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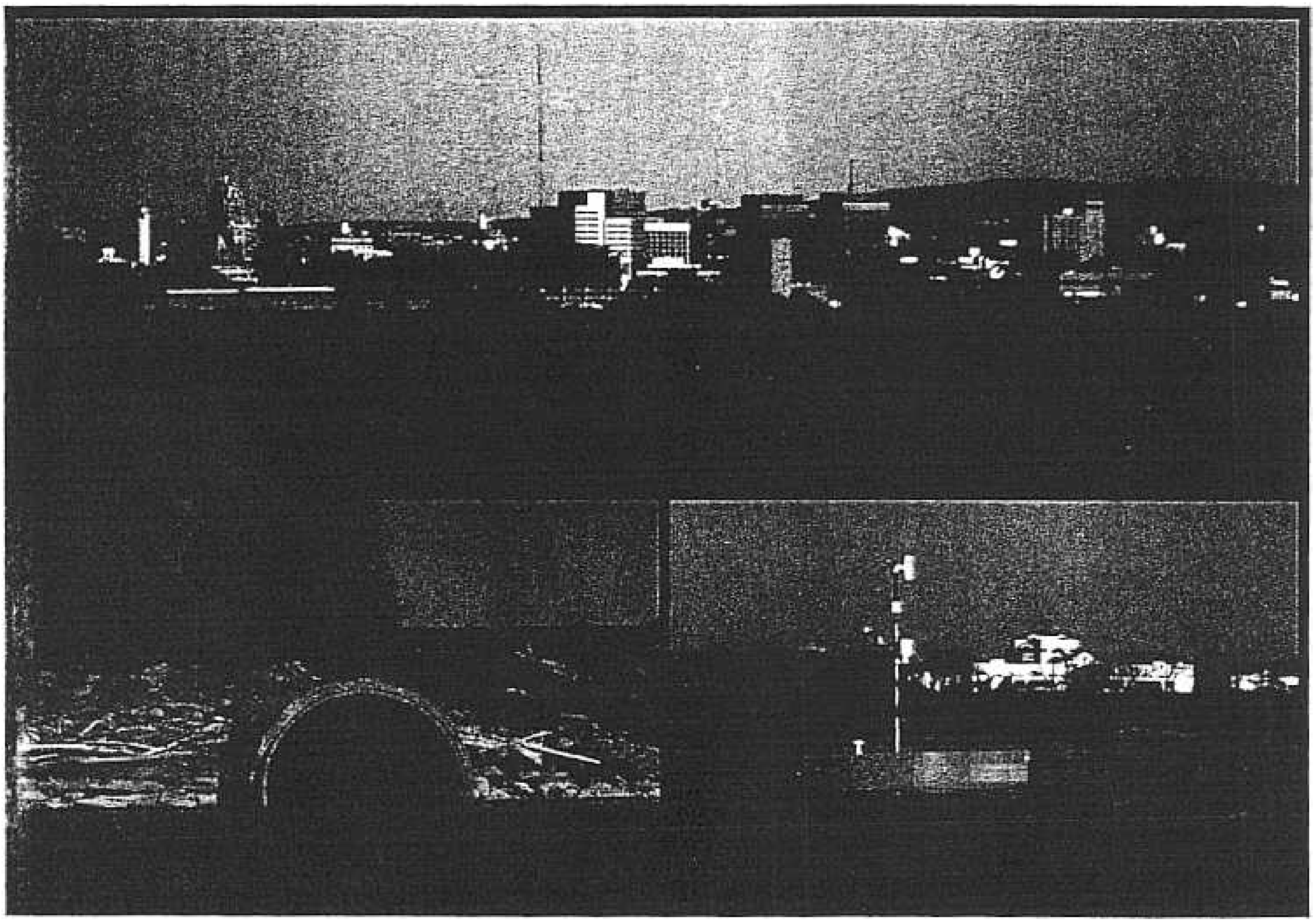

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Limnological and Loading Information and a Phosphorus Total Maximum Daily Load (TMDL) Analysis for Onondaga Lake[1]

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ABSTRACT

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The phosphorus (P) total maximum daily load (TMDL) analysis and associated management plan for culturally eutrophic Onondaga Lake, NY, are critically evaluated based on available input/discharge and limnological information for the system. The evaluation is based on: (1) results from a long-term monitoring program conducted on the lake, its tributaries, and the adjoining river that receives the lake's outflow, (2) algal bioassay experiments of the bioavailability of particulate P (PP) in inputs to the lake, (3) loading rate calculations for forms of P in these inputs, (4) calculations of water densities in inflows and the lake, (5) model analyses of plunging interflows and responses to seasonal material loading, and (6) mass balance calculations for a tracer conducted around the lake outlet and the receiving river to estimate inflow to the lake from the river. Several important system-specific characteristics were found not to be accommodated in the current TMDL analysis, including: (1) a P load from the river back into the lake, (2) seasonal plunging of tributaries to depths below the productive layers of the lake, (3) incomplete and different bioavailabilities of PP in the various inputs, (4) the different settling velocities of PP from these sources, (5) false high estimates of TP loading from tributaries associated with turbidity interferences in P analyses, and (6) the implications of the high flushing rate of the lake for strong seasonality in the relative impacts of externals loads. The TMDL analysis is demonstrated to understate the present role of the dominant point source and overstate the importance of non-point sources. Recommendations are made to upgrade the TMDL analysis through an integrated program of model development, testing and application, supporting process studies and monitoring, and re-evaluation of management options.

Key Words: bioavailability, deposition, phosphorus loading, seasonal flushing, TMDL analysis, underflows.

[1] Contribution No. 213 of the Upstate Freshwater Institute.

Effective integration of limnological information into regulatory initiatives is important in developing and executing strategies to rehabilitate impacted lakes and reservoirs (Cooke et al. 1993). A primary vehicle for this integration in the United States is the "total maximum daily load" (TMDL) process, that has been established by the U.S. Environmental Protection Agency (USEPA 1991a) as a quantitative regulatory framework to guide rehabilitative efforts to meet standards in impacted systems designated as "water quality limited" (as per section 303d of the Federal Clean Water Act). The TMDL is defined as the pollutant loading rate ($kg \cdot d^{-1}$) that will result in standards being met. The TMDL is the summation of three components (USEPA 1991a): (1) wasteload allocation (WLA; $kg \cdot d^{-1}$), the portion of the loading capacity allocated to point source inputs, (2) load allocation (LA; $kg \cdot d^{-1}$), the portion of the loading capacity attributed to non-point sources of pollution plus natural background inputs, and (3) margin of safety (MOS; $kg \cdot d^{-1}$). The MOS is intended to take into account any lack of knowledge concerning the relationship between the loadings of the pollutant and related features of receiving water quality; e.g., uncertainties in estimates of loads and the supporting model analysis, and variations in ambient conditions. The TMDL process advances earlier WLA programs (e.g., Thomann and Mueller 1987) by integrating all sources within a watershed, as well as MOS. A TMDL analysis is expected to accommodate important system-specific characteristics, critical environmental conditions, and recurring features of seasonality (USEPA 1991a).

A number of factors influence the complexity and outcome (i.e., rehabilitation costs) of TMDL analyses, including: (1) the identity and behavior of the target constituent, (2) the magnitude and format of the standard, (3) the number and character of sources of the constituent in the target watershed, (4) characteristics of the receiving water system, (5) characteristics and credibility of the scientific model, and (6) related features of regulatory policy. The understanding of the behavior of various pollutants ranges from poor to good, often corresponding to the level of effort devoted to related research. Standards are established by the states; these exist in numeric form for many pollutants. Watershed situations vary greatly in complexity; e.g., from the simple case of dominance of loading of the target constituent from a single point source, to significant contributions from a number of point and non-point sources. Despite extensive guidance for the conduct of the TMDL analysis process (e.g., USEPA 1991a, 1991b, 1997), there remain discretionary components (e.g., regulatory policy) that can have important implications for the outcome of the analysis and thereby the success of related rehabilitation efforts.

The necessary quantitative linkage between external loads and receiving water concentrations of a pollutant in a TMDL analysis is an appropriate mathematical model. The model should represent a synthesis of the understanding of the system and the behavior of the constituent, and therefore the results of supporting monitoring and related scientific (e.g., process) studies (Thomann and Mueller 1987, Chapra 1997). Further, the model's structure should be consistent with the format of the standards (USEPA 1991b); e.g., appropriate resolution in time and space.

Despite major advancements in the removal of phosphorus (P) at wastewater treatment plants (WWTPs) and control of non-point sources of P over the last several decades (Metcalf and Eddy, Inc. 1991, Cooke et al. 1993), numerous lakes and reservoirs continue to suffer water quality impacts of cultural eutrophication driven by excessive P loading. This paper reviews salient features of a TMDL analysis and resulting management plan for P for a culturally eutrophic urban lake. The TMDL analysis and management plan for the lake are critically evaluated within the context of their consistency with available tributary/discharge and limnological information for the system. This case study is valuable not only because of the great challenge to rehabilitation offered by this extremely degraded system, but also because issues of broad concern are addressed, including: (1) contrasting bioavailability of different P sources, (2) the importance of considering all sources, (3) the occurrences and implications of input(s) entering as an underflow(s) (i.e., entry into sub-surface layers of a lake), (4) heterogeneity in origins, character, and distribution of P sources, and (5) the effects of the magnitude and seasonality of lake flushing.

System Description and TMDL Analysis

Onondaga Lake and Inflows

Onondaga Lake is located (lat. 43° 06' 54"; long. 76° 14' 34") in metropolitan Syracuse, NY, in Onondaga County (Fig. 1). The lake is an alkaline dimictic system. This lake has a volume of 131×10^6 m^3, a surface area of 12.0 km^2, a maximum depth of ~20 m and a watershed area of 738 km^2. The lake flushes rapidly (average of ~4 times $\cdot$ y^{-1}, on a completely-mixed basis), though strong seasonal and interannual variations occur (Effler and Whitehead 1996). The lake discharges through a single outlet (1.9 km long, 4.5 m depth) to the Seneca River (Fig. 1). Intrusion of Seneca River water into

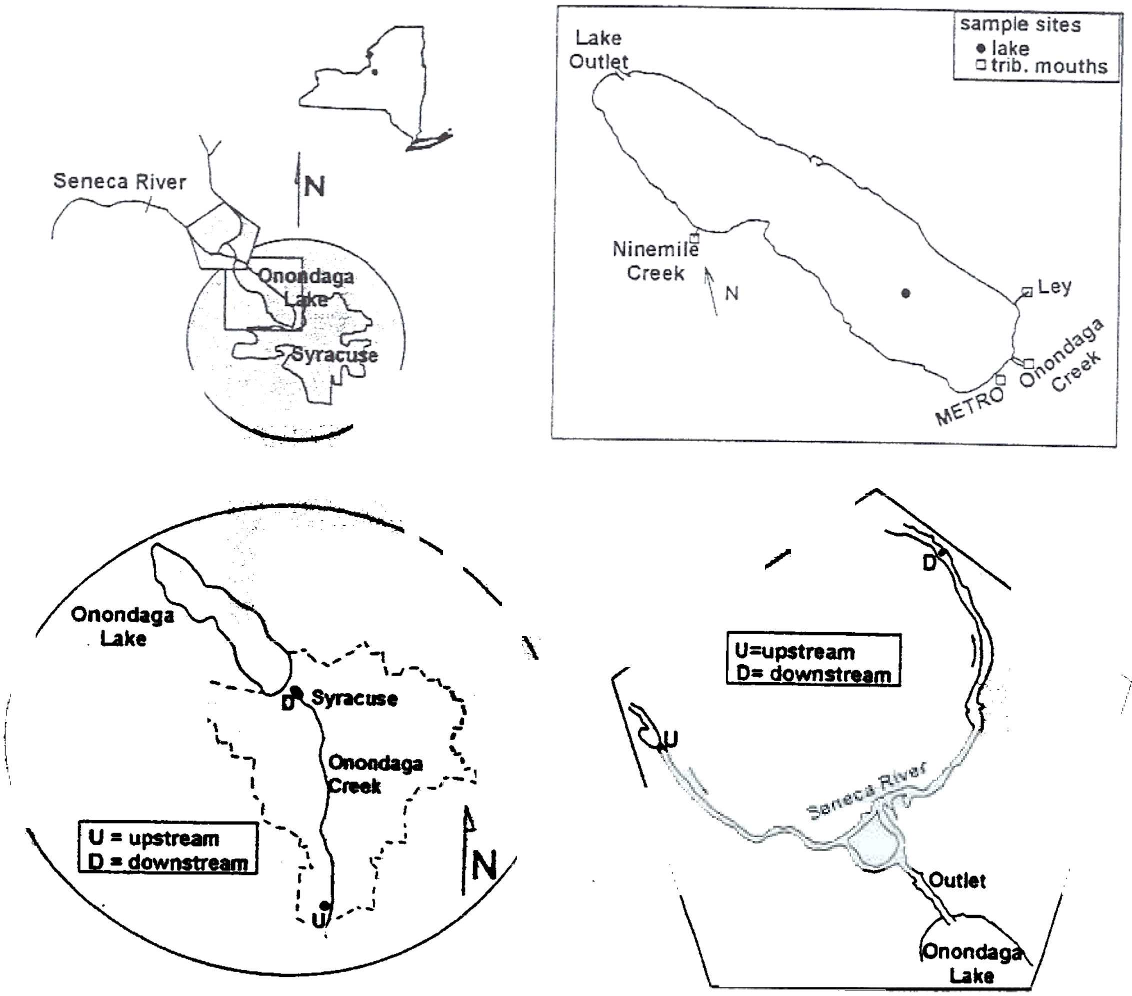

Figure 1.–Onondaga Lake, selected tributaries, Metro (WWTP), adjoining portions of the Seneca River, and long-term monitoring sites.

Onondaga Lake occurs during certain low flow intervals (Owens and Effler 1996a). Hydrologic budgets (Effler and Whitehead 1996; including the above flushing rates) and mass balance models for the lake (Effler 1996) have generally ignored this inflow.

Three tributaries, Onondaga Creek, Ninemile Creek, and Ley Creek (Fig. 1), represent more than 85% of the total tributary flow (Table 1; exclusive of WWTP input and river intrusion). Onondaga Creek and Ninemile Creek are the largest tributaries, having nearly equal watershed areas and average flow rates (Table 1). The lower reaches of Onondaga Creek drain a substantial portion of the City of Syracuse; ~20% of the creek's watershed drains this urban area. The Syracuse sewer system has 66 combined sewer overflows (CSOs; in 1998) that discharge, mostly to Onondaga Creek, during high runoff intervals when the system's capacity is exceeded. Portions of the Onondaga Creek watershed located upstream of the city (~80%) have a rural setting (Fig. 1). Much of the lower portion of the Ley Creek watershed is urban. Ninemile Creek is bordered by waste beds associated with soda ash manufacture over the lower 3 km of the stream (Fig. 1). Both Ninemile Creek and Onondaga Creek have unusually high salinities (S ‰; Effler et al. 1996c). Most of the salinity in Ninemile Creek is a result of residual ionic waste inputs from soda ash manufacturing received in the lower reaches of the stream (Effler et al. 1991, 1996c). The elevated S of Onondaga Creek largely reflects ground water input of sodium chloride from brine deposits in its watershed (Effler et al. 1996c).

The Metropolitan Syracuse Wastewater Treatment Plant (Metro) has been located on the southern shore of Onondaga Lake, and has discharged in the adjoining near-shore area of the lake (Fig. 1), since the 1920s. The existing Metro facility, operated by Onondaga County, is designed to treat an average waste flow of 80 MGD ($3.5\ m^3 \cdot s^{-1}$). Metro presently serves ~300,000 residents and a number of industries, and has an average discharge of about $3.0\ m^3 \cdot s^{-1}$ (68 MGD). The treated eff-

Table 1.–Onondaga Lake tributaries and inputs, drainage areas, TP loading, and contributions to total inflow and TP loading.

Source	Watershed Area (km²)	% Contribution[a] to Hydrologic Load	TP Load[b] (kg · d⁻¹)	% Contribution to Total TP Load
Onondaga Creek	298	31.4	34.5	14
rural	229			
urban	69[c]			
Ninemile Creek	298	30.4	28.2	
Ley Creek	77	7.7	14.0	6
minor tributaries	65	11.5	2.1	1
Metro		18.9	167.2 (160[d])	68
shoreline			150.4	
bypass			16.8 (9.6[d])	
Total	738		246.4	

[a] 1971-1989; Effler and Whitehead (1996), does not include Seneca River inflow.
[b] estimated annual averages for 1990-1995 interval; presented by NYSDEC (1998); based on Onondaga County monitoring.
[c] urban watershed area above the downstream (near mouth) USGS gauge is 53 km².
[d] based on 9 years of monitoring bypass (NYSDEC 1998).

luent enters the lake as a surface shoreline discharge. A small fraction of the discharge from this WWTP (~2%; as an annual average) irregularly enters the lake at a depth of 6 m that has by-passed full treatment, during particularly severe runoff events (Effler 1996). Metro presently makes an extraordinary contribution to the total inflow to Onondaga Lake, approaching 20% of the annual inflow on average; this discharge is usually the largest single source of water in August (Effler et al. 1996a; Table 1).

Cultural Eutrophication

Onondaga Lake was oligo-mesotrophic before European settlement in the late 1700s (Rowell 1996). Pollution that accompanied development of the watershed lead to severe degradation and loss of uses of the lake, including: (1) loss of the cold-water fishery by the late 1800s (Tango and Ringler 1996), (2) closure to ice harvesting in 1901, (3) closure to swimming in 1940, and (4) closure to fishing in 1970 (Effler and Harnett 1996). Onondaga Lake has been described as perhaps the most polluted lake in the United States (Onondaga Lake Restoration Act 1989, Hennigan 1990).

Onondaga Lake is highly eutrophic because of the high loading rate of P received in its inputs (Effler et al. 1996a). Water quality manifestations of this eutrophy include: (1) severe phytoplankton blooms, including nuisance cyanobacteria (Auer et al. 1990, Effler et al. 1996a, Matthews et al. 2001), (2) poor clarity, often below the state swimming safety standard of 1.2 m (Perkins and Effler 1996), (3) rapid loss of oxygen from the hypolimnion (Effler et al. 1996a), (4) subsequent accumulations of reduced by-products of anaerobic metabolism (Effler et al. 1988, Addess and Effler 1996), and (5) severe depletion of dissolved oxygen (DO) in the upper waters during the fall mixing interval associated with the oxidation of these constituents (Effler et al. 1996a, Gelda and Auer 1996). A large fraction of the lake's fish have been observed to migrate to the Seneca River during the interval of depressed DO (Tango and Ringler 1996).

Analyses of summertime water column conditions, including the concentrations of dissolved forms of P, cellular P pools, and phytoplankton biomass, over the 1989-1992 interval lead Connors et al. (1996) to conclude that very little nutrient limitation of phytoplankton growth occurred in Onondaga Lake (i.e., nearly nutrient-saturated). However, phosphorus is the nutrient present in least quantities relative to algal needs for growth, and is thus the appropriate target for management (Auer et al. 1990, Connors et al. 1996). The summer average epilimnetic total P (TP) concentration ranged from 56 to 76 μg · L⁻¹ for these four years (Connors et al. 1996), levels representative of high degrees of eutrophy (Vollenweider 1982, Auer et al. 1986). As recently as 1999, the summer average TP concentration in Onondaga Lake remained very high at 54 μg · L⁻¹ (Matthews et al. 2001). These levels exceed by a wide margin the New York State guidance value for the summer (mid-May to mid-September) epilimnetic average TP concentration of 20 $\mu g \cdot L^{-1}$, intended to protect against water quality degradation from cultural eutrophication.

External Total Phosphorus Loads

Total P loads from Metro for the 1977-1993 interval were reviewed by Effler et al. (1996a); Matthews et al. (2001) updated the analysis through 1999. Approximately a twenty-fold reduction in loading was achieved over the 1977-1993 interval associated with a ban on high-P detergents in Onondaga County and increased treatment (addition of secondary and tertiary) at Metro (Effler et al. 1996a). Concentrations of TP in Metro's principal discharge averaged 0.55 to 0.6 $mg \cdot L^{-1}$ in the 1990s, substantially below the effluent standard of 1 $mg \cdot L^{-1}$ (protection of the Great Lakes; in place through 1997 for Metro). The New York State Department of Environmental Conservation (NYSDEC) reported an average daily TP load from Metro to the lake over the 1990-1995 interval of 167 $kg \cdot d^{-1}$, 90% from the principal shoreline discharge and 10% via the bypass (Table 1, NYSDEC 1998). The loading estimates for the principal discharge are based on analyses (by Onondaga County) of daily flow-weighted 24 hr composite (hourly) samples. Bypass loads are based on "grab" type samples.

The NYSDEC (1998) used tributary (bi-weekly grab sampling) and Metro effluent and bypass annual loading estimates reported by Onondaga County over the 1990–1995 interval to represent prevailing loading levels and apportion the contributions according to the sources in the TMDL analysis (Table 1; load from the Seneca River inflow not considered). The estimated total daily TP loading rate to the lake for the 6 year period was 246.4 $kg \cdot d^{-1}$ (Table 1). The average contributions of Metro and the tributaries to the total TP annual loads were 68 and 32%, respectively (NYSDEC 1998; Table 1), based on estimates reported by the County. The contributions of Onondaga Creek, Ninemile Creek, and Ley Creek were estimated to be 14, 11, and 6%, respectively (Table 1). NYSDEC (1998) reported that the prevailing average daily load of TP from CSOs was 16.9 $kg \cdot d^{-1}$, or nearly 50% of the Onondaga Creek loading rate and 10% of the Metro rate (Table 1).

TMDL Analysis for Phosphorus

NYSDEC identified Onondaga Lake as a priority water body for TMDL development through its inclusion on the 1996 303(d) list of water quality limited systems. NYSDEC (1998) conducted a TMDL analysis for P for the lake that was accepted by the USEPA. A phased approach was adopted in establishing the TMDL for P in the lake, an atypical strategy that is reserved for receiving waters with serious and complex water quality problems; only two other cases were cited as examples at the time of submission of the analysis (NYSDEC 1998). The guidance value for TP developed for New York, based on empirical relationships of TP concentration with aesthetic effects for primary and secondary recreation (Kishbaugh 1993), represented the numerical goal. This numerical limit of 20 $\mu g \cdot L^{-1}$ for the summer epilimnetic average TP concentration is generally consistent with the upper bound of mesotrophy proposed for TP concentrations by several researchers (Vollenweider 1975, 1982, Chapra and Dobson 1981, Auer et al. 1986).

The model used for the analysis is a system-specific mechanistic tool, supported by monitoring and process studies (Doerr et al. 1996). The model and the TMDL analysis only consider TP. The lake is modeled as two vertical completely-mixed layers that correspond approximately to the epilimnion and hypolimnion. Accordingly, external loads enter the upper layer of the lake from its tributaries and Metro, and the lake outflow to the river leaves the same layer. The model accommodates the P cycling processes of settling, sediment release, and vertical mass transport between the two layers, and the dynamics of external TP loading (Doerr et al. 1996). The model performed well in a continuous simulation of four consecutive years (1987-1991) over which the summer average epilimnetic TP ranged from 74 to 140 $\mu g \cdot L^{-1}$ (Doerr et al. 1996). A management version of the model supports evaluations of reductions in TP loading from Metro and the tributaries for the runoff conditions of two selected years that were identified as high and low runoff years. The management model allows the user to vary the Metro effluent flows and concentrations and to reduce tributary loads in 10% increments (applied uniformly through the time-series of tributary loads). The model can be used to make iterative simulations for different external loading conditions to establish loading rates that are predicted to meet the guidance value. The irregular loads from the Metro bypass and potential inputs from the river were not considered over the interval of model testing; nor are these sources considered in the management version of the model.

Based on their review of prevailing annual external loading conditions (Table 1) and application of the model, NYSDEC (1998) determined that a Phase I TMDL of 63.5 $kg \cdot d^{-1}$ (Table 2) would meet the in-lake guidance value of 20 $\mu g \cdot L^{-1}$. A modest increase in the design average flow of Metro of ~5% (3.7 $m^3 \cdot s^{-1}$; 84.2 MGD) was invoked for the analysis – an assumption

Table 2.–Prevailing annual loads compared to TMDL allocations, as presented by NYSDEC (1998)[a].

Sources	Prevailing TP Loads (kg · d⁻¹)	TMDL TP Loads (kg · d⁻¹)	% Reductions
Non-point/tributaries	78.8	41.3 (LA)	48
Metro	160.0	15.9 (WLA)	90
MOS		6.3	–
total	238.8	63.5 (TMDL)	73

[a] based on Onondaga County monitoring data, and annual loading estimates.

that is conservative for prevailing conditions. Partitioning between WLA and LA was based on invoking an effluent concentration of 20 $\mu g \cdot L^{-1}$ for Metro, specifying a MOS of 10% of the TMDL, and calculating the necessary LA as the residual (LA = TMDL – WLA – MOS; Table 2). No abatement action for the bypass portion of the WLA was incorporated in the TMDL analysis (i.e., assumed that it will remain unchanged). According to the TMDL analysis, nearly a 75% reduction in the annual external TP loading will need to be achieved, that will be accomplished through nearly a 50% reduction in tributary loading and a 90% decrease in the Metro input (Table 2). Seasonality of loading was not considered, apparently because of an insensitivity of model predictions of the summer average epilimnetic TP concentration for the two disparate annual flow regimes used in the management applications of the model (NYSDEC 1998). Apparently, this was the basis for the timing features of the effluent limit specified for Metro, that stipulated the required loading rates or concentrations as twelve month moving (rolling) averages (NYSDEC 1998).

The implementation plan for the phase I (with three stages) P TMDL for Onondaga Lake (part of a larger $400 million plan to clean up the domestic waste problems of the lake) has three stages that will extend from 1998 to the end of 2012 (NYSDEC 1998). The phase I plan calls for a continuing in-lake discharge of the Metro effluent. Diversion of this discharge to the Seneca River had been a leading alternative (Effler 1996), until the assimilative capacity of the river for oxygen-demanding wastes decreased in response to the zebra mussel invasion (starting in 1993; Effler et al. 1996b). The stage I limits for TP for Metro (181.8 $kg \cdot d^{-1}$) correspond approximately to levels achieved through the mid-1990s (Table 1); CSO limits reflect "best management practices" implemented according to regulatory guidelines that have resulted in the annual capture of 62% of the wet weather combined sewage (NYSDEC 1998). Stage II (by 2006) requires that TP concentrations in the Metro effluent be reduced to 120 $\mu g \cdot L^{-1}$, and an increase in the annual capture of the wet weather combined sewage from 62 to 85% (NYSDEC 1998). If it is determined during stages I and II that continued discharge to the lake will not achieve water quality standards, Metro will be required to implement other alternatives, which may include relocation of the discharge to the Seneca River (NYSDEC 1998). An average reduction in tributary loading of 10.3 $kg \cdot d^{-1}$ (13% reduction from prevailing conditions, Table 1) has been assumed associated with the implementation of the CSO component of stage II (NYSDEC 1998). Stage III requires that the Metro effluent concentration be reduced to 20 $\mu g \cdot L^{-1}$ by the end of 2012 (NYSDEC 1998). Effluent levels of TP < 120 $\mu g \cdot L^{-1}$ had not been demonstrated at any full scale WWTPs of similar size to Metro at the time of the TMDL analysis (NYSDEC 1998). Specific plans to achieve the additional reductions in tributary loading (~35%; accepting the assumed reductions of 13% from increased capture of wet weather combined sewage) necessary to reach the LA, and thereby the TMDL, were not specified in the TMDL analysis (NYSDEC 1998). Listed possibilities include reductions in agricultural and stream bank erosion inputs in rural areas and elimination of possible sewer leaks in urban areas (NYSDEC 1998). A phase II TMDL analysis will be conducted by the beginning of 2009 (NYSDEC 1998).

Methods

This evaluation of the Phase I TMDL analysis for P for Onondaga Lake relies on: (1) an independent long-term monitoring program conducted on the lake, its tributaries, and adjoining portions of the Seneca River (e.g., Effler 1996, Effler et al. 1996a; Table 3), (2) algal bioassay experiments of the bioavailability of particulate P in inflows to the lake (DePinto et al. 1981, Auer et al. 1998), (3) loading rate calculations for forms of P for lake inputs, (4) calculations of water densities of inflows and the lake with an appropriate equation of state (Effler 1996), (5) analyses conducted with a previously

Table 3.–Features of long-term monitoring program used in this evaluation.

System	Frequency	Interval/ Duration	Sites[a]	Measurements Instrumentation	Laboratory[c]
Onondaga L.	weekly	April-October/ 1990-2001	L1	temperature (T), specific conduct. (SC)	Cl
lake tribs.	bi-weekly	year-round/ 1991-2000		T, flow[b]	Cl, TP, SRP, TDP
Onon. Cr.			ON_u, ON_d		
Ninemile Cr.			NM		
Ley Cr.			LC		
Metro			MT		
Seneca R	weekly	May-Sept. 1993-2000	S317	T, flow[a]	Cl, TP, SRP,

[a] sites shown on Fig. 1.
[b] Onondaga L. tributaries and Seneca R. flow measurements continuously by United States Geological Survey; Metro continuously measured by Onondaga County.
[c] abbreviations for laboratory analytes: Cl – chloride; TP, SRP and TDP – total, soluble reactive, and total dissolved P.

tested one-dimensional density stratification model (Owens and Effler 1996b) and a simpler two-layer framework (Doerr et al. 1996), and (6) mass balance calculations conducted around the lake outlet and river to estimate river inflow into the lake. The long-term lake monitoring site (Fig. 1) is generally representative of lake-wide conditions (Effler 1996). The mouths of the three primary tributaries and an upstream site on Onondaga Creek are monitored (Fig. 1). The upstream Onondaga Creek site supports bracketing the urban portion of that watershed. The Metro effluent P was monitored bi-weekly over the 1995–1997 interval to partition this input according to dissolved and particulate components. Chloride measurements made in the summers of 1990 and 1991 (Nauman 1993) and 1993 and 1994 on samples collected from the Seneca River, upstream and downstream of the lake outlet (Fig. 1), supported estimates of river inflow into the lake. Tributary flows at the three tributary mouths, the upstream Onondaga Creek site and the Seneca River (Fig. 1) were those reported by the United States Geologic Survey from continuous gauging stations; the Metro effluent Q is measured continuously by Onondaga County. All laboratory analyses (Table 3) were performed according to standard methods (APHA 1992). Spectrophotometric TP analyses were corrected for the effects of turbidity (method 4500-PE; APHA 1992).

The bioavailability of particulate P from Onondaga Creek at its mouth and at the upstream urban/rural boundary, at the mouth of Ninemile Creek and the Metro effluent was determined for a single sample from each of these sites collected during a dry weather interval of 1996 (October 7–22), using a modification (Auer et al. 1998) of the dual culture diffusion apparatus of DePinto (1982). Accordingly, P released from the particulates diffuses across a membrane and is immobilized by P-starved algae (*Selenastrum capricornutum*). The algal cells were harvested at 3 d intervals over a 30 d period and the P content determined. Three features of bioavailability were quantified by the experiments (Auer et al. 1998): (1) the ultimate concentration of P available, normalized to the mass of suspended solids (P_{ult}; $\mu gP \cdot gSS^{-1}$), (2) the fraction of the P associated with the suspended solids that is available (f), and (3) the reaction rate (k; d^{-1}) for conversion of PP to algal P.

Loading rates of forms of P from tributaries were calculated from measurements on bi-weekly grab samples and the continuous Q measurements (Table 3) through time interpolation, as strong concentration - Q relationships were not observed (Effler and Whitehead 1996). Estimates of water density were made with an equation of state (Effler 1996) that incorporates the density-temperature (T, °C) relationship for pure water of Millero et al. (1976) and the S dependence reported by Chen and Millero (1978). This expression performs as well as the system-specific expression developed by Effler et al. (1986), for the density difference issues addressed here. Values of S for Ninemile Creek, Onondaga Creek and Onondaga Lake were estimated from chloride (Cl) concentration according to relationships presented by Effler (1996).

Net inflow into the lake from the Seneca River ($Q_{l/s}$) was calculated according to the following steady-state mass balance expression (Owens 1993) for Cl, widely used as a conservative tracer

$$Q_{SEN}(Cl_{SEN/u} - Cl_{SEN/d}) + Q_{TRIB}(Cl_L - Cl_{SEN/d}) + Q_{I/S}(Cl_L - Cl_{SEN/u}) = 0 \quad (1)$$

where Q_{SEN} and Q_{TRIB} are daily flows of the Seneca River at Baldwinsville (Fig. 1) and the summed inflows to the lake (tributaries plus Metro), respectively, and $Cl_{SEN/u}$, $Cl_{SEN/d}$, and Cl_L are the Cl concentrations in the river upstream of the lake, downstream of the lake, and in the epilimnion of the lake.

A previously tested (Owens and Effler 1989, 1996b) system-specific one-dimensional hydrothermal density stratification model for the lake was applied here to simulate the occurrence and extent of the underflow phenomenon (plunging of a dense inflow below the upper mixed lake layer) for the conditions of 1999. This is an integral, or mixed layer, model that accommodates the effects of meteorological and hydrologic conditions on vertical transport of heat and mass in simulating vertical density stratification. This model partitions the water column of the lake vertically into segments of about 1 m thickness (Owens and Effler 1989). Turbulent kinetic energy supplied by surface wind stress and convective cooling is used to overcome the gradient at the base of the expanding surface mixed layer (e.g., Ford and Stefan 1980, Harleman 1982, Owens 1998). Turbulent diffusion below the epilimnion is also accommodated (Owens 1998). The heat budget of the model includes terms for evaporative heat loss, short- and long-wave radiation, convection, conduction and back radiation. The model also includes a submodel that simulates the effects of dense plunging inflows (e.g., Hebbert et al. 1979, Owens and Effler 1996b). An earlier version of the model successfully simulated the substantial seasonality of the effective depth of plunging of inflows made dense by saline waste inputs (Owens and Effler 1989). Model inputs include the T, S, and Q of inflows and meteorological data collected at a nearby (8.5 km) National Weather Service station. Model calibration procedures were described by Owens and Effler (1996b).

The simpler two-layer transport framework of Doerr et al. (1996) and the above multi-layer stratification model were used to explore the implications of the magnitude and seasonality of the lake flushing rate on summertime concentrations of a tracer, discharged from Metro. The tracer was loaded at a uniform concentration each month separately and the average summertime (mid-May to mid-September) epilimnetic concentration predicted to depict the relative responsiveness of the lake's productive layers to the seasonality of this load; e.g., within the context of the timing and depth features of the TP guidance value. Application of the density stratification model was limited to the conditions documented for 1999. The two-layer framework was applied for 30 years of continuous hydrologic loading documented for the system for the 1971-2000 period to represent the effects of natural variations in runoff (e.g., Gelda et al. 2001).

Results and Discussion

Hydrology

An accurate hydrologic budget is necessary to support material budget calculations and related mass balance water quality models (Chapra 1997). Thus it is important to include all significant hydrologic inputs. Within the context of a P TMDL analysis, this is particularly critical for sources that make disproportionately large contributions to the overall load of available P. The TMDL analysis did not consider two minor tributaries on the lake's east shore that probably represent <5% of the total tributary flow. This omission is not considered important to the analysis, as associated TP loads from these inputs are not disproportionately high (Effler and Whitehead 1996).

A much more important shortcoming of the TMDL analysis is the omission of the input from the Seneca River. There are three aspects of concern in this regard: (1) the magnitude of this inflow has not been quantified, (2) the river is relatively rich in P during the summer months (e.g., TP > 60 µg·L^{-1}; Effler et al. 1996b) compared to levels anticipated in the lake following implementation of the management plan, and (3) a shift to increased availability of this source of P (more in dissolved forms) has occurred since the zebra mussel invasion of the river (Effler et al. 1996b). Two anthropogenic effects are responsible for the unusual flow regime in the lake's outlet that results in irregular inputs from the river: (1) the elevated density of lake water relative to the river, that is at least in part due to residual industrial saline waste inputs (Effler et al. 1997), and (2) the elimination of the natural elevation gradient from the lake to the river through lowering of the lake and control of the river elevation for hydropower and navigation purposes (Owens and Effler 1996a). The flow regime in the lake's outlet has been found to be extremely dynamic and complex (Seger 1980, Owens and Effler 1996a), such that the application of traditional techniques (e.g., flow gauging) to quantify net river inflow (incorporated into the lake's water column) and lake outflow has been confounded. A bidirectional flow regime is commonly encountered in the lake outlet, with the less dense river water flowing toward the lake in the upper layer (Owens and Effler 1996a). Analyses to date indicate the river inflow phenomenon occurs during low flow (e.g., summertime) intervals, and that mitigating factors include operational influences on the river elevation and wind conditions for the lake (Owens and Effler 1996a).

The dynamics of net river inflow ($Q_{i/s}$) for 1990 and 1991, estimated through application of the Cl mass balance around the lake and river [Eq. (1)], depict strong short-term and year-to-year differences in the magnitude of this source (Fig. 2a and b). Extension of the approach to 1993 and 1994, for which there are less Cl data for the river to support the calculations, resulted in average $Q_{i/s}$ values of 0.8 and 1.5 $m^3 \cdot s^{-1}$ for the May–September interval. There are substantial sources of uncertainty that may compromise the reliability of the estimates, and particularly the short-term patterns, of $Q_{i/s}$ including: (1) the known high temporal variability of the phenomenon, (2) temporal limitations in Cl measurements, (3) effects of travel time and mixing processes in the river, (4) errors in flow measurements, and (5) limitations in the assumptions invoked in the development of the mass balance (Owens 1993). The average of the estimates of $Q_{i/s}$ for the May–September interval for the four years, ~1.5 $m^3 \cdot s^{-1}$, is a reasonable first approximation of the magnitude of this inflow. Based on the long-term hydrologic record for the lake's tributaries (e.g., Gelda et al. 2001), this represents on average about 50% of the Metro discharge and ~13% of the total flow into the lake for the May–September interval. The implications of this additional source of P within the context of the other inputs and the TMDL analysis are considered subsequently.

Implications of Flushing

The seasonal character of New York's TP guidance value (mid-May to mid-September) has extremely important implications with respect to the potential seasonality, or short-term variations, in the performance of P treatment at Metro, and the extent to which averaging of effluent concentrations over 12 months is protective for Onondaga Lake. This is demonstrated here through a lake "response curve" (Fig. 3a) that presents predicted epilimnetic concentrations of a conservative tracer for the mid-May to mid-September interval that results from uniform loading from Metro through each month separately (i.e., 12 simulations of response to single month loading to form the response curve). Means of the 30 years of simulations with the simple two-layer framework, and measures of inter-annual variations from natural hydrologic variability (± 1 standard deviation) incorporated in the 30 year record of inflows, are presented (Fig. 3a). Predictions with the more replete one-dimensional hydrothermal model were well within these variability limits, and accommodation of reasonable levels of river inflow have only a modest effect on the character of the predicted response (Fig. 3a). Clearly, the impacts of loads received from early fall through the following early spring interval are modest (because of rapid flushing/turnover) compared to the inputs received over the April-August interval (Fig. 3a).

Conditions over the designated critical interval of the TP guidance value are largely driven by loading conditions over much of the same interval (Fig. 3). This

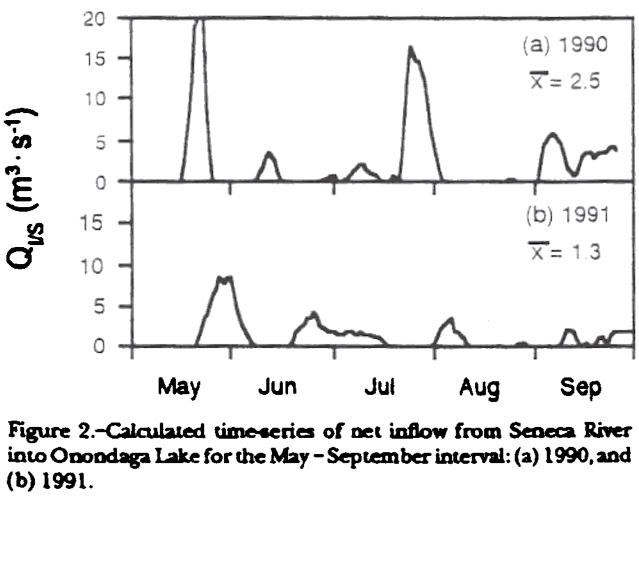

Figure 2.–Calculated time-series of net inflow from Seneca River into Onondaga Lake for the May – September interval: (a) 1990, and (b) 1991.

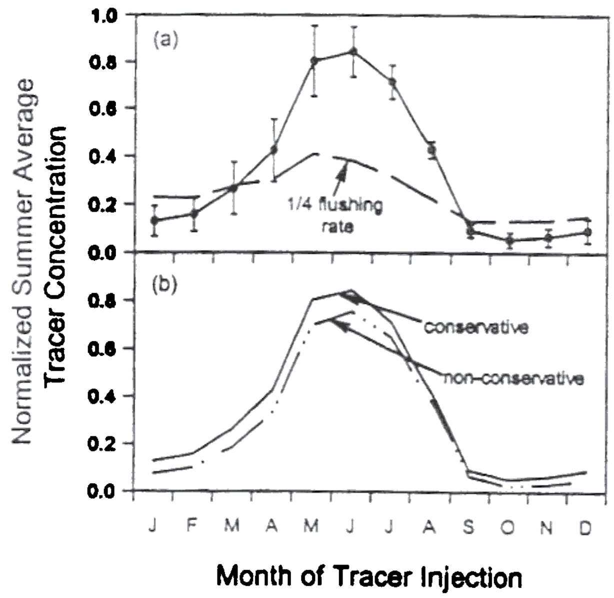

Figure 3.–Predicted "response curves," epilimnetic average tracer concentrations normalized by maximum value for the mid-May to mid-September interval, from monthly tracer injections at Metro: (a) conservative tracer, points are means for 30 year tributary hydrologic record, vertical bars are ± 1 standard deviation, dashed line is predicted average response for the case of lake flushing rate at 25% of the prevailing, and (b) comparison of response curves for non-conservative versus conservative tracer.

is a manifestation of the high flushing rate of Onondaga Lake. The response curve demonstrates substantially less seasonality for a lower average flushing rate of 25% of the prevailing rate (inflows reduced by 25% for this scenario; Fig. 3a). Similar seasonality emerges for a reactive substance (e.g., P). This is demonstrated here by repeating the response curve analysis for a tracer with average settling velocity of 0.03 $m \cdot d^{-1}$ (Fig. 3b), corresponding to the net loss behavior of P incorporated in a widely used simple P model (Vollenweider 1975).

The predicted character of the summertime response of the lake to the timing of loads has at least four important implications for managers. First, this response diminishes the potential importance of internal loading of P from the enriched hypolimnion of the lake mediated by vertical mixing. Large quantities of P accumulate in Onondaga Lake's anoxic hypolimnion during summer stratification associated primarily with sediment release (Auer et al. 1993, Doerr et al. 1996, Penn et al. 2000), as observed in many eutrophic lakes (Wetzel 2001). A number of well-known case studies (Larsen et al. 1981, Welch et al. 1986, Marsden 1989) have demonstrated that sediment feedback can retard a lake's response to P-based rehabilitation efforts. Auer et al. (1993) demonstrated the highest internal load (by a wide margin) to the productive layers of Onondaga Lake from this feedback occurs during late September and early October, associated with the entrainment of the enriched hypolimnion that accompanies the approach to complete fall turnover (Effler and Owens 1996). Loading in that interval does not contribute substantively to the epilimnetic P pool of the lake during the critical summer months, as these inputs are removed from the water column before the subsequent summer through export or redeposition (Fig. 3). This is promising for timely lake recovery, as assessed in the summer months, following adequate reductions in external loading. Internal loading from the hypolimnion during the summer months is limited to much smaller fluxes (average of about ~3 $kg \cdot d^{-1}$ in late 1980s; Auer et al. 1993), mediated by small scale vertical mixing (Wodka et al. 1983).

Second, protective permit limits for Metro effluent P concentrations need to reflect the seasonality of the lake's response (Fig. 3b) driven by its high flushing rates. Timing features of New York's TP guidance value dictate that permit limits for Metro to protect summertime conditions in the lake should specify the highest performance over the April–August interval. The year long (12 month) averaging presently incorporated in the facility's permit is inconsistent with the basic timing features of the response of this lake (Fig. 3). It is not protective of the lake, as it would allow occurrences of relatively poor summertime P treatment to be compensated for by better levels of treatment in non-critical months.

Third, the strong seasonality in lake response also challenges the representation of external P loading rates, based on year-round monitoring, such as included in the P TMDL analysis for Onondaga Lake (Tables 1 and 2). These rates should instead reflect levels that prevail over the April–August interval. While Metro loading rates have been relatively uniform seasonally, tributary loading rates of P have been significantly (α = 0.05) lower for the April – August interval than on an annual basis at the mouths of both Onondaga Creek (TP, TDP) and Ninemile Creek (TDP; Fig. 4a and b).

Fourth, the wide interannual variations predicted for both the conservative and non-conservative tracer analysis (Fig. 3a and b) suggest the potential for substantial interannual differences in lake TP in the future (following the major reductions in Metro inputs), driven by natural variations in runoff. These effects probably cannot be fairly represented through *a priori* selection of two case years perceived as bounding the range of runoff effects, as presently adopted in the P TMDL analysis (NYSDEC 1998). Long-term monitoring of P in the tributaries and the extensive flow record for these inputs (Effler 1996) offer the opportunity to more fully represent these effects and objectively identify critical conditions (Gelda et al. 2001).

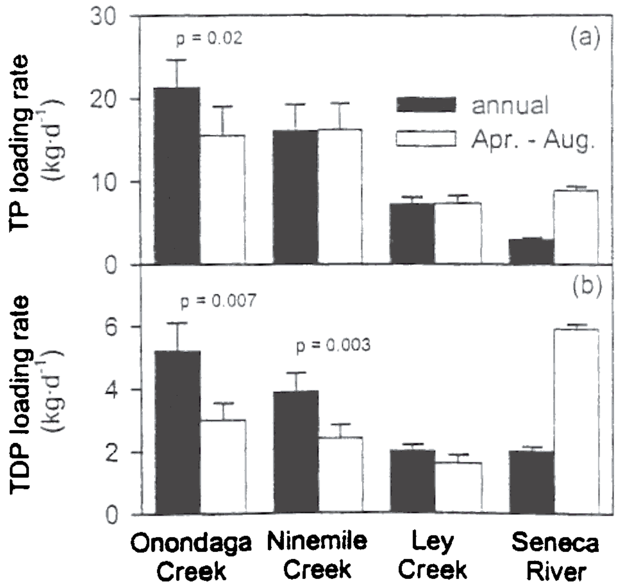

Figure 4.–Comparison of annual average versus April–August average P loading rates for the 1995–2000 interval for inputs to Onondaga Lake: (a) total P (TP), and (b) total dissolved P (TDP). Error bars represent one standard error of the means for six years; p-values are included to indicate significant differences for the two averaging periods.

Salinity, Density and Inflow Patterns

Density differences between inflows and a receiving lake can influence the effective depth of entry of the inputs (Effler and Owens 1986, Alavian et al. 1992). Inflows that are less dense than the surface waters of a lake tend to enter onto the surface of the lake (overflow) and are readily incorporated into the upper mixed layer. Inflows that are more dense than the surface waters tend to plunge in the lake, and can enter as an "underflow." Ambient lake waters are entrained into an underflow as it plunges, thereby reducing density differences. The underflow enters the water column at a depth where its density equals that of the water column (neutral buoyancy depth; becoming an "interflow"). Local mixing conditions in the area of the point of entry of an inflow into a lake basin influences this phenomenon; e.g., often the turbulence is adequate to eliminate plunging where modest differences in density develop seasonally. Both T (ΔT) and S (ΔS) differences play important roles in regulating the seasonal dynamics of density differences ($\Delta \rho$) between the surface waters of Onondaga Lake and its inflows and the occurrence of plunging underflows (Fig. 5).

Substantial differences between the temperatures of the two largest tributaries and the lake's surface waters develop annually; the seasonal trends are represented here by polynomial fits of observations from the long-term monitoring program (Fig. 5a). The average temporal distributions are very similar for Ninemile Creek and Onondaga Creek. These tributaries remain colder than the lake surface for the May-October interval (i.e., $\Delta T < 0$), with a maximum difference of nearly 7 °C common in early August. This timing is widely observed, associated with the disparate responses of lotic and lentic systems to the seasonality of heat flux components in this climate. The temporal differences between the Metro effluent and lake surface temperatures demonstrate a widely different character. The temperature of this effluent remains warmer than the two main tributaries, and often by a wide margin. The effluent temperature becomes colder than the surface lake waters (i.e., $\Delta T < 0$) for a shorter interval (June-August), and the seasonal maximum temperature difference during this interval is much less (~3 °C). The Metro effluent remains much warmer than the lake surface in early spring and late fall.

Salinity levels in the two major tributaries are generally substantially greater than in the upper waters of the lake over the summer months (i.e., $\Delta S > 0$), while Metro concentrations are usually lower ($\Delta S < 0$), as represented by the polynomial fits of long-term observations (Fig. 5b). These relationships are subject to more short-term variability than those for temperature, associated with runoff events (e.g., dilution) in

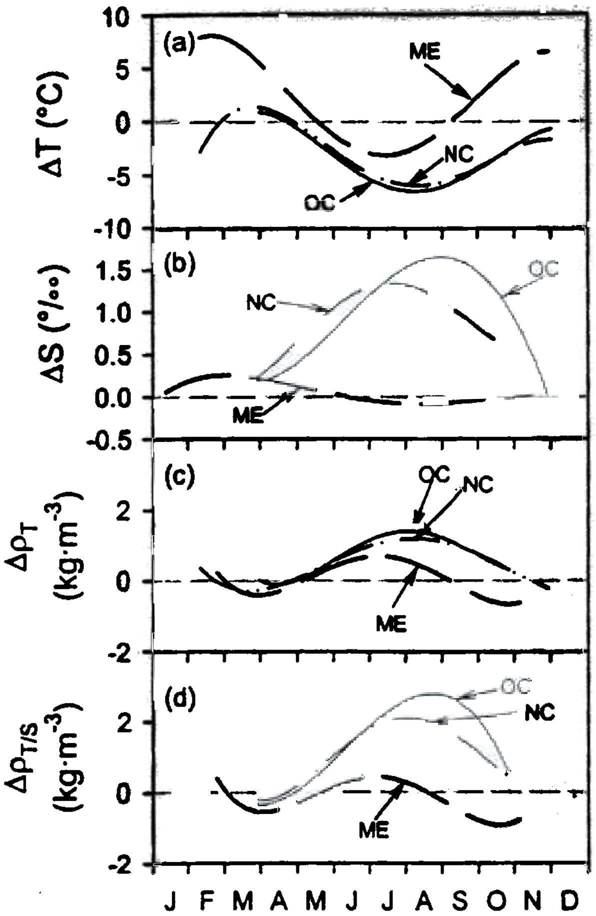

Figure 5.–Seasonality of differences in density and its components between inflows and the lake, represented as polynomial fits of long-term monitoring data: (a) temperature differences, ΔT, (b) salinity differences, ΔS, (c) density differences associated with ΔT, $\Delta \rho_T$, and (d) density differences associated with ΔT and ΔS, $\Delta \rho_{T/S}$. Designations for the inflows are ME for Metro, OC for Onondaga Creek, and NC for Ninemile Creek.

the case of tributaries, and with irregular inputs from industrial waste in the case of Metro (Effler et al. 1996c). Somewhat higher S levels presently prevail in Onondaga Creek compared to Ninemile Creek (Fig. 5b), a noteworthy reversal associated with decreases in residual industrial loading to Ninemile Creek (Matthews and Effler 2001). The timing of the maximum S difference between the tributaries and the lake coincide approximately with that of the T differences (Fig. 5a and b).

The colder temperatures tend to make these two tributaries, and to a lesser extent the Metro effluent, denser than the surface waters of the lake during summer and early fall (i.e., $\Delta \rho_T$, includes temperature effects only; Fig. 5c). Inclusion of the S effect is critical in this analysis as it modifies the density differences substantially (i.e., $\Delta \rho_{T/S}$; Fig. 5d). The higher S of the tributaries increases the density differences with the lake surface associated with lower T by about a factor of two (Fig. 5c and d). The higher tributary S values combined with the lower Metro S causes these inputs to diverge strongly in the potential to plunge. Metro effluent is denser than the lake surface waters by a relatively small amount for only a brief interval (Fig. 5d).

The occurrence of plunging of one or both of the saline dense tributaries during summer is established by the coincident annual development of metalimnetic peaks in S over the summer to early fall interval (unpublished data, Upstate Freshwater Institute), illustrated here through specific conductance profiles in early and late summer in several recent years (Fig. 6a–d). Very little vertical structure is observed in early summer but conspicuous sub-surface peaks are manifested by late summer. Interannual differences in the vertical position and magnitude of the peaks (Fig. 6a–d) may reflect the effects of variations in runoff (e.g., dilution of tributary S) and meteorological conditions (e.g., ambient lake mixing, and ΔT).

Phosphorus loads carried in the plunging interflow(s) enter below the upper mixed productive epilimnion and are not immediately available to support phytoplankton growth (below the photic zone; Perkins and Effler 1996). Though some portion of this interflow subsequently makes its way to the upper layers through mixing processes, the associated P load is likely further diminished in the interim. The one-dimensional density stratification model is used here to provide a first approximation of the seasonality of the plunging underflow phenomenon for the two major tributaries for the conditions of 1999. This preliminary analysis assumes uniform near-shore geometry and near-field mixing for the two saline tributaries and Metro as they enter the lake basin. Following initial calibration to the observed features of the thermal stratification regime, calibration focused on the amount of entrainment (a "one-way" transport process into the plunging inflow; Fischer et al. 1979) of ambient lake water necessary to approximately match the sub-surface maxima in S (e.g., Owens and Effler 1989, 1996b). Thus differences in the predicted dynamics of plunging for the various inflows depended solely on the dynamics of density differences between the inflows and the lake (e.g., 5d). The calibrated model performed well in simulating the sub-surface maxima in S (e.g., 7a–c). The potential for calibration through different combinations of conditions (e.g., near-field mixing and entrainment) that could affect the relative importance of these tributaries in regulating the in-lake signature, and the relative extent of plunging of these two inflows, is acknowledged.

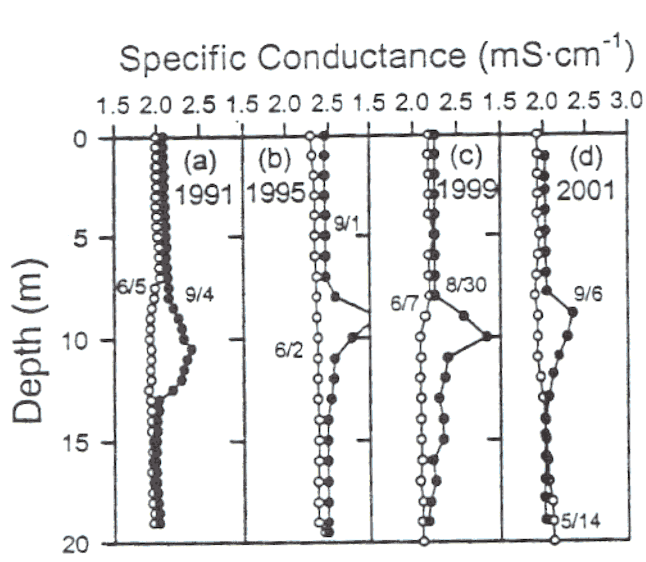

Figure 6.–Recurrence of the development of a sub-surface maximum of salinity, depicting the effects of a plunging underflow, from comparisons of early and late summer specific conductance profiles collected in Onondaga Lake: (a) 1991, (b) 1995, (c) 1999, and (d) 2001.

The calibrated model was applied to estimate the percent of the Metro, Onondaga Creek, and Ninemile Creek inflows that entered the upper mixed productive layers of the lake over the April–October interval 1999. The Metro effluent either entirely or mostly entered the upper mixed layers throughout the interval (Fig. 7d). In strong contrast, a substantial portion of the Onondaga Creek inflow entered below these layers over most of the May–September interval (Fig. 7e), while the extent of plunging of Ninemile Creek was less over that period (Fig. 7f). Plunging of both tributaries was predicted to be conspicuously diminished during the prolonged interval of elevated tributary flow starting in mid-June (Fig. 7g; Onondaga Creek Q generally a good indicator of overall tributary Q; Effler and Whitehead 1996), that was at least in part driven by dilution-based reductions in the S of these inflows (Effler et al. 1996c). These interactions suggest the potential for strong interannual variations in this phenomenon, associated with natural variations in meteorological conditions. Approximately 27% of the TP load and 31% of the TDP load for Onondaga Creek entered below the upper productive layers over the April–August interval of 1999; the percentages for Ninemile Creek were 28 and 25%.

Bioavailability and Deposition

As the TP guidance value is intended to protect against water quality degradation from cultural eutrophication, only those components of the P load that can support phytoplankton growth should be

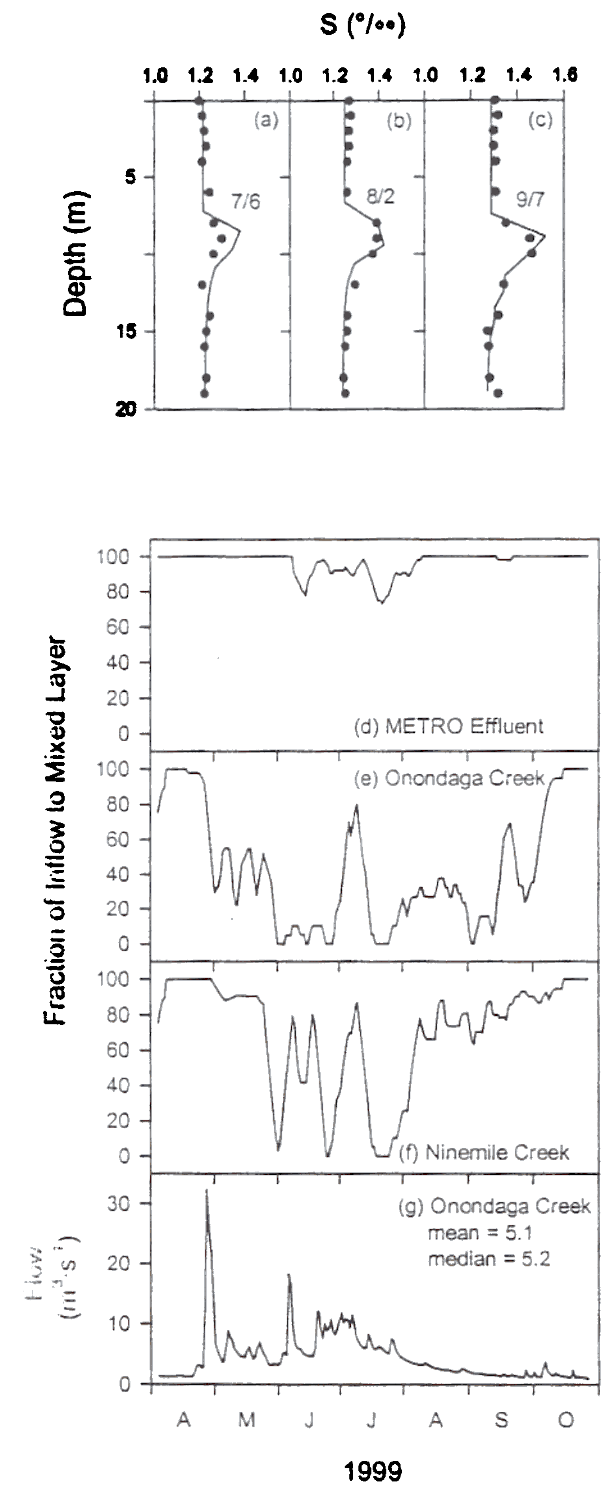

Figure 7.–Calibration of the plunging interflow feature of the density stratification model and application to assess the extent and seasonality of plunging of inflows below the upper productive layers, for the conditions of 1999: (a) performance for July 6, (b) performance for August 2, (c) performance for September 7, (d) predicted fraction of Metro inflow into upper mixed layer (UML), (c) predicted fraction of Onondaga Creek inflow into UML, (f) predicted fraction of Ninemile Creek inflow into UML, and (g) time-series of flow at Onondaga Creek mouth.

considered in the TMDL analysis. Dissolved forms of P are used by phytoplankton to support growth (Hutchinson 1973). Forms of dissolved P included in the SRP fraction are widely considered to be immediately available to phytoplankton, while dissolved organic P (DOP ~TDP-SRP) can be made available through enzymatic hydrolysis (Gage and Gorham 1985, Currie et al. 1986). Utilizing the same experimental bioassay protocols adopted here for PP, researchers have reported that most (Young et al. 1982), to essentially all (Auer et al. 1998), of the TDP pool in lake inputs is available to support phytoplankton growth. The bioavailability of PP is substantially lower (Young et al. 1982, Auer et al. 1998), and a portion of this material may be lost from productive layers through deposition. Thus the concentration ratio TDP:TP is a valuable indicator of relative availability of P in external loads.

The average TDP:TP ratios for the April-August interval for the 1995–2000 interval were very similar for the upstream site on Onondaga Creek (0.21 ± 0.06; 95% confidence interval) and for the mouth of Ninemile Creek (0.20 ± 0.03). The ratio shifted higher for Ley Creek (0.24 ± 0.02) and over the urban reach of Onondaga Creek, to an average value of 0.29 ± 0.03 at its mouth, reflecting enrichment in TDP. This increase is also observed during dry weather conditions, consistent with other circumstantial evidence (e.g., bacterial indicators; Effler and Whitehead 1996) that suggests contributions from leaky sewers within the urban portion of the watershed. The Metro P load has been substantially more potent for the support of phytoplankton growth, as the average TDP:TP ratio for the April-August interval was 0.51 ± 0.06 (1995–1997). Since the zebra mussel invasion (e.g., 1993; Effler and Siegfried 1994), the irregular inputs from the Seneca River have been the most potent, with an average ratio value of 0.68 ± 0.07. No partitioning of P concentration between dissolved and particulate forms has been specified for Metro stage II and stage III upgrades.

Wide differences in the bioavailability of PP were found for the four tested sources, as characterized by the single set of experiments (Table 4). The solids in the Metro effluent were substantially enriched in available P (P_{ult}) compared to the tributaries; the Metro P_{ult} (10,150 μg P · $gTSS^{-1}$) was approximately 30-fold greater than found for the tributaries draining rural areas, and more than 6-fold greater than the mouth of Onondaga Creek (Table 4). Further, the reaction rate (k) for the Metro effluent was substantially greater than for the tributaries, and the fraction (f) of PP that was available was distinctly higher for Metro and at the mouth of Onondaga Creek than the other tributary sites (Table 4). These results, though temporally limited, indicate the Metro load of PP is substantially more potent in supplying P to support phytoplankton growth

Table 4.–Bioavailability of particulate phosphorus (PP) in inputs to Onondaga Lake as determined by bioassay experiments, with 95% confidence intervals for the determinations.

Source	Bioavailable P Concentration, (P_{ult}, μgp . g TSS^{-1})	Bioavailable P Fraction (f)	Rate Constant (k, d^{-1})
Onondaga Creek, rural	390 ± 130	0.32 ± 0.10	0.08 ± 0.06
Onondaga Creek mouth	1650 ± 200	0.52 ± 0.06	0.13 ± 0.05
Ninemile Creek	300 ± 50	0.22 ± 0.03	0.14 ± 0.06
Metro effluent	10,150 ± 1570	0.58 ± 0.09	0.31 ± 0.13

in the lake than the tributaries. This is consistent with the literature, as other investigators have also reported greater P bioavailability in solids loads from WWTPs compared to tributaries, using similar experimental procedures (DePinto et al. 1981, Young et al. 1982, 1985). The increase in bioavailability of P in suspended sediments at the downstream site on Onondaga Creek (Table 4) is also consistent with other circumstantial evidence (Effler and Whitehead 1996) that suggests contributions from leaky sewers.

The process of deposition further reduces the availability of PP received in external loads. Certain particles carrying available P may settle out of the productive layers before the P can be made available for uptake. All but the smallest inorganic particles are lost rapidly from the watercolumns of lakes and reservoirs because of their relatively high densities. Settling velocities of inorganic particles have been widely reported to be much higher than for organic particles (e.g., 10 to 20-fold) in the open waters of lakes and reservoirs in sediment trap studies (e.g., Effler et al. 2001). The larger of the particles received from tributaries tend to settle out in the near-shore zone proximate to the inflows. Typical settling velocities for organic and inorganic particles in pelagic waters are 0.2 and 3 m $\cdot$ d^{-1}, respectively (Effler et al. 2001), corresponding to water column loss rates of approximately 0.03 and 0.5 d^{-1}, respectively, for a common epilimnion depth of 6 m. The transport of PP to the underlying sediments through deposition is largely unidirectional in the pelagic zone of Onondaga Lake, as very little sediment resuspension occurs in the deep portions of the lake (Effler and Brooks 1998). The rapid deposition of inorganic particles received from Onondaga Creek and Ninemile Creek in the lake is manifested by localized near-shore lake deposits adjoining the mouths of these tributaries (Auer et al. 1996, Effler 1996).

Selective extraction analyses performed on the tributary and Metro particulate samples before and after the bioassays (Needham 2000) demonstrated most of the PP in the Metro effluent, and mobilized in the experiment, was associated with organic particles. In contrast, most of the PP for the upstream Onondaga Creek and Ninemile Creek samples, and the limited mobilization, was associated with inorganic particles (Needham 2000). Intermediate conditions were observed with respect to association and mobilization from the PP at the mouth of Onondaga Creek (Needham 2000). These associations are consistent with the dominant components of particle populations of these inputs established through individual particle analysis techniques (Yin and Johnson 1984, Effler et al. 1992).

Settling velocities reported in the literature (e.g., Effler et al. 2001) for organic versus inorganic particles, together with the limited site-specific characterizations of particulate associations of P mobilized in the bioavailability experiments (Needham 2000), suggests much more of the bioavailability potential of PP from Metro is exerted in the lake compared to the tributaries. A first approximation of the diminishment of the potential of the bioavailable PP from settling is represented as the ratio of the settling loss rate (adopting 0.2 m $\cdot$ d^{-1} for organic and 3 m $\cdot$ d^{-1} for inorganic) to the sum of this rate and the reaction rate observed in the bioavailability experiment (k, Table 4). Accordingly, approximately 10% of the bioavailable PP from Metro (assumed organic particles) would be expected to settle before it could be mobilized, while about 75% of the bioavailable PP from Ninemile Creek (assumed inorganic particles) is probably lost from the productive layers through deposition before it could release the available P. An intermediate level (~40%) of diminishment was estimated for Onondaga Creek, assuming that the increase in P_{ult} from the upstream site to the mouth was mostly (80%) associated with an influx of organic particles.

Analytical Issues and Tributary Unit Area Loads

Samples that contain substantial turbidity will have false high TP concentrations if appropriate corrections for the effects of this particulate material on spectrophotometric measurements (APHA 1992) are not made. Failure to correct for turbidity effects on the TP analysis (e.g., Tables 1 and 2; implicit in the use of contemporary auto-analyzers) results in substantial overestimation of loads from these tributaries (Fig. 8). The impact is the greatest for Onondaga Creek and is more important for annual than April–August loads (Fig. 8a and b). These relationships are consistent with the interference of suspensoids, as Onondaga Creek has the highest solids concentrations and these levels tend to be higher for all the tributaries outside of the April–August interval (Yin and Johnson 1984, Effler 1996). The differences in loads for the uncorrected (for turbidity) and corrected TP measurements were significant ($\alpha = 0.05$, paired two sample t-test for means on log transformed observations) in all cases. Nearly 90% of the uncorrected TP measurements for Onondaga Creek were false high by more than 20% and nearly 35% were false high by more than 60% (Fig. 8c). Annual loads at both Onondaga Creek sites were overestimated by nearly a factor of two in 2000, in the absence of corrections for turbidity (Fig. 8a). Uncorrected estimates for Ninemile Creek and Ley Creek were false high by ~15%. Overestimates of tributary loading for the April–August interval of 2000 associated with failure to correct for turbidity were about 20% for the two Onondaga Creek sites, and about 10% for the other two tributaries (Fig. 8b). The effect of this analytical interference for the Metro effluent is less certain; irregular checking suggests the TP concentrations in this discharge have been largely unaffected compared to the tributaries. Analytical results for dissolved forms of P (e.g., TDP, SRP) are determined from filtered (0.45 µm) samples (for all sites) and thus are unaffected by turbidity interference. Particulate P (PP) concentrations determined by difference (PP = TP − TDP) will be false high by the same magnitude as TP concentrations, though the relative error will be higher for PP.

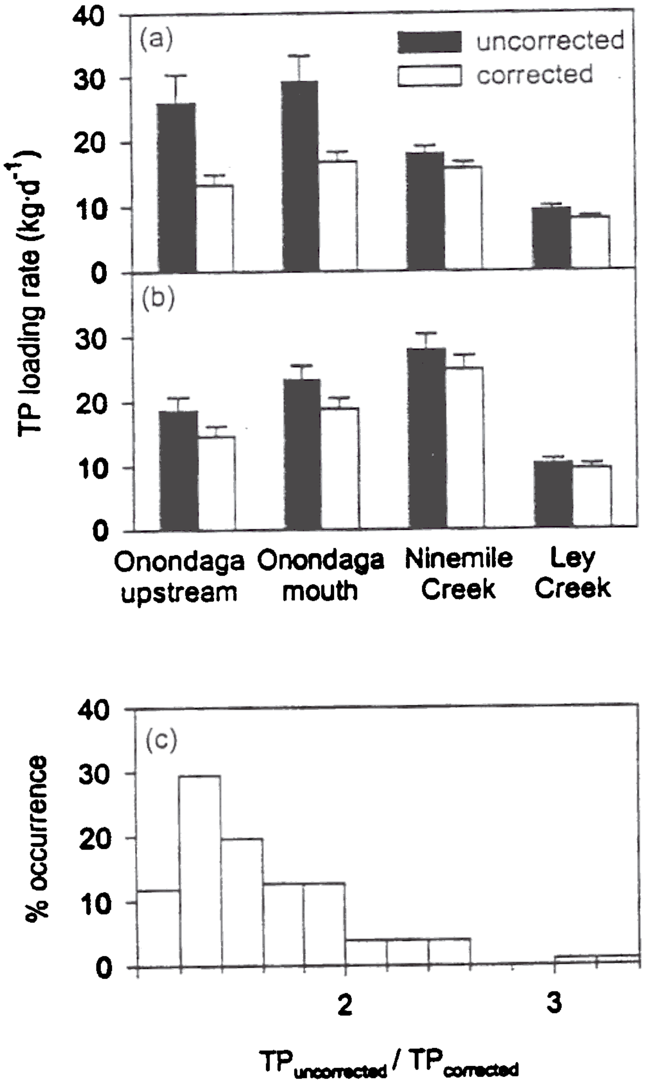

Figure 8.–Effects of turbidity interference on estimates of average TP loads for Onondaga Lake tributaries for conditions of 2000: (a) annual, (b) April–August, and (c) distribution of the ratios of uncorrected TP concentrations to corrected TP concentrations for Onondaga Creek. Error bars represent one standard error of the means calculated from daily loading estimates.

Examination of tributary loadings normalized by contributing area (unit area loads, UALs) is valuable in delineating the relative richness of these sources within the lake's watershed and it facilitates comparisons with conditions in other basins reported in the literature (Table 5). UALs are presented for TP, TDP, and SRP for the rural and urban portions of Onondaga Creek and for the overall watersheds of Ninemile Creek and Ley Creek, as averages for the 1995–2000 period. Both annual (based on year-round monitoring) and April through August values are presented (Table 5). The annual estimates support comparison to literature values, while the seasonal values are consistent with the system-specific response time described here (Fig. 3). "Most frequently reported ranges" of UALs presented in the review of Budd and Meals (1994) for forested, agriculture, and urban land uses are included for comparison (Table 5). No systematic decreases have been noted in the UALs for the tributaries since the completion of the TMDL analysis.

The UALs for all three forms of P are shifted substantially lower for the April–August interval compared to the annual levels for both the rural and urban portions of the Onondaga Creek watershed (Table 5).

Table 5.–Unit area loads (UALs) for Onondaga Lake tributaries[a], compared to literature values for different land uses. Standard deviations of the averages for six years are included in parentheses.

Tributary/Landuse	TP UALs ($kg \cdot km^{-2} \cdot y^{-1}$)		TDP UALs ($kg \cdot km^{-2} \cdot y^{-1}$)		SRP UALs ($kg \cdot km^{-2} \cdot y^{-1}$)	
	Annual	April-Aug	Annual	April-Aug.	annual	April-Aug.
Onondaga Creek						
rural	20 (11)	15 (10)	3.7 (1.4)	2.1 (0.7)	1.4 (0.9)	0.8 (0.6)
urban	56 (29)	43 (28)	24 (20)	13 (7)	17 (14)	8.5 (5.9)
Ninemile Creek	20 (9)	20 (11)	4.8 (1.7)	3.3 (1.5)	2.6 (1.3)	1.6 (1.0)
Ley Creek	35 (10)	35 (12)	9.2 (2.3)	8.1 (3.7)	7.0 (5.8)	4.9 (2.9)
Forested[b]	4-24		–		3-7	
Agriculture[b]	25-81		–		9-22	
Urban	100-191		–		21-100	

[a]averages for 1995 – 2000 period.
[b]most frequently reported ranges (Budd and Meals 1994)

The shifts are less dramatic for the other two tributaries. The highest UALs for all three forms of P prevail for the urban portion of the watershed, a widely reported condition for other lakes with urban areas in their basin. The ordering of the four basins/sub-basins for each of these forms of P was Onondaga Creek urban > Ley Creek > Ninemile Creek > Onondaga Creek rural (Table 5). More than 40% of the P load for the urban area of Onondaga Creek was in a dissolved form, while these forms represented ≤26% in the other areas. Comparison of TP and SRP UALs to the common literature ranges indicates these basins/sub-basins do not represent particularly rich targets for major reductions in tributary loading (Table 5). The TP UALs for rural Onondaga Creek and Ninemile Creek have been within the range of forested watersheds, while Ley Creek has been in the lower portion of the range for agriculture. The TP and SRP UALs for the urban portion of Onondaga Creek are distinctly lower than the most frequently reported range for urban areas. Yet further reductions in loading from this urban area will result from the on-going CSO abatement program. The benefit claimed for the CSO program from stage I to stage II appears to be unreasonably optimistic (reductions in TP load ~10 $kg \cdot d^{-1}$; NYSDEC 1998) as this corresponds to a reduction in the urban Onondaga Creek TP UAL (for 53 km^2 watershed; footnote c, Table 1) of nearly 70 $kg \cdot km^{-2} \cdot y^{-1}$ (more than the total prevailing level; Table 5). The SRP UALs for rural Onondaga Creek and Ninemile Creek fall somewhat below the most frequently reported range for forested areas (Table 5). Ley Creek is more enriched; this SRP UAL falls within the upper portion of the forested range.

The goal of a 50% reduction in total non-point P loading (NYSDEC 1998), without regard to the aforementioned attenuating processes, appears to be unrealistically high based on prevailing conditions (Table 5). The enrichment observed within the urban portion of Onondaga Creek during dry weather indicates further reductions are possible. An optimistic reduction of 50% in loading from the urban portion of Onondaga Creek would result in an UAL of ~30 $kg \cdot km^{-2} \cdot y^{-1}$, that would be unusually low for an urban area (Table 5). Ley Creek is probably the next best target, though its contribution to overall tributary loading is modest. A 20% reduction in P loading from Ley Creek may be an attainable goal. Given the relatively low UALs that prevail for the non-urban portions of the lake's watershed (Table 5), and published results from areas where "best management practices" have been implemented (Johnson et al. 1978), a 10% reduction in P loading from these areas is a reasonable upper bound. According to these assumed individual reductions, about a 20% reduction in total tributary P loading represents a reasonable upper bound goal. Much of the present loading of bioavailable P from the tributaries probably corresponds to natural background inputs.

Synthesis: Effective P Loading, Partitioning Contributions

Partitioning contributions of pollutants is fundamental management information to support rehabilitation programs. Several processes and factors have been described here that influence the effective loading of P; i.e., that will regulate summer average epilimnetic

TP concentrations and related features of water quality. Despite substantial uncertainties that prevail concerning the quantitative effects of certain of these processes and factors, it is important to provide a preliminary synthesis of the information so that managers recognize appropriate targets for management action, and to bring focus to future research needs to reduce uncertainties. Phosphorus loading has been partitioned according to the TDP (Fig. 9a) and PP (Fig. 9b) fractions, and the sources according to the fully treated Metro effluent, the Metro bypass, the tributaries, and the Seneca River. Apparent average loading rates are attenuated (moving left to right in Fig. 9; exception, Seneca River TDP load) differently for the P fractions and various sources, according to the preceding analyses.

The apparent PP loading rates are modified by more effects than the TDP fraction, these include (Fig. 9a): (1) errors from not correcting TP analyses for turbidity effects, (2) adjustments for the response of the lake to the seasonality of loading, (3) the bioavailability of the inputs, (4) deposition out of the productive layers, and (5) plunging to depths below the productive layers. The TDP fraction of the external load is influenced only by the seasonality and plunging factors (Fig. 9b). The effectiveness of the existing Metro PP load (i.e., potential to support phytoplankton growth) is presently only substantially diminished by its incomplete bioavailability (Fig. 9a). In sharp contrast, the apparent annual PP load of the lake tributaries (e.g., Table 1) is attenuated substantially by each of the factors; about 35% by analytical error, 10% by seasonality of the load, 45% by the limited bioavailability, 45% by deposition, and 20% by plunging below the productive layers (percentages based on preceding loading rate, Fig. 9a). Accordingly, less than 15% of the apparent annual PP load from the lake's tributaries (e.g., Table 1) is expected to be manifested as an effective load (Fig. 9a). The speculative treatment of the Metro bypass included here invoked several assumptions: (1) equal partitioning of P between the PP and TDP fractions, (2) bioavailability and deposition characteristics of the PP fraction equivalent to the fully treated Metro effluent, and (3) the analytical errors observed for Onondaga Creek on a annual basis. The relative effect of the river inflow phenomenon, which largely coincides with the critical loading interval (Figs. 2 and 3), is understated if it is distributed over an annual period (Fig. 9a and b). Attenuation of the modest PP loading from this source is highly speculative; behavior identical to the Metro input was assumed. The TDP load from Metro remains largely unattenuated by these processes, while the effects of apparent annual tributary TDP inputs are diminished by seasonality (~30%) and plunging (~20%; Fig. 9b).

This analysis has demonstrated that effective P loading from Metro has been understated and tributary inputs have been overstated in the phase I TMDL, associated with the following factors: (1) false high measurements of TP for tributaries, (2) lack of recognition of the greater bioavailability of the Metro input, (3) lack of recognition of the importance of seasonal versus annual loading for the tributaries, (4) lack of recognition of the higher settling velocities of tributary (inorganic) versus Metro (organic) bioavailable PP, and (5) lack of recognition of the plunging of a substantial fraction of the tributary inputs during the critical summer interval. Accommodation of these factors results in a much more dominant contribution (> 85%) to effective P loading from Metro (fully treated plus bypass) under the prevailing conditions compared to the TMDL analysis (68%, Table 1), for the case of no river inflow (Fig. 10a). Including the river inflow, the estimated average contributions of Metro, the tributaries, and the river, under prevailing conditions, are about 80, 13, and 7%, respectively (Fig. 10b).

It is critical to recognize the different extents to which each of the sources is manageable. The contribution from the fully treated Metro effluent is entirely manageable; this input could be eliminated through diversion to the river (Effler and Doerr 1996), though the adopted plan instead calls for a 90% reduction in apparent P loading (Table 2; reduction in effective P loading remains uncertain in the absence of more information concerning the future effluent; e.g., TDP:TP ratio, bioavailability and settling character of PP fraction). A variety of treatment options can be used to reduce the availability of P in irregularly occurring inputs such as the Metro bypass (Cooke et al. 1993). The previously unidentified river input can be eliminated through changes in operation of the river system (e.g., Owens and Effler 1996a), modification of the upstream river channel, and/or installation of control facilities on the lake outlet. In contrast to these inputs, a substantial portion (~80%) of the tributary effective P load is not subject to reduction, as it corresponds to natural background inputs. Consideration of the scenario of successful implementation of phase III treatment at Metro (effluent TP = 20 $\mu g \cdot L^{-1}$), with all other inputs at prevailing levels (Fig. 10c), is valuable in considering the relative richness of the remaining potential targets for additional loading reductions. The fully treated Metro effluent P is assumed to be (conservative) completely bioavailable, consistent with filtration-type treatment options (e.g., all dissolved P in effluent) that will be necessary to meet this rigorous effluent standard (e.g., Delaware Engineering and New York City Department of Environmental Protection 2000). Under these conditions, the single largest source of effective P loading would be the lake tributaries,

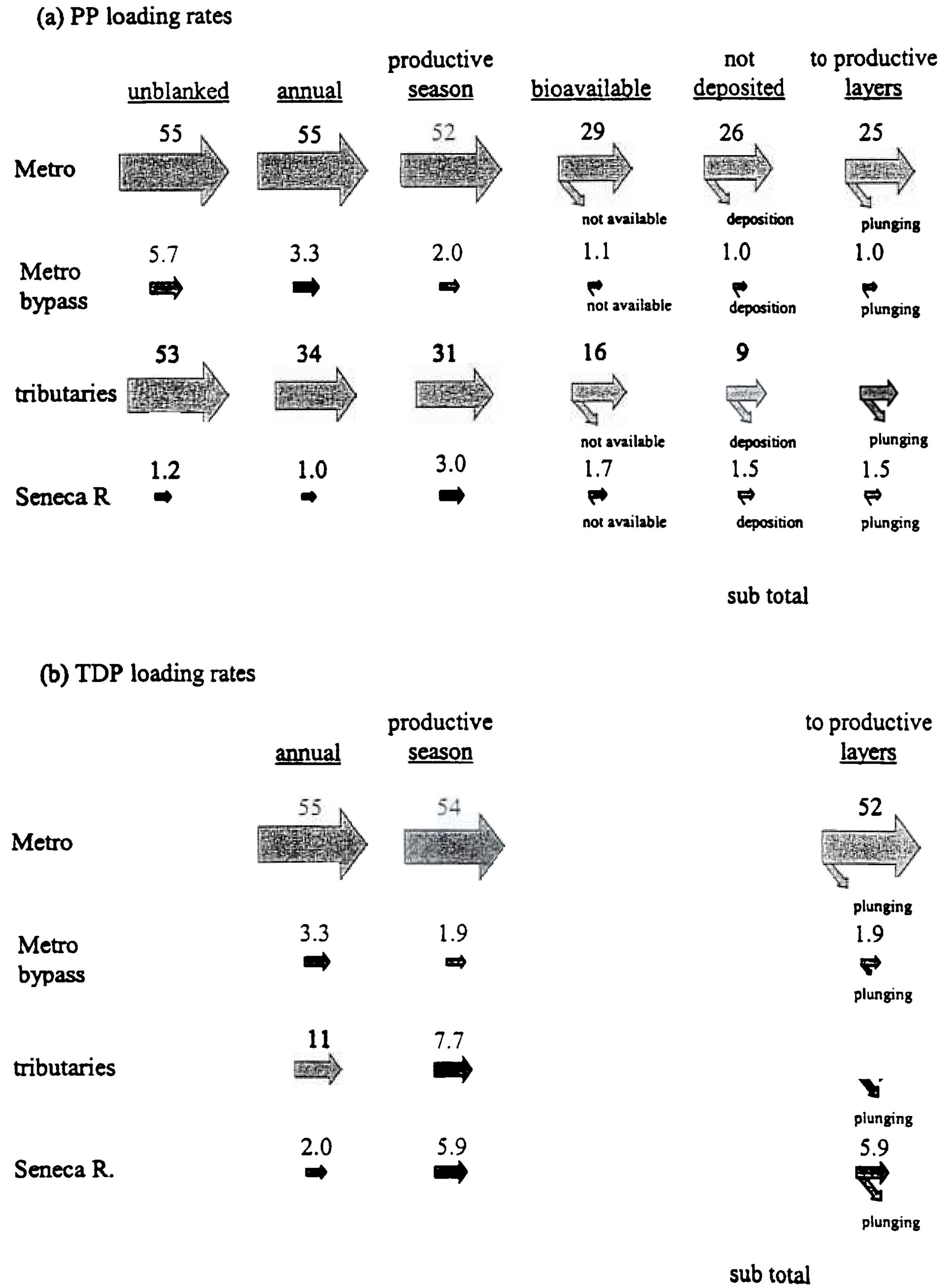

Figure 9.–Synthesis of factors modifying the effectiveness of phosphorus (P) loads (kg·d^{-1}) to Onondaga Lake, according to sources and P fractions: (a) particulate P, and (b) total dissolved P. Estimates of effective average loads are in the righthand column.

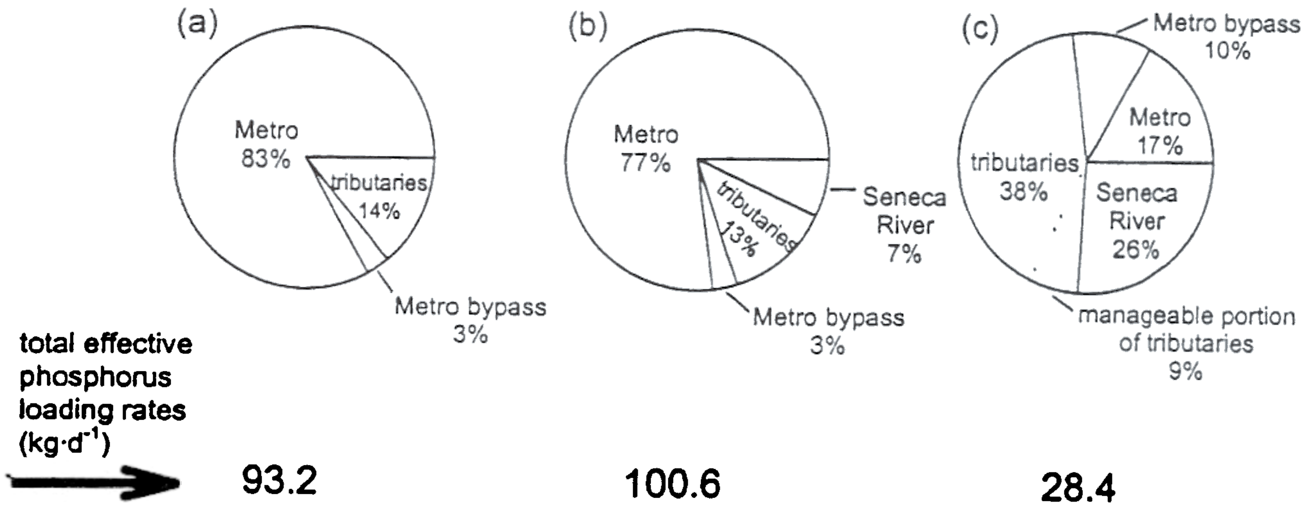

Figure 10.–Contribution to effective phosphorus (P) loading to Onondaga Lake: (a) prevailing conditions, for the case of no inflow from the Seneca River, (b) prevailing conditions, including inflow from the river, and (c) stage III Metro effluent limit for April–September interval, other sources at prevailing rates. Total effective P loading rate for each case appears below the corresponding pie chart.

however, the portion subject to management would represent the smallest of the sources (Fig. 10c). The load associated with the Seneca River inflow would be the single largest manageable input. The load carried by the irregularly operating Metro bypass is estimated to approach that associated with the fully treated Metro effluent. If loading reductions beyond those associated with the phase III Metro effluent P limit are found to be necessary to reach related water quality goals for the lake, the richest targets appear to be the Seneca River inflow (e.g., eliminate), the Metro bypass (e.g., P treatment), and the Metro effluent (e.g., diversion).

Conclusions

Based on the limitations identified here, the existing P TMDL analysis for Onondaga Lake cannot be considered a reliable basis to guide rehabilitation of Onondaga Lake's extreme problems associated with cultural eutrophication and to meet the specified numeric goal. Specifically, the TMDL analysis did not accommodate important system-specific characteristics, such as: (1) the substantial P load from the Seneca River, (2) the seasonal plunging of tributaries in the lake, (3) the different bioavailabilities of particulate P in the various sources, (4) the different settling velocities of particulate P from these sources, (5) the need to correct TP laboratory analyses on samples from these tributaries for turbidity effects, and (6) the implications of the high flushing rate of the lake for relative seasonal impacts of external loads.

It is doubtful that critical environmental conditions, particularly related to natural variations in hydrologic inputs, were adequately accommodated in the analysis. The model adopted in the TMDL analysis was inappropriate because it could not accommodate the array of factors and processes identified here as important. The short-comings in the P TMDL analysis resulted in understatement of the present role of the Metro discharge(s) in regulating summertime epilimnetic P concentrations and related features of water quality, and the overstatement of the importance of tributary contributions. The goal for non-point loading reductions incorporated in the TMDL analysis is almost certainly not feasible. Further, reductions in sediment and associated P loading that may result from erosion control in rural areas, identified as a potential target in the analysis, should not be expected to substantially reduce effective P loading because of the low bioavailability and high settling velocity of this material. The time averaging feature (twelve month rolling average) adopted in the permit for the TP loads and concentrations in the Metro effluent is inconsistent with the response time of the lake and is not protective of important summertime conditions in the lake.

Recommendations

An upgraded P TMDL analysis for Onondaga Lake is required that will provide a credible quantitative basis to develop a management plan(s) that will lead to meeting the applicable numeric goal and related features of water quality. Key components of this upgrade should include: (1) the development, testing, and application of an appropriate mechanistic model that can accommodate the phenomena/processes identified here, as well as represent P cycling within the water column, (2) the conduct of a number of specialty studies to quantify these, and other important, processes in the model, (3) representative monitoring of the water column of the lake, all significant inputs, and other environmental forcing conditions, to quantify state variables and other model inputs, (4) a re-evaluation by managers, supported by the above efforts, of the richness of the various potential targets to reduce or eliminate P loads, and (5) application of the model and supporting information to identify feasible management alternatives to meet the applicable numeric goal.

A number of process studies need to be conducted to more completely specify processes identified here as important, including: (1) determination of the magnitude and seasonality of the inflow of Seneca River into the lake, for a range of runoff conditions, (2) determination of the magnitude and seasonality of the plunging underflow phenomenon on a tributary-specific basis for a range of runoff conditions, (3) evaluation of the fate of P that enters via a plunging inflow(s), (4) quantification of depositional losses and settling velocities of PP in the lake, according to particle classes, and (5) specification of the bioavailability of all existing and future significant P sources, on a seasonal basis, for a range of runoff conditions. Uncertainty in prevailing loading estimates needs to be reduced and variability characteristics need to be better defined, for the tributaries, the Metro bypass, CSOs, and the inflow from the river. Further, the magnitude of internal loading from sediment releases during winter should be assessed and integrated into the upgraded TMDL analysis.

Equipped with the information presented here, an upgraded model, and improved processes and monitoring information, managers should reconsider the strategies manifested in the existing TMDL analysis. The previously omitted inflow from the river could, according to the existing plan, represent the largest manageable source of P, and could be responsible for not meeting the water quality goal. Elimination of this inflow, and the associated stratified flow regime imparted to downstream portions of the river (Effler et al. 1997), would also eliminate related water quality problems in the river (Canale et al. 1995). If yet further reductions in external P loading are found to be necessary, managers should consider attainable decreases in loading from the Metro by-pass and non-point sources, and if necessary, diversion of the Metro effluent to the river.

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CONTENTS OF LAKE AND RESERVOIR MANAGEMENT

Volume 18, No. 2, June 2002

Cover Photos: An urban lake setting, Onondaga Lake: top (Bruce Wagner) City of Syracuse skyline overlooking the lake, lower left (Bruce Wagner) – discharge pipe for effluent of metropolitan wastewater treatment facility, southern end of lake, lower right (David O'Donnell) – robotic monitoring buoy used in the Upstate Freshwater Institute's ongoing research program.

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