Domestic Waste Inputs of Nitrogen and Phosphorus to Onondaga Lake, and Water Quality Implications[1]

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ABSTRACT

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The recent history of loading of total phosphorus (TP) and nitrogen (N) species to Onondaga Lake, NY, from an adjoining domestic waste treatment facility (METRO), and related impacts on the lake, are documented. The discharge from METRO represents an extraordinary contribution to the inflow of the lake; e.g., in most years it is the largest source of water during the month of August. Approximately a 20-fold reduction in effluent TP concentration has been achieved by METRO over the 1970-1993 period, in response to a detergent ban and increased levels of treatment. The TP loading from the facility decreased about 5-fold over the 1977-1992 interval. A significant level of nitrification has been achieved during the warmer months since the late 1980s, but this has been attended by increased effluent concentrations of nitrite (NO_2^-). METRO presently contributes approximately 60, 90, and 80% of the total external loads of TP, ammonia, and total N, respectively. The extremely high prevailing external total loads of TP and total N, approximately 8 and 200 g/m²/y, respectively, severely impact the lake. Recurring degradations associated with the cultural eutrophication of the lake include excessive concentrations of phytoplankton, poor clarity, rapid loss of oxygen from the hypolimnion, and lake-wide oxygen depletion during the fall mixing period. Related water quality standards for transparency and oxygen are violated. A state guidance value for summer epilimnetic TP concentration of 20 μg/L is exceeded by a factor of ≥ 3 annually. Standards to protect aquatic life against the toxic effects of free ammonia and NO_2^- are violated by a large margin routinely in the upper waters in summer. METRO is the appropriate focus for related remediation efforts.

Key Words: domestic waste, phosphorus, nitrogen, external loads, cultural eutrophication, clarity, free ammonia, nitrite, standards, violations.

Anthropogenic inputs of phosphorus (P) and nitrogen (N) can have important water quality implications for receiving waters. Phosphorus loading is the principal regulator of primary productivity in most lakes (e.g., Vollenweider 1975, 1982, Wetzel 1983). Extreme cases of cultural eutrophication are manifested as conspicuous water quality problems, such as phytoplankton blooms, poor clarity, and rapid depletion of hypolimnetic dissolved oxygen (DO) associated with the decay of depositing phytoplankton. Reduction of external P loading has been a frequently applied lake restoration technique to eliminate or ameliorate these unwanted characteristics (Cooke et al. 1993). A rather wide range of lake response (e.g., degree of success) to P-based management has been observed (e.g., Edmondson and Lehman 1981, Larsen et al. 1981, Welch et al. 1986). Nitrogen, in the form of ammonium ion (NH_4^+) and nitrate (NO_3^-), is also an important plant nutrient, though rarely limiting in fresh waters (Harris 1986, Wetzel 1983). Ammonia and its organic precursors can represent a significant oxygen sink (Bowie et al. 1985, Harris 1986), and unionized ammonia (or free ammonia, NH_3) and nitrite (NO_2^-) are toxic to fish (Russo and Thurston 1977, USEPA 1985) at low concentrations.

Effluent from domestic wastewater treatment plants is an important anthropogenic source of P and N (Brezonik 1972, Thomann and Mueller 1987, Vollenweider 1975); concentrations vary greatly depending on the level of treatment (Thomann and Mueller 1987). Here we characterize and quantify the loading of P and N from a domestic wastewater treatment plant to Onondaga Lake, NY, and document recent changes in loading in response to management actions. Further, selected features of the impacts of this discharge on the water quality of the lake are described.

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Particular emphasis is given to the status of the lake with respect to numerical regulatory standards.

Recent History of METRO

In 1960, Onondaga County completed construction of a domestic wastewater treatment facility, the Metropolitan Syracuse Wastewater Treatment Plant (METRO), that served most of the residents in the Onondaga Lake watershed. The facility, located on the southeastern shore of the lake (Fig. 1), provided primary treatment. A combined sewer system (i.e., carries domestic waste and runoff) serves this region. METRO was designed to treat an average flow of 2.19 m^3/s (50 million gallons/d (MGD)), and could accommodate a peak flow of 7.45 m^3/s (170 MGD). According to the original plans for the facility, the METRO effluent was to be pumped around the lake, combined with effluent from a small treatment facility located about 2 km east of the lake, and discharged to the Seneca River (Fig. 1). The diversion part of the plan was never executed; instead the METRO effluent was discharged to the southern end of Onondaga Lake (Fig. 1). By the early 1970s the METRO facility was overloaded, with associated manifestations of poor performance (USEPA 1974).

Major upgrades to METRO were made from the late 1970s to the early 1980s. The facility provided secondary treatment by 1979, with the contact stabilization modification of activated sludge. This form of biological treatment was not expected to achieve significant ammonia removal (e.g., nitrification) in this climate (e.g., USEPA 1974). Advanced, or tertiary treatment, in the form of P removal by chemical precipitation, was on line by mid-1981. This process was unusual in that P was precipitated by addition of calcium enriched ionic waste (0.28 m^3/s (6.5 MGD), by design) from a nearby chemical manufacturer (Effler and Hennigan 1996). The effluent standard for total P (TP), established for facilities of this size in the Great Lakes watershed, is 1.0 mg/L. The upgraded facility was designed to treat an average flow of 3.5 m^3/s (80 MGD). Flows up to 5.26 m^3/s (120 MGD) receive full treatment and are discharged to the lake via a shoreline outfall. Peak flows of 9.77 m^3/s (223 MGD) can be accommodated. Flows in excess of 5.26 m^3/s (120 MGD; e.g., during runoff events) receive incomplete treatment (primary treatment and chlorination; "by-pass" flow) and are discharged at a depth of about 6 m. The alternatives of discharging METRO effluent to the Seneca River or directly to Lake Ontario were dismissed during these upgrades because the assimilative capacity of the river was judged to be inadequate and the cost for the Lake Ontario discharge option was considered too great (USEPA 1974).

Tertiary treatment was interrupted for significant periods during 1981, 1982, and 1983, usually in response to operational problems caused by the reception of the industrial ionic waste (excessive precipitation of $CaCO_3$, which occluded pipes and coated tanks). The closure of the industry in 1986 required the development of an alternative chemical treatment methodology. Phosphorus is now precipitated with ferrous sulfate, a much more widely used approach. The facility irregularly fails to meet its New York State permit requirements for total suspended solids, BOD, setteable solids, pH, TP, and residual chlorine.

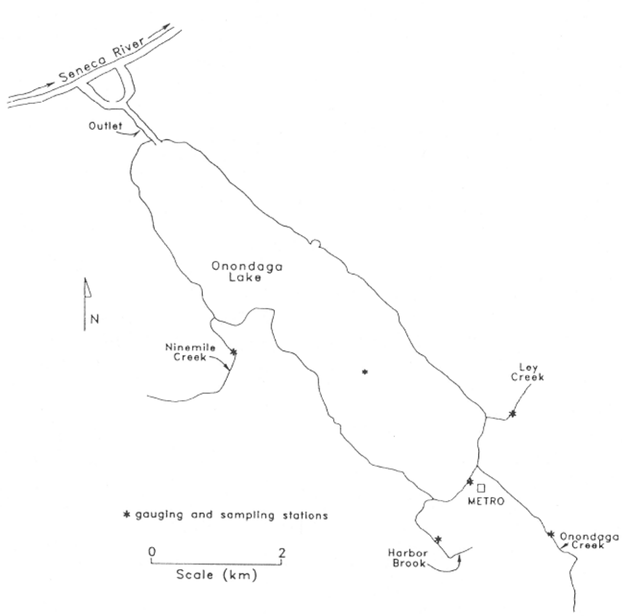

Figure 1.–Onondaga Lake with the positions of METRO, significant tributaries, and approximate locations of flow gauges and monitoring sites.

Methods

Onondaga Lake is a medium sized (surface area of 12 km^2 and mean depth of 10.9 m), high flushing rate (average of 3.9 flushes/y), dimictic, urban lake that adjoins Syracuse, NY. Since closure of the chemical manufacturer (1986), summer stratification has been established by May and the onset of complete fall turnover has occurred by mid-to-late October (Effler and Owens 1996). More background information on the lake, including the history of the development of the area, the lake's setting, morphometry, hydrology, and selected features of its degraded state, is available from Effler and Hennigan (1996).

The analysis presented here uses data collected as

Table 1: Salient features of supporting monitoring programs.

Monitoring Component	Responsible Party	Parameters[1]	Description
hydrology			
tributary	USGS	flow	continuous gauges since early 1970s
METRO	Onondaga County[2]	flow	continuous measurements
METRO effluent concentrations	Onondaga County	TIP	2/mo, 1970-1979
		$T\text{-}NH_3$, TKN	2/mo, 1970-1979
		T-NOx	2/mo, since 1970
	Onondaga County[2]	TP	daily, since 1980
	Onondaga County[2]	$T\text{-}NH_3$	5d/wk, since 1989
	Onondaga County[2]	TKN	2d/wk, since 1989
	authors	NO_2^-	2/mo 1991 and 1992; 2-3 d/wk part of 1993
tributary concentrations	P - by authors N - Onondaga County and authors	TP, $T\text{-}NH_3$, NO_3^-, NO_2^- TKN	2/mo routinely, augmented in certain years by more frequent measurements by authors[3]
lake measurements	authors	$T\text{-}NH_3$, NO_2^-, TP -	general: weekly, routine deep water site (19 m), ≤ 2 m depth profiles
		DO, temp., pH	Hydrolab Surveyor 3 (since 1991)
		Secchi disc,	12 cm diameter, black/white quadrant
		Chl[4]	

[1] chemical analyses according to APHA (1980, 1985).
[2] Onondaga County (METRO) permit requirements, reported monthly to NYSDE.
[3] details of augmented portions of program presented by Effler and Whitehead (1996).
[4] chlorophyll.

part of several monitoring programs, including: 1) measurements made at METRO as part of its permit requirements, 2) tributary flow measurements made by the United States Geological Survey, starting in the early 1970s, 3) measurements made by the authors, and 4) results reported by Onondaga County (1971-1994) as part of an annual monitoring program. Salient features of these programs that support this analysis are specified here (Table 1).

As the impact of the METRO facility has come under closer scrutiny in recent years, sampling frequency and the number of parameters measured in the facility's effluent have both increased (Table 1). Daily TP monitoring commenced in 1980. Earlier monitoring of P in the plant's effluent was limited to bi-monthly (2/mo) analyses of total inorganic P (TIP; described as "total acid hydrolyzable phosphorus" by the American Public Health Association (APHA 1989)). Values of TIP are systematically lower than TP measurements because organic (particularly particulate) forms of P are not detected (weaker digestion procedure for TIP (APHA 1989)). Onondaga County (1971-1990) also measured TIP, instead of TP, in Onondaga Lake and its tributaries until 1989. The more frequent monitoring of the various N species in the METRO effluent was dictated by modifications in the permit requirements in 1989 (Table 1). Samples from the lake tributaries, and METRO until 1980, were grab type. Samples from METRO over the 1980-1988 interval were time composites, thereafter they have been flow-weighted composites.

We commenced our monitoring program for TP in the tributaries and the lake in 1987. All N and P analytes were measured according to standard methods (e.g., APHA 1985, 1989). Chlorophyll *a* and phaeophytin concentrations in the lake were measured according to Lorenzen (1967) through 1986. Thereafter total chlorophyll was measured according to Parsons et al. (1984). The lake was monitored at a deep water location in the south basin (Fig. 1) found to be generally representative of lake-wide conditions (Effler 1996).

Annual loads were based on summation of daily loading estimates for METRO and the tributaries (Fig. 1); other sources of N and P are insignificant by comparison. Daily estimates for METRO were the product of daily flows and daily concentrations. Concentrations for days without measurements (e.g., < 1980) were estimated by time interpolation. Estimates for the pre-1980 period utilized an empirical expression based on paired TP and TIP measurements, developed by Effler and Whitehead (1996). Tributary daily loading estimates were developed using FLUX (version 4.4, 1990), loading analysis software developed by Walker (1987). The details of the development of the tributary loads were described by Effler and Whitehead (1996).

The USEPA (1985) developed criteria intended to protect against the toxic effects of NH_3, that were subsequently adopted as standards by the State of New York. Both the standards (acute and chronic) and the fraction of total ammonia (T-NH_3) that exists as NH_3 are functions of temperature and pH. Thus they are subject to substantial seasonal variations in systems such as Onondaga Lake (e.g., Effler et al. 1990). The protocols to estimate NH_3 concentrations and the standards, as a function of pH and temperature, were reviewed by Effler et al. (1990). These procedures were used to evaluate the status of Onondaga Lake with respect to these standards.

The depletion of hypolimnetic DO in Onondaga Lake is represented by the "areal hypolimnetic oxygen deficit" (AHOD; $g/m^2/d$). AHOD has been widely used as an indicator of trophic state (Hutchinson 1957, Mortimer 1941, Wetzel 1983). Mortimer (1941) proposed limits of 0.25 $g/m^2/d$ for the upper limit of oligotrophy and 0.55 $g/m^2/d$ for the lower limit of eutrophy. Estimates of AHOD (e.g., Wetzel and Likens 1991) could not be made for those years in which spring turnover did not occur (an impact of the chemical manufacturing facility, see Effler and Owens 1996, Effler and Perkins 1987), as the lower layers were not replenished with oxygen following the winter depletion (e.g., Effler and Perkins 1987).

METRO Inputs

Hydrology

Selected features of the hydrologic budget of Onondaga Lake for the 1971-1989 period were reviewed by Effler and Hennigan (1996). The effluent flow into the lake from METRO over this period was about 19% of the total inflow. We know of no other lake in the United States that receives this much of its inflow as treated wastewater. The next largest contribution of treated domestic waste to an inland lake in New York State is the Ithaca WWTP (0.44 m^3/s or 10 MGD), which represents < 2% of the total inflow received by Cayuga Lake.

The situation for Onondaga Lake is even more severe than implied by the annual average inflow, when the seasonality of runoff for the watershed is considered (Fig. 2). The relatively constant volume of effluent discharged from METRO in combination with the strong seasonality in hydrologic loading from tributaries

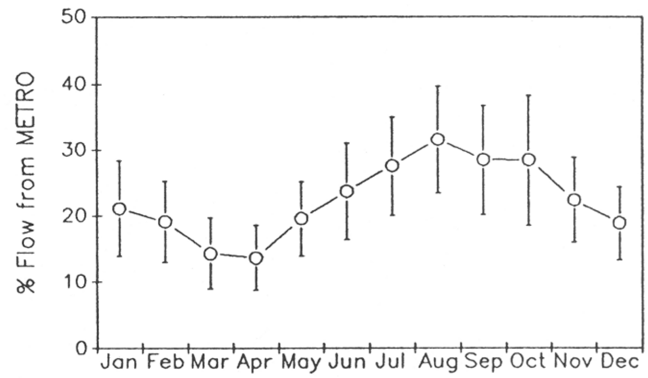

Figure 2.–Seasonality in the contribution of the METRO discharge to total inflow to Onondaga Lake, average monthly % for period 1971-1989, with ± 1 standard deviation bars.

(Effler and Hennigan 1996) results in major seasonality in the contribution of METRO to total inflow, though substantial interannual variability is observed (Fig. 2). METRO's contribution is smallest in spring (e.g., about 13% on average for April) and largest in mid- to late summer (e.g., about 31% on average for August; Fig. 2), when primary productivity (e.g., Wetzel 1983) and recreational demands are greatest. The METRO discharge represented 28%, on average, of the total inflow to the lake over the critical water quality interval of June-September, for the 1971-1989 period. This wastewater input was the single largest source of water to the lake for the month of August in 14 y over the 1971-1989 interval.

Phosphorus

Annual average TP concentrations in the METRO effluent for the 1970-1993 period are presented here (Fig. 3a), but corresponding estimates of loads are not presented for the 1970-1976 interval (Fig. 3b), because short-comings in this portion of the data base make estimates for those years unreliable. Reliable estimates of total loading from tributaries are limited to the 1987-1993 interval (Fig. 3b). Relative contributions of METRO to total loading are presented for 1987 and 1989 as insets (Fig. 3b). METRO loads correspond to the fully treated effluent, with the exception of 1993. The relative contributions of partially treated "by-pass" (e.g., runoff events) discharges to the facility's TP load were minor (e.g., < 8%; Effler and Whitehead 1996), except in 1993, when about 22% of the released flow did not receive full treatment. This was due to unusually high spring runoff and construction activity at METRO. Substantial disparity exists within this monitoring record with respect to temporal coverage and analytical methods (e.g., TIP versus TP; Table 1). METRO effluent TP concentrations and loading information became more reliable starting in 1980 when daily monitoring of TP commenced (Table 1).

Substantial reductions in effluent TP concentrations (Fig. 3a) and loads (Fig. 3b) have occurred in response to related management efforts, which include a ban on high-P detergents in Onondaga County (Murphy 1973), the addition of secondary treatment, the addition of tertiary treatment, and, most recently, improvements in tertiary treatment. Though poorly quantified, the single largest reduction was apparently associated with the detergent ban (Fig. 3a). Substantial short-term variations in TP concentrations in the plant's effluent have been observed throughout the period of daily monitoring (since 1980). Concentrations in excess of the standard (> 1 mg/L) have been documented annually, though only infrequently since 1989 (Effler and Whitehead 1996); 213 violations occurred in 1983, but only 25 in 1989. The annual average TP concentration in the METRO effluent over the 1989 - 1992 interval was about 0.6 mg/L, substantially less than the state standard of 1.0 mg/L. This represents a nearly twenty-fold reduction in the annual average TP concentration in the facility's effluent since 1970 (Fig. 3a).

The time course of estimated annual TP loads from METRO over the 1977-1993 interval (Fig. 3b) generally tracks that of the average concentrations (Fig. 3a). Note the increased load for 1993 (e.g., about 70% greater than for the previous 3 y), associated with the unusually high spring runoff (about 50% of the 70% increase) and construction activity (about 20% of the 70% increase) in the absence of an increase in the estimated tributary load (Effler et al. 1994). Tributary loading estimates for the 1991-1993 interval are less than those obtained for the 1987-1990 period. The cause(s) of this apparent decrease is presently unknown; it is not in response to a specific management action. METRO's contribution to the total annual TP load in 1987 was nearly 75%, and in 1989 it was about 60% (Fig. 3b). The 1987 case is considered generally representative of the facility's contribution when it operated at the 1 mg/L effluent standard, while the 1989 case reflects the improved performance of tertiary treatment (e.g., 0.6 mg/L) achieved in recent years since switching to ferrous sulfate treatment (e.g., Fig. 3a).

The prevailing external TP loading rate (i.e., sum of METRO and tributary inputs) is extremely high from the perspective of widely used semi-empirical, or screening, trophic state models (e.g., Vollenweider 1975). The average annual areal TP loading rate for the 1989-1992 interval, 7.8 g/m^2/y, for an average hydraulic overflow rate (annual average total inflow rate for the same period, divided by the surface area of the lake) of 39 m/y, corresponds to highly eutrophic conditions (cf. Vollenweider 1975).

A substantial portion of the total TP load to Onondaga Lake is subject to management reduction (Fig. 4). This includes the entire METRO contribution, which could be eliminated by diversion around the lake (also see Effler and Doerr 1996). Additionally, a portion of the tributary load could be eliminated. For example about 5-10% of the total TP load to the lake emanates from combined sewer overflows (Effler and Whitehead 1996), which discharge to lake tributaries after runoff events (Effler and Hennigan 1996). Perhaps another 3-6% of the total TP load could be eliminated by effective management practices in the rural portions of the lake's watershed (cf. Effler and Whitehead 1996). About 25-32% of the total load (or 60-80% of the tributary load) is probably not subject to further

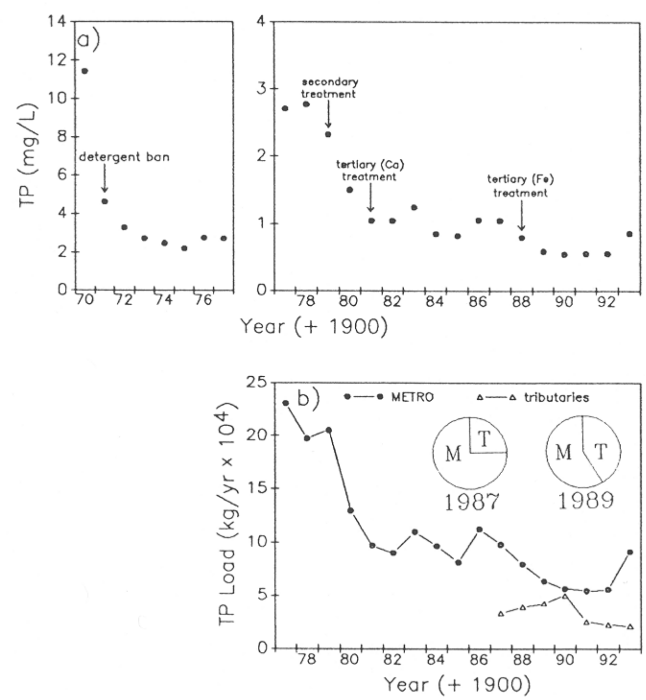

Figure 3.–Annual conditions: (a) METRO effluent average TP concentrations (y-axis segmented in time to resolve recent changes), and (b) METRO and tributary TP loads, with contributions for two selected years; M-METRO, and T-tributaries.

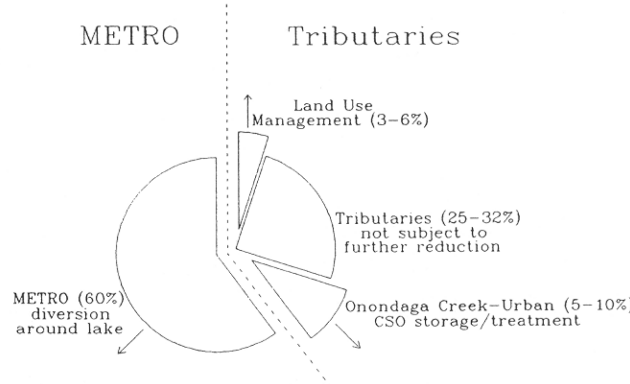

Figure 4.–Partitioning external TP loading to Onondaga Lake; identification of manageable fractions (arrows).

reduction. Clearly METRO should be the primary focus for P-based remediation efforts for the lake. Internal loading from the enriched hypolimnion to the productive upper waters during summer stratification, mediated by vertical mixing, presently is only about 10% of the annual external load (e.g., Auer et al. 1993). This internal load would decrease by more than a factor of two if the METRO input was eliminated (e.g., following the establishment of new steady-state conditions for the lake's sediments (Penn 1994)).

Nitrogen Species

Essentially all of the N discharged from METRO before the addition of biological treatment was as organic-N or T-NH_3. More than 50% was as organic-N (Effler and Whitehead 1996). Essentially no nitrification was achieved in the facility following the addition of biological treatment (e.g., see Fig. 5a) until the late 1980s (e.g., Fig. 5b and c), though there was a shift to greater concentrations of T-NH_3 and reduced concentrations of organic-N (Effler and Whitehead 1996). The seasonal distributions of N species in METRO's effluent presented for 1988 (Fig. 5b) and 1989 (Fig. 5c), that depict nitrified effluent for portions of the warmer (e.g., Fig. 5d) months, are generally representative of conditions since 1988.

The most conspicuous seasonal pattern in nitrogen species for 1988 and 1989 is associated with the operation of the nitrification process in the warmer months; in particular, T-NH_3 concentrations were lower and T-NO_x (NO_3^- plus NO_2^-) concentrations were higher (Fig. 5b and c), while total N concentrations remained relatively constant. The seasonality in nitrification in 1988 and 1989 (Fig. 5b and c) largely reflects the influence of temperature (Fig. 5d) on the kinetics of this microbial process (Bowie et al. 1985), and is consistent with observations for other contact stabilization facilities in north temperate climates (USEPA 1975). The details of the timing and extent of the nitrification process have differed year-to-year. For example, nitrification was achieved over a shorter period, and to a less complete extent, in 1988 (Fig. 5b) compared to 1989 (Fig. 5c). These interannual differences are probably attributable to modest differences in plant operation (e.g., sludge age).

Monitoring of the NO_2^- species in the METRO effluent has not been a permit requirement. However, limited monitoring conducted by the authors (Fig. 6) indicates NO_2^- represents a significant fraction of T-NO_x during the active nitrification period of summer. The dynamics of NO_2^- generally tracked those of T-NO_x during the summer of 1993. At times NO_2^- represented as much as 30% of T-NO_x during this interval (Fig. 6).

Increases in NO_2^- concentrations have been observed in the plant's effluent in other recent years during summer (unpublished data, S.W. Effler). Thus, in an effort to reduce the load of T-NH_3 to Onondaga Lake, the loading of another potentially toxic species, NO_2^-, has increased. These results, and the subsequently documented violations of the state standard for this

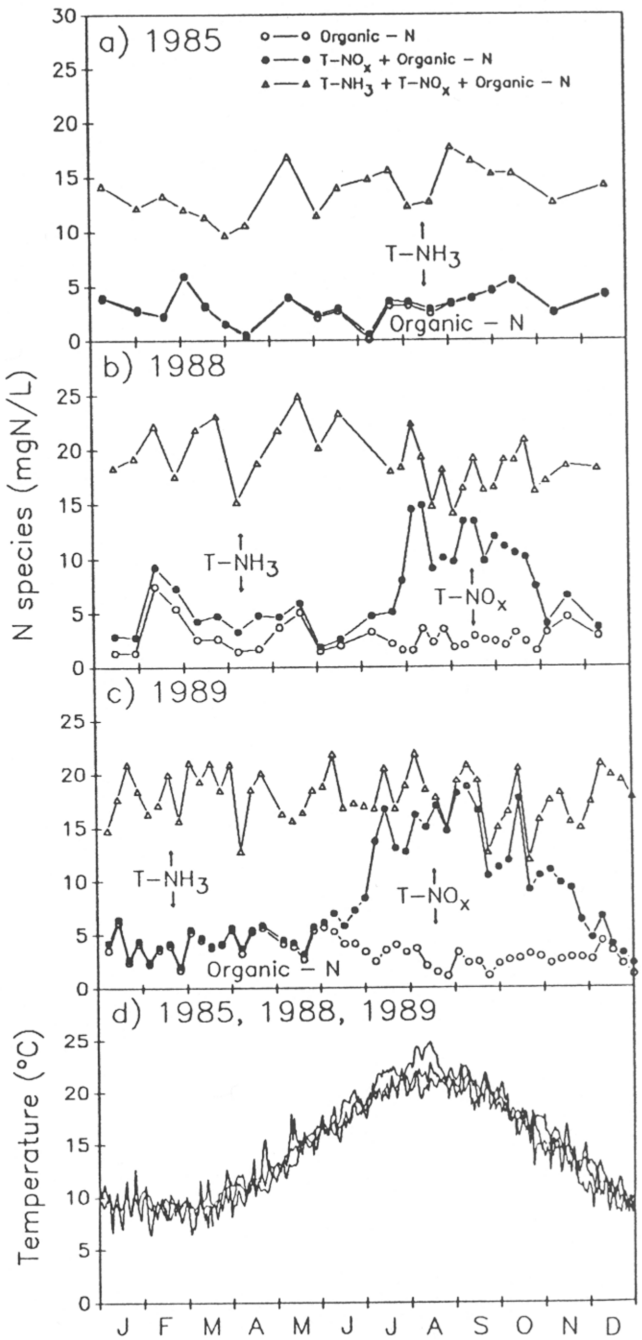

Figure 5.–Temporal distributions in the METRO effluent; (a) concentrations of N species in 1985, (b) concentrations of N species in 1988, (c) concentrations of N species in 1989, and (d) temperature for 1985, 1988, and 1989.

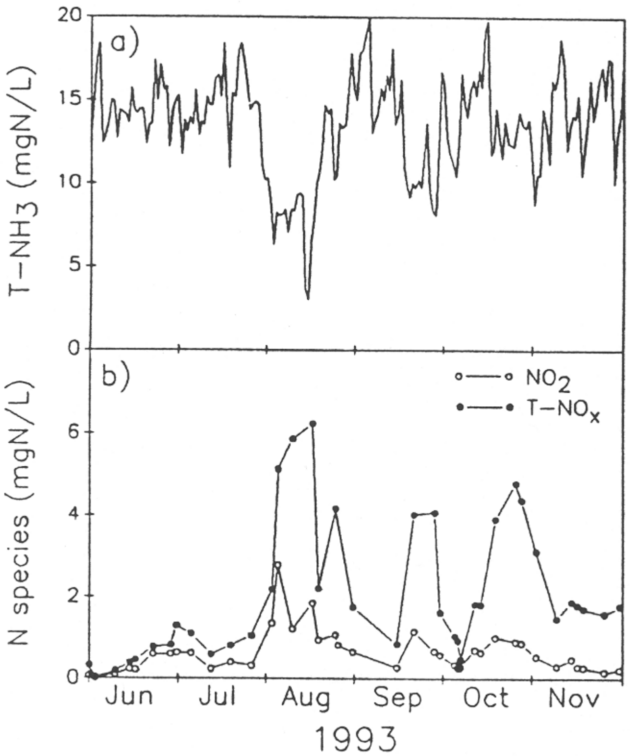

Figure 6.–Time series of selected N species in METRO's effluent in the June-November interval of 1993: (a) T-NH$_3$, and (b) T-NO$_x$ and NO$_2^-$.

species in the lake, indicate that NO$_2^-$ should be monitored routinely in the plant's effluent, and that the "trade-off" between the benefit of seasonal nitrification and the increase in NO$_2^-$ loading to the lake

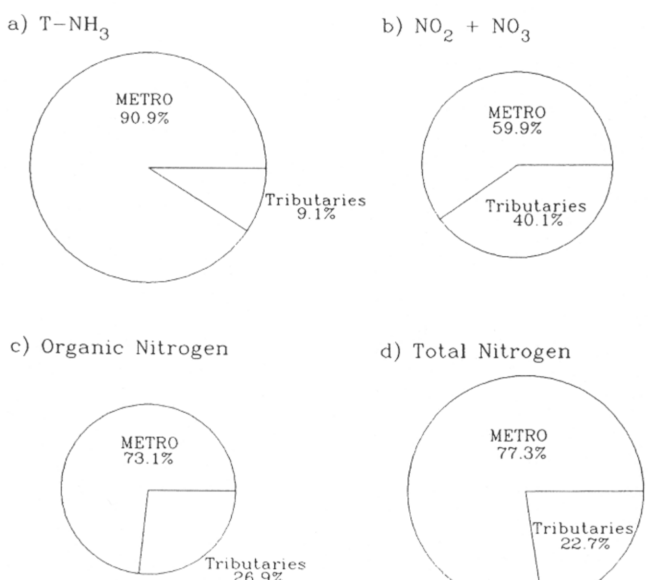

Figure 7.–Contributions of METRO and tributaries to loading of N species and total N to Onondaga Lake in 1989: (a) T-NH$_3$, (b) T-NO$_x$, (c) organic-N, and (d) total N. Relative areas of "pies" for three N species proportional to contribution of each to total N loading.

should be evaluated.

METRO is the dominant source of N to Onondaga Lake. The contributions of the METRO discharge to the loading of selected N species and total N (Fig. 7) for 1989, are representative of conditions that continue. The annual areal loading rate for total N to the lake (about 200 gN/m^2/y) exceeds by a factor of 1.5 the highest value included in the N loading compilation of Brezonik (1972). METRO represents about 90% of the T-NH$_3$ load (Fig. 7a), 60% of the T-NO$_x$ load (Fig. 7b), 70% of the organic-N load (Fig. 7c), and about 80% of the total N load (Fig. 7d).

Manifestations of Cultural Eutrophication

The paleolimnological analysis of Rowell (1996) shows that Onondaga Lake was oligo-mesotrophic to mesotrophic before colonial development of the watershed. Here we describe the highly eutrophic conditions that continue to prevail in the lake despite the reductions in P loading achieved at METRO (Fig. 3).

Lake TP Concentrations

The TP concentration in the upper productive layers of a lake is a widely used indicator of trophic state (e.g., Auer et al. 1986, Chapra and Dobson 1981, Vollenweider 1975, 1982). There is no numerical standard for TP in the State of New York, but there is a narrative standard, ".... none in amounts that will result in growths of algae, weeds, and slimes that will impair the best usages", and a recently established guidance value (open to some regulatory discretion) of 20 µg/L, as the mean summer epilimnetic TP concentration (New York State Department of Environmental Conservation (NYSDEC) 1993). The guidance value is based on empirical relationships with aesthetic effects for primary and secondary recreation (e.g., Kishbaugh 1993). This value is generally consistent with the upper bounds of mesotrophy proposed for TP concentrations by various researchers (see Auer et al. 1986, Chapra and Dobson 1981, Vollenweider, 1975, 1982).

Concentrations of TP were high in 1987, the highest in 1988, were lower in the 1989-1992 interval, and increased in 1993 (Table 2). The lowest margin of exceedance of the NYSDEC guideline value was nearly a factor of 3. Further, the TP concentration in all the years was substantially above levels described as indicative of eutrophy (e.g., Auer et al. 1986, Chapra and Dobson 1981, Vollenweider 1975, 1982).

Table 2.–Indicators of eutrophy for Onondaga lake, for the mid-May to mid-September interval.

Year	TP* (µg/L) $\bar{x}$	Chl⁺ (µg/L) $\bar{x}$	Chl⁺ (µg/L) median	SD** (m) $\bar{x}$	SD** (m) median	AHOD (g/m³/d)
1978	-	70*	55	.93	1.0	-
1981	-	-	-	.92	1.0	1.7
1985	-	63	59	.84	.85	-
1986	-	37	20	.77	.75	-
1987	105	72	46	1.42	1.3	1.7
1988	140	33	29	1.93	1.5	-
1989	74	36	24	2.22	1.7	1.9
1990	76	28	21	2.32	1.7	1.1
1991	56	36	21	2.90	2.0	1.4
1992	63	20	20	2.66	1.95	1.5
1993	90	26	22	3.05	1.8	-

* TP – epilimnetic summer average TP concentration; volume-weighted average for 0-8 m, mid-May to mid- September.
+ Chl - epilimnetic average chlorophyll; summation of chlorophyll *a* plus phaeophytin for 1978, 1985, 1986; as total chlorophyll in other years.
** SD - Secchi disc transparency.

The central role that loading from METRO plays in regulating TP concentrations in Onondaga Lake is evidenced by the strong linear relationship that has prevailed between the summertime mean epilimnetic TP concentration and the TP load from METRO over the April-September interval (Fig. 8). The lake responds rapidly to significant changes in loading because of its high flushing rate (Effler and Hennigan 1996). Approximately 97% of the year-to-year differences in epilimnetic TP concentrations over the 1987-1993 interval (with the exception 1988) can be explained by changes in METRO TP loading. The year 1988 is an outlier, with several possible explanations. For instance, the tributary monitoring program (Table 1) may not have captured the added loading associated with a sewer line break along one of the tributaries in that year (Effler and Whitehead 1996). More probably, incorrect high measurements of TP concentration in the lake were included in the observations for that year. Clearly lake TP concentrations are highly responsive to loads received from METRO (Fig. 8). It is unfortunate that the appropriate monitoring TP data for the lake are not available to document the response to earlier reductions in METRO loading (Fig. 3).

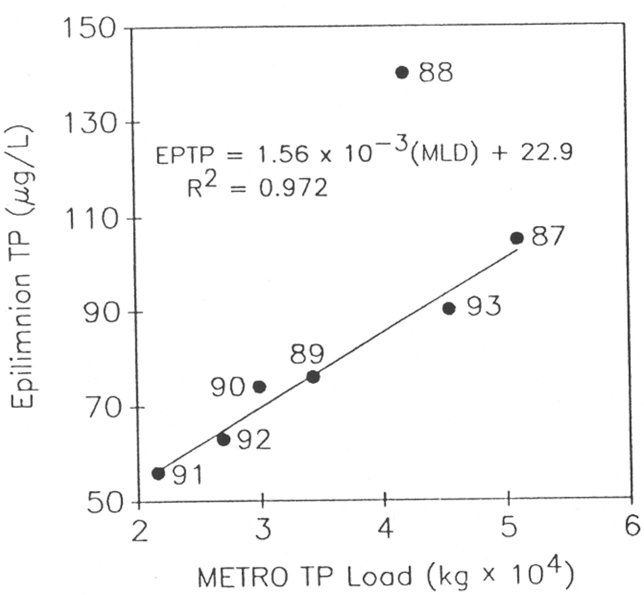

Figure 8.–Relationship between epilimnetic summer average TP concentration in Onondaga Lake and METRO loading over the April-September interval. Years designated by last two digits.

Phytoplankton and Clarity

Before closure of the soda ash facility (1986), chlorophyll concentrations tended to remain high (e.g., > 30 µg/L; Auer et al. 1996) throughout the productive period of late spring to early fall. Stronger fluctuations have been observed since 1987, characterized by distinctly lower and more abrupt minima (e.g., < 3 μg/L) compared to earlier years (Auer et al. 1996). The temporal changes in clarity in the lake are largely driven by the dynamics of phytoplankton biomass (Effler and Perkins 1996). Thus, abrupt increases in Secchi disc transparency (described as "clearing events") have attended the abrupt decreases in phytoplankton biomass and seasonal measures of clarity (Table 2) have improved. Summer average and median chlorophyll concentrations and Secchi disc tranparencies throughout the 1978-1993 interval (Table 2) are indicative of eutrophy (cf. Carlson 1977). The median is provided as a more robust indicator for the highly dynamic conditions that have prevailed annually since the closure of the soda ash/chlor-alkali facility (Auer et al. 1996).

Phosphorus is the limiting nutrient in Onondaga Lake (Auer et al. 1996). However, the available evidence (e.g., Connors et al. 1996) indicates that, despite substantial reductions in TP loading (Fig. 3) and lake concentrations (e.g., Fig. 8, Table 2) over the period of record, only a small degree of P limitation of phytoplankton growth has been achieved (i.e., other factors such light and temperature are regulating (e.g., Field and Effler 1983)). This is consistent with the literature; e.g., Auer et al. (1986) found that TP concentrations > 40 μg/L were essentially saturating in Green Bay, Lake Michigan. Reductions in phytoplankton biomass and increases in clarity in Onondaga Lake since closure of the soda ash/chlor-alkali facility (Table 2) are not a result of increased nutrient limitation, but can instead be attributed to increased zooplankton grazing ("top-down" control; Siegfried et al. 1996). The timing of changes in chlorophyll and Secchi disc transparency, when compared to temporal patterns of TP loading and lake TP concentrations (Table 2), further support this position. First, the distinct improvements in clarity and reductions in phytoplankton biomass (Table 2) emerged before reductions of external TP loading (Fig. 3) and coupled decreases in lake TP concentrations (Fig. 8) occurred. Second, both chlorophyll concentrations and clarity were insensitive to the increases in epilimnetic TP concentration that occurred in 1993 (and perhaps 1988, Table 2). One possible side-effect of increased zooplankton pressure has been the proliferation of nuisance filamentous cyanobacteria (particularly *Aphanizomenon flos-aquae*) which are resistant to grazing. Yet greater reductions in external TP loading would be required to systematically improve related features of water quality (see Effler and Doerr 1996) through P limitation.

Oxygen

The rapid depletion of DO from the lake's hypolimnion and the calculation of the associated AHOD is illustrated here for 1991 (Fig. 9a), based on volume-weighted concentrations (from profiles) below a depth of 10 m. The AHOD value for 1981 (Table 2) has been presented previously (Effler et al. 1986). The average AHOD for the six years for which deter-minations have been made was 1.54 $g/m^2/d$, the range was 1.1 to 1.9 $g/m^2/d$. These values are indicative of a highly eutrophic system (e.g., > 0.55 $g/m^2/d$; Mortimer 1941), and they approach the highest values in data bases used to develop predictive empirical expressions for AHOD (Cornett and Rigler 1979, Walker 1979, Welch and Perkins 1979). We assign no significant temporal trends to the results (Table 2). The substantial year-to-year variability is most likely a result of year-to-year differences in the magnitude and timing of vertical mixing in the lake (Effler et al. 1986, Owens and Effler 1989).

In Onondaga Lake, anoxia is established first in the lowermost layers of the hypolimnion because of oxygen-demanding processes at the sediment-water interface (Gelda et al. 1995). The depth interval of anoxia expands upward in a generally progressive manner, and may for brief periods extend from the lake bottom to within 5-6 m of the surface (e.g., Fig. 9b). The by-products of anaerobic metabolism, particularly methane (CH_4; Addess and Effler 1996) and hydrogen sulfide (H_2S; Effler et al. 1988), start to accumulate after the onset of anoxia. The highest concentrations of these anaerobic by-products develop in the lowermost layers (Fig. 9b), consistent with the localization of methanogenesis and sulfate reduction in the sediments (Effler et al. 1988, Fallon et al. 1980, Rudd and Hamilton 1978).

Lake-wide depletions of DO down to concentrations that violate surface water standards for New York State (< 5 mg/L, as a daily average; < 4 mg/L, at anytime within a day) are common (Fig. 9c) in the upper waters of Onondaga Lake during the approach to fall turnover. This fall DO "sag" phenomenon is illustrated as the average DO concentration in the upper 5 m of the lake over the September-November interval of 1990 (Fig. 9d). During that period, the DO concentration violated the standard for more than 2 weeks. Substantial year-to-year variations in the timing and severity of this phenomenon (e.g., Fig. 9c) are attributed largely to meteorological variability (Effler 1996, Gelda and Auer 1996). The exodus of most fish from the lake (e.g., to the Seneca River) during this low DO period has been documented by Tango and Ringler (1996). The severe oxygen depletion in autumn that results from oxidation of anaerobic metabolites that accumulate in the hypolimnion during summer (e.g., Fig. 9b) is a particularly severe and conspicuous manifestation of the cultural eutrophication of the lake (Effler et al. 1988, Addess and Effler 1996).

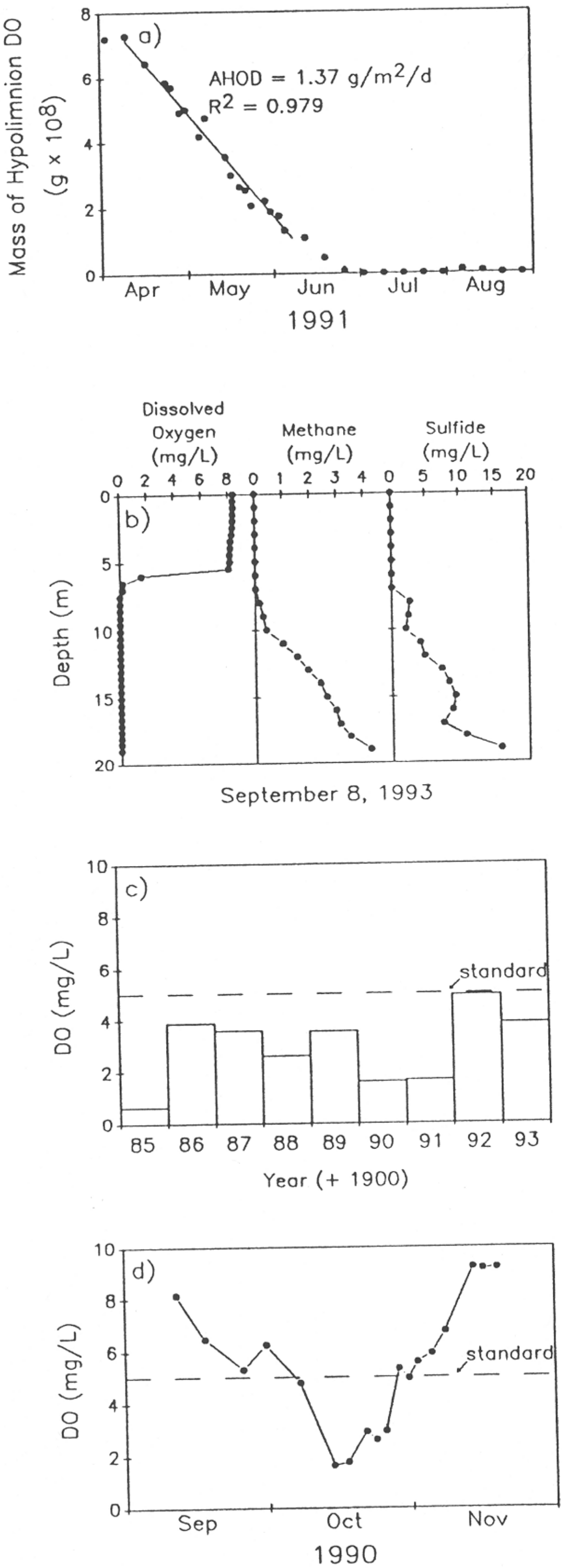

Figure 9.–Oxygen resources of Onondaga Lake: (a) hypolimnetic DO depletion and calculation of AHOD for 1991, (b) selected paired vertical profiles of DO, CH_4, and H_2S (Sept. 8, 1993), (c) annual fall DO minimum for upper 5 m, 1985-1993, and (d) fall DO (0-5 m average) sag and recovery in 1990.

Lake Status With Respect to NH_3 and NO_2^- Standards

NH_3

Ammonia (NH_3) concentrations in the upper waters of Onondaga Lake during the 1988-1993 period routinely violated the chronic non-salmonid toxicity standard and occasionally exceeded the acute non-salmonid standard (Fig. 10). Note that allowable levels of NH_3 are not a constant, but are a function of ambient temperature and pH (cf. Effler et al. 1990) which together determine the unionized fraction of T-NH_3 and the toxicity limits of NH_3. The 1988 conditions were documented earlier by Effler et al. (1990). The spring to early fall interval addressed in this treatment is the most critical because the higher temperatures (Effler and Owens 1996) and pH (Driscoll et al. 1994, Effler et al. 1990) values increase the fraction of T-NH_3 that exists as NH_3. Further, the most severe conditions are observed in the upper waters, rather than the hypolimnion, for the same reasons.

Violations of the acute standard were documented in 1988 and 1989, but not in subsequent years. The acute standard was undoubtedly exceeded, and more nearly approached, on more days than indicated in this analysis (Fig. 10a-f) as a result of photosynthetically - induced diurnal increases in pH. Violations of the chronic NH_3 standard, by a wide margin, is a recurring problem for the upper waters of the lake (Fig. 10 a-f). Violations persist for at least 2 months, and in some years for more than 5 months (Fig. 10a-f).

The year-to-year differences in the severity of NH_3 violations (Fig. 10) are related primarily to interannual differences in the T-NH_3 concentration in spring, and, to a lesser extent, pH. The spring T-NH_3 concentration is regulated largely by the dilution provided by tributary flow. Inflow from the tributaries is highly variable year-to-year (Effler and Hennigan 1996; Fig. 2), while the T-NH_3 load from METRO to the lake during the antecedent winter period is relatively uniform (e.g., Fig. 5b and c). Thus, the highest T-NH_3 concentrations in spring are expected in those years with the lowest tributary inflow during the proceeding winter. We have evaluated the relationship between spring (April) lake T-NH_3 concentrations and antecedent (November-March) tributary inflow within the context of a "dilution" (Manczak and Florczyk 1974) model (Fig. 11). Variations in the inverse of total antecedent tributary inflow explained 65% of the variability in T-NH_3 concentrations observed in the lake in April for the 6 y of the 1988-1993 interval (Fig. 11). The approximate linearity of the relationship is consistent with the

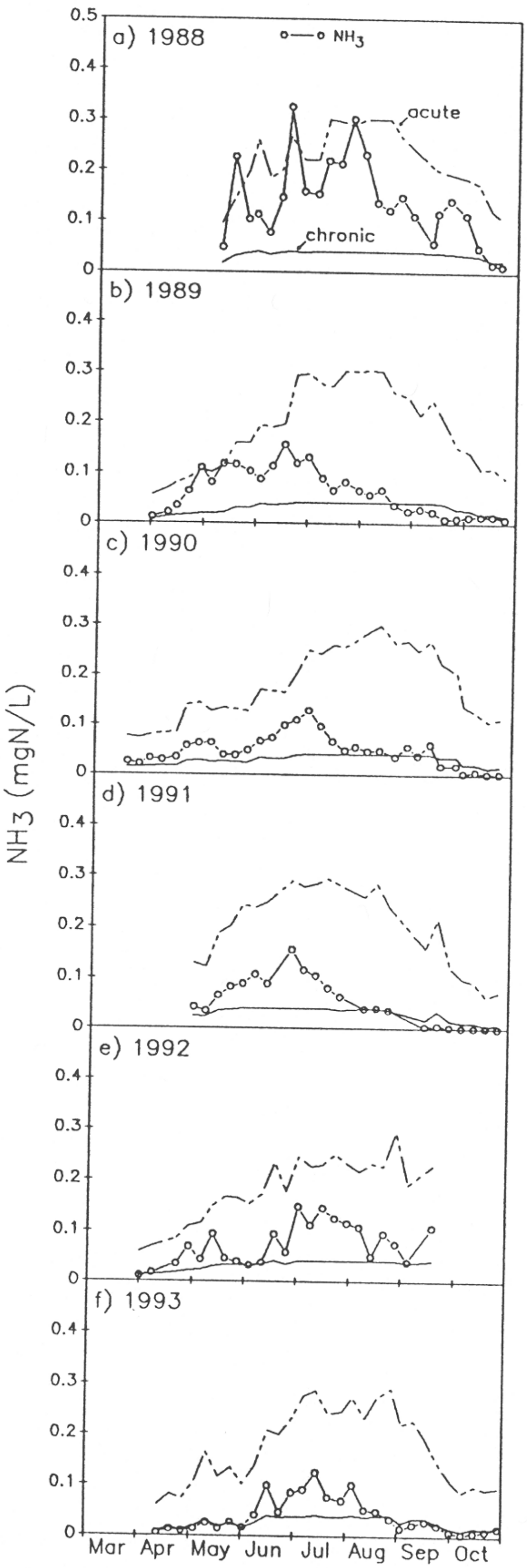

Figure 10.–Comparison of time series of NH_3 concentrations and chronic and acute standards for non-salmonids in the near surface (1 m) waters of Onondaga Lake: (a) 1988, (b) 1989, (c) 1990, (d) 1991, (e) 1992, and (f) 1993.

"dilution" model identified for some fluvial systems (Manczak and Florczyk 1974), where a relatively uniform loading source is diluted to a varying extent according to natural variations in runoff. This analysis indicates that year-to-year differ-ences in T-NH_3 (and therefore NH_3) concentrations in the lake are primarily a result of natural variations in runoff instead of differences in treatment performance at METRO. Year-to-year differences in phytoplankton growth (a sink for T-NH_3) may also contribute importantly to variations in T-NH_3 concentrations in the upper waters of the lake (Canale et al. 1996).

The prevalence and margin of violation of the chronic non-salmonid toxicity criterion for NH_3 and the more infrequent occurrence of violations of the acute standard in the upper waters of Onondaga Lake are consequences of the discharge of domestic waste water. The extent to which the prevailing high NH_3 concentrations impact the diversity, health, and abundance of aquatic life (particularly fish) in the lake is presently unknown. Delineation of the influence(s) of this pollutant is complicated by the myriad of other water quality problems of the lake (Effler 1996).

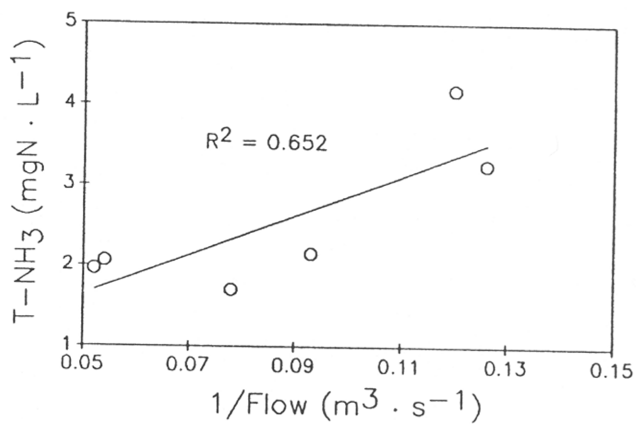

Figure 11.–Relationship between the inverse of the summed tributary inflow for the November-March interval and the subsequent April T-NH_3 concentration in Onondaga Lake, e.g., according to a dilution model.

NO_2^-

Nitrite is an intermediate product in the two stage biochemical process of nitrification. Nitrite is depleted rapidly from the hypolimnion of Onondaga Lake following the onset of anoxia associated with the operation of the denitrification process (e.g., Brooks and Effler 1990). In sharp contrast, generally progressive increases in NO_2^- concentrations are observed in the upper waters of the lake until August (Fig. 12a-f). Violations of the non-salmonid toxicity standard have been documented in each year over the 1989-1993

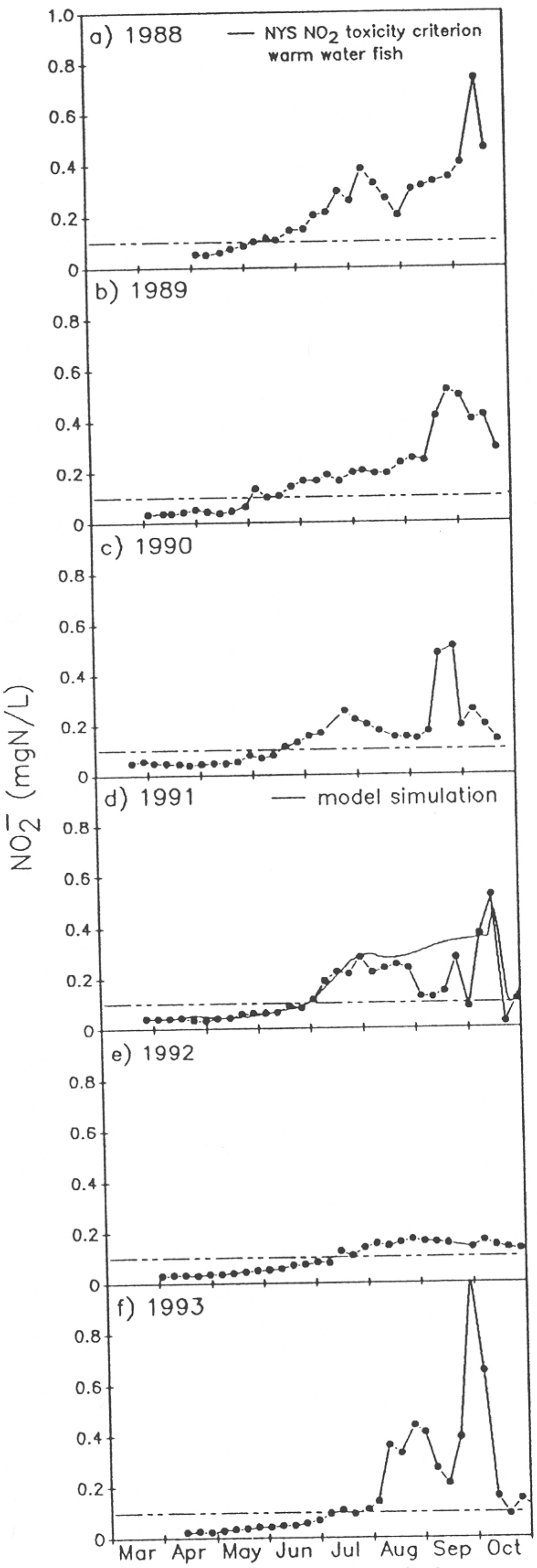

Figure 12.–Time series of NO_2^- concentrations in the near surface waters of Onondaga Lake: (a) 1988, (b) 1989, (c) 1990, (d) 1991, (e) 1992, and (f) 1993. Standard for non-salmonid fish, and mass balance model simulation for 1991, included.

interval (Fig. 12). Rather strong differences have been common after early August, with rather abrupt changes in NO_2^- concentration evident in most years (Fig. 12). Extremely high concentrations of NO_2^-, that represent severe violations of the standard, were observed over the August-October interval in 5 of the 6 years (Fig. 12). Clearly, severe violation of the NO_2^- toxicity standard has been a recurring problem in the upper waters of the lake. Hutchinson (1957) observed that the occurrence of "appreciable" NO_2^- concentrations in surface waters is indicative of sewage contamination. The toxic effects of NO_2^- are known to be mitigated by Cl^- (e.g., Lewis and Morris 1986), which is present in Onondaga Lake in unusually high concentrations (Doerr et al. 1994, Effler 1996).

A major part of the NO_2^- problem in the lake is the high external loads of NO_2^- received in the warmer months from METRO (e.g., Fig. 6). Application of a conservative substance mass balance model (e.g., loads and flushing accommodated, but no reactions) for NO_2^- has been successful in simulating the observed temporal structure through mid- to late summer in 1991 (Fig. 12d), supporting the important role METRO (95% of the NO_2^- load over the simulation interval of 1991) plays in this problem. It is noteworthy that the available evidence indicates the rate of nitrification in Onondaga Lake in summer is low (Canale et al. 1996), compared to values adopted in the modeling literature (e.g., Bowie et al. 1985). The conservative NO_2^- model fails in the late summer to fall interval (Fig. 12d). The operation of a major sink process in early September and a major source in early October of 1991 is indicated (Fig. 12d), which are not accommodated by the model. The non-conservative behavior of NO_2^- in fall (Fig. 12d) probably reflects temporal imbalances in the rates of the first and second stages of the overall nitrification process, as well as the abrupt increase in internal loading of T-NH_3 to the upper waters of the lake that accompanies the fall mixing period (Brooks and Effler 1990).

Management Perspectives

Clearly the extremely high loads of P and N species received by Onondaga Lake from METRO severely impact the lake, as manifested by the routine violations of several related water quality standards in the lake (Table 3). Degradations from cultural eutrophication, including excessively high concentrations of phytoplankton biomass, poor transparency, rapid depletion of oxygen from the hypolimnion, and lake-wide oxygen depletion during the fall mixing period, are conspicuous and recurring. Standards to protect

Table 3.–Violations/exceedances of numerical standards/guidance value for New York State in Onondaga Lake related to METRO P and N discharges.

Constituent/Attribute	Resource/Use	Standard/Guideline
free ammonia (NH_3)	fishing	toxicity; standard function of pH and temperature; differ for salmonid and non-salmonid fisheries
nitrite (NO_2^-)	fishing	toxicity; < 100 μg NO_2^-/L for non-salmonid, 20 < μgNO_2^-/L for salmonid
dissolved oxygen (DO)	fishing	≥ 5 mg/L, daily average; ≥ 4 mg/L, minimum within a day
clarity (Secchi disc)	swimming	standard for opening a public bathing beach; ≥ 4 ft (or 1.2 m)
total phosphorus (TP)	-	guidance value; epilimnetic summer average ≤ 20 μgP/L

against the toxic effects of NH_3 and NO_2^- on aquatic life are violated routinely and by a wide margin. Rarely have the signatures of N and P pollution been so clear (Table 3). METRO plays the primary role in all of these problems, and is the appropriate target for related remediation efforts.

At the heart of these water quality problems is the large volume of the METRO discharge to Onondaga Lake. This input of treated waste water represents nearly 30%, on average, of the total inflow received by the lake during summer, and is usually the largest single source of water in the month of August. Despite the major reductions in METRO effluent TP concentrations, and thereby loads and lake concentrations, achieved from increased treatment and other management actions, the lake remains highly eutrophic due to excessive external loading of P. The severity of the violations of the ammonia toxicity standard(s) reflects the lack of year-round nitrification treatment at METRO. While the seasonal nitrification achieved during the warmer summer months at the facility in recent years may have resulted in modest amelioration of the NH_3 problem in the lake, the attendant high NO_2^- loads from the plant contribute greatly to the violations of the NO_2^- toxicity standard in the lake.

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