Forecasting Impacts of a Hypolimnetic Wastewater Discharge on Lake Water Quality[1]

S. M. Doerr and S. W. Effler
Upstate Freshwater Institute
P.O. Box 506, Syracuse, NY 13214

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

ABSTRACT

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Water quality impacts of shifting the shoreline discharge of a major (3.5 m^3/s) domestic wastewater treatment facility (METRO) on polluted, eutrophic Onondaga Lake, NY, to a deep-water location are evaluated with three mechanistic water quality models. Transport and mixing inputs for the simulations are specified from the output of a separate hydrothermal model (Owens and Effler 1996). Model simulations indicate, that by trapping enriched effluent in the lower layers, summertime concentrations of total phosphorus (TP) and total ammonia ($T\text{-}NH_3$) would decrease in the upper mixed layer (0 to 4.5 m depth interval). Further reductions in the concentrations of those constituents would be realized by reducing sediment release, through maintenance of oxia in the hypolimnion. However, the decreases that could be achieved with the prevailing METRO effluent characteristics would not result in substantial reductions in phytoplankton growth nor prevent violation of the $T\text{-}NH_3$ standard. An extremely high degree of nitrification would be needed at METRO, particularly during the late fall to early spring interval (e.g., $T\text{-}NH_3$ concentrations < 3.6 mgN/L in effluent), to avoid violation of the $T\text{-}NH_3$ standard. Oxygen concentrations in the existing METRO effluent would have to be maintained at approximately 70 mg/L to ensure the hypolimnion remains oxic. To provide oxygen concentrations ≥ 5 mg/L in the hypolimnion, oxygen concentrations of ≈ 130 mg/L would have to be maintained in METRO's effluent.

Key Words: stratification, mixing, transport, water quality models, phosphorus, free ammonia, oxygen, effluent, treatment, deep-water discharge.

This study evaluates water quality impacts of a proposed deep-water discharge of effluent from a major (3.5 m^3/s (80 MGD)) municipal wastewater treatment plant (METRO) on Onondaga Lake, NY. The deleterious impacts of the existing shoreline discharge/loading conditions from METRO on the lake's water quality have been described elsewhere in this issue (Effler et al. 1996a). The fundamental management strategy for a deep-water discharge (e.g., Fischer et al. 1979) is to isolate external loads of nutrients and toxic species from upper productive layers during summer, by discharging through a submerged diffuser (e.g., Owens and Effler 1996).

The two-layer framework adopted for the Onondaga Lake total phosphorus (TP; Doerr et al. 1996), nitrogen (Canale et al. 1996), and oxygen (Gelda and Auer 1996) models has been widely used in modeling efforts for other dimictic lakes (e.g., Chapra and Reckhow 1983, Thomann and Mueller 1987), and is appropriate to evaluate the load-reduction and diversion alternatives considered for METRO by Effler and Doerr (1996). However this framework had to be modified to accommodate the altered stratification/transport regime that would prevail for a deep-water discharge (Owens and Effler 1996). The analysis presented in this manuscript documents the: 1) development and testing of a two-layer transport framework for the water quality models that accommodates the modified regime that would result from a deep-water discharge, 2) interfacing of the

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hydrothermal and water quality models, and 3) simulation of water quality characteristics of the lake for selected deep-water discharge scenarios.

A Modified Two-Layer Transport/Mixing Framework

Owens and Effler (1996) used a calibrated/verified multi-layer hydrothermal model to demonstrate the major impact a deep-water discharge of METRO would have on stratification/ transport/mixing in Onondaga Lake, and delineated a reasonable range of design features for a diffuser for a deep-water discharge. Appropriate accommodation of buoyancy (both positive and negative) effects was crucial to their analysis (Owens and Effler 1996) and is essential for the related water quality modeling analyses presented here.

Some of the shoreline inflows to Onondaga Lake, particularly Ninemile Creek, are negatively buoyant (Effler and Owens 1996) because of ionic enrichment (Effler et al. 1996b), and plunge in the lake (Owens and Effler 1996; see Fig. 1a). Plunging shoreline inflows entrain lake water and effectively transport lake water from shallow to deep layers (Fig. 1a). The proposed deep-water discharge of METRO (see Owens and Effler (1996) for preliminary design features of this proposal and Effler et al. (1996a) for typical METRO effluent characteristics) would be positively buoyant and tend to rise in the water column as a plume. Momentum and buoyancy-induced mixing of a submerged buoyant inflow would transport lake water upward from deep layers (Fig. 1a). These buoyancy effects induce a vertical cycling of water (advective transport), with associated heat and mass, by entraining ambient lake water.

The modified transport framework for the water quality models, shown schematically in Fig. 1b, retains the desired simplicity (Doerr et al. 1996) of the fixed boundary two-layer representation, yet accommodates the inflow and coupled advective transport processes associated with the atypical buoyancy effects. Inflows directly enter either of the two layers or are split between the layers. The advective transport from the upper layer to the lower layer associated with the entrainment of water from the upper layer (epilimnion) into a plunging inflow is designated Q_{EH} (Fig. 1b)

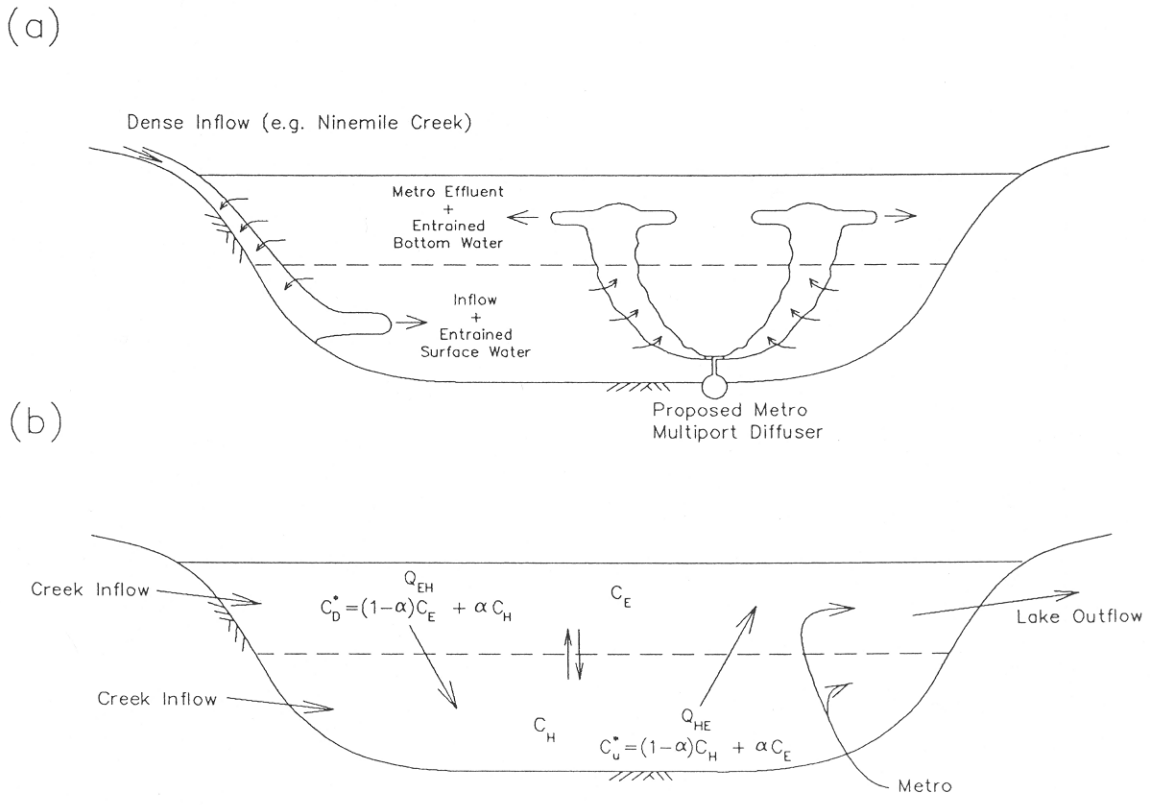

Figure 1.–(a) Transport processes associated with underflows and deep-water discharge of buoyant inflow, and (b) modified transp framework for two layer water quality models, to accommodate processes of (a).

Upward advective transport from the bottom layer (hypolimnion) to the top layer, Q_{HE} (Fig. 1b), results from the entrainment of ambient lake water from the lower layer into the rising plume. These advective transport components are also adjusted (either Q_{EH} or Q_{HE} is increased) to maintain the fixed layer water balance constraints of the framework (calculated by difference). The modifications in the transport framework (Fig. 1b) represent an increase in complexity that make the water quality models more mechanistically realistic by accommodating the prevailing plunging inflow phenomenon (Effler and Owens 1996, Owens and Effler 1996). Further, this modified framework provides for the alterations in vertical transport that would accompany the proposed deep-water discharge (Owens and Effler 1996).

Our modeling approach linked the multi-layer hydrothermal model (Owens and Effler 1996; via its output) with the modified two-layer water quality models directly (as input), and indirectly, via the vertical diffusion submodel (see Doerr et al. 1996) for the two layer framework (Fig. 2). Simulations of the stratification/mixing regime by the hydrothermal model are driven by meteorology, inflow hydrology and density (e.g., temperature and salinity (Effler and Owens 1996)), and , for METRO deep-water discharge scenarios, diffuser configuration (Owens and Effler 1996). Time series of volume-weighted temperatures of the two layers, inflows to the two layers, and exchange flows between the layers (according to a user-specified layer boundary) are generated as output files from the hydrothermal model. These drive the two-layer water quality models and vertical diffusion submodel (Fig. 2). The kinetic frameworks of the water quality models remain unchanged, though the kinetics of certain biochemical processes may be influenced by the deep-water discharge option due to resultant changes in layer temperatures (Owens and Effler 1996).

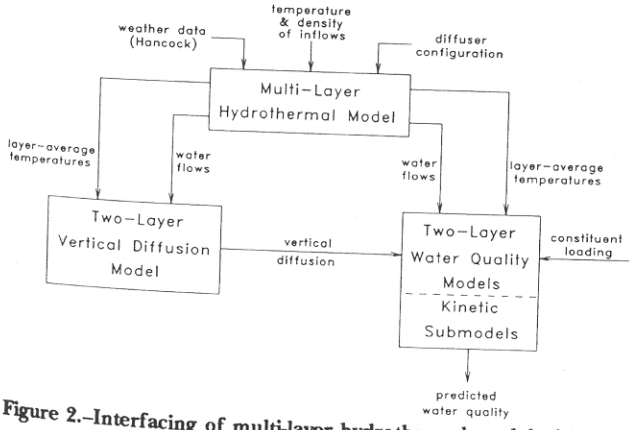

Figure 2.–Interfacing of multi-layer hydrothermal model with the more complex transport framework version of the two-layer water quality models, and the vertical diffusion submodel.

The concentrations C_U^* and C_D^* introduced in the modified framework (Fig. 1b) represent the constituent concentrations of water moving upward and downward across the layer boundary associated with the flows Q_{HE} and Q_{EH}, respectively. These concentrations are intermediate between the concentrations C_E and C_H (Fig. 1b) predicted by the two-layer water quality models. The relationships between the exchange flow concentrations and layer concentrations are described here by:

$$C_U^* = (1-\alpha)\, C_H + \alpha\, C_E \quad (1a)$$

$$C_D^* = (1-\alpha)\, C_E + \alpha\, C_H \quad (1b)$$

in which α = a dimensionless weighting factor between 0 and 1. Note that the representation of Eq. (1) is only applicable when advective vertical transport needs to be accommodated (e.g., Fig. 1a).

A comparative, conservative tracer analysis was conducted with the multi-layer hydrothermal model (the basis of evaluation) and the modified transport framework of the two-layer models (Fig. 1b) to determine α (Eq. (1)) and to establish the veracity of the modified framework in simulating the impact(s) of the proposed deep-water discharge. Loading conditions for the tracer analysis were set to mimic those for TP, a particularly critical constituent for this culturally eutrophic system (e.g., Effler et al. 1996a). In this simulation, the flow rates of the tributaries and METRO were set to their 1989 average values. The TP concentration of the tributary inflows was set to 110 µg/L, while that for METRO was specified as 550 µg/L. These values are representative of average TP concentrations for these inputs. The flow-weighted input concentration is 195 µg/L, which would also be the steady-state lake concentration for a completely-mixed system. Meteorological, tributary hydrology, and inflow density conditions were set to those measured in 1989 (e.g., Owens and Effler 1996). METRO discharge specifications were a discharge depth of 18 m and the diffuser configuration of Owens and Effler (1996). Simulations were initialized at spring turnover at a tracer concentration of 195 µg/L.

The predictions of the multi-layer hydrothermal model were compressed into two layers (volume weighting of multi-layer output) according to the boundary depth of 8.5 m used in the original (Canale et al. 1996, Doerr et al. 1996, Gelda and Auer 1996) two-layer water quality model simulations. The hydrothermal model predictions depict decreases in tracer concentration in the upper layer with the onset of summer stratification. These are attributable to the "trapping" of the METRO discharge in the lower layer and continued flushing of the upper layer (Fig. 3). The subsequent increases in tracer concentration in the upper layer in late summer largely reflect intrusion of

buoyant, enriched METRO inflow above the 8.5 m boundary. Increases in concentration in the lower layer throughout the stratification period are predicted (Fig. 3) in response to the deep-water discharge. Predictions with the two-layer model were made according to model linkages described previously (Fig. 2) for various specified values of α, with the goal of obtaining the optimal match with the multi-layer simulations (Fig. 3). The best closure of the two models was achieved with $\alpha \geq 0.5$. Performance of the two-layer framework was insensitive to increases in α above 0.5, as this influence was compensated for by changes in the computed distribution of the diffusion coefficient (see Fig. 2, diffusion submodel). Consequently, a value of $\alpha = 0.5$ was adopted for all the modified water quality models. The veracity of the modified two-layer framework is depicted by its nearly-equivalent simulation of tracer concentration to that obtained with the multi-layer model (Fig. 3). Performance for the upper layer, the primary focus of management concern (e.g., Effler

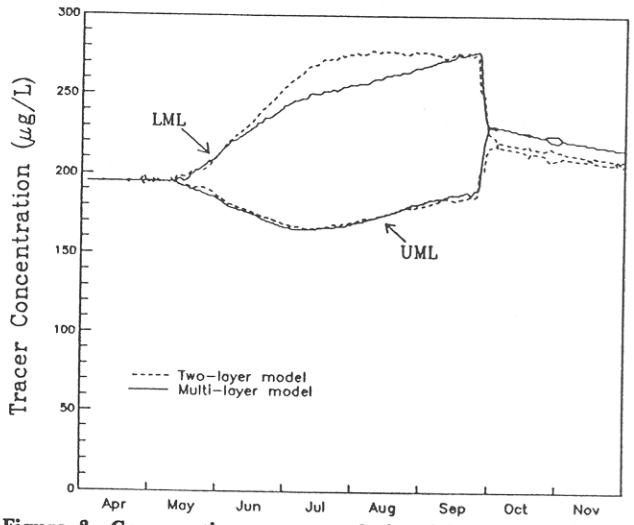

Figure 3.–Comparative tracer analysis with the multi-layer hydrothermal model (results presented for two-layer segmentation) and modified two-layer transport framework ($\alpha = 0.5$) of the water quality models. Demarcation depth of 8.5 m. LML - lower mixed layer; UML - mixed layer.

Table 1.–Summary of features of water quality models.

Feature(s)	Model		
	TP	N	DO
reference	Doerr et al. 1996	Canale et al. 1996	Gelda et al. 1996
management version	Effler and Doerr 1996	Effler and Doerr 1996	Effler and Doerr 1996
dimensionality	1 - vertical	1 - vertical	1 - vertical
layers	2	2	2
time	year-round, daily time step of loads and model solution	same as TP	same as TP
vertical diffusion estimates	from T profiles, with submodel (Doerr et al. 1996)	same as TP	same as TP
constituents externally loaded	TP	$T\text{-}NH_3$, NO_x, DON, d-PON	DO, CBOD
processes/kinetics	settling, vertical mixing sediment release	settling of PON, nitrification, denitrification, sediment release of $T\text{-}NH_3$ volatilization of NH_3 net uptake of $T\text{-}NH_3$/ formation of p-PON, due to phyto. growth hydrolysis of DON decomposition of d-PON	reaeration net production, due to phyto. growth exertion of NBOD exertion of CBOD RS oxidation SOD
constituents simulated	TP	p-PON, d-PON, PON, DON, $T\text{-}NH_3$, NO_x, TKN, TN	RS, DO, CBOD

abbreviations:
$T\text{-}NH_3$ - total ammonia
NH_3 - free ammonia
NO_x - sum of nitrate (NO_3) plus nitrite (NO_2)
DON - dissolved organic N
PON - particulate organic N
p-PON - phytoplankton PON
d-PON - detrital PON
TKN - total Kjeldahl N
TN - total N
T - temperature
SOD - sediment oxygen demand
RS - reduced species (in oxygen equivalents
CBOD - carbonaceous BOD
NBOD - nitrogenous BOD

prediction of spring concentrations. The under-the-ice simulations of the modified framework should be considered an enhancement that may track reality more closely. This change does not significantly influence subsequent simulations during the critical water quality period of summer (Fig. 4).

Water Quality Simulations

The analysis presented here is a screening-level effort intended to predict selected features of water quality in Onondaga Lake in response to a deep-water discharge for the METRO effluent. It should not be confused with a regulatory analysis. Note, for instance, there is no exhaustive effort to arrive at the most appropriate set of critical discharge (e.g., diffuser design, as described by Owens and Effler (1996)) and ambient environmental conditions, that may be necessary to support regulatory decision(s) and final engineering design. Rather, this analysis serves to evaluate the potential of realizing certain water quality benefits from the proposal.

All deep-water discharge evaluations have adopted the "selected" diffuser design specified by Owens and Effler (1996); depth of 18 m, diffuser length of 75 m, with 35 ports, and a port diameter of 25 cm. The effect of reasonable variations in the details of diffuser design (Owens and Effler 1996) on water quality predictions is minor. The analysis assumes that the deep-water discharge operates continuously, except as indicated. METRO discharge flows and constituent concentrations were those measured in 1989, except as specified otherwise. The seasonal distribution of "net" phytoplankton growth rate (see Canale et al. (1996) for definition) in the N model was recalibrated for a layer demarcation depth of 4.5 m (justified subsequently), assuming an unchanged distribution of chlorophyll (i.e., 1989) for the upper mixed layer. However, it is

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vertical diffusion estimates	from T profiles, with submodel (Doerr et al. 1996)	same as TP	same as TP
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constituents simulated	TP	p-PON, d-PON, PON, DON, $T\text{-}NH_3$, NO_x, TKN, TN	RS, DO, CBOD

abbreviations: $T\text{-}NH_3$ - total ammonia; NH_3 - free ammonia; NO_x - sum of nitrate (NO_3) plus nitrite (NO_2); DON - dissolved organic N; PON - particulate organic N; p-PON - phytoplankton PON; d-PON - detrital PON; TKN - total Kjeldahl N; TN - total N; T - temperature; SOD - sediment oxygen demand; RS - reduced species (in oxygen equivalents); CBOD - carbonaceous BOD; NBOD - nitrogenous BOD

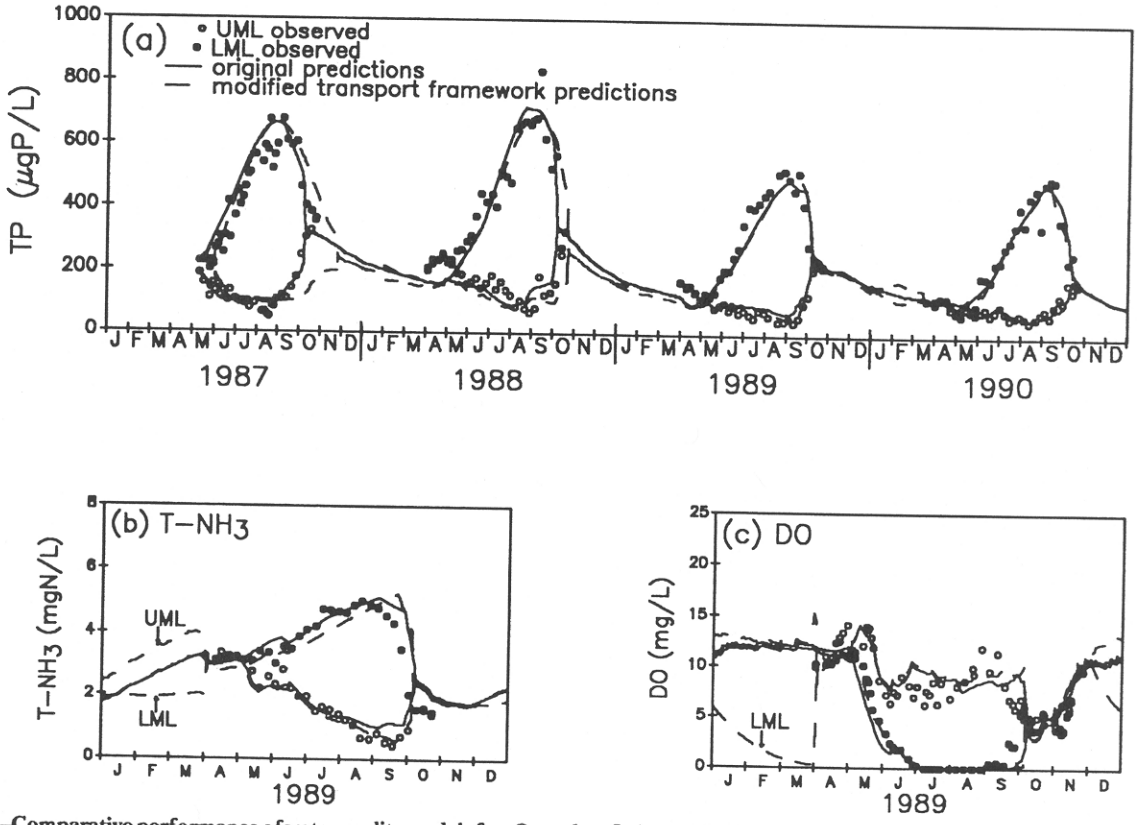

Figure 4.–Comparative performance of water quality models for Onondaga Lake, original and modified transport frameworks: (a) TP model, 1987-1990, (b) N model, for T-NH_3, 1989, and (c) DO model, 1989. LML - lower mixed layer; UML-upper mixed layer.

important to note that the decreases in the depth of the upper mixed layer predicted for the deep-water discharge scenarios (Owens and Effler 1996) would likely cause a systematic shift to higher concentrations of phytoplankton in that layer (e.g., Lorenzen and Mitchell 1973, 1975, Oskam 1973, Stefan et al. 1976), if existing nutrient conditions continued (Connors et al. 1996).

All model predictions correspond to the fourth consecutive year of simulations, to represent steady-state conditions for this rapidly flushed (Effler and Hennigan 1996) system. The base cases are the retest simulations made with the modified transport framework versions of the water quality models (Fig. 4).

TP Simulations

Simulations of TP for the upper layer illustrate the benefits of isolating METRO inflow from the upper productive waters (Fig. 5a), and the potential for reduced internal loading of phosphorus associated with maintaining oxia in the hypolimnion for the April-September interval (Fig. 5b). The oxia benefit may be described alternately as a "kinetic" benefit (Fig. 5b), as it is associated with changes in the values of kinetic coefficients attributable to oxia. Predicted summer

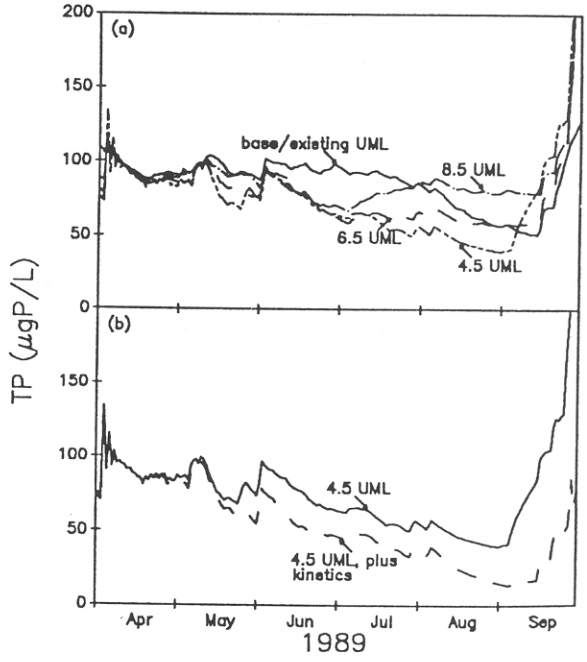

Figure 5.–Simulations of TP concentration for the upper mixed layer of Onondaga Lake for the METRO deep-water discharge proposal (18 m discharge depth, diffuser design according to Owens and Effler (1996)), for the conditions of 1989: (a) for model demarcation depths of 4.5, 6.5, and 8.5 m, compared to existing (base case) shoreline case, and (b) additional benefit of optimistic changes in kinetic coefficients (see Table 2) for a demarcation depth of 4.5 m, relative to the 4.5 m demarcation depth case with kinetic coefficients reflecting existing conditions.

average (mid-May to mid-September) epilimnetic TP concentrations ($[TP]_e$, µg/L) are presented for the various model runs (Table 2). The guidance value for $[TP]_e$ established in New York State is 20 µg/L (NYSDEC 1993; also see Effler et al. 1996a). Simulations are presented for demarcation depths of the two-layer model of 8.5 (adopted for original and retested models (e.g., Fig. 4)), 6.5, and 4.5 m (Fig. 5a). Kinetic conditions were the same as those developed for the original models (e.g., Table 2) for these runs. The substantial sensitivity to the demarcation depth (Fig. 5a and Table 2) largely results from predicted impacts of the deep-water discharge on the lake's stratification regime (Owens and Effler 1996). The upper mixed layer would be shallower, and the TP enriched buoyant METRO discharge would intrude into the lake's water column over the depth interval of 4.5 to 8.5 m in late summer (Owens and Effler 1995). The case of 4.5 m as the demarcation depth best reflects conditions that would prevail in the upper mixed layer throughout the summer with a deep-water discharge in operation (Owens and Effler 1996). The predicted value of $[TP]_e$ for this vertical segmentation is 63 µg/L. The progressively higher predicted concentrations (Fig. 5a, Table 2) for deeper model segmentations reflect greater relative contributions from the rising METRO discharge in late summer (Owens and Effler 1996). It should be noted that the vertical segmentation effect indicated here for $[TP]_e$ for the deep-water discharge (Table 2) is not manifested for this depth range under existing conditions, e.g., $[TP]_e$ values calculated from measured profiles generally do not differ significantly for these same layer boundary depths. Note the early and abrupt enrichment of the upper waters in September for the deep-water discharge scenario (Fig. 5) reflects a shortening of the stratification period, another impact predicted with the hydrothermal model (Owens and Effler 1996).

Table 2.–Summer average epilimnetic TP concentrations [TP]predicted with modified TP model for selected scenarios, for 1989 METRO conditions.

Case No.	Description	$[TP]_e$ µg/L
1.	existing conditions	83
2.	diffuser, 8.5 m model boundary, existing kinetics	82
3.	diffuser, 6.5 m model boundary, existing kinetics	69
4.	diffuser, 4.5 m model boundary, existing kinetics	63
5.*	diffuser, 4.5 m model boundary, kinetics changed - SRR = 3 mg/m²/d, $(PP/TP)_h$ set equal to measured conditions of $(PP/TP)_e$	42
6.	diffuser, 4.5 m model boundary, existing kinetics, effluent TP concentration = 0.2 mg/L (effluent filtration)	50

* $(PP/TP)_h$ - ratio value for bottom model layer; $(PP/TP)_e$ - ratio value for top model layer; SRR - sediment release rate for phosphorus.

The concern for TP concentration in the upper waters of a lake is predicated largely on the potential to support phytoplankton growth (e.g., Auer et al. 1986, Chapra and Dobson 1981, Vollenweider 1975, 1982). This interaction between the concentration of the critical nutrient and phytoplankton growth is operative only within the upper layers that receive adequate irradiance (Kirk 1983, Wetzel 1983). This interplay is limited to the upper mixed layer (Stefan et al. 1976) in waters with poor light penetration such as Onondaga Lake (Perkins and Effler 1996). For these reasons, we conclude that the 4.5 m demarcation depth scenario is the best representation to evaluate the trophic state implications of the deep-water discharge.

The proposed deepwater discharge could also change internal loading of P within the lake. Internal loading of P within the framework of the Onondaga Lake TP model is represented by two kinetic/stoichiometric inputs, the sediment release rate (SRR) and the particulate P (PP) to TP ratio (PP/TP; Doerr et al. 1996). Reduced, or negligible, values of SRR are expected for sediments with an oxidized microzone (i.e., oxic hypolimnion) relative to systems with anoxic hypolimnia (e.g., Bostrom and Petterson 1982, Fillos and Swanson 1975, Holdren and Armstrong 1980, Kamp-Nielson 1974, Mortimer 1941, 1942, 1971). However, significant P release from sediments under oxic conditions has been reported (e.g., Jensen and Andersen 1992, Kamp-Nielsen 1975, Lee et al. 1977, Sridharan and Lee 1974). Penn (1994) reported the seasonality of SRR for Onondaga Lake based on experiments with intact sediment cores. The minimum value of approximately 3 mgP/m²/d measured by Penn (1994) on cores collected in spring has been adopted here for the oxic hypolimnion analysis. This value is substantial for oxic conditions (e.g., Bostrom and Petterson 1982, Kamp-Nielson 1974, 1975). The PP/TP ratio in the hypolimnion results, in part, from the sediment release of P. To provide an upper bound for the potential benefit of maintaining oxygenated conditions, we used a time distribution for PP/TP for the lower layer equal to that specified for the upper layer (Doerr et al. 1996).

The conditions adopted for the oxygenated hypolimnion scenario are highly optimistic, representing a lower bound of $[TP]_e$ (Case 5, Table 2, 42 µg/L). If oxic conditions cannot be maintained, the most representative prediction for [TP]e for the deep-water discharge alternative corresponds to the 4.5 m segmentation depth simulation that does not invoke kinetic benefit (Case 4, Table 2). Additionally, a management scenario was evaluated that includes increased treatment associated with effluent filtration

at METRO (effluent TP concentration reduced to 0.2 mg/L (Effler and Doerr 1996)). The predicted summer average epilimnetic TP concentration for this case was 50 µg/L (Case 6, Table 2). At these concentrations, or even for the most optimistic projection (Table 2), little improvement in the lake's culturally eutrophic state would be expected. These concentrations exceed values associated with the eutrophic classification (Auer et al. 1986, Chapra and Dobson 1981, Vollenweider 1975, 1982).

T-NH_3 Simulations

Simulations with the modified N model (Fig. 6) address three issues: 1) the desirability of operating the deep-water discharge only during summer stratification instead of year-round (Fig. 6a), 2) the benefit of maintaining oxia in the hypolimnion to reduce T-NH_3 in the epilimnion (Fig. 6b), and 3) the METRO effluent concentrations that would be necessary to meet the T-NH_3 standard established for Onondaga Lake (Fig. 6c). The T-NH_3 standard established by the State of New York for the lake (0.77 mgN/L) to protect against the toxic effects of free ammonia (NH_3), based on analysis of lake pH and temperature data, appears for reference with all the simulations. All simulations are based on a 4.5 m boundary for the model layers, and 1989 METRO effluent and ambient characteristics (Fig. 7), except as specified otherwise.

The operation of a deep-water discharge year-round would have a deleterious effect on late spring/early summer T-NH_3 concentrations in the upper waters, compared to a "summer only" discharge scheme (Fig. 6a). Violations of the NH_3 standard are often most severe in the upper waters in this period (e.g., Effler et al. 1990, Effler et al. 1996a). The higher concentrations for year-round operation result from trapping much of the T-NH_3 from the METRO effluent in the lower layer during winter (Owens and Effler 1996). Distribution of the accumulated T-NH_3 throughout the water column with the onset of spring turnover causes an abrupt increase in concentrations in the upper layer (Fig. 6a). A seasonal operation scheme for the deep-water discharge (i.e., use of existing shoreline discharge for winter months and deep-water discharge in summer) would reduce the spring turnover concentration by allowing a portion of the wintertime METRO load to exit the lake before spring. However, severe violations of the T-NH_3 standard would continue to occur in the upper waters of the lake, particularly in late spring/early summer and fall (Fig. 6a), if METRO effluent were discharged into the hypolimnion of the lake with the prevailing N species concentrations (e.g., existing case in Fig. 7).

Further reductions in T-NH_3 concentrations in the upper layer could be achieved by maintaining oxic conditions and thereby promoting nitrification (see Canale et al. 1996) in the hypolimnion (see DO analysis in subsequent section). The benefit is depicted in simulations (Fig. 6b) for which a "summertime only" deep-water discharge scheme was adopted. Part of the apparent oxia benefit (Fig. 6b) is attributable to predicted increases in hypolimnetic temperature associated with the deep-water discharge (Owens and

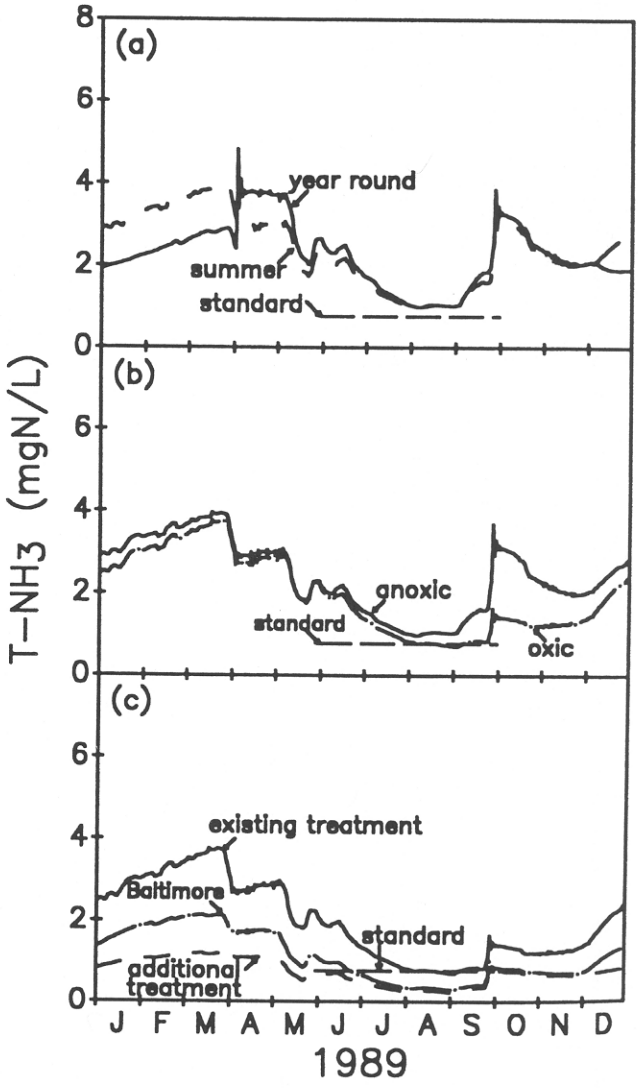

Figure 6.–Simulations of T-NH_3 for the upper mixed layer of Onondaga Lake for the METRO deep-water discharge proposal (18 m discharge depth, diffuser design according to Owens and Effler (1996), model demarcation depth of 4.5 m), for conditions of 1989: (a) evaluation of effect of year-round versus summertime only operation of the deep-water discharge, (b) benefit of maintaining oxia versus anoxia in hypolimnion, for a summertime only deep-water discharge, and (c) evaluation of the effect of increased treatment at METRO, for summertime only deep-water discharge and oxic hypolimnion, for three levels of treatment. Standard for T-NH_3 (0.77 mgN/L) included for reference.

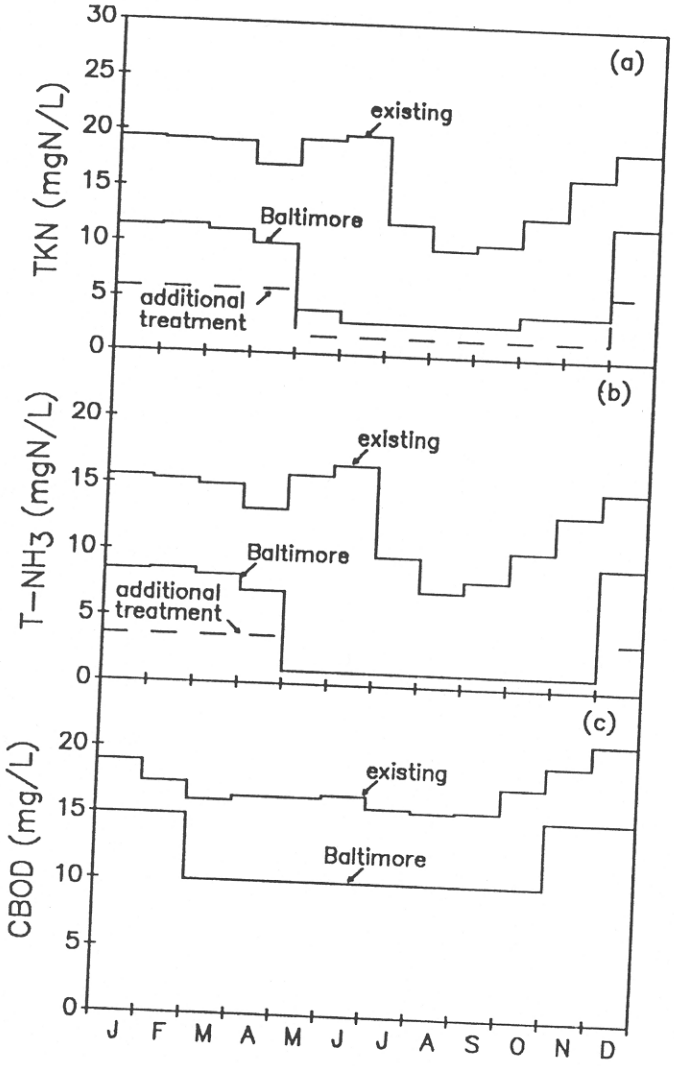

Figure 7.–Monthly average concentrations for METRO effluent for three deep-water discharge management scenarios, existing conditions at METRO, conditions for the City of Baltimore wastewater treatment facility, and concentrations necessary to avoid violations of the T-NH_3 standard in the upper mixed layer of Onondaga Lake ("additional treatment") for the tributary runoff conditions of 1989: (a) TKN, (b) T-NH_3, and (c) CBOD.

Effler 1996), as nitrification only operates in the N model at temperatures > 10°C (Canale et al. 1996). The effect on the upper layer for the critical late spring/early summer period would be small (Fig. 6b). However, the benefit becomes relatively greater subsequently, particularly with the approach to fall turnover (Fig. 6b), because of a reduced hypolimnetic pool of T-NH_3. For the case of existing METRO effluent characteristics, operation of the deep-water discharge during summer, and maintenance of oxic conditions in the hypolimnion, continued severe violation of the T-NH_3 standard is predicted (Fig. 6b).

Two model runs address the level of treatment necessary at METRO to avoid violation of the T-NH_3 standard. Maintenance of hypolimnetic oxia and summertime (only) operation of a deep-water discharge have been assumed for both analyses. The distributions of monthly average TKN and T-NH_3 concentrations adopted for the first scenario correspond to the operating characteristics of the Baltimore WWTP (~170 MGD) that achieves a high level of nitrification during the late spring-fall interval, but substantially less TKN/T-NH_3 treatment during the winter months (Fig. 7a and b). Unfortunately, wintertime nitrification performance is critical in controlling late spring/early summer T-NH_3 concentrations in Onondaga Lake (Effler et al. 1996a). Though the duration and margin of violation of the T-NH_3 standard for the upper waters of the lake would be greatly reduced by this improved level of treatment at METRO, violations would continue to occur in late spring/early summer for the conditions of 1989 (Fig. 6c).

The second simulation scenario corresponds to additional nitrification at METRO over the cold months (November through April) adequate to avoid violation of the T-NH_3 standard for the conditions of 1989. The T-NH_3 concentration of the effluent would have to be reduced to about 3.6 mgN/L during this interval (Fig. 7b; and TKN to about 5.8 mgN/L (Fig. 7a)) to avoid violation of the standard. Even greater levels of treatment would be necessary to avoid violations for reasonable critical conditions of low tributary runoff (e.g., 1987 runoff, one in ten year low flow, as described by Effler and Doerr (1996)), and projected increases in METRO discharge rates (Onondaga County 1994).

DO Simulations

Part of the proposal for the deep-water discharge of the METRO effluent calls for supersaturation of the effluent with oxygen, with the stated purpose of "maintaining oxic conditions in the hypolimnion, thereby preventing the recycle of P from the sediments" (Onondaga County 1994). We used the modified DO model to determine the necessary DO concentrations for the METRO effluent to reach two goals: 1) maintain oxic conditions within the hypolimnion throughout summer, and 2) maintain hypolimnetic concentrations of $\geq$ 5 mg/L (e.g., support fish survival in the hypolimnion).

The effluent DO concentration required to reach these goals decreases with increasing levels of treatment for oxygen demanding constituents, primarily carbonaceous and nitrogenous (e.g., T-NH_3 and organic N) BOD. Two different levels of treatment were evaluated: 1) existing treatment, and 2) the prevailing level of treatment at the Baltimore facility (Fig. 7). The

time distributions of TKN and CBOD in the METRO effluent are presented here as monthly averages for these two cases (Fig. 7).

We used the 4.5 m demarcation depth for simulations with the modified DO model, and we limited oxygenation of the METRO effluent to the period of summer stratification. Simulations of DO in the lower layer, for the improved level of treatment case for METRO (Baltimore), are presented for three different effluent DO concentrations (Fig. 8). The lower layer is predicted to become anoxic by mid-July, without effluent oxygenation; just slightly later than observed in 1989 (Gelda and Auer 1996). Increasing the effluent DO concentration to about 40 mg/L is predicted to be adequate to maintain oxic conditions through summer stratification; about 100 mg/L would be necessary to maintain ≥ 5 mg/L throughout the summer period (Fig. 8). The METRO effluent DO concentrations required to maintain oxia and ≥5 mg/L in the hypolimnion for the existing effluent quality are substantially higher; 70 and 130 mg/L, respectively.

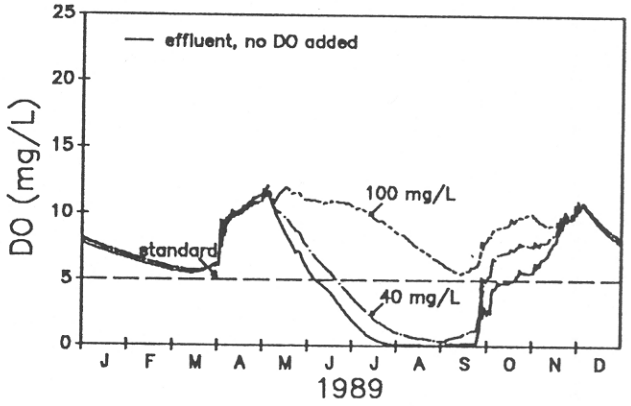

Figure 8.–Simulations of hypolimnetic DO concentrations in Onondaga Lake for the ambient conditions of 1989, for the deep-water discharge of METRO (18m discharge depth, diffuser design according to Owens and Effler (1996), model demarcation depth of 4.5 m), for the Baltimore WWTP TKN and CBOD effluent concentrations of Fig. 7 for three different effluent DO concentrations - existing, 40, and 100 mg/L. Predictions indicate 40 mg/L DO in effluent would maintain oxic conditions in the hypolimnion; 100 mg/L in effluent would maintain DO concentrations in the hypolimnion ≥ 5 mg/L.

Management Perspectives

This model analysis demonstrates that trapping of the METRO effluent below the upper mixed layer during summer stratification, by discharging this treated wastewater to the near bottom waters of the lake, would result in decreases in TP and T-NH_3 concentrations in the lake's upper waters. These reductions would be achieved at the price of profoundly altering the stratification/mixing regime of the lake (Owens and Effler 1996). Further, the decreases in TP and T-NH_3 concentrations would be inadequate to reach related water quality goals for the lake without additional treatment at METRO that may be beyond established technologies for large scale facilities. Phytoplankton growth probably would not be substantially limited. The in-lake standard for T-NH_3, to protect against the toxic effects of free ammonia, would continue to be violated unless major reductions in METRO's effluent T-NH_3 concentration (3.6<mgN/L) could be achieved for the non-summer months. Attention needs to be given to establishing appropriate critical conditions to support specification of effluent T-NH_3 limits for this management alternative. The deep-water discharge management alternative for METRO falls short of what could be achieved by diverting METRO around the lake (Effler and Doerr 1996), and has little to offer as an interim measure, for the lake's TP and T-NH_3 problems.

The situation is somewhat different for the oxygen resource implications of the proposal. It has been predicted that diversion of METRO would eventually eliminate the violation of DO standards in the upper waters during the fall mixing period (Effler and Doerr 1996). Hypolimnetic anoxia would be delayed compared to existing conditions, but not eliminated (Effler and Doerr 1996). The benefits of maintaining barely-oxic conditions in the hypolimnion are predicted to be marginal, as the associated reductions in internal loading of TP and T-NH_3 to the upper waters are inadequate to substantially influence phytoplankton growth or the lake's status with respect to the T-NH_3 standard. However, the maintenance of substantially higher hypolimnetic oxygen concentrations (≥5 mg/L) may be attractive to lake managers because of the potential recovery of fish habitat. The feasibility of attaining the oxygen concentrations in the METRO effluent necessary to achieve these in-lake oxygen concentrations needs to be established.

Managers should also be aware of a potential for increased concentrations of phytoplankton in the upper waters of the lake during late summer that might result from the reduction in the upper mixed layer depth associated with a deep-water discharge. The literature clearly supports this response, for the case of nutrient-saturated phytoplankton growth (e.g., Lorenzen and Mitchell 1973, 1975, Oskam 1973, Stefan et al. 1976).

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