THE 2018 WATER QUALITY MONITORING REPORT, OWASCO LAKE, NY.

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INTRODUCTION

Since the initial Finger Lake Institute (FLI) water quality survey of the eastern Finger Lakes in 2005, Owasco Lake and its watershed has been the focus of additional research due to the lake's poor water quality in comparison to neighboring Finger Lakes. This focus established a monitoring program of Owasco Lake and its watershed to: (1) document spatial and temporal trends in pertinent water quality / water clarity / limnological parameters; (2) investigate the source and magnitude of nutrients in the watershed, as their inputs promote algal growth and thus degrade water quality; (3) investigate linkages between the water quality data and the recent rise in blue-green algae and their associated toxins; and, (4) promote the development of comprehensive and effective watershed management policies to improve water quality in Owasco Lake. This decade[+] effort was supported by numerous sponsors including: the Fred L. Emerson Foundation, Auburn, NY, New York State funds secured by New York State Senator Michael Nozzolio, the Owasco Lake Watershed Association (OWLA), the Town of Fleming, Cayuga County Soil and Water Conservation District, Finger Lakes – Lake Ontario Watershed Protection Alliance and most notably the Cayuga County Legislature. Thank you all for your support.

The ongoing monitoring effort has identified the following results to date:

- The trophic status (productivity level) of Owasco Lake fluctuates above and below the oligotrophic (good water quality) – mesotrophic (intermediate water quality) boundary.
- Phosphorus is the limiting nutrient in Owasco Lake. Additional inputs of phosphorus would stimulate additional algal growth and degrade water quality.
- The lake has experienced late-summer blooms of blue-green algae. Blue-green algae are a concern due to their affiliation with impaired / eutrophic (poor water quality) water bodies, their ability to form unsightly, surface water, algal scums. More importantly, some species of blue-greens may produce toxins that have health implications for humans and other warm blooded organisms.
- Nutrient and sediment sources include point sources like wastewater treatment facilities and onsite wastewater (septic) systems, and nonpoint sources like animal and crop farms, lawn fertilizers, soil erosion, stream bank erosion, roadside ditches, drainage tiles, and construction activities.
- A 2007 DEC mandated reduction of phosphorus in the effluent of the Groton Wastewater Treatment Facility starting in has reduced nutrient loading to the Owasco Inlet and thus Owasco Lake.

- The increased adoption of agricultural best management practices in the watershed and follow through on recommendations made by the Watershed Inspector's Office and the Owasco Lake Watershed Management Council should also reduce nutrient loads to the lake as well.
- Annual nutrient load estimates positively correlated to precipitation totals, especially precipitation in the spring season.
- Streams and tributaries are the primary source of nutrients and sediments to the lake, especially during "wet" years but also in "dry" years.
- Event *vs.* base flow analysis of daily nutrient and sediment loads from Dutch Hollow Brook indicated that over 90% of the loads are delivered during precipitation/runoff events, especially in the spring season.
- Since 2011, annual phosphorus budgets for Owasco Lake typically revealed larger inputs than outputs. Continued net accumulation of phosphorus in the lake, i.e., when nutrient inputs exceed outputs, will continue to degrade water clarity and water quality.
- Phosphorus loading must be curtailed to improve water quality in Owasco Lake. This must be accomplished sooner rather than later because if the loads were curtailed today, it would still take a minimum of five water retention times, i.e., approximately a decade, for the lake to naturally cleanse itself of excess phosphorus and improve water quality.

Water quality research is moving into an exciting phase. NY State funds to Cayuga County Soil and Water Conservation District and Owasco Lake Watershed Association should establish preliminary BMPs in the Owasco Lake watershed. Support has been secured by Cayuga County Planning to expand the recent Owasco Lake Watershed Management & Waterfront Revitalization Plan to develop an EPA Nine Key Elements Plan. DEC established the Finger Lakes HUB, group to oversee efforts to improve water quality in the 11 Finger Lakes. One immediate outcome was additional monitoring of the lake and streams. For example, they contracted with the USGS to deploy a water quality buoy in the lake. Finally, we all must take advantage of Governor Cuomo's $65 million investment to combat the blue-green algae problems in Owasco and other lakes.

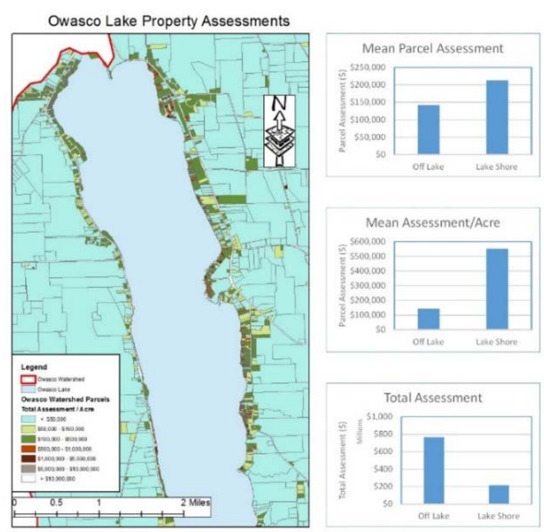


Fig. 1. Owasco Lake property assessments per acre of land. Lakeshore property assessments are significantly larger than properties away from the lake.

Numerous economic reasons mandate remediation efforts in the watershed (Fig. 1). First and foremost, the lake is a public drinking water source for Auburn, Owasco and neighboring communities, and supplies numerous private systems for lakeshore residents. Second, the lake is the focus for recreational, sports fisheries, and other tourism industries in the region. In both cases, poor water quality yields economic challenges. Declining water quality will also negatively impact local property values and tax revenues. Recent[1] property assessments in the watershed, like other Finger Lake watersheds, reveal significantly larger property assessments

[1] Assessment data and parcel's map acquired from Cayuga County in the summer 2016.

per acre for parcels adjacent to the lakeshore than parcels away from the lake (Fig. 1). Therefore, municipal officials should do everything in their power to maintain water quality in the lake. More importantly, we owe it to future generations to leave this lake in better shape than when we got it.

METHODS

The offshore lake and stream sample sites and field/laboratory methods used in 2018 were similar to the 2005 – 2017 programs.

Owasco Lake: The 2018 lake survey sampled Sites 1 and 2 on a monthly basis from late May through late September (Table 1, Fig. 2). These two sites have been sampled since the initial 2005 survey, and have been deemed representative of the open water limnology in Owasco Lake. The specific 2018 monthly survey dates were: 5/22, 6/19, 7/24, 8/14 & 10/3. A three year award from the Emerson Foundation supported the deployment and retrieval of the water quality monitoring buoy, increased the number of lake surveys to weekly surveys in July, August and September, and added six nearshore lake sites (Table 1). Additional funding from Owasco Watershed Lake Association (OWLA) enabled a test of various blue-green algae mitigation devices at a number of shoreline locations. The results of the nearshore and mitigation technology efforts will be discussed in separate reports[2]. This report will focus on the offshore lake and stream results.

The lake-monitoring field methods were identical to the earlier research. A CTD profile, bbe FluoroProbe profile, Secchi disk depth, vertical plankton tow (80-μm mesh), and surface and bottom water samples were collected at each site. The CTD electronically measures water column profiles of temperature (°C), conductivity (reported as specific conductance, μS/cm, a measurement proportional to salinity), dissolved oxygen (mg/L), pH, turbidity (NTUs), photosynthetic active radiation intensities (PAR, $\mu E/cm^2$-s), and fluorescence (a measure of chlorophyll-a, μg/L) using a SeaBird SBE-25 CTD. The CTD was lowered from the surface to ~1m above the lake floor, collecting data every 0.5 seconds (~0.2 meters) along the downcast. The bbe bbe FluoroProbe electronically measures water column profiles of four different algal groups and yellow substances based on their accessory pigments. It distinguishes among: 'green' algae (Chlorophyta and Euglenophyta), 'brown' algae (diatoms: Baccillariophyta, Chyrsophyta, and Dinophyta), 'blue-green' algae (Cyanophyta), and 'red' algae (Cryptophyta). It was deployed attached to the CTD. The bbe FluoroProbe was under repairs during the May survey. The plankton collected by each tow were preserved in a 6-3-1, water-alcohol-formalin solution and enumerated to species level by Barbara Halfman back in the laboratory under a microscope. Water samples were analyzed onsite for temperature (°C), conductivity (specific conductance, μS/cm), pH, dissolved oxygen (mg/L), and alkalinity (mg/L, $CaCO_3$) using hand-held probes and field titration kits, and analyzed back in the laboratory for total phosphate (μg/L, P), dissolved phosphate (SRP, μg/L, P), nitrate (mg/L, N), chlorophyll-a (μg/L), dissolved silica (μg/L, Si), and total suspended solid (mg/L) concentrations.

[2] Halfman, et al., 2018. Blue-green algae in Owasco Lake, the 2017 Update. The 2018 Annual Report to the Fred L. Emerson Foundation.
Halfman, et al., 2018. Final Report on the Owasco Lake HAB Inhibiting Technologies Assessment. A report to the Owasco Watershed Lake Association.

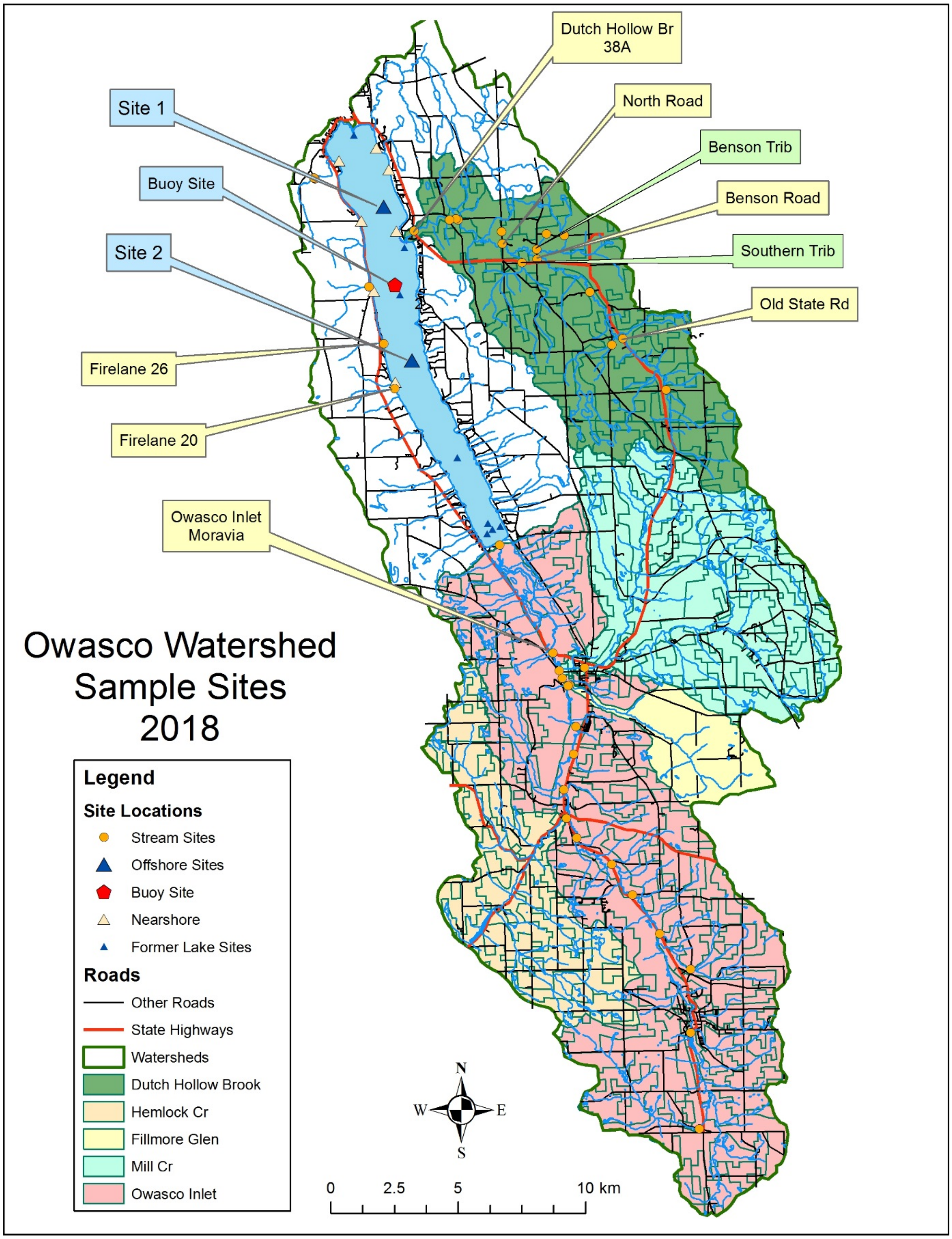


Fig. 2. The 2018 lake and stream sites. The 2018 stream sites focused on previously sampled sites within Dutch Hollow Brook and the Owasco Inlet. The small tributaries near the terminus of Fire Lane 20 and, for the first time, Fire Lane 26 were also sampled.

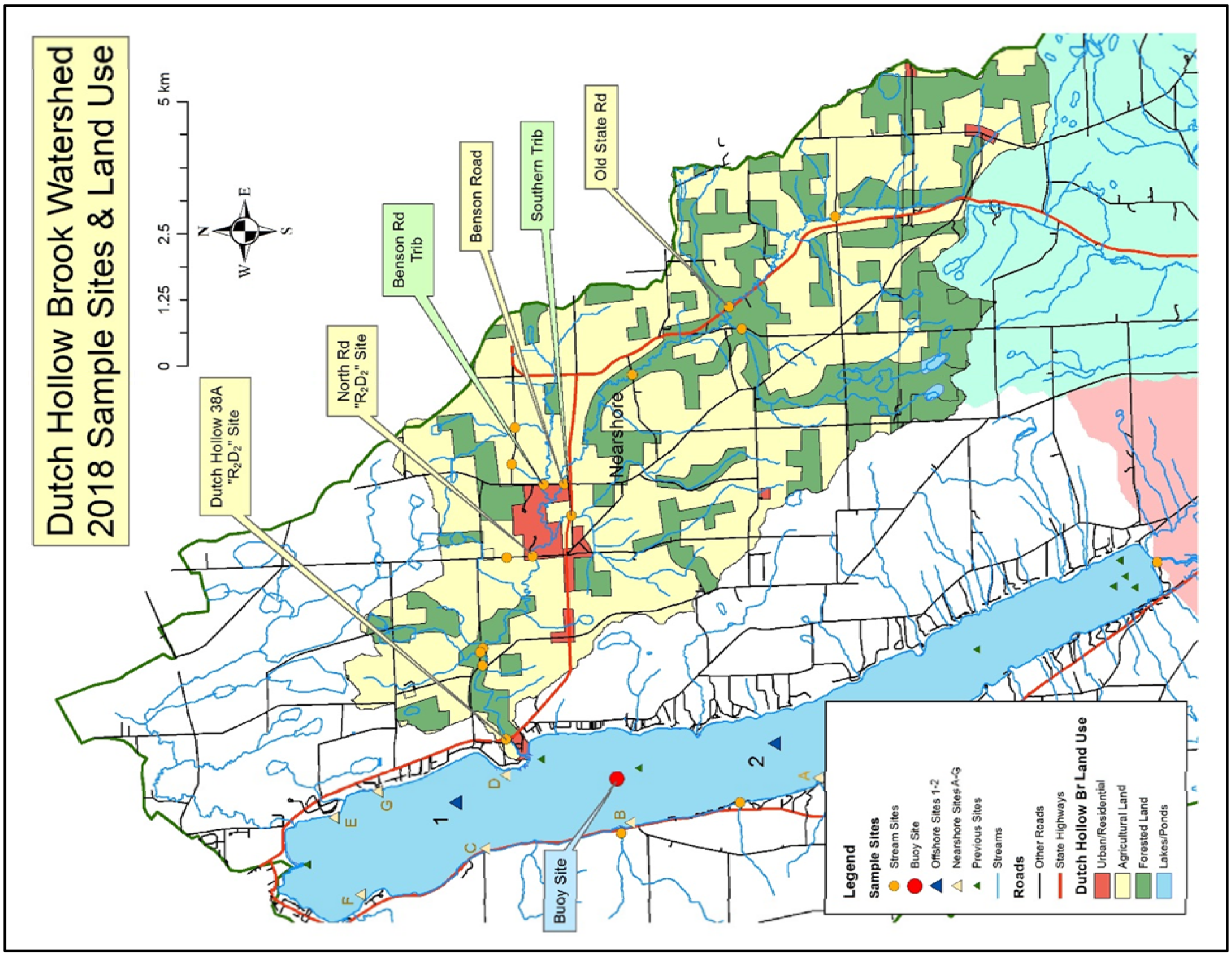


Fig. 2 continued. 2018 site locations and land use within Dutch Hollow Brook watershed.

Table 1. Owasco Lake Site Locations and Water Depths.

Site Name	Latitude	Longitude	Water Depth
Offshore Sites:			
Site 1	42° 52.40’ N	76° 31.35’ W	34 m
Site 2	42° 49.15’ N	76° 30.45’ W	52 m
Buoy Site	42° 50.35’ N	76° 30.85’ W	49 m
Nearshore Sites:			
A – Fire Lane 20	42° 48.69’ N	76° 30.92’ W	2 - 3 m
B – Wyckoff Rd	discontinued		
C – Stone School Rd	42° 52.01’ N	76° 31.98’ W	2 - 3 m
D – Burtis Pt	42° 51.89’ N	76° 30.96’ W	2 - 3 m
E – Martin Pt	42° 53.64’ N	76° 31.59’ W	4 - 5 m
F – Buck Pt	42° 53.35’ N	76° 32.65’ W	2 - 3 m
G – Yacht Club	42° 53.23’ N	76° 31.23’ W	5 m

Drone Flights: A drone was flown at 50 m above the lake along the six nearshore locations to investigate its suitability to measure water clarity and other parameters, and a few additional shoreline locations to assess the effectiveness of the BGA mitigation technologies (Fig. 3). DJI’s Phantom 3 Advanced with a Sony EXMOR gimbaled camera was used. It captured 12 megapixel digital images. Each image spanned an area of ~100 by 150 meters at a flight altitude of 50 m. Multiple, overlapping images were collected at every nearshore site to investigate attached algae and macrophyte distributions, the distribution of blue-green algae blooms and open water algal concentrations (Table 1, Fig. 2). The overlapping images were spatially aligned in Adobe Photoshop or Drone Deploy. The composite image was then georeferenced in ArcGIS to 2015 satellite digital orthoimagery (NYS Clearinghouse data). Only one vertical and two oblique images looking up and down along the shoreline were collected at the non-overlapping BGA mitigation technology sites. Flights dates were: 7/11, 7/18, 8/1, 8/8, and 8/15, plus a few additional partial flights. Drones will no longer be flown in high winds (>10 mph) or rainy weather as a drone crashed in 11 mph winds last year.

Fig. 3. The drone used in this study, a Phantom 3 Advanced by DJI.

Owasco Buoy: The FLI meteorological and water quality monitoring buoy manufactured by YSI/Xylem was redeployed at its mid-lake site from 4/25 through 10/29 (Table 1, Fig. 2). As in previous deployments, the buoy was programmed to collect water column profiles using a YSI/Xylem EXO2 water quality sonde every 12 hours (noon and midnight). The sonde measured temperature (°C), conductivity (μS/cm, reported as specific conductance), dissolved oxygen (mg/L & % saturation, by optical sensor), turbidity (NTUs by backscattering), and fluorescence (RFUs). The fluorescence sensor measured both total chlorophyll and blue-green algae phycocyanin concentrations (after specific pigment excitation by different wavelengths of light). Data were collected every 1.5 meters down the water column starting at 1 m. The buoy also contained a standard suite of meteorological sensors that, as in previous years, recorded five-minute, mean, air temperature, barometric pressure, relative humidity, light intensity, wind speed and wind direction data every 30 minutes. The accumulating raw data were periodically transferred to HWS by cellular phone ~1 hour after collection and is available on a website (http://fli-data.hws.edu/buoy/owasco/). Buoy hardware and software issues prevented collection of water quality data from 4/17 – 5/7, 5/17, 5/26, 10/15, 10/18, and after 10/29 (the later dates due to power issues during the unrelenting cloudy/rainy weather); and meteorological data from 4/17 through 4/23.

Owasco Streams: The 2018 stream monitoring focused on six sites within the Dutch Hollow Brook watershed, the terminus of Owasco Inlet, and the terminus of two small tributaries entering the western side of the lake, one at the end of Fire Lane (FL) 20 and the other at the end of FL 26. Stream sites were visited seven times in 2018, specifically 4/4, 4/18, 5/18, 5/28, 6/4, 6/12, and 6/13, for onsite analyses and collection of water samples for nutrient and sediment analyses back in the laboratory. The first two survey dates were performed by Halfman's Environmental Hydrology class, and these surveys only sampled Dutch Hollow Brook at three sites, 38A, North Rd and Old State Rd due to class time restrictions. In all, more dates were surveyed than were proposed a no additional cost to Cayuga County.

The last five surveys sampled Dutch Hollow Brook at six sites in 2018 (Fig. 2). Progressing upstream, four sites were sequentially located along 38A, including the terminus at Rt 38A, and sequentially upstream at North Rd, Benson Rd, and near Old State Rd. Two unnamed tributaries in the watershed were also sampled. The South tributary was sampled at Rt 38A just east of the Owasco town center. The Benson tributary was sampled along Benson Rd just north of the Benson Rd site. These sites duplicated those used in the past in this watershed.

Owasco Inlet was sampled just upstream of the Owasco Flats area just north of Moravia where the Inlet crosses Rt 38 (Fig. 2). Budget pressures forced the reduction in the number of stream sites that were sampled within this drainage in previous years. Two small tributaries that enter along the western edge of the lake at the ends of FL 20 and FL 26 were also sampled.

Stream discharge, water temperature, conductivity, dissolved oxygen, pH and alkalinity were measured onsite at each site using hand-held probes or field titration kits. Water samples were also collected and subsequently analyzed in the laboratory for total phosphate (TP), dissolved phosphate (SRP), nitrate (NO3) and total suspended sediment (TSS) concentrations. Stream discharge (the volume of water per unit time flowing past a site) was calculated from measured stream width, depth and velocity data (using a 30 m tape, wading rod and HACH FH950 portable velocity flow meter with electromagnetic sensor). Both velocity and stream depth were measured at ten (or five) equally distributed segments aligned perpendicular to stream flow. The velocity was measured at ~60% of the stream depth and assumed the average velocity for each segment. Ten segments were utilized when the stream was wide (>10 m) or more accuracy was necessary, e.g., the Inlet site and along Dutch Hollow Brook at 38A and North Rd. Stream discharge (water volume per unit time, e.g., m^3/s) is necessary to calculate the flux (loading) of nutrients and suspended sediments, because flux of a substance (its mass/time, e.g., kg/day) equals stream discharge (volume water/time, e.g., m^3/s) times its concentration (mass/volume water, e.g., mg/L).

Runoff/Event Flow versus Base Flow Variability: A Teledyne ISCO automated water sampler, and two pairs of *ONSET* HOBO U20L-04 loggers were deployed at the Rt 38A site in Dutch Hollow Brook from 4/12 to 11/4 to investigate the impact of event *versus* base flow variability on the delivery of nutrients and sediments to the lake (Figs. 4a & 4b). One logger in each pair was deployed in air and the other deployed underwater at a fixed elevation. The configuration accounted for changes in atmospheric pressure to isolate pressure changes detected by the submerged logger to changes in water level by these unvented pressure transducers. The configuration also provided air and water temperature measurements at the site. Deploying two pairs of loggers hedged against losing a pair of logger to a flood, vandalism or any other unfortunate issue.

The autosampler was programmed to collect 1-L of water every day (4 am). This frequency collected both event and base flow samples in 2016 and 2017. At each site, stream discharge was measured and the autosampler was serviced every one to two weeks. Each sample was analyzed for suspended sediment and nutrients. In 2018, three samples were not analyzed over the 207 day deployment for various reasons (e.g., dumping out the filtered water before saving the filtrate for the dissolved nutrient analyses).

Fig. 4a. Servicing "R_2D_2" the Teledyne ISCO automated water sampler located at the Rt 38A site. It collected 1-liter of water daily (4 am) and was serviced every one to two weeks.

Fig. 4b. An ONSET HOBO U20L-004 data logger. Each pair of data loggers measured hourly water and air pressure to calculate hourly stream stage (height), and air and water temperatures.

The data loggers were programmed to record hourly pressure and temperature data. The stage data and the stream discharge measurements from the weekly to bi-monthly site visits established a rating curve, the relationship between stream height and stream discharge. The rating curve was then used to calculate a stream discharge for every ISCO water sample.

Laboratory Analyses: Laboratory analyses for nutrient, chlorophyll-a (only lake samples), and total suspended sediment concentrations were determined in Halfman's research lab following standard limnological techniques[3]. Briefly, an aliquot of each water sample was analyzed for total phosphate using a colorimetric analysis by spectrophotometer after digestion of any organic-rich particles in hot (100ºC) persulfate for 1 hour. Additional sample water was filtered immediately on our return from the field through pre-weighed, 0.45 μm glass-fiber filters and the filtrate was stored at 4°C until dissolved phosphate (SRP), nitrate and dissolved silica colorimetric analyses by spectrophotometer. The filter and residue were dried at 80°C for at least 24 hours. The weight gain and filtered water volume determined the total suspended sediment concentration. A known volume of lake water was also filtered through a Gelman HA 0.45 μm membrane filter, and the filtered residue was kept frozen until chlorophyll-a analysis by spectrophotometer after pigment extraction in 90% acetone. Laboratory precision was determined by periodic replicate analyses resulting in the following mean standard deviations: total suspended sediments ±0.2 mg/L, phosphate ±0.1 μg/L (both TP and SRP), silica ±5 μg/L, and nitrate ±0.1 mg/L. For the plankton enumerations, over 100 individuals were identified to genus (and typically species) level and reported as date averaged relative percentages. Multiple reagent blanks and standards were run for each analysis for a constant check on data quality. The nitrate triplicate blanks and standards occasionally yielded concerns.

PRECIPITATION IN 2018 RESULTS & DISCUSSION

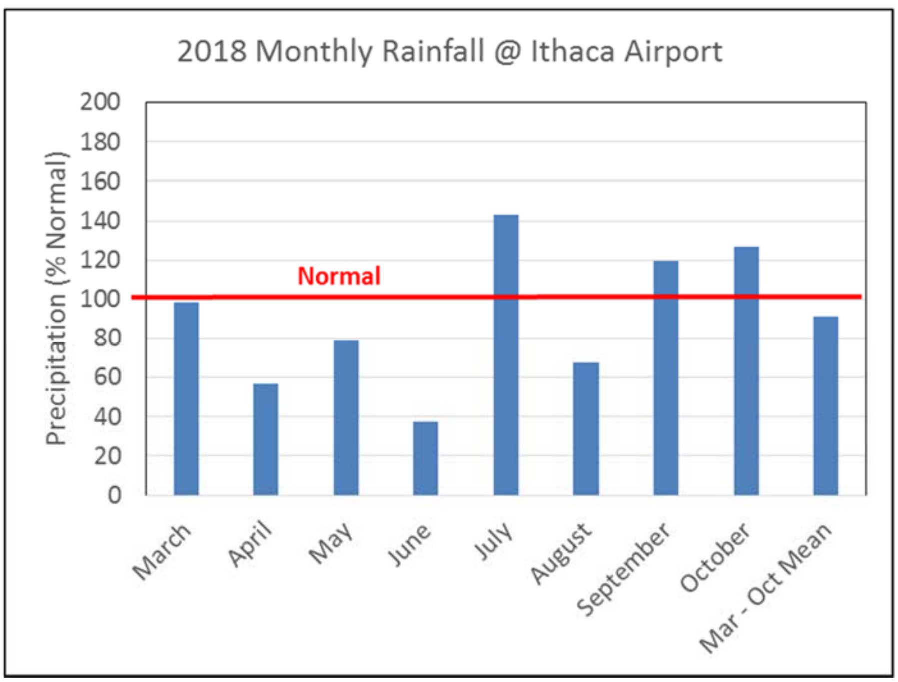
Previous reports concluded that annual rainfall totals, its seasonal variability and individual rainfall event intensities influence the delivery of nutrients and sediments to the lake and thus water quality in the lake. On annual time scales, rainfall was proportional to runoff and associated soil erosion. More rainfall delivers proportionally more nutrients and sediments to the lake, especially rainfall totals in the spring season. Seasonality also influences soil saturation,

[3] Wetzel and Likens, 2000. Limnological Analyses, 3rd Edition. Springer-Verlag, New York.

water infiltration and evapotranspiration rates, and the extent of plant cover on agricultural lands (e.g. spring tillage for planting). Thus it influences the percentage of rainfall that enters runoff that in turn dictates and the delivery of nutrients and sediments by streams to the lake. During the spring and early summer, saturated or nearly saturated soils and less evapotranspiration dominate. Soils become increasingly more unsaturated and evapotranspiration increases in the summer. The fall is typically in between. The percentage of rainfall that enters runoff increases with less infiltration and less evapotranspiration, and more soil erosion results from ground without vegetation, i.e., unplanted fields. Thus, a spring rainstorm produces proportionally more runoff and more soil erosion than a summer or fall event.

Rainfall in 2018 was below normal in the spring (62% below normal) and was closer to normal during the second half of the summer (92% normal) and above normal in the fall (123% above normal, Fig. 5). Thus, the reduced rainfall in the spring through early summer should decrease runoff and nonpoint source nutrient and sediment loads to the lake but loads should become more typical in the fall.

Compared to earlier years, 2018 was drier than 2011, 2013, 2014 and 2015, and 2018 was similar to or slightly wetter than 2012 and 2016 (Gig. 20). Compared to 2017, 2018 was much drier in the spring. It suggests that Owasco Lake water quality should remain the same or slightly improve from 2017 to 2018.



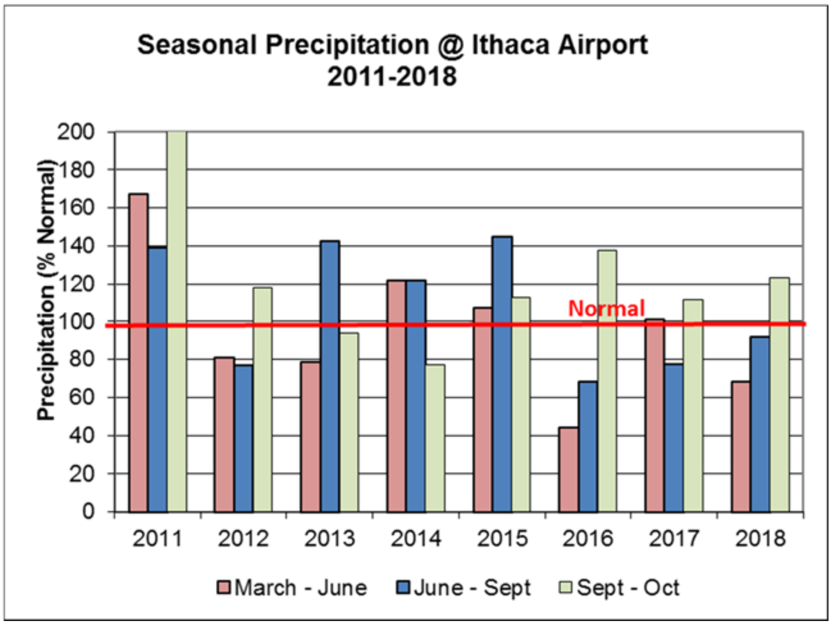


Fig 5. 2018 Monthly (above) and 2011 – 2018 spring, summer and fall seasonal precipitation totals (below) compared to normal totals at the Ithaca Airport.

LAKE MONITORING RESULTS & DISCUSSION

Lake CTD & Bbe FluoroProbe Profiles: The 2018 offshore water temperature profiles revealed a slightly warmer than typical late spring through early fall transition (Fig. 6). The 5/23 cast revealed the initial establishment of seasonal stratification, the initiation of less dense and warmer epilimnion (surface water) overlying the denser and uniformly cold hypolimnion (bottom water). The thermocline, the boundary between the surface and bottom waters was between 10 and 15 meters for most of the stratified season. Epilimnetic temperatures ranged from 10ºC (~50ºF) in late May to 25ºC (~77ºF) in early August, and cooled to 19ºC (66ºF) through the last cruise of the survey (10/3). Hypolimnetic temperatures remained cold, warming from 4º to 5ºC (~39ºF) through the survey.

Epilimnetic salinity (specific conductance) ranged from 293 to 325 μS/cm in 2018 (~150 ppm TDS). Like previous years, epilimnetic salinity in 2018 decreased by ~25 μS/cm (~10 ppm TDS, a small amount) from the largest values detected in the late spring to the lowest values in late summer as the epilimnion was progressively diluted by less saline precipitation and stream runoff. The 2018 early spring specific conductance was similar to those detected in in previous years (2015-2017), and all four years (2015-2018) were slightly larger than earlier years. The annual change in salinity is interpreted to reflect the extent of road salt application during the preceding winter, e.g., the larger salinity in 2015 was due to more snowfall and road salt the previous winter, concentrations which carried over into the spring of 2018.

The 2018 hypolimnetic specific conductance data were just above 330 μS/cm and increased slightly to 338 μS/cm over time and depth (Fig. 6). These values were similar to those in 2017 and slightly smaller to those in 2015 and 2016, and all three years were 10 to 20 μS/cm larger than previous years. Again, the use of an estimated 10,000 tons of additional road de-icing salt from the larger and more frequent snowfall in 2014 & 2015 probably maintained the slightly larger hypolimnetic salinity in 2014 and 2015 than earlier years, and the larger concentrations carried over into 2016 until the dilution by spring rains in 2017.

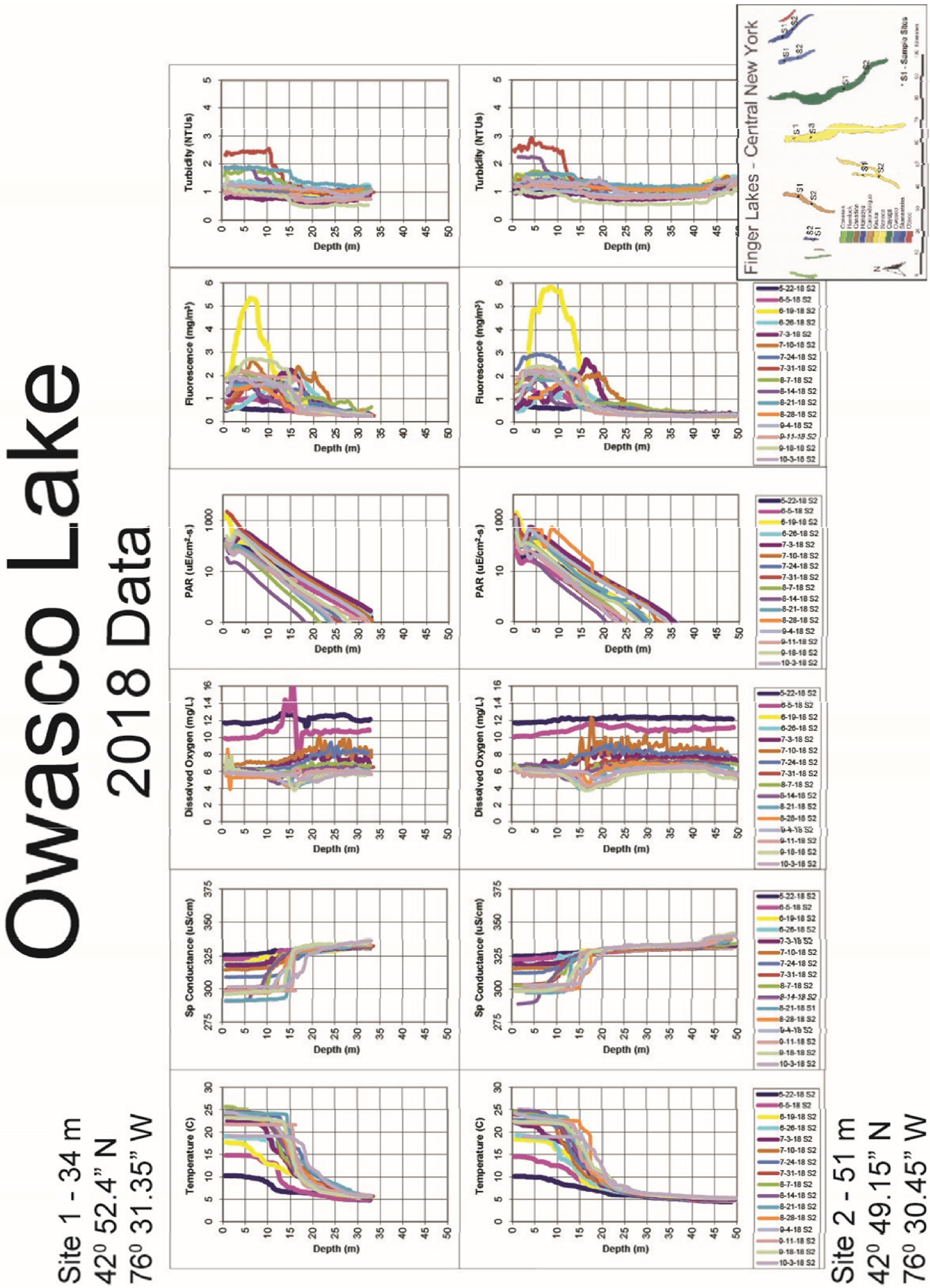


Fig. 6. CTD profiles from Sites 1 & 2 in 2018. The PAR (light) data are plotted on an exponential scale, so that the expected exponential change in light intensity with water depth appears as straight lines.

The 2018 epilimnetic dissolved oxygen (DO) concentrations remained between 8 and 15 mg/L, and near 100% saturation (Fig. 6). However, hypolimnetic DO concentrations were progressively depleted below saturation through the stratified season to just above 5 mg/L (~50% saturation) in the upper hypolimnion and 7 mg/L (~60% saturation) in the lowest hypolimnion by late summer. These lowest saturation levels approached the threshold for respiratory stress in sensitive organisms. The decrease is interpreted to reflect hypolimnetic bacterial respiration and decomposition of dead algae. The depletion was slightly less severe in 2018 than the past three years and most likely reflected a decrease in algal productivity during 2018.

Profiles of photosynthetic available radiation (PAR), i.e., light intensity, in 2018 were similar to earlier results (Fig. 6). Light decreased exponentially with water depth from a maximum intensity of a few 100 to a few 1,000 $\mu E/cm^2$-s at the surface to 1% of surface light intensities within the epilimnion at water depths of 10 to 15 m. The observed decrease in light reflects the preferential and normal exponential 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. The range in surface intensities reflected the season, the extent of cloud cover, and the turbidity of the water (suspended sediment and/or algal density) on the survey date. The 1% of surface light threshold represents the minimum amount of light required for algae to photosynthesize enough biomass to survive. Thus, algal photosynthesis and growth was restricted by light to the epilimnion in Owasco Lake. Many of the profiles revealed a marked decrease in light at 2 or 3 meters. It corresponded to the sensor passing through the shadow of the boat.

Fluorescence, a measure of total algal concentrations, revealed peaks in algal abundance within the epilimnion at approximately 5 to 15 and occasionally 20 m below the lake's surface (Fig. 6). Peak concentrations exceeded 4 μg/L (mg/m^3) on 6/19, and above 3 μg/L (mg/m^3) on 7/3, and were lower, between 1 and 2 μg/L, on the other survey dates. These concentrations were slightly smaller than previous years. It parallels the decrease in spring and early summer rainfall, as runoff is the primary source of new nutrients for large algal blooms. Internal loads from the sediments and benthic organisms (mussels) may provide another viable source. Hypolimnetic concentrations were consistently below 1 μg/L, i.e., algae are typically absent in the dark bottom waters.

The turbidity profiles revealed uniform or nearly uniform turbidities from 1 to 2 NTUs down to the lake floor (Fig. 6). On 7/3, it was above 5 NTUs in the epilimnion, and aligned with an algal bloom, suggesting that 2018 epilimnetic turbidities reflected algal densities more than runoff of suspended sediments. At Site 2, the typical increase in turbidity just above the lake floor detected in earlier years was not observed in 2018. It parallels the lack of spring and early summer rainfall bringing in turbid runoff. The change in benthic turbidities from year to year typically parallels the change in rainfall and wind velocities, as the primary source of suspended sediments (turbidity) is runoff events from precipitation and snowmelt and resuspension events by waves.

The bbe FluoroProbe data revealed the dominance of diatoms and green algae (peaks up to 10 μg/L), and lesser amounts of cryptophytes (below 5 μg/L) and blue green algae (below 0.1 μg/L). Mean epilimnetic total fluorescence concentrations exceeding 10 μm/L (mesotrophic/eutrophic threshold) was detected at one or more offshore sites on six of the thirteen surveys of the lake (Fig. 7).

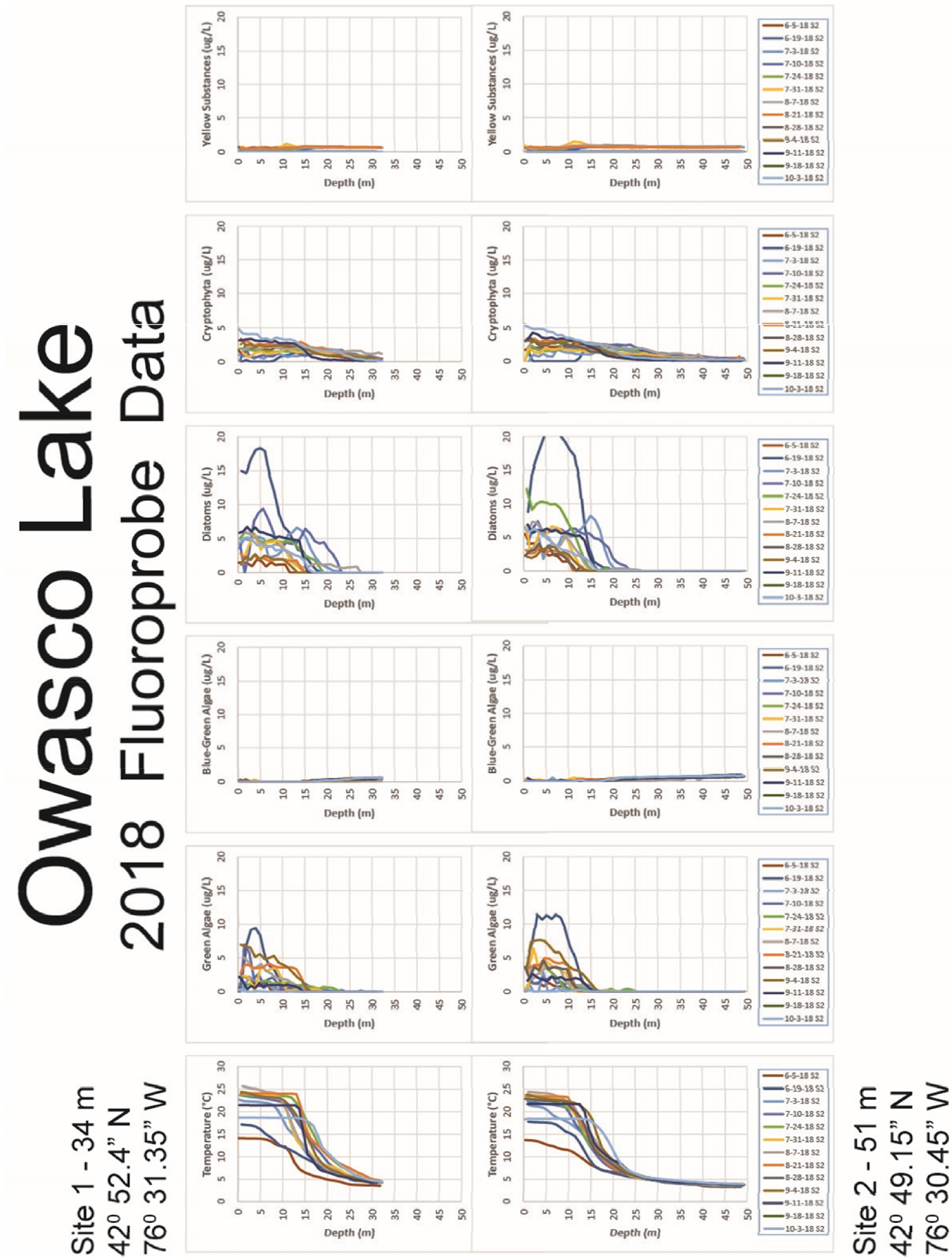


Fig. 7. Bbe FluoroProbe profiles from Sites 1 & 2 of the four algal groups.

Limnology & Trophic Status: Date averaged mean chlorophyll concentrations in the epilimnion ranged from 0 to 6.6 μg/L in 2018 (Table 2 in appendix, Fig. 8). The largest values were on 9/4, a day when of a number of significant BGA blooms were also detected around the shoreline. The annual mean of 2.4 μg/L was below the 4 to 6 μg/L not to exceed DEC threshold for potable water bodies[4]. Nitrate concentrations ranged from 0.4 to 0.8 mg/L, and an order of magnitude (10 times) below the 10 mg/L maximum contaminant level (MCL) established by the EPA. The lake was not impaired due to phosphorus, as the annual mean total phosphate concentration was 13.6 μg/L, below the 20 μg/L total phosphate (TP) threshold used by the DEC to designate impaired (eutrophic) water bodies. Three dates, 6/5, 9/4 and 9/11 were an exception, with a date-averaged TP concentrations of 21, 30 and 21 μg/L, respectively. September 4th and 12th experienced a blue-green algae blooms at numerous shoreline sites around the lake. Secchi disk depths ranged from 2.1 to 6.1 meters, and averaged 3.9 meters in 2018 (Fig. 8). This annual average was in between annual averages detected in earlier years by the FLI monitoring effort. Total suspended sediments (TSS) date-averaged concentrations ranged from 0.8 to 3.1 mg/L and averaged 1.9 mg/L. The 2018 TSS data were in between values measured in previous years.

Annual mean Secchi depths disk gradually deepened from 2009 through 2012 but shallowed since, except for a reversal in 2016 and again in 2018. It suggests that the major trigger for the decline in water quality during 2014 and 2015 and again in 2017 was the larger spring rainfalls and/or more intense rainfall events in those years. It also suggests that the "dry" conditions in 2016 and the reduced spring though mid-summer rainfall in 2018 allowed the lake to recover.

Since 2005, annual mean total phosphate (TP) concentrations have increased from ~8 to over 17 μg/L by 2015 with a slight dip in 2013 (Fig. 8). After another dip in 2016, TP increased to 16.2 μg/L in 2017. Since then, TP decreased to 13.6 μg/L in 2018. Annual mean soluble reactive phosphate (SRP) concentrations returned to low concentrations in 2018. The only years with significantly larger SRP concentrations were 2006 and 2017 (1.9 μm/L), and especially in 2014 (5.8 μm/L, Fig. 8). The large 2014 mean was biased by a sample collected immediately after intense May rains. Interestingly, mean annual SRPs in 2016, a "dry" year, and 2017 an "in-between" year were 2nd largest to 2014. Reduced external sources in 2016 suggests that decomposition of organics within the lake may provide a critical SRP source. Chlorophyll-a concentrations were larger in 2009, 2010, and again in 2014, 2015 and 2016 (3.9, 3.7, 3.2, 3.8 & 3.5 μg/L, respectively) than 2011, 2012 and 2013 (1.9 to 2.3 μg/L; Fig. 8). The 2018 annual mean concentration continued the decreasing trend from 3.8 in 2015 to 2.4 μg/L in 2018. The total suspended sediment (TSS) annual mean concentrations continued their decline from a peak of 3.5 in 2014, down to 2.1 in 2015, 1.8 mg/L in 2016, 1.9 mg/L in 2017, and 1.7 mg/L in 2018 (Fig. 8). In summary, 2014 and 2015 revealed the worst water quality for the lake. Water quality improved in 2018. The trends parallel changes in rainfall. It indicates that rainfall totals, intensities and seasonal timing impact water quality.

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[4]Callinan, C.W., J.P Hassett, J.B. Hyde, R.A. Entringer & R.K. Klake. 2013. Proposed nutrient criteria for water supply lakes and reservoirs. American Water Works Association Journal, E157-E172.

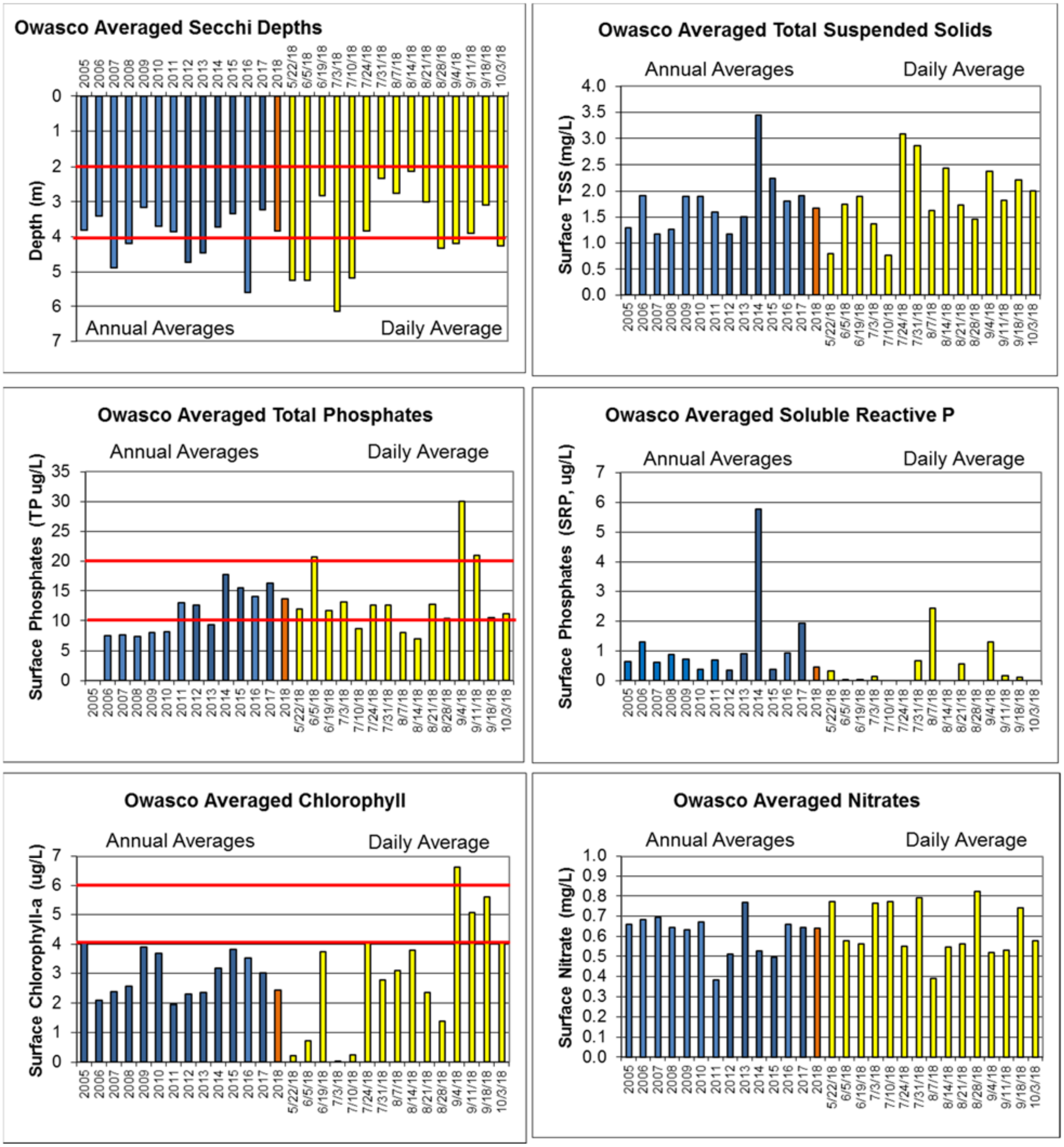

Fig. 8. Annual average surface water concentrations from 2005 (blue) to 2018 (orange), and date averaged offshore surface water data from 2018 (yellow). When appropriate, boundaries for oligotrophic, mesotrophic and eutrophic concentrations are marked.

The 2018 annual mean Secchi disk, total phosphate, chlorophyll-a and hypolimnetic dissolved oxygen saturation data (from the buoy) placed Owasco Lake slightly above the oligotrophic-mesotrophic trophic boundary (Table 3, Fig. 8). Nitrogen, measured by nitrate concentrations, and chlorophyll data placed Owasco Lake below the boundary. Thus, the trophic status of Owasco Lake improved slightly over the past few years. The fluctuations above and below the boundary however, indicates that the lake is in a delicate balance. Any increase or decrease in nutrient loads from one year to the next influence the lake's water quality.

Table 3. Concentration ranges for Oligotrophic (low productivity), Mesotrophic (mid-range productivity), and Eutrophic (high productivity) lakes. The bold entries reflect Owasco's 2018 annual mean values.

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

A few additional observations about the limnological data are noteworthy. First, the mean, surface water, dissolved phosphate to nitrate ratio in the lake, the two nutrients that typically limit algal growth, averaged 1:1,460 in 2018. The P:N ratio required by algae is 1:7 (Redfield Ratio). The measured ratios indicate that phosphate has consistently been (since the start of the FLI monitoring effort) the limiting nutrient in Owasco Lake. The limiting nature of phosphorus is unlikely to change because fluvial sources yield 30 times more nitrogen than phosphorus, and fluvial sources of nitrates are augmented by additional sources of nitrogen to the lake (e.g., atmospheric acid rain nitrates) not available to phosphorus. Second, variability was observed in every parameter from one survey date to the next (Fig. 8). The extent of the variability is best observed in the box and whiskers plots (Fig. 9). It reflects, for example, that algal blooms do not persist the entire summer but are instead episodic and bloom for a week or so at a time before nutrient limitations or grazing by zooplankton and mussels decrease the algal concentrations. Third, the dissolved nutrient concentrations revealed slightly larger concentrations in the hypolimnion than the epilimnion. The annual mean surface and bottom water concentrations were 0.4 and 1.0 μg/L for SRP, 0.6 to .9 mg/L for nitrate, and 790 to 1,430 μg/L for dissolved silica. Chlorophyll-a concentrations revealed the expected decrease from the epilimnion to the hypolimnion of 2.9 to 0.4 μg/L. The separation highlights the expected algal uptake of nutrients in the epilimnion and bacterial decomposition of organic materials and release of nutrients in the hypolimnion. Finally, 2017 mean TP, TDP, chlorophyll-a and nitrate surface concentrations determined by C-SLAP efforts were similar to the results from this study (Fig. 9). C-SLAP's TP, TDP and N were within the "box" of the box and whisker plots, and chlorophyll just above the "box". This is good.

Plankton Data: The phytoplankton (algal) species in Owasco Lake during 2018 were dominated by diatoms, primarily *Synedra* and *Asterionella*, with smaller numbers of *Diatoma*, and *Fragillaria*, (Table 4 in appendix, Fig. 10). Unlike previous years, *Synedra* replaced *Asterionelia* and *Diatoma* as the dominant taxa for most of the summer. The reason for the mid-summer *Synedra* dominance is unclear at this time and it has never dominated the algal population in the past. We openly speculate that the ecology of the lake may be changing. Besides blue-greens, other phytoplankton species included a few *Dinobryon* and *Coalcium*. Zooplankton species were dominated by rotifers, namely *Keratella* with some cladocerans, like *Copepods*, and *Cercopagis*, the fishhook water flea. Zebra and quagga mussel larvae were also detected in the plankton tows.

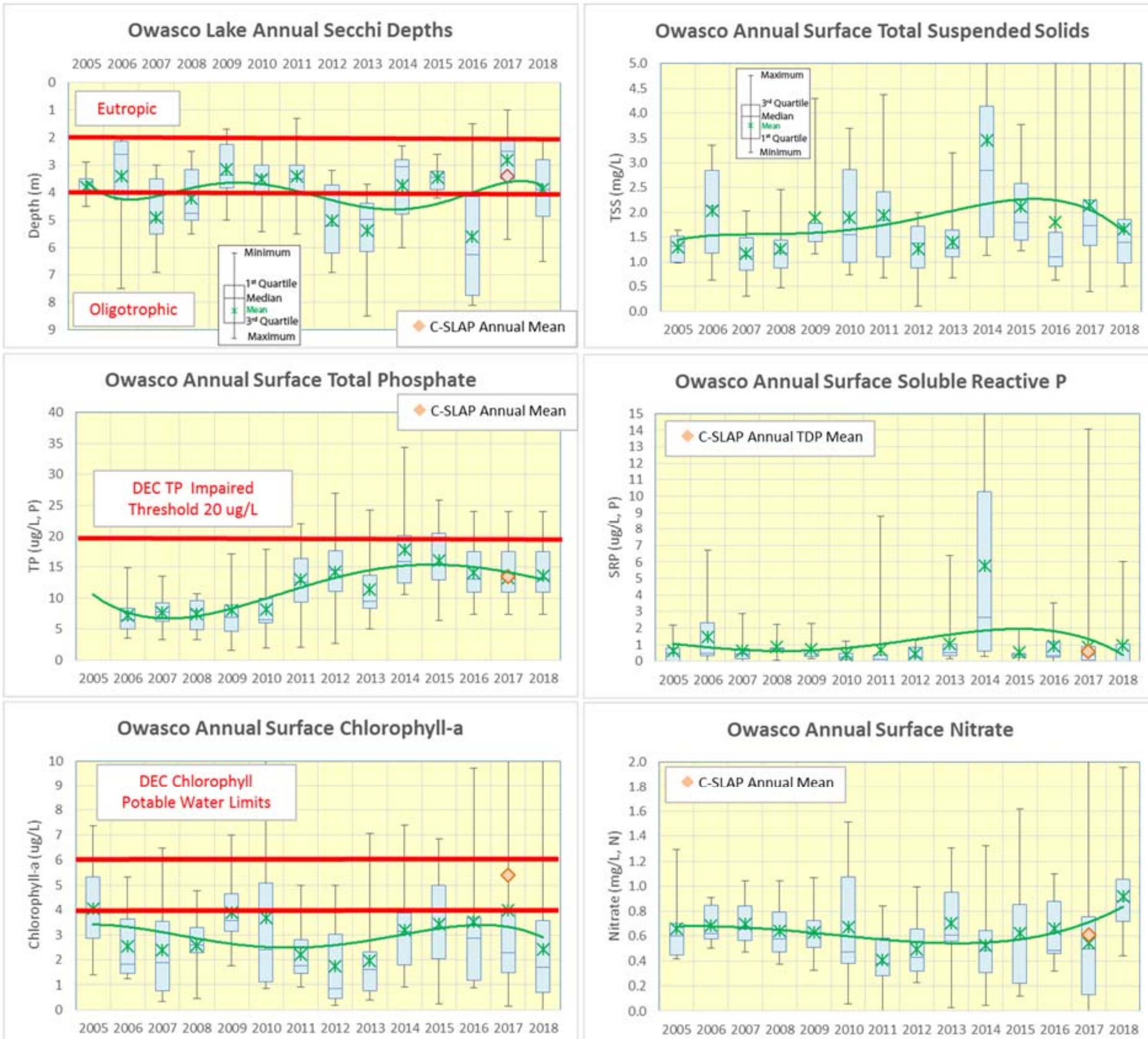

Fig. 9. Box and Whisker plots of the annual nutrient, chlorophyll and Secchi disk data. The mean C-SLAP results are also plotted (2017 only, orange diamonds). DEC measured TDP instead of SRP.

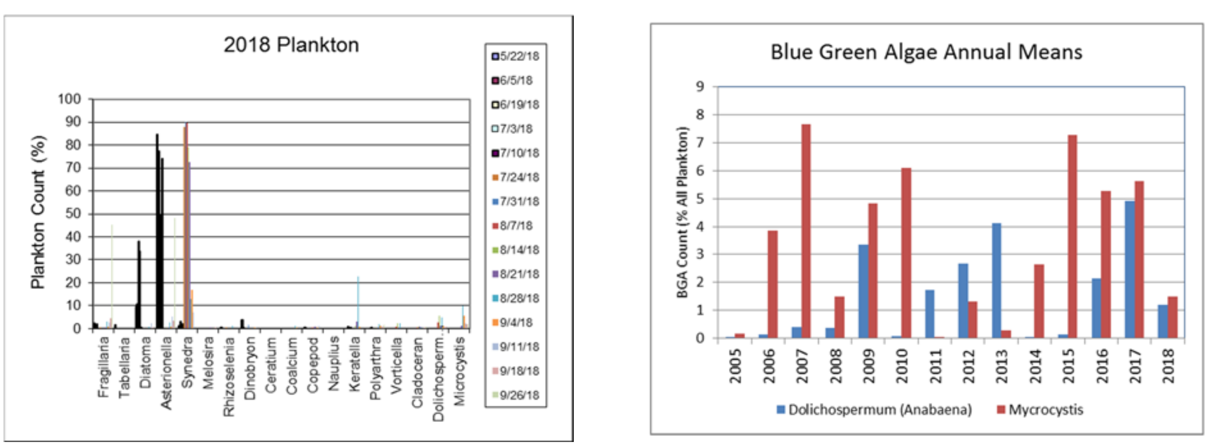

Fig. 10. Date averaged plankton data for 2018 (left) and the mean annual abundance of blue-green algae species since 2005 (right).

Two blue-green genera, *Microcystis* and *Dolichosperma* (*Anabaena*), were prevalent in the late summer and early fall 2018 surveys (Fig. 10). Detection of BGA in the lake is not new. BGA were always detected in the open water of Owasco Lake since the initial FLI surveys in 2005. However, the annual means never exceeded 10% for any BGA species at these open water sites (Fig. 10). Typically the largest populations were restricted to the late summer and/or early fall, with *Microcystis* representing nearly 40% of the plankton counts during a late summer survey in 2007, 2010, 2014, 2015 and 2017, and *Anabaena* making up 30% of the late-summer counts in 2013. In fact, blue-green species were detected in a neighboring Finger Lake as long ago as 1914[5]. However, major blooms of BGA have been increasingly detected along the shoreline in Owasco Lake since 2012[6]. In 2018, the offshore presence of blue-green algae had decreased, with a maximum relative percentage of 10% during one survey. The companion reports on the 2018 nearshore analyses and the 2018 mitigation technology assessment have more details.

Finger Lake Water Quality Ranks: The 2018 Finger Lake water quality ranks still place Owasco Lake as one of the worst lakes among the eight easternmost Finger Lakes (Table 5 in appendix, Figs. 11 & 12). The ranks were based on annual average Secchi disk depths, and surface water concentrations of chlorophyll-a, total and dissolved phosphate, nitrate and total suspended sediments collected by the May through October, monthly FLI survey. These ranks revealed similar trends as other comparative water quality / trophic state methods like the oligotrophic-eutrophic trophic states (discussed above), and Carlson's Trophic Indices[7] that quantitatively combine chlorophyll-a, total phosphorus and Secchi depth data (Fig. 11). In 2018, water quality in Owasco ranked poorer than Canandaigua, Keuka and Skaneateles Lakes, and slightly better than Seneca, Cayuga, and Otisco Lakes, and much better than Honeoye Lake. Interestingly, all of the lakes except Seneca revealed generally better water quality in 2018 than earlier years (2014-2017). It indicates that rains and/or intense rain events in 2014, 2015 and 2017 induced sufficient nutrient and sediment loads to degrade water quality in all the Finger Lakes, and 2016 and 2018, relatively "dry" years, were years of recovery.

The change in water quality among lakes is also influenced by a number of other sometimes competing and always intertwined factors. First and foremost, the degree of water quality protection legislation and its implementation, that protect the lakes from nutrient and sediment loading issues. So does ecological, "top-down" pressures by zebra and quagga mussels, Asian clams and *Cercopagis*, the fishhook water flea.

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[5] Bloomfield, J.A. (ed.), 1978. Lakes of New York State. Vol.1: The Ecology of the Finger Lakes. Academic Press.
[6] http://www.dec.ny.gov/chemical/83332.html
[7] Carlson, R.E. 1977. A trophic state indicator for lakes. Limnology & Oceanography, 22:361-369.

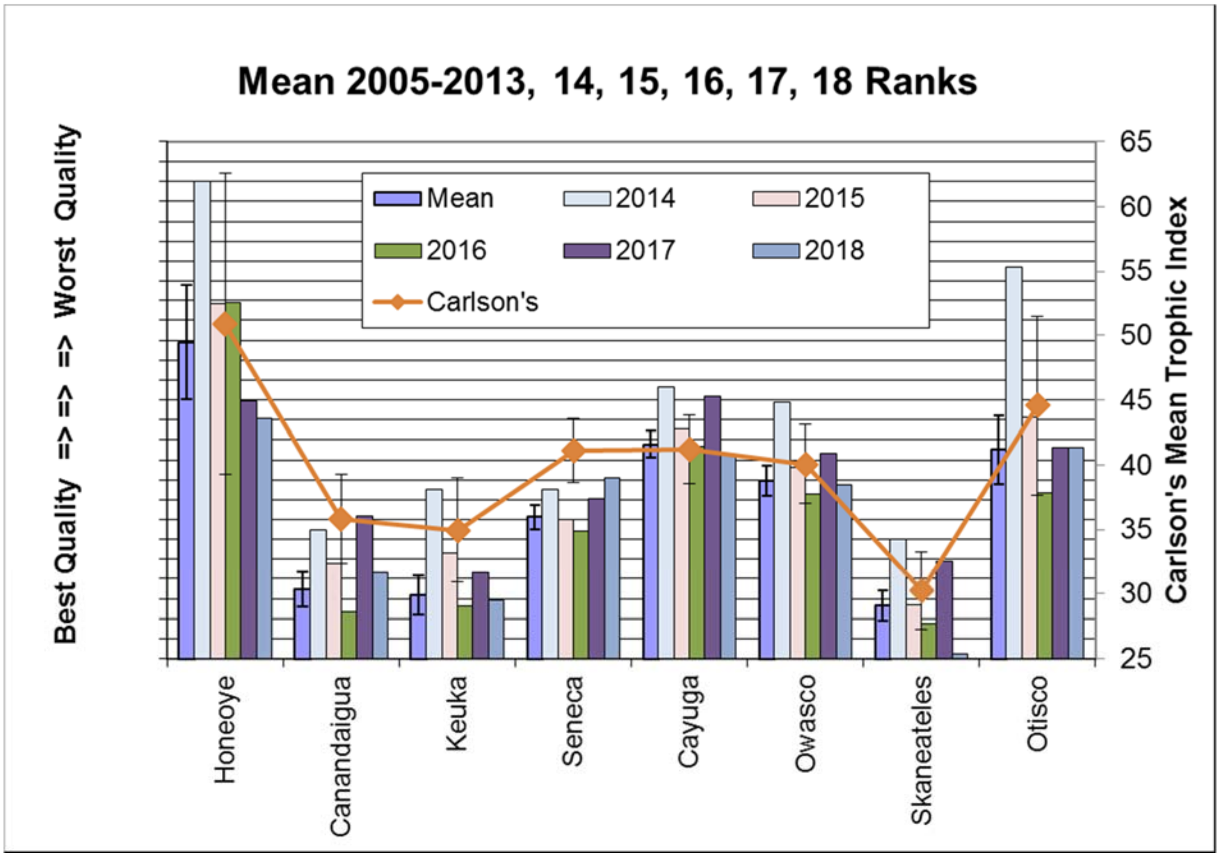


Fig. 11. Annual Water Quality Ranks from 2005 – 2018 for the eight easternmost Finger Lakes. The “mean” dark blue bar averaged the 2005 - 2013 ranks for each lake with a 1σ standard deviation error bar. Carlson’s mean trophic indices of the mean Secchi depths, total phosphate and chlorophyll concentrations are also shown.

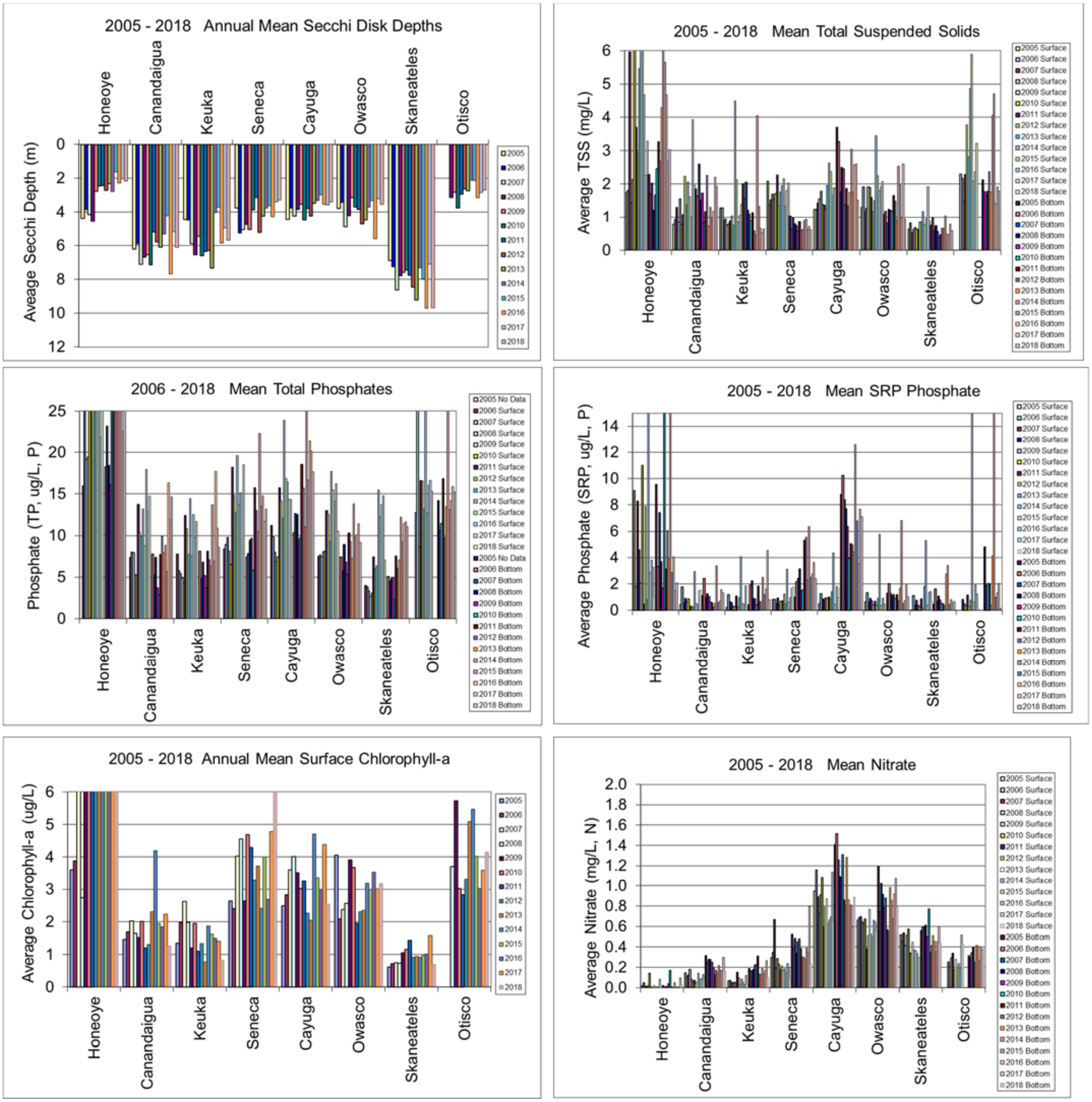


Fig. 12. 2005 – 2018 Annual mean limnological data from the eight eastern Finger Lakes.

DRONE FLIGHTS

Unfortunately BGA blooms were not detected in the drone images during the specific site visits during 2018. On a few occasions, it was noted that the bubbles from the aeration mitigation devices appeared to push surface accumulations of BGA ~30 meters away from where the bubbles break the surface and form an outward propagating circle of waves. Apparently the concentric waves formed by the bubble activity on relative calm days were able to push and concentrate BGA (Fig. 13). This phenomena is discussed further in the nearshore mitigation technology report.

Complete spectra (from 340 to 823 nm at ~0.5 nm intervals) of the upwelling and down-welling light were again collected at a number of sites to find a better indicator of open-water algal concentrations (Fig. 14). The intent was to determine if the difference between upwelling and down-welling spectra could resolve algal concentrations. The 2018 results confirmed the preliminary 2017 results (shown in Fig 14) and revealed potential algal signatures in the near infrared portions of the light spectrum where plants emit the most light (wavelength of 750 nm). More work must be done next summer to improve these techniques, and we propose to continue our periodic drones flights and recovery of spectral signatures on more surveys to assess water quality in Owasco and neighboring lakes, and map the distribution and concentration of nearshore macrophytes, attached algae and blue-green algae blooms in the years ahead.

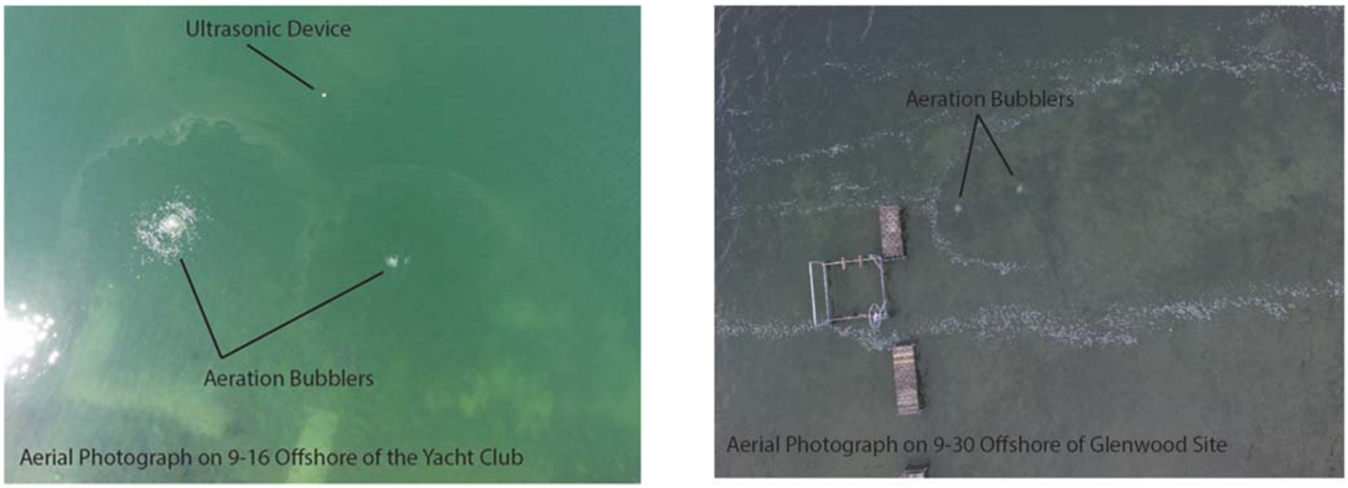


Fig. 13. Drone images from the Yacht Club and Glenwood Rd. BGA mitigation sites. Notice the circular rings of very low concentrations of BGA (left) and the white foam from Langmuir cells around the bubbles formed by the bubbler mitigation technologies.

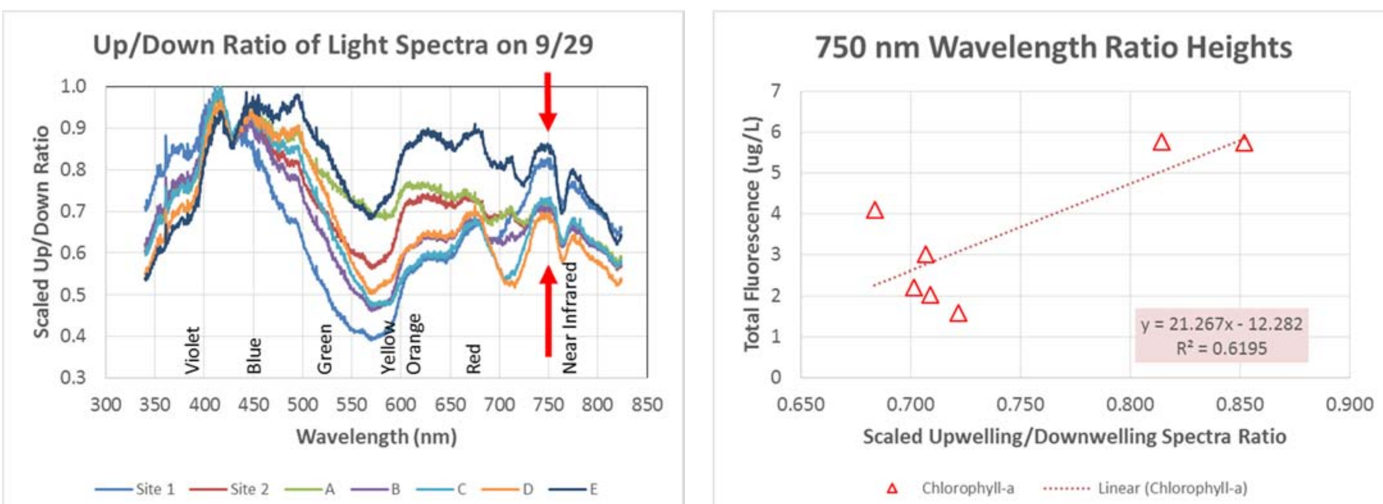


Fig. 14. Ratio of upwelling and down-welling spectra from Sites 1, 2 and A – E on 9/27/17 (left). Plant signatures are typically in the near infrared (~750 nm). Peak heights at 750 nm for each up/down spectral ratio vs. total fluorescence measured by bbe FluoroProbe (right).

FOUR YEARS OF BUOY DATA

The FLI meteorological and water quality monitoring buoy was redeployed in Owasco Lake during the 2018 field season. It revealed higher resolution but otherwise consistent changes in the water column as described in the CTD section (Fig. 15). Epilimnetic (surface water) temperatures increased from 3.5ºC (38ºF) in late April up to 25.7ºC (78ºF) in mid-July through mid-August. Surface waters oscillated between 25 and 22ºC (72ºF) until mid-September when it then cooled down to 13ºC (38ºF) by the end of the deployment (10/29). Hypolimnetic temperatures slowly increased from below 4.5 to 5.5ºC (42ºF) during the deployment (10/29).

In general, all five years revealed similar seasonal patterns explained by the typical daily, weekly and seasonal changes in climate. The epilimnion was slightly warmer in 2018 than 2014, 2015 and 2017, and the hypolimnion was slightly warmer in 2017 that previous years. The 2017 timing of the epilimnetic peak temperature was similar to 2015, warming to 25ºC by late July whereas it warmed to 25ºC by early July in 2014. The seasonal cooling in the fall started earlier in 2014 as well, i.e., the surface waters cooled below 20ºC by mid-September in 2014 but was two weeks later in 2015, 2016 and 2017. The change probably reflected the earlier onset and longer duration of the very cold 2014/2015 winter season. As in earlier years, a few ~2.5°C dips in water temperature were observed in 2018, one in mid-August and another in early-September. It indicates that the surface temperatures responded to variability in rainfall and air temperature.

The depth of the thermocline, the boundary between the epilimnion and hypolimnion, gradually increased through the field season from < 10 m to > 20 m in all five years. The thermocline depth deepened faster during September and October reflecting the vertical mixing of surface water to deeper depths as the epilimnion cooled into the fall, i.e., the gradual decay of summer stratification. It also revealed daily oscillations of 1 to 2 meters in response to internal seiche and/or internal wave activity throughout the deployment in all five years.

The epilimnetic specific conductance (proportional to salinity) decreased from 324 μS/cm in May through early June to 292 μS/cm by mid-August, and then increased by ~20 μS/cm by the end of October (Fig. 15). These changes are small (~20 ppm). The decrease probably reflects the dilution of the epilimnion by stream inputs and rainfall, but their impact is complicated by the relative salinities of the numerous subwatersheds flowing into the lake, and thus the delay in the decrease in 2018 compared to earlier years is consistent with the reduced spring to mid-summer rains. The subsequent surface water increase in salinity reflects the mixing of slightly more saline hypolimnetic water into the epilimnion as the surface waters cool and vertically mixed with deeper water in the fall. The hypolimnetic salinity increased from 323 μS/cm to 334 μS/cm from late April to mid-September, then decreased by a few μS/cm until buoy recovery in late October in 2018. Similar hypolimnetic trends were observed in previous years. The long term trend revealed slightly smaller salinities from 2015 through 2018.

The epilimnetic turbidity was small in 2018 (Fig. 15). The largest turbidities were just above 4 NTUs in late July just after the first major rain storm of the 2018 field season. Runoff associated turbidity spikes were also detected in previous years. With only a few exceptions, epilimnetic turbidities were 1 NTU or smaller reflecting the small rainfall totals in the late spring and early summer. Lake-floor turbidities were also small in 2018 almost as small as those detected in 2016. The differences parallel the amount of precipitation. The change is interpreted to reflect the amount and intensity of the early spring rains and wind/wave resuspension events in the high turbidity years.

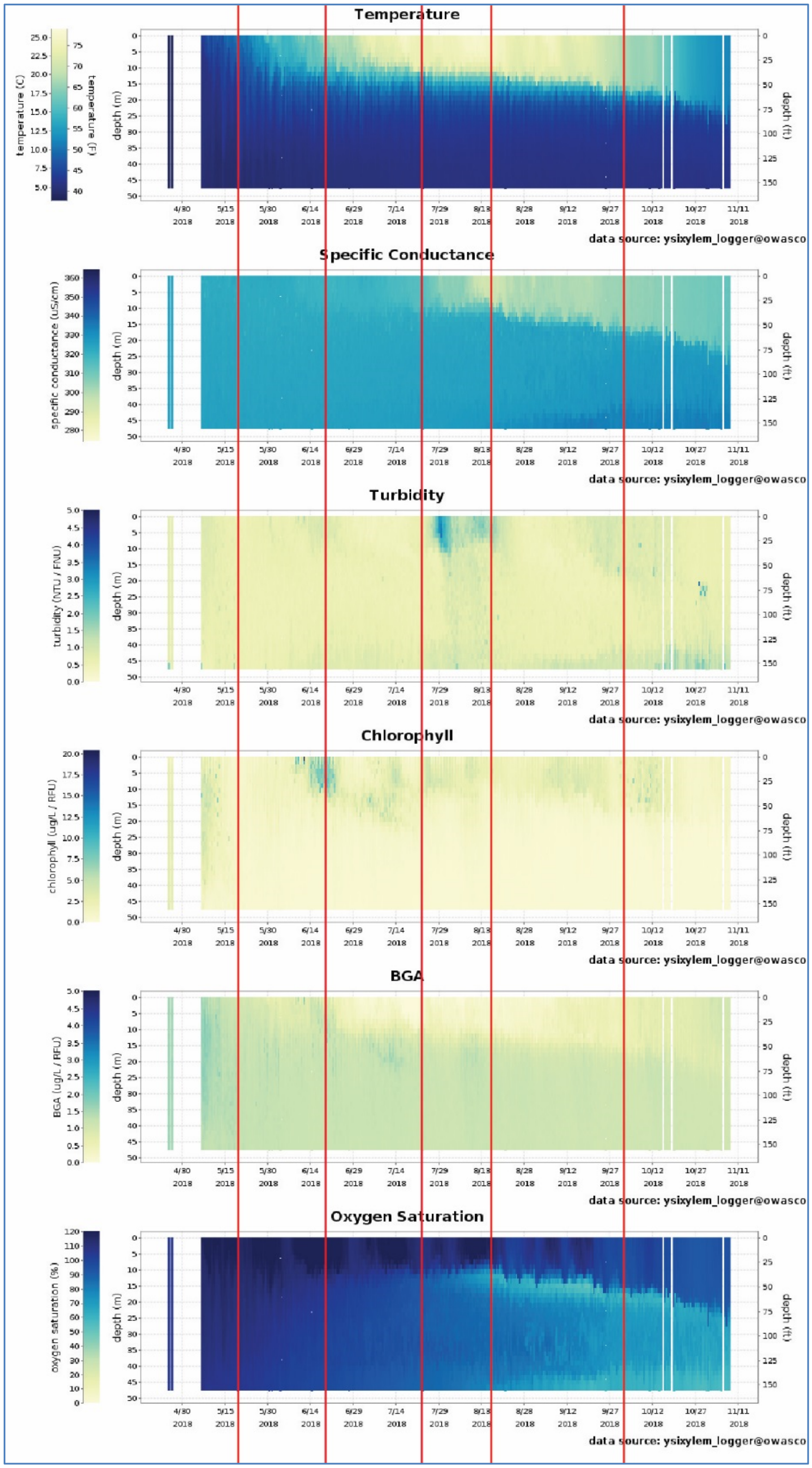
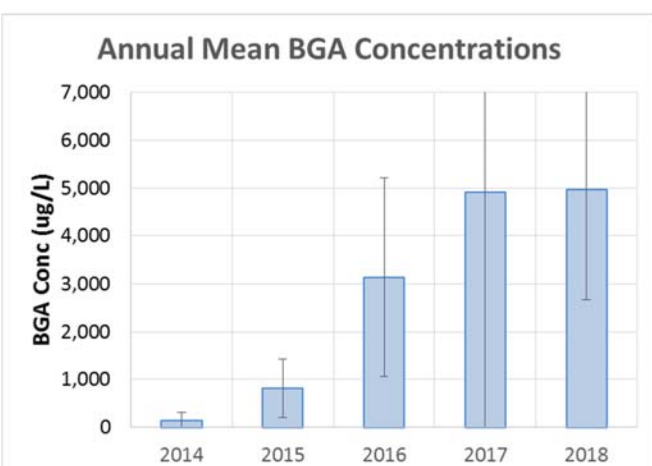


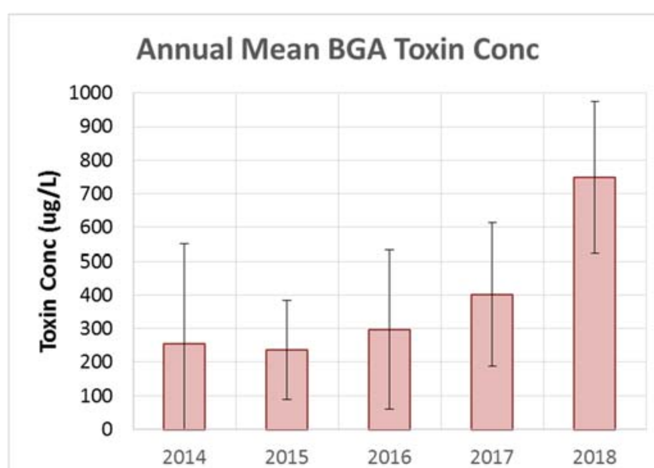
Fig. 15. Buoy water quality data in 2018. Note, algal concentrations shown above are raw uncalibrated values. The red lines depict the monthly monitoring cruise dates.

The chlorophyll-a concentrations measured by the buoy were small and only changed from ~1 to over 6 μg/L in 2018 (Fig. 15). A week long bloom in late June was the only time concentrations exceeding 5 μg/L. The algae were contained within the upper epilimnion (shallower than 10 m). More algae is expected in the surface water (epilimnion) because algae require light for photosynthesis. Smaller blooms were detected in September and October that probably responded to nutrient inputs from rain events, internal seiche activity and/or the decay of the seasonal thermal stratification and mixing of nutrient-rich hypolimnetic waters into the epilimnion. Mean algal concentrations in 2018 were much smaller than previous years and reversed the steady increase from 2014 through 2017. This is good and probably reflects the decrease in early spring and to mid-summer rains in 2018, and any improvements to farming practices, roadside ditches, and other BMPs in the watershed.

BGA concentrations never exceeded a few μg/L at the buoy, compared to the nearshore bloom concentrations of a few hundred to a maximum of just above 14,000 μg/L in 2018 (DEC and Watershed Inspector data, by permission, Fig. 16). The small, open-water, BGA concentrations were confirmed by the bbe FluoroProbe water column profiles (Fig. 7). The discrepancy therefore reflects the surface water and shoreline hugging distribution of BGA blooms. It also confirms that minimal concentrations detected by the buoy in previous years was also due to the mid-lake, open-water, deployment of the buoy and a sensor that cannot measure any shallower than 1 m. It appears that shoreline annual mean BGA concentrations have increased over the past four years from ~10 to ~5,000 μg/L from 2014 to 2017, but stagnated afterwards (Fig. 16). The number of blooms detected along the shoreline declined since 2016, nearly 50% smaller in 2018 than 2016. These trends however might be artifacts of sample bias and/or sampling protocols, as the BGA concentrations typically change radically within a few meters of the shoreline and with water depth.

Finally, epilimnetic dissolved oxygen (DO) concentrations in 2018 were at or just above saturation throughout the deployment (Fig. 15). Hypolimnetic DO concentrations decreased from nearly saturated concentrations in late May to below 50% saturation just below the thermocline by the end of August and down to 60% saturation along the lake floor by the end of September. The depletion reflects the respiration by bacteria, zooplankton and other animals at these depths. A similar pattern in DO was observed in previous years but the depletion was less severe in 2018 than previous years, i.e., it depleted to ~30% saturation in 2015, and extended later into the fall, i.e., into September during 2015 and 2017.





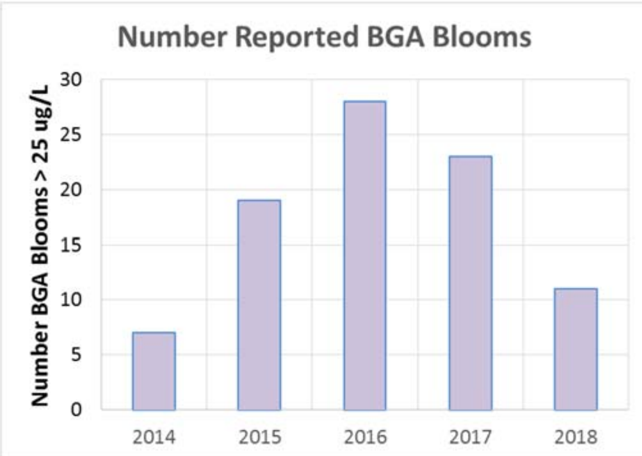


Fig. 16. Annual mean BGA and BGA toxin concentrations (with standard deviations) and number of reported blooms in Owasco shoreline areas.

STREAM MONITORING RESULTS & DISCUSSION

Stream Discharge: Stream discharge data from the five May & June stream survey dates in 2018 ranged from nearly dry, 0.01 m^3/s (6/13) conditions at Fire Lane 20 & 26 to 2.7 m^3/s in Owasco Inlet at Moravia (Table 6 in appendix, Fig. 17). These flows detected at each stream were typically much smaller in 2018 than past years during the same interval of time due to lower rainfall totals in the spring and early summer of 2018. Flows were larger in the April surveys by the Geo Hydrology class, and in August through October during the autosampler service dates. The increase in discharge reflected the spring melt in April and more rainfall during the late summer and fall.

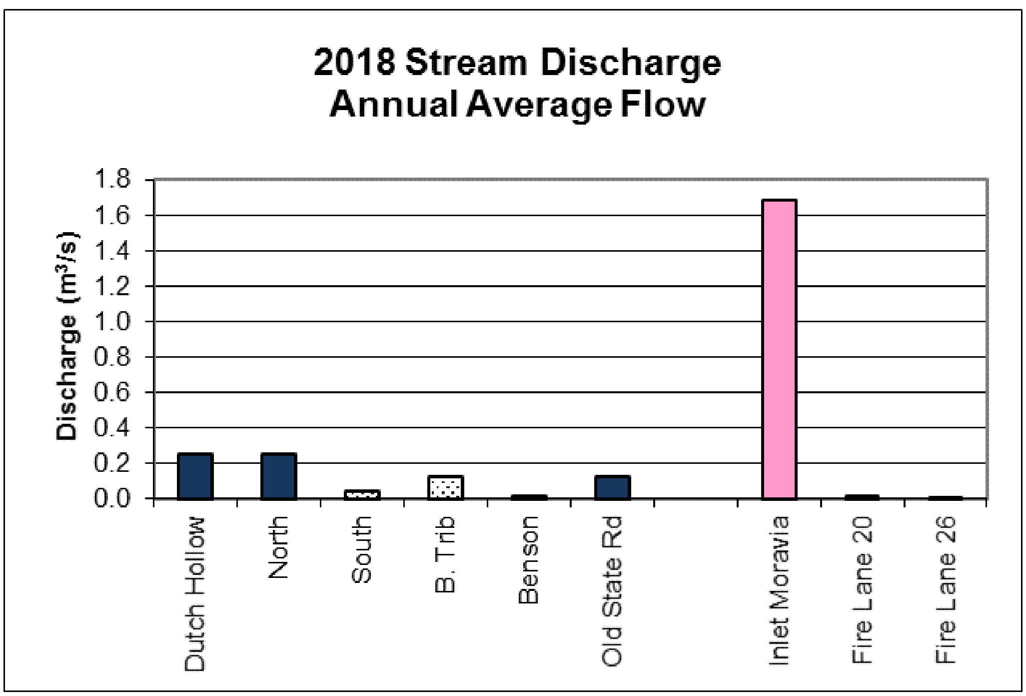


Fig. 17. Annual average stream discharge using the five May & June surveys at each stream site in the Dutch Hollow Brook (purple), Owasco Inlet (pink), Fire Lane 20 and Fire Lane 26 watersheds. Tributary sites along Dutch Hollow Brook are stippled. Sites are arranged, left to right, from downstream to upstream.

Within Dutch Hollow Brook, mean annual discharge at each site, focusing on the May and June surveys, typically equaled or was slightly larger than the sum of the discharges at the next upstream site and any measured tributaries entering along the segment between sites (Fig. 17). For example, the sum of the mean annual discharge at North Rd was similar to the sum of the discharges at South, Benson tributary, and Benson Rd sites. Mean annual discharge was the same from North Rd to 38A, and not a decreasing trend observed in earlier "dry" years. It suggests that surface runoff and shallow groundwater flow persistently contributed to and increased stream discharge from North Rd down to Rt 38A during "wet" years. Whereas, the stream probably lost water to evapotranspiration by plants, and/or into the permeable sand and gravel aquifer at the Dutch Hollow Brook delta, during "dry" years.

Discharge for the Owasco Inlet at Moravia was again much larger than Dutch Hollow Brook because the Owasco Inlet drains a significantly larger watershed than Dutch Hollow Brook (299 vs. 77 km^2).

Seasonal Variability: Seasonally, the largest discharges of 2018 were detected in the spring at Dutch Hollow and in the fall at the Owasco Inlet whereas the smallest were during the summer based on the data logger estimated discharge data for Dutch Hollow at the Rt 38A site and the USGS gauge data for Owasco Inlet (Fig. 18). A larger fall discharge is unusual for the Owasco Inlet. Typically the spring has the largest discharge, by a significant margin. The unusual 2018 trend parallels below normal rainfall in the spring and early summer months and larger than normal rainfall in the fall.

Differences to Earlier Years: The 2018 annual mean discharge was "in-between" those detected at the downstream sites of Dutch Hollow Brook and Owasco Inlet over the past decade (Fig. 19). These differences parallel changes in precipitation.

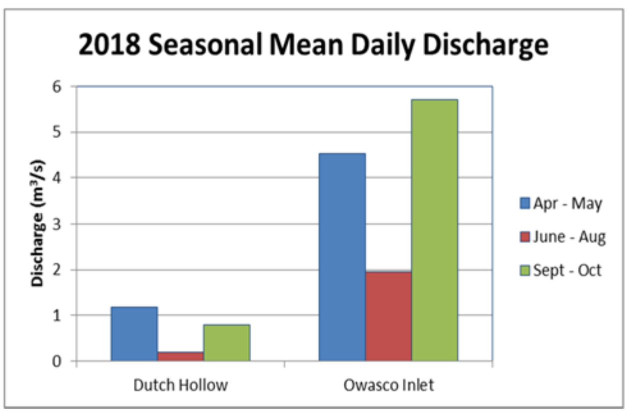


Fig. 18. Seasonal averaged stream discharge for the Rts. 38A and 38 sites, the terminal sites on Dutch Hollow Brook and Owasco Inlet, respectively.

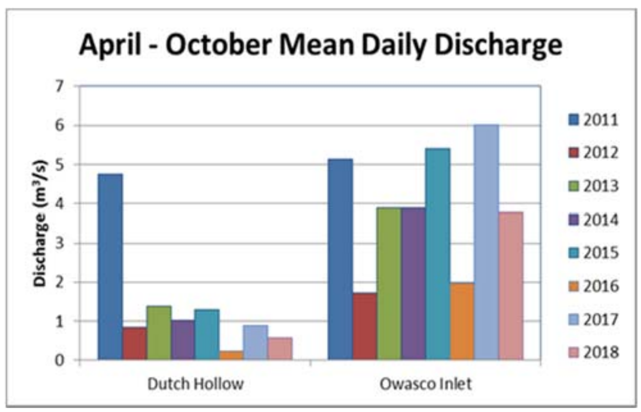


Fig. 19. Field season annual average stream discharge for the Rts. 38A and 38 sites. This plot used the estimated Dutch Hollow Brook data logger and USGS daily Owasco Inlet discharge data.

The Owasco Inlet (USGS Gauge, 4235299) mean, field-season, daily discharge of 3.8 m^3/s in 2018 indicates an "in between" year, rather than the 2011, 2015 and 2017 "wet" years and 2012 and 2016 "dry" years. The field season mean discharges were 5.1, 1.7, 3.9, 3.0, 5.4, 2.0, 6.0 and 3.8 m^3/s from 2011 through 2018, respectively (Fig. 20). Similar variability was observed for the Owasco Outlet (USGS Gauge, 4235440, Fig. 20). Annual mean daily outflows were 11.4, 5.9, 9.1, 8.6, 9.4, 7.3, 12.1 and 8.3 m^3/s for 2011 through 2018, respectively. Clearly, 2013, 2014 and 2018 were "in-between" years compared to the 2011, 2015 and 2017 "wet", and 2012 and 2016 "dry" years. The analysis parallels the dry spring and early summer but above normal rains in the fall of 2018.

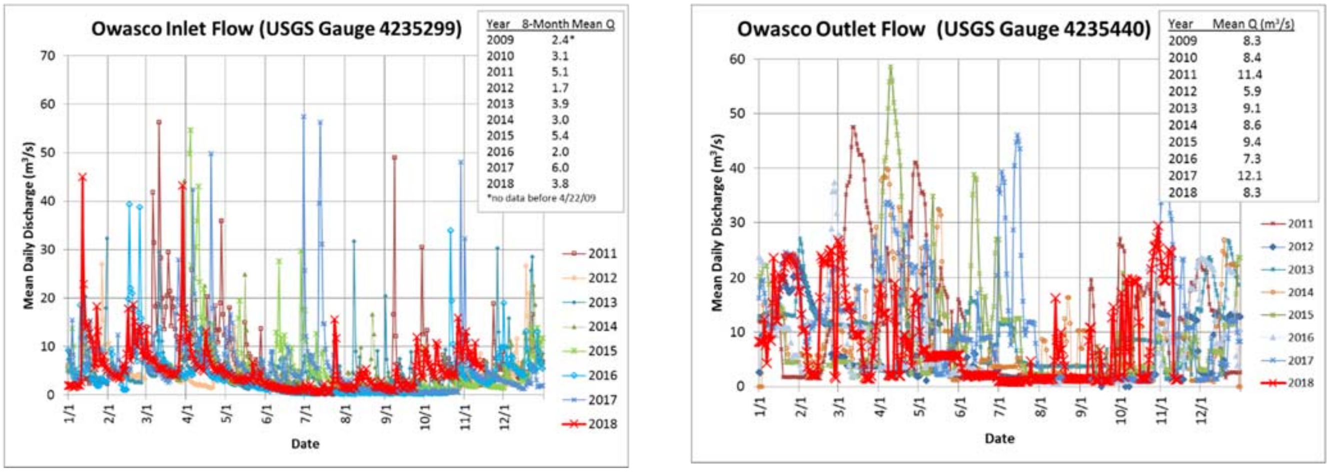


Fig. 20. 2018 annual average stream discharge for the Owasco Inlet near Moravia (left) – USGS Stream Gauge 4235299 and the Owasco Outlet (right) – USGS Stream Gauge 4235440.

Extreme Events: Both the Owasco Inlet and Dutch Hollow revealed precipitation induced events in their hydrology. Box and whisker plots of mean daily USGS discharge data for the May – June period at Owasco Inlet revealed larger mean flows in 2011, 2015 and 2017 and lower mean flows during 2012, 2013, 2014, 2016 and 2018 (Fig. 21). The top whisker in the B&W plot also revealed significantly larger events during 2011, 2014, 2015 and 2017 than the other years in the record as well. A similar pattern is observed at Dutch Hollow Brooks as well. However, its largest events were in 2011, 2014, 2015 and 2017, and 2017 was more important if the event during early July is included (2017* in Fig. 21). This inclusion is important because large events have an exponentially greater impact on nutrient and sediment loads to the lake. The largest events in 2018 were smaller than the largest events in the other years at both Owasco Inlet and Dutch Hollow Brook.

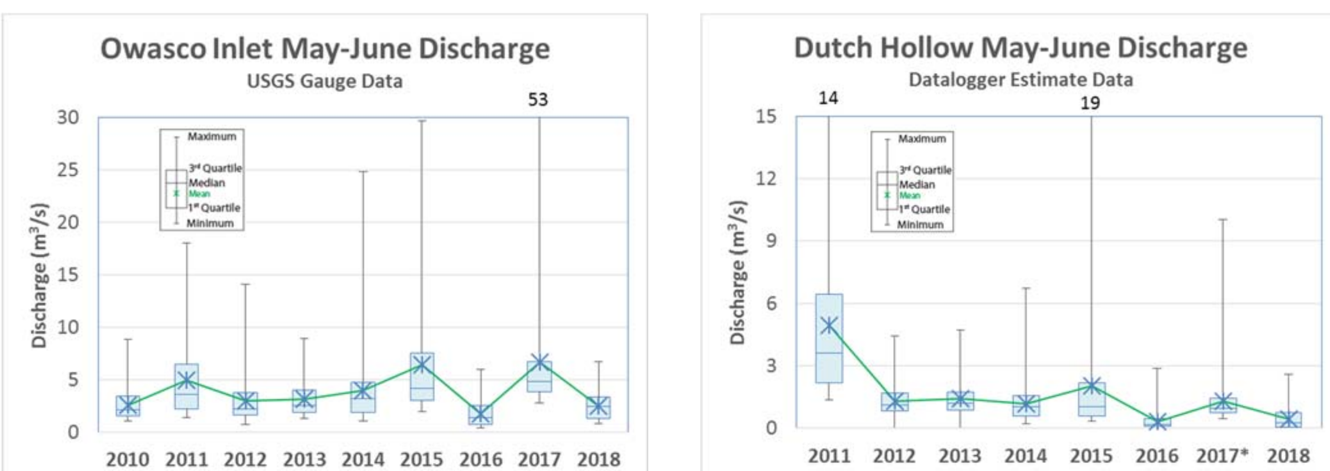
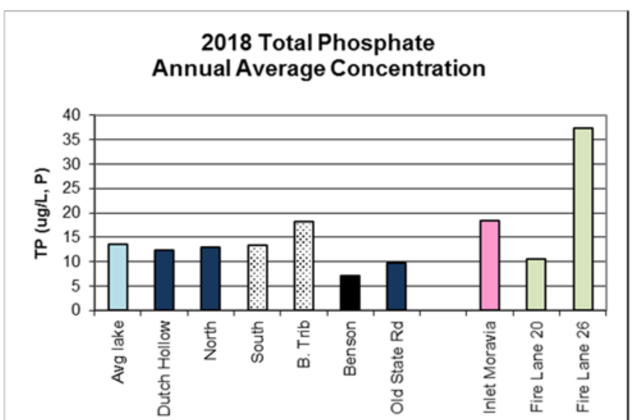
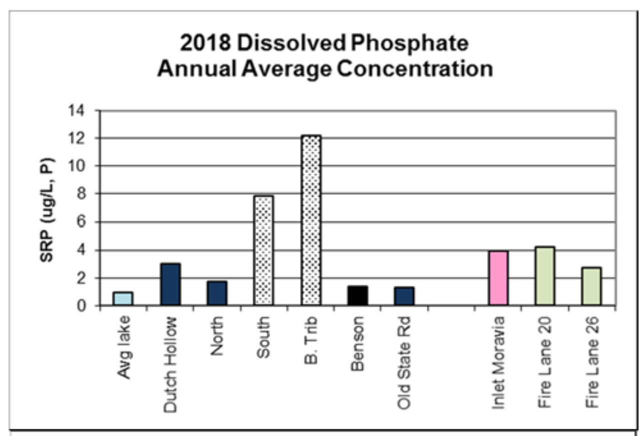


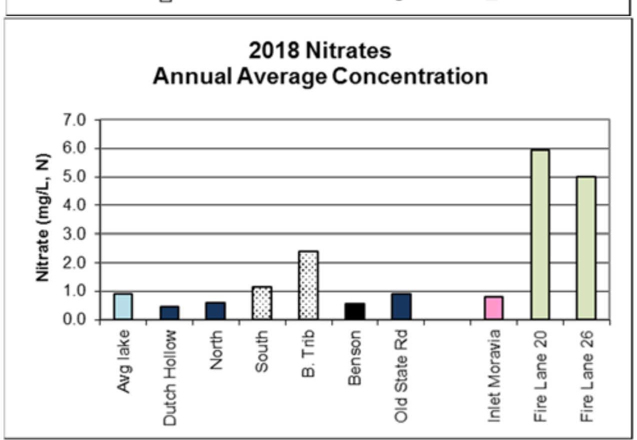
Fig. 21. Box and whisker plots of daily mean discharge during the May – June period over the past eight years for Owasco Inlet (left) near Moravia using the USGS Stream Gauge 4235299 data and Dutch Hollow Brook (right) using the Rt 38A data logger data.

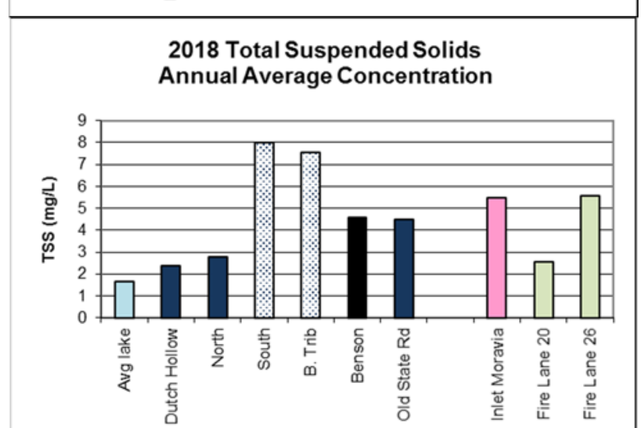
Stream Nutrient & TSS Concentration Data: Total phosphate (TP) concentrations in 2018 ranged from 3 to 53 μg/L, and averaged 12 μg/L at Rt 38A in Dutch Hollow Brook, ranged from 4 to 34 μg/L, and averaged 18 μg/L at RT 38 in Owasco Inlet and ranged from 6 to 142 μg/L, and averaged 24 μg/L at the two Fire Lane sites (Table 6 in appendix, Fig. 22).

Along Dutch Hollow Brook, the Benson Tributary site revealed a slightly larger annual mean TP concentration than the other sites, whereas the Benson Rd site revealed the smallest total phosphate concentration. This trend is more pronounced in the soluble reactive phosphates (SRP), total suspended solids (TSS), nitrates (N) and specific conductance (salinity) annual averages. For example, total suspended sediment (TSS) annual mean concentrations were largest at the Benson and South Trib. sites (8.0 & 7.6 mg/L) compared to elsewhere in the stream. TSS was a few mg/L smaller (<5 mg/L) upstream and even smaller (< 3 mg/L) downstream. It suggests that the low flow in 2018 allowed for deposition and/or dilution of the tributary and upstream TP, SRP and TSS inputs within plant uptake of the nutrients along the stream and deposition of the suspended sediments in the many pools along the main stream channel. Benson tributary has consistently revealed larger annual mean concentrations of TP, SRP and specific conductance (salinity) since 2009 and suggests that agricultural inputs impact the tributary but the impact is masked by larger stream volumes downstream (Fig. 23). It appears that the 2016 reduction of nutrients and suspended sediments loads at the Benson tributary site and a subsequent return to earlier values in 2017 and 2018 were due to decreased rainfall in 2016 and not due to other causes as previously speculated. The South tributary that drains agriculturally rich land to the south also revealed larger SRP, TSS, and nitrates.









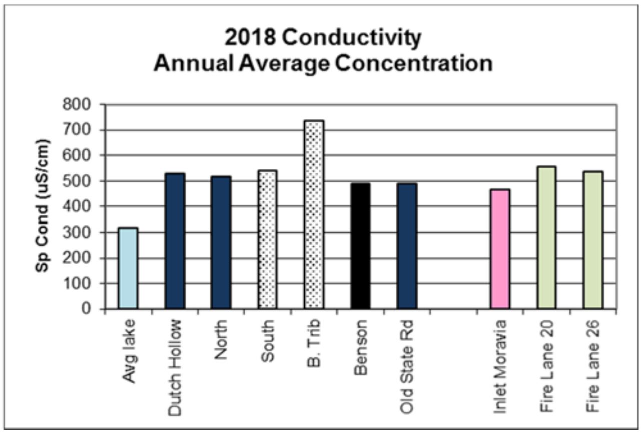
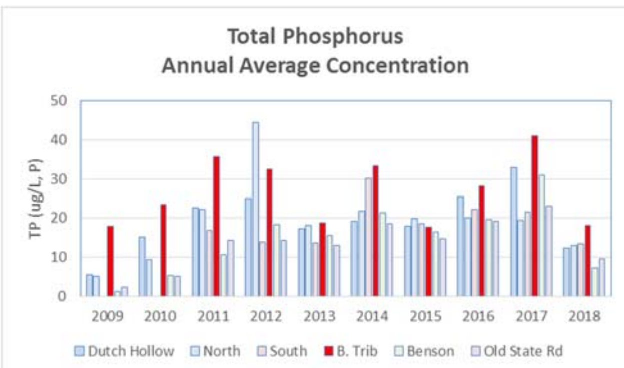
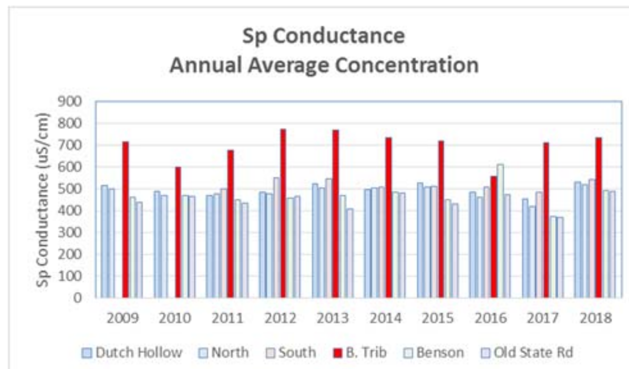


Fig. 22. Site averaged lake (light blue) and stream nutrient and suspended sediment concentrations for Dutch Hollow Brook (blue), Owasco Inlet (pink), and Fire Lane 20 & 26 (green). Tributary sites are stippled. Sites are arranged from downstream (left) to upstream (right).





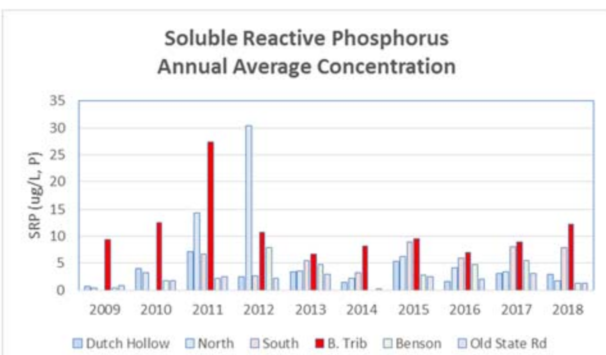


Fig. 23. Mean annual concentrations of total phosphorus, specific conductance and soluble reactive phosphorus at the routinely sampled Dutch Hollow Brook sites.

Nutrient and sediment concentrations in Owasco Inlet at Moravia were similar to those at Dutch Hollow Brook. Dissolved phosphate, total suspended sediment and specific conductance concentrations at the two Fire Lane sites were similar to the other stream sites. In contrast, significantly larger total phosphate concentrations were detected at Fire Lane 26, and significantly larger nitrate concentrations at both Fire Lane sites. These larger nutrient concentrations potentially reflect larger agricultural impacts at the headwaters of these tributaries. The timing of manure spreading and other agricultural inputs is not known for a more robust conclusion.

Stream Fluxes: Dutch Hollow Brook revealed smaller fluxes of nutrients and sediments than Owasco Inlet (TP 0.1 vs. 2.7 kg/day; SRP 0.1 vs. 0.6 kg/day; TSS 428 vs. 473 kg/day; N 12 vs. 115 kg/day, respectively, Fig. 24). Similar concentrations of nutrients and sediments between these two streams, but significantly larger discharges down the larger Owasco Inlet, resulted in its larger fluxes to the lake. As before, fluxes in the Owasco Lake watershed were sensitive to discharge and basin size. The annual mean fluxes measured in 2018 based on the five spring to early summer grab samples were similar to those detected in previous "dry" years (Fig. 25).

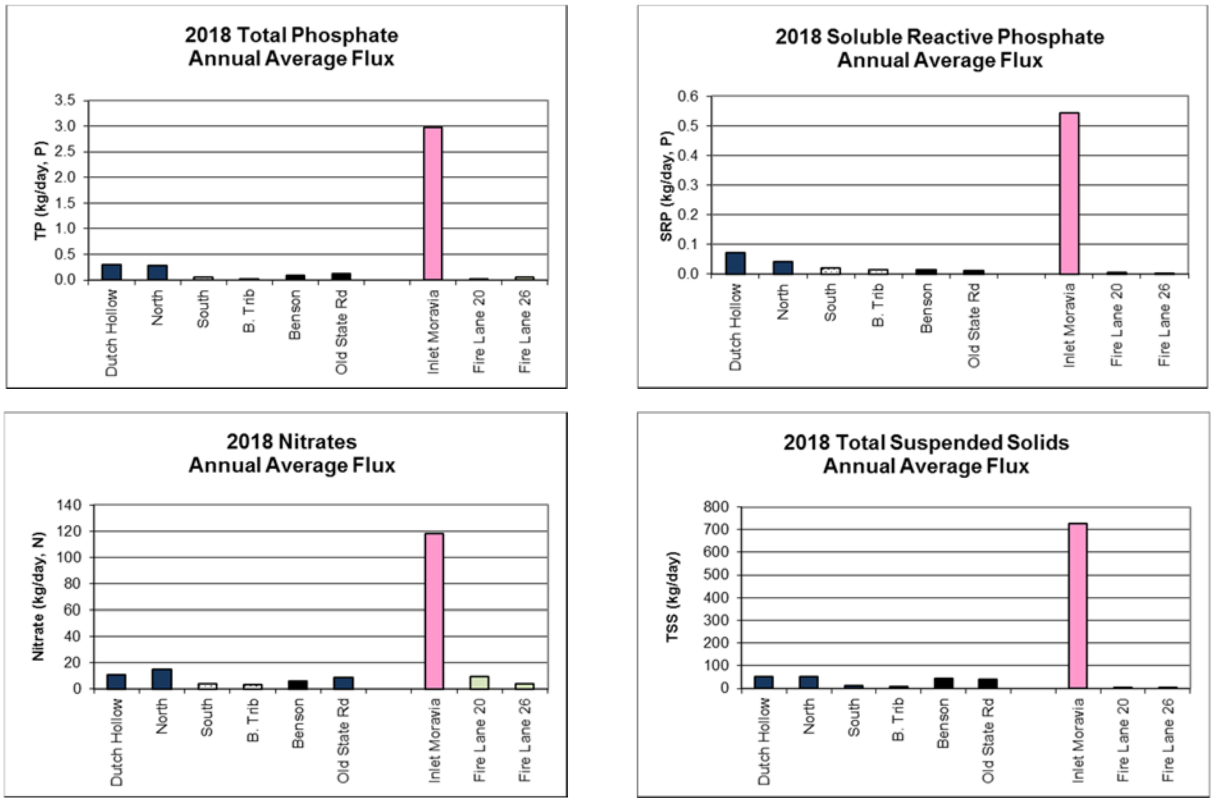


Fig. 24. Site averaged nutrient and sediment fluxes for Dutch Hollow (blue), Owasco Inlet (orange), and Fire Lane 20 & 26 (green). Tributary sites along Dutch Hollow Brook are stippled. Sites are arranged from downstream to upstream.

At the small end of the spectrum, fluxes at the Dutch Hollow Brook tributary sites (Benson and South sites) and the two Fire Lane sites were smaller than the other sites in the survey. These three small fluxes parallel the smaller discharges at these sites. It follows that smaller watersheds with smaller discharges delivered the smallest fluxes, and larger watersheds with larger discharges delivered the largest fluxes. However, many small, 1st or 2nd order, tributaries (~40 in Fig. 2) like Fire Lane 20 and 26 drain into Owasco Lake. The combined TP load by all these small tributaries, assuming they have similar concentrations as Fire Lane 20 & 26 is estimated to be similar to the load from Dutch Hollow Brook (see phosphorus loading section below for tally of loads by source).

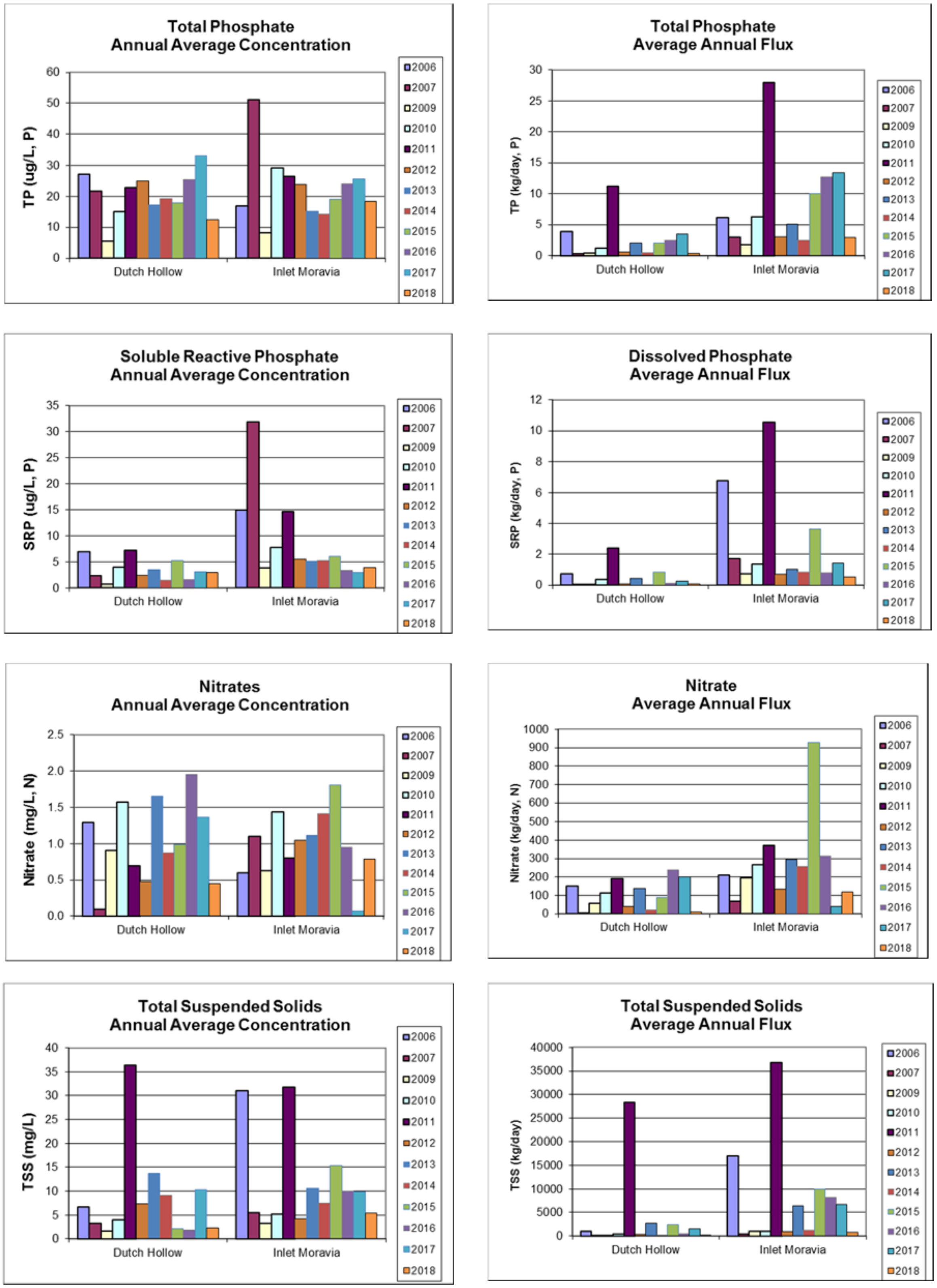


Fig. 25. 2018 Annual average stream grab sample concentrations (left) and fluxes (right).

As in previous years, no one tributary to Dutch Hollow Brook had a significantly larger flux of nutrients. Thus, no one segment of this stream was the "primary" source of nutrients and sediments. Instead, Dutch Hollow Brook steadily gained nutrients along its entire course up to North Rd, a conclusion consistent with the pervasive nature of nonpoint sources throughout the watershed, and the drainage of agricultural land, animal feedlot operations, road-side ditches, golf courses, suburban homes and other nonpoint sources. The implications are critical. To remediate Dutch Hollow Brook's nonpoint source nutrient loading problem is more challenging than remediating a point source like Groton's wastewater treatment facility, because nonpoint source remediation efforts must be applied throughout the entire watershed, demanding cooperation by every land owner in the watershed.

The Groton wastewater treatment facility contribution to the total phosphorus load along the Owasco Inlet has been significantly smaller since the DEC mandated P load reduction in 2007 than earlier years (Fig. 26). The load contributed by the Moravia WWTF was also very low. Both facilities averaged ~0.1 to 0.2 kg/day.

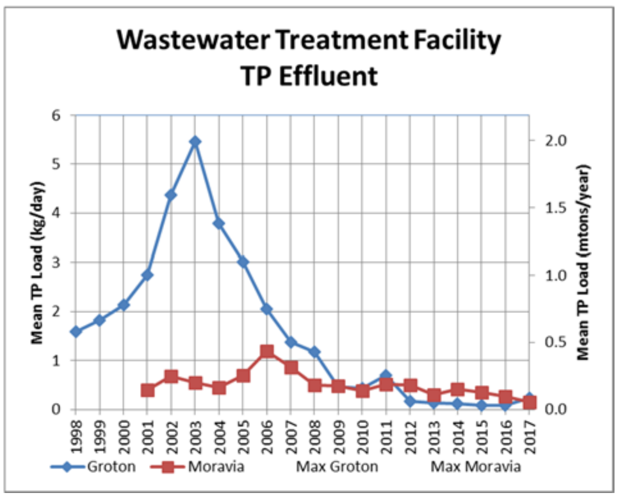


Fig. 26. Phosphorus loads from the Moravia and Groton wastewater treatment facilities.

Event Sampling Along Dutch Hollow Brook

Detailed Stage Data @ 38A along Dutch Hollow Brook: The 2018 mean stage data revealed textbook responses to precipitation events superimposed on the gradual spring through mid-summer decline in stage (discharge) as groundwater inputs decreased over this interval of time (Fig. 27). Each increase in stage corresponded to a precipitation event. Changes in stream height ranged from 5 to more than 100 cm above the adjacent base flow levels. Not all precipitation events induced a proportional stream response, especially during the spring when increases in stage were larger for similar sized precipitation events than the other seasons. The differences are interpreted to reflect seasonal changes in, for example, ground saturation, rainfall intensity, runoff/infiltration ratios and evapotranspiration. Similar seasonal and day to day precipitation/event influenced changes in stage and temperature were detected during the past five years as well (Fig. 28). Far fewer events were detected in 2018 than 2017, and 2018 was more similar to the earlier "dry" years, especially during the spring and early summer (Fig. 27). Events were more common in 2018 after mid-July.

Detailed "Event vs Base Flow" Results @ 38A: Nutrients and sediment concentrations increased markedly during precipitation events in 2018 (Fig. 29). Total suspended sediments (TSS) increased from an average base flow concentration of ~8 mg/L to an average event flow concentration of 70 mg/L, and rose to an event maximum of 315 mg/L on 10/28. The large TSS concentrations were restricted to storm events, and declined quickly to base flow turbidities, typically before the stream stage returned to base flow. It indicates that peak flow, runoff events compared to base flow transported significantly more soil particles to and had a greater impact on water quality in the stream. It highlights the importance of large events. The 2018 event concentrations were smaller in 2018 than previous years, especially the previous "wet" years.

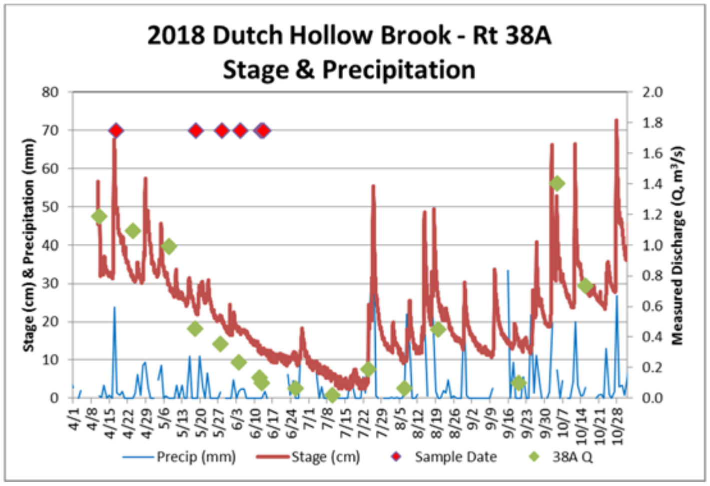


Fig. 27. Dutch Hollow Brook estimated discharge, precipitation, stream sample dates and measured discharge data for 2018 at Rt 38A. Precipitation data was from NY-CY-8, a CoCoRaHS station.

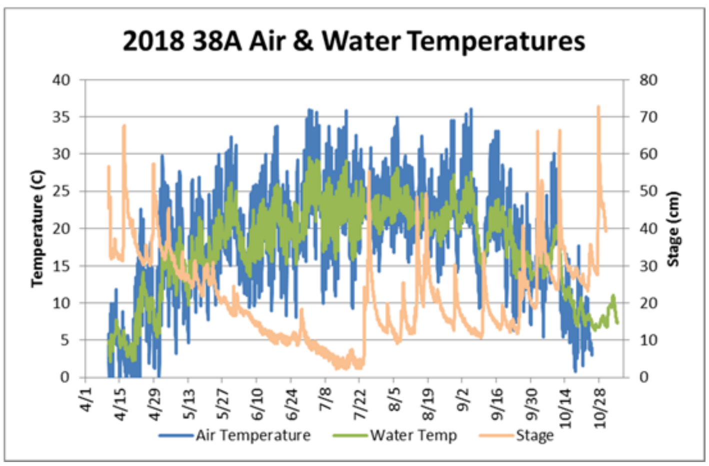


Fig. 28. Data logger mean stage, air and water temperature data at 38A.

Total (TP) and soluble reactive (SRP) phosphorus revealed event responses as well. Mean TP and SRP event concentrations were significantly larger than base flow concentrations, increasing from base flow means of 18 and 3 μg/L to event means of 43 and 13 μg/L, respectively. Maximum event concentrations were 75 μg/L for TP and 42 μg/L for SRP. Again, 2018 event concentrations suggest a direct linkage to and the importance of precipitation induced runoff events for phosphorus loading to the lake. Thus, the remediation steps to reduce phosphate loading are similar to remediating suspended sediment, i.e., reduce the movement of soil particles from the watershed to the lake. The literature indicate that drain tile are an important source of SRP as well. Tiles increase the release of dissolved and particulate phosphorus from the soils. Although not specifically measured in this watershed, drain tiles and the ditches that tiles drain into should be mapped, sampled and remediated.

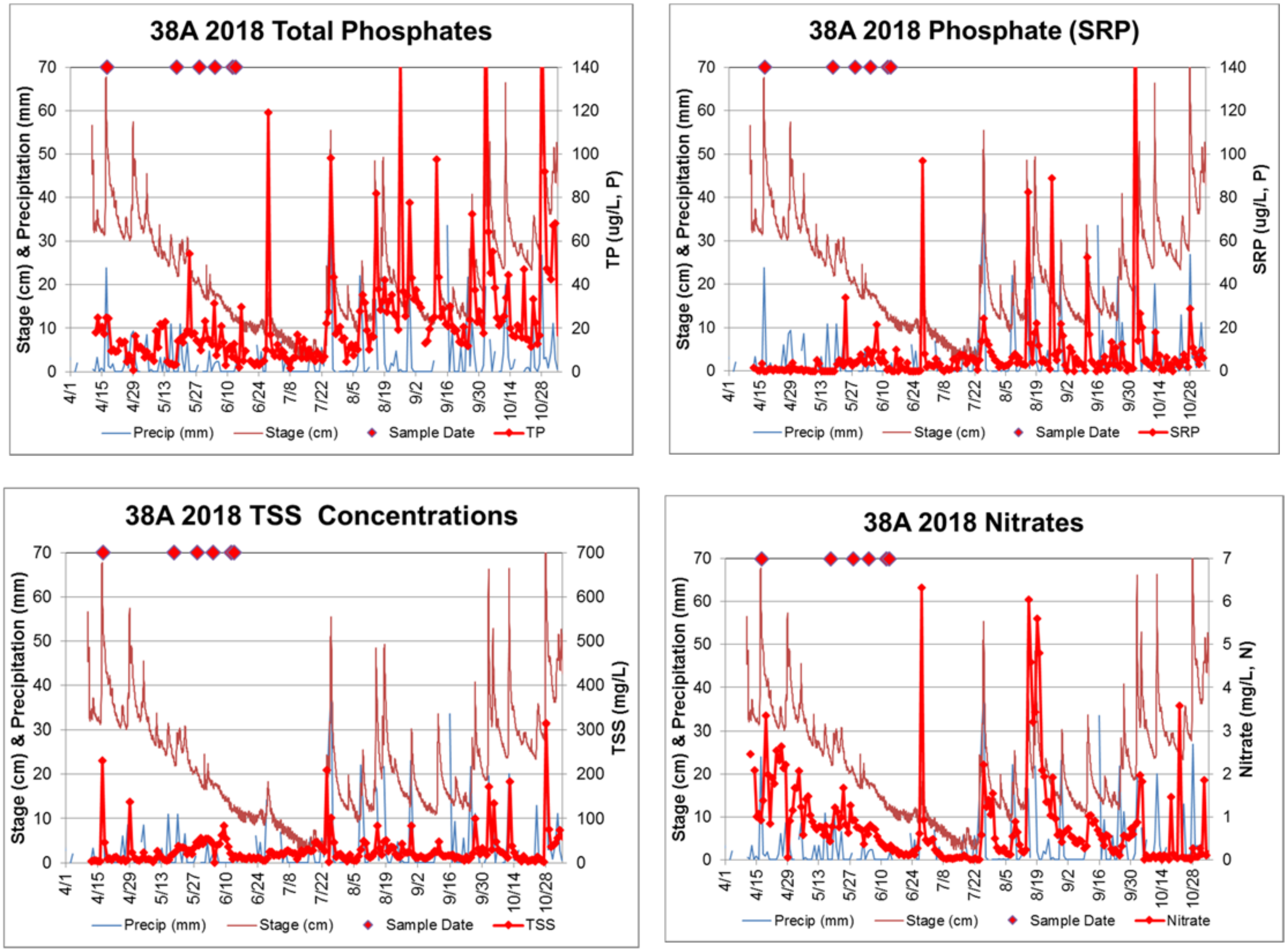


Fig. 29. 2018 daily nutrient and suspended sediment concentrations at Rt 38A.

The event *versus* base flow results suggests a number of potential remediation practices to reduce TSS impairments as mentioned in previous reports. For example, roadside ditches should be hydro-seeded and/or utilize other flow reducing structures to decrease water velocities and the erosion potential of the draining water. Slower water speeds allows for greater deposition of the particles with the attached phosphorus before it enters the stream. This is more critical if drain tile effluent enters a roadside ditch as drain tiles efficiently transport phosphorus from the fields to the ditch. In agricultural areas, buffer strips of vegetation should be established and maintained alongside each stream course, because the vegetated strips reduces the runoff velocity and allows particles with attached phosphorus to settle out before entering the stream. Installation of gully plugs, vegetation strips and retention ponds in low lying areas provide another mechanism to retard the movement of suspended sediments before the runoff spills into the nearby stream. Farmers should also plant a winter crop cover, as it reduces topsoil erosion from their fields during the late fall, winter and early spring seasons. This is most critical in the early spring when the soils are thawed and still saturated, conditions ripe for the largest erosion rates. All of these practices worked in the Conesus Lake watershed. These practices however remove tillable acreage from the farmer and/or require additional time on the fields to, e.g., plant winter cover crops, and thus reduce her/his annual income.

Nitrates, once again, revealed a slightly different response to events than TSS, TP and SRP. The largest nitrate concentrations still correlated with events. However, the maximum event concentration was 6.3 mg/L, only slight larger than the mean base flow concentration of 1.1 mg/L, and a much smaller difference than those observed in the TSS, TP and SRP data. The increase to the peak concentration and subsequent decline to base flow conditions took slightly longer for nitrates as well. It indicates that runoff provided extra nitrates to the stream. However, nitrates have a different event/base flow response than TSS, TP and SRP because nitrates are water soluble and not bound to particles, thus they can enter a stream by both runoff and groundwater routes. In contrast, phosphates are typically particle bound, thus groundwater does not transport TP, SRP and TSS. Precipitation events also rejuvenate near-surface groundwater flow, which contributed to the nitrate load as well, extending the nitrate response to the event as runoff flows faster than groundwater flow.

Event vs. Base Flow Fluxes @ 38A: To calculate daily fluxes from Dutch Hollow Brook, a discharge was determined for each stage using a best-fit, 2nd order, polynomial relationship between the data logger stage data and weekly to bi-monthly discharge measurements at 38A (r^2 = 0.99). It established a stage/discharge rating curve for the site (Fig. 30). A 2nd-order, polynomial fit provided a better match for the 2016, 2017 and 2018 stage/discharge data than a linear fit.

The TSS, TP, SRP and N fluxes were clearly event driven (Table 7, Fig. 31). In 2018, TSS, TP, SRP and N event *vs.* base flow fluxes at 38A averaged 6,950 *vs.* 160, 4.2 *vs.* 0.3, 1.3 *vs.* 0.1 kg/day and 110 *vs.* 21 kg/day, respectively. During the 2018 April through October deployment, Dutch Hollow provided 678,000 kg of sediment to the lake during events, but only 17,700 kg during base flow conditions. In a similar light, the 2018 events delivered 400 kg of TP, 124 kg of SRP and 10,450 kg of N to the lake compared to base flow contributions of 39 kg of TP, 7 kg of SRP and 2,300 kg of N. In conclusion, each year revealed significantly larger event than base flow loads for TSS, TP, SRP and N along Dutch Hollow Brook (Table 7).

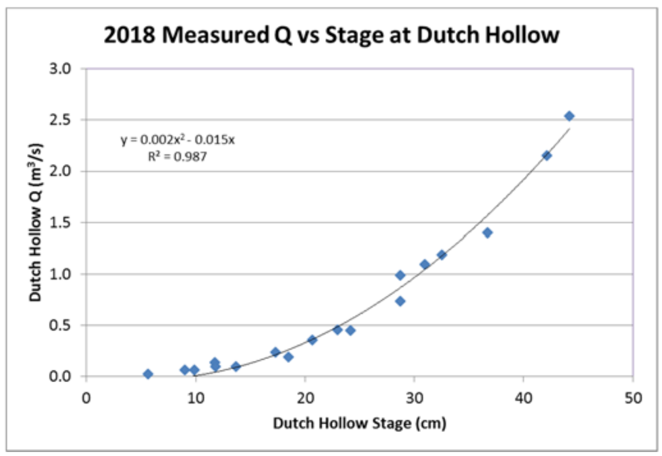


Fig. 30. Best fit correlations between weekly to bi-weekly measured discharge and data logger stage data at 38A site.

Annual changes were also observed. The 2018 mean annual fluxes for TSS were smaller than previous years, second only to 2012, a “dry” year. It reflected the lack of events in the spring and early summer, and 2017, a “wetter” year, probably washed out most of the easily erodible materials from the stream (Fig. 32). Similar small loads were noted in 2018 for TP, SRP, and nitrate where 2018 mean annual fluxes were consistently below the eight year average, and typically 2nd or 3rd smallest in the time series.

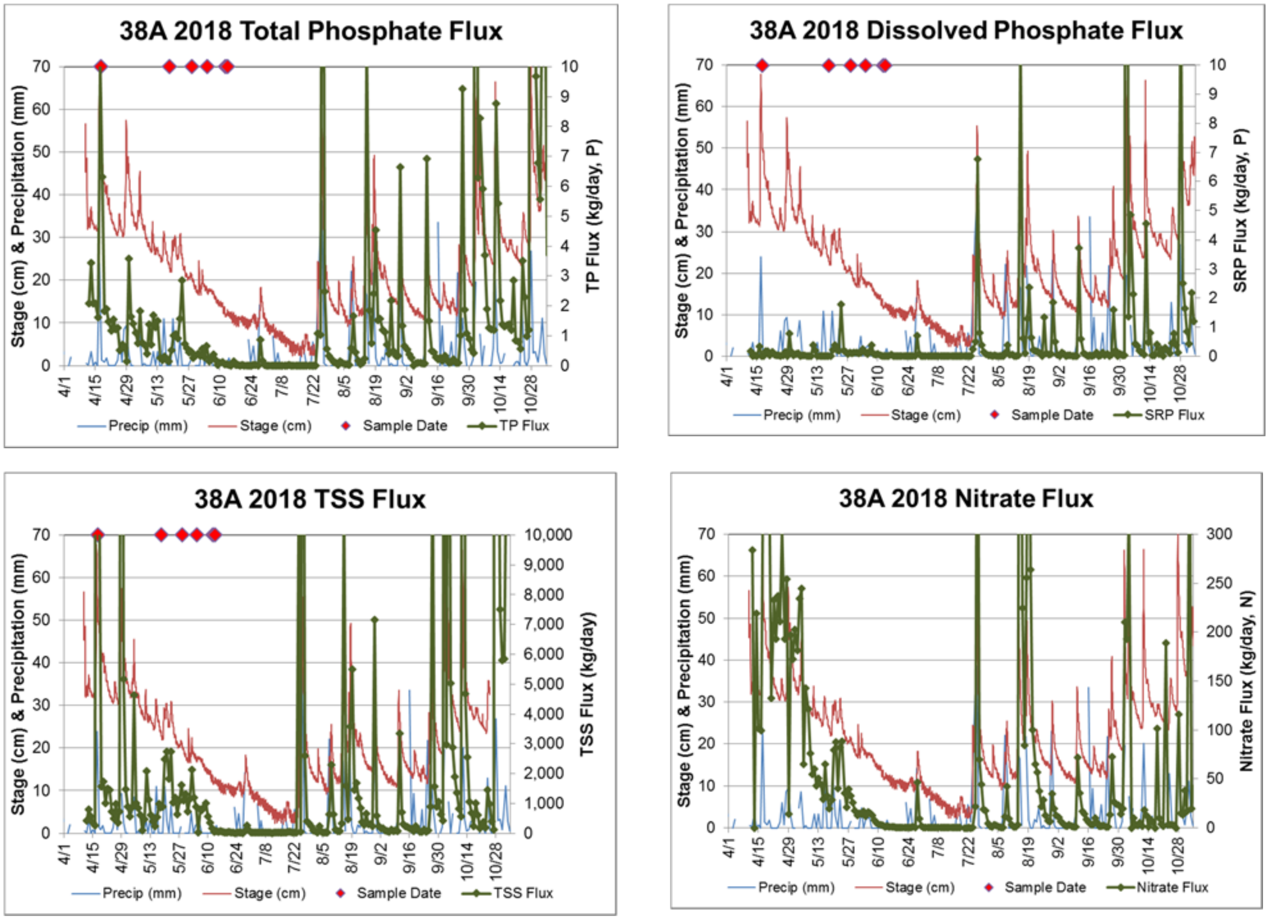


Fig. 31. Autosampler nutrient and suspended sediment fluxes.

These relationships were even more apparent when rainfall totals focused on May through June; a time frame when soils are saturated and thus more rain is directed to runoff than infiltration, but the soils are thawed enough to enable soil erosion (Fig. 33). Plant life is also absent or just rebounding from winter dormancy at this time, and thus not available to retard runoff velocities and reduce runoff volumes by evapotranspiration. Farm fields are also typically tilled bare of vegetation in preparation for spring planting of crops at this time, increasing their potential for erosion as well. In 2018, the watershed experienced minimal events during the spring and early summer when the fields are bare and soils saturated, and more typical events in the late summer and fall when the fields are planted and evapotranspiration are high.

The event *versus* base flow data also indicate that grab samples underestimated annual fluxes down a stream. For example, the 2018 autosampler estimated a mean sediment flux of 3,300 kg/day, total phosphates 2.1 kg/day, dissolved phosphates 0.6 kg/day, and nitrates 62 kg/day; whereas the grab sampling estimated an annual mean flux of 430 kg/day for sediments, 0.1 kg/day for total phosphates, 0.1 kg/day for dissolved phosphates, and 12 kg/day for nitrates. The grab samples estimates were smaller because the site visits were during May and early June, the dry period of 2018, and these samples were usually biased to base flows. Grab samples are therefore less accurate for detailed flux estimates compared to the daily data collected by the autosampler and data loggers. However, grab samples are essential for stream segment analysis and the investigation of nutrient and sediment sources from within a watershed.

Table 7: 2011 – 2018 Autosampler Fluxes at RT 38A Dutch Hollow Brook.

2011 (6/9-11/4)	TSS	Nitrate	TP	SRP
Mean (kg/day)	8,700	75	2.7	1.7
Event (kg/day)	24,500	180	6.9	4.5
Base Flow (kg/day)	115	19	0.4	0.1
% by events	99%	84%	90%	96%
2012 (3/20-11/2)	**TSS**	**Nitrate**	**TP**	**SRP**
Mean (kg/day)	2,400	69	1.9	0.4
Event (kg/day)	6,850	150	4.0	0.6
Base Flow (kg/day)	190	28	0.9	0.2
% by events	95%	73%	70%	60%
2013 (4/10-10/29)	**TSS**	**Nitrate**	**TP**	**SRP**
Mean (kg/day)	7,550	270	4.4	1.3
Event (kg/day)	12,000	370	6.4	1.8
Base Flow (kg/day)	290	100	1.3	0.3
% by events	99%	85%	89%	91%
2014 (4/19-10/28)	**TSS**	**Nitrate**	**TP**	**SRP**
Mean (kg/day)	14,600	115	3.5	1.6
Event (kg/day)	36,000	185	6.5	3.2
Base Flow (kg/day)	300	67	1.5	0.5
% by events	99%	65%	74%	81%
2015 (4/19-10/28)	**TSS**	**Nitrate**	**TP**	**SRP**
Mean (kg/day)	35,600	180	3.7	2.3
Event (kg/day)	81,500	370	7.7	5.2
Base Flow (kg/day)	185	27	0.5	0.0
% by events	99%	93%	94%	99%
2016 (4/13-10/25)	**TSS**	**Nitrate**	**TP**	**SRP**
Mean (kg/day)	7,482	1,323	1.4	0.7
Event (kg/day)	25,844	4,602	4.7	2.3
Base Flow (kg/day)	137	11	0.1	0.0
% by events	99%	99%	97%	99%
2017 (4/25-11/25)	**TSS**	**Nitrate**	**TP**	**SRP**
Mean (kg/day)	14,770	84	2.2	1.1
Event (kg/day)	29,195	167	4.2	2.1
Base Flow (kg/day)	176	9	0.3	0.1
% by events	99%	94%	92%	96%
2018 (4/12-11/4)	**TSS**	**Nitrate**	**TP**	**SRP**
Mean (kg/day)	3,277	62	2.1	0.6
Event (kg/day)	6,953	110	4.2	1.3
Base Flow (kg/day)	158	21	0.3	0.1
% by events	97%	82%	91%	95%

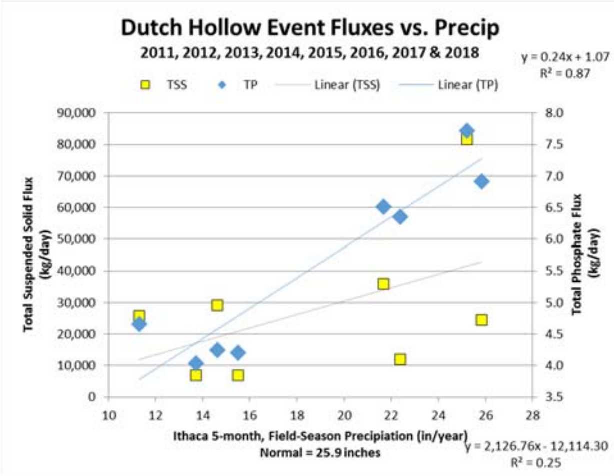


Fig. 32. Estimated annual total phosphorus loads vs field season (5-month) rainfall at the Ithaca Airport.

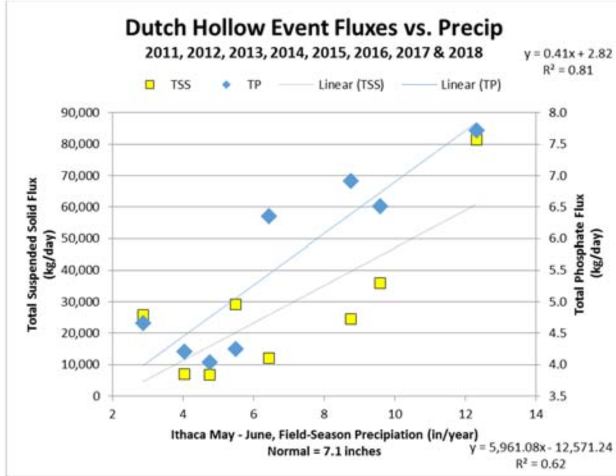


Fig. 33. Estimated annual total phosphorus loads vs May-June spring rainfall at the Ithaca Airport.

PHOSPHORUS BUDGET:

Phosphorus loads are critical to the health and water quality of Owasco Lake because phosphorus limits algal growth and thus impairs water quality and clarity. The recent development of blue-green algae blooms, some with life threatening concentrations of toxins, also highlight its importance. Clearly, stream loads dominate the inputs, even in "dry" years. However, the stream inputs are only one part of the equation. A complete budget must also include other inputs like atmospheric loading, onsite septic systems and lakeshore lawns. Outputs must also be calculated to estimate the net change in phosphorus for the lake (Fig. 34). The net change is critical because the amount of phosphorus will increase in the lake, if inputs exceed outputs. Phosphorus will decrease in the lake, if inputs are less than outputs. Finally, phosphorus remains the same, i.e., at equilibrium, when inputs equal outputs. To improve water quality, the inputs of phosphorus must be smaller than outputs for a number of years (multiple water retention times). A sustained reduction allows phosphorus in the lake to leave by the outlet or be buried in the sediments, and increasingly limit algal growth and improve water quality and clarity. The required "cleansing" time frame in the Owasco watershed is a decade or more.

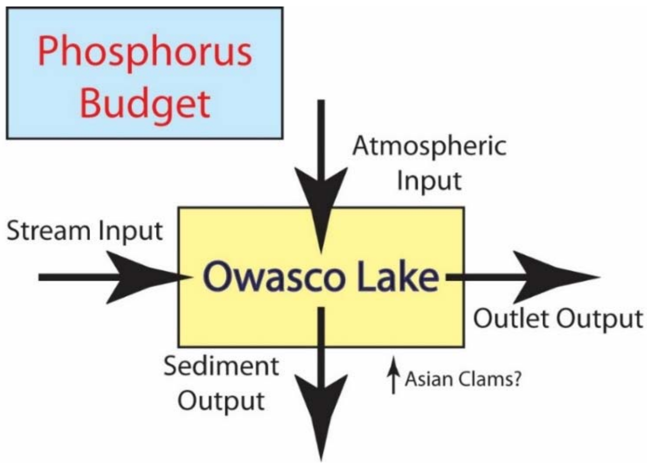


Fig. 34. The Owasco Lake phosphorus budget: Sources and sinks for phosphorus in Owasco Lake. Water quality improves if **inputs are less than outputs**, for a number of years.

The Inputs: The Moravia and Groton WWTF added a combined 0.2 metric tons P to the Owasco Inlet in 2018[8]. The detailed 38A autosampler data calculated a mean total phosphate flux of 2.1 kg/day from Dutch Hollow Brook in 2018. Owasco Inlet delivered 3.0 kg/day based on the available 2018 stream grab data. The 2018 load from Owasco Inlet was estimated at 8.1 kg/day assuming a proportional change between the mean grab sample total phosphorus loads to the detailed autosampler loads from Dutch Hollow Brook. An extrapolation of fluxes and surface areas from Dutch Hollow Brook and Owasco Inlet to the entire Owasco watershed, estimated an annual input of 4.9 metric tons of phosphorus from every stream to the lake in 2018. The stream extrapolation incorporates all the 1st and 2nd order (small) tributaries like Fire Lane 20 and 26. The 2011 report estimated atmospheric and septic system inputs at 0.1 metric tons/year and ~1 metric ton/year, respectively. Loads from water fowl were estimated at 0.1 mton/year assuming 1,000 geese poop on average 3 times/day, yielding 1.5 g dry poop/dropping at 1% phosphorus content[9], and live on Owasco Lake for the entire year. This load was deemed insignificant and ignored as the estimate exaggerated geese numbers and length of stay. The contribution from clams/mussels (Asian clams and zebra/quagga mussels) and decaying macrophytes is unclear at this time as mussel and plant lake-floor densities are not well known. The extent mussels redistribute nutrients from the offshore to nearshore locations is also unclear. These should be investigated in the near future as nutrient inputs from lake-floor sediments are speculated to be a significant source for nearshore BGA blooms (see companion reports). Macrophytes release P taken up from the sediments whereas zebra/quagga mussels redirect P from the open water algae to the nearshore lake floor.

The total 2018 influx of phosphorus is estimated at 6.1 metric tons/year.

The Outputs: Phosphorus is lost from the lake through the Outlet in the form of algae, dissolved organic-rich compounds, organic-rich particulates, and the occasional larger organism (e.g., fish). Approximately 3.6 metric tons of phosphorus escaped out the Outlet in 2018 assuming a 2018 annual mean total phosphate concentration in the lake of 13.6 μg/L, and a 2018 mean daily discharge of 8.3 m^3/s through the Owasco Outlet (USGS Owasco Outlet Gauge #04235440). The 2011 report estimated the flux of phosphorus to the sediments of a few metric tons per year and this estimate is again used here. The earlier report cautioned that more work was required to firm up this sediment burial estimate, because the flux was based on only a few sediment cores.

The total 2018 efflux of phosphorus is estimated at 6.4 metric tons/year.

The Net Flux: Owasco Lake thus lost approximately 0.3 metric tons of phosphorus in 2018. Since 2011, the lake gained phosphorus during five years and gained almost or slightly more phosphorus as it lost in the past three years (Fig. 35). Since 2011, the mean annual input was 8.2 mtons/year, 1.6 metric tons/year more than the mean output of 6.6 mtons/year. Earlier estimates of annual nutrient fluxes were based on limited summer-season grab samples thus not included in this analysis. The pervasive positive net fluxes and fewer negative net fluxes indicates that significant remediation efforts must take place to move Owasco Lake to a negative balance and eventually improve water quality. The decrease in loads and near equilibrium conditions in 2016, 2017 and again in 2018 is encouraging and perhaps Owasco Lake is responding to the

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[8] http://cfpub.epa.gov/dmr/facility_search.cfm Groton: NY0025585, Moravia: NY0022756.
[9] Fleming and Fraser, 2001. The impact of water fowl on water quality – a literature review. Ridgetown College, U of Guelph, Ontario, Canada.

various remediation efforts in the watershed, but inputs must be much less than outputs to improve water quality.

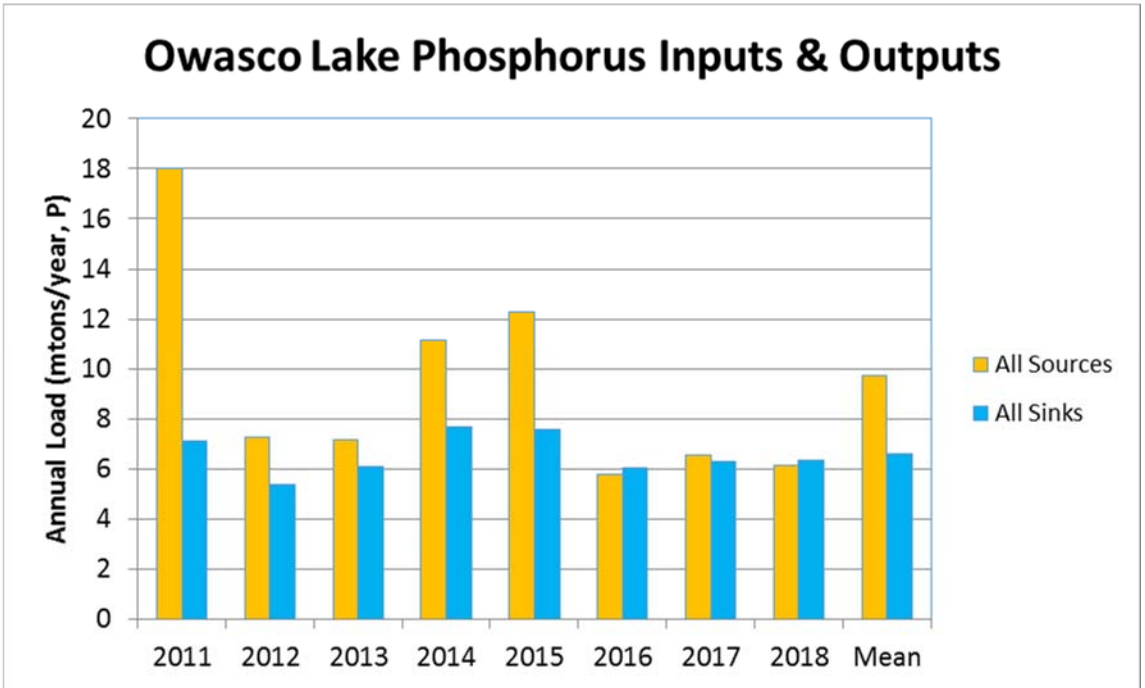

Fig. 35. Estimated annual total phosphorus inputs and outputs for Owasco Lake.

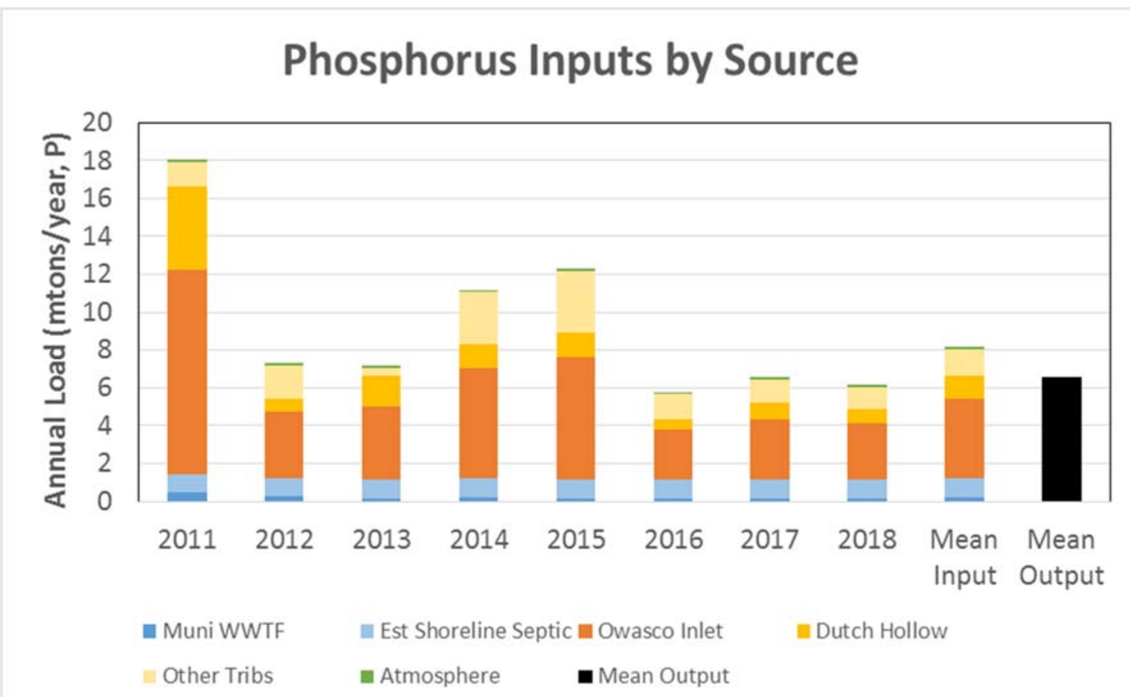

Fig. 36. Estimated annual inputs of phosphorus by source for Owasco Lake.

The contribution from stream sources changed from year to year, whereas the inputs from other sources were relatively constant over the past seven years (Fig. 36). For example the annual contribution from the Owasco Inlet, excluding the wastewater treatment facilities (WWTFs), ranged from 0.5 to 10.0 metric tons since 2011. The variability reflects changes in precipitation intensity, seasonality and totals. In contrast, the WWTFs contributed from 0.1 to 0.4 metric tons/year over the same time interval. Despite the variability in stream inputs, streams are always the largest contributors of phosphorus to the lake, ranging from 78 to 91% of the total load, and averaged 85% of the load. The stream dominance clearly pinpoints where additional remediation efforts should be focused to reduce phosphorus loads to the lake. Remediation therefore should include reduction in inputs from non-point sources like agricultural areas, both animal farms (manure spreading and barnyard runoff) and crop farms (drain tile effluent), road-side ditches and construction sites.

Finally, the large nutrient and sediment inputs, especially in 2011, 2014, and 2015 were coincident with and probably "triggered" the recent BGA blooms[10]. Even though coincidence does not prove causation, these four years of excessive loads were unique over the past decade. Assuming these large loads were the trigger, everyone must significantly reduce nutrient loading in the watershed. Otherwise the lake will continue to degrade and experience more BGA blooms into the future.

CONCLUSIONS & RECOMMENDATIONS:

This report confirms and expands on earlier findings.

Owasco Lake Water Quality:

- Owasco Lake is a borderline oligotrophic – mesotrophic lake. The improvements in water quality from 2011 through 2013 were lost in 2014 and 2015. Water quality improved in 2016 and 2018 with a reversal in 2017.
- Phosphorus is the limiting nutrient for algae growth in Owasco Lake.
- The water quality degradation in 2014, 2015 and 2017 is attributed to the heavy rains and/or intense precipitation events in those years, especially in the spring.
- The below normal rainfall in the spring and early summer and above normal rainfall in the fall made 2018 an unusual year. Loads were low in the early part of the field season but increased in the fall.
- The water quality buoy provided a more robust view of water quality in the lake by detecting algal blooms and other events missed by the monthly lake surveys over the past five years.

Blue-Green Algae Blooms:

- Blue-green algae (BGA) concentrations were small at the open-water sites, typically much smaller (up to 10,000 x's smaller) than the shoreline blooms.
- The annual mean BGA concentrations in the shoreline blooms steadily rose from 2014 through 2017 but leveled off in 2018. Many of the blooms contained toxins, both microcystins and anatoxins over the past five years.
- Bubblers, ultrasonic vibrators, benthic mats and other in-lake strategies might provide stop-gap measures to reduce the extent of the BGA blooms until the nutrient loading issue is

[10]Halfman, J.D., 2017. Water quality of the eight eastern Finger Lakes, New York: 2005 – 2016. Finger Lakes Institute, Hobart and William Smith Colleges. 51 pg.
Halfman, J.D., 2017. Decade-scale water quality variability in the eastern Finger Lakes, New York. Clear Waters. Fall 2017, v. 47, No. 3, pg. 20-32. http://nywea.org/clearwaters/uploads/Decade-ScaleWater7.pdf

resolved. Unfortunately, a test of bubbler and ultrasonic technologies in 2018 were inconclusive because blooms did not frequent the test sites during the survey dates.

- NEVER use herbicides in Owasco Lake because it is a drinking water source for over 44,000 people, and herbicides are toxic to humans.
- Lakeshore owners that draw water from the lake need affordable mechanisms (or financial support) to reduce their risks from the BGA toxins.
- More details on BGA blooms and the associated HABs are contained in a companion report that focuses on the water quality analysis of a number of nearshore sites.

Stream Loads & Watershed Phosphorus Budget:

- Daily discharge data for Owasco Inlet and Dutch Hollow Brook revealed the worst events in 2011, 2014, 2015 and 2017. The 2018 events were small in comparison and skewed to the latter portion of the field season but increased once it started to rain.
- The excessive nutrient loads during 2012, 2014 and 2015 were coincident with and perhaps triggered the onset of the blue-green algae in Owasco and many other Finger Lakes. Once these loads triggered the initial blooms, BGA have typically returned in larger numbers to the same nearshore locations. Perhaps the dry spring and early summer in 2018 decreased nutrient loads thus the number of BGA blooms compared to earlier years. Windier and cloudier conditions may have hampered the 2018 blooms as well.
- Segment analysis lacked significant point source signatures along Dutch Hollow Brook.
- Both the Moravia and Groton municipal wastewater treatment facilities have done an amazing job keeping their phosphorus loads to a minimum.
- The event *versus* base flow analysis at Dutch Hollow Brook highlights the dominance of events and associated runoff of nonpoint sources for the delivery of phosphorus to the lake. It also provided more accurate load estimates than grab samples, especially in those years when surveys were limited to base-flow conditions during the summer months.
- The estimated phosphorus budget for Owasco Lake indicated that the lake neither gained of lost significant amounts of phosphorus in 2018.
- Partitioning the total loads by major sources, i.e., Owasco Inlet, Dutch Hollow Brook and other streams, lakeshore onsite septic systems, municipal wastewater treatment facilities, and the atmosphere, confirms that streams are the primary source of nutrients and sediments to the lake over the decade long monitoring effort. Owasco Inlet was always the largest fluvial contributor and Dutch Hollow Brook was not too far behind. The stream inputs however vary from year to year, dependent on the amount, intensity and seasonality of rainfall. Contributions from geese and other large water fowl are insignificant.
- More research is required to assess the impact of zebra/quagga mussels and macrophytes on the nutrient budget and internal nutrient redistributions in the lake.

Remediation Strategies:

The ongoing nutrient loads and their negative impact on the lake's water quality commands the immediate use of additional remediation strategies. The sooner they are installed the sooner the lake might return to its original oligotrophic state.

- o More BMPs should be installed, where necessary, to reduce nutrient and sediment loading from agriculturally-rich watersheds. The critical areas for BMPs are along stream banks and in the low lying and other water saturated areas in each field. The BMPs include buffer strips, gully plugs, vegetation strips, barnyard cleanup, and other means to slow down and stop the runoff of nutrients and sediments.

- Roadside ditches, especially those that accept drain tile effluents, should be hydro-seeded, have catch basins installed and employ other strategies to retain the nutrient and sediment load on land before the runoff enters the lake. The ditches and catch basins will require periodic cleaning to be effective.
- The time is also ripe to strengthen existing regulations and uniformly adapt these new/strengthened regulations across the entire watershed. The areas of focus should include: onsite septic systems, disposal of animal wastes and deer (and other) carcasses, drain tiles, phosphorus-free fertilizers, disposal of grass-clipping and leaf litter, and other areas of concern to reduce the nutrient and sediment loading to the lake. Even though many of these issues are already regulated, the current regulations are clearly **NOT** stringent enough because Owasco Lake currently has a nutrient loading problem.
- Perhaps all human and animal wastes in the watershed should be treated at a municipal wastewater treatment facility instead of the current practices. This option would be expensive though.
- Additional flood basins should be built near the terminus of Owasco Inlet and initiated in the Dutch Hollow Brook watershed. Floating wetlands should be anchored just offshore of tributary mouths, as the vegetation would utilize some of the nutrients and thus reduce nutrient loads to the lake. Do not let the vegetation die and decompose in the water and thus release these sequestered nutrients back into the lake.
- Nutrients should also be removed from the lake, when feasible. For example, macrophytes should be harvested from the nearshore areas in the late summer and disposal outside the watershed. Those macrophytes and attached algae that wash up on the shoreline should be removed before they decompose along the lakeshore. The BGA blooms themselves should be vacuumed before they disappear.
- Owasco Lake is probably too large and the existing phosphorus concentration too small for phosphorus sequestration techniques like using binders like Alum and Phoslock (bentonite clay) that remove available phosphorus from the water column and bury it into the sediments. However phosphorus binding materials should be used in road-side ditches and at the opening of drain tiles to reduce phosphorus loads from this source.
- The sports fisheries is too vital for the local economy to use bio-manipulation.
- Finally, the financial burden to install the remediation efforts cannot be placed solely on farmers, lakeshore landowners or other individual groups. Water quality is a watershed-wide issue. Everyone benefits from a cleaner lake. Thus, everyone must support the remediation efforts. Yeah for Governor Cuomo’s financial support!

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Table 2. 2018 Lake Data.

2018 Owasco Lake Site Averaged and Date Averaged Data

Site Averaged Surface Water Data

Site	Secchi Depth	Suspended Solids	Total Phosphate	Dissolved Phosphate	Nitrate	Silica	Chlorophyll
	(m)	(TSS, mg/L)	(TP, ug/L)	(SRP, ug/L)	(N, mg/L)	(Si, ug/L)	(a, ug/L)
1	3.8	2.1	14.3	0.5	0.6	753	3.0
Buoy	3.7	---	---	---	---	---	---
2	3.9	1.7	11.6	0.4	0.7	801	3.1
A	3.8	1.9	15.1	0.4	0.8	843	1.8
B	2.0	1.7	15.7	0.2	0.6	745	2.9
C	--	1.1	12.3	0.6	0.6	733	2.1
D	4.2	1.4	11.6	0.8	0.7	802	2.4
E	3.4	2.2	14.4	0.5	0.6	758	2.9
F	--	1.2	13.9	0.4	0.5	730	1.7
Average	**3.5**	**1.7**	**13.6**	**0.5**	**0.6**	**771**	**2.5**

Site Averaged Bottom Water Data

Site	Secchi Depth	Suspended Solids	Total Phosphate	Dissolved Phosphate	Nitrate	Silica	Chlorophyll
	(m)	(TSS, mg/L)	(TP, ug/L)	(SRP, ug/L)	(N, mg/L)	(Si, ug/L)	(a, ug/L)
1	---	1.7	13.8	2.8	0.1	988	0.2
Buoy	---	---	---	---	---	---	---
2	---	1.4	15.2	8.5	0.7	683	0.8
Average	**---**	**1.6**	**14.5**	**5.7**	**0.4**	**836**	**0.5**

Date Averaged Surface Water Data

Date	Secchi Depth	Suspended Solids	Total Phosphate	Dissolved Phosphate	Nitrate	Silica	Chlorophyll
	(m)	(TSS, mg/L)	(TP, ug/L)	(SRP, ug/L)	(N, mg/L)	(Si, ug/L)	(a, ug/L)
5/22/18	5.3	0.8	12.0	0.3	0.8	981	0.2
6/5/18	5.3	1.8	20.7	0.0	0.6	968	0.7
6/19/18	2.8	1.9	11.6	0.0	0.6	1004	3.7
7/3/18	6.1	1.4	13.1	0.1	0.8	981	0.0
7/10/18	5.2	0.8	8.6	0.0	0.8	949	0.2
7/24/18	3.8	3.1	12.6	0.0	0.6	796	4.1
7/31/18	2.3	2.9	12.6	0.7	0.8	823	2.8
8/7/18	2.8	1.6	8.0	2.4	0.4	712	3.1
8/14/18	2.1	2.4	6.9	0.0	0.5	741	3.8
8/21/18	3.0	1.7	12.7	0.6	0.6	733	2.4
8/28/18	4.3	1.5	10.4	0.0	0.8	796	1.4
9/4/18	4.2	2.4	30.1	1.3	0.5	757	6.6
9/11/18	3.9	1.8	21.0	0.2	0.5	647	5.1
9/18/18	3.1	2.2	10.5	0.1	0.7	434	5.6
10/3/18	4.3	2.0	11.1	0.0	0.6	523	4.0
Average	**3.9**	**1.9**	**13.5**	**0.4**	**0.6**	**790**	**2.9**

Date Averaged Bottom Water Data

Date	Secchi Depth	Suspended Solids	Total Phosphate	Dissolved Phosphate	Nitrate	Silica	Chlorophyll
	(m)	(TSS, mg/L)	(TP, ug/L)	(SRP, ug/L)	(N, mg/L)	(Si, ug/L)	(a, ug/L)
5/22/18	---	0.9	8.9	1.1	0.6	1103	0.1
6/5/18	---	0.7	9.7	0.2	0.7	1123	0.1
6/19/18	---	0.5	8.1	0.0	0.8	1375	0.2
7/3/18	---	0.8	11.0	0.2	0.8	1414	0.0
7/10/18	---	0.8	6.2	0.0	1.2	1202	0.1
7/24/18	---	1.1	8.3	0.0	0.9	1101	0.0
7/31/18	---	0.8	8.8	3.8	1.0	1626	0.3
8/7/18	---	0.8	6.9	5.2	0.6	1480	0.4
8/14/18	---	1.1	12.7	2.2	0.8	1526	3.1
8/21/18	---	0.8	6.4	0.5	0.9	1520	0.3
8/28/18	---	1.3	6.4	0.0	1.3	1553	0.3
9/4/18	---	0.3	19.7	0.0	1.0	1500	0.4
9/11/18	---	0.6	21.7	0.0	0.9	1581	0.0
9/18/18	---	0.6	18.5	0.0	1.5	1446	0.4
10/3/18	---	1.1	6.2	1.1	0.9	1917.6	0.1
Average	**---**	**0.8**	**10.6**	**1.0**	**0.9**	**1431.1**	**0.4**

Table 4. Annual Average Plankton Data from 2005 through 2018, and Daily Average Data for 2018.

Plankton Group	Diatoms							Dinoflagellates			Rotifers & Zooplankton						Blue Greens	
Plankton Species	Fragillaria %	Tabellaria %	Diatoma %	Asterionella %	Melosira %	Synedra %	Rhizoselenia %	Dinobryon %	Ceratium %	Coalcium %	Copepod %	Nauplius %	Keratella %	Polyarthra %	Vorticella %	Cladoceran %	Dolichospermum (Anabaena) %	Mycrocystis %
2005 Average	34.9	1.4	0.0	9.9	0.2	5.6		14.6	4.5		0.9	1.1	2.5	3.2	10.3	2.8		0.3
2006 Average	24.3	1.7	0.0	7.1	1.4	0.7	2.6	41.5	0.7		0.2	0.1	2.4	0.8	0.3	0.6	0.1	3.8
2007 Average	30.0	0.5	0.0	23.3	0.2	2.1	3.8	12.9	0.7		0.4	0.4	0.6	0.4	3.8	2.8	0.4	7.7
2008 Average	52.3	0.1	0.0	14.6	0.2	0.1	1.2	18.7	0.6	0.2	0.4	0.5	0.3	0.9	4.3	0.6	0.4	1.5
2009 Average	9.7	7.1	0.0	12.3	0.2	1.0	7.8	26.6	0.7	2.0	0.7	0.6	3.6	0.7	4.3	2.1	3.4	4.8
2010 Average	36.8	0.5	0.0	19.1	0.2	1.4	0.7	4.6	0.0	2.6	0.6	0.8	3.3	0.7	3.2	5.6	0.1	6.1
2011 Average	26.0	14.1	0.0	15.0	0.4	1.4	15.0	5.3	0.5	1.8	0.9	0.7	2.8	1.0	3.9	2.0	0.2	2.6
2012 Average	27.0	25.5	0.0	10.9	13.0	2.2	1.1	8.1	0.3	0.2	0.5	0.5	0.3	1.5	0.9	0.6	0.3	0.8
2013 Average	27.6	0.3	26.9	3.9	3.8	0.0	5.9	0.0	0.1	2.1	0.5	0.9	1.3	2.4	1.2	4.1	0.3	0.6
2014 Average	21.8	0.3	5.8	15.2	0.2	1.5	2.5	20.2	0.1	0.0	2.7	2.7	1.1	6.4	1.8	1.1	0.1	2.6
2015 Average	28.6	7.5	1.0	20.2	0.3	0.8	3.9	3.7	0.1	0.1	0.7	0.9	1.8	3.5	0.8	3.1	0.1	7.3
2016 Average	11.5	2.8	6.7	13.7	1.2	0.3	0.2	11.7	0.0	0.1	0.7	1.0	4.0	5.0	1.7	1.5	2.1	5.3
2017 Average	11.8	0.1	0.1	6.4	1.5	0.0	11.1	5.4	0.3	0.5	0.6	1.5	2.0	2.7	4.0	2.3	4.9	5.6
5/22/18	2.2	1.5	9.5	84.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6/5/18	0.0	0.0	10.6	77.2	0.0	1.1	0.2	3.8	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
6/19/18	2.0	0.0	37.9	47.1	0.0	3.1	0.0	0.4	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0
7/3/18	0.0	0.0	33.9	49.6	0.0	2.1	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.3	0.0	0.0	0.0	0.0
7/10/18	0.0	0.0	0.7	74.1	0.0	2.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.2	0.0	0.0	0.0	0.0
7/24/18	0.3	0.0	0.0	0.0	0.0	87.8	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
7/31/18	0.4	0.0	0.0	0.0	0.0	90.0	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
8/7/18	0.0	0.0	0.0	0.0	0.0	89.7	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.7	0.4	2.7	0.0
8/14/18	0.0	0.0	0.0	0.3	0.0	79.0	0.0	0.0	0.0	0.0	0.0	0.0	1.1	0.5	2.1	0.8	5.5	0.0
8/21/18	0.4	0.0	0.0	0.6	0.2	72.6	0.0	0.0	0.0	0.0	0.7	0.0	3.0	0.4	0.2	0.9	1.2	1.2
8/28/18	3.1	0.0	0.7	2.6	0.0	12.7	1.2	0.0	0.0	1.2	0.0	0.0	22.7	1.9	2.4	0.2	4.8	10.1
9/4/18	0.8	0.0	0.0	0.8	0.0	17.0	0.0	0.5	0.0	0.3	0.3	0.0	0.8	1.0	0.0	0.0	1.3	5.5
9/11/18	2.7	0.0	2.4	5.1	0.0	6.9	0.0	0.3	0.0	0.0	1.1	0.0	0.0	0.8	0.0	0.3	0.3	2.4
9/18/18	4.6	0.0	0.2	3.3	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0	0.2	0.7	1.8
9/26/18	45.1	0.0	0.0	48.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.5	0.0	0.7	0.0	1.8
2018 Average	1.2	0.1	6.9	24.7	0.0	33.1	0.1	0.5	0.0	0.1	0.2	0.0	2.0	0.4	0.4	0.2	1.2	1.5

Table 5: Annual Average 2017 Lake Data from the Finger Lake Survey.

2018 Average Values (± 1s)	Honeoye	Canandaigua	Keuka	Seneca	Cayuga	Owasco	Skaneateles	Otisco
Secchi Depth (m)	2.2 ± 0.9	6.1 ± 1.7	5.7 ± 1.3	3.3 ± 1.4	3.4 ± 1.1	3.6 ± 1.1	9.7 ± 4.0	2.7 ± 0.8
Total Suspended Solids (mg/L), Surface	3.3 ± 2.2	2.0 ± 2.2	1.4 ± 0.9	2.0 ± 1.4	1.9 ± 0.9	2.1 ± 0.8	0.9 ± 0.4	3.2 ± 1.5
Total Suspended Solids (mg/L), Bottom	3.0 ± 2.2	1.9 ± 2.2	0.6 ± 0.4	0.6 ± 0.4	1.5 ± 0.6	0.9 ± 0.3	0.6 ± 0.2	1.8 ± 0.4
Total Phosphate (µg/L, TP), Surface	28.7 ± 10.9	14.7 ± 9.2	11.7 ± 3.9	18.5 ± 11.1	14.3 ± 4.0	10.5 ± 3.7	7.0 ± 2.4	15.3 ± 3.8
Total Phosphate (µg/L, TP), Bottom	31.8 ± 11.4	14.7 ± 15.1	8.6 ± 2.5	13.2 ± 4.8	17.7 ± 4.4	9.2 ± 5.0	11.1 ± 9.8	15.3 ± 5.4
Dissolved Phosphate (µg/L, SRP), Surface	1.0 ± 1.0	1.5 ± 2.3	0.1 ± 0.2	1.8 ± 2.8	1.7 ± 5.3	0.1 ± 0.1	0.7 ± 1.7	0.2 ± 0.3
Dissolved Phosphate (µg/L, SRP), Bottom	2.1 ± 3.0	1.3 ± 1.4	0.8 ± 1.7	2.4 ± 3.4	7.1 ± 4.3	1.0 ± 1.5	0.7 ± 1.2	0.2 ± 0.3
Nitrate as N (mg/L), Surface	0.0 ± 0.0	0.1 ± 0.0	0.0 ± 0.0	0.1 ± 0.1	0.6 ± 0.2	0.5 ± 0.1	0.3 ± 0.2	0.4 ± 0.4
Nitrate as N (mg/L), Bottom	0.0 ± 0.0	0.2 ± 0.1	0.1 ± 0.1	0.2 ± 0.2	0.9 ± 0.3	0.8 ± 0.2	0.4 ± 0.3	0.4 ± 0.4
Silica (SR µg/L), Surface	934 ± 372	927 ± 79	1044 ± 137	222 ± 79	335 ± 159	803 ± 180	680 ± 129	775 ± 268
Silica (SR µg/L), Bottom	884 ± 306	1219 ± 234	1308 ± 350	419 ± 139	902 ± 205	1415 ± 326	770 ± 151	1071 ± 385
Chlorphyll a (µg/L), Surface	12.0 ± 10.1	1.3 ± 1.3	0.8 ± 1.0	10.2 ± 19.3	2.5 ± 2.0	3.2 ± 2.7	0.7 ± 0.7	4.2 ± 3.4
Chlorphyll a (µg/L), Bottom	14.7 ± 13.2	0.2 ± 0.2	0.1 ± 0.1	1.0 ± 1.2	0.0 ± 0.1	0.7 ± 1.5	0.0 ± 0.1	1.9 ± 1.6

Table 6. 2018 Stream Data.

2018 Stream Segment Analysis Data							
Date & Location	**Discharge**	**Specific Conductance**	**Water Temp**	**Nitrate**	**Suspended Solids**	**Total Phosphate**	**Phosphate SRP**
	(m^3/s)	(µS/cm)	(^oC)	(mg/L, N)	(mg/L)	(µg/L, TP as P)	(µg/L, SRP as P)
4/4/2018	Data Collected by GEO-210 Envionmental Hydrology Class						
Dutch Hollow, Rt 38A	4.0	459.0	7.3	1.7	17.1	31.6	0.6
Dutch Hollow, North Rd	2.8	417.0	7.7	2.8	33.3	28.3	1.5
Dutch Hollow, Old State Rd	1.5	320.0	7.2	2.2	44.0	26.4	0.6
4/18/2018	Data Collected by GEO-210 Envionmental Hydrology Class						
Dutch Hollow, Rt 38A	2.54	481.0	5.5	2.0	10.0	19.6	2.0
Dutch Hollow, North Rd	2.27	444.0	4.6	2.2	7.1	16.3	0.3
Dutch Hollow, Old State Rd	0.81	365.0	4.9	1.3	4.9	15.6	0.8
5/18/2018							
Dutch Hollow 38A	0.45	500	16.0	0.4	1.3	12.8	2.9
Dutch Hollow North St	0.42	487	15.4	0.7	1.8	8.4	0.0
Dutch Hollow South Trib	0.06	496	14.5	1.1	0.9	5.8	0.0
Dutch Hollow Benson Trib	0.02	679	15.9	2.5	1.6	5.8	0.0
Dutch Hollow Benson Rd	0.01	454	16.3	0.6	1.8	9.4	2.2
Dutch Hollow Old State Rd	0.25	454	16.1	0.9	3.2	13.9	0.0
Owasco Inlet Moravia Rt 38	2.69	407	17.3	0.6	3.3	33.7	0.0
Fire Lane 20	0.02	559	14.4	5.1	1.0	14.7	0.0
Fire Lane 26	0.02	554	16.0	6.1	2.6	141.6	0.0
5/28/2018							
Dutch Hollow 38A	0.35	521	18.9	0.6	3.6	13.2	3.9
Dutch Hollow North St	0.35	504	18.6	0.7	2.7	15.5	3.2
Dutch Hollow South Trib	0.06	523	19.8	1.5	1.9	25.0	8.7
Dutch Hollow Benson Trib	0.03	720	20.0	2.1	3.0	16.0	5.8
Dutch Hollow Benson Rd	0.24	463	20.0	0.6	3.2	7.3	1.6
Dutch Hollow Old State Rd	0.13	456	19.9	0.7	3.6	7.1	1.8
Owasco Inlet Moravia Rt 38	2.13	411	20.2	1.2	5.4	14.5	1.9
Fire Lane 20	0.04	554	18.6	4.5	2.4	10.9	3.5
Fire Lane 26	0.02	525	19.8	3.8	4.8	13.0	2.4
6/4/2018							
Dutch Hollow 38A	0.23	544	16.7	0.8	2.1	20.4	5.1
Dutch Hollow North St	0.27	534	16.9	0.7	2.3	15.1	4.2
Dutch Hollow South Trib	0.05	552	15.6	0.9	0.8	13.9	9.1
Dutch Hollow Benson Trib	0.01	757	16.6	2.9	3.2	53.3	37.9
Dutch Hollow Benson Rd	0.18	490	16.3	0.7	1.8	11.8	0.9
Dutch Hollow Old State Rd	0.08	497	16.2	0.8	2.8	9.0	1.1
Owasco Inlet Moravia Rt 38	1.65	588	16.4	0.9	4.9	14.7	15.0
Fire Lane 20	0.02	575	15.2	5.0	2.4	9.2	1.2
Fire Lane 26	0.00	536	15.4	4.6	2.3	6.5	3.2

Table 6. 2018 Stream Data (continued)

6/12/2018							
Dutch Hollow 38A	0.14	553	16.9	0.3	3.1	10.5	0.0
Dutch Hollow North St	0.11	530	17.3	0.4	4.2	14.0	0.0
Dutch Hollow South Trib	0.01	564	16.0	1.1	34.8	11.4	10.4
Dutch Hollow Benson Trib	0.01	764	17.4	2.4	7.0	7.0	7.5
Dutch Hollow Benson Rd	0.08	519	17.2	0.5	6.9	4.1	0.0
Dutch Hollow Old State Rd	0.05	509	17.0	1.2	6.7	4.1	1.0
Owasco Inlet Moravia Rt 38	1.09	464	17.4	0.6	7.3	4.1	0.0
Fire Lane 20	0.01	547	15.9	5.0	5.0	7.0	7.0
Fire Lane 26	0.01	536	15.5	4.2	6.3	14.5	2.0
6/13/2018							
Dutch Hollow 38A	0.10	529	19.3	0.2	1.7	4.9	2.9
Dutch Hollow North St	0.10	533	18.9	0.5	2.8	11.6	1.3
Dutch Hollow South Trib	0.01	572	17.2	1.0	1.5	11.0	11.1
Dutch Hollow Benson Trib	0.01	759	18.4	2.1	23.0	9.0	9.8
Dutch Hollow Benson Rd	0.10	531	18.5	0.4	9.2	3.2	2.0
Dutch Hollow Old State Rd	0.09	522	18.4	0.9	6.1	14.1	2.5
Owasco Inlet Moravia Rt 38	0.88	466	18.6	0.6	6.4	25.1	2.8
Fire Lane 20	0.01	547	16.6	10.0	2.0	10.9	9.5
Fire Lane 26	0.00	530	16.8	6.2	11.9	10.7	6.1
2018 Average Values	excluding April surveys						
Dutch Hollow 38A	0.25	529.40	17.56	0.45	2.36	12.35	2.98
Dutch Hollow North Rd	0.25	517.60	17.42	0.61	2.76	12.91	1.75
Dutch Hollow South Trib	0.04	541.40	16.62	1.14	7.98	13.42	7.87
Dutch Hollow Benson Trib	0.02	735.80	17.66	2.40	7.56	18.21	12.20
Dutch Hollow Benson Rd	0.12	491.40	17.66	0.57	4.58	7.15	1.37
Dutch Hollow Old State Rd	0.12	487.60	17.52	0.89	4.48	9.66	1.28
Owasco Inlet Rt 38 Moravia	1.69	467.20	17.98	0.79	5.46	18.43	3.95
Fire Lane 20	0.02	556.40	16.14	5.93	2.56	10.53	4.25
Fire Lane 26	0.01	536.20	16.70	4.99	5.58	37.27	2.74
2018 Average Fluxes				N kg/day	TSS kg/day	TP kg/day	SRP kg/day
Dutch Hollow 38A				9.9	51.9	0.3	0.1
Dutch Hollow North Rd				13.3	60.3	0.3	0.0
Dutch Hollow South Trib				3.8	26.4	0.0	0.0
Dutch Hollow Benson Trib				3.4	10.8	0.0	0.0
Dutch Hollow Benson Rd				6.0	48.6	0.1	0.0
Dutch Hollow Old State Rd				9.3	46.8	0.1	0.0
Owasco Inlet Rt 38 Moravia				115.0	795.7	2.7	0.6
Fire Lane 20				9.9	4.3	0.0	0.0
Fire Lane 26				4.3	4.8	0.0	0.0