Modeling the Impacts of a Proposed Hypolimnetic Wastewater Discharge on Stratification and Mixing in Onondaga Lake[1]

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Emmet M. Owens
Department of Civil and Environmental Engineering
Syracuse University, Syracuse, NY 13244

Steven W. Effler
Upstate Freshwater Institute
P.O. Box 506, Syracuse, NY 13214

ABSTRACT

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A one-dimensional hydrothermal model is used to forecast the impact of a proposed hypolimnetic discharge of treated municipal wastewater on stratification and mixing in Onondaga Lake, NY. 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. Features of the hydrothermal 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 successfully simulated six consecutive years of historical 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, such as intermittent formation of chemical stratification during spring and fall mixing.

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

The location of a wastewater outfall in a lake is generally selected to provide dilution of the effluent with ambient lake waters and to minimize the impact on beneficial uses of the lake. In many cases, this has been achieved by constructing an outfall pipe to an offshore location, thereby taking advantage of the increased water depth and more active ambient mixing relative to a shoreline location. An offshore location also tends to reduce the effects of the discharge on shoreline activities, such as bathing.

At sites on the coastal ocean where water depth increases relatively rapidly offshore, wastewater outfalls are often located below the thermocline (or more generally the pycnocline) in order to "trap" the wastewater below the upper mixed layer (Fischer et al. 1979). Trapping is achieved by inducing mixing of the effluent with deep ambient water, creating a mixed effluent density that is greater than that of ambient surface waters. Such mixing is typically induced by construction of a multiport diffuser on the outfall pipe (Grace 1978).

Onondaga Lake has been receiving effluent from wastewater treatment facilities serving metropolitan Syracuse, NY since the early part of this century (Effler and Hennigan 1996). Domestic wastewater treatment for a portion of Onondaga County that includes Syracuse was consolidated into a single facility, located at the southeast end of the lake, known as METRO (Effler and Hennigan 1996). Since this consolidation occurred, two outfall locations have been used for the METRO effluent. Prior to 1980, treated effluent was discharged approximately 520 meters from the southeast shore at a depth of 6 meters. As a part of a plan to divert saline industrial wastewater to the plant for use in phosphorus removal, the outfall was moved to the southeast shoreline in 1980 in order to minimize impact of the saline effluent on density stratification in the lake (USEPA 1974). At present, the outfall remains

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at this shoreline location despite the cessation of the saline industrial wastewater discharge in 1986.

In an effort to reduce water quality impacts associated with this effluent, Onondaga County (1994) has proposed to discharge oxygenated effluent from METRO to a diffuser located in the hypolimnion of Onondaga Lake. The discharge of treated wastewater below the thermocline of a lake is unusual, particularly considering the small ratio of lake volume to wastewater flow rate (i.e. low detention time) that would exist in this case. Potential benefits from this management action include increased isolation of METRO's phosphorus and ammonia load from the epilimnion by trapping the wastewater at the thermocline, and direct oxygenation of the lake's lower stratified layers during summer and winter. Presently, anoxia develops rapidly in the hypolimnion after the onset of summer stratification, followed by the accumulation of oxygen-demanding reduced species and lake-wide oxygen depletion in the upper waters during the fall mixing period (Addess and Effler 1996, Effler et al. 1996).

Here we analyze the lake-wide effects of this alternative discharge location on stratification and vertical mixing, particularly as they may influence water quality. The analysis is based on 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 1996). A submodel is included to describe the near-field mixing associated with a multiport diffuser to allow forecasting of the effects of the hypolimnetic discharge. Certain predictions of this model are used to specify thermal and transport/mixing information in a separate model analysis to forecast the water quality impacts of this alternate discharge location (Doerr et al. 1996).

Hypolimnetic Discharge Alternative

General specifications of the hypolimnetic discharge alternative were contained in a planning document (Onondaga County 1994), which called for discharge through an outfall pipe to a diffuser located at a depth of 14 m, at a peak flow of 7.0 m^3/s, or 160 million gal/day (MGD). The proposal also called for supersaturation of the effluent with 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 would flush the hypolimnion (assumed to be the volume below a depth of 8.5 m, or 47 x 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 consideration is the buoyancy of the METRO effluent relative to the waters of the hypolimnion of Onondaga Lake. Analysis of the annual variation of the METRO effluent temperature and of Onondaga Lake at a depth of 14 m (Effler and Owens 1996) for the 1987-1993 interval shows that the discharge is warmer throughout the year (Fig 1a). 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. With the outfall and associated diffuser located in the hypolimnion (depth D, Fig. 2), the effluent tends to rise in the water column, entraining ambient lake water as it rises (Fig. 2). The mixed effluent continues to rise until it reaches a depth where its density equals the density of the ambient lake (Fig. 2), where it then spreads horizontally. If the thermal stratification in the lake is weak or nonexistent, the mixed effluent rises to the water surface. The depth at which the mixed effluent spreads horizontally may be described as the effective inflow depth D_e (Fig. 2), which can be substantially less than

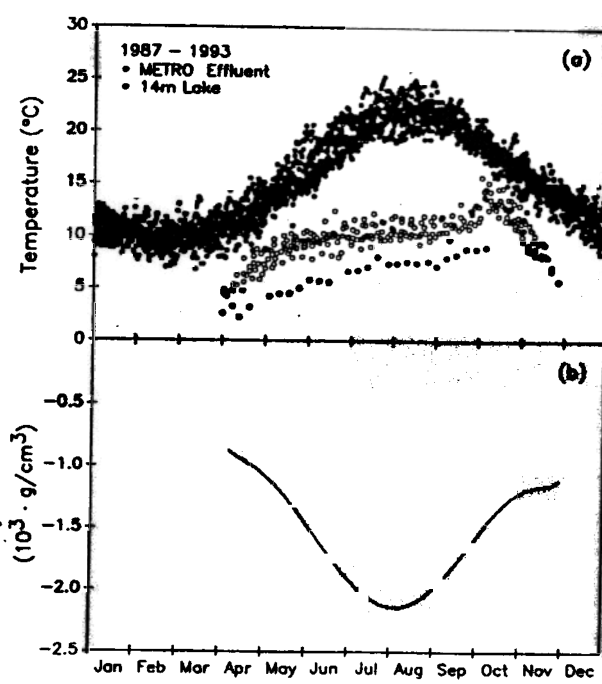

Figure 1.–Relative temperature and density of METRO effluent and Onondaga Lake (14 m). Lake temperatures measured at other depths during the winter months are below 4°C. Density difference is effluent less lake.

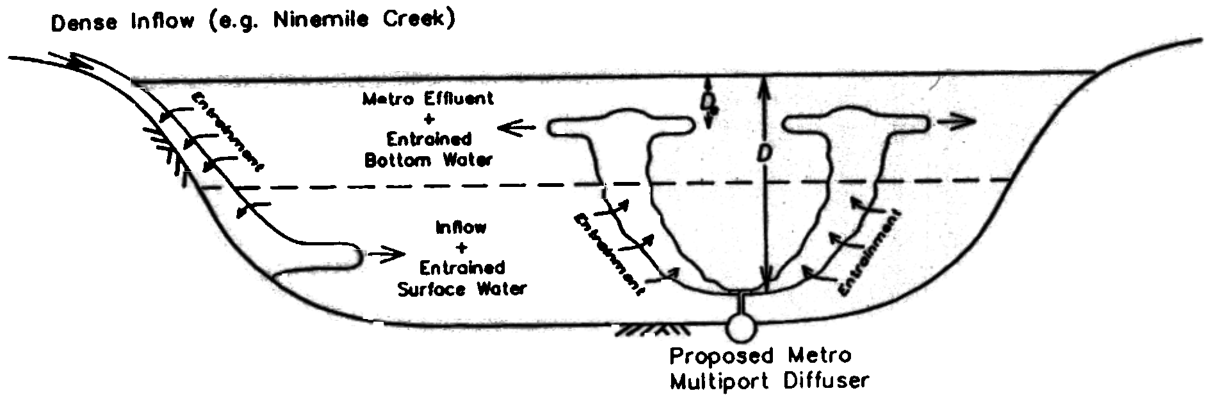

Figure 2.–Schematic of plunging shoreline inflows and buoyant diffuser effluent. Dense inflows entrain surface waters, and resulting mixed inflow enters the water column at depth. Buoyant effluent from the proposed hypolimnetic discharge rises in the water column, entraining ambient lake water, and spreads at depth of neutral buoyancy.

the diffuser depth D.

During winter, the METRO and lake temperatures straddle the temperature of maximum density (~3.5°C in Onondaga Lake due to its elevated salinity), which may allow for unusual buoyancy effects (Ford and Johnson 1983). The weak thermal stratification occurring in the fall-winter-spring period creates the conditions under which the salinity stratification in the lake (Effler and Owens 1996) can affect vertical transport and mixing.

Hydrothermal Model

Governing Equations

The model is one-dimensional (vertical), and thus predicts areally-averaged quantities; this general type of model was described by Harleman (1982). Due to the influence of salinity on buoyancy in Onondaga Lake (Effler and Owens 1996), both temperature and chloride ion (a surrogate measure of salinity) were simulated. The governing equations are the one-dimensional heat and mass conservation equations, which are:

$$\frac{\partial T}{\partial t} + w\frac{\partial T}{\partial z} = \frac{1}{A}\frac{\partial}{\partial z}\left(A K \frac{\partial T}{\partial z}\right) + \frac{1}{\rho c A}\frac{\partial}{\partial z}(A\phi) + \sum \frac{q_i}{A}(T_i - T) \tag{1}$$

$$\frac{\partial C}{\partial t} + w\frac{\partial C}{\partial z} = \frac{1}{A}\frac{\partial}{\partial z}\left(A K \frac{\partial C}{\partial z}\right) + \sum \frac{q_i}{A}(C_i - C) \tag{2}$$

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 near-field mixing submodels; two of these, associated with negatively-buoyant (plunging) shoreline inflow 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

$$-K\frac{\partial T}{\partial z} = k_s(T_S - T_E) + \frac{\beta\phi_{so}}{\rho c} \tag{3}$$

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 appropriateness of the one-dimensional assumption can be evaluated 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

$$F_d = \frac{l}{d_m}\frac{Q}{V}\sqrt{\frac{\rho d_m}{\Delta\rho g}} \tag{4}$$

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, $V = 1.3 \times 10^8$ 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 m^3/s yields $\Delta\rho > 6 \times 10^{-5}$ 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 buoyant diffuser plume 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 m^3/s yields Q = 195 m^3/s and $\Delta\rho >$ 0.012 kg/m^3 for the one-dimensional assumption, which is exceeded in virtually all stratified conditions including winter stratification (Effler and Owens 1996).

The model 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

$$\frac{dh}{dt} = \frac{C_p \sigma}{C_T + Ri} + \frac{1}{A_I}(Q_i - Q_o) \tag{5}$$

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_p, C_T, and η are empirical coefficients, $Ri = \sigma^2/g'h$ is the bulk Richardson number for the mixed layer, $g' = \Delta\rho g/\rho$, $\Delta\rho$ is the density increment at the base of 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 ($\sigma > 0$), Eq. 5 is used to compute its growth. When this is not the case, h will decrease such that $\sigma = 0$. Values of $C_p = 0.5$ and $C_T = 3.6$ 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 entrainment driven by plungi[ng] tributary inflows and buoyant treatment plant infl[ows] (Fig. 2). Plunging shoreline inflows entrain water fro[m] the surface, which then re-enters the water column [at] a greater depth. In an analagous manner, the buoya[nt] jets from the proposed multiport diffuser remove wat[er] from the surrounding ambient waters as they rise. Th[e] computation of such flows is described below.

The model uses an equation of state to comput[e] water density, and thus buoyancy, from temperatur[e] and chloride concentration. Based on measurement[s] of chloride ion (C, mg/L) and each of the major ion[s] contributing to salinity, an empirical relationship wa[s] developed to estimate salinity (S, ppt) from C (Effle[r] 1996). This relationship between C and S is given by

$$S = \begin{cases} 0.00152C + 0.589, \text{ if } C > 800 \\ 0.00178C + 0.381, \text{ if } C < 800. \end{cases} \tag{6}$$

Density was calculated based on T and S with an equation of state (Effler 1996) 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, Owens and Effler 1989). These impacts have been substantially reduced, but not eliminated, by the closure of the plant (Effler and Owens 1996).

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

$$K = \frac{C_H \varepsilon}{(N^2 + N_o^2)^r} \tag{7}$$

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.

The process associated with plunging shoreline inflows that must be quantified in the model is the entrainment of ambient lake water (Alavian et al. 1992). To describe entrainment in the vicinity of the shoreline, a constant entrance mixing coefficient S_g was used, such that the inflow rate of the density current after entrance was increased by a factor $(1+S_g)$. The temperature and chloride concentration of the inflow are also adjusted based on the characteristics of the surface layer of the lake. Following entrance mixing, entrainment of ambient lake water into the density current flowing down the sloping lake basin is described by

$$\frac{dQ}{dz} = S_l Q \qquad (8)$$

where Q is the flow rate of the plunging inflow, and S_l 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. This approach is consistent with investigations of plunging inflows as reviewed by Alavian et al. (1992).

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 horizontally in a normal direction to the axis of the diffuser manifold, and alternate in direction (Fig. 2). As wastewater exits an individual port, entrainment of ambient lake water is computed first by considering each round buoyant jet, assuming an equal distribution of flow to each port. After sufficient spreading 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 of Wright et al. (1982) were modified 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 assumes 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 effectively enters the water column over a range of depth centered on the effective inflow depth D_e (Fig. 2). The upper limit of this range is the depth of neutral buoyancy of the jet, while the lower limit is either the depth at which the jet ceases to rise or zero, whichever is greater. Based on the buoyancy of the METRO effluent (Fig. 1), the effluent may 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 performance of the stratification model was tested using data from the six-year period (1987-92) that followed closure of the soda ash/chlor-alkali facility. Ample data are available to describe the forcing conditions of meteorology and inflow hydrology over this period, and to provide in-lake information 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 National Weather Service (NWS) station; incident solar radiation measurements were used when available. Daily average flow rates of natural tributaries were measured by U. S. Geological Survey, 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 (1996). Weekly underwater measurements of ϕ_s yielded values of the extinction coefficient (Perkins and Effler 1996); daily values were estimated by interpolation. Model predictions were made at a 1-day time interval, so that diurnal variations in stratification were not considered.

Model Calibration/ Verification

Model calibration was performed by adjusting model coefficients in order to optimize agreement between predicted and measured profiles of temperature and chloride for the spring-fall period of 1992. The following values of model coefficients were determined: $\eta = 2.1$, $C_H = 0.00002$, $r = 0.30$, $S_E = 1.5$, and $S_I = 0.06$ 1/m. The values of four 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). To demonstrate model verification, predicted vertical profiles of temperature and chloride for the spring-fall interval of 1991 accurately reproduced

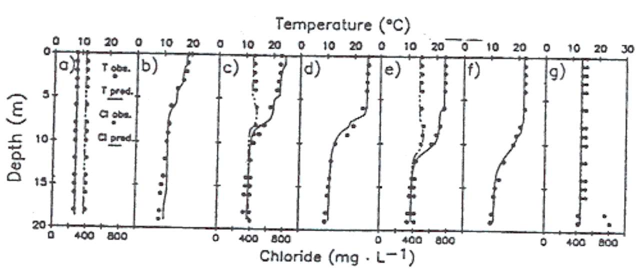

Figure 3.–Comparison of predicted and measured mid-month profiles for 1991. Specific dates are (a) 17 April; (b) 15 May; (c) 19 June; (d) 17 July; (e) 21 August; (f) 18 September; and (g) 16 October. The April, June, August, and October profiles show temperature and chloride; others show temperature only.

measured profiles (Fig. 3); the average differences between predicted and measured temperature and chloride for this simulation were 0.8°C and 28 mg/L, respectively. A shortcoming of this prediction is the failure to reproduce the elevated chloride which forms at the lake bottom in mid-October.

In order to summarize the predictions for the six-year period, statistics of the predicted and measured stratification conditions were determined. Stratification statistics computed were volume-weighted average T of the upper and lower (demarcated by the depth of 8.5 m) layers (Fig. 4), and total (top to bottom) density difference and the contribution of salinity to this density difference (Fig. 5). These statistics represent a substantial consolidation of the observed and simulated profiles, yet they describe important features of the lake's seasonal stratification regime and facilitate evaluation of model performance for multiple years. These statistics were based on a continuous model simulation for the six-year period, using measured initial conditions on April 8, 1987, and continuing through the end of 1992.

Simulations of the layer-average T (Fig. 4) show

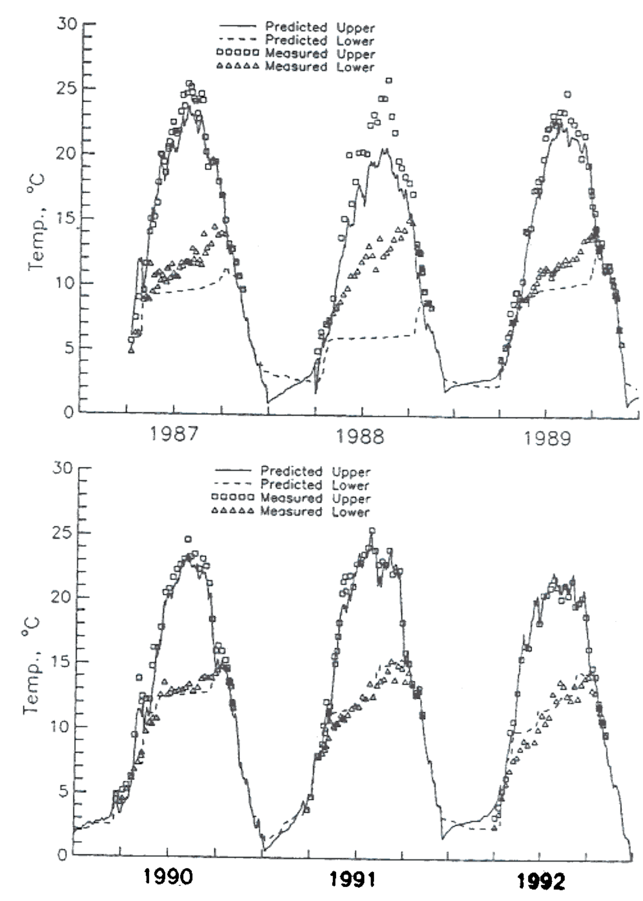

Figure 4. –Comparison of predicted and measured layer-average temperatures for 1987-92. Predicted results are for a continuous model simulation over this period.

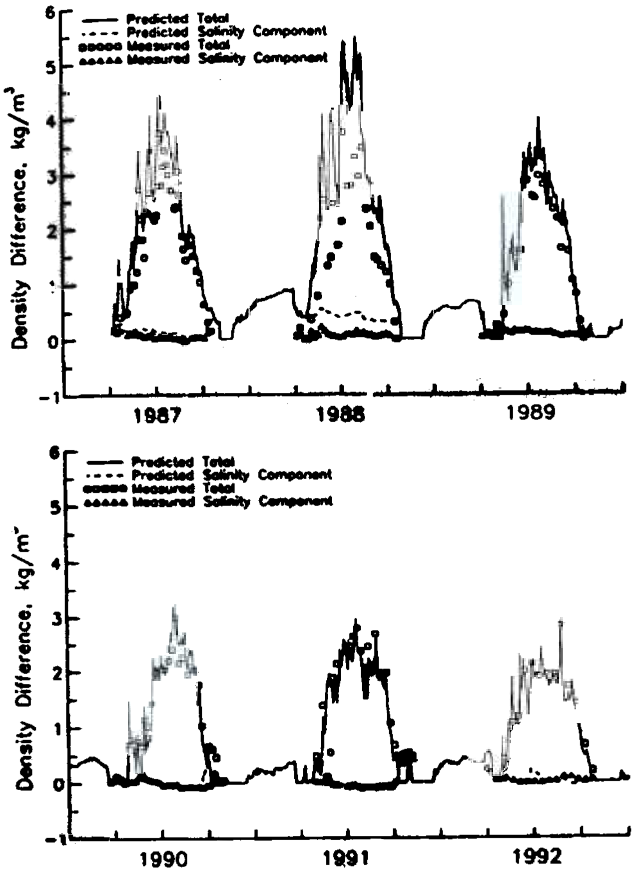

Figure 5.–Comparison of predicted and measured top-bottom density difference for 1987-92. Predicted results are for a continuous model simulation over this period.

that the model accurately predicted the timing and duration of stratification and the temperature of the upper waters for all six years. Simulation of the temperature of the lower waters was good for 1990 through 1992 (average error in layer-average T was 0.7°C), but was less accurate for 1987 through 1989 (average error 2.8°C). 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 1996).

Projected Effects of Hypolimnetic Discharge

Simulations of the impact of the proposed discharge were made for the same six 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. In order to calculate near-field mixing with this model, the following multiport diffuser characteristics must be specified: diffuser depth D, manifold length L, number of ports n, and port diameter d. Before addressing specific diffuser designs, the following general 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 Eqs. 1 and 2) to supply water for this entrainment.

The effect of moving the discharge to the hypolimnion was determined by comparing model predictions for historical periods (a hindcast) to forecasts for the same historical meteorological and hydrologic conditions with the exception that the METRO outfall is moved. Forecasts were made using the "selected" values of L, n, and d described below, and with D equal to 14 (consistent with the Onondaga County proposal) and 18 m. Temperature profiles in June and September (Fig. 6) show the effect of the relatively warm METRO effluent in a region above the diffuser. The simulations 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 (Fig. 6). 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 (Effler 1996) would likely continue below 15 m. The predicted changes in fall turnover temperature and vertical structure during ice cover were relatively small (Fig. 6).

Simulations also indicate that increasing the diffuser depth D leads to an increase in the depth D_t at

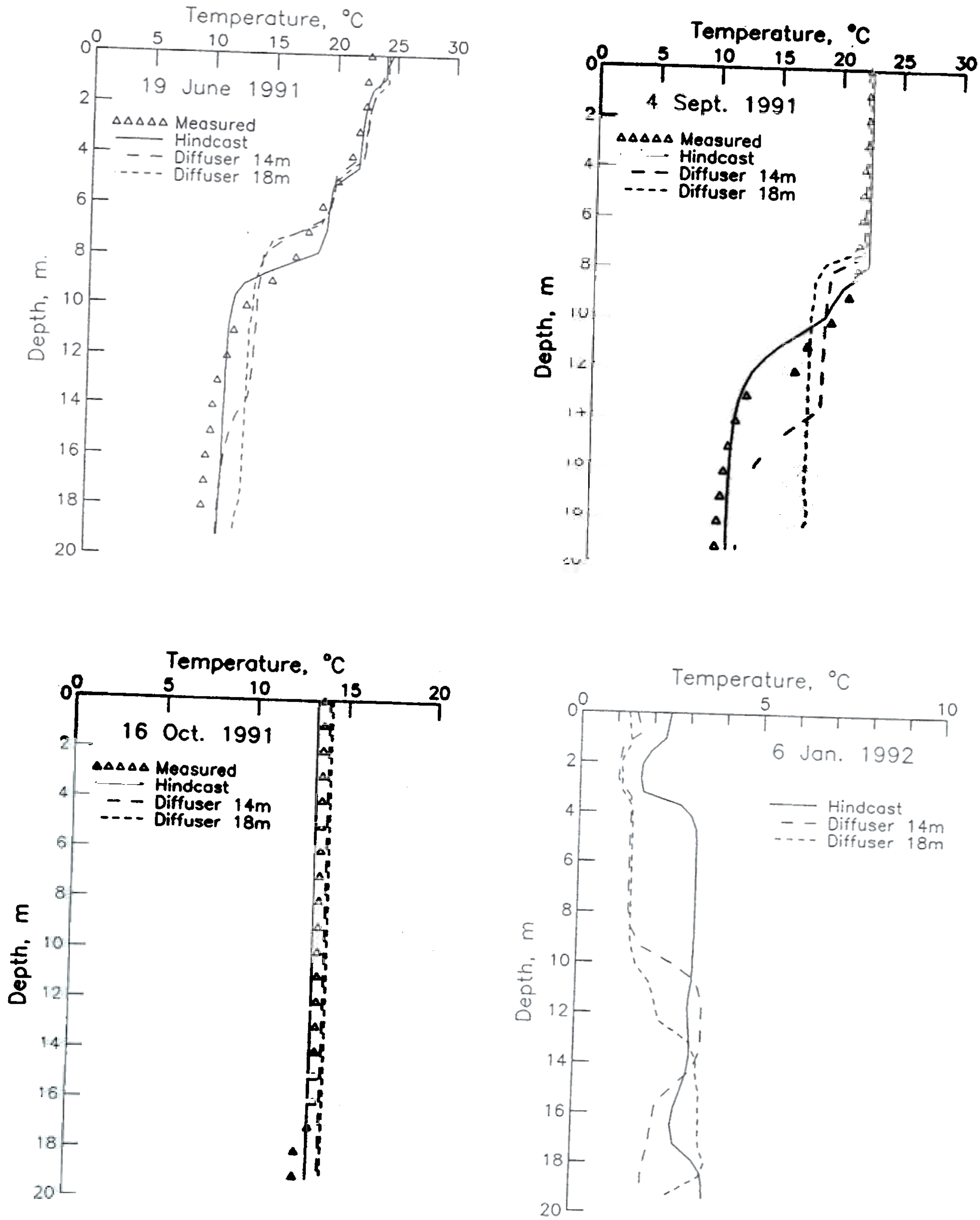

Figure 6. –Selected temperature profiles from 1991-92. The symbols indicate measured temperatures on these dates. The hindcast is a model prediction for the actual conditions during this period. The remaining predictions are with the selected diffuser design located at depths *D* equal to 14 and 18 m.

which the METRO effluent effectively enters the water column of the lake (Fig. 2). 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 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).

A number of diffuser configurations were investigated with $D = 18$ m. It was determined that the values of L, n, and d also affected the effective inflow depth D_e (Fig. 2). The only limitation on the values of L, n, and d considered was that the outfall system from the treatment plant to the lake operate by gravity (without pumping). 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 7.0 m^3/s (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, which can be achieved with various combinations of n and d. For many such combinations of n and d, the effective inflow depth D_e can be maximized by spacing the ports over a sufficiently large diffuser length L so that the jets issuing from individual ports do not 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 conditions used, a diffuser design which resulted in maximum D_e was as follows: $L = 75$ m, $n = 35$, and $d = 25$ cm. For the purposes of this study, this is described as the "selected" design.

The predicted variation of the effective inflow depth range (Fig. 2) for the selected design for the period of April 1989 through March 1990 was computed by the model (Fig. 7). 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 predicted inflow depth range 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 inflow depth range is roughly 6 to 9 m.

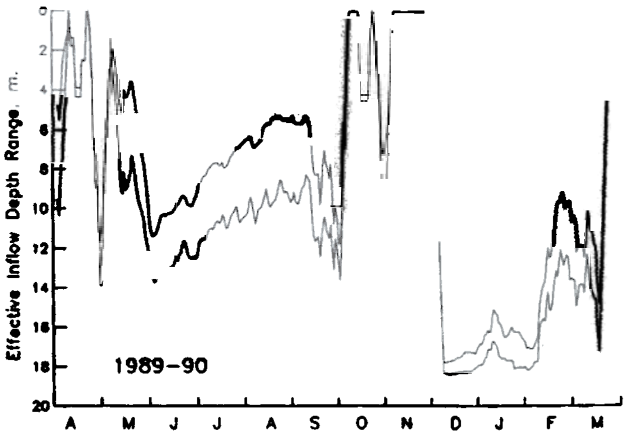

Figure 7.– Predicted range of the effective depth of inflow (D_e) for the METRO effluent for April 1989 through March 1990 using the selected diffuser design at a depth $D = 18$ m.

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 mixed effluent that is neutrally buoyant 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 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, homogenizing, and increasing the vertical extent of 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 locations (Table 1) shows the average predicted increase in maximum bottom temperature for the 6 years to be 4.7°C. This significant increase could eliminate the possibility of reclaiming the lake's cold-water fishery (Effler and Hennigan 1996). The greater vertical uniformity within the hypolimnion would cause various reactive substances associated with the sediment processes which presently demonstrate strong vertical

Table 1.–Model predictions of the maximum temperature at the bottom of Onondaga Lake averaged over a 7-day period. The hindcast is for historical conditions, while the forecast is with the proposed hypolimnetic discharge.

	Temperature, degrees C		
Year	Hindcast	Forecast	Change
1987	10.4	16.6	
1988	8.1	15.3	
1989	12.4	15.5	
1990	14.7	17.0	
1991	14.2	18.5	
1992	11.9	17.2	

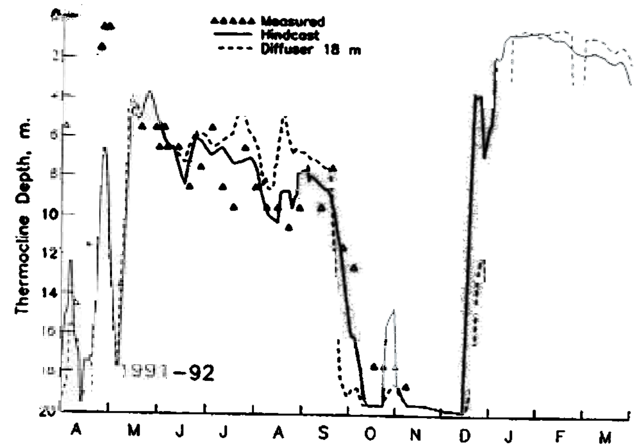

Figure 8.–Variation of the thermocline depth for April 1991 through March 1992. Measured values were determined as the depth of the maximum density gradient determined from measured temperature and chloride profiles. The hindcast is the model prediction for these historical conditions. The other prediction is for the selected diffuser design at depth *D* = 18 m.

gradients within the hypolimnion (Addess and Effler 1996, Effler 1996) to be more uniformly distributed with depth. Additionally, this would tend to reduce the occurrence or magnitude of salinity stratification that has persisted since closure of the soda ash/chlor-alkali facility (Effler and Owens 1996).

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 (equivalent to a temperature change from 8.0 to 9.6°C) 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) indicate 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 (Table 2). 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 model can be used to predict the 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) for the conditions of April 1991 through March 1992 (Fig. 8). The discharge is predicted to cause the thermocline to move higher in the water column during summer stratification. 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 effects of the proposed discharge are linked to the high temperature

Table 2.–Model predictions of the date of the onset of summer stratification and fall turnover. Hindcast predictions are for historical conditions, while the forecast is for the proposed hypolimnetic discharge.

	Spring Onset			Fall Turnover		
Year	Hindcast		Change, days	Hindcast		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

of the METRO effluent relative to the hypolimnion of Onondaga Lake. Clearly, the hypolimnetic discharge 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 in the hypolimnion. 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.

The temperature difference between the discharge and the receiving water at the point of discharge raises the question of whether the proposed hypolimnetic discharge should be treated as a thermal discharge. New York State regulations require: "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 to be 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 the hypolimnion causes the excess heat to be retained below the surface during stratification.

The potential positive impacts of the proposed hypolimnetic discharge 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 (Doerr et al. 1996). 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.

Simulations presented here (Fig. 8) indicate that the METRO effluent would be trapped at depth during winter stratification. However, reduction of the input of pollutants 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 pollutant concentrations at spring turnover to be higher than if the existing shoreline outfall was used in winter. This suggests that the use of the hypolimnetic discharge on a seasonal basis (during summer stratification only) may have some advantages.

The management issue addressed in this study is the effect of the location of a wastewater treatment outfall on the physical characteristics of a lake. The model simulations indicate that moving the METRO outfall from the shoreline to the hypolimnion of Onondaga Lake would have significant impact on lake temperature, stratification, and mixing. The hypolimnetic discharge alternative does effectively trap the effluent below the thermocline during much of the summer stratification period, but the effluent also causes the thermocline to move up during this period. The temperature of the hypolimnion is increased, and the duration of summer stratification is decreased.

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