20,30) CONECTIV 316A - IR
D2(20,30) CONECTIV 316a - IR
WQ2(16,29) CONECTIV 316a - IR
C1(15,29) CONECTIV 316a - IR
WQ4(21,26) CONECTIV 316a - IR
30
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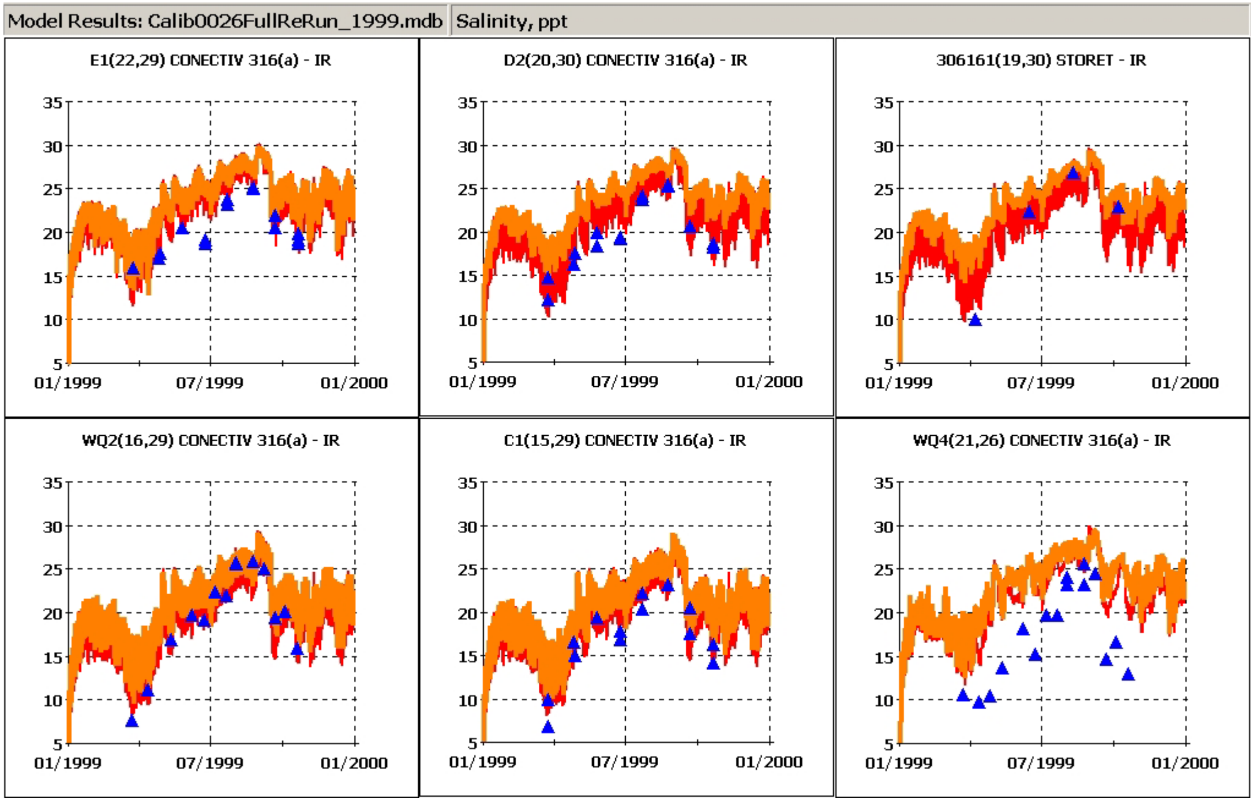


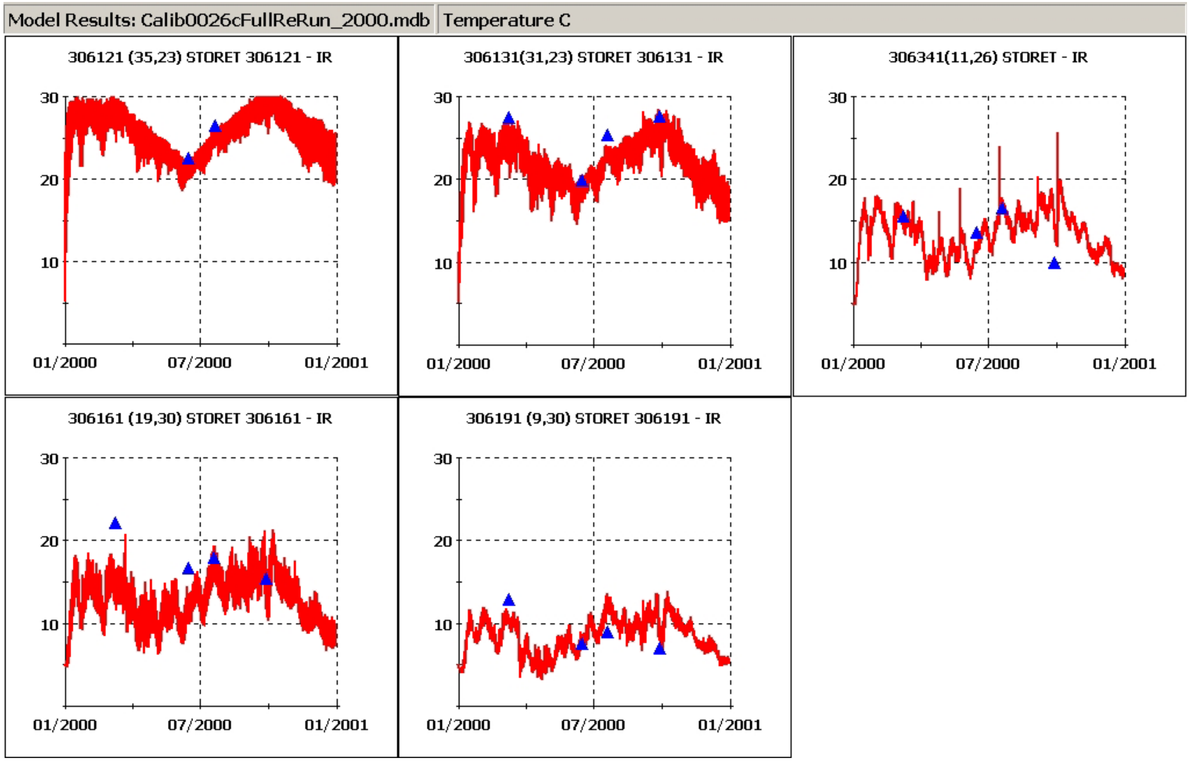


FIGURE 13. TOP 6 PANELS SHOW MODEL DATA COMPARISONS FOR SALINITY ALONG THE AXIS OF INDIAN RIVER BAY IN 1998 WHILE THE MIDDLE 6 PANELS SHOW THE SAME COMPARISON FOR 1999 AND THE LAST 5 PANELS SHOW THE MODEL-DATA COMPARISON FOR 2000.

Another way of viewing the salinity fit is a simple linear regression between modeled and observed salinity. For example, in 1998, the linear regression is shown in Figure 14. There appears to be a model bias toward overprediction of mesohaline (15-20) waters. Future modeling efforts should consider horizontal mixing carefully as it can potentially influence DO calculations significantly in tributaries.

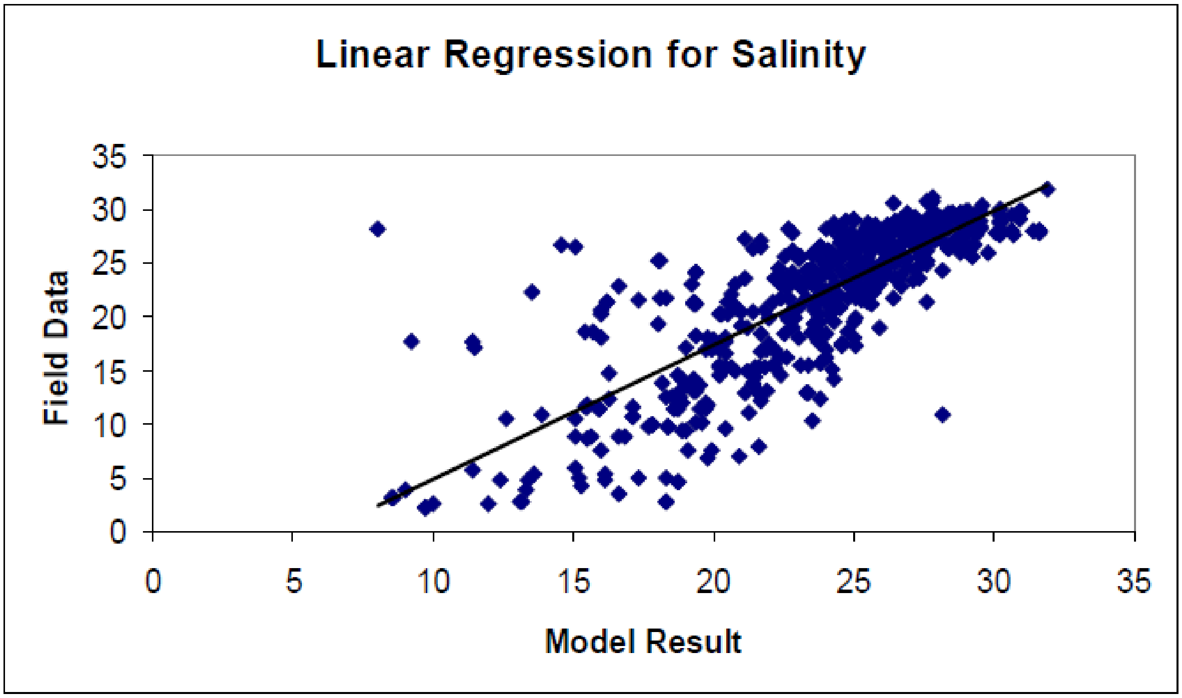


FIGURE 14. MODELED VERSUS OBSERVED SALINITY IN 1998

Dissolved Oxygen in Tributaries

The current understanding of DO dynamics in Delaware's Inland Bays comes largely from the sampling of DNREC and the University of Delaware (Tyler et al. 2009). This analysis has resulted in new understanding of the spatial and temporal dynamics of diel-cycling hypoxia. Much of this understanding has been built on the monitoring of sentinel tributary sites, such as Pepper Creek, Indian River, Herring Creek, and Love Creek. For example, Figure 15 shows the intra-annual and interannual trends in DO in the headwaters of Pepper Creek. Generally, the daily minimum occurs from 06:00-08:00 and the DO maximum occurs from 16:00-20:00. However, productivity, and hence respiration, can be so high, that even short term cloudiness can result in the formation of hypoxia in mid-summer. Most years the daily range is from 2-14 mg O_2 L^{-1}, but this can be exacerbated to 0-25 mg O_2 L^{-1} during particularly bad summers (e.g., 2001, 2004-2005).

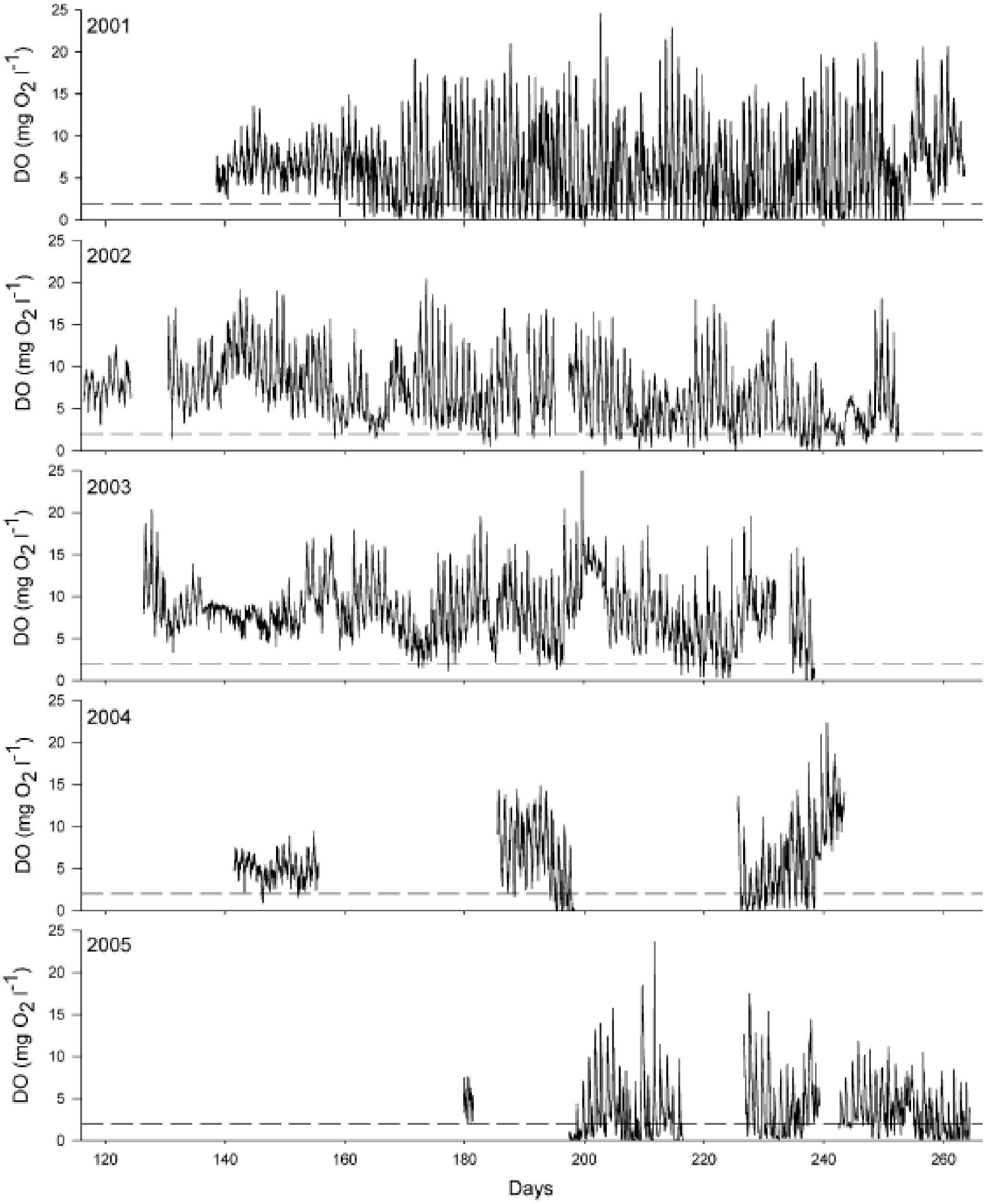


FIGURE 15. DO RECORD FOR THE HEAD WATERS OF PEPPER CREEK, DELAWARE, FROM 2001-2005

Indian River, Herring Creek and Love Creek show similar patterns (see top three panels in Figure 16). Additionally, there are strong gradients along the axis of these creek with more hypoxic waters upstream of open bay sites.

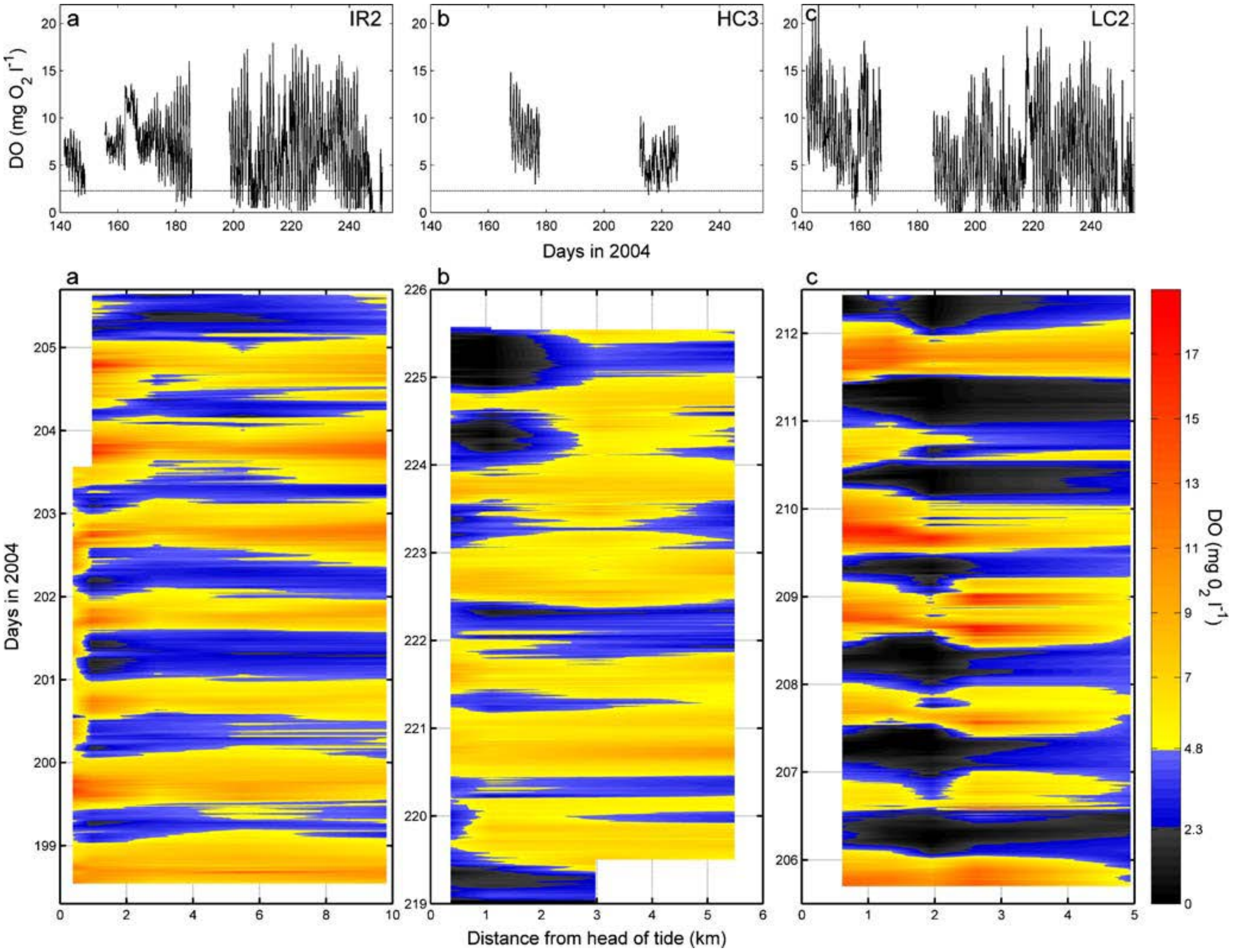


Figure 16. DO time series for (a) Indian River, (b) Herring Creek, and (C) Love Creek. Bottom panels show the spatial gradient in DO with the acute DO criteria in black (2.3 mg O_2 L^{-1}) and the Chronic DO criteria in blue (<4.8 mg O_2 L^{-1})

The remaining analysis will focus on model validation sites in these four systems.

Pepper Creek

The closest stations for Pepper Creek in the model validation dataset were sites IP-1, IRB-5, WQ6, and DS3. Model-data comparisons at Station IP-1 reveal salinity over prediction as is commonly observed in mesohaline stations in GEMSS. The salinity record indicates that a pulse of freshwater in early summer caused salinity to drop to 10 and then a summer drought resulted in a gradual increase to 25. This may be an indication that the model has missed a large pulse of organic material and nutrients from the watershed. Of particular interest in this study is the DO panel. First, DO is generally supersaturated and does not fall below 7.0 mg O_2 L^{-1} (approximately 100% saturation during summer temperatures and salinity). The data used for validation is also supersaturated; presumably because it has been sampled in the early to mid-afternoon period. The maximum daily excursion in modeled DO is 4 mg O_2 L^{-1}. It should also be noted that although the trend ('rise-then-fall') in chlorophyll is well simulated, the chlorophyll value is underpredicted and model data comparisons are presented on a log scale. As a result, total nitrogen (TN) is significantly underpredicted.

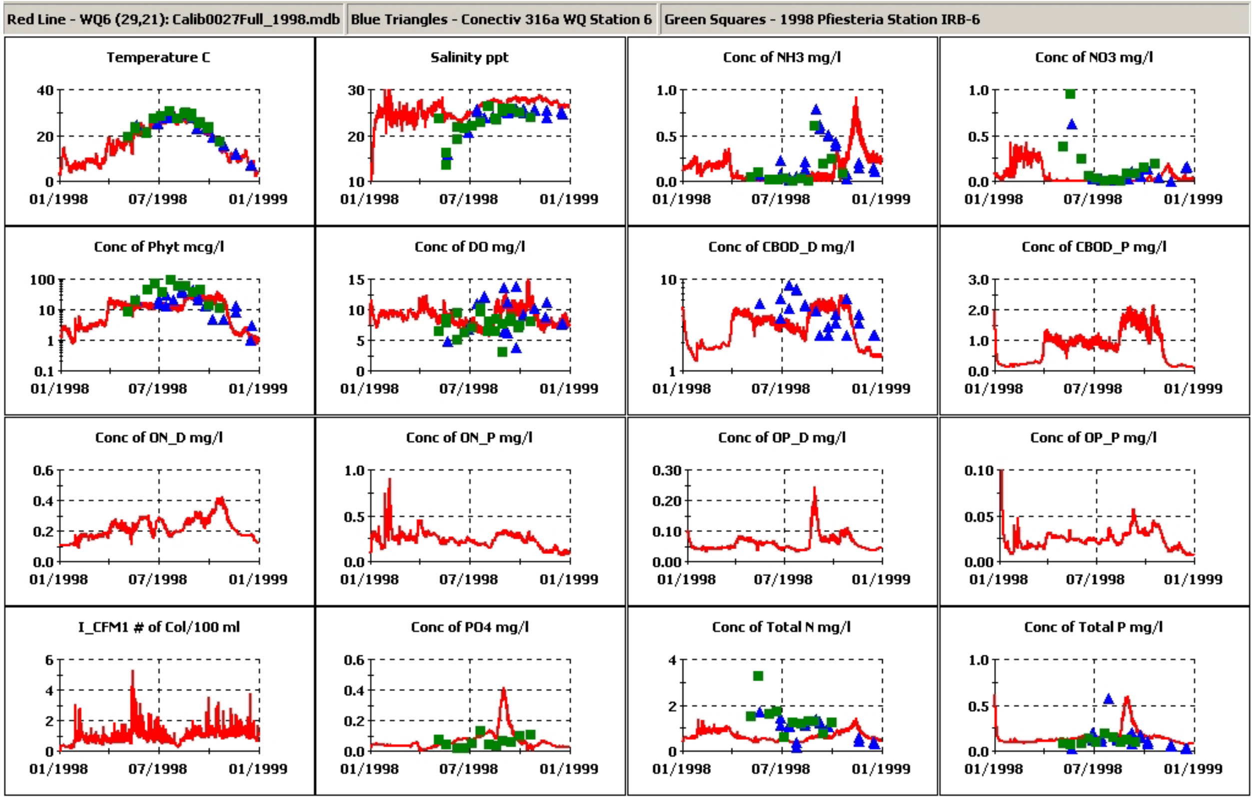


FIGURE 17. MODEL-DATA COMPARISON FOR DO, BOD, AND NUTRIENTS AT STATION IP-1 (NEAR PEPPER CREEK, DELAWARE) FOR 1998. GREEN SQUARES REPRESENT DATA FROM PFIESTERIA MONITORING AND BLUE TRIANGLES REPRESENT DATA FROM CONECTIV

Indian River

Comparatively, Indian River is the most data-rich tributary in the database. Stations in Indian River and the adjacent Indian River Bay include: Storet stations 306161, 306341, 306331, & 306131, Pfiesteria stations IR-4, IR-3, IRB-6, & IR-2, Conectiv Stations A2-I2, WQ1-WQ7 (excluding WQ6 which was in Pepper Creek), and DS1, DS2, & DS4, Citizen Monitoring Station IR02. For the sake of brevity, I chose 4 stations along the axis of Indian River to show in this report in order of upstream to downstream: 306161, IRB-6, WQ3, and 306131 (Figures 18-20). Figure 18 shows data from the most upstream Indian River Station. Of note, salinity is well simulated at this site, as is total nitrogen. The salinity time series may be the result of the site's proximity to the well-known boundary conditions at Millsboro Dam. However, the model simulates DO very poorly. Relatively high algal growth (50 µg CHL L^{-1}) causes supersaturated conditions throughout the year. In contrast, the data indicates, the late summer bloom should have been larger and concomitant increases in respiration may have held DO lower than the simulation.

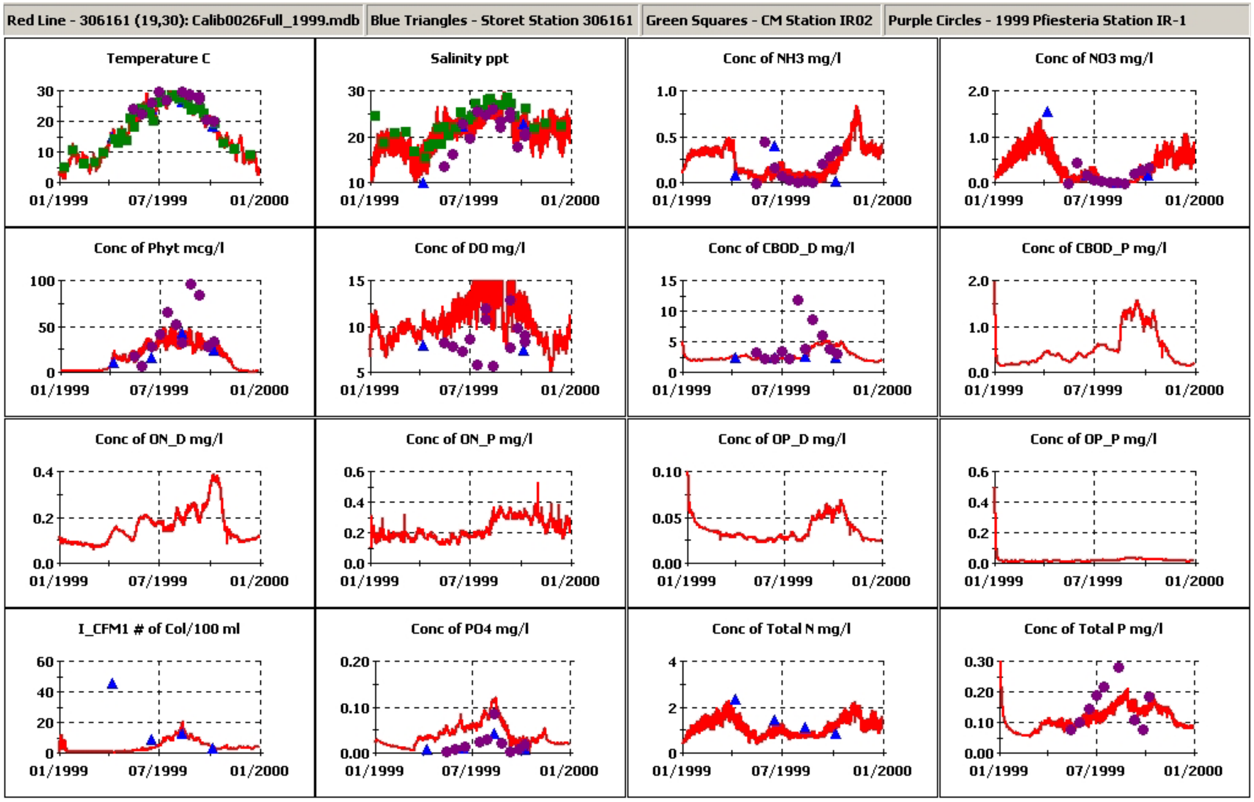


FIGURE 18. MODEL-DATA COMPARISON FOR AN UPSTREAM (HEAD OF TIDE) STORET STATION 306161

The next downstream station, Figure 19: WQM3 and IRB-6, demonstrates that the model calculates a standard DO sag. That is, as the calculation moves from the head of tide toward the ocean, the DO decreases as material from the fall line is processed in the river. Modeled DO is lower in the mid-summer and salinity & TN have transitioned from well-simulated to overpredicted. Boundary inputs of organic material and nutrients are advected downstream and used by phytoplankton which results in a seasonal DO minima in the mid-river site (Figure 19). While this is standard model and system behavior in estuaries, data indicates that DO is likely more controlled by daily fluctuations in insolation than seasonal advective and bloom dynamics.

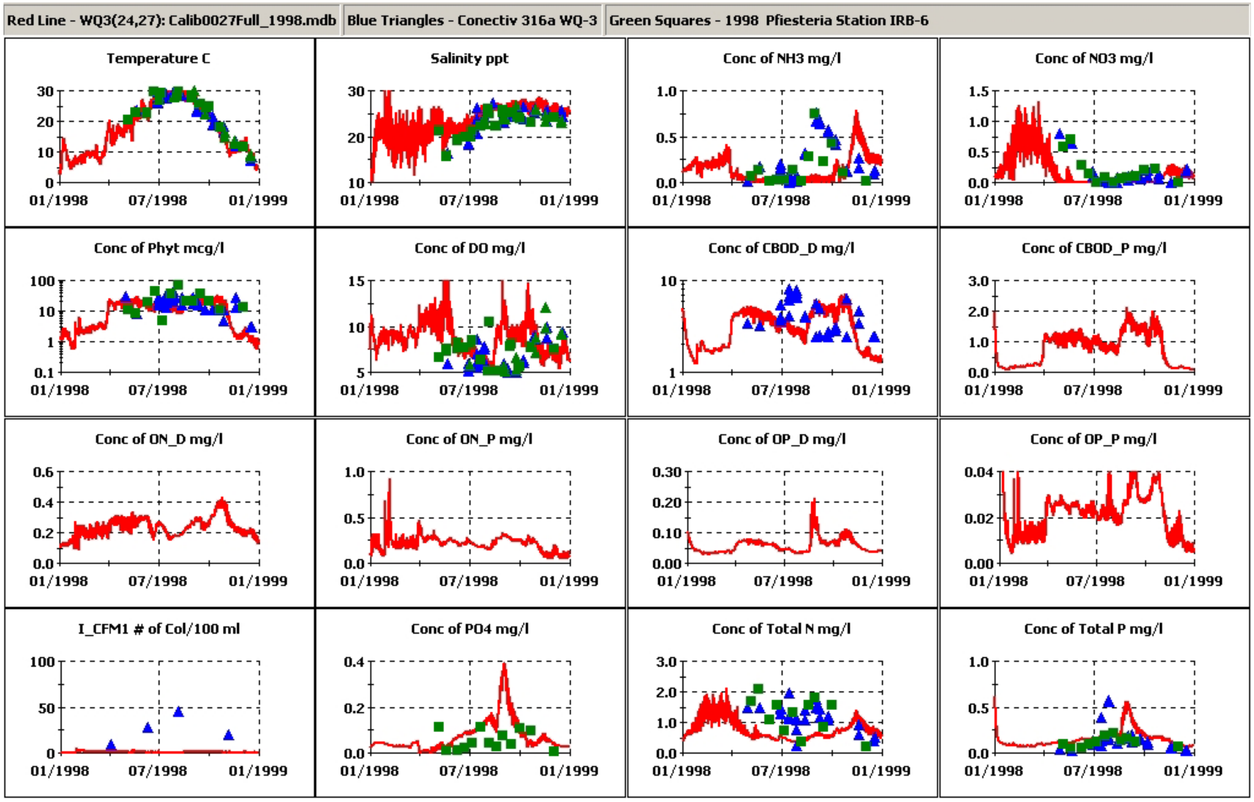


FIGURE 19. MODEL-DATA COMPARISON FOR AN MID-RIVER CONECTIV STATION WQ3 AND PFIESTERIA IRB-6

Finally, the model-data comparison for the open bay Storet Station at the mouth of Indian River is shown in Figure 20 for both 1998 and 2000. The model data comparison is very similar to the trends observed at the mid-River station with DO showing seasonal changes (summer downshift) instead of diel-changes and lower than observed chlorophyll and TN predictions.

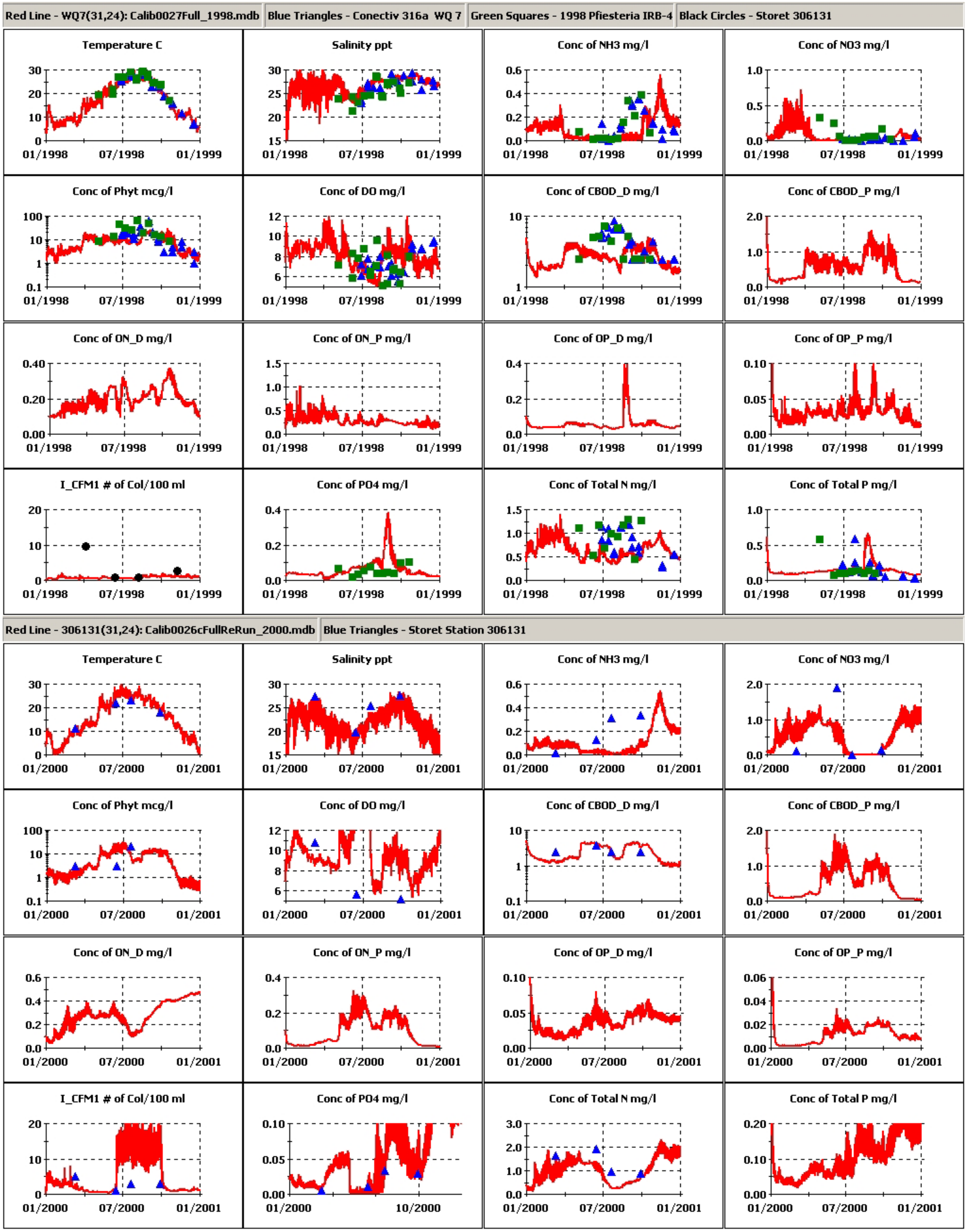


Figure 20. Model-data comparison for an open Indian River Bay Conectiv Station WQ7 and Pfiesteria IRB-4, Storet Station 306131 (for 1998 in the top panels and 2000 in the bottom 6 panels)

Herring Creek

Applicable sites for Herring Creek were Storet Station 308031, Pfiesteria RH-2, and Citizen Monitoring RB-40, and RB-04. Storet Station 308031 is a boundary condition at Burton's Pond and therefore, the model is fit to the data. RH-2 is the most appropriate model-data comparison to analyze and is shown in Figure 21. Water Quality at this site is relatively well simulated for chlorophyll, salinity, and temperature. Modeled DO is consistently higher than observed DO, which was often below the chronic DO criteria (4.8 mg O_2 L^{-1}). Total nitrogen and ammonium appear to be underpredicted.

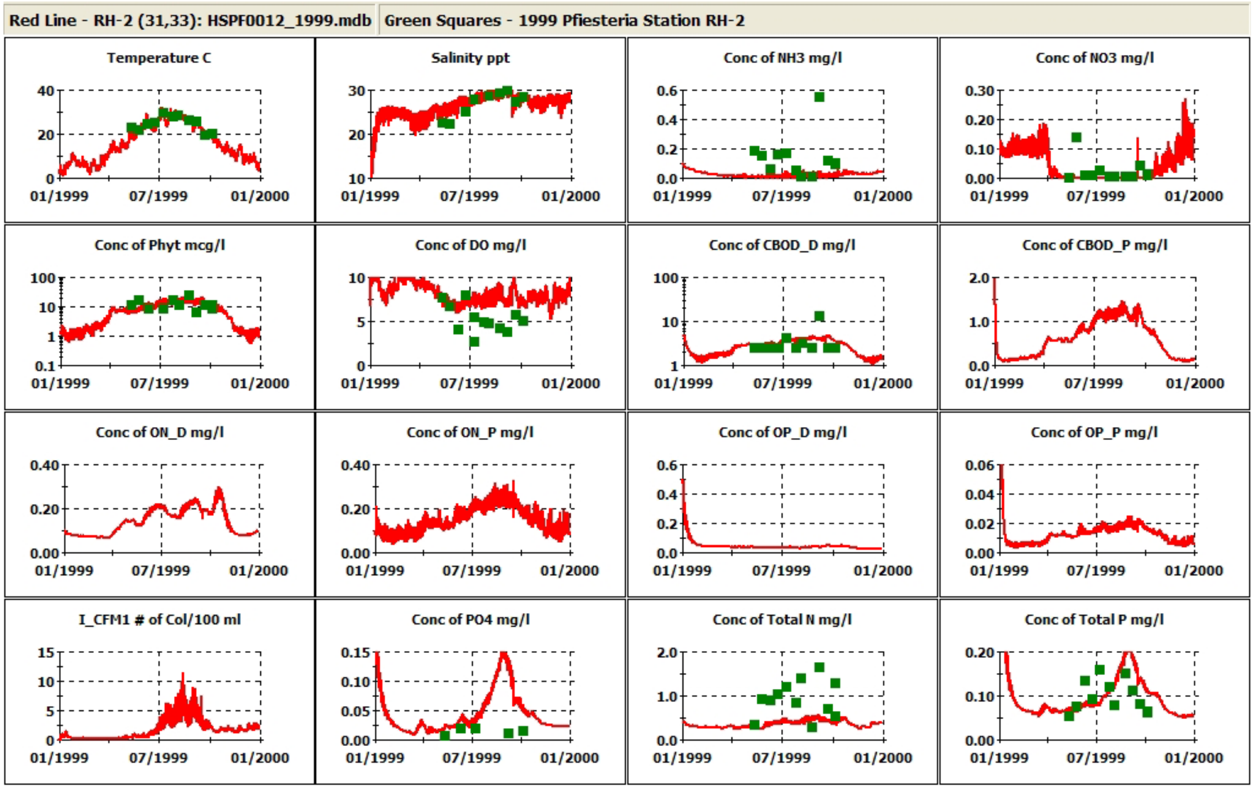


FIGURE 21. MODEL-DATA COMPARISON FOR HERRING CREEK (PFIESTERIA RH-2) FOR 1999

Love Creek

Love Creek, according to Tyler, Brady, and Targett (2009), was the most consistently hypoxic site in Delaware's Inland Bays by frequency, duration, and severity. Love Creek sites available for model-data comparison include Storet Stations 308291, 308021, Pfiesteria RLC-2 & RLC-1, and Citizen Monitoring sites RB34, RB01, RB38. Storet Station 308291 was a boundary condition stations and was fit to data. RLC-1 is the only appropriate station to analyze Love Creek model-data comparisons (Figure 22), however, it should be noted that RLC-1 is a station outside the creek proper. The model simulates DO as generally decreasing as the summer progresses from 12 to 4 mg O_2 L^{-1}. During mid-July, DO rebounds due to a strong bloom in the model. It is common for DO in Love Creek to vary between 0 and 20 mg O_2 L^{-1} in any given day. Seasonal trends in DO at this site tend to be in the amplitude of DO fluctuations and not in the mean DO level.

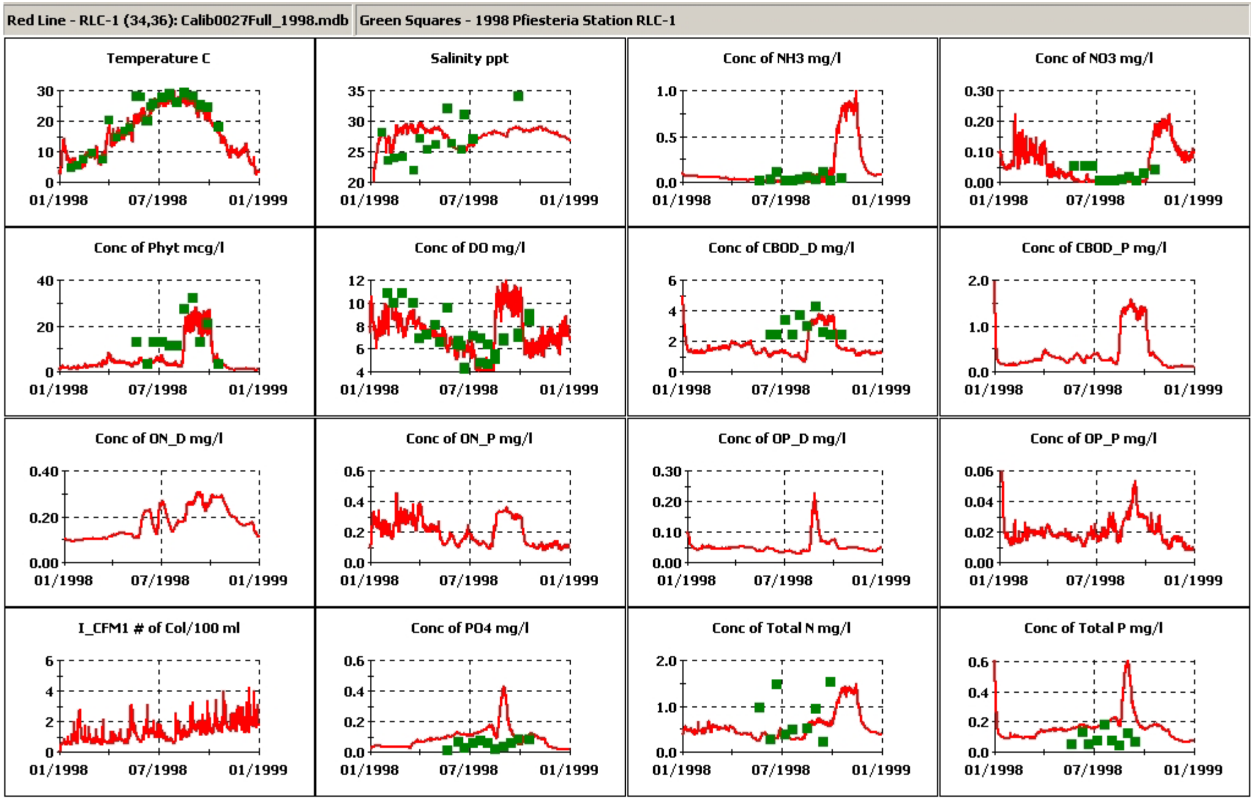
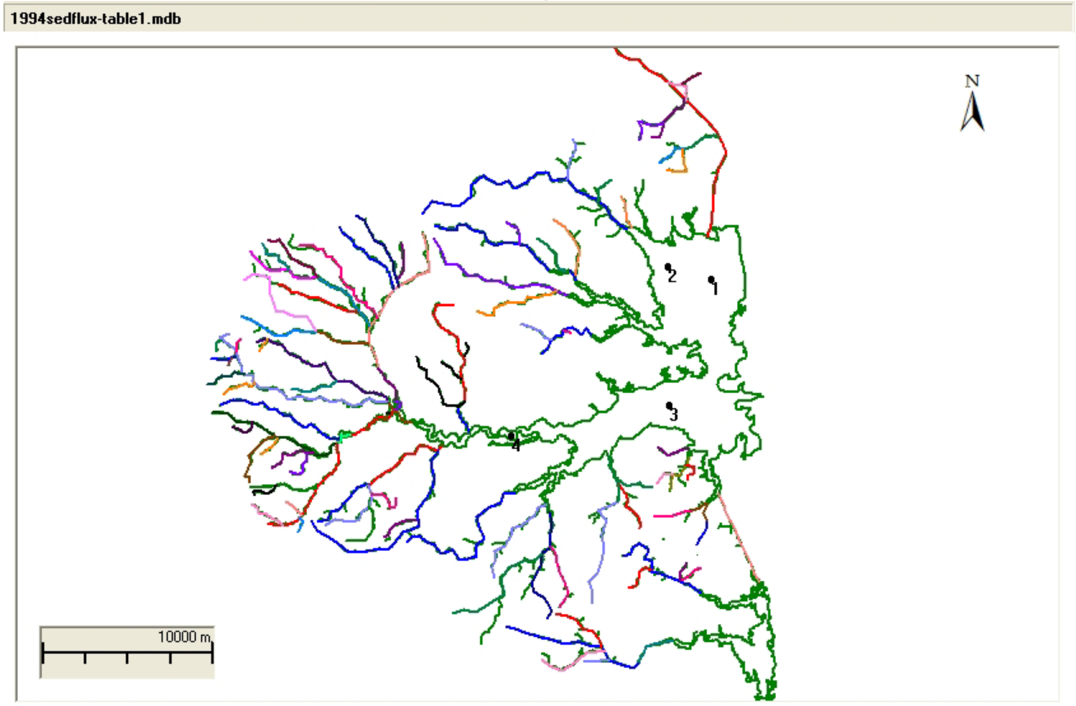


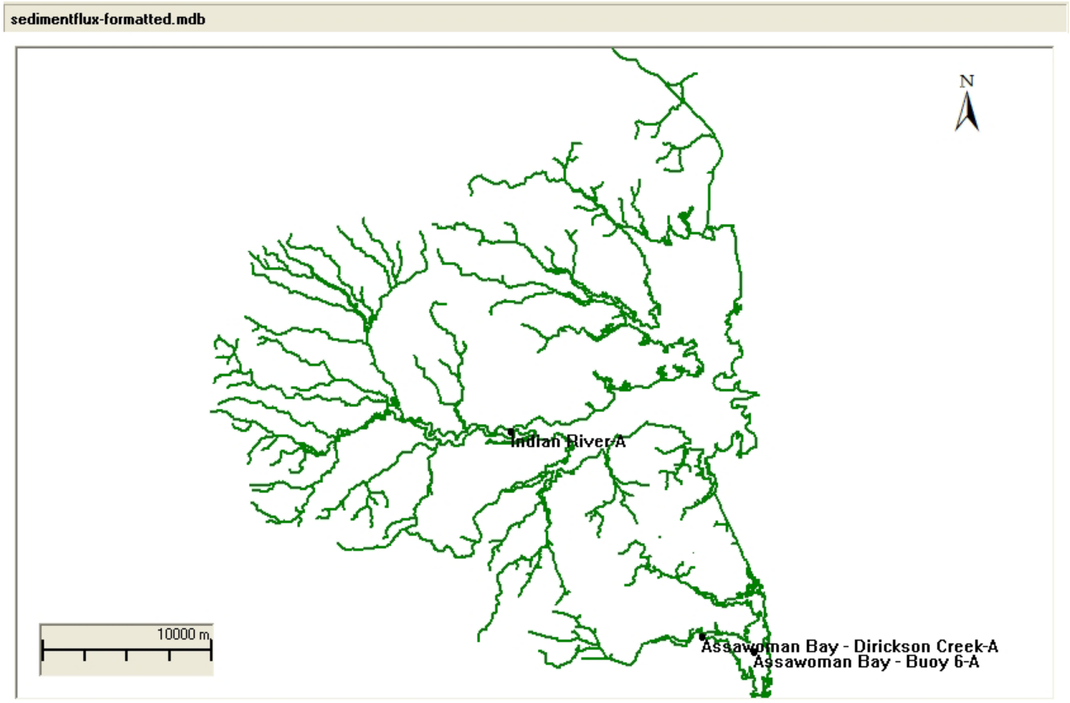
FIGURE 22. MODEL-DATA COMPARISONS FOR LOVE CREEK SITE RLC-1 (OPEN BAY ADJACENT TO LOVE CREEK) FOR 1998

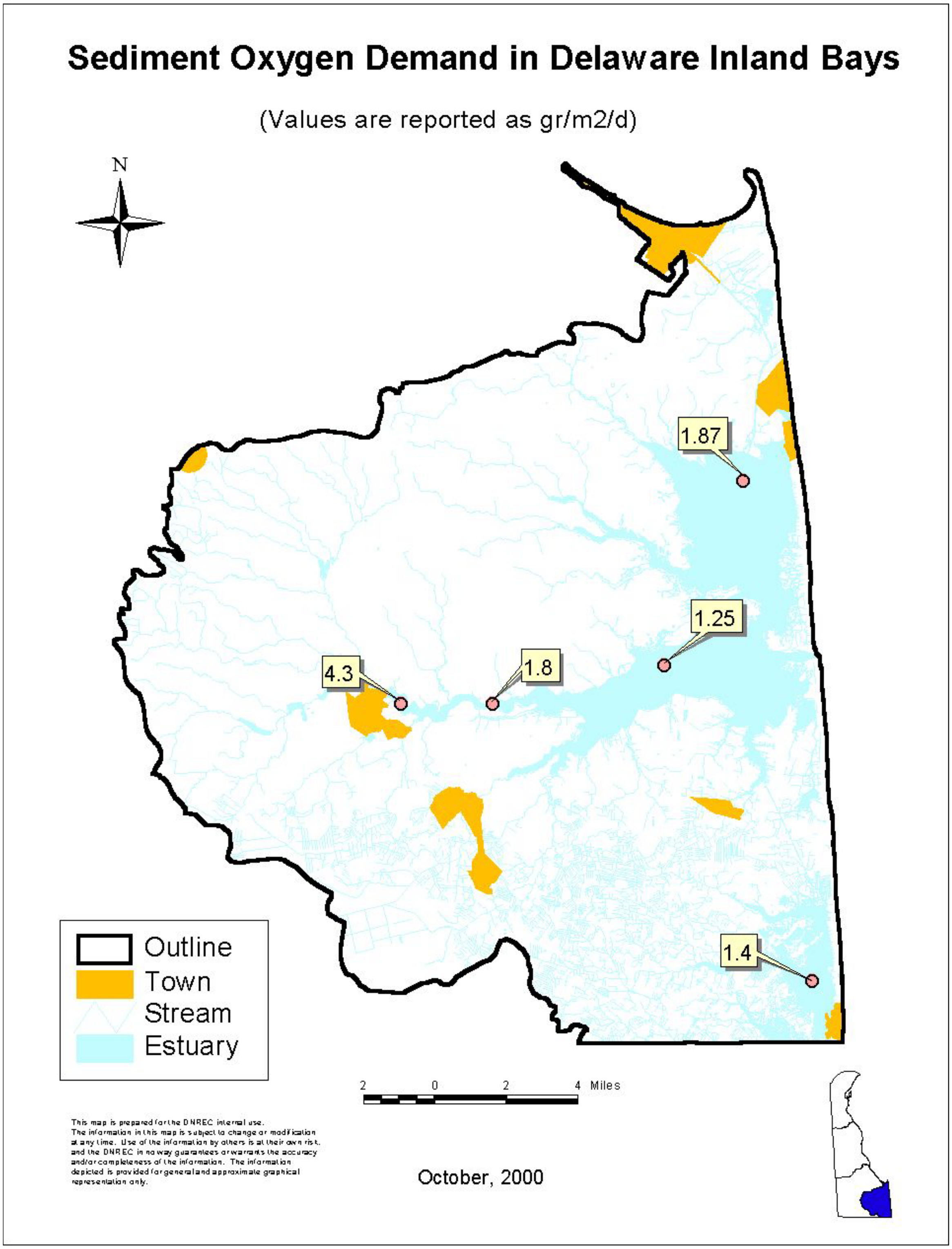
<u>RECOMMENDATIONS</u>

- In this shallow estuary, benthic pelagic coupling between the water column and sediments is potentially a large source of oxygen demand. The current model uses fluxes measured from 1992-1993 and 2001. There is no mechanistic sediment flux model associated with this modeling effort. More recent flux measurements, a sediment flux model, and explicit incorporation of benthic algae will almost certainly be necessary to complete nutrient budgets. There is also potential for the sediment to play a role in time lags between the implementation of the Pollution Control Strategy and response in the estuary that cannot be explored in the current modeling framework.

It is not entirely clear what empirical relationships between overlying water column characteristics (overlying DO and nutrient concentrations) that GEMSS used to predict sediment fluxes of oxygen and nutrients (although the locations of the observed fluxes can be seen in Figure 23). Ironically, the older TMDL model implemented by the Army Corp of Engineers included a mechanistic sediment flux model with a benthic algal model (Figure 24). The reasons for this inclusion are clear: in an estuary as shallow as Delaware's Inland Bays, benthic algae can dampen resuspension and intercept a significant amount of inorganic nutrients before they flux into the water column. Additionally, an explicit sediment model would decouple nitrification from denitrification and recycle more nutrients during early morning hypoxic periods. This negative feedback is common to eutrophied estuaries worldwide, but these non-linear dynamics require a sediment model capable of storage and release.



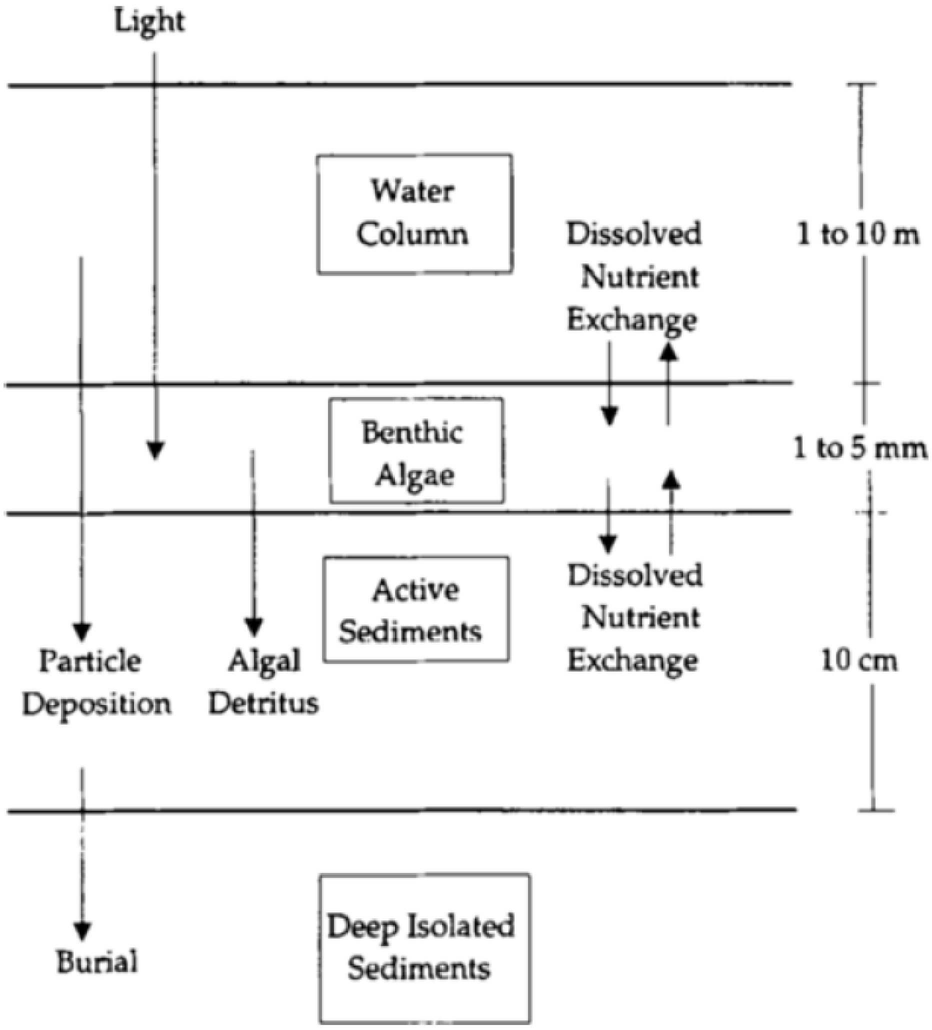




This map is prepared for the DNREC internal use.
The information in this map is subject to change or modification at any time. Use of the information by others is at their own risk, and the DNREC in no way guarantees or warrants the accuracy and/or completeness of the information. The information depicted is provided for general and approximate graphical representation only.

October, 2000

FIGURE 23. TOP PANEL: SEDIMENT FLUX MEASUREMENTS FROM 1988, MIDDLE PANEL SEDIMENT FLUX MEASUREMENTS TAKEN IN 2001, AND BOTTOM PANEL SEDIMENT OXYGEN DEMAND USED IN GEMSS


Light
Water Column
Dissolved Nutrient Exchange
1 to 10 m
Benthic Algae
1 to 5 mm
Active Sediments
Particle Deposition
Algal Detritus
Dissolved Nutrient Exchange
10 cm
Burial
Deep Isolated Sediments

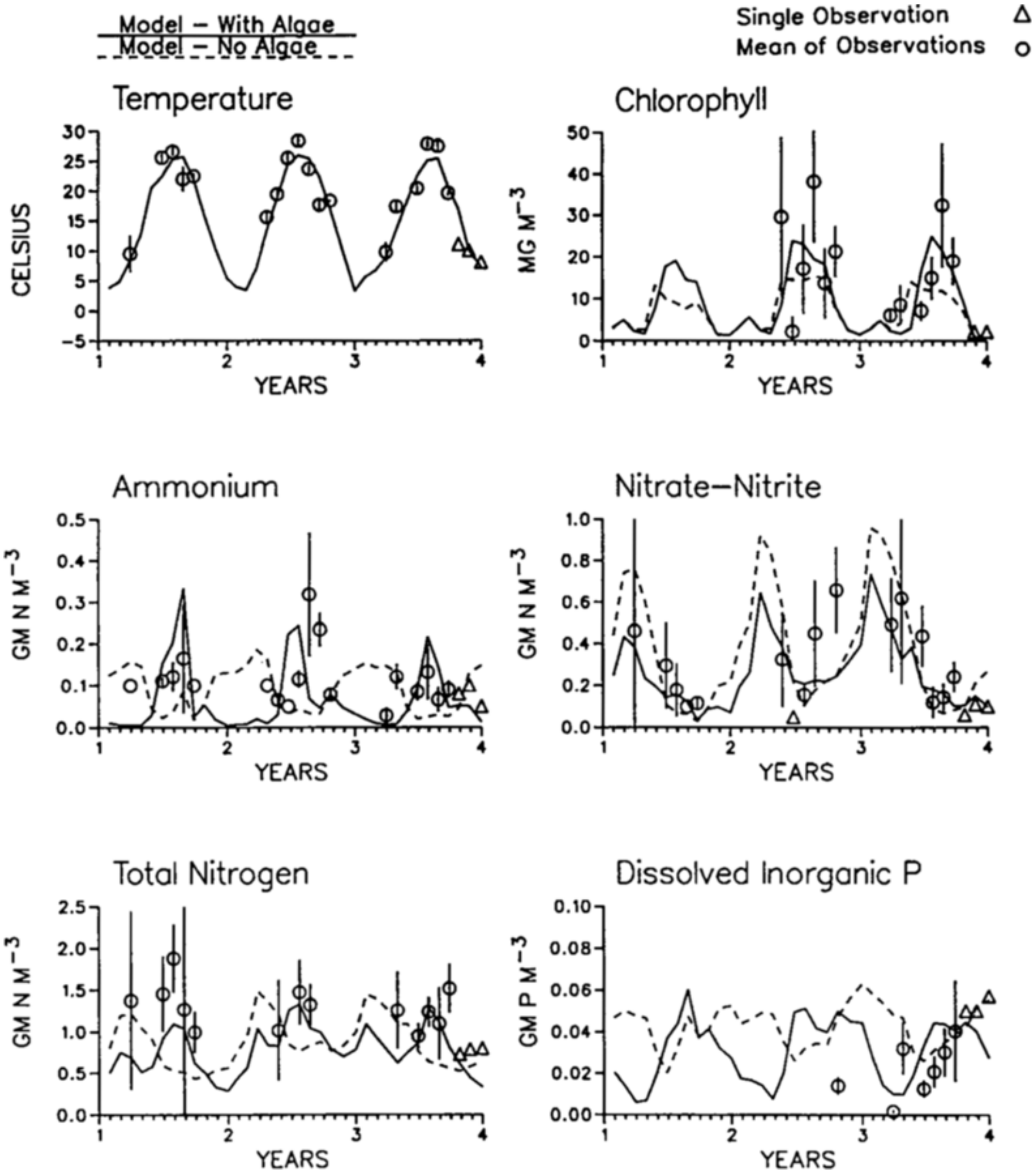


FIGURE 24. TOP PANEL: BENTHIC ALGAL MODEL INCORPORATED INTO A SEDIMENT FLUX MODEL (CERCO AND SEITZINGER 1997), THE BOTTOM PANEL SHOWS RESULTS OF BENTHIC ALGAL MODEL SIMULATIONS WITH AND WITHOUT BENTHIC ALGAE. NOTICE THAT BENTHIC ALGAE EXERTS STRONG CONTROL ON AMMONIUM AND NITRATE IN THE OVERLYING WATER COLUMN DUE TO UPTAKE

- Incorporation of primary production and respiration rates into model calibration. Seasonal respiration appears well calibrated, but daily respiration rates are clearly not large enough to generate hypoxia in the early to late morning.

It is clear that GEMSS is capable of simulating large phytoplankton blooms and the attendant DO supersaturation. Respiration of algae and bacteria at night is clearly not high enough to produce diel-cycling hypoxia in this model. In fact, no rates of primary production or respiration were utilized during model calibration and this should be revisited in a future TMDL effort.

- Increased spatial resolution particularly in tidal headwaters where recent fish tagging evidence has highlighted potential fish exposure mechanisms reliant on spatial gradients in DO

The spatial gradient in DO along the axis of tributaries may ultimately determine fish exposure to hypoxia and potentially the frequency in fishkills. Figure 25 shows an example calculation along the axis of Indian River (maximum, minimum, and median DO, CHL, TN, and TP. While it is encouraging that the model captures the increase in DO amplitude as the distance from head of tide decreases, there is some concern that the model calculates a maximum of 35 mg O_2 L^{-1} (or almost 500% saturation) and that the DO minimum is fairly unresponsive to the spatial gradient. It is the recommendation of this report that future modeling efforts increase the spatial resolution in the tributaries to increase the models ability to capture spatial gradients.

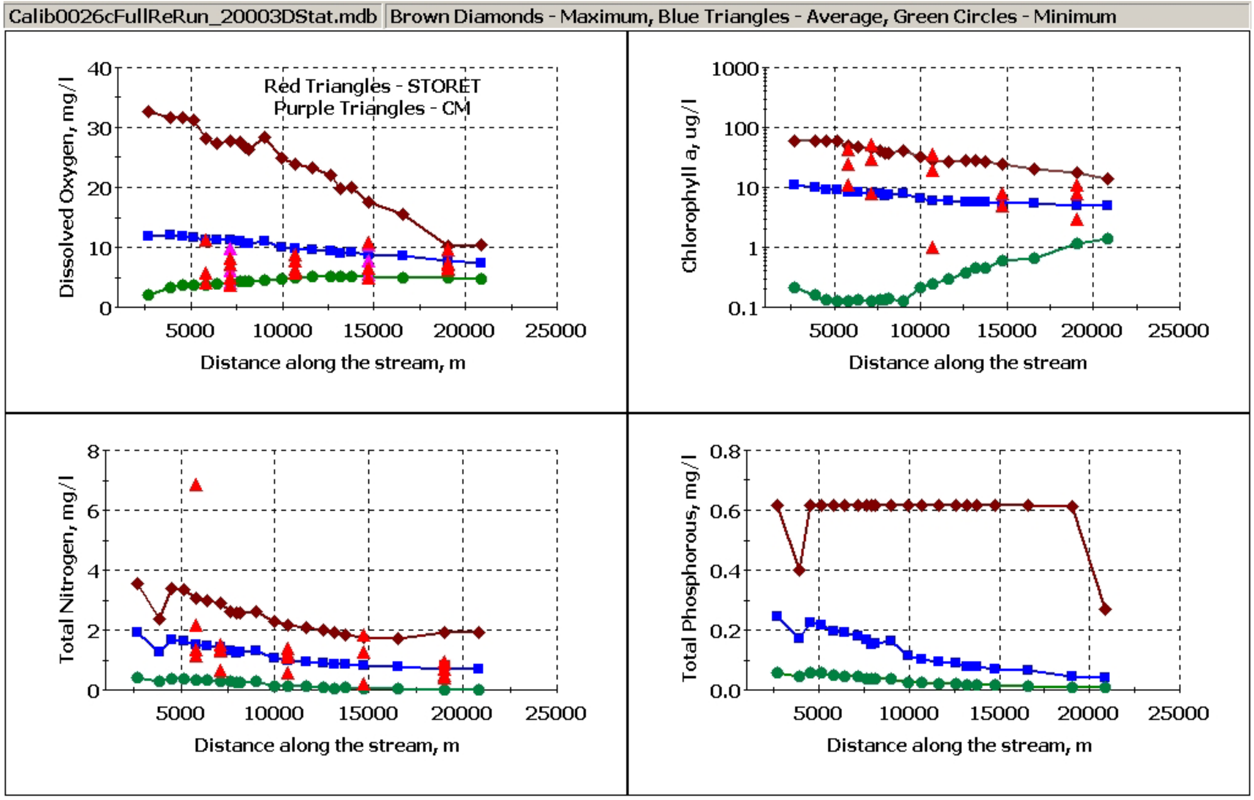
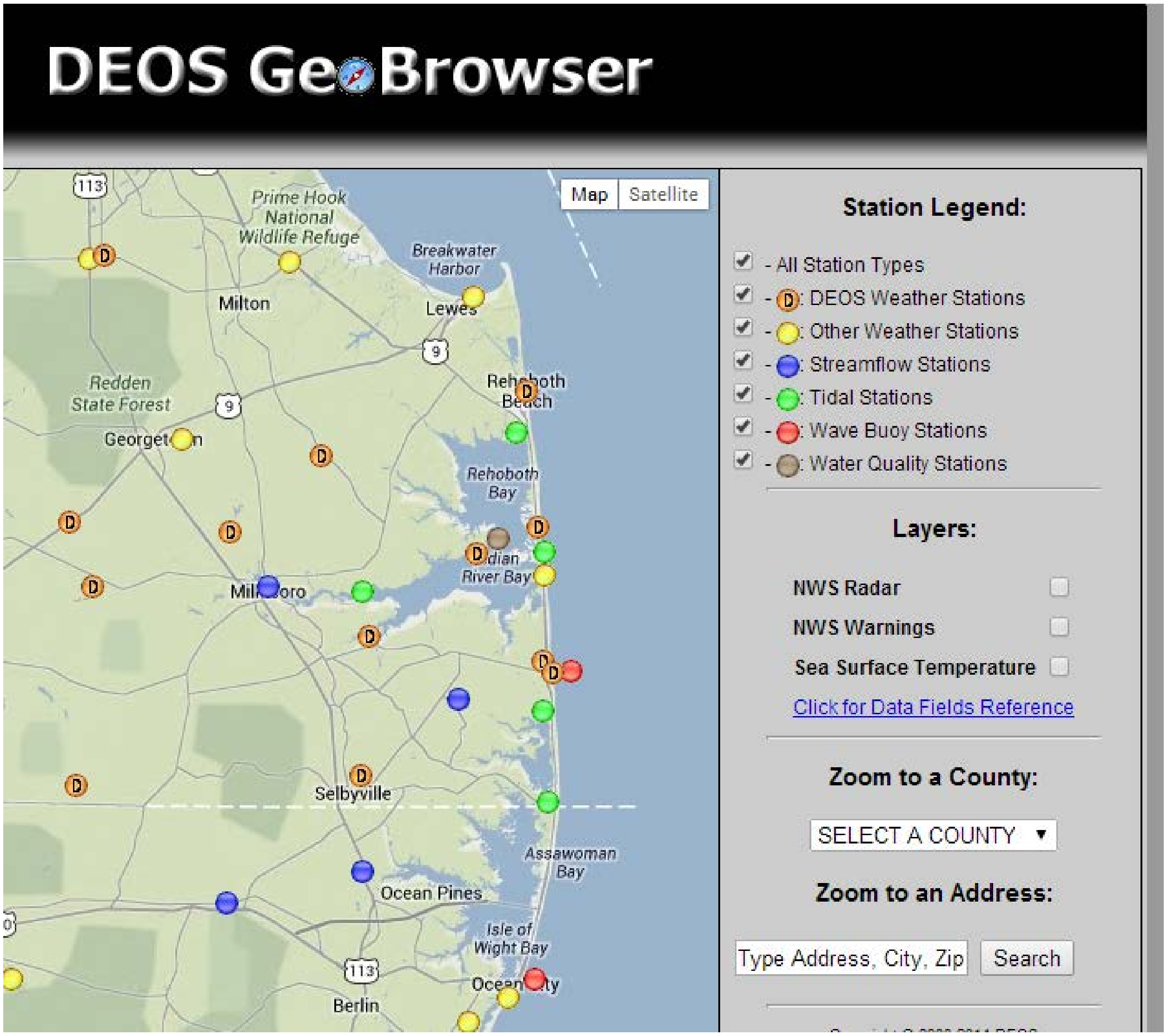


FIGURE 25. SPATIAL DYNAMICS IN (MAXIMUM, MEDIAN, AND MINIMUM) DO, CHL, TN, AND TP ALONG THE AXIS OF INDIAN RIVER IN 2000

- Incorporation of multiple meteorological records that were unavailable or offline during the calibration years (1998-2000) made available by the Delaware Environmental Observing System

Finally, since 2000, many more Delaware Environmental Observing System stations are available (Figure 26). The reason this could lead to increased accuracy is that the dynamics water quality constituents in shallow estuaries tends to be synoptically linked to short term meteorological phenomena. For example, brief periods of cloudiness can allow respiration to deplete DO, which in turn can lead to short term increase in nutrient sediment, which in turn, can increase chlorophyll. Additionally, the severity of hypoxia in Delaware's Inland Bays appears to be carefully coupled to the frequency and period of precipitation events.



Station Legend:

☑ - All Station Types
☑ - (D): DEOS Weather Stations
☑ - ●: Other Weather Stations
☑ - ●: Streamflow Stations
☑ - ●: Tidal Stations
☑ - ●: Wave Buoy Stations
☑ - ●: Water Quality Stations

Layers:

NWS Radar ☐
NWS Warnings ☐
Sea Surface Temperature ☐

Click for Data Fields Reference

Zoom to a County:

SELECT A COUNTY ▼

Zoom to an Address:

Type Address, City, Zip | Search

FIGURE 26. DELAWARE DEOS STATIONS

Delaware's Inland Bays calculated a TMDL load for data collected from 1988-1990 and then again for data collected from 1998-2000. It has been 14 years since data collected in Delaware's Inland Bays has been integrated into a 3-D hydrodynamic-water quality model. No model is perfect, some models are useful. The benefits of the modeling exercise in addition to prediction and scenario forecasting, is that they use our current level of understanding to form a narrative of how a system works. Model residuals inform us where more understanding is needed. The GEMSS effort at once integrated a large database and identified areas where improved understanding is necessary. This report argues that a new modeling effort may be necessary to incorporate new data and knowledge for better management of this ecosystem.

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