

Pilot survey of Canadarago Lake and its tributaries, 2008

Carter Bailey¹ and M.F. Albright

INTRODUCTION

Canadarago Lake (N42°48.9', W75°00.4'; 390 m (1280ft) above sea level (Harr et al. 1980)) is a dimictic lake of glacial origin located in the Towns of Richfield, Otsego and Exeter, Otsego County, NY. It has a maximum depth of 13 m (43 ft), a surface area of 760 ha (1,877 ac) and a watershed of 175 km² (43,240 ac). Together with its sister lake, Otsego, they form the headwaters of the Susquehanna River. The drainage basin of the Canadarago watershed is largely agricultural in nature, with some wooded hill tops and wetlands. This, along with its shallow depth, is believed responsible for Canadarago's eutrophic character. The Richfield Springs wastewater treatment plant is located on Ocquionis Creek, discharging treated effluent 0.8 km (0.5 mi) upstream of the lake (Figure 1). Alum precipitation has been instituted since 1973 for phosphorus removal (Harr et al. 1980).

Concerted efforts to manage Canadarago Lake and its watershed have been limited, as have the availability of funds to meet that end, largely due to the lack of a scientifically-based management plan. Funds have recently been committed for the development of a "State of the Lake" report, which is intended to serve as the basis for the development of a management plan. That work, to be conducted by the Biological Field Station, is slated to commence in January 2009. This 2008 summer pilot survey was intended to provide insight into the nature of the lake and its watershed so that a more effective 2009-2010 work plan could be developed. The work summarized here included monitoring temperature, dissolved oxygen, conductivity, pH, nutrients, and fecal coliform in the lake, its tributaries and outlet (Oaks Creek). Secchi disk transparency and surveys of zooplankton and aquatic plants were conducted on the lake as well. Suspended sediment samples were also collected from the tributaries and the outlet.

MATERIALS AND METHODS

Canadarago Lake and corresponding tributary sampling sites were visited once a week from 11 June to 23 July. A summary of the sampling sites is given in Table 1. Tributary and lake sampling sites are shown in Figure 1. The lake was sampled in profile at the deepest spot encountered (approximately 12 m; 40 ft) (Figure 1). Transparency was measured with a standard Secchi disk. Temperature, dissolved oxygen, conductivity, and pH was collected in 1 m intervals on the lake and once at each tributary sampling spot using a Eureka Amphibian/Manta[®] multiporobe digital microprocessor which had been calibrated according to manufacturer's instruction immediately prior to use (Eureka Environmental Engineering 2004). Lake samples were collected for nutrients (ammonia, nitrite+nitrate, total nitrogen and total phosphorus) in 4 m depth intervals using a

¹ Canadarago Lake Improvement Association Assistantship. Current affiliation: SUNY ESF.

Kemmerer water sampler from the surface to just off bottom. Samples for nutrient analysis were also collected from seven tributary sites using 125 mL bottles. Where appropriate, sampling involved using a bucket and rope in a tossing fashion in order to gather a sample from the middle of the creek, where the water was less stagnant. A summary of methodologies employed for chemical analyses are given in Table 2.

Fecal coliform samples were collected using 1 L Pyrex® glass bottles from each of the tributary and lake sites, excluding the profile sampling site (Figure 1). Samples were iced during transportation until analysis. Laboratory analysis followed the membrane filter technique (APAHA 1989). Sample volumes were selected in attempt to produce 20-80 fecal coliform colonies per filter. These set volumes, ranging from 5-200 mL, were low-pressure vacuum filtered through 0.45-micron membrane filters. Filters were then placed and sealed inside sterile petri dishes on absorbent pads saturated in F C Base by Bacto® growth media. All cultures were incubated in a water bath at 44.5°C for 24±2 hours. Colonies were counted after incubation period and reported as colonies per 100 mL.

Tributary & Outlet Sampling Sites

<u>Oaks Creek</u>	<u>Abbreviation: OK. C.</u>
East of the Village of Schuyler Lake on County Route 22; sampled north of bridge.	
<u>Herkimer Creek</u>	<u>Abbreviation: HK. C.</u>
North of the Village of Schuyler Lake on State Route 28; sampled east of bridge.	
<u>Hyder Creek</u>	<u>Abbreviation: HY. C.</u>
South of Dennison Road (NYSP boat launch access road) on State Route 28; sampled west of bridge.	
<u>Trout Brook (Mink Creek)</u>	<u>Abbreviation: T.B.</u>
Just north of Canadarago Lake on Elm Street Extension; sampled east of bridge.	
<u>Ocuionis Creek North</u>	<u>Abbreviation: O.C. 1</u>
The beginning of Elm Street Extension, just south of Bronner Street; sampled south of bridge.	
<u>Ocuionis Creek South</u>	<u>Abbreviation: O.C. 2</u>
End of North Shore Lane, must walk to site through home owner's yard; sampled near mouth of creek.	
<u>Waste Treatment Plant</u>	<u>Abbreviation: W.T.</u>
End of Bloomfield Drive, through gate and into plant, sampled from effluent pipe	

Lake Sampling Sites

<u>Lake Profiling Site</u>	<u>Abbreviation: None</u>
Deepest spot encountered (11.48-12.45 m; 37.5-41 ft). (N 42° 49.339' W 75° 00.032')	
<u>Canadarago 1:</u>	<u>Abbreviation: C.L. 1</u>
West side of Lake, north of boat launch. (N 42° 49.776' W 75° 00.398')	
<u>Canadarago 2:</u>	<u>Abbreviation: C.L. 2</u>
West side of Lake, south of boat launch. (N 42° 49.788' W 75° 00.414')	
<u>Canadarago 3:</u>	<u>Abbreviation: C.L. 3</u>
West side of Lake, southern most lake sampling spot. (N 42° 48.870' W 75° 00.901')	

Table 2. Descriptions and locations of sampling sites on Canadarago Lake's tributaries and outlet.

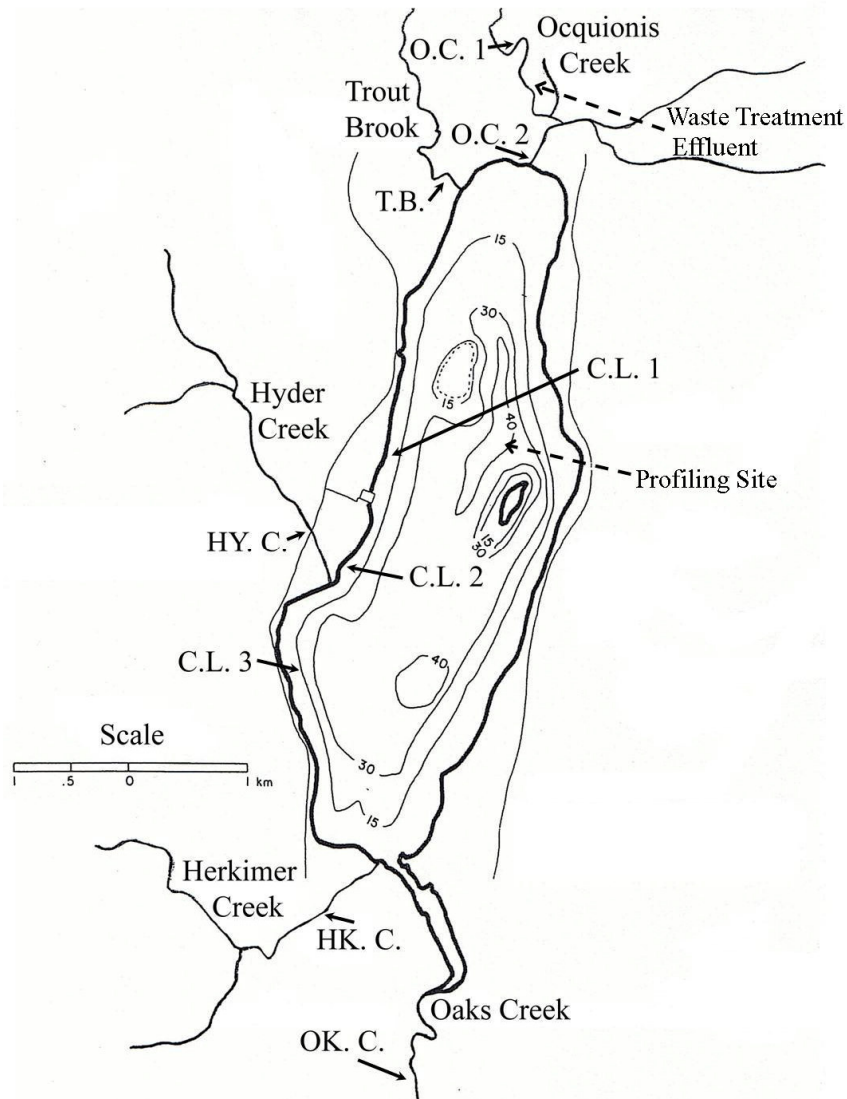


Figure 1. Canadarago Lake, New York, showing location of sites sampled over summer 2008.

Sediment samples were collected along with fecal coliform using the same 1 L Pyrex[®] glass bottle per tributary site. Total suspended and inorganic sediment samples were run according to standard methods (APHA 1989). Prior to sample processing, Whatman[®] GF/C 47mm filters were prepared by low-pressure vacuum filtering distilled water through them and then placing in numbered aluminum planchets. These filters were then dried in an oven at 105°C for 24±2 hours. After cooling, the filters were massed and recorded. A recorded volume water sample (500-900 mL) was passed through each filter which then were dried in an oven at 105°C for 24±2 hours. The mass was recorded, allowing for the calculation of total suspended sediment. The filters were then heated to 550°C for an hour, removed from the oven, cooled and massed. This allowed for the calculation of organic versus inorganic suspended sediment in each sample. All sediment data were reported in mg/L.

Parameter	Sample volume	Preservation	Method	Reference
Total Phosphorus-P	10 ml	H ₂ SO ₄ to pH<2	Persulfate digestion followed by single reagent ascorbic acid	Liao and Marten 2001
Total Nitrogen-N	5 ml	H ₂ SO ₄ to pH<2	Cadmium reduction method following peroxodisulfate digestion	Pritzlaff 2003; Ebina et. al 1983
Nitrite+Nitrate-N	10 ml	H ₂ SO ₄ to pH<2	Cadmium reduction	Pritzlaff 2003
Ammonia-N	10 ml	H ₂ SO ₄ to pH<2	Phenolate	Liao 2001

Table 1. Summary of laboratory methodologies employed.

Zooplankton were collected using a 20 cm, weighted 63 µm mesh plankton net. The net was lowered 4-5 m (13-16 ft) by an attached nylon cord and retrieved. The sample was placed in a plastic bottle on site and preserved with 70% ethanol after returning to the laboratory. One-milliliter sub-samples were viewed on a Sedgwick-Rafter cell under a compound microscope having digital imaging and analysis capabilities, where zooplankton were measured and identified (Pennak 1989). The relative abundance of each genus was calculated.

The Point Intercept Rake Toss Relative Abundance Method (PIRTRAM) (Lord et al. 2006) was used to evaluate species collected from eleven sites around the lake (Figure 2). Two garden rake heads were attached by weld in a back-to-back fashion to form a double sided rake. This was connected to a 10 m (33 ft) long nylon cord. At each of the eleven sites the rake apparatus was tossed into the water and then retrieved after it had settled at or below submergent plant level. Plant species present at each site were recorded. Emergent plants were collected by hand.

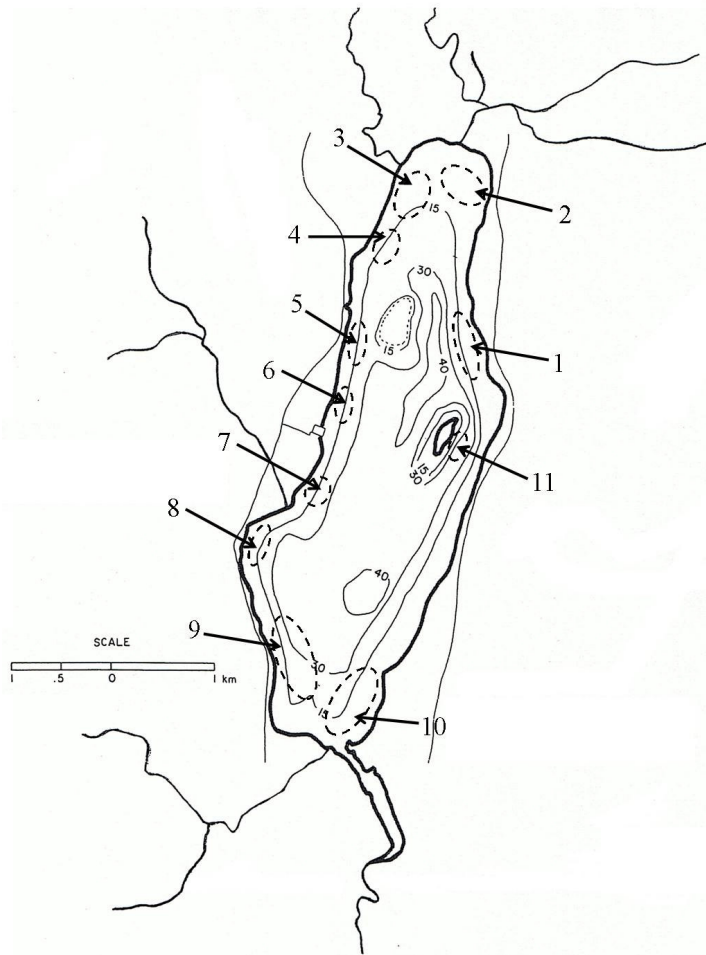


Figure 2. Canadarago Lake, New York, showing sites sampled for aquatic plants, summer 2008.

RESULTS AND DISCUSSION

Lake Temperature

Figure 3 shows temperature profiles over the course of the study. Thermal stratification was well evident at the onset of sampling. The warmest temperature recorded (24.49°C) was observed at the surface through 1 m on 9 July. The coldest temperature (10.49°C) was recorded on 16 June at the bottom.

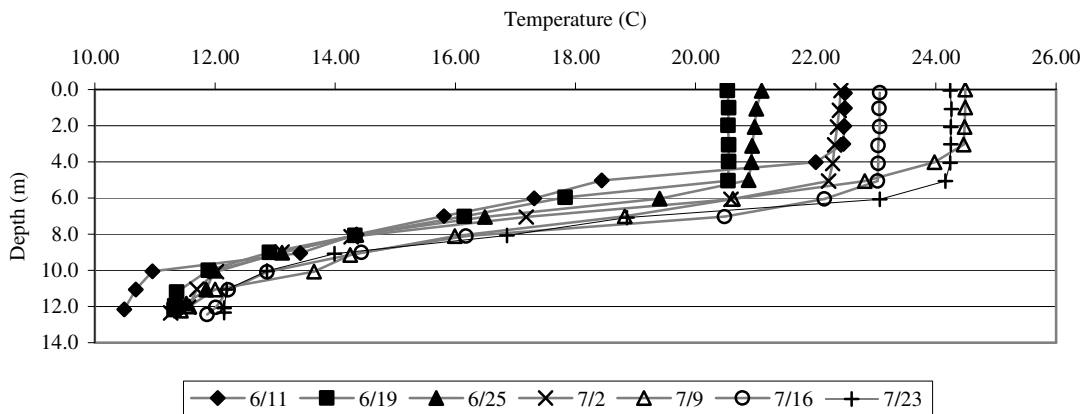


Figure 3. Canadarago Lake 2008 temperature data shown in depth profile, by date.

Dissolved Oxygen

Figure 4 shows dissolved oxygen profiles over the summer. Hypolimnetic concentrations were substantially depressed by mid June and gradually declined further by late July. Near bottom conditions were anoxic by 10 July. These conditions are consistent with eutrophic conditions, with hypolimnetic oxygen being depleted as a result of algal respiration and/or decomposition.

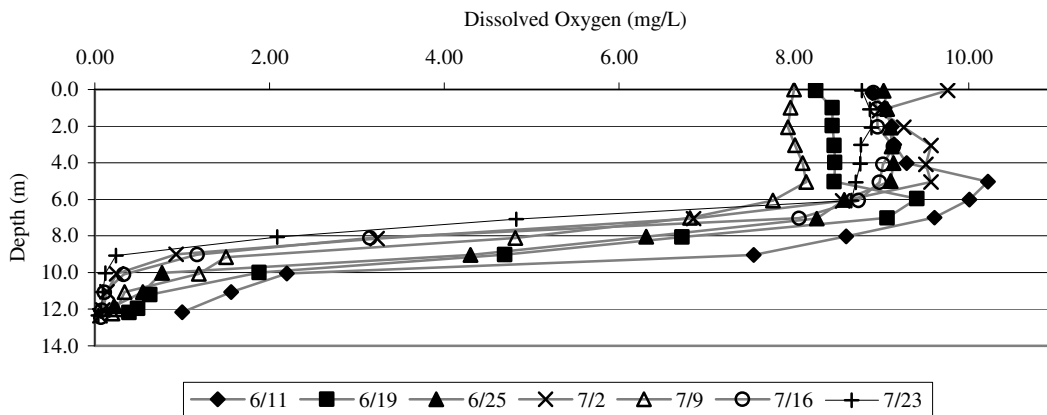


Figure 4. Canadarago Lake 2008 dissolved oxygen data shown in depth profile, by date.

Conductivity

Conductivity (an indirect measure of ions in solution) values ranged from 0.320 ms/cm from a surface sample on 11 June to 0.386 ms/cm in the bottom waters on 16 July (Figure 5). The increase in conductivity with depth is likely due to byproducts of bacterial decomposition and/or the release of ions from the sediments resultant of a reducing (anoxic) environment (Figure 4).

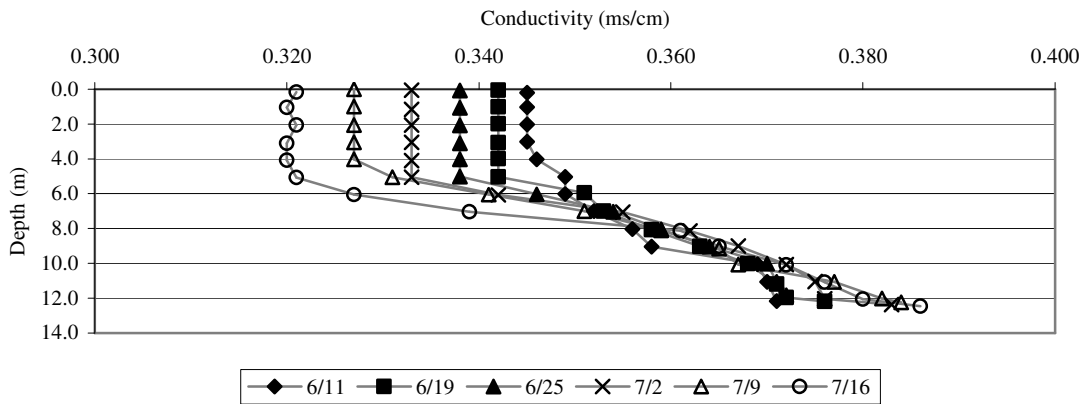


Figure 5. Canadarago Lake 2008 conductivity data shown in depth profile, by date.

pH

pH levels of the lake were quite high, ranging from 7.58 at the bottom on 19 and 25 June to 8.66 at the surface on 2 July. A reduction in pH in hypolimnetic waters is common, as decomposition results in the addition of carbonic acid (Wetzel, 2001). The lake's basic nature (pH > 7) is a function of its limestone rich watershed (Harr et al. 1980), and as a result the lake is well buffered against acid deposition and accompanied pH swings.

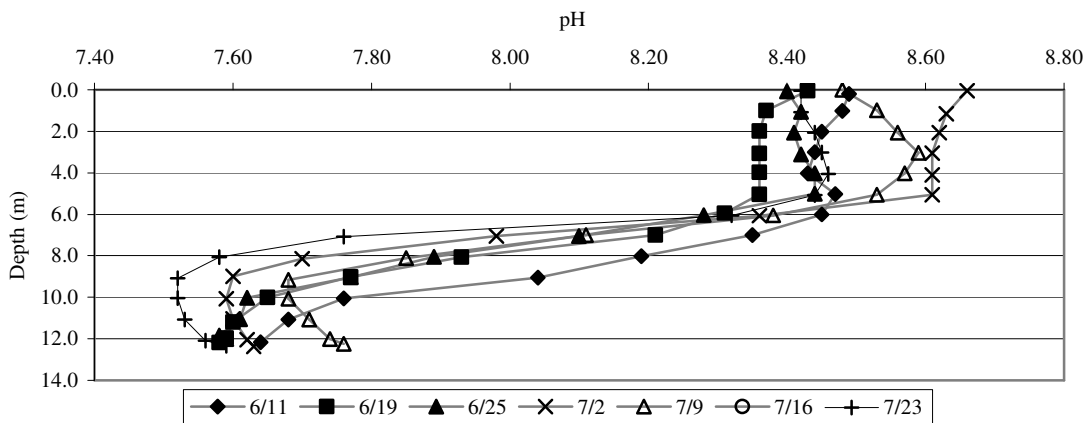


Figure 6. Canadarago Lake 2008 pH data shown in depth profile, by date.

Nutrients

Profiles of total phosphorus are provided in Figure 7. Internal loading of phosphorus was evident, being somewhat modest at the onset of the monitoring program and becoming more pronounced over the course of stratification. This situation is typically encountered in lakes where oxygen-deprived waters overlies sediments. The resultant reducing environment brings about the release of phosphorus and iron

associated complexes (i.e., Marsden et al. 1989). Generally, the dynamics of phosphorus observed in 2008 was similar to that described in 1968 (Harr et al. 1980) and in 2001 (Harman et al. 2002). However, mean total phosphorus documented in epilimnetic waters (surface to 4 m) during this study, averaging 9.5 ug/l, were substantially lower than that observed in 2001 (19 ug/l).

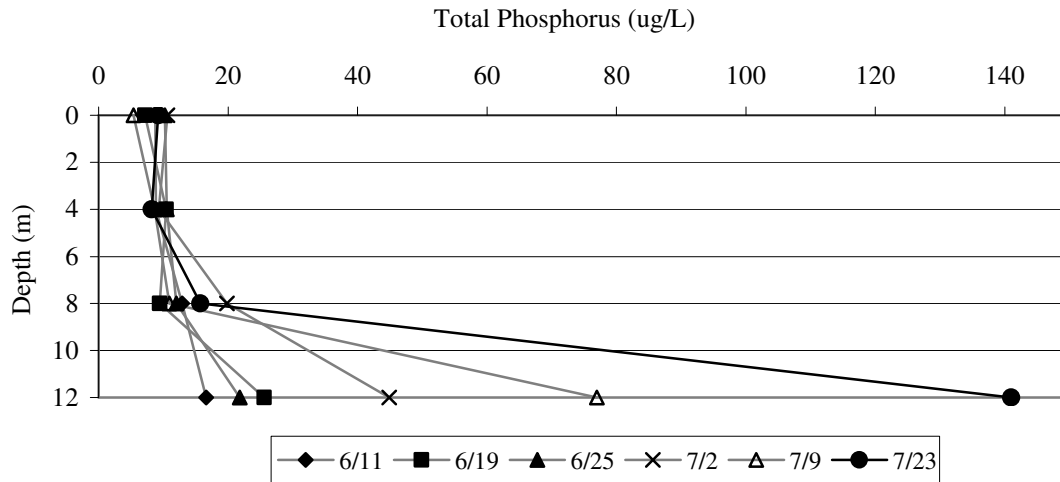


Figure 7. Canadarago Lake 2008 total phosphorus data shown in depth profile, by date.

Ammonia profiles, presented in Figure 8, follow trends similar to those of total phosphorus. Concentrations are comparatively low in epilimnetic waters and increase below the thermocline; deep water concentrations likewise increased over the course of the summer. This is expected, as nitrates, the soluble oxidized fraction, were reduced to ammonia concurrent with the decline of oxygen (Figure 4).

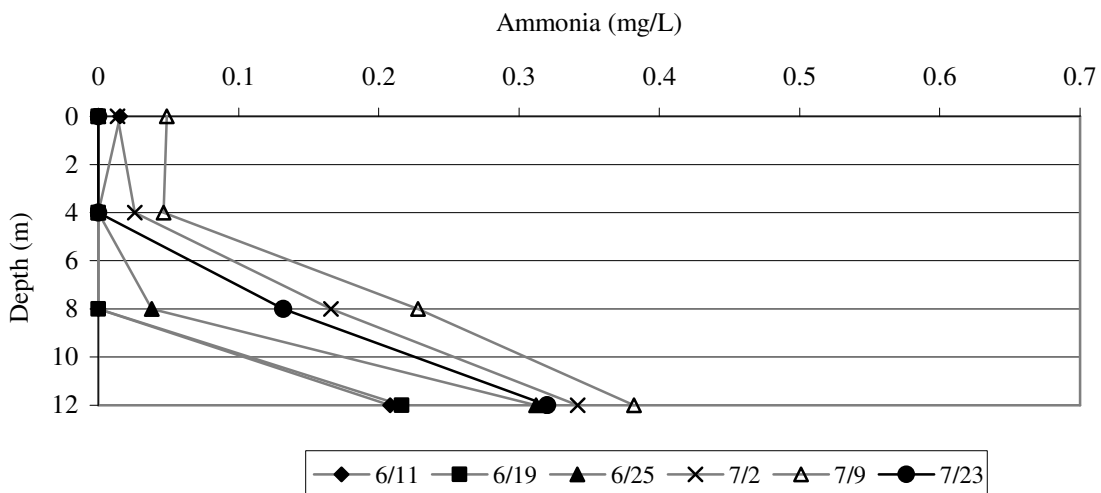


Figure 8. Canadarago Lake 2008 ammonia data shown in depth profile, by date.

Profiles of nitrate+nitrite are given in Figure 9. On each given sampling date, concentrations were homogeneous through the epilimnion, increase below the thermocline, then decline near the bottom. A temporal decline of mean column-wide concentrations was evident.

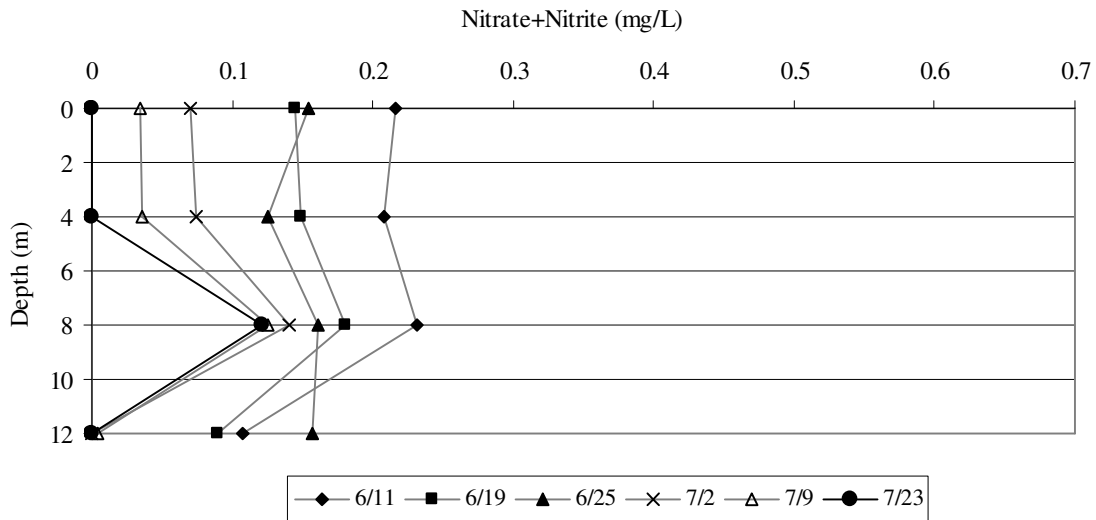


Figure 9. Canadarago Lake 2008 nitrate+nitrite data shown in depth profile, by date.

Profiles of total nitrogen, the sum of ammonia, nitrite+nitrate and organic nitrogen, are given in Figure 10. Concentrations were relatively homogeneous through the water column at the onset of the study. Over time, concentrations declined in the epilimnion and increased in the hypolimnion, probably related to the settling of nitrogen bound in organic particles, namely dead and senescent algal cells.

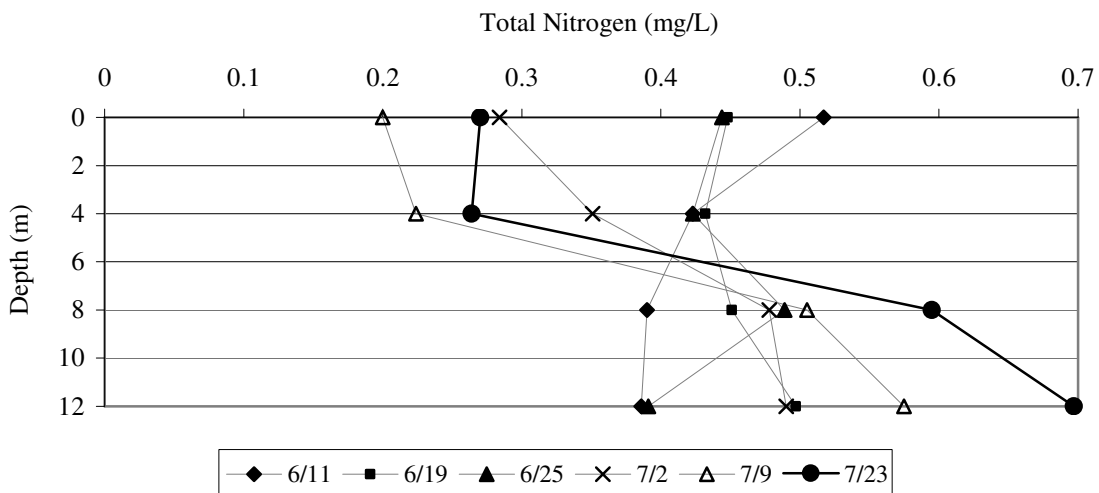


Figure 10. Canadarago Lake 2008 total nitrogen data shown in depth profile, by date.

Past studies have been conflicting as to whether algal production is limited by phosphorus or nitrogen. Harr et al. (1980) suggested phosphorus limitation. The total nitrogen:total phosphorous (TN:TP) ratio in algal biomass is generally 7-10 (Vallentyne, 1974). Phosphorus limitation is expected when in-lake concentrations exceed that ratio. Work conducted in 2001 revealed a nitrite+nitrate:total phosphorus rate of about 5 (Harman et al. 2002), suggesting nitrogen limitation (though ammonia and organic nitrogen were not considered). Work in 2008 revealed a TN:TP of about 30, suggesting phosphorus limitation. (This difference from previous data results not only from the inclusion of ammonia and organic nitrogen, but also from a marked decrease in total phosphorus in the epilimnion). Internal loading of phosphorus, due to the deoxygenation of waters overlying sediments (Marsden et al. 1980), was evident throughout the sampling dates, as it was in 1968 (Harr et al. 1980), and 2001 (Harman et al. 2002)). However, the loss over time of bioavailable nitrogen (namely, nitrate) over the course of the summer would imply limitation by that nutrient.

Secchi disk transparency

Water transparencies ranged from 4.50 m on 9 July 9 to 7.50 m on 11 June. Increased transparencies since 2001, when Secchi transparency averaged 2.5 m (Harman et al. 2002), are most likely due to the introduction and establishment of a zebra mussel (*Dreissena polymorpha*) population in the lake. The introduction is estimated to have occurred between 2000 and 2001, based on the size of the largest individuals collected in 2002 (Horavath and Lord 2002). Filtration by zebra mussels has long been recognized to lead to increased clarity (D'Itri, 1997).

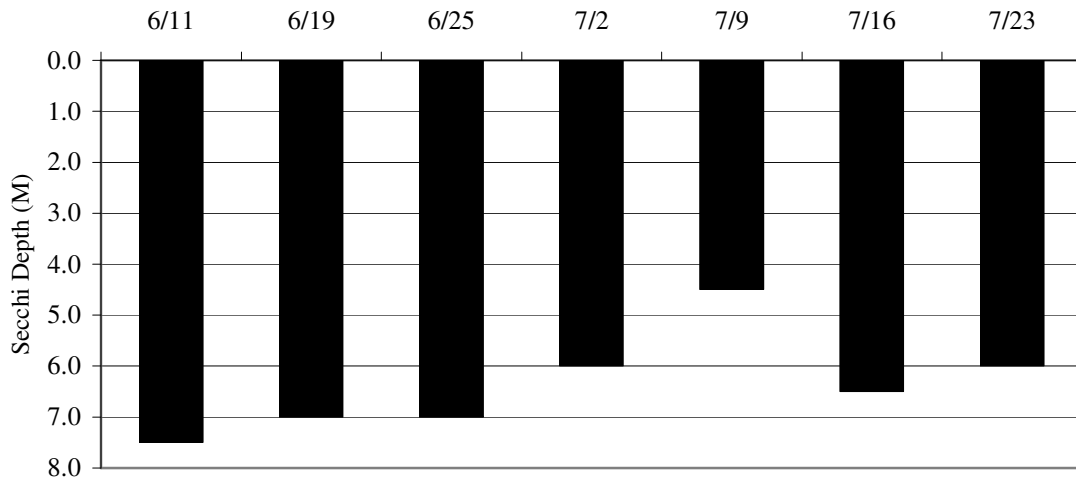


Figure 11. Canadarago Lake 2008 Secchi transparencies collected at lake profiling site.

Fecal coliform

Fecal coliform densities were quite variable at two of the three lake sites monitored (Figure 12; see Figure 1 for site locations), indicated by standard error bars for sites 1 and 2. Potential sources of coliform bacteria are wildlife (such as waterfowl), agricultural runoff or inadequately treated household sewage. The water is well below 200 colonies per 100 mL, which is the standard for safe swimming or any other full-body contact activities (Kaufmann & Cleveland 2008). More vigorous testing, including that associated with high lake level conditions, would be appropriate.

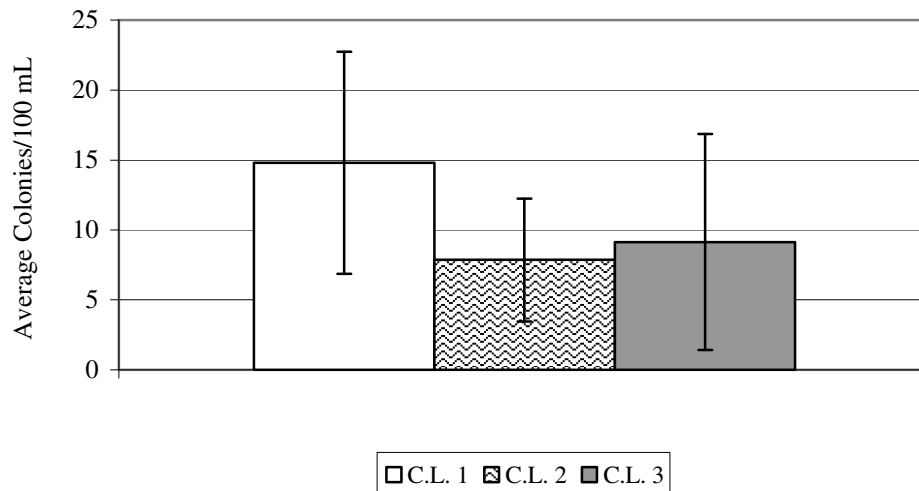


Figure 12. Average number of fecal coliform colonies (+/- 1 standard error) from 25 June to 23 July 2008, collected from the corresponding sites around the lake (see Figure 1).

Plants

The plant surveys conducted on 19 and 25 June yielded 17 plant species, submergent and emergent, from 11 different sampling sites around the lake (Table 3; see Figure 2 for collection sites). Current plant diversity is slightly lower than reported in a 1935 survey, when 21 submergent species were considered abundant, common or frequent (Muenscher, 1936), though that survey was likely more intense than that conducted during this study. The three most abundant species noted in 1935 (Muenscher, 1936), *Potamogeton pectinatus*, *Zosterella dubia*, and *Ceratophyllum demersum* are still dominant. Two non-native species were encountered, *Potamogeton crispus* and *Myriophyllum spicatum*. They are considerably common in early summer. Anecdotal evidence suggests that plant densities have increased in recent years. If so, this probably is related to higher water clarity resultant of the establishment of zebra mussels.

Scientific Name	Common Name	Site Collected
<i>Chara vulgaris</i>	Stonewort	1,4,5,6,7,8,9
<i>Potamogeton illinoensis</i>	Illinois pondweed	1,9
<i>Elodea canadensis</i>	Waterweed	1,9,11
<i>Myriophyllum spicatum</i> *	Eurasian water milfoil	1,5,9,11
<i>Ceratophyllum demersum</i>	Coontail	1,7,8,11
<i>Vallisneria americana</i>	Eel grass	1,4,7
<i>Potamogeton richardsonii</i>	Richardson's pondweed	2
<i>Zosterella dubia</i>	Yellow star flower	2,4,11
<i>Nuphar variegata</i>	Yellow pond-lily	4,3
<i>Potamogeton crispus</i> *	Curly pondweed	4,5,9
<i>Potamogeton amplifolius</i>	Bass weed	9
<i>Potamogeton pectinatus</i>	Sago pondweed	10
<i>Ranunculus longirostris</i>	White water buttercup	5,7,10
<i>Pontederia cordata</i>	Pickerelweed	5
<i>Schoenoplectus tabernaemontani</i>	Soft-stem bulrush	7,10
<i>Carex</i> spp.	Sage	10
<i>Typha latifolia</i>	Broadleaf cattail	10

* non-native species

Table 3. Submergent and emergent plants collected and observed from 19 and 25 June 2008 plant survey. See Figure 2 for site locations

Zooplankton

Zooplankton abundances, in relative abundance, are given in Table 4. Rotifers comprised the bulk of the community. The zooplankton community also included large-bodied crustaceans. Cyclopoid copepods and nauplius larva were found to account for about 20% of the community. Few cladocerans were found. Zebra mussel veligers, which are the free-swimming planktonic larva stage of the zebra mussel, were also recorded. Further work would be appropriate in order to better understand trends due to seasonal variation in zooplankton populations, but even with limited data, observations suggest little changes in the zooplankton community as outlined by Harr et al. (1980). Further efforts could reveal impacts by the exotic zebra mussel and alewife on the zooplankton community.

Taxa	Percent Abundance
Cladocera	
<i>Bosmina sp.</i>	>1%
<i>Daphnia sp.</i>	1%
Copepoda	
<i>Cyclopoid sp.</i>	12%
<i>Calanoid sp.</i>	1%
<i>Nauplius sp.</i>	8%
Rotifera	
<i>Asplanchna sp.</i>	4%
<i>Killicottia sp.</i>	1%
<i>Keratella sp.</i>	5%
<i>Polyarthra sp.</i>	54%
<i>Unknown 1</i>	2%
<i>Unknown 2</i>	9%
Other	
Zebra Mussel Veliger, <i>Dreissena polymorpha</i>	2%

Table 4. Zooplankton collected from the mid-lake sampling site on Canadarago Lake, 11, 19 and 25 June 2008, displayed in relative abundance.

Tributaries & Outlet

Temperature

Mean temperature of the tributary sites are given in Figure 13. Colder water temperatures tend to contain high levels of oxygen due to the exothermic nature of the dissolution, and hold fewer nutrients in solution (Kaufmann & Cleveland 2008). Mean temperatures ranged from 21.22°C at Oaks Creek (OK. C.) to 17.75°C at Ocquionis Creek North (O.C. 1). Temperatures were relatively consistent with the largest standard error being $\pm 1.13^\circ\text{C}$ occurring at Trout Brook (T.B.).

Dissolved Oxygen

Mean dissolved oxygen is graphed in Figure 14. Values ranged from 4.00 mg/L at Ocquionis Creek South (O.C. 2) to 8.07 mg/L at Hyder Creek (HY. C.). A trend of decreased dissolved oxygen was apparent between sites O.C. 1 and O.C. 2, with the downstream site containing little more than half of the dissolved oxygen of the initial sampling. Site O.C. 2 also contained the largest amount of variability with a standard error value of ± 0.90 mg/L. On some sampling days this site was recorded to be below 3 mg/L, which is often considered insufficient for many aquatic organisms (Harr et al. 1980). This could be due to the stagnant wetland nature of the water near the mouth of Ocquionis Creek.

Specific Conductance and pH

Mean specific conductance and pH for the tributaries are given in Figures 15 and 16, respectively. Both datasets imply relatively hard, well buffered waters.

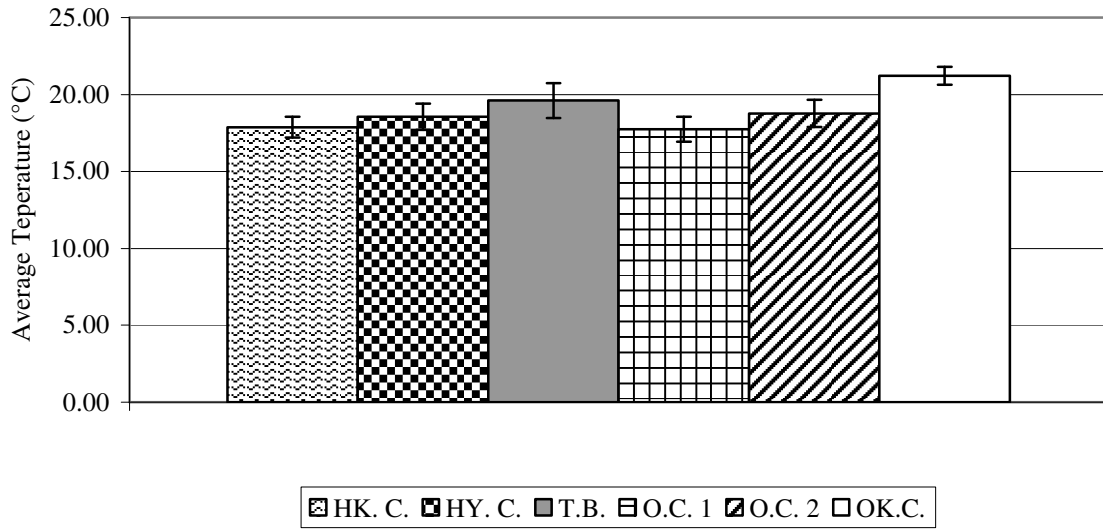


Figure 13. Mean temperatures for sampling sites of the Canadarago Lake watershed, summer 2008.

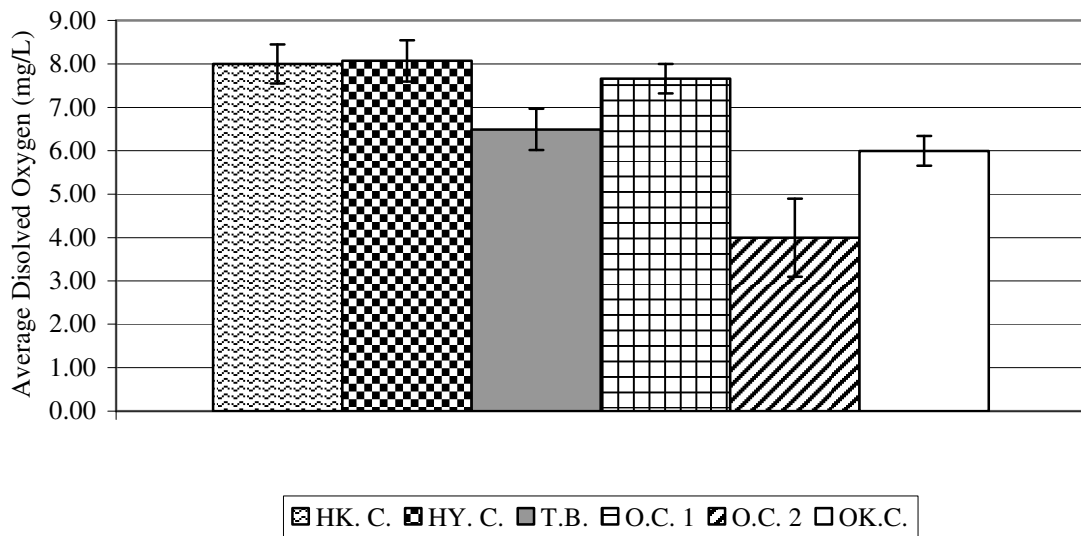


Figure 14. Mean concentrations of dissolved oxygen at each sampling site of the Canadarago Lake watershed, summer 2008.

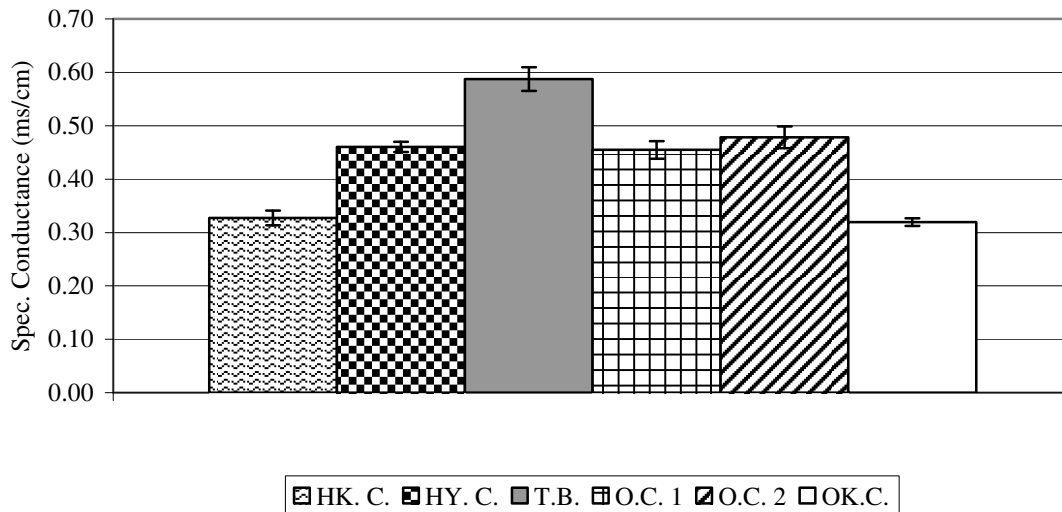


Figure 15. Mean specific conductance (ms/cm) at each sampling site of the Canadarago Lake watershed, summer 2008.

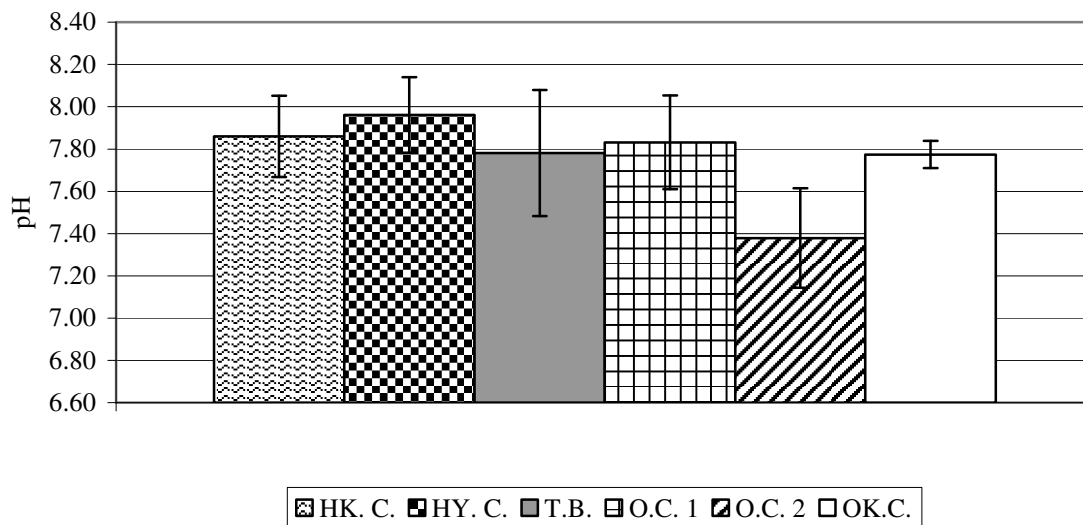


Figure 16. Mean pH at each sampling site of the Canadarago Lake watershed, summer 2008.

Nutrients

The mean concentrations of ammonia, nitrite+nitrate, total nitrogen and total phosphorus are given in Figures 15 through 18, respectively. (Note difference in scale between Figure 15 and Figures 16 and 17). The outlet of Ocquionis Creek has the highest concentrations of all nutrient fractions evaluated. Concentrations of total nitrogen and total phosphorus there are approximately twice that of the upper Ocquionis Creek site, implying either inputs by the wastewater treatment plant (concentrations of the effluent

are given in Table 5), or by release from the relatively stagnant stretch of the stream/wetland between the treatment plant and the mouth.

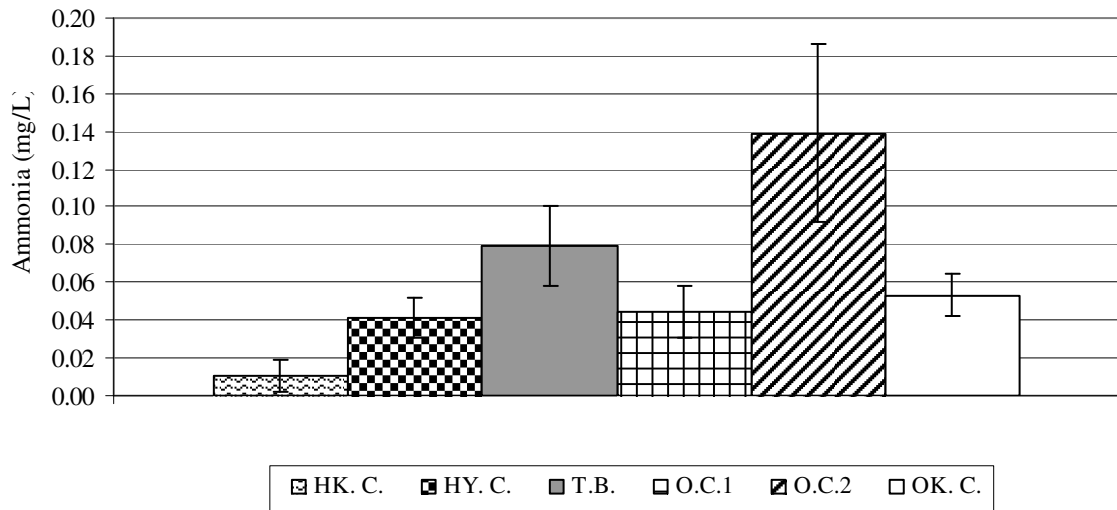


Figure 15. Mean ammonia concentrations at each sampling site of the Canadarago Lake watershed, summer 2008. Note different scale from Figures 16 and 17.

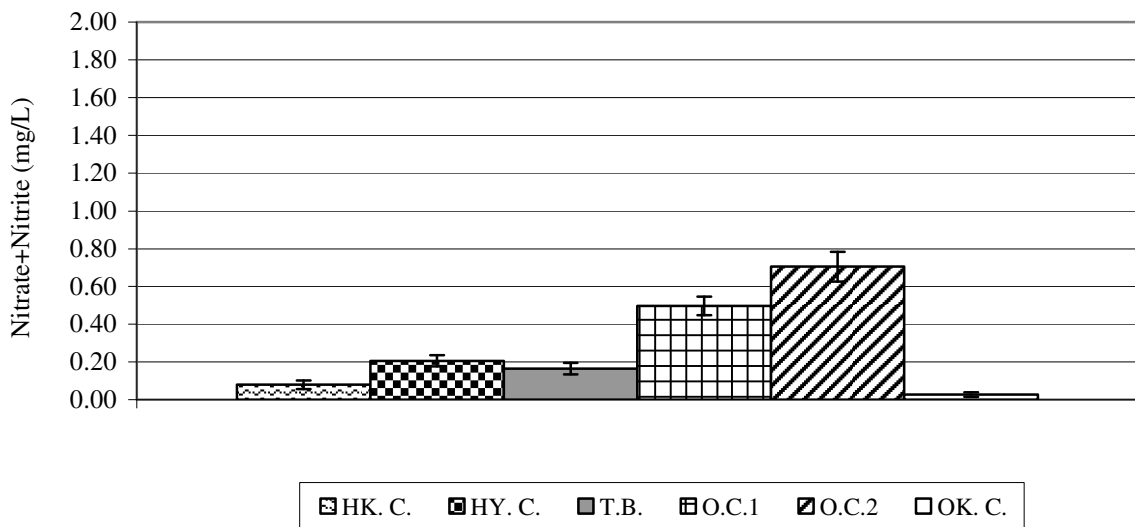


Figure 16. Mean nitrate + nitrite concentrations at each sampling site of the Canadarago Lake watershed, summer 2008. Note different scale from Figure 15.

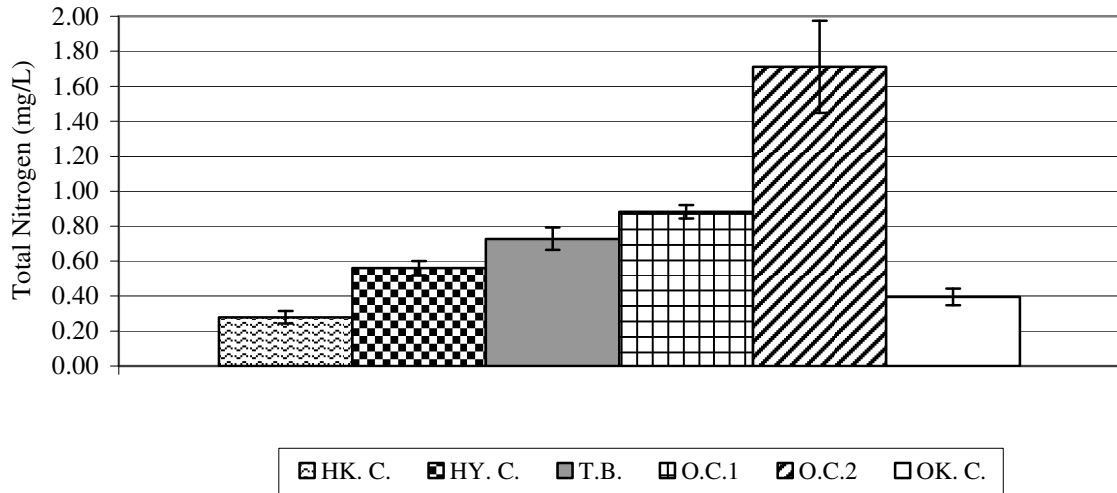


Figure 17. Mean total nitrogen concentrations at each sampling site of the Canadarago Lake watershed, summer 2008. Note different scale from Figure 15.

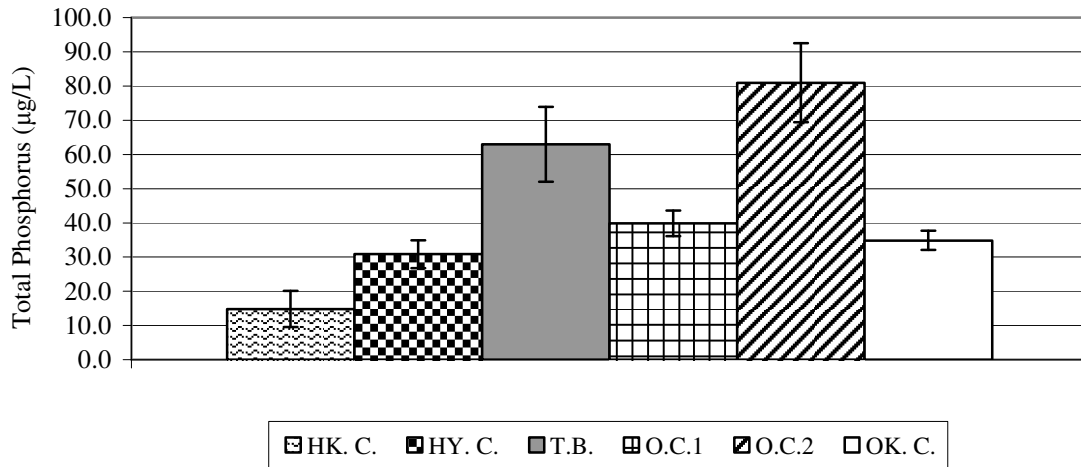


Figure 18. Mean total phosphorus concentrations at each sampling site of the Canadarago Lake watershed, summer 2008.

	NH ₃ (mg/l)	NO ₂ +NO ₃ (mg/l)	TN (mg/l)	TP (ug/l)
W.T.P.	1.47	7.81	9.85	213.4
(SE)	1.30	1.34	0.80	104.3

Table 5. Mean nutrient concentrations and standard errors of effluent collected from the Richfield Springs waste treatment plant, summer 2008.

Fecal Coliform

Mean fecal coliform concentrations are given in Figure 19. Trout Brook (T.B.) contained the highest levels, with an average of 930 colonies per 100 mL and Oaks Creek (OK. C.) contained the lowest concentrations with an average of 163 colonies per 100 mL. There was an increase in fecal coliform from Ocquionis Creek North (O.C. 1) to Ocquionis Creek (O.C. 2), though the effluent of the Richfield Springs waste treatment plant revealed concentrations below 5 colonies/100 mL. Increases in coliform bacteria counts did correspond to days following large rainstorms, linking fecal coliform levels (at least in part) to the agricultural nature of the surrounding watershed.

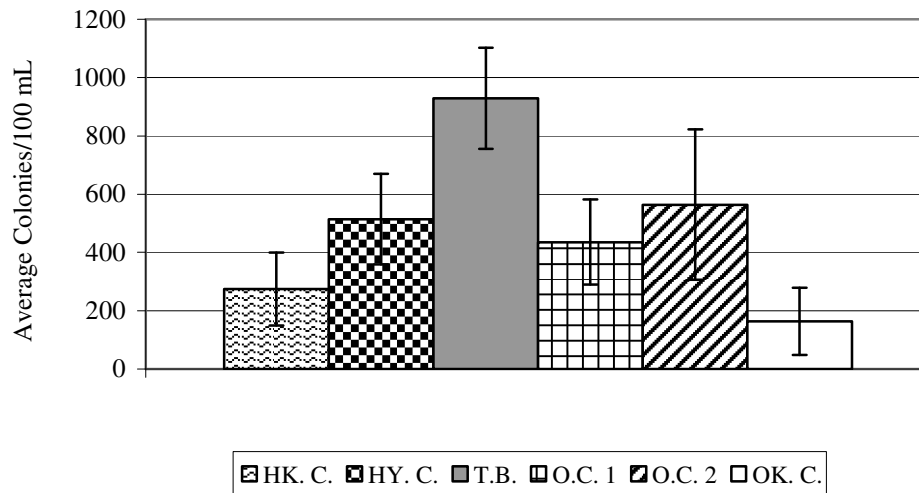


Figure 19. Average number of fecal coliform colonies from 25 June to 23 July 2008, collected from the corresponding Canadarago watershed sites.

Sediment

Figures 20-22 summarize the concentrations of total, organic and inorganic suspended sediment at the tributary sites. Ashing of the filters did not begin until the 2 July, compared to total sediment which began on the 18 June. The highest suspended sediment value recorded (106.8 mg/L) occurred on 16 July at Ocquionis Creek South (O.C. 2). This sample also contained the highest values in both organic (28.8 mg/L) and inorganic (78.0 mg/L) suspended sediment. This could be possibly due to the muddy banked wetland separating the two sites and increased water flow because of rain the previous night/morning of sampling. Inorganic sediment tended to be in higher proportion than organic at most of the sampling locations.

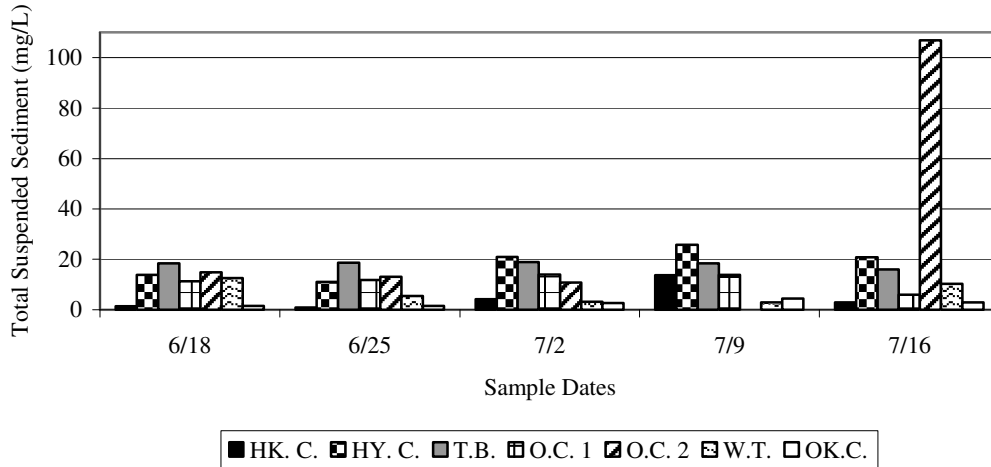


Figure 20. Total suspended sediment from each Canadarago watershed site, by date, summer 2008.

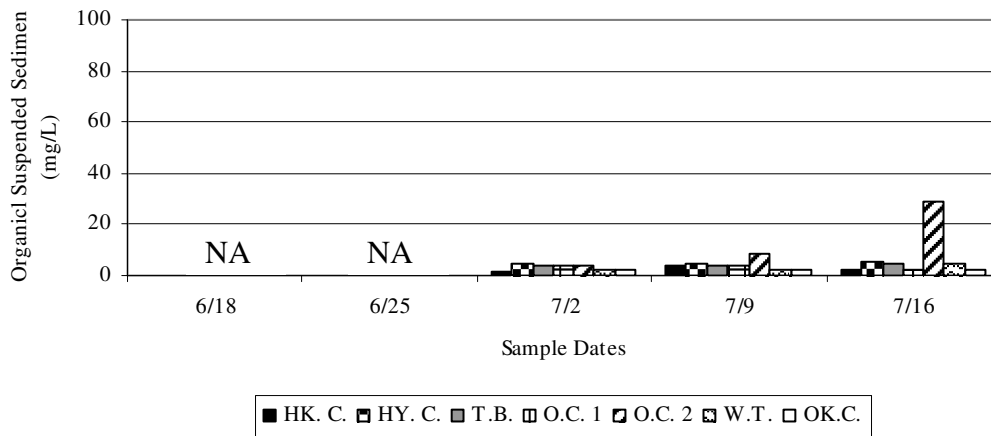


Figure 21. Organic suspended sediment from each Canadarago watershed site, by date, summer 2008.

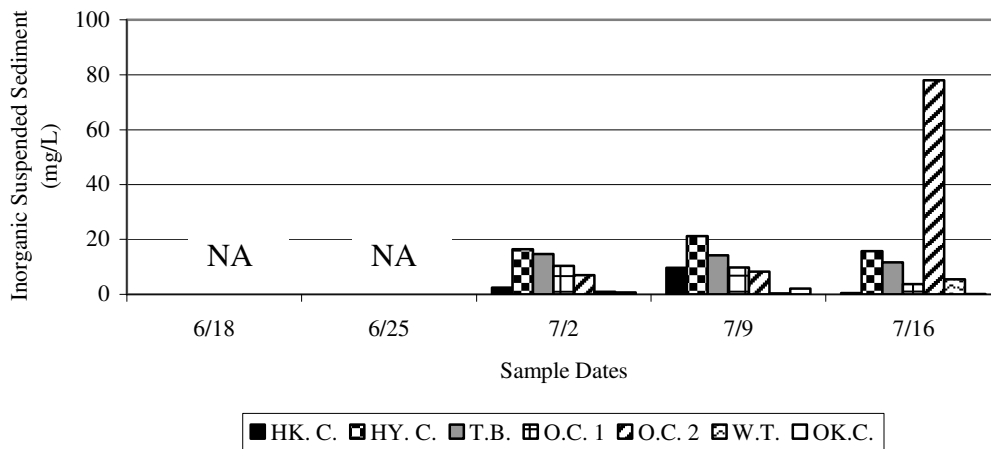


Figure 22. Inorganic suspended sediment from each Canadarago watershed site, by date, summer 2008.

CONCLUSION

One of the most obvious changes in Canadarago Lake since 2001 (Harman et al. 2002) is the water clarity. The average Secchi disk reading that year was 2.6 m (Harman et al. 2002) compared to the 2008 average of 6.8 m. This is likely the result of the establishment of zebra mussels, highly efficient filter feeders (D'Itri 1997). Although clarity has increased the lake still exhibits a eutrophic nature, similar to that described in 1968 (Harr et al. 1980). Bottom waters become anoxic by mid-summer, and the resultant reducing environment leads to the conversion of nitrates to ammonia as well as the release of phosphorus from bottom sediments (known as internal phosphorus loading). Redistribution of that nutrient during periods of turnover will likely stimulate algal growth.

External sources of nutrient could include agricultural activities and near-shore development. The Richfield Springs waste treatment plant effluent is below the regulated concentration of 0.5 mg/L phosphorus, with an average concentration being 0.21 mg/L. Concentrations of nitrogenous compounds are substantial (with mean TN approximately 10.0 mg/l), as nitrogen is not regulated. Agricultural activities are also suspected to have contributed to the observed fecal coliform levels of the lake tributaries, mainly Trout Brook (T.B.). Suspended sediment levels were consistently low (<20 mg/l) across all tributaries, the exception being Ocquionis Creek South (O.C. 2) following a rain event. Herkimer Creek (HK. C.), which enters into the south end of the lake and is the topic of a possible dredging campaign, contributed the lowest concentrations of suspended sediments and all fractions of nutrients.

Aquatic plant species observed were generally similar to those recorded by Muenscher (1936). Two species currently common are curly leaved pondweed and Eurasian milfoil, both exotic nuisance species. Water chestnut (*Trapa natans*) was not recorded, despite its presence in other water bodies of Otsego County.

The establishment of zebra mussel and alewife, both nuisance exotic animals, have had detrimental effects on the lake. The extent to which they will impact the lake's ecology and recreational is yet to be known.

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