Spring 4-19-2019

Fish Assemblage Succession Within a Recovering Urban Lake

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FISH ASSEMBLAGE SUCCESSION WITHIN A RECOVERING URBAN LAKE

by

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A thesis
submitted in partial fulfillment
of the requirements or the
Master of Science Degree
State University of New York
College of Environmental Science and Forestry
Syracuse, New York
April 2019

Department of Environmental and Forest Biology

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ACKNOWLEDGEMENTS

I would like to thank my steering committee: Dr. Donald Stewart, Dr. Stephanie Johnson, and Danielle Hurley, for their time and expertise. I am especially grateful to my major professor, Dr. Neil Ringler, for his support and guidance throughout my graduate program.

I would like to give a special thank you to the Onondaga Lake field crews, particularly the 2017 and 2018 cohorts: Erik Hazelton, Deb Hummel, Zach Davis, Ryan Shaw, Sean Korbas, Michaela Kenward, Jane Van Vessem, Carrick Palmer, Joe Sullivan, and Jacob Wojcik. The long-term monitoring of Onondaga Lake would not have been possible without their countless hours of work. Research on Onondaga Lake has been supported by Honeywell International, Incorporated and Parsons Corporation, with significant guidance by Mark Arrigo, Anne Burnham, Jesse Carr, and Matt McDonough.

The support of my graduate colleagues has been invaluable; I owe Alex Kua, Ben Gallo, John Zeiger and Kate Abbott for helping to preserve my sanity. Lastly, I'd like to thank my parents for their encouragement and enthusiasm, even when they were not sure what I was talking about.

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ABSTRACT

G.R. Kronisch. Fish Assemblage Succession Within a Recovering Urban Lake. 66 pages, 6 tables, 10 figures, 2019. AFS style guide used.

Onondaga Lake in Syracuse, New York was once the site of prolific chemical and municipal sewage dumping. However, over the last two decades it has become the target of restoration efforts including the rehabilitation of the fish assemblage. This study compared species richness and Shannon diversity between lake basins and over time, in conjunction with multivariate ordination to assess changes in fish assemblage structure. Species richness of offshore fish increased in this timeframe; however, both richness and diversity declined for the nearshore fish assemblage. There was significant annual variability in species composition for both offshore and nearshore samples based on permutational analyses of variance, but only the composition of offshore samples was significantly different between basins. These results suggest that offshore fish have been responding positively to increasing water quality, while the nearshore fish assemblage has likely been negatively impacted by nearshore habitat homogenization from introduced aquatic invasives.

Key Words: Restoration, Onondaga Lake, Community Ecology, Remediation, Species Richness, Rarefaction, Diversity, PERMANOVA, Nitellopsis

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PREFACE

Onondaga Lake (Onondaga County, New York) has been negatively impacted by centuries of anthropogenic disturbances (Table 1). The City of Syracuse grew around the southeastern shore of Onondaga Lake largely because of the natural brine springs nearby and subsequent commercial salt production (Effler 1996). Once a mesotrophic aquatic system, Onondaga Lake became eutrophic during the early 1800s and then hypereutrophic in the late 1940s from excessive municipal wastewater inputs from Syracuse containing high nutrient levels (Hennigan 1989; Rowell 1996). In addition to municipal effluent dumping, ionic "Solvay waste" byproducts, mostly comprised of calcium chloride and calcium carbonate, were introduced along the western shoreline as the salt industry transitioned to soda ash production (Effler et al. 1996). The filling of local wetlands with Solvay waste was done to combat malaria. This resulted in an increased trophic state of Onondaga Lake and corresponded with the loss of native coldwater species, including Atlantic Salmon (*Salmo salar*) and the Onondaga Lake whitefish (likely *Coregonus artedii)*(Ringler et al. 1996). The fish assemblage of Onondaga Lake has since transitioned to favor warmwater species, including Common Carp (*Cyprinius carpio*), Largemouth Bass (*Micropterus salmoides*), and Yellow Perch (*Perca flavescens*) (Beauchamp 1908; Greeley 1928; Ringler et al. 1996).

Onondaga Lake was already toxic with sewage and ionic waste before the construction of a chlor-alkali facility on the western shoreline in 1946, which manufactured chlorine and other organic chemicals which further contributed to the pervasive degradation of the lake. Due to high turbidity, pollution, and poor substrate only eight fish species and a single macrophyte species (*Potamogeton pectinatus*) were found in the lake at this time (Stone and

Pasko 1946). Although the US Department of Justice took legal action against the chlor-alkali facility in 1951 to reduce mercury inputs, nearly 76,000 kg of mercury had been discharged by the time of its closure in 1986 (Effler 1987; Effler and Hennigan 1996). At that time mercury concentrations in fish were comparable to those in the St. Claire River (Effler 1987).

The Metropolitan Wastewater Treatment Plant (METRO) underwent numerous upgrades between 1960 and 1981 to improve effluent quality and reduce organic phosphate, which corresponded to an increase in lake fish species richness from 8 in 1946, to 16 in 1969, and to 22 in 1980 (Stone and Pasko 1946; Noble and Forney 1971; Chiotti 1981). This phosphorus reduction directly correlates to increases in species richness, likely attributed to improved water quality, fish habitat, and reductions in algal blooms (Murphy et al. 2015). However, continued effluent discharge exceedances spurred legal action by New York State against Onondaga County in 1989. This lawsuit resulted in green infrastructure projects throughout Syracuse to reduce combined sewer overflow and METRO upgrades to further reduce ammonia in the effluent to the lake (Mahoney 2017).

Shortly after the legal suit began against Onondaga County, AlliedSignal, the owner of the chlor-alkali manufacturing plant, was sued by New York State under the federal Comprehensive Environmental Response, Compensation and Liability Act (CERCLA), or "Superfund", for polluting Onondaga Lake with ionic and mercury wastes (Effler and Hennigan 1996). The lake was formally added to the Superfund High Priority List in December 2004 (USEPA 2018). AlliedSignal, now Honeywell Incorporated, was mandated to remediate the mercury-polluted sections of Onondaga Lake, which was mainly conducted via dredging and capping of the southwestern shoreline between 2012 and 2016 (Parsons 2018). More recently this operation has been enhanced by whole-lake nitrate additions to inhibit methylation of benthic mercury in uncapped lake sections of Onondaga Lake, construction and enhancement of existing wetlands, and the addition of nearly 2000 habitat structures (Matthews et al. 2013; McAuliffe 2017).

Table 1. Condensed Timeline of Human Impacts on Onondaga Lake (adapted from Effler 1987; Hennigan 1989; Effler and Hennigan 1996; Rowell 1996; Tango and Ringler 1996; Spada et al. 2002; Matthews et al. 2013; Smith et al. 2015).

Year	Event
1794	Salt commercially produced on Onondaga Lake
1822	Onondaga Lake drawn down to match Seneca River; Lake area reduced by 20%
1825	City of Syracuse, New York established
1884	Soda ash production begins via the Solvay process
~1890	Ionic waste from Solvay process directly dumped into Onondaga Lake
1896	Sewers completed to direct raw sewage into lake
1898	Loss of cold water fishery in Onondaga Lake (whitefish, eels)
~1900	Commercial salt production switches to industrial development
1928	Syracuse begins primary treatment of wastewater
1940	Swimming banned
1946	Begin of mercury loading into lake from steel manufacturing and chlor-alkali process
1970	Fishing banned; Allied Chemical sued by US Attorney General to stop mercury dumping
1970s	Significant reduction in fish mercury concentrations
1979-81	METRO upgraded to include secondary and tertiary treatments
1989	Judgement on Consent filed in federal court against Onondaga County for allowing METRO to exceed permitted effluent limits; NY State sues Allied Chemical under CERCLA aka “superfund” legislation for polluting Onondaga Lake with organic chemicals/heavy metals
1992	Allied Chemical agrees to conduct a remediation feasibility investigation
1994	Onondaga Lake added to USEPA National Superfund Priorities List
1999	Honeywell buys Allied Chemical’s parent company, Allied Signal Inc.
1999	NYSDEC lifts fishing ban on Onondaga Lake
2000	First successful Zebra Mussel (*Dreissenia polymorpha*) colonization
2005	Metro upgrade for ammonia (NH_4) treatment
2010	Starry Stonewort (*Nitellopsis obsusa*) first observed
2011	Beginning of Round Goby (*Neogobius melanostomus*) invasion
2012	Honeywell begins dredging and capping operation
2016	Capping concludes & nearshore habitat enhancements begin

INTRODUCTION

Onondaga Lake, in central New York, has been degraded by the dumping of industrial and municipal wastewater for over a century, which led to alterations of its chemical properties and subsequently the lake ecology (Effler 1996). Onondaga Lake originally supported multiple economically-important cold water fisheries, but they were lost in the late 1800s (Ringler et al. 1996). Natural brine springs throughout the watershed brought salt production and other industries to the area, eventually leading to the establishment of Syracuse, New York in 1825 (Rowell 1996). The brine springs were later used for soda ash production that generated large quantities of ionic wastes; chemical manufacturing shifted to chlor-alkali production in the late 1940s (Effler 1987). This process used mercury as an electrode, but large quantities were dumped into Onondaga Lake within the waste products.

Anthropogenic impacts on waterways have resulted in the degradation of aquatic ecosystems worldwide, particularly from industrial and agricultural pollution. The need to protect United States waterways led to the establishment of the Federal Water Pollution Control Act of 1972, better known as the Clean Water Act, and the Comprehensive Environmental Response, Compensation and Liability Act of 1980 (CERCLA) which respectively provide water quality guidelines and hold negligent parties accountable.

Mercury pollution from manufacturing is a significant health hazard globally, including North America, and continues to be a significant hazard to human health (Trip and Allan 2000; Wang et al. 2012). Mercury, particularly in the methylmercury form, is neurotoxic. Exposure is strongly correlated to neurological diseases, including Amyotrophic Lateral Sclerosis (ALS), Alzheimer's disease, and Parkinson's disease, as well as birth defects and cardiovascular diseases, particularly heart attacks (Wang et al. 2004, 2012; Stern 2005). The first major case of mercury poisoning from environmental dumping occurred in Minamata, Japan, where more than 1000 people died and more than 2000 more were permanently paralyzed from the release of industrial effluent during in the 1950s and 1960s (Kudo and Miyahara 1991). The main pathway for human uptake of methylmercury is from consumption of contaminated foods, most commonly fish that have bioaccumulated methylmercury in aquatic systems with low water concentrations (Wang et al. 2004; Scheuhammer et al. 2007). Few fish species are significantly impaired by methylmercury, but long-lived piscivores such as Walleye (*Sander vitreus*) and Northern Pike (*Esox lucius*) can exhibit hormonal deficiencies and altered reproductive behavior after accumulating high tissue concentrations of methylmercury (Scheuhammer et al. 2007).

A major source of mercury in aquatic systems worldwide is from chlor-alkali manufacturing plants, which produce industrial organic chemicals from brine through the use of mercury cathodes (Effler 1987; Trip and Allan 2000; Arribére et al. 2003; Southworth et al. 2004; Ullrich et al. 2007; Benejam et al. 2010; Zheng et al. 2011). This process is extremely prone to mercury loss into the waste material, resulting in high accumulation among these ionic wastes. Under hypoxic conditions, mercury mainly enters the food web when it is methylated by sulfate-reducing bacteria (Benoit et al. 2003); these bacteria are subsequently consumed by aquatic organisms (Scheuhammer et al. 2007). Controlling inputs and reducing existing quantities of mercury in waterbodies is imperative for human health and safety worldwide, and the global reduction in aquatic methylmercury is also beneficial for the aquatic community.

Multiple methods are used to treat polluted aquatic systems, especially when the toxicants are restricted to lake sediments. However, the most common is a combination of contaminant immobilization and then removal via aquatic dredging (Wang et al. 2012). In many cases this involves the use of a chemical binder, such as activated carbon, to prevent the movement and dissemination of the contaminant during removal. The southern basin of Onondaga Lake contains the majority of mercury waste from chlor-alkali production and elevated nutrient levels from METRO effluent (Effler 1987; Matthews et al. 2015). As a result, this southern portion was most affected by anthropogenic pollution and is the focus of ongoing dredging and nearshore restoration efforts.

Sediment dredging can potentially provide another beneficial alteration to the aquatic community. Zhang et al. (2010) found that this practice ultimately reduced the trophic status of shallow eutrophic lakes by decreasing phosphorus, total suspended solids, organic matter, and chlorophyll-a. Fish are rarely impacted directly, but dredging does alter habitats (Fischer et al. 2012). Effects of sediment resuspension also vary widely by fish species and life stage; for example, eggs and larvae are more susceptible than adults (Wilber and Clarke 2001). However, riverine communities around dredging operations are usually comprised of more silt-tolerant species, such as centrarchids and Common Carp (Cross et al. 1982); suggesting that fish assemblages are altered not only by pollution, but also by the methods of remediation.

Freshwater fishing is a popular recreational activity worldwide; anglers were estimated to have spent $331 million USD annually in New York state in 2007, equating to more than $403 million in 2018 (Connelly and Brown 2009). Recreational fishing in the northeastern United States, including Onondaga Lake, primarily targets black bass, salmonids, Walleye, and Yellow

Perch (*Perca flavescens*). Many of these populations are managed via stocking (Zale et al. 2012), but understanding and managing sportfish populations requires a whole-community approach. Stocking is an effective measure to introduce and boost sportfish populations, but population maintenance is dependent on the health of the entire aquatic community (Brazner and Beals 1997; Cvetkovic et al. 2010; Bhagat and Ruetz 2011). Without adequate habitat or food sources, species of interest are susceptible to larval predation, and migratory species are more likely to leave the system, requiring annual stocking to maintain populations.

Abiotic and biotic factors, such as water chemistry and food sources, drive community composition in lakes (Tonn and Magnuson 1982). These factors are also interdependent; changes in one can have a cascading effect on the entire community. For example, warming of water bodies correlates to higher predation on zooplankton by fish, subsequent reductions in grazing on algae, and increased trophic state (Jeppesen et al. 2010). Fish and the environment are interconnected; fisheries management must account for effects that they have on each other.

Fish habitat includes spawning substrate and cover and are required for the establishment and replenishment of a viable population, while cover is also necessary to sustain the population to maturity and provide for predator avoidance. Fish habitat in lotic systems is predominantly large woody debris (Dustin and Vondracek 2017). Macrophyte cover is the other major adult fish habitat in lakes and ponds; this coverage increases seasonally in spring and then dies back in fall and winter. Macrophyte diversity plays a major role in species richness of a system (Tonn and Magnuson 1982). Aquatic plants usually occur nearshore where light penetration is high and the depth to which they grow is directly related to water clarity and subsequent light attenuation (Barko and Smart 1986).

As with fish spawning, substrate is also a key factor for macrophyte location and growth. Lake flushing dynamics and sedimentation drive the movement of substrate. Wind, and subsequently wave exposure, can also play significant roles in the distribution of macrophytes by directly uprooting seedlings and physically damaging mature plants (Keddy 1982). Instances of heightened wave exposure can further accelerate erosion of sediments and contribute to ice scour in winter. Because of its northwest to southeast orientation, westerly prevailing winds may have a significant impact on the nearshore communities of Onondaga Lake (Effler 1996). Macrophytes cannot establish without anchoring sediments, so areas less protected from wind have fewer aquatic plants and a diminished benefit to nearshore and juvenile fish.

Aquatic biodiversity is also reduced by anthropogenic habitat degradation (Dudgeon et al. 2006; Freedman et al. 2013). Lakeshore habitat modifications and development are directly correlated to decreases in habitat complexity, and subsequent losses in fish species richness (Dustin and Vondracek 2017). The response and recovery of the aquatic community to disturbances is ultimately dependent on the diversity of the ecosystem and habitat (Lake 2000). Shoreline development will alter the lake community, making it less resistant to disturbances, anthropogenic and otherwise. An improved understanding of the fish assemblage response to water quality changes and disturbances will contribute to better management of lake resources.

Most studies on fish response to habitat alterations have focused on riverine habitats. This study aims to evaluate the response of the fish assemblage to the remediation of Onondaga Lake to develop management expectations and better inform goals for the remediation of other aquatic systems. The questions addressed are: how the fish assemblage of Onondaga Lake has responded to remediation over time, between basins, and whether nearshore assemblage differs from offshore. Spatial and temporal analyses will allow for comparisons of fish assemblages between basins as well as before and after the major remediation efforts.

METHODS

System Description

Onondaga Lake has two basins, a total surface area of 11.6 km^2 , maximum depth of 18 m (Effler 1996; Effler and Hennigan 1996). Its watershed covers approximately of 642 km^2, which is comprised of 40% forest, 30% agriculture, 21% urban, and 9% residential land (Coon and Reddy 2008). Onondaga Lake currently receives most of the water from the Metropolitan Wastewater Treatment Plant (METRO); four natural tributaries have been directly polluted by municipal or industrial waste, including Onondaga Creek, Ninemile Creek, Ley Creek, Harbor Brook, and Bloody Brook (Effler and Hennigan 1996). The lake discharges to the Seneca River system, which joins the Oneida River to form the Oswego River, a major tributary to Lake Ontario (Figure 1).

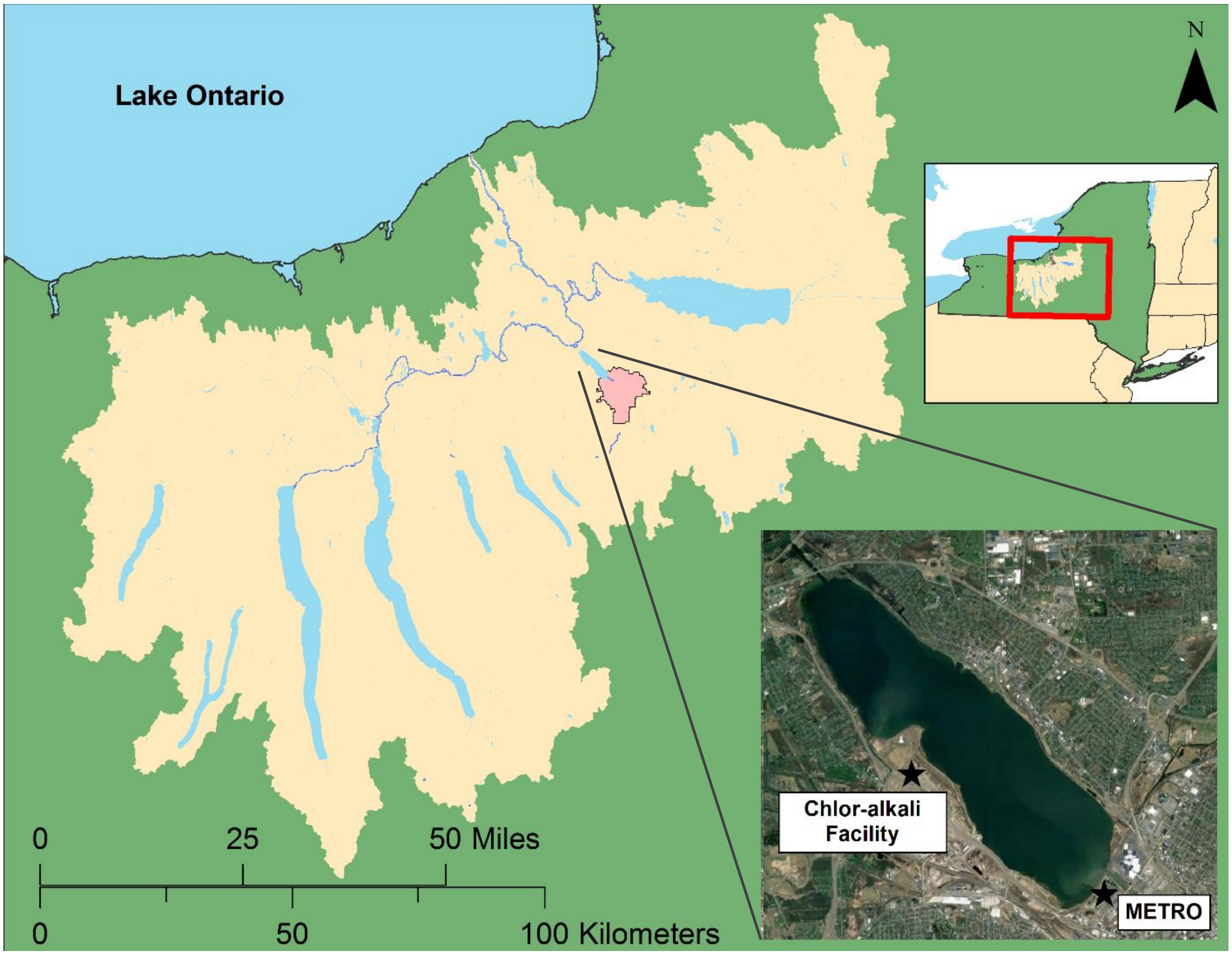

Figure 1. Location of Onondaga Lake and Syracuse (red) within the Oswego River system, a tributary to Lake Ontario. METRO and the Chlor-alkali facility are each marked on the insert map with a black star.

Site Selection

Trap and gillnet sampling was conducted at sites originally chosen by Gandino (1996) to evaluate the fish assemblage structure of Onondaga Lake. These locations were refined by Siniscal (2009) and incorporated into the Parsons/Honeywell Onondaga Lake Maintenance and Monitoring Plan (Parsons 2017) to consist of eleven trap net and twelve gillnet sites (Figure 2). Additional sites were added in 2008 to fulfill monitoring requirements in the southern basin for the Honeywell remediation of Onondaga Lake, specifically to increase sampling within each Sediment Management Unit of the capping area (Parsons 2017). Remediation efforts in the south basin obstructed certain sites, leading to inconsistent sampling locations in some years (Tables 2 & 3).

These sites are positioned throughout the lake but are not distributed equally between the north basin and the significantly larger south basin (Figure 2). There are four gillnet and trapnet sites each in the north basin, but eight gillnet and seven trapnet sites in the south basin.

Field Data Collection

The Onondaga Lake fish assemblage was sampled May through October every year from 2008 to 2018 to capture a wide range of seasonal patterns, including spawning and recruitment. Efforts beyond the summer months allowed for more comprehensive evaluation of the lake fish assemblage.

Table 2. Trapnet locations sampled per study year (2008-2018). Some sites were inaccessible during the dredging and capping operation. The Wastebeds site moved to 690 Point in 2017 due to the construction of a nearby public dock.

Year	690 Point	Harbor Brook	Iron Bridge	Ley Creek	Maple Bay	Marina	Metro	Ninemile	Parsons	PHM	Wastebeds	Willow Bay
2008		x	x	x	x	x	x	x	x	x	x	x
2009		x	x		x	x	x	x	x	x	x	x
2010		x	x		x	x	x	x	x	x	x	x
2011		x	x		x	x	x	x	x	x	x	x
2012		x	x		x	x	x	x	x	x	x	x
2013			x	x	x	x		x	x	x	x	x
2014			x	x	x	x		x	x	x	x	x
2015			x	x	x	x		x	x	x	x	x
2016			x	x	x	x		x	x	x	x	x
2017	x		x	x	x	x	x	x	x	x		x
2018	x	x	x	x	x	x	x	x	x	x		x

Table 3. Gillnet locations sampled per study year (2008-2018). Some sites were inaccessible during the dredging and capping operation

Year	690 Point	Harbor Brook	Hiawatha Point	Iron Bridge	Ley Creek	Marina	Metro	Ninemile	Outlet	Parsons	PHM	Wastebeds
2008	x	X	x	x	x	x		x	x	x	x	x
2009	x	X	x	x		x		x	x	x	x	x
2010	x	X	x	x		x	x	x	x	x	x	x
2011	x	X	x	x		x		x	x	x	x	x
2012	x	X	x	x		x		x	x	x	x	x
2013	x		x	x	x	x		x	x	x	x	x
2014	x	X	x	x	x	x		x	x	x	x	x
2015			x	x	x	x		x	x		x	x
2016	x	X	x	x	x	x		x	x		x	x
2017	x	X	x	x	x	x	x	x	x	x	x	x
2018	x	X	x	x	x	x	x	x	x	x	x	x

South Dakota-style trap nets were used to evaluate the shallow-water assemblage. These nets were constructed with 0.64cm (1/4 in) nylon mesh, which stretched over two 1.52m x 1.22m (5ft x 4ft) aluminum box frames and five 0.91m (3ft) hoops to end in a cinched "cod end". A 25.9m (85ft) main leader and two 10.66m (35ft) wings connect to the front box, each with float and weight lines. The main leader was attached to shore and the wings placed at 45-degree angles to the leader to create funnels for the fish on each side. The main box was intentionally not submerged to prevent fish from swimming over the net and inhibit suffocation of non-fish caught in the traps, such as turtles, muskrats, and northern water snakes. Wings were staked in place and the cod end was weighted with a concrete block to ensure the net would not move. These nets were deployed by boat once per month for 24 hours at each site. All fish captured were collected, identified to species, enumerated, and released.

Gillnets were used to evaluate larger pelagic fish in the lake. These monofilaments nets were constructed of 15.2cm (6in) stretch mesh that was 2.44m (8ft) tall and 38.4m (126ft) long with foam-core float line and lead-core sinking line at the top and bottom, respectively. Gillnets were set at each of the sites from a boat once per month, but exclusively after dark for 1-hour sets. Nets were placed perpendicular to shore starting at 3m depth but the deeper end of these nets varied significantly based on site bathymetry; the deeper ends were set at an average depth of 7 m, but this ranged from 3.5 to 11m. Timing of these net sets were staggered by 30 minutes to allow synchronous net sets, minimize time overlap when pulling nets, and minimize catch mortality.

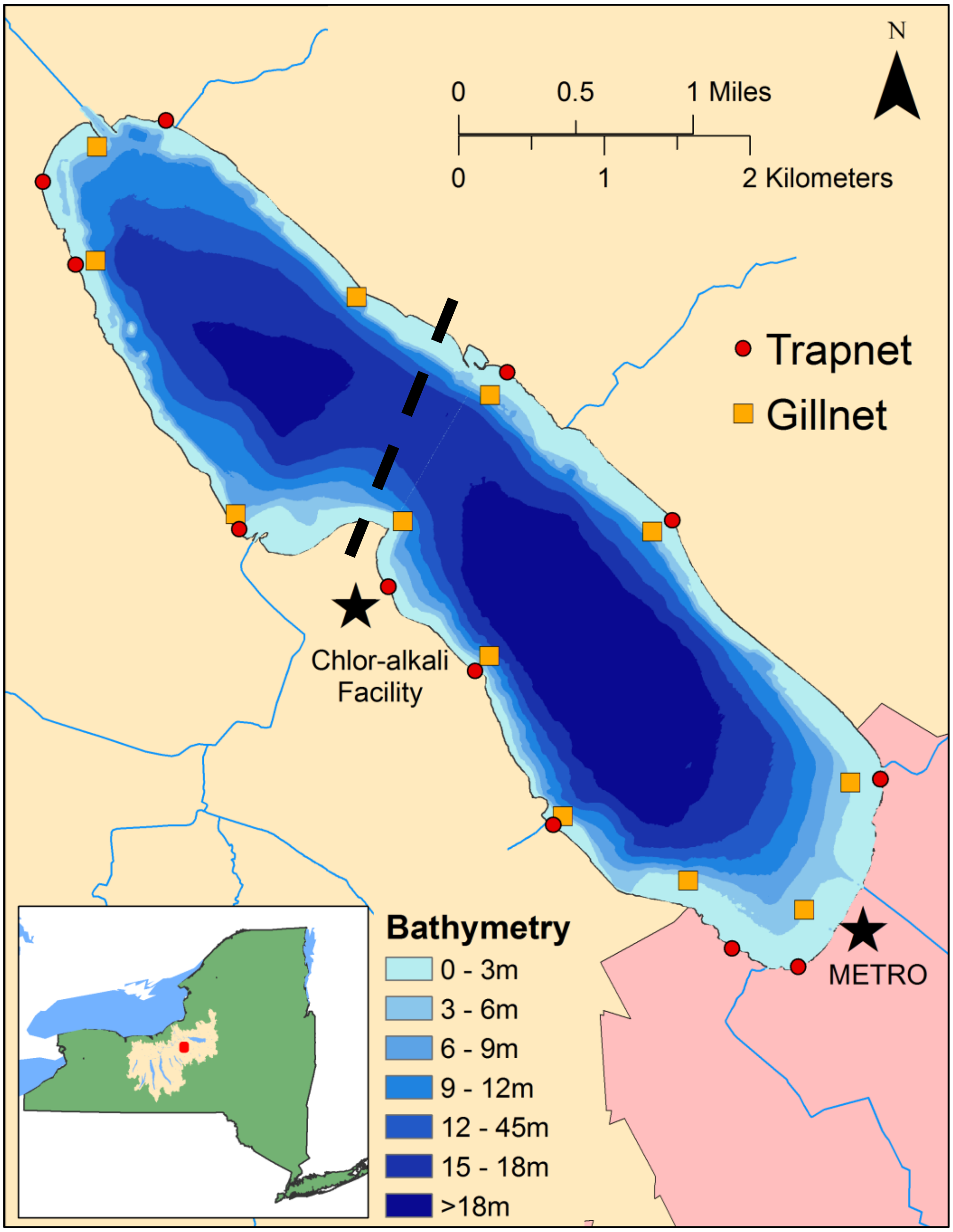

Figure 2 Trap net (red circle) and gillnet (yellow square) sampling locations on Onondaga Lake 2008-2018. The insert at bottom left shows the location of Onondaga Lake in the Oswego River drainage of central New York. METRO and the Chlor-alkali facility are each marked with a black star. The dashed line denotes the split between north and south basins.

Environmental Data Sources

Chemical and physical data for Onondaga Lake between May 1, 2008 and October 31, 2018 were obtained from the Upstate Freshwater Institute, US Geological Survey, and Onondaga County Department of Water Environmental Protection (Table 4). Two datasets were created to compare the environmental data with respect to the fish sampling methods; 1-2 m for trap nets and 3-10 m for gill nets. The physical and chemical water data were collected at relevant water depths per sampling method (Table 4); weather data and lake height were attributed to both datasets. Environmental data were averaged for each year.

Daily water conditions were obtained from the Onondaga Lake South Deep autonomous monitoring buoy operated by the Upstate Freshwater Institute (UFI). This monitoring platform samples temperature, specific conductivity, pH, dissolved oxygen, and turbidity at 1-meter intervals from 1 m to 18 m every 12 hours. For comparability, only the noon sampling data were used.

Daily lake height was recorded by USGS water stage recorder 04240495, located at the Onondaga Park Marina in Liverpool, New York (US Geological Survey 2018). Weather data were procured from the Onondaga County Department of Water Environment Protection (WEP) weather station at the Syracuse Metropolitan Wastewater Treatment Plant (METRO). This weather station reports rainfall, air temperature, and air pressure in 5-minute intervals.

Wind direction recordings were also taken from the METRO weather station and translated into one of 16 cardinal directions, each 22.5° wide, such as north and east-southeast. Monthly average wind speeds per cardinal direction were used and records of no wind (still air)

Table 4. Environmental data, units, and sources for multivariate analyses. Thirteen variables were selected and some were split by sampling depth to correspond to trapnet and gillnet net depths.

Variable	**Units**	**Data Summary**	**Source**	**Trapnet Depth (m)**	**Gillnet Depth (m)**	**Sampling Scale**
Lake Height	m	Average	USGS Lake Height Gage	NA	NA	Year
Precipitation	cm	Total	METRO Weather Station	NA	NA	Year
WaveIndex	Index ($P_iV_iM_i$)	Total	METRO (calculated)	NA	NA	Year & Basin
WaterTemp	C	Average	UFI	1-2	3-10	Year
Conductivity	us/cm	Average	UFI	1-2	3-10	Year
pH	pH	Average	UFI	1-2	3-10	Year
DO	mg/L	Average	UFI	1-2	3-10	Year
Turbidity	NTU	Average	UFI	1-2	3-10	Year
TP	mg/L	Average	OCDWP LIMS	0 – 1	3, 6, 9	Year & Basin
Hg-methyl	ng/L	Average	OCDWP LIMS	3	3	Year
NH3-N	mg/L	Average	OCDWP LIMS	3	3	Year
Fcoli	cfu/100ml	Average	OCDWP LIMS	0	0	Year & Basin

were removed. Fetch per cardinal direction for each sampling site was calculated as Euclidian distance in ArcGIS as the average distance from the sampling site to the edge of Onondaga Lake. The wind data were analyzed after Kirby (2009) by using a wind exposure index for each site (E) using the equation from Keddy (1982):

$$E = \sum_{i=1}^{16}(V_i * P_i * F_i)$$

Where i is one of the 16 cardinal directions, V_i is wind speed per cardinal wind direction, proportion of wind blowing in a given direction (P_i), and site fetch per wind direction (F_i).

Additional water quality metrics were collected by the Onondaga County Department of Water Environment Protection between 2008 and 2018. Of the nearly two dozen parameters, four were selected for their consistency over the sampling period: water concentrations of fecal coliform (CFU/100mL), total phosphorus (mg/L), methyl-mercury (µg/L), and ammoniacal nitrogen (mg/L) (Appendix 3). These conditions were analyzed at separate depths and frequencies throughout the May to October field season and were each averaged per year (Appendix 3).

Changes in Richness and Diversity Over Time

Linear regressions were conducted to compare species richness and diversity over time and between basins. However, effort and catch were not uniform over time and between the north and south basins. Catch data were aggregated by sampling year and site before standardization by rarefaction using the equation (Hurlbert 1971):

$$E(S_m) = \sum_i \left[1 - \frac{\binom{N - N_i}{m}}{\binom{N}{m}} \right]$$

Where $E(S_m)$ is the expected number of species in the collection, N and N_i are respectively the total number of individuals and individuals per species i, and m is the standardized number of individuals. This method scales down species richness at all sites relative to the smallest number of individuals captured and is especially useful when sampling effort is non-uniform or unknown (Gotelli and Colwell 2001; Brewer and Williamson 1994). Shannon diversity was also calculated for the samples aggregated by year and sampling site using the equation (Spellerberg and Fedor 2003):

$$H = -\sum_{i=1}^{n} (p_i \times \ln p_i)$$

Where *H* is the Shannon diversity, *n* is the number of species present in the sample, p_i is the proportion of individuals from species *i* within the sample. Whole-lake trends in both expected richness and diversity were analyzed year using linear regressions.

There have been numerous alterations to Onondaga Lake between 2008 to 2018 with significant impacts on water quality as part of the Honeywell remediation project. Piece-wise regressions were calculated for the assemblage indices to determine whether there were different response patterns over distinct time ranges (Ryan and Porth 2007). These regressions were computed in R using the ‘segmented’ package to determine breakpoints and then conduct a Davies Test to compare the piece-wise and original linear regressions (Davies 2002; Muggeo 2008). This test was used to detect non-constant responses of the regression over time.

North and South Basin Comparisons

Rates of change in expected richness and Shannon diversity were calculated for each site and then compared between basins. These metrics were first analyzed with one sample T-tests to determine if the sites were significantly similar within basin groupings and then with a Welch's Two-Sample T-test to evaluate between basins. Additionally, expected richness and Shannon diversity for all years were directly compared between basins using Welch's Two-Sample T-tests.

Multivariate Analysis of Factors Influencing Fish Assemblages

Fish samples were aggregated by year and basin and then species abundances were standardized by unit effort (CPUE) by net soak time: 24 hours for trapnet and 1 hour for gillnet. CPUE was then log-transformed [LN(x+1)] to reduce the influence of abnormally high catches (Boesch 1977). Non-metric multidimensional scaling (NMDS) ordinations using the Bray-Curtis dissimilarity index were employed to relate species compositions among basins and years, and to visualize compositions with respect to influential lake habitat variables (Galacatos et al. 2004; Clarke 1993). Significant relationships between environmental variables and the assemblages were assessed in R using the envfit function in the 'vegan' package (Oksanen et al. 2019). Differences in catch, in this case fish species CPUE, were statistically tested using a permutational multivariate analysis of variance (PERMANOVA), a non-parametric partitioning test for geometric distances of multiple response variables (Anderson 2017). As with the NMDS, CPUE were also log-transformed [LN(x+1)] to prevent issues with the Bray-Curtis dissimilarity calculation.

RESULTS

Changes in Richness and Diversity Over Time

Throughout the 2008 to 2018 period, Onondaga Lake sampling locations for both inshore trap nets and offshore gillnets varied, particularly during the capping and dredging period due to limited shoreline access. Nearshore catches ranged from 55 fish at the Willow Bay site in 2015 to 11636 at the Metro Site in 2018; offshore ranged from 10 individuals at the PHM sampling site in 2013 to 117 at the Iron Bridge Site in 2008. Species richness was rarefied to 55 and 10 individuals for near- and offshore assemblages, respectively.

Significant negative trends were observed for both nearshore expected richness ($p < 0.001$) and diversity ($p < 0.001$)(Figure 3). These negative trends for both regressions appeared to be nonlinear, which was supported by the use of Davies test to evaluate the piecewise regression of different responses over distinct time periods (Davies 2002). Expected richness and diversity had statistically significant breakpoints between the 2013 and 2014 field seasons (2013.6; Figure 3) ($p < 0.001$ and $p = 0.004$, respectively).

The five most abundant species captured nearshore between 2008 and 2018 were Banded Killifish (*Fundulus diaphanus*), Largemouth Bass, Bluegill (*Lepomis macrochirus*), Gizzard Shad (*Dorosoma cepedianum*), and Alewife (*Alosa pseudoharengus*). These five species comprised approximately 20% of the assemblage in 2008

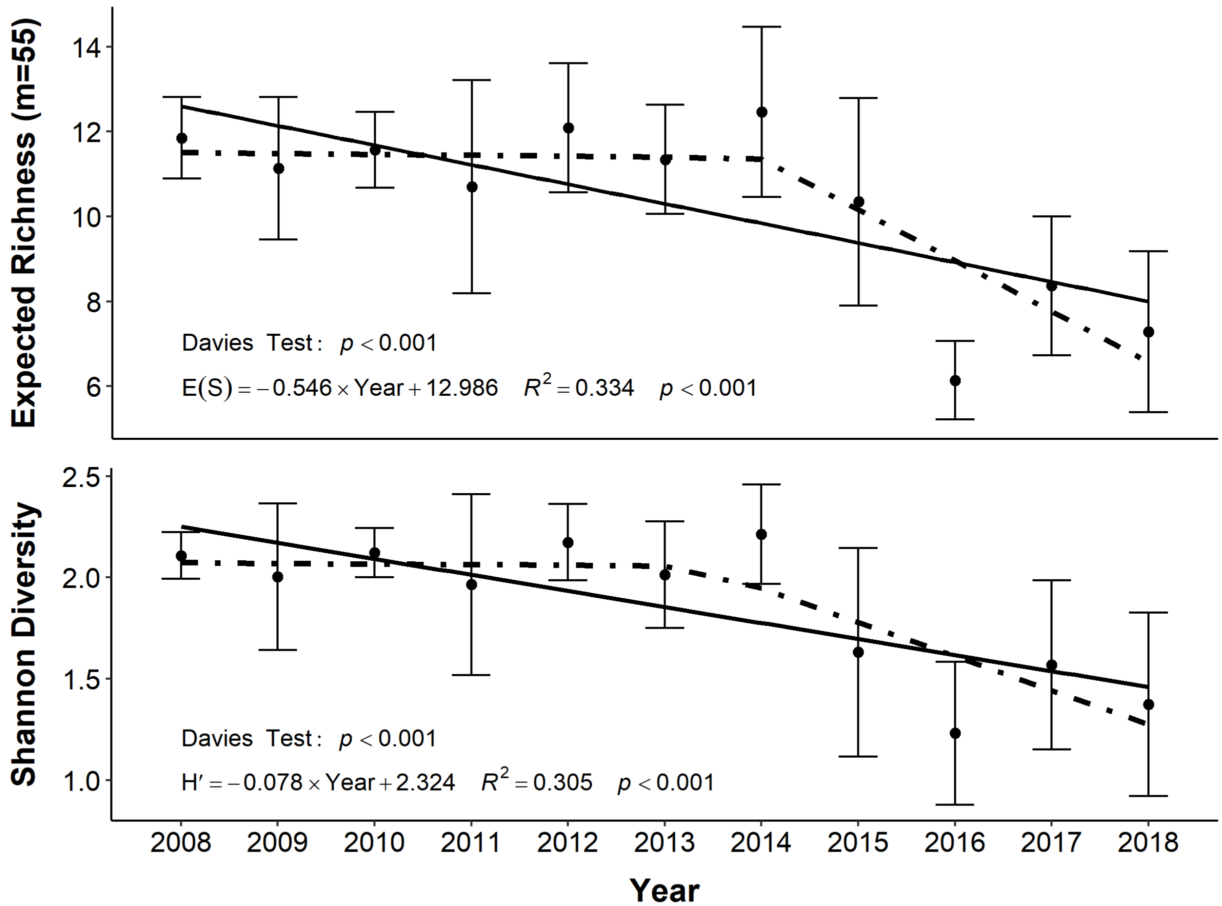

Figure 3. Linear regressions and 95% error bar intervals of expected richness (top; m=55) and Shannon diversity (bottom) vs. year for nearshore fish surveys on Onondaga Lake for 2008 to 2018. Dashed lines show the fitted piecewise regressions.

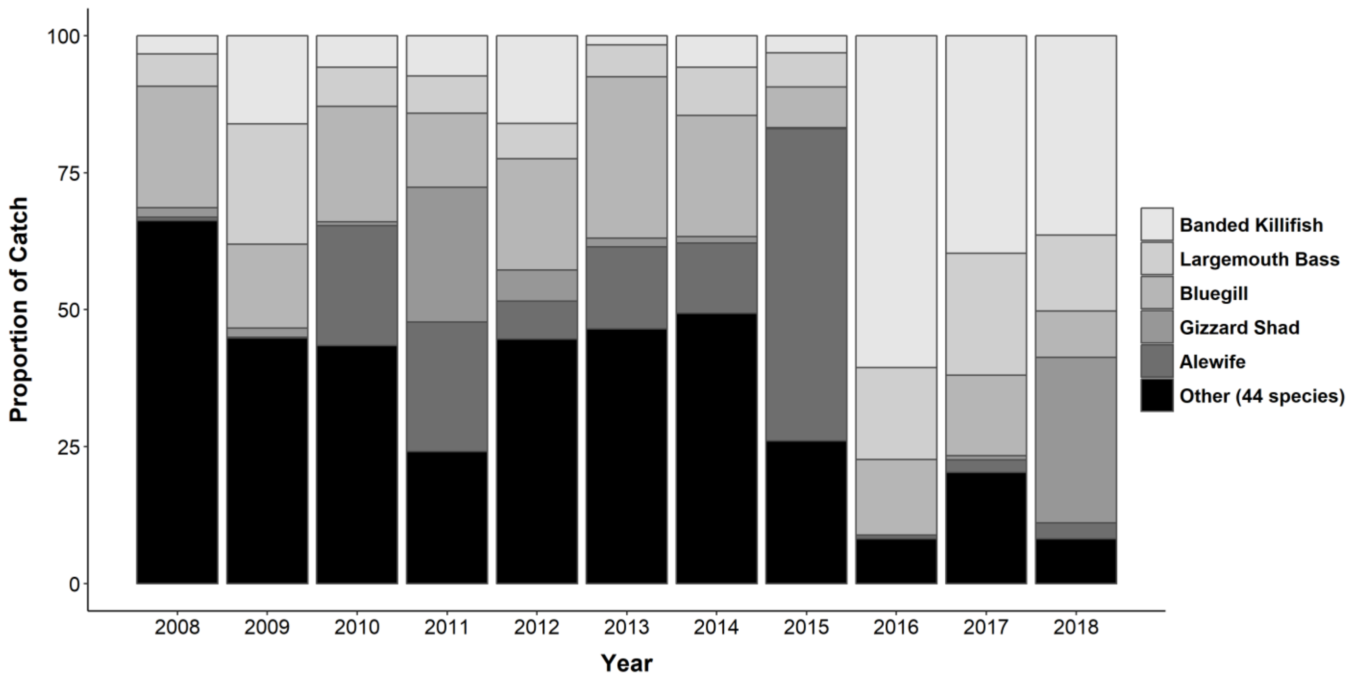

Figure 4. Stacked bar chart for the Onondaga Lake nearshore assemblage composition from 2008 to 2018. Forty-nine species were encountered in the nearshore surveys; the five most abundant were Banded Killifish, Largemouth Bass, Bluegill, Gizzard Shad, and Alewife.

but accounted for nearly 90% in 2016 and 2018 (Figure 4). The nearshore assemblage prior to 2016 was mostly composed of Bluegill and Alewife but was dominated by Banded Killifish in 2017 and 2018.

Offshore, the expected richness significantly increased over this period ($p = 0.038$), while there was no significant trend in Shannon diversity ($p = 0.112$)(Figure 5). Additionally, neither richness nor diversity had significant piecewise regression breakpoints; $p = 0.953$ and $p = 0.832$, respectively, suggesting a linear relationship if any.

The composition of the larger, offshore fishes was uniform between 2008 and 2018 (Figure 6). Walleye were consistently the dominant catch in gillnets; however, the proportions of Common Carp (*Cyprinius carpio*), Channel Catfish (*Ictalurus punctatus*), Gizzard Shad, and Freshwater Drum (*Aplodinotus grunniens*) varied. These top five species accounted for at least 80% of the annual catch each sampling year. The proportion of Channel Catfish gradually decreased over this time, while the proportion of Freshwater Drum increased. There also appears to be some periodicity to the relative abundance of Gizzard Shad, which were most common in 2011 and 2015.

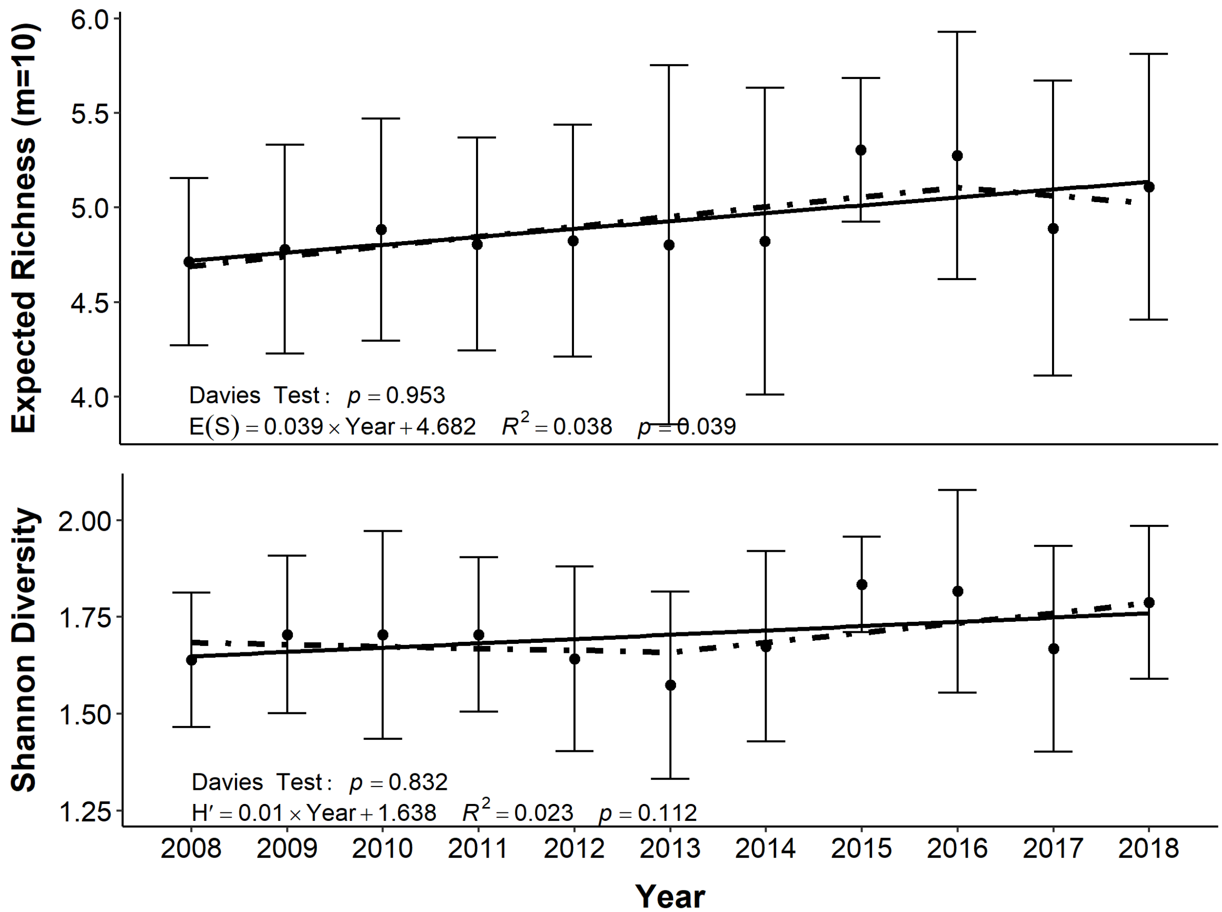

Figure 5. Linear regressions and 95% error bar intervals of expected richness (top; m=10) and Shannon diversity (bottom) vs. year for offshore fish surveys on Onondaga Lake for 2008 to 2018. Dashed lines show the fitted piecewise regressions.

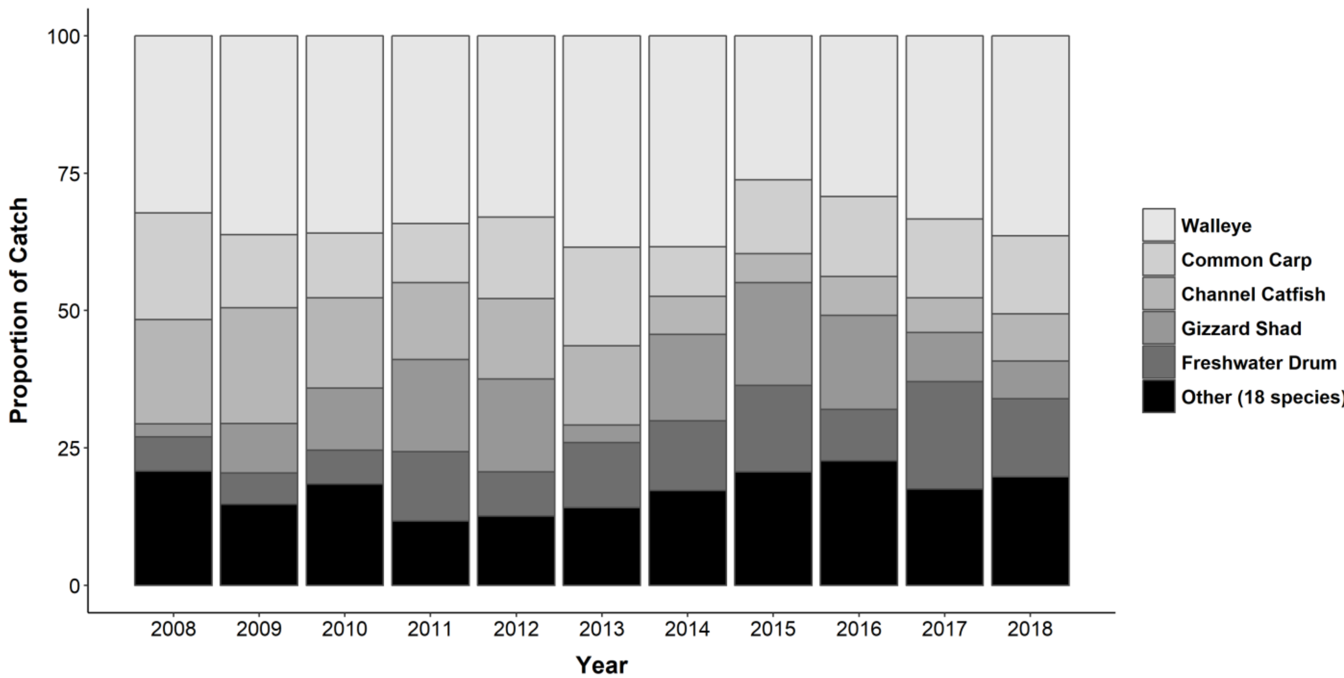

Figure 6. Stacked bar chart for the Onondaga Lake offshore assemblage composition from 2008 to 2018. Twenty-three species encountered in the offshore surveys; the five most common were Walleye, Common Carp, Channel Catfish, Gizzard Shad, and Freshwater Drum.

North and South Basin Comparisons

The rates of change in nearshore expected richness and diversity for each basin were all significant (North richness $p = 0.0051$ and diversity $p = 0.0076$; South richness and diversity $p < 0.001$); however, the basins were not significantly different in richness or diversity ($p = 0.5118$ and $p = 0.201$, respectively; Figure 7). Trends in the offshore indices differed from trends determined for nearshore habitats. The slopes observed for offshore expected richness were not significantly different within basin groups (North $p = 0.123$; South $p = 0.282$), nor between basins ($p = 0.708$). The same trend was observed for Shannon diversity, which had no significant similarities within (North $p = 0.098$; South $p = 0.406$) or between basins ($p = 0.728$; Figure 8).

The average values for richness and diversity were not significantly different for nearshore or offshore assemblages; however, the proportion of catches per species in each basin appeared different (Figure 7). Nearshore assemblages in the north basin were proportionally dominated by Banded Killifish, which comprised nearly 50% of all catches. However, the south basin nearshore was comprised of approximately 20% Gizzard Shad, which were rarely observed in the north basin sites. The distribution of the five most common fish species in offshores appears nearly identical between basins (Figure 8).

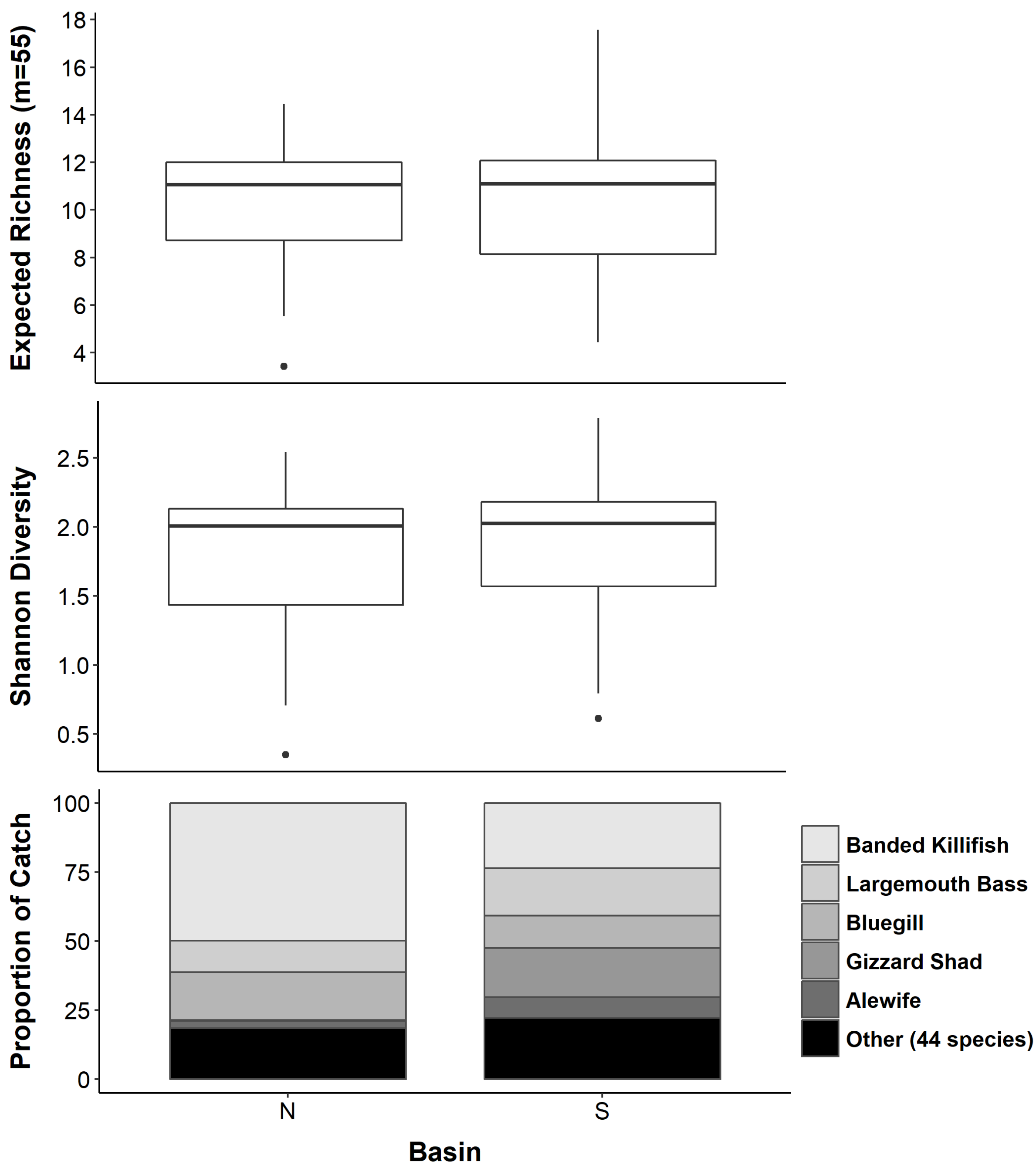

Figure 7. Nearshore expected richness (m=55; top) and Shannon diversity (middle) for north and south basins of Onondaga Lake for the entire sampling period 2008-2018. Stacked bar chart (bottom) represents the proportion of nearshore catches between north and south basins of Onondaga Lake. Forty-nine species encountered in the nearshore surveys; the five most common were Banded Killifish, Largemouth Bass, Bluegill, Gizzard Shad, and Alewife.

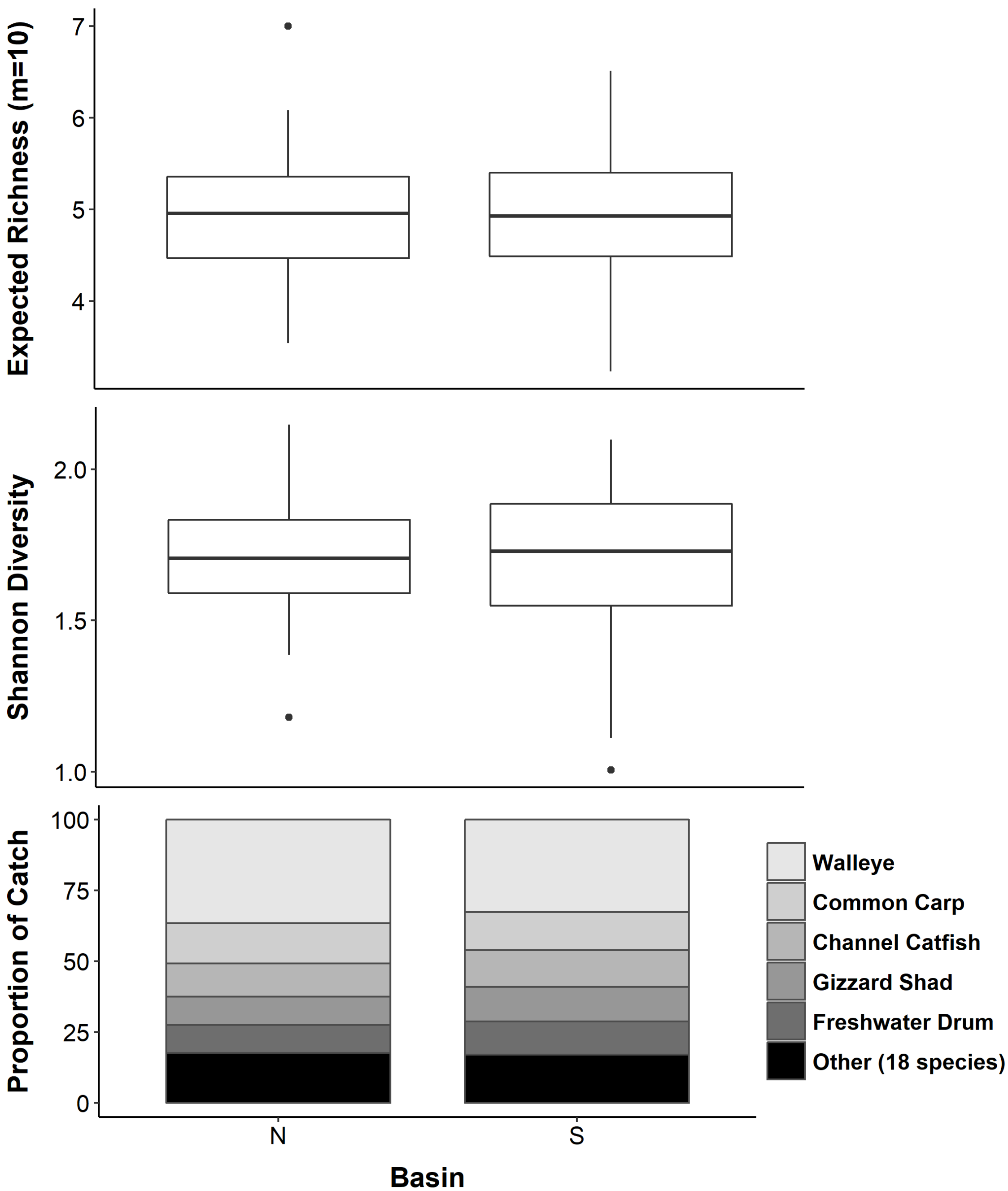

Figure 8. Gillnet expected richness (m=10; top) and Shannon diversity (middle) for north and south basins of Onondaga Lake for the entire sampling period 2008-2018. Stacked bar chart (bottom) representing the proportion of offshore catches between north and south basins of Onondaga Lake. Twenty-three species encountered in the offshore surveys; the five most common were Walleye, Common Carp, Channel Catfish, Gizzard Shad, and Freshwater Drum.

Multivariate Analysis of Factors Influencing Fish Assemblages

The NMDS ordination for trap nets produced a stress value of 0.067 for the first two dimensions, which suggests a good fit (Figure 9). Samples generally ordinated together by year, except for the 2009 north basin catches, which were very dissimilar from all other catches. Two of the selected thirteen habitat variables were significantly fitted to the ordination of trap net catch data: methylmercury (Hg.methyl) and water temperature (Table 5). The methylmercury variable ordinated inversely to the progression of time; mercury concentrations in the water were highest in the first few sampling years and then dropped precipitously during remediation. Water temperature plotted closely to methylmercury; also indicating a relationship to time. The PERMANOVA found significant differences between years ($p = 0.001$), but not between basins ($p = 0.705$; Table 6).

The offshore assemblage appeared homogenous over time and between basins (Figures 6 and 8), which was reflected in the NMDS ordination (Figure 10). The ordination was calculated with a stress value of 0.171, which indicates a usable, albeit weak ordination that may be misleading (Clarke 1993). Unlike with the trap net NMDS plot, the two points per year were not as tightly ordinated together. Five of the thirteen habitat variables were significantly associated with the trends in the offshore species compositions: conductivity, lake height, methylmercury, pH, and turbidity (Table 5). Similarly to nearshore trends, higher methylmercury may be indicative of pre-remediation conditions and higher turbidity is associated with the capping and dredging years: 2012 to 2016 (Parsons 2017). Most of the dredge material was composed of Solvay waste, rich with calcium carbonate; it acted as an alkaline buffer and removal coincides with reductions in pH. Turbidity and conductivity appear to be inversely correlated, as are lake

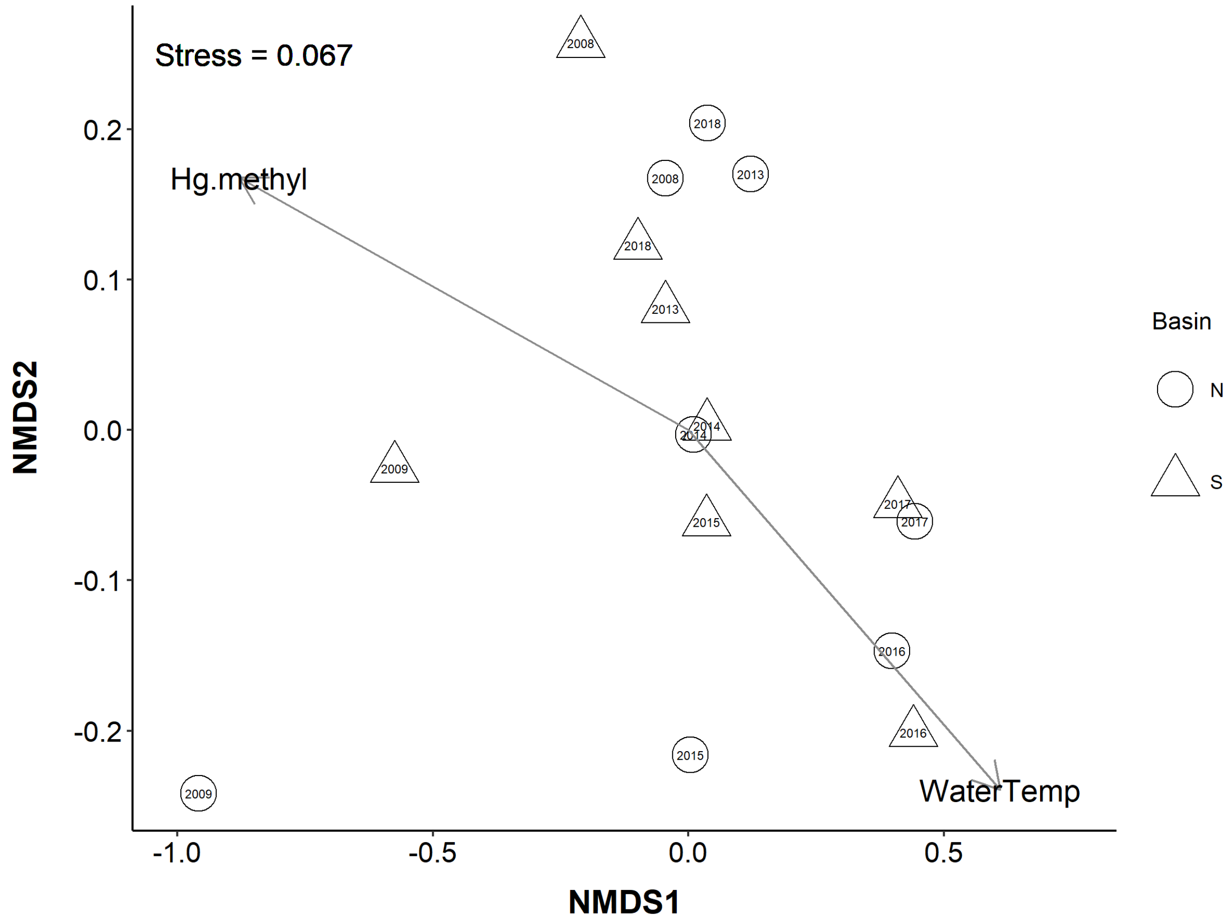

Figure 9. Non-metric multidimensional scaling ordination of trap net samples aggregated by year and basin. Circular and triangular points represent north and south basins, respectively. Year is shown inset within each shape.

Table 5. Fit of environmental variables to the nonmetric multidimensional scaling (NMDS) analysis of fish assemblage data. Only variables significantly related to the ordination were included in the ordination; these are denoted with an asterisk (*).

Variable	**Nearshore *P***	**Offshore *P***
Lake Height	0.165	0.020 *
Precipitation	0.080	0.120
Wave Index	0.958	0.204
Water Temperature	0.028*	0.063
Conductivity	0.112	0.038 *
pH	0.513	0.025 *
Disssolved Oxygen (DO)	0.730	0.123
Turbidity	0.189	0.049 *
Total Phosphorus (TP)	0.382	0.120
Methylmercury (Hg.methyl)	0.001*	0.001 *
Ammoniac Nitrogen (NH3-N)	0.079	0.144
Fecal Coliform (Fcoli)	0.524	0.297

Table 6. Permutational analyses of variance (PERMANOVAs) analyzing the effect of factors Year and Basin for the nearshore and offshore assemblages based on Bray-Curtis dissimilarties of catch-per-unit-effort (CPUE). An asterisk (*) denotes significant differences.

Nearshore Source of Variation	*df*	*SS*	*F*	*P*
Year	7	3.4253	8.3457	0.001*
Basin	1	0.0397	0.6776	0.705
Residuals	59	3.4592		
Total	67	6.9242		

Offshore Source of Variation	*df*	*SS*	*F*	*P*
Year	10	2.7605	3.0698	0.001 *
Basin	1	0.1991	2.2139	0.027 *
Residuals	103	9.2624		
Total	114	12.2220		

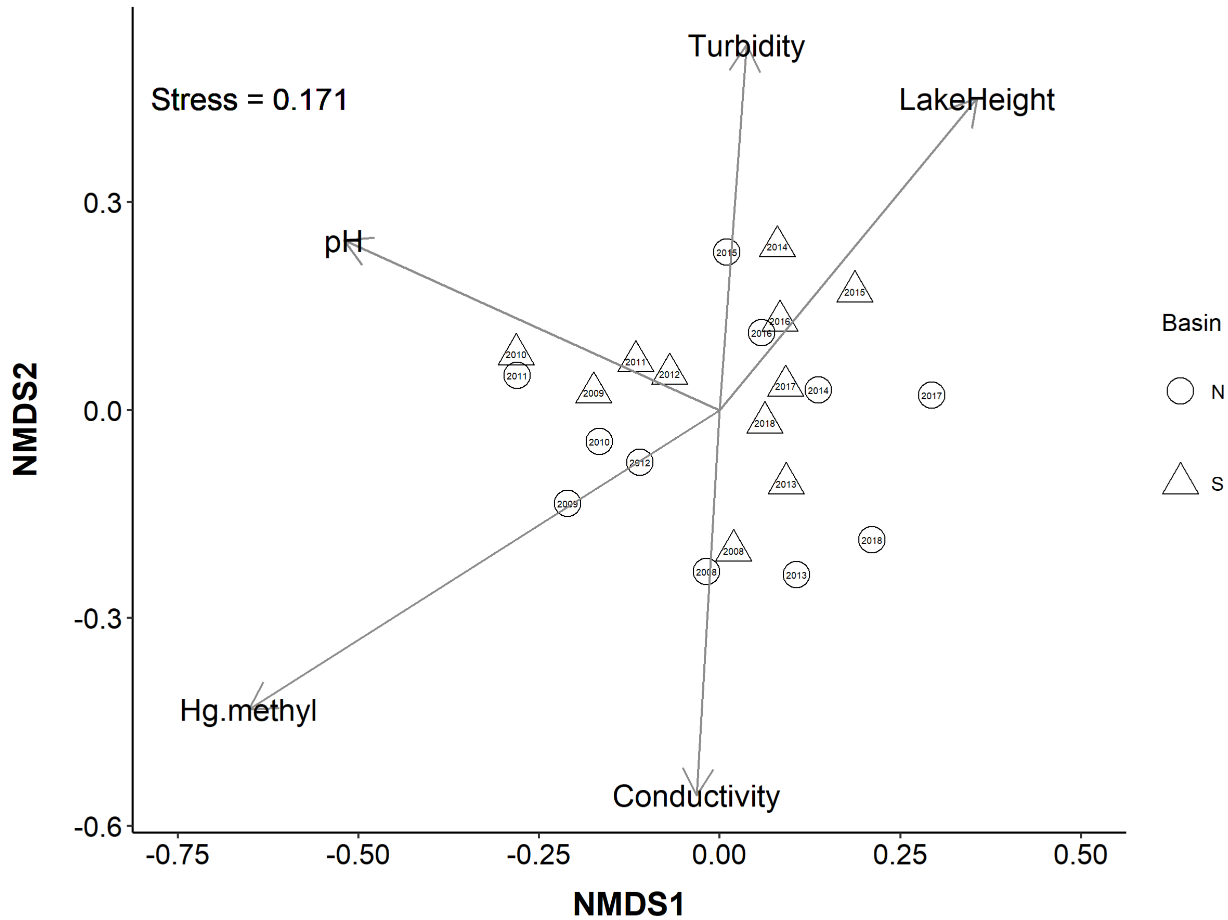

Figure 10. Non-metric multidimensional scaling ordination of offshore samples aggregated by year and basin. Circular and triangular points represent north and south basins, respectively. Year is shown inset within each shape.

height and methylmercury. Additionally, a PERMANOVA found significant differences in offshore assemblages between years ($p = 0.001$) and basins ($p = 0.027$)(Table 6).

DISCUSSION

Overarching Trends in the Onondaga Lake Fish Assemblage

The nearshore and offshore assemblages behaved differently during the timeframe of this study (2008 to 2018), which may be attributed to differences in response to remediation efforts. The species composition of the offshore assemblage changed over time and between basins, but this effect was not evident when using common metrics of species richness and diversity. The nearshore assemblage was significantly different over time when analyzed using both univariate and multivariate assessments. Species composition was not shown to be different between basins using either method.

Murphy et al. (2015) and Tango and Ringler (1996) found that both near- and offshore lake fish assemblages had been increasing in richness and diversity since 1946, which was most likely due to lake water quality improvements and enhancements to the METRO wastewater treatment plant. These studies provide baseline trajectories of the assemblages prior to dredging and capping. Gradual increases in species richness of the offshore assemblage in this study were consistent with these prior studies, but declines in nearshore richness and diversity in this study differ from these long-term trends.

These results suggest that the offshore assemblage has been minimally affected by the remediation and dredging of Onondaga Lake. Most of the fish species caught in the gillnets are highly mobile. Kirby et al. (2017) found that Walleye, the most abundant offshore assemblage constituent, often move nearly the entire length of Onondaga Lake in a 24-hour period. These fish are tolerant of changes in specific parts of the lake and have been shown to escape to external refugia, such as the Seneca River and Onondaga Lake tributaries, when water quality declines (Tango and Ringler 1996). Diversity of the offshore assemblage did not change over time or between basins, but significant differences in composition suggest that whole-lake species succession is gradually occurring.

The whole-lake nearshore assemblage was negatively impacted during the capping and dredging period; expected richness and diversity sharply declined in both basins of Onondaga Lake beginning in the physical remediation period 2012-2016. However, dredging and capping was exclusive to the south basin of Onondaga Lake (Parsons 2017), and the structure of the nearshore assemblage did not significantly differ between the north and south basins. This departure of the nearshore assemblage from its long-term trend may not be directly due to dredging, but rather a function of whole-lake water quality, ecological disturbance, and the introduction of invasive species.

Mechanisms of Ecological Change

Fish assemblage diversity and productivity are strongly correlated to habitat quality and complexity, particularly macrophyte diversity (Randall et al. 1996; Benson and Magnuson 1992; Tonn and Magnuson 1982). Kirby and Ringler (2015) found a significantly higher macroinvertebrate abundance in Onondaga Lake in diverse macrophyte beds than in monocultures of macroalgae. Macroinvertebrates, particularly chironomids and amphipods, are common prey for many of the littoral fish species found in Onondaga Lake (Kirby and Ringler 2015; Werner 2004). Increased littoral vegetation coverage also provides necessary nursery and foraging habitat for many young-of-year fish species, including Largemouth Bass and Pumpkinseed (Hinch and Collins 1993; Valley and Bremigan 2002).

Cvetkovic et al. (2010) determined that macrophyte assemblage composition is a consistently better predictor of fish community than abiotic water quality metrics. However, macrophyte coverage and composition are still tied to water quality. Murphy et al. (2015) concluded that increases in fish species richness in Onondaga Lake corresponded to decreases in ammoniac nitrogen and subsequent increases in macrophyte diversity and coverage. Reductions in ammoniac nitrogen also corresponded to the invasion of Zebra Mussels (*Dreissena polymorpha)*, a Eurasian bivalve shown to increase water clarity and facilitate macrophyte expansion (Spada et al. 2002; Skubinna et al. 1995; Effler and Siegfried 1998). Increasing macrophyte coverage stabilizes lake sediments, which subsequently decreases sedimentation rates, further lowers turbidity, and allows for greater light penetration for macrophyte growth at greater depths (Chambers and Kaiff 1985).

However, macrophyte diversity in lakes can be reduced by disturbances, such as fluctuations in lake height, which inhibit the establishment of rooted macrophytes in favor of mat-forming species (Wilcox and Meeker 1991). Onondaga Lake underwent numerous water height fluctuations during the study period, ranging from 0.02m to 0.16m, with the lowest lake height observed in 2012. Many of the submergent macrophyte taxa commonly found in Onondaga Lake, such as *Myriophyllum* and *Elodea*, are intolerant of drawdowns, while macroalgae are more moderately tolerant (Siver et al. 1986; Mjelde et al. 2013; Upstate Freshwater Institute et al. 2017). Macroalgae *and Potamogeton* are quick to colonize recently-excavated lake substrate (Geest et al. 2005), which suggests that annual fluctuations in water height and the depauperate substrate from capping of Onondaga Lake may favor these quick-growing macrophytes.

These lake-wide disturbances coincide with the arrival and proliferation of two aquatic invaders: Starry Stonewort (*Nitellopsis obtusa*) and Round Goby (*Neogobius melanostomus*). *Nitellopsis* is an aggressively-invasive macroalgae that was first discovered in Onondaga Lake in 2010 and quickly became a major constituent of the nearshore macrophyte assemblage (EcoLogic, LLC 2011; Kirby and Ringler 2015; Personal Observation 2018). *Nitellopsis* was first found in the St. Lawrence River in 1978 (Geis et al. 1981) and has since spread throughout the Great Lakes watershed via recreational boaters (Sleith et al. 2015; Midwood et al. 2016). It has been an effective invader due to its tolerance to high conductivity, production of inhibitory allelopathic chemicals, and late seasonal proliferation (Hilt and Gross 2008; Kovalenko et al. 2010; Midwood et al. 2016). In many North American lakes *Nitellopsis* has been able to outcompete other invasive macrophytes, including Eurasion Water Milfoil (*Myriophyllum spicata*) and Curly Leaf Pondweed (*Potamogeton crispus*), and reduce fish habitat diversity (Pullman and Crawford 2010; Brainard and Schulz 2016).

Nugent (2018) suggested that the expansion of *Nitellopsis* may have been a significant factor in the succession of the Onondaga Lake nearshore assemblage from Pumpkinseed and Bluegill in 2008 to juvenile Largemouth Bass and Banded Killifish after 2016. Between July and Sepember of 2017 and 2018, most of the north basin littoral zone was heavily inundated with

Nitellopsis, as were large portions of the southeastern shoreline (Personal Observation 2018). Dense macroalgae beds inhibit nest building by centrarchid gamefish species, such as Pumpkinseed, Bluegill, and Smallmouth Bass (*Micropterus dolomieu*) (Pullman and Crawford 2010). The nearshore inundation of *Nitellopsis* throughout Onondaga Lake constitutes a loss in habitat diversity and favors small-bodied preyfish species that can hide among the dense mats, such as Banded Killifish (Kapuscinski and Farrell 2014).

Round Goby (*Neogobius melanostomus*) also benefit from dense macrophyte beds and drive aquatic community composition (Cooper et al. 2007; Kipp and Ricciardi 2012). The Round Goby is an egg predator native to the Baltic Sea that arrived in the Great Lakes in 1990 via shipping ballast water and has since become a nuisance species throughout (Kornis et al. 2012). High densities of Round Goby have been shown to cause trophic cascades by lowering macroinvertebrate diversity and biomass via predation, the subsequent reduction of grazing allows for the proliferation of algal biomass (Pagnucco and Ricciardi 2015).

Round Goby were first observed in Onondaga Lake in 2010 and catches have since increased exponentially. Dense *Nitellopsis* mats in Onondaga Lake provided good habitat, but the lake became even more favorable when the polluted substrates in the south basin were replaced with larger sediments and substrate (Parsons 2018), which were effectively enhanced spawning habitat for Round Goby (Kornis et al. 2012). Round Goby are also a notable predator of the Dreissenid mussels, which arrived in Onondaga Lake in 2000, as well as fish eggs including Smallmouth Bass (Steinhart et al. 2004). However, this predation is reciprocal; Round Goby can be a major component of Smallmouth Bass diet and constitute a major link in the toxicant bioaccumulation pathway of many fishes of Onondaga Lake (Johnson et al. 2005; Henry and Driscoll 2017).

CONCLUSION

The Future of Onondaga Lake

The future management of the Onondaga Lake fishery is dependent on water and habitat quality. Dredging, capping, and active use of nitrate additives have been imperative to minimize methylmercury accumulation in Onondaga Lake fishes (Matthews et al. 2013; Murphy et al. 2015). These practices are primarily to the benefit of human health by obstructing mercury biomagnification pathways. Improvements to the fish assemblage to date have mostly been due to enhancements to whole-lake water quality from wastewater nutrient reductions and subsequent habitat diversification. During the height of the pollution, only 10 species were found in Onondaga Lake (Greeley 1928); however, since then more than 66 fish species have been observed (Murphy et al. 2015).

Nearshore remediation of Onondaga Lake has likely been beneficial to the lake fish assemblage; however, many of the positive effects are likely confounded by negative influences from the invasion of Round Goby and *Nitellopsis*. These introductions were likely significant factors in the succession of the lake fish assemblage during remediation; the structure of the Onondaga Lake fish assemblage will continue to be altered by future invasions. Pagnucco et al. (2015) expect the near-future of the Great Lakes basin will be marred by range expansions of many species due to the breakdown of thermal barriers from anthropogenic global warming and ultimately the continued introduction of destructive aquatic invasive species.

Lake Management Implications

Remediation of polluted aquatic systems is critical for preserving ecosystem and human health. However, effective management of these system hinges on adequate evaluation methodology. This study showed that different aspects of lake fish assemblages can differ in response to remediation and water chemistry changes. Many fisheries studies rely on the use of a single gear type or a few target species which may not sufficiently sample the breadth of fish assemblage responses in a lacustrine system. However, the use of a wide variety of sampling methodologies can ensure that temporal effects are more properly evaluated.

In addition to sampling methodology, choice of analyses is paramount. Shannon diversity and species richness are simple and ubiquitous metrics for evaluating the state of a community, but may not detect changes in composition, as seen with the Onondaga Lake offshore assemblage. Multivariate analyses are more difficult to interpret than indices but are useful exploratory tools to detect specific community interactions. These initial analyses are useful for lake and ecosystem managers to better evaluate and further develop rehabilitation goals.

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Appendix 1. List of fish species observed and abbreviation codes used in data analysis.

Species Code	Common Name	Scientific Name
ALE	Alewife	*Alosa pseudoharengus*
ATS	Atlantic Salmon	*Salmo salar*
BAK	Banded Killifish	*Fundulus diaphanus*
BLC	Black Crappie	*Pomoxis nigromaculatus*
BLG	Bluegill	*Lepomis macrochirus*
BLN	Bluntnose Minnow	*Pimephales notatus*
BLS	Blacknose/Blackchin Shiner	*Notropis heterolepis/heterodon*
BOW	Bowfin	*Amia calva*
BRB	Brown Bullhead	*Ameiurus nebulosusz*
BRS	Brook Stickleback	*Culaea inconstans*
BRT	Brown Trout	*Salmo trutta*
BSLV	Brook Silverside	*Labidesthes sicculus*
CARP	Common Carp	*Cyprinus carpio*
CHC	Channel Catfish	*Ictalurus punctatus*
CHP	Chain Pickerel	*Esox niger*
COM	Common Shiner	*Luxilus cornutus*
CRC	Creek Chub	*Semotilus atromaculatus*
EMS	Emerald Shiner	*Notropis atherinoides*
FHM	Fathead Minnow	*Pimephales promelas*
FWD	Freshwater Drum	*Aplodinotus grunniens*
GOH	Golden Redhorse	*Moxostoma erythrurum*
GOF	Goldfish	*Carassius auratus*
GOS	Golden Shiner	*Notemigonus crysoleucas*
GSF	Green Sunfish	*Lempomis cyanellus*
GZS	Gizzard Shad	*Dorosoma cepedianum*
LMB	Largemouth Bass	*Micropterus salmoides*
LNG	Longnose Gar	*Lepisosteus osseus*
LOG	Common Logperch	*Percina caprodes*
MGMT	Margined Madtom	*Noturus insignis*
NHS	Northern Hogsucker	*Hypentelium nigricans*
NOP	Northern Pike	*Esox lucius*
PUD	Pumpkinseed	*Lepomis gibbosus*
QLB	Quillback	*Carpiodes cyprinus*
RBT	Rainbow Trout	*Oncorhyncus mykiss*
ROB	Rock Bass	*Ambloplites rupestris*
ROG	Round Goby	*Neogobius melanostomus*

Appendix 1 Continued.

Species Code	Common Name	Scientific Name
RUD	Common Rudd	*Scardinius erythrophthalmus*
SFS	Spotfin Shiner	*Cyprinella spiloptera*
SHR	Shorthead Redhorse	*Moxostoma macrolepidotum*
SMB	Smallmouth Bass	*Micropterus dolomieu*
SVR	Silver Redhorse	*Moxostoma anisurum*
TED	Tesselated Darter	*Etheostoma olmstedi*
TGM	Tiger Muskellunge	*Esox masquinongy x Esox lucius*
TPM	Tadpole Madtom	*Noturus gyrinus*
WAE	Walleye	*Sander vitreus*
WHP	White Perch	*Morone americana*
WHS	White Sucker	*Catostomus commersoni*
YEB	Yellow Bullhead	*Ameiurus natalis*
YEP	Yellow Perch	*Perca flavescens*

Appendix 2. Environmental data fitted to Non-Metric Multidimensional Scaling ordination. Sources and associated sample depths are listed in Table 4.

Year	DataSource	Basin	Lake Height	Precipitation	Wave Index	Water Temp	Conductivity	pH	DO	Turbidity	TP	Hg-methyl	NH3-N	Fcoli
2008	Trapnet	N	110.69	45.84	3.46	17.17	2182.47	7.99	9.74	1.78	0.01	0.13	0.11	8.89
2009	Trapnet	N	110.71	55.69	4.02	17.38	1871.68	8.11	9.94	2.7	0.02	0.21	0.09	56.47
2010	Trapnet	N	110.74	69.06	3.5	20.2	1813.95	8.14	9.72	3.18	0.02	0.17	0.11	12.77
2011	Trapnet	N	110.84	64.79	3.85	20.42	1692.08	8.13	9.81	5.47	0.02	0.07	0.07	7.71
2012	Trapnet	N	110.68	42.55	3.79	20.91	2182.72	8.15	9.63	3.82	0.02	0.08	0.09	15.61
2013	Trapnet	N	110.78	64.06	3.67	20.2	1982.69	8.16	9.74	4.32	0.02	0.1	0.09	22.42
2014	Trapnet	N	110.81	58.34	3.9	19.64	1780.53	8.1	9.71	4.09	0.02	0.05	0.09	18.82
2015	Trapnet	N	110.85	72.67	3.86	20.41	1744.65	8.13	9.8	4.96	0.02	0.06	0.07	30.82
2016	Trapnet	N	110.72	58.82	3.94	20.52	2009.8	8.03	9.98	3.45	0.02	0.05	0.08	30.67
2017	Trapnet	N	110.86	65.48	3.86	19.98	1615.05	8.1	10.17	3.6	0.02	0.03	0.07	42.23
2018	Trapnet	N	110.78	58.83	4.01	21.17	1936.85	8.03	10.26	3.44	0.02	0.06	0.22	17.54
2008	Trapnet	S	110.69	45.84	6.79	17.17	2182.47	7.99	9.74	1.78	0.01	0.13	0.11	19.43
2009	Trapnet	S	110.71	55.69	6.51	17.38	1871.68	8.11	9.94	2.7	0.02	0.21	0.09	33.37
2010	Trapnet	S	110.74	69.06	7.76	20.2	1813.95	8.14	9.72	3.18	0.02	0.17	0.11	224.19
2011	Trapnet	S	110.84	64.79	7.17	20.42	1692.08	8.13	9.81	5.47	0.02	0.07	0.07	23.9

Appendix 2 Continued.

Year	DataSource	Basin	Lake Height	Precipitation	Wave Index	Water Temp	Conductivity	pH	DO	Turbidity	TP	Hg-methyl	NH3-N	Fcoli
2012	Trapnet	S	110.68	42.55	7.71	20.91	2182.72	8.15	9.63	3.82	0.02	0.08	0.09	25.45
2013	Trapnet	S	110.78	64.06	7.42	20.2	1982.69	8.16	9.74	4.32	0.02	0.1	0.09	23.26
2014	Trapnet	S	110.81	58.34	7.88	19.64	1780.53	8.1	9.71	4.09	0.02	0.05	0.09	38.24
2015	Trapnet	S	110.85	72.67	7.66	20.41	1744.65	8.13	9.8	4.96	0.02	0.06	0.07	179.09
2016	Trapnet	S	110.72	58.82	7.95	20.52	2009.8	8.03	9.98	3.45	0.02	0.05	0.08	32
2017	Trapnet	S	110.86	65.48	7.55	19.98	1615.05	8.1	10.17	3.6	0.02	0.03	0.07	44.21
2018	Trapnet	S	110.78	58.83	7.39	21.17	1936.85	8.03	10.26	3.44	0.02	0.06	0.22	27
2008	Gillnet	N	110.69	45.84	6.18	15.39	2244.23	7.81	8.37	1.76	0.02	0.13	0.11	8.89
2009	Gillnet	N	110.71	55.69	6.54	16.17	1912.8	7.91	8.47	2.95	0.01	0.21	0.09	56.47
2010	Gillnet	N	110.74	69.06	6.58	17.94	1853.06	7.92	8.03	2.97	0.02	0.17	0.11	12.77
2011	Gillnet	N	110.84	64.79	6.53	17.54	1713.43	7.8	7.18	5.38	0.02	0.07	0.07	7.71
2012	Gillnet	N	110.68	42.55	6.77	18.04	2229.54	7.87	7.13	3.51	0.02	0.08	0.09	15.61
2013	Gillnet	N	110.78	64.06	6.51	17.28	2087.51	7.83	7.31	4.07	0.02	0.1	0.09	22.42
2014	Gillnet	N	110.81	58.34	6.97	17.54	1818.81	7.84	7.49	4.02	0.03	0.05	0.09	18.82
2015	Gillnet	N	110.85	72.67	6.74	17.52	1790.27	7.84	7.47	4.77	0.02	0.06	0.07	30.82
2016	Gillnet	N	110.72	58.82	6.93	18.13	2030.89	7.88	7.79	3.39	0.02	0.05	0.08	30.67
2017	Gillnet	N	110.86	65.48	6.85	17.39	1659.15	7.8	7.19	3.42	0.02	0.03	0.07	42.23
2018	Gillnet	N	110.78	58.83	6.89	18.16	1980.56	7.75	7.42	3.48	0.02	0.06	0.22	17.54

Appendix 2 Continued.

Year	DataSource	Basin	Lake Height	Precipitation	Wave Index	Water Temp	Conductivity	pH	DO	Turbidity	TP	Hg-methyl	NH3-N	Fcoli
2008	Gillnet	S	110.69	45.84	11.1	15.39	2244.23	7.81	8.37	1.76	0.02	0.13	0.11	19.43
2009	Gillnet	S	110.71	55.69	11.11	16.17	1912.8	7.91	8.47	2.95	0.02	0.21	0.09	33.37
2010	Gillnet	S	110.74	69.06	12.87	17.94	1853.06	7.92	8.03	2.97	0.02	0.17	0.11	224.19
2011	Gillnet	S	110.84	64.79	11.59	17.54	1713.43	7.8	7.18	5.38	0.02	0.07	0.07	23.9
2012	Gillnet	S	110.68	42.55	12.14	18.04	2229.54	7.87	7.13	3.51	0.02	0.08	0.09	25.45
2013	Gillnet	S	110.78	64.06	11.62	17.28	2087.51	7.83	7.31	4.07	0.02	0.1	0.09	23.26
2014	Gillnet	S	110.81	58.34	12.36	17.54	1818.81	7.84	7.49	4.02	0.02	0.05	0.09	38.24
2015	Gillnet	S	110.85	72.67	12.24	17.52	1790.27	7.84	7.47	4.77	0.02	0.06	0.07	179.09
2016	Gillnet	S	110.72	58.82	13.15	18.13	2030.89	7.88	7.79	3.39	0.02	0.05	0.08	32
2017	Gillnet	S	110.86	65.48	11.69	17.39	1659.15	7.8	7.19	3.42	0.02	0.03	0.07	44.21
2018	Gillnet	S	110.78	58.83	11.71	18.16	1980.56	7.75	7.42	3.48	0.02	0.06	0.22	27

Appendix 3. Log-transformed nearshore (trapnet) catch-per-unit-effort (CPUE) for Non-Metric Multidimensional Scaling ordination.

Species codes are listed in Appendix 1.

Year	Basin	SoakTime	ALE	BAK	BLC	BLG	BLN	BLS	BOW	BRB	BRS	BSLV	CARP	CHC	CHP	COM	CRC	EMS	FHM	FWD	GOF	GOH
2008	N	3.00	0.51	1.39	0.85	3.04	0.00	0.29	0.51	1.61	0.00	0.00	2.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2009	N	1.04	0.00	0.67	0.00	1.76	0.00	0.00	1.07	0.00	0.00	0.00	0.67	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2013	N	20.76	0.00	3.37	0.89	3.43	0.05	0.05	1.04	1.59	0.09	0.00	1.09	0.29	0.00	0.05	0.00	0.00	0.00	0.13	0.00	0.00
2014	N	21.23	1.85	2.14	0.17	2.90	0.05	0.29	1.04	1.79	0.05	0.00	0.64	0.17	0.05	0.00	0.05	0.05	0.05	0.13	0.00	0.00
2015	N	23.35	2.09	0.77	0.04	2.99	0.00	0.33	0.81	1.39	0.08	0.00	0.50	0.04	0.00	0.00	0.00	0.39	0.00	0.08	0.00	0.00
2016	N	25.70	2.05	4.88	0.04	2.88	0.30	0.00	0.66	1.28	0.04	0.04	0.36	0.00	0.00	0.00	0.00	0.24	0.00	0.00	0.00	0.00
2017	N	24.11	2.37	5.08	0.29	3.99	0.63	0.00	0.56	2.09	0.00	0.12	1.55	0.00	0.04	0.04	0.00	0.15	0.04	0.04	0.00	0.00
2018	N	24.35	0.08	3.17	0.25	3.23	0.04	0.00	0.89	1.96	0.19	0.00	1.10	0.12	0.00	0.00	0.00	0.08	0.04	0.08	0.04	0.00
2008	S	4.88	1.18	0.19	0.00	2.66	0.00	0.00	0.34	0.60	0.00	0.00	1.81	0.00	0.00	0.00	0.00	0.19	0.00	0.00	0.00	0.00
2009	S	4.04	0.00	0.22	0.00	2.91	0.00	0.00	0.56	0.22	0.00	0.00	0.56	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2013	S	32.42	0.27	1.81	0.64	2.83	0.06	0.00	0.82	1.38	0.06	0.00	0.79	0.09	0.03	0.00	0.00	0.03	0.00	0.06	0.00	0.00
2014	S	30.13	1.82	1.80	0.62	3.07	0.06	0.00	0.87	1.91	0.00	0.00	0.67	0.15	0.00	0.00	0.00	0.09	0.00	0.09	0.00	0.00
2015	S	26.29	1.09	1.63	0.04	3.64	0.11	0.00	0.76	1.62	0.27	0.00	0.52	0.00	0.00	0.00	0.00	0.85	0.00	0.04	0.00	0.00
2016	S	26.34	1.46	4.95	0.04	2.55	0.74	0.00	0.50	0.71	0.00	0.04	0.72	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2017	S	33.19	2.55	6.04	0.09	4.61	0.58	0.00	0.60	1.69	0.00	0.11	0.66	0.09	0.00	0.00	0.03	0.71	0.14	0.03	0.00	0.00
2018	S	39.63	0.23	1.89	0.05	2.48	0.05	0.00	0.89	1.49	1.11	0.00	0.75	0.62	0.00	0.10	0.00	0.02	0.00	0.18	0.00	0.02

Appendix 3 Continued.

Year	Basin	SoakTime	GOS	GSF	GZS	LMB	LNG	LOG	MGMT	NOP	PUD	QLB	ROB	ROG	RUD
2008	N	3.00	2.48	0.00	1.95	1.61	0.00	0.00	0.00	0.00	3.01	0.00	1.30	0.00	0.00
2009	N	1.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	2.67	0.00	0.00	0.00	0.00
2013	N	20.76	1.62	0.05	0.13	4.43	0.00	0.00	0.00	0.05	3.08	0.00	0.87	0.00	0.00
2014	N	21.23	1.84	0.00	0.45	1.84	0.32	0.00	0.00	0.00	2.57	0.00	0.71	0.25	0.59
2015	N	23.35	0.95	0.08	0.39	1.78	0.19	0.00	0.00	0.16	1.80	0.00	0.16	0.62	0.44
2016	N	25.70	0.70	0.98	0.07	2.46	0.07	0.00	0.00	0.00	1.72	0.00	1.16	2.45	0.21
2017	N	24.11	0.73	1.16	0.67	3.32	0.04	0.00	0.04	0.08	1.55	0.00	1.46	2.44	0.12
2018	N	24.35	2.68	0.00	0.51	2.00	0.73	0.00	0.00	0.04	2.81	0.04	0.45	0.00	0.75
2008	S	4.88	1.88	0.00	0.60	1.55	0.00	0.00	0.00	0.00	3.20	0.00	0.19	0.00	0.00
2009	S	4.04	0.56	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.19	0.00	0.69	0.00	0.00
2013	S	32.42	1.78	0.00	0.20	1.54	0.24	0.00	0.00	0.12	3.04	0.00	1.40	0.00	0.12
2014	S	30.13	1.15	0.00	1.00	2.28	1.31	0.03	0.00	0.15	2.36	0.00	0.77	0.09	0.09
2015	S	26.29	1.40	0.11	0.69	2.05	0.61	0.00	0.00	0.11	2.16	0.00	0.76	0.32	0.27
2016	S	26.34	0.52	1.20	0.11	4.13	0.04	0.00	0.00	0.00	1.55	0.00	1.01	2.14	0.32
2017	S	33.19	0.86	0.58	0.39	3.37	0.09	0.03	0.09	0.06	2.00	0.00	1.94	2.13	0.29
2018	S	39.63	1.73	0.00	0.71	1.44	0.46	0.00	0.00	0.02	3.07	0.00	0.89	0.00	0.24

Appendix 3 Continued.

Year	Basin	SoakTime	SFS	SHR	SMB	SVR	TAM	TED	TGM	WAE	WHP	WHS	YEB	YEP
2008	N	3.00	0.00	0.00	0.29	0.00	0.00	0.00	0.00	0.00	0.69	0.69	0.29	1.61
2009	N	1.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2013	N	20.76	0.00	0.09	0.57	0.05	0.09	0.05	0.00	0.09	1.57	1.25	0.65	1.87
2014	N	21.23	0.00	0.00	0.09	0.05	0.00	0.05	0.00	0.00	0.88	0.64	0.39	1.28
2015	N	23.35	0.00	0.00	0.00	0.04	0.00	0.04	0.00	0.00	0.44	0.08	0.00	1.40
2016	N	25.70	0.00	0.00	0.00	0.00	0.18	0.00	0.00	0.00	0.33	0.04	0.00	0.70
2017	N	24.11	0.00	0.00	0.00	0.00	0.48	0.04	0.00	0.00	0.29	0.08	0.38	1.91
2018	N	24.35	0.00	0.12	0.64	0.00	0.00	0.00	0.00	0.00	1.56	0.45	0.00	0.48
2008	S	4.88	0.00	0.00	0.48	0.00	0.00	0.00	0.00	0.00	2.26	0.60	0.00	1.05
2009	S	4.04	0.00	0.00	0.56	0.00	0.00	0.00	0.00	0.00	0.40	0.22	0.00	0.40
2013	S	32.42	0.00	0.00	0.52	0.03	0.00	0.00	0.00	0.03	0.93	0.50	0.20	1.74
2014	S	30.13	0.00	0.03	0.15	0.03	0.00	0.06	0.03	0.03	1.31	1.13	0.40	1.80
2015	S	26.29	0.00	0.00	0.04	0.00	0.43	0.00	0.00	0.00	0.32	0.63	0.04	1.72
2016	S	26.34	0.00	0.00	0.11	0.00	0.17	0.00	0.00	0.00	0.94	0.11	0.00	0.32
2017	S	33.19	0.03	0.00	0.11	0.00	0.24	0.03	0.00	0.00	0.43	0.24	0.41	2.15
2018	S	39.63	0.00	0.34	0.36	0.02	0.00	0.00	0.00	0.00	1.76	1.13	0.12	1.38

Appendix 4. Log-transformed offshore (gillnet) catch-per-unit-effort (CPUE) for Non-Metric Multidimensional Scaling ordination. Species codes are listed in Appendix 1.

Year	Basin	SoakTime	ALE	BOW	BRB	BRT	CARP	CHC	FWD	GRR	GZS	LAS	LMB	LNG
2008	N	24.25	0.00	0.00	0.08	0.88	0.84	0.35	0.00	0.04	0.04	0.04	0.00	0.04
2009	N	28.20	0.07	0.03	0.03	0.93	1.11	0.47	0.00	0.38	0.10	0.03	0.03	0.00
2010	N	26.75	0.00	0.00	0.26	0.87	0.62	0.26	0.00	0.73	0.00	0.07	0.04	0.00
2011	N	22.02	0.00	0.00	0.13	0.78	1.10	0.65	0.04	1.20	0.04	0.00	0.04	0.00
2012	N	24.17	0.04	0.00	0.00	0.86	0.63	0.32	0.00	0.65	0.08	0.00	0.04	0.12
2013	N	20.93	0.00	0.00	0.05	0.69	0.54	0.36	0.00	0.09	0.29	0.00	0.00	0.09
2014	N	28.18	0.00	0.07	0.10	0.43	0.49	0.72	0.00	0.40	0.00	0.03	0.03	0.13
2015	N	26.38	0.00	0.00	0.40	0.65	0.35	0.78	0.00	1.04	0.00	0.07	0.00	0.26
2016	N	27.57	0.00	0.07	0.20	0.46	0.34	0.46	0.00	0.66	0.04	0.17	0.00	0.10
2017	N	24.82	0.08	0.11	0.00	0.59	0.25	0.76	0.00	0.15	0.00	0.04	0.04	0.18
2018	N	24.50	0.00	0.00	0.04	0.90	0.50	0.89	0.45	0.15	0.08	0.19	0.00	0.04
2008	S	53.65	0.02	0.00	0.00	0.78	0.78	0.33	0.00	0.19	0.04	0.05	0.02	0.00
2009	S	36.30	0.05	0.08	0.18	0.63	1.00	0.35	0.00	0.72	0.08	0.05	0.00	0.05
2010	S	37.00	0.03	0.05	0.10	0.72	1.18	0.62	0.00	0.78	0.05	0.03	0.03	0.08
2011	S	36.40	0.00	0.00	0.15	0.75	0.79	0.96	0.00	0.91	0.03	0.00	0.00	0.13
2012	S	28.42	0.00	0.07	0.00	0.51	0.72	0.51	0.00	0.83	0.16	0.00	0.00	0.07
2013	S	37.93	0.00	0.08	0.00	0.65	0.58	0.55	0.00	0.19	0.00	0.00	0.00	0.00
2014	S	32.13	0.00	0.09	0.00	0.50	0.27	0.50	0.00	0.93	0.03	0.00	0.00	0.12
2015	S	25.52	0.04	0.04	0.18	0.68	0.27	0.70	0.00	0.58	0.00	0.04	0.00	0.11
2016	S	34.00	0.03	0.11	0.06	0.80	0.41	0.48	0.00	0.80	0.00	0.03	0.00	0.08
2017	S	47.57	0.00	0.02	0.00	0.71	0.39	0.87	0.00	0.60	0.00	0.06	0.00	0.02
2018	S	45.83	0.00	0.02	0.02	0.77	0.60	0.77	0.18	0.62	0.00	0.06	0.02	0.10

Appendix 4 Continued.

Year	Basin	SoakTime	LNG	NOP	QLB	QUB	RUD	SHR	SMB	SVR	TGM	WAE	WHP	WHS
2008	N	24.25	0.04	0.00	0.00	0.00	0.22	0.58	0.04	0.00	1.21	0.00	0.08	0.00
2009	N	28.20	0.00	0.00	0.10	0.00	0.10	0.07	0.03	0.00	1.34	0.00	0.25	0.07
2010	N	26.75	0.00	0.00	0.00	0.00	0.14	0.07	0.11	0.00	1.20	0.00	0.32	0.00
2011	N	22.02	0.00	0.04	0.00	0.00	0.31	0.09	0.00	0.00	1.81	0.04	0.31	0.00
2012	N	24.17	0.12	0.00	0.00	0.00	0.29	0.00	0.00	0.00	1.20	0.04	0.12	0.04
2013	N	20.93	0.09	0.05	0.00	0.00	0.05	0.48	0.00	0.00	1.07	0.00	0.05	0.00
2014	N	28.18	0.13	0.00	0.00	0.03	0.22	0.33	0.00	0.07	1.51	0.00	0.10	0.00
2015	N	26.38	0.26	0.07	0.00	0.04	0.04	0.26	0.14	0.07	1.20	0.04	0.07	0.00
2016	N	27.57	0.10	0.00	0.00	0.07	0.14	0.48	0.07	0.00	1.20	0.04	0.07	0.00
2017	N	24.82	0.18	0.08	0.00	0.18	0.34	0.47	0.00	0.00	1.19	0.08	0.04	0.08
2018	N	24.50	0.04	0.08	0.00	0.00	0.08	0.37	0.00	0.00	1.62	0.00	0.12	0.00
2008	S	53.65	0.00	0.11	0.00	0.00	0.14	0.60	0.02	0.00	1.08	0.02	0.07	0.02
2009	S	36.30	0.05	0.00	0.00	0.00	0.15	0.24	0.15	0.00	1.49	0.03	0.31	0.05
2010	S	37.00	0.08	0.13	0.00	0.00	0.52	0.17	0.13	0.00	1.72	0.08	0.58	0.03
2011	S	36.40	0.13	0.00	0.00	0.00	0.31	0.22	0.00	0.00	1.34	0.00	0.31	0.00
2012	S	28.42	0.07	0.03	0.00	0.03	0.10	0.10	0.00	0.00	1.12	0.03	0.19	0.00
2013	S	37.93	0.00	0.00	0.00	0.00	0.05	0.19	0.00	0.03	1.13	0.00	0.05	0.00
2014	S	32.13	0.12	0.00	0.00	0.00	0.17	0.32	0.00	0.22	1.03	0.00	0.03	0.00
2015	S	25.52	0.11	0.00	0.00	0.24	0.04	0.27	0.00	0.04	0.85	0.04	0.08	0.04
2016	S	34.00	0.08	0.03	0.00	0.11	0.08	0.42	0.03	0.08	0.93	0.00	0.06	0.03
2017	S	47.57	0.02	0.04	0.00	0.02	0.12	0.32	0.02	0.02	1.15	0.00	0.04	0.00
2018	S	45.83	0.10	0.04	0.00	0.10	0.08	0.63	0.00	0.00	1.35	0.06	0.08	0.00

CURRICULUM VITA

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EDUCATION

State University of New York College of Environmental Science and Forestry

- Master of Science in Fish and Wildlife Biology and Management, expected May 2019
  - Thesis: Fish Assemblage Succession Within a Recovering Urban Lake
  - GPA 3.86
- Bachelor of Science in Aquatic and Fisheries Science (Cum Laude), May 2015
  - Marine Sciences Minor, Honors Program, GPA 3.32

PROFESSIONAL EXPERIENCE

Graduate Research Project Assistant - State University of New York College of Environmental Science and Forestry, Syracuse, NY (January 2017-Present)

- Field crew lead and operation of watercraft
- Collection and identification of fish species with: trap net, gill net, beach seine, and boat electrofishing
- Lake-wide macrophyte and Centrarchid nesting surveys
- Lake Sturgeon mark/recapture via PIT and Carlin tags
- Aging of multiple sportfish species via otolith cross section and pressed scales
- Extensive data wrangling and management
- Teaching Assistant for EFB 486 Ichthyology Spring 2017 semester (January - May)

Fish Culturist (Contractor) - Tunison Laboratory of Aquatic Science (U.S. Geological Survey), Cortland, NY (June 2016 - January 2017)

- Aquaculturist for Atlantic Salmon (*Salmo salar*), Lake Herring (*Coregonus artedi*), and Bloater (*Coregonus hoyi*)

Fisheries Research Technician (Contractor) - Tunison Laboratory of Aquatic Science (U.S. Geological Survey), Cortland, NY (June 2015 - June 2016)

- Phase III of the Fish Enhancement, Mitigation, and Research Fund (FEMRF) project: Evaluation of Threatened, Endangered, and Declining Species of the Major Tributaries to the St. Lawrence River
  - Fish collection/identification, site characterization, GIS, and data management
  - Evaluation of fish assemblage above and below Hogansburg Hydroelectric Dam on the St. Regis River prior to removal
- Coordinated data exchange between federal and state agencies
- Lake Sturgeon population assessment in New York tributaries to the St. Lawrence River
- Experimental and field eDNA sample collection in the St. Regis River, NY

Undergraduate Researcher - SUNY College of Environmental Science and Forestry, Syracuse, NY

- Honor's Research Quantifying Decomposition of Allochthonous Matter in Forest Pools of Central New York (September 2014 - May 2015)
    - Role of microbial decomposition on water chemistry and pool ecosystem
- Teaching Assistant - General Biology I Laboaratory, SUNY ESF (Fall 2013)
    - Instructed 3 hour weekly general biology lab along with a graduate TA
- Lab Research for Dr. Kim Schulz (October 2012 - May 2013)
    - Benthic invertebrate sorting and macrophyte sample measurements

Student Intern - John Carroll University, Cleveland, OH (May - August 2014)

- Field research studying evolutionary trends in color morphology of the eastern red-backed salamander, *Plethodon cinereus*, across the Great Lakes Region
- Collected photographs and tail tips for six hours at each of twenty-five sites
- Used ImageJ to quantify stripe area of over 1000 photographed individuals

AWARDS

- 2015 Maple Leaf Award for Academic Excellence, Outstanding Leadership Involvement, and Volunteer Service (SUNY ESF)
- Eagle Scout: 9/29/2010

SKILLS

- Computer: ArcGIS programs, ImageJ, Microsoft Office, R
- Outdoor: Knot tying, Map and Compass, SCUBA/Snorkeling, Camping, Backpacking
- Technical: Trailering/Operating Motor Boats, Standard Vehicle Operation
- Field Sampling: Fish/Amphibian/Macrophyte/Invertebrate Identification, Li-COR and YSI Meter, Operation/Maintenance, Boat and Backpack Electrofishing, Seine/Gill/Trap/Plankton/Trawl netting, eDNA Collection
- Laboratory Sampling: Basic Phytoplankton/Zooplankton identification, Aquatic Invertebrate Identification, Spectrophotometry, Electron Microscopy, Acid Washing

ACTIVITIES

- Graduate Student Association Treasurer SUNY ESF (2017 - Present)
    - Created budget and managed financial operations for independent non-profit graduate organization
- Member National American Fisheries Society (2018 - Present)
- Member New York State Chapter American Fisheries Society (2013 - Present)
- Undergraduate Student Association Treasurer SUNY ESF (2014 - 2015)

PRESENTATIONS

Kronisch, GR and NH Ringler. Evaluation of Fish Assemblage Response to the Remediation of an Urban Lake. 2019. New York Chapter and Northeast Division of the American Fisheries Society Annual Meeting, Poughkeepsie, NY. (Oral)

Limburg, K, D Breitburg, J Cramer, SS Ekoh, A Gårdmark, Y Heimbrand, **G Kronisch**, J Le, LA Levin, LR McCormick, S McNulty, A Orio, MA Samson, S Shatto and KM Smith. 2018. Valuing Ecosystem Services at Risk from Deoxygenation of Oceans, Estuaries, and Coastal Seas. American Fisheries Society 2018 National Meeting, Atlantic City, NJ. (Oral)

Kronisch, GR, DJ Stewart, and NH Ringler. Longitudinal Changes of Fish Assemblages in Nine Mile Creek, Tributary to a Recovering Urban Lake. 2018. New York Chapter of the American Fisheries Society Annual Meeting, Cooperstown, NY. (Poster)

Kronisch, GR, JA DiRado, JH Johnson, and MA Chalupnicki. Re-establishing Atlantic Salmon into Lake Ontario via the Salmon River, New York. 2017. New York Chapter of the American Fisheries Society Annual Meeting, Buffalo, NY. (Poster)

Kronisch, GR, KS Hanak, and JE McKenna. Young of Year Habitat Preference in St. Lawrence River Tributaries. 2016. New York Chapter of the American Fisheries Society Annual Meeting, Cooperstown, NY. (Poster)