An ecological assessment of the trophic structure of York Pond in Coos County Milan, NH

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Abstract

We examined the physical, chemical and biological properties of York Pond in Coos County Milan, NH as a part of a 6-lake study. Chemical and physical characteristics measured included: total phosphorus and nitrogen, turbidity, light profiles with Secchi disk depth, specific conductivity, oxidation-reduction potential, dissolved oxygen, and temperature. Biological analysis included: phytoplankton percentage, chlorophyll *a* fluorescence, zooplankton abundance and zooplankton biomass. York Pond had eutrophic levels of several parameters in the epilimnion, including: chlorophyll *a* (39.4 ± 1.04 µgL^{-1}), total phosphorus (46.3 ± 0.67 µgL^{-1}), total nitrogen (843.3 ± 18.48 µgL^{-1}), turbidity (22.0 ± 0.09 NTU), and percentage of cyanobacteria at (42.0 ± 9.60 %). Cyanobacteria were even more dominant deeper in the water column, making up more than 70% of the net phytoplankton. The primary source of nutrient loading appears to be effluent from the fish raceways at the Berlin fish hatchery. Eutrophication has shifted the phytoplankton of York Pond toward cyanobacteria, with *Anabaena* as the dominant genus in the fall. The could have impacts across multiple trophic levels. Comparisons of York Pond to other study lakes suggest that Secchi disk, chlorophyll *a* and total phosphorus may be useful predictors of cyanobacteria dominance in New England lakes.

Introduction

Water quality degradation has steadily increased over the past half century due to anthropogenic impacts such as nutrient loading through poor farming practices, development of wetland systems and riparian zones, introduction of invasive flora and fauna, and physical alteration from damming. These impacts have had deleterious effects such as biotoxins and lakes with unstable fish populations. Human development and upland practices have negatively affected lake biodiversity (species richness, species evenness, species composition and interspecific interactions), which is the driving force behind biocomplexity of the ecosystem functioning. Biocomplexity arises from the multiple behavioral, biological, chemical and physical interactions between the biotic (flora & fauna) and abiotic (habitat) (Symstad *et al.*, 2003).

The value of biodiversity for human use is the sum of the functions we can extract from it. Ecosystem functions can range from biomass of a fish population for a food crop and recreation to maintaining a pristine lake for drinking water and aesthetic purposes. Zedler et. al (2001) looked at the relationship between ecosystem functions and biodiversity through marsh restoration by evaluating how plant communities affect ecosystem functions. The three key functions include aboveground biomass for bird habitats and carbon storage, recruitment, and belowground biomass for carbon and nitrogen accumulation; the plots with a more diverse plant community had higher functions than the monoculture (dominated by a single species). Their conclusions were based on the theory that each species has a functional trait that contributes to the ecosystem. Furthermore, the interspecific

competition causes the species to utilize the ecosystem resources more efficiently. Such results reinforce the idea that biodiversity is positively correlated with ecosystem functions.

Biodiversity is most commonly degraded through eutrophication (an increase in trophic condition of a lake) (Lampert & Sommer, 1997). The major cause is the enrichment of key nutrients (nitrogen and phosphorus) into a water body leading to overall high productivity. Rapid reproduction of primary producers ultimately leads to high mortality rates of algae and decomposition by bacteria. Decomposition depletes the dissolved oxygen levels in lakes (Dillon & Rigler, 1974). Anoxia increases in the hypoliminion spreading vertically, leading to a habitat with a decreased fitness of its inhabitants. Fish suffer from anoxia not only because they require high amounts of dissolved oxygen, but their eggs sink to the bottom where reproductive success is reduced from increased egg mortality. Eutrophication can also cause a shift to extensive populations of harmful cyanobacteria. *Anabaena* and *Microcystis* are two common cyanobacteria that can alter the chemical and physical properties of a lake. These organisms can increase anoxic waters upon degredation, interfere with zooplankton feeding (Gliwicz et. al, 1980), and release harmful biotoxins (Claska & Gilbert, 1998) that bioaccumulate up the food chain.

Eutrophication, decreased biodiversity, and biotoxins have forced policy makers to create a regional prevention and mitigation plan. The first step in any type of mitigation project involves a detailed evaluation. York Pond is the primary site in a six lakes study evaluated in a regional comparative lakes study. York Pond was chosen to examine the lake for possible effects of an adjacent fish hatchery. It appear that few studies have focused their research on the potential impacts from fish hatchery effluent in freshwater ecosystems. Our study took place on York Pond on September 25th, 2003. It included an evaluation of the physical, chemical, and biological characteristics that might contributed to the overall lake conidtion in terms of biodiversity and functioning. It is necessary to examine many parameters when the goal is to better understand the biocomplexity of a lake web. Our goal was to assess the trophic structure and biodiversity of the lake through a multiple parameter study.

Methods

Study Site- The assessment of York Pond (YP) (44°30.48 N, 71°20.257 W) in Coos County Berlin, NH (Fig. 1) included biological, chemical and physical parameters. The five other lakes tested in the same manner are Cedar Pond (CP), Christine Lake (CL), Lower Sawyer (LS), Upper Sawyer (US), and Townhouse Pond (TP) (Table 1). These tests were taken September 25, 2003 on a 60-70 °C day with minimal overcast in the deepest point of the lake. York Pond is located at an elevation of 451.1 m (1479.9 ft). Its watershed is relatively small at 165.5 Ha and consists mainly of forested area with minimal development. The pond has an area of 8.5 Ha, a volume of 23.5 Ha-m, and its watershed is 165.5 Ha (Fig. 1). Its mean depth is 2.8 m (9.2 ft), and its maximum depth is 5.2 m (17.0 ft). It is classified as a eutrophic lake by the state of NH from the visible amounts of high primary productivity. York Pond had one house in the riparian zone with a road bordering within 30.5 m (100 ft), including power lines running along side. Also, a fish hatchery was located immediately upland and was discharging effluent directly into York Pond.

Chemical and Biological Parameters and Water Collection - The maximum depth of York Pond was measured with a hand held sonar. The YSI 650 data logger in conjunction with the YSI Sonde 6600M lake probe was used to obtain a multiparameter profile of the pond. This instrument was calibrated immediately before use for atmospheric pressure, pH, chlorophyll *a*, turbidity and dissolved oxygen. We measured depth profiles of temperature, specific conductivity, percent and concentration of dissolved oxygen (DO), pH, redox, turbidity, chlorophyll *a*, and chlorophyll *a* fluorescence at a depth resolution of 3 cm. The multiparameter was lowered at a 0.5 m.min^{-1} rate until it reached the lake sediments. Data were automatically logged every 3 seconds or approximately every 2.5 cm in depth.
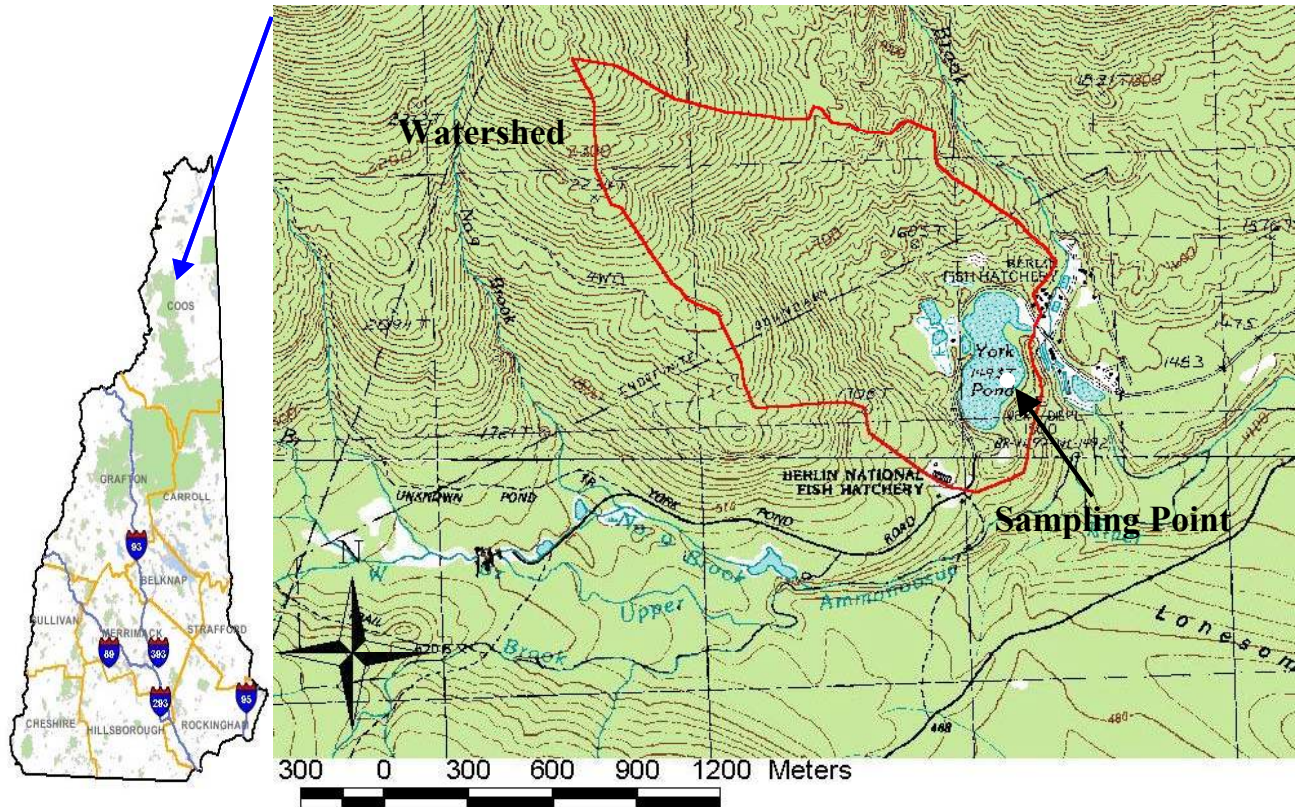
Figure 1- York Pond Milan, NH (44° 30, 71° 20) delineated watershed as indicated by the solid line including its town and state. Approximate sampling point is indicated by the white dot in York Pond.

At the beginning of each study, temperature was measured at 0.5 m intervals to determine the depth of the thermocline boundaries for integrated water sampling. An integrated tube sampler was lowered to the bottom limit of the epilimnion to obtain water samples. Samples were placed into containers pre-rinsed with lake water. Triplicate samples were then shaken and poured into 500 ml containers for phosphorous and total nitrogen.

The stratification was determined with a multiparameter probe (YSI model 6600). A discrete sampler water bottle with a volume of 5.27 L was lowered to eight assigned depths (1 m, 1.5 m, 2 m, 2.5 m, 3 m, 3.5 m, 4 m, & 4.5 m) and then closed with a messenger. Then, 50 ml was placed into a cup labeled whole lake water, and additional two 50 ml were filtered through a 50 μm ring net. One of these 50 μm filtrates was filtered through a 2 μm filter for plankton analysis.

Samples collected with the integrated tube sampler and discrete water column samples were size fractioned. These samples were mixed and filtered at four different fractions: whole lake water (WLW), <50 μm, <2 μm, and <0.2 μm. All samples except WLW were first run through a 50 μm mesh size ring. Then, they were either filtered through 2.0 μm filter Millipore isopore membrane filters (47 mm) or 0.2 μm filter Millipore isopore membrane filters (47 mm). These fractionated filtrates were placed in 250 ml plastic containers for chlorophyll fluorescence. A sample of 250 ml of epilimnion water was filtered through 0.45 μm Millipore white HAWP filters (47 mm) for chlorophyll *a* and dissolved color from the 0.45 μm filtrate. Chlorophyll filters were placed into containers filled with desiccant. Also, 50 ml of filtrate collected through discrete sampling was filtered through a 2 μm filter. Then, 60 ml of the filtrate was poured into separate containers for dissolved color (CDOM) and nitrate/nitrogen analysis, and the remaining

filtrate was frozen for later microcystin analysis and chlorophyll *a* fluorometry. The acid neutralizing capacity (ANC) of the lake was measured in triplicate (Lind, 1985). The titrant used was 0.002 N H_2SO_4.

Light and Biotic Measurements - A Turner Fluorometer 10-AU was used to measure the chlorophyll *a* fluorescence in triplicate. The instrument was first blanked with distilled water. A cuvette was filled to the same volume and placed in the same orientation each use. The instrument read each sample for 15 seconds to obtain a consistent recording. The temperature was taken for each sample. This procedure was then repeated for every sample, which included WLW, <50 μm, and <2 μm fraction samples at all discrete depths.

The Licor LI-1000 instrument was assembled in full sunlight, while the light meter was submerged directly beneath the lake surface water. The light meter was then calibrated, and lowered every half meter until it reached the bottom or the end of the cord (9 m). Secchi disk measurements were taken by lowering it until it was not visible using a scope. It was then raised until it just became visible. The measurement was the average of the two depths. These measurements above were done in triplicate.

A vertical plankton tow net (80 μm) with a diameter of 0.25 m was lowered 3.5 m. Sampling efficiency was estimated to be 70%. The plankton net was raised at a constant rate (0.5 m s^{-1}). The process was repeated to obtain four samples, one live sample and three samples preserved with formalin-sucrose (3 ml per 50 ml sample). Discrete samples were passed through a funnel with a 50 μm mesh. Meshes with particulates (net plankton) were then preserved with 4% formalin/sucrose for lab analysis. Next, a discrete plankton sampler (Haney chamber) was lowered to the same depths as the discrete water sampler. The 5.27 L sample was then filtered out onto a 50-μm mesh filter, and the zooplankton were preserved in 4% formalin-sucrose mixture.

A sample from the lake sediment was taken with a benthic gravity corer, ARI Gravity Corer, at the lakes maximum depth. The corer was lowered carefully into the sediment, and closed with a messenger weight. The upper 1 cm of the sediment layer was removed from the core for microcystins analysis.

Lab Analysis - The total nitrogen, phosphorous, CDOM and chlorophyll *a* concentrations were analyzed in the laboratory using the frozen water samples from the integrated tube. A scanning spectrophometer (Cary 50) with the aid of software was used to obtain total nitrogen absorbance values using a second derivative for each sample (Crumpton *et al*, 1992). A 5 cm (path length) Quartz cuvette was used for all parameters, except total phosphorus which used a 10 cm path length cuvette. Chlorophyll *a* (Lind, 1985) and CDOM (Cuthbert & Giorgio, 1992) absorbance values were calculated with a Milton Roy Spectronic 1001+ spectrophotometer with a 2 nm bandwidth. This instrument was also used to calculate the total phosphorus absorbance values (APHA, 1998).

Phytoplankton densities were counted using the samples collected through the vertical plankton tows and the discrete plankton samples, with a sub-sampling method and compound microscope. At least 100 organisms were identified to genus whenever possible to obtain a relative count.

Zooplankton densities were counted from the samples collected through the vertical plankton tows and the discrete plankton samples. All organisms except rotifers were identified to genus and counted for each sample. Counts of sub-samples were conducted for both the vertical tows (> 100 organisms) and the smaller volumes of the discrete samples (> 50 organisms). The average body lengths of zooplankton in the vertical tow samples (> 30 organisms) were measured with an image analysis program called Metamorph. Zooplankton dry weights were calculated from average body length of each genus (Dumount *et al.*, 1975).

Composite, integrated whole lake water (WLW) samples were placed in HDPE sample cups (250 ml) and immediately placed on ice in the field and stored frozen (–40° C). The WLW samples were frozen/thawed (3X) to release intracellular microcystin (MC). Samples (15 ml)

Table 1- Morphometric data for all six lakes. Data were complied from the NH Department of Enivornmental Services.

	York Pond	Townhouse Pond	Cedar Pond	Christine Lake	Lower Saywer	Upper Sawyer
Abbreviation	YP	TP	CP	CL	LS	US
Town	Berlin	Milton	Milan	Stark	Livermore	Livermore
Sampling Date	25-Sep-03	18-Sep-03	26-Sep-03	26-Sep-03	11-Sep-03	11-Sep-03
Lattitude	44° 30.48'N	43 26.052 N	44° 35.739'N	44° 37.8'N	44 03.041 N	44° 03.16 N
Longitude	71° 20.257'W	70 58.867 W	71° 16.209'W	71° 24.192'W	71 23.065 W	71° 22.800 W
Elevation (m)	451.1	126.5	349.6	360.9	621.8	621.8
Max Depth (m)	5.2	10.1	17.7	19.5	29.6	10.0
Mean Depth (m)	2.8	3.3	8.3	6.9	12.2	5.0
Relative Depth %	1.6	1.3	2.8	2.1	6.0	5.6
Watershed: Lake Volume	7.4	186.7	4.9	2.6	1.4	2.1
Watershed Area (Ha)	165.5	29525.9	1243.2	1165.5	310.8	23.6
Lake Area (Ha)	8.5	48.0	31.5	68.8	19.0	2.5
Lake Volume (Ha-m)	23.5	158.4	261.5	474.7	231.4	12.5
HRT (yrs)	2.5	0.0	0.4	0.8	1.0	N/A
MEI	9.3	36.4	8.0	3.7	2.1	4.1

were thoroughly mixed, transferred to borosilicate glass serum bottles (30 ml), and frozen on edge to maximize exposed surface area. The frozen samples were lyophilized in a freeze-dry system (LabConco Freezone 4.5) under vacuum (~30 X 10^{-3} mbar) at -50° C for 18-24 h. Dried material was then re-hydrated with 1.5 ml of water and mixed on a shaker table for 16-24 h. This procedure achieved a 10-fold concentration of materials and increased the sensitivity range of the ELISA methods used. About 1 ml of the re-hydrated sample was filtered through a 13 mm, 0.2 μm Whatman PTFE syringe filter into a 1.5 ml polypropylene centrifuge tube, which was stored frozen (–40° C) before ELISA analysis for MC.

Benthic core samples were taken at the deep site of each lake with a gravity corer (Aquatic Research Instruments) for analysis of nutrient (TP and TN) concentrations, MC concentrations, and enumeration of benthic cyanobacteria. The corer was equipped with a 7.5 cm diameter, 45 cm long, clear polycarbonate coring tube. The coring device was gently lowered into the sediment without disturbing the upper layers of benthos so that sediment filled one-half to three-quarters of the tube. A brass messenger was deployed to release a piston that created suction so that the corer could be retrieved with an intact benthic sample and minimal disturbance of layers. A rubber stopper was placed in the bottom of the coring tube as it was held just beneath the water's surface as the coring unit was retrieved. The corer head was gently removed; the top 2 cm of benthic sediments and overlaying/interstitial water (~225 ml) were collected in a sample cup

Table 2. Physical and chemical averages from the multiparameter probe for York Pond 9-25-03. Standard errors are in parenthesis.

	Depth	Temp	SpCond	Dis. Oxy.	Dis. Oxy.	pH	Eh7	Turbidity	Chl.
	(m)	(C)	(μS cm^{-1})	(%)	(mg L^{-1})		(mV)	(NTU)	(μg L^{-1})
Eppilimion Ave.	1.6 (0.1)	15.7 (0.1)	24.8 (0.1)	109.6 (0.9)	10.9 (0.1)	7.4 (0.1)	186.1 (0.3)	22.0 (0.1)	39.4 (0.4)
Metalimnion Ave.	3.8 (0.1)	15.1 (0.1)	27.0 (0.3)	67.4 (4.1)	6.8 (0.4)	6.7 (0.1)	169.3 (0.6)	20.0 (0.2)	31.8 (0.6)

(250 ml), and stored on ice until they were split into nutrient and MC sub samples.

For ELISA analyses, several HDPE scintillation vials (20 ml) were filled about three-quarters full (~15 ml) with thoroughly mixed benthic sample and kept on ice in the field, and frozen (-40° C) in the laboratory. The samples were subjected to 3 freeze-thaw cycles, after

which ~1.0 ml of thoroughly mixed benthic sample was filtered through a 13 mm, 0.2 μm Whatman PTFE syringe filter into a 1.5 ml centrifuge tube. The centrifuge tubes were stored frozen (–40° C) until ELISA analysis for MC.

All ELISA analyses for microcystins were performed using Microcystin 96-Well-Plate Kits (EnviroLogix Inc. Portland, ME). The three MC
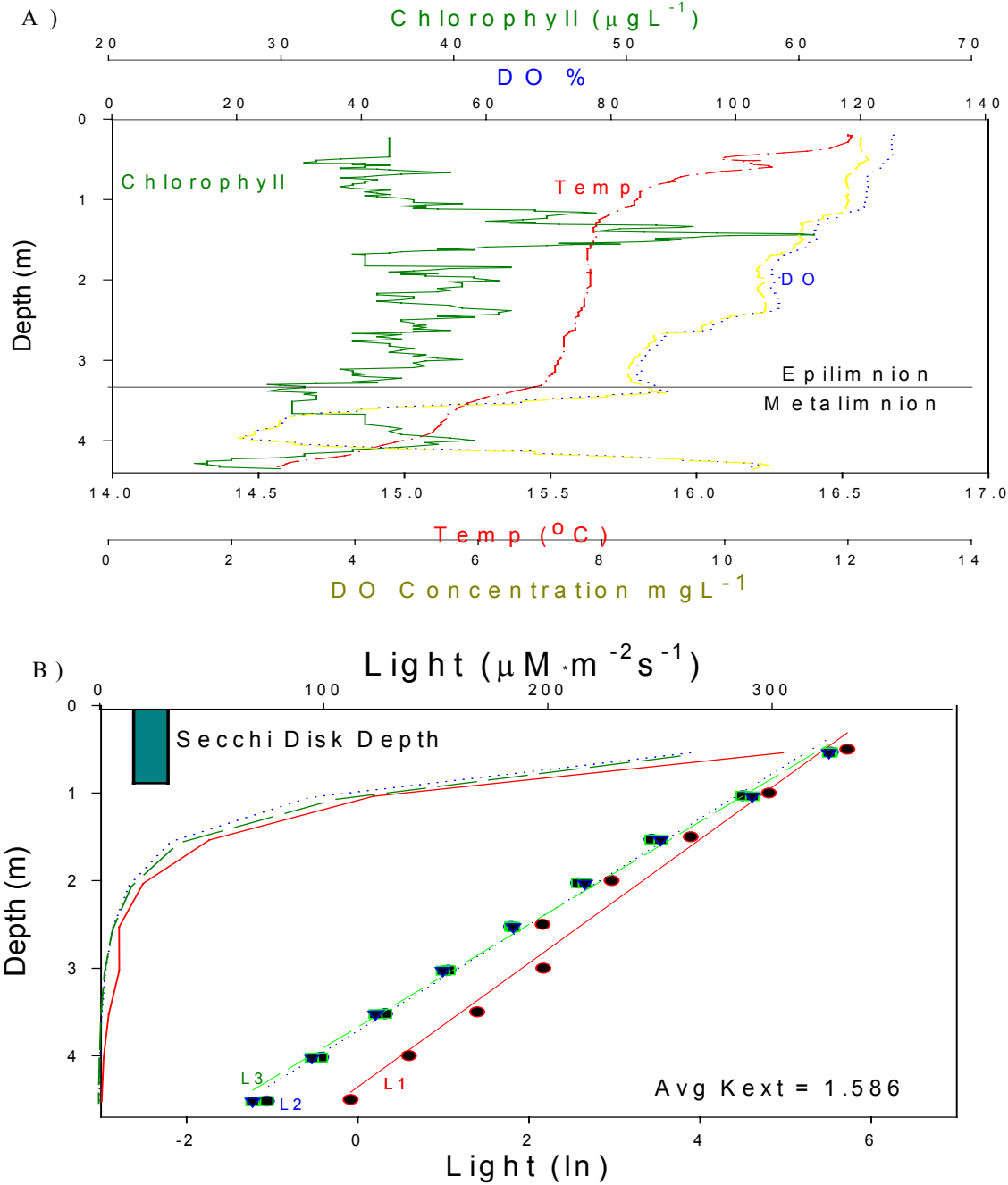

Fig. 2. Lake profiles of York Pond 9-25-03. A) Chlorophyll (μgL^{-1}), dissolved oxygen (percentage and concentration (mgL^{-1})), and temperature. B) Light profile including the three repetitions of light (mM m^{-2} s^{-1}) and light extinction. For replicate one adjusted r^2 = 0.98 and K$_{ext}$ = 1.41, for replicate two adjusted r^2 = 0.99 and K$_{ext}$ = 1.64, for replicate three adjusted r^2 = 0.99 and K$_{ext}$ = 1.70. The average SDD is 0.85 (<0.01) m.

kit standards provided (160, 500, and 1600 pg ml^{-1}) were further diluted with water to make 4 additional standards (15, 25, 53, and 100 pg ml^{-1}) to expand the standard curve and increase the sensitivity range of the assay.

Graphics and Statistics- Graphics and statistics were conducted with SigmaPlot and Systat software. Where triplicate analyses were appropriate, ANOVAs were used. A probability value of $p<0.05$ was used to determine significance. Data were log transformed to
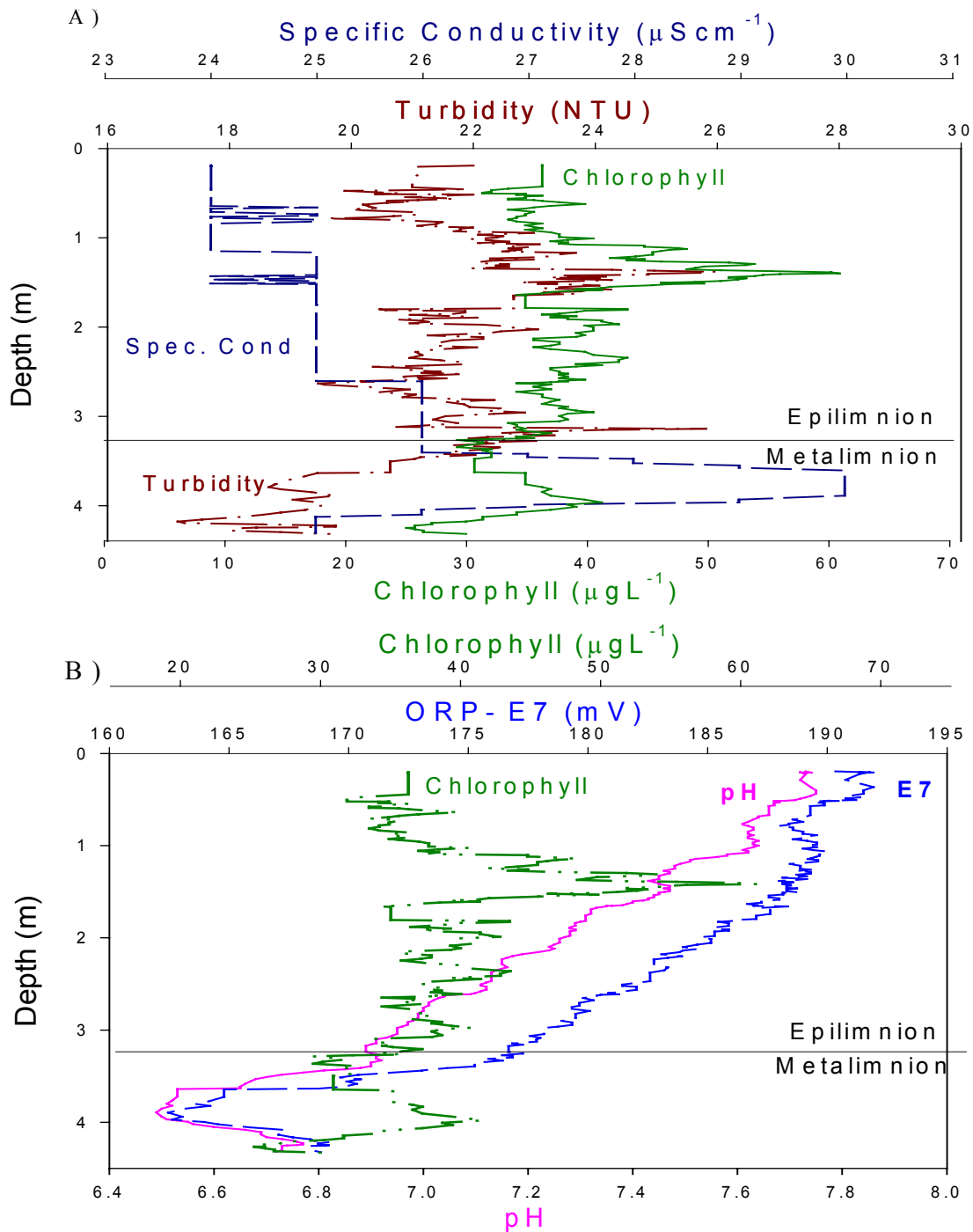

Fig. 3. Lake profiles of York Pond 9-25-03. A) Turbidity (NTU), specific conductivity (μScm^{-1}) and chlorophyll (μgL^{-1}). B) Chlorophyll (μgL^{-1}), pH and oxidation-reduction potential (mV).

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stabilize variance where appropriate.

The fluorescence chlorophyll *a* values were converted to concentration, and corrected by multiplying the multiparameter readings by 0.82 μgL^{-1} to account for variability in the multiparameter probe for high readings. The values were further corrected to the average epilimnetic integrated tube sample determined through chlorophyll *a* extraction. The ORP was corrected to pH 7.0 by removing the pH influence on it: +57 mV per every decrease of 1.0 pH unit below pH 7.0 and -57 mV per increase of 1.0 pH unit above pH 7.0.

Results

The area of the delineated watershed was 90.4 Ha (Fig. 1). Watershed area including the lake area to lake volume ratio was 7.4 (Table 1). The hydraulic retention time was 2.5 years. The relative depth, which is the maximum depth as a percent of the average surface diameter, is 1.58 m. The Morphoedaphic Index (Chow-Fraser, 1991) for York Pond from the average specific conductivity is 9.25.

Physical and chemical averages were calculated from the multiparameter probe data (Table 2). The temperature was 16.5 °C near the surface where there was a decline of one degree in the first meter marking a temporary thermocline (Fig. 2). Then, it remained constant until 3.25 m where the thermocline was reached (15.49 °C). The temperature decreased to the bottom of the lake where it reached its minimum of 14.57 °C. The chlorophyll curve had two major peaks, at 1.41 m depth (60.88 $\mu g\ L^{-1}$) and 3.97 m (41.19 $\mu g\ L^{-1}$). Turbidity closely followed the same trend as chlorophyll *a* until the start of the metalimnion, where it inversely correlates with chlorophyll (Fig. 3). The dissolved oxygen (DO) concentration and saturation curve are nearly identical from top to bottom, decreasing
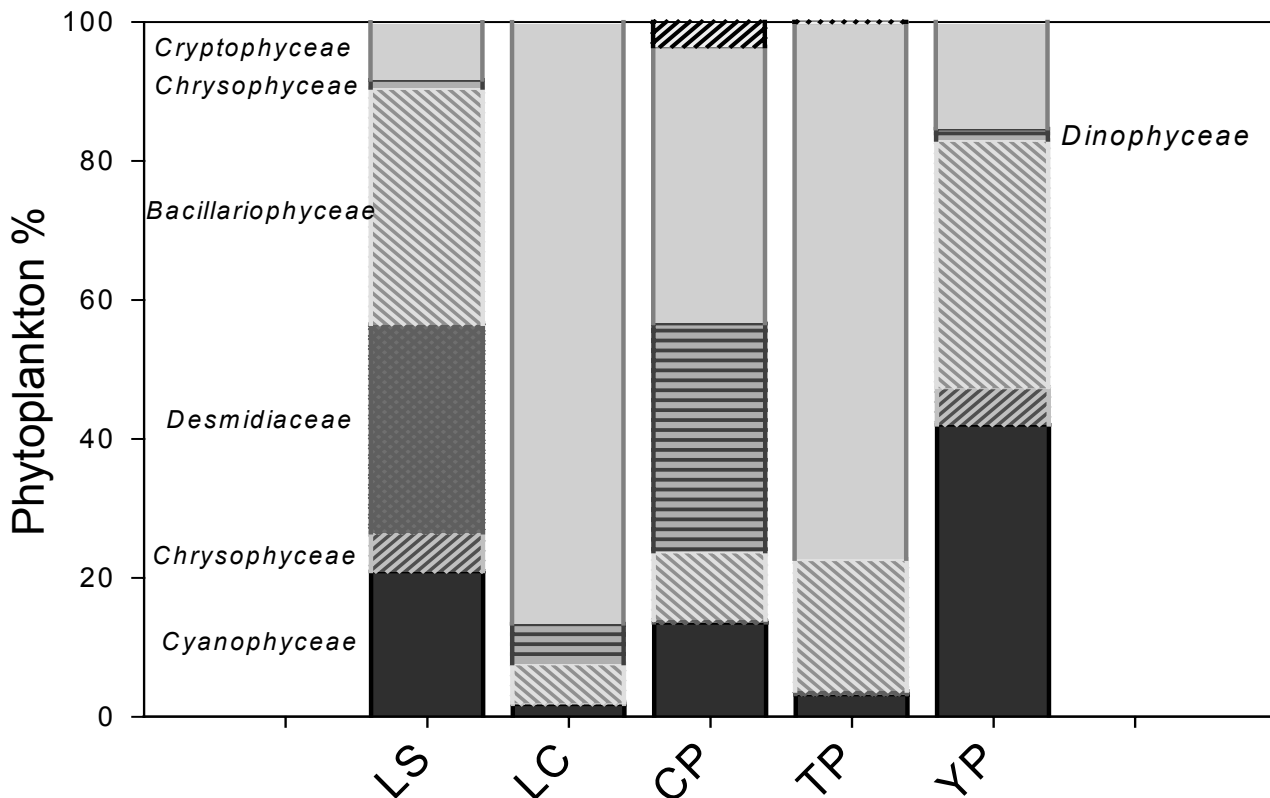

Fig. 4. Phytoplankton percentage composition of six lakes using netplankton in York Pond 9-25-03. *Cyanophyceae* (Bluegreens) are represented by solid black, *Chlorophyceae* (Green algae) by diagonal top right to bottom left light gray lines, *Desmidiaceae* (Desmids) by crosshatched dark gray, *Bacillariophyceae* (Diatoms) by diagonal top left to bottom right light gray, *Dinophyceae* (Dinoflagellates) by horizontal lines in dark gray, *Chrysophyceae* (Golden algae) by solid light gray, and *Cryptophyceae* (Cryptomonads) by diagonal top right to bottom left black lines. Upper Sawyer data is unavailable.

from the surface to the metalimnion, and then begin to rapidly decrease to a minimum of 21.1% saturation and 2.13 mg L^{-1} at 3.84 m. Dissolved oxygen then rapidly increases to over 100% saturation and 10.5 mg L^{-1} at the bottom of the lake. E7 (ORP corrected for pH) reached a minimum of 162.4 mV along with a pH minimum of 6.49 corresponding with the DO minimum depth. However, specific conductivity reached a maximum at a similar depth (3.89 m) of 30 μScm^{-1}. The light intensity at 1 m depth was nearly 300 mM m^{-2} s^{-1} and then sharply declines to approximately 16 mM m^{-2} s^{-1} at 2 m (Fig. 3). The average Secchi disk depth was 0.85 (± 0.00)
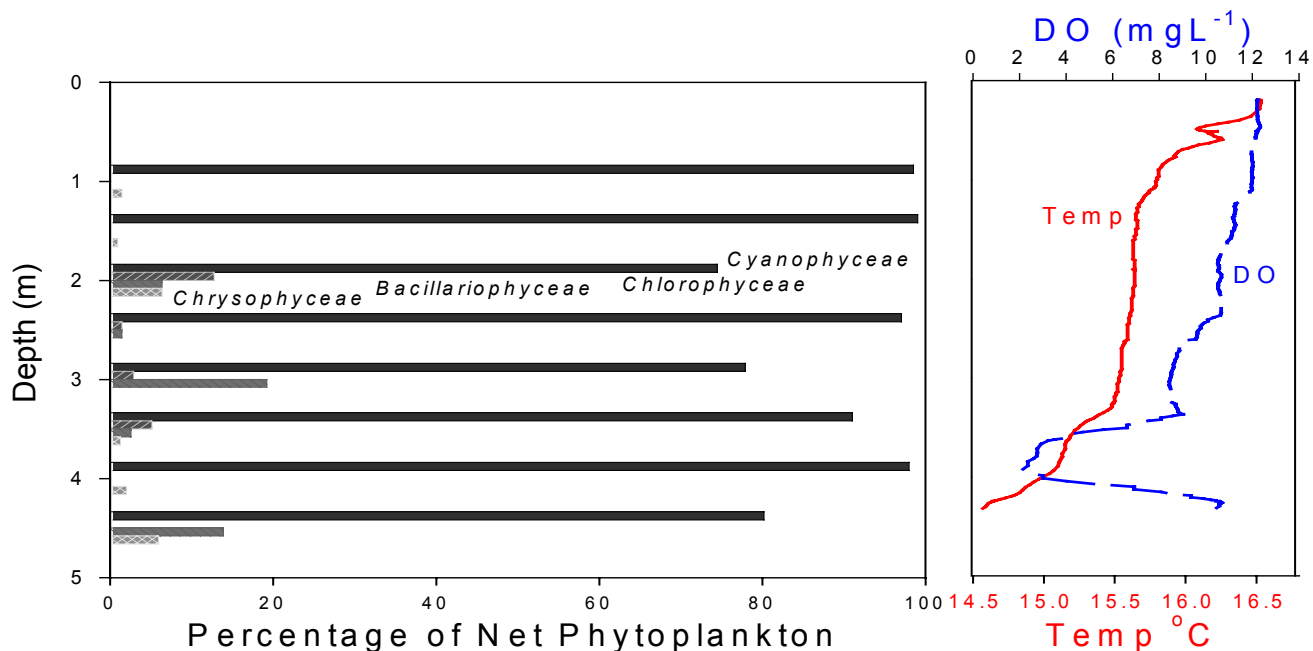

Fig. 5. Phytoplankton percentage profile obtained from discrete sampling using netplankton along with the temperature and dissolved oxygen curve of York Pond 9-25-03. *Cyanophyceae* (Bluegreens) are represented by solid black, *Chlorophyceae* (Green algae) by diagonal top right to bottom left lines, *Bacillariophyceae* (Diatoms) by diagonal top left to bottom right lines, and *Chrysophyceae* (Golden algae) by cross hatched light gray lines.
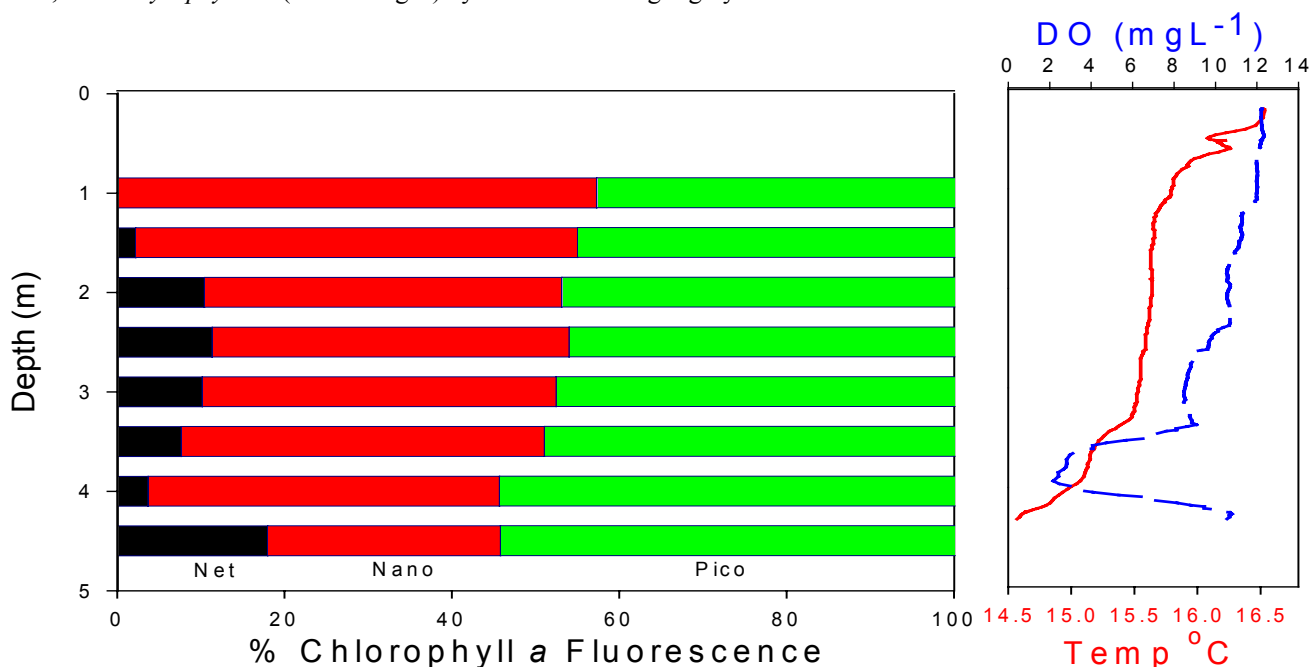

Fig. 6. Composition of phytoplankton sizes for York Pond 9-25-03 including the temperature and dissolved oxygen curve. Netplankton are represented by solid black, nanoplankton by solid dark gray, and picoplankton by light gray.

Table 3. Secchi disk and light extinction comparisons. SE of means are given in parentheses (Haney & Ikawa, 2000).

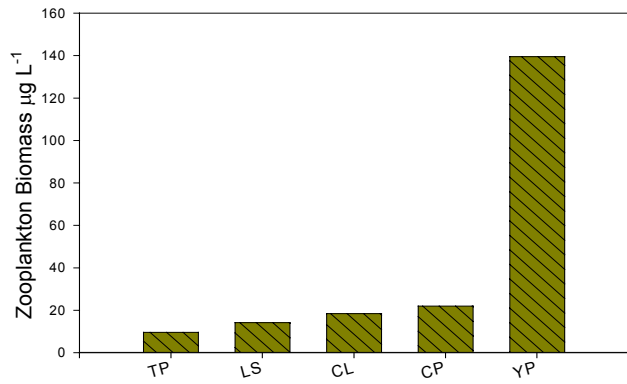| Lake	AVG SDD	AVG Kext
Cedar Pond	2.42 (0.02)	1.10 (0.03)
Christine Lake	7.35 (<0.01)	0.45 (0.02)
Lower Sawyer	9.78 (0.19)	0.39 (0.01)
Townhouse Pond	5.33 (0.09)	0.30 (<0.01)
Upper Sawyer	8.05 (0.23)	0.49 (<0.01)
York Pond	0.85 (<0.01)	1.59 (0.09)
NH Avgs	5.03 (0.26)	
NH Max	13.97	
NH Min	1.81	

Fig. 7. Zooplankton biomass (μgL^{-1} dry wt.) estimated from average body length and density for the six lakes. Zooplankton counts are unavailable for Upper Sawyer.

m and the average light extinction coefficient was 1.59 (Table 3).

York Pond had the highest concentration of eplimnetic average chlorophyll at 39.43 ($\pm$ 1.04) $\mu g \ L^{-1}$. York Pond also had the highest percentage of *Cyanophyceae* (bluegreen bacteria) at 42%, mainly *Anabaena* and M*icrocystis* (Fig. 4). The filamentous organism *Melosira,* the only *Bacillariophyceae* (diatom) at 35.5% found in York Pond, was the next most dominant

phytoplankton. *Chrysophyceae* (golden algae) at 15.5%, *Chlorophyceae* (green algae) at 5.4%, and *Dinophyceae* (dinoflagellates) at 1.6% were also present in a smaller percentage.

Cyanophyceae comprised more than 70% of the phytoplankton community at all eight depths from discrete sampling and was greater than 97% in more than half the depths (Fig. 5). From the chlorophyll fluorescence (Fig. 6), York Pond had
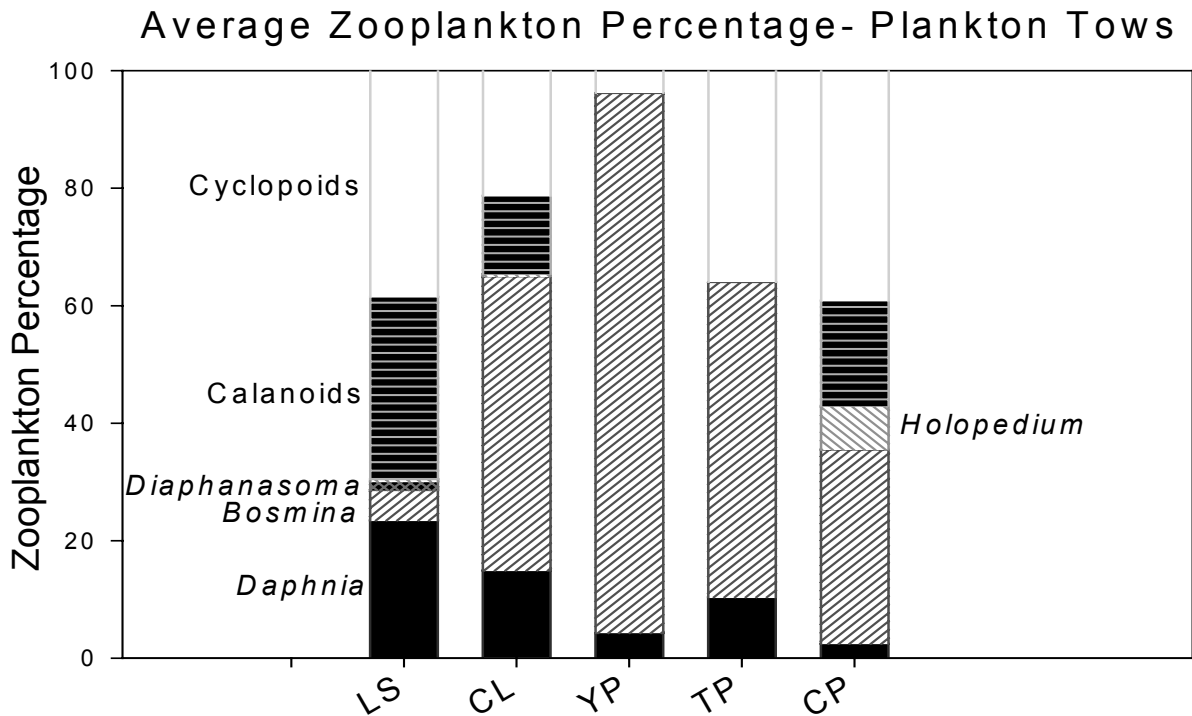

Fig. 8. Zooplankton percentage composition of the six lakes. Cyclopoids are represented by solid white, Calanoids by horizontal dark gray lines, Holopedium by diagonal top left to bottom right gray lines, Diaphanasoma by cross hatched lines, Bosmina by diagonal top right to bottom left gray lines, and Daphnia by solid black. Upper Sawyer data is unavailable.
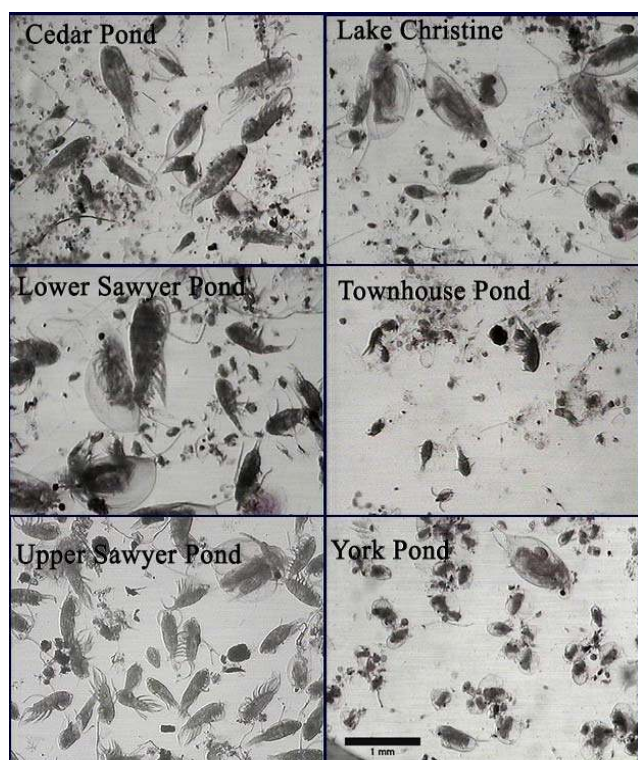
Fig. 9- Zooplankton photos of relative dominance for the six lakes in Fall 2003.

the highest percentage of nanoplankton (70.8%). Lower Sawyer, an oligotrophic lake, had the highest percentage of picoplankton (88.6%). In York Pond, the percentage of nanoplankton gradually decreased with depth while percentages of picoplankton and netplankton increased with depth. The most distinct change occurs at the upper region of the thermocline. Nanoplankton and picoplankton nearly account for the total chlorophyll fluorescence.

Zooplankton densities (273.34 ($\pm$ 32.06) ind. L^{-1}) from the vertical tows were greater than the other six lakes studied and the biomass (139.36 μg L^{-1}) was also high (Fig 7). *Bosmina* (91.9%) dominated the zooplankton community of York Pond with *Daphnia* (4.3%) and *cyclopoid* Copepods (3.8%) present in smaller amounts (Fig. 8). York Pond had the highest dominance of one organism in any of the lakes studied (Fig. 9). *Bosmina* was present in all the lakes, and in some cases, composed a high percentage of the zooplankton community. The maximum density (407.1 ind. L^{-1}) of zooplankton in York Pond was found at 3.5 m on the upper region of the

thermocline (Fig. 10). The lowest density (37.2 ind. L^{-1}) was found at 1.5 m depth

York Pond had the highest amounts of chlorophyll (39.4 ($\pm$ 1.04) μg L^{-1}), total phosphorus (46.3 ($\pm$ 0.67) μg L^{-1}), total nitrogen (843.3 ($\pm$ 18.48) μgL^{-1}; Fig. 11), turbidity (24.7 ($\pm$ 0.09) NTU), and *Cyanophyceae* (42.0 ($\pm$ 6.90) %; Fig. 12). However, it had the lowest molar total nitrogen to total phosphorus ratio of 40.3. York Pond had the concentration preserved of benthic microcystin with an average of 3,331.5 pg mL^{-1}, plankton microcystin at (30.8 pg mL^{-1}), and the second highest concentration of whole lake water microcystin at (28,120.2 ng g^{-1} wet wt.) (Fig. 13). New Hampshire average, maximum and minimum values were obtained from a 50-lake study (Haney & Ikawa, 2000). York Pond's chlorophyll value was nearly twice as much as the NH maximum value of 22.2 μg L^{-1}. The above values classify York Pond as a eutrophic lake.

A significant inverse relationship (p<0.01) appeared between average Secchi disk depth (SDD) and average chlorophyll within these six lakes (Fig. 14). As total phosphorus increased, total chlorophyll significantly (p<0.01) increased with it. Total phosphorus shows a positive relationship with percentage *Cyanophyceae*, however it proved not to be significant (p=0.10). Total chlorophyll significantly (p<0.01) increases along with TN: TP. Carlson's (1977) trophic state index (TSI) was calculated for all the lakes of study (Fig. 15). York Pond had the highest TSI for total phosphorus (59), chlorophyll (67) and Secchi disk (62). York Pond also had higher values of TSI than the NH maximum for Secchi disk and chlorophyll.

Discussion

York Pond can be classified as a eutrophic water body by multiple parameters and multiple classification limits. A biotic method of classification of a lake is to measure the total biomass and production of primary producers. The chlorophyll concentration is a proxy measurement for phytoplankton in the water column, and York Pond has nearly double the

amount of the NH maximum recorded value. This pond also has the highest zooplankton biomass, which makes it even more remarkable that the average chlorophyll concentration is at 39.4 ($\pm$ 1.04) μg L^{-1} because of the high grazing rates. Secchi disk is used as a surrogate of chlorophyll, which measures transparency. York Pond's Secchi disk depth was 0.85 ($\pm$ 0.00) m, which is less than the NH minimum depth. The other method of classification of a lake is to use abiotic parameters. The main parameter for this is total phosphorus because it is considered a limiting nutrient for most freshwater bodies. York Pond's average total phosphorus values are the highest in this six-lake study and nearly match NH maximum-recorded value. These three parameters place York Pond well into the classification of a eutrophic lake according to several different classification schemes (Table 4).

An excessive loading of nutrients including point sources and non-point sources causes eutrophication. York Pond's watershed is 165.5 Ha with a relatively small watershed to lake ratio of 7.4. Its watershed is mostly forested with barely any human development. Such characteristics indicate that watershed runoff is most likely low in nutrients, although no chemical analysis was conducted. However, the fish hatchery located adjacent to York Pond, adds untreated effluent directly into it. Although no analysis of this effluent has been conducted for nutrient concentration, it is highly likely for fish hatchery effluent to be enriched with nutrients and oxygen.

A DO curve in a eutrophic lake typically decreases steadily to zero before the sediments of the lake is reached. This is usually due to the high mortality rates of phytoplankton that accumulate near the bottom. In York Pond, there is strong evidence that decomposition is taking place between the 3 m to 4 m interval. At 4 m depth, there was a peak of chlorophyll with a corresponding DO minimum of 2.13 mg L^{-1}. A maximum specific conductivity at this depth supports the idea that the chlorophyll peak is due to decomposing inactive phytoplankton because decomposition releases ions into the water column. Also, the pH decreased at this depth.
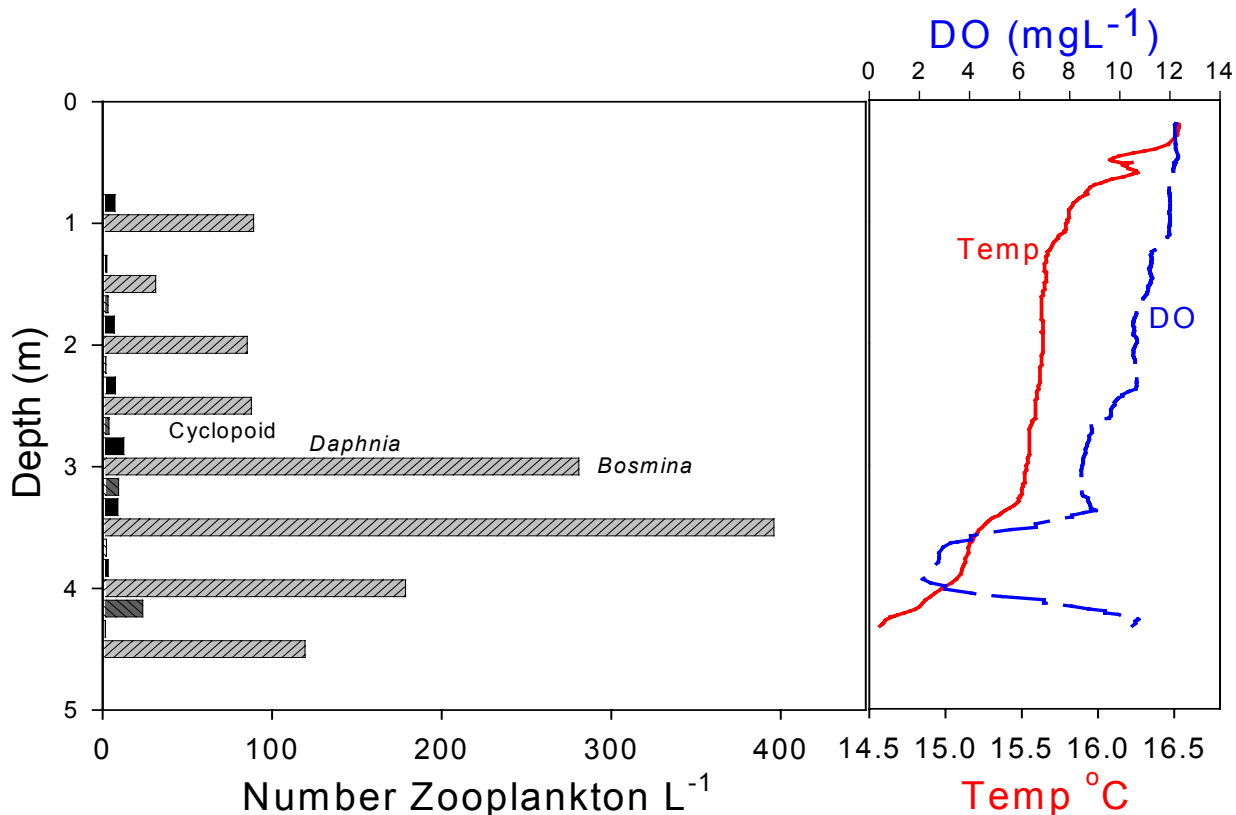

Fig. 10. York Pond's (9-25-03) zooplankton percentage profile obtained from the discrete sampler along with the temperature and dissolved oxygen curve. Cyclopoid are represented by diagonal top left to bottom right light gray lines, *Daphnia* by solid black, and *Bosmina* by solid light gray.

Active chlorophyll usually increases pH because it removes CO_2 from the water column. The less CO_2 available, the less carbonic acid can be formed. York Pond's DO curve slowly decreased down to about 2 mg L^{-1}, but never reached zero. This curve actually increases towards the bottom of the lake. My hypothesis is that the stream input from the fish hatchery is the cause of this anomaly. The stream's temperature was measured to be 15.3°C, which was the temperature of the water column directly above when the DO begins to drop. The density

differences between the effluent and the lake would likely cause this input to flow to the bottom of the lake. It is unlikely that this increase of DO is due to groundwater inputs because groundwater is usually low in DO (Allan, 1995). Furthermore, the lake sediments were tested for DO and were nearly zero.

In a water body with one of the highest concentrations of chlorophyll in the state, there were very few *Daphnia*. *Daphnia* are effective filter-feeding zooplankton with the largest prey size range averaging between 1 μm to 50 μm
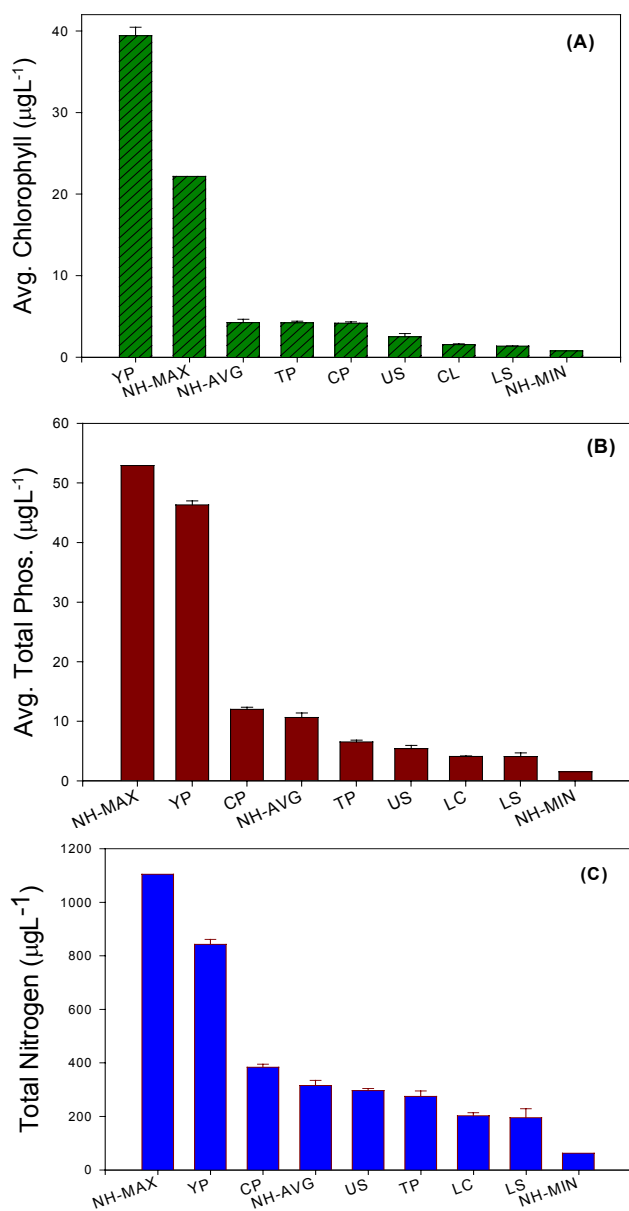

Fig. 11. Multi-graph lake comparisons including A) Chlorophyll, B) Total phosphorus, C) Total nitrogen (Haney & Ikawa, 200)
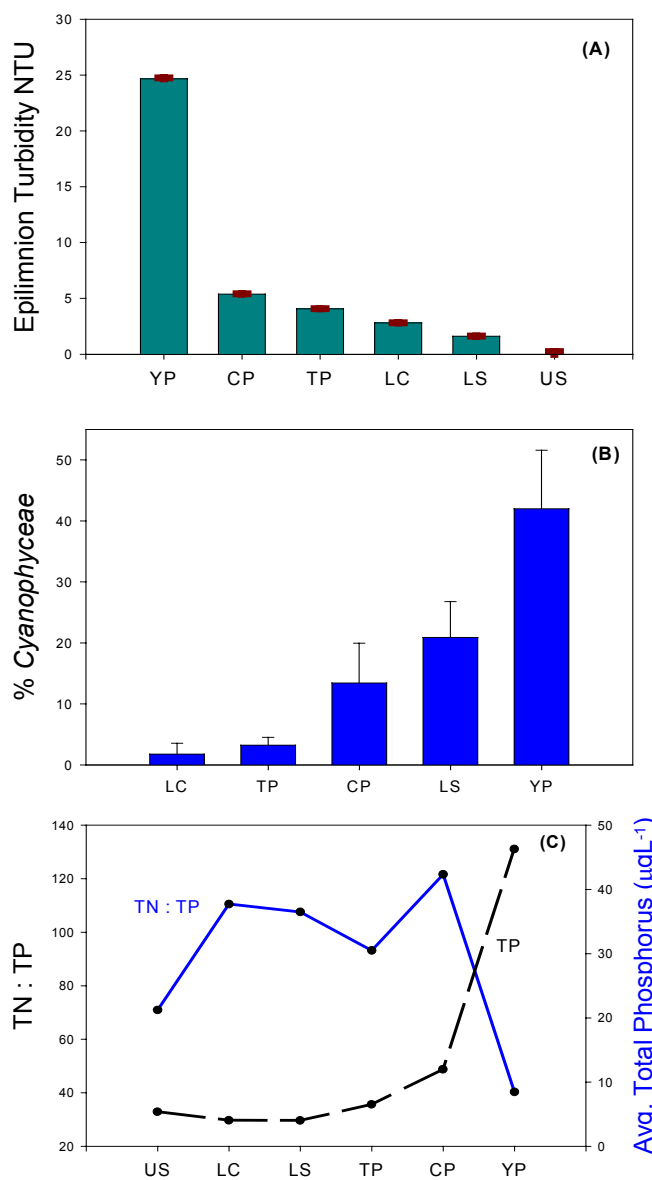

Fig. 12. Multi-graph lake comparisons including A) Epilimnetic turbidity, B) % *Cyanophyceae*, and C) TN : TP. Upper Sawyer data is unavailable for graph B.
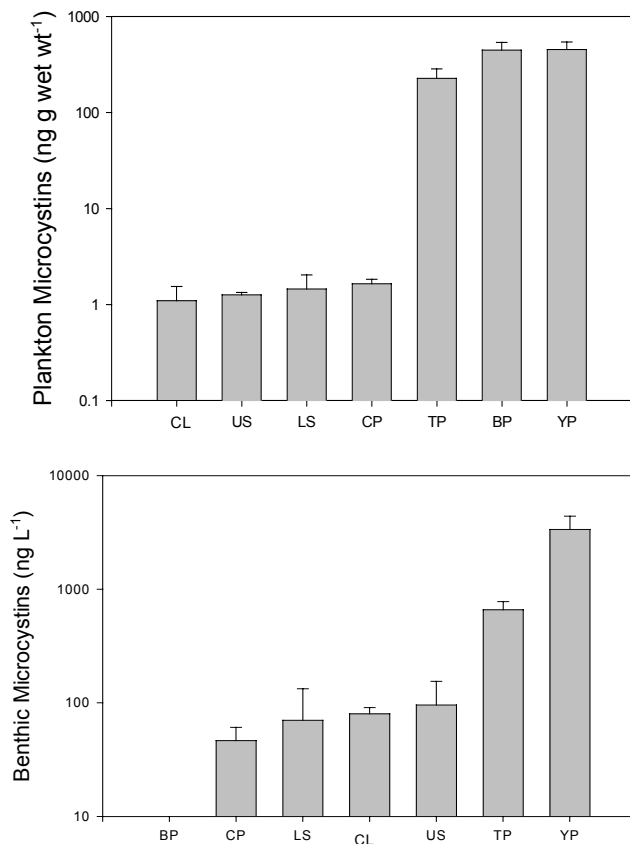
Fig. 13. Comparisons of microcystin concentrations in the plankton upper graph and in the benthos, lower graph. Note the log scale used in the lower graph. Barbadoes Pond is added to the plankton microcystin data set only.

dependent on the species (Lampert & Sommer, 1997) and have even been seen to ingest filamentous *Cyanophyceae* (Gliwicz, 1969). Abundant supply of food and lack of vertebrate predators would suggest this pond to be a suitable habitat for *Daphnia*. However, it has been observed that certain types of filamentous algae can inhibit the feeding rate, survival, and reproduction of *Daphnia* (Infante & Abella, 1985). York Pond has the highest percentage of *Cyanophyceae,* primarily *Anabaena*. Hardy & Gunther (1935) suggest that *Daphnia* are inhibited by exposure to bluegreen bacteria because of the potentially harmful biotoxins these organisms produce. However, Gliwicz et. al (1980) found that *Daphnia* narrow the gape between the margins of the carapace, limiting the larger sized netplankton from entering when placed in high concentrations of bluegreen bacteria. Such a response would suggest that this is not a chemical inhibition because large

filamentous bacteria and small unicellular species both produce biotoxins. It is more likely that *Anabaena* interferes with *Daphnia's* feeding effectiveness, forcing *Daphnia* to expend more energy, thus reducing its overall fitness. Claska & Gilbert (1998) found evidence supporting both previous studies, which showed *Daphnia pulex* to be inhibited by both the biotoxins and the feeding disruption. It is likely both stresses account for the small percentage of *Daphnia* in York Pond. This would then favor zooplankton with a smaller filtering range that avoid large filamentous algae and only be subject to one stress. Correspondingly, York Pond is dominated by *Bosmina,* a cladoceran that selects intermediate sized food particles (Lampert & Sommer, 1998).

The effects of biotoxins moving through the food web are not well documented. Cyanotoxins such as microcystins are potentially harmful biotoxins found in York Pond. York Pond has both the highest dominance *Anabaena* and *Microcystis* in the net plankton community, as well as, the highest concentration of benthic and planktonic microcystin. Microcystins are serious liver toxins and may cause liver tumors and complete liver failure in mammals, including humans (Best & Eddy, 2003). However, the effects on the aquatic ecosystem are poorly known. A few studies have shown harmful effects on fish populations (Li & Liu, 2003) and other phytoplankton (Infante & Abella, 1985). York Pond appears to have a relatively simplified food web (Fig. 16), which reflect in part the high concentration of biotoxins in this lake.

Fish communities may also be impaired by the phytoplankton abundance. Fish, particularly trout, require high levels of dissolved oxygen. Although phytoplankton produces oxygen, they also produce biotoxins, which can inhibit fish communities (Li & Liu, 2003). Furthermore, phytoplankton decomposition can lead to anoxic waters spreading vertically. York Pond is relatively shallow, thus making it more susceptible to having completely anoxic waters, especially in the summer and the winter. Also, fish mainly rely on visual hunting of prey. The Secchi disk transparency was the shallowest of all the lakes studied and even created a new minimum value for the NH fifty lake study

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fish. It is likely that York pond may have summer and winter fish die offs due to hypoxia, although it is not known which fish currently inhabit the lake.

Water transparency is usually dependent upon biotic factors such as chlorophyll concentrations. Our regression analysis (Fig. 14) supports this statement because a significant relationship ($p<0.01$) shows that as Secchi disk depth decreases, chlorophyll levels increase. Secchi disk can be a useful surrogate of chlorophyll in lakes with low dissolved color. The method is simple and inexpensive when compared to chlorophyll measurements. In York Pond's profile (Fig. 3), the turbidity is accounted for by chlorophyll since both curves vary with each other until the bottom of the lake where decomposition takes over. Turbidity is the measure of the scattering of light, and is accounted for by mainly phytoplankton cells in
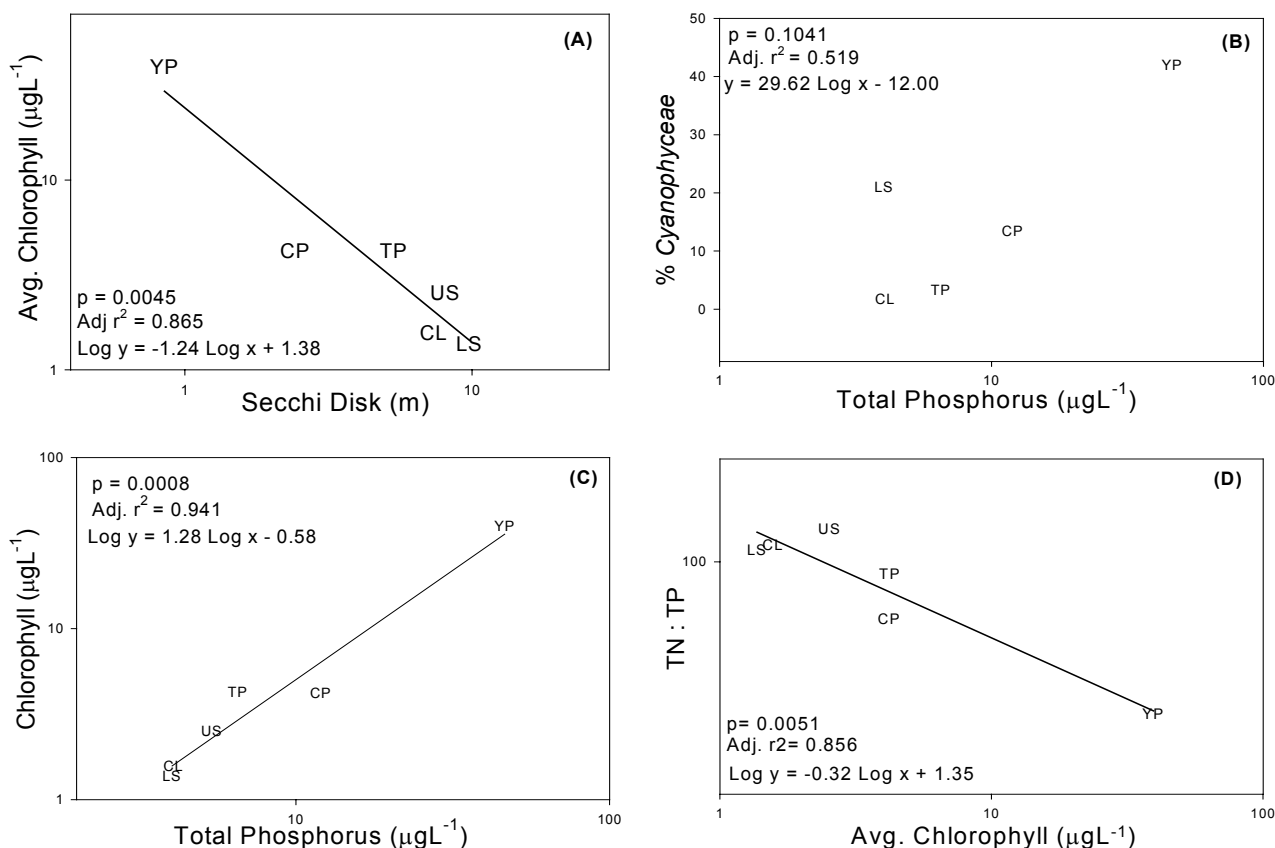

Fig. 14. Regression analysis for lake comparison. A) Chlorophyll vs. Secchi disk, B) % *Cyanophyceae* vs. Total phosphorus, C) Chlorophyll vs. Total phosphorus and D) TN : TP vs. Chlorophyll. Upper Sawyer data for % Bluegreen algae is unavailable.

York Pond. Therefore, Secchi disk proves to be a significant predictor of chlorophyll concentration in these lakes. Cedar Pond may not follow this relationship because its high dissolved color may significantly (81.03 ($\pm$ 0.87) CPU) affect water transparency.

Chlorophyll concentration can be a useful predictor of other lake characteristics such as nutrient concentration. Total phosphorus is primarily considered the limiting nutrient in freshwater ecosystems (Forsberg & Ryding, 1980). We found a significant relationship between average chlorophyll and total phosphorus ($p<0.05$) in the six lakes studied. This would suggest that chlorophyll levels are dependent upon phosphorus concentrations. We then hypothesized that the higher phosphorus levels would shift phytoplankton communities towards *Cyanophyceae*. Although this relationship was not significant ($p = 0.104$), it did show a trend. Lower Sawyer appeared to weaken this
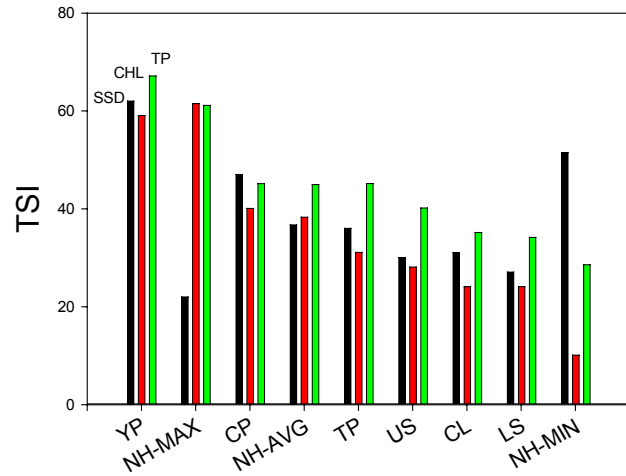

Fig. 15. Carlson's trophic state index for all lakes studied and NH lakes (Mean, Minimum, & Maximum; Haney & Ikawa, 200)). Secchi Disk (SSD) is represented by solid black, total phosphorus by black with white dots, and chlorophyll (CHL) by white and black dots.

relationship because of its high percentage (not density) of *Cyanophyceae* in an oligotrophic lake.
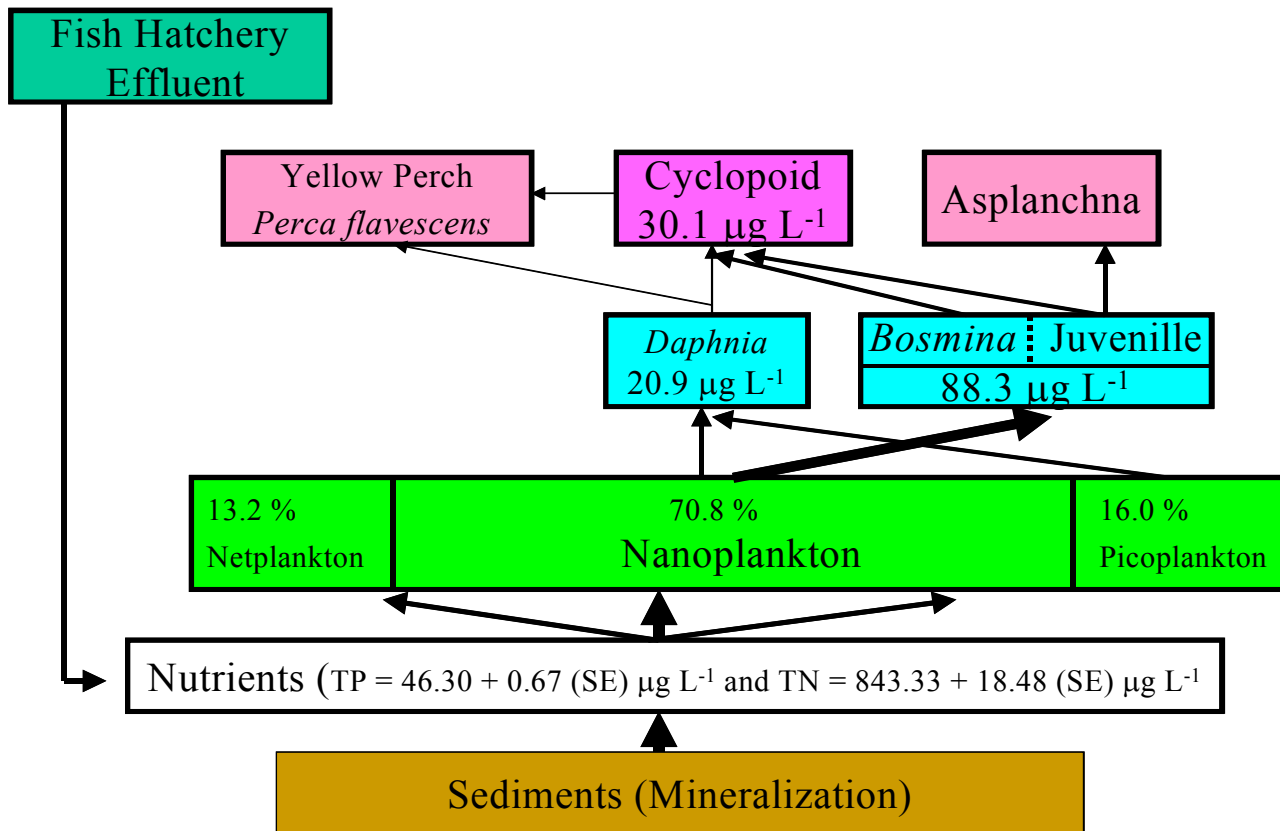

Fig. 16. A preliminary model of York Pond's pelagic food web constructed on the basis of nutrient, zooplankton and phytoplankton samples from this study.

Table 4. Proposed trophic state from Forsberg & Ryding, 1980

	Chlorophyll *a* (µgL^{-1})			Total phosphorus (µgL^{-1})			Transparency (m)		
	Oligo	Meso	Eutro	Oligo	Meso	Eutro	Oligo	Meso	Eutro
Rast & Lee, 1978	<2	2 - 6	>6	<10	10 - 20	>20	>4.6	2.7 - 4.6	>2.7
Sakamoto, 1966	<2.5	2.5 - 5	>5						
Carlson, 1977	<2.5	2.5 - 6.5	>6.5	<12	12 - 24	>24	>4	2 - 4	<2
Forsberg & Ryding, 1980	<3	3 - 7	>7	<15	15 - 25	>25			
NAS 1972	<3.5	3.5 - 10	>10						
Dobson et al. 1974	<4	4 - 9	>9				>4	2.5 - 4	<2.5
US EPA 1974	<7	7 - 12	>12	<12	10 - 20	>20	>3.7	3.7 - 2	<2

Summary - York Pond clearly fits the classification of a eutrophic lake based on its total phosphorus, chlorophyll and Secchi disk depth measurements. The most likely cause of eutrophication is the untreated effluent from the fish hatchery, although no chemical analysis has been accomplished. As a result of eutrophication, *Anabaena* and *Bosmina* are the two most dominant organisms across all trophic levels tested. *Anabaena* appears to be inhibiting *Daphnia*, fish communities and possibly other phytoplankton. *Anabaena* has been shown to paralyze *Chlorophyceae*, which causes algae to settle out and frees up competition (Kearns & Hunter, 2001). Impacts from eutrophication in York Pond may have lowered the biodiversity of the lake. Further sampling should include testing of the concentration of nutrients and organic matter in the fish hatchery effluent. This effluent is the most obvious source of excess phosphorus and nitrogen entering York Pond. Possible mitigation measures should be considered, including retention ponds and diversion of effluent through wetlands. Future studies should further investigate the impacts of fish hatchery effluent on a lake and stream ecosystem. It is also important that further examination be made on the presence of biotoxins such as microcystins in the food web of York Pond, as biotoxins are becoming a growing concern due to their effects on organisms such as fish, birds and mammals, including humans (Gupta *et al.*, 2003).

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