THE 2019 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. The resulting monitoring program of Owasco Lake and its watershed was designed 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 Watershed Lake 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 highlighted 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 stimulates additional algal growth and degrades water quality.
- The lake has experienced late-summer / early fall 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-green algae 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 has significantly reduced nutrient loading to the Owasco Inlet and thus Owasco Lake.

- The increased adoption of agricultural best management practices in the watershed and initial follow through on recommendations made by the proposed Owasco Watershed Rules and Regulations, Cayuga County Soil and Water Conservation District, Watershed Inspection and Protection Division's Office (formerly the Watershed Inspector's Office) and the Owasco Lake Watershed Management Council should also reduce nutrient loads to the lake as well.
- Streams and tributaries are the primary source of nutrients and sediments to the lake, especially during "wet" years but also "dry" years.
- Annual nutrient load estimates at Dutch Hollow Brook positively correlated to precipitation totals, especially precipitation during the spring season.
- Daily nutrient and sediment loads measured near the terminus of Dutch Hollow Brook indicated that over 90% of the loads are delivered during precipitation/runoff events, especially in the spring season.
- The large nutrient and sediment inputs during 2011, 2014, and 2015 were coincident with and probably "triggered" the recent BGA blooms[1]. Even though coincidence does not prove causation, these excessive loads were unique over the past decade.
- Since 2011, annual phosphorus budgets for Owasco Lake initially revealed larger inputs than outputs. A continued net accumulation of phosphorus in the lake, i.e., when nutrient inputs exceed outputs, will continue to degrade water clarity and water quality. In the past few years however, the balance has turned and inputs have become smaller than outputs.
- Phosphorus loading must be curtailed even more to significantly improve water quality in Owasco Lake. This effort must be intensified because if all 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. Cayuga County Planning will complete the EPA Nine Key Elements Plan in the coming year. 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, DEC 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.

Numerous economic reasons mandate remediation efforts in the watershed (Fig. 1). First and foremost, the lake is a public drinking water source for the City of Auburn, Town of 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. Third, declining water quality will negatively impact local property values and tax revenues. 2016 property assessments in the watershed, like neighboring Finger Lake watersheds, reveal significantly larger property assessments per acre for parcels adjacent to the lakeshore than parcels away from the lake. Therefore, municipal officials should do everything in their

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[1]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

power to maintain water quality in the lake. Finally, we owe it to future generations to leave this lake in better shape than when we inherited it.

This report highlights the 2019 monitoring efforts within the Owasco Lake watershed. The research included water quality analyses of the lake and selected tributaries. The lake monitoring for this project focused on offshore sites. The stream effort included the analysis of springtime grab samples from Dutch Hollow Brook, Owasco Inlet and two small tributaries entering along the western shoreline, and April through October daily loads at Dutch Hollow Brook. This data enabled estimation of P-budgets for the watershed. We also present the results from ongoing drone flights along selected nearshore sites. Finally, relatively inexpensive instrument arrays were tested to monitor nearshore blue-green algae blooms, and potential causes for the blooms.

METHODS

The offshore lake and stream sample sites and field/laboratory methods used in 2019 were similar to the 2005 – 2019 programs.

Owasco Lake: The 2019 lake survey sampled Sites 1 and 2 on a monthly basis from late May through late September (Table 1, Fig. 1). 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 2019 monthly survey dates were: 5/29, 6/25, 7/23, 8/13 & 9/24. 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, samples macrophyte and mussel densities and populations at selected sites/dates, and surveyed six nearshore lake sites. The results of the Emerson Foundation effort will be discussed in a separate report[2]. Finally, Cayuga County Fl-LOWPA funds supported a pilot of nearshore instrumentation (FLI Sensor Nodes, automated cameras and weather stations).

The lake-monitoring field methods were identical to the earlier research. A CTD profile, bbe FluoroProbe profile, Secchi disk depth, vertical plankton tow (integrate upper 15 m, 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, μE/cm^2-s), and fluorescence (a measure of total chlorophyll, 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 FluoroProbe electronically measures water column profiles of four different algal groups and yellow substances based on their accessory pigments. It distinguishes: '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 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), dissolved oxygen (mg/L) and alkalinity (mg/L, CaCO$_3$) using hand-held probes and field titration kits, and

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[2] Halfman, et al., 2019. Blue-green algae in Owasco Lake, the 2019 Update. The 2018 Annual Report to the Fred L. Emerson Foundation.

analyzed back in the laboratory for total phosphate (μg/L, P), soluble reactive phosphate (SRP, μg/L, P), nitrate (mg/L, N), dissolved silica (μg/L, Si), chlorophyll-a (μg/L) and total suspended solid (mg/L) concentrations.
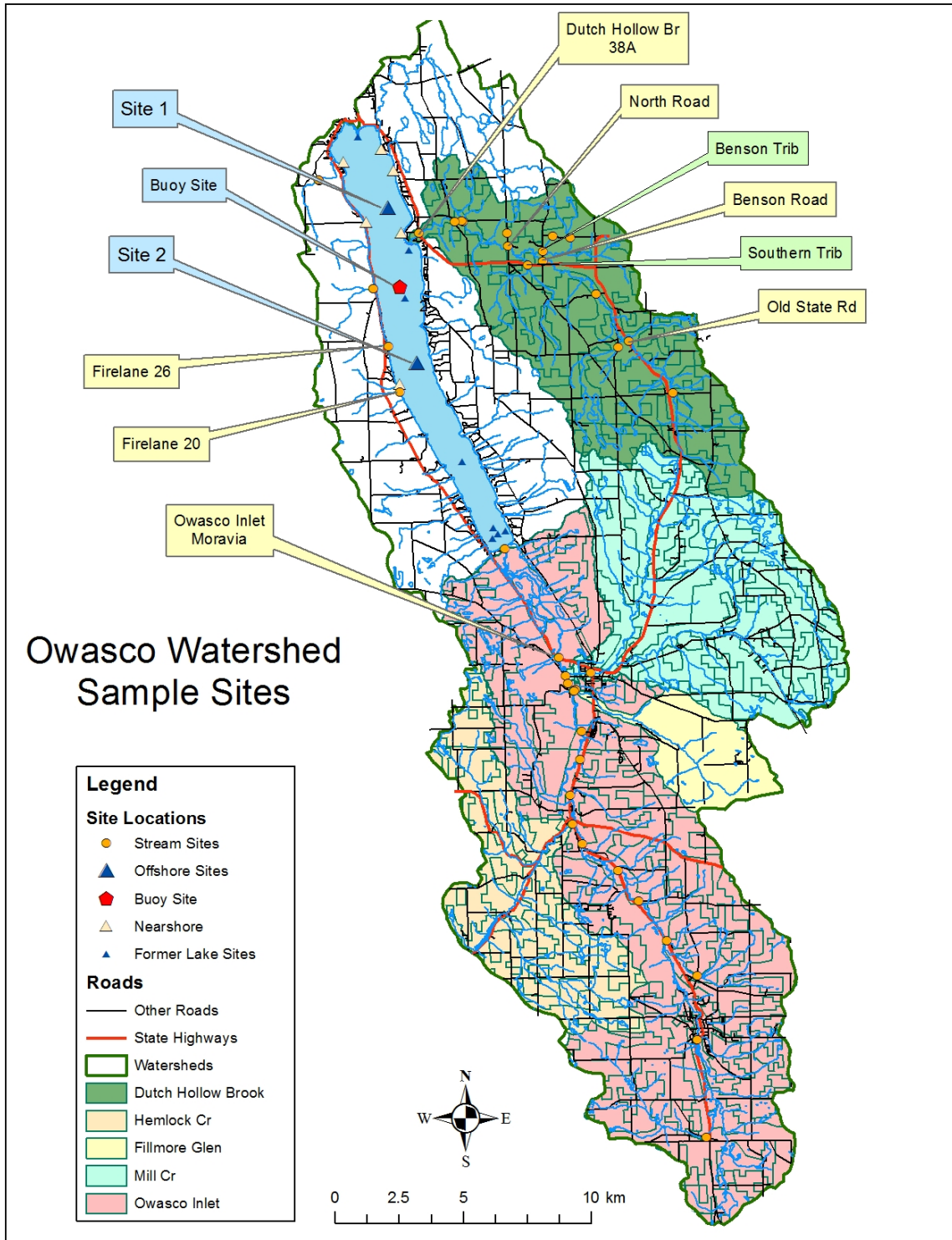

Owasco Watershed Sample Sites

Site 1
Buoy Site
Site 2
Firelane 26
Firelane 20
Owasco Inlet Moravia

Dutch Hollow Br 38A
North Road
Benson Trib
Benson Road
Southern Trib
Old State Rd

Legend

Site Locations
- ● Stream Sites
- ▲ Offshore Sites
- ⬠ Buoy Site
- △ Nearshore
- ▴ Former Lake Sites

Roads
- —— Other Roads
- —— State Highways
- ▢ Watersheds
- ▨ Dutch Hollow Brook
- ▨ Hemlock Cr
- ▨ Fillmore Glen
- ▨ Mill Cr
- ▨ Owasco Inlet

0   2.5   5   10 km

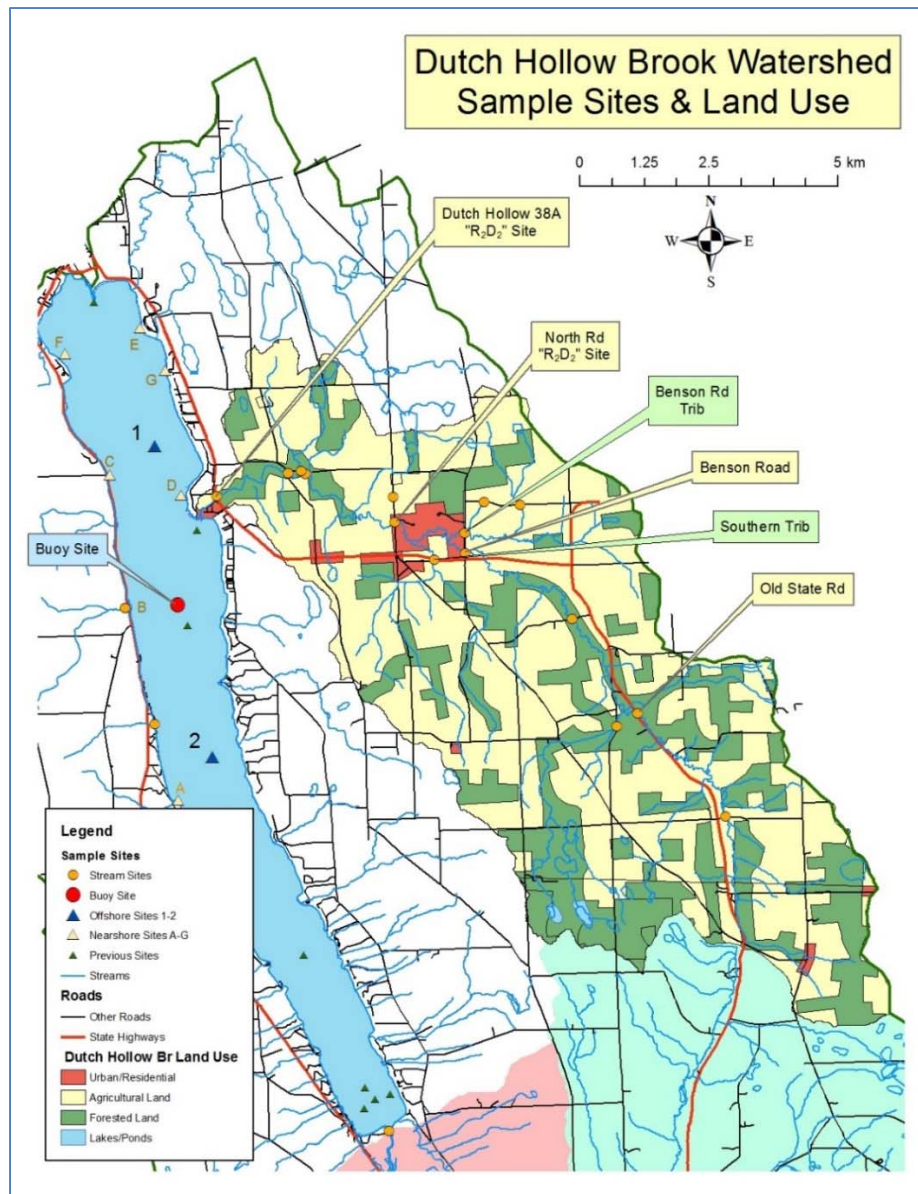Fig. 1. The lake and stream sites.

Fig. 1 continued. 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 in 2018		
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

Owasco Buoy: The FLI meteorological and water quality monitoring buoy manufactured by YSI/Xylem was redeployed at its mid-lake site from 4/15 through 10/30 (Table 1, Fig. 1). As in previous deployments, the buoy was programmed to collect water column profiles with a YSI/Xylem EXO2 water quality sonde every 12 hours (noon and midnight). The sonde detected 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 below the surface. The buoy also contained a standard suite of meteorological sensors recording five-minute mean, air temperature, barometric pressure, relative humidity, light intensity, wind speed and wind direction data every 30 minutes. Raw data were periodically transferred to HWS by cellular phone ~1 hour after collection and archived in a database on a website (http://fli-data.hws.edu/buoy/owasco/). Minimal solar power due to unrelenting cloudy/rainy weather prevented collection of water quality data from 4/15 – 4/20, 4/22 – 4/24, 4/28, 4/30, 5/13, 5/22, and 10/20; and meteorological data from 4/17 through 4/23.

Owasco Streams: The 2019 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 at the end of Fire Lanes 20 and 26 (Fig. 1). Stream sites were visited four times in 2019, specifically 5/21, 5/24, 6/6 and 6/18. Dutch Hollow Brook was sampled at four sites along its main course, 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. Owasco Inlet was sampled at one site just upstream of the Owasco Flats just north of Moravia where the Inlet crosses Rt 38. Two small tributaries that enter along the western edge of the lake at the ends of Fire Lanes 20 and 26 were also sampled. The selected sites duplicated those sampled in the past in this watershed.

At each site, stream discharge, water temperature, conductivity, dissolved oxygen and alkalinity were measured onsite using hand-held probes or field titration kits. Water samples were also collected and subsequently analyzed back in the laboratory for total phosphate (TP), soluble reactive phosphate (SRP), nitrate (N) 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 Owasco Inlet and Dutch Hollow Brook at both 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). Stream discharge was not measured this year at the Owasco Inlet site due to high flows. The USGS gauge flow measured at Moravia (Site 4235299) was used instead.

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 4/10 to 10/29 to investigate the impact of event *versus* base flow variability on nutrient and sediment loads to the lake (Figs. 2a & 2b). One logger in each pair was deployed in air and the other deployed underwater at a fixed elevation just above the stream bed. Both loggers were attached to a fence post driven into the stream bed. The deployment 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 simgle pair of loggers to a flood, vandalism or any other unfortunate issues. The duplication was useful in 2019 because a flood event on 5/10 and again on 5/15 dislodged both loggers from the stream bed, and one logger was lost and replaced with a new one in June.

Fig. 2a. 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).

Fig. 2b. 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 autosampler was programmed to collect 1-L of water every day (4 am). This frequency successfully collected both event and base flow samples in previous years. 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 nutrient concentrations. In 2019, two samples were not analyzed over the 203 day deployment for various reasons (e.g., dumping out the filtered water before saving the filtrate for the dissolved nutrient analyses).

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 estimate the stream discharge for every ISCO water sample.

Laboratory Analyses: Plankton enumerations identified over 100 individuals to genus (and typically species) level under a microscope and reported as date averaged relative percentages. 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

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

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. The stream filtrate was stored at 4°C until soluble reactive 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. The lake filtrate was stored at 4°C until SRP, nitrate and dissolved silica analyses. The filtered residue was kept frozen until chlorophyll-a analysis by spectrophotometer after pigment extraction in 90% acetone. Multiple reagent blanks and standards were run for each analysis for a continuous check on data quality. The nitrate triplicate blanks and standards occasionally yielded concerns. 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.

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 (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. Multiple, overlapping images were collected at each nearshore site (Sites A-G) to investigate algae and macrophyte distributions, the distribution of blue-green algae blooms and open water algal concentrations (Table 1, Fig. 1). The overlapping images were spatially aligned in Adobe Photoshop or Drone Deploy. Each composite image was georeferenced in ArcGIS to 2015 satellite digital orthoimagery (NYS Clearinghouse data).

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

Fig. 4. A weather station and automated camera at a dock site.

Nearshore/Dockside Water Quality Monitoring: Seed funding in 2019 from FL-LOWPA tested the suitability of weather stations, water temperature loggers, automated cameras and FLI Sensor Nodes with dissolved oxygen sensors to detect and elucidate occurrences of nearshore BGA blooms, and the bloom's precursor weather and water quality data. The test locations were Sites D and E, located at Martin and Burtis Points in the northeast side of the lake (Fig. 1). At each site, a weather station (Ambient 1002-WS) recorded air temperature, rainfall, barometric pressure, humidity, light intensity, wind speed and direction at 30 minute intervals (Fig. 4). An

ONSET HOBO U20L-04 data logger were strapped to dock posts in ~1 m of water to record water temperature and lake level at 30 minute intervals. A Brinno TLC-200 automated camera was deployed on the weather station pole 3 to 4 m above the lake's surface to collect images of the lake's surface every 10 minutes from dawn to dusk. At this deployment height, the 60° field of view allowed the camera to image 2 x 3 to 3.5 x 5 meter areas of the lake's surface. The purpose was to image nearshore water quality, i.e., clear, turbid water, and hopefully capture nearshore BGA blooms. Finally, prototype FLI Sensor Nodes were deployed at each site to record water temperature and dissolved oxygen. Each site was visited bi-monthly to measure water temperature (by handheld meter), dissolved oxygen concentrations (by titration) and atmospheric data (Kestral 5000), and swap SD memory cards in the camera for image analysis in the laboratory.

RESULTS & DISCUSSION

2019 PRECIPITATION

Previous reports concluded that annual rainfall totals, its seasonal variability and individual storm 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 nutrient and sediment loads to the lake, especially rain events in the spring season. Smaller, seasonal scales, influence the relative soil saturation, 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 seasonal 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 2019 was very close to "normal" throughout the field season, i.e., spring was at 106% above "normal", summer at 103% above "normal", and fall at 104% above "normal" (Fig. 5). Normal as defined by the weather bureau. Thus, the near normal rainfall throughout the field season should dictate near normal runoff and nonpoint source nutrient and sediment loads to the lake.

Compared to earlier years, 2019 was slightly drier than 2013, 2014 and 2015, and wetter than 2012, 2016 and 2018 (Fig. 5). Compared to 2017 and 2018, 2019's return to slight above normal rainfall suggests that Owasco Lake water quality should decline.
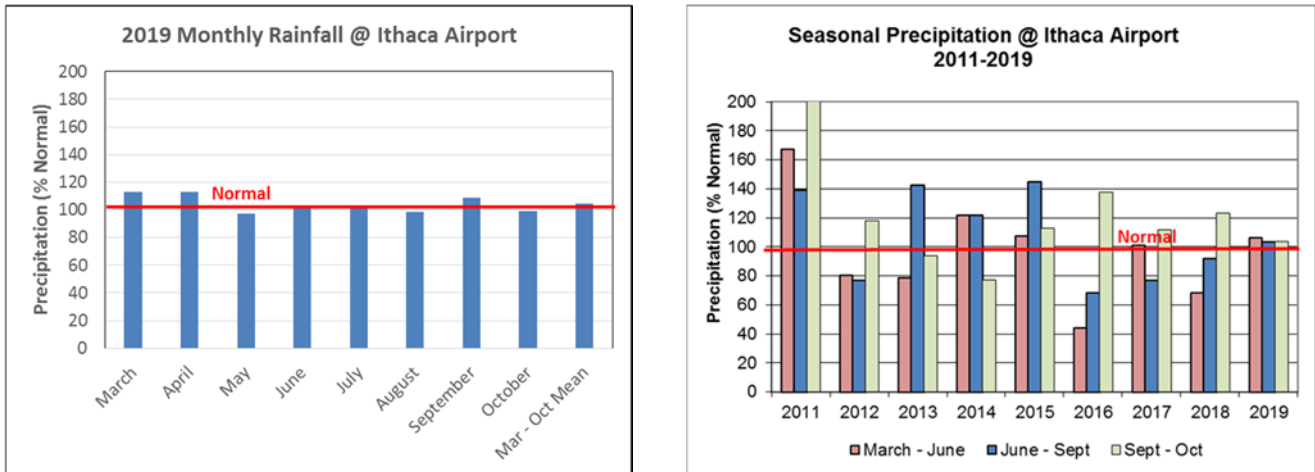
Fig 5. 2019 Monthly (left) and 2011 – 2019 spring, summer and fall seasonal precipitation (right) compared to normal totals at the Ithaca Airport.

LAKE MONITORING

Lake CTD & bbe FluoroProbe Profiles: The 2019 offshore water temperature profiles revealed a slightly cooler late spring surface temperatures, warmer summer temperatures and cooler fall temperatures than earlier years (Fig. 6). The 5/29 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. The depth of the thermocline deepened seasonally, and occasionally oscillated up and down by a few meters due to internal seiche activity Epilimnetic temperatures ranged from 10ºC (~50ºF) in late May to 25ºC (~77ºF) in late July and early August, and cooled to 17ºC (63ºF) through the last cruise of the survey (9/24). Hypolimnetic temperatures remained cold, warming from 4º to 5ºC (39-41ºF) through the survey.

Epilimnetic salinity (specific conductance) ranged from 299 to 329 μS/cm in 2019 (~150 ppm TDS). Like previous years, epilimnetic salinity in 2019 decreased by ~30 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 early spring surface water specific conductance declined slightly since 2015. 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. Since 2015, presumably less snow and thus less road salt accumulated in the lake, allowing salinity in the lake to decline.

The 2019 hypolimnetic specific conductance data were between 330 and 340 μS/cm and increased slightly over time (Fig. 6). These values were similar to those detected in 2017 and 2018, slightly smaller than those in 2016, and all four years were 10 to 20 μS/cm smaller than 2015. Again, the use of an estimated 10,000 tons of additional road de-icing salt from the larger and more frequent snowfall over the 2014 - 2015 winter 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.
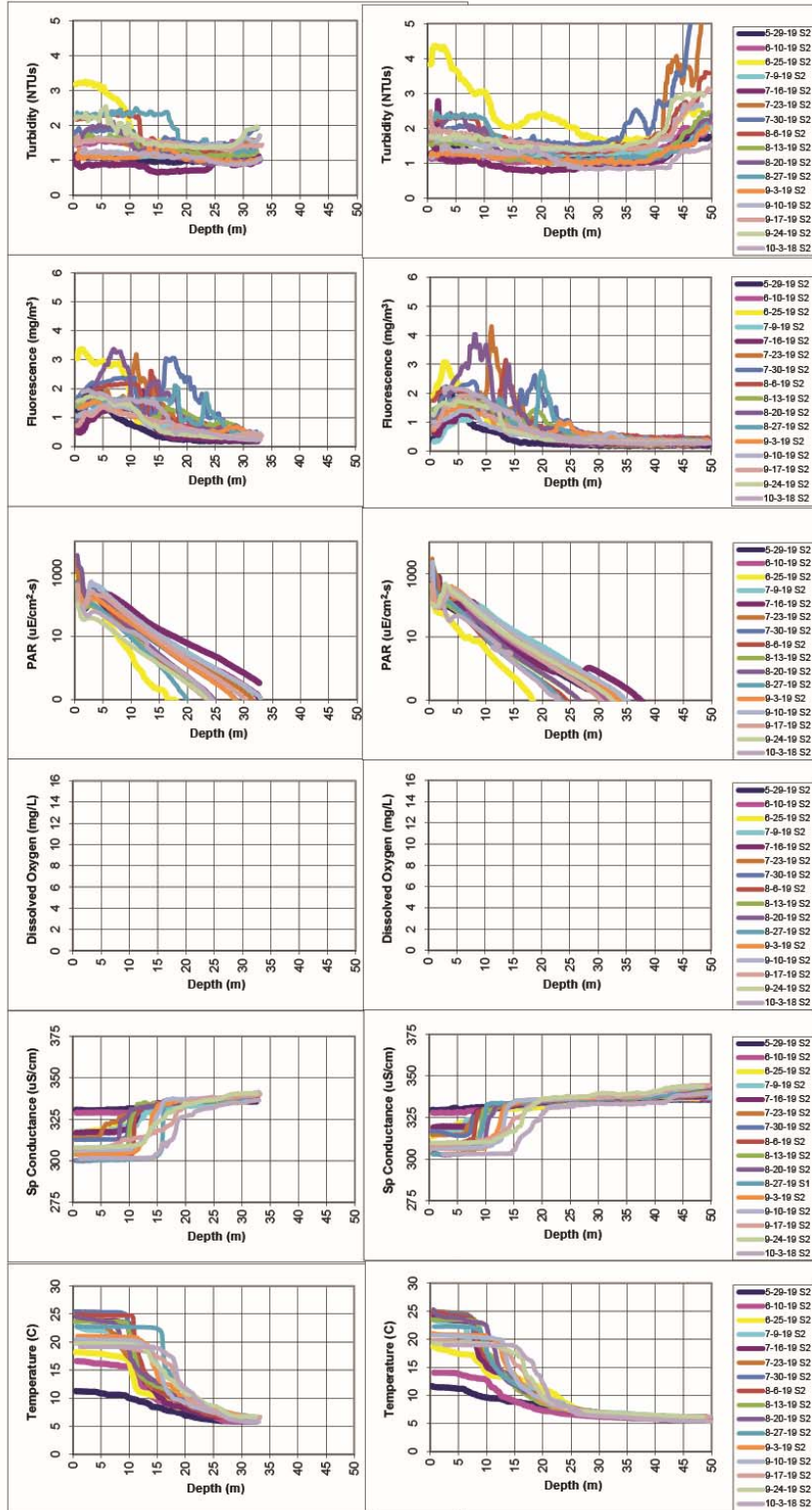
Fig. 6. CTD profiles from Sites 1 & 2 in 2019. The PAR (light) data are plotted on an exponential scale, so that the expected exponential change in light intensity with water depth appears as straight lines. The DO sensor was broken.

The dissolved oxygen (DO) sensor was not working in 2019. Using the FLI buoy data, epilimnetic DO concentrations remained between 8 and 15 mg/L, and near or lightly above 100% saturation. In contrast, 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 5.5 mg/L (~55% 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 2019 than the past four years and most likely reflected a slight decrease in algal productivity during 2019.

Profiles of photosynthetic available radiation (PAR), i.e., light intensity, in 2019 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 expected 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 defines the maximum depth for the photic zone, i.e., water depths above 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 intensity 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 $\mu g/L$ (mg/m^3) on 7/23 and 8/20, and 3 $\mu g/L$ (mg/m^3) on 6/25 and 8/6, and were lower, between 1 and 2 $\mu g/L$, on the other survey dates. Fluorescence measures algal pigment concentrations and not algal populations. Thus, the algal peak at depth may reflect an increase in algal biomass or more pigments per cell in the lower light conditions. These concentrations were similar to 2018 and slightly smaller than earlier years. It parallels changes 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 $\mu g/L$, i.e., algae are typically absent in the dark bottom waters.

The turbidity profiles revealed uniform or nearly uniform turbidities of 1 to 2 NTUs down to a distinct benthic nepheloid layer at the deep site (Fig. 6). On 6/25, the turbidity was above 4 NTUs in the epilimnion, and aligned with an algal bloom and an intense rainstorm on 6/21. At Site 2, the typical increase in turbidity just above the lake floor to concentrations above 7 NTUs was observed in 2019. It parallels the normal rainfall in 2019 bringing in turbid runoff. The change in benthic turbidities from year to year typically parallel 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.
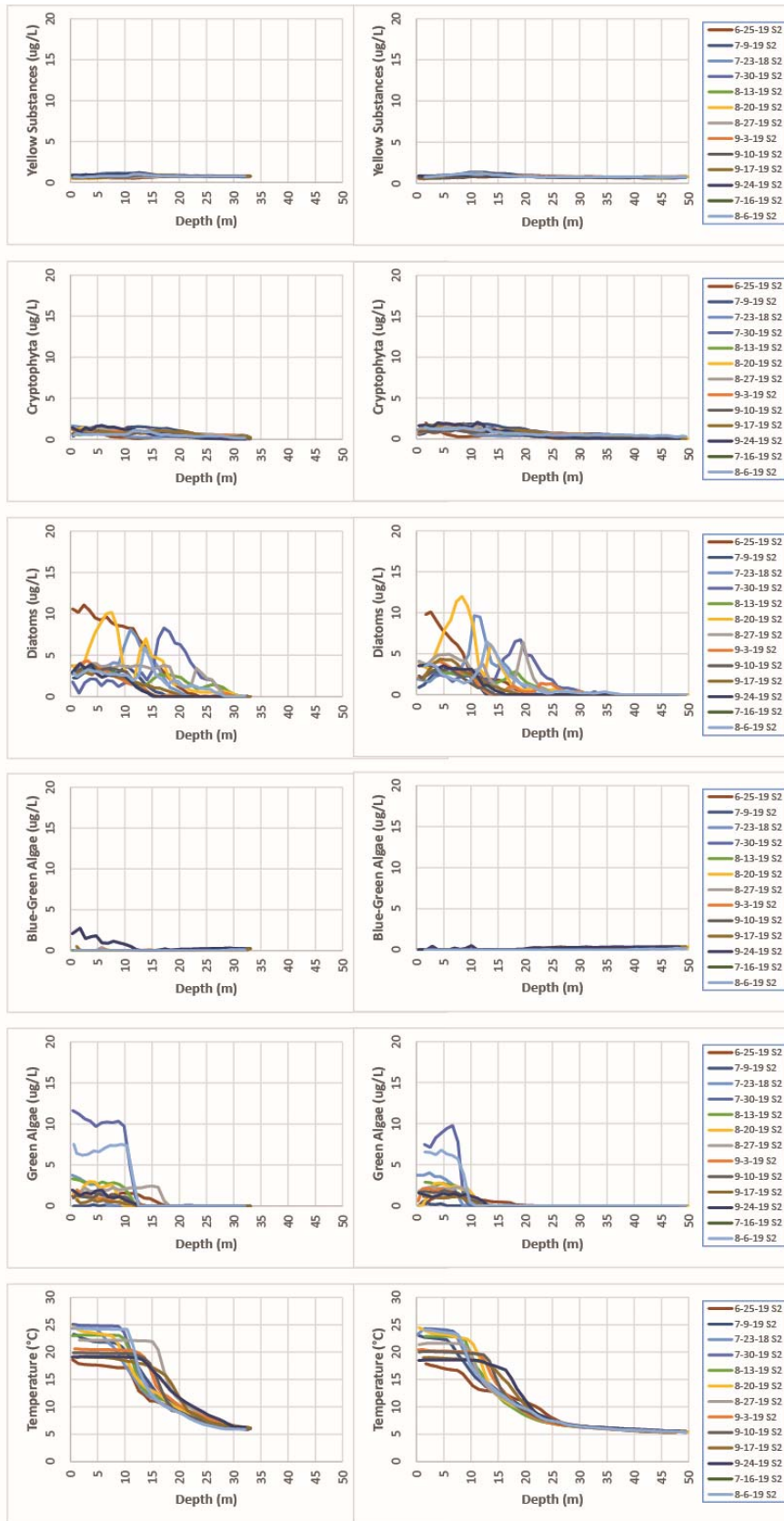
Fig. 7. bbe FluoroProbe profiles of the four algal groups from Sites 1 & 2.

The bbe FluoroProbe data revealed the dominance of diatoms and green algae (above 12 μg/L on 7/30), and lesser amounts of cryptophytes (below 2 μg/L) and blue green algae (below 1 μg/L). Mean epilimnetic total fluorescence concentrations exceeded 10 μm/L (mesotrophic/eutrophic threshold) at the offshore sites on two of the thirteen surveys of the lake (Fig. 7).

Limnology & Trophic Status: Date averaged mean chlorophyll-a concentrations in the epilimnion ranged from 0.1 to 5.7 μg/L and averaged 2.7 μg/L in 2019 (Table 2 in appendix, Fig. 8). The largest values were on 7/30, 8/20 and 8/27. The chlorophyll-a method measured only one algal pigment whereas total chlorophyll and fluorescence data measured many pigments, thus chlorophyll-a data presented here should be smaller than other datasets. The annual mean was mid-range compared to earlier years but still below the 4 to 6 μg/L not to exceed DEC threshold for potable water bodies[4]. Nitrate concentrations ranged from 0.4 to 1.2 mg/L, and an order of magnitude (10 times) below the 10 mg/L maximum contaminant level (MCL) established by the EPA. However, the annual mean concentration was ~0.1 μg/L larger than those detected in earlier years. The lake was not impaired due to phosphorus, as the annual mean total phosphate concentration was 10.5 μg/L and below the 20 μg/L total phosphate (TP) threshold used by the DEC to designate impaired (eutrophic) water bodies. The impaired waterbody threshold was never exceeded in 2019 with the maximum date-averaged TP concentration of 16 μg/L on 7/16. Secchi disk depths ranged from 1.7 to 5.8 meters, and averaged 4.0 meters in 2019 (Fig. 8). This annual average was in between to slightly deeper (~0.5 m) than annual averages detected in recent years by the FLI monitoring effort. Total suspended sediments (TSS) date-averaged concentrations ranged from 1.1 to 7.8 mg/L and averaged 2.6 mg/L. The 2019 TSS data were slightly larger than those measured in previous years.

Overtime, annual mean Secchi depths disk gradually deepened from 2009 through 2012, then shallowed to 2017, except for a reversal in 2016. Deeper depths were detected again in 2018 and 2019. It suggests that the major trigger for the decline in water clarity 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 and "normal" rainfall in 2019 allowed the lake to recover.

Since 2006, annual mean total phosphate (TP) concentrations have increased from ~8 to over 17 μg/L by 2014 with a slight dip in 2013 (Fig. 8). After another dip in 2015 and 2016, TP increased to 16.2 μg/L in 2017. Since then, TP decreased to 10.5 μg/L in 2019. Annual mean soluble reactive phosphate (SRP) concentrations in 2019 remained very small ~0.1μm/L, similar to 2010, 2012, 2015 and 2018, compared to years with significantly larger SRP concentrations, i.e., 2006 and 2017 (1.9 μm/L), and especially 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 (3.9 & 3.7 μg/L), and again in 2014, 2015 and 2016 (3.2, 3.8 & 3.5 μg/L, respectively) than 2011, 2012 and 2013 (1.9 to 2.3 μg/L; Fig. 8). The 2019 annual mean concentration of 2.7 μg/L reversed the decreasing trend from 3.8 in 2015 to 2.4 μg/L in 2018. The total suspended sediment (TSS)

[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.

annual mean concentrations in 2019 (2.6 mg/L) reversed a decline since 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). Overall, 2014 and 2015 revealed the worst water quality for the lake. Water quality improved in 2018 and did not change significantly in 2019. The trends parallel changes in rainfall.
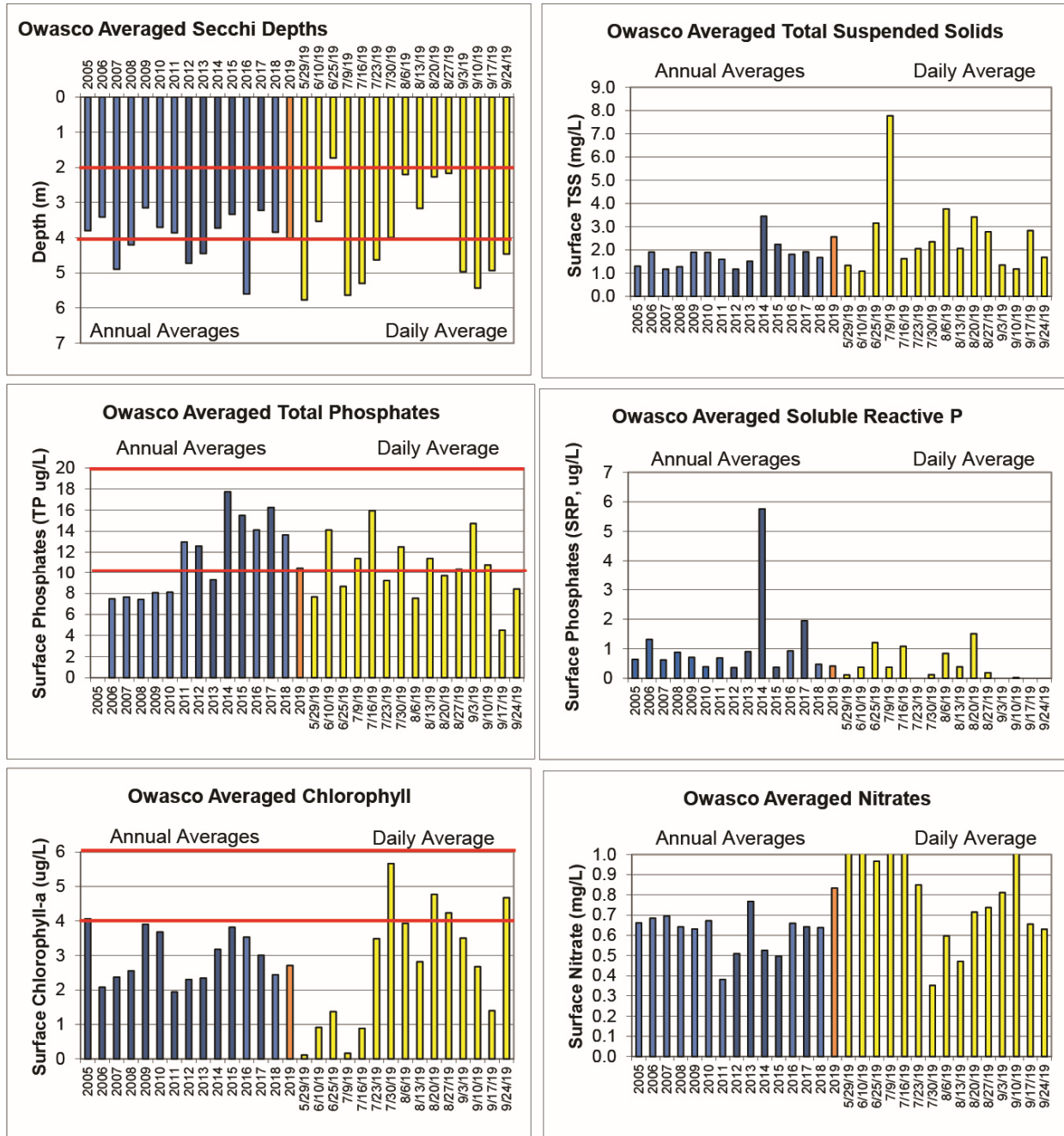

Fig. 8. Annual average surface water concentrations from 2005 (blue) to 2019 (orange), and date averaged offshore surface water data during 2019 (yellow). When appropriate, boundaries for oligotrophic, mesotrophic and eutrophic concentrations are marked with red lines.

The 2019 annual mean total phosphate, and hypolimnetic dissolved oxygen saturation data (from the buoy) placed Owasco Lake just above the oligotrophic-mesotrophic trophic boundary (Table 3). Nitrogen, measured by nitrate concentrations, Secchi disk 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, indicate that the lake is in a delicate balance. Any increase (or decrease) in nutrient loads from one year to the next decreases (or increases) the lake's water quality.

Table 3. Concentration boundaries for Oligotrophic (low productivity), Mesotrophic (mid-range productivity), and Eutrophic (high productivity) lakes. The bold entries reflect Owasco's 2019 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 1:1,900 in 2019. The P:N ratio required by algae is 1:7 (Redfield Ratio). These 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 10 to 100 times more nitrogen than phosphorus, and fluvial sources of nitrates are augmented by additional sources of nitrogen to the lake (e.g., atmospheric deposition of 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 only bloom for a week or so at a time before nutrient limitations and/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 to 0.9 µg/L for SRP, 0.8 to 1.1 mg/L for nitrate, and 720 to 1,410 µg/L for dissolved silica. Chlorophyll-a concentrations revealed the expected decrease from the epilimnion to the hypolimnion of 2.7 to 0.6 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 and 2018 mean TP, total dissolved phosphorus (TDP, i.e., dissolved organic matter and soluble reactive phosphate), total chlorophyll and nitrate surface concentrations determined by C-SLAP were similar to the results from this study (Fig. 9). C-SLAP's TP and N were within or just below the "box" of the box and whisker plots, and TDP and total chlorophyll just above the "box". The variability reflects natural scatter in the data as the limnology of the lake constantly changes; and both TDP and chlorophyll determined by C-SLAP measured more "stuff" that the SRP and chlorophyll-a data presented in this report.

Plankton Data: The phytoplankton (algal) species in Owasco Lake during 2019 were dominated by diatoms, primarily *Fragillaria* and *Asterionella*, with smaller numbers of *Diatoma,* and other species (Table 4 in appendix, Fig. 10). Unlike 2018, the distribution was similar to earlier years. The reason for the mid-summer *Synedra* dominance in 2018 is unclear at this time and it has

never dominated the algal population in the past. 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.
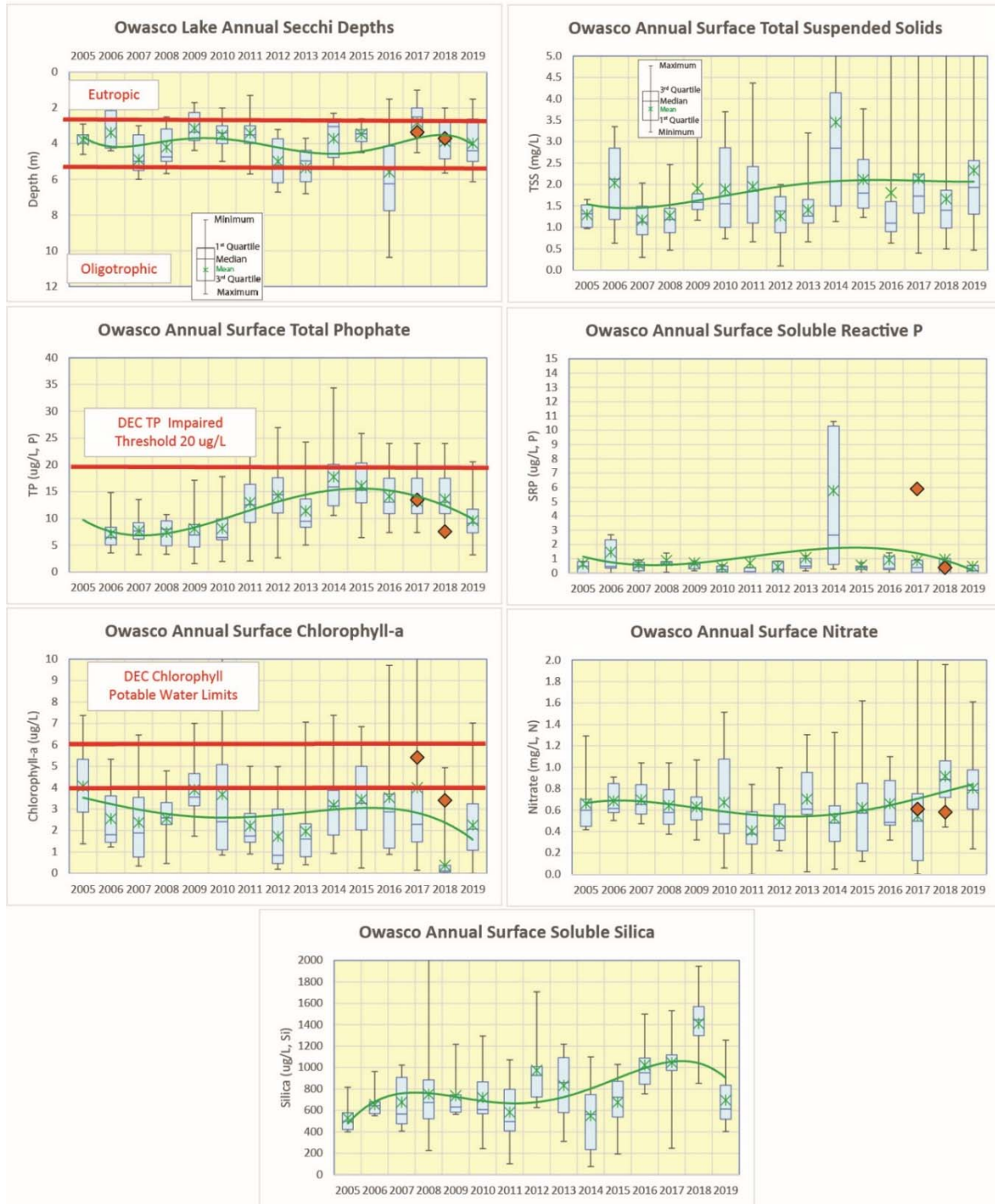

Fig. 9. Box and Whisker plots of the annual nutrient, chlorophyll and Secchi disk data. The annual mean DEC C-SLAP results are also plotted (orange diamonds). DEC measured TDP instead of SRP and total chlorophyll instead of chlorophyll-a, and should result in larger values.
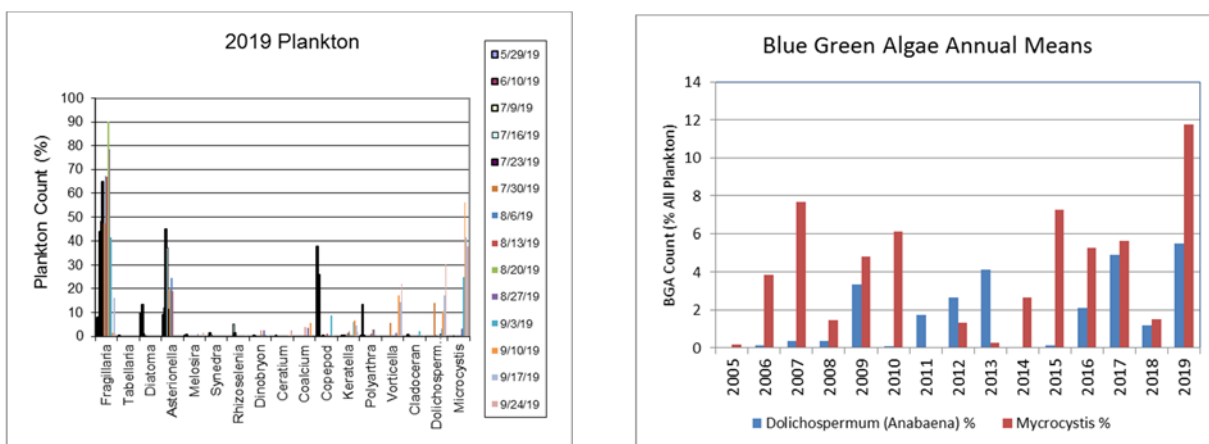
Fig. 10. Date averaged plankton data for 2019 (left) and the mean annual abundance of blue-green algae species since 2005 (right).

Two blue-green genera, *Microcystis* and *Dolichosperma* (previously *Anabaena),* were prevalent in the 2019 late summer and early fall 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. 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]. The largest BGA populations were restricted to the late summer and/or early fall, with *Microcystis* representing over 40% of the plankton counts during a late summer survey in 2007, 2010, 2014, 2015, 2017 and 2019, and *Dolichosperma* making up 30% of the late-summer counts in 2013. In 2019, the offshore presence of blue-green algae increased over 2018, with a maximum relative percentage of 56% for *Microcystis* during the 9/10 survey.

Finger Lake Water Quality Ranks: The 2019 Finger Lake water quality rank for Owasco Lake improved over 2018 (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 2019, water quality in Owasco ranked poorer than Canandaigua, and Skaneateles Lakes, and slightly better than Keuka, Cayuga, and Otisco Lakes, and much better than Seneca and Honeoye Lakes.

Owasco Lake water quality seemingly and slightly improved yet rainfall increased in 2019, it suggests that besides nutrient loading issues, the observed change in water quality in these lakes may also be 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. The recently

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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.

installed BMPs must be beginning to work. So does ecological, "top-down" pressures by zebra and quagga mussels, Asian clams and *Cercopagis*, the fishhook water flea.
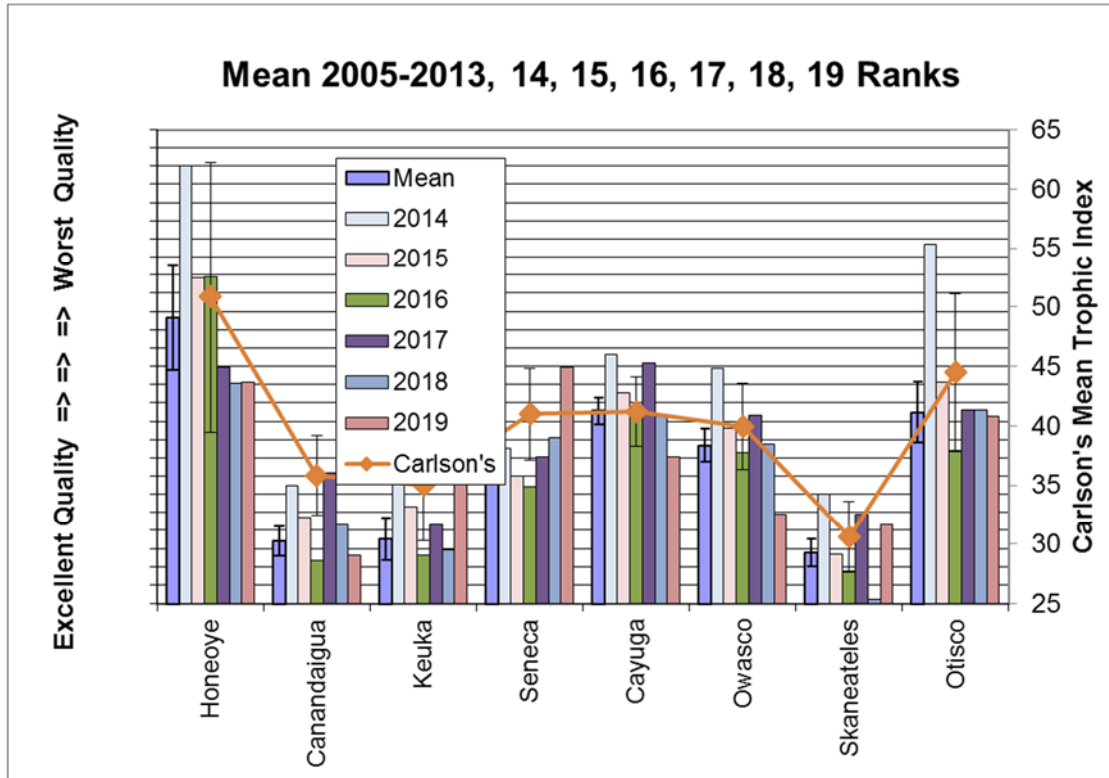

Fig. 11. Annual Water Quality Ranks from 2005 – 2019 for the eight easternmost Finger Lakes. The "mean" dark blue bar averaged the 2005 - 2013 ranks for each lake with a 1σ standard deviation error bar. Carlson's mean trophic indices of the mean Secchi depths, total phosphate and chlorophyll concentrations are also shown.
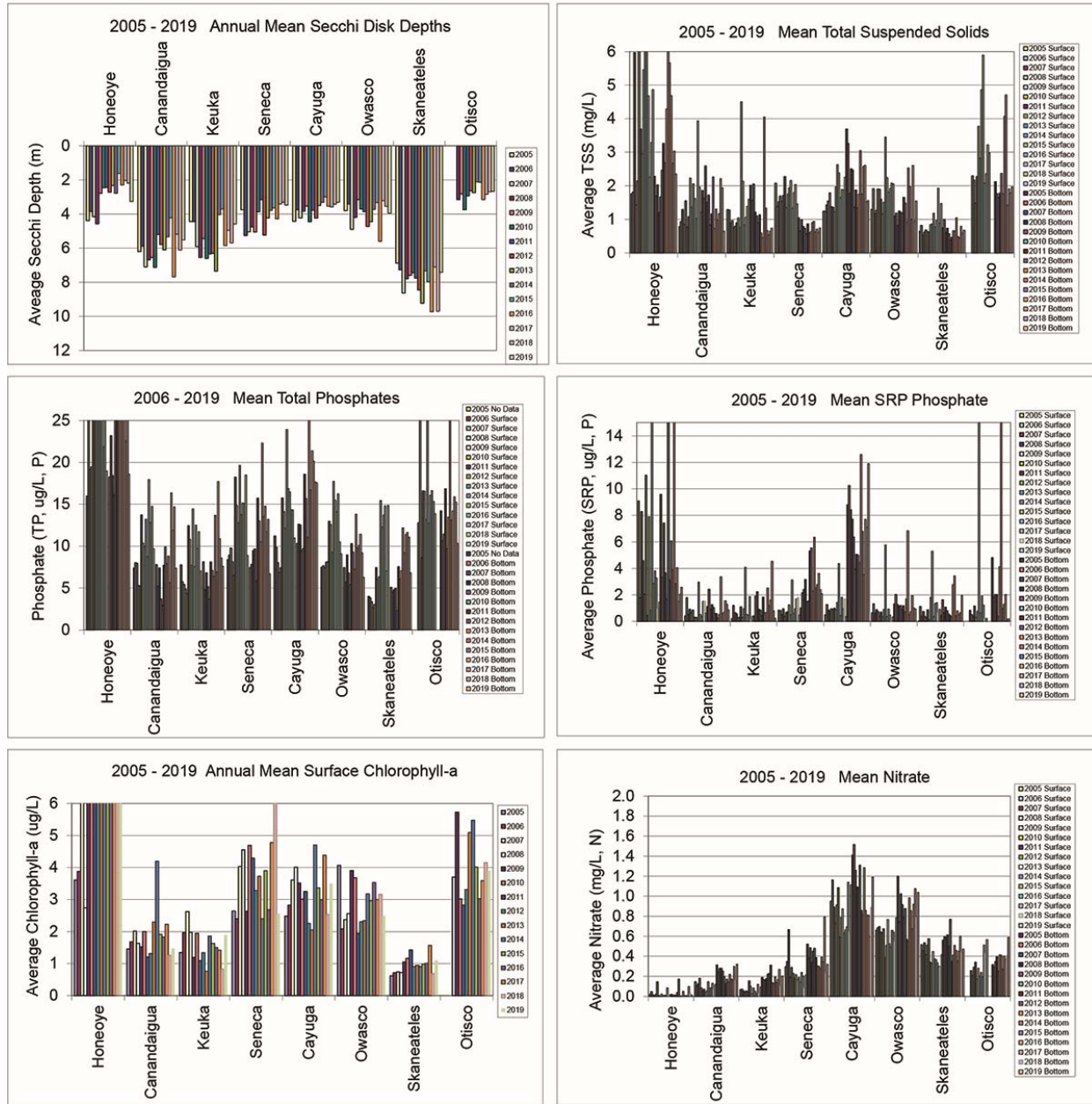
Fig. 12. 2005 – 2019 annual mean limnological data from the eight eastern Finger Lakes.

DRONE FLIGHTS

Unfortunately BGA blooms were not detected in the drone images during the specific site visits during 2019. 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 offshore sites to find a better indicator in the drone images of open-water algal concentrations (Fig. 13). The intent was to determine if the difference between upwelling and down-welling spectra could resolve algal concentrations. The 2019 results confirmed the earlier findings (Fig 13) 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.
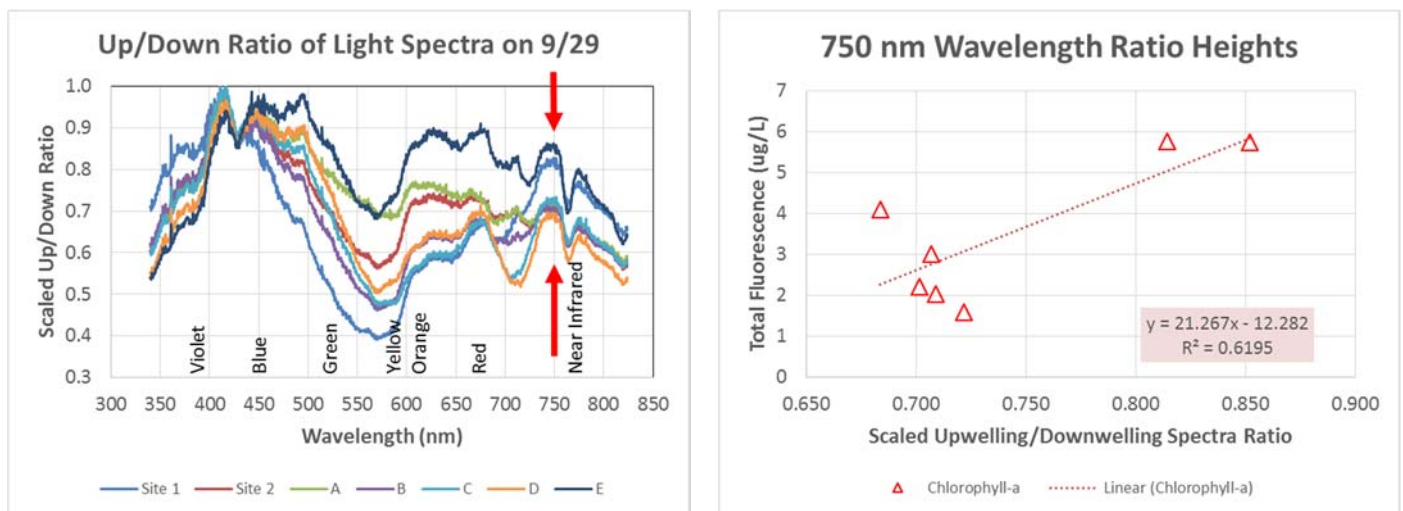

Fig. 13. Ratio of upwelling and down-welling spectra from Sites 1, 2 and A – E on 9/27/17 (left). Algal signatures are typically in the near infrared (~750 nm). Peak heights at 750 nm for each up/down spectral ratio vs. total fluorescence measured by bbe FluoroProbe (right).

SIX YEARS OF FLI BUOY DATA

The FLI meteorological and water quality monitoring buoy was redeployed in Owasco Lake during the 2019 field season with support from the Emerson Foundation. It revealed higher resolution but otherwise consistent changes in the water column as described in the CTD section (Fig. 14). More information can be found in the companion report.
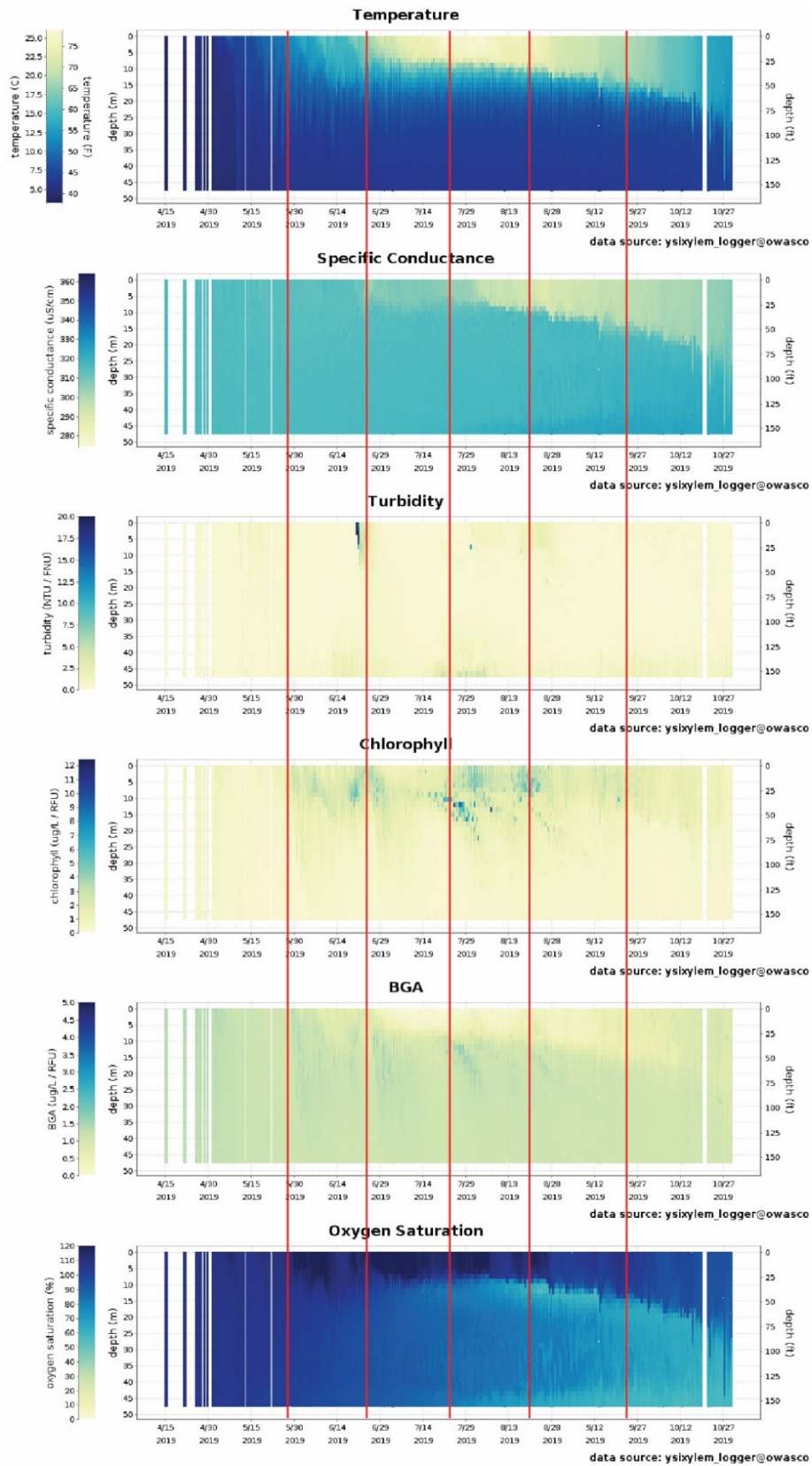
Fig. 14. Buoy water quality data in 2019. The data shown above are raw uncalibrated values. Subsequent calibrations altered the turbidity and chlorophyll data slightly (see companion report). The red lines depict the monthly monitoring cruise dates. The spike in turbidity occurred just after a major rainfall event on 6/21.

NEARSHORE/DOCKSIDE MONITORING RESULTS & DISCUSSION

FLI Sensor Nodes: The FLI Sensor Node is a student designed and built, low cost, data logger designed to record a host of environmental variables. Each self-contained unit integrates an Arduino processor, SD memory card, solar panel, associated electronic components and up to four commercial sensors at an estimated cost of $500 with a simple temperature sensor. The FLI Sensor Node reads each sensor and records the data on an SD card at preset time intervals, thus it provides a reliable sensor array / data logger at a fraction of the cost of comparable commercial counterparts. Two sensors were deployed at Sites D and E in 2019 to test the sensor's suitability in the field. Unfortunately, the deployed units had power, memory and other issues and could not be suitably tested in the field.

The recoverable temperature data from both sites (~two weeks) was informative (Fig. 15). The calibrated temperatures data from duplicate thermistors at each site were very similar to the HOBO results at the same site (r^2 of 0.90 and 0.94). They revealed daily oscillations, warming during the daylight and cooling during nighttime darkness. Similar daily temperature variability was detected at the other nearshore sites in the lake.

The duplicate but uncalibrated dissolved oxygen FLI Sensor Node probes also revealed daily oscillations in DO concentration that co-varied with water temperature (Fig. 15). Sufficient independent DO data were not available to calibrate the Sensor Node results. Regardless, the available data are informative. Water temperature inversely controls saturated dissolved oxygen concentrations. However the diffusion of oxygen between the atmosphere and water column to achieve saturation is a relatively slow process compared to biological activity, i.e., photosynthesis and respiration. If biological activity is intense enough, it will alter DO concentrations by adding oxygen to the water column during daylight hours through photosynthesis, and removing oxygen from the water column during nighttime darkness through respiration. The covariance between the temperature and DO fluctuations indicates that biology had a major impact on nearshore dissolved oxygen concentrations and suggests that bacterial respiration of organic debris in the water column and sediments is important in Owasco Lake nearshore areas, and might even provide a source of nutrients to the water column for the growth of blue-green algae. It dictates a redeployment of temperature and DO sensors in the future and at additional sites to determine if this DO covariance is common everywhere in the lake.

Automated Cameras: The Brinno cameras faithfully recorded ~3 x 4 meter images of the lake every 10-minutes from 7 am to 6 pm over the 105 day deployment. They detected a number of BGA blooms at both sites (Table 6, Fig. 16). The images also differentiated between turbid (lake floor not visible) and clear water (lake floor visible). Glare from the sun did impact a portion of the afternoon images, especially at Martin Point as its camera pointed west into the setting sun. Unfortunately, the available volunteer HABs data from the DEC lack detection locations to determine which method was better at detecting blooms. However, Seneca Lake Pure Waters Association volunteers detected a similar number of blooms as identical cameras deployed at 8 sites around Seneca Lake this past summer, but blooms were typically detected on different dates by the different methods. The volunteer weekly surveys may have missed blooms on other days of the week, and their survey of an entire shoreline zone may have detected shoreline blooms outside of the camera's 3x4 meter field of view. On one day with BGA blooms, a twig influenced the shoreline parallel, northward migration of the bloom at Martin Point (Fig. 16). Bloom migrations parallel to the shoreline were also noted by HABs volunteers. Although expensive, perhaps nearshore currents, e.g., longshore drift, should be measured in the future.
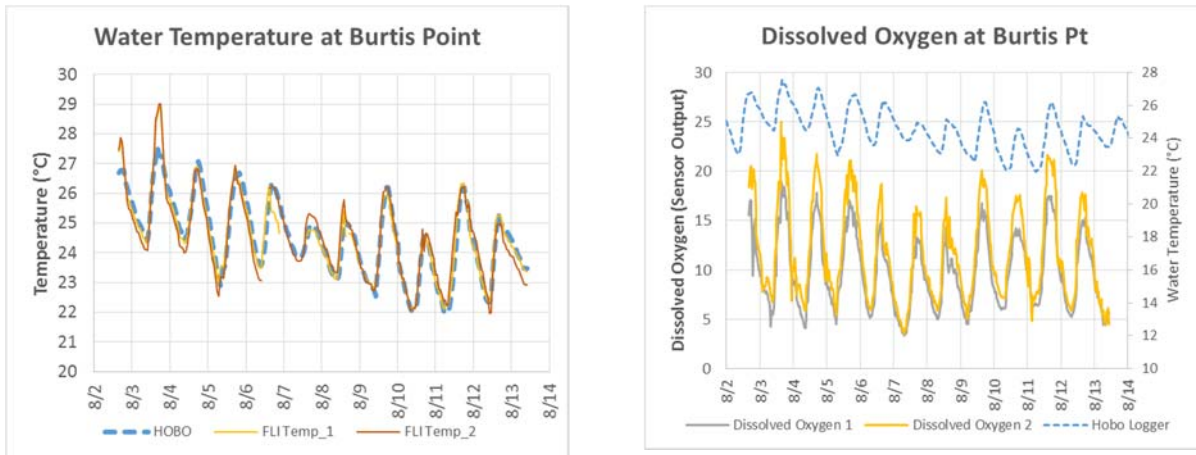
Fig. 15. Dockside water temperatures (above left) and dissolved oxygen concentrations (above right) measured with a HOBO logger and FLI Sensor Nodes at Burtis Point. Similar results were observed at Martin Pt. The FLI DO Sensors were not calibrated.

**Table 6. Brinno Automated Camera Results**

Camera Results (in days)	Martin Pt	Burtis Pt
Blooms	10	14
Clear Water (bottom was visible)	94	80
Turbid Water (bottom not visible)	7	17
Glare Impacted Image	10	2
Camera Malfunctions	0	0

Fig. 16. Example dockside BGA blooms imaged by the Brinno camera, left Burtis Pt, right Martin Pt. Notice how glare impacts the image quality and a twig influenced BGA movement at Martin Pt.

Weather Station Data: Air temperature, rainfall, solar radiation, wind speed and direction data are consistent with BGA blooms (Fig. 17). Air temperature declines, solar radiation is high, wind speed declines and rainfall precedes most blooms at these two sites. For example, over the course of the deployment, the mean wind speed during blooms is 1.8 and 2.7 mph, at Martin and Burtis Pts, respectively, compared to a mean wind speed of 2.1 and 3.8 mph at Martin and Burtis

Pts. It suggests that additional data at more sites should be collected in the future for a more complete statistical analysis of these potential correlations. Interestingly, the mean wind velocities significantly declined at the nearshore sites compared to the FLI buoy site (Fig 18). The dominate wind direction significantly differed at these three sites as well (Fig. 18). The shoreline orientation at each dock site can determine which regional winds are "felt" (onshore winds) and which are not "felt" (offshore winds) or its "felt" impact is greatly reduced (shore parallel winds) at each site. It suggests an interesting speculation. If one shoreline is experiencing calm and sunny conditions, and a bloom appears, then enough variability in winds exists so that not every shoreline should experience the same calm and sunny conditions, and thus a bloom. Thus, BGA blooms can appear at different segments of shoreline on different days. Additional sites should be monitored in the future to confirm this speculation.
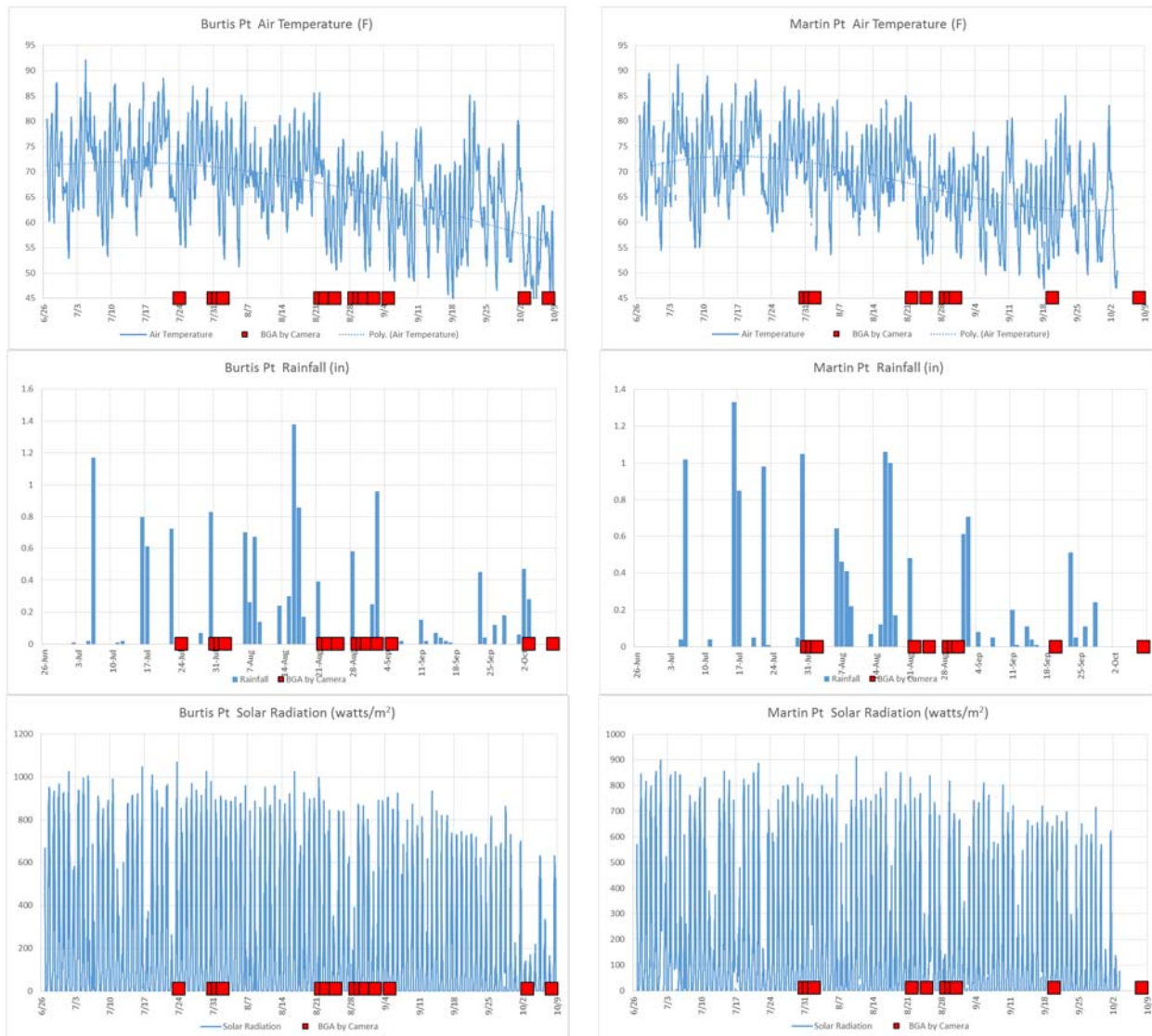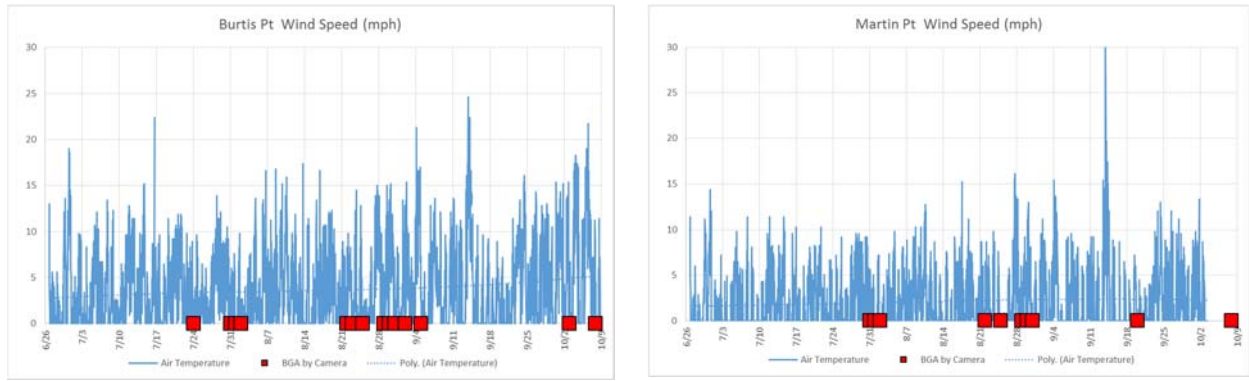
Fig. 17. Dockside air temperature, daily rainfall, solar intensity and wind speed data from Burtis Pt (Left) and Martin Pt (right). The red boxes are dates when BGA were observed in the camera images at the site.
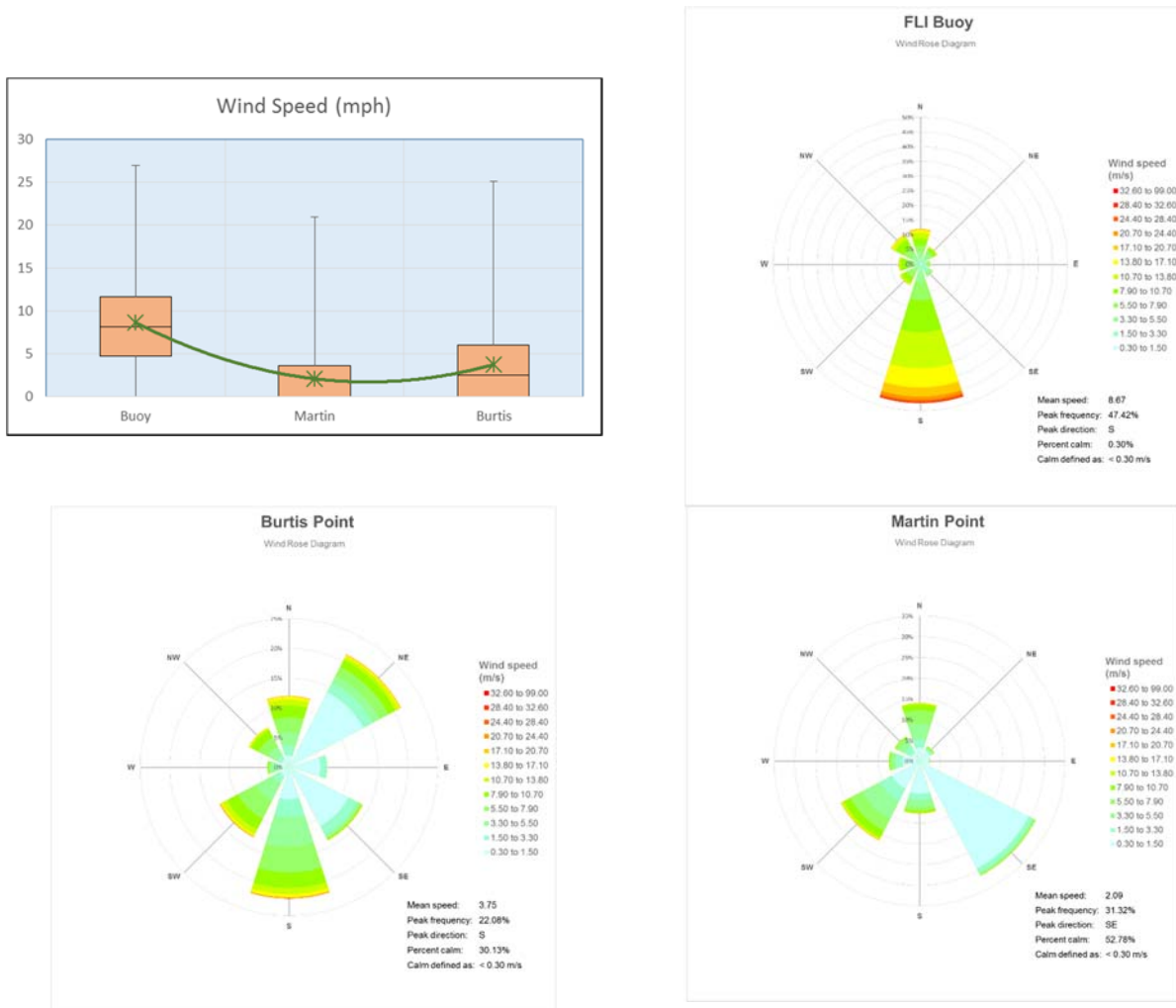

Fig. 18. Box and Whisker plots of wind speed (upper left) and wind roses from the FLI Buoy (upper right), Burtis Pt (lower left) and Martin Pt (lower right) over the deployment.

STREAM MONITORING

Stream Discharge: Stream discharge data from the four May & June stream surveys in 2019 ranged from nearly dry conditions, 0.01 m³/s at Fire Lane 20 & 26 on 6/6 & 6/18, to 5.5 m³/s in Owasco Inlet at Moravia on 5/21 (Table 7 in appendix, Fig. 19). The 2019 flows were in between those detected in past years, and relates to the near "normal" rainfall in 2019. Flows were largest during the early spring reflecting the spring melt in April, saturated ground and near "normal" rainfall during this season.
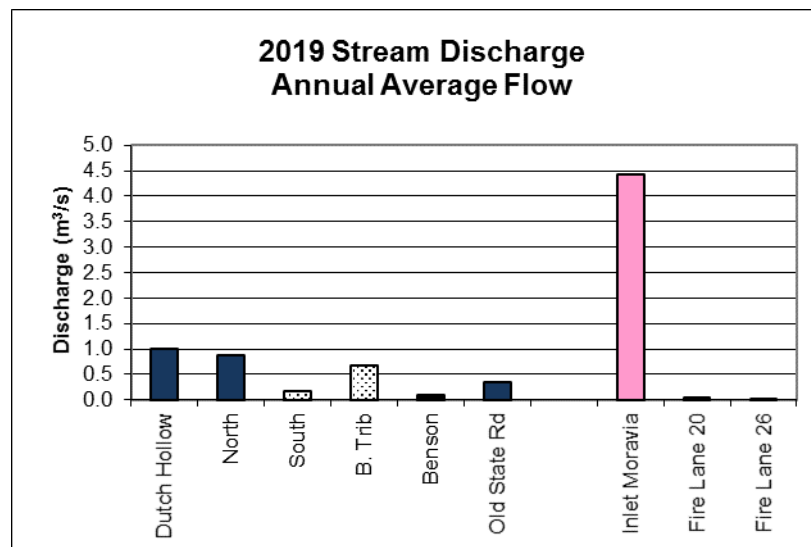

Fig. 19. Annual average stream discharge at each stream site in the Dutch Hollow Brook (purple), Owasco Inlet (pink), Fire Lane 20 and 26 watersheds. Tributary sites along Dutch Hollow Brook are stippled.   Sites are arranged, left to right, from downstream to upstream.

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. 19). 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 slightly smaller at North Rd than 38A this year like previous "wet" years, whereas the opposite is typically true during "dry" years. It suggests that surface runoff and shallow groundwater flow contributed to and increased stream discharge from North Rd down to Rt 38A during "wet" and "normal" 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 proportionally larger (by watershed area) than Dutch Hollow Brook because the Owasco Inlet drains a significantly larger watershed than Dutch Hollow Brook (299 vs. 77 km²). Discharge was proportionally smaller at the two Fire Lane sites due to their smaller watersheds.

Seasonal Variability: Seasonally, the largest discharges of 2019 were detected in the spring at both Dutch Hollow and the Owasco Inlet whereas the smallest were during the summer based on the data logger estimated discharge for Dutch Hollow at the Rt 38A site and the USGS gauge for Owasco Inlet (Fig. 20). The seasonal pattern is typically for this watershed during "wet" and "normal" years.

Differences to Earlier Years: The 2019 annual mean discharge was "in-between" those detected at Dutch Hollow Brook and Owasco Inlet over the past decade (Fig. 21). These differences parallel changes in precipitation.
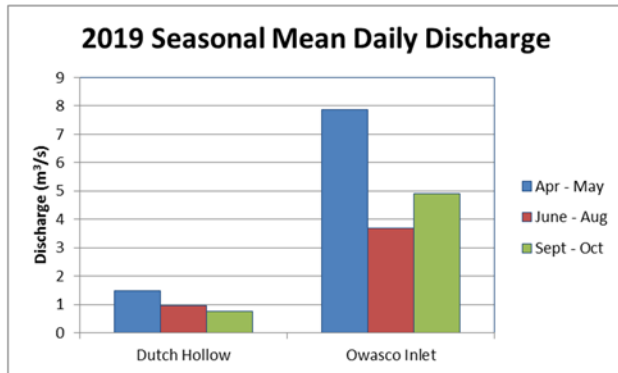

Fig. 20. Seasonal averaged stream discharge for the Rts. 38A and 38 sites, the terminal sites on Dutch Hollow Brook and Owasco Inlet, respectively.
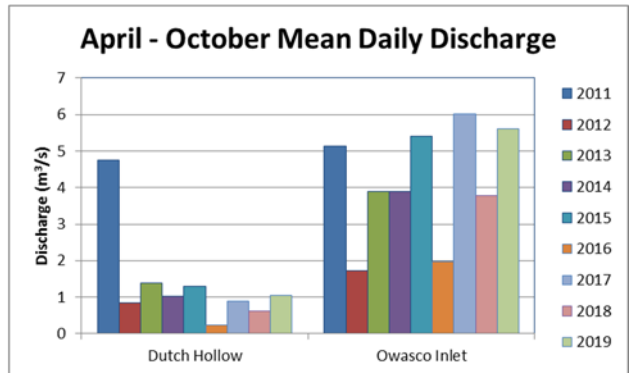

Fig. 21. Field season annual average stream discharge for the Rts. 38A and 38 sites. This plot used the estimated Dutch Hollow Brook data logger and USGS daily Owasco Inlet discharge data.

The Owasco Inlet (USGS Gauge, 4235299) mean, field-season, daily discharge of 5.1 m^3/s in 2019 indicates an "in between" year, smaller than the following "wet" years, rather than the 2011, 2015 and 2017 "wet" years and larger than the 2010, 2012, 2016 and 2018 "dry" years (Fig. 22). Similar variability was observed for the Owasco Outlet (USGS Gauge, 4235440, Fig. 22). 2013, 2014, 2019 were "in-between" years compared to the 2011, 2015 and 2017 "wet" years, and 2012, 2016 and 2018 "dry" years. The trends were consistent with annual and seasonal rainfall. Flow out the Owasco Outlet does not parallel the Inlet because the Outlet has additional mandates on flow besides rainfall that include stabilizing lake levels to the seasonal rating curve levels, decreasing downstream flooding and other concerns.
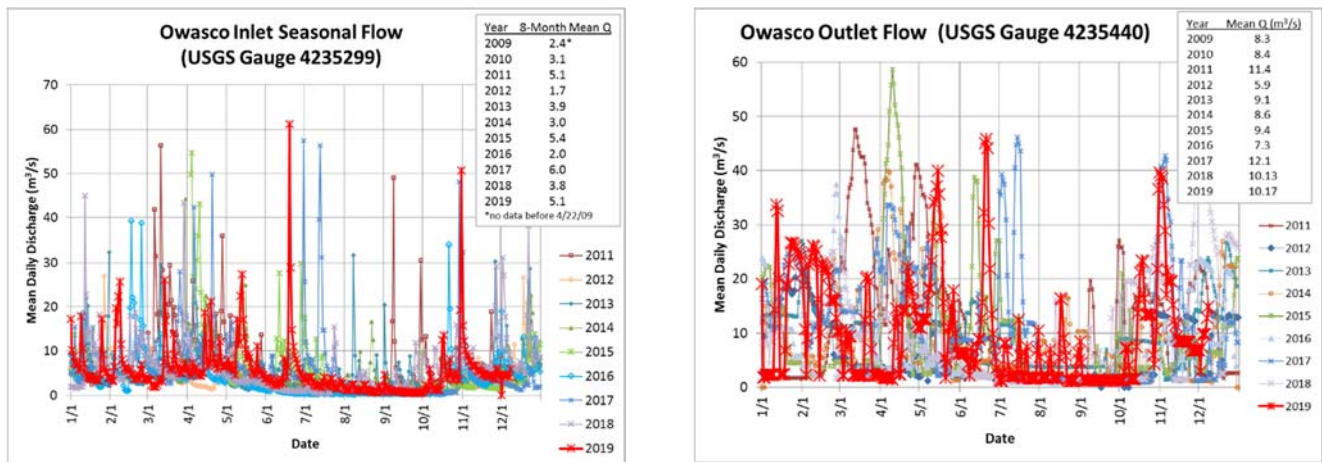
Fig. 22. 2018 annual average stream discharge for the Owasco Inlet near Moravia (left) – USGS Stream Gauge 4235299 and the Owasco Outlet (right) – USGS Stream Gauge 4235440.

Extreme Events: Both the Owasco Inlet and Dutch Hollow Brook 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, 2017 and

2019, and lower mean flows during 2010, 2012, 2013, 2014, 2016 and 2018 (Fig. 23). The top whisker in the B&W plot also revealed significantly larger events than the majority of the flow during 2011, 2014, 2015, 2017 and 2019 than the other years in the record as well. A similar pattern is observed at Dutch Hollow Brooks if the event during early July is included in the Dutch Hollow record (2017* Fig. 23). These events are critical because large events have an exponentially greater impact on nutrient and sediment loads to the lake. The largest events in 2019 were larger than the largest events in any other year. The largest dislodged the data logger deployment at Rt. 38A.
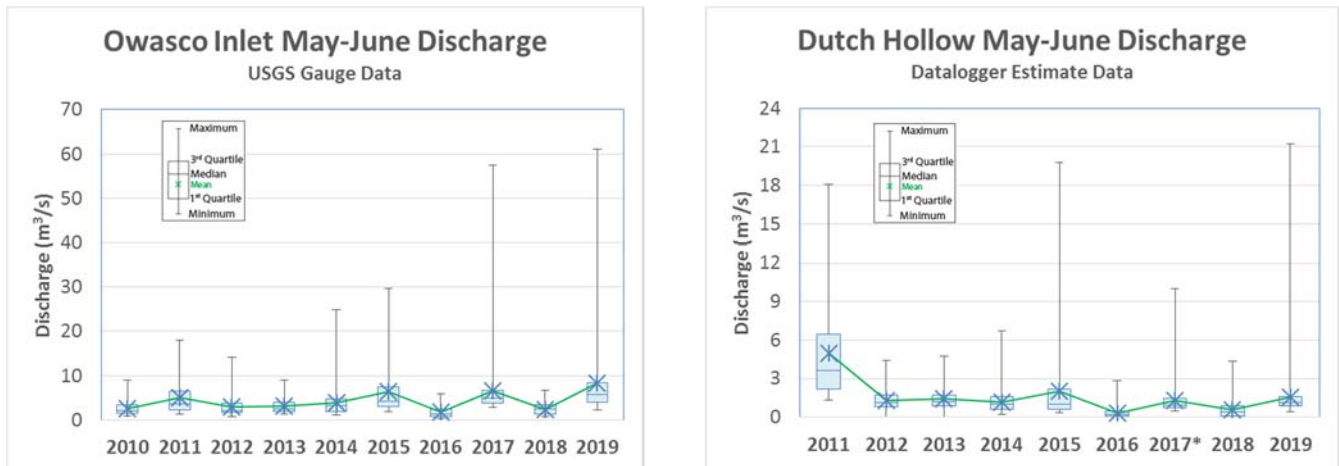

Fig. 23. Box and whisker plots of daily mean discharge during the May – June period over the past nine years for Owasco Inlet (left) near Moravia using the USGS Stream Gauge 4235299 data and Dutch Hollow Brook (right) using the Rt 38A data logger data.

Stream Nutrient & TSS Concentration Data: Total phosphate (TP) concentrations in 2019 ranged from 3 to 25 μg/L, and averaged 10 μg/L at Rt 38A in Dutch Hollow Brook, ranged from 5 to 13 μg/L, and averaged 9 μg/L at Rt 38 in Owasco Inlet and ranged from 7 to 17 μg/L, and averaged 12 μg/L at the two Fire Lane sites (Table 7 in appendix, Fig. 24). The ranges and means are similar to previous years.

Along Dutch Hollow Brook, the Benson tributary site revealed the largest annual mean TP, SRP, nitrate, and specific conductance (salinity) than the other sites in 2019, and revealed the smallest TSS concentration. For example, the SRP annual mean concentration were 7.1 μg/L at the Benson tributary site and the next largest concentration was 3.3 μg/L at the South Trib site and other sites in this basin were below 3.0 μg/L. Benson tributary consistently revealed larger annual mean concentrations of TP, SRP and specific conductance (salinity) since 2009 and suggests that agricultural inputs impact the tributary (Fig. 25). Its impact downstream was diluted by larger stream volumes downstream. It appears that the 2016 reduction of nutrients and suspended sediments concentrations at the Benson tributary site compared to other sites in the basin and a subsequent return to earlier values in 2017 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 TP, SRP, and nitrates.

2019 Total Phosphate Annual Average Concentration

2019 Dissolved Phosphate Annual Average Concentration

2019 Nitrates Annual Average Concentration

2019 Total Suspended Solids Annual Average Concentration

2019 Conductivity Annual Average Concentration
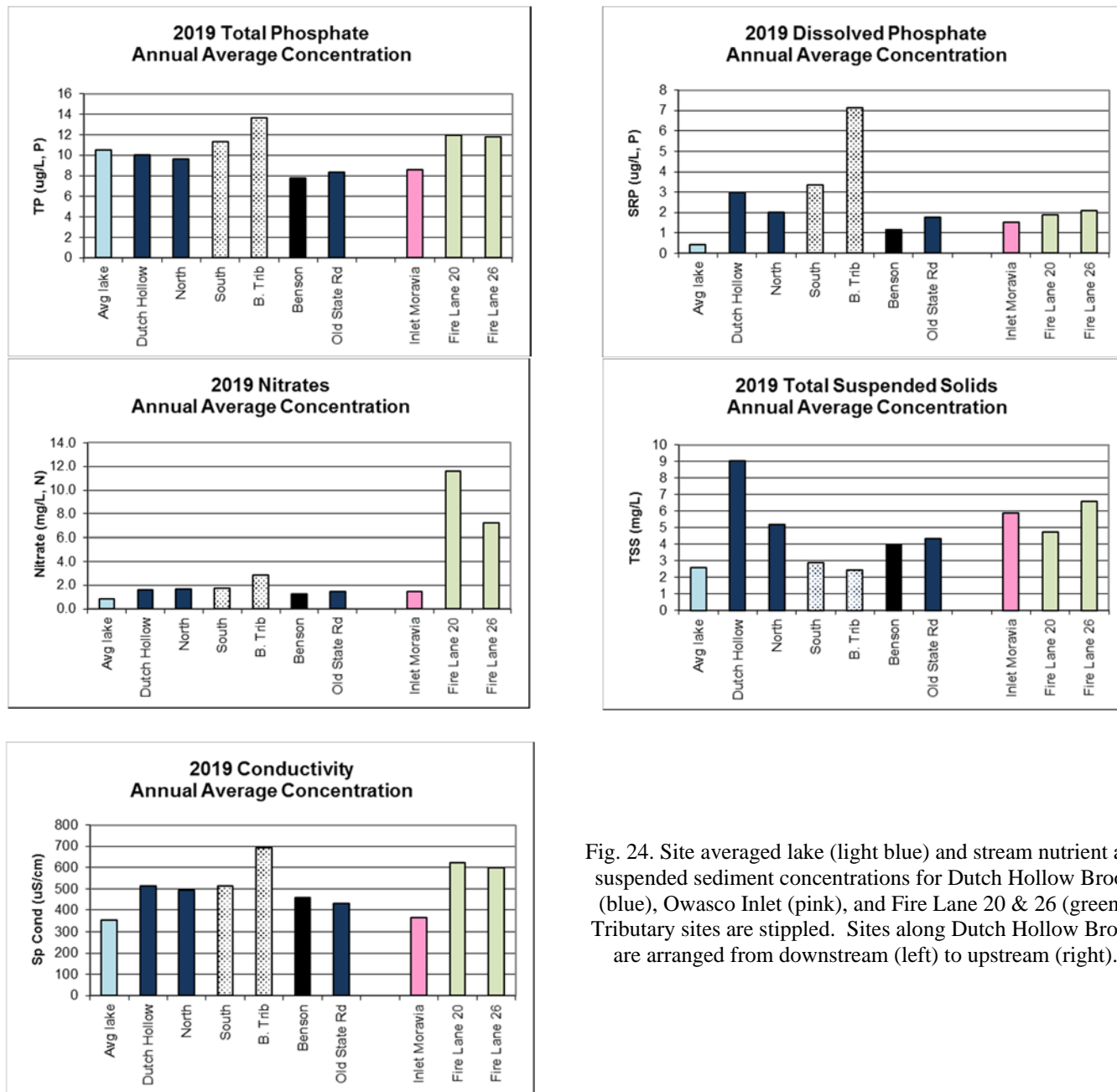
Fig. 24. Site averaged lake (light blue) and stream nutrient and suspended sediment concentrations for Dutch Hollow Brook (blue), Owasco Inlet (pink), and Fire Lane 20 & 26 (green). Tributary sites are stippled. Sites along Dutch Hollow Brook are arranged from downstream (left) to upstream (right).

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 as well. In contrast, significantly larger total phosphate and nitrate concentrations were detected 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, thus a more robust association cannot be concluded.

Total Phosphorus
Annual Average Concentration

Sp Conductance
Annual Average Concentration

Soluble Reactive Phosphorus
Annual Average Concentration
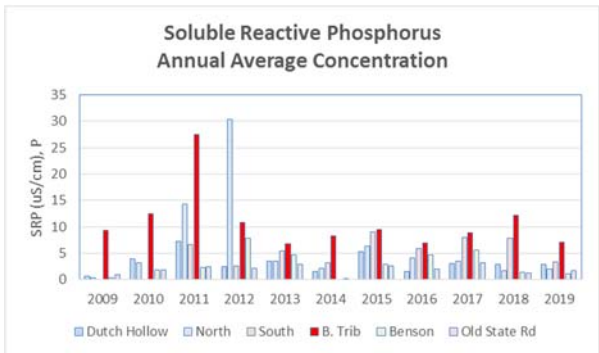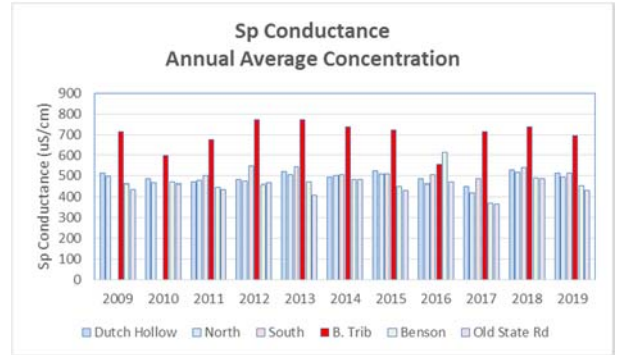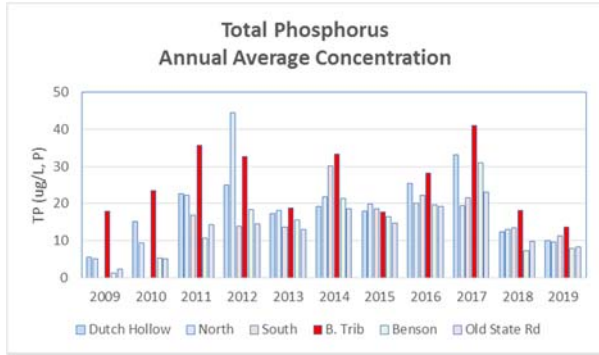
Fig. 25. Mean annual concentrations of total phosphorus, specific conductance and soluble reactive phosphorus at the Dutch Hollow Brook sites. Sites are arranged from downstream (left) to upstream (right).

Stream Fluxes: Dutch Hollow Brook revealed smaller mean fluxes of nutrients and sediments than Owasco Inlet (TP 0.5 vs. 3.3 kg/day; SRP 0.1 vs. 0.6 kg/day; TSS 402 vs. 512 kg/day; N 80 vs. 548 kg/day, respectively, Fig. 26). 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. Therefore, fluxes to Owasco Lake are sensitive to discharge and basin size. The annual mean fluxes measured in 2019 based on the four spring to early summer grab samples were similar to those detected in previous "normal" years (Fig. 27).

2019 Total Phosphate
Annual Average Flux

2019 Soluble Reactive Phosphate
Annual Average Flux
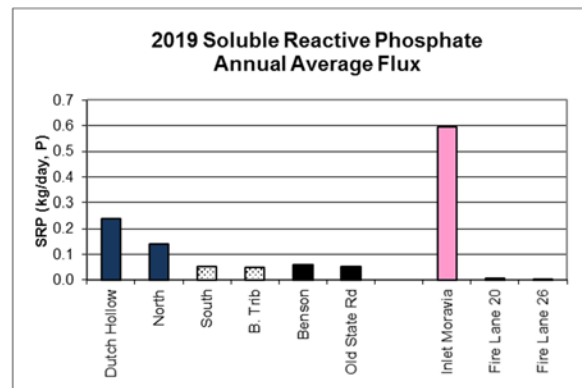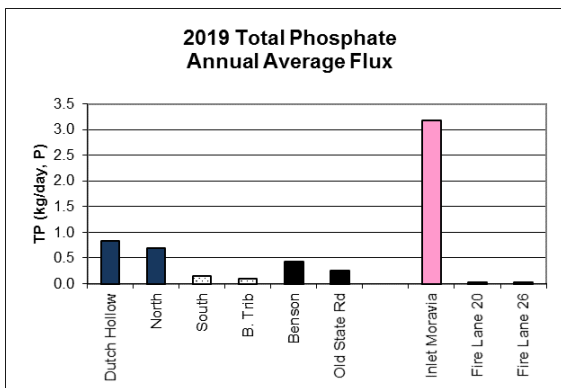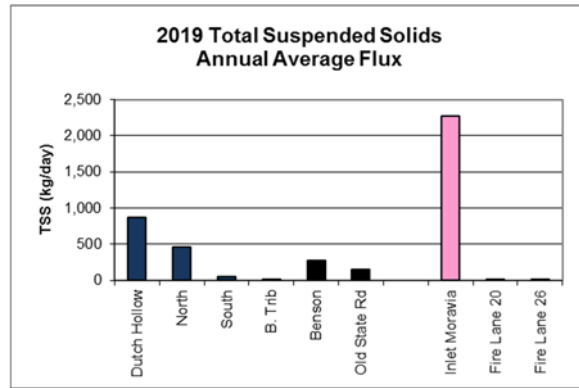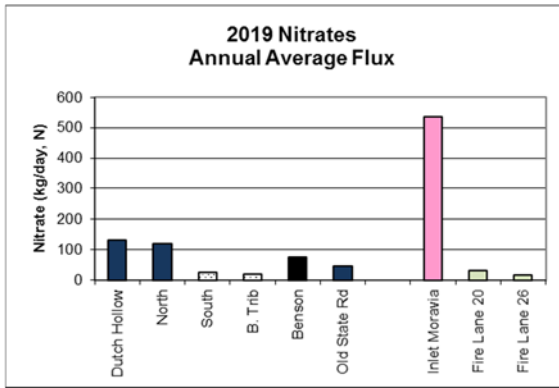
Fig. 26. Site averaged nutrient and sediment fluxes for Dutch Hollow (blue), Owasco Inlet (orange), and Fire Lane 20 & 26 (green). Tributary sites along Dutch Hollow Brook are stippled. Sites along Dutch Hollow Brook are arranged from downstream (left) to upstream (right).

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 small fluxes parallel the smaller discharges at these sites and smaller drainage areas. 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. 1) 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).
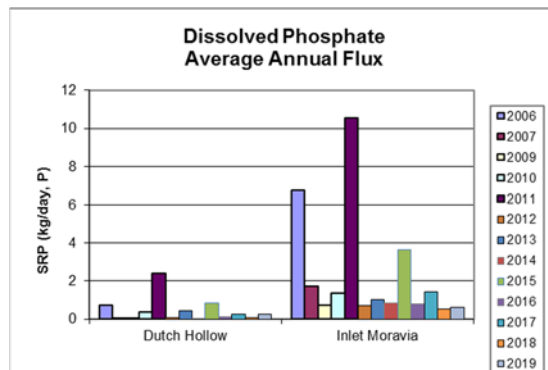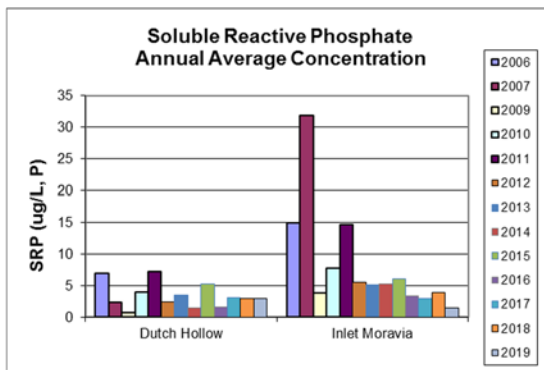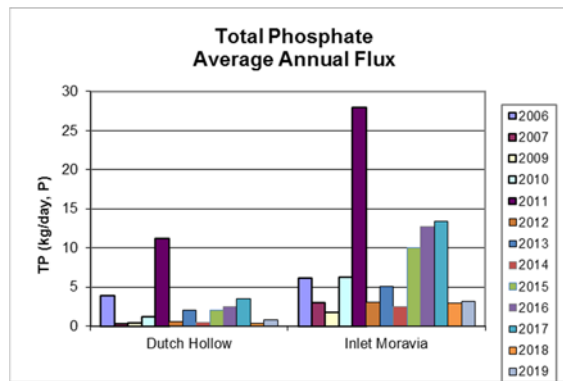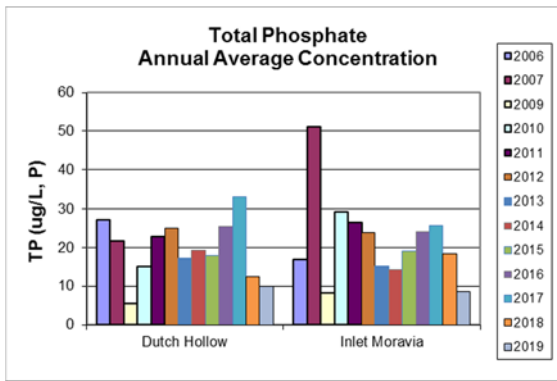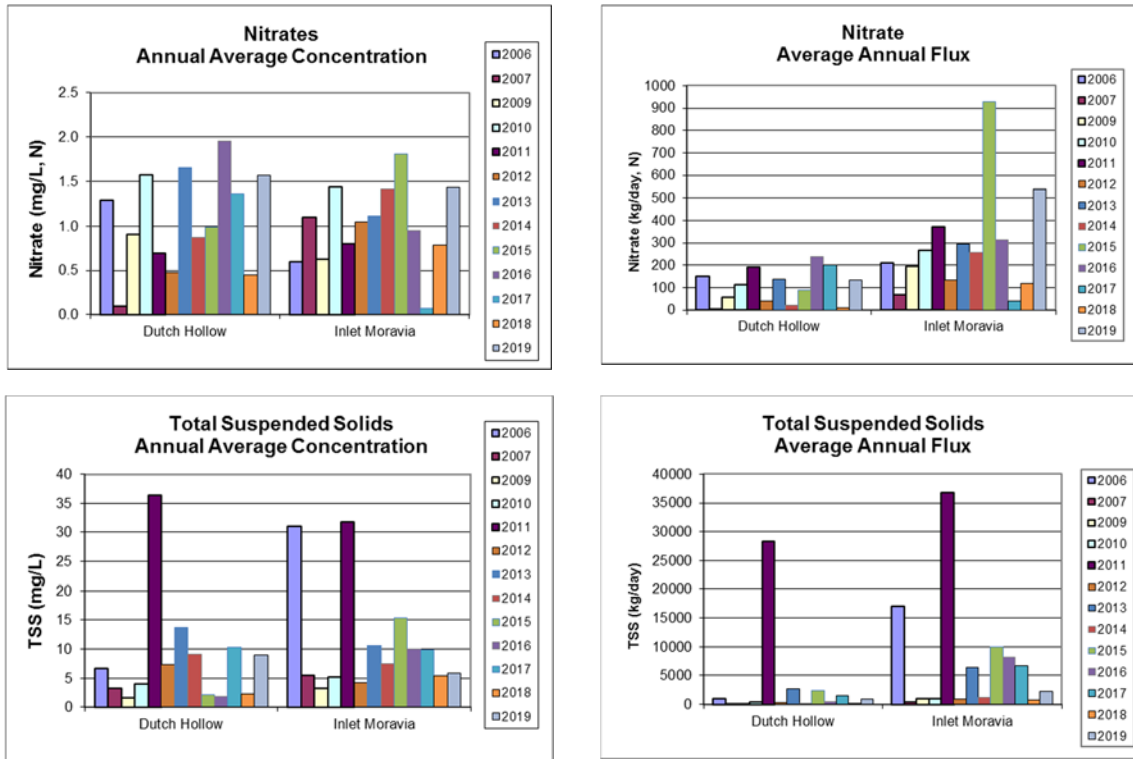
Fig. 27. 2006 through 2019 Annual average stream grab sample concentrations (left) and fluxes (right).

As in previous years, no site along 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, drainage tiles, 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 total phosphate contribution to the Owasco Inlet by Groton wastewater treatment facility has been significantly smaller since 2007 when the DEC mandated its P load reduction in the facility's effluent (Fig. 28). The load contributed by the Moravia WWTF was also very low. Since 2010, both facilities averaged ~0.1 to 0.3 kg/day.
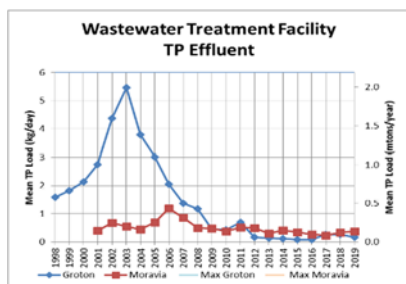
Fig. 28. Phosphorus loads from the Moravia and Groton wastewater treatment facilities.

EVENT SAMPLING ALONG DUTCH HOLLOW BROOK

Detailed Stage Data @ 38A along Dutch Hollow Brook: The 2019 stage data revealed textbook responses to precipitation events superimposed on a subdued but gradual spring through mid-summer decline in stage (Fig. 29). Each increase in stage corresponded to a precipitation event. Changes in stream height measured by the loggers ranged from 5 to more than 100 cm above the preceding 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. Precipitation events influenced changes in water temperature as well (Fig. 30). More events were detected in 2019 than 2018, and 2019 was more similar to the earlier "normal to wet" years, especially during the spring and early summer. Discharge did not decrease as much in 2019 as earlier years, in that, Dutch Hollow Brook did not decline to nearly dry conditions as in previous years, perhaps reflecting more rain events throughout 2019 (and perhaps a decline in agricultural withdrawals in 2019).
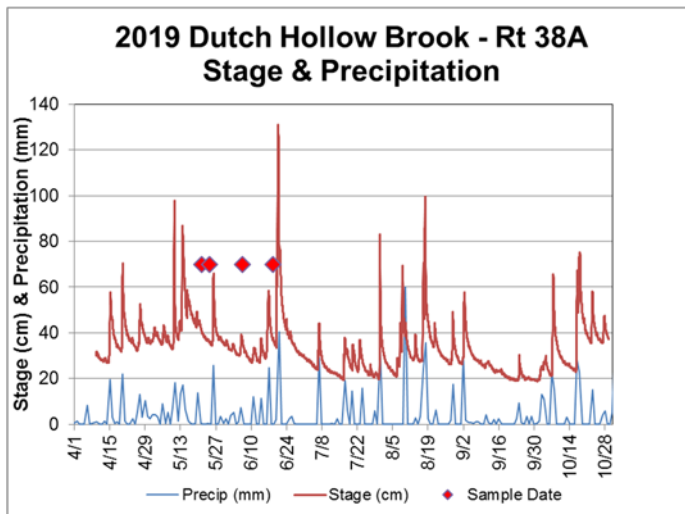
Fig. 29. Dutch Hollow Brook estimated discharge, precipitation, stream sample dates and measured discharge data for 2018 at Rt 38A. Precipitation data was from NY-CY-8, a CoCoRaHS station.
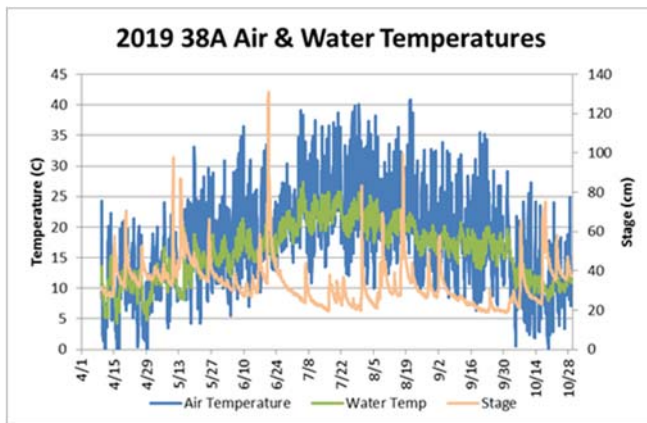
Fig. 30. Data logger mean stage, air and water temperature data at 38A.

Detailed "Event vs Base Flow" Results @ 38A: Nutrient and sediment concentrations increased markedly during precipitation events in 2019 like earlier years (Fig. 31). Total suspended sediments (TSS) increased from an average base flow concentration of ~10 mg/L to an average event flow concentration of 315 mg/L, and rose to an event maximum of 1,338 mg/L on 6/21.

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 the peak-flow runoff events transported significantly more soil particles than base flow and had a greater impact on water quality in the stream. It highlights the importance of large events. The concentrations measured for the 2019 events were in between those of events in previous years, and parallels the 2019 in between rainfall amounts.

Total (TP) and soluble reactive phosphorus (SRP) revealed event responses as well. Mean TP and SRP event concentrations were significantly larger than base flow concentrations, increasing from base flow means of 2 and 14 µg/L to event means of 10 and 29 µg/L, respectively. Maximum event concentrations were 81 µg/L for TP and SRP. Again, 2019 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 indicates that drain tiles are an important source of SRP as well. Tiles increase the release of dissolved and particulate phosphorus from the soils. Drain tiles and the ditches that tiles drain into should be mapped, sampled and remediated in the watershed. Drone flown infrared photography might differentiate drain tile locations, and LiDAR elevations.
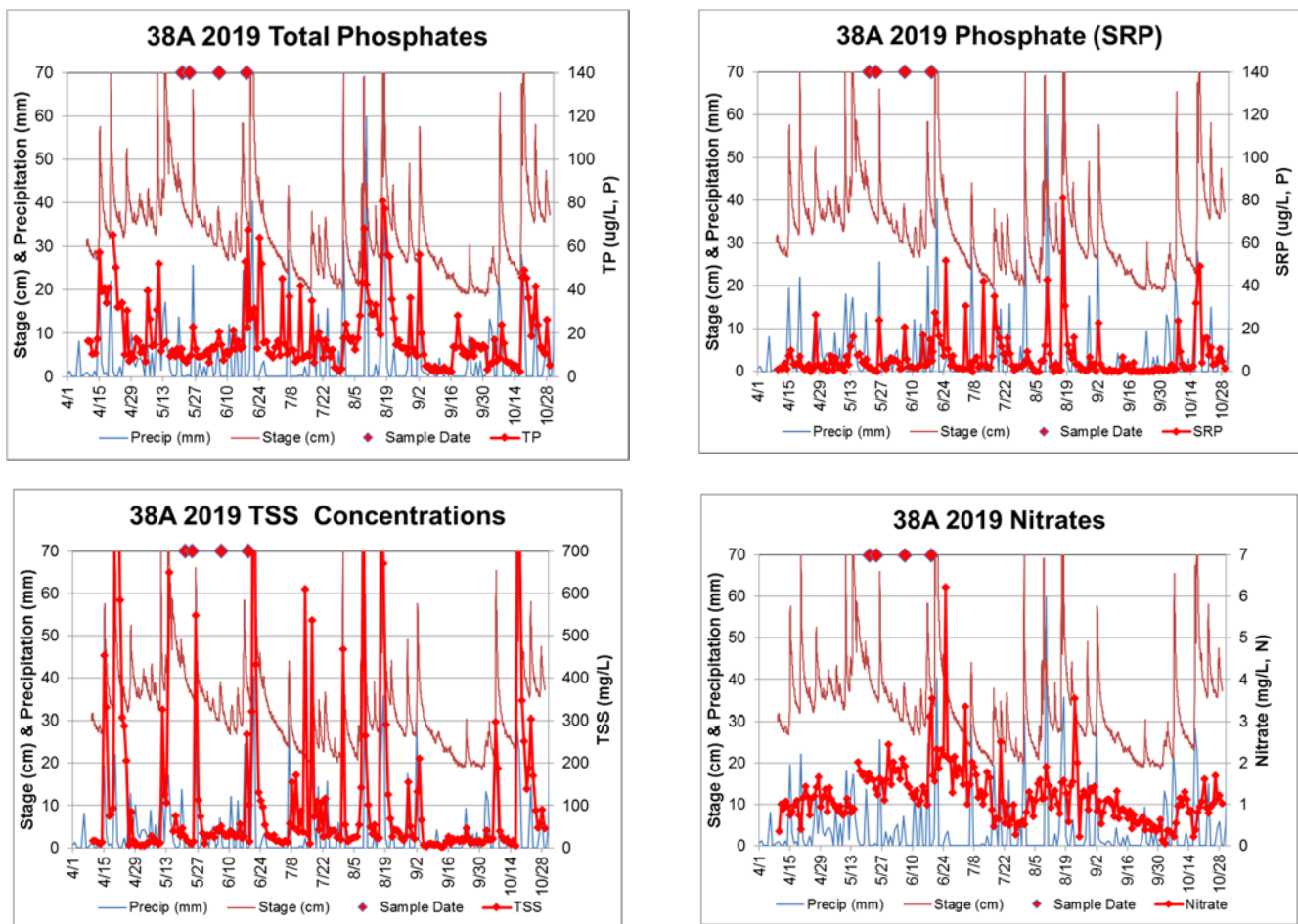
Fig. 31. 2019 daily nutrient and suspended sediment concentrations at Rt 38A.

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 more 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 reduce the runoff velocity and allows particles with attached phosphorus to settle out before entering the stream. Installation of gully plugs, vegetation strips, layers of wood chips 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. These infield practices are also hampered by drain tiles which bypass ground surface remediation practices, and suggests more settling structures in ditches and other locations where drain tiles discharge effluent. It's good that more farmers have recently adopted more BMPs in the watershed (Cayuga County Soil & Water communication).

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 6.2 mg/L, only slight larger than the mean base flow concentration of 0.9 mg/L, and a much smaller difference than those observed in the TSS, TP and SRP data. The increase to 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 at 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 ($r^2 = 0.93$). It established a stage/discharge rating curve for the site (Fig. 32).

The TSS, TP, SRP and N fluxes were clearly event driven (Table 8, Fig. 33 & 34). In 2019, TSS, TP, SRP and N event *vs.* base flow fluxes at 38A averaged 34,200 *vs.* 330, 3.2 *vs.* 0.3, 1.6 *vs.* 0.1 kg/day and 150 *vs.* 29 kg/day, respectively. During the 2019 April through October deployment, Dutch Hollow provided 5,060,000 kg of sediment to the lake during events, but only 18,000 kg during base flow conditions. In a similar light, the 2019 events delivered 467 kg of TP, 323 kg of SRP and 22,230 kg of N to the lake compared to base flow contributions of 17 kg of TP, 4 kg of SRP and 1,600 kg of N. The data indicate that typically over

Fig. 32. Best fit correlations between weekly to bi-weekly measured discharge and data logger stage data at 38A site.
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90% of the suspended sediments, total phosphorus, soluble reactive phosphate and nitrates were delivered to Owasco Lake by Dutch Hollow Brook during events over the course of this monitoring program (Table 8).

Annual changes were also observed. When TSS and TP are plotted against seasonal rainfall, the correlations explained ~60% of the variance (Fig. 35 & 36). It suggests that other variables, for example, sediment storage, extent of cover crops, and nutrient management must come into play. The extremely large nitrate flux in 2016 is due to a season ending precipitation event and exceedingly large nitrate concentrations.
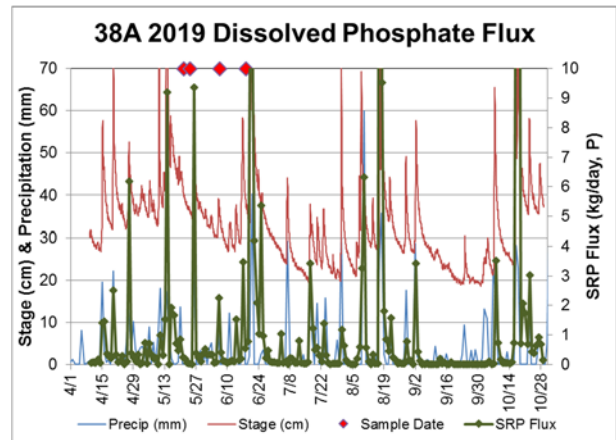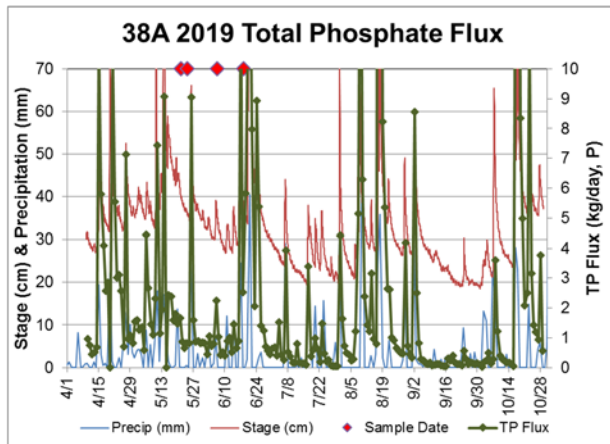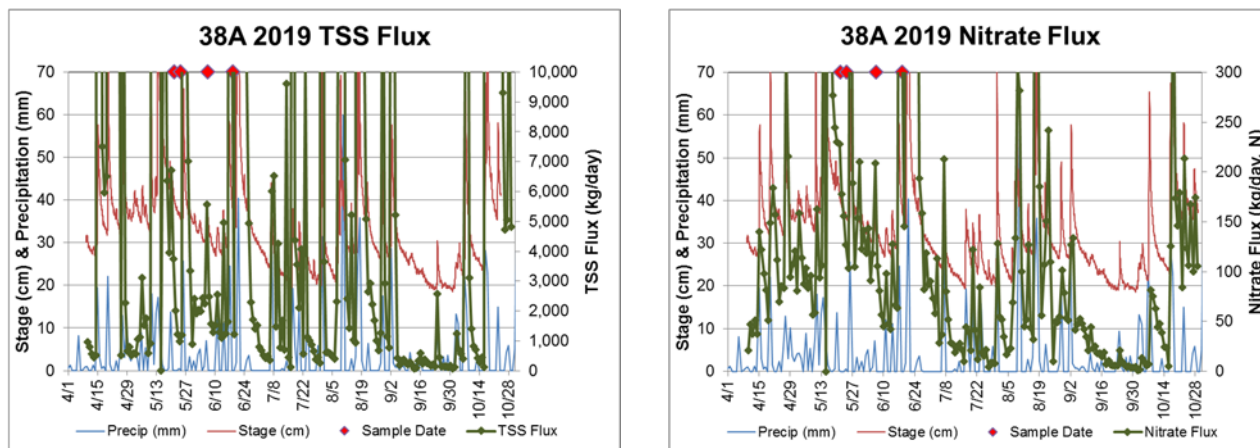
Fig. 33.   Autosampler nutrient and suspended sediment fluxes.

The nutrient/rainfall relationship is 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.

The event *versus* base flow data also indicate that grab samples underestimated annual fluxes from a stream. For example, the 2019 autosampler estimated a mean sediment flux of 25,000 kg/day, total phosphates 2.4 kg/day, dissolved phosphates 1.2 kg/day, and nitrates 120 kg/day; whereas the grab sampling estimated an annual mean flux of 670 kg/day for sediments, 0.8 kg/day for total phosphates, 0.2 kg/day for soluble reactive phosphates, and 130 kg/day for nitrates. The grab samples estimates were smaller because these samples were 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 8: 2011 – 2019 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**

	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%
2019 (4/10-10/29)	**TSS**	**Nitrate**	**TP**	**SRP**
Mean (kg/day)	25,018	117	2.4	1.2
Event (kg/day)	34,191	150	3.2	1.6
Base Flow (kg/day)	331	29	0.3	0.1
% by events	99%	93%	97%	98%
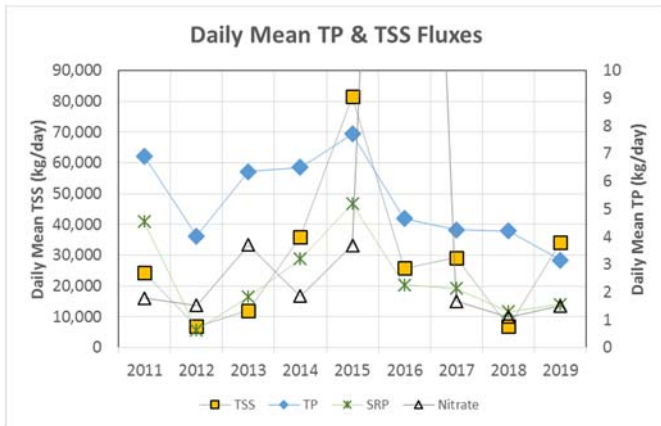
Fig. 34. Estimated mean daily total suspended solids, total phosphorus, soluble reactive phosphorus and nitrate (N/100) loads at Dutch Hollow Brook since 2011.
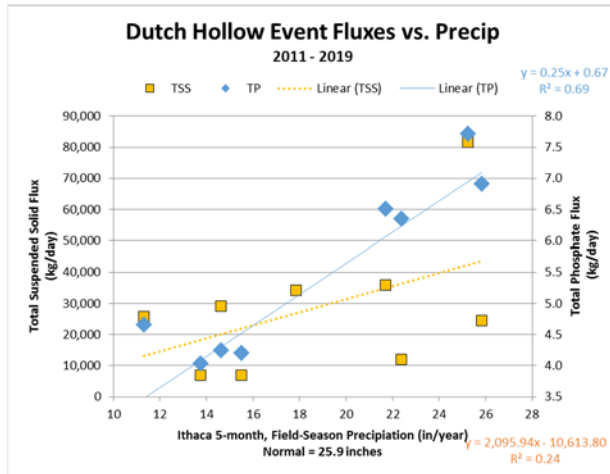
Fig. 35. Estimated annual total phosphorus loads vs May-September (5-month) rainfall at the Ithaca Airport.
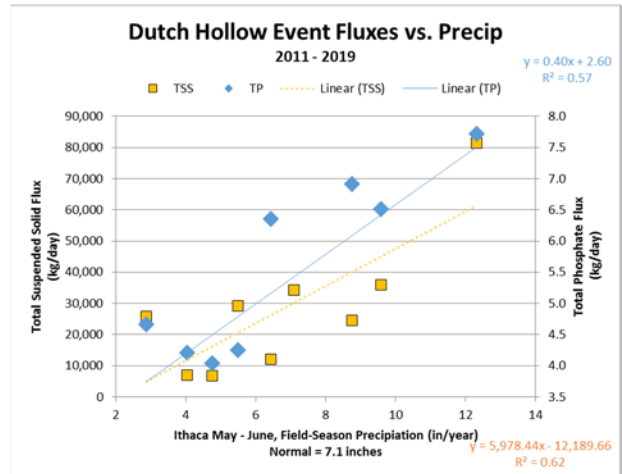
Fig. 36. Estimated annual total phosphorus loads vs May-June (3-month) rainfall at the Ithaca Airport.

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 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. 37). 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 decade or more (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.
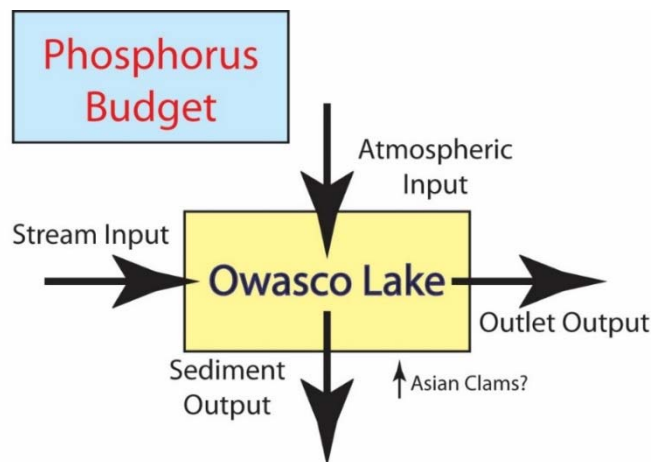
Fig. 37. The Owasco Lake phosphorus budget: Sources and sinks for phosphorus in Owasco Lake. Water quality improves if **inputs are less than outputs, for a number of years**.

The Inputs: The Moravia and Groton WWTF added a combined 0.2 metric tons P to the Owasco Inlet in 2019[8]. The detailed 38A autosampler data calculated a mean total phosphate flux of 3.8 kg/day from Dutch Hollow Brook in 2019. Owasco Inlet delivered 3.2 kg/day based on the available 2019 stream grab data. The 2019 load from Owasco Inlet was estimated at 12.9 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 7.7 metric tons of phosphorus from every stream to the lake in 2019. 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 nutrient 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 2019 influx of phosphorus is estimated at 6.4 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 4.4 metric tons of phosphorus escaped out the Outlet in 2019 assuming a 2019 annual mean total phosphate concentration in the lake of 13.6 μg/L, and a 2019 mean daily discharge of 10.2 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 2019 efflux of phosphorus is estimated at 7.2 metric tons/year.

The Net Flux: Owasco Lake thus lost approximately 0.8 metric tons of phosphorus in 2019. Since 2011, the lake gained phosphorus during five years and lost phosphorus during four years (Fig. 38). The mean annual input was 9.0 mtons/year, 2.1 metric tons/year more than the mean output of 3.9 mtons/year. More importantly, the last four years registered net P losses. The recent, multi-year, negative net-P flux suggests that the remediation efforts in the watershed were working, but the reduction apparently was not sufficient enough and/or spanned enough time to significantly improve water quality in the lake.

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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.
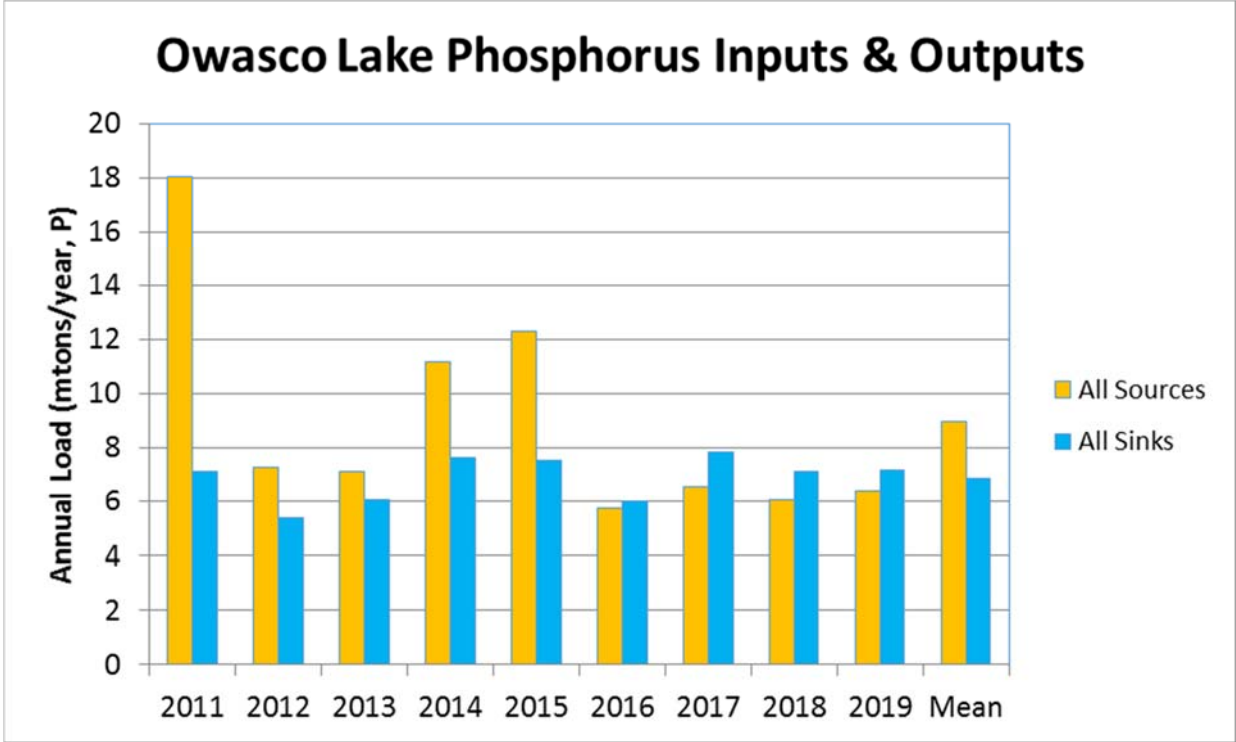
Fig. 38. Estimated annual total phosphorus inputs and outputs for Owasco Lake.
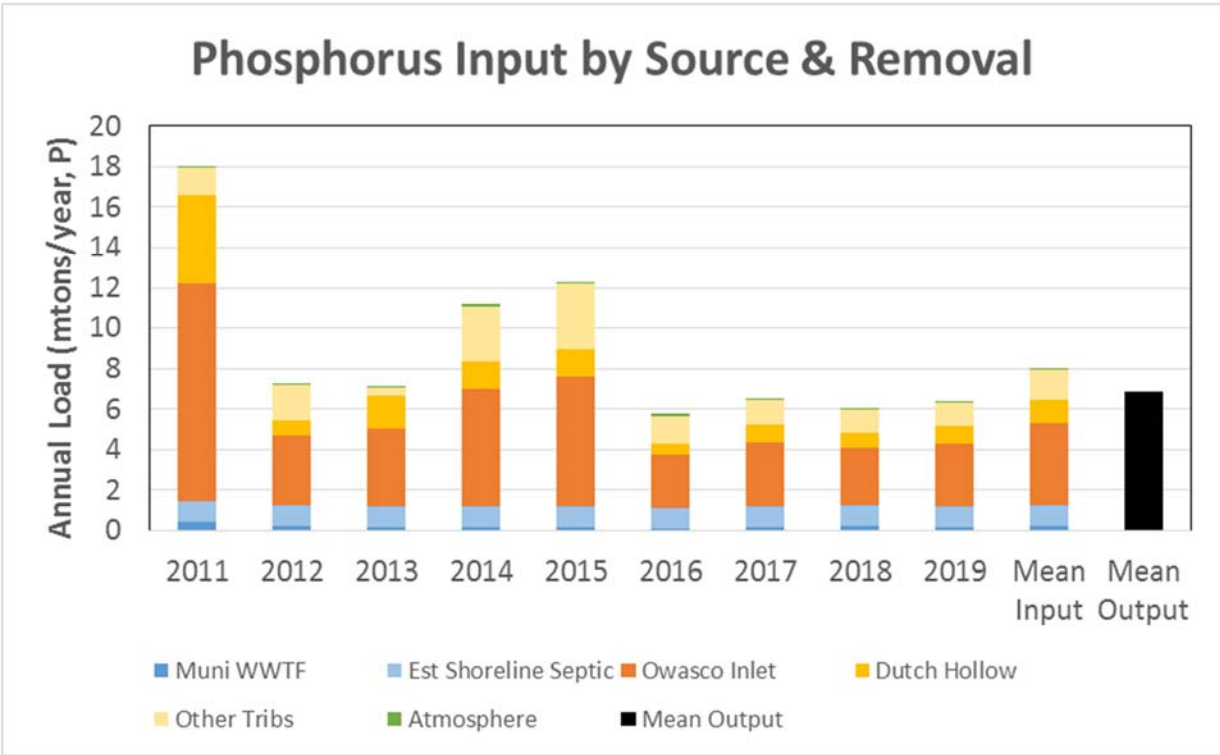
Fig. 39. Estimated annual inputs of phosphorus by source for Owasco Lake.

The contribution from stream sources changed from year to year, whereas the inputs from other sources were relatively constant over the past nine years (Fig. 39). For example the annual contribution from the Owasco Inlet, excluding the wastewater treatment facilities (WWTFs), ranged from 3 to 10 metric tons since 2011. The variability reflects changes in precipitation intensity, seasonality and totals. In contrast, the two 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% and averaging 83% of the total 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.

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 again in 2016, 2018 and 2019 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.
- Nutrient and sediment loads were near normal in 2019, and related to the near "normal" rainfall.
- 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:
- Details on BGA blooms and the associated HABs are contained in a companion report that also 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 flood events in 2011, 2014, 2015, 2017 and 2019.
- 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.
- Segment analysis did not identify significant point sources along Dutch Hollow Brook.
- Both the Moravia and Groton municipal wastewater treatment facilities have done an amazing job keeping their annual phosphorus loads to a minimum.
- The event *versus* base flow analysis at Dutch Hollow Brook highlighted the dominance of events and associated runoff of nonpoint sources for the delivery of sediments and nutrients 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 phosphorus budget for Owasco Lake indicated that the lake neither gained nor lost significant amounts of phosphorus in 2019. However, the negative balance, i.e., inputs < outputs, over the past four years is encouraging, and suggests the remediation efforts in the watershed are beginning to work.
- 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 and Dutch Hollow Brook were always the largest and 2nd largest fluvial contributors. The stream inputs however proportionally vary from year to year, dependent primarily on the amount, intensity and seasonality of rainfall.
- Contributions of phosphorus 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 small but negative net P fluxes for the watershed over the past four years is encouraging but has yet to dictate a discernable improvement in the lake's water quality. It commands the immediate use of additional remediation strategies. The sooner they are installed the sooner the lake might return to its original oligotrophic state.
- 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, tile drains, 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 experienced a nutrient loading problem.
- Perhaps all human and farm 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.
- Alternatively, P-adsorption strategies should be utilized on roadside ditches, drainage tile effluent and manure before it is spread on nearby fields.
- Additional flood retention 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 as long as the vegetation does not die and decompose in the lake and thus release these sequestered nutrients back into the lake.
- Nutrients should also be physically removed from the lake, when feasible. For example, macrophytes should be harvested from the nearshore areas in the late summer and

disposed 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.

- o Owasco Lake is probably too large and the existing phosphorus concentration too small for phosphorus sequestration techniques 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, at the opening of drain tiles and manure before spreading to reduce phosphorus loads from these sources.
- o Bio-manipulation is a poor option because the recreation is too vital for the economy.
- o 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.

WEATHER STATION, WATER TEMPERATURE AND FLI SENSOR NODE QA/QC
The weather station wind speeds, wind directions and air temperatures were periodically (35 times, ~ 4 times at each site) compared to data from a Kestral 5000 pocket weather meter and a cell phone compass. Even though the handheld and compass data were not measured at exactly the same time as the weather station recordings, the mean deviation in wind speeds was 0.04 mph, mean deviation in wind direction was 1.3° and mean deviation in air temperature was 2.0°C. The linear correlation between different data sources were $r^2 = 0.51$, $r^2 = 0.88$, and $r^2 = 0.89$, respectively. The low correlation for wind speeds could be due to, in part, the variability in the data measured by the weather station as the mean range in weather station wind speeds between the three closest readings was 3.3 mph. Unfortunately, the FLI Sensor node was not operating properly to compare the dissolved oxygen sensor to the measured, bi-weekly, titration data. The available FLI Sensor Node water temperature data was compared to the adjacent TidBit or HOBO data logger deployed at two sites. The r^2 were 0.98 and 0.93, and mean deviation of the temperature readings were 0.04°C and 0.001°C. Finally, a comparison of TidBit and HOBO water temperature data deployed at the same site resulted in an r^2 of 0.99 and a mean deviation of 0.02°C. Unequal deployment depths probably influenced the mean difference in temperature due to the observed daily onset and decay of thermal stratification by up to 4°C at these sites.

Table 2. 2019 Lake Data.

2019 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	4.0	2.4	11.8	0.3	0.9	696	2.6
Buoy	4.0	--	--	--	--	--	--
2	4.1	2.7	9.1	0.6	0.8	743	2.9
A	--	3.2	8.8	0.3	0.9	772	2.4
C	--	2.1	9.4	0.6	0.8	672	1.9
D	--	2.7	10.7	0.3	0.8	667	2.0
G	--	2.0	8.8	0.6	0.8	664	2.2
E	--	1.7	8.8	0.5	0.7	666	1.8
F	--	1.8	9.0	0.7	0.8	670	1.9
Average	**4.0**	**2.3**	**9.6**	**0.5**	**0.8**	**694**	**2.2**

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	---	1.3	9.7	0.7	1.1	1328	0.7
Buoy	---	---	---	---	---	---	---
2	---	2.6	8.0	1.1	1.0	1489	0.6
Average	**---**	**1.9**	**8.9**	**0.9**	**1.1**	**1408**	**0.6**

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)
5/29/19	5.8	1.3	7.7	0.1	1.0	1061	0.1
6/10/19	3.5	1.1	14.1	0.4	1.1	1078	0.9
6/25/19	1.7	3.2	8.7	1.2	1.0	1139	1.4
7/9/19	5.6	7.8	11.4	0.4	1.2	958	0.2
7/16/19	5.3	1.6	16.0	1.1	1.2	786	0.9
7/23/19	4.6	2.1	9.2	0.0	0.9	615	3.5
7/30/19	4.0	2.4	12.5	0.1	0.4	527	5.7
8/6/19	2.2	3.8	7.5	0.8	0.6	542	3.9
8/13/19	3.2	2.1	11.4	0.4	0.5	610	2.8
8/20/19	2.3	3.4	9.7	1.5	0.7	517	4.8
8/27/19	2.2	2.8	10.4	0.2	0.7	452	4.2
9/3/19	5.0	1.4	14.7	0.0	0.8	543	3.5
9/10/19	5.4	1.2	10.8	0.0	1.2	616	2.7
9/17/19	4.9	2.8	4.5	0.0	0.7	681	1.4
9/24/19	4.5	1.7	8.4	0.0	0.6	668	4.7
Average	**4.0**	**2.6**	**10.5**	**0.4**	**0.8**	**720**	**2.7**

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	---	1.2	6.6	3.4	1.0	1158	0.0
6/5/18	---	0.9	12.3	0.4	1.2	1226	0.2
6/19/18	---	1.6	8.1	0.0	1.0	1280	0.1
7/3/18	---	7.4	14.7	0.2	1.3	1290	0.2
7/10/18	---	1.4	7.7	1.7	1.2	1304	0.2
7/24/18	---	2.6	5.9	0.4	1.3	1243	0.6
7/31/18	---	2.4	10.1	0.9	0.7	1276	0.9
8/7/18	---	2.9	3.7	1.7	0.9	1239	0.7
8/14/18	---	1.1	5.0	0.8	0.9	1439	1.1
8/21/18	---	1.1	21.3	0.0	1.3	1656	2.5
8/28/18	---	1.2	9.3	1.1	1.2	1659	0.6
9/4/18	---	1.0	10.6	1.4	1.1	1637	0.7
9/11/18	---	1.0	8.4	0.4	1.1	1572	0.5
9/18/18	---	2.4	3.4	0.7	0.8	1596	0.8
10/3/18	---	1.3	5.7	0.1	0.9	1553	0.6
Average	**---**	**1.9**	**8.9**	**0.9**	**1.1**	**1408.5**	**0.6**

Table 4. Annual Average Plankton Data from 2005 through 2019, and Daily Average Data for 2019.

Plankton Group			Diatoms					Dinoflagellates			Rotifers & Zooplankton					Blue Greens	
Plankton Species	Fragillaria %	Tabellaria %	Diatoma %	Asterionella %	Melosira %	Synedra %	Rhizoselenia %	Dinobryon %	Ceratium %	Coalcium %	Copepod %	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	0.0	0.9	2.5	3.2	10.3	2.8	0.0	0.3
2006 Average	24.3	1.7	0.0	7.1	1.4	0.7	2.6	41.5	0.7	0.0	0.2	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	0.0	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	0.3	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	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	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	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	0.3	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	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	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	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	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	2.0	2.7	4.0	2.3	4.9	5.6
2018 Average	1.2	0.1	6.9	24.7	0.0	33.1	0.1	0.5	0.0	0.1	0.2	2.0	0.4	0.4	0.2	1.2	1.5
5/29/19	7.7	0.3	9.7	9.2	0.3	0.0	0.0	0.0	0.0	0.0	38.0	0.0	0.3	0.0	0.0	0.0	0.0
6/10/19	8.1	0.0	13.3	12.1	0.7	0.0	0.0	0.0	0.0	0.0	26.1	0.0	13.4	0.0	0.0	0.0	0.0
7/9/19	43.8	0.0	1.0	44.7	0.0	1.3	0.0	0.3	0.1	0.0	0.0	0.3	0.1	0.0	0.7	0.0	0.0
7/16/19	48.1	0.0	0.3	37.4	0.0	0.6	5.1	0.3	0.0	0.0	0.0	0.3	0.0	0.0	0.3	0.0	0.2
7/23/19	64.8	0.0	0.0	11.4	0.0	0.0	1.6	0.0	0.0	0.0	0.3	0.3	0.0	0.0	0.0	0.0	0.0
7/30/19	47.2	0.0	0.0	19.5	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.7	0.0	5.5	0.0	14.1	0.0
8/6/19	67.3	0.0	0.0	24.6	0.0	0.0	0.0	0.0	0.0	0.0	0.3	1.3	0.3	0.3	0.0	0.6	0.3
8/13/19	66.9	0.0	0.0	19.0	0.0	0.0	0.0	2.4	0.0	3.9	0.9	2.1	0.9	0.0	0.0	0.0	0.0
8/20/19	89.9	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0	0.4	0.8	2.9	0.0	0.0	0.4	0.8
8/27/19	78.6	0.0	0.0	0.0	1.0	0.0	0.0	2.4	0.0	3.6	0.3	0.0	2.7	1.4	0.2	1.2	3.3
9/3/19	41.3	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	8.8	6.1	0.0	0.5	2.3	3.1	25.0
9/10/19	1.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	5.5	0.5	6.8	0.0	17.1	0.3	10.5	56.0
9/17/19	16.2	0.0	0.0	0.0	0.0	0.0	0.3	0.3	0.0	0.0	0.3	4.4	0.3	14.3	0.0	17.2	41.2
9/24/19	0.9	0.0	0.0	0.0	1.4	0.0	0.0	0.0	2.4	0.0	0.0	1.7	0.0	22.0	0.4	30.1	38.0
2019 Average	41.6	0.0	1.7	12.7	0.2	0.1	0.5	0.5	0.2	0.9	5.4	1.8	1.5	4.4	0.3	5.5	11.8

Note: Only included plankton from Offshore Sites with at least 2% of the total counts on any survey day, in any year.

Table 5: Annual Average 2019 Lake Data from the Finger Lake Survey.

2019 Average Values (± 1s)	Honeoye	Canandaigua	Keuka	Seneca	Cayuga	Owasco	Skaneateles	Otisco
Specific Conductance (µS/cm), Surface	286.5 ± 8.0	436.4 ± 11.8	351.8 ± 19.1	679.7 ± 36.6	449.9 ± 15.3	336.0 ± 18.3	304.0 ± 9.9	406.2 ± 18.3
Specific Conductance (µS/cm), Bottom	288.1 ± 9.5	455.0 ± 11.0	356.4 ± 9.0	718.0 ± 50.1	492.5 ± 10.0	362.8 ± 9.2	318.1 ± 14.4	438.7 ± 8.8
Secchi Depth (m)	3.3 ± 0.9	5.5 ± 1.5	4.6 ± 1.1	3.5 ± 0.8	3.3 ± 0.5	4.0 ± 1.5	7.4 ± 3.8	2.7 ± 1.0
Total Suspended Solids (mg/L), Surface	4.9 ± 3.5	1.2 ± 0.5	1.6 ± 0.7	1.5 ± 0.4	2.3 ± 0.7	2.1 ± 0.7	1.5 ± 0.7	3.0 ± 1.5
Total Suspended Solids (mg/L), Bottom	2.4 ± 1.0	0.7 ± 0.3	0.7 ± 0.3	0.7 ± 0.5	1.6 ± 0.6	1.5 ± 0.7	0.7 ± 0.5	2.0 ± 1.2
Total Phosphate (µg/L, TP), Surface	19.0 ± 13.0	9.7 ± 4.9	7.0 ± 4.2	8.9 ± 2.9	11.0 ± 6.1	9.1 ± 3.8	14.9 ± 26.8	13.9 ± 3.9
Total Phosphate (µg/L, TP), Bottom	18.6 ± 8.1	7.4 ± 4.0	7.6 ± 5.4	6.7 ± 2.2	17.6 ± 6.4	6.3 ± 1.4	6.8 ± 2.6	10.3 ± 3.0
Dissolved Phosphate (µg/L, SRP), Surface	1.5 ± 1.8	0.6 ± 0.9	0.4 ± 0.4	0.5 ± 0.7	0.4 ± 0.3	0.3 ± 0.3	0.9 ± 2.5	0.2 ± 0.3
Dissolved Phosphate (µg/L, SRP), Bottom	2.6 ± 4.6	0.9 ± 1.0	0.2 ± 0.3	2.1 ± 2.2	11.9 ± 5.6	0.9 ± 1.9	2.0 ± 2.4	0.2 ± 0.2
Nitrate as N (mg/L), Surface	0.0 ± 0.0	0.1 ± 0.1	0.1 ± 0.1	0.2 ± 0.1	1.1 ± 0.3	0.8 ± 0.3	0.4 ± 0.2	0.6 ± 0.2
Nitrate as N (mg/L), Bottom	0.0 ± 0.0	0.3 ± 0.3	0.2 ± 0.0	0.3 ± 0.1	1.2 ± 0.2	1.0 ± 0.2	0.5 ± 0.1	0.6 ± 0.2
Silica (SR µg/L), Surface	1188.2 ± 402.7	932.3 ± 55.9	951.6 ± 75.4	249.0 ± 105.7	147.6 ± 89.0	818.8 ± 250.2	505.9 ± 146.9	715.6 ± 281.8
Silica (SR µg/L), Bottom	1222.1 ± 432.4	1181.3 ± 152.7	1275.1 ± 108.8	389.9 ± 139.3	945.4 ± 183.9	1334.6 ± 178.2	808.0 ± 56.7	1289.0 ± 367.0
Chlorophyll a (µg/L), Surface	19.2 ± 18.9	1.5 ± 1.6	1.9 ± 1.6	2.6 ± 1.7	3.5 ± 1.8	2.5 ± 1.8	1.1 ± 0.7	3.9 ± 3.2
Chlorophyll a (µg/L), Bottom	5.5 ± 2.9	0.4 ± 0.2	0.3 ± 0.3	0.9 ± 1.3	0.3 ± 0.3	0.5 ± 0.4	0.2 ± 0.2	1.4 ± 1.0

Table 7. 2019 Stream Data.

2019 Stream Segment Analysis Data

Date & Location	Discharge (m³/s)	Specific Conductance (µS/cm)	Water Temp (°C)	Nitrate (mg/L, N)	Suspended Solids (mg/L)	Total Phosphate (µg/L, TP as P)	Phosphate SRP (µg/L, SRP as P)
5/21/2019							
Dutch Hollow 38A	1.18	470	14.4	1.2	27.1	6.0	1.1
Dutch Hollow North St	1.10	458	14.3	1.3	15.8	4.4	0.1
Dutch Hollow South Trib	0.20	481	13.3	1.3	7.7	7.5	1.0
Dutch Hollow Benson Trib	0.07	662	15.6	3.1	4.3	7.0	7.6
Dutch Hollow Benson Rd	0.92	421	14.3	1.1	9.3	3.5	0.0
Dutch Hollow Old State Rd	0.43	401	14.1	1.4	7.5	4.7	0.8
Owasco Inlet Moravia Rt 38*	5.52	255	14.8	1.1	6.6	6.0	1.8
Fire Lane 20	0.06	616	12.8	10.0	10.8	6.8	2.0
Fire Lane 26	0.05	597	12.8	6.8	4.8	13.7	1.8
5/24/2019							
Dutch Hollow 38A	1.02	503	14.4	1.3	1.6	10.0	0.3
Dutch Hollow North St	0.77	485	13.7	2.0	1.9	11.7	0.0
Dutch Hollow South Trib	0.12	501	12.7	1.9	1.2	10.2	1.1
Dutch Hollow Benson Trib	0.10	724	12.7	1.9	1.1	10.4	3.0
Dutch Hollow Benson Rd	0.59	453	13.0	1.3	2.6	7.5	0.2
Dutch Hollow Old State Rd	0.37	436	12.5	1.7	6.8	10.9	1.0
Owasco Inlet Moravia Rt 38*	4.11	388	13.0	1.9	7.6	4.6	1.1
Fire Lane 20	0.03	619	12.2	10.7	4.0	8.0	1.3
Fire Lane 26	0.02	597	12.0	6.8	3.1	9.3	1.1
6/6/2019							
Dutch Hollow 38A	0.88	540	17.3	1.6	5.7	16.7	2.5
Dutch Hollow North St	0.78	514	16.8	1.5	1.2	13.6	2.5
Dutch Hollow South Trib	0.15	524	15.8	1.4	0.8	14.9	0.9
Dutch Hollow Benson Trib	0.07	634	18.7	2.6	2.7	24.0	3.9
Dutch Hollow Benson Rd	0.61	491	16.7	1.4	2.0	11.8	0.5
Dutch Hollow Old State Rd	0.29	453	17.2	1.5	1.4	9.5	1.3
Owasco Inlet Moravia Rt 38*	3.99	424	17.6	1.6	7.3	12.6	0.8
Fire Lane 20	0.01	614	15.8	19.2	2.8	16.5	2.0
Fire Lane 26	0.03	608	15.1	6.2	18.5	8.9	1.1
6/18/2019							
Dutch Hollow 38A	0.90	546	17.5	2.1	1.7	7.2	8.0
Dutch Hollow North St	0.83	516	16.8	1.8	1.9	8.7	5.4
Dutch Hollow South Trib	0.18	547	15.9	2.5	1.8	12.5	10.3
Dutch Hollow Benson Trib	0.08	757	17.5	3.8	1.7	13.3	14.0
Dutch Hollow Benson Rd	0.61	463	16.8	1.4	2.0	8.3	3.9
Dutch Hollow Old State Rd	0.33	432	16.0	1.4	1.7	8.2	3.9
Owasco Inlet Moravia Rt 38*	4.11	395	17.5	1.2	2.0	11.0	2.4
Fire Lane 20	0.03	648	15.6	6.6	1.4	16.5	2.2
Fire Lane 26	0.01	599	16.0	9.0	0.0	15.4	4.3
Used USGS Gauge at Moravia for Discharge Data							
2019 Average Values							
Dutch Hollow 38A	1.00	514.75	15.90	1.57	9.03	9.99	2.97
Dutch Hollow North Rd	0.87	493.25	15.40	1.64	5.20	9.60	2.00
Dutch Hollow South Trib	0.16	513.25	14.43	1.76	2.88	11.28	3.33
Dutch Hollow Benson Trib	0.08	694.25	16.13	2.86	2.45	13.66	7.11
Dutch Hollow Benson Rd	0.68	457.00	15.20	1.28	3.98	7.78	1.15
Dutch Hollow Old State Rd	0.36	430.50	14.95	1.49	4.35	8.33	1.76
Owasco Inlet Rt 38 Moravia	4.43	365.50	15.73	1.43	5.88	8.54	1.53
Fire Lane 20	0.03	624.25	14.10	11.62	4.75	11.96	1.87
Fire Lane 26	0.03	600.25	13.98	7.20	6.60	11.82	2.08
2019 Average Fluxes				N kg/day	TSS kg/day	TP kg/day	SRP kg/day
Dutch Hollow 38A				135.3	777.4	0.9	0.3
Dutch Hollow North Rd				123.3	390.3	0.7	0.1
Dutch Hollow South Trib				24.8	40.5	0.2	0.0
Dutch Hollow Benson Trib				75.5	234.7	0.5	0.1
Dutch Hollow Benson Rd				20.0	17.1	0.1	0.0
Dutch Hollow Old State Rd				45.7	133.8	0.3	0.1
Owasco Inlet Rt 38 Moravia				548.3	2249.1	3.3	0.6
Fire Lane 20				34.2	14.0	0.0	0.0
Fire Lane 26				16.7	15.3	0.0	0.0