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Forecasting Impacts of a Hypolimnetic Wastewater Discharge on Lake Stratification and Mixing

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ABSTRACT

Application of a one-dimensional hydrothermal model to polluted Onondaga Lake, New York, is documented. Features of the model include simulation of entrainment associated with plunging inflows, capability for simulations during ice cover, and a submodel to simulate the near-field mixing associated with a multiport diffuser. The model is successfully calibrated for 6 consecutive years of stratification conditions. The model performs well in simulating the dimensions and temperatures of layers, and the timing/duration of stratification. The model is less successful in simulating the more subtle effects of dense saline inflows that linger from a recently closed soda ash/chlor-alkali facility. The calibrated model is used to forecast the impact of a proposed hypolimnetic discharge of treated municipal wastewater on stratification and mixing. Important simulated impacts are increased temperatures in the hypolimnion, reductions in density stratification, increased mixing and homogeneity within the hypolimnion, and reductions in the duration of summer stratification. Predictions from this analysis serve as input to water quality models that simulate related impacts of this management action.

Key Words: Stratification, vertical mixing, hydrothermal model, diffuser.

Discharge of treated wastewater to a receiving water through a submerged diffuser is commonly used to increase dilution of wastewater in the vicinity of the discharge (Wright et al. 1982). Such dilution is induced by discharge of effluent through small diameter nozzles or ports, resulting in turbulent entrainment of ambient receiving water. In cases where the wastewater is buoyant relative to the receiving water, additional mixing is induced as the effluent rises through the ambient water column. Additionally, when the diffuser is located at a depth below the thermocline, the wastewater plume may be "trapped" at depth, effectively preventing direct inflow of effluent to surface waters (Fischer et al. 1979). Onondaga County (1994) has proposed to discharge aerated effluent from the METRO municipal wastewater treatment plant to a diffuser located in the hypolimnion of Onondaga Lake to reduce water quality impacts associated with the present shoreline discharge. Potential benefits from this management action include increased isolation of phosphorus and ammonia from the epilimnion and direct oxygenation of the anoxic hypolimnion during summer and winter when this dimictic lake is thermally stratified.

Here we provide an analysis of the lake-wide effects of this proposed hypolimnetic discharge on stratification and vertical mixing, particularly as they may influence water quality. The analysis is based on the application of a mechanistic mathematical model of vertical heat and mass transport. Credibility of the model is established through extensive calibration to conditions that have prevailed since the adjoining soda ash/chlor-alkali plant closed (Effler and Owens 1994). A submodel is included to describe the near-field mixing associated with a multiport diffuser to allow forecasting of the effects of the proposed discharge. Certain predictions of this model are used to specify thermal and transport/mixing information in a separate model that is used to forecast the water quality impacts of this proposed discharge (Effler et al. 1994).

Proposed Hypolimnetic Discharge

General specifications of the proposed discharge were contained in a planning document (Onondaga County 1994). The proposal called for discharge through a 1 km long, 275 cm diameter outfall pipe to a diffuser located at a depth of 14 m at a peak flow of $7.0\,m^3/s$ (160 mgd). The proposal also called for supersaturation of the effluent with 9100 kg/d (20,000 lbs/d) of dissolved oxygen, resulting in an effluent concentration of 27 mg/l at the $3.9\,m^3/s$ (88 mgd) average flow. The ability to trap effluent in the lower waters of Onondaga Lake is affected by the magnitude of this discharge relative to the size of the hypolimnion of the lake. The METRO discharge at the average flow of $3.9\,m^3/s$ would flush the hypolimnion (assumed to be the volume below a depth of 8.5 m, or $47 \times 10^6\,m^3$ (Owens 1987)) in about 5 months, which is roughly the duration of summer stratification in the lake. Thus, lake-wide stratification conditions can be expected to be modified by this discharge.

An additional feature of this proposal is the buoyancy of the METRO effluent relative to the waters of the hypolimnion of Onondaga Lake. The annual variation of the METRO effluent temperature and of Onondaga Lake at a depth of 14 m (Effler and Owens 1994) is depicted for the 1987-1993 interval in Fig. 1a. Year-to-year variations in lake temperatures are due to differences in weather conditions (Owens and Effler 1989). The discharge is warmer than existing lake temperature at a depth of 14 m throughout the year. The difference is particularly great in mid-summer during the period of strong thermal stratification. Under these conditions, the discharge would be buoyant (Fig. 1b) and tend to rise in the water column of the lake as a plume that entrains ambient lake water as shown schematically in Fig. 2. During winter, the METRO and lake temperatures straddle the temperature of maximum density ($\sim 3.5^{\circ}C$ in Onondaga Lake due to its elevated salinity), which may allow for unusual buoyancy effects (Ford and Johnson 1983). The weak stratification occurring in the fall-winter-spring period creates the conditions under which the salinity stratification in the lake (Effler and Owens 1994) can affect vertical transport and mixing.

Hydrothermal Model

Governing Equations

The model used is based on the one-dimensional assumption, and thus predicts areally-averaged quantities. This general type of model was described by Harleman (1982) and is referred to as a hydrothermal model. Due to the influence of salinity on buoyancy in Onondaga Lake (Effler and Owens 1994), both temperature and chloride ion (a surrogate measure of salinity) were simulated in this model. The governing equations are the one-dimensional heat and mass conservation equations, which are:

$$\frac{\partial T}{\partial t} + w\frac{\partial T}{\partial z} = \frac{1}{A}\frac{\partial}{\partial z}\left(A\,K\frac{\partial T}{\partial z}\right) + \frac{1}{\rho c A}\frac{\partial}{\partial z}(A\phi_S) + \sum \frac{q_I}{A}(T_I - T) \quad (1)$$

$$\frac{\partial C}{\partial t} + w\frac{\partial C}{\partial z} = \frac{1}{A}\frac{\partial}{\partial z}\left(A\,K\frac{\partial C}{\partial z}\right) + \sum \frac{q_I}{A}(C_I - C) \quad (2)$$

where T and C are water temperature and chloride concentration, t is time, z is vertical position, w is the vertical velocity, A is the plan area of the lake, K is the turbulent (eddy) diffusion coefficient, ρ and c are the density and specific heat of water, ϕ_S is the flux of solar radiation, q_I is the lake inflow per unit vertical distance, and T_I and C_I are the temperature and chloride concentration of an individual inflow. The summation on the right side of Eqs. 1 and 2 indicates that the individual characteristics of 5 inflows were considered. The vertical distribution of q_I is determined by several near-field mixing submodels; two of these, associated with negatively-buoyant (plunging) shoreline inflows and the buoyant diffuser discharge, are described below.

Boundary conditions for the solution of Eqs. 1 and 2 describe the fluxes of heat and chloride at the water surface and lake bottom. At the water surface, the heat flux is given by

$$-K\frac{\partial T}{\partial z} = k_S(T_S - T_E) + \beta\phi_{SO} \quad (3)$$

where k_S is a heat transfer coefficient, T_S is the water surface temperature, T_E is the equilibrium temperature, and β is the fraction of the net (incident less reflected) solar radiation ϕ_{SO} that is absorbed at the water surface. The quantities k_S and T_E are functions of meteorological quantities and of T_S, and represent heat transfer due to long-wave atmospheric radiation, back radiation, evaporative and conductive transfer. The flux of heat at the lake bottom, and the flux of chloride at the water surface and lake bottom, are assumed equal to zero.

The model used here assumes that a well-mixed layer of depth h extends down from the water surface; the depth of this layer is determined from an integral, parameterized form of a turbulent kinetic energy balance for this layer (Harleman 1982). This "entrainment" relationship is given by

$$\frac{dh}{dt} = \frac{C_F \sigma}{C_T + Ri} + \frac{1}{A_I}(Q_I \quad Q_O) \tag{4}$$

where $\sigma^3 = \eta^3 u_*^3 + w_*^3$, u_* is the shear velocity in water due to surface wind stress, w_* is a velocity scale related to buoyancy effects in the mixed layer, C_F, C_T, and η are empirical coefficients, $Ri = \sigma^2/g'h$ is the bulk Richardson number for the mixed layer, A_I is the plan area of the interface at the base of the mixed layer, and Q_I and Q_O are the inflow to and outflow from the mixed layer. Under conditions where there is turbulent energy available for the mixed layer to grow ($\sigma > 0$), this equation is used to compute its growth. When this is not the case, h will decrease such that $\sigma \quad 0$. Values of $C_F = 0.5$ and $C_T \quad 3.6$ have been determined from experiments on mixed layer dynamics (Sherman et al. 1978, Harleman 1982); η is not expected to have a universal value as it includes site-specific characteristics such as anemometer location, basin setting, and fetch.

The equations above are solved numerically using a finite difference approximation, employing a Lagrangian approach to represent the effects of the vertical velocity w in Eqs. 1 and 2 (Imberger and Patterson 1981) Thus, the discrete layers used to represent variations in the vertical direction may expand, contract, and move vertically in response to the vertical distribution of inflow q_I and outflow per unit depth q_O. Two forms of outflow are computed. First, water flow out of the lake basin is computed from measured inflow rates and change in lake level. In addition, "internal" outflows from the water column are associated with near-field mixing of tributary and treatment plant inflows, as shown in Fig. 2. Plunging shoreline inflows remove (entrain) water from the surface, which then re-enters the water column at a greater depth. In an analagous manner, the buoyant jets from the proposed multiport diffuser remove water from the water column as they rise. The computation of such flows is described below

An important component of the model was the equation of state used to compute water density, and thus buoyancy, from temperature and chloride concentration. Based on measurements of chloride ion and each of the major ions contributing to salinity, an empirical relationship between the two was determined. This relationship between chloride concentration C in ppm and salinity S in ppt is given by

$$S = \begin{cases} 0.00152C + 0.589, & \text{if } C > 800 \\ 0.00178C + 0.381, & \text{if } C < 800. \end{cases} \tag{5}$$

Density was calculated based on T and S with an equation of state that combines the pure water T temperature relationship presented by Millero et al. (1976) with the S dependence of Chen and Millero (1978).

Model Enhancements

Owens and Effler (1989) calibrated a similar model for Onondaga Lake for conditions that prevailed before the soda ash/chlor-alkali facility closed in 1986. The effort focused on the impact of the plunging inflows, made dense from the facility's saline waste discharge, on the lake's stratification regime. The model performed well in simulating the complex stratification regime of the lake that prevailed before closure of the facility, and in supporting identification of related impacts (Effler 1987). These impacts have been substantially reduced, but not eliminated, by the closure of the plant (Effler and Owens 1994)

The earlier model (Owens and Effler 1989) was enhanced in this study in several ways. First, the method of solution of the mixed-layer conservation equations was modified to yield a more accurate numerical solution of integral conservation equations for this layer. In addition a modified expression for the turbulent diffusion coefficient was used. The new turbulence closure is given by

$$K = \frac{C_H \epsilon}{(N^2 + N_o^2)^r}$$

where C_H is a model coefficient to be determined by calibration, ϵ is the rate of dissipation of turbulent energy per unit mass averaged over the lake volume, N is the local buoyancy frequency in the water column, N_o is the minimum value of N due to compressibility of water, and r is an empirical coefficient. This expression is similar to those used by others for the stratified layers of lakes (Fischer et al. 1979, Aldama et al. 1988). Additional changes to the model involve the submodels describing plunging inflows, the proposed buoyant diffuser discharge, and an ice cover submodel to allow year-round simulations.

Vertical transport associated with plunging shoreline inflows has been reviewed by Alavian et al. (1992). In general, the entrainment of ambient lake water can be computed if there is knowledge of the cross-sectional geometry of the density current as it flows down the slope. This geometry can be defined when the inflow is laterally confined, as in the case of an arm of a dendretic reservoir (e.g. Hebbert et al. 1979). In Onondaga Lake, Ninemile Creek, Onondaga Creek, and the METRO effluent have been found to be more dense than lake surface waters at certain times of the year (Effler 1994). Available information on lake bathymetry indicates that density currents from these inflows are not confined laterally to any significant extent. In addition, there are no known measurements of the geometry of these currents.

Due to the lack of information on the geometry of plunging inflows in Onondaga Lake, a very simple approach was used here. To describe mixing in the vicinity of the shoreline, a constant entrance mixing coefficient S_E was used, such that the inflow rate of the density current after entrance was increased by a factor $(1 + S_E)$. The temperature and chloride concentration of the inflow are also adjusted based on the characteristics of the surface layer. Following entrance mixing, entrainment of ambient lake water into the density current flowing down the sloping lake basin is described by

$$\frac{dQ}{dz} = S_I Q$$

where Q is the flow rate of the plunging inflow, and S_I is a mixing coefficient with dimensions of inverse length indicating the rate of entrainment per unit vertical distance. Again, temperature and chloride of the plunging inflow are adjusted to reflect entrainment of ambient lake water. The plunging inflow submodel assumes that the density current continues to plunge until it reaches a depth of neutral buoyancy, where the density of the current is equal to that of the lake. It was assumed that the inflow enters the water column over a 2-meter depth interval centered at the neutral buoyancy depth.

In order to forecast the entrainment and vertical transport due to the proposed buoyant diffuser discharge, the near-field mixing theory of Wright et al. (1982) was used. The approach assumes that the multiport diffuser is made up of an array of ports that discharge in a normal direction to the axis of the diffuser manifold, and alternate in direction (Fig. 2). As wastewater exits an individual port, mixing of ambient lake water is computed first by considering each round buoyant jet, assuming an equal distribution of flow to each port. After sufficient entrainment occurs to cause adjacent jets to merge, mixing is calculated assuming identical two-dimensional jets on either side of the diffuser manifold. The entrainment relationships originally proposed by Wright et al. (1982) were revised to reflect recent experience (S. J. Wright personal communication 1994).

The diffuser submodel proposed by Wright et al. (1982) is based on integral conservation equations for water volume, horizontal and vertical momentum, and buoyancy. The buoyancy conservation equation is based on the assumption of a linear relationship between buoyancy and temperature or salinity. While this assumption is reasonable in some applications, it is not in this case, particularly during winter conditions. Thus the buoyancy conservation equation was replaced by separate integral jet conservation equations for temperature and chloride; jet buoyancy was then computed using the equation of state. The submodel computes the character (Q, T and C) of the buoyant jet as it rises. The mixed effluent then enters (intrudes) into the lake water column over an "intrusion depth interval", the upper limit of which is the depth of neutral buoyancy of the jet The lower limit of this depth interval is either the depth at which the jet ceases to rise or zero, whichever is greater. Based on the buoyancy of the METRO effluent depicted in Fig. 1, the effluent may effectively enter the water column anywhere between the depth of the diffuser and the lake water surface.

The simulation of ice cover is included as a part of the heat conservation equation. Ice cover is assumed to form as soon as water surface temperature below the freezing point is computed; when this occurs, water volume below the freezing point is converted to an equivalent volume of ice at the freezing point. When ice exists, the surface heat transfer relationship is modified to simulate transfer at an ice-air interface. This modified relationship considers solar and long-wave radiation, sublimation and conduction. Heat transfer through the ice layer is computed by assuming a linear vertical distribution of temperature. Change in ice thickness is computed using a simple heat balance at the upper and lower surfaces of the ice cover. Ice thickness increases when conduction from the lower surface into the ice layer exceeds transfer from the water to the ice. Melting occurs at either the upper or lower surface when there is transfer to the surface. This approach to ice modeling is a simplification of models used by others (Ashton 1986). The accumulation of snow on the ice surface is not directly simulated, and the mechanical effects of wind in breakup of ice are not considered.

Model Inputs

The stratification model is tested here for a six year period (1987-92) following closure of the soda ash/chlor-alkali facility. A substantial data base is available to describe the forcing conditions of meteorology and inflow hydrology over this period, and of in-lake measurements used to initialize and calibrate/verify the model. The morphometry of the lake was specified according to the bathymetric analysis of Owens (1987). Daily average meteorological data were obtained from the nearby (8.5 km) Syracuse NWS monitoring station; incident solar radiation measurements were used when available. Daily average flow rates of natural tributaries were measured by USGS, and daily METRO flows were available from plant records. Inflow T and

C were measured at a frequency that varied from daily to biweekly. Daily average values of tributary T, for days lacking measurements, were estimated with an empirical model (Ford et al. 1978), that fits the measured seasonality with a harmonic function and predicts daily variations from the seasonal trend using measured air temperature. Daily average values of tributary C, for days lacking measurements, were estimated by Effler (1994). Weekly underwater measurements of ϕ_S yielded values of the extinction coefficient (Perkins and Effler 1994); daily values were calculated by interpolation. Model predictions were made at a 1-day time interval, so that diurnal variations in stratification were not considered.

Model Calibration/Verification

The performance of model simulations in hindcasting documented stratification conditions for Onondaga Lake is shown in two different formats, as vertical profiles of T and C for individual days, and as statistics that summarize selected features of the lake's stratification regime. Stratification statistics presented are volume weighted T of the upper and lower layers (demarcated by the depth of 8.5 m), and total (top to bottom) density difference and the contribution of salinity to this density difference (Effler and Owens 1994). These statistics represent a substantial consolidation of the observations and simulations, yet they describe important features of the lake's seasonal stratification regime and facilitate evaluation of model performance for multiple years.

Model calibration was performed by operating the model for the spring-fall period of 1987 and 1989. The following values of model coefficients were determined: $\eta = 2.1$, C_H 0.00002, $r = 0.30$, $S_E = 1.5$, and $S_I = 0.06\,\mathrm{m}^{-1}$. The values of 4 of these coefficients are in the same range as values reported in related studies: η (Sherman et al. 1978; Aldama et al. 1988), C_H (Ford and Johnson 1983), r (Jassby and Powell 1975; Ford and Johnson 1983), and S_E (Ford and Johnson 1983). Performance of the model in simulating the observed features of stratification for each of the six years over the 1987-1992 interval is presented in a series of figures. Predicted and measured mid-month T profiles for 1991 are shown in Fig. 3; profiles of

C are included for several dates. This model run was initialized by the measurements of March 26, during spring turnover. The model performed well in simulating the principal features of the lake's stratification regime, these being the duration of the stratification period and the temperatures of the layers. Errors in some of the finer features of stratification are apparent. The model slightly over-predicted bottom water temperatures, particularly in the early part of the stratification period, and depicted somewhat greater uniformity within the hypolimnion than actually observed The model performed well in simulating the mid-depth peaks in C observed during summer stratification that are associated with the plunging of saline Ninemile Creek (Effler and Owens 1994), but failed to predict the salinity stratification that formed in the lowermost waters in mid-October (Fig. 3g)

Model performance for the entire 6-year period is depicted in Figs. 4 and 5. In this simulation, the model was initialized using profiles measured on April 8, 1987, and was run continuously through the end of 1992. The model performed well in simulating the timing of stratification and the temperature dynamics of the upper waters in all six years. Performance in simulating the temperatures of the lower waters was good in 1990 through 1992, but was not as good in 1987 through 1989 (Fig. 4). The temporary establishment of stratification in early spring was successfully modeled in 1987, 1990, and 1991, but was overpredicted for 1989. The model performed well in simulating overall density stratification within the lake's water column (Fig. 5). Year-to-year differences in performance are generally coupled to the model's success in simulating the temperatures of the lower layers. Stratification was overpredicted the most in 1988 because of the underprediction of the temperatures of the lower layer. Stratification was somewhat overpredicted for the early to mid-summer period of 1989 and slightly underpredicted for portions of 1991 and 1992. Minor salinity-based restratification was predicted during early fall of 1987 through 1991, but was only clearly documented in 1991 (Effler and Owens 1994).

Projections for Hypolimnetic Discharge

In this study, simulations of the impact of the proposed discharge were made for the same 6 years of historical meteorologic and hydrologic conditions used in calibration and verification. The proposal was investigated by changing only the description of the METRO discharge in the model from its present shoreline location to the hypolimnion. The design characteristics of a multiport diffuser which must be specified in the diffuser submodel are the diffuser depth D, manifold length L, number of ports n, and port diameter d. The following observations are made based on numerous simulations using various diffuser configurations. During periods when the water column is homogeneous (turnover), the METRO effluent rises to the lake surface. During summer stratification, the diffuser creates a well-mixed region of the water column beginning just below the level of the diffuser and extending up to a level just below the thermocline, where the METRO effluent enters the water column Within this region, the METRO effluent rises and entrains ambient lake water, and water moves downward in the water column (w is negative in Eq. 1) to supply water for this entrainment

Predicted and measured temperature profiles for four dates in 1989-90 (1991-92) are shown in Fig. 6. The predicted profiles are the hindcast prediction (shoreline METRO discharge) and for the "selected" values of L, n, and d described below for D equal to 14 (consistent with the Onondaga County proposal) and 18 m. The June and September profiles show the effect of the relatively warm METRO effluent in a region above the diffuser. These profiles also show that with $D = 14$ m, a "three-layer" structure develops during summer stratification, where the diffuser influence is limited to the middle layer and temperature below 15 m is not significantly affected. Thus, the supersaturated effluent would not significantly affect dissolved oxygen in the lake below this depth. Anoxic conditions and associated sediment release of reduced species and phosphorus would likely continue below 15 m (Effler 1994). The predicted changes in fall turnover temperature and vertical structure during ice cover were small by comparison (Fig. 6).

Simulations also indicate that increasing the diffuser depth D tends to increase the depth at which the METRO effluent effectively enters the water column of the lake. Thus, in order to maximize the efficiency of hypolimnetic oxygenation and the isolation of the effluent from the surface waters, D should be set at as large a value as possible. However, placing the diffuser close to the maximum lake depth of 19.5 m would risk disturbing the contaminated sediments of the lake in the vicinity of the diffuser. For this reason, the maximum value of D considered in this study of lake-wide impacts was somewhat arbitrarily chosen to be 18 m. Selection of the maximum depth that is acceptable would have to consider both construction and operational effects on sediment disturbance. An outfall pipe length of about 1.5 km would be required to reach the 18 m depth (Owens 1987).

With $D = 18$ m established, a number of diffuser configurations were investigated. It was assumed that the proposed hypolimnetic discharge would achieve the maximum water quality benefits if the depth at which the METRO effluent is "trapped" below the thermocline during summer stratification is maximized. The only limitation on diffuser design considered was that head loss through the outfall pipe and diffuser not limit operation of the outfall by gravity under critical peak flow and lake level conditions. The maximum head available to drive the discharge by gravity was estimated to be 1.85 m (6 ft), based on design standards which call for continuous treatment at the peak flow of 160 mgd under the conditions of a 25 year flood Using assumed head loss coefficients for the outfall pipe and diffuser risers and ports, only those diffuser designs whose head loss did not exceed the maximum available head were considered to be feasible. Other design factors (cost, construction) would of course be considered in later design phases.

Based on the assumed coefficients, the head loss associated with a given design is dependent only on n and d. Analysis of various designs indicate the trapping of the effluent can be maximized by choosing a design which uses the maximum head available. There are various combinations of n and d which utilize the maximum head loss. For many such combinations of

n and d, trapping can be maximized by spacing the ports over a large enough diffuser length L so that the jets issuing from individual ports never merge. However, in some cases an optimal, intermediate value of L can be determined. Of the diffuser designs considered here and for the historic meteorologic and hydrologic condtions used, a diffuser design which resulted in maximum trapping was as follows: $L = 75$ m, $n = 35$, and $d = 25$ cm. For the purposes of this preliminary study, this will be described as the "selected" design.

The predicted variation of the "intrusion depth interval" for the selected design for the period of April 1989 through March 1990 is shown in Fig. 7. It can be seen that during spring (April 1989) and fall (Oct., Nov. 1989) turnover, the METRO effluent would most often rise to the water surface, due to the general lack of stratification during these periods. At the beginning of summer stratification, the depth interval is 9 to 12 m, and progressively becomes more shallow over the summer due to progressive warming and mixing of the hypolimnion. By the end of August, the intrusion depth interval is roughly 6 to 9 m.

Under winter conditions, the water column generally has weak "inverse" stratification, with water at the freezing point in contact with the ice cover overlying water at the bottom which is near the temperature of maximum density (~ 3.5°C). The METRO effluent, which has a temperature of 8 to 11°C during winter, is buoyant and begins to rise. However, entrainment of ambient lake water which is at a temperature below the temperature of maximum density results in a jet which becomes nonbuoyant after only a small vertical rise. For this reason, the METRO effluent is trapped at a depth just slightly above the diffuser depth for most of the winter, until the entire lower water column is heated to above the temperature of maximum density.

Discharge of the METRO effluent through the selected diffuser configuration at $D = 18$ m would have the effect of warming, thickening and homogenizing the hypolimnion during summer stratification (Fig. 6). The predicted maximum 7-day average lake bottom temperatures for each of the six years for the shoreline and hypolimnetic discharge conditions is given in Table

1. The average increase in maximum bottom temperature for the 6 years was predicted to be 4.7°C. The significant increase could eliminate the possibility of reclaiming the lake's cold-water fishery (Effler and Hennigan 1994). The greater vertical uniformity within the hypolimnion would cause various reactive substances associated with the sediment processes, that presently demonstrate strong vertical gradients within the hypolimnion (Addess and Effler 1994, Effler 1994), to be more uniformly distributed with depth. Additionally, this would tend to reduce the occurence or magnituce of salinity stratification that has persisted since closure of the soda ash/chlor-alkali facility (Effler and Owens 1994)

The predictions indicate that the increased vertical mixing in the water column would result in a decrease in the duration of summer stratification To quantify this effect, the onset of summer stratification is assumed to occur when a top-to-bottom density difference of at least 0.1 kg/m^3 is established and persists for the duration of the summer. Fall turnover is assumed to occur on the first date in fall when the density difference falls below this value. The dates of "spring onset" and "fall turnover' predicted by the model for each of the six years (stratification was established at the start of simulation in 1987) are listed in Table 2. The model predicted that spring onset would be delayed by an average of 12 days, and fall turnover would be advanced by an average of 16 days, such that the overall duration of summer stratification would be shortened by an average of 28 days. This reduction in duration of stratification by itself (not considering the effect of oxygenated METRO effluent) would likely have a positive impact on hypolimnetic anoxia, as the duration of isolation of the hypolimnion from surface reaeration would be reduced

The predicted impact of the hypolimnetic discharge on the dynamics of the depth of the thermocline (defined as the depth at which the maximum vertical gradient in density occurs) are depicted in Fig. 8 for the conditions of April 1991 through March 1992. The discharge is predicted to cause the thermocline to move higher in the water column. The predicted average decrease in the thermocline depth for the July to mid-September period is about 1 m.

Management Implications

This analysis indicates that the impacts of this proposed discharge are related to the high temperature of the METRO effluent relative to the hypolimnion of Onondaga Lake. Clearly, this proposal can achieve its intended purpose of improving lake water quality only if the effluent can be made to enter and mix with the waters of the hypolimnion during summer stratification. However, the buoyancy of the effluent tends to work against trapping of the effluent. When trapping is enhanced through the use of a multiport diffuser, the temperature of the hypolimion is increased. Potential deleterious effects of increased temperature are the loss of low temperatures necessary to support a cold-water fishery and increased rates of biochemical processes involved in oxygen depletion and nutrient cycling.

New York State has regulations which apply to thermal discharges. A section of the regulations state the following: "In lakes subject to stratification as defined in Part 652 of this Title, thermal discharges that will raise the temperature of the receiving water shall be confined to the epilimnion" (New York State 1986). It is not clear if a source of treated municipal wastewater can be considered a thermal discharge simply due to vertical location of the outfall in a lake. Nonetheless, the intent of the regulation appears to be to prevent a significant rise in the ambient temperature of lake waters. The advantage of releasing a typical thermal discharge (such as industrial cooling water) to the epilimnion rather than the hypolimnion is that a portion of the excess heat can be dissipated to the atmosphere through surface heat transfer processes. Trapping the METRO effluent in hypolimnion causes the excess heat to be retained in that layer during stratification.

The potential positive impacts of this proposal are the isolation of pollutants from the surface layer during summer stratification and the oxygenation of the hypolimnion. These effects have been evaluated quantitatively using mechanistic water quality models, as described in an accompanying paper (Effler et al. 1994). Predictions of this hydrothermal model which describe water temperature and vertical transport were used as inputs to these water quality models.

The model used herein to evaluate the hypolimnetic discharge proposal is based on the one-dimensional assumption; the model is thus not capable of simulating horizontal variation. Measurements of temperature and numerous mass constituents at stations in the two basins of the lake indicate horizontal uniformity. Such uniformity also existed in the years before the 1986 closure of the soda ash/chlor-alkali facility, which operated a hypolimnetic cooling water intake and epilimnetic diffuser at a flow roughly equal to the METRO discharge. Impacts of this proposal that would not be uniformly distributed over the plan area of the lake would tend to be more severe than predicted by this one-dimensional model. A more general three dimensional analysis may be appropriate to evaluate the details of near-field flow, temperature and concentration distributions, if the predicted lake-wide impacts are judged to be acceptable.

The appropriateness of the one-dimensional assumption can be judged quantitatively using a criterion based on a densimetric Froude number (Orlob 1983). For a lake or reservoir, the densimetric Froude number F_d is defined as

$$F_d = \frac{l}{d_m}\frac{Q}{V}\sqrt{\frac{\rho\, d_m}{\Delta\rho\, g}} \tag{8}$$

where l is the length of the lake, d_m is the mean depth, Q is flow through the lake, V is the lake volume, $\Delta\rho$ is the top-to-bottom density difference, and g is the acceleration of gravity. Orlob (1983) states that the one-dimensional assumption is valid if $F_d < 1/\pi$. Using $l = 7600$ m, $d_m = 11$ m, $V = 1.3 \times 10^8\,\mathrm{m}^3$ for Onondaga Lake, the one-dimensional assumption is valid if $\Delta\rho > 3.1 \times 10^{-7} Q^2$. Using Q equal to the average annual lake inflow of roughly $14\,\mathrm{m}^3/\mathrm{s}$ yields $\Delta\rho > 6 \times 10^{-5}\,\mathrm{kg/m}^3$, a value which is exceeded with any measurable stratification. If the conservative assumption is made that the entrained flow of ambient lake water into a diffuser acts as an "outflow" from the lake, the effective value of Q could be much higher. Using an entrained flow equal to 50 times the average METRO flow of $3.9\,\mathrm{m}^3/\mathrm{s}$ yields $Q = 195\,\mathrm{m}^3/\mathrm{s}$ and $\Delta\rho > 0.012\,\mathrm{kg/m}^3$ for the one-dimensional assumption, which is exceeded in virtually all stratified conditions including winter stratification (Effler and Owens 1994).

Simulations presented here (Fig. 8) indicate that the METRO effluent would be trapped at depth during winter stratification. However, reduction of the input of phosphorus and ammonia to the epilimnion is not as much of a concern in winter as it is in summer. As lake outflow occurs from the surface waters, use of the hypolimnetic discharge during winter may cause phosphorus and nitrogen concentrations at spring turnover to be higher than if the existing shoreline outfall was used in winter. These effects are investigated with the water quality models (Effler et al. 1994).

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Table 1. Predicted maximum 7-day average lake bottom temperature

Year	*Temperature, degrees C*		
	Hindcast	*Forecast*	*Change*
	10.4	**16.6**	**+6.2**
	8.1	**15.3**	**+7.2**
	12.4	**15.5**	**+3.1**
	14.7	**17.0**	**+3.3**
	14.2	**18.5**	**+4.3**
	11.9	**17.2**	**+5.3**

Table 2. Predicted change in spring onset of stratification and fall turnover

	Spring Onset			*Fall Turnover*		
Year	*Hindcast*	*Forecast*	*Change, days*	*Hindcast*	*Forecast*	*Change, days*
1987	–	–	–	Nov. 8	Oct. 4	31
1988	Apr. 15	Apr. 17	2	Oct. 26	Oct. 12	14
1989	May 9	May 12	3	Oct. 9	Sept. 29	10
1990	Apr. 25	May 25	30	Oct. 4	Sept. 23	11
1991	Apr. 25	May 10	15	Oct. 9	Sept. 27	12
1992	Apr. 17	May 9	12	Oct. 17	Sept. 29	18

Figure Captions

Figure 1. Relative temperature and density of METRO effluent and Onondaga Lake.

Figure 2. Schematic of plunging shoreline inflows and buoyant diffuser effluent.

Figure 3. Comparison of predicted and measured profiles for 1991

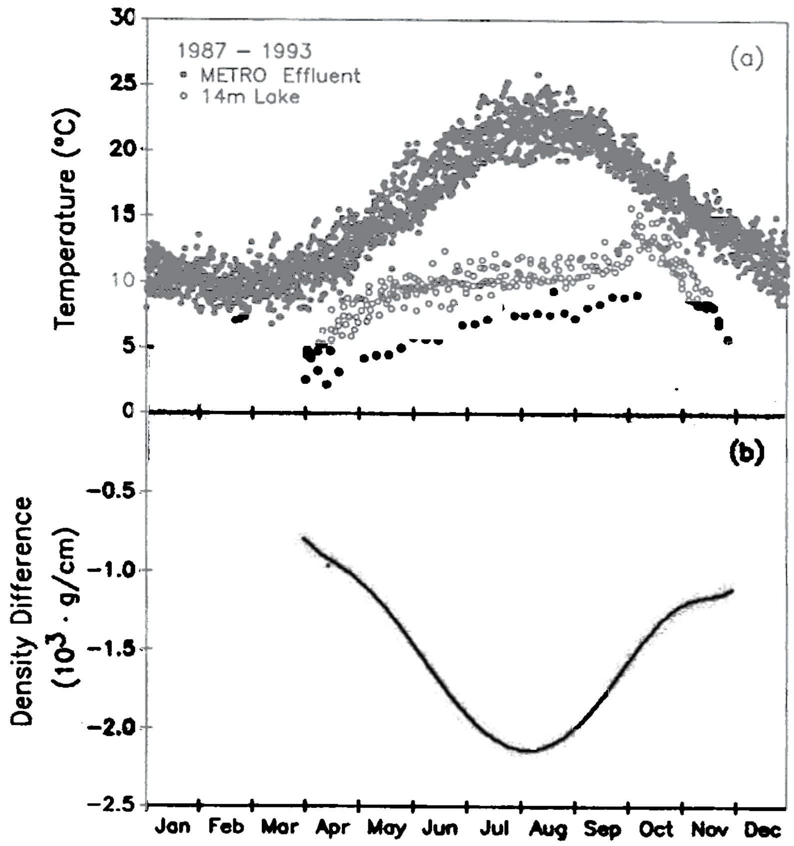
Figure 4. Comparison of predicted and measured layer-average temperatures for 1987-92.

Figure 5. Comparison of predicted and measured top-bottom density difference for 1987-92.

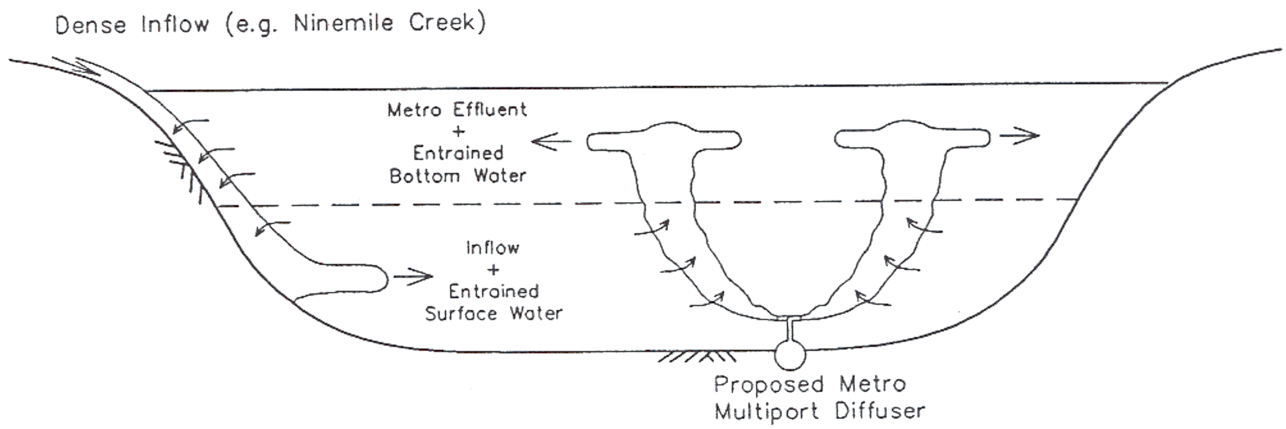
Figure 6. Selected predicted and measured temperature profiles from 1991-92.

Figure 7. Predicted intrusion depth interval for April 1989 through March 1990.

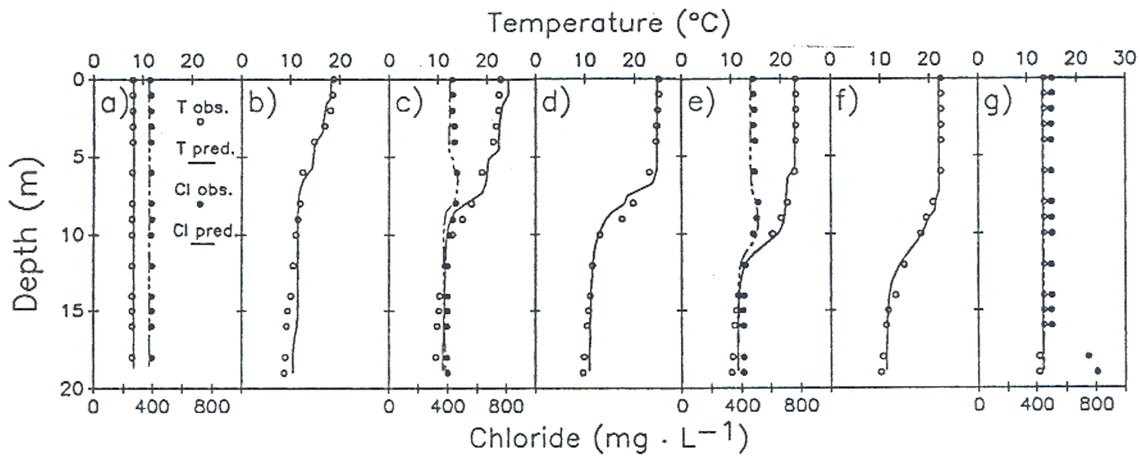
Figure 8. Predicted and measured thermocline depth for 1991-92.

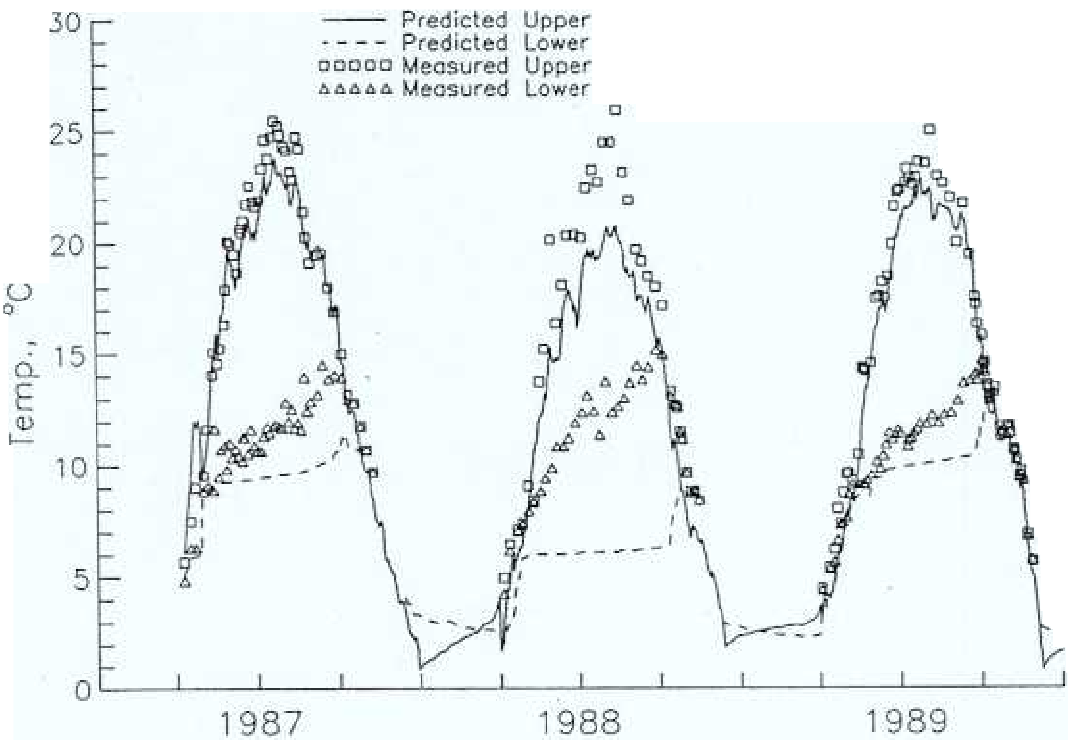


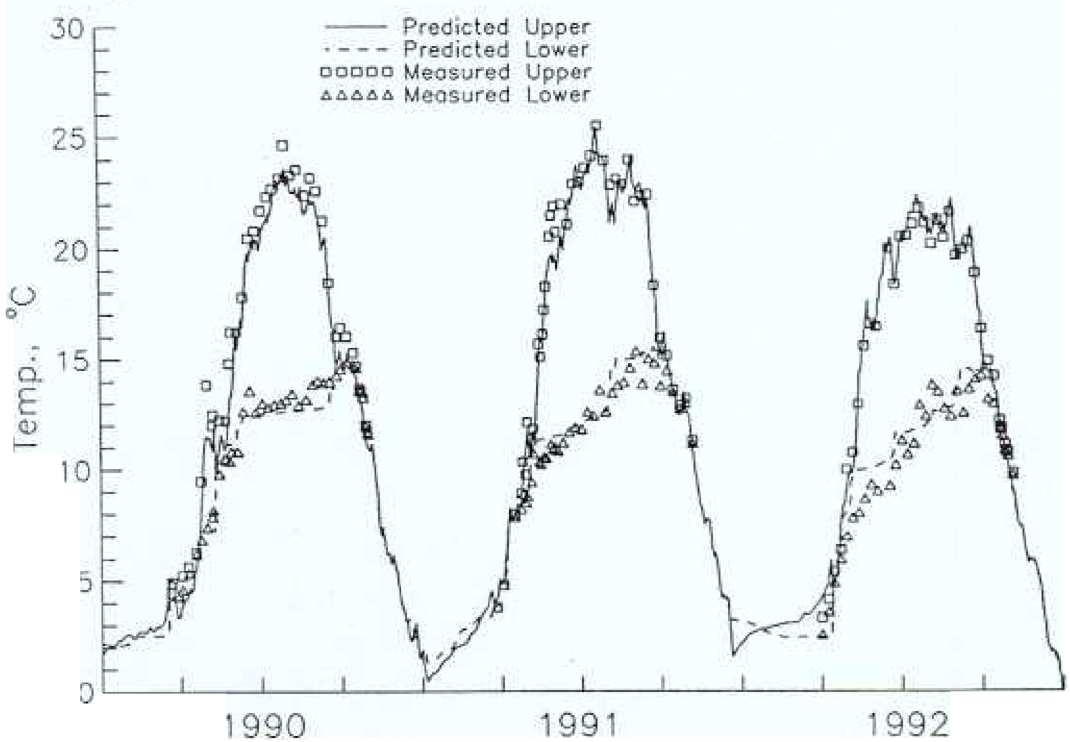
1987 – 1993
METRO Effluent
14m Lake
(a)
(b)
Temperature (°C)
Density Difference ($10^3 \cdot$ g/cm)
30
25
20
15
10
5
0
-0.5
-1.0
-1.5
-2.0
-2.5
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec

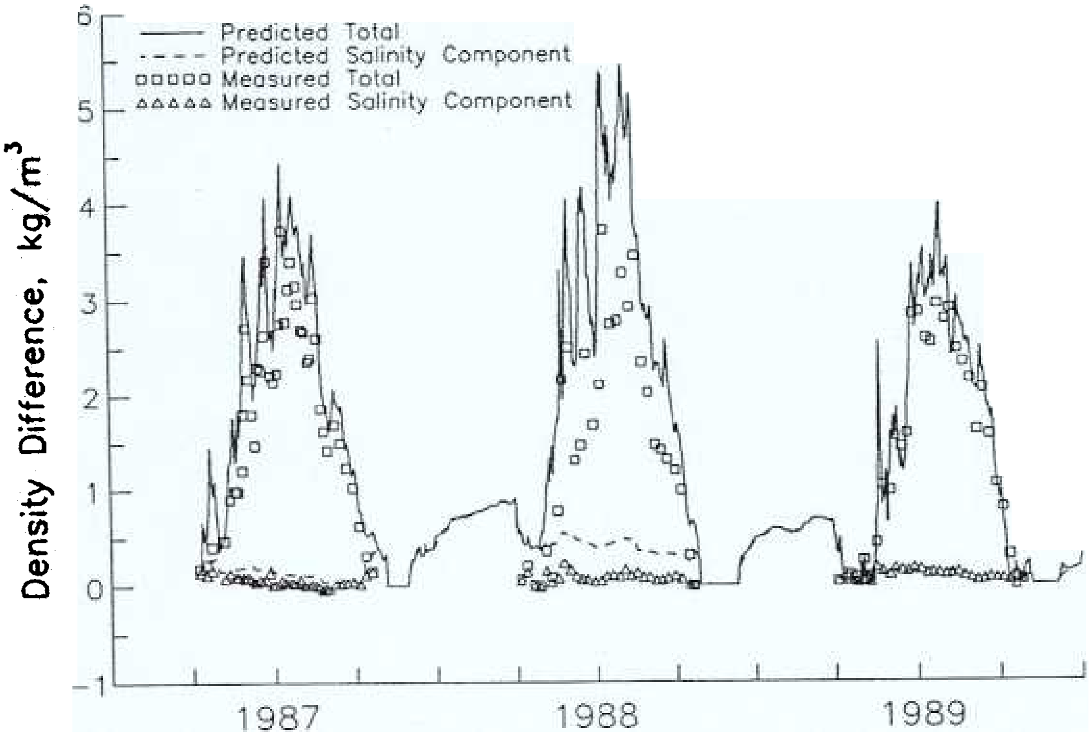


Schematic of Inflow Processes

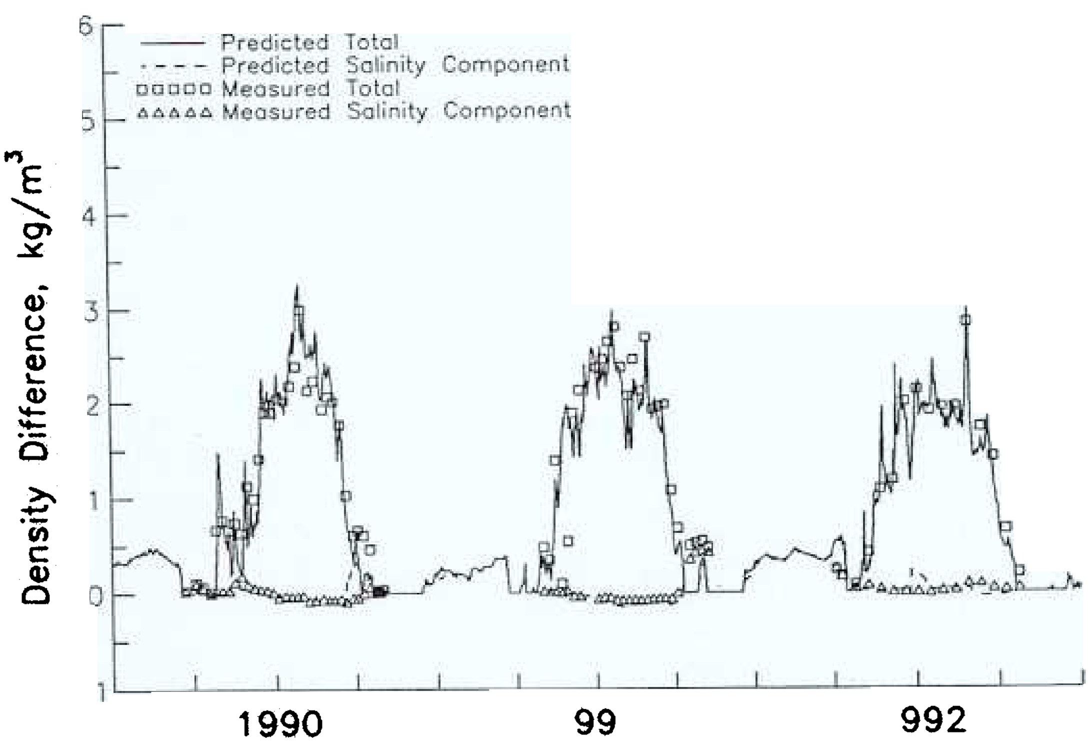

Temperature (°C)
0 10 20 0 10 20 0 10 20 0 10 20 0 10 20 0 10 20 0 10 20 30
a) b) c) d) e) f) g)
T obs.
T pred.
Cl obs.
Cl pred.
Depth (m)
0 5 10 15 20
0 400 800 0 400 800 0 400 800 0 400 800
Chloride (mg · L−1)


Predicted Upper
Predicted Lower
Measured Upper
Measured Lower
Temp., °C
30
25
20
15
10
5
0
1987
1988
1989

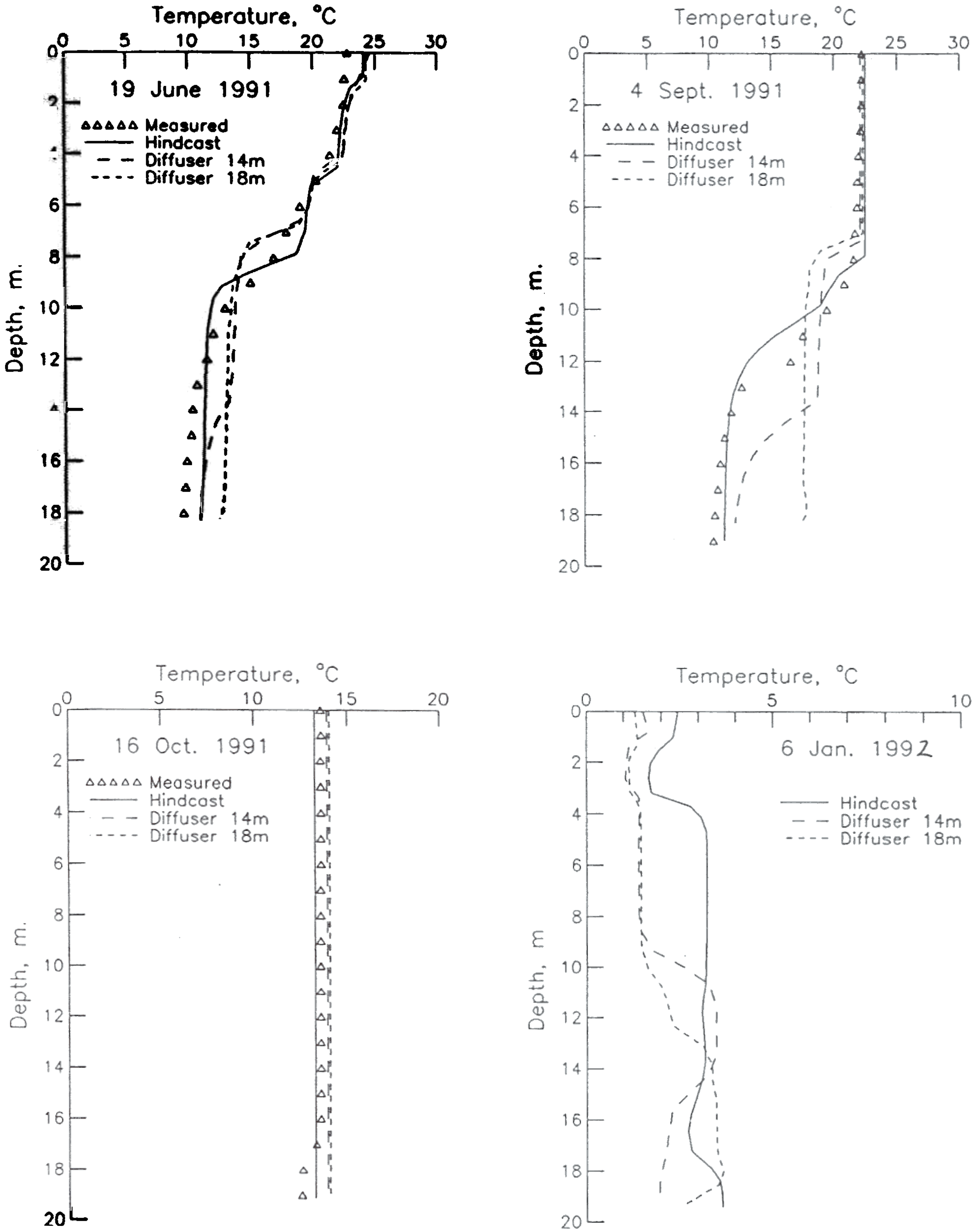

Predicted Upper
Predicted Lower
Measured Upper
Measured Lower
Temp., °C
30
25
20
15
10
5
0
1990
1991
1992



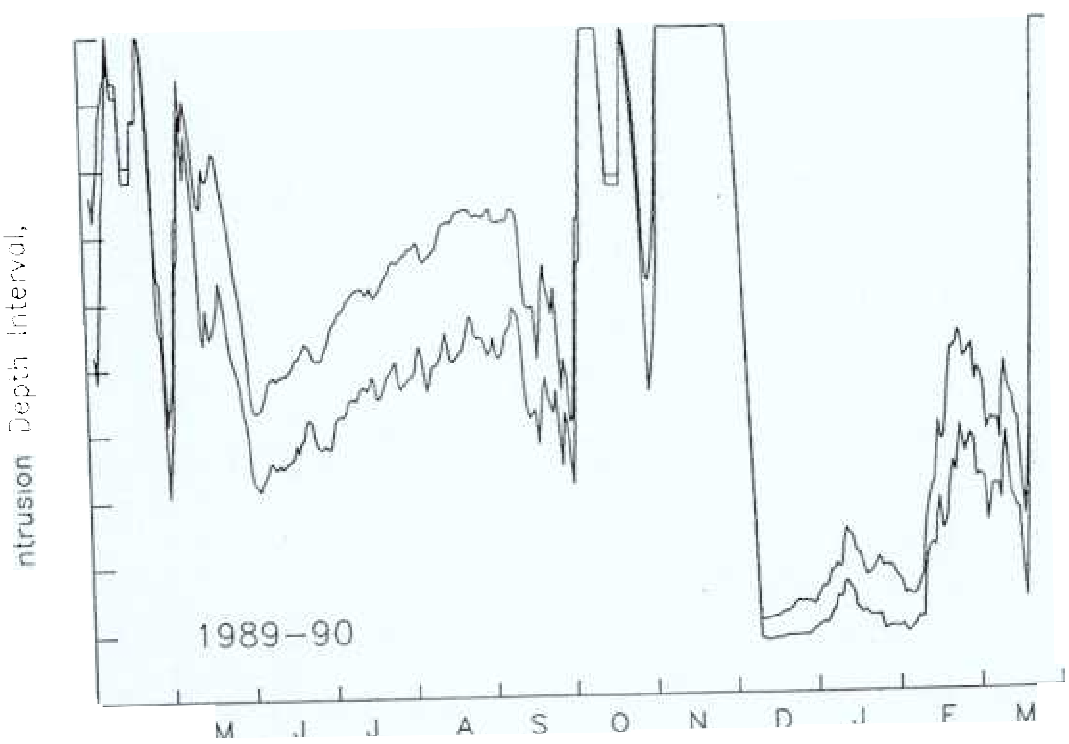
Predicted Total
Predicted Salinity Component
Measured Total
Measured Salinity Component
Density Difference, kg/m³
6
5
4
3
2
1
0
−1
1987
1988
1989

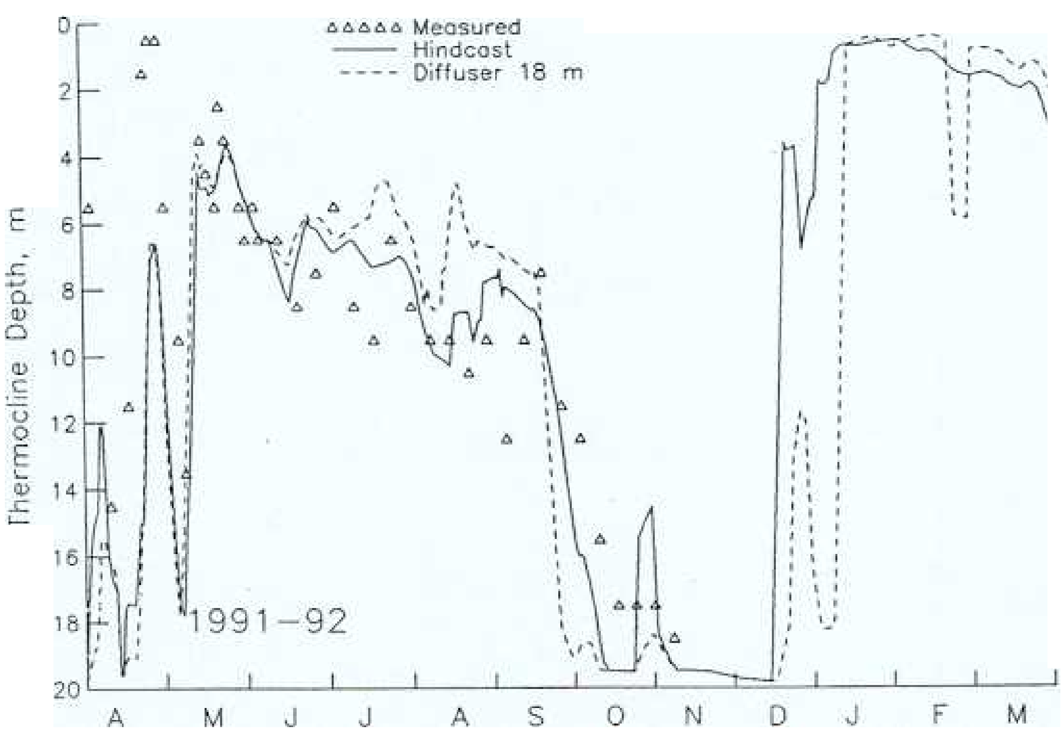


Predicted Total
Predicted Salinity Component
Measured Total
Measured Salinity Component
Density Difference, kg/m³
6
5
4
3
2
1
0
1
1990
99
992



Temperature, °C
19 June 1991
Measured
Hindcast
Diffuser 14m
Diffuser 18m
Depth, m.
4 Sept. 1991
16 Oct. 1991
6 Jan. 1992
Depth, m


ntrusion Depth Interval,
1989-90
M
J
J
A
S
O
N
D
J
F
M



Measured
Hindcast
Diffuser 18 m
Thermocline Depth, m
1991–92
0
2
4
6
8
10
12
14
16
18
20
A
M
J
J
A
S
O
N
D
J
F
M