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# A Limnological Analysis of Cannonsville Reservoir, NY[1]


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## ABSTRACT


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A limnological analysis of Cannonsville Reservoir, NY, is presented that focuses on features related to primary production. Monitoring data collected in 1995, a major drawdown year for the impoundment, and long-term data (since 1974), are evaluated. The reservoir demonstrates eutrophic characteristics in most summers, though upper mesotrophic conditions have been observed in some years. The concentration of chlorophyll is found to be the most reliable indicator of trophic state for the impoundment, as tripton (non-living particulate material) interferes with the measures of Secchi disc transparency and total phosphorus (P) concentration as indicators. Evidence is presented that the sediment resuspension process introduced tripton into the water column in 1995 as the reservoir was drawn down. Oxygen was depleted from the hypolimnion during the summer months of 1995; anoxia prevailed above the deep-water sediments for about 1 month. However, a major release of P from the sediments did not occur during this period. Evidence is presented that nitrogen became limiting to phytoplankton growth in mid-summer, and that a sink process(es) operates for the soluble reactive P released in the hypolimnion from the decomposition of organic material. Longitudinal gradients in trophic state indicators and other features of water quality prevail. Bounds for the riverine, transition, and lacustrine zones are presented; the lacustrine zone represents about 80% of the full reservoir volume.


Key Words: reservoir, impoundment, drawdown, monitoring, eutrophication, trophic state, sediment resuspension, longitudinal gradients, riverine/transition/lacustrine zones.

Reliable tools, in the form of rigorously tested mechanistic mathematical models, are widely desired by lake and reservoir managers to guide management decisions to improve or maintain water quality (e.g., Chapra 1997, Thomann and Mueller 1987). Such models represent a quantitative synthesis of the scientific understanding of processes regulating the concentration(s) of constituents of interest (e.g., nutrients, phytoplankton, oxygen, toxics, etc.). Often it is not recognized that these models can only be as good as the scientific information that supports them. The most reliable models for non-conservative substances are based on detailed limnological and system specific kinetic/process studies, and monitoring of critical forcing (e.g., meteorological, hydrologic, and material loading) conditions. Effective integration of these studies with a modeling program serves to enhance the yield of each and the credibility and utility of the resulting model(s) (see Chapra 1997, Thomann and Mueller 1987).

This issue of the journal documents the findings of integrated interdisciplinary studies that focus primarily on a single flow augmentation/water supply impoundment, Cannonsville Reservoir, New York (NY). Contributions in descriptive limnology, hydrology, hydrodynamics, kinetic and process studies, material loading analysis, and hydrothermal and water quality modeling are presented.

[1]Contribution No. 169 of the Upstate Freshwater Institute.

This manuscript characterizes selected features of the limnology of Cannonsville Reservoir, with particular emphasis on aspects of primary production and related characteristics. The reservoir, its setting, and monitoring programs are described. Historic data are reviewed that depict the system's trophic state and the apparent dependence of related indicator measurements and the character of its stratification regime on reservoir operation. Patterns in time and space are presented and analyzed for a year of particularly intensive monitoring. Important trends, phenomena, and processes are identified from the analysis that have served to guide the design of model frameworks and other studies incorporated in the overall program of study for this system, described in subsequent manuscripts in this issue.

## Cannonsville Reservoir

Cannonsville Reservoir (Latitude 42 02' 46", Longitude 75 22' 24", at dam) is located in Delaware County in upstate NY, approximately 190 km from New York City (NYC, Fig. 1). This impoundment is NYC's newest and third largest (of 19) water supply reservoir. Filling of the reservoir began September 30, 1963, after completion of an earth-filled, rock-faced dam downstream from the confluence of the West Branch of the Delaware River (WBDR) and Trout Creek, 2.9 km upstream of Stilesville, NY. The beds of these two streams divide the upper reaches of the impoundment into two arms. The upstream boundary of the WBDR

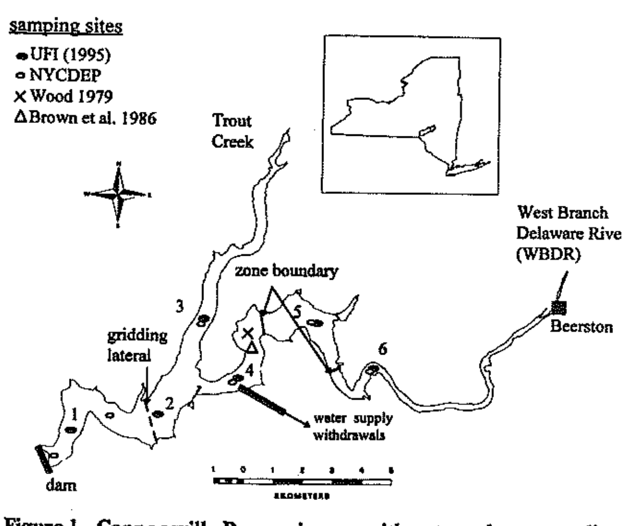


**Figure 1.–Cannonsville Reservoir map, with water column sampling locations for intensive 1995 program, sites monitored adjacent to water supply intakes by various programs, position of a lateral transect for three dimensional "gridding," and locations of reservoir within New York State, water supply intakes, dam, approximate bounds of reservoir (e.g., lacustrine) zones and Beerston, NY.**

arm of the reservoir is at Beerston, NY (Fig. 1). The reservoir is used primarily as a drinking water supply and to maintain flows in downstream portions of the Delaware River (mandated in a 1954 Supreme Court determination), though recreational fishing is also supported. Cannonsville Reservoir has a crest capacity of $373 \times 10^6 m^3$, a surface area of $19.3 \times 10^6 m^2$ (i.e., mean depth, when full, of ~19 m), and a maximum depth near the dam (Fig. 1) of ~49 m. The reservoir has a maximum length of 27.4 km and a shoreline length of about 74 km (Wood 1979).

A dimictic stratification regime prevails for Cannonsville Reservoir (Bader 1993, Brown et al. 1986, Wood 1979); e.g., isodensity conditions occur in spring following ice-out and again in fall (spring and fall turnover), strong thermal stratification develops in summer, and weak inverse stratification occurs under ice-cover in winter. The reservoir is a soft water system with limited buffering capacity. The concentration of $Ca^{2+}$ averages about 7.5 $mg \cdot L^{-1}$, and alkalinity averages about 16 $mg \cdot L^{-1}$ (e.g., NYCDEP 1997).

The annual average completely-mixed flushing rate of Cannonsville Reservoir over the 1969 - 1995 interval was 2.6 $y^{-1}$ (coefficient of variation (cv) = 18%; Owens et al. 1998b). The interannual variability is attributable primarily to natural, meteorologically-based variations in runoff from the 1,162 $km^2$ watershed, and related variations in reservoir operation (Owens et al. 1998b). Substantial interannual and seasonal variations in the water surface elevation (WSE) of Cannonsville Reservoir occur. A detailed treatment of the history of operation and the hydrology of the reservoir is presented by Owens et al. (1998b). Approximately one-third of the annual inflow is received in early spring, associated with snowmelt (Owens et al. 1998b). The WBDR contributes about 80% of the inflow to the reservoir, Trout Creek ~ 5%; the remainder is associated with smaller tributaries and direct inflow (Owens et al. 1998b). Water exits the reservoir in one of three ways: 1) over the dam (spillway) when the reservoir is full (e.g., in spring), 2) via one of three withdrawals (intakes for water supply at depths of 10, 20 and 37 m below the spillway elevation; Fig. 1), and 3) downstream releases (flow augmentation and stream conservation, ~ 35 m below spillway adjacent to dam, Fig. 1). The maximum withdrawal rate for the water supply is $3.0 \times 10^6 m^3 \cdot d^{-1}$, the maximum release rate is ~ $5.7 \times 10^6 m^3 \cdot d^{-1}$. Flow augmentation requirements for the lower Delaware River can be met by releases from Cannonsville Reservoir or from nearby Pepacton and Neversink Reservoirs (two other NYC impoundments).

The watershed, located in the northwestern section of the Catskill Mountains, has variable topography, with elevations that range from 350 m (at the dam) to 1010 m above sea level; its area is 1160 $km^2$ (WBDR sub-basin is 78% of total). It drains in a southwestern direction into the reservoir. The annual average precipitation within the watershed is ~ 100 cm · $yr^{-1}$ [New York City Department of Environmental Protection (NYCDEP) records]. The underlying bedrock is made up of consolidated sandstone, siltstone and shales, covered by gravel, sand, unconsolidated fill and clay (Soren 1963). Most of the agriculture is located along the watershed's stream corridors, where the soils have superior drainage capability. The vegetative cover in the watershed is about 70% forest, 24% grass, and 3% corn or alfalfa. Almost all of the agricultural land is associated with dairy farming. There are about 210 farms, with a total of approximately 20,000 dairy cows (Longabucco and Brown 1990). Approximately 18,000 people resided in the Cannonsville watershed in 1980; 8,400 were served by 5 municipal wastewater treatment facilities (WWTP) (Brown et al. 1986). Two other small WWTPs operate in the watershed to treat waste from agriculture products. The largest WWTP is located in Walton, 7 km upstream of the mouth of WBDR (Fig. 1). NYC owns the land immediately adjacent to the reservoir. Thus the shoreline is protected from development and direct nutrient input from septic systems. Subsequently in this issue, Longabucco and Raferty (1998) review several aspects of external material loading to Cannonsville Reservoir, with particular emphasis on phosphorus.

# Monitoring Programs for the Reservoir

The NYCDEP conducts an extensive monitoring program on all 19 of the reservoirs in NYC's water supply system. All outflows and WSE are monitored continuously for Cannonsville Reservoir by NYCDEP (data record reviewed by Owens et al. 1998). Inflow from WBDR (8 km upstream of Beerston, Fig. 1) and flow downstream (2.25 km) of the reservoir are monitored continuously by the United States Geologic Survey. NYCDEP monitors the water quality of WBDR and Trout Creek, outflows from the reservoir, and the water column of the reservoir. The in-reservoir program includes field measurements and chemical and biological laboratory analyses (Table 1) conducted according to standard methods. Instrument profiles are collected at 1-m-depth intervals (NYCDEP 1995). Samples for laboratory analyses are collected at three or four depths for this program, extending from the upper layers to the near bottom, at six longitudinal stations [five along the main axis of the reservoir, one in the Trout Creek branch (Fig. 1)]. Sampling frequency was about monthly over the 1988-1993 interval, but increased to bi-weekly in 1994 (some differences, according to analyte). Temporary additions are made to the routine long-term monitoring program on a "needs" basis, to respond to pertinent short-term (e.g., events) concerns. Findings of this long-term monitoring program are published regularly by NYCDEP (e.g., NYCDEP 1992, 1993).

A more intensive in-reservoir monitoring program was executed in 1995, with the intention of enhancing the resolution of limnological processes and to support the development and testing of mechanistic hydrothermal (see Gelda et al. 1998, Owens 1998b) and nutrient-phytoplankton (see Doerr et al. 1998) models for Cannonsville Reservoir. Salient results of the 1995 program are a primary focus of this paper. Additionally, several process studies were conducted in 1995 to support model development, including: 1) identification and quantification of hydrodynamic processes (Owens 1998a), 2) runoff event-based material loading estimates (Longabucco and Rafferty 1998), 3) optical characterizations (Effler et al. 1998b), 4) quantification of deposition of particulate constituents (Effler and Brooks 1998), 5) characterization of the plankton community, 6) quantification of phytoplankton kinetics (Auer 1998), and 7) determination

**Table 1–NYCDEP monitoring program for Cannonsville Reservoir; analytes.**

| Field Measurements | Chemical/Physical Analysis |
|---|---|
| temperature | turbidity |
| pH | color |
| dissolved oxygen | alkalinity |
| conductivity | chloride |
| Secchi disc depth | ammonia |
| scalar irradiance | nitrate plus nitrite |
| | total nitrogen |
| | soluble reactive phosphorus |
| **Biological Analyses** | total dissolved phosphorus |
| fecal coliform | total phosphorus |
| heterotrophic plate count | dissolved organic carbon |
| fecal strep | total organic carbon |
| phytoplankton | suspended solids |
| zooplankton | volatile suspended solids |
| | major metals - Ca, Mg, K, Mn, Ni, Al, Fe |
| | trace metals - Cu, Zn, As, Cr, Hg, Cd, Pb, Se, Ag, Be |
| | silica |
| | sulfate |
| | chlorophyll *a* |

of sediment-water chemical exchange rates (Erickson and Auer 1998).

The intensive 1995 monitoring program focused primarily on nutrient-phytoplankton issues (Table 2). The six longitudinal sites (designated as UFI, Fig. 1) adopted were positioned at the locations used for the long-term program. Additionally, a "gridding" of field measurements of temperature, transmissometry [as an indicator of turbidity ($T_n$); e.g., Kitchen et al. 1982], and fluorescence [as a indicator of chlorophyll (Chl); e.g., Cullen 1982] was conducted with a Seabird Sealogger Profiler (Model SBE 25) on six occasions in early fall of 1994 and the summer of 1995. Readings with the profiler were recorded at a rate of 8 $s^{-1}$, using an instrument descent rate of about 1.2 $m \cdot s^{-1}$, which produces highly resolved (e.g., nearly continuous) vertical profiles. The gridding included 45 sites, and was intended to identify and characterize a significant three-dimensional structure in the distribution of phytoplankton biomass and $T_n$ in the reservoir. Additionally, the Seabird Sealogger was used to collect profiles of these measures, plus scalar irradiance, and specific conductance, at each of the six longitudinal sites weekly, over the April-October interval. Dissolved oxygen and pH profiles (1- to 2-m-depth resolution; Table 2) were also collected weekly, at site 4 (Fig. 1), at ~ 1000 h. Samples for laboratory analyses (Table 2) were collected at 3-m-depth intervals from the surface to the bottom at sites 1, 4 and 5, and at depths of 0, 3 m, and 1 m above the bottom, at sites 2, 3 and 6 (Fig. 1). Other monitoring programs for the reservoir, from which data are used in the subsequent historic analysis, were described by Wood (1979) and Brown et al. (1986).

# Review of Historic Data

Limnological studies of Cannonsville Reservoir have focused on the impoundment's trophic state (e.g., Brown et al. 1986, USEPA 1974, Wood 1979). The most widely used indicators of trophic state are the concentrations of total chlorophyll (Chl) and total phosphorus (TP) in the epilimnion and Secchi disc (SD) transparency (Carlson 1977). High levels of primary productivity (i.e., eutrophy) are commonly manifested as high concentrations of Chl and TP and low SD. Chlorophyll concentration should be considered the primary indicator of these three

**Table 2.–Analytes and methods for intensive 1995 monitoring program at Cannonsville Reservoir**

| Parameters | Description |
|---|---|
| **Field Measurements** | |
| Seabird Sealogger Profiler | |
| temperature | Sea-Bird Inc. |
| specific conductance | Sea-Bird Inc. |
| scaler irradiance | Li-Cor Inc. |
| transmissometry | Sea Tech Inc. |
| fluorescence | Sea Tech Inc. |
| Hydrolab Surveyor 3 | |
| dissolved oxygen | |
| pH | |
| Secchi disc depth | 12-cm-diameter, black and white quadrant |
| **Laboratory Measurements** | |
| chlorophyll (total, Chl) | Parsons et al. 1984 |
| phosphorus | APHA (1992) |
| soluble reactive (SRP) | method 4500 P, E |
| total (TP) | method 4500 P, E |
| total dissolved (TDP) | method 4500 P, B, E |
| nitrogen | USEPA 1983 |
| ammonia ($T\text{-}NH_3$) | method 350.1 |
| nitrate plus nitrite ($NO_x$) | method 353.2 |
| total suspended solids (TSS) | APHA 1992, method 2540D |
| turbidity ($T_n$) | APHA 1992, method 2130B |

parameters, as high concentrations of tripton (inanimate particles) are known to compromise the value of TP and SD as trophic state indicators (e.g., Carlson 1977). The summer average concentration of Chl that demarcates mesotrophy and eutrophy ranges from 8 to 12 $\mu g \cdot L^{-1}$, depending on the literature citation adopted (e.g., Dobson et al. 1974, National Academy of Science 1972, Great Lakes Group 1976). Several TP concentrations have been proposed for the mesotrophy/eutrophy boundary (e.g., Auer et al. 1986, Chapra and Dobson 1981, Vollenweider 1975, Vollenweider 1982). Probably the most often cited is a TP concentration equal to 20 $\mu g \cdot L^{-1}$ (Vollenweider 1975), which is also the guidance value (mean summer epilimnion concentration) established by the New York State Department of Environmental Conservation for lakes and reservoirs in NY.

Time series of summer average (May-September) values of Chl and TP concentrations and SD observed in Cannonsville Reservoir over the period of record, at sites adjacent to the water supply intakes (Fig. 1), are presented here (Fig. 2a-c), along with the corresponding average WSE value (Fig. 2d). Secchi disc and Chl data are available for 14 of 31 years of operation; TP data are available for 12 years. Data are not available for the first 10 years of operation of the reservoir. Data from four different programs contribute to the time series (Fig. 2). Differences in the locations of the monitoring sites (Fig. 1) are not believed to substantively influence the observed long-term distributions. These trophic state indicators were tracked in the reservoir by two different programs in each of the years in the 1992-1995 interval. The results reported by the different programs were quite similar throughout the interval (Fig. 2). Temporal coverage within years has varied greatly over the period of record. For example, there were only 3 surveys in 1974 and 5 in 1975 (Wood 1979), 10 or 11 over the interval 1980-1982 (Brown et al. 1986), 6 to 7 by NYCDEP over the 1988-1991 interval, 20 surveys (multiple programs) for the 1992-1995 period, and 13 in 1996 by NYCDEP.

Cannonsville Reservoir is considered to be the most eutrophic of NYC's reservoirs (e.g., NYCDEP 1992). The long-term average concentrations of Chl and TP have been 14.5 and 23.6 $\mu g \cdot L^{-1}$, respectively; the average SD has been about 2.1 m. These conditions are consistent with eutrophy. However, upper mesotrophy has been indicated in several years (Fig. 2). For example, the summer average Chl was slightly less than 12 $\mu g \cdot L^{-1}$ in 5 years, and TP was < 20 $\mu g \cdot L^{-1}$ in 4 years. Relative variability within individual years has been the greatest (of the three indicators) for Chl (Fig. 2); e.g., the average coefficient of variation (cv) for Chl for the period of record was 0.49, compared to 0.29 and 0.31 for SD and TP, respectively. No long-term trends are apparent for these three indicator parameters, but substantial interannual variations in the summer average values were observed; e.g., cv = 0.30, 0.23, and 0.22 for Chl, SD, and TP, respectively.

Concentrations of Chl and TP tended to be higher and SD lower in the years of greater drawdown (i.e., lower summer average WSE; Fig. 2a-d and Fig. 3). For example, the best water quality conditions observed

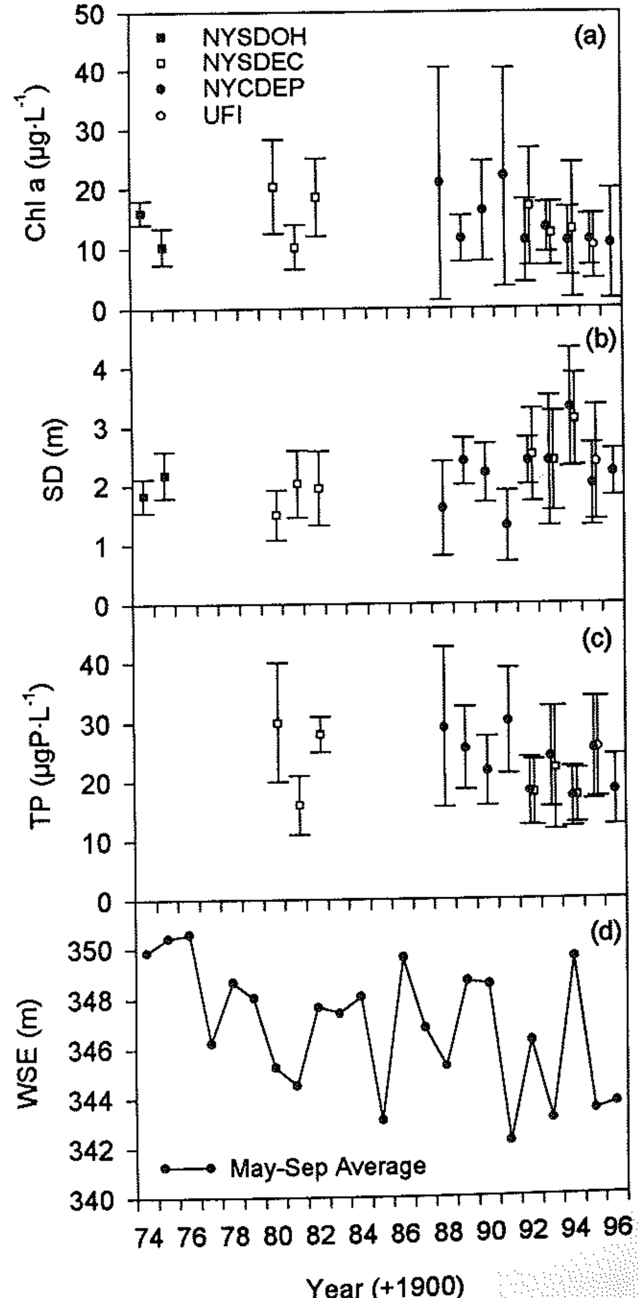


Figure 2.–Time series of summer (May-September) arithmetic mean values for Cannonsville Reservoir for the 1974 - 1996 period: a) Chl concentration, b) SD, c) TP concentration, and d) water surface elevation (WSE). Dimensions of bars correspond to ± 1 standard deviation.

over the 1988-1996 interval were in 1994, a year in which the reservoir remained nearly full (Fig. 2d), the worst in 1991, the year of greatest drawdown for the interval (see Fig. 3). Differences in summer average WSE explained 30%, 41%, and 50% of the interannual variations in average Chl, SD, and TP values, for the interval, respectively, based on linear least squares regression (Fig. 3). This empirical analysis suggests there is a water quality "cost" for the operation of the reservoir for its intended use, when runoff from the watershed does not compensate for the required releases and withdrawals. Evidence is presented subsequently in this article and elsewhere in this issue (Effler et al. 1998a, b) that TP and SD are compromised as trophic state indicators in Cannonsville Reservoir during periods of major drawdown as a result of the influx of resuspended sediment (i.e., tripton) into the water column.

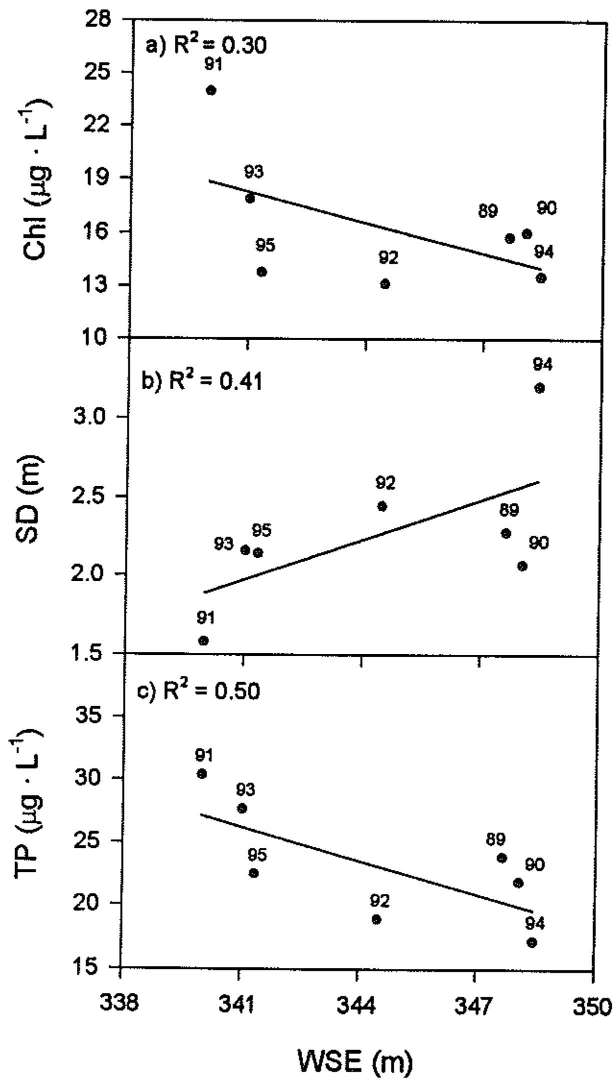


**Figure 3.–Evaluation of the relationships between summer average measures of trophic state and WSE for Cannonsville Reservoir: a) TP, b) Chl, and c) SD.**

Year-to-year variations in the thermal stratification regime of the reservoir result not only from natural variability in meteorology (e.g., Effler et al. 1986), but also in response in interannual differences in reservoir operation (also see Gelda et al. 1998, Owens 1998b). For example, the duration of stratification, DSTRAT [defined here as the interval (days) during which the vertical density difference from the surface to the bottom exceeded 0.1 kg · $m^{-3}$], has been shorter for years of greater drawdown (Fig. 4). The value of DSTRAT differed by as much as 40 days in the years of the 1989-1995 interval. The following relationship, developed from linear least squares regression, explained 82% of the variance observed in DSTRAT over the 1989-1995 interval (Fig. 4).

$$\text{DSTRAT} = 2.77\ \text{WSE} - 765.5 \qquad (1)$$

At least two factors contributed to the shortening of DSTRAT in years of greater drawdown: 1) reductions in the depth of the water column, and 2) selective withdrawal of water from the lower stratified layers (Owens 1998a, b). Other features of the reservoir's stratification/mixing regime are impacted by operations (see Owens 1998b, Gelda et al. 1998). These relationships are important to identify and quantify, as various measures of water quality are known to be influenced by features of the stratification/mixing regime (e.g, Effler and Owens 1996, Stauffer and Lee 1973, Stefan et al. 1976).

# Selected Limnological Characteristics, 1995

## *Temporal Patterns*

Several salient, and in most cases recurring, seasonal features of the limnology of Cannonsville Reservoir are represented here through time series of measurements made over the April-October interval of 1995 (Fig. 5) at site 4, adjacent to the NYC water supply intakes (Fig. 1). A major drawdown was experienced in the reservoir in 1995. The WSE decreased (from full, ~351 m above sea level) about 20 m from April to early October (Fig. 5a), to about 23% of its crest capacity (see hypsographic data for the reservoir presented by Owens et al. 1998). The WSE dropped most rapidly from late June to early July and from mid-August to mid-September (Fig. 5a). In strong contrast, the reservoir remained much more nearly full through the spring-fall interval of 1994 (Fig. 5a). Interannual and seasonal variations in WSE over the period of operation of the reservoir are reviewed by Owens et al. (1998b).

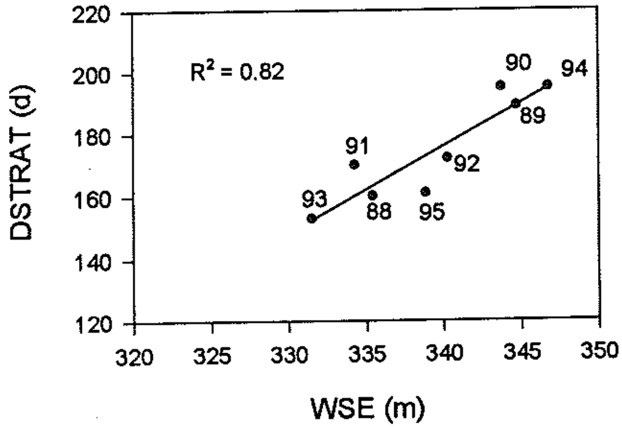


**Figure 4.–Evaluation of the relationship between duration of stratification (DSTRAT) and summer average WSE for Cannonsville Reservoir.**

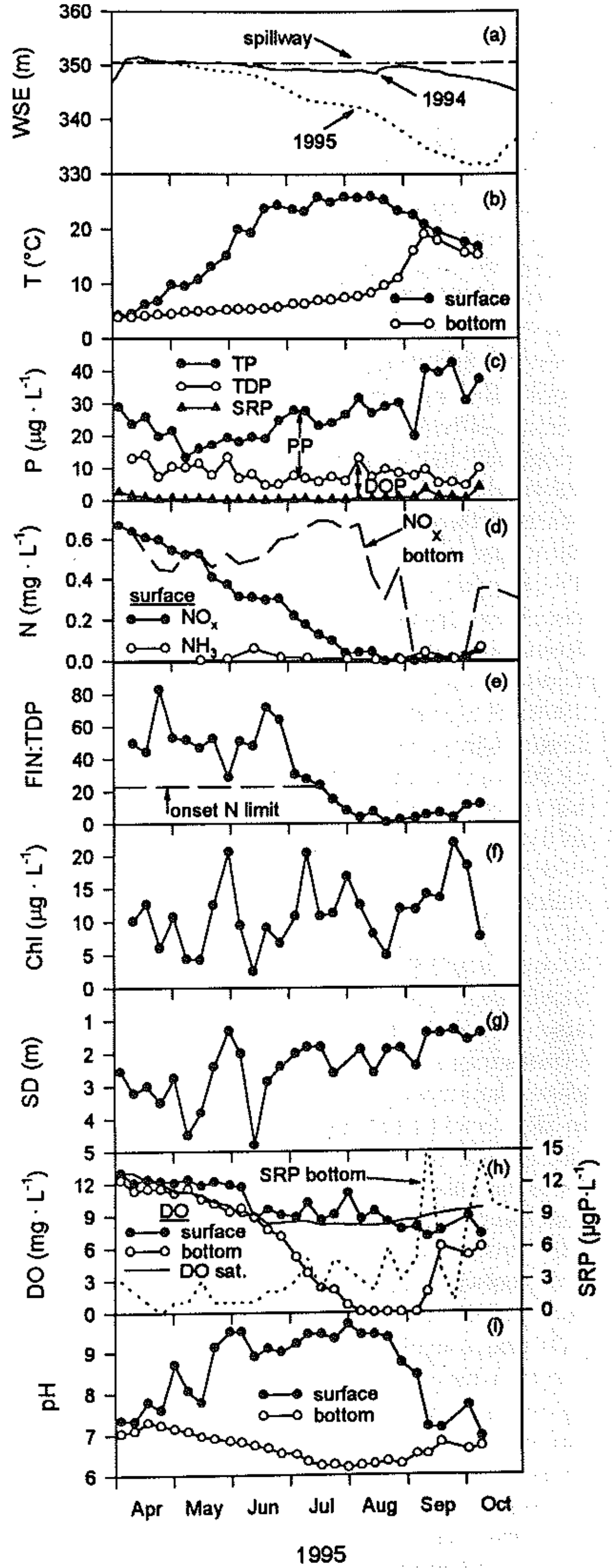


**Figure 5.–Time series for Cannonsville Reservoir of site 4, for the April - October interval of 1995: a) WSE (including 1994), b) T, c) P species in the upper waters (0-6 m average), d) N species, e) ratio FIN:TDP in the upper waters, f) Chl in the upper waters, g) SD, h) DO (including saturation values for the upper waters) and SRP in the lower waters, and i) pH.**

Thermal stratification is an ubiquitous characteristic for lakes and reservoirs of the depth of Cannonsville Reservoir in temperate climates (e.g., Hutchinson 1957, Wetzel 1983). Spring turnover continued in the reservoir in 1995 until mid-April (Fig. 5b). The surface waters warmed rapidly through mid-June with further, but slower and more erratic increases in temperature into early August (Fig. 5b). The maximum vertical temperature difference (~20°C) occurred in mid-July in 1995 (Fig. 5b). The upper mixed layer (or epilimnion; depth interval of approximately uniform temperature, extending from the surface) was generally about 6 m thick during the summer of 1995. The hypolimnion warmed progressively through summer, but at a much lower rate than the epilmnion (Fig. 5b). This modest heating of the hypolimnion is attributable to the limited vertical mixing that occurs between the epilimnion and hypolimnion (e.g., Stauffer and Lee 1973, Effler and Field 1983) that was enhanced in 1995 by the drawdown of the reservoir (Owens 1998b). The rapid cooling of the upper waters and heating of the lower layers that started in late August (Fig. 5b) reflect increased mixing and the loss of some heat from the system to the atmosphere. By mid-September the reservoir was nearly completely turned over; the upper mixed layer extended to within 2 to 3 m of the bottom. The minor thermal stratification that persisted through the end of the 1995 study period is a manifestation of the operation of the density underflow process [associated with plunging of a tributary (e.g., Effler and Owens 1996, Thornton et al. 1990)] that was doubtless eliminated when the temperatures of WBDR and the reservoir became more nearly equal later in fall. Owens (1998a) presents a detailed account of the operation of the underflow phenomenon in the reservoir.

Phosphorus has long been recognized as the most critical nutrient controlling phytoplankton growth in most fresh waters (e.g., Carlson 1977, Dillon 1975, Hutchinson 1973, Vollenweider 1968). The concentration of TP in the upper waters (0 to 6 m average) proximate to the water supply intakes before the onset of stratification in 1995 was about 25 to 30 $\mu g \cdot L^{-1}$. Concentrations decreased through early May, but thereafter increased progressively through September (~ 40 $\mu g \cdot L^{-1}$; Fig. 5c). The average concentration for the May-September interval was 25.6 $\mu g \cdot L^{-1}$. The increases in TP after May were in the particulate P (PP) pool, as concentrations of total dissolved P (TDP) remained (by comparison) relatively uniform, in the range of 5 to 10 $\mu g \cdot L^{-1}$ (Fig. 5c). The vast majority of PP in the epilimnion of most eutrophic lakes and reservoirs is incorporated in algal biomass. However, in the upper waters of Cannonsville Reservoir in 1995, the PP (Fig. 5c) and Chl (Fig. 5f) distributions were uncorrelated (particularly after May). The increase in PP from mid-to-late summer relative to Chl reflects contributions to the PP pool from tripton (Effler et al. 1998a). Other data and analyses presented subsequently in this manuscript and elsewhere in this issue (Effler et al. 1998a, Doerr et al. 1998) support this position. Soluble reactive (SRP) concentrations remained very low ($\leq 3\ \mu g \cdot L^{-1}$, usually $\leq 1\ \mu g \cdot L^{-1}$) in the productive layers of the reservoir from spring through early fall (Fig. 5c), reflecting the efficient uptake of these available forms of P by the phytoplankton. Minor enrichment may have occurred (concentrations close to detection limits) in August and September, perhaps associated with the approach to fall turnover (Fig. 5b i.e., entrainment of hypolimnetic layers), the shift to N limitation of phytoplankton growth (Fig. 5e), and/or the increases in SRP concentrations in WBDR over the same period (Longabucco and Rafferty 1998). Most of the dissolved P was dissolved organic P (DOP, calculated here as the residual of TDP and SRP, = TDP-SRP, though some DOP probably is included in the SRP analysis; Fig. 5c). This pool was relatively stable in 1995; modest depletions occurred in June-July and September. A fraction of the DOP pool is available to support phytoplankton growth, as certain forms of DOP can be utilized directly by algae (Bentzen et al. 1992, Cotner and Wetzel 1992) and others can be made available through enzymatic activity. The most well known enzymes active in this conversion are the alkaline phosphatases (e.g., Currie et al. 1986, Gage and Gorham 1985, Healey and Hendzel 1979). Alkaline phosphatase activity (APA) data collected during this study indicates DOP was utilized to support phytoplankton growth in the reservoir in 1995.

Ammonium ($NH_4^+$) and $NO_3^-$ ions are the principal forms of N used to support phytoplankton growth (Harris 1986, Wetzel 1983). The sum of the concentration of these N species is referred to as fixed inorganic N (FIN). Ammonium is preferred over $NO_3^-$ for energetic reasons, though concentrations of $NH_4^+$ are so low in many systems that $NO_3^-$ is the principal source of N used to support primary production (Wetzel 1983). This was the case for Cannonsville Reservoir in 1995 as concentrations of total ammonia (T-$NH_3$; $NH_4^+$ plus unionized ammonia) in the productive layers were usually < 20 $\mu g \cdot L^{-1}$ (e.g., Fig. 5d), except for the single observation in early June. We attribute this pulse to zooplankton excretion associated with a "clearing event" (abrupt reduction in phytoplankton biomass from grazing; e.g., Lampert et al. 1986). A very strong signature of phytoplankton uptake is imparted to the seasonal distribution of $NO_x$ ($NO_3^-$ + $NO_2^-$) in the reservoir, manifested as the progressive depletion of $NO_x$ in the upper productive layer (Fig. 5d). The $NO_x$ concentration was depleted from about 0.7 mgN $\cdot L^{-1}$ in early April to essentially undetectable levels by late August of 1995 (Fig. 5d). Starting in July of 1995, the concentrations of FIN were in the range typically associated with N limitation of non-N-fixing phytoplankton (e.g., Bowie et al. 1985). There is ample evidence (e.g., see Hardy et al. 1973) that the process of fixation of atmosphere N supported phytoplankton growth in the reservoir in late summer and fall. For example, nitrogen-fixing filamentous cyanobacteria (previously known as blue-green algae) were common in early July and were the dominants of the phytoplankton assemblage from mid-July through mid-August (unpubl. data, Siegfried 1998). Heterocysts, which function in the N-fixation process for these cyanobacteria, where observed in large numbers (e.g., see Horne and Goldman 1972) during the period of dominance of these cyanobacteria (unpubl. data, Siegfried 1998). Floating mats and scums of the cynanobacteria, widely described as a nuisance condition by lake managers (Cooke et al. 1993), were common during this period. Clearly N fixation was effective in sustaining the growth of this component of the phytoplankton in this reservoir, as rather high Chl concentrations (Fig. 5f) were observed over the period of low FIN concentrations (Fig. 5d).

The nutrient present in the shortest supply relative to the needs of the phytoplankton is described as the "limiting" nutrient; its availability usually is the most important factor controlling growth during the productive spring-fall interval (Hutchinson 1973). Phosphorus is the limiting nutrient in the vast majority of lakes and reservoirs (Harris 1986, Wetzel 1983). Ratio values of FIN:TDP within the upper productive layers have been used to delineate ambient conditions of P versus N limitation; P is considered limiting when the atomic N:P ratio exceeds 10-30:1 (10-20:1, Goldman et al. 1979; 16:1, Redfield 1958; 30:1, Rhee 1978). This corresponds to weight-based ratio limits of 23-69:1. The time course of the FIN:TDP ratio in 1995 indicates a shift from P to N limitation by at least the month of July (Fig. 5e). These conditions prevailed through at least September (Fig. 5e). This timing for a shift to N limitation is also supported by the observed emergence of N-fixing cyanobacteria as dominant during the same period (unpubl. data, Siegfried 1998; also see Hardy et al. 1973, Fogg 1974). The N limitation phenomenon is important in Cannonsville Reservoir because it promotes the development of the nuisance conditions that accompany the dominance of N-fixing cyanobacteria.

The concentration of Chl, the most widely used measure of phytoplankton standing crop, was highly dynamic in the upper waters of Cannonsville Reservoir in 1995. Peak concentrations of >20 $\mu g \cdot L^{-1}$ were observed in late May, early July, and late September (Fig. 5f). The first peak was a diatom bloom, the second and third were composed primarily of cyanobacteria and secondarily of diatoms (unpubl. data, Siegfried 1998). The average Chl concentration for the upper waters, proximate to the intakes (site 4) over the April-September interval was about 10.5 $\mu g \cdot L^{-1}$, indicative of mesotrophic to eutrophic conditions (Carlson 1977, Dobson et al. 1974, Great Lakes Group 1976, National Academy of Science 1972). Concentrations of Chl were <5 $\mu g \cdot L^{-1}$ on four (of 27) occasions (Fig. 5f). Chlorophyll *a* and Chl were highly correlated over the study period (R = 0.99), and chlorophyll *a* represented 92% of Chl, on average. The early July bloom was in part manifested as an increase in the PP pool (Fig. 5c). The time series of $NO_x$ concentrations (Fig. 5d) was a better indicator of phytoplankton growth than those for P species (Fig. 5c) in the first half of the study; e.g., the most rapid depletions of $NO_x$ were observed during the late May and early July blooms.

Clarity, as measured by SD, was highly variable in the reservoir from April through June of 1995, ranging from about 1 to almost 5 m; thereafter clarity was low and more invariant (1.2 to 2.7 m; Fig. 5g). The Chl concentration appears to have regulated light penetration until rapid drawdown was initiated in late June (Fig. 5a); variations in Chl explained nearly 80% of the variance in clarity (as 1/SD) observed at site 4 (Fig. 1) over the April to late-June interval (Fig. 6a). Thereafter clarity was largely insensitive to the substantial changes in phytoplankton biomass observed (Fig. 6a), supporting the position that tripton particles were important in regulating clarity in the reservoir during the period of major drawdown. The stronger relationship between SD and $T_n$ (Fig. 6b), compared to that observed for SD and Chl (Fig. 6a), further supports the position that tripton is important in regulating clarity in the reservoir during drawdown periods. A more detailed characterization of the optics of the reservoir, including an analysis of regulating processes and materials, and the role of tripton, is presented in this issue by Effler et al. (1998b). Effler et al. (1998a) provide evidence that the increase in tripton turbidity in Cannonsville Reservoir during the period of rapid drawdown is a result of resuspension of deposited sediment.

Dissolved oxygen (DO) concentrations in the upper well-mixed waters tend to track concentrations that are in equilibrium with the air (i.e., saturation concentrations; Fig. 5h), mediated by exchange at the air-water interface (e.g., Gelda and Auer 1996). Thus the reductions in DO concentrations in the surface waters from spring to summer are largely attributable to the increase in the temperature of the upper waters. Modest deviations from equilibrium concentrations

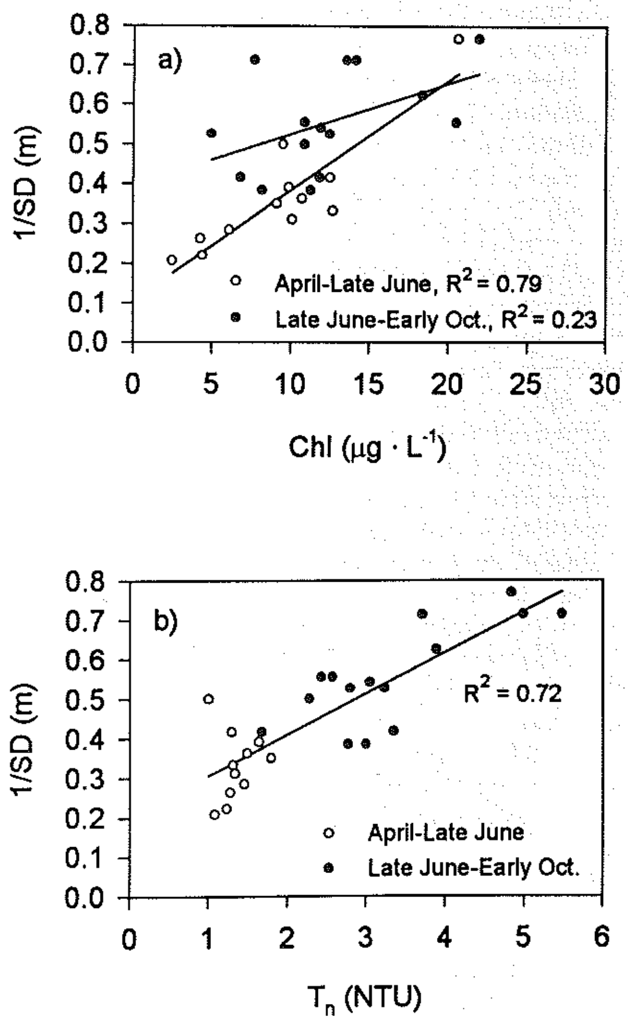


**Figure 6.–Evaluation of components regulating clarity (SD) in Cannonsville Reservoir in 1995: a) Chl, and b) $T_n$.**

were observed in 1995. The positive inflections and periods of oversaturation (Fig. 5h) reflect the effect of photosynthesis (e.g., Fig. 5f); the decreases and undersaturation preceding the onset of complete fall turnover (Fig. 5h) were a result of mixing with the depleted lower layers (e.g., Fig. 5b). The observed progressive depletion of DO from the lower layers of the reservoir (Fig. 5h) is a widely occurring phenomenon in stratifying lakes and reservoirs during summer stratification. This depletion reflects the demand for DO associated with oxidation processes exceeding the limited sources of DO. Severe depletions (e.g., $< 4\ mg \cdot L^{-1}$) can eliminate the potential to support a cold water fishery. The occurrence of anoxia (DO ~ $0.0\ mg \cdot L^{-1}$) and subsequent anaerobic conditions are a water quality concern because of the potential for mobilization of phosphorus and release of reduced substances from the underlying sediments (e.g., Mortimer 1941, 1971). Oxygen demand in the hypolimion is usually localized at the sediment-water interface (represented as sediment oxygen demand; see Erickson and Auer 1998) and thus DO concentrations are lowest, and anoxia is observed first, just above the sediments. Oxygen was lost rapidly from the bottom waters of Cannonsville Reservoir from June until the onset of anoxia in early August (Fig. 5h). The lowermost waters were anoxic at this site for about a month. The oxygen resources of the lowermost layer were recovered rapidly in September by mixing with overlying oxygenated waters that accompanied the approach to fall turnover (Fig. 5b and h).

It is valuable to consider the dynamics of SRP in the lowermost layers of the reservoir within the context of the observed oxygen depletion (Fig. 5h; e.g., Gächter and Mares 1985). The mineralization (decay) processes responsible for the observed oxygen depletion are accompanied by the release of SRP (e.g., cellular P). The extent to which released SRP accumulates can be evaluated by comparing the rate of increase in SRP ($\Delta SRP/\Delta t$) and the rate of DO depletion ($\Delta DO/\Delta t$). According to Redfield ratio(s) stoichiometry for phytoplankton (Redfield et al. 1963, Imboden and Gächter 1978), the ratio of these rates ($\Delta SRP/\Delta t{:}\Delta DO/\Delta t$) should be about $-0.007\ mgP \cdot mg^{-1}O_2$ (Gächter and Mares 1985), if the released SRP has not been taken up by other particles. This calculation was made for Cannonsville Reservoir (at site 4) over the major DO depletion interval of early June to mid-July of 1995 (Fig. 5h). The increase in SRP over this interval was small, the ratio of the rates was estimated to be about $-0.0005\ mgP \cdot mg^{-1}O_2$, indicating the operation of a loss process(es) (e.g., conversion to a particulate form) for SRP released in the lower layers of the reservoir. Gächter and Mares (1985) reported similar evidence for the operation of a sink process(es) for SRP produced from mineralization of settling organic particles for several Swiss lakes. Potential regulating mechanisms include uptake of SRP into bacterial biomass and adsorption to inorganic (e.g., iron-rich, clay) particles (Gächter and Mares 1985). The surficial sediments of Cannonsville Reservoir are enriched with clay and iron-rich particles, and were resuspended into the water column of the impoundment in 1995 (Effler et al. 1998a). Such particles are known to have a high adsorptive capacity for phosphorus (Böstrom et al. 1988, Mortimer 1971, Wetzel 1983). The lack of a major increase in SRP in the lowermost waters of the reservoir during anoxia indicates no significant sediment release of phosphorus occurred in that interval. This is not inconsistent with the Mortimer (1941, 1971) model for the regulation of sediment phosphorus release by iron activity, as other electron acceptors such as $Mn^{4+}$ and $NO_3^-$ are reduced following the onset of anoxia before $Fe^{3+}$, based on thermodynamic considerations (Froelich et al. 1979, Kelly et al. 1988). The pool of $NO_x$ in the bottom waters of Cannonsville Reservoir in 1995 was partially depleted, but not eliminated during the anoxic interval (Fig. 5d), thus iron-based release of phosphorus should not have been expected (e.g., Mortimer 1971, Wetzel 1983). Increases in hypolimnetic $NO_x$ before the onset of anoxia (Fig. 5d) reflect the nitrification of T-$NH_3$ released from the sediments (see Erickson and Auer 1998).

The temporal and vertical distributions of pH in productive systems are primarily mediated through the dynamics of photosynthetic consumption and respiratory/decomposition-based production of $CO_2$ (Wetzel 1983). Increases in pH are widely observed in the upper waters of productive lakes in late spring and summer, as a result of photosynthetic uptake exceeding the inputs of $CO_2$ (e.g., Effler 1984, Effler and Driscoll 1985, Effler et al. 1982, Kelts and Hsu 1978). Whereas, decreases in pH are observed in hypolimnia because of the dominance of respiration/decomposition processes (e.g., Effler et al. 1982, Kelts and Hsu 1978, Wetzel 1983). Vertical differences in pH emerged in the reservoir in 1995 (Fig. 5i) with the onset of stratification (Fig. 5b). The late April peak and the strong increase in pH in the upper waters in late May were coincident with two maxima in Chl (Fig. 5f). Values of pH were greater than 9 for most of the late May-August interval in 1995 (Fig. 5i; at the time of sampling, rather substantial variations are to be expected within a day related to diurnal variations in photosynthesis driven by incident light). These very high values reflect the limited buffering capacity of the reservoir, associated with its low alkalinity (NYCDEP 1997). The pH of the lower waters decreased progressively to a minimum of about 6.1 in late July; the vertical difference in pH was a maximum at that time (~ 3.5 units; Fig. 5i). The pH of the upper and lower layers reconverged with the approach to fall turnover, at a value of about 7.

## *Spatial Patterns*

Longitudinal gradients in measures of water quality prevail along the main axis of Cannonville Reservoir (April-October, Fig. 7a-e). Such gradients are widely observed in reservoirs, associated with the transition from a riverine to a lacustrine environment (e.g., Cooke et al. 1993, Kennedy et al. 1982, Kimmel and Groeger 1984, Thornton et al. 1990). Progressive improvements in average water quality conditions occur along the WBDR branch of the reservoir toward the dam, though the differences between sites 1, 2 and 4 were minor (Fig. 7). Average conditions at site 3 in the Trout Creek arm of the reservoir (Fig. 1) tend to be very similar to those measured near the intakes at site 4 (e.g., Fig. 7). This general longitudinal distribution was observed for nearly all the samplings of 1995 (and previously in the long-term monitoring program), though substantial variability in the details of the structure occurred. Temporal variability (represented by ± 1 standard deviation bars in Fig. 7) was the greatest at site 6 in 1995 for TP, Chl, total suspended solids (TSS), and $T_n$, reflecting variations in particulate and phosphorus inputs, that were driven in part by natural variations in runoff (Longabucco and Rafferty 1998), as well as the dynamics of sediment resuspension (Effler et al. 1998a). We consider site 6 representative of the riverine zone of this reservoir (Fig. 1). The intermediate conditions of site 5 (Fig. 1) with respect to water quality conditions and relative variability (Fig. 7) are consistent with the "transition zone" (from riverine to lacustrine zones; see Fig. 1) described by Kimmel and Groeger (1984). The other four sites are representative of the lacustrine zone of the reservoir (Fig. 1); this zone represents about 80% of the full reservoir volume. This zonation (Fig. 1) is further supported by the character of longitudinal gradients of other (in addition to SD) optical parameters (Effler et al. 1998b) and in deposition rates for particulate constituents (Effler and Brooks 1998), reported subsequently in this issue.

A trophic gradient is manifested along the main axis of the reservoir (e.g., average Chl concentrations of ~ 21 and 9 µg · L$^{-1}$ at sites 6 and 1 in 1995, respectively; Fig. 7b), consistent with the gradient in TP concentrations (e.g., average concentrations of ~ 60 and 20 µg · L$^{-1}$ at sites 6 and 1 in 1995, respectively; Fig. 7a). However, a substantial fraction of the TP in the reservoir in 1995, particularly at the upstream sites, was apparently associated with tripton (e.g., terrigeneous origins) (Effler and Brooks 1998, Effler et al. 1998a). The average values of SD over the April-August interval

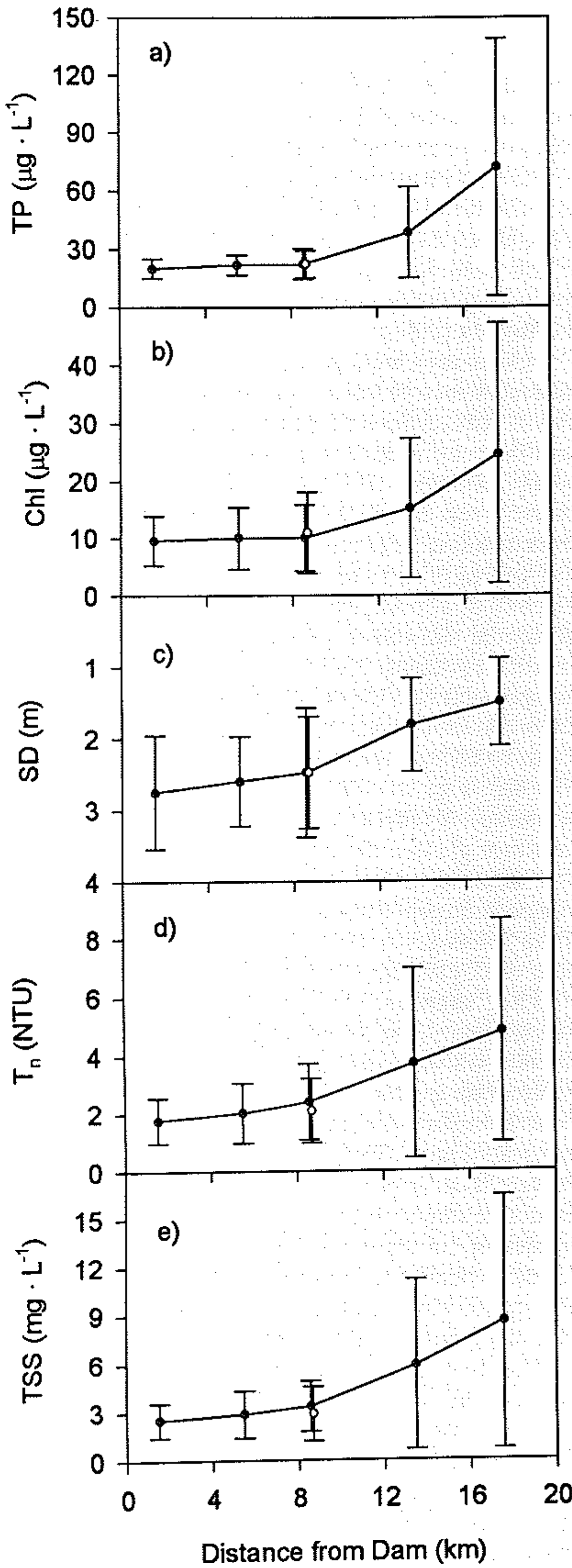


**Figure 7.–Longitudinal profiles of seasonal average values of selected water quality parameters for the upper waters of Cannonsville Reservoir in 1995: a) TP, b) Chl, c) SD, d) $T_n$, and e) TSS. Dimensions of vertical bars (±1 standard deviation) describe temporal variability.**

ranged from about 1.5 m at site 6 to about 2.8 m at site 1 (Fig. 7c). Gradients in $T_n$ and TSS concentrations, as measured in Cannonsville Reservoir (Fig. 7d and e), are widely observed in reservoirs, and consistent with the transition from a riverine (e.g., advection dominates transport processes) to a lacustrine (e.g., diffusion becomes relatively more important to transport; Chapra and Reckhow 1983) environment. The loss of portions of the allochthonous and resuspended sediment loads in the reservoir through localized deposition in upstream portions of the reservoir is depicted by Effler and Brooks (1998).

Systematic changes in the vertical distributions of particulate materials accompanied the major drawdown (Fig. 5a) of Cannonsville Reservoir in 1995 (Fig. 8) that have been attributed to sediment resuspension processes (Effler et al. 1997a). In early summer, when the reservoir was nearly full, no major increases in the concentrations of particulate materials were observed with the approach to the sediment-water interface (Fig. 5a). By late August the reservoir had experienced substantial drawdown [~ 12 m decrease in surface elevation by August 29 (Fig. 8b)]. Substantial sediment resuspension (see Bloesch 1995) was evidenced in this late summer interval by the sharp increases in PP, $T_n$, and TSS just above the sediment water interface (Fig. 8b); e.g., the PP concentration just above the sediments was more than 50 μg·L$^{-1}$ on August 29, about five times greater than it was in early summer. The various manifestations of the operation of the sediment resuspension process, and its coupling to the drawdown of the reservoir in 1995, are more fully resolved by Effler et al. (1998a).

The increase in the non-phytoplankton contribution to the PP pool above the thermocline of the reservoir was indicated in late summer of 1995 by the increase in the PP/Chl ratio; e.g., the ratio was about 1 on June 6 compared to approximately 2 on August 29 (Fig. 8). The generally low concentrations of SRP and TDP in the reservoir hypolimnion and the absence of major increases with the approach to the sediment-water interface (Fig. 8) are consistent with the less vertically resolved data presented previously in this manuscript (Fig. 5h), indicating the sediments are not a major source of dissolved P to this layer.

The three-dimensional "griding" of the reservoir conducted in 1994 and 1995 failed to resolve substantial lateral variations in fluorescence (e.g., Fig. 9) and beam attenuation. The absence of recurring lateral structure has important implications with respect to the hydrodynamic/transport framework needs of related water quality models for the reservoir. The prevalence of lateral gradients in phytoplankton biomass could perhaps dictate the need for a three-dimensional hydrodynamic framework, capable of simulating transport in the lateral, as well as vertical and longitudinal (simulated with a two-dimensional model; e.g., Gelda et al. 1998) dimensions. The lack of recurring lateral structure (e.g., Fig. 9) eliminates the need for this added level of model complexity, as well as the inclusion of lateral sites in the long-term water quality monitoring program for Cannonsville Reservoir.

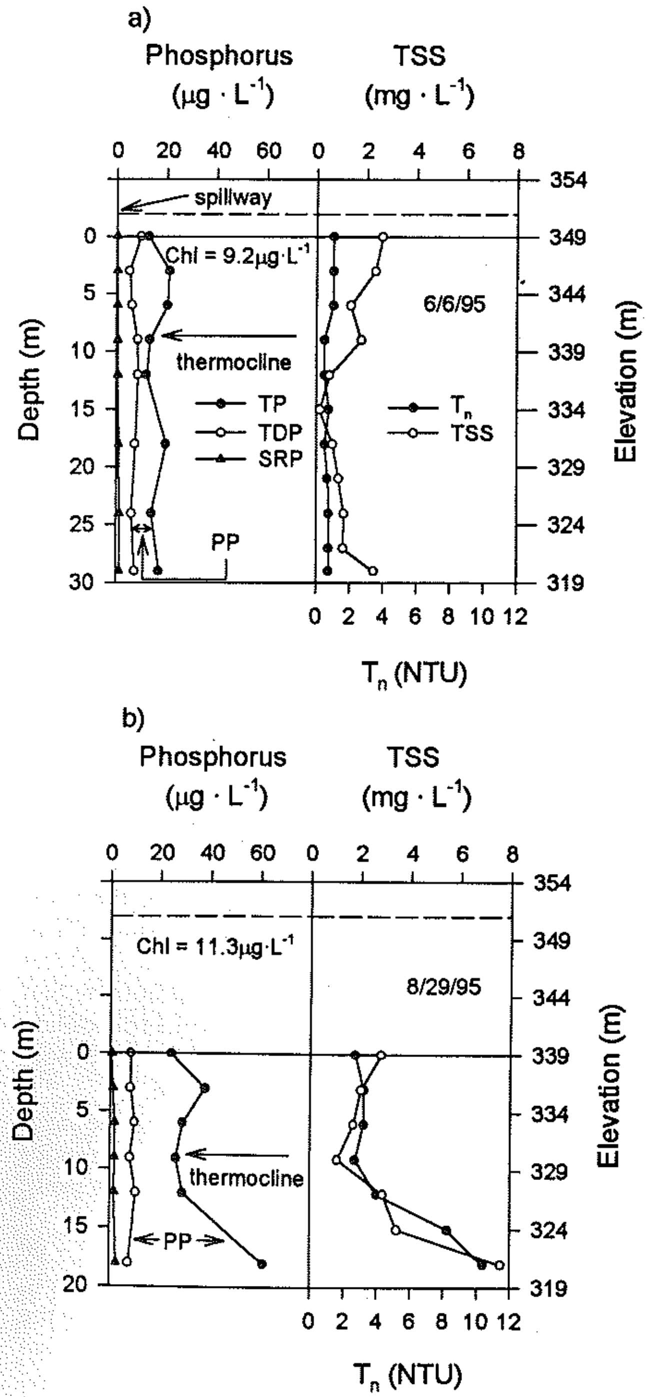


**Figure 8.**–Vertical profiles of P species, $T_n$ and TSS at site 4 in Cannonsville Reservoir, 1995: a) June 6, and b) August 29.

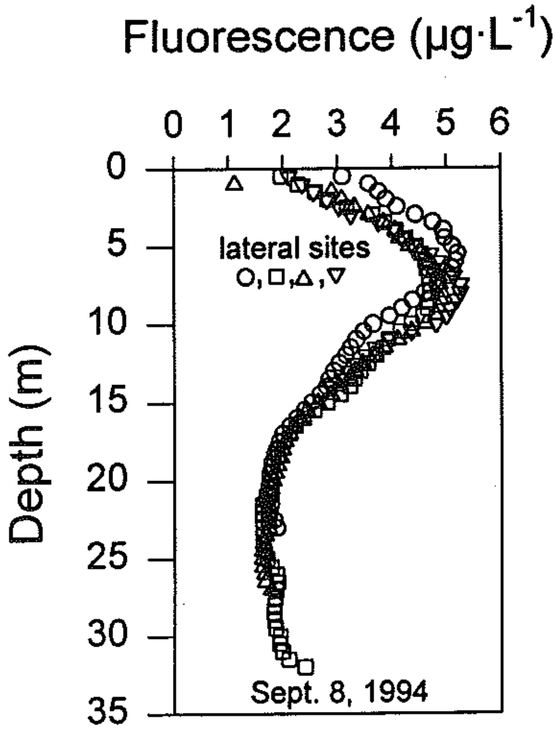


**Figure 9.–Profiles of fluorescence along a lateral transect (see Fig. 1) in Cannonsville Reservoir, collected on September 8, 1994.**

# Management Perspectives

Long-term monitoring has established that the trophic state of Cannonsville Reservoir in most years has been eutrophy (upper mesotrophy in other years). Greater degrees of eutrophy and shorter periods of summer stratification have been observed in years of increased drawdown, indicating there is a water quality cost for operation of the reservoir. More detailed monitoring in 1995, a year of major drawdown, supported these observations and was invaluable in: 1) identifying certain processes, 2) identifying the needs of model frameworks, and 3) supporting calibration testing of hydrothermal models (Gelda et al. 1998, Owens 1998b) and a eutrophication model (Doerr et al. 1998), presented subsequently in this issue. These models are to serve as management tools for this reservoir (e.g., Owens et al. 1998a) and as forerunners of tools to be developed for other reservoirs in NYC's water supply system.

Evidence is presented that non-phytoplankton particles (tripton) compromised the concentration of total phosphorus (TP) and Secchi disc transparency (SD) as measures of trophic state during the major drawdown period of the summer of 1995. Additional evidence indicates the sediment resuspension process was at least in part responsible for introducing tripton into the reservoir's water column (also see Effler et al. 1998a). Abrupt and large variations in phytoplankton biomass [as measured by the concentration of chlorophyll (Chl)] are common in the reservoir's upper layers. Conspicuous signatures of primary production (phytoplankton activity) to support testing of a eutrophication model for the reservoir include the progressive depletion of $NO_x$ (nitrate plus nitrite) from the upper layers and the maintenance of low concentrations of soluble reactive phosphorus (SRP) in those layers during summer. The proliferation of nuisance nitrogen-fixing cyanobacteria (blue-green algae) in the late summer of 1995 was indicative of the development of nitrogen-limitation of phytoplankton growth; this is a recurring feature in the reservoir. Oxygen is depleted progressively from the lower layers (hypolimnion) of the reservoir. Anoxia (absence of oxygen) prevailed in the lowermost layers for about one month in 1995, but no major release of phosphorus (P) from the sediments occurred. Substantial sediment release is to be expected if the period of anoxia increases, e.g., in response to an increase in anthropogenic phosphorus loads. Evidence is presented that a sink (loss) process(es) operates for the dissolved P released in the hypolimnion as part of decomposition of deposited organic material.

Distinct gradients in water quality and measures of trophic state prevail along portions of the main (longitudinal) axis of the reservoir; approximate bounds of the riverine, transition, and lacustrine zones of the reservoir were identified. Relatively modest differences were observed within the lacustrine zone (also see Effler and Brooks 1998, Effler et al. 1998a, b, Stepczuk et al. 1998), which represents about 80% of the volume of the reservoir when it is full. No substantive and recurring lateral gradients in phytoplankton biomass and related optical characteristics were observed in spatially intensive profiling (e.g., $n \geq 45$ sites) of the reservoir. These spatial features of water quality have important implications for the transport frameworks adopted in modeling analyses presented subsequently for the reservoir in this issue (Doerr et al. 1998, Gelda et al. 1998, Owens 1998b).

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