THE 2020 WATER QUALITY MONITORING REPORT, OWASCO LAKE, NY.

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12/30/2020

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 ongoing water quality 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 cyanobacteria blooms 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, the Finger Lakes – Lake Ontario Watershed Protection Alliance and most notably the Cayuga County Legislature. Additional funds to hire summer research students came from the Provost's Office and the Finger Lakes Institute at Hobart & William Smith Colleges. 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, based on open lake, dissolved nutrient (soluble reactive phosphate and nitrate) concentrations in the lake. Additional inputs of phosphorus stimulates additional algal growth and degrades water quality.
- The lake has experienced late-summer / early fall blooms of cyanobacteria (blue-green algae, BGA). Cyanobacteria 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 cyanobacteria may produce toxins (Harmful Algal Blooms, HABs) 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, more importantly, 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 Municipal Wastewater Treatment Facility has significantly reduced nutrient loading to the Owasco Inlet and thus Owasco Lake.
- Updated regulations to improve water quality and agricultural best management practices in the watershed were recently highlighted in the Owasco Lake Watershed Rules and Regulations revisions undertaken by the Owasco Lake Watershed Management Council on behalf of the City of Auburn and the Town of Owasco; a collaborative effort by numerous state, county and local groups and other stakeholders within the watershed. (see: Owasco-Watershed-Rules-and-Regulations)
- Streams and tributaries are the primary source of nutrients and sediments to the lake, especially during "wet" years but also "dry" years.
- Daily nutrient and sediment loads measured near the terminus of Dutch Hollow Brook revealed that over 90% of the loads are delivered during precipitation/runoff events, especially in the spring season.
- Annual suspended sediment, total and soluble reactive phosphate load estimates for Dutch Hollow Brook positively correlated to precipitation totals, especially precipitation during the spring season.
- The large nutrient and sediment inputs during 2011, 2014, and 2015 were coincident with and probably "triggered" the onset of the recent cyanobacteria blooms[1]. Even though coincidence does not prove causation, these excessive loads were unique over the past decade and coincident with the first bloom sightings.
- Since 2011, estimated 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. Since 2016, the balance has turned and inputs have become similar or smaller than outputs.
- Phosphorus loading reductions must continue to significantly improve water quality in Owasco Lake. This effort must be intensified because if all loads were significantly curtailed today, it would still take a minimum of five water retention times, i.e., approximately a decade or two, for the lake to naturally cleanse itself of excess phosphorus and improve water quality. Phosphorus stored within the sediments will take longer to flush out.

Water quality research has moved into an exciting phase. Cayuga County Planning will complete the EPA Nine Key Elements Plan for Owasco Lake/Watershed soon. The DEC's Finger Lakes HUB stimulated additional monitoring of the lake and streams. For example, DEC contracted with the USGS to deploy a water quality buoy in the lake and supported renewed and expanded C-SLAP and winter season sampling. Finally, the revision of the 1984 Owasco Lake Watershed Rules and Regulations was approved by local governments this past fall and is now waiting state approval.

[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

This report highlights our 2020 monitoring efforts within the Owasco Lake watershed, including water quality analyses of the lake and selected tributaries.

METHODS

The 2020 offshore lake and stream sample sites and field/laboratory methods were similar to the 2005 – 2019 programs.

Owasco Lake: The 2020 lake monitoring program sampled Sites 1 and 2 eleven times, monthly from late May to August, and then weekly through the end of 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 previous surveys of Owasco Lake.

The lake-monitoring field methods were similar to the earlier monitoring efforts. A CTD 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, $\mu E/cm^2$-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, and downcast profiles are utilized in these reports. The CTD pump was inoperative in 2020 until its replacement on 9/2. Without a pump, conductivity and dissolved oxygen data occasionally revealed spikes, especially on the upcast. These spikes should be ignored. The pump's replacement was delayed by COVID closure of the factory. 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) and alkalinity (mg/L, $CaCO_3$) using hand-held probes and field titration kits, and analyzed back in the laboratory for total phosphate (TP, μg/L, P), soluble reactive phosphate (SRP, μg/L, P), nitrate (NOx, mg/L, N), chlorophyll-a (μg/L) and total suspended solid (TSS, mg/L) concentrations. Rather than collect bbe FluoroProbe profiles in the field, surface and bottom water grab samples were analyzed by FluoroProbe in the lab to differentiate four different algal groups and yellow substances based on their accessory pigments to distinguish the relative concentrations of: 'green' algae (Chlorophyta and Euglenophyta), 'brown' algae (diatoms: Baccillariophyta, Chyrsophyta, and Dinophyta), 'blue-green' algae (Cyanophyta), and 'red' algae (Cryptophyta). Dissolved silica was not determined in 2020 due to COVID induced staff reductions.

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
FLI Buoy Site	42° 50.35' N	76° 30.85' W	49 m
Nearshore Sites:			
Martin Pt North	42° 53.64' N	76° 31.59' W	dockside
Martin Pt South	42° 53.31' N	76° 31.48' W	dockside
Burtis Pt	42° 51.89' N	76° 30.96' W	dockside
Fire Lane 20	42° 48.69' N	76° 30.92' W	dockside

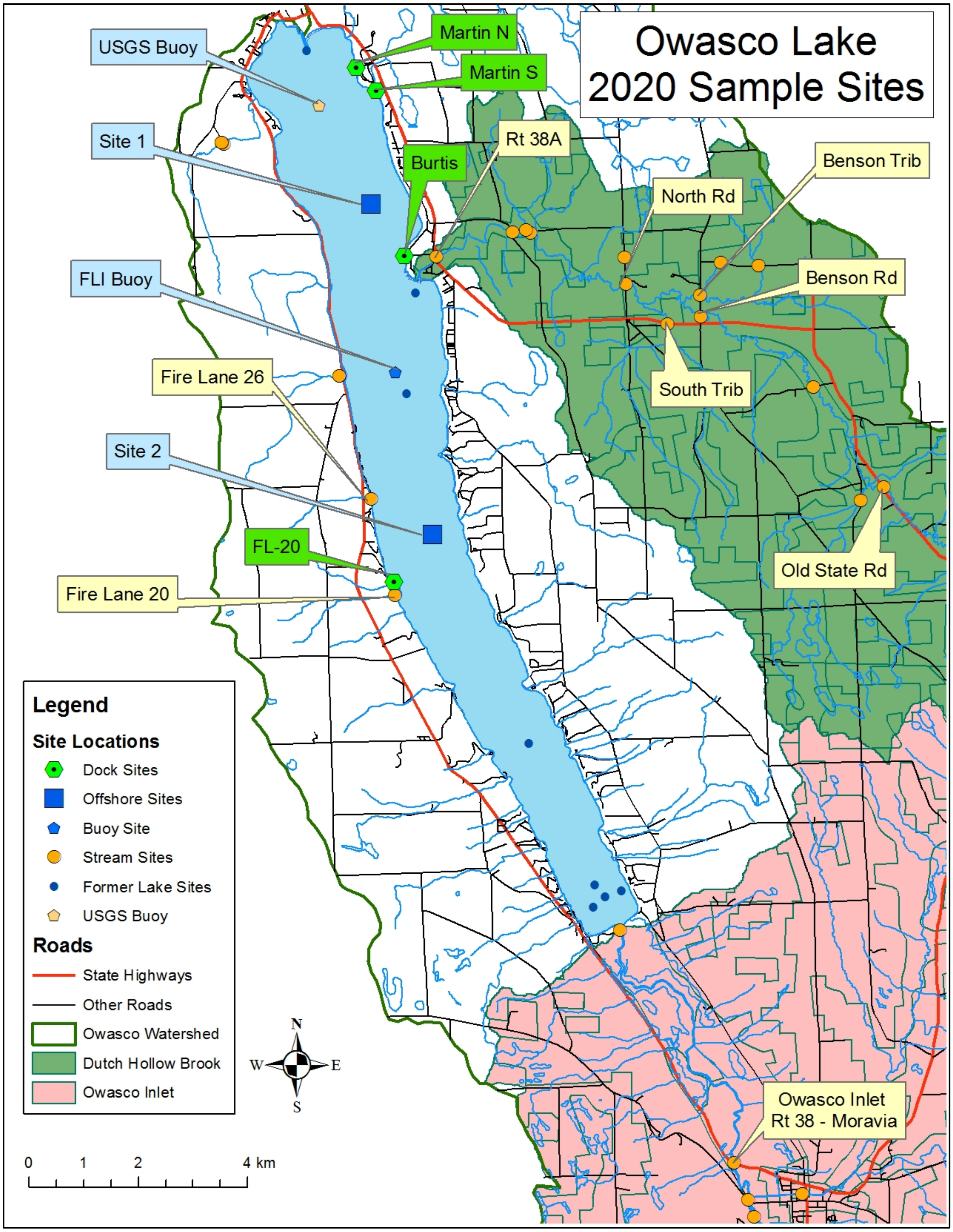

Fig. 1. The lake monitoring (blue), buoy (blue), dock (green) and stream (yellow) sites.

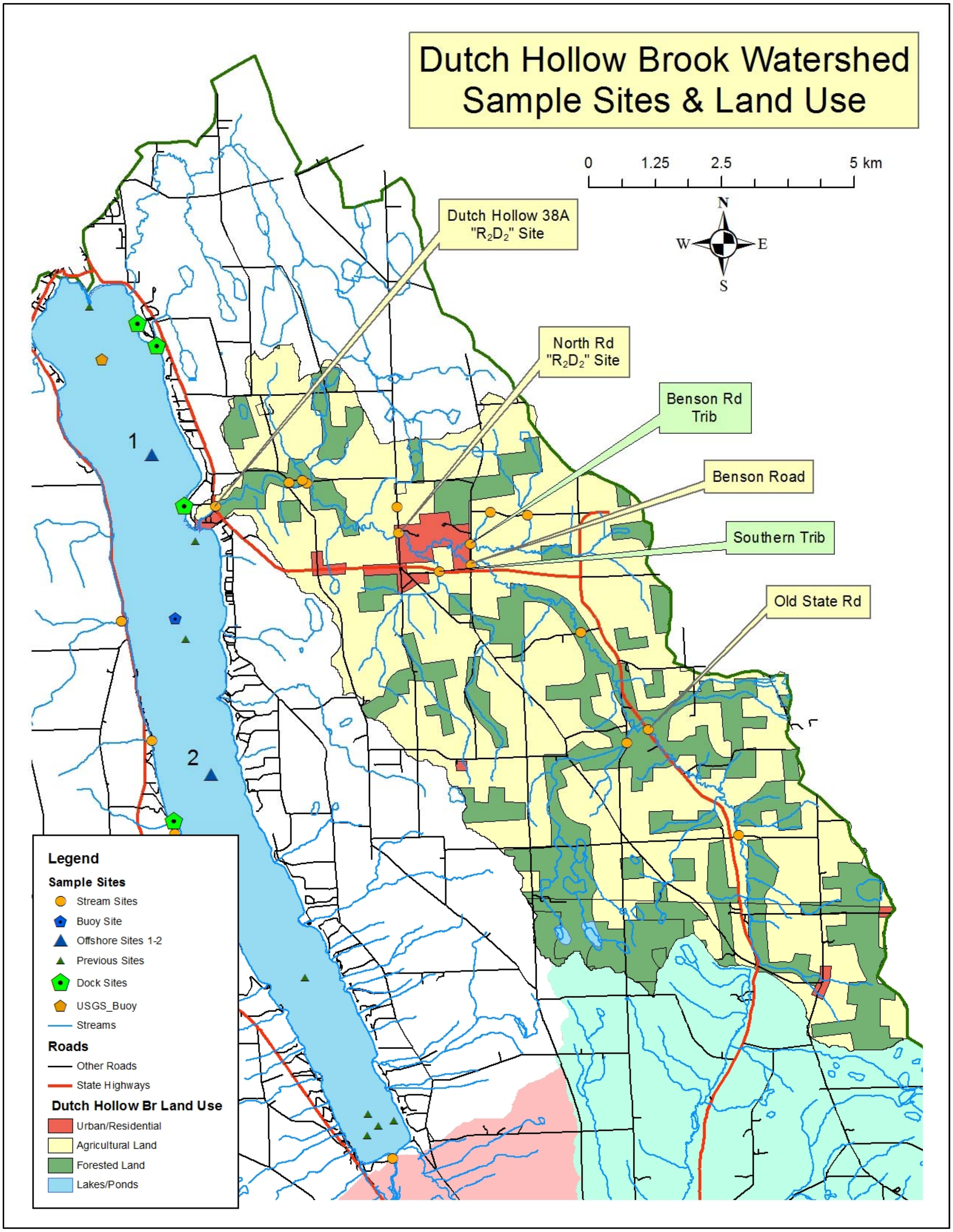

Fig. 1 continued. Site locations and land use within Dutch Hollow Brook watershed.

Owasco Streams: The 2020 stream monitoring program 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 2020. 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 recent years.

At each site, stream discharge, water temperature, conductivity, 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 (NOx) 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., Dutch Hollow Brook at both 38A and North Rd. The published USGS gauge flow measured at Moravia (USGS Gauge 4235299) was used for the Owasco Inlet discharge in 2020. 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., $\mu g/L$).

Runoff/Event Flow versus Base Flow Variability: A Teledyne ISCO 6712, full size, automated water sampler, and two pairs of *ONSET* HOBO U20L-04 data loggers were deployed at the Rt 38A site in Dutch Hollow Brook from 4/15 to 11/4 (185 days) to investigate the impact of event *versus* base flow variability on nutrient and sediment loads to the lake (Figs. 2a & 2b). The autosampler was programmed to collect 1-L of water daily (4 am). This periodicity 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 water sample was analyzed for suspended sediment and nutrient (TP, SRP and NOx) concentrations. In 2020, a sample was not collected on 5/2 as a flood displaced the sampler’s intake, and sand clogged the sampler’s water intake from 8/21 through 9/1 and again from 10/30 through 11/4.

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.

One logger in each pair was deployed in air and the other deployed underwater at a fixed elevation just above the stream bed, i.e., attached to a fence post driven into the stream bed. The paired air/water deployment accounted for changes in atmospheric pressure to isolate 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 single pair of loggers to a flood, vandalism or any other unfortunate issues. The duplication was useful in 2020 because the 5/1 flood event dislodged both loggers from the stream bed. They were successfully redeployed on 5/4, one was reattached to its fence post and the other attached to a 90 lb anchor and chain wrapped around some large rocks for the remainder of the field season. Neither redeployment was uprooted afterwards. Stage data from 5/1 through 5/4 at Dutch Hollow Brook were interpolated from Owasco Inlet USGS data.

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 for 2020, the relationship between stream stage (height) and stream discharge. The rating curve was then used to estimate the stream discharge for every ISCO water sample and thus daily flux of nutrients and suspended sediments.

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[2]. Briefly, an aliquot of each water sample was analyzed for total phosphate using a colorimetric analysis by spectrophotometer after digestion of any organic-rich particles in hot (100ºC) persulfate for 1 hour. An additional sample water (~1L for stream, and ~3L for lake samples) 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) and nitrate (NOx) 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

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

determined the total suspended sediment concentration. A known volume (~1L) 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 and NOx 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 during the analysis of each group of samples for a continuous check on data quality. The NOx 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) and nitrate ±0.1 mg/L.

Owasco Buoy: A renewed three-year award by the Fred L. Emerson Foundation supported the redeployment of the FLI meteorological and water quality monitoring buoy manufactured by YSI/Xylem. It was redeployed at its mid-lake site from 5/22 through 10/24 (Table 1, Fig. 1). COVID issues delayed the normal April deployment. The buoy was again 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 cyanobacteria 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 user accessible website (http://fli-data.hws.edu/buoy/owasco/). Minimal solar power and other issues prevented collection of water quality data from 7/29 – 8/1, 9/19 – 9/24 and 9/25 – 9/28. The meteorological data collection was not interrupted.

Drone Flights & Spectrophotometer Measurements: FL-LOWPA funds supported drone flights and collection of the signatures of light emitted from the lake. Unfortunately, COVID restrictions on student hires prevented drone flights in 2020. Instead, we used an Ocean Insight R400 VIS-NIR spectrophotometer to measure the spectral signature of the upwelling and down-welling radiation (light) over 350 to 800 nm in 0.5 nm steps at the offshore and dock (Fig. 3). This data will be presented in the companion report.[3]

Dockside Water Quality & Meteorological Monitoring: Funding from the Fred L. Emerson Foundation enabled deployment of a weather station, water quality sonde, water temperature loggers, and an automated camera were deployed at four docks in Owasco Lake (Fig. 4). The equipment was deployed to elucidate occurrences of nearshore cyanobacteria blooms, and the bloom's precursor meteorological and limnological data. The program follows up on promising initial results from previous years[4]. The deployment locations were at the northern and southern sides of Martin Point, Burtis Point, and at the end of Fire Lane 20 (Fig. 1, Table 1).

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[3] Halfman, et al., 2020. Cyanobacteria on Owasco and Seneca Lakes, the 2020 Update. The 2020 Annual Report to the Fred L. Emerson Foundation, Seneca Lake Pure Waters Association and the Finger Lakes – Lake Ontario Watershed Protection Alliance.

[4] The early results were supported by funds from the Finger Lakes - Lake Ontario Watershed Protection Alliance, Seneca Lake Pure Waters Association and the Finger Lakes Institute.

Fig. 3. The spectrophotometer in the field.

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

At each dock, a weather station (Ambient 1002-WS or WS-2000 Osprey) recorded air temperature, rainfall, barometric pressure, humidity, light intensity, wind speed and direction at 30 minute intervals. An *ONSET* HOBO U20L-04 logger was strapped to a dock post originally at 1 m of water and HOBO Tidbit MX logger was attached to a surface float to record water temperatures at both depths 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 to log nearshore water quality, i.e., log clear vs. turbid water, and obvious surface cyanobacteria blooms. At this deployment height, the camera's 60° field of view imaged 2 x 3 to 3.5 x 5 meter area of the lake's surface. Finally, an *In-Situ* Aqua Troll 600 water quality sondes with temperature, conductivity, total chlorophyll and cyanobacteria phycocyanin sensors was deployed at each dock except one where a *YSI/Xylem* EXO2 water quality sonde with temperature, conductivity, dissolved oxygen, turbidity, and total chlorophyll and cyanobacteria phycocyanin fluorescence was deployed instead. The results of this dockside investigation will be discussed in the companion report[5].

[5] Halfman, et al., 2020. Cyanobacteria on Owasco and Seneca Lakes, the 2020 Update. The 2020 Annual Report to the Fred L. Emerson Foundation, Seneca Lake Pure Waters Association and the Finger Lakes – Lake Ontario Watershed Protection Alliance.

Results & Discussion

2020 Precipitation

Previous reports concluded that annual rainfall, its seasonal variability and individual storm intensities influenced 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 its associated nutrient and sediment loads to the lake, especially rain events in the spring season. On smaller, seasonal scales, runoff is influenced by, for example, changes in soil saturation, water infiltration rates, evapotranspiration rates, and the extent of plant cover on agricultural lands (e.g., spring tillage for planting, harvesting in the fall). Thus, numerous variables influence the percentage of rainfall that entered runoff rather than infiltrates into the ground that, in turn, dictates 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 runoff over land surfaces without vegetation, i.e., unplanted fields. Thus, a spring rainstorm produces proportionally more runoff, erodes more soil and increases nutrient and sediment loads to the lake than a summer or fall event.

Rainfall in 2020 was significantly (56%) below normal during the field season (Fig. 5). Seasonally, spring was 82% of “normal”, summer 40% of “normal”, and fall 12% of “normal”. Normal as defined by the weather bureau. Thus, the lack of rainfall throughout the field season should dictate reduced runoff and nonpoint source nutrient and sediment loads to the lake.

The 2020 field season was drier than every year since 2011 (Fig. 5). The other low rainfall years include 2012, 2016, 2018 and 2019 at 88%, 76%, 93%, and 85% of normal, respectively. The 2020 rainfall amounts suggest that runoff and nutrient loading to Owasco Lake should have declined, and water quality in the lake should have improved.

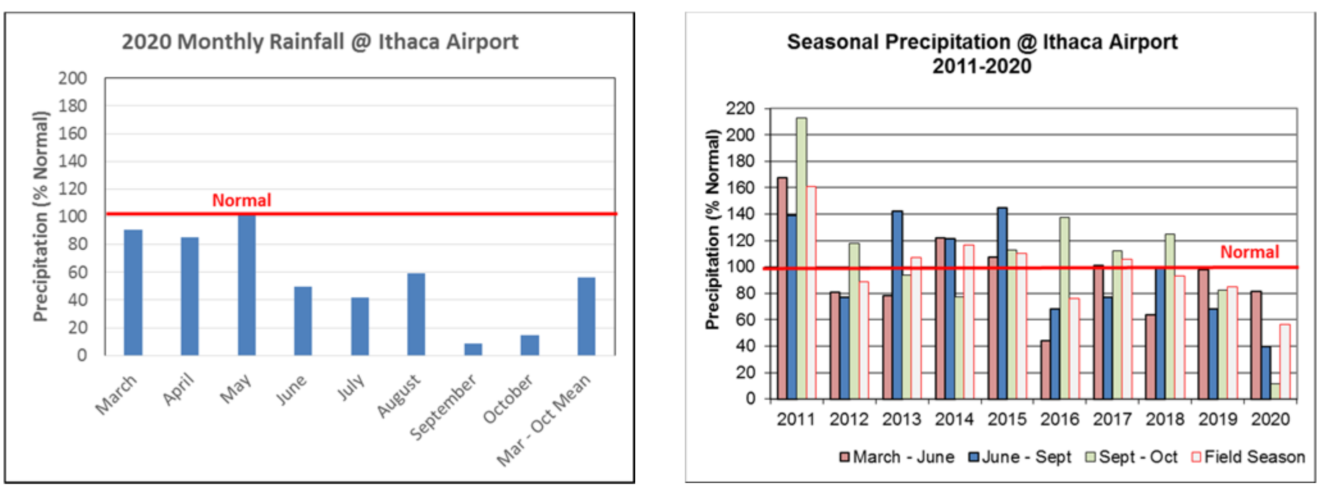
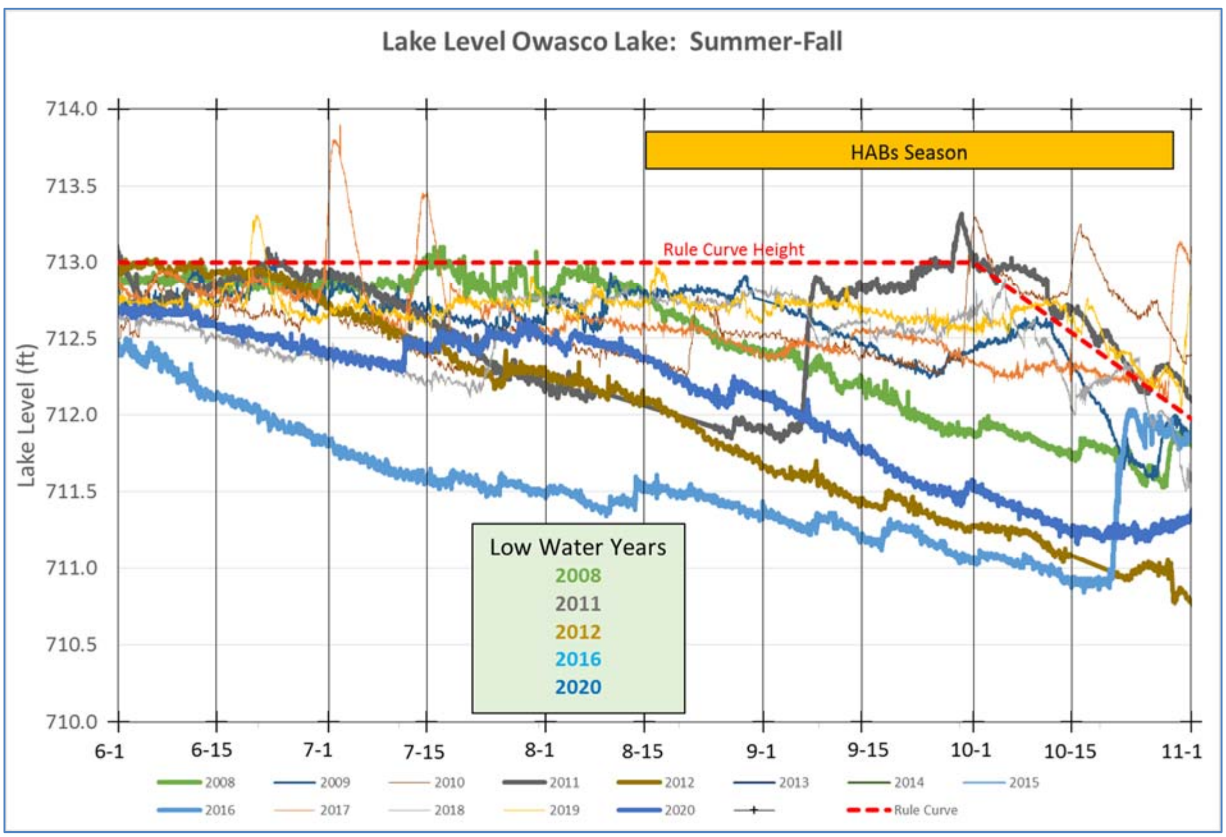
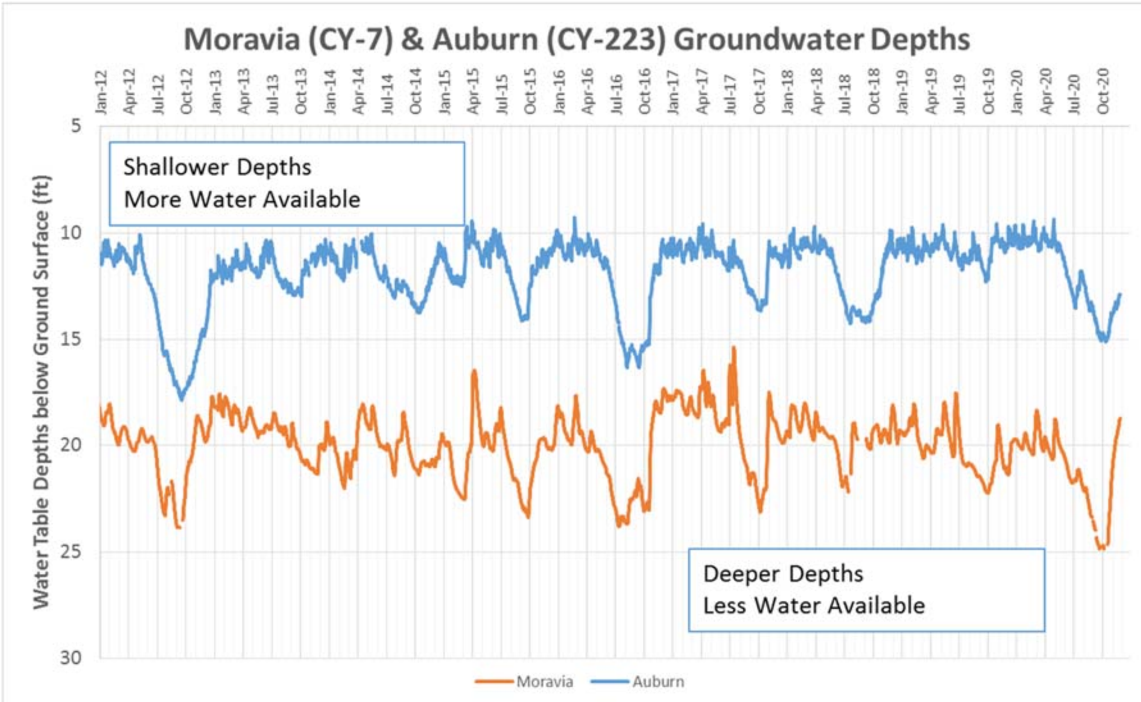


Fig 5. 2020 Monthly (left) and 2011 – 2020 spring, summer and fall seasonal, and field season precipitation (right) compared to normal totals at the Ithaca Airport.

Owasco Lake Levels: Another indicator for dry conditions is lake level and water table elevations. Records of both are available within the Owasco Watershed. Lake levels in the late summer and early fall were approximately 1.5 to 2 ft lower in 2020 than the rule curve summer elevation of 713 ft (Fig. 15). This past year was the 3rd lowest summer/fall-season lake levels since 2008, after 2012 and 2016. 2020 also recorded the lowest water table depths at Moravia (USGS Site: CY-7), and the third lowest water table depth at Auburn (USGS Site CY-122) monitoring wells over the past decade (Fig. 15). These records all highlight very dry conditions in 2020.

Speculatively, the timing of the low lake levels was coincident with the HABs season. We openly wonder if lower lake levels in 2020 contributed to the large number of HABs events recorded by the DEC. HABs sightings started in 2012 in Owasco Lake, a low lake level year, and the number of sightings each year peaked in 2016, the lowest recorded lake level since 2008 (Fig. 15). Lower lake levels may have exposed nutrient rich sediments to increased wave action and erosion, and the decline through the summer might have allowed more macrophytes to be abandoned and rot along the shoreline. Both processes would provide more nutrients to the lake and support more blooms. Unfortunately, significantly fewer blooms were observed at Seneca Lake despite similar precipitation, lake level and water table conditions.





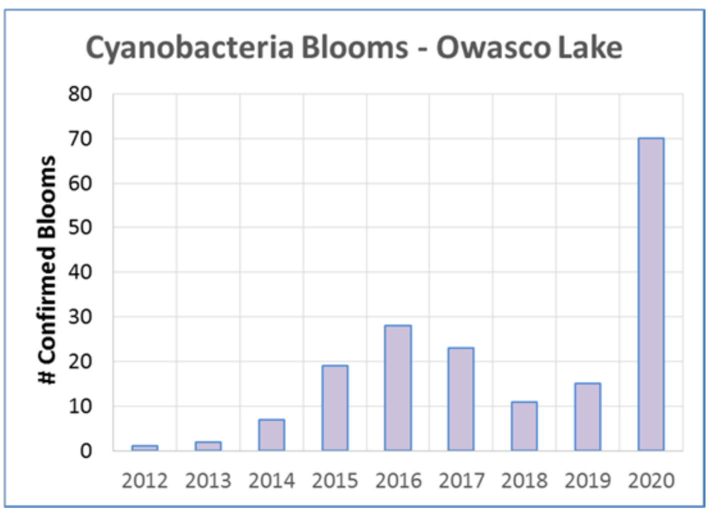


Fig. 15. 2008 through 2020 lake levels for the summer/fall season (above). 2012 through 2020 groundwater water table depths below the ground's surface (middle). Number of confirmed cyanobacterial blooms in Owasco Lake tabulated by the DEC NYS-HABs web site (left).

LAKE MONITORING

Lake CTD Profiles: The 2020 offshore water temperature profiles revealed warmer surface water temperatures than earlier years (Fig. 6). The seasonal stratification, the initiation of less dense and warmer epilimnion (surface water) overlying the denser and uniformly cold hypolimnion (bottom water), was already established by the first cruise. The thermocline, the boundary between the surface and bottom waters, was again between 10 and 15 meters for most of the stratified season. The depth of the thermocline deepened seasonally, and oscillated up and down by a few meters due to internal seiche activity Epilimnetic water temperatures ranged from 8ºC (~46 ºF) in late May peaked at 25.5ºC (~78ºF) in early August, and cooled to 18ºC (63ºF) by the last cruise of the survey (9/22). Hypolimnetic water temperatures remained cold, warming from 5.9º to 6.2ºC (42-43ºF) through the field season.

Owasco Lake experienced the warmest mean surface water CTD temperatures (1 to 10 m average) in 2020 than any other year since 2005. The CTD temperature record is consistent with buoy temperatures when both records overlapped in time (Fig. 7). A best-fit, linear interpolation of the CTD surface temperatures revealed a mean warming of 0.2°C/year (0.0006°C/day). Water temperatures deviated above and below the linear trend and were probably influenced by natural climatic variability including the amount of cloud cover, rainfall, mean wind speed and global phenomena like El Nino. For example, 2009 and 2016 were slightly warmer whereas 2013 and 2014 were slightly cooler than the linear trend. The long term warming, however suggests that Owasco Lake is influenced by Global Warming. Speculatively, the warmest water was coincident with the largest number of cyanobacteria blooms in Owasco Lake. Warmer temperatures can promote faster and more complete bacterial decay of the dead organic matter and release more nutrients for algal uptake. However, Seneca Lake surface water was also warmest on record but Seneca experience very few blooms in 2020 compared to previous years.

Epilimnetic salinity (specific conductance) ranged from 300 to 335 μS/cm in 2020 (~150 ppm TDS). Like previous years, epilimnetic salinity in 2020 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 (Fig. 8). The early spring, surface water, specific conductance was slightly larger in 2020 than 2019 (~15 μS/cm), and reversed the steady but very small decline from 2015 to 2019. 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 salinities in 2020 was slightly larger than previous years. It may have reflected decreased rainfall in 2020 and less dilution of the previous winter's salt input than earlier years.

The 2020 hypolimnetic specific conductance data were between 333 and 345 μS/cm and increased slightly over time (Fig. 6). These values were similar to those detected in in previous years. Small annual fluctuations can again reflect the suspected inputs of road salt during the previous winter and dilution by spring rainfalls, i.e., 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.

The epilimnetic dissolved oxygen (DO) concentrations remained between 8 and 12 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 (~60% saturation) in the upper hypolimnion and 8 mg/L (~60% saturation) in the lowest hypolimnion by late summer. These lowest saturation levels approached the threshold for respiratory stress in sensitive organisms. The decrease is interpreted to reflect hypolimnetic bacterial respiration and decomposition of dead algae. The hypolimnetic depletion was slightly less severe in 2020 than the past five years.

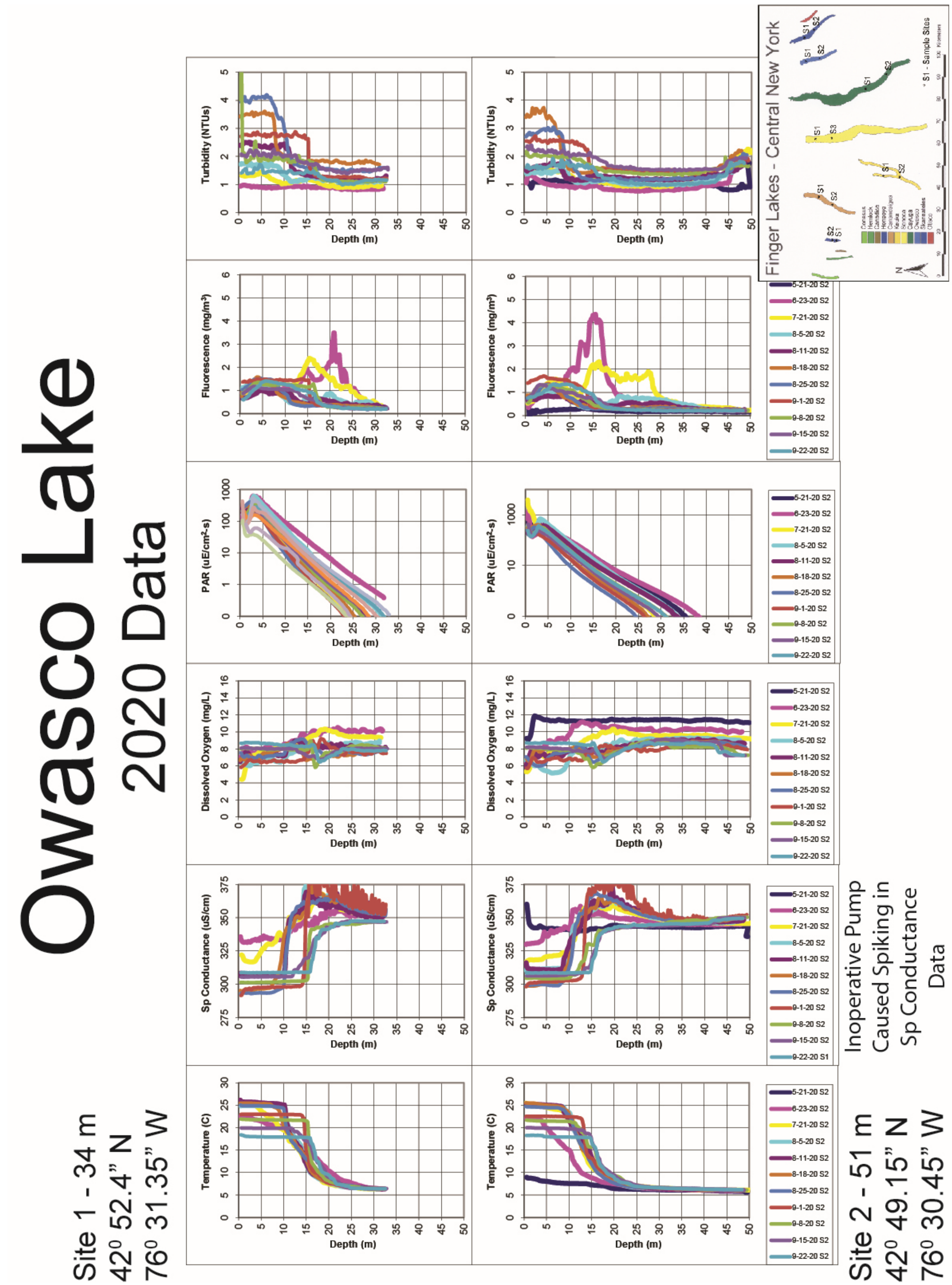


Fig. 6. CTD profiles from Sites 1 & 2 in 2020. The PAR (light) data are plotted on an exponential scale, so that the expected exponential change in light intensity with water depth appears as straight lines.

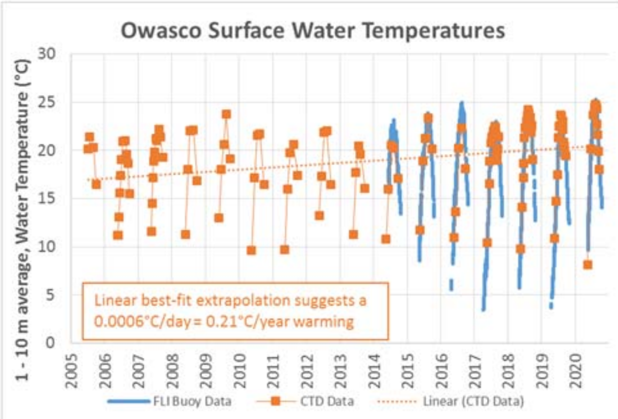


Fig. 7. The available CTD and FLI buoy mean surface water temperatures (1 to 10 m average) since 2005.

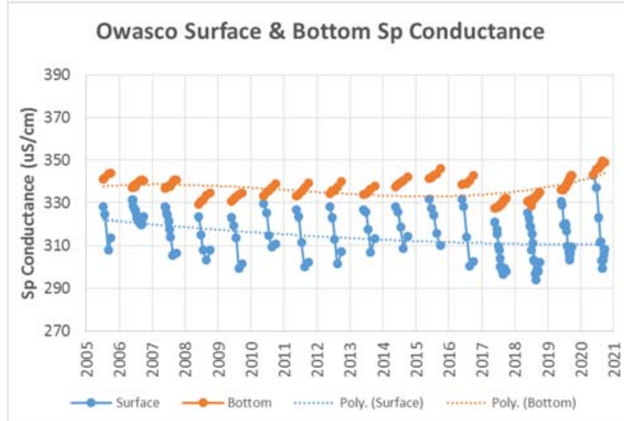


Fig. 8. Mean surface and bottom water salinities by CTD since 2005.

Profiles of photosynthetic available radiation (PAR), i.e., light intensity again 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 in 2020 (Fig. 6). 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. The surface light intensities were slightly smaller, slightly below 1,000 in 2020 compared to slightly above 1,000 $\mu E/cm^2$-s in earlier years. Increased surface water turbidity and increased relative percentages of cyanobacteria concentrations in the plankton may explain the decrease.

Fluorescence, a measure of algal pigment concentrations, revealed an occasional peak in algal abundance within the lower epilimnion at approximately 15 to 20 m below the lake's surface (Fig. 6). Peak concentrations exceeded 4 μg/L (mg/m^3) on 6/23, and were above 2 μg/L on 7/21. Algal concentrations were lower, between 1 and 2 μg/L, on the other survey dates. Fluorescence measures algal pigment concentrations and not algal populations. Thus, the algal peak at depth may reflect some combination of an increase in algal biomass or more pigments per cell in the lower light conditions. These peak concentrations and frequency of peaks were smaller in 2020 than earlier years. It parallels the decrease in spring and early summer rainfall, as runoff is the primary source of new nutrients for large algal blooms. Hypolimnetic concentrations were consistently below 1 μg/L, i.e., algal pigments were nearly absent in the dark bottom waters.

The turbidity profiles typically revealed uniform or nearly uniform turbidities of 1 to 2 NTUs down to a poorly defined benthic nepheloid layer at the deep site (Fig. 6). On 8/18, 8/25 and 9/1, the turbidity in the epilimnion rose above 2 NTUs to 3 or 4 NTUs. These peaks did not align with an algal bloom but instead occurred just after prolonged wind events. It suggests that erosion of nearshore sediments by wind driven waves provides a viable source of turbidity to the epilimnion before it settles out to the lake floor. The change in benthic turbidities from year to year typically parallel the change in rainfall and wind velocities, as the primary source of bottom-water suspended sediments (turbidity) is runoff events from precipitation and snowmelt, and resuspension events by waves. For example, the poorly defined benthic nepheloid layer in 2020 compared to better developed nepheloid layers in earlier years, probably reflected the decreased rainfall in 2020 compared to earlier years.

The bbe FluoroProbe data revealed similar concentrations of green, diatoms, cryptophyte and cyanobacteria in surface water grabs at the offshore sites in 2020 (Fig. 9). The total algal concentrations ranged from 3 μg/L in the early summer to 15 μg/L on 8/28. From 2017 to 2020, the algal population changed from mostly cryptophytes and diatoms with lesser amounts of green algae and trace amounts of cyanobacteria in 2017 to similar amounts of all four algal groups in 2020. The increased presence of cyanobacteria in 2020 might reflect decreased rainfall and nutrient loading in 2020 compared to the earlier years. In support, larger relative percentages of cyanobacteria compared to other algal groups were detected in the algal populations in neighboring oligotrophic lakes like Skaneateles, Keuka and Canandaigua, compared to mesotrophic and eutrophic Finger Lakes like Honeoye, Seneca and Cayuga. It suggests that cyanobacteria may dominate algal populations in nutrient starved habitats.

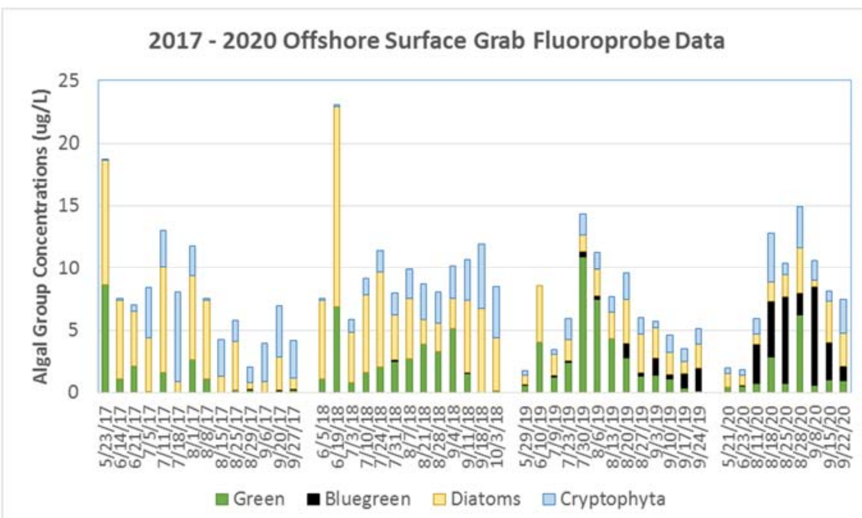


Fig. 9. Offshore, surface grab, date averaged, bbe FluoroProbe data revealing the relative concentrations of the four algal groups from 2017 through 2020.

Limnology & Trophic Status: Date averaged mean chlorophyll-a concentrations in the epilimnion ranged from 0.5 to 7.2 μg/L and averaged 3.7 μg/L in 2020 (Table 2 in appendix, Fig. 10). The largest values were detected during August and September. The chlorophyll-a concentrations used in this report were smaller than CTD total chlorophyll and FluoroProbe fluorescence data because the former only measures the concentration of one algal pigment whereas the other fluorescence datasets measured the concentration of all pigments. The chlorophyll-a annual mean concentration was slightly larger in 2020 compared to earlier years but still below the 4 to 6 μg/L not to exceed DEC threshold for potable water bodies[6]. This increase is surprising as 2020 was dry and nutrient loads should have been reduced. However,

[6]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.

the increased presence of cyanobacteria and their surface floating tendencies compared to other algal groups might have increased the chlorophyll-a concentrations in the surface grab samples compared to earlier years.

Secchi disk depths ranged from 1.7 to 5.1 meters, and averaged 3.4 meters in 2020 (Fig. 10). Annual mean depths 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 but shallowed slightly in 2020. The timing 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. However, shallower Secchi depths in 2020 are not consistent with drier conditions. Perhaps the proportional cyanobacteria increase in the algal populations, and their ability to float near the surface, obstructed light penetration and Secchi disk depths in 2020, reversing the annual trends. TSS and chlorophyll-a trends, and lower light intensities measured by the CTD were consistent with this hypothesis.

The lake was not impaired due to phosphorus, as the annual mean total phosphate (TP) concentration ranged from 3 to 6.9 μg/L. The annual average of 4.7 μg/L was below the 20 μg/L TP threshold used by the DEC to designate impaired (eutrophic) water bodies. The impaired waterbody threshold was never exceeded on any sample date in 2020 with a maximum date-averaged TP concentration of 6.9 μg/L on 8/18. Since 2006, annual mean TP concentrations have increased from ~8 to over 17 μg/L by 2014 with a slight dip in 2013 (Fig. 10). After another dip in 2015 and 2016, TP increased to 16.2 μg/L in 2017. Since 2017, TP decreased to 4.7 μg/L in 2020, fluctuations that parallel rainfall trends. Annual mean soluble reactive phosphate (SRP) concentrations in 2020 remained very small 0.5 μg/L, similar to 2010, 2012, 2015, 2018 and 2019, compared to years with significantly larger SRP concentrations, i.e., 2006 and 2017 (1.9 μg/L), and especially 2014 (5.8 μg/L, Fig. 10). 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 and 2020 suggests that decomposition of organics within the lake may provide a critical SRP source. The consistently low SRP concentrations indicates that SRP is the limiting nutrient in the open lake.

Nitrate concentrations ranged from 0.3 to 1.0 mg/L, and an order of magnitude (10 times) below the 10 mg/L maximum contaminant level (MCL) established by the EPA.

Total suspended sediments (TSS) concentrations ranged from 1.1 to 3.5 mg/L and averaged 1.0 mg/L. The total suspended sediment (TSS) annual mean concentrations in 2019 (2.6 mg/L) and 2020 (2.1 mg/L) reversed a declining trend since a peak of 3.5 in 2014, down to 1.7 mg/L in 2018 (Fig. 10). Overall, 2014 revealed the worst water turbidity for the lake. Water turbidity decreased through 2018 but increased slightly in 2019 and 2020. The trends, for the most part, parallel changes in rainfall. Perhaps wind driven waves increased suspended sediment concentrations in 2019 and 2020. Alternatively, proportionally more cyanobacteria in 2019 and 2020 may have increased the TSS concentrations in surface grab samples.

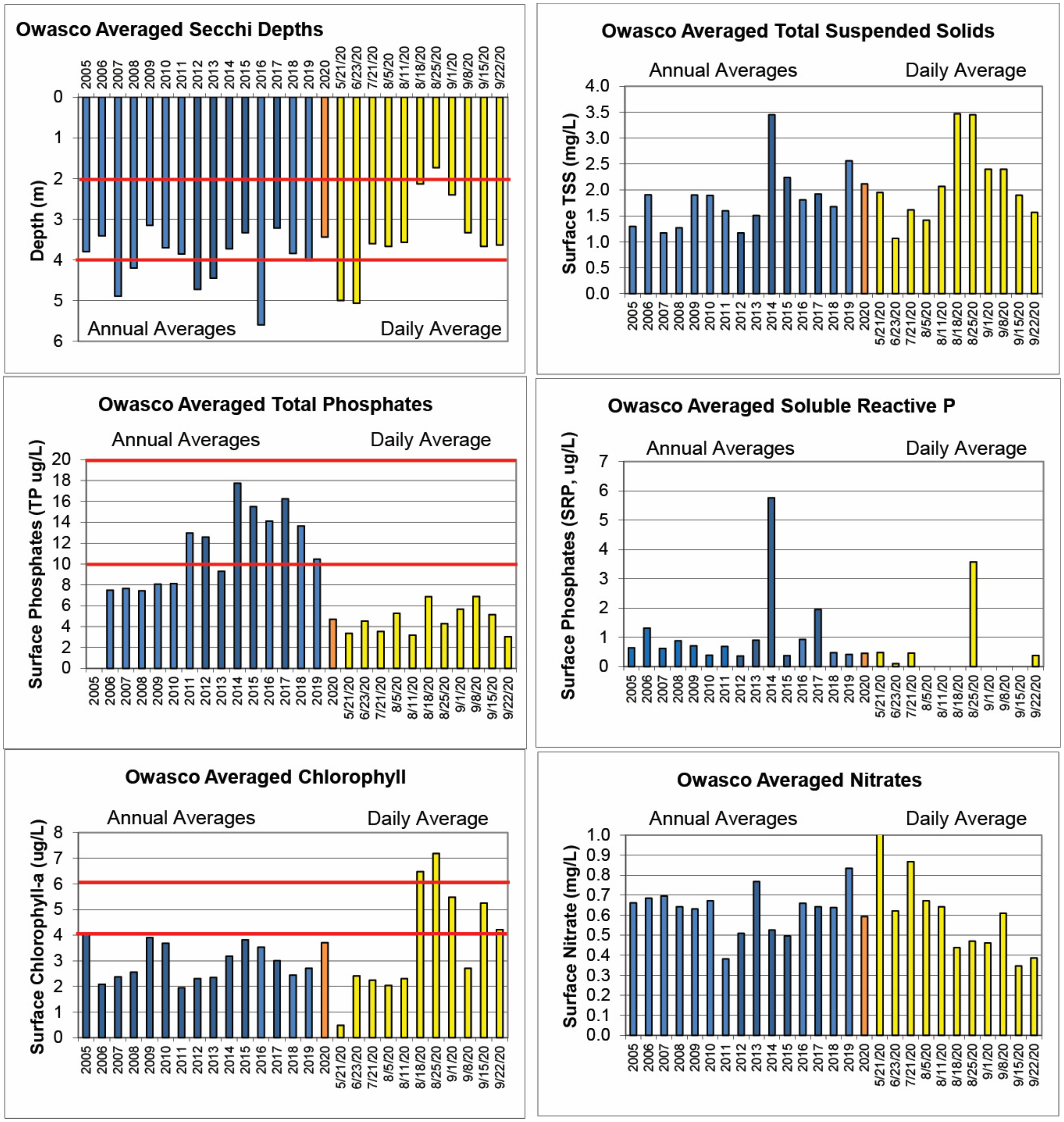


Fig. 10. Annual average surface water concentrations from 2005 (blue) to 2020 (orange), and date averaged offshore surface water data during 2020 (yellow). When appropriate, boundaries for oligotrophic, mesotrophic and eutrophic concentrations are marked with red lines.

The 2020 trophic status indicators again yield a mixed signal. Annual mean Secchi depth and hypolimnetic dissolved oxygen saturation data placed Owasco Lake just above the oligotrophic-mesotrophic trophic boundary (Table 3). Nitrogen, measured by NOx concentrations, total phosphate and chlorophyll data placed Owasco Lake below the boundary. Thus, the overall trophic status of Owasco Lake in 2020 is oligotrophic. The fluctuations above and below the boundary over the past decade 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 ranges for Oligotrophic (low productivity), Mesotrophic (mid-range productivity), and Eutrophic (high productivity) lakes. The bold entries reflect Owasco's 2020 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 are noteworthy.

First, the mean, surface water, soluble reactive phosphate (SRP) to nitrate (NOx) ratio in the lake, the two nutrients that typically limit algal growth, averaged 1:2,300 in 2020. 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 NOx are augmented by additional sources of nitrogen to the lake (e.g., atmospheric deposition of acid rain NOx) not available to phosphorus. Mesocosm experiments suggest that ammonium (NH_4^+), the reduced and preferred source of nitrogen for algae, might also limit algal growth.

Second, variability was observed in every parameter from one survey date to the next (Fig. 10). The extent of the variability is best observed in the buoy data (next section) and in the box and whiskers plots (Fig. 11). 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. For example, the largest 2020 algal bloom occurred between the May and June surveys.

Third, the dissolved nutrient concentrations revealed slightly larger concentrations in the hypolimnion than the epilimnion. The annual mean surface and bottom water concentrations were 0.5 to 1.0 μg/L for SRP, and 0.6 to 1.1 mg/L for NOx. Chlorophyll-a concentrations revealed the expected decrease from the epilimnion to the hypolimnion of 3.7 to 0.2 μg/L. The separation highlights the expected algal uptake of nutrients in the epilimnion and bacterial decomposition of organic materials (e.g., algae) and release of nutrients in the hypolimnion.

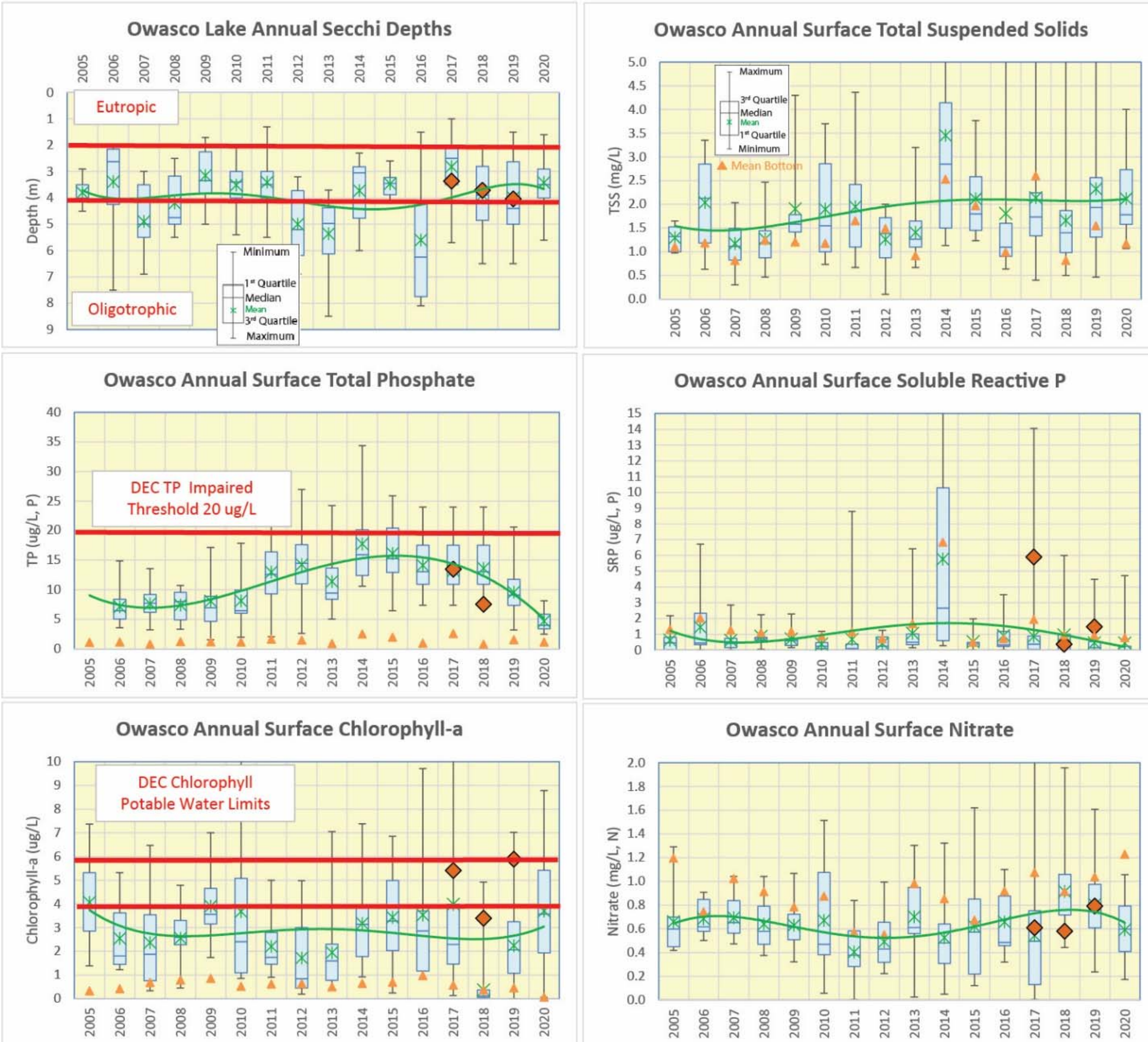


Fig. 11. Box and Whisker plots of the annual nutrient, chlorophyll and Secchi disk data. The annual mean bottom water concentrations (orange triangles) and annual mean DEC C-SLAP data (orange diamonds) are also plotted. DEC measured TDP in 2017 instead of SRP, and total chlorophyll instead of chlorophyll-a. Both differences should result in larger values. 2019 TP data, and all 2020 C-SLAP data were not available.

Finally, 2017 through 2019 mean TP (no 2019 data), SRP, total chlorophyll and NOx surface concentrations determined by C-SLAP were similar to the results from this study (Fig. 11). 2020 C-SLAP data are not yet available. C-SLAP's TP, SRP and NOx were within or just above the "box" of the box and whisker plots, and total chlorophyll above the "box". Larger concentrations reflected by C-SLAP's chlorophyll and 2017 total dissolved phosphate (TDP) concentrations may reflect the natural day to day variability in these parameters. However, total chlorophyll determined by C-SLAP measured more "stuff" that chlorophyll-a data presented in this report, and C-SLAP's TDP measures more stuff than SRP presented here as TDP includes both dissolved organic matter and dissolved phosphorus (SRP) in the sample.

Plankton Data: The phytoplankton (algal) species in Owasco Lake during 2020 were dominated by diatoms in the early part of the field season, primarily *Asterionella*, with smaller numbers of

Fragillaria and *Diatoma*, and cyanobacteria, primarily *Mycrocystis* with some *Dolichospermum* (formerly *Anabaena*), and other species (Table 4 in appendix, Fig. 12). Like previous years, the distribution was dominated by diatoms in the early part of the field season, but by early August, the population quickly transformed to and was dominated by cyanobacteria species, primarily *Mycrosystis* for the remainder of the field season. The overall diversity decreased in 2020 from earlier years. Other phytoplankton species detected included a few *Synedria*, and *Dinobryon*. Zooplankton species were dominated by rotifers, namely *Vorticella*, *Keratella* with some cladocerans, like *Copepods*, and *Cercopagis*, the fishhook water flea. Zebra and quagga mussel larvae were also detected in the plankton tows.

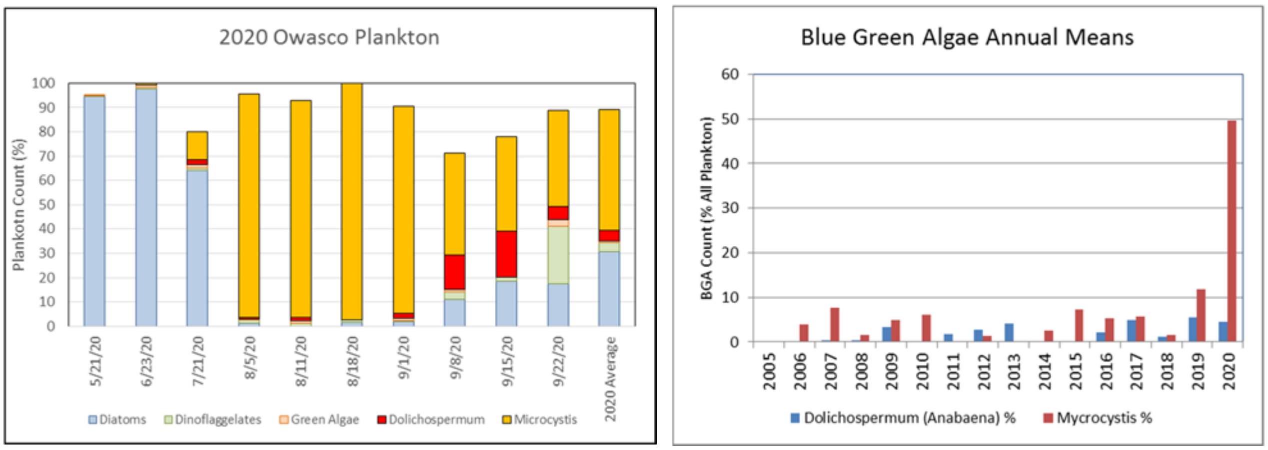


Fig. 12. Date averaged plankton data for 2020 (left) and the mean annual abundance of cyanobacteria species since 2005 (right).

Cyanobacteria genera, *Microcystis* and to a lesser degree *Dolichosperma*, were dominant in the 2020 August through September surveys (Fig. 12). Detection of cyanobacteria in the lake is not new, and they were always detected in the open water of Owasco Lake since the initial FLI surveys in 2005. In fact, cyanobacteria species were detected in a neighboring Finger Lake as long ago as 1914[7]. However, major blooms of cyanobacteria have been increasingly detected along the shoreline in Owasco Lake since 2012[8]. 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, 2019, and 2020 and *Dolichosperma* making up 30% of the late-summer counts in 2013. In 2020, the offshore presence of cyanobacteria increased dramatically over previous years, with a maximum relative percentage of 92% for *Microcystis* during the 8/5 survey. This is consistent with the Fluoroprobe results.

Finger Lake Water Quality Ranks: The 2020 Finger Lake water quality rank for Owasco Lake was better in 2020 compared to previous years in the study, except for 2019 (Table 5 in appendix, Figs. 13 & 14). 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

[7] Bloomfield, J.A. (ed.), 1978. Lakes of New York State. Vol.1: The Ecology of the Finger Lakes. Academic Press.
[8] http://www.dec.ny.gov/chemical/83332.html

oligotrophic-eutrophic trophic states (discussed above), and Carlson's Trophic Indices[9] that quantitatively combine chlorophyll-a, total phosphorus and Secchi depth data (Fig. 13). In 2020, water quality in Owasco ranked poorer than Canandaigua, Keuka, Seneca and Skaneateles Lakes, and better than Cayuga and Honeoye Lakes.

Owasco Lake water quality improved from 2014 through 2019, and declined slightly in 2020. It suggests that the observed change in water quality in these lakes are not only influenced by nutrient loading and rainfall totals but are probably 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 trend suggests that the recently adopted BMPs are be beginning to work. Algal populations are also influenced by "top-down" ecological pressures from zebra and quagga mussels, Asian clams and *Cercopagis*, the fishhook water flea. Finally changing offshore algal populations to larger percentage of cyanobacteria in 2020 could have arbitrarily increased the chlorophyll-a and TSS data, worsening Owasco's annual rank compared to earlier years.

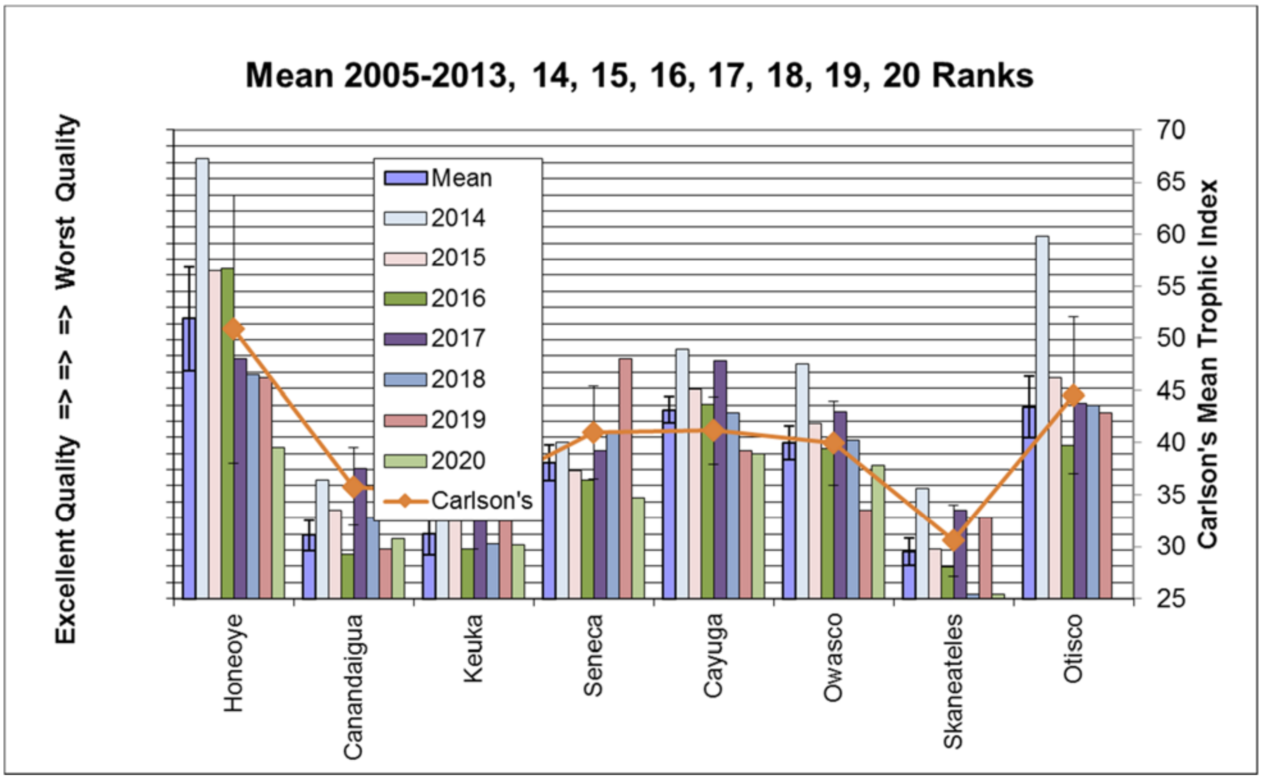


Fig. 13. Annual Water Quality Ranks from 2005 – 2020 for the eight easternmost Finger Lakes. Otisco was excluded from the survey in 2020 due to COVID issues. The "mean" dark blue bar averaged the 2005 - 2013 ranks for each lake with a 1σ standard deviation error bar. Carlson's mean trophic indices of the mean Secchi depths, total phosphate and chlorophyll concentrations are also shown.

[9] Carlson, R.E. 1977. A trophic state indicator for lakes. Limnology & Oceanography, 22:361-369.

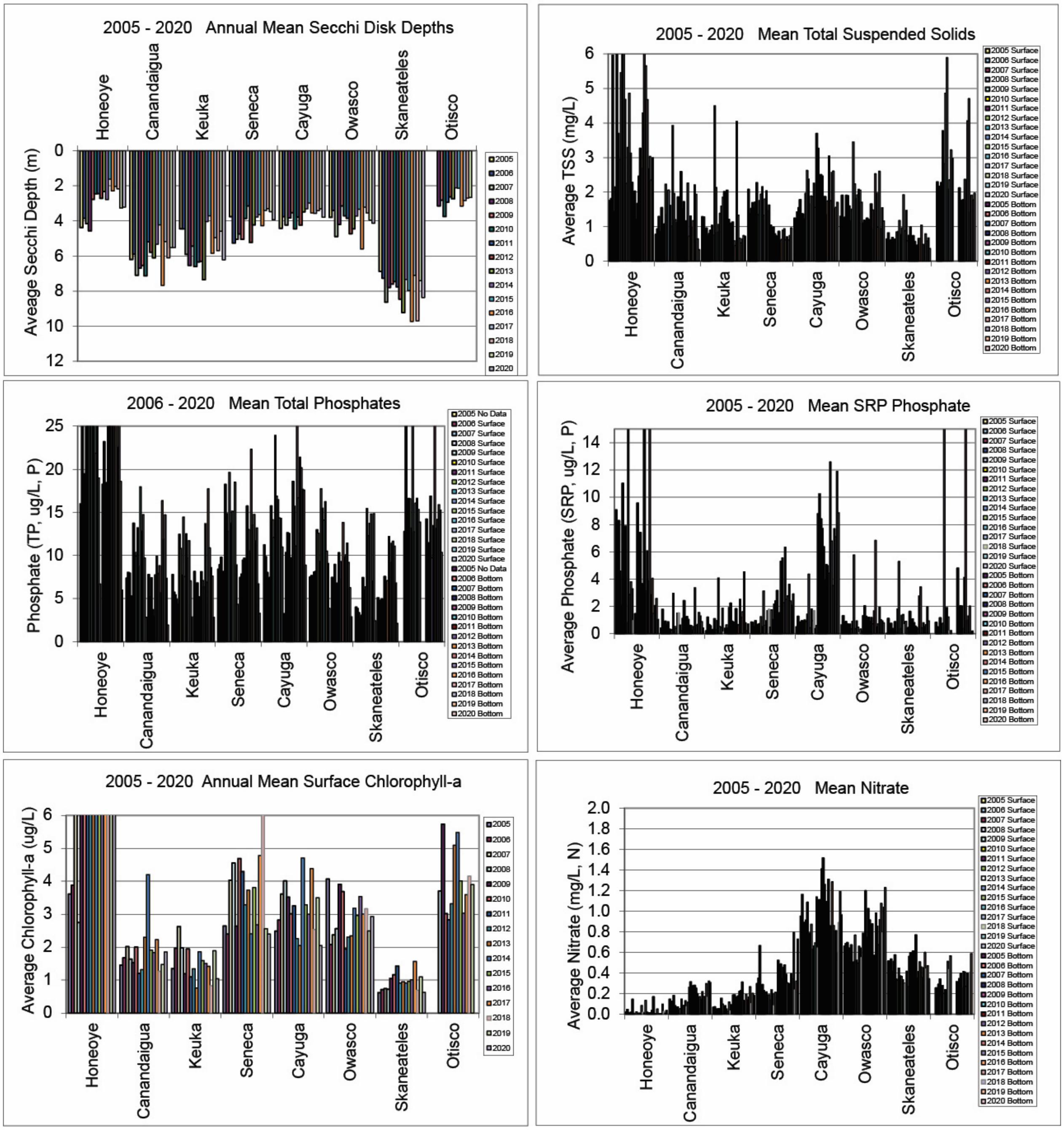
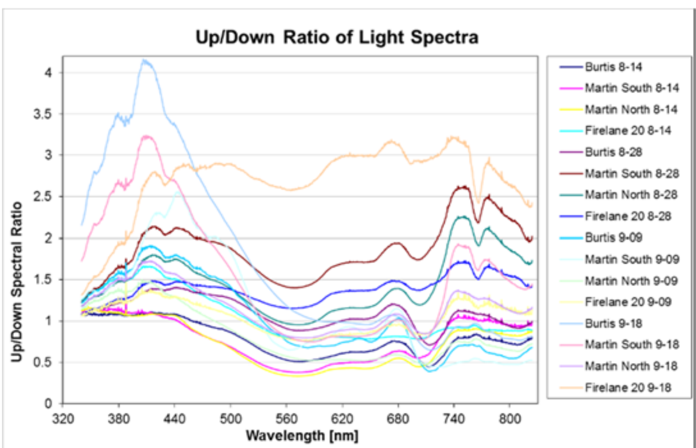


Fig. 14. 2005 – 2020 annual mean limnological data from the eight eastern Finger Lakes.

DRONE FLIGHTS & NEARSHORE SPECTRAL DATA

Drone images were not collected in 2020 due to COVID issues. Instead, we focused our imaging efforts on the spectra of light reflected by from the lake and its relationship to the algal populations and concentrations. Complete spectra (from 340 to 823 nm at ~0.5 nm intervals) of the upwelling and down-welling light were collected multiple times at the two offshore and four dock sites to find an indicator concentrations (Fig. 16). The intent was to determine if the ratio between upwelling and down-welling spectra could resolve algal concentrations. The 2020 results are very encouraging and revealed potential algal signatures in the near infrared portions of the spectrum where algae reflect the most light (wavelength of ~700 nm). The offshore sites provided similar results although backscattering by waves and the reflections of the boat influenced the spectra results and suggests using a cosine filter in the future. More work must be done next summer to improve the spectroscopic techniques. We also suggest to continue our periodic drone flights to map the distribution and concentration of nearshore macrophytes, attached algae and cyanobacteria blooms in the years ahead.



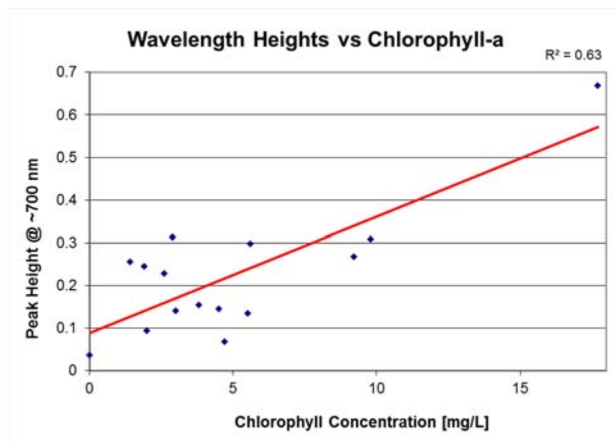


Fig. 16. Ratio of upwelling and down-welling spectra from the dock sites (left). Algal signatures are typically in the near infrared (~700 nm). Peak height at 700 nm for each up/down spectral ratio vs. chlorophyll-a concentration (right). Data from Martin N dock site on 8/28 was omitted from the comparison due to interference from turbid water.

FLI BUOY & DOCKSIDE DATA

The FLI meteorological and water quality monitoring buoy was redeployed in Owasco Lake during the 2020 field season. It revealed higher resolution but otherwise consistent changes in the water column as described in the CTD section (Fig. 17). More information is available in the companion report.

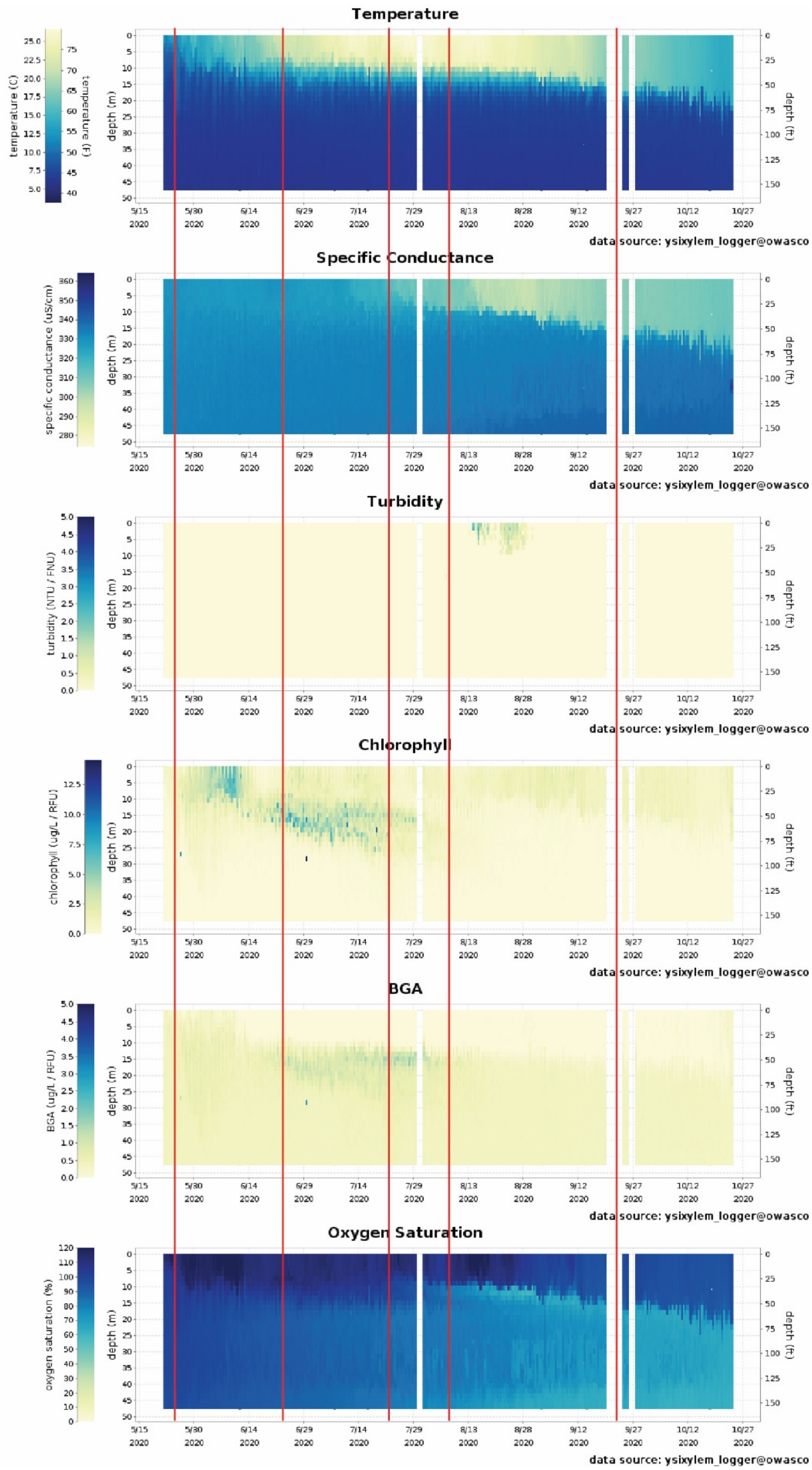


Fig. 17. Buoy water quality data in 2020. The data shown above are raw uncalibrated values. Subsequent calibrations altered the turbidity and chlorophyll data slightly (see companion report). The red lines depict the monthly monitoring cruise dates. The late August spike in turbidity occurred just after a major wind event.

STREAM MONITORING

Stream Discharge: The 2020 stream discharge data ranged from nearly dry conditions, 0.01 m^3/s in the small tributaries at Fire Lane 20 & 26 on 6/2 & 6/9, to 8.72 m^3/s in Owasco Inlet at Moravia on 5/5 (Table 6 in appendix, Fig. 18). These flows were significantly smaller than those detected in past years, and relates to the dry conditions in 2020. Flows were largest during the early spring reflecting more rain and saturated ground.

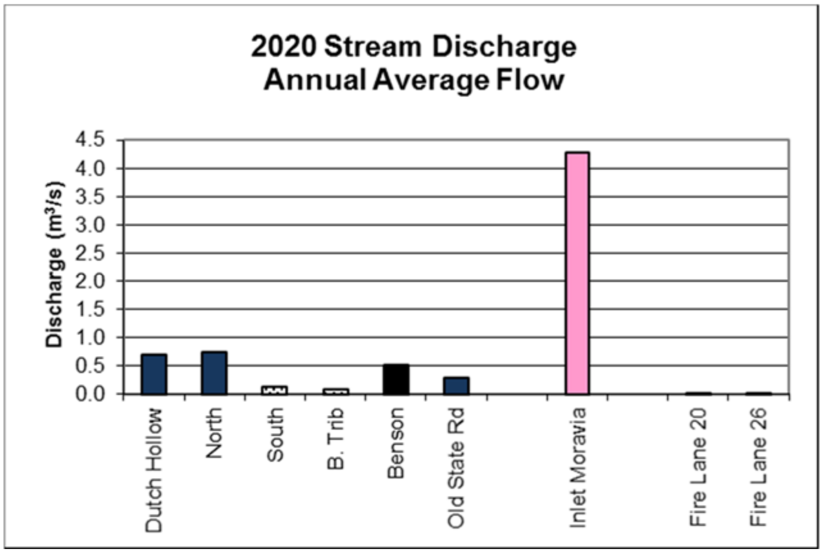


Fig. 18. Annual average stream discharge at each stream site in the Dutch Hollow Brook (purple), Owasco Inlet (pink), Fire Lane 20 and 26 watersheds. Tributary sites along Dutch Hollow Brook are stippled. Sites are arranged, left to right, from downstream to upstream.

Within Dutch Hollow Brook, mean annual discharge at each site 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. 18). 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 larger at North Rd than 38A this year like previous “dry” years, whereas the opposite was typically true during “wet” years. It suggests that surface runoff and groundwater inputs contributed to and increased stream discharge from North Rd down to Rt 38A during “wet” and “normal” years. Whereas, the stream loses water by evapotranspiration by plants, and/or infiltration into the permeable sand and gravel aquifer at the Dutch Hollow Brook delta, during “dry” years. The abnormally low water table depths in 2020 and other dry years support this conclusion.

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 km^2). Discharge was proportionally smaller at the two Fire Lane sites due to their smaller watersheds.

Seasonal Variability: Seasonally, the largest discharges in 2020 were detected in the spring at both Dutch Hollow and the Owasco Inlet whereas discharge in the summer and fall was progressively smaller based on the data logger estimated discharge for Dutch Hollow Brook at the Rt 38A site and the USGS gauge for Owasco Inlet (Fig. 19). The seasonal pattern is typical for this watershed during “dry” years. Fall flows typically increase over the summer minimums in “wet” and “normal” rainfall years.

Differences to Earlier Years: The 2020 annual mean discharge was “smaller” than those detected at Dutch Hollow Brook and Owasco Inlet over the past decade, except for 2016 for both streams and 2012 at Owasco Inlet (Fig. 20). These differences parallel changes in precipitation, lake levels and water table depths.

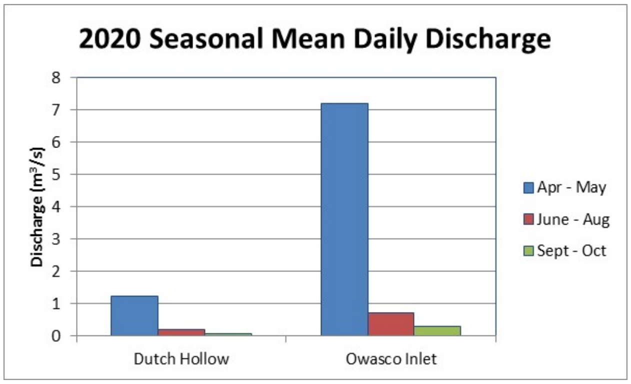


Fig. 19. Seasonal averaged stream discharge for the Rts. 38A and 38 sites, the terminal sites on Dutch Hollow Brook and Owasco Inlet, respectively.

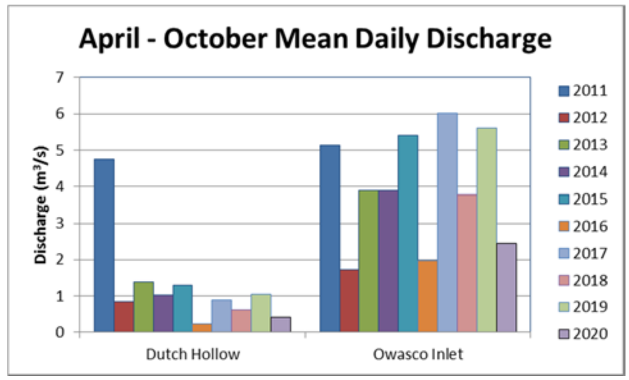


Fig. 20. Field season annual average stream discharge for the Rts. 38A and 38 sites. This plot used the estimated Dutch Hollow Brook data logger and USGS daily Owasco Inlet discharge data.

The Owasco Inlet (USGS Gauge, 4235299) mean, field-season, daily discharge of 4.1 m^3/s in 2020 indicates a very “dry” year, smaller than every year since 2009 but one, 2012 (Fig. 21). Similar variability was observed for the Owasco Outlet where the 2020 flow was smaller than every year in the record except for 2012 (USGS Gauge, 4235440, Fig. 21). The trends were consistent with annual and seasonal rainfall, lake levels and water table depths. Flow variability during any one year at the Owasco Outlet does not precisely parallel the Inlet because the Outlet has additional mandates on flow besides rainfall that include stabilizing lake levels to the seasonal rating curve levels, minimizing downstream flooding and other concerns.

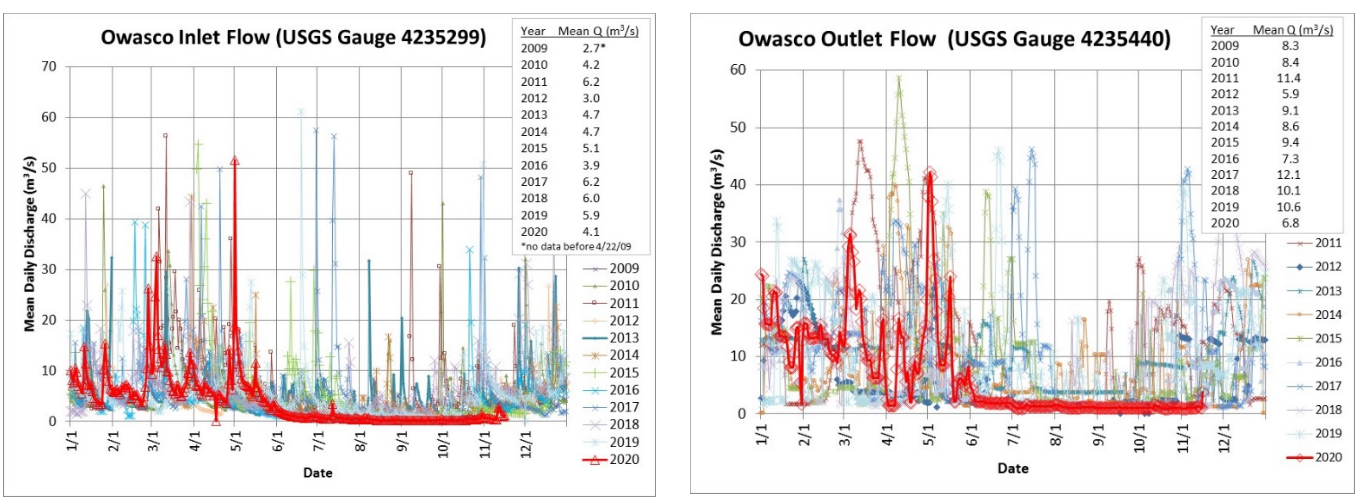
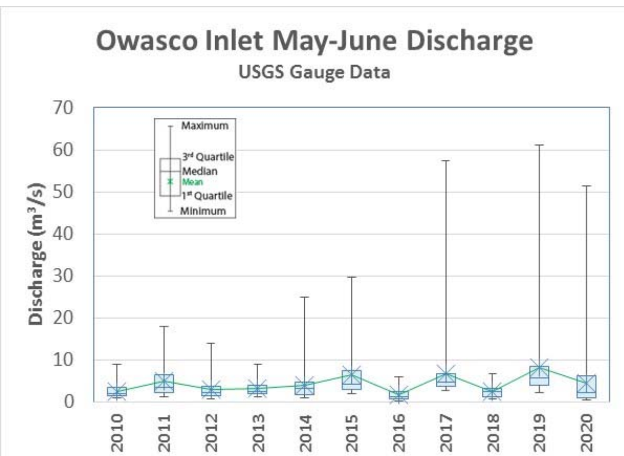


Fig. 21. Stream discharge for the Owasco Inlet near Moravia (left) – USGS Stream Gauge 4235299 and the Owasco Outlet (right) – USGS Stream Gauge 4235440.

Extreme Events: Both the Owasco Inlet and Dutch Hollow Brook revealed extreme 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 slightly lower mean flows during 2010, 2012, 2013, 2014, 2016, 2018 and 2020 (Fig. 22). The top whisker in the B&W plot, which reveals the maximum daily discharge, revealed significantly larger events in 2014, 2015, 2017, 2019 and 2020 than the other years in the record as well. A similar pattern is observed at Dutch Hollow Brook if the event during early July is included in the Dutch Hollow record (2017* Fig. 22). These events are critical because large events resulted in an exponentially larger impact on nutrient and sediment loads to the lake. The largest events in 2020 were similar to the largest events in any other "event" year and dislodged the data logger deployment at Rt. 38A. More intense rainfall events is another predicted outcome of Global Warming.



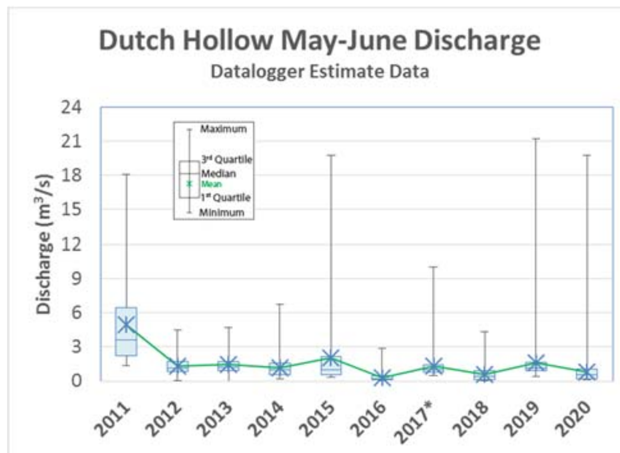


Fig. 22. Box and whisker plots of daily mean discharge during the May – June season over the past nine years for Owasco Inlet (left) near Moravia using the USGS Stream Gauge 4235299 data and Dutch Hollow Brook (right) using the Rt 38A data logger data.

Stream Nutrient & TSS Concentration Data: Total phosphate (TP) concentrations in the 2020 grab samples ranged from 1.3 to 9.0 μg/L, and averaged 4.0 μg/L at Rt 38A in Dutch Hollow Brook, ranged from 2.8 to 7.8 μg/L, and averaged 4.5 μg/L at Rt 38 in Owasco Inlet, and ranged from 1.3 to 9.0 μg/L, and averaged 3.5 μg/L at the two Fire Lane sites (Table 6 in appendix, Fig. 23). The ranges and means for TP, SRP, NOx, and TSS were similar to or slightly smaller than previous years (e.g., Dutch Hollow Brook data in Fig 24).

Along Dutch Hollow Brook, the Benson tributary site revealed the largest annual mean NOx, TSS and specific conductance (salinity) data than the other sites in 2020 (Fig 23). For example, the NOx annual mean concentration was 3.8 mg/L at the Benson tributary site. The next largest concentration was 2.4 mg/L at the South Trib site, and the other sites in this basin were below 1.5 mg/L. This year did not reveal the largest TP and SRP concentrations at the Benson tributary site although its SRP concentration was the 2nd largest (Fig. 24). The decline is attributable to the lack of runoff and/or successful BMPs in the huge farms upstream from the sample site. Its downstream impact was diluted by larger stream volumes downstream. It appears that the 2015/6 reduction of nutrients, suspended sediment concentrations and salinity at the Benson tributary site compared to other sites in the basin and a subsequent return to larger values in 2017 were due to decreased rainfall in 2016 and not due to other causes as previously speculated, as the decline was duplicated in 2020. The South tributary that drains agriculturally rich land to the south also revealed larger TP, SRP, and NOx.

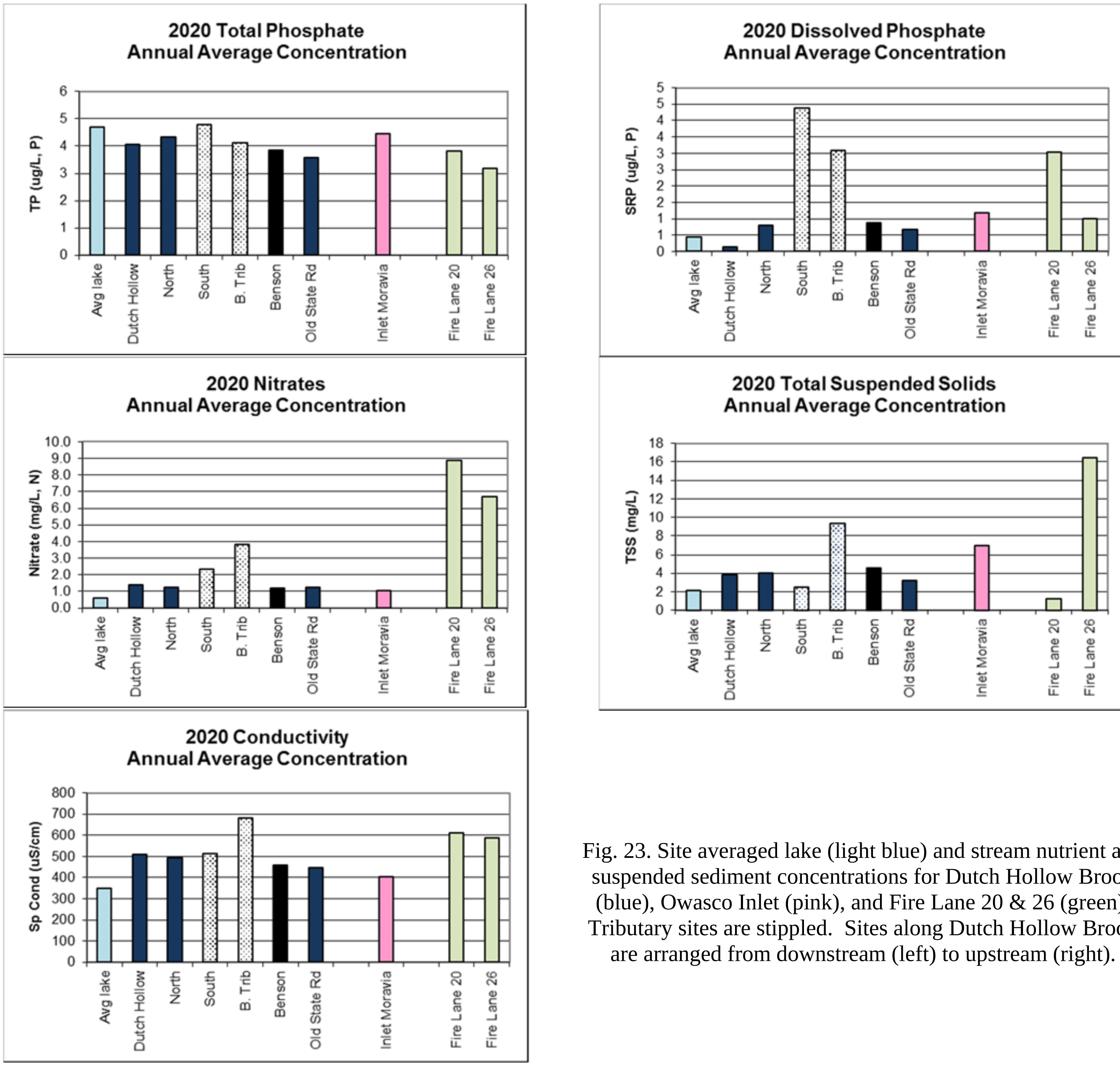


Fig. 23. Site averaged lake (light blue) and stream nutrient and suspended sediment concentrations for Dutch Hollow Brook (blue), Owasco Inlet (pink), and Fire Lane 20 & 26 (green). Tributary sites are stippled. Sites along Dutch Hollow Brook are arranged from downstream (left) to upstream (right).

Nutrient and sediment concentrations in Owasco Inlet at Moravia were similar to those at Dutch Hollow Brook. Total phosphate, soluble reactive phosphate and specific conductance concentrations at the two Fire Lane sites were similar to the other stream sites as well. In contrast, significantly larger NOx concentrations were detected at both Fire Lane sites, and larger total suspended sediments at Fire Lane 26. The larger NOx concentrations potentially reflect larger agricultural impacts at the headwaters of these tributaries. Previous years detected larger TP and TSS at these two small tributaries as well. Perhaps the dry conditions retarded the runoff of sediments and attached phosphorus and emphasizes contributions of NOx-rich groundwater to these tributaries. The timing of manure spreading, other agricultural inputs and groundwater nutrient concentrations are unknown, thus a more robust association cannot be concluded.

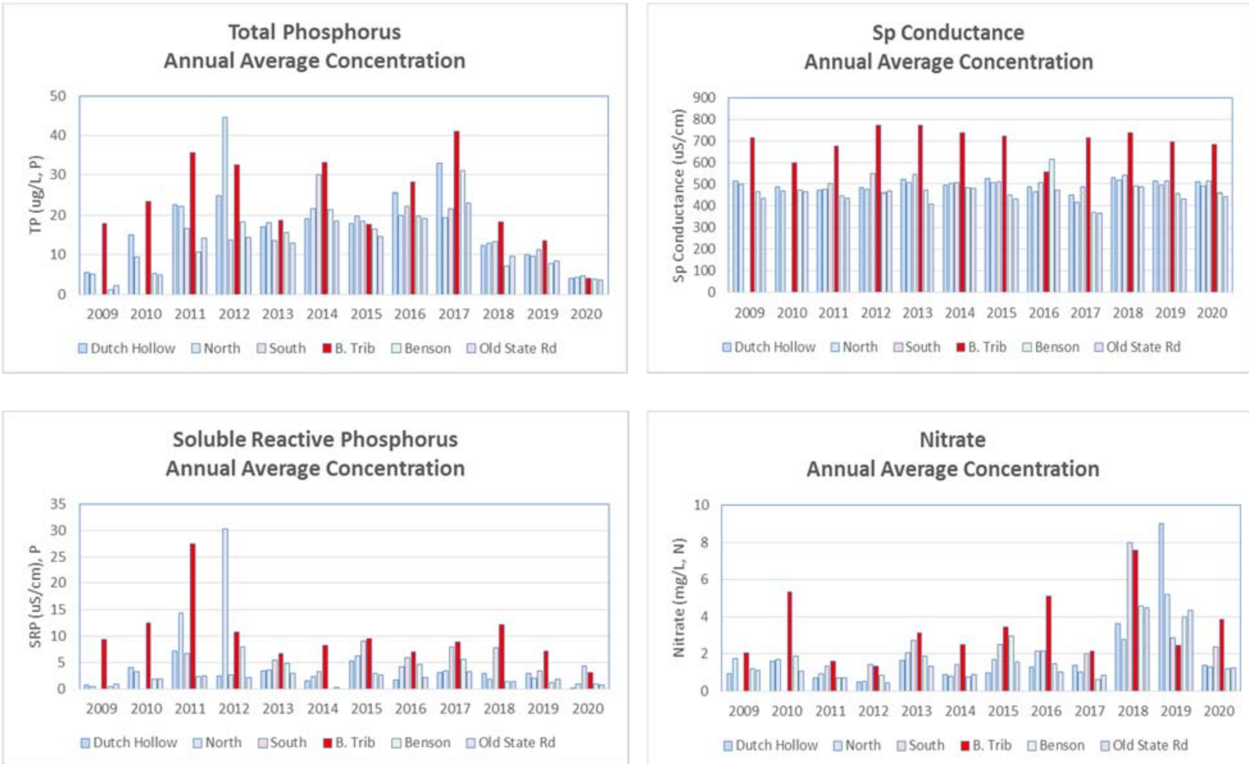
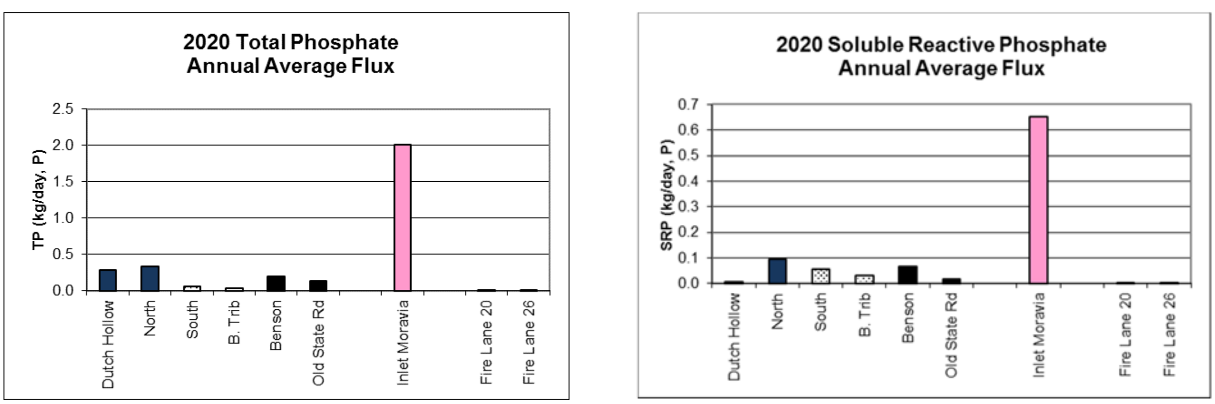


Fig. 24. Mean annual concentrations of total phosphorus, specific conductance soluble reactive phosphorus and nitrate at the Dutch Hollow Brook sites. Sites are arranged from downstream (left) to upstream (right) in each year.

Stream Fluxes: Dutch Hollow Brook revealed smaller mean fluxes of nutrients and sediments than Owasco Inlet (TP 0.1 vs. 1.6 kg/day; SRP 0.1 vs. 0.4 kg/day; TSS 400 vs. 610 kg/day; N 65 vs. 380 kg/day, respectively, Fig. 25). 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 2020 based on the four spring to early summer grab samples were similarly small like previous “dry” years (Fig. 26).



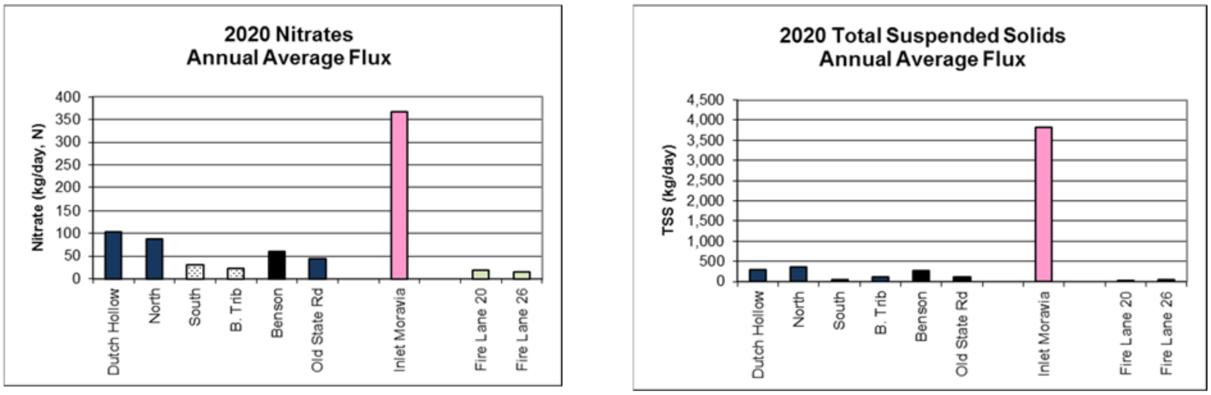
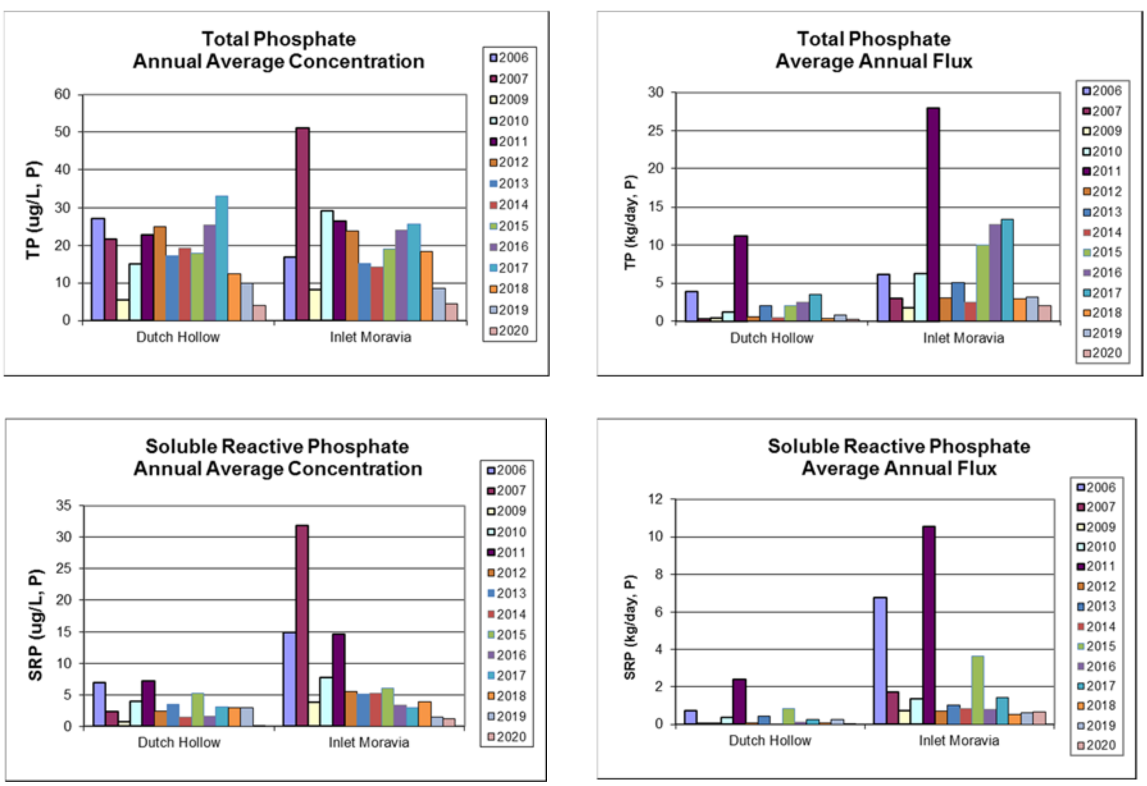


Fig. 25. Site averaged nutrient and sediment fluxes for Dutch Hollow (blue), Owasco Inlet (orange), and Fire Lane 20 & 26 (green). Tributary sites along Dutch Hollow Brook are stippled. Sites along Dutch Hollow Brook are arranged from downstream (left) to upstream (right).

At the small end of the spectrum, fluxes at the Dutch Hollow Brook tributary sites (Benson and South tributary 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. The phosphorus loading section below estimates loads by source.



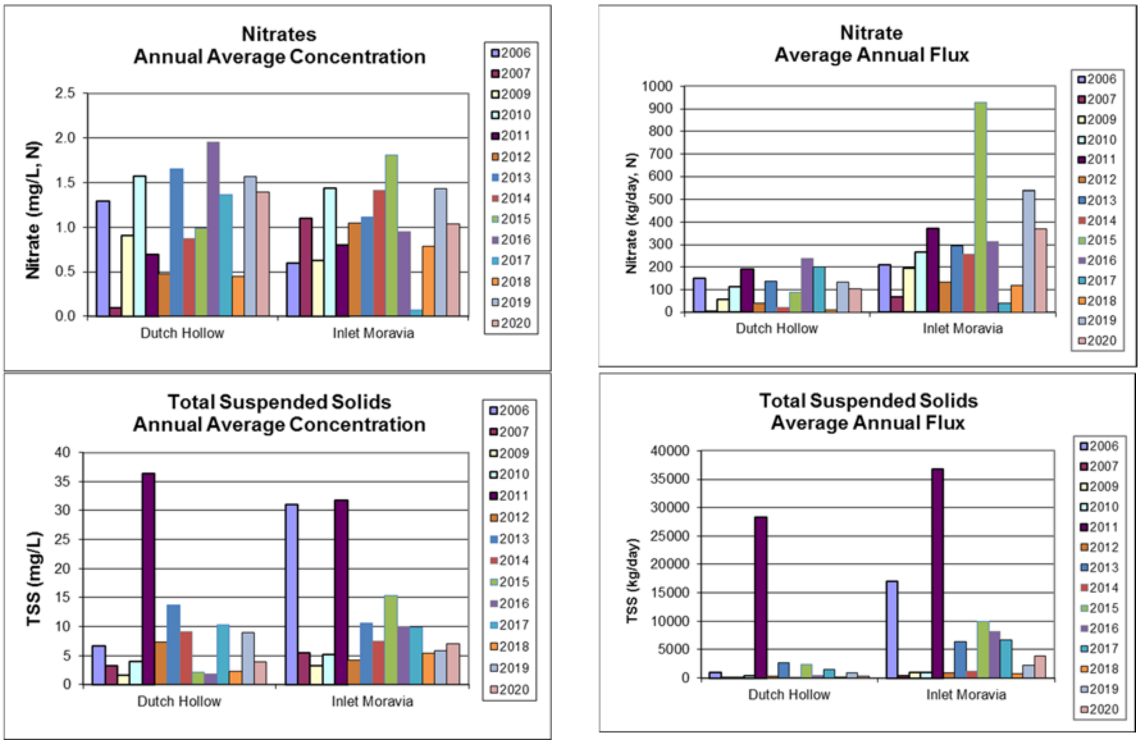


Fig. 26. 2006 through 2020 Annual average stream grab sample concentrations (left) and fluxes (right).

As in previous years, Dutch Hollow Brook steadily gained nutrients along its entire course up to North Rd. No one 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. This is 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. Interestingly, total phosphate, soluble reactive phosphate and total suspended sediment concentrations and fluxes decreased from the North St site downstream to 38A. Presumably the decline was due to the reduced flow, sediment deposition and nutrient uptake between these two sites.

The total phosphate contribution to the Owasco Inlet by Groton wastewater treatment facility has been significantly smaller since 2007 when the DEC mandated a phosphorus load reduction in the facility’s effluent (Fig. 27). The load contributed by the Moravia WWTF has been and continues to be very low as well. Since 2010, both facilities averaged ~0.1 to 0.3 kg/day.

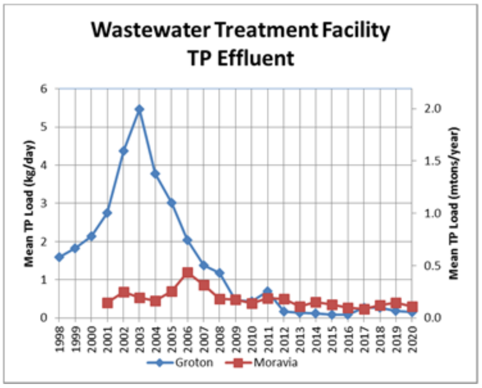


Fig. 27. Phosphorus loads from the Moravia and Groton wastewater treatment facilities.

EVENT SAMPLING AT DUTCH HOLLOW BROOK

Detailed Stage Data @ 38A along Dutch Hollow Brook: The 2020 stage data at Dutch Hollow Brook revealed textbook responses to the occasional precipitation event superimposed on a subdued but gradual spring through fall decline (Fig. 28). Each increase in stage corresponded to a precipitation event. Abrupt increases in stream stage for individual events ranged from approximately 5 to 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. 29). Significantly fewer precipitation events were detected in 2020 than earlier years, and except for the early May event, the size of these events were smaller than earlier years as well. The stream was close to dry conditions for most of the summer and fall, however it never completely dried up.

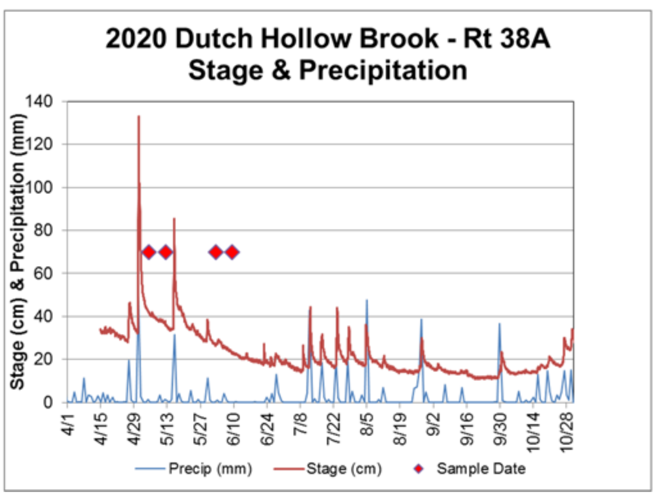


Fig. 28. Dutch Hollow Brook estimated discharge, precipitation, stream sample dates and measured discharge data for 2018 at Rt 38A. Precipitation data was from NY-CY-8, a CoCoRaHS station.

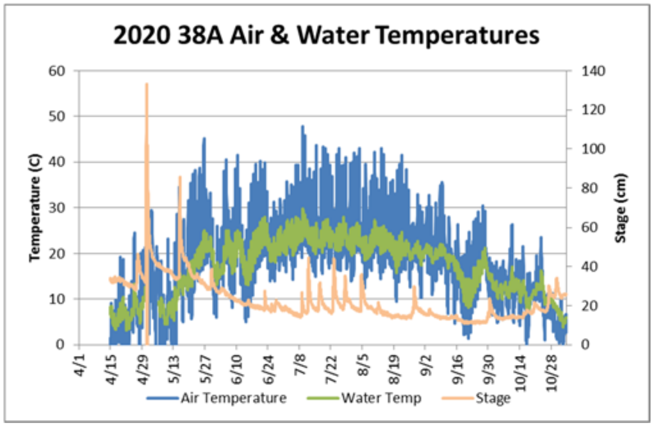


Fig. 29. Data logger mean stage, air and water temperature data at 38A.

Detailed "Event vs Base Flow" Results @ 38A: Nutrient and suspended sediment concentrations increased markedly during precipitation events in 2020 like earlier years (Fig. 30). Total suspended sediments (TSS) increased from an average base flow concentration of ~15 mg/L to an average event flow concentration of 260 mg/L, and rose to an event maximum of 2,500 mg/L on 7/28. Note: The event sample on 5/2 was not collected, as the flood dislodged the sampler intake. 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 highlights the importance of large events in 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. A few TSS concentrations were arbitrarily large because the pump intake was buried in the stream bed from 8/21 through 9/1 and again on 10/22, and sucked sediment from the stream bed.

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 11 and 9 μg/L to event means of 29 and 26 μg/L, respectively. Maximum event concentrations were 450 μg/L for TP and SRP. Again, 2020 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.

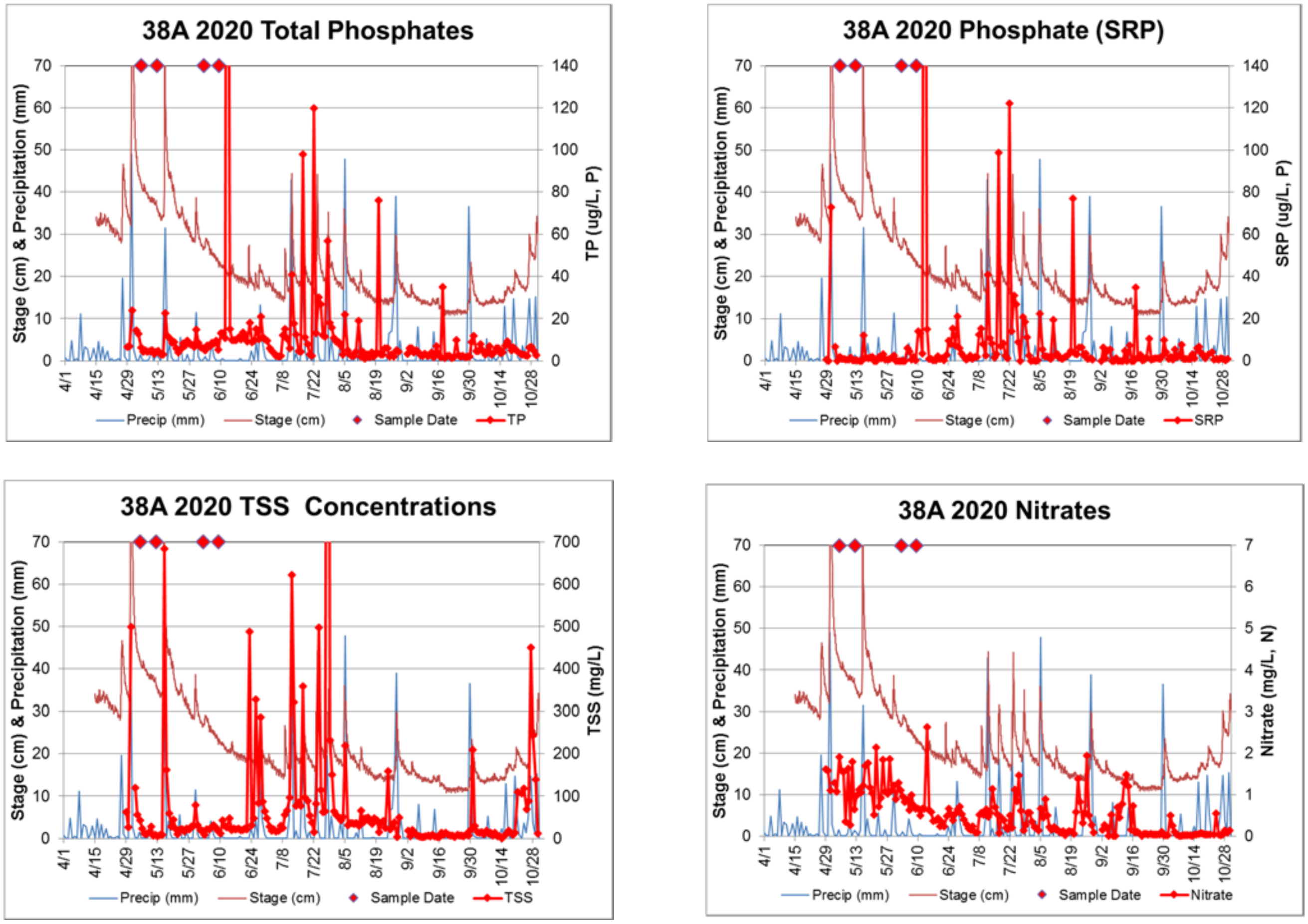


Fig. 30. 2020 daily nutrient and suspended sediment concentrations at Rt 38A.

The event *versus* base flow results suggests a number of potential remediation practices to reduce TP, SRP and 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 using 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).

Annual mean nitrate concentrations did not correlate to TP, SRP and TSS ($r^2 < 0.2$), suggesting a unique source. The maximum event NOx concentration was 2.6 mg/L, larger than the mean base flow concentration of 0.4 mg/L, but the event increase was much smaller 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 NOx as well. It indicates that runoff provided extra NOx to the stream but not to the same extent as the other nutrients. The difference us explained by nitrates unique chemistry. 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, and particles do not readily flow through groundwater systems. Thus groundwater does not transport TP, SRP and TSS. Precipitation events also rejuvenate near-surface groundwater flow, which contributed to the delayed NOx load as well. It extended the NOx events response as runoff flows faster than groundwater flow. The dry conditions in 2020 contributed to a larger groundwater contribution to stream flow, and larger base flow NOx concentrations as well.

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 ($r^2 = 0.95$). It established a stage/discharge rating curve for the site (Fig. 31).

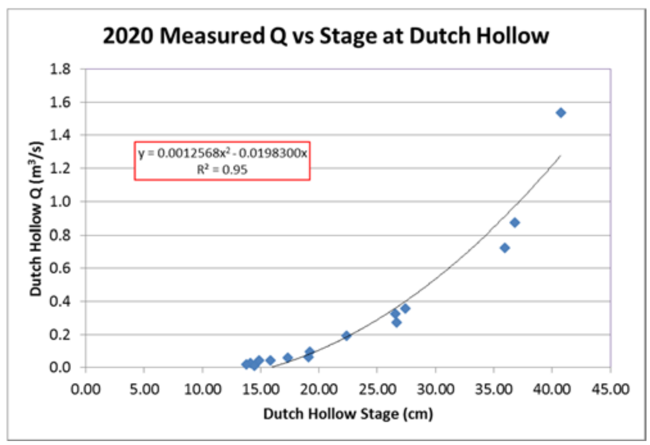


Fig. 31. Best fit correlations between weekly to bi-weekly measured discharge and data logger stage data at 38A site.

The TSS, TP, SRP and N fluxes were clearly event driven (Table 7, Fig. 32 & 33). In 2020, TSS, TP, SRP and N event *vs.* base flow fluxes at 38A averaged 19,560 *vs.* 175, 1.2 *vs.* 0.1, 1.9 *vs.* 0.1 kg/day and 80 *vs.* 4 kg/day, respectively. During the 2020 April through October deployment, Dutch Hollow provided 1,560,000 kg of sediment to the lake during events, but only 18,300 kg during base flow conditions. In a similar light, the 2020 events delivered 177 kg of TP, 152 kg of SRP and 6,510 kg of N to the lake compared to base flow contributions of 14 kg of TP, 11 kg of SRP and 450 kg of N. Over 90% of the suspended sediments, total phosphorus and soluble reactive phosphate were delivered to Owasco Lake by Dutch Hollow Brook during events over the course of this monitoring program (Table 7).

Annual changes were also observed. When TSS and TP are plotted against seasonal rainfall, the correlations explained up to 60% of the variance (Fig. 34 & 34). It suggests that other variables, for example, sediment storage, extent of cover crops, and nutrient management must come into play. The TP relationship to rainfall is stronger if the 2019 and 2020 data are excluded from the correlation ($r^2 = 0.8$ vs. 0.5), and suggests that remediation efforts in the watershed have recently decreased the TP loads below what rainfall totals predicted.

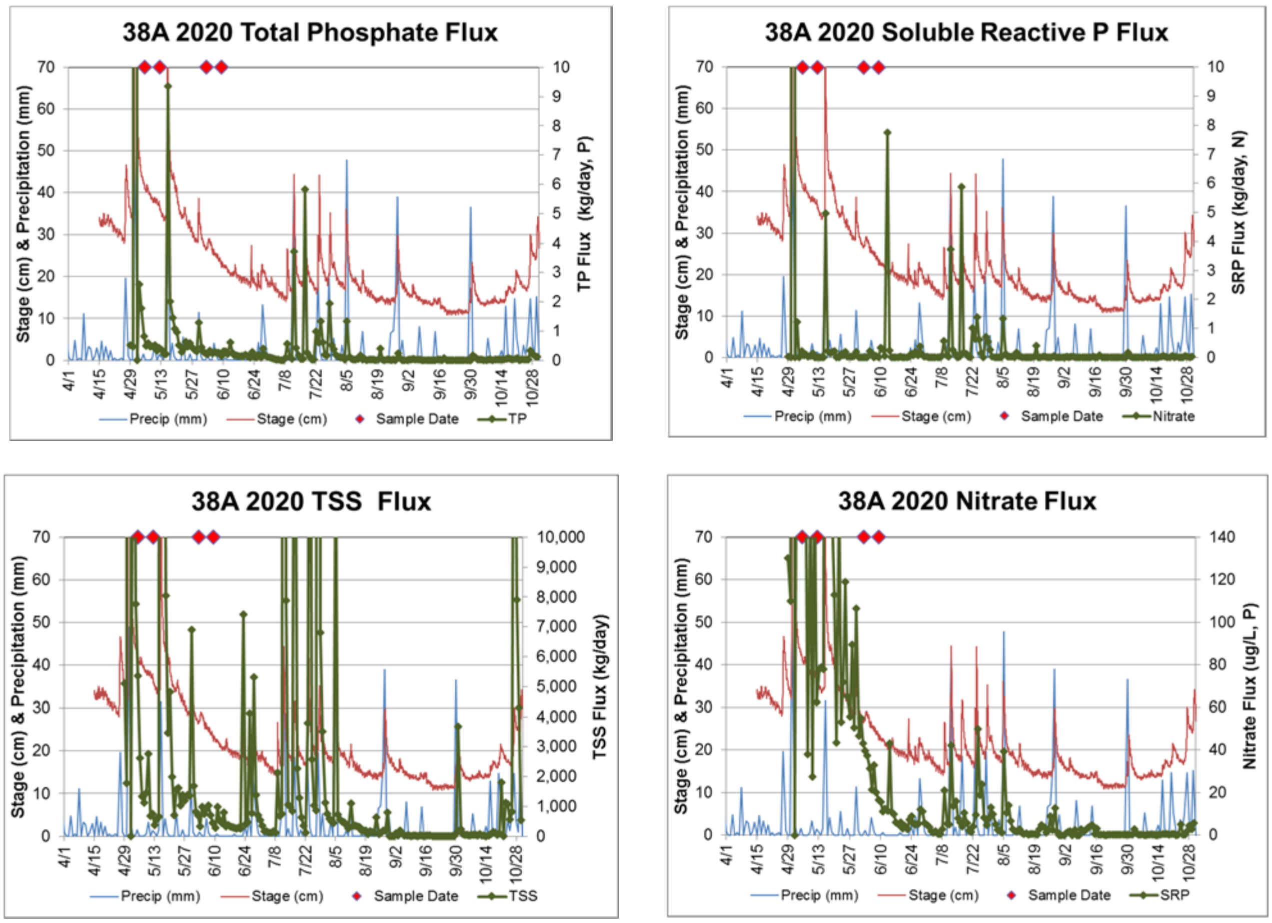


Fig. 32. Autosampler nutrient and suspended sediment fluxes.

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. 32). 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. Finally, farm fields are also typically tilled bare of vegetation in preparation for spring planting of crops, 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 2020 autosampler estimated a mean sediment flux of 8,500 kg/day, total phosphates 1.1 kg/day, dissolved phosphates 0.9 kg/day, and NOx 38 kg/day; whereas the grab sampling estimated an annual mean flux of 400 kg/day for sediments, 0.1 kg/day for TP, 0.1 kg/day for SRP, and 66 kg/day for NOx. The grab samples estimates for TSS, TP and SRP 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, i.e., the investigation of nutrient and sediment sources from within a watershed.

Table 7: 2011 – 2020 Autosampler Fluxes at Rt 38A Dutch Hollow Brook.

2011 (6/9-11/4)	TSS	NOx	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**	**NOx**	**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**	**NOx**	**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**	**NOx**	**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**	**NOx**	**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**	**NOx**	**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**	**NOx**	**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**	**NOx**	**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**	**NOx**	**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%
2020 (4/29-10/30)	**TSS**	**NOx**	**TP**	**SRP**
Mean (kg/day)	8,556	38	1.0	0.9
Event (kg/day)	19,557	81	2.2	1.9
Base Flow (kg/day)	175	4	0.1	0.1
% by events	99%	94%	93%	93%

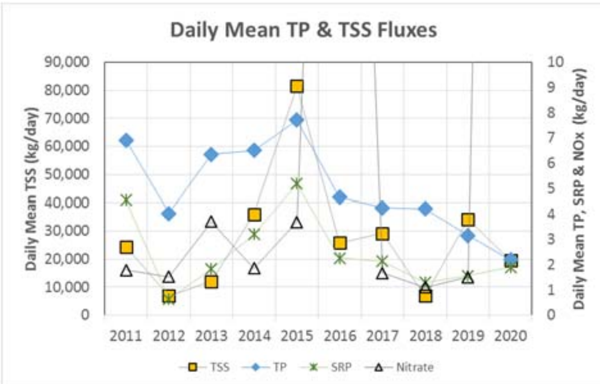


Fig. 33. Estimated mean daily total suspended solids, total phosphorus, soluble reactive phosphorus and nitrate loads at Dutch Hollow Brook since 2011.

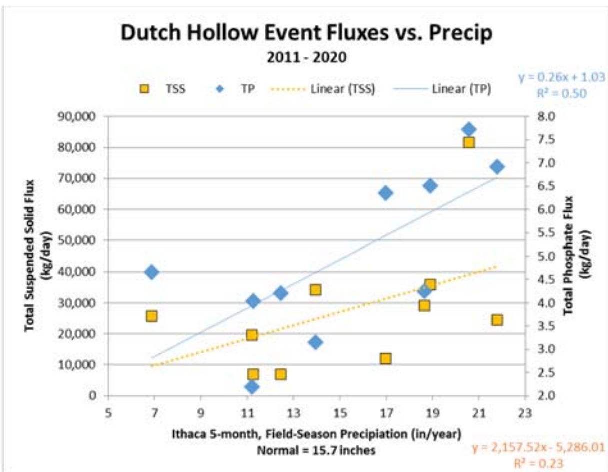


Fig. 34. Estimated annual total phosphorus loads vs May-September (5-month) rainfall at the Ithaca Airport.

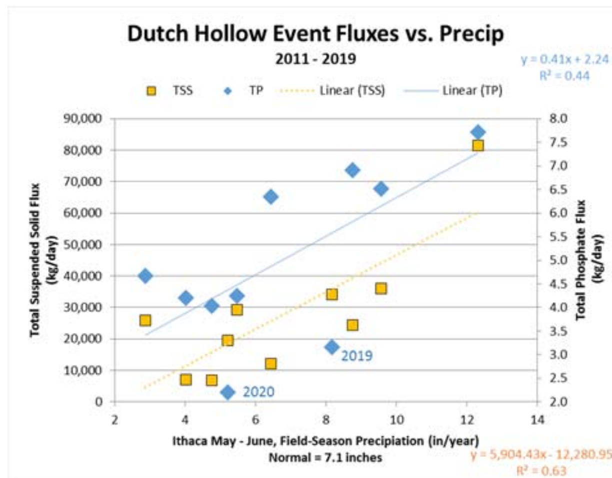


Fig. 35. Estimated annual total phosphorus loads vs May-June (2-month) rainfall at the Ithaca Airport.

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 increase of cyanobacteria 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 lawn fertilizers. Outputs must also be calculated to estimate the net change in phosphorus for the lake (Fig. 36). 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.

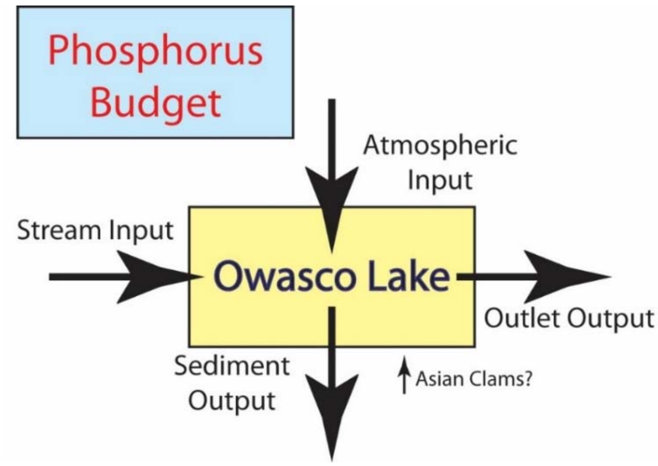


Fig. 36. The Owasco Lake phosphorus budget: Sources and sinks for phosphorus in Owasco Lake. Water quality improves if **inputs are less than outputs, for a number of years**.

The Inputs: The detailed 38A autosampler data calculated a mean total phosphate flux of 1.0 kg/day from Dutch Hollow Brook in 2020. Owasco Inlet delivered 2.0 kg/day based on the available 2020 stream grab data. The 2020 load from Owasco Inlet was estimated at 4.0 kg/day assuming a proportional change between the mean grab sample total phosphorus loads to the detailed autosampler loads from Dutch Hollow Brook. An extrapolation of fluxes and surface areas from Dutch Hollow Brook and Owasco Inlet to the entire Owasco watershed, estimated an annual input of 4.3 metric tons of phosphorus from every stream to the lake in 2020. The stream extrapolation incorporates all the 1st and 2nd order (small) tributaries like Fire Lane 20 and 26. The Moravia and Groton WWTF added a combined 0.2 metric tons P to the Owasco Inlet in 2020[10]. 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[11], 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. Lawn fertilizers are supposed to be phosphorus free, thus not a source. 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 cyanobacteria blooms (see companion report). 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 2020 influx of phosphorus is estimated at 5.6 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 1.0 metric tons of phosphorus escaped out the Outlet in 2020 assuming a 2020 annual mean total phosphate concentration in the lake of 4.7 μg/L, and a 2020 mean daily

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

discharge of 6.8 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 was 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 2020 efflux of phosphorus is estimated at 3.8 metric tons/year.

The Net Flux: Owasco Lake gained 1.8 metric tons of phosphorus in 2020. The input declined compared to earlier years but the output decline was proportionally larger. It suggests that more water evaporated from the lake in 2020 than years past, as the outflow was proportionally smaller in 2020 than previous years, removing less P than previous years. The long-term recent record is encouraging however. Since 2016, net P losses or steady state conditions were recorded. The recent, multi-year, negative or steady state P fluxes 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.

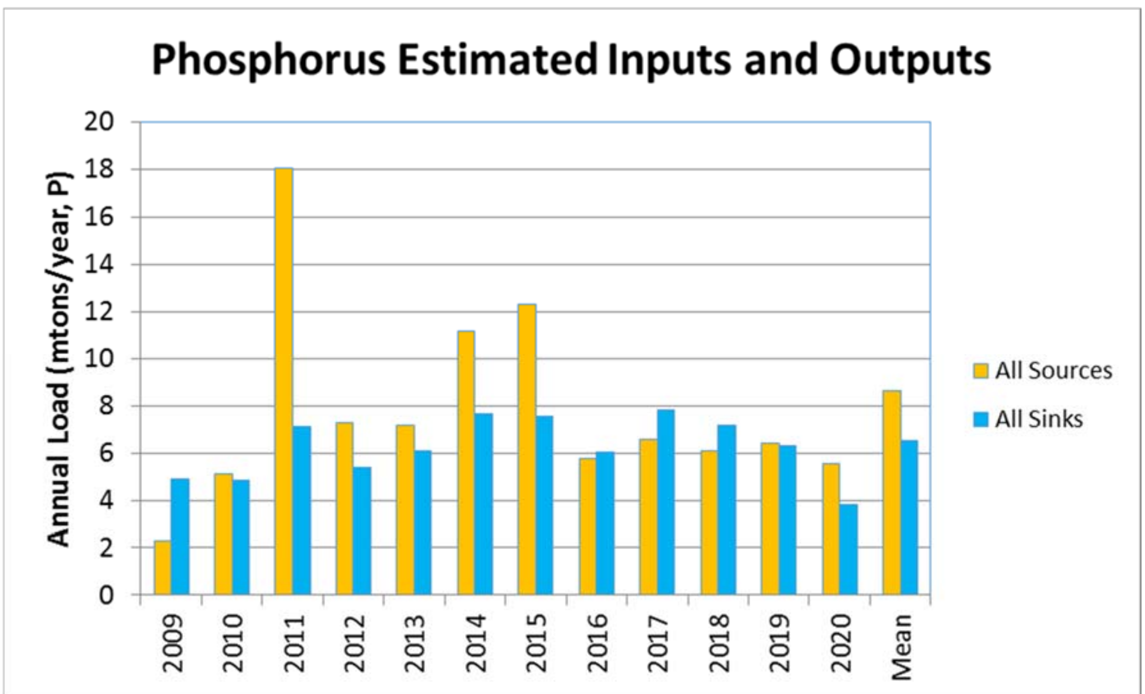


Fig. 37. Estimated annual total phosphorus inputs and outputs for Owasco Lake.

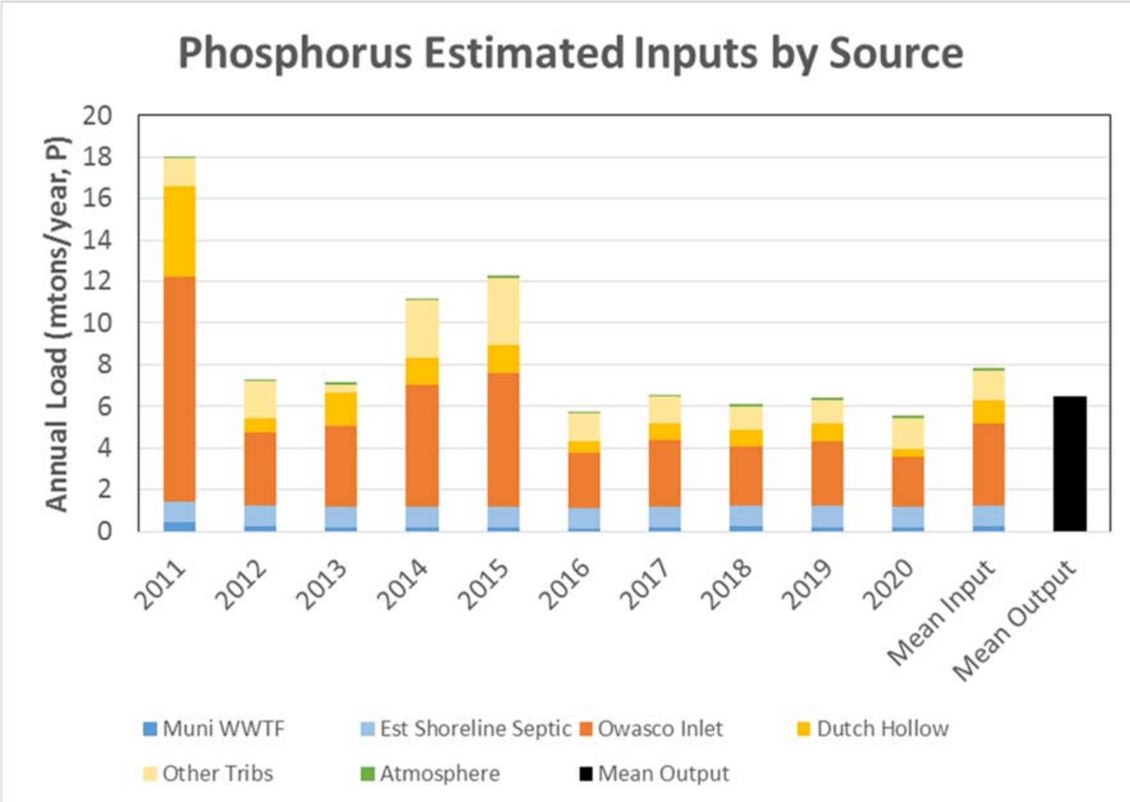


Fig. 38. Estimated annual inputs of phosphorus by source for Owasco Lake.

The total contribution from stream sources changed from year to year, whereas the inputs from other sources were relatively constant since 2011 (Fig. 38). 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 67 to 91% and averaging 82% of the total load. The lowest contribution by streams was in 2020, a very "dry" year, but even in 2020, streams still dominated the source of nutrients and sediments to the lake. 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, and small decline in 2020.
- Based on surface water soluble reactive phosphate and nitrate concentrations, 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.
- Even though nutrient loaded were reduced in 2020 (see below), the water quality in the lake declined slightly in 2020. The nutrient exports were even smaller and suggests that the net phosphorus budget might be a better predictor of water quality than nutrient loads or rainfall.
- Lake levels were low in 2020. The lower levels might have allowed more dead macrophytes to rot along the shoreline and/or eroded nutrient rich sediments previously too deep for disturbance by wave action. We speculate that lower lake levels may contributed to the larger numbers of cyanobacteria blooms in the lake in 2020.
- 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.

Cyanobacteria Blooms:

- The relative contribution of cyanobacteria to the algal population in the open lake increased in 2019 and increased some more in 2020.
- Details on cyanobacteria blooms are contained in a companion report that also focuses on the limnological and meteorological analysis at a number of dock 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, 2019 and 2020.
- The excessive nutrient loads during 2012, 2014 and 2015 were coincident with and perhaps triggered the onset of the cyanobacteria blooms in Owasco and many other Finger Lakes. Once these loads triggered the initial blooms, cyanobacteria have typically returned in larger numbers to the same nearshore locations.
- Segment analysis did not identify significant point sources along Dutch Hollow Brook, and indicates that non-point sources of nutrients and sediments dominate loads in this watershed.
- Both the Moravia and Groton municipal wastewater treatment facilities have done an amazing job to keep 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 nutrients and sediments 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 and/or the summer months.
- Total phosphorus, soluble reactive phosphates and total suspended sediments correlated to spring precipitation total although less phosphorus were transported in 2019 and 2020 than rainfall would have predicted. Perhaps recent remediation efforts reduced TP loads. The watershed’s phosphorus budget indicated that Owasco Lake gained phosphorus in 2020. The estimated input of phosphorus declined from earlier years but the output declined even more due to the dry conditions and lower flow out the outlet. The negative or near equilibrium conditions since 2016 is very encouraging, and suggests that the remediation efforts in the watershed are beginning to work.
- Partitioning phosphorus loads by source, 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, even in very “dry” years.

- 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 slightly negative or zero net phosphorus fluxes for the watershed since 2016 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. Clearly, implementation of the revised Owasco Lake Watershed Rules and Regulations is critical to the health and wellbeing of the lake. The sooner they are followed the sooner the lake might return to its original oligotrophic state. Example strategies follow:

- 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.
- 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.
- 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 the 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 cyanobacteria blooms themselves should be vacuumed before they disappear (feasible?).
- 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.
- Bio-manipulation is a poor option because lake-wide recreation is too vital for the economy.
- 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.

ACKNOWLEDGEMENTS
The 2020 research was supported by Cayuga County Legislature, the Fred L. Emerson Foundation, and the Finger Lakes – Lake Ontario Watershed Protection Alliance. We thank members of the Cayuga County Planning Department, Cayuga County Water Quality Management Agency, Owasco Lake Watershed Management Council, Cayuga County Health Department, Owasco Watershed Lake Association, the Cayuga County Soil and Water District, the Institute for the Application of Geospatial Data, the Finger Lakes – Lake Ontario Watershed Protection Alliance, and NYS Department of Environmental Conservation for their help over the past few decades of effort. Numerous individuals helped with many aspects of this study including Senator Mike Nozzolio, Barbara Halfman, Steven Cuddeback, Bill Graney, Mike Didio, Gary Searing, Ed Wagner, Keith Batman, Eileen O-Connor, Bruce Natale, Steve Lynch, Anthony DeCaro, Katie Jakaub, Joe Wasileski, Charlie Green, Jim Beckwith, Dana Hall, Peter Rogers, Joe Leonardi, Bob Brower, Ron Podolak, Judy Wright, Doug Kierst, Andrew Snell, Timothy Schneider, Michele Wunderlich, Adam Effler, Marion Balyszak, Lisa Cleckner, Roxanne Razavi, Martha Bond, Jerry Buckley, Scott Kishbaugh, Rebecca Gorney, Anthony Prestigiacomo, Lew McCaffrey, Aimee Clinkhammer, Scott Cook, Todd Walter and David Eckhardt. Hopefully, I didn't forget to acknowledge someone, and my apologies to those I omitted.

Table 2. 2020 Lake Data.

2020 Owasco Lake Site Averaged and Date Averaged Data

Site Averaged Surface Water Data

Site	Secchi Depth	Suspended Solids	Total Phosphate	Dissolved Phosphate	Nitrate	Silica	Chlorophyll
	(m)	(TSS, mg/L)	(TP, ug/L)	(SRP, ug/L)	(N, mg/L)	(Si, ug/L)	(a, ug/L)
1	3.3	2.3	5.4	0.3	0.6		3.7
Buoy	3.2						
2	3.8	1.9	4.0	0.6	0.6		3.7
Martin N Dock	--			16.7	0.2		8.0
Martin S Dock	--			0.3	0.7		8.1
Burtis Dock	--			0.5	0.4		5.3
FL-20 Dock	--			0.0	0.5		3.2
Average	**3.4**	**2.1**	**4.7**	**3.1**	**0.5**	**No Data**	**5.3**

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	---	0.9	3.6	1.2	1.1		0.2
Buoy							
2	---	1.1	3.7	0.9	1.2		0.2
Average	**---**	**1.0**	**3.7**	**1.0**	**1.1**	**No Data**	**0.2**

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)
5/21/20	5.0	2.0	3.4	0.5	1.0		0.5
6/23/20	5.1	1.1	4.5	0.1	0.6		2.4
7/21/20	3.6	1.6	3.5	0.5	0.9		2.2
8/5/20	3.7	1.4	5.3	0.0	0.7		2.0
8/11/20	3.6	2.1	3.2	0.0	0.6		2.3
8/18/20	2.1	3.5	6.9	0.0	0.4		6.5
8/25/20	1.7	3.5	4.3	3.6	0.5		7.2
9/1/20	2.4	2.4	5.7	0.0	0.5		5.5
9/8/20	3.3	2.4	6.9	0.0	0.6		2.7
9/15/20	3.7	1.9	5.1	0.0	0.3		5.2
9/22/20	3.6	1.6	3.0	0.4	0.4		4.2
Average	**3.4**	**2.1**	**4.7**	**0.5**	**0.6**	**No Data**	**3.7**

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)
5/21/20	---	2.0	2.3	1.6	1.0		0.0
6/23/20	---	0.8	3.8	0.5	0.7		0.1
7/21/20	---	1.3	2.9	0.1	1.2		1.1
8/5/20	---	0.6	2.6	1.2	0.9		0.0
8/11/20	---	1.7	3.0	0.0	1.5		0.0
8/18/20	---	0.9	8.4	0.4	1.4		0.0
8/25/20	---	1.1	6.1	5.1	1.1		0.3
9/1/20	---	0.7	2.7	0.2	0.5		0.3
9/8/20	---	0.6	3.0	0.0	1.6		0.4
9/15/20	---	0.5	2.6	0.0	2.0		0.2
9/22/20	---	0.8	2.7	1.9	0.9		0.1
Average	**---**	**1.0**	**3.7**	**1.0**	**1.1**	**No Data**	**0.2**

Table 4. Annual Average Plankton Data from 2005 through 2020, and Daily Average Data for 2020.

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		0.9	2.5	3.2	10.3	2.8		0.3
2006 Average	24.3	1.7	0.0	7.1	1.4	0.7	2.6	41.5	0.7		0.2	2.4	0.8	0.3	0.6	0.1	3.8
2007 Average	30.0	0.5	0.0	23.3	0.2	2.1	3.8	12.9	0.7		0.4	0.6	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
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
5/21/20	2.7	0.3	10.9	78.3	0.6	0.0	0.0	0.1	0.0	0.0	0.9	0.0	0.0	0.0	0.0	0.0	0.0
6/23/20	0.0	0.0	0.0	97.7	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.6
7/21/20	0.8	0.0	61.1	1.7	0.0	0.0	0.0	1.2	0.0	0.0	0.6	12.2	0.0	3.6	0.7	2.2	11.5
8/5/20	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.9	0.9	0.2	0.5	0.3	0.4	92.1
8/11/20	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.3	0.0	0.0	2.2	0.5	1.2	89.5
8/18/20	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	97.5
9/1/20	1.5	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	3.7	0.0	5.2	0.9	1.8	85.4
9/8/20	8.8	0.0	0.0	0.0	0.0	0.2	0.0	1.3	0.0	0.0	0.2	9.7	0.0	17.5	0.2	14.2	41.8
9/15/20	13.4	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.4	3.6	0.0	17.8	0.0	19.0	38.7
9/22/20	13.9	0.0	0.0	0.0	0.0	2.9	0.0	23.1	0.0	0.0	0.2	3.4	0.0	3.9	0.2	5.5	39.5
2020 Average	4.1	0.0	7.2	17.8	0.1	0.3	0.0	2.8	0.0	0.0	0.4	3.3	0.0	5.1	0.3	4.4	49.6

Table 5: Annual Average 2020 Lake Data from the Finger Lake Survey.

2020 Average Values (± 1s)	Honeoye	Canandaigua	Keuka	Seneca	Cayuga	Owasco	Skaneateles	Otisco
Secchi Depth (m)	3.2 ± 1.6	5.5 ± 1.1	6.2 ± 1.1	3.9 ± 1.3	3.8 ± 1.0	4.1 ± 0.8	8.4 ± 1.7	Not Sampled
Total Suspended Solids (mg/L), Surface	3.1 ± 3.0	1.1 ± 0.8	1.9 ± 2.4	1.6 ± 1.2	1.6 ± 0.7	1.7 ± 0.5	0.6 ± 0.3	
Total Suspended Solids (mg/L), Bottom	3.0 ± 2.5	0.3 ± 0.1	0.7 ± 0.6	1.0 ± 1.2	1.7 ± 1.3	1.2 ± 0.8	0.4 ± 0.2	
Total Phosphate (µg/L, TP), Surface	6.7 ± 1.6	2.8 ± 1.1	2.9 ± 1.0	4.3 ± 3.3	3.3 ± 0.6	3.8 ± 1.2	2.4 ± 0.7	
Total Phosphate (µg/L, TP), Bottom	6.0 ± 1.6	1.9 ± 0.6	2.8 ± 1.2	3.3 ± 1.9	8.9 ± 5.2	2.9 ± 1.0	2.1 ± 1.3	
Dissolved Phosphate (µg/L, SRP), Surface	1.5 ± 1.6	0.2 ± 0.2	0.6 ± 0.9	1.8 ± 2.3	0.6 ± 0.9	0.2 ± 0.3	0.4 ± 0.5	
Dissolved Phosphate (µg/L, SRP), Bottom	1.0 ± 1.3	0.4 ± 0.4	0.7 ± 0.7	2.9 ± 2.6	8.9 ± 5.9	0.8 ± 1.3	0.9 ± 1.5	
Nitrate as N (mg/L), Surface	0.1 ± 0.3	0.3 ± 0.5	0.2 ± 0.3	0.2 ± 0.3	0.7 ± 0.4	0.6 ± 0.3	0.3 ± 0.1	
Nitrate as N (mg/L), Bottom	0.0 ± 0.0	0.3 ± 0.4	0.2 ± 0.1	0.7 ± 1.5	1.0 ± 0.3	1.2 ± 0.6	0.3 ± 0.1	
Silica (SR µg/L), Surface	Not Measured							
Silica (SR µg/L), Bottom	Not Measured							
Chlorphyll a (µg/L), Surface	7.3 ± 6.5	1.9 ± 1.5	1.0 ± 0.6	2.4 ± 1.8	2.0 ± 1.5	2.9 ± 2.0	0.6 ± 0.5	
Chlorphyll a (µg/L), Bottom	5.5 ± 8.6	0.1 ± 0.1	0.3 ± 0.2	1.3 ± 2.0	0.2 ± 0.3	0.1 ± 0.1	0.2 ± 0.3	

Table 6. 2020 Stream Data.

2020 Stream Segment Analysis Data							
Date & Location	**Discharge**	**Specific Conductance**	**Water Temp**	**Nitrate**	**Suspended Solids**	**Total Phosphate**	**Phosphate SRP**
	(m^3/s)	(µS/cm)	(^oC)	(mg/L, N)	(mg/L)	(µg/L, TP as P)	(µg/L, SRP as P)
5/5/2020							
Dutch Hollow 38A	1.54	530	7.7	1.9	4.8	5.4	0.1
Dutch Hollow North St	1.47	485	7.3	1.2	7.3	7.5	2.7
Dutch Hollow South Trib	0.31	487	8.0	2.5	4.2	5.6	6.0
Dutch Hollow Benson Trib	0.17	496	10.0	2.0	29.2	6.4	7.0
Dutch Hollow Benson Rd	0.97	423	7.5	1.1	9.3	6.1	3.1
Dutch Hollow Old State Rd	0.70	393	9.1	2.0	5.7	7.4	0.2
Owasco Inlet Moravia Rt 38*	8.72	350	8.7	0.4	14.1	7.8	1.7
Fire Lane 20	0.04	631	8.1	8.4	1.3	3.0	0.1
Fire Lane 26	0.04	628	9.4	6.5	3.6	5.9	0.2
5/12/2020							
Dutch Hollow 38A	0.72	482	7.4	1.7	7.8	3.3	0.0
Dutch Hollow North St	0.88	461	6.7	2.1	5.0	3.0	0.5
Dutch Hollow South Trib	0.13	492	6.8	3.2	3.6	4.0	2.3
Dutch Hollow Benson Trib	0.08	713	8.6	6.4	2.9	2.1	1.1
Dutch Hollow Benson Rd	0.63	428	7.1	2.1	4.3	2.0	0.0
Dutch Hollow Old State Rd	0.30	413	7.7	1.7	2.4	1.8	2.0
Owasco Inlet Moravia Rt 38*	5.52	373	7.7	1.9	8.4	2.8	2.7
Fire Lane 20	0.03	629	6.5	16.8	1.6	1.4	0.4
Fire Lane 26	0.03	599	8.5	9.6	52.7	1.3	1.1
6/2/2020							
Dutch Hollow 38A	0.32	506	14.7	1.2	1.5	4.3	0.0
Dutch Hollow North St	0.37	511	14.3	0.7	1.9	3.0	0.0
Dutch Hollow South Trib	0.06	524	13.4	2.6	1.1	4.8	4.2
Dutch Hollow Benson Trib	0.03	752	14.0	4.2	2.7	4.4	4.0
Dutch Hollow Benson Rd	0.29	490	14.1	1.0	2.0	4.8	0.3
Dutch Hollow Old State Rd	0.08	480	13.5	0.2	2.4	2.8	0.5
Owasco Inlet Moravia Rt 38*	1.62	433	13.9	0.9	2.9	2.8	0.3
Fire Lane 20	0.02	600	12.4	7.2	1.0	1.9	1.8
Fire Lane 26	0.01	573	12.4	9.3	2.5	2.8	2.5
6/9/2020							
Dutch Hollow 38A	0.19	525	17.3	0.7	1.4	3.3	0.4
Dutch Hollow North St	0.21	511	17.5	1.1	2.0	3.9	0.0
Dutch Hollow South Trib	0.02	551	16.0	1.1	1.0	4.6	5.0
Dutch Hollow Benson Trib	0.03	765	18.4	2.7	2.7	3.6	0.3
Dutch Hollow Benson Rd	0.15	499	17.8	0.6	2.5	2.6	0.0
Dutch Hollow Old State Rd	0.08	494	17.2	0.9	2.2	2.4	0.0
Owasco Inlet Moravia Rt 38*	1.21	465	18.3	0.9	2.7	4.4	0.0
Fire Lane 20	0.01	582	16.0	3.1	1.0	9.0	9.9
Fire Lane 26	0.01	554	15.9	1.4	6.9	2.8	0.1
**Used USGS Gauge at Moravia for Discharge Data*							
2020 Average Values							
Dutch Hollow 38A	0.69	510.68	11.78	1.39	3.88	4.07	0.13
Dutch Hollow North Rd	0.73	491.90	11.45	1.26	4.05	4.34	0.81
Dutch Hollow South Trib	0.13	513.55	11.05	2.36	2.48	4.78	4.38
Dutch Hollow Benson Trib	0.08	681.45	12.75	3.83	9.38	4.13	3.09
Dutch Hollow Benson Rd	0.51	460.00	11.63	1.21	4.53	3.85	0.87
Dutch Hollow Old State Rd	0.29	444.95	11.88	1.23	3.17	3.59	0.68
Owasco Inlet Rt 38 Moravia	4.27	405.35	12.15	1.03	7.03	4.45	1.18
Fire Lane 20	0.02	610.25	10.75	8.87	1.23	3.83	3.04
Fire Lane 26	0.02	588.30	11.54	6.69	16.43	3.19	1.00
2020 Average Fluxes				N kg/day	TSS kg/day	TP kg/day	SRP kg/day
Dutch Hollow 38A				102.4	297.5	0.28	0.0
Dutch Hollow North Rd				87.3	351.0	0.3	0.1
Dutch Hollow South Trib				29.7	40.1	0.1	0.1
Dutch Hollow Benson Trib				23.0	116.2	0.0	0.0
Dutch Hollow Benson Rd				60.4	274.9	0.2	0.1
Dutch Hollow Old State Rd				44.0	110.2	0.1	0.0
Owasco Inlet Rt 38 Moravia				367.8	3830.0	2.0	0.7
Fire Lane 20				19.4	2.5	0.0	0.0
Fire Lane 26				14.0	36.0	0.0	0.0