FINAL REPORT:

The Development of a Community Model of Nutrient Transport and Cycling for Long Island Sound

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Long Island Sound Study

A Partnership to Restore and Protect the Sound

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Executive Summary

Prior work has identified low rates of dissolved oxygen production and respiration and the arbitrary adjustments of the vertical eddy flux coefficients in the western Sound as central weaknesses in the System Wide Eutrophication Model (SWEM). This project addressed these issues by eliminating the mixing adjustment and by reformulation of the algal growth kinetics and dissolved oxygen budget and a systematic recalibration using the 88-89 and 94-95 data sets to make the model both hydrodynamically and biogeochemically consistent. The principal modifications are the implementation of the Jassby-Platt formulation of the algal production and the introduction of Network Common Data Form (NETCDF) output.

Most of the initial model calibration effort focused on water years 1989 and 1995, since those years received the most focus when developing the hydrodynamic and water quality models during the initial SWEM development. Simulations for 1999-2002 were also assessed. The calibration and evaluations showed the model was unable to simulate the lowest values of near bottom dissolved oxygen (DO) observed in the Sound during the critical summer period. Although model discrepancies varied from year-to-year, and the model even under-predicted observed bottom water DO in August 2000, typical 10-day average minimum values were approximately 1-3 mg/L above those observed. However, when comparing the absolute minimums computed by the model against the observed DO data, the discrepancies were much smaller, on the order of 0.25-1 mg/L.

Evaluation of the sensitivity of the minimum DO values to the selection of the parameters of the Jassby-Platt model suggests that DO could be reduced another 0.5 mg/l in the 1-day average minimums, however, this would lead to unreasonably large summertime production rates and an unreasonable reduction in the wintertime rates. If mixing rates in the model were reduced by a factor of 10 then another 0.5 mg/l reduction could be achieved.

We explored the role of the very limited resolution of lateral variation in the bathymetry in SWEM by comparing the circulation and density patterns produced by a high resolution version of the Finite Volume Community Ocean Model (FVCOM) to the SWEM simulations. The results show that the later transport is unresolved in SWEM and that it may, at times, be significant.

Steps to allow the scientific community to access the SWEM computer code and the solutions were limited by the complicated architecture of the code that has evolved over the last two decades. It is impossible to modularize the code without rewriting it. However, documentation, the original and revised code, input files and solutions have been made available through the SWEM.UCONN.EDU web site. In addition, high resolution output in NETCDF and a translator to convert the standard binary files to NETCDF were developed and released via the web site.

The model is now more consistent with the scientific communities understanding of mixing and circulation in estuaries than the previous edition and is now more consistent with observed estimates of primary production and community respiration. However, the model's failure to predict the very low DO that is observed must be addressed. The effort associated with revising the existing code, a consequence of the complexity of the architecture, suggests that a major revision of the programming of the model must be considered. The strategy should follow open source coding standards and interoperable data exchange standards to ensure broad access and the ability to contribute to model development.

Further, enhancing the spatial resolution of both the hydrodynamics and water quality model components must be undertaken. Evidence from the application of a higher resolution circulation model suggests that the across-Sound transport of water at the bottom as a consequence of by wind forcing could be significant, yet is not represented in coarse resolution models such as the Estuarine, Coastal and Ocean Model (ECOM) hydrodynamic model used in SWEM.

1. Introduction

Very low levels of dissolved oxygen (DO) have occurred in the near bottom waters of western Long Island Sound (LIS) each summer since measurements began in 1988 and this causes stress to the marine life in the region. To mitigate the extent and duration of hypoxia in LIS, the EPA and the States of New York and Connecticut have developed a Comprehensive Conservation and Management Plan (CCMP) for nitrogen loadings to the Sound using a computer model, known as the System-Wide Eutrophication Model (SWEM), to assess the likely impact of reductions in nitrogen discharged from wastewater treatment plants, combined sewer overflows (CSOs), storm water overflows (SWOs) and other non-point sources, riverine inputs and atmospheric deposition directly impinging onto the waters of the Sound. SWEM was developed and tested by HydroQual to simulate the biogeochemistry and circulation in Long Island Sound and adjacent waters and is being used to reassess the effectiveness of the CCMP and a total maximum daily load (TMDL) for nitrogen. SWEM is a complex model with many parameters that represent the rates of the processes that influence primary production and DO concentrations. O'Donnell et al. (2010) reported the results of a sensitivity study of the SWEM predictions of DO concentrations to parameter choices and boundary conditions and nitrogen loads, and assessed the effect of year-to-year variations in precipitation and wind patterns. They found some weaknesses in the model formulation and performance and this project is directed at improving these weaknesses. In addition, we propose strategies to make the model system more accessible to the Long Island Sound research community.

SWEM has two major modules. Circulation and mixing are simulated by solving the equations describing the hydrodynamics of the coastal ocean with boundary conditions that represent river flow, winds and the state of the ocean at the model boundaries. This component is called the Estuarine, Coastal and Ocean Model (ECOM). The products of this module (velocities and vertical eddy coefficients) are passed to the water quality module, known as RCA, to compute the evolution of nutrients, plankton, dissolved oxygen etc. During the development and calibration of SWEM, an ad-hoc reduction to the vertical eddy coefficients predicted by the ECOM module was introduced to reduce near-bottom dissolved oxygen in the western Sound in the summer.

Recent work on mixing in the coastal ocean and comparison of ECOM results to recent observations in the Sound suggest that the original ECOM values were actually realistic and that the absolute values of vertical mixing imposed by the *ad-hoc* reductions were much too small. By comparing recent observations in LIS and SWEM predictions, O'Donnell et al (2010) found that both respiration and production were significantly underestimated in SWEM while the levels of mixing predicted by the original circulation module were in a realistic range. They conclude that it is likely that the underestimation of respiration was the cause of the problems that led to the need for the artificial reduction of vertical mixing rates.

In this report we describe a reformulation of the DO budget to enhance the ecosystem respiration and production rates that are necessary to better match observations of DO trends and rate measurements when realistic mixing rates are employed. We then quantitatively evaluate the impacts on the 1988-9 and 1994-5 simulations. We report an assessment of the impact of these modifications on the expected response to nitrogen discharge management by replicating the experiments using data from the four additional years (1998-99, 1999-2000, 2000-2001, and 2001-2002).

To begin the development of a more modular system that facilitates the engagement of research groups interested in the nutrient cycles and oxygen budget of the Sound we have: (1) established a web site (swem.uconn.edu) to archive and distribute documents describing the evolution of the model, the computer code, and input files; (2) developed, and shared via the web site, the original computer programs to convert the SWEM standard output files to NETCDF format; (3) developed and shared the revised code and a version that has a modification to allow the output of solutions at higher frequency in binary and NETCDF format. Advice on running the code is also provided.

Since there is concern that the original hydrodynamic simulation in SWEM is too course to allow the lateral circulation and the exchange with the estuaries of Long Island and Connecticut to be influenced by the deeper parts of the Sound, we have performed an preliminary analysis of the influence of resolution by comparing the circulation in SWEM to that of a high resolution implementation of FVCOM.

In the concluding section of this report we summarize the results, comment on the limitations of our work and make recommendations on the use of this model and for additional work.

2. Results

2.1 Project Objective 1: Revise and assess SWEM

2.1.1 Task 1a: *Remove mixing limitation and revise algae/DO system*

Appendix 1 provides a comprehensive discussion of the model revisions implemented in this project and an extensive discussion of the calibration and evaluation. The model captures the central features of water quality in Long Island Sound (LIS) and is largely consistent with our understanding of circulation and mixing in the Sound. The model predictions of the west to east gradients in total nitrogen, NH_4, NO_2+NO_3 and TP and PO_4 are consistent with the survey results of the CTDEEP. Analysis of the predicted nitrogen concentration in the summer demonstrate that it is the limiting nutrient in the Sound as is widely accepted in the literature. The spatial distribution of the algal biomass, primary production and respiration are predicted to be higher in the western Sound as than in the central and eastern Sound and bottom water dissolved oxygen is predicted to be lower in the western Sound.

However, the model does not fully capture the degree of hypoxia in the western LIS that is observed in the bottom water DO data. Potential contributing factors to this problem, particularly for the water years 1999-2002, may be attributed to limitations of the hydrodynamic model. The SWEM original model was extended to include 1999-2002 for the CARP project and the calibration effort tended to focus on NY/NJ Harbor, the Hudson River and the NJ tributaries. In reviewing the model versus data plots contained in the Appendix, it can be noted that the hydrodynamic model often fails to capture the vertical structure observed in the salinity and temperature data, and, therefore, does not properly capture density difference between surface and bottom waters. This limitation may contribute to the inability of the model to fully capture the minimum DO concentrations observed in the data. Future efforts should be focused on improving the hydrodynamic model calibration in Long Island Sound and the East River. In particular, the exchange between the other portions of the Harbor, the East River and the Sound should be re-examined. One reason to do this is that the parameterization of benthic filter feeding in the East River that was implemented to reduce phytoplankton biomass in the East River results in a very high sediment oxygen demand (SOD).

However, DO is still over-estimated in the East River by about 1 mg/L on average. Given the relatively high SOD, the fact that the model over-estimates DO is surprising. Contributing factors to the mis-calibration to the DO data may be related to choices in algal growth and respiration rates and grazing rates, both for the winter and summer algal groups. There are suggestions from the NH4 and NO2+NO3 levels in the model that additional algal growth could, in some years, be supported (the model does not always fully capture the nutrient limiting conditions that occur in the western Sound). However, the model does compare favorably to gross primary production (GPP), net primary production (NPP), and respiration and biological oxygen demand (BOD) data in that region, suggesting that the values in use are close to optimal.

This revision and recalibration of SWEM is more consistent than the previous version with the community consensus on the magnitude of vertical and horizontal transport rates, and production and respiration rates. However, given its limited ability to re-produce the observed bottom water hypoxia in the Sound, SWEM as presently calibrated is limited for use as a management tool to address hypoxia in the Sound by the LIS Management Committee. Although it could be exercised with various levels of nutrient reduction to evaluate the effect of nutrient reduction on phytoplankton biomass as well as the response of DO and hypoxia to potential management scenarios, the latter predictions should be viewed with caution.

2.1.2 Task 1b: *Quantitative skill and sensitivity analysis.*

O'Donnell et al. (2010) performed extensive skill assessments with the original SWEM to assess the sensitivity of predictions to parameter choices. The basic approach was to perform annual simulations in which single parameters were perturbed one at a time. In this project we implemented two new algal growth formulations in the eutrophication submodel. We investigated the Laws-Chalup (1990) algal growth model and the Jassby-Platt (1976) algal growth model. Both algal growth models are similar to that used in the Standard SWEM eutrophication model except for how they handle light. The major differences between the Laws-Chalup model and the Jassby-Platt model and the standard SWEM model are as follows:

In the Laws-Chalup model, the algal chlorophyll to carbon model is a function of nutrient and light availiability, while in the standard model the chlorophyll to carbon uses a fixed or constant stoichiometry. In addition, the Laws-Chalup formulation partitions algal respiration into a resting or basal component and a growth related component. It was thought that it might be possible to balance levels of chlorophyll computed by the model, changes in the formulation of the respiration term and dissolved oxygen. However, this was not successful.

Goebel and Kremer (2007) provide direct measurements of photosynthetic parameters and community respiration in the Sound. The information was compatible with the Jassby-Platt model formulation. We decided to replace the Laws-Chalup formulation with the Jassby-Platt formulation. The major difference from the standard SWEM formulation is that the Jassby-Platt formulation does not consider photo-inhibition of algal growth under high light conditions. Therefore, we believed that using this formulation might effectively increase algal growth computed by the model and, thus, allow us to balance the additional growth by increasing algal respiration to maintain previously computed chlorophyll levels, which had been consistent with observations, while at the same time allowing an effective increase in DO losses via increased respiration.

Switching to Jassby-Platt formulation also allowed us to base some of our model coefficients for algal growth and light to site-specific data collected in LIS by Goebel and Kremer (2007). Appendix 1 describes the calibration. We then assessed the sensitivity of the DO predictions to the parameters controlling phytoplankton growth in the Jassby-Platt formulation: PBMAX, the maximum photosynthesis rate; ALPHA, the initial slope of the production vs irradiance curve; K1GZ, the zooplankton grazing rate; and FLOCX, the fraction of production going to algal exudates.

(a)

(b)

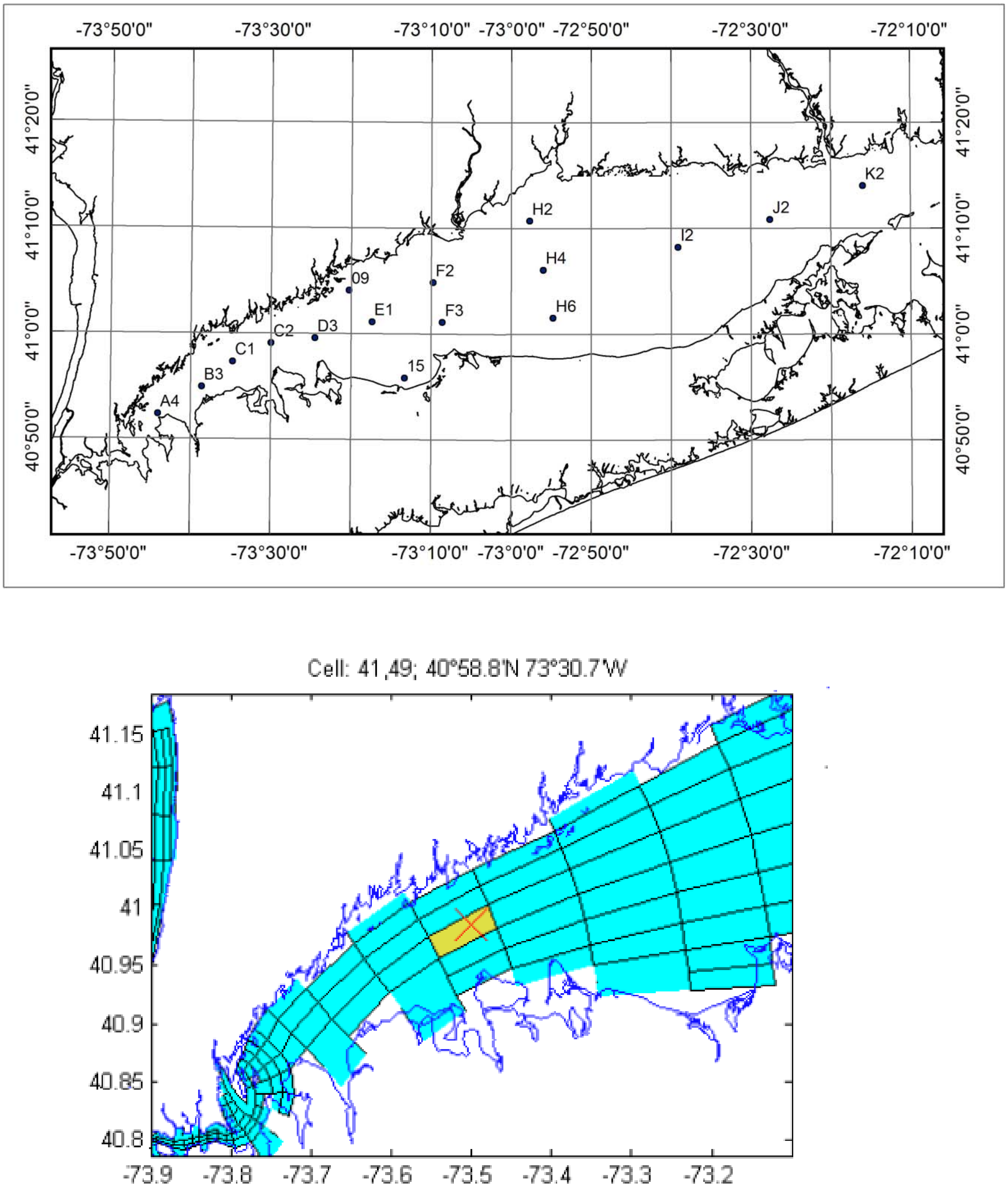

Figure 1. (a) Map of the LIS with the location of CTDEEP survey station locations. **(b)** Map of the western LIS with the location of CTDEEP station C2 with the SWEM cell surrounding it shaded yellow.

Figure 1a shows a map of the Long Island Sound coastline with the locations of the CTDEEP sampling locations that were used to calibrate and evaluate the model. Figure 1b shows a map of the western end of Long Island Sound at higher resolution with the SWEM grid as an overly. The red plus symbols shows the location of CTDEEP station C2 and the yellow shading indicates the model grid point closest to the observation station.

The plots in Figure 2 shows the model predicted minimum 10 day mean DO value at the bottom at the cell near CTDEEP station C2 as a function of the parameter values in the Jassby- Platt formulation of the algal production rates for the two phytoplankton populations modeled. The top row shows parameter values effecting the PHYT1 winter/ spring assemblage while the bottom row shows parameter values effecting the PHYT2 summertime assemblage. The blue circled points show the parameter choices used in the discussion of model calibration in Appendix 1. The red circled points represent a parameter set that shows a lower minimum DO.

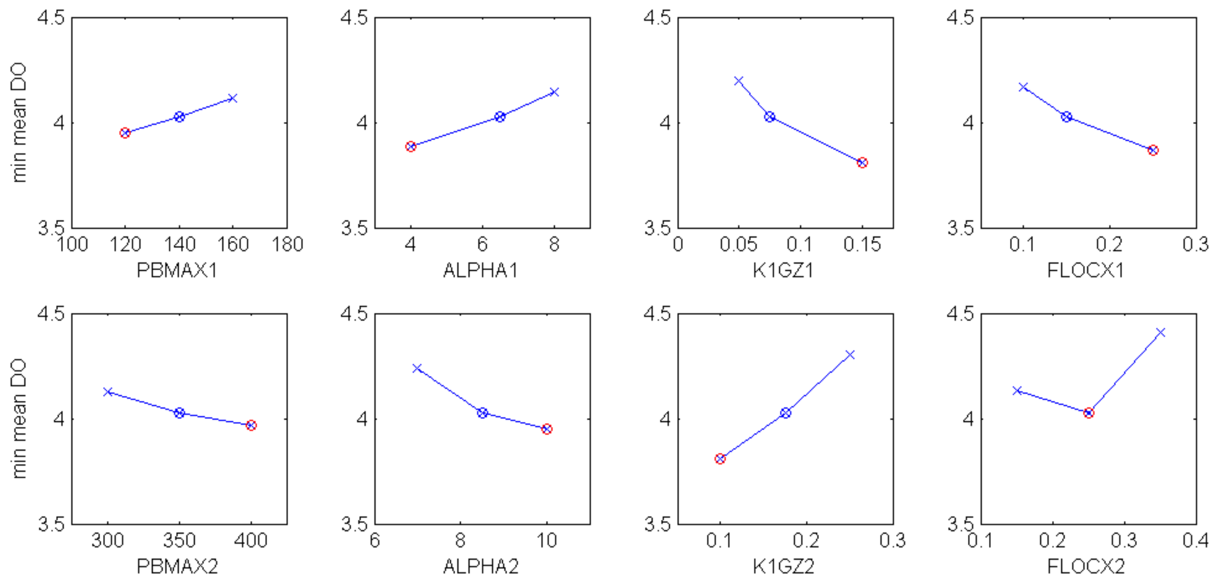

Figure 2. Variation in the predicted minimum 10-day mean DO at the bottom at the cell near CTDEEP station C2 (*y*-axes) as a function of the value of the parameters of the Jassby-Platt formulation of algal production rate (*x*-axes). The top row shows the results of varying the parameter values used for the PHYT1 population (representing winter/ spring diatom assemblage) while the bottom row shows the parameter values for PHYT2 (representing the summertime dinoflagellate assemblage). PBMAX is the maximum photosynthesis rate, ALPHA is the initial slope of the production vs irradiance curve, K1GZ is the (hard coded) zooplankton grazing rate, and FLOCX is the proportion of production going to algal exudate.

It is clear from Figure 2 that there is a weak sensitivity of the model to these parameter values that, in combination, could reduce the minimum DO by approximately 0.5 mg/l. Though this would be a slight improvement in the DO prediction, we find that the winter/ spring production rates would be slightly less than those observed and the summertime rates somewhat higher than those observed.

The vertical mixing rates used in SWEM are those predicted by the well established model of Mellor and Yamada (1982). This model predicts eddy diffusivity and eddy viscosity values that have been shown to be consistent with measurements in a wide variety of atmospheric and oceanic flows. However, in stratified estuaries measurements are difficult and the agreement between observations and predictions in the pycnocline is generally within a factor of 10 (Simpson et al., 2002). To assess the sensitivity of the model DO predictions to the predicted mixing rates we simulated the 1988 water year with all eddy coefficients multiplied by a factor in the range 0-1.5. The blue line in Figure 3 shows the variation of the predicted minimum 10-day average DO at C2 using the model coefficients defined in the Appendix A. Using the minima found in the plots shown in Figure 2 leads to the red line. As expected, less vertical mixing leads to lower DO near the bottom and a factor of 10 reduction leads to approximately a 0.5 mg/l reduction in both sets of calculations.

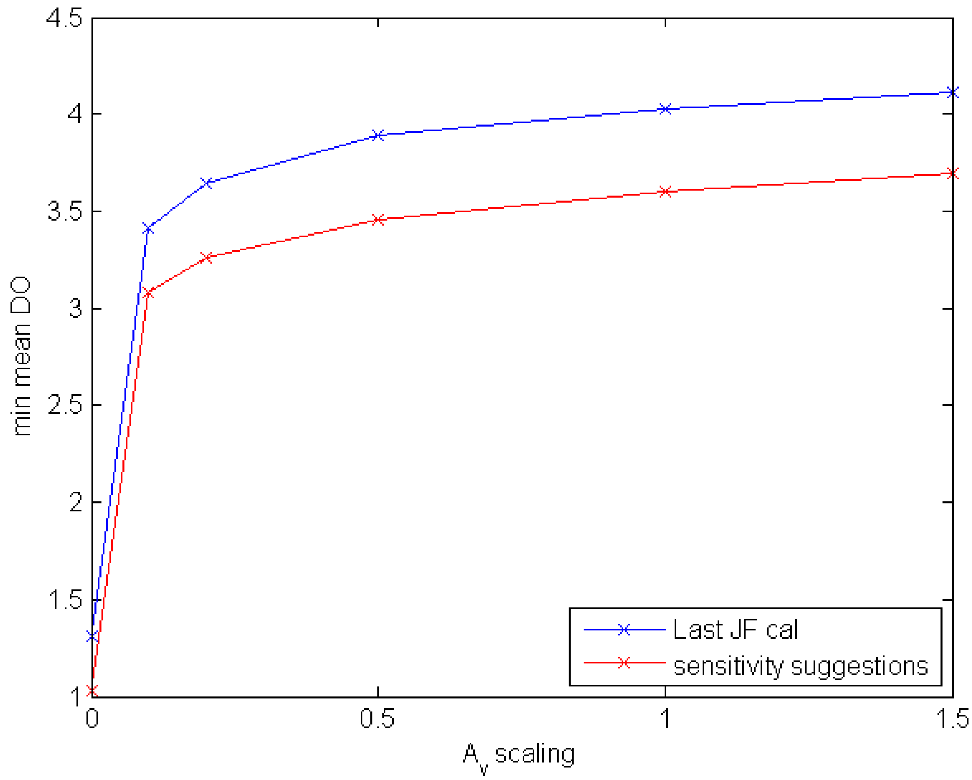

Figure 3. Minimum 10-day mean DO at the bottom at the cell containing C2 as a function of modification to the vertical mixing in the model. The abscissa values indicate the factor by which the ECOM vertical mixing values used in SWEM were modified.

The quantitative assessment of the simulation of other parameters can be assessed by computing the skill statistic S= 1 - $\sigma_{md}^2 / \sigma_d^2$, where σ_{md}^2 is the variance in the difference between the model and climatology of observations and σ_d^2 is variance in data, for each variable (total particulate P, total dissolved P, dissolved phosphate, total dissolved N, ammonium, nitrate, total particulate Si, total particulate organic carbon, total dissolved organic carbon, dissolved oxygen, and chlA) separately. With this definition S=1 when the model and data are in perfect agreement. If S>0 then the model can discriminate between years. The skill values we obtain for each model vary depending upon the year we test (88-89 or 94-95) but are negative for all variables in 94-95 and positive for only dissolved phosphate and nitrogen, dissolved and particulate silica and total dissolved organic carbon in 1988-89. The skill in the revised model is substantially reduced when realistic mixing is added and none of the adjustments to the revised model formulation are effective in increasing it.

2.2 Objective 2: Modify SWEM to facilitate access to the model, data and solutions

2.2.1 Task 2a: *Make Documentation available on the website*

We developed a website (SWEM.UCONN.EDU) to distribute information about the development of SWEM and the user manual to facilitate broader community access to the code. The original and revised model codes and input files are also shared through this web site.

2.2.2 Task 2b: *Add NETCDF IO capability*

To begin the transition of SWEM to a community modeling framework we have developed computer code to translate the standard binary output files from SWEM to NETCDF. NETCDF (Network Common Data Format) is a set of software libraries and self-describing, machine-independent data formats that facilitates the creation and sharing of scientific data. NetCDF is becoming a *de facto* standard in the scientific computing community for sharing and displaying model input and output files. There are a number of freeware tools available with which to visualize NetCDF compatible data files. The translator is available at SWEM.UCONN.EDU.

In addition, the changes to the model output that allow evaluation of high frequency results have been developed for both binary and NETCDF output. This code is available at SWEM.UCONN.EDU.

2.2.3 Task 2c: *Create Project Wiki*

We had proposed to develop a multiuser software management system, however, the difficulties arising from the program architecture, in revising the code limited the value of this functionality and the Wiki was not implemented. The revisions were largely limited to temporary changes to facilitate debugging and testing and were not useful as part of a code development process.

2.2.4 Task 2d: *Initiate model revision management*

Community model development and code sharing is accelerated by the use of version control software. We had proposed to implement Apache SVN (http://subversion.apache.org/features.html) to maintain the model system. The difficulties, however, in revising the code due to the program architecture limited the value of this functionality and SVN was not implemented. There is really only two versions, the original and the final. These are shred via the model web site SWEM.UCONN.edu.

2.2.5 Task 2e: *Install and test Model Coupling Toolkit*

Difficulties associated with the development, implementation, and testing of revisions to the SWEM code meant that the version used in the simulations was not available until late in the project period. Since the architecture of the program was established over two decades ago it is not modular and this foreclosed the option to develop coupling while testing and developing the ecosystem components of the model.

2.3 Objective 3: Evaluate Assimilation Strategies

2.3.1 Task 3a: *Assess assimilation FVCOM*

Assimilation of observations of salinity, temperature and density, and velocity in hydrodynamic circulation models has been demonstrated to improve the ability of models to describe the state of the system in numerous applications. We proposed to assess the effectiveness of assimilation on the simulations with FVCOM but the effort required to revise and test the SWEM code foreclosed this.

2.3.2 Task 3b: *Assess effect of assimilation on DO skill*

Since the time we had budgeted for implementation of the water quality of FVCOM was reallocated to the revision of SWEM we were unable to evaluate the effectiveness of assimilation.

2.4 Objective 4: Assess sensitivity to Meteorology

2.4.1 Task 4a: *Repeat 6 years with revised SWEM*

To conduct this assessment, it would have been desirable to rerun the hydrodynamic model, in order to address some of the deficiencies, i.e., salinity and temperature stratification, identified during the course of this work. However, re-running and re-calibrating the hydrodynamic model was outside of the scope and budget of this assignment. Therefore, this assessment was limited to using the available hydrodynamics and exercising the water quality model. In reviewing, the spatial model versus data plots that are presented in the calibration section of this report, it is difficult to discern whether the model is sensitive to changes in meteorological conditions, since we are looking at just one set of model outputs along the center of the Sound. Therefore, we post-processed model output for all segments in the Long Island Sound portion of the SWEM model domain for each of the 6 years in our simulation period. We determined the DO area-days for hypoxic bottom water for each of the years. DO area-days were computed by multiplying the spatial area of a segment or model cell for which the bottom water DO was less than 3.0 mg/L by the length of time the model segment was below 3.0 mg/L. Table 1 contains a summary of the hypoxic DO area-days, the number of days during which some portion of the Long Island Sound bottom waters were below 3 mg/L as computed by the model and the duration of hypoxia as reported by the Long Island Sound Study (LISS) office (1997) and the Connecticut Department of Energy and Environment (2011).

Table 1. Comparison of the number of days during which some portion of LIS bottom waters were below 3 mg/L in the model and the duration of hypoxia as reported by the Long Island Sound Study office (1997) and the Connecticut Department of Energy and Environment (2011).

Water Year	DO Area-Days (km^2-days)	Model duration of hypoxia (days)	LISS/CTDEEP Duration (days)
1988-1989	569	20	63
1994-1995	642	12	35
1998-1999	2,060	17	51
1999-2000	5,530	31	35
2000-2001	6,930	36	66
2001-2002	1,480	24	65

With the exception of water year 1999-2000, SWEM computes between a factor of two to three smaller number of days of hypoxia as compared the duration estimates provided by the LISS based on the monitoring program. However, the LISS duration estimate is based on the beginning date when hypoxia was first observed and when it was last observed. Aside from possible error in the biogeochemical model, the model estimates may be smaller than the duration dates for two reasons: (1) because the water quality model is subject to the limitations of the hydrodynamic model, particularly in 1995, when it appears as if there is little or no vertical stratification computed by the hydrodynamic model, and (2) the LISS duration estimates based on the monitoring program can miss meteorological-induced re-ventilation events, which may mix the upper and lower layers of the water column for a short period of time. This is an important process in the Sound as has been discussed by O'Donnell et al., (2008). If these ventilation events are missed by the 2-week CTDEEP sampling interval, the estimates of the total number of days during which hypoxia occurs may be high.

One of the key questions being asked of the model as part of this evaluation was, "Can the model explain the year-to-year variations in bottom water dissolved oxygen observed in the data?" As was mentioned in the previous paragraph, in order to truly perform this assessment it would have been desirable to re-calibrate the Long Island Sound portion of the hydrodynamic model. However, since we used the hydrodynamic model "as is" from the CARP study, the emphasis, which was on the Hudson River and New York/New Jersey Harbor, the calibration of the Long Island Sound portion of the SWEM domain is not as robust as might be desired. This is evidenced by the calibration plots of salinity and temperature presented in the Appendix. Interestingly, the hydrodynamic model computations of salinity do not vary significantly from August to August of each year, particularly with respect to the degree of vertical stratification. The model compares reasonably well to the observed August spatial profiles surface and bottom salinity for five of the six years (1989, 1995, 1999, 2001, and 2002). The model computes little vertical stratification in salinity in August 1995, which is consistent with the observed data, but over-estimates surface and bottom salinity by ~ 1ppt. The model also appears to capture the spatial profile of salinity stratification in 2000, but over-estimates the observed surface and bottom salinities by ~ 2ppt. However, the hydrodynamic model does not perform as well with respect to surface and bottom water temperature. The hydrodynamic model compares favorably to the observed spatial profiles of surface and bottom water temperature for August 1989, but, in general, over-estimates the vertical gradient in surface and bottom water temperatures for the remaining years, with the exception of August 1995, wherein the model computed little or no temperature stratification. Also, with the exception of August 1995, it appears as if the hydrodynamic model computes roughly the same spatial and vertical profiles in August water temperature, which is not consistent with the year-to-year variability observed in the data.

Perhaps, as a consequence of this behavior of the hydrodynamic model, the resulting spatial profiles of August surface and bottom computed by the water quality model do not differ from year-to-year, with the exception of 1995. It is not clear as to the degree that the problems with the hydrodynamic model contribute to the failure of the water quality model to reproduce the year-to-year variation in surface and bottom dissolved oxygen as compared to issues with the biochemical framework of the water quality model, but it is certainly a contributory factor. The Long Island Sound Study should certainly consider funding to improve and strengthen the calibration of the hydrodynamic model before consideration of providing additional resources to improve the water quality model.

A more complete diagnoses of the response to "meteorology" is presented in the Appendix.

2.4.2 Task 4b: *Nutrient scenario assessment*

Under this task, we conducted a single nutrient assessment model run, wherein point source and nonpoint source (CSO and SWO) nitrogen loadings were reduced by 58.5 percent, consistent with the nitrogen load reduction called for in the LIS nitrogen TMDL. In general, levels of algal biomass were reduced and bottom water DO concentrations increased, with a decrease in the spatial and temporal extent of hypoxic bottom waters in the Sound. Post-processing the model computations result in the following reductions in bottom water hypoxia DO area-days and the number of days of hypoxia. Table 2 summarizes these results.

Table 2. Comparison of Changes in Hypoxia Between the Baseline and the Nutrient Reduction Scenario

Water Year	Baseline		Nutrient Reduction Scenario	
	DO Area-Days (km^2-days)	Number of Days	DO Area-Days (km^2-days)	Number of Days
1988-1989	569	20	51	4
1994-1995	642	12	<1	<1
1998-1999	2,060	17	53	1
1999-2000	5,530	31	175	3
2000-2001	6,930	36	2,456	13
2001-2002	1,480	24	77	6

Table 2 indicates an average reduction in the number of days of hypoxia of 84% under the nutrient reduction scenario. This compares closely with the nutrient reduction scenarios run using the previous version of SWEM where the number of days of non-attainment with the DO water quality standards for the States of CT and NY decreased by 83%. Additional details and results from the nutrient scenario assessment are presented in the Appendix under OBJECTIVE 4: Assess Sensitivity To Meteorology Task 4B: Nutrient Scenario Assessment.

2.4.3 Task 4c: *Climate change scenario*

Under this task, we conducted a climate change scenario, which assumed that air temperatures would increase by about 3 degrees centigrade. This is the average expected air temperature increase from a number of different climate change models as reported by the USEPA. We further assumed that this increase of 3 degrees would result in an increase of 3 degrees centigrade in the LIS water column. To accomplish this increase, we took the output from the SWEM hydrodynamic model (ECOMSED) that is read and used by SWEM and increased the water temperatures throughout the water column, surface to bottom, by 3 degrees.

The climate warming scenario results in a decrease in the bottom water DO of between 0.8-1.2 mg/L. We post-processed the results for this model scenario in a similar fashion as is presented in Task 4B. Table 3 presents the DO hypoxic area-days and length of hypoxia for the baseline and climate change scenario. As can be seen the DO area-days increases by a factor of almost 30, while the number of hypoxic days increases by a factor of 3, suggesting a large increase in the area of bottom water hypoxia might be expected as a result of changes associated with future climate warming. Part of this change in bottom water dissolved oxygen is due to the change in water temperatures resulting in a decrease in the solubility of DO in water. At a salinity of 25 ppt, a change of temperature of 3 °C results in a reduction in DO saturation of about 0.25 mg/L. Other changes are related to increases in bottom water temperature, which increases sediment oxygen demand and increases rates of bacterial respiration.

Table 3. Comparison of Changes in Hypoxia Between the Baseline and the Climate Change Scenario

Water Year	Baseline		Climate Change Scenario	
	DO Area-Days (km^2-days)	Number of Days	DO Area-Days (km^2-days)	Number of Days
1988-1989	569	20	15,934	62

Additional details and results from the climate change scenario are presented in the Appendix under OBJECTIVE 4: Assess Sensitivity To Meteorology Task 4C: Climate Change Scenario.

2.4.4 Task 4d: *High Resolution ECOM+RCA*

Due to problems encountered during the re-calibration of the SWEM water quality model, we were not able to run the high resolution version of ECOM/RCA. While we were able to setup the hydrodynamic model for water year 1995, we did not have remaining time and budget to fully develop all of the model inputs for the water quality model on the high resolution grid and to run the model. We were able to setup the hydrodynamic model and get it running over an annual cycle for 1995, but were not able to complete the development of all of the model inputs for the water quality portion of the SWEM model. Additional details are provided in the Appendix under OBJECTIVE 4: Assess Sensitivity To Meteorology Task 4D: High Resolution ECOM+RCA.

2.4.5 Task 4e: *High Resolution FVCOM*

In conjunction with other projects (O'Donnell 2012, Babb et al. 2013) we have developed an implementation of the model FVCOM (Chen et al., 2007) for Long Island Sound and performed extensive calibrations in the area of Stratford Shoals and the Eastern Sound to ensure that the major features of the circulation are reproduced. The bathymetry used in the western Sound is shown in Figure 4. The horizontal resolution is approximately 250m in most of the area shown but larger in the eastern Sound. Only limited tests have been executed for the western Sound as a consequence of the limited scope of the project, however, Figure 5 shows an example comparison between the predicted vertical average tidal current ellipse and that based on observations from bottom mounted acoustic Doppler Profilers. The agreement is very good, within 20% of the amplitude considering no attempt has been made to adjust the friction coefficients. This suggests that the model is a reasonable representation of the circulation and that the predicted patterns of flow are useful as representation of flow in the area.

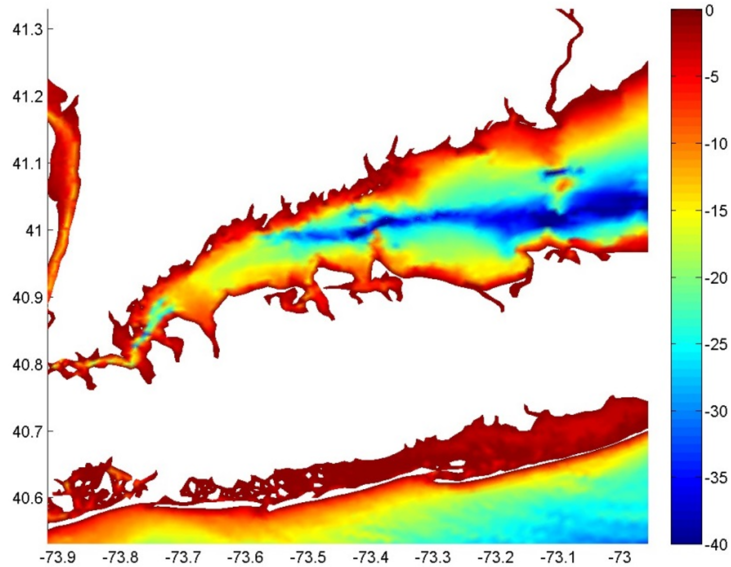

Figure 4. Bathymetry (depth in m) used in high resolution model of the circulation in the western Long Island Sound. The spatial separation of grid points is variable to resolve the topographic complexity with a minimum of 250m.

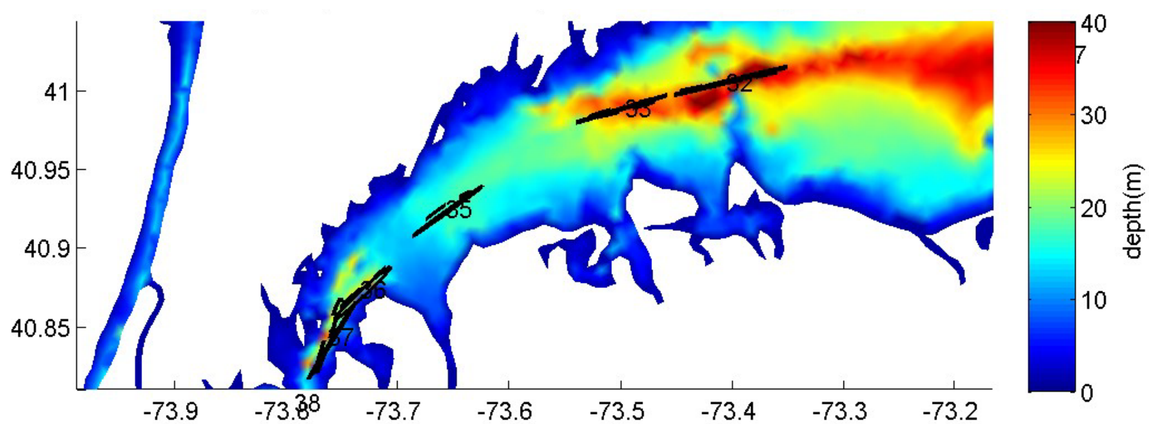

Figure 5. Comparison of the predicted M2 tidal current ellipse with that based on data from a bottom mounted ADCP.

Figure 6 shows the simulated mean bottom flow in the summer of 2013. The color scale is defined on the right of the figure. Flow is to the south west everywhere with highest speeds in the deeper areas and low values in the shallows. The winds can modify this flow substantially. Figures 7 and 8 show the mean flow at the bottom averaged over all the times in the summer of 2013 when the winds are from the north east, (characteristic of summer storms), and from the south east. During the northeast wind events the normally westward mean flow shown in Figure 6 is slowed and reversed in much of the area of the Sound prone to hypoxia. There is also a significant southeastward mean transport of bottom water towards the inlets of the north shore of Long Island. When the winds are from the southeast, Figure 8 shows that the bottom flow is in the opposite direction and again towards the north shore of Long Island. The magnitudes of the velocities are as high as 5 cm/s and it is likely that these transfer dissolved oxygen from areas of higher concentration into the deeper water and perhaps into the embayments. These fine-scale across-Sound flows are not represented in SWEM.

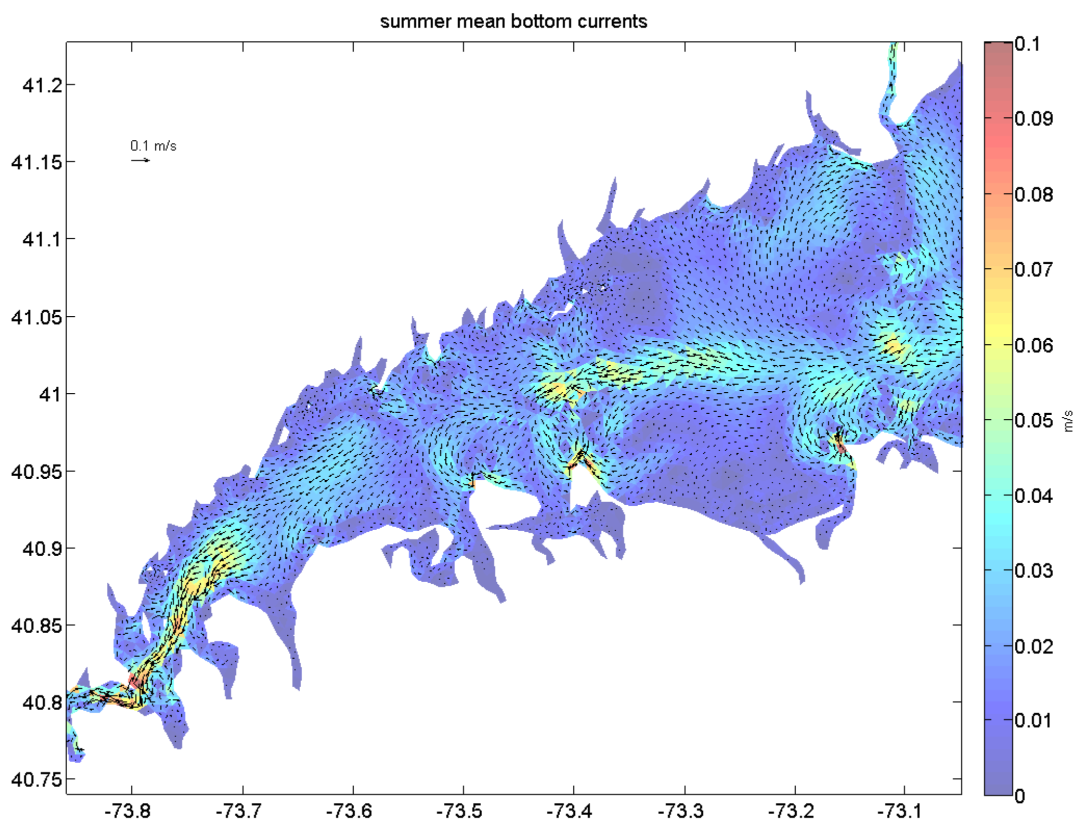

Figure 6. Mean bottom current over the summer (Jun-Aug) of a simulation of 2013. The color scale on the right shows the magnitude of the current. It is negative (to the west) everywhere and correlated with bathymetry with larger values in the deeper water.

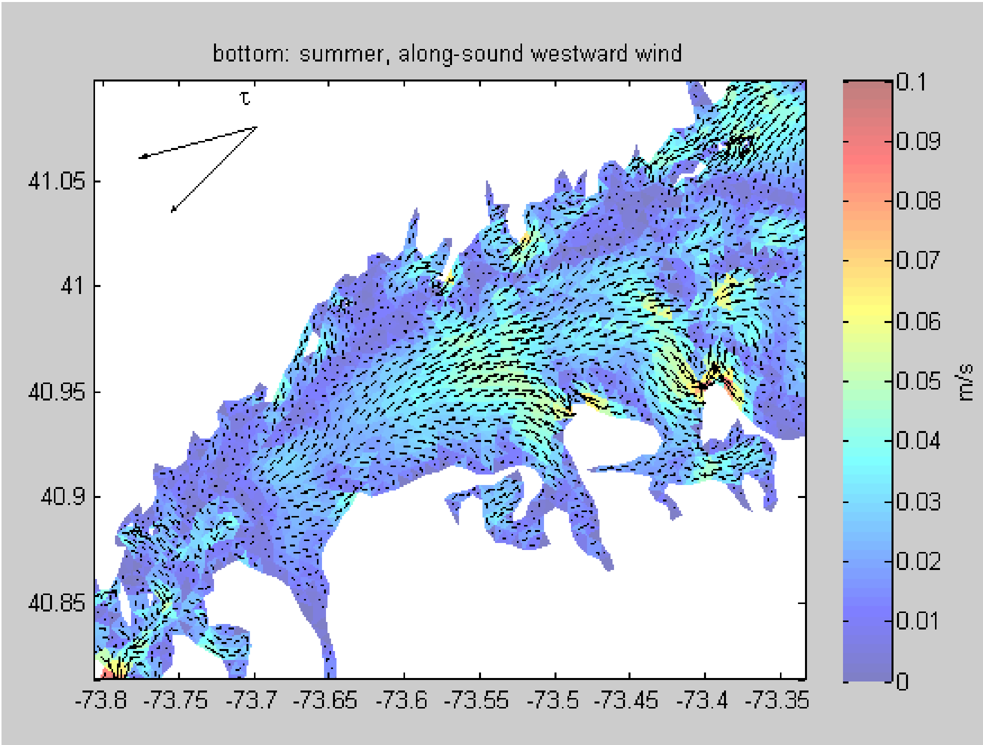

Figure 7. Mean flow at the bottom in the summer of 2013 when the wind was from the north east. The magnitude is shown by color with the code on the right**.**

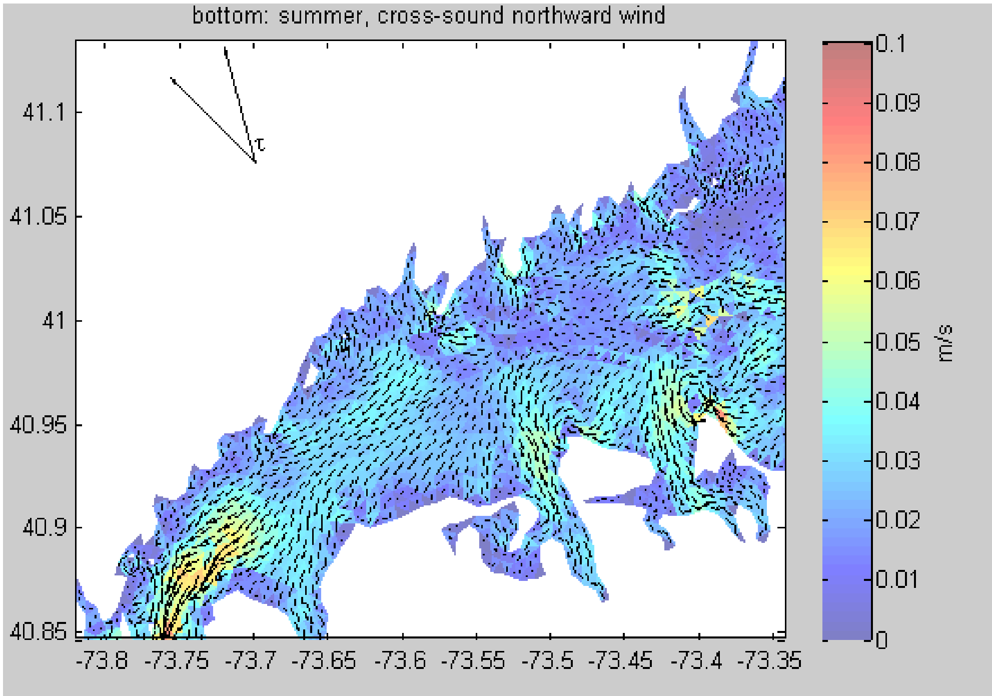

Figure 8. Mean flow at the bottom in the summer of 2013 when the wind was from the south east. The magnitude is shown by color with the code on the right**.**

2.5 Objective 5. Add mechanistic approach to modeling shellfish and kelp

2.5.1 Task 5a. *Implement Chesapeake Bay Filter Feeder Model (CBFFM)*

As per our proposal, we implemented the suspension and deposit filter feeder model developed by HydroQual, Inc. (2000) for the USACE (Waterways Experiment Station, now ERDC) and the USEPA (Chesapeake Bay Program Office). The full details of the model theory and model framework may be found at http://www.chesapeakebay.net/content/publications/cbp_12264.pdf and an overview of the model was published by Meyers et al. (2000). Essentially the CBFFM calculates suspension feeder biomass based on the assimilation of organic particles filtered from the water column. Biomass can be lost through respiration, predation and mortality due to hypoxia. The suspension feeders can filter both organic and inorganic suspended matter from the water column and either assimilate it or deposit it to the sediment. Respiratory end-products are returned to the water column as components of inorganic nutrient fluxes. Oxygen consumption, as a part of suspension feeder respiration, is incorporated into the sediment flux model's calculation of sediment oxygen demand.

2.5.2 Task 5b. *Test Revisions And Document Code Changes*

After implementing the CBFFM code into SWEM, we performed numerical checks of the model output against hand-computations to check source/sink terms in the code and to confirm that the CBFFM as linked to the water column and sediment flux model (SFM) maintained mass balances. We also modified the RCA (the computational platform upon which SWEM is based) User's Manual to provide an overview of the model formulation and provide details as to the inputs required to run the CBFFM model in conjunction with SWEM. The updated User's Manual is provided under separate cover from this report.

In order to fully implement the suspension feeder model, it is required to have sufficient spatial and temporal estimates of bottom suspension feeder biomass with which to calibrate the model. Required data include taxonomic identifications to the lowest possible level with abundance and biomass (as ash-free dry weight (AFDW) for each taxon. Biomass should either be directly measured or estimated based on morphometric (length, width) relationships to individual mass (e.g., Ranasinghe et al., 1996). Sampling should also include observations of water quality (e.g., temperature salinity, and dissolved oxygen) collected contemporaneously with benthic sampling. From these data a relational database can be developed and used to extract and analyze data for dominant individual species and in the reductionist, lumped categories to be employed by the benthic modeling framework. Also, it would be highly desirable to obtain measurements of rate processes associated with suspension feeders (i.e., filtration and food assimilation rates, respiration rates, etc.) with which to parameterize the model.

3. Conclusions

Making the hydrodynamics and geochemical transport consistent is the main improvement to SWEM that this project initiated. The original version of SWEM used transport rates for dissolved oxygen that were substantially different from that used to predict the temperature and salinity distributions. This was inconsistent with widely accepted science. The model has now been modified so that the geochemical transport is consistent with the hydrodynamics. However, this required that other processes in the water quality component of SWEM be substantially modified for the predictions of dissolved oxygen to be consistent with observations. This was a difficult challenge, and although the revised model is now consistent with expected transport rates, the dissolved oxygen levels predicted by the current model calibration have larger residual errors than the original model.

The model captures several important features of water quality in Long Island Sound. These include:

- The west to east gradients in total nitrogen, NH_4, NO_2+NO_3 and TP and PO_4,
- That nitrogen is the limiting nutrient in the Sound,
- That algal biomass, primary production and respiration are higher in the western Sound as compared to the central and eastern Sound,
- That bottom water oxygen is lower in the western Sound as compared to the central and eastern Sound.

However, the model does not fully capture the extent of hypoxia that is observed in the bottom water DO data. Potential contributing factors to this problem, particularly for the water years 1999-2002, may be attributed to mis-calibration of the hydrodynamic model. The SWEM model was extended to include 1999-2002 for the CARP project and the calibration effort tended to focus on NY/NJ Harbor, the Hudson River and the NJ tributaries. Future efforts should be focused on improving the hydrodynamic model calibration in Long Island Sound and the East River.

In particular, although we did not fully explore whether additional spatial resolution could benefit the hydrodynamic and water quality models, this should be explored. One could also take the existing SWEM model and increase the vertical resolution by a factor of two (20-sigma layers rather than 10-sigma layers) to see if this could improve the water quality model. Even though the hydrodynamic model seems to be computing the correct scale for the vertical diffusivities, the 10-sigma layer resolution may have some artificial numerical mixing that contributes too much vertical exchange of DO.

In addition, the exchange between the other portions of the Harbor, the East River and the Sound should be re-examined. One reason to do this is that the parameterization of benthic filter feeding in the East River, necessary to reduce phytoplankton biomass in the East River, results in a very high SOD. Despite this high SOD, DO is still over-estimated in the East River by about 1 mg/L on average. Other contributing factors to the mis-calibration of DO may be related to choices in algal growth, respiration, and grazing rates, both for the winter and summer algal groups. There are suggestions from the model values of NH_4 and NO_2+NO_3 that additional algal growth could be supported (the model does not fully capture the nutrient limiting conditions that occur in the western Sound). However, the model does compare favorably to GPP, NPP and respiration and BOD data in that region, suggesting that it may be difficult to increase algal growth much further.

The model as presently calibrated should be exercised with various levels of nutrient reduction to evaluate the effect of nutrient reduction on phytoplankton biomass as well as the response of DO and hypoxia to potential management scenarios. This would help inform the LISS program and Management Committee as to the utility of the SWEM model for use as a management tool to address hypoxia in Long Island Sound.

Reviewers have noted several additional areas for future study. The suggestion that POC levels were too low and that the loading from rivers might not accurately represent the source of allochthonous carbon to the Sound. The magnitude of this uncertainty is difficult to estimate and additional observations are warranted. The transient effects of wind was a fruitful suggestion that prompted the FVCOM experiments but the influence these have on the DO budget remains to be assessed and addition modeling at high resolution is required. However, the addition of local winds is likely to increase near bottom DO. *In situ* determination of rates of production, respiration, and nutrient variations at several depths in the western Sound for at least a week to resolve variability is essential to constraining the rates in the model.

Several tasks in this project associated with transforming SWEM to a model that could have more community participation in the development were not completed as a consequence of the difficulties encountered in modifying the formulation and calibration of the biogeochemistry. The biogeochemistry component of the model (the fundamental equations and parameterizations) was originally implemented (coded) in the Formula Translator computer language (FORTRAN) and has been revised and modified in many occasions as it has been applied to different estuaries. In principle, experienced scientists should be able to substitute alternative formulations of the biogeochemical cycles or extract values of variables in a straightforward manner. The fact that the changes implemented in this project were extremely time consuming reflects the complexity of the current version of the computer program. We had planned to streamline and better document the code to facilitate future testing and evaluation but time only permitted the limited implementation of device- independent solution output formats (NETCDF). Achieving the goal of transitioning SWEM into a community model requires either a complete reformulation of the programming of SWEM, or the implementation of the equations and parameterizations in the current SWEM in an existing, well programmed and documented model system. The essential characteristics of the revised model should include:

(1) an open-source modular design that facilitates implementation of alternative parameterizations
(2) NETCDF input and output files
(3) a revision management system
(4) documentation (e.g. a user guide)
(5) solution file sharing
(6) complementary analysis and visualization tools
(7) an ability to work with alternative hydrodynamics models.

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APPENDIX 1

TASK 1A: REMOVE MIXING LIMITATION AND REVISE ALGAE/DO SYSTEM RE-CALIBRATION ANALYSIS

INTRODUCTION

Very low levels of dissolved oxygen (DO) have occurred in the near bottom waters of western Long Island Sound (LIS) each summer since measurements began in 1988 and this causes stress to the marine life in the region. Similar patterns of low DO occur in many urbanized estuaries and the phenomenon is termed hypoxia. To mitigate the extent and duration of hypoxia in LIS, the EPA and the States of New York and Connecticut have developed a Comprehensive Conservation and Management Plan (CCMP) using a computer model to assess the likely impact of reductions in nitrogen discharged from water treatment plants and non-point sources. This model, known as LIS 3.0, was also used by the States of Connecticut and New York to develop a nitrogen TMDL for the Sound.

Subsequently, an improved model, the System Wide Eutrophication Model (SWEM) was developed and tested by HydroQual, Inc. to simulate the biogeochemistry and circulation in Long Island Sound and adjacent waters and is being used to reassess the effectiveness of the plan. The model, which is comprised of a hydrodynamic submodel, ECOM (Blumberg and Mellor, 1987, Blumberg, 1996), and a water quality eutrophication module, RCA (HydroQual, 2004), has a large number of state-variables and accompanying parameters that represent the rates of processes that influence primary production, algal biomass, nutrient cycling and for the purposes of the CCMP and the TMDL, dissolved oxygen. In 2005, the Long Island Sound Study provided funding to evaluate the effectiveness of SWEM and to identify additional studies that will improve the ability of natural resource managers to predict the impact of management strategies on the water quality of Long Island Sound. The University of Connecticut performed a detailed sensitivity analysis of SWEM to model parameters, model formulation, and inter-annual variations in weather and river discharge and provided an independent, quantitative evaluation of the model and its utility as a management tool. In their 2010 report, O'Donnell et al. (2010) identified a number of issues with the SWEM model as it had been developed by HydroQual. They included:

- an ad-hoc reduction to the vertical eddy coefficients predicted by the ECOM hydrodynamic module was introduced into the RCA eutrophication module to reduce near bottom dissolved oxygen in the western Sound in the summer. This ad-hoc reduction was found not to be supportable by observations of vertical mixing in the Sound,
- SWEM appeared to under-estimate algal production and community respiration rates as observed in the Sound.

It was concluded that the inability of the SWEM model to reproduce the observed algal production and community respiration rates was the likely cause of HydroQual's need to artificially reduce the vertical mixing rates in order to reproduce the observed DO in the western Sound. O'Donnell et al. subsequently modified some of the algal model coefficients and were able to reproduce the observed DO in the western Sound without the need for the vertical mixing adjustments. However, a subsequent review of the changes in the model parameters suggests that they are not realistic from a biological perspective. Subsequently, the LISS issued an RFP that would provide funding for making improvements to the SWEM model, to make it a community model, and to add additional ecosystem modules (suspension feeding bivalves, seagrasses, etc.) to the SWEM framework. A joint University of Connecticut and HDR|HydroQual team was the successful applicant for this RFP and was awarded the contract. This report provides a summary of the results of this effort, including a re-calibration of the SWEM model to 1988-89 and 1994-1995, as well as extending the calibration period to include the water years 1999, 2000, 2001, and 2002.

Model Calibration

As mentioned above, the initial calibration period for SWEM was for water years 1989 and 1995. The model validation was extended to include water years 1999-2002. However, the focus of the extended calibration period was to be on the calibration of the water quality model, RCA, and not on the hydrodynamic model, ECOM. As a consequence, the project team utilized available hydrodynamic files and calibration based on the application of SWEM to address contaminant fate and transport in the NY/NJ Harbor complex. The development and application of SWEM to include toxic contaminants found within the NY/NJ Harbor complex was funded by the Contamination Assessment and Reduction Project (CARP), which brought together a number of federal, state, and non-government partners. Funding for CARP was provided by the Port Authority of New York and New Jersey, the NJ Department of Transportation, the Empire State Development Corporation, the USACE, the Hudson River Estuary Management Program, the USEPA Harbor Estuary Program and the Hudson River Foundation. The development of the CARP hydrodynamic/water quality/contaminant model was performed by HydroQual (2007a, 2007b), while under a sub-contracting agreement with the Hudson River Foundation.

The CARP version of SWEM was expanded beyond the initial 1989-1989 (water year 1989) and 1994-1995 (water year 1995) SWEM calibration to include water years 1999, 2000, 2001, and 2002. While the CARP study used the SWEM model framework, ECOM hydrodynamics and RCA water quality, and thus included Long Island Sound within the model domain, the principal focus of the CARP study was on NY/NJ Harbor and little attention was paid attention to the calibration of the CARP model in Long Island Sound. Therefore, as will be shown in some of the calibration results to follow, problems with the hydrodynamic model limit the ability of the SWEM eutrophication model to reproduce the observed bottom water DO in the Western Sound. It is important to note, though, that not all of the DO mis-calibration results can be attributed to the ECOM hydrodynamic model. Some of the mis-calibration appears to be due to either a problem in the overall framework used for the eutrophication model or the choice of model coefficients or a combination of both. This will be explored subsequently.

DATA REVIEW

The first step in preparing to extend the calibration period to include water years 1999-2002 was to perform a review of the Long Island Sound data. Physical, chemical and biological data were collected in 1999-2002 by the Connecticut Department of Environmental Protection (now the Connecticut Department of Energy and Environmental Protection (DEEP)) at a number of stations along the length of the Sound (Figure 1). This report will present a sequence of time-series plots from three stations located along the length of the Sound going from the western Sound towards the eastern Sound (the complete set of time-series plots for the main LIS stations are contained in Appendix A of this report). The first series of plots (Figures 2 through Figure 6) show the physical (salinity, temperature, density, total suspended solids (TSS), light attenuation (K_e)), biological (phytoplankton biomass as indicated by Chl-a), and chemical parameters (DO and various forms of organic and inorganic nutrients – carbon, nitrogen, phosphorus and silica) for the period January 1998 through December 2002 for Station A4. Station A4 is located in the western Sound at Execution Rocks with a mean water depth of 32.6 meters (m). The annual cycle of temperature observed at this station (Figure 2a) is quite similar from year to year, with winter temperatures at a minimum in the January-February time period and summer maxima occurring in the July-August time period. There are some small differences between the years with the lowest temperatures being observed in the winters of 2000 and 2001, and the highest summer values being recorded in the summers of 1998, 1999, and 2002. However, the differences between years in winter low and summer high temperatures is only 2-3 °C. Salinity, on the other hand, shows both a long term increase in salinity as well as inter-annual variability (Figure 2b). Salinities usually show minima in the spring to early summer period with maxima usually observed between November and January. Bottom water salinity in 1998 ranged from highs of 27 psu, observed in January and December to a minimum of about 24 psu in late April. Over the next four years observed bottom water salinity increased at this station to a maximum of about 28.5 psu in January of 2002, with spring/summer minima of about 26.5-27 psu, resulting in a long-term average change of about 2 psu. A similar pattern was observed in the surface water salinity as well (Figure 2b). Similar to the long term increase in surface and bottom salinity, there was a long term increase in surface and bottom water density (Figure 2c). Interestingly, however, there did not appear to be a long-term increase in the density difference between the surface and bottom waters (Figure 2f). There was, however, a strong seasonal cycle in the density differences between surface and bottom with the maximum density differences being observed in the summer period, when hypoxia is observed in the bottom waters of this station (Figure 2e). As can be observed (Figure 2e) DO has a strong seasonal cycle with maximum values observed in January-February and minima observed during the mid- to late-summer period (July-August). As can also be noted maximum surface to bottom differences are also observed in the summer period, when water temperatures and density differences peak. The most interesting features of this data set are the observations of low phytoplankton biomass, as indicated by chlorophyll-a (Chl-a) observed in 1998 and 1999 (Figure 2d), as compared to 2000-2002. Chl-a levels in 1998 peak during the winter/spring at about 10 ug/L with summer values on the order of 3-4 ug/L. In 1999, maximum values of Chl-a are only about 5-6 ug/L. This is in sharp contrast to maximum values of Chl-a in the range of 15-40 ug/L that are observed throughout 2000-2002. These low levels of Chl-a observed in 1998 and 1999 are surprising because they are significantly lower than the Chl-a levels observed in 1988-1989 and 1994-1995, as well as 2001-2002 and would not be in response to changes in nutrient loadings to the Sound, since, point source and fall-line loadings did not significantly decrease in 1998 or 1999 as will be shown later in this report.

Inorganic nutrient data available at station A4 suggest a potential for greater nitrogen limitation than phosphorus limitation. Ammonium (NH_4) and nitrite+nitrate (NO_{23}=NO_2+NO_3) nitrogen concentrations minimum values occur during the summer growing season (Figure 2a and 3b) and occasionally fall below the commonly accepted Michaelis-Menton value of 0.010 mg N/L (indicated by the solid blue line) used in a large number of eutrophication modeling studies, while PO_4 minima tend to occur in the late spring/early summer (Figure 3d) and almost always are above the Michaelis-Menton value of 0.001 mg P/L commonly used in eutrophication modeling studies. In addition, with the exception of the winter/early spring of 1998, the ratio of DIN/DIP is always below the Redfield ratio of 7, another commonly used metric that supports that this station is generally nitrogen limited, as compared to phosphorus limited. Dissolved silica minima are usually observed in the late winter/spring period (Figure 3e). Assuming a value of 0.020 mg Si/L for the Michaelis-Menton value suggests that there may be some silica limitation in the spring of some years that may limit the extent of the winter/spring silica bloom (Figure 3e). This also seems to be supported by the data, observing that in the spring of 1999-2002 the ratio of DIN/Si is well above the Redfield ration of 0.5 mg N/mg Si for diatoms (Figure 3f). Interestingly, minimum dissolved inorganic nutrient concentrations in 1998 and 1999 appear to be similar to those observed in 2000, 2001, and 2002 suggesting that, despite lower concentrations of Chl-a in 1998 and 1999, phytoplankton uptake of nutrients did occur in 1998 and 1999.

Time-series plots of particulate organic carbon (C), nitrogen (N), and phosphorus (P) (Figures 4a, 4b, 4c, respectively), generally show maximum concentrations in the late spring through late summer period, with minimum concentrations observed in the winter period. Again it is interesting to note, that the maximum concentrations of particulate C, N, and P observed in 1998 and 1999 are also similar to those observed in 2000 through 2002, again suggesting that the 1998 and 1999 Chl-a data may be suspect. However, an alternative hypothesis is that the phytoplankton biomass may have been grazed by a larger than usual community of zooplankton, whose biomass may be showing up in the high concentrations of particulate nutrients. However, zooplankton data were not collected as part of the Connecticut DEEP monitoring program during those years. Concentrations of dissolved organic carbon (DOC) are generally between 1-3 mg C/L (Figure 4d), with the exception of the late summer/fall of 2002, where maximum concentrations were between 3.5 and 4.2 mg C/L. In most cases, surface concentrations are slightly elevated as compared to bottom data, the exception being the late summer/fall data of 2002, where bottom data are about 0.5 mg C/L higher than the surface data. There appears to be a slight seasonal cycle in the DON data, with the concentrations of DON generally increasing from winter through the summer/fall period (Figure 4e). Concentrations of DOP (Figure 4f) do not indicate any strong seasonal periodicity, but slightly higher concentrations of DOP were observed in the summer/fall period of 2002, similar to the pattern observed in DOC.

Time-series plots of total organic carbon (TOC) and total phosphorus (TP), (Figures 5a, 5c, respectively) show a slight seasonal behavior, with TOC concentrations peaking during the middle of the years and TP generally peaking late in the year. Mid-summer TP data also trended upwards between 1998 and 2002, increasing by approximately 0.05 mg/L between 1998 and 2002. Total nitrogen (TN) (Figure 5b) data do not show a clear seasonal cycle. Biogenic silica (BSi) data did not show evidence of either a strong seasonal cycle or a strong temporal trend (Figure 5d). Although there was a suggestion of an increase in BSi between 1998/1999 and 2000/2001, BSi concentrations in 2002 were similar to those observed in 1998. There was a general trend, however, for bottom water concentrations of BSi to exceed surface water concentrations. Total suspended solids (TSS) did not show evidence of either a seasonal cycle or an increasing or decreasing trend over the five year data period (Figure 5e). However, similar to BSi, bottom water TSS tended to be slightly higher than surface water concentrations. From 1998 through 2002 data were available from vertical profiles of photosynthetically available radiation (PAR) from which light attenuation or vertical extinction coefficients (K_e) could be estimated. From 2000 through 2002, CTDEEP also made simultaneous measurements of secchi disk depth (SDD), which could also be used to estimate K_e. The SDD data were converted to K_e using the standard Poole=Atkins coefficient of 1.7 and were compared to the value of K_e estimated from the PAR data. Generally, the K_e estimated from SDD were higher than those estimated from PAR (Figure 5f). However, the seasonal trends from both estimates were quite similar with higher K_e values during the summer period (Figure 5f). The PAR and SDD data were also processed so as to provide an estimate of the site-specific Poole-Atkins coefficient, which yielded a value of 1.23 for station A4.

A final set of time-series data for station A4 are presented in Figure 6. The first variable plotted is the ratio of particulate carbon to phytoplankton Chl-a (Figure 6a). While these data alone do not provide a true estimate of the carbon to chlorophyll ratio (which will be presented later in this section) since the particulate carbon pool also include detrital particulate carbon as well as phytoplankton carbon, there is a clear difference in the ratio of carbon to chlorophyll in 1998, 1998 and the winter of 2000, as compared to the rest of 2000 and 2001 and 2002. The ratio of particulate carbon to particulate nitrogen tends to be distributed around the Redfield ratio of 5.67C:1N with a slight increase of the ratio between 1998 and 2002 (Figure 6b). The ratio of particulate carbon to particulate phosphorus is also distributed about the Redfield ratio of 40C:1P and is significantly above that ratio in 1999 and the first six month of 2000. The ratio of particulate carbon to BSi is well below the Redfield ratio of 2.8C:1Si except for a few values in1998, 1999 and 2002 (Figure 6d). With the exception of 1999, the ratio of particulate nitrogen to particulate phosphorus is almost equally distributed around the standard Redfield ratio (7N:1P), with the 1999 data being well above that ratio (Figure 6e), suggesting a potential for phosphorus limitation, although that is not supported by the available PO_4 data. Similarly, the particulate nitrogen to BSi data (Figure 6f) would suggest a potential for silica limitation, since the ratio is well below the Redfield ratio. However, the dissolved inorganic silica data do not show a strong silica limitation at this station.

Moving to approximately mid-Sound, we reviewed data from station H4 and observed a number of similar trends in the water quality data as were observed at station A4. First, temperatures in January, February and March of 1998 and 2002 tended to be a little warmer as compared to temperatures in the same months in 1999, 2000, and 2001 (Figure 7a), and the summer of 2000 appeared to be about 2-3 degrees cooler than the other years. Also the increasing trend in salinity over the 5-year data record can also be observed (Figure 7b) at station H4. An increasing trend in density can also be observed over the 5-year data record at station H4 (Figure 7c). Interestingly, surface to bottom density differences tend to be a little stronger in the summer at station (Figure 7e) as compared to station A4 (Figure 2e), while winter density differences are actually smaller at H4 versus A4. Phytoplankton Chl-a concentrations are greatly reduced at station H4 (Figure 7d) as compared to station A4. However, there the anomaly observed in Chl-a between 1998 and 1999 versus 2000-2002 is still in evidence. Minimum bottom water concentrations of dissolved oxygen are range from 3.5 to 4.5 mg/L during the summer months (Figure 7e) and are about 2 mg/L greater than summer bottom water data observed at A4. Winter values of dissolved oxygen are also about 1-3 mg/L lower as compared to station A4, also suggesting that winter primary production is lower at H4 as compared to A4.

Evidence of a stronger nitrogen limitation at H4 during the summer months can be seen in the time-series plots of NH_4 (Figure 8a) and NO_{23} (Figure 8b) with much more of these data at or below the Michaelis-Menton value of 0.010 mg N/L and with all of the DIN/DIP data well below the Redfield ratio (Figure 8c). With the exception of a few scattered data points, the PO_4 data are well above the phosphorus Michaelis-Menton value (Figure 8d) and the Si data are also well above the Michaelis-Menton value for diatoms (Figure 8e). The DIN:Si ratio data are also well below the Redfield ratio (Figure 8f) again providing strong evidence for nitrogen limitation in the Sound.

Concentrations of spring/summer particulate carbon are about a factor of 2-3 times lower at station H4 as compared to A4, while late fall/winter concentrations are about the same. The data also indicate that summer values at H4 are higher than winter values (Figure 9a). The ratios of particulate nitrogen and particulate phosphorus at station A4 are also about a factor of 2-3 times higher as compared to the data at station H4 (Figure 9b and 9c, respectively). In contrast the concentrations of DOC at H4 are only slightly higher (10-30%) than those observed at A4. In addition, the H4 DOC data also show elevated concentrations in the summer/fall/early winter in 2002 as compared to the other years (Figure 9d). There is considerable year-to-year variability in the DON data observed at station H4 (Figure 9e), with values observed in 1999 generally being lower as compared to 1999, 2000 and 2002. DON data observed in the early part of 2002 are similar to levels observed in 1999, while DON data in the fall of 2001 are higher than in the fall of 1999. Station H4 DOP data also show considerable variability (Figure 9f) and no clear seasonal nor temporal trends.

In general, H4 TOC data show a less pronounced seasonal signal (Figure 10a) as compared to station A4 (Figure 3a), but there is a slight indication of summer maxima. The summer concentrations of TOC at H4 are also about a factor of 2-3 time lower as compared to the summer data at A4. Station H4 TN (Figure 10b) and TP (Figure 10c) data are about a factor of two lower than data observed at A4. Also the TP data show a similar increasing trend in concentration at H4 as is observed at A4. BSi concentrations are also lower at H4 as compared to A4 and also show a similar trend where bottom water concentrations generally tend to be greater than surface values (Figure 10d). A similar pattern can be observed in the TSS data (Figure 10e). Generally, light attenuation values tend to be lower at H4 and do not show the a strong season signal with a near range of between 0.4 and 0.6 m^{-1}.

Similar to the particulate carbon to Chl-a ratio data observed at A4, the data at H4 show a clear difference between 1998/1999 and 2000-2002 (Figure 11a). The PC:PN data are slightly more elevated above the Redfield ratio at H4 (Figure 11b) and may suggest a slight increasing trend, but it is not clear if it is statistically significant, given some of the within year variability. Interestingly, the PC:PP data also tend to be above the Redfield ratio, but the trend appears to be slightly downward between 1998 and 2002. Again, however, it is not clear if this is a statistically significant trend. The PC:BSi ratio data are all well below the Redfield ratio (Figure 11d), while the PN:PP data vary about the Redfield ratio and also show evidence of a trend towards greater potential nitrogen limitation between 1998 and 2002 (Figure 11e). With the exception of a few data points, the PN:BSi ratio data (Figure 11f) show a strong potential for nitrogen limitation versus silica limitation.

Finally, moving to the eastern Sound and looking at the water quality data from station M3, we see that water temperatures (Figure 12a) are somewhat moderated by the coastal ocean. Maximum temperatures are 3-4 °C cooler during the summer as compared to station A4. Winter temperatures also appear to be 1-2 degrees warmer at M3 as compared to A4. As might be expected being closer to the Atlantic Ocean salinity is also higher (Figure 12b) by 2-2.5 psu at M3 as compared to A4. Station M3 also shows an increasing trend of salinity between 1999 and 2002, although perhaps by only 1 psu as compared to the 2 psu observed at A4. Density shows a strong seasonal pattern (Figure 12c) with minimum densities observed in the mid- to late-summer period. Bottom to surface density differences tend to be smaller at station M3 (Figure 12f) as compared to A4. The concentrations of Chl-a tend to be very low at station M3 (Figure 12d), generally being less than 5 ug/L. Dissolved oxygen is significantly different at station M3 as compared to A4. Little difference between surface and bottom concentrations is observed in the data (Figure 12e) and the concentrations generally range between 6.5 and 11.5 mg/L.

Concentrations of NH_4 and NO_{23} tend to similar to those concentrations observed at H4, with nutrient limitation generally indicated to occur during the summer months (Figure 13a and 13b). The ratio of DIN:DIP is well below the Redfield ratio (Figure 13c) clearly showing nitrogen limitation as compared to phosphorus limitation. Phosphorus levels at M3 are also similar to those observed at H4 and are with the exception of a few data points well above the Michaelis-Menton coefficient for phosphorus limitation (Figure 13d). Silica concentrations are also well above the Michaelis-Menton coefficient for silica limitation (Figure 13e). Similarly, the data indicate that the ratio of DIN:Si (Figure 13f) is well below the Redfield ratio for diatoms and again provides strong support for nitrogen limitation at this station.

Concentrations of PC, PN, and PP are generally at the lowest levels at M3 (Figures 14a, 14b, and 14c, respectively) and do not really show a seasonal pattern. The concentrations of DOC are also lower at M3 (Figure 14d) as compared to stations A4 (Figure 3d) and H4 (Figure 8d) and also show the unusual increase in DOC during the later portion of the year in 2000. However, the peak concentrations of DOC at M3 are slightly lower than A4 or H4, so it is not clear if the peak data represent an ocean source of DOC. The concentrations of DON are of a similar magnitude at M3 (Figure 14e) as observed at H4, while the concentrations of DOP (Figure 14f) are slightly lower.

Concentrations of TOC, TN, TP, BSi and TSS (Figures 15a, 15b, 15c, 15d and 15e, respectively) are generally 20-40 percent lower at M3 as compared to H4 and do not show any strong seasonal or temporal patterns. Observations of light attenuation as estimated from vertical casts of PAR (Figure 15f) are also about 30-40 percent lower than those estimated at H4.

Evidence of the unusual behavior of the carbon to chlorophyll ratio is also seen at station M3 (Figure 16a). The ratio of PC:PN observed at M3 (Figure 16b) shows more variability than observed at H4, but in general, tends to be above the Redfield ratio of 5.67C:1N. Similarly, there appears to be more scatter in the PC:PP ratio at M3 as compared to H4, but again the ratio tends to be greater than the Redfield ratio for most of the data. The ratio of PC:BSi (Figure 16d) are for the most part below the Redfield ratio for diatoms, while the ratio of PN:PP (Figure 16e) generally tends to be above the Redfield ratio, except for the spring/summer/fall period in 2000, where it is below the Redfield ratio. The PN:BSi ratio (Figure 16f) is also below the Redfield ratio for diatoms (with the exception of a couple of data points), but the ratio is not as low as is observed at station H4.

In general, the following conclusions can be drawn from this brief data analysis:

- There is a strong gradient in phytoplankton biomass and nutrient concentrations going from west to east, with the higher biomass and nutrient concentrations observed in the western Sound.
- There were unusually low (and suspect) concentrations of Chl-a observed in 1998, 1999 and early 2000. However, concentration data of particulate carbon, nutrients and dissolved oxygen do not suggest that primary production did not take place in those years.
- The data suggest that primary production in the Sound is strongly nitrogen limited.
- The occurrence of hypoxia is greatest during the summer months and in the western Sound. The occurrence of hypoxia also occurs when the greatest stratification is observed in the water column.
- For the 5-year period between 1999 and 2002, there appeared to be an increasing trend of salinity in the Sound.
- The late summer/fall period of year 2000 was marked by an unusual increase in the concentrations of DOC.

As mentioned in the above data analysis, there appeared to be an unusual anomaly in the observed Chl-a data in 1998 and 1999. We looked further into the analysis and performed regression analysis of POC and Chl-a. Figure 17 presents comparisons of the regression analyses of carbon to chlorophyll for all years and then for each year separately. As can be seen the regression slopes for 1998 and 1999 are a factor of 3 (1998) and a factor of almost 6 (1999) as compared to the slopes estimated for 2000-2002. The slopes themselves of 154 mg C/mg Chl-a (converted from 0.154 mg C/ug Chl-a) and 297 mg C/mg Chl-a are generally well above more conventional values observed in the marine environment. The regression analysis was further expanded to look at the regression analyses on a month-by-month and on a yearly basis. Data analyses for 1998 (Figure 18) indicate that the carbon to chlorophyll ratios vary from nearly 100 to over 500 mg C/mg Chl-a. With the exception of October 1999, which had a low C:Chl-a ratio of 21, the other monthly values ranged from a low of 138 to a high of 420, with most months above 200 mg C/mg Chl-a (Figure 19). Regression analysis for January and February 2000, showed negative slopes for the C:Chl-a ratio, while March 2000 had a exceptionally high value of over 1900 (Figure 20). April and May also had had ratios of 179 and 224, respectively. Values for the rest of the year 2000, ranged from a low of 6 to 89 mg C/mg Chl-a. Regression results from 2001 (Figure 21) and 2002 (Figure 22), appear to be more consistent with conventional C:Chl-a ratios, with values ranging from 35 to 70-75 in 2001 (the exception being a high value of 97 in April, but most of the Chl-a concentration data in that month were below 7 ug/L) and ranging from 33 to 69 in 2002 (the exception being a low of 17 in October). Based on these regression analysis results, as well as other nutrient and DO, we believe that the 1998-early 2000 Chl-a data are questionable and, therefore, we will not focus our efforts on calibrating to these data.

LOADINGS

Before beginning the extended calibration period for SWEM, we decided to update the nutrient loadings that discharge to Long Island Sound for the new years being considered in the model validation, i.e., water years 1999 through 2002. The first set of loadings updated were the point source loadings. These loadings were provided by Kelly Streich from the CTDEEP. The next set of loadings to be updated were the fall-line loadings. We followed a similar process as was used to develop fall-line loadings for LIS3.0 and the initial SWEM model. This involved performing regression analysis of concentration vs. flow. The first step in the analysis was to plots time-series data from the various tributaries to see if there were any significant trends in nutrient concentrations between the mid-1990's and the early 2000's. The only tributary in which we observed a significant trend was in the Naugatuck River, wherein total nitrogen and ammonium nitrogen decreased significantly in early 2000. This is likely in response to the Waterbury wastewater treatment plant being upgraded in 2000 to a full BNR facility. Therefore, for the Naugatuck the regression analysis was split into two periods 1993 through April 2000 and from May 2002 through December 2003. Figure 23 presents the regression results for the fall-line (Beacon Falls, CT) of the Naugatuck River for the period 1993-April 2000, while Figure 24 presents the regression results for the period May 2000 though December 2003. As can be seen there is a significant decrease in the concentrations of TN at low flows, while concentrations remain similar in magnitude at mid- to high flows. Concentrations of ammonia are also significantly lower (about 1-2 orders of magnitude) at low flows, and again are similar at mid- to high flows. Concentrations of PON are about a factor of 5 lower at low flows post May 2002, but are similar at mid- to high flows. Concentrations of NO_{23} are similar in magnitude during both regression periods, as are concentrations of DON. By way of contrast, the regression analyses for the Connecticut River are presented in Figure 25. (A complete set of regression plots are included in the Appendix). Similar to what is observed at the fall-line of other rivers, the concentration of TN tends to decrease at higher flow. For the Connecticut River there is about a factor of two decrease in TN between low flow and high flow conditions. Most of the decrease results from a reduction in NO_{23} nitrogen. Ammonia nitrogen tends to only decrease a small amount between low- and high flow conditions. Concentrations of PON tend to increase slightly with increasing flows, while DON tends to decrease with increasing flow.

Figure 26 presents some of the results for phosphorus, TOC and Si. As can be seen, the slope of the regression line for TP is almost flat. The data suggest a slight decrease in TP from low to moderate flow and then perhaps a slight increase from moderate to high flows. However, there is quite a bit of variability in the high flow data, therefore, we did not attempt to break the regression analysis into two flow ranges. Concentrations of DOP tend to decrease as flow increases, as does dissolved inorganic phosphorus (DIP or PO_4). On the other hand, the concentrations of POP tend to increase with increasing flow, perhaps a higher concentration of POP associated with runoff under high flow conditions. TOC is nearly independent of flow conditions. The silica data tend to suggest an increase from low to intermediate flows and then perhaps leveling off to a constant level even as flows continue to increase. However, we did not attempt to break the regression into two flow ranges, but instead set a maximum value for Si at high flows. In performing our regression analysis for DO it was decided to break the data into the four seasons of the year, since DO is strongly influenced by DO saturation, which in turn is strongly influenced by temperature, as are rates of biological activity. Figure 27 presents the regression results for DO for the Connecticut River. As can be seen, with the exception of the fall period, the concentrations of DO are not strongly influenced by flow. However, there is a strong seasonal signal in the DO data. Using the individual regression results for each season, we constructed a comparison of the annual DO behavior as a function of flow. The composite estimate of DO based on flow and season tends to go through the averages of the flow-binned data.

After performing the regression analysis, we then generated time-series plots of observed and estimated concentrations for the water quality variables of interest. We will present a limited sampling of those plots at this time, while the time-series plots for all tributary fall-lines and all water quality variables are included in the Appendix. Figures 28 presents time-series plots of estimated and observed concentrations of TN, ammonia, NO_{23}, while Figure 29 presents time-series plots for estimated and observed concentrations of PON and DON and the observed daily flow for the Naugatuck River fall-line. As can be seen (Figure 25) the regression model performs quite well for TN, reproducing the increased concentrations of TN during the summer low flow period. The regression model also reproduces the decrease in TN that begins in 2000. The model also does quite well in reproducing the sharp drop in ammonia concentrations that also begins in early 2000. The regression model also appears to capture the seasonal variation in NO_{23}, although it tends to under-estimate the maximum concentrations observed in some years by 10-15 percent. The regression model tend to go through the mid-range of the observed PON data, reproducing a weak seasonal signal with higher concentrations of PON in the summer months (Figure 26). The model does miss some of the extreme high and low observed data. Interestingly, the regression model does not predict a seasonal signal in PON post May 2000. The model appears to perform slightly better with respect to DON, matching a more pronounced seasonal signal in the observed DON data. This trend continues post May 2000. As with PON, the regression model does not reproduce some of the extremes observed in the DON data.

Figures 30 and 31 present a similar set of time-series plots for the Connecticut River. The regression model compares reasonably well to the observed TN data (Figure 30) although it does not fully capture some of the observed high and lows. The regression model performs less favorably for ammonia, although it does appear to go through the average of the data. The regression model performs more favorably for NO_{23}, which perhaps is more important, since the concentrations of NO_{23} appear to be about a factor of 2-3 times higher than the ammonia data. The regression model also appears to go though the mean of the PON and DON data (Figure 31), but does miss some of the extreme values observed in the data.

The next series of figures presents time-series of estimated versus observed loads for the Naugatuck River. In general, the regression model appears to provide a good comparison to the observed TN loads (Figure 32), as well as the ammonia loads, including the marked reduction in ammonia observed beginning in 2000. Whereas, ammonia loads only varied by a small degree before 2002, there are significant variations (albeit the absolute magnitudes of the loads are considerably smaller than the 1993-April 2000 period) post May-2000, which the model reproduces well. The regression model also performs well for NO_{23}. In general, the regression model comparisons for PON and DON (Figure 33) are not as favorable as for TN, NH_4, and NO_{23}, but the majority of the estimated loads are close to the observed loading data.

An alternate way of evaluating the performance of the regression model is to present scatter plots of the observed and estimated loads. Figure 34 presents such an analysis for the Naugatuck River. As can be seen for most of the water quality variables, the estimated vs. observed loadings tend to be uniformly distributed about the 1:1 line, indicating that there is not a bias in the regression model. One can see that the regression model generally performs better for nitrogen (i.e., correlation coefficients for DON, and inorganic nitrogen (NH_4 and NO_{23})) are greater than for DON and PO_4, but reverse for PON and POP) than it does for phosphorus. The regression model also performs well for TOC, Si and DO (correlation coefficients of 0.72, 0.87, and 0.88, respectively). Figure 35 presents a comparison for the Connecticut River. For the Connecticut River, the regression model performs well for all forms of nitrogen as compared to the phosphorus forms. The correlation coefficients for all forms of nitrogen are greater than 0.7, with DON having a correlation coefficient of 0.95. While the correlation coefficient for NH_4 is 0.72 (meaning that the model explains ~50% of the variability observed in the data) and while the estimated distribution is centered around the 1:1 line for the higher loads, the regression model appears to over-estimate loadings at the lower range of the observed NH_4 loads. The model does not do as well for PON, as was observed also for the Naugatuck River, but does provide a more favorable comparison for DON. The model appears to be biased high for POP and TP, but is more centered for DOP and DIP and the correlation coefficients are all greater than 0.60. The regression model also performs well for TOC, Si and DO, with correlation coefficients of 0.86, 98, and 1.0, respectively, similar to the performance for these variables at the Naugatuck fall-line.

During discussions with the Model Evaluation Group, it was suggested that we consider using the USGS LOADEST program to perform the fall-line loading analysis. We were able to obtain a copy of the program and performed a LOADEST analysis for all tributaries. In general, the results from both the HDR|HydroQual regression analysis and the USGS LOADEST program were quite similar. Figure 36 presents the time-series loading estimates for the Naugatuck River and may be compared against Figure 28. The estimates of TN are quite similar for the two models. There are some periods of time, where one model tends to estimate high relative to the other model, but there are also time where the situation is reversed. Both model provide a good comparison to the observed TN loadings. The big difference between the two models, can be seen in the NH_4 comparisons. The HDR|HydroQual regression performs better than LOADEST, but this is largely due to the fact that we broke the regression analysis into two periods for the Naugatuck River, pre- and post-May 2000. The HDR|HydroQual regression and the LOADEST regression results are also quite similar for NO_{23}, with neither model showing a better fit to the observed loads. Comparisons between the two model can also be made for PON and DON. Figure 37 presents the LOADEST comparisons for PON and DON and they can be contrasted to the estimates provided by the HDR|HydroQual regression as shown on Figure 29. In general, there is a suggestion that the HDR|HydroQual regression might be performing better for PON. The LOADEST model tends to over-estimate the lower PON loadings and also tends to compute higher/flashier peaks than does the HDR|HydroQual regression. Both models tend to be quite similar in their estimates of DON.

A similar set of comparisons is presented for the Connecticut River, contrasting Figures 38 and 39 versus Figures 30 and 31, respectively. The comparisons for TN (Figures 38 and 30) are both very similar, with perhaps the LOADEST program providing slightly higher estimates of the maximum TN loads on a few occasions. For NH_4, the situation is reversed with the HDR|HydroQual estimates being slightly higher than the LOADEST estimates. The estimates for NO_{23} are very much the same across the two methods. In general, the LOADEST program tends to provide higher loading estimates of PON as compared to the HDR|HydroQual regression model, but both are quite similar for DON (Figures 39 vs. Figure 31).

A final comparison can be made between the two estimation techniques by contrasting the scatter plots. The LOADEST loading scatter plot for the Naugatuck River is presented in Figure 40 and can be compared to Figure 34. Similar to what was noted in the time-series comparisons, the HDR|HydroQual scatter plots for TN and NH_4 appear to provide a better loading estimate as compared to LOADEST. Although the correlation coefficients for TN are quite similar (0.38 for the HDR|HydroQual regression model vs. 0.39 for the LOADEST model), the HDR|HydroQual estimates appear to be more tightly clustered around the 1:1 line. Similarly, the correlation coefficients for NH_4 are quite similar (0.38 vs. 0.35), but again the HDR|HydroQual estimates are more tightly clustered around the 1:1 line, particularly at the lower end of the loading values. Again this is likely due to the fact that we broke the regression analysis into two periods for the Naugatuck River. Other than for DIP (or PO_4), wherein the LOADEST program appears to perform slightly better, there are not significant differences between the two load modeling approaches.

Similar comparisons can be made for the Connecticut River. Figure 41 presents the scatter plots for the LOADEST program and they can be contrasted to Figure 35. The two methods provide very similar loading estimates for TN, DON, NO_{23}, DOP, DIP (or PO_4), Si, and DO. The LOADEST programs appears to provide a slightly more favorable comparison for PON and NH_4, i.e., a tighter band around the 1:1 line, as well as POP where HDR|HydroQual appears to be biased high. Given that for the most part, that both model performed quite similarly (i.e., the correlation coefficients are quite similar for most water quality variables) and that neither model provided a clear advantage, we chose to stay with the HDR|HydroQual analysis rather than to re-generate new input files for model calibration.

MODEL CALIBRATION ANALYSIS

The model calibration analysis included the water-years 1989, 1995, and 1999-2002. Most of the initial model calibration effort focused on water years 1989 and 1995, since those years received the most focus when developing the hydrodynamic and water quality models during the initial SWEM development. As was mentioned earlier, we are using hydrodynamic model output that was developed for the CARP project and as a consequence not as much emphasis was placed on the calibration of the Long Island Sound portion of the model. A review of the CARP hydrodynamic model calibration results suggests that there are periods of time, when the LIS portion of the CARP model is less than ideal. This will be discussed, as appropriate, during the presentation of the model calibration results below.

During the calibration effort, we performed a large number of model simulations, adjusting model coefficients in order to improve the model calibration to dissolved oxygen in the Sound and in particular in the western Sound, where summer hypoxia takes place. As called for in our proposal, we also modified the SWEM code to utilize the Laws-Chalup (1990) algal growth model. Basically, the Laws-Chalup model permits variable carbon to chlorophyll ratios as driven by light and nutrients and variable carbon to nutrient ratios for the phytoplankton as driven by inorganic nutrient availability. The Laws-Chalup model also partitions algal respiration into a basal or resting component and a photosynthetic component. We thought that the latter process would be important to helping us to improve the model's ability to reproduce some of the reported levels of gross and net primary production that has been reported in the Sound. However, after a number of calibration runs, we did not achieve the expected results.

During the calibration process, we also paid attention to other water quality variables, such as phytoplankton biomass (Chl-a), the various forms of nutrients and the available process data, such as primary production, community respiration and available sediment oxygen demand and nutrient flux rates. We also reviewed the most recent literature for the Sound, which has provided new estimates of primary production and community respiration (Anderson and Taylor, 2001, Goebel et al., 2006, Collins et al., 2012). As a result, we abandoned the Laws-Chalup algal growth model and implemented the Jassby-Platt model (Jassby and Platt, 1976). The Jassby-Platt algal growth model is essentially the same formulation as is used in the standard SWEM eutrophication model except for how it handles light. The Jassby-Platt formulation, as implemented in the SWEM code, essentially eliminates photo-inhibition effects on algal growth, and thus allowed us to achieve one of our modeling goals, which was to increase the models computation of primary productivity. Switching to Jassby-Platt formulation also allowed us to base some of our model coefficients for algal growth and light to site-specific data collected in LIS by Goebel and Kremer (2007).

We will present a series of calibration figures for May and August for each of the calibration years. May is an important month because it is usually near the end of the spring algal bloom, which when it ends provides a significant quantity of particulate organic matter to the sediment bed. This particulate organic matter then fuels sediment oxygen demand (SOD) and nutrient flux during the summer months, when bottom water temperatures increase and, therefore, increase bacterial processes in the sediment bed. August results are also presented, since that is usually the month when summer hypoxia is observed within the Sound. Each of the calibration figures will present surface (open upward-pointing diamonds) and bottom (filled downward-pointing diamonds) data (average and range) and the model computed monthly average for the surface (solid line) and bottom (dashed line) layers. In the case of phytoplankton Chl-a and dissolved oxygen, the monthly maximum values and monthly minimum values for the surface and bottom layers, respectively will also be plotted (very fine dashed lines). For the spatial plots, milepoint (MP) 0 is located at the Battery at the lower end of the East River and the southern tip of Manhattan Island.

Figure 42, presents a spatial profile of salinity, temperature, Chl-a, and DO for May 1989. As can be observed, the hydrodynamic model slightly over-estimates bottom and surface salinities, with the over-estimation being greater in the surface layer. The hydrodynamic model does, however, appear to capture both the surface and bottom water temperatures, including the degree of thermal stratification well. The model tend to over-estimate the surface values for Chl-a from MP 40 and eastward, but does compare favorably to the observed data in the bottom waters. The model under-estimates both the surface and bottom Chl-a between MP 20 and 30, and then over-estimates Chl-a in the lower East River (MP 0-10), which has long been a problem with SWEM. The 1988/1989 sampled for DO using two measurement techniques, via DO titration and via DO probes. On the DO calibration panel, the DO titration data are shown using open upward-pointing triangles for surface data and closed downward-pointing triangles for bottom data, while the DO probe data use open circles and closed circles for surface and bottom data respectively. The model captures the average values for the surface DO in the eastern part of the Sound, but does not reproduce the maxima observed in this portion of the Sound. The model also tends to under-estimate the bottom water DO in this portion of the Sound by about 1 mg/L. Moving towards the central Sound, the model tends to match the surface titration data, but over-estimates the surface probe data (MP 40-70). It is interesting to note, however, that there is about a 1-2 mg/L difference in the surface titration vs. probe data. In the western portion of the Sound, the model computations of the 10-day average DO concentrations tend to over-estimate both the surface and bottom DO data by about 1 mg/L, except for two DO titration observations near MP 25 and near MP 35. The model also over-estimates DO in the lower East River (MP 0-20) by about 2 mg/L. A comparison of the model computed minimum bottom water DO does, with the exception of observed data at MP 16 and MP 55 compare more favorably to the bottom water minimum DO.

The next series of spatial plots present the calibration results for the various forms of phosphorus (Figure 43). The model over-estimates inorganic phosphorus between MP 20-70 by about a factor of two. Both the data and model clearly show that the inorganic phosphorus data are well above the Michaelis-Menton coefficient for phosphorus limitation. The model also over-estimates DOP by a factor of 2-3 throughout the Sound, but especially so in the lower East River. While the model provides a favorable comparison to both the surface and bottom water particulate phosphorus between MP 35-125, the model over-estimates the particulate phosphorus in the East River. This may suggest that the splits between effluent dissolved and particulate organic phosphorus may be incorrect. The model also tends to over-estimate the total phosphorus by 50-75% throughout the Sound.

Figure 44 presents model vs. data comparisons for silica. The model significantly under-estimates the particulate (biogenic silica) throughout the Sound, except in the extreme eastern portion of the Sound. Between MP 0 and 40, the under-estimate ranges by a factor of 2-4. There is quite a bit of variation in the observed data, as well as between surface and bottom data, which we cannot explain. Looking at the dissolved silica data and the model computations, the model is computing silica limitation for the diatoms between MP 20 and 75. The data, however, are not quite at limiting concentrations, assuming a Michaelis-Menton coefficient of 0.020 mg Si/L. The model also under-estimates total silica in the western and central Sound.

Calibration results for the various forms of nitrogen are presented in Figure 45. The model matches the TON data in the lower 10 miles of the East River and most of the central and eastern Sound, but under-estimates TON between MP 15 and 70. The model also fails to capture the near nitrogen limiting conditions in the central and western portion of the Sound, wherein, the sum of NH_4 and NO_2+NO_3 approaches the Michaelis-Menton coefficient of 0.010 mg N/L. In particular, the model over-estimates concentrations of NO2+NO3 by a factor of 2-4 between MP 20 and 55. A potential reason for this mis-match may be related to the fact that the model is computing a silica limitation at this time and, therefore, inorganic nitrogen is not being removed from the water column by additional phytoplankton growth and uptake. The model comparison to the observed TN data is fairly good and captures the gradient in TN observed between MP 0-10 and MP 10-30.

Model calibration results for organic carbon, BOD, and light attenuation are presented on Figure 46. The model reproduces the observed DOC in the lower 15 miles of the East River, before under-estimating DOC between MP 16-70 and then over-estimating DOC in the eastern most portion of the Sound. In general, the model tends to under-estimate POC from the Battery to about MP 70 and then over-estimates POC in the rest of the Sound. The model over-estimates BOD between MP 0-20 by almost 2.5 mg/L and then over-estimates BOD by about 0.6 mg/L in the surface at MP 21, but only over-estimates the bottom BOD by about 0.1 mg/L. The model matches the observed and bottom BOD data at MP 27, and then over-estimates the observed BOD data from MP 35- 67 by 0.3-0.4 mg/L. The model under-estimates the observed BOD data by about 0.3 mg/L near MP 82, and then over-estimates the BOD data between MP 90 and MP 125. It is interesting though that the BOD data are so low in the East River, given the relatively higher concentrations of DOC and POC. With the exception of missing the very high value of light attenuation (K_e) at MP 0.0, the model compares favorably to the observed K_e data.

There were no primary production data with which to compare to the model in May 1989, but there were some limited respiration data with which to compare to the model. The model over-estimated two of the three available surface data by about 0.5 mg O_2/L-day and under-estimates the maximum value of about 1.7 mg O2/L-day near MP 27 (Figure 47). It should be noted through that the maximum observed datum was about 1.5 7 mg O2/L-day higher than the other two data values. There were also measured sediment oxygen demand (SOD) data to compare against model computations. The data ranged from 0.7 to about 1.2 gm O_2/m^2-day between MP 15 and 40 and then from 0.3 to 0.75 gm O2/m2-day between MP 55 and 70. The model considerably under-estimates the observed SOD data during the month of May. Similarly the model also under-estimates the observed ammonium (J_{NH4}) and silica (J_{Si}) fluxes during May (Figure 48) and then over-estimates the observed fluxes of nitrate (J_{NO3}). The model computes fluxes of NO_3 from the water column to the sediment bed, while the data indicate fluxes of NO_3 from the sediment to the overlying water column.

Comparisons of model and data for the month of August 1989 are presented in Figures 49 through 55. In general, the model computations of salinity and temperature are well produced by the data, capturing both the spatial distributions and vertical gradients observed in the data (Figure 49). The model also compares favorably to the observed surface concentrations of Chl-a, although it slightly under-estimates Chl-a between MP 15 and 35 and slightly over-estimates the observed data in the central and eastern Sound. The model also significantly under-estimates the very high concentrations of Chl-a in the vicinity of the upper East River. Interestingly the bottom data are about a factor to two higher than the observed surface data. The model does not compare well to the observed DO data in August 1989. The model 10-day averages over-estimate the surface and bottom DO data in the lower East River between MP 0 and 15 by about 1 mg/L. The model compares well to the observed surface data between MP 15 and 20 and then over-estimates the observed data (both titration (open upwards-pointing diamonds) and probe (open circles) data) between MP 20 and 30 by 1-2 mg/L. Interestingly, the model compares favorably to the titration data between MP 37 and 41, while over-estimating the probe data by about 2 mg/L. The model then compares favorably to the observed surface data between MP 45 and 85 and then lies between the titration and probe data between MP 80 and 130. The model also over-estimates bottom water observations of DO, particularly in the western Sound. Unlike the surface titration and probe data, the differences between the titration and probe data in the bottom waters of the Sound between MP 0 and 65 are generally less than a few tenths of a mg/L. The model 10-day averaged DO over-estimates the observed average DO by 1 mg/L between MP 20 and MP 30, but the model computed minimum value over-estimates the observed data minimum DO values by 1-2 mg/L. Model computed average DO concentrations compare more favorably to the observed average DO values between MP 30 and MP 65, although biased about 0.2 mg/L high between MP 47 and MP 65, but again the model computed minima tend to over-estimate the observed DO data minima by 1.5-2 mg/L. At MP 67, the model over-estimates the average DO probe data by only about 0.2 mg/L, but over-estimates the titration data by about 2.5 mg/L. The model compares favorably to the observed bottom water probe data between MP 80 and MP 130.

In general, the model compares favorably to both the observed surface and bottom DIP data, except between MP 55 and 67, where the model over-estimates the surface data (Figure 50). The model over-estimates the observed DOP data by a factor of 2-4 throughout the Sound. The model tends to under-estimate the total particulate phosphorus in the lower East River, before beginning to over-estimate the observed data in the rest of the Sound. The model tends to match the observed TP data between MP 0-70 and then slightly over-estimates the TP data from MP 80 and eastward.

The model severely under-estimates the particulate silica data between MP 0 and 70, before performing better between MP 80-130 (Figure 51). The model also tends to perform poorly when compared to the surface values of dissolved silica. The model over-estimates surface and bottom silica in the East River by about 0.2 mg Si/L. The model also over-estimates the observed surface data by 0.1-0.3 mg Si/L between MP 20 and MP 90, although a more favorable comparison can be observed versus the bottom data between MP 35 and 80. It is possible that there was a greater population of diatoms present in August of 1989 than was computed by the model; this is also suggested by the particulate silica data. The model computations of surface silica compare reasonably well to the observed surface data, but the model tends to under-estimate the bottom total silica data.

Model versus data comparisons for the nitrogen variables are presented in Figure 52. The model over-estimate TON in the western Sound by about 0.1 mg/L. The model computed surface data tend to compare more favorably to the surface data in the rest of the Sound, but in general, the model tends to under-estimate the observed bottom TON data. While the model approximately reproduces the observed 10:1 gradients in NH_4 and NO_2+NO_3 observed between the East River and the far western Sound, the model does not compute as strong a nutrient limitation for dissolved inorganic nitrogen (DIN) as is observed in the data. The surface DIN data are between 0.01 and 0.02 mg N/L between MP 20 and 30, which would reduce the algal growth rate by between a third and a half, while the model computes DIN concentrations of between 0.04 and 0.07 mg/L, levels at which significant growth reduction would not be expected to occur. This suggests that perhaps that either the maximum algal growth rate should be increased or that factors (respiration, grazing or settling) contributing to reductions in algal production should be decreased or a combination of both. Both observed and computed concentrations of DIN tend to be near or at growth limiting concentrations throughout the rest of the Sound. The model computation of TN compares favorably to the data.

The model tends to compare favorably to the observed west to east gradient in DOC (Figure 53) although there is considerable variability in the observed data. The model also compares favorably to the observed POC data in the East River and far western Sound, although it tends to over-estimate the observed POC data between MP 35-45 and from MP 80 and eastward. Similar to what was observed in May, the model over-estimates BOD in the East River, but compares more favorably to the observed data in the rest of the Sound. The model comparison to K_e is mixed, over-estimating the observed data in the far eastern Sound, comparing favorably to the data between MP 80-95 and then within the range of the observed data in the western Sound and the East River.

Model computations of gross primary production (GPP) and net primary production (NPP) are compared to primary production data reported by Welsh and Eller (1991). We are comparing the GPP and NPP to the same production data, although the Welsh and Eller paper suggests that the data are closer to being a measure of NPP. The model computed GPP values are about a factor of two greater than the observed Welsh and Eller production data, but do show maximum production to occur near MP 20, consistent with the data (Figure 54). The model computations of NPP are also higher than the observed production data, but now perhaps only by a factor of 1.0 near MP 20. Model comparisons of community respiration versus the observed Welsh and Eller data provide a mixed comparison. The model over-estimates the observed surface and bottom data near MP 16. The model also over-estimates the surface respiration data near MP 21, but compare favorably to the bottom data. The model also over-estimates the surface data near MP 23.5, but under-estimates the bottom respiration data. The model compares more favorably to the observed surface and bottom respiration data near MP 27, but under-estimate the observed surface and bottom data near MP 41 (although these data are considerably greater than the other observed data, which tend to be in the region with higher algal biomass). These model versus data comparisons (production and respiration) would perhaps suggest that the algal growth rate is not too high nor is the algal respiration rate too low, as was suggested as a correction for the mis-match between computed and observed DIN data. Perhaps, then, the adjustment needs to be made to the algal settling rate or to the grazing rate on phytoplankton. Model computed versus observed SOD comparisons indicate that the model reproduces the observed SOD between MP 35 and 80. The model appears to over-estimate the observed SOD between MP 15 and 30. However, these data are surprising low and may be an artifact of the methods used. If the cores were not aerated before being run for measuring SOD, it is possible that with low initial overlying water column concentrations of DO, the DO may have been rapidly depleted with subsequent possible production and release of hydrogen sulfide (H_2S), which was not measured, but would be an equivalent SOD. Modeled and observed fluxes of PO_4, NH_4, and NO_3 compare well to the observed data (Figure 55). The model does, however, under-estimate the fluxes of dissolved silica.

Spatial plots for the May 1995 calibration period are presented on Figures 56 and 57. Figure 56 presents model versus data comparisons for salinity temperature, Chl-a, DO and the various forms of phosphorus. The model tends to over-estimate salinity by 1-1.5 ppt and over-estimates May temperatures by 3-5 degrees throughout the Sound. Both the data and model computations for Chl-a are generally low, less than 10 ug/L, except for a single data value of almost 20 ug/L at MP 21. The model computations and observations of DO are also in general agreement throughout the Sound. The model over-estimates DIP by a factor of 2-3 throughout the Sound, but both model and data are well above the Michaelis-Menton coefficient of 0.001 mg P/L, indicating no phosphorus limitation. The model compares favorably to the observed DOP data, but over-estimates the particulate phosphorus data. The model compares favorably to the TP data for the May calibration period.

The model under-estimates the DOC data between MP 30 and 70, but then compares favorably to the data in the remaining portions of the Sound (Figure 57). The model also under-estimates the POC data throughout the Sound, while comparing favorably to the observed BOD data. The model under-estimates the biogenic silica data in the western portion of the Sound, but then provides a better comparison to the observed data in the central and eastern Sound. The model under-estimates dissolved silica limitation that is not supported by the data. In general, with the exception of the lower East River, the model under-estimates the observed TON data. The model also over-estimates both the ammonium and nitrite+nitrate data. The NH_4 and NO_2+NO_3 data suggest that nitrogen is at limiting concentrations. However, the model computations do not show evidence of nitrogen limitation. These results may be due to the fact that the model is projecting silica limitation, which reduces the uptake of DIN by the phytoplankton. The model computation of TN compares favorably to the observed data, including the observed spatial gradient in TN going from west to east, suggesting that the total nitrogen loading to the Sound is correct, but that the component species are not being computed correctly.

The lack of a successful calibration for August 1995 may, in part, be attributed to the failings of the hydrodynamic model. While the model, over-estimates the observed salinity by 1-2 ppt in August, it does show the relative lack of stratification between the surface and bottom salinities observed in the data (Figure 58). However, the model does not capture the difference between surface and bottom temperatures, about 2-3 °C, observed in the data, except between RM 60-90. While the model does reproduce the observed August Chl-a well, as well as the surface observations of DO, it does not capture the observed bottom water DO concentrations, particularly between MP 20-60. The model over-estimates the observed DO near MP 20 by almost 4 mg/L and the remaining bottom water DO data between MP 30 and 50 by 2 mg/L and MP 56 by 1 mg/L. The hydrodynamic model's inability to capture the density difference in temperature may contribute to the failure to capture the bottom water DO to some degree.

The model does a better job in reproducing the observed August DIP and particulate phosphorus data then it did in May, but it still tends to over-estimate the surface DIP slightly and under-estimate the surface particulate phosphorus slightly. The model also under-estimates the observed surface DOP between MP 20-70, but compares well to the observed TP data, except for over-estimating TP in the lower East River (MP 0-10).

The model under-estimates both the observed DOC and POC throughout most of the Sound (Figure 59). The mismatch between model and data for DOC is greatest in the lower East River and less so through the remaining portion of the Sound. It is interesting to note, however, that there is quite a bit of variation in the DOC data, with some stations reporting surface DOC being greater than bottom water DOC and some stations reporting the reverse. With respect to the POC model calibration, the model under-estimates the observed surface POC data by 1-1.5 mg C/L in the East River and western Sound and by 0.3-0.6 mg C/L throughout the rest of the Sound. The model compares favorably to the observed BOD data, suggesting perhaps that SOD may be contributing the most to the bottom water oxygen losses. The model under-estimates the observed biogenic silica (BSi), suggesting that there was a greater population of diatoms present in the Sound than the model computed. There is also some evidence of this in the modeled versus observed dissolved silica, where the model tends to over-estimate the observed dissolved silica data.

There are not sufficient surface TON, NH_4 and NO_2+NO_3 data with which to make model versus data comparisons for August. However, comparing the available surface data for NH_4 and NO_2+NO_3 suggest that the model is biased slightly high, about a factor of 2 to 3, but the computed concentrations are approaching growth limiting conditions. The model also over-estimates TN in the lower East River, but compares favorably to the remaining Sound TN data.

Figures 60 through 64 provide model versus data comparisons for the available water quality variables for May 1999, while Figures 65 and 66 provide model computations of primary production, community respiration and nutrient fluxes. The model appears to provide a reasonable reproduction of the surface to bottom salinity gradient observed for May 1999 and in particular the greater surface to bottom salinity difference observed in the eastern Sound, but the model does over-estimate the observed salinity in the central and western Sound by about 1ppt (Figure 60). Interestingly, there is a greater surface to bottom temperature gradient in the western Sound than is observed in the central and eastern Sound and the model reproduces this feature in the data, although the model does appear to compute too large a surface to bottom temperature gradient between MP 40-56. The model over-estimates the observed Chl-a data for May 1999, computing surface Chl-a concentrations of 5-10 ug/L between RM 0-130, versus less than 2 ug/L in the observed data. However, as was noted in the data review section, these Chl-a data should be viewed with some question as to their validity. The model compares favorably to the surface observed DO data, which are at or slightly above saturation, except for MP 20-30, where the model under-estimates the observed DO data by 2-4 mg/L. The model also tends to under-estimate the bottom water DO data by a mg/L in the western Sound.

With the exception of the western Sound, the model computation of surface DIP compares well to the observed surface data (Figure 61). In the western Sound the model over-estimates the surface DIP data by 0.01-0.02 mg P/L, except for one unusually low datum at MP 27.5, where the model over-estimates the data by about 0.035 mg P/L. Again both the model and data are well above the Michaelis-Menton constant for phosphorus limitation of phytoplankton growth. The model tends to under-estimate the observed DOP data in the western Sound (MP 20-50), but compares favorably to the observed data in the central and eastern Sound. The model is biased high with respect to the observed total particulate phosphorus (TPP) data. The model is also biased slightly high to the observed TP data between MP 20-95, but compares well to the remaining TP data.

The model significantly under-estimates the particulate Si in the western Sound, which shows an increasing concentration of particulate Si between MP 60 to MP 20, suggesting that the model is under-estimating the spring diatom bloom (Figure 62). However, the model does appear to be computing some level of a bloom as the model shows a decrease in dissolved silica between MP 70 to MP 20, and in fact is computing that Si is limiting phytoplankton growth between MP 20 and about MP 40. However, with the exception of a single surface datum at MP 27.5, the data do not suggest that silica is limiting phytoplankton growth during this May 1999 period. The model under-estimates the total silica data by a factor of about in between MP 20-55.

With the exception of a couple of data points in the western Sound (MP 20-30 and MP 55), the model compares well to the observed TON data (Figure 63). The model calibration to the observed ammonium and nitrite+nitrate data is mixed. Between MP 20-30, the model over-estimates both NH_4 and NO_2+NO_3 and fails to reproduce the observations that suggest that DIN is limiting phytoplankton growth in this region. The model compares more favorably to the observed surface data between MP 30-70, which although the model under-estimates the NH_4 data, does suggest that DIN is limiting phytoplankton production in this region. The model then under-estimates surface NH_4 and NO_2+NO_3, suggesting DIN limitation of phytoplankton growth, which is not supported by the data. The model does, however, compare favorably to the observed TN data in May.

The model provides a good comparison to the observed DOC data (Figure 64). The model also compares favorably to the observed POC data except for the far western Sound, where POC concentrations increase to about 2 mg C/L between MP 20-30 from less than 0.7 mg C/L from MP 37 and eastward.

Model computations of GPP and NPP for May 1999 (Figure 65) are about 10-20 percent lower than were computed by the model for the same time period of May 1989 (Figure 47). Similarly, computed community respiration is also lower in May 1999 as compared to May 1989. Computations of SOD, however, between the two periods do not show much difference. Fluxes of PO_4 and NH_4 for the two periods are virtually the same (Figure 66), while there are differences between the fluxes of NO_3 with May 1999 showing a zero flux rate throughout the entire Sound versus the May 1989 period showing a flow of NO_3 into the sediment bed from the overlying water column. The computed fluxes of Si are quite similar for both May periods, except for the lower East River, where the May 1999 fluxes are higher than the computed May 1989 fluxes.

For August 1999, the model provides a very good comparison to the observed salinity data, comparing well to both the spatial and vertical trends in the data (Figure 67). The model tends to over-estimate surface and bottom temperature data in western and portions of the central Sound, but tends to approximate the vertical gradient in the observed temperature data. The model compares better to the observed data between MP 80-95 and then reverts back to over-estimating the observed data by 1-2 degrees in the eastern Sound. The model also severely over-estimates the observed Chl-a data, but has been noted earlier, the observed data have been called into question. The model does not compare well to the observed bottom water DO data in the western Sound. The model computed averaged bottom water DO over-estimates the observed bottom water averaged data by about 2 mg/L between MP 20 and 30 and then by about 1 mg/L at MP 32, before comparing well to the bottom water averaged DO data between MP 35-90. The model then under-estimates the bottom water DO data in the eastern portion of the Sound. With respect to the minimum computed DO data versus the minimum observed DO data, the model still over-estimates the data, but the discrepancy is smaller than the averaged data, i.e., 1 mg/L versus 2 mg/L. The model also tends to over-estimate the surface averaged DO data by 1.5-2.5 mg/L in the western Sound, but compares more favorably to the data in the central and eastern Sound.

The model provides a favorable comparison to the bottom water DIP throughout the Sound, but tends to over-estimate the observed DIP data in the western and central Sound (MP 20-85). Both the model and data again do not show any evidence of phosphorus limitation (Figure 68). With the exception of a few data points the model compares favorably to the observed DOP data, but over-estimates the TPP data, particularly in the western Sound. The model versus observed comparisons for TP are, in general, quite good.

The model comparison to the various forms of silica tends to be quite poor (Figure 69), with the model severely under-estimating the observed data. However, it is interesting to note that the reported August 1999 Si data, including particulate, dissolved and total Si, are all about a factor of three times greater than were observed in August of 1989. This would be the expected ratio if the 1999 silica data were being reported at SiO_2 versus as Si. However, in inquiring if the 1999 (through 2002) data were reported as SiO_2 we were informed that this was not the case. However, it is difficult, then, to explain why water column silica concentrations would have increased about 3-fold from the 1988-1989 and 1994-1995 periods.

The model versus data comparisons for TON, NH_4, NO_2+NO_3 and TN are all quite favorable (Figure 70). The model appears to capture the observed spatial and vertical gradients in TON, NH_4 and NO_2+NO_3, including the strong DIN nitrogen limitation between MP 20 and 130, as observed in the data.

The model tends to over-estimate the vertical gradient in DOC as compared to the data (Figure 71), but tends to compare favorably to the surface data between MP 20 and 70, before under-estimating DOC in the eastern Sound. The model also tends to over-estimate the surface POC by 10-20 percent in the western Sound, but does tend to compare favorably to the observed POC data in the central and eastern Sound.

While model computations of GPP and NPP in August are quite similar to those computed in August 1989, respiration and SOD in August 1999 are about 10-20 percent lower than computed in August 1989 (Figure 72 versus Figure 54, respectively). A similar reduction can be observed in computed values for J_{PO4}, J_{NH4} and J_{Si} in August 1999 versus August 1989 (Figure 73 versus Figure 55, respectively).

With the exception of a few surface water values (MP 56, 92 and 103), the model compares favorably to the observed May 2000 salinity data (Figure 74). The model, however, compares less favorably to the observed temperature data, over-estimating the surface data by 1-3 °C throughout the Sound and over-estimating the vertical stratification in the observed temperature data between MP 45-70. The model over-estimates the Chl-a data, but again the observed data during this period are being questioned. Similar to what was observed in May 1999, the model under-estimates the observed surface DO in the western Sound, which is about 2.5-3 mg/L above saturation, again suggesting that an algal bloom was occurring at that time. The model also tends to under-estimate the bottom water DO throughout the Sound by 0.5-1 mg/L.

The model over-estimates the observed DIP data, particularly in the western Sound, where two of the data (MP 20-30) suggest that phosphorus limitation may be occurring; something that the model does not indicate at all (Figure 75). The model also over-estimates DIP between MP 80-95, another region where the concentrations of DIP are approaching growth limiting levels. Again the model is well above growth limiting concentrations of DIP. The model also under-estimates DOP throughout most of the Sound, although there is quite a bit of spatial variability between MP 20-80 in the observed data. Interestingly, the model compares favorably to the observed surface TPP data, but does under-estimate the bottom water data between MP 20-30.

The model also compares favorably to the observed TP data, except between MP 20-30, where it does not capture the difference between observed surface and bottom TP data, but instead the model surface and bottom goes through the middle of the surface and bottom data.

As was observed with the August 1999 model versus data comparison, the May 2000 model computations of Si, severely under-estimates the observed forms of silica (Figure 76). Again, it is interesting to note that the May 2000 data are about a factor of 2-3 times greater than the silica data observed in May 1989.

The model under-estimates the observed TON data between MP 20 and 35, but compare favorably to the observed data in the remaining regions of the Sound (Figure 77). The data suggest that the spring bloom is being limited by nitrogen throughout the Sound. The model also indicates that nitrogen is limiting, but only from MP 30 and eastward. The model over-estimates the observed ammonium and nitrite+nitrate data between MP 20-30. This may be due to the fact that the model is predicting Si to be limiting in this portion of the Sound (Figure 76).

The model computations of DOC are about 50% lower than the observed data between MP 20 and MP 90 (Figure 78) and the model also under-estimates the POC data between MP 20 and MP 55, with the greatest discrepancy observed in the western Sound (MP 20-40).

Maximum computed GPP and NPP (about MP 15) are about a third lower in May 2000 (Figure 79) as compared to May 1989 (Figure 47). Computed respiration is also about 20 percent lower in May 2000 as compared to May 1989. Model computations of SOD differ only by about 10 percent between the two May periods, with May 2000 being lower than May 1989. Model computed fluxes of PO_4 and NH_4 do not differ appreciably between the two May periods (Figure 80 versus Figure 48), while fluxes of silica are higher in the lower East River in May 2000 as compared to May 1989.

It appears as if the hydrodynamic model did not perform well in the central and western portions of the Sound in August 2000. The model over-estimates surface and bottom salinity in the central and western Sound between MP 20 and approximately 82 by 1-1.5 ppt, although it does get the surface to bottom gradient approximately correct (Figure 81). The model does over-estimate the surface to bottom gradient in observed temperatures by 2-3 degrees between MP 20-

80, although it does compare favorably to the surface data. Unlike the other monthly low observations of Chl-a data, the August 2000 surface maxima approach 20 ug/L in the western Sound and the model captures those data well. However, the model under-estimates the bottom water Chl-a data in the western Sound. With respect to bottom water DO, for the month of August 2000, this is the exception to the general model calibration results wherein the model under-estimates the observed data, particularly in the western Sound. The data show bottom averaged DO data to be a little less than 6 mg/L and then decline to about 5 mg/L near MP 42. The model, however, computes bottom averaged DO concentrations to be about 3.5 mg/L near MP 20 and then increase slightly to about 4 mg/L by MP 42. The model also under-estimates observed bottom water DO data from MP 90 to 130. A possible factor contributing to the under-estimation of the DO data in the western Sound may be the temperature stratification computed by the model in the western Sound.

The model follows the general spatial trend observed in the DIP data (Figure 82), but does not reproduce some of the spatial variability observed in the surface data, although it does provide a slightly better calibration to the bottom water data in the central and western Sound. The model over-estimates the bottom water DIP in the eastern Sound. Neither model nor data give any suggestion of phosphorus being a limiting nutrient, except in the far eastern Sound. The model tends to over-estimate the DOP in the Sound although there are stations for which the model under-estimates the observed data. The model also tends to over-estimate the vertical gradient observed in the TPP data and over-estimates the surface data, while under-estimating the bottom TPP data. Model versus data comparisons to TP are, in general, consistent with the observed data. As has been observed in the 1999 and post-data, the model significantly under-estimates the observed forms of silica (Figure 83).

The model tends to compute the small spatial gradient from west to east observed in the TON data (Figure 84), but does tend to over-estimate the vertical gradient observed in the data. The model does compute the very strong nitrogen limitation observed in the data and although the model computed NH_4 concentrations are right at the Michaelis-Menton values, the observed data are well below the Michaelis- Menton value. Interestingly, the bottom water ammonium data are also well below the Michaelis-Menton coefficient and are at the same concentrations as are the surface data, while the bottom model computations are above limiting conditions. To see ammonium concentrations this low in the bottom of the water column is surprising. Model versus data comparisons of TN are good.

The model under-estimates both the surface and bottom water DOC by about a factor of two (Figure 85). It is interesting to note the sharp spatial gradient (about a factor of 2) observed in the DOC data that occurs around MP 100. The model computation of surface POC also appears to be biased high compared to the observed data, except at MP 27.5, where both the model surface and surface data agree. The model is also biased high to the observed K_e data by about 15-20 percent.

Model computations of August 2000 GPP, NPP, and respiration (Figure 86) are again about 10-15 percent lower than the corresponding period in August 1989 (Figure 54) and are quite similar to the levels computed in August 1999 (Figure 72). Similar observations can be made for SOD (Figure 86) and the other nutrient fluxes (Figure 87).

We did not have May 2001 data with which to make model versus data comparisons, so instead we chose April 2001 to compare the model to the observed data. In general the model tended to over-estimate the observed vertical gradient in salinity in most of the Sound, the exceptions being MP 48 and MP 102 (Figure 88). The model tends to provide a better comparison to the observed surface data as opposed to the bottom data, in which case the model tends to over-estimate the data. The model also over-estimates the observed surface and bottom temperature data in April by about 2.5 degrees. The model also tends to over predict the observed vertical gradient in temperature by about 2 degrees. The model also over-estimates the surface Chl-a data observed in April. This over prediction may be attributed to in part the over-estimation of the vertical stratification (salinity and temperature), which may provide a greater residence time in the surface layer and thus allow the algae to grow. It may also be attributed to, in part, the questionable Chl-a data themselves. The model under-estimates the observed DO data, both surface and bottom water averaged data and also over-estimates the vertical gradient in the observed data. This lack of calibration may be in part due to the hydrodynamic model's over-estimation of the salinity and temperature gradient in the observed data.

The model under-estimates the surface DIP throughout most of the Sound, except in the far western Sound, MP 20-30 (Figure 89). The model over-estimates the observed DOP data except for a few random surface data in the vicinity of MPs 42, 67, and 82. The model tends to over-estimate the surface TPP data, but compares favorably to the bottom water TPP data. The model also compares favorably to the observed TP data. As has been observed with other silica data measured in 1999 and 2000, the model under-estimates the observed forms of Si (Figure 90). The model also computes a strong nutrient limitation by dissolved silica between MP 17.5-30, which is not evident in the observed data.

The model provides a reasonable calibration to the observed TON data for April 2001 (Figure 91). The model only slightly over-estimates the observed surface and bottom NH_4 data between MP 20-35 and also under-estimates the observed bottom water NH_4 throughout the rest of the Sound. The model approximates the observed surface NH_4 at MP 30-50, under-estimates the data near MP 56, approximates the surface data at MP 67, under-estimates the data at MP 82 and then approximately reproduces the observed data eastward of MP 90. The model severely under-estimates the observed NO_2+NO_3 throughout the Sound in April 2001. Interestingly, the model computes a much stronger limitation to DIN than is observed in the data. The model also over-estimates the TN data slightly throughout the Sound.

With the exception of some observed surface and bottom DOC data between MP 25-40, the model compares favorably to the observed DOC data (Figure 92). The model comparison to the observed surface POC data looks to be reasonable, although there are some stations where the model slightly under-estimates the observed data and there are some stations where the model slightly over-estimates the observed data. The model is also biased high relative to the observed light attenuation (K_e) data. Spatial profiles for GPP, NPP, respiration, SOD and nutrient fluxes are presented in Figures 93 and 94, although there are no data with which to make model-data comparisons against.

Model versus data comparisons for the August 2001 data are shown in Figures 94-101. Model computations of salinity compare well against the observed spatial and vertical gradients in salinity (Figure 95). The model also roughly captures the vertical gradient observed in temperature in the western most Sound (MP 27.5-42), but then over-estimates the vertical gradient between MP 45-70, before capturing the vertical gradient in the eastern portion of the Sound. The model does tend to over-estimate the surface and bottom temperatures in the western Sound by 1-2 degrees. Except for the high datum observed near MP 21, the model tends to over-estimate the surface Chl-a data, particularly in the eastern Sound. The model also over-estimates the bottom averaged DO data in the western Sound (MP 20-55). With the exception of MP 21, where the difference in the averaged model computation and the averaged datum is about 2 mg/L, the difference between the bottom averaged model computation and the bottom averaged data is about 1 mg/L. The difference between the minimum values computed by the model and the observed data is generally less than 1 mg/L between MP 20-70. In the eastern Sound the model tends to under-estimate the bottom DO data by 1-2 mg/L.

The model tends to under-estimate the surface and bottom DIP data between MP 20 and MP 55 and slightly over-estimates the surface DIP data in the rest of the Sound, while comparing favorably to the bottom water DIP (Figure 96). In general, the model under-estimates the DOP data throughout the Sound. In general, the model over-estimates the surface TPP data between MP 20-42 and slightly under-estimates the bottom water TPP data in that same region. With the exception of a few data points the model calibration to the observed TP data is favorable. As with other silica data, the model substantially under-estimates the observed silica data for particulate, dissolved and total silica in August 2001 (Figure 97).

With the exception of a high surface value near MP 82, the model compares favorably to the August 2001 TON data (Figure 98). The model also compares reasonably well to the surface and bottom ammonium data in the western Sound (MP 20-70), although it does not reproduce some of the very low (0.001-0.002 mg N/L) concentrations of observed ammonium. The model over-estimates the observed surface and bottom ammonium data east of MP 80. The model provides a favorable comparison to the observed surface NO_2+NO_3 data between MP 20 and MP 95, but over-estimates the surface NO_2+NO_3 east of that point. The model also over-estimates the bottom water NO_2+NO_3 throughout the Sound. The model and data both show strong DIN limitation during August of 2001. The model provides a favorable comparison to the TN data, except in the far eastern Sound (MP 115-130), where it begins to over-estimate the observed data. The model-data comparisons for DOC and POC are, in general, good (Figure 99). With the exception of over-estimating DOC in the far eastern Sound (MP 115-130) and a high value of

POC near MP 27.5, the model captures most of the spatial and vertical gradients observed in the DOC and POC data. The model also tends to be high relative to the observed light attenuation (K_e) data.

Spatial profiles of GPP, NPP, respiration, SOD (Figure 100) and the nutrient fluxes (Figure 101) for August 2001 are virtually the same as were computed for August 1999 (Figures 72 and 73) and August 2000 (Figures 86 and 87).

Model computations of salinity compare well to the observed data for May 2002 (Figure 102) with the model computations of surface and bottom salinity differing less than a few tenths of a ppt from the observed data. The model is biased slightly high (about 1 °C) relative to the surface and bottom temperature data, but appears to do a good job of getting the vertical stratification correct. The model computations of surface and bottom Chl-a compare well to the observed data between MP 20 and 70, but then over-estimate the observed Chl-a data eastward of MP 80. The model tends to over-estimate the surface DO data by a few tenths of a mg/L and tends to under-estimate the bottom data by as much as 1 mg/L between MP 30 and 60.

The model comparison to the observed DIP data is good between MP 20-70, but then slightly under-estimates the observed data between MP 80 and 95 (Figure 103). There is quite a bit of variability in the DOP data and the model tends to capture the lower end of the observed data, but misses the higher values observed in the data. In general, the model comparison to the TPP data and the TP are good, with the exception of a few data point where both the DOP and TPP data were very high.

Again, in May of 2002, we see the model significantly under-estimate the various forms of the silica data (Figure 104). The model does indicate silica limiting conditions to occur between MP 15 and about MP 37. The model compares favorably to the observed TON data (Figure 105). The model also compares favorably to the observed NH_4 and NO_2+NO_3 data in the western Sound, approximately capturing the spatial gradient observed in the DIN data between MP 20 and 40, wherein DIN goes from non-limiting to limiting conditions for phytoplankton growth. The model also captures in part the reversal in the vertical gradients in NO_2+NO_3 between MP 20 and 50, where between MP 20-45 the surface waters are higher in NO_2+NO_3 as compared to the bottom waters and then between MP 50-80, the signal is reversed with the bottom waters being higher in NO_2+NO_3 as compared to the surface waters. The model does, however, over-estimate both NH_4 and NO_2+NO_3 in the eastern Sound. Model-data comparisons for TN are good for May 2002.

The model compares favorably to the observed DOC data in the western Sound and portions of the central Sound, but under-estimate the observed data eastward of MP 80 (Figure 106). The model also compares well to the observed surface POC in most of the Sound, but does tend to under-estimate the bottom water POC data throughout most of the Sound. The model also compares fairly well to the observed K_e data in the western Sound, but over-estimates the K_e data by 0.2-0.5 m^{-1} in the central and eastern Sound. Spatial profiles of GPP, NPP, respiration, SOD and nutrient fluxes are presented in Figures 107 and 108.

The spatial and vertical profiles of salinity compare quite well for August 2002 (Figure 109). The comparisons between model and data for temperature are not so favorable though. The model over-estimates the degree of vertical temperature stratification by about 2-2.5 degrees between MP 20 and 70. The model also tends to over-estimate the degree of temperature stratification in the eastern Sound as well. The model also compares favorably to the surface and bottom Chl-a throughout most of the Sound, the exception being in the far eastern Sound, where the model over-estimates the observed Chl-a data by 5 ug/L. The model does not compare well to the observed DO data in the western Sound, over-estimating both the surface and bottom DO by 1.5-2.5 mg/L between MP 20-40, before comparing more favorably between MP 40-105. It is interesting to note, that despite relatively high Chl-a data observed between MP 20-40, the surface data are well below DO saturation.

The model provides a fair comparison to the observed DIP data (Figure 110), except in the far eastern Sound, where the model over-estimates both the surface and bottom data. The model computations of DOP tends to be biased low to the observed data, while model computations of TPP tend to be biased high to the observed surface data and biased low to the observed bottom data. The model compares well to the observed TP data and the west to east gradient observed in the data. Model versus silica (Figure 111) is consistent with the results from other months, i.e., the model is low compared to the observed data.

The model appears to biased low relative to the observed measurements of TON (Figure 112). The model does indicate DIN is limiting algal growth in August 2002 as supported by the data. The model does not match some of the very low (0.001-0.004 mg N/L) surface NH4 data, nor does it reproduce the very low bottom water concentrations of NH4 observed at a large number of the monitoring stations. The model compares more favorably to the observed NO_2+NO_3 data, approximating the surface to bottom gradient observed in the data between MP 20 and 70. The model tends, however, to over-estimate surface and bottom NO_2+NO_3 data eastward of MP 80. The model also under-estimates the observed surface TN data in August of 2002.

The model also under-estimates the observed DOC data, particularly in the bottom layer in August 2002 (Figure 113), but provides a fair comparison to the surface POC in most of the Sound but does under-estimate some high values of POC observed between MP 65-85. The model is also biased low as compared to the bottom water POC in the western Sound, but compares favorably to bottom POC data in the rest of the Sound. The model is also biased high by 0.2-0.4 m^{-1} to the observed K_e data throughout the Sound.

Spatial profiles of modeled GPP, NPP and respiration are presented in Figure 114. Similar to other August periods, the maximum values of GPP and NPP are computed to occur near MP 20. The peaks in August of 2002, interestingly, tend to be greater than the computed August peaks for 1999-2001. Interestingly, however, the spatial peak in SOD is about 20 percent lower in August 2002 as compared to the peak SOD computed in 1989. Spatial profiles of the nutrient fluxes are presented in Figure 115 and when compared to those computed in August 1989 (Figure 55), indicate that the fluxes of PO_4 and Si are lower in 2002, but that the fluxes of NH_4 are about the same.

CONCLUSIONS AND RECOMMENDATIONS

The model does capture some of the features of water quality in Long Island Sound. These include:

- The west to east gradients in total nitrogen, NH_4, NO_2+NO_3 and TP and PO_4,
- That nitrogen is the limiting nutrient in the Sound,
- That algal biomass, primary production and respiration are higher in the western Sound as compared to the central and eastern Sound,
- That bottom water oxygen is lower in the western Sound as compared to the central and eastern Sound.

However, the model does not fully capture the extent of hypoxia that is observed in the bottom water DO data. Potential contributing factors to this problem, particularly for the water years 1999-2002, may be attributed to mis-calibration of the hydrodynamic model. The SWEM model was extended to include 1999-2002 for the CARP project and the calibration effort tended to focus on NY/NJ Harbor, the Hudson River and the NJ tributaries. Future efforts should be focused on improving the hydrodynamic model calibration in Long Island Sound and the East River. In particular, although we did not fully explore whether additional spatial resolution could benefit the hydrodynamic and water quality models, this should be explored. One could also take the existing SWEM model and increase the vertical resolution by a factor of two (20-sigma layers rather than 10-sigma layers) to see if this could improve the water quality model. We have had some internal discussions as to whether, even though the hydrodynamic model seems to be computing the correct scale for the vertical diffusivities, perhaps, the 10-sigma layer resolution may have some artificial numerical mixing that contributes too much vertical exchange of DO. In addition, the exchange between the other portions of the Harbor, the East River and the Sound should be re-examined. One reason to do this is very high SOD that is created by the model parameterization of benthic filter feeding in the East River. However, DO is still over-estimated in the East River by about 1 mg/L on average. Other contributing factors to the mis-calibration to the DO data, may be related to choices in algal growth and respiration rates and grazing rates, both for the winter and summer algal groups. There are suggestions from the NH_4 and NO_2+NO_3 that additional algal growth could be supported (the model does not fully capture the nutrient limiting conditions that occur in the western Sound). However, the model does compare favorably to GPP, NPP and respiration and BOD data in that region, suggesting that it may be difficult to push those model coefficients too much further.

In order to help assess its continued utility as a management tool, the model as presently calibrated should be exercised with various levels of nutrient reduction to evaluate the effect of nutrient reduction on phytoplankton biomass as well as the response of DO and hypoxia to potential management scenarios. This would help inform the LISS program and Management Committee as to the utility of the SWEM model for use as a management tool to address hypoxia in Long Island Sound.

OBJECTIVE 4: ASSESS SENSITIVITY TO METEOROLOGY
TASK 4A: REPEAT 6 YEARS WITH REVISED SWEM

The results of the SWEM re-calibration have been presented above. In this task, we will be taking a closer look at the ability of the SWEM model to respond to changes in meteorology. Water quality, particularly chlorophyll and dissolved oxygen, responds to both meteorological conditions, i.e., freshwater inflows, wind-mixing, sunlight, air-temperature, and tides, and nutrient inputs. While the SWEM hydrodynamic model is being driven by year-to-year variability in the meteorological conditions, the emphasis of the model extension to include water years 1999-2002 during its application in the CARP project was on the NY-NJ Harbor portion of the domain and not Long Island Sound. It appears from some of the hydrodynamic model calibration results presented in the water quality calibration section of this Appendix that additional refinements to the hydrodynamic model calibration in the Sound are warranted. However, despite this limitation, an analysis of the year-to-year responses to the applied meteorology is presented here.

The first evaluation considers the duration and spatial extend of hypoxia (DO $\leq$ 3.0 mg/L) computed by the model. In the case of the spatial duration of hypoxia the model computed a variable known as DO-area-days. The definition of DO area-days is as follows:

- For each model segment or cell a running summation is computed, wherein, if the computed DO is found to be less than or equal to 3 mg/L, the area of that model segment is multiplied by the time-step of the model (i.e., SWEM uses a 10 minute time-step) and is added to the running total for that model segment. At the end of the simulation for that year, then the running totals for each segment are added together to get the total DO area-days for the entire Long Island Sound domain.

Table 4A-1 presents the DO area-days and the number of days that hypoxia was found to occur within the Sound. The number of days reflects only the period of time that hypoxia was actually computed to occur within the model. For example, if hypoxia was computed to occur only in one model cell in the Sound from midnight and into early/mid-morning (say 10 AM) and then the DO rose above 3 mg/L for the remaining portion of the sunlight period of the day (say 7 PM) before falling below 3 mg/L again, then the duration of hypoxia for that day is only 15 hours and not 24 hours. This computation differs a little from the USEPA LISS reporting of the "duration" of hypoxia, which is determined as the number of days between the first and last observations of hypoxia within the Sound. Therefore, it might be expected that the reported duration may be greater than the number of days of hypoxia computed by the model, since duration does not know about re-ventilation events that may occur during a reporting season due to limitations in the frequency of sampling. Re-ventilation has been shown to be an important process in the Sound and is driven by meteorological events (O'Donnell et al., 2008).

Table 4A-1. Model Computations of Hypoxic DO Area-Days and Days of Hypoxia vs. Hypoxic Duration as Reported by LISS

Water Year	DO Area-Days (km^2-days)	Number of Days	LISS/CTDEEP Reported Duration (days)
1988-1989	569	20	63
1994-1995	642	12	35
1998-1999	2,060	17	51
1999-2000	5,530	31	35
2000-2001	6,930	36	66
2001-2002	1,480	24	65

It is interesting to note, that, with the exception of water year 1999-2000, SWEM computes between a factor of two to three smaller number of days of hypoxia as compared the duration estimates provided by the LISS. Part of this discrepancy may be due to differences in the two ways that the computations are determined, as mentioned above, but a more likely factor is due to limitations in the ability of the SWEM model to properly reproduce all of the factors that are contributing to hypoxia in the Sound, i.e., limitations of the existing SWEM model. It is also interesting to note the fact that the model computes increases in the DO area-days computations between the summers of 1989/1995 and 1999 though 2001, before a reduction in the DO area-days is computed for the summer of 2002.

In order to get further insight in the effects of meteorology on bottom water hypoxia, we have also prepared a series of figures that show "contour" plots of the 10-day averaged bottom water DO and the corresponding 10-day minimum bottom water DO computed by the model. The plots represent 10-day averaged results from the beginning of July, through August and ending in mid-September. The results presented on these plots are consistent with the values of DO area-days presented in Table 4A-1, but the 10-day to 10-day differences within and between years are quite spatially variable. These contour plots are included with the plots attached to the document.

OBJECTIVE 4: ASSESS SENSITIVITY TO METEOROLOGY
TASK 4B: NUTRIENT SCENARIO ASSESSMENT

Due to time and budget limitations introduced into the project that resulted from efforts to improve the calibration and validation of the water quality model, we only performed one nutrient scenario assessment run. For this run, we reduced point source and non-point source (CSO and SWO) nitrogen loadings to the system by 58.5% (consistent with the LIS nitrogen TMDL). Post-processing the model computations result in the following reductions in bottom water hypoxia DO area-days and the number of days of hypoxia (Table 4B-1).

Table 4B-1. Comparison of Changes in Hypoxia Between the Baseline and the Nutrient Reduction Scenario using the revised SWEM model.

Water Year	Baseline		Nutrient Reduction Scenario	
	DO Area-Days (km^2-days)	Number of Days	DO Area-Days (km^2-days)	Number of Days
1988-1989	569	20	51	4
1994-1995	642	12	<1	<1
1998-1999	2,060	17	53	1
1999-2000	5,530	31	175	3
2000-2001	6,930	36	2,456	13
2001-2002	1,480	24	77	6

We have also prepared a series of "contour" plots (similar to those presented for Task 4A) that show the changes, increases and decreases, in bottom water DO that result from the reduction in TN being delivered to the Sound. In general, the model computes that average and minimum bottom water DO increases as a result of the reduction in nitrogen loading to the system, but for those shallow water segments (i.e., embayments) of the Sound, bottom water DO tends to decrease. This is due to the fact that since those segments are so shallow, there was no surface to bottom stratification, i.e., surface and bottom water DO concentrations were about the same and since primary production was reduced in the embayments due to nutrient limitation, the DO also decreased, therefore, leading to a reduction in surface and bottom water DO. These plots are attached to this document.

During the review of this document a question was asked as to how the results of this nutrient reduction scenario performed using the revised version of SWEM compares to those computed by the previous version of SWEM. Unfortunately, given time and lack of available budget we were not able to perform a direct comparison to answer this question, as the current analysis looked at DO Area-Days based on 3 mg/L, while previous analyses looked at days of non-attainment using both the State of Connecticut and State of New York acute DO standards, which were never less than 3.5 mg/L in the open waters of the Sound and never less than 3.0 mg/L for the State of Connecticut and State of New York standards, respectively. However, we believe the following analysis provides a qualitative assessment that, in part, responds to the question.

Table 4B-2 presents a summary of the results of days of non-attainment with the DO water quality standards for the States of CT and NY based on the previous version of SWEM. The baseline period was 1988/1989 and the table shows the days of non-attainment for the baseline condition and the Phase IV TMDL Nitrogen TMDL for each Response Region of the Sound as well as the percent reduction in the days of non-attainment for each of the state standards.

Table 4B-2. Comparison of Days of Non-Attainment of DO Water Quality Standards for the States of CT and NY for the Baseline and LIS TMDL Nitrogen TMDL using the previous version of SWEM.

	Response Region Average Non-Attainment Duration										
RUN	**Standard**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Baseline	CT Acute	---	49.8	37.0	24.8	26.3	5.4	5.58	0.52	---	0.0
	NY Acute	3.6	24.6	21.0	11.5	9.0	0.8	0.1	0.0	0.0	0.0
TMDL	CT Acute	---	13.9	9.9	3.4	6.6	0.0	0.0	0.0	---	0.0
	NY Acute	0.7	6.9	2.7	0.9	0.6	0.0	0.0	0.0	0.0	0.0
	% Reduction in Non-Attainment										
	CT Acute	---	72.1	73.2	86.2	74.9	100	100	100	---	---
	NY Acute	80.6	72.0	87.1	92.2	93.3	100	100	---	---	---

As can be seen from the above table, runs using the previous version of SWEM showed that the LIS Nitrogen TMDL reduced the number of days of non-attainment with the DO standards from between 70 and 100% with an average reduction of about 83%. Runs using the revised model show reductions in DO –Area Days of between 65 and 100% with an average of 91% and reductions in the numbers of days of hypoxia of from 64 to 100% with an average reduction of 84% (based on Table 4B-1).

OBJECTIVE 4: ASSESS SENSITIVITY TO METEOROLOGY

TASK 4C: CLIMATE CHANGE SCENARIO

For this subtask, we took a mid-range value of 3 °C for future increases in air temperatures as reported by the USEPA (http://www.epa.gov/climatechange/science/future.html) and the Intergovernmental Panel on Climate Change (https://www.ipcc.ch/publications_and_data/ar4/wg1/en/spmsspm-projections-of.html) and assumed that water temperature would increase by that same amount. Again, given uncertainty as how to specify temperature and salinity changes at the boundaries of SWEM without being able to run a "global" hydrodynamic model to predict these boundaries as a function of climate change, as well as due to budget and time limitations, we did not run the hydrodynamic model, but rather added the 3 °C to the existing temperature fields provided to SWEM by the hydrodynamic model. The increase in temperature was added to all vertical layers in the model. So if the hydrodynamic model computed a surface, mid-depth, and bottom temperature of 20 °C, 18 °C, and 14.5 °C, respectively, after the modification the SWEM model (and the temperature driven biogeochemical rates) would see temperatures of 23°C, 21 °C, and 17.5 °C. The end result is there is no change in the vertical stratification used to drive the water quality model, rather the biogeochemical reaction rates (i.e., phytoplankton growth and respiration, nutrient recycle rates, and sediment diagenesis rates) used in the water quality model, will, in general, increase in response to this temperature perturbation.

The climate warming scenario results in a decrease in the bottom water DO of between 0.8-1.2 mg/L. We post-processed the results for this model scenario in a similar fashion as is presented in Task 4B. Table 4C-1 presents the DO hypoxic area-days and length of hypoxia for the baseline and climate change scenario. As can be seen, the DO area-days increases by a factor of almost 30, while the number of hypoxic days increases by a factor of 3, suggesting a large increase in the area of bottom water hypoxia.

Table 4C-1. Comparison of Changes in Hypoxia Between the Baseline and the Climate Change Scenario

Water Year	Baseline		Climate Change Scenario	
	DO Area-Days (km^2-days)	Number of Days	DO Area-Days (km^2-days)	Number of Days
1988-1989	569	20	15,934	62

Similarly, we have also prepared a series of "contour" plots that show the changes, increases and decreases, in bottom water DO that result from the climate change scenario. In general, the model computes that average and minimum bottom water DO decreases in response to elevated water temperatures. Part of the reduction in DO concentrations results from the fact that the change in water temperature results in a decrease in the solubility of DO in water (i.e., DO saturation). This acts to reduce the amount of surface water DO that can diffuse into the lower layers of the water column. For example, at a salinity of 25 ppt, a change of temperature of 3 °C could result in a reduction of about 0.5 mg/L, while in the summer the reduction in DO saturation would be about 0.25 mg/L. The plots showing the model computations of bottom water DO follow immediately below.

OBJECTIVE 4: ASSESS SENSITIVITY TO METEOROLOGY

TASK 4D: HIGH RESOLUTION ECOM+RCA

The purpose of this task was to evaluate whether grid resolution may be contributing to some of the difficulties SWEM has in computing the proper distribution of bottom water DO in the Sound. We were not able to complete this task, due to the challenges we faced in attempting to calibrate the existing SWEM water quality model. We made more than 50 runs of the water quality model in our efforts to improve the ability of the model to capture the temporal and spatial extend of hypoxia in the Sound. These model runs included testing of both the previous version of the algal growth model, as well as the development and revisions to the SWEM computer code to explore the Laws-Chalup algal growth model and the Jassby-Platt algal growth model. We were able to setup the hydrodynamic model (Figure 4C-1) and get it running over an annual cycle for 1995, but were not able to complete the development of all of the model inputs for the water quality portion of the SWEM model.

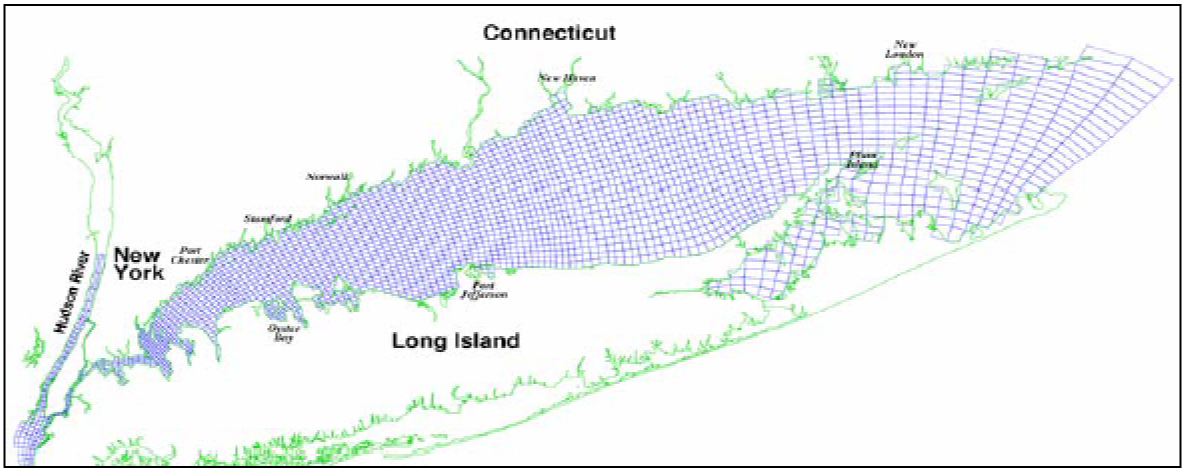

Figure 4D-1. High Resolution Grid of Long Island Sound Domain

OBJECTIVE 5: ADD MECHANISTIC APPROACH TO MODELING SHELLFISH
TASK 5A. IMPLEMENT CHESAPEAKE BAY FILTER FEEDER MODEL (CBFFM)
TASK 5B. TEST REVISIONS AND DOCUMENT CODE CHANGES

Under these two subtasks, we implemented the suspension and deposit filter feeder model developed by HydroQual, Inc. (2000) for the USACE (Waterways Experiment Station, now ERDC) and the USEPA (Chesapeake Bay Program Office). The full details of the model theory and model framework may be found at http://www.chesapeakebay.net/content/publications/cbp_12264.pdf , while a short overview paper of the model was published by Meyers et al. (2000). Essentially the CBFFM calculates suspension feeder biomass based on the assimilation of organic particles filtered from the water column. Biomass can be lost through respiration, predation and mortality due to hypoxia. The suspension feeders can filter both organic and inorganic suspended matter from the water column and either assimilate it or deposit it to the sediment. Respiratory end-products are returned to the water column as components of inorganic nutrient fluxes. Oxygen consumption, as a part of suspension feeder respiration, is incorporated into the sediment flux model's calculation of sediment oxygen demand.

After implementing the CBFFM code into SWEM, we performed numerical checks of the model output against hand-computations to check source/sink terms in the code and to confirm that the CBFFM as linked to the water column and sediment flux model (SFM) maintained mass balances. We also modified the RCA (the computational platform upon which SWEM is based) User's Manual to provide an overview of the model formulation and provide details as to the inputs required to run the CBFFM model in conjunction with SWEM.

In order to fully implement the suspension feeder model, it is required to have sufficient spatial and temporal estimates of bottom suspension feeder biomass with which to calibrate the model. Required data include taxonomic identifications to the lowest possible level with abundance and biomass (as ash-free dry weight (AFDW) for each taxon. Biomass should either be directly measured or estimated based on morphometric (length, width) relationships to individual mass (e.g., Ranasinghe et al., 1996). Sampling should also include observations of water quality (e.g., temperature salinity, and dissolved oxygen) collected contemporaneously with benthic sampling. From these data a relational database can be developed and used to extract and analyze data for dominant individual species and in the reductionist, lumped categories to be employed by the benthic modeling framework. Also, it would be highly desirable to obtain measurements of rate processes associated with suspension feeders, i.e., filtration and food assimilation rates, respiration rates, etc. with which to parameterize the model.

References

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eutrophication model. Water Quality and Ecosystem Modeling, 1, 123-140, 2000.

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Fig 1. CTDEEP monitoring stations

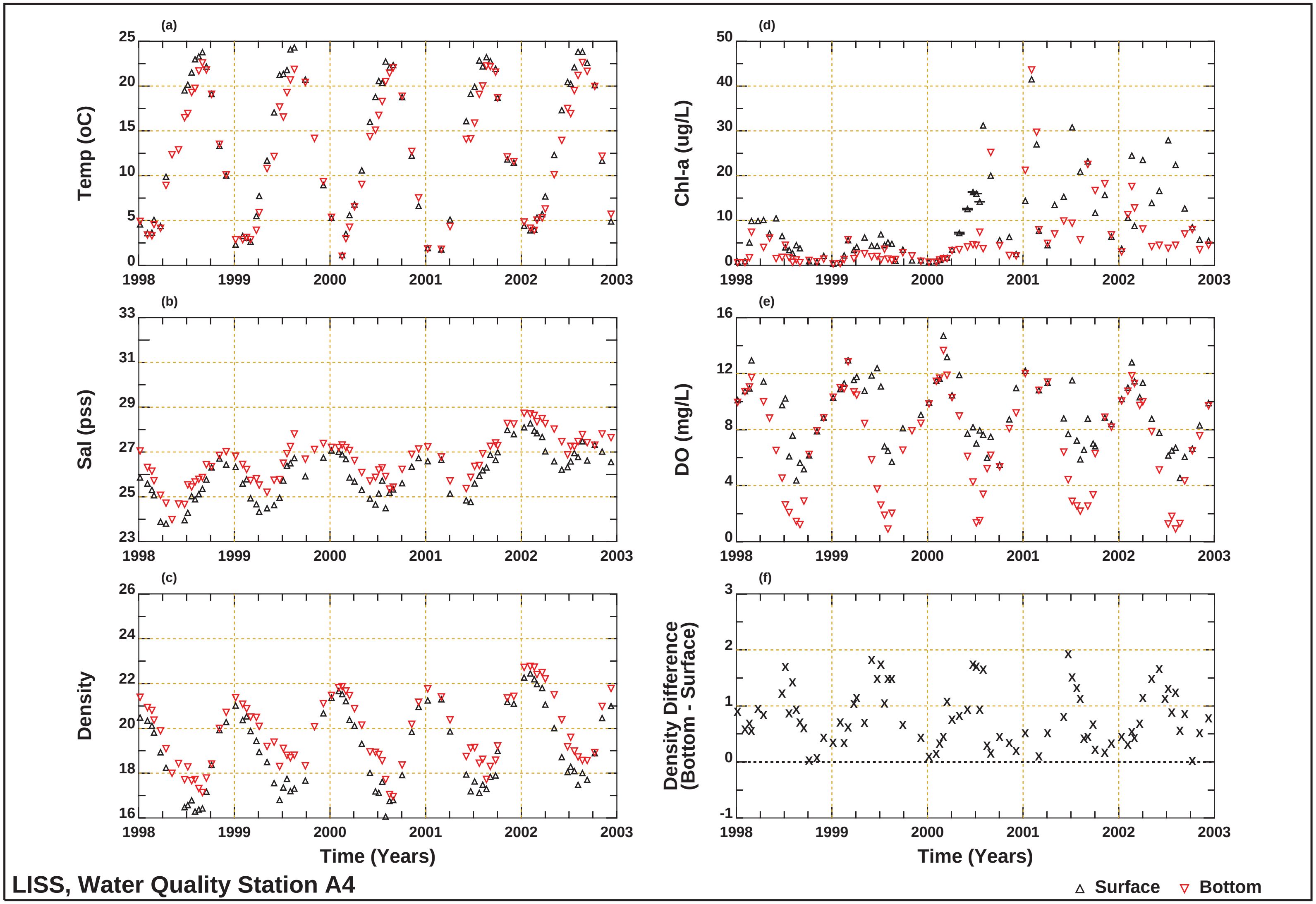


Figure 2. Temperature, salinity, density, phytoplankton chlorophyll, and dissolved oxygen data for station A4

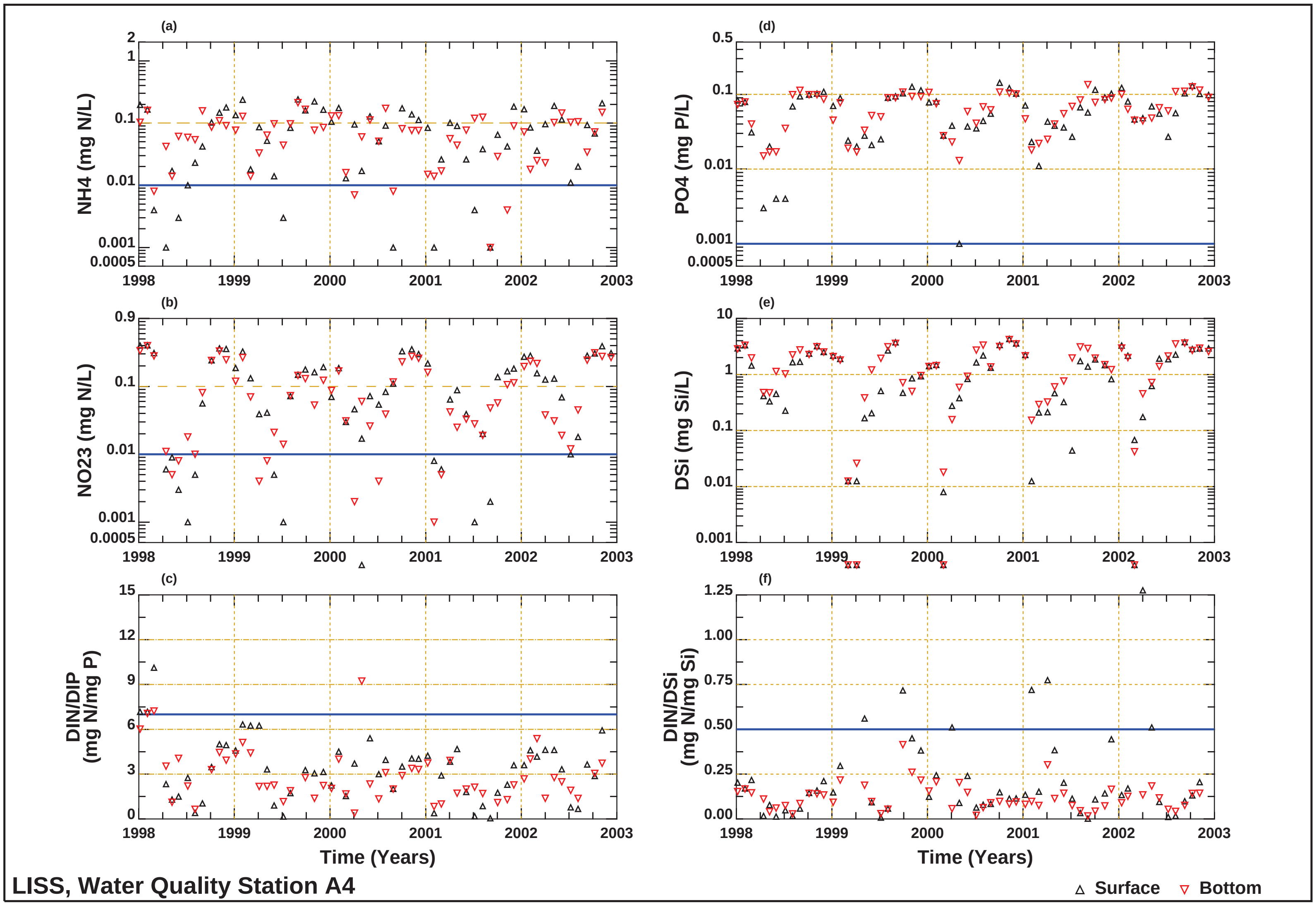


Figure 3. Ammonium, nitirite+nitrate, DIN:DIP, ortho-phosphate, dissolved silica and DIN:Si data for station A4

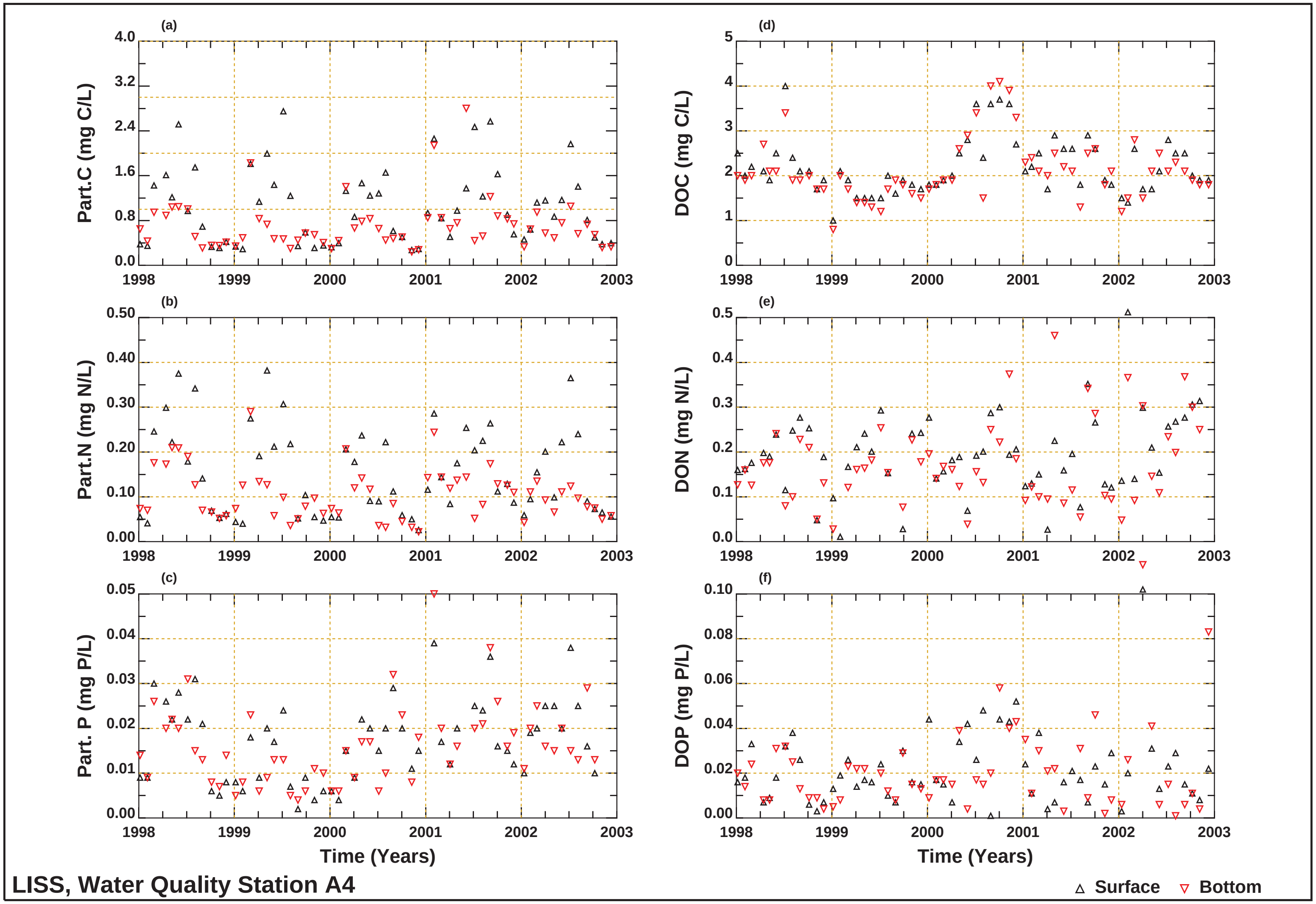


Figure 4. Particulate carbon, particulate nitrogen, particulate phosphorus, dissolved organic carbon, dissolved organic nitrogen, and dissolved organic phosphorus data for station A4

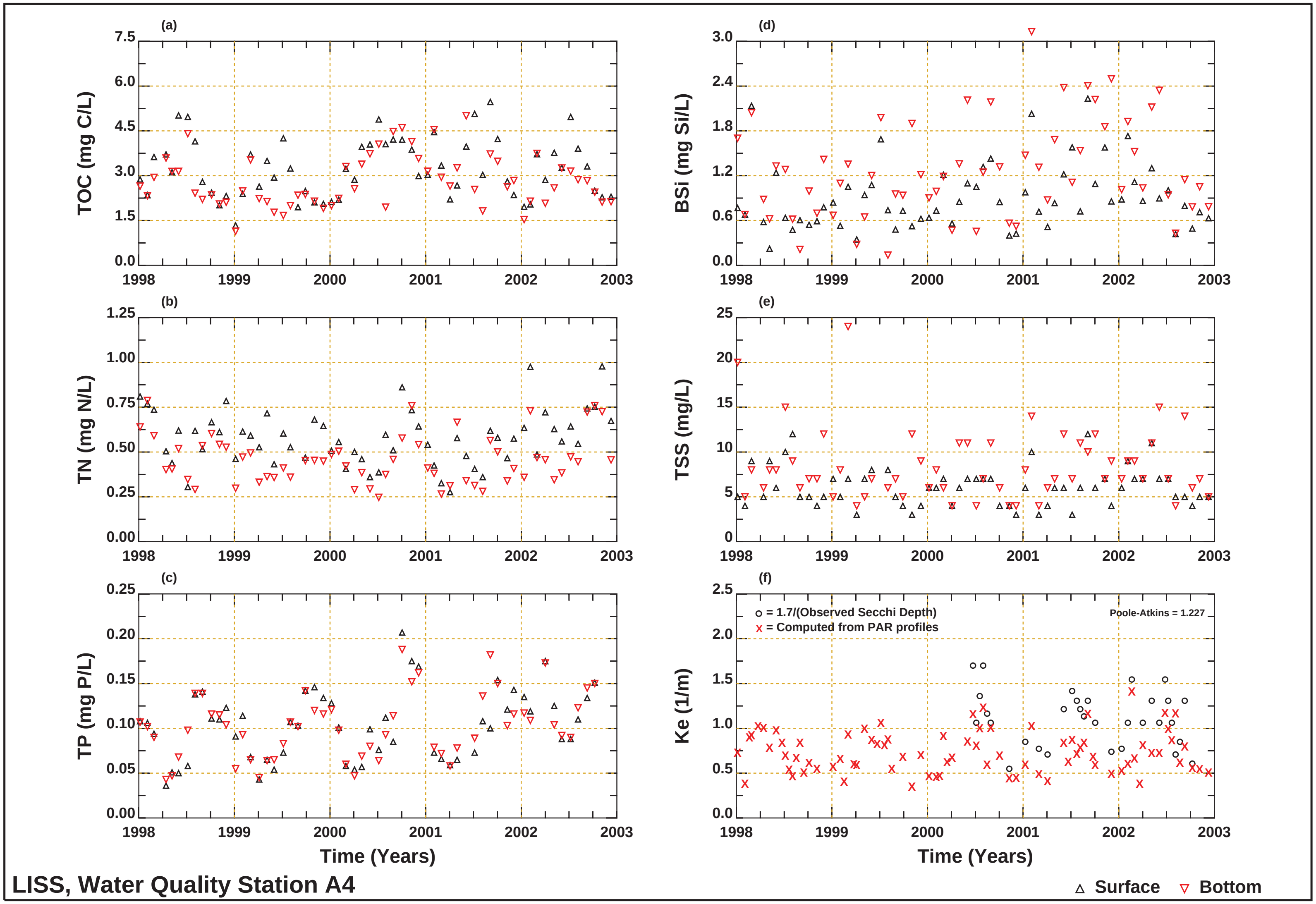


Figure 5. Total organic carbon, total nitrogen, total phosphorus, biogenic silica, total suspended solids, and light attenuation data for station A4

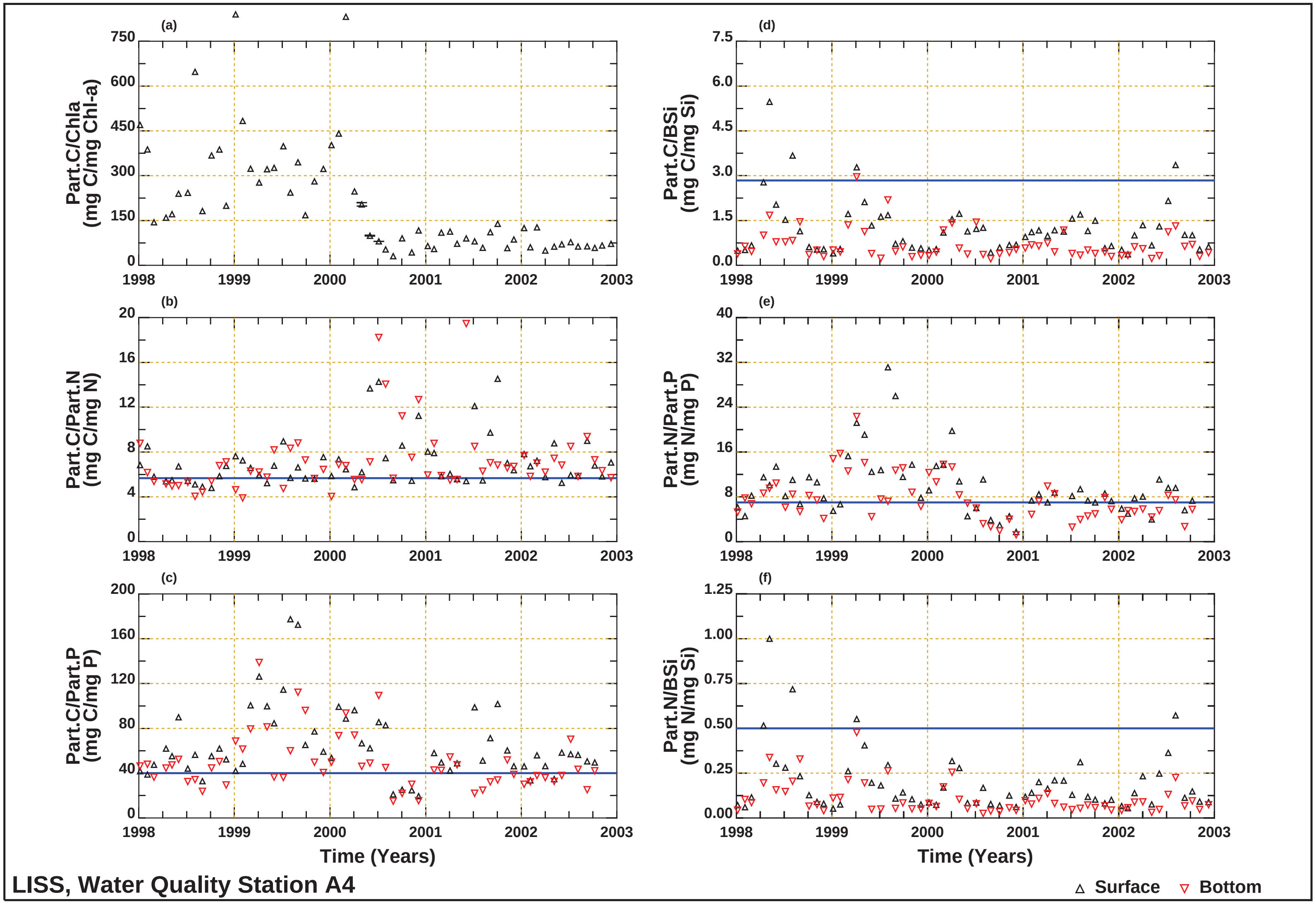


Figure 6. PC:Chl-a ratio, PC:PN ratio, PC:PP ratio, PC:BSi ratio, PN:PP and PN:BSi ratio data for station A4

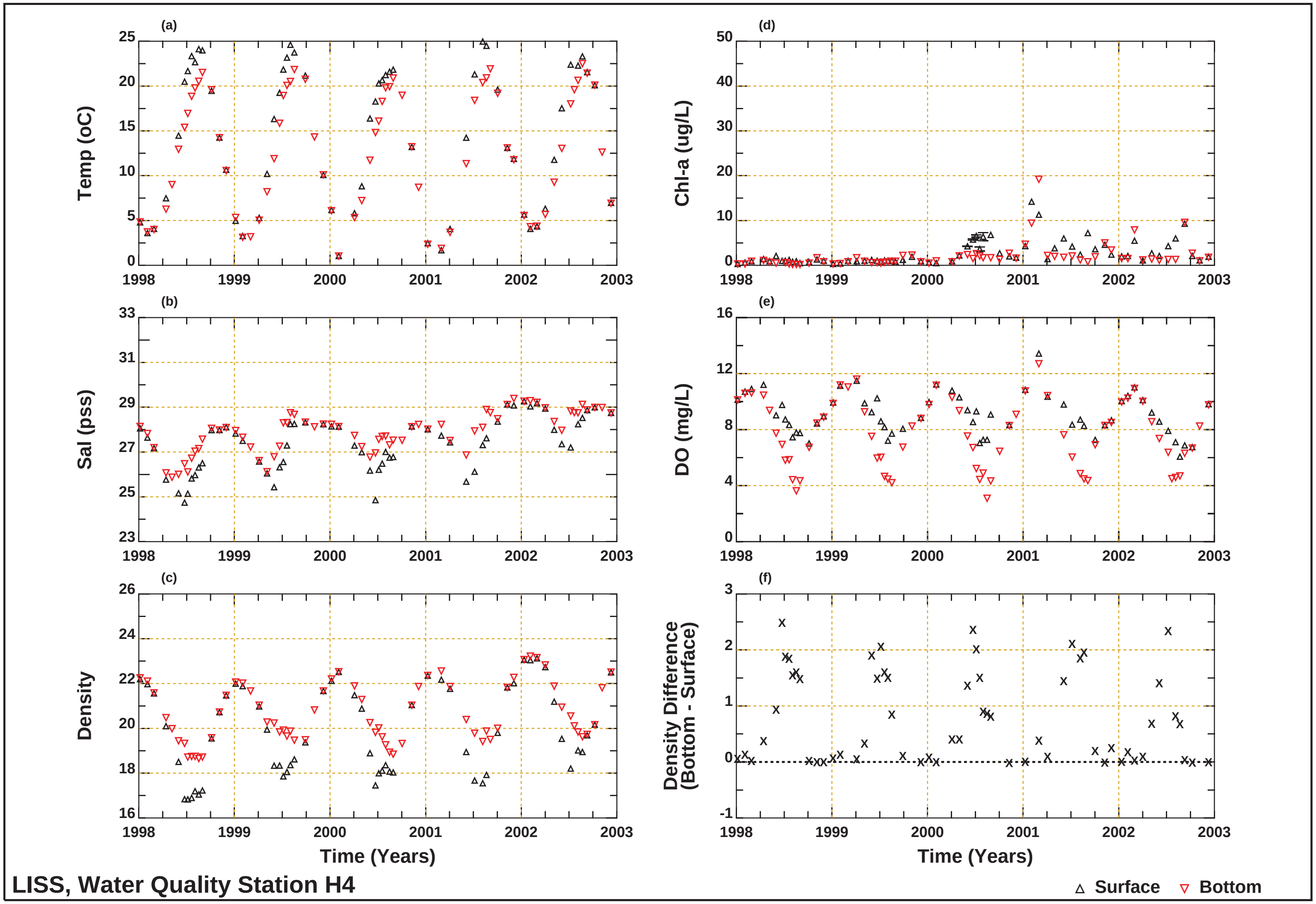


Figure 7. Temperature, salinity, density, phytoplankton chlorophyll, and dissolved oxygen data for station H4

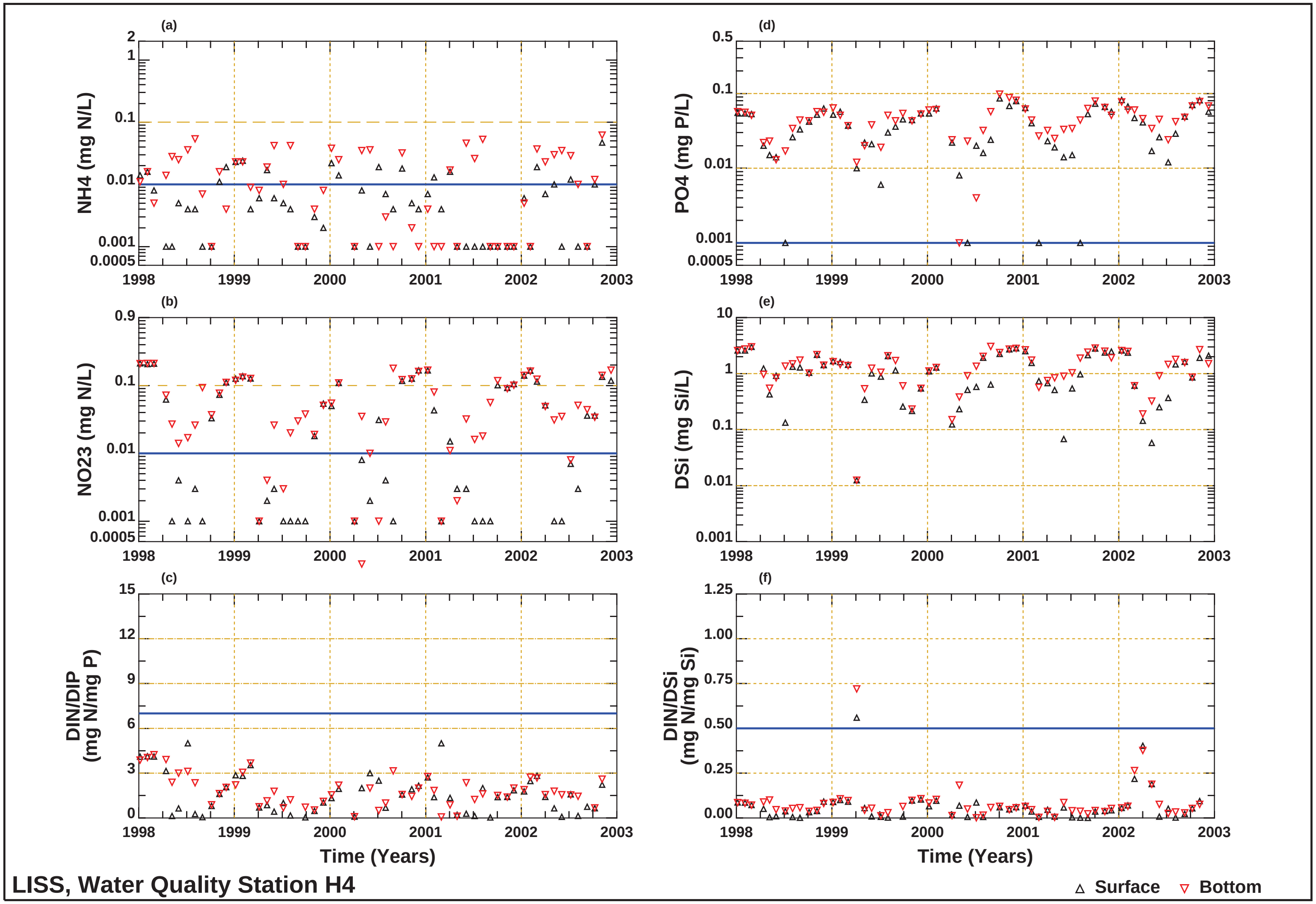


Figure 8. Ammonium, nitirite+nitrate, DIN:DIP, ortho-phosphate, dissolved silica and DIN:Si data for station H4

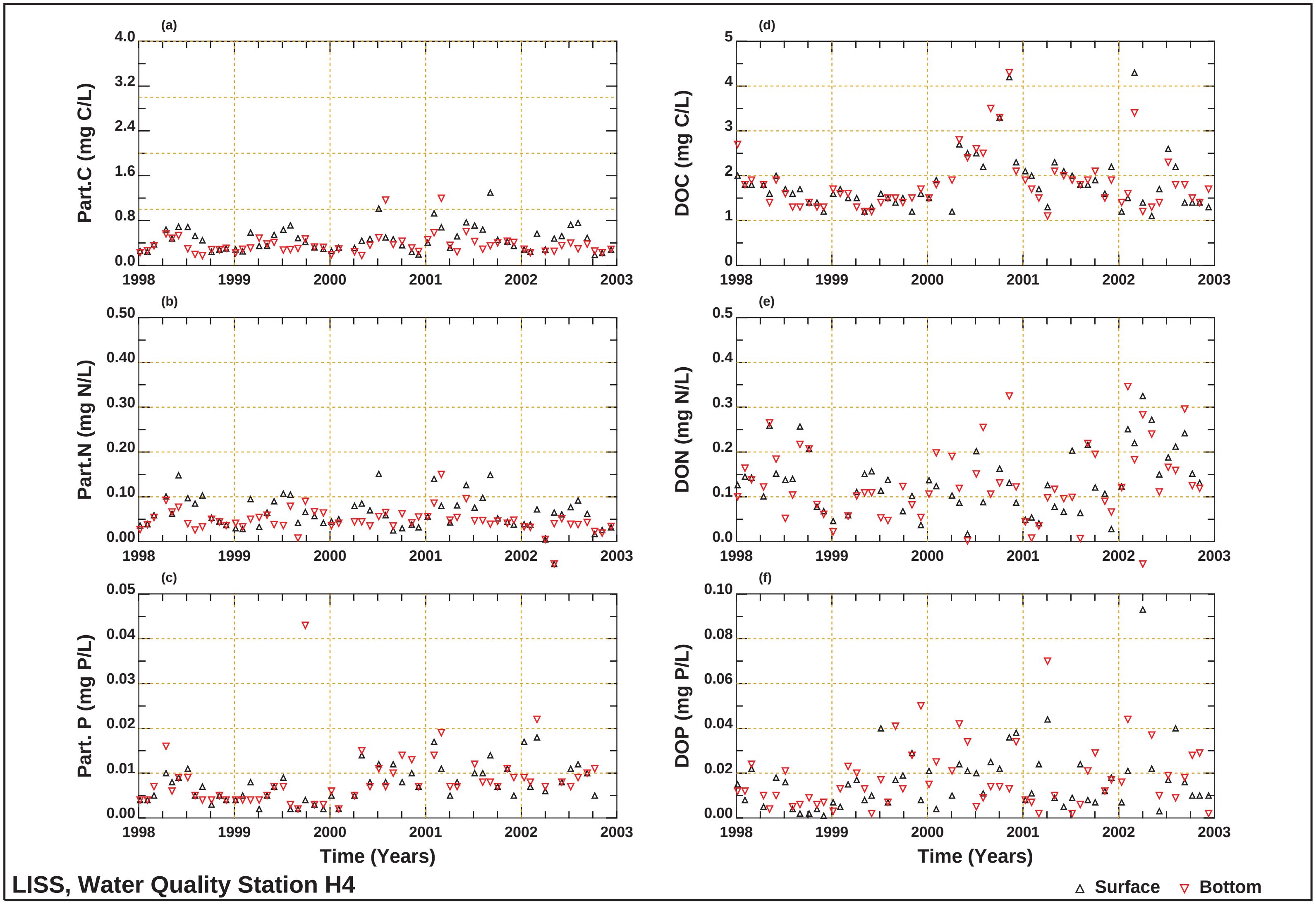


Figure 9. Particulate carbon, particulate nitrogen, particulate phosphorus, dissolved organic carbon, dissolved organic nitrogen, and dissolved organic phosphorus data for station H4

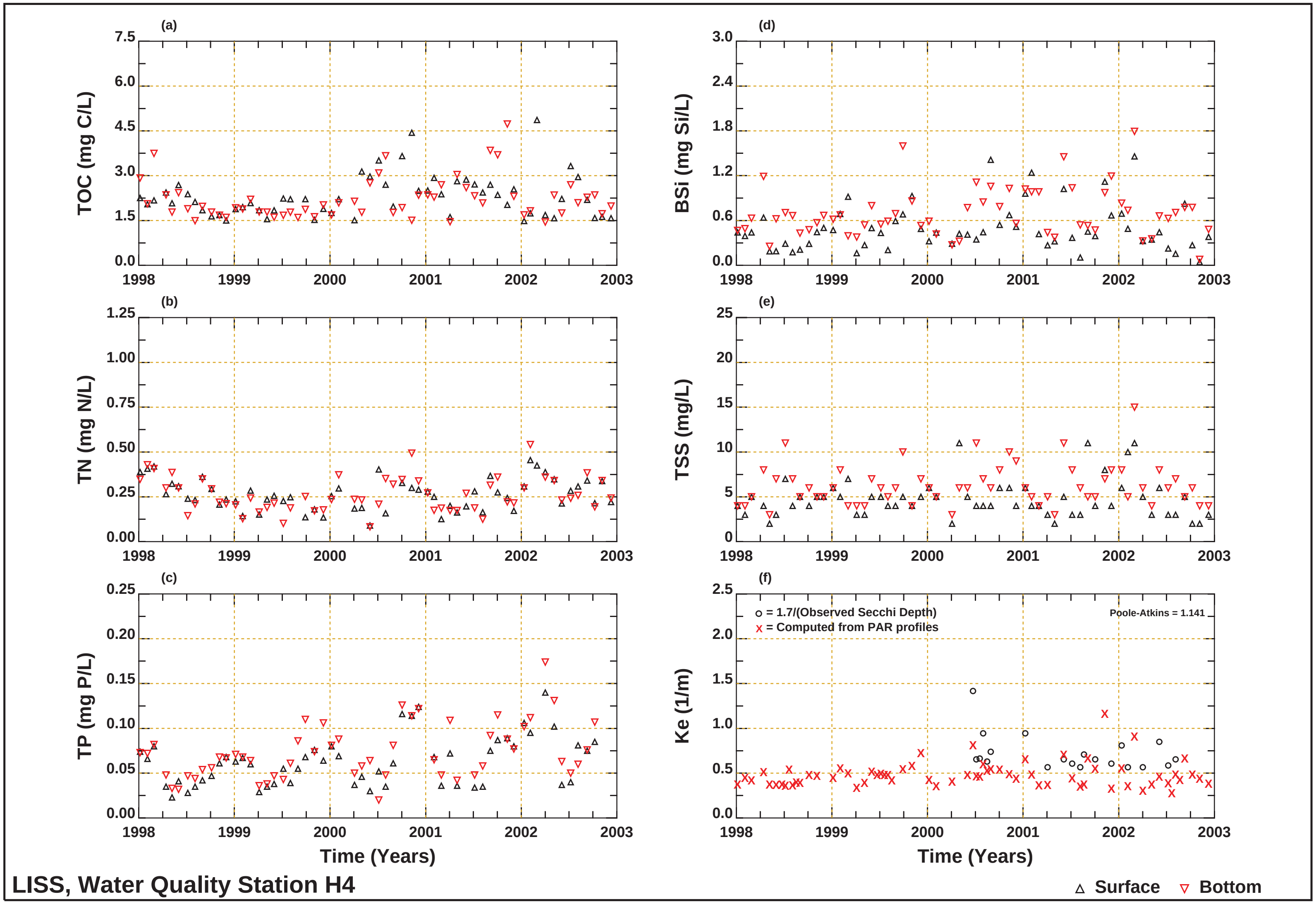


Figure 10. Total organic carbon, total nitrogen, total phosphorus, biogenic silica, total suspended solids, and light attenuation data for station H4

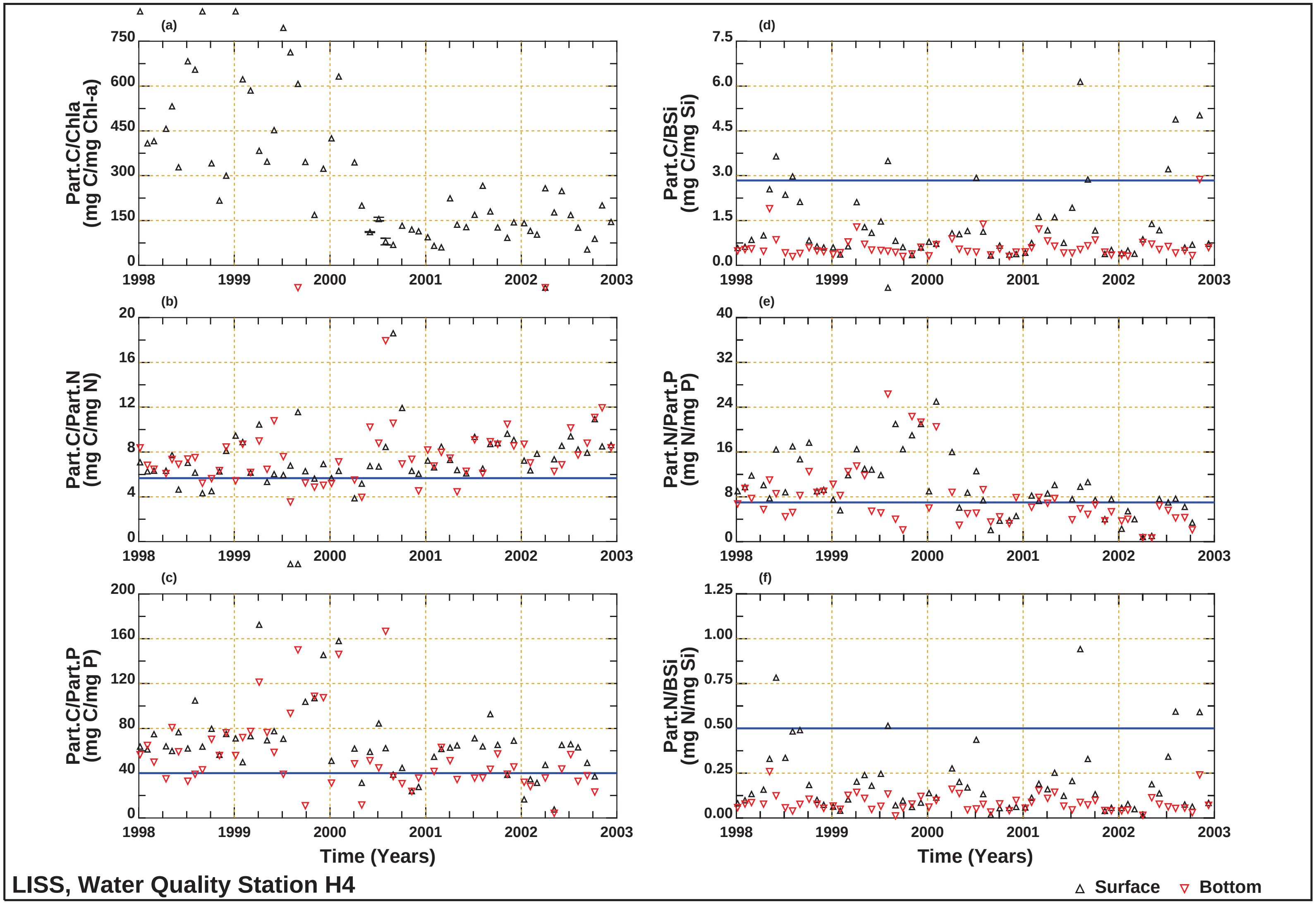


Figure 11. PC:Chl-a ratio, PC:PN ratio, PC:PP ratio, PC:BSi ratio, PN:PP and PN:BSi ratio data for station H4

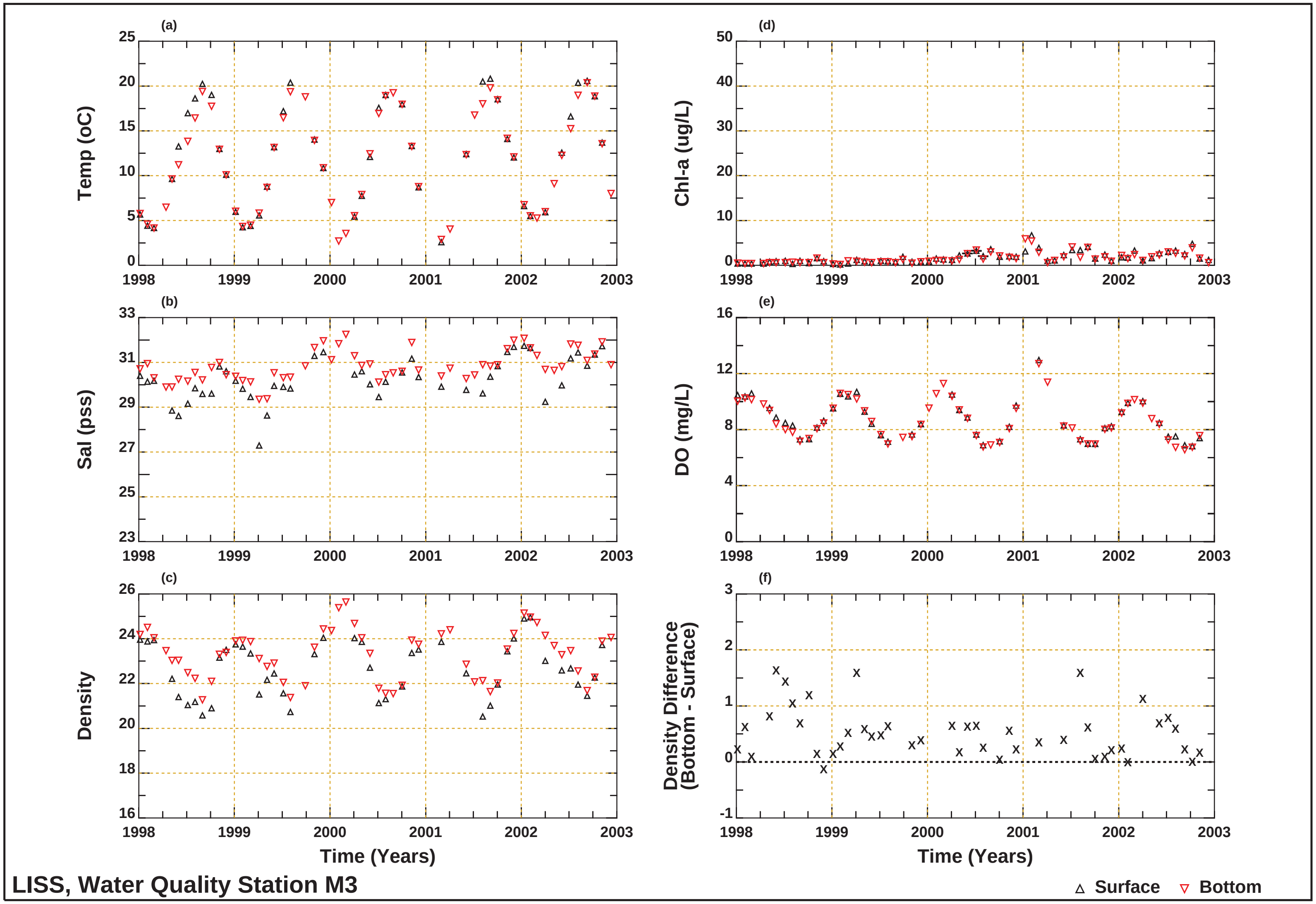


Figure 12. Temperature, salinity, density, phytoplankton chlorophyll, and dissolved oxygen data for station M3

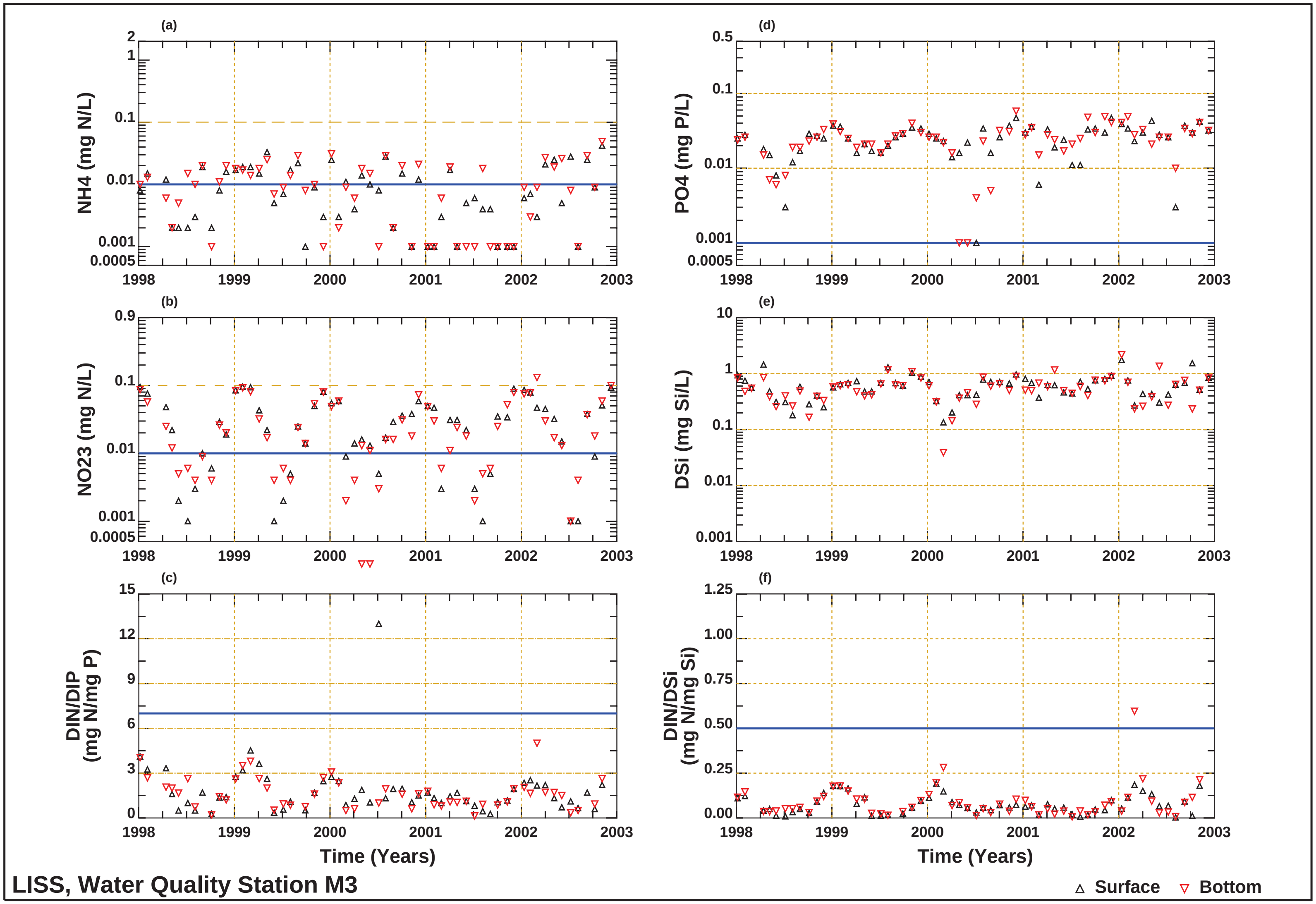


Figure 13. Ammonium, nitirite+nitrate, DIN:DIP, ortho-phosphate, dissolved silica and DIN:Si data for station M3

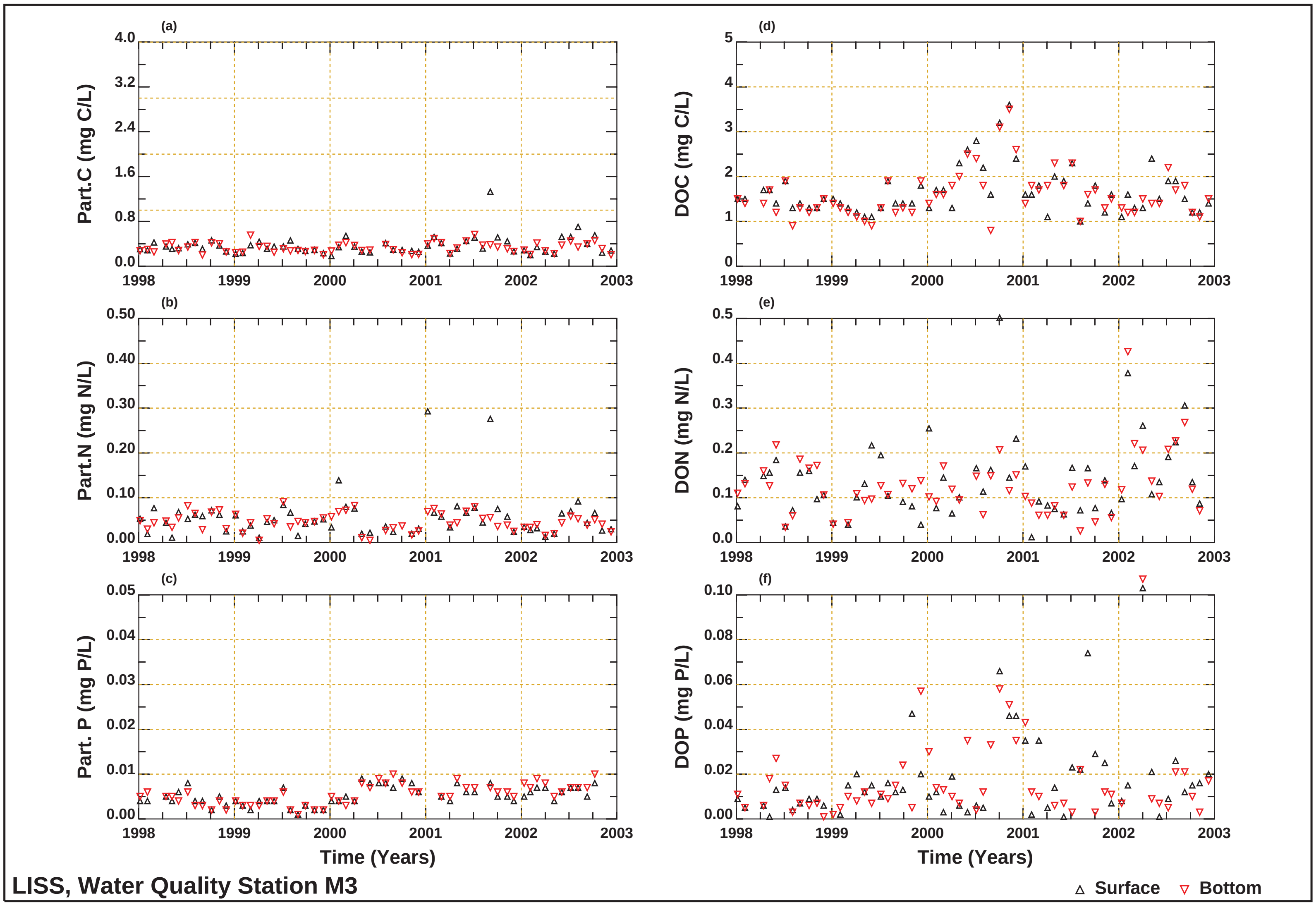


Figure 14. Particulate carbon, particulate nitrogen, particulate phosphorus, dissolved organic carbon, dissolved organic nitrogen, and dissolved organic phosphorus data for station M3

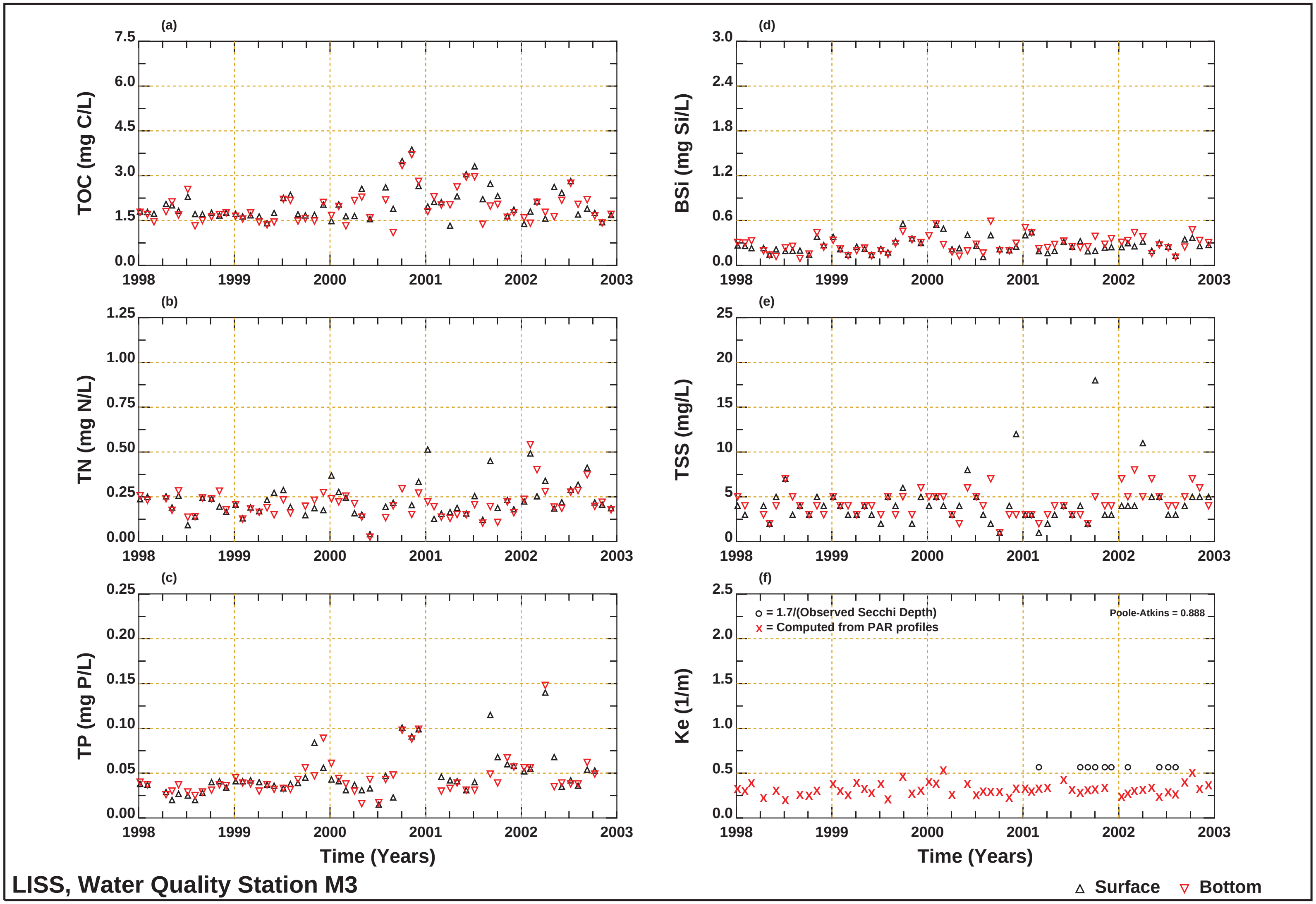


Figure 15. Total organic carbon, total nitrogen, total phosphorus, biogenic silica, total suspended solids, and light attenuation data for station M3

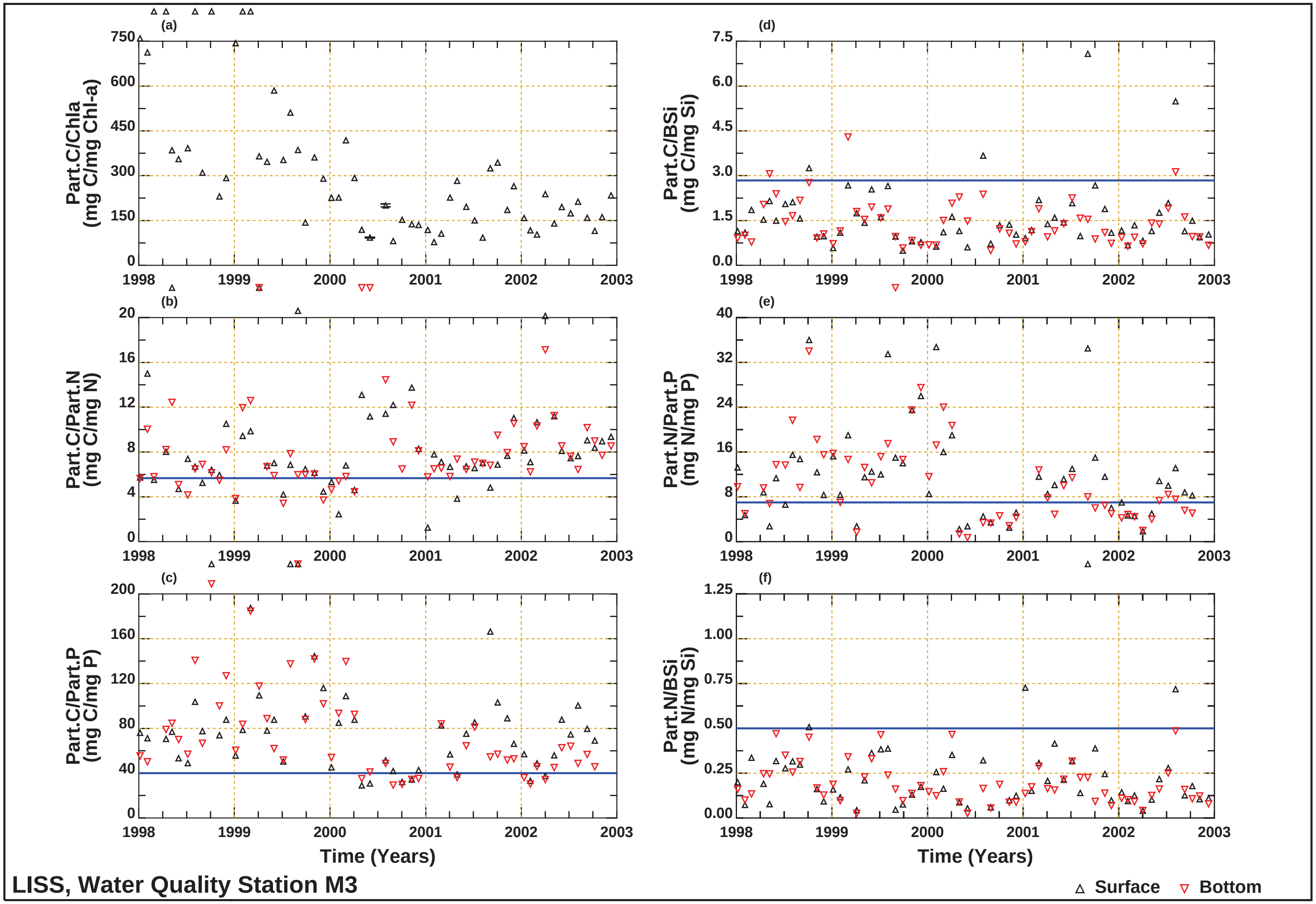


Figure 16. PC:Chl-a ratio, PC:PN ratio, PC:PP ratio, PC:BSi ratio, PN:PP and PN:BSi ratio data for station M3

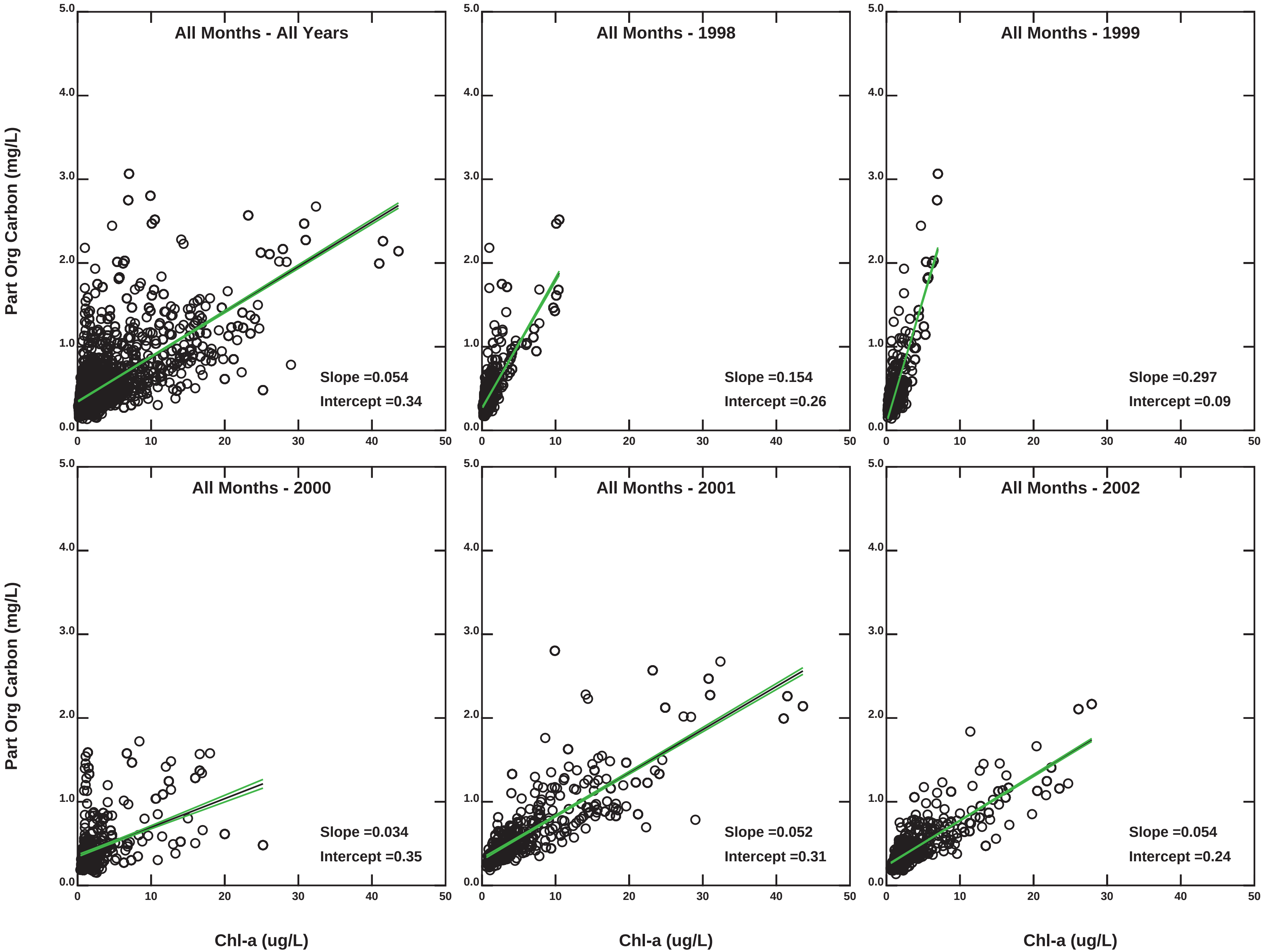


Figure 17. Yearly carbon to chlorophyll ratio data for all years

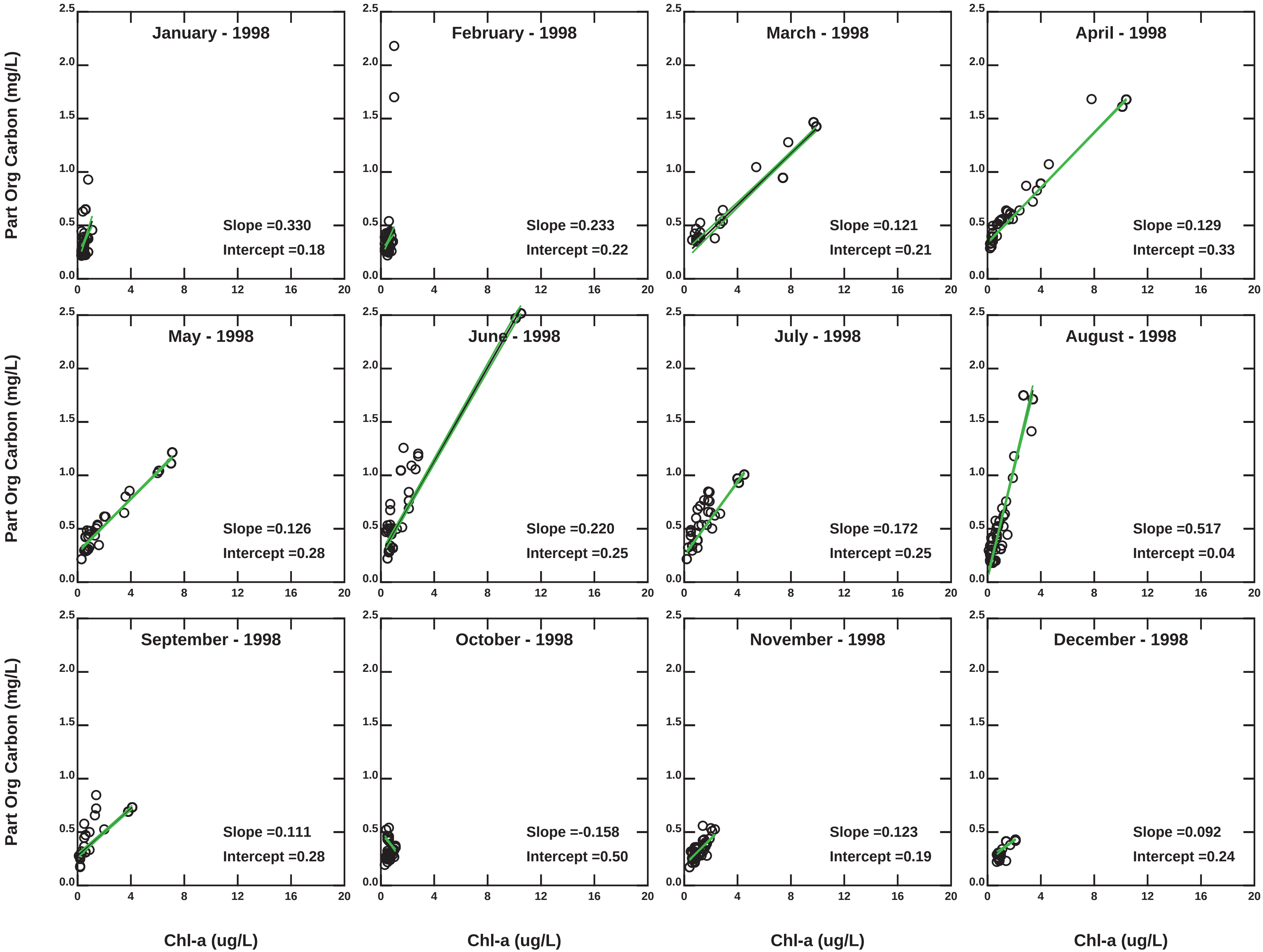


Figure 18. Monthly carbon to chlorophyll ratio data for 1998

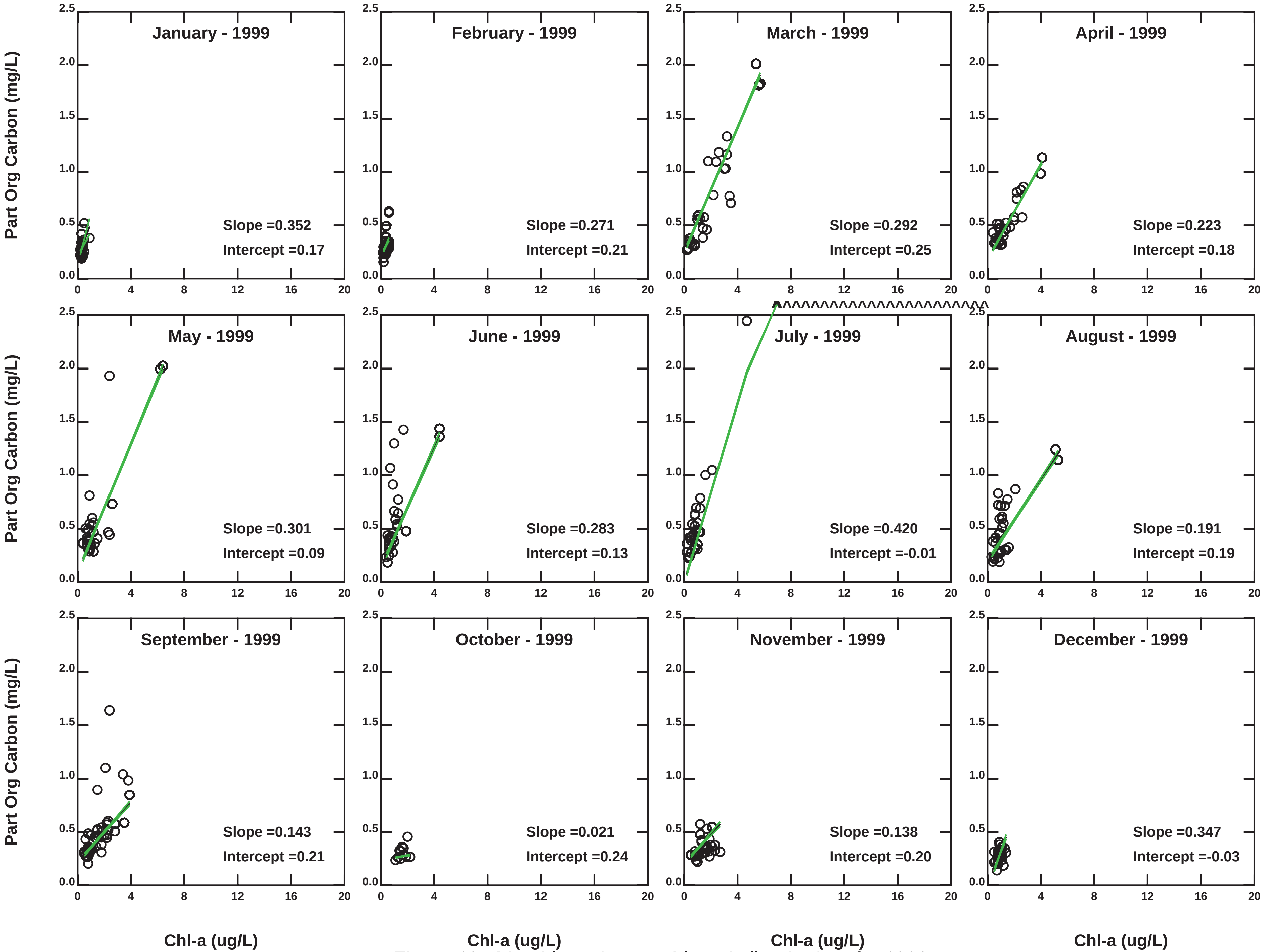


Figure 19. Monthly carbon to chlorophyll ratio data for 1999

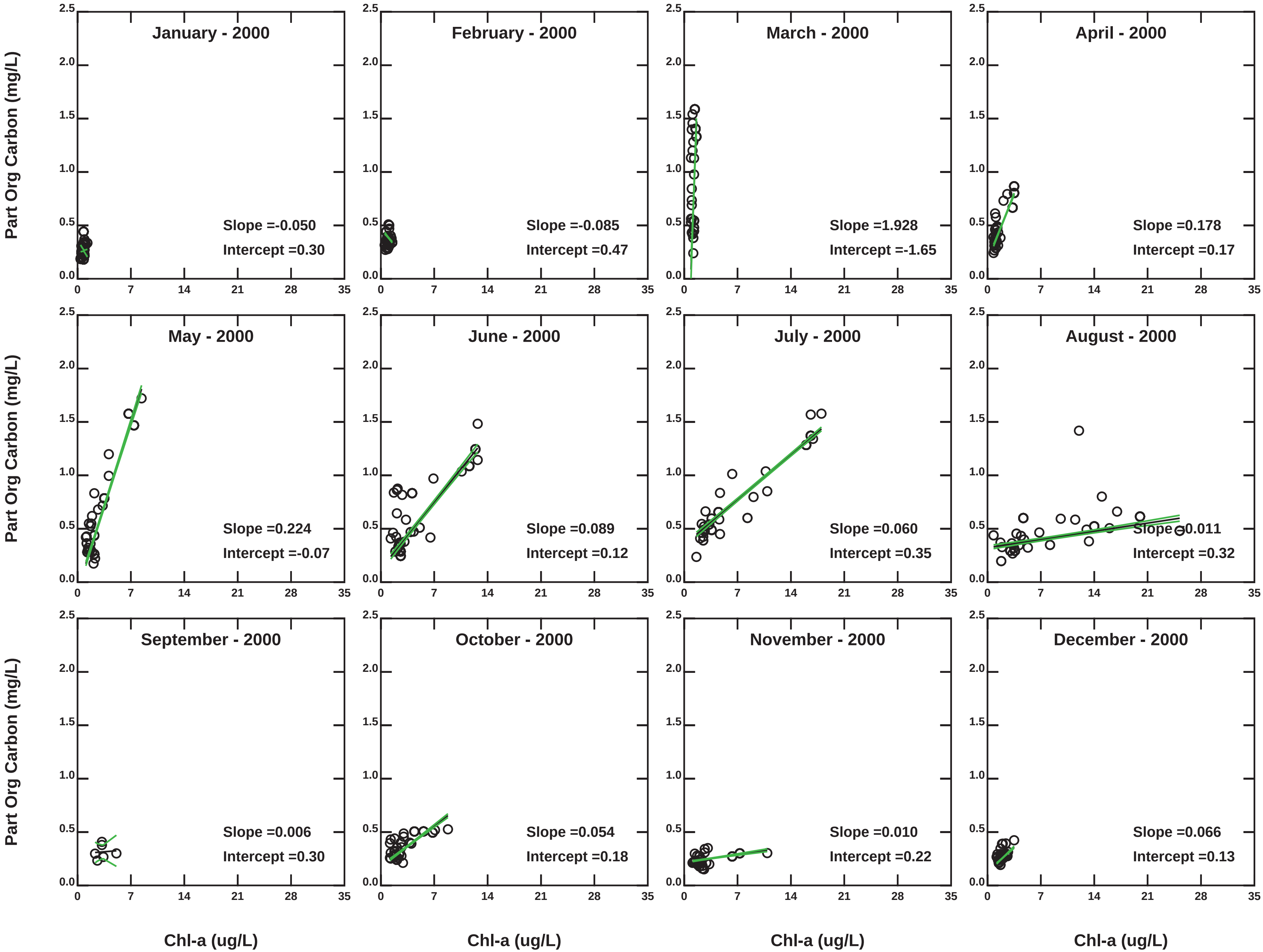


Figure 20. Monthly carbon to chlorophyll ratio data for 2000

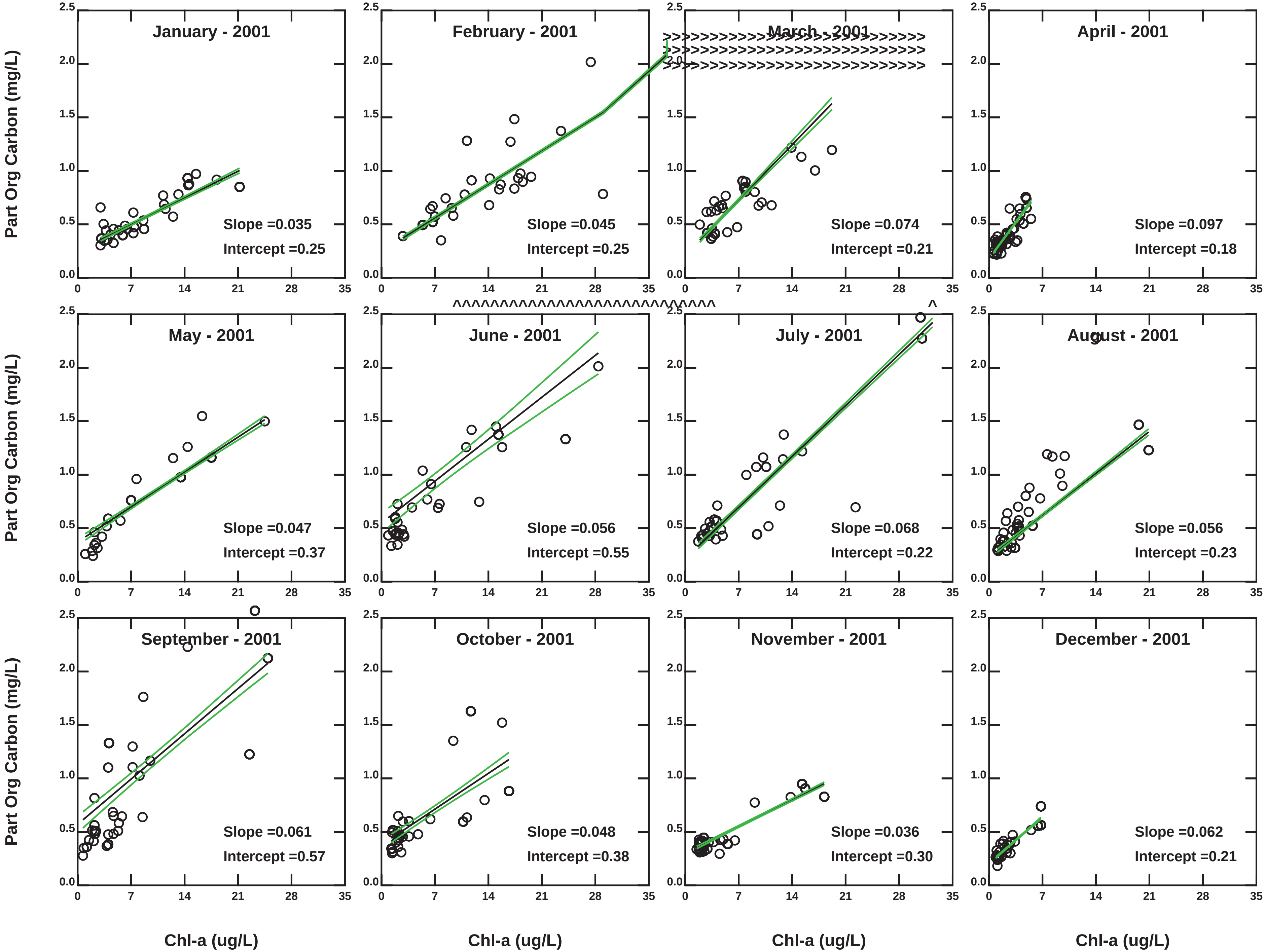


Figure 21. Monthly carbon to chlorophyll ratio data for 2001

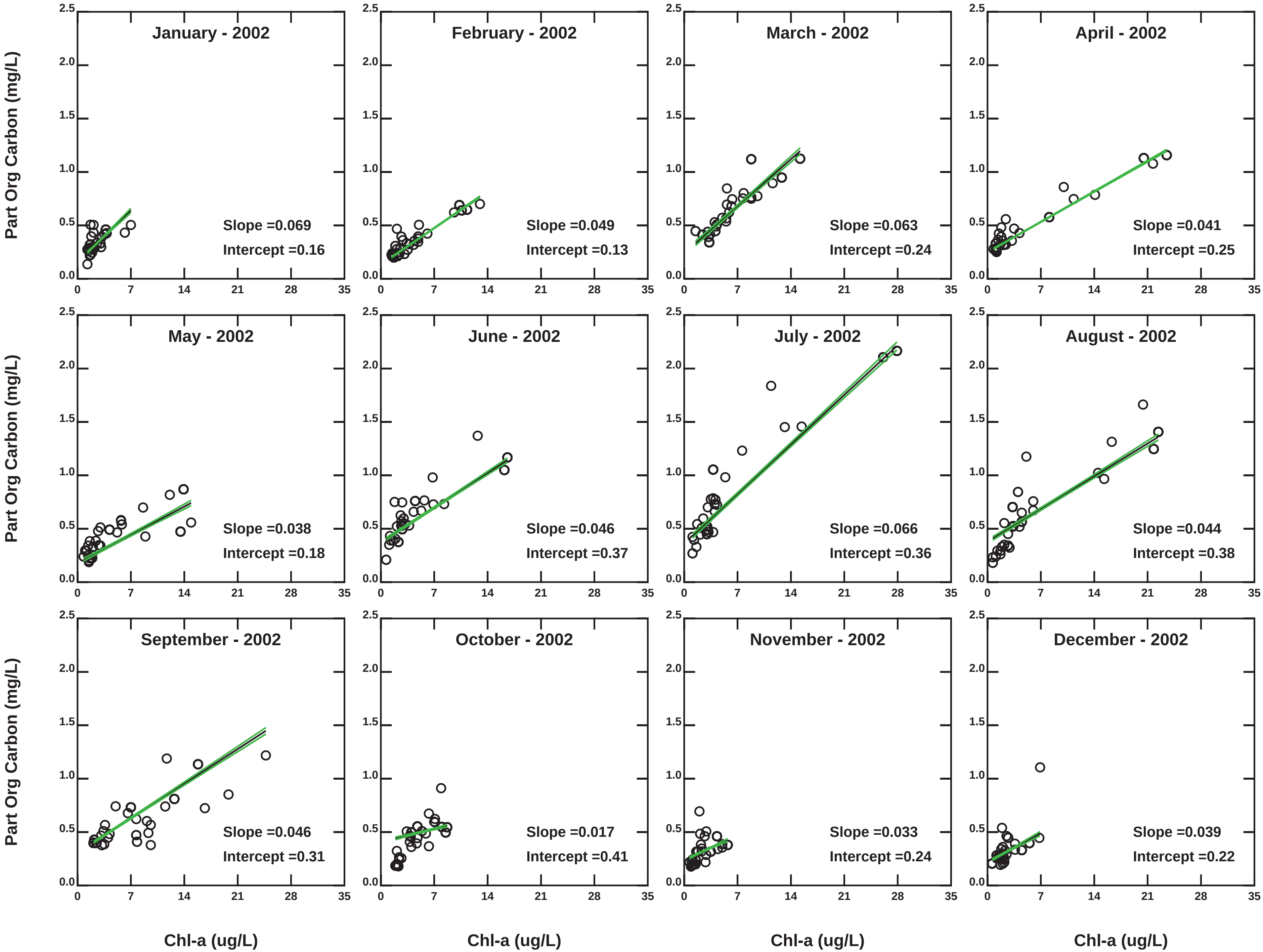
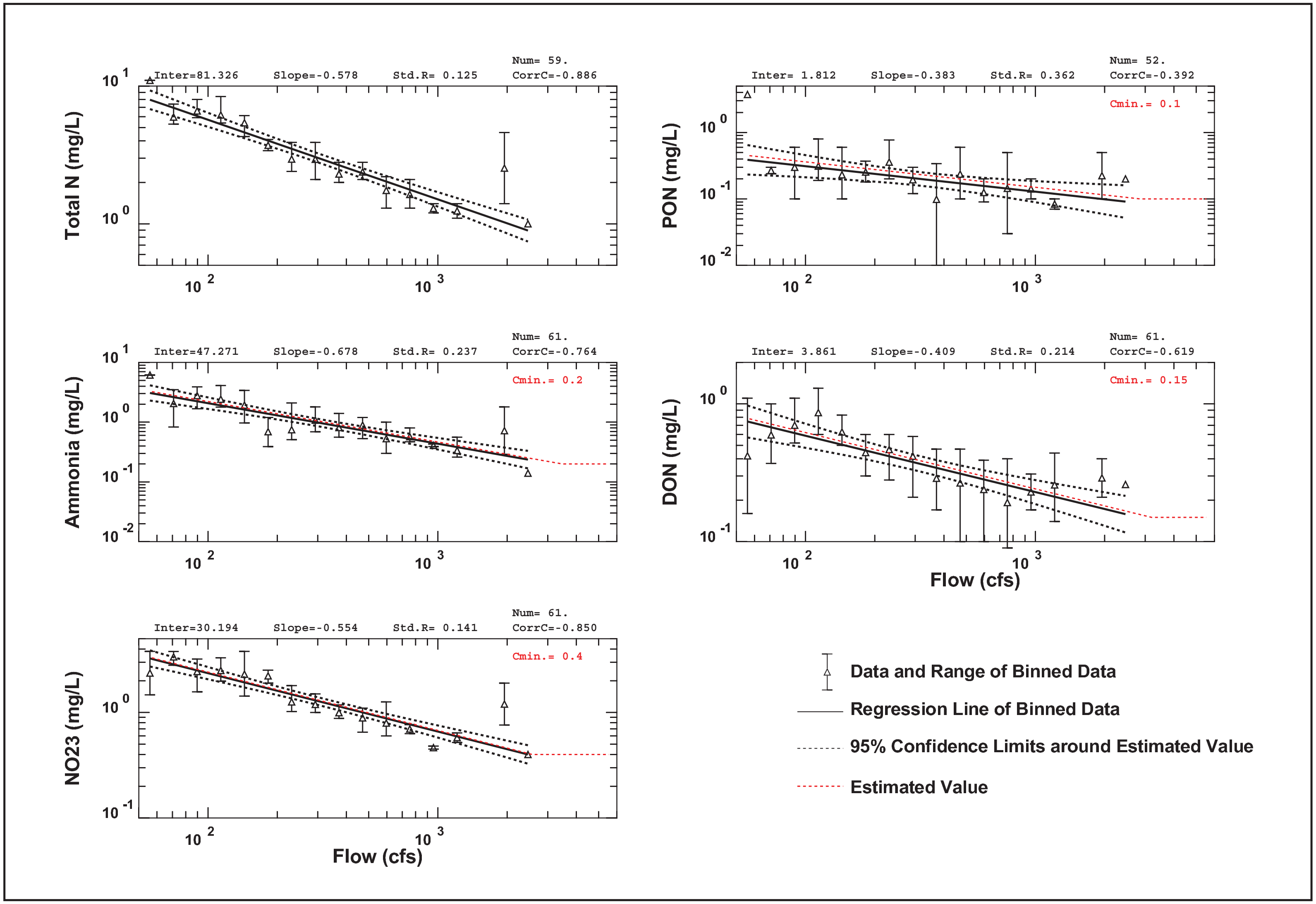


Figure 22. Monthly carbon to chlorophyll ratio data for 2002

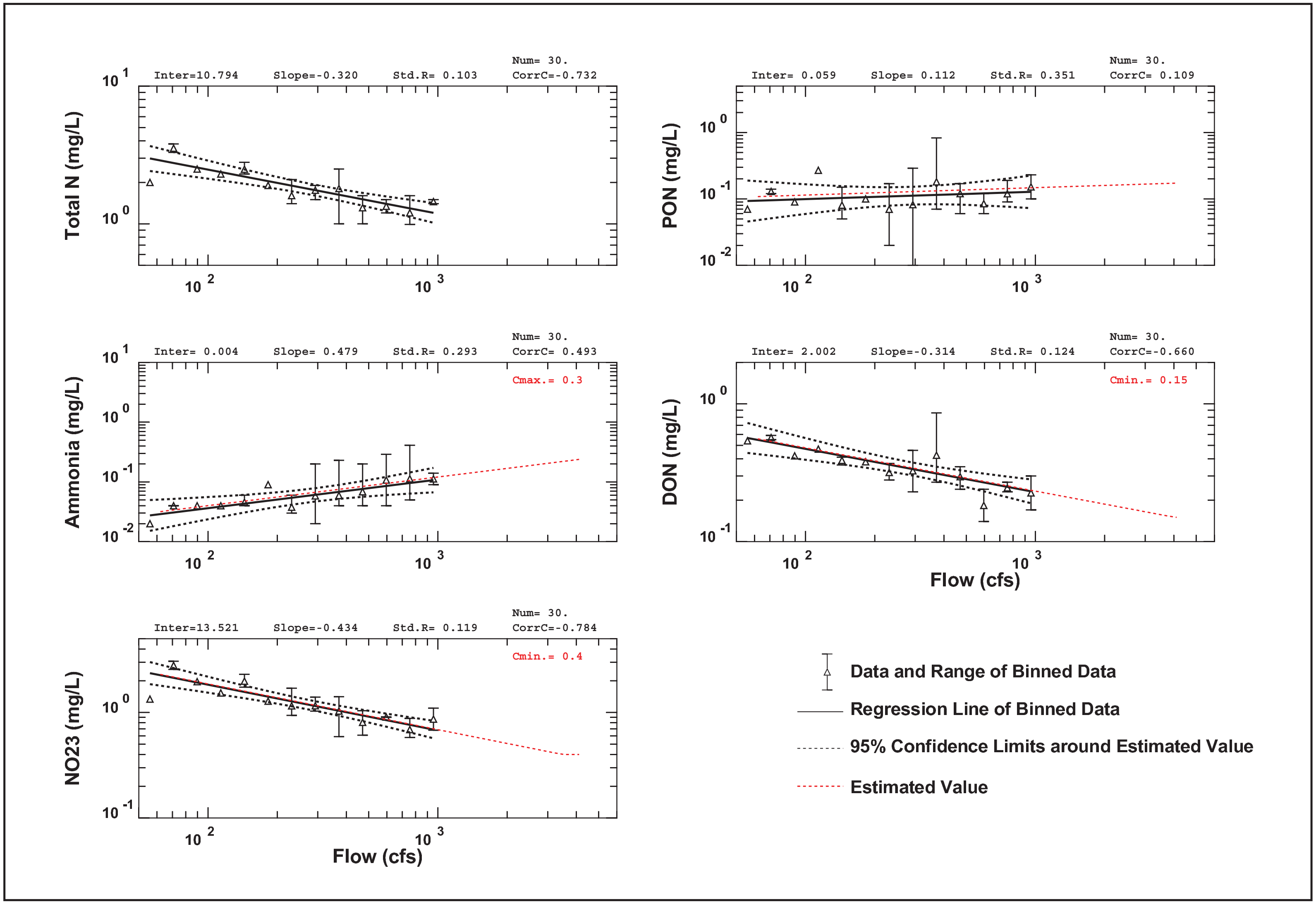
Nitrogen 1993 - 2000



Fall-Line Concentration versus Flow, Naugatuck River at Beacon Falls, CT.

Figure 23. Nitrogen regression analyses for the Naugatuck River January 1993-April 2000

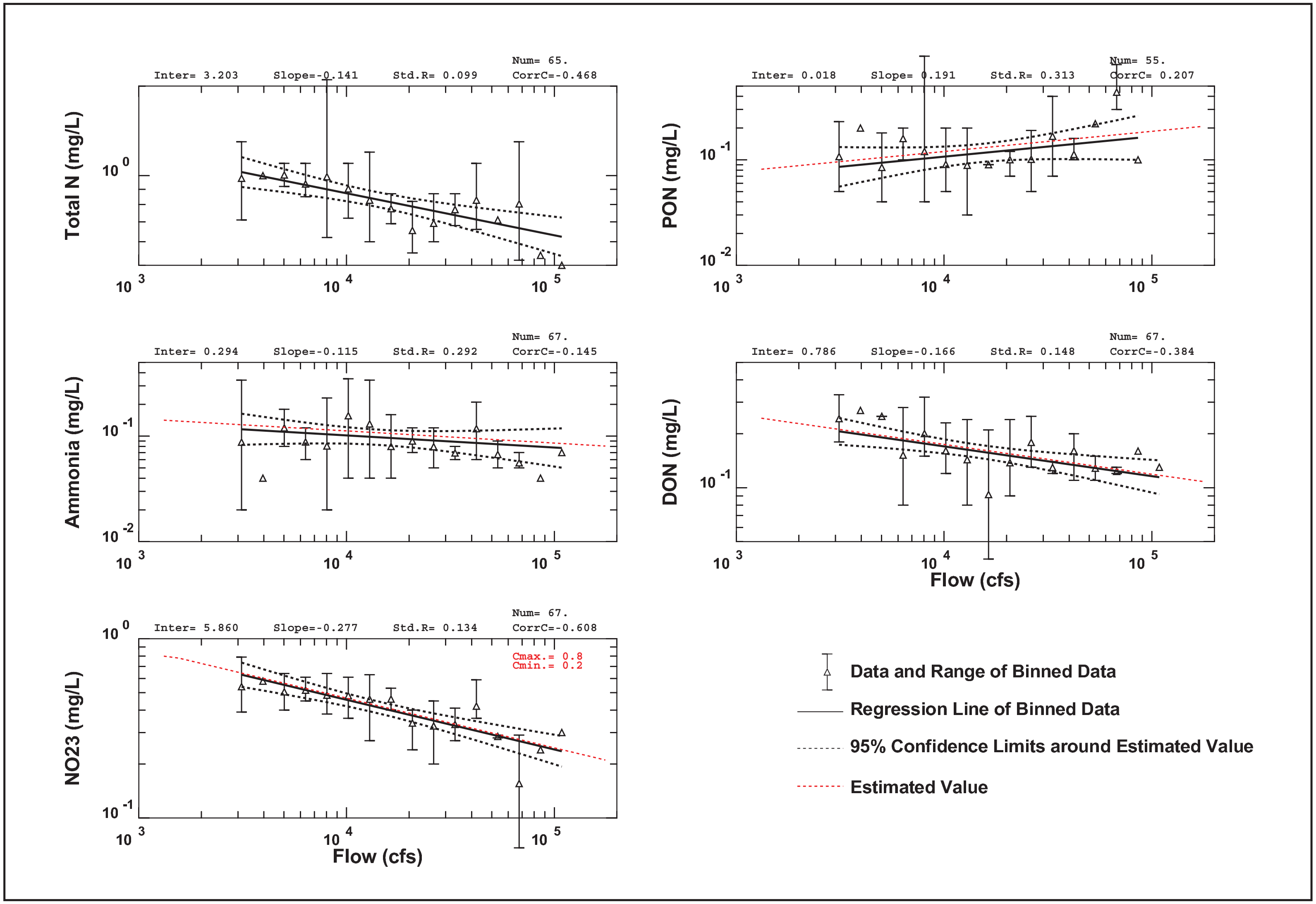
Nitrogen 2000 - 2003



Fall-Line Concentration versus Flow, Naugatuck River at Beacon Falls, CT.

Figure 24. Nitrogen regression analyses for the Naugatuck River May 2000-December 2003

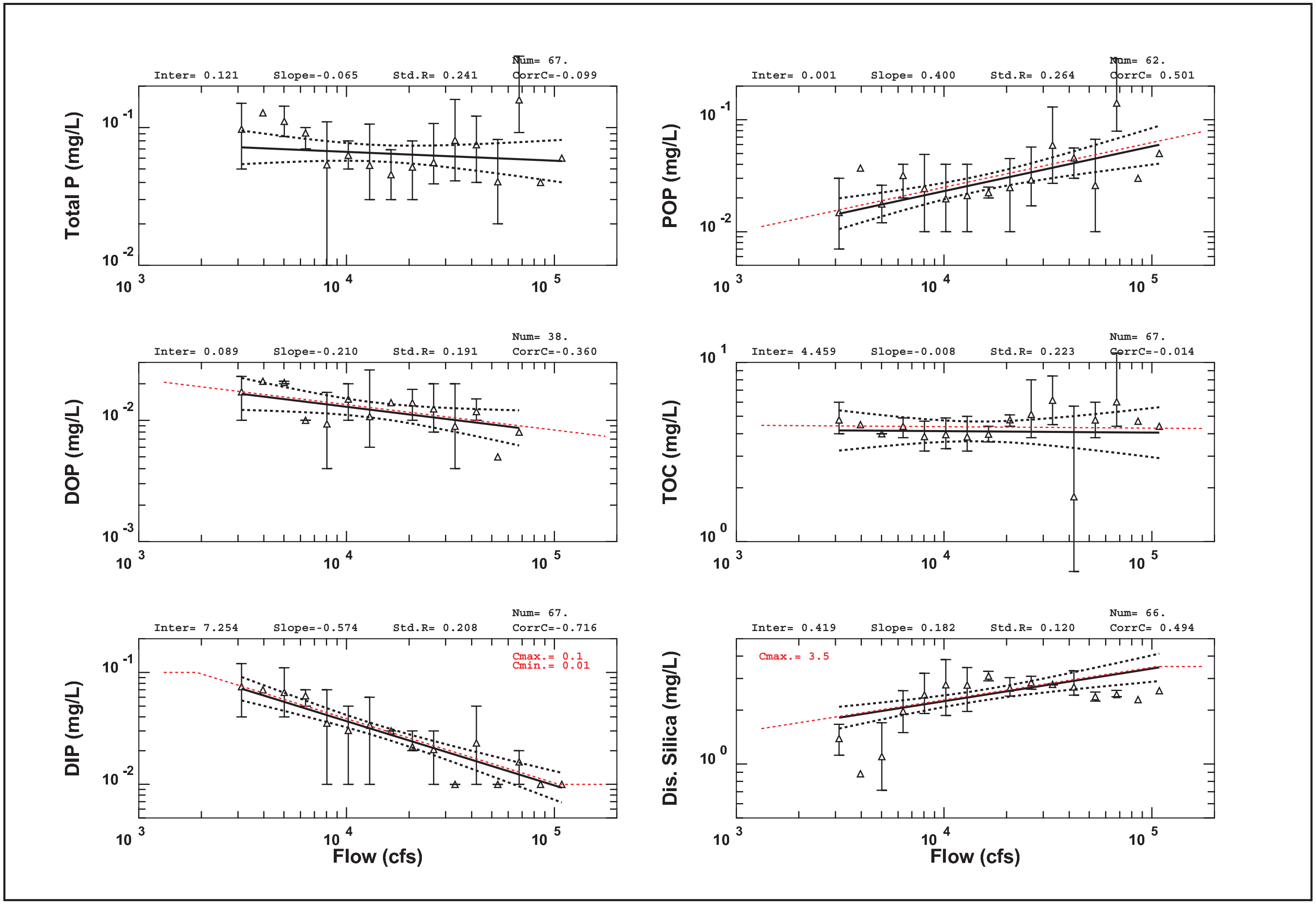
Nitrogen 1993 - 2003



Fall-Line Concentration versus Flow, Connecticut River at East Haddam, CT.

Figure 25. Nitrogen regression analyses for the Connecticut River

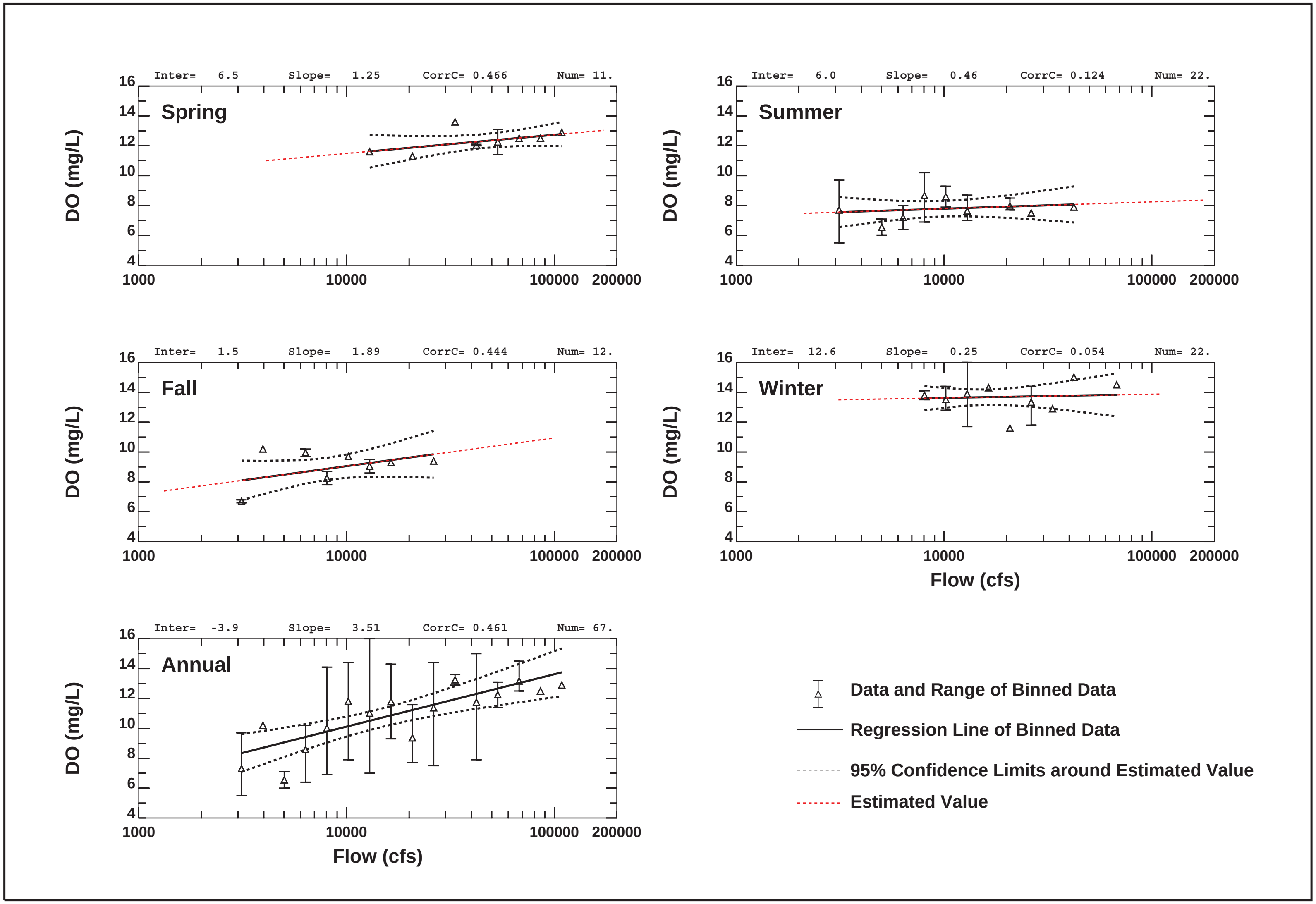
P, C and Si 1993 - 2003



Fall-Line Concentration versus Flow, Connecticut River at East Haddam, CT.

Figure 26. Phosphorus, carbon and silica regression analyses for the Connecticut River

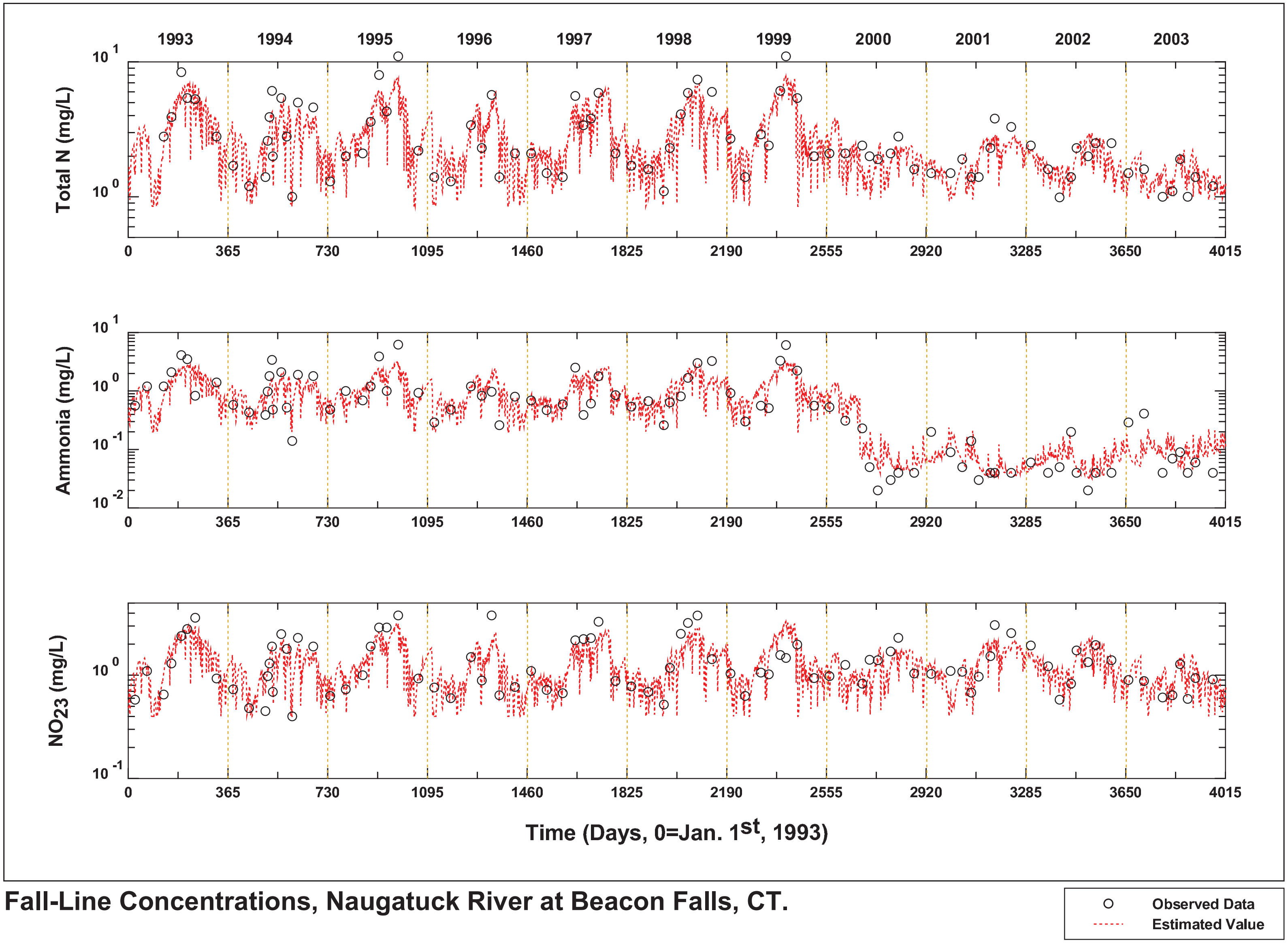
Dissolved Oxygen (mg/L) 1993 - 2003



Fall-Line Concentration versus Flow, Connecticut River at East Haddam, CT.

Figure 27. Dissolved oxygen regression analyses for the Connecticut River

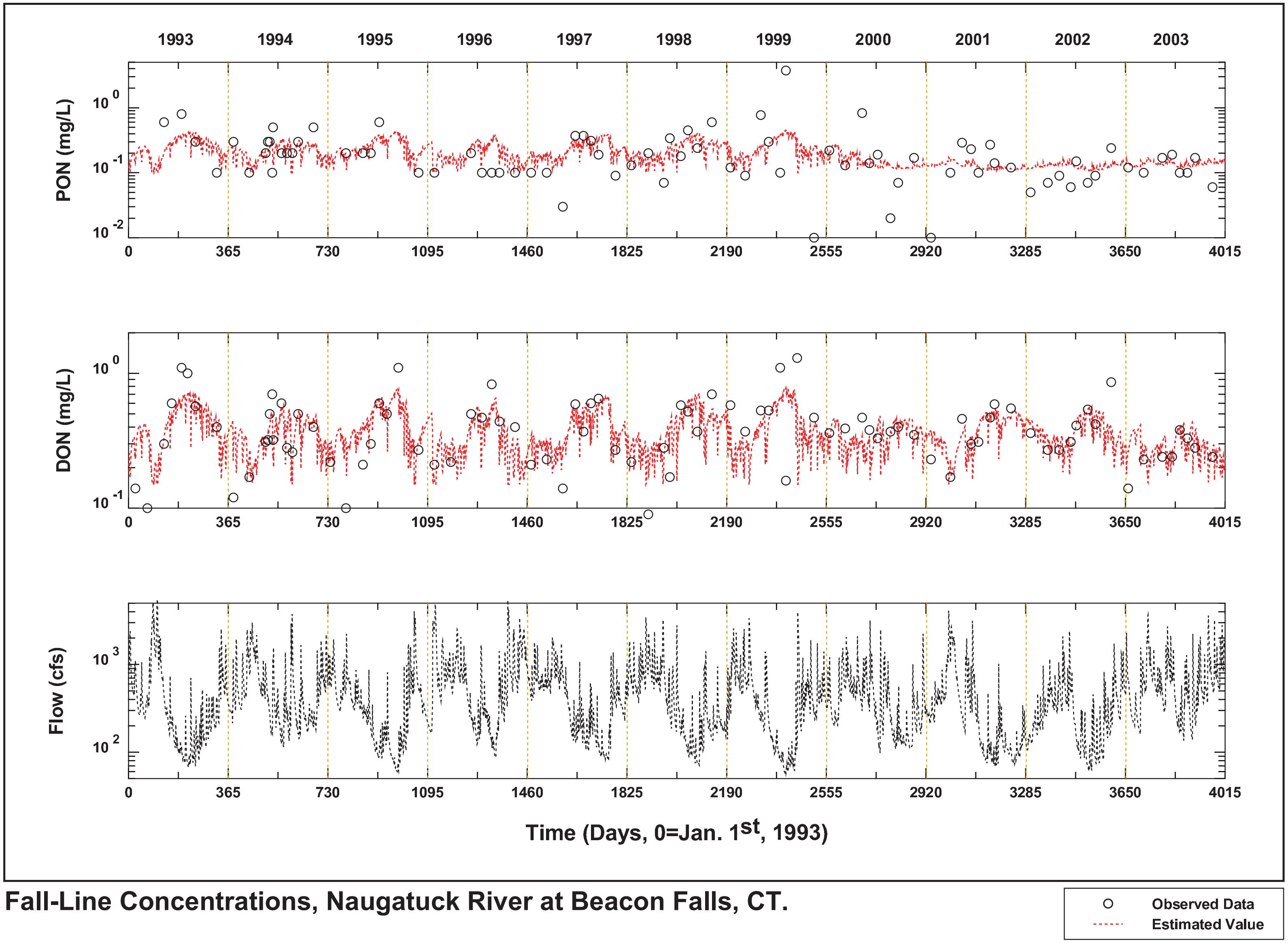
TN, NH_3 and NO_{23} (1993 - 2003)



Fall-Line Concentrations, Naugatuck River at Beacon Falls, CT.

Figure 28. Observed and estimated total nitrogen, ammonium and nitrite+nitrate concentrations for the Naugatuck River fall-line

PON, DON and Flow (1993 - 2003)



Fall-Line Concentrations, Naugatuck River at Beacon Falls, CT.

Figure 29. Observed and estimated PON and DON concentrations and observed flow for the Naugatuck River fall-line

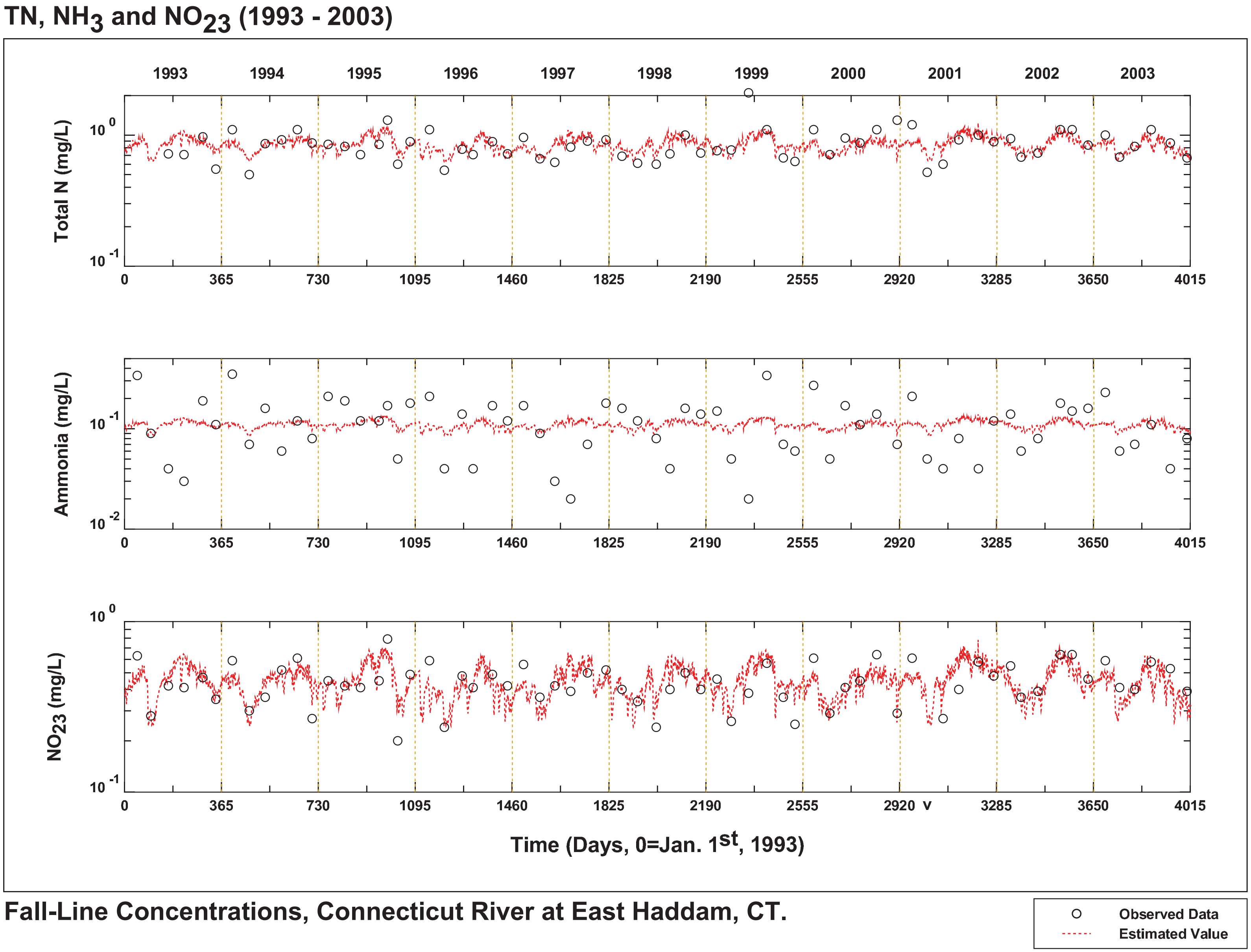


Figure 30. Observed and estimated total nitrogen, ammonium and nitrite+nitrate concentrations for the Connecticut River fall-line

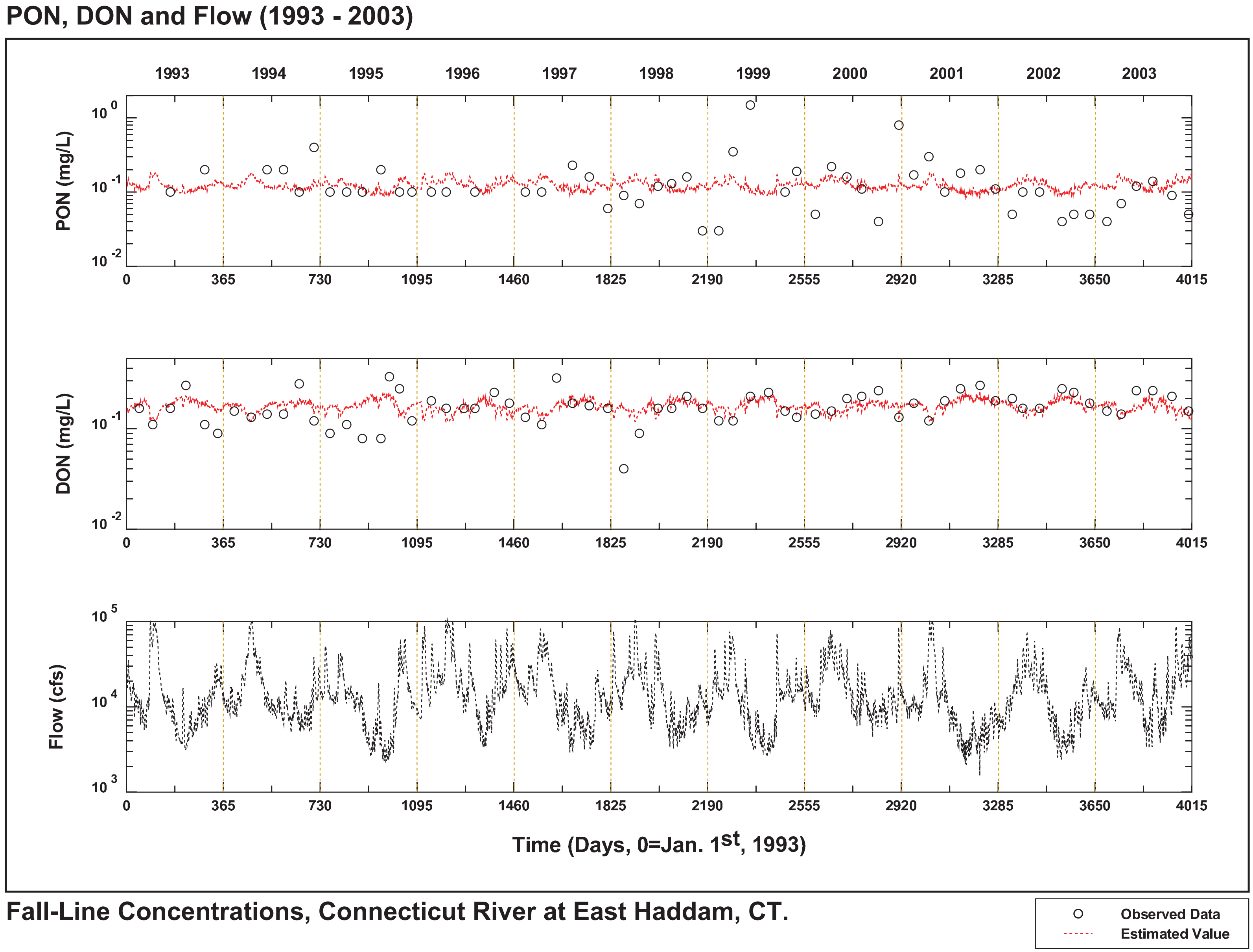


Figure 31. Observed and estimated PON and DON concentrations and observed flow for the Connecticut River fall-line

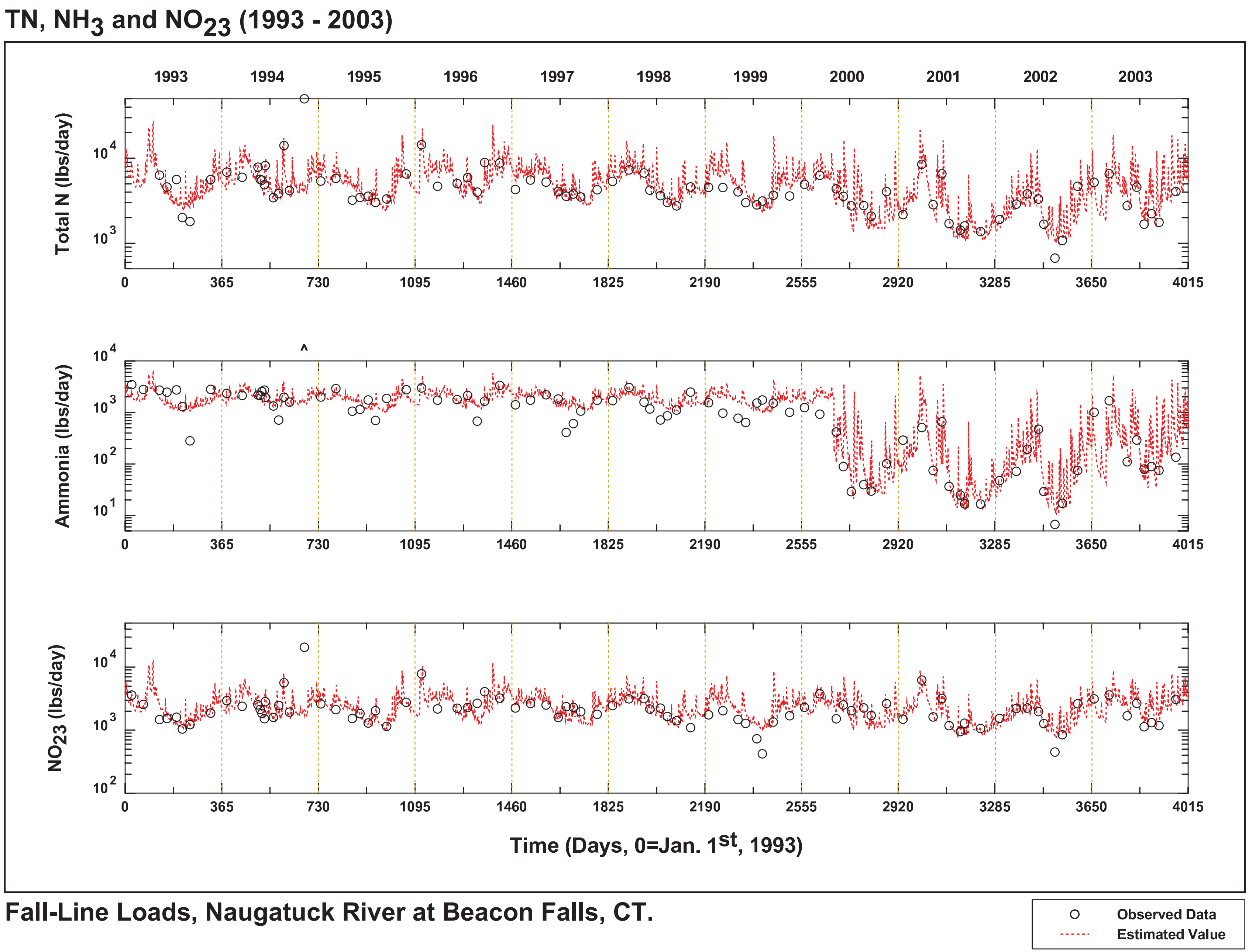
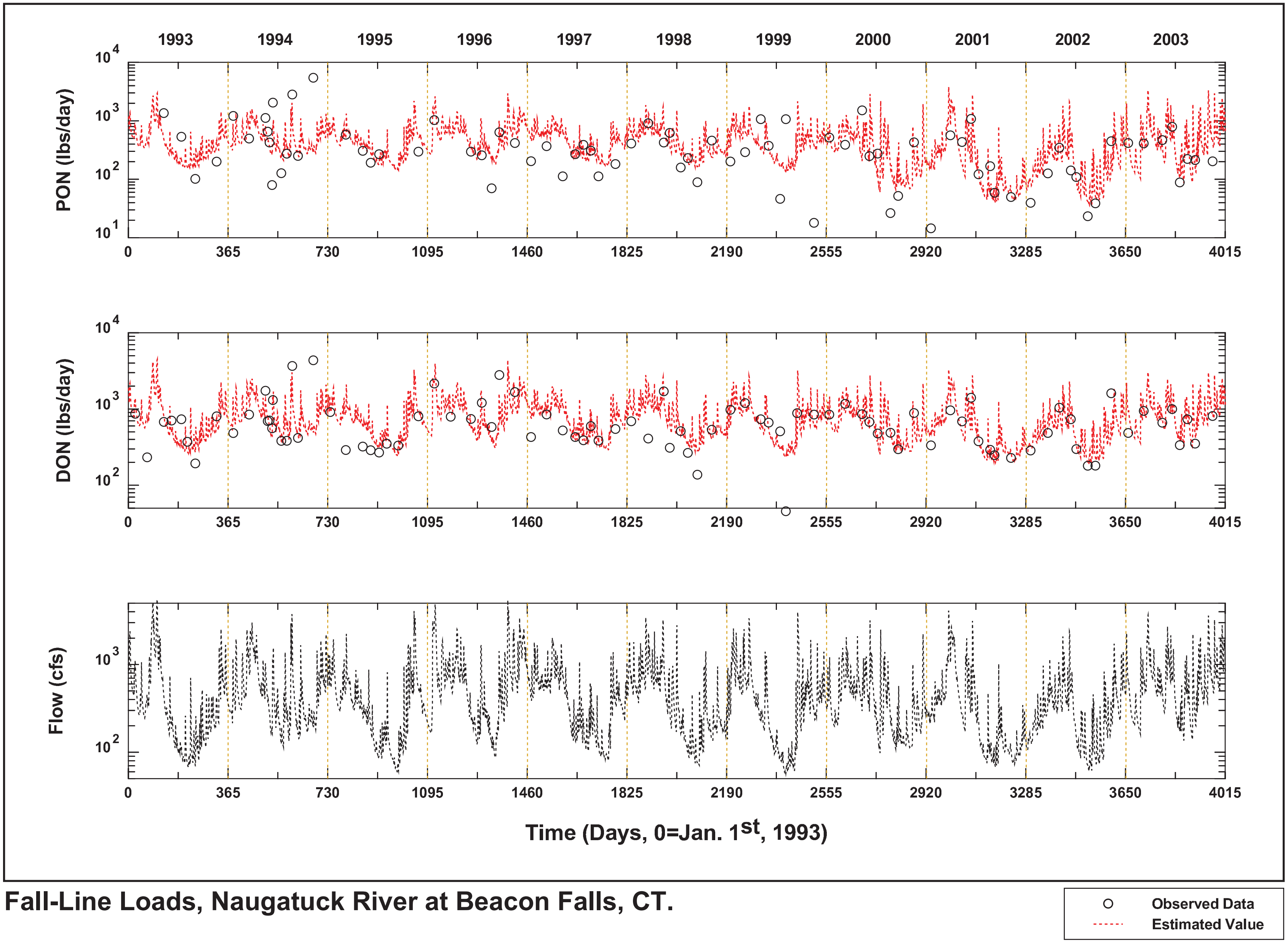


Figure 32. Observed and estimated total nitrogen, ammonium and nitrite+nitrate loadings for the Naugatuck River fall-line

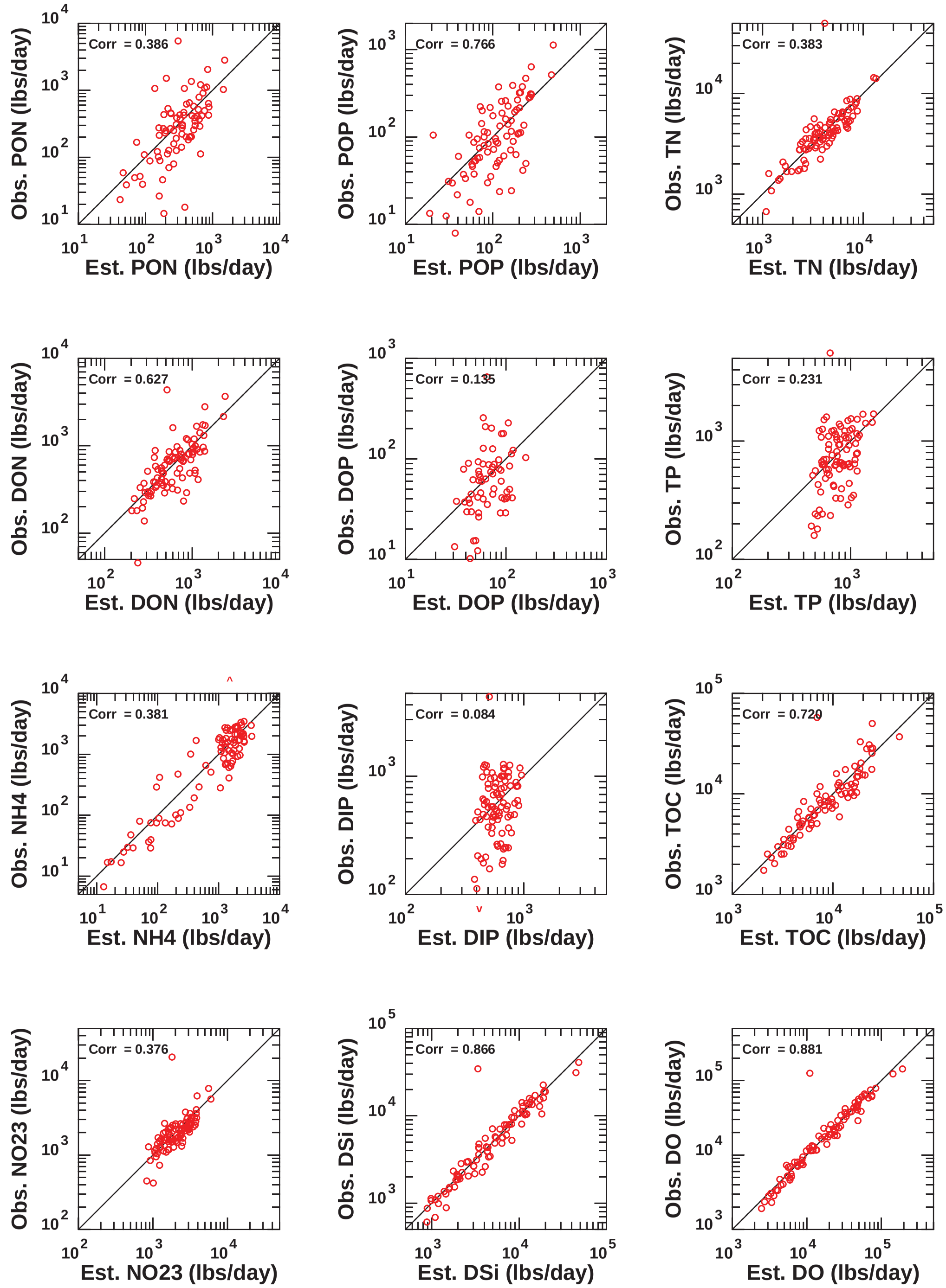
PON, DON and Flow (1993 - 2003)



Fall-Line Loads, Naugatuck River at Beacon Falls, CT.

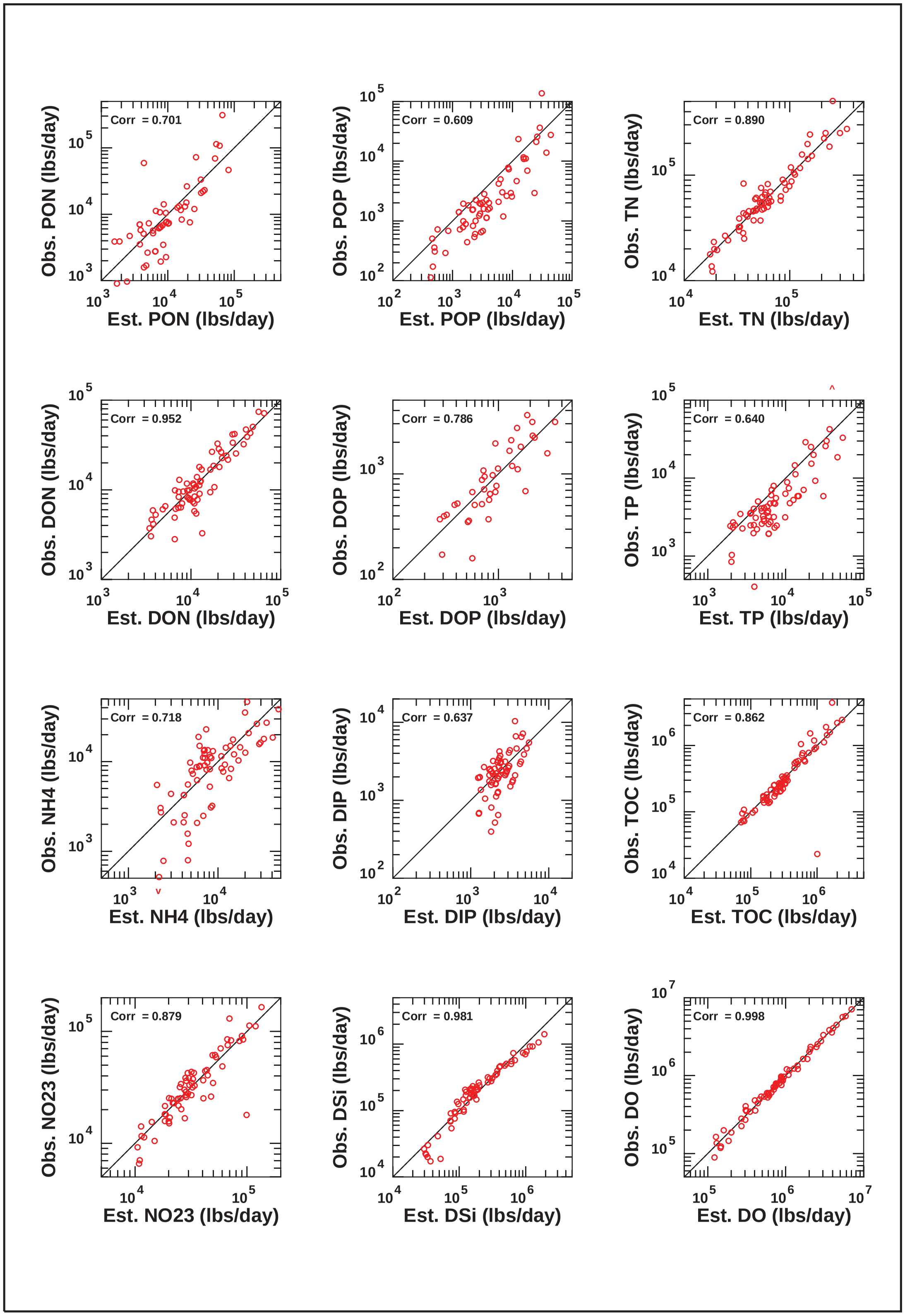
Figure 33. Observed and estimated PON and DON loadings and observed flow for the Naugatuck River fall-line

1993 - 2003 Observed Loads vs. Estimated Data



Naugatuck River at Beacon Falls, Connecticut

1993 - 2003 Observed Loads vs. Estimated Data



Connecticut River at East Haddam, Connecticut

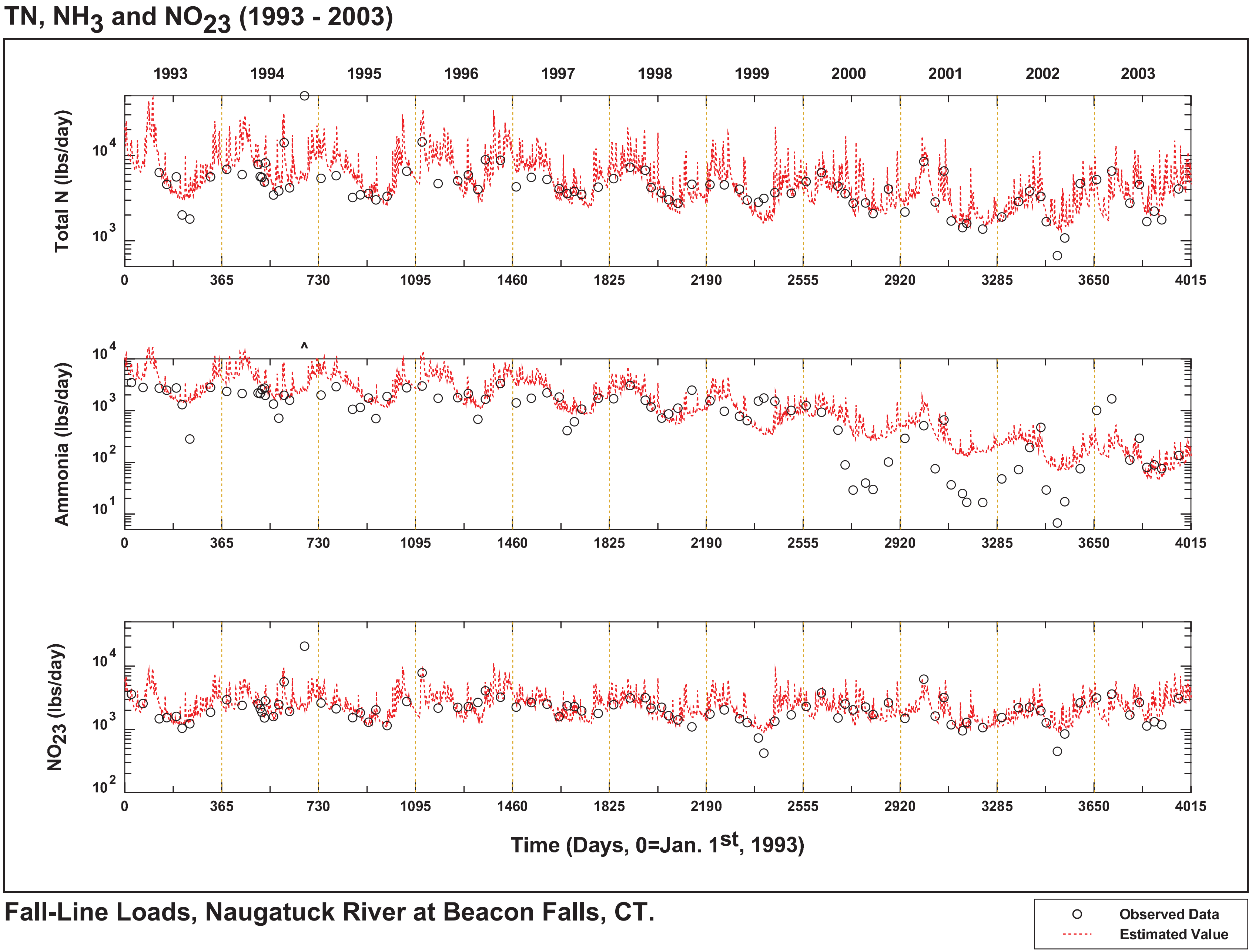


Figure 36. Observed and LOADEST estimated total nitrogen, ammonium and nitrite+nitrate concentrations for the Naugatuck River fall-line

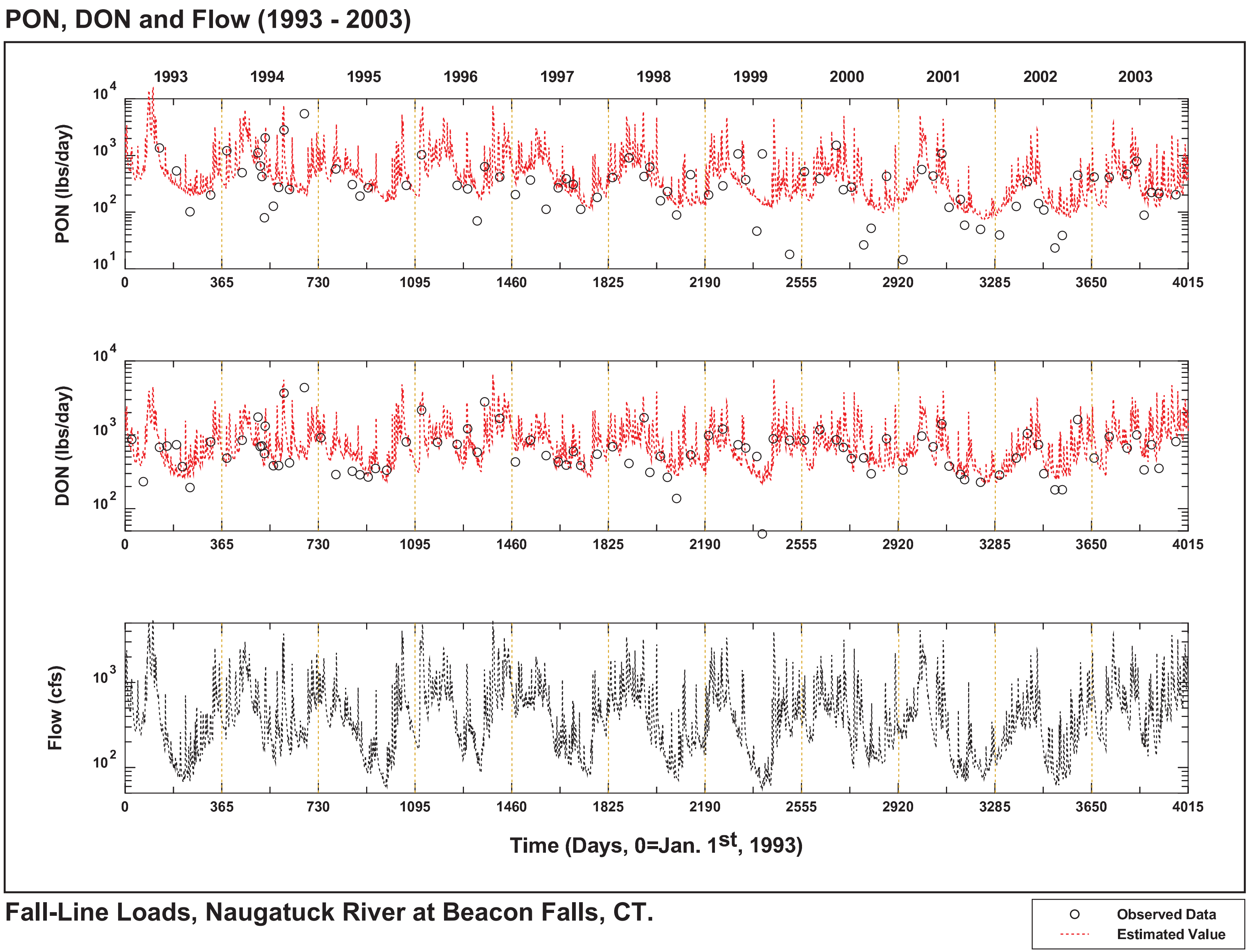


Figure 37. Observed and LOADEST estimated PON and DON loadings and observed flow for the Naugatuck River fall-line

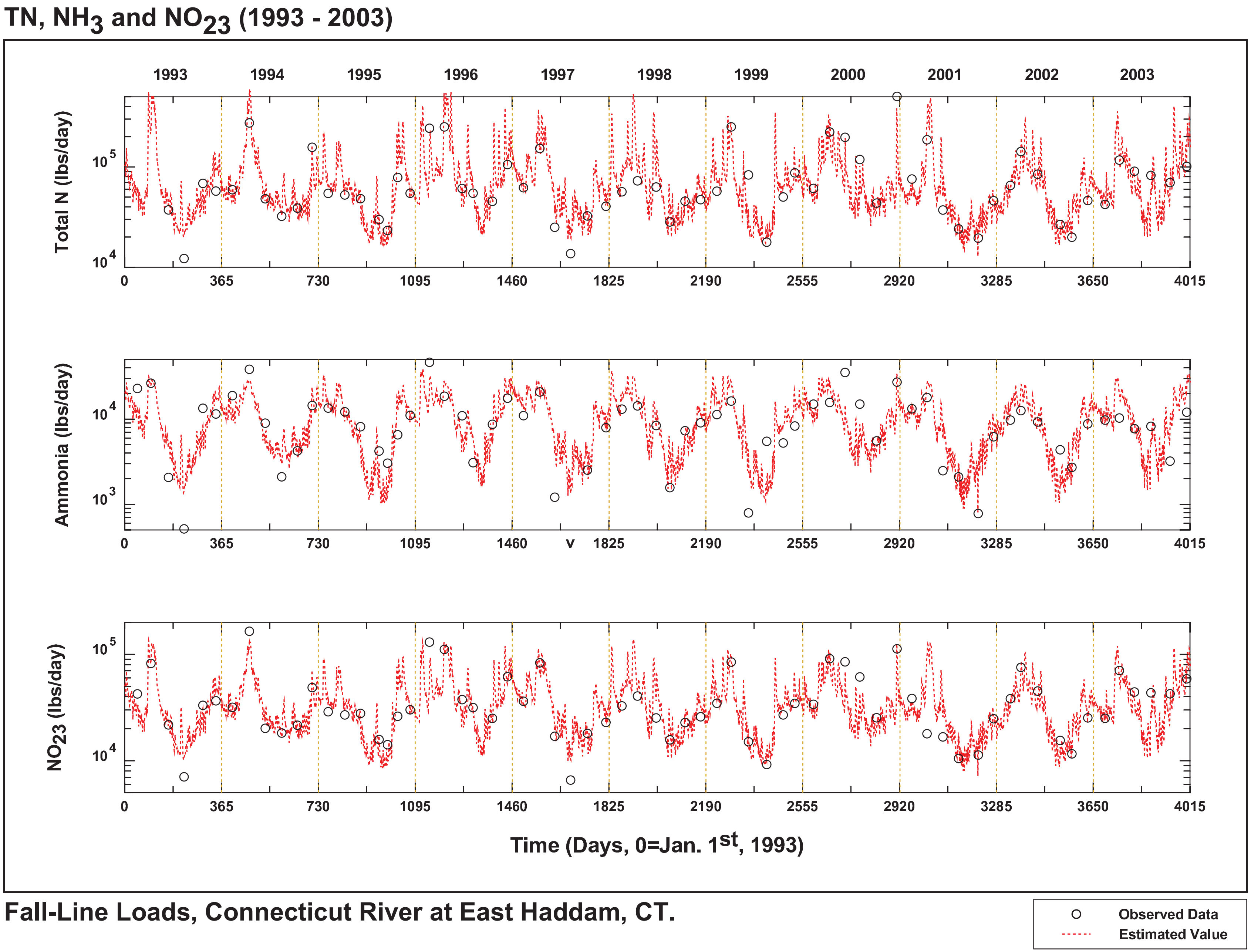


Figure 38. Observed and LOADEST estimated total nitrogen, ammonium and nitrite+nitrate loadings for the Connecticut River fall-line

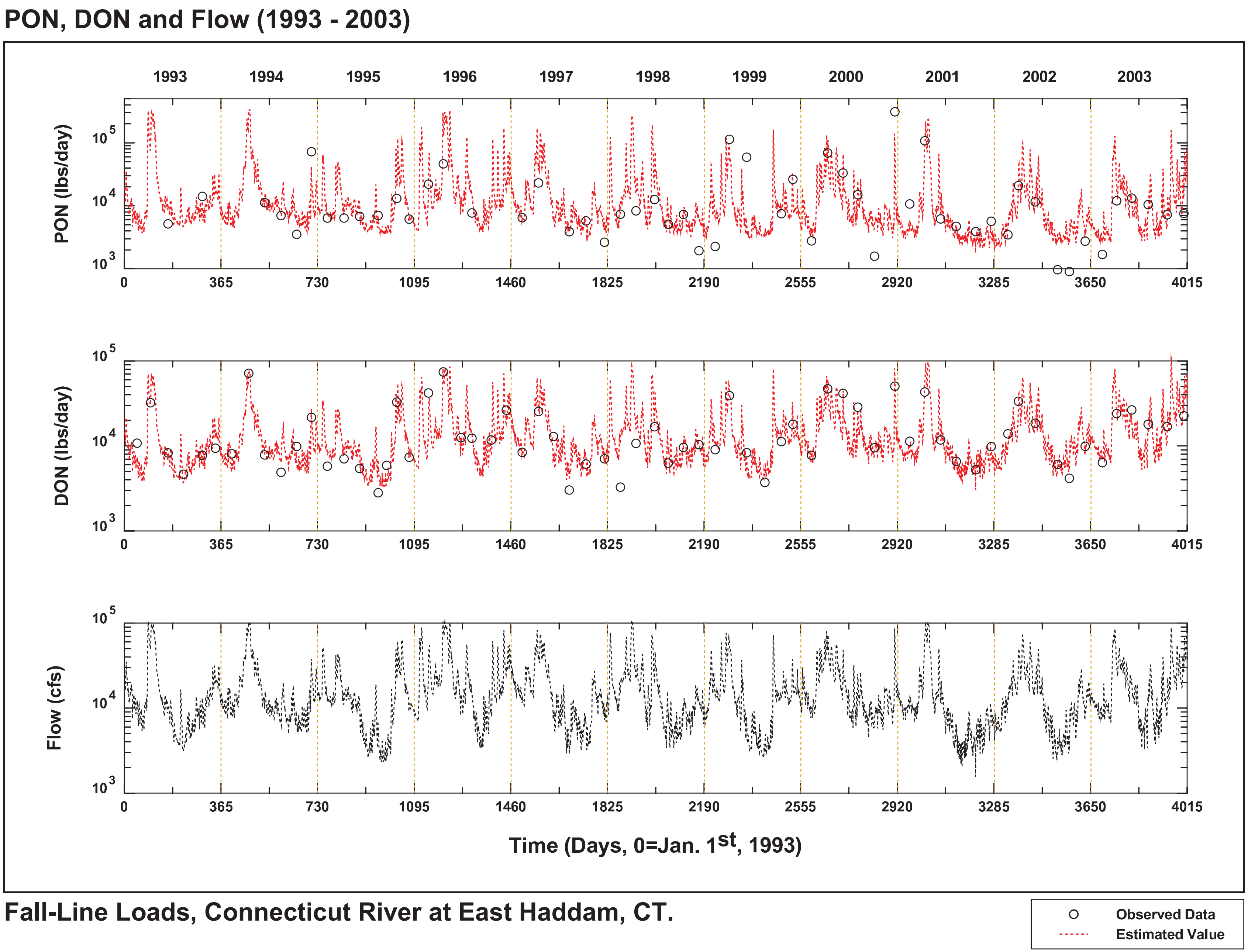
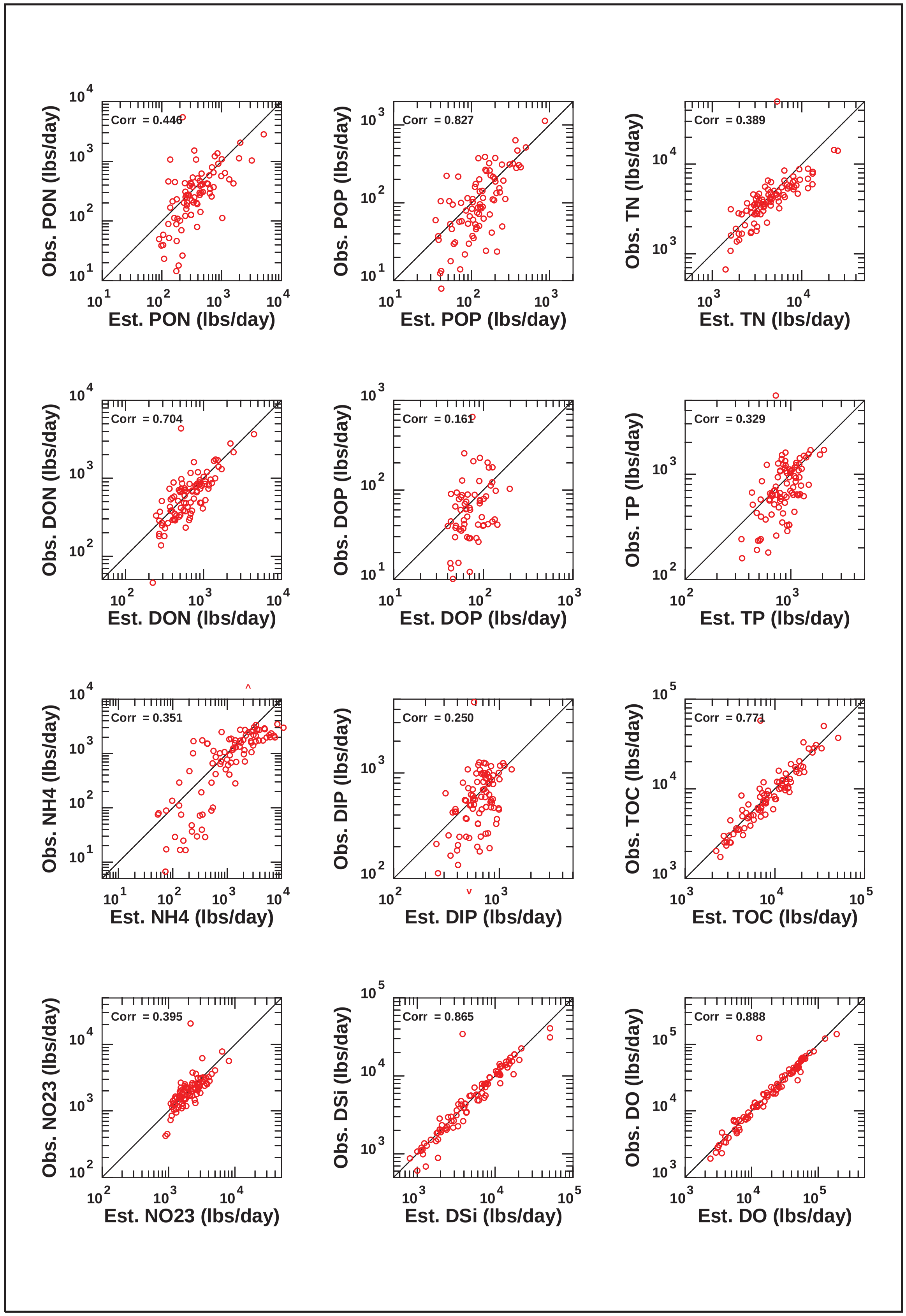


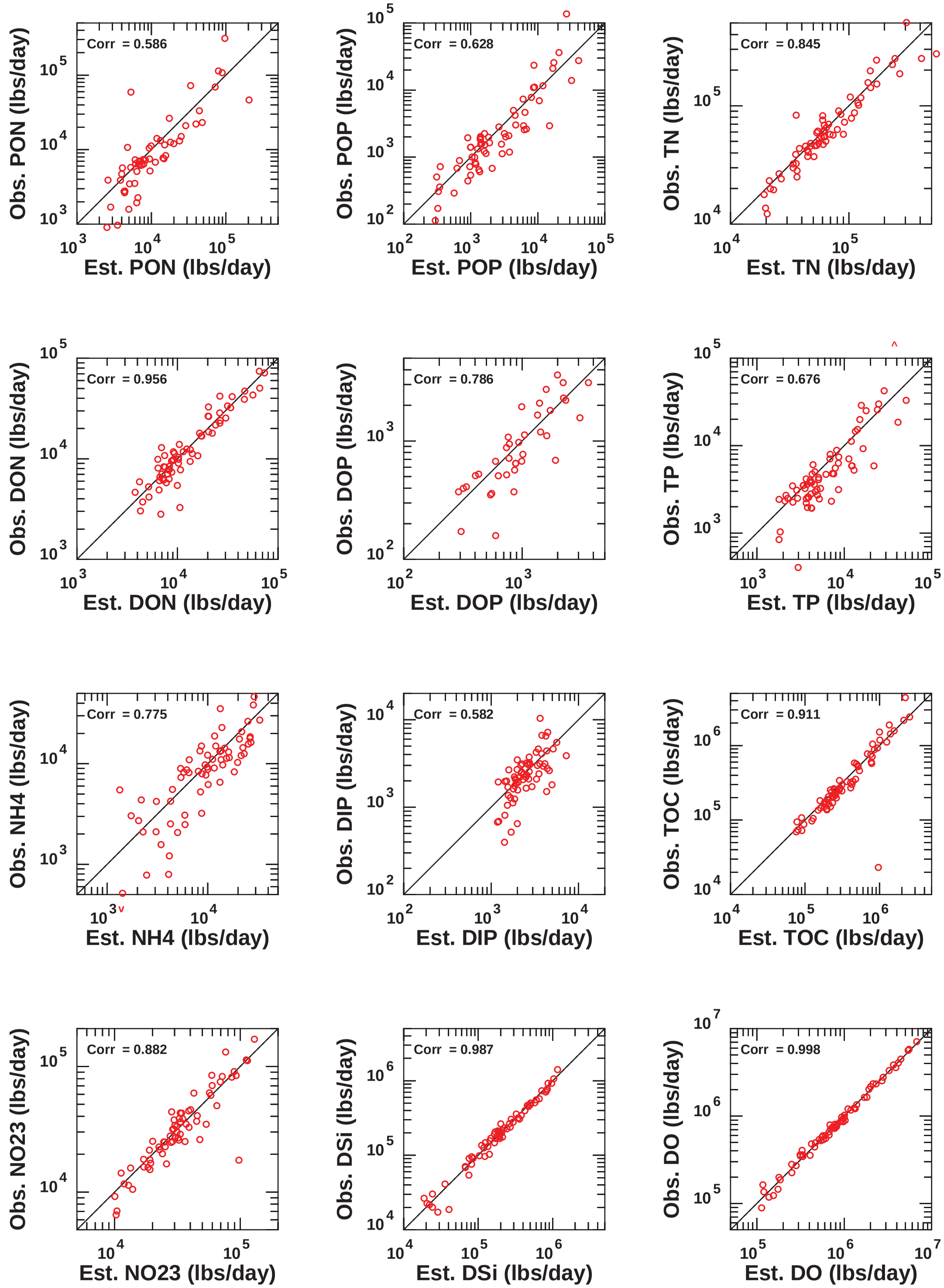
Figure 39. Observed and LOADEST estimated PON and DON loadings and observed flow for the Connecticut River fall-line

1993 - 2003 Observed Loads vs. Estimated Data

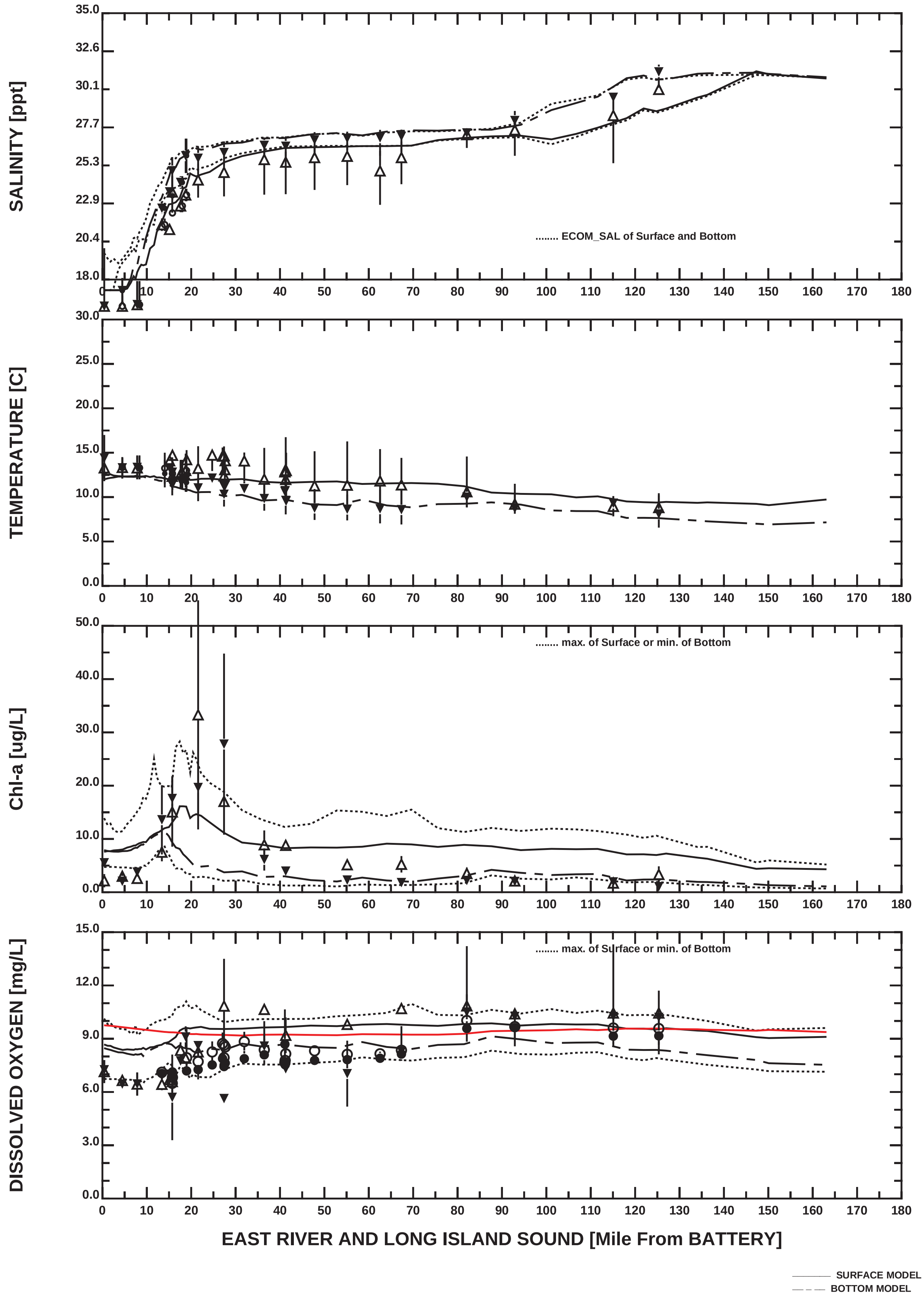


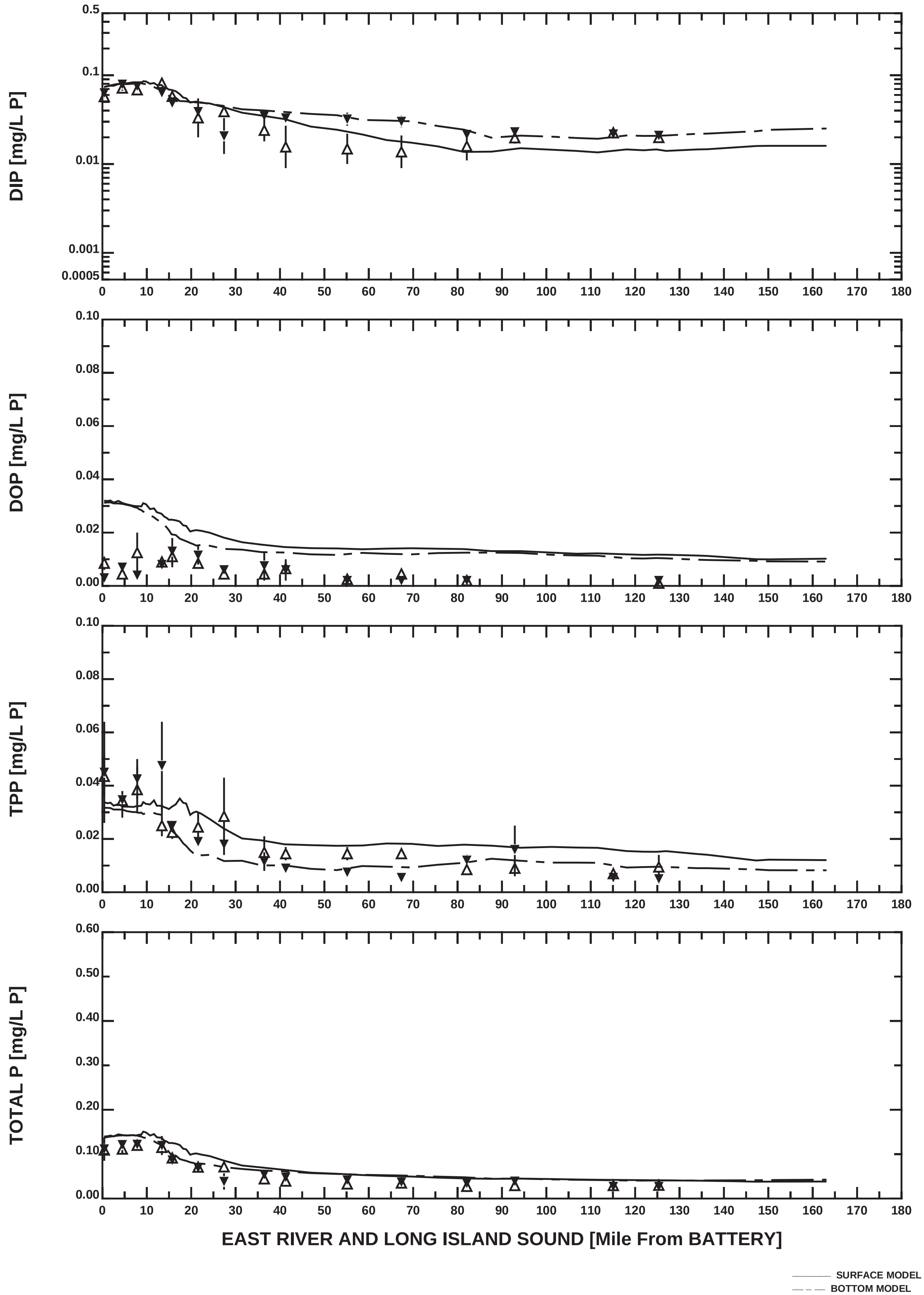
Naugatuck River at Beacon Falls, Connecticut   Loadings Estimated from LOADEST

1993 - 2003 Observed Loads vs. Estimated Data

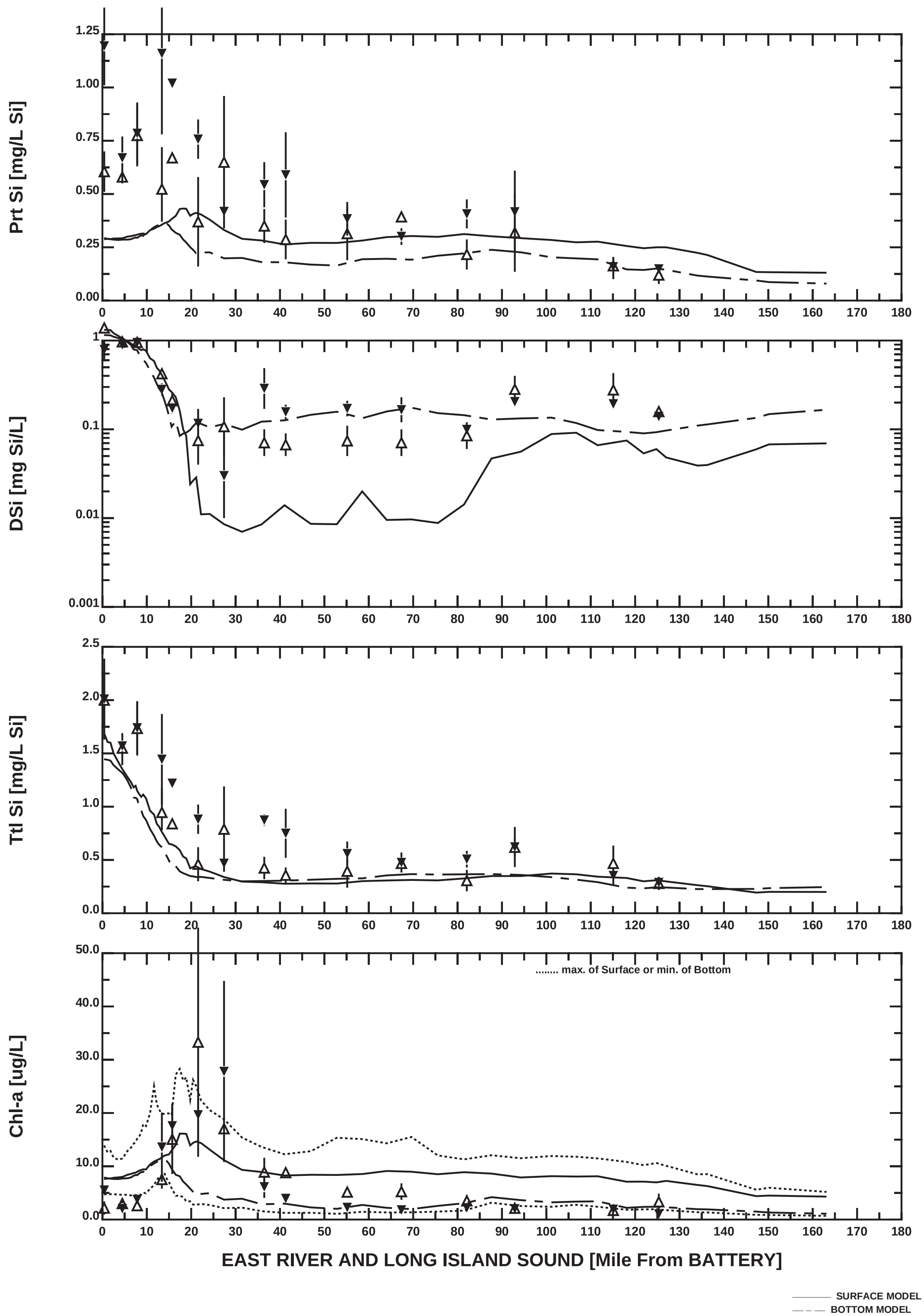


Connecticut River at East Haddam, Connecticut Loadings Estimated from LOADEST

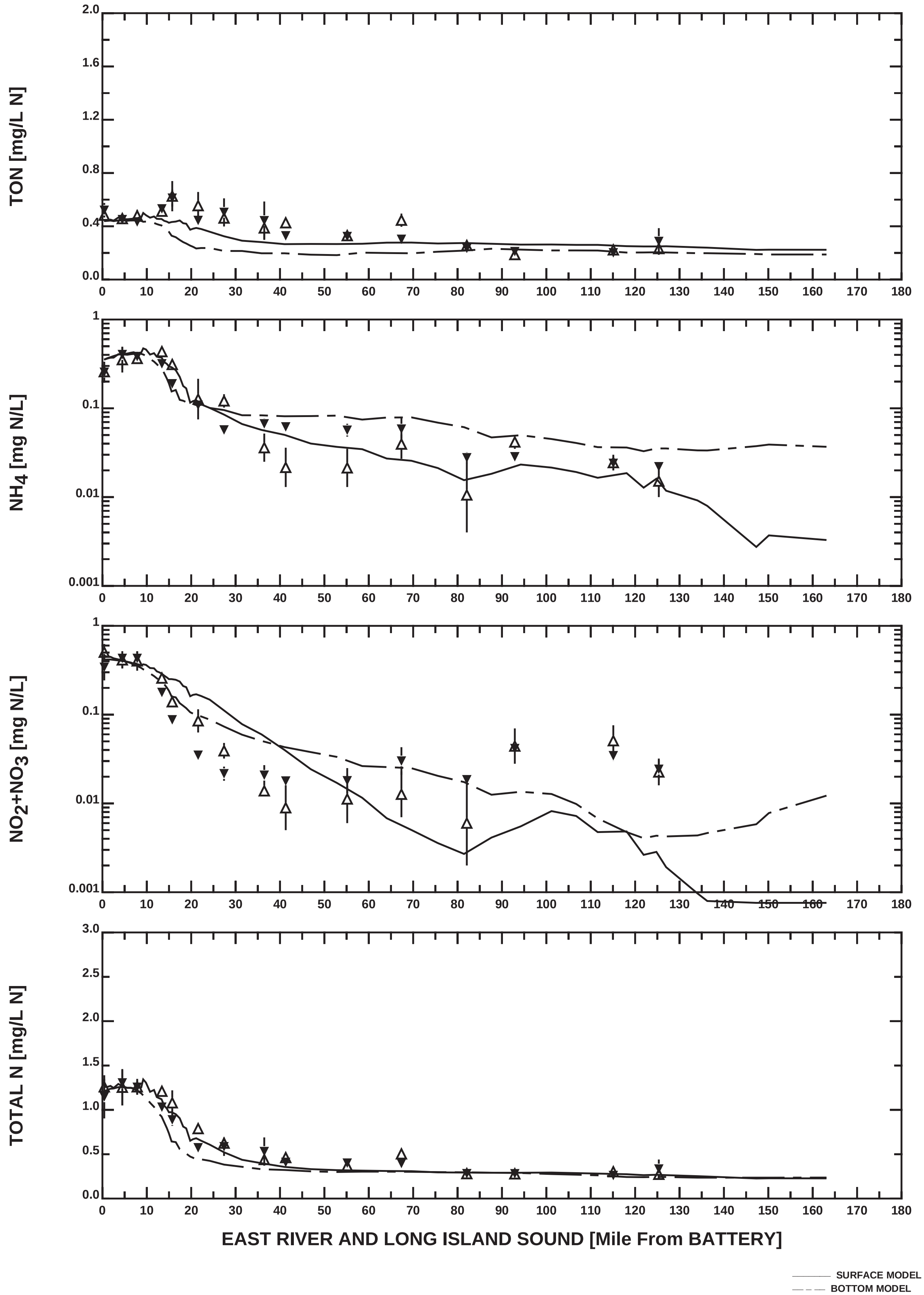

SALINITY [ppt]
ECOM_SAL of Surface and Bottom
TEMPERATURE [C]
Chl-a [ug/L]
max. of Surface or min. of Bottom
DISSOLVED OXYGEN [mg/L]
max. of Surface or min. of Bottom
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL

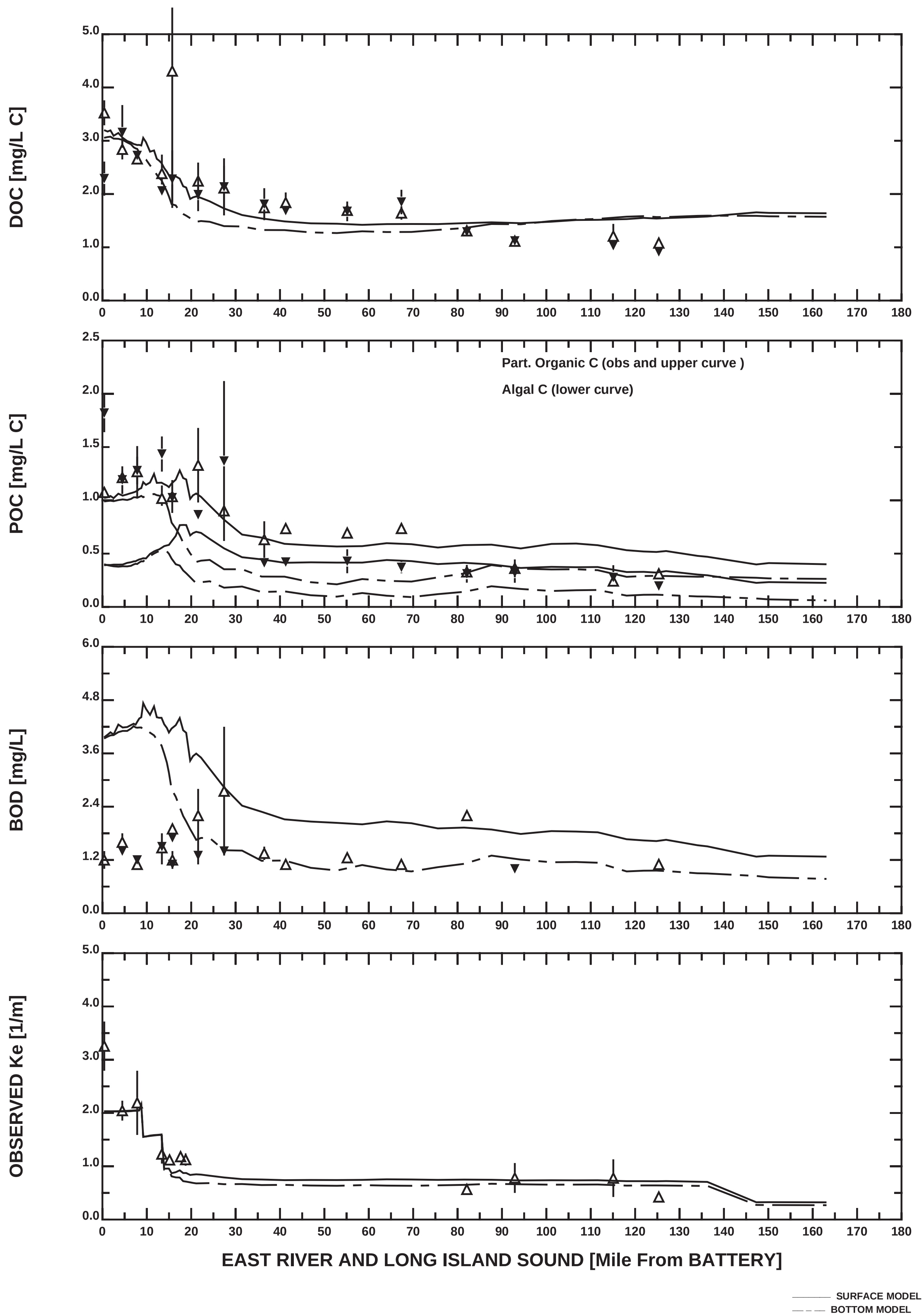


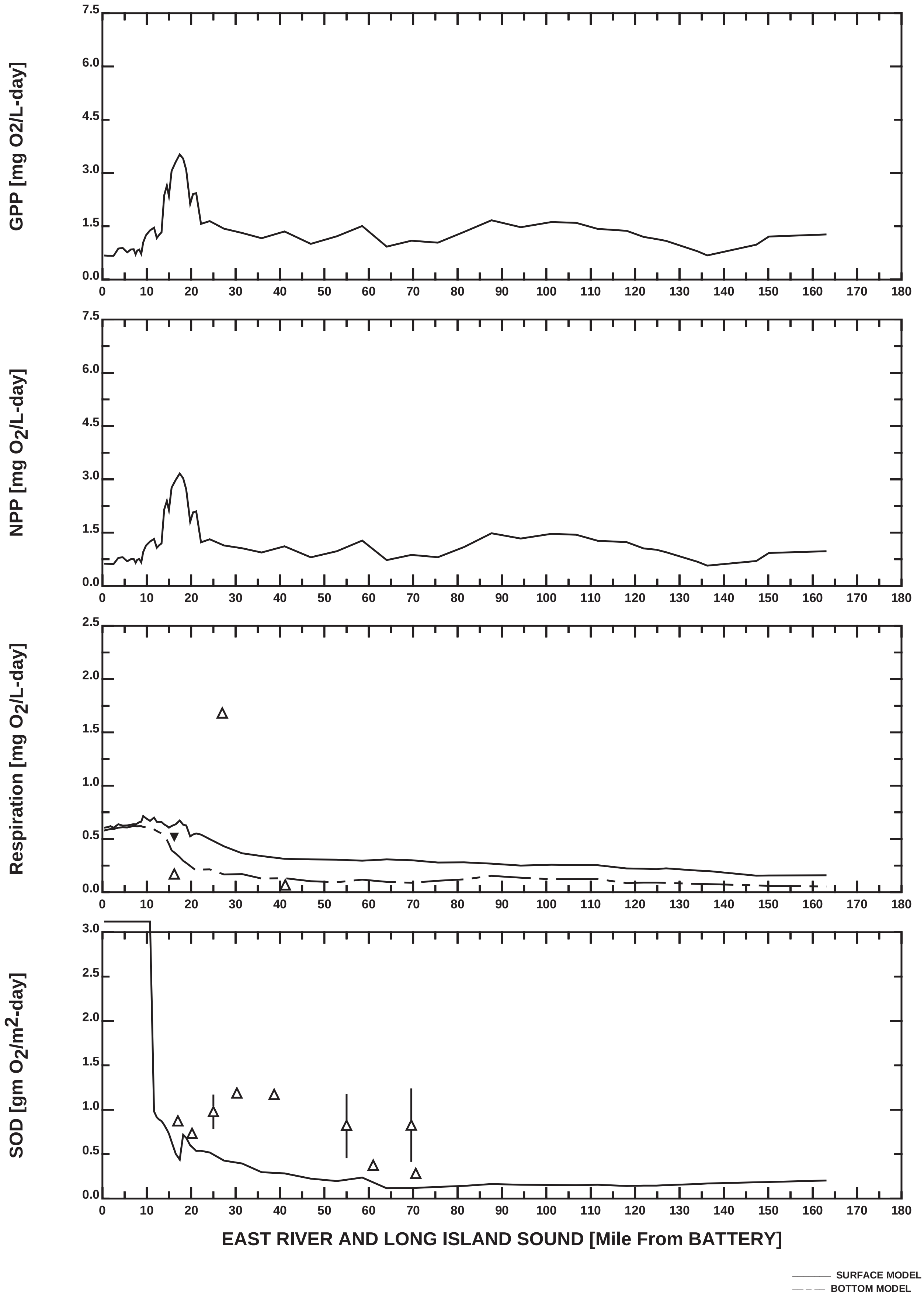
DIP [mg/L P]
DOP [mg/L P]
TPP [mg/L P]
TOTAL P [mg/L P]
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL



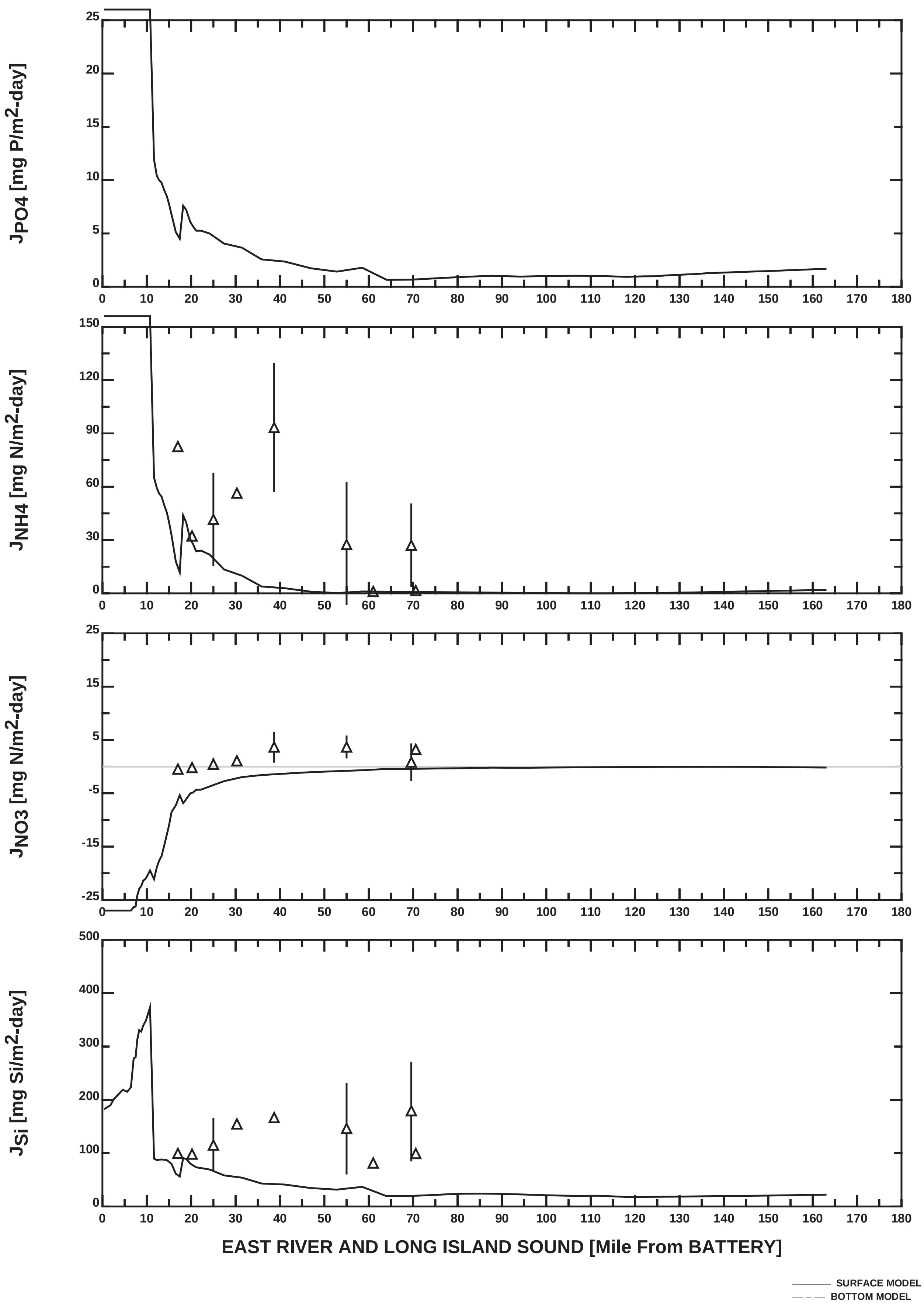
Prt Si [mg/L Si]
DSi [mg Si/L]
Ttl Si [mg/L Si]
Chl-a [ug/L]
.... max. of Surface or min. of Bottom
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL

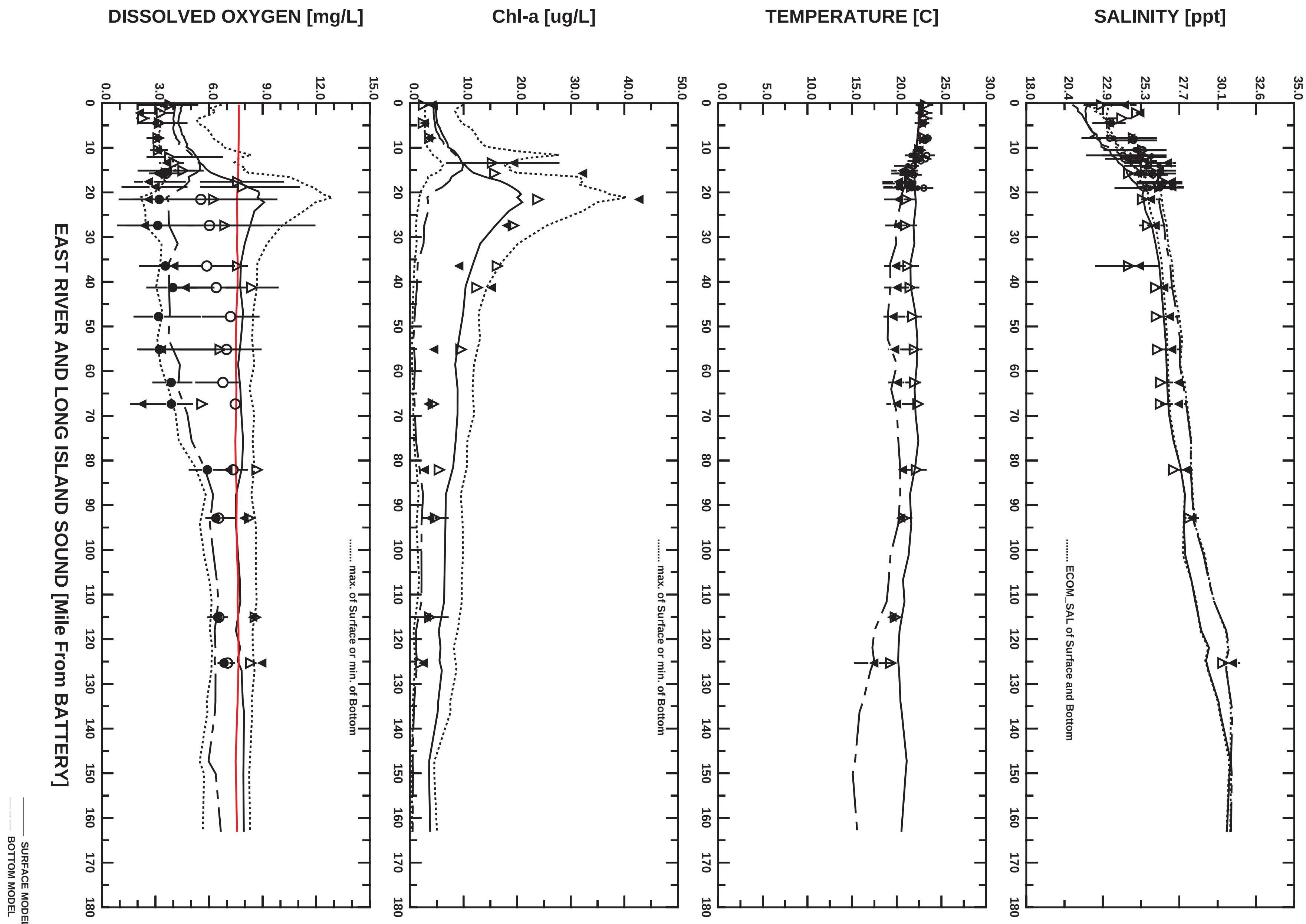

TON [mg/L N]
NH4 [mg N/L]
NO2+NO3 [mg N/L]
TOTAL N [mg/L N]
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL

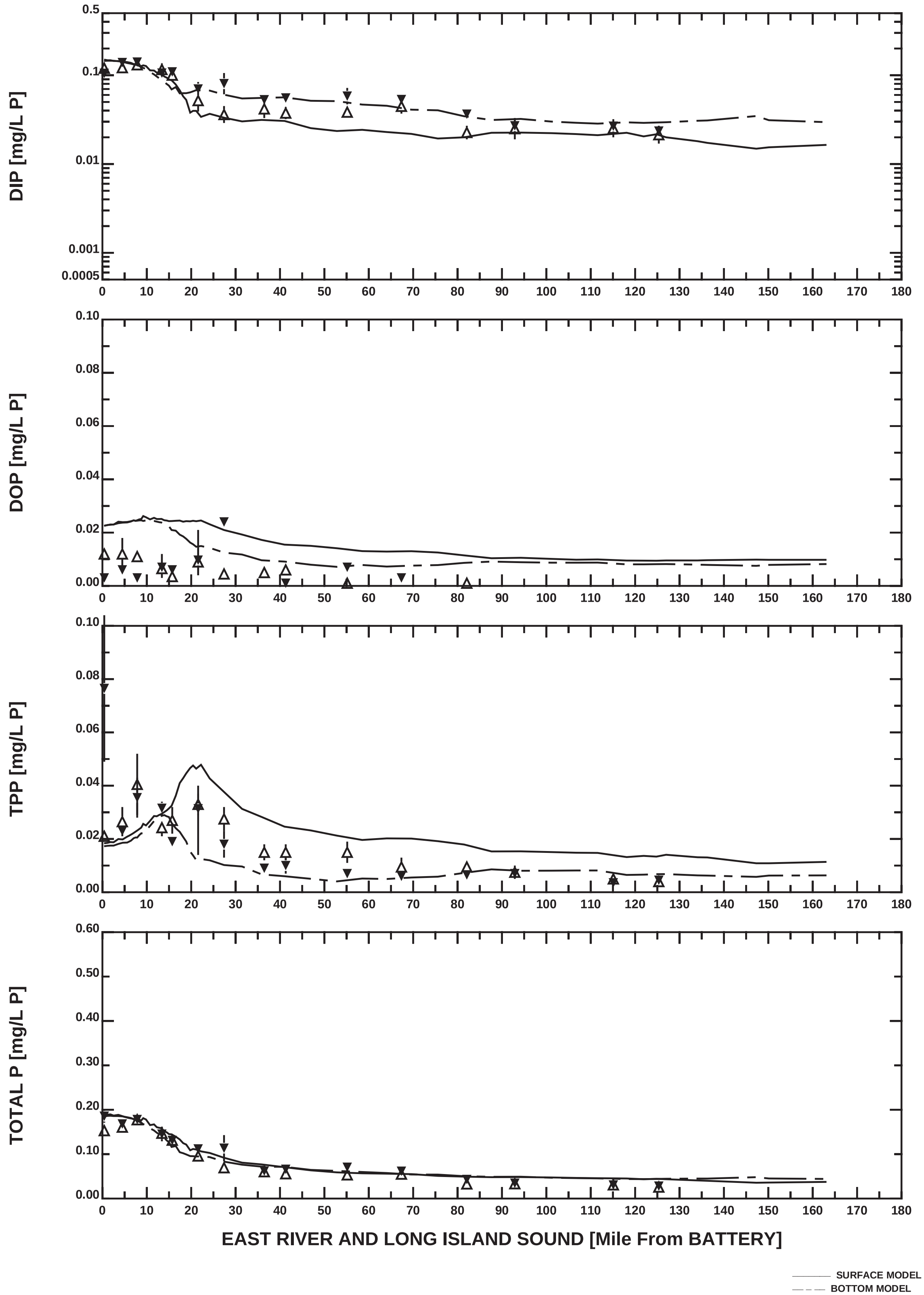

DOC [mg/L C]
POC [mg/L C]
Part. Organic C (obs and upper curve)
Algal C (lower curve)
BOD [mg/L]
OBSERVED Ke [1/m]
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL



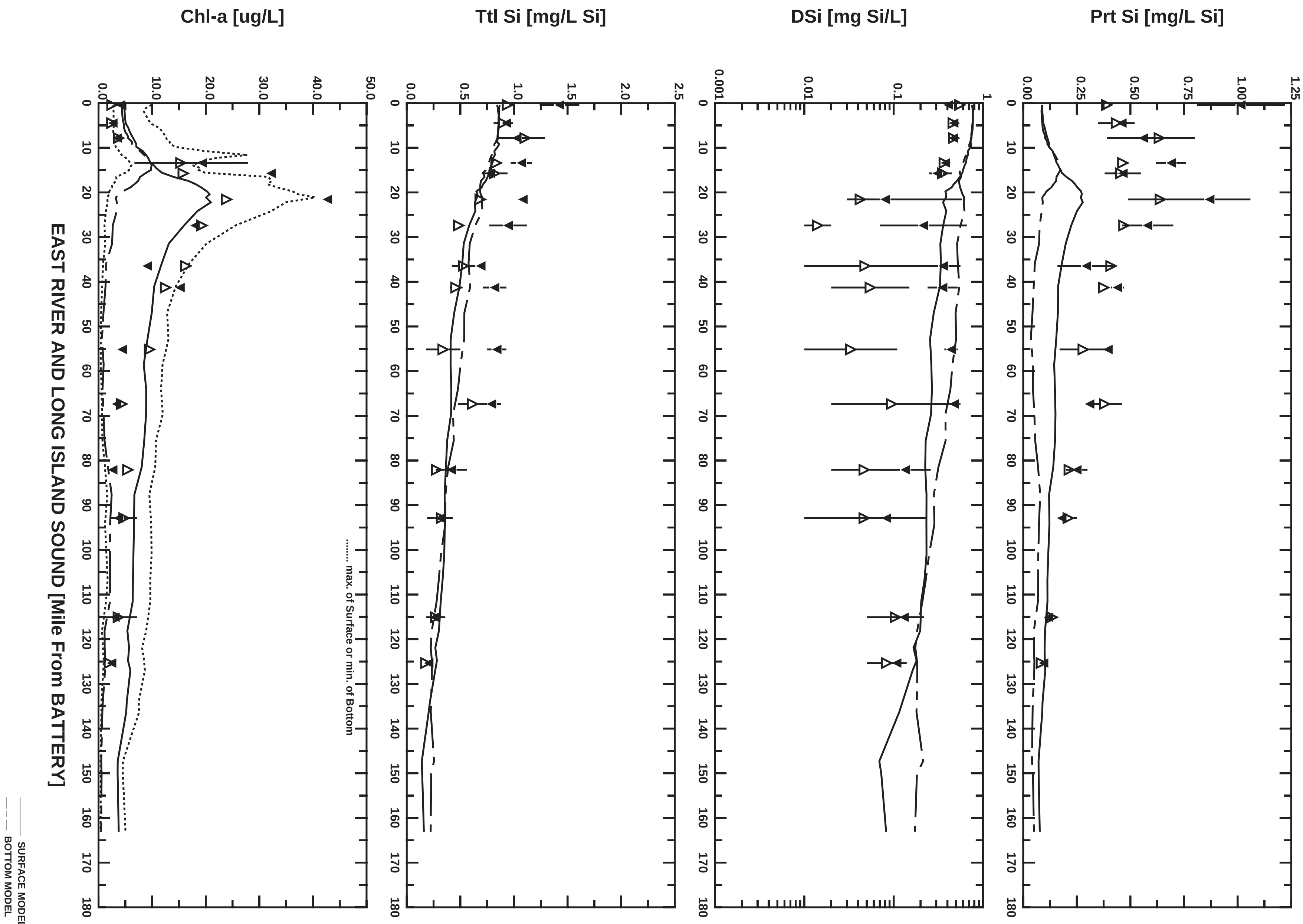
GPP [mg O2/L-day]
NPP [mg O2/L-day]
Respiration [mg O2/L-day]
SOD [gm O2/m2-day]
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL


J_PO4 [mg P/m2-day]
J_NH4 [mg N/m2-day]
J_NO3 [mg N/m2-day]
J_Si [mg Si/m2-day]
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL

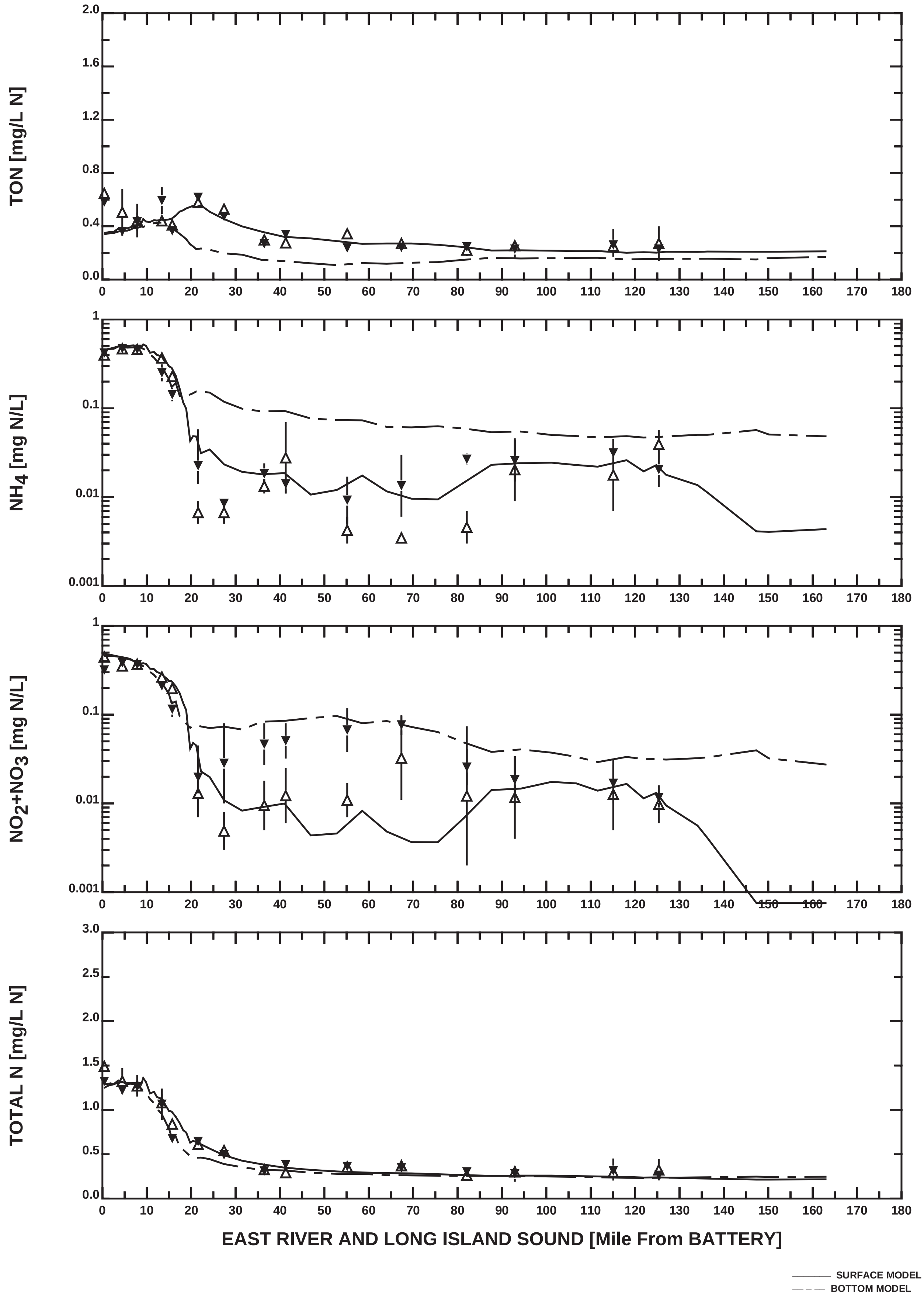

DISSOLVED OXYGEN [mg/L]
Chl-a [ug/L]
TEMPERATURE [C]
SALINITY [ppt]
...... max. of Surface or min. of Bottom
...... max. of Surface or min. of Bottom
...... ECOM_SAL of Surface and Bottom
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL



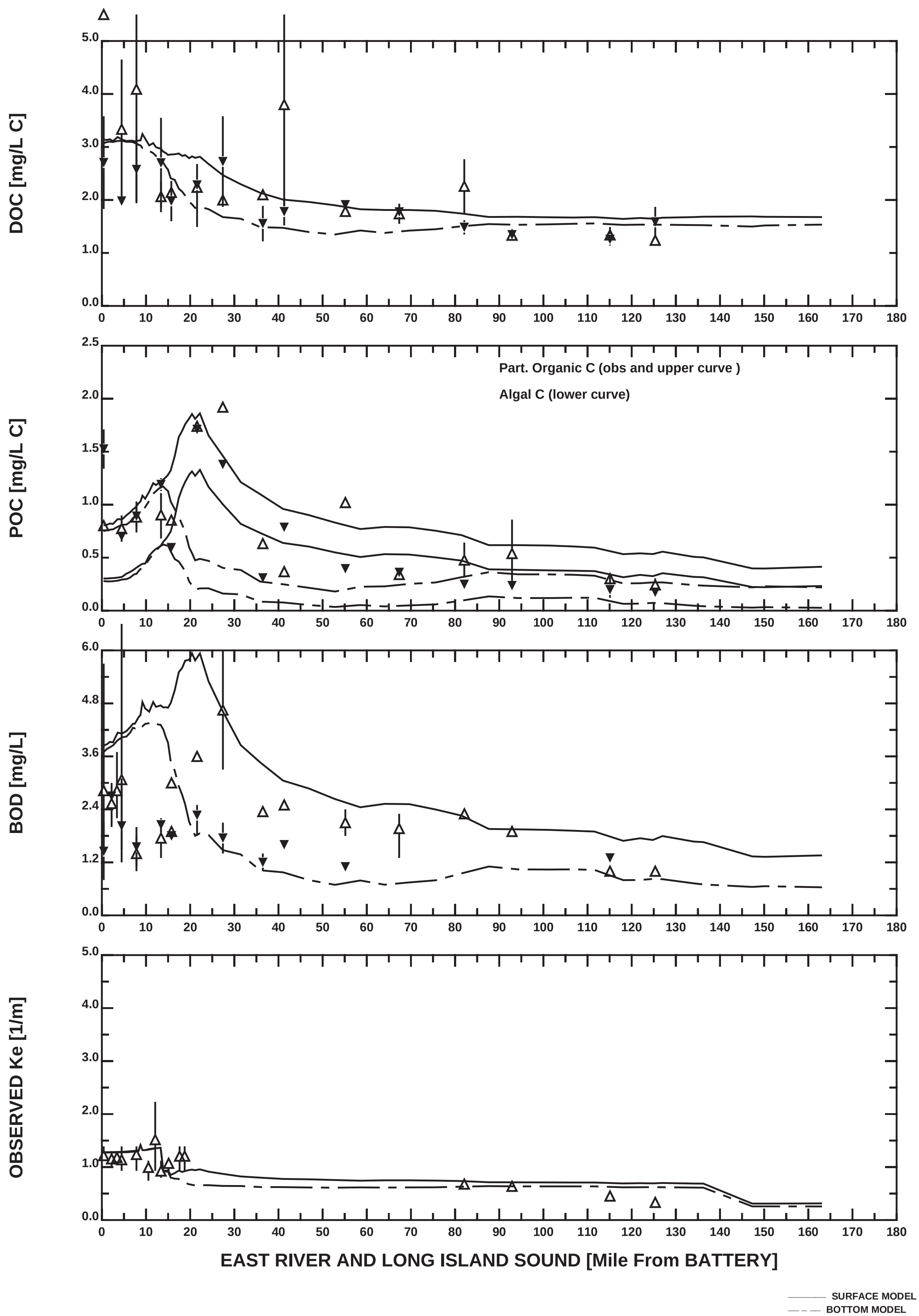
DIP [mg/L P]
DOP [mg/L P]
TPP [mg/L P]
TOTAL P [mg/L P]
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL

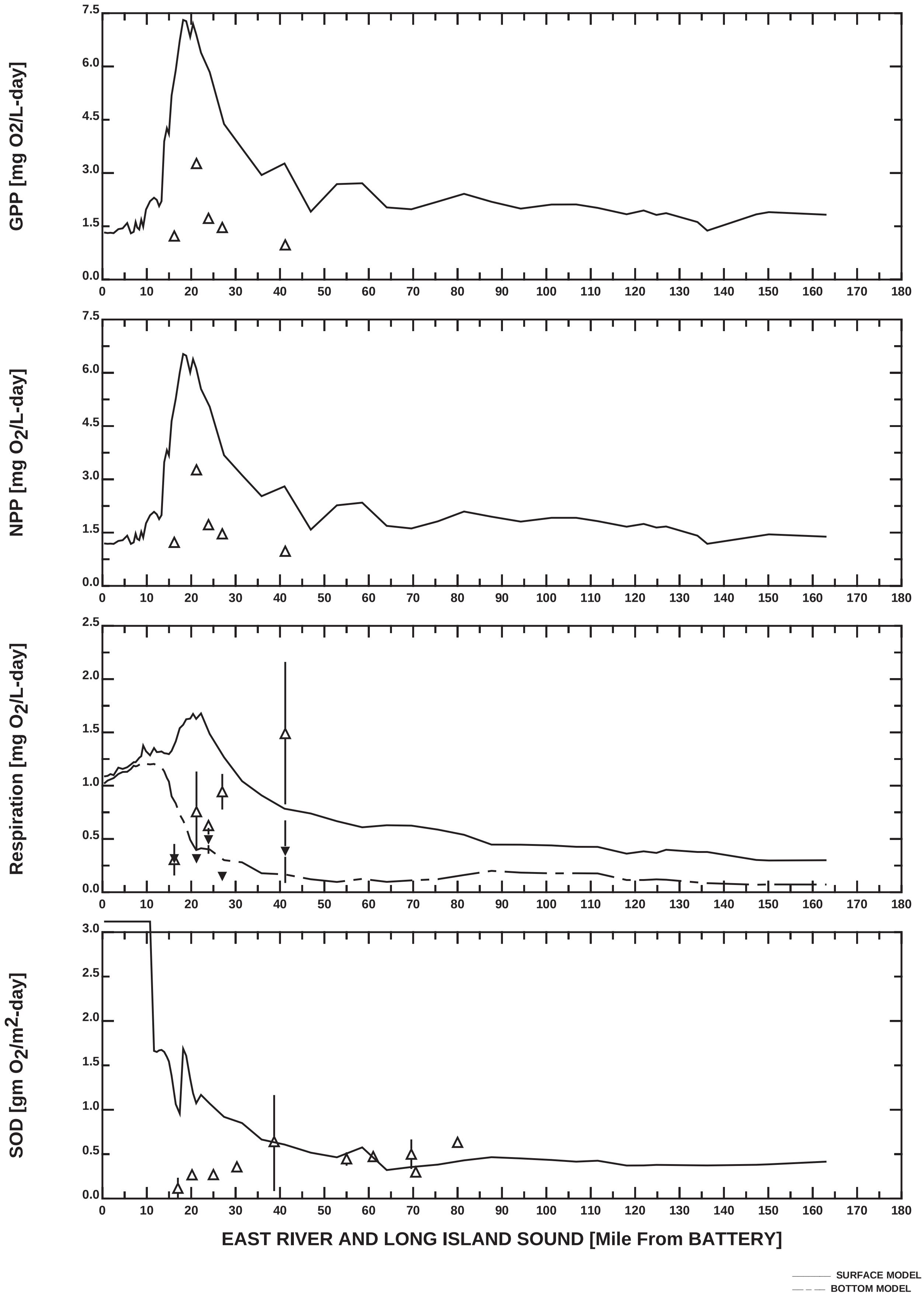


Prt Si [mg/L Si]
DSi [mg Si/L]
Ttl Si [mg/L Si]
Chl-a [ug/L]
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
max. of Surface or min. of Bottom
SURFACE MODEL
BOTTOM MODEL

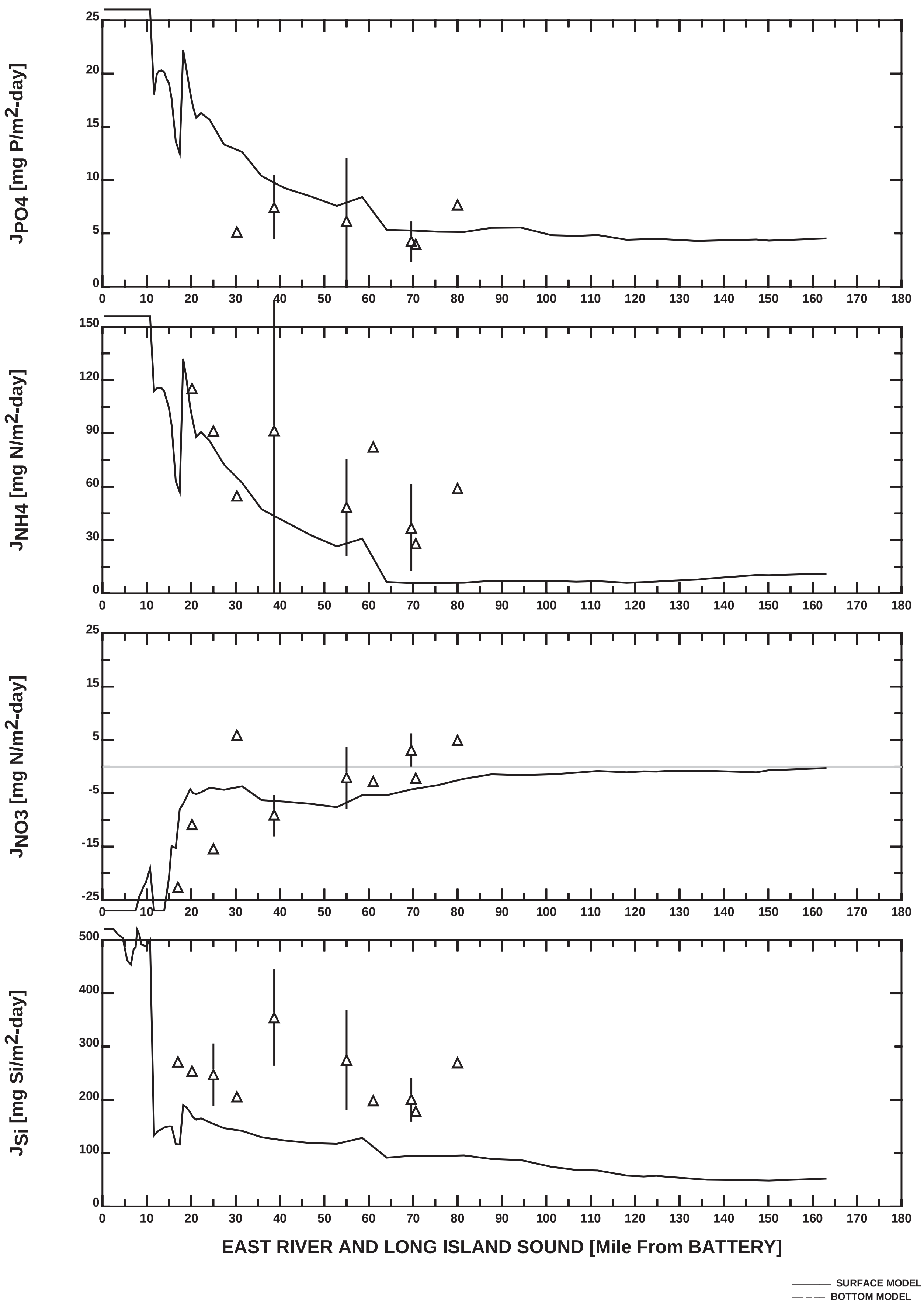


TON [mg/L N]
NH4 [mg N/L]
NO2+NO3 [mg N/L]
TOTAL N [mg/L N]
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL

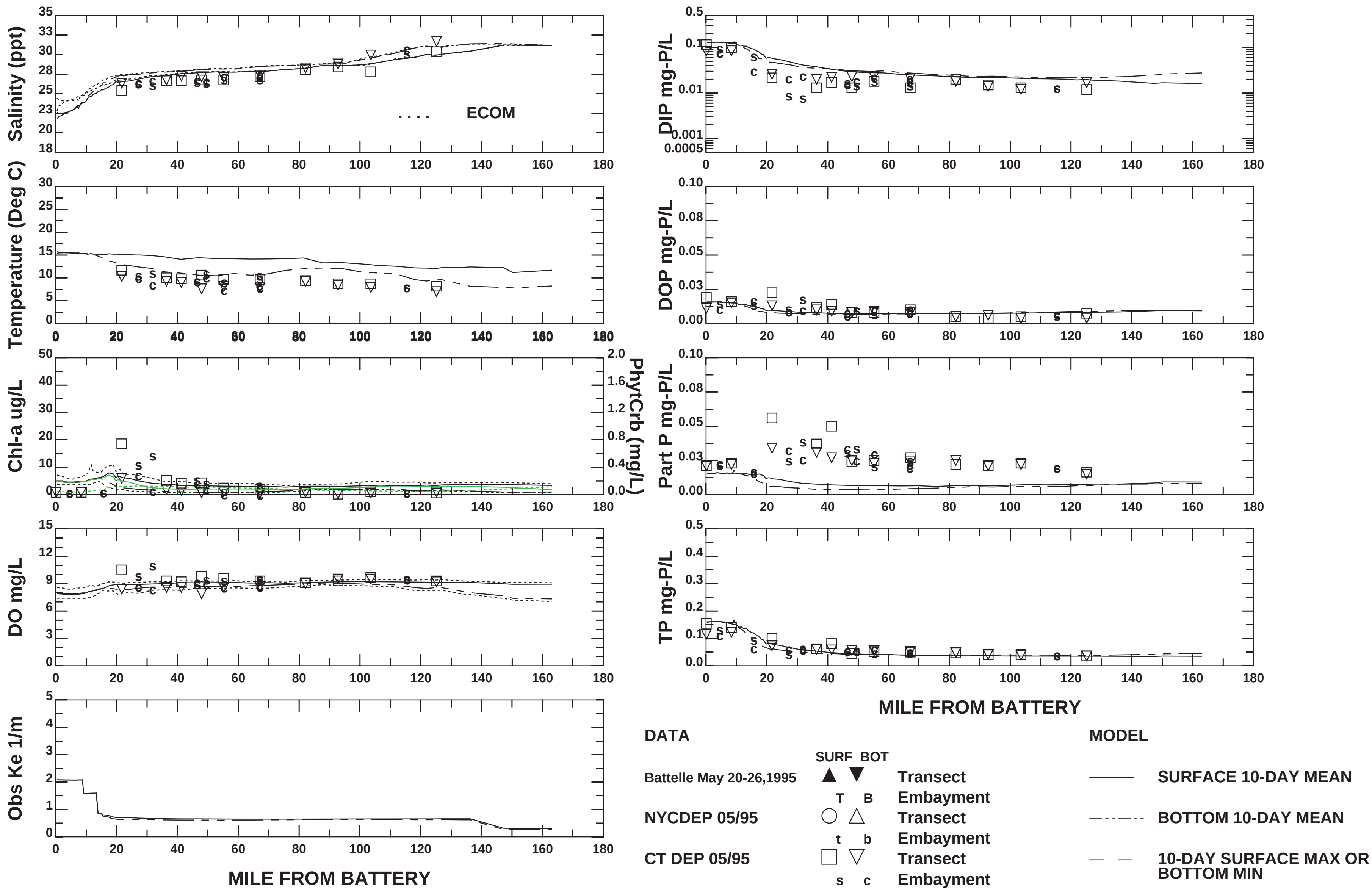

DOC [mg/L C]
POC [mg/L C]
Part. Organic C (obs and upper curve)
Algal C (lower curve)
BOD [mg/L]
OBSERVED Ke [1/m]
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL



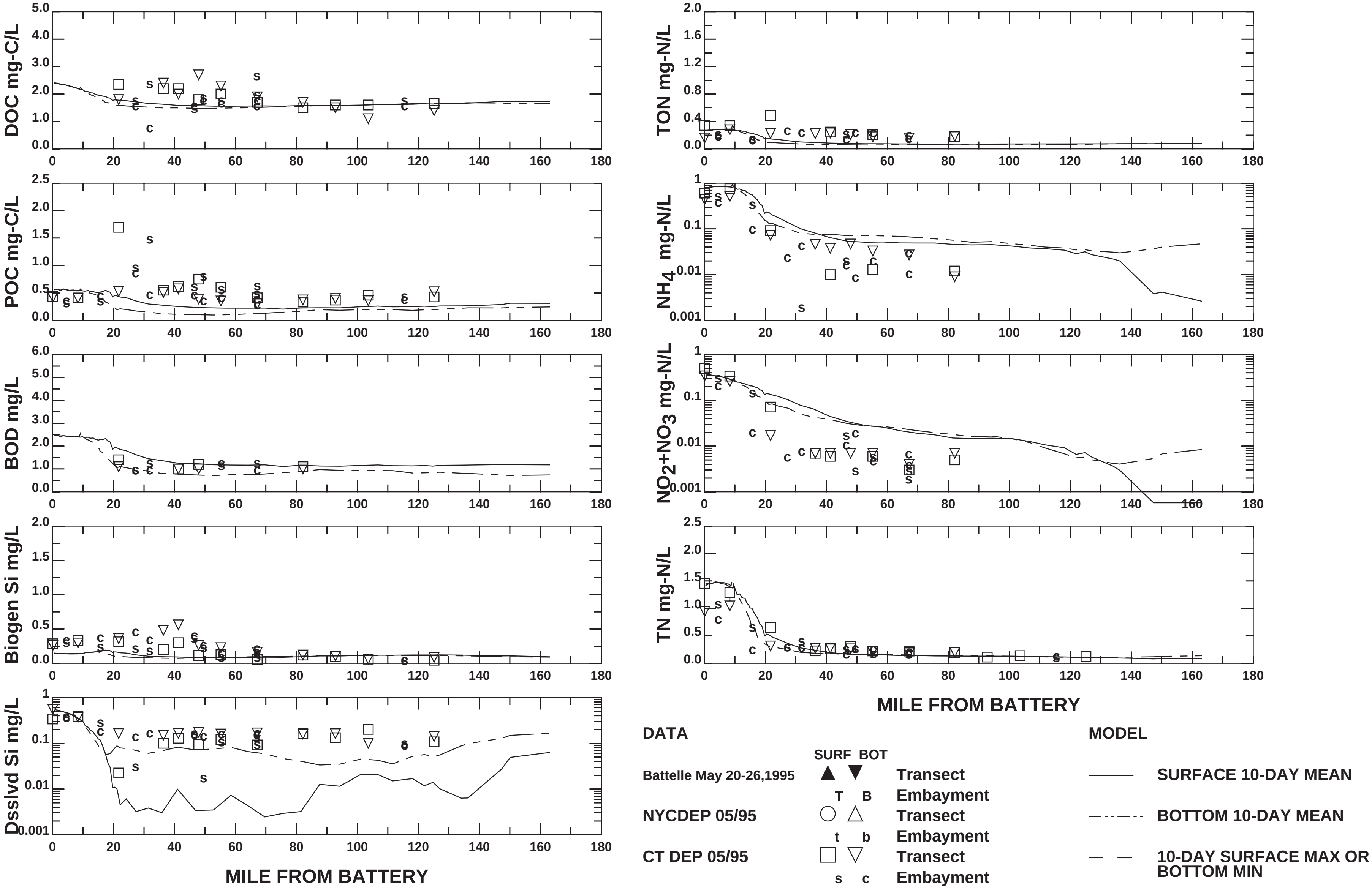
GPP [mg O2/L-day]
NPP [mg O2/L-day]
Respiration [mg O2/L-day]
SOD [gm O2/m2-day]
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL



J_{PO4} [mg P/m^2-day]
J_{NH4} [mg N/m^2-day]
J_{NO3} [mg N/m^2-day]
J_{Si} [mg Si/m^2-day]
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL



EAST RIVER AND LONG ISLAND SOUND



EAST RIVER AND LONG ISLAND SOUND

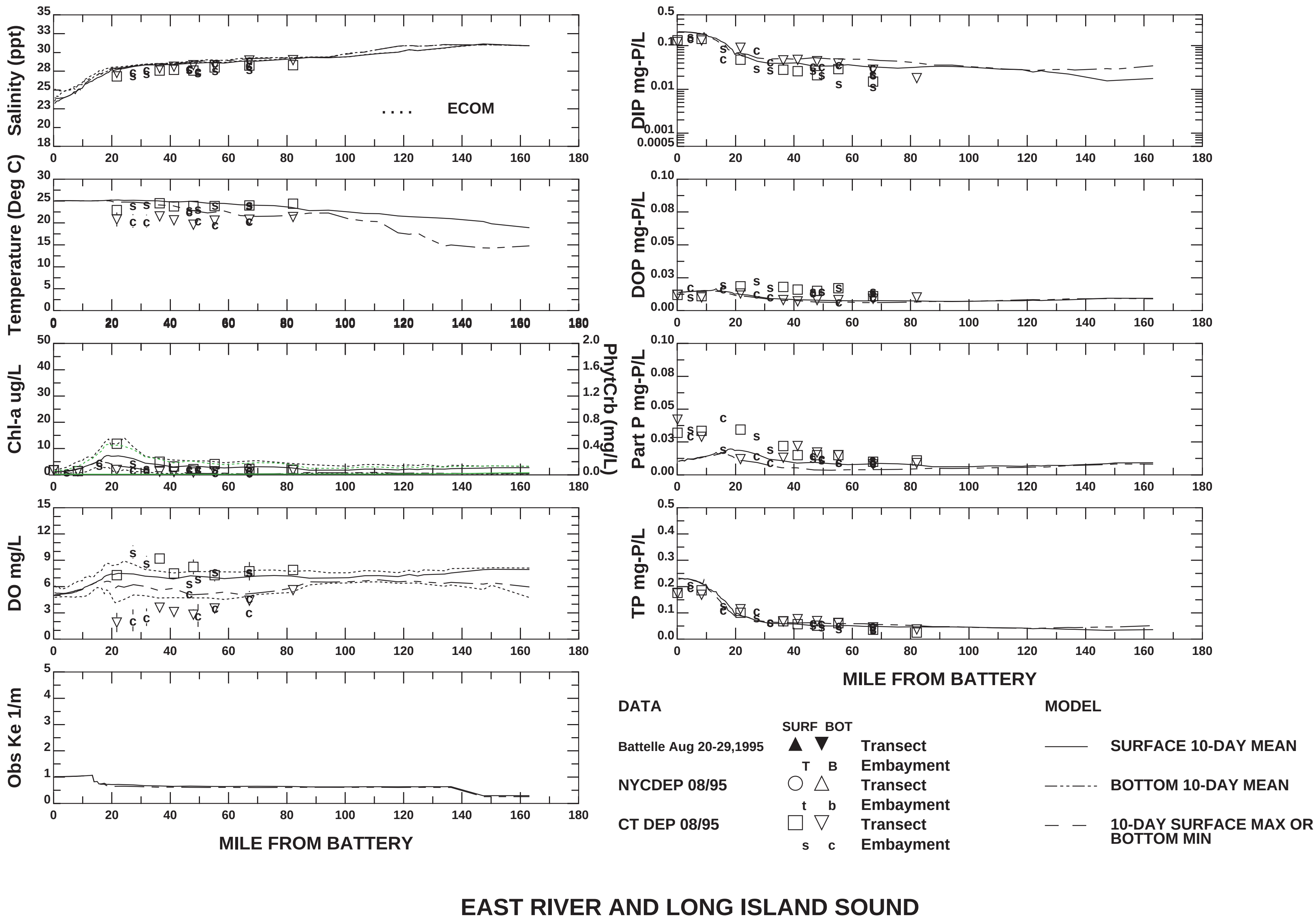
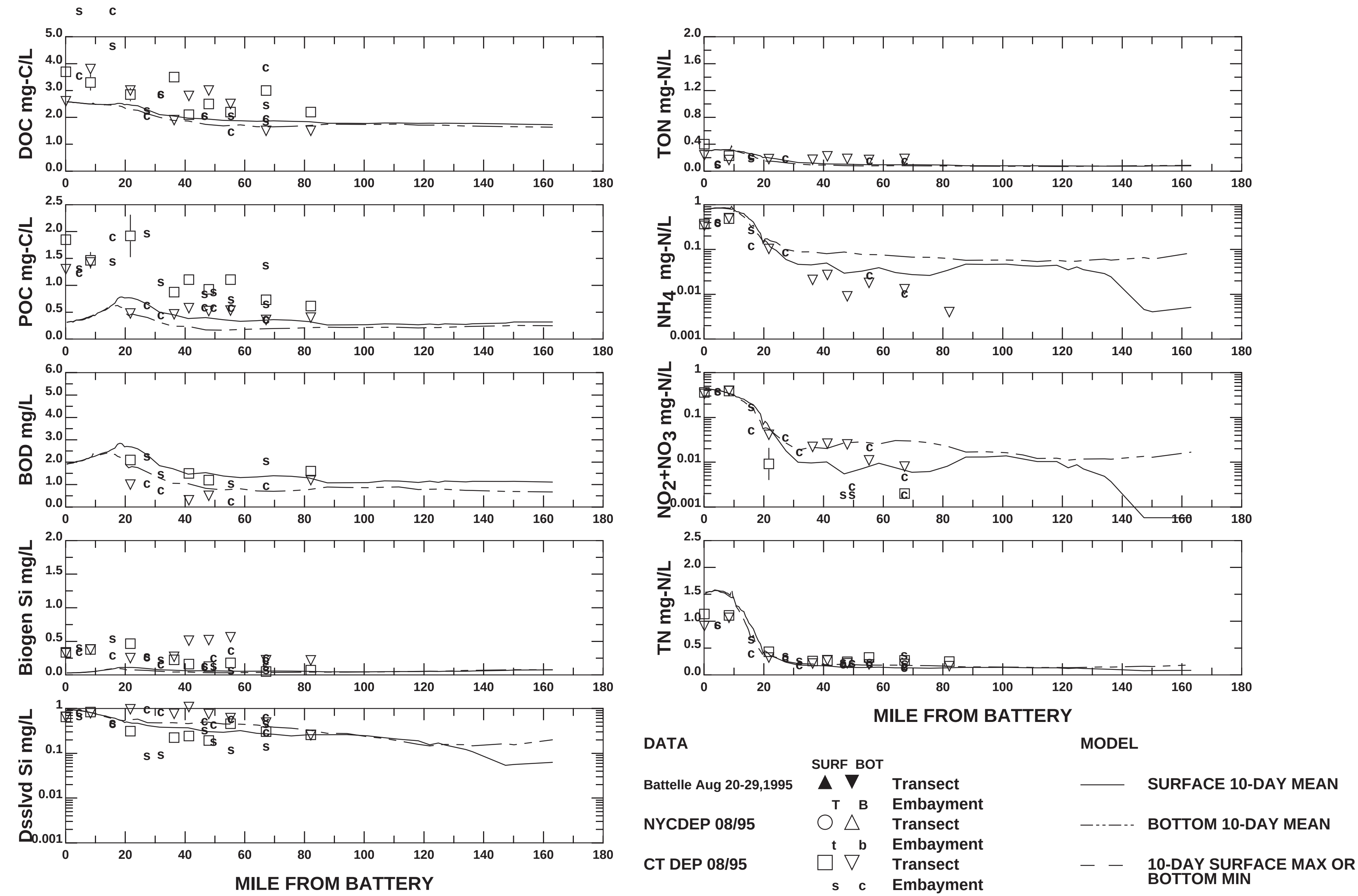


Figure 58. Calibration Results for August 1995 - Part 1



EAST RIVER AND LONG ISLAND SOUND

Figure 59. Calibration Results for August 1995 - Part 2

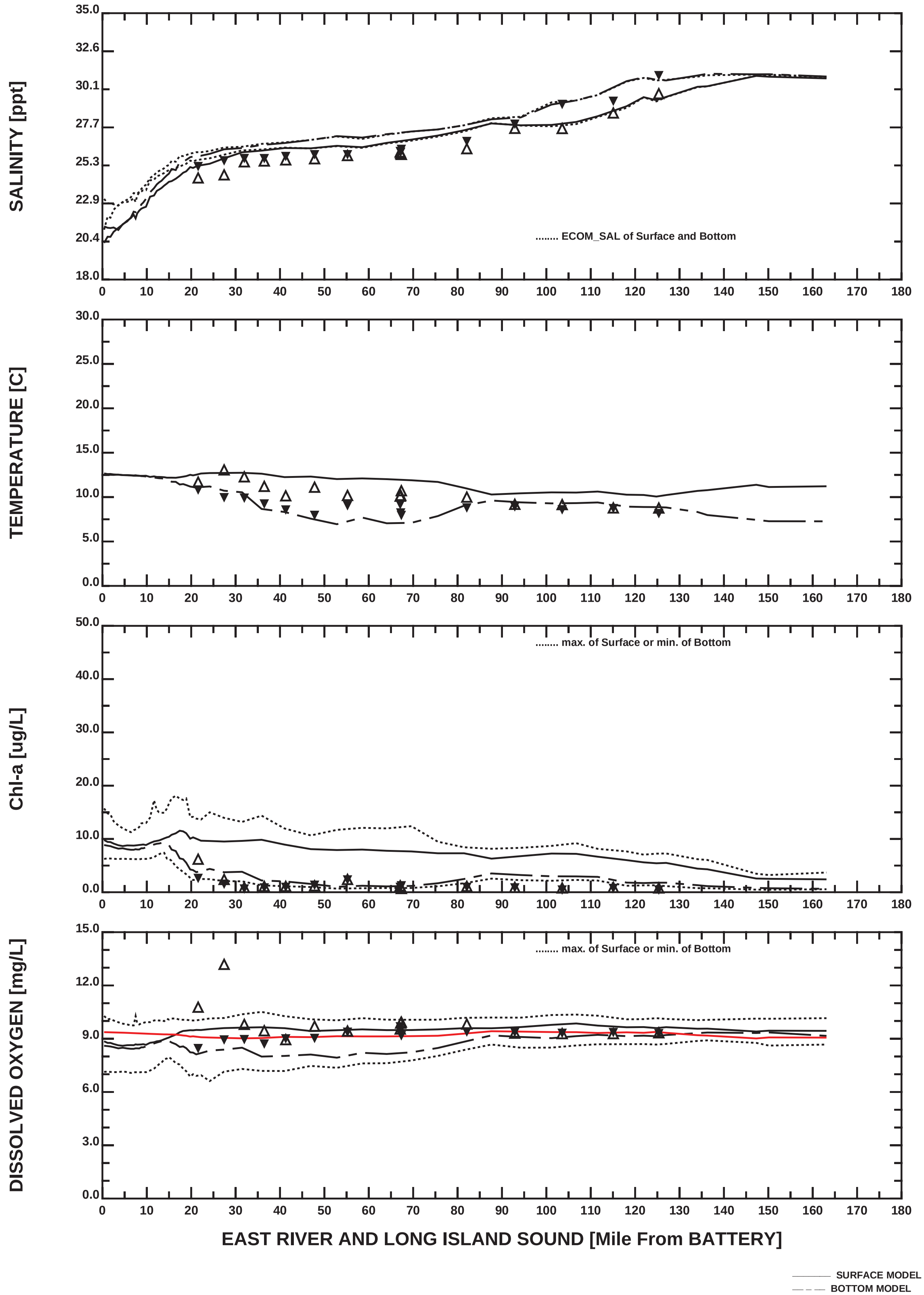


Figure 60. Salinity, temperature, chlorophyll and DO calibration results for May 1999

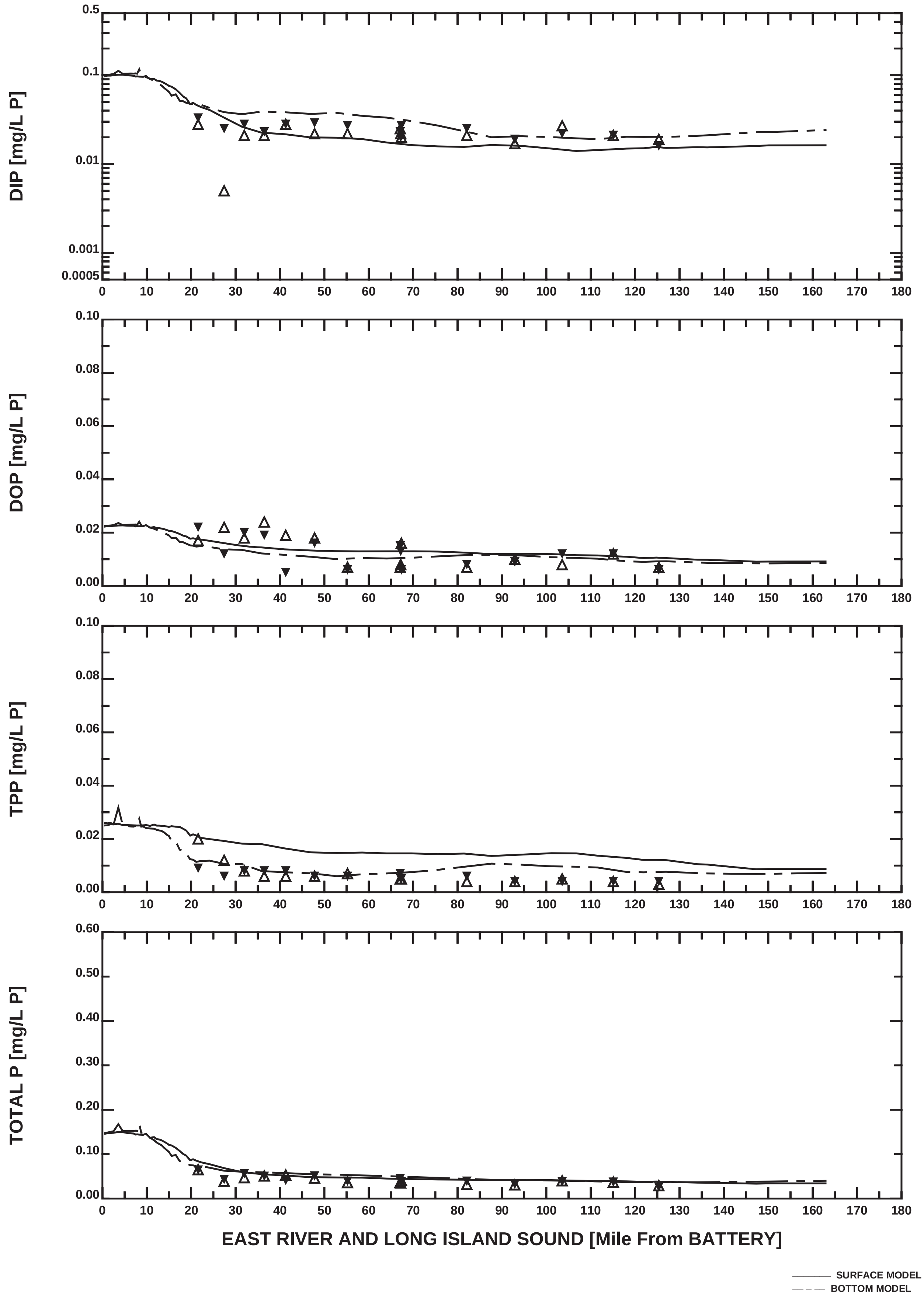


Figure 61. Dissolved inorganic phosphorus, DOP, total particulate phosphorus and total phosphorus calibration results for May 1999

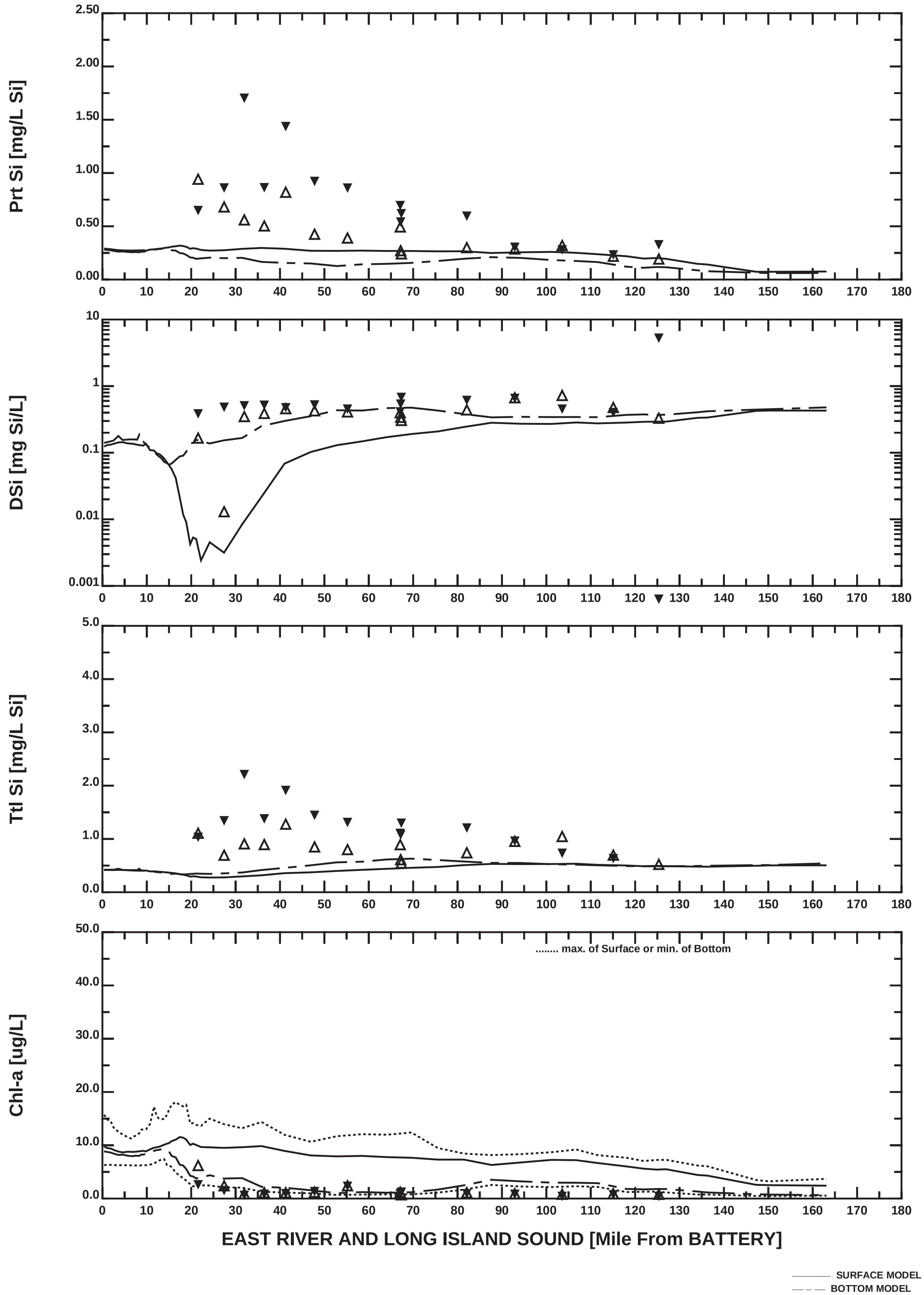


Figure 62. Particulate silica, Si, total silica, and chlorophyll calibration results for May 1999

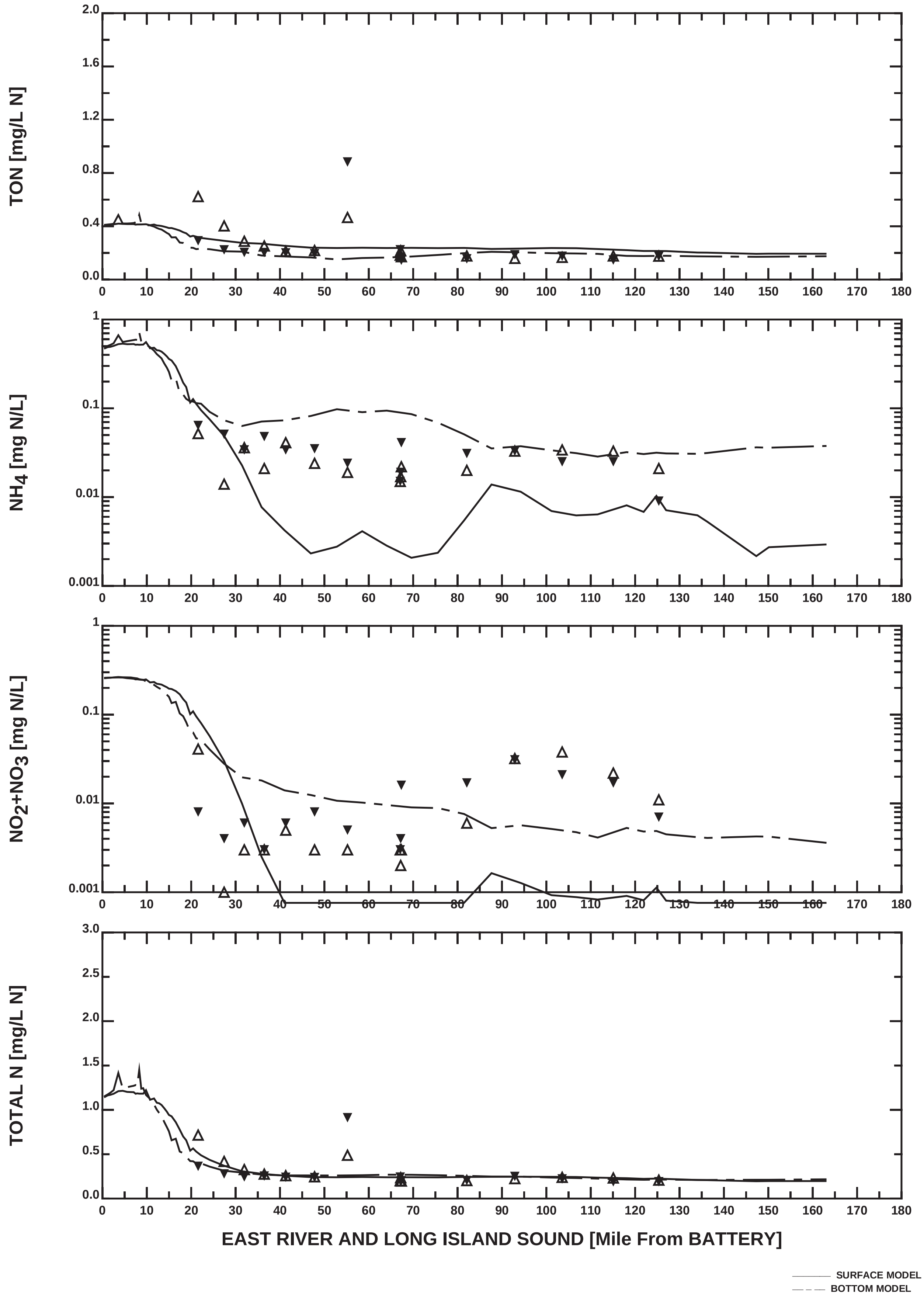


Figure 63. Total organic nitrogen,NH4, NO2+NO3 and total nitrogen calibration results for May 1999

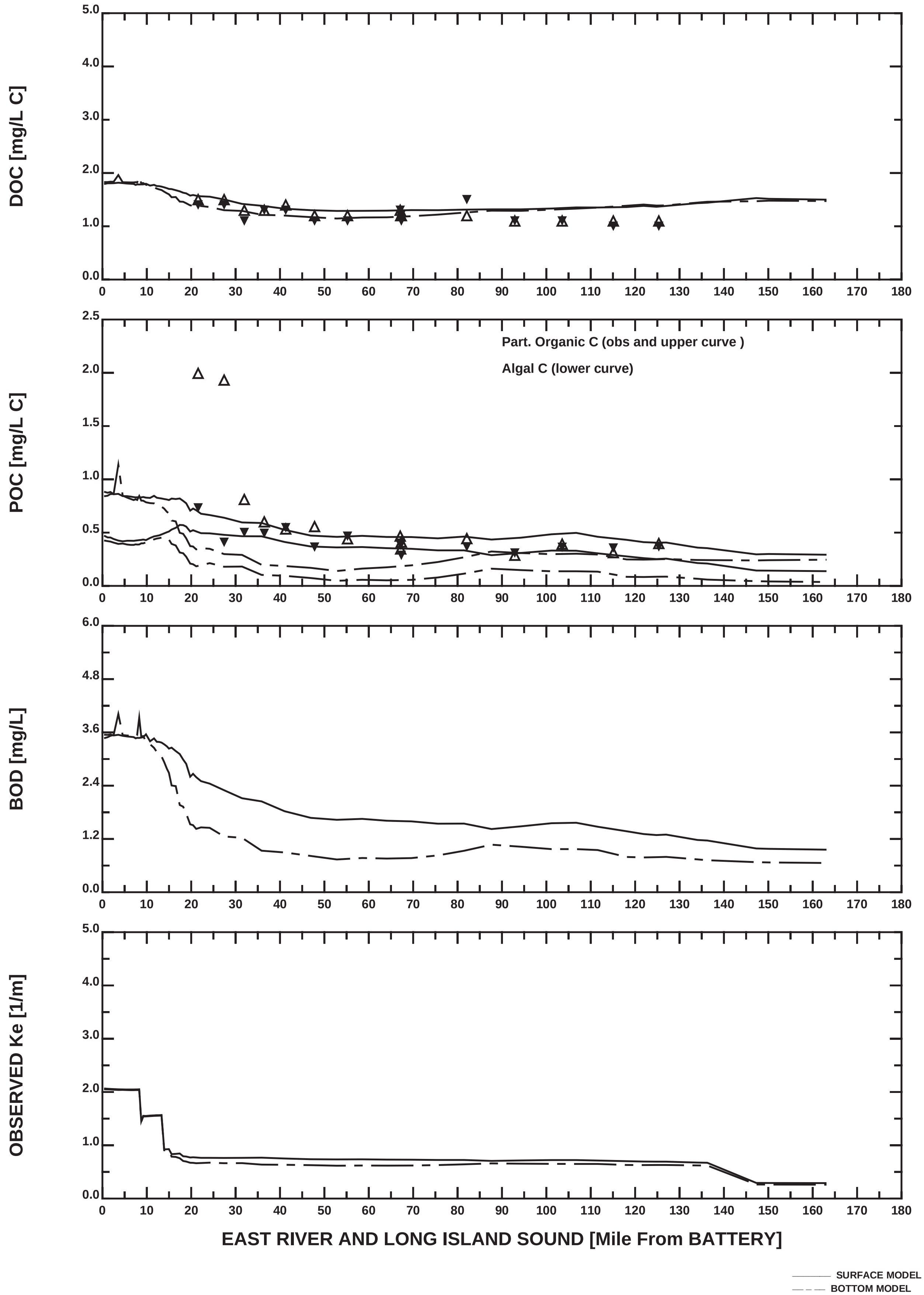


Figure 64. Dissolved organic carbon, POC, BOD and light attenuation calibration results for May 1999

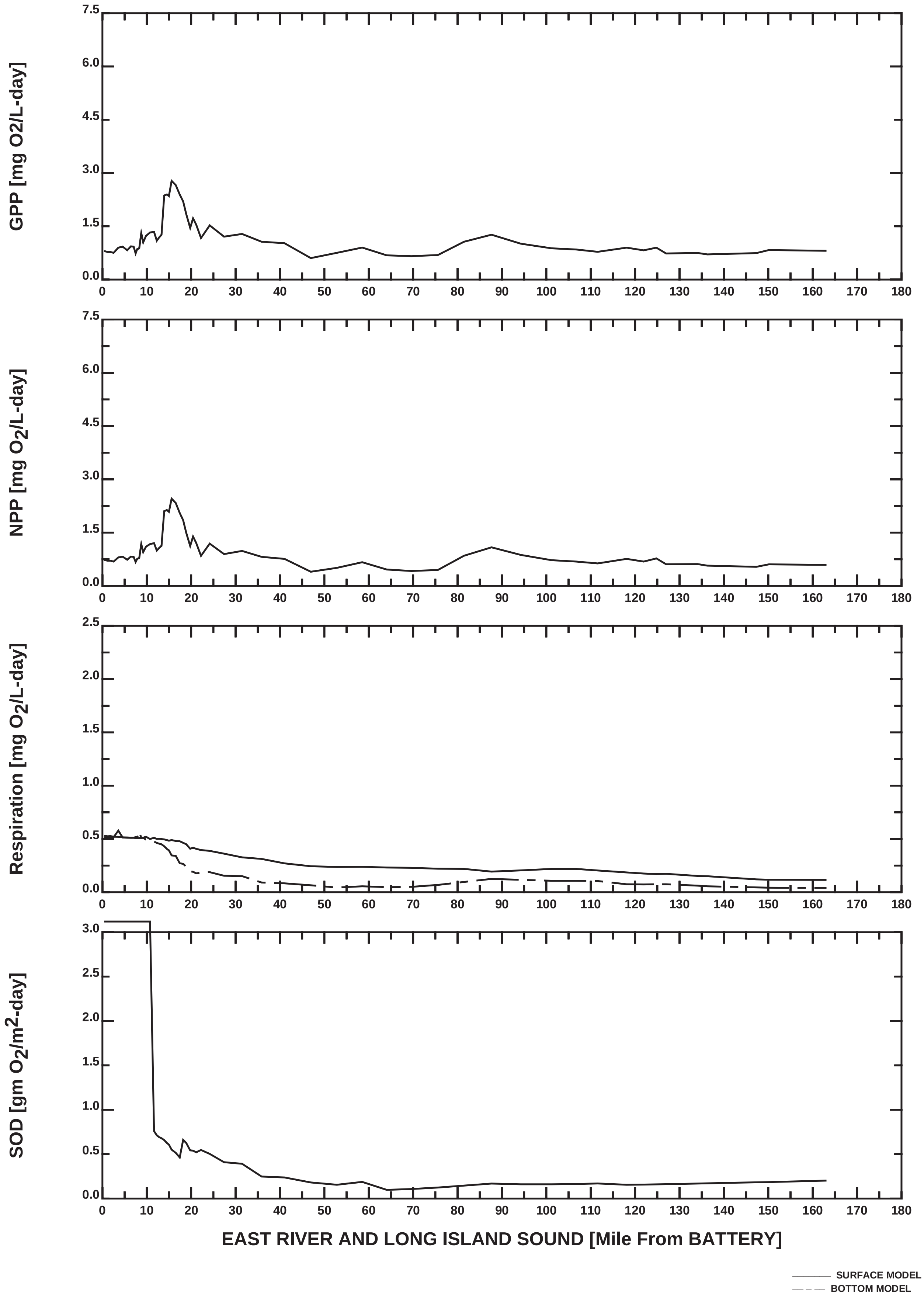


Figure 65. Gross primary production, net primary production, community respiration, and sediment oxygen calibration results for May 1999

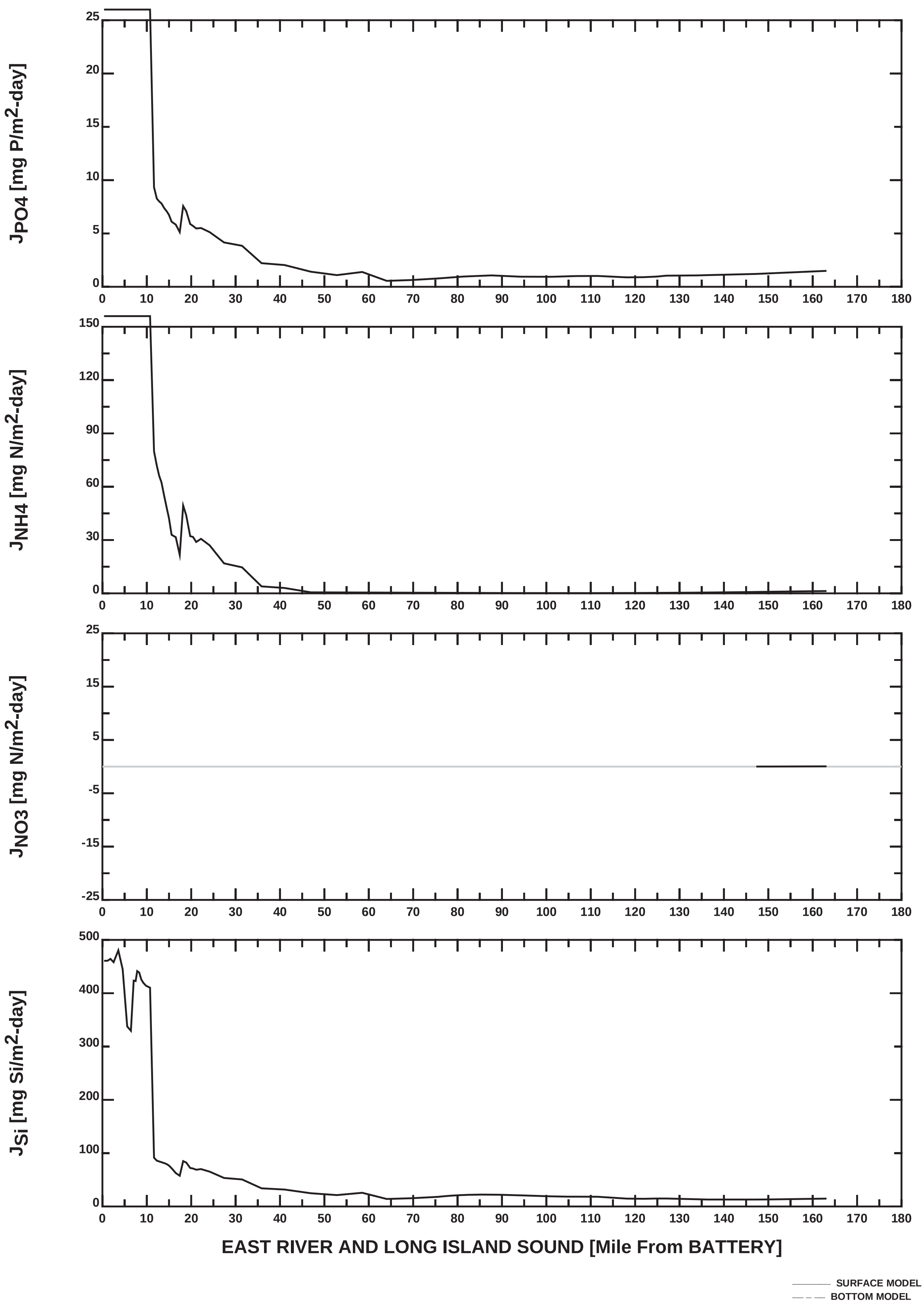
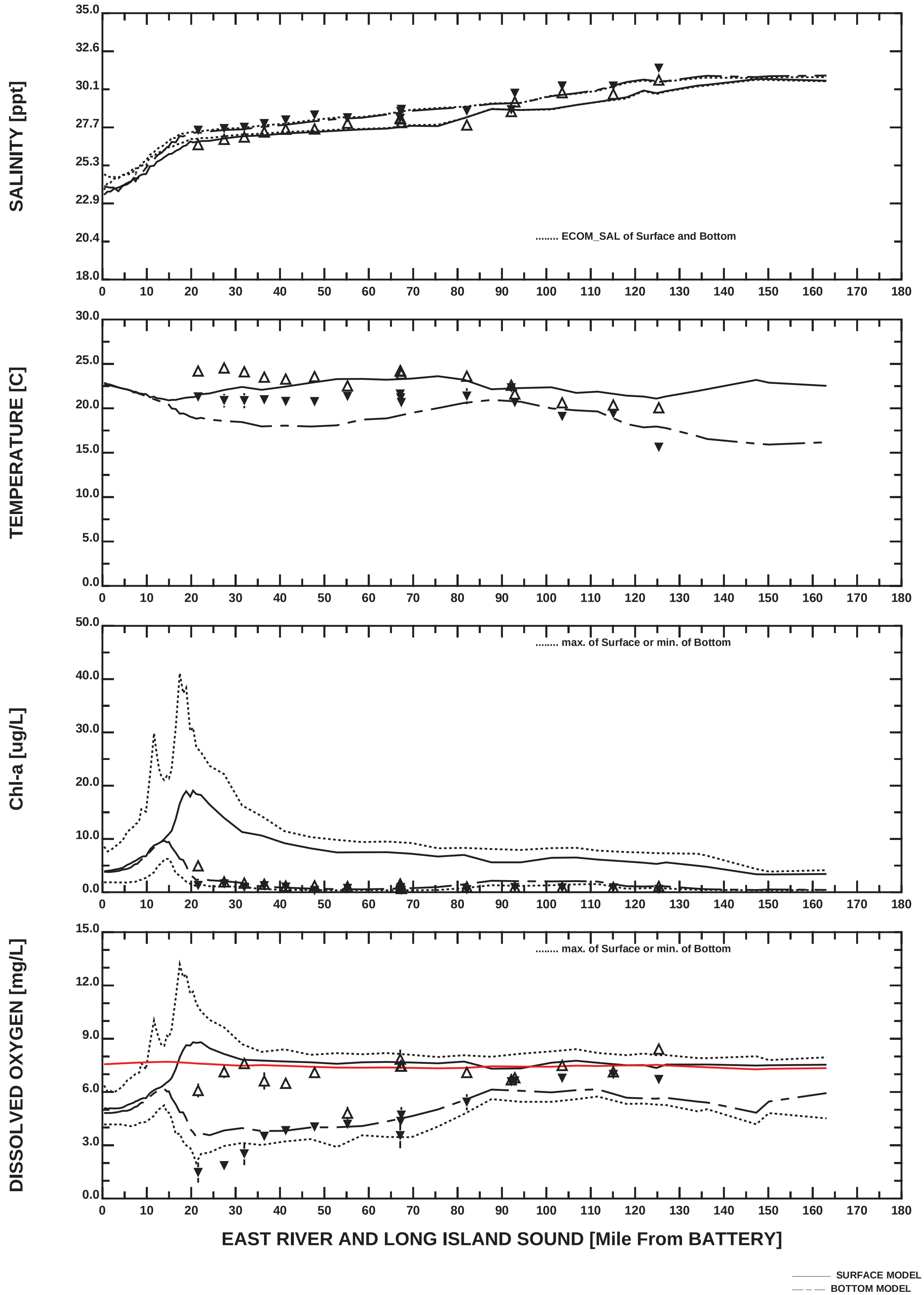
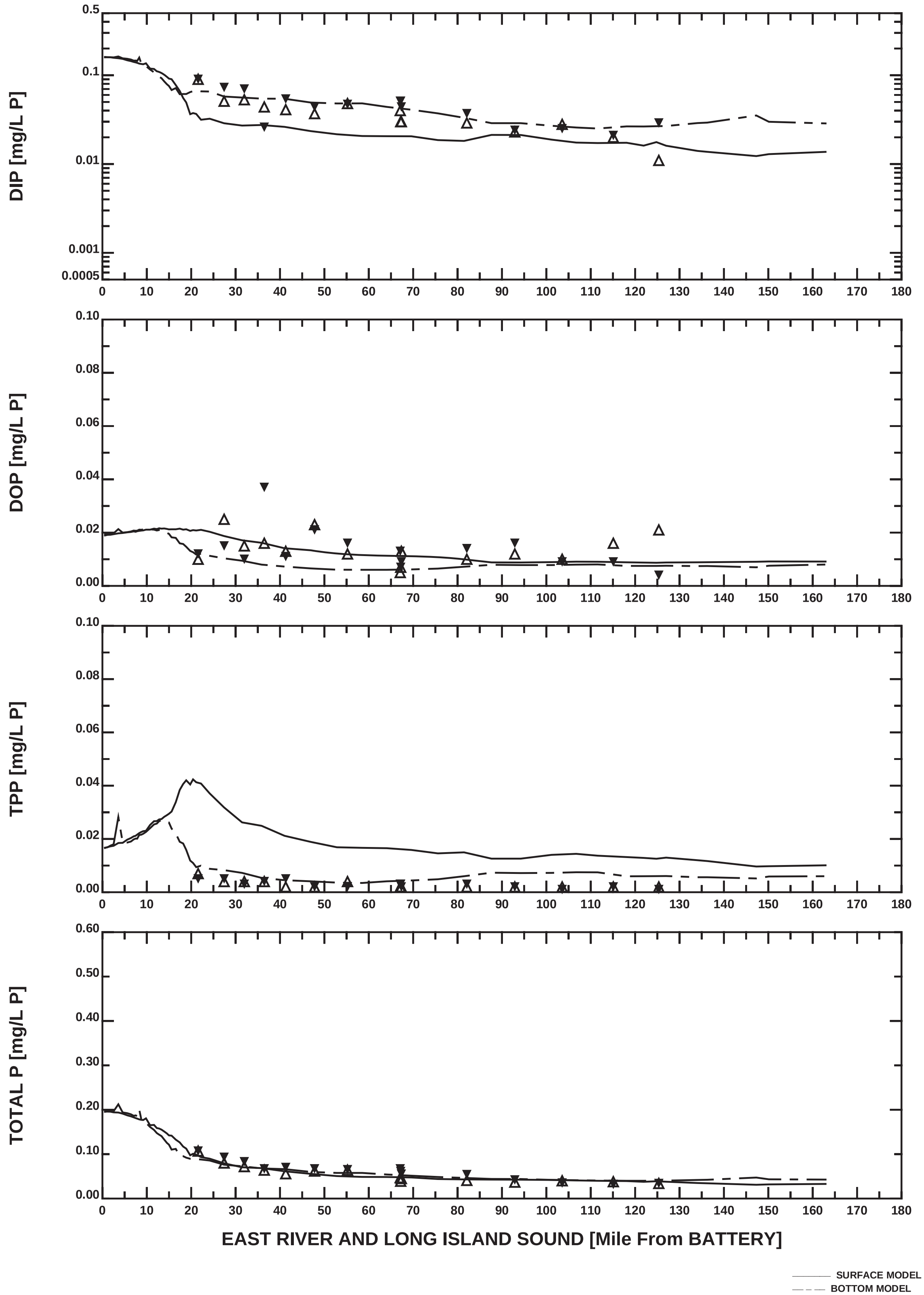
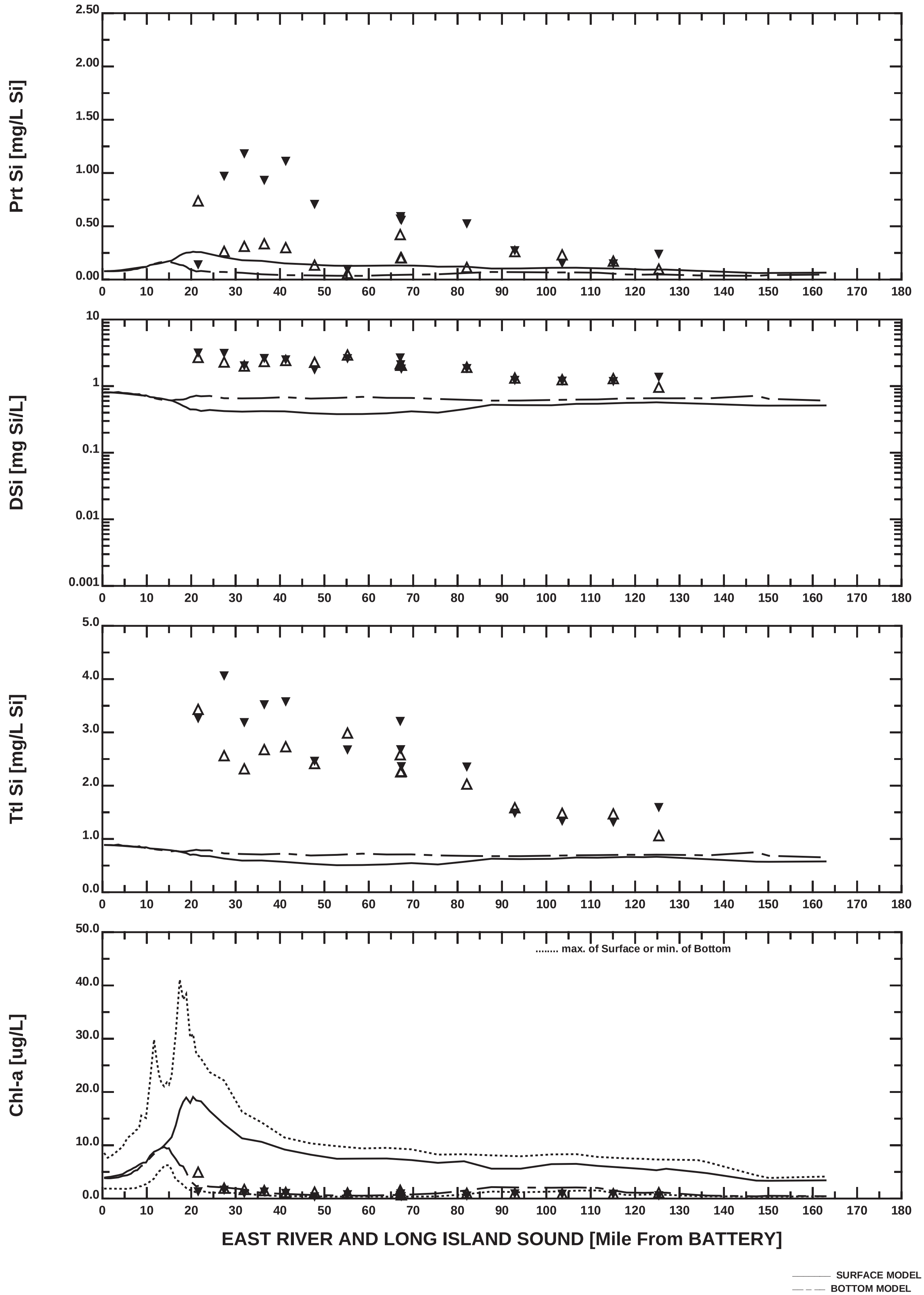


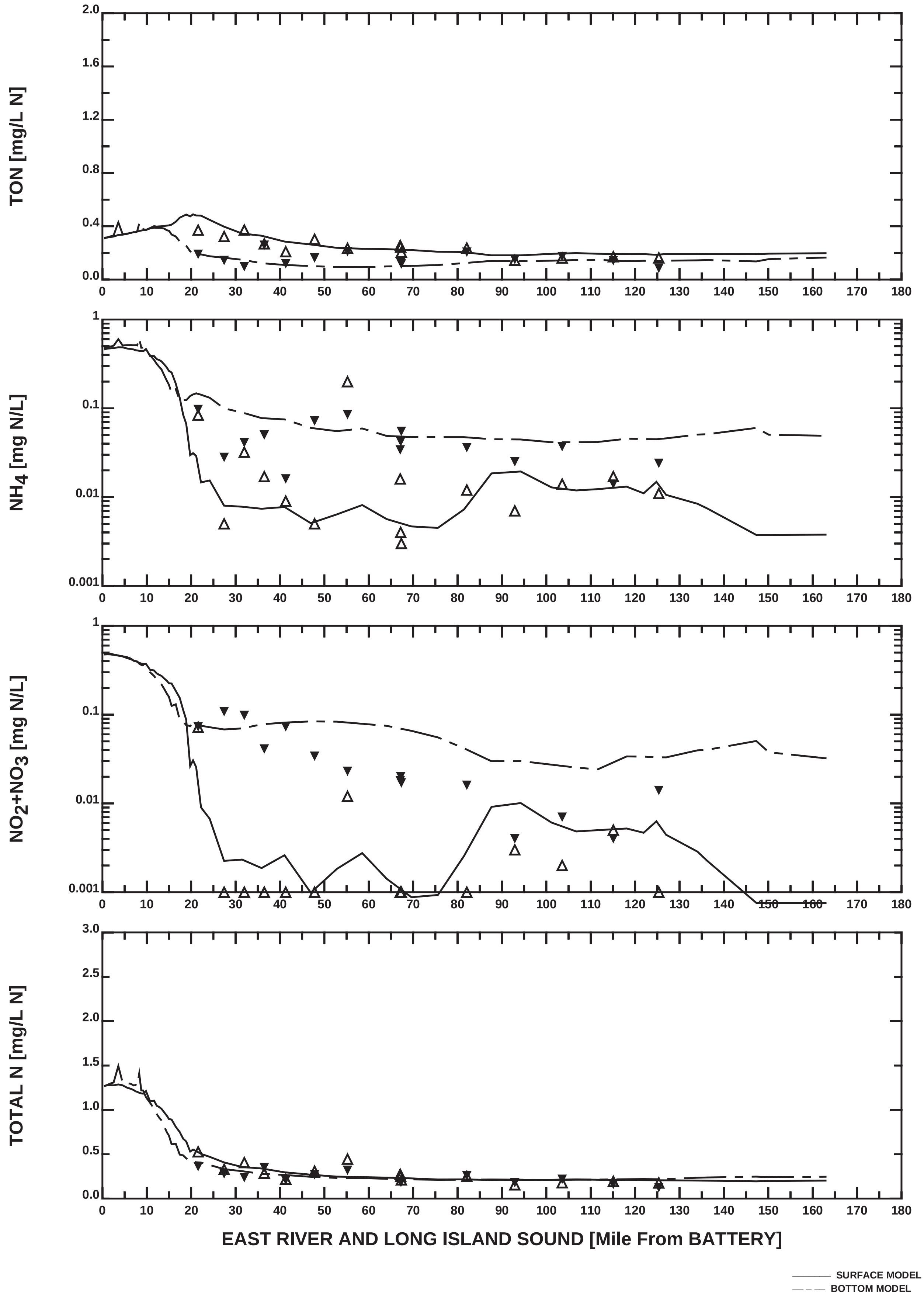
Figure 66. Calibration results for sediment nutrient fluxes of PO4, NH4, NO3 and Si for May 1999


SALINITY [ppt]
.... ECOM_SAL of Surface and Bottom
TEMPERATURE [C]
Chl-a [ug/L]
.... max. of Surface or min. of Bottom
DISSOLVED OXYGEN [mg/L]
.... max. of Surface or min. of Bottom
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL

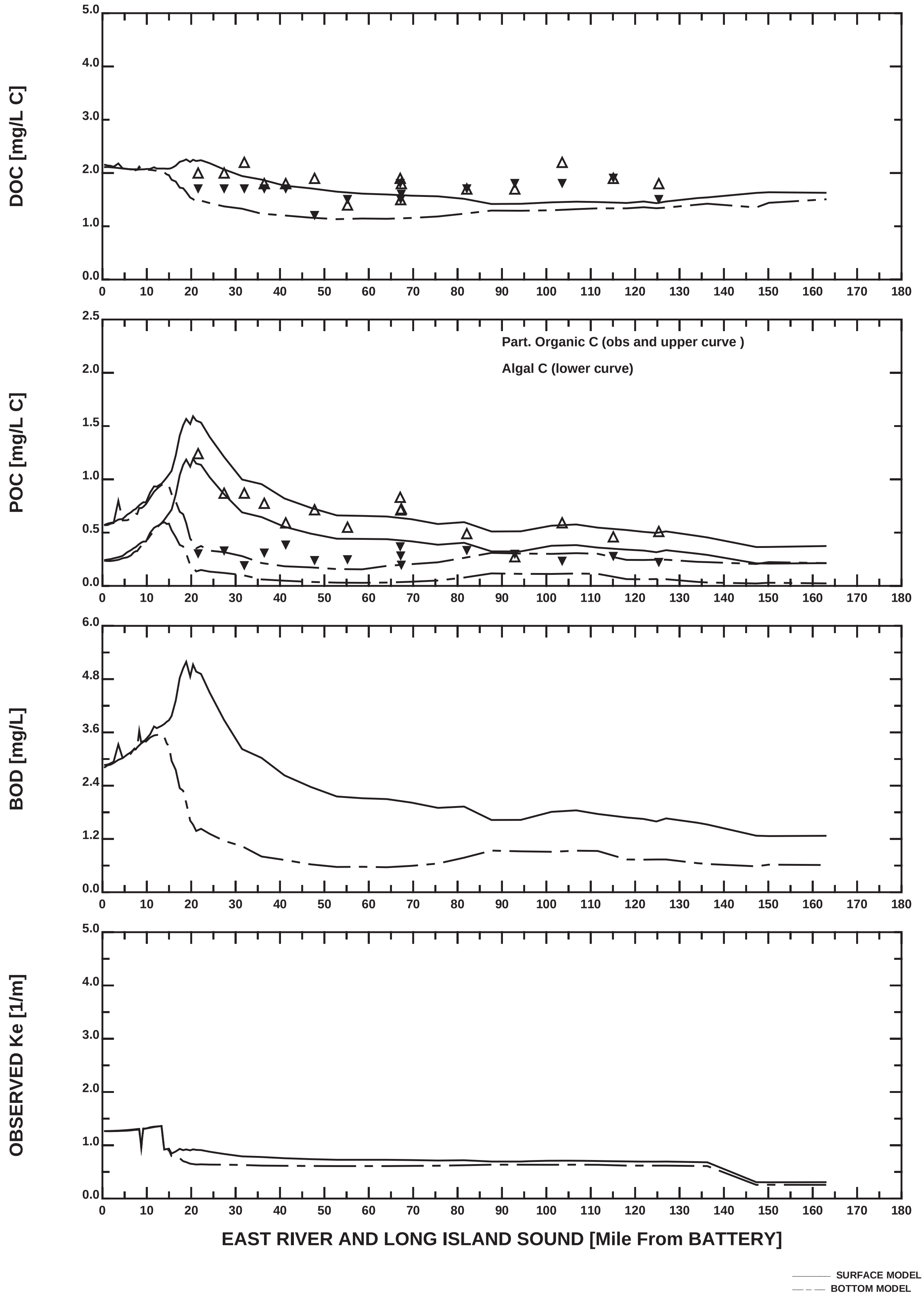

DIP [mg/L P]
DOP [mg/L P]
TPP [mg/L P]
TOTAL P [mg/L P]
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL



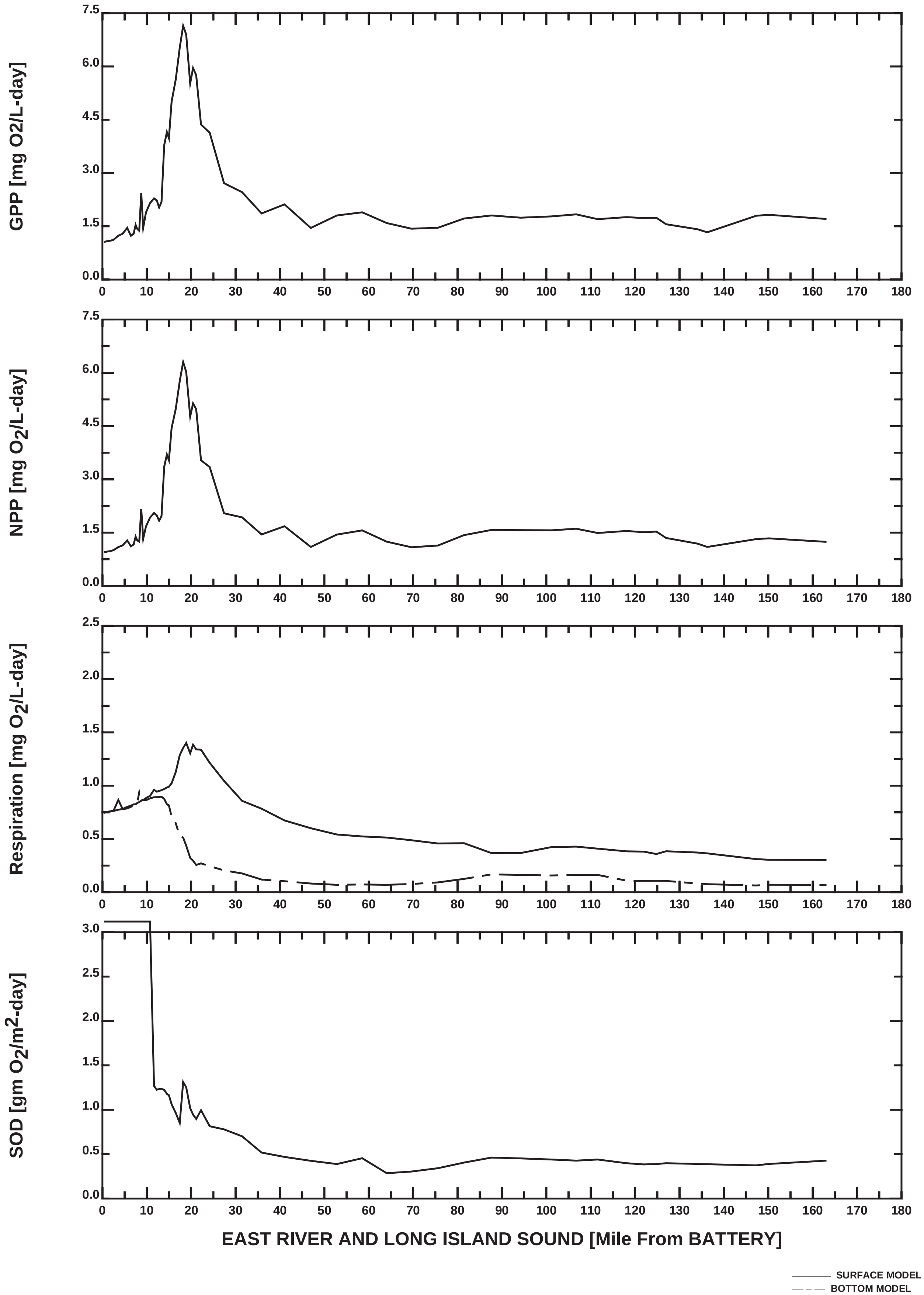
Prt Si [mg/L Si]
DSi [mg Si/L]
Ttl Si [mg/L Si]
Chl-a [ug/L]
.... max. of Surface or min. of Bottom
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL



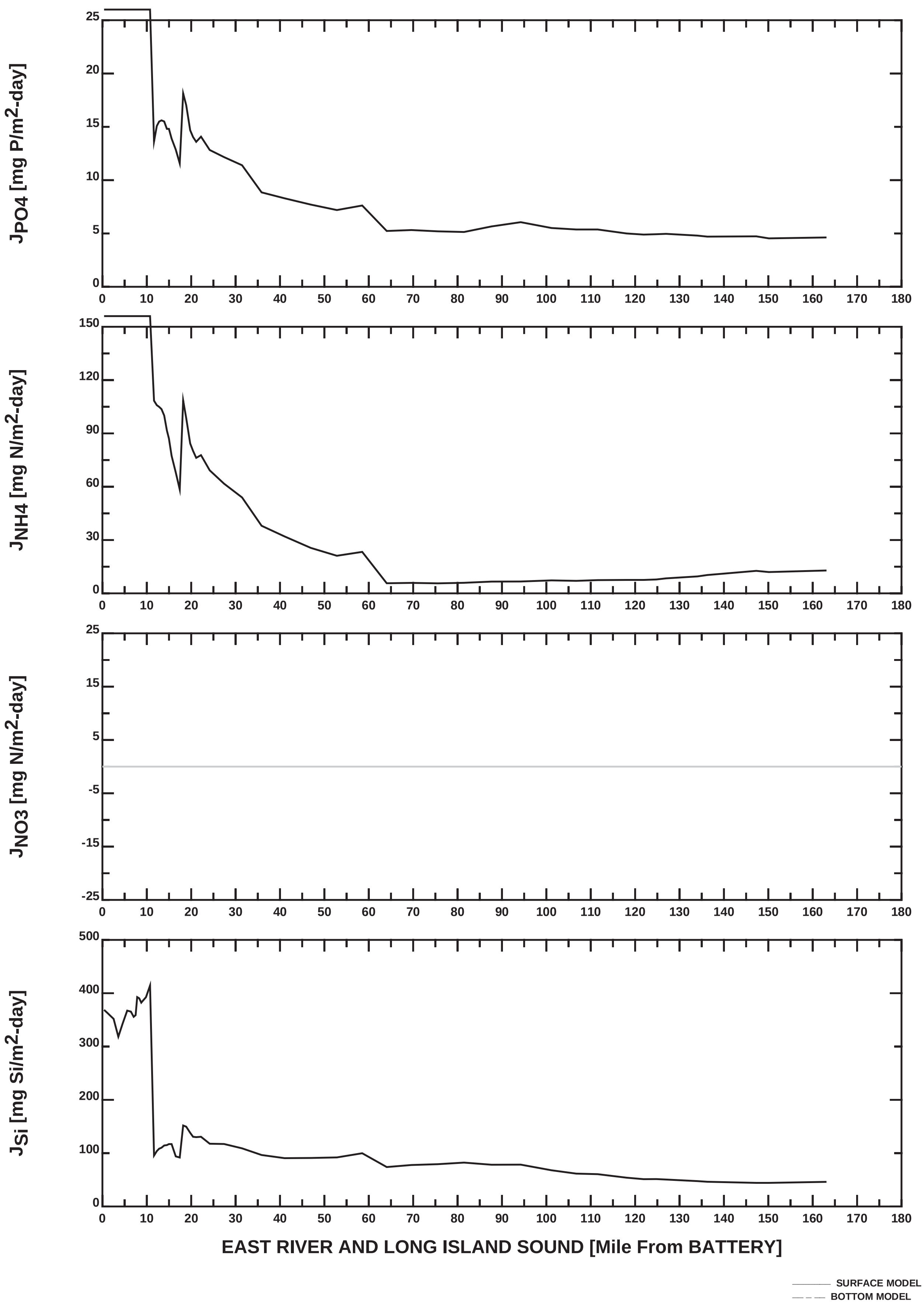
TON [mg/L N]
NH4 [mg N/L]
NO2+NO3 [mg N/L]
TOTAL N [mg/L N]
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL



DOC [mg/L C]
POC [mg/L C]
Part. Organic C (obs and upper curve)
Algal C (lower curve)
BOD [mg/L]
OBSERVED Ke [1/m]
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL



GPP [mg O2/L-day]
NPP [mg O2/L-day]
Respiration [mg O2/L-day]
SOD [gm O2/m2-day]
0.0
1.5
3.0
4.5
6.0
7.5
0.5
1.0
2.0
2.5
0
10
20
30
40
50
60
70
80
90
100
110
120
130
140
150
160
170
180
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL



JPO4 [mg P/m2-day]
JNH4 [mg N/m2-day]
JNO3 [mg N/m2-day]
JSi [mg Si/m2-day]
EAST RIVER AND LONG ISLAND SOUND [Mile From BATTERY]
SURFACE MODEL
BOTTOM MODEL

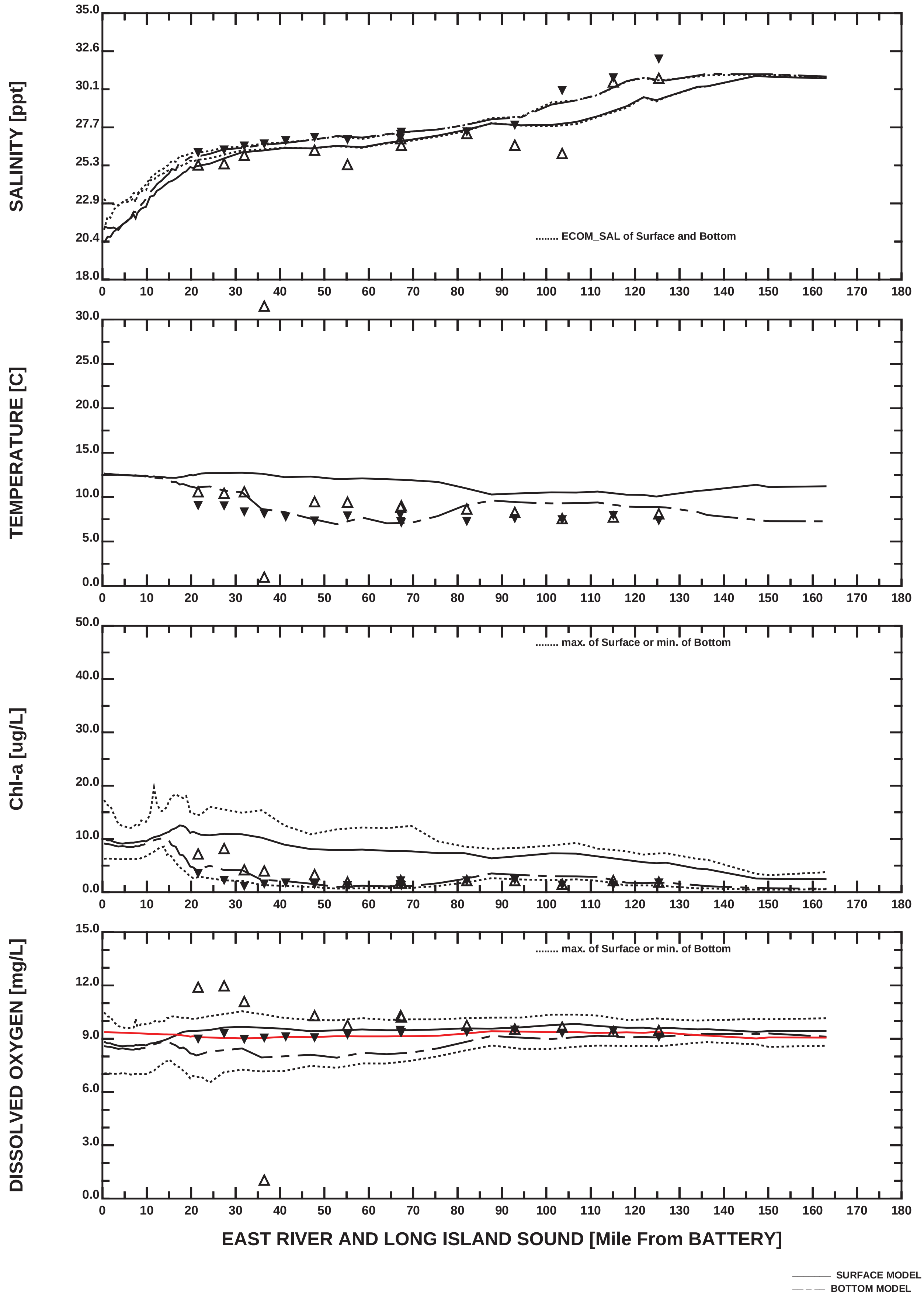


Figure 74. Salinity, temperature, chlorophyll and DO calibration results for May 2000

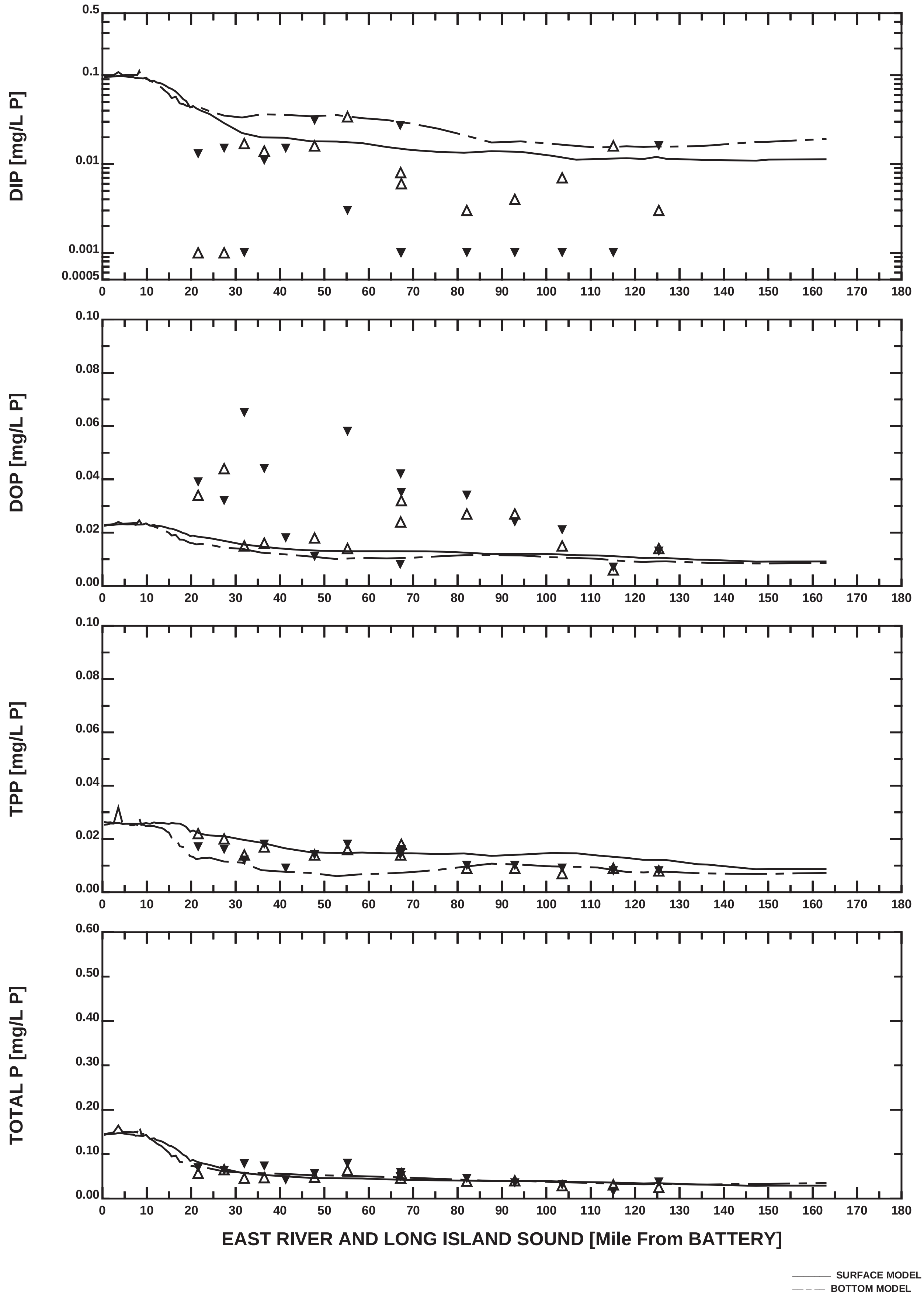


Figure 75. Dissolved inorganic phosphorus, DOP, total particulate phosphorus and total phosphorus calibration results for May 2000

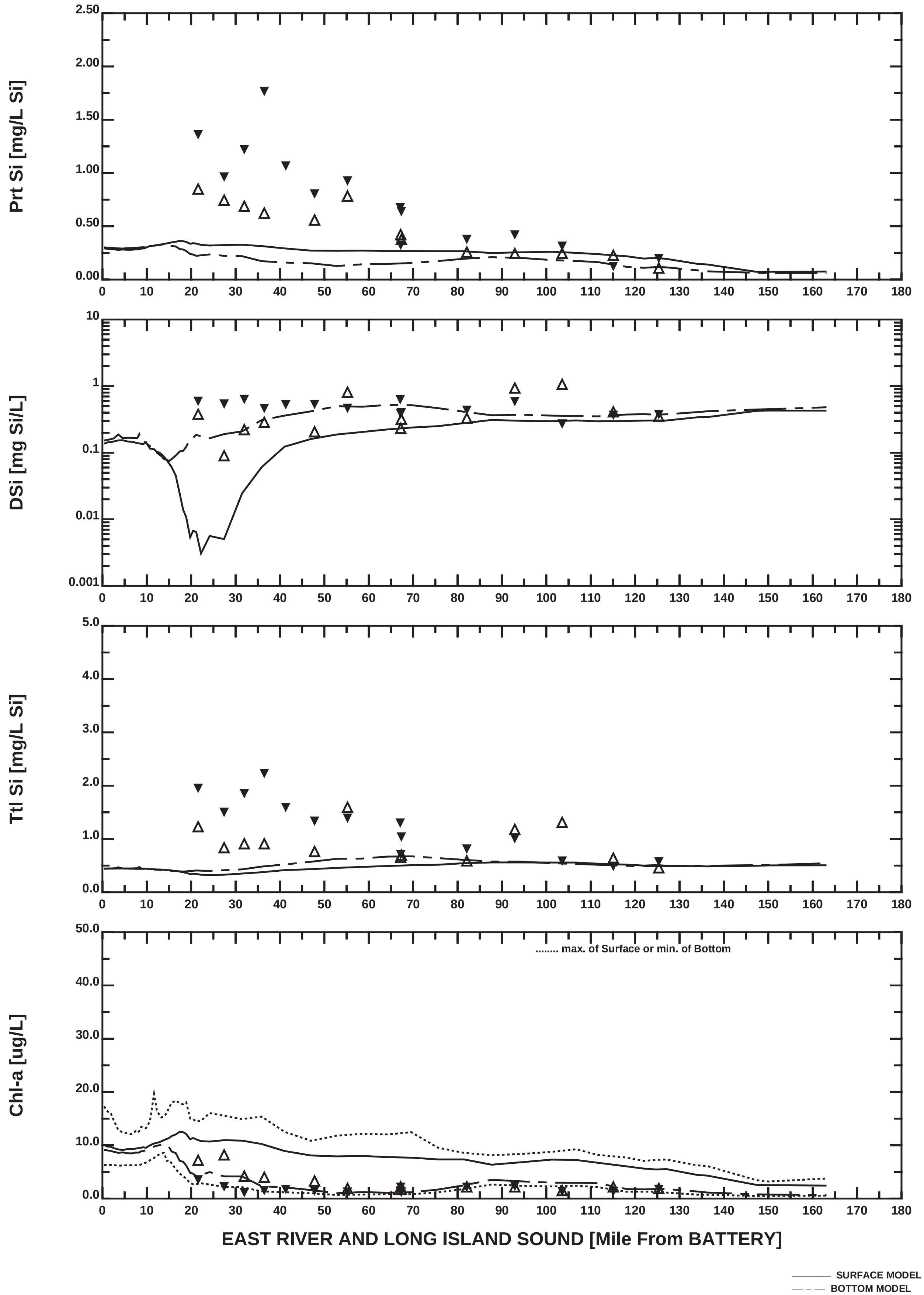


Figure 76. Particulate silica, Si, total silica, and chlorophyll calibration results for May 2000

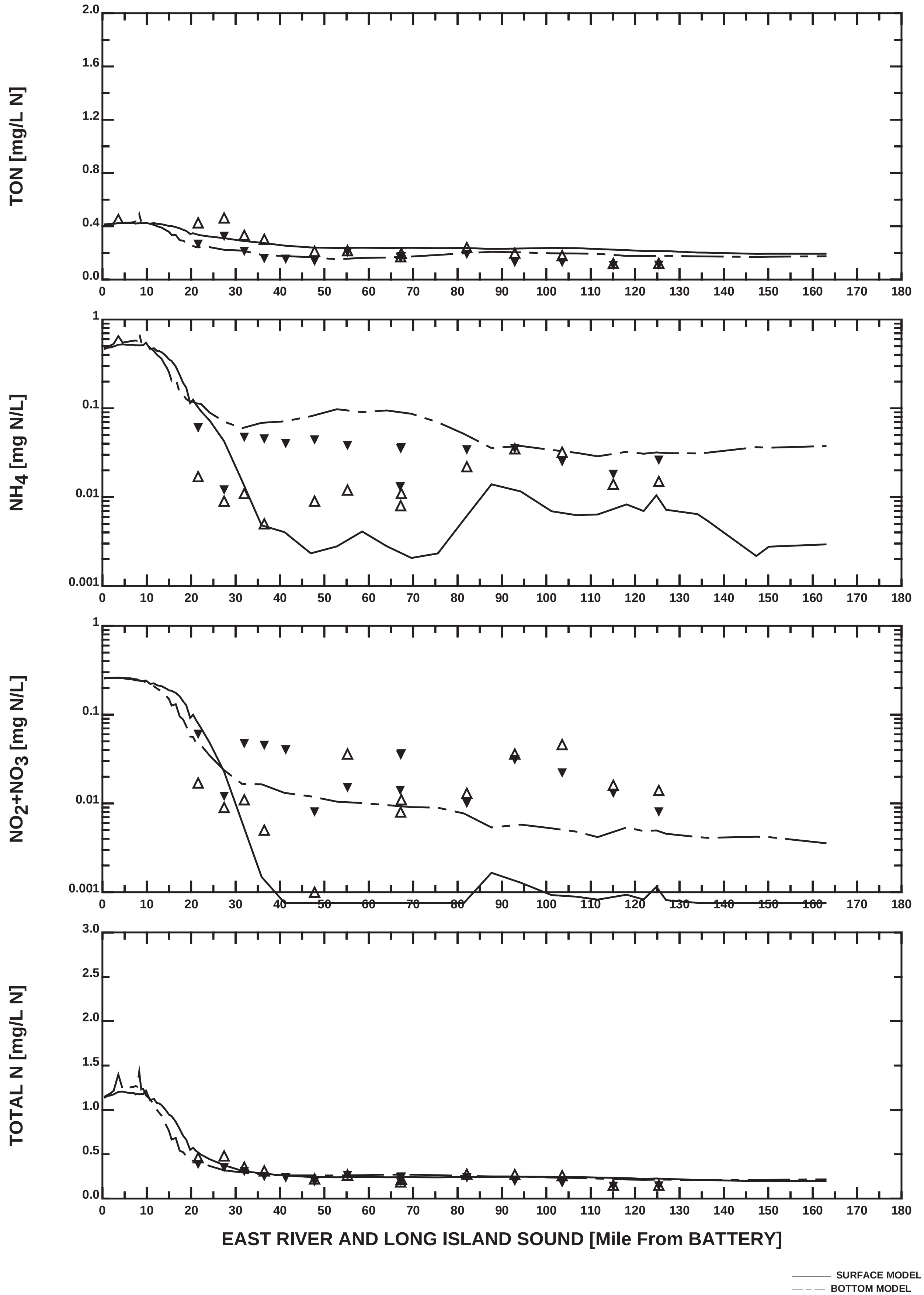


Figure 77. Total organic nitrogen,NH4, NO2+NO3 and total nitrogen calibration results for May 2000

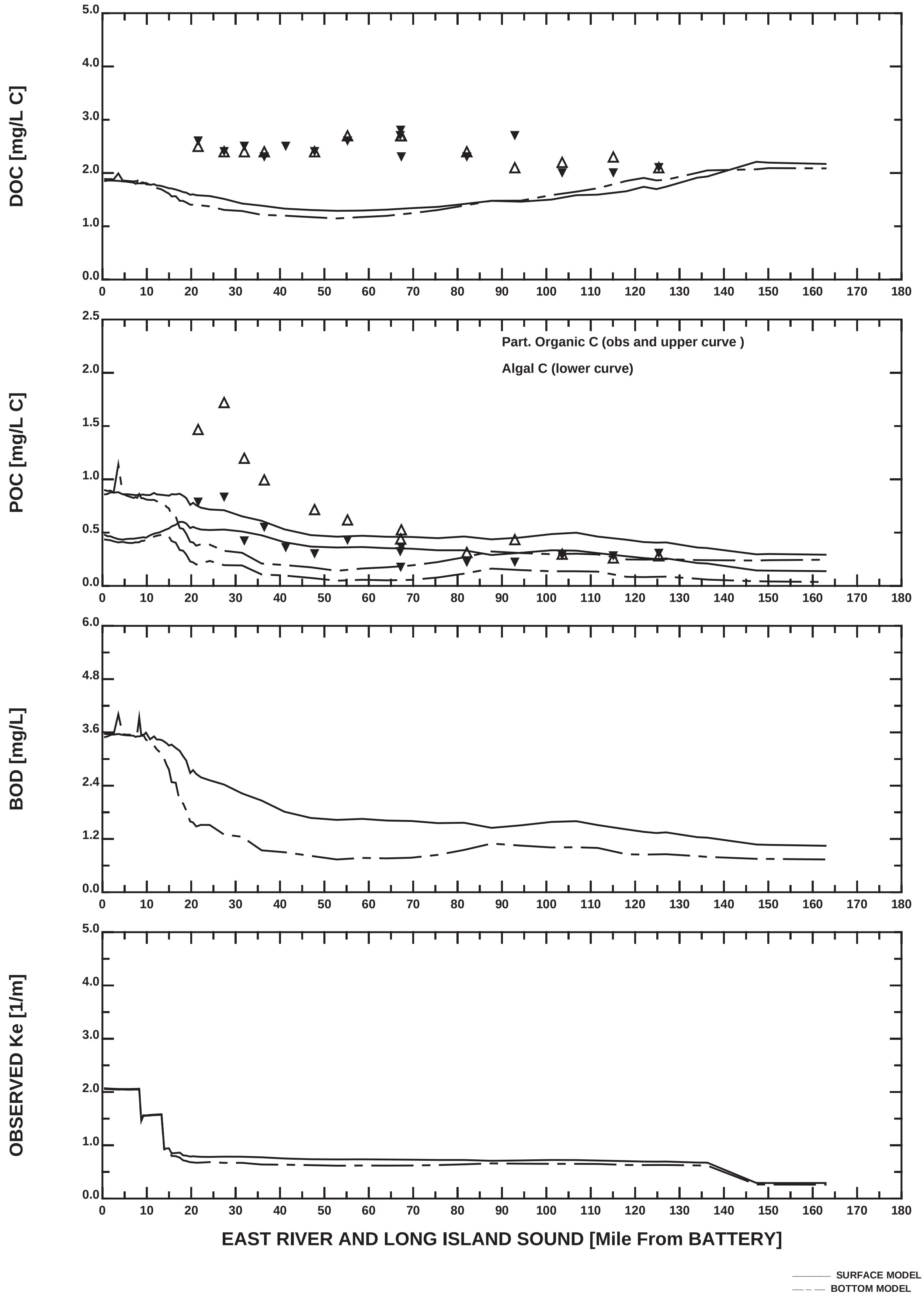


Figure 78. Dissolved organic carbon, POC, BOD and light attenuation calibration results for May 2000

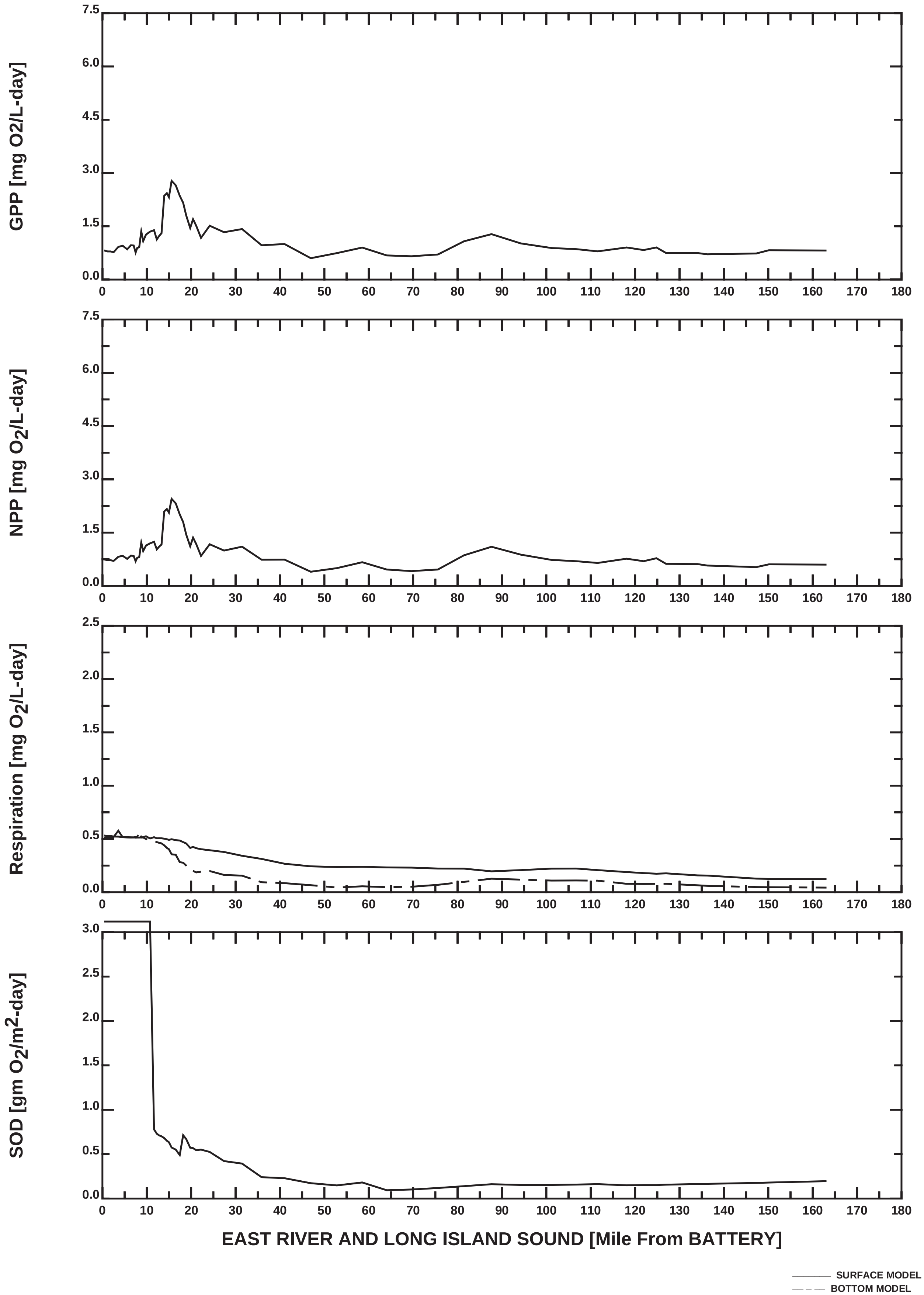


Figure 79. Gross primary production, net primary production, community respiration, and sediment oxygen calibration results for May 2000

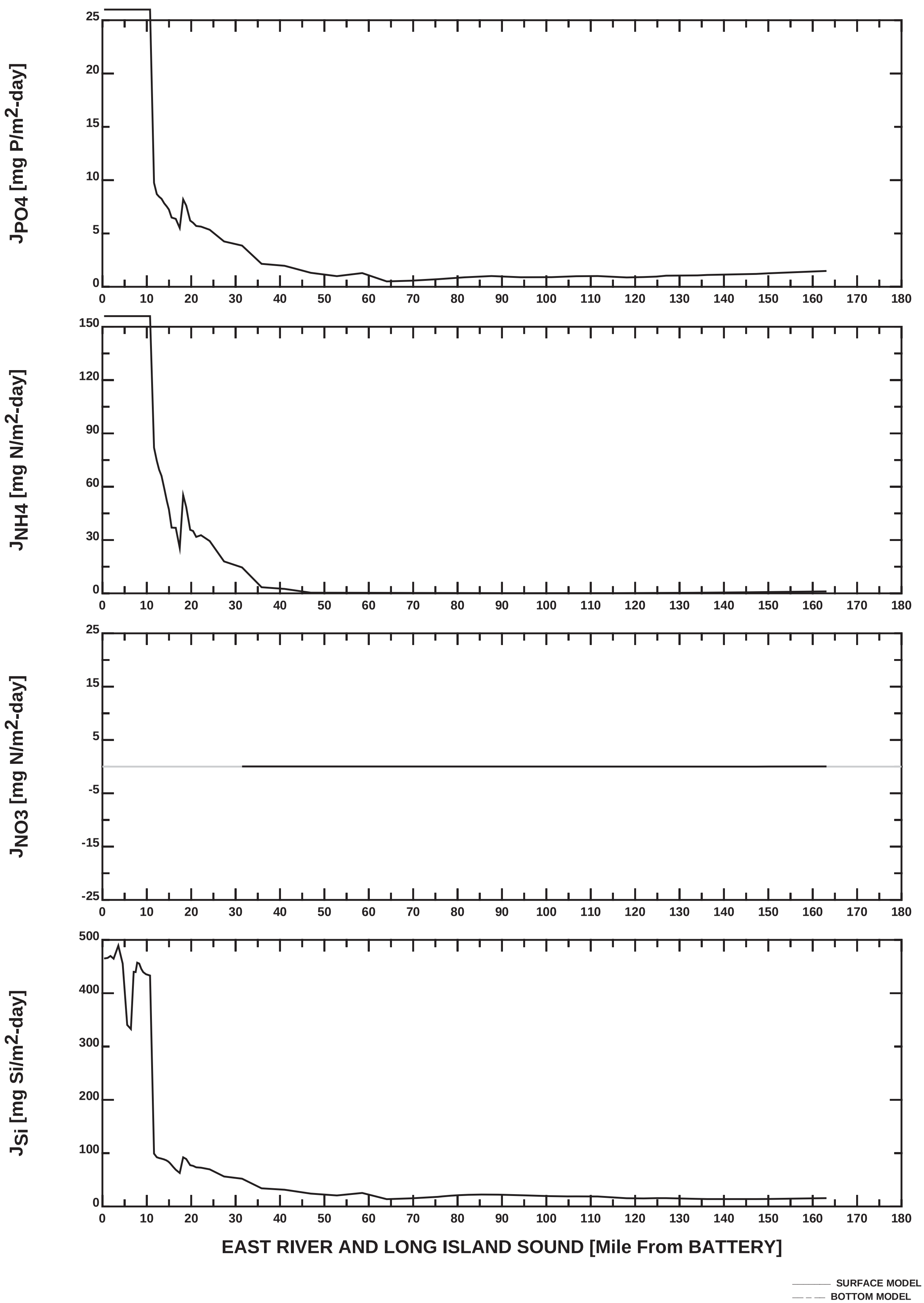


Figure 80. Calibration results for sediment nutrient fluxes of PO4, NH4, NO3 and Si for May 2000

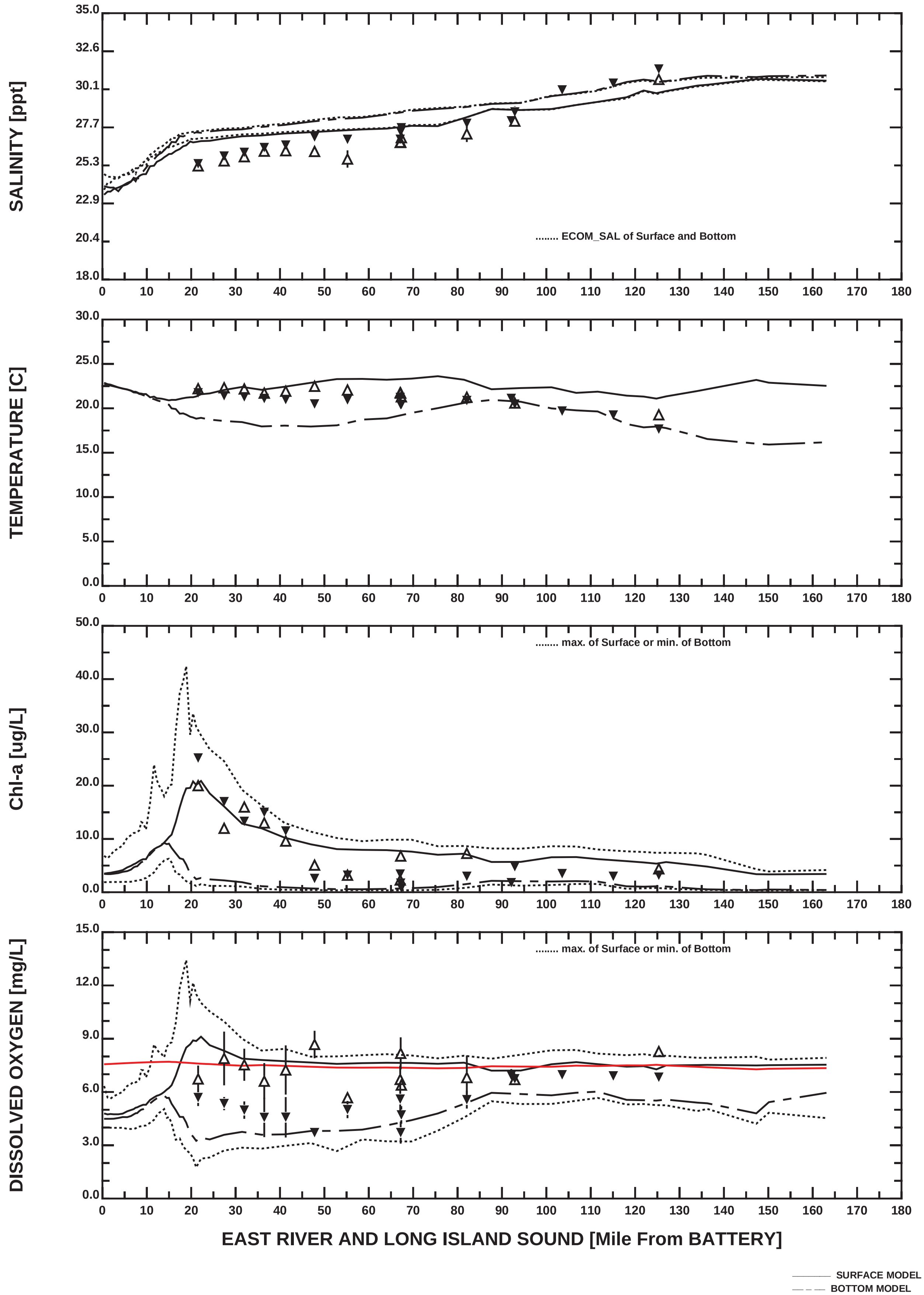


Figure 81. Salinity, temperature, chlorophyll and DO calibration results for August 2000

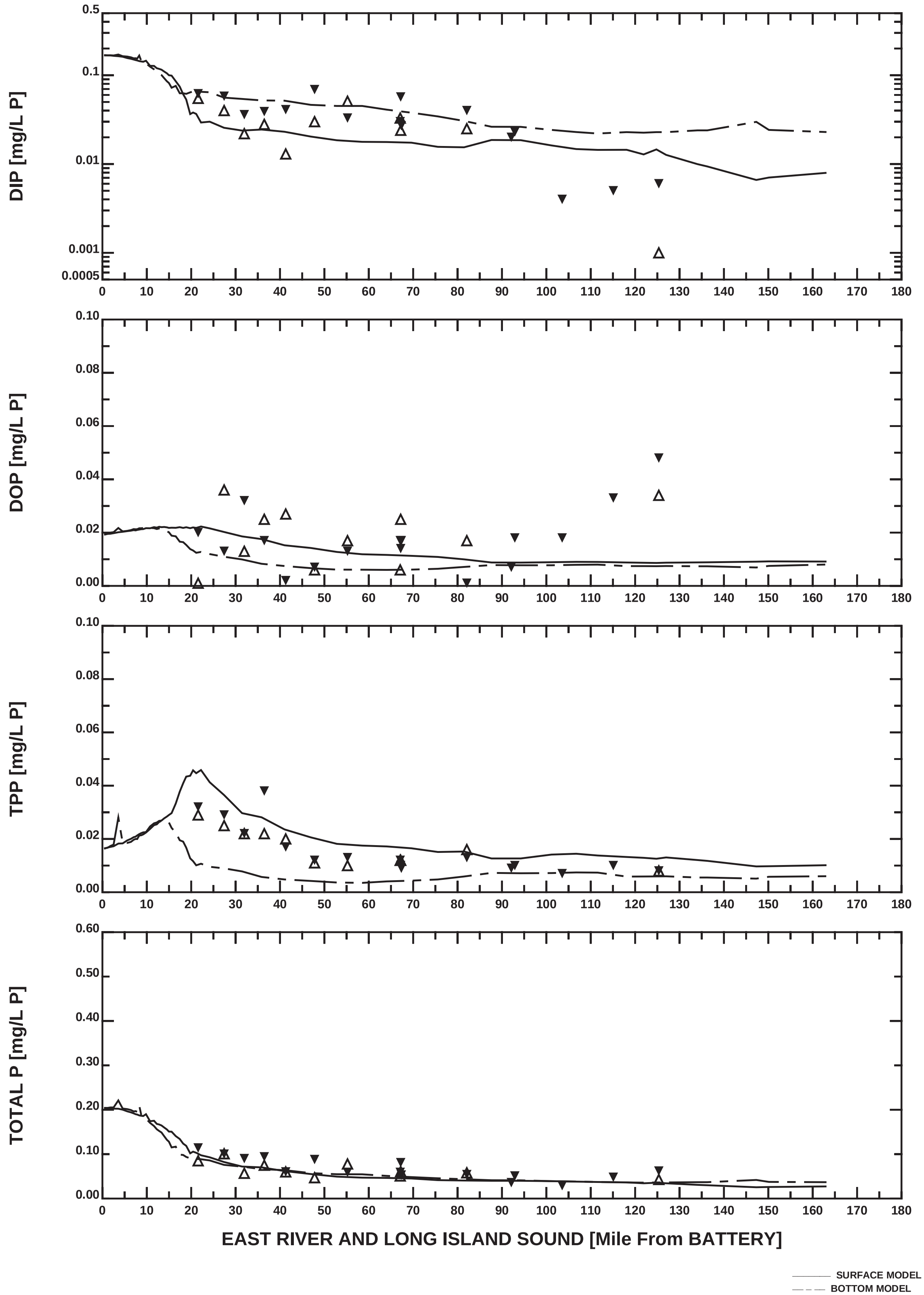


Figure 82. Dissolved inorganic phosphorus, DOP, total particulate phosphorus and total phosphorus calibration results for August 2000

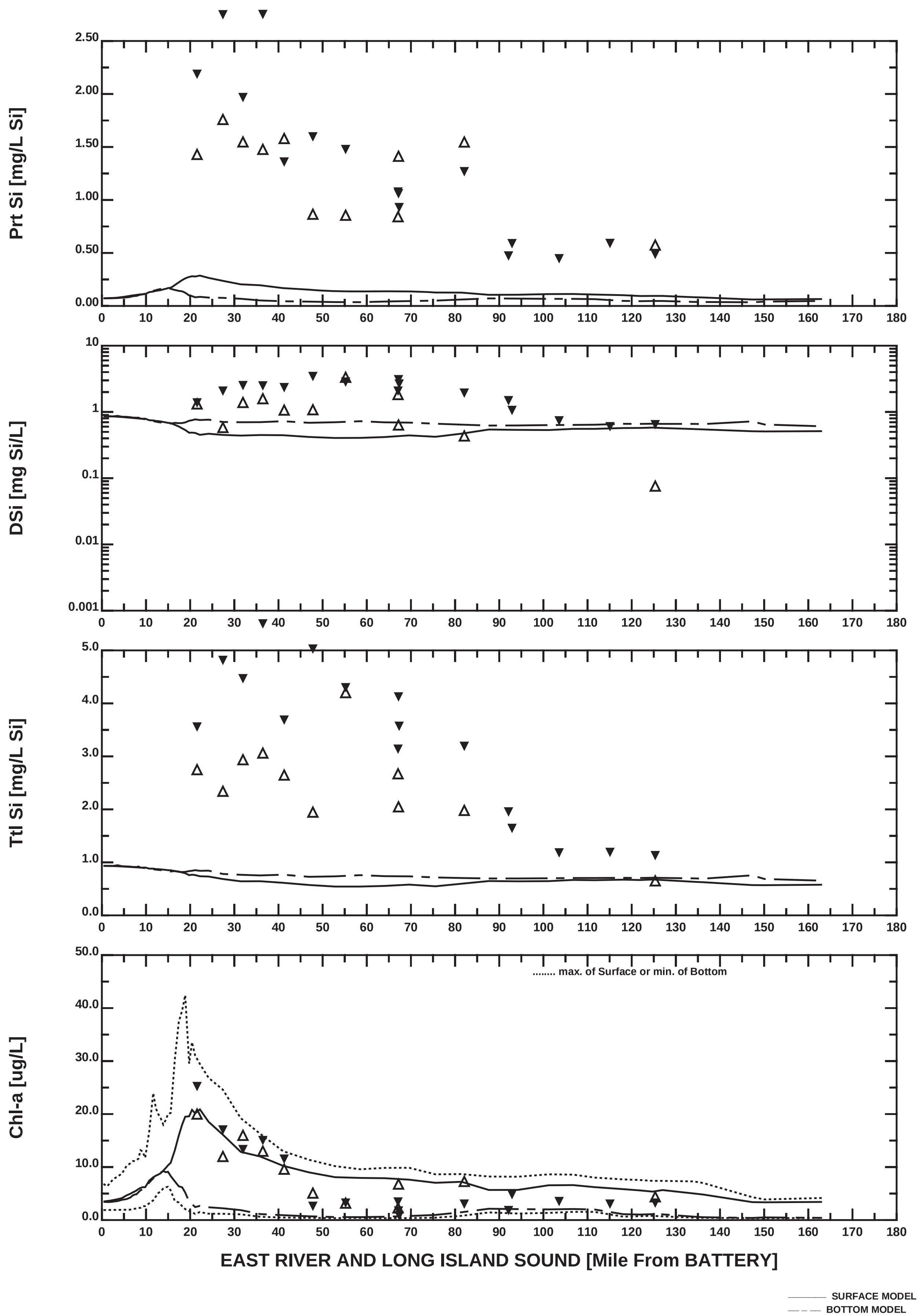


Figure 83. Particulate silica, Si, total silica, and chlorophyll calibration results for August 2000

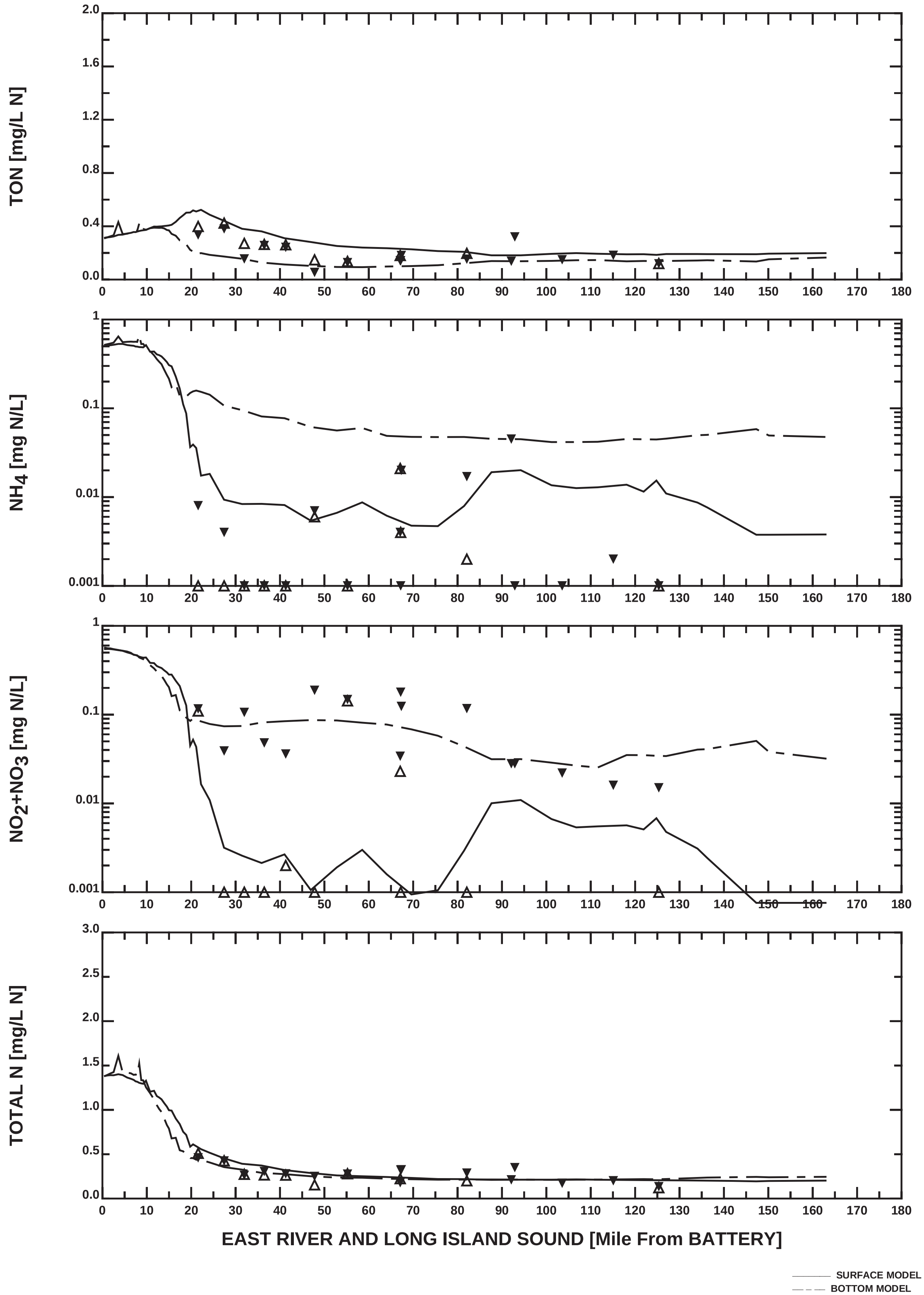


Figure 84. Total organic nitrogen,NH4, NO2+NO3 and total nitrogen calibration results for August 2000

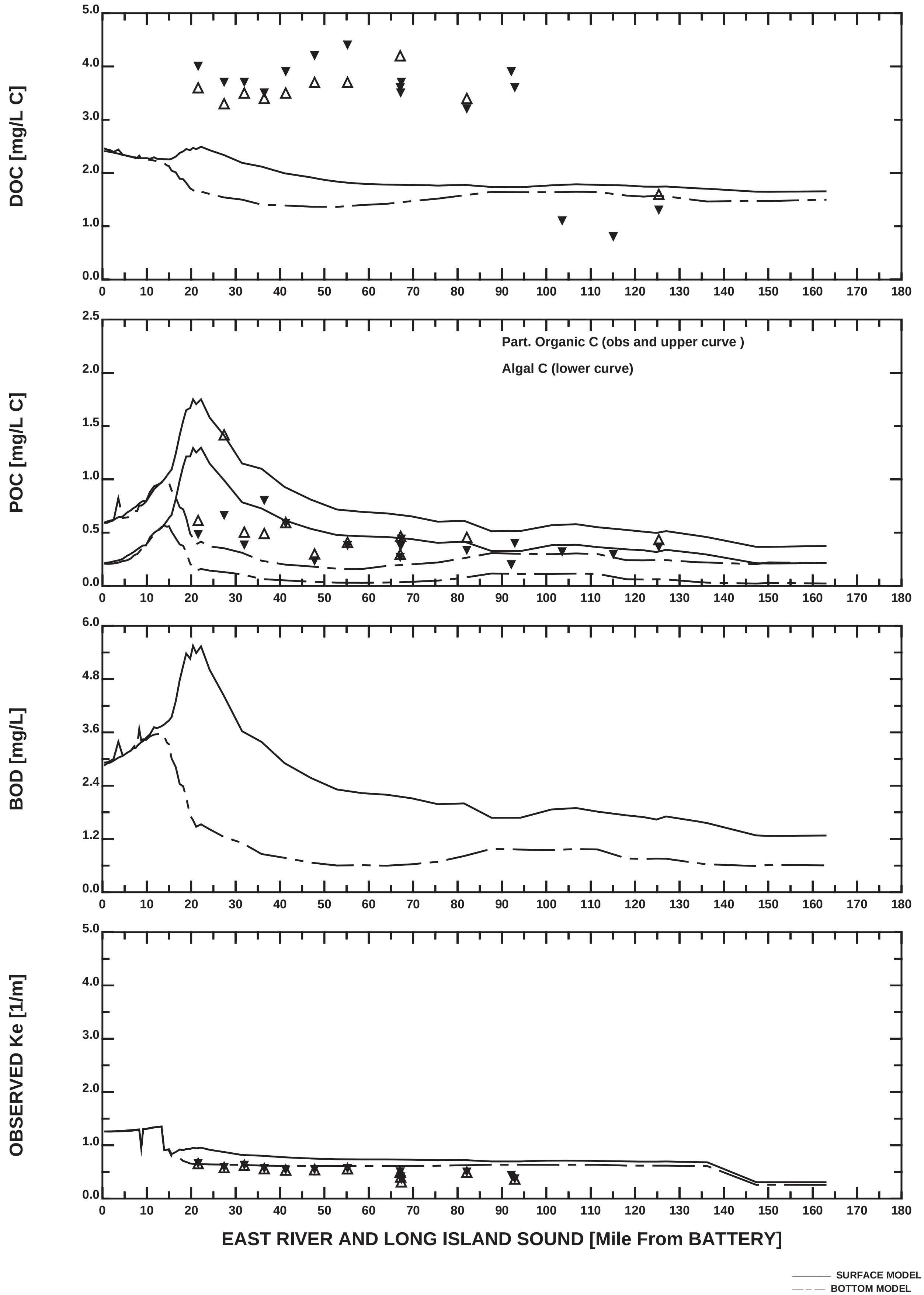


Figure 85. Dissolved organic carbon, POC, BOD and light attenuation calibration results for August 2000

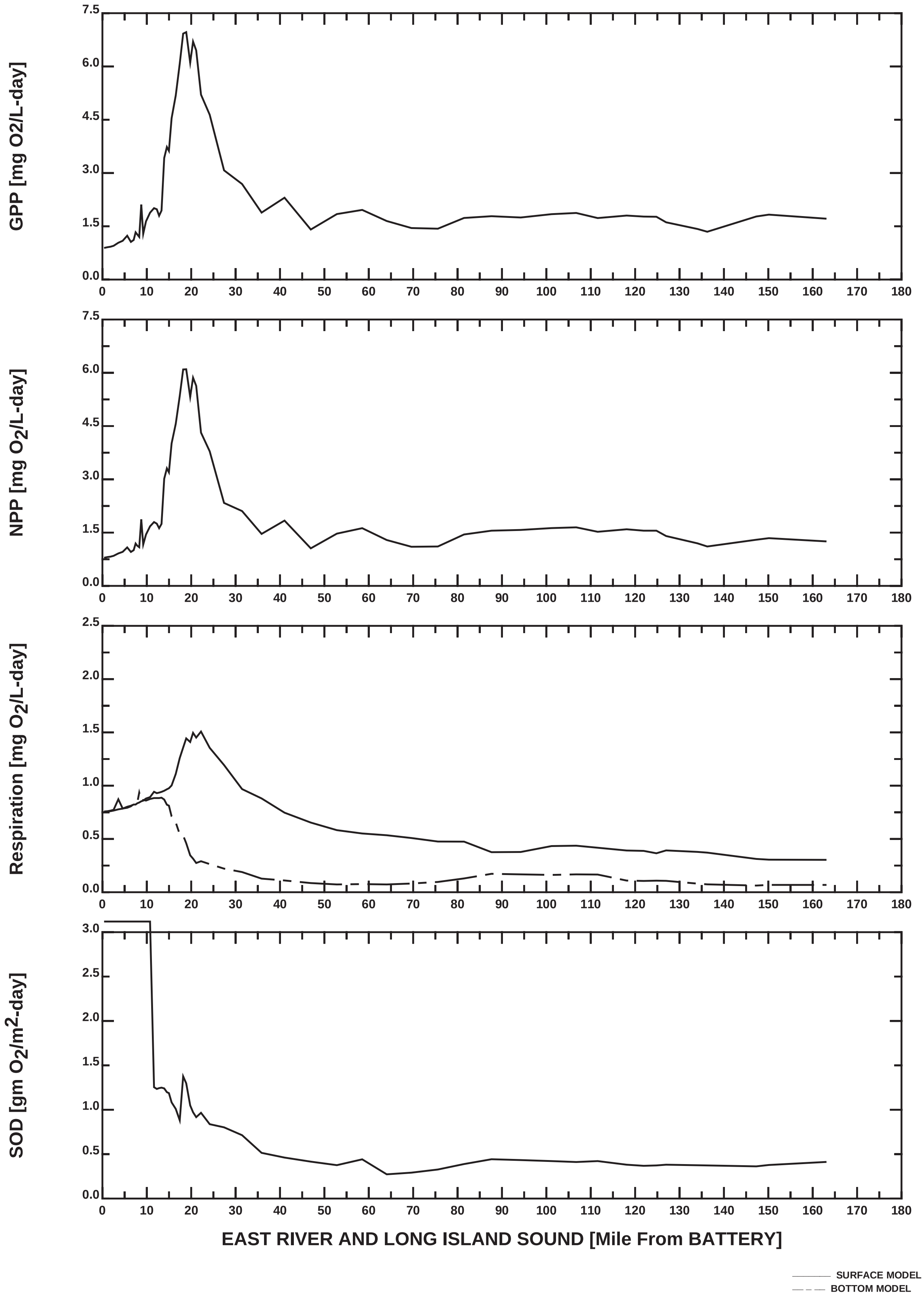


Figure 86. Gross primary production, net primary production, community respiration, and sediment oxygen calibration results for August 2000

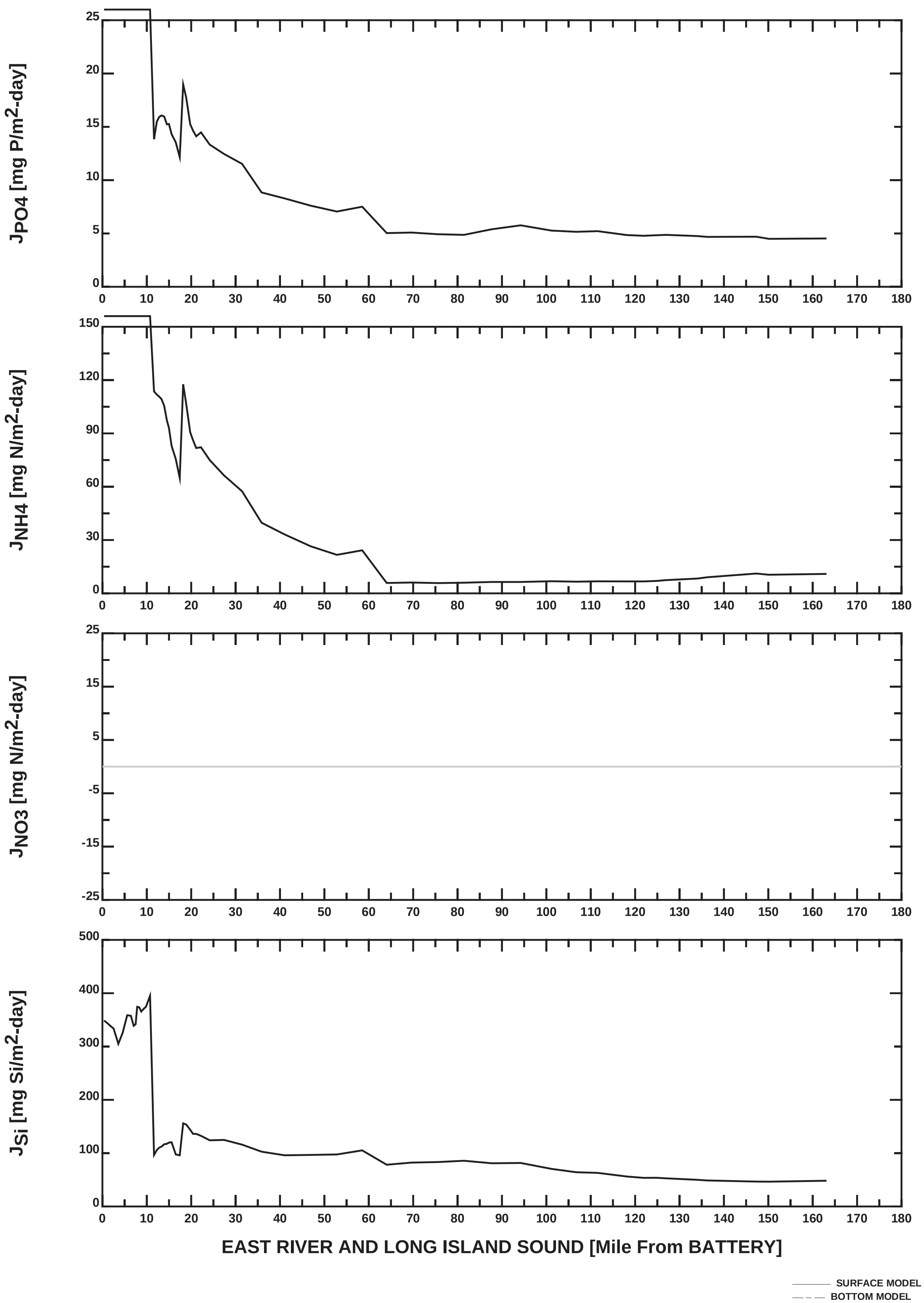


Figure 87. Calibration results for sediment nutrient fluxes of PO4, NH4, NO3 and Si for August 2000

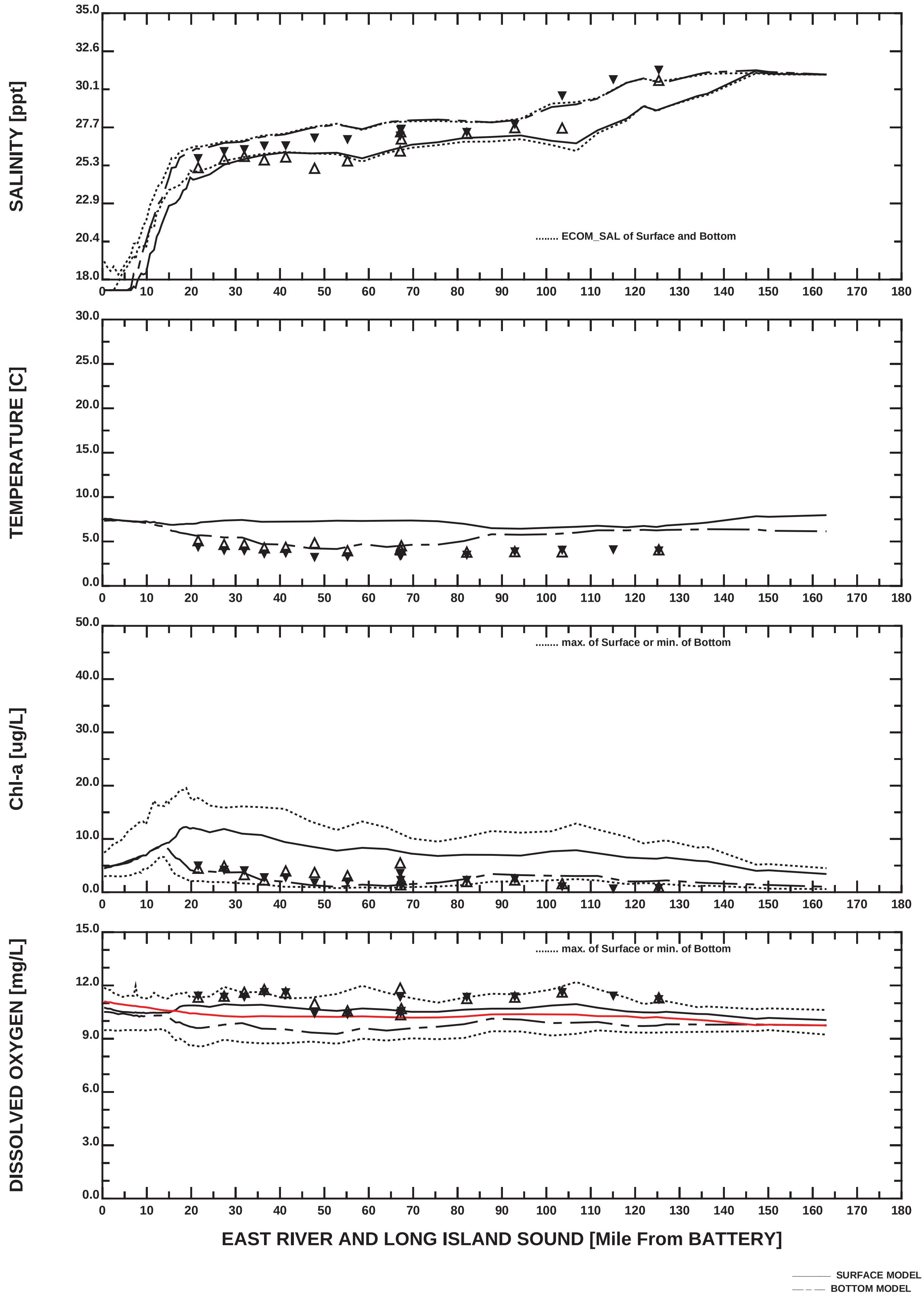


Figure 88. Salinity, temperature, chlorophyll and DO calibration results for April 2001

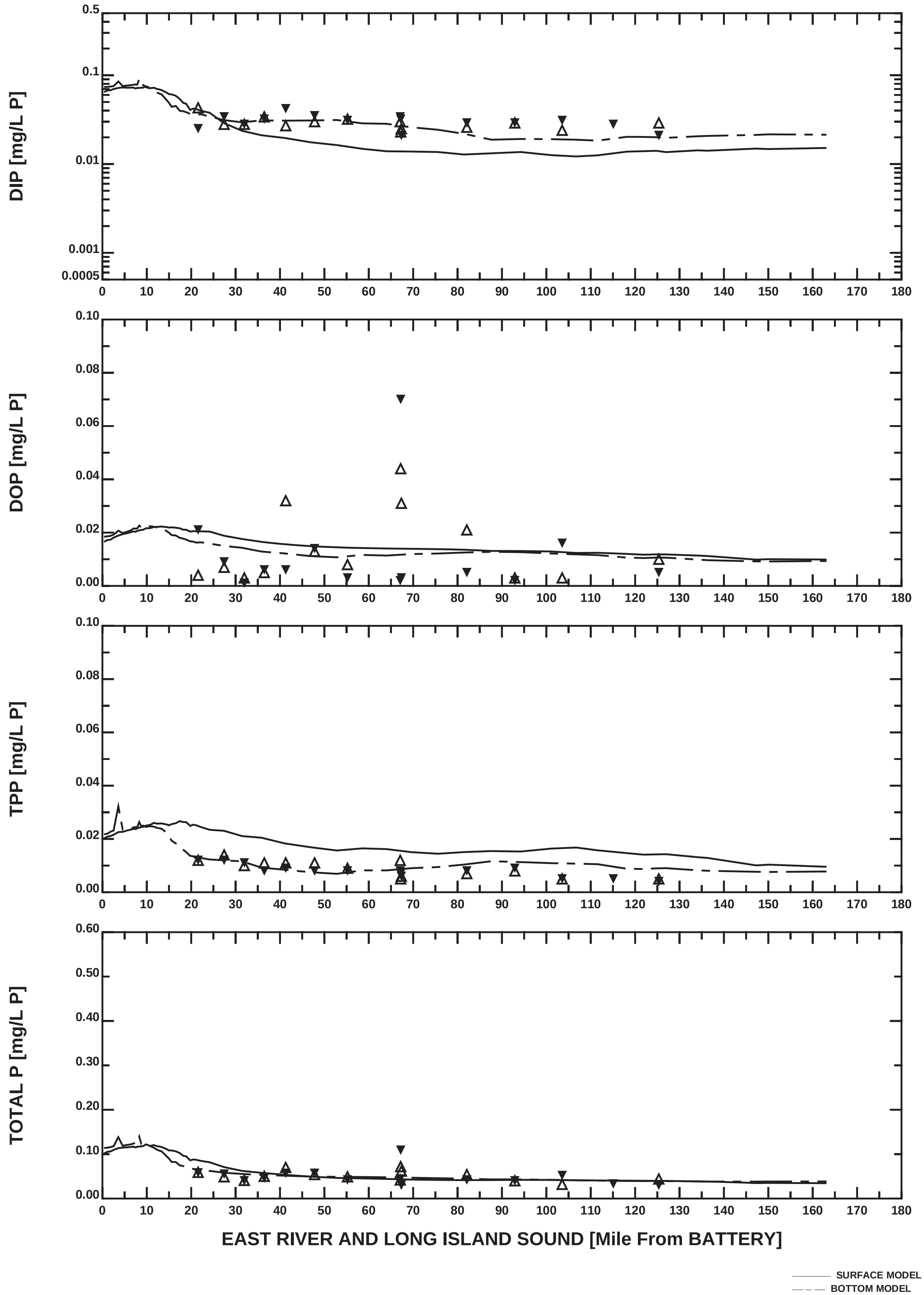


Figure 89. Dissolved inorganic phosphorus, DOP, total particulate phosphorus and total phosphorus calibration results for April 2001

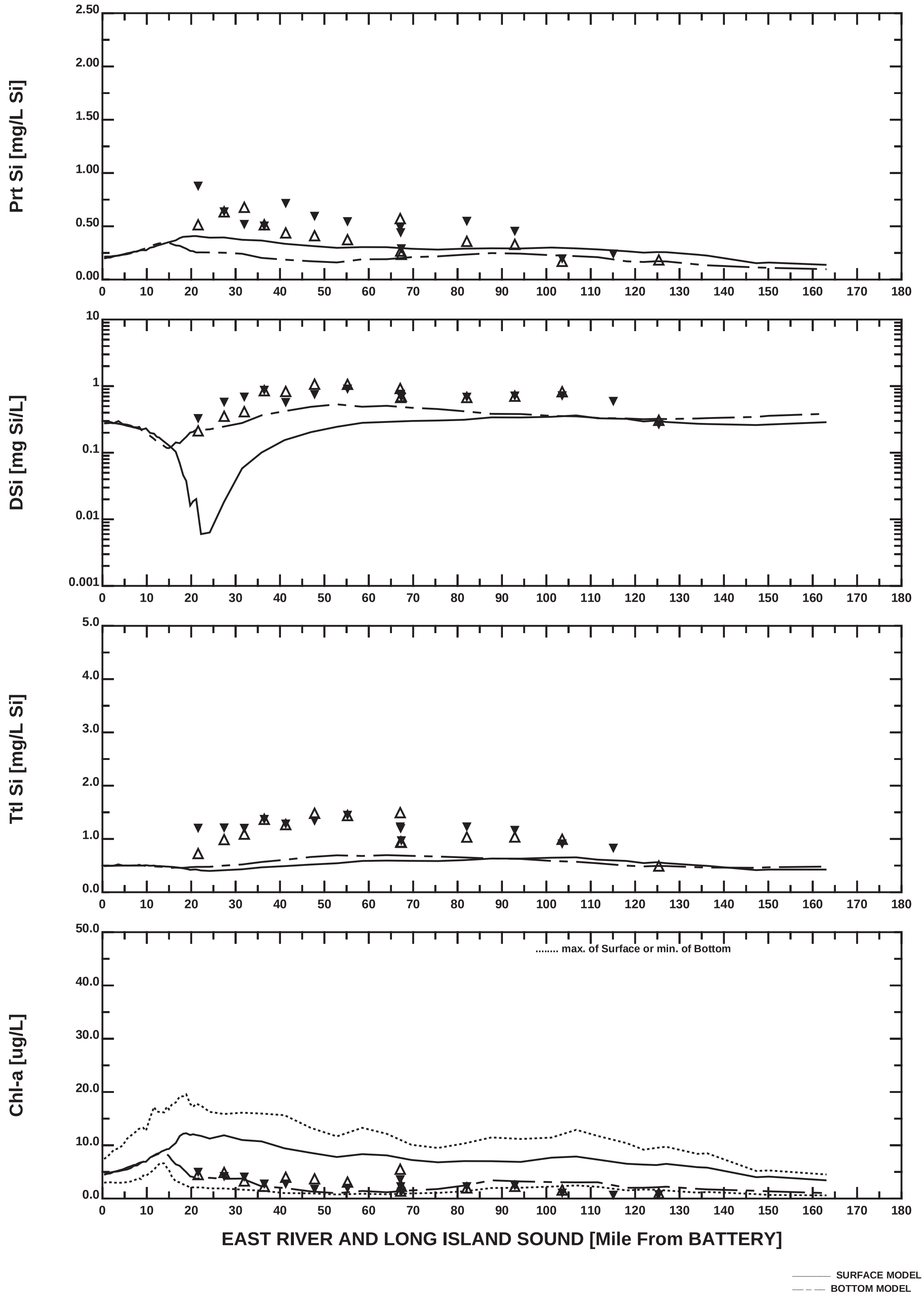


Figure 90. Particulate silica, Si, total silica, and chlorophyll calibration results for April 2001

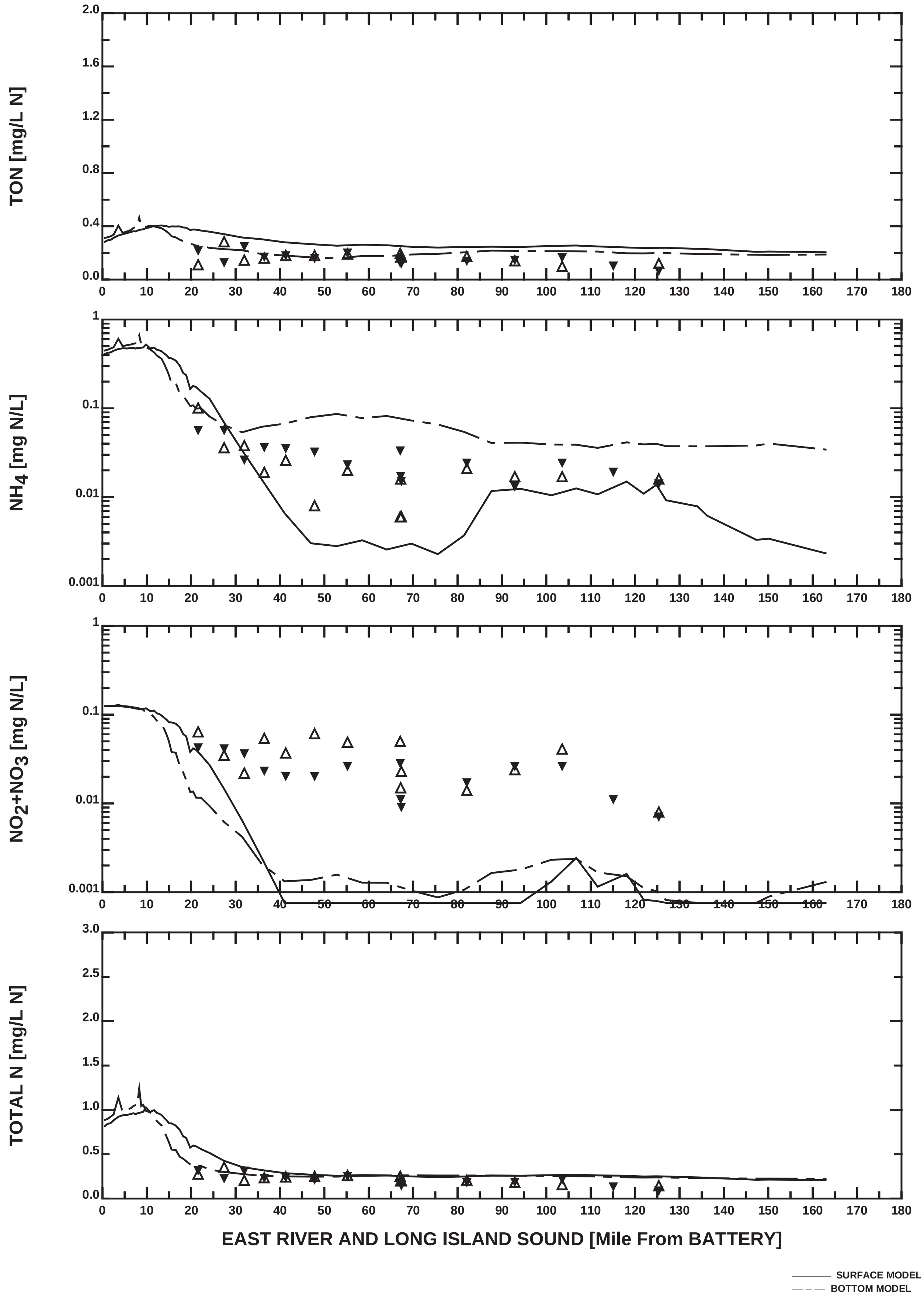


Figure 91. Total organic nitrogen,NH4, NO2+NO3 and total nitrogen calibration results for April 2001

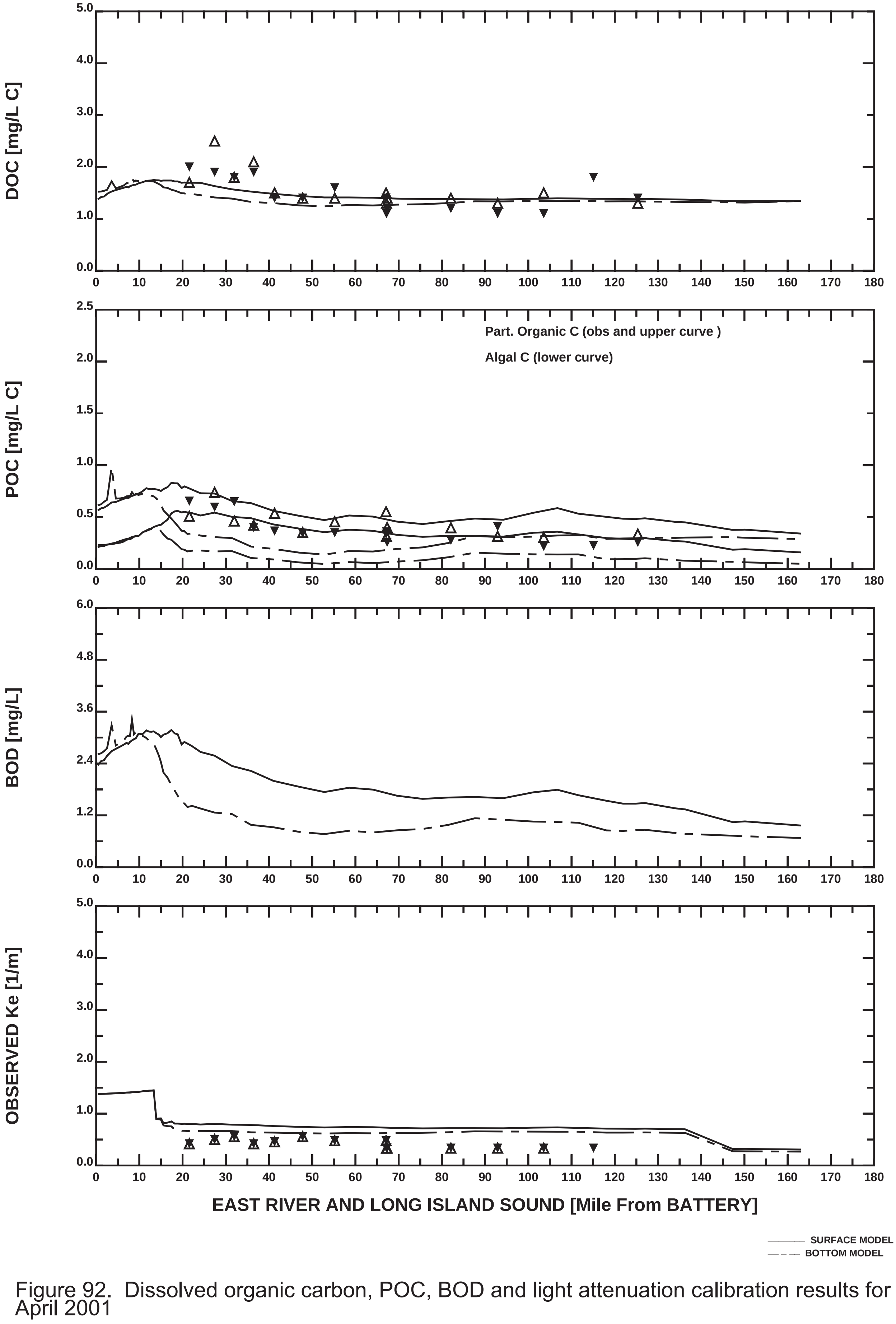


Figure 92. Dissolved organic carbon, POC, BOD and light attenuation calibration results for April 2001

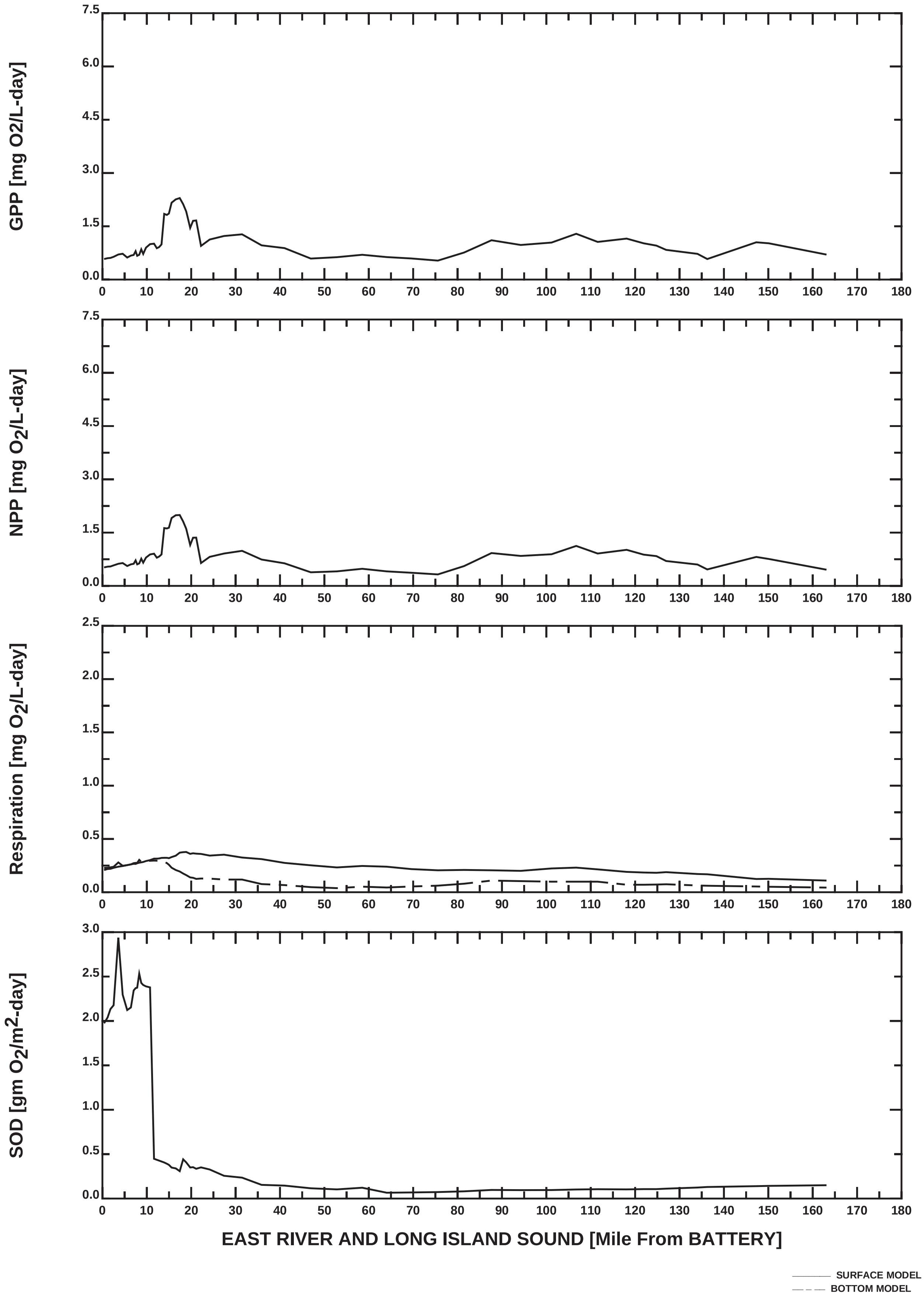


Figure 93. Gross primary production, net primary production, community respiration, and sediment oxygen calibration results for April 2001

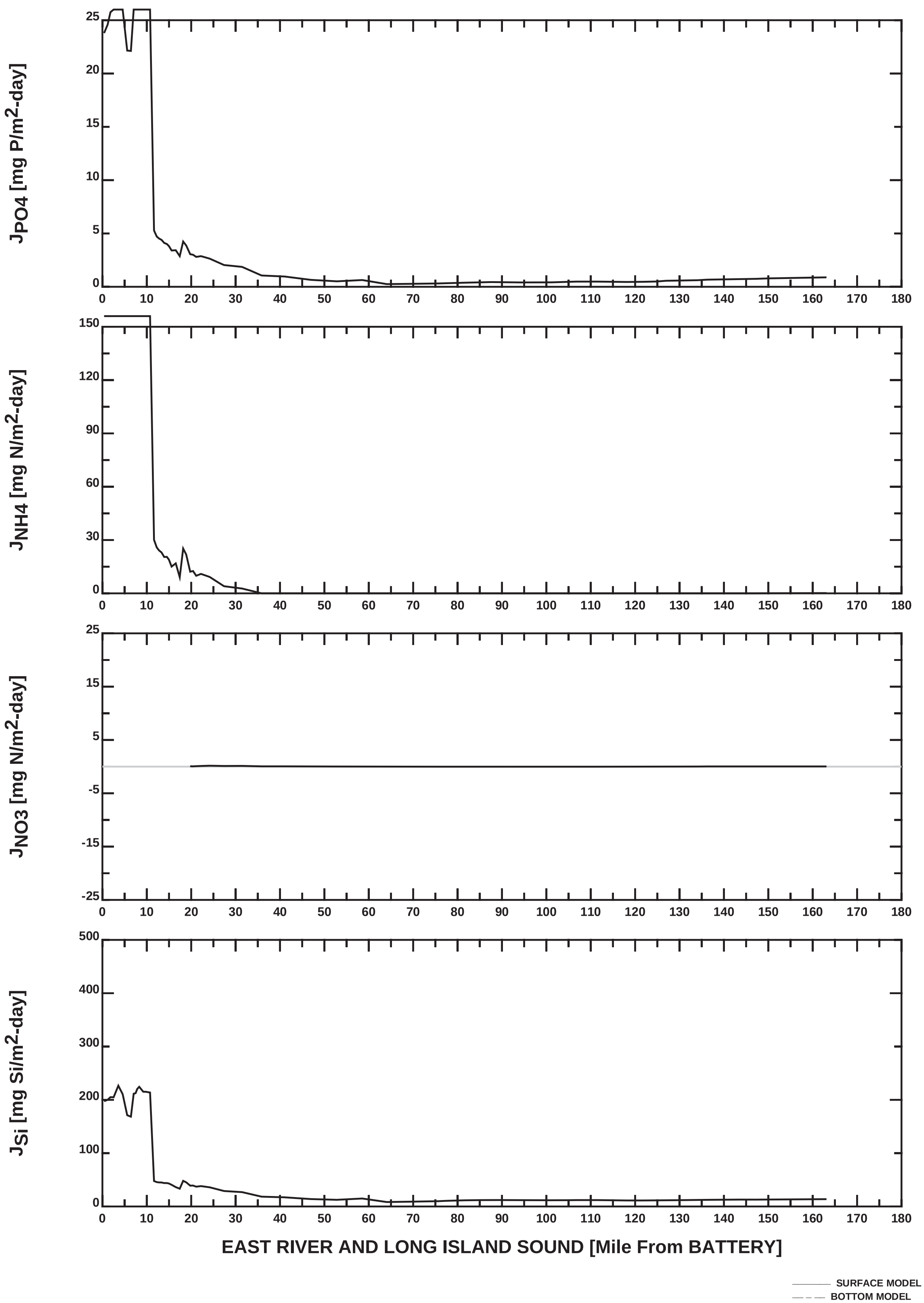


Figure 94. Calibration results for sediment nutrient fluxes of PO4, NH4, NO3 and Si for April 2001

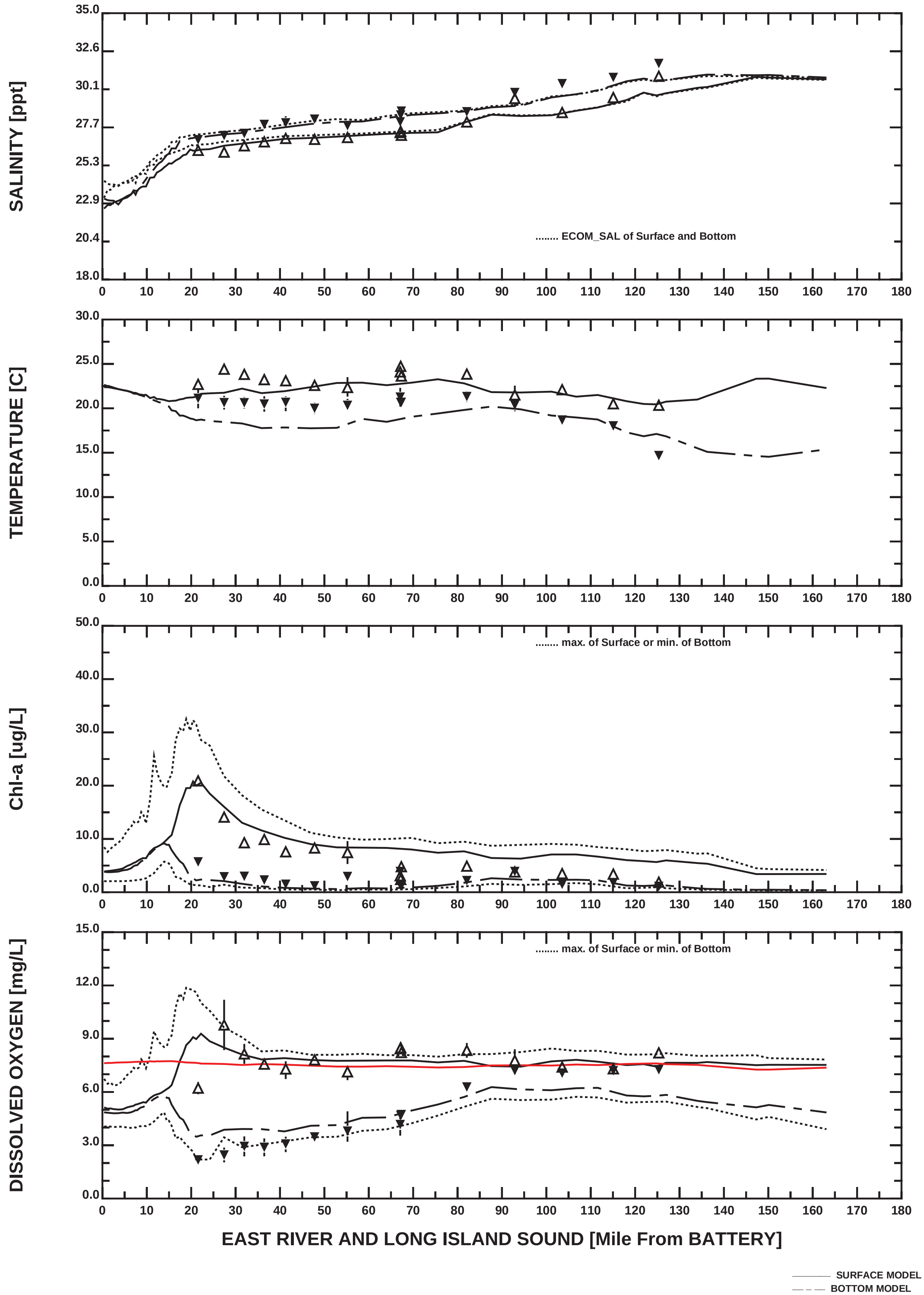


Figure 95. Salinity, temperature, chlorophyll and DO calibration results for August 2001

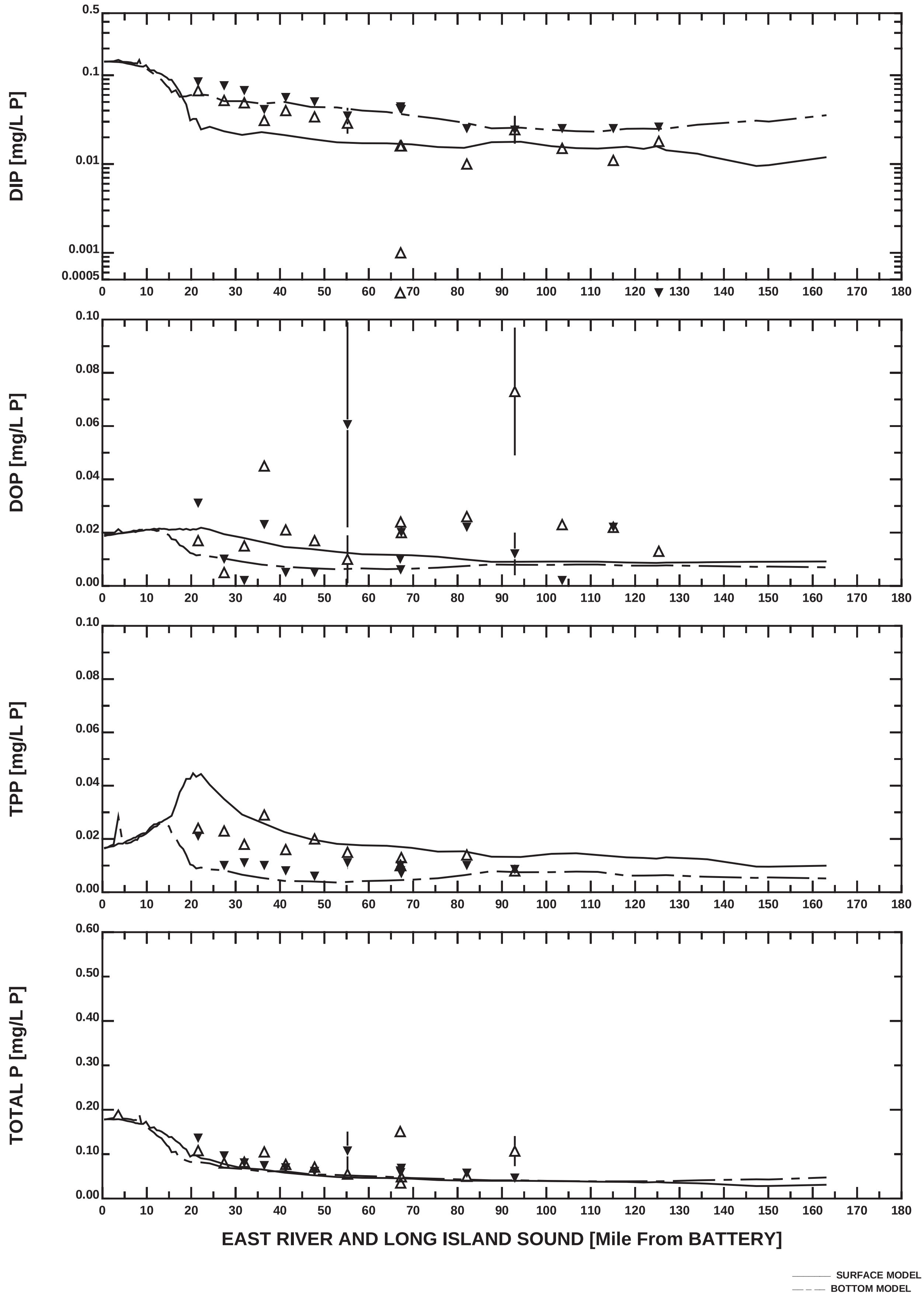


Figure 96. Dissolved inorganic phosphorus, DOP, total particulate phosphorus and total phosphorus calibration results for August 2001

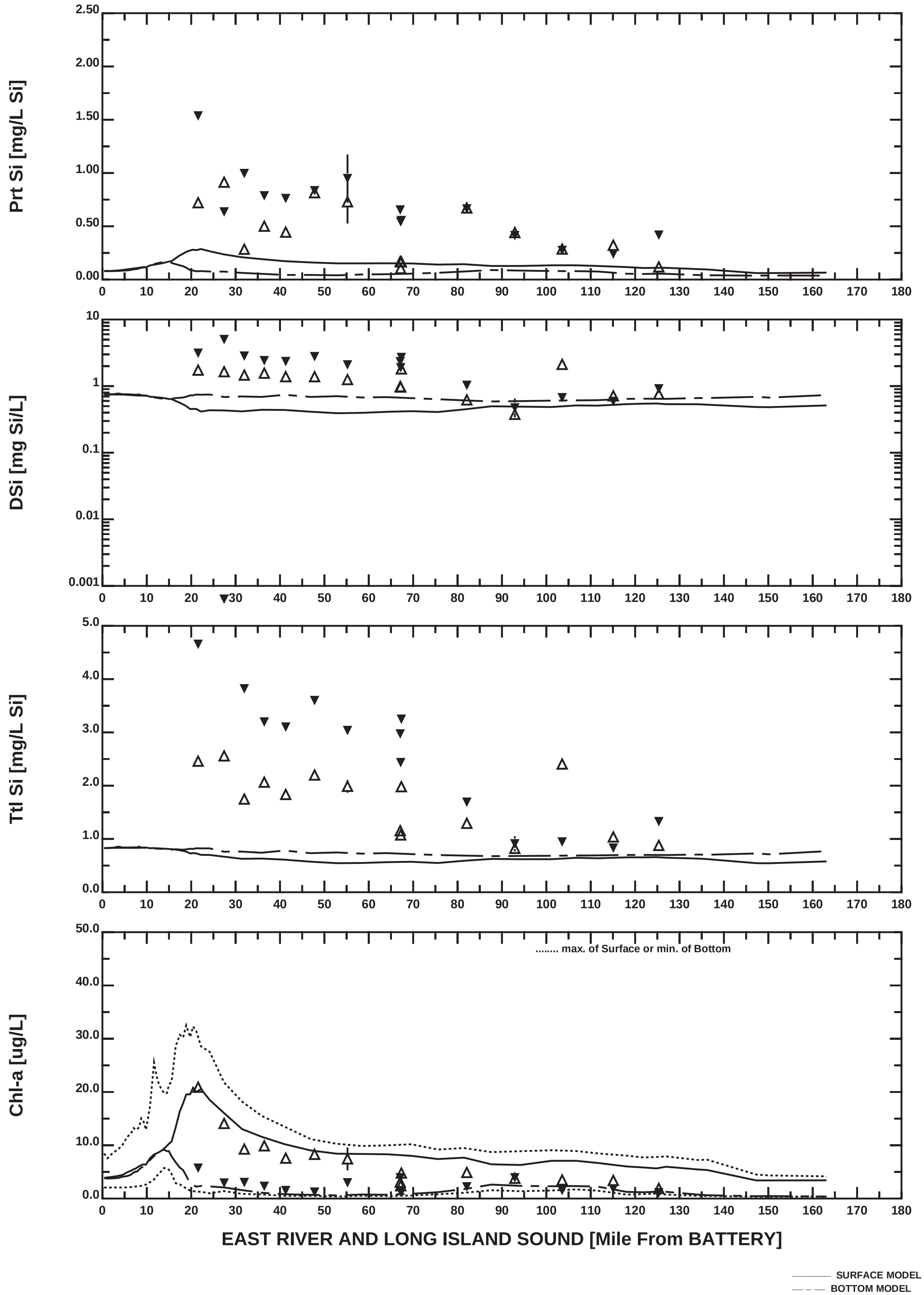


Figure 97. Particulate silica, Si, total silica, and chlorophyll calibration results for August 2001

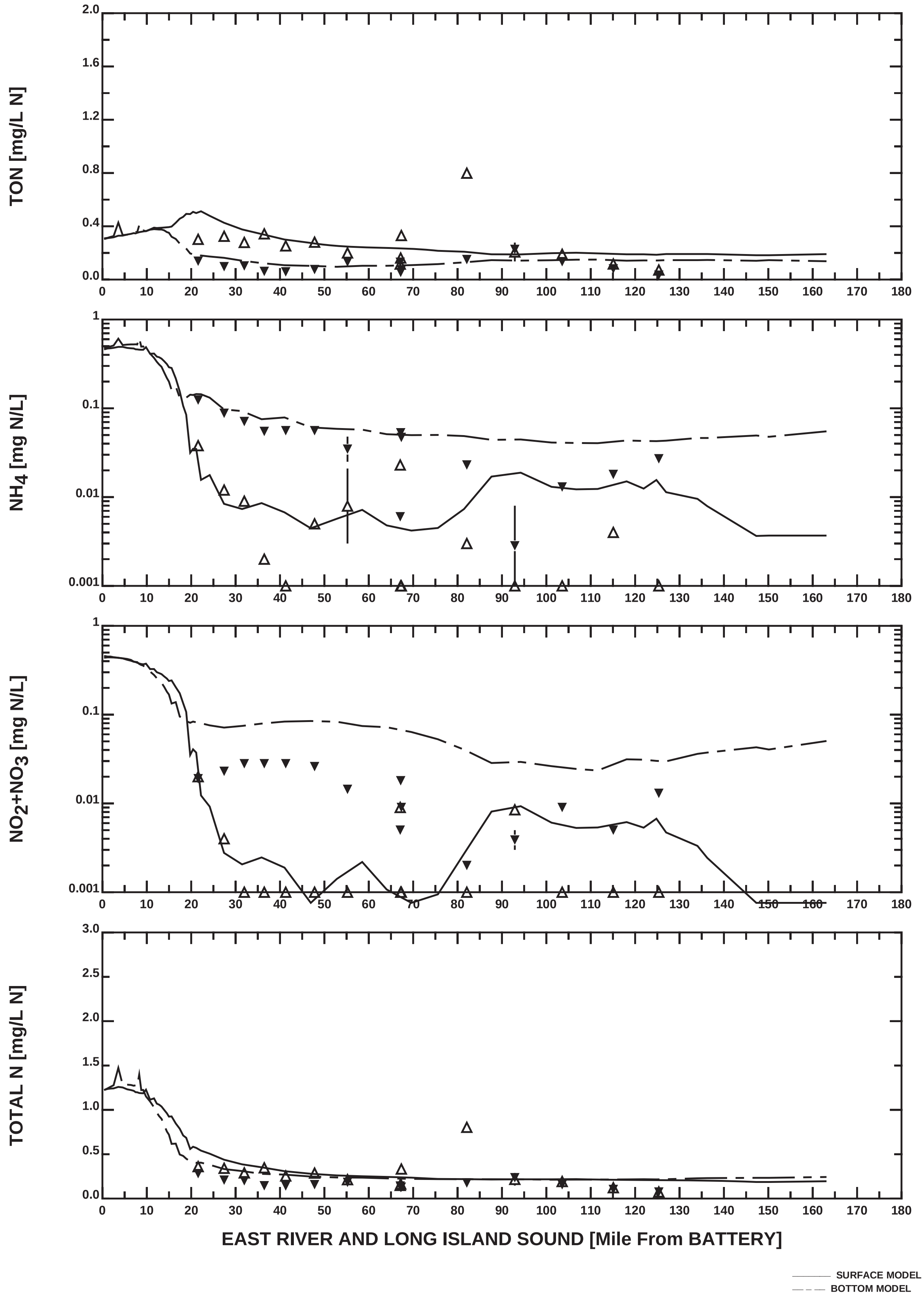


Figure 98. Total organic nitrogen,NH4, NO2+NO3 and total nitrogen calibration results for August 2001

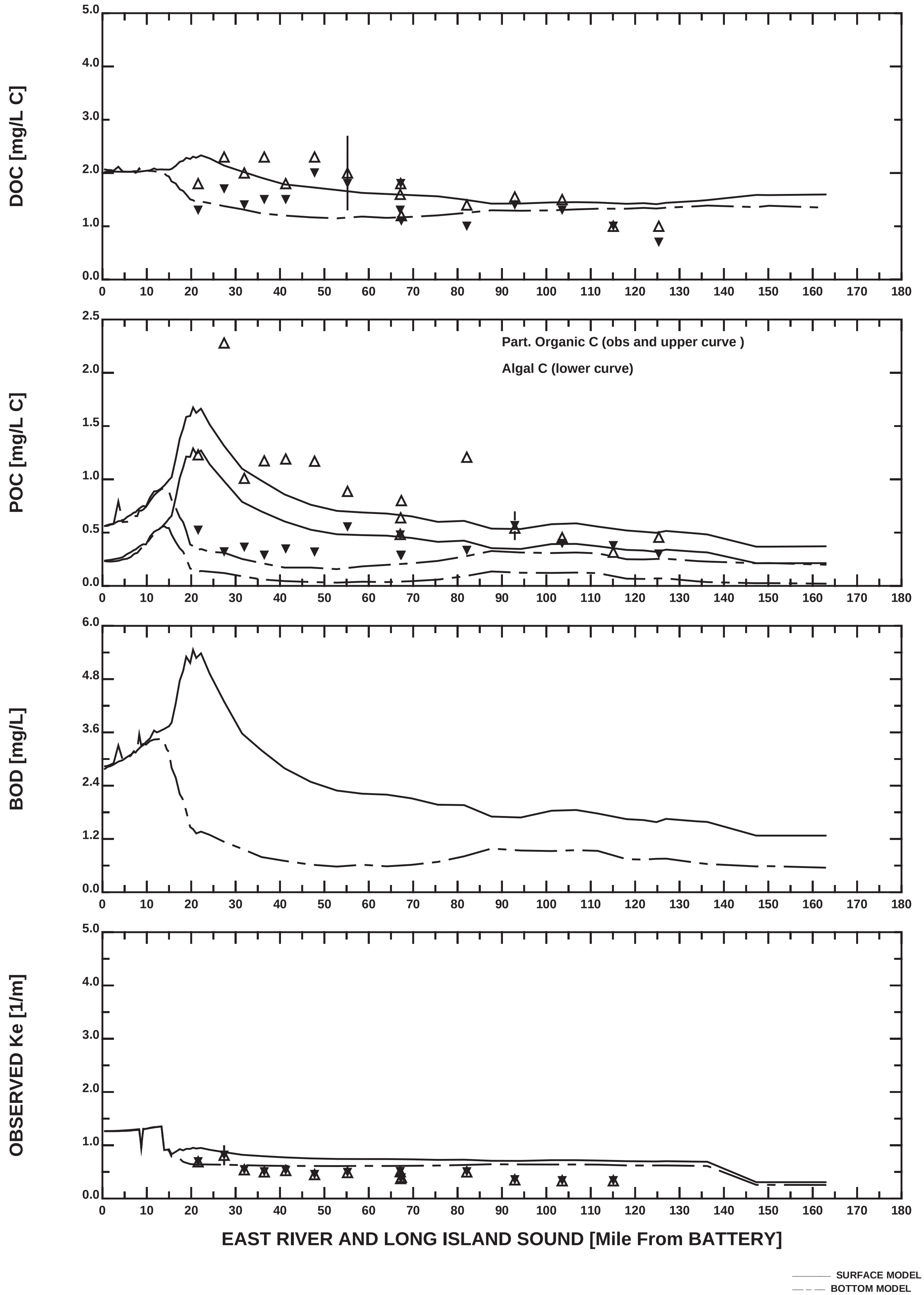


Figure 99. Dissolved organic carbon, POC, BOD and light attenuation calibration results for August 2001

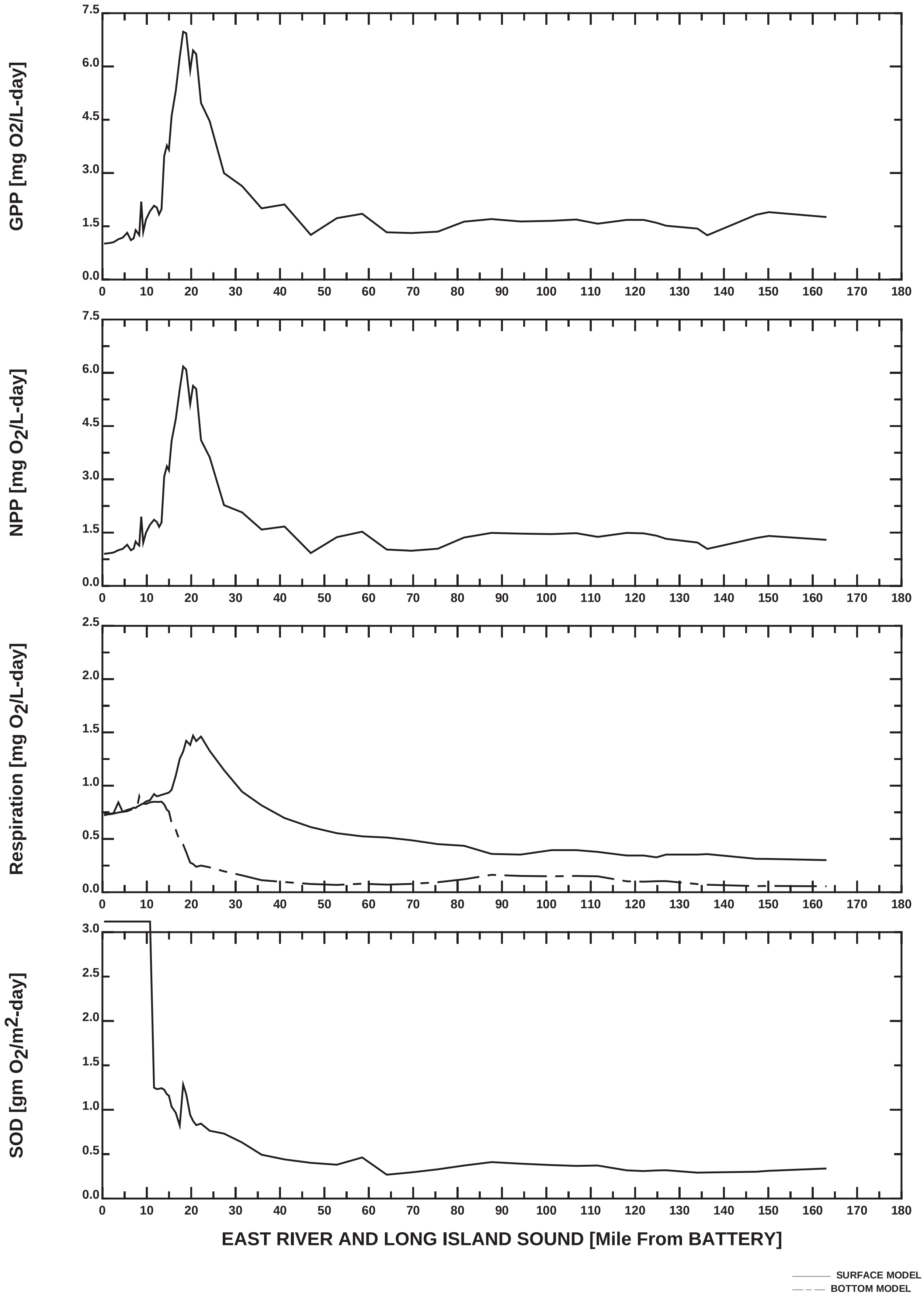


Figure 100. Gross primary production, net primary production, community respiration, and sediment oxygen calibration results for August 2001

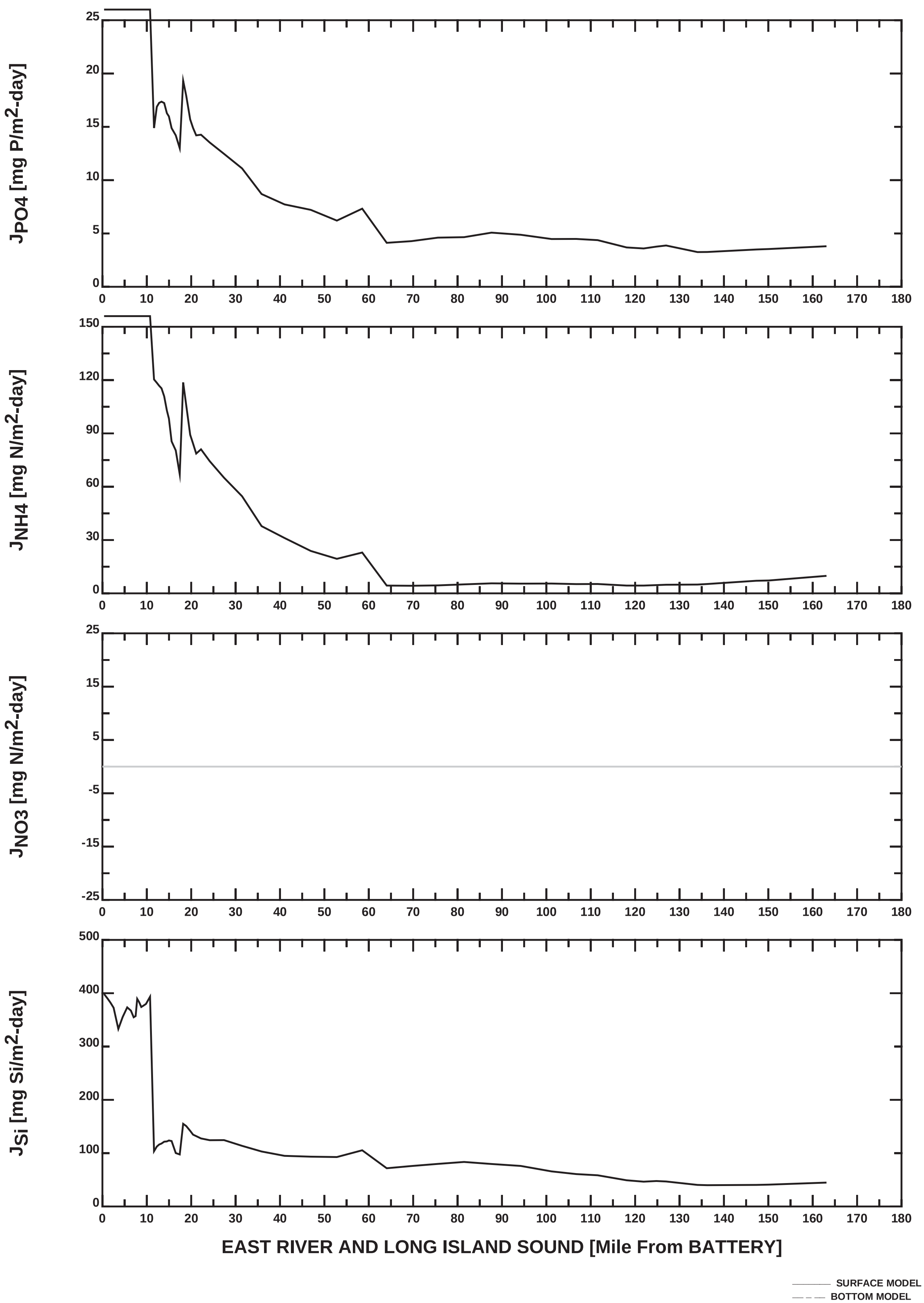


Figure 101. Calibration results for sediment nutrient fluxes of PO4, NH4, NO3 and Si for August 1999

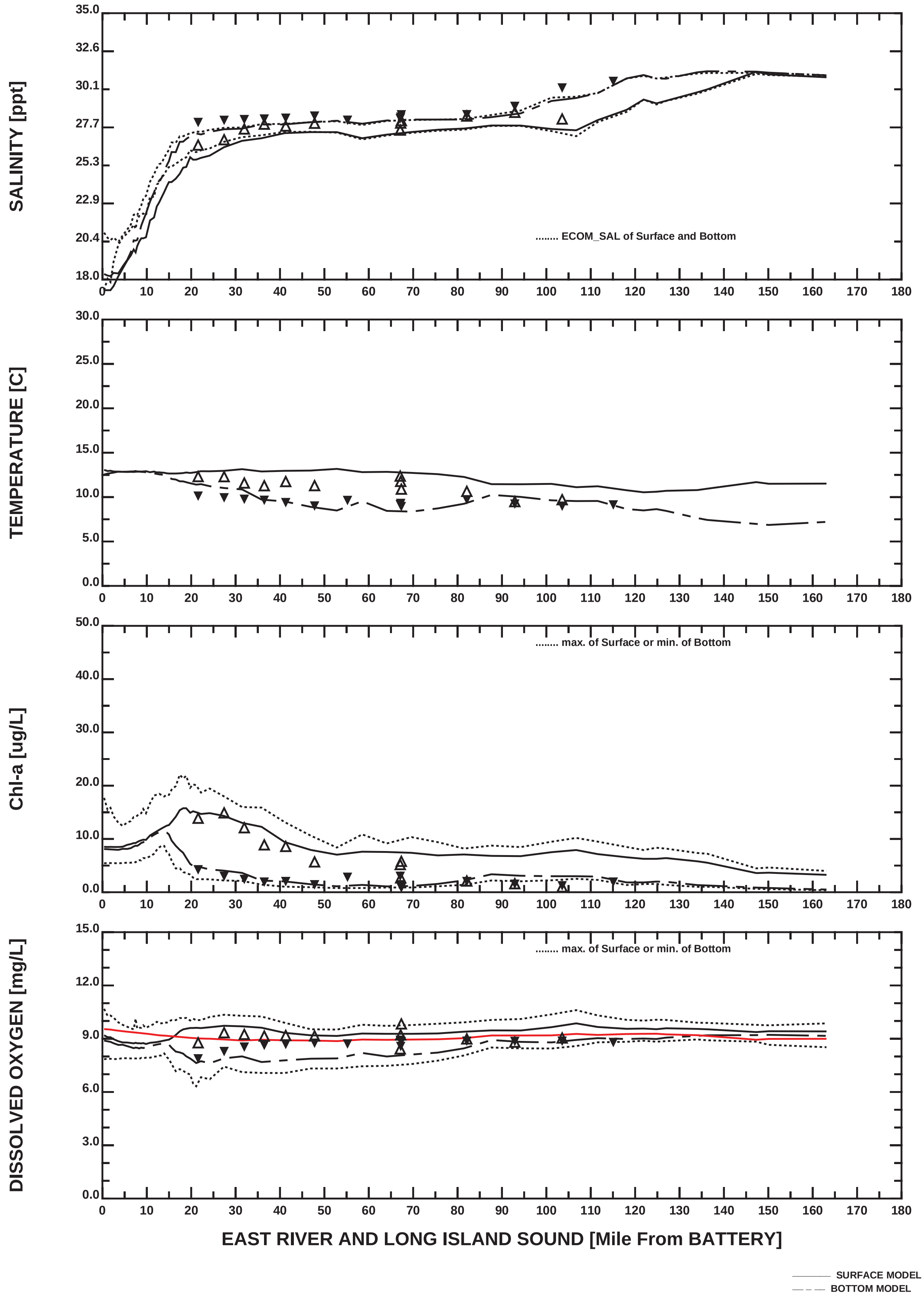


Figure 102. Salinity, temperature, chlorophyll and DO calibration results for May 2002

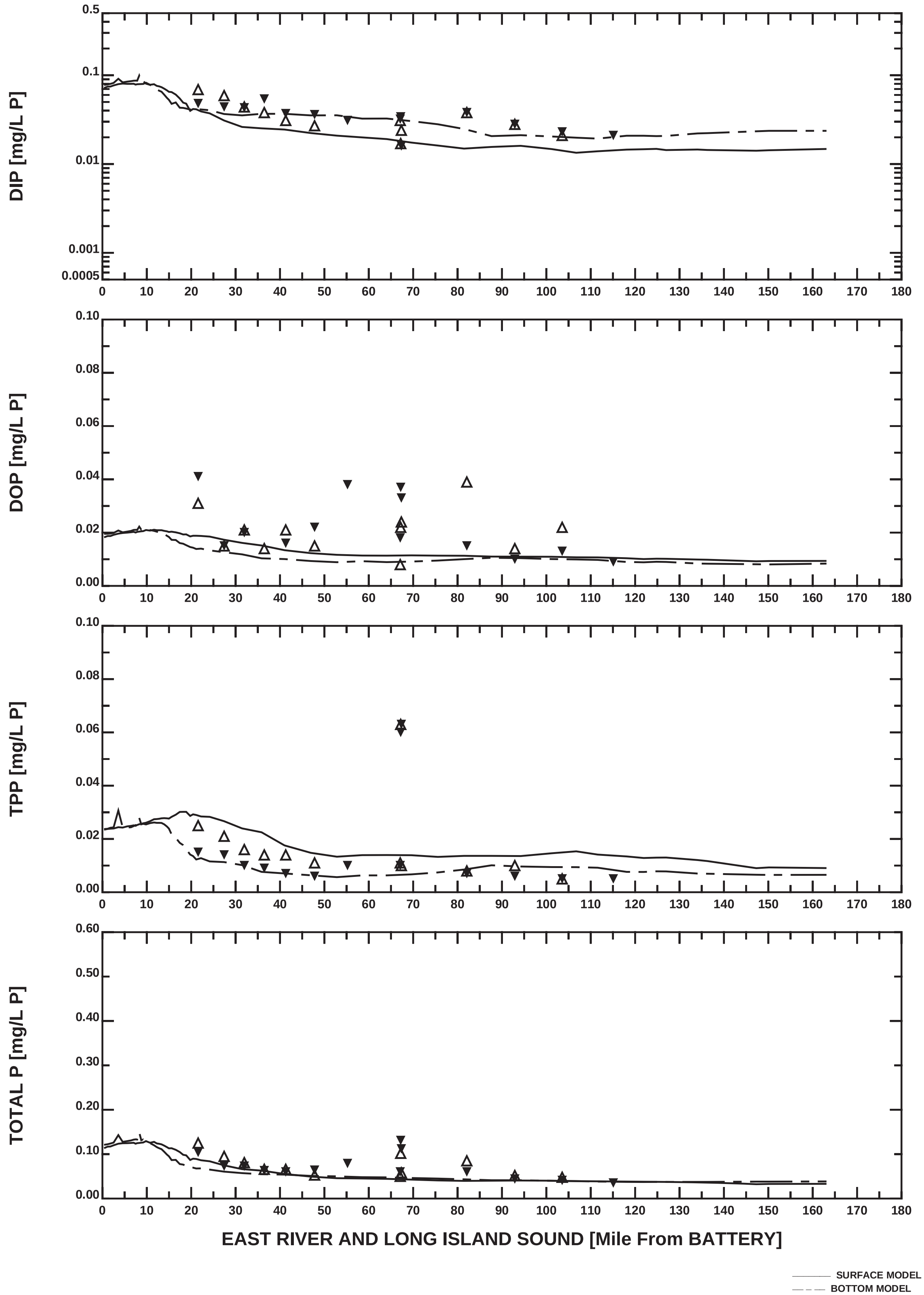


Figure 103. Dissolved inorganic phosphorus, DOP, total particulate phosphorus and total phosphorus calibration results for May 2002

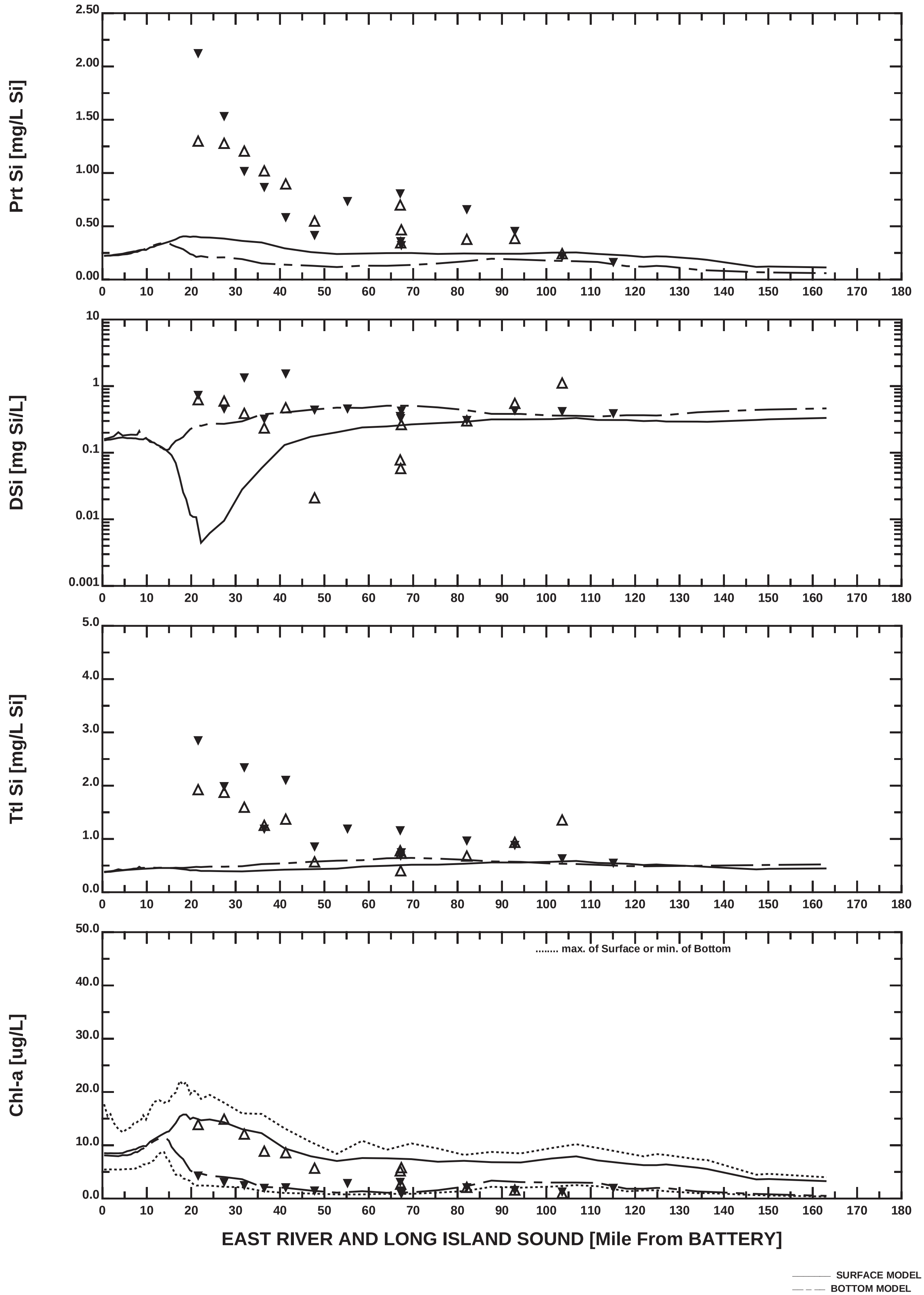


Figure 104. Particulate silica, Si, total silica, and chlorophyll calibration results for May 2002

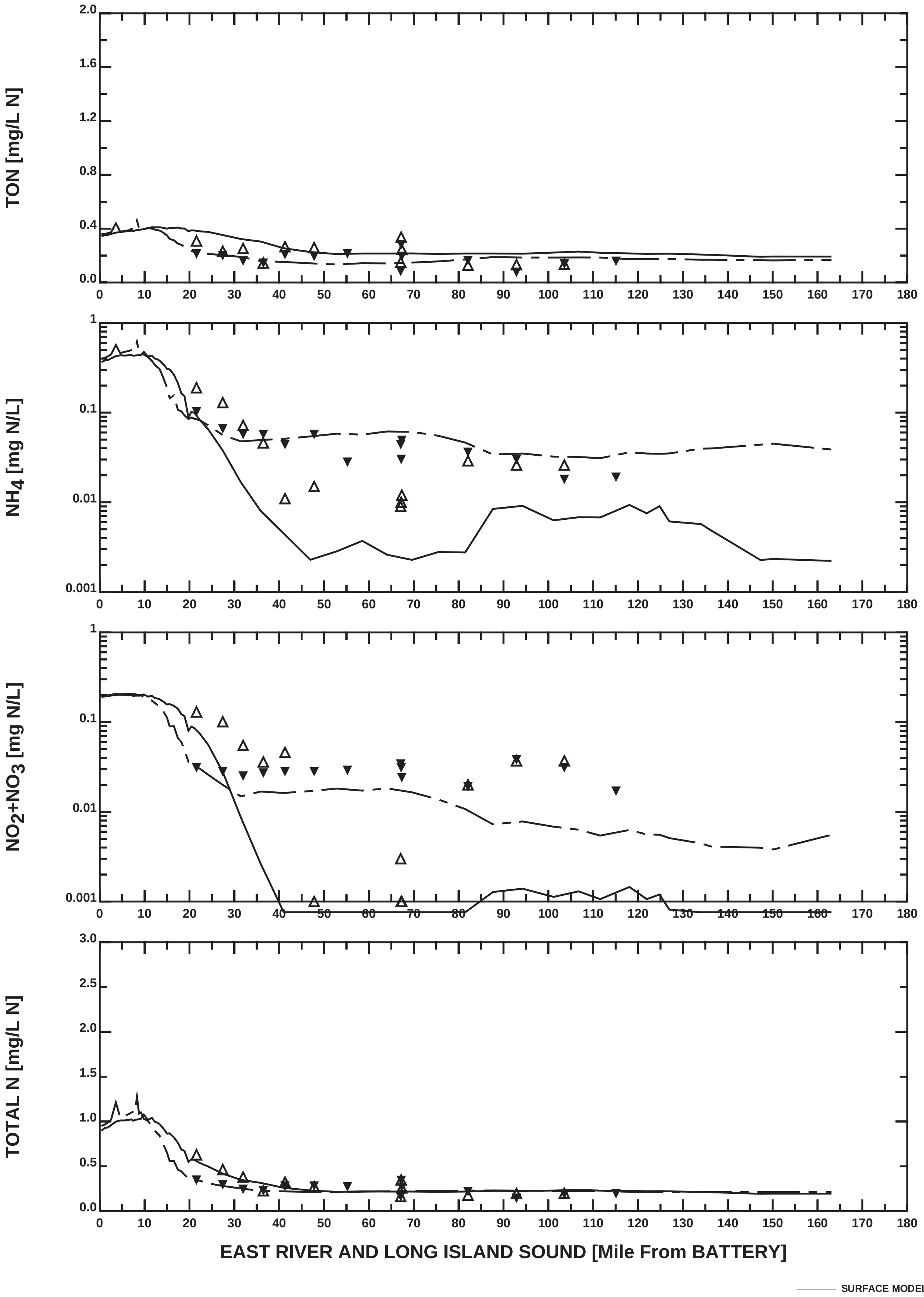


Figure 105. Total organic nitrogen,NH4, NO2+NO3 and total nitrogen calibration results for May 2002

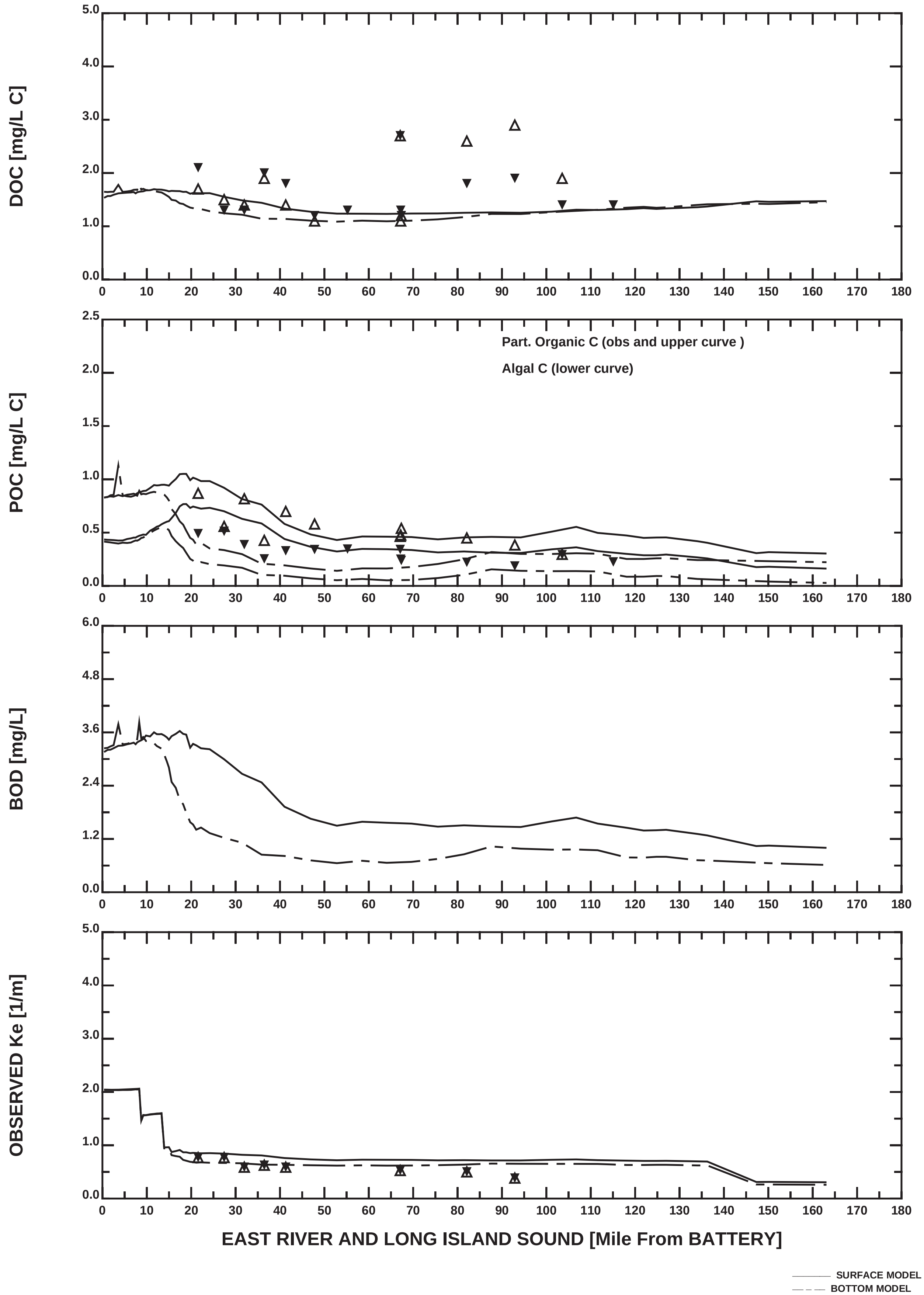


Figure 106. Dissolved organic carbon, POC, BOD and light attenuation calibration results for May 2002

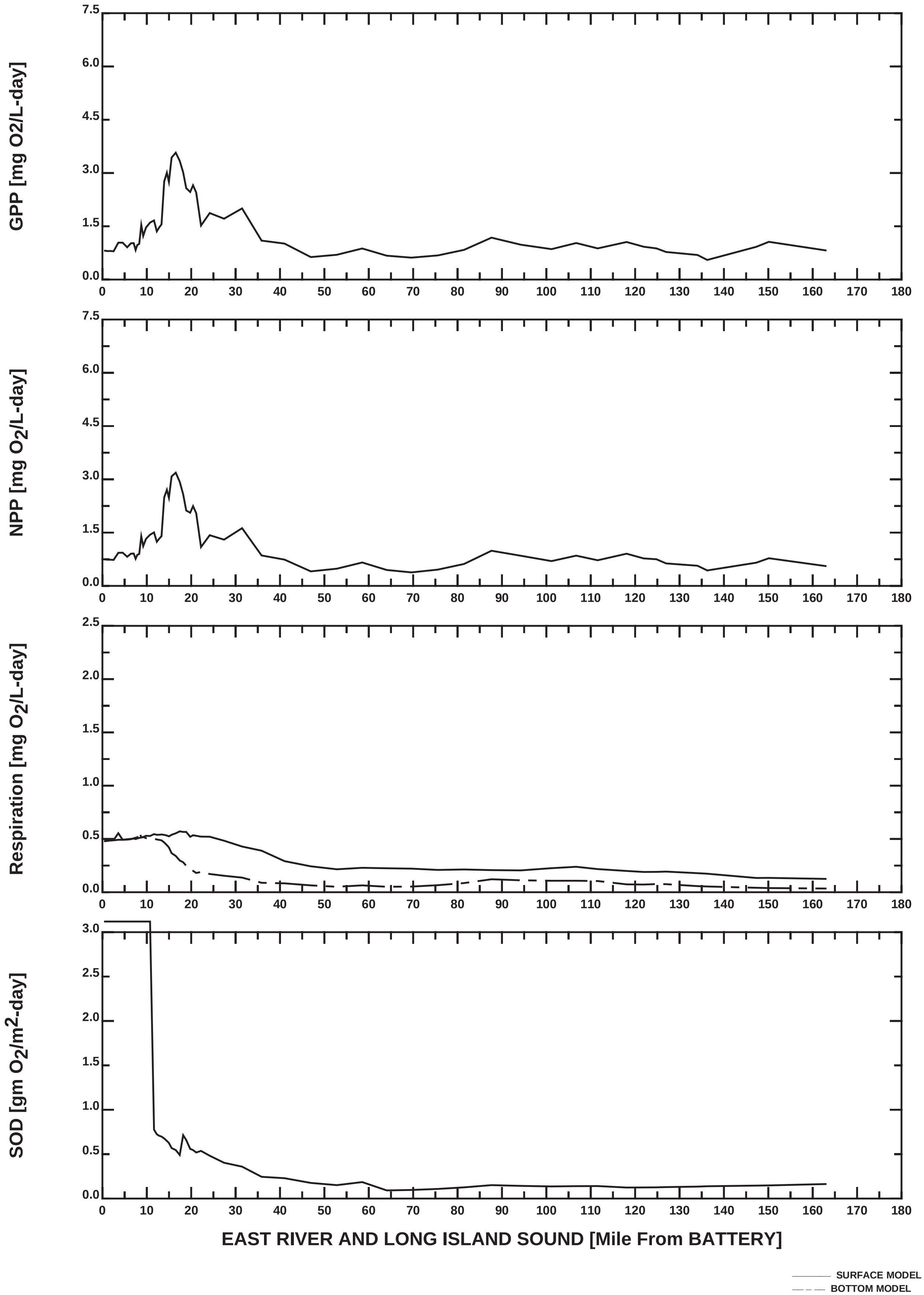


Figure 107. Gross primary production, net primary production, community respiration, and sediment oxygen calibration results for May 2002

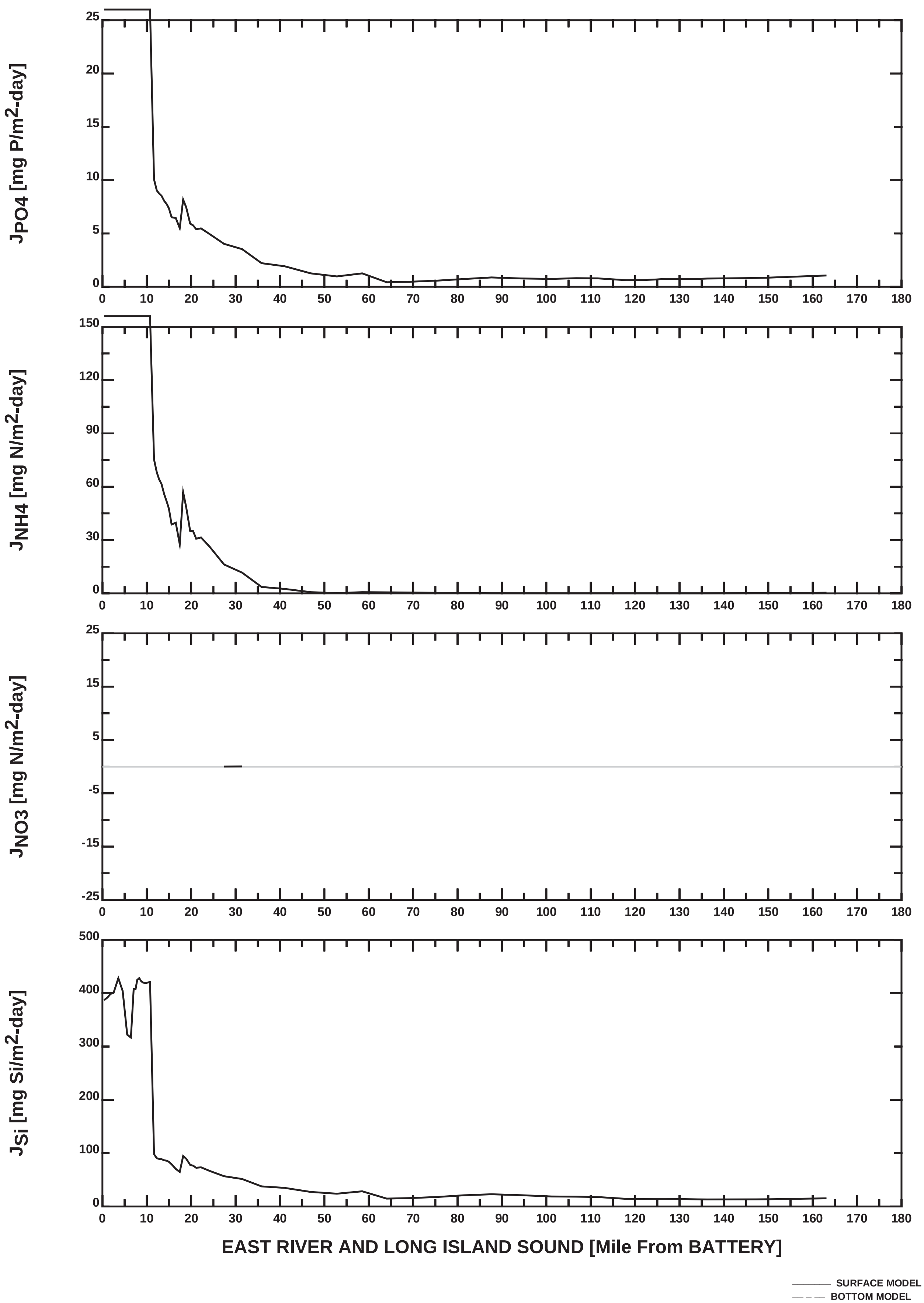


Figure 108. Calibration results for sediment nutrient fluxes of PO4, NH4, NO3 and Si for May 2002

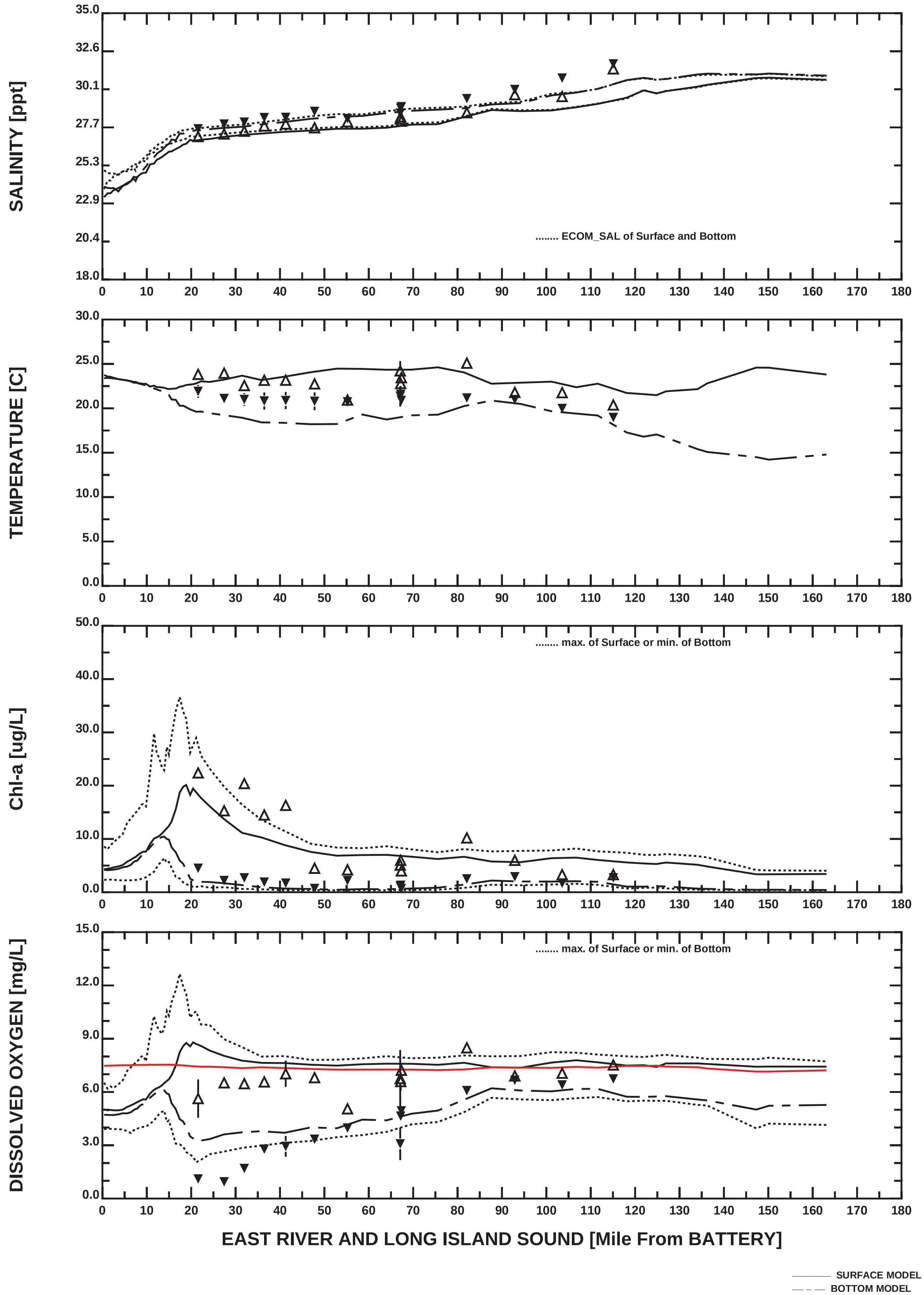


Figure 109. Salinity, temperature, chlorophyll and DO calibration results for August 2002

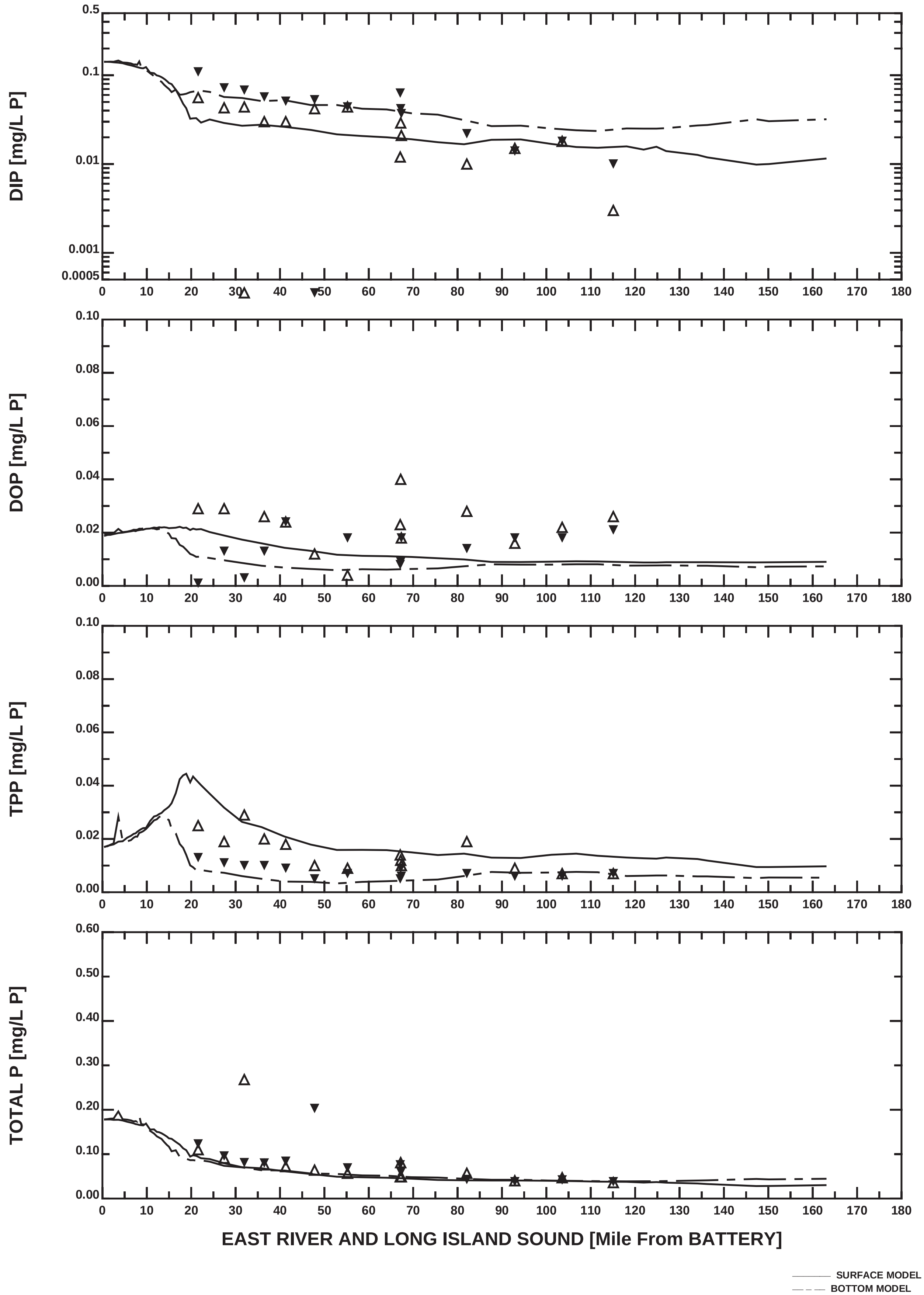


Figure 110. Dissolved inorganic phosphorus, DOP, total particulate phosphorus and total phosphorus calibration results for August 2002

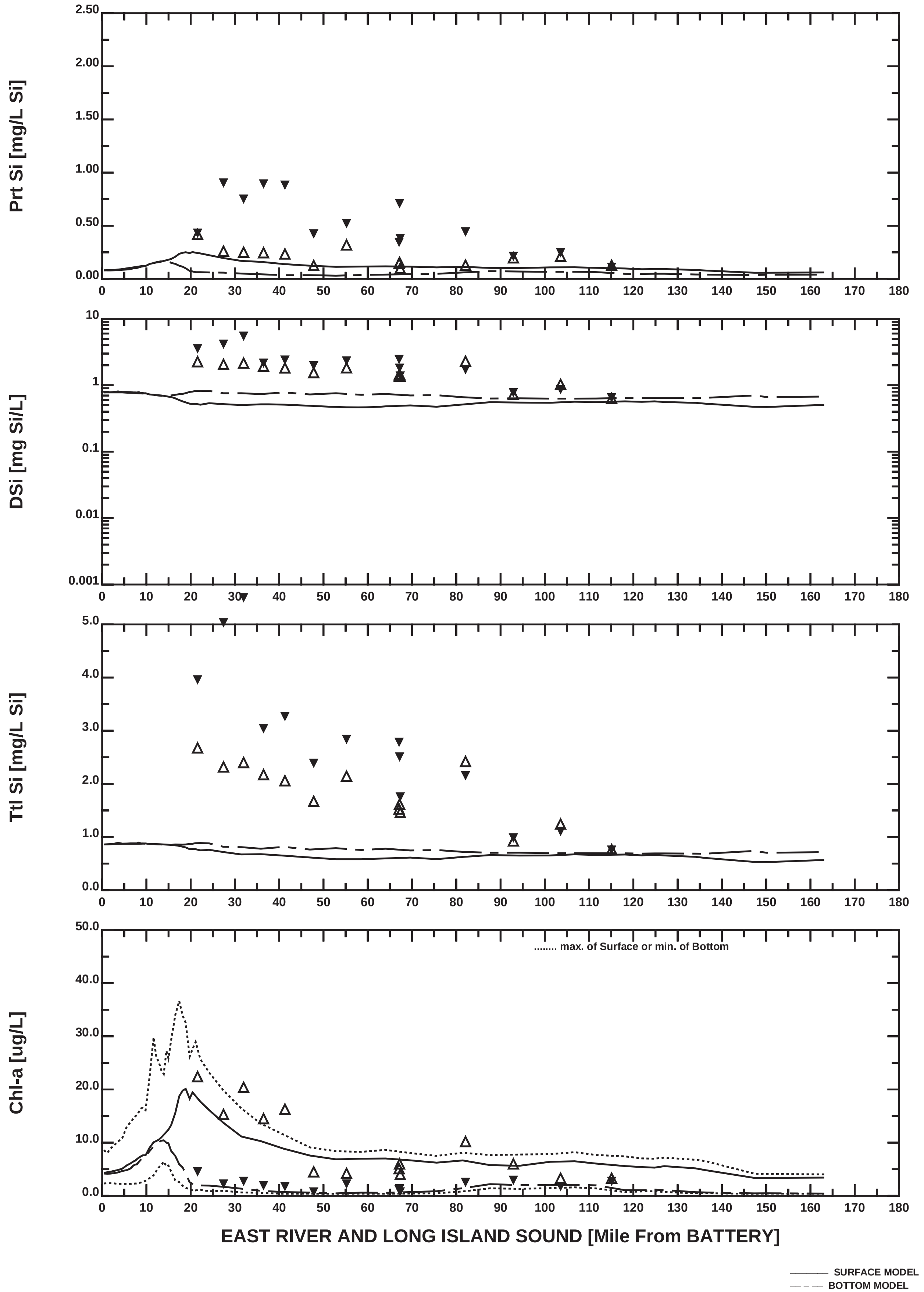


Figure 111. Particulate silica, Si, total silica, and chlorophyll calibration results for August 2002

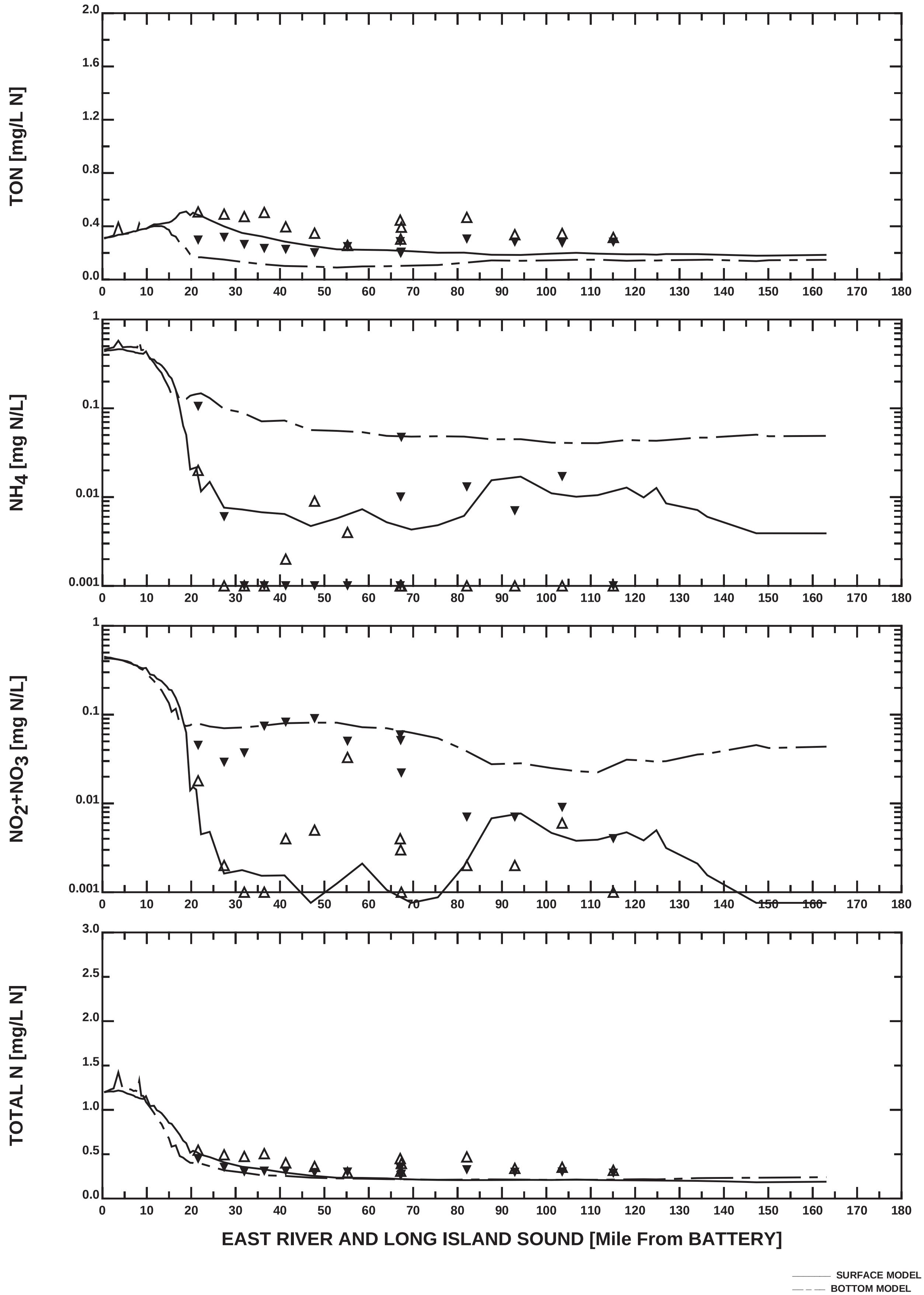


Figure 112. Total organic nitrogen,NH4, NO2+NO3 and total nitrogen calibration results for August 2002

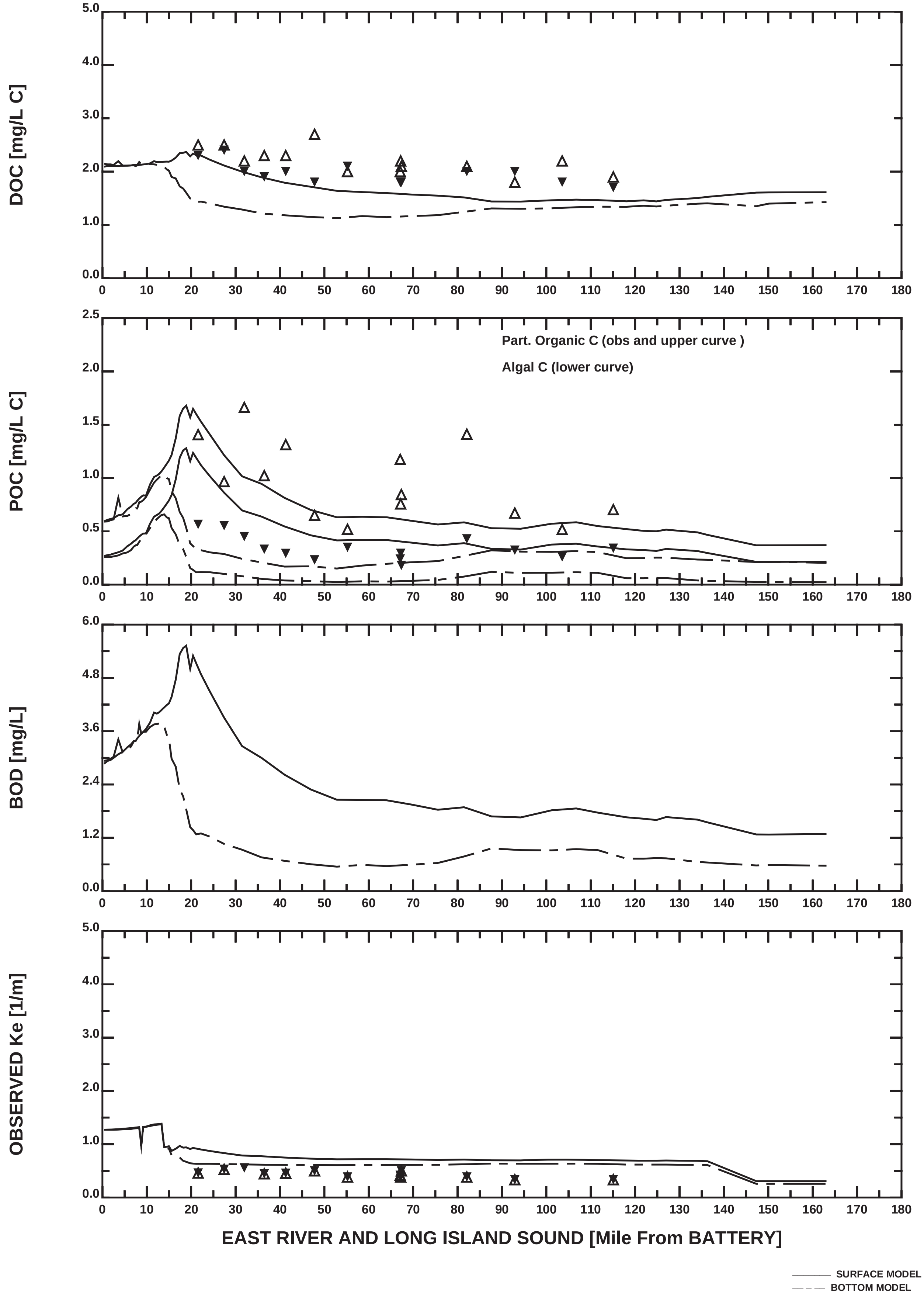


Figure 113. Dissolved organic carbon, POC, BOD and light attenuation calibration results for August 2002

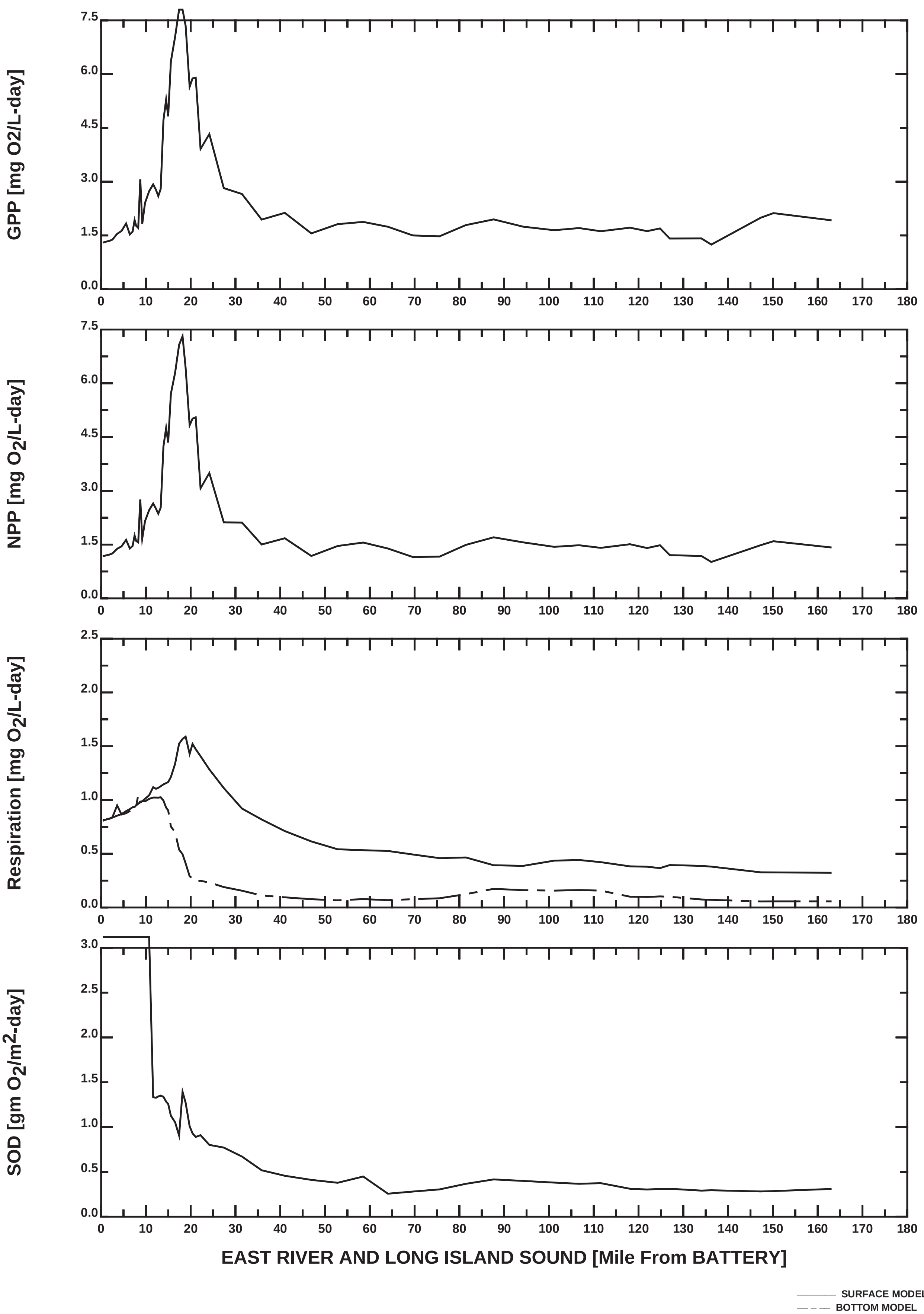


Figure 114. Gross primary production, net primary production, community respiration, and sediment oxygen calibration results for August 2002

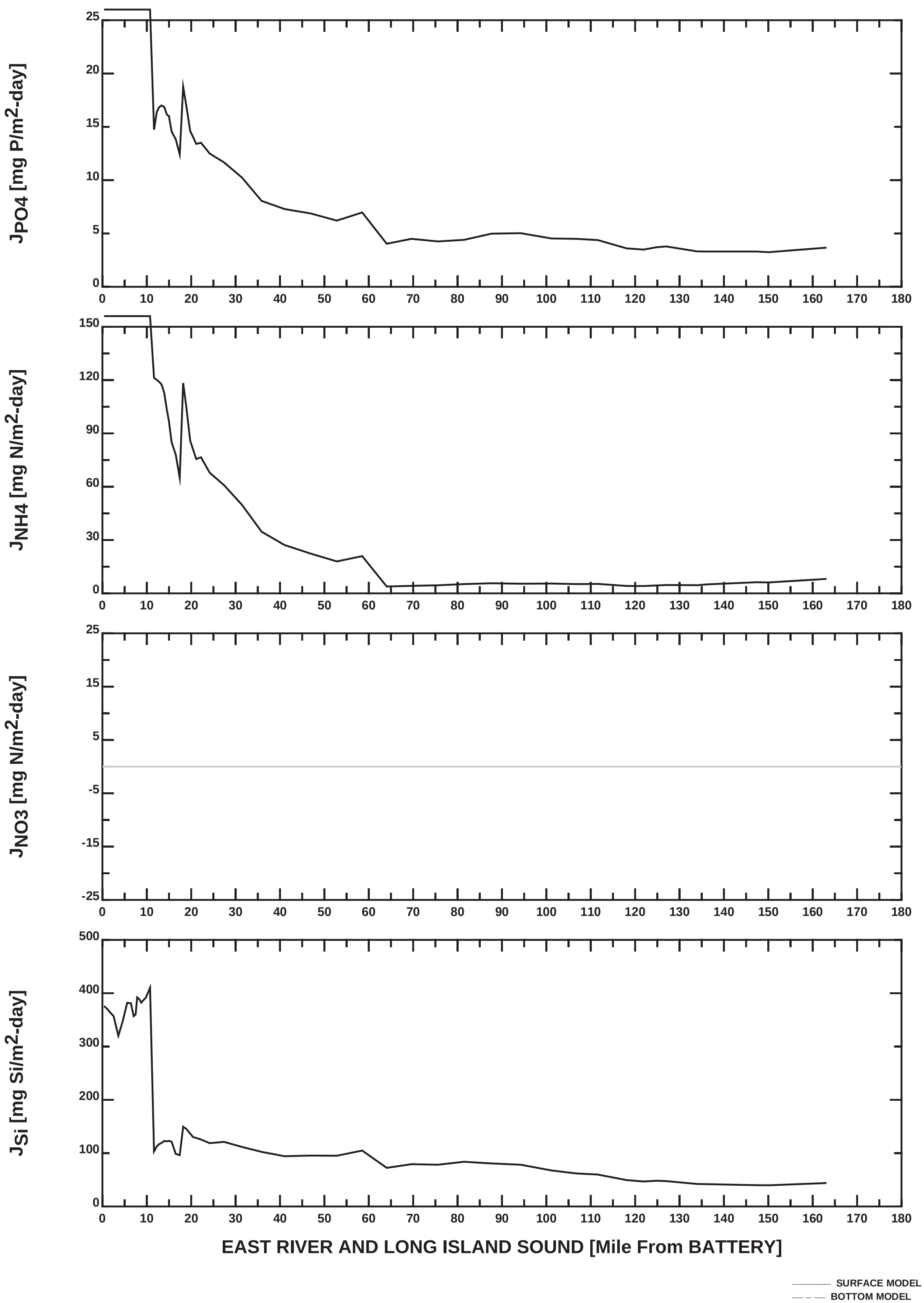
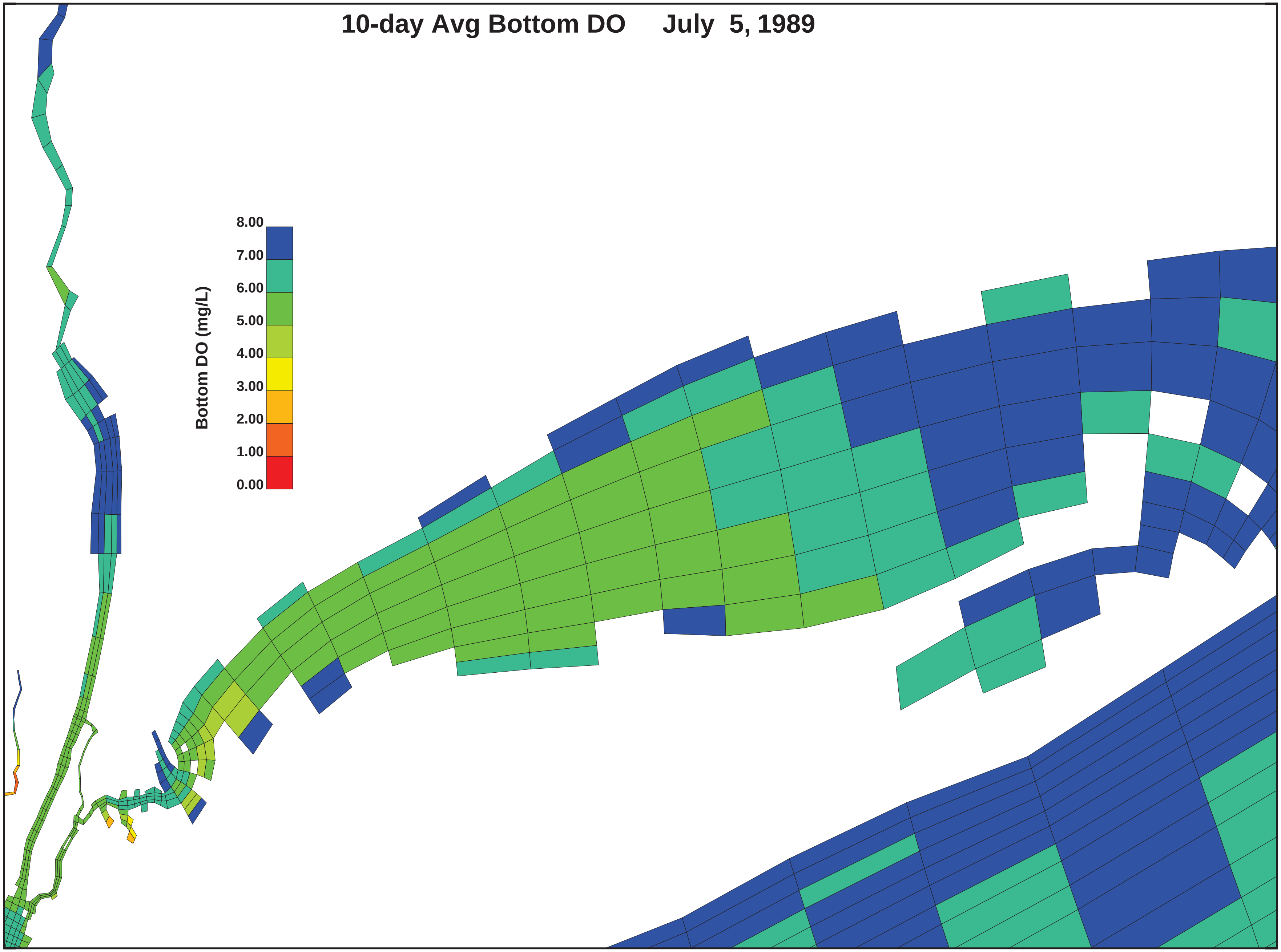


Figure 115. Calibration results for sediment nutrient fluxes of PO4, NH4, NO3 and Si for August 2002


10-day Avg Bottom DO July 5, 1989
Bottom DO (mg/L)
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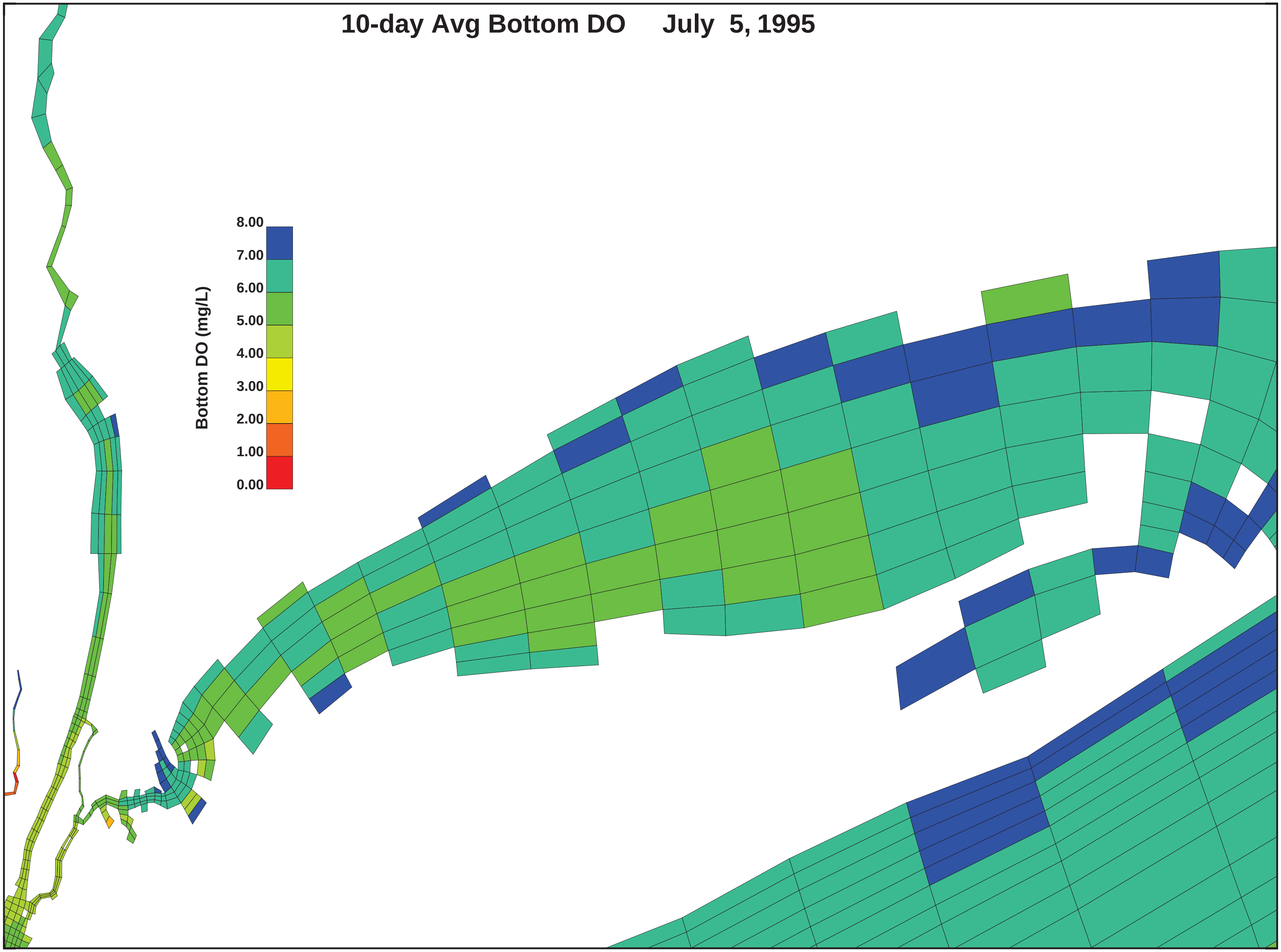

10-day Avg Bottom DO July 5, 1995
Bottom DO (mg/L)
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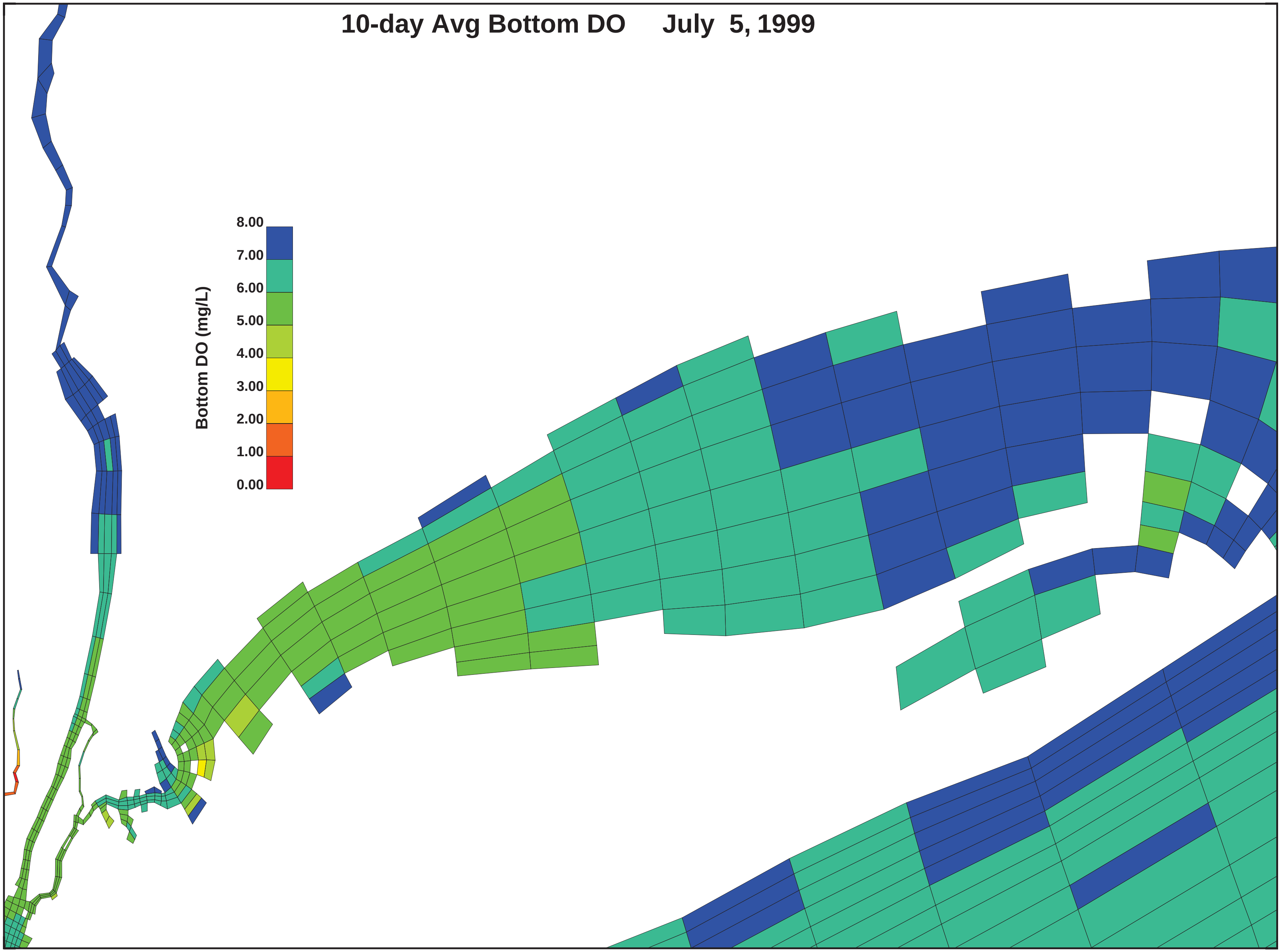

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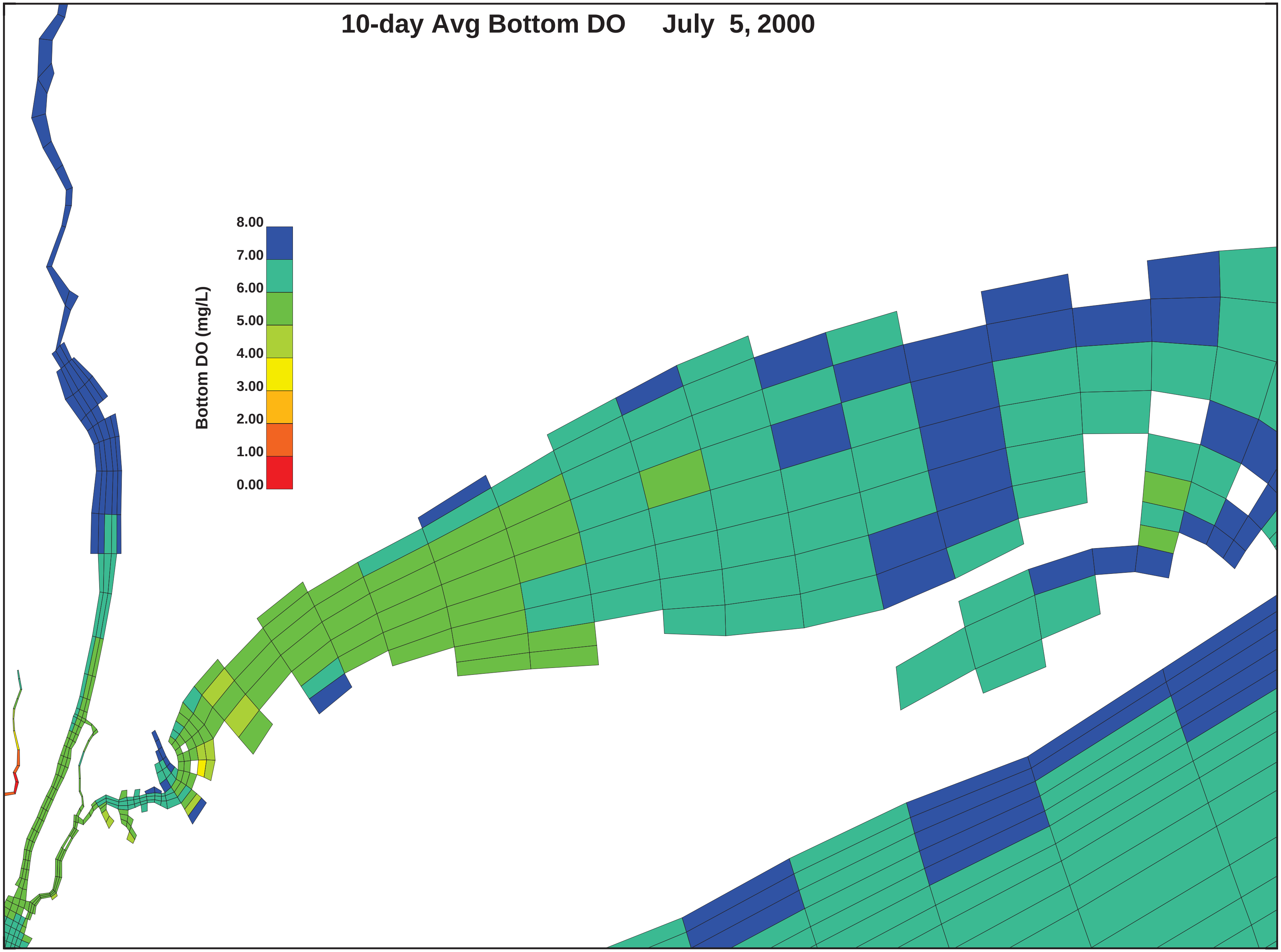
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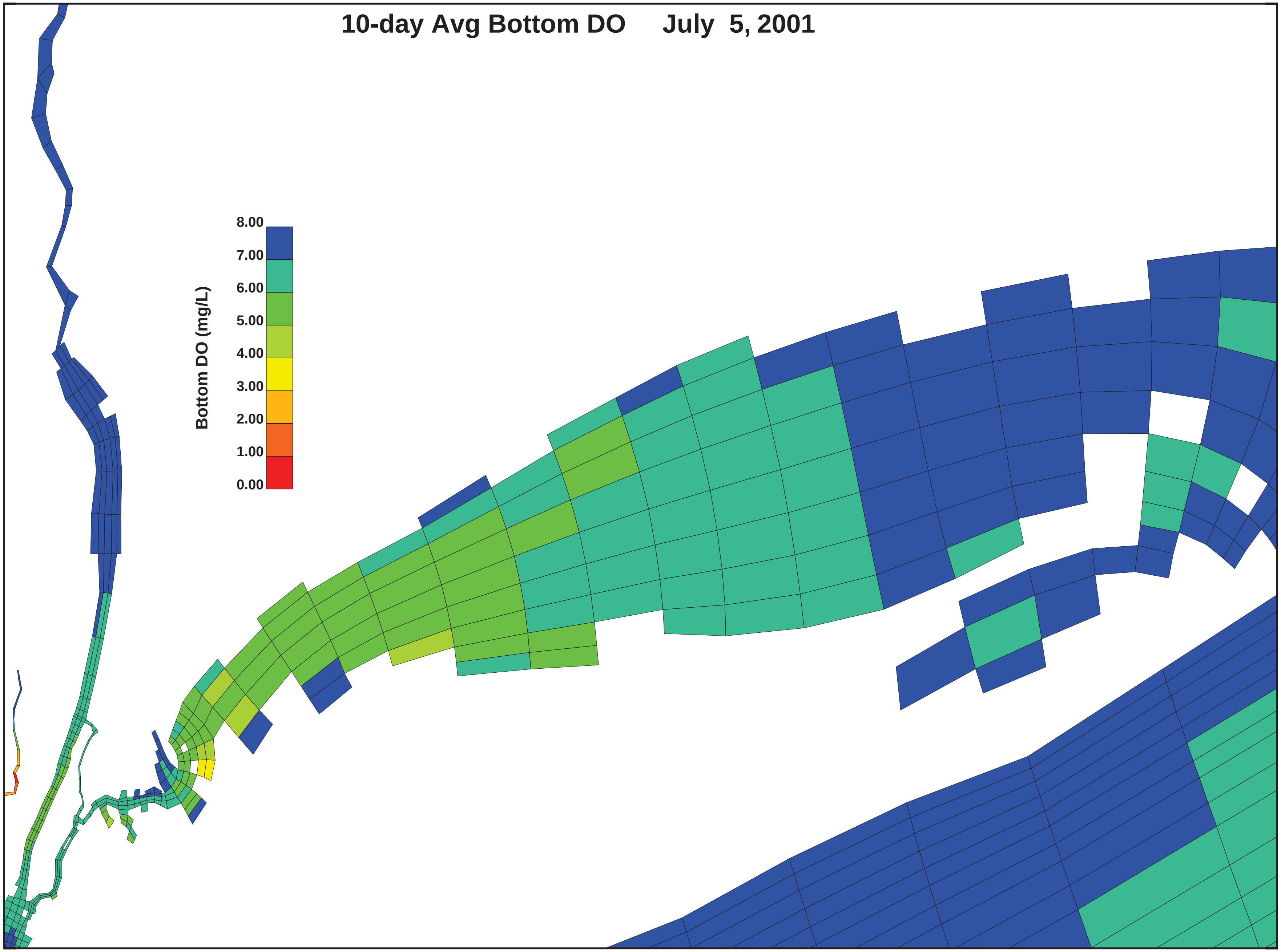

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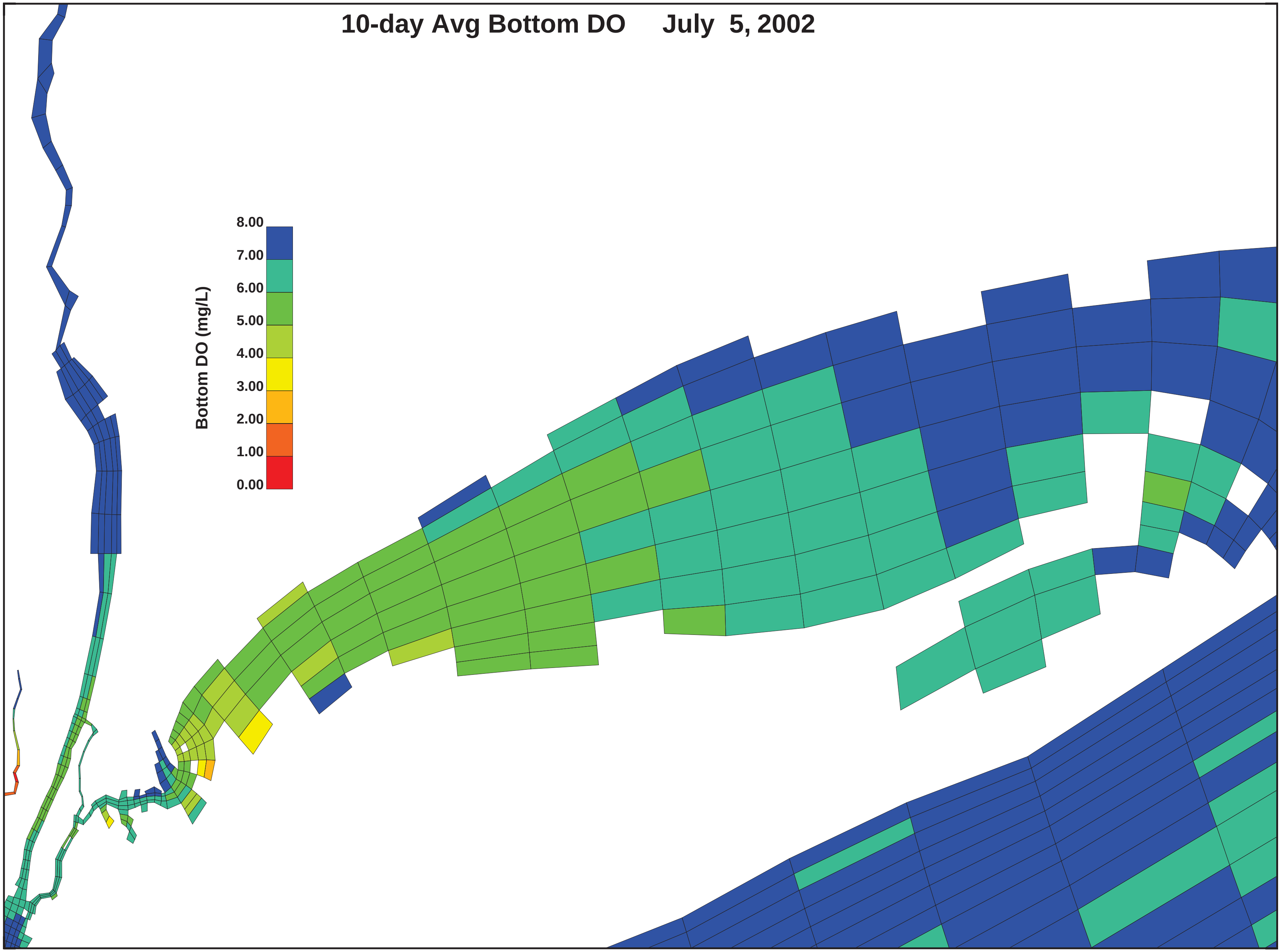

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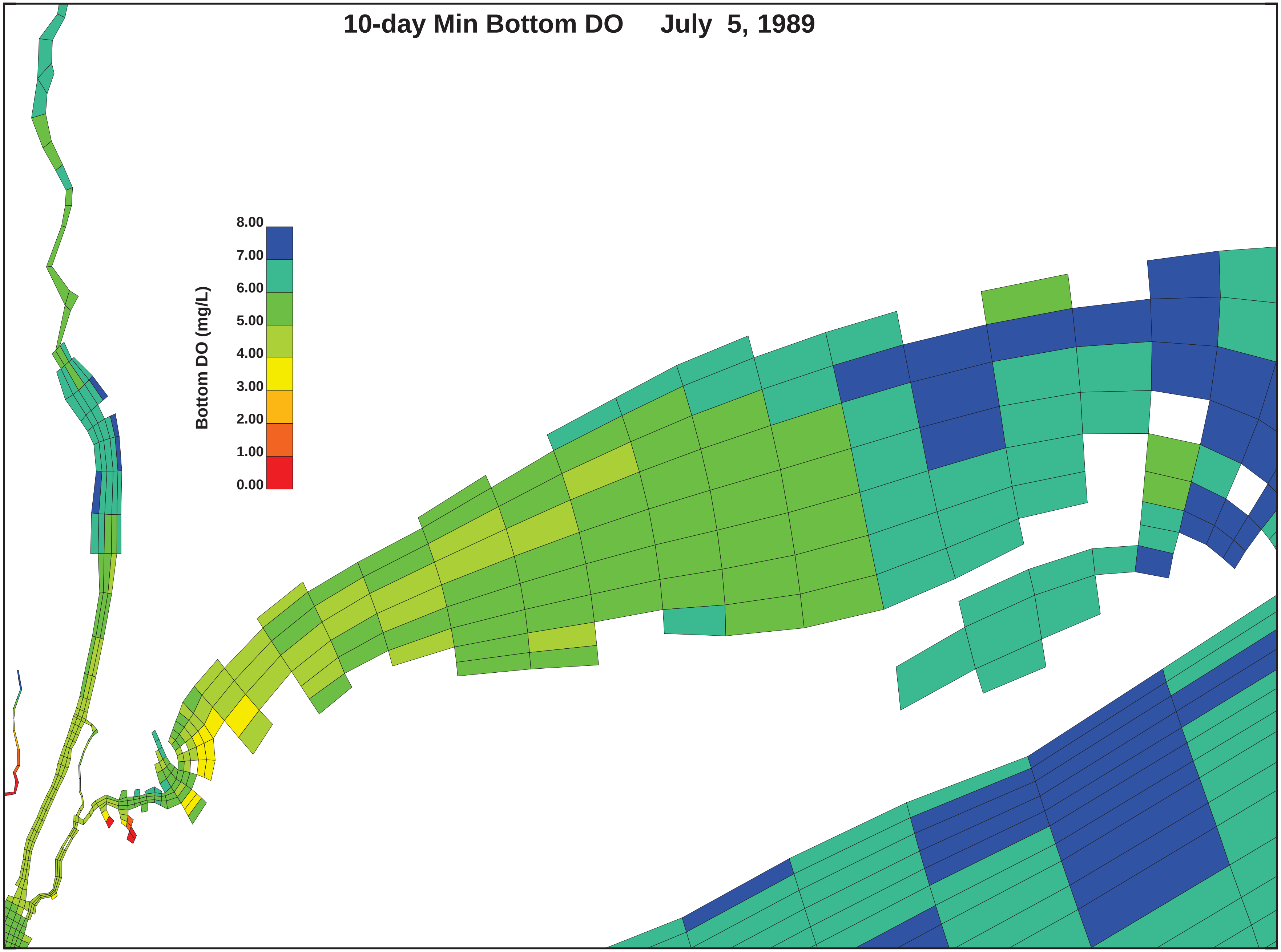
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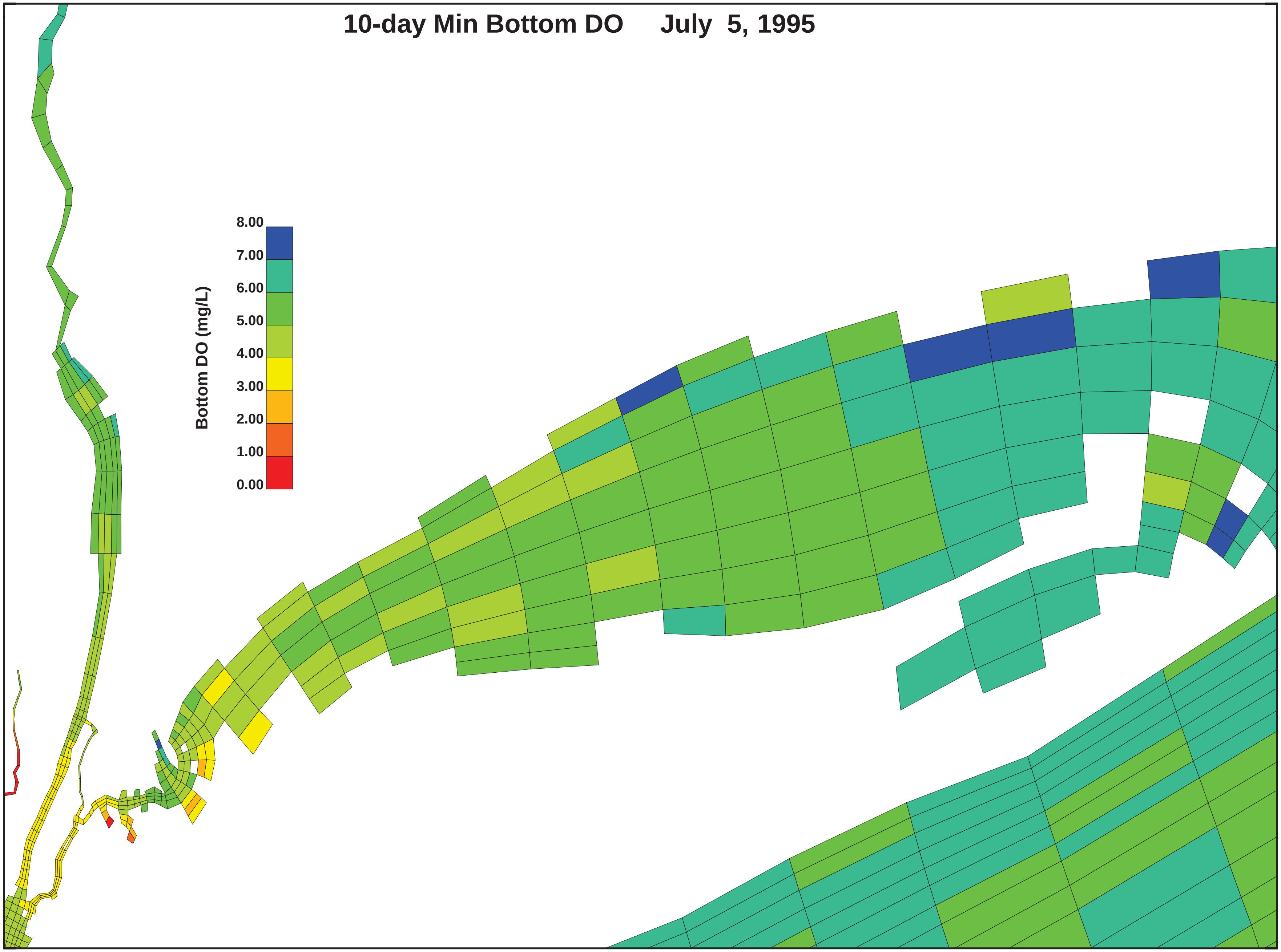
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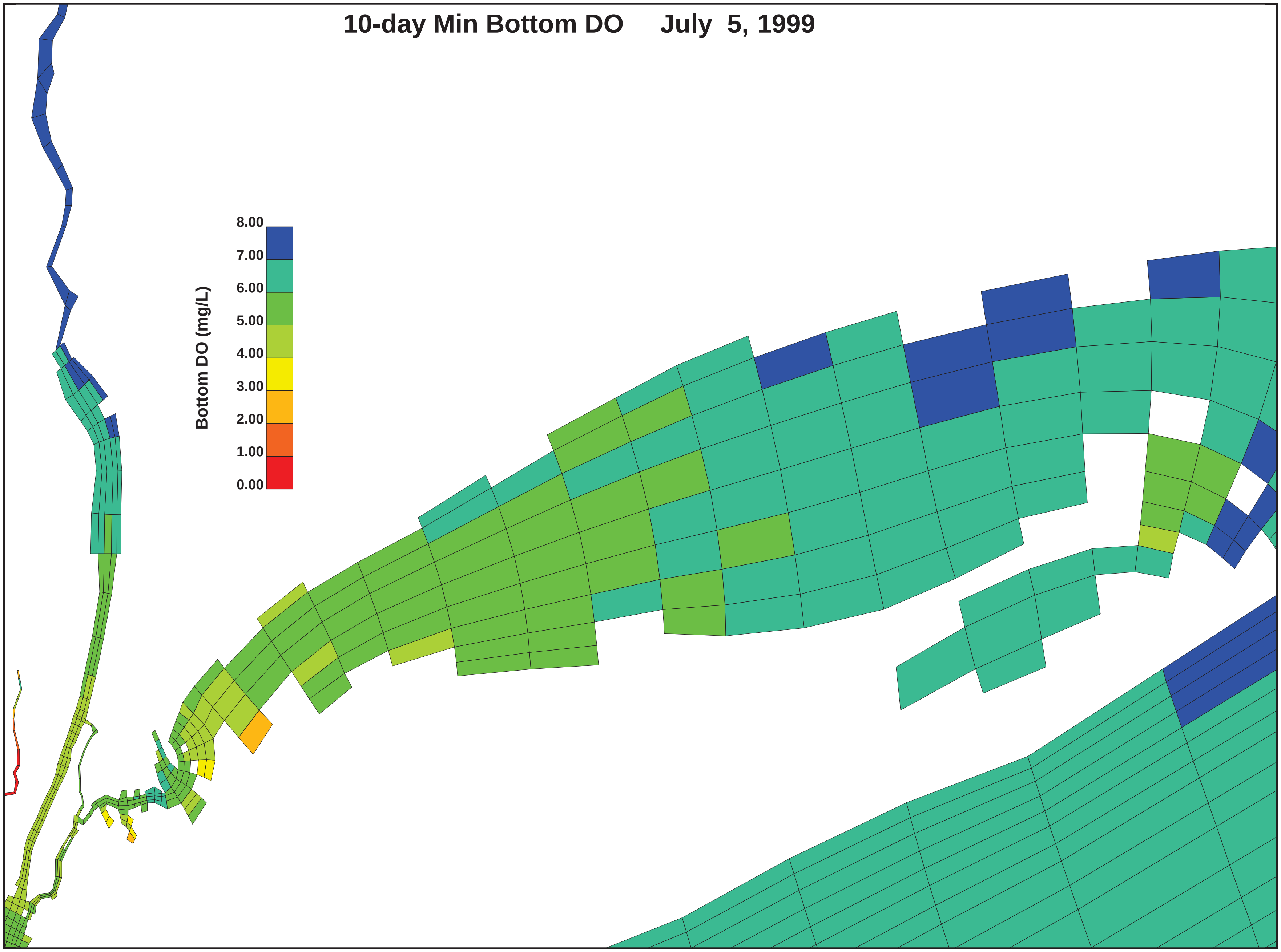

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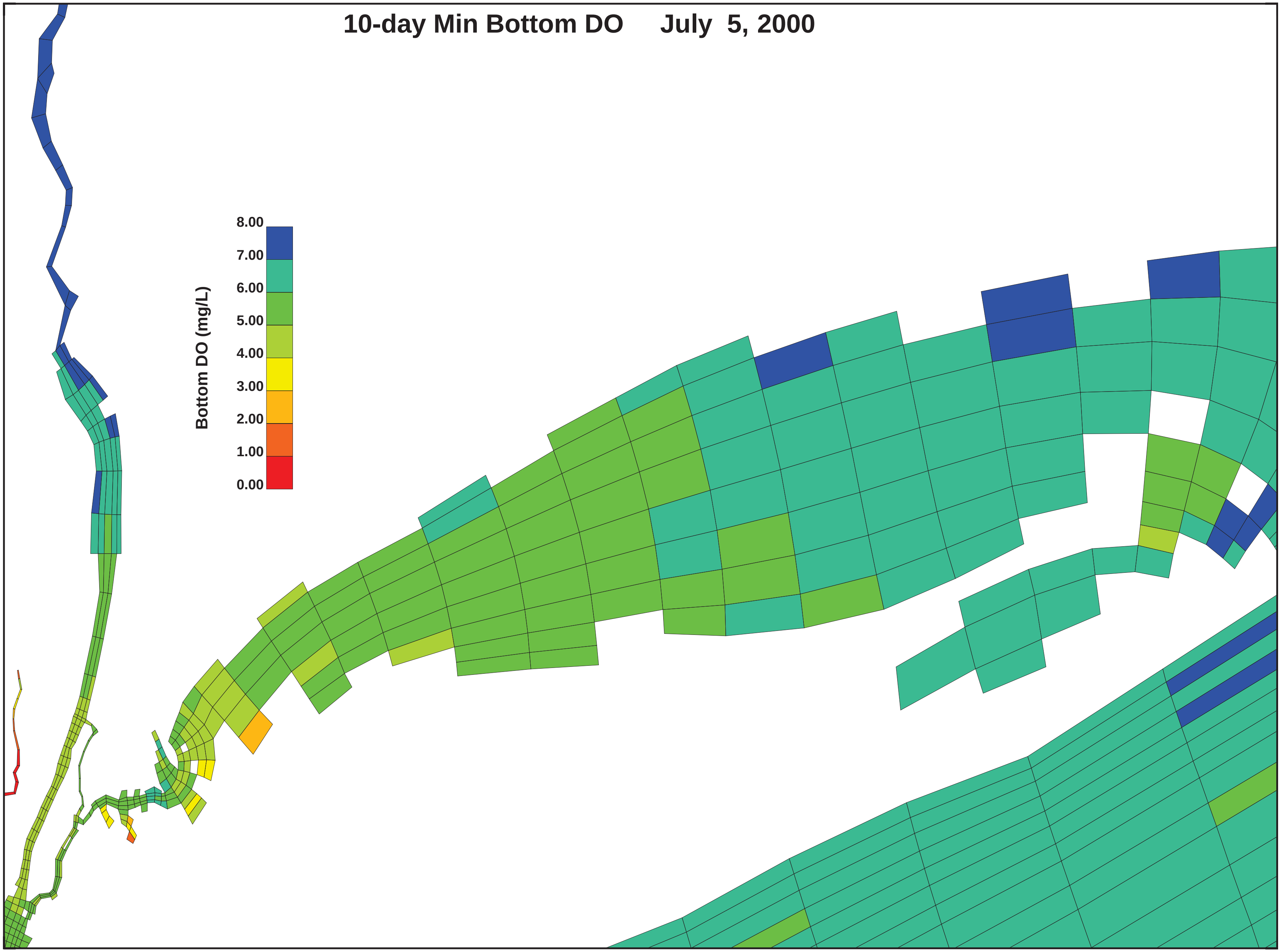

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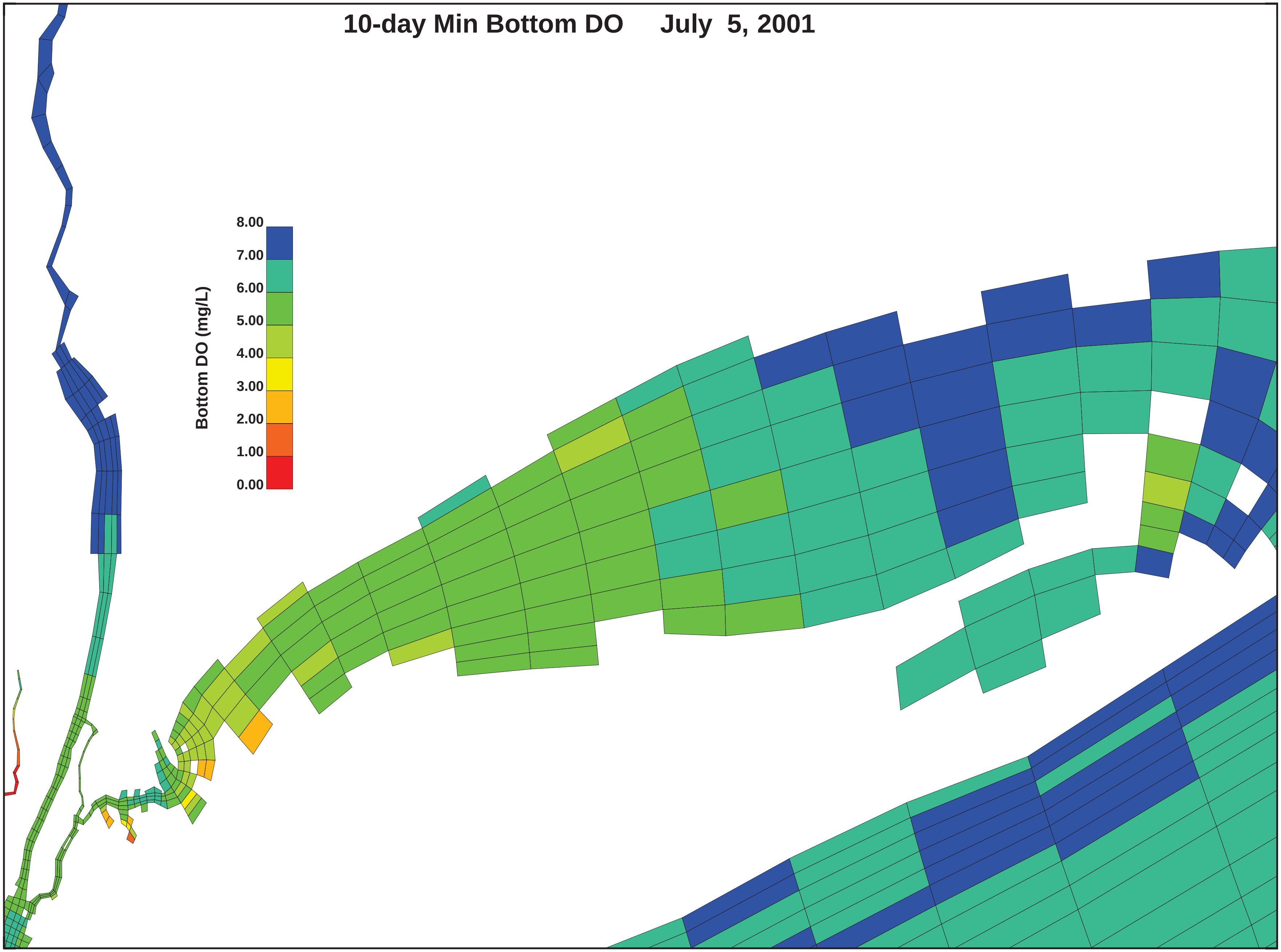
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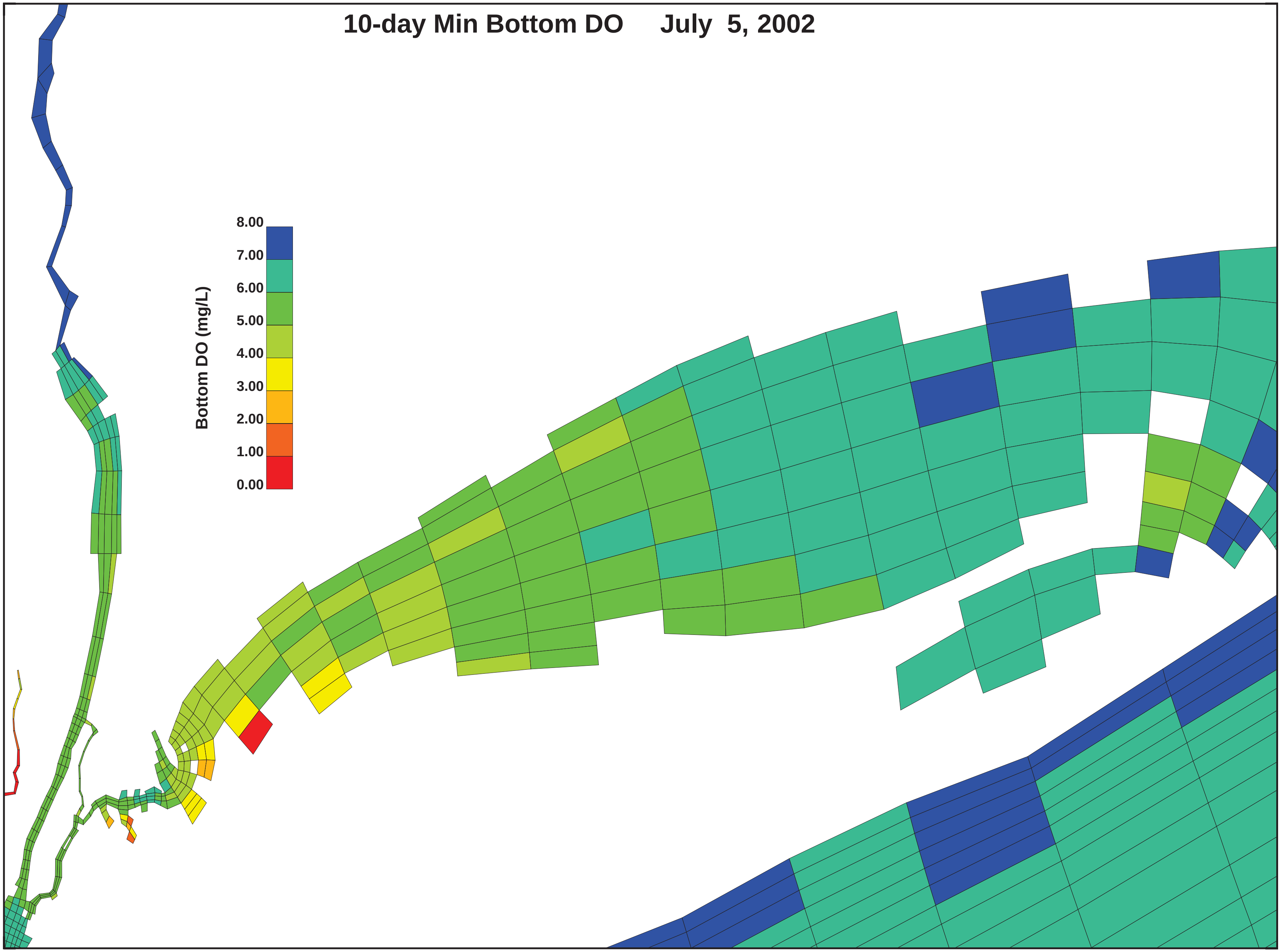
Bottom DO (mg/L)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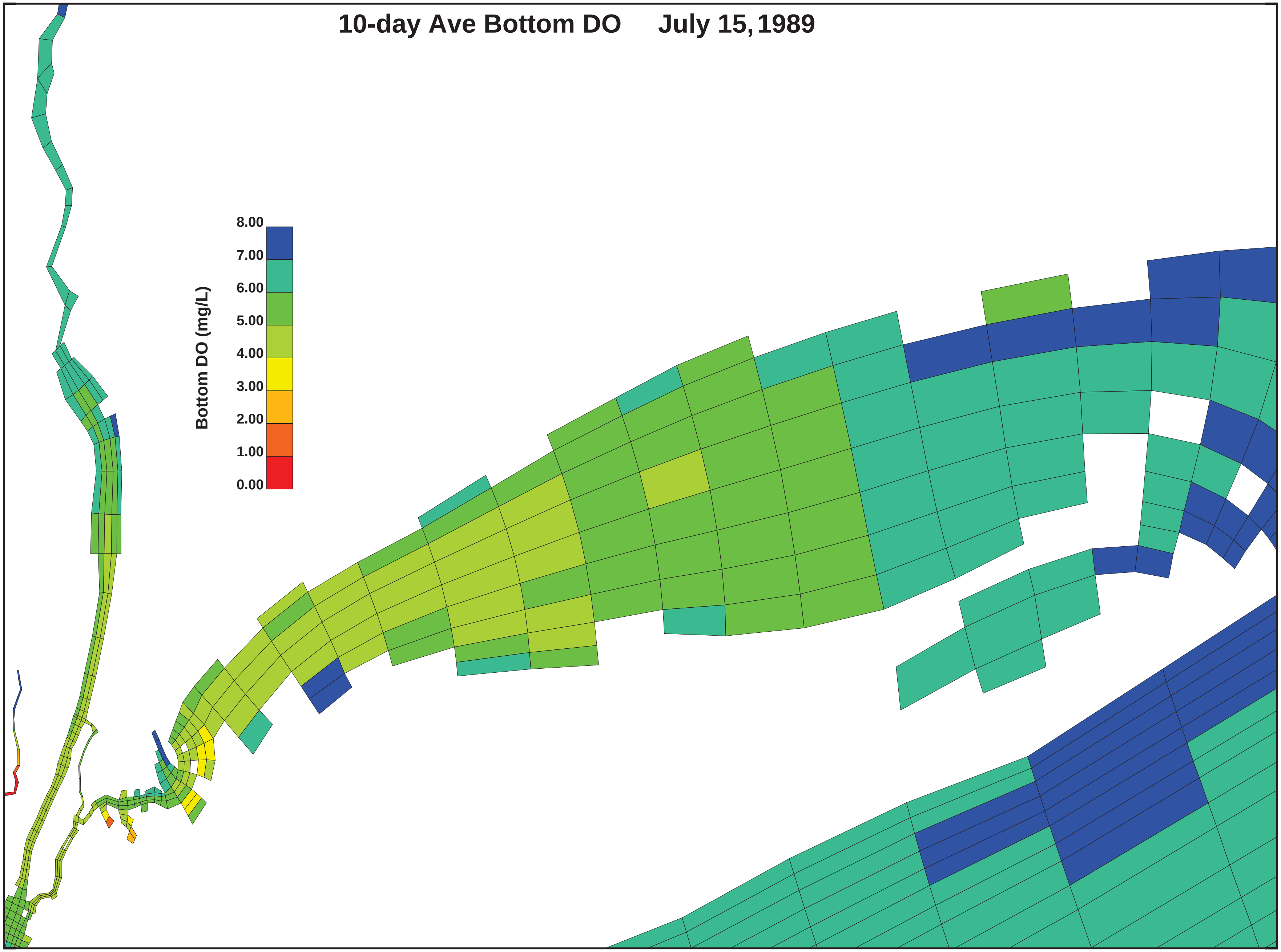
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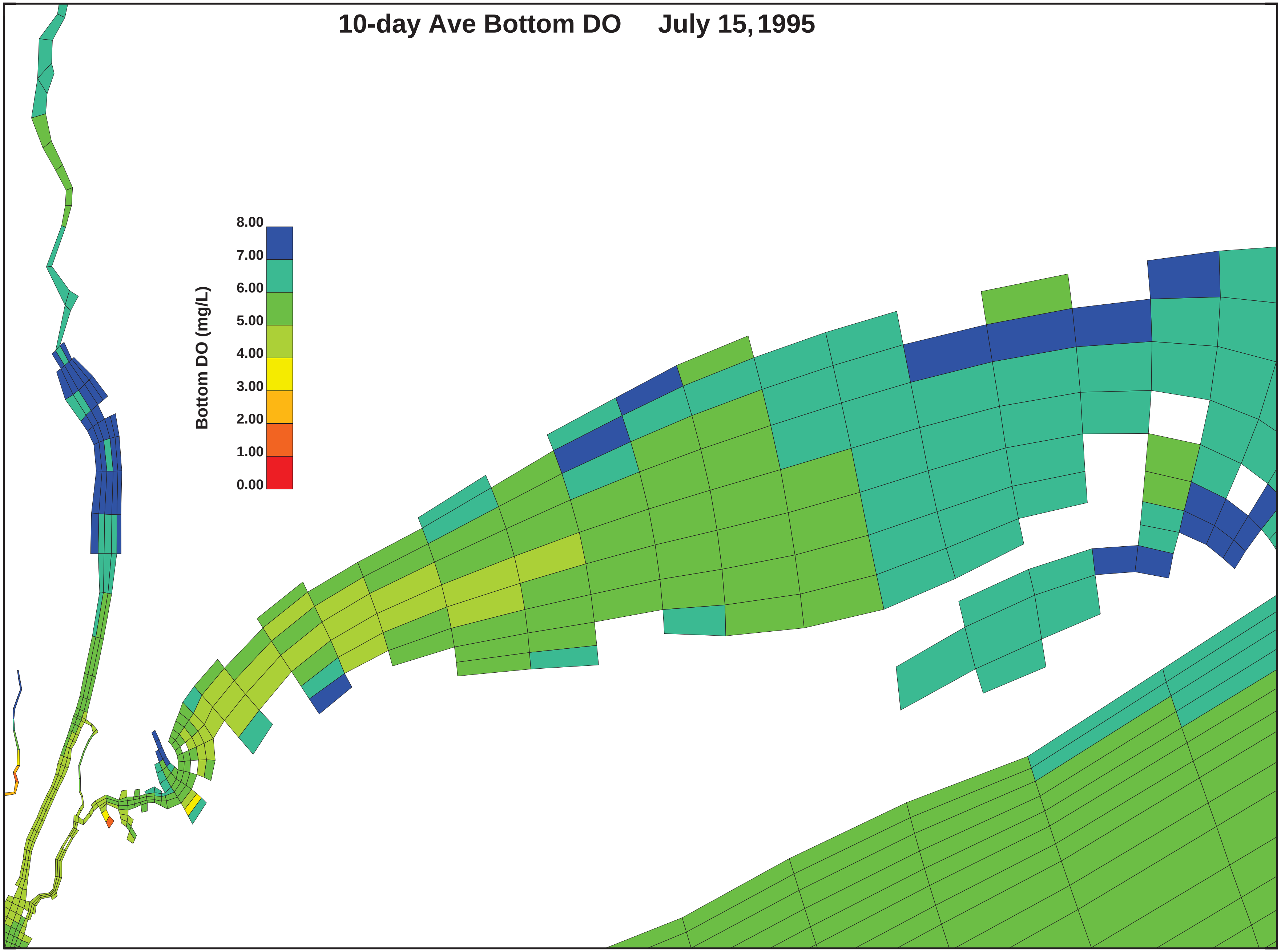
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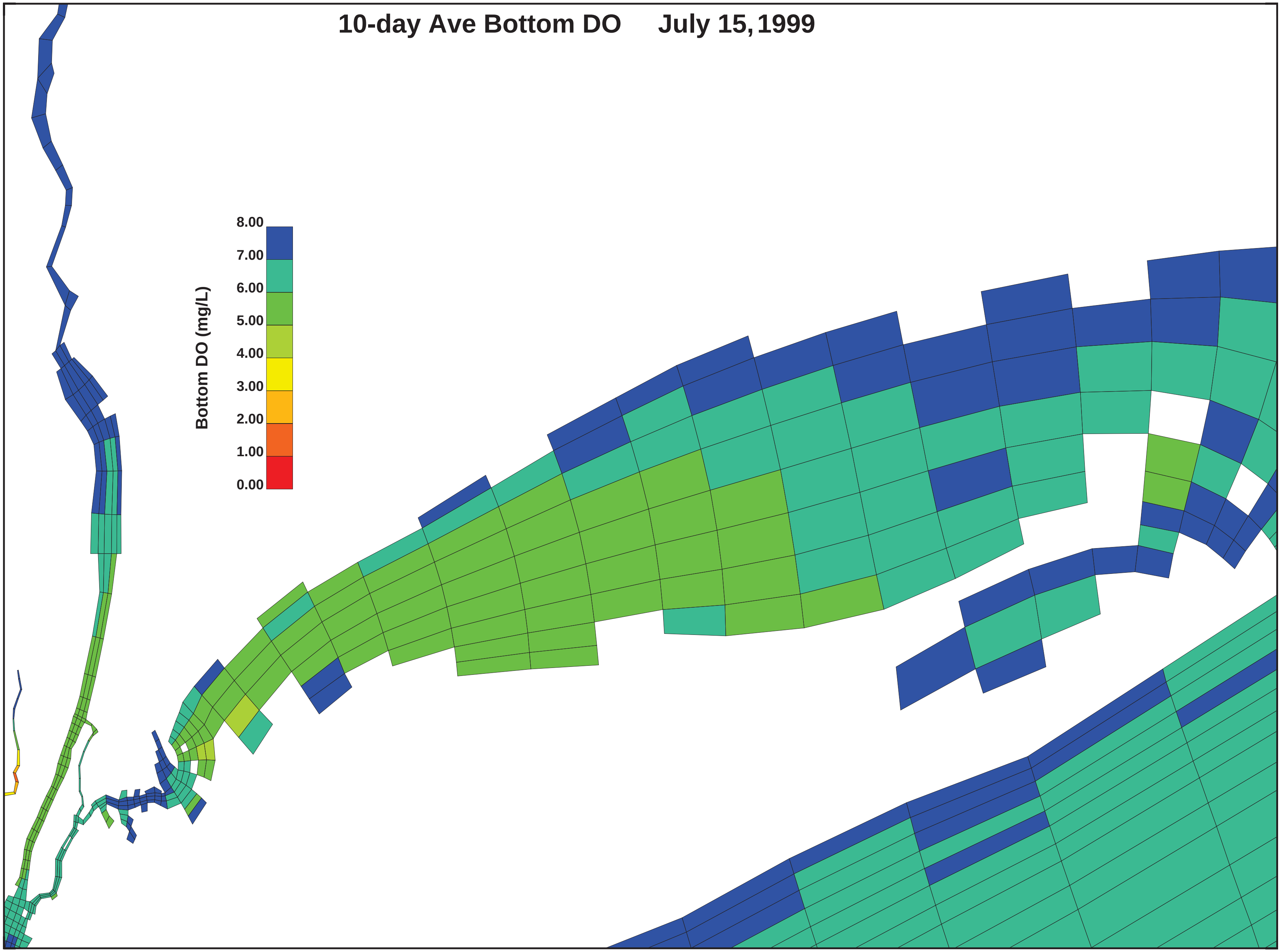

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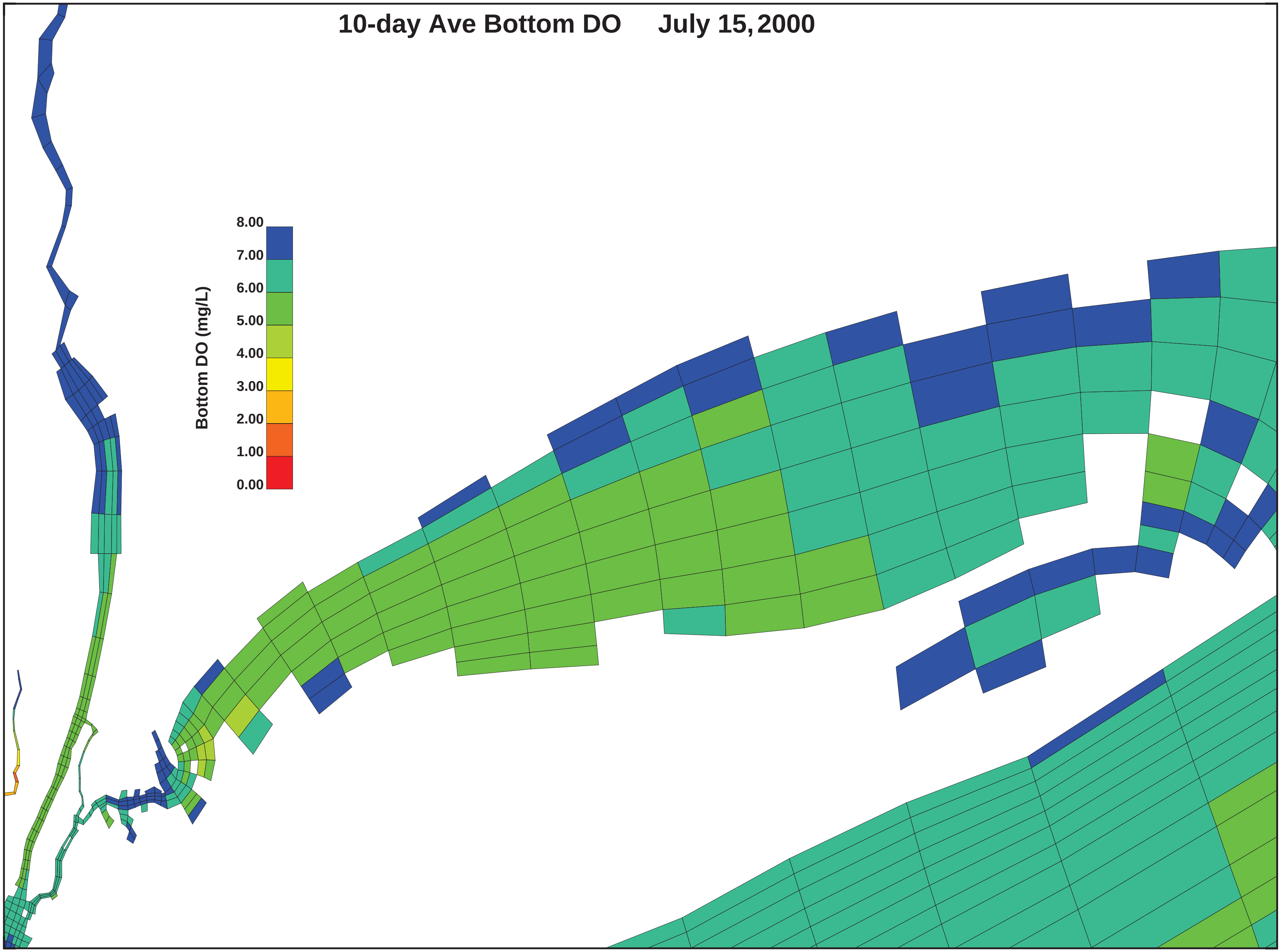
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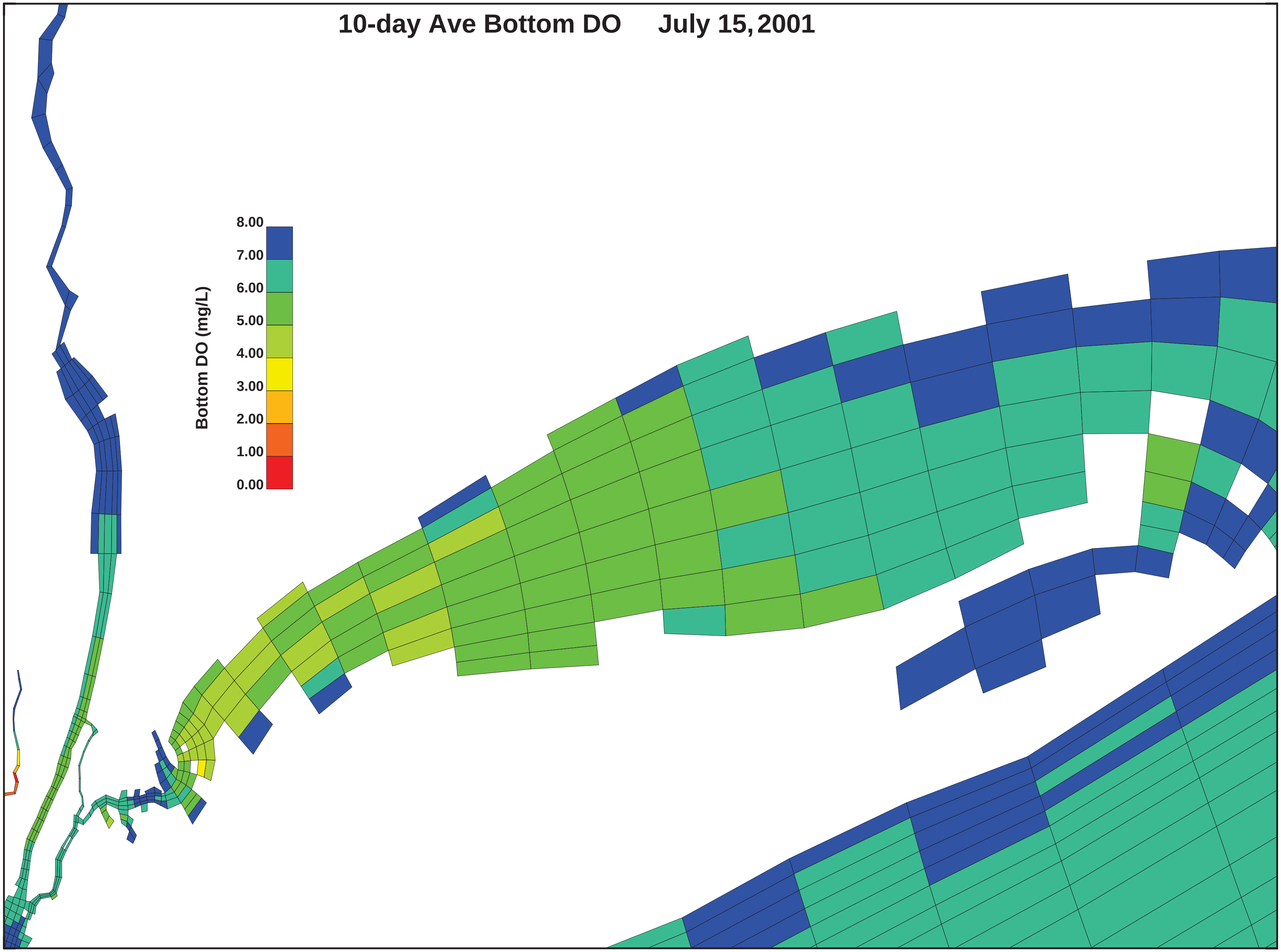
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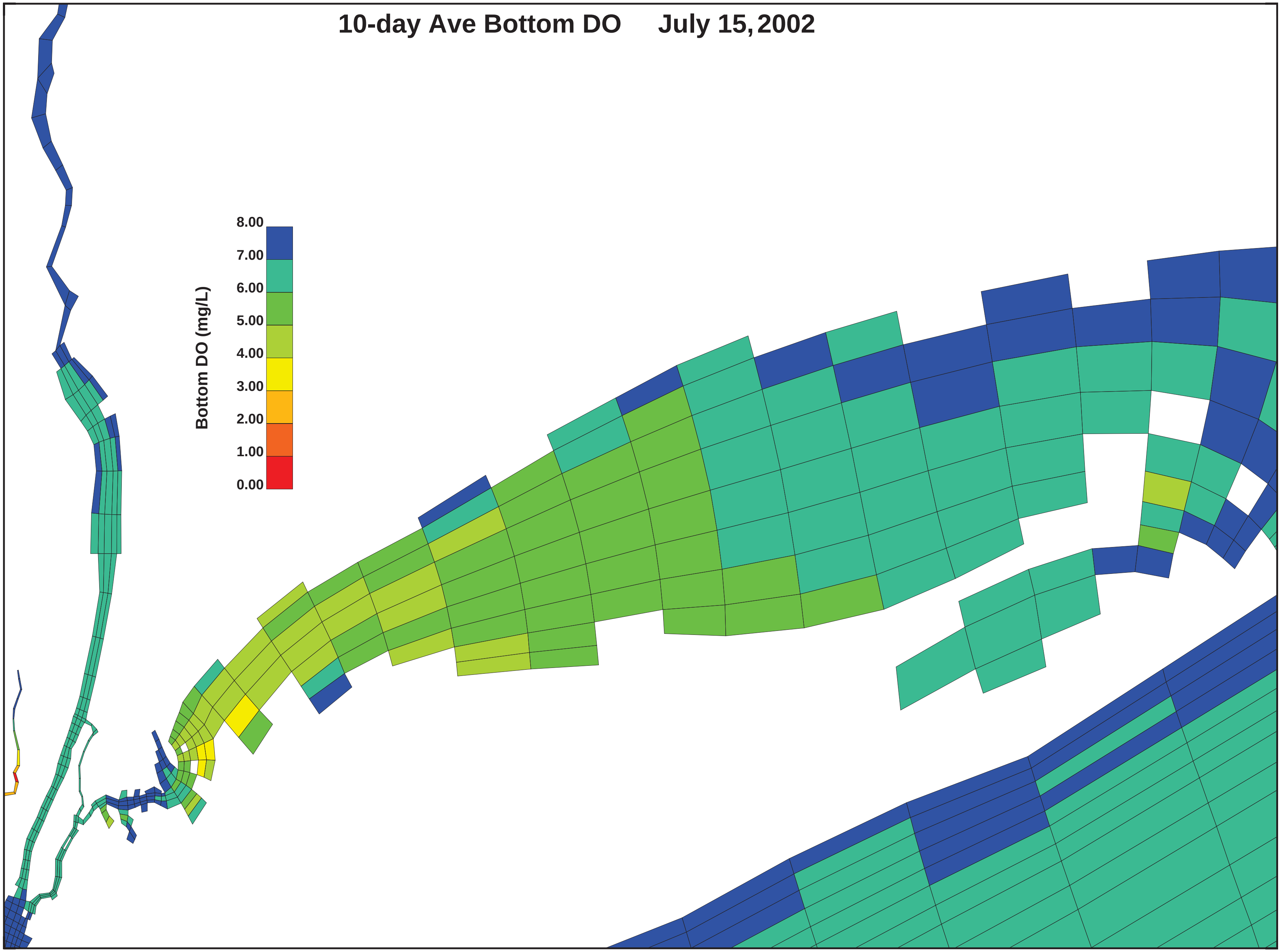
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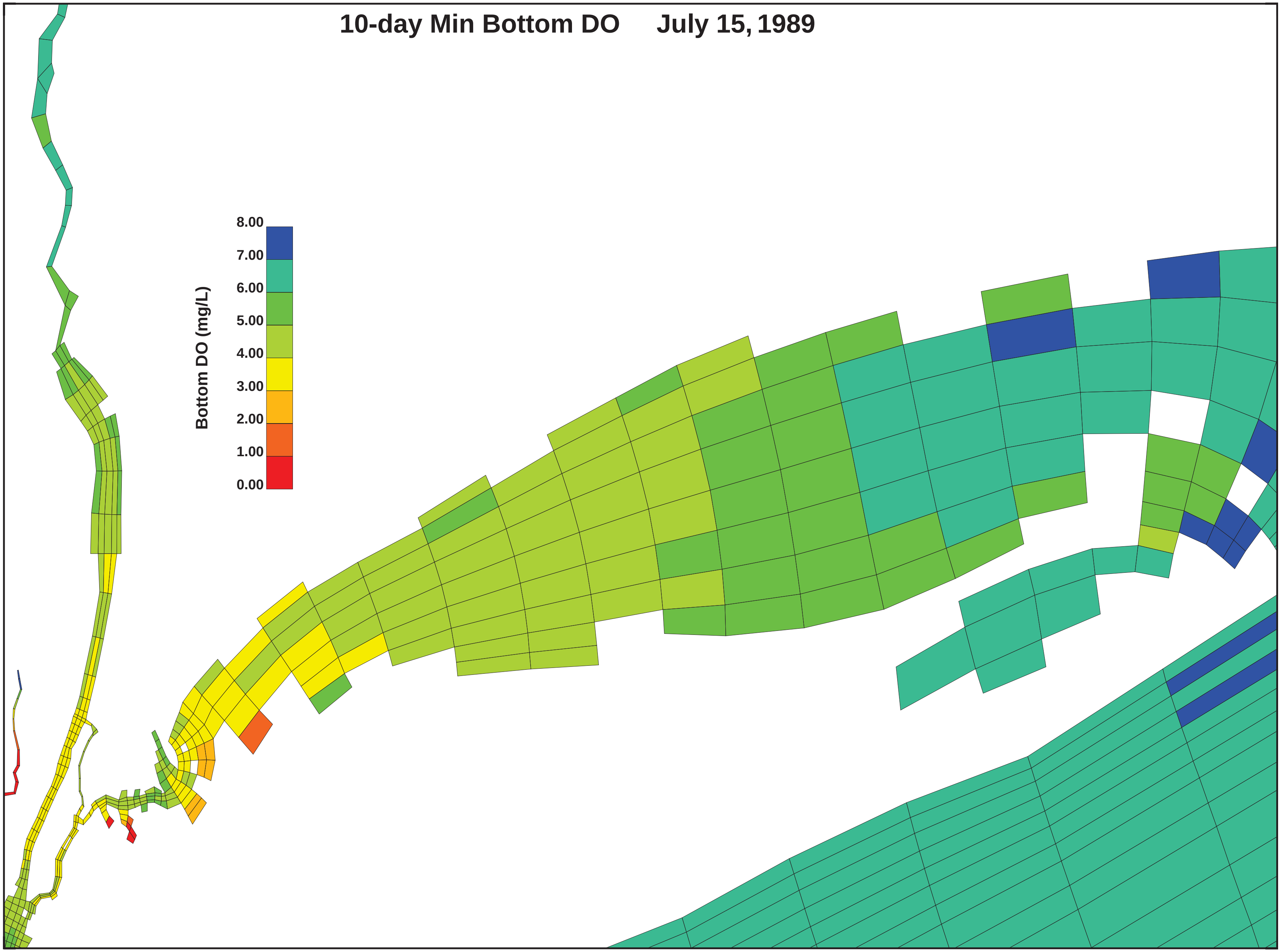
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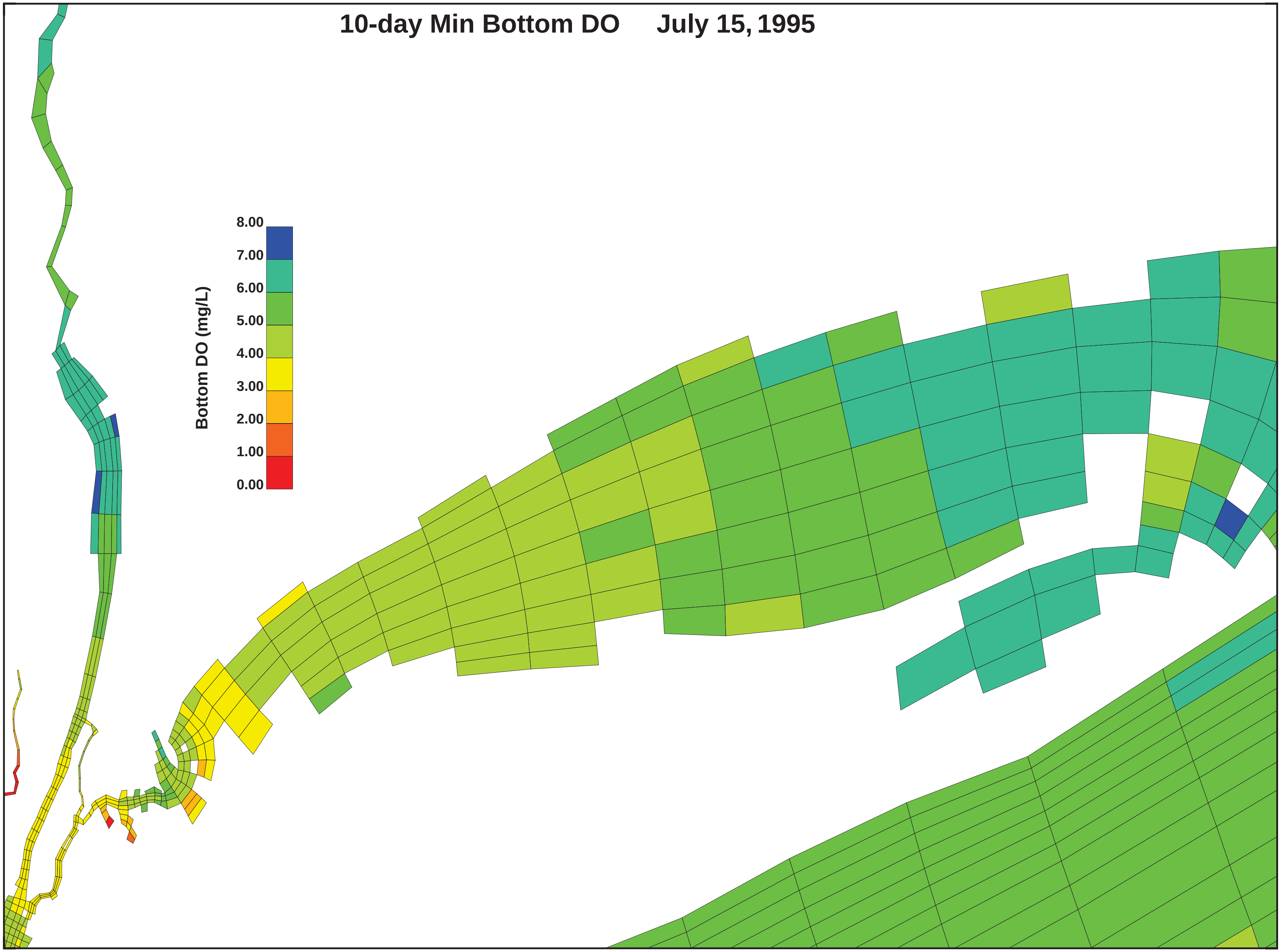
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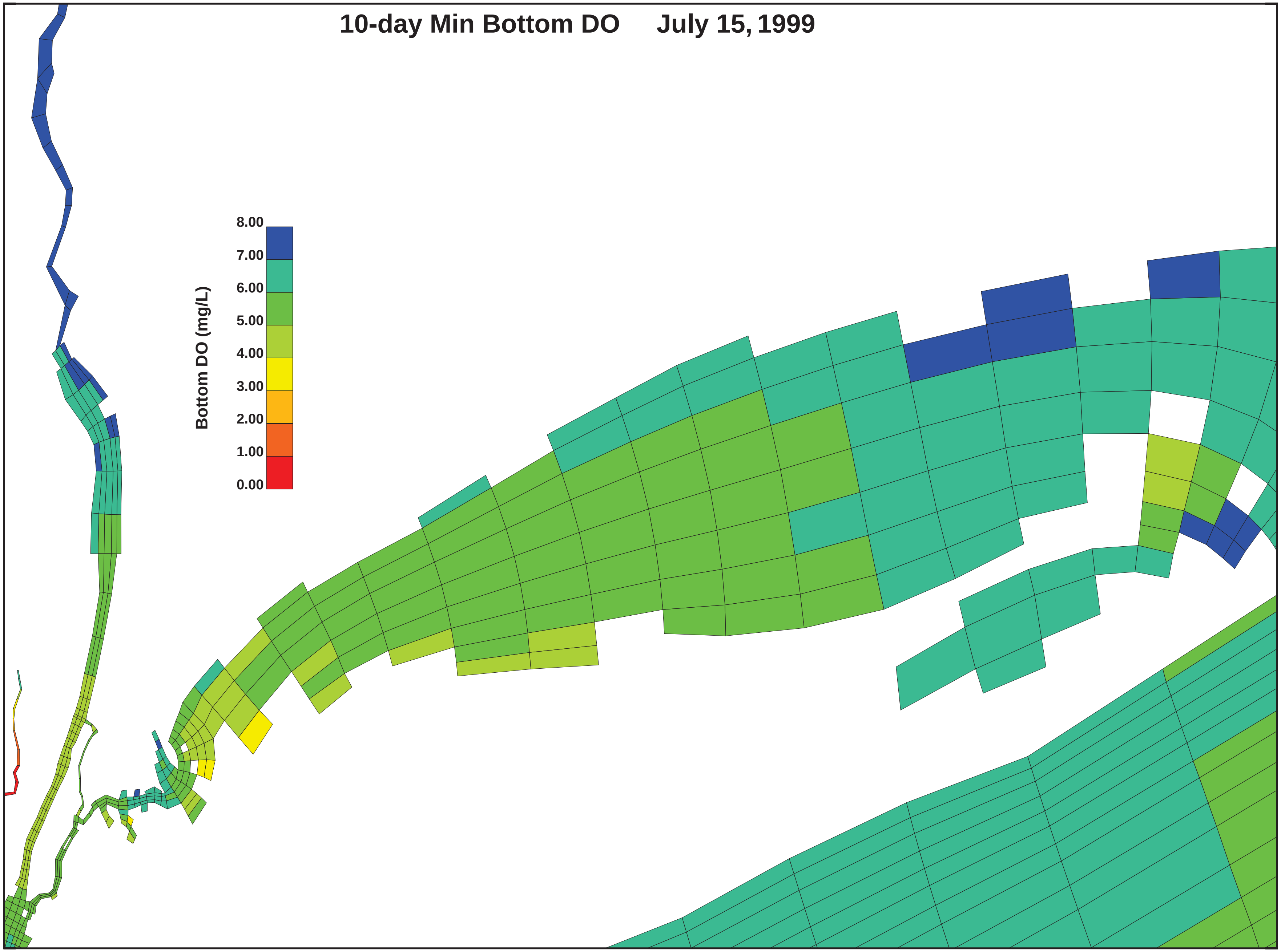
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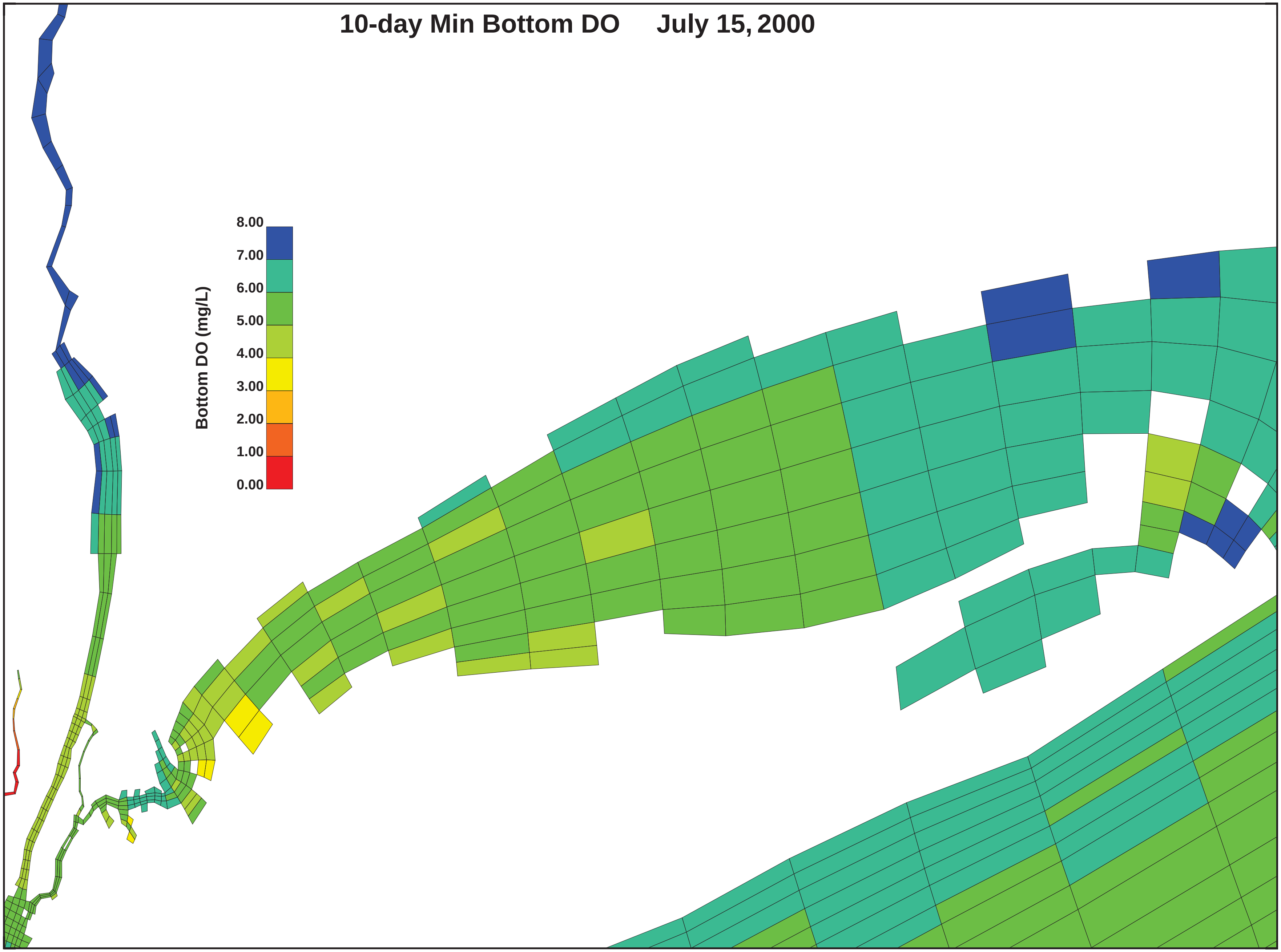
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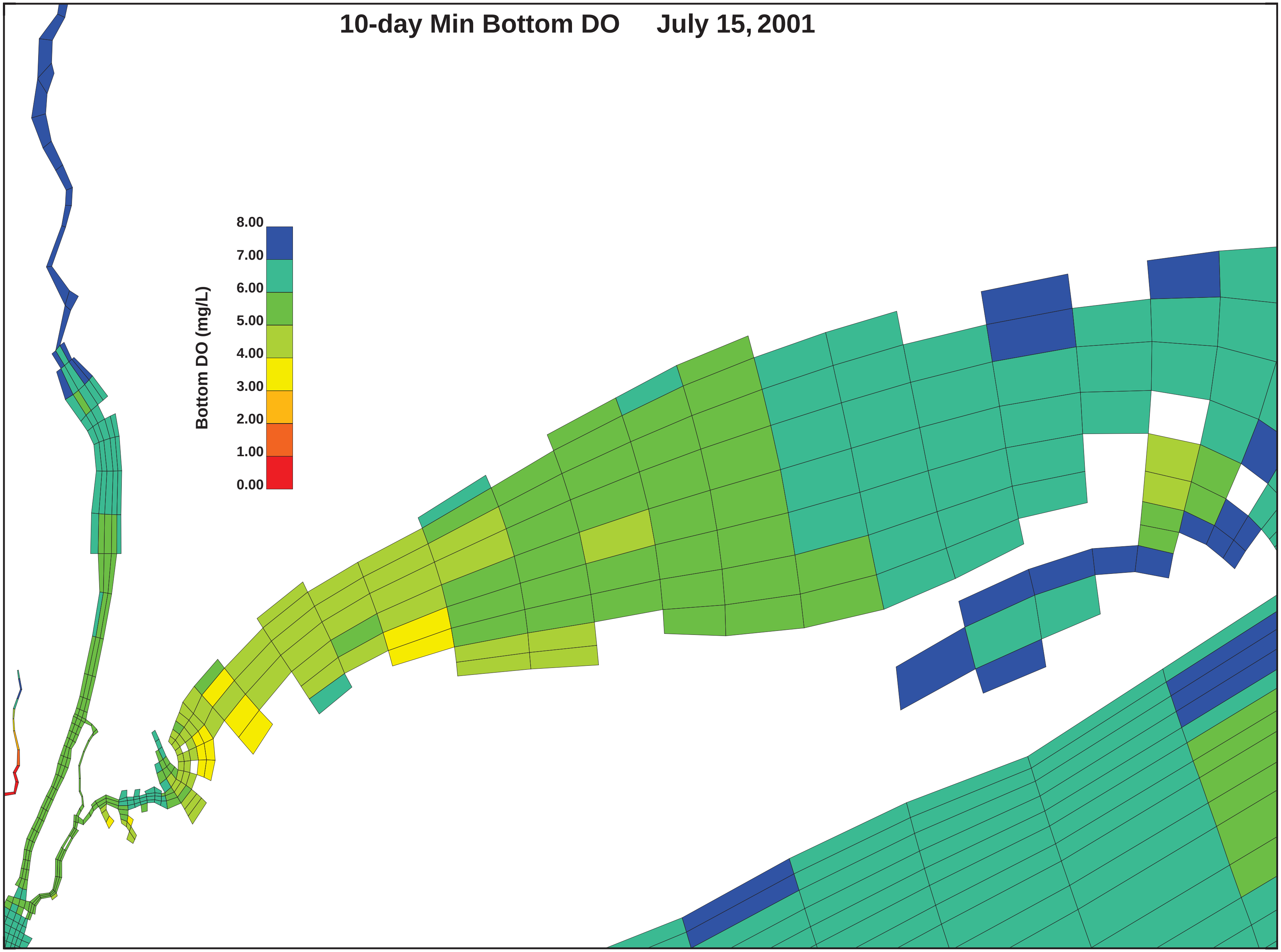

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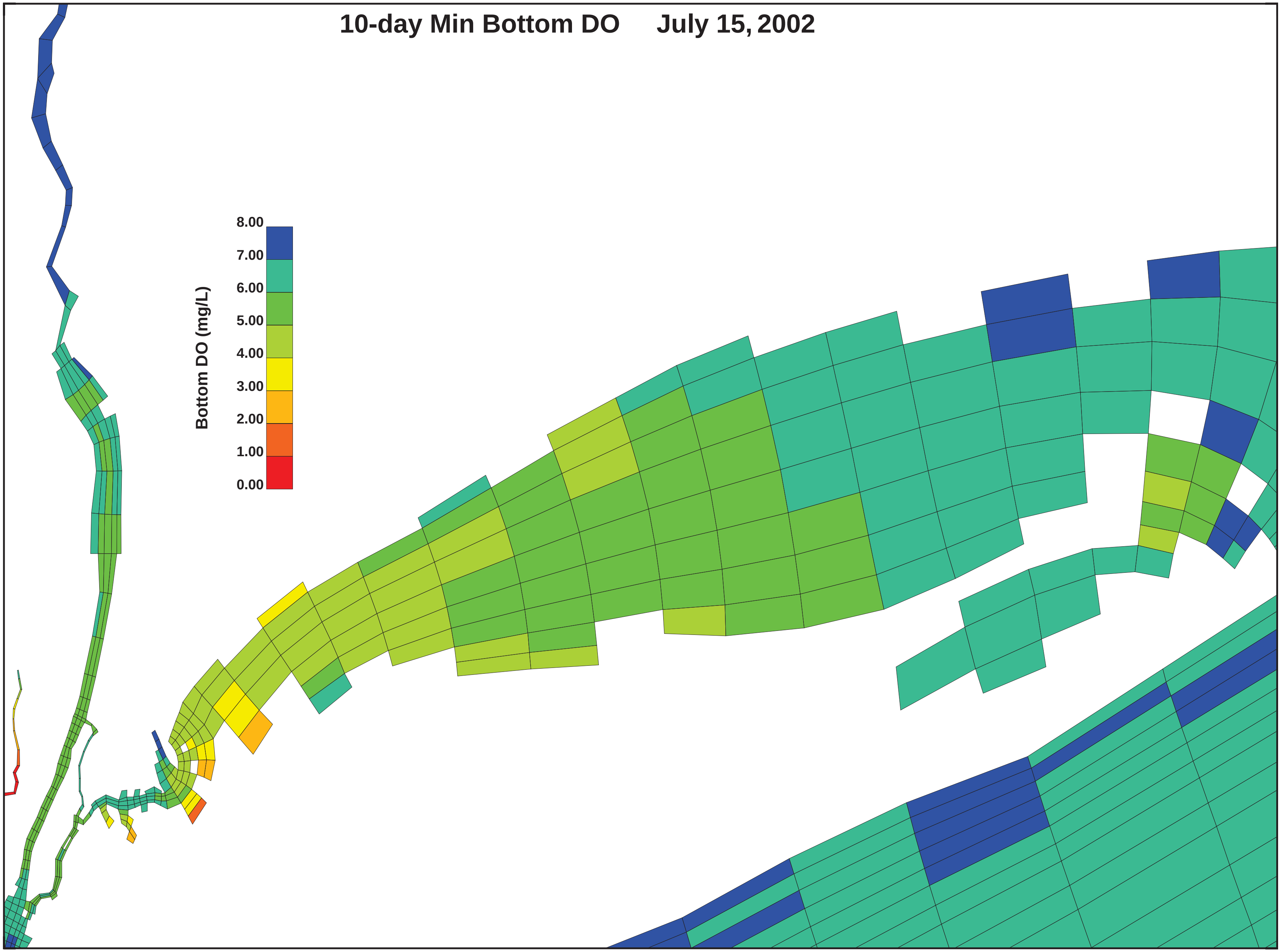
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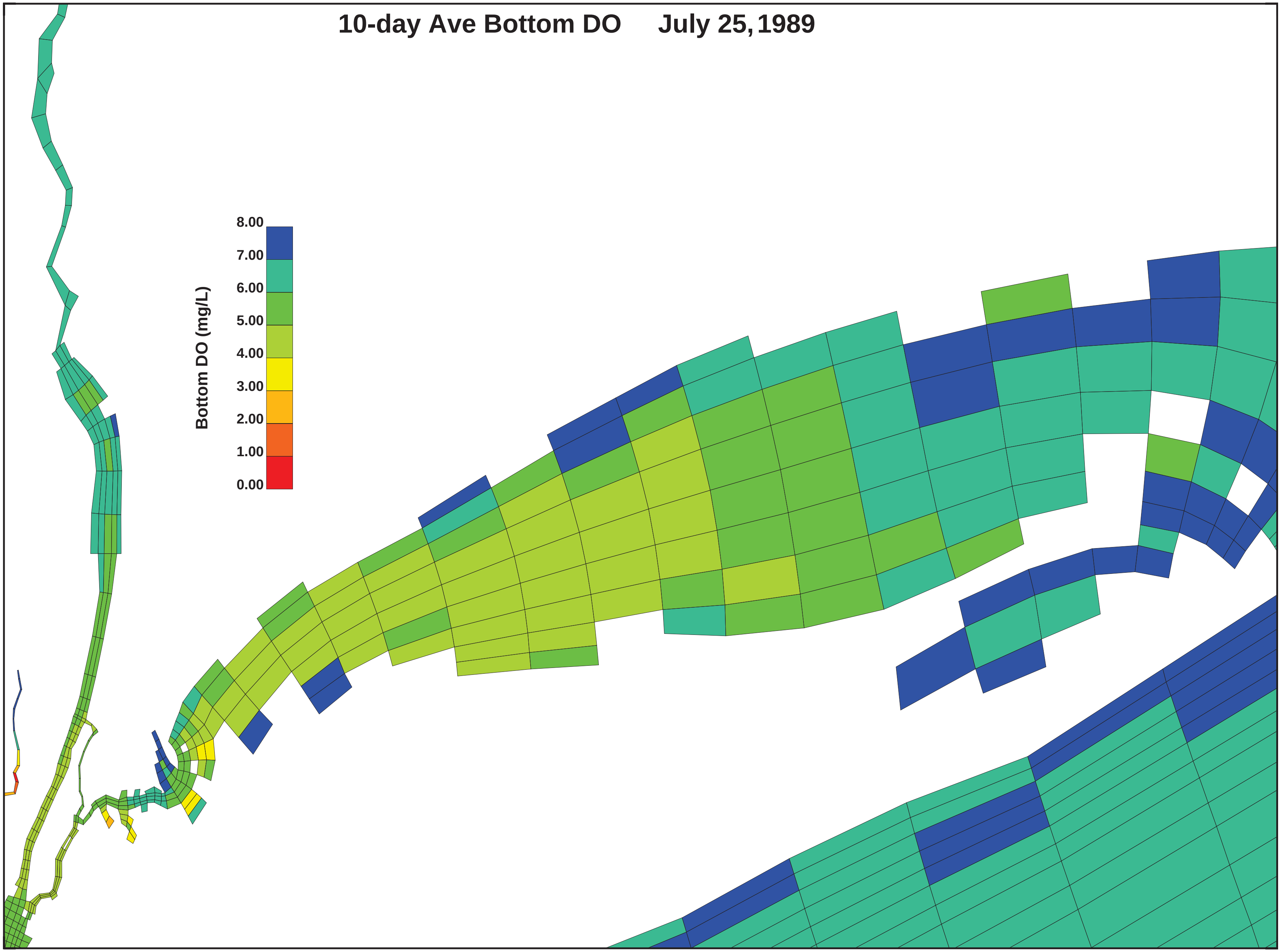

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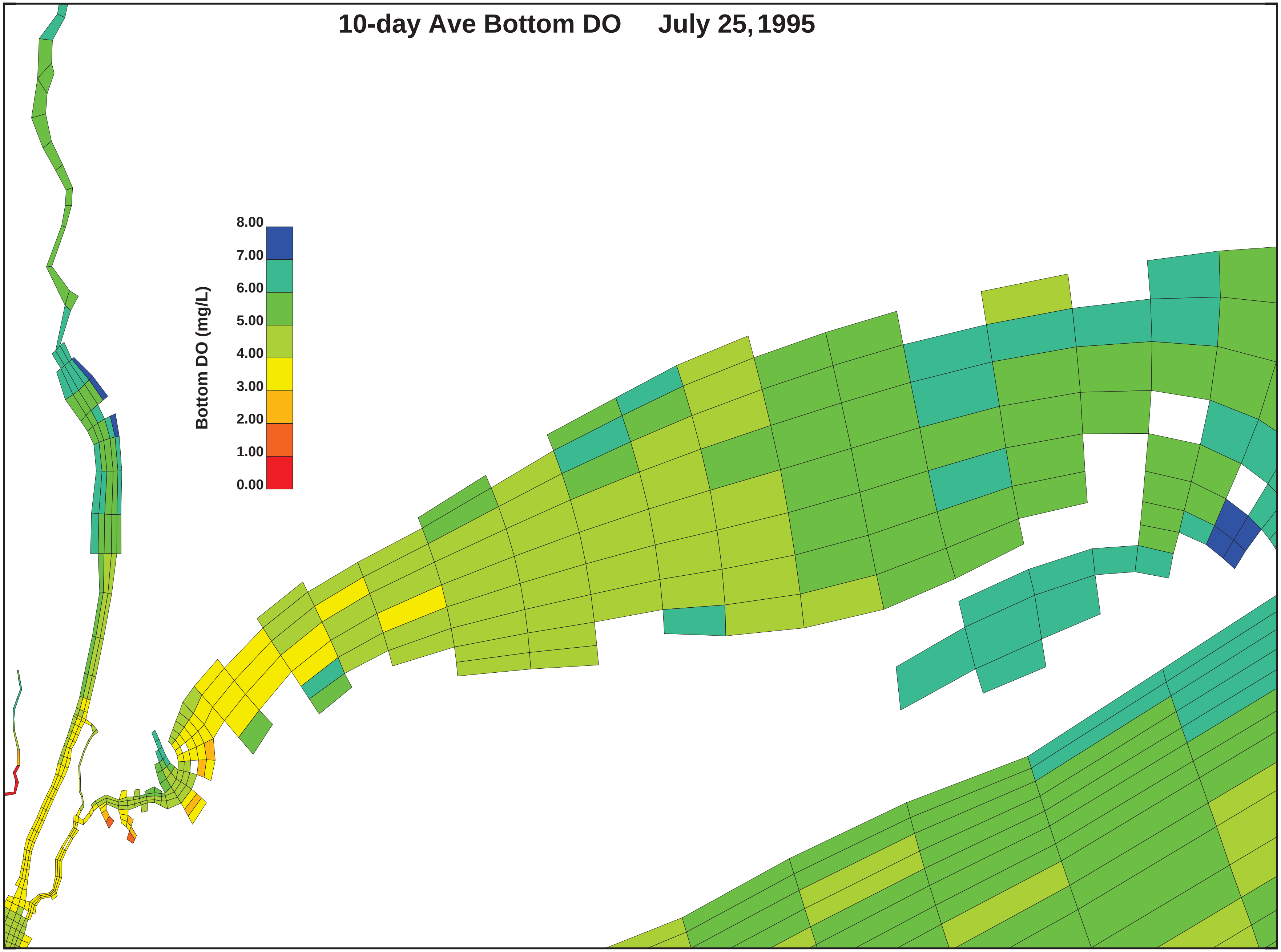
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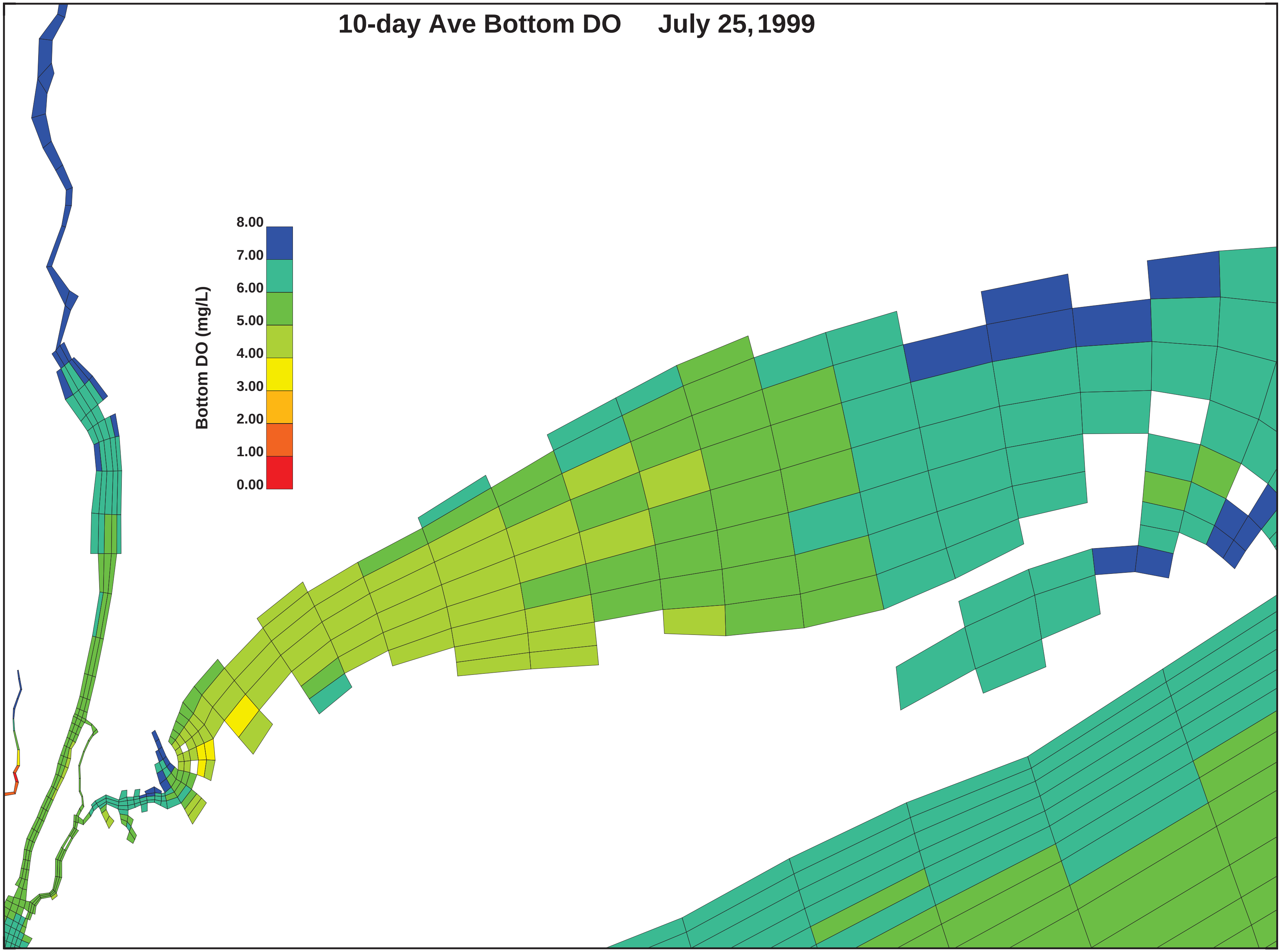
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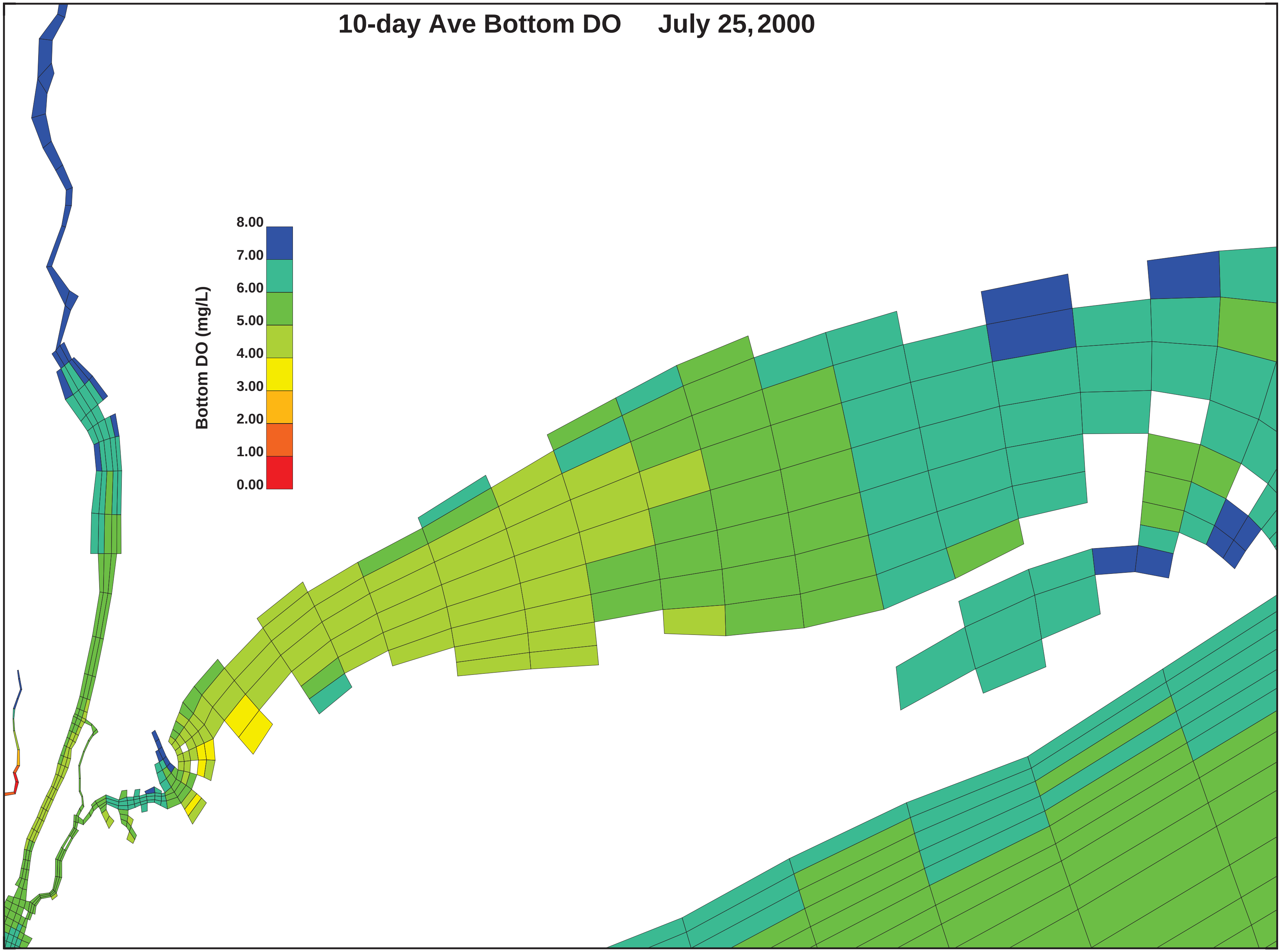
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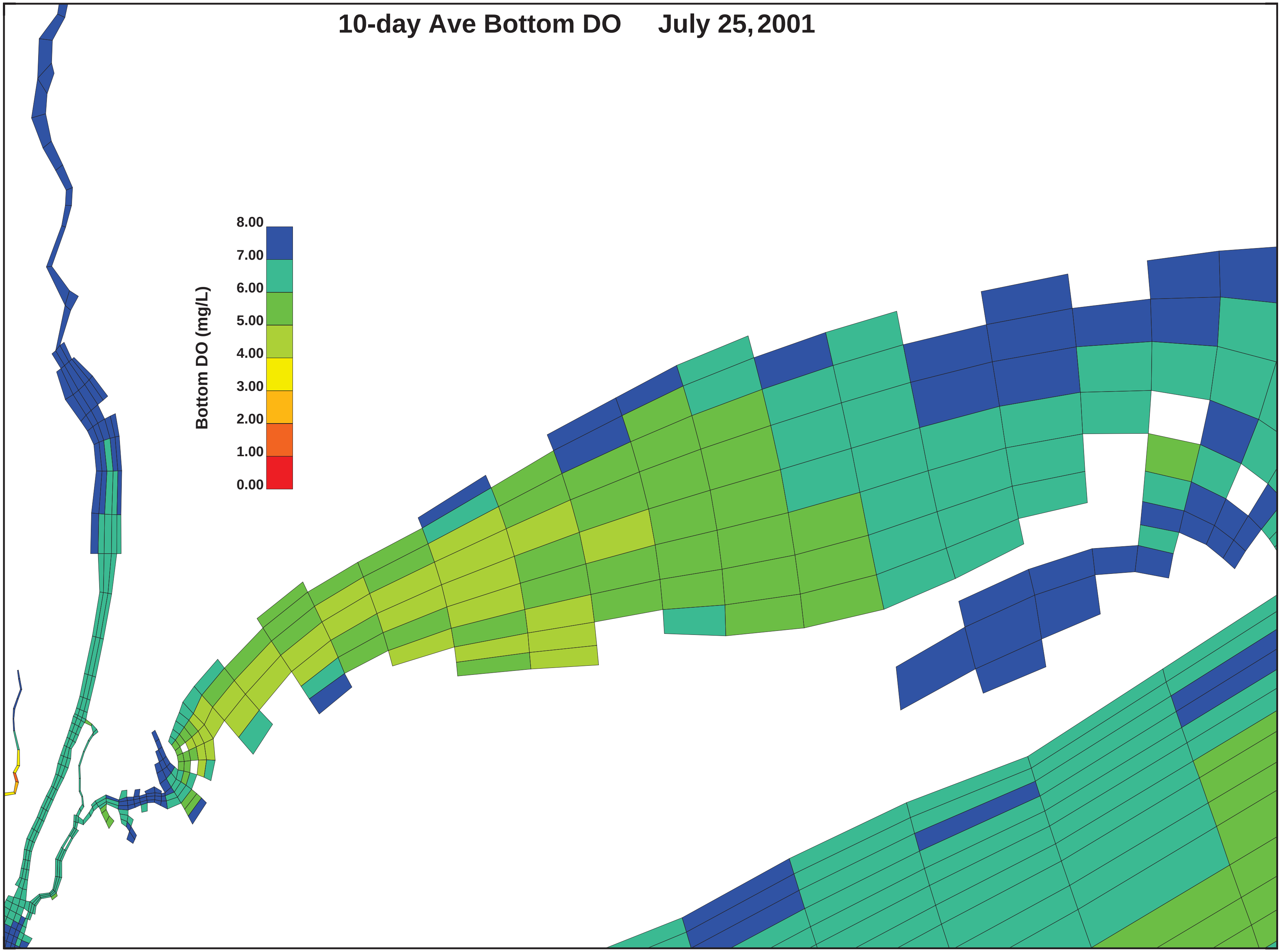

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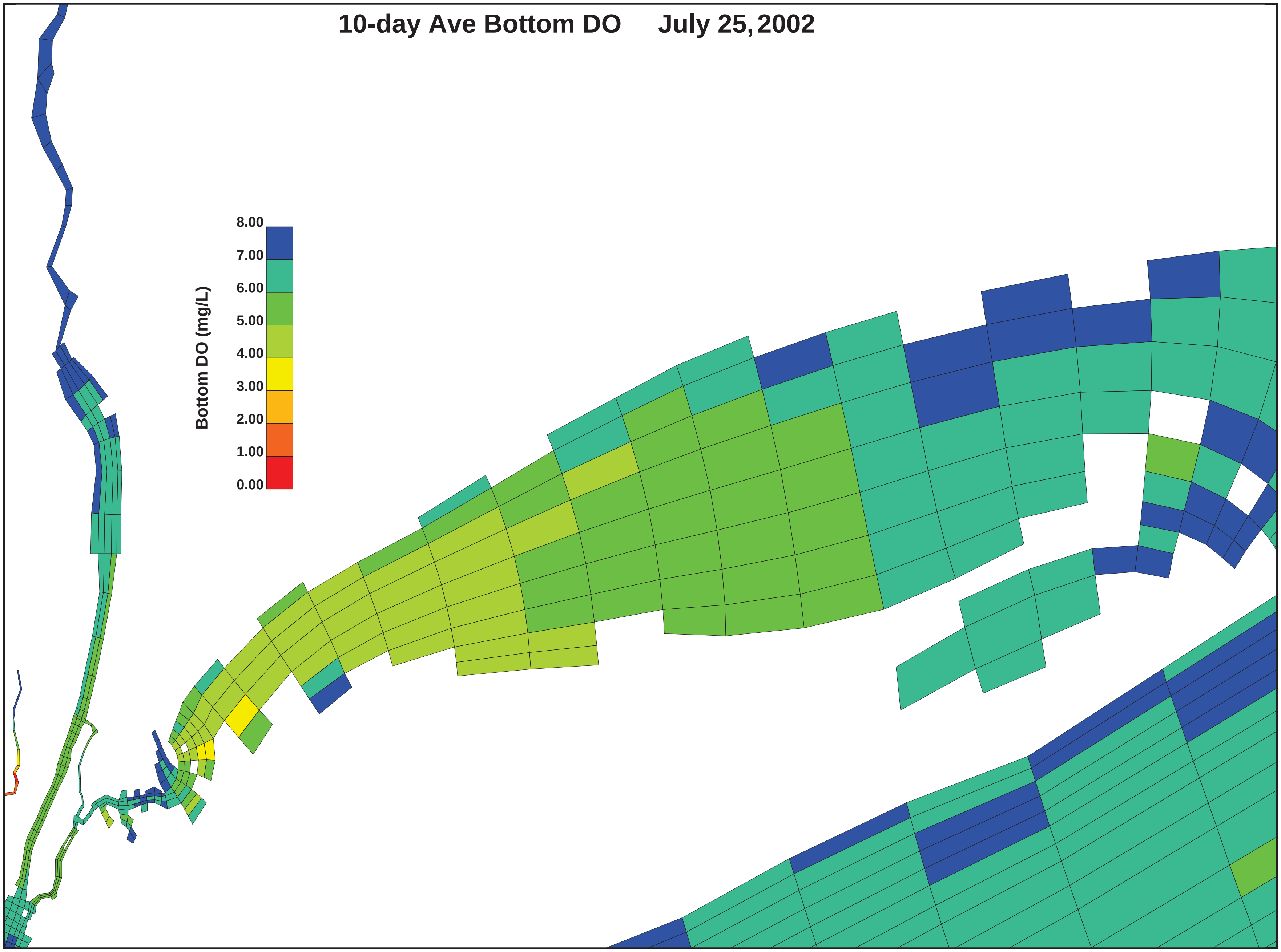
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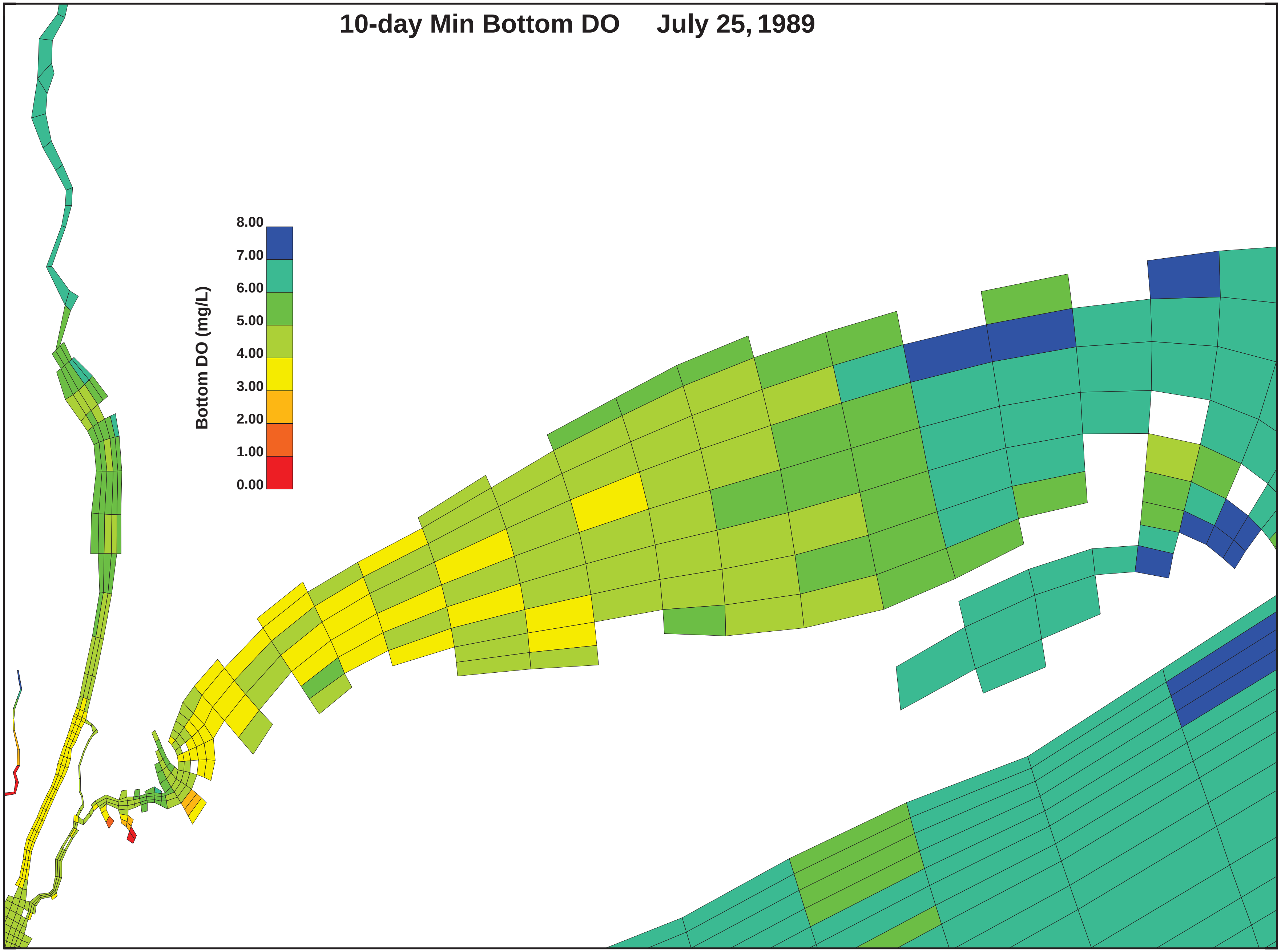
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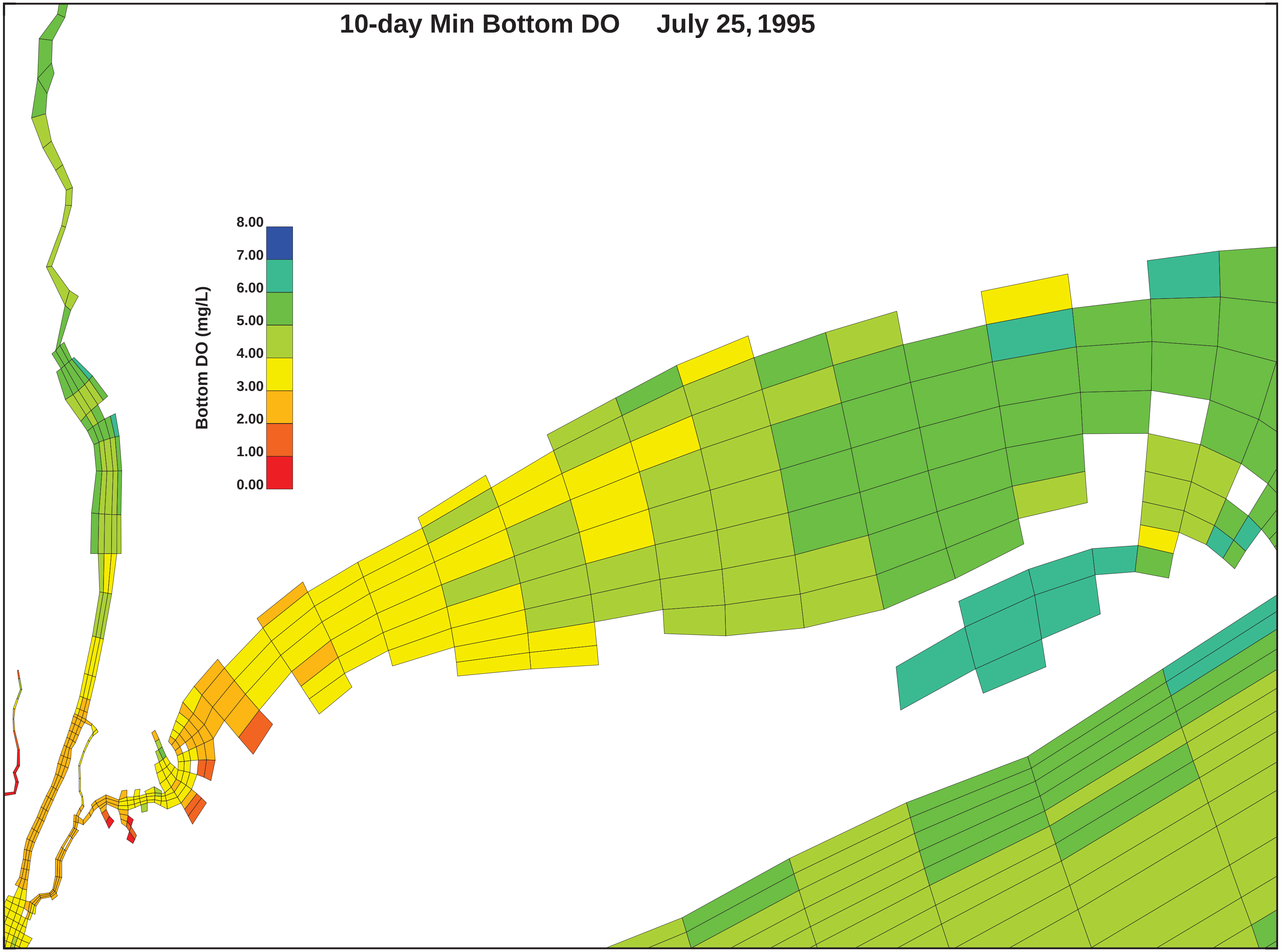
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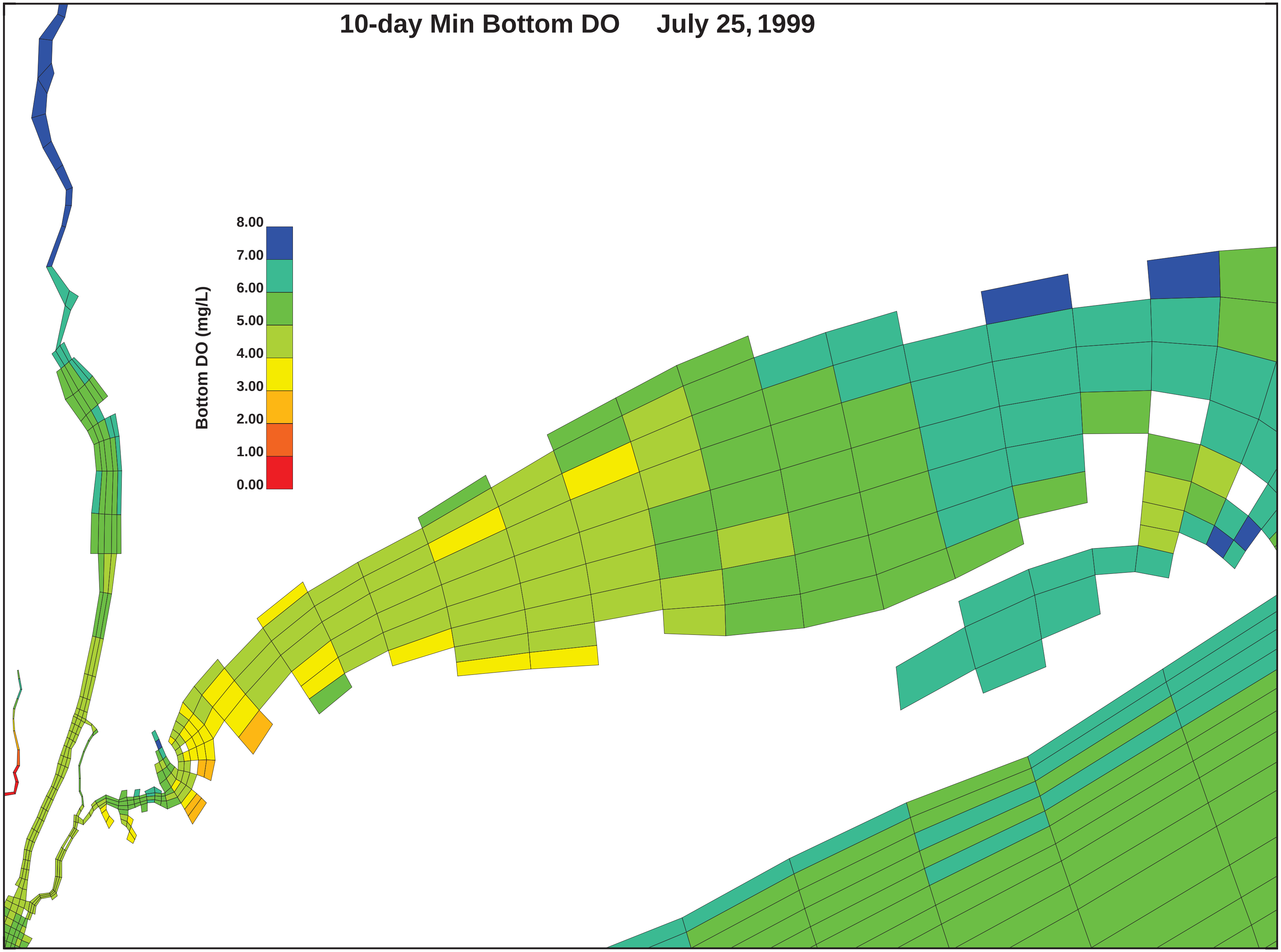

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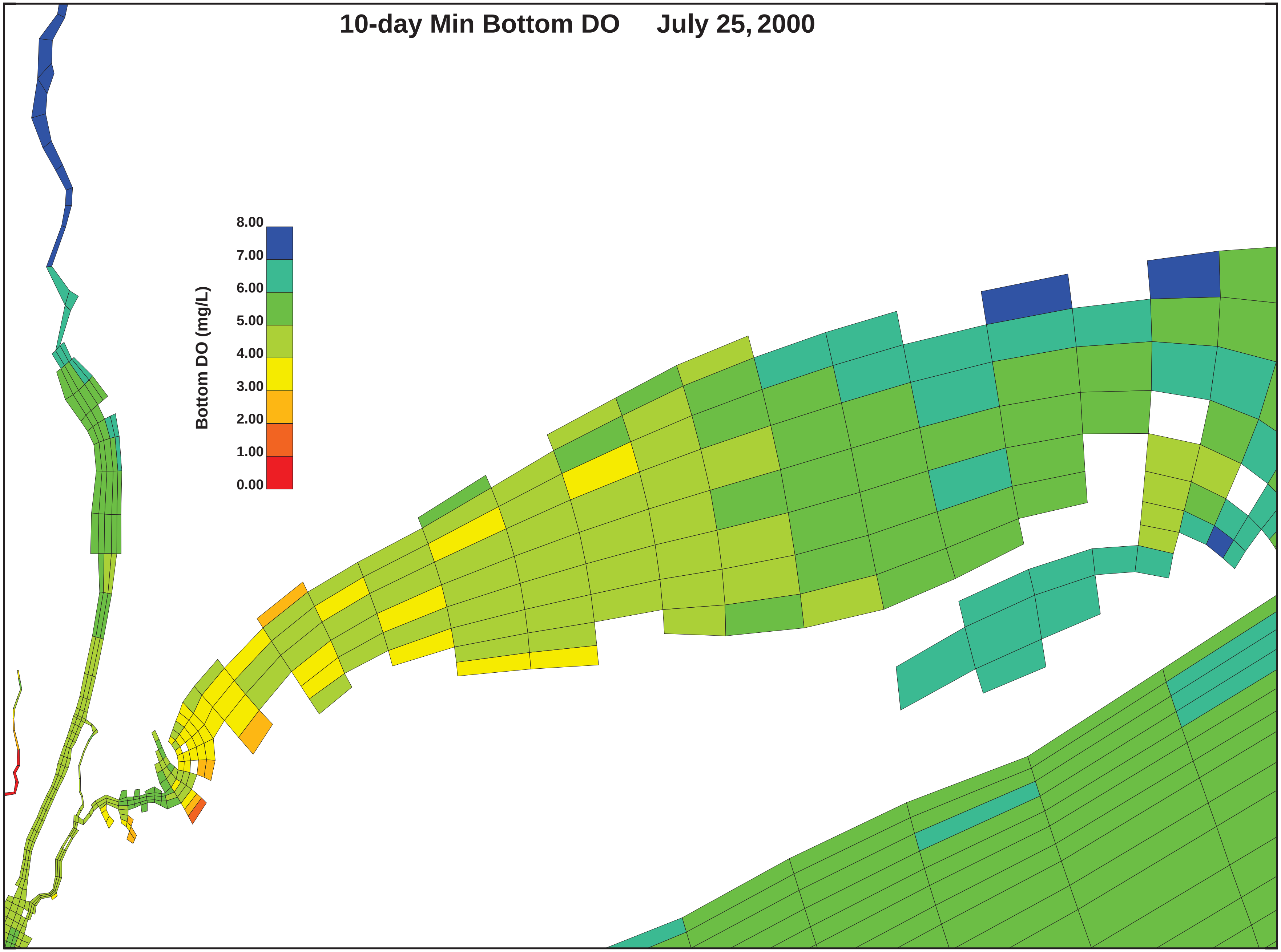
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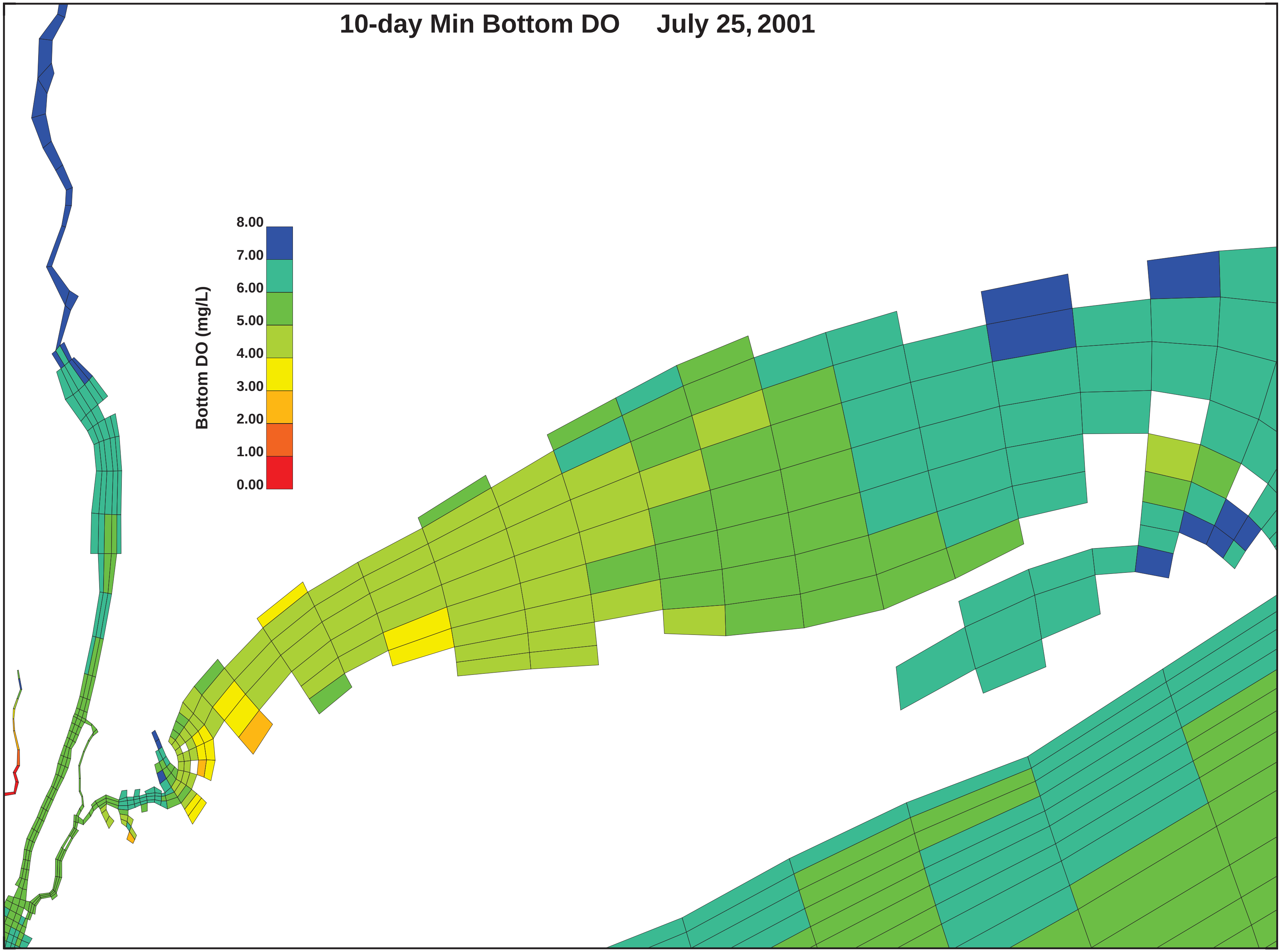
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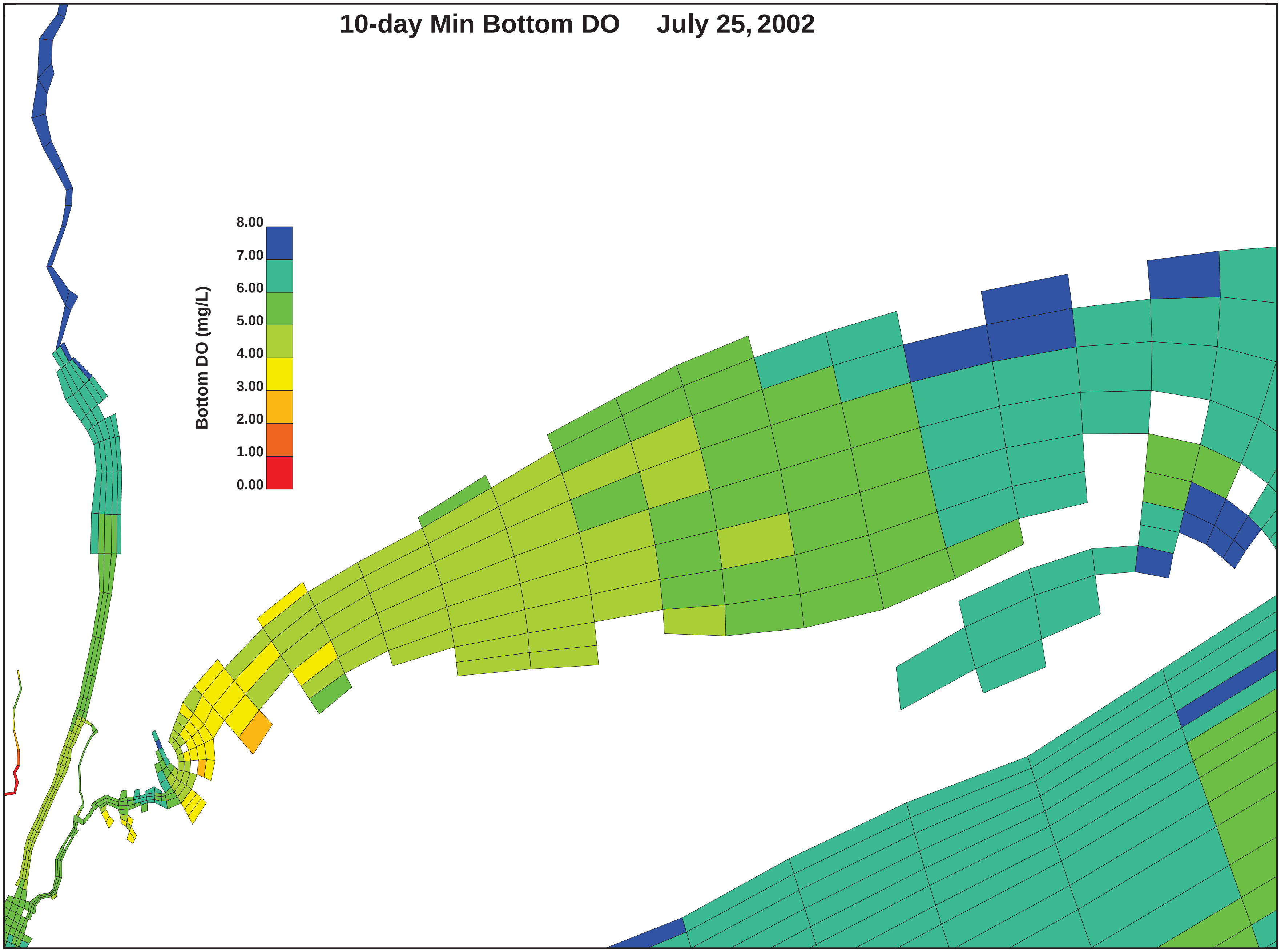
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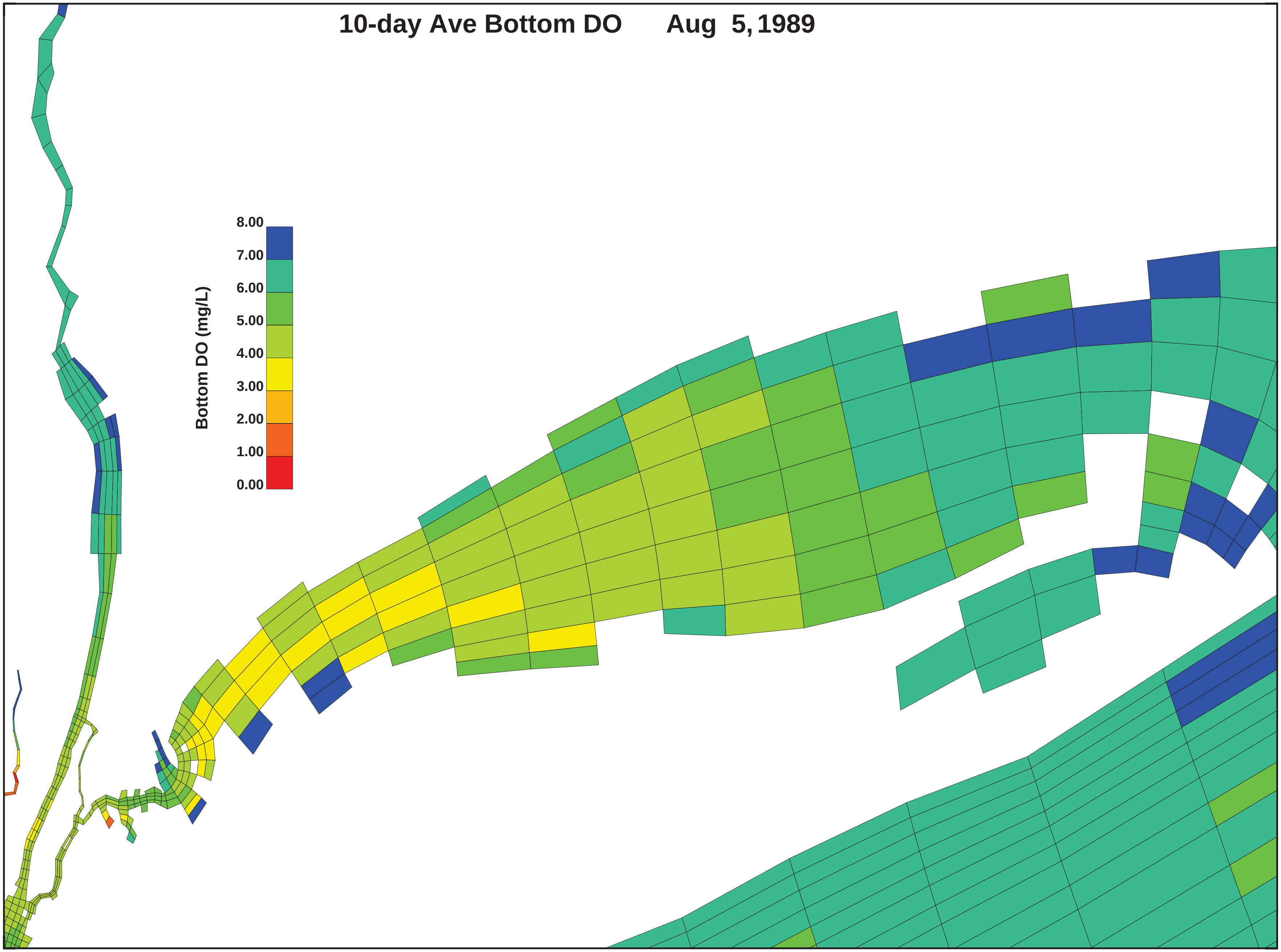

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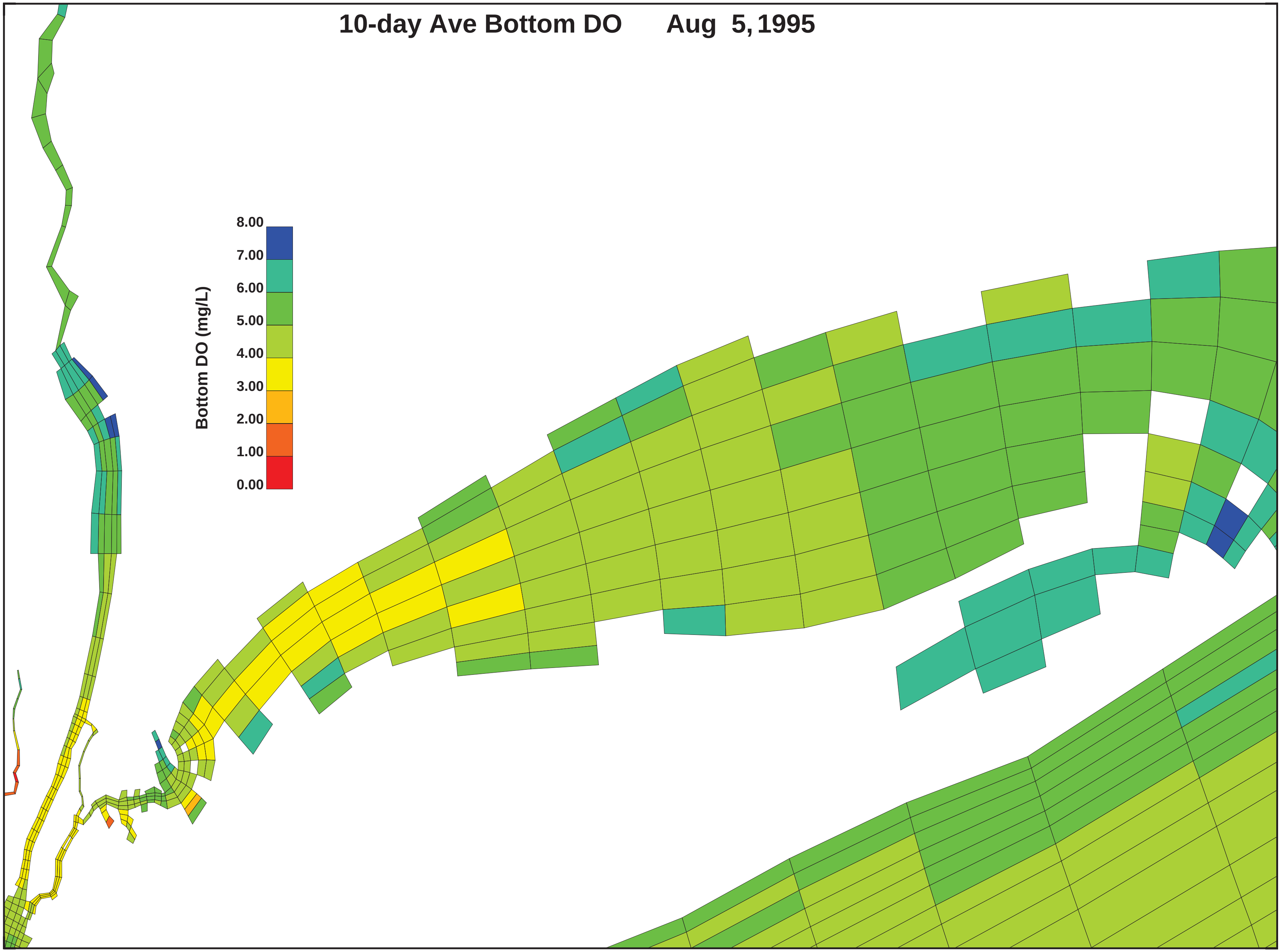
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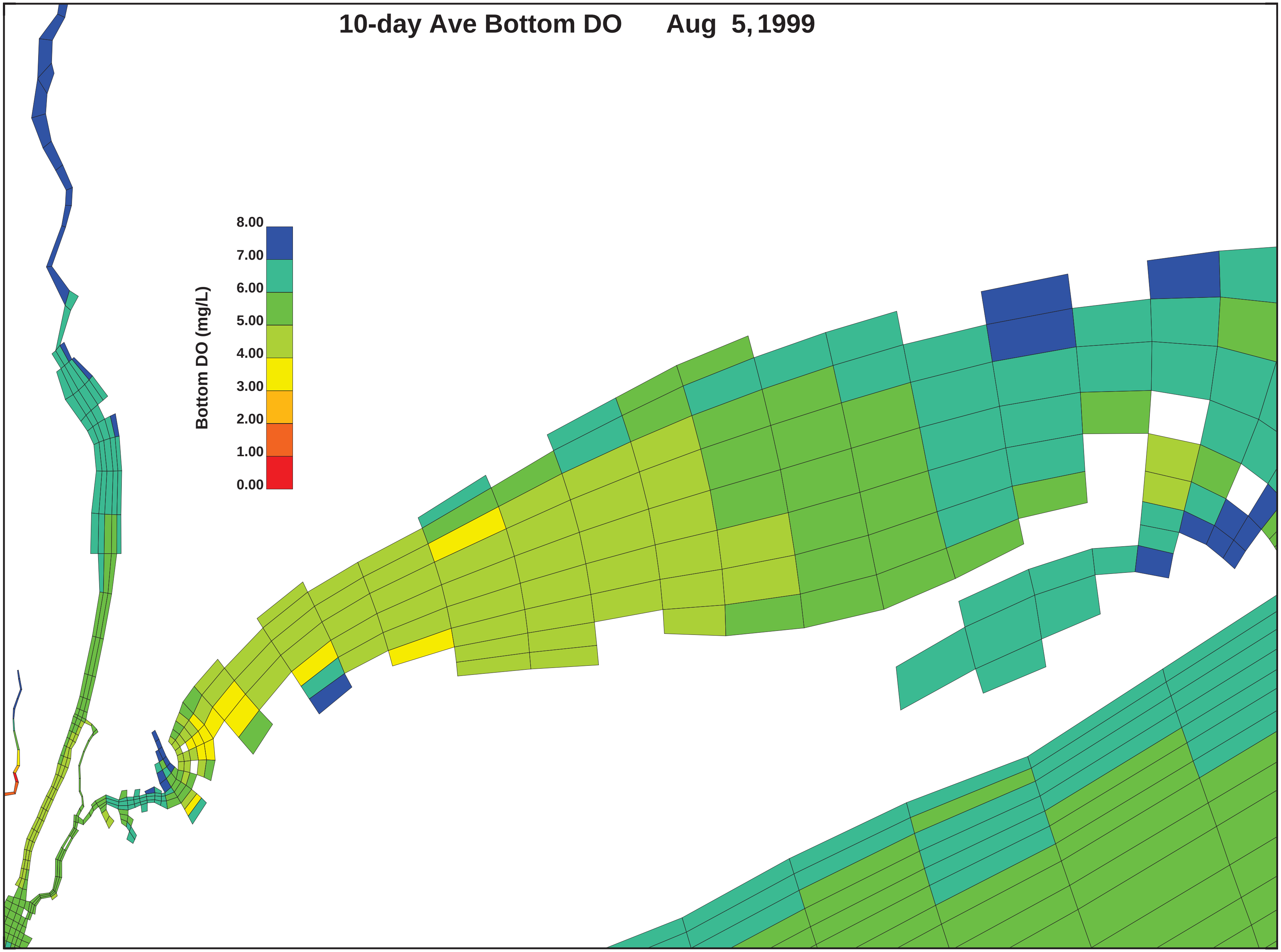
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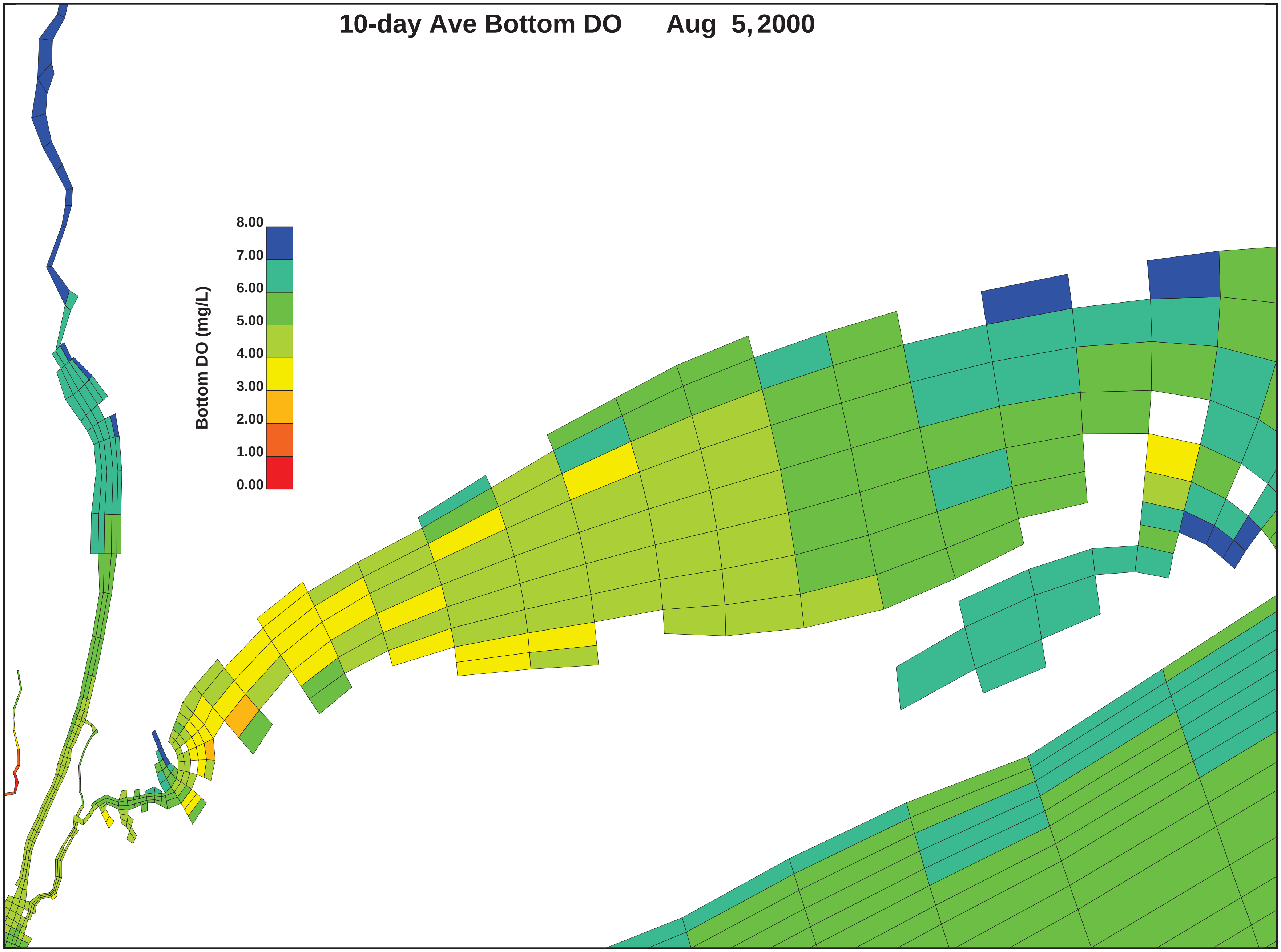
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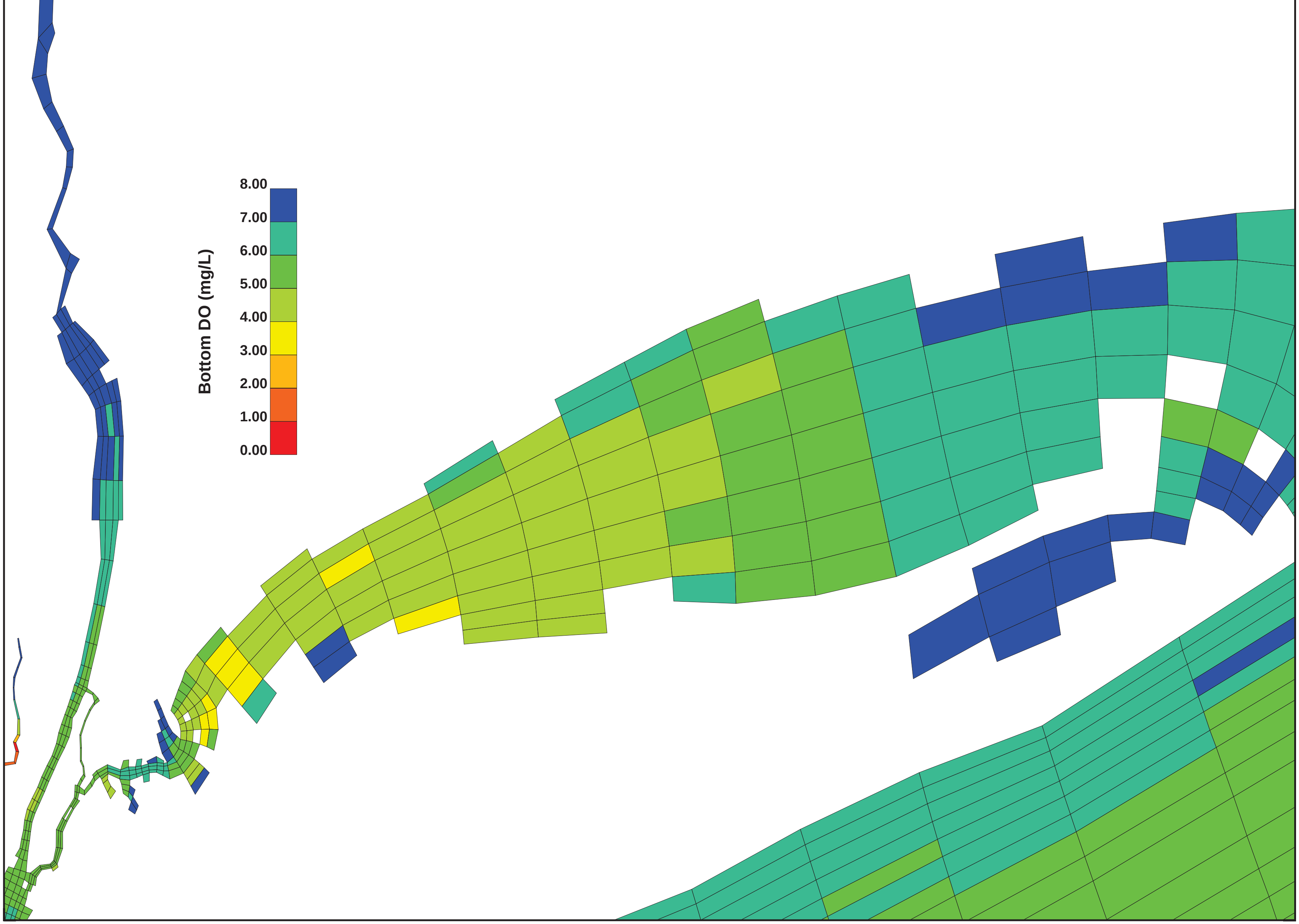
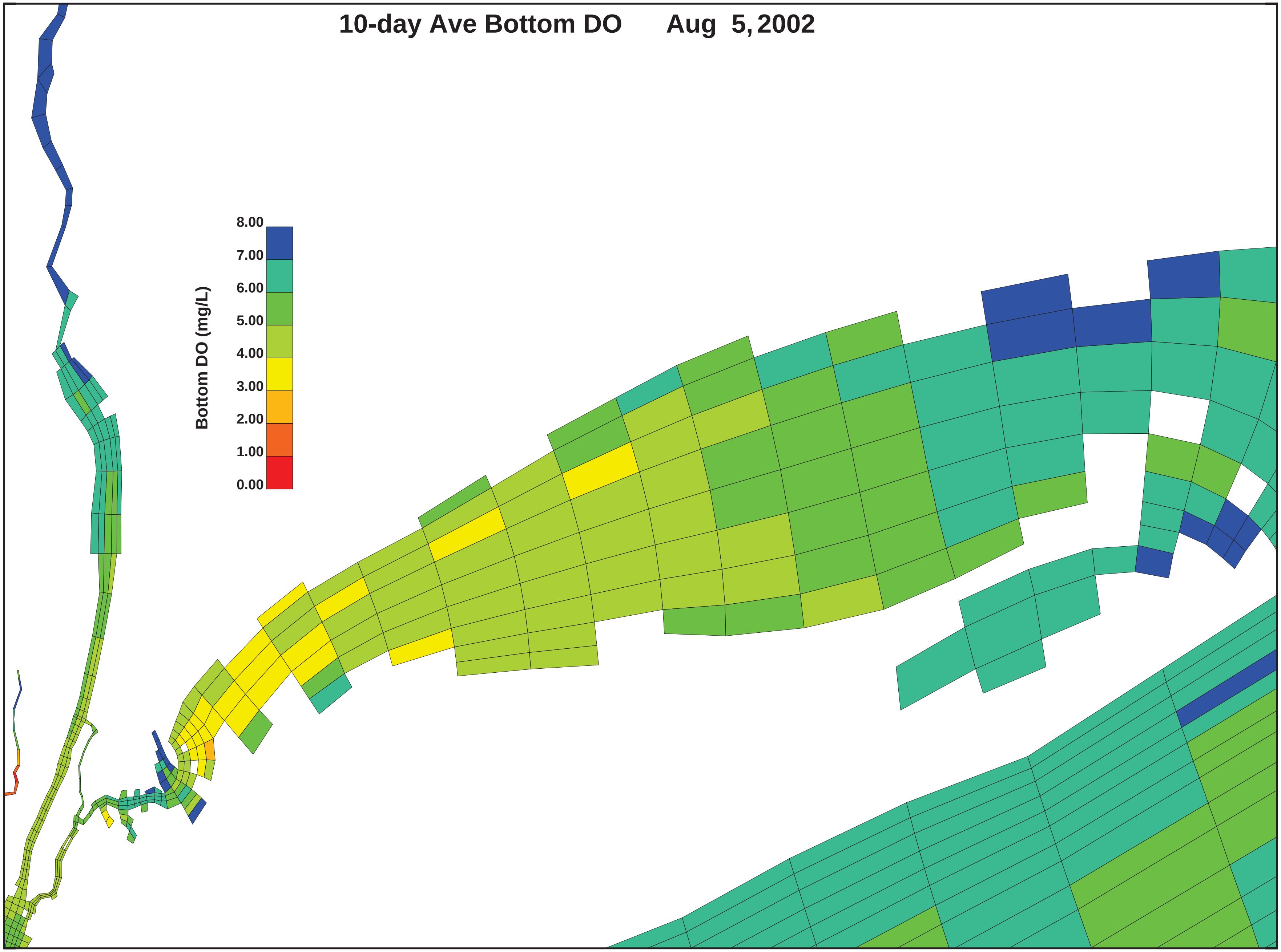
Bottom DO (mg/L)

10-day Ave Bottom DO   Aug 5, 2001





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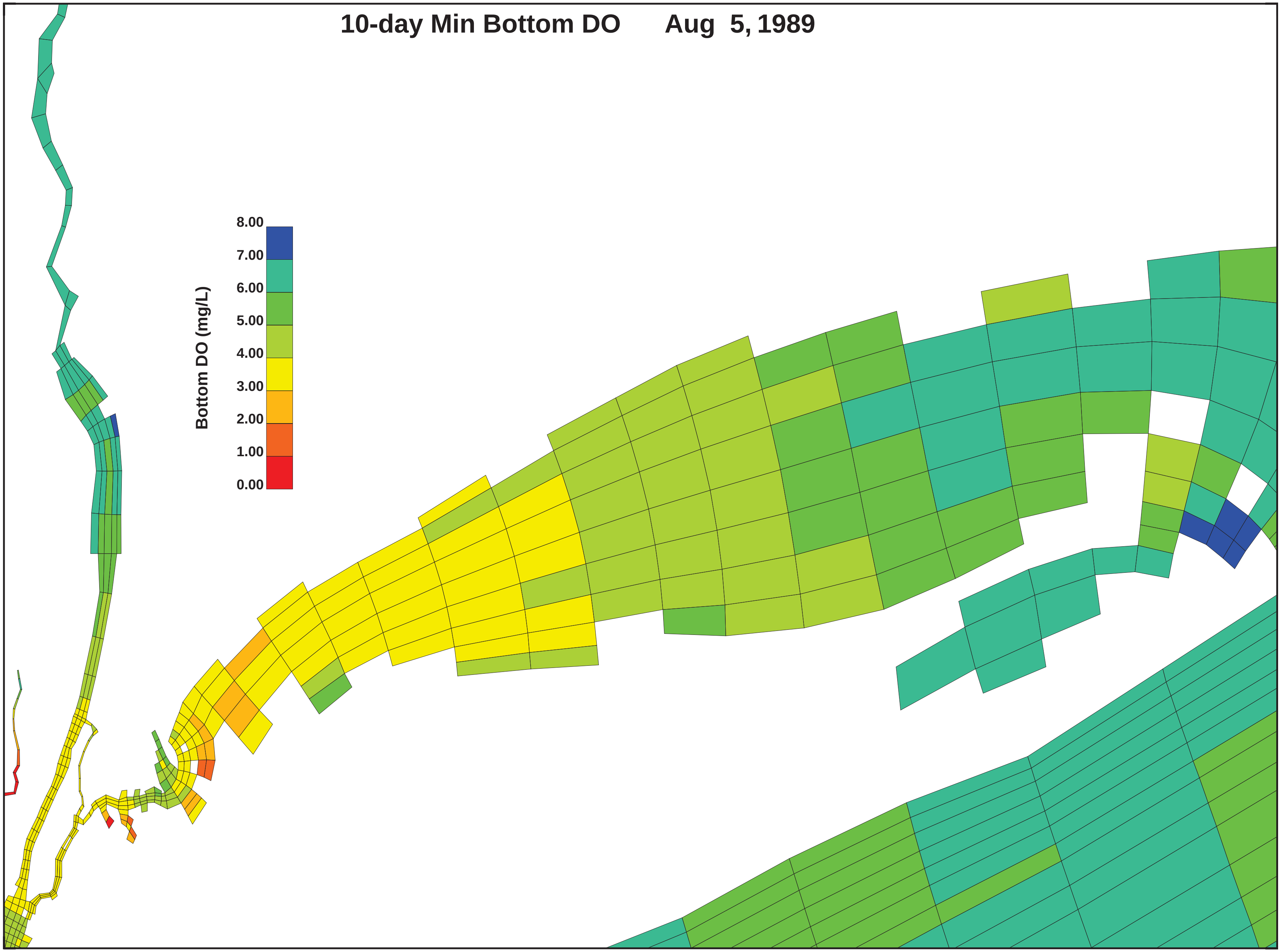
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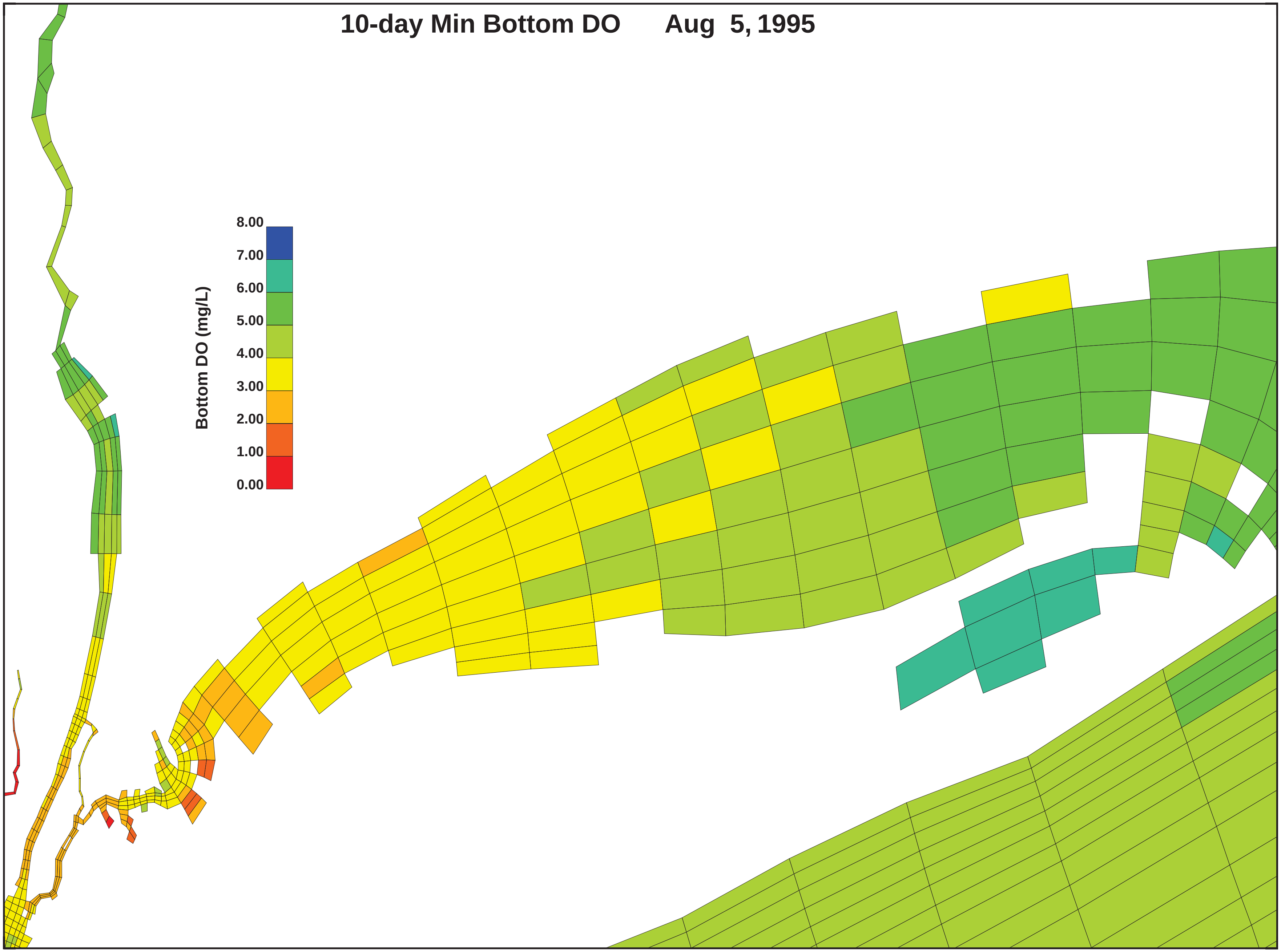
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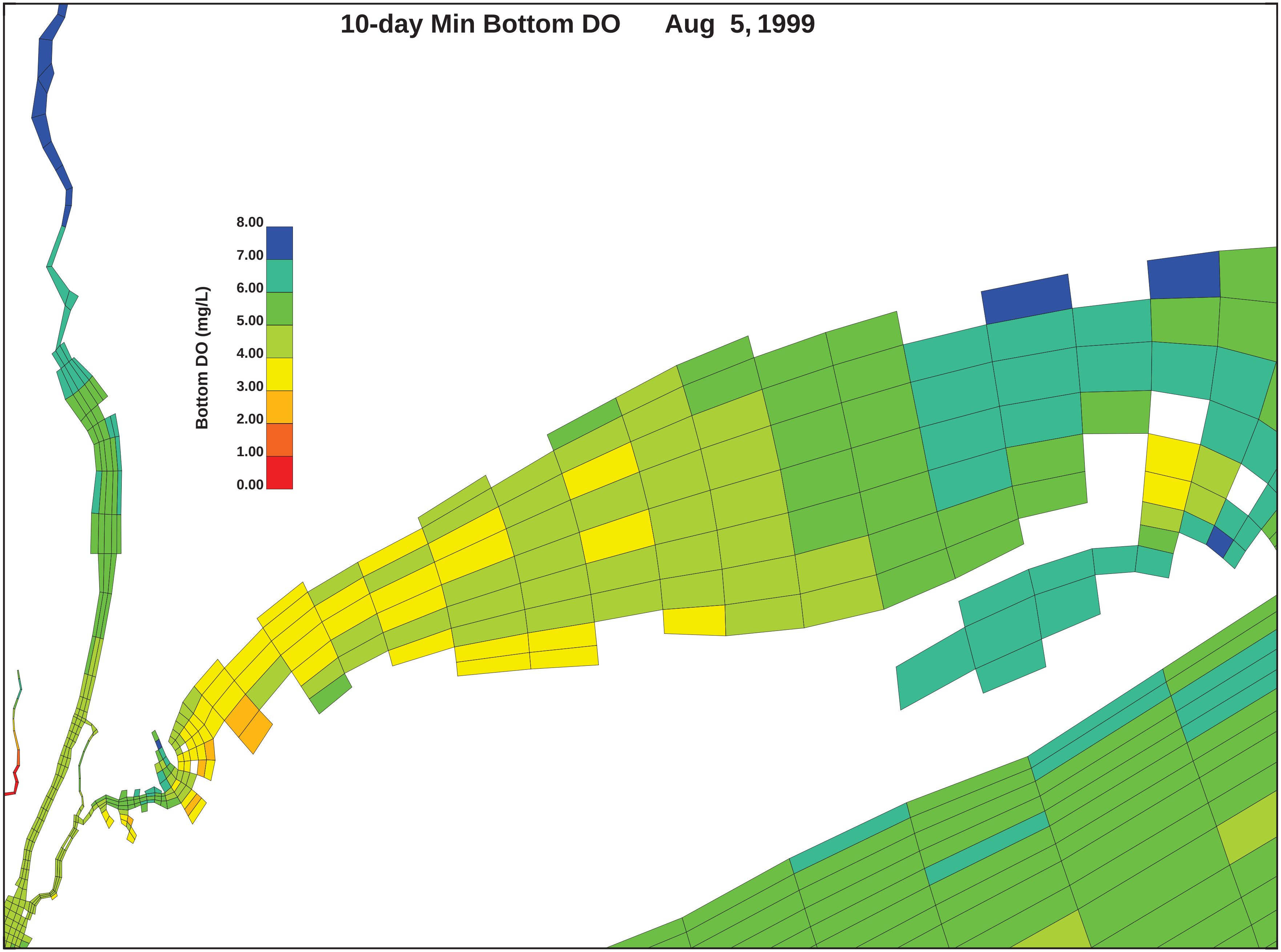
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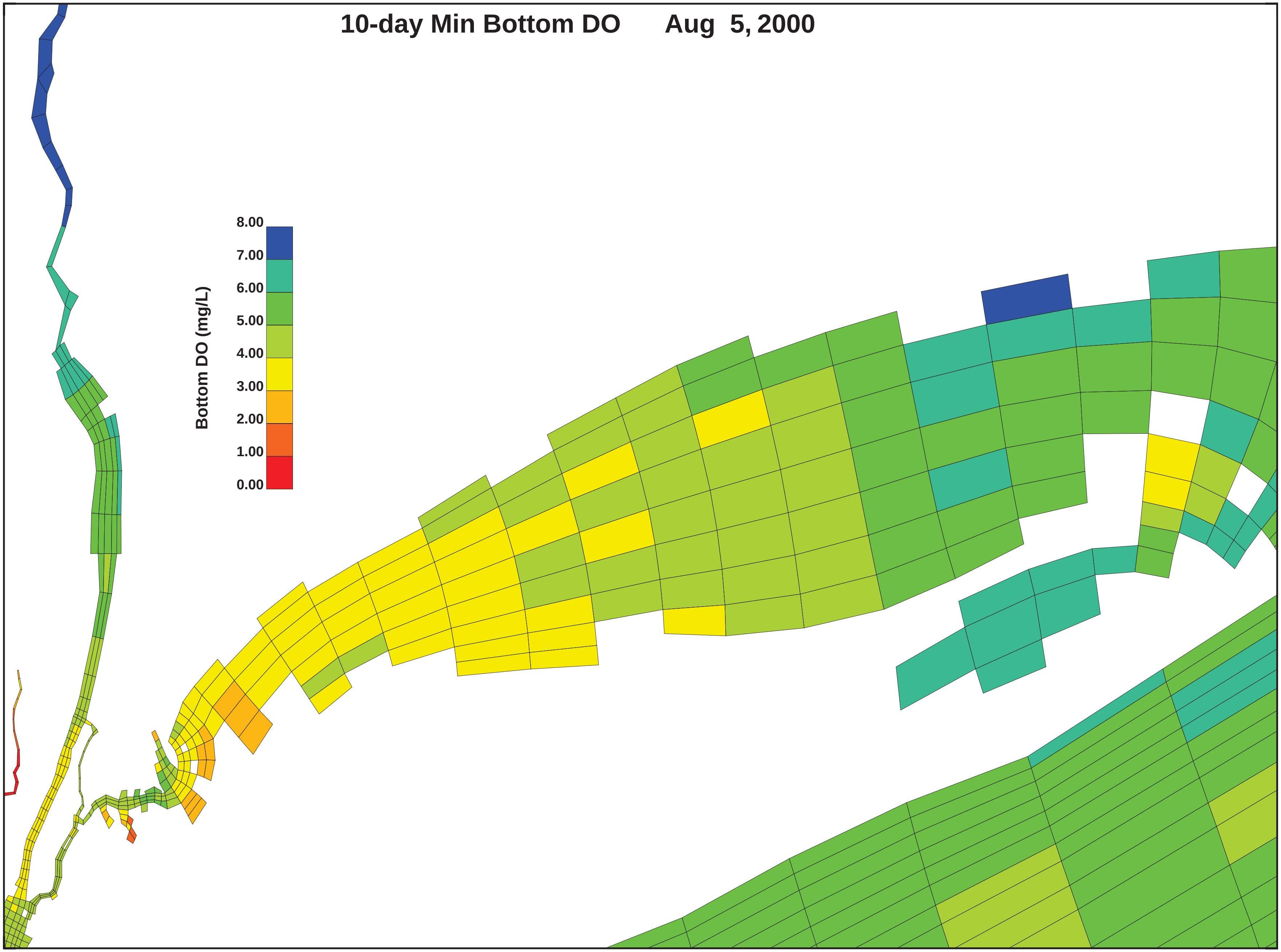

10-day Min Bottom DO Aug 5, 2000
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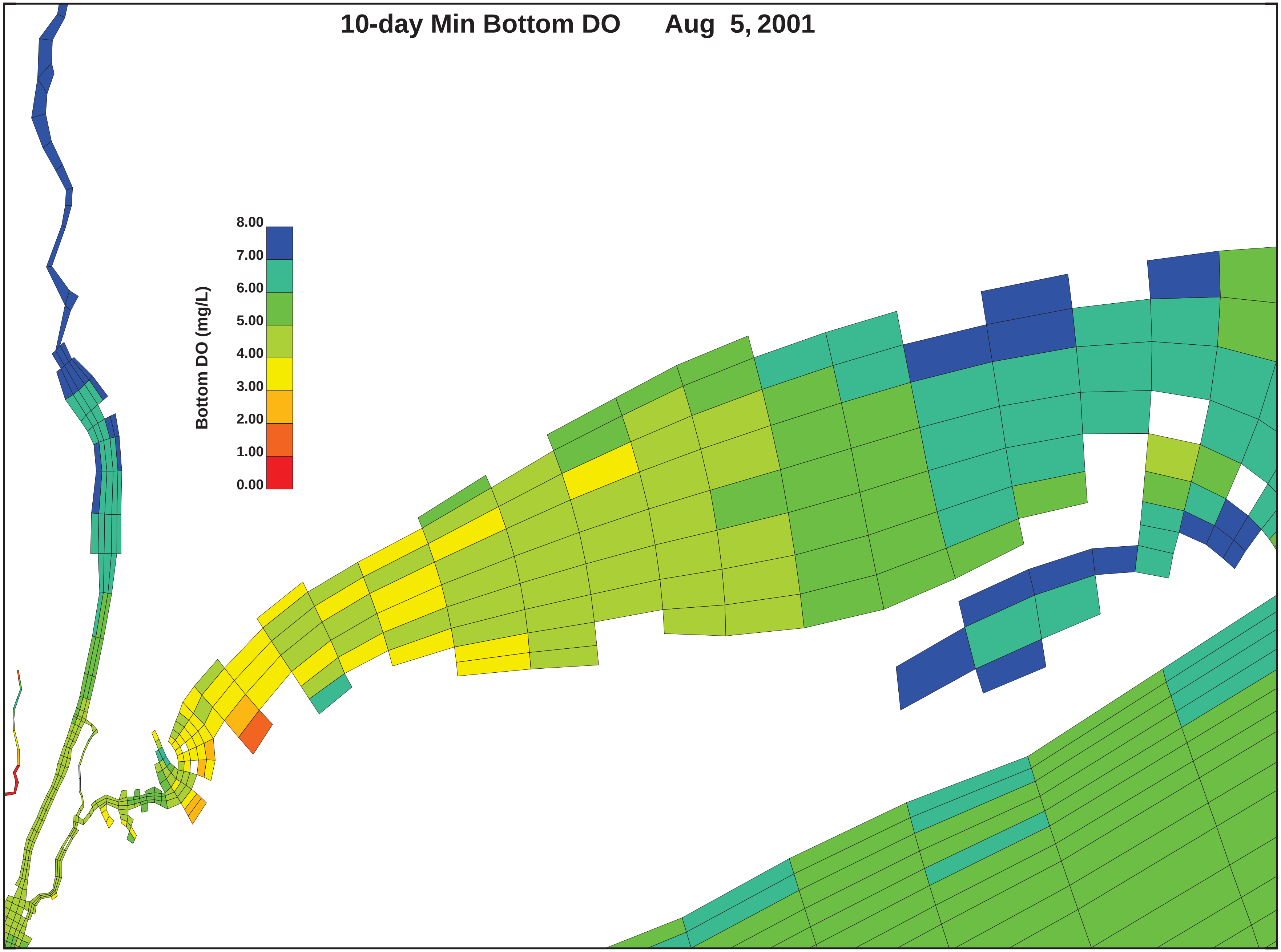
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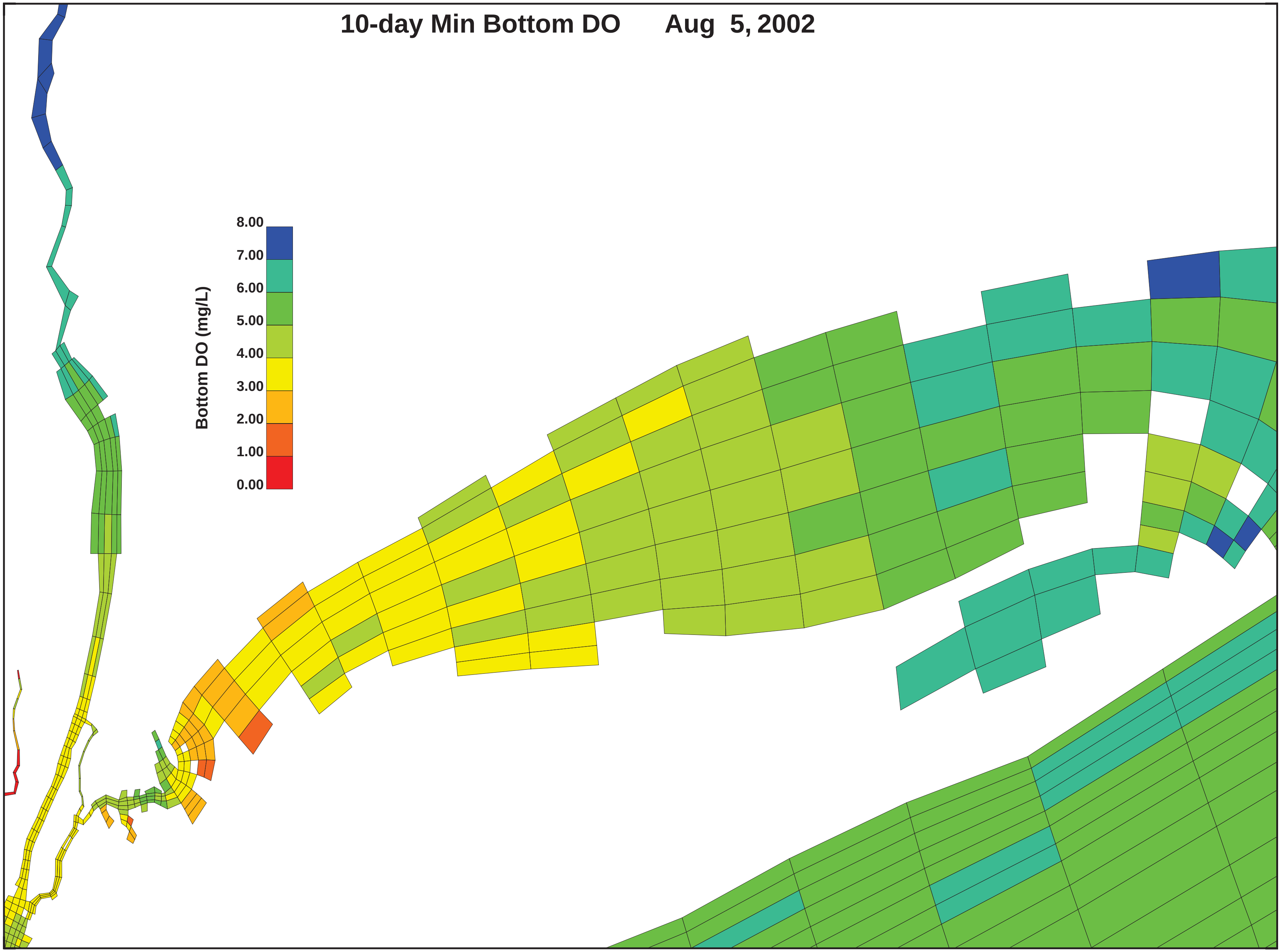
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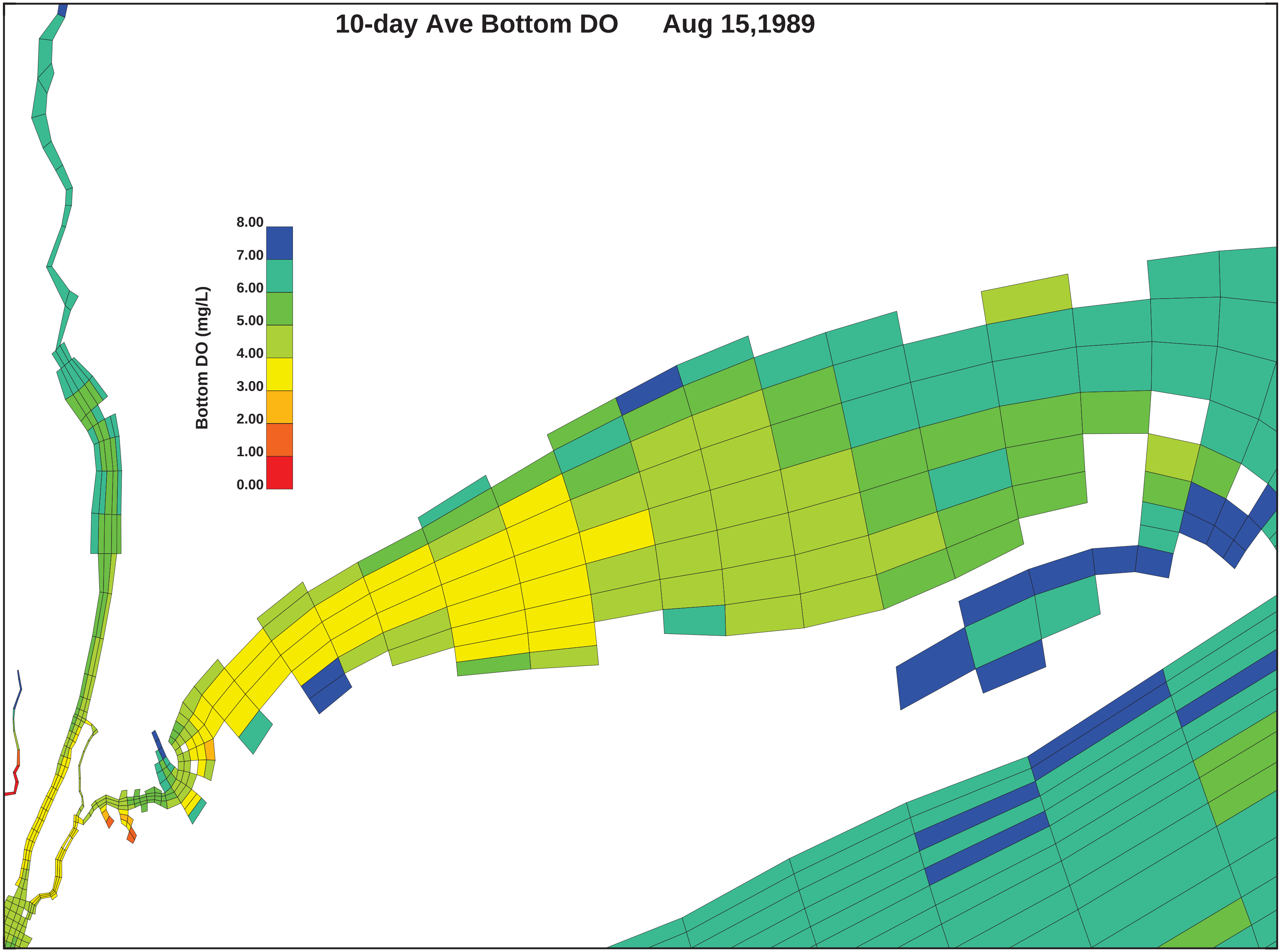

10-day Ave Bottom DO Aug 15,1989
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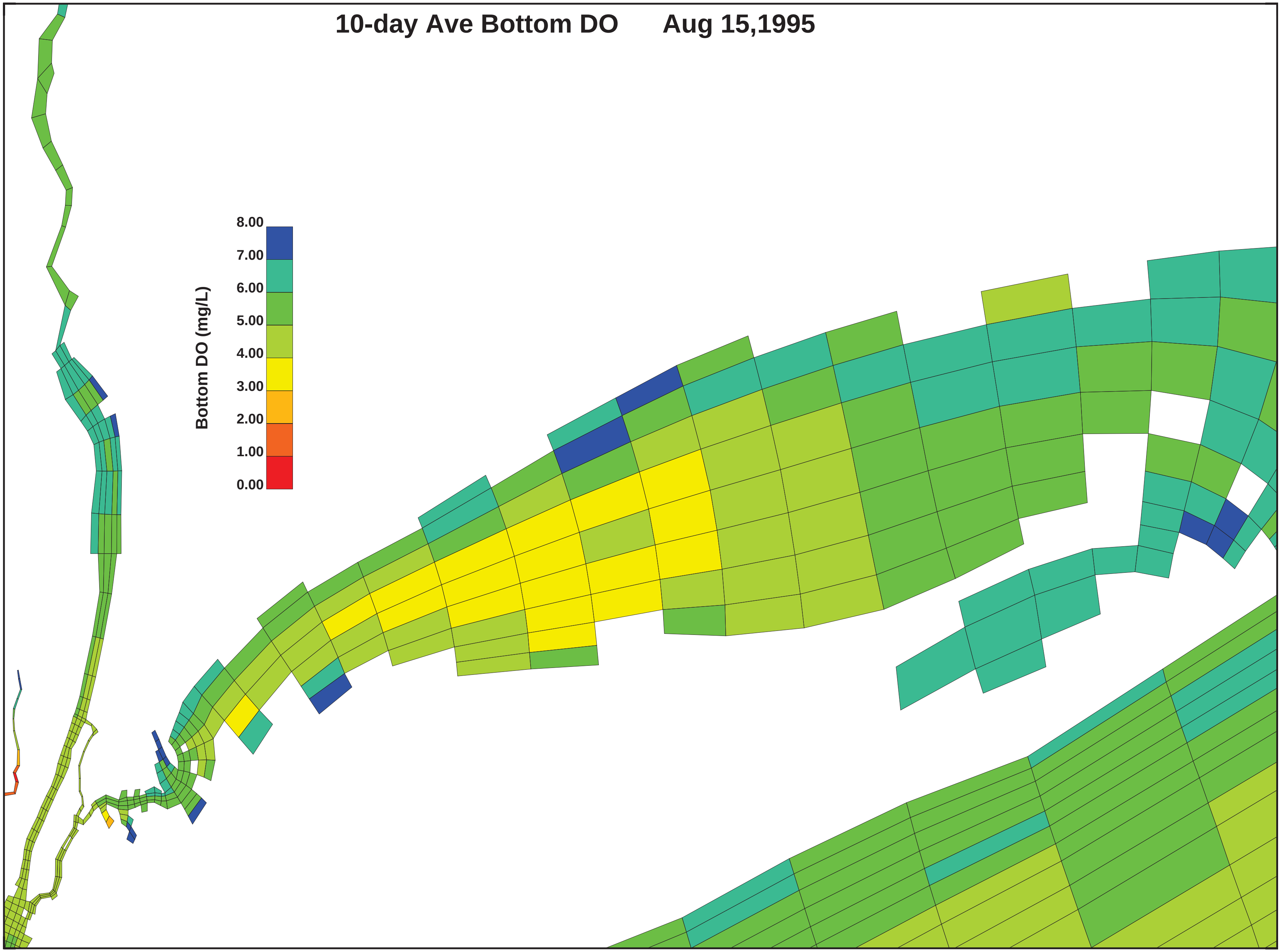

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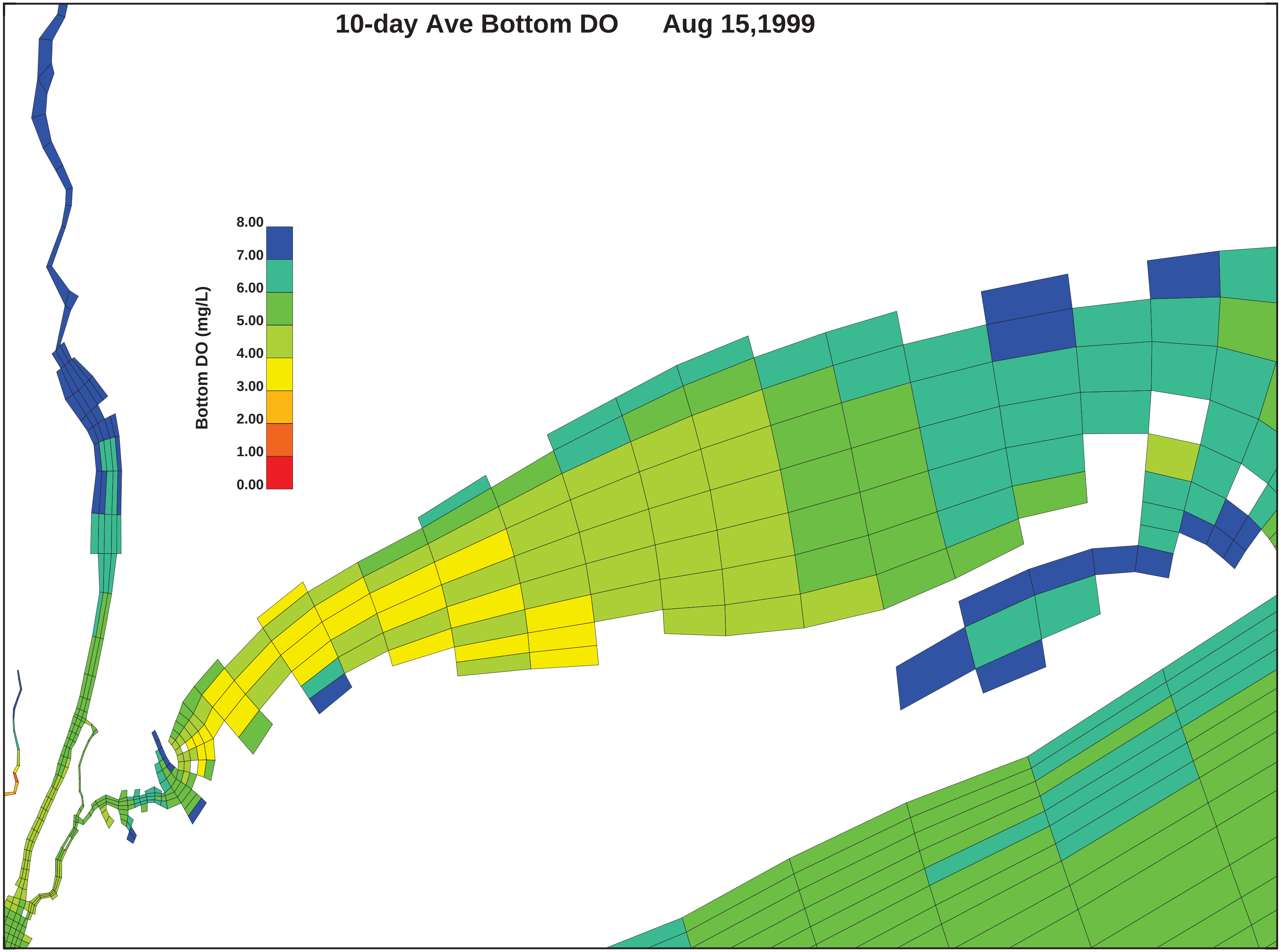

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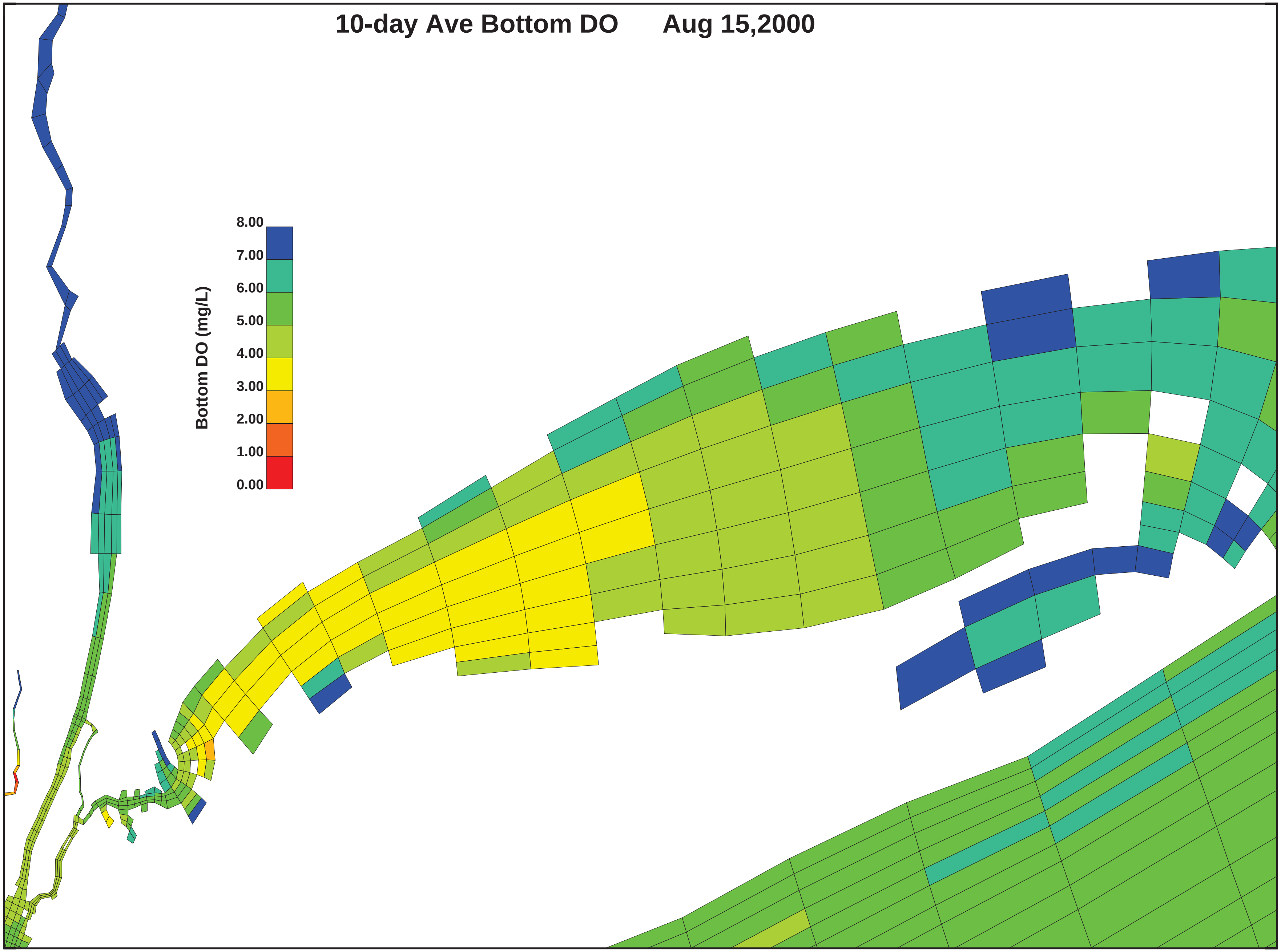
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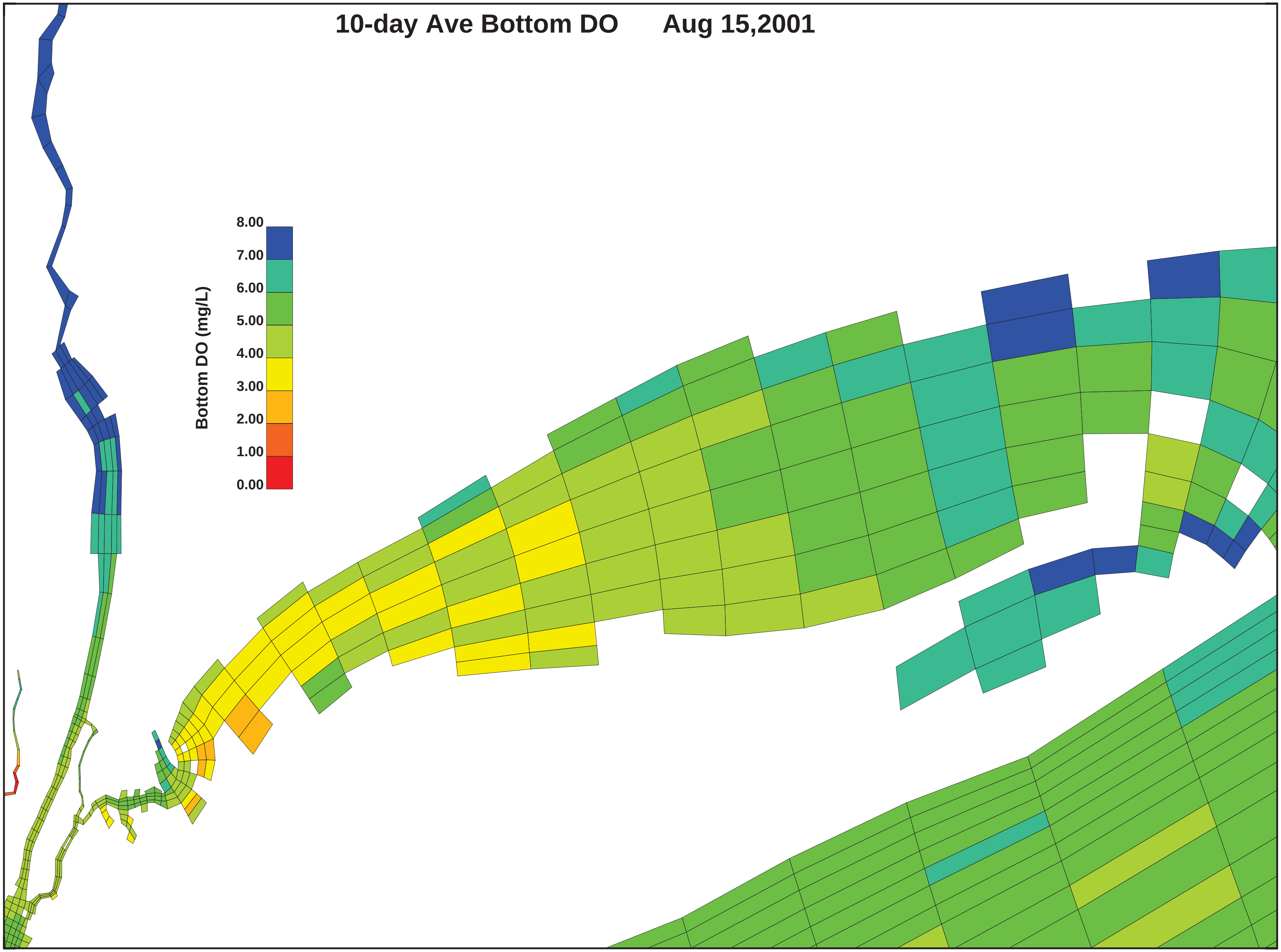
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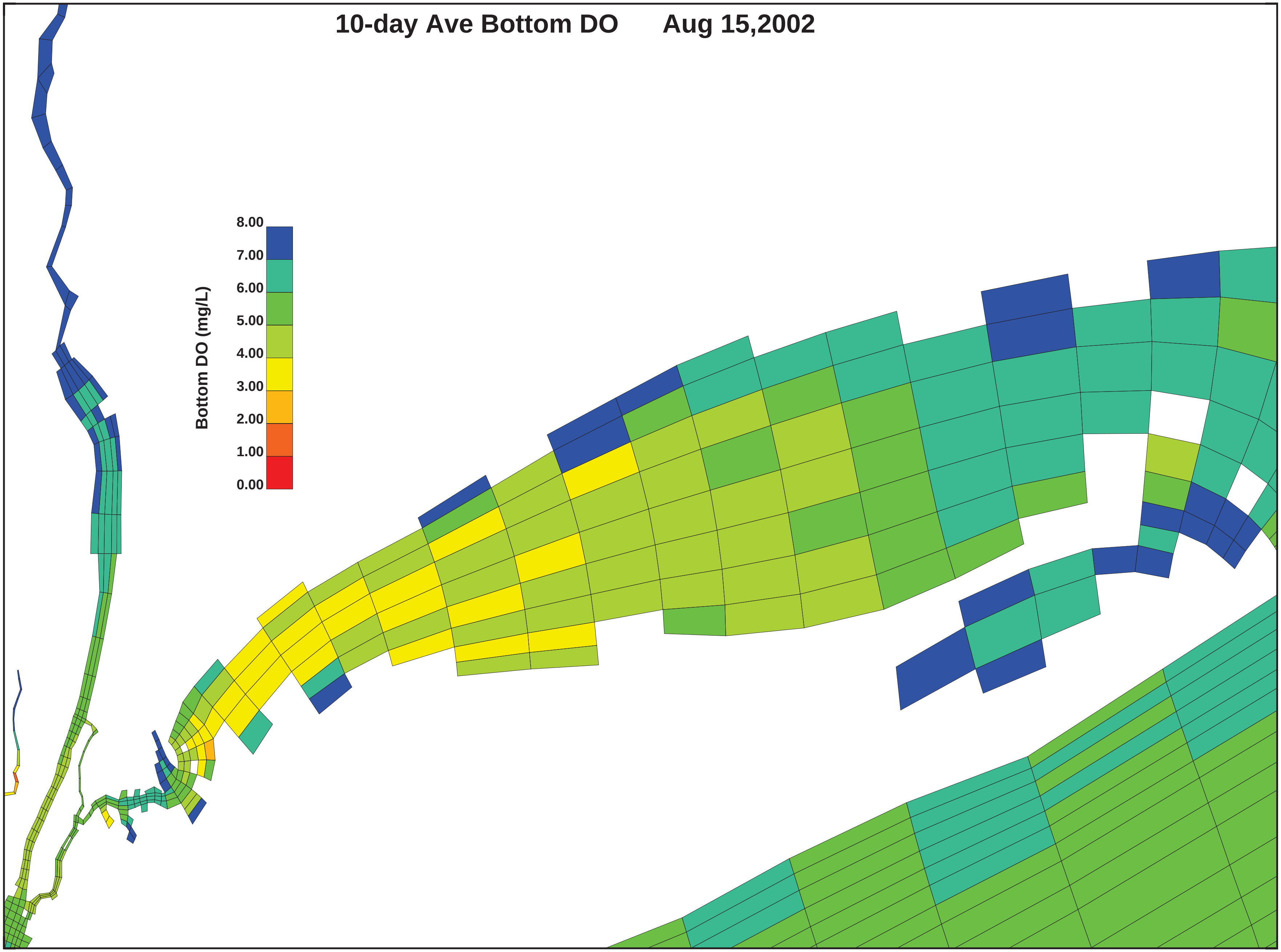

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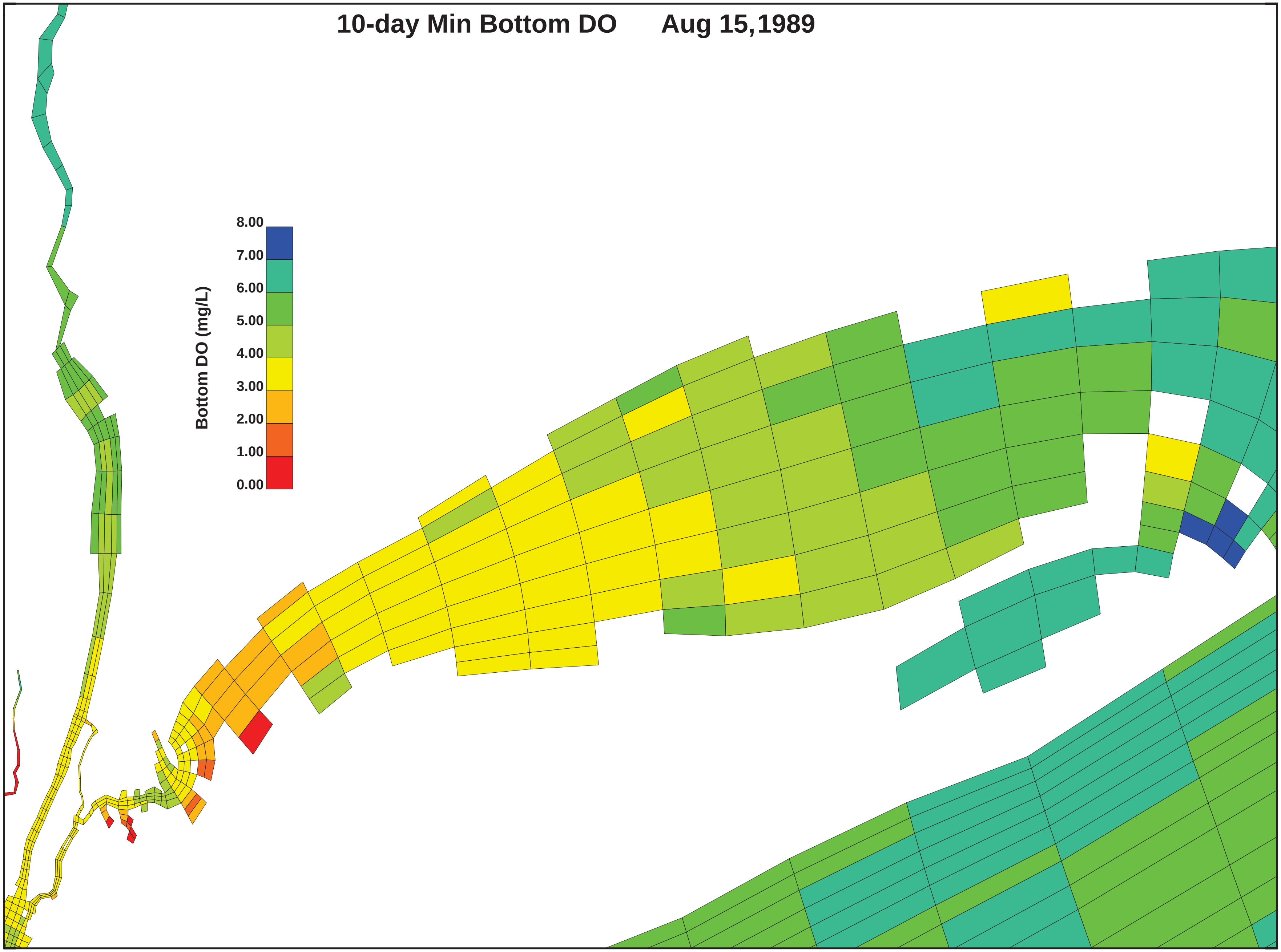
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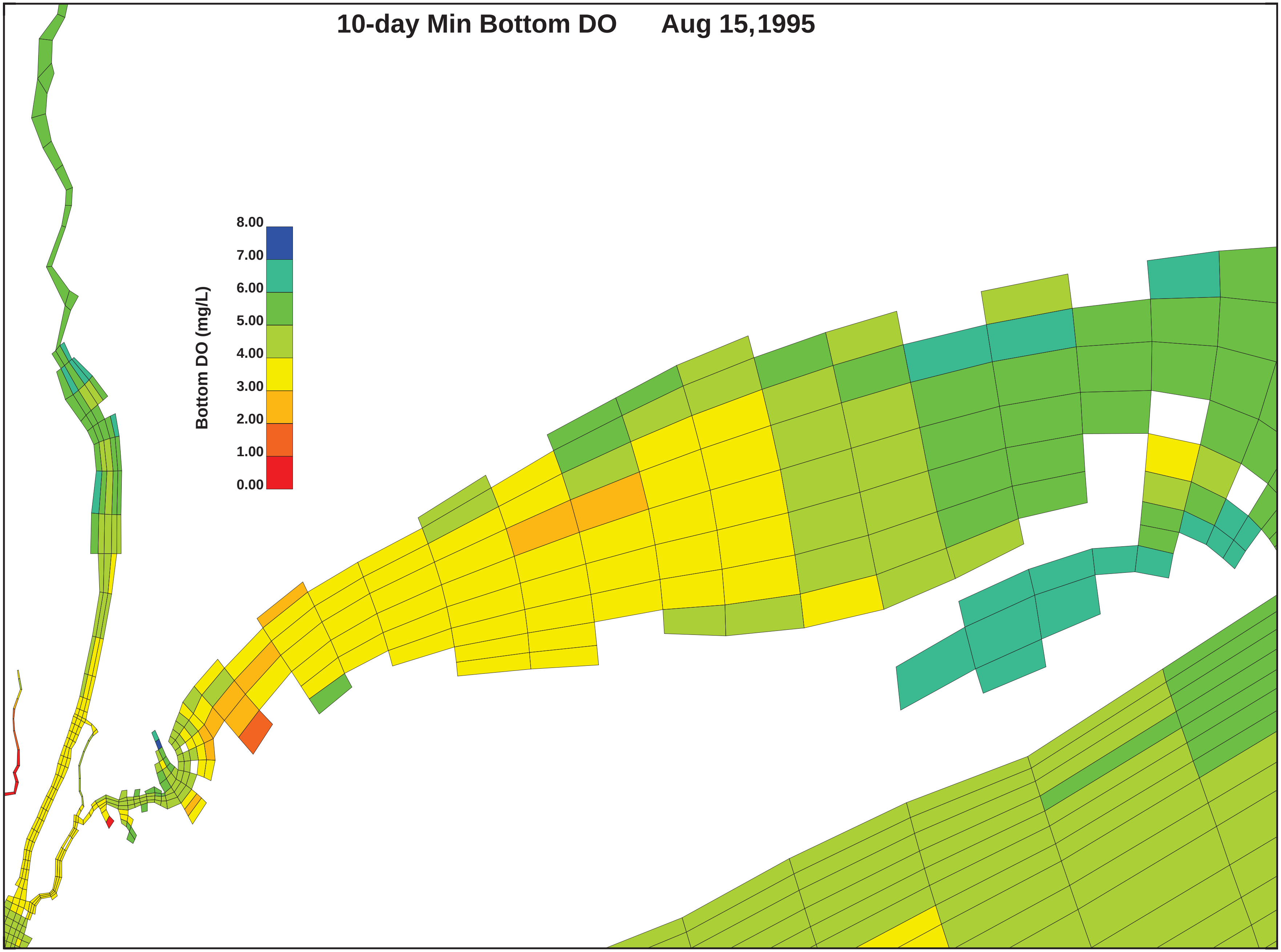
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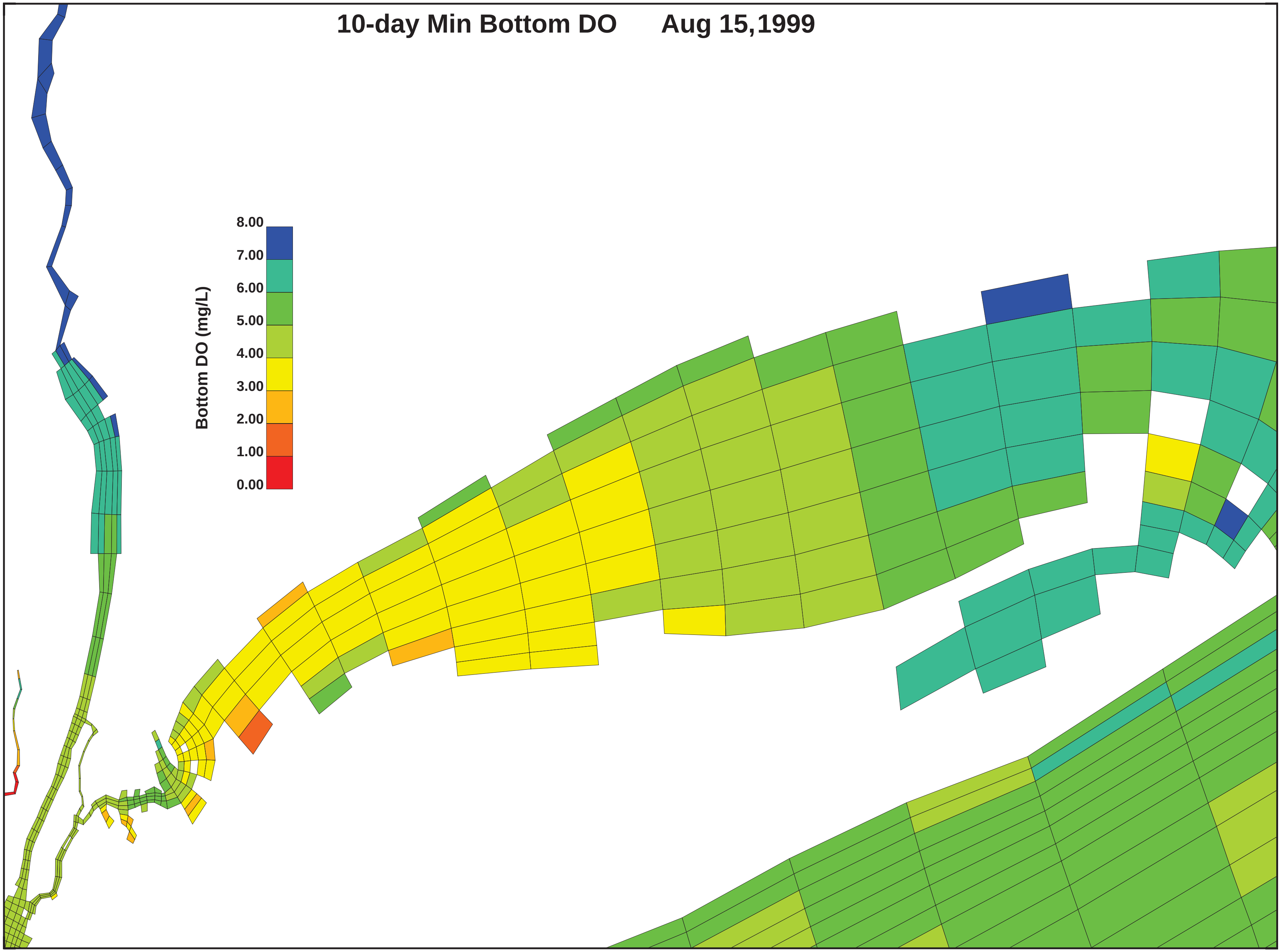

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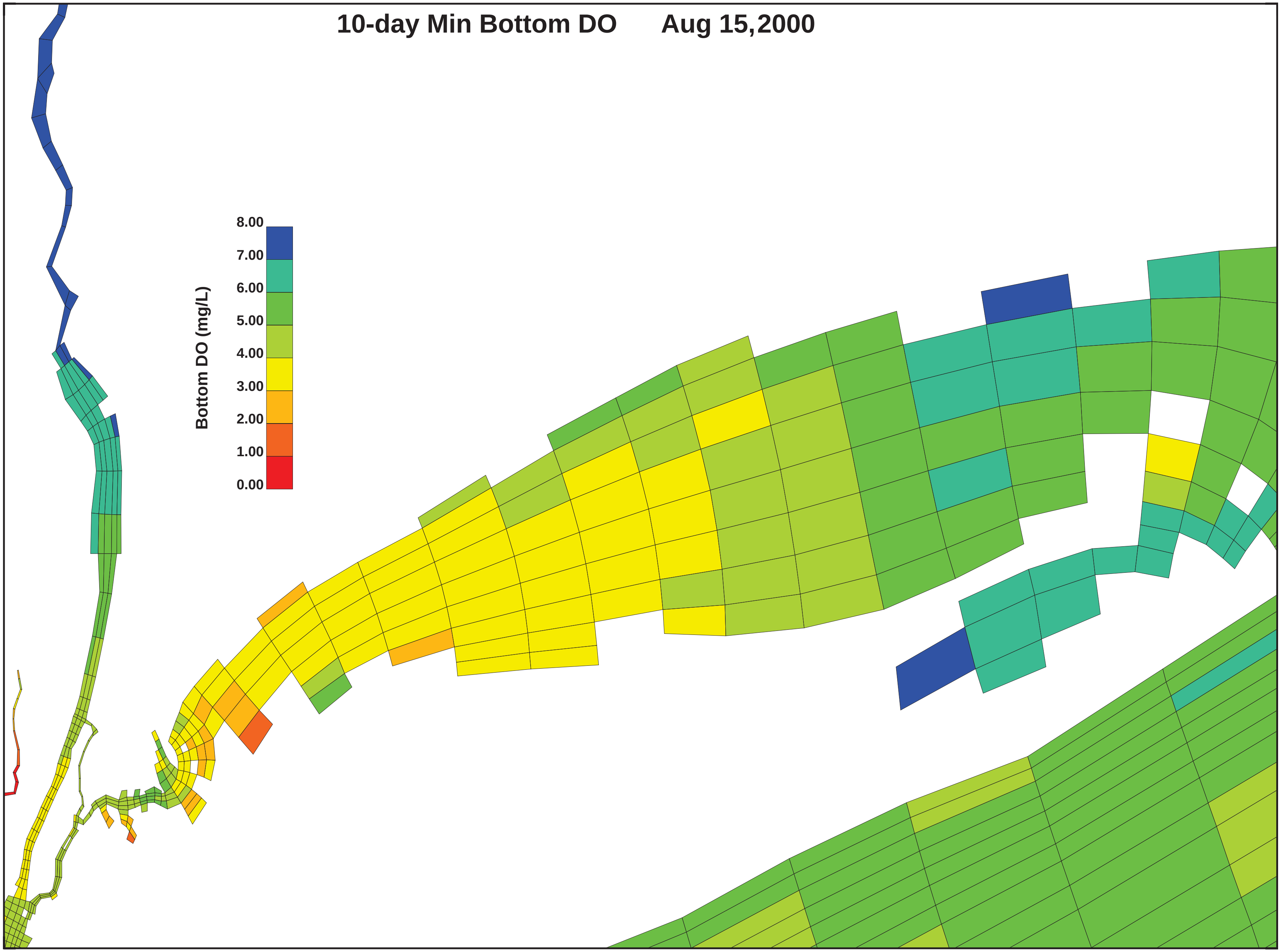
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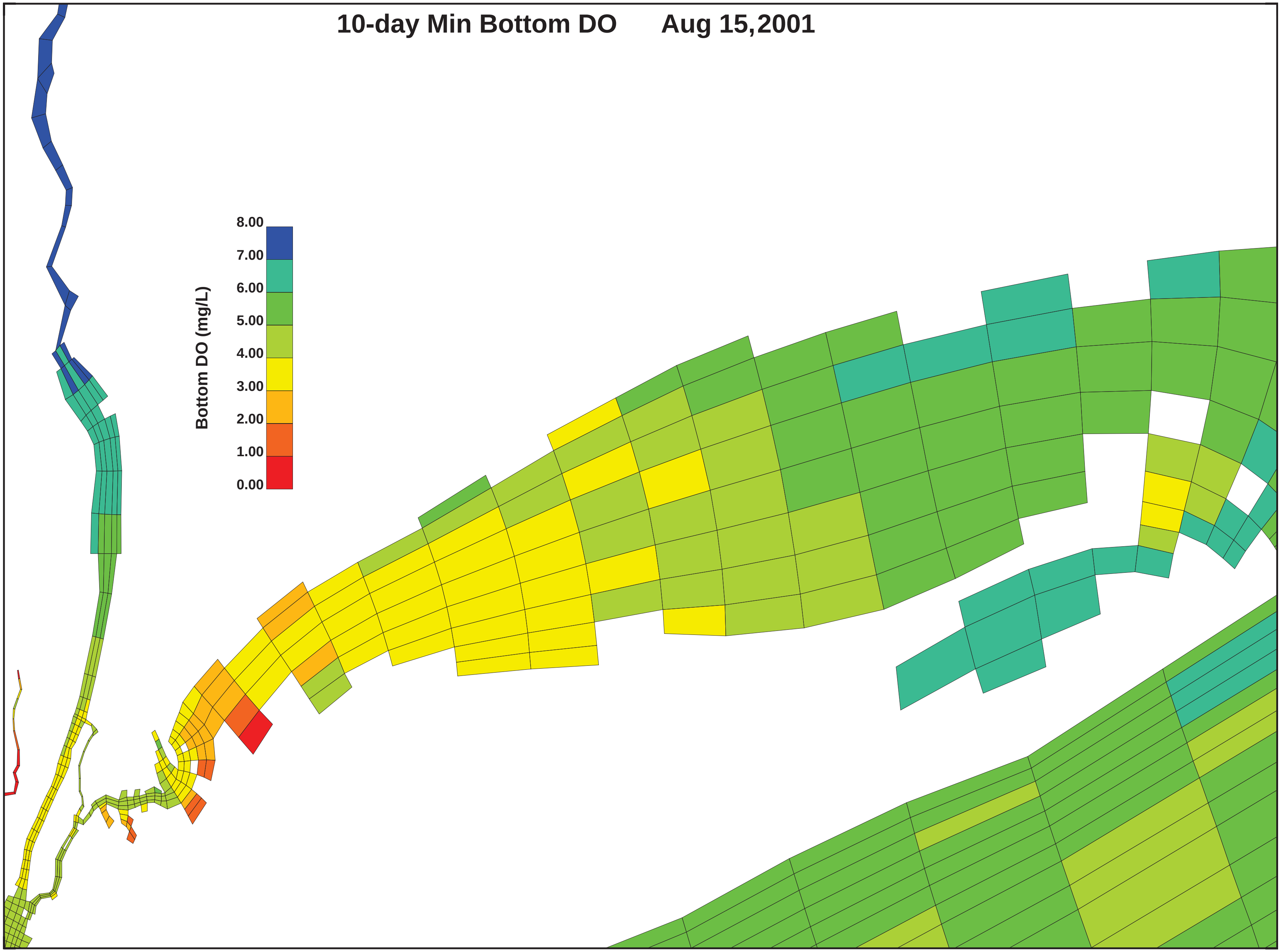
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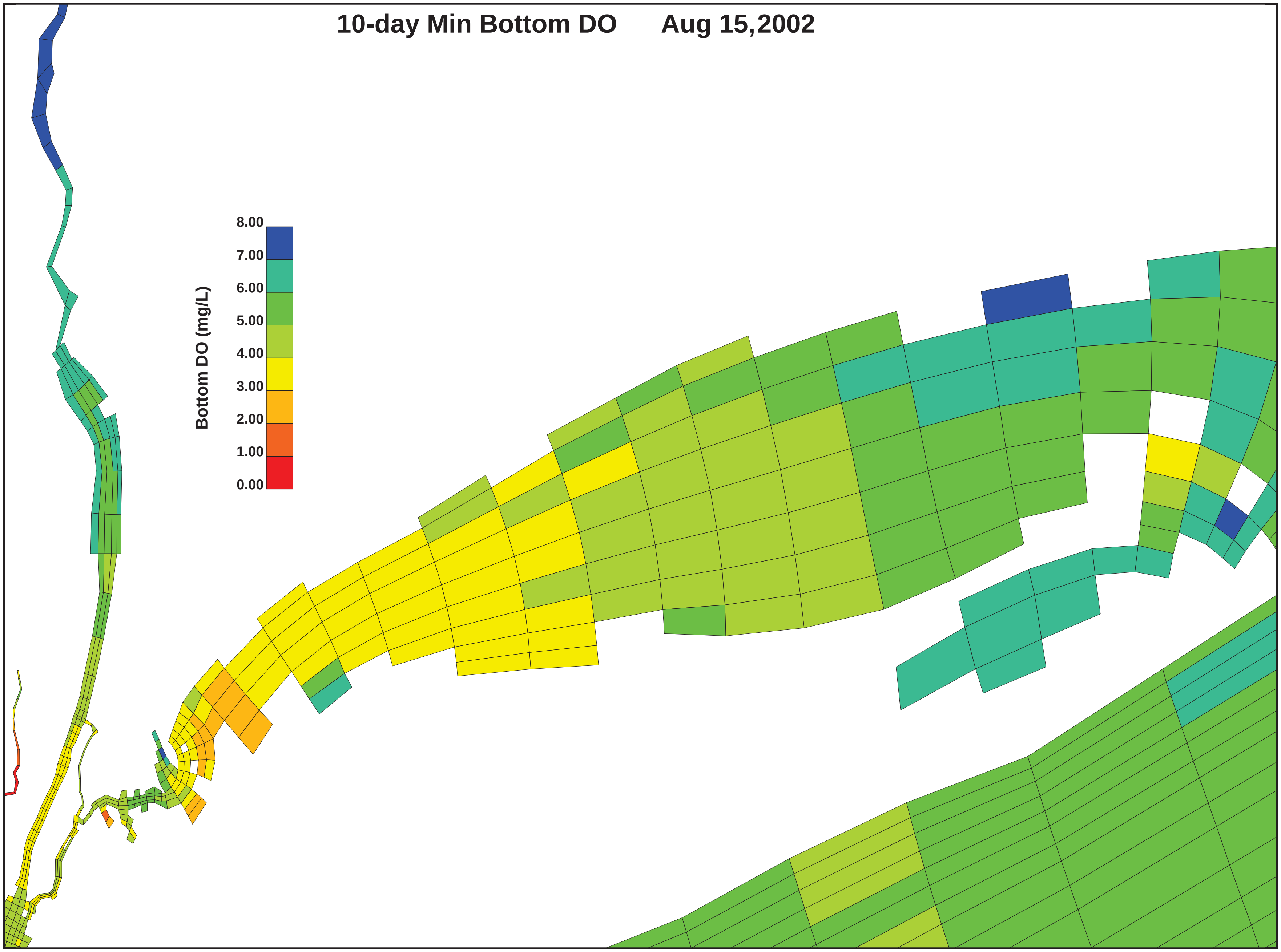

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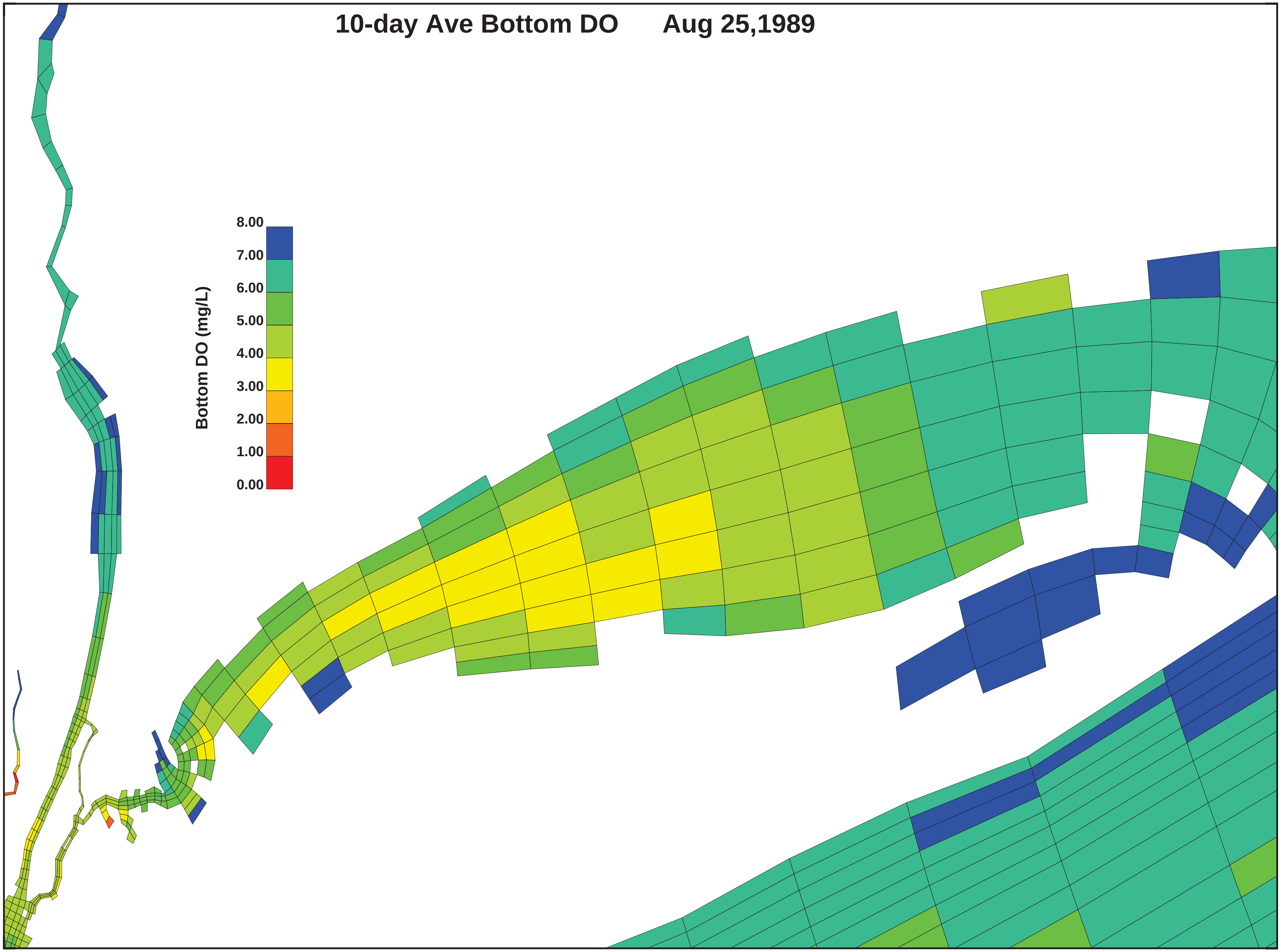

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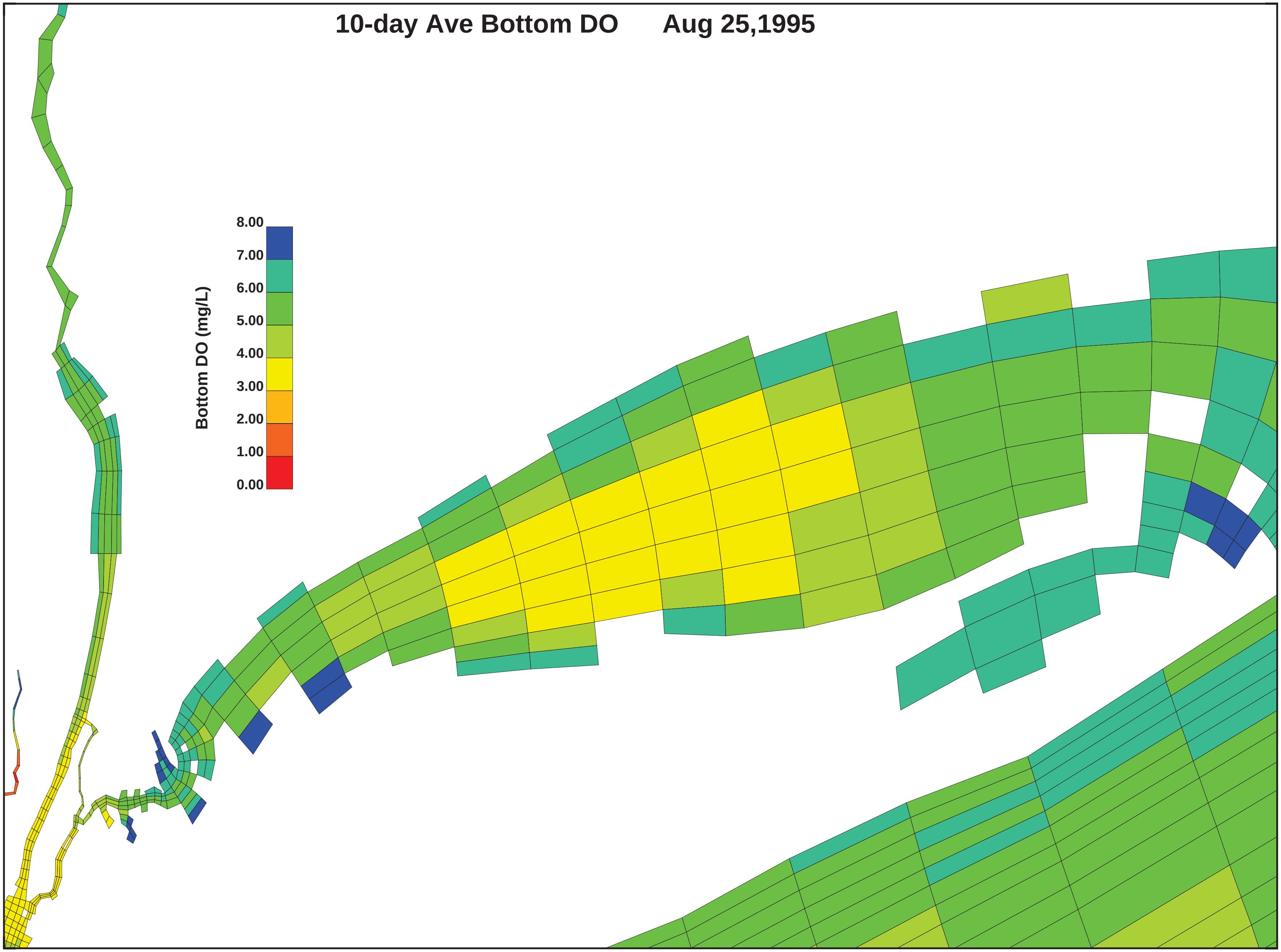

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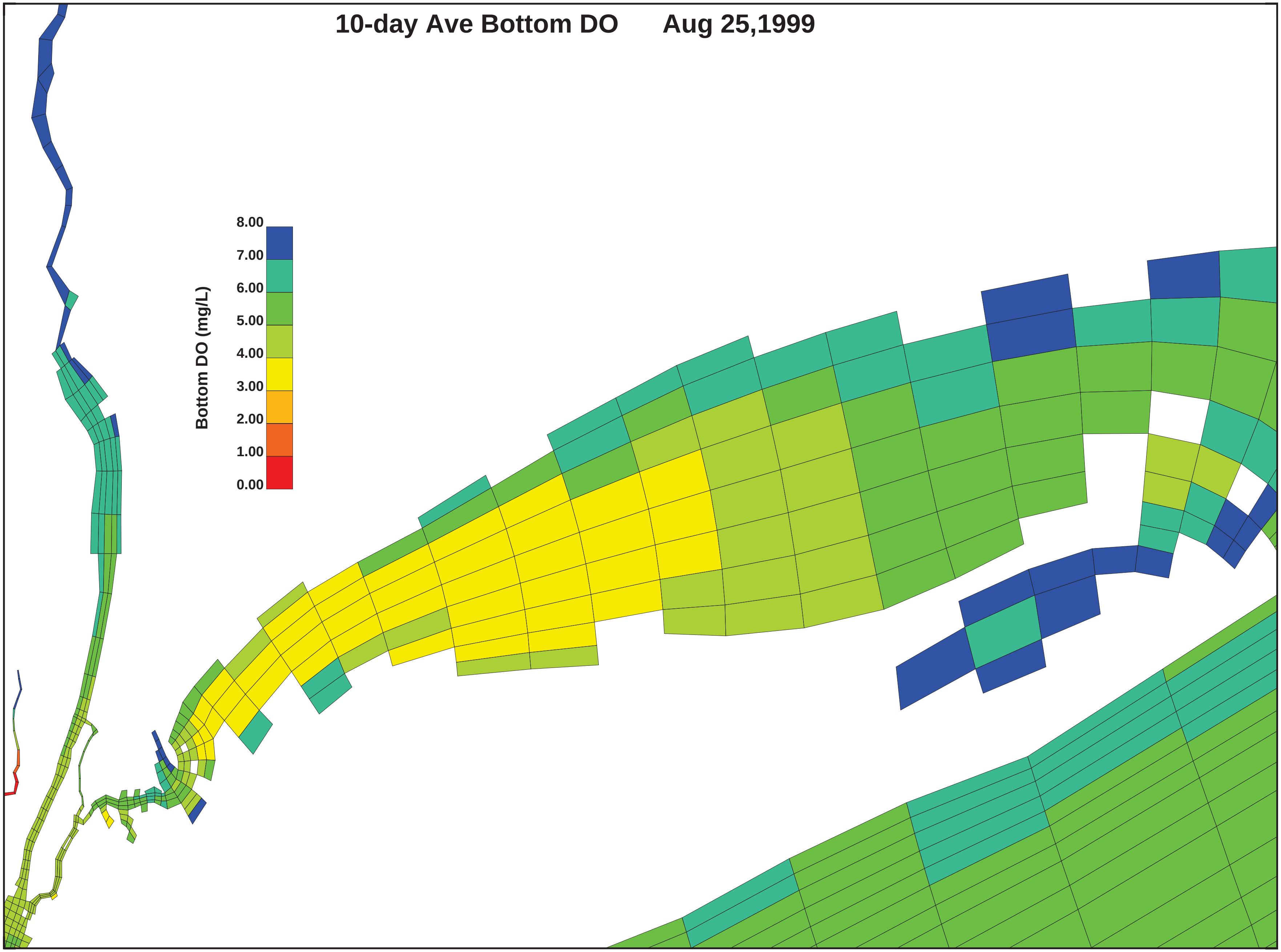
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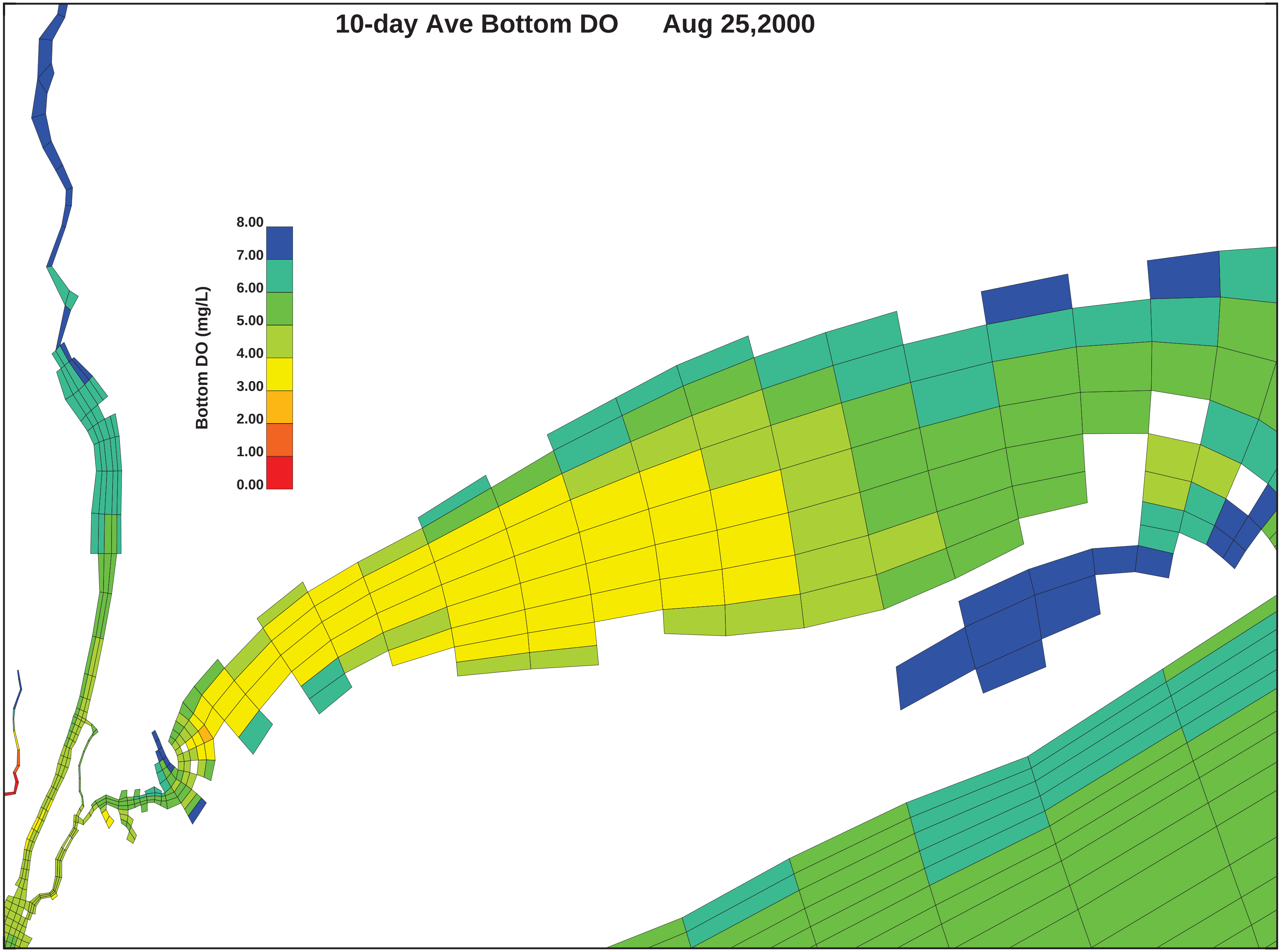
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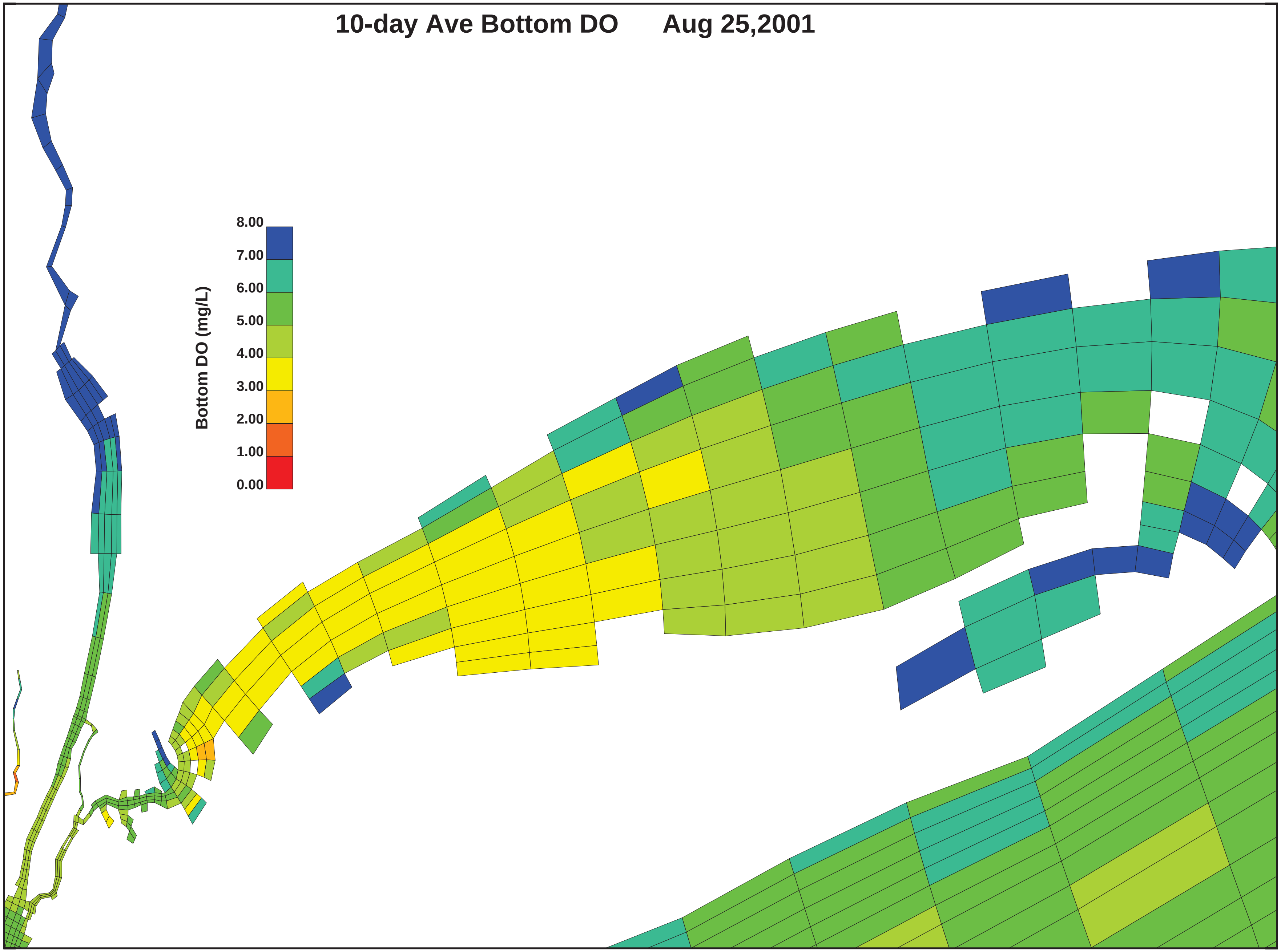
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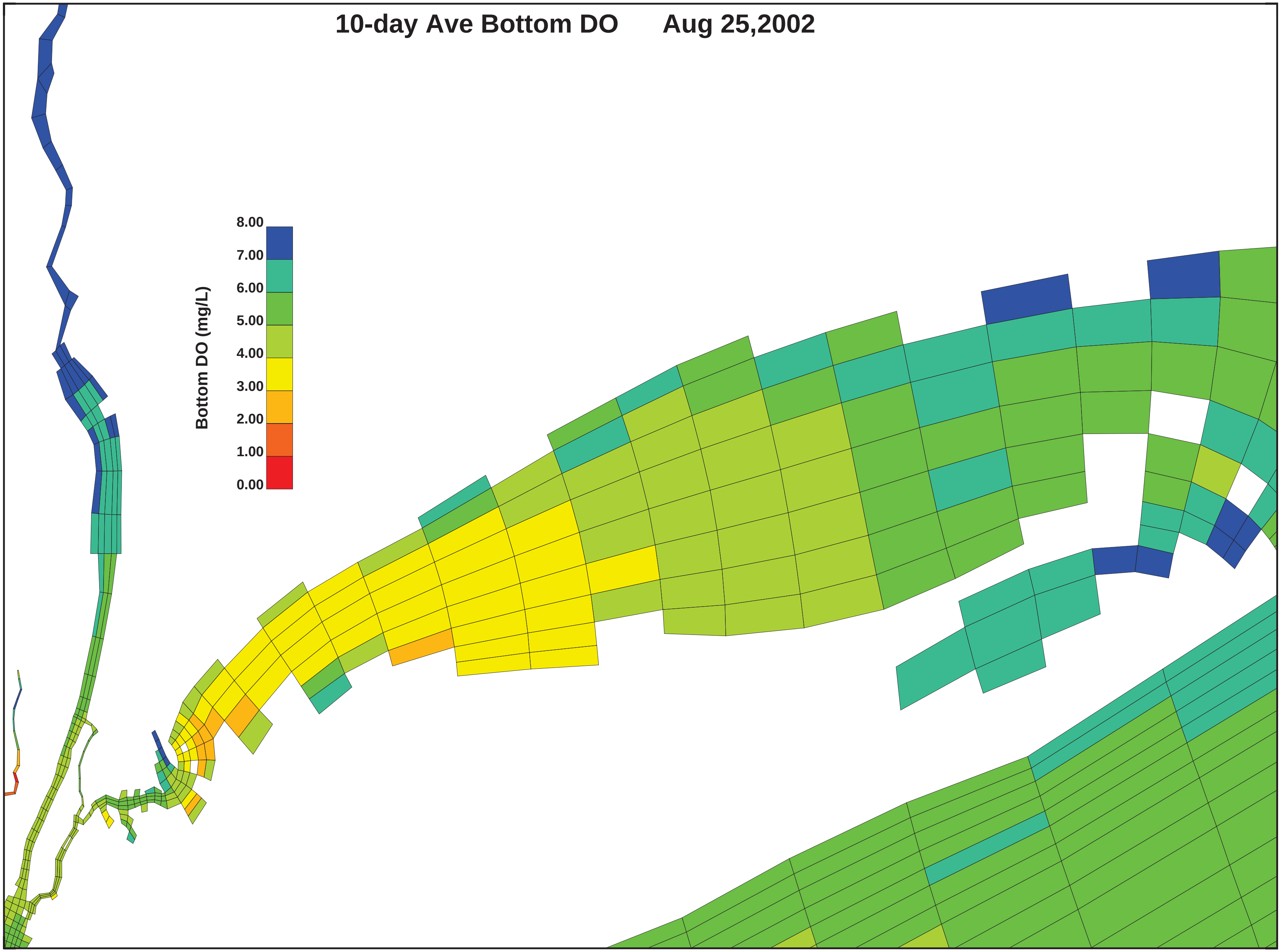

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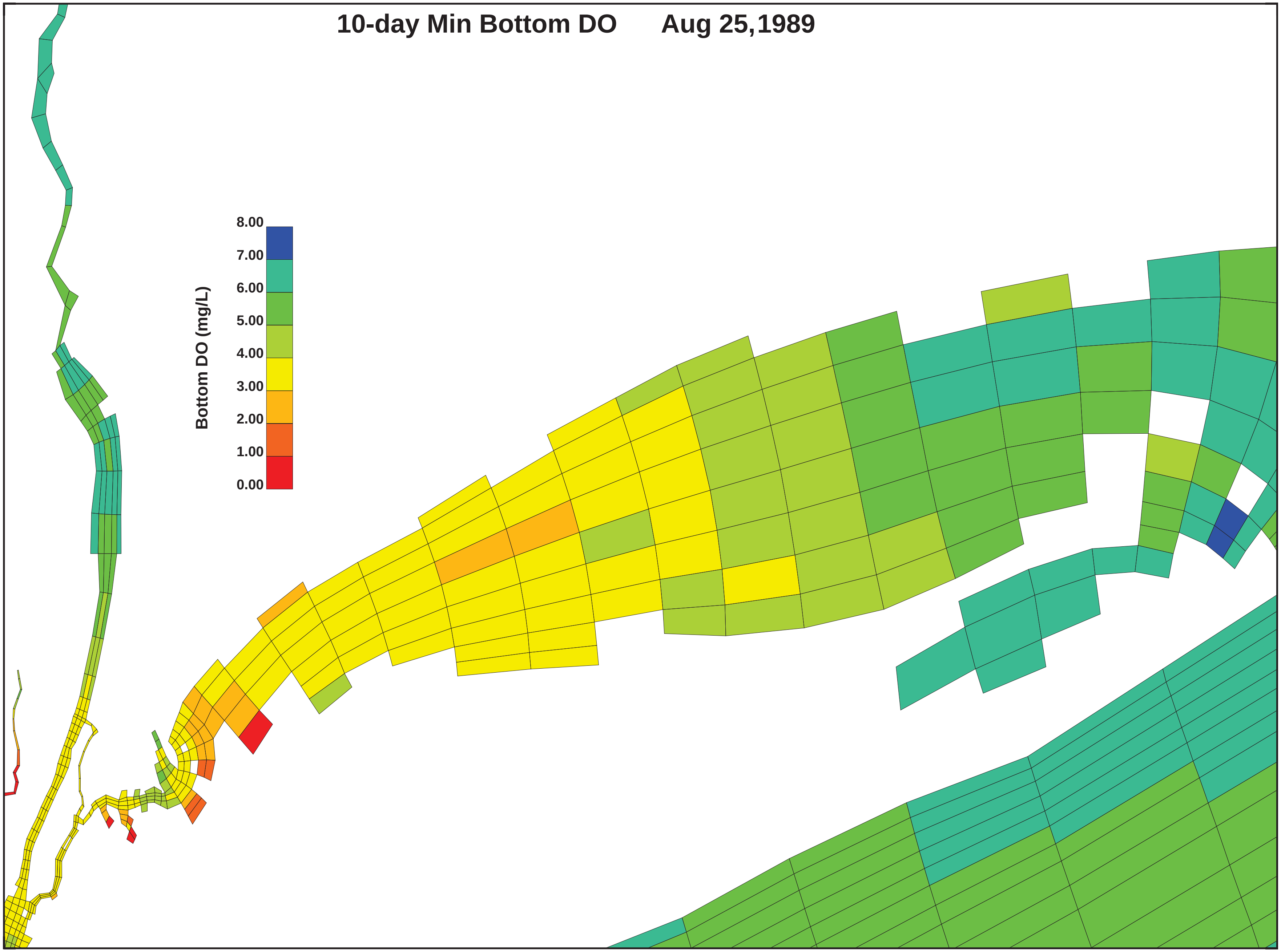
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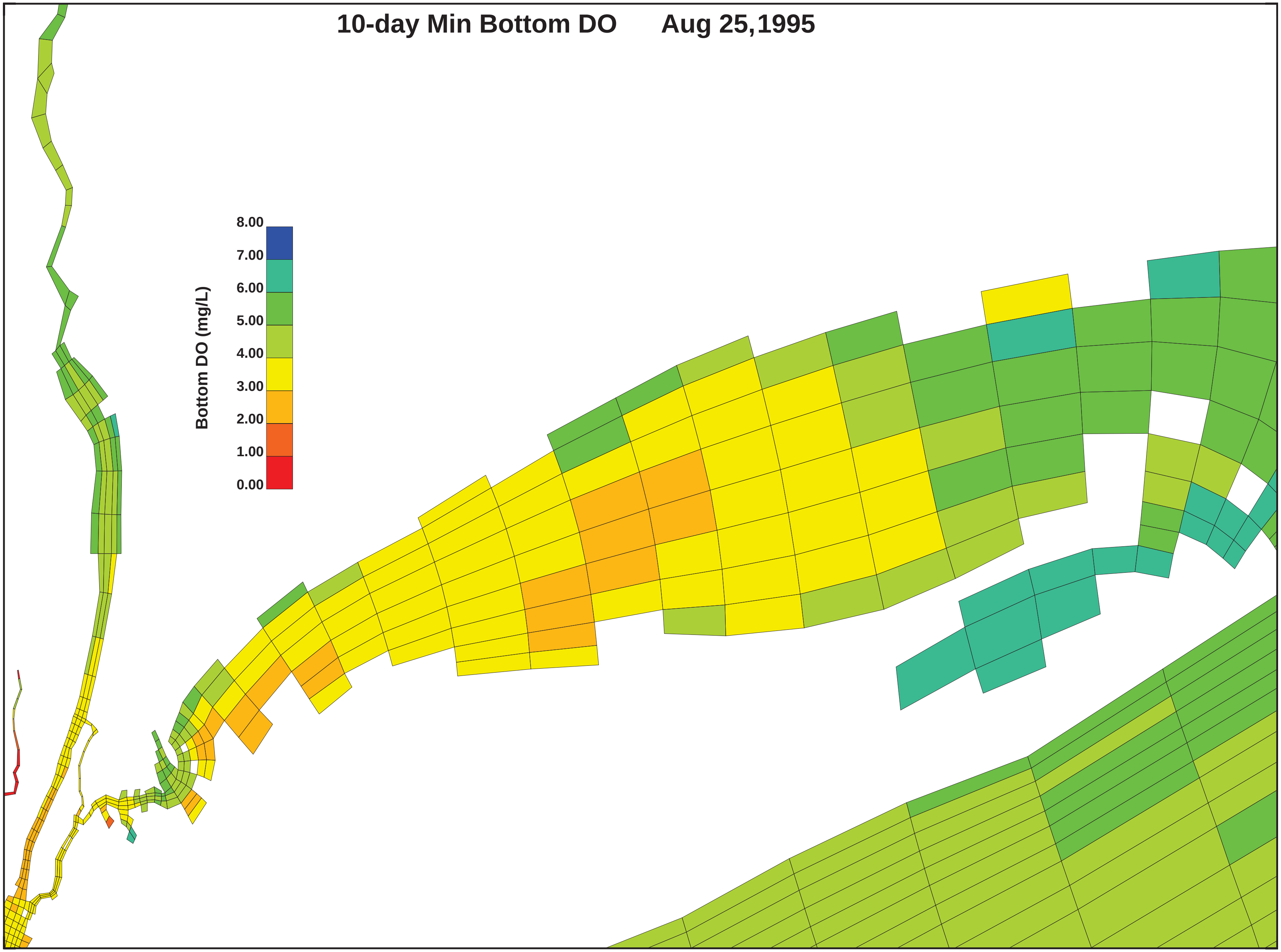

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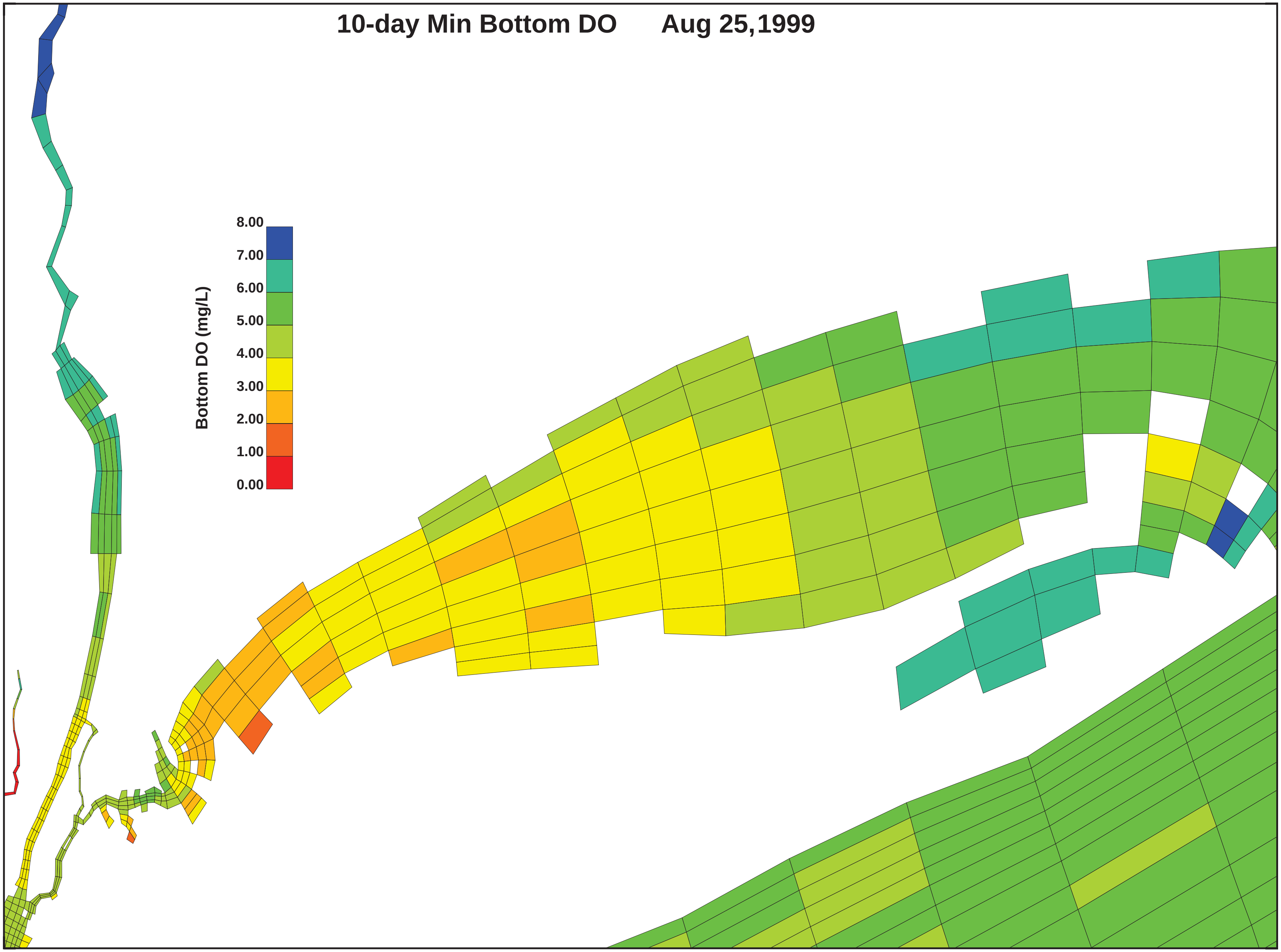

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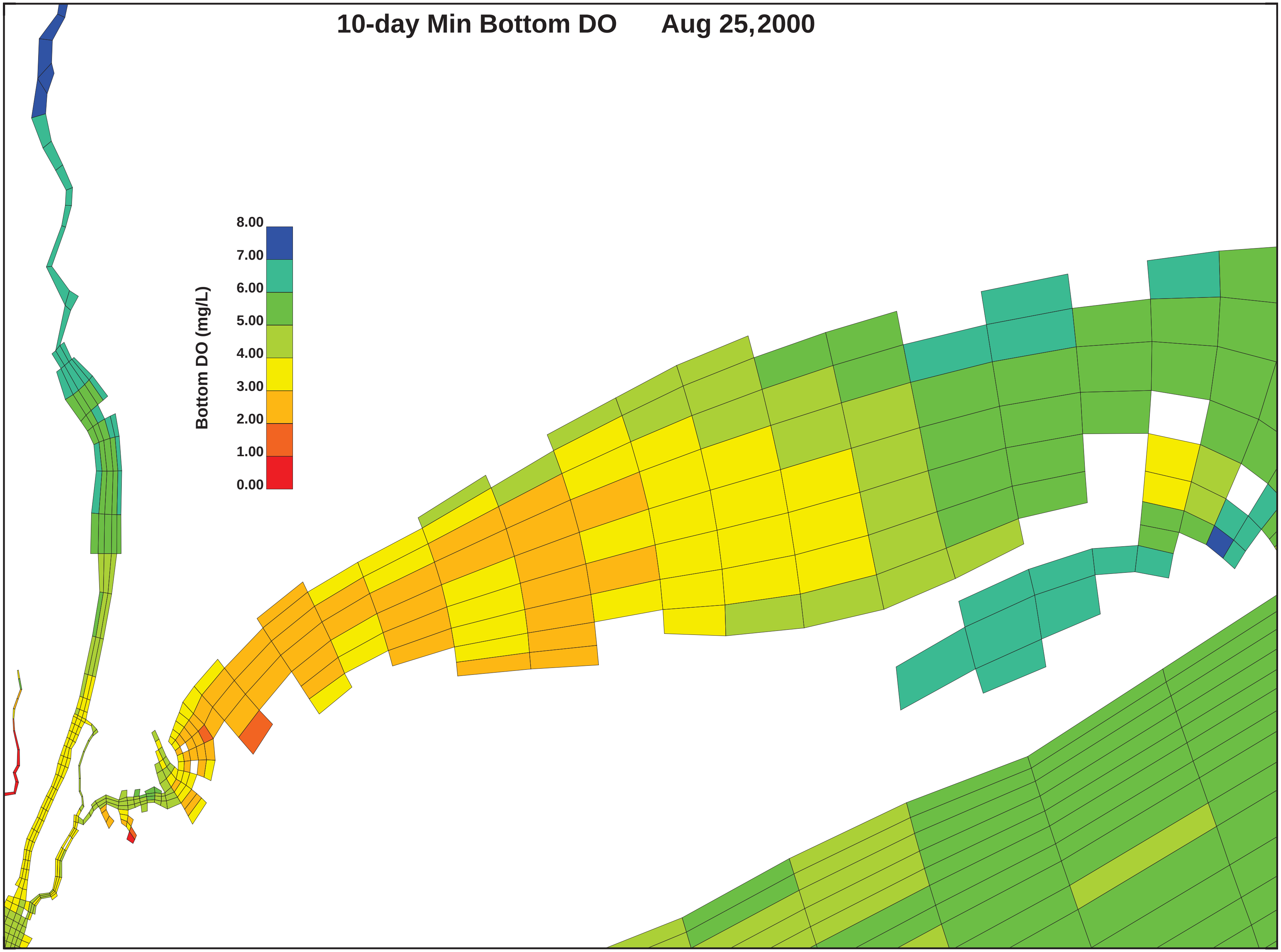
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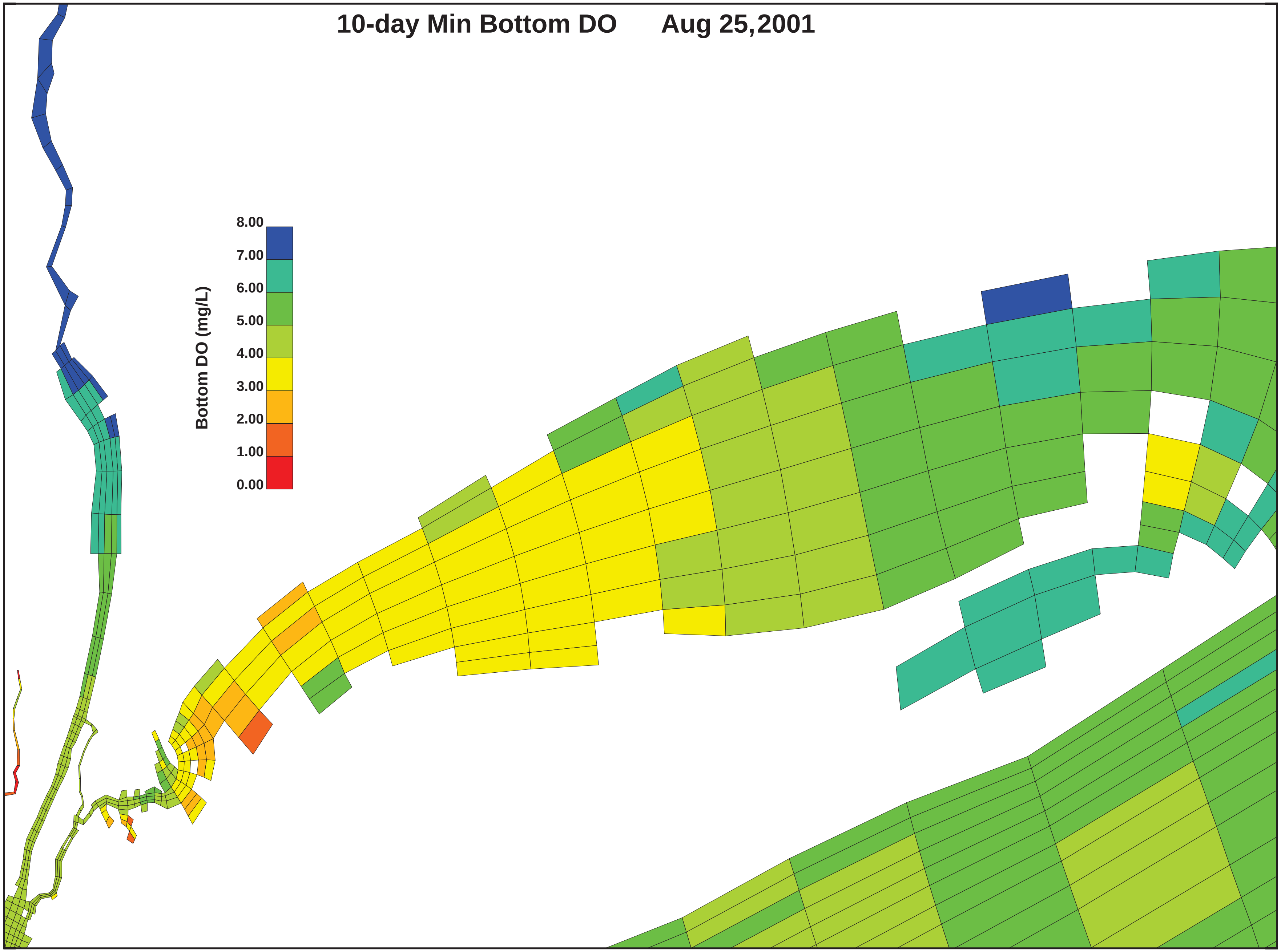
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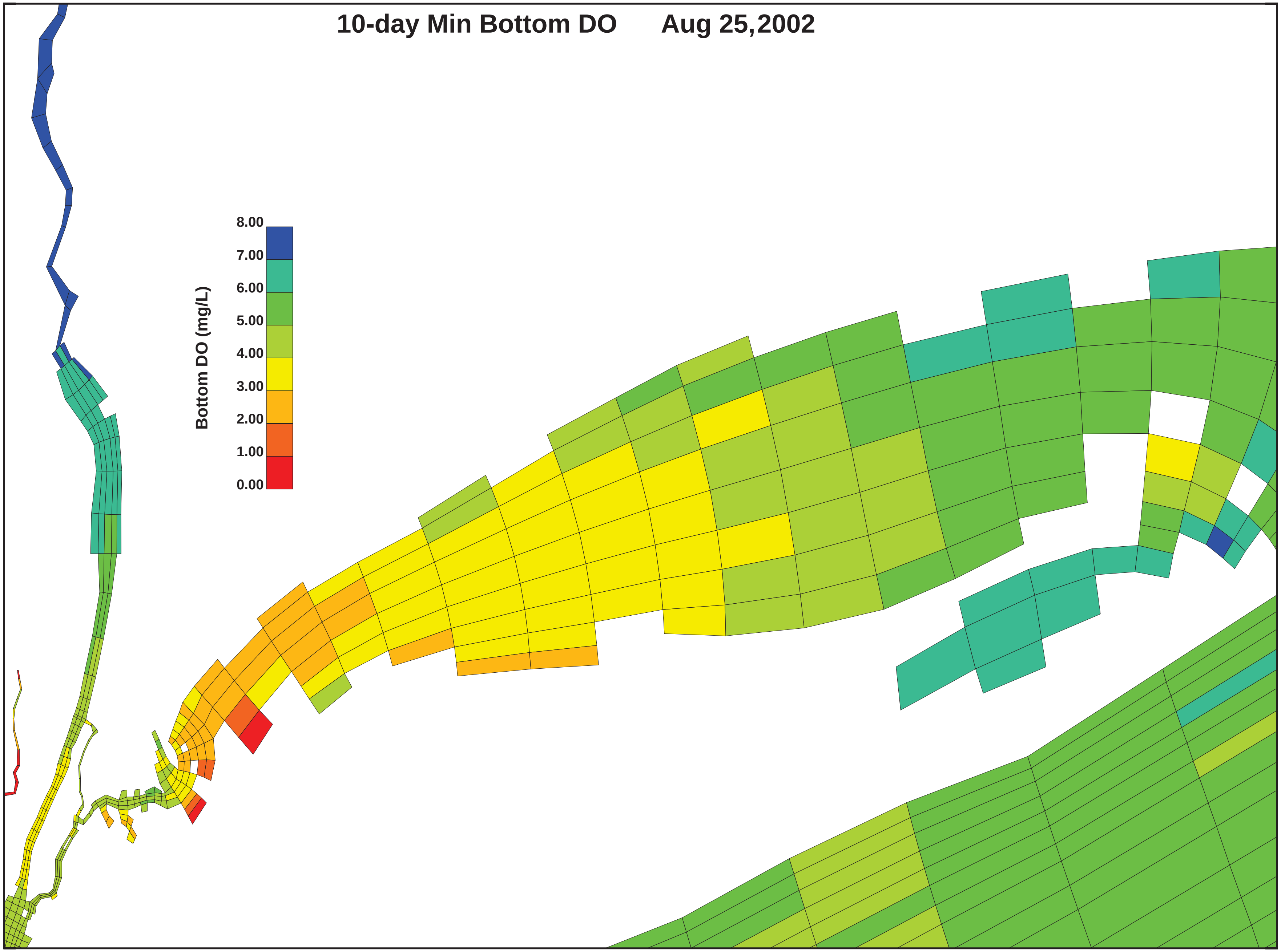
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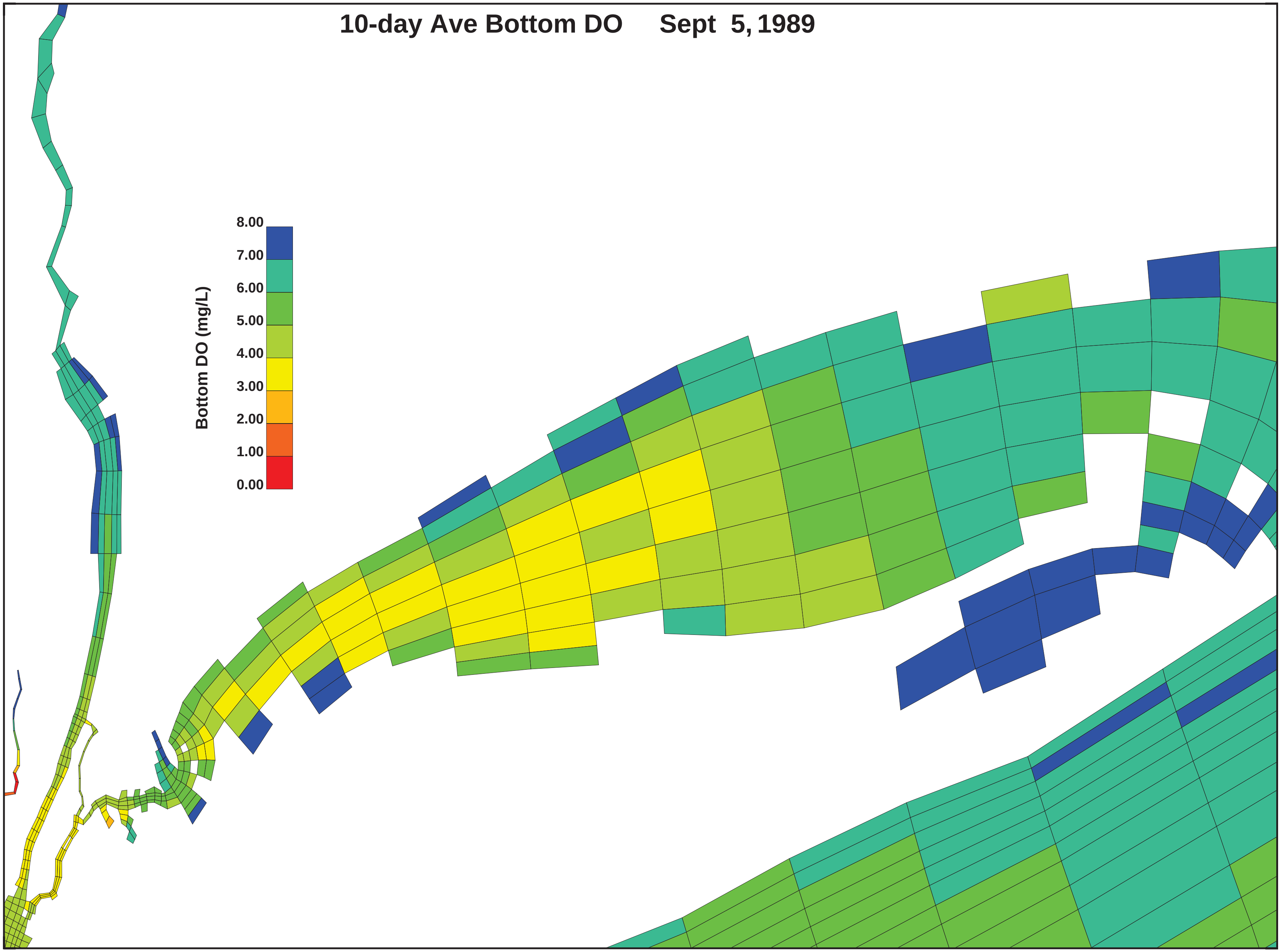
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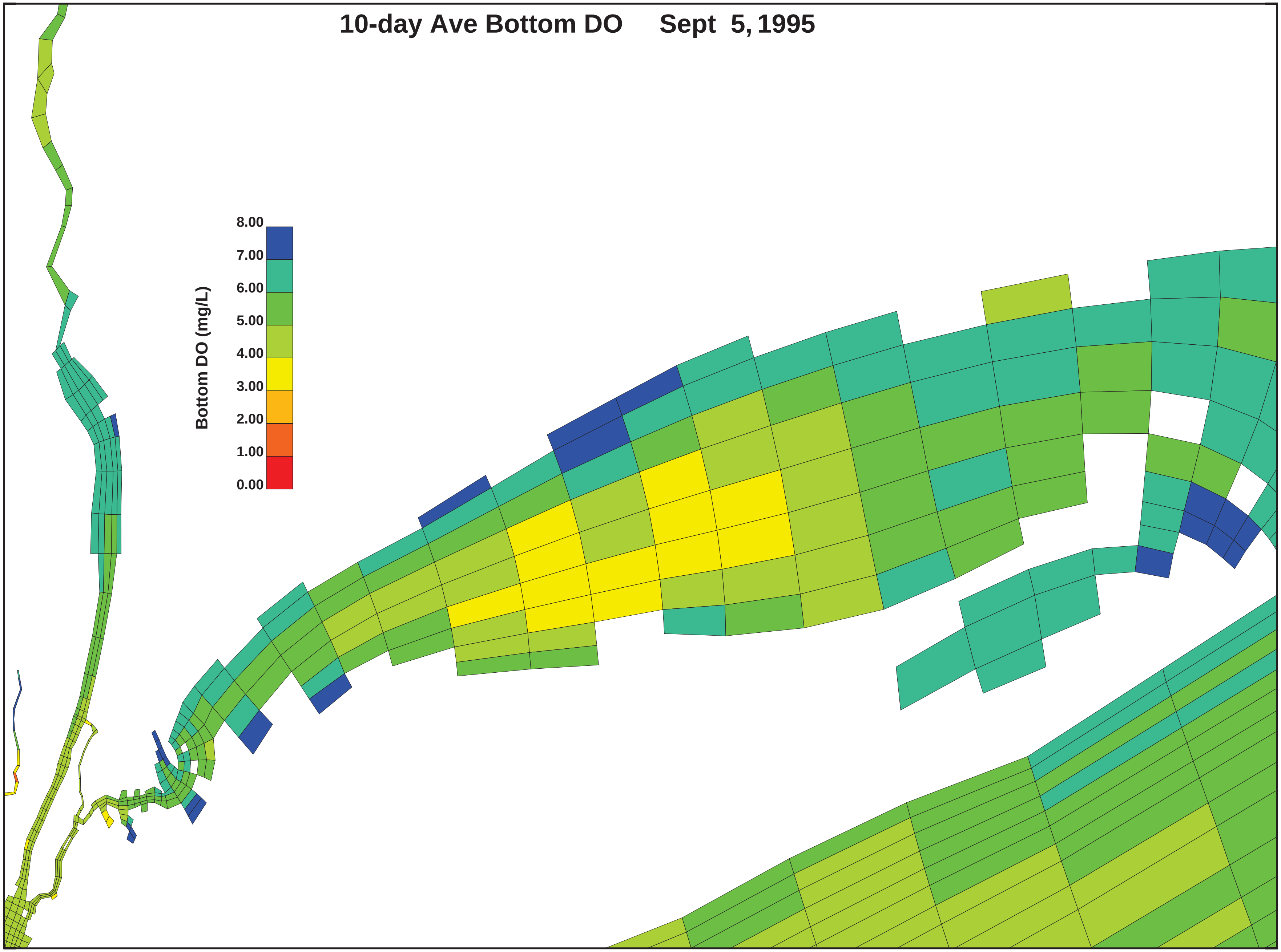

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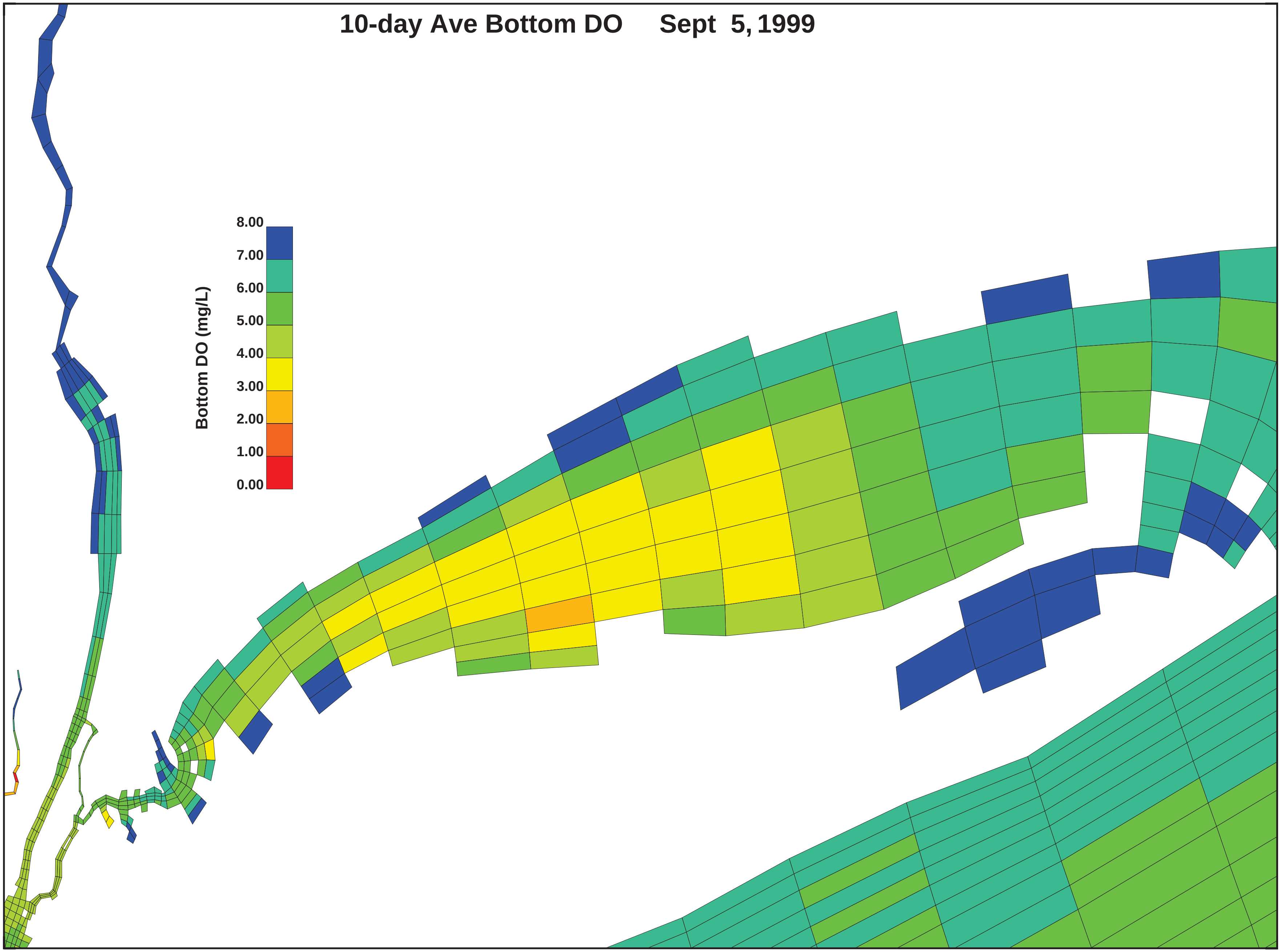

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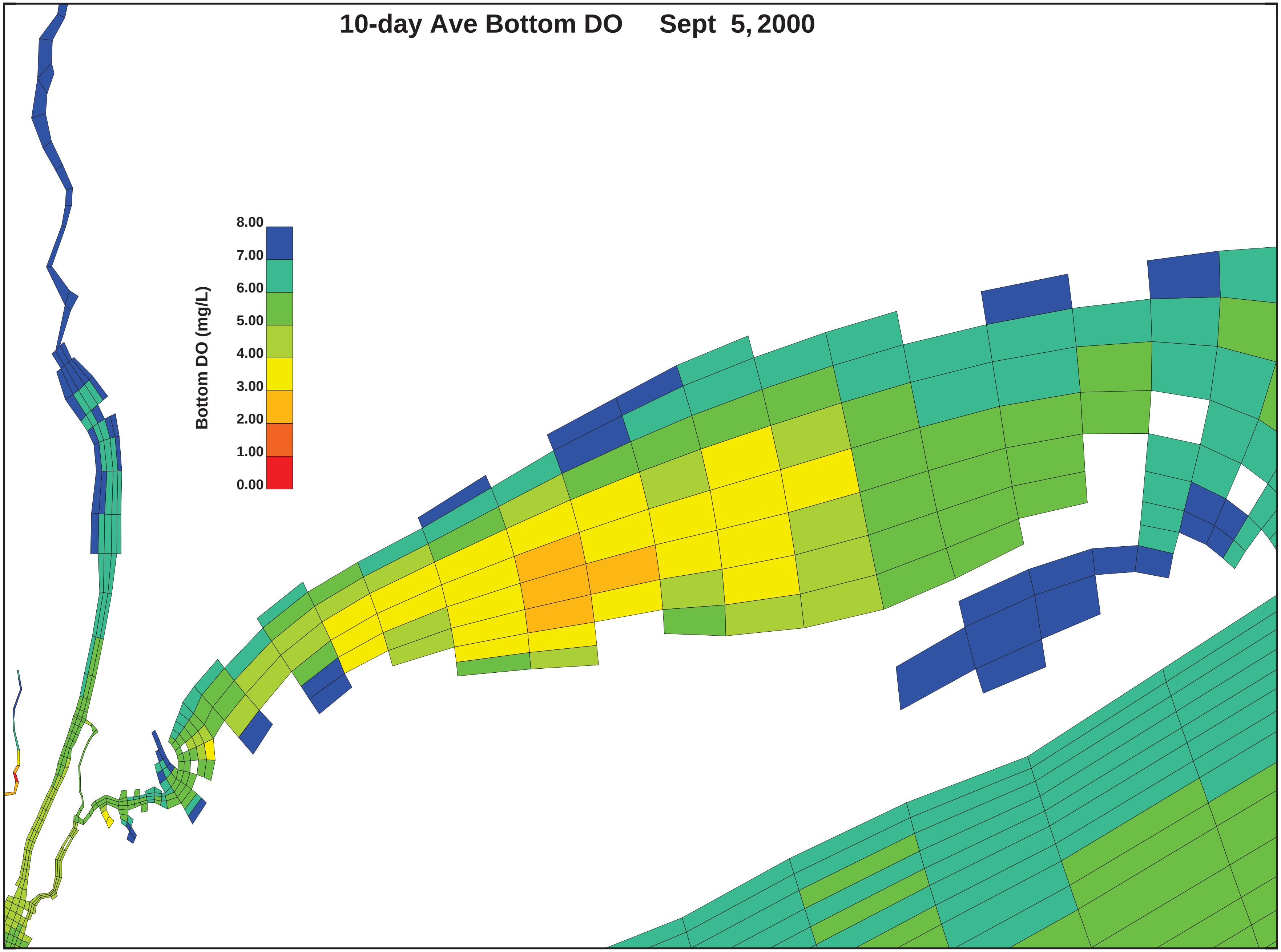
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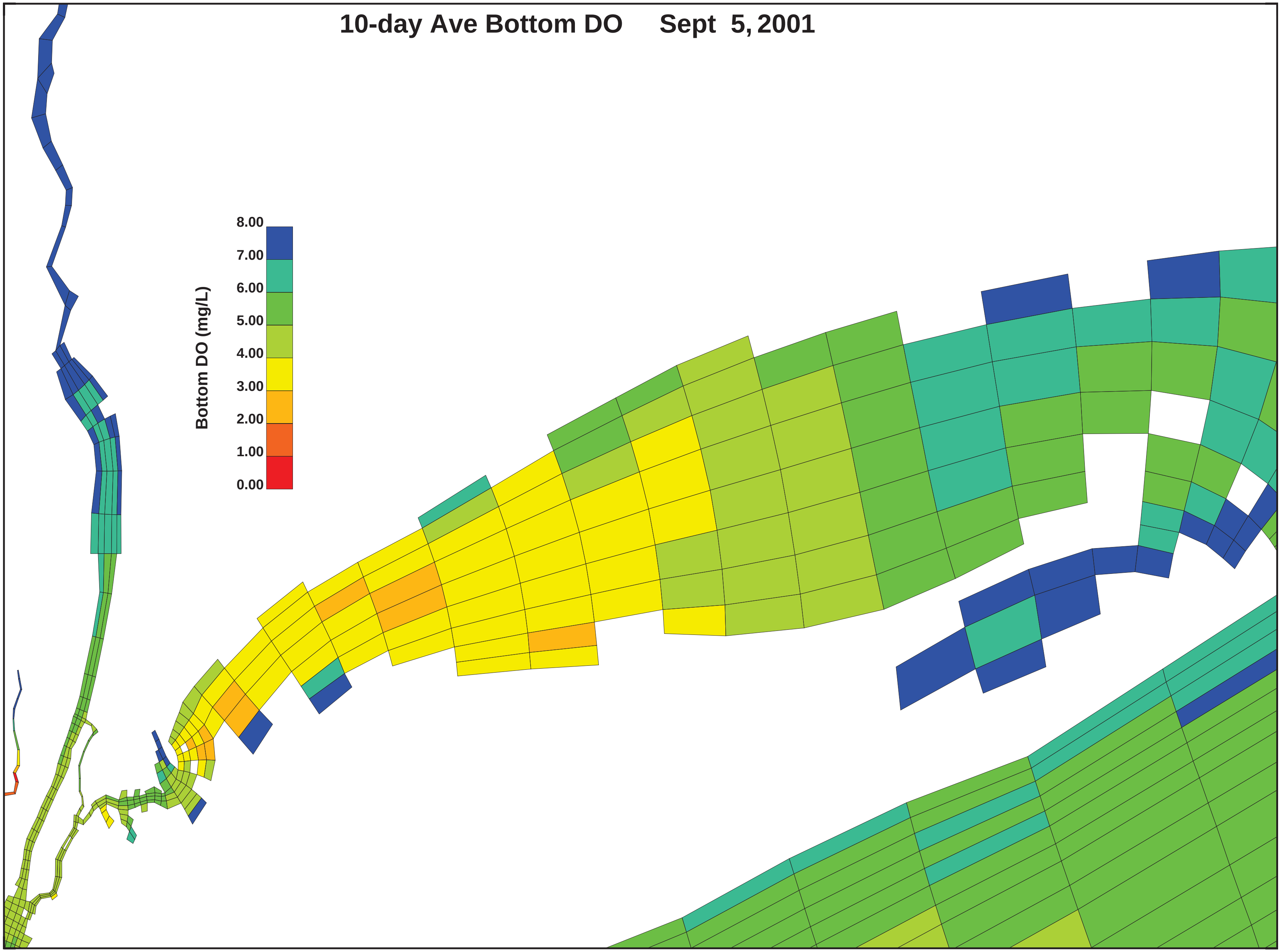
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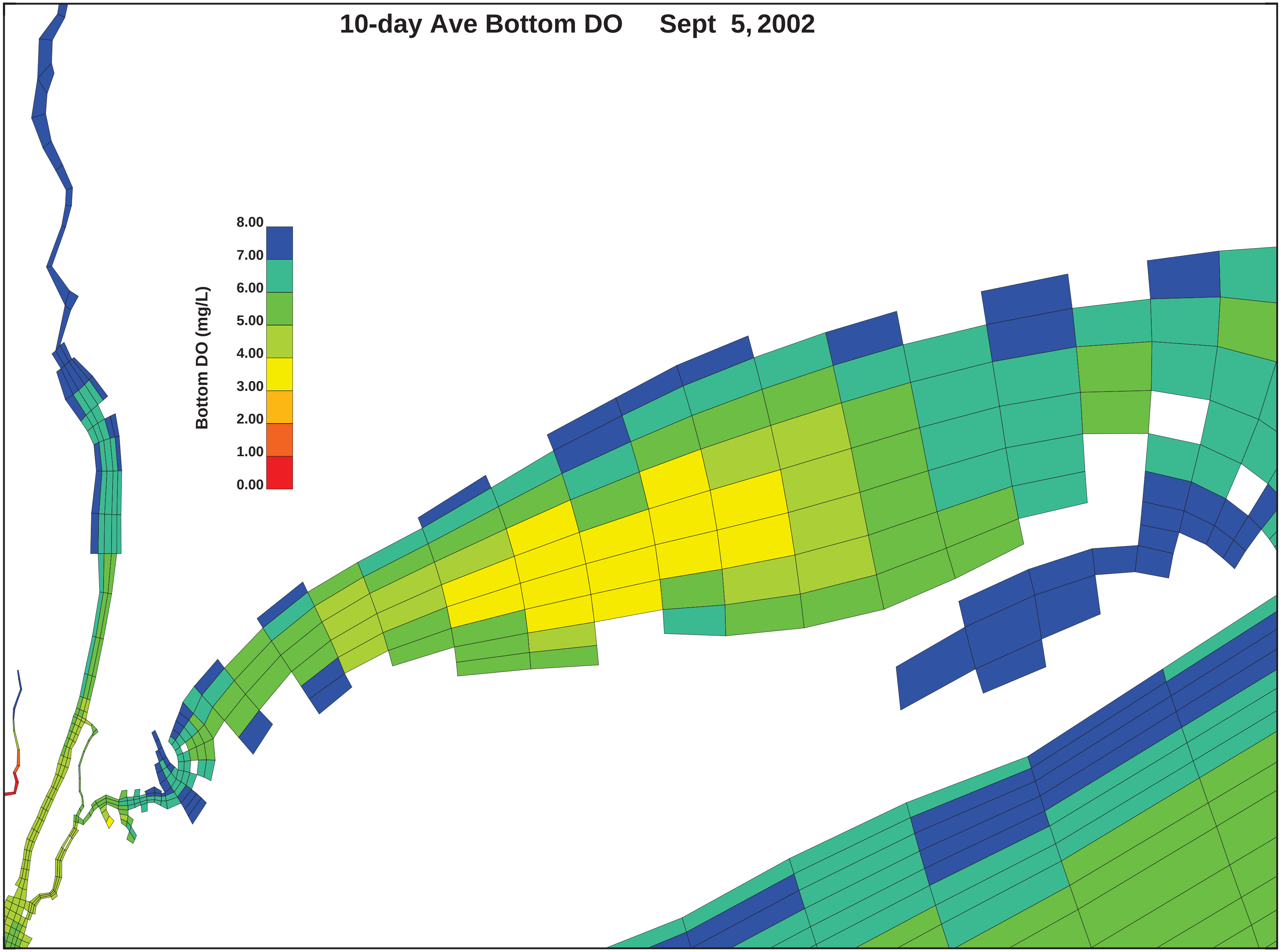

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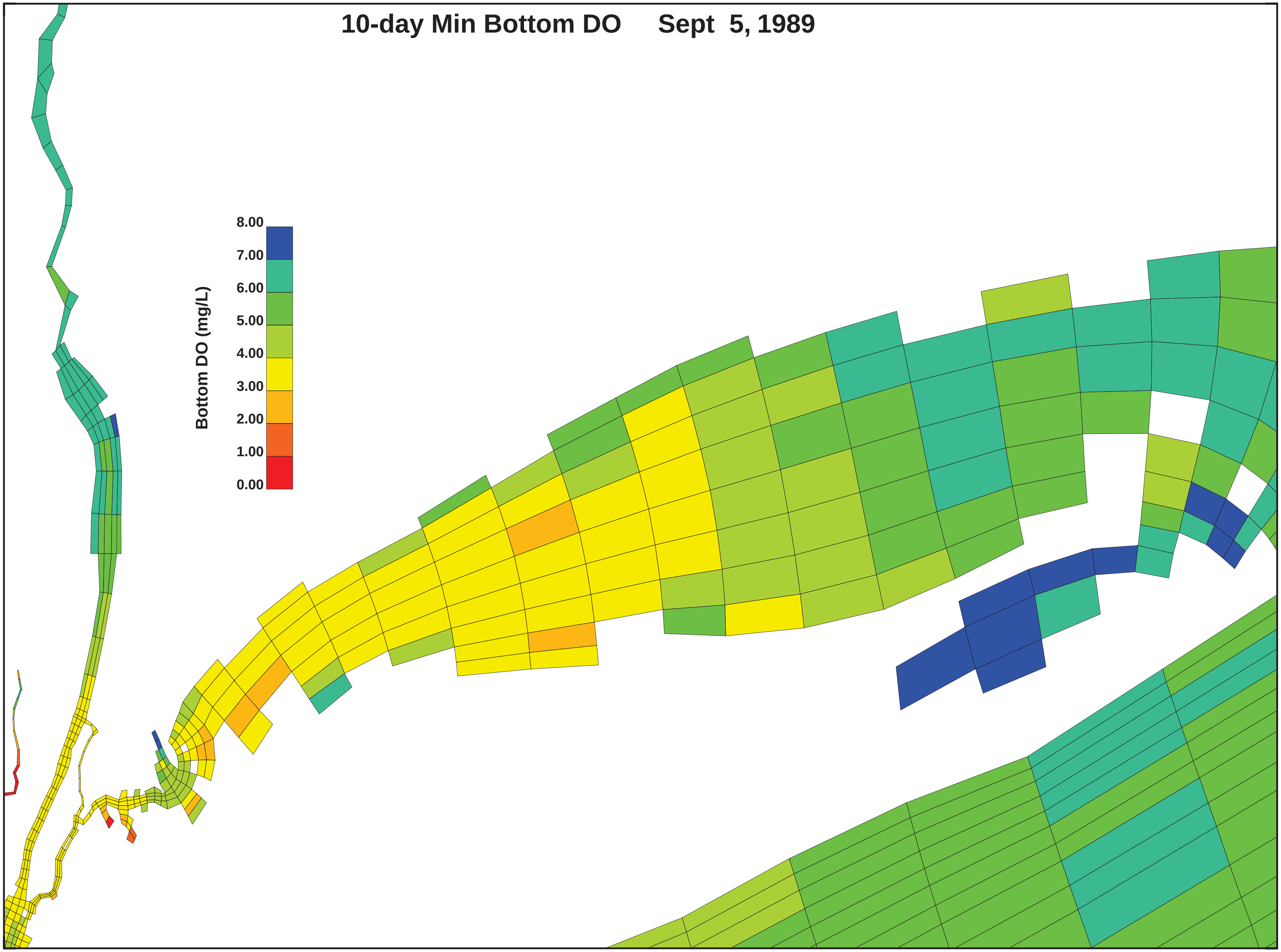
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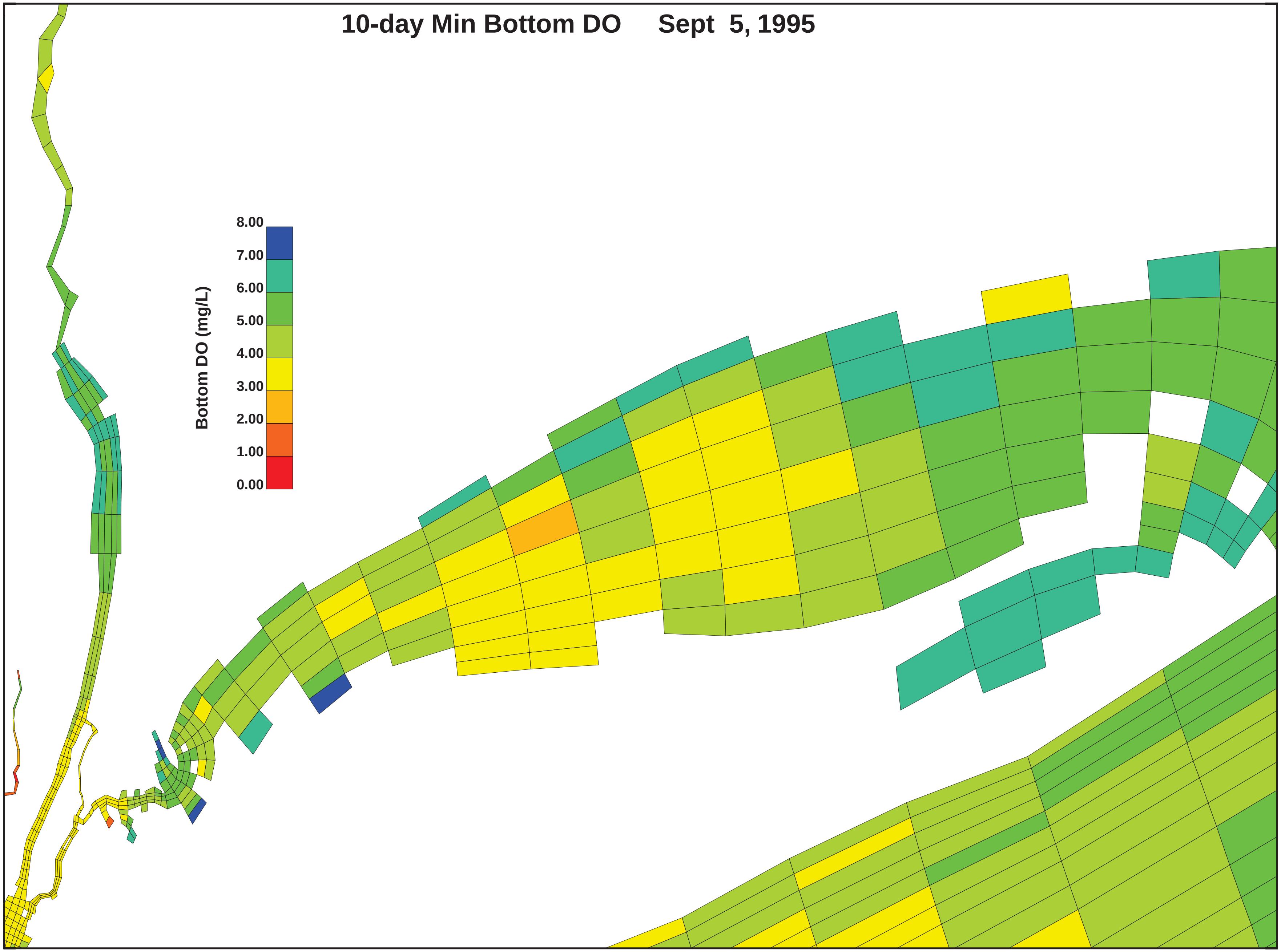
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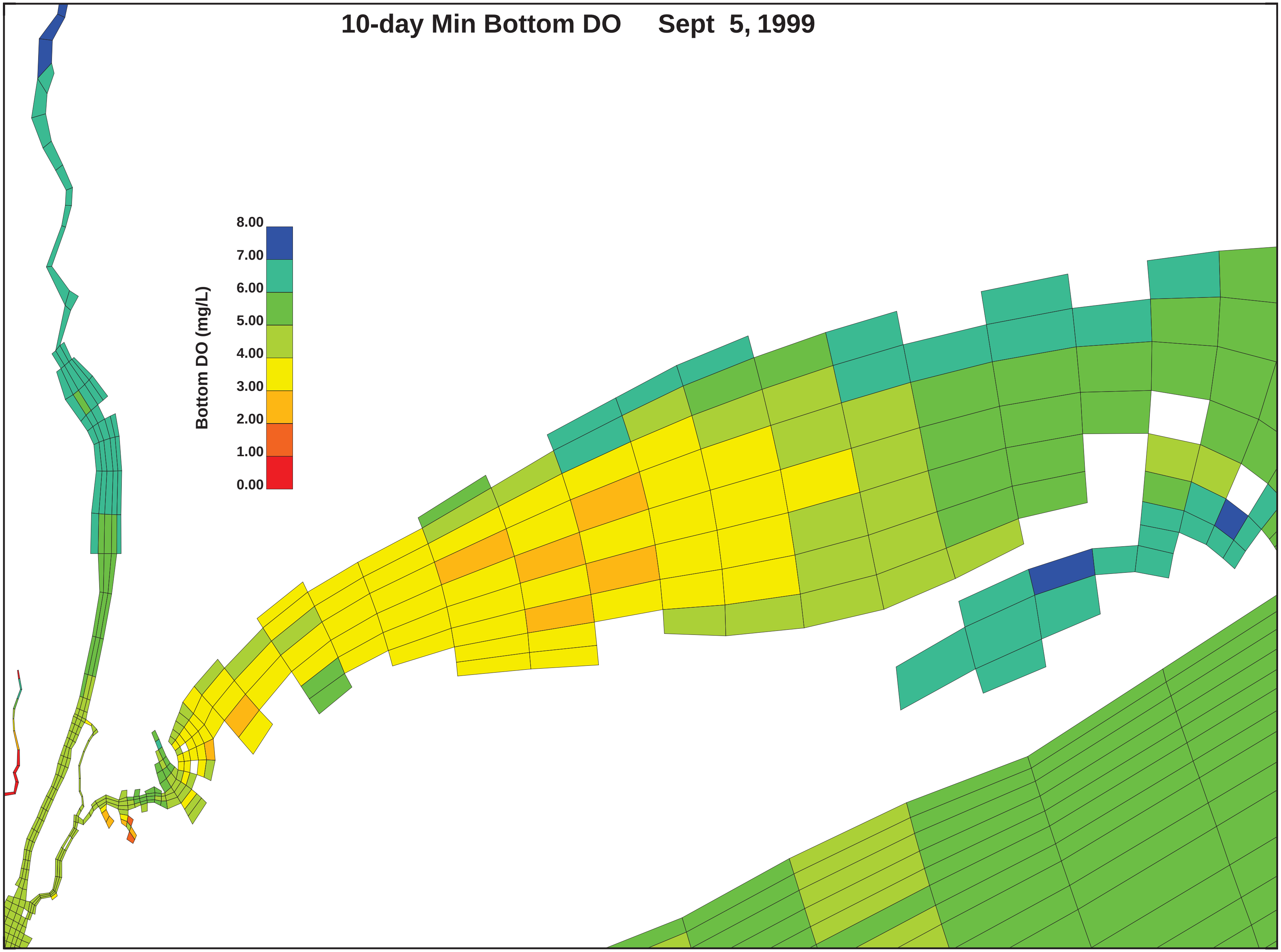
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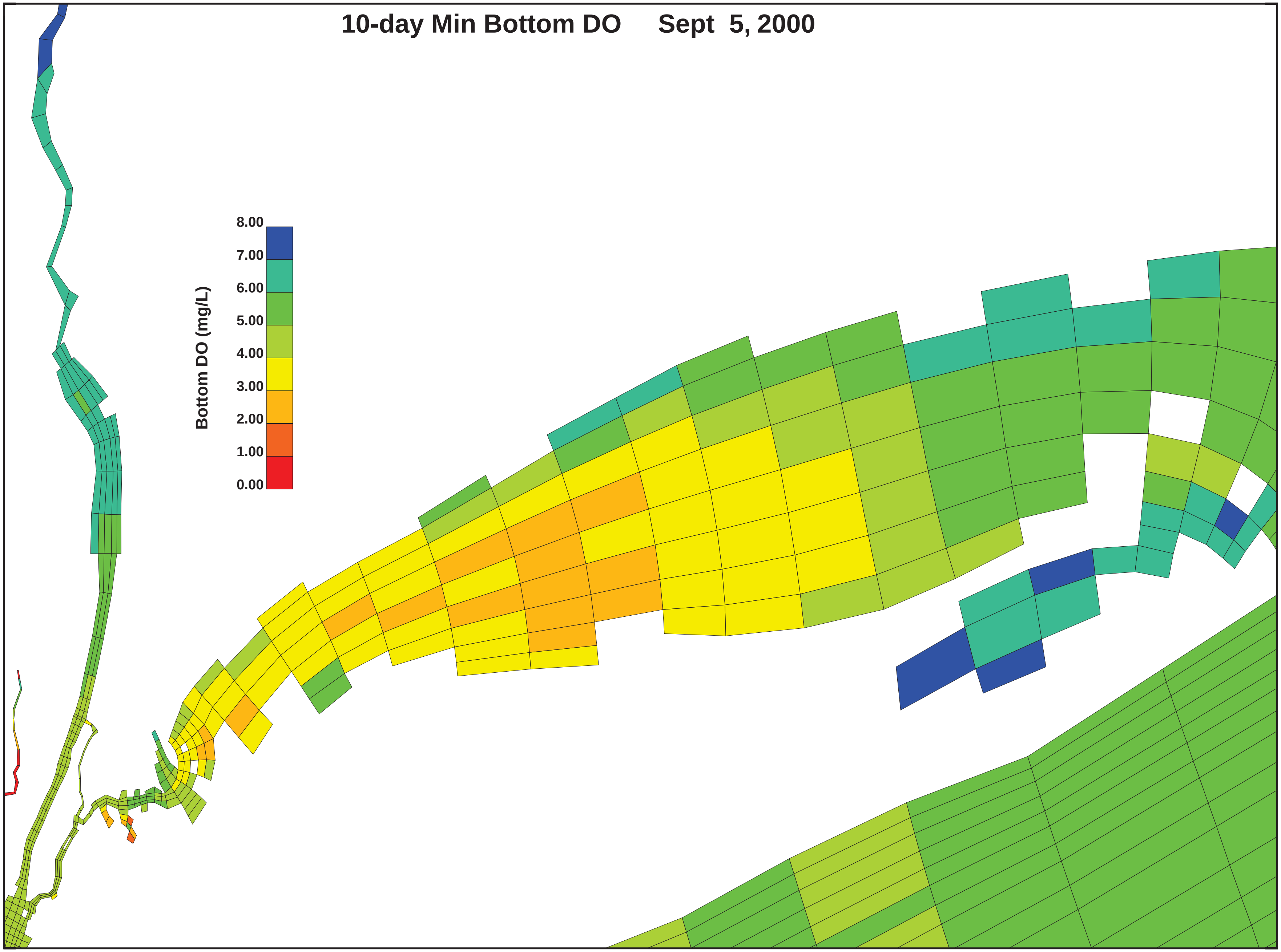

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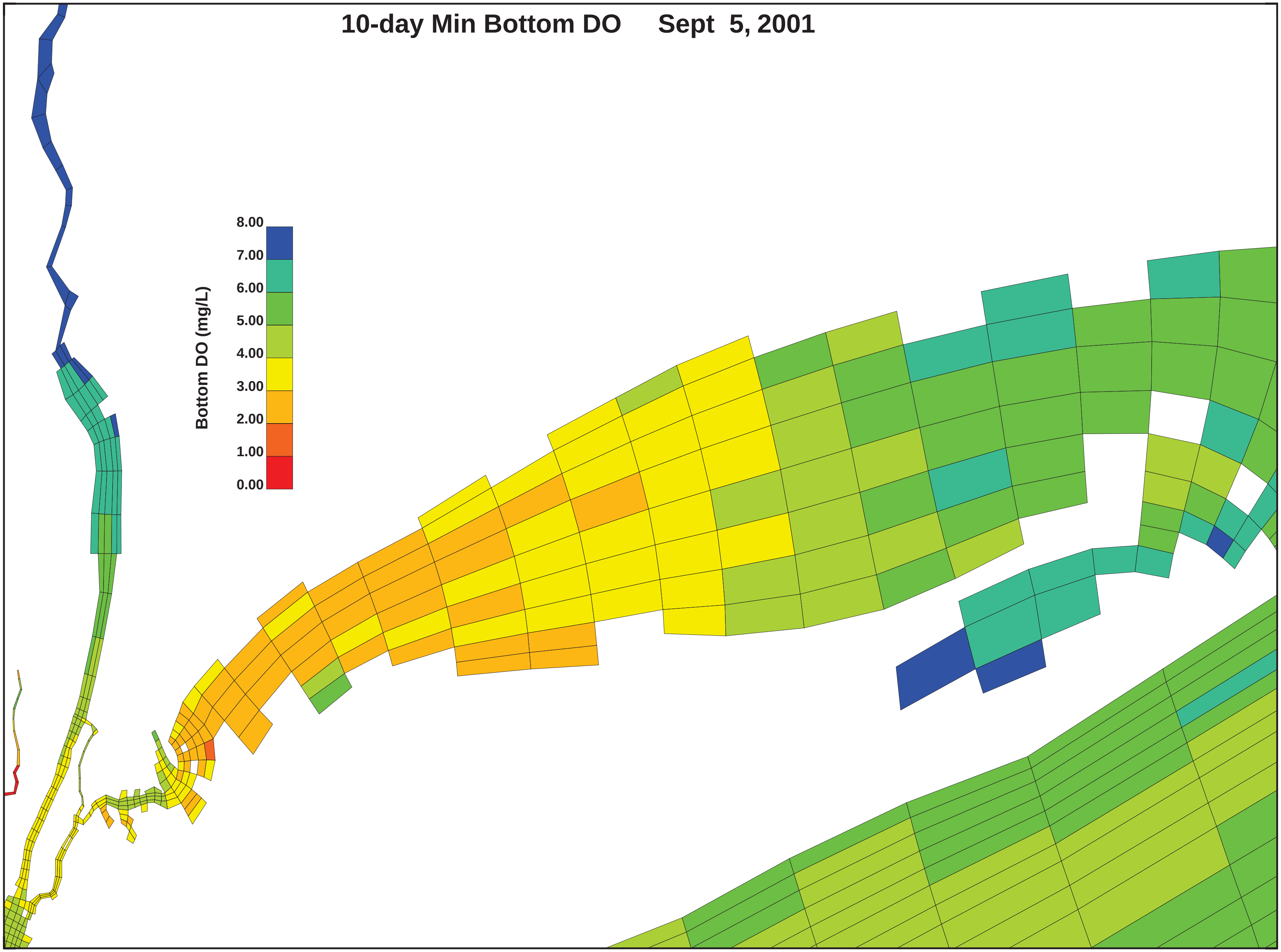
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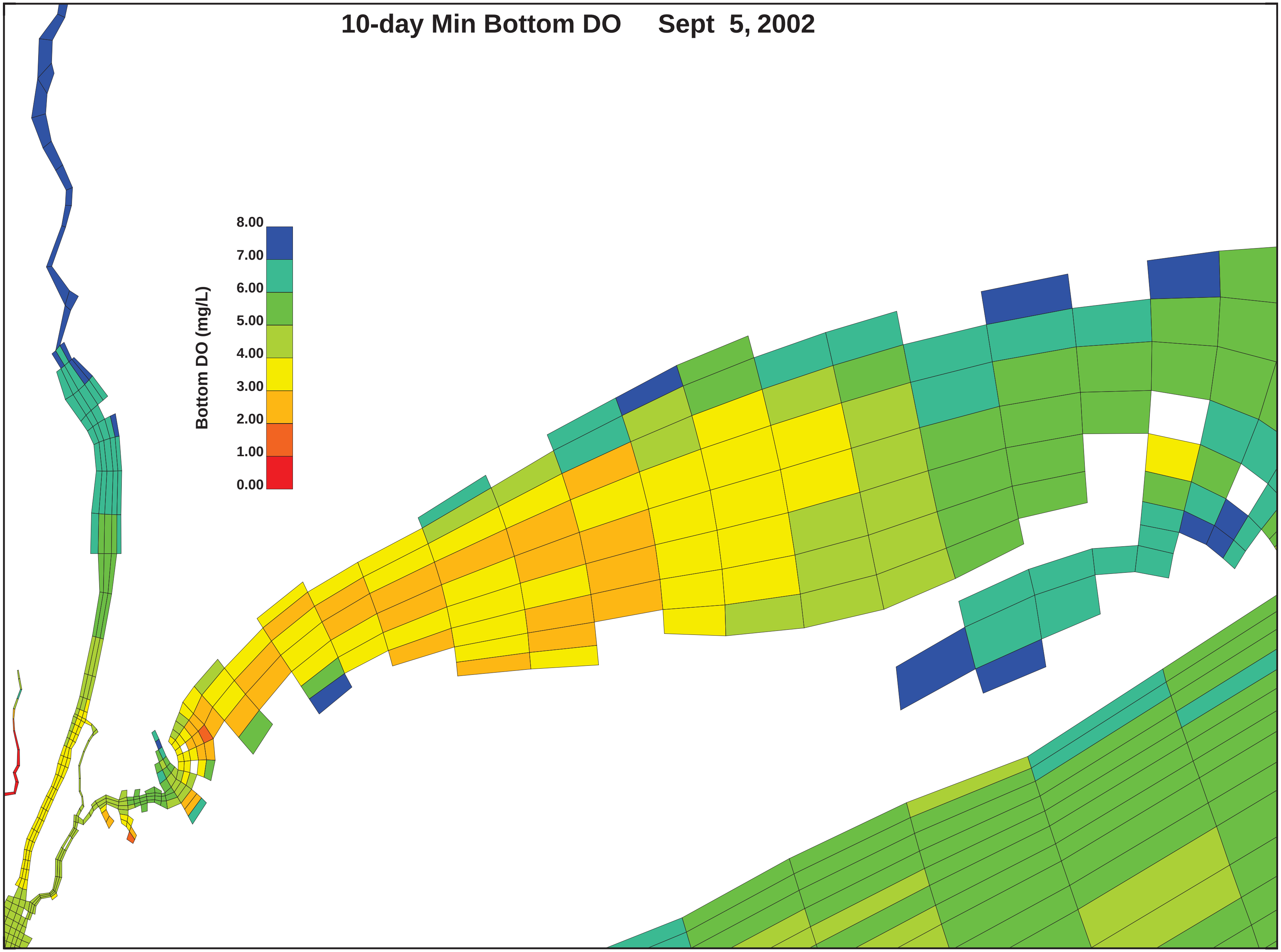

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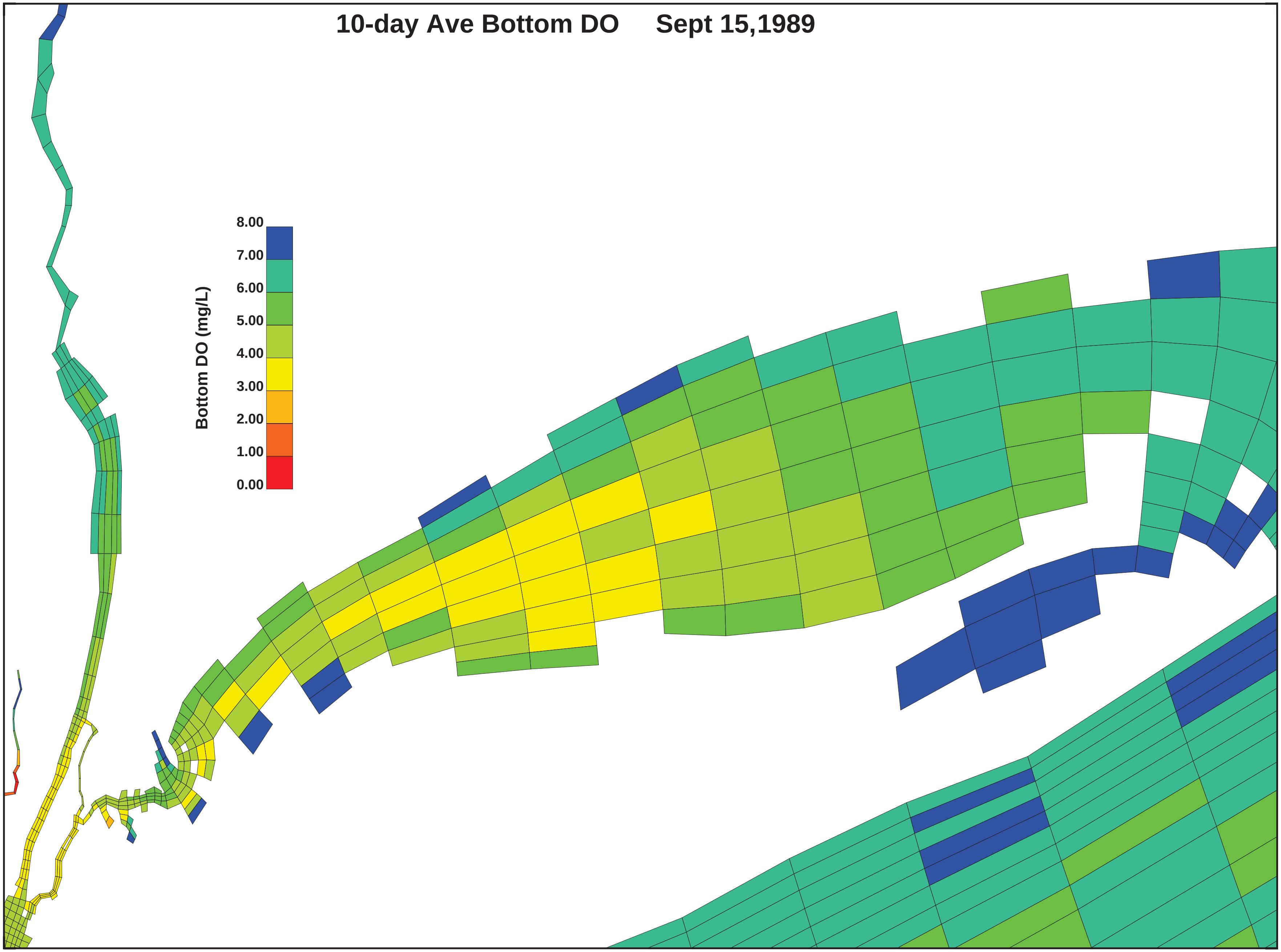
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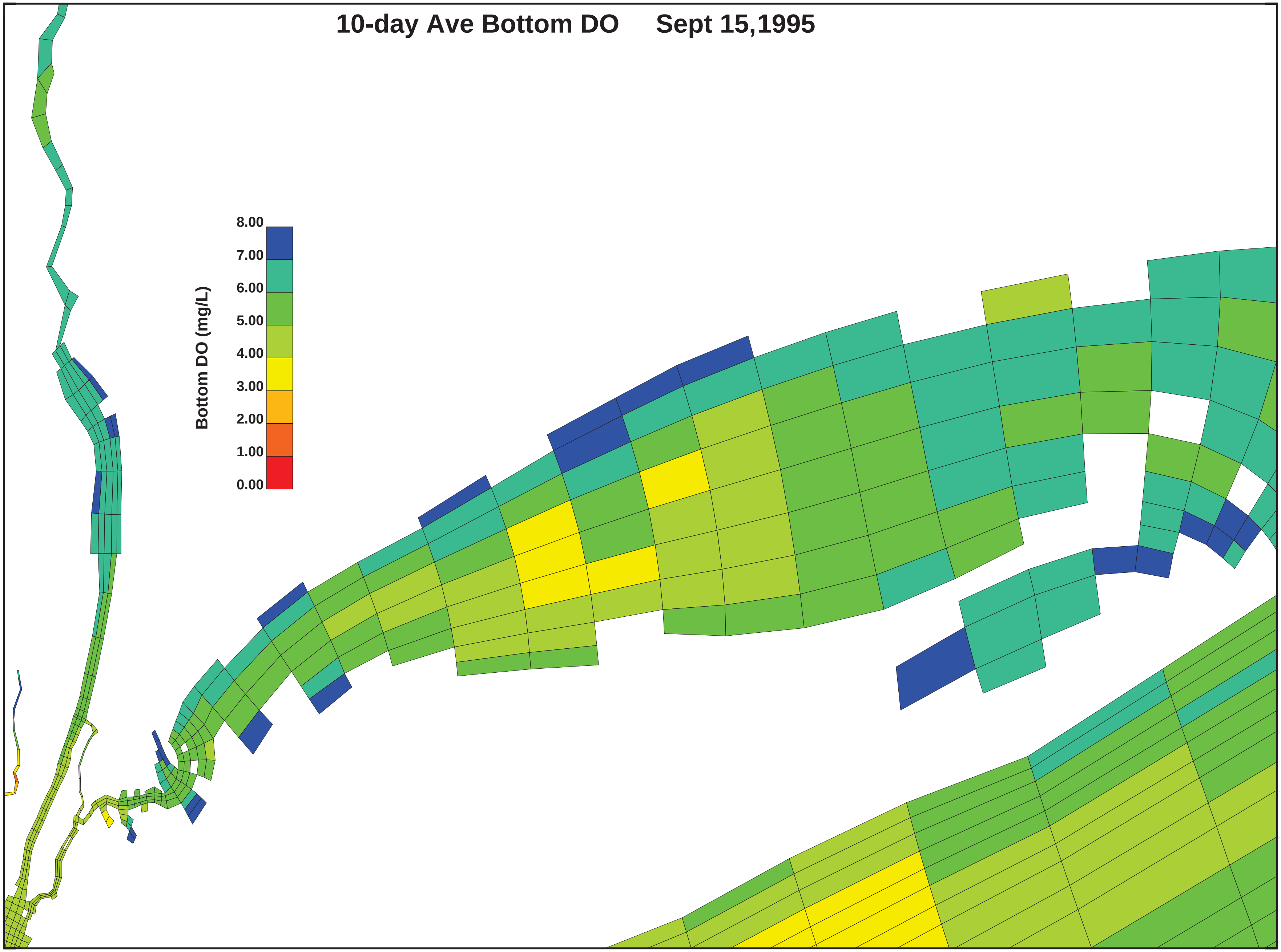

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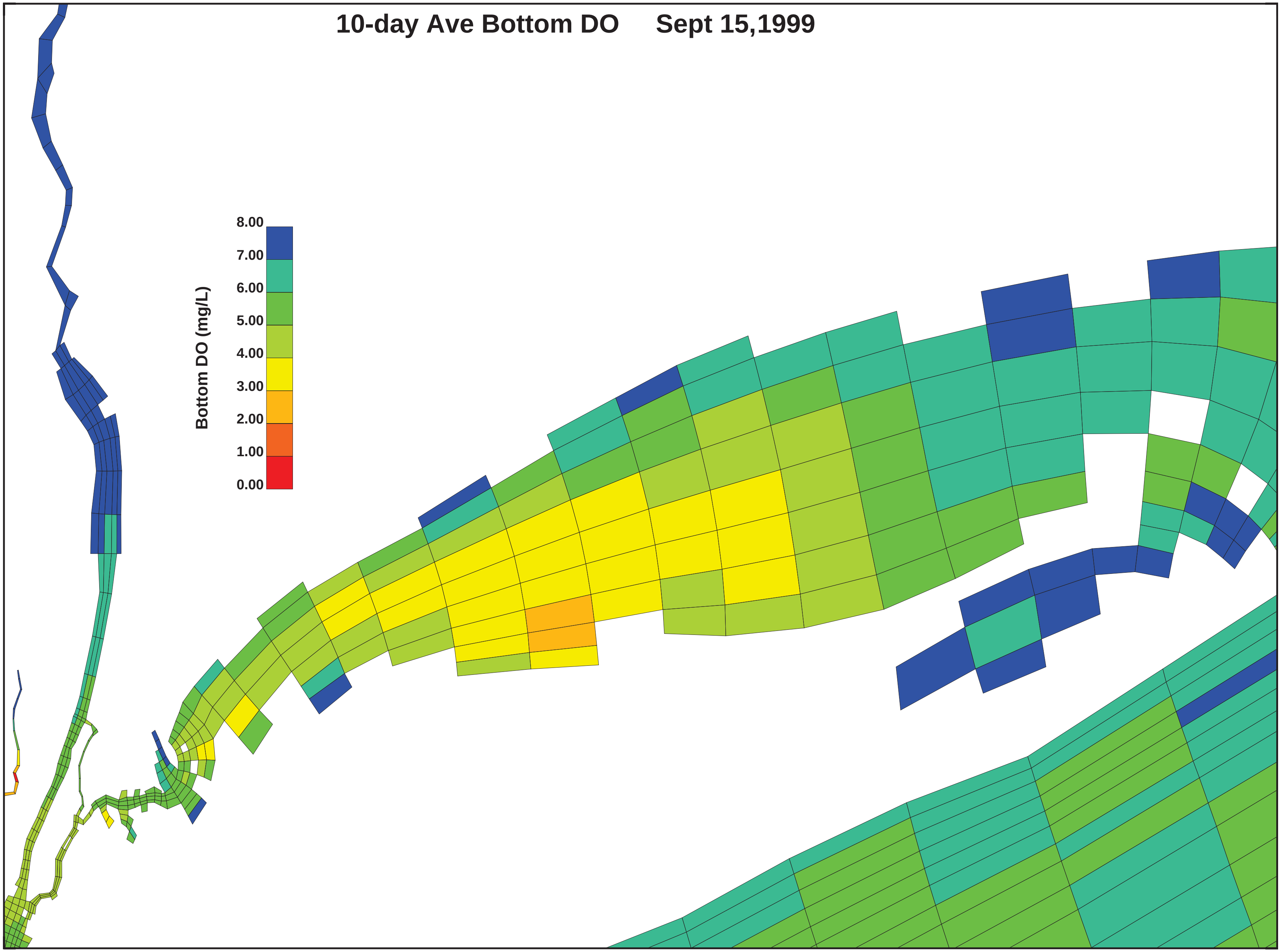
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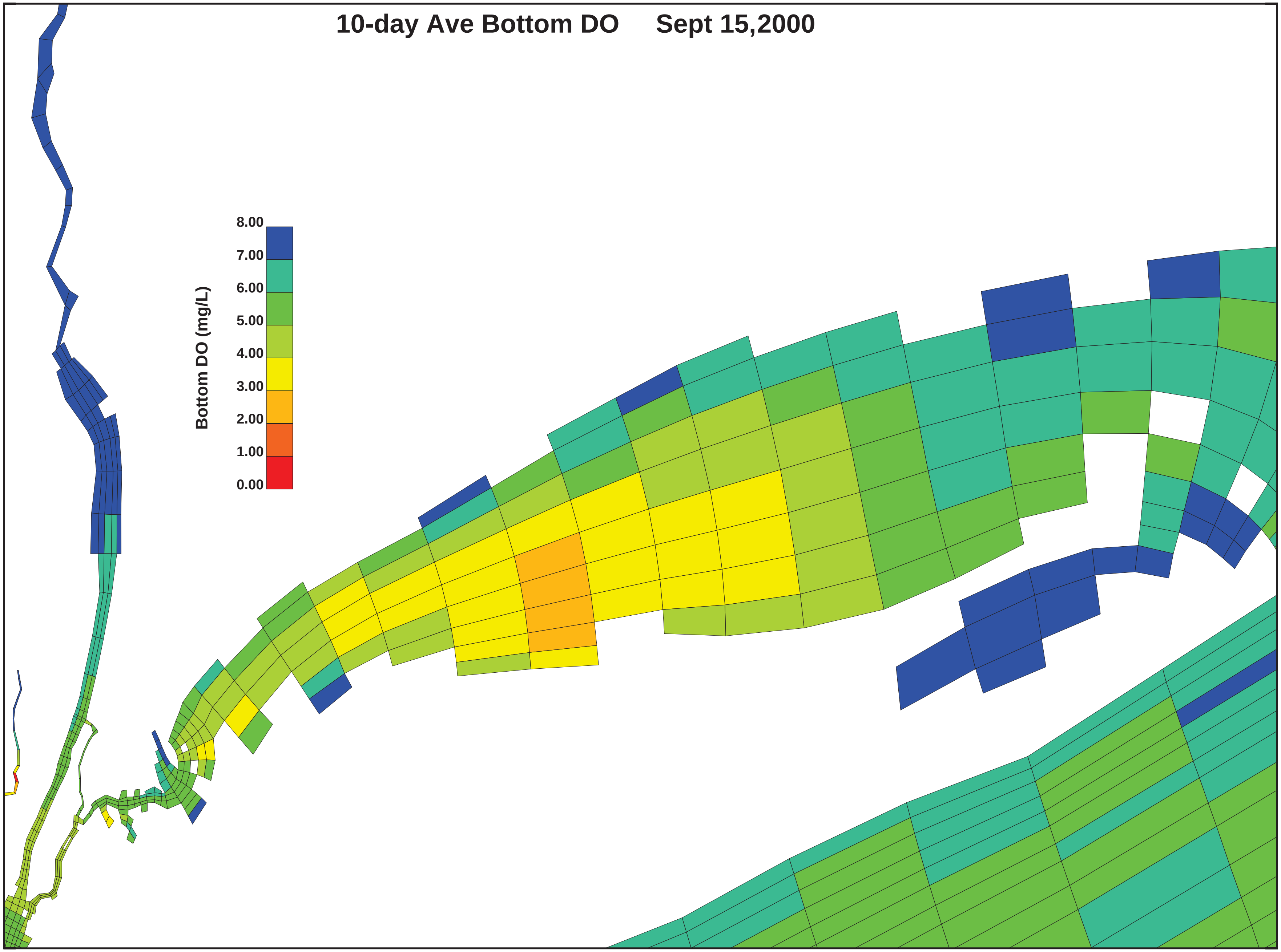
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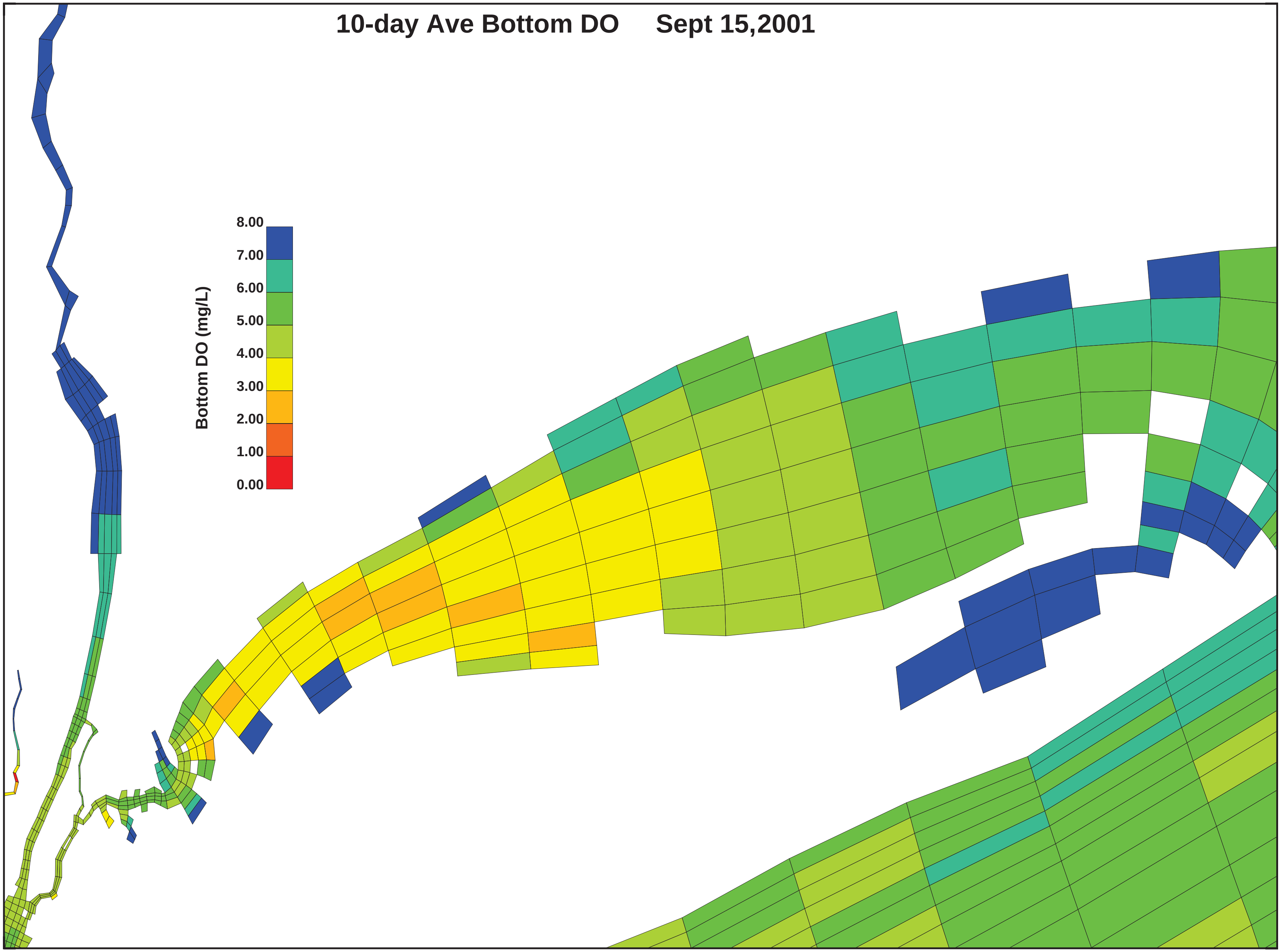
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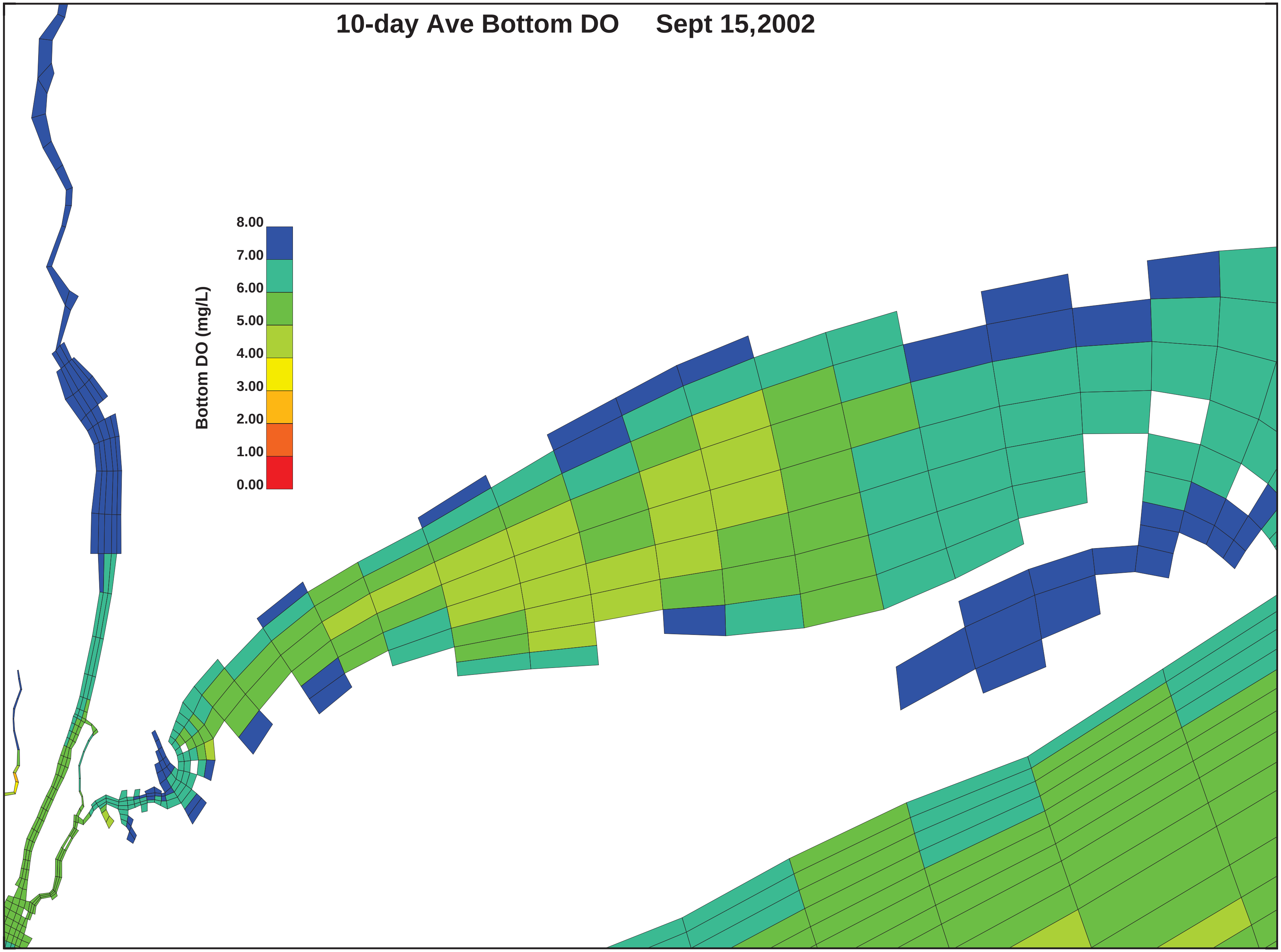

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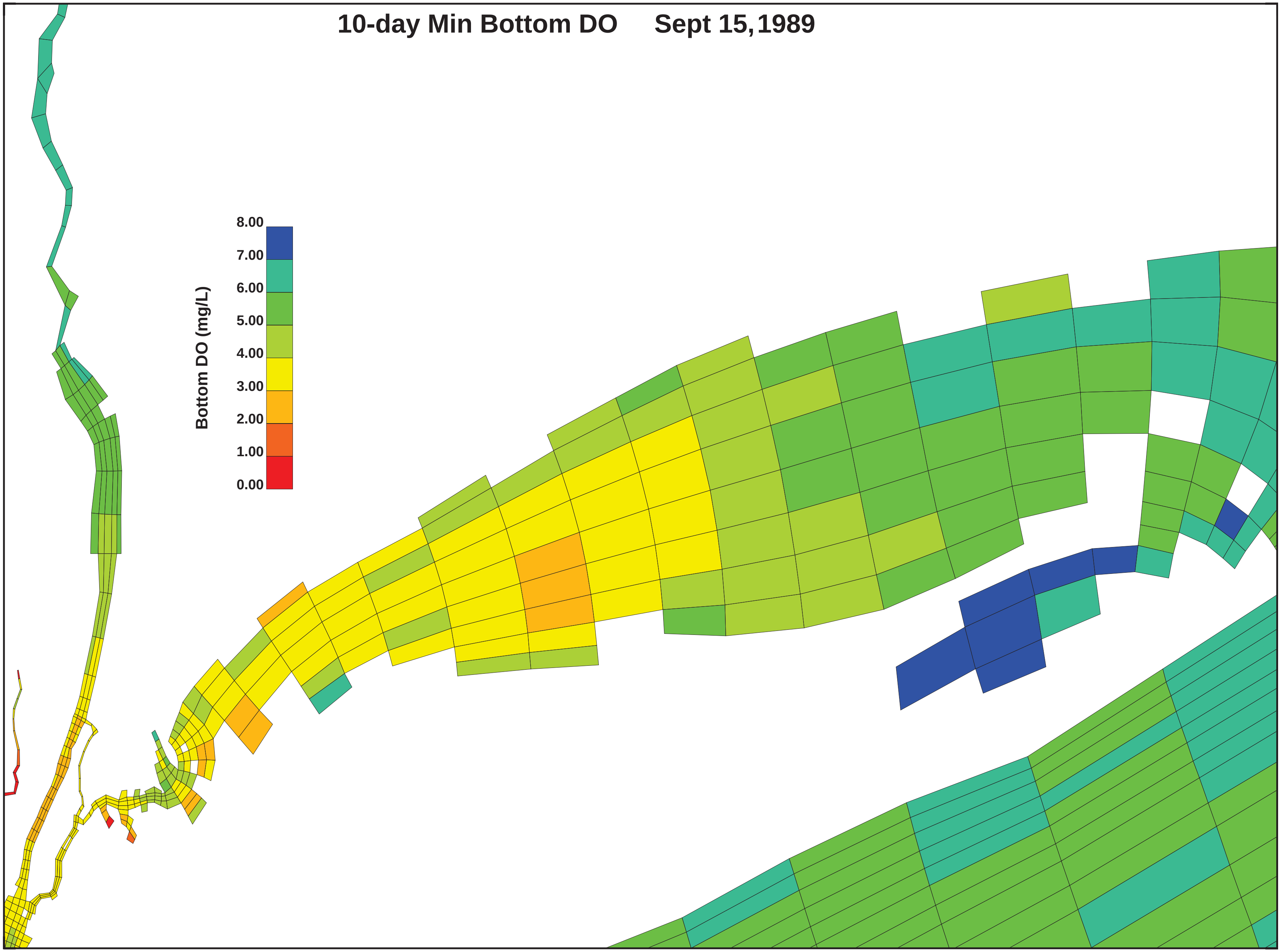

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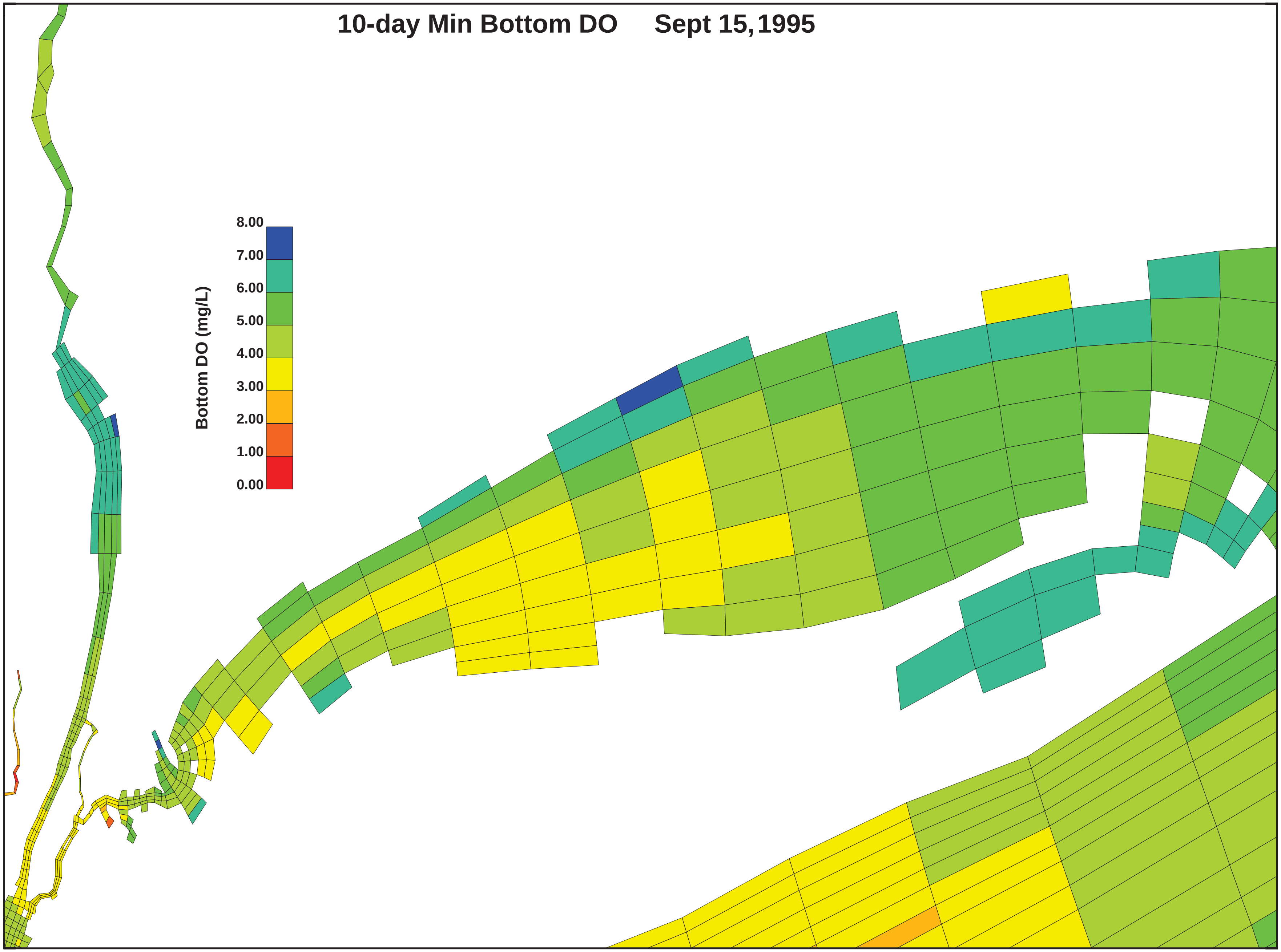

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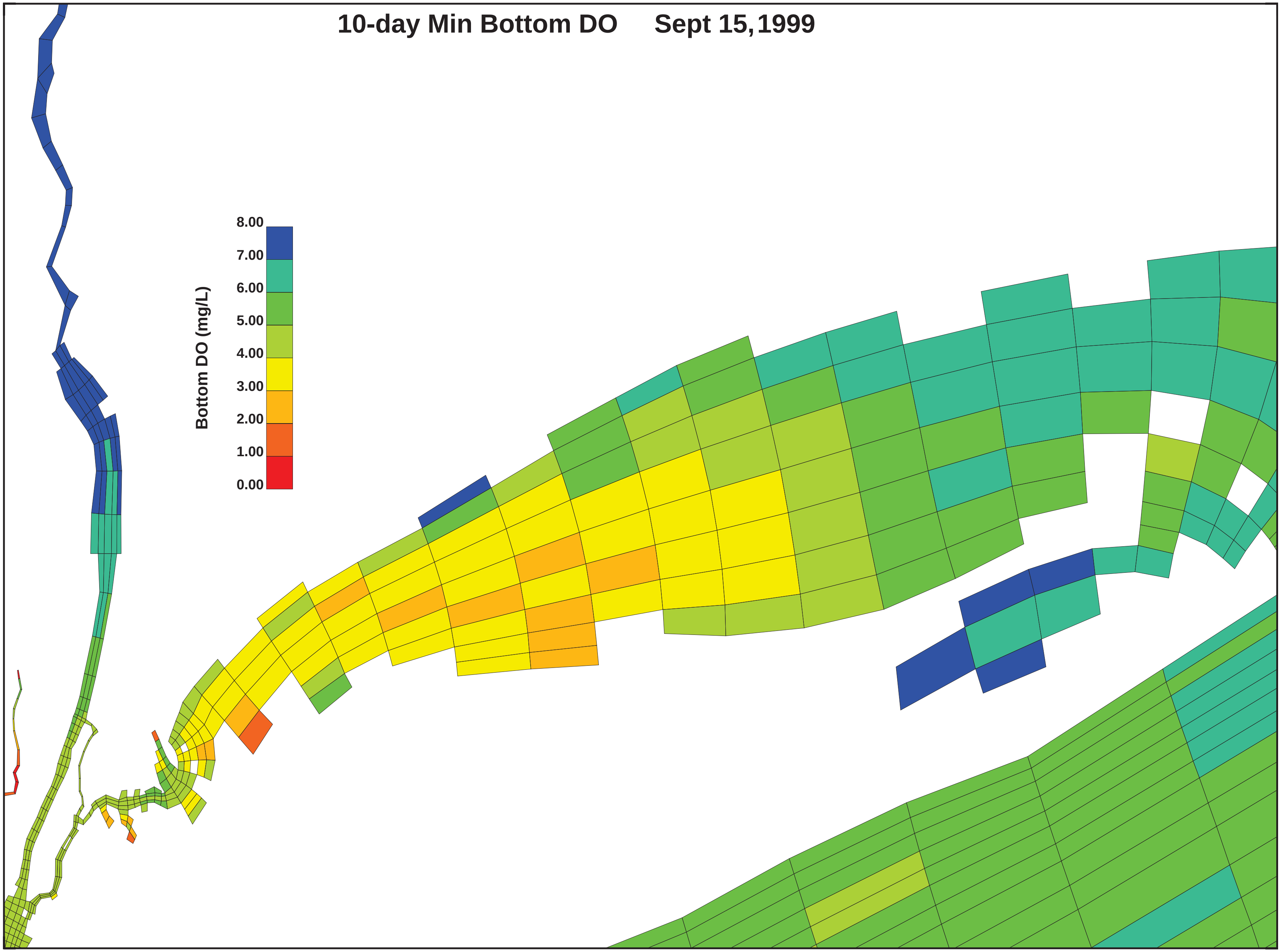

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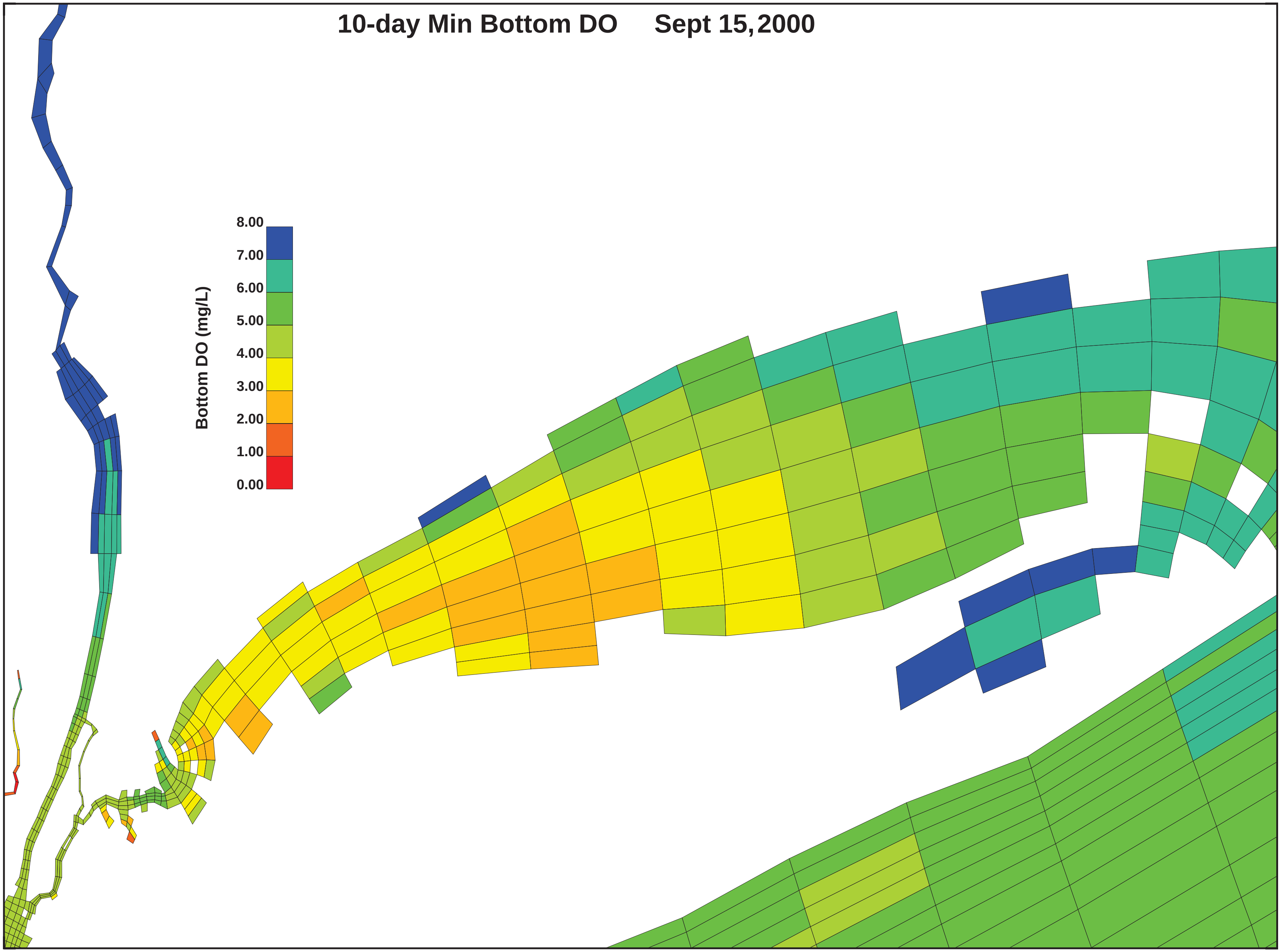

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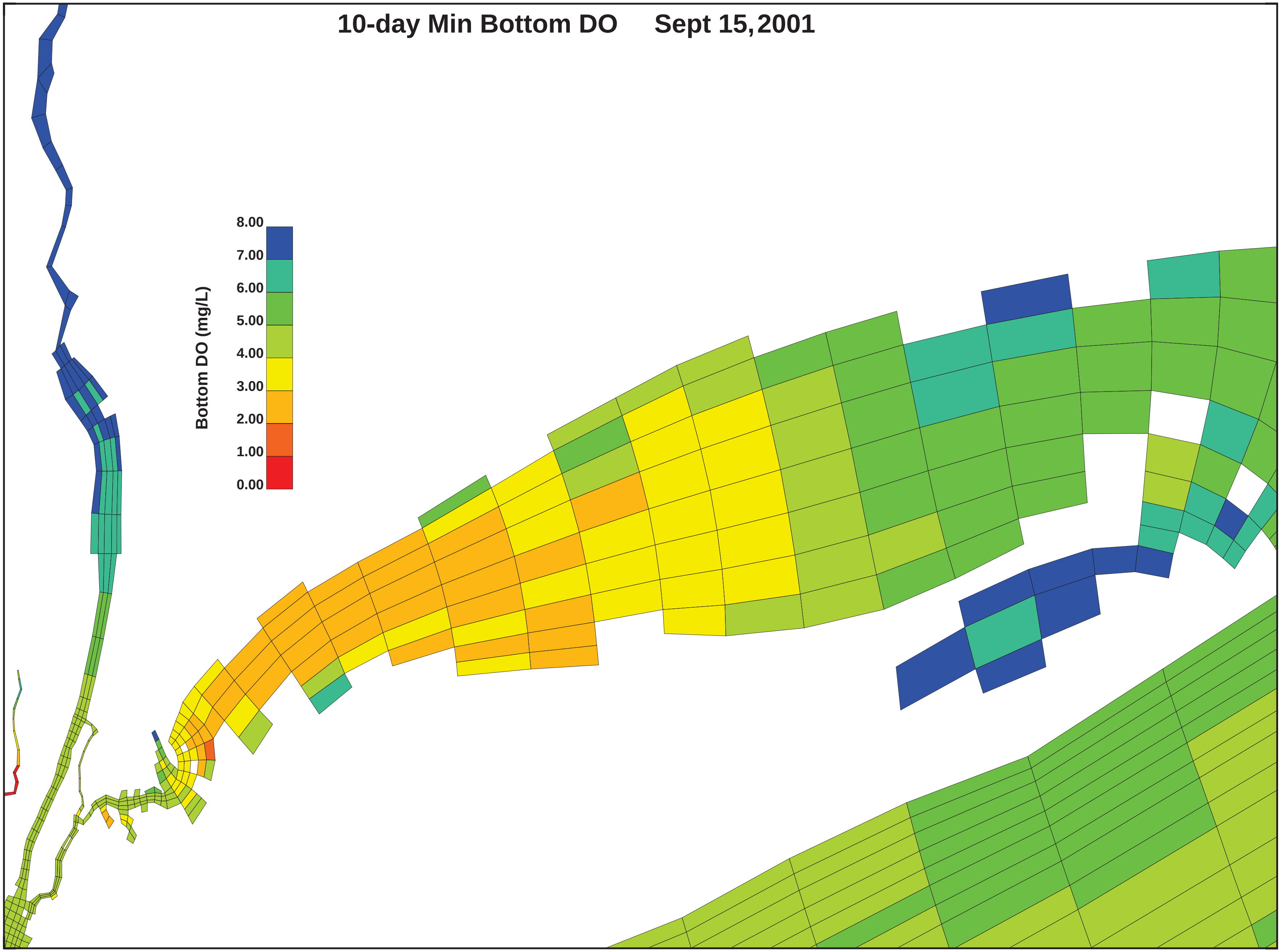
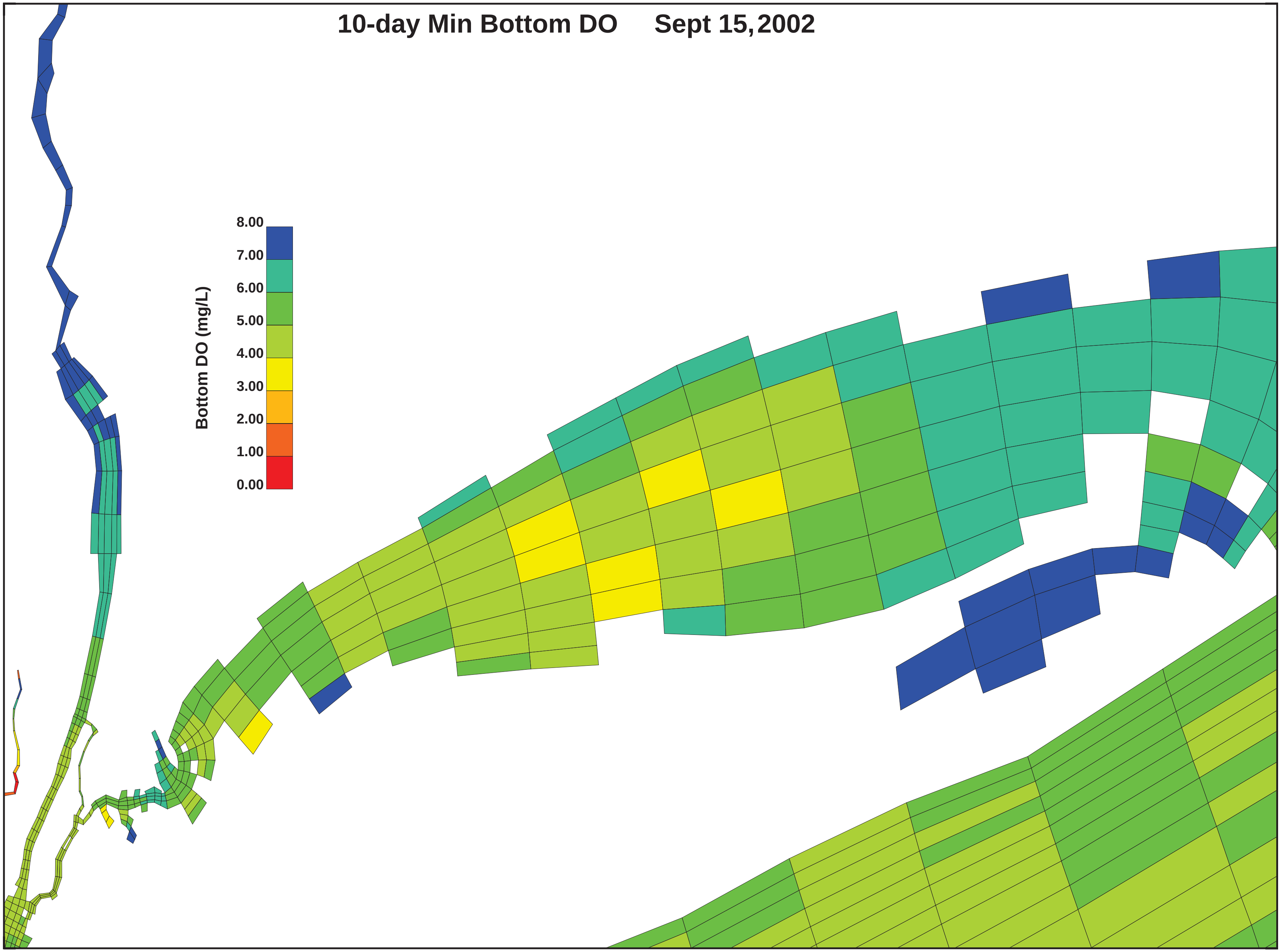
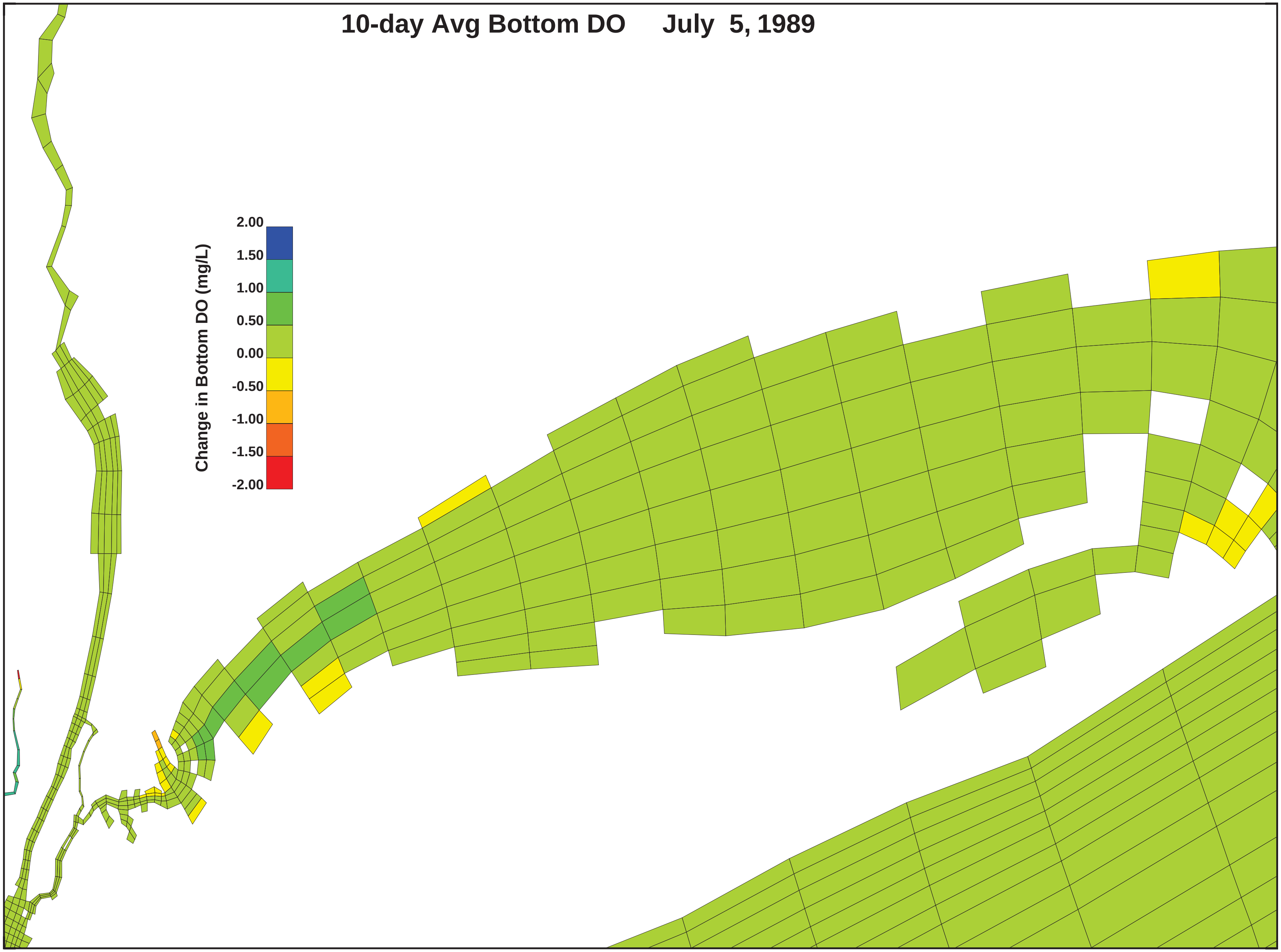
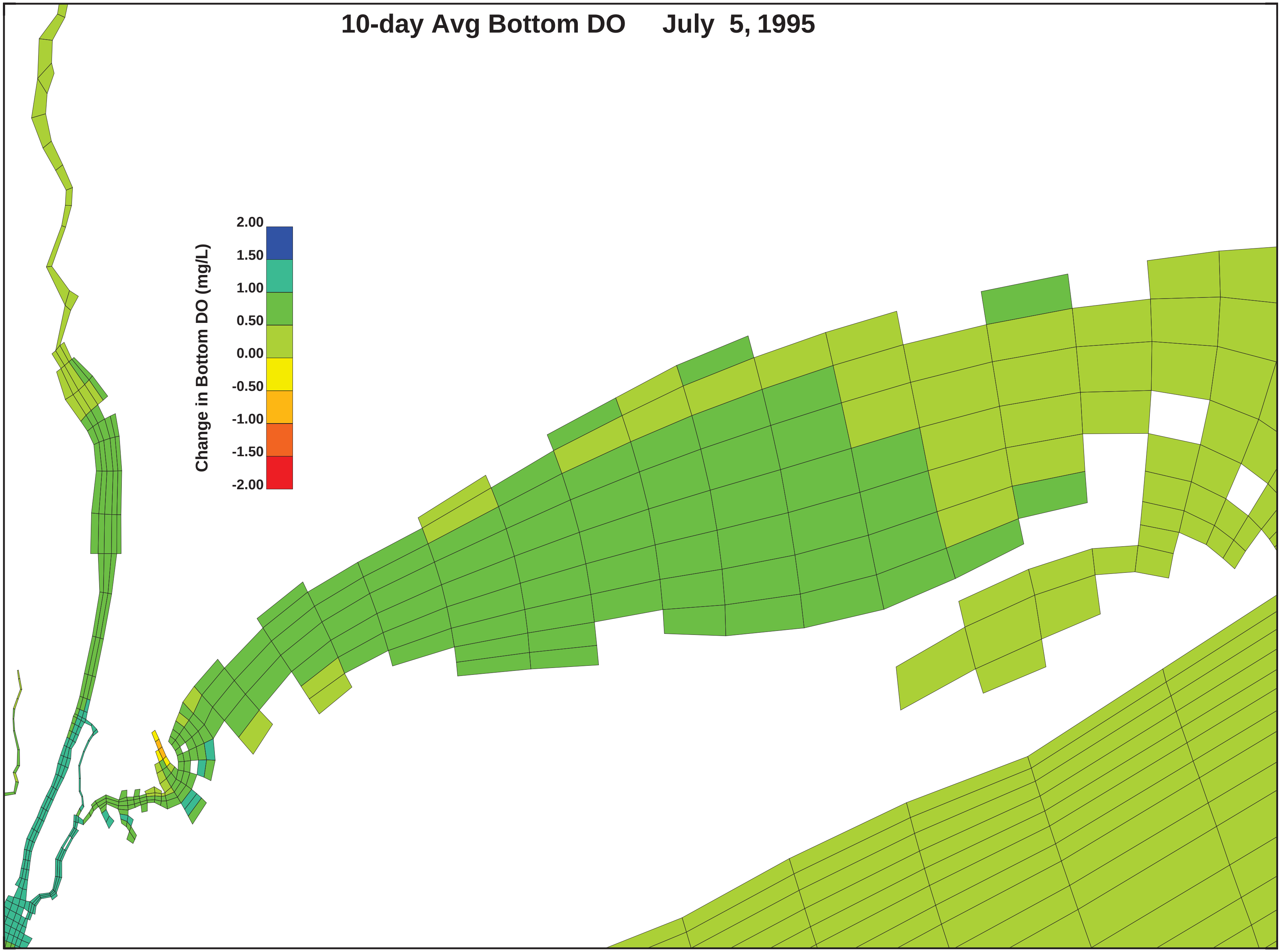

10-day Min Bottom DO Sept 15, 2001
Bottom DO (mg/L)
8.00
7.00
6.00
5.00
4.00
3.00
2.00
1.00
0.00

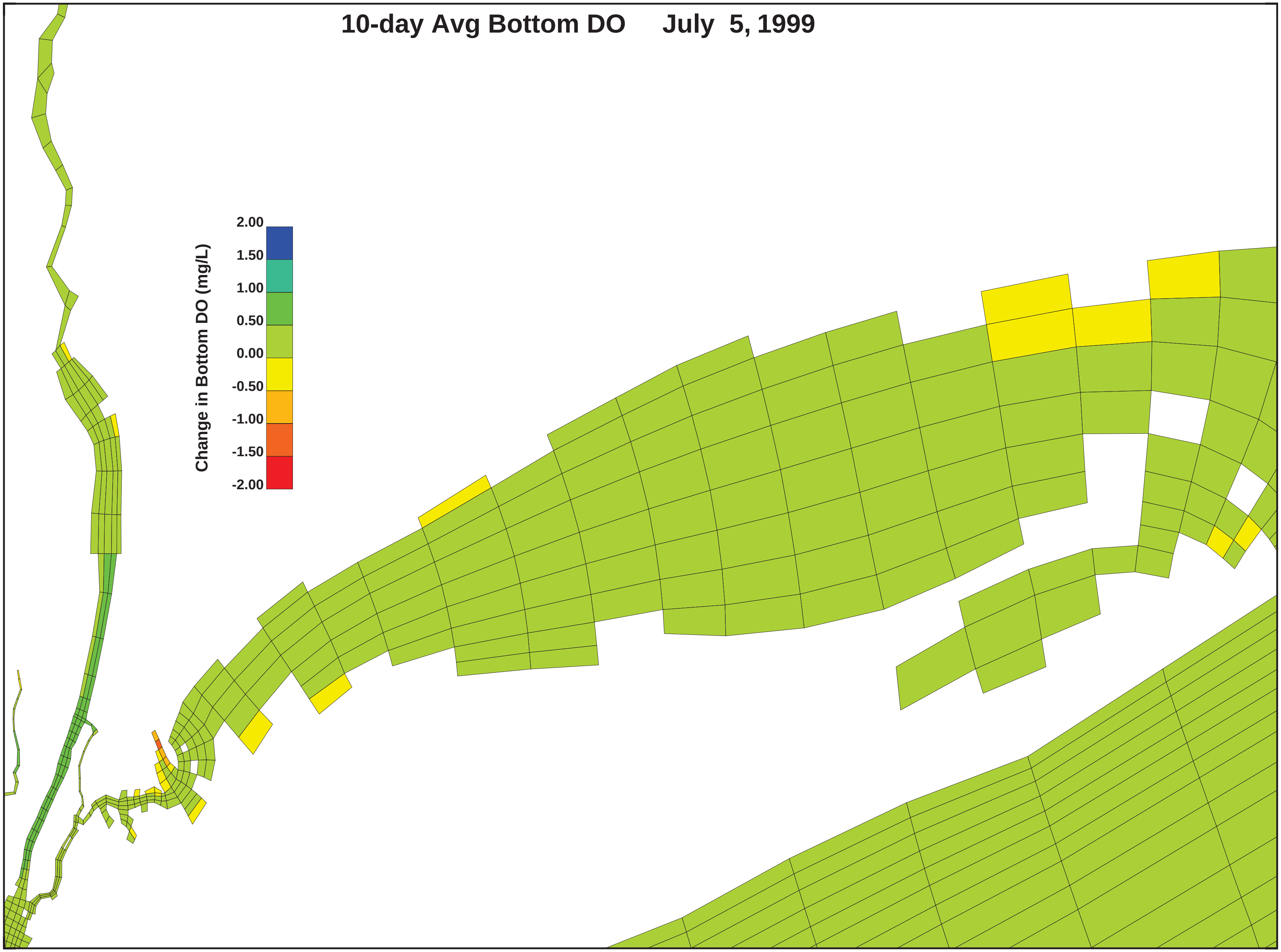
10-day Min Bottom DO   Sept 15, 2002

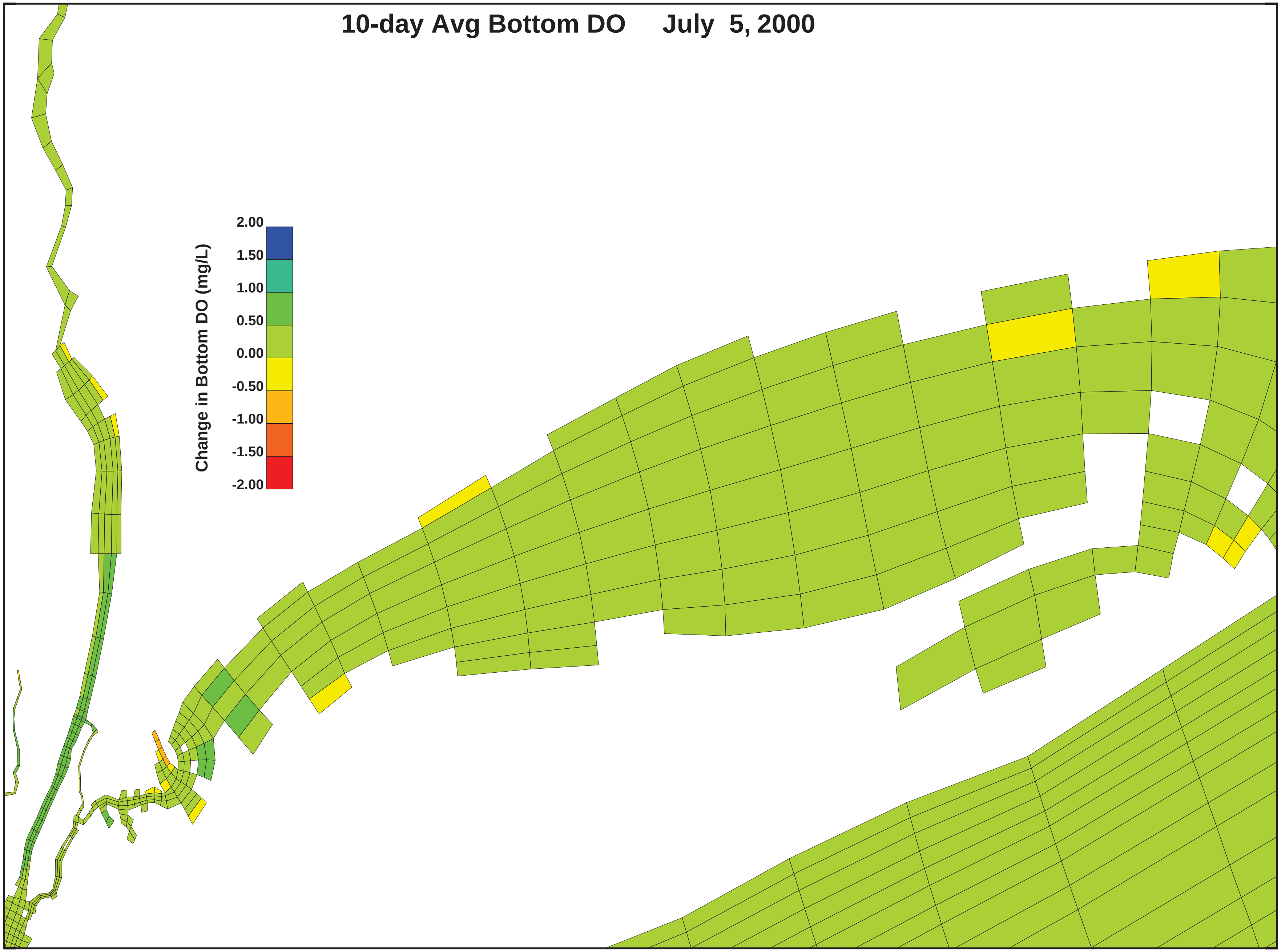


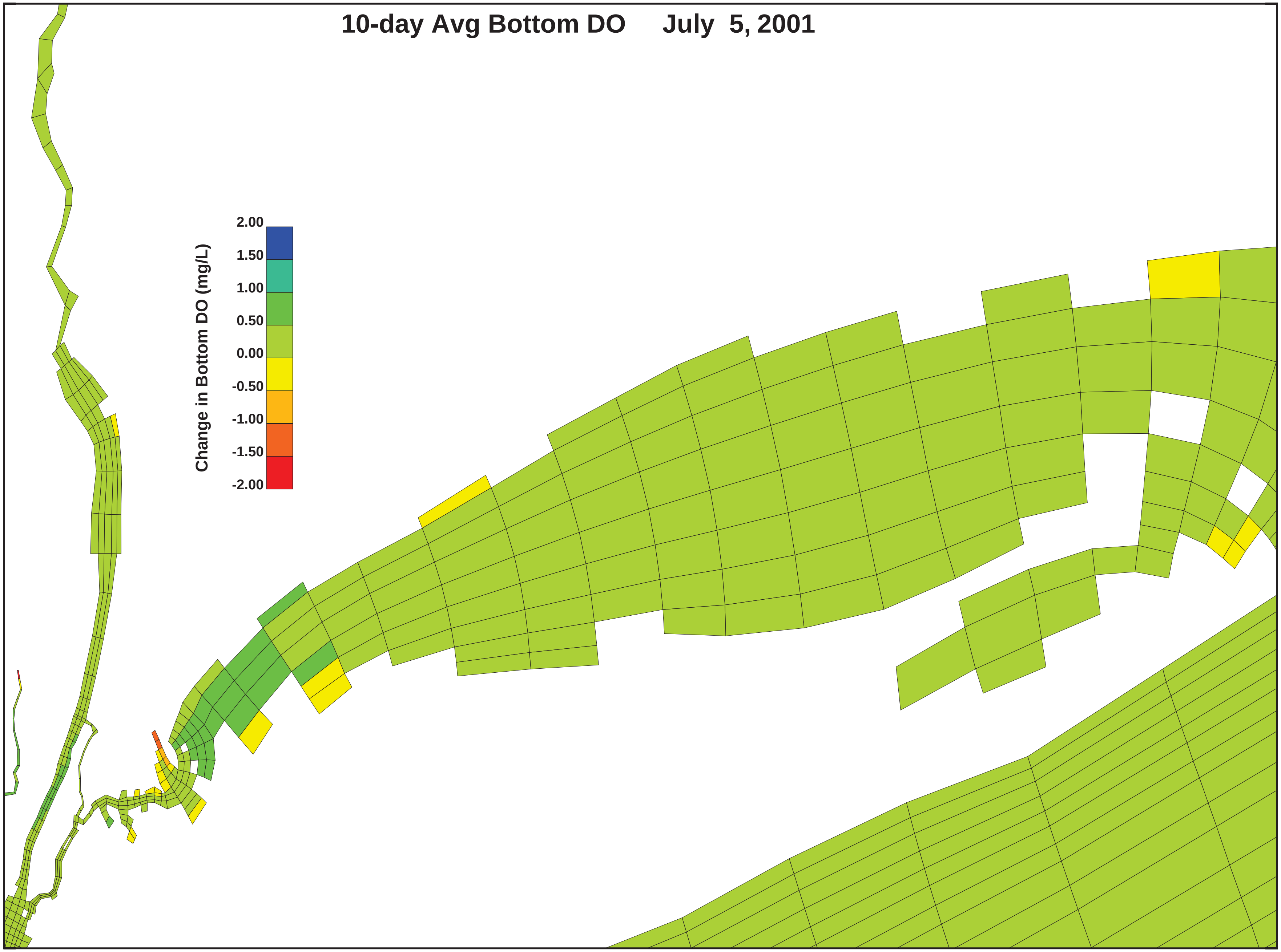

10-day Avg Bottom DO July 5, 1989
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00

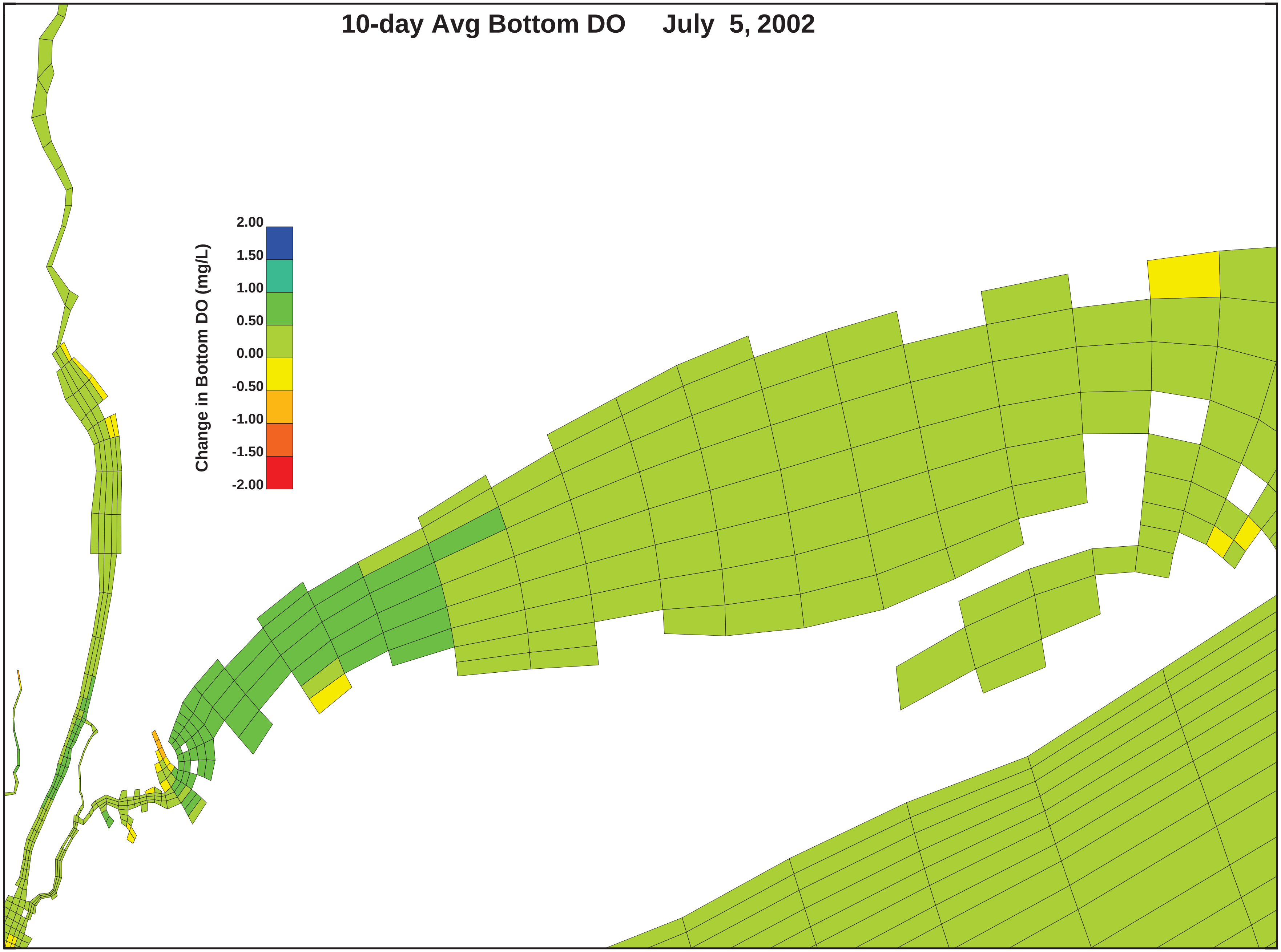


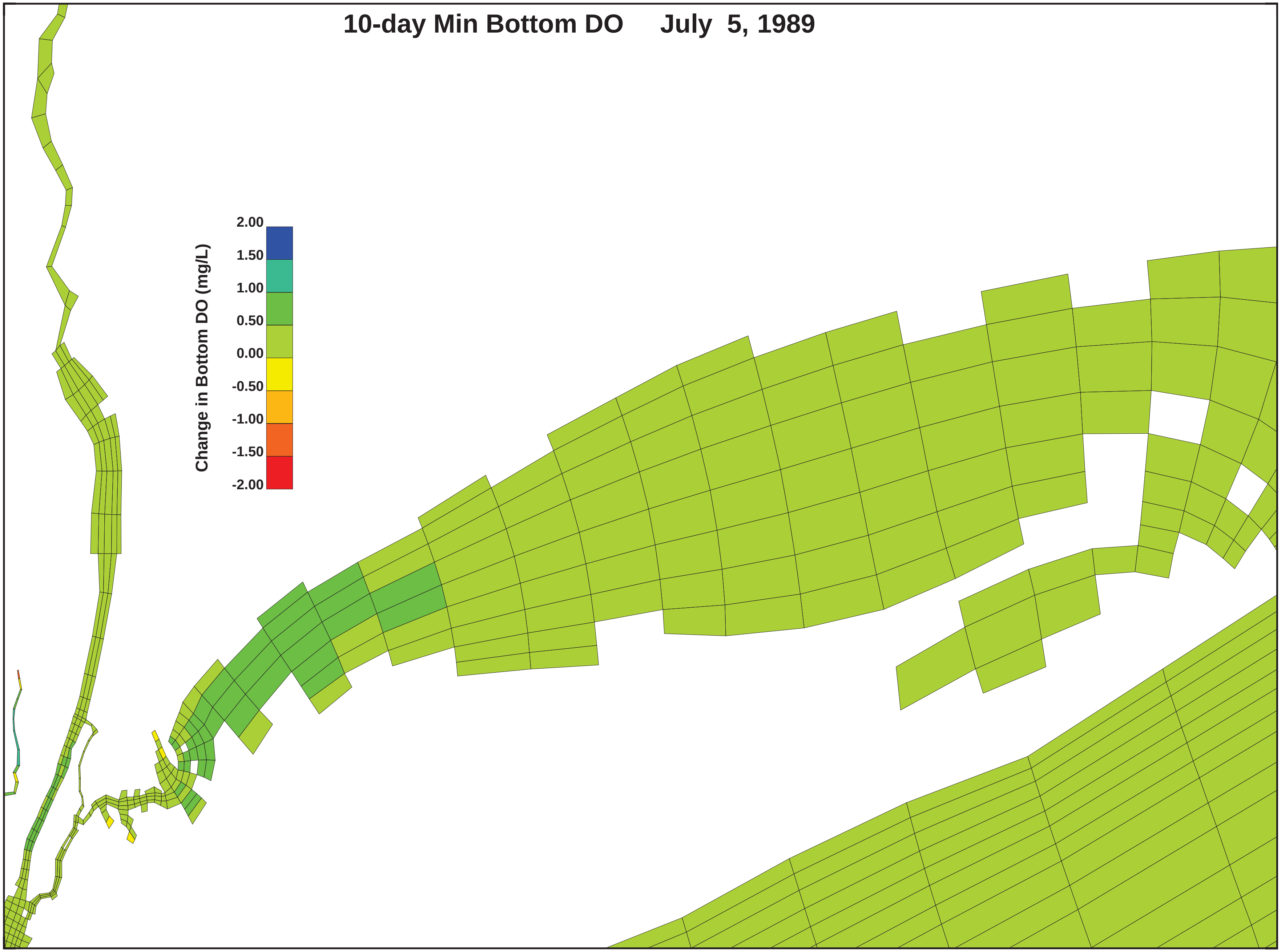
10-day Avg Bottom DO   July 5, 1995
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00

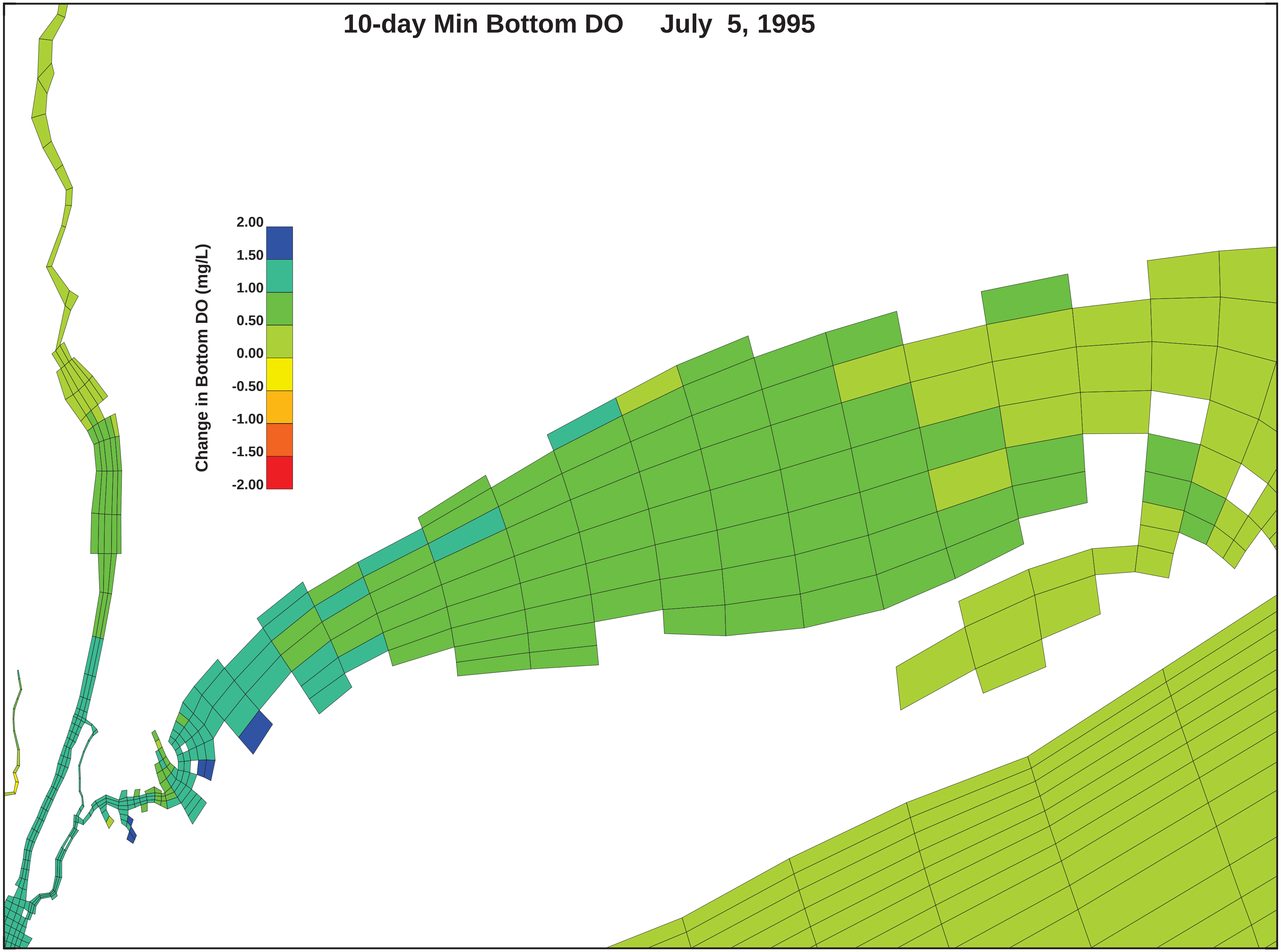

10-day Avg Bottom DO July 5, 1999
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00


10-day Avg Bottom DO July 5, 2000
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00

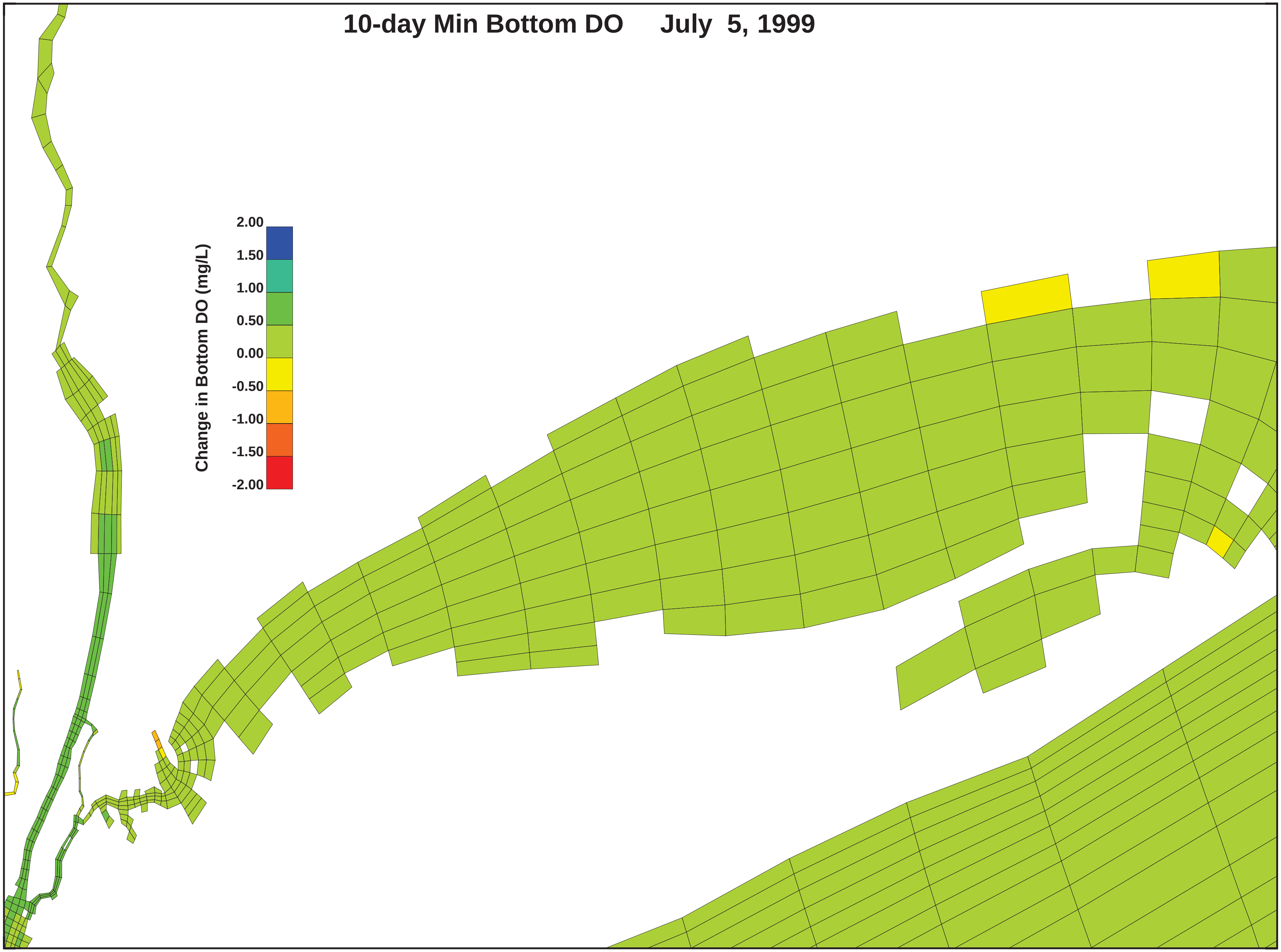

10-day Avg Bottom DO   July 5, 2001
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00

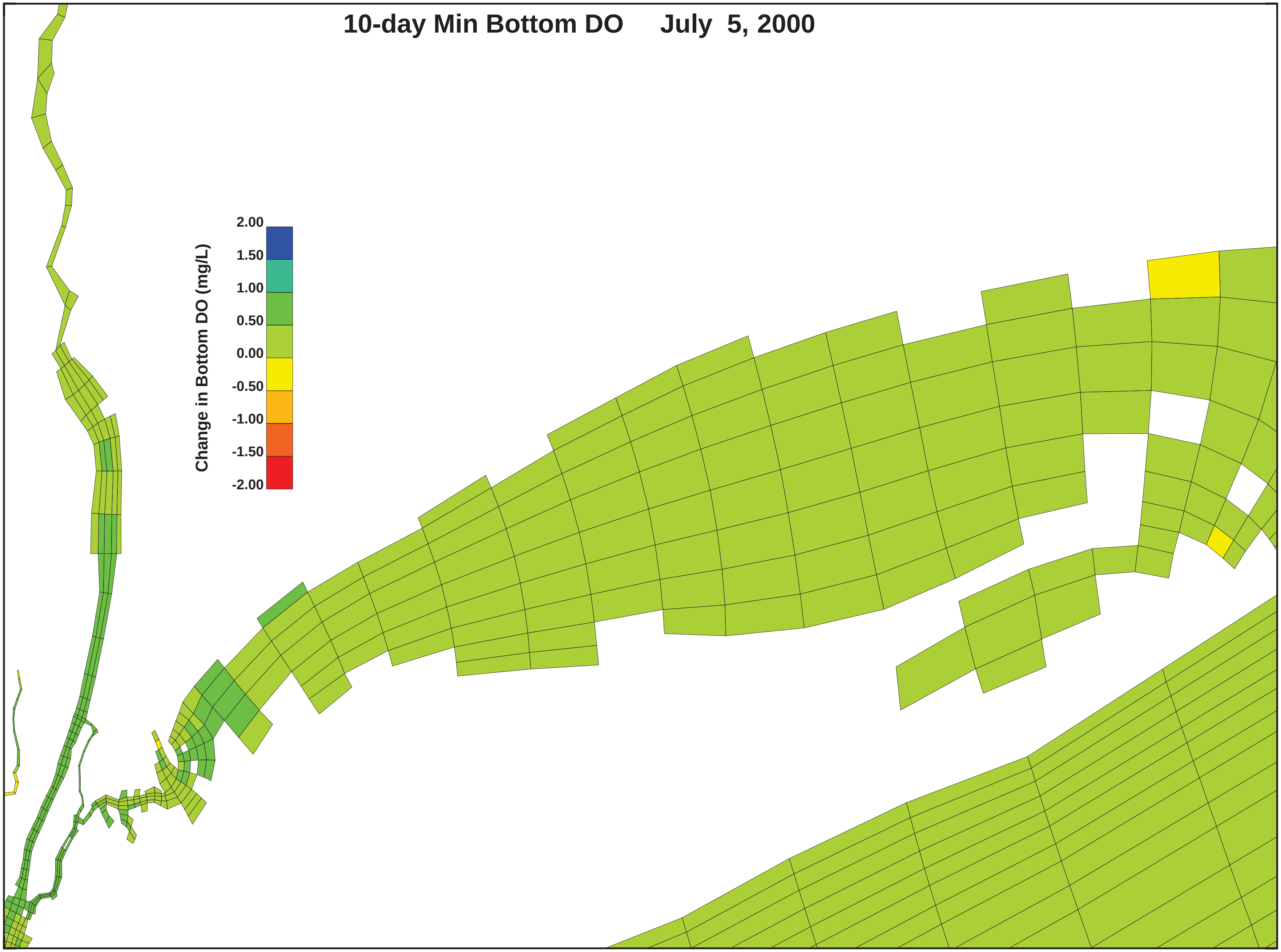

10-day Avg Bottom DO July 5, 2002
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00


10-day Min Bottom DO July 5, 1989
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00

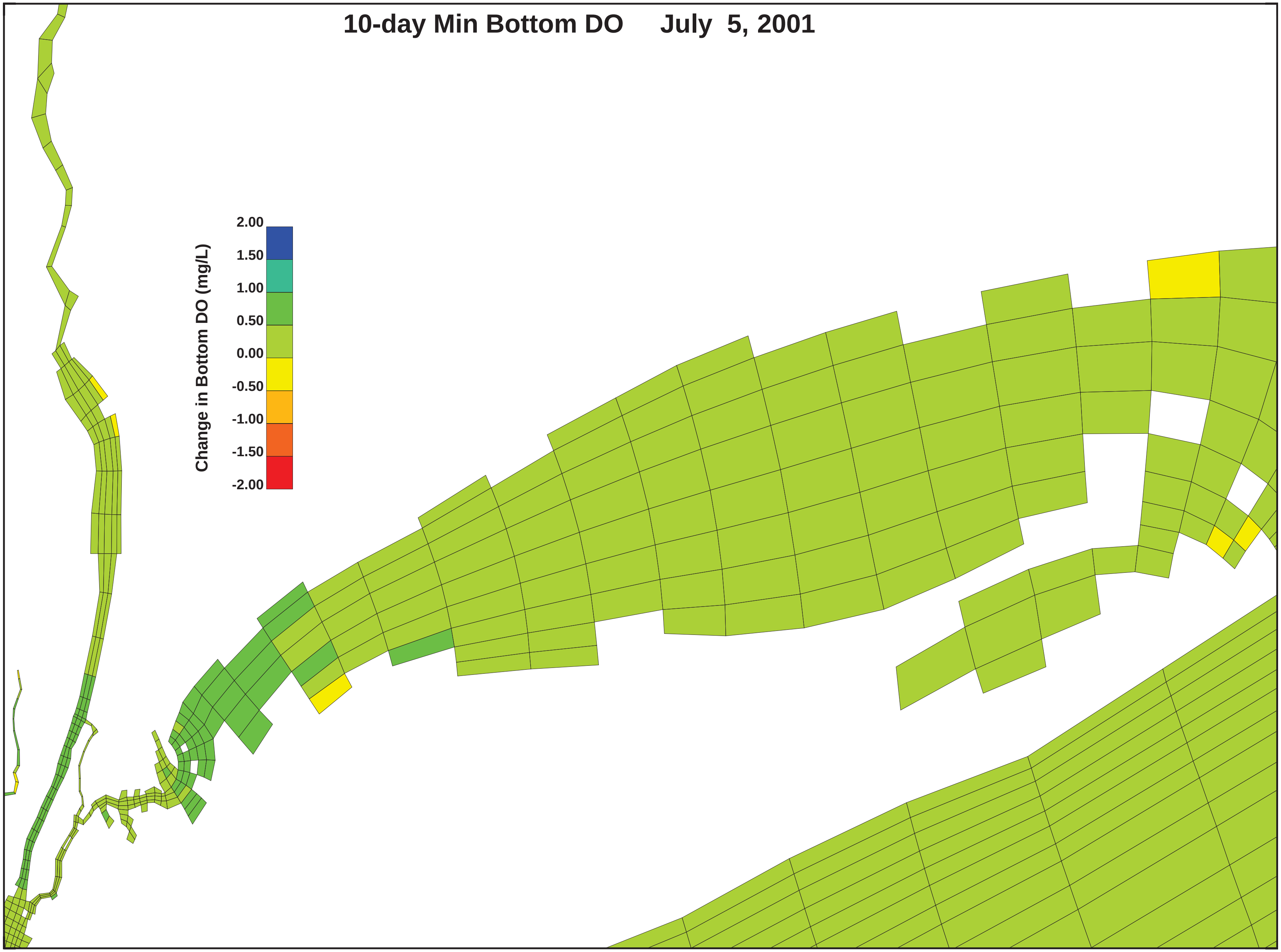


10-day Min Bottom DO July 5, 1995
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00

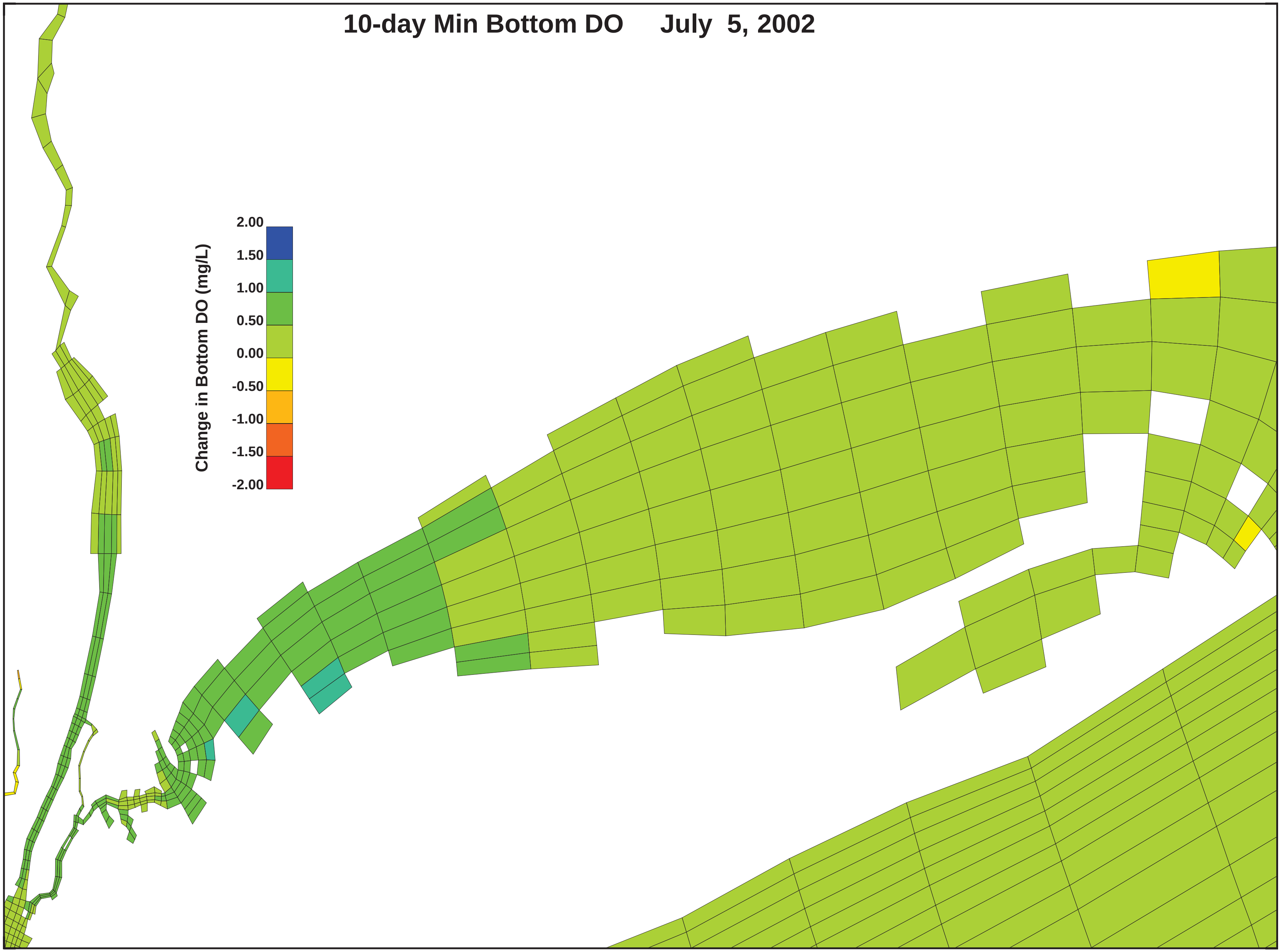

10-day Min Bottom DO July 5, 1999
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00

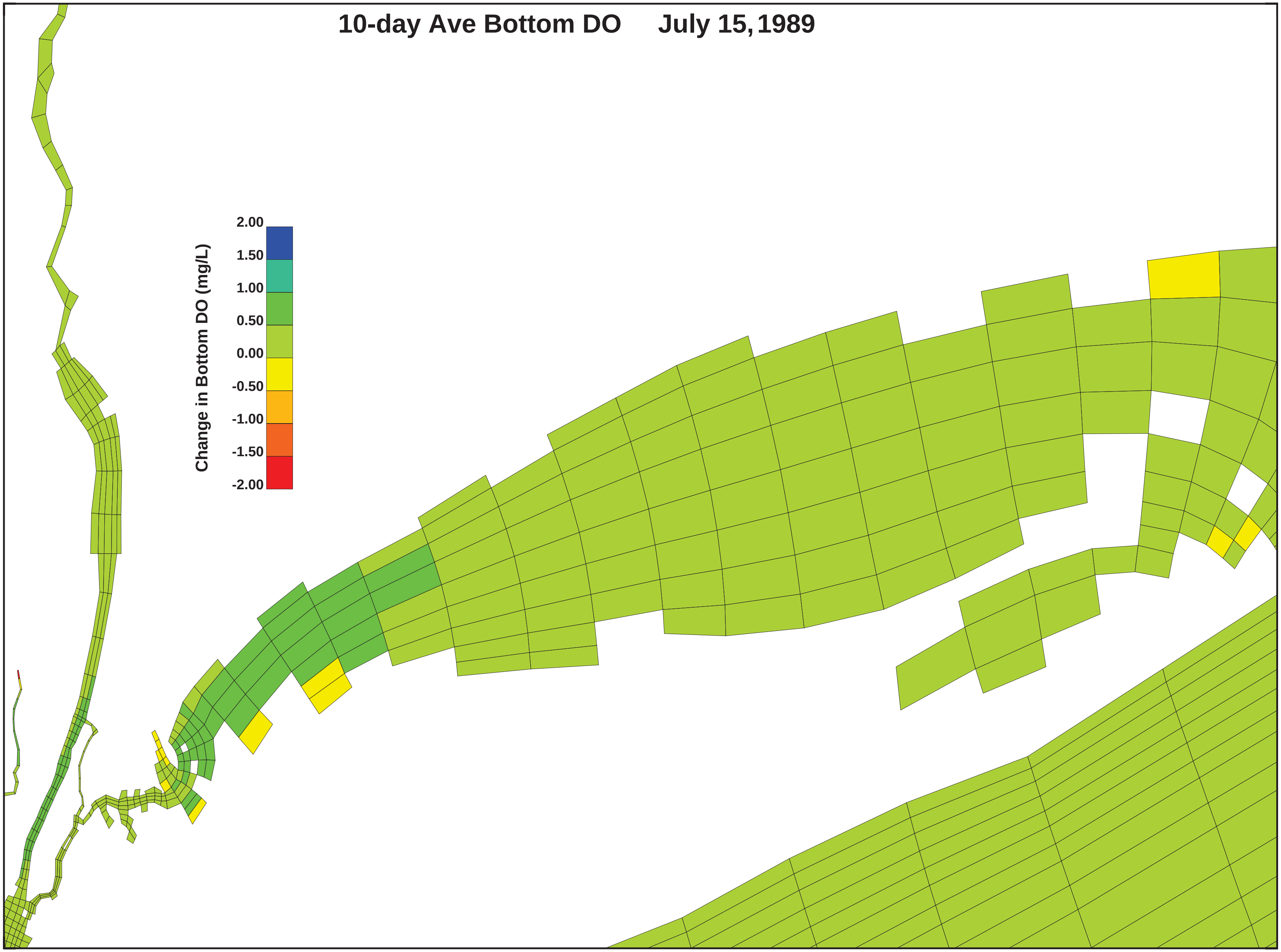


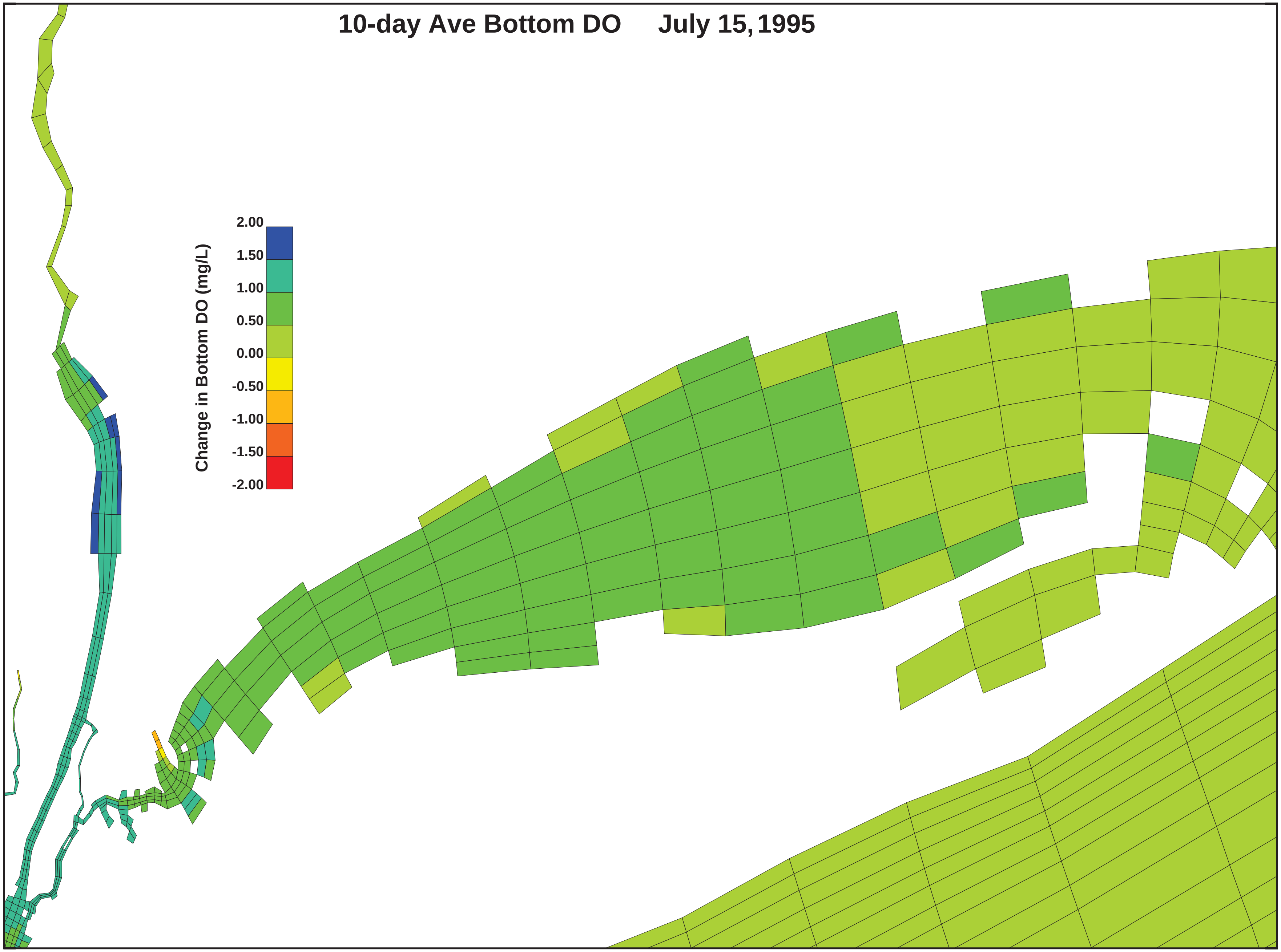
10-day Min Bottom DO July 5, 2000
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00

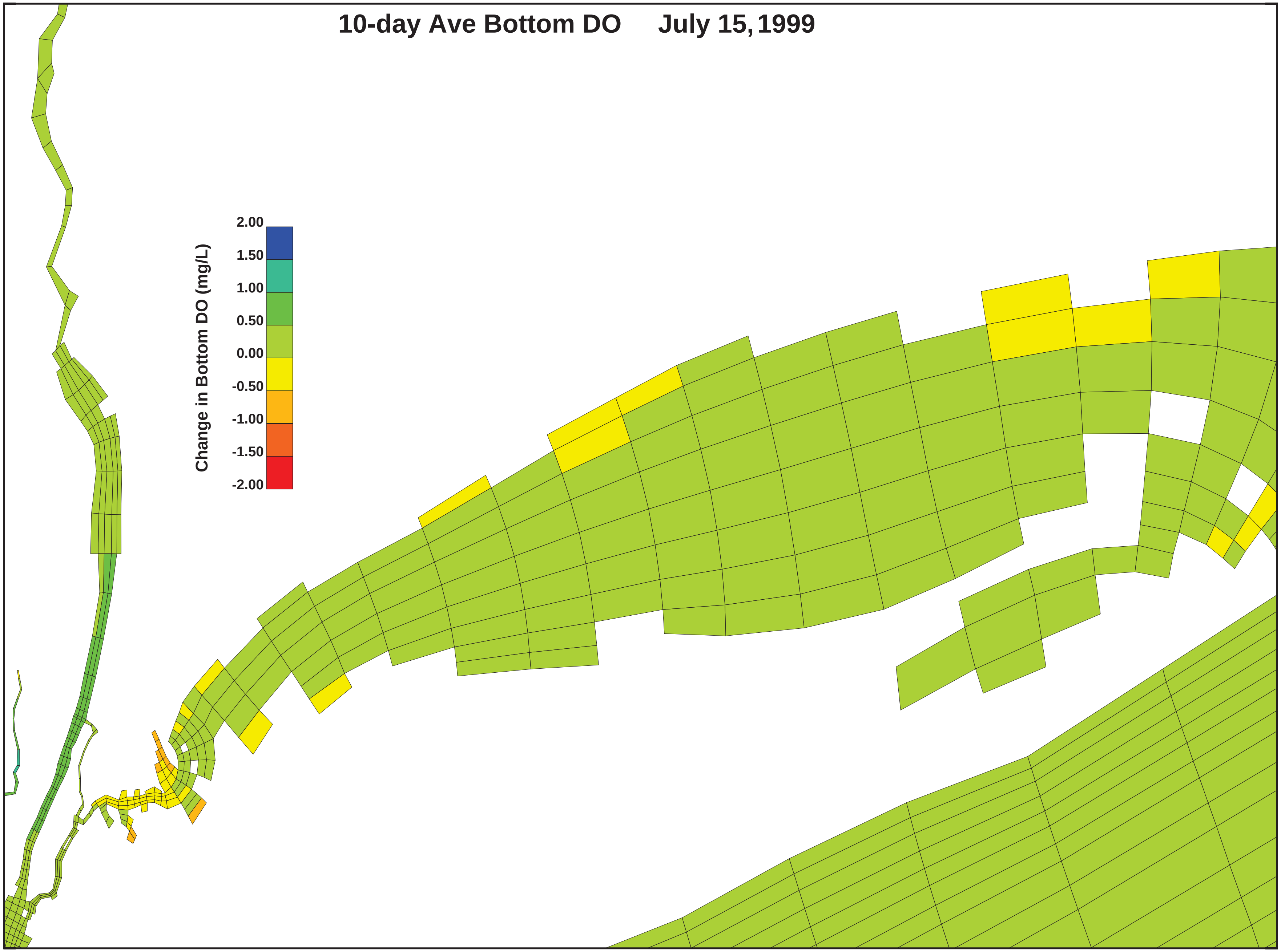


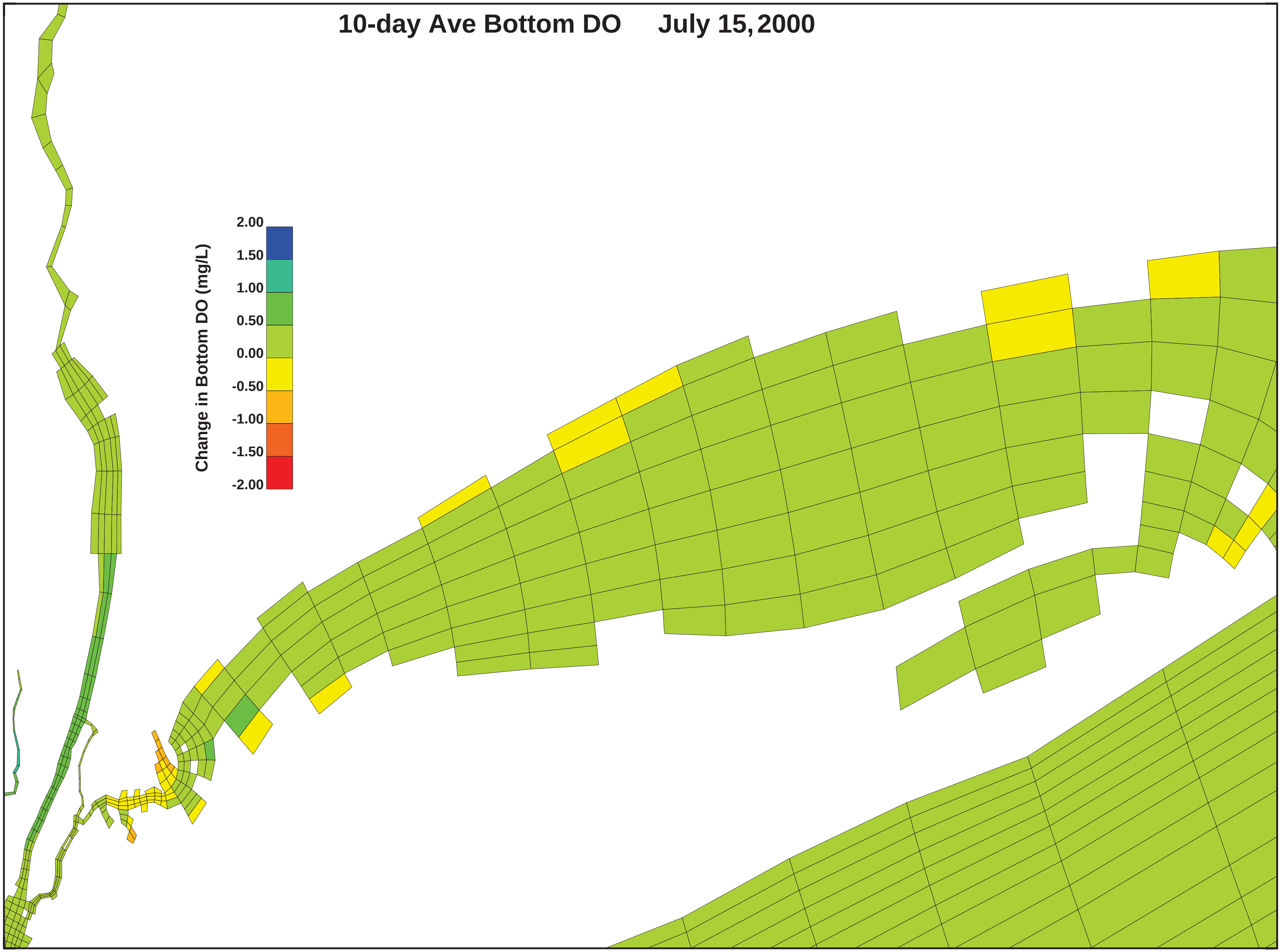
10-day Min Bottom DO July 5, 2001
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00


10-day Min Bottom DO July 5, 2002
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00

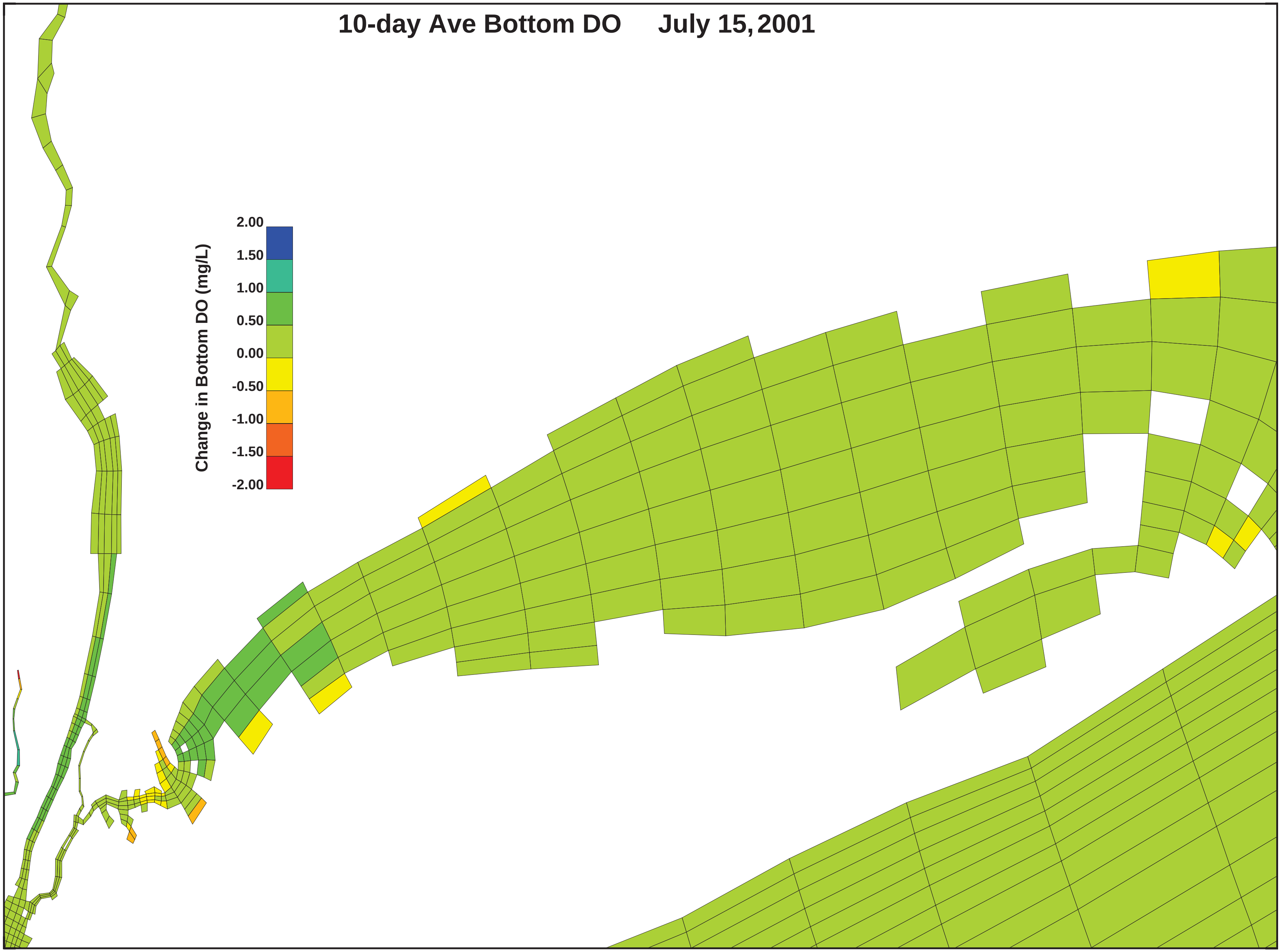

10-day Ave Bottom DO   July 15, 1989
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00


10-day Ave Bottom DO July 15, 1995
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00

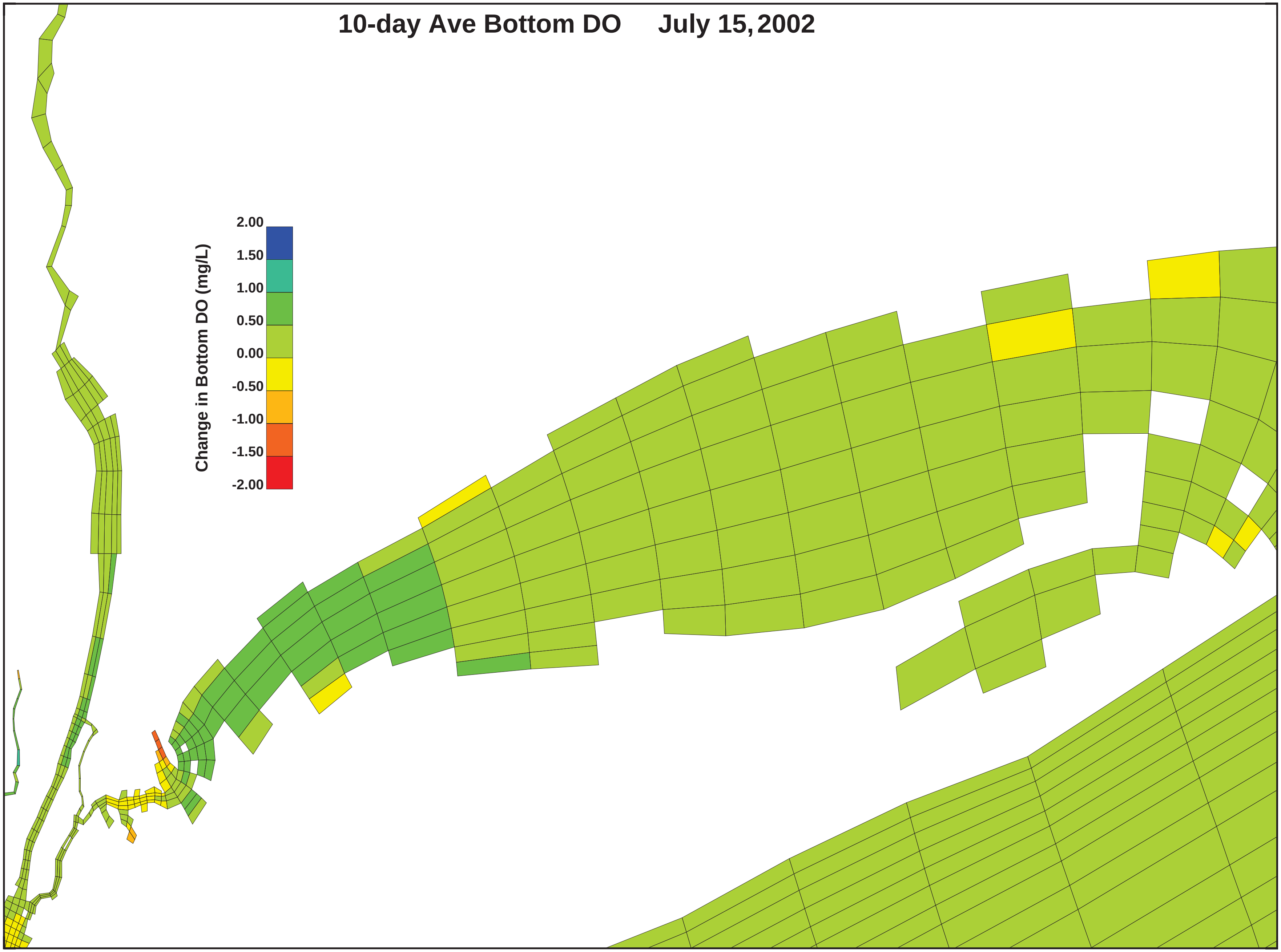

10-day Ave Bottom DO July 15, 1999
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00



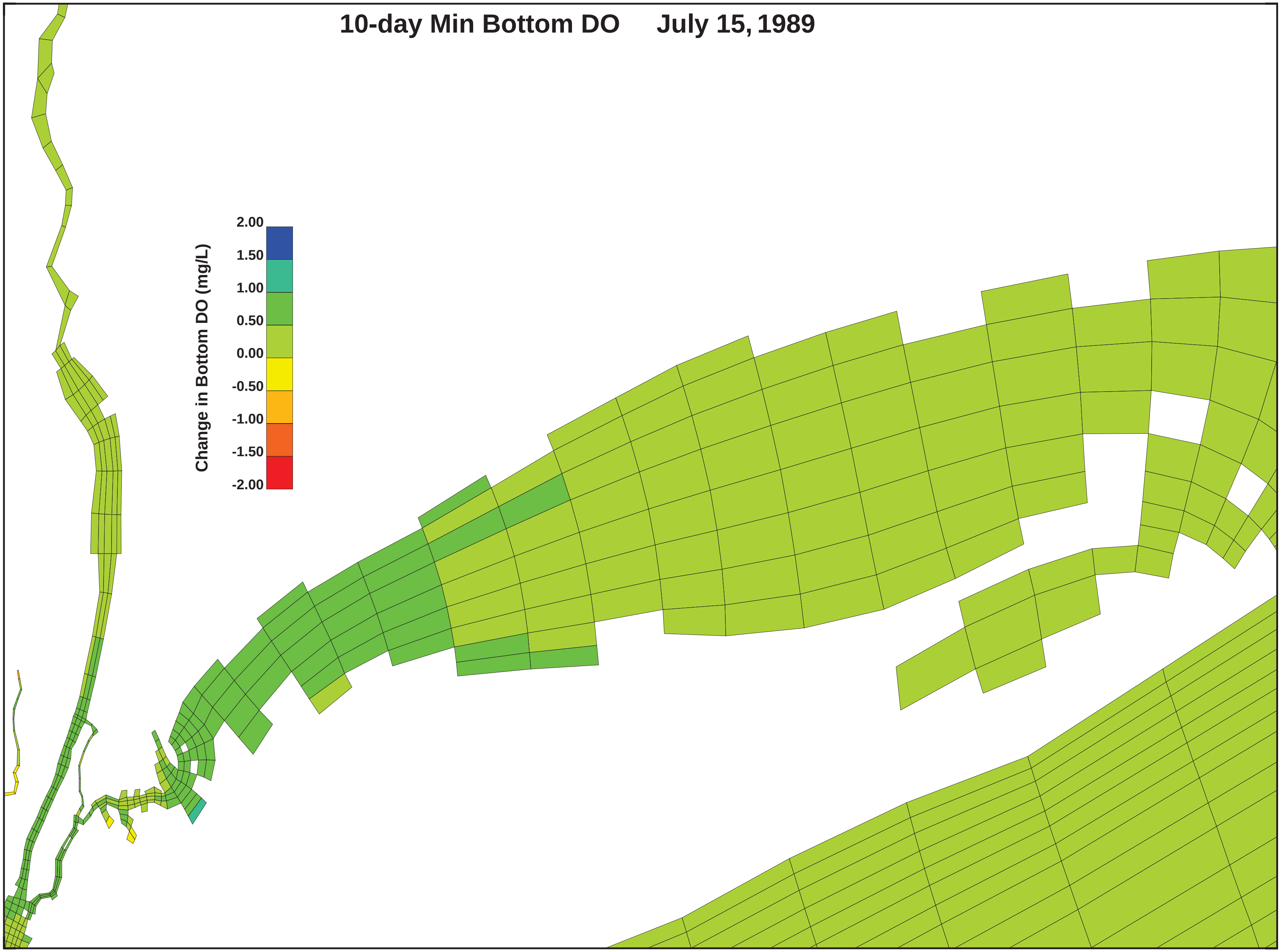
10-day Ave Bottom DO July 15, 2000
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00



10-day Ave Bottom DO July 15, 2001
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00



10-day Ave Bottom DO July 15, 2002
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00


10-day Min Bottom DO July 15, 1989
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
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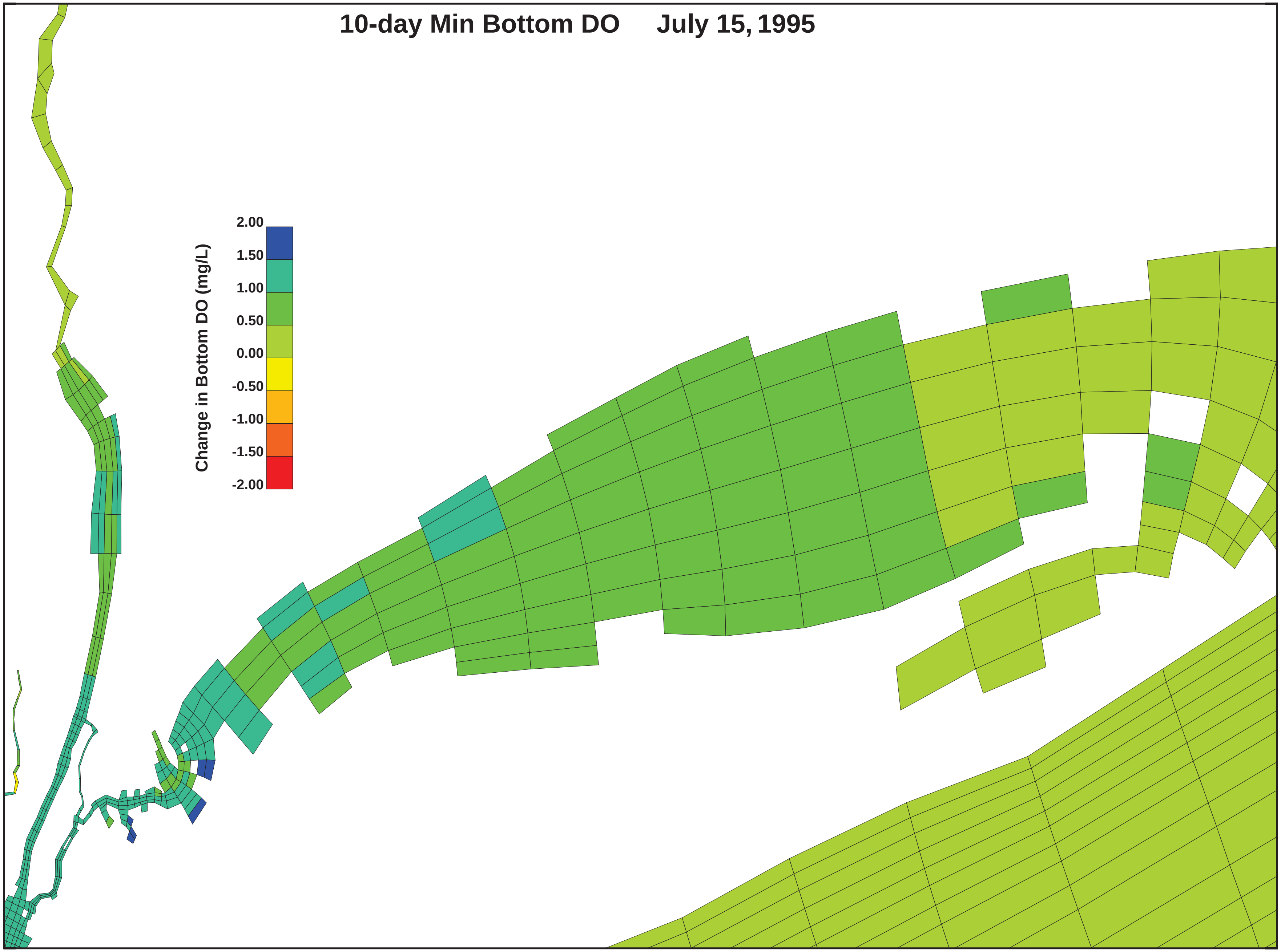
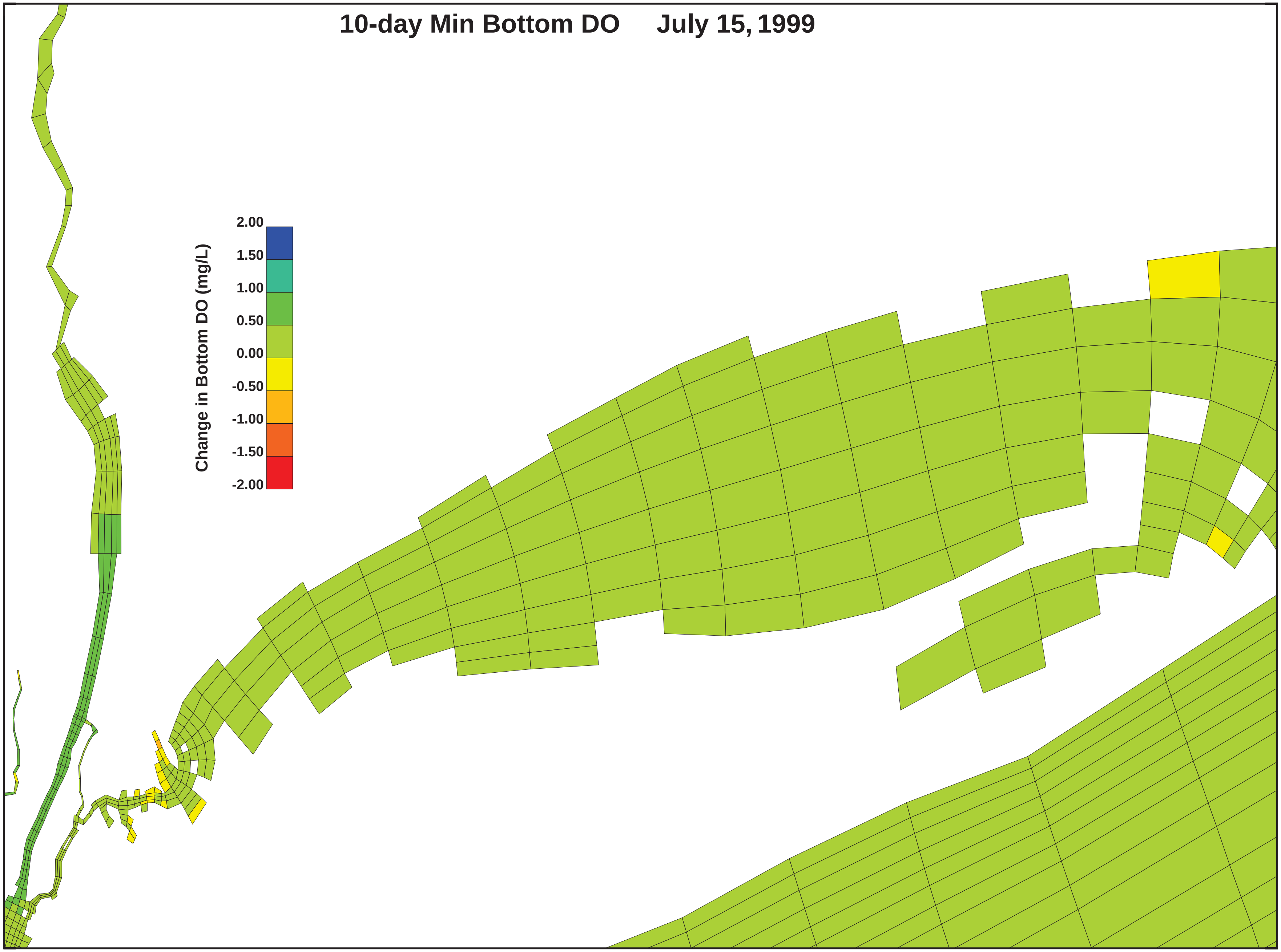
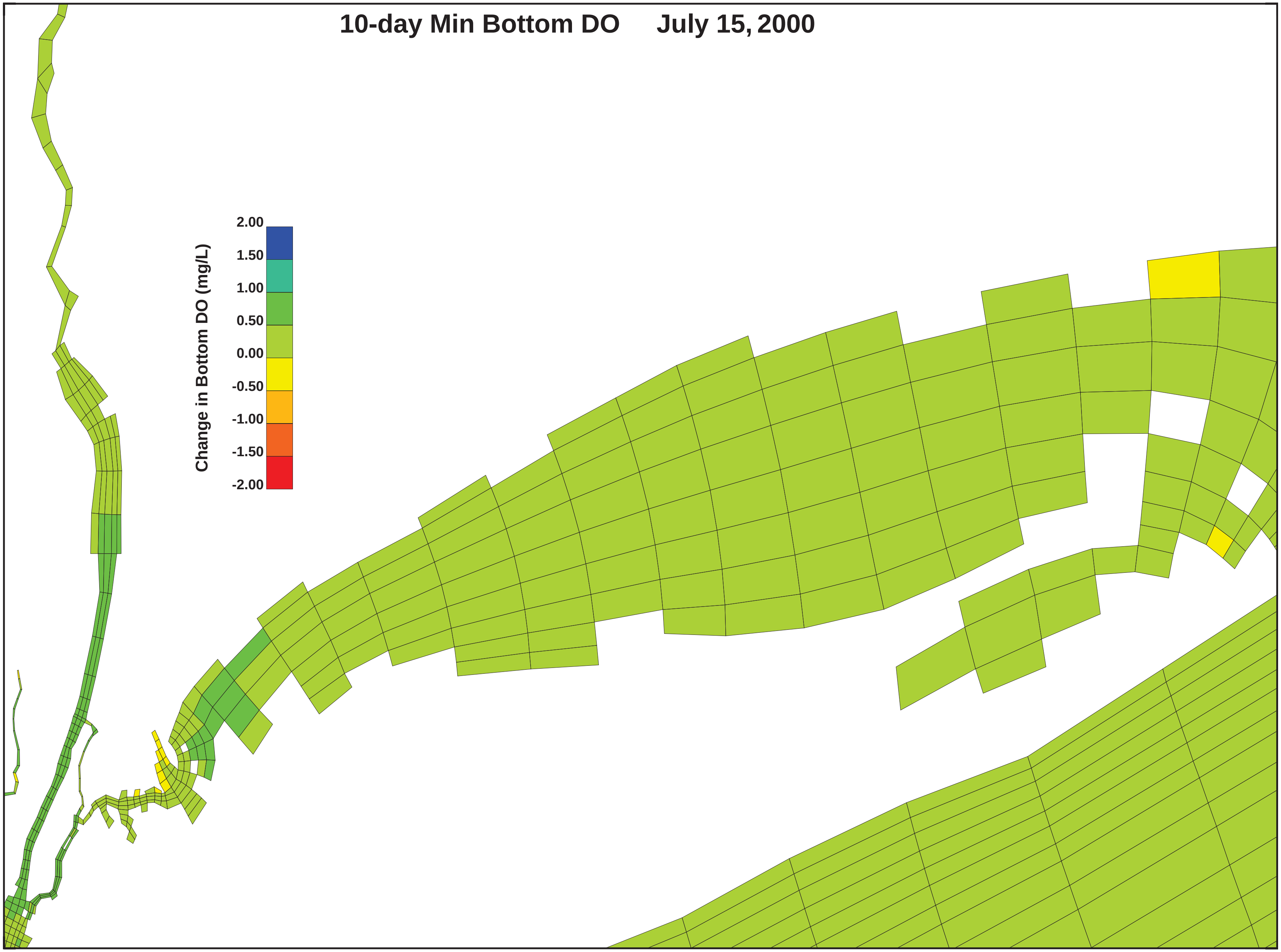
-1.50
-2.00



10-day Min Bottom DO July 15, 1995
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00


10-day Min Bottom DO July 15, 1999
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00


10-day Min Bottom DO July 15, 2000
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
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-0.50
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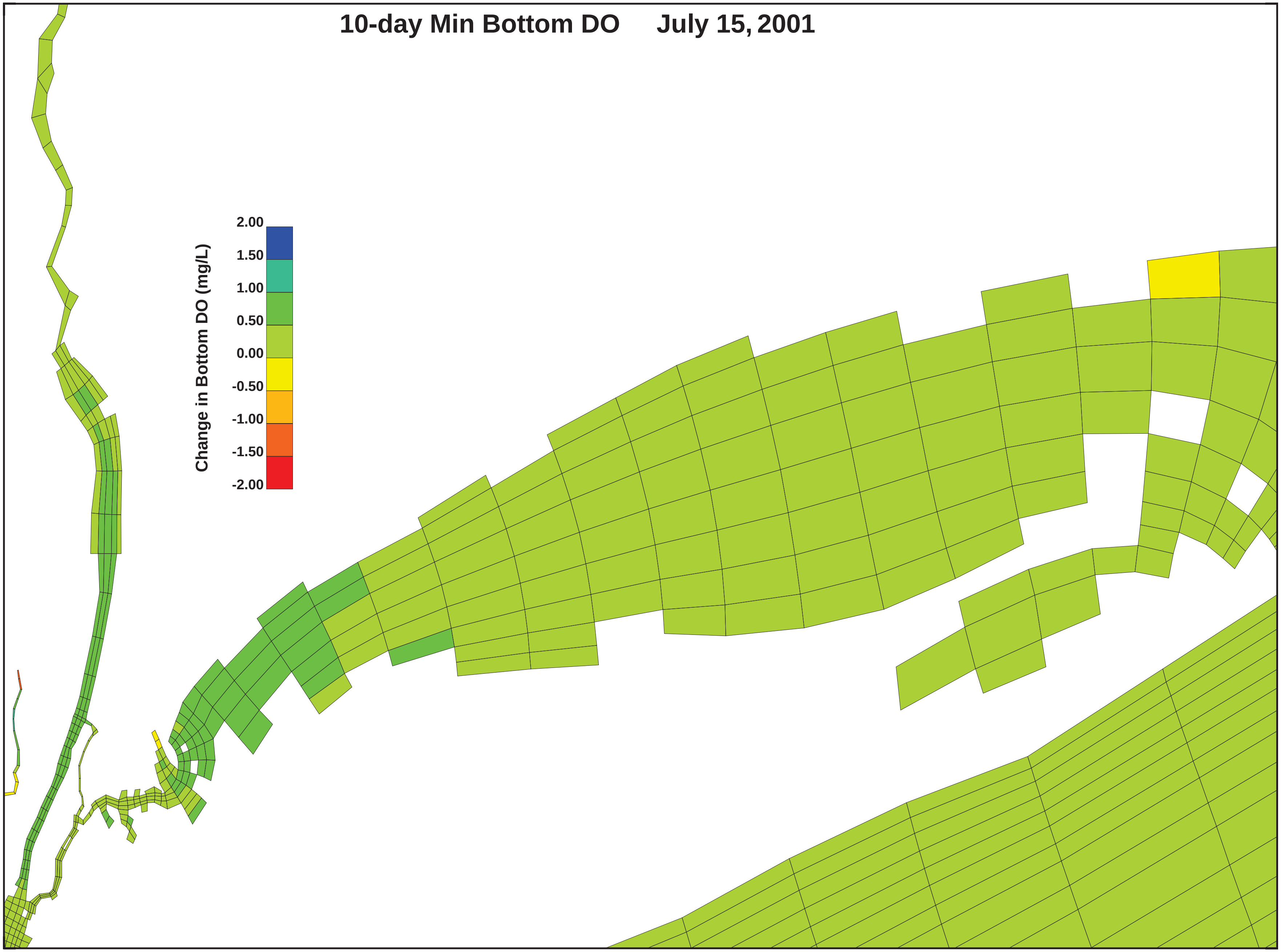
-2.00



10-day Min Bottom DO July 15, 2001
Change in Bottom DO (mg/L)
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1.50
1.00
0.50
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-0.50
-1.00
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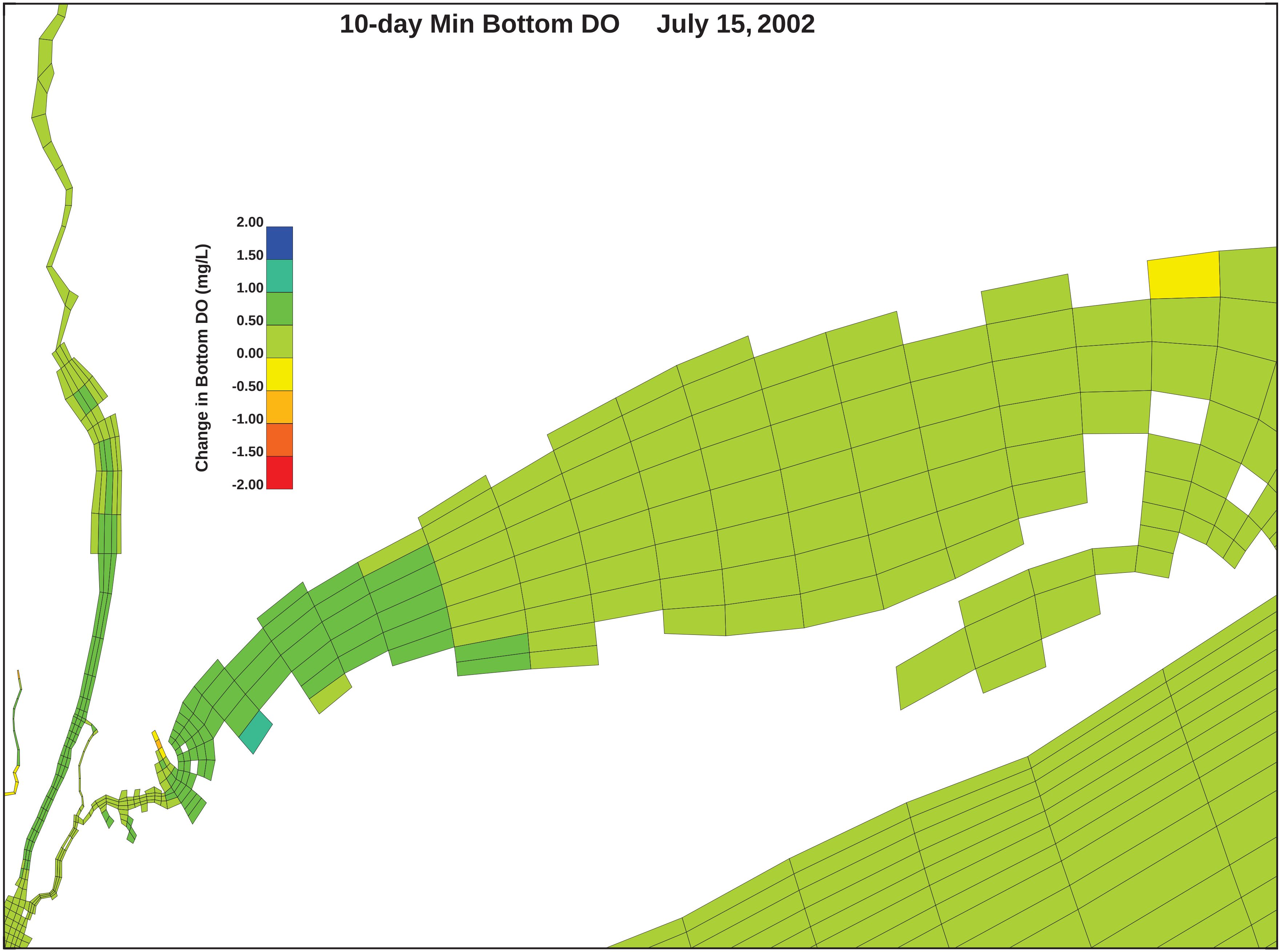
-2.00



10-day Min Bottom DO July 15, 2002
Change in Bottom DO (mg/L)
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1.50
1.00
0.50
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-0.50
-1.00
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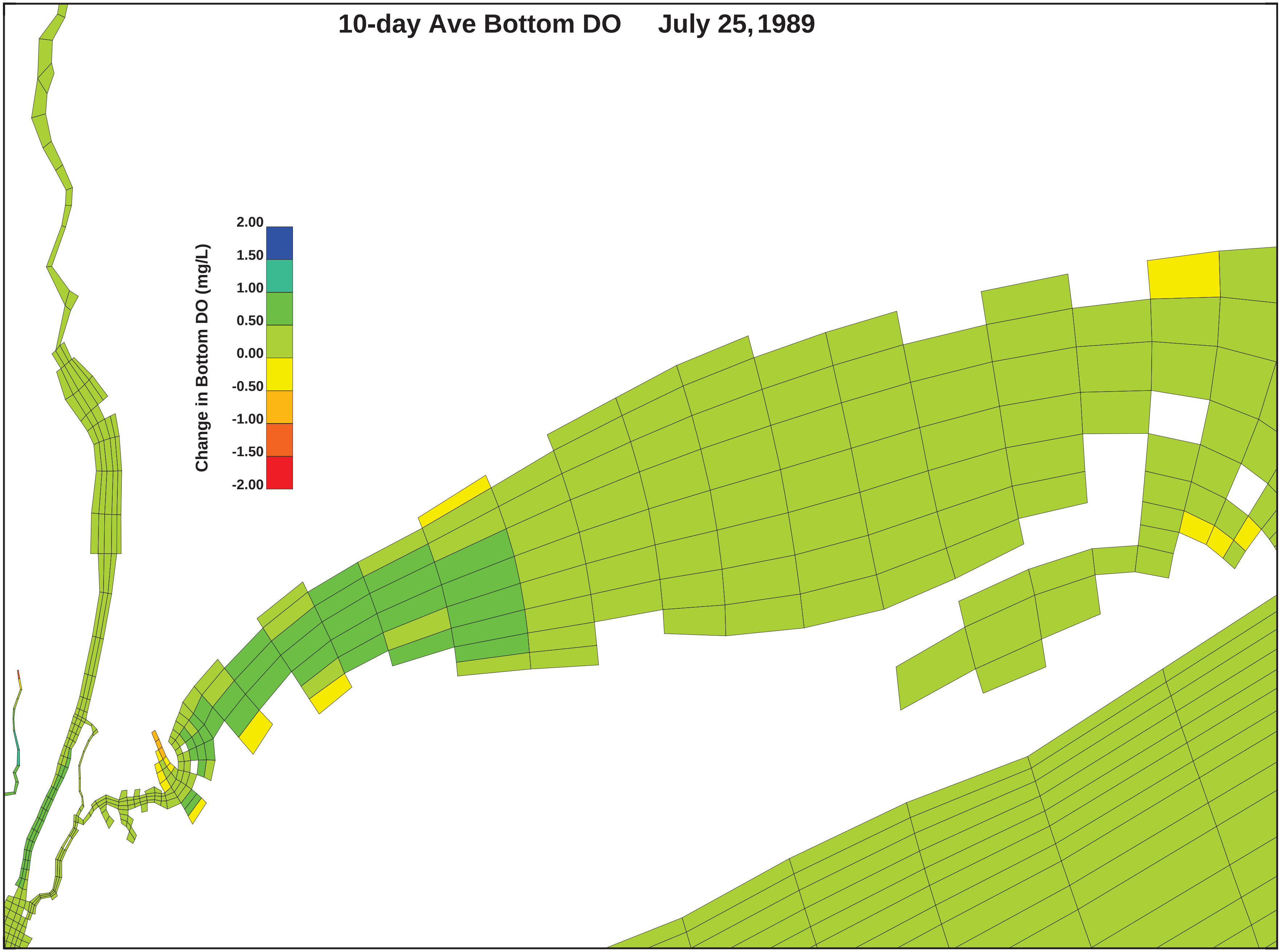
10-day Ave Bottom DO July 25,1989
Change in Bottom DO (mg/L)
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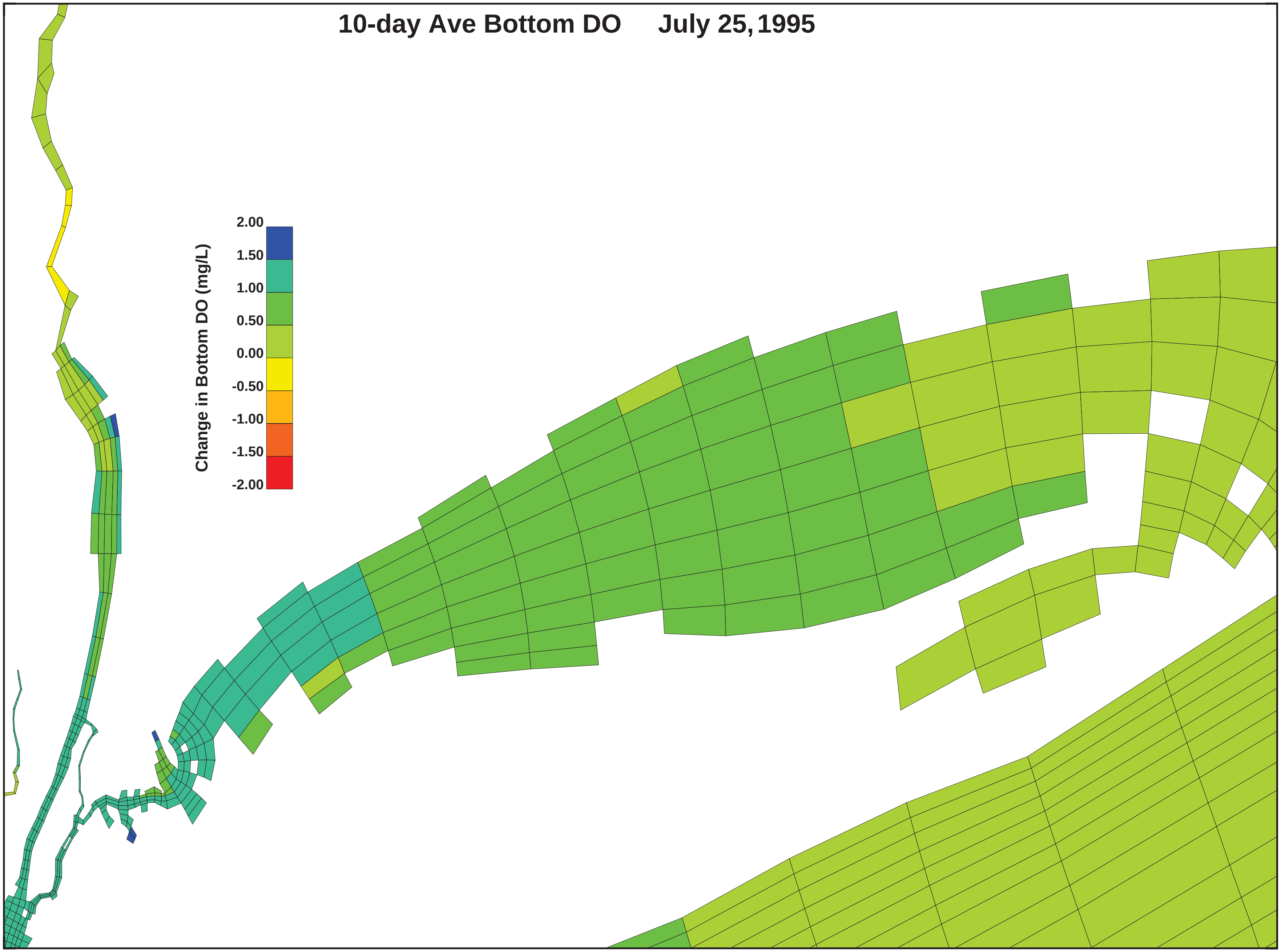
-2.00


10-day Ave Bottom DO July 25, 1995
Change in Bottom DO (mg/L)
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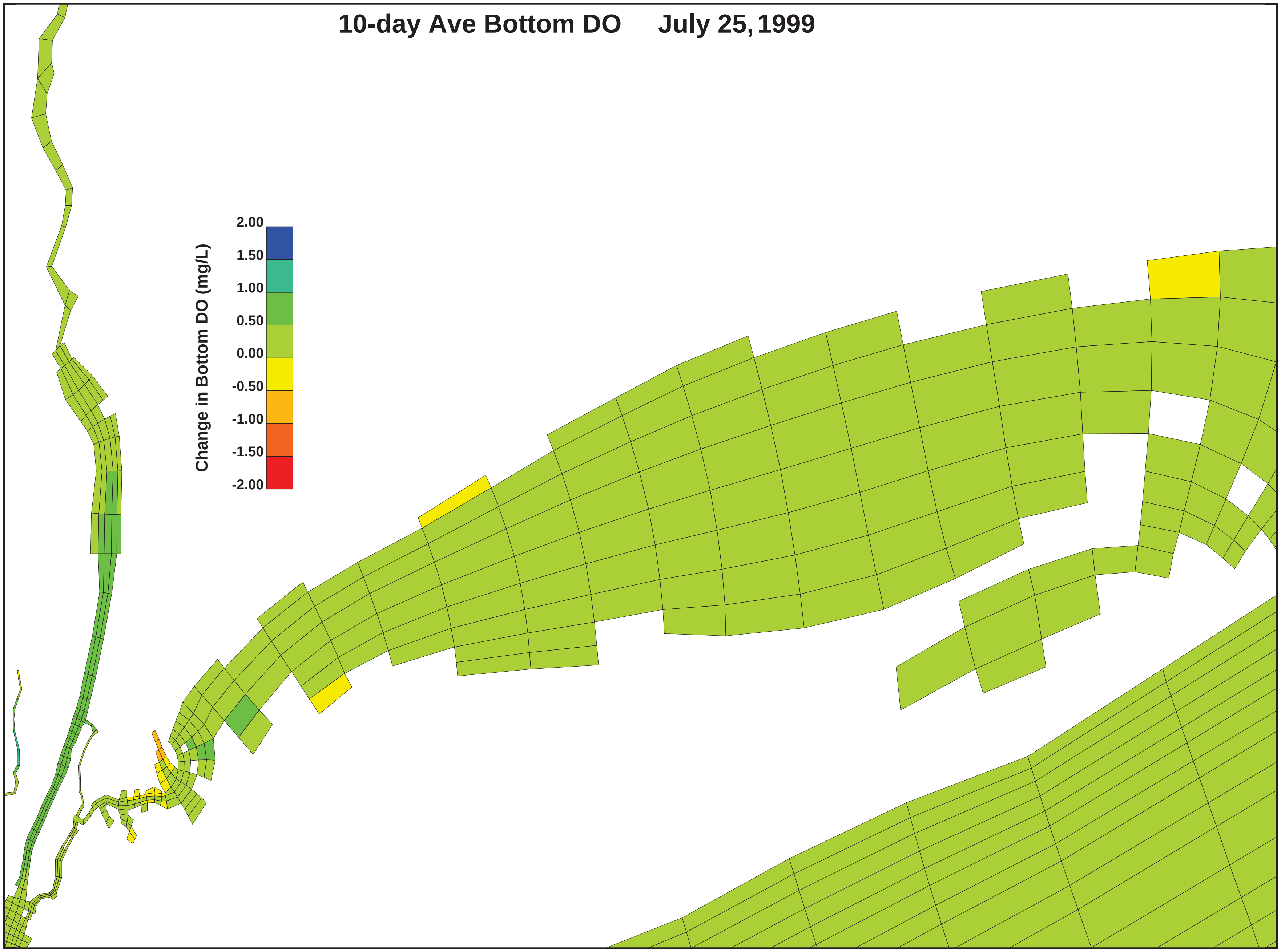
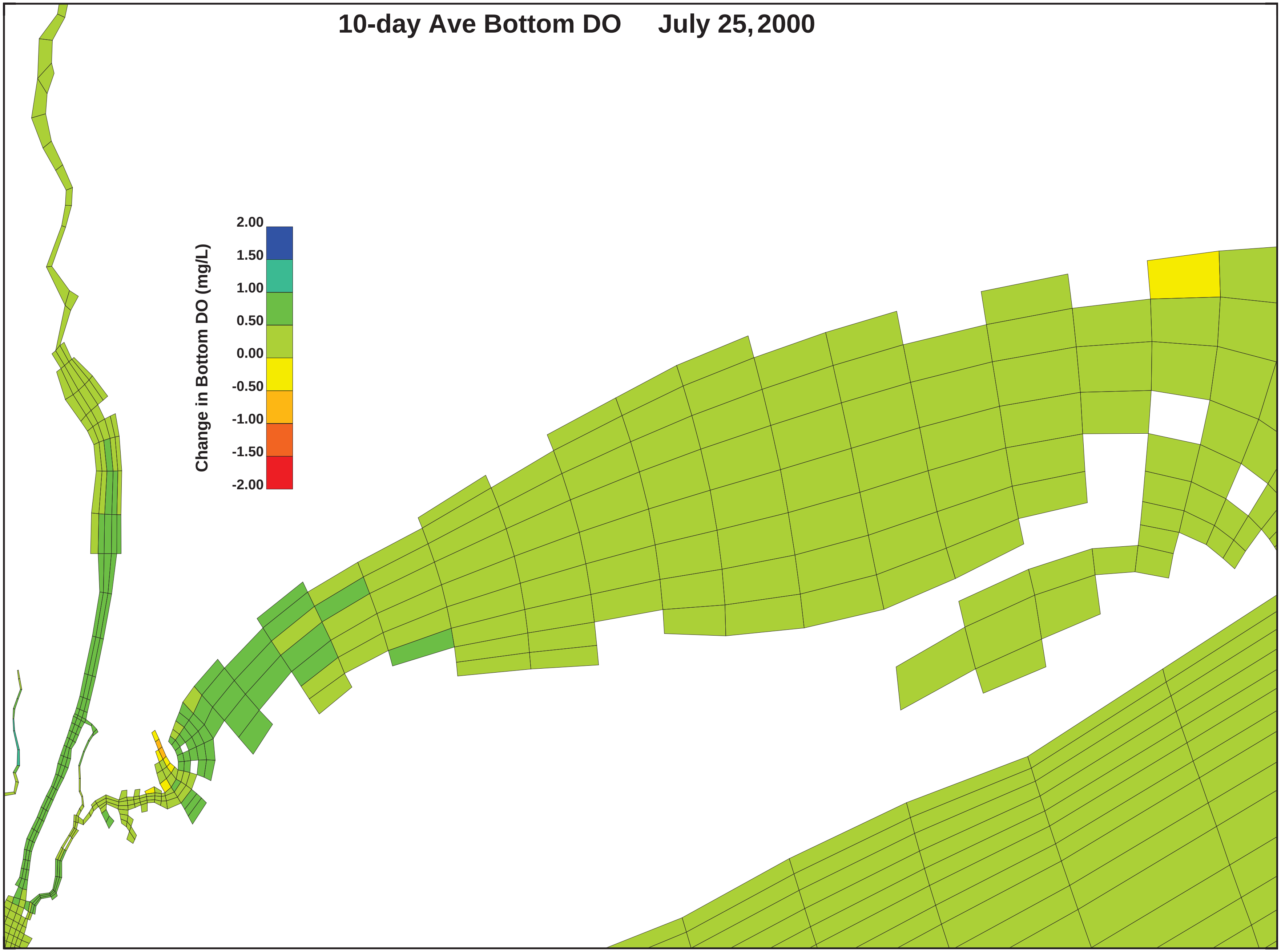
0.50
0.00
-0.50
-1.00
-1.50
-2.00

10-day Ave Bottom DO July 25, 1999




10-day Ave Bottom DO July 25, 2000
Change in Bottom DO (mg/L)
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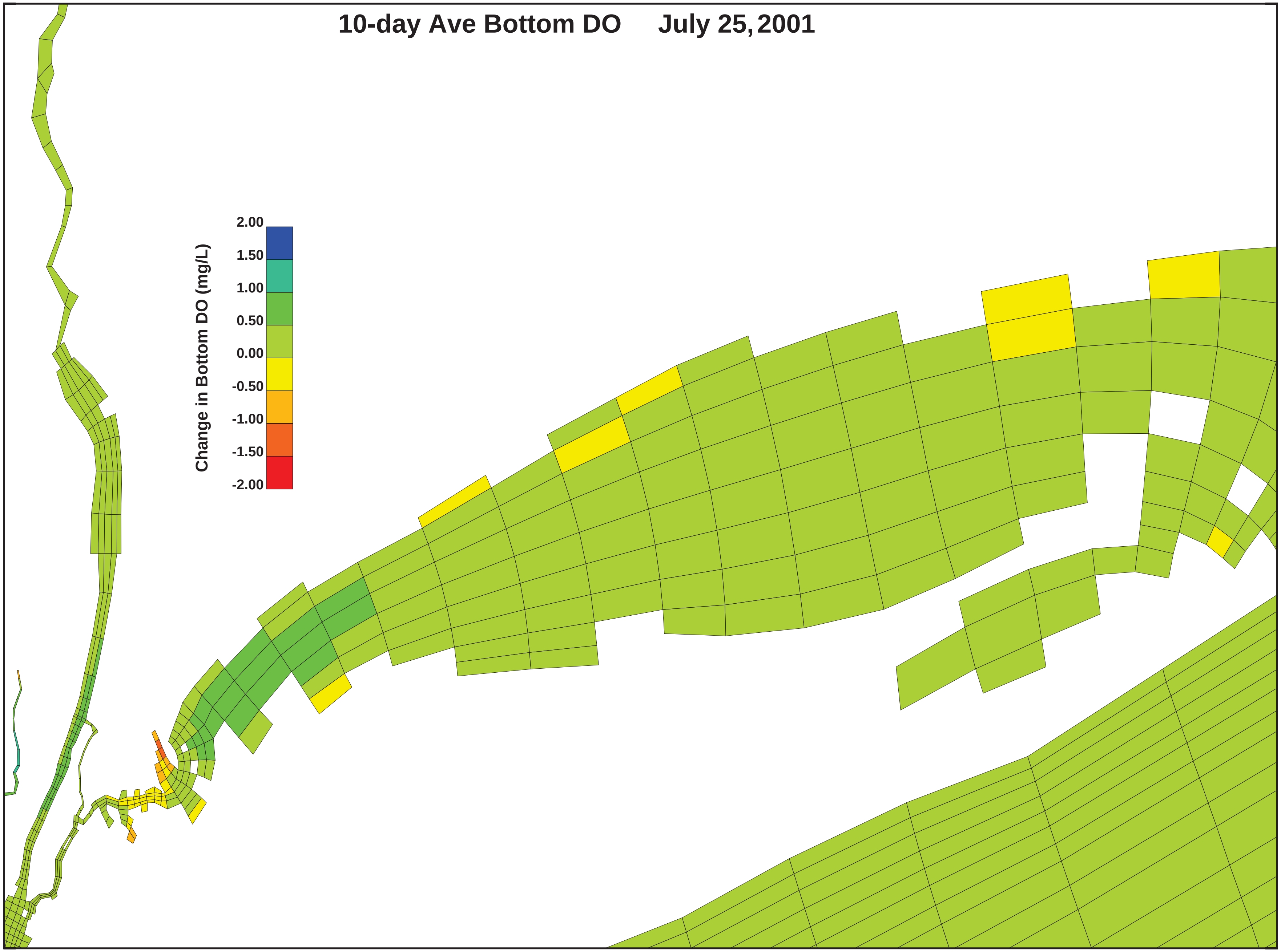
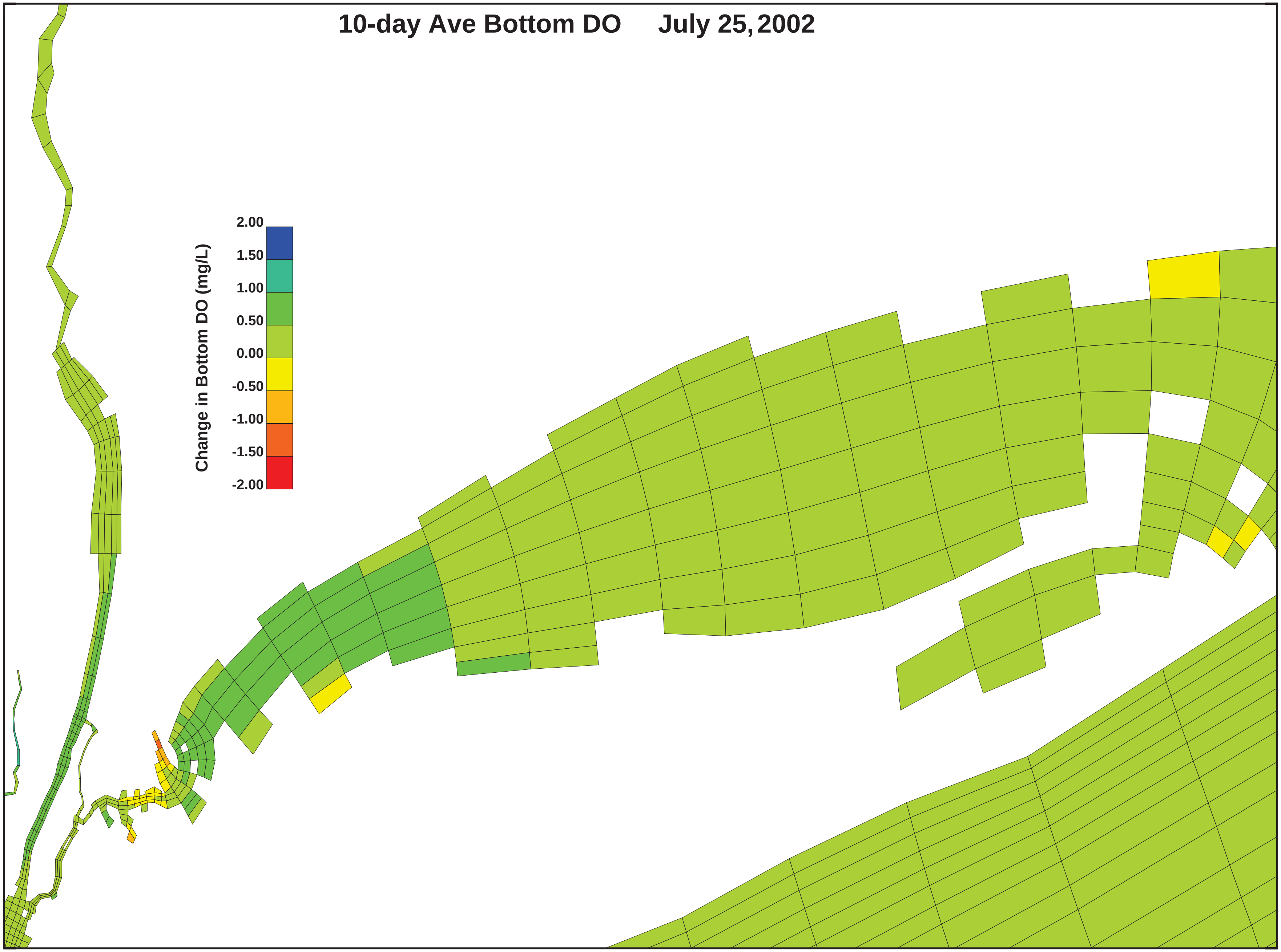
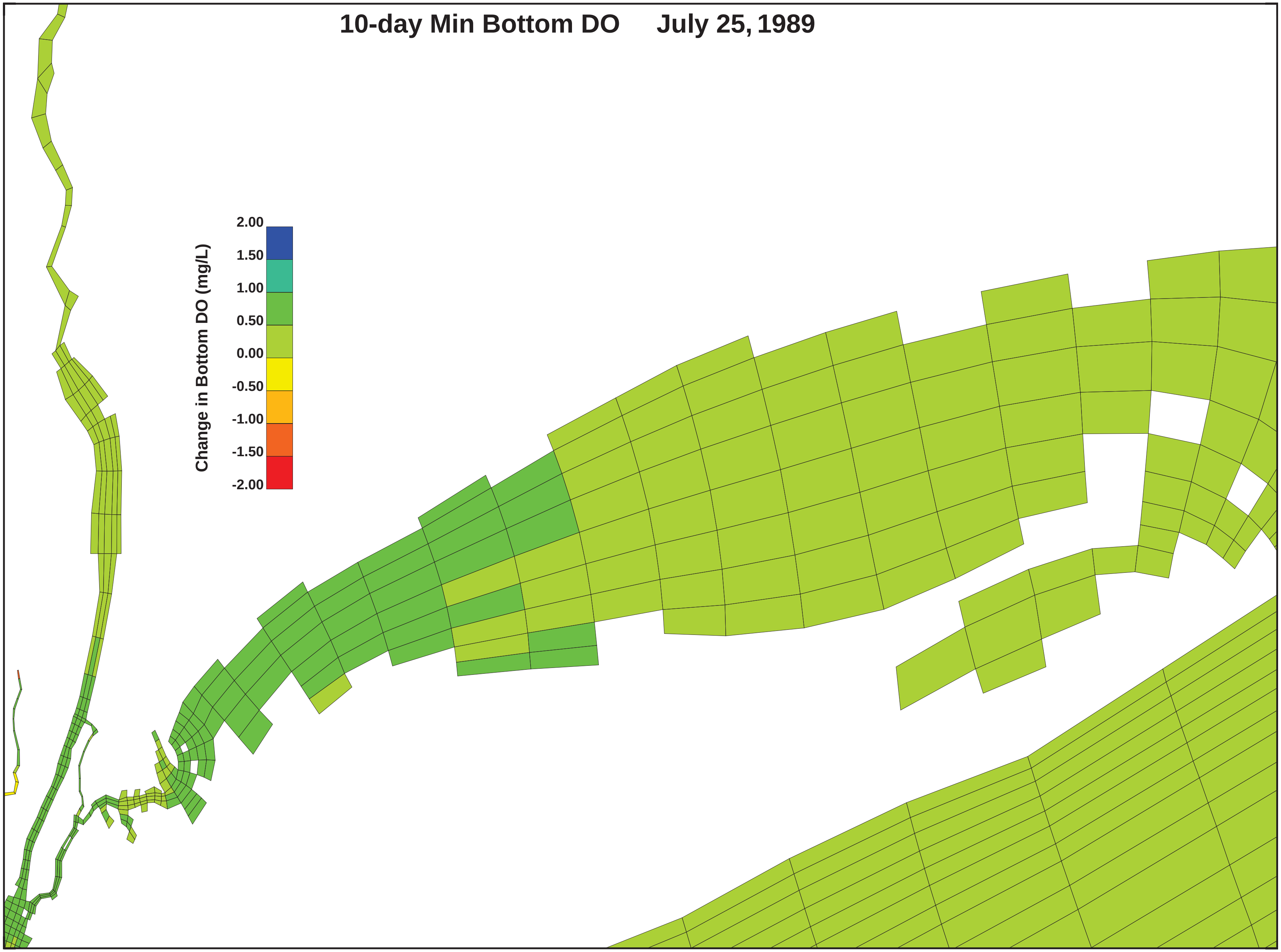
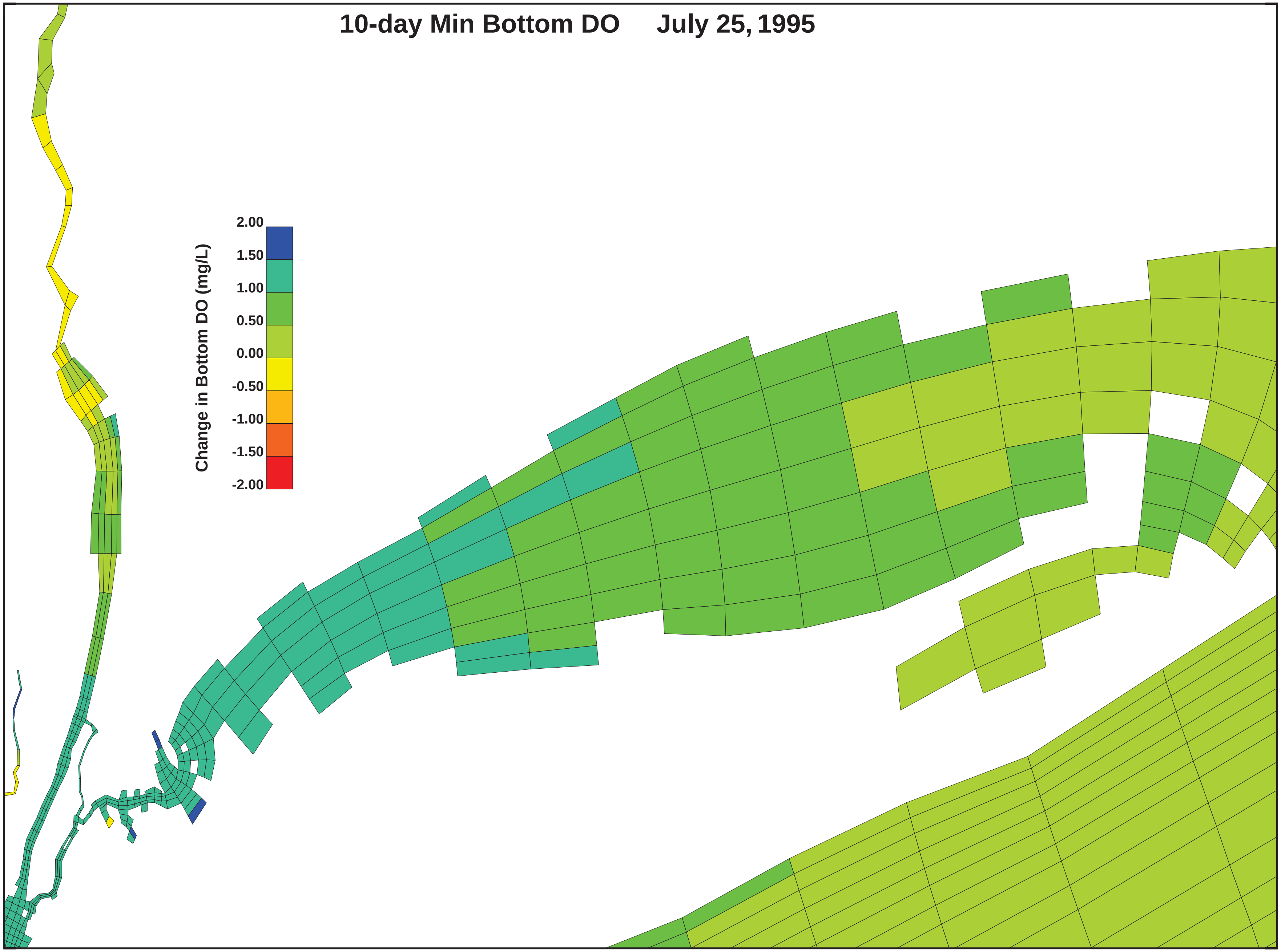
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00


10-day Ave Bottom DO July 25, 2001
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00

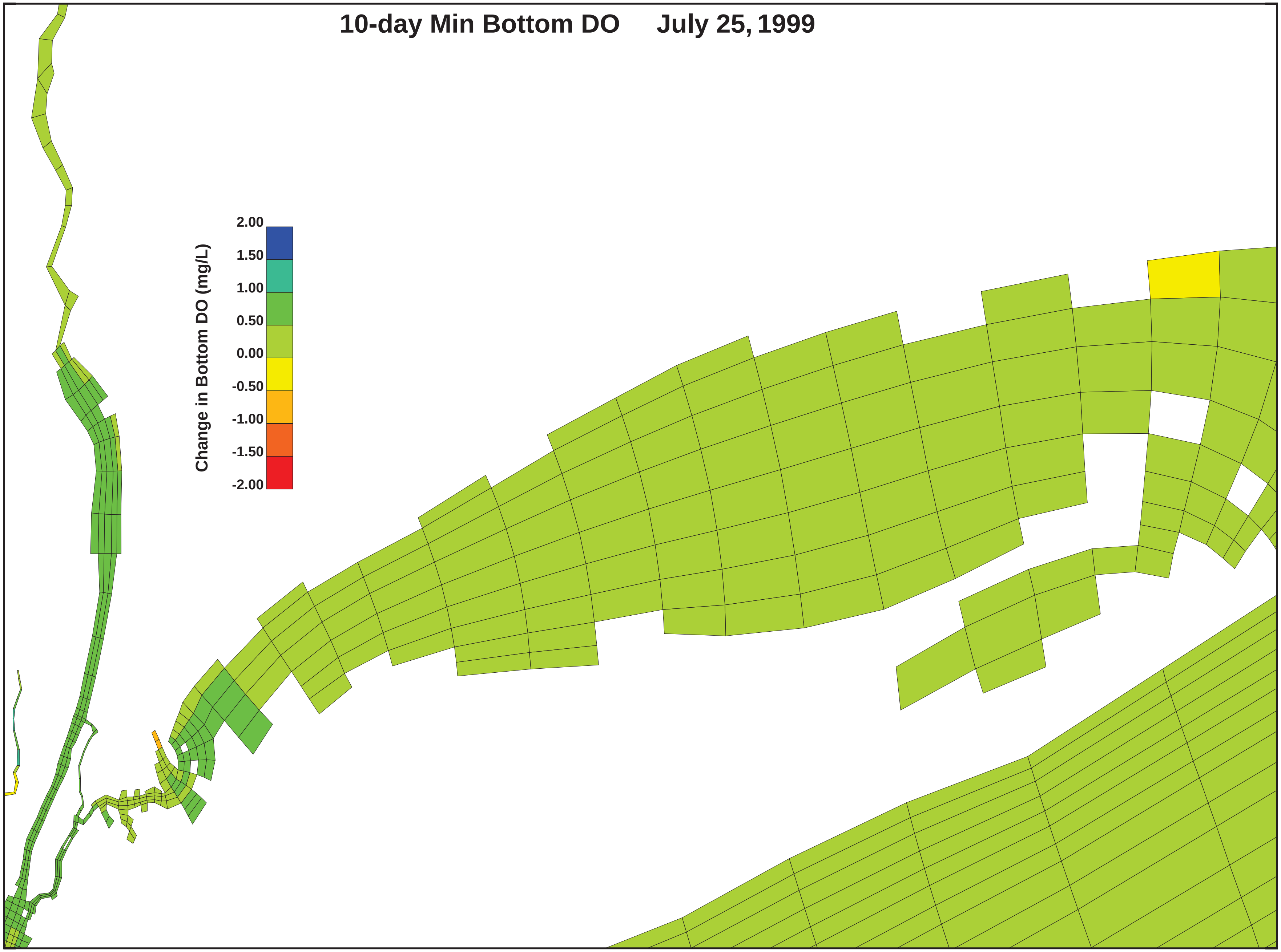


10-day Ave Bottom DO July 25, 2002
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00


10-day Min Bottom DO July 25, 1989
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00



10-day Min Bottom DO July 25, 1995
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00


10-day Min Bottom DO July 25, 1999
Change in Bottom DO (mg/L)
2.00
1.50
1.00
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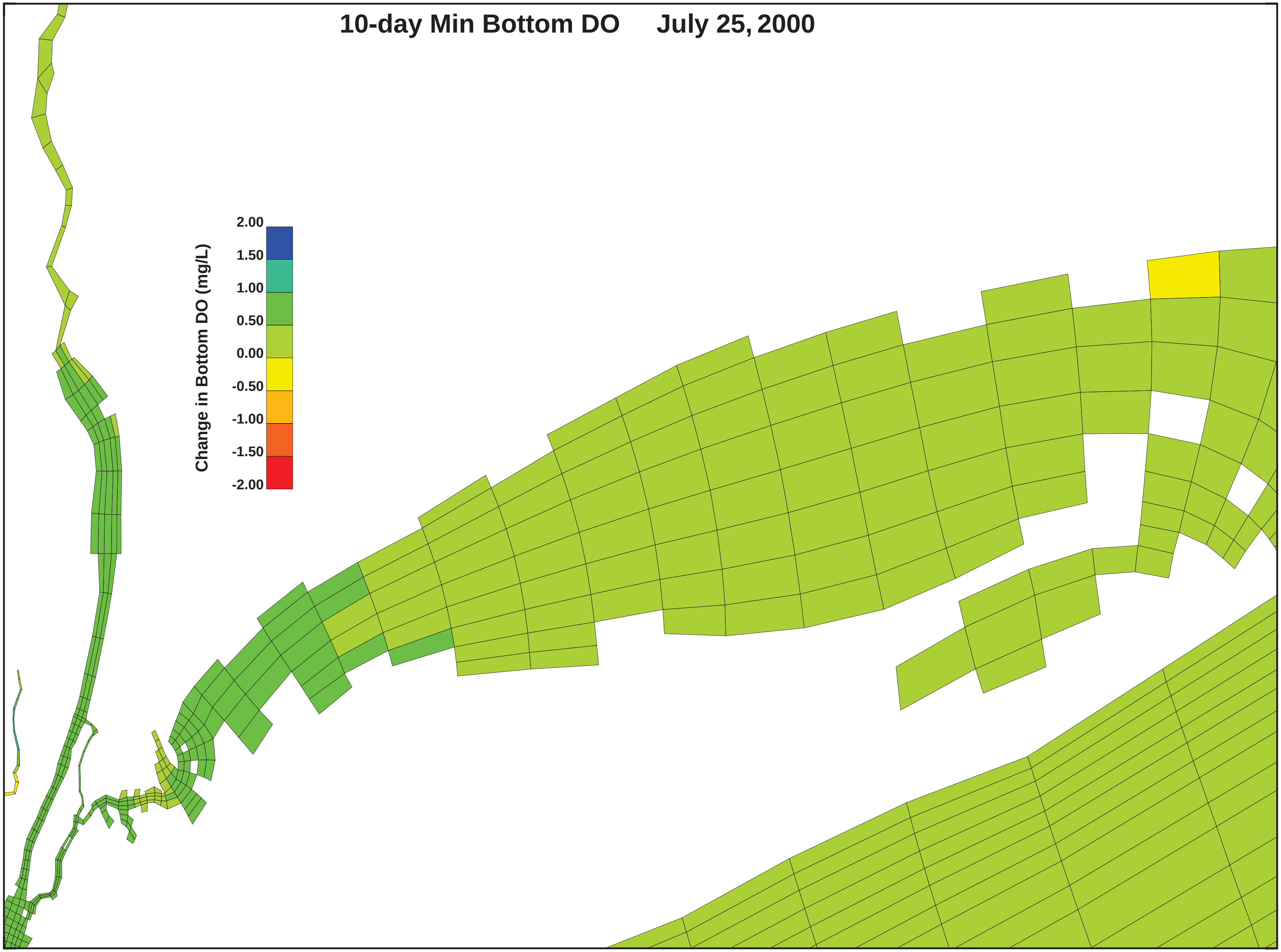
0.00
-0.50
-1.00
-1.50
-2.00


10-day Min Bottom DO July 25, 2000
Change in Bottom DO (mg/L)
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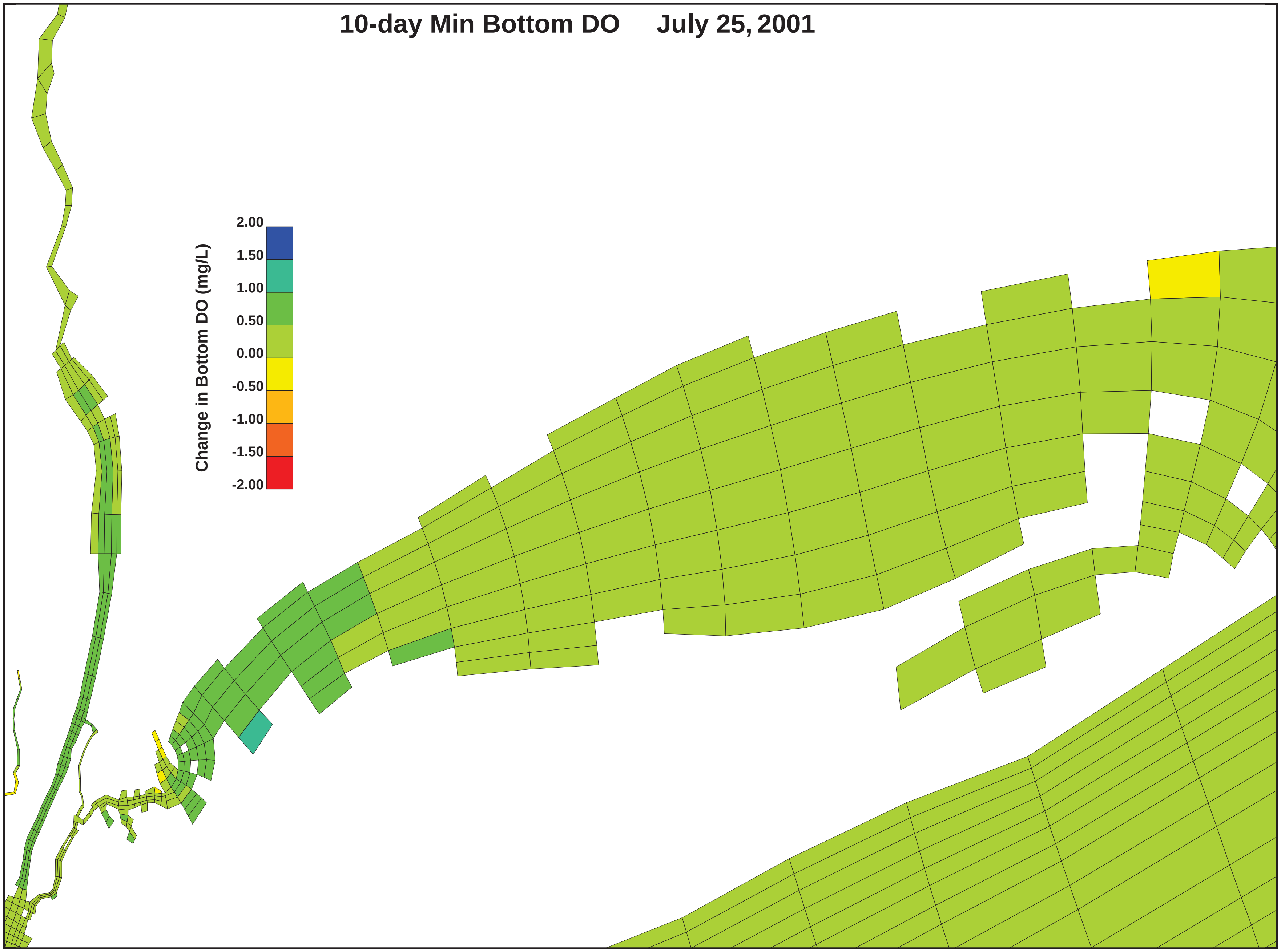
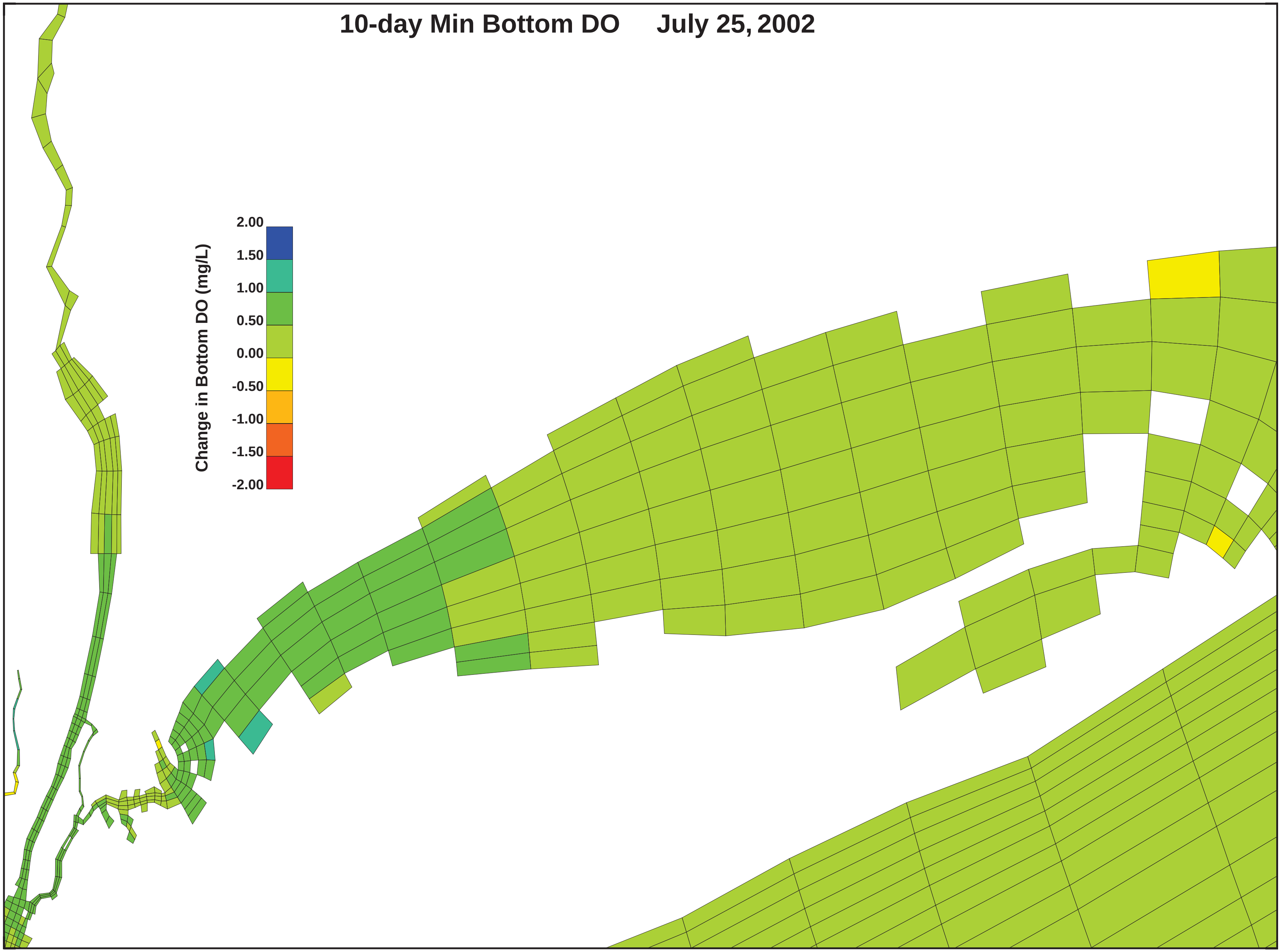
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00


10-day Min Bottom DO July 25, 2001
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00


10-day Min Bottom DO July 25, 2002
Change in Bottom DO (mg/L)
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1.00
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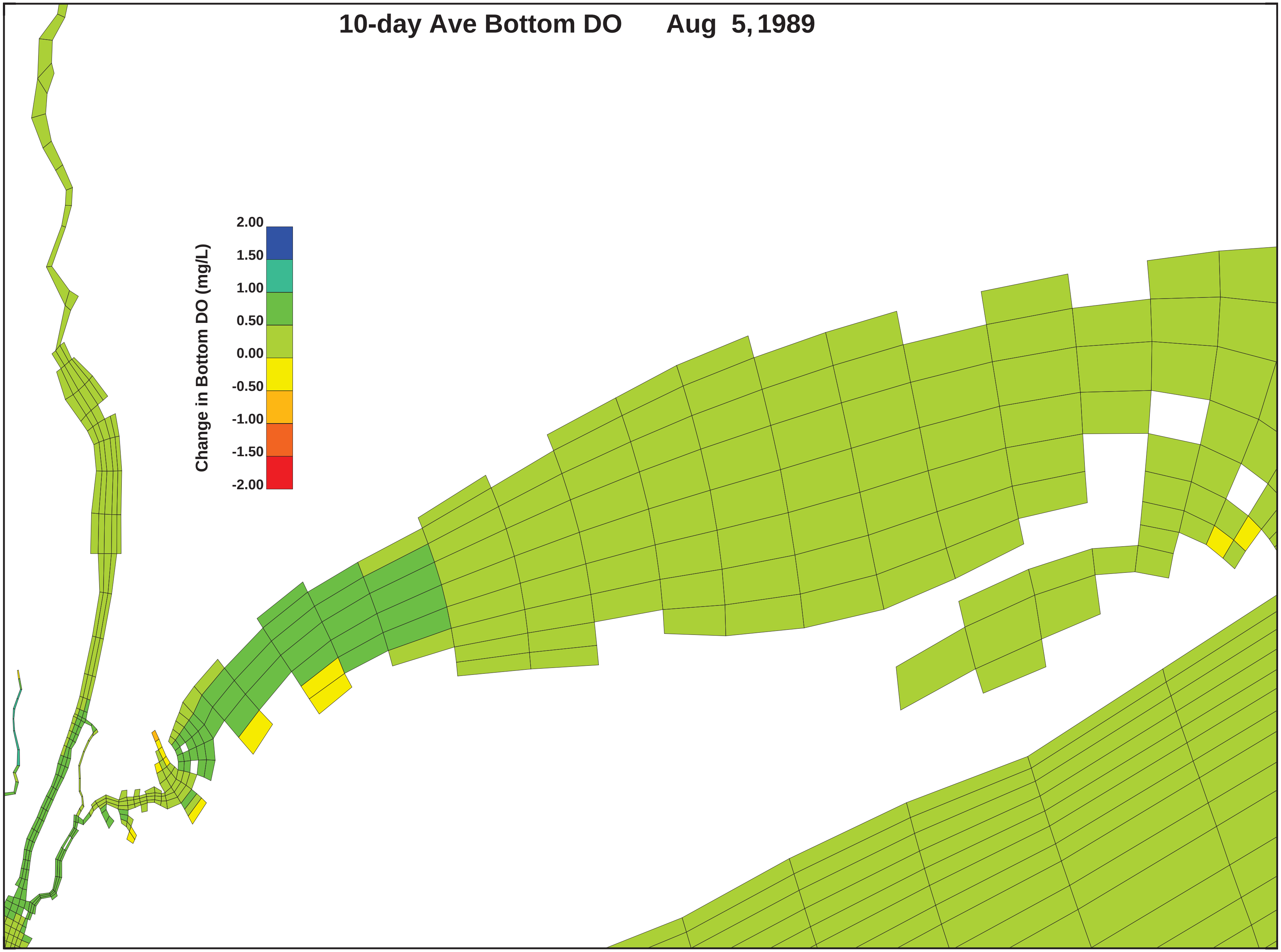
0.00
-0.50
-1.00
-1.50
-2.00


10-day Ave Bottom DO Aug 5, 1989
Change in Bottom DO (mg/L)
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1.00
0.50
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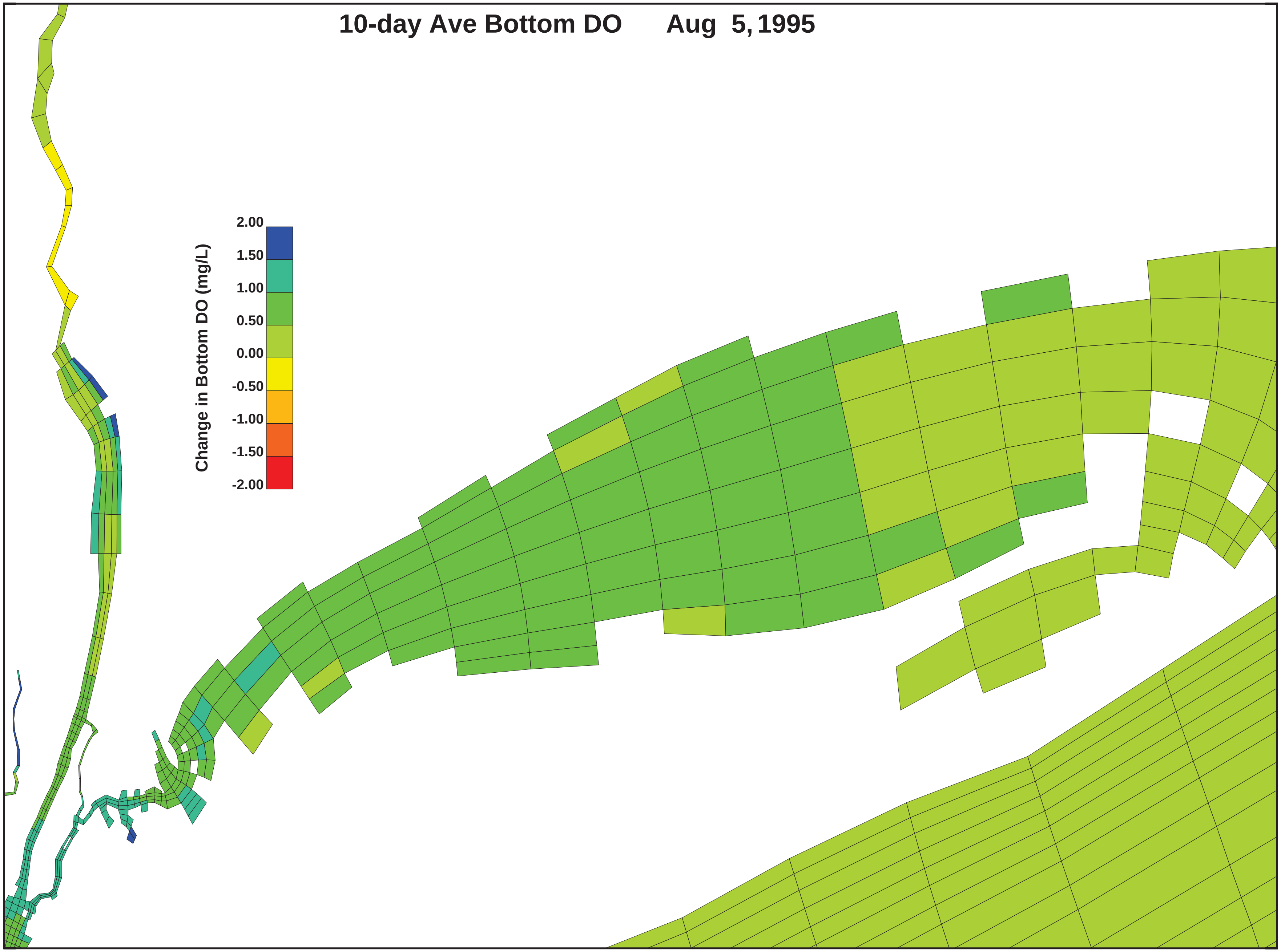
-0.50
-1.00
-1.50
-2.00


10-day Ave Bottom DO Aug 5, 1995
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
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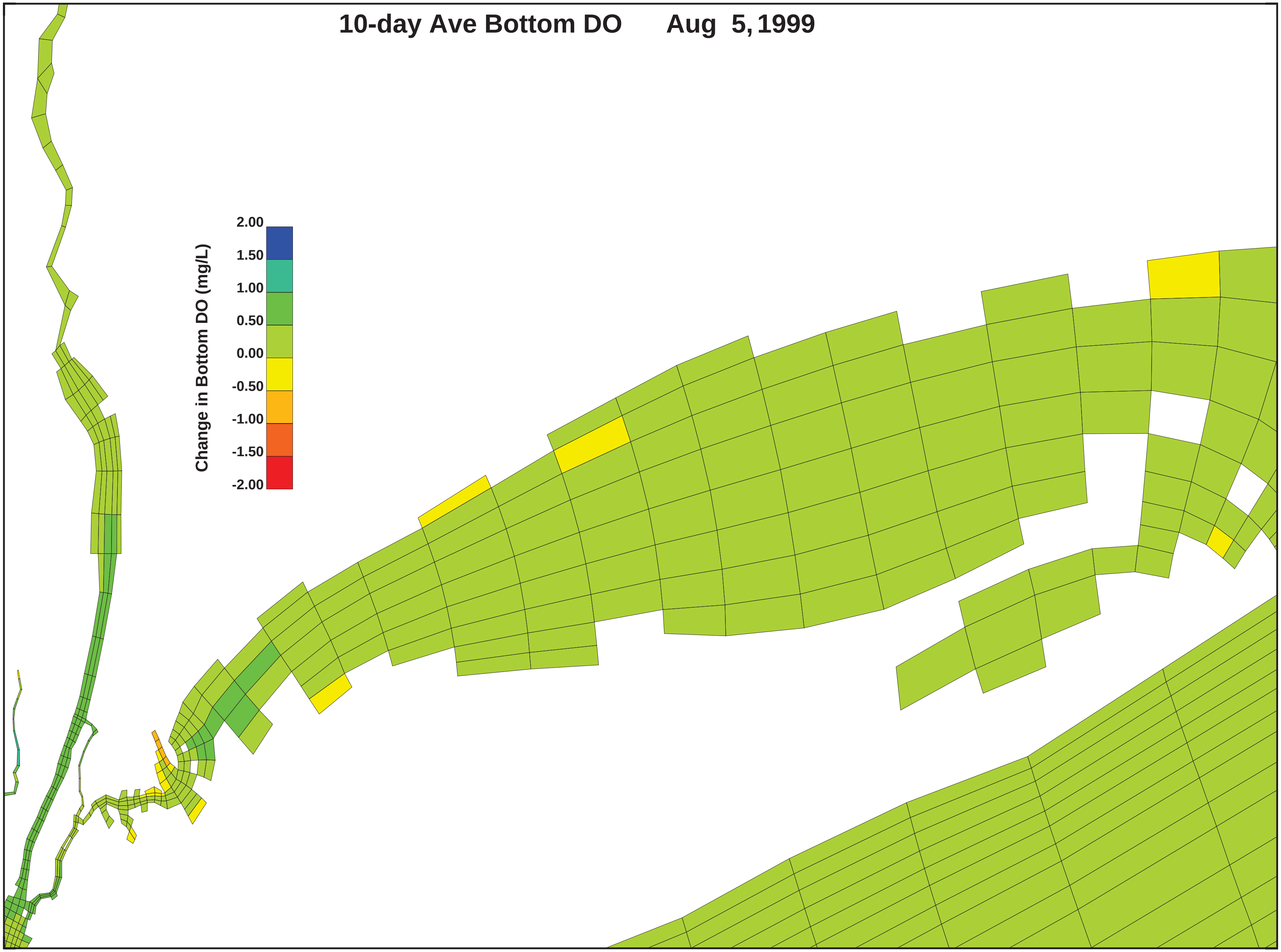
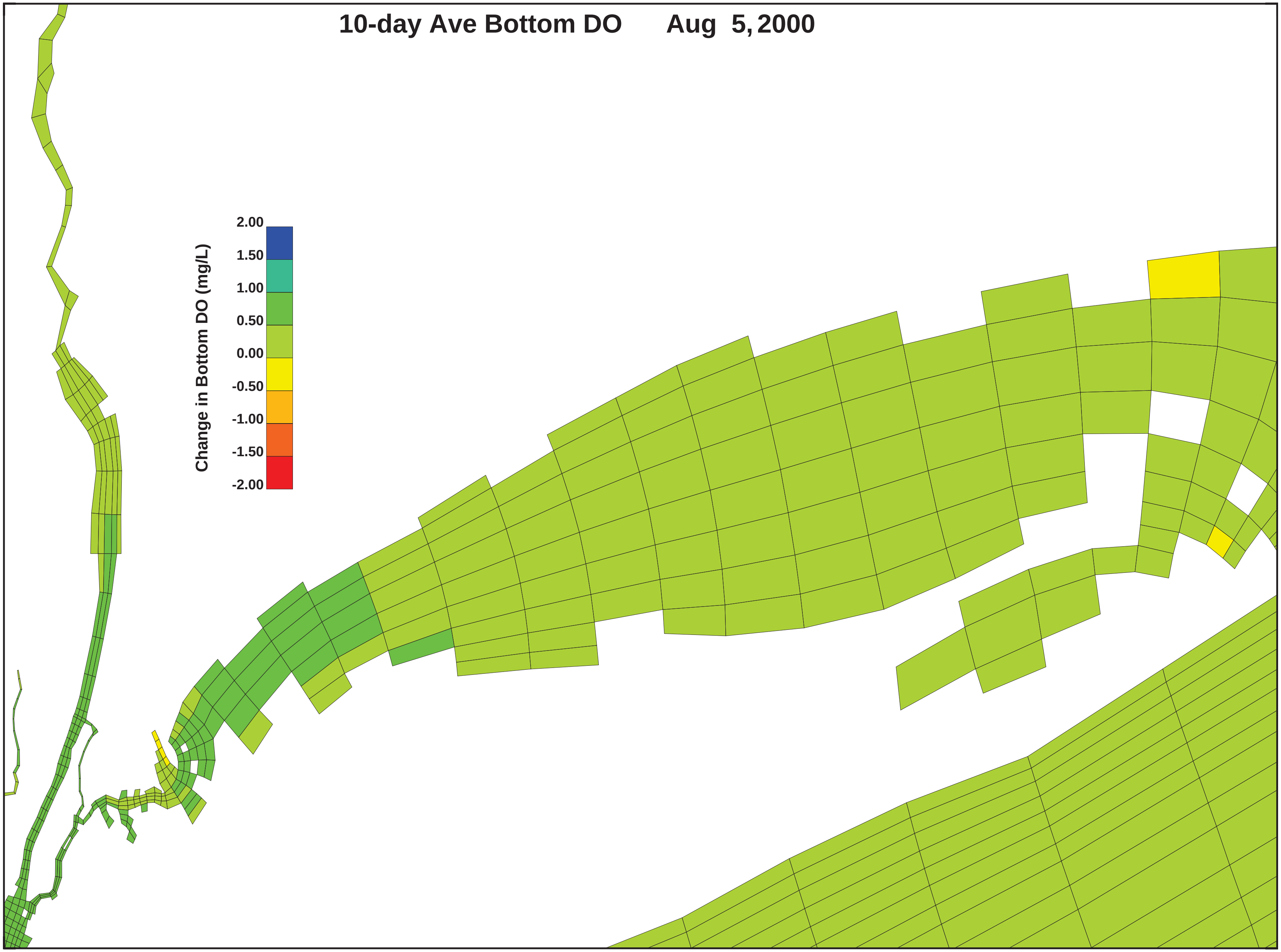
-0.50
-1.00
-1.50
-2.00


10-day Ave Bottom DO Aug 5, 1999
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00


10-day Ave Bottom DO Aug 5, 2000
Change in Bottom DO (mg/L)
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1.00
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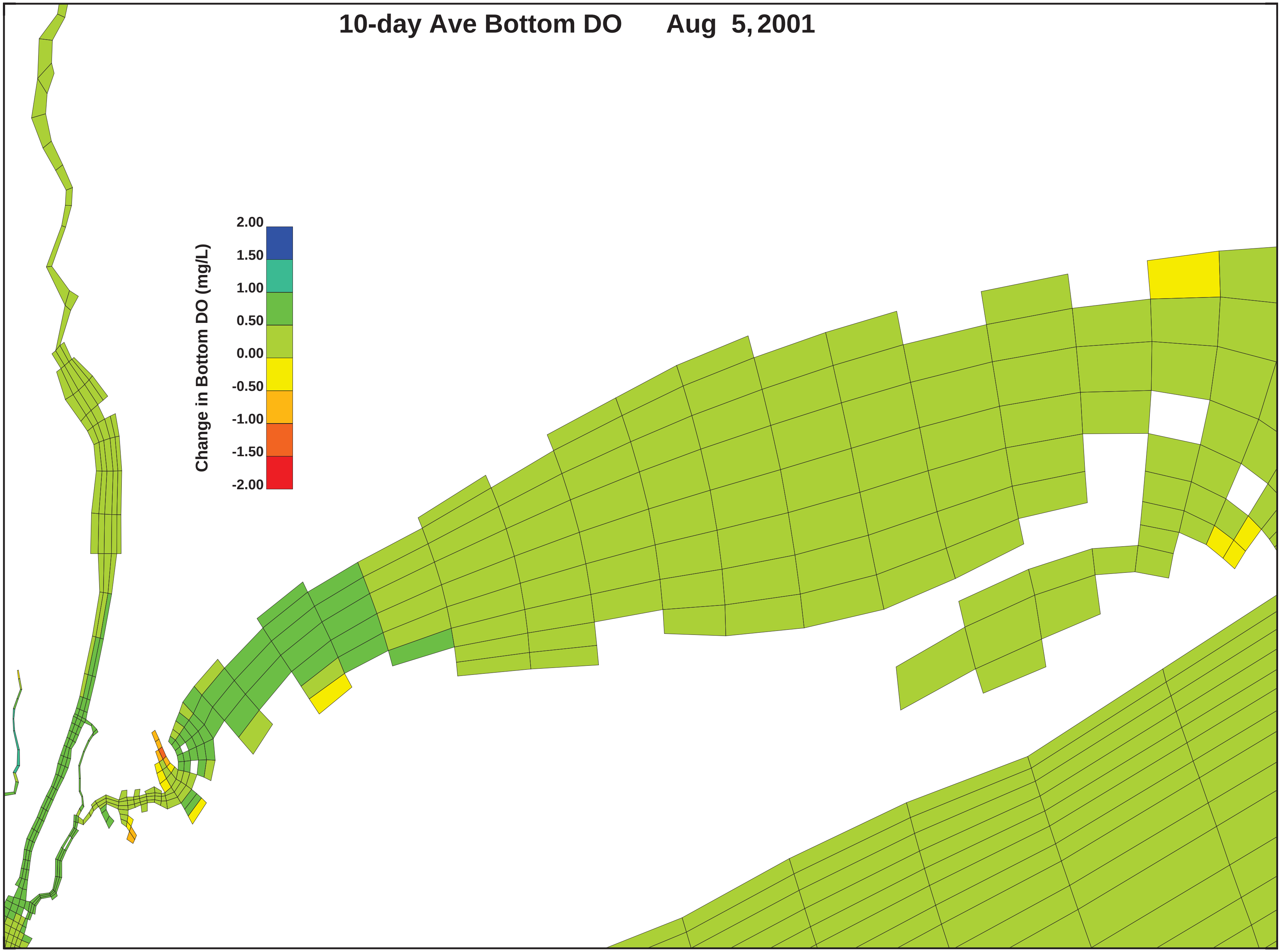
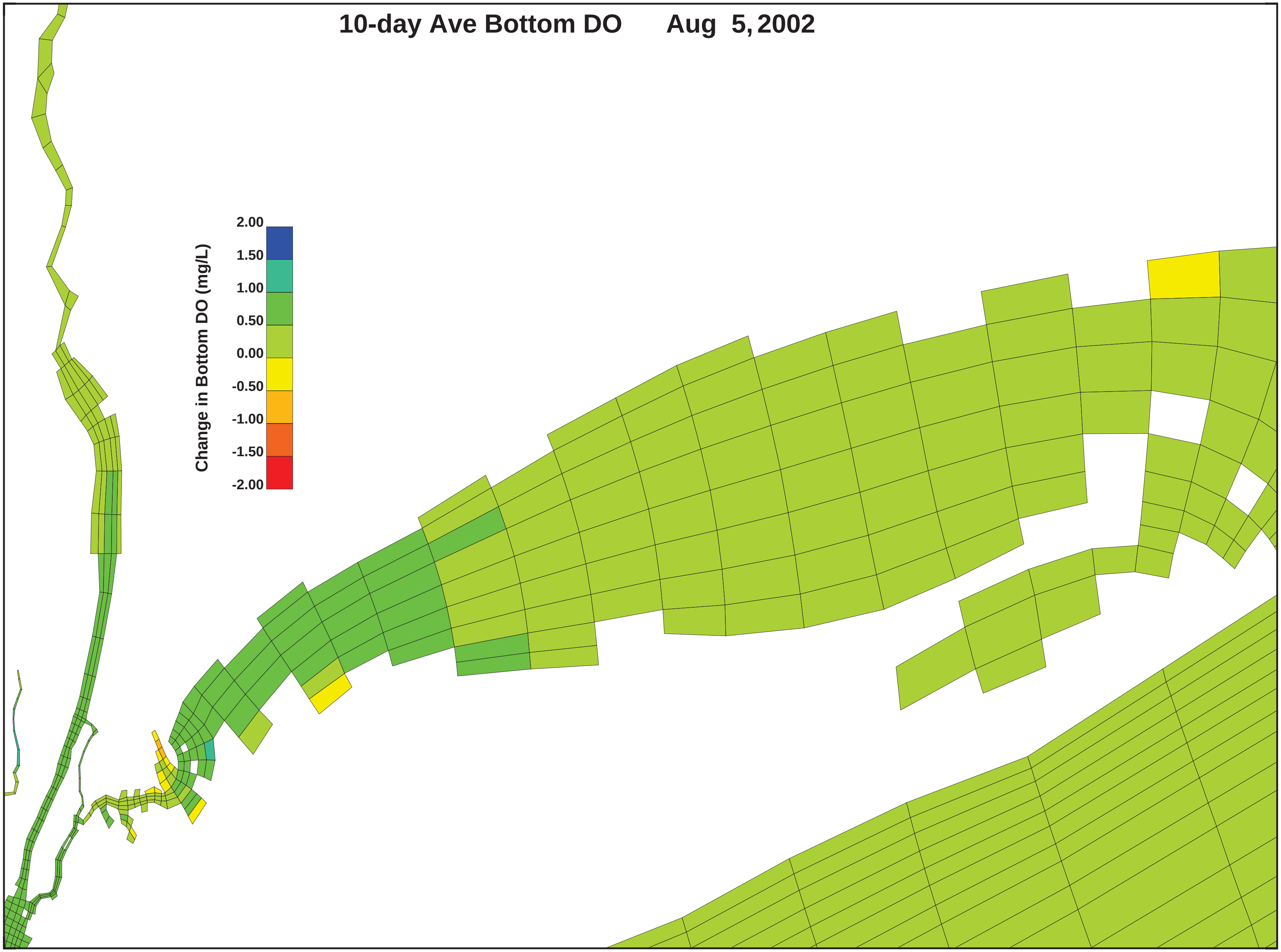
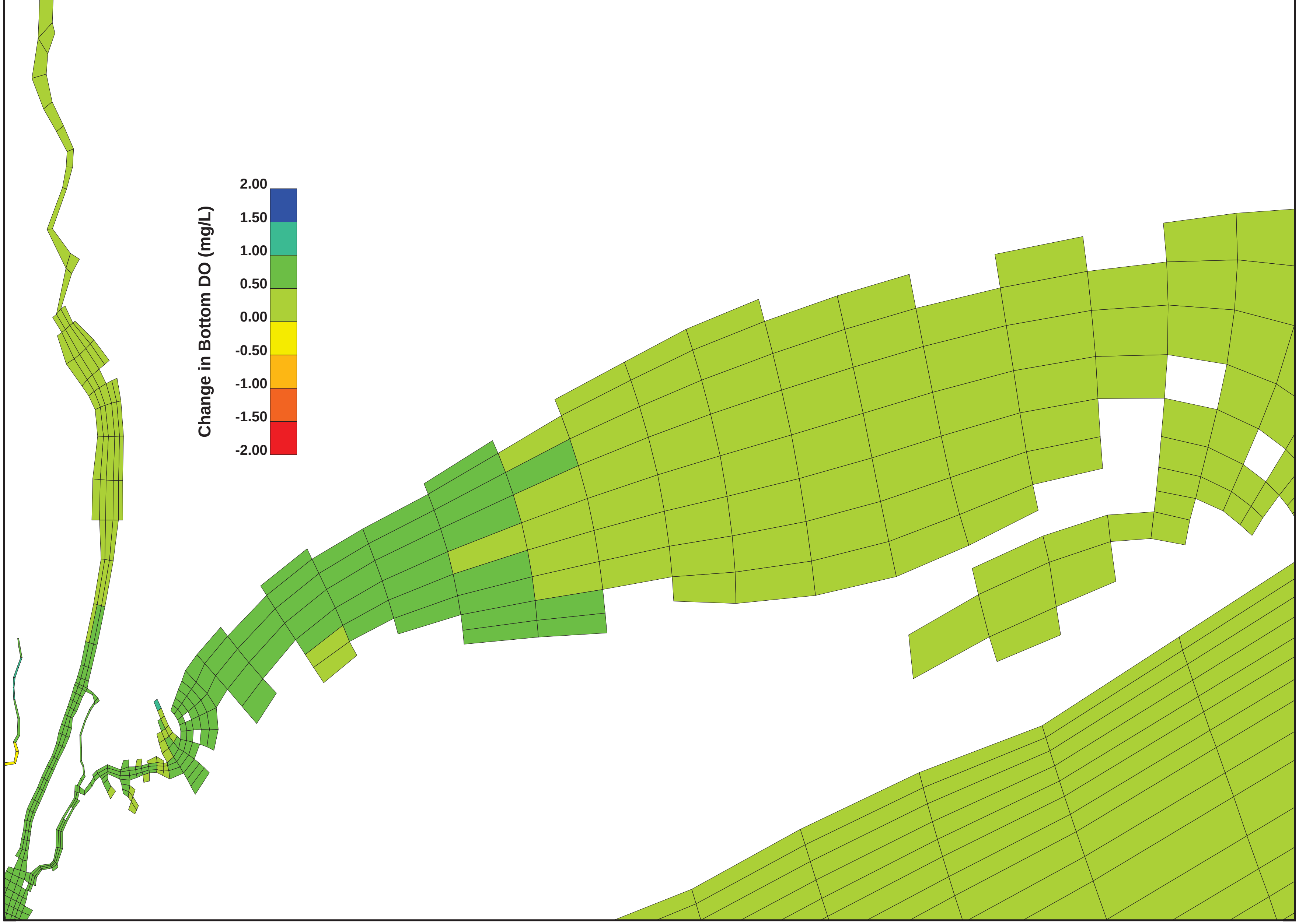
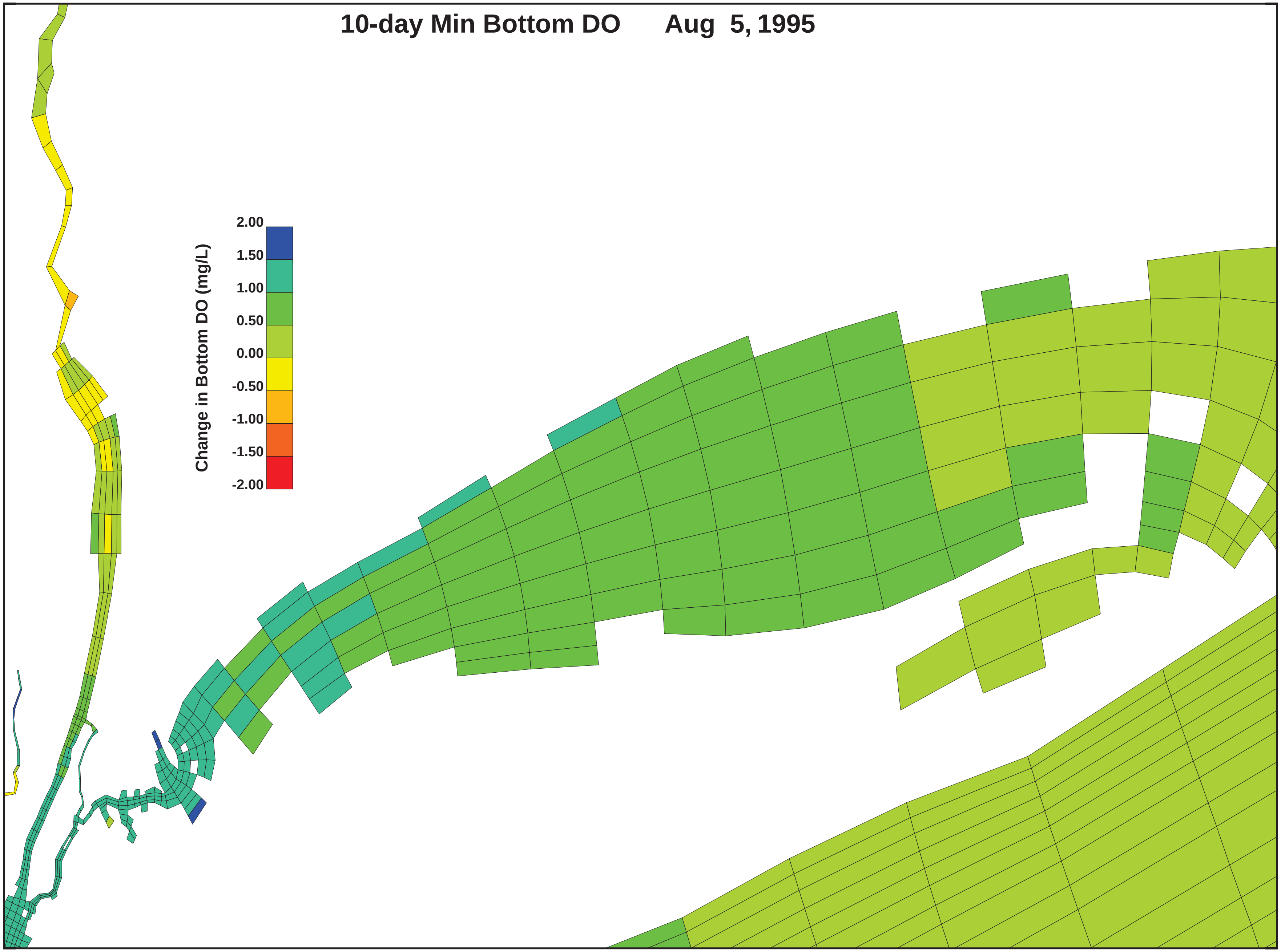
0.00
-0.50
-1.00
-1.50
-2.00


10-day Ave Bottom DO Aug 5, 2001
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00

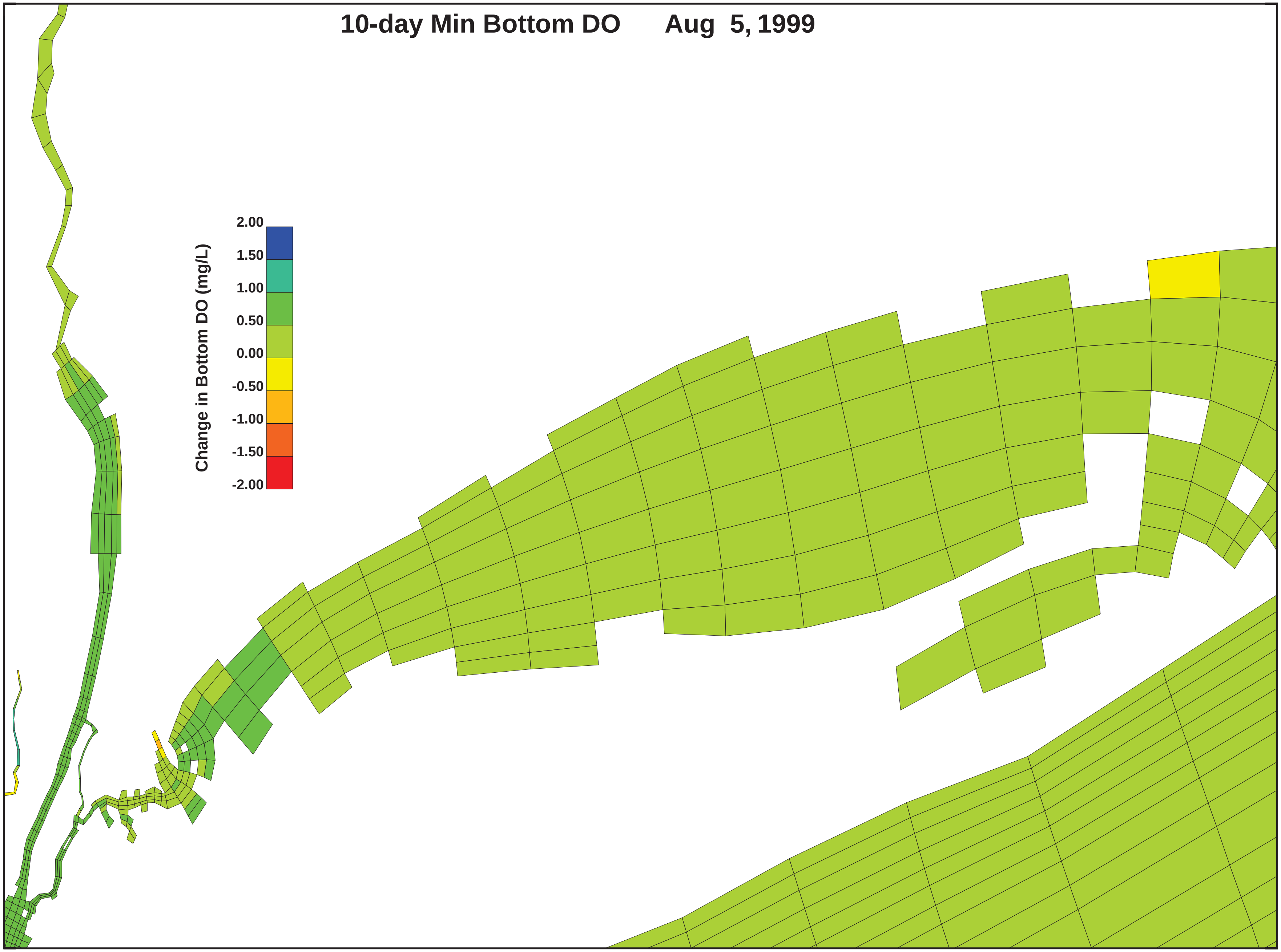

10-day Ave Bottom DO Aug 5, 2002
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00

10-day Min Bottom DO Aug 5, 1989

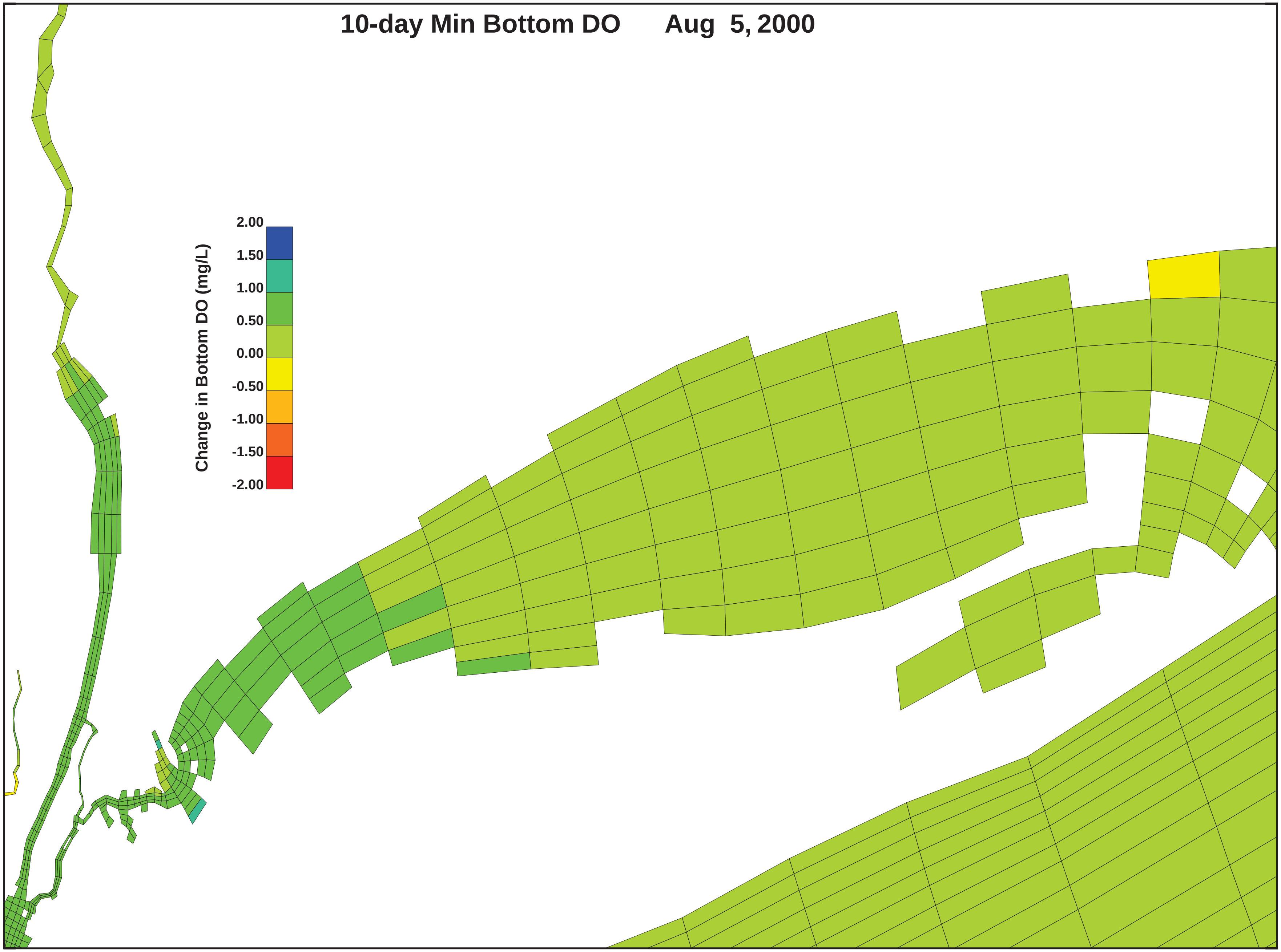




10-day Min Bottom DO Aug 5, 1995
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00



10-day Min Bottom DO Aug 5, 1999
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00


10-day Min Bottom DO Aug 5, 2000
Change in Bottom DO (mg/L)
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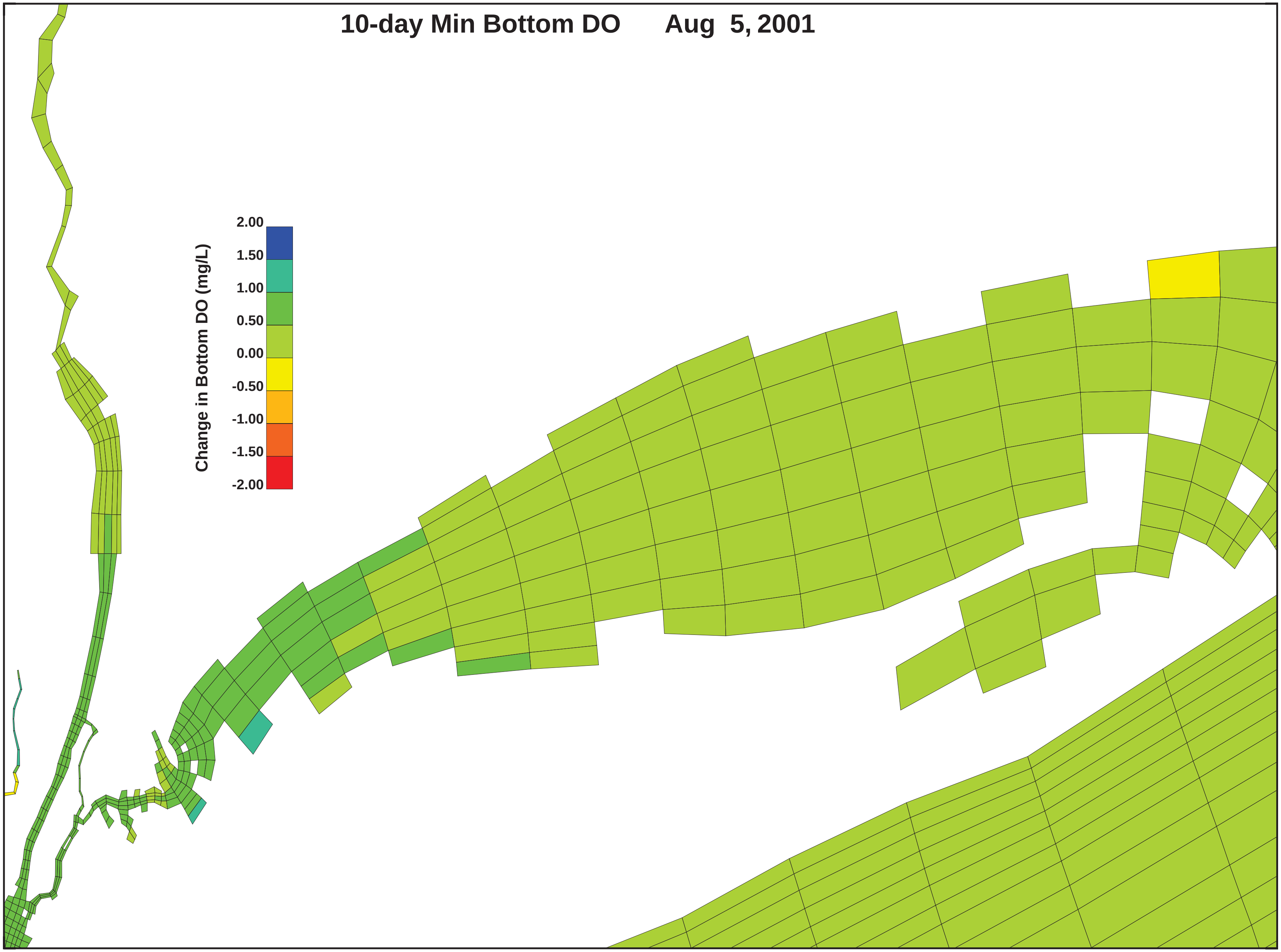
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Change in Bottom DO (mg/L)
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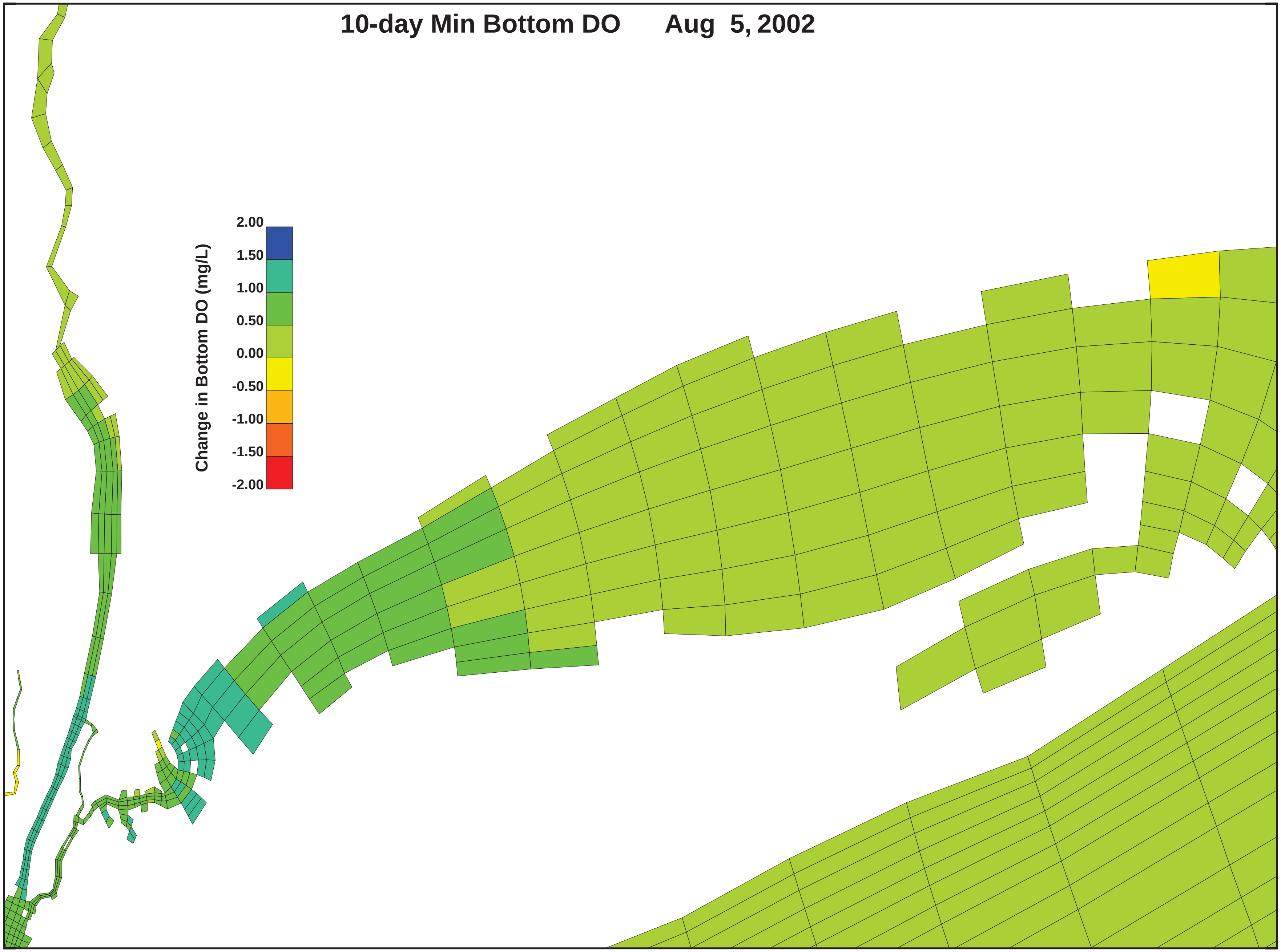

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Change in Bottom DO (mg/L)
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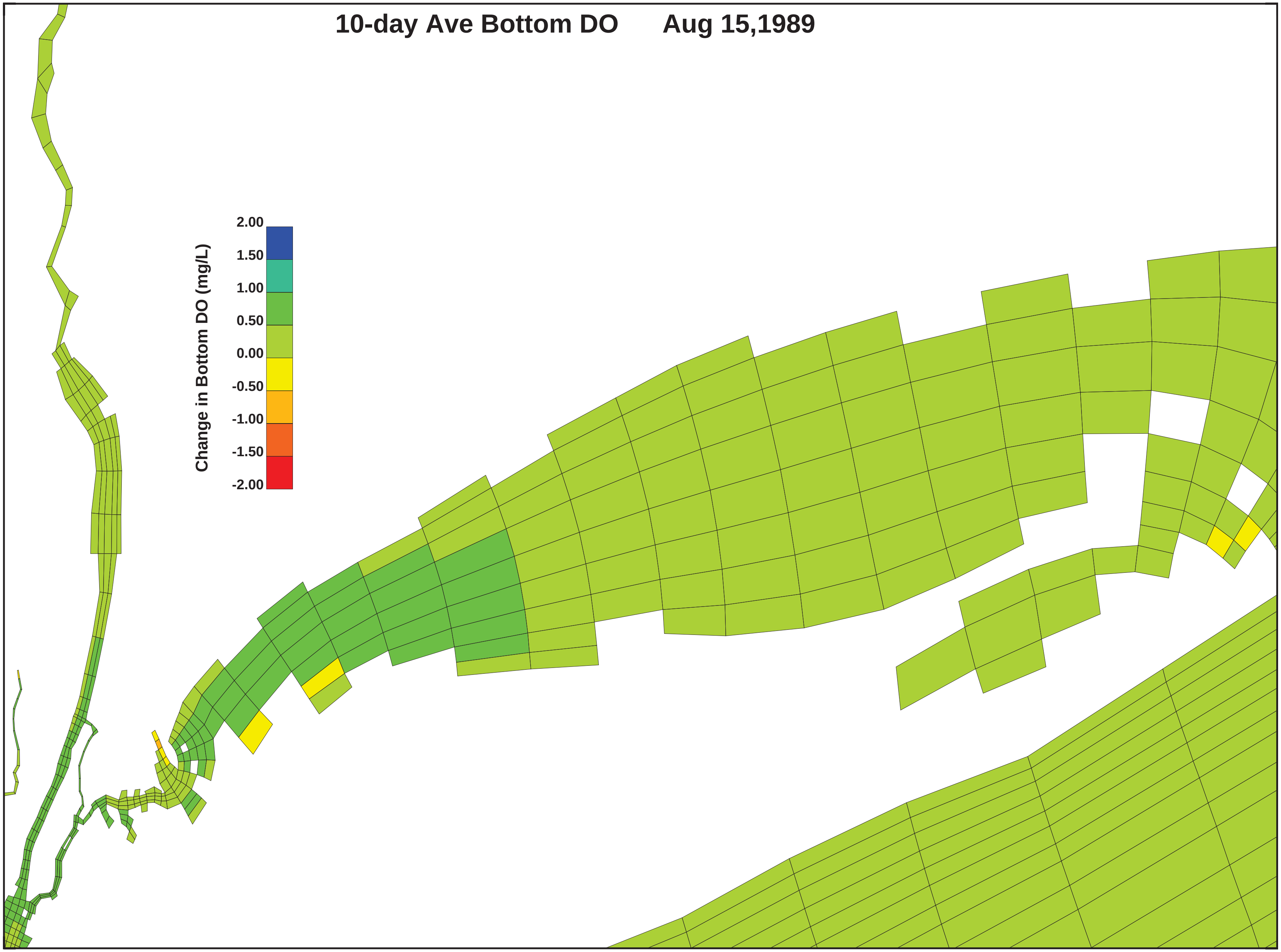
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Change in Bottom DO (mg/L)
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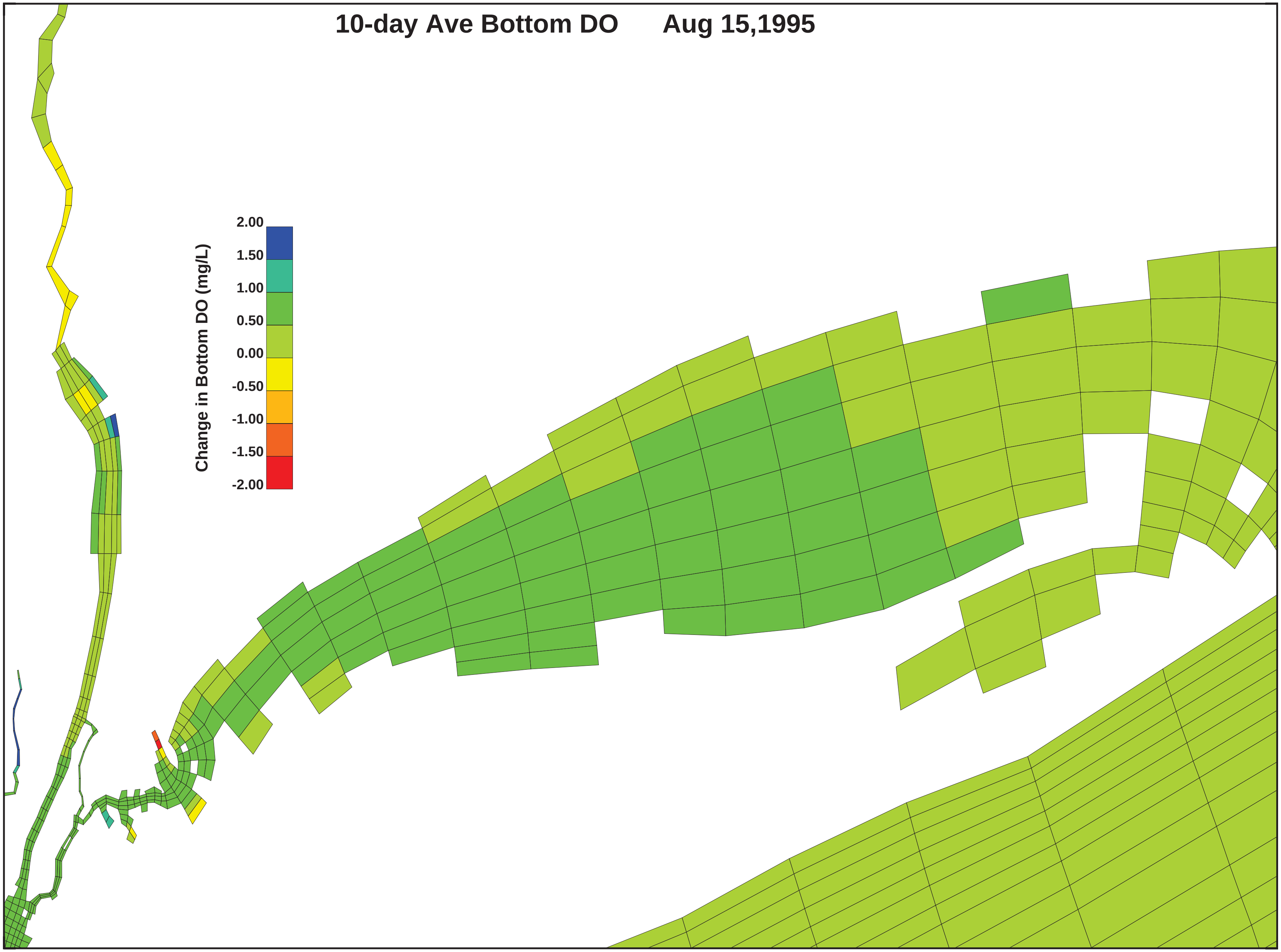
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Change in Bottom DO (mg/L)
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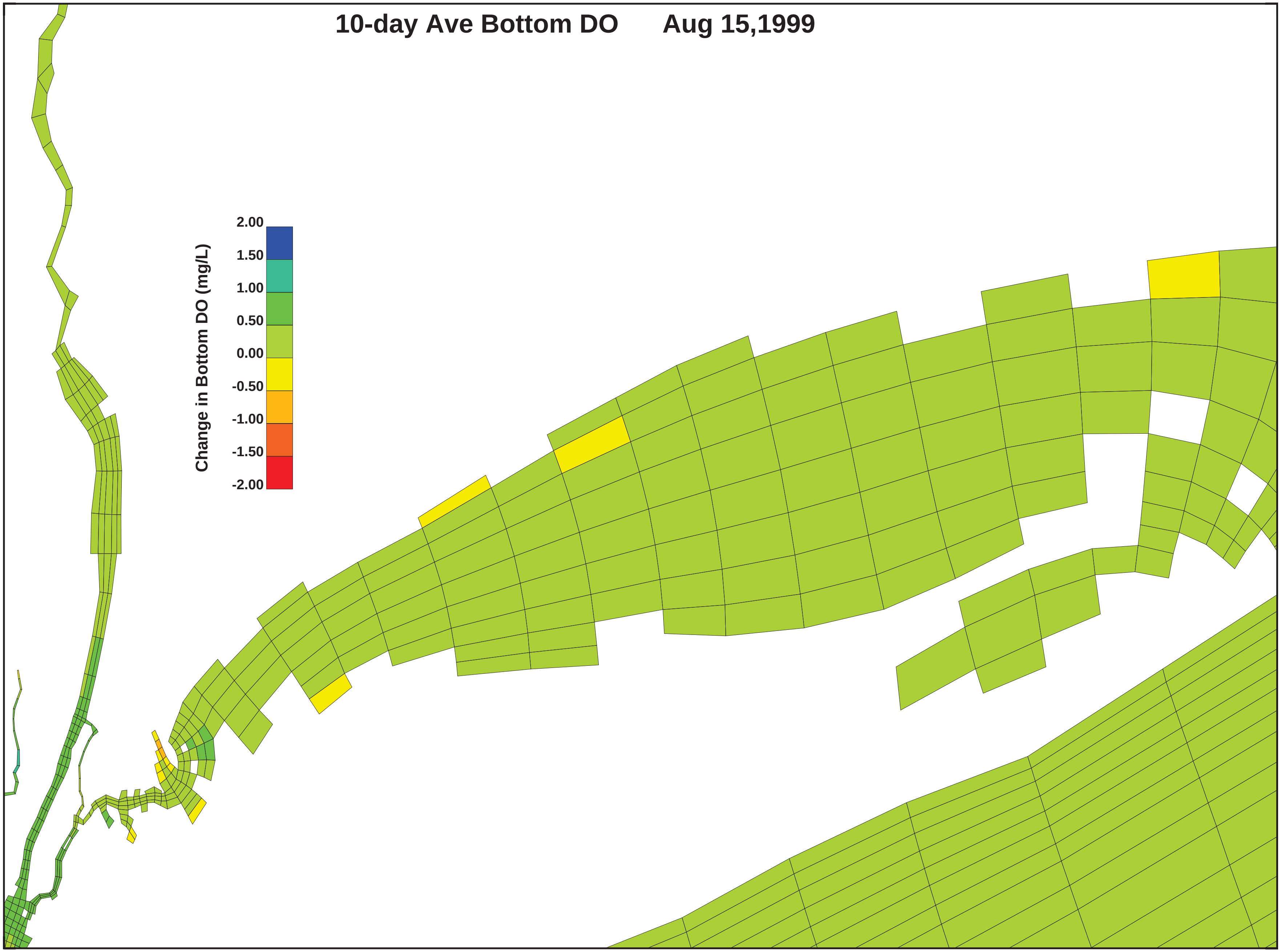
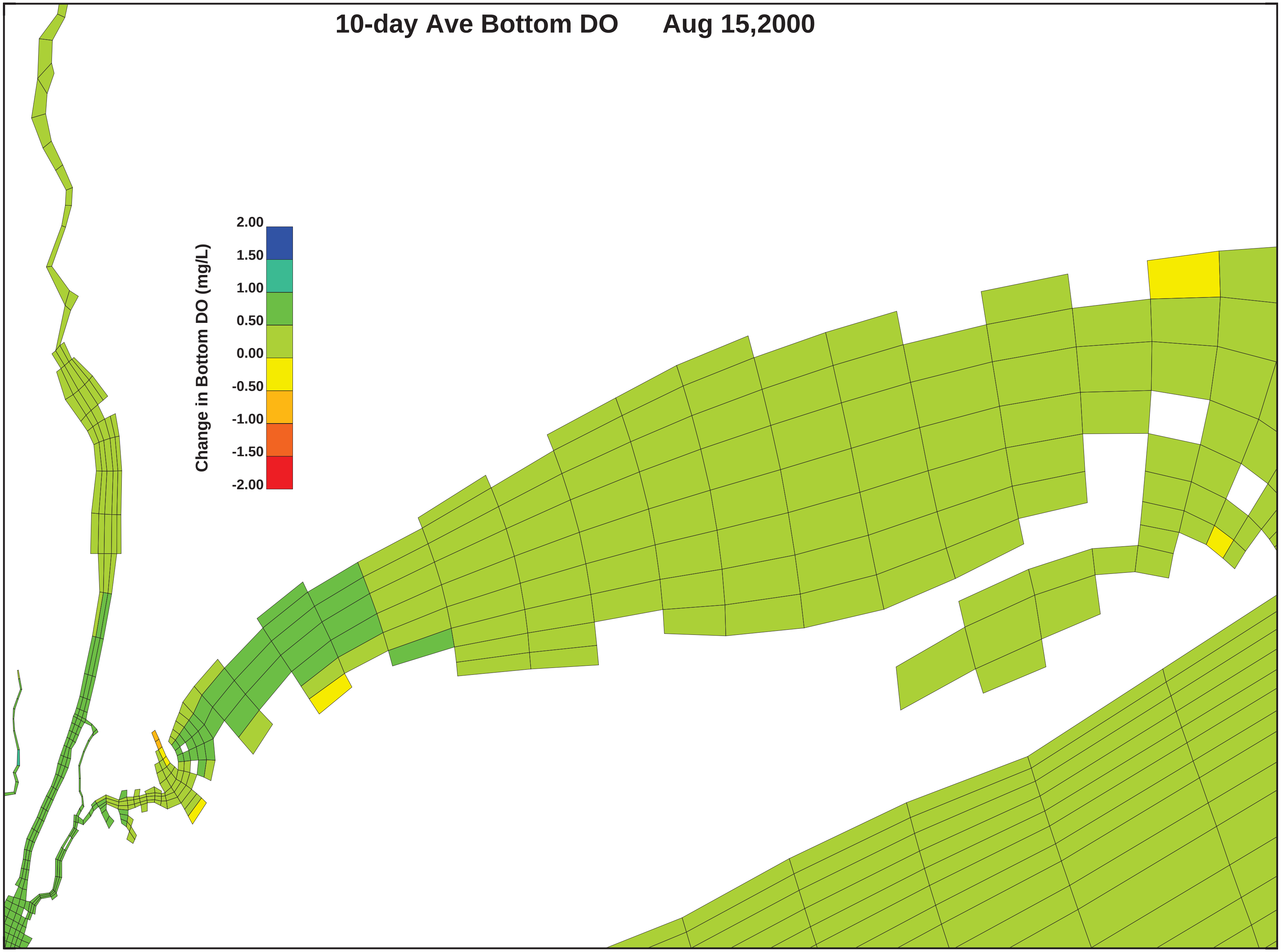
10-day Ave Bottom DO   Aug 15,1999





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Change in Bottom DO (mg/L)
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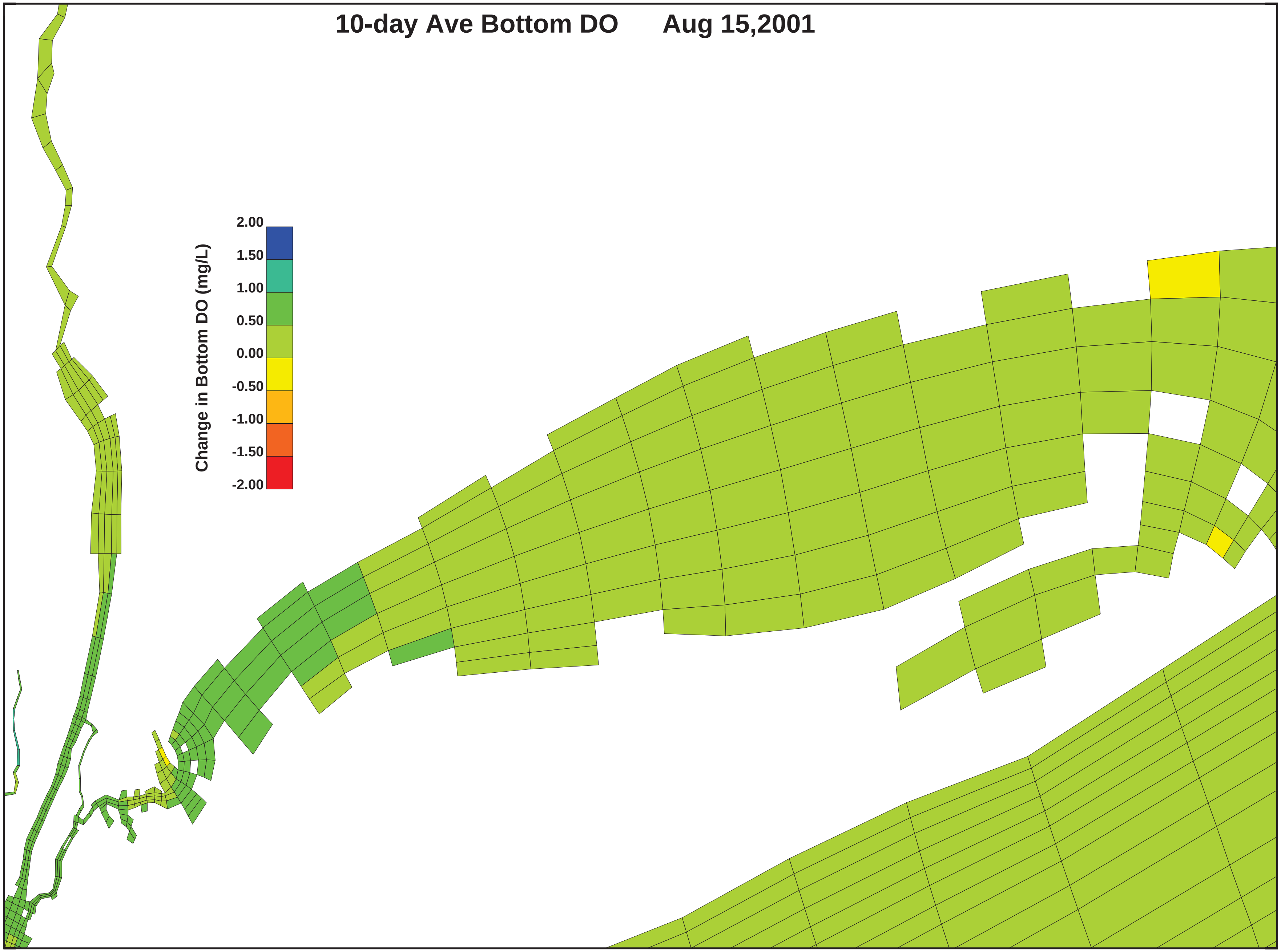
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Change in Bottom DO (mg/L)
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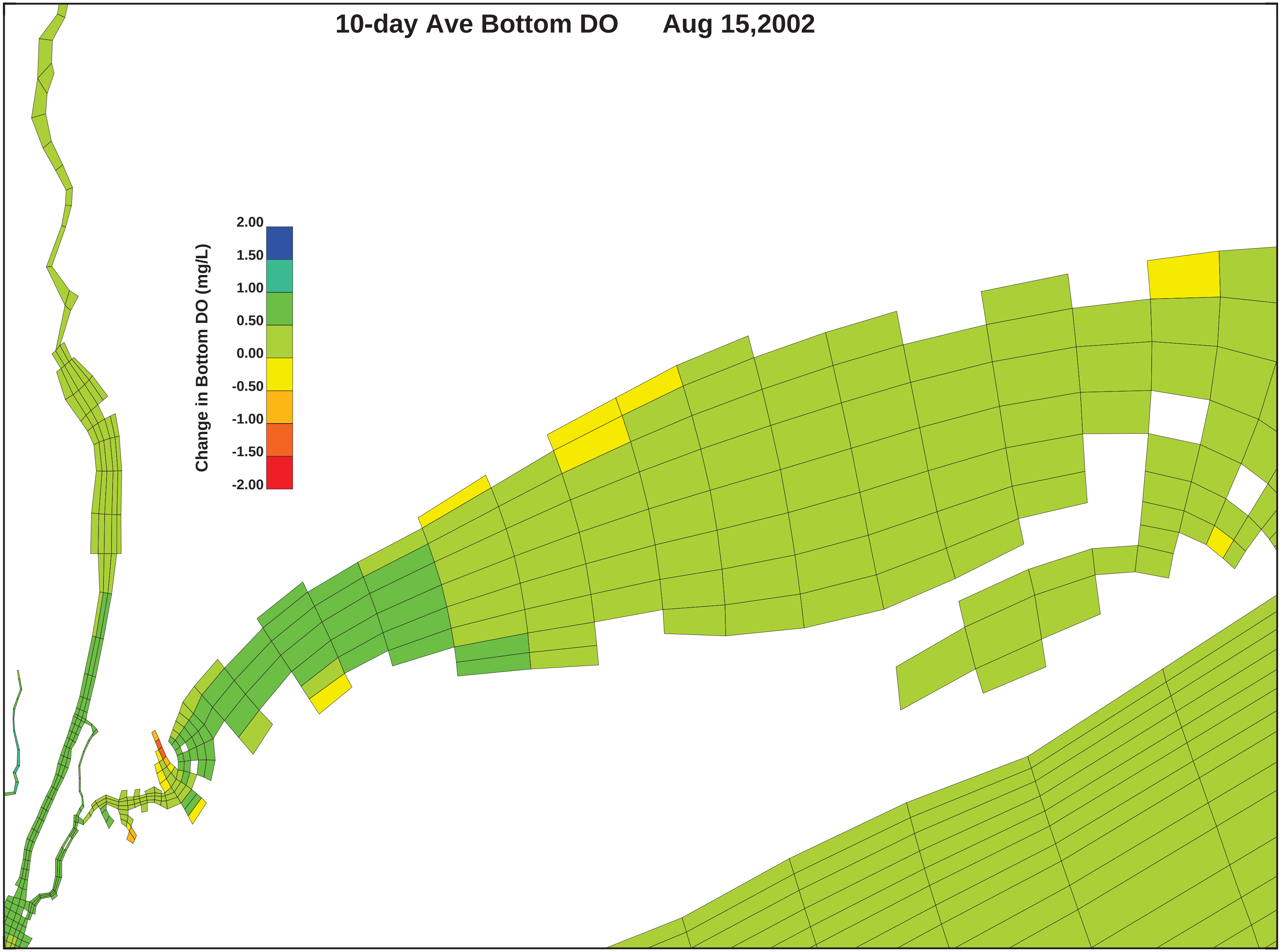
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Change in Bottom DO (mg/L)
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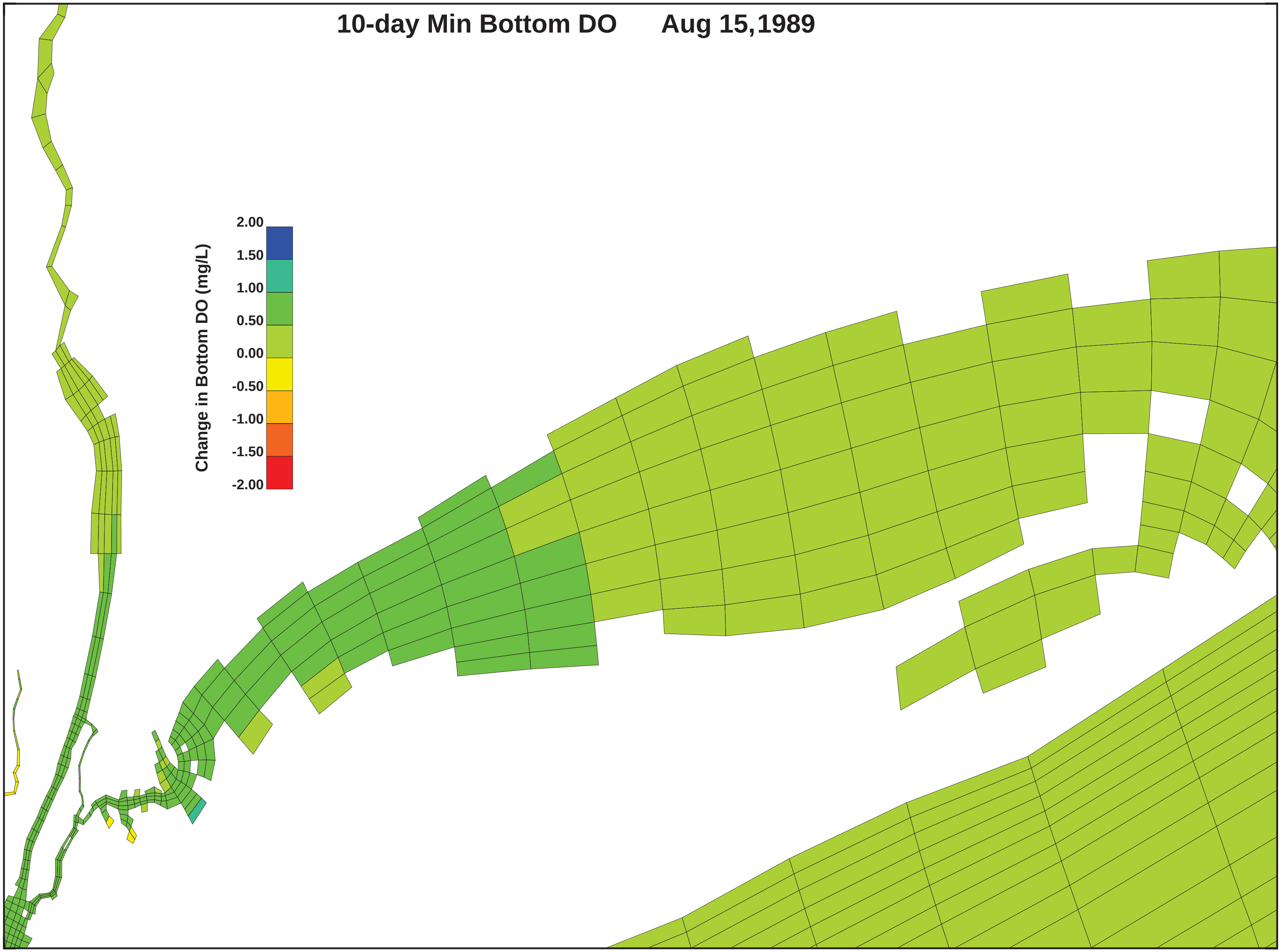

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Change in Bottom DO (mg/L)
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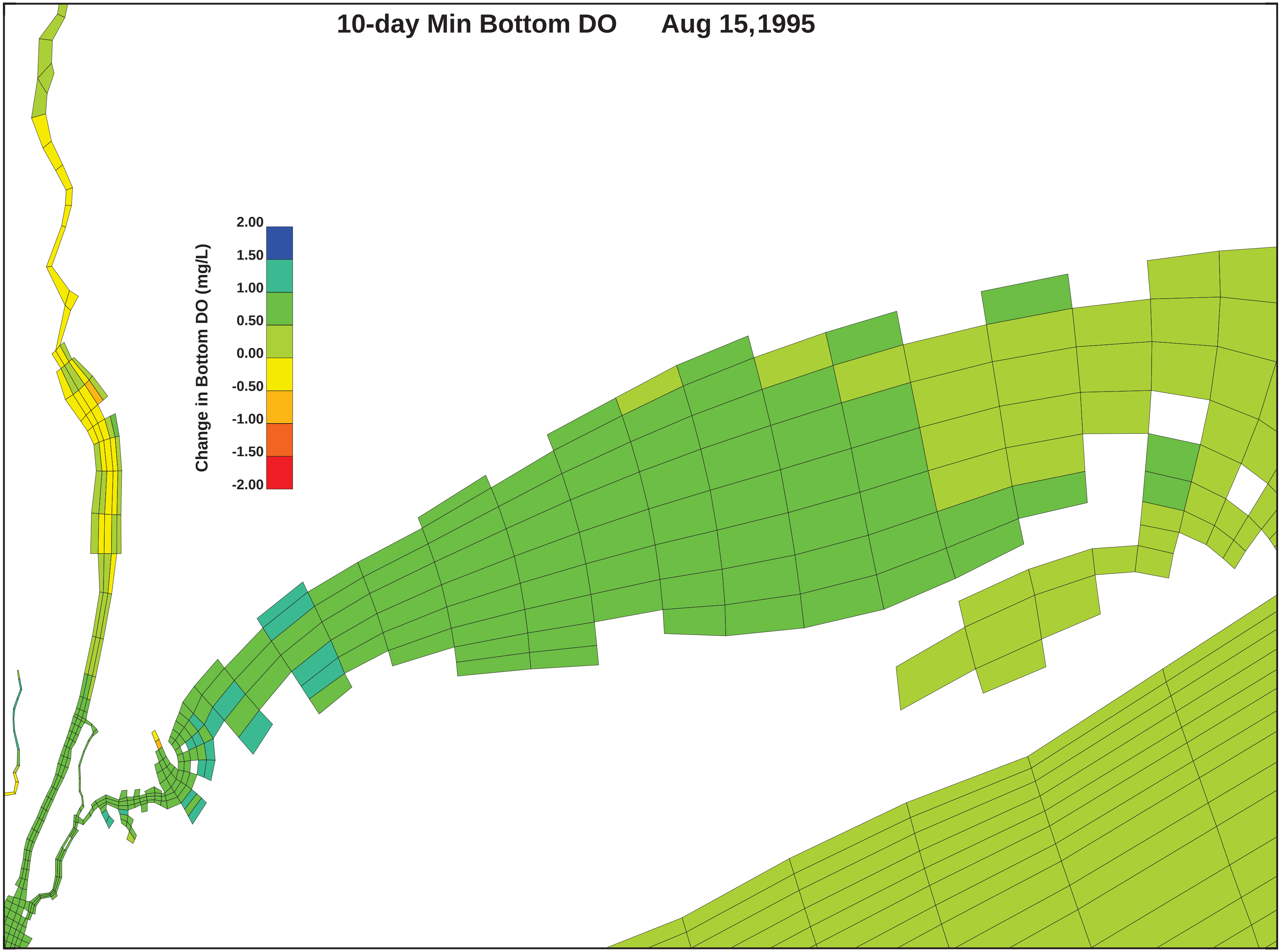

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Change in Bottom DO (mg/L)
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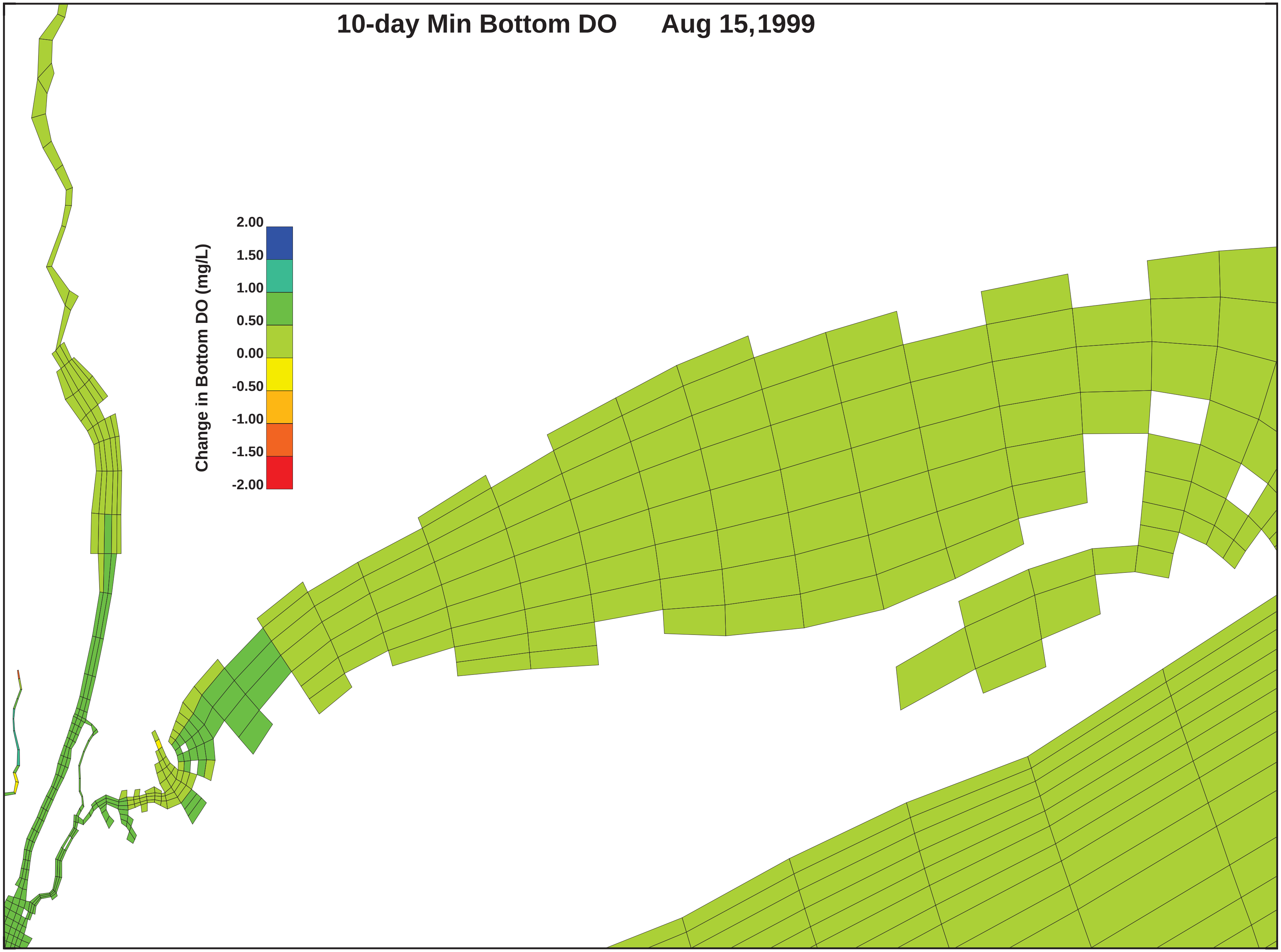
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Change in Bottom DO (mg/L)
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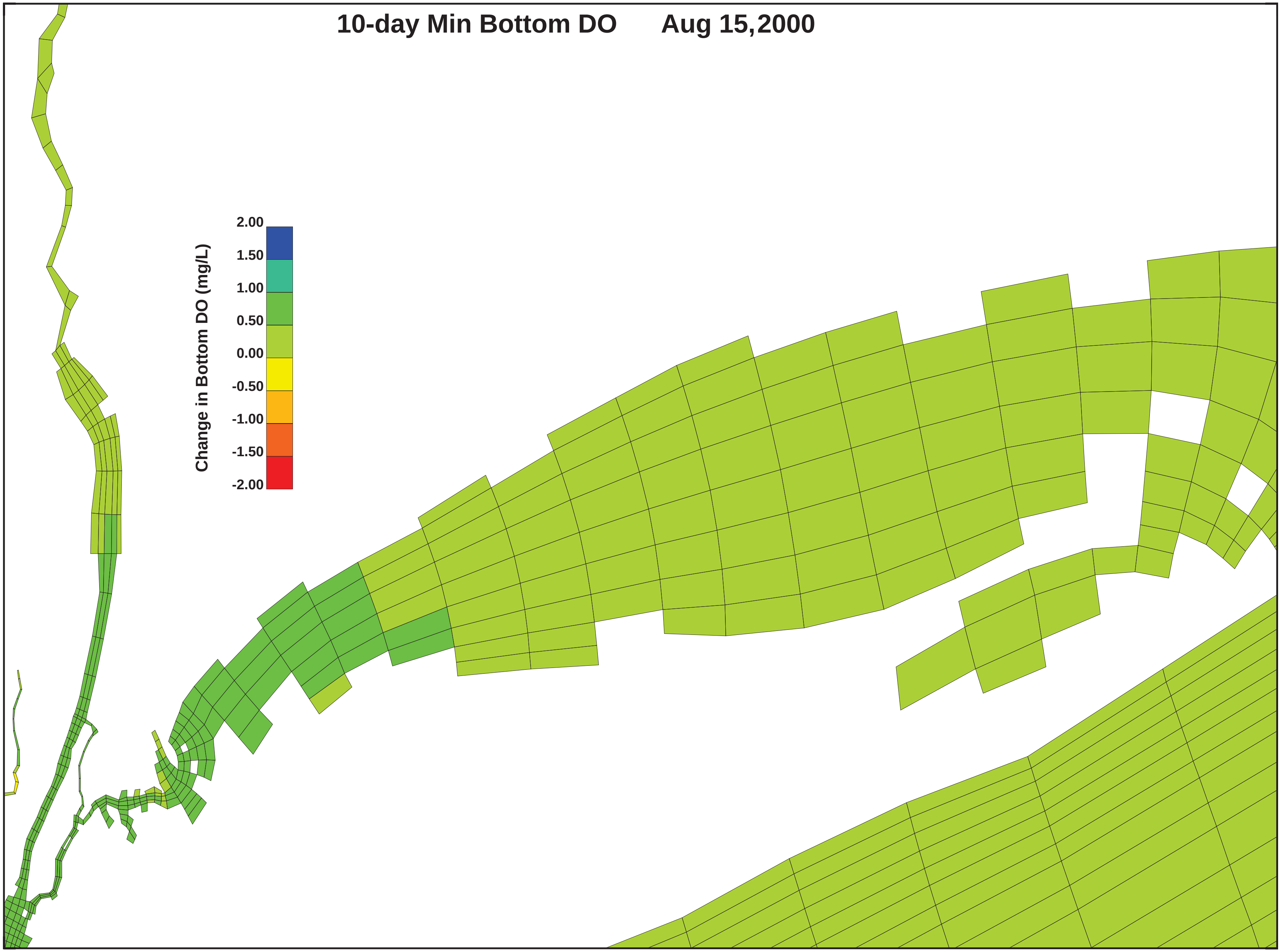
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Aug 15,2000
Change in Bottom DO (mg/L)
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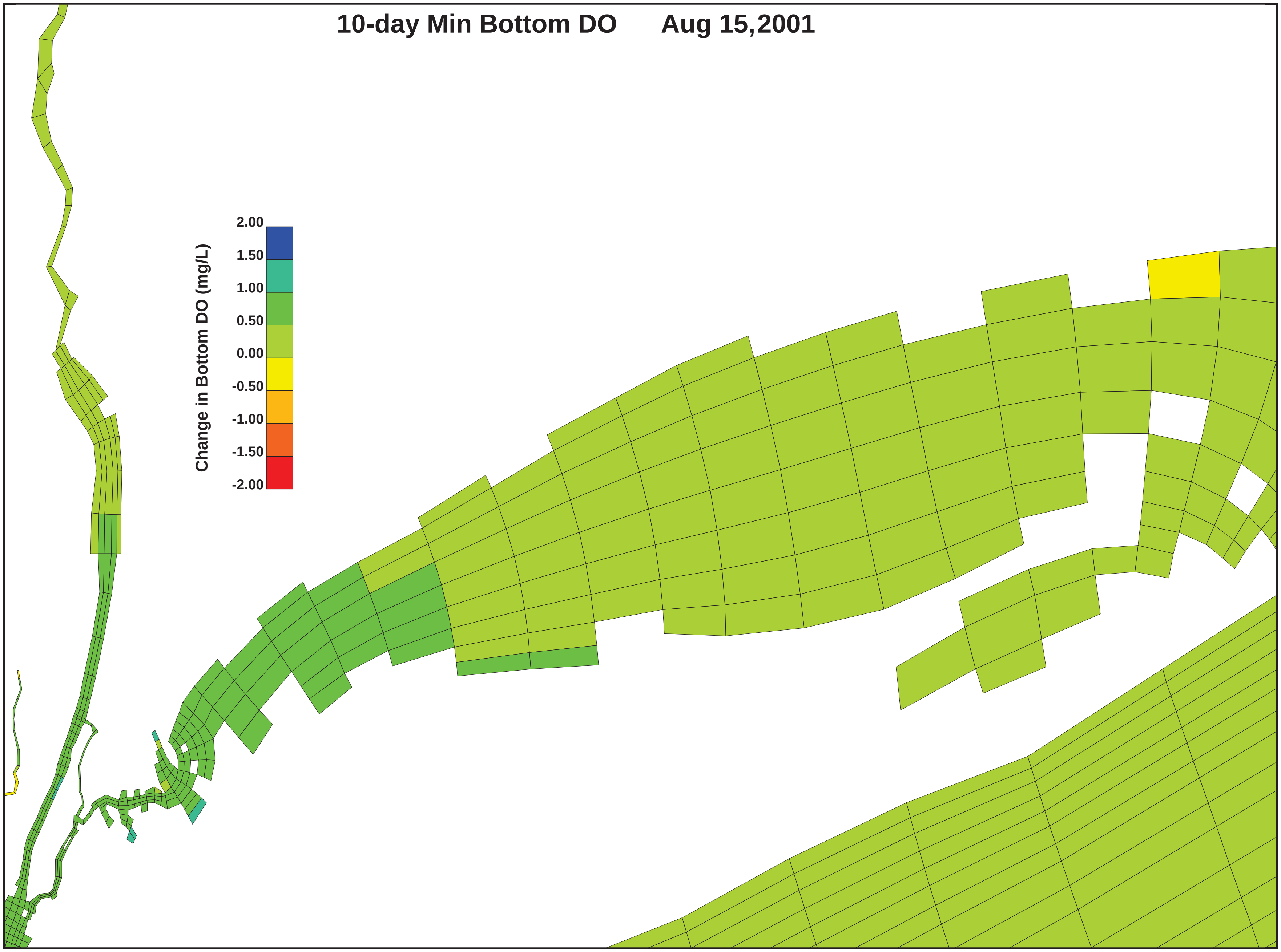
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Change in Bottom DO (mg/L)
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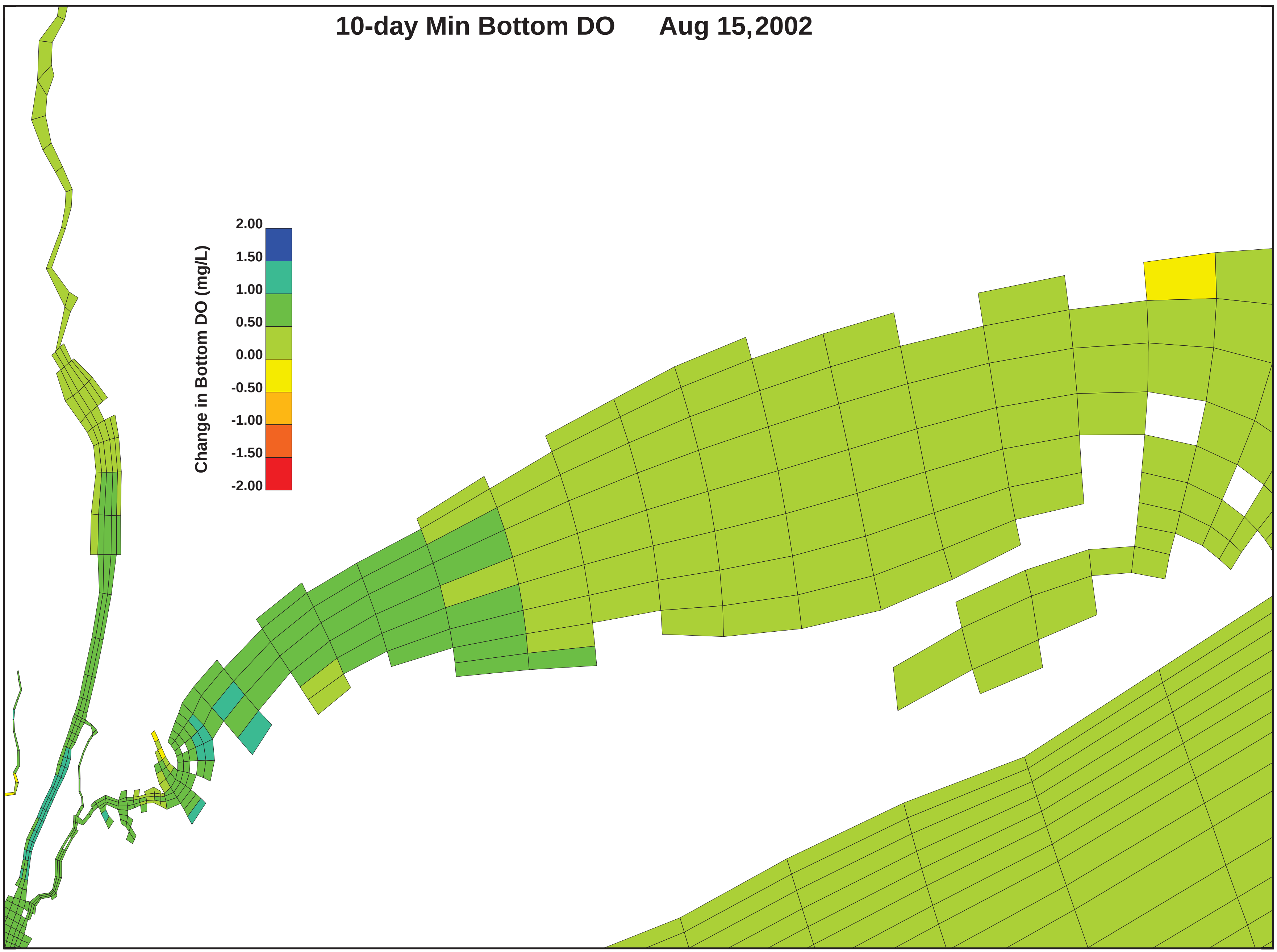

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Change in Bottom DO (mg/L)
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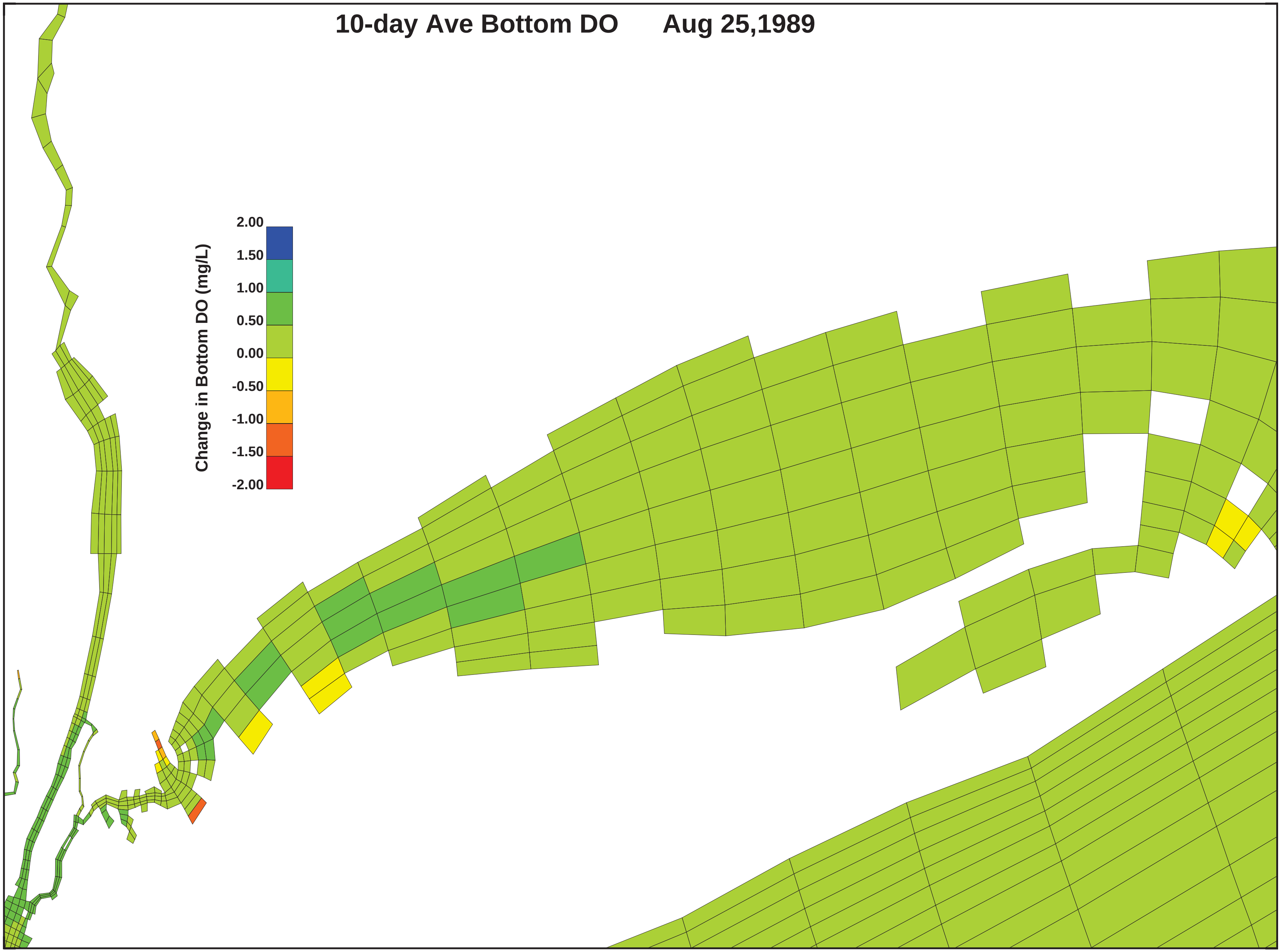
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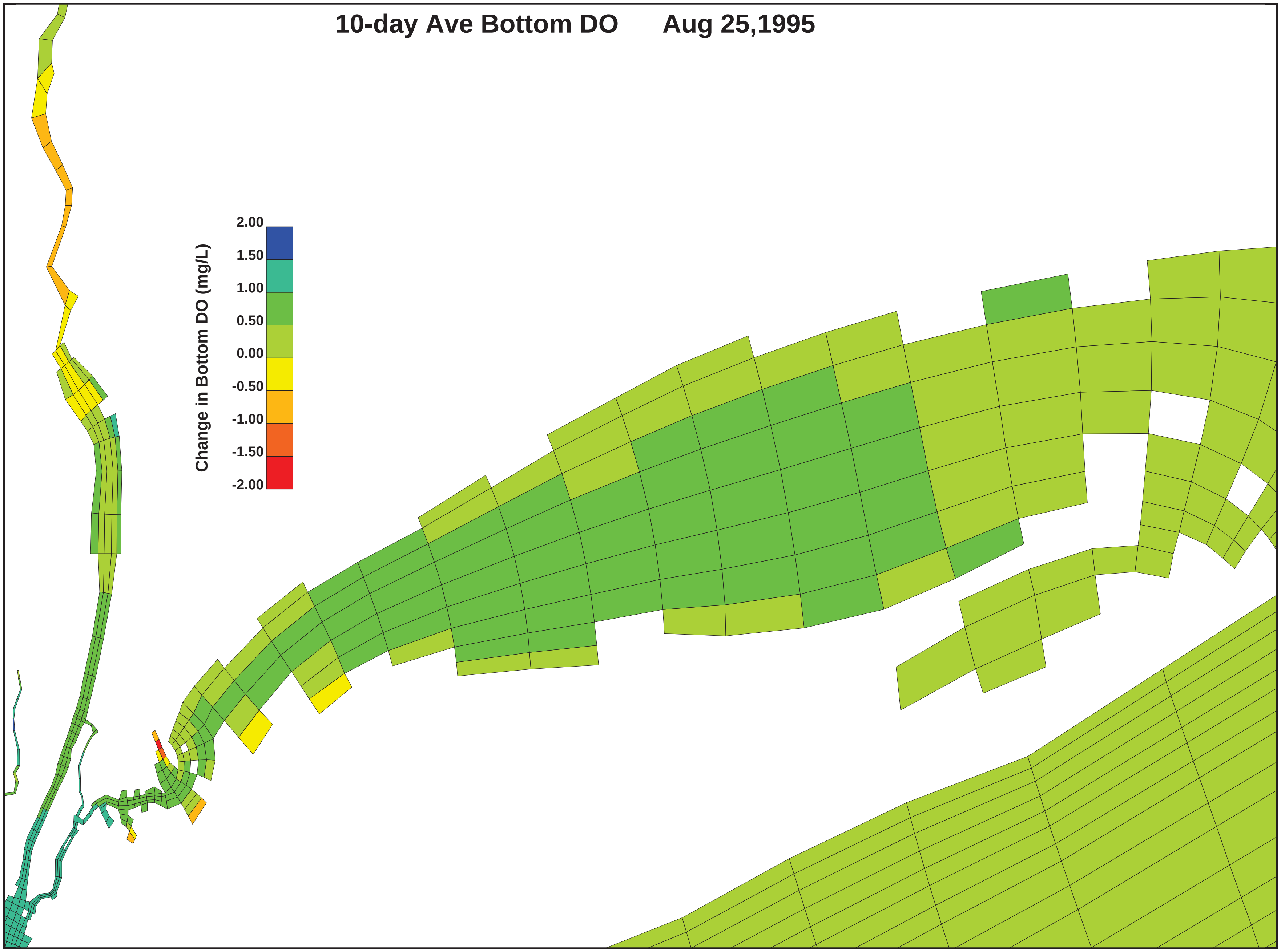
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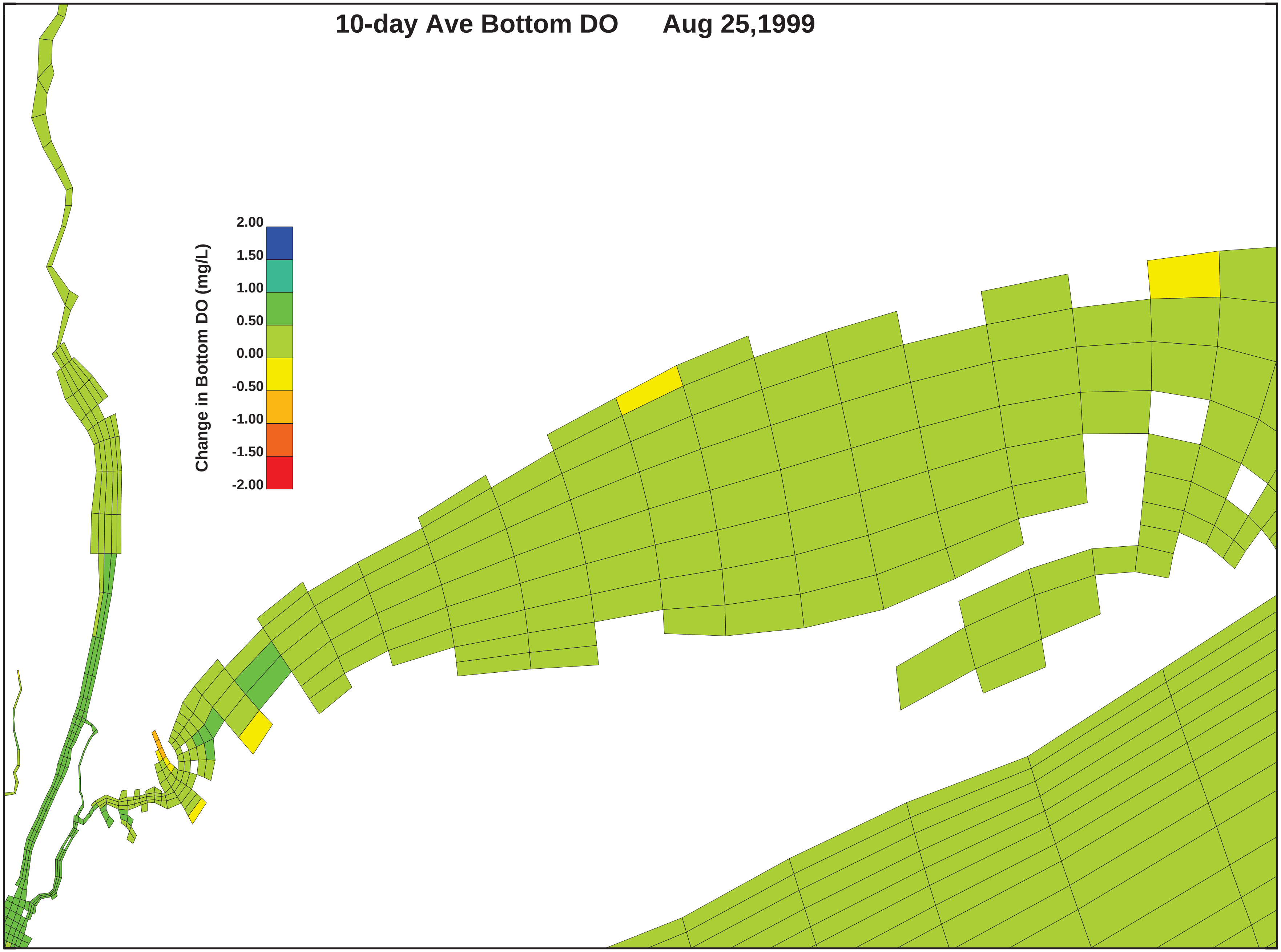
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Change in Bottom DO (mg/L)
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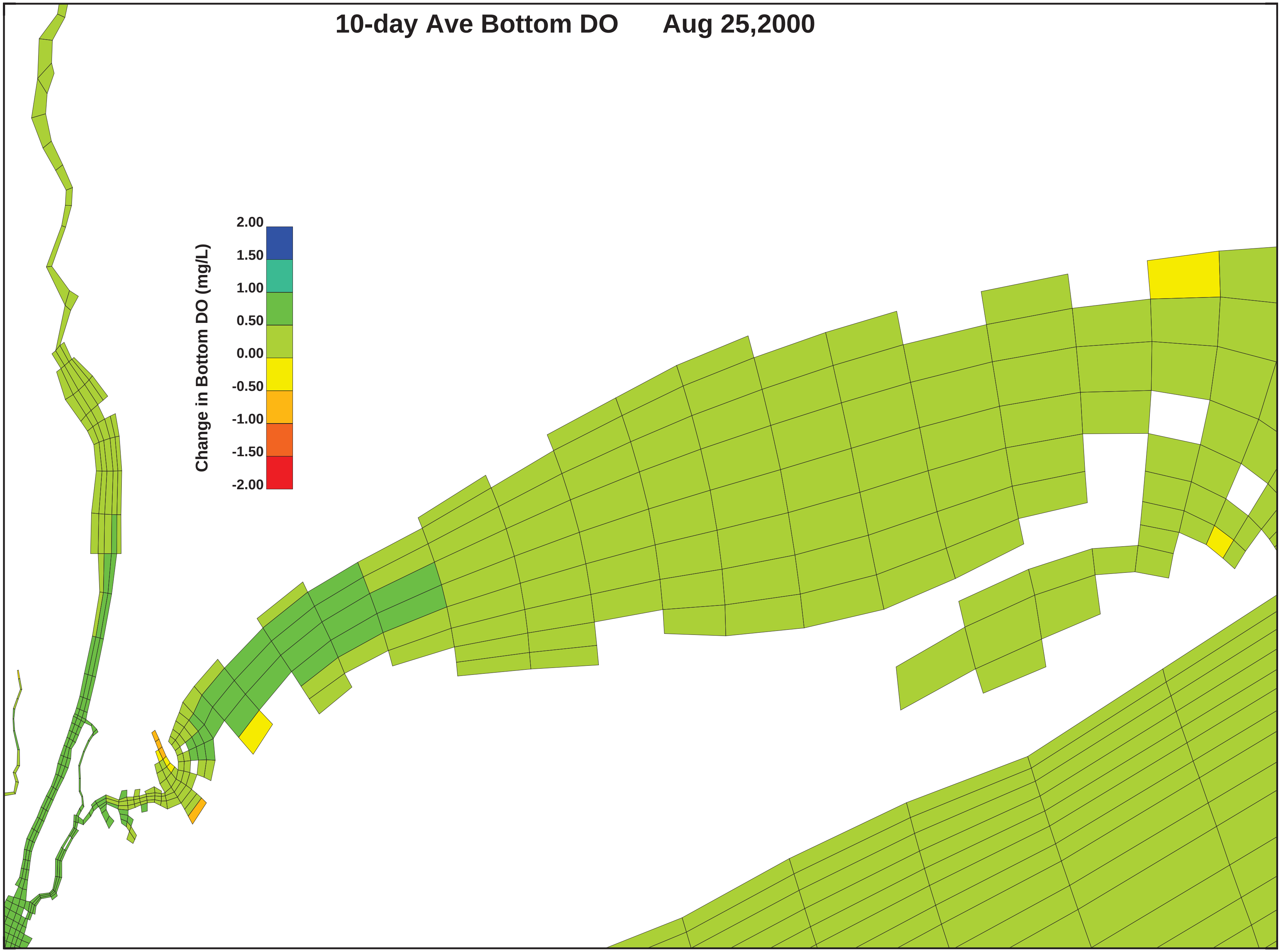

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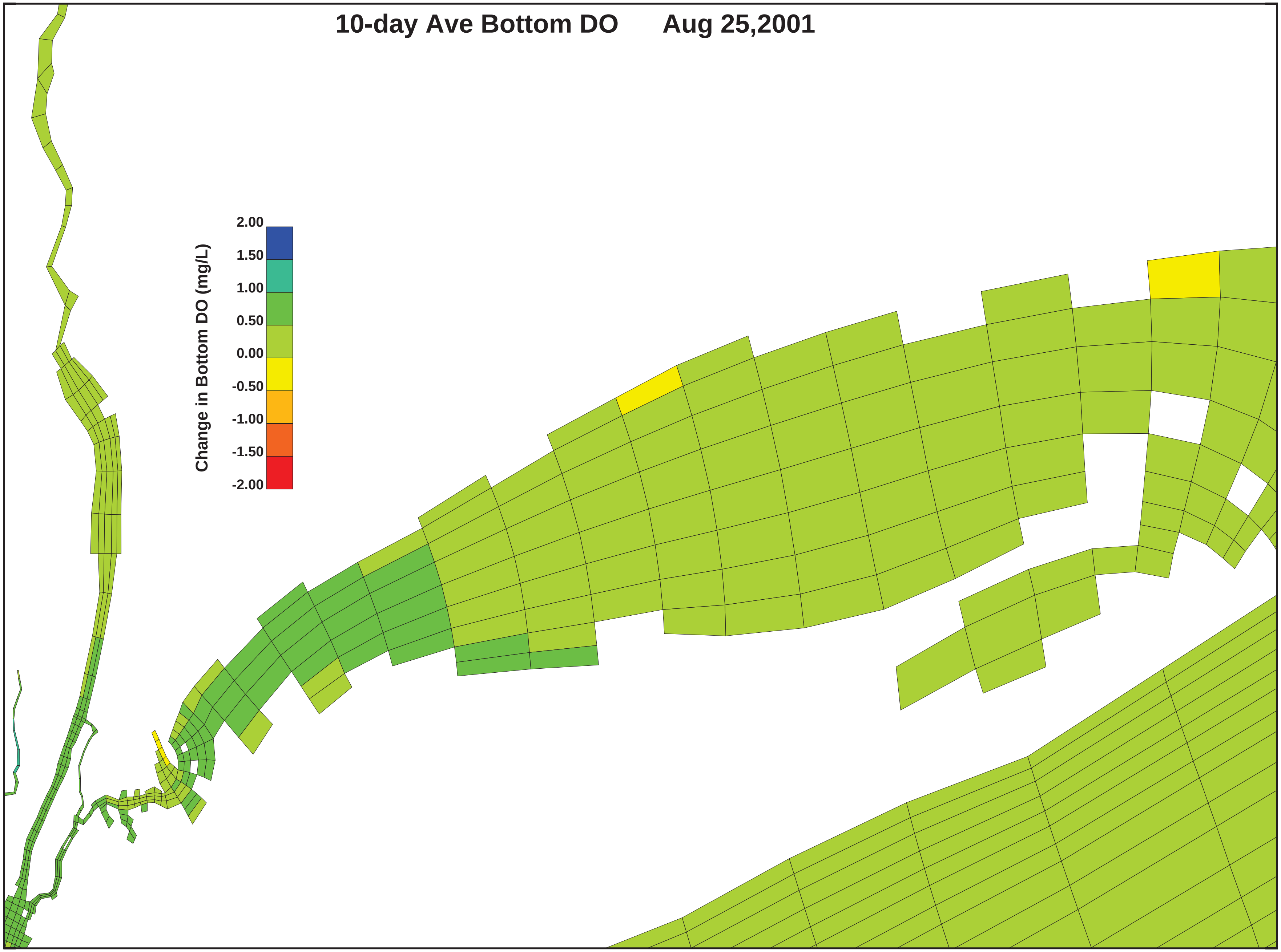
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Change in Bottom DO (mg/L)
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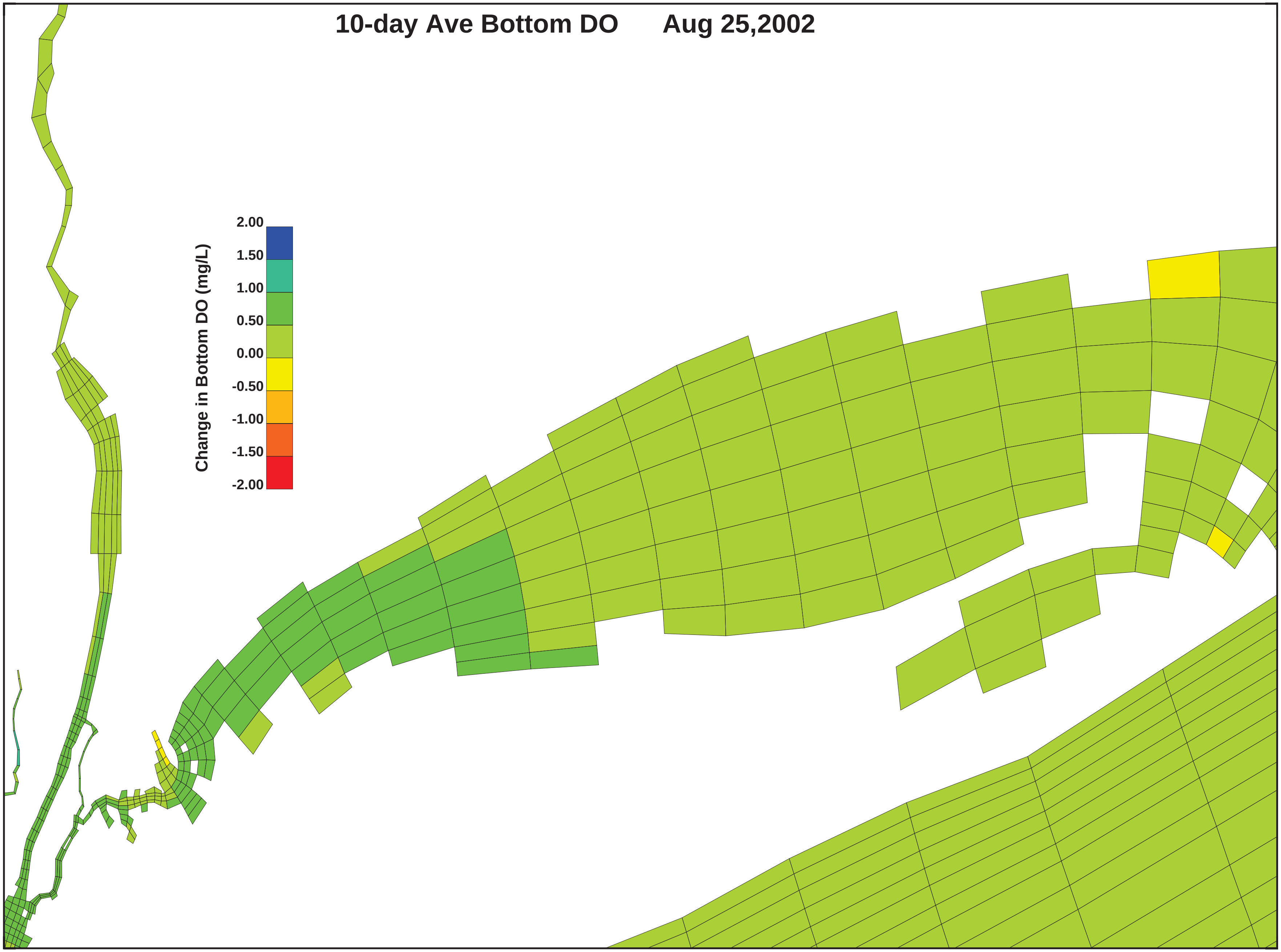

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Change in Bottom DO (mg/L)
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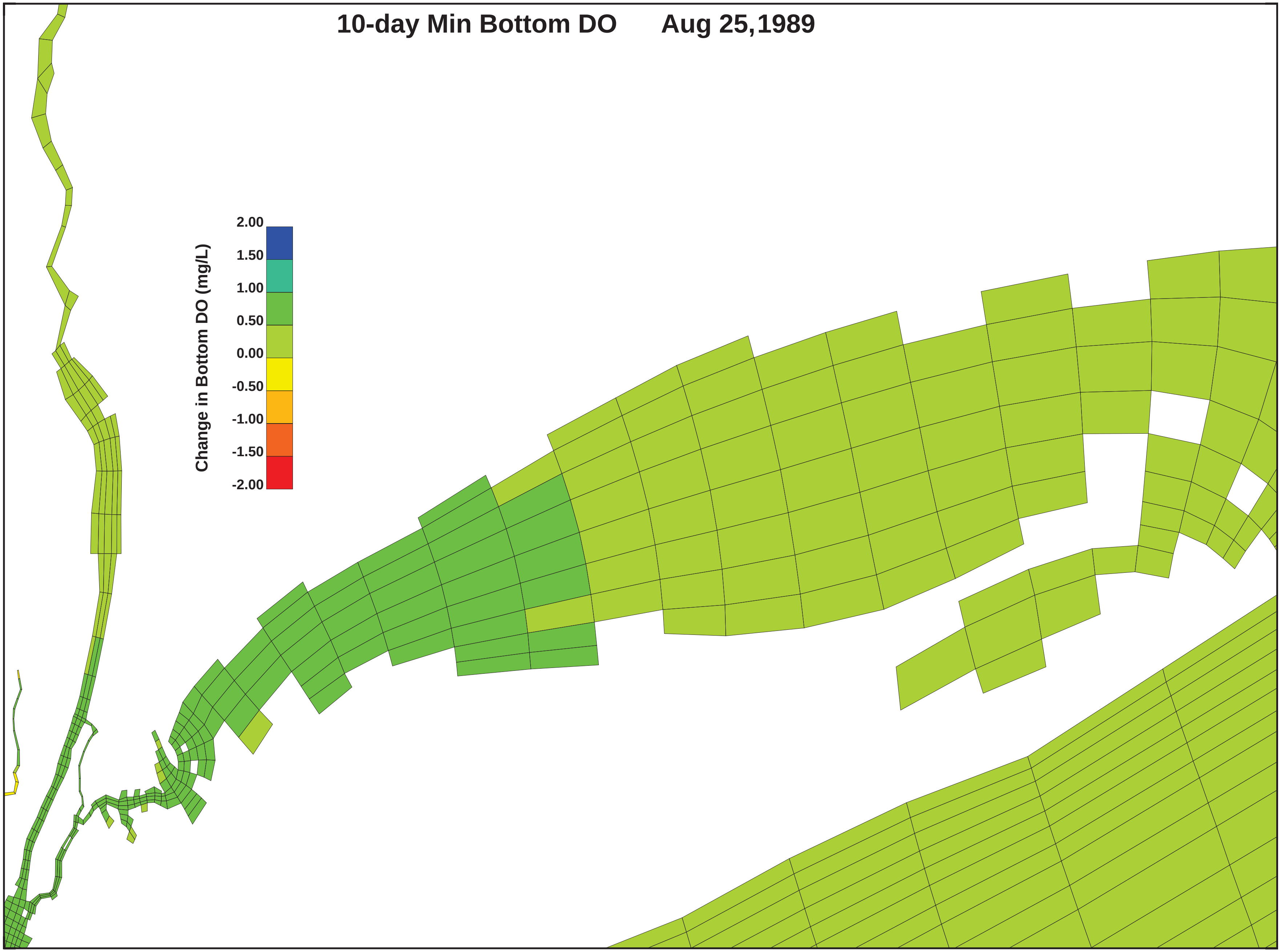

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Change in Bottom DO (mg/L)
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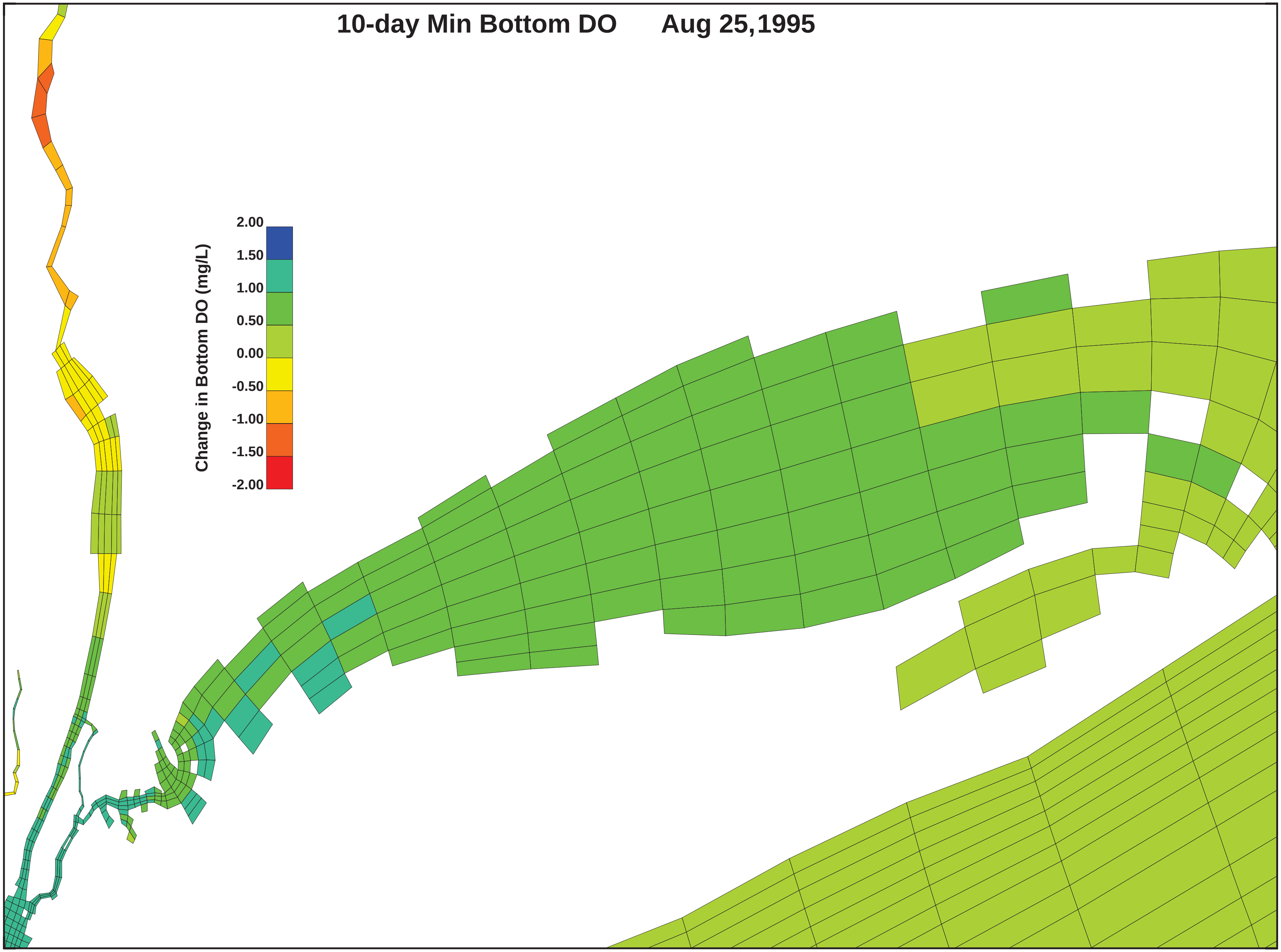
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Change in Bottom DO (mg/L)
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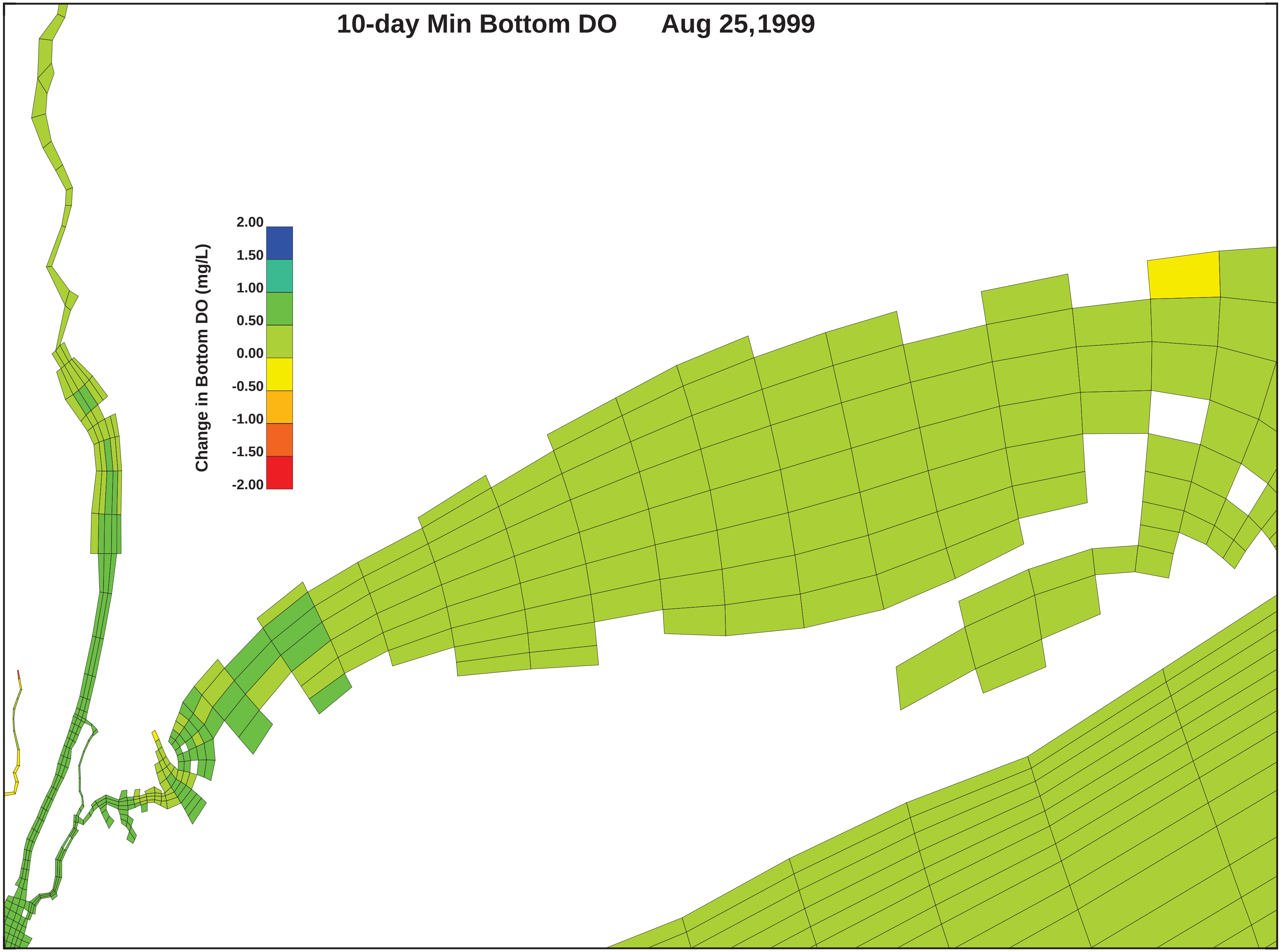

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Change in Bottom DO (mg/L)
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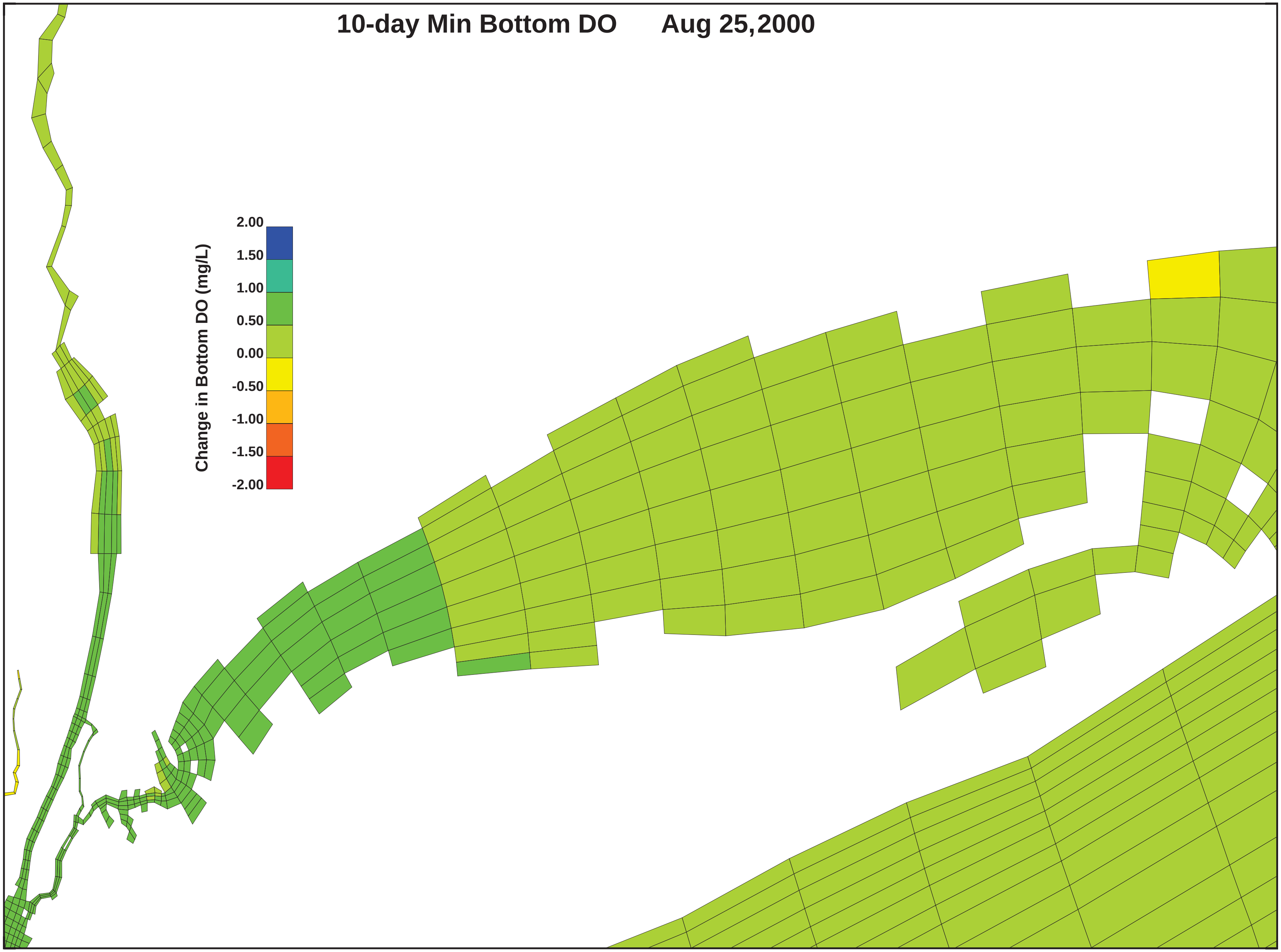

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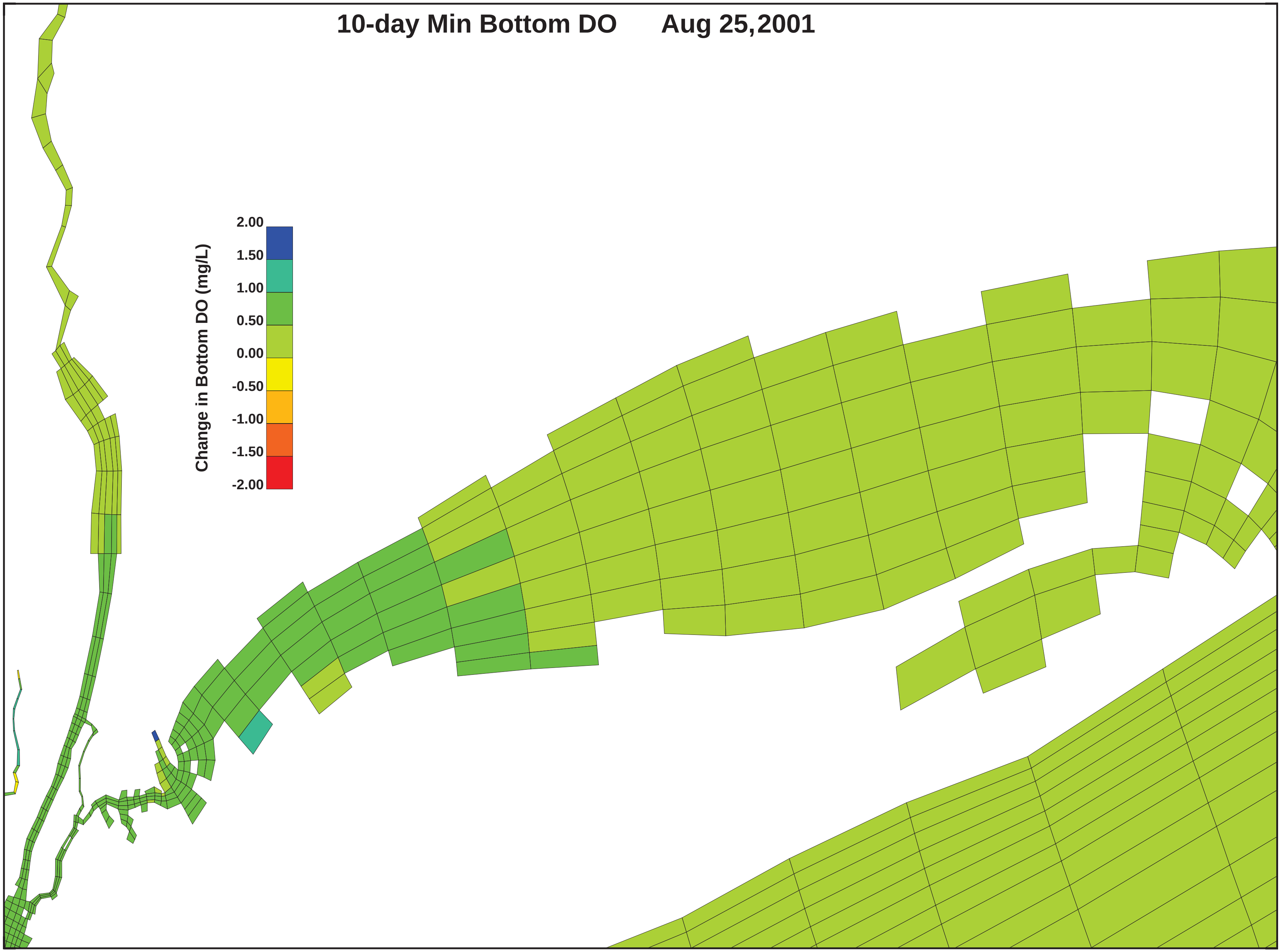
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Change in Bottom DO (mg/L)
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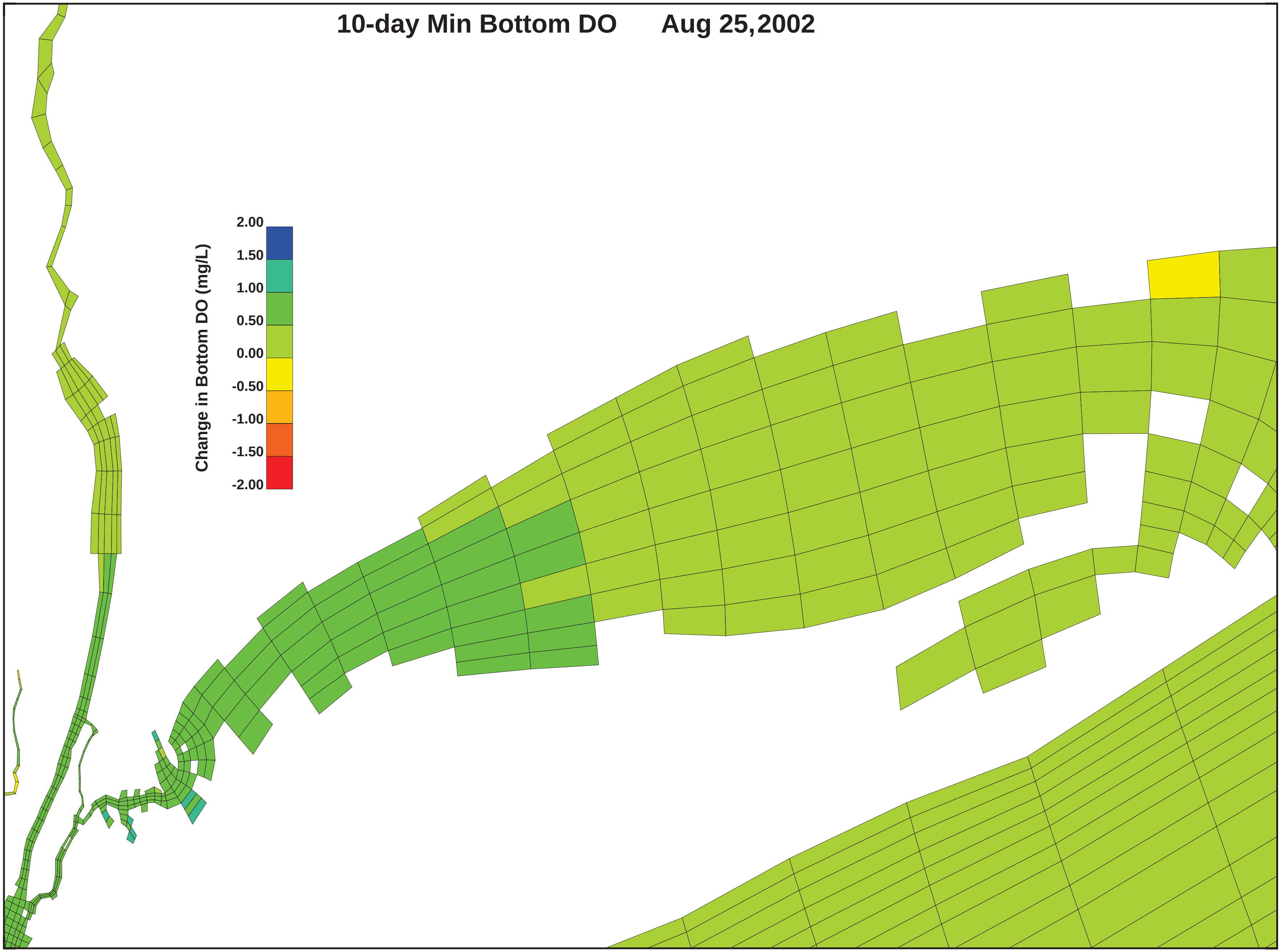
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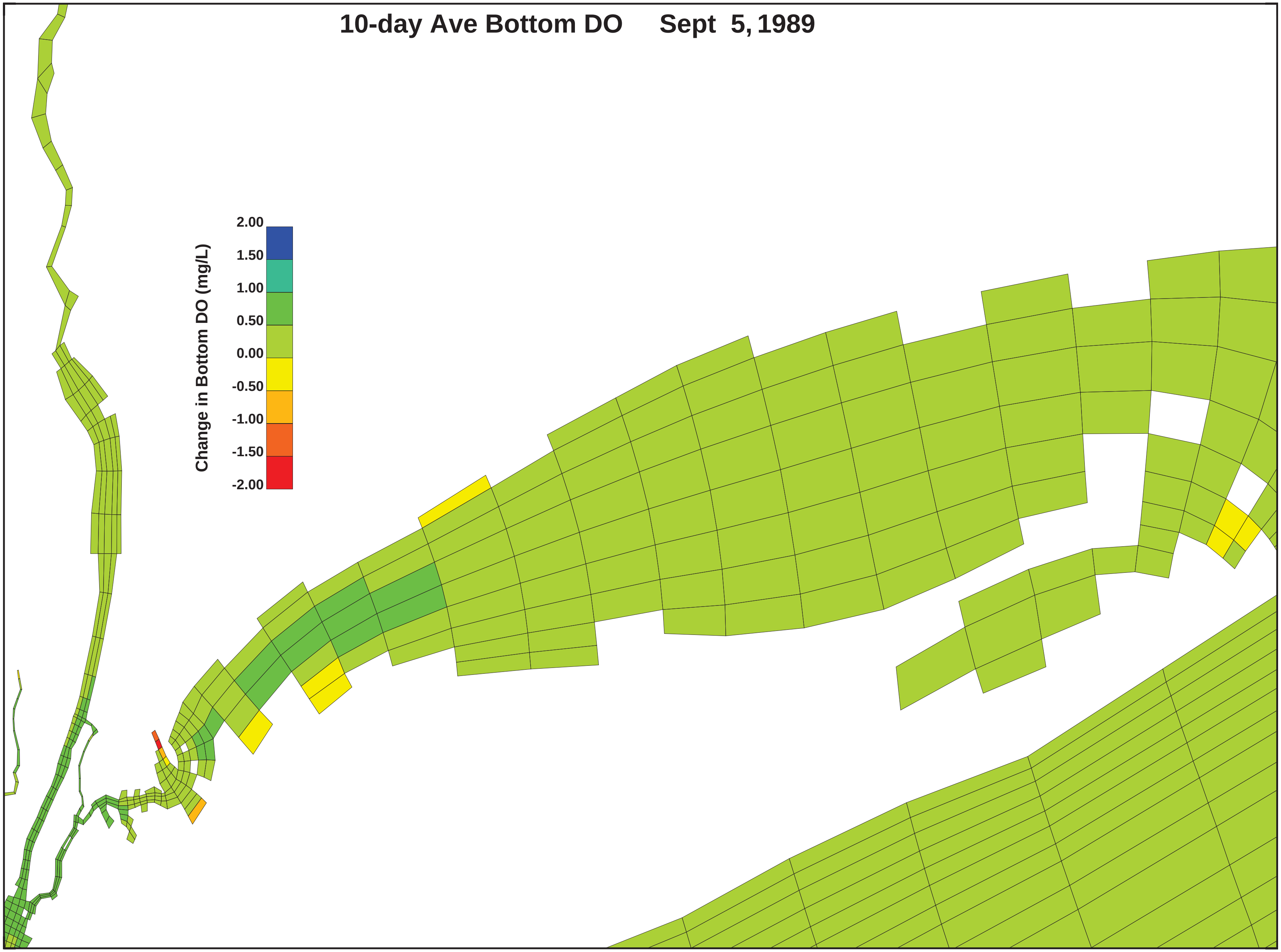

10-day Ave Bottom DO Sept 5, 1989
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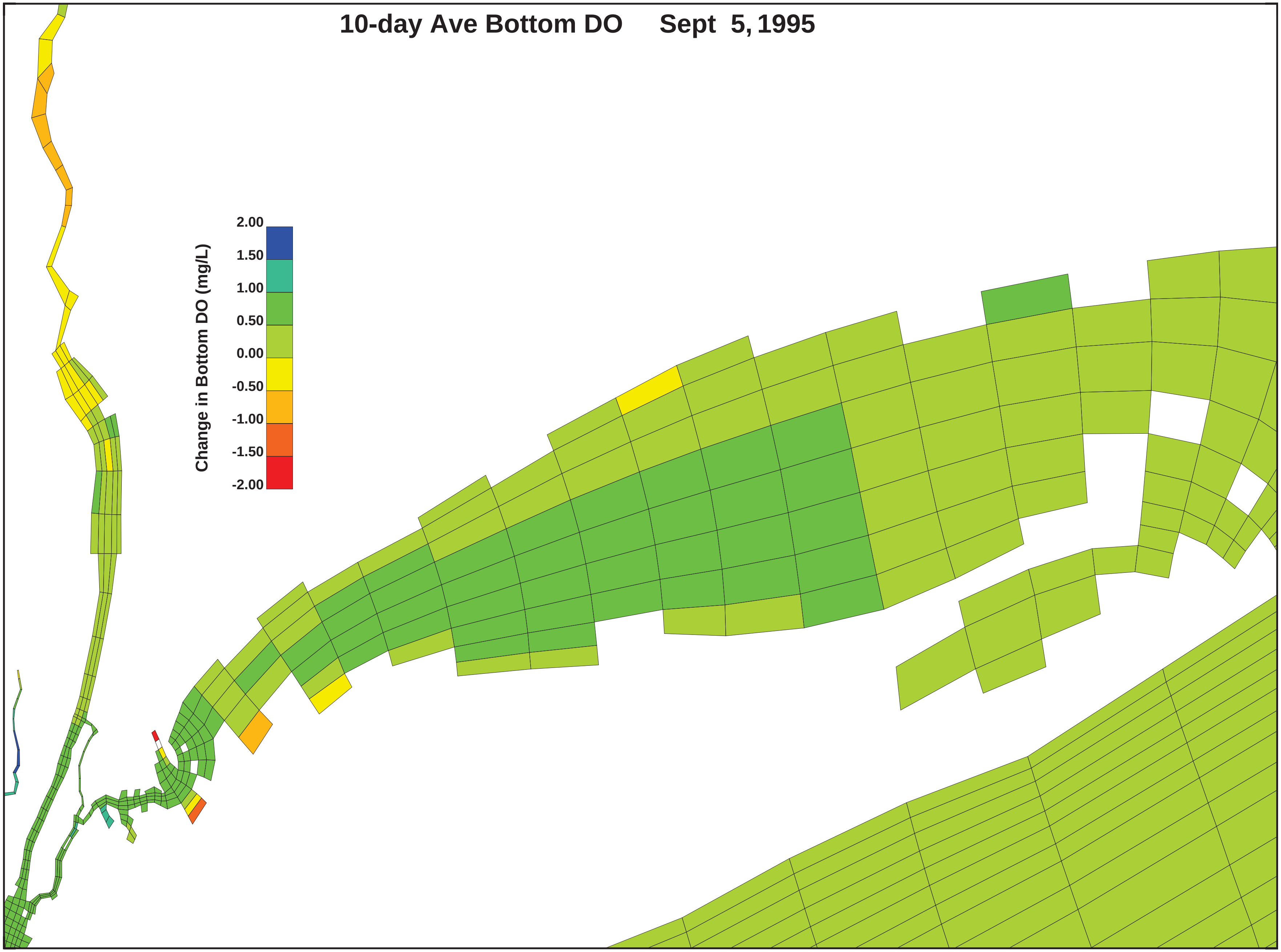

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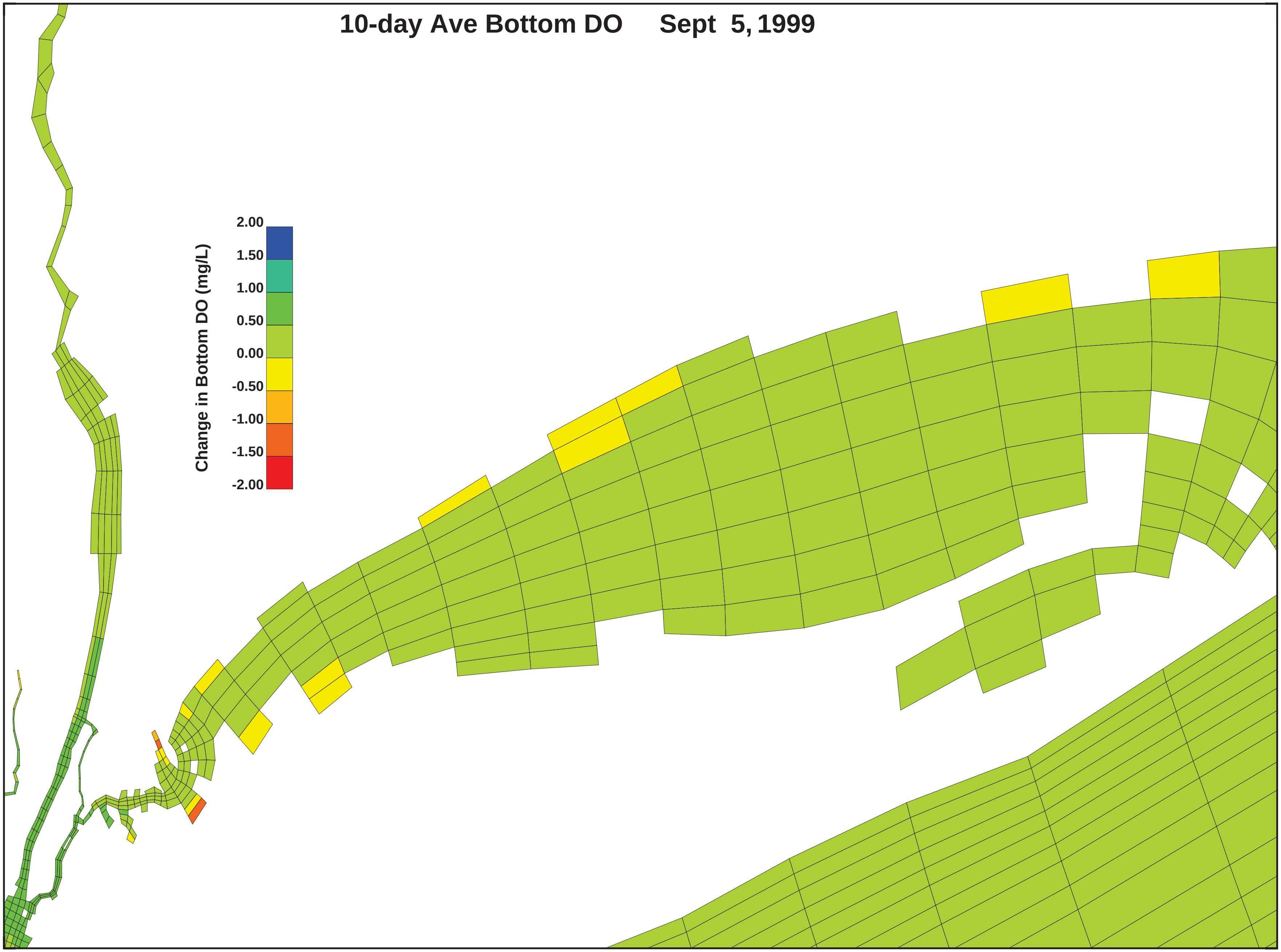
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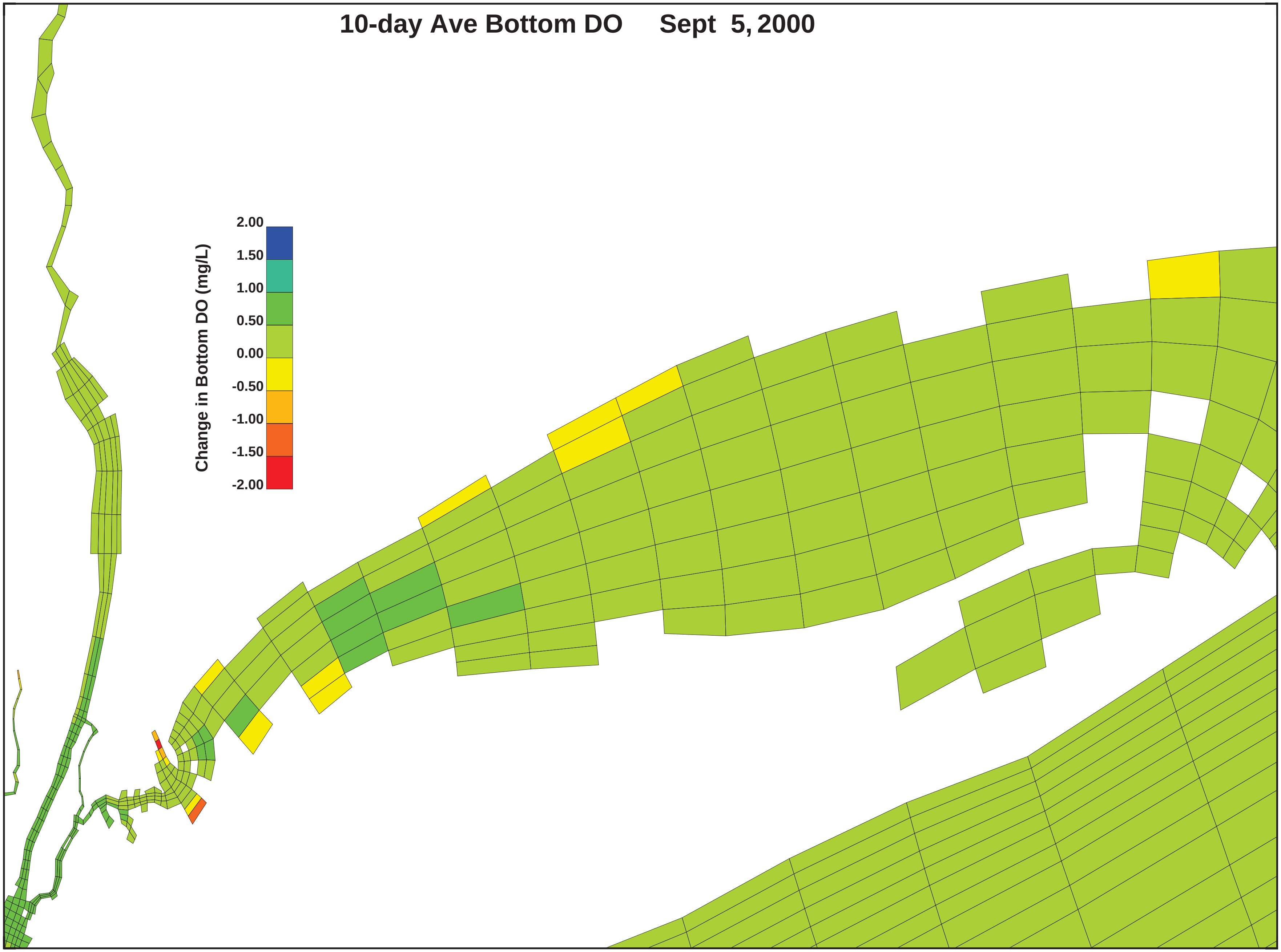

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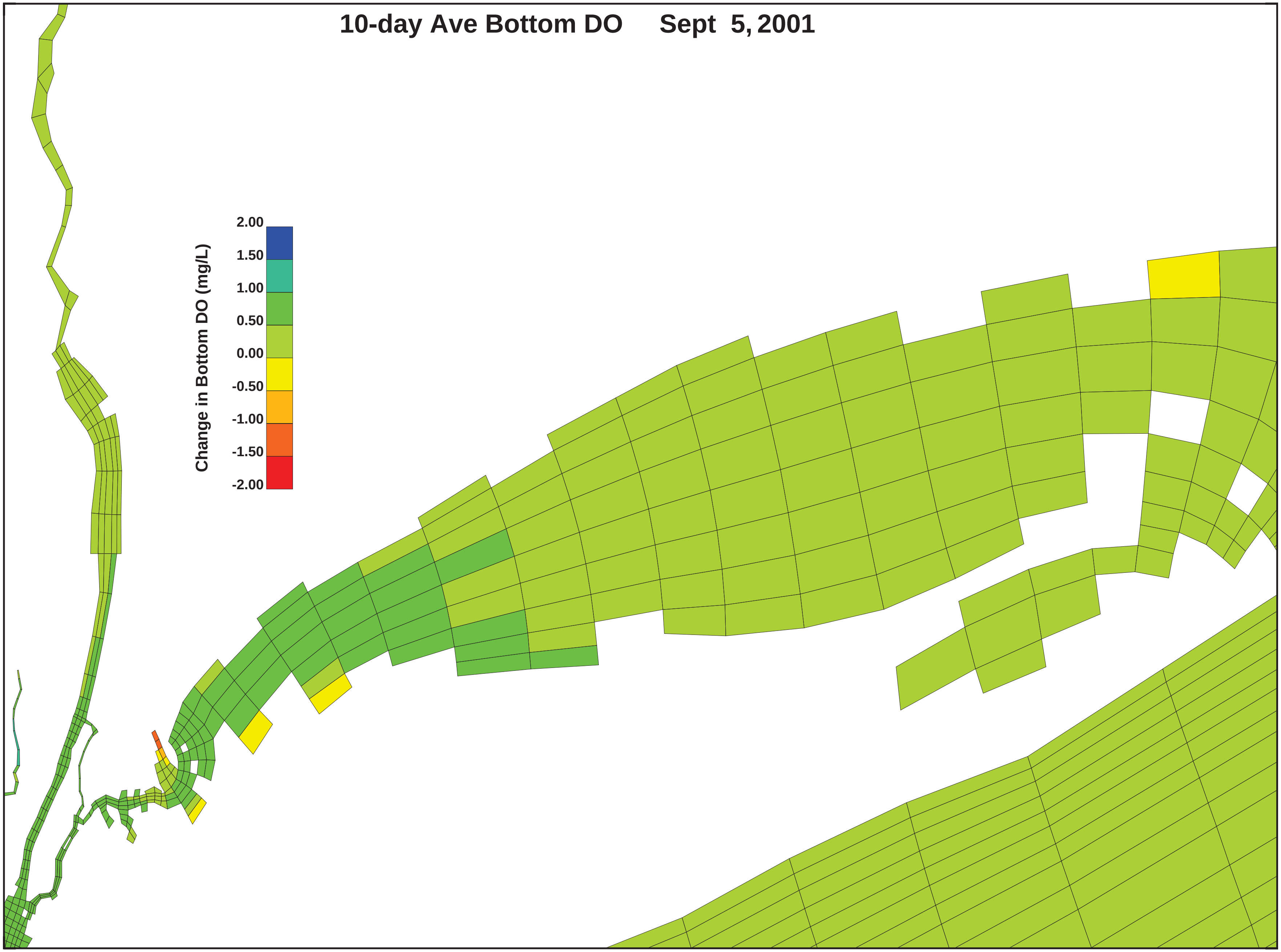
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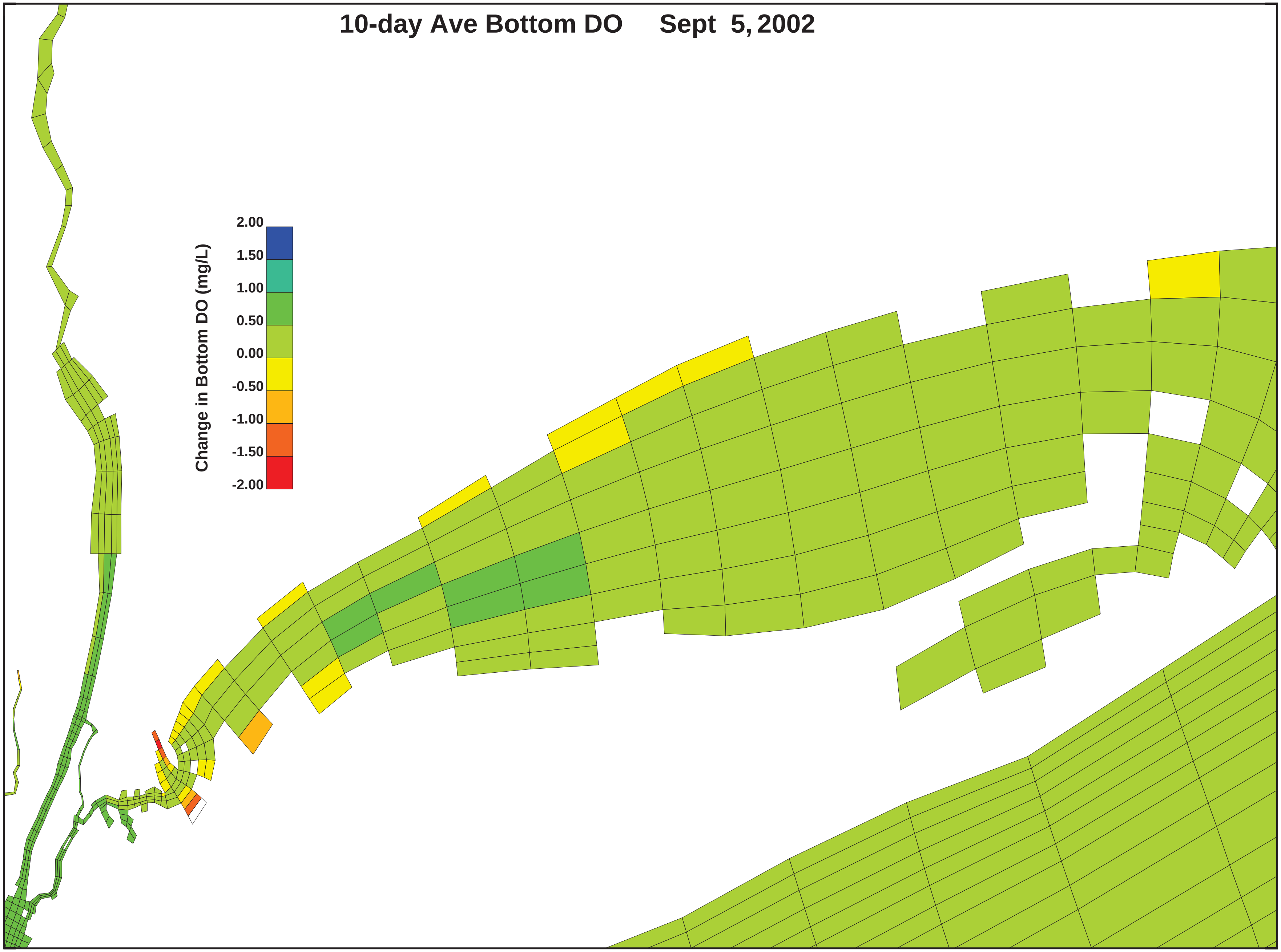
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Change in Bottom DO (mg/L)
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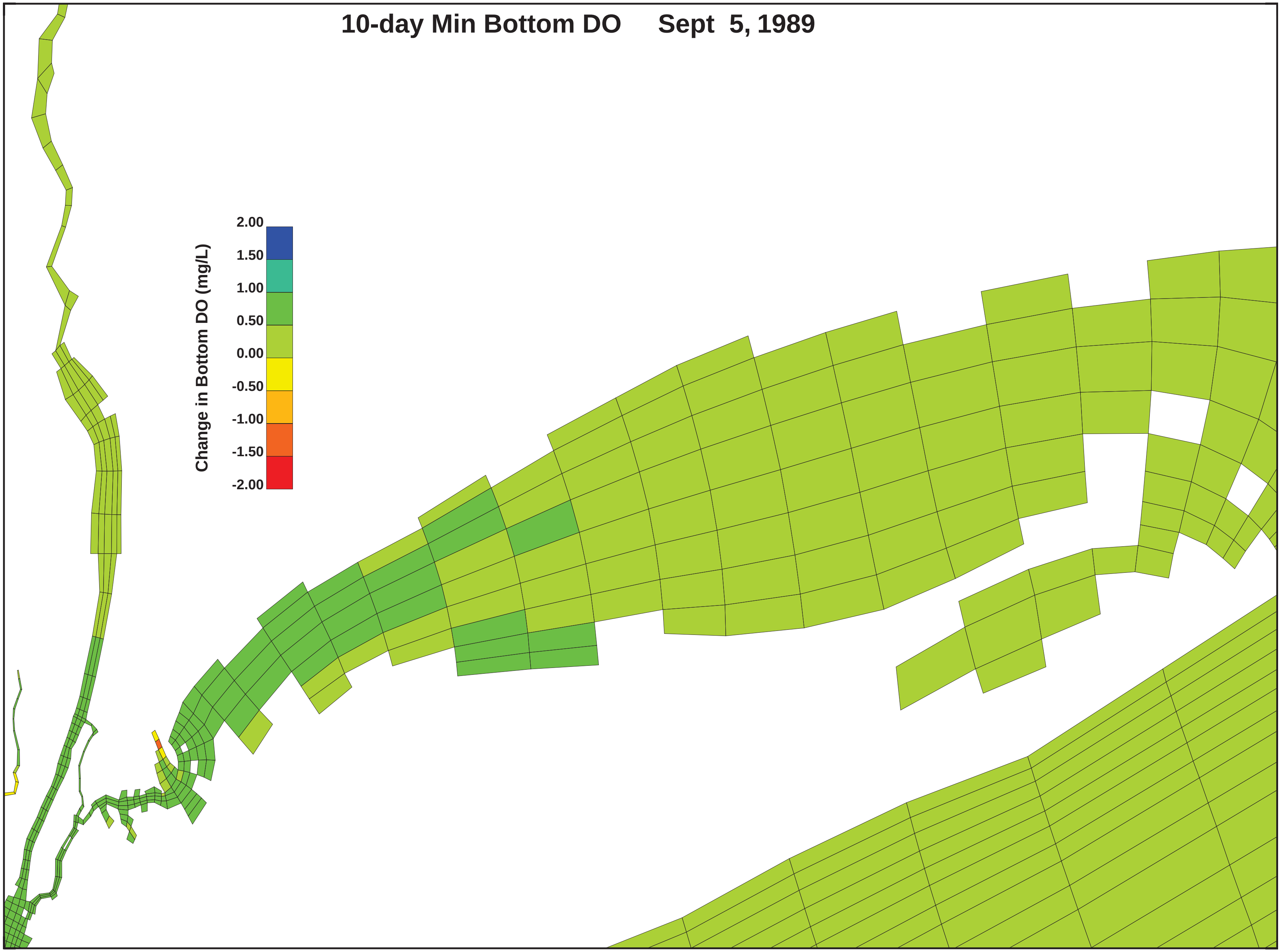
10-day Min Bottom DO   Sept 5, 1989
Change in Bottom DO (mg/L)
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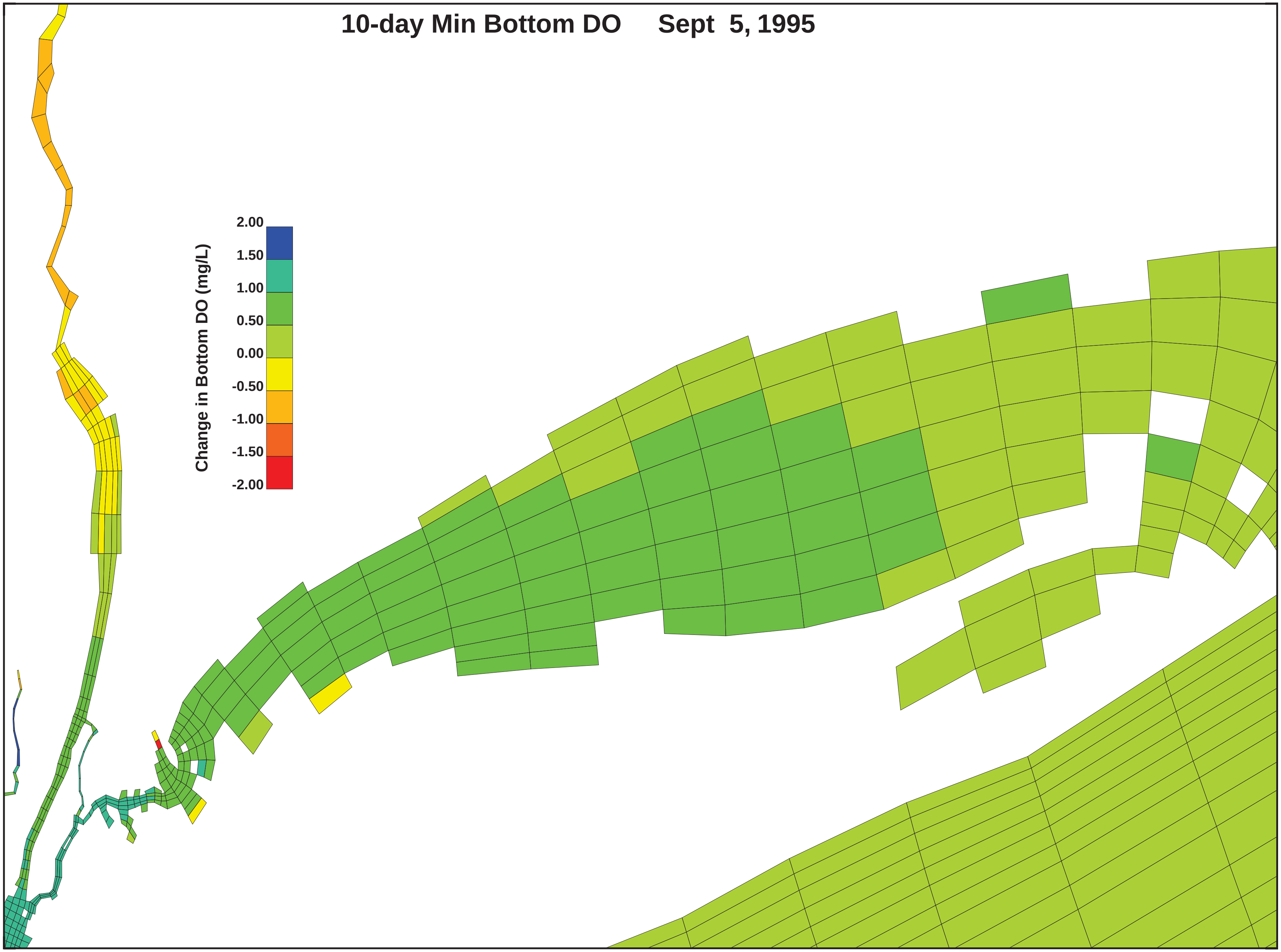

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Change in Bottom DO (mg/L)
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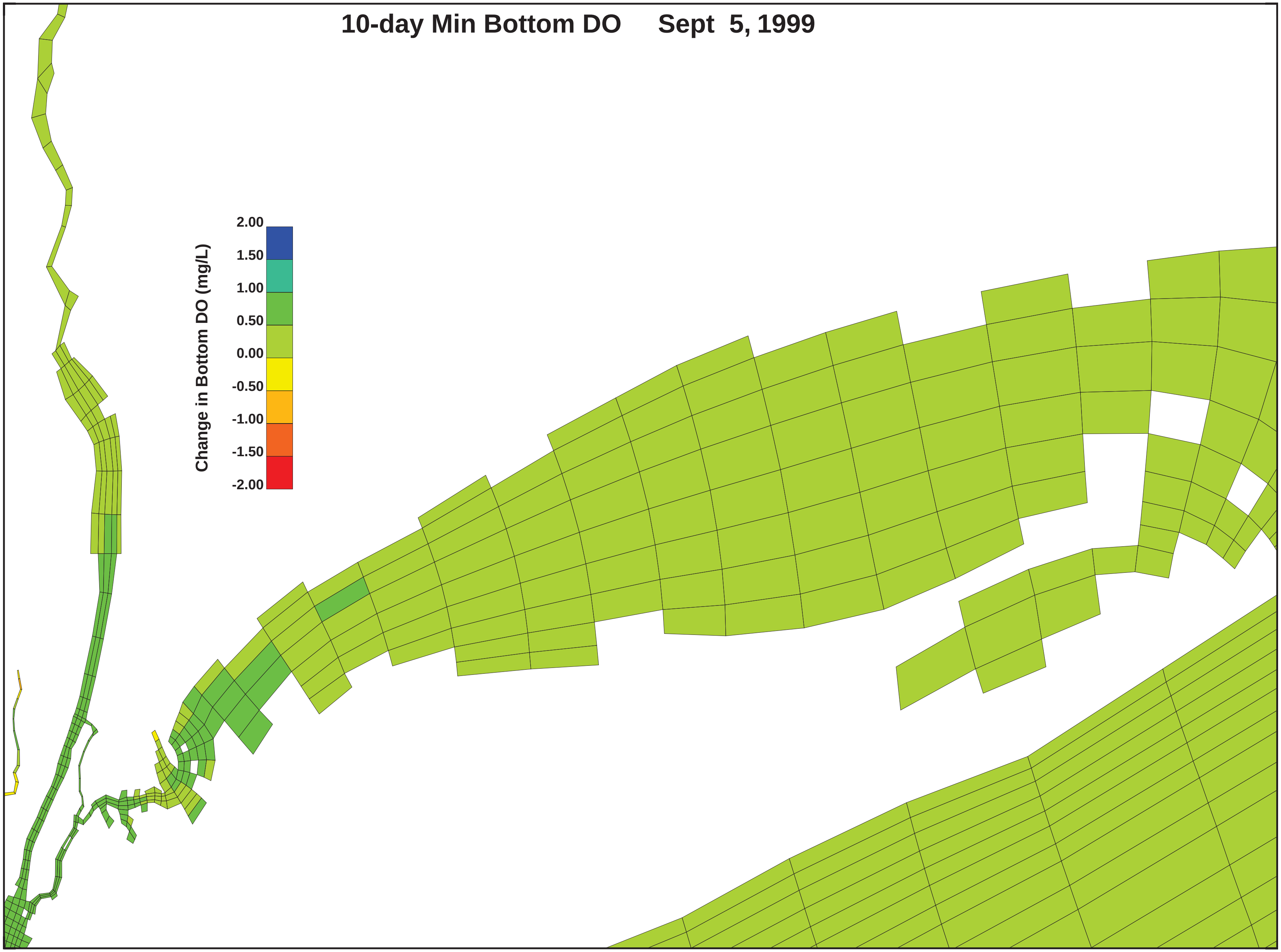

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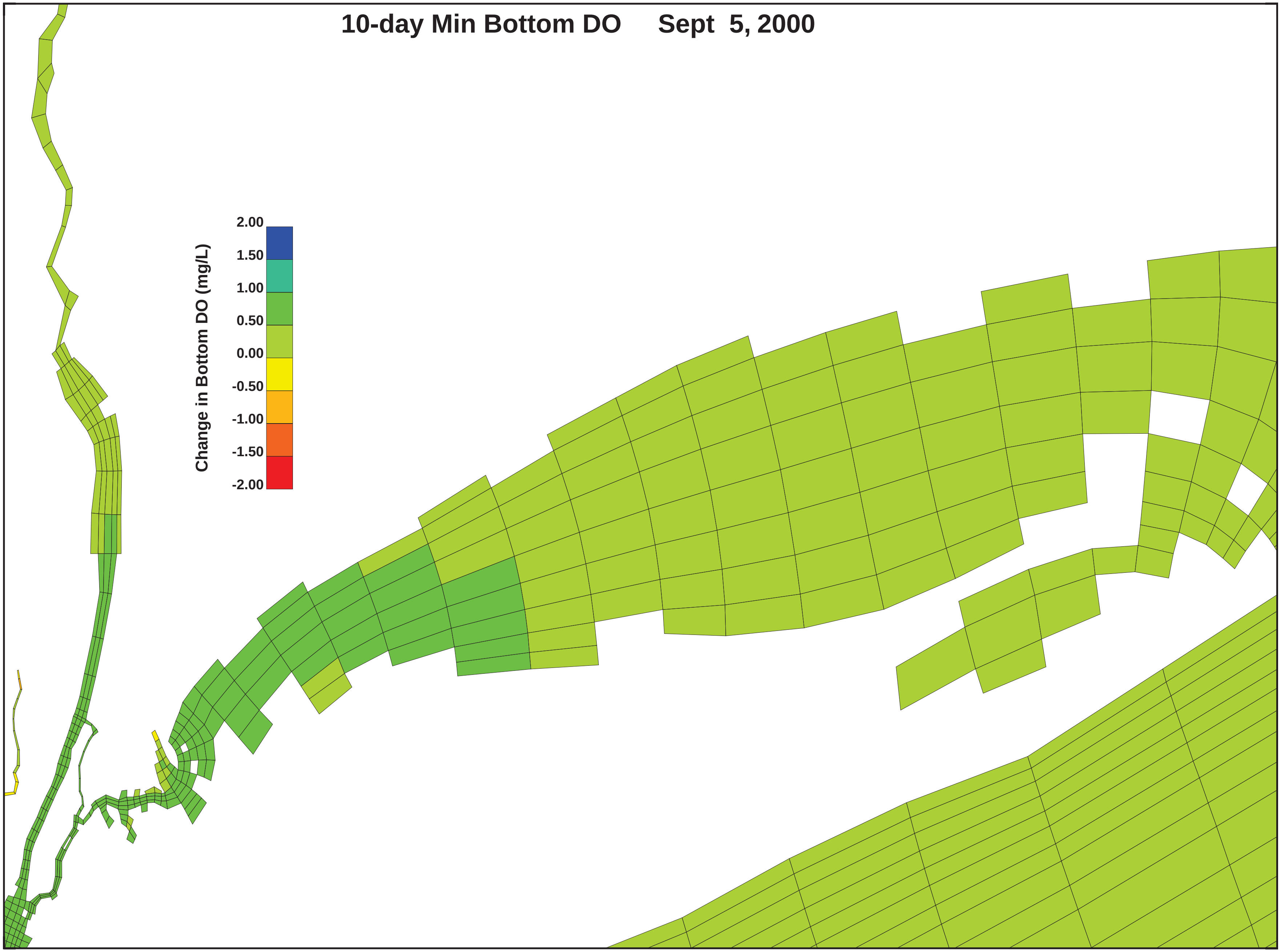
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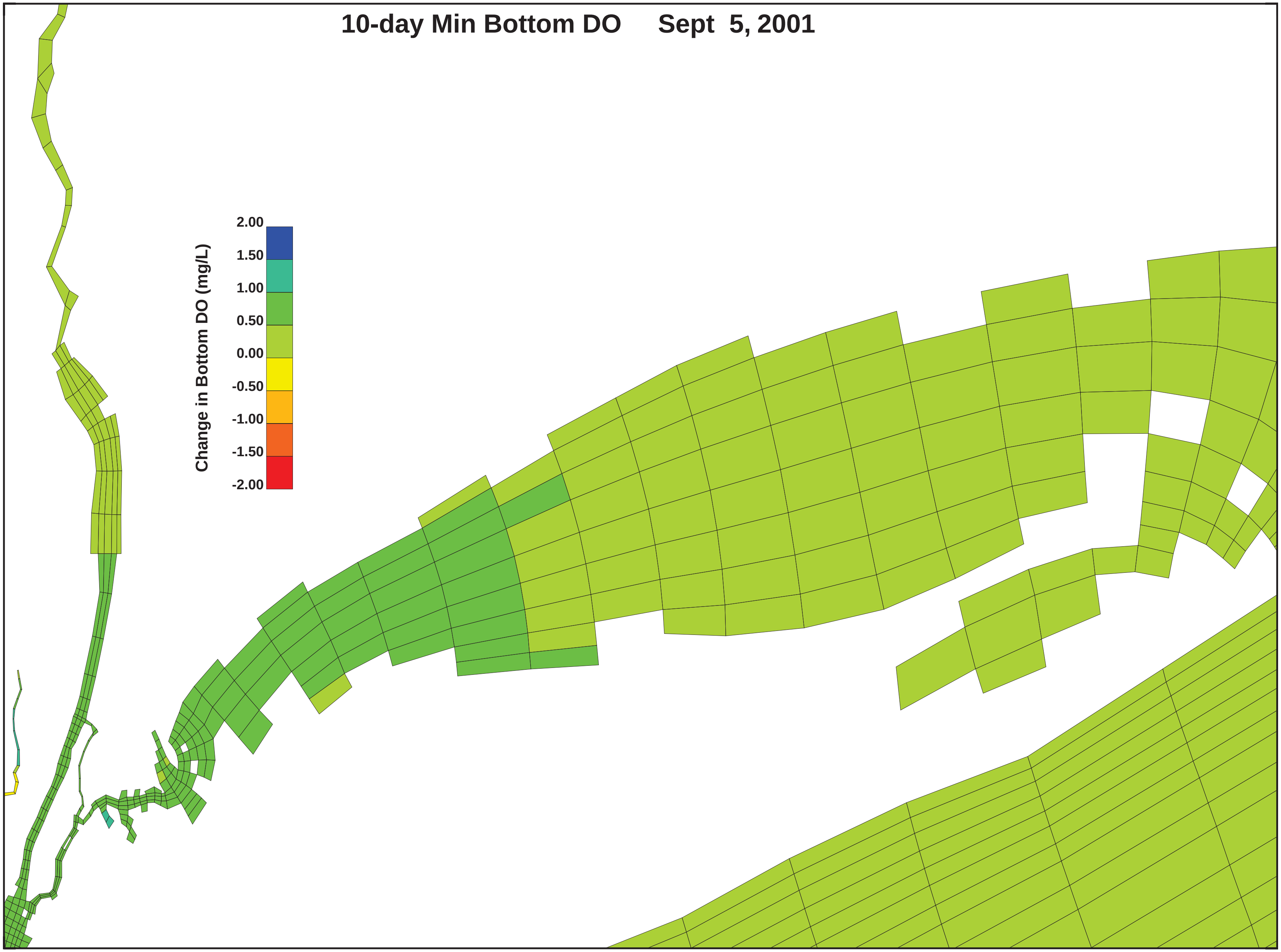
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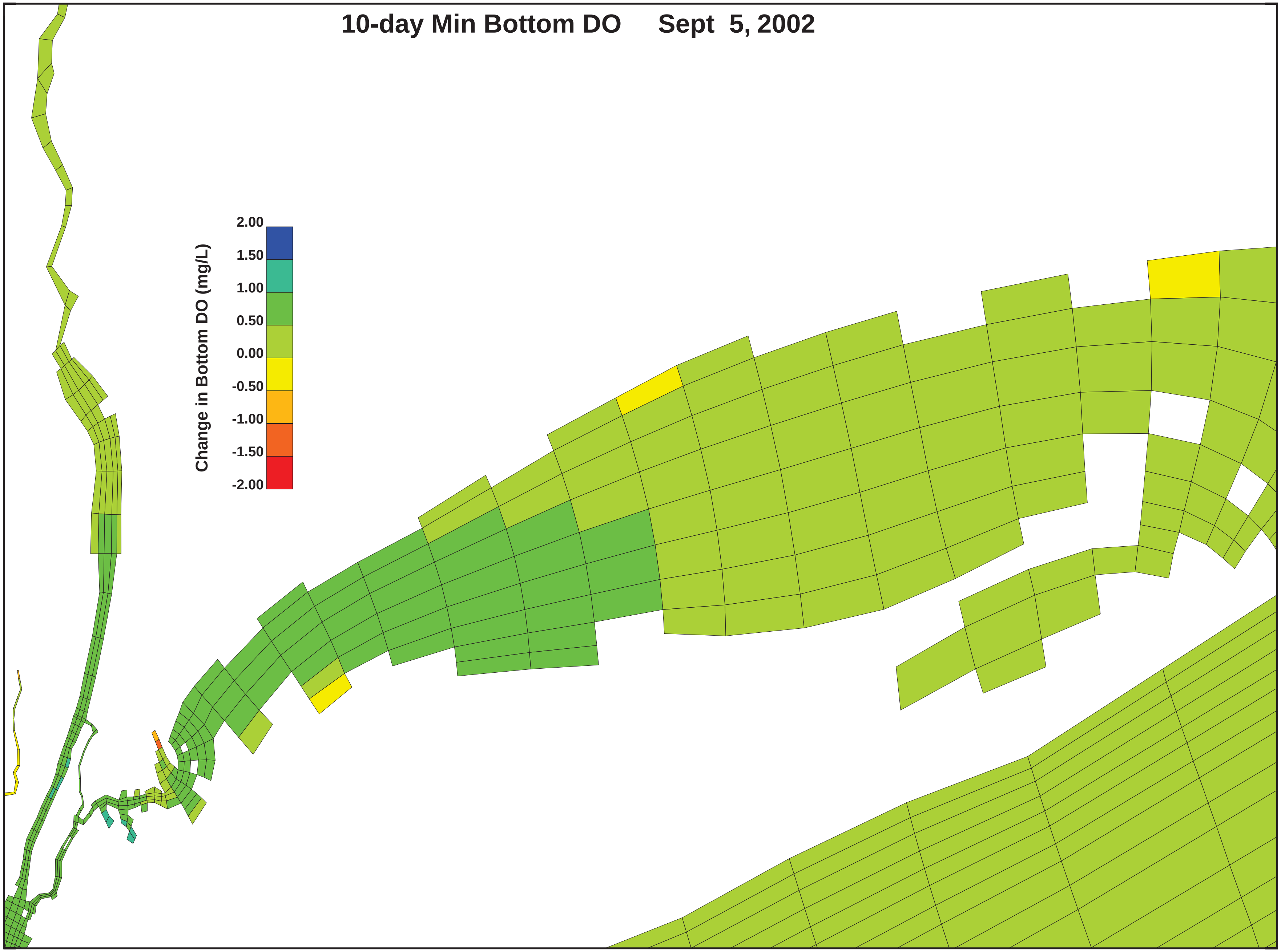
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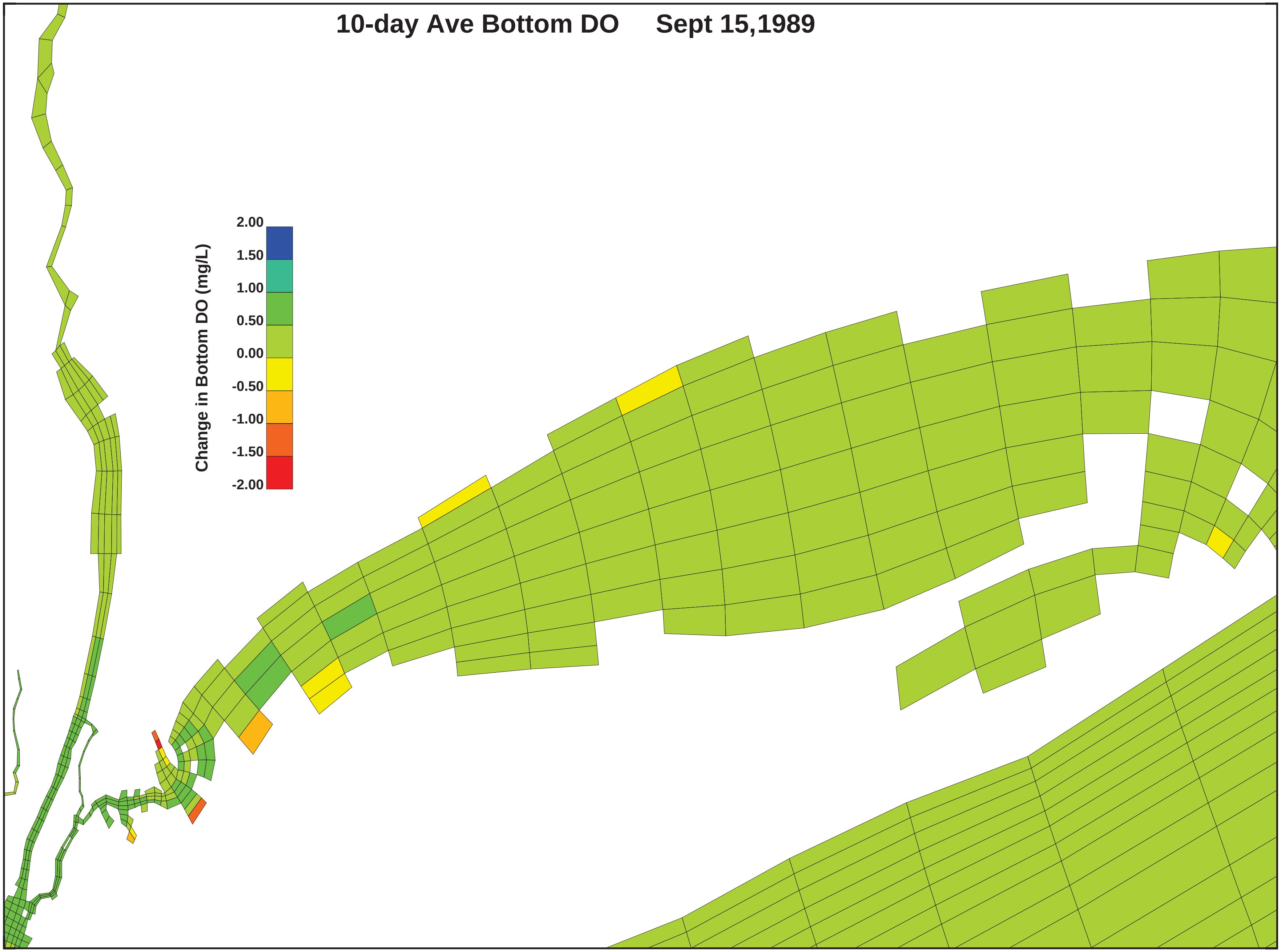
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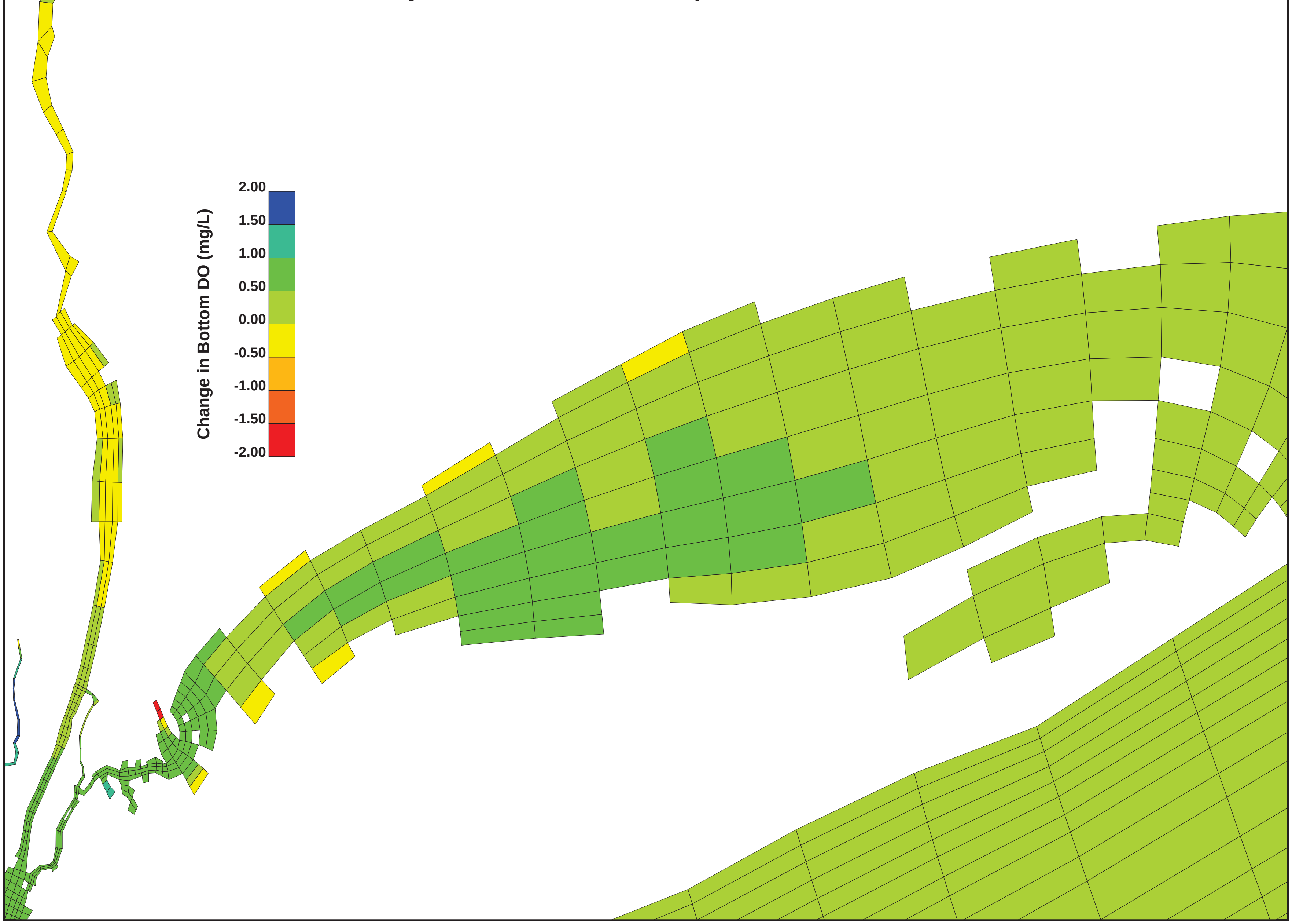
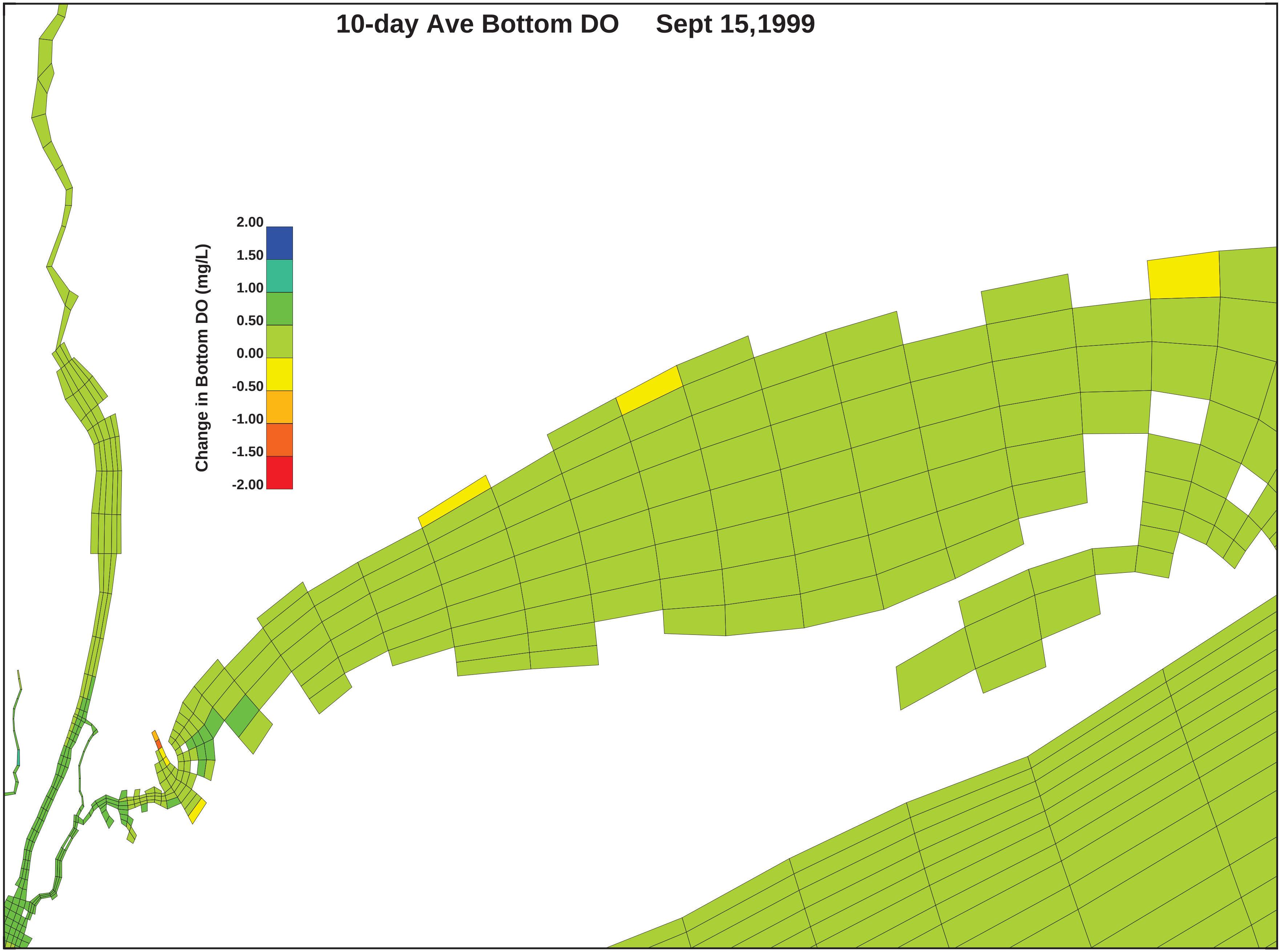
10-day Ave Bottom DO   Sept 15,1995

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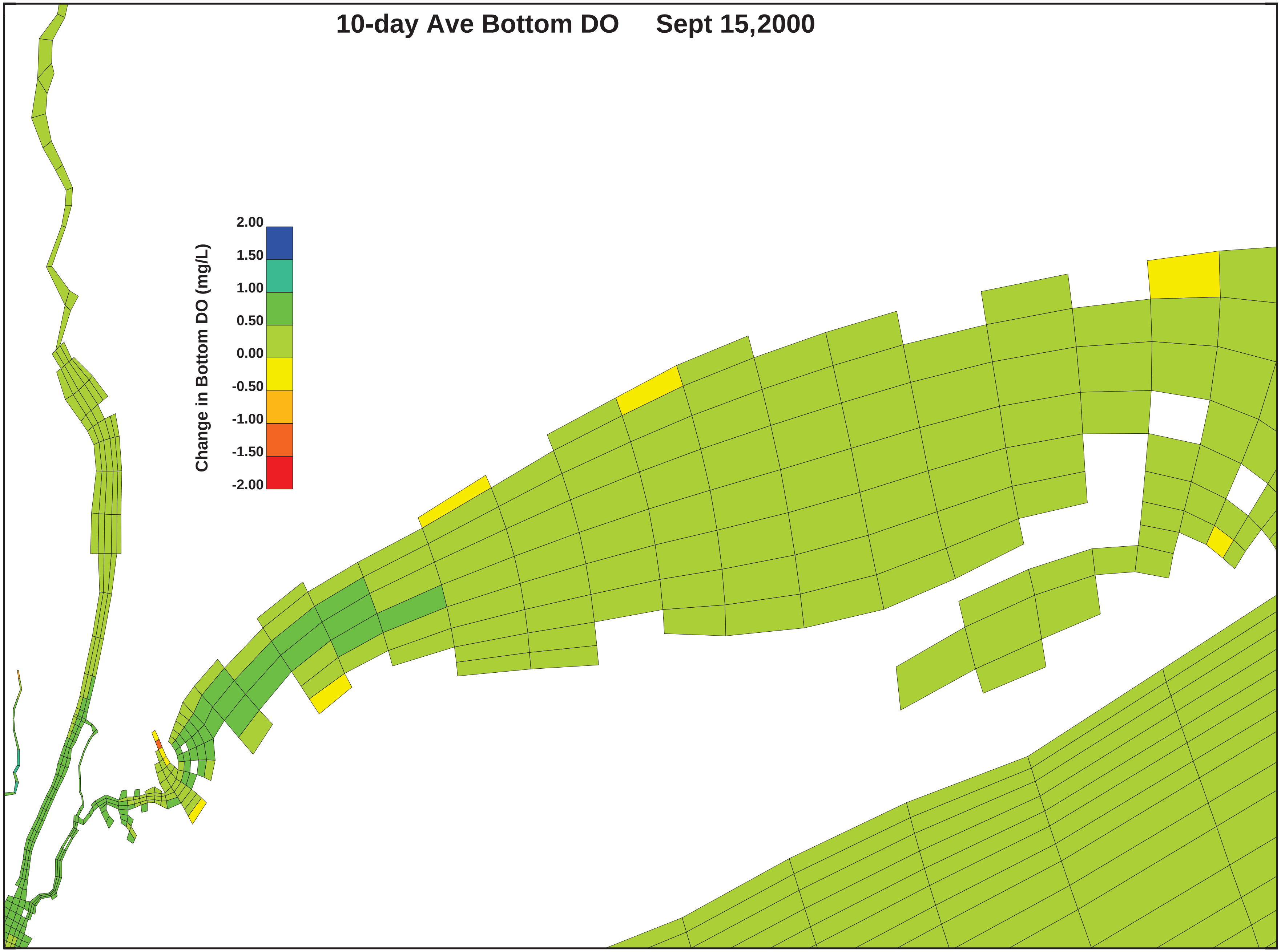

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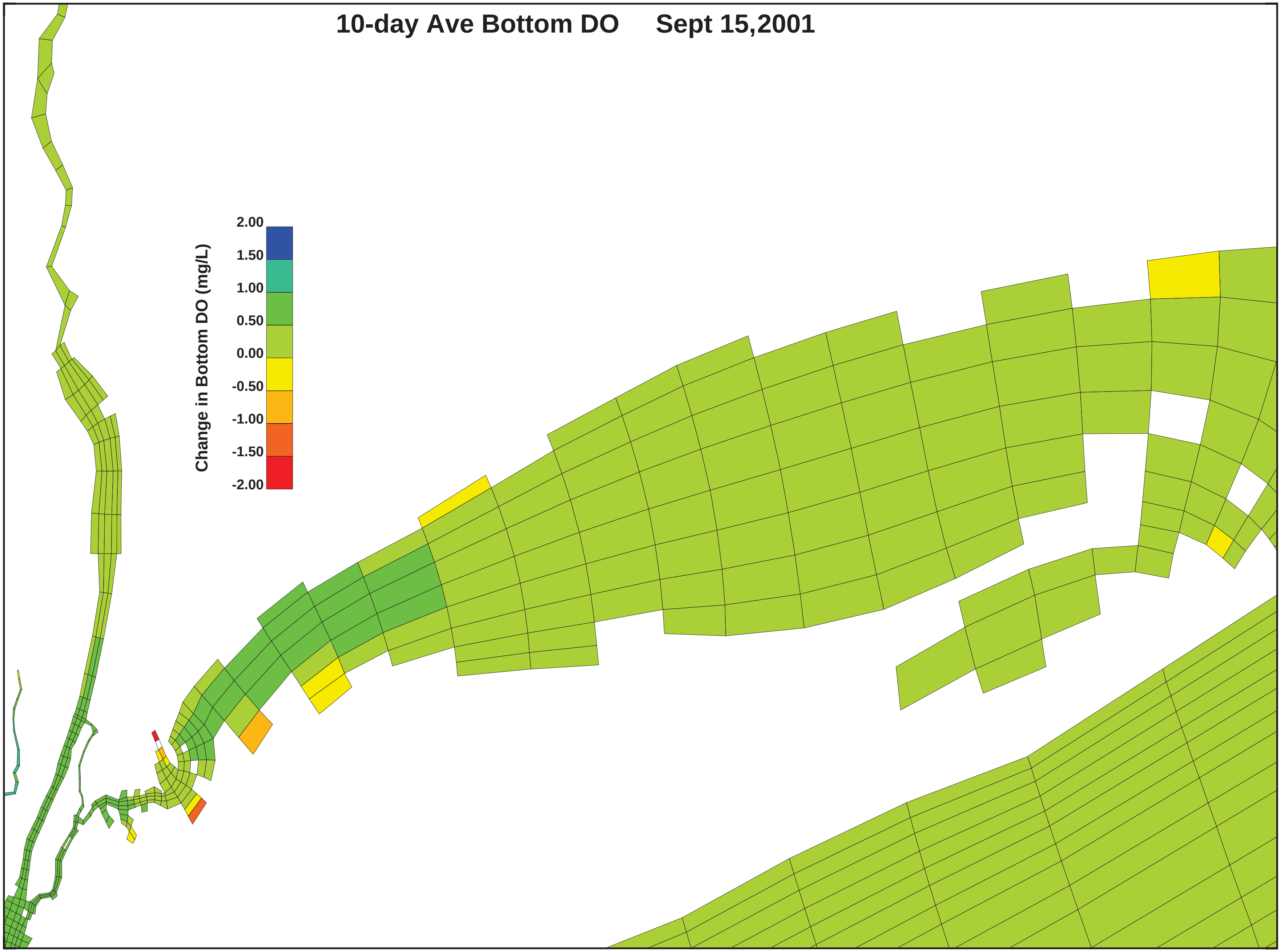
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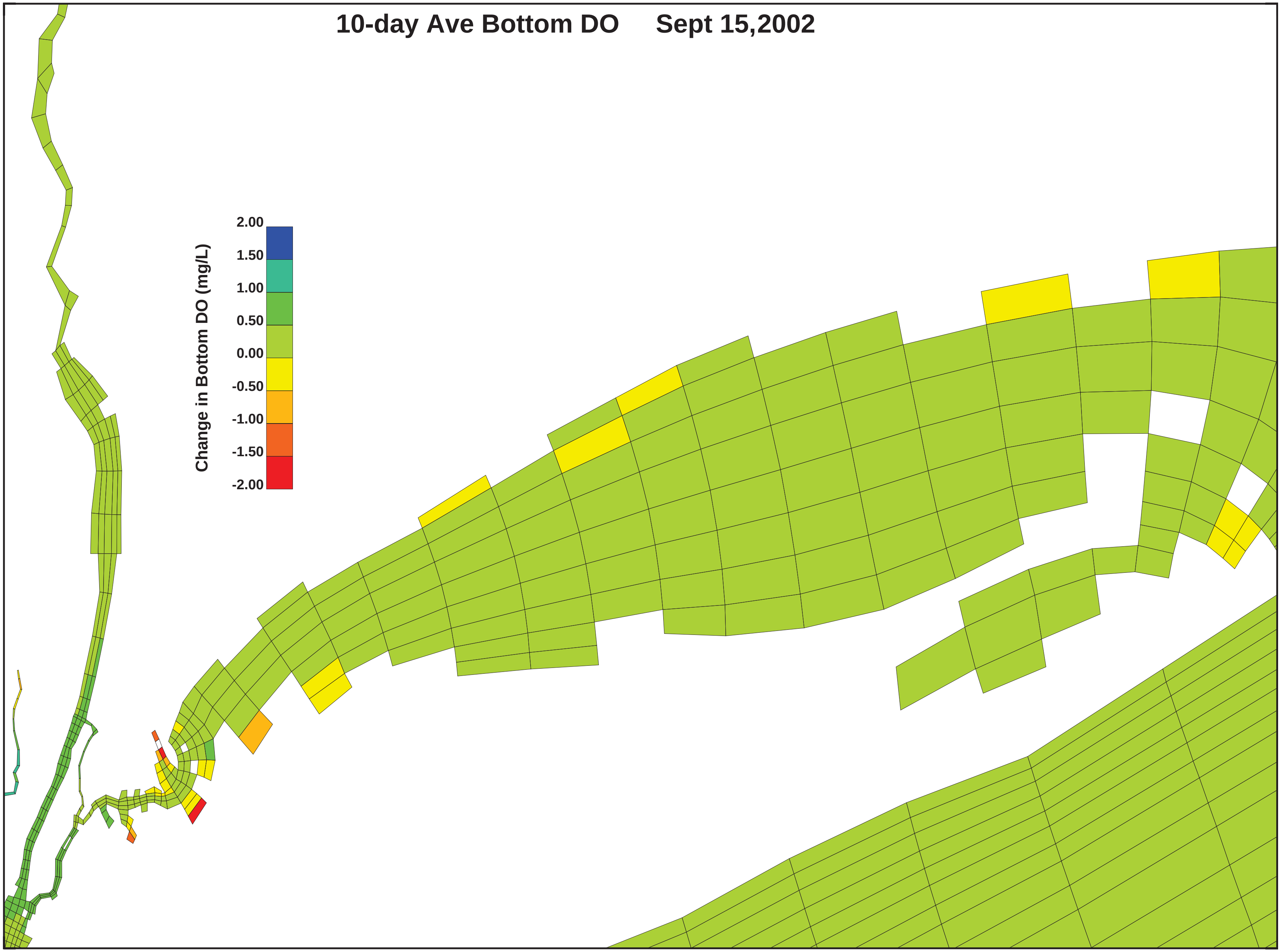
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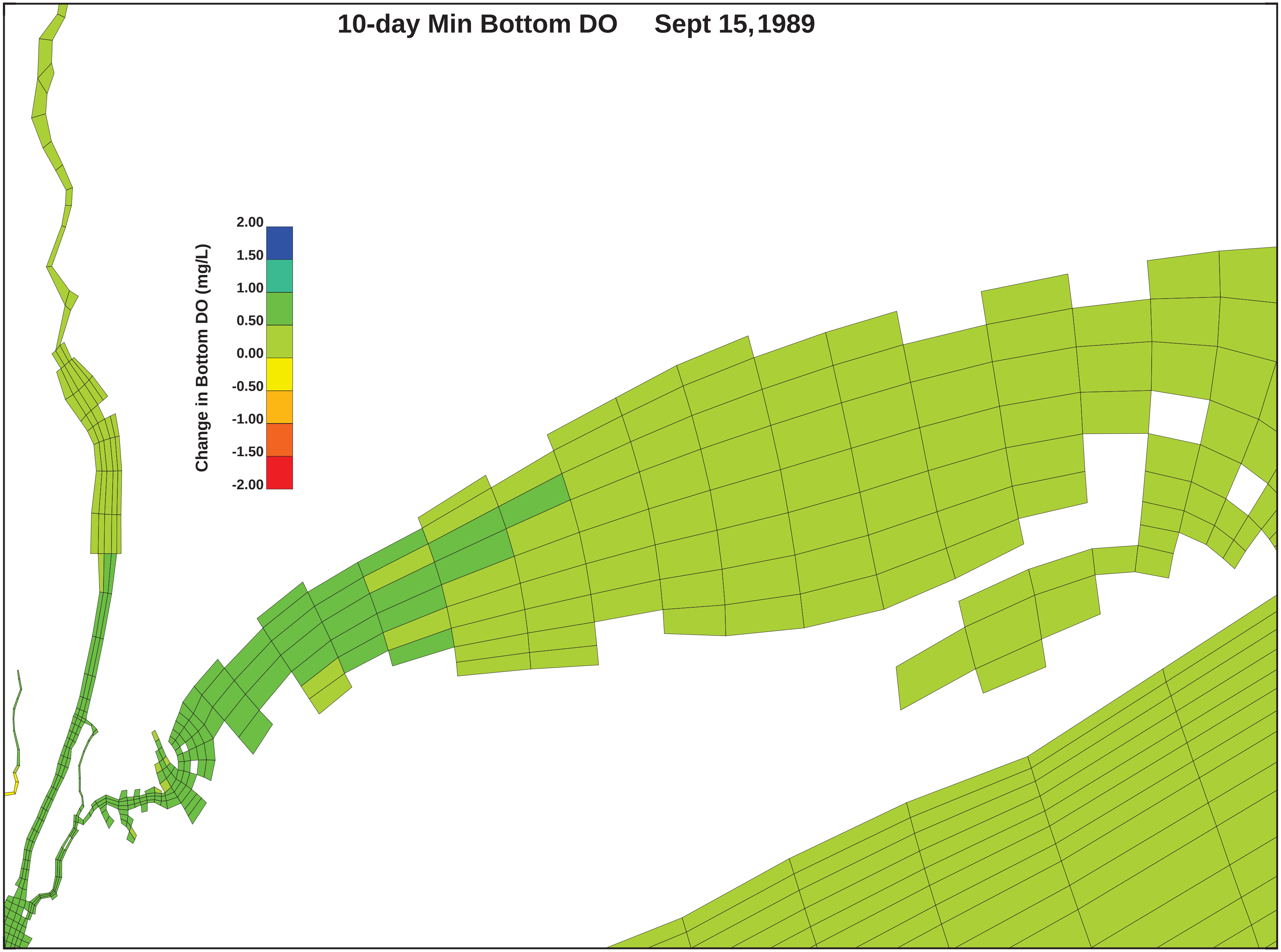
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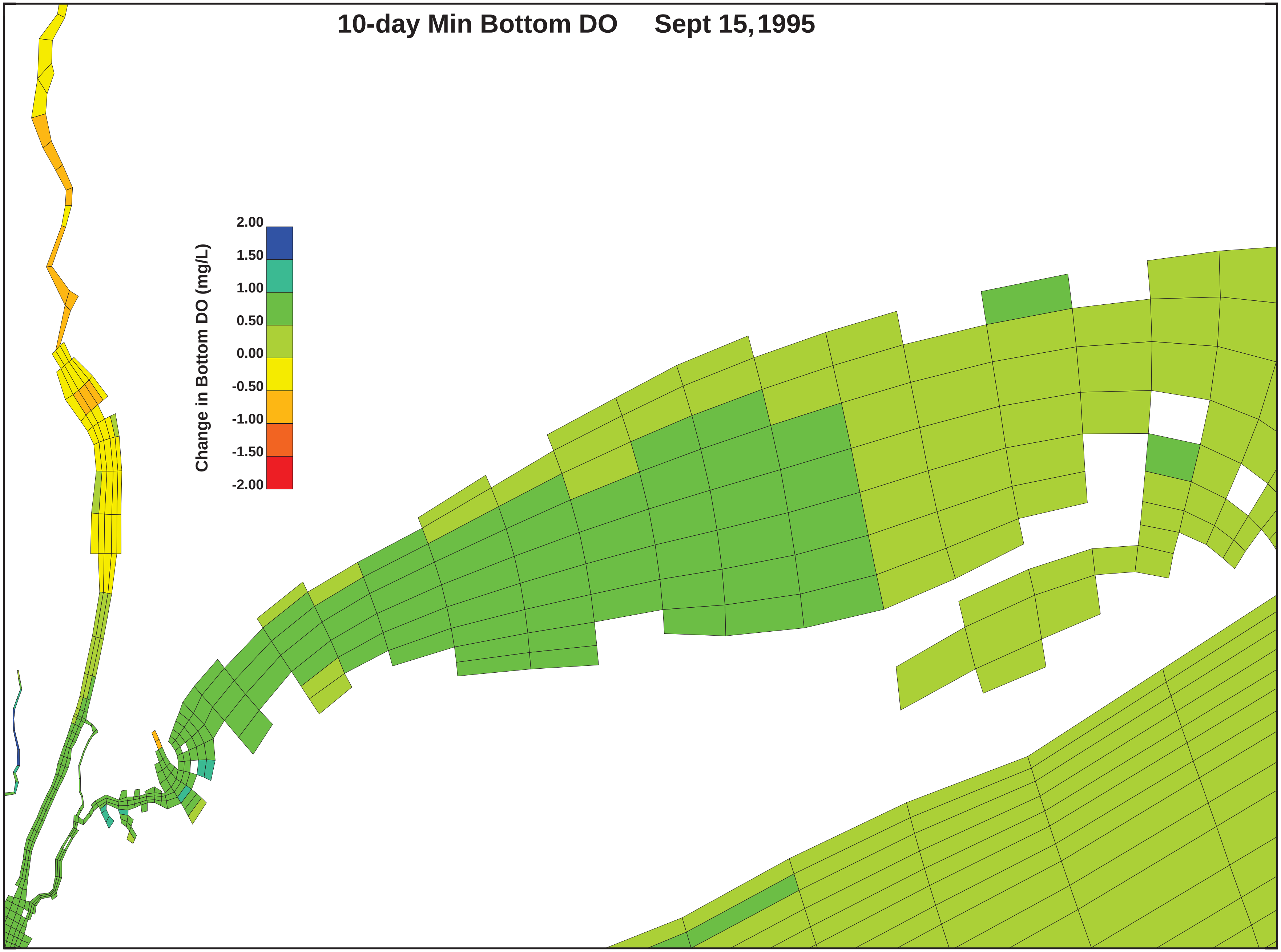
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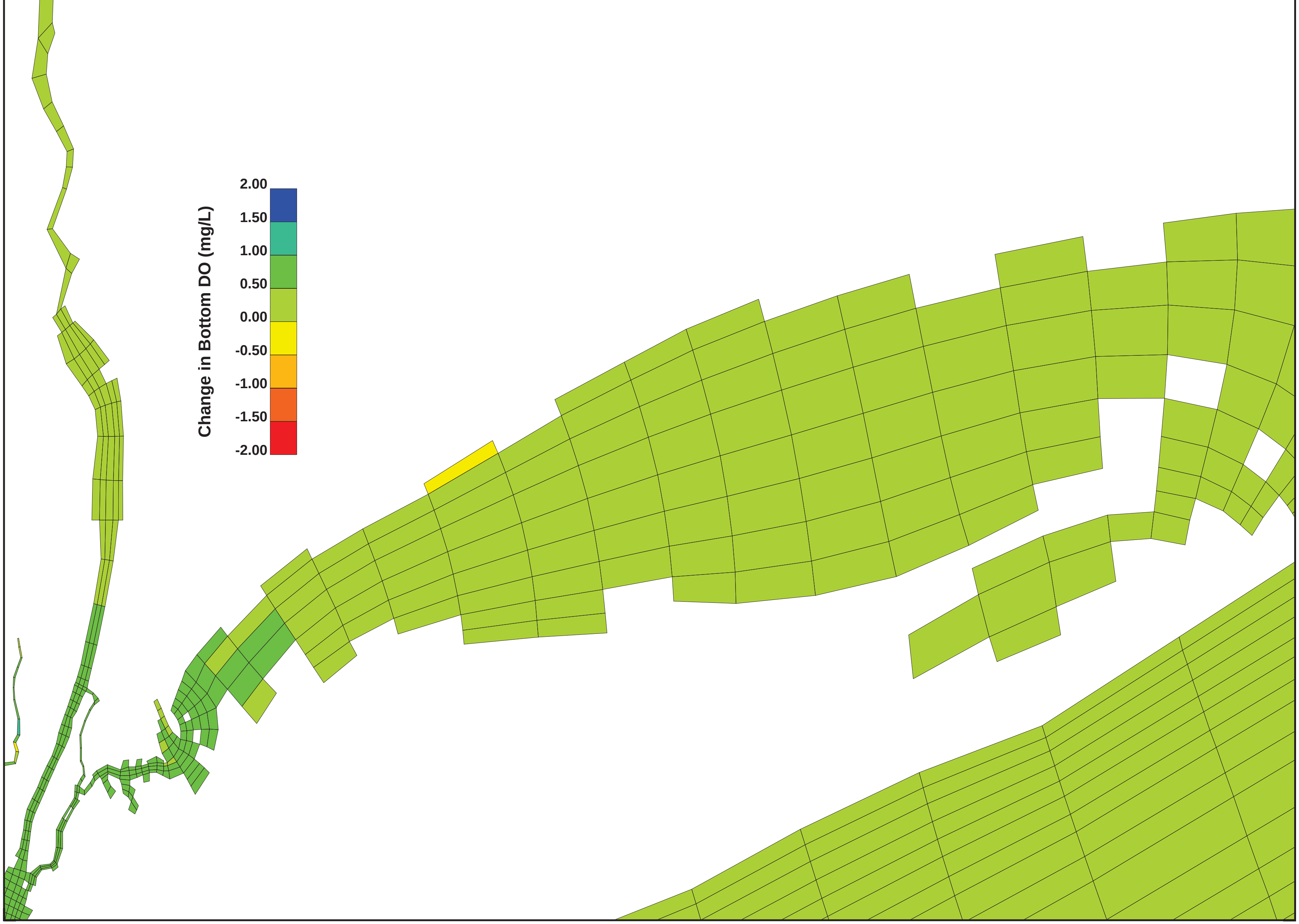
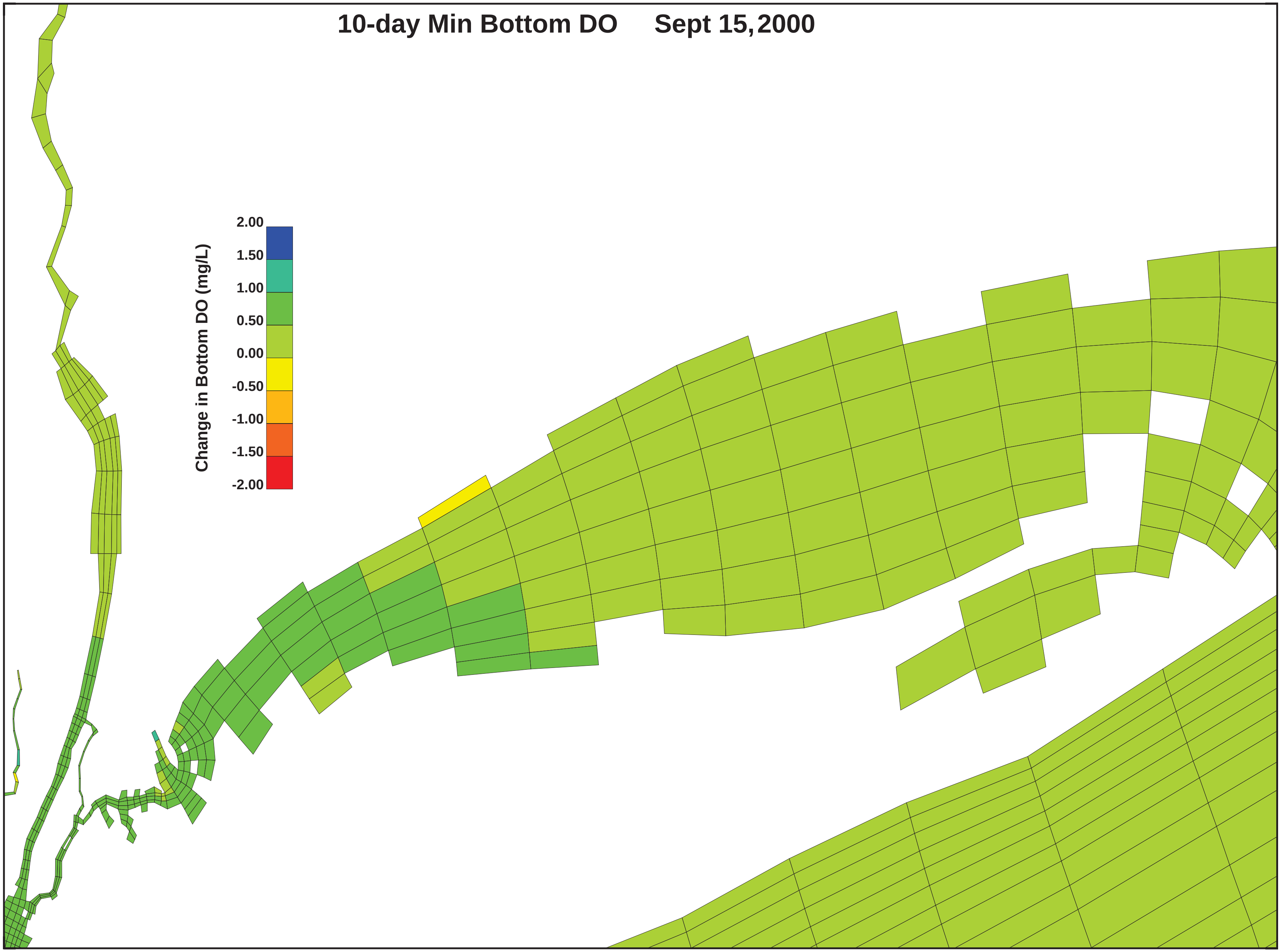
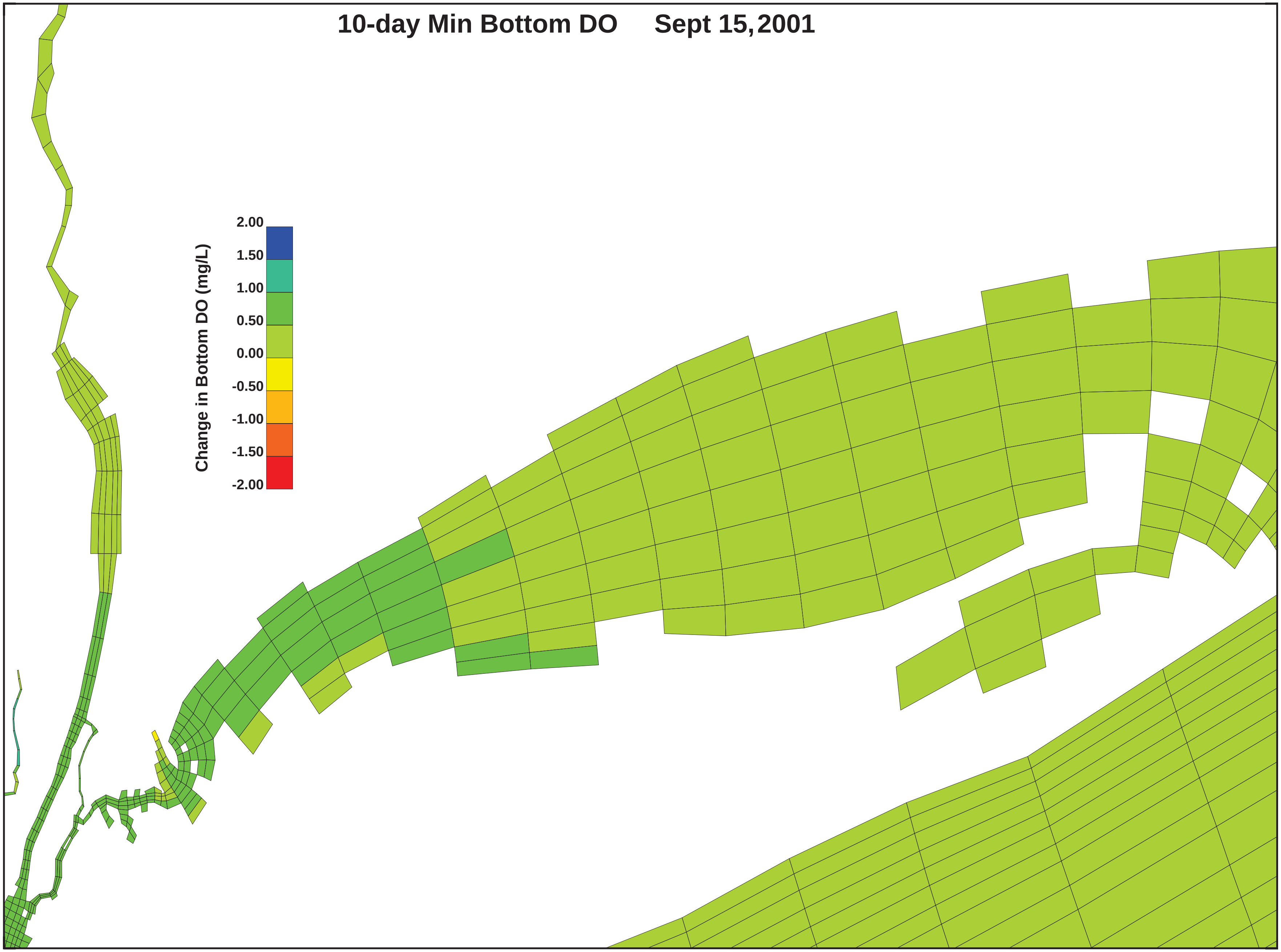
-2.00

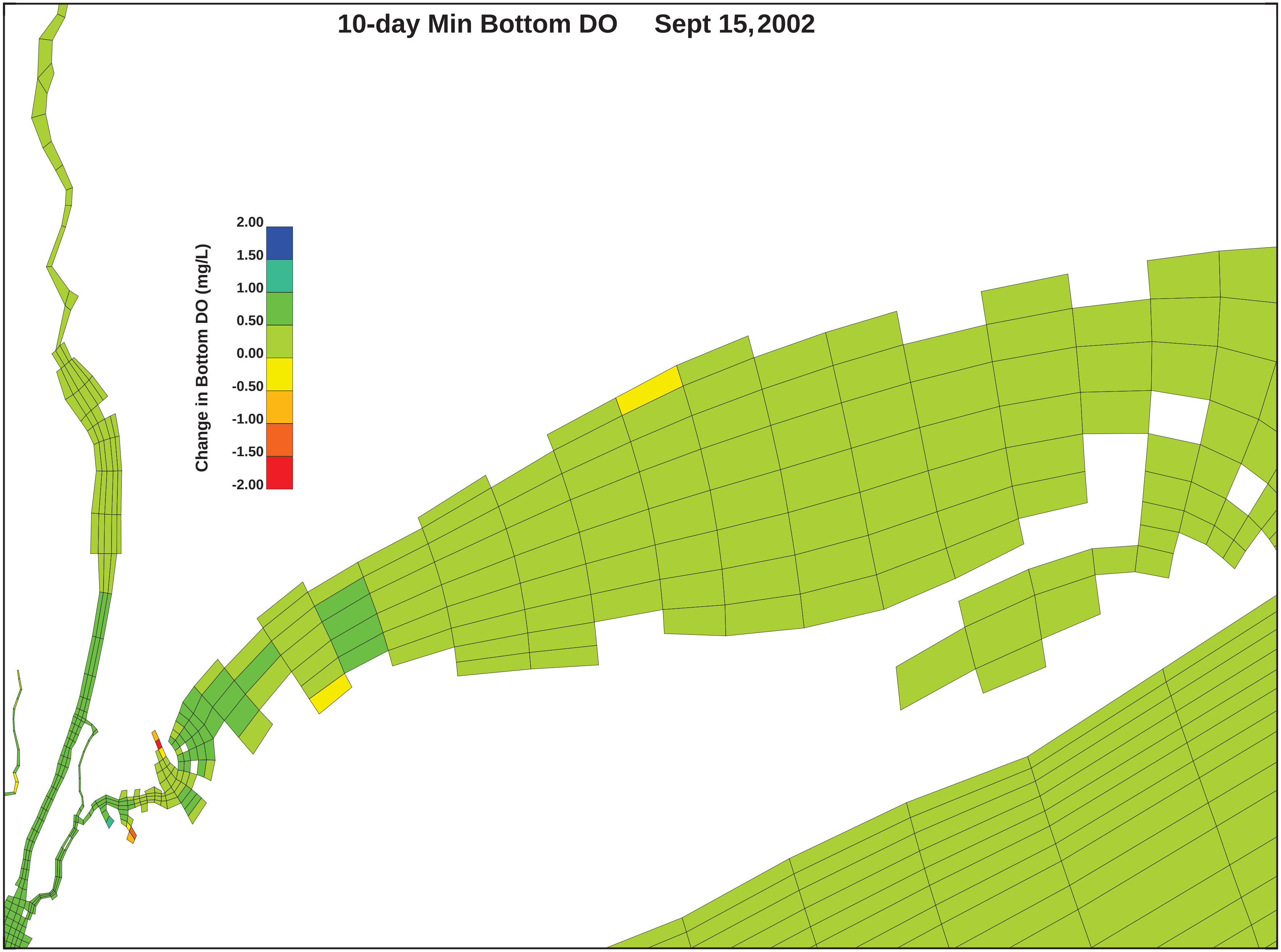
10-day Min Bottom DO Sept 15,1999

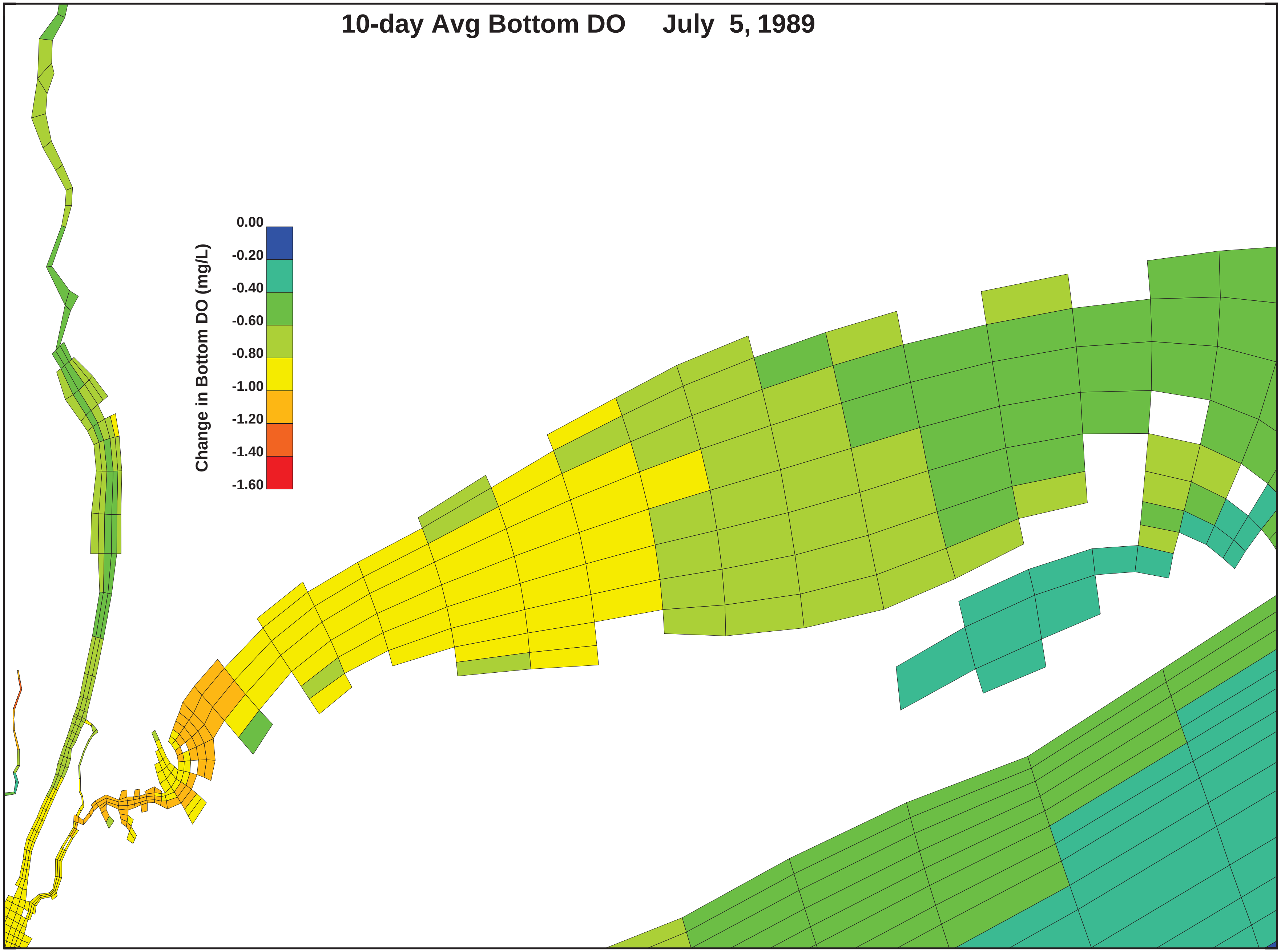




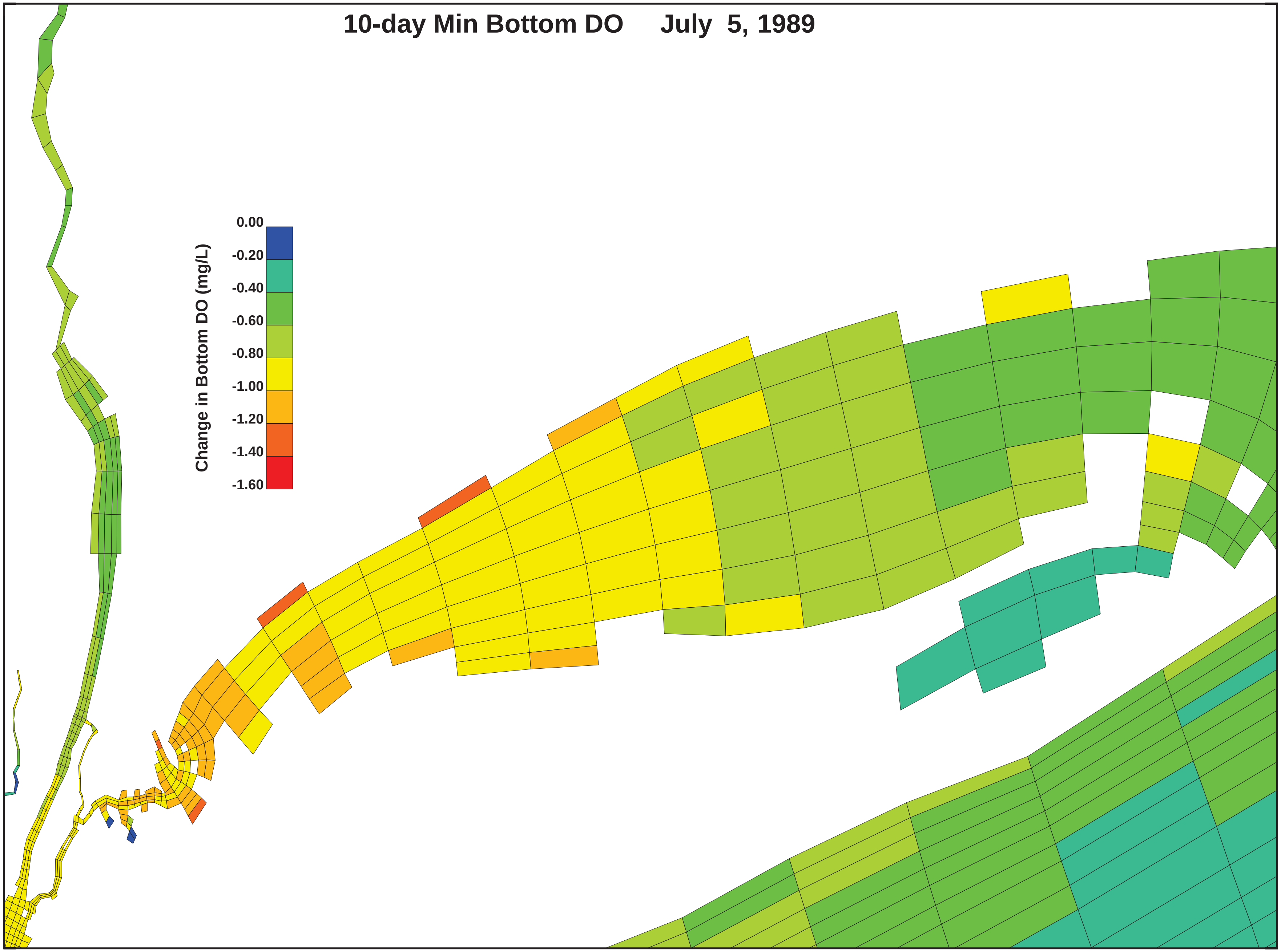
10-day Min Bottom DO Sept 15, 2000
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00

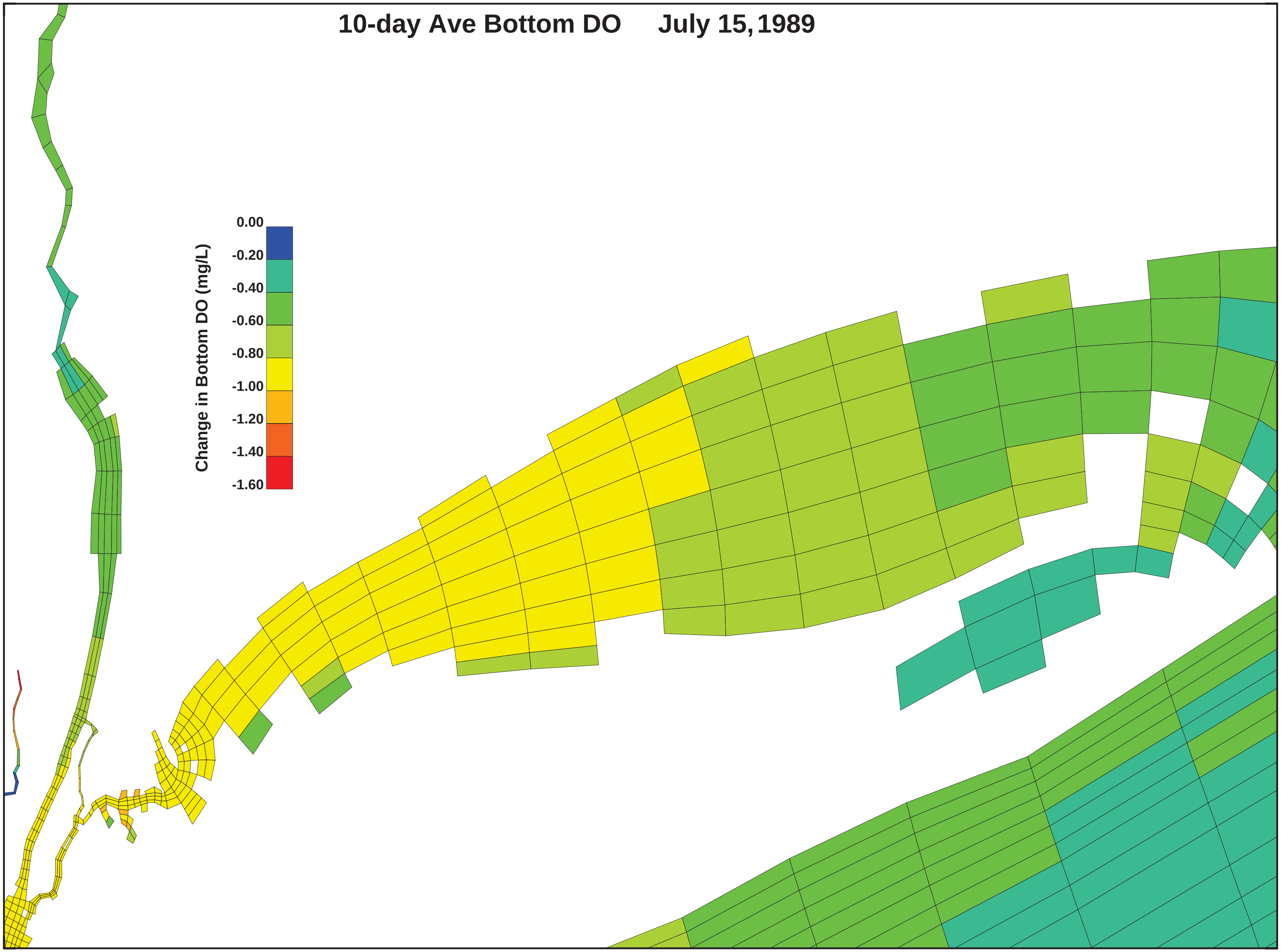

10-day Min Bottom DO Sept 15, 2001
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00


10-day Min Bottom DO Sept 15, 2002
Change in Bottom DO (mg/L)
2.00
1.50
1.00
0.50
0.00
-0.50
-1.00
-1.50
-2.00

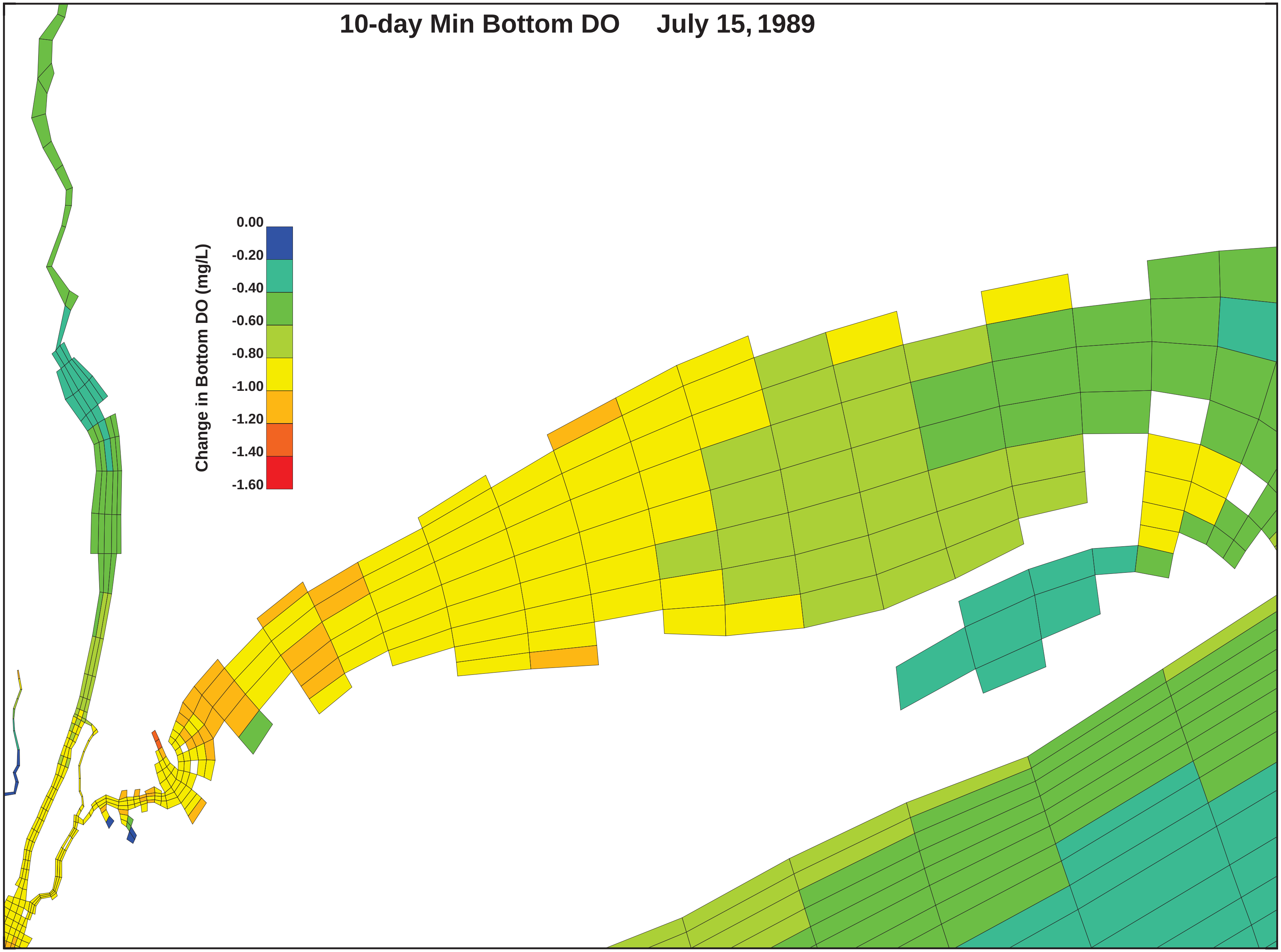


10-day Avg Bottom DO July 5, 1989
Change in Bottom DO (mg/L)
0.00
-0.20
-0.40
-0.60
-0.80
-1.00
-1.20
-1.40
-1.60

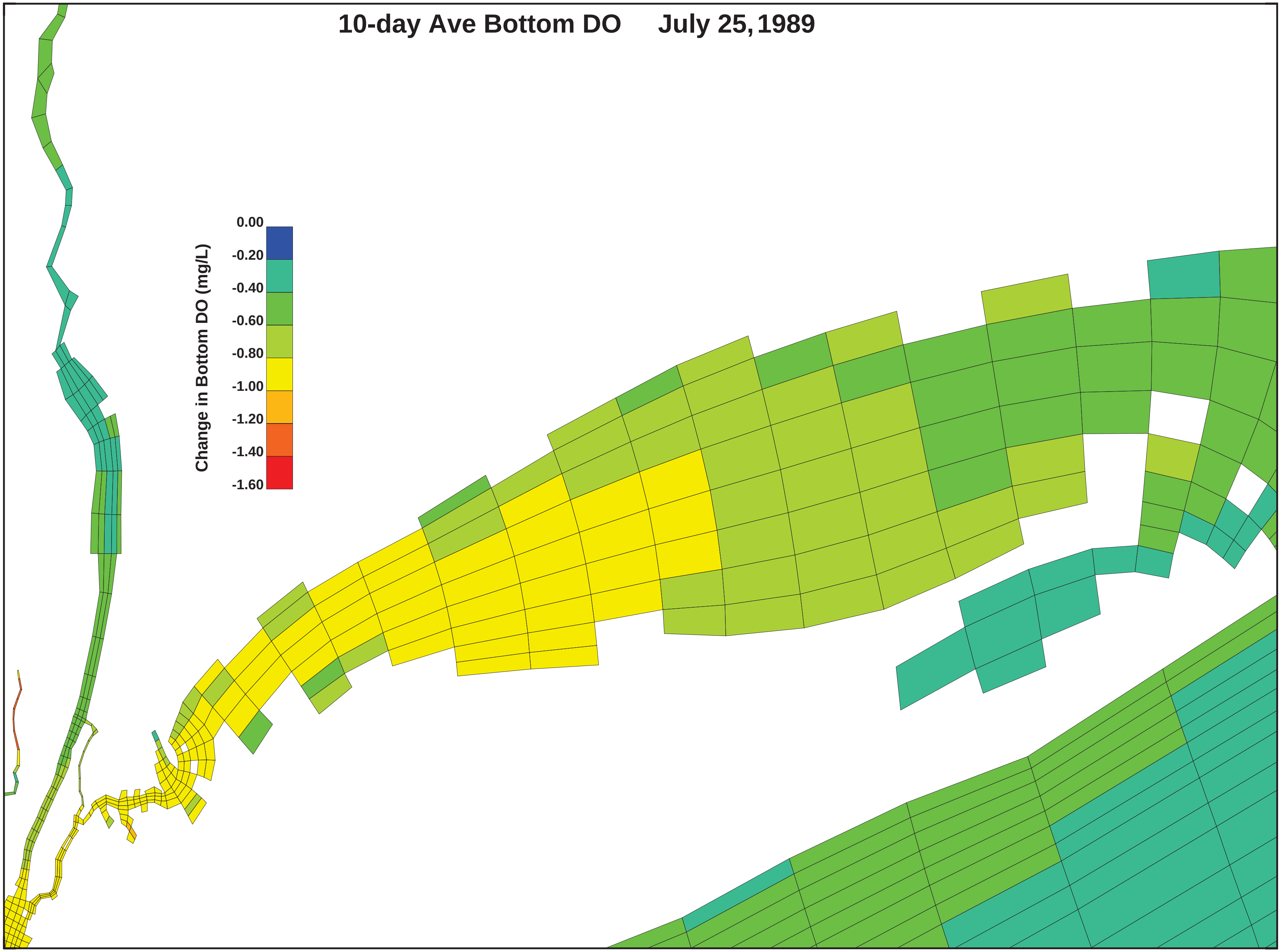

10-day Min Bottom DO July 5, 1989
Change in Bottom DO (mg/L)
0.00
-0.20
-0.40
-0.60
-0.80
-1.00
-1.20
-1.40
-1.60

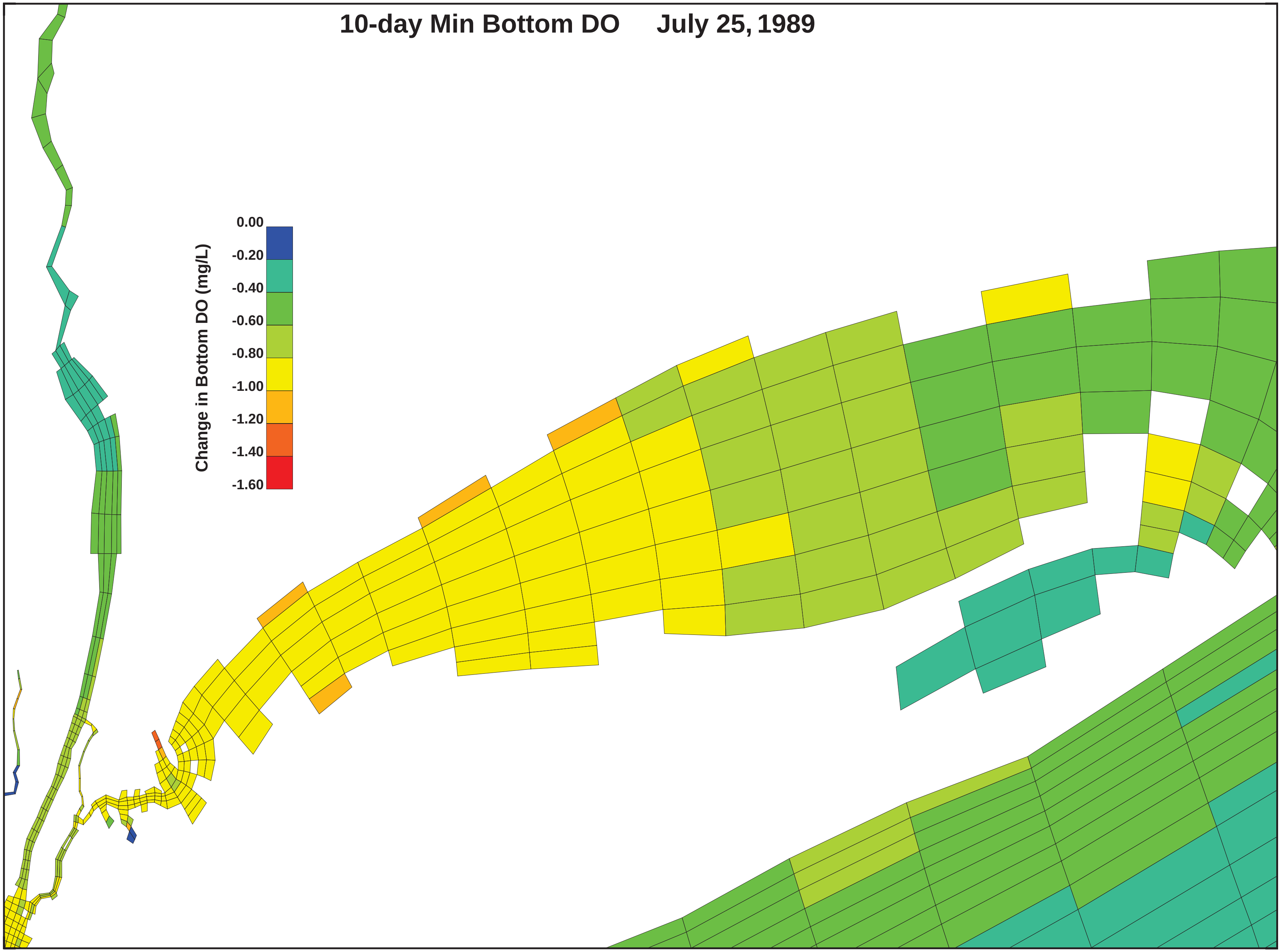


10-day Ave Bottom DO July 15, 1989
Change in Bottom DO (mg/L)
0.00
-0.20
-0.40
-0.60
-0.80
-1.00
-1.20
-1.40
-1.60

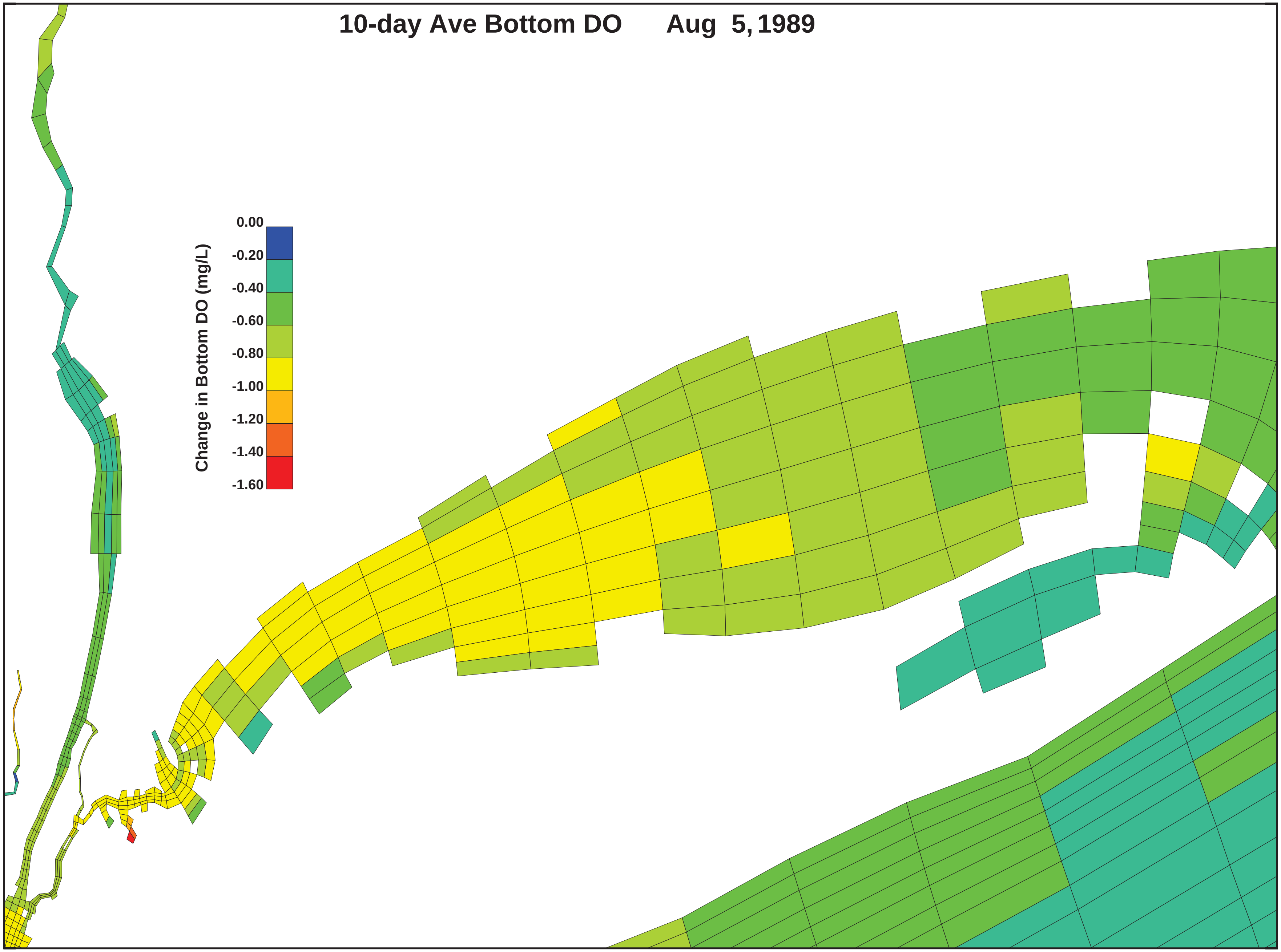


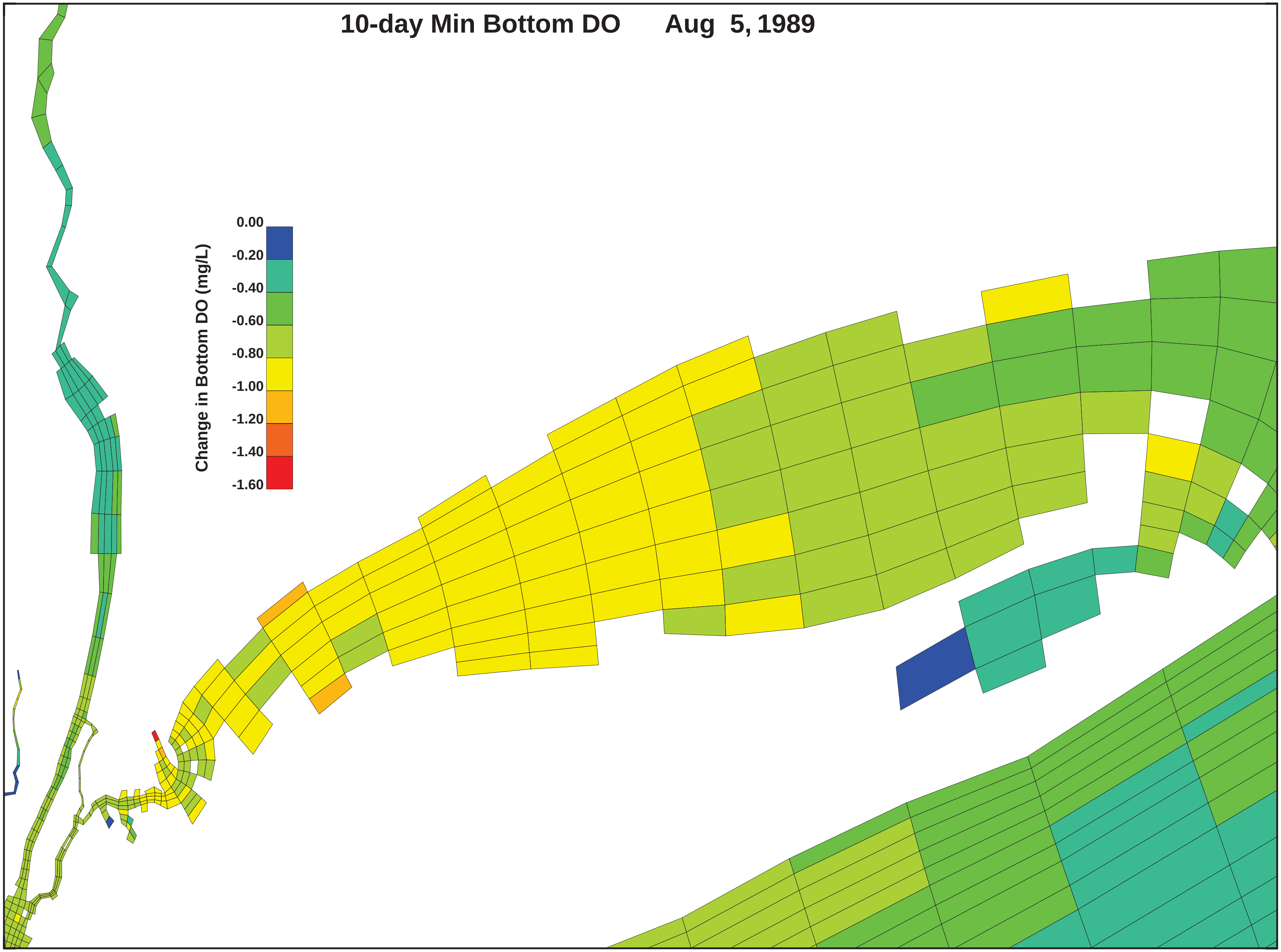
10-day Min Bottom DO July 15, 1989
Change in Bottom DO (mg/L)
0.00
-0.20
-0.40
-0.60
-0.80
-1.00
-1.20
-1.40
-1.60

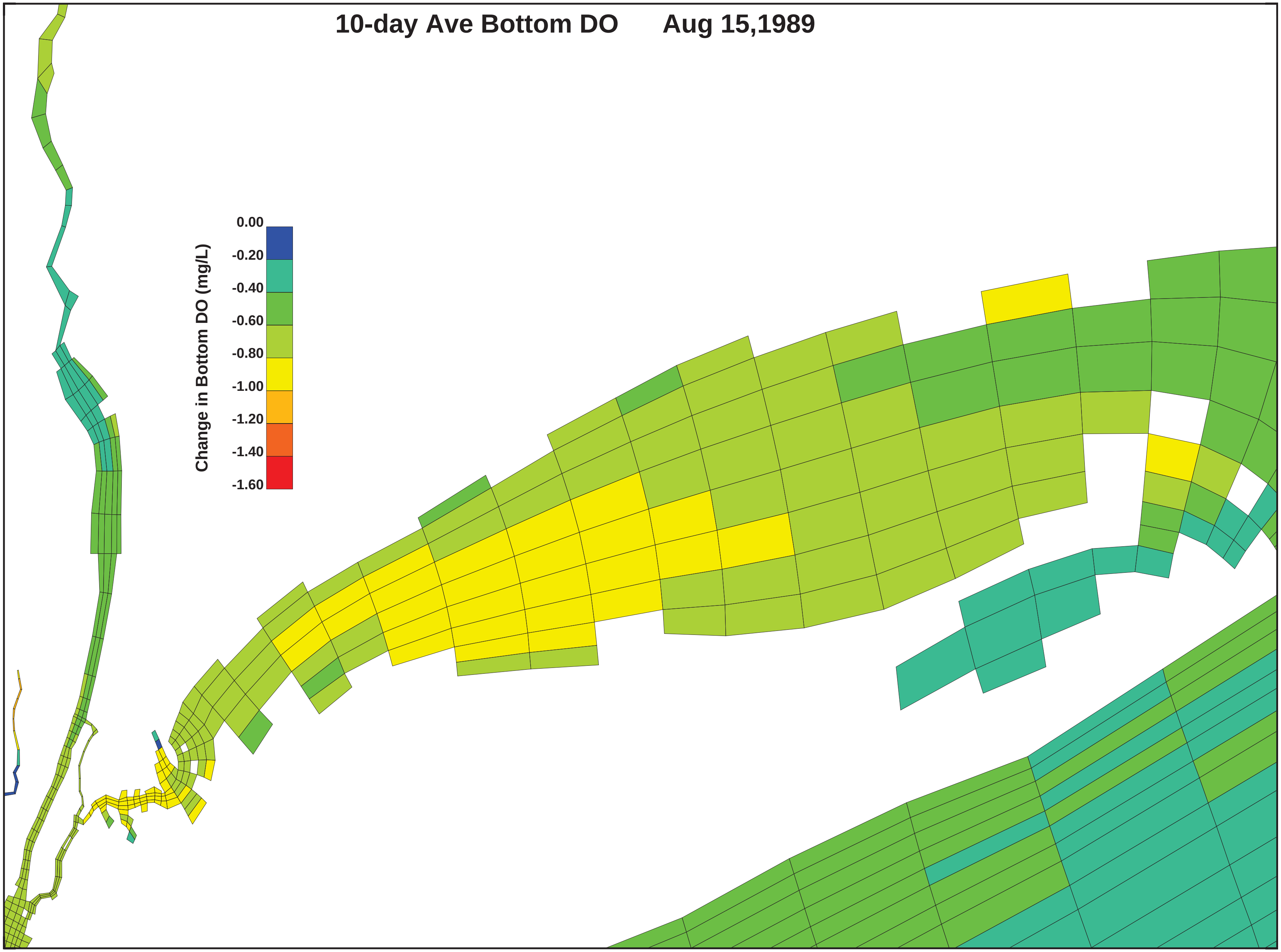

10-day Ave Bottom DO July 25,1989
Change in Bottom DO (mg/L)
0.00
-0.20
-0.40
-0.60
-0.80
-1.00
-1.20
-1.40
-1.60

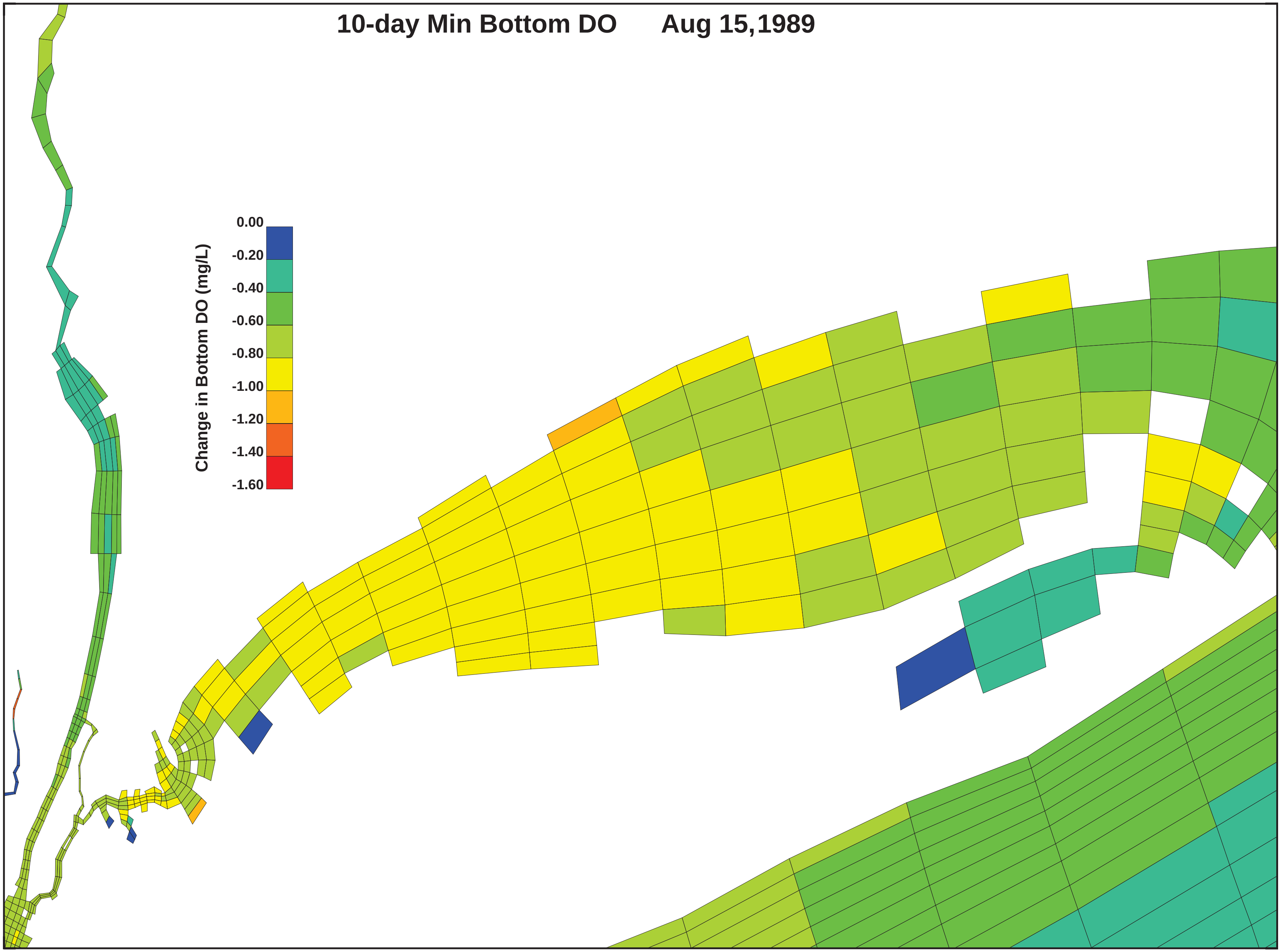


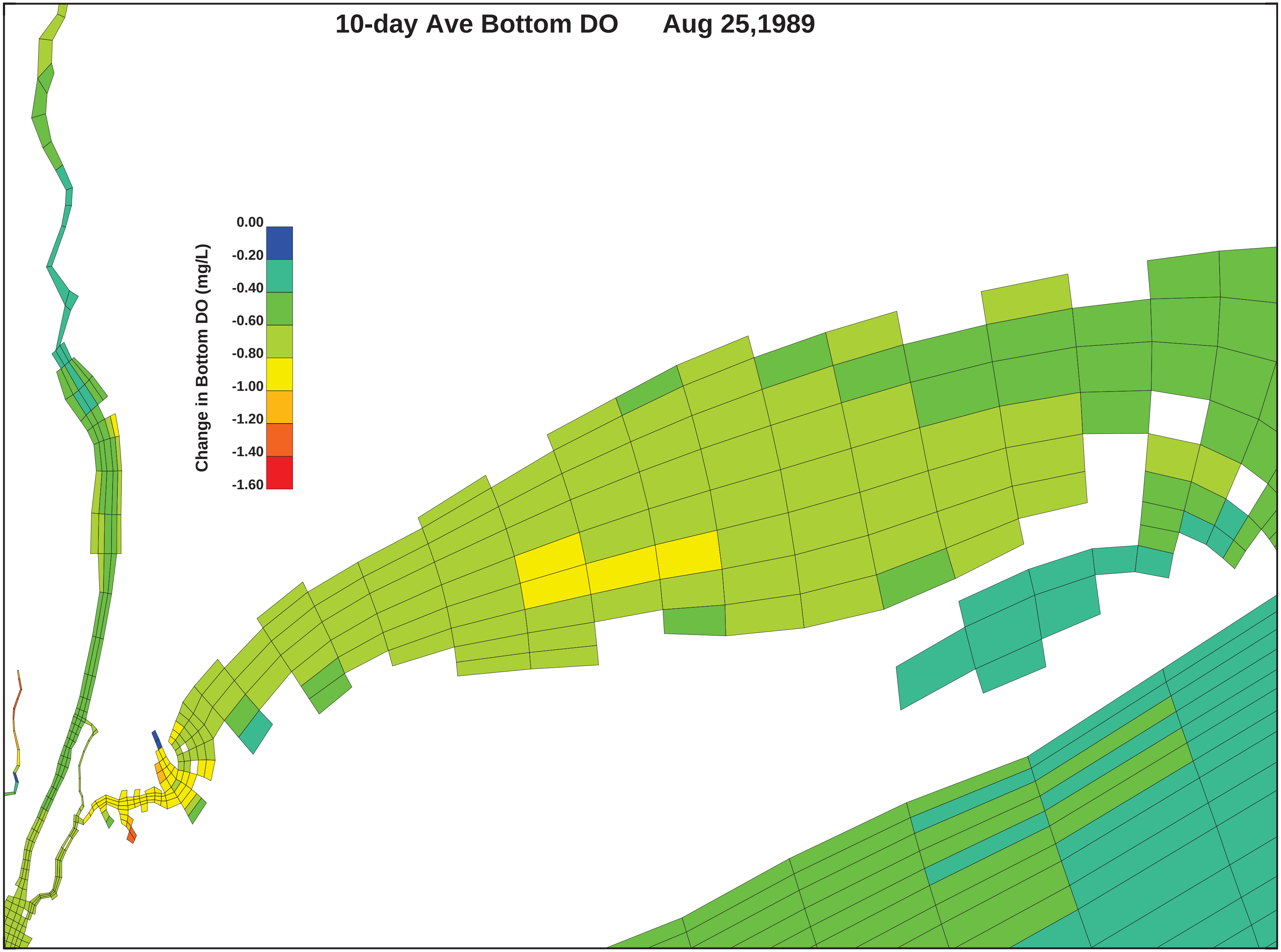
10-day Min Bottom DO July 25, 1989
Change in Bottom DO (mg/L)
0.00
-0.20
-0.40
-0.60
-0.80
-1.00
-1.20
-1.40
-1.60


10-day Ave Bottom DO Aug 5, 1989
Change in Bottom DO (mg/L)
0.00
-0.20
-0.40
-0.60
-0.80
-1.00
-1.20
-1.40
-1.60

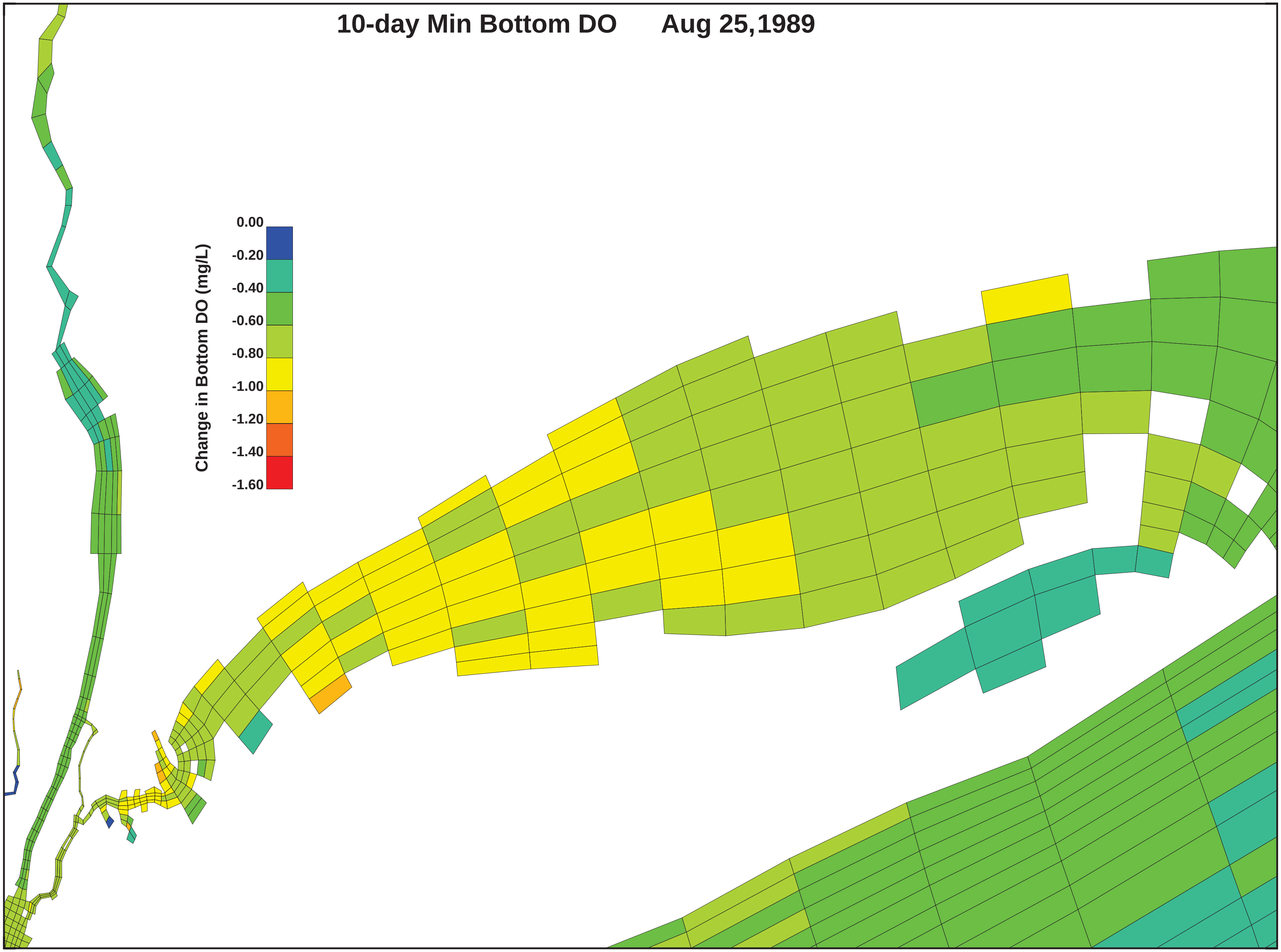

10-day Min Bottom DO Aug 5, 1989
Change in Bottom DO (mg/L)
0.00
-0.20
-0.40
-0.60
-0.80
-1.00
-1.20
-1.40
-1.60


10-day Ave Bottom DO Aug 15,1989
Change in Bottom DO (mg/L)
0.00
-0.20
-0.40
-0.60
-0.80
-1.00
-1.20
-1.40
-1.60

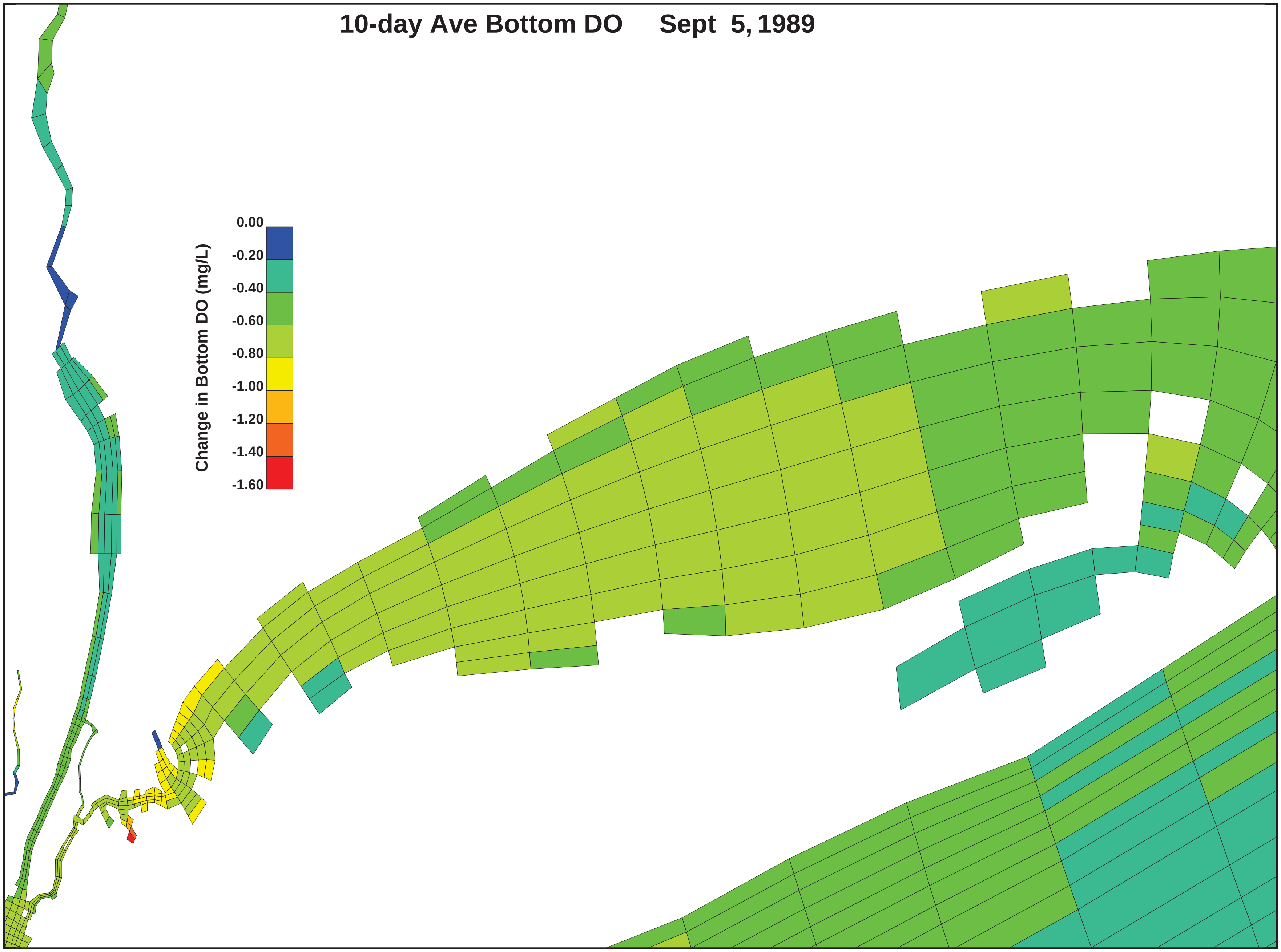

10-day Min Bottom DO Aug 15,1989
Change in Bottom DO (mg/L)
0.00
-0.20
-0.40
-0.60
-0.80
-1.00
-1.20
-1.40
-1.60

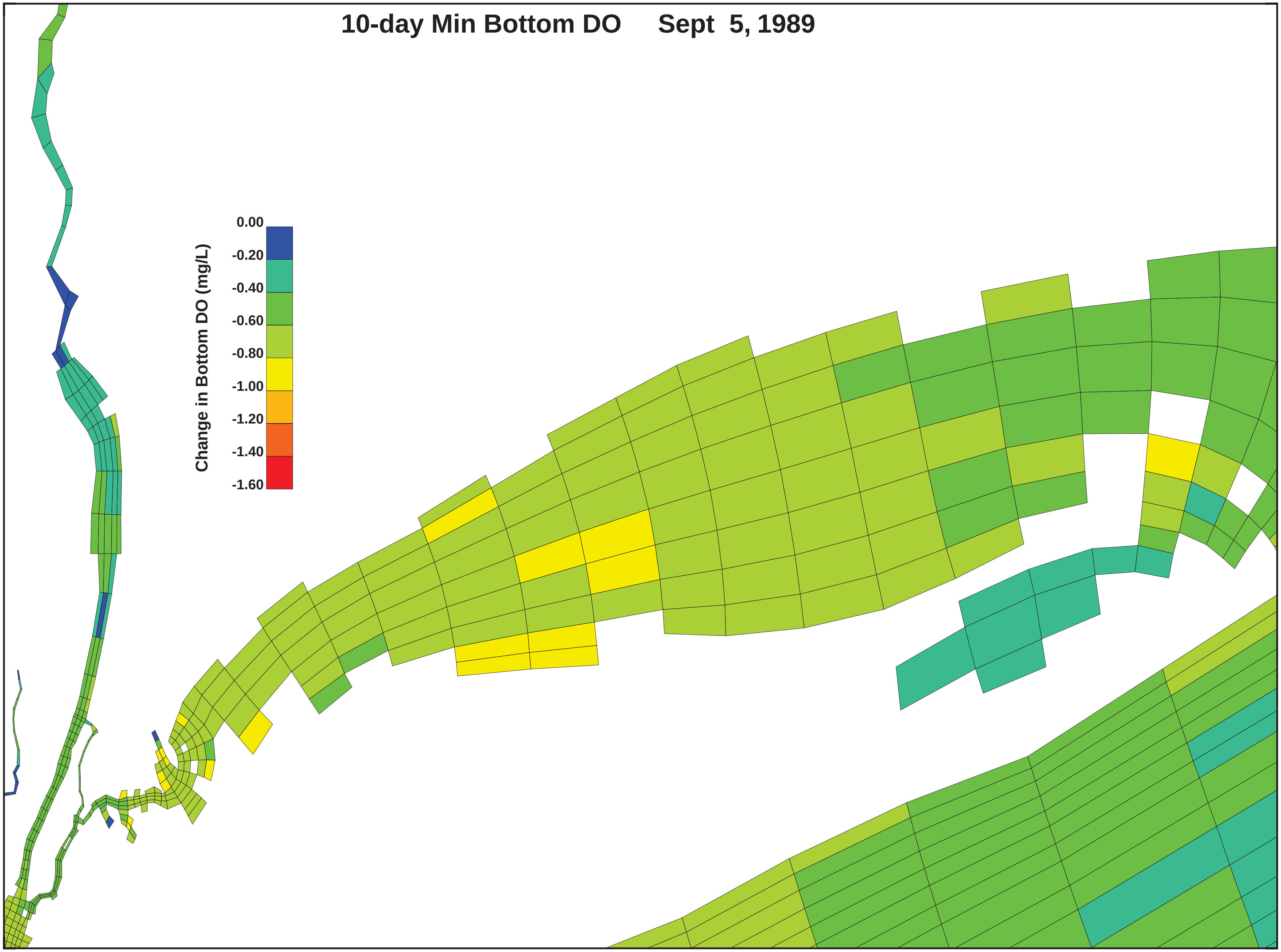


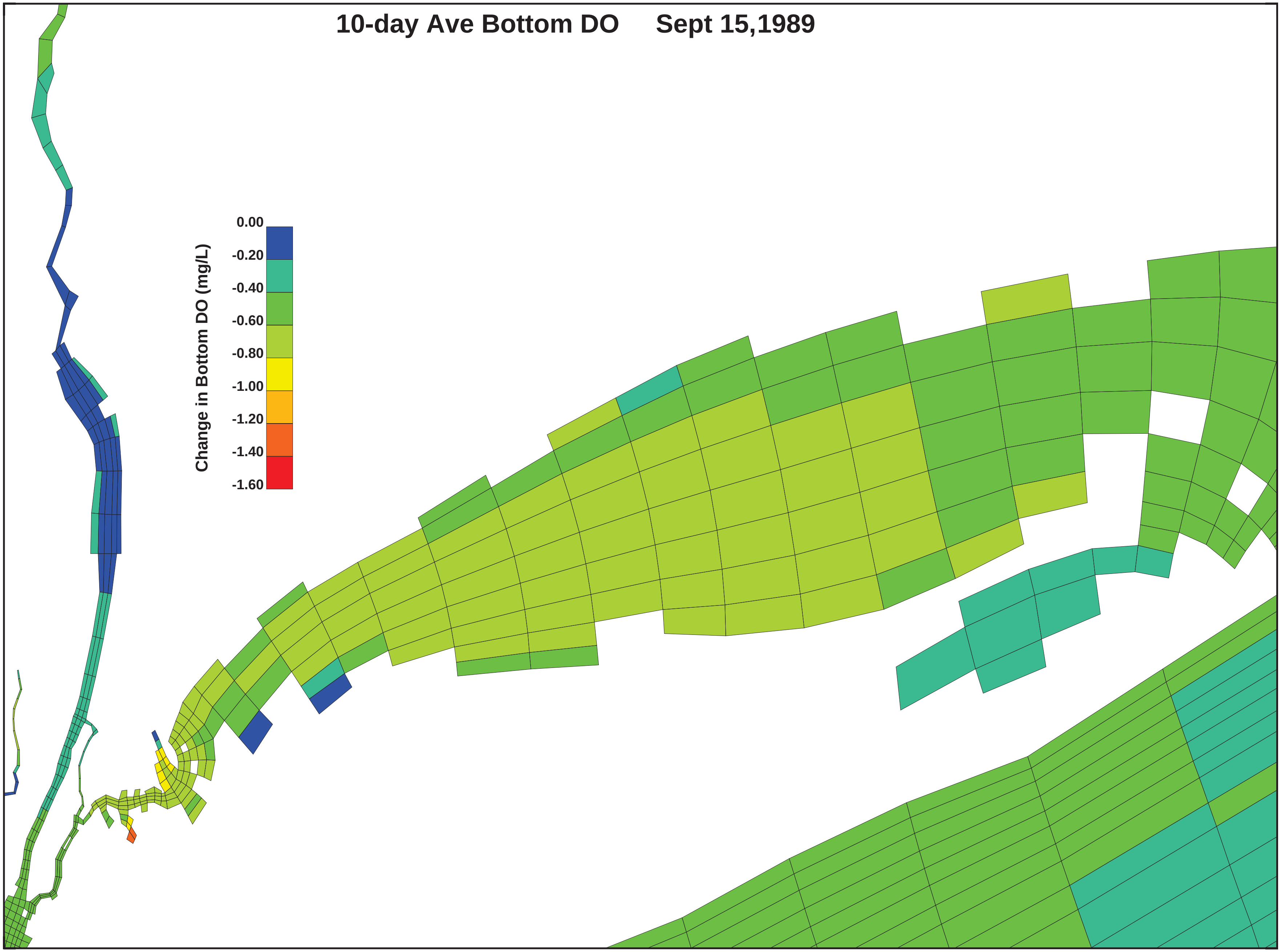
10-day Ave Bottom DO Aug 25,1989
Change in Bottom DO (mg/L)
0.00
-0.20
-0.40
-0.60
-0.80
-1.00
-1.20
-1.40
-1.60

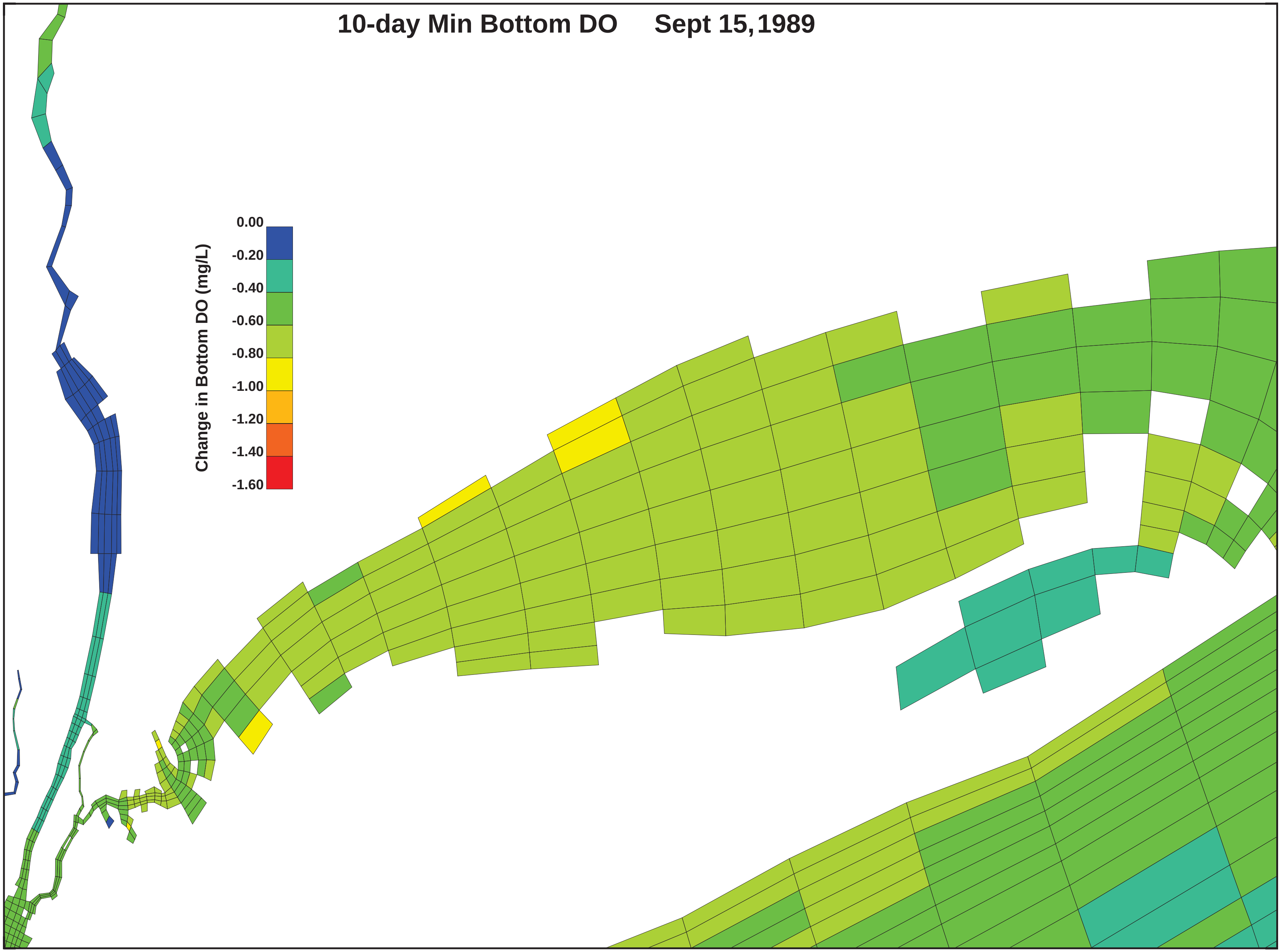


10-day Min Bottom DO Aug 25,1989
Change in Bottom DO (mg/L)
0.00
-0.20
-0.40
-0.60
-0.80
-1.00
-1.20
-1.40
-1.60


10-day Ave Bottom DO Sept 5, 1989
Change in Bottom DO (mg/L)
0.00
-0.20
-0.40
-0.60
-0.80
-1.00
-1.20
-1.40
-1.60


10-day Min Bottom DO Sept 5, 1989
Change in Bottom DO (mg/L)
0.00
-0.20
-0.40
-0.60
-0.80
-1.00
-1.20
-1.40
-1.60


10-day Ave Bottom DO Sept 15,1989
Change in Bottom DO (mg/L)
0.00
-0.20
-0.40
-0.60
-0.80
-1.00
-1.20
-1.40
-1.60


10-day Min Bottom DO Sept 15,1989
Change in Bottom DO (mg/L)
0.00
-0.20
-0.40
-0.60
-0.80
-1.00
-1.20
-1.40
-1.60