Bays Eutrophication Model (BEM): Modeling analysis for the period 1992-1994

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ABSTRACT

This report documents the application of the Bays Eutrophication Model (BEM), a water quality model of the eutrophication processes of Massachusetts and Cape Cod Bays, to the Harbor and Outfall Monitoring (HOM) data set for the period 1992-1994. The report summarizes the evaluation of existing data; describes the application and calibration of BEM to existing water quality data; performs a system-wide nitrogen balance for the Massachusetts Bays system; and describes the results of some diagnostic simulations designed to evaluate the relative importance of nutrient inputs from MWRA versus those entering the system from the open boundary.

The water quality model uses the results of a three-dimensional time-variable hydrodynamic model of the Bays system, developed for MWRA by the United States Geological Survey. The hydrodynamic model was used to develop realistic circulation patterns for the water quality model calibration period of January 1992 through December 1994. As judged by model versus observed data comparisons, the water quality model provides a realistic representation of the physical, chemical, and biological processes that determine eutrophication within the study area. The water quality model has been used to investigate and define the relationships between bay circulation, nutrient loadings, primary productivity, and dissolved oxygen in Massachusetts and Cape Cod Bays. However, some limitations of the model have been identified and are discussed in the report.

The calibrated water quality model has also been used to develop a mass balance for total nitrogen for the Massachusetts Bays system. Model computations indicate that the largest input of nitrogen to the bays is via import from the Gulf of Maine through the open water boundary of the system. Additional analysis indicates that the open water boundary has a large spatial and temporal effect on water quality at the near field stations, under current or future outfall conditions.

ACKNOWLEDGMENTS

The work detailed in this report was conducted for the Massachusetts Water Resources Authority (MWRA) Harbor Studies Program, as part of the Harbor and Outfall Monitoring III Task Order Contract, while under sub-contract to Battelle Ocean Sciences. Many people contributed to the material in this report and the research effort which it describes. In particular, the authors wish to acknowledge Dr. Michael Mickelson (MWRA), project manager for Harbor and Outfall Monitoring Contract and project manager for this study and Mr. Ken Keay (MWRA). We wish to thank them for their program assistance and careful review of the Draft Report. Their suggestions have resulted in a much improved final report. The authors are also grateful to Dr. Carlton Hunt (Battelle) for providing liaison with MWRA during the course of this study. The authors also acknowledge the significant contribution of Dr. Scott Libby (Battelle), who assisted in the evaluation of model performance relative to the 1993 fall algal bloom and 1994 minimum dissolved oxygen event. In particular, Scott contributed significantly to the development and writing of Section 5 of this report.

The authors also acknowledge the significant contribution of Dr. Richard Signell (USGS), who was responsible for the development and calibration of the hydrodynamic model, which is a key underpinning to the water quality model. His continued enthusiasm for this project and his willingness to perform "yet another model run" to assist the water quality modeling effort was much appreciated.

Finally, recognition and appreciation is extended to Ms. Linda Jensen (HydroQual), who typed this report.

Mr. James Fitzpatrick served as Principal Engineer for this study and was responsible for the overall technical direction of the project. Mr. Richard Isleib served as project Manager on this study and was responsible for the development of model inputs, model calibration, and the development of the major portions of this report.

CONCLUSIONS AND RECOMMENDATIONS

The Bays Eutrophication Model (BEM) developed for the Massachusetts Water Resources Authority (MWRA) and previously calibrated for the periods October 1989 through April 1991 and January through December 1992, was applied to additional water quality data from the ongoing Harbor and Outfall Monitoring (HOM) program. The model simulation period was extended from January 1993 through December 1994. However, due to changes required to calibrate the Massachusetts Bays hydrodynamic model for 1993 and 1994, and due to additional insights concerning the behavior of water quality at the Massachusetts Bay/Gulf of Maine boundary gained from the relocation of a HOM monitoring station closer to Cape Ann, the original 1992 calibration period was rerun as part of this analysis.

The application of the BEM to the Massachusetts Bays system provided an opportunity to evaluate model performance against a unique set of water quality conditions observed during the 1992-1994 time period. In particular, the ability of the model to reproduce the occurrence of a large phytoplankton bloom in the fall of 1993 and the occurrence of low concentrations of dissolved oxygen in the fall of 1994 were considered to be a key test of the model.

CONCLUSIONS

Based on data analysis and model versus data comparisons, the following conclusions and inferences may be drawn.

Hydrodynamic

1) Based on model versus data comparisons, the hydrodynamic model appears to capture the major temporal, horizontal, and vertical gradients of salinity and temperature within the Bays system. These comparisons suggest that the model is properly reproducing the time-dependent, three-dimensional circulation and stratification in Massachusetts and Cape Cod Bays. The major model versus data discrepancies and uncertainties probably relate to the inability to exactly specify the forcing and boundary conditions for the model simulations.

Water Quality

1) The water quality model appears to reproduce or partially reproduce the principal temporal and vertical features of the water quality within the Bays during the 1992-1994 time period. These features include:
   - the onset of the winter/spring phytoplankton bloom earlier in Cape Cod Bay than the rest of the Massachusetts Bay system,
   - the decrease in phytoplankton biomass, as indicated by chlorophyll-a and fluorescence, in Massachusetts Bay as one moves further away from Boston Harbor,
   - the limitation of the winter/spring phytoplankton bloom in Cape Cod Bay by silica,
   - the non-limiting concentrations of inorganic phosphorus, which suggest that the bays are not phosphorus limited,
   - the limitation of summer primary productivity by inorganic nitrogen,
   - the observed vertical gradients in the inorganic nutrients with low levels in the surface (due to algal uptake) and higher levels in the bottom waters of the Bays (due to remineralization of detrital algal biomass and reduced algal uptake),
   - the annual cycle of dissolved oxygen, with super-saturated conditions occurring between late winter/early spring and late summer and with minimum levels occurring in late September and October, and
   - the annual cycle of primary productivity in northwest Massachusetts Bay.

2) The water quality model, however, was not able to fully reproduce two of the key features of the 1992-1994 data set. In particular, the model was not able to:
   - reproduce the large *Asterionellopsis glacialis* bloom observed in the late summer and fall of 1993, nor
   - reproduce the minimum levels of dissolved oxygen observed at some near field stations in September/October of 1994.

3) The model also failed to reproduce the observed increase in bottom water dissolved oxygen that occurred between May and June in 1992 and 1993. However, this increase in bottom water dissolved oxygen was not as evident in the summer 1994 data.

4) While the inability of the model to reproduce the fall *A. glacialis* bloom is disappointing, it is important to note that the magnitude of the bloom may be overstated due to the unique carbon to chlorophyll-a ratios specific to *A. glacialis*. When compared to measurements of particulate organic carbon, the

model does not appear to under estimate phytoplankton biomass in the fall of 1993 as severely as it does when compared to measurements of chlorophyll-a.

5) The inability of the model to reproduce the fall *A. glacialis* bloom does not appear to have an adverse impact on the ability of the model to compute bottom water dissolved oxygen during the fall of 1993.

6) While the model did not fully reproduce the minimum values of bottom water dissolved oxygen in the fall of 1994, the model was able to compute bottom water concentrations of dissolved oxygen in 1994 that were approximately 1 mg O_2/L lower than computed for either 1992 or 1993.

RECOMMENDATIONS

1) The water quality model should continue to be applied and refined using water quality data provided by the ongoing HOM.

2) While the current HOM program has provided some additional information concerning the water quality conditions at the open water boundary, should funding be available and opportunities arise, additional sampling of the boundary stations should be considered.

3) Consideration should be given to collecting additional field data which might provide insights as to the cause of the increase in bottom water dissolved oxygen that occurs in some years; in particular, 1992 and 1993 of this study. First priority should be given to the measurement of benthic diatoms, which may be generating oxygen through primary production in the shallower waters (5-18 meters) of the near field.

4) Should data collected as part of Recommendation 3 indicate high benthic diatom populations, consideration should be given to expanding the BEM to include a benthic diatom state-variable.

5) If upon further review it is believed that it is important for the BEM to better predict fall diatom blooms, consideration should be given to expanding the BEM framework to include an additional functional phytoplankton group, representing fall-blooming diatoms.

SECTION 1

INTRODUCTION

The Massachusetts Water Resources Authority (MWRA) is constructing an ocean outfall, which will divert treated effluent from the Deer Island Wastewater Treatment Plant. The present (1998) Deer Island outfall is located at the entrance to Boston Harbor and will be relocated into Massachusetts Bay at a distance approximately 15 km east of Deer Island and at a water depth of 32 meters (Figure 1-1). The relocation of the wastewater discharge from within the confines of Boston Harbor, together with improved sewage treatment at the Deer Island facility, is expected to result in a significant improvement in water and sediment quality within the Boston Harbor area.

In order to develop a more rigorous and detailed understanding of the potential impact of the outfall relocation on the water quality of the Massachusetts Bays system (Boston Harbor, Massachusetts Bay, and Cape Cod Bay), the MWRA has funded or partially funded a number of studies within the Massachusetts Bays system. Included among these studies are a number of hydrographic and water quality surveys of the Bays. These surveys have collected data on salinity, temperature, phytoplankton, primary productivity, nutrients and dissolved oxygen throughout the Massachusetts Bays system. The MWRA has also funded a number of benthic nutrient flux and sediment oxygen demand surveys for stations within Boston Harbor and Massachusetts Bay. Both of these programs have been expanded and incorporated into the Harbor and Outfall Monitoring program (HOM) and will provide information concerning baseline water quality and sediment conditions, as well as annual and inter-annual variability, prior to the new outfall going online. In addition, the HOM program will be used to detect possible changes from baseline water quality and sediment conditions that may result from the new outfall. The HOM program began in February, 1992 and is still ongoing.

The MWRA also entered into a cooperative agreement with the U.S. Geological Survey (USGS) to develop a time-variable three-dimensional hydrodynamic model of the bays (Signell et. al., 1993, Signell et. al., 1996), and funded the development and calibration of a time-variable water quality model (HydroQual and Normandeau, 1995). These models were calibrated for the

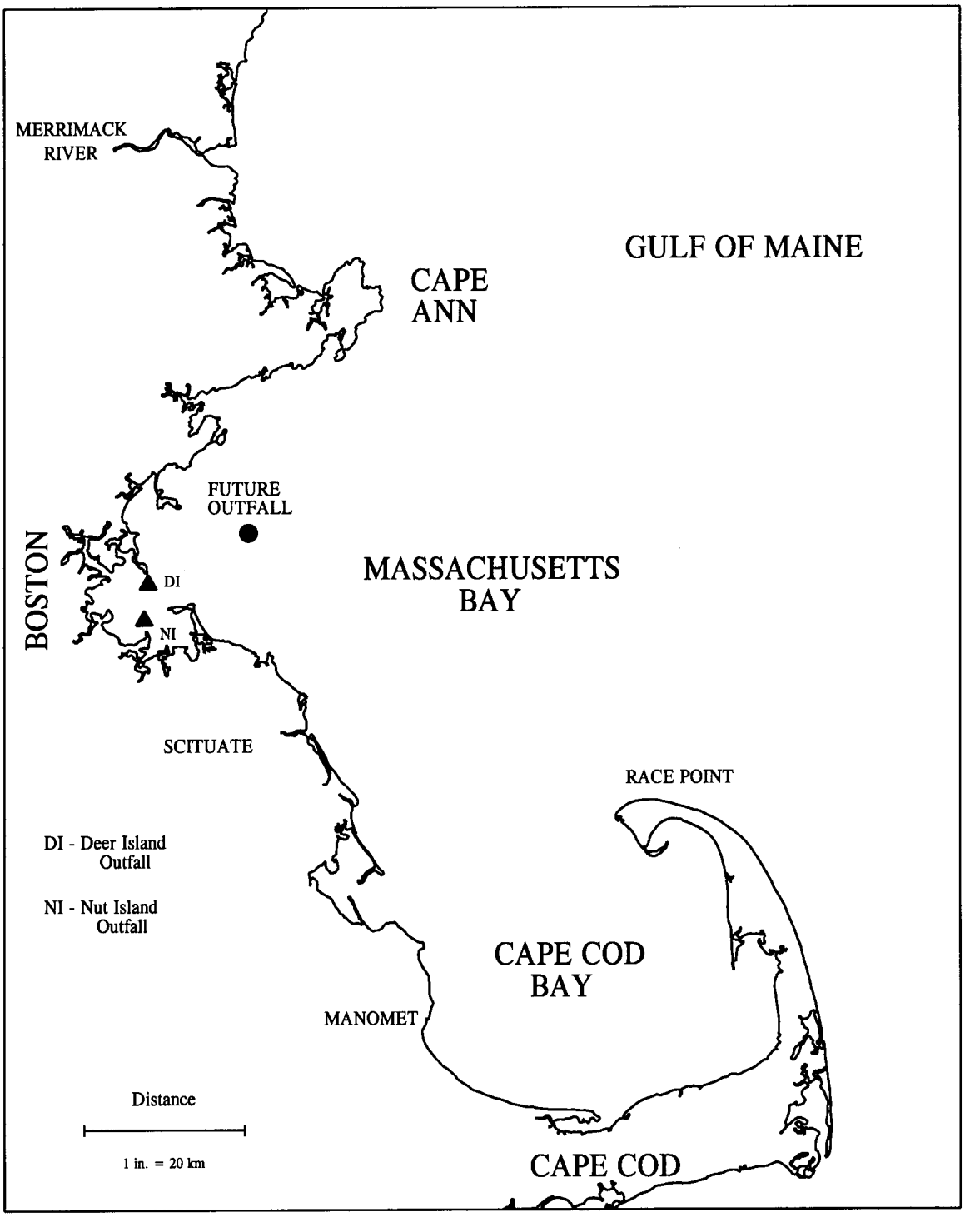


Figure 1-1. Massachusetts Bay Study Area

periods of October 1989 through April 1991 and January 1992 through December 1992 (Signell, 1993; HydroQual and Normandeau, 1995). As part of MWRA's permit for the new outfall, these models are to be run as HOM data become available. This report details the results obtained from applying the Bays Eutrophication Model (BEM) for the years 1993 and 1994. As will be discussed in Section 3, unique water quality conditions were observed in both 1993 and 1994, relative to water quality observed during the years against which the BEM was originally calibrated. Ideally, the BEM should be able to reproduce observed 1993 and 1994 water quality without adjusting any of the model constants or parameters; the only changes to the model should be to exogenous inputs such as pollutant loads, meteorological conditions (e.g., wind, solar radiation, rainfall, etc) and/or tidal circulation and density stratification. However, as is often the case, as subsequent data are collected, new insights into the workings of the system being modeled are gained. In particular, the addition of two new monitoring stations to the HOM program in 1994 provided additional information concerning the behavior of water quality at the northeastern boundary of the BEM model, which we felt were appropriate to incorporate in the 1992 simulation. In addition, in the hydrodynamic model modifications were made to the sea surface cooling in December and January to more properly account for the observed rate at which heat was extracted from the system (Signell, pers. comm.). Also between the time of the original calibration effort and the present study the numerical scheme used in the hydrodynamic model for scalar advection was replaced with a more accurate scheme (Signell, pers. comm.). As a result, minor modifications were made to the original hydrodynamic and water quality model calibrations for 1992. Therefore, the model results for 1992 are presented in this report. This report describes the application of the BEM to hydrographic, biological and chemical data collected during the period 1992-1994.

SECTION 2

POLLUTANT LOADINGS

2.1 INTRODUCTION

The pollutant loadings used for this modeling analysis were developed in the same manner as for the 1990 and 1992 calibration periods with some minor exceptions which will be discussed below. The reader is referred to HydroQual and Normandeau (1995) for details on the procedures and assumptions used to develop pollutant loadings. This section will present information concerning year to year variability in loadings of carbon (C), nitrogen (N), and phosphorus (P) for the period 1992 through 1994.

2.2 LOADING COMPARISON

Figure 2-1 presents a comparison between the total carbon, total nitrogen and total phosphorus loadings from various sources for the years 1992, 1993, and 1994. The loading sources include MWRA wastewater treatment plants (WWTPs), non-MWRA WWTPs, non-point sources (which include combined sewer overflows (CSOs), storm sewers and groundwater), riverine sources and atmospheric sources. In general, the carbon, nitrogen, and phosphorus loadings to the Massachusetts Bays system were quite similar for all three years considered in this analysis. Perhaps the greatest differences were in the loading estimates for the MWRA WWTPs, although even these varied only by 5 to 10 percent for carbon and nitrogen and about 15 percent for phosphorus. An analysis of the average daily flows from the Nut Island and Deer Island facilities from 1992 through 1994 indicated that the minimum annual average daily combined effluent flow of 371.4 MGD occurred in 1992, while the maximum annual average daily combined effluent flow of 382.5 MGD occurred in 1994. Since this range in effluent flow is small, it would suggest that differences in the carbon, nitrogen, and phosphorus loadings may have been due to varying levels of treatment efficiencies over time as well as possibly due to natural sampling variability in the effluent wastewaters.

Lacking additional information, the non-MWRA treatment plant loads were held the same for all three years. The annually-averaged non-point sources developed for the model did not vary significantly over the three year period (1992-1994). Again lacking additional information, the

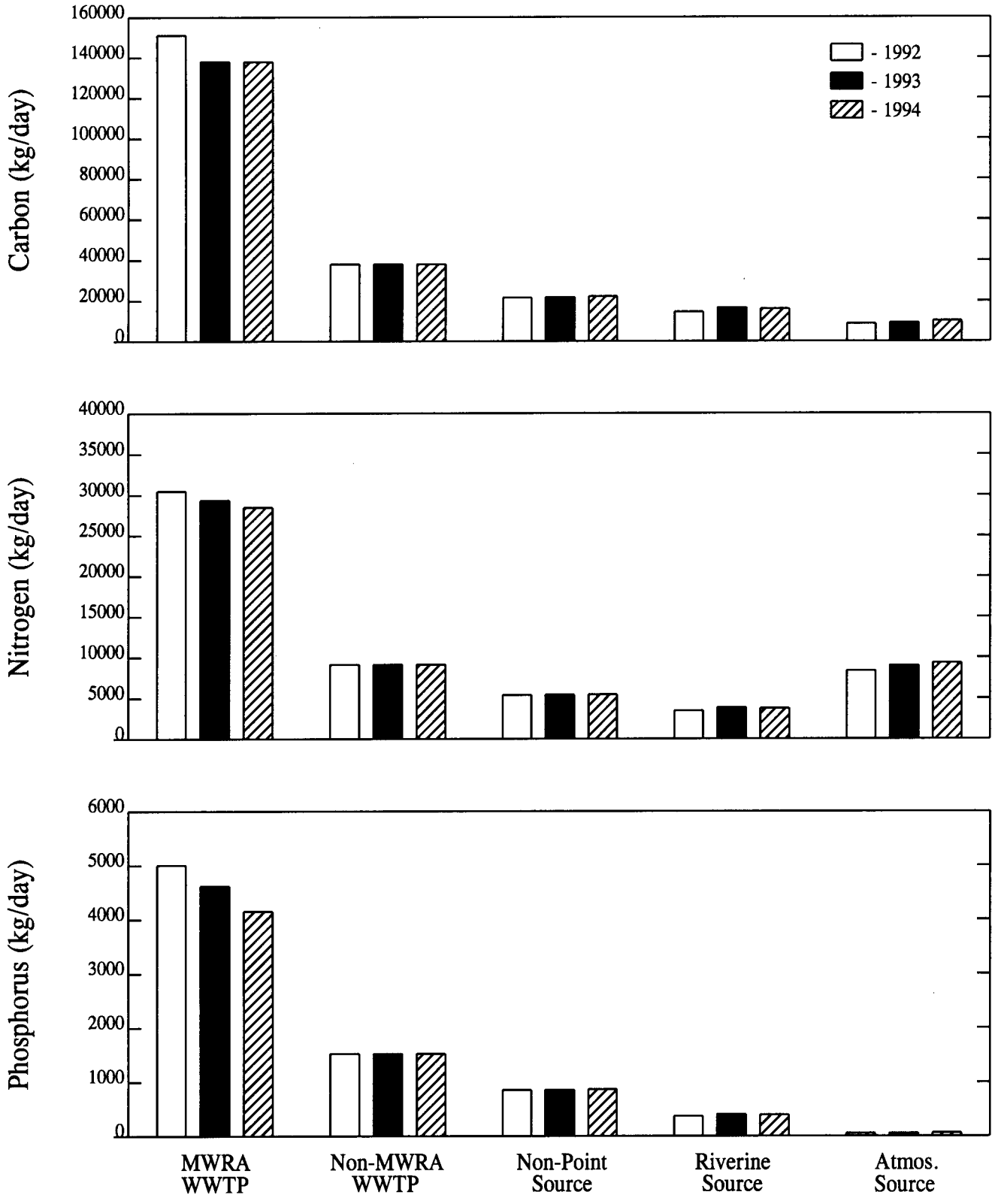


Figure 2-1. Loading Comparison of Sources for 1992 through 1994

groundwater and stormwater loading were held the same for all three years. For this analysis the CSO loadings were calculated in a slightly different manner than the original 1992 calibration. Volumes of CSO captured and treated by MWRA as well as the ratio of the total estimated CSO volume versus captured volume were obtained from MWRA. From this information, the total flow was estimated. The total flow less the flow from Cottage Farm, which discharges above the dam on the Charles River, was multiplied by the estimated CSO concentrations developed by Alber and Chan (1994) to develop the CSO loads. The estimated volume ratios were 1.96, 1.91, and 1.86 for 1992, 1993, and 1994, respectively.

Pollutant loads for the Charles, Mystic, and Chelsea Rivers varied year to year in response to freshwater flow as reported by the USGS. The riverine loads outside of Boston Harbor (but within the Massachusetts Bays system) were the same for all three years. The dry fall loads for the atmospheric loads remained the same for all three years while the wet fall loads were modified according to annual rainfall. The total annual rainfall measured at Logan Airport was 43.7, 43.3, and 47.6 inches for 1992, 1993, and 1994 respectively.

Two observations can be made after viewing Figure 2-1: (1) the MWRA effluent is the major internal source of nitrogen, phosphorus and anthropogenic carbon to Massachusetts and Cape Cod Bays, and (2) the year to year variation in the pollutant loads are small, at least for the period 1992-1994. However, this ignores the importance of the Gulf of Maine as a potential source of nutrients to the Massachusetts Bays system. Since there is no way to determine the magnitude of this nutrient source by direct observation, the BEM was utilized to evaluate the magnitude of the boundary input. Results of this analysis will be presented in Section 5.

SECTION 3

WATER QUALITY DATA

3.1 INTRODUCTION

The credibility of model calculations is judged, to a large degree, on the basis of the agreement of the model with observed data. Beyond the constraint that the model must behave reasonably well and conform with general scientific principles, observed data offer perhaps the only external criteria available to assess the validity and, hence the utility of a complex hydrodynamic/water quality model. A properly validated deterministic model is a powerful tool because it represents, as closely as possible, the mechanisms which affect the system.

The data used herein come from the MWRA funded Harbor and Outfall Monitoring program (HOM). The monitoring program was not designed specifically for use in a water quality model, but rather was developed to detect possible changes in water quality conditions that might result in Massachusetts Bay once the new WWTP effluent outfall goes on line. However, the data collected do provide sufficient information for calibration purposes.

To date, there have been several reports sponsored by MWRA that provide extensive analyses of the HOM water quality data (Kelly, 1993, Gilbin et al., 1994, Kelly and Turner, 1995, Gilbin et al., 1995, Kelly and Doering, 1995). This section will not repeat those efforts. Rather, this section will summarize the observed differences between the water quality conditions occurring in 1992, 1993 and 1994.

3.2 HOM WATER QUALITY DATA

The HOM water quality monitoring program consists of 46 stations. These stations consist of 21 near field stations which are in the vicinity of the future outfall location, and 25 far field stations which cover the majority of Massachusetts and Cape Cod Bays. Figure 3-1 presents the locations of the stations during 1992 and 1993. Some modifications to the locations of sampling stations were made in 1994, in particular, the addition of two stations located closer to the

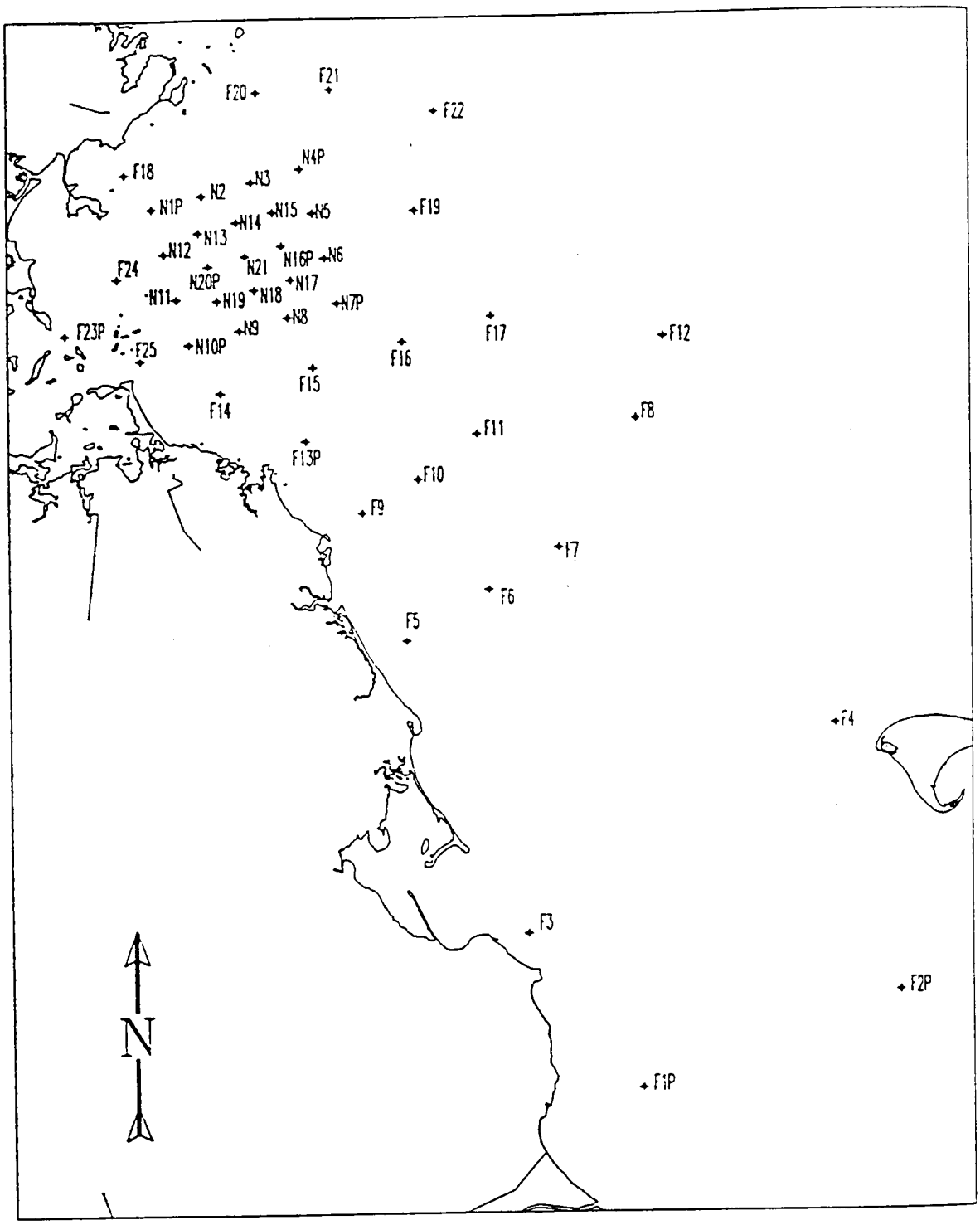


Figure 3-1. Water Quality Sampling Stations in Massachusetts and Cape Cod Bays 1992 and 1993

Massachusetts Bays/Gulf of Maine boundary, near Cape Ann. These stations are shown in Figure 3-2.

3.2.1 Physical Water Quality Parameters

3.2.1.1 Temperature

Water column temperature is recognized as an important parameter affecting both hydrodynamic and biological processes that occur in all water bodies. The distribution of water column temperature in Massachusetts Bay is a function of both surface heat flux and the transport of water into and out of the system from the Gulf of Maine. Figure 3-3 displays temporal distributions of water column temperature for the 1992-94 period at four locations in the Massachusetts Bay system. The first location is HOM station F23 in Boston Harbor; the second location is station N10 just outside Boston Harbor; the third location is station F01 in southwest Cape Cod Bay; while the fourth location presents the averaged data observed at stations N06 and N07 which are near the future outfall location.

While the annual cycles of temperature warming and cooling and vertical temperature stratification were similar for the three years studied in this analysis there were also some notable differences. For example, during the winter, the 1992 water column temperatures were 1-2 degrees Celsius higher than the other years. Of the three years, 1993 had the highest temperatures during the late spring, summer and early fall. In 1994, the surface temperatures remained 2-3 degrees Celsius warmer in the late fall. In 1992, bottom waters tended to be warmer by one degree Celsius during the spring. During the summer and fall the 1992 and 1993 bottom temperatures were very similar. However, in 1994 the bottom waters warmed very rapidly towards the end of summer and by the fall the temperatures were 3-5 degrees Celsius warmer than 1992 and 1993. Temperature stratification between the surface and bottom appeared to occur at approximately the same time from year to year. Initial stratification occurred in the April/May time period and generally, the fall overturn occurred between late October and late November. The fall of 1992 appeared to stay stratified slightly longer than was observed in the other years.

Figures 3-4 and 3-5 present probability distribution comparisons of surface and bottom temperatures, respectively, for 1992-1994 at the near field and far field stations. These figures present another way to observe the year to year and season to season variability of the

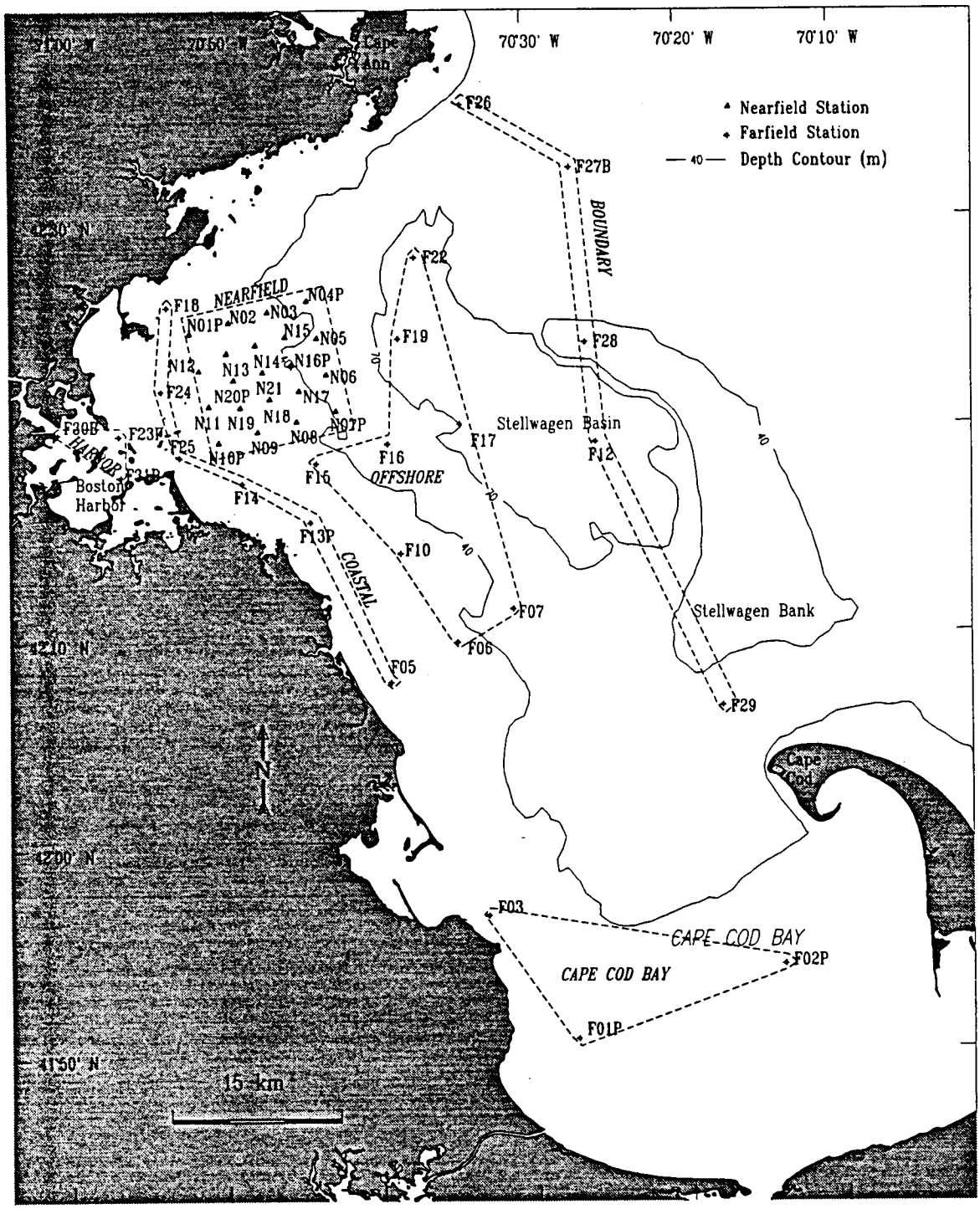


Figure 3-2. Water Quality Sampling Stations in Massachusetts and Cape Cod Bays 1994

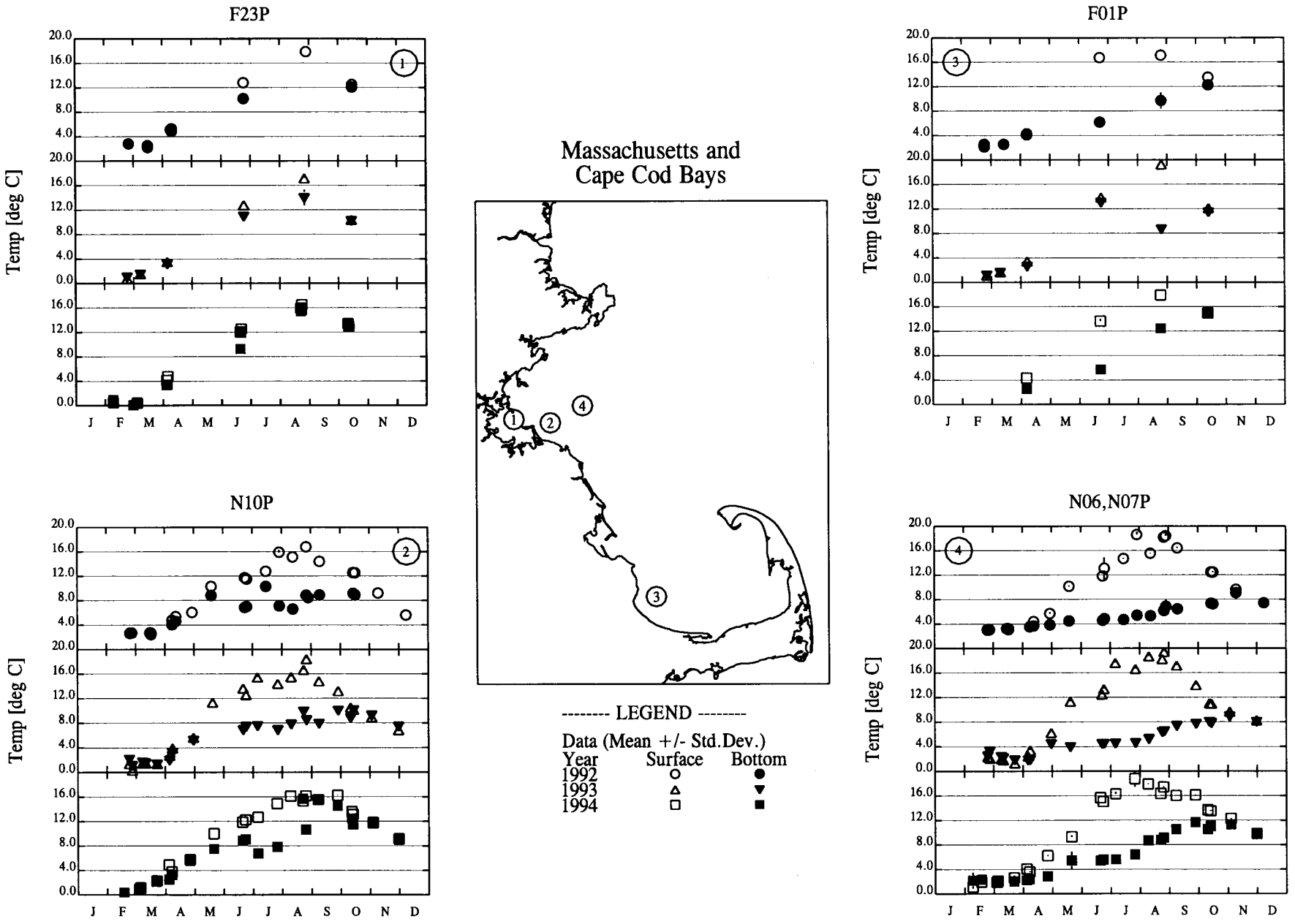


Figure 3-3. Temporal Comparison of Temperature Data

temperature. For this analysis winter is defined December and February (no data were collected during January), spring is March-May, summer is June-August, and fall is September-November. While the selection of these "seasonal" averaging periods may appear arbitrary, it attempted to recognize that the spring diatom bloom usually occurred in the March-May time period and the lowest dissolved oxygen concentrations were observed in October and November just before the complete overturn of the water column in December. The annual distributions should be viewed with some caution as no attempt was made to evenly weight the data by season. Therefore the summer data contribute more to the distribution than do the winter data.

As seen in Figure 3-4, there is a 5-7°C gap in temperature during the winter months for 1993 and 1994. This is in part an artifact resulting from the absence of data sampling in January 1993 and 1994. This gap is not observed in 1992 because the data in February, 1992 were slightly warmer than February 1993/1994, while temperature data in December 1992 were slightly colder than December 1993/1994. The winter surface temperature distributions show a higher median temperature in 1992. In general, spring surface temperatures were similar for all three years, although 1992 had higher minimum temperatures and 1993 had slightly higher maximum temperatures. Summer temperatures were also similar during the three years. During 1994 there were higher surface temperatures observed for part of the fall. On an annual basis, the three years appear similar.

The bottom water temperature data (Figure 3-5) show a greater variation from year to year than did the surface data. Limited data make the winter comparisons difficult, but the maximum winter temperatures in 1994 are approximately 2 degrees Celsius higher than observed in 1993 and approximately 6 degrees Celsius higher than observed in 1992. During the spring, the 1992 bottom water temperatures were a degree Celsius or so higher than observed in 1993 or 1994. Summer bottom temperatures were similar for all three years except that maximum summer temperatures were 1-2 degrees Celsius higher in 1994. The fall data for 1992 and 1993 are very similar. However, in 1994 bottom water temperatures were on average 3-4 degrees Celsius warmer than observed in 1992 and 1993.

3.2.1.2 Salinity

Massachusetts Bay has a large open boundary with the Gulf of Maine. Besides freshwater inputs from WWTPs, Massachusetts and Cape Cod Bays have only a few small tributaries which

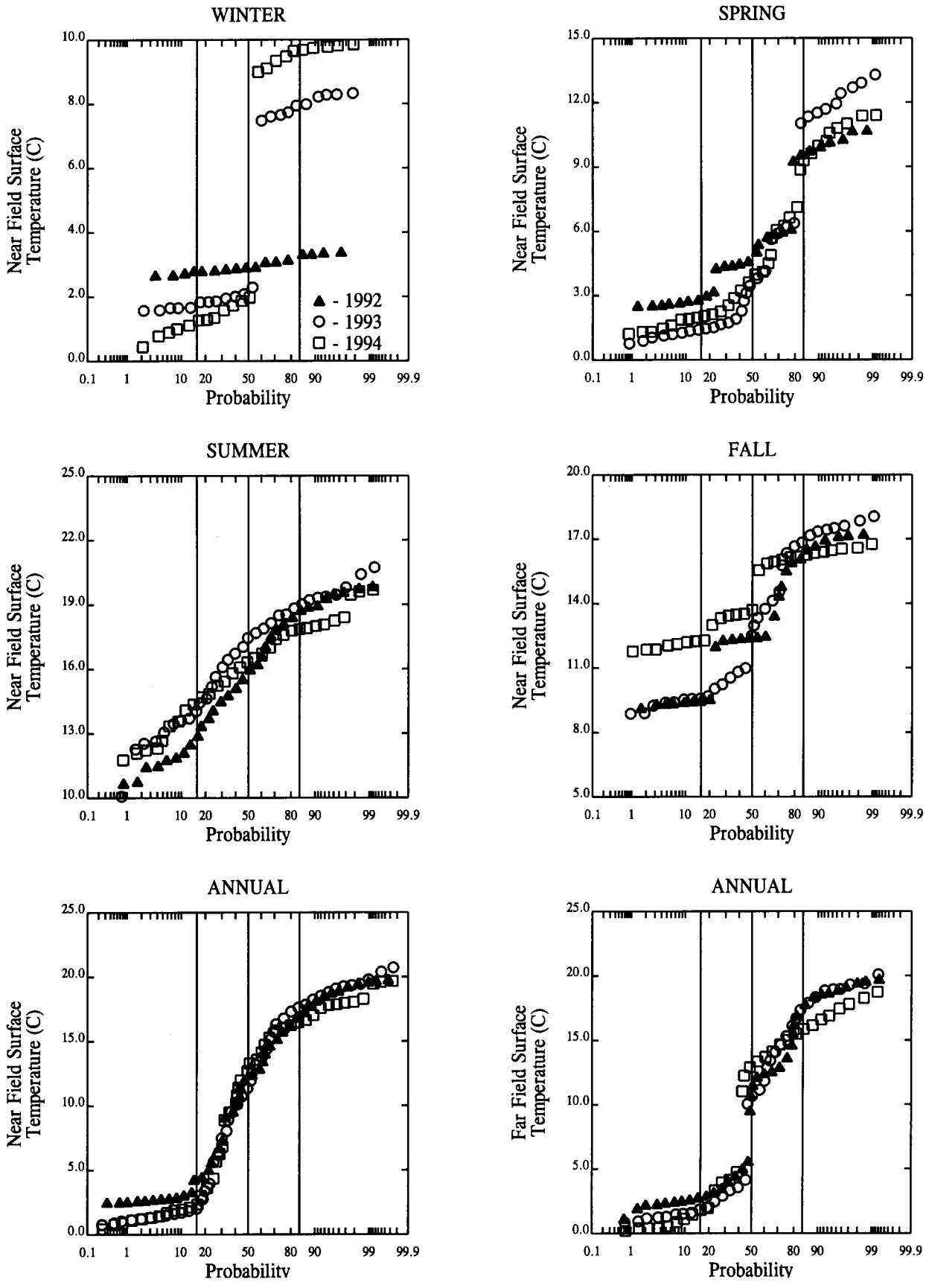


Figure 3-4. Seasonal and Annual Probability Distributions of Surface Temperature

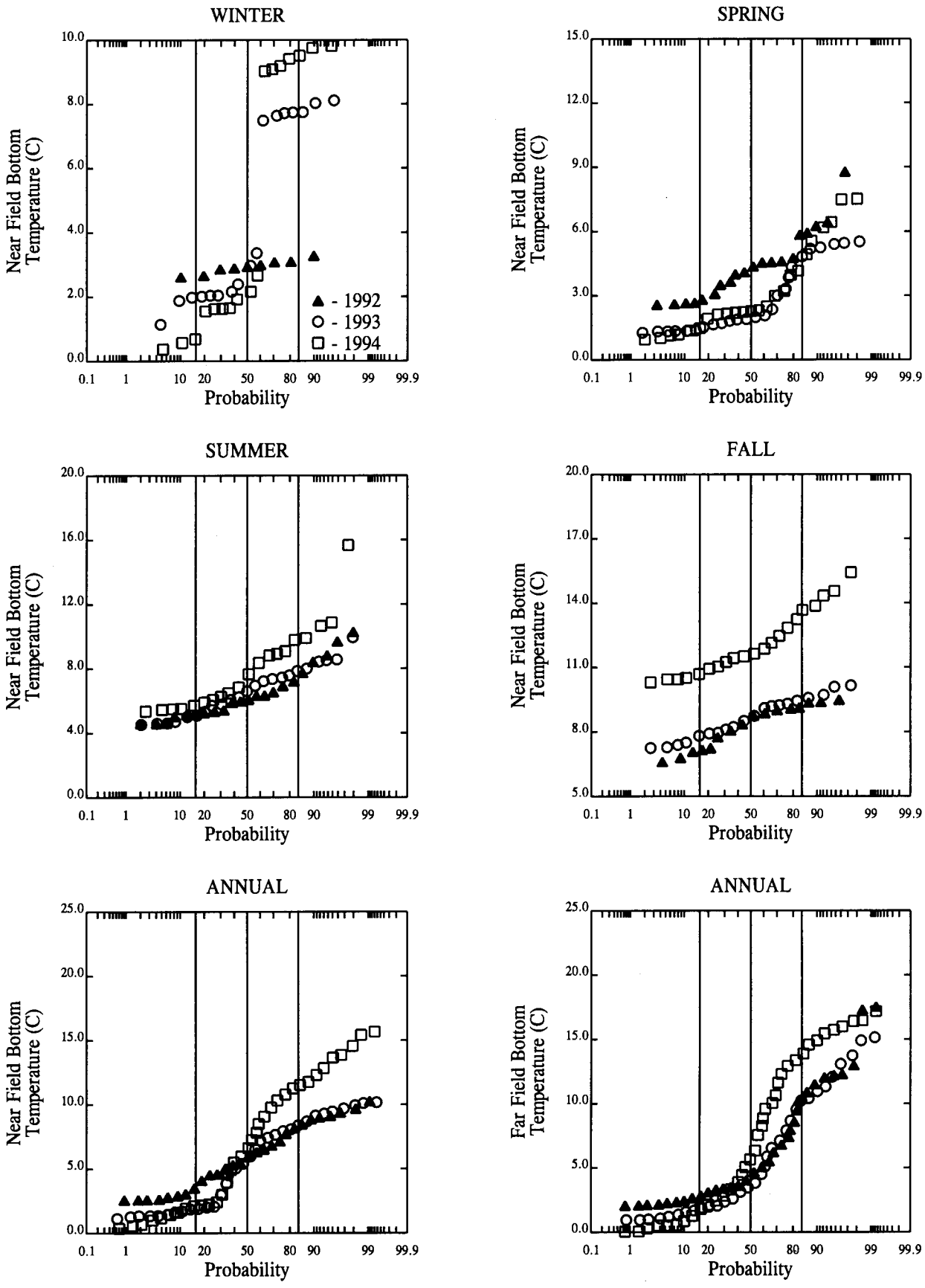


Figure 3-5. Seasonal and Annual Probability Distributions of Bottom Temperature

contribute fresh water to the system. Therefore, most of the water within the Bays is coastal ocean water with a relatively constant salinity. During the spring, however, as a consequence of snow melt and spring runoff, pulses of fresh water, or freshets, enter the northern portion of the Massachusetts Bay. These freshets originate from the Merrimack River and other northern rivers flowing into the Gulf of Maine. They are transported south along the Maine/Massachusetts coastline by the Gulf of Maine coastal current and enter Massachusetts Bay just south of Cape Ann.

Figure 3-6 presents the Merrimack River flows for 1992-94. In general, the highest flows occur during the spring and the lowest during the summer. During 1992, the earliest high flow of the three years was observed in mid-March. The spring flows in 1992 were lower than the other years, but extended over a longer period to time. The summer flows were highest in 1992. During 1993, a peak flow of approximately 1500 m^3/s was observed and high flow lasted from late March to early May. During 1994, peak flows were not as great in magnitude as 1993, but high flows persisted over a longer period of time. On an annual basis the three years did not differ greatly. The average Merrimack River flows for 1992, 1993 and 1994 were 214.1, 227.2 and 234.5 m^3/s, respectively.

Temporal distributions of surface and bottom salinity data are presented in Figure 3-7. It is difficult to discern the temporal pattern of salinity at station F23 due to the paucity of samples. However, some evidence of salinity stratification can be observed in March, 1992 and April of 1993 and 1994. There is also stratification shown in the data for August 1994. While some of the salinity stratification at this station may be related to local runoff, it may also result from the stage of the tide and/or wind conditions which bring a freshwater signal from the Deer Island effluent plume into the region being sampled. Evidence of salinity stratification can also be observed at station F01 during June and August 1992, and April and June 1994. In 1993, both the surface and bottom salinity concentrations declined in June, possibly a consequence of the high spring freshet.

Salinity data were collected much more frequently at the three near field stations, namely N10, N06 and N07. A more discernable pattern in seasonal stratification/destratification is evident in the data from these stations. In 1992, lower salinities, resulting from the spring freshet, were observed in May. In the spring of 1993, salinities at these station were lower than 1992 due to the higher flows associated with the spring freshet of 1993. The lowest salinity concentrations

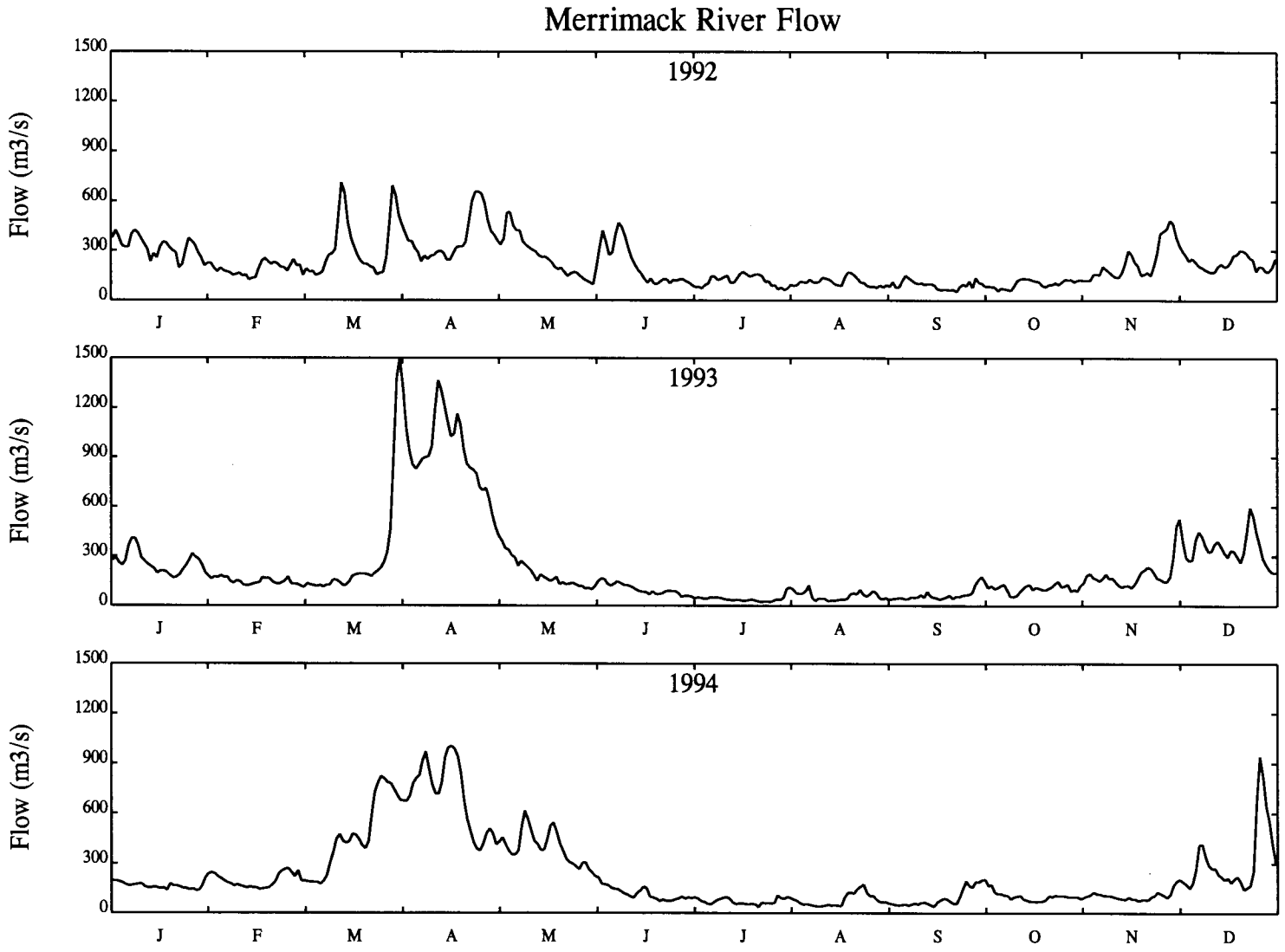


Figure 3-6. Merrimack River Flows 1992, 1993 and 1994

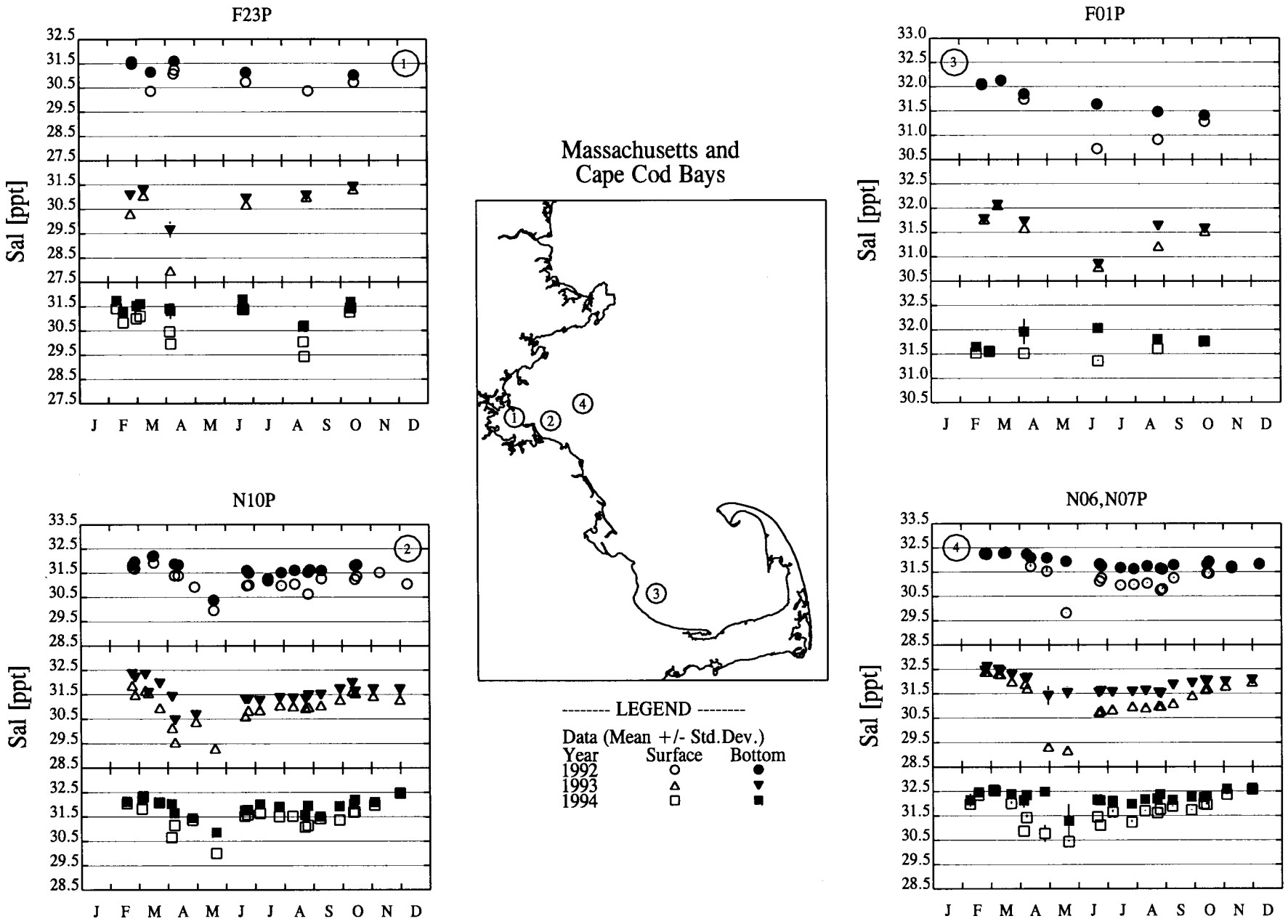


Figure 3-7. Temporal Comparison of Salinity Data

measured during 1992-94 were observed at this time. The freshet in 1994 occurred in March, April and May. It can also be observed that salinity concentrations tended to be higher in both the surface and bottom waters during the second half of 1994 compared to the other years, suggesting perhaps, a greater intrusion of more saline waters from the Gulf of Maine than occurred in 1992 or 1993.

Figures 3-8 and 3-9 present probability distributions of surface and bottom salinity data, respectively. The surface salinity in 1992 and 1993 at the near field stations appear quite similar across seasons except that the more intense freshet of 1993 resulted in lower spring salinities. More saline conditions were observed in 1994 in the surface waters of the near field stations, especially in the summer and fall. The annual distribution of salinity at the far field stations do not show significant year to year variability. The bottom salinity data show similar patterns as were observed in the surface data, except that differences due to the spring freshet are not as pronounced.

3.2.1.3 Density

The density of a parcel of water is determined by its salinity and temperature composition. Therefore, vertical density stratification is primarily a function of differences in temperature and salinity throughout the water column. Figure 3-10 presents the difference in density between the surface and bottom layers in units of sigma-t. Density was not measured directly, but was calculated using temperature and salinity data. During the winter, the water column was well mixed and only small differences in density were observed. In the spring, the greatest degree of stratification was observed in 1993 and was probably associated with the spring freshet. The year to year variability in the summer and fall was relatively minor. However, it is interesting to note that the combination of higher salinity and higher bottom water temperature in the fall of 1994 resulted in a smaller vertical density gradient than was observed in 1992 and 1993.

3.2.2 Chemical and Biological Water Quality Parameters

3.2.2.1 Chlorophyll-a

Annual variation in phytoplankton biomass is an important issue as MWRA readies to discharge at the ocean outfall. One measure of phytoplankton biomass is chlorophyll-a. Figures 3-11 and 3-12 present a series of seasonal and annual probability distributions of near field and

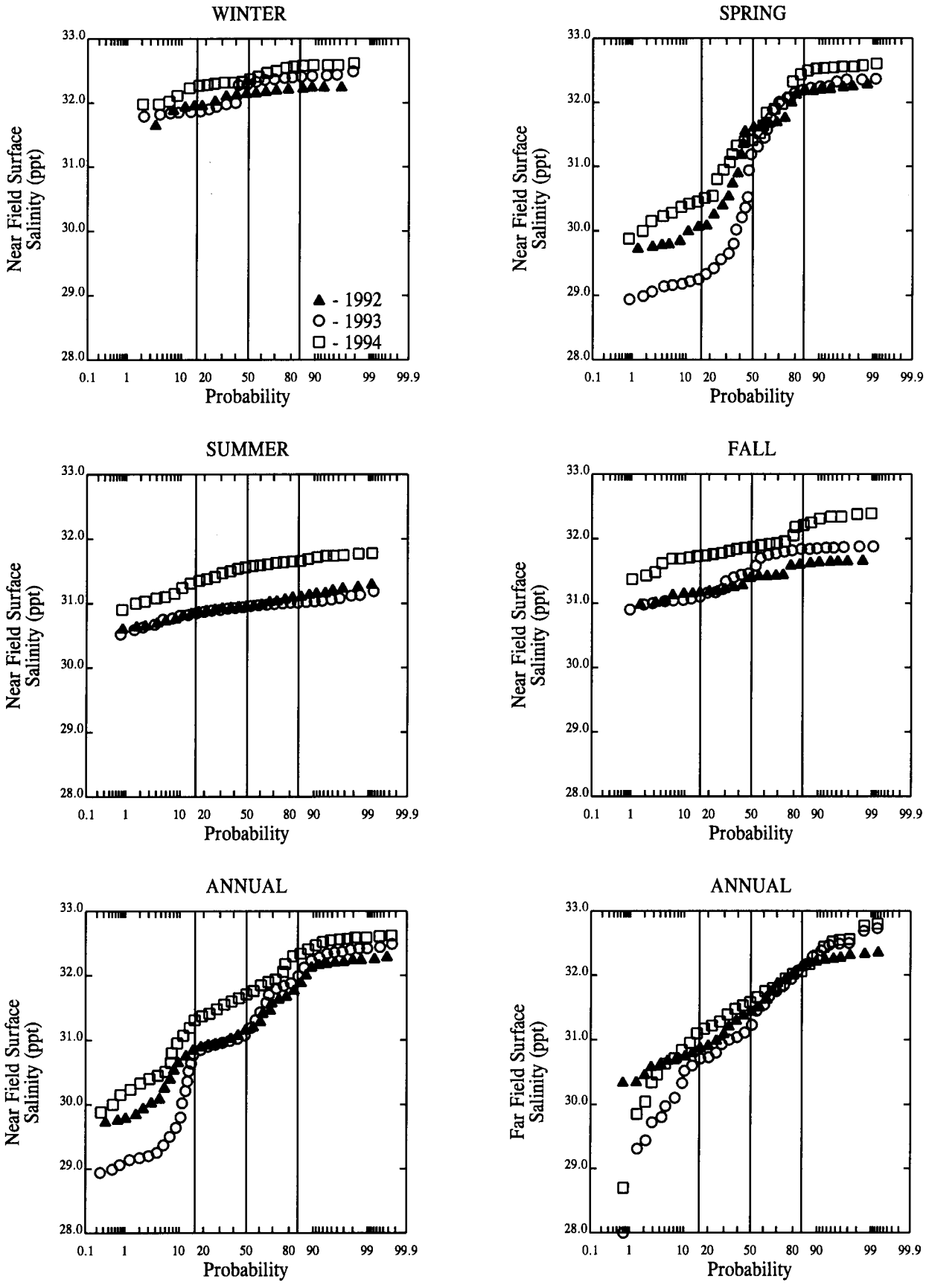


Figure 3-8. Seasonal and Annual Probability Distributions of Surface Salinity

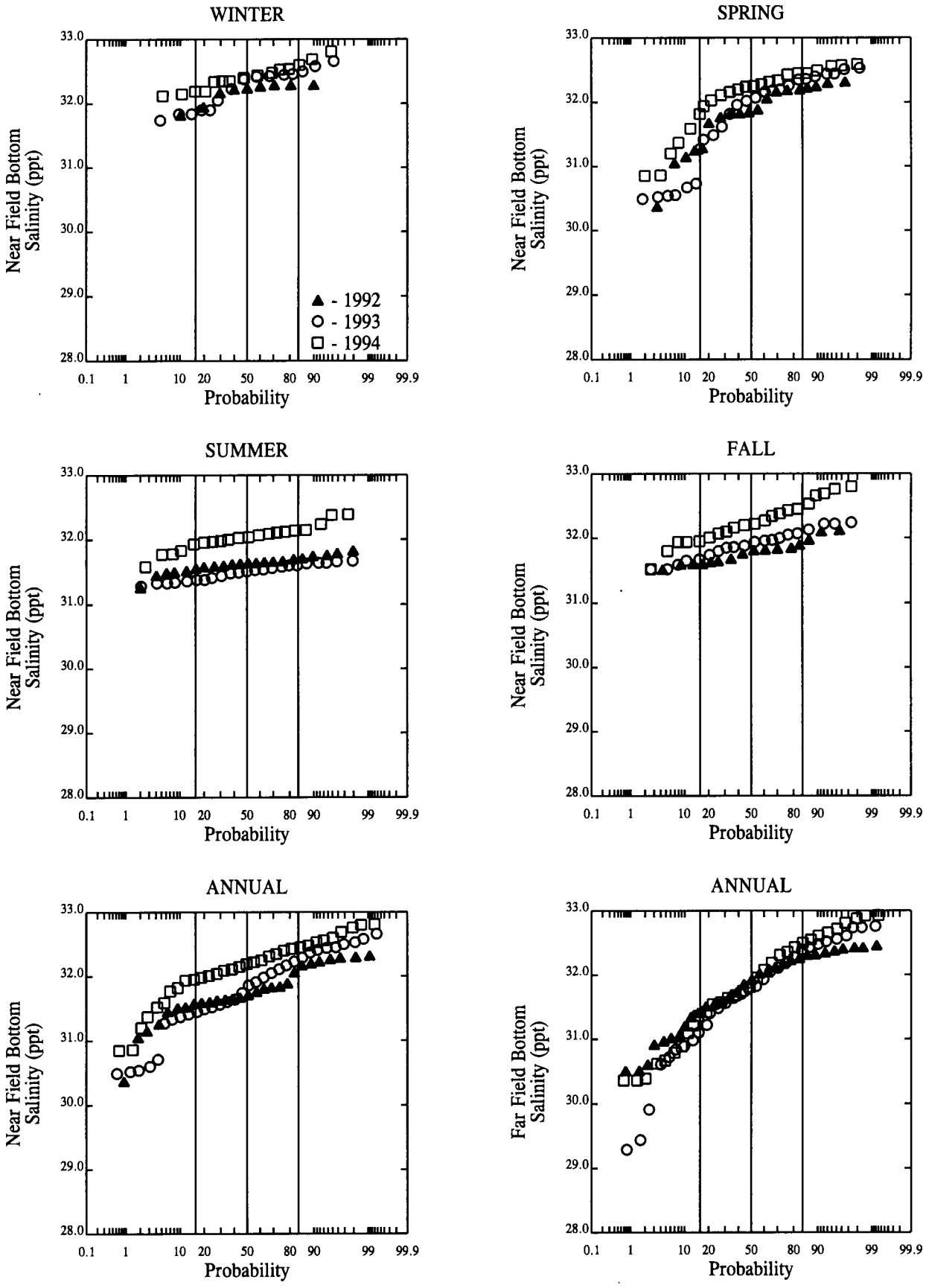


Figure 3-9. Seasonal and Annual Probability Distributions of Bottom Salinity

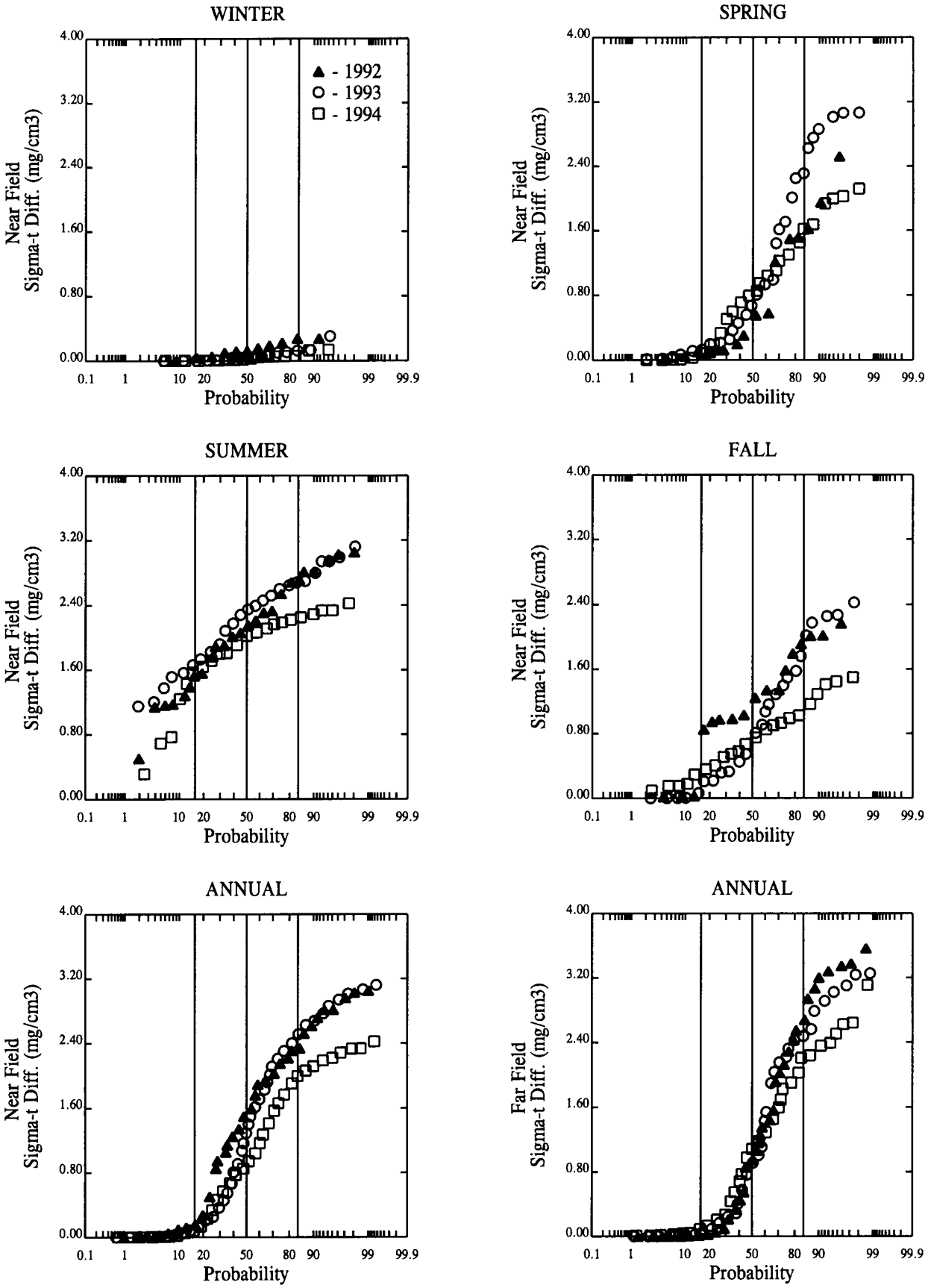


Figure 3-10. Seasonal and Annual Probability Distributions of Density Stratification

far field surface and mid-depth chlorophyll-a data, respectively. A review of the surface chlorophyll-a data (Figure 3-11) suggests that algal blooms occurred in different seasons during the three years. The highest winter chlorophyll-a concentrations were observed in 1992. Spring chlorophyll-a values were highest in 1994. Summertime chlorophyll-a concentrations were similar for all three years. Generally, the highest surface chlorophyll-a concentrations were observed in the fall, with 1993 having the highest concentrations of the three years.

The mid-depth chlorophyll-a data are presented in Figure 3-12. In this analysis, mid-depth was defined as occurring between 12.5 and 17.5 m from the water surface. In general, these data show the same seasonal pattern as the surface data. During the winter chlorophyll-a concentrations were higher at the surface than at mid-depth. The summer chlorophyll-a surface and mid-depth chlorophyll-a data had similar medians, however, the mid-depth data did not show as much variation. The spring and fall surface and mid-depth concentration data were similar to one another for the three years of data.

Estimates of phytoplankton biomass as chlorophyll-a were measured using two techniques in the HOM program: (1) as discrete chlorophyll-a measurements from bottle samples taken at fixed depths (Figures 3-11 and 3-12), and (2) using a continuous fluorometric recorder that was attached to the CTD recorder. The latter data are presented in Figures 3-13 (surface) and 3-14 (mid-depth) as probability distributions. These data show similar seasonal medians and distributions when compared to the discrete chlorophyll-a data. The highest winter fluorescence concentrations were measured in 1992, the highest spring fluorescence levels, were measured in 1994 and the highest fall fluorescence concentration were observed in 1993. During the summer all three years had similar fluorescence levels. On an annual basis, the median fluorescence concentrations were similar for all three years. 1993 had the highest observed fluorescence measurements and 1994 had the highest minimum observed fluorescence concentrations. Higher fluorescence measurements were observed at the far field stations during 1993 than at the near field stations.

The mid-depth fluorescence data (Figure 3-14) show more clearly the differences between seasons and years than can be discerned using the sparse mid-depth chlorophyll-a data. In general, the mid-depth fluorescence data have higher concentrations than the surface fluorescence data. Exceptions are the winter of 1992 and the fall of 1993 when the levels are comparable. The mid-depth fluorescence data clearly show a spring bloom in 1994 and a fall bloom in 1993.

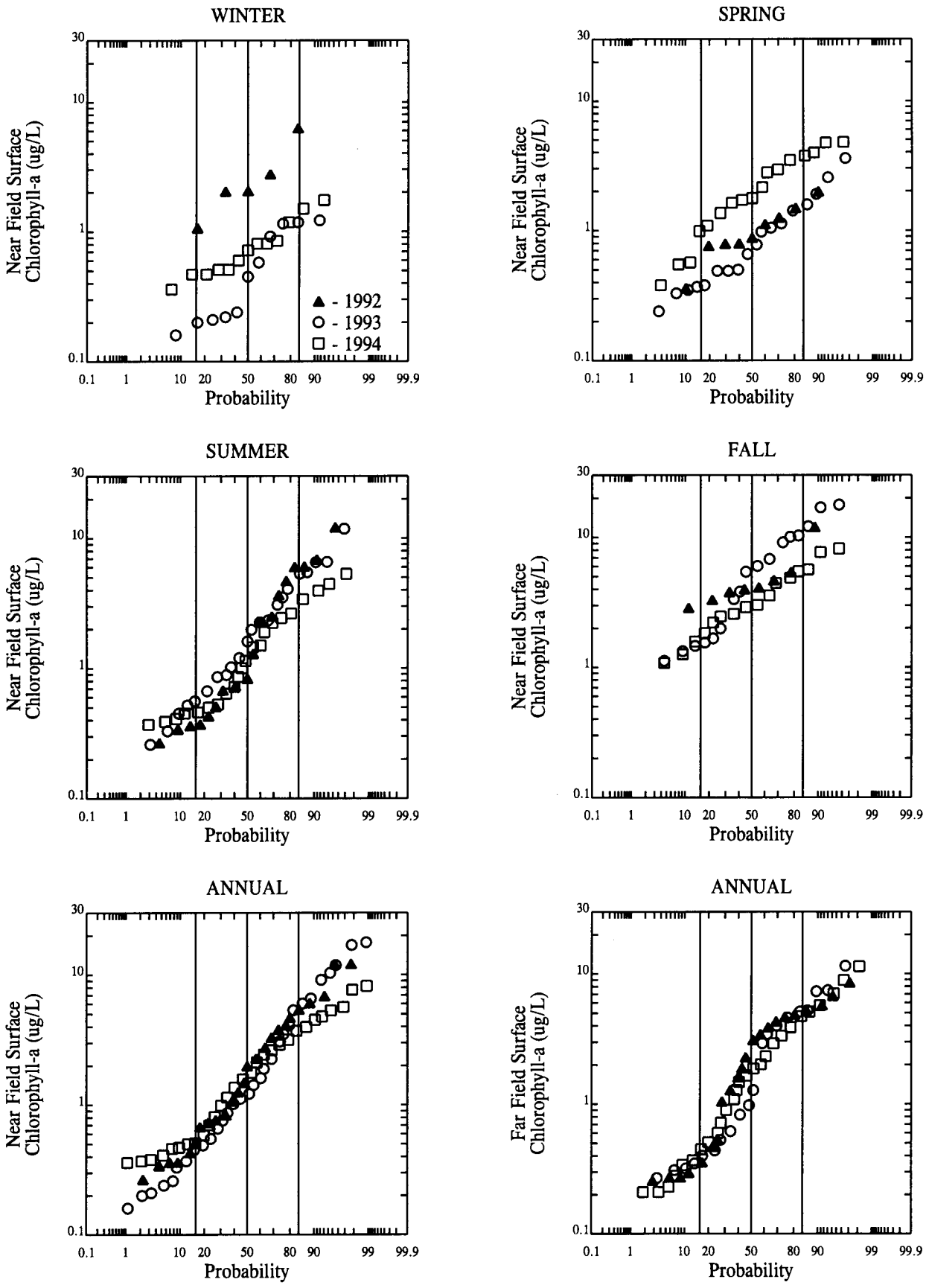


Figure 3-11. Seasonal and Annual Probability Distributions of Surface Chlorophyll-a

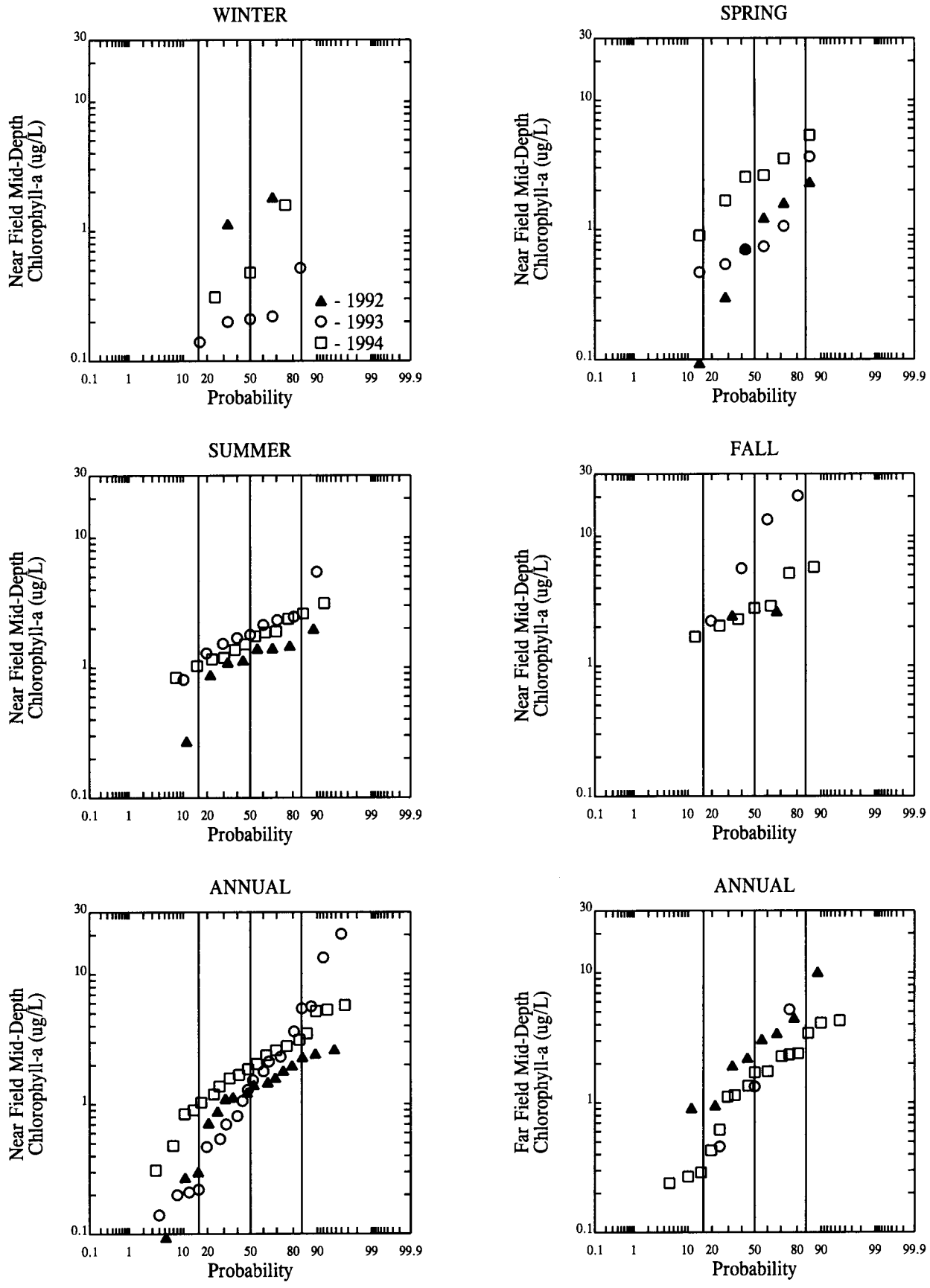


Figure 3-12. Seasonal and Annual Probability Distributions of Mid-Depth Chlorophyll-a

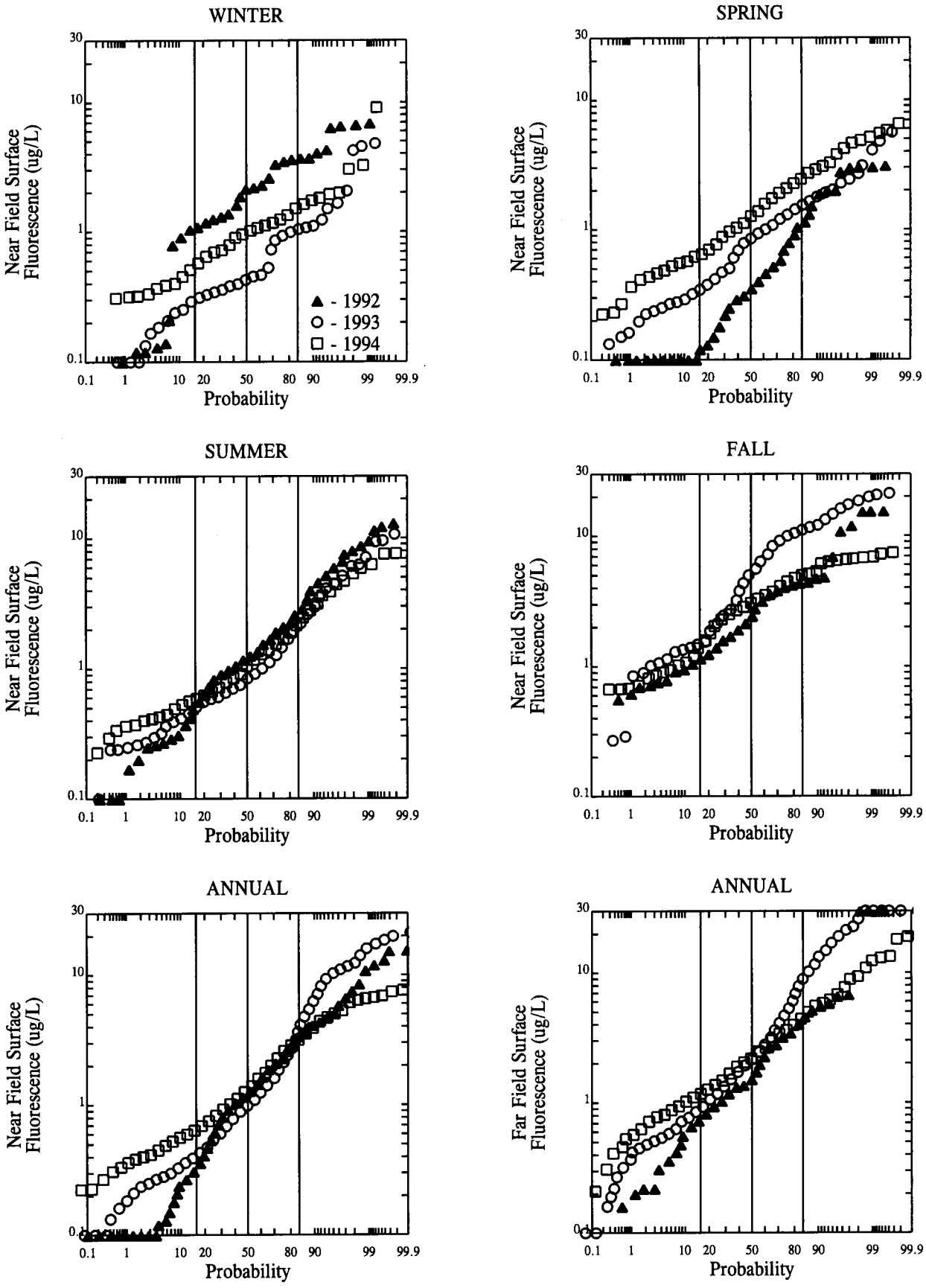


Figure 3-13. Seasonal and Annual Probability Distributions of Surface Fluorescence

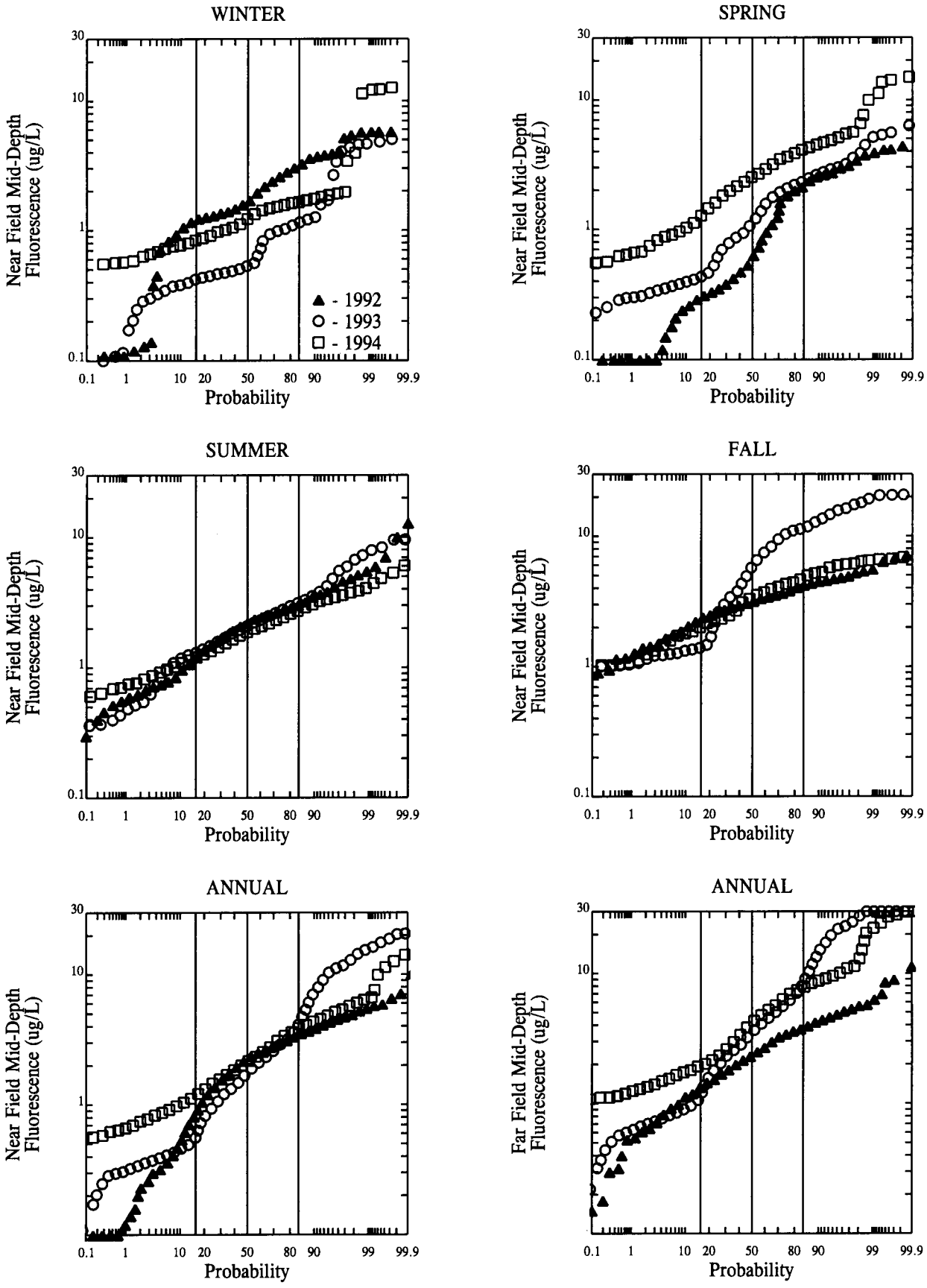


Figure 3-14. Seasonal and Annual Probability Distributions of Mid-Depth Fluorescence

3.2.2.2 Particulate Organic Carbon

Particulate organic carbon (POC) can be used as an indicator of algal biomass in regions removed from anthropogenic sources. In areas where algal biomass dominates the composition of the POC sample, POC may be a better indicator of algal biomass than chlorophyll-a because phytoplankton can adjust their internal concentration of chlorophyll-a in response to certain environmental conditions. Figure 3-15 presents the surface nearfield and farfield POC data probability distributions for 1992-94. Analysis of the POC probability distributions provides some contrast to the chlorophyll-a data. For example POC concentrations observed in 1992 tend to be higher year-round, while the 1994 POC concentrations were generally the lowest year-round of the three years.

3.2.2.3 Phosphorus

Inorganic phosphorus (ortho-phosphate, PO_4) is an important nutrient for phytoplankton growth. Based on an analysis of the data, phosphorus does not appear to be a limiting nutrient in Massachusetts Bay. Variations in the concentrations of PO_4 in Massachusetts Bay appear to be associated with changes in algal biomass and Gulf of Maine inputs associated with the spring freshet. Figure 3-16 presents seasonal and annual probability distributions for surface PO_4. A comparison of the winter data shows that the 1992 PO_4 concentrations were lower than the other two years. This may have resulted from the fact that chlorophyll-a concentrations were higher in 1992 compared to the other years so the uptake of PO_4 by algae may account for the lower PO_4 concentrations observed in 1992. Comparisons of spring PO_4 data show little difference from year to year. During the summer months, 1994 had the highest concentrations of PO_4 while 1993 had the lowest. It is not clear why this is the case, since our estimates of anthropogenic phosphorus loads were lowest for 1994.

3.2.2.4 Nitrogen

Dissolved inorganic nitrogen (DIN) is another important nutrient for algal growth. DIN is the sum of ammonium (NH_4), nitrate (NO_3) and nitrite (NO_2) nitrogen. Figure 3-17 presents seasonal and annual probability distributions for surface DIN. During the winter months for these three years the DIN concentrations were very similar. In the spring, surface DIN concentrations decreased significantly in response to algal growth and primary production. The median spring

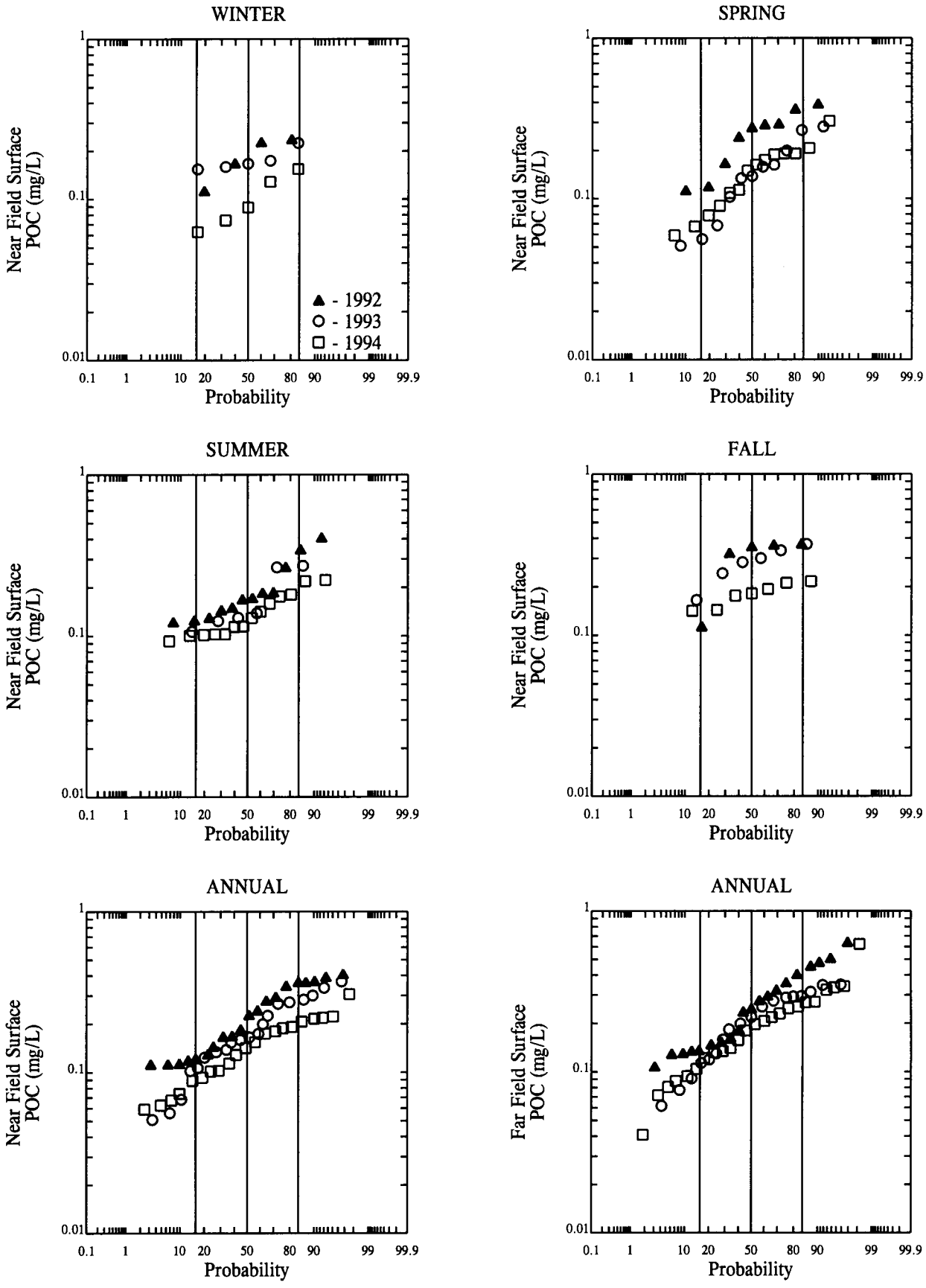


Figure 3-15. Seasonal and Annual Probability Distributions of Surface POC

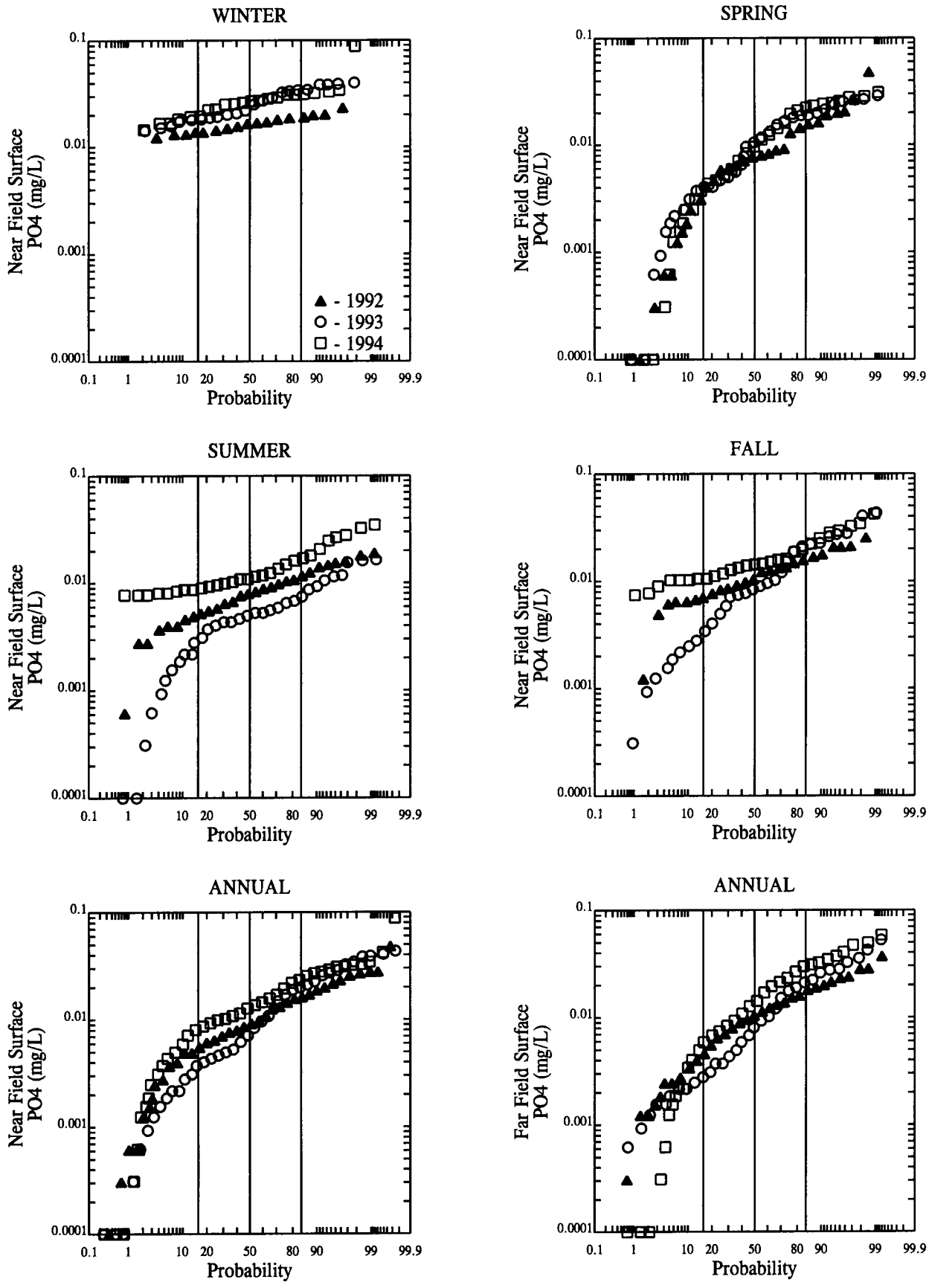


Figure 3-16. Seasonal and Annual Probability Distributions of Surface PO_4

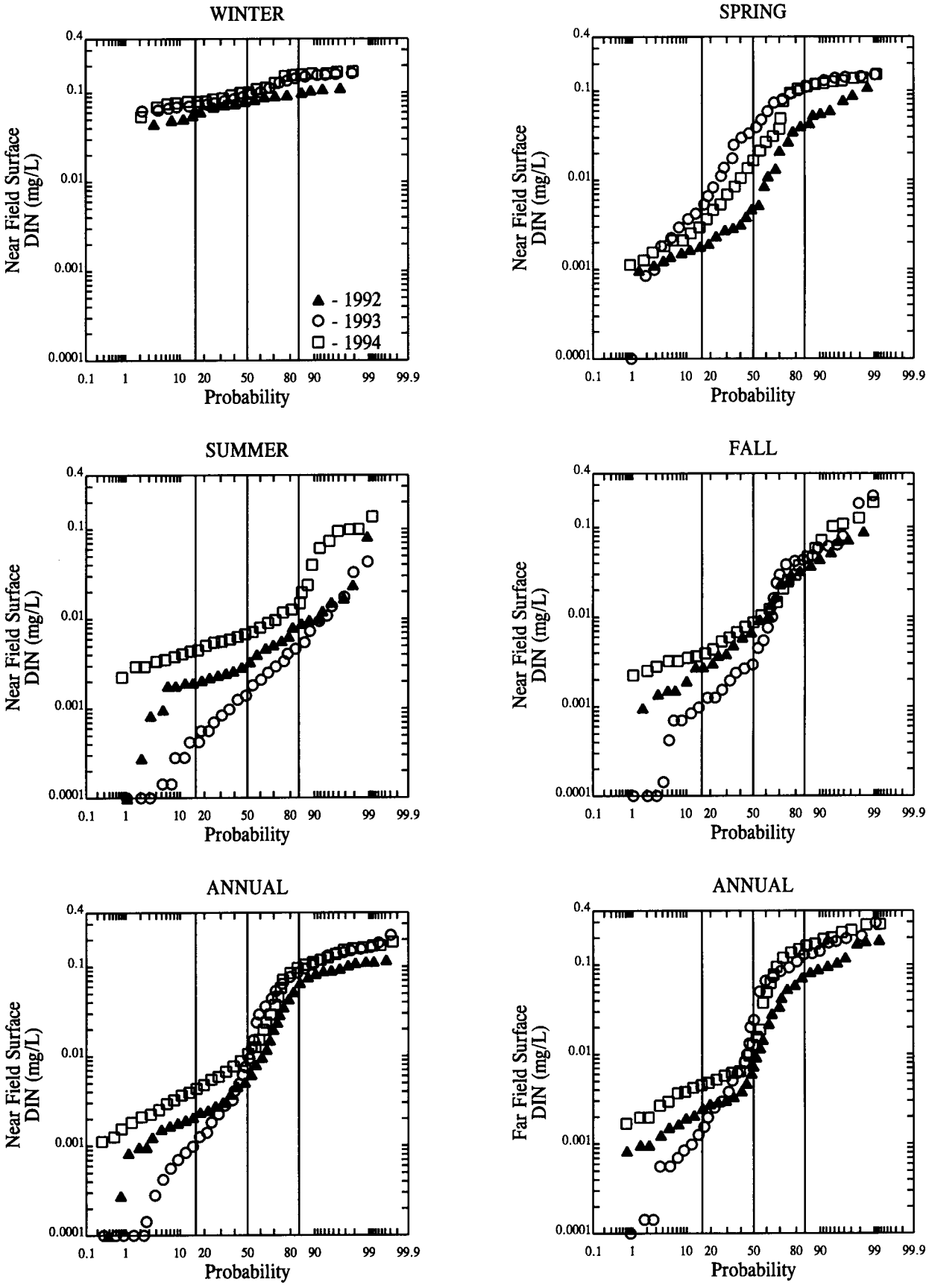


Figure 3-17. Seasonal and Annual Probability Distributions of Surface DIN

DIN concentration in 1992 was an order of magnitude less than the median concentration in 1993 consistent with higher spring concentrations of POC, but inconsistent with chlorophyll-a data. However, it may be consistent with the occurrence of a *Phaeocystis* bloom reported in the spring of 1992. In addition, the *Phaeocystis* may have had a lower carbon to nitrogen ratio and a higher carbon to chlorophyll ratio than are usually associated with "normal" spring algal blooms. The summer surface DIN concentrations were lowest in 1993. However, most of the summer DIN data for all three years were at levels thought to limit algal growth. In the fall, 1993 continued to have the lowest DIN concentrations, consistent with the fall 1993 algal boom.

3.2.2.5 Silica

Dissolved silica (DSi) is an important nutrient for diatom growth. During diatom blooms Massachusetts Bay can become silica limited. Figure 3-18 shows the seasonal and annual surface DSi probability distributions for 1992-94. The lowest silica concentrations on these figures correspond very well to periods of time when the highest chlorophyll-a concentrations were observed during a particular year, indicating algal uptake of silica during these periods. Summer and fall 1993 DSi concentrations were low, compared to the other two years, consistent with a large late summer/fall diatom bloom observed during that same period.

3.2.2.6 Nitrogen to Phosphorus Ratio

The Redfield ratio for phytoplankton can be used to determine the potentially limiting nutrient for phytoplankton growth. For nitrogen and phosphorus, a DIN to DIP ratio of approximately 7, on a weight basis, would indicate that both DIN and DIP limit phytoplankton equally. A DIN/DIP greater than 7 would indicate potential phosphorus limitation while a ratio less than 7 suggests potential nitrogen limitation. However, this ratio only indicates potential limitation and one must look at the actual concentrations of DIN and DIP. Figure 3-19 presents the DIN to DIP ratio probability distributions for the surface concentrations at the near and far field stations for 1992-94. The figure shows that for all three years, nitrogen is the potentially limiting nutrient year round with the exception of a few observations in the spring.

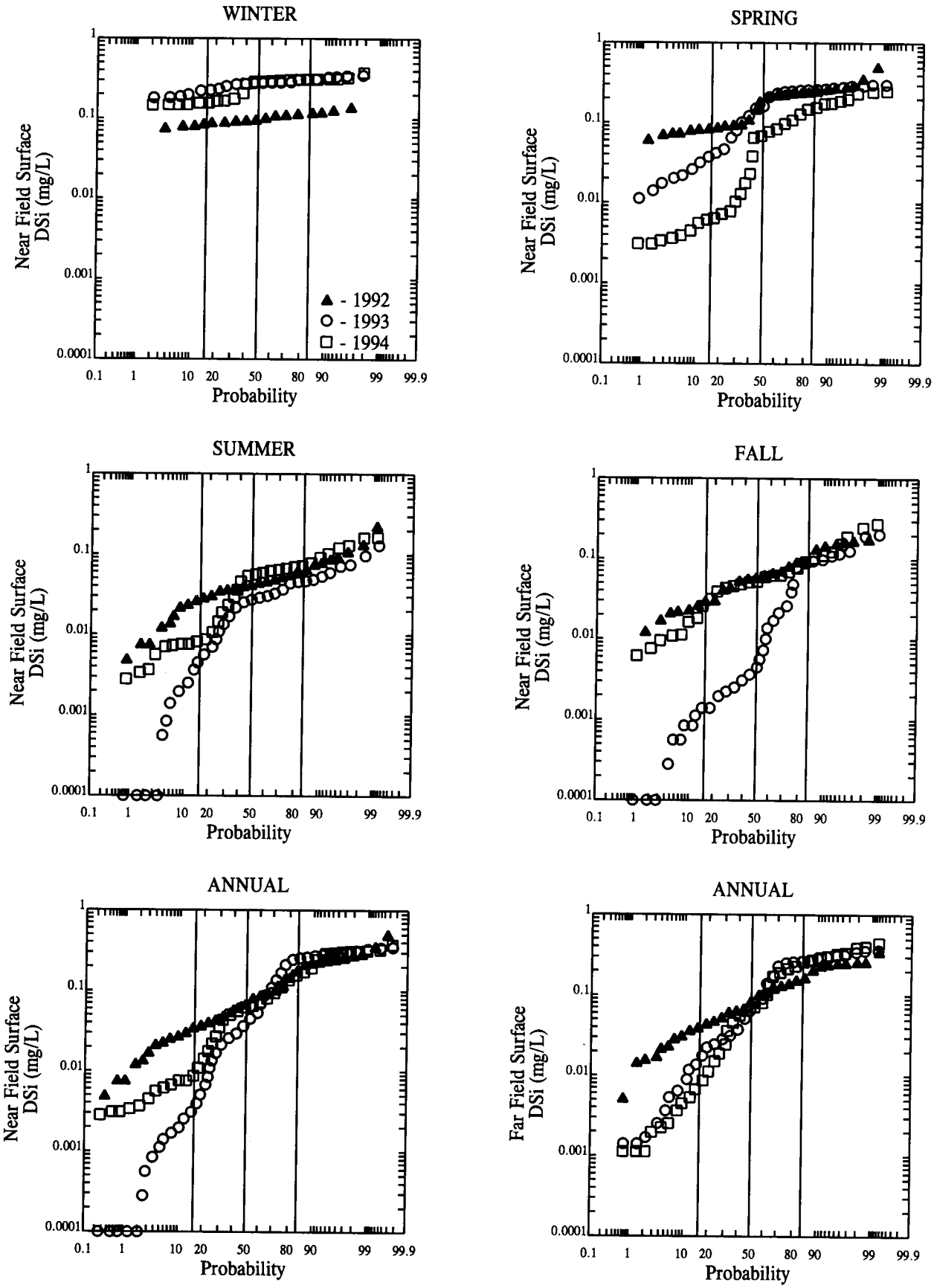


Figure 3-18. Seasonal and Annual Probability Distributions of Surface DSi

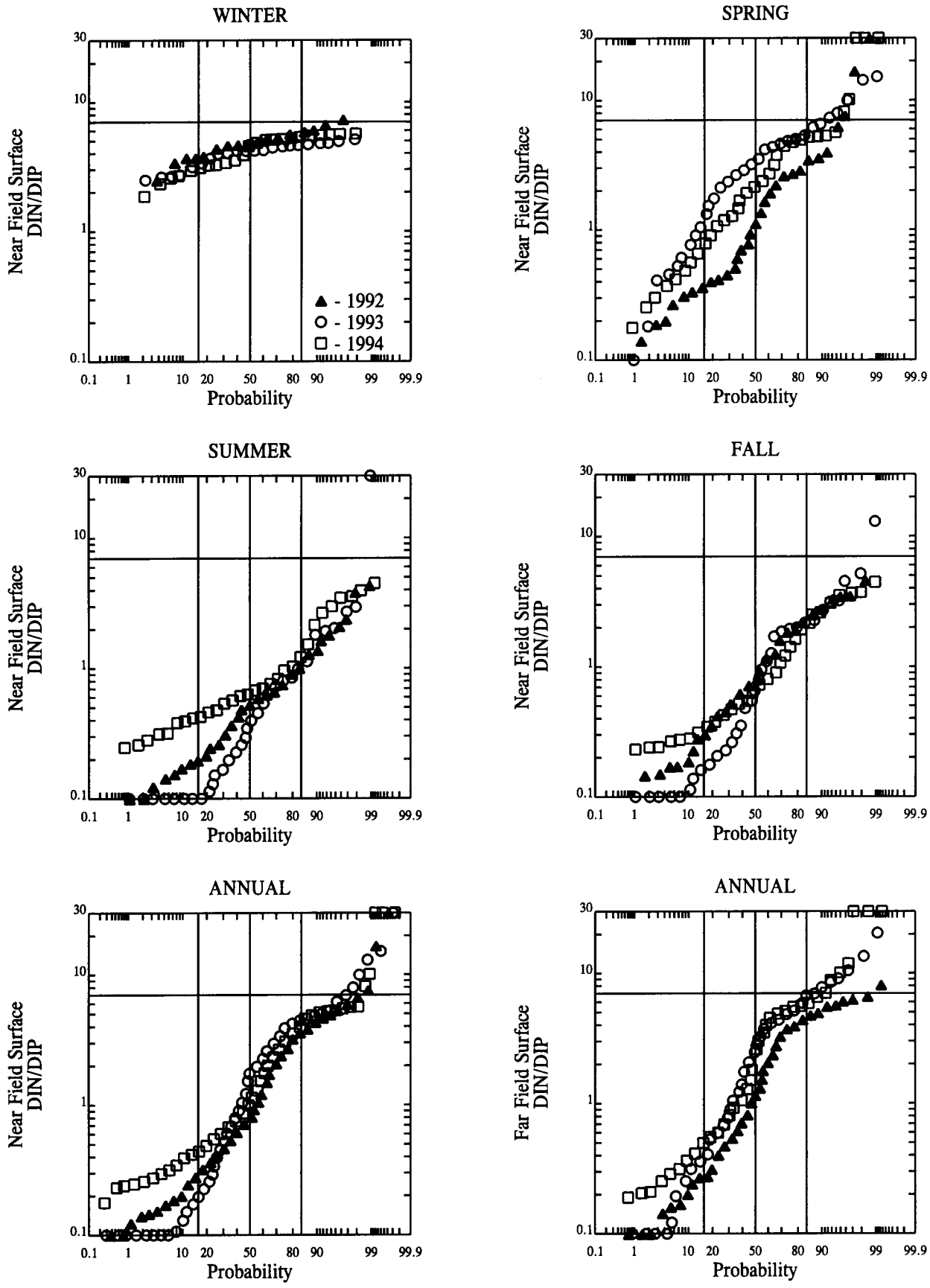


Figure 3-19. Seasonal and Annual Probability Distributions of DIN/DIP

3.2.2.7 Nitrogen to Silica Ratio

Determining a ratio at which nitrogen or silica is the potentially limiting nutrient is not as straightforward as it is for nitrogen and phosphorus. Diatoms are the only algal group that require silica, so that when a mixed assemblage of algae are present the silica concentration levels may only be limiting to diatom growth. Assuming, however, that only diatoms are present, a DIN to DSi ratio of less than 0.5 suggests that DIN may potentially limit diatom growth, while a ratio greater than 0.5 suggests that silica may limit diatom growth. Figure 3-20 presents the probability distributions for surface DIN to DSi ratio at the near field and far field stations. During 1992, diatom growth appears to be potentially limited by DSi in the winter and DIN during most of the remainder of the year. In 1993, diatom growth was potentially limited by DIN more so than DSi. In 1994, diatom growth was potentially limited by DIN and DSi almost equally.

3.2.2.8 Dissolved Oxygen

The dissolved oxygen (DO) of a water body is one of the more important water quality parameters. Since DO is necessary to maintain aquatic life, it reflects the general health level of a water body and indicates the ability of an aquatic ecosystem to support a balanced habitat. When periods of hypoxia (DO < 3.0 mg O_2/L) occur, the ability of fish and other aquatic life to reproduce and grow may be impaired. In Massachusetts and Cape Cod Bays, the state DO standard is never less than 6.0 mg O_2/L.

Figure 3-21 presents seasonal and annual bottom DO probability distributions for 1992-94 (note the scale differences for each season). The fact that fewer DO samples were taken in the winter of 1992 make comparisons to the other years more difficult for the winter period. The 1993 and 1994 winter bottom DO data are similar. The spring of 1992 had the lowest DO while the spring of 1993 had the highest. For the summer period, the 1992 and 1993 DO data were nearly the same. Bottom water DO data were markedly lower (1-2 mg O_2/L) in 1994. Generally, the lowest concentration of DO were observed in the fall of each year. Data from the fall of 1994 indicate that 1994 was a more critical year with 20 percent of the near-field bottom water at or below the state standard.

It is important to recall from the loading analysis presented in Section 2 that the nutrient loads entering Massachusetts and Cape Cod Bays from land based sources and the atmosphere did not vary

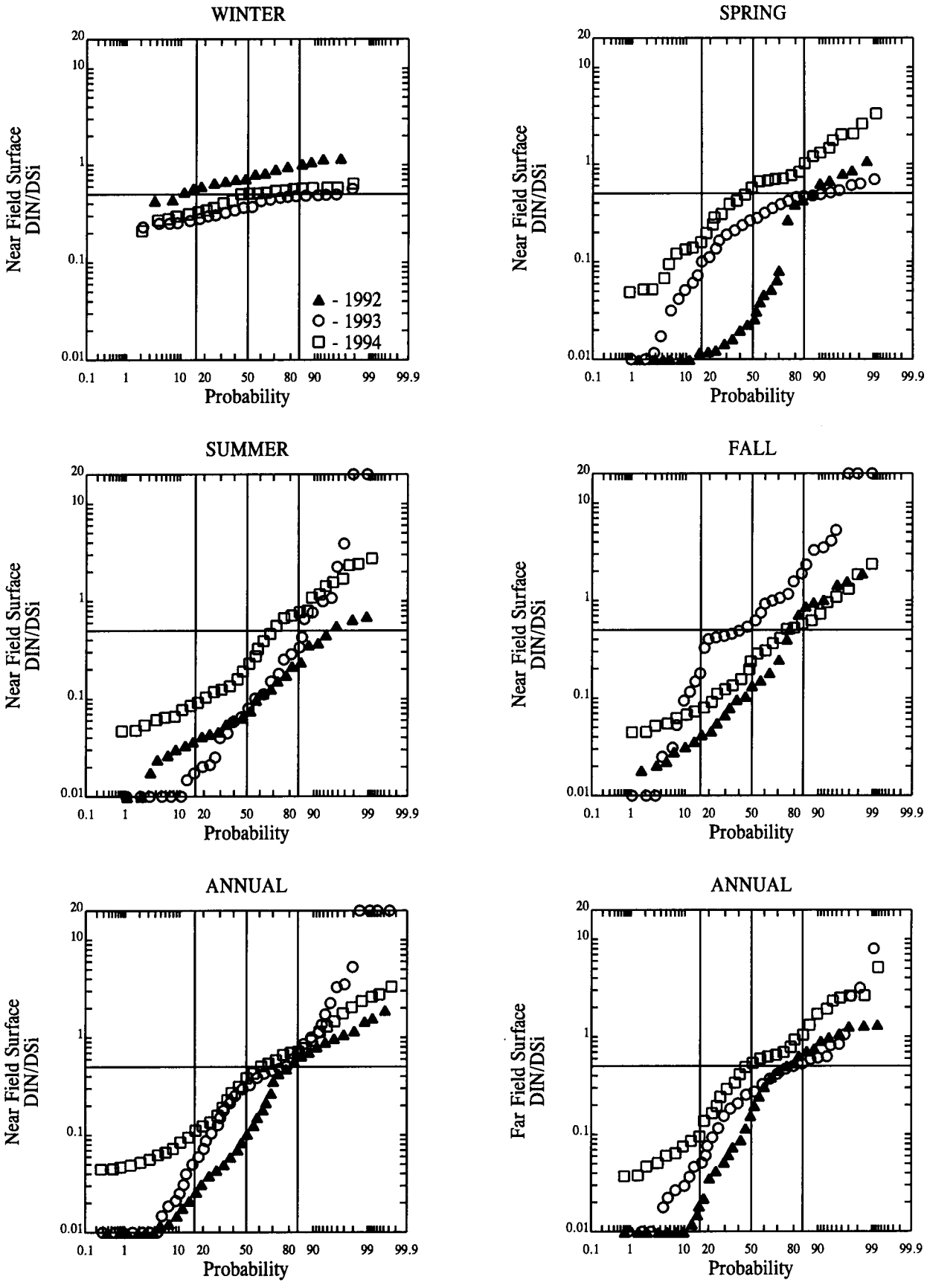


Figure 3-20. Seasonal and Annual Probability Distributions of DIN/DSi

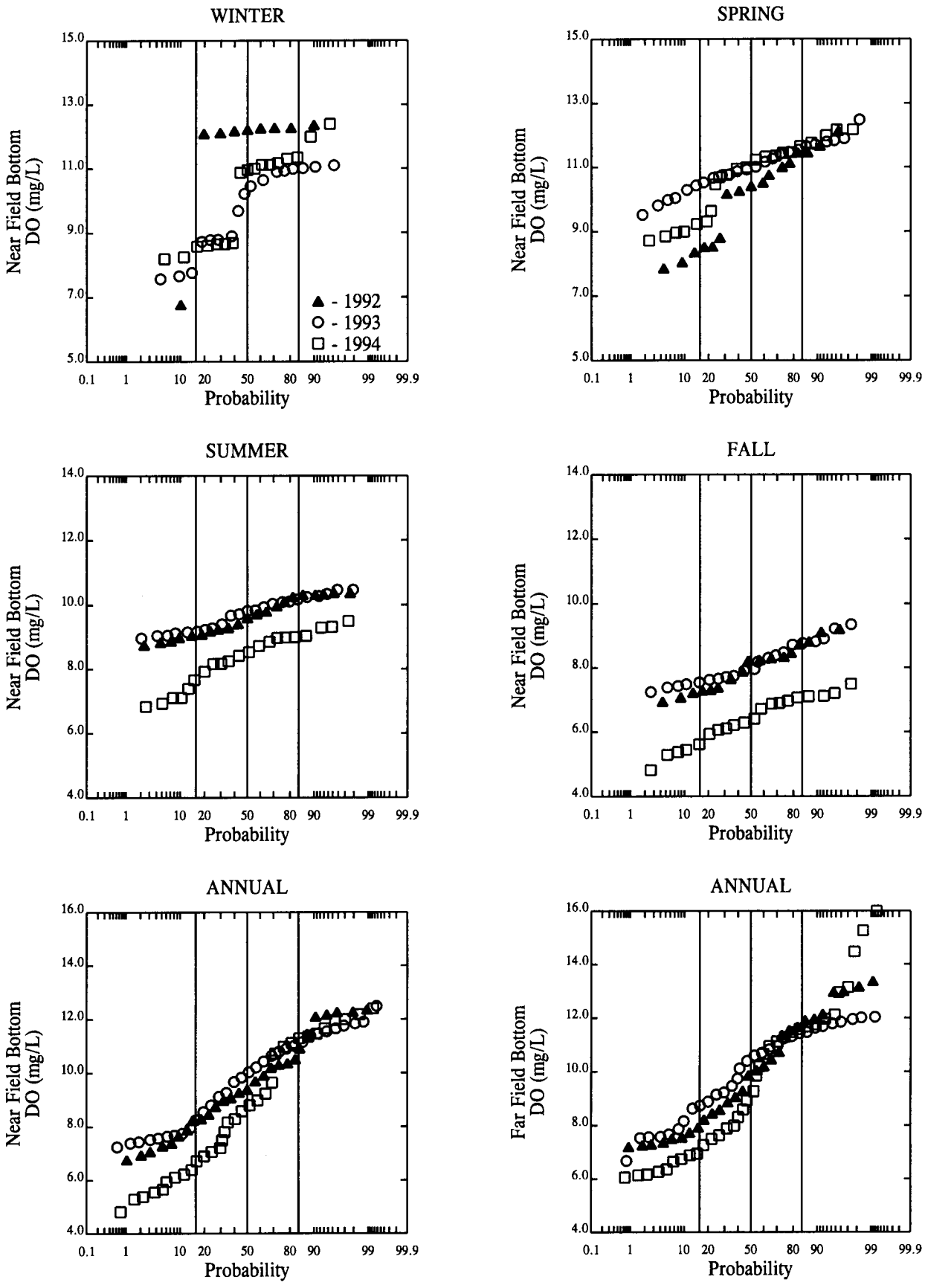


Figure 3-21. Seasonal and Annual Probability Distributions of Bottom DO

greatly year to year from 1992-1994. Therefore, it is improbable that changes in the anthropogenic loading caused the variability observed in the bottom water DO data. It is more likely that the higher bottom water temperatures in the late summer and fall of 1994 resulted in higher rates of bottom water and sediment metabolism which consumed oxygen at a higher rate that the other two years. It is also possible that waters low in DO intruded from the Gulf of Maine into Massachusetts Bay as suggested by bottom water temperature and salinity data. However, the one conclusion that can be made is that there is a good deal of natural year to year variation in the Massachusetts and Cape Cod Bays.

3.3 BOUNDARY DATA

The 1992 through 1994 boundary conditions were determined using data from the HOM program. Far field surveys were conducted in February, March, April, June, August, and October of each year. Because of their relative proximity to the boundary of the model domain (Figure 3-22) five stations (F21, F22, F12, F08, and F04) were chosen to guide the determination of the boundary conditions for the original 1992 calibration. However, these stations are still a distance from the actual model boundary and are on the interior side of the boundary. Furthermore, these stations were sampled for only a limited number water quality parameters, i.e., PO_4, NH_4, NO_2+NO_3, DSi, and DO. As a consequence there was some uncertainty in the estimations of the original boundary conditions. Therefore, engineering judgement and calibration analysis, i.e., comparisons of model computations to observed data at these five stations, were also used to determine the boundary conditions for the original 1992 calibration period. In 1993, the same sampling stations were monitored. Based, in part, upon recommendations made by HydroQual, additional stations (F26, F27, F28 and F29), which were located closer to the boundary of BEM, were added to the HOM program, while some others (F08 and F04) were deleted. These new sampling stations provided better data from which to assign boundary conditions. In addition, chlorophyll-a, POC, PON, TOC, TDN, and TDP were included in the constituents being sampled at station F27.

As a consequence of the new 1994 sampling locations and additional variables sampled, the boundary conditions for chlorophyll-a, POC, PON, DOC, DON, and DOP for all three years being modeled were, in part, guided by the 1994 data. Figures 3-23 and 3-24 present the data collected in 1994 at station F27. Although there was an improvement in the ability to assign boundary conditions with the 1994 data, there were still many data gaps which required engineering judgement to fill.

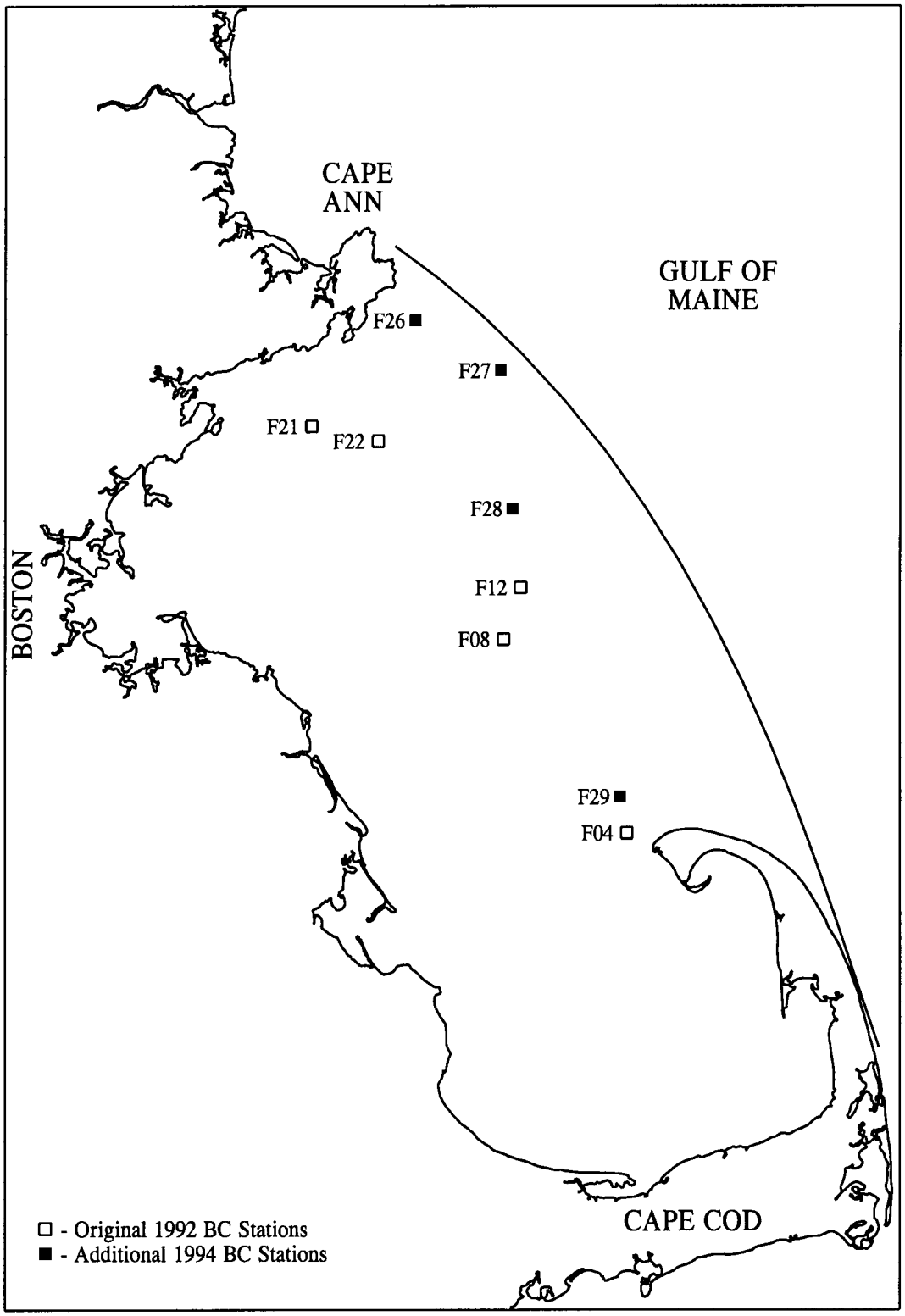


Figure 3-22. Stations Used for Boundary Condition Data

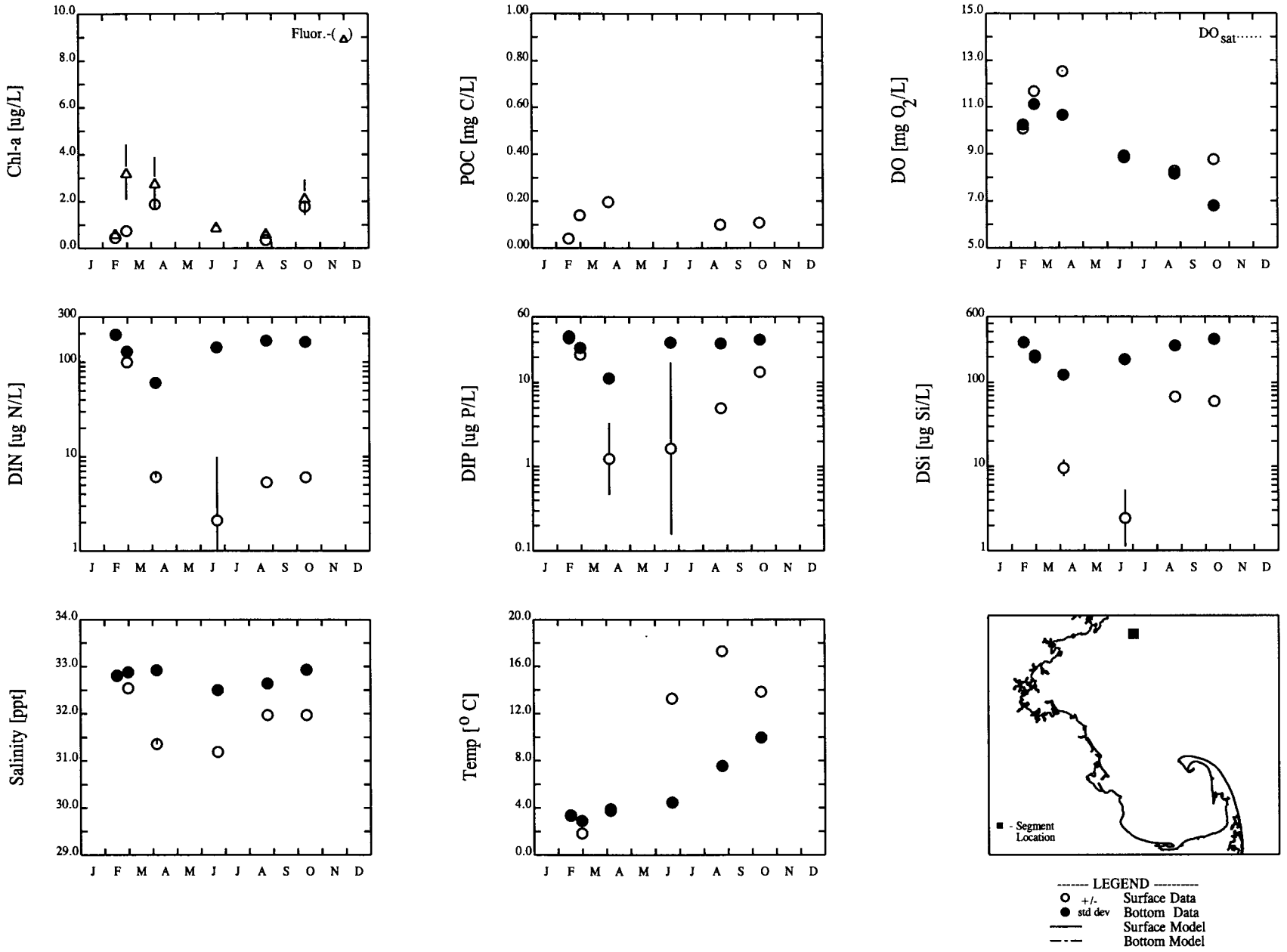


Figure 3-23. Example Boundary Condition Data

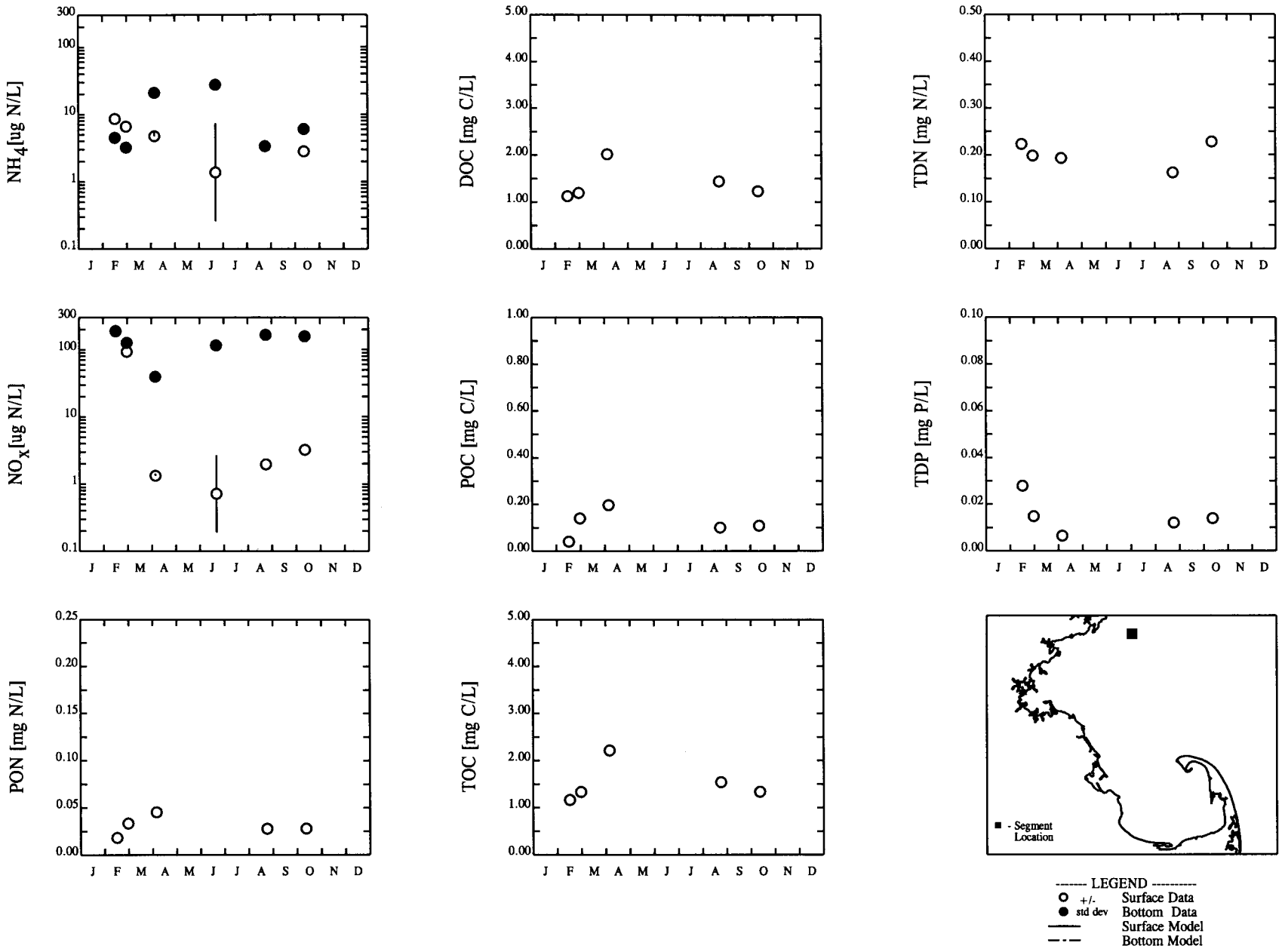


Figure 3-24. Additional Example Boundary Condition Data

3.4 SEDIMENT DATA

The physical, chemical, and biological processes that occur in the sediments of coastal environments play an important role in the decomposition of organic matter and recycling of nutrients. The end-products of these sediment processes can have an important influence on primary productivity and dissolved oxygen in the overlying water column. During organic matter decomposition in sediments there is considerable oxygen demand which must be supplied from the overlying water column. This sediment oxygen demand (SOD) may comprise a substantial fraction of the total system oxygen consumption. Over lengthy time scales (e.g., years to decades), the sediments can act as an ultimate sink of nutrients and other substances discharged to the water column. Over lesser time scales (e.g., months to seasons), however, sediment release of previously-deposited nutrients can be a net source to the water column. Therefore, in order to gain insight into the potential effects of benthic SOD and nutrient regeneration on the coastal ecosystem resulting from the relocation of the Boston Harbor outfall, MWRA funded a series of field and laboratory sampling programs. These studies included measurements of oxygen uptake, nutrient flux and sediment pore water.

Sediment flux and pore water data are available from two sources: the Ecosystems Center of the Marine Biological Laboratory (Giblin et al., 1991, 1992, 1993, 1994, 1995; Tucker et al., 1993) and a program conducted by a joint Battelle Ocean Sciences and University of Rhode Island team (Kelly and Nowicki, 1992, 1993). Data were available for 1990 through 1994. Prior to 1993 data were available only for Boston Harbor and the immediate vicinity of Boston Harbor in Broad Sound and Massachusetts Bay. In 1993, some additional nutrient flux measurements were taken in Stellwagen Basin and Cape Cod Bay. Cape Cod Bay was not revisited in 1994.

The 1990 study, conducted by Marine Biological Laboratory (MBL) consisted of eight stations: three in the Outer Harbor, two in Broad Sound and three in Massachusetts Bay near the proposed outfall. Sediment measurements of SOD, NH_4 flux, NO_3 flux, and PO_4 flux were made in September and October. In 1991, only four stations were sampled by MBL during September, all in the Outer Harbor region of Boston Harbor. Measurements were made for SOD as well as NH_4, NO_3, and PO_4 flux. Pore water samples were analyzed for DIN, PO_4, alkalinity, and sulfide. During 1992, a more extensive sampling program was conducted. Boston Harbor stations were sampled in April, May, June, August, and November. Massachusetts Bay stations were sampled in October and November. Fluxes were measured for SOD, NH_4, NO_3, and PO_4. Pore water profiles were measured for DIN, PO_4, and sulfide. Sediment denitrification was also estimated using both direct methods (Kelly and Nowicki, 1993) and indirect stoichiometric techniques (Giblin et al, 1993).

In 1993, a total of 11 sites were sampled: three in Boston Harbor, six in Massachusetts Bay and two in Cape Cod Bay. Surveys were conducted in late February/early March, May, July, August, and October. Two sites in Boston Harbor and two sites in Massachusetts Bay were sampled during all five surveys. The Cape Cod Bay sites were sampled during August. Flux measurements were made for NH_4, NO_3, PO_4, DSi, urea, DIC, SOD, and N_2. Pore water profiles were determined for NH_4, NO_3, PO_4, urea, H_2S, pH, and alkalinity.

Eleven sites were sampled in 1994: four in Boston Harbor and seven in Massachusetts Bay. Surveys were conducted in March, May, July, August, and October. Three Massachusetts Bay sites were sampled during all five surveys and four stations were sampled only once during 1994. The same suite of flux and pore water measurements made in 1993 were made in 1994.

Figure 3-25 presents SOD data collected at four stations during the period of 1992 through 1994. Station BH02 is located in northern Boston Harbor near the Inner Harbor, while station BH08 is a sandy site in Hingham Bay in southern Boston Harbor. Station MB03 is located in northern Massachusetts Bay within the perimeter of the nearfield water quality stations and station MB05 is located in Stellwagen Basin where water depths are approximately 70 m. The highest SOD observed at those four sites were found at Station BH02. The highest SOD at BH02 was observed in May 1993 when an SOD of 3.3 gm O_2/m^2-d was measured. Except for this May SOD measurement, the SODs observed at this station generally followed a seasonal pattern related to bottom water temperatures, i.e., high SODs at high bottom water temperatures and low SODs at low bottom water temperatures. At station BH08 the observed SODs were significantly lower than observed in northern Boston Harbor. This may have been due, in part, to the choice of station location for BH08. The site chosen for BH08 was a sandy non-depositional area of Hingham Bay. Measurements of SOD at stations MB03 and MB05, in Massachusetts Bay were also low when compared to BH02. Maximum SOD measurements were observed in October and November at stations MB03 and MB05. These seasonal observations also appear to follow the seasonal trend of bottom water temperatures. The deeper waters of Massachusetts Bay reach their maximum temperatures in October and November.

Ammonia fluxes (J_{NH4}) measured during the period of 1992 through 1994 are shown in Figure 3-26. The J_{NH4} data followed a similar trend to the SOD data. In northern Boston Harbor, the highest J_{NH4} were observed during the warmer periods of the year. Lower J_{NH4} were observed in Hingham Bay than at station BH02. At station MB03, a clear seasonal trend was observed in all three years with J_{NH4} generally lower in the beginning of the year and rising steadily into October and November. The ammonia fluxes measured at station MB05 did not display an observable seasonal pattern, and were generally lower than ammonia fluxes measured at other sampling locations.

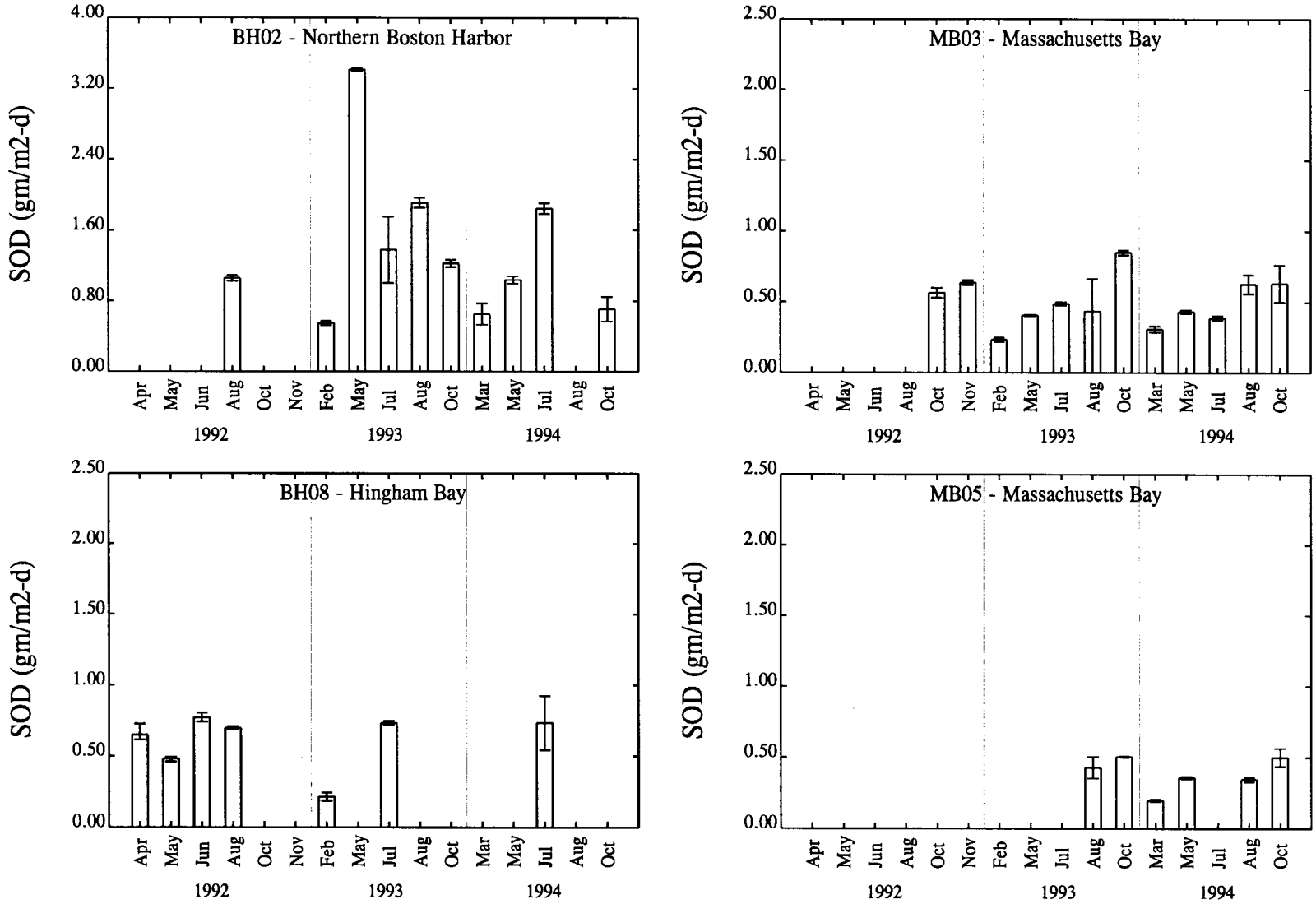


Figure 3-25. SOD Data

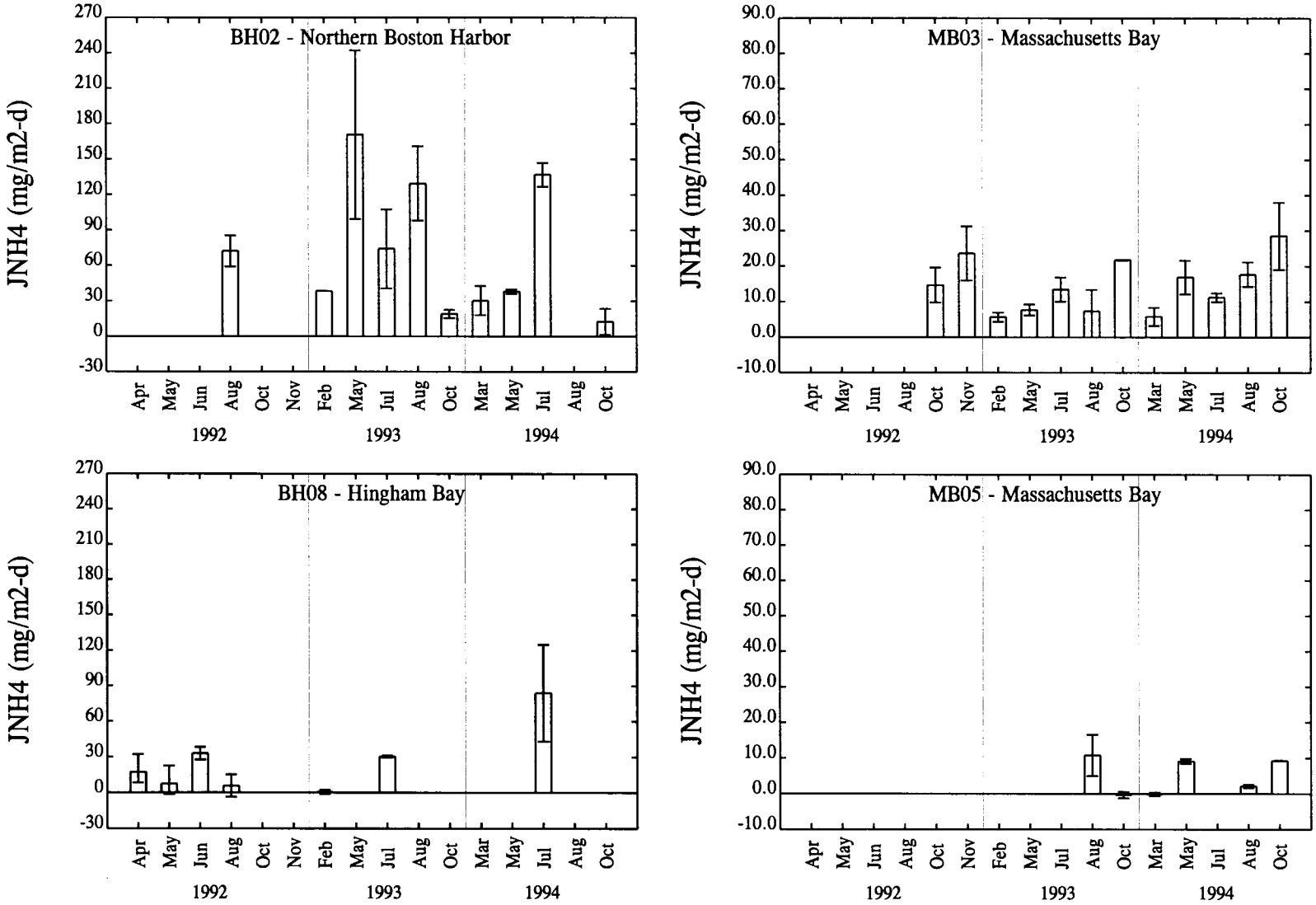


Figure 3-26. Ammonium Flux Data

SECTION 4

MODEL RESULTS

4.1 MODEL INPUTS

4.1.1 Hydrodynamics

A key factor affecting water quality conditions in Massachusetts and Cape Cod Bays is the time-variable circulation and vertical density structure of the bays. These transport processes determine the movement of particulate and dissolved constituents in the bays. In order to simulate these transport processes, a variant of the Blumberg-Mellor (1987) hydrodynamic model was developed and calibrated by the USGS for 1992 (Signell et. al., 1996) and extended to include 1993 and 1994 (Signell, pers. comm.). This time-variable three-dimensional hydrodynamic model is coupled with the water quality model in order to perform mass balance calculations and water quality simulations in the bays. The hydrodynamic model provides all of the transport information to the water quality model as well as temperature and salinity information. The success of the water quality model calibration relies heavily on the success of the hydrodynamic model calibration. The hydrodynamic model grid is presented in Figure 4-1.

In order to reduce the computational burden faced by the water quality model if it were to run on the same grid as the hydrodynamic model, it was decided to use spatial aggregation. Once the hydrodynamic model was calibrated, the transport, temperature and salinity information was spatially aggregated to the water quality grid by combining a 3x3 grid of horizontal hydrodynamic cells into one water quality segment. However, to maintain spatial integrity in the region near Race Point on the northern end of Cape Cod, only a 2x2 aggregation was used. The resulting water quality grid is presented in Figure 4-2. In addition to horizontal aggregation, a partial vertical aggregation was used wherein the top three sigma levels were combined into one water quality segment equal to the depth of each of the bottom nine segments.

The grid shown in Figure 4-2 was used in the original 1992 calibration effort. However, as reported in the HydroQual and Normandeau (1995) model documentation report, this grid was not sufficient to resolve the plume dynamics of the future outfall and a more detailed grid was

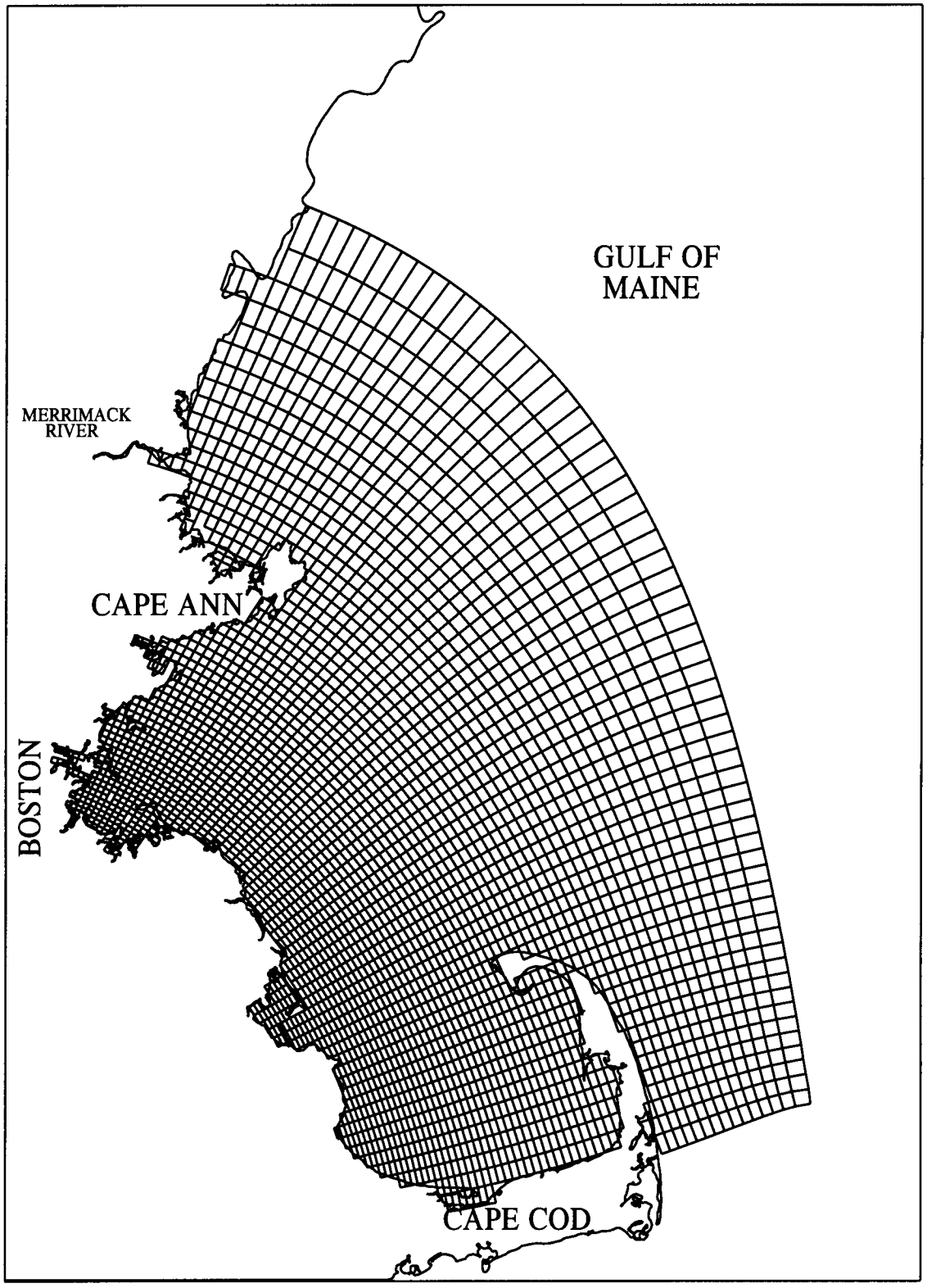


Figure 4-1. Hydrodynamic Model Grid

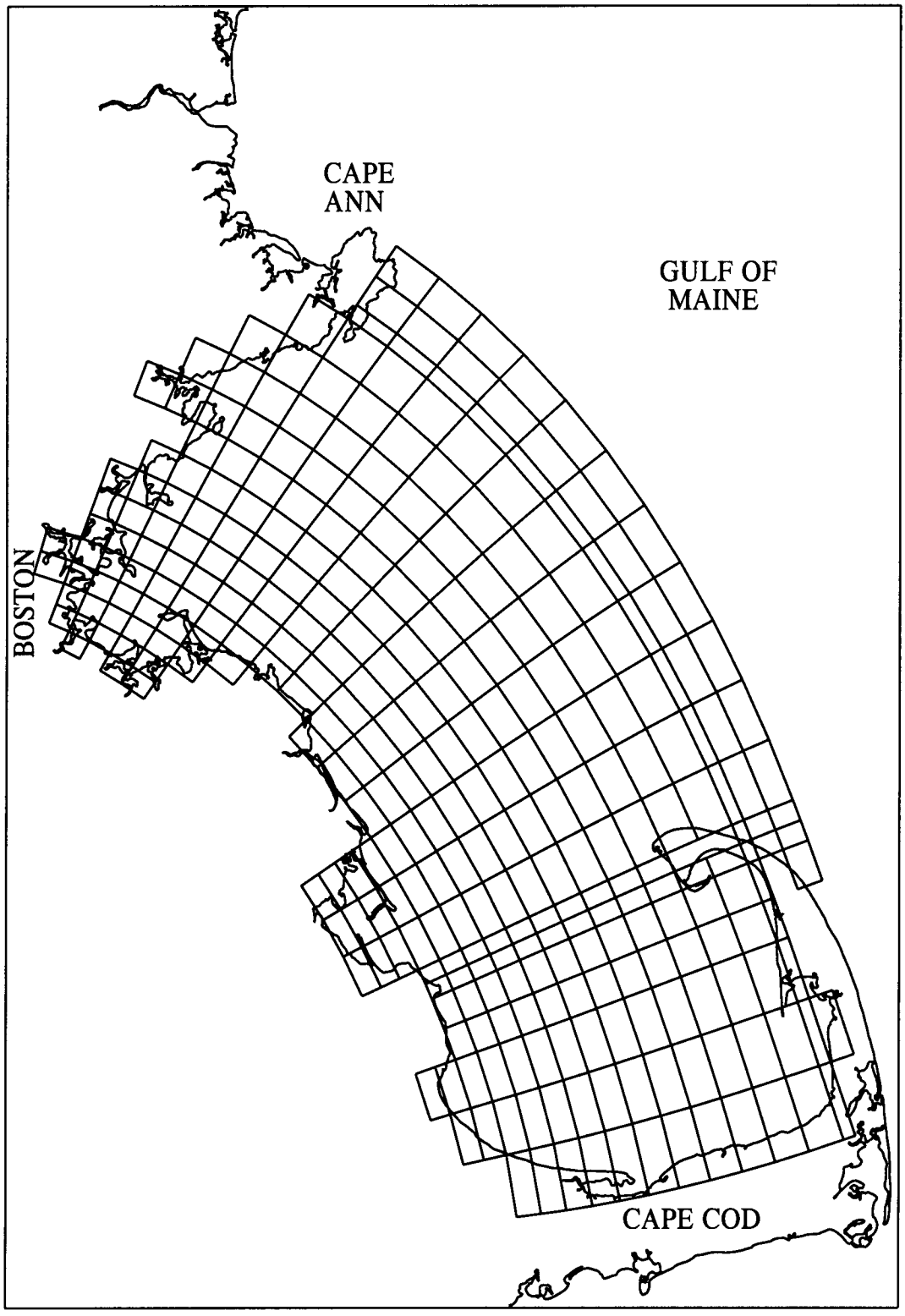


Figure 4-2. Water Quality Model Grid for Massachusetts Bay

used for projections. However, since projections are not being evaluated as part of this analysis, the original calibration grid (Figure 4-2) was used in this study.

4.1.2 Boundary Conditions

Massachusetts and Cape Cod Bays have a large open boundary connection to the Gulf of Maine and the Atlantic Ocean. As a consequence, waters from the Gulf of Maine play a major role in the water quality conditions within the bays. A required model input is the specification of the temporal and spatial distribution, along the boundary, of all water quality state-variables being simulated in the model. However, as was noted in Section 3, there were limited data with which to assign boundary conditions. Data from stations F04, F08, F12, F21 and F22 were used to guide 1992 and 1993, while data from stations F26, F27, F28, and F29 were used to guide 1994. To some degree the determination of model boundary conditions were also guided by comparisons to interior data stations. In addition, where appropriate, an attempt was made to keep the year to year variations of the boundary conditions at a minimum. Figures 4-3 through 4-6 present the surface and bottom boundary conditions used for the model calibration. The thicker solid line represents 1992 boundary conditions, the thinner solid line represents the 1993 boundary conditions and the dashed line represents the 1994 boundary conditions.

The major differences in the boundary conditions from year to year are associated with the temperature, salinity, dissolved silica and DO. The temperature and salinity boundary conditions come directly from the hydrodynamic model. Silica boundary conditions varied the most between years during the spring, perhaps in response to the timing, magnitude, and duration of the spring freshet. Dissolved oxygen boundary conditions between the years varied seasonally, perhaps in response to changes in water column temperature or phytoplankton blooms in the Gulf of Maine. The lowest bottom water values for DO were specified in 1994, and appear to be related to the intrusion of Gulf of Maine water as suggested by bottom water temperature and salinity.

4.1.3 Time Variable Inputs

Figure 4-7 presents the wind, solar irradiance, and fraction of daylight information supplied to the model. The wind data came from the Boston buoy, which is operated by the USGS, and is used to determine the reaeration coefficient. Solar irradiance data came from the Woods Hole Oceanographic Institute's weather tower. Solar irradiance affects phytoplankton growth. The daily fractions of daylight which depends on the latitude of the location of interest

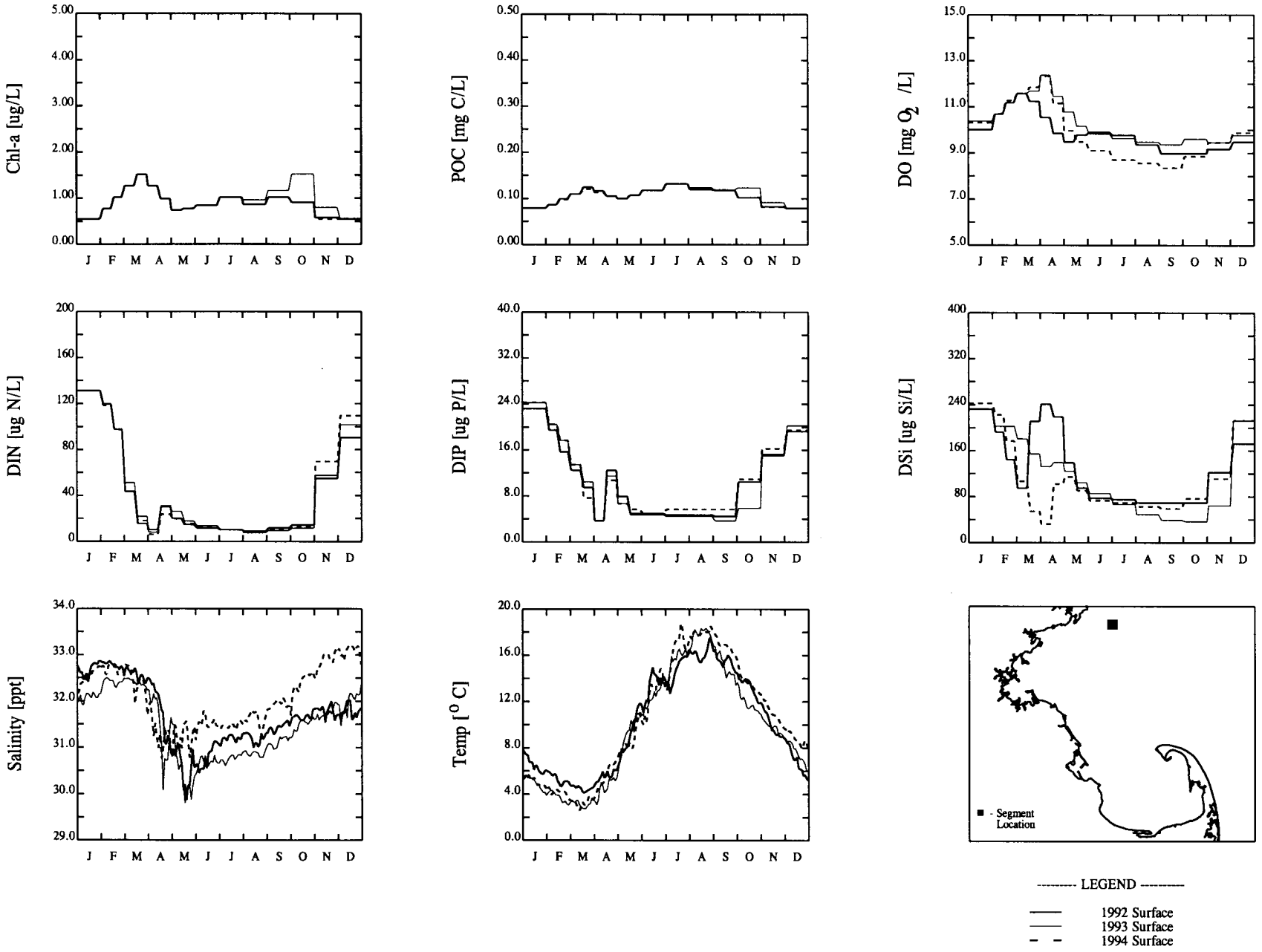


Figure 4-3. Model Surface Boundary Conditions

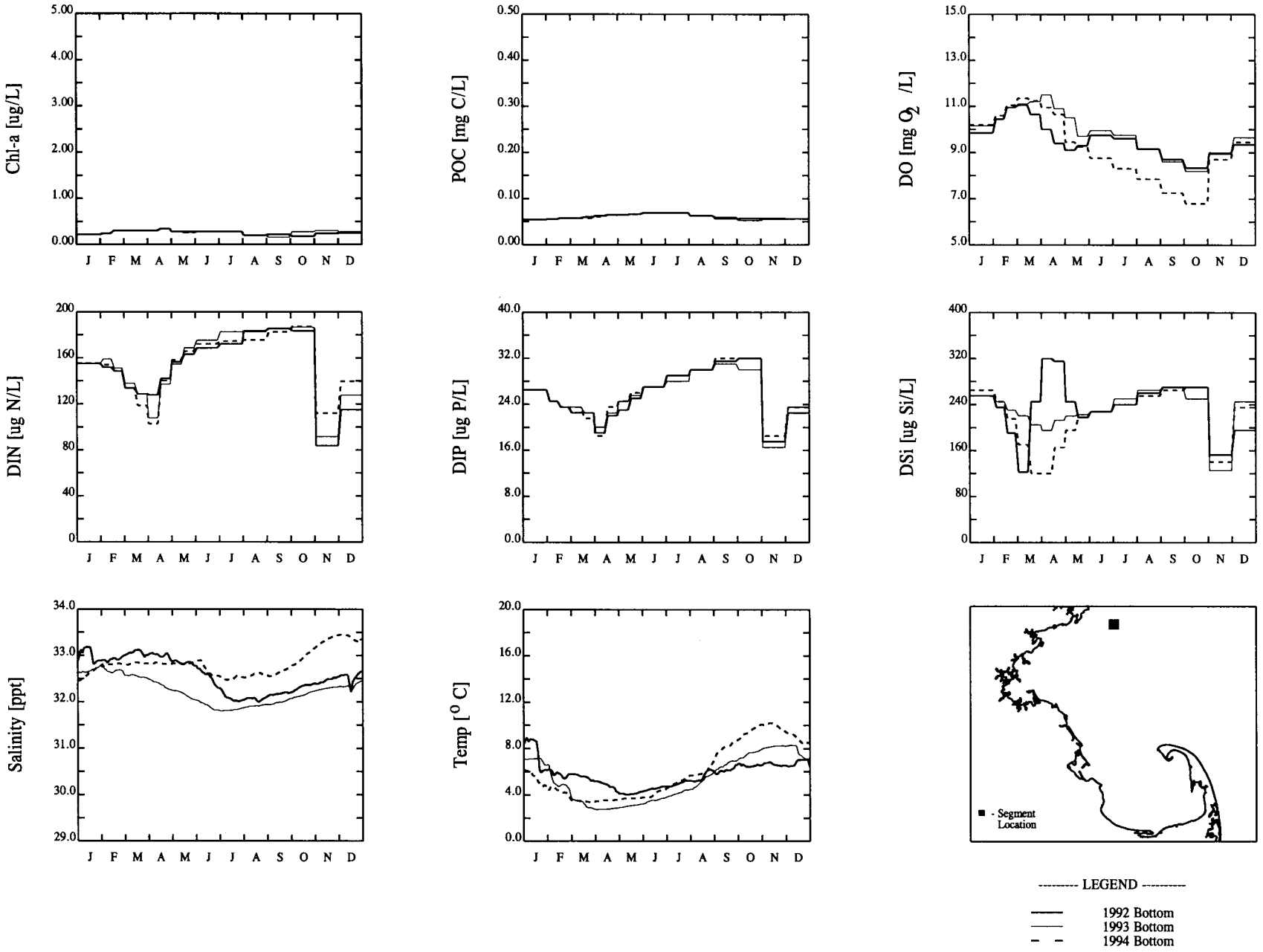


Figure 4-4. Model Bottom Boundary Conditions

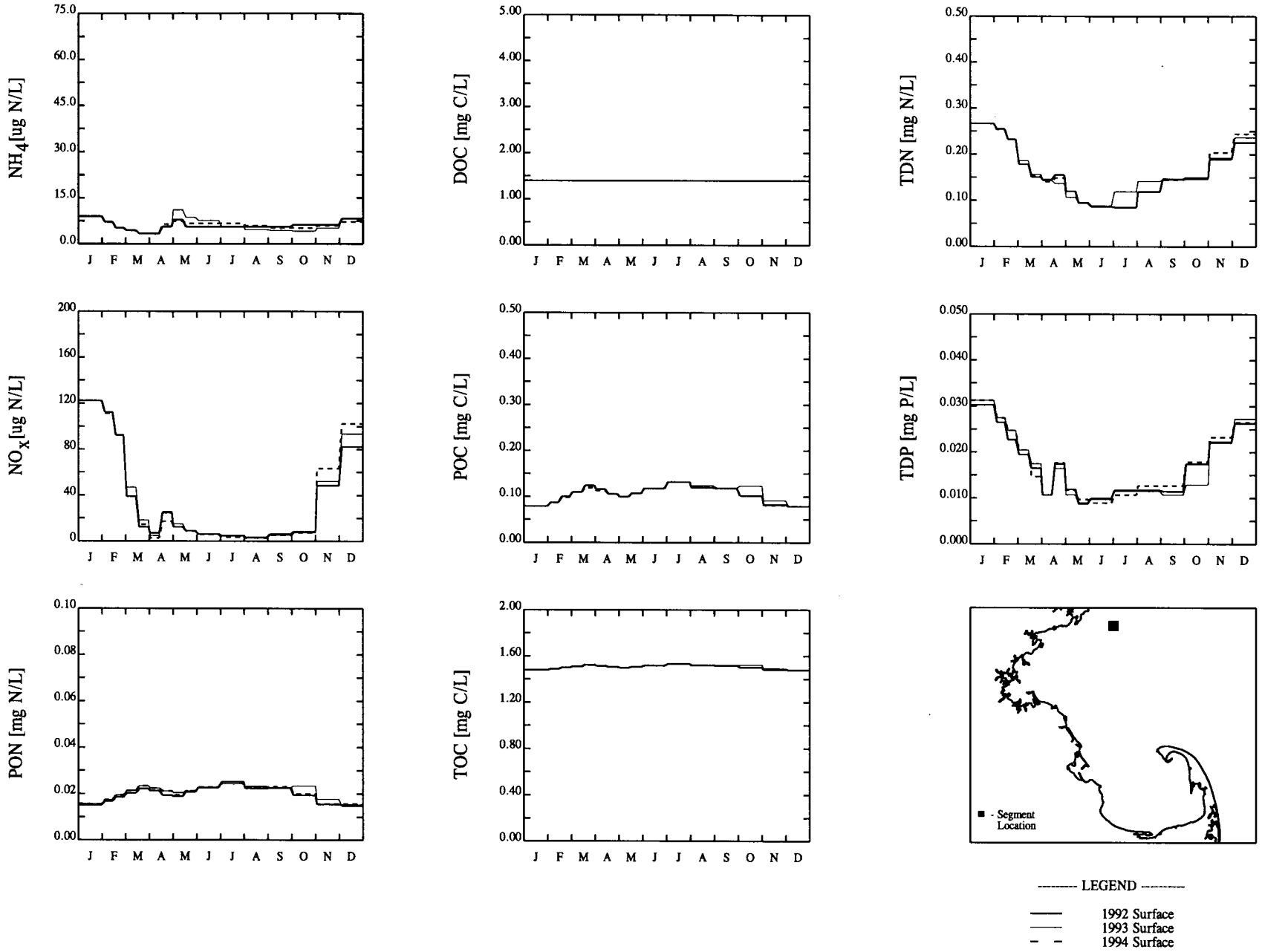


Figure 4-5. Model Surface Boundary Conditions Cont'd

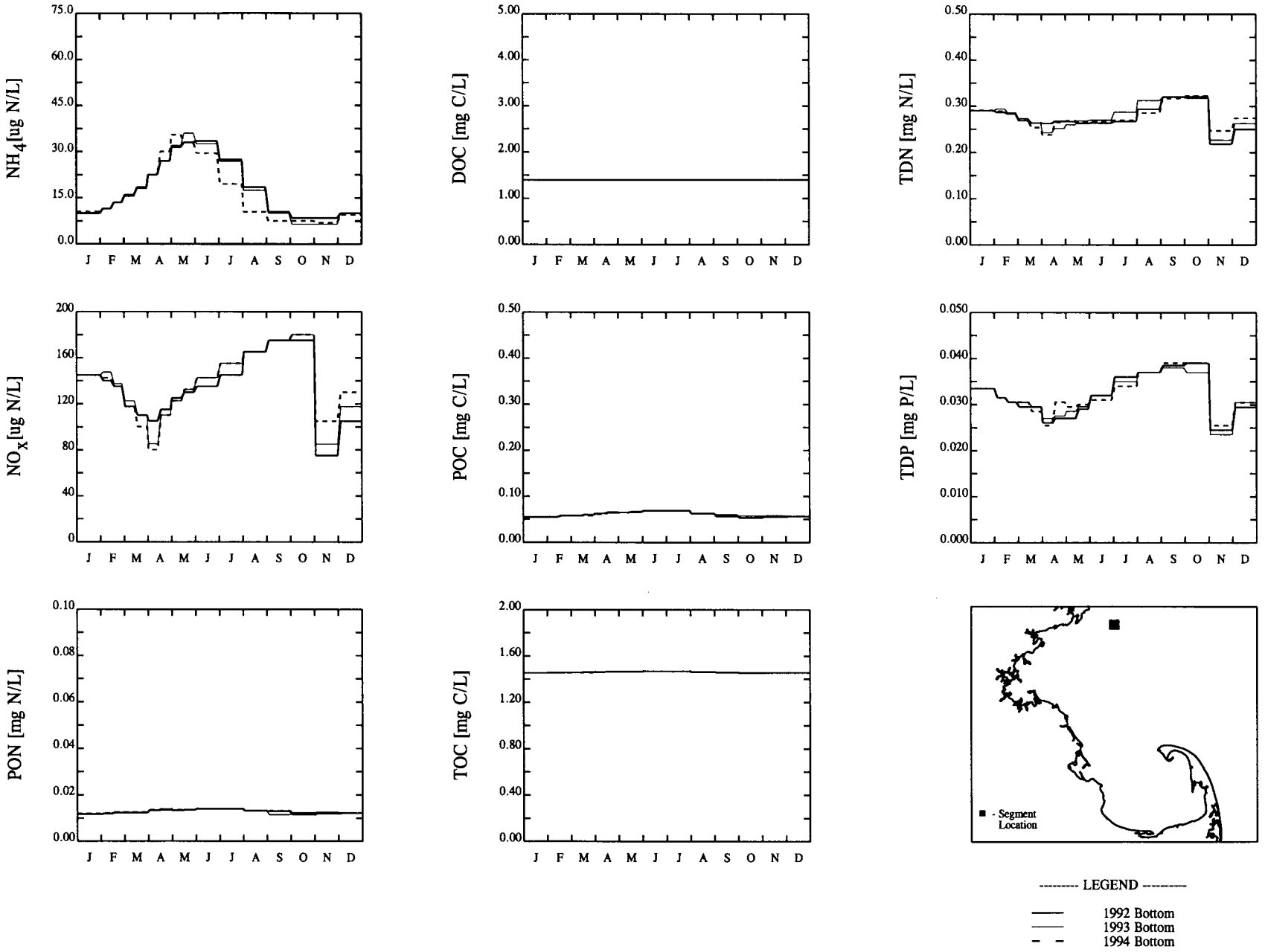


Figure 4-6. Model Bottom Boundary Conditions Cont'd

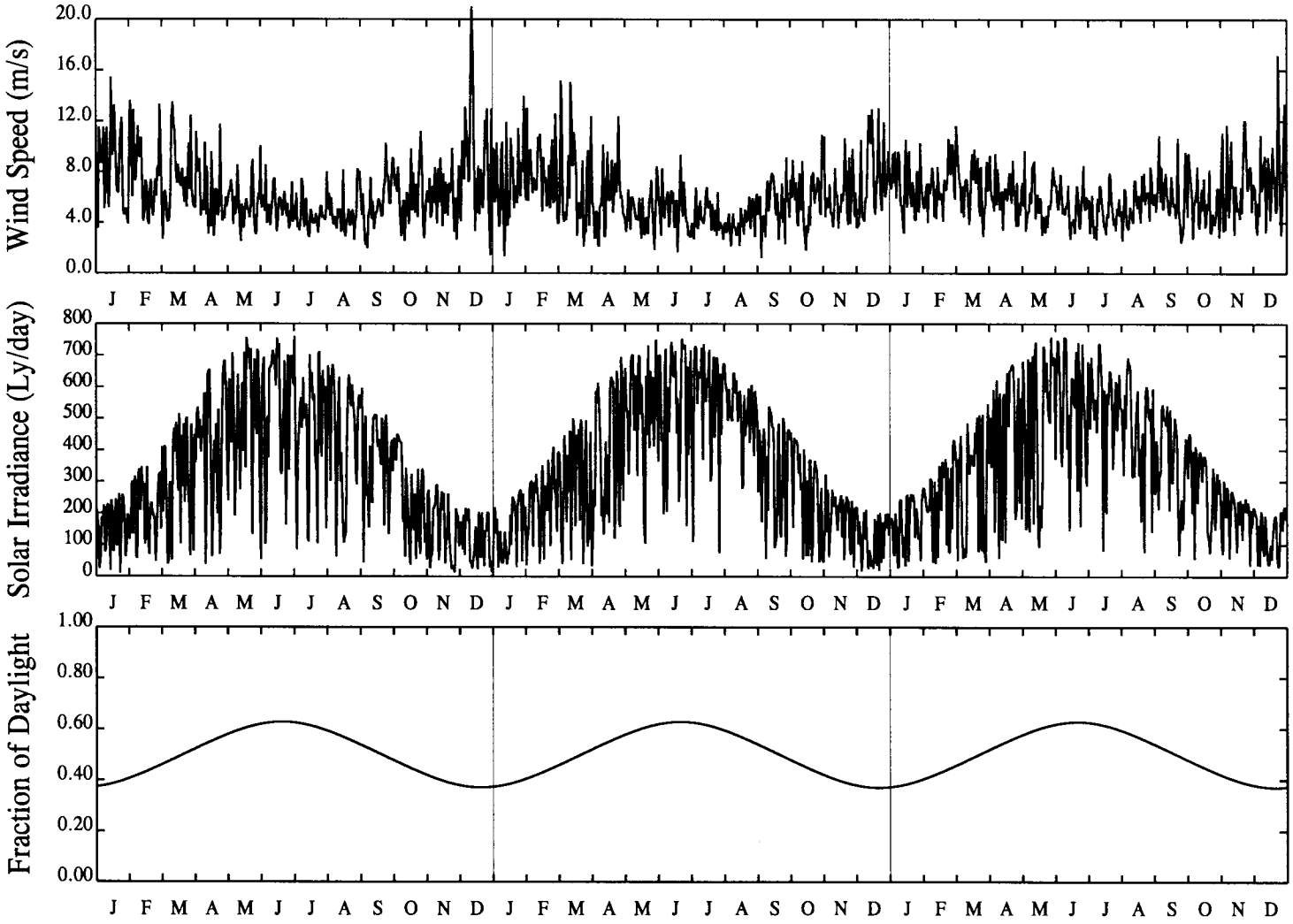


Figure 4-7. Time Variable Inputs

and the declination of the sun as a function of the time of year, were calculated using a method developed by Duffie and Beckman (1970).

4.2 MODEL VERSUS DATA RESULTS

The original BEM was calibrated to data collected from October 1989 through April 1991 and January 1992 through December 1992. The model's ability to reproduce year to year variation was to be tested as additional years of data became available. As data from 1993 and in particular the 1994 boundary data became available, the influence of the Gulf of Maine boundary on water quality in Massachusetts and Cape Cod Bays became more apparent. Subsequently, the model was recalibrated to all the data from 1992 through 1994. The following section compares model results to the observed water quality data.

The BEM is a simplification of the actual Massachusetts and Cape Cod Bays system. When comparing the model results to data it is important to understand the complexities of the system being modeled and the inherent variability in the measurements of the concentrations of water quality variables. The variability may be due to natural processes, e.g., algal patchiness or the effects of local cloud cover or wind mixing on local phytoplankton primary productivity. There is also spatial variability introduced when observed data from one or two randomly selected sampling locations are compared to the output of a completely mixed model segment. Finally, variability may also be due to measurement imprecision or measurement error. Therefore, given this variability it is unrealistic to expect a model to exactly reproduce all of the observed water quality data. However, the model should reproduce the seasonal and spatial trends in the data, the interrelationships between variables and some of the year to year variations in the observed data.

The model versus data comparisons will be presented in three ways. First, a series of nine-panel figures will be presented at a specific location for all three years in order to present the interrelations between the model variables. Second a series of six-panel figures for a particular variable will be presented to demonstrate the model's ability to reproduce spatial variability. Finally a series of plots showing probability distributions will be presented to evaluate the model's ability to reproduce seasonal trends.

4.2.1 Time Series Comparisons

Time-series comparisons of model versus data for 1992, 1993, and 1994 will be presented here. The water quality parameters included are phytoplankton chlorophyll-a, POC, DO, DIN, PO_4 or DIP, and DSi. Model versus data comparisons are also presented for salinity and temperature, to illustrate the calibration status of the hydrodynamic model. The figures will present model versus data comparisons for surface (open circles) and bottom water (closed circles) data and the surface (solid line) and bottom (dashed line) layer computations from the water quality model. When there was more than one sampling station within a water quality grid cell or if multiple samples were taken on a given day the data were averaged and mean and standard deviation are presented. The annual cycle of DO saturation for the surface layer is also included on these figures for reference purposes. The model results represent a two day average of the model computations. The model versus data comparisons for the nutrients are displayed on a log scale to better discern which nutrient is limiting during different parts of the year. The HOM stations chosen for comparison are N16, N17, and N21 all of which lie within the spatial bounds of model segment (11,18).

4.2.1.1 1992 Results

Figure 4-8 presents model versus data results for the 1992 calibration period. There is a good comparison between the hydrodynamic model results and the observed temperature and salinity data. The model predicts the annual cycle of water column temperatures and the timing of temperature and salinity stratification and fall turnover, as well as the timing and magnitude of the effects of the spring freshet on salinity.

The water quality model also appears to be fairly well calibrated. The seasonal trends observed in the water quality data are generally reproduced by the model. In the upper left hand corner of Figure 4-8, the surface chlorophyll-a and fluorescence data show higher concentrations in late February, late June/early July and mid-October suggesting the occurrence of short-term algal blooms. The ranges of the chlorophyll-a and fluorescence data during the year at this station are from approximately 0.1 to 5.6 µg/L. The model does not fully reproduce the observed chlorophyll and fluorescence data, but does predict the annual range of the data. The high frequency variations in the model computed chlorophyll are due to highly variable patterns in wind speed and direction which, if the winds are blowing offshore for a prolonged period of time, advect algal biomass and nutrients from Boston Harbor out to this location.

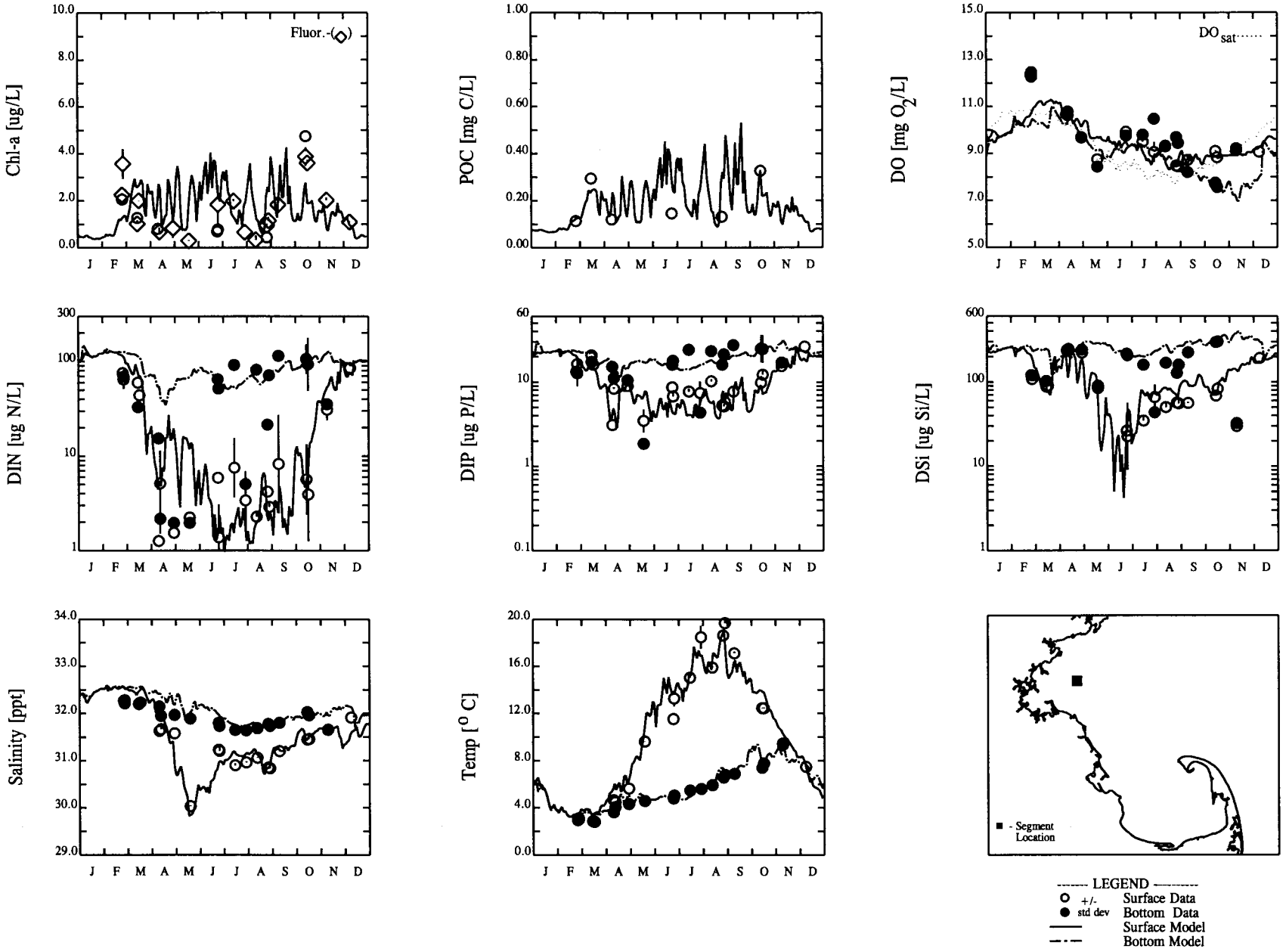


Figure 4-8. 1992 Temporal Calibration Results for Grid Cell (11,18) vs. Data Station N16P, N17, N21

The POC data are relatively sparse, taken only during biological/productivity surveys, which occur only six times a year. The data appear to indicate small algal blooms in March and October which correspond well to the chlorophyll-a and fluorescence data. On a point by point basis the model matches the data very well, missing only the June observation.

The DIN data follow a seasonal cycle influenced by density stratification and algal growth. In early March, the concentrations of DIN are relatively high and the water column appears well mixed as shown by the similarity between the surface and bottom water DIN concentrations. Subsequently there is a rapid decrease in the DIN concentrations in late March as the phytoplankton populations increase. Kelly (1993) reported the surface concentration of NO_2+NO_3 as being 0.0 µg N/L (or below the detection limit of 0.28 µg N/L) in late March through June. However, it is suspect that the NO_2+NO_3 concentrations were actually 0.0 µg N/L for this long a period of time. Therefore, in computing the "observed" concentrations of DIN, we assumed a minimum value of 0.42 µg N/L for any value of NH_4 reported as zero and a value of 0.28 µg N/L for any value of NO_2+NO_3 reported as zero. The surface DIN concentrations remain low, less than 10 µg N/L, through the summer and early fall and then increase as the water column turns over in November. During the summer months there is strong vertical stratification within the DIN data, with surface concentrations generally less than 10 µg N/L and bottom water concentrations ranging from 50 to 120 µg N/L. The model reproduces this feature of the DIN data well. In general, the model reproduces the annual cycle and magnitude of DIN concentrations.

Model versus data results for DIP are presented in the center panel of Figure 4-8. As can be seen, there is a decrease in the surface DIP concentrations in late March/early April as phytoplankton biomass increases. DIP concentrations then increase slightly as the spring freshet begins to advect nutrients into the bay. As spring progresses into summer, the surface and bottom DIP concentrations become stratified with surface DIP concentrations decreasing to between 5 to 8 µg P/L. The model reproduces this feature reasonably well. Both the data and model show that surface DIP concentrations do not reach levels thought to limit algal growth.

The seasonal pattern observed in the DSi data is similar to the other nutrients. Surface and bottom DSi concentrations begin to decrease in February and early March before the arrival of the spring freshet from the Gulf of Maine. Between May and June there is a rapid decrease in surface DSi concentrations followed by a gradual increase to near winter levels. The model follows the timing of the spring decline

and recovery of surface DSi concentrations very well and reproduces the bottom water DSi concentrations reasonably well.

Dissolved oxygen in the water column is strongly influenced by temperature, since temperature affects both the saturation of oxygen in the water column and the rates of microbial activity which can consume dissolved oxygen. Dissolved oxygen concentrations were highest in February, due to a combination of low temperature (i.e., high dissolved oxygen saturation) and primary production resulting from a February algal bloom. Concentrations of DO then decreased rapidly through May. Surface and bottom water DO concentrations increased about 1 mg/L between May and June. Surface waters then generally declined through the summer into fall, while bottom water DO remained slightly elevated before declining markedly beginning in mid-August. There was relatively little vertical stratification in DO until the fall. The model reproduces the general trends observed in the DO data, but does not reproduce the full range of the data. For example the model does not fully reproduce the high February DO concentrations. Also the model only partially reproduces the observed decline through May. While the model reproduces the general trend of surface DO in the summer months, it tends to underestimate bottom water DO in June, July and early August. However, the model does reproduce the lowest observed DO concentrations and the vertical stratification in concentrations that occurred in the fall.

4.2.1.2 1993 Results

Model versus data comparisons for 1993 are presented in Figure 4-9. Again the hydrodynamic model appears to reproduce the salinity and temperature data reasonably well. In particular, the hydrodynamic model reproduces the bottom water salinity very well. The timing and magnitude of change in salinity caused by the spring freshet also appears to be modeled well. However, during the fall, salinity stratification is over predicted by the model. The hydrodynamic model temperature calibration is very good. In general, the timing and magnitude of the temperature changes are reproduced by the model.

The 1993 chlorophyll-a and fluorescence data differ somewhat from the 1992 chlorophyll-a and fluorescence data. The 1993 chlorophyll-a and fluorescence data do not show strong evidence of a late winter bloom nor do they indicate a significant summer bloom. However, spring concentrations were slightly more elevated in 1993 than in 1992. In addition, a large phytoplankton bloom occurred during the fall of 1993 with the highest chlorophyll-a and fluorescence values observed during the 1992-94

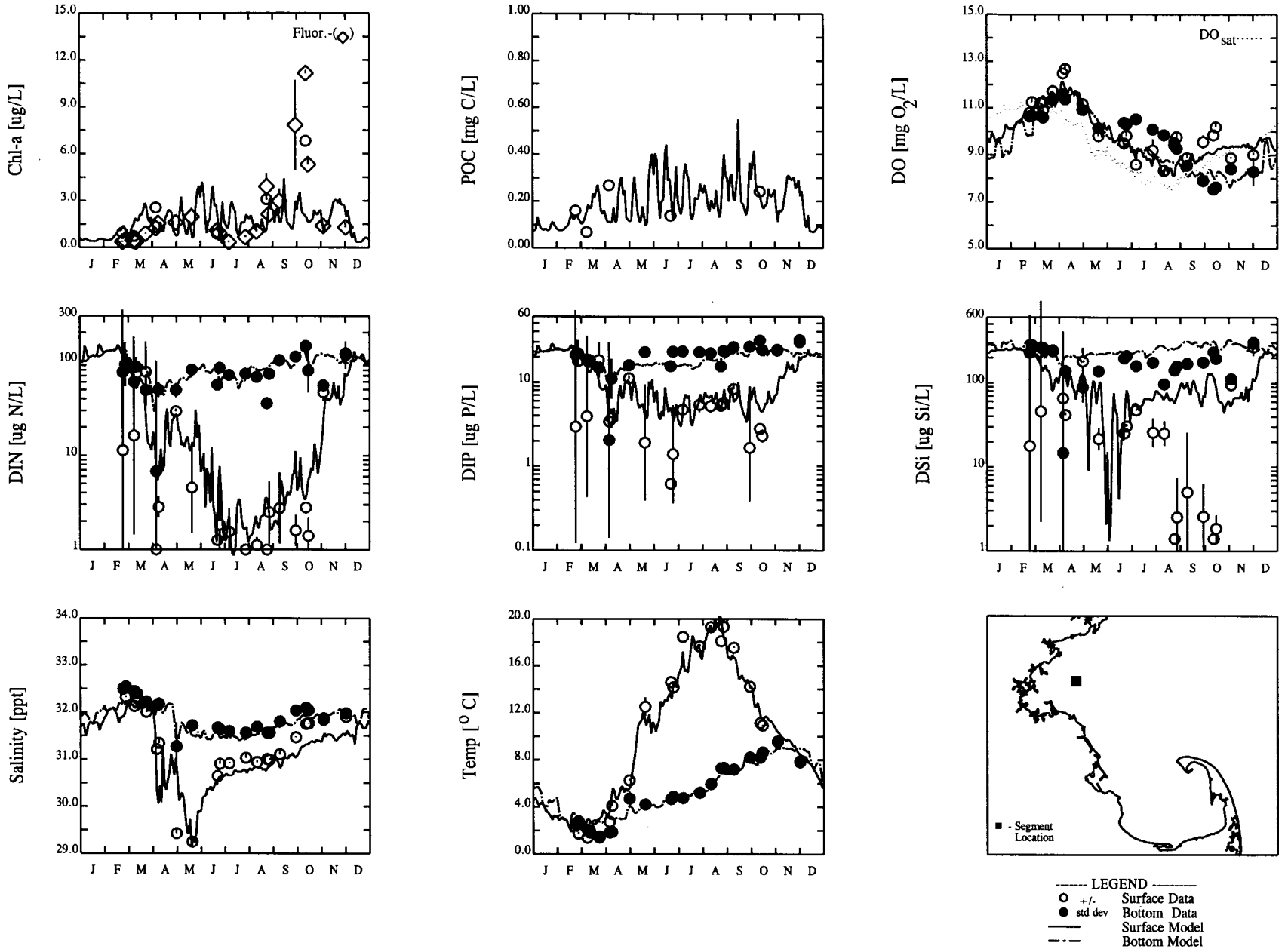


Figure 4-9. 1993 Temporal Calibration Results for Grid Cell (11,18) vs. Data Station N16P, N17, N21

period. In general, the model follows the temporal patterns and magnitude of chlorophyll-a and fluorescence data but fails to reproduce the fall phytoplankton bloom. The failure of the model to reproduce the fall bloom is also evidenced in model versus data comparisons for other state-variables as well, e.g., surface DO, DIN, DIP, and DSi.

In general, the model reproduces the magnitudes of the limited POC data that exists at this location. It is interesting to note that the magnitude of the fall bloom is not as apparent in the POC data. This in part may be due to the fact (as will be discussed later in this section) that the fall bloom had lower carbon to chlorophyll-a ratios than normally observed during the fall in the Massachusetts Bay System.

With the exception of the fall period the model compares very favorably with the DIN data. Bottom water concentrations of DIN began the year at elevated levels before decreasing in March and April. This was followed by a gradual increase through the remainder of the year. The model follows the trend of the bottom data nicely. Surface water concentrations of DIN also began the year at elevated levels before decreasing in March and early April. This was followed by an increase in late April, coincident with the intrusion of the spring freshet from the Gulf of Maine, before decreasing again in July. DIN concentrations remain low (< 3 μg N/L) through mid-October at which time the water column overturns over and surface DIN concentrations once again increase. While the model reproduces the surface DIN data reasonably well, the model does not reproduce the lowest values observed in the late summer and fall. This is due to the fact that the model failed to reproduce the fall algal bloom.

Again with the exception of the fall period, the model compares favorably with the DIP data (Figure 4-9, middle panel). The temporal trend of the bottom DIP is similar to that observed in bottom water DIN data. There is a decrease in bottom water DIP concentrations in March and April followed by a gradual increase into the fall. The model follows this trend, but under predicts the magnitude of the DIP concentrations during the summer. The surface DIP data show a similar decline and recovery in spring concentrations that was observed in the surface DIN data. Low surface DIP concentrations are observed in the fall, when the fall bloom occurred. The model reproduces the surface DIP data very well except for the period of the fall bloom.

The DSi data also follow a seasonal trend similar to the DIP and DIN data. There is a decrease in the bottom water concentrations of DSi in the spring. The surface DSi concentrations decrease in March, again in May, and during the late summer and fall. Very low DSi concentrations were observed

in the late summer/early fall, during the period of high chlorophyll-a concentrations. The model does not reproduce this feature of the data, since the model failed to reproduce the fall bloom.

Model versus data comparisons for DO are presented in the upper right hand panel of Figure 4-9. The model tracks the surface DO data very well missing only the highest concentrations that were observed in April and October. The model matches the bottom DO data well from February to May. However, the model fails to reproduce the elevated levels of DO observed in the bottoms waters from June through August. It is not obvious as to the cause of these elevated concentrations. In September, the model begins to follow the bottom DO data again. The model predicts DO stratification from September through November, consistent with the observed data.

As noted above (and for 1992 - Figure 4-8), the model fails to reproduce the increase in bottom water DO that occurred between May and June. While it is not obvious as to the cause of this phenomenon a few hypotheses can be considered. First, as has been suggested by Cibik et. al. (1998), in their analysis of a similar pattern in bottom water DO observed in 1996, the increase in bottom water DO may occur as a consequence of ventilation via the mixing of the bottom water with oxygenated surface water. However, if this was the case one would expect to see a coincident increase in bottom water temperature, which is not the case. Cibik et. al. (1998) also suggested that advection of oxygen-rich water could be the cause. However, the source of this oxygen-rich water remains to be identified. An additional hypothesis is that the increase in bottom water oxygen may be due to production from benthic diatom communities. These communities could develop in water depths of up to 15-18 meters given observed light extinction coefficients. However, this hypothesis may not be the full answer since the late-spring-summer increase in bottom DO does not occur every year (e.g., 1994 to follow). This feature in the seasonal change in bottom water DO merits further study.

4.2.1.3 1994 Results

Figure 4-10 presents results of the BEM calibration for 1994. Computations from the hydrodynamic model match the temperature and salinity data fairly well. The effects of the spring freshet do not appear to be as strong as those that occurred in 1992 and 1993. In particular, the surface salinity concentrations were generally higher than those observed in 1992 and 1993. In general, the model reproduces the temporal and vertical features of the salinity data, although it does over predict the bottom salinity in November and December. The hydrodynamic model also matches the temperature data very well, although it does over-estimate the bottom temperatures slightly in February and early March. Note, the high bottom water temperatures that are observed and computed in the fall are much higher than were observed or computed in 1992 or 1993.

The 1994 chlorophyll-a and fluorescence data (Figure 4-10, upper left panel) indicate a small winter bloom occurring in late February and a larger fall bloom occurring in late September and October. The model generally reproduces the magnitude of the annual chlorophyll-a and fluorescence data and reproduces most of the features of the fall bloom. The POC data (Figure 4-10, middle panel) do not show much of an annual trend and were fairly narrow in their range of concentration (0.06 - 0.18 mg C/L). While the model computations of POC are consistent with the winter and spring POC data, the model appears to overestimate observed POC during the summer and fall periods.

Model versus data comparisons for DIN are presented in the middle left panel of Figure 4-10. While the model follows the seasonal trends observed in the DIN data, the model does not always reproduce the magnitude of the observed concentrations. For example, during the summer months the model under predicts the bottom water DIN concentrations. The model also over-estimates surface DIN concentrations in March and early April, while under predicting surface DIN concentrations in June and July. It is interesting to note that June and July 1994 surface DIN concentrations are 5 to 10 µg N/L higher than observed in June and July 1992 and 1993. In general though, the model does reproduce the seasonal changes in the vertical stratification of the water column with respect to DIN.

The DIP data observed in 1994 (Figure 4-10 middle panel) are fairly similar to the data observed in 1992 and 1993. The unique feature to the 1994 DIP data set is the marked decrease that occurs in the surface data during late March and early April. The model tends to under

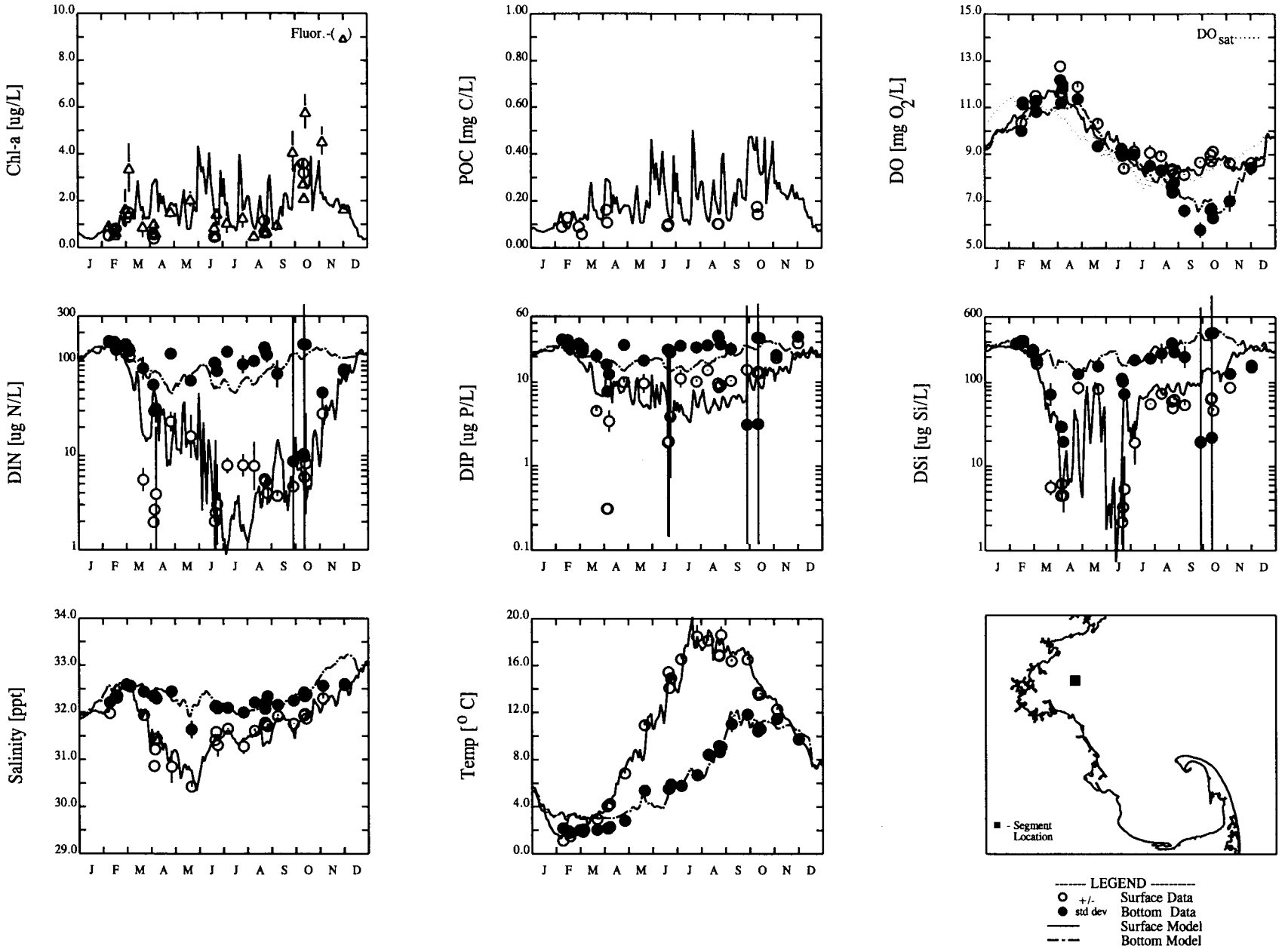


Figure 4-10. 1994 Temporal Calibration Results for Grid Cell (11,18) vs. Data Station N16P, N17, N21

estimate both surface and bottom DIP concentrations year round. However, the model does reproduce the approximate stratification between the surface and bottom data.

The model compares favorably with the 1994 DSi data (Figure 4-10, middle right panel). While the model over estimates the decrease in bottom water DSi concentrations that occurred in March and April, the model does reproduce the remaining surface and bottom data very well. The observed decrease in the surface DSi concentrations in March followed by the increase in April and subsequent decrease in June is reproduced by the model.

The model matches the 1994 DO data at this location reasonably well (Figure 4-10 upper right panel). The model's computation of surface layer DO compares favorably to the data, missing only the highest DO concentrations observed in April. The comparison of the model's computations to the bottom water DO data is also very favorable. The model reproduces most of the seasonal variability observed in the bottom water data, missing only the lowest DO values that were recorded in late September.

4.2.2 Spatial Model versus Data Comparisons

The previous calibration figures presented the calibration at one location to show the interaction between the various state-variables in the model. The following calibration figures will present a single parameter at six locations to show the model's ability to spatially reproduce the water quality conditions throughout the bays. The six locations include: station F23P, located near the mouth of Boston Harbor; station N10P, located just outside of Boston Harbor; station F01P, located in western Cape Cod Bay; station N04P, in northern Massachusetts Bay; stations N06 and N07P, near the future outfall location; and station F08, located in Stellwagen Basin. In 1994, station F08 was discontinued, so station F29 will be presented in its place. Station F29 is located near the tip of Cape Cod. The model computations are presented as before, as two-day averages. Model computations for chlorophyll-a are banded by shading that represents the range of the maximum and minimum concentrations computed by the model during each two-day interval.

4.2.2.1 1992 Results

Figure 4-11 presents model versus data comparisons for chlorophyll-a for the 1992 calibration period. The model reproduces the general spatial pattern observed in the chlorophyll-a data. The

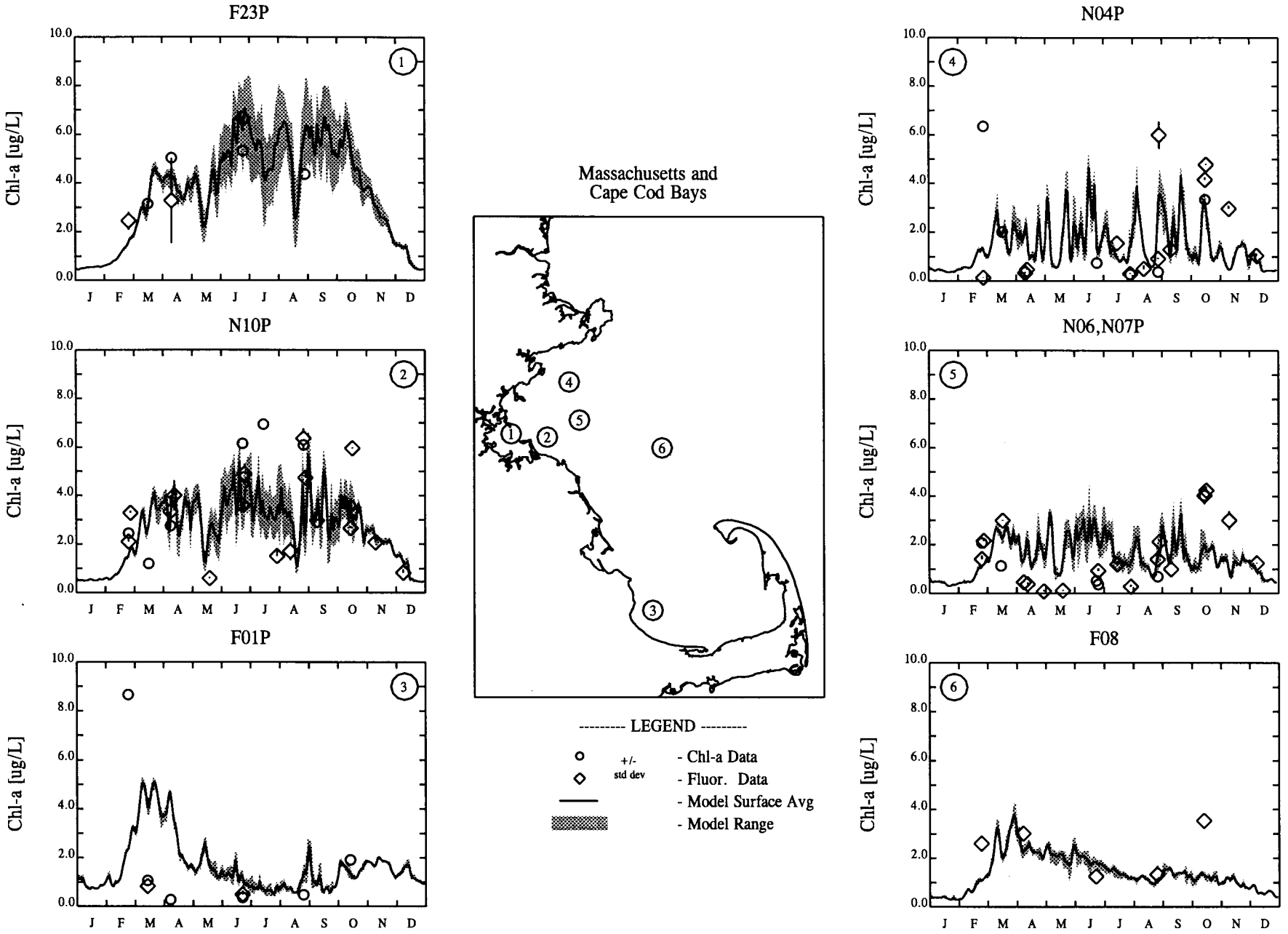


Figure 4-11. 1992 Model vs. Surface Chlorophyll-a Data

data. The highest chlorophyll-a and fluorescence values are observed in the vicinity of Boston Harbor and then decrease with distance from the harbor. The model reproduces this trend. Another spatial feature that can be observed in the chlorophyll-a data is that in Cape Cod Bay maximum chlorophyll concentrations were observed in February and then remained low throughout the rest of the year. This contrasts to the other stations which generally had as high or higher concentrations during the summer and fall months as compared to February. Although the model is approximately a month late in the timing of the bloom, it does reproduce the general temporal trend of the data. Overall the model does a fair job reproducing the magnitude of the observed chlorophyll-a and fluorescence data at each location.

Another feature of the Bays water quality that the model appears to reproduce is the temporal and spatial distribution of DSi (Figure 4-12). A number of researchers (Becker, 1992, Kelly, 1993) have noted that the late winter/early spring diatom bloom virtually depletes silica in Cape Cod Bay. This phenomenon of spring silica depletion is generally not observed in Massachusetts Bay. Figure 4-12 shows that station F01P was virtually depleted of DSi in late March/early April, as opposed to other Massachusetts Bay stations which were observed to have higher DSi concentrations during this period. The model appears to reproduce this spatial feature reasonably well. A reason silica may not be depleted in Massachusetts Bay in the spring may be due to the influx of silica rich waters accompanying the spring freshet.

Figure 4-13 presents a temporal model versus data comparison for 1992 surface and bottom DIN. In general, the DIN concentrations were highest during late winter and late fall. During the spring there was a decrease in the surface and bottom DIN concentrations with the surface DIN concentrations declining more significantly than the bottom DIN concentrations. Stratification between the surface and bottom DIN concentrations generally lasted from March to November. The model computed the highest DIN concentrations near Boston Harbor, generally consistent with the observations. The model computes lower DIN concentrations at station N10P and reproduces the data fairly well. A rapid decrease was observed in DIN concentrations in February/March and the model reproduces this feature. The model also reproduces the low surface DIN concentrations that were observed through October. The two northern Massachusetts Bay sites N04P and N06, N07P had similar DIN concentrations during 1992. The model reproduces the low surface DIN concentrations and somewhat higher bottom DIN concentrations observed at these two sites. In Cape Cod Bay (Station F08), the model reproduces the generally low DIN concentrations that were observed in 1992.

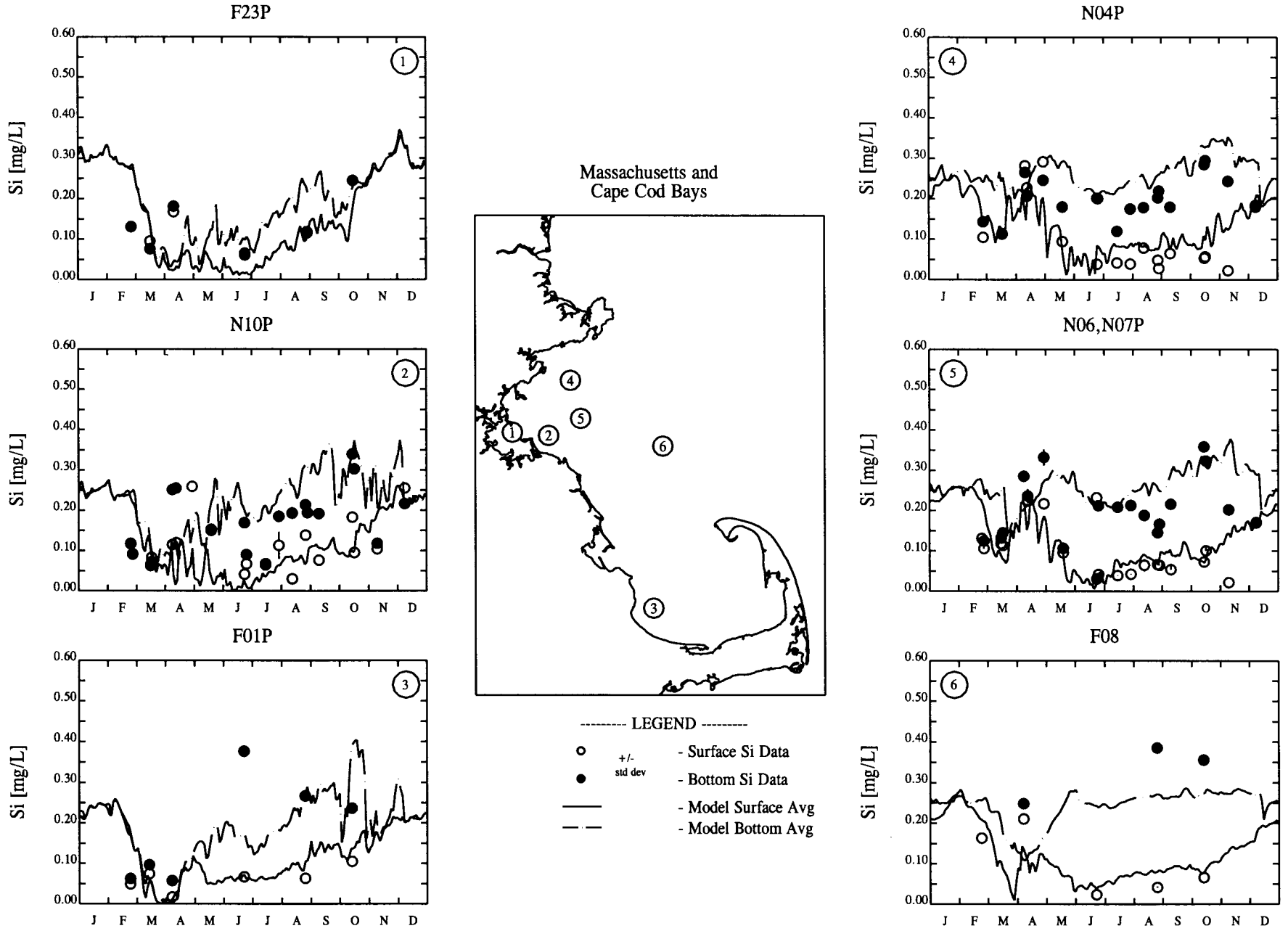


Figure 4-12. 1992 Model vs. Surface and Bottom DSi Data

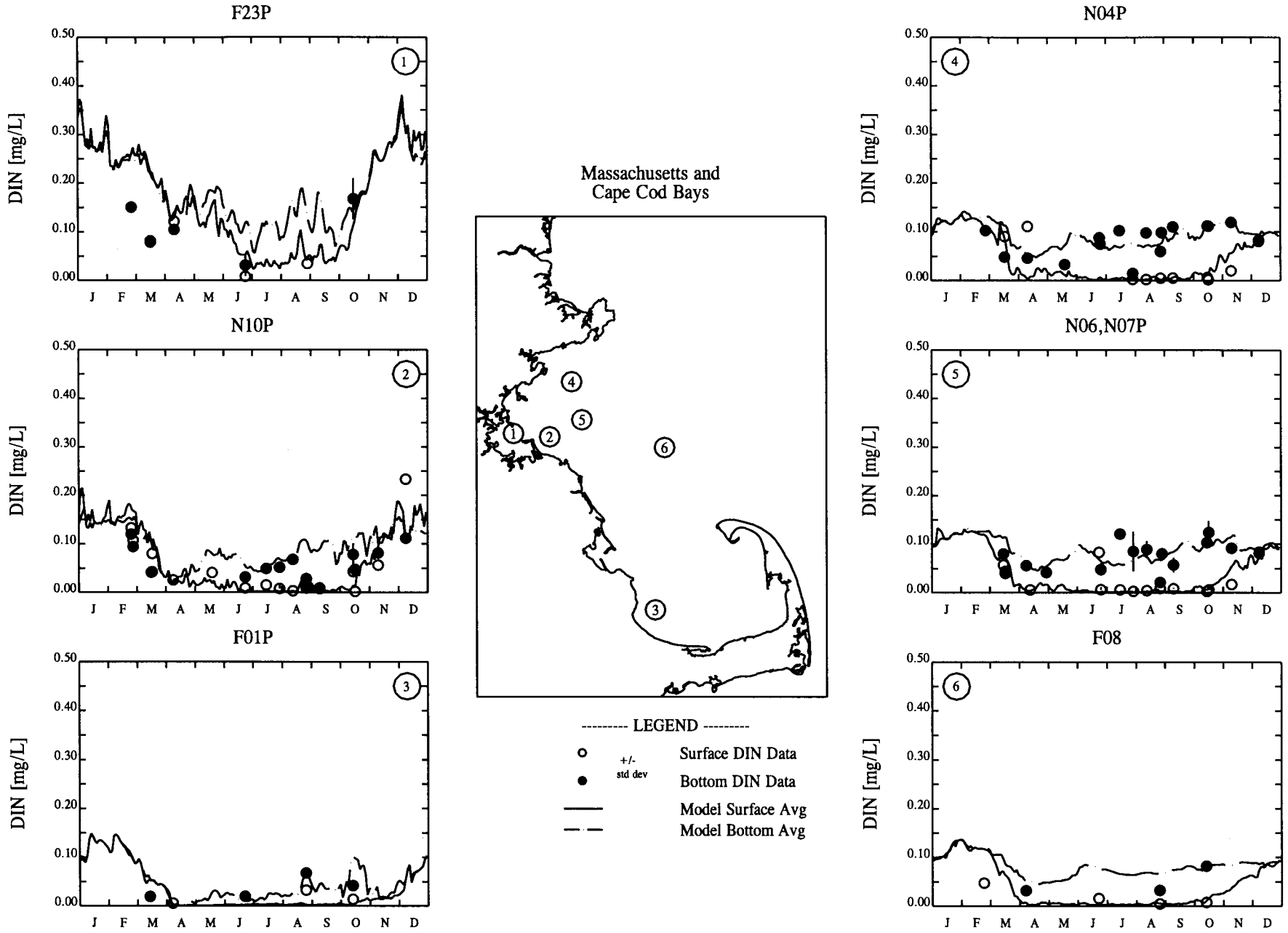


Figure 4-13. 1992 Model vs. Surface and Bottom DIN Data

Model versus data comparisons for the water column DIN to DIP ratio are presented in Figure 4-14. The majority of the data indicate that the Massachusetts/Cape Cod Bays system was potentially nitrogen limited during 1992 with DIN to DIP ratios less than 7. The model reproduces this feature while also reproducing the timing and magnitude of changes in the ratio of DIN to DIP throughout the bays. Although the data are few in number, the model and data suggest that the F23P site is not as potentially nitrogen limited as are the other locations in Massachusetts and Cape Cod Bays.

The DIN to DSi model versus data comparison for 1992 is shown in Figure 4-15. From the data it appears that the bays were more potentially silica limited in the spring and late winter (November and December), while for the remainder of the year nitrogen was more potentially limiting. In Boston Harbor, the model over predicts the DIN to DSi ratio for the first half of the year. At the five other sites displayed, the model does a fairly good job matching the data. The are a few occasions where the model computes very low dissolved silica concentrations and consequently predicts very high DIN/DSi. At the easternmost stations, the model does not compute these high DIN to DSi ratios.

Figure 4-16 presents model versus data comparisons for dissolved oxygen for the 1992 calibration year. The model reproduces some of the features observed from the DO data. The model does not fully reproduce the high DO concentrations observed in February which were followed by a rapid decrease in DO into May. This feature in the DO data may be due to the breakup and rapid settling of a bloom of colonial *Phaeocystis* that occurred in March and April of 1992. Occasionally, and at some stations more than others, the model is able to reproduce the fact that during the early to mid-summer bottom water DO levels are higher than surface values. However, the model does not compute bottom water DO as high as are observed in the data. The model also reproduces the low DO levels observed in October and November as well as the degree of DO stratification observed in the fall.

4.2.2.2 1993 Results

Model versus data comparisons for chlorophyll-a and fluorescence 1993 concentrations are presented in Figure 4-17. In general, the model reproduces the magnitude and trends observed in the 1993 chlorophyll-a and fluorescence data. However, with the exception of Boston Harbor, there was a system-wide bloom that occurred in September and October that the model does not reproduce. Further discussion of this mis-calibration will be presented later in this report.

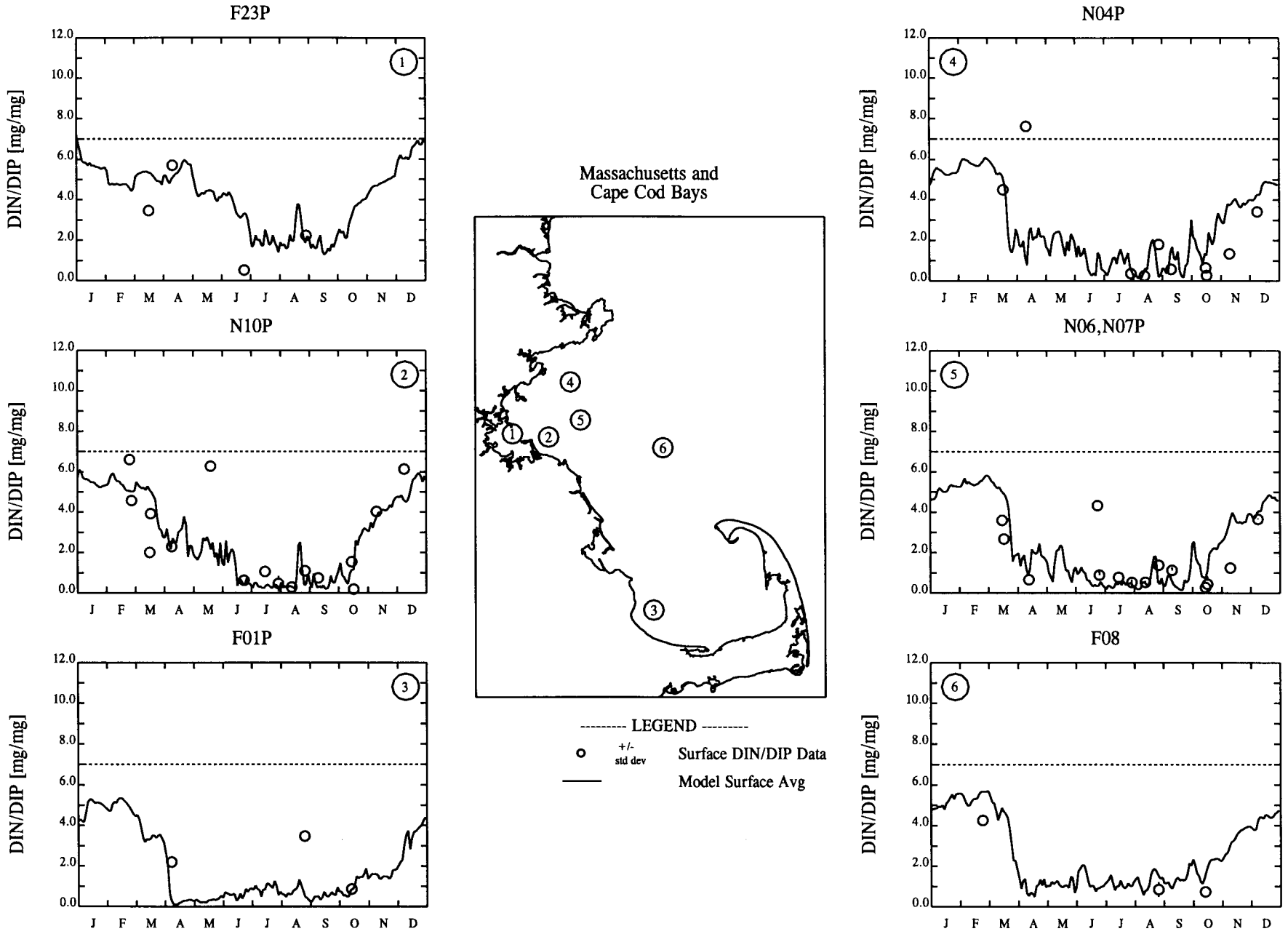


Figure 4-14. 1992 Model vs. Surface DIN/DIP Data

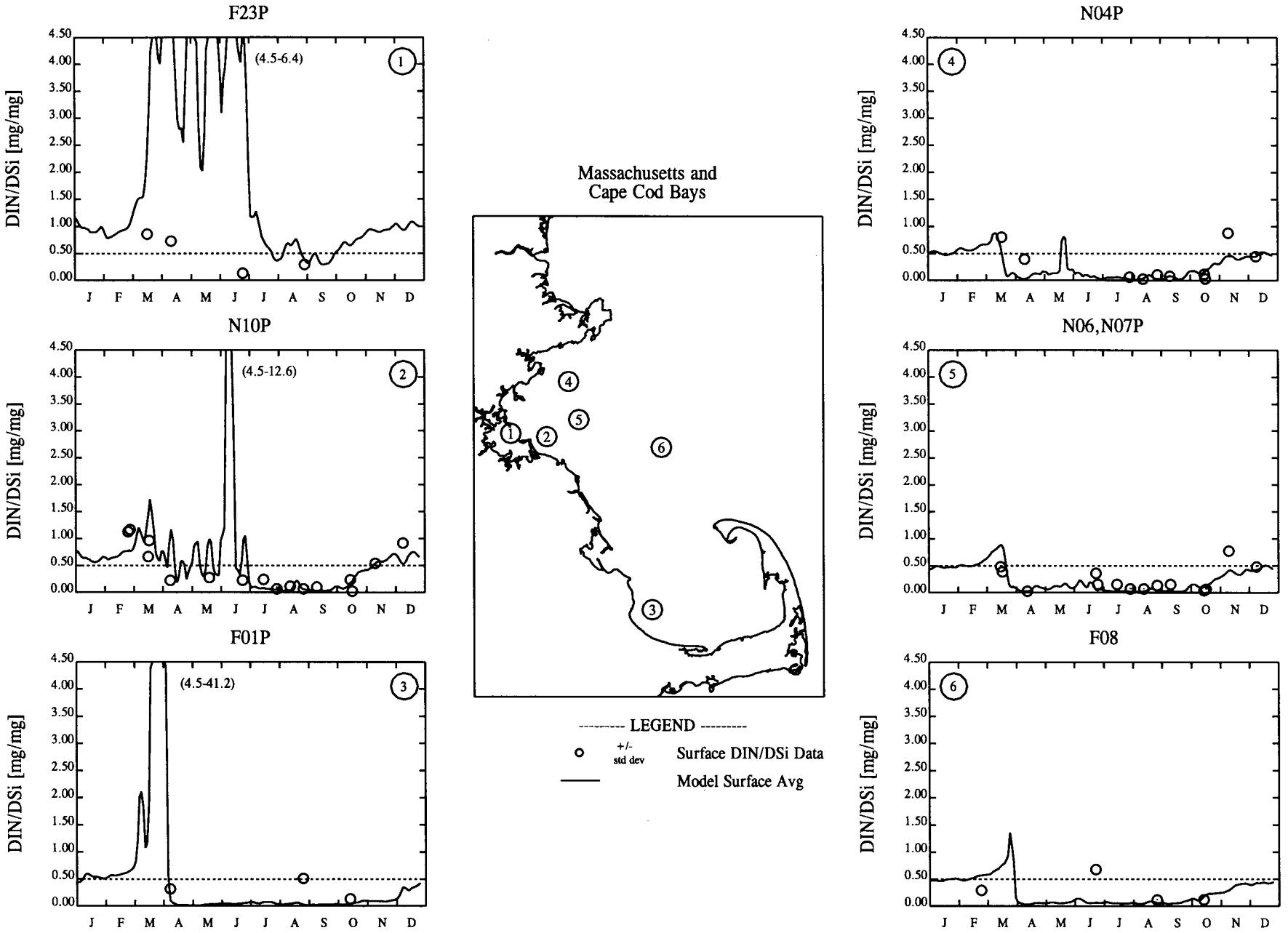


Figure 4-15. 1992 Model vs. Surface DIN/DSi Data

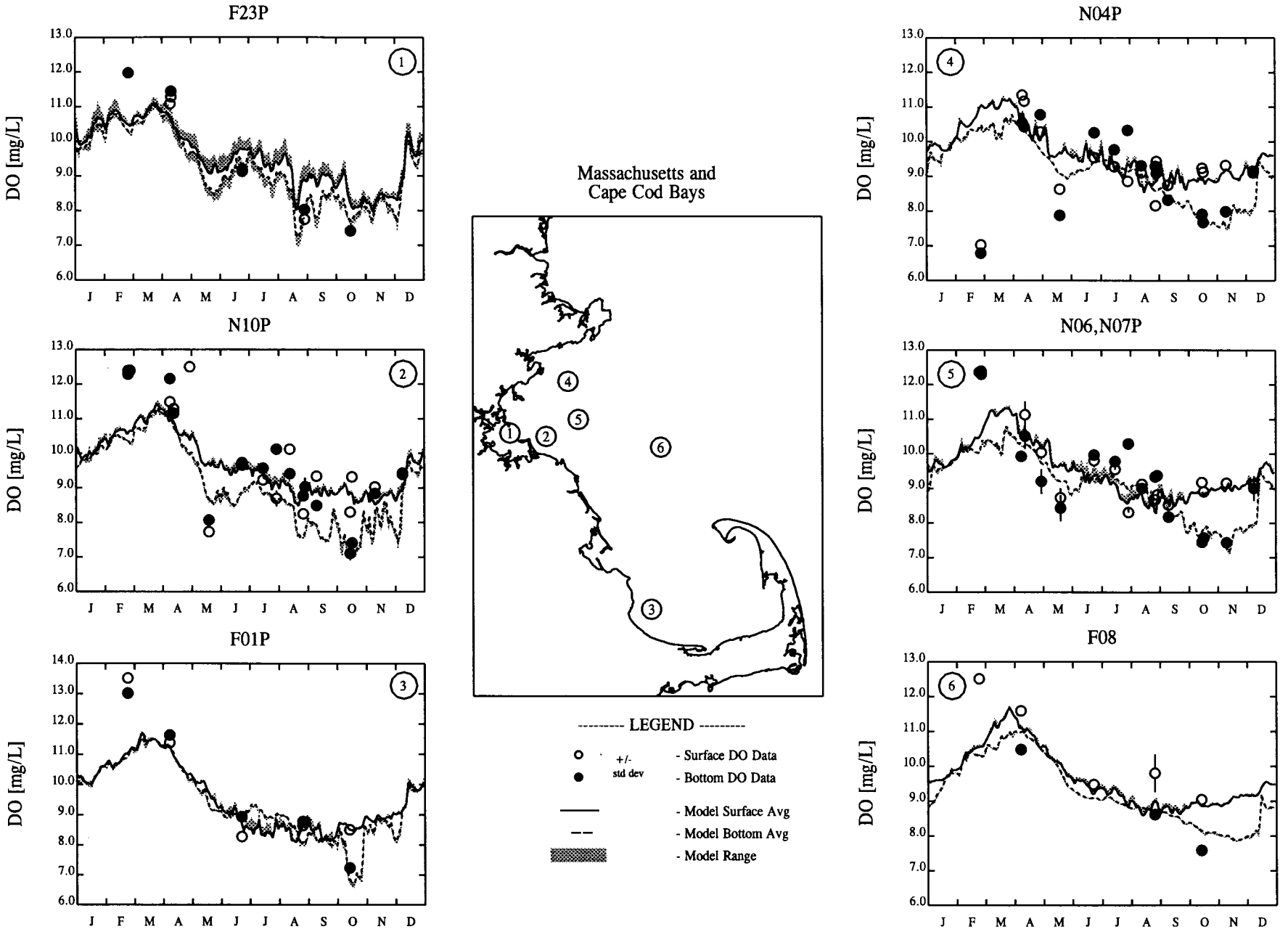


Figure 4-16. 1992 Model vs. Surface and Bottom DO Data

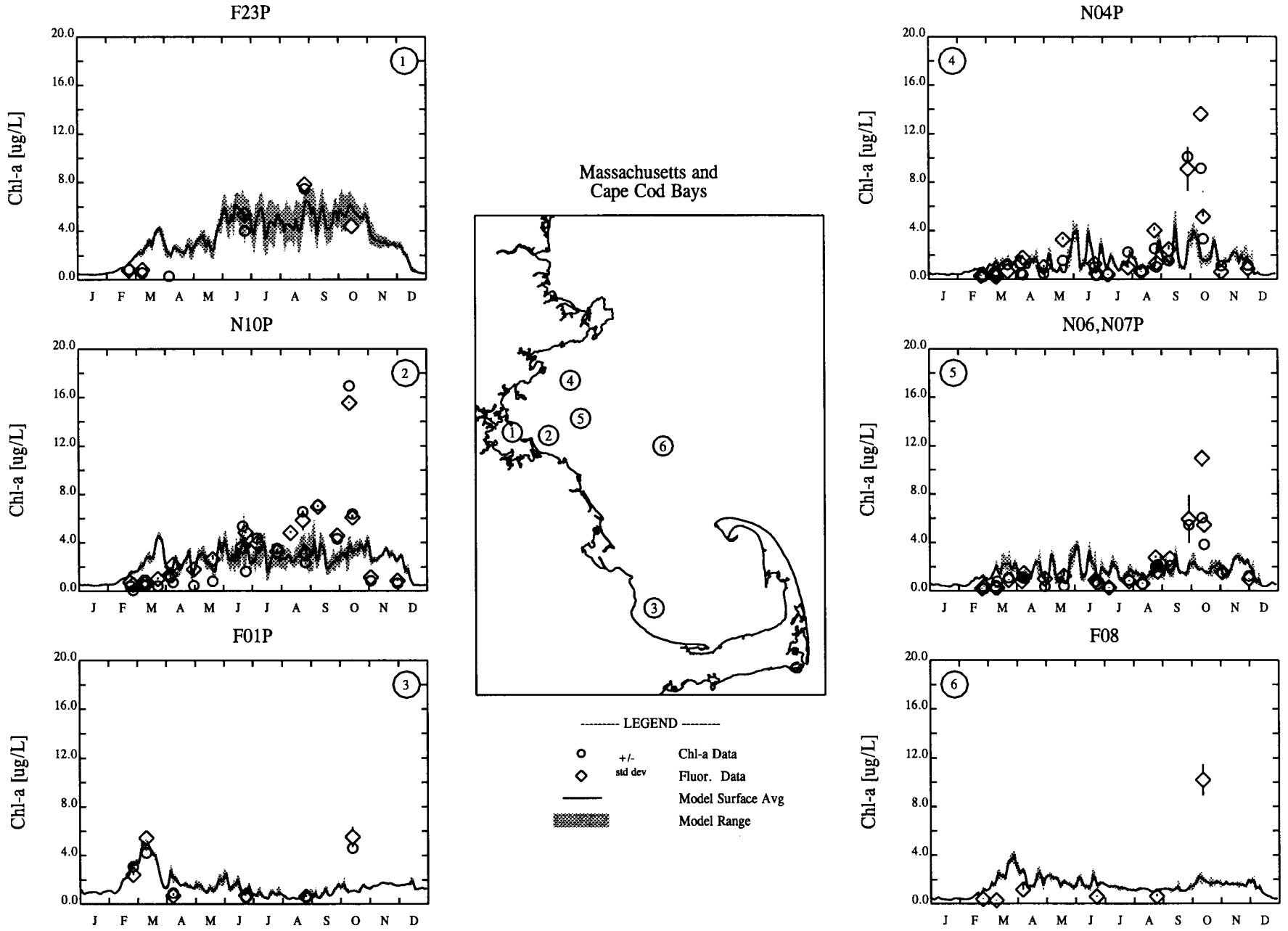


Figure 4-17. 1993 Model vs. Surface Chlorophyll-a Data

Figure 4-18 presents model versus data comparisons for dissolved silica for 1993. Differences in the March and April boundary conditions account for some of the differences observed between the 1992 and 1993 model results. Again, the model computes nearly depleted DSi in Cape Cod Bay during March with somewhat higher DSi concentrations observed throughout the rest of the bays system during March. As has been noted previously, since the model fails to reproduce the fall algal bloom, the model overestimates surface DSi concentrations for all stations presented in Figure 4-18.

Model versus data comparisons for surface and bottom DIN concentrations in 1993 are shown in Figure 4-19. The highest DIN concentrations observed in 1993 were found in the vicinity of Boston Harbor. On a year round basis, higher DIN concentrations were observed in the bottom waters of the deeper, easternmost stations. The model matches both the temporal and spatial patterns, observed in the DIN data, very well. Near Boston Harbor (F23P), the timing of the seasonal decline and recovery of DIN concentrations are reproduced fairly well by the model. At the northern sites in Massachusetts Bay, the model to data comparisons are very favorable. The model reproduces the lower bottom water DIN concentrations observed in the bottom waters of the western station (N10P) and the higher DIN concentrations observed in the bottom waters of the eastern stations (N04P, N06, and N07P). The period of lower surface DIN concentrations is also reproduced by the model.

Figure 4-20 presents the model versus data comparisons for the surface DIN to DIP ratio for 1993. As opposed to 1992, the data indicate that for those stations (F23P and N10P) near Boston Harbor phosphorus was potentially the more limiting nutrient (as compared to DIN) in the spring. However, for the remainder of the year and for the other stations presented the data indicate nitrogen to be the more limiting nutrient. The model reproduces most of DIN/DIP data fairly well, but misses the higher ratios observed in the spring for the stations (F23P and N10P) near Boston Harbor. The model also under predicts DIN to DIP ratios observed in the latter part of 1993 for station N10P. However, in general, the model reproduces the observed trends in the DIN to DIP ratios.

The 1993 model versus data comparisons for surface DIN to DSi ratios are presented in Figure 4-21. The DIN to DSi ratio data indicate that during the winter and early spring either DIN or DSi could have been the potentially limiting nutrient. Beginning in late spring and into the summer the DIN to DSi ratios suggest DIN to be the more potentially limiting nutrient. In the fall, when the fall

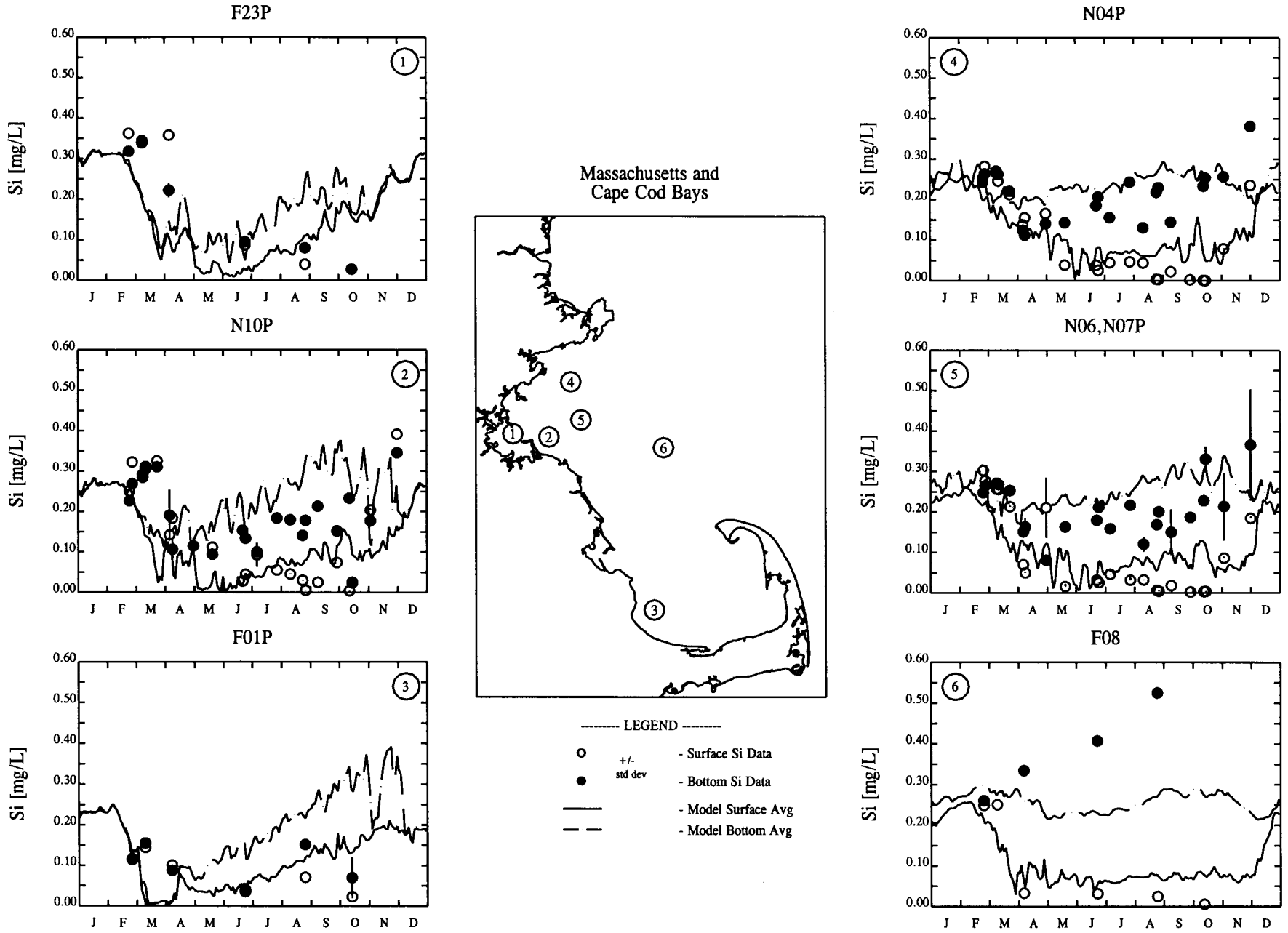


Figure 4-18. 1993 Model vs. Surface and Bottom DSi Data

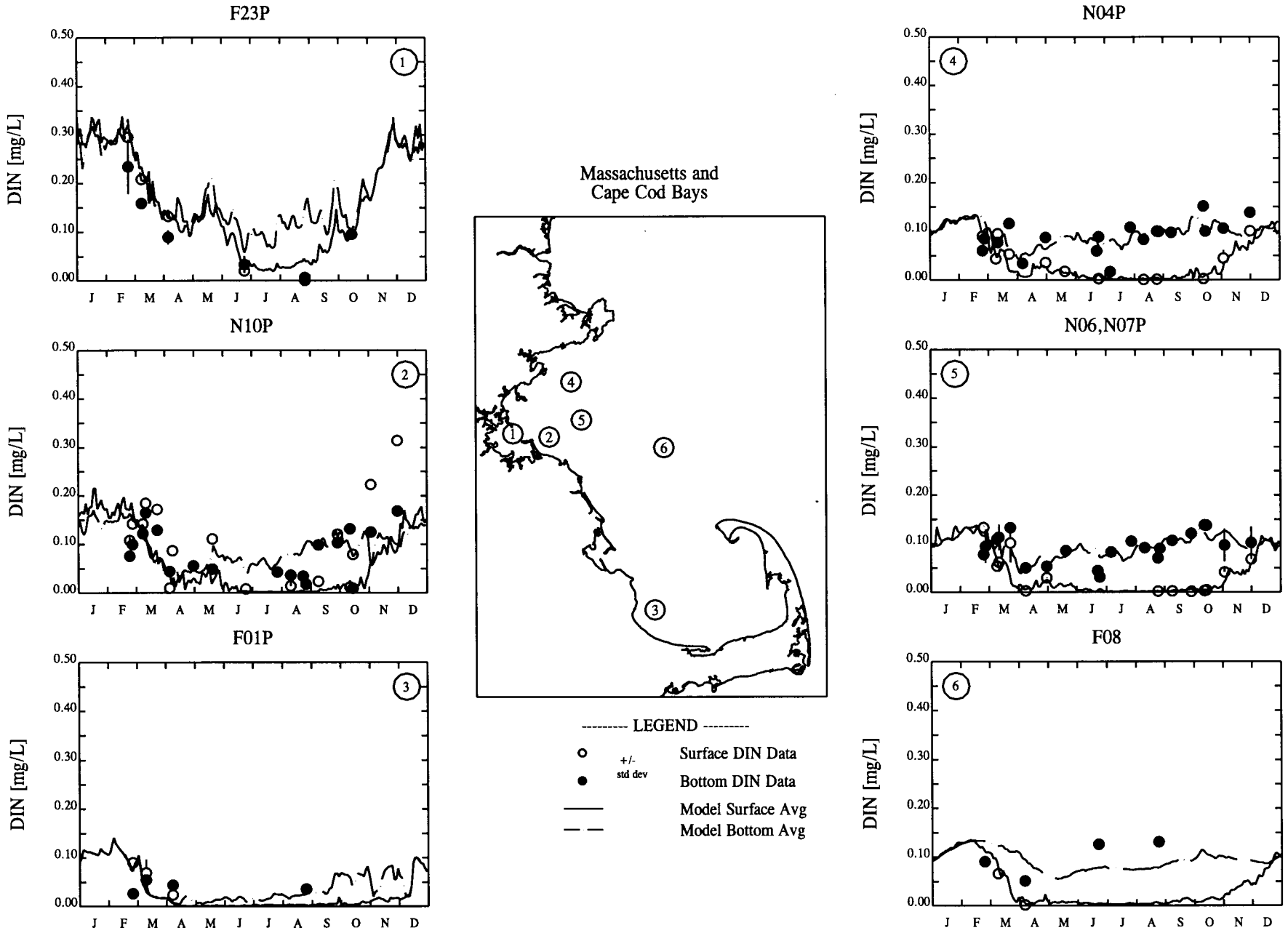


Figure 4-19. 1993 Model vs. Surface and Bottom DIN Data

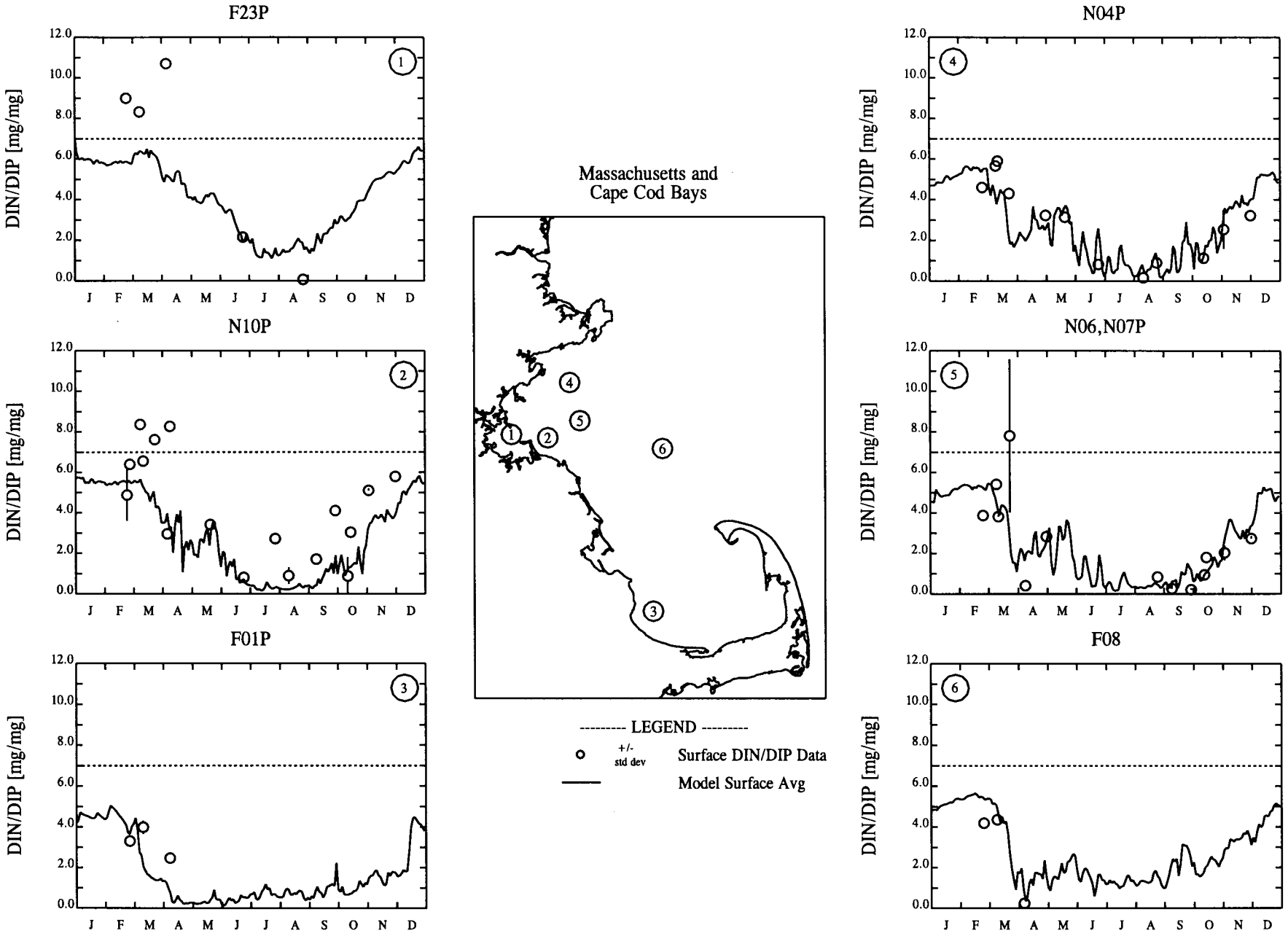


Figure 4-20. 1993 Model vs. Surface DIN/DIP Data

diatom bloom occurred, the DIN to DSi ratios suggest DSi to be potentially more limiting. The model reproduces some of the features that were observed, but does not reproduce the observed DIN to DSi ratios during the fall bloom. This is to expected since the model did not capture the fall diatom bloom.

Model versus data comparisons for 1993 DO concentration data are presented in Figure 4-22. Differences between the 1992 and 1993 model output are due, in part, to differing model boundary conditions. The model reproduces the high DO concentrations observed in April and generally reproduces the surface DO data. However, the model does not reproduce the high bottom water DO concentrations observed at a few locations during June and July.

4.2.2.3 1994 Results

The 1994 model versus data comparisons for chlorophyll-a are presented in Figure 4-23. Overall, the model reproduces the surface chlorophyll-a and fluorescence data fairly well. While the model over predicts the observed chlorophyll-a measurements at station F23P, at the entrance to Boston Harbor, during the fall, it compares favorably to the data for the remainder of the year. The model also reproduces the data collected in northern Massachusetts Bay quite well. The model also compares favorably to the observed chlorophyll data in Cape Cod Bay (F01P) and near Race Point (F29), although the model under estimates the chlorophyll data in late August and early October.

Comparisons of model versus data for surface and bottom DSi in 1994 are shown in Figure 4-24. Silica concentrations were lower in the surface and bottom waters during March and April 1994 as compared to the same months in 1992 and 1993. In addition, bottom waters appear to be higher in DSi in the fall of 1994 as compared to the fall periods in 1992 and 1993. The model is able to reproduce the spatial distribution of DSi throughout the Bays, approximately reproducing the timing of increases and decreases in silica concentrations and the observed stratification between surface and bottom concentrations. The lone exception is the Cape Cod Bay station (F01P) where the model over predicts DSi during the second half of 1994.

Figure 4-25 presents the comparisons of model versus data for surface and bottom DIN in 1994. Again in 1994, the highest DIN concentrations were observed at the stations near Boston Harbor (F23P and N10P). DIN concentrations appear to decline more rapidly in the spring of 1994 than in the previous two years. In addition, summer bottom water DIN concentrations were slightly higher, at the eastern stations, than were observed during the same period in 1992 and 1993. The

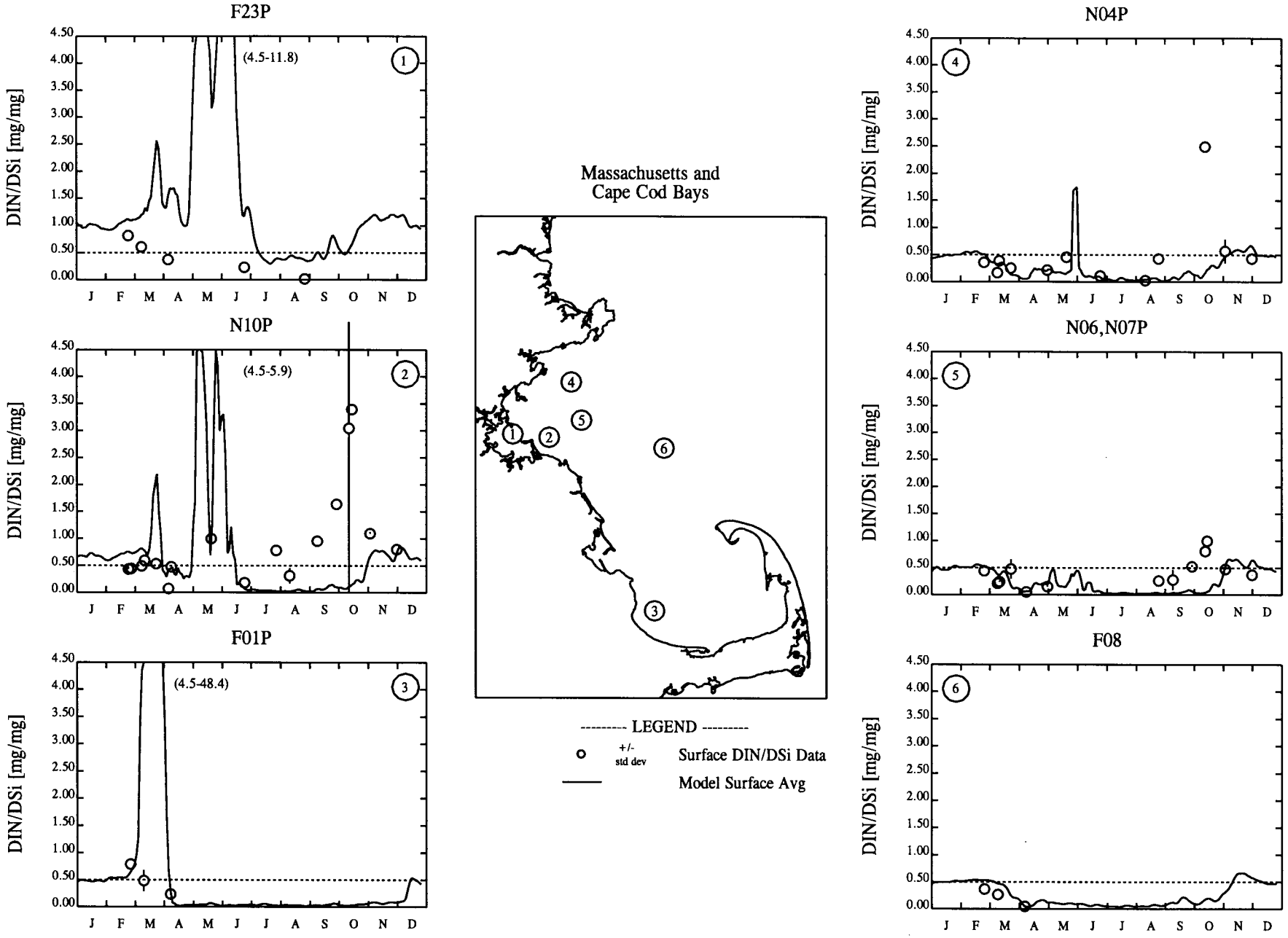


Figure 4-21. 1993 Model vs. Surface DIN/DSi Data

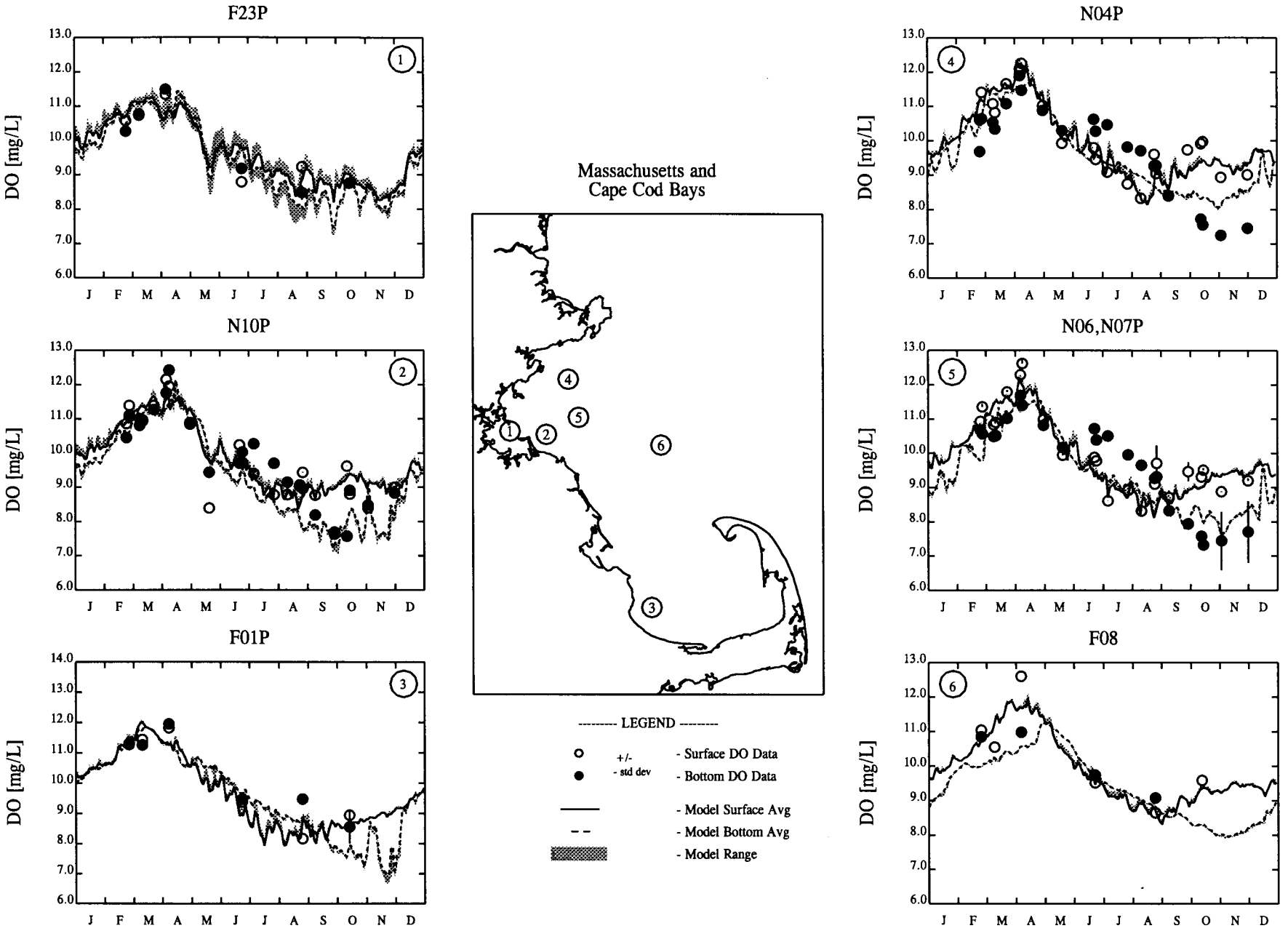


Figure 4-22. 1993 Model vs. Surface and Bottom DO Data

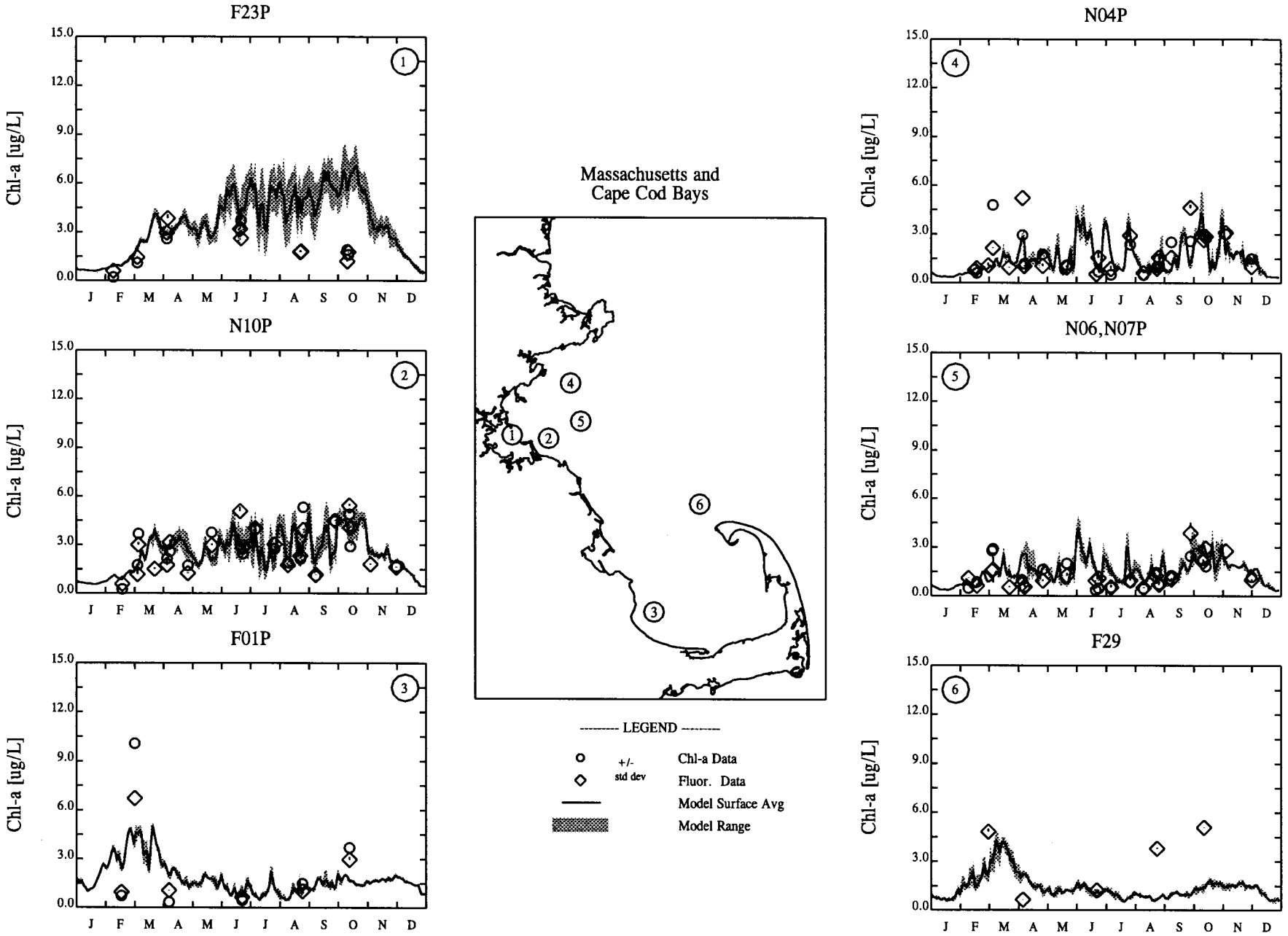


Figure 4-23. 1994 Model vs. Surface Chlorophyll-a Data

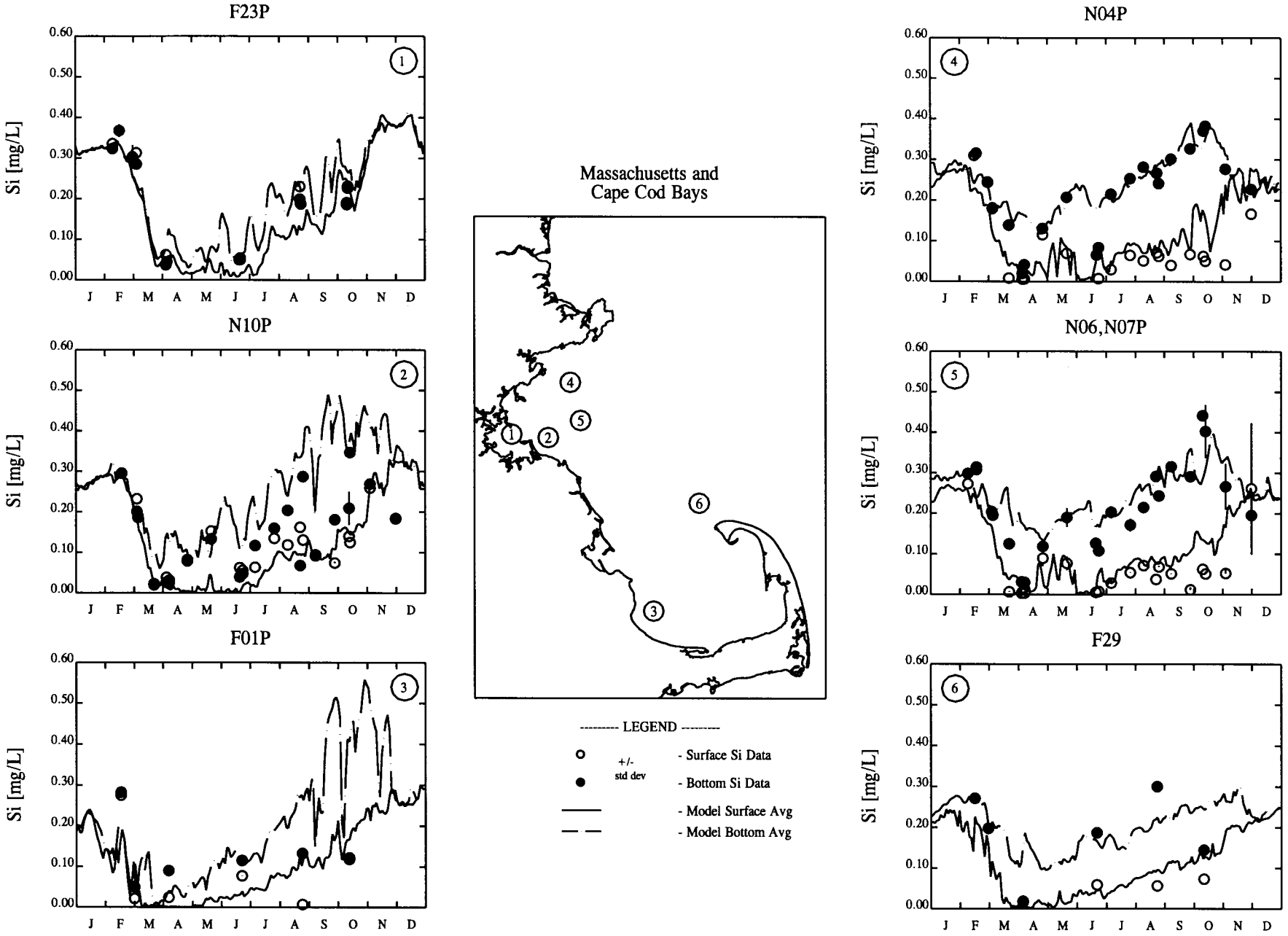


Figure 4-24. 1994 Model vs. Surface and Bottom DSi Data

model does not reproduce the rapid decline of DIN concentrations observed in early April at F23P but compares favorably with the data during the remainder of the year. There was considerable variability in the concentrations of DIN observed at station N10P, and the model only reproduces part of that variability. At the remaining sites, the model compares favorably with the surface measurements of DIN, but tends to under estimate the bottom water DIN concentrations in the northern portions of Massachusetts Bay.

Figure 4-26 presents the model versus data comparisons for surface DIN to DIP ratios. With the exception of one data point in April in Boston Harbor, the ratio data for these six locations indicate that nitrogen was the more potentially limiting of the two nutrients in the Massachusetts and Cape Cod Bays system throughout 1994. The model clearly reproduces this feature of the system. With the exception of July through September at F23P and N10P, the model provides an excellent comparison to the observed DIN to DIP ratios.

Model versus data comparisons for the surface DIN to DSi ratios for 1994 are presented in Figure 4-27. The data indicate that in the vicinity of Boston Harbor (stations F23P and N10P), silica was the more potentially limiting nutrient for much of the year. At the remaining sites, based on DIN/DSi ratios, either nitrogen or silica could have been potentially limiting during the late winter and early spring, while nitrogen was more likely to be limiting during the latter portion of the year. In general, the model DIN/DSi ratios compare favorably to the observed data for stations N04P, N06, N07P, and F29, although the model may overstate the degree of potential silica limitation. For stations F23P and N10P, the model suggests that silica is the more potentially limiting nutrient for much of the year but over estimates the magnitude of DIN/DSi. In addition, the model computes DIN to become the potentially limiting nutrient in July for station N10P, which is not supported by the data.

Figure 4-28 presents the model versus data comparisons for the 1994 surface and bottom dissolved oxygen. The lowest dissolved oxygen of the three years was observed in 1994. Also, the phenomenon of higher bottom DO concentrations than surface DO concentrations which was observed during the summer of 1992 and 1993, was not as evident in 1994. The model reproduces many of the features observed in the 1994 DO data, but does not reproduce the lowest DO concentrations observed in the bottom waters during September and October. However, the model does reproduce the fact that 1994 fall bottom water DO concentrations were lower than the corresponding time periods in 1992 and 1993.

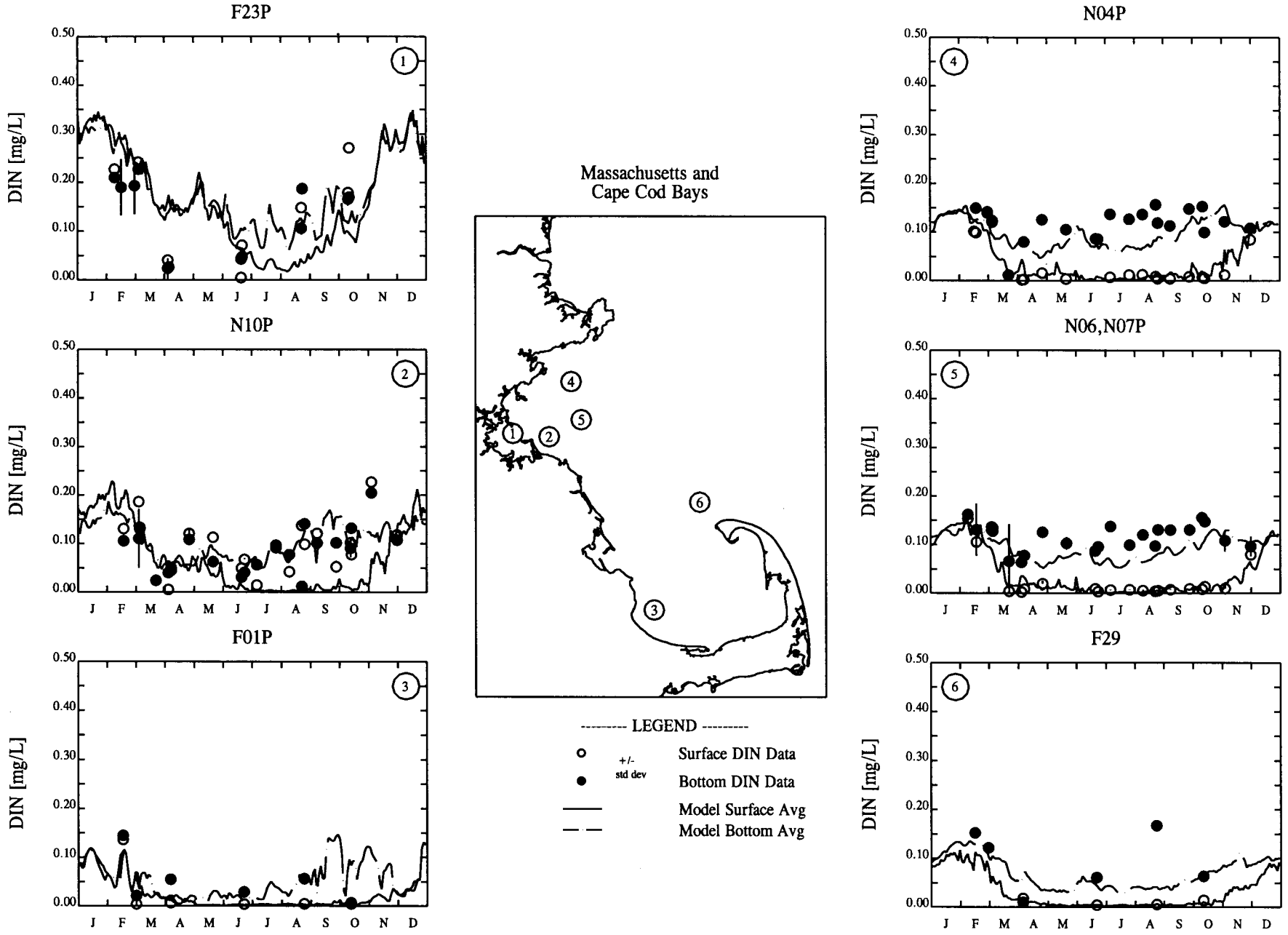


Figure 4-25. 1994 Model vs. Surface and Bottom DIN Data

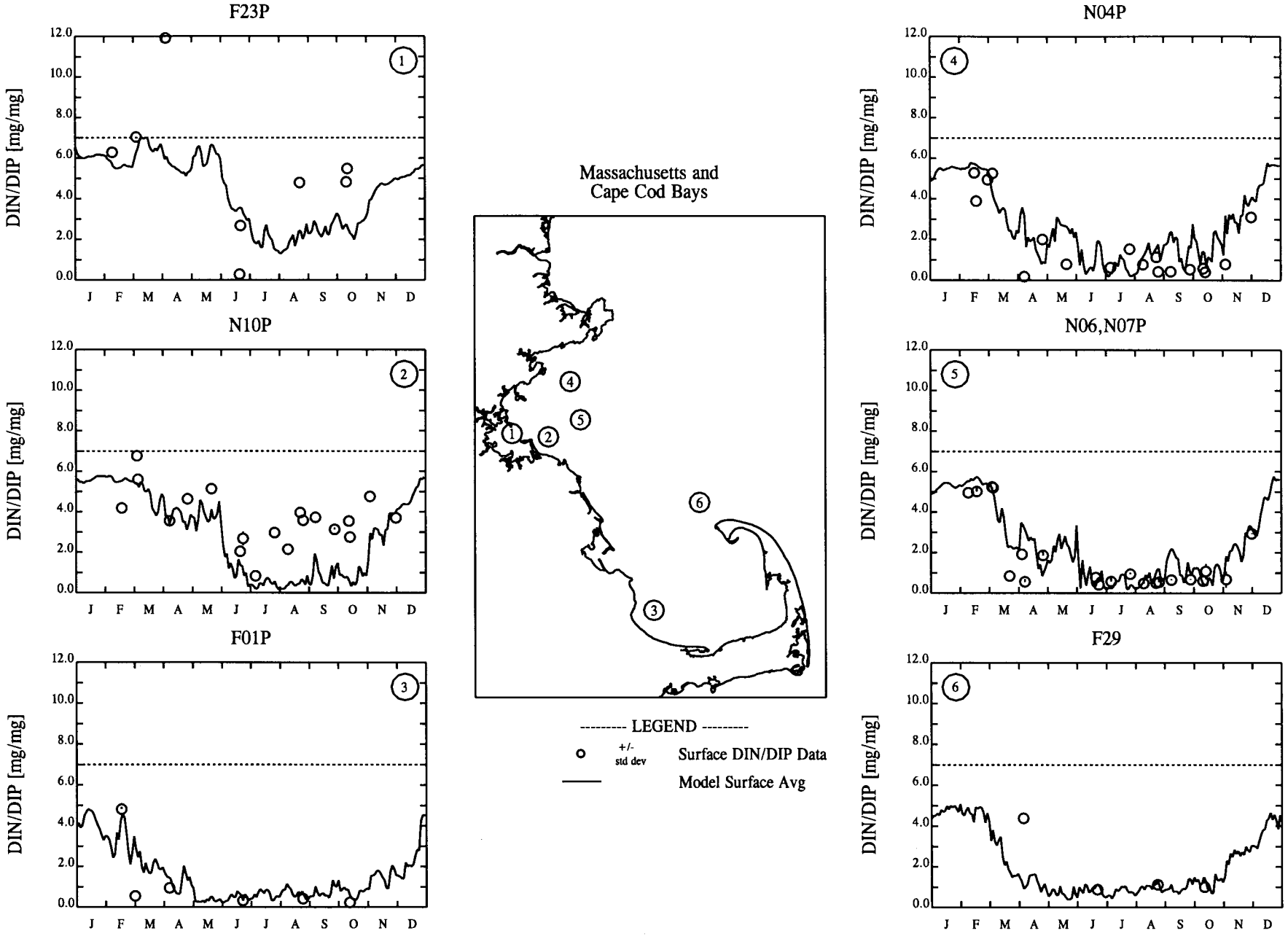


Figure 4-26. 1994 Model vs. Surface DIN/DIP Data

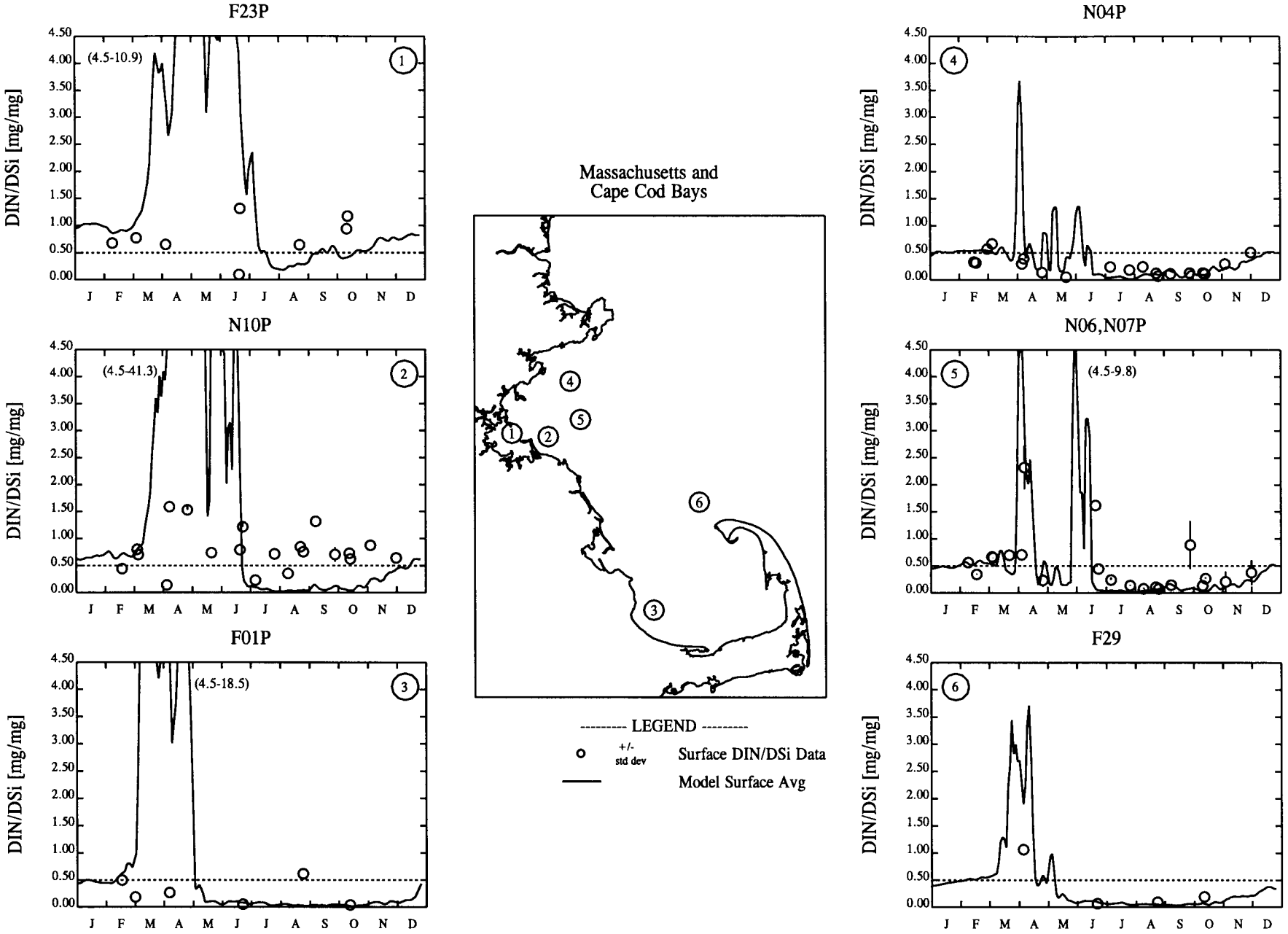


Figure 4-27. 1994 Model vs. Surface DIN/DSi Data

4.2.3 Model Versus Data Probability Distributions

The model versus data comparisons of Figures 4-8 through 4-28 tend to be rather exacting tests of the model since the data represent grab samples at a fixed point in time and space. Kelly (1992, 1993) has shown that there is considerable variability in both space and time in the observed data using tow-yo measurements. These data show high frequency variability, which suggest that a grab sample may not always be representative of the water quality conditions of a large spatial area such as would be encompassed in a model segment that is 4-9 km^2. Therefore, in order to evaluate the model calibration on another level, i.e., the degree to which the model reproduces the variability observed in the field data on a seasonal basis, model output is compared to field data using probability distribution plots.

The model versus data comparisons are separated into near field and far field analyses. The near field data were divided seasonally as follows: winter (December and February), spring (March through May), summer (June through August), fall (September through November), and on an annual basis (February through December). No data were collected in January. The model output was separated in the same manner. Since temporal sampling was conducted less frequently for the far field stations, the far field analysis was performed only on an annual basis.

Figure 4-29 presents model versus data comparisons for surface chlorophyll-a during 1992. The filled circles represent discrete chlorophyll-a data while the open diamonds represent chlorophyll fluorescence data. The model computations are plotted as a solid line and represent the range of two day averaged chlorophyll-a concentrations. For plotting purposes, chlorophyll-a concentrations less than 0.1 µg/L were plotted at 0.1 µg/L, and chlorophyll-a concentrations greater than 20 µg/L were plotted at 20 µg/L. During the winter the model under predicts the observed chlorophyll-a concentrations. The median winter chlorophyll-a concentrations are under predicted by approximately 1.5 µg/L and the maximum winter chlorophyll-a concentrations are under predicted by approximately 5 µg/L. During the spring the model over predicts the median chlorophyll-a concentration by approximately 1.5 µg/L while reproducing the maximum observed concentrations. The discrepancy between the winter/spring chlorophyll-a data and the winter/ spring model results suggests a phasing problem in computing the timing of the late winter/early spring algal bloom for the near field stations. Although it does miss the extremes, the model does a good job reproducing the observed summer chlorophyll-a data at the near field stations. During the fall the model under predicts the median observed chlorophyll-a concentrations by approximately 2 µg/L. The bottom two panels of Figure 4-29 present the near field and far field annual model versus data comparison for

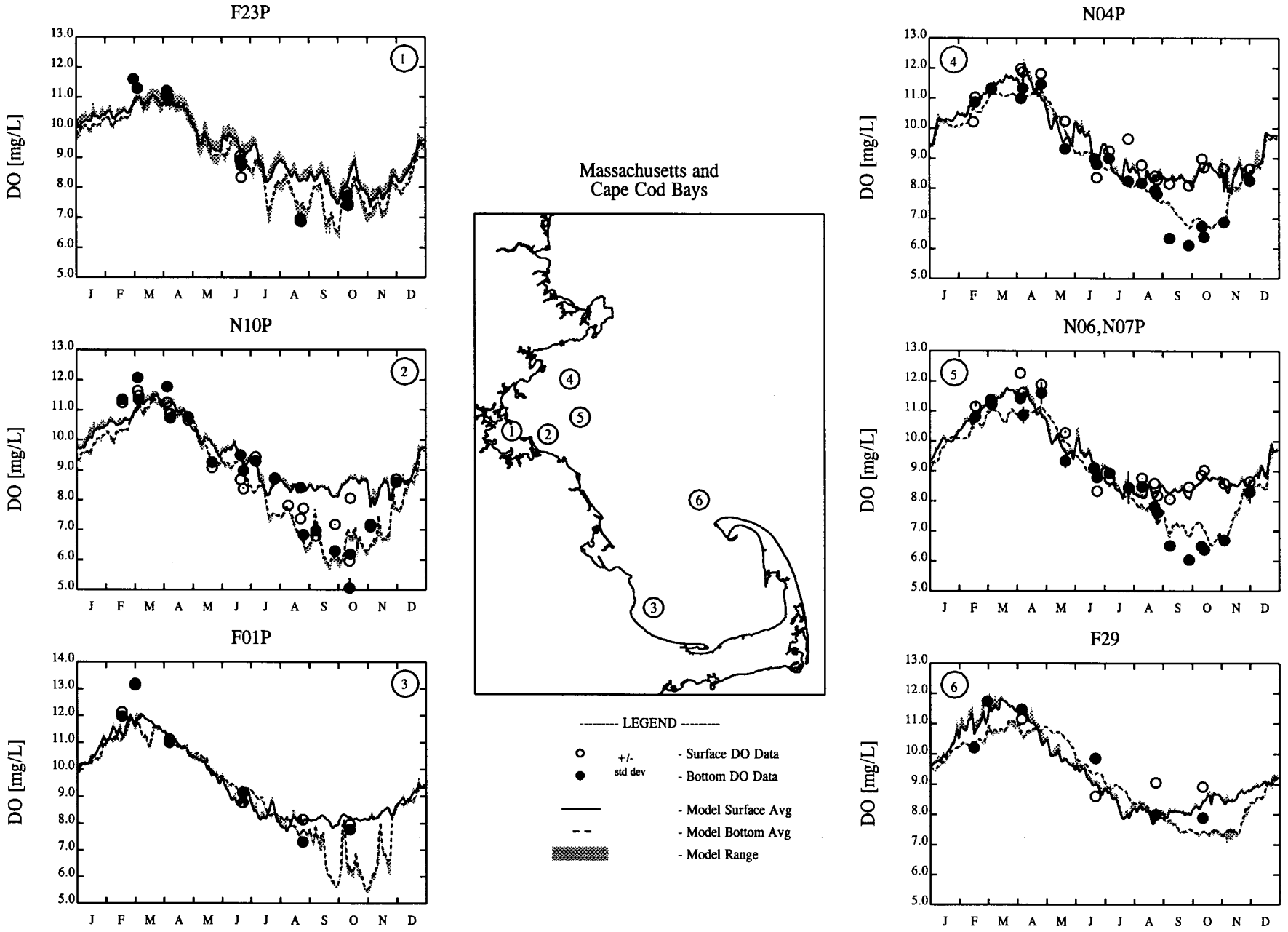


Figure 4-28. 1994 Model vs. Surface and Bottom DO Data

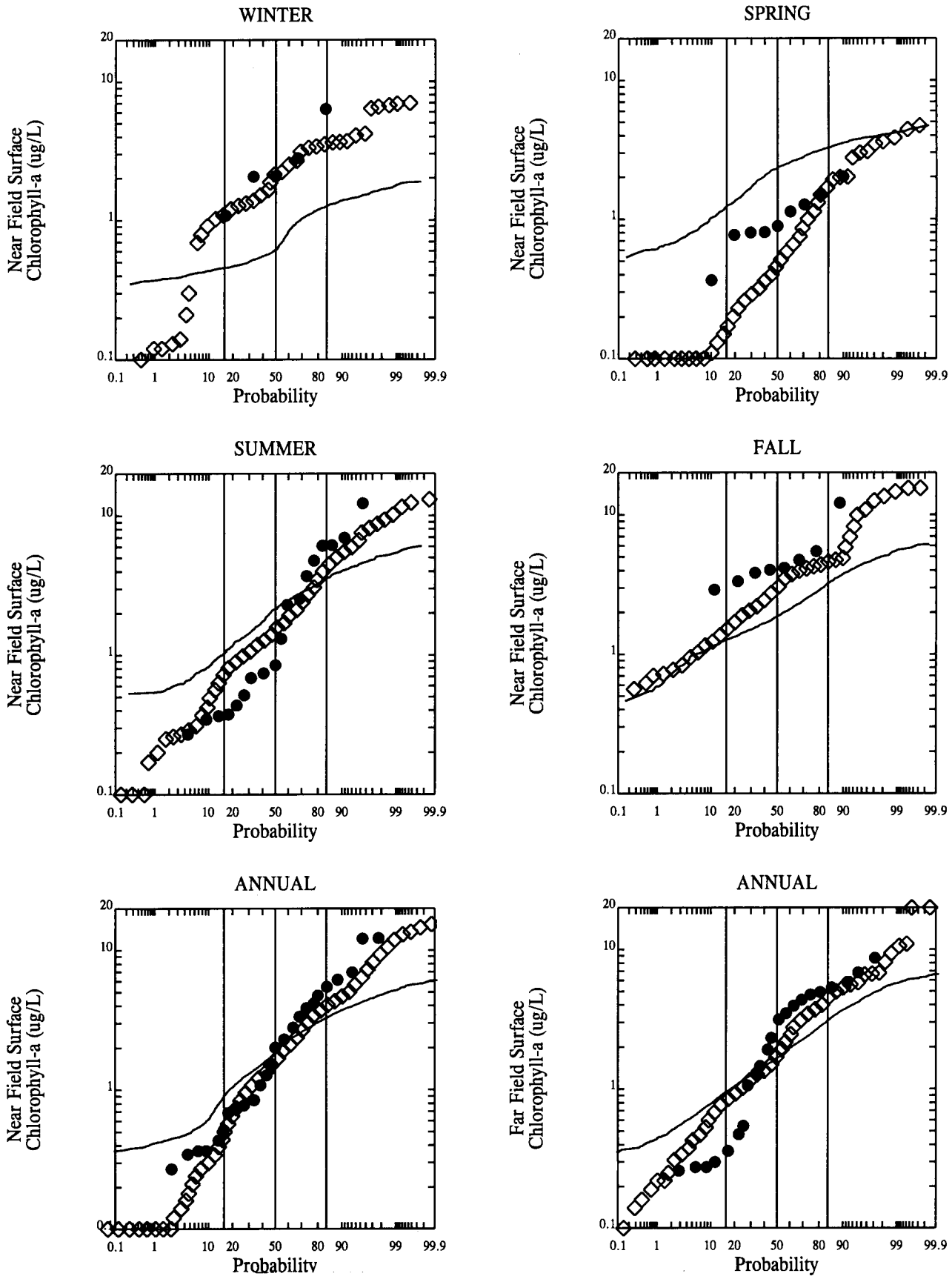


Figure 4-29. Model vs Data Probability Comparisons for 1992 Surface Chlorophyll-a

chlorophyll-a. Although the number of samples taken during each season was not consistent, no attempt was made to weight the data or model results to account for this sampling bias. On an annual basis, the model performs fairly well, reproducing the chlorophyll-a and fluorescence data within a standard deviation on both sides of the median. However, due in part to the fact that the present model is not capable of reproducing phytoplankton patchiness, the model does not reproduce the maxima and minima observed in the data.

Figure 4-30 presents the mid-depth (12.5-17.5 m) probability comparisons of discrete and fluorescence chlorophyll-a data to model computations for 1992. The model has similar calibration biases to the observed mid-depth chlorophyll-a data as were computed by the model relative to the surface chlorophyll-a data. The model under-estimates the observed chlorophyll-a concentrations in the winter and fall, over-estimates the observed spring time chlorophyll-a concentrations, and does a good job for the summer chlorophyll-a concentrations. On an annual basis the model does a good job for both the near field and far field stations. Both the model and data show less variability at mid-depth than was observed at the surface.

Model versus data comparisons for the bottom water dissolved oxygen concentrations for 1992 are presented in Figure 4-31. The model and data are presented in the same manner as for chlorophyll-a above. However, note that the scales change from panel to panel. During the winter, the model does a poor job reproducing the observed bottom water dissolved oxygen concentrations, missing the median observed concentrations by more than 2 mg O_2/L. This under prediction is due, in part, to the underestimation of the phytoplankton biomass during the winter. Inappropriate boundary conditions could be another reason for the underestimation of DO during the winter. During the spring, summer and fall, the model provides reasonable comparisons to the observed bottom water DO concentrations. For the most part, the model predictions are within 1 mg O_2/L of the data. On an annual basis, the model reproduces the lowest 80 percent of the DO concentrations very well at the near field stations, while missing the highest observed DO values. At the far field stations, the model reproduces the median and low end DO concentrations but, does not reproduce the high end DO concentrations.

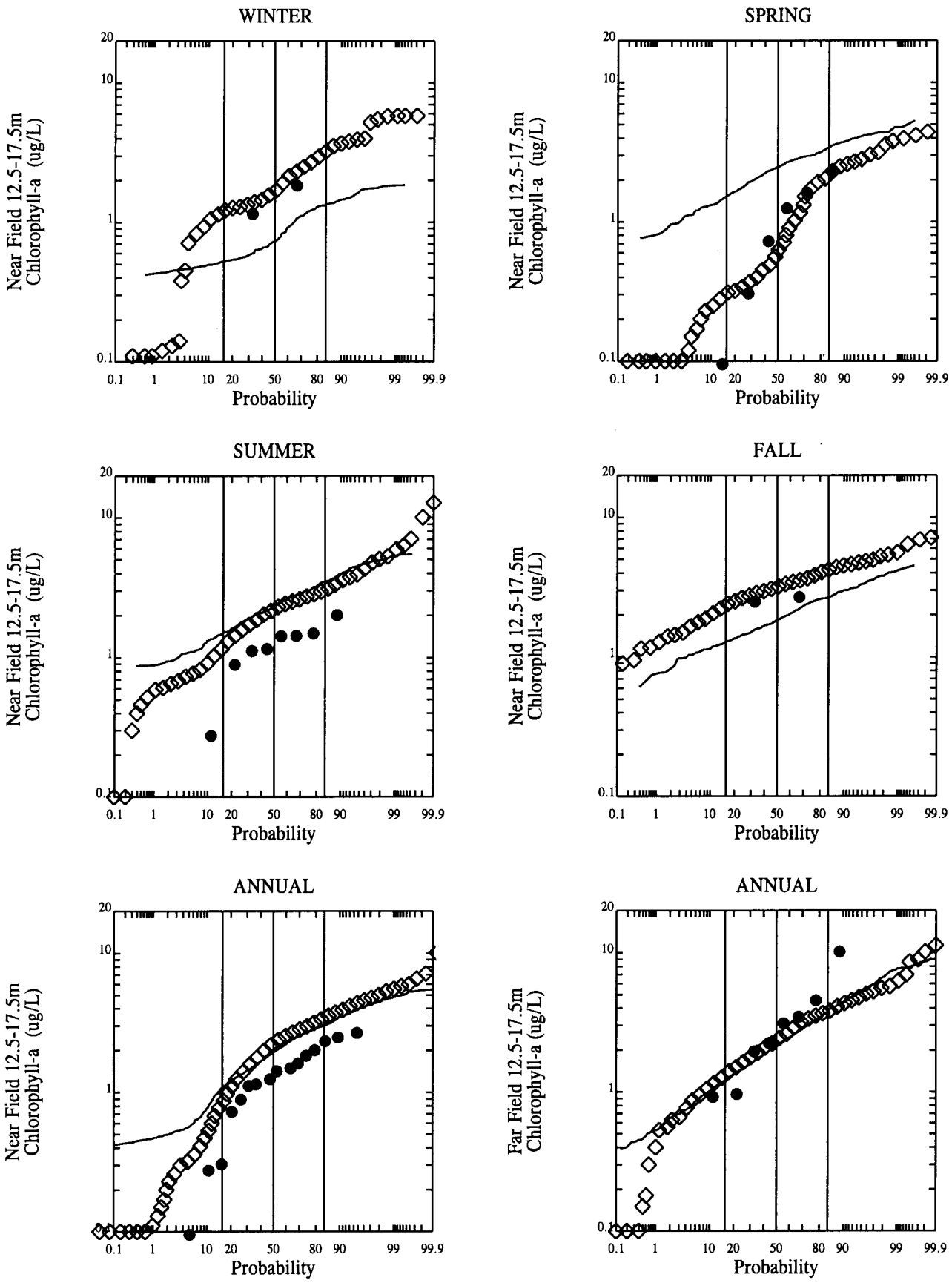


Figure 4-30. Gross Primary Productivity 1993

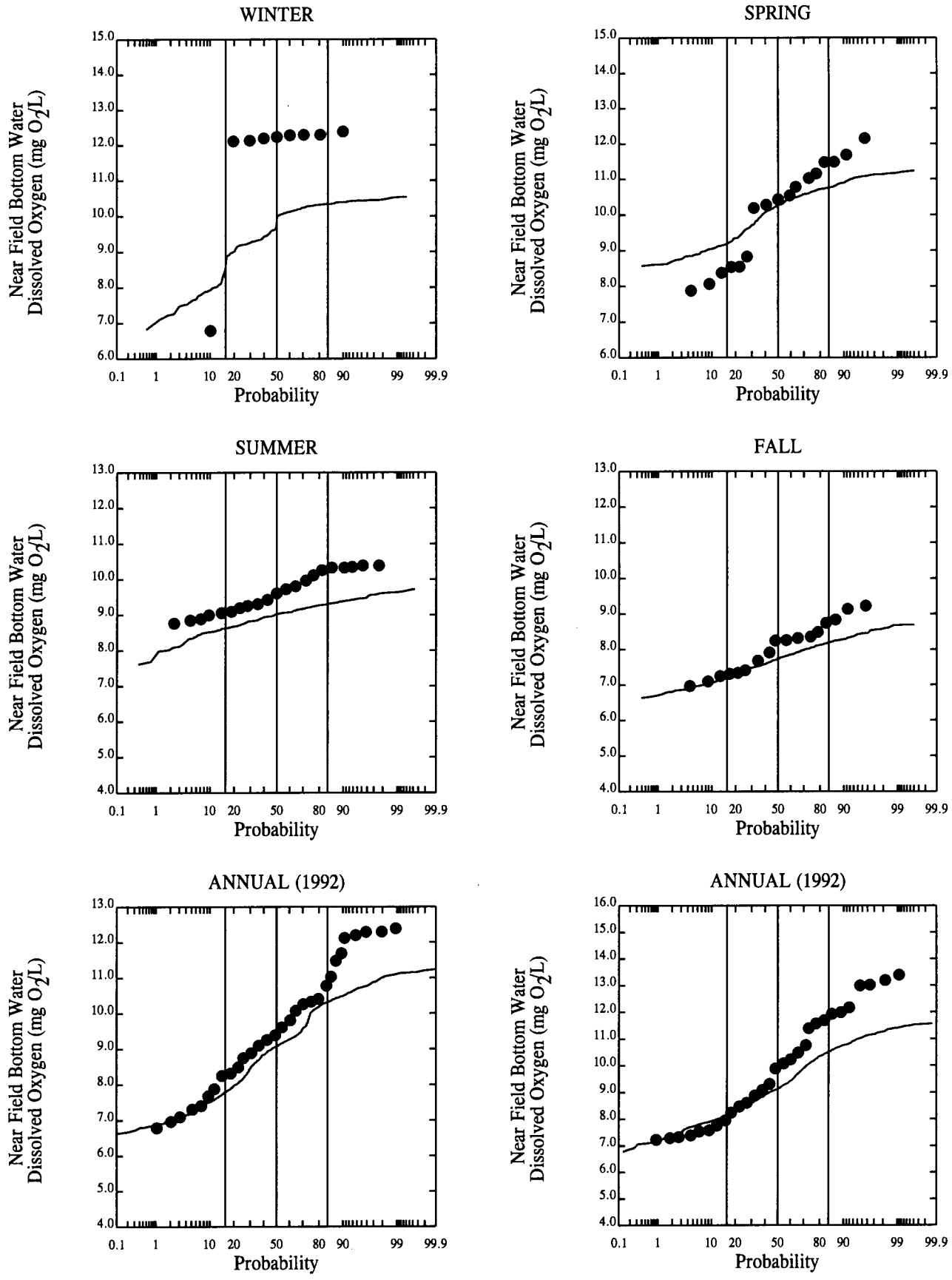


Figure 4-31. Model vs Data Probability Comparisons for 1992 Bottom DO

Figures 4-32 through 4-34 present model versus data probability distributions for surface chlorophyll-a, mid-depth chlorophyll-a, and bottom DO respectively for 1993. As shown in Figure 4-32 the model does a much better job reproducing the observed winter surface chlorophyll-a concentrations for 1993 than for 1992. It should be noted, however, the differences in model computations between the two years are not great. The model matches the median plus or minus one standard deviation (or 66 percent of the data) chlorophyll-a concentrations well, while over-estimating the maximum chlorophyll-a concentration by approximately 1 µg/L during the winter. During the spring of 1993, the model over-predicts the observed chlorophyll-a consistently, but never by more than 1 µg/L. The summertime near field surface chlorophyll-a data show a wide range of concentrations. The model reproduces the median chlorophyll-a concentration fairly well, but does not fully reproduce the variability observed in the data. In the fall of 1993, the largest phytoplankton bloom of the three-year calibration period was observed. The model did not reproduce the high chlorophyll-a concentrations observed during this season. On an annual basis, the model reproduces the median concentration at both the near field and far field stations, but does not reproduce the observed range of the chlorophyll-a data. In particular, the model under estimates the observed chlorophyll maxima by a factor of two to three.

In Figure 4-33, the model matches the observed mid-depth chlorophyll-a concentrations somewhat better than it matched the surface chlorophyll-a concentrations. In general the model performs reasonably well for the winter, spring and summer periods, but misses the fall algal bloom. On an annual basis, for both the near and far field stations, the model and data medians are slightly higher at mid-depth than the surface concentrations.

Figure 4-34 presents the model versus data probability analysis of bottom DO concentration for 1993. On a seasonal basis the model does a good job reproducing the observed DO concentration data. During the winter the model is within 0.5 mg O_2/L throughout the entire data set. During the spring and fall there is almost an exact match between the model and data. During the summer the model under predicts the observed DO concentration by less than 1 mg O_2/L. The model reproduces range of the DO data on an annual basis very well for both the near field and far field stations.

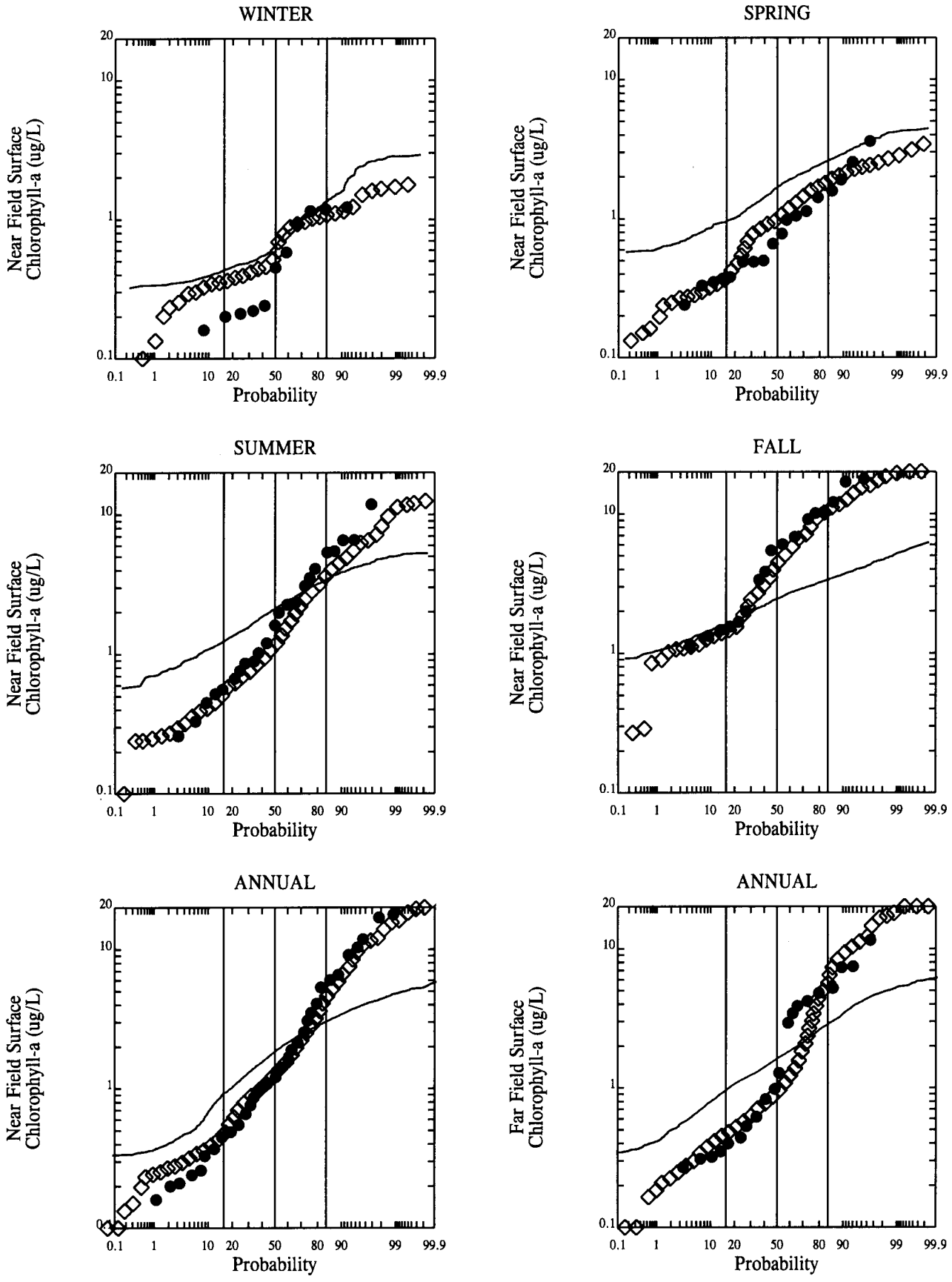


Figure 4-32. Model vs Data Probability Comparisons for 1993 Surface Chlorophyll-a

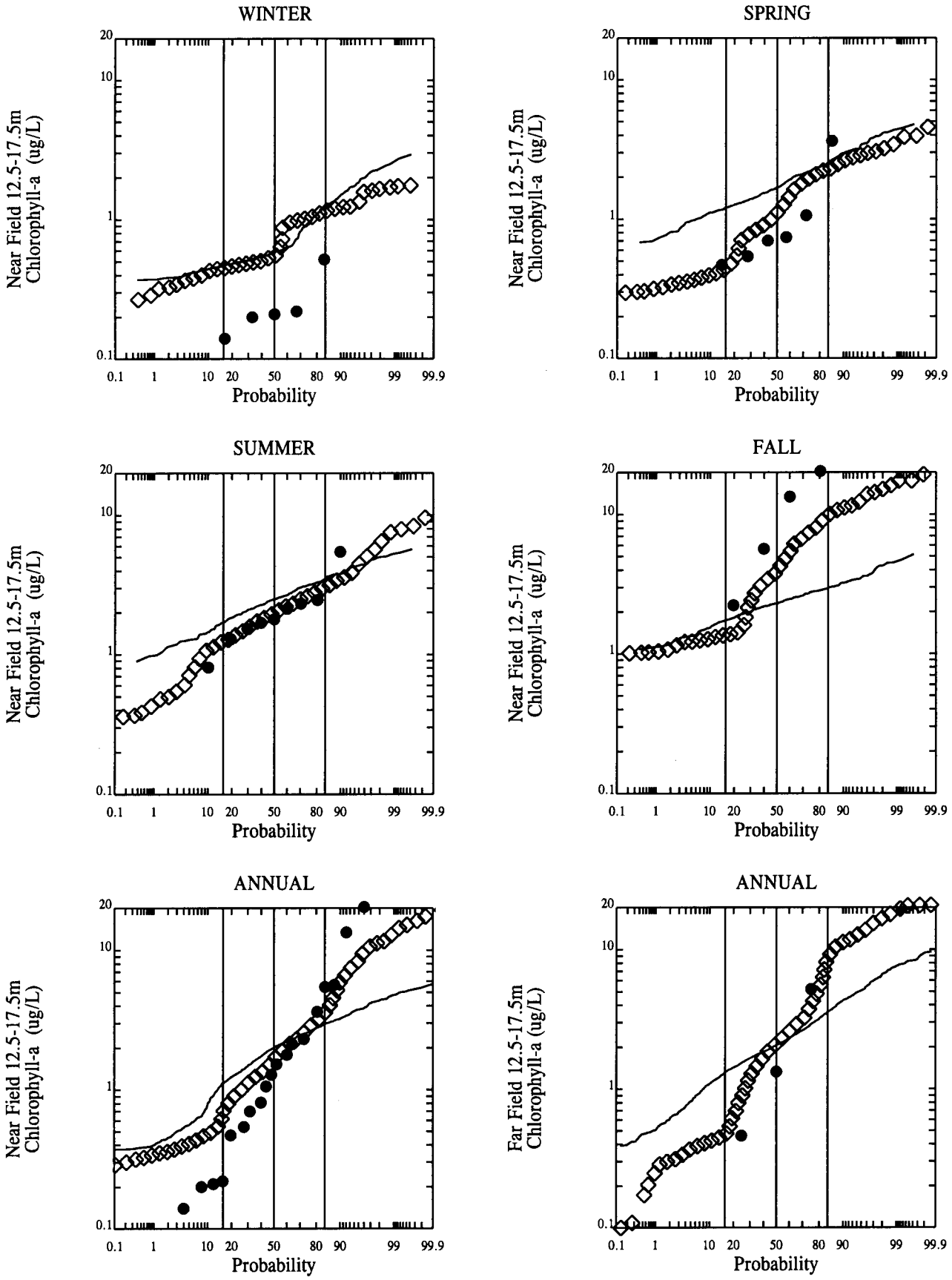


Figure 4-33. Model vs Data Probability Comparisons for 1993 Mid-Depth Chlorophyll-a

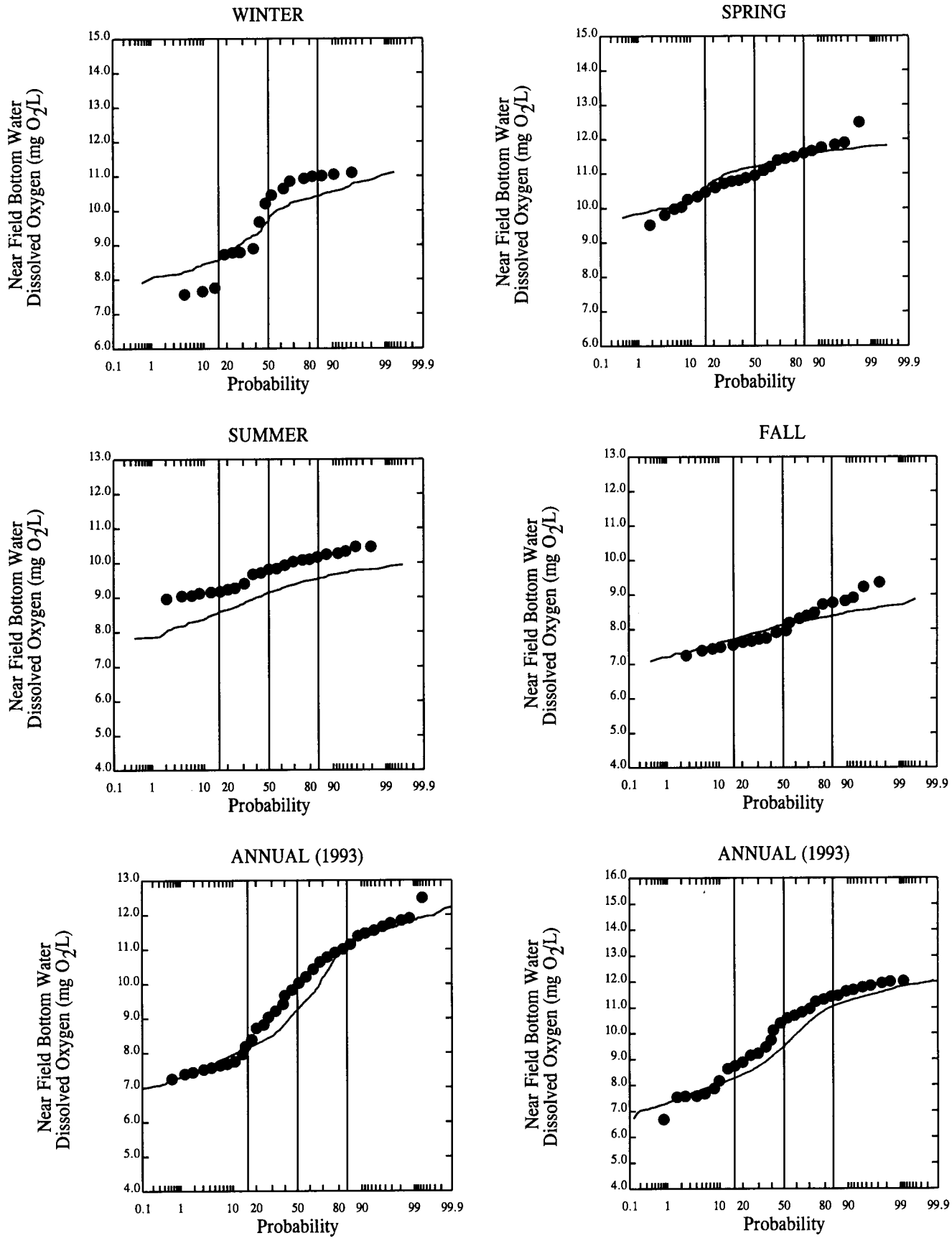


Figure 4-34. Model vs Data Probability Comparisons for 1993 Bottom DO

The 1994 model versus data probability distributions for surface chlorophyll-a, mid-depth chlorophyll-a and dissolved oxygen are presented in Figures 4-35, 4-36, and 4-37, respectively. Figure 4-35 shows the comparison between the model and data for chlorophyll-a to be quite good. During the winter, the model computed surface chlorophyll-a concentrations match the observed data. During the spring, the model computations are consistent with the majority of the observed data, missing only the extremes in the data. For the summer months the model over-estimates the chlorophyll-a concentrations by about 1 µg/L. As would be expected, since the seasonal distributions are reasonably well matched by the model the annual comparisons are favorable as well. Again, the difference in the model results between 1994 and the other years is not that great.

The model also compares favorably to the mid-depth chlorophyll-a concentrations for 1994 as shown in Figure 4-36. The model is within 0.2 to 0.3 µg/L of the low-end mid-depth chlorophyll-a concentrations, within 0.5 to 1.0 µg/L of the median mid-depth chlorophyll-a concentrations and within 0.1 to 3 µg/L of the high-end chlorophyll-a concentrations during all seasons at the near field stations. On an annual basis, the comparison of mid-depth chlorophyll-a data to model computations at the near field stations during 1994 is very favorable. At the far field stations, the model under predicts the highest 10 percent of the observed chlorophyll-a concentrations at mid-depth.

Figure 4-37 presents the model versus data probability comparison for bottom DO concentrations in 1994. During the winter there is a discontinuity in the data distribution reflecting the fact that some of the data is from the February 1994 sampling cruises and some of the data is from the December 1994 sampling cruises. The model reproduces the lower observed winter DO concentrations, but under-estimates the maximum DO concentrations between 1 to 1.5 mg O_2/L. During the spring the model does a fair job of reproducing the observed bottom water DO concentrations. The model does an excellent job reproducing the observed summer bottom DO concentrations. In the fall, the model generally over-estimates the observed bottom water DO concentrations. In particular, the model computes minimum concentrations of just under 6 mg O_2/L versus the observed minimum of just under 5 mg O_2/L. However, the model does compute fall DO concentrations to be 0.8 - 1.0 mg O_2/L lower than were computed for 1992 and 1993. This is similar to the observed data which show the fall of 1994 to have a median DO concentration of approximately 1 mg O_2/L lower than 1992 or 1993. The model also computes the lowest bottom DO during this period, but the model over-estimates the DO concentrations by approximately 1 mg O_2/L during this time. On an annual basis the model reproduces the majority of the distribution observed bottom DO concentrations in 1994.

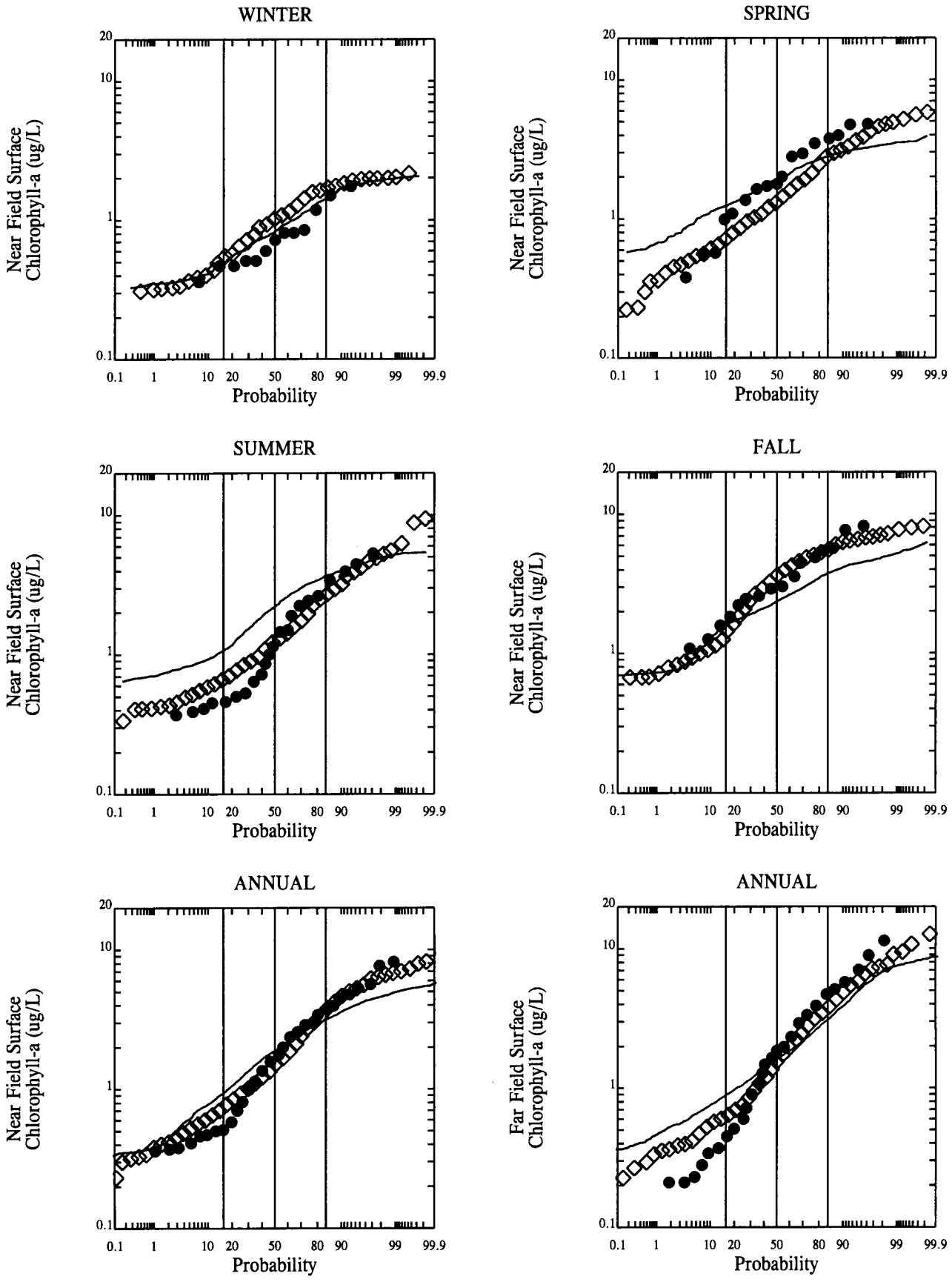


Figure 4-35. Model vs Data Probability Comparisons for 1994 Surface Chlorophyll-a

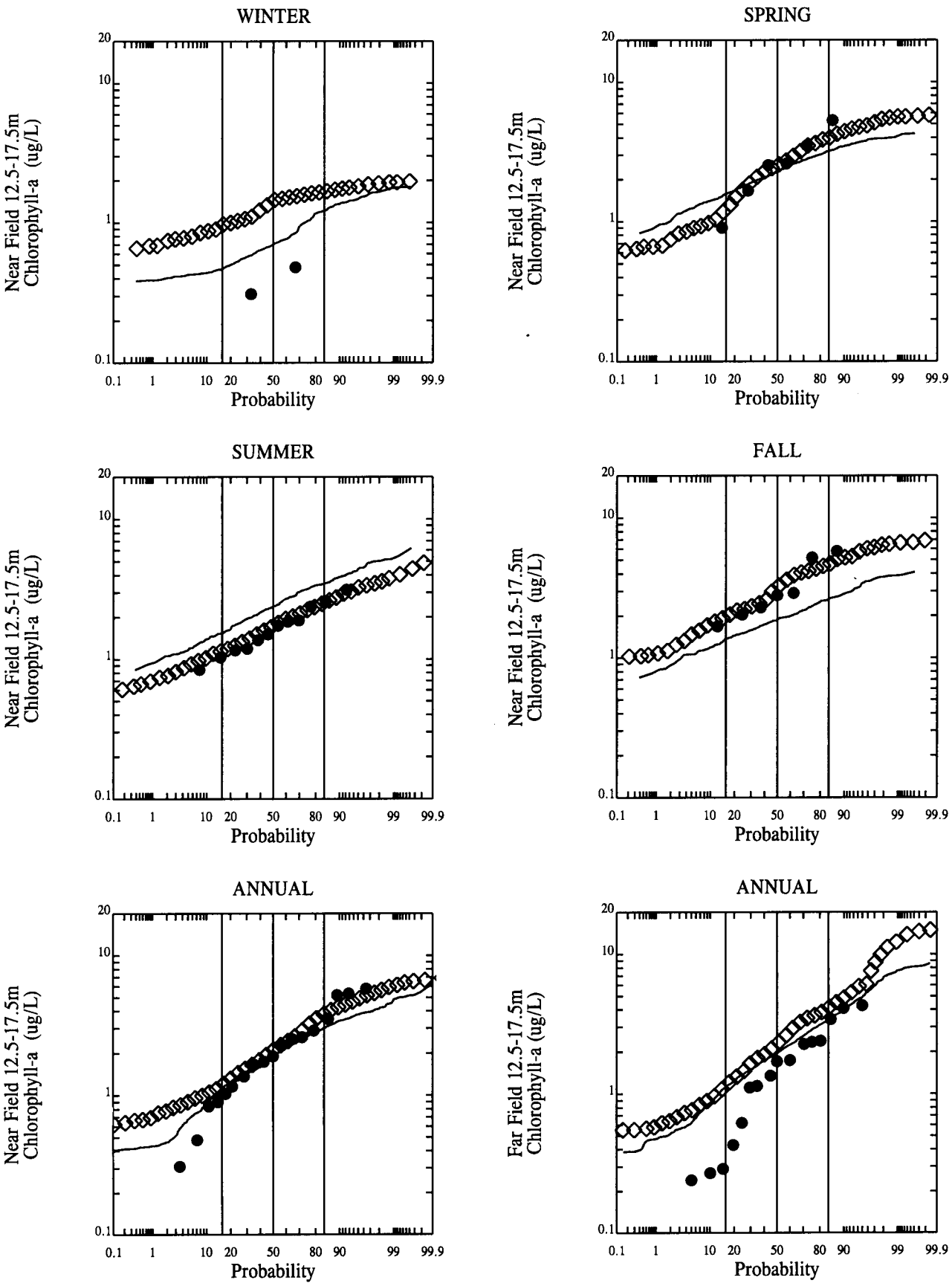


Figure 4-36. Model vs Data Probability Comparisons for 1994 Mid-Depth Chlorophyll-a

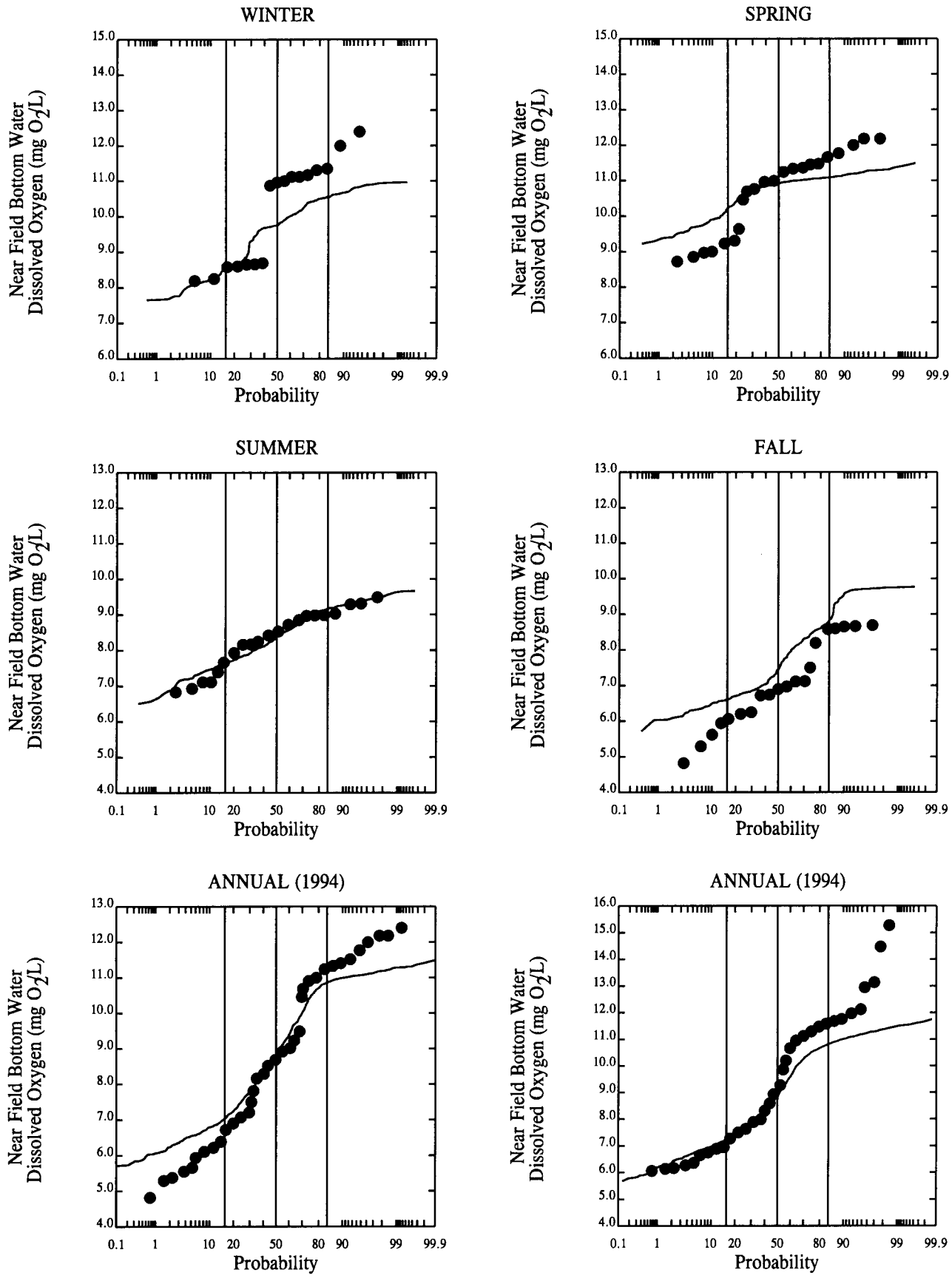


Figure 4-37. Model vs Data Probability Comparisons for 1994 Bottom DO

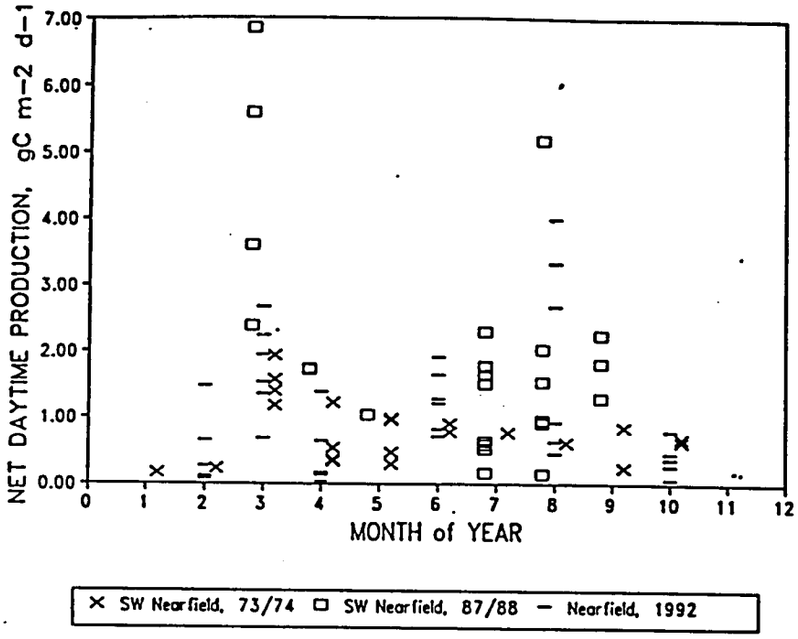
4.2.4 Primary Productivity and Community Respiration

4.2.4.1 Primary Productivity

Another measure of the water quality model calibration can be determined by comparing model computations of system metabolism (the production and consumption of organic matter) versus observed data. Two measures of metabolism which can be estimated from field data are primary productivity and community respiration.

In 1992, oxygen-based light bottle-dark bottle equipment were used to estimate primary production. In 1993 and 1994 oxygen light-dark bottles were replaced by ^{14}C techniques. Measurements of primary production were made during February, March, April, June, August and October of each year. In 1992, six near field stations were used to estimate primary productivity. In 1993, 10 stations were used to estimate primary productivity. These stations included the six near field stations, two far field stations in Cape Cod Bay, and two coastal far field stations. However, in 1994 only two stations were used to estimate primary productivity, namely station F23P, a coastal station, and N01P, a near field station.

Since there is some controversy as to whether gross or net primary production is measured by oxygen light-dark bottles and ^{14}C techniques due to (1) the length of the time period used for the incubation and (2) the assumptions used to convert short and mid-length incubations to daily production estimates, the model computations of both net primary production (NPP) and gross primary production (GPP) will be compared to the data. Figure 4-38 presents a comparison between the 1992 near field estimates of primary productivity and the water quality model estimates of NPP and GPP. The data figure also includes: 1973/1974 *in situ* measurements of primary productivity, as reported by Parker (1974), taken a few kilometers south of the futureoutfall diffuser; and a 1987/1988 data set derived from incubator measurements from three stations located in the vicinity of the current near field stations, in western Massachusetts Bay, as reported by MWRA (1988,1990). The data presented on this panel show a marked variability not only between data sets, but also within individual studies. Maximum rates of productivity in 1992 were observed in March, June and August; low rates of productivity were found during the winter months.



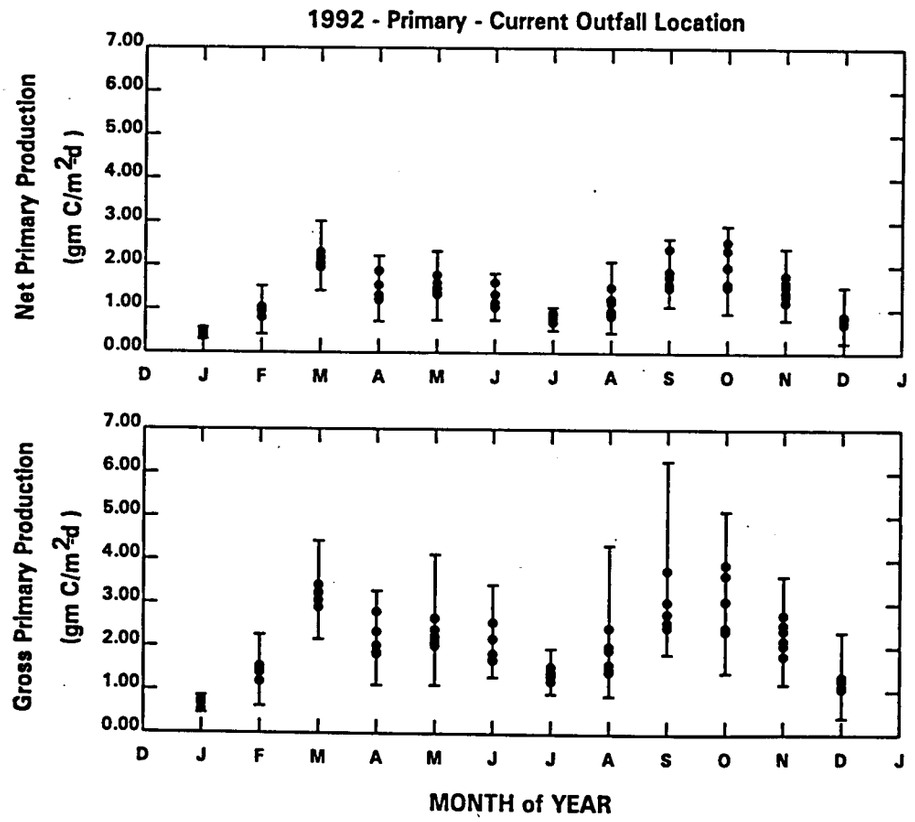


Figure 4-38. 1992 Primary Productivity Comparisons

The model results for NPP and GPP in 1992 differ by approximately 50 percent with the GPP being higher. The model computations only partially match the observed data. During the early part of 1992, model and data trends are similar with a peak in primary production in March followed by a drop off in late spring. The model predicts low productivity in June while the data indicate a small peak in June. The data indicate high primary productivity in August followed by low productivity in October. While the model predicts increasing primary productivity in August it does not peak until September and October.

A comparison of model versus data for 1993 primary productivity is presented in Figure 4-39. This figure provides a spatial as well as seasonal picture of primary productivity throughout the Massachusetts Bays system. In Cape Cod Bay the highest primary productivity was measured in February. Lower primary productivity rates were observed in March, April, June and August followed by slightly higher rates in October. At the near field and coastal sites, low primary productivity rates were observed in February and March, followed by elevated rates in April, June and August. The rates at the near field and coastal sites diverge in October when near field rates increase and coastal rates remained similar to August rates.

The model matches the 1993 production data fairly well. At the nearfield stations, the model primary productivity begins to increase earlier than the data, peaking in March rather than April. The model computed GPP rates match the June and August observations fairly well. Both the model and data show the highest primary production rates in October. In Cape Cod Bay, the model predicts higher productivity than at the other locations for January, but both model NPP and GPP rates are lower than the February data. The model computes higher primary productivity in the summer and lower primary productivity in the fall than were observed in Cape Cod Bay. The model reproduces the February and March coastal data, but misses the timing of the increase in April. Beginning in June the model reproduces the coastal station data fairly well. Again the observation can be made that the model computed GPP is approximately 50 percent greater than the NPP.

Model versus data comparisons for the 1994 primary productivity are shown in Figure 4-40. The two stations sampled show similar patterns except that the near field station has a peak primary productivity rate in March and again in October, whereas the harbor edge station does not peak until April and the October rates are not much higher than the August rates. Overall, 1993 had the highest measured primary productivity rates and 1992 and 1994 had similar primary productivity rates. However, the estimates of primary productivity can vary a great deal based on the weather conditions on the date a site was sampled and the patchiness of the phytoplankton biomass.

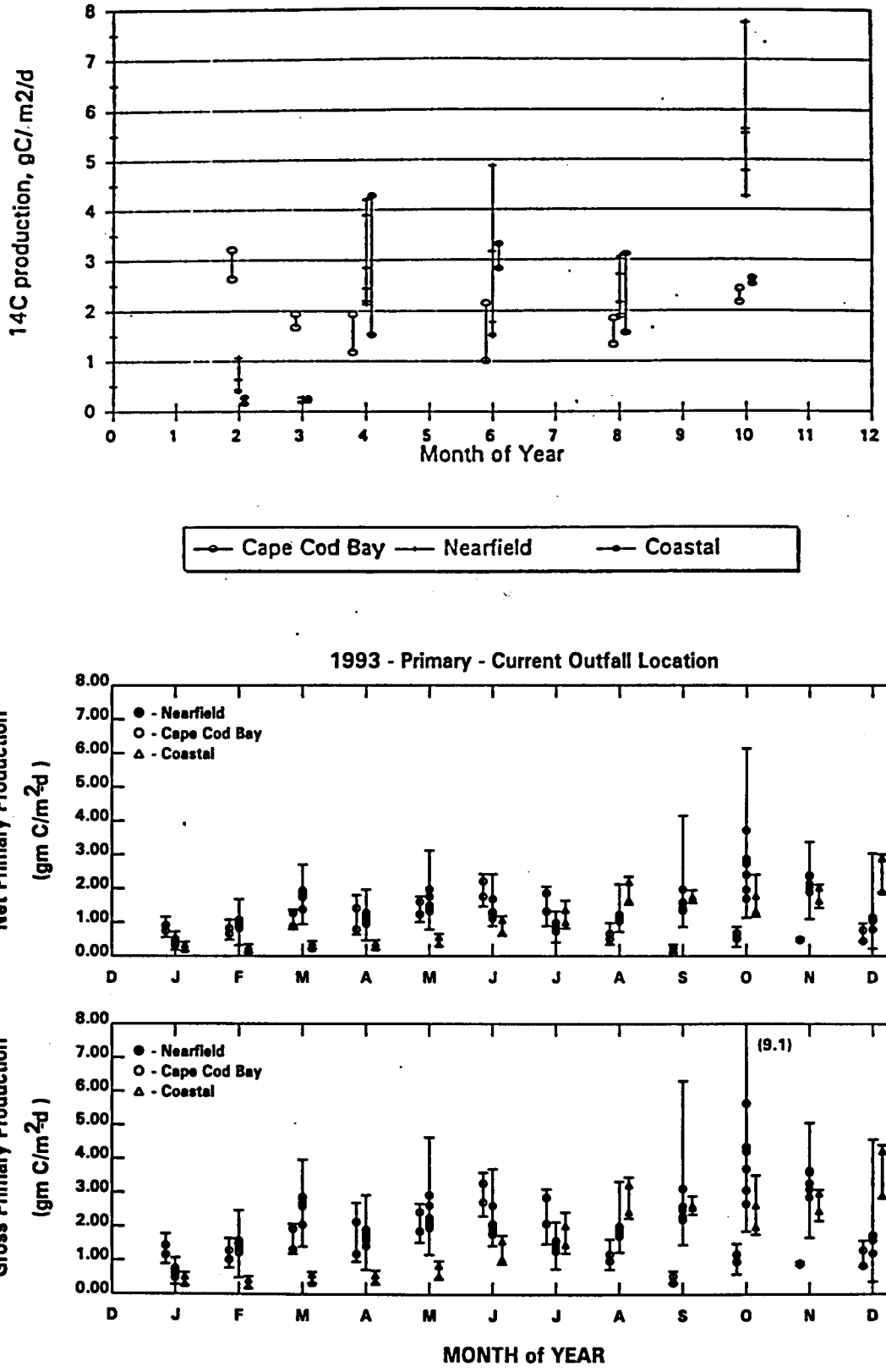
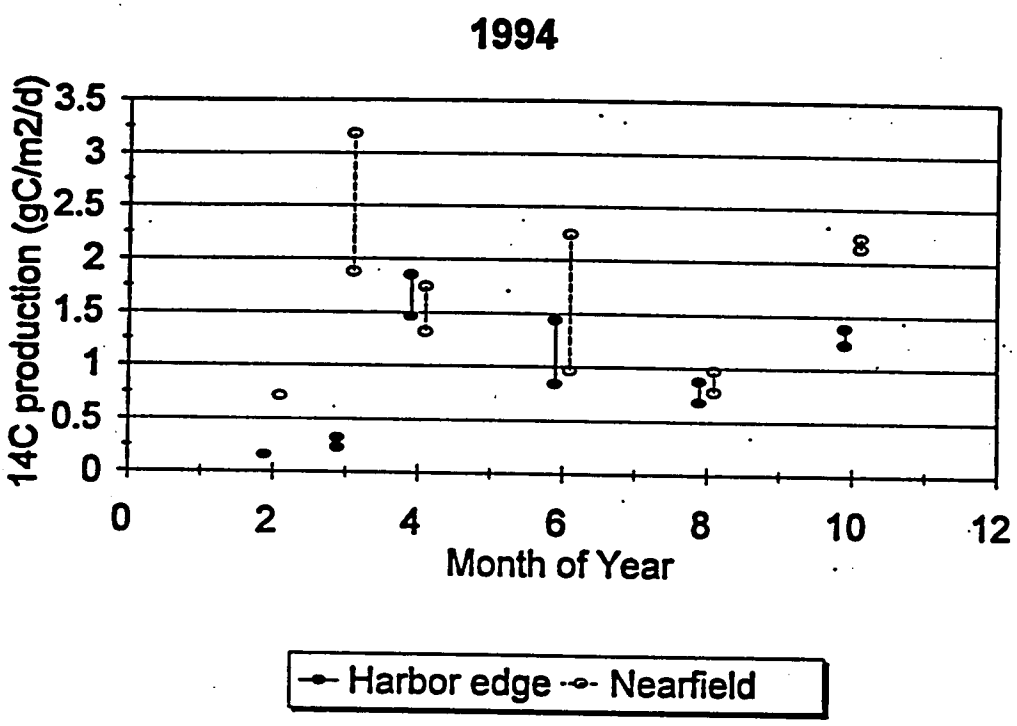


Figure 4-39. 1993 Primary Productivity Comparisons



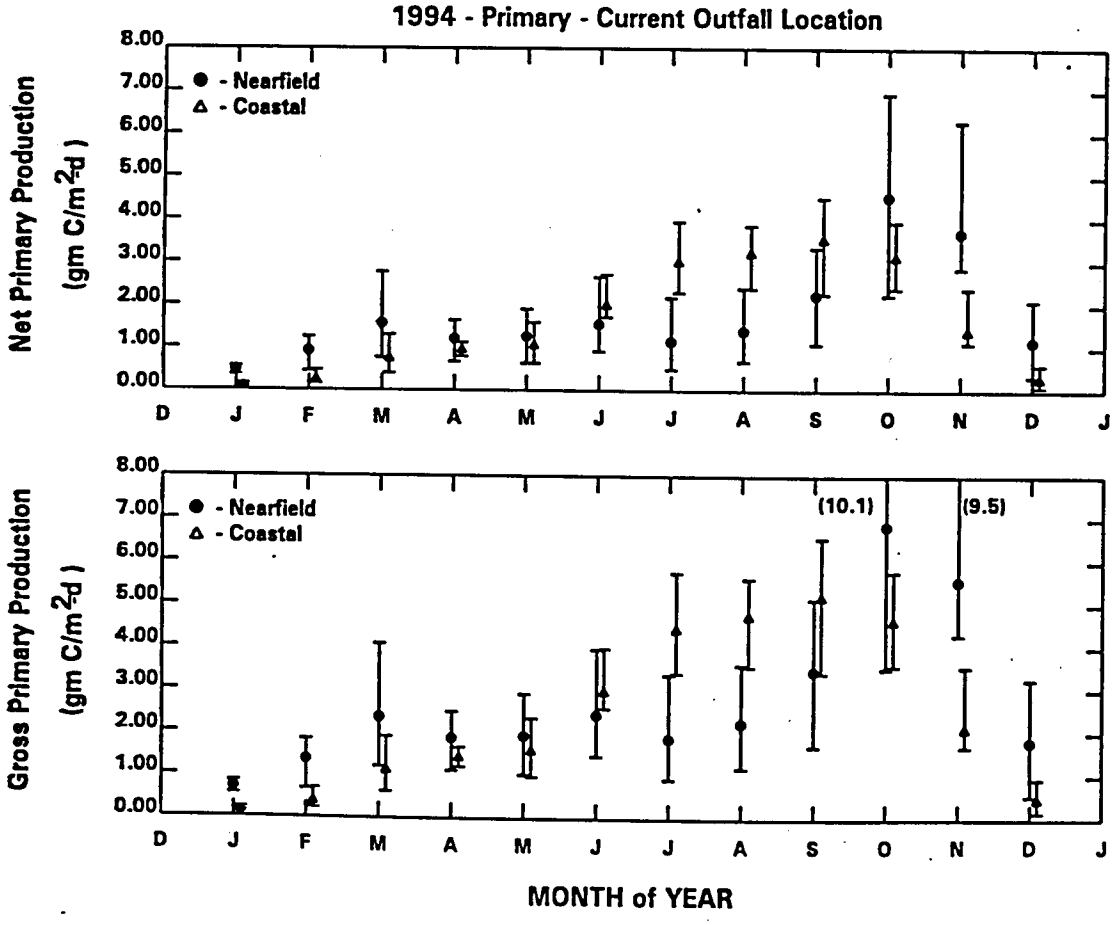


Figure 4-40. 1994 Primary Productivity Comparisons

The model computations for 1994 for both NPP and GPP appear to be high relative to the observed data, particularly in the later part of the year. The model reproduces the February through April data fairly well, reproducing the peak observed in March. The model GPP over predicts the June and August ^{14}C production estimates, while the model NPP computations compare more favorably. Both the model NPP and GPP greatly over predict the October data.

4.2.4.2 Community Respiration

Figures 4-41, 4-42, and 4-43 present model versus data comparisons for bottom water DO at near field and far field stations (located in Stellwagen Basin) for 1992, 1993, and 1994, respectively. The top two panels on each figure represent DO measurements taken at depths below 20 meters at the near field stations and below 50 meters in Stellwagen Basin. The bottom two panels represent model computations (two-day averages) at the same locations and depths at which the data were taken.

Figure 4-41 presents model versus data comparisons of bottom water DO for 1992. For the near field stations, the model reproduces the general trend in the data. The highest DO concentrations are observed and computed in the late winter/early spring. Both the model and the data show decreasing oxygen concentrations from the end of March through May, although the model does not compute as marked a decrease as observed in the data. Both the model and data show an increase in oxygen levels from May through July, although the model does not compute as large an increase as observed in the data. Both the model and data show minimum concentrations of approximately 6.5 mgO_2/L in October. In general for the near field stations, the range of model computations encompasses most of the observed data. However, the model does not reproduce the high DO concentrations observed in February, nor does it reproduce the magnitude of the DO decrease that is observed to occur between April and May. The exact cause for the observed DO decline that occurred between April and May remains unclear. Kelly (1993) attributed the sharp decline to respiration processes associated with the rapid die-off of the *Phaeocystis* bloom that occurred in the spring of 1992. However, the possibility does exist that the decline in DO could be due to advective transport of waters low in DO from the Gulf of Maine. The observed decline in DO concentrations does coincide with the freshet that occurs in late April/May. Unfortunately, there are no boundary DO data to confirm or refute this alternate hypothesis. It is also possible that processes associated with the two hypotheses acted together to result in the rapid decline in DO observed in the February through early May time period.

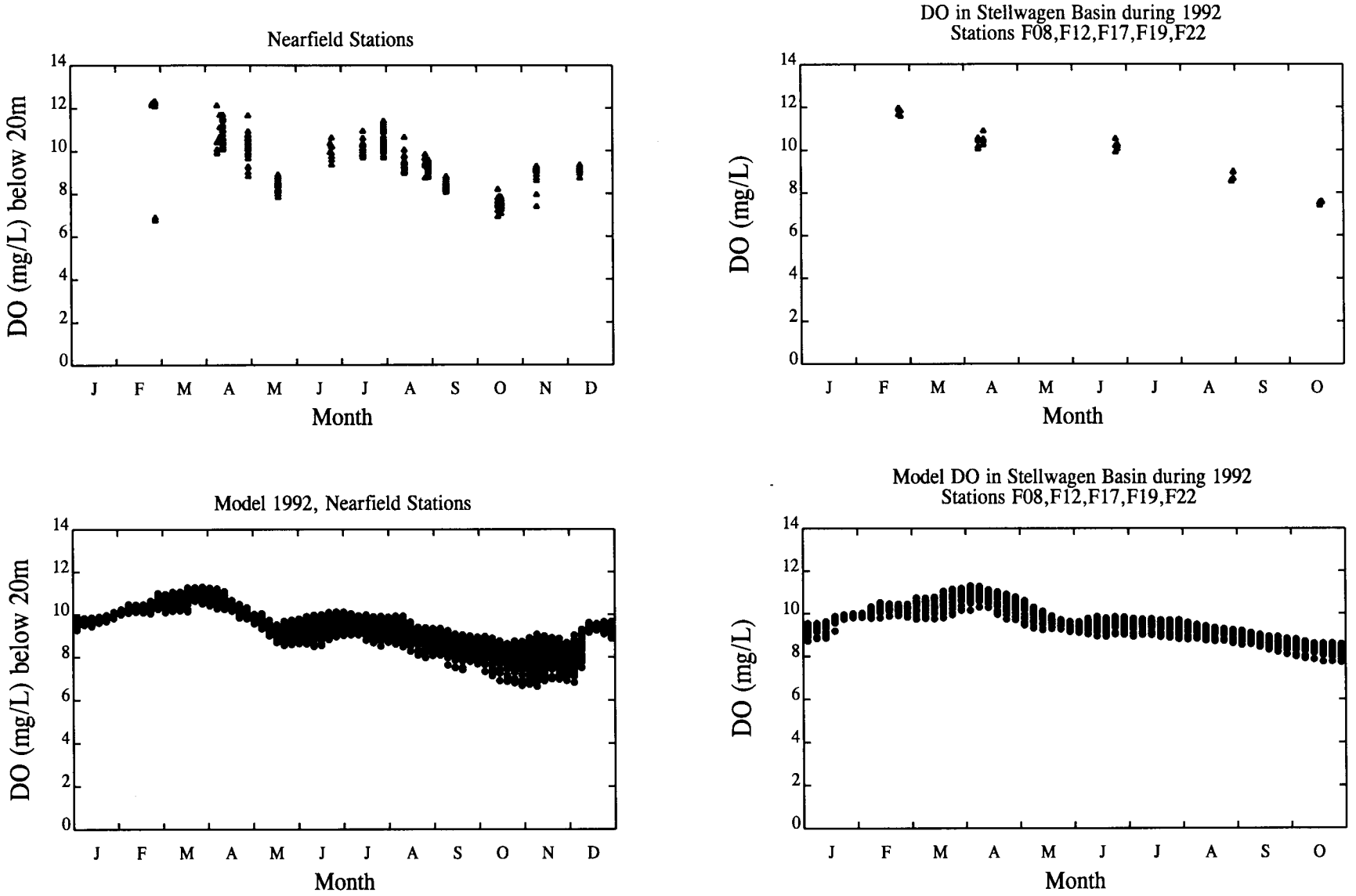


Figure 4-41. 1992 Model vs Data DO Comparisons 1992

The model also does not compute the magnitude of the increase in DO concentrations that is observed for the nearfield stations between the end of May and late July. Again this process is not fully understood. Kelly (1993) offered two possible mechanisms. First, primary production at low light levels within the pycnocline may contribute. Second, the variability in the density structure of many near field stations suggests that physical stratification may, on occasion, be partially disrupted, thus permitting the addition of oxygen to the bottom depths of the water column. However, one would expect to see changes in bottom water temperature if the water column were to destratify and this is not observed in the data. A third hypothesis which has been mentioned earlier relates the increase in DO to primary production by benthic algae. This mechanism is not currently in the model.

The model does, however, seem to compute the approximate range of DO concentrations observed in the nearfield from month to month, and does reproduce the long-term decline in DO concentrations observed between February and October. Kelly (1993) performed a regression analysis on the DO data from April to mid-October and computed a decline of 0.0136 mg O_2/L-d. The model estimates a decline of 0.0102 mg O_2/L-d during this period.

In Stellwagen Basin, the model compares reasonably well to the observed DO data. Although the model does not quite reproduce the high DO concentrations observed in February, the model does capture the range of much of the observed data. It is interesting to note that the lowest DO concentrations are not observed in Stellwagen Basin, the deepest part of the bays, but rather the lowest measured DO concentrations are observed in the near field area. The model reproduces this feature of the data set. Kelly(1993) also performed a regression analysis on the data in Stellwagen Basin and computed a decline of 0.015 mg O_2/L-d between April and mid-October. The model computes a decline of 0.0109 mg O_2/L-d.

Figure 4-42 presents model versus data comparisons for DO at the near field and Stellwagen Basin stations for 1993. The model reproduces the DO data collected at the near field stations from February to May very well. The model does not reproduce the observed increase in DO that occurs from May to July nor do the model computed DO concentrations decline as rapidly as do the data from July to September. However, the range of DO concentrations computed by the model does encompass most of the observed data collected from August through November. Kelly (1994) repeated his regression analysis for the 1993 near field data and estimated a decline of 0.0180 mg O_2/L-d for the period of April to mid-October. As a comparison, a regression on model results determined a decline of 0.0170 mg O_2/L-d.

The model to data comparison in Stellwagen Basin is more favorable. The model captures most of the range of the observed data and the slowly declining DO trend. The model does not quite compute the highest DO concentrations observed in July 1993 in Stellwagen Basin and the model DO concentrations do not decline as rapidly from July to October as do the DO data. The estimated decline in DO concentrations from data collected between April to mid-October was 0.0181 mg O_2/L-d (Kelly, 1994) while the model estimated decline was 0.0157 mg O_2/L-d.

Model versus data comparisons for DO concentrations at the near field and Stellwagen Basin stations for 1994 are presented in Figure 4-43. In general, the model does a reasonable job reproducing the observed DO concentrations at the near field stations. However, the model does not reproduce the highest DO concentrations observed in the spring nor does the model fully reproduce the lowest observed DO concentrations in the fall. The model does, however, compute a range of DO concentration which encompass most of the observed data. The May through June increase in DO concentration that was observed in 1992 and 1993 is not as evident in 1994. The lowest DO concentrations of the three years were observed in 1994. The model reproduces this feature of the three year data set well. The model also does a good job reproducing the observed DO concentrations in Stellwagen Basin during 1994. The model reproduces most of the observed range of DO concentrations in Stellwagen Basin. Kelly (1995) estimated rates of decrease in DO concentrations between April and mid-October in the near field area and Stellwagen Basin were 0.027 mg O_2/L-d and 0.022 mg O_2/L-d, respectively. The same analysis performed for model computations resulted in estimates of 0.0224 mg O_2/L-d and 0.0218 mg O_2/L-d for the near field area and Stellwagen Basin, respectively.

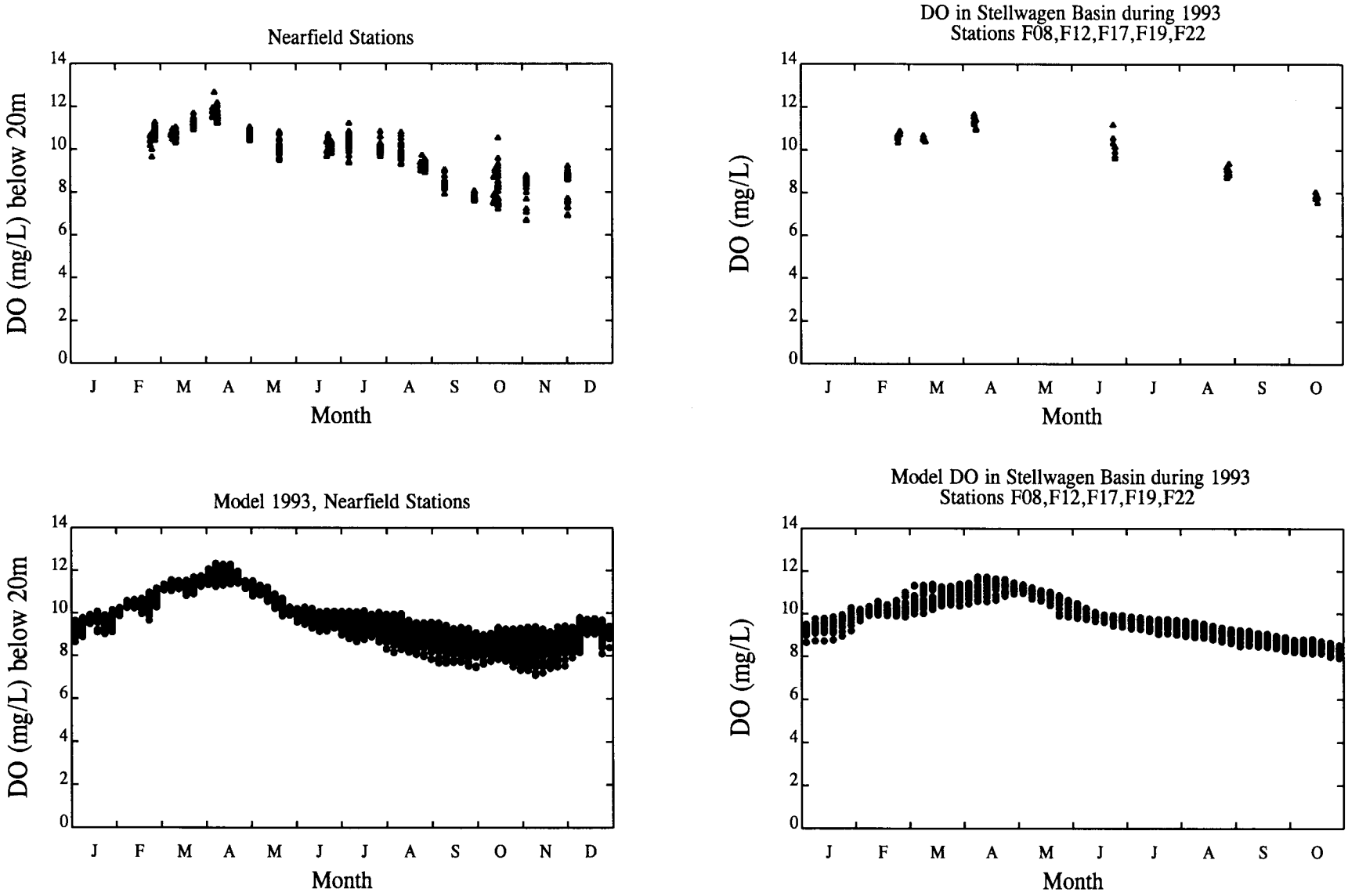


Figure 4-42. Model vs Data DO Comparisons 1993

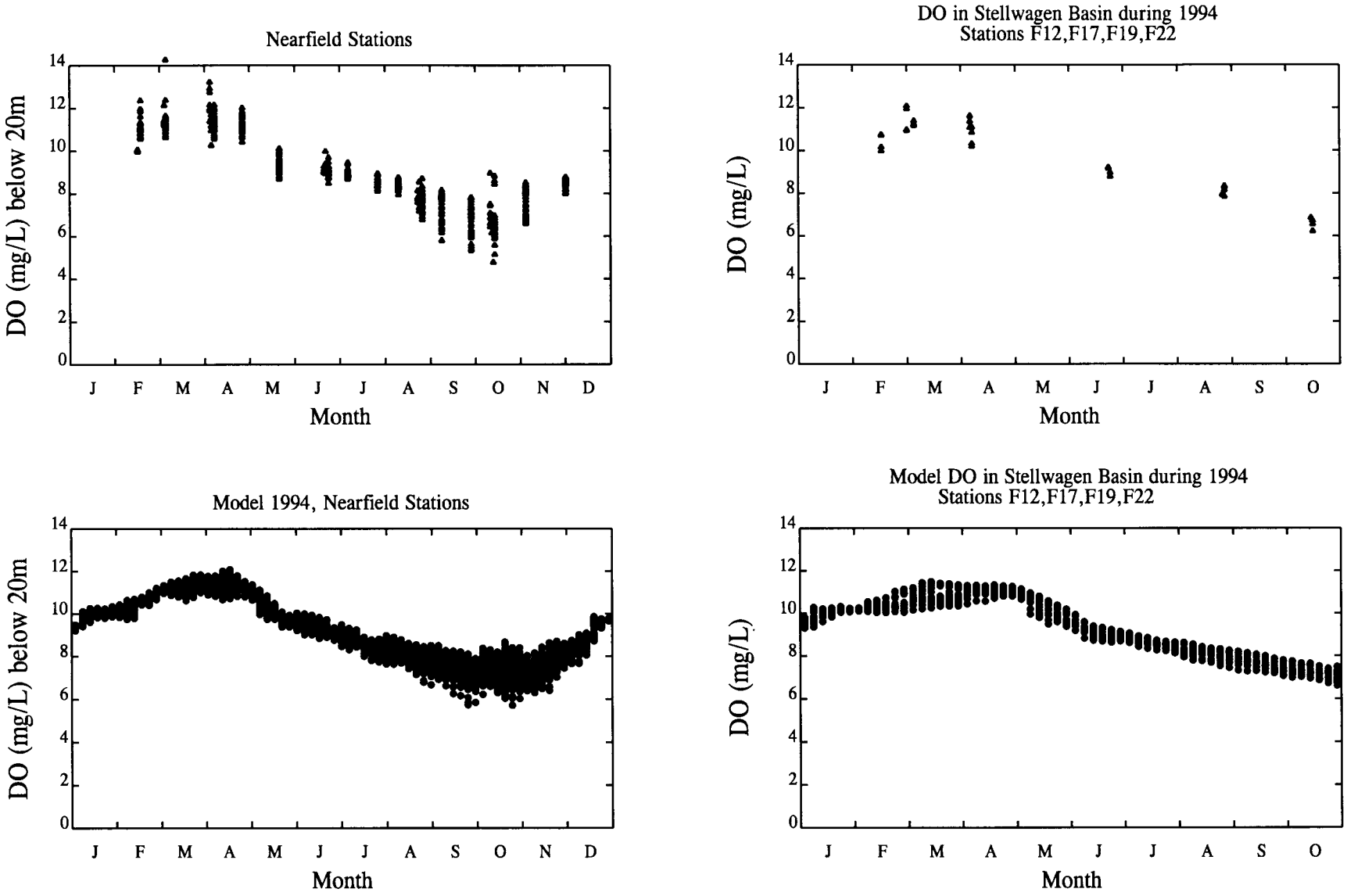


Figure 4-43. 1994 Model vs Data DO Comparisons

4.3 SEDIMENT MODEL RESULTS

This section presents comparisons between observed sediment flux data and model computations for 1992-94. The model results are compared to the data collected within the particular year being analyzed. The model versus data comparisons are shown for two model segments in Boston Harbor, a model segment located near the future outfall site, and a segment located in Stellwagen Basin. Each sediment figure contains a map showing the locations of the stations being presented.

Figure 4-44 presents model versus data comparisons of sediment oxygen demand (SOD) for 1992 through 1994. Panel 1 shows the SOD for a station (BH03) in the northern portion of Boston Harbor. Except for a high SOD recorded in May of 1993, the model provides a favorable comparison to the observed data. During the winter, when water temperatures are at their lowest, biological activity in the sediment bed is also at its lowest level. Consequently, sediment oxygen demand is low, 0.3 gm O_2/m^2-day, during this time. As the water column warms, primary productivity increases and as a result organic carbon deposition increases. In addition, biological activity in the sediment increases. These processes interact together to contribute to an increase in sediment oxygen demand. The model computes its maximum rates of SOD in July 1994 of approximately 2.75 gm O_2/m^2-day, during which time bottom water temperatures are near their highest levels in Boston Harbor. The model generally compares favorably with the highest observed summertime SOD in each year. As the water temperature begins to decrease, both the model and data show a decrease in SOD.

Panel 2 shows the sediment model versus data comparison for a station located in Hingham Bay, in the southern portion of Boston Harbor. The model does not compare as favorably to the data in this segment. The model and data show some evidence of a seasonal cycle, but the model overpredicts the SOD by as much as 1.5 gm O_2/m^2-day during portions of the year. One reason for the lack of agreement between model and data at this station may be due to an artifact of the data. The data presented in Panel 2 of Figure 4-44 (and Figures 4-45 through 4-49) are from BH08. This site was specifically selected by Anne Giblin to represent a low deposition sand environment within the harbor. This was done so as to partially bound the range of observed fluxes and to allow the construction of Harbor-wide flux estimates given the known distribution of sediment types within the Harbor. However, the majority of the sediment bed in Hingham Bay is comprised of depositional muds (Knebel, 1993). Therefore, data from this site may not be representative of the fluxes to

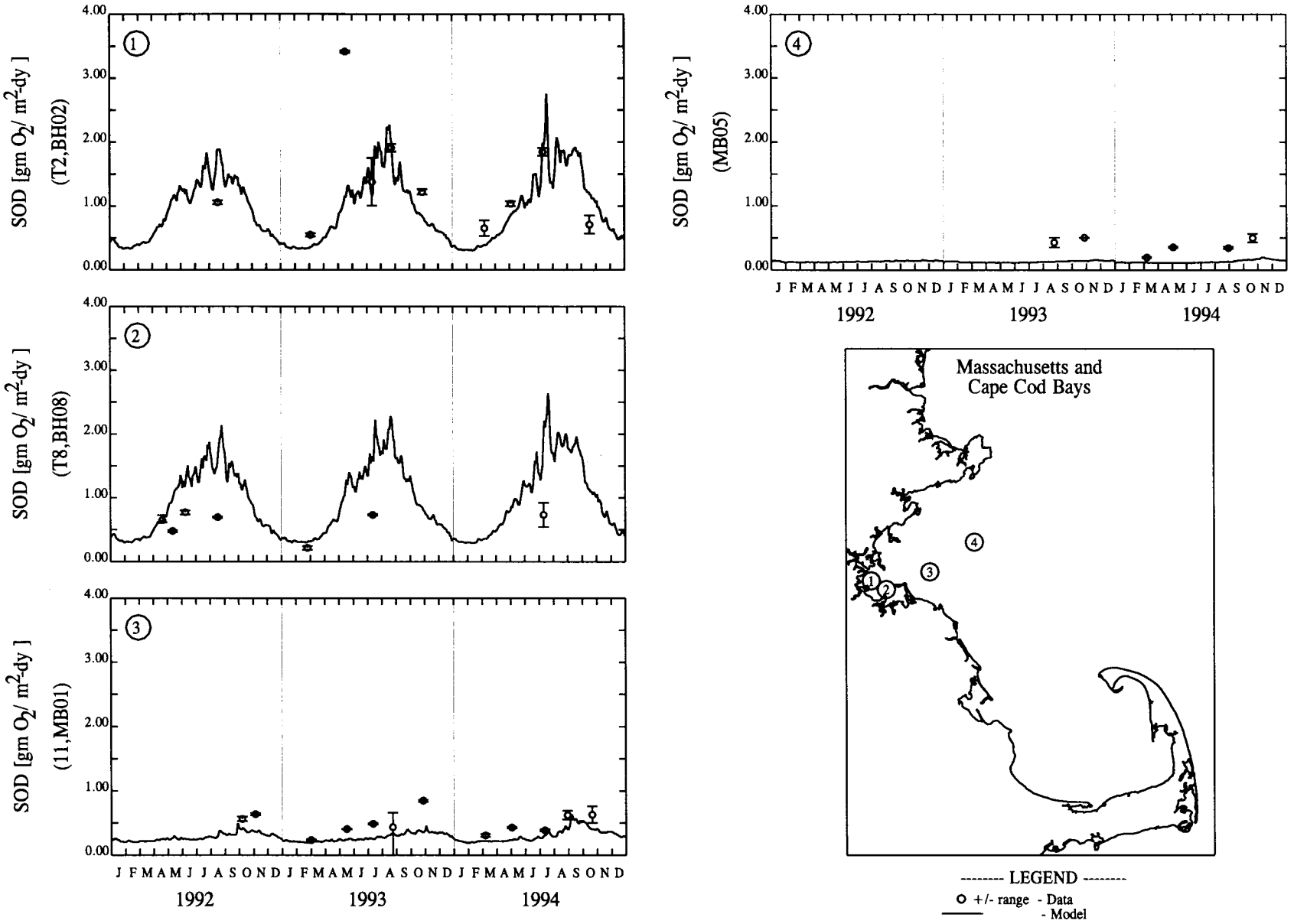


Figure 4-44. Model vs Data Temporal Comparisons for SOD (1992 - 1994)

be found in Hingham Bay. In fact, this site was abandoned after 1994 and flux measurement made at a nearby depositional mud station between 1995 and 1997 are more consistent with the fluxes measured at BH03 (Ken Keay, MWRA, pers. comm.) and, therefore, provide a more favorable comparison between model and data.

Panel 3 presents calibration results for a station located near the future outfall site. The SOD measurements at this location are slightly lower than observed in Hingham Bay during May through August, averaging about 0.6 gm O_2/m^2-day. For 1993, the model computes slightly lower rates than were observed but compares more favorably to the data for 1994. One reason that the model may be computing low at this site also may be due to a sampling artifact. Giblin (1995) has indicated that all of the sites chosen for nutrient flux sampling in the nearfield region of Massachusetts Bay were depositional and were probably accumulating and concentrating organic material. There were other locations in the immediate vicinity of these stations, however, which were found to be non-depositional. The model segments, which are a number of square kilometers in size, encompass both depositional and non-depositional bottom segments. Since the model spreads the organic matter over the entire are of the sediment bed within the model cell, it is not surprising that the observations of SOD are slightly underestimated. The model does, however, reproduce the feature observed in the data wherein higher SODs are observed in the fall as compared to the summer. This is probably due to the fact that the deeper bay waters reach their maximum temperatures in the fall rather than the summer. The model computes the highest SODs in 1994 when the highest bottom water temperatures were observed. Calibration results for the Stellwagen Basin segment are presented in Panel 4. At this station the observed SODs are slightly lower than observed at the future outfall site. This may be, in part, due to the fact that this station is located in deeper waters, and therefore, it is likely that a greater portion of the particulate organic carbon will be oxidized in the water column before it reaches the sediment. In addition, this portion of the bay is more nutrient limited, so there may be less production of organic matter in the surface waters. The observed data at this location average about 0.39 gm O_2/m^2-day, while the model computes an SOD rate of approximately 0.13 gm O_2/m^2-day. What is encouraging, however, is that the model does reproduce the marked differences in SOD between Boston Harbor and the rest of Massachusetts Bay.

Computed and observed ammonium fluxes (J_{NH4}) are presented in Figure 4-45 for the same four model segments for 1992 through 1994. Comparisons between observed and computed NH_4 fluxes show a similar spatial pattern to that observed for SOD. There is a favorable comparison between the model and data in northern Boston Harbor. The highest observed NH_4 fluxes and model computed fluxes are observed in northern Boston Harbor. Both the model and data show similar temporal patterns from year

to year. In southern Boston Harbor, the model over estimates the NH_4 flux. Lower NH_4 fluxes are observed at the future outfall site and in Stellwagen Basin than in Boston Harbor and the model reproduces the declining spatial trend very well. The observed NH_4 data and flux model computations appear to be sensitive to the water column temperature as was observed for the SOD.

Computed and observed nitrate fluxes (J_{NO3}) for 1992 through 1994 are presented in Figure 4-46. The two model segments shown for Boston Harbor provide an interesting contrast to the SOD and ammonia flux model/data comparisons. The model over estimates much of the winter and early spring nitrate flux data at the northern Boston Harbor segment, while model-data comparisons for the southern Boston Harbor station are quite good. The model does a fairly good job reproducing the observed NO_3 fluxes at the future outfall site, but in general, the model slightly under estimates the NO_3 fluxes in the first half of the year and slightly over estimates the NO_3 fluxes during the latter portion of the year. The NO_3 flux data at the future outfall site do not appear to be as strongly correlated to bottom water temperatures as the SOD rate or the NH_4 fluxes. The model, however, displays the same temperature signal as was observed for SOD and the NH_4 flux. In Stellwagen Basin the model computes a small NO_3 flux into the sediment while the data indicate there was a small NO_3 flux out of the sediment when measurements were taken in 1993 and 1994.

Figure 4-47 presents the model versus date comparisons for rates of sediment denitrification for 1992 through 1994. The model estimates of sediment denitrification compare favorably to the observed data with the exception of the northern Boston Harbor station, especially in 1993. However, it should be noted that there are also differences between the two denitrification data sets (Giblin et. al., 1992, 1993, 1995 versus Nowicki, 1994) in northern Boston Harbor. Overall there is little seasonal pattern to be observed in the data at any of the stations and the model goes through many of the data points at the future outfall site and in Stellwagen Basin. There is some suggestion of higher denitrification rates during the warmer months in Boston Harbor; however, the model results are counter to this observation.

The 1992 through 1994 model versus data comparisons for inorganic phosphorus fluxes (J_{PO4}) are presented in Figure 4-48. Excluding one exceptionally high rate observed in Boston Harbor in May of 1993, the data indicate the PO_4 flux rates to be correlated to temperature. The highest rates in Boston Harbor are observed during the summer, and in Massachusetts Bay the highest PO_4 flux rates are observed in the fall. The model reproduces this temporal pattern. Spatially the model results are similar to those for

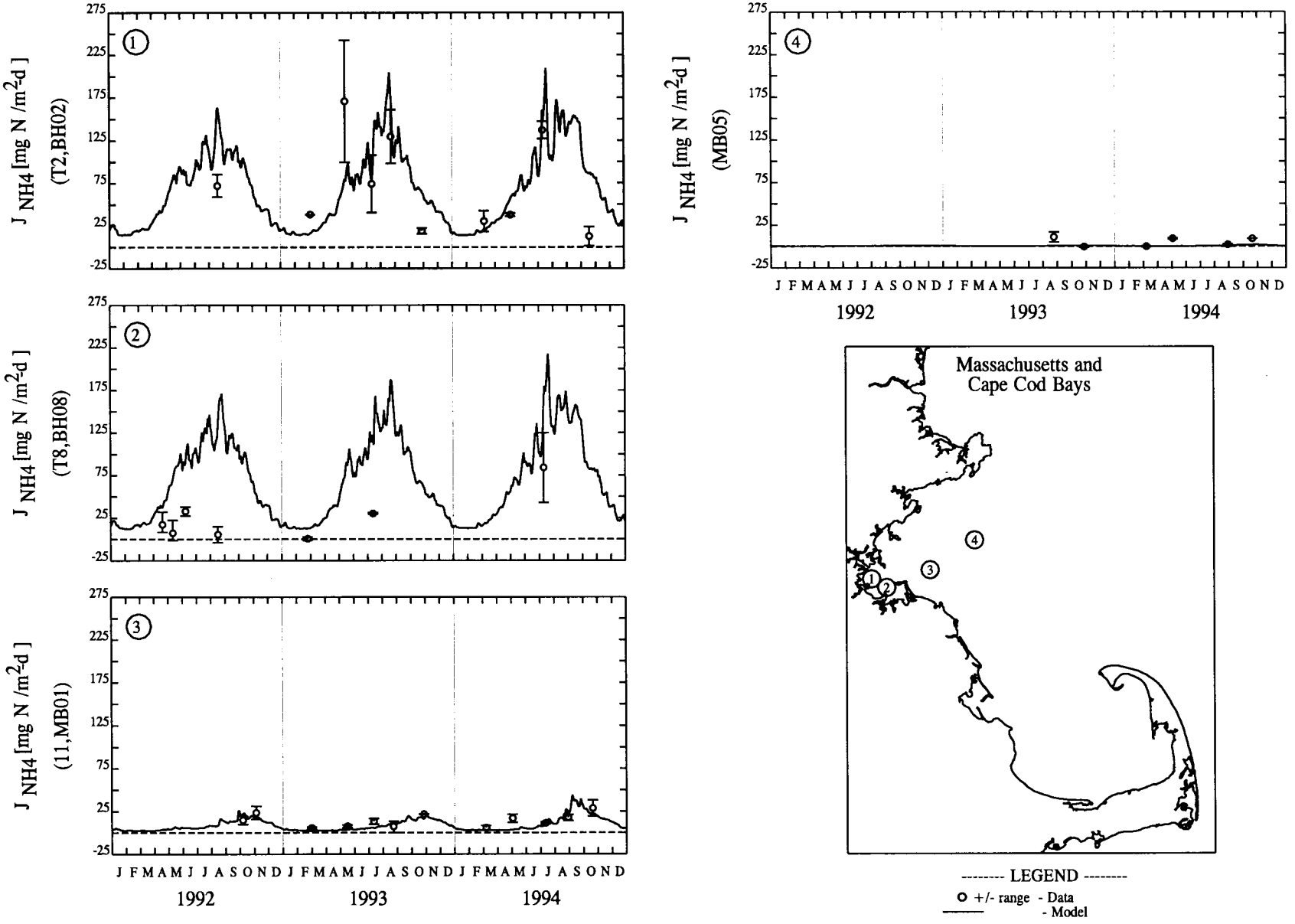


Figure 4-45. Model vs Data Temporal Comparison for Ammonia Flux (1992 - 1994)

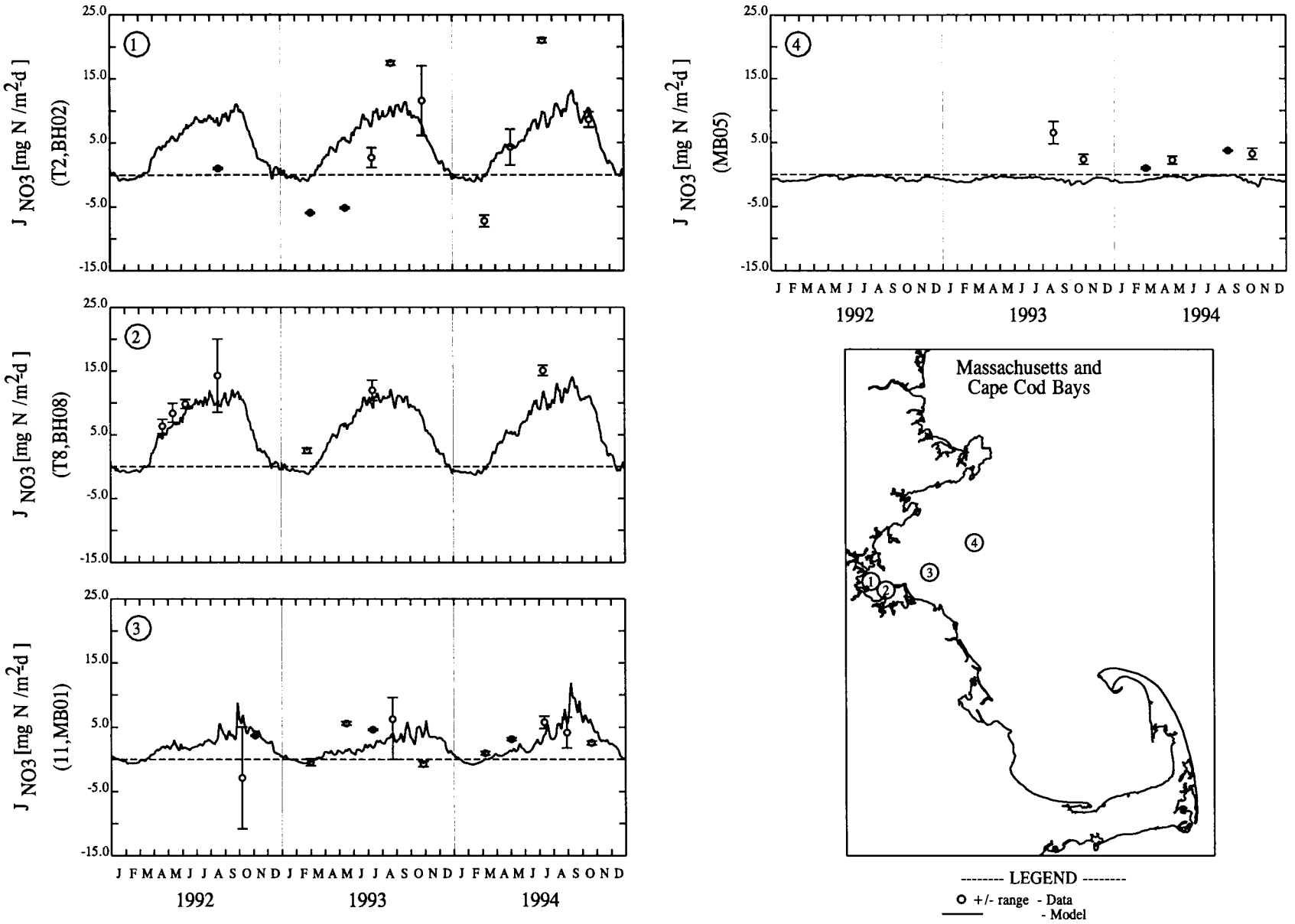


Figure 4-46. Model vs Data Temporal Comparison for Nitrate Flux (1992 - 1994)

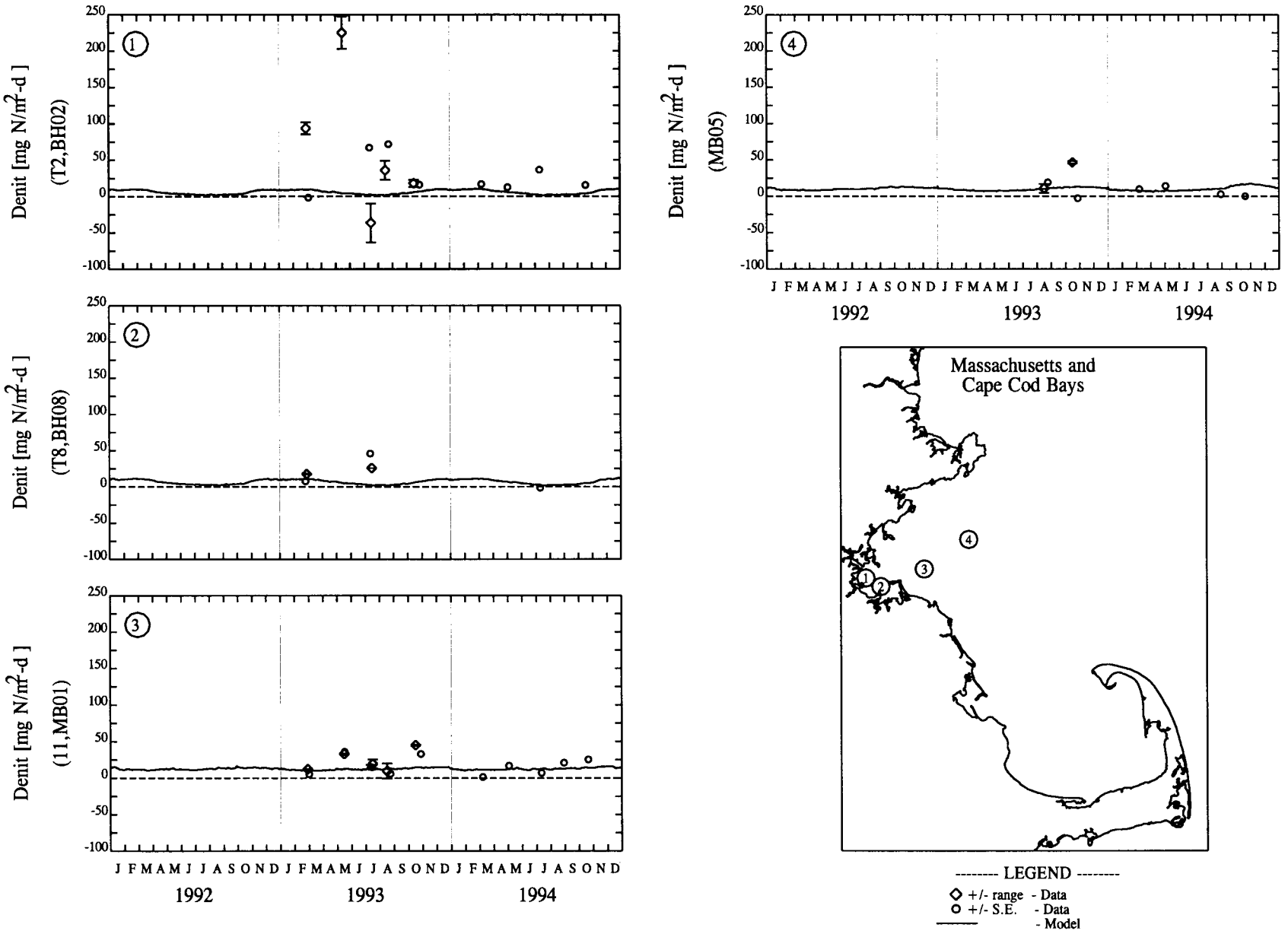


Figure 4-47. Model vs Data Comparison for Denitrification (1992 - 1994)

SOD and J_{NH4}. The model does a fairly good job reproducing the fluxes in northern Boston Harbor, while over predicting the flux rates in southern Boston Harbor. At the future outfall site and the station in Stellwagen Basin the PO_4 flux rates are quite a bit lower than in Boston Harbor and the model reproduces these rates very well.

Comparisons of the model versus data for silica fluxes (J_{Si}) are displayed in Figure 4-49 for 1992 through 1994. For the northern Boston Harbor segment, the model compares somewhat favorably to the data except for the high rate measured in May of 1993 and the extremely high rate observed in July of 1994. The model over predicts the July 1993 measurement in Hingham Bay. Unlike the other fluxes measurements that were made, the silica flux does not appear to have the same spatial trend. Silica fluxes in Massachusetts Bay are as high or higher than those observed in Boston Harbor. The model significantly under predicts the silica fluxes outside of Boston Harbor. During this analysis an attempt was made to improve the sediment sub-model calibration of silica flux by increasing the detrital source of silica to the sediment. While this action increased the computed silica fluxes, it did not do so to the degree required to fit the silica flux data. More data on the pore water and solid phase composition of the sediment may help in the understanding of the model's shortcoming of computed fluxes. However, the underestimation of silica fluxes does not appear to have a significant adverse impact on the overall water quality model comparison to data.

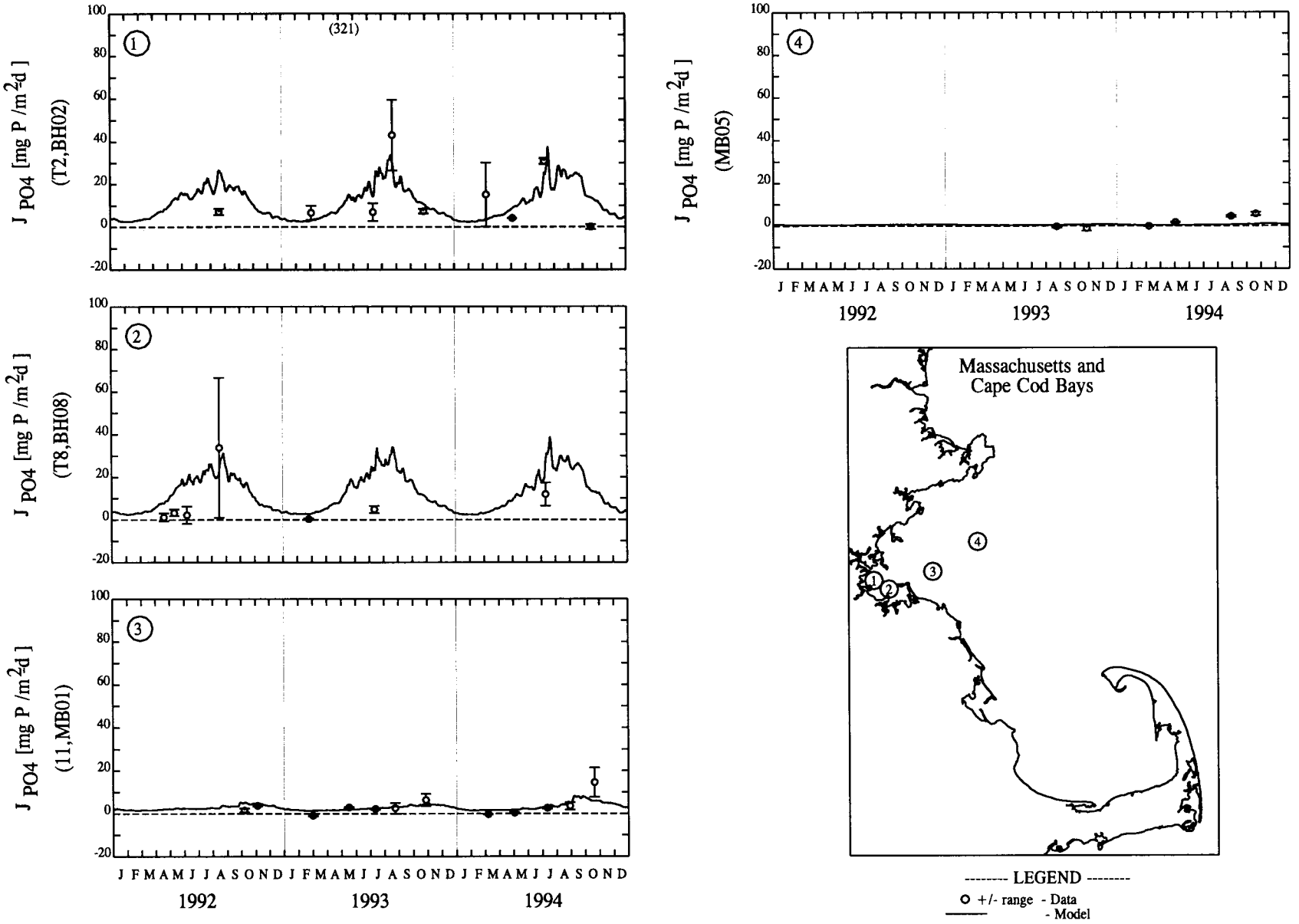


Figure 4-48. Model vs Data Comparison for Phosphate Flux (1992-1994)

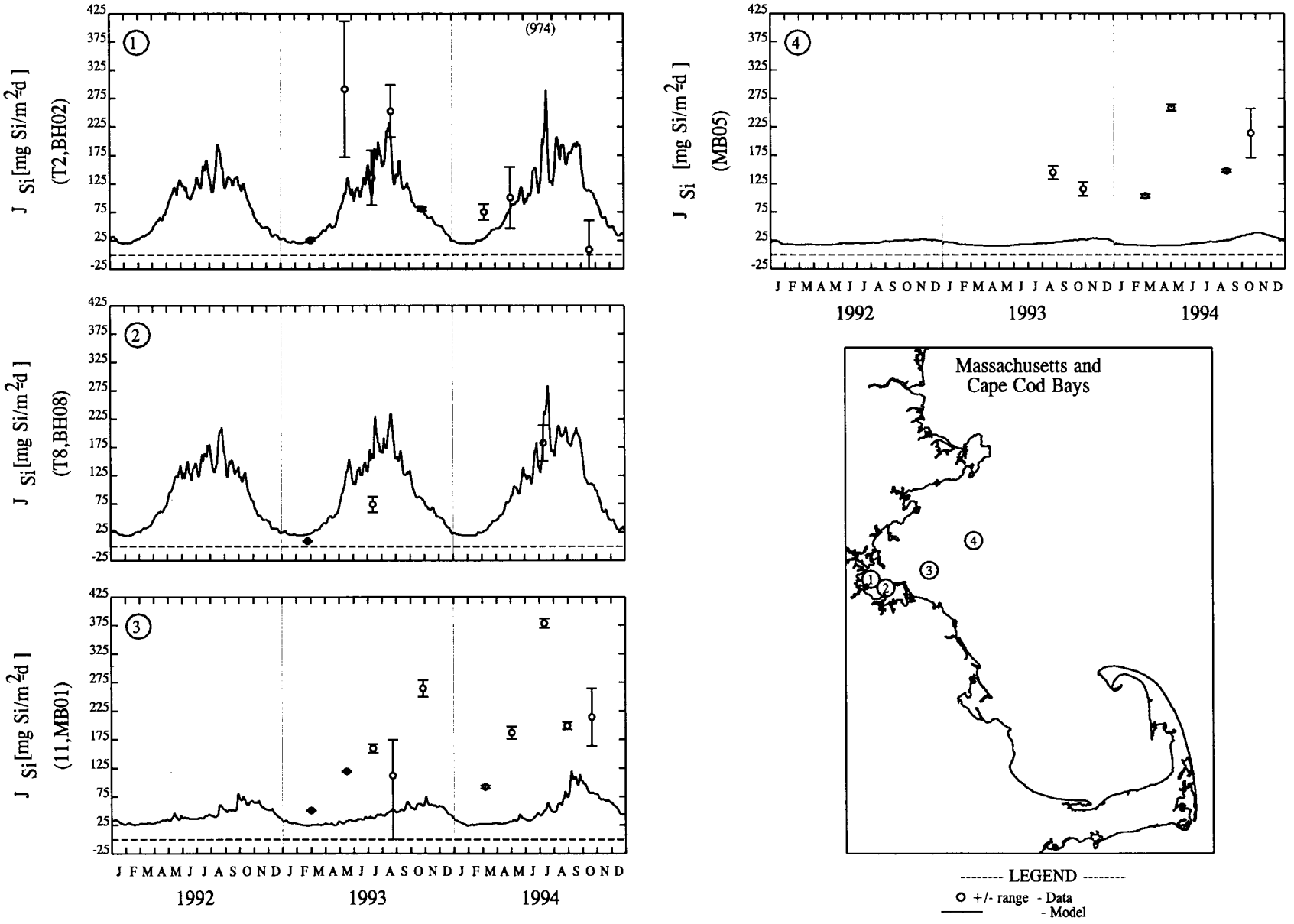


Figure 4-49. Model vs Data Temporal Comparison for Silica Flux (1992-1994)

SECTION 5

OVERALL ASSESSMENT OF MODEL PERFORMANCE

5.1 INTRODUCTION

MWRA baseline monitoring has shown that there are consistent seasonal trends in water quality parameters and, as discussed in this report, the Bays Eutrophication Model (BEM) represents most of these trends very well. However, as noted in numerous MWRA reports (e.g. Kelly and Turner, 1995), there are several important water quality parameters that vary substantially from year-to-year. It is these extremes of variability that allow for an examination of the limits of predictability of the model. In this section, we shall evaluate the inability of the BEM to reproduce the unusual water quality characteristics that were observed in 1993 and 1994 (a strong fall algal bloom and low bottom water dissolved oxygen concentrations, respectively) and provide a comprehensive explanation as to the reason these events were missed by the model.

5.2 1993 FALL BLOOM OF *ASTERIONELLOPSIS GLACIALIS*

The major water quality event of 1993 was the bay-wide bloom of the diatom *Asterionellopsis glacialis* in late August through October. The model computations for chlorophyll concentrations for 1993 captured the winter-spring bloom at Cape Cod Bay stations (F01 & F02) and were generally in agreement with the monitoring data, with the exception of the high chlorophyll concentrations associated with the fall bloom (Figure 5-1). Along the western edge of the nearfield area (stations N01 and N10), chlorophyll concentrations were underestimated for most of the summer and early fall. It is the inability of the model to reproduce this water quality event that is examined below.

High productivity, high chlorophyll, and high cell counts usually are concomitant with high biomass concentrations, but a review of the water quality data does not show evidence of higher than normal particulate organic carbon (POC) or nitrogen (PON). As shown in Figure 5-2, POC concentrations were elevated during the winter-spring bloom in Cape Cod Bay, but in comparison to the elevated chlorophyll observed in October there is relatively little change in the measured POC concentrations going from spring

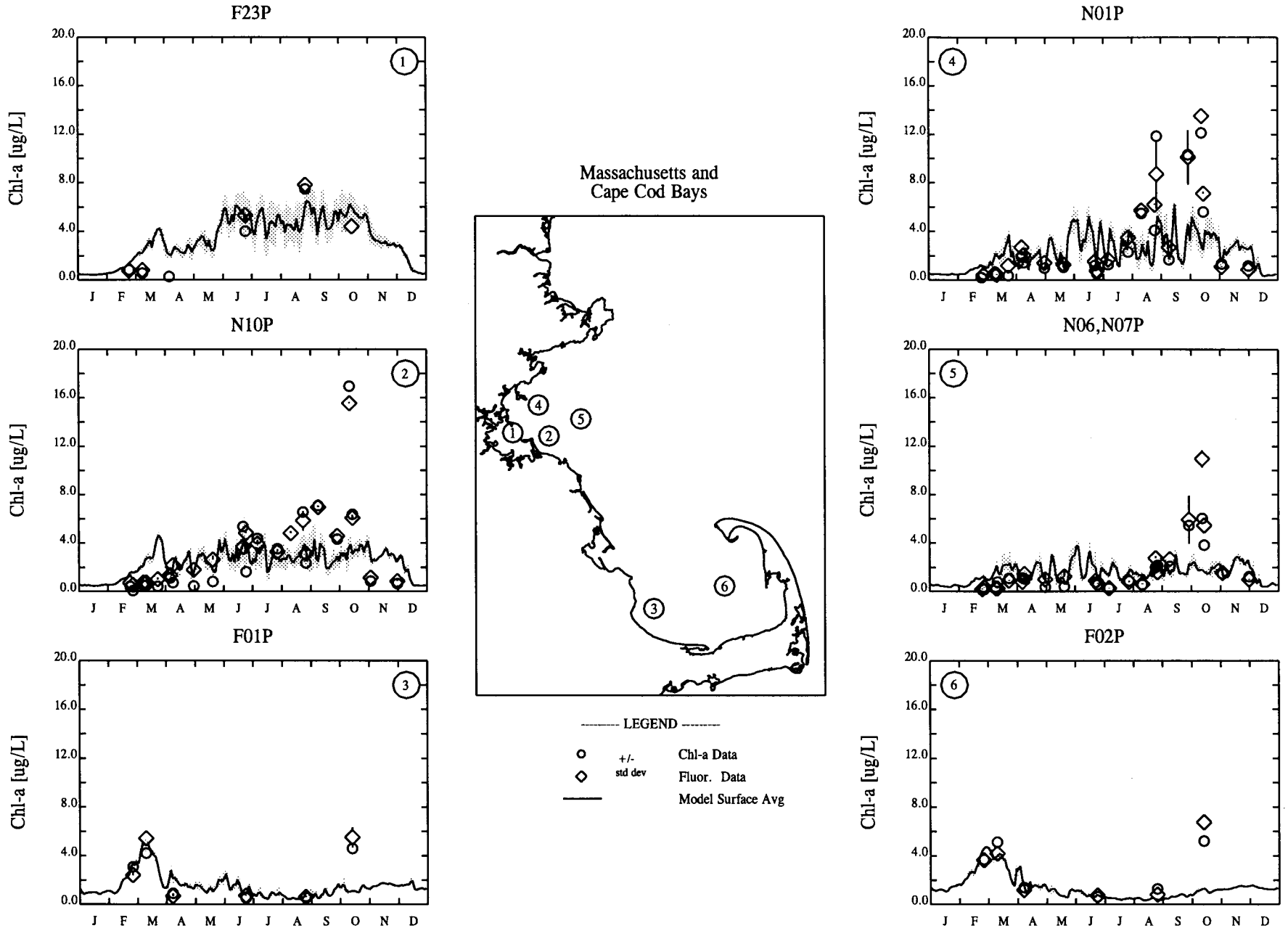


Figure 5-1. 1993 Temporal Calibration Results for Surface Chlorophyll-a

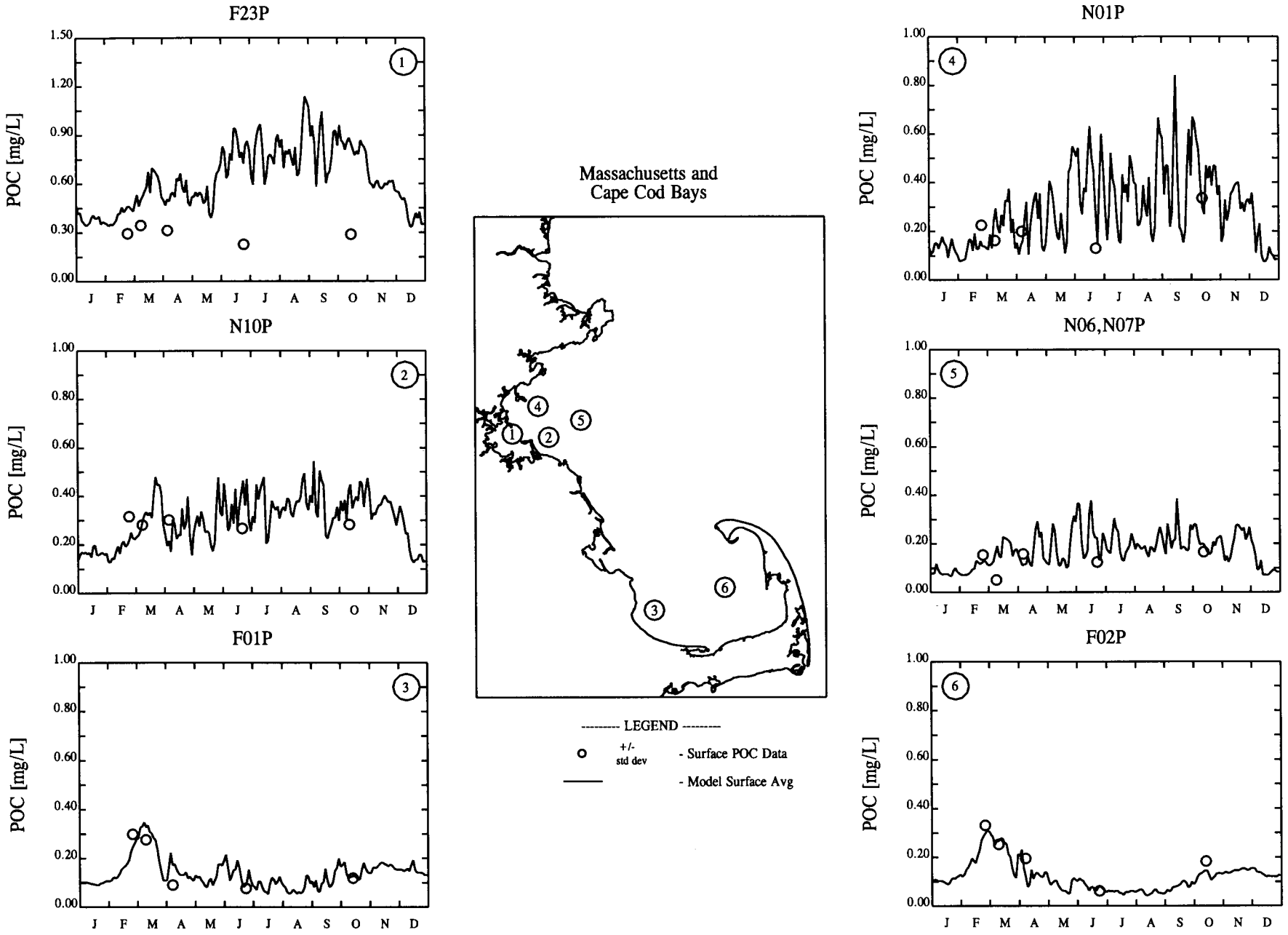


Figure 5-2. POC 1993 Temporal Calibration Results

through summer to fall. The exception is station N10P, which shows an increase of approximately 0.1 mg C/L over spring/summer values. Given this contradiction in the biomass parameters, it is possible to argue that even though the model did not capture the fall *A. glacialis* bloom event it was able to accurately represent the fall trends in carbon for this system. The questions that need to be addressed are:

- Why is there an apparent contradiction in the various biomass parameters that were measured in October 1993?
- Why didn't the model capture the fall bloom of 1993?
- Is missing the bloom (chlorophyll) a problem if the model is able to accurately reproduce trends in the other important parameters (carbon, dissolved oxygen, etc.)?

At all Massachusetts Bay Bioproductivity stations the fall *A. glacialis* bloom was dominant at both surface and chlorophyll-maximum depths. As presented in Table 5-1, the elevated chlorophyll concentrations were coincident with high phytoplankton counts (1 to 6.5 million *A. glacialis* cells/L) and very high production rates (4-8 g $C/m^2/d$ at the nearfield stations). POC concentrations ranged from 0.10 to 0.45 mg C/L at the Bioproductivity stations. In Figure 5-2, it appears that there is only a slight increase in POC during October in comparison to the large increase in measured chlorophyll concentrations. Though the concentration of POC may be relatively uniform, the constituents that makeup the POC were changing from summer to fall conditions. The mixed phytoplankton assemblage (microflagellates and cryptomonads dominating) and relatively high C/Chla ratios, due to both species characteristics and high light intensity, were being replaced by a diatom monoculture with higher per cell chlorophyll concentrations due to decreasing irradiance.

To evaluate this further, we have calculated an estimate of phytoplankton carbon (PPC) based on the cell counts of *A. glacialis* and a per cell carbon equivalence of 73.09 pg/cell (Lacouture, 1999). The estimated PPC was then compared to the measured POC to evaluate what percentage of the total POC was comprised of carbon from the bloom species. At the nearfield and coastal stations, PPC represented 60% to 160% of the POC. Though the errors involved in this type of estimation may be large, the evaluation suggests that the POC concentrations that were measured are reasonable for the chlorophyll conditions that were observed during the *A. glacialis* bloom in October 1993.

Table 5-1. Phytoplankton, Chlorophyll, Organic Carbon, and Productivity Data Collected at BioProductivity Stations October 1993. (Phytoplankton Carbon (PPC) was calculated as #cells X 73.09 pg/cell)

Station	Depth (m)	# cells A. glacialis	Chla (ug/L)	Production (mg C/M^2-d)	POC (mg C/L)	PPC (mg/L)	%PPC of POC	C/Chla Ratio
F01	2.35	86000	4.61	2324	0.12	0.00	0%	26
F01	6.83	61000	4.46	2002	0.10	0.00	0%	23
F02	2.29	853000	5.23	2661	0.18	0.06	34%	35
F02	13.54	808000	5.20	2194	0.17	0.06	35%	32
F13	1.59	4835000	11.51	2384	0.28	0.35	128%	24
F13	11.09	3783000	11.28	2895	0.27	0.28	102%	24
F23	2.34	1239000	4.82	2129	0.29	0.09	31%	60
F23	10.93	1629000	4.90	2929	0.33	0.12	36%	67
N01	0.88	5775000	12.12	7973	0.34	0.42	126%	28
N01	13.32	6072000	13.41	7535	0.30	0.44	149%	22
N04	2.5	4674000	9.14	4308	0.30	0.34	114%	33
N04	11.92	3669000	8.00	4227	0.37	0.27	72%	46
N07	2.14	1476000	5.91	4715	0.17	0.11	63%	29
N07	8.7	1669000	4.34	6354	0.20	0.12	60%	47
N10	1.05	5775000	17.67	4555	0.28	0.42	149%	16
N10	8.96	5249000	16.01	5008	0.32	0.38	119%	20
N16	1.16	2766000	6.82	5356	0.24	0.20	84%	35
N16	19.21	5133000	12.45	5875	0.23	0.38	164%	18
N20	1.45	6505000	17.76	3795	0.37	0.48	129%	21
N20	16.66	6398000	20.37	4743	0.45	0.47	105%	22

In one sense it is relatively easy to say why the model did not reproduce the fall *A. glacialis* bloom and that is the fact that the model was not designed to reproduce monoculture or monospecific blooms. The question to answer before any water quality model can simulate the growth of a specific algal species is, "What causes (or caused) a specific species to bloom?" The complexity of the nutrient, biological and physical factors that contribute to the ability of a specific phytoplankton species to outcompete other species and achieve bloom concentrations has been the focus of research for many years. The scientific community has yet to adequately characterize these factors and their complex interactions, so we cannot expect a model to fully capture these ecological responses, especially when they do not occur with regularity, e.g., the *A. glacialis* bloom occurred in the fall of 1993 but not 1992 or 1994. This said, there are a number of biological factors that can be examined to determine if adjustments in the relationships and/or assumptions within the model might result in more accurate representation of the monitoring data for October 1993.

A change in the top-down control of phytoplankton has been suggested as a potential factor in the occurrence of the *A. glacialis* bloom. A comparison of 1992 and 1993 data indicated that the general zooplankton trends and species were similar between the two years except for much higher zooplankton counts in August 1992. Elevated zooplankton numbers in August (and potentially September though no data on plankton were collected during this month) may have resulted in higher grazing pressure and lower phytoplankton counts from August through the fall months in 1992. The lower concentrations of zooplankton in August 1993 and the concomitant lower grazing pressure could have been a factor in the initiation and development of the *A. glacialis* bloom. By adjusting the interaction between grazing pressure and phytoplankton growth, the model might be able to more accurately track the development of fall blooms. While this may be a reasonable approach, i.e., to treat the effects of zooplankton grazing as a forcing function with year to year variability, it begs the question, "Why were the 1993 zooplankton levels lower than 1992?"

The phytoplankton assemblages in the model are broken down into a summer "mixed assemblage" and a winter "diatom assemblage". There are a number of important rate constants that vary according to which assemblage is being used: temperature optimum, carbon to chlorophyll-a (C/Chl-a) ratio, and carbon/silicate (C/BSi) ratio. By varying the assemblage used and the associated rate constants, the choice of mixed or diatom assemblages effects not only the potential production rate, but also the chlorophyll and silicate concentrations (along with other parameters that have relationships that are more deeply imbedded in the model). In the model, the summer assemblage is used during the summer/fall months (June-Aug) and the winter assemblage during winter/spring months (Sept-May). The temperature optimum for summer and winter assemblages are 18° and 8° C, respectively. In October 1993, surface water temperatures were 11±1°C throughout the Bays. Despite introducing the winter diatom group into the model in September, the winter group was not able to achieve the biomass levels of the fall *A. glacialis* bloom as suggested by the chlorophyll and dissolved silica data. In part this is due to the fact that the BEM winter diatom group represents a generic diatom population and not a specific monoculture. In order to capture the fall bloom, it would probably be necessary to add a third phytoplankton species to the model.

Another factor contributing to under estimating the 1993 fall algal bloom is the carbon to chlorophyll ratio of the *A. glacialis*. The C/Chl-a ratio used by the model for the summer and winter assemblages are 65 and 40 mg C/mg Chl-a, respectively. In calculating chlorophyll concentrations, the underestimation of phytoplankton carbon for October 1993 is compounded by the overestimate of the C/Chl-a ratio. As a simplistic example, we used a value of 0.3 mg C/L for the fall phytoplankton biomass based on observed

data at stations N01P and N10P. By using the associated C/Chl-a ratio, we calculated chlorophyll concentrations of 4.6 Fg Chl-a/L using the summer assemblage condition and 7.5 Fg Chl-a/L using the winter assemblage condition. Both of these values are below the maximum chlorophyll concentrations of 12-16 Fg/L observed at these stations.

To take this example one step further, the C/Chl-a ratio of 40 used for the winter diatom assemblage might be too high for the particular conditions observed in October 1993. Figure 5-3 presents a comparison of the C/Chl-a ratio and *A. glacialis* cell counts at the BioProductivity stations (the C/Chl-a ratios are also listed in Table 5-1). At the stations where the bloom was strongest (>5.0E+06), the C/Chl-a ratio was approximately 20. If we use this ratio, we calculate a chlorophyll concentration of 15 ug Chl-a/L which is more consistent with the data. Once again the calculation is simplistic, but it does provide a possible explanation of how the model may have missed the bloom in chlorophyll. It also suggests that chlorophyll may not be the best indicator of algal biomass, since the chlorophyll content of phytoplankton may vary as a function of species, as well as light and nutrient status.

A similar case can be made as to why the model did not accurately reproduce the draw down of dissolved silica (DSi) that was observed in the monitoring data (Figure 5-4). The C/BSi ratios used by the model for the summer and winter assemblages are 7.0 and 2.5 mg C/ mg Si. The fact that the model computes a fall phytoplankton population that is largely comprised of the summer mixed assemblage rather than the winter diatom assemblage means that silica uptake is under estimated by the model.

One of the goals of the BEM is to be able to forecast potential problems and this is accomplished by evaluating extreme conditions for specific indicators (DO, chlorophyll, etc.). The inability of the model to produce the fall *A. glacialis* bloom of October 1993 does not seriously effect this goal. First, while comparisons of model computations to observed chlorophyll and dissolved silica data suggest that the model missed the fall bloom model, data comparisons for POC (Figure 5-2) suggest that the model is predicting algal biomass, as measured by organic carbon, reasonably well. Second, since the bloom occurred in the fall just prior to the seasonal overturn of the water column, it did not appear to strongly effect bottom water DO concentrations. During the winter-spring bloom, the increase in phytoplankton biomass eventually leads to an increase in bottom water respiration and a decrease in DO concentrations. If the model were to miss a strong winter-spring bloom, it would be difficult to have any confidence in the model's representation of the summer DO concentrations. However, in this case, both the model and monitoring data support the notion that the fall bloom (or lack thereof) had little effect on bottom water DO concentrations.

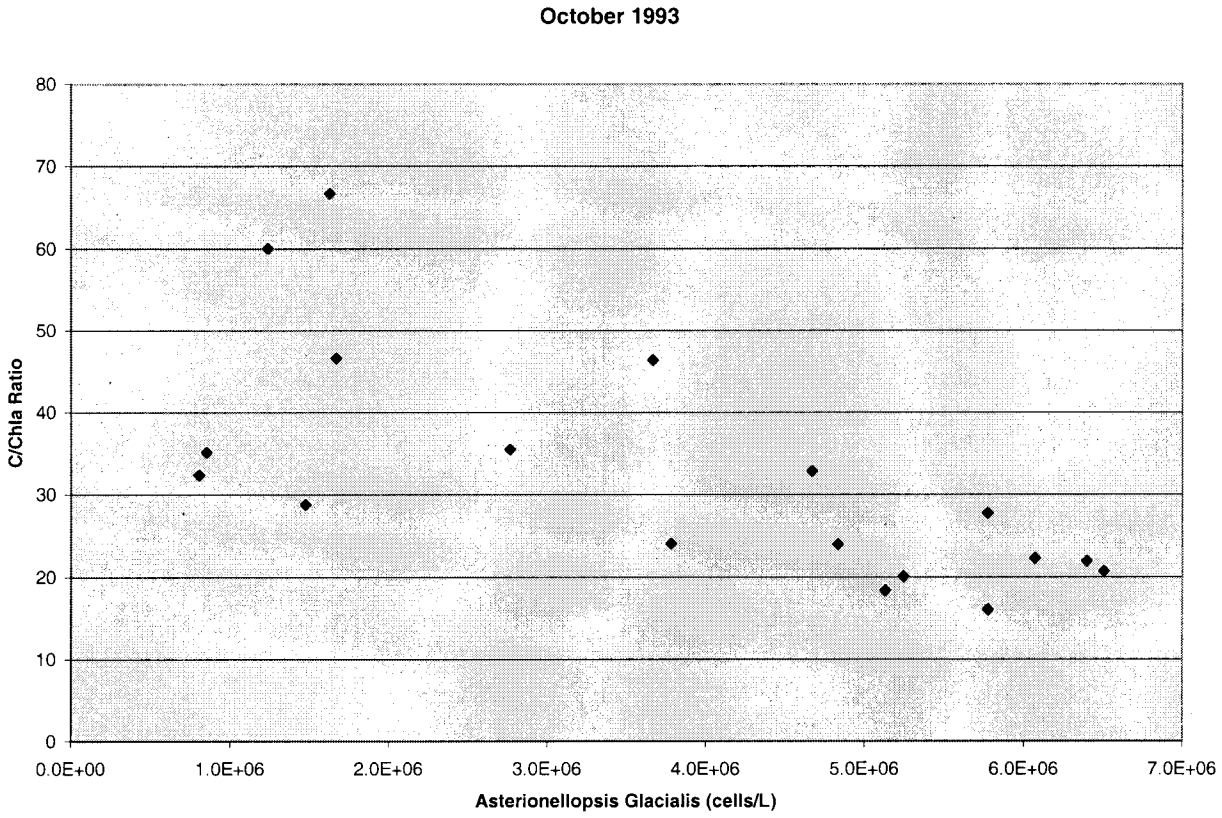


Figure 5-3. C/Chlorophyll-a Ratio vs. *A. glacialis* concentration for October 1993

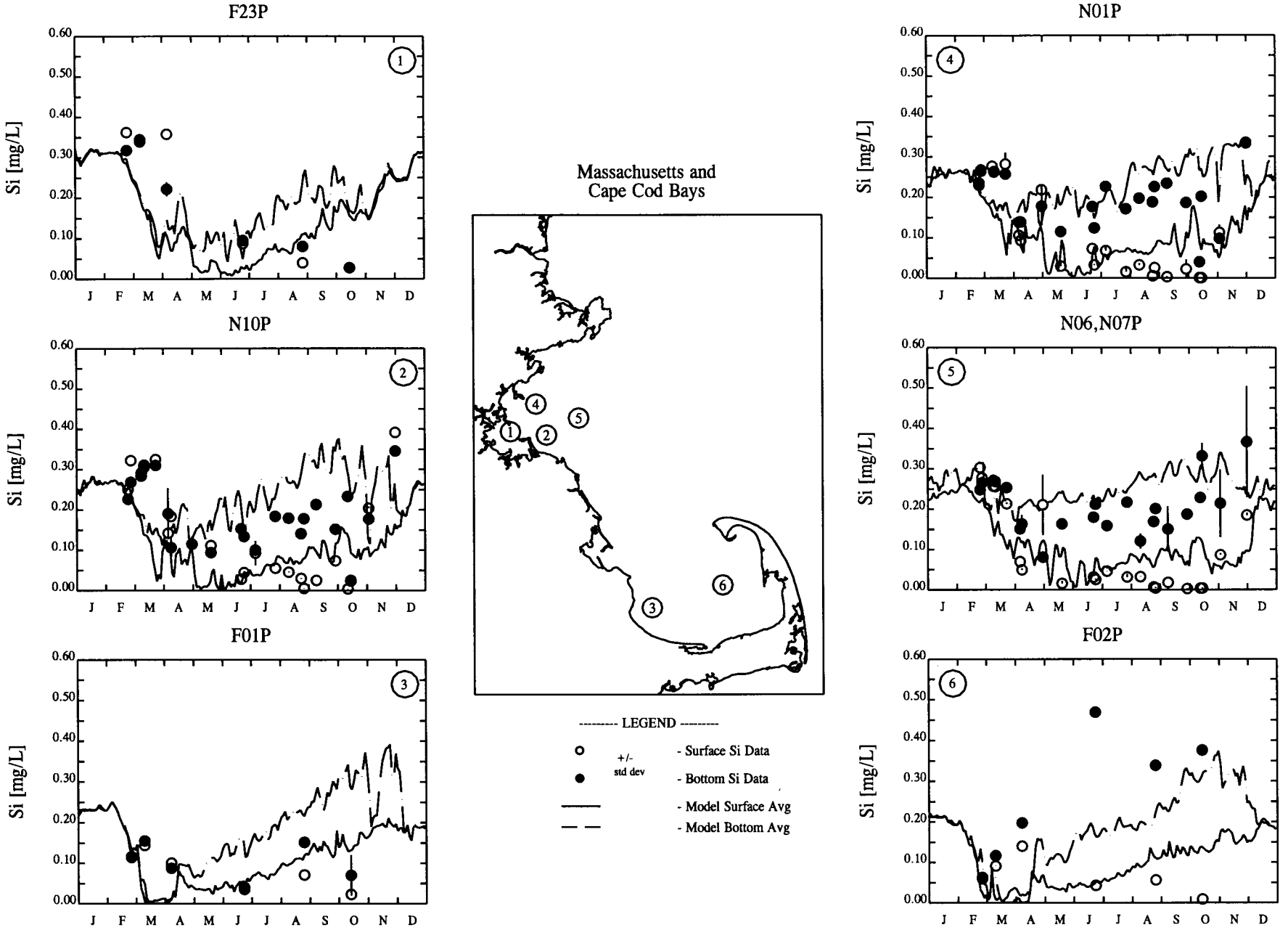


Figure 5-4. 1993 Temporal Calibration Results for DSi

5.3 1994 OCTOBER DO MINIMA

During each of the years from 1992 to 1996, the data have shown the strong seasonal decline in bottom water DO over the summer and early fall. The model was able to track the seasonal decline of DO concentrations reasonably well for 1992, 1993, and 1994, but was not able to achieve the extreme low concentrations that were observed in the nearfield for October 1994 (Figure 5-5). The data collected from other regions in October 1994 were represented quite accurately by the model with the exception of station F07, which is in the vicinity of the nearfield area (Figure 5-6). Given the importance of understanding the DO characteristics of the nearfield area, we have examined the monitoring data to learn why the model was unable to predict the low DO concentrations that were observed in 1994.

There are a number of physical and biological factors that lead to the seasonal decline of bottom water DO concentrations and control the extent of the decline. The establishment of stratification of the water column in the spring or early summer separates the water column into a surface layer, which is generally saturated or supersaturated with DO, and a bottom layer that is isolated from sources of oxygen and becomes under saturated with respect to DO. The surface layer has two sources of oxygen: 1) autotrophic production and 2) direct aeration from the atmosphere. At the nearfield and other offshore stations, the *in situ* production of oxygen by phytoplankton is generally restricted to the upper layers of the water column (including subsurface chlorophyll maximum near the pycnocline) due to light limitation at greater depths. Thus the bottom layer is isolated from sources of oxygen while stratified conditions exist and, as water temperatures increase, there is an increase in water column and sediment respiration that leads to the decline in oxygen concentrations in the bottom water.

This process occurs each year, yet, as was indicated by the monitoring data collected in 1992-1996 (Table 5-2) , the extent of DO decline varies. In order to determine why the model was unable to achieve the low DO concentrations of 1994, we need to understand the factors that lead to the variations observed in the monitoring data. Important physical and biological factors that affect the magnitude of the DO decline include:

- Initial bottom water DO concentration at the establishment of stratified conditions
- Biological processes (production and respiration)
- Variations in stratification (duration, strength, ventilation events, depth of pycnocline, bottom layer thickness)

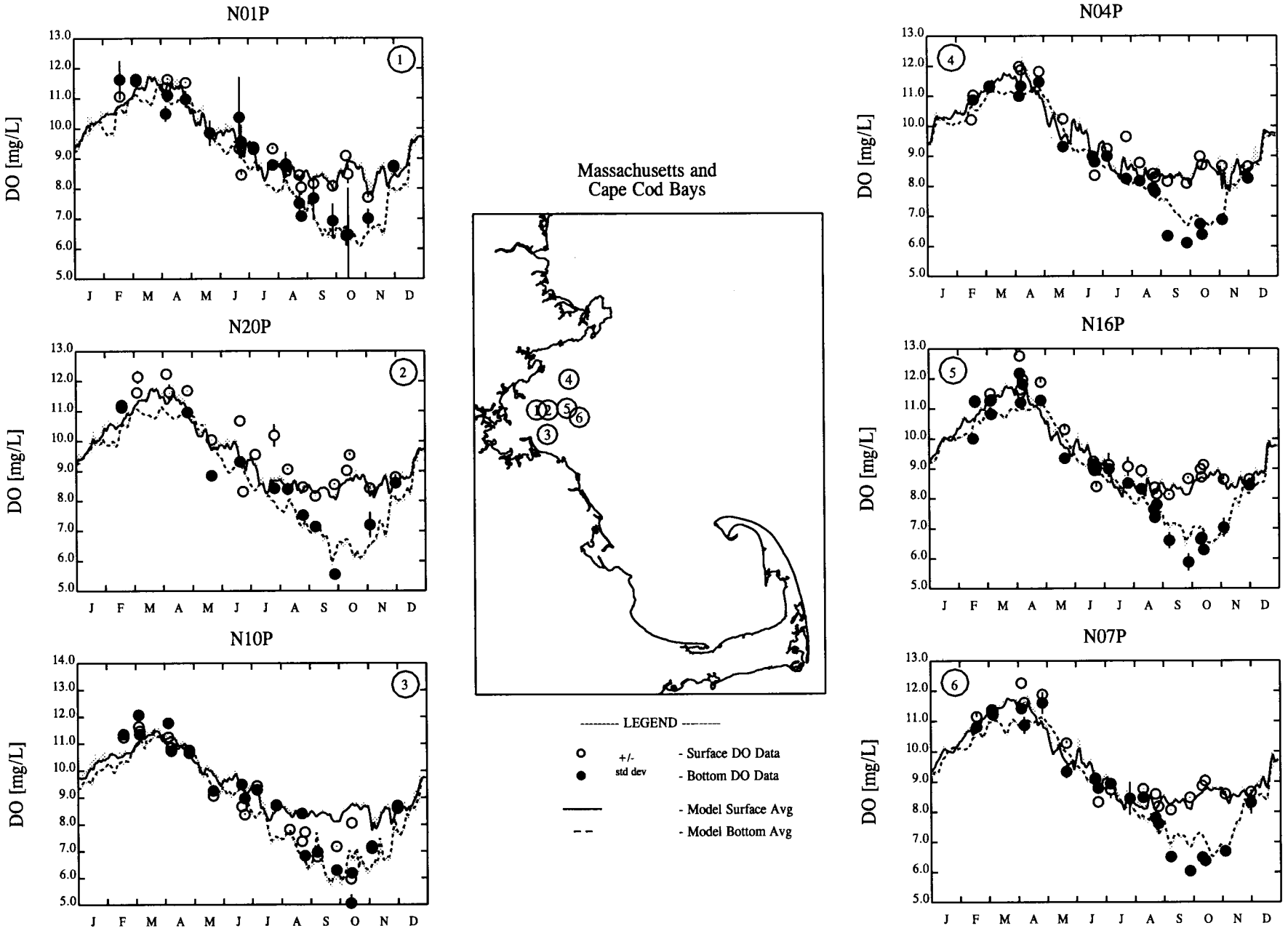


Figure 5-5. 1994 Temporal Calibration Results for DO at Six Nearfield Stations

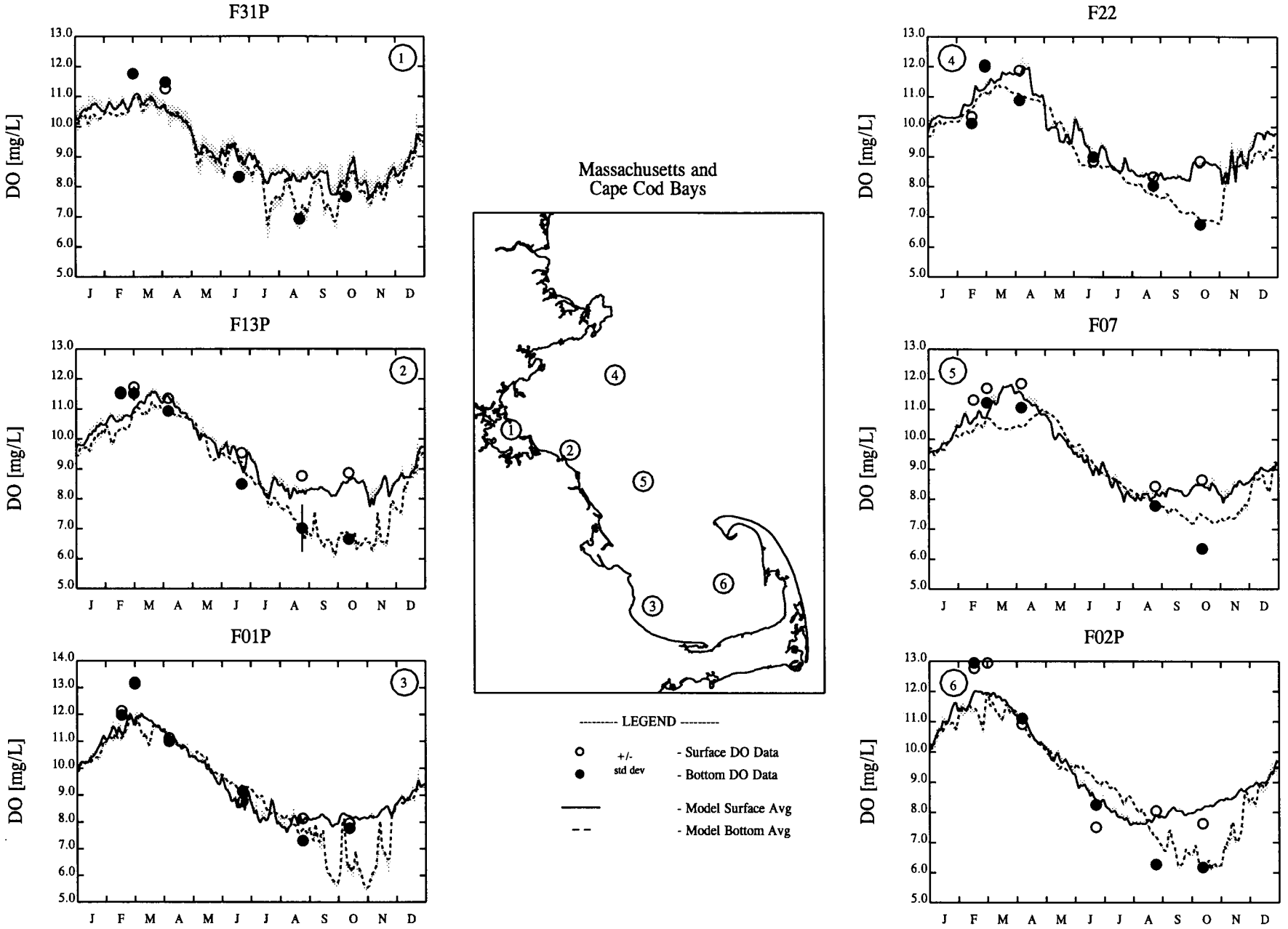


Figure 5-6. 1994 Temporal Calibration Results for DO at Six Farfield Stations

Table 5-2. Bottom Water Dissolved Oxygen Concentrations in the Nearfield Area at the Establishment of Stratified Conditions and the Minimum Values Observed.

Year	Mean DO (mg/L) June Survey	Rate of Decline (mg/L/d)	Mean DO (mg/L) Survey Minimum	DO Minimum (mg/L)
1992	9.91	-0.024	7.57	7.10
1993	10.18	-0.025	7.85	6.7
1994	9.06	-0.031	6.2	4.82
1995	~9.2	-0.027	6.6	5.67
1996	~9.9	-0.025	7.55	7.27

*1992-1994 data taken from Kelly and Doering 1998

**1995 & 1996 data taken from Cibik et al. 1996 and Cibik et al. 1998 (mean DO in June estimated from figure)

In 1994, stratified conditions were established in the nearfield by the June survey and the mean bottom water DO concentration was 9.06 mg/L, which was about 1 mg/L lower than the DO concentrations observed in 1992 and 1993 at the onset of stratified conditions. It was also lower than the mean bottom water DO concentrations observed in subsequent monitoring years, 1995 and 1996 (Table 5-2). Kelly and Doering (1995) have suggested that the rate of DO decline over the course of the summer and fall is relatively consistent and therefore the concentration of bottom water DO at the early phases of seasonal stratification is a good indicator of potential bottom water DO problems. It follows that if the model incorrectly represents bottom water DO at the onset water column stratification (i.e. too high a concentration), the model may also fail to reproduce the DO minimum concentrations observed in October. A review of the model output for 1994 presented in Figure 5-5 shows that the model was able to adequately reproduce the bottom water DO concentrations from April through June. Therefore, though initial bottom water DO concentrations may be a good early-warning indicator of low fall DO concentrations, it is not the reason that the model was unable to reproduce the minimum DO observed in the nearfield in October 1994. It should be noted that the rate of oxygen decline is just a straight forward mathematical computation and does not offer any explanation as to causality. It should also be noted that the 1994 rate of decline was 23% higher than the average rate of decline for 1992, 1993, 1995, and 1996.

Two processes could contribute to the observed decline in bottom water DO observed in 1994, biological and physical. Biological processes could control the reduction of DO concentrations, since water column and sediment respiration contribute to the decrease in DO concentrations in bottom waters. The amount of organic material that is available sets an upper limit on the rate of metabolism and, in the nearfield bottom waters, the main sources of organic material are primary production in the upper water column and advection of particulate organic matter from coastal waters. It would be expected that the measurement

of high biomass concentrations or production rates in the nearfield would correlate with low bottom water DO concentrations, in that the more organic material available, the higher the *potential* metabolism and decrease in DO concentrations. Biomass concentration (chlorophyll, POC, phytoplankton) and primary production, however, were not elevated in 1994 compared to other years. As mentioned above, high chlorophyll concentrations, phytoplankton counts, and production rates were observed in 1993, yet unusually low DO concentrations were not detected. Thus it appears, for these monitoring years, other factors (temperature, sedimentation rates, mixing, etc.) must have effected the extent to which the biological processes regulated DO concentrations in 1993 and 1994.

The relative importance of biological and physical processes has been discussed by a number of researchers in other forums. In this instance, however, the important point is that the monitoring data do not indicate that annual variations in biological parameters (chlorophyll, production, etc.) directly influenced the trends that were observed for bottom water DO concentrations in 1993 and 1994. With the exception of the 1993 fall bloom, the model adequately reproduced the trends in biological parameters for 1993 and 1994. This suggests then, as do the monitoring data, factors other than biology contributed to the unusually low DO concentrations observed in 1994.

Concurrent with relatively lower DO concentrations in 1994, field monitoring showed high bottom water temperatures (>12°C or 3-4°C above 1992-1993) and a deepened pycnocline prior to the fall overturn of the water column. Kelly and Doering (1998) concluded that these factors interact with metabolic processes leading to the observed trends in DO decline and the spatial/temporal variability in DO minima. The effect of temperature on the rates of metabolic processes is well characterized and, since the model reproduces the temperature trends accurately, variations in temperature appear not to be the reason the model missed the low DO concentrations in the bottom water. Variability in water column stratification, however, may have an effect on bottom water DO decline that is not sufficiently captured by the model.

The most obvious way that stratification might vary from year to year is the duration of stratification. The data, however, indicate that for 1992 to 1996 the duration of stratification and the timing of the DO minima (October survey) are relatively consistent. The “strength” of stratification refers to the ability of the bottom water to communicate with the DO rich surface waters. This primarily occurs through event driven inputs of DO rich waters or by vertical diffusion. Cibik et al. (1998) have shown that for the 1996 monitoring year there were a number of storm events that led to ventilation of the nearfield bottom waters. This was not a factor in 1994, as the weather was relatively calm and the only way ventilation events would

have affected the model result would have been if there were "storm events" within the model data, which was not the case. Kelly and Doering (1998) model results suggest that vertical diffusion of DO from surface to bottom water has minimal influence on the rate of DO decline in bottom waters. The eddy-diffusion coefficient used in their analysis was a standard nominal rate of 0.1 cm^2/sec and a doubling of this rate had limited effect on their results. It is expected that variations in vertical diffusion in the BEM did not result in the higher than measured bottom water DO concentrations.

An interesting result from Kelly and Doering (1998) was the effect that changes in bottom layer thickness had on DO concentrations and the rate of decline. The thickness of the bottom layer in the nearfield area varies temporally based on the depth of the pycnocline and spatially based on variations in water column depth due to bathymetric sloping and topographic irregularities. They found that the late-season deepening of the pycnocline in 1994 had a significant effect on DO decline. The deeper pycnocline meant that there was a thinner bottom water layer and, by default, an enhanced contribution of sediment respiration to DO decline. They took this a step further and suggested that, in contrast to the effect of late-season deepening, a shallow pycnocline at the onset of stratification may moderate seasonal DO decline by allowing mid-water primary production to introduce DO to the bottom water layer. Spatially the variability in bottom water DO results from changes in bottom layer thickness, as well as due to a general west to east increase in depth, and the topographical heterogeneity across the nearfield area.

This finding has interesting implications on how the BEM deals with changes in pycnocline depth and the resulting effect on bottom water DO concentrations. The BEM is vertically grouped over 10 sigma levels or 10 layers of uniform thickness based on the overall depth of the water column. In the nearfield, the thickness of these layers range from 2.4 to 5.5 m based on the nominal station depths. The resolution of the model is such that if the pycnocline deepens the thickness of the "bottom layer" will not decrease until the pycnocline enters a lower sigma level. To illustrate this point, we use station N04 as and example. This station has a nominal depth of 50 m and the model sigma levels are equal to 5 m. Assuming the pycnocline is at 40 m, there is a 10-m thick bottom layer. If we use a moderate sediment oxygen demand of 1 g O_2/m^2-d, there is a 0.1 mg O_2/L-d rate of DO decline within this bottom layer. If the pycnocline deepens to 43 m, there would be a 0.14 mg O_2/L-d rate of DO decline within the bottom layer. Since the resolution of the model will not adjust the pycnocline depth (or bottom layer thickness) until it deepens to the next sigma level, the model would still account for a 0.1 mg O_2/L-d or a 30% lower decrease in bottom water DO. This example clearly illustrates that the deepening of the pycnocline during late-season stratification can result in significant differences between the rates of *in situ* and modeled DO decline.

Another physical factor that may have contributed to the low DO concentrations observed in 1994 is the horizontal advection of low DO concentration waters from the Gulf of Maine. Some evidence to support this hypothesis is suggested by the increases in bottom water salinity and temperature in the late summer and fall of 1994 (Figure 4-10) versus 1992 and 1993 (Figures 4-8 and 4-9, respectively). Since the only way bottom salinity can increase is from import from the Gulf of Maine and since the increase in bottom water temperature does not appear related to vertical exchange with surface waters, because the bottom waters do not appear to be re-oxygenated with oxygen-rich surface waters, this might suggest a horizontal influx from the Gulf of Maine.

Though it is important for the model to accurately reproduce the low DO concentrations observed in the fall, the fact that the model was unable to achieve the unusually low concentrations in October 1994 should not be overemphasized. The model represents mean trends and may have more difficulty expressing localized or fleeting events. For example, the model was able to reproduce the trend in lower DO concentrations for the bottom waters in 1994 versus those in 1993 even though it was unable to achieve the extreme low values observed in October 1994. The actual *in situ* variability in DO was quite high in that time period. The lowest DO concentration, 4.82 mg/L, was observed at station N10 on October 13th, but on the previous day the bottom water DO was 6.25 mg/L. There are a number of factors that may have caused this change (or discrepancy) including tidal aliasing. Kelly et al. (1995) noted tidal effects at stations including N10P that are relatively shallow and in the midst of irregular bathymetry. The combination of tidal changes and irregular bathymetry make repeated sampling at the same height above bottom difficult. As discussed above, minor variations in sampling depth could significantly alter the observed DO concentration especially during the critical season when near-bottom temperature is high, the bottom layer is thin, and sediment oxygen demand is high.

SECTION 6

MASS BALANCE ANALYSIS

6.1 INTRODUCTION

One issue that has been very controversial during the design and construction of the future outfall pipe is the question as to the impact of the MWRA effluent on the Massachusetts Bays system, including Cape Cod Bay. An analysis of the contribution of MWRA's effluent to the internal loading to the Massachusetts Bay's system estimates that approximately 55 percent of the nitrogen and 65 percent of the phosphorus directly entering the bay are attributable to MWRA (Figure 2-1). This analysis ignores, however, a potentially large source of nutrients and organic matter entering the Massachusetts Bays system from the Gulf of Maine. In order to evaluate this contribution a global mass balance calculation was performed.

6.2 1992 NITROGEN MASS BALANCE

A mass balance analysis for nitrogen during 1992 was conducted with the calibrated model to estimate the relative importance of each source and sink of nitrogen in Massachusetts and Cape Cod Bays. Figure 6-1 presents a diagram of the mass balance that was performed. The analysis was completed for total nitrogen as well as dissolved inorganic and organic nitrogen. Sources of nitrogen to the bay included in the analysis were the MWRA WWTPs, other WWTPs, non-point sources, riverine sources, atmospheric deposition, and the Gulf of Maine. Sinks of nitrogen from the entire system are burial to the sediment, denitrification, and nitrogen associated with flow exiting Massachusetts and Cape Cod Bays. The mass balance around the water column included the deposition of PON to the sediment and the flux of NH_4 and NO_3 into or out of the sediment. In general, the net flow was directed into Massachusetts Bay near Cape Ann and directed out of Cape Cod Bay near Race Point.

Figure 6-2 repeats the mass diagram on the left side of the figure and includes model computations (kg/day) for each of the sources and sinks of nitrogen on the right. The summary table presented in Figure 6-2 indicates that only 3% of the TN entering Massachusetts and Cape Cod Bays is from the MWRA treatment plants while the boundary contributes 92% of the TN. The remaining sources contribute 5% of

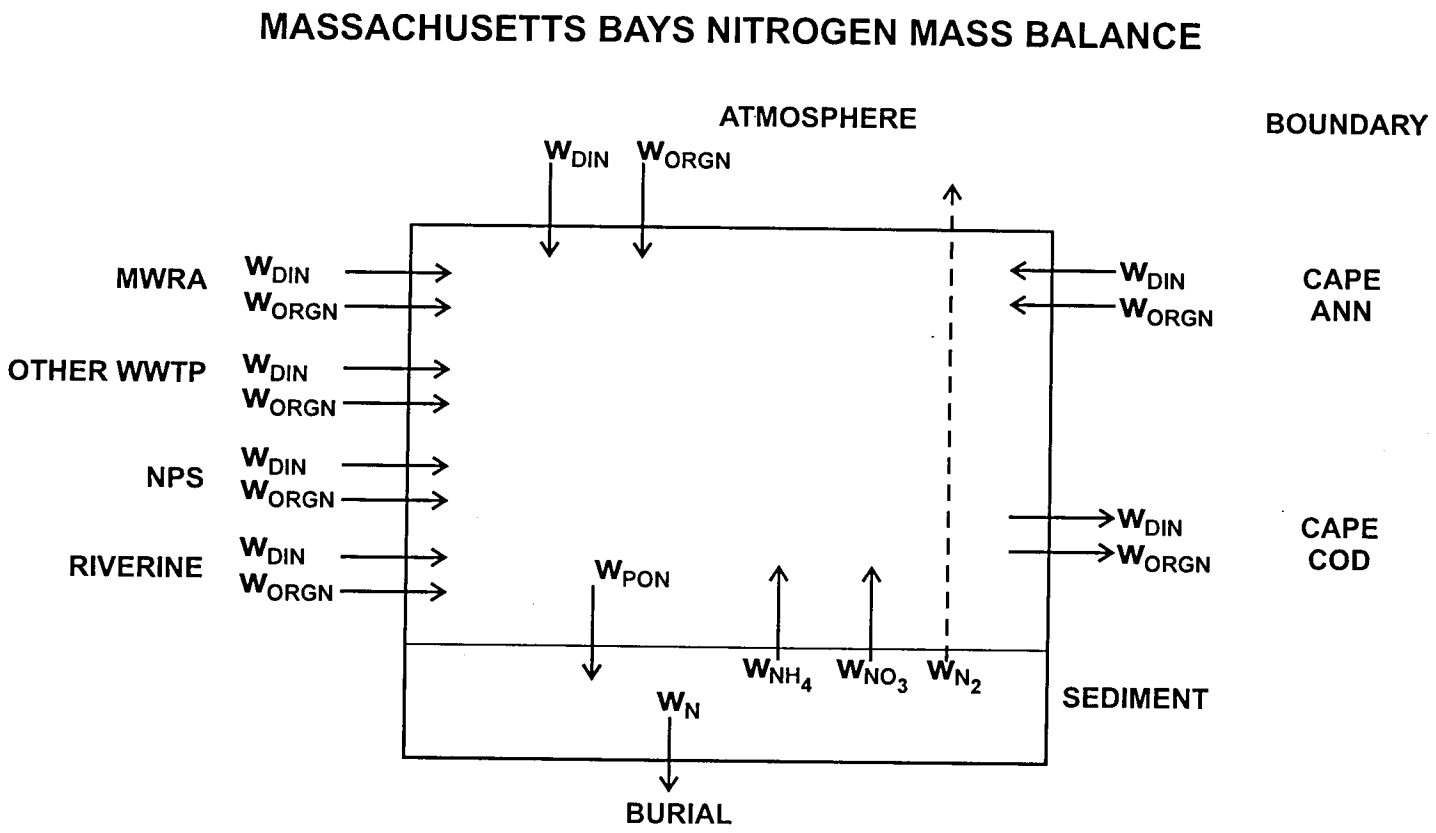
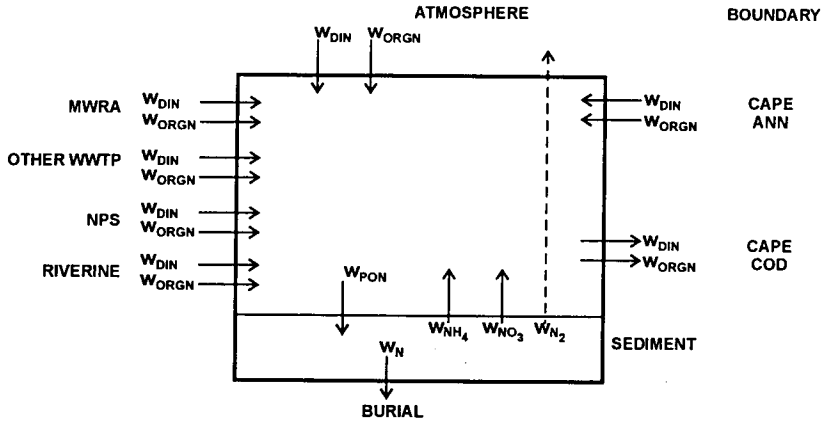
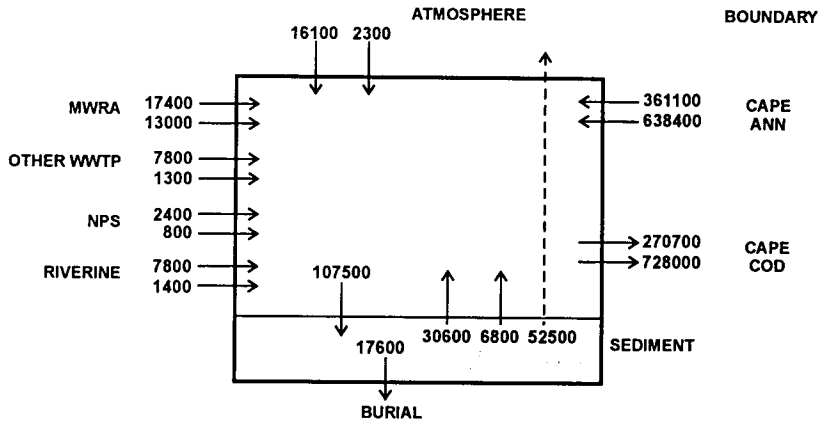


Figure 6-1. Massachusetts Bay Nitrogen Mass Balance

MASSACHUSETTS BAYS NITROGEN MASS BALANCE FOR 1992





(All units are kg/day)

SUMMARY	DIN	ORGN	TN	%TN
MWRA	17400	13000	30400	3
NON-MWRA	34100	18800	52900	5
BOUNDARY	361100	638400	999500	92

Figure 6-2. Massachusetts Bay Nitrogen Mass Balance for 1992

the TN load to Massachusetts and Cape Cod Bays. The MWRA WWTPs contribute 4% of the DIN load to Massachusetts and Cape Cod Bays. Only 1.6% of the TN entering system is lost by sediment burial and 4.8% of the TN is lost from the system through denitrification. The remaining nitrogen exits with the flow at Race Point although it may be in a different form than when it entered. It is interesting to note, that the model predicts that nearly 50% of the PON that is deposited to the sediment is later lost from the system through denitrification.

6.3 REGIONAL ANALYSIS

The analysis just presented performed a global mass balance for nitrogen. It indicates that over an annual cycle only 3 percent of the total nitrogen entering the Massachusetts Bays system originates from MWRA. However, this analysis does not provide any seasonal or regionalized overview of MWRA effluent within the Massachusetts Bays system. In order to provide some insight into this question two additional model runs were performed. These runs involved the use of two conservative tracers. One tracer represented an input along the current or future MWRA outfall, while the second tracer represented an input at the northern boundary to Massachusetts Bay, near Cape Ann. The tracer at the MWRA outfall utilized a time-variable concentration equal to the total nitrogen effluent concentration discharged by MWRA in 1992. The tracer at the northern boundary utilized a time-variable concentration equal to the total nitrogen prescribed for the 1992 calibration. The time-series of these tracer concentrations are presented in Figure 6-3. As can be seen, the MWRA tracer concentration ranges between approximately 13.5 and 27 mg/L, while the boundary tracer ranges between 0.11 and 0.27 mg/L in the surface and 0.26 and 0.34 mg/L in the bottom. In the first simulation, the model was run using the 1992 hydrodynamics, the MWRA effluent (Figure 6-3) discharged at the current outfall site, the prescribed boundary conditions (Figure 6-3), and assuming zero initial conditions. The model also assumed that all of the tracer material entered the system in the dissolved form and that the tracer was conservative. While total nitrogen entering the Massachusetts Bays system is comprised of both dissolved and particulate forms, the latter of which can settle, and while nitrogen is not necessarily a conservative material, the results provided from this run should provide some insight into the possible spatial and temporal distribution of MWRA's effluent within Massachusetts Bay and Cape Cod Bay. In the second simulation, the model was run using the MWRA effluent (Figure 6-3) discharged at the future outfall site, the associated future outfall hydrodynamics, the boundary conditions as prescribed in Figure 6-3, and assuming zero initial conditions.

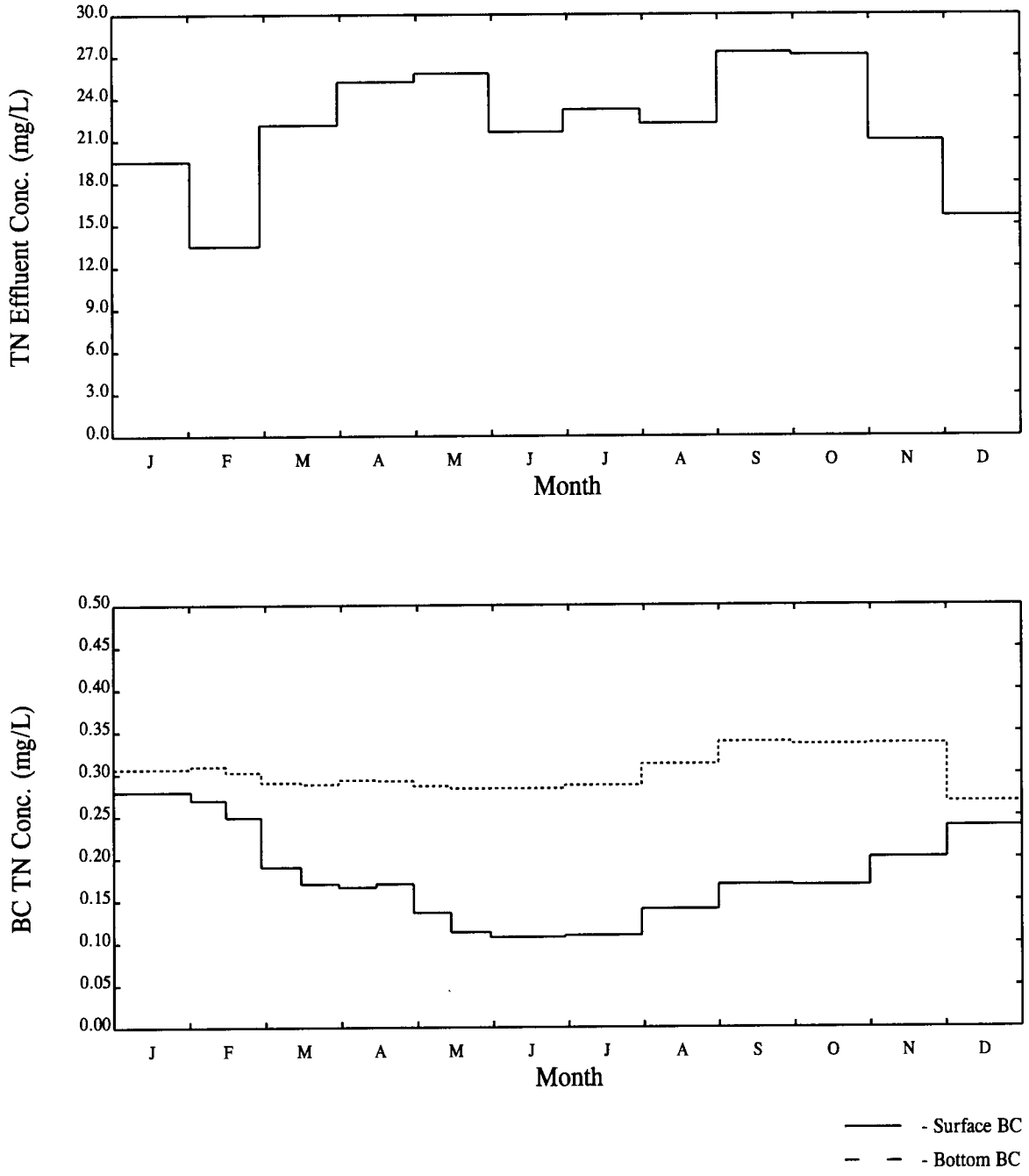
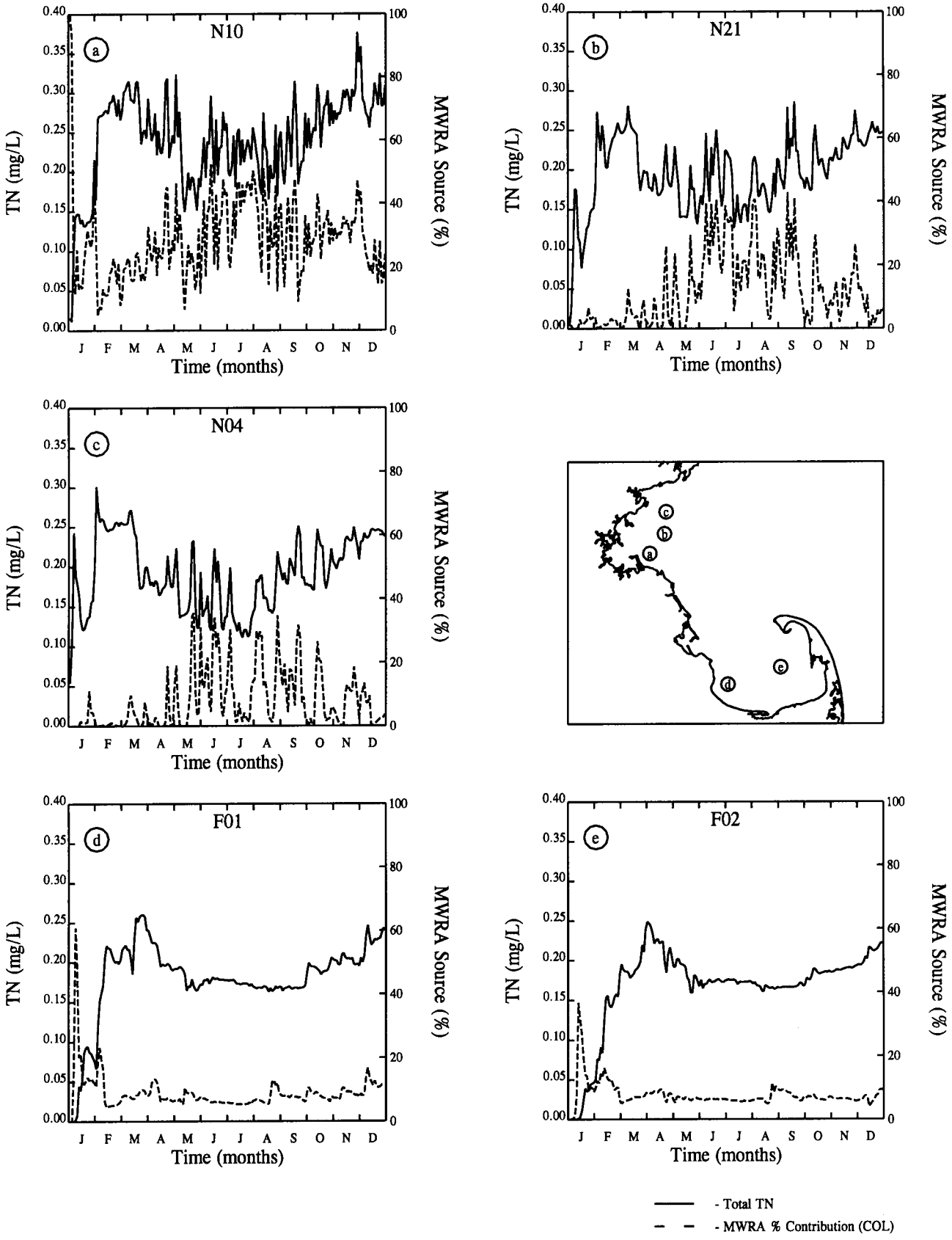


Figure 6-3. MWRA and Open Boundary Tracer Concentrations

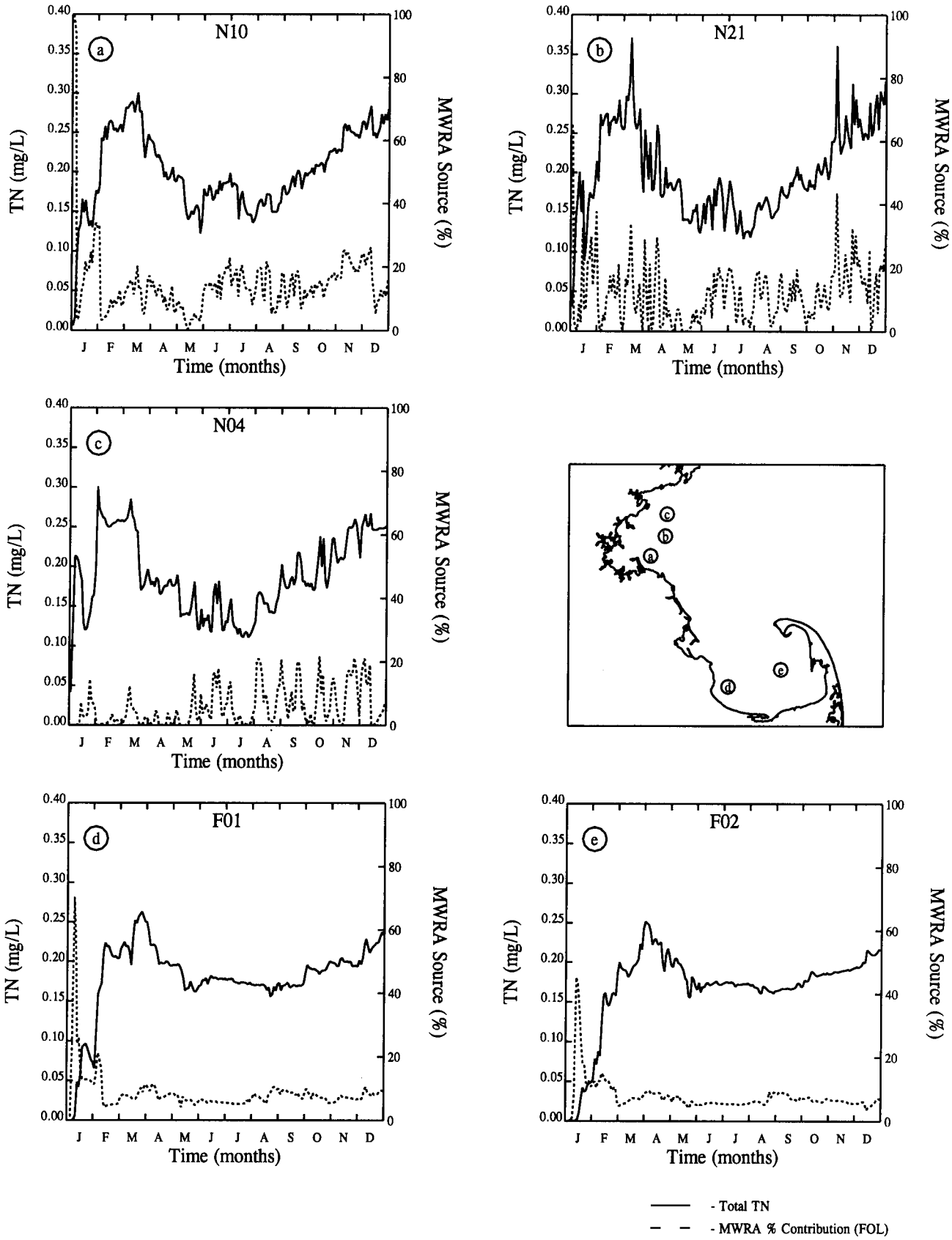
Figure 6-4 provides a time-series analysis of tracer concentrations i.e., the current outfall location (COL) for five locations in the Massachusetts Bays system. These locations included three segments in northwestern Massachusetts Bay (corresponding to sampling locations N10, N21, and N04 in the vicinity of the future outfall) and two segments in Cape Cod Bay (corresponding to F01 and F02). The solid line represents the time-series of the total tracer concentration, while the dashed line represents the time-series of the percent contribution of the MWRA tracer effluent to the total tracer concentration. As can be seen, even for the stations located near the present MWRA discharge, the tracer associated with the boundary condition dominates for much of the year. The exception is during the summer months, when the surface tracer concentrations at the boundary decrease relative to the winter months. This change in the boundary condition results in the MWRA contribution to the total tracer concentration increasing to between 40 and 50 percent. One can also observe a dilution gradient for the three northern locations, with the N04 site having considerably less MWRA tracer than N10. The two Cape Cod Bay locations show even less of a contribution from MWRA to the total tracer concentration, a maximum of less than 20 percent. In addition, it appears as if total tracer reaching these two locations is less time-variable. In fact, minimum summer tracer concentrations in Cape Cod Bay are slightly higher than the minimum summer concentrations computed at the three Massachusetts Bay segments. This may be due to the circulation of the Massachusetts Bays system, wherein the largest influx of water into Cape Cod Bay from Massachusetts Bay occurs during the late winter and spring, coincident with the spring freshet entering Massachusetts Bay from the Gulf of Maine.

The results of the second simulation run for the future outfall location (FOL) are presented in Figure 6-5. A number of interesting features relative to the current outfall run (Figure 6-4) can be noted. The first feature is that the concentration of total tracer is lower at station N10 for the FOL run and that the MWRA contribution is lower year round. These results occur because N10, which is close to Boston Harbor, received a significant portion of the tracer in the waters leaving Boston Harbor under the COL, while for the FOL apparently the circulation of the Massachusetts Bays system does not advect a significant portion of the tracer from the FOL to N10. In addition, the MWRA contribution does not increase during the summer months at N10 (as well as the other nearfield stations N21 and N04) due to the fact that the tracer is trapped below the pycnocline during the summer months and does not reach the surface layer. The second feature to note is that total tracer concentrations are higher during the winter/early spring and late fall/winter months and lower during the late spring/summer/early fall for the FOL.



SURFACE LAYER - SOURCE DURING ENTIRE YEAR

Figure 6-4. Results of Tracer Analysis for the Current Outfall Location



SURFACE LAYER - SOURCE DURING ENTIRE YEAR

Figure 6-5. Results of Tracer Analysis for the Future Outfall Location

MWRA's contribution to the total tracer for the FOL run follows a reverse pattern to the COL run, higher during the winter months and lower for the summer months. These results occur because N21 is in the immediate vicinity of the future outfall site and during the winter months the MWRA tracer reaches the surface waters, while during the summer it remains trapped in the subsurface waters. The final feature to note is that there is little or no difference between the COL and FOL runs for the two stations in Cape Cod. The western Cape Cod station (F01) receives between 10 and 18 percent of the MWRA tracer, while the eastern station (F02) receives less than 10 percent (other than a short spike associated with spinning up the model from the zero initial conditions).

SECTION 7

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