WATER QUALITY OF SENECA LAKE, NEW YORK: A 2011 UPDATE.

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INTRODUCTION

Seneca Lake provides Class AA drinking water to ~100,000 people with total permitted withdrawals of ~9 million gallons of water per day (Callinan, 2001). The lake is also essential to the economic and social structure of the region by injecting ~$100 million per year into the local economy through tourism and recreation alone, and influencing a tax base of over $1 billion. Seneca Lake is the largest Finger Lake on many measures. Its volume of 15.5 km^3 is over 50% of the water contained in the eleven Finger Lakes. It also has the largest surface area 175 km^2, the deepest depth 186 m, and the longest residence time 18.6 years (Schaffner and Oglesby, 1978, Wing et al., 1995, Mullins et al., 1996, Callinan, 2001). Its watershed area of 1,586 km^2 and length of 57 km are second only to Cayuga Lake (1,870 km^2 and 61 km). Because the watershed spans portions of Ontario, Seneca, Yates, Schuyler, Stueben (Keuka watershed) and Chemung counties and various cities, villages and towns, New York's home rule status and thus its multi-jurisdiction watershed sets additional hurdles for uniform watershed rules and pollution prevention regulations. Thus, Seneca Lake is a critical resource for the region and everyone must stringently protect it, because once stressed, perturbed or degraded, it will take multiple generations to restore the lake back to its former, less stressed state.

A 2005 water quality survey, conducted under the direction of Dr. John Halfman, Finger Lakes Institute at Hobart and William Smith Colleges, ranked water quality parameters for Skaneateles, Owasco, Cayuga, Seneca, Keuka, Canandaigua and Honeoye Lakes (Fig. 1., Halfman and Bush, 2006). The ranking was based on monthly (May - October) secchi disk depths, and surface water quality analyses for total phosphate (TP), dissolved phosphate (SRP), nitrate, chlorophyll-a and total suspended solid concentrations from at least two mid-lake, deep-water sites in each lake. Water samples were also analyzed for total coliform and *E. coli* bacteria in 2005. The ranking indicated that Seneca, Owasco, and Honeoye Lakes had the worst water quality, whereas Skaneateles, Canandaigua and Keuka Lakes had the best water quality. Cayuga Lake fell in between these end-members. The 2005 preliminary report also noted a correlation between the ranking and

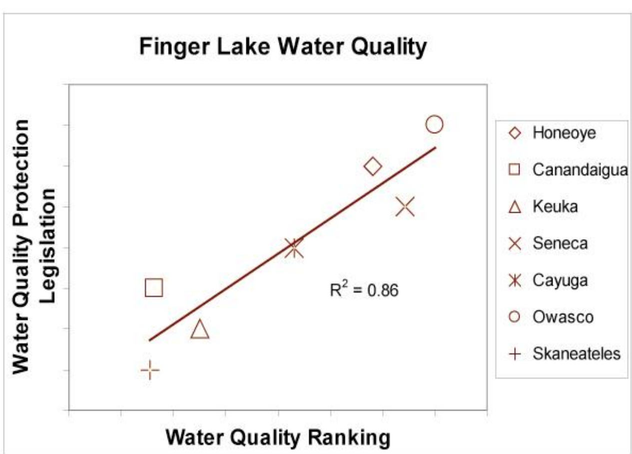


Fig. 1. The 2005 water quality ranking of 7 Finger Lakes.

a first-order, qualitative assessment of water quality protection legislation. Subsequent research also indicated that land use activities, precipitation amounts, and/or the impact of recent exotics like zebra and quagga mussels and *Cercopagis pengoi*, the fishhook water flea play a role.

Here, we report on our current understanding of the limnology and hydrogeochemistry of the lake based on ongoing monitoring efforts at Hobart and William Smith Colleges (HWS). The program was initiated in the early 1990s with the addition of a limnologist on the faculty of HWS. This position was first occupied by Prof. M. Wing and then Prof. J.D. Halfman. Since the mid-1990's, the program comprised of weekly monitoring of four lake sites in the northern end of the lake, and seasonal sampling of a number of streams in or adjacent to the watershed.

The objectives were to:

(1) establish consistent and comprehensive monitoring to document spatial and temporal trends in the limnology and other water quality parameters in the lake;
(2) bring particular focus to the extent and sources of nutrients to the lake and associated watershed-lake interactions;
(3) provide water quality data to local government agencies, watershed protection groups and concerned citizens; and,
(4) promote the development of effective and comprehensive watershed management policies to initiate the protection and if required the remediation of Seneca Lake.

This 2011 report builds on the lake and stream water quality chapters in the 1999 State of the Seneca Lake Watershed Report (Halfman and many undergraduate students, 1999), and subsequent reports, publications and presentations by Halfman and his students, especially an update by Halfman and Franklin (2007).

WATER QUALITY

Water is a critical resource used for domestic, agricultural and industrial uses. In the US, water withdrawals exceed 900,000 million gallons/day from both surface (80%) and groundwater (20%) sources (USGS 2005 water facts). However, water is susceptible to abuse and pollution.

Pollutants are categorized by source and/or by composition. *Point source pollutants* are discharged from an identifiable spot, irregardless of its composition, like the outlet of a municipal wastewater treatment plant, a factory drain pipe, or power plant cooling-water outflow. These identifiable sources are typically regulated, monitored and controlled. For example, the treatment of organic wastes by a municipal wastewater treatment plant is regulated by the New York State Pollutant Discharge Elimination System (SPDES), a permit program approved by the US Environmental Protection Agency (EPA) in accordance with the Clean Water Act. SPDES Permits are overseen by NYS Department of Environmental Conservation (DEC) to limit and/or monitor the quality and quantity of discharges to a water body and maintain water quality standards consistent with public health and economic considerations (http://www.dec.ny.gov/permits/6308.html). *Nonpoint source pollutants* originate from large or dispersed land areas, such as runoff of road salt or agricultural fertilizer. Due to their diffuse nature, nonpoint source pollutants are more difficult to regulate, monitor and control, but are increasingly becoming the focus of federal and state legislation.

Pollutants are also categorized into organic, agricultural, and industrial wastes. Organic wastes are primarily materials sent down the drain or flushed down the toilet by humans. Agricultural wastes include, for example, the runoff of soils, fertilizers, herbicides and pesticides, and byproducts from Concentrated Animal Feeding Operations (CAFOs). Industrial wastes include, for example, the disposal of heavy metals, manufactured organic compounds, their byproducts and heat. Each group has its own degree of legislation and control. Organic wastes and agricultural runoff are more relevant to the Seneca watershed.

Both municipal wastewater treatment facilities in high population density areas and individual onsite wastewater treatment systems (septic systems) in low population density areas are designed to remove solid and dissolved organic materials from wastewater in two steps. First, raceways and tanks allow particulates to settle out and microbes to digest some of the solid materials. Then, aerobic bacteria decompose the dissolved organics, and converts the organic matter into carbon dioxide (colorless and odorless) and dissolved nutrients. Smaller or older facilities will then discharge the nutrient-rich effluent to a nearby stream or lake. Larger and newer municipal facilities and more sophisticated onsite systems include additional tertiary treatments that chemically reduce the concentration of dissolved nutrients and other compounds in the effluent before releasing the treated wastewater.

Current legislation also attempts to control the impact from crop and animal farms through various recommended "Best Management Practices" (BMPs), such as contour plowing, settling ponds, buffer vegetation strips, minimal tillage farming, manure digesters, gully plugs and other methods. BMPs often reduce agricultural impact on nearby waterway. However, the potential financial burden for farmers is significant if they had to bear the cost to establish and maintain the BMPs, and the lost revenue from the BMP displaced farmland.

WATER QUALITY INDICATORS

The Seneca watershed is dominated by a rural landscape with a mix of agriculture (46%), forests (38%), and smaller amounts of urban (5%) and other land uses (Fig. 2). Keuka Lake occupies the remaining 12% of the watershed. The distribution suggests that the primary water quality threat to the lake is nutrient loading from organic wastes and agricultural runoff, and its stimulation of algal and nearshore plant growth. Continued nutrient loading progressively reduces water quality, and the lake eventually becomes eutrophic. A basic limnological primer is required to understand these implications, various measures of water quality, and typical seasonal variability.

Algae, the primary producers in an aquatic ecosystem, require sunlight and nutrients to grow. Isothermal conditions during spring overturn mix the available dissolved nutrients throughout the water column. These isothermal conditions also mix the algae into and out of the photic zone, and thus algae are typically light limited. Increase sunlight during the spring, and the algae bloom just as the lake becomes thermally stratified and the algae remain in the sunlit epilimnion (sunlit, warmer, less dense, surface waters). Summer stratification however isolates photosynthesis and the uptake of dissolved nutrients to the epilimnion. Thus, epilimnetic nutrients become scarce and it limits algal populations for the remainder of the stratified season. Another limit is their predation by herbaceous zooplankton. Nutrient concentrations increase during the summer in the hypolimnion (dark, colder, more dense, bottom waters) due to bacterial decomposition (respiration) of organic matter. Algae bloom once again in the fall, during the

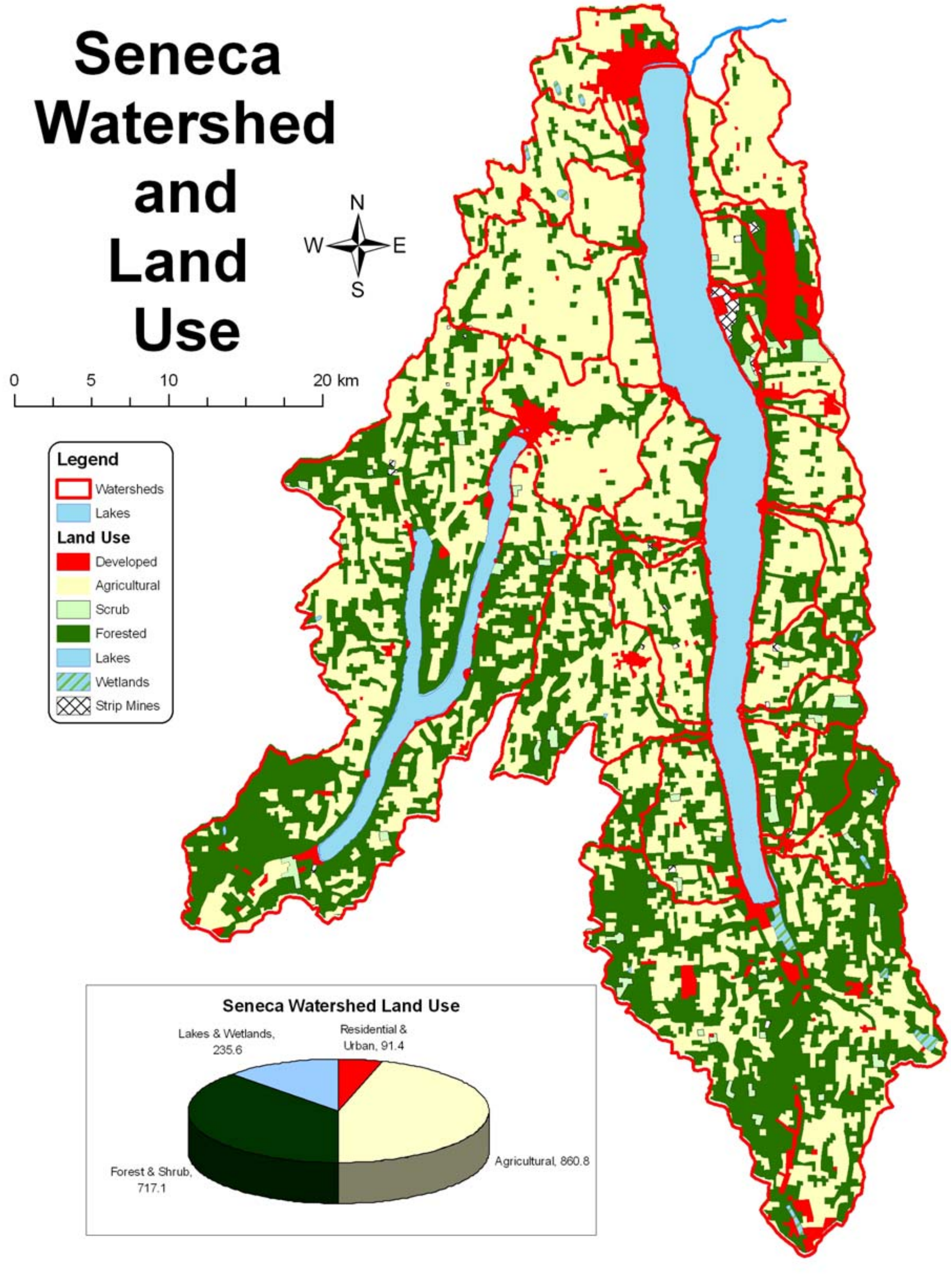


Fig. 2. Land use in the Seneca watershed. Kendig Creek in the northeast corner is just outside the Seneca watershed but it was routinely sampled because of its large agricultural land use.

thermal decay of the epilimnion and mixing of nutrient-rich bottom water into the sunlight. Runoff, nutrient loading, internal seiche activity, waves & currents and other events can also introduce nutrients to the epilimnion and stimulate algal growth. Reduced light limits algal growth in the isothermal and winter stratified seasons.

Aquatic ecosystems are extremely efficient at recycling nutrients (Fig. 3). Nutrients dissolved in water are assimilated by algae and nearshore plants. These nutrients are passed up the food chain to other organisms when the algae are eaten. When any of these organisms die, bacteria decompose the organic material and release almost all of the nutrients back into the water column. The nutrients are then available for algal uptake once again, completing the nutrient cycle.

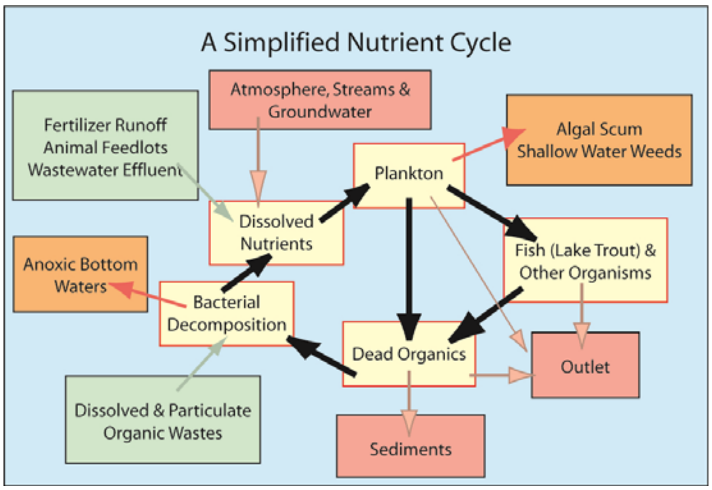


Fig. 3. A typical nutrient cycle for lake ecosystems (yellow boxes and black arrows). Typical human-induced additions (green), their impact (orange), and natural sources and sinks (red) from the nutrient cycle are also shown.

Nitrates (NO_3^-) and phosphates (PO_4^{-3}) are imperative for amino acids, proteins, RNA, DNA, Krebs Cycle and other compounds and processes in organisms. Dissolved silica ($H_2SiO_4^{2-}$) is required by diatoms to form their frustules (shells). If dissolved silica is scarce, below a few 100 μg/L, then other forms of algae dominate in the lake, that do not require silica tests (e.g., green algae, blue-green algae). These nutrients enter nearby waterways from a variety of natural and human-induced sources. Natural sources include the weathering of bedrock, erosion of soils, and gases (e.g., NO_x) dissolved in rainwater. Human-induced nutrient sources include municipal wastewater treatment plants, onsite wastewater systems (septic systems), runoff from agricultural areas and manicured lawns, and other sources. These sources are water quality concerns because they "fertilize" a lake and make it more productive. Unfortunately, once nutrients enter a lake, the ecosystem typically remains enriched in nutrients because they are continually and efficiently recycled, and continue to "fertilize" plant growth at enhanced levels. If the aquatic system becomes too productive, i.e., eutrophic, then a foul smelling/tasting scum of blue-green algae typically dominates the phytoplankton community (base of the food chain) and covers the surface of the lake with a green slime. Macrophytes, rooted nearshore plants, prosper as well. The increase in algae decreases water clarity (transparency). The extra algae also hamper and significantly increase the cost of water filtration for municipal water supplies. For example, Syracuse, NY, spends large sums of money each year to keep Skaneateles Lake clean, clean enough to eliminate the need for even more expensive filtration costs. The net savings are passed on to the residents of Syracuse. Thus, nutrients provide one indicator of water clarity, lake productivity, the ecological health of the lake and local economics.

When the algae die, bacteria naturally decompose the organic matter and consume dissolved oxygen in the process. If the removal of oxygen from the summer-time hypolimnion decreases concentrations below 6 mg/L (or 4 mg/L, each species has its own level of tolerance), it induces respiratory stress. Complete de-oxygenation of the bottom waters in eutrophic lakes induces fish kills, and, if sulfur is available, releases a "rotten egg" odor caused by the bacterial release of hydrogen sulfide. Anoxic bottom waters also allow phosphates, previously buried and trapped in the sediments as oxides in oxygen-rich oligotrophic systems, to dissolve, re-enter and re-fertilize the water column.

Nutrient loading is not the only means to induce algal blooms. Zooplanktivorous fish like alewife or carnivorous zooplankton like *C. pengoi* the fishhook water flea can induce algal blooms as well. Their predation on herbaceous zooplankton reduces zooplankton predation on algae, thus reduces this natural means for algal population control during the summer. Thus, both "bottom up", i.e., nutrient loading, and "top down", i.e., predation on herbaceous zooplankton, can stimulate algal blooms and decrease water quality in a lake.

Are excess nutrients health threats to humans? Excess nitrate are, especially for infants, as they can increase the risk for blue-baby syndrome or methemoglobinemia. The EPA has set a maximum contaminant level (MCL), concentration limits for safe drinking water, at 10 mg/L for nitrates. Phosphate and silica, at natural concentrations up to 100 μg/L (P) and 100 mg/L (Si), respectively, do not pose health risks. Excess phosphates however do contribute to the fertilization and eutrophication of waterways because phosphorus is typically the limiting nutrient (see below). Thus, NYS DEC uses a total phosphate concentration of 20 μg/L, P, as the threshold for impaired or overly productive lakes.

Phosphorus typically limits algal growth in temperate lakes because the natural sources of phosphates are limited to weathering of relatively rare rocks. Once weathered, phosphate delivery to a lake is even further hampered because phosphates easily bind to soil particles and are nearly insoluble in water. Even when they are mobilized, they typically absorb onto sediment particles and are buried in the sediment.

In contrast, nitrates are typically non-limiting. Nitrates do not bind to soil particles. They are more water soluble, and are delivered to the lake by both stream runoff and groundwater flow. The atmosphere provides additional sources of usable nitrogen compounds. Nitrates created from acid rain nitrogen oxides enter lakes by both dry and wet deposition, and some blue green algae "fix" otherwise inert nitrogen gas (N_2) from the atmosphere.

Algal concentrations, measured directly by the concentration of chlorophyll, and indirectly by fluorometer, total suspended solids and secchi disk depths, are another indicator of water clarity, lake productivity, and the ecological health of a lake. Chlorophyll content is a direct measure of algal concentrations, as all algae utilize chlorophyll to harness energy from sunlight to perform photosynthesis. Its measurement, however, requires filtration of water samples and digestion of the algae in acetone before analysis by spectrophotometer. A fluorometer measures the concentration of algae *in situ* or in the laboratory. Algal concentrations are proportional to the intensity of the fluoresced red light after excitation by blue light. Filter a known volume of water through a pre-weighed filter and measure the dry mass of the filtered material, determines the total suspended solids (TSS) concentrations. TSS measures all suspended materials, organic and inorganic. The secchi disk is a weighted disk, 20 cm in diameter, painted with two black and two white quadrants. It is slowly lowered into the water column until it disappears, and this water depth is noted. The disk is lowered some more, and then slowly pulled up until it reappears, and this second depth is noted. The secchi disk depth is the average of these two depths. In very transparent waters (plankton poor, ultra-oligotrophic, very low productivity systems), a secchi disk depth can be 100 feet (30 m) or more. Whereas in eutrophic, highly productive systems, like a pond on a farm, secchi disk depths can be as shallow as a few centimeters. Secchi disk depths, however, are also influenced by suspended sediments delivered by streams or resuspended by waves and currents. Careful comparison of these parameters differentiates if the impairment to water clarity is due to algae, suspended sediments, or both (Georgian and Halfman, 2008).

Dissolved oxygen (DO) concentration is the last primary indicator of algal productivity and the trophic status of lakes. Atmospheric oxygen is the primary source of dissolved oxygen to a lake. Temperature is the primary control on the saturation concentration. Cold water exponentially hold more oxygen than warmer water at saturation, e.g., 12 to 13 mg/L at 4°C, and 8 mg/L at 25°C. DO concentrations are also influenced by photosynthesis and respiration. Photosynthesis releases oxygen to the water. Because photosynthesis occurs mostly, if not entirely, in the epilimnion, photosynthesis and the exchange of oxygen with the atmosphere maintains a saturated epilimnion. The epilimnion DO concentrations are thus typically dictated by seasonal changes in water temperature. In contrast, oxygen is consumed by respiration, especially bacterial respiration, throughout the lake. Bacteria dominate DO concentrations in the dark and isolated hypolimnion of the lake, and therefore hypolimnetic DO concentrations can progressively become depleted below saturation during the stratified season given enough organic matter to decompose. Stratification also isolates the hypolimnion from potential oxygen sources, the atmosphere and photosynthesis. The extent of hypolimnetic depletion depends on the amount of organic mater, both aquatic and terrestrial, to decompose, and the volume and initial stratification saturation conditions for DO in the hypolimnion. Dissolved oxygen is depleted from saturated concentrations (12 to 13 mg/L) in the spring to anoxic conditions (0 mg/L) by mid- to late summer in the hypolimnion of typical eutrophic lakes.

Thus, a combination of nutrient concentrations, algal concentrations, secchi disk depths, and dissolved oxygen concentrations are utilized to document the degree of productivity from oligotrophic (poorly productive), mesotrophic (intermediate) to eutrophic (highly productive) systems (trophic status) and progressively more impaired water quality in aquatic systems (Table 1). These ranges serve as guidelines and not every parameter must be within each range for one trophic level to be classified that trophic level. For example, a gargantuan hypolimnion may not turn anoxic even though every other parameter falls within the eutrophic ranges.

Table 1. Oligotrophic, Mesotrophic and Eutrophic Indicator Concentrations (EPA).

Trophic Status	Secchi Depth (m)	Total Nitrogen (N, mg/L, ppm)	Total Phosphate (P, µg/L, ppb)	Chlorophyll a (µg/L, ppb)	Oxygen (% saturation)
Oligotrophic	> 4	< 2	< 10	< 4	> 80
Mesotrophic	2 to 4	2 to 5	10 to 20	4 to 10	10 to 80
Eutrophic	< 2	> 5	> 20 (> 30)	> 10	< 10

HYDROGEOCHEMICAL & OTHER WATER QUALITY INDICATORS

Major ions, pH, coliform bacteria, herbicides, pesticides, heavy metals, trihalomethanes and other materials, are additional but not exhaustive list of water quality indicators. The major ions, chloride, sulfate, bicarbonate/carbonate, sodium, potassium, calcium and magnesium, combine to determine the salinity of the lake. Chloride concentrations above 250 mg/L (ppm) pose a health risk because the associated sodium hamper the development of kidneys in small children, and aggravate heart problems in older individuals. Water with total dissolved ion concentrations above 2,000 mg/L is too salty to drink because it promotes nausea and dehydration. Seawater, at ~35,000 mg/L, significantly exceeds drinking water limits.

Acceptable pHs for natural waters is neutral to slight basic, i.e., between 6.5 and 8.5. Lakes lacking a buffering capacity in the underlying soils, tills and bedrock have been acidified through the dry and wet deposition of acid rain compounds, sulfuric and nitric acids, from burning of fossil fuels. Lakes in New York are threatened by acid rain, if the watershed lacks a buffering capacity, because the pH of rainfall is 4 to 4.5.

Finally, industrial release and/or agricultural runoff of heavy metals, herbicides, pesticides and other pollutants impact water quality. For example, atrazine is a common herbicide used to control broadleaf weeds in corn, sorghum, sugar cane, pineapple and other crops. It is also health threat, and has been shown to cause adrenal degradation and congestion of the lungs, liver and kidneys. It has been banned in the European Union since 2004. Its likelihood as an endocrine disrupter and its epidemiological connection to low sperm counts in men, has led the call to ban its use in the United States. Many water supplies in the mid-west corn belt are contaminated with atrazine, with concentrations above the EPA's 3 μg/L MCL.

Total coliform and *E. coli* bacteria are used to monitor for the presence of human organic wastes and associated disease causing organisms. However, these bacteria themselves pose minimal health threats, except for a few toxic strains of *E. coli*. Coliform sources also include geese, dogs, deer and other warm blooded, wild and domesticated, animals. MCLs for total coliform and *E. coli* are 2,400 CFUs/100mL and 235 CFUs/100mL, respectively, calculated as geometric means. Trihalomethanes (e.g., chloroform, bromoform, bromodichloromethane) are disinfection byproducts predominantly formed when chlorine is used to disinfect water. The MCL for total THMs is 80 μg/L.

Hardness is defined by the concentration of divalent cations (ions with a 2^+ charge) with hard water exceeding a total concentration of 80 mg/L ($CaCO_3$). The concentration of calcium and magnesium in surface and groundwater reflect the availability of limestone and other easily weathered calcium and/or magnesium rich minerals in the underlying bedrock and/or glacial till for the individual drainages. Hard water is not harmful to drink as long as the total salt concentration is less than 2,000 mg/L, and some studies have shown that drinking hard water may reduce the risk of heart disease. However, hard water is a nuisance problem by preventing the lathering of soap, yellowing of white clothes, and precipitating a carbonate scum on pots, pans, and inside hot water pipes, water heaters and other plumbing. Water softeners reduce water hardness by passing the water through an ion-exchange material. The process typically exchanges calcium (or magnesium) for sodium (or potassium). Rock salt (NaCl) is used to periodically recharge the ion-exchange mineral with sodium (or KCl recharging with potassium). Softeners can effectively reduce the hard water problem for a household but waste 10% of water used in the home and the sodium salt ion exchange softeners create new problems for those on low-sodium diets.

METHODS

Lake Research: Monitoring of Seneca Lake at HWS started in 1991 but was progressively more frequent and included more parameters after 1996. Lake sites focused on Sites 1 through 4 located in the northern portion of the lake (Table 2, Fig. 4). The four sites maximized the nearshore/offshore and shallow/deep water diversity in Seneca Lake while minimizing the expense during a ½ day cruise aboard the HWS Scandling, the Colleges' 65 ft steel-hulled research vessel. These sites are also used for the Finger Lake Institute's comparative water quality survey of the eight easternmost Finger Lakes. Water quality samples were collected from surface (< 1m) and bottom water (within 2m of the lake floor) depths at Sites 1 & 3, but only surface water samples at the other two sites. Sites were typically sampled weekly from mid-April through mid-October each year. In 1998, 1999, 2010 and 2011 additional data were collected from five additional sites distributed from the northern survey area southward along the central axis of the lake to Watkins Glen to assess the similarity of the northern sites to the rest of the lake.

Table 2. Lake & Buoy Site Locations and Site Water Depths.

Site	Latitude	Longitude	Water Depth (m)
Weekly			
Site 1	42° 50.80' N	76° 57.56' W	22
Site 2	42° 46.82' N	76° 55.99' W	25
Site 3	42° 46.28' N	76° 57.00' W	115
Site 4	42° 45.79' N	76° 58.01' W	25
Occasionally			
Site 5	42° 43.50' N	76° 56.50' W	135
Site 6	42° 41.00' N	76° 55.00' W	140
Site 7	42° 36.50' N	76° 53.70' W	179
Site 8	42° 30.00' N	76° 54.00' W	157
Site 9	42° 25.00' N	76° 52.60' W	129
Buoy Site	42° 49.13' N	76° 57.61' W	57

A conductivity, temperature and depth (CTD) water quality profile, secchi disk depth and plankton tows, were also collected at each site. The CTD was lowered from the surface to ~2 m above the lake floor, and collected data every 0.5 seconds (~0.5 m) along the downcast. Before 2007, a SeaBird SBE-19 CTD collected temperature, conductivity (reported as specific conductance), dissolved oxygen, pH, and light transmission (water clarity, inversely proportional to turbidity) profiles. In 2007, the CTD was upgraded to a SeaBird SBE-25 with additional sensors for photosynthetically active radiation (PAR), turbidity by light scattering and chlorophyll-a by fluorescence.

Water samples were analyzed onsite for temperature, conductivity, dissolved oxygen, pH and alkalinity using hand-held probes and field titration kits. Water samples were also analyzed for total phosphate (starting in 2006), dissolved phosphate, nitrate, dissolved silica, chlorophyll-a, total suspended solids and major ions in the laboratory. Surface and depth integrative (20 m) horizontal and vertical plankton tows were collected using 85 μm mesh, 0.2m diameter opening plankton nets, preserved in a formalin/alcohol solution for subsequent enumeration in the lab. Additional water samples were analyzed for atrazine, a common herbicide, in 1999 and 2000, trihalomethanes in 2010, and total coliform and *E. coli* bacteria assays in 2003 through 2005.

Finally, a water quality (WQ) buoy (YSI 6952 platform) was deployed from April through November of each year starting in 2006, at a mid-lake location between Sites 1 and 4 in approximately 55 m of water. It collected two profiles every day using a YSI 6600-D logger measuring depth, temperature, conductivity, turbidity and fluorescence (chlorophyll) every 1.5 m through the water column. It also collected hourly averaged meteorological data including air temperature, barometric pressure, light intensity, relative humidity, wind speed and direction. The data are publically available at: http://fli-data.hws.edu/clarkpt/.

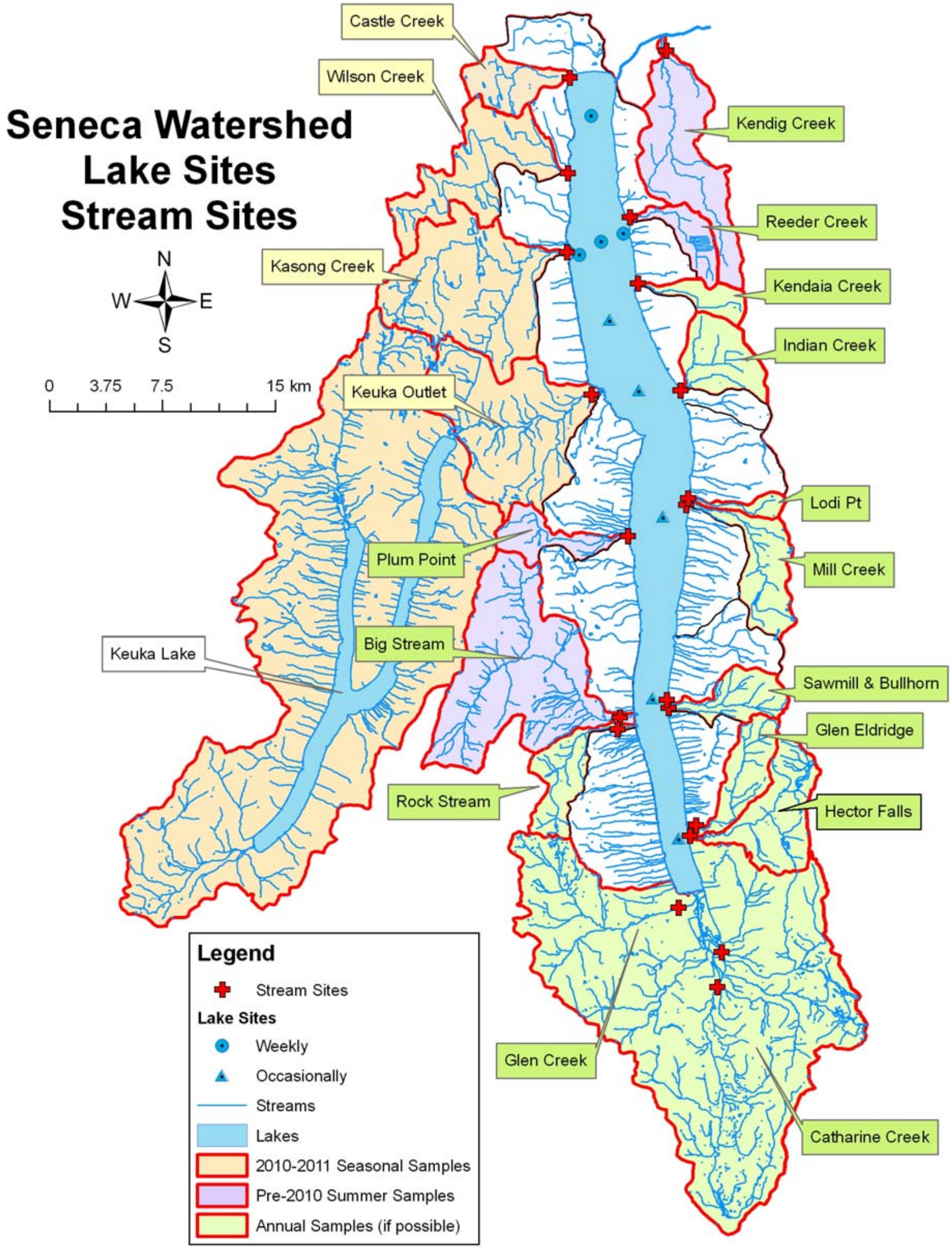


Fig. 4. Lake and stream sites.

Stream Research: HWS stream monitoring started in 1998. The stream sites from 1998 through 2009 focused on six major tributaries that entered the western and northeastern sides of the lake and included Wilson Creek, Kashong Creek, Keuka Outlet, Plum Pt Creek, Big Stream, Reeder Creek and Kendig Creek (Table 3, Fig. 4). Kendig Cr. was included in the survey even though it was not in the Seneca watershed because it drained one of the most agricultural landscapes in this group. Sample sites were located near the terminus of the stream, typically at the intersection of the stream and Rt. 14. Keuka Outlet was sampled at Milo Street in Dresden, Plum Pt. Creek at Hall Road, Reeder Creek at East Lake Road, and Kendig Creek at Marshall Road. These streams were typically sampled weekly from late May through July (unless they dried up earlier in the summer). Once or twice a summer, samples were also collected from the other major tributaries to the lake including Rock Stream, Glen Creek, Hector Falls, Glen Eldridge, Sawmill, Bullhorn, Mill, Lodi Pt, Indian, and Kendaia Creeks. Segment analysis, analysis at a series of locations distributed from the mouth to the headwaters of the stream, was performed along Big Stream in 2001 and Keuka Outlet in 2003 to pinpoint sources of nutrients and other contaminants (Bowser, 2002; Hintz, 2004).

Table 3. Watershed Areas and Percent Land Use.

Watershed/Subwatershed	Area (km2)	Residential & Urban (%)	Agriculture (%)	Forest & Shrubs (%)	Lakes & Wetlands (%)
Cathrine Creek Subwatershed	329.75	4.3	36.7	57.2	1.4
Reading Drainage	50.55	2.9	49.6	47.1	0.3
Rock Stream Drainage	20.13	0.0	49.1	50.9	0.0
Big Stream Drainage	96.26	1.8	53.1	45.0	0.2
Starkey Drainage	48.63	2.1	63.1	34.2	0.5
Plum Point Subwatershed	15.49	2.9	53.0	43.7	1.0
Long Point Drainage	38.26	2.9	72.8	24.3	0.0
Keuka Lake Outlet Subwatershed	80.13	7.4	76.9	15.6	0.1
Benton Drainage	21.33	2.6	82.6	14.8	0.0
Kashong Creek Subwatershed	80.53	0.9	83.3	15.7	0.1
Reed Point Drainage	22.21	2.3	87.0	10.6	0.0
Wilson Creek Subwatershed	46.67	1.3	78.6	18.4	1.6
Geneva Drainage	55.55	30.2	54.3	14.4	1.2
Sunset Bay Drainage	18.77	6.6	78.0	15.4	0.0
Reeder Creek Subwatershed	12.73	56.3	17.3	22.9	3.5
Wilcox Creek Drainage	13.65	5.8	51.0	41.9	1.2
Kendaia Subwatershed	10.13	61.6	9.9	27.3	1.2
Sampson State Park Drainage	14.01	17.4	9.1	49.4	24.0
Indian Creek Subwatershed	22.86	20.5	38.7	39.9	0.9
Simpson Creek Subwatershed	8.36	26.3	54.1	19.1	0.5
Sixteen Falls Creek Drainage	31.30	1.3	69.6	28.9	0.3
Lodi Point Subwatershed	5.00	9.6	62.4	28.0	0.0
Mill Creek Subwatershed	25.61	0.9	58.6	40.3	0.2
Lamoreaux Landing Drainage	26.74	1.5	59.4	38.3	0.0
Valois Drainage	28.42	2.3	51.1	43.5	1.4
Sawmill/Bullhorn Creek Subwatershed	17.25	1.4	36.2	62.4	0.0
Satterly Hill Drainage	22.45	0.4	38.5	60.6	0.5
Glen Eldridge Subwatershed	20.13	1.4	28.8	69.0	1.4
Hector Falls Creek Subwatershed	33.54	2.1	26.5	70.2	1.2

Starting in 2010, Castle, Wilson, Kashong and Keuka Outlet were sampled weekly throughout the year. Castle Creek was added because it drains a smaller basin with the greatest percentage of urban land use. The site was just downstream from Exchange St in Geneva. Catharine Creek was sampled every other survey in 2011 at typically two sites. Catharine Creek was added to provide baseline data before the initiation of hydrofracking in the southern end of the watershed. One site was at Rt 224 just downstream from Montour Falls and the other at Genesee St just upstream from Montour Falls.

Water samples were measured onsite for temperature, conductivity, dissolved oxygen, pH and alkalinity using hand-held probes or field titration kits. Samples were also collected and analyzed in the laboratory for total phosphate (starting in 2006), dissolved phosphate, nitrate, dissolved silica, total suspended solids and major ion concentrations.

Stream discharge (the volume of water per unit time) was calculated from measured stream width, depth and water velocity data during each visit. The velocity was measured at ~60% of the depth from the top using a Marsh-McBirney flow meter. Velocity and depth were measured at five or ten equally distributed locations perpendicular to the stream to enable an accurate estimate of total discharge. Discharge was not measured if the stream was too high to cross safely. Thus, a few of the spring discharge data at Keuka Outlet on 3/24, 3/31, 5/5 in 2010; and 3/2, 3/9, 3/11, 3/23, 4/6, 4/21, 5/4 & 5/12 in 2011 were supplemented with mean daily discharge USGS gauge data recorded at the sample site. Unfortunately, the priority to maintain Keuka Lake levels at the outlet dam overshadowed a natural stream hydrograph at the downstream sample site and thus precluded using this record to interpolate flows at neighboring streams.

Laboratory Analyses: Laboratory procedures for nutrients, chlorophyll-a, and total suspended solid analyses followed standard limnological techniques (Wetzel and Likens, 2000). Once back in the lab, a known volume of sample water was filtered through pre-weighed, 0.45 μm glass-fiber filters for total suspended sediment (TSS) analysis. The glass-fiber filter and residue were dried at 80°C for 24 hours, and stream water filtrate stored at 4°C until nutrient analyses. The weight gain and filtered volume determined the total suspended sediment concentration. A known volume of lake water was also filtered onsite through a Gelman HA 0.45 μm membrane filter for chlorophyll analysis. The membrane filter and residue was kept frozen until chlorophyll analysis after acetone extraction by spectrophotometer. The filtrate was stored at 4°C until soluble reactive (dissolved) phosphate, nitrate and dissolved silica colorimetric analyses by spectrophotometer. A third, unfiltered water sample was collected and analyzed for total phosphates by spectrophotometer following the SRP procedures after digestion of the organic-rich particulates in hot (100ºC) persulfate for 1 hour. The phosphate procedures utilized a 10-cm cuvette for added precision. Each spectrophotometer analysis measured blanks and standards in triplicate. Laboratory precision was determined by analyzing replicate tests on the same water sample on a number of occasions resulting in the following mean standard deviations: total suspended solids ±0.2 mg/L, phosphate ±0.1 μg/L, silica ±5 μg/L, and nitrate ±0.1 mg/L.

Major ion concentrations, specifically, sodium (Na^{+}), calcium (Ca^{+2}), magnesium (Mg^{+2}), potassium (K^{+}), chloride (Cl^{-}), and sulfate (SO_4^{-2}), were measured on filtered (0.45 μm) water samples by a Dionex DX-120 ion chromatograph with a precision of 1.0 mg/L. Atrazine, a common herbicide, was analyzed using the Atrazine RaPID Assay Kit that utilizes an enzyme-linked immunosorbent assay to detect atrazine (Omicron Environmental Diagnostics). The precision of the procedure was 0.1 μg/L. Water samples for total coliform and *E. coli* bacteria analyses were collected in sterile bags, kept at 4ºC until analysis in the lab using Hach's m-ColiBlue24 broth filtration technique, and reported as colony forming units per 100 mL of water (CFU/100mL). The precision was 260 for total coliform and 33 CFUs/100 mL for *E. coli* bacteria. Trihalomethanes (THMs) were analyzed on selected samples from the 2010 spring fieldwork. The methods utilized a HACH test kit (10132). For the plankton enumerations, over 100 individuals were identified to genus level and reported as date averaged relative percentages.

Halfman and Franklin (2007) detailed quality control measures performed in 2007 using over 100 sample splits that were analyzed by our research lab at HWS and a commercial laboratory. The comparison revealed good correlations (r^2 of 0.8 to 0.9). The HWS lab also had lower detection limits, enabling detection of concentrations as low as 0.1 mg/L for nitrates and 1 μg/L HWS for phosphates. Thus, the commercial lab could not provide data for 15% of the nitrate samples, 70% of the SRP samples, and 40% of the TP samples.

SENECA LAKE LIMNOLOGY

CTD Profiles, Results and Interpretations: The CTD temperature profiles were typical for a relatively deep lake in central New York (Fig. 5). A thermocline typically developed in early May as the epilimnion (surface waters) warmed above 4ºC and eventually warmed to ~25ºC by mid to late summer. The lake was thermally stratified throughout the remainder of each field season as the surface waters never cooled back to isothermal conditions (4ºC) by the last cruise of the year in November. The Scandling was unavailable during the winter season. Thus, data are not available to determine if the lake was dimictic or warm monomictic, i.e., undergoes overturn twice a year in the spring and the fall, or overturn only once a year throughout the winter and into the early spring each year. The last time Seneca Lake was completely covered with ice was in 1912 (Finger Lakes Times, 1/13/2012). The thermocline, the boundary between the warm epilimnion and cold hypolimnion (bottom waters), was typically at a depth of 20 m. However, its depth oscillated vertically in response to internal seiche activity, epilimnetic mixing by storm waves and spring warming and fall cooling of the epilimnion. Seiche activity was more obvious in the WQ buoy data (see below). Its presence and depth are fundamental to lake processes because it represents the stratification boundary between the warmer, less-dense, sunlit and nutrient poor epilimnion above the colder, more-dense, dark and nutrient enriched hypolimnion.

Specific conductance (salinity) was isopycnal (well mixed) in the isothermal early spring. It was just over 700 μS/cm (or ~0.33 ppt) in 2011. The salinity was not life threatening. The epilimnetic specific conductance slowly decreased by ~50 μS/cm throughout the stratified season and reflected its dilution by precipitation events and associated runoff. The hypolimnion salinity remained relatively constant when the lake was stratified but decreased from one year to the next when seasonal overturn mixed the less saline epilimnion into the remainder of the lake. The lake wide specific conductance decreased by an average of 10 μS/cm each year over the past decade. However, the decrease is less evident in the profiles during the past 5 years.

The epilimnetic dissolved oxygen (DO) concentrations decreased from the spring to summer and increased again in the fall. The seasonal progression reflected the seasonal warming and cooling of the epilimnion as DO concentrations remained saturated or nearly saturated throughout the field season. DO progressively decreased to 6 mg/L or ~40% of saturation just below the thermocline by the end of the stratified season. Seneca Lake is apparently deep enough and decomposition load low enough to restrict the bulk of the oxygen depletion to the upper hypolimnion. Over the past 5 years, the deficit in the upper hypolimnion has increased from concentrations around 7 mg/L down to 6 mg/L, however this is a short term trend as CTD profiles from 2005 and 2006 revealed deficits as low as 5 mg/L.

Photosynthetic active radiation (PAR) intensities decreased exponentially from a few 100 to a few 1,000 μE/cm^2-s at the surface to ~1% surface intensities at 10 to 30m depth, typically near the base of the epilimnion. The 1% surface light depth represents the minimum amount of solar energy required for algae to photosynthesize enough biomass to survive (net production = zero). The surface intensity reflected the season and cloud cover. The observed exponential decrease in light reflects the normal preferential absorption and conversion of longer wavelengths of light (infrared, red, orange, yellow) to heat, and scattering of shorter wavelengths of light (ultraviolet, violet, blue) back to the atmosphere. Seasonal changes were observed. Light penetration was deeper in the early spring, shallower in the summer months, and was inversely proportional to the density of algae in the water column. Occasionally PAR intensities would deviate to lower light levels at or near the surface from the typical exponential decay as the CTD travelled through the shadow of the boat.

Algae, as detected by fluorometer, were found throughout the epilimnion and occasionally extended into the metalimnion of the lake. Algal peak concentrations were up to 7 or 8 μg/L during algal blooms, and were typically located 5 to 20 m below the water's surface. The depth typically rose and fell with light availability (i.e., algal density), and depth or absence of the thermocline. The hypolimnion concentrations were less than 0.5 μg/L, as expected, because it is too dark for photosynthesis. During isothermal or nearly isothermal conditions algae were detected in deeper water and they were probably (with regret) vertically mixed into and out of the photic zone.

The turbidity was larger in the epilimnion, especially during algal blooms, with values up to 2 NTUs, much larger than 0.5 NTUs or less detected in the hypolimnion. The distribution reflected epilimnetic algae, and on rare occasions algal blooms with calcite precipitation (whiting events). Benthic nepheloid layers, lake floor zones of increased turbidity observed in other Finger Lakes (Halfman and Basnet, 2009), were not detected at the northern sites. In 2011, profiles from the early spring revealed larger turbidities through the mixed water column. The timing is concurrent with heavy rains in the early spring and/or significant nearshore erosion by storm waves and currents, both could introduce suspended sediments to the lake.

Buoy Data: The Seneca Lake WQ buoy revealed similar results water column profiles (Fig. 6). Two daily profiles compared to weekly CTD data however, revealed a number of new insights. First, the thermocline depth oscillated vertically on a quasi-weekly time frame by 10 to 15 meters. It suggests that wind stress set up internal seiche activity in the lake and its subsequent vertical seesaw motions oscillated the depth of the thermocline. Once set, the basin geometry sets the theoretical oscillation period at 1 to 2 days. Second, the specific conductance profiles revealed a similar oscillation of the thermocline, and confirm the decline in epilimnetic salinities through the stratified season. Third, the chlorophyll data revealed spring and fall algal blooms as expected by the onset and decay of summer stratification. However additional algal blooms were detected mid-summer during the stratified season. Some of these mid-summer blooms may be related to the "bottom up" inputs of nutrients, especially new phosphates entering the epilimnion by major runoff events, and/or mixing of hypolimnetic waters by internal seiche activity (e.g., Baldwin et al., 2001, Baldwin, 2002,). Alternatively, these blooms may be induced by the reduction of herbaceous zooplankton by "top down" ecological stressors like the carnivorous fishhook water flea, *C. pengoi*, and/or zooplanktivorous fish. The decrease in herbaceous zooplankton reduces their algal predation. Finally, the weather data revealed annual-scale variability (Fig. 6). For example, annual wind rose diagrams revealed more prevalent and faster southerly winds in 2011 than 2010. This variability impacts the lake because winds blowing parallel to the lake's long axis are more likely to set up internal seiche activity, and a change in prevailing weather patterns is also associated with wet vs. dry years.

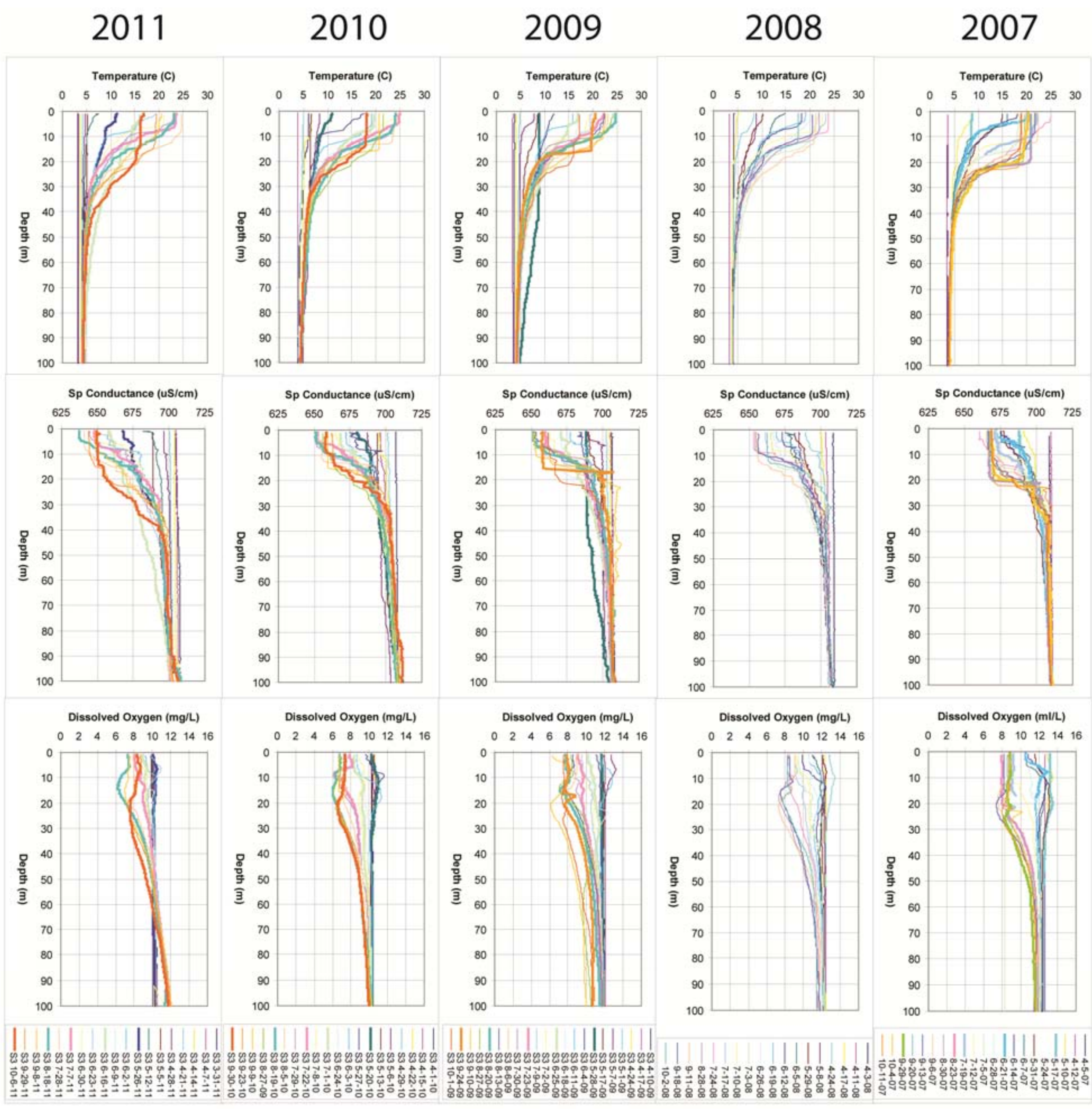


Fig. 5a. Site 3 Temperature, specific conductance and dissolved oxygen CTD profiles from 2007 through 2011.

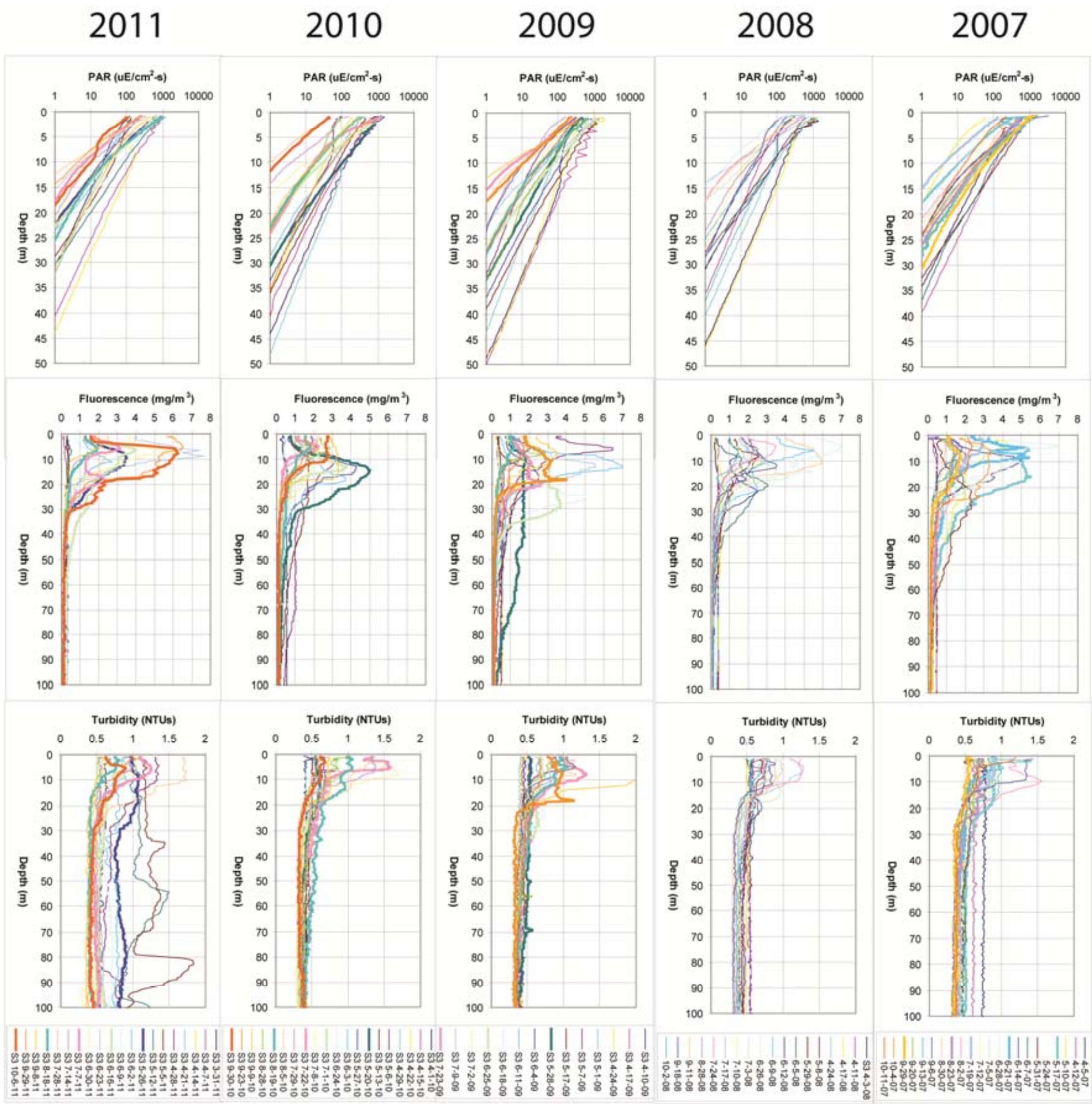
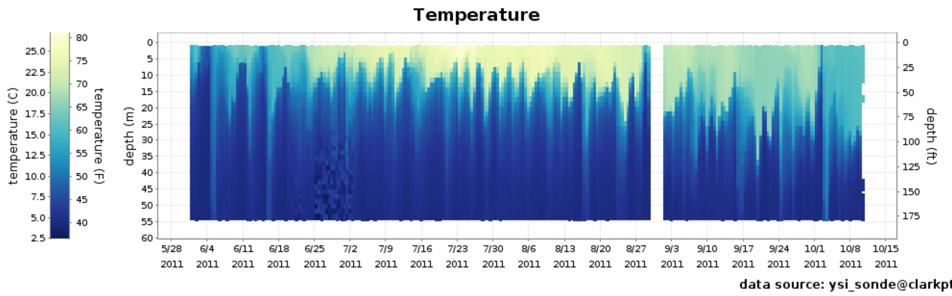
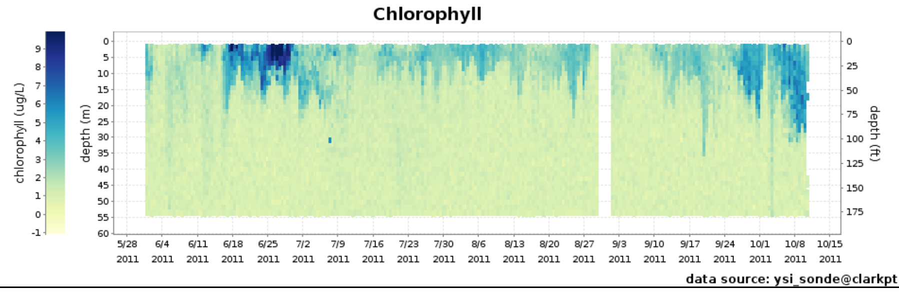


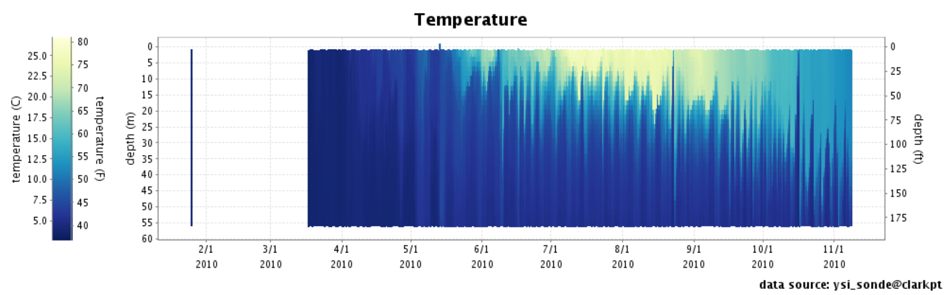
Fig. 5b. Site 3 PAR (light), fluorescence and turbidity CTD profiles from 2007 through 2011.

Lake Nutrient, Chlorophyll-a, Secchi Disk, & TSS Data: The total phosphate, soluble reactive phosphate, dissolved silica, nitrate, chlorophyll-a, secchi disk and TSS concentrations were not a health threat nor dictated an impaired lake (Table 4 and Fig. 7). The nitrate concentrations were always below the US EPA's MCL of 10 mg/L and total phosphate concentrations were always below NYS DEC's 20 μg/L threshold for impaired water bodies. The epilimnion to hypolimnion increase in nutrient concentrations and decrease in chlorophyll-a concentrations over the stratified season are consistent with the normal seasonal progression of algal growth and uptake and removal of nutrients from the epilimnion, and algal decomposition and nutrient release by bacteria to the hypolimnion. Silica concentrations were occasionally at the lower limit for diatom growth, and point to a potential reason for algal succession through the spring, summer, and fall seasons.



2011 Data





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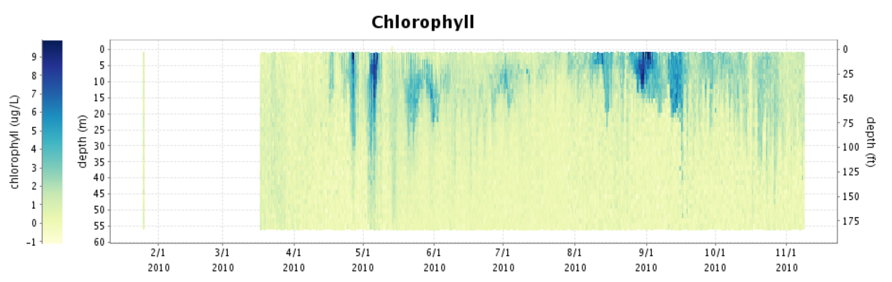
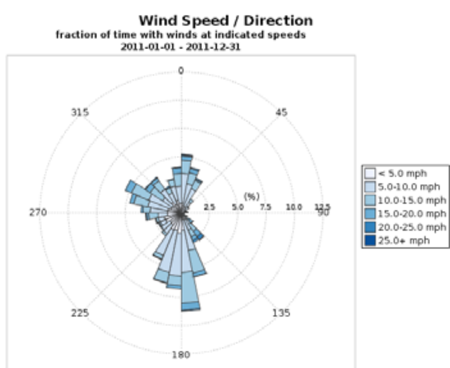
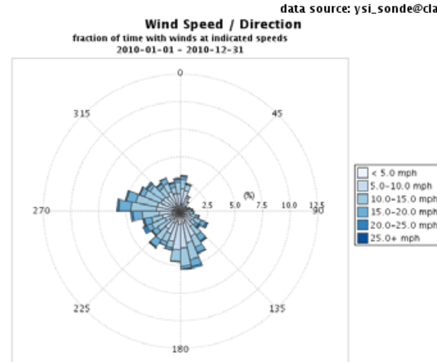




2011 Data



2010 Data

Fig. 6. Selected Seneca Lake WQ buoy data from 2010 and 2011. The other years revealed similar patterns.

Table 4. Annual Mean Lake Data.

Year	Secchi Depth (m)	TSS (mg/L)		Total Phosphate (TP, μg/L, P)		Phosphate (SRP, μg/L, P)		Nitrate (mg/L, N)		Silica (SRSi μg/L, Si)		Chlorophyll-a (μg/L or mg/m^3)	
		Surface	Bottom	Surface	Bottom	Surface	Bottom	Surface	Bottom	Surface	Bottom	Surface	Bottom
1991	4.4	0.7				0.4						3.7	
1992	3.7	0.8				0.3	1.5	0.4	0.5	100	340	4.5	
1993	4.4	0.8				0.5						4.6	
1994	5.7	0.8				1.2	3.2			125	262	2.2	1.1
1995	7.8	0.7				1.5	2.1			471	447	2.0	0.7
1996	7.5	0.8								1089	1481	1.6	0.7
1997	8.3	0.8	0.8			0.5	1.2	0.1	0.1	1426	1821	0.6	0.4
1998	6.2	0.8	0.6			2.4	3.6	0.3	0.3	267	518	1.5	0.4
1999	7.2	0.9	0.7			3.3	3.5	0.3	0.3	216	268	1.2	0.9
2000	7.3	1.3	0.9			2.1	3.3	0.2	0.2	252	356	1.2	1.0
2001	6.9	0.9	0.7			1.7	2.0	0.2	0.3	239	323	1.7	0.8
2002	7.0	1.2	0.8			1.1	3.3	0.3	0.4	215	300	0.6	0.4
2003	7.1	0.7	0.5			1.4	1.5	0.4	0.5	301	382	1.0	0.6
2004	6.3	1.2	0.9			0.9	1.8	0.3	0.4	387	475	2.3	1.0
2005	6.4	1.3	0.9			1.9	2.3	0.4	0.5	252	402	1.7	1.0
2006	6.3	1.2	0.6	9.1	10.7	1.3	2.9	0.4	0.5	237	377	2.0	0.5
2007	5.5	1.5	0.9	8.6	8.3	1.5	3.4	0.4	0.4	246	367	3.4	1.0
2008	6.6	1.4	0.7	10.3	10.3	1.6	3.1	0.2	0.4	303	472	3.5	1.0
2009	6.4	1.5	0.7	7.6	8.0	0.8	1.6	0.3	0.4	338	647	3.1	0.9
2010	5.3	1.4	0.5	7.3	6.2	1.1	2.3	0.3	0.5	294	414	3.2	0.5
2011	5.4	1.8	0.9	15.3	14.5	2.0	4.5	0.3	0.4	337	527	2.8	0.6
2000-2011 Average	6.3	1.1	0.7	9.7	9.7	1.4	2.6	0.3	0.4	373	536	2.3	0.7

Note: mg/L ~ ppb, and mg/L ~ ppm

Seneca Lake Concentration Trends: As initially discussed in Halfman and Franklin (2007), significant changes were observed in the epilimnetic limnology of Seneca Lake over the past 20 years. The data divided into two primary, decade-scale trends: 1992 to 1997, and 1998 to 2011, with some smaller-scale deviations along each trend. For example, annual average secchi disk depths became progressively deeper from 3 to 4 m in the early 1990's to 7 to 8 m by the end of 1997, and since then decreased to nearly 5 m by 2011. Imprinted on the overall trend are annual deviations from the decade-scale trend, for example, deeper depths were observed in 2008 and 2009 and shallower depths in 1998 than the overall trend. Chlorophyll-a concentrations decreased from an annual average of ~4.5 μg/L in the early 1990's to 0.6 μg/L by 1997, and then steadily increased to 2.5 to 3.5 μg/L by 2010 and 2011 with a deviation to larger concentrations, up to 3 to 4 μg/L, in 2007. Dissolved silica surface water concentrations rapidly increased from ~100 μg/L in 1994 to 1,800 μg/L in 1996, to then reverse and rapidly decreased to below 300 μg/L by 1998 and remain low through the end of the survey. Nitrate concentrations were 0.1 mg/L in 1997 (earlier data was not available) and increased to 0.3 mg/L by 2011 with deviations to larger concentrations, up to 0.42 mg/L in 2007. Dissolved phosphate (SRP) concentrations above 3.0 μg/L in 1999, and typically between 1.0 to 1.5 μg L before and since. The pre-1995 SRP data were scarce. Total suspended solid concentrations remained steady at 0.7 mg/L until 1997 then increased steadily each year to 1.8 mg/L by 2011.

The 1992 through 1997 trends were consistent with increased grazing by the growing population of filter-feeding zebra mussels in the early 1990's (Halfman et al., 2001, Halfman and Franklin, 2007). Zebra mussels were first detected in 1992, and successfully colonized Seneca Lake within a few years. Their introduction and establishment had implications on the limnology of the lake (Fig. 8). As more algae were consumed, algal concentrations progressively declined. The nutrients were progressively sequestered into the mussel biomass. Fewer nutrients promoted declining algal productivity. In contrast, dissolved silica concentrations significantly increased from 1995 through and including 1997. The increase was consistent with mussel predation. Declining diatoms populations within the peak of mussel grazing presumably reduced the diatom uptake of dissolved silica from the lake.

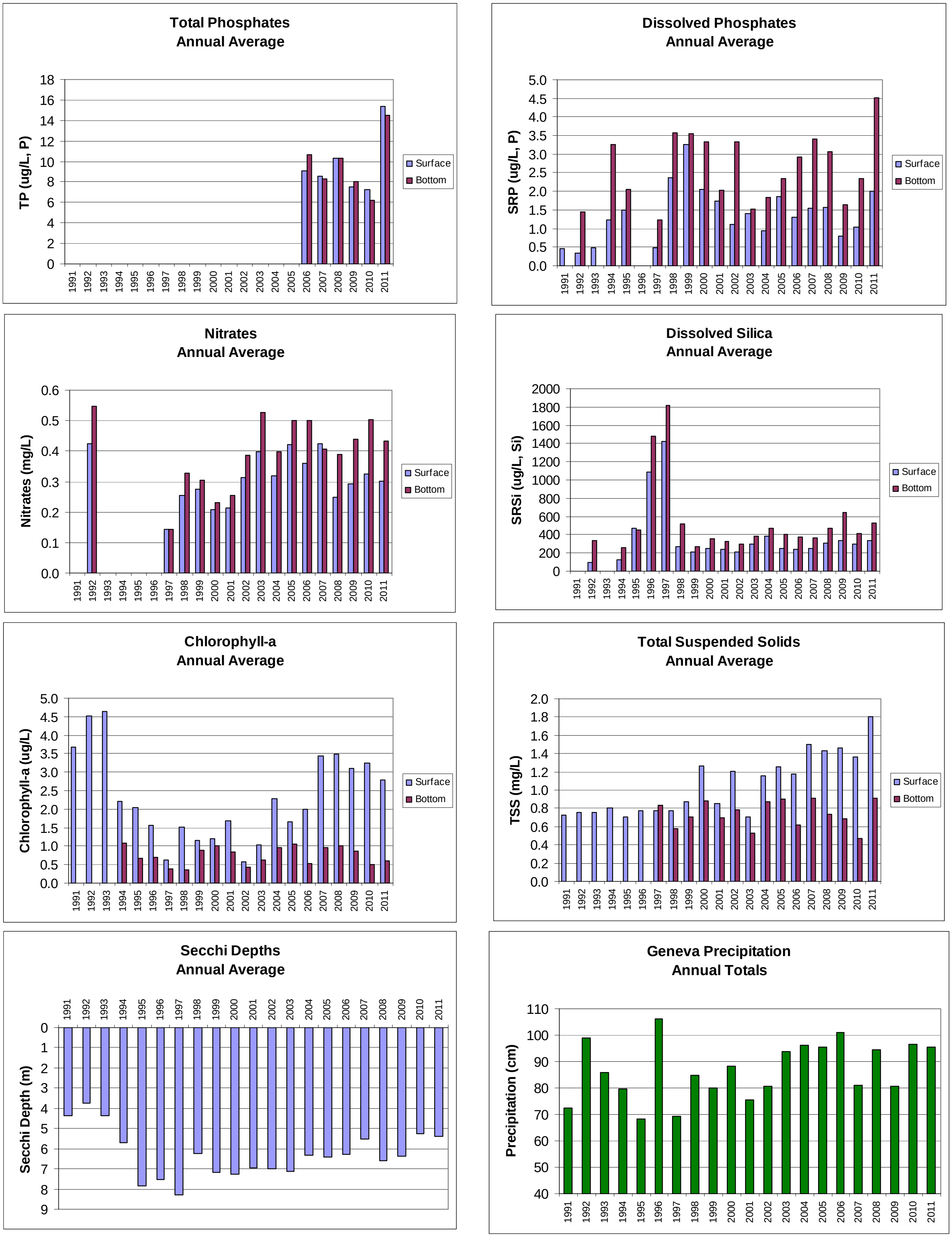


Fig. 7. Annual average secchi disk depths and surface (blue) and bottom water (magenta) total phosphate (TP), dissolved phosphate (SRP), nitrate, dissolved silica (SRSi), chlorophyll-a, and total suspended solid (TSS) concentrations. Annual precipitation totals were from Cornell's Agricultural Experiment Station in Geneva, NY.

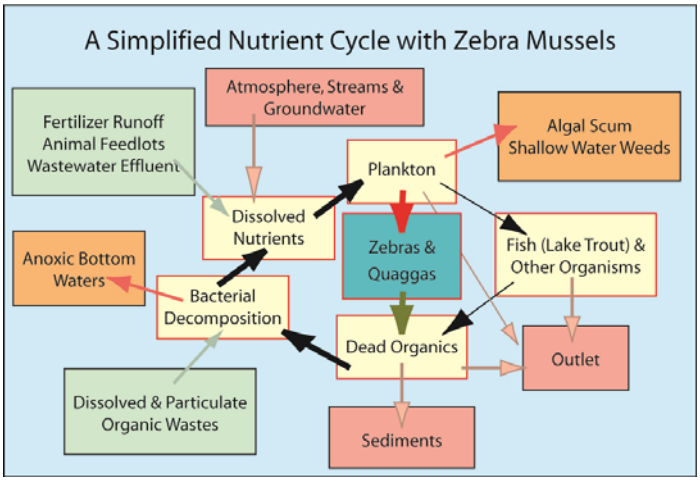


Fig. 8. Zebra and quagga mussel impact on the nutrient cycle and algal concentrations. Initial sequestering of nutrients in their biomass before their 1998 die off and subsequent release of the sequestered nutrients after the die off are shown by the red and green arrows, respectively, into and out of the zebra biomass box.

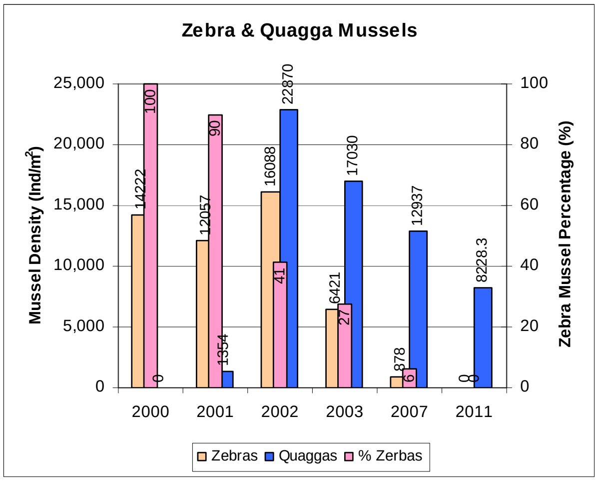
The trend reversed after the major die off of zebra mussels in 1998. The lake became progressively more impaired since, as shown by shallower secchi dish depths and larger chlorophyll concentrations (Hoering and Halfman, 2010, Halfman et al., 2010). These trends continued throughout the past decade as the lake transformed from an oligotrophic lake to a borderline oligotrophic-mesotrophic lake. Earlier reports suggested that the die off and associated bacterial decomposition of the mussel biomass released the previously sequestered nutrients back into the water column during 1998 and 1999, increasing TP, N, SRP and algal concentrations and decreasing secchi disk depths.

Various factors contributed to the decline in water quality over the past decade. First, the both zebra and quagga mussel populations declined since 2002 (B Zhu and D Dittman, unpublished data, Shelley et al., 2003, Fig. 9a). Zebra mussels posted the largest decline, to <1% of the total mussel population in 2011. Thus, the mussel impact on and reduction of the algal populations probably decreased as well. Unfortunately, these conclusions are tentative because the data was collected from a variety of water depths and site locations, and mussel populations are depth dependent (Fig. 9b). Second, nutrient loading could have stimulated algal growth and decreased secchi disk depths. The stream hydrogeochemistry and the watershed phosphorus budget sections below highlight the nutrient loading issue. Finally, “top down” predation pressures on herbaceous zooplankton would create a decline in water quality. For more details, see Brown, 2011.

Seasonal patterns in the limnology of the lake were also observed (Fig. 10). Secchi disk depths became progressively deeper from 2001 to 2011 in the early spring but were progressively shallower in the summer and fall. Parallel trends were also detected in the chlorophyll, TSS and SRP data, i.e., smaller algal concentrations in the spring but progressive larger algal concentrations in the summer and the fall. Total phosphate, nitrate and dissolved silica data were less consistent. The exact reason for the increased water clarity in the spring was unclear but perhaps related to mussels grazing and light limitations as both limit algal growth in the near isothermal spring. Clearly, shallow secchi depths and larger chlorophyll concentrations in the summer and fall were critical to the overall change in the annual concentrations over the past decade.

Entire Lake Surveys: Surface water analyses from the full lake surveys in 1998, 1999 and again in 2010 and 2011, revealed minimal systematic changes from one end of the lake to the other, beyond the expected patchiness in open water, minor variability from one site to the other (Fig.

11). Secchi disk depths revealed shallower depths towards the southern end of the lake on a few surveys in 2010 and 2011. However, the trend was not detected on every survey, the variability in secchi depths was within the scatter in the data, and the trend was not consistent with total suspended solids or chlorophyll data.



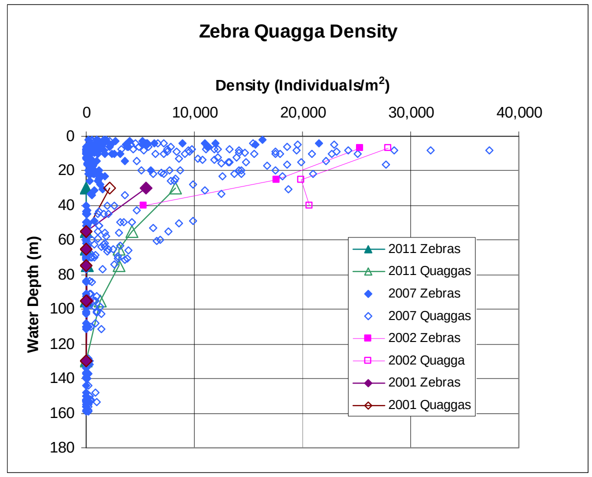
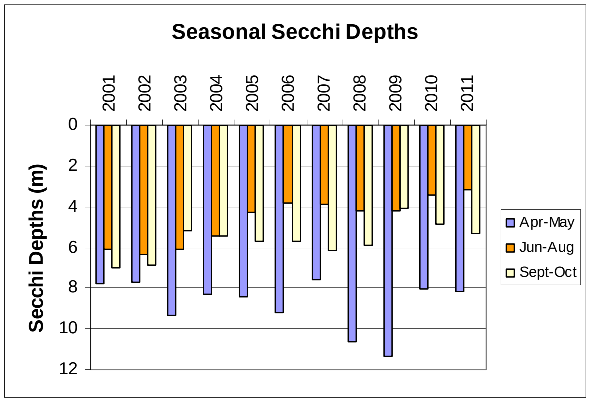
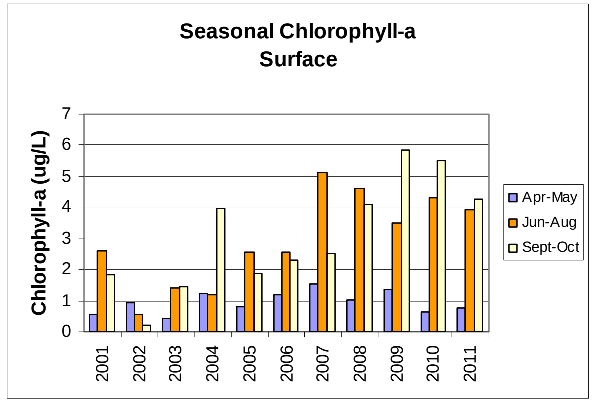


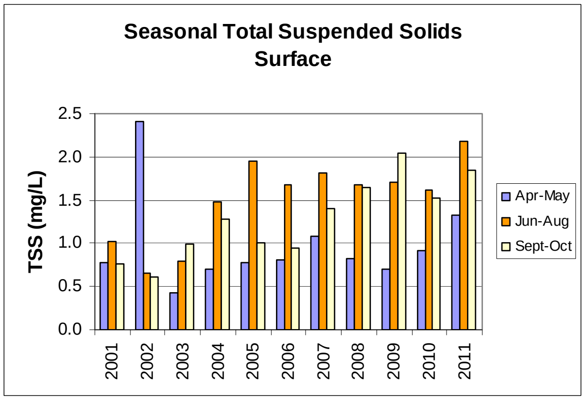
Fig. 9. Zebra and quagga mussel populations (left) and depth distributions (right) over the past decade (Geo-330 class data '00, '01, '03, B Zhu '07, B Shelley '02, D Dittman '01 & '11, unpublished data, Shelley et al., 2003).

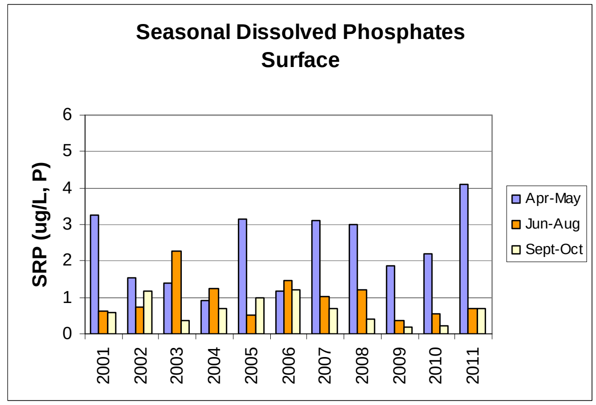
The full-lake CTD profiles from 2010 and 2011 were, in general, similar to the northern sites, except for three observations (Fig. 12). First, dissolved oxygen concentrations progressively declined in the deepest water of the hypolimnion from early spring to late summer. The bulk of the decline was just below the thermocline with smaller decreases at deeper depths in the hypolimnion. It reaffirmed that the huge volume of water in the hypolimnion has a tremendous capacity to accommodate biological respiration before it becomes anoxic. Second, benthic nepheloid layers detected in many Finger Lakes (Halfman and Basnet, 2009), were not observed in the deepest Seneca Lake profiles, except on 8/28/10. Benthic nepheloid layers commonly result from the accumulation of fluvial sediments, resuspension events and/or algal remains. Perhaps the survey dates missed major runoff or resuspension events, lacked enough dead algae, and/or their inputs were insignificant compared to the lake size. Finally, the specific conductance (salinity) profiles revealed more saline water just above the lake floor in the deepest depths in the lake. The difference in salinity was minimal, 0.02 ppt, but was unique to Seneca among the other Finger Lakes. Saline groundwater or salt mine waste water inputs would be dense enough to accumulate and form the layer along the lake floor.

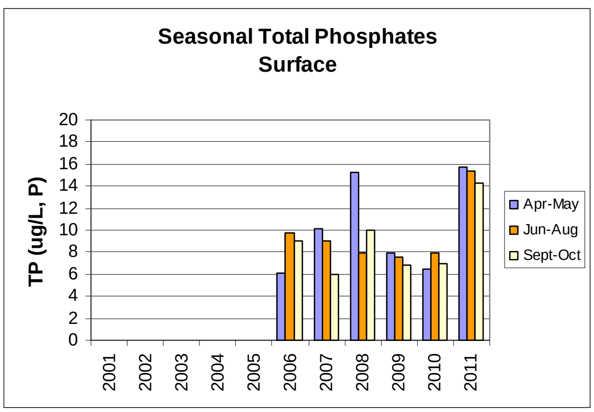
Phosphorus Limitation & Trophic Status: In Seneca Lake, P:N ratios in the water column ranged from 1:60 to 1:360 based on annual average nitrate and dissolved phosphate data, and averaged 1:160 over the past decade (Fig. 13). The P:N ratio required by algae is 1:7 (Redfield Ratio), so the significantly larger Seneca Lake ratios dictate that phosphate was the limiting nutrient, like most of the other Finger Lakes. It also implies that phosphate loading would stimulate more algal growth and move the lake to a more productive system.

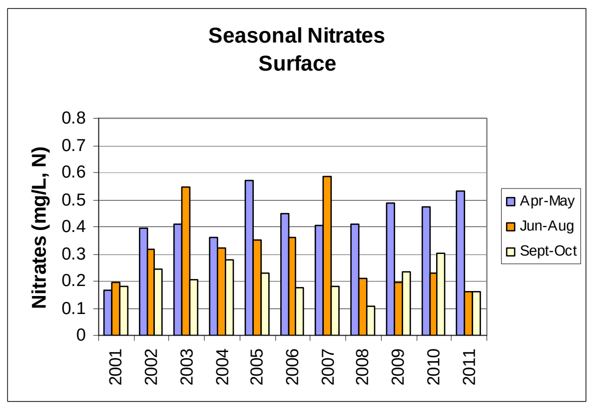












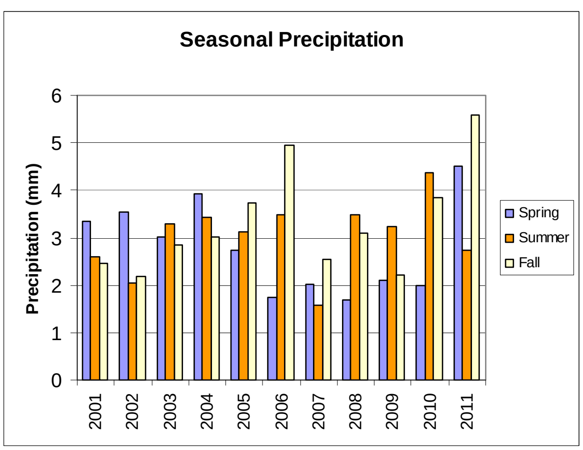


Fig. 10. Seasonal changes in secchi disk depths and surface water chlorophyll, total suspended solids, total phosphate, dissolved phosphate and nitrate concentrations. Seasonal precipitation data are from Cornell's NYS Agricultural Experiment Station in Geneva, NY.

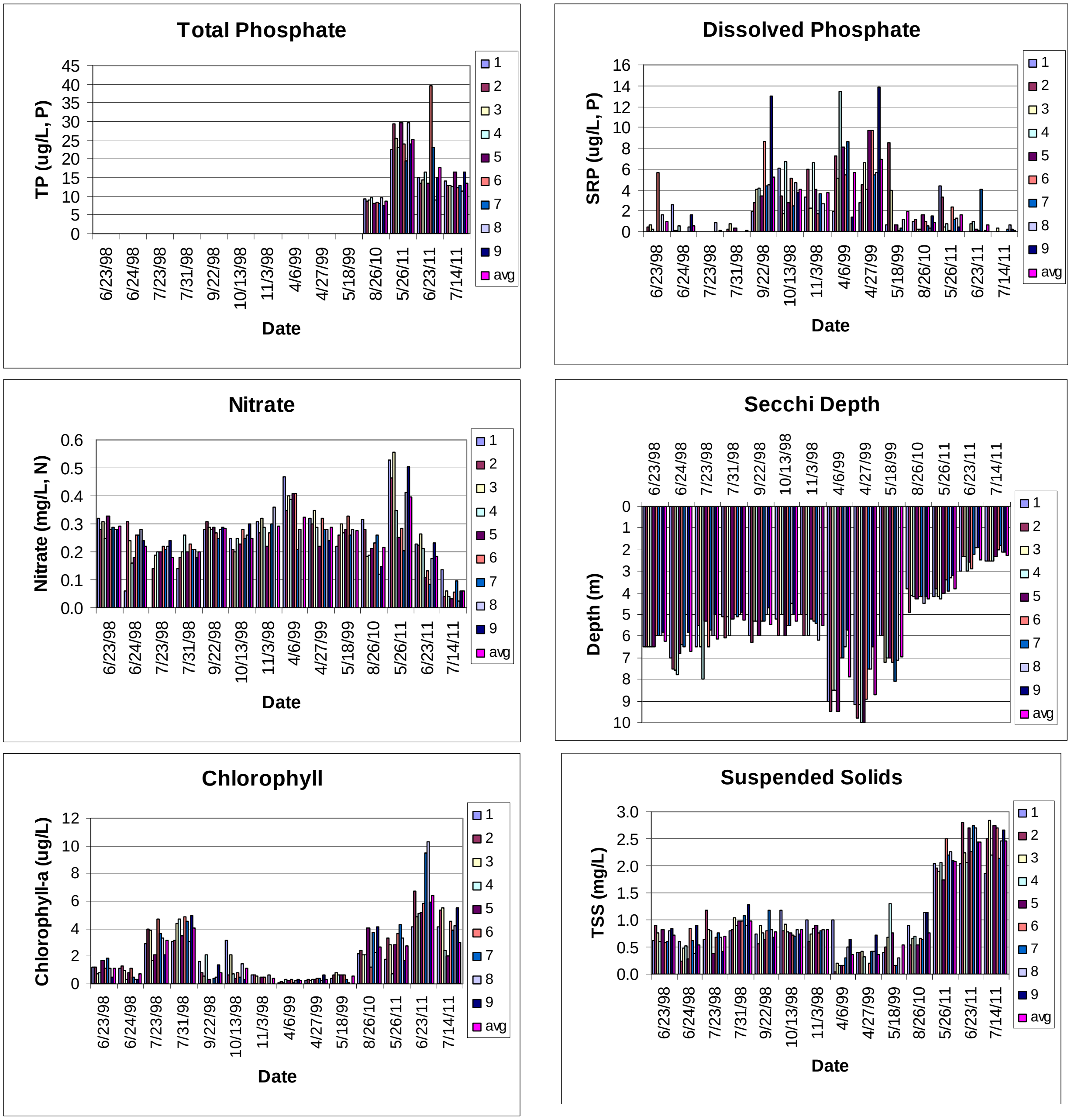


Fig. 11. Surface, water quality data from the full lake surveys. Data from each site and survey date are displayed.

Comparing the 2011 mean annual total phosphate, nitrate, dissolved oxygen, chlorophyll-a and secchi disk depths in Seneca Lake to the trophic status ranges for oligotrophic through eutrophic lakes indicates a borderline oligotrophic-mesotrophic lake. Annual mean total phosphate concentrations and hypolimnetic oxygen saturation data in 2011 were within the mesotrophic range however, secchi disk depths, chlorophyll and nitrate concentrations were oligotrophic (Table 1 & 4, Fig. 7). In 2006 and 2007, all of the parameters were in the oligotrophic range, although some were near the oligotrophic-mesotrophic cutoff. Thus, Seneca Lake has migrated to a borderline oligotrophic-mesotrophic lake, and water quality has declined over time.

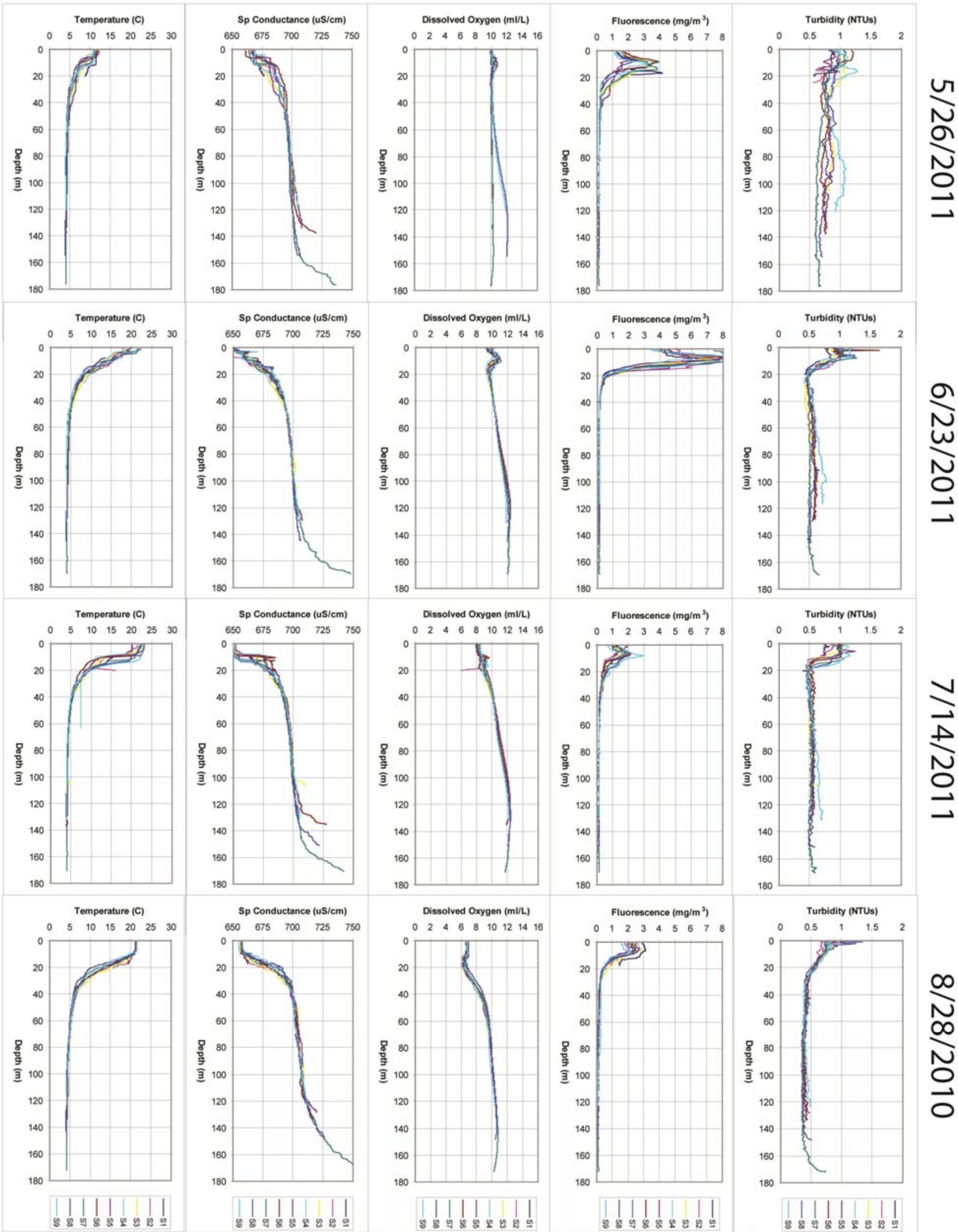


Fig. 12. CTD profiles from every full-lake survey in 2010 and 2011.

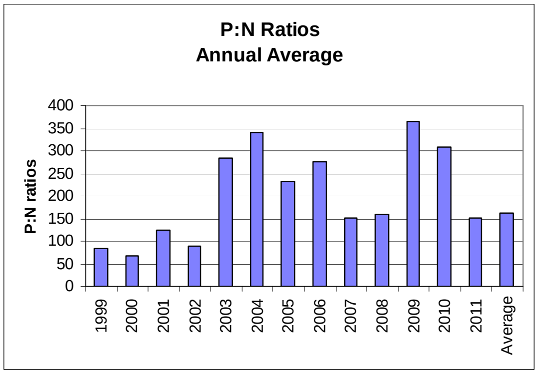


Fig. 13. Annual P:N ratios using annual mean nitrate and dissolved phosphate data.

STREAM DATA & NUTRIENT LOADING

Stream Discharge: The measured annual average stream discharge ranged from less than 0.1 to 7.9 m^3/s in the watershed (Table 5, Fig. 14). The smallest discharge was detected in the smallest watersheds, Plum Pt and Castle Creek, and largest was detected in the largest watersheds, Keuka Outlet and Catharine Creek. Thus, basin size is the primary determinant for stream discharge (Fig. 15).

Nutrient Concentrations in Streams: Nutrient loading impacted the watershed (Halfman & Franklin, 2007). All of the nutrient and TSS concentrations were larger in the streams than the lake (Table 5, Fig. 14). For example, fluvial total phosphate concentrations averaged over 45 μg/L but were below 10 μg/L in the lake, fluvial nitrate concentrations averaged 0.7 mg/L but averaged 0.3 mg/L in the lake over the past decade. Thus, Seneca has a nutrient loading problem, as do most agriculturally-rich watersheds in the Finger Lakes.

Nutrient concentrations varied from stream to stream. Wilson Creek, Kendig Creek, Castle Creek, Big Stream, and especially Reeder Creek revealed larger phosphate concentrations compared to the other tributaries. Unfortunately, no one reason accounts for these differences (Spitzer, 1999; Halfman and Franklin, 2007). Wilson and Kendig Creeks have high nutrient concentrations because they drain larger portions of agricultural land. The loading characteristically increased during an intense runoff event at Wilson Creek, and similar results were found elsewhere the Finger Lakes region (Kostick and Halfman, 2003; Halfman et al, 2011). Thus, runoff from agricultural nonpoint sources probably contributes to their larger phosphate concentrations.

In contrast, Big Stream drained much less agricultural land but had larger phosphate concentrations than Wilson and Kendig. Stream segment analysis in 2001 indicated that the Dundee wastewater treatment (WWT) facility and runoff of lawn fertilizers in the town were important sources of nutrients but still maintained stream concentrations below MCLs (Bowser, 2002). A similar segment analysis along the Keuka Outlet indicated that the Penn Yan wastewater treatment facility was not a significant point source of nutrients to Keuka Outlet (Hintz, 2004). Forested land was more prevalent in the other streams thus their lower concentrations. Keuka Outlet also has Keuka Lake in its headwaters. The lake serves as a large "retention pond", and sequesters nutrients delivered to Keuka Lake before they reached Seneca.

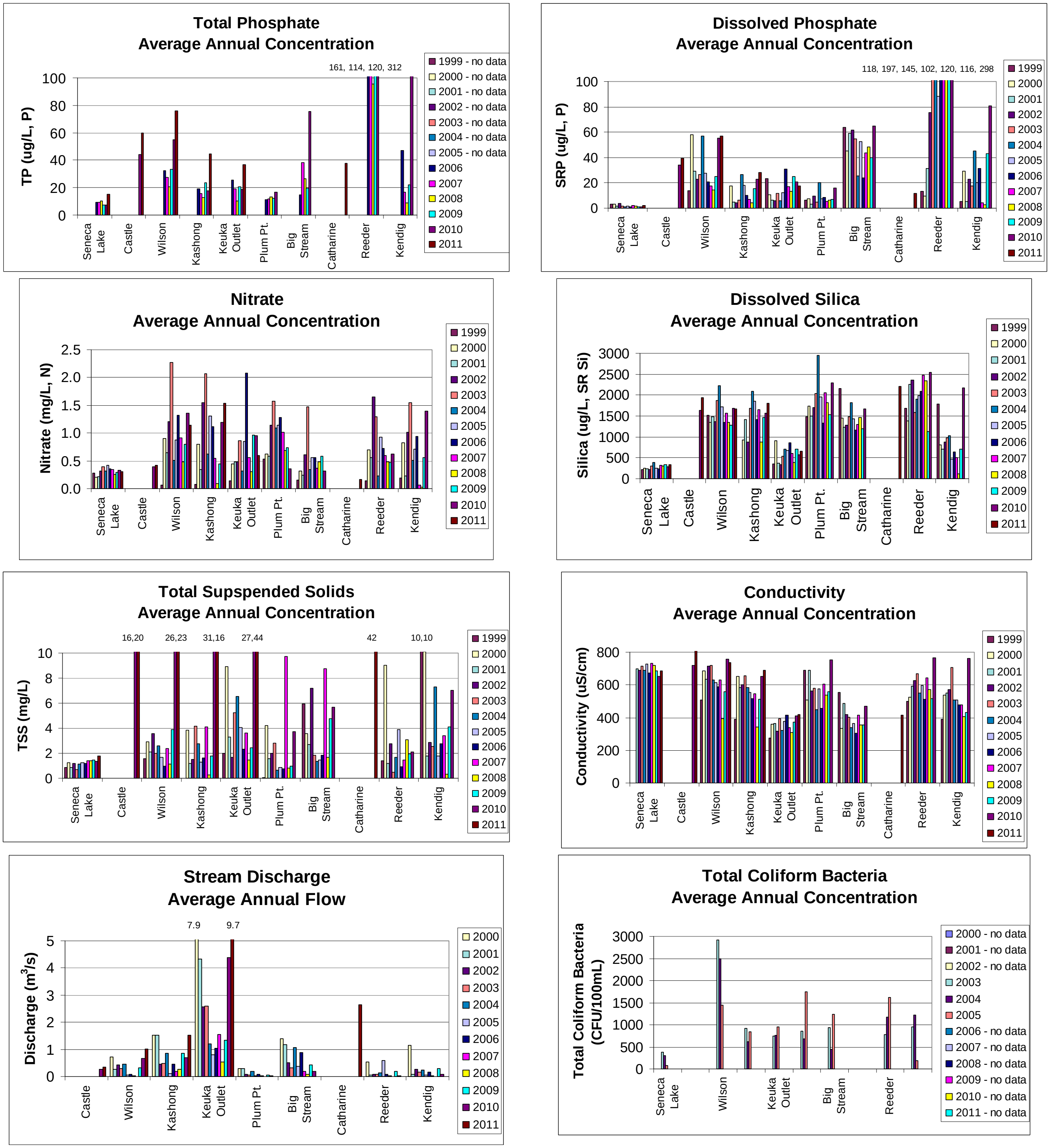


Fig. 14. Annual stream site-averaged discharge and water quality data. Castle Creek was added to the survey in 2010 and Catharine Creek in 2011. Annual Seneca Lake concentrations are included for comparison.

Table 5. Site-Averaged Annual Stream Data.

Conductivity (μS/cm)	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	Average
Seneca Lake			699	688	715	688	728	672	733	718	684	652	685	696
Castle												720	968	844
Wilson	509	687	636	715	721	630	615	587	632	395	560	756	737	629
Kashong	388	651	584	602	657	586	554	518	548	345	514	654	689	561
Keuka Outlet	275	359	362	316	393	321	378	413	339	310	374	409	419	359
Plum Pt.	690	510	689	563	580	447	575	456	603	537	560	756		580
Big Stream	555	335	485	417	404	341	365	305	414	355	355	469		400
Catharine													416	416
Reeder	499	524	591	626	668	550	595	514	643	569	517	766		589
Kendig	388	538	551	570	709	510	506	480	477	406	432	763		527
Nitrate (ppm, N)														
Seneca Lake	0.3	0.2	0.2	0.3	0.4	0.3	0.4	0.4	0.3	0.2	0.3	0.3	0.3	0.3
Castle												0.4	0.4	0.4
Wilson	0.1	0.9	0.6	1.2	2.3	0.5	0.9	1.3	0.9	0.5	0.8	1.4	1.1	1.0
Kashong	0.1	0.8	0.3	1.5	2.1	0.6	1.3	1.1	0.6	0.1	0.4	1.2	1.5	0.9
Keuka Outlet	0.1	0.4	0.5	0.5	0.9	0.3	0.9	2.1	0.6	0.3	1.0	0.9	0.6	0.7
Plum Pt.	0.5	0.6	0.6	1.1	1.6	1.1	1.1	1.3	1.0	0.7	0.7	0.4		0.9
Big Stream	0.1	0.3	0.2	0.6	1.5	0.3	0.6	0.6	0.4	0.5	0.6	0.3		0.5
Catharine													0.2	0.2
Reeder	0.1	0.7	0.6	1.6	1.3	0.2	0.9	0.7	0.6	0.5	0.5	0.6		0.7
Kendig	0.2	0.8	0.2	1.0	1.5	0.5	0.7	0.9	0.1	0.0	0.6	1.4		0.7
Total Phosphate (μg/L, P)														
Seneca Lake								9.1	9.1	10.3	7.6	7.3	15.3	9.8
Castle												44.2	59.7	51.9
Wilson								32.5	27.3	20.5	33.5	54.9	76.1	40.8
Kashong								19.3	15.8	12.9	23.6	17.7	44.5	22.3
Keuka Outlet								25.5	19.0	10.3	20.5	18.5	36.7	21.7
Plum Pt.								11.4	11.6	13.1	12.5	16.6		13.0
Big Stream								14.5	38.3	26.7	19.7	75.6		34.9
Catharine													37.9	37.9
Reeder								161.0	114.0	95.6	119.6	312.0		160.4
Kendig								47.1	16.8	9.0	21.9	105.7		40.1
Phosphate (μg/L, P)														
Seneca Lake	3.3	3.0	1.7	3.5	1.4	0.9	1.8	1.3	2.3	1.6	0.8	1.1	2.0	1.9
Castle												34.2	39.6	36.9
Wilson	13.6	57.8	29.4	22.8	26.5	57.0	27.7	20.8	17.3	14.5	25.1	55.5	56.7	32.7
Kashong		17.6	4.8	4.1	6.3	26.5	18.2	10.1	7.1	4.3	15.3	23.1	28.0	13.8
Keuka Outlet	23.3	10.9	6.5	5.7	11.5	5.7	12.4	31.1	16.9	13.4	24.7	20.9	17.8	15.4
Plum Pt.	6.6	7.6	3.2	9.6	4.7	20.1	7.5	8.5	5.1	6.2	6.9	15.9		8.5
Big Stream	63.9	45.3	59.2	61.6	54.8	25.4	52.7	24.0	43.6	48.3	40.1	64.8		48.6
Catharine													11.4	11.4
Reeder	13.5	9.5	31.6	75.4	117.5	196.8	88.3	145.3	102.4	120.0	115.7	297.8		109.5
Kendig	5.2	29.4	5.1	22.9	17.5	45.1	20.1	31.2	4.5	2.8	43.1	80.8		25.6
Silica (ug/L, Si)														
Seneca Lake	216	252	239	215	301	387	254	237	326	303	338	294	337	284
Castle												1641	1933	1787
Wilson	1518	1356	1491	1359	1876	2232	1717	1343	1567	1352	1277	1679	1666	1572
Kashong		921	1423	878	1679	2083	1848	1418	1643	875	1472	1567	1795	1467
Keuka Outlet	350	904	367	336	545	710	670	855	603	390	711	572	654	590
Plum Pt.	1480	1740	1502	1705	2033	2952	1959	1323	2062	1820	1535	2296		1867
Big Stream	2150	1445	1228	1285	1488	1821	1439	1166	1290	1465	1193	1663		1469
Catharine													2211	2211
Reeder	1694	1382	2255	2352	1584	1896	1994	2087	2480	2341	1128	2539		1978
Kendig	1794	802	705	875	980	1026	465	640	505	113	703	2175		899
Suspended Solids (mg/L)														
Seneca Lake	0.9	1.3	0.9	1.2	0.7	1.2	1.3	1.2	1.4	1.4	1.5	1.4	1.8	1.2
Castle												17.0	20.5	18.7
Wilson	1.6	2.9	2.1	3.6	2.0	2.6	1.7	1.0	2.4	1.1	3.9	25.7	23.1	5.7
Kashong		3.8	1.2	1.5	4.2	2.7	1.3	1.6	4.1	0.3	1.8	31.3	16.0	5.8
Keuka Outlet	2.0	8.9	3.3	1.7	5.2	6.5	4.0	2.3	3.6	1.5	2.4	27.5	43.7	8.7
Plum Pt.	0.0	4.2	1.6	2.0	2.8	0.7	0.9	0.8	9.7	0.8	1.0	3.7		2.3
Big Stream	5.9	3.6	2.7	7.2	1.9	1.3	1.5	1.9	8.8	1.7	4.8	5.6		3.9
Catharine													42.5	42.5
Reeder	1.4	9.0	1.2	2.7	0.5	1.7	3.9	0.9	1.5	3.1	2.0	2.1		2.5
Kendig	10.4	10.1	1.8	2.9	2.6	7.3	1.8	2.8	3.4	0.3	4.1	7.1		4.5
Discharge (m^3/s)														
Castle												0.3	0.3	0.3
Wilson		0.7	0.3	0.4	0.3	0.4	0.1	0.1	0.0	0.0	0.3	0.7	1.0	0.4
Kashong		1.5	1.5	0.4	0.5	0.9	0.1	0.5	0.2	0.3	0.9	0.7	1.5	0.7
Keuka Outlet		7.9	4.3	2.6	2.6	1.2	0.8	1.0	1.6	0.5	1.3	4.4	9.7	3.2
Plum Pt.		0.3	0.3	0.1	0.1	0.2	0.0	0.1	0.0	0.0	0.1	0.0		0.1
Big Stream		1.4	1.2	0.5	0.3	1.1	0.4	0.9	0.2	0.1	0.4	0.2		0.6
Catharine													2.6	2.6
Reeder		0.5	0.1	0.1	0.1	0.1	0.6	0.1	0.0	0.0	0.2	0.0		0.2
Kendig		1.2	0.1	0.3	0.2	0.2	0.0	0.2	0.0	0.0	0.3	0.1		0.2

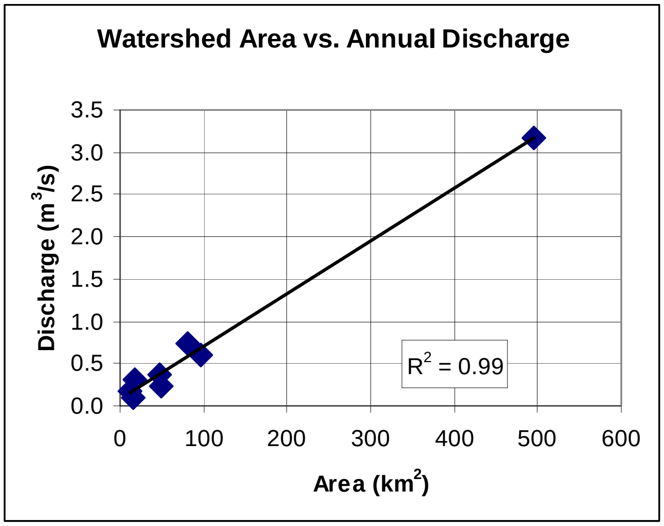


Fig. 15. Annual mean discharge vs. watershed area.

Catharine Creek revealed larger total suspended solid concentrations, but smaller phosphates (both TP and SRP), nitrates and specific conductance than the other streams. It drains more forested land than the other surveyed watersheds (~60% compared to 15 to 18% forested land). As expected, forests yield fewer nutrients to the lake than agricultural watersheds. The larger suspended solid concentrations in Catharine were inconsistent with forested watersheds, and perhaps reflected logging, construction and/or gravel pit practices in the watershed. Fillmore Creek in the Owasco watershed, and North McMillan Creek in the Conesus watershed, both heavily forested watersheds, revealed similar minimal nutrient loading to their lakes compared to their neighboring agricultural-rich streams (Halfman et al., 2011).

The largest concentrations of SRP and TP were consistently detected in Reeder Creek. This "honor" started in 2002 when SRP concentrations rose from typical tributary values of ~20 μg/L to 100 μg/L or more. Pig farms entered the region, and the former Seneca Army Depot was systematically disposing of and exploding old, phosphate-rich, munitions at this time. Both could contribute to the initial increase in 2002 but only pig farms persisted through 2011.

Finally, annual mean discharges, TP and TSS concentrations were larger in 2010 and 2011 in Castle, Wilson, Kashong, and Keuka Outlet than previous years, and most of the other tributaries. These four streams were sampled year round since 2010, whereas all streams were only sampled in the summer season during earlier years. The seasonal analysis (see below) revealed larger discharges, concentrations and fluxes during the winter and/or spring compared to the summer months, which dictated this difference.

Nutrient Fluxes: Flux is defined as the mass of a substance per unit time. Water analyses measure concentrations, a mass per unit volume of water. The flux of a substance is calculated by multiplying its concentration by the stream discharge, or dimensionally, multiplying mass/volume by volume/time, resulting in a mass/time. This distinction is important because a stream with a large concentration but small discharge will provide a similar flux as another stream with a small concentration but large discharge. Fluxes are better than concentrations at identifying sources to and sinks from a body of water because they directly measure the movement of the substance instead of concentrations which are also influenced by changing the amount of water. Fluxes are also the basis for any mass balance calculations to determine if nutrients are increasing or decreasing in a lake over time.

The largest fluxes were from streams with the largest basin areas (Table 6, Fig. 16). For example, Keuka Outlet and Catharine Creek, the largest streams sampled, revealed annual average fluxes of 7.6 and 8.7 kg/day for total phosphates compared to loads below 2 kg/day in the other streams, and 4,800 and 9,700 kg/day for total suspended solids vs. 500 kg/day in the other streams. The fluxes were artificially elevated in Catharine Creek as its average was based on samples throughout the year whereas the other streams also averaged summer-only data collected before 2010. The smallest fluxes were from the smallest watersheds, like Plum Pt. and Reeder Creek, adding only 0.04 and 0.8 kg/day for TP, and 20 and 65 kg/day for TSS over the past decade. These trends were interesting because Keuka Outlet revealed one of the smaller concentrations and Redder Creek one of the largest concentrations for TP. Castle Creek, another small watershed, discharged as much TP (1.4 kg/day), SRP (1.0 kg/day), and TSS (500 kg/day) as its larger and more agricultural neighbors, Wilson and Kashong Creeks. Perhaps Castle Creek's elevated flux reflected drainage of an urban area, and/or annual averages from year round samples. TP, SRP, TSS and nitrate fluxes correlated to basin size (r^2 from 0.63 and 0.85, e.g., Fig. 17).

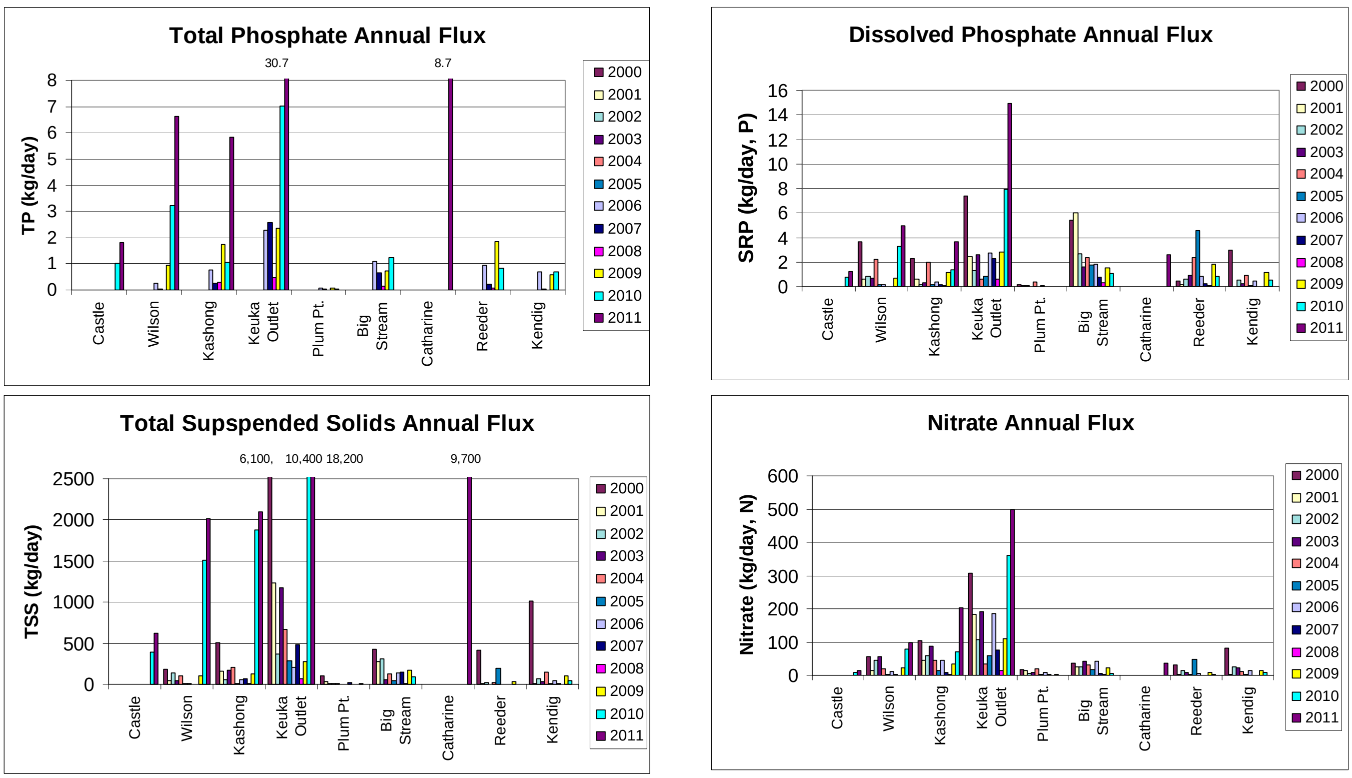


Fig. 16. Annual average fluxes of nutrients and suspended sediments to Seneca Lake.

Seasonal Changes in Stream Data: Stream discharge, concentration and flux of nutrients and suspended sediments changed seasonally (Fig. 18). These changes are critical for long term comparisons because the pre-2010 samples were typically restricted to the late spring and early summer, whereas the post-2010 samples were collected year round. Stream discharge was largest in the winter and/or spring and smallest in the summer and fall. Whether the season was winter or spring was dependent on the timing of snow melt and "spring" rains. The late and wet spring in 2011 compared to 2010 delayed the seasonal maximum discharge in 2011 from the winter to the spring. The anomalous large fall discharge at Keuka Outlet reflects the dam on Keuka Lake. Water releases focused on maintaining lake levels within acceptable limits for Keuka Lake and the occasional kayak competition along its course. Water was routinely released in the fall to lower the lake to winter levels. TSS concentrations were larger in the winter or spring and related to the timing of the early spring rains and snow melt. Seasonality for the other parameters was most apparent in their fluxes, with more material entering the lake in the late winter or early spring.

Table 6. Site-Averaged Annual Stream Fluxes.

Nitrate Flux (kg/day, N)	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	Average
Castle											9.1	12.8	10.9
Wilson	56.6	14.1	45.6	56.9	19.7	4.0	10.4	1.5	0.0	22.4	79.2	99.5	34.2
Kashong	105.0	45.0	58.9	87.5	45.5	13.2	44.0	9.2	2.0	32.5	71.7	201.9	59.7
Keuka Outlet	306.7	183.1	107.1	192.2	32.8	59.8	185.5	75.2	14.1	110.2	359.8	497.3	177.0
Plum Pt.	15.5	14.3	6.9	8.8	18.6	3.2	9.1	2.1	0.5	3.9	0.8		7.6
Big Stream	37.7	24.9	25.9	42.1	32.1	18.2	41.9	6.3	2.9	22.1	5.1		23.6
Catharine												37.4	37.4
Reeder	32.3	3.2	12.7	9.7	2.7	48.0	4.3	1.2	0.3	7.3	1.7		11.2
Kendig	82.6	1.5	24.3	21.7	10.1	3.0	14.0	0.2	0.0	14.2	9.2		16.4
TP Flux (kg/day, P)													
Castle											1.0	1.8	1.4
Wilson							0.3	0.0	0.0	0.9	3.2	6.6	1.8
Kashong							0.8	0.3	0.3	1.7	1.1	5.8	1.7
Keuka Outlet							2.3	2.6	0.5	2.4	7.0	30.7	7.6
Plum Pt.							0.1	0.0	0.0	0.1	0.0		0.04
Big Stream							1.1	0.7	0.2	0.7	1.2		0.8
Catharine												8.7	8.7
Reeder							1.0	0.2	0.1	1.9	0.8		0.8
Kendig							0.7	0.0	0.0	0.6	0.7		0.4
SRP Flux (kg/day, P)													
Castle											0.8	1.2	1.0
Wilson	3.6	0.6	0.9	0.7	2.2	0.1	0.2	0.0	0.0	0.7	3.3	4.9	1.4
Kashong	2.3	0.6	0.2	0.3	2.0	0.2	0.4	0.1	0.1	1.1	1.4	3.7	1.0
Keuka Outlet	7.4	2.4	1.3	2.6	0.6	0.9	2.8	2.3	0.6	2.8	7.9	14.9	3.9
Plum Pt.	0.2	0.1	0.1	0.0	0.3	0.0	0.1	0.0	0.0	0.0	0.0		0.1
Big Stream	5.4	6.0	2.6	1.6	2.4	1.7	1.8	0.7	0.3	1.5	1.0		2.3
Catharine												2.6	2.6
Reeder	0.4	0.2	0.6	0.9	2.3	4.6	0.9	0.2	0.1	1.8	0.8		1.2
Kendig	2.9	0.0	0.6	0.2	0.9	0.1	0.5	0.0	0.0	1.1	0.5		0.6
Silica Flux (kg/day, Si)													
Castle											37.9	58.2	48.1
Wilson	85.0	32.7	51.3	47.1	85.8	7.8	10.6	2.6	0.1	35.6	98.3	145.1	50.2
Kashong	121.1	186.2	33.5	71.0	153.7	18.7	55.6	27.5	20.7	108.7	94.2	235.7	93.9
Keuka Outlet	617.1	137.9	74.8	121.8	73.4	46.9	76.2	81.4	18.0	81.7	216.9	547.6	174.5
Plum Pt.	43.2	37.3	10.4	11.4	50.7	5.4	9.4	4.3	1.3	8.0	4.8		16.9
Big Stream	173.6	125.1	55.0	42.7	168.5	47.4	87.8	22.1	8.9	44.7	26.7		72.9
Catharine												505.1	505.1
Reeder	63.8	12.9	18.1	12.0	22.6	103.4	12.4	4.9	1.5	17.5	6.8		25.1
Kendig	80.1	4.5	21.1	13.7	20.6	1.9	9.5	1.4	0.1	18.1	14.4		16.9
TSS Flux (kg/day)													
Castle											392	617	505
Wilson	182	46	134	50	100	7	8	4	0	108	1507	2016	347
Kashong	503	156	58	177	202	13	63	69	6	130	1878	2099	446
Keuka Outlet	6073	1228	369	1171	673	282	205	489	68	281	10407	36636	4824
Plum Pt.	105	39	12	16	12	2	5	20	1	5	8		20
Big Stream	427	277	307	53	123	48	140	150	10	178	91		164
Catharine												9703	9703
Reeder	417	7	21	4	20	201	5	3	2	31	6		65
Kendig	1009	12	69	36	146	7	41	9	0	106	47		135

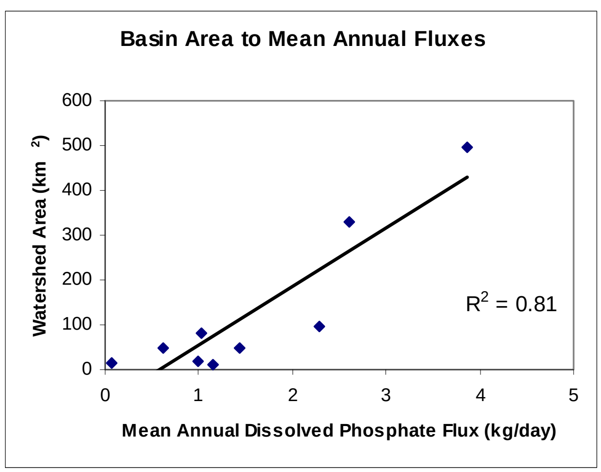


Fig. 17. Comparison of basin areas to annual average SRP fluxes to the lake.

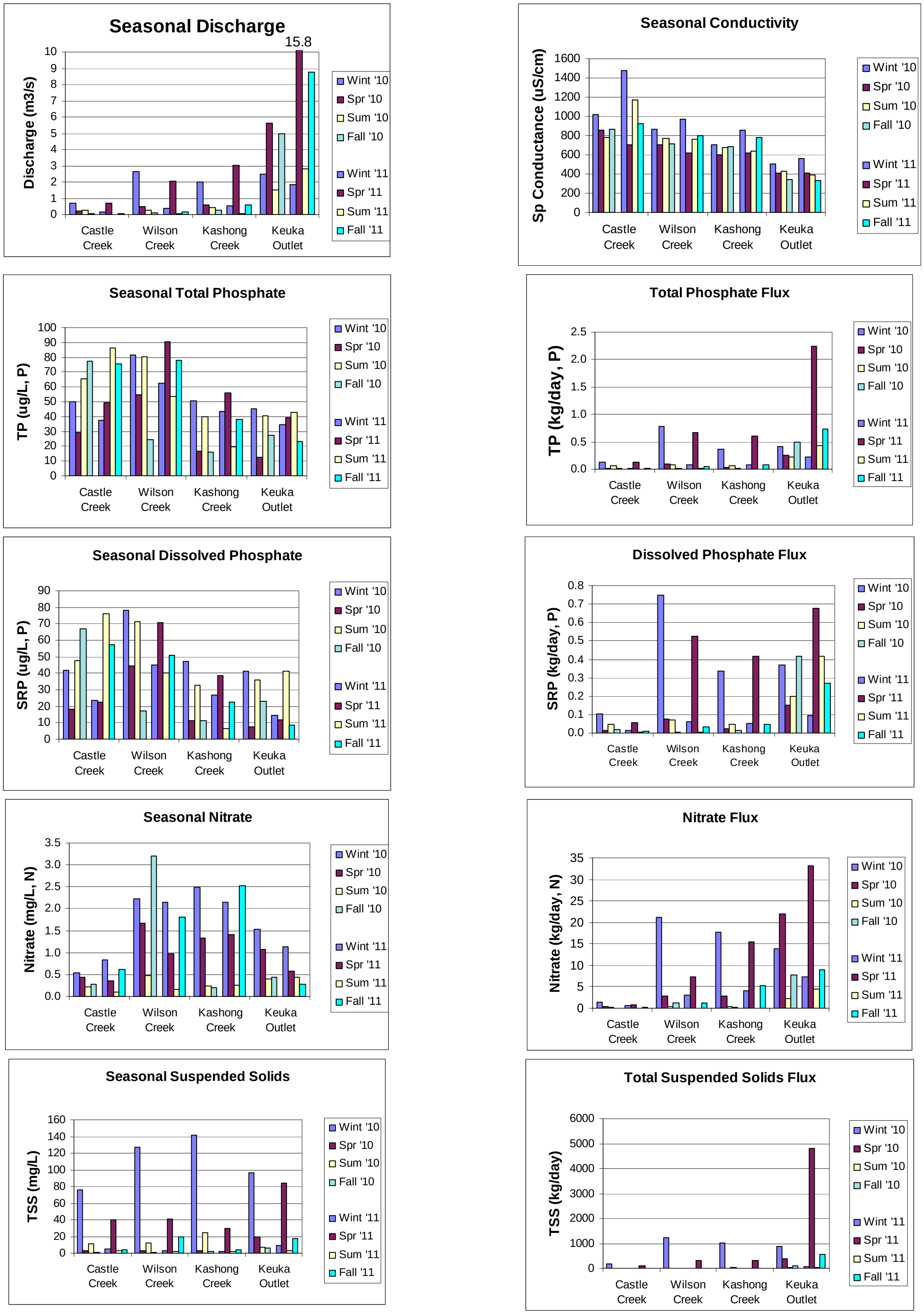


Fig. 18. Seasonal concentration and flux data for four watersheds in 2010 and 2011.

PHOSPHATE BUDGET FOR SENECA LAKE:

Phosphorus is critical to the health and water quality of Seneca Lake because it limits algal growth. The stream concentrations and fluxes suggest that a nutrient loading problem exists. However, stream inputs are only one part of the equation. A phosphorus budget must also include additional inputs and outputs. Other inputs to the lake include atmospheric loading, lakeshore lawn care fertilizers, lakeshore septic systems and municipal wastewater treatment facilities (Fig. 19). Outputs include the outflow of phosphorus-bearing, dissolved and particulate materials through the Seneca River and organic matter burial into the sediments. Comparing all of the inputs and outputs determines the net change in phosphorus for the lake. Phosphorus will increase in the lake, if inputs are larger than outputs, decrease if inputs are smaller than outputs, or remaining the same if inputs equal outputs. Nutrient loading implies inputs exceed outputs. To improve water quality, inputs must be smaller than outputs.

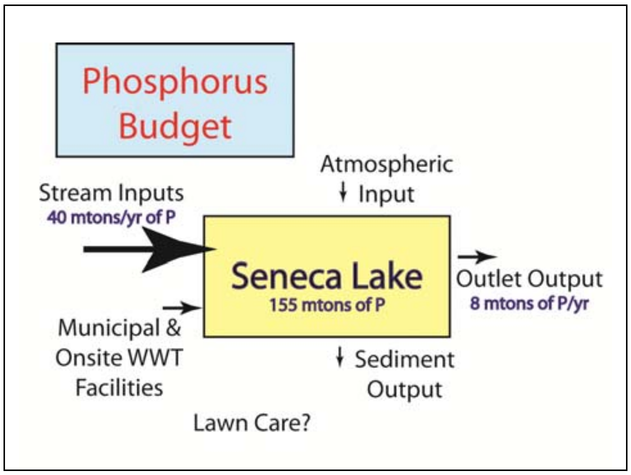


Fig. 19. Estimated phosphorus fluxes into and out of Seneca Lake. The arrow size is proportional its flux.

Inputs: The total fluvial flux of phosphate to the lake is, on average, 40 metric tons/year, assuming a mean stream total phosphate concentration of 47 μg/L, and an estimated annual stream discharge of 863 x 10^6 m^3 (Wing et al., 1995). This stream influx is almost three times larger than the 17 metric tons/year estimated earlier using the same annual discharge (Halfman and Franklin 2007). The difference reflected the inclusion of year round samples in the more recent calculation.

Other notable inputs include lakeshore septic systems, lakeshore lawn care, atmospheric deposition and municipal wastewater treatment facilities that do not discharge into a sampled stream. Extrapolating a septic input per km of shoreline estimate for Owasco Lake, the lakeshore septic influx is approximately 5 metric tons/yr (Halfman, et al., 2011). The septic loading assumed an average household residency of 3 people for 140 day/yr (summer months), an average TP concentration in the septic tanks (~10 mg/L), and an average per capita water use (200 L/capita/day). The atmospheric loading of 0.8 metric tons/year was estimated from National Atmospheric Deposition Program data collected at Ithaca, NY (Site NY67). This site reported an average phosphorus concentration in rainfall of 3 to 4 μg/L in 2009 and 2010, which equated to a mean annual deposition rate of 4.5 mg/m^2 or 0.0045 metric tons/km^2 (http://nadp.isws.illinois.edu/). New Jersey, Connecticut and Minnesota have reported similar phosphorus deposition rates of 0.004 to 0.025 metric tons/km^2 (Koelliker et al., 2004, Hargen et al., 2011). Finally, the Geneva (Marsh Creek) wastewater treatment facility discharged approximately 2.4 metric tons of phosphorus per year (http://www.epa-echo.gov/echo/). Unfortunately, phosphate data was not publically available for the Waterloo and other facilities, and estimating a lawn care/fertilizer flux is too tenuous at this time.

Combining all the inputs, the influx of phosphorus to the lake was approximately 55 metric tons/year. This estimate was probably low due to the lack of some, albeit minor, contributions and simplifying assumptions.

Losses: Nutrient budgets must also include losses from the lake to determine the net gain or loss over time. Phosphates were lost from Seneca Lake through the outlet and into the sediments. The flux of phosphorus out through the outlet is estimated at ~8 metric tons per year, assuming a mean lake TP concentration of 10 μg/L, and an outflow discharge of 760 x 10^6 m^3/year. Unfortunately, very few sediment cores have both total phosphate and sedimentation rate data. Extrapolation from the limited number of cores estimated a flux of 1.5 metric tons/year to the sediments. The sediment burial estimate is tentative at this time.

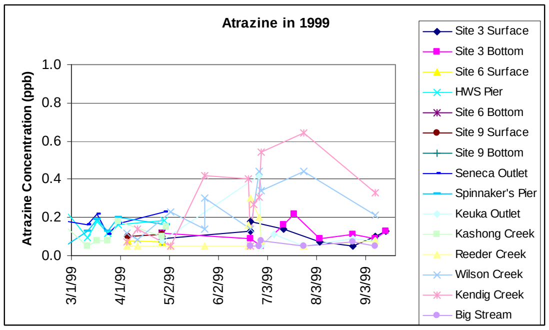
Combining all the outputs, the efflux of phosphorus from the lake was approximately 10 metric tons/year. This total efflux is less certain than the inputs.

Budget: The inputs at 55 mt/yr are much larger than the outputs at 10 mt/yr, thus Seneca Lake experienced a significant nutrient loading problem over the past two decades (Fig. 19). The total amount of phosphorus in the lake was 155 metric tons estimated from the 2011 mean lake total phosphate concentration of 10 mg/L and a lake volume of 15.5 km^3. Assuming a net positive flux of 45 metric tons/year, the lake is destined to become eutrophic. Predicting when this will happen is difficult to estimate. For example, larger algal productivity from nutrient loading induces larger effluxes out the outlet and to the sediments. Changes in rainfall, thus runoff and discharge, proportionally influence the fluvial flux. However, the budget highlights the tenuous nature of the lake, and the need to proactively decrease nutrient loading, and especially loading from streams.

In conclusion, the Seneca Lake watershed has a number point and nonpoint sources of nutrients. They included municipal wastewater treatment facilities and onsite wastewater treatment (septic systems), atmospheric loading, runoff from agricultural land both crop farming and animal husbandry, and runoff of nutrients and other products from well manicured lawns. The preliminary analysis indicated that runoff from streams dominated all inputs to the lake. Clearly, the phosphorus budget indicates that inputs overshadow outputs. This is consistent with the observed degradation in water quality degradation over the past decade.

OTHER HYDROGEOCHEMICAL WATER QUALITY INDICATORS

Herbicides and Coliform Bacteria: The source of atrazine in the Seneca Lake watershed was investigated in 1999 and 2000 (Fig. 20, McSweeney, 1999; Baldwin and Halfman, 2000; Halfman and Franklin, 2007). Atrazine concentrations were typically below 1.0 μg/L through out 1999. In 2000, concentrations were similar to 1999 values up to the end of May. After May, stream concentrations rose to or very close to 3 μg/L, the EPA's MCL, with the largest detected concentration of 8 μg/L at Kendig Creek (August, 2000). The study concluded with following spatial and temporal changes. First, streams draining more agricultural land had larger atrazine concentrations. Second, atrazine concentrations peaked during June, July and August, a timing that corresponds with the application of atrazine in the fields. Third, the amount of rainfall co-varied with the concentration of atrazine in the runoff. The largest concentrations were detected during a large rainfall event. The smaller concentrations in 1999 compared to 2000 corresponded to lower rainfall in 1999. Finally, none of the lake concentrations exceeded 1 μg/L, consistently below the EPA's MCL.



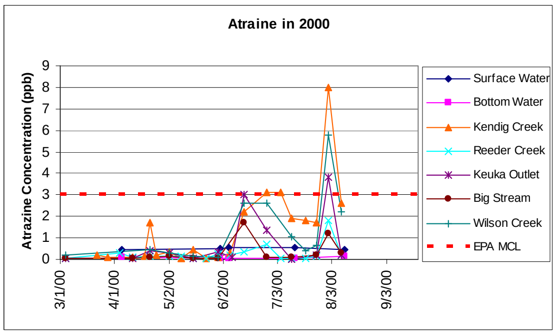


Fig. 20. Atrazine concentrations in Seneca Lake and its major tributaries from 1999 and 2000. Note the scale change between years (McSweeney, 1999; Baldwin and Halfman, 2000).

Total coliform and *E. coli* bacteria concentrations in 2003, 2004 and 2005 were typically below the EPA's MCL (Bush and Halfman, 2006; Bush, 2006). Lake samples were typically ten times less concentrated than stream water, and lacked any temporal or spatial trends. Bacteria concentrations were largest in the streams during runoff events, and a runoff event influenced the large mean counts in 2005. Wilson and Hector Falls regularly had larger bacteria concentrations than the other streams, especially during runoff events. It suggests that agricultural and rural landscapes with aging septic systems input more bacterial than the other drainage systems, and pose potential but currently not detrimental threats to the Seneca Lake watershed.

Trihalomethanes (THMs) concentrations were not above analytical detection limits for all the analyzed stream and lake water samples during the 2010 spring field season.

Seneca Lake Major Ions & pH: The following summarizes previous reports and publications (Halfman et al., 2006, Halfman and Franklin, 2007):

- Chloride, bicarbonate (measured as alkalinity), sodium and calcium dominate the salinity of Seneca Lake with lesser amounts of sulfate, magnesium and potassium. The ions reflected the weathering of carbonate-rich bedrock, tills and soils and groundwater inputs (Table 7, Fig. 21).
- Their concentrations were not drinking water concerns but their source required elaboration to assess if it may change in the future.
- The fluvial flux of chloride and sodium to the lake was insufficient to provide the concentrations measured in Seneca, and to a lesser extent, Cayuga, but accounts for the chloride and sodium concentrations in neighboring Finger Lakes.
- A groundwater source for chloride and sodium was hypothesized to compliment fluvial sources. The bedrock floor of Seneca, and to a lesser extent, Cayuga, is deep enough to intersect the Silurian beds of commercial-grade rock salt located ~450-600 m below the surface (Wing et al., 1995, Halfman et al., 2006).
- Historical chloride data revealed two century-scale patterns among the Finger Lakes (Jolly, 2005, Jolly, 2006, Sukeforth and Halfman, 2006) (Fig. 22). Chloride concentrations were low ~40 mg/L in 1900, rose to ~170 mg/L by the 1960's, and subsequently decreased since 1980 to the present day concentration of ~120 mg/L in Seneca Lake with parallel changes in Cayuga Lake (Jolly, 2005, 2006). Seneca CTD profiles confirm the decrease over the past two decades (Fig. 23).

- Chloride data from Canadice, Hemlock and Skaneateles increased from below 10 mg/L to above 30 mg/L from 1920 to the present day and were interpreted to reflect increased use of road salt on our major roadways (Sukeforth and Halfman, 2006). A groundwater source was still required in Seneca and Cayuga, however the flux of salt from the ground must have varied during the past century. Perhaps the historical change was dictated by solution salt mining activity at the southern end of the watershed.
- Sulfate, alkalinity, calcium and magnesium comprise the other major ions in the lake (Halfman et al., 2006). Mass-balance arguments indicated that sulfate also has an additional groundwater source, perhaps originating from the underlying gypsum-rich ($CaSO_4 \cdot H_20$), Bertie Formation.
- The calcium and magnesium data indicate that Seneca Lake water is moderately hard. Calcium, magnesium and alkalinity concentrations are smaller in the lake than predicted by stream inputs, and are removed from the water column by the precipitation of fine-grained, calcium carbonate ($CaCO_3$) during algal bloom induced whiting events and formation of carbonate shells for zebra & quagga mussels, clams, snails and other shelled animals.
- The pH of Seneca Lake was between 8 to 9. Thus, acid rain has had a minimal impact on the acidity of the lake due to limestone and other acid neutralizing materials are in the Seneca watershed.

Table 7. Site-Averaged Major Ion Data (Halfman et al., 2006).

Lake Sites	Chloride mg/L, Cl	Sulfate mg/L, SO_4	Sodium mg/L, Na	Alkalinity mg/L, $CaCO_3$	Potassium mg/L, K	Calcium mg/L, Ca	Magnesium mg/L, Mg
Site 1S	140.3 ± 7.5	38.0 ± 1.8	78.9 ± 5.9	107.4 ± 14.0	2.8 ± 37	41.6 ± 5.1	11.0 ± 1.1
Site 2S	138.3 ± 7.7	37.6 ± 2.1	77.8 ± 6.0	106.1 ± 12.0	2.7 ± 35	42.2 ± 2.6	10.8 ± 1.0
Site 3S	137.2 ± 8.7	37.4 ± 2.1	78.1 ± 7.0	107.0 ± 14.0	2.7 ± 34	41.9 ± 3.4	10.8 ± 1.1
Site 4S	140.0 ± 7.5	37.8 ± 1.8	79.1 ± 6.4	104.7 ± 11.0	2.8 ± 37	42.6 ± 2.8	11.0 ± 1.1
Site 1B	139.9 ± 6.4	38.2 ± 1.4	78.5 ± 6.0	104.6 ± 15.6	2.7 ± 37	42.7 ± 3.1	10.9 ± 1.0
Site 3B	141.2 ± 7.2	38.3 ± 1.5	81.3 ± 7.0	108.6 ± 16.0	2.7 ± 35	43.6 ± 2.7	10.9 ± 0.9
Average Lake	139.5 ± 7.5	37.9 ± 1.8	79.0 ± 6.4	106.4 ± 13.8	2.7 ± 35.8	42.4 ± 3.3	10.9 ± 1.0
Stream Sites							
Glen	11.9 ± 1.2	13.6 ± 0.9	9.7 ± 0.2	135.0 ± 7.1	1.6 ± 2	36.3 ± 1.8	11.7 ± 6.2
Rock Stream	29.2 ± 5.8	20.1 ± 2.9	19.3 ± 3.6	128.5 ± 31.3	2.0 ± 4	46.2 ± 31.8	20.1 ± 18.1
Big Stream	41.6 ± 13.2	24.1 ± 5.1	22.4 ± 6.8	156.3 ± 26.7	2.5 ± 20	49.1 ± 8.0	14.1 ± 4.5
Plum Pt.	71.4 ± 29.6	38.7 ± 9.1	33.5 ± 15.1	164.3 ± 20.6	2.7 ± 16	53.7 ± 15.3	16.2 ± 5.6
Keuka	31.2 ± 8.1	28.7 ± 5.0	16.1 ± 3.9	129.6 ± 24.5	2.5 ± 34	41.3 ± 11.2	14.1 ± 9.0
Kashong	50.7 ± 6.7	38.9 ± 6.8	21.4 ± 2.3	255.0 ± 42.3	3.3 ± 27	77.6 ± 25.0	30.1 ± 15.3
Wilson	54.2 ± 15.1	40.0 ± 10.1	25.7 ± 5.2	248.4 ± 52.6	4.4 ± 22	90.6 ± 19.1	29.0 ± 9.2
Kendig	39.1 ± 12.2	42.0 ± 10.7	19.3 ± 6.5	221.4 ± 35.8	3.0 ± 17	74.2 ± 20.1	22.5 ± 10.3
Reeder	37.1 ± 18.7	43.2 ± 11.2	24.4 ± 9.9	238.4 ± 45.1	2.4 ± 27	79.1 ± 25.5	17.4 ± 6.8
Kendaia	29.8 ± 6.6	37.3 ± 5.4	17.2 ± 6.2	240.0 ± 40.7	3.4 ± 4	77.4 ± 13.5	17.2 ± 7.9
Indian	27.5 ± 6.7	37.3 ± 3.9	15.4 ± 3.0	228.8 ± 35.7	2.2 ± 4	71.8 ± 9.8	20.8 ± 8.1
Lodi Pt.	60.9 ± 13.8	39.2 ± 0.1	32.9 ± 6.4	287.0 ± 103.2	3.6 ± 2	88.2 ± 11.1	17.2 ± 2.2
Mill Cr	8.4 ± 0.8	20.5 ± 2.3	9.3 ± 3.2	122.3 ± 22.6	1.4 ± 4	30.6 ± 20.3	8.2 ± 6.5
Bullhorn	32.2 ± 3.6	28.2 ± 1.6	18.3 ± 1.2	152.0 ± 14.4	2.1 ± 3	60.2 ± 6.4	15.2 ± 8.2
Sawmill	14.3 ± 5.0	21.9 ± 2.4	12.5 ± 7.3	134.5 ± 39.4	1.8 ± 4	43.8 ± 8.4	11.3 ± 6.4
Glen Eld.	9.6 ± 3.8	17.6 ± 2.2	6.6 ± 1.0	138.7 ± 56.8	1.1 ± 3	38.6 ± 10.7	10.3 ± 7.1
Hector Falls	18.6 ± 1.4	14.6 ± 0.5	10.4 ± 2.8	138.0 ± 21.1	1.4 ± 3	42.8 ± 6.0	11.7 ± 6.4
Average Stream	33.4 9.0	29.8 4.7	18.5 5.0	183.4 36.5	2.4 11.5	58.9 14.4	16.9 8.1

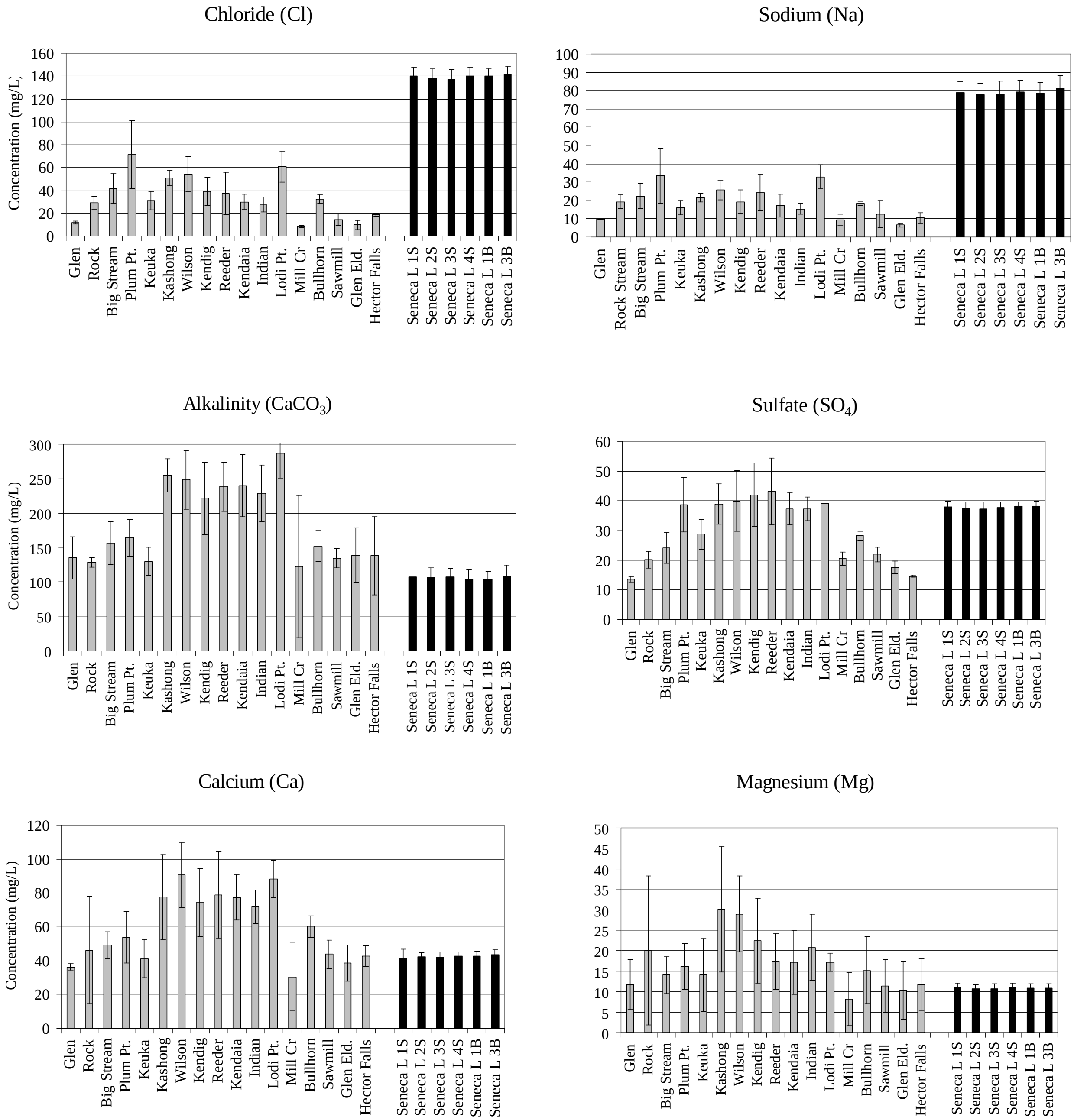


Fig. 21. Major ion mean concentrations in Seneca Lake and its major tributaries (Halfman et al., 2006).

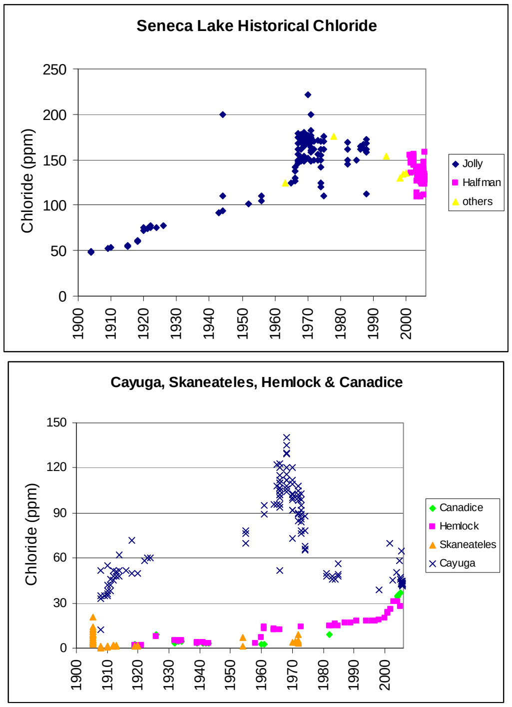


Fig. 22. Historical chloride data in Seneca and Cayuga Lakes (Jolly, 2005, 2006), and in Canadice, Hemlock and Skaneateles Lakes (Sukefoth and Halfman, 2006).

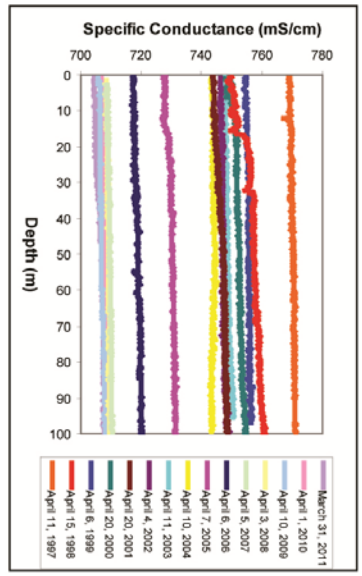


Fig. 23. Multi-decade early spring CTD specific conductance data.

CONCLUSIONS & RECOMMENDATIONS:

None of the water quality parameters were life threatening. The limnological trends over the past two decades revealed significant impacts by zebra and quagga mussels in the mid-1990s. Since then, secchi disk depths, and chlorophyll, nutrient and suspended sediment concentrations indicated that water quality has declined over the past decade from an oligotrophic lake to a borderline oligotrophic-mesotrophic lake by 2011. Clearly, nutrient loading from the watershed contributed to the degradation. The loading, if unchecked, will degrade water quality in the lake into the future if not addressed in a meaningful and sustainable manner. Identified sources of phosphorus include wastewater treatment facilities, onsite wastewater systems, lawn fertilizers, atmospheric loading, agricultural activities, soil erosion, roadside ditches and construction activities. The bulk of the influx was by streams. The preliminary phosphorus budget highlights greater inputs than outputs, an imbalance consistent with the declining water quality trends. Declining mussel populations and “top down” ecological stressors may have also played a role and their relative contributions should be investigaed in the future.

Due to the variety of nutrient sources and their measured impact on water quality, remediation efforts should be multifaceted to improve Seneca Lake.

Municipal Wastewater Facilities: Wastewater treatment facilities should evaluate the treatment and removal of nutrients from their effluent. Those with minimal phosphorus removal should add appropriate tertiary treatment systems. Selection of the best method must balance the effectiveness of the removal process with its costs. Any additional treatment procedure adds to the capital and operating costs, costs that will be passed on to the public.

Agricultural BMPs: Various "Best Management Practices" (BMPs) are available to reduce the nonpoint source contributions from the agricultural sector, both crop farming and animal husbandry. Potential BMPs are numerous, and include contour plowing, gully plugs, settling ponds, buffer vegetation strips, minimal tillage farming, manure digesters, and others. Steps should be taken to investigate which practices are most effective for this watershed and what must be accomplished so that the entire economic burden is not placed solely on the farmer. In some cases education is critical to gain acceptance and follow through with adaptation of these practices. A USDA funded-research project of various BMPs in the Conesus watershed by Dr. J. Makarewicz and his team at SUNY Brockport, County Soil and Water, and colleagues, indicated that a combination of BMPs significantly reduced nutrient and suspended sediment inputs from agricultural watersheds (Makarewicz, 2009).

Watershed Inspector: A watershed inspector's office with sufficient budget and authority should be established to protect and preserve Seneca Lake. This is a significant challenge because funding and jurisdiction is at odds with existing town, county and state agency resources and political boundaries. The inspector's office should work in conjunction with existing County Soil and Water Offices, Seneca Lake Pure Waters Association, Seneca Lake Area Partners in Five Counties (SLAP-5), Cornell Cooperative Extension Offices, concerned citizens and the Finger Lakes Institute perhaps brought together into a Seneca Lake Watershed Council to promote and implement best management practices that reduce nutrient loading from agricultural landscapes; initiate onsite system inspection programs and lawn care fertilizer and herbicide reduction seminars for lakefront property owners; mitigate the impact of construction and roadside ditch drainage activities; and, update watershed protection legislation for the entire Seneca Lake Watershed. Taken together, these steps would reduce nutrient loading to and increase water quality in the lake. Similar organizations have protected water quality in Keuka, Canandaigua and Skaneateles watersheds. Sufficient funding is essential. One model to support the Inspector, related staff and its operations budget, would be a tax on water users. If levied at a few dollars per user per year, the income would provide a viable operational budget to support the program. Once in operation, an inspector's office might save money by reducing the cost of filtering algae and suspended sediment from the water before it is delivered to the consumer.

Other Industrial Facilities: Permitted discharges should be evaluated to assess the impact on the lake and its ecosystem. For example, more research is required to determine the source and fate of chloride and sodium salts in the lake from mining activities, application of road salt and potential storage of gas in the abandoned salt caverns.

Education: Education is also critical for local residents to understand the need for properly functioning onsite wastewater treatment systems and environmentally friendly (less fertilized) lawns to help reduce the seepage and runoff of nutrients to the lake. The education starts at primary and secondary levels but should also include the average homeowner, government officials, and environmental protection associations. The Finger Lakes Institute has an active K-12 outreach program that utilizes our 65-ft research vessel on Seneca Lake and various exercises that teachers can bring into their classrooms. It is currently underutilized by the numerous secondary programs throughout the watershed. FLI sponsors a successful community outreach program as well.

Stream and Lake Monitoring: Finally, water quality monitoring of selected streams and lake sites should continue into the future. Both peak flow and base flow fluxes must be quantified for phosphates, total suspended solids and nitrates as the distinction was critical in preliminary studies in Wilson Creek and other studies in Owasco and Conesus watersheds. This minimal effort will evaluate the effectiveness of the nutrient reduction strategies in the largest tributaries to the lake. Any monitoring program must sample all of the other major tributaries in the watershed, sample year round, and analyze for herbicides, pesticides, heavy metals, organic compounds like PCBs and DDT, human and concentrated animal healthcare products like antibiotics, estrogens and other hormones, and total coliform and *E. coli* bacteria because protection of our water supply is worth the cost. Biological macrophyte monitoring would be a useful compliment to the hydrogeochemical efforts. Lake sites should also be monitored for the same list of parameters. The Finger Lakes Institute plans to continue some monitoring of the lake and watershed in the years to come. The nature of the FLI monitoring program and its diversity of analyses, however depends on funding.

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