The role of internal nitrogen loading in supporting non-N-fixing harmful cyanobacterial blooms in the water column of a large eutrophic lake

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Running head: Internal N loading supports cyanoHABs in large lakes

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Abstract

Western Lake Erie cyanobacterial harmful algal blooms (cyanoHABs) occur every summer as a result of anthropogenic nutrient loading. While the physiological importance of nitrogen (N) in supporting bloom biomass and toxin production is established, the role of internal N recycling in the water column to support bloom maintenance is lacking. Over three field seasons (2015–2017), we collected water from western Lake Erie and employed bottle incubations with ^{15}N-ammonium (NH_4^+) enrichments to determine NH_4^+ regeneration and potential uptake rates in the water column. Potential NH_4^+ uptake rates followed spatial and seasonal patterns, with greatest rates measured nearest the Maumee River inflow and during peak bloom months (August and September). Regeneration followed a similar spatial pattern but was greatest in early summer (June and July) and supported ~20–60% of potential NH_4^+ demand during the height of the bloom. Basin-wide internal NH_4^+ regeneration during the April–October period could supply NH_4^+ at 60–200% of annual external N loading to the western basin. These results help explain how non-N-fixing cyanoHABs in Lake Erie, and other large, eutrophic lakes, continue producing biomass and N-rich toxins long after spring nutrient loads are exhausted or transported to other areas. Internal N loads are ultimately driven by external N loads; in low precipitation years, external nutrient loads result in smaller blooms, producing less substrate for subsequent internal

N loads. Overall, these findings, along with others, confirm that both internal N loading and external N loading must be considered when evaluating cyanoHAB management strategies.

Introduction

Microcystis-dominated cyanobacterial harmful algal blooms (cyanoHABs) in western Lake Erie have increased in severity since the mid-1990s (Steffen et al. 2014). Research on and management of high nutrient loads from agricultural watersheds have focused on phosphorus (P) as the key driver of cyanoHABs in Lake Erie (e.g., Martin et al. 2021) and globally (Paerl et al. 2016). Although total P and total nitrogen (N) loads have decreased in recent decades, the proportion of dissolved reactive P in the total P load to Lake Erie has increased and correlates with annual bloom size (Jarvie et al. 2017, but see River and Richardson 2019). Likewise, the fraction of the total N (TN) load comprised of chemically reduced N forms (as total Kjeldahl N; TKN) delivered to western Lake Erie has increased concurrently and is also related to cyanoHAB biomass (Newell et al. 2019).

Uncertainties in dissolved reactive P analyses related to nanoparticulate P passing through 0.45 μm filters may result in over-estimation of Maumee River fluxes to western Lake Erie by ~50% (River and Richardson 2019). The temporal disconnect, exceeding the basin residence time, between spring dissolved reactive P loads and cyanoHABs in late summer is also problematic (Newell et al. 2019). In contrast, higher TKN (including highly bioavailable ammonium (NH_4^+) and urea) proportions in the Maumee River TN load in mid-summer coincide well with cyanoHAB initiation in July (Newell et al. 2019). Cyanobacteria, particularly non-N fixing taxa, are excellent at scavenging NH_4^+ and often outcompete eukaryotic organisms (e.g., diatoms) for NH_4^+ (Blomqvist et al. 1994). *Microcystis* and many other toxin-producing, bloom-

69 forming taxa (e.g., *Planktothrix*) cannot fix atmospheric dinitrogen (N_2) and thus rely on
70 combined N, particularly NH_4^+, in the water column (Paerl et al. 2016). Maximum uptake rates
71 (V_{max}) for NH_4^+ are 4–6 times greater than those for nitrate (NO_3^-) in *Microcystis* (Takamura et
72 al. 1987), while diatoms are often more competitive for NO_3^- (Glibert et al. 2016).
73 Microcystin and biomass concentrations in non-N fixing cyanobacteria increase in
74 response to NH_4^+ additions (Chaffin et al. 2018), and NH_4^+ and urea induce upregulation of
75 microcystin-associated genes (Harke et al. 2016) more quickly than NO_3^- (Chaffin et al. 2018).
76 Conversely, low NH_4^+ concentrations inhibit toxin production due to *ntcA* (global cyanobacteria
77 N regulation gene) activation, the product of which binds to the *mcy* gene cassette responsible for
78 microcystin production and represses its activity (Kuniyoshi et al. 2011). NH_4^+ bioreactivity
79 leads to rapid assimilation and recycling within the sediments and water column, making it
80 difficult to accurately characterize in situ NH_4^+ concentrations (McCarthy et al. 2013). Thus,
81 snapshot sampling and monitoring efforts for NH_4^+ concentrations cannot accurately estimate
82 NH_4^+ availability in situ, which requires measuring NH_4^+ assimilation and turnover/recycling
83 rates (Gardner et al. 2017).
84 Over three field seasons (April through October, 2015–2017), NH_4^+ regeneration and
85 potential uptake rates were quantified in the western Lake Erie water column to evaluate the
86 importance of internal N loading in supporting non-N-fixing cyanoHAB biomass and toxin
87 production. Potential NH_4^+ uptake includes both assimilation into primary producer biomass and
88 nitrification (conversion of NH_4^+ to NO_3^-). NH_4^+ regeneration includes, but is not limited to,
89 remineralization of organic compounds, microplankton excretions (including algal exudation),
90 and sloppy feeding by grazers (Hopkinson et al. 1987). We also evaluated the capacity of NH_4^+

regeneration to support water column NH_4^+ demand across the bloom season relative to external TN loading.

We expected that NH_4^+ regeneration and potential uptake rates would follow patterns described previously (i.e., photic exceeding dark NH_4^+ uptake, highest NH_4^+ uptake during peak blooms, potential NH_4^+ uptake exceeding regeneration) and follow spatial and temporal trends consistent with seasonality and distance from nutrient sources. Specifically, we hypothesized that: (1) both NH_4^+ regeneration and potential uptake rates would decrease with distance from the Maumee River mouth; (2) both NH_4^+ regeneration and potential uptake rates would increase from late spring/early summer to August and September (peak bloom); and (3) rates measured in near-surface waters would be greater than those from near-bottom waters. We anticipated that the proportion of internal NH_4^+ regeneration capable of supporting potential NH_4^+ uptake would increase throughout from bloom initiation to bloom peak. Given the importance of NH_4^+ to the metabolic demands of cyanoHABs and the difficulty of accurately determining in situ NH_4^+ concentrations, quantifying these NH_4^+ cycling pathways is critical to understanding and managing non-N-fixing cyanoHABs in western Lake Erie and other large eutrophic lakes. These kinds of measurements are exceedingly rare in the Great Lakes and other freshwater systems but are critically needed for informing and validating ecosystem models and management actions.

Materials

Field sampling

Figure 1. Western Lake Erie sampling stations. Stations are located at the following coordinates (latitude, longitude): WE6 (41.70555, -83.386533), WE2 (41.7638, -83.330617), WE4 (41.8269833, -83.192117), and WE13 (41.7429167, -83.138783).

Water samples were collected in conjunction with the NOAA Great Lakes Environmental Research Laboratory (GLERL) weekly HABs monitoring program in 2015, 2016, and 2017. Further details on bloom severity, initiation, and duration are presented in supplemental material. Three stations (WE6, WE2, WE4; Fig. 1) were sampled monthly from June–September 2015 (Table S1), and these plus WE13 (Fig. 1) were sampled in 2016 and 2017 during May–October (with an additional sampling at WE2 in April 2016). These stations represent a spatial and depth gradient moving away from the Maumee River inflow. Not all stations were sampled during each month, and sampling varied across years due to opportunity and weather constraints, particularly

123 in 2016, when frequent storms interrupted scheduled sampling events. At each station, water was
124 collected from two depths (~1 m below the surface and ~1 m above the sediment-water interface)
125 with a 5-L Niskin bottle. 10 L of water was collected from each depth and transferred into 20-L
126 polyethylene carboys. 12 ml ambient nutrient (orthophosphate (o-PO_4^{3-}), NH_4^+, urea, nitrite
127 (NO_2^-), NO_3^-) samples were filtered immediately on-site (0.22 μm Nylon filters) into 15-ml
128 polypropylene sample tubes and frozen in the field (dry ice) until transfer into a -20 °C lab
129 freezer. Other biological and physicochemical parameters (water temperature, dissolved oxygen
130 (DO), total P, Secchi depth, conductivity, photosynthetically active radiation (PAR), turbidity,
131 total suspended solids, and chlorophyll *a*, phycocyanin, and microcystin concentrations) were
132 collected from surface waters and analyzed by NOAA GLERL and CIGLR (NOAA Great Lakes
133 Environmental Research Laboratory; Cooperative Institute for Great Lakes Research, University
134 of Michigan, 2019). All physicochemical data other than nutrients are available in Supplemental
135 Information. The N and P mass of Maumee River loads was obtained from the National Center
136 for Water Quality Research (NCWQR) database, and other ambient environmental
137 characteristics of interest (average daily wind speed, wind direction, air temperature) were
138 selected from buoy data hosted by NOAA's National Data Buoy Center
139 (https://www.ndbc.noaa.gov; last accessed 11/30/2020) and Michigan Technological
140 University's Upper Great Lakes Observing System (UGLOS; http://uglos.mtu.edu; last accessed
141 11/30/2020).

142 **Incubations**

143 NH_4^+ regeneration and potential uptake rates were quantified via enrichment with ^{15}N-
144 labeled NH_4^+ (McCarthy et al. 2013). 1-L aliquots of water from each station and depth were
145 amended with $^{15}NH_4Cl$ (98 atom % ^{15}N, Sigma Aldrich; final concentration 8–24 µM, depending

on sampling date and bloom conditions; Table S1), mixed, and distributed among six clear, colorless, 125 ml Nalgene bottles for triplicate light and dark (foil-wrapped) incubations. Initial samples from each bottle were collected immediately using a syringe and canula, filtered (0.22 μm), and frozen at -20 °C until analysis. Bottles were incubated in a mesh bag for 16–25 hours (Table S1) in simulated ambient lake conditions (outdoor water bath, not temperature-controlled) before being filtered (0.22 μm) into 15-ml polypropylene sample tubes and immediately frozen at -20 °C until total NH_4^+ analysis ($^{14}N + {}^{15}N$). Pre- and post-incubation samples for $^{15}NH_4^+$ analyses were filtered into 12-ml Exetainers (Labco) with no headspace, sealed with double-wadded septa caps, and stored in the dark at room temperature (17 °C) until analysis (within one week).

Sample analysis and rate calculations

Total NH_4^+ concentrations were determined via colorimetric flow-injection analysis (Lachat Quikchem 8500). $^{15}NH_4^+$ concentrations were quantified via OX-MIMS (Yin et al. 2014). Additional details on OX-MIMS are provided in supplemental material. Total NH_4^+ concentrations were used to determine the $^{15}NH_4^+$ to total NH_4^+ ratio for use in isotope dilution models (Blackburn 1979; Caperon et al. 1979). Potential NH_4^+ uptake rates were qualified as potential rates due to saturating-level isotope additions (McCarthy et al. 2013), but rates determined from saturating-level isotope amendments tend to converge with actual rates in highly productive aquatic systems (Glibert et al. 1988).

In August 2015, $^{15}NH_4^+$ was undetectable after 21 hours of incubation. However, a clear decrease (up to 50%) in $^{15}NH_4^+$ concentration was observed between the known amendment and initial sampling (~15 min). Therefore, we calculated NH_4^+ regeneration and potential uptake rates for the abbreviated period. Rates for one station near WE6 were measured from a separate

sampling event five days prior to the others, and these data are included in the Supplemental Materials and briefly mentioned in the Discussion, but not included in any statistical analyses.

Statistical analyses

All analyses were performed in R (version 3.6.1; R Core Team 2020). Rate data were visually examined for assumptions of normality and heteroscedasticity and via Shapiro-Wilk tests; all failed to meet normality assumptions, so they were log-transformed prior to analysis. All linear models were conducted with the *lm* function from the “lme4” package (Bates et al. 2015). Differences among treatment means were determined via Tukey’s HSD post hoc tests using the *emmeans* function from the "emmeans" package (Lenth et al. 2019). More detail about linear models can be found in Supplemental Information.

Spearman’s Rank correlations were calculated using the *rcorr* function from “Hmisc” package (Harrell 2019) to inform relationships between NH_4^+ regeneration and potential uptake and environmental variables. For correlations we added bloom day, defined as days plus or minus the bloom initiation day (as determined by the first report of cyanobacteria in NOAA GLERL HABs bulletins), as a variable.

Results

Ambient conditions

Table 1. Nutrient concentrations (μmol N or P L^{-1}) for ammonium (NH_4^+), nitrite (NO_2^-), nitrate (NO_3^-), orthophosphate (o-PO_4^{3-}), and urea near the water surface. BDL = below method detection limit.

Date	Station	NH_4^+ (μM)	NO_2^- (μM)	NO_3^- (μM)	o-PO_4^{3-} (μM)	Urea (μM)
22-Jun-15	WE6	7.01	1.21	260	4.10	2.71
	WE2	4.03	0.320	315	1.31	1.35
	WE4	0.646	0.120	22.0	0.040	1.18
20-Jul-15	WE6	5.36	1.93	196	4.19	2.71
	WE2	1.45	2.17	216	3.27	1.44
	WE4	BDL	1.58	49.5	0.040	0.692
10-Aug-15	WE6	BDL	0.740	91.2	0.510	0.470
	WE2	BDL	0.730	99.0	1.21	0.310
	WE4	BDL	0.260	4.98	0.050	0.310
28-Sep-15	WE6	0.650	BDL	0.032	5.56	0.500
	WE2	4.86	BDL	0.024	20.7	0.500
	WE4	0.300	BDL	0.014	6.92	1.50
19-Apr-16	WE2	4.21	0.730	45.6	0.170	0.450
30-May-16	WE4	1.72	BDL	9.50	0.050	0.600
	WE13	6.04	0.370	15.1	0.600	0.400
27-Jun-16	WE6	9.72	1.57	48.8	1.35	2.08
	WE2	5.49	0.500	26.1	0.280	0.560
	WE4	0.54	0.410	11.9	0.010	0.410
11-Jul-16	WE6	0.250	7.40	326	0.080	0.650
	WE2	2.10	0.870	30.6	0.130	0.390
	WE4	0.670	0.560	18.0	0.060	0.250
8-Aug-16	WE6	0.020	0.130	4.97	0.060	0.490
	WE2	0.420	0.570	12.1	0.040	0.300
	WE4	BDL	0.790	8.84	0.050	0.570
	WE13	0.840	0.200	5.40	0.070	1.30
19-Sep-16	WE6	0.800	0.510	9.49	0.790	2.10
	WE2	2.41	0.710	5.78	0.890	0.770
17-Oct-16	WE4	1.86	0.430	19.0	0.570	0.470
	WE13	0.950	2.56	7.82	1.13	0.560
	WE6	1.87	0.650	29.5	0.800	1.20
	WE2	0.430	0.740	20.0	0.380	0.360
30-May-17	WE6	6.48	4.88	373	4.51	2.86
	WE2	2.87	4.85	314	2.65	2.26
	WE4	4.63	0.290	34.2	0.030	0.355
	WE13	1.88	0.450	36.8	0.040	0.423
12-Jun-17	WE6	7.30	4.94	352	4.10	2.71
	WE2	1.55	3.63	230	1.31	1.35
	WE4	2.75	0.540	40.0	0.040	1.18
	WE13	2.11	0.270	33.4	0.040	0.553
17-Jul-17	WE6	6.29	5.13	400	4.32	2.71
	WE2	2.55	5.38	364	2.89	1.44
	WE4	2.52	1.10	42.6	0.080	0.692
	WE13	0.760	0.860	37.5	0.090	0.501
14-Aug-17	WE6	0.330	1.46	122	0.530	0.820
	WE2	0.290	0.450	34.8	0.060	0.722
	WE4	0.720	1.65	77.0	0.090	0.446
	WE13	0.800	1.00	51.4	0.030	0.368
18-Sep-17	WE6	0.270	BDL	0.390	0.040	0.536
	WE2	BDL	0.020	0.700	0.063	0.290
	WE4	0.290	1.19	18.9	0.038	0.345
	WE13	0.260	0.030	0.990	0.039	0.424
10-Oct-17	WE6	BDL	0.020	0.390	0.170	0.539
	WE2	0.040	0.050	0.450	0.210	0.340
	WE4	0.630	0.720	20.7	0.180	0.326
	WE13	0.150	0.480	15.7	0.060	0.219

NH_4^+ concentrations peaked in June and decreased through the bloom season (Table 1), also decreasing along the distance gradient from the river input. NO_3^-+NO_2^- (NO_x) concentrations were usually 5–65 times greater than NH_4^+ and followed a similar spatial trend, with peak NO_x concentrations observed near the Maumee River discharge. NO_x concentrations generally peaked later than NH_4^+ concentrations before decreasing to less than 1 μM at the westernmost stations late in the sampling season. Urea concentrations followed similar patterns to those for NH_4^+, with higher concentrations in June decreasing through the sampling season. o-PO_4^{3-} concentrations in the westernmost part of the basin decreased from mid-summer until bloom initiation. High o-PO_4^{3-} concentrations (5–20 μM) were observed in surface waters in September 2015, despite most N forms being depleted. o-PO_4^{3-} concentrations in 2016 did not exhibit the decreasing trend observed in the other two years.

220 Table 2. Ambient biological and cyanotoxin (microcystins) concentrations in surface waters.

221 BDL = below method detection limit.

Date	Station	Phycocyanin (µg/L)	Chlorophyll *a* (µg/L)	Dissolved MC (µg/L)	Particulate MC (µg/L)
22-Jun-15	WE6	0.155	1.63	BDL	BDL
	WE2	0.218	3.84	BDL	BDL
	WE4	0.389	1.01	BDL	BDL
20-Jul-15	WE6	0.450	2.88	BDL	BDL
	WE2	BDL	6.59	BDL	BDL
	WE4	155	120	0.150	8.46
10-Aug-15	WE6	241	353	0.150	3.10
	WE2	46.8	187	0.150	9.19
	WE4	9.89	86.3	0.280	2.29
28-Sep-15	WE6	51.8	55.0	0.340	1.99
	WE2	22.4	28.0	0.140	0.910
	WE4	17.6	22.2	0.210	0.260
19-Apr-16	WE2	–	--	–	–
30-May-16	WE4	0.36	2.60	BDL	BDL
	WE13	0.24	4.80	BDL	BDL
27-Jun-16	WE6	0.993	4.25	BDL	BDL
	WE2	0.122	4.03	BDL	BDL
	WE4	0.105	4.31	BDL	BDL
11-Jul-16	WE6	4.46	36.0	0.155	0.547
	WE2	0.453	4.14	BDL	0.137
	WE4	0.131	1.82	BDL	BDL
8-Aug-16	WE6	14.6	62.2	0.300	4.70
	WE2	4.20	39.1	0.210	3.02
	WE4	0.251	7.77	0.110	0.270
	WE13	7.79	7.20	0.140	2.27
19-Sep-16	WE6	6.02	8.80	0.280	1.41
	WE2	0.312	5.52	0.130	0.210
17-Oct-16	WE4	0.056	2.71	0.100	0.010
	WE13	0.300	8.09	0.220	0.140
	WE6	0.164	0.400	0.190	BDL
	WE2	0.239	5.20	0.150	BDL
30-May-17	WE6	0.098	2.98	BDL	BDL
	WE2	0.092	2.56	BDL	BDL
	WE4	BDL	2.45	BDL	BDL
	WE13	0.167	10.1	BDL	BDL
12-Jun-17	WE6	BDL	3.41	0.110	BDL
	WE2	0.079	26.9	0.099	BDL
	WE4	BDL	7.61	0.109	BDL
	WE13	BDL	1.94	0.250	BDL
17-Jul-17	WE6	0.103	3.25	BDL	BDL
	WE2	0.461	19.8	BDL	BDL
	WE4	BDL	2.18	0.100	BDL
	WE13	BDL	3.16	BDL	BDL
14-Aug-17	WE6	453	532	0.590	21.6
	WE2	23.2	27.1	0.140	4.00
	WE4	19.3	30.6	0.370	3.70
	WE13	2.37	12.4	0.130	0.320
18-Sep-17	WE6	19.3	33.5	0.180	0.760
	WE2	68.7	35.1	0.180	0.300
	WE4	4.75	9.62	0.230	0.250
	WE13	109	53.1	BDL	0.550
10-Oct-17	WE6	26.1	40.9	0.210	0.560
	WE2	16.8	27.7	BDL	0.450
	WE4	0.423	5.11	BDL	BDL
	WE13	9.86	14.5	BDL	0.160

222

223 Phycocyanin (cyanobacteria) and chlorophyll *a* (Table 2), followed similar patterns
224 during each year and sampling season, peaking at WE6 in August, when ambient NH_4^+
225 concentrations were at or near undetectable levels. Maximum chlorophyll *a* and phycocyanin in
226 2017 were 1.5–2 times those from the same stations in 2015, although these peak values
227 represent samples from just below the surface, not within surface scums. Chlorophyll *a* was
228 correlated with bloom day, all N forms (except NO_2^- and TN), the TN mass and N:P ratio of the
229 previous week's Maumee River load, Secchi depth, water temperature, PAR, turbidity, total
230 suspended solids, and both dissolved and particulate microcystins (Table S2). Phycocyanin was
231 similarly correlated with bloom day, all N forms (except TN), the TN mass and N:P ratio of the
232 Maumee River load from the previous week, Secchi depth, turbidity, total suspended solids, and
233 both toxin measures, but was additionally correlated with wind speed (Table S2). Particulate and
234 dissolved microcystin concentrations followed similar spatial and temporal trends in 2016 and
235 2017, with maximum values at WE6 in August. In 2015, maximum dissolved and particulate MC
236 concentrations were observed at WE6 in September and WE2 in August. In 2017, peak
237 particulate microcystins (21.6 μg L^{-1}) were measured at WE6 in August, coinciding with
238 chlorophyll *a* and phycocyanin maxima. Both dissolved and particulate microcystins were
239 correlated with bloom day, several N concentrations (NH_4^+, NO_3^-, total dissolved N, dissolved
240 inorganic N, and particulate organic N), the TN mass of the Maumee River load from the
241 previous week, total suspended solids, chlorophyll *a*, and phycocyanin (Table S2). Neither
242 biomass nor toxin concentrations were correlated with any P form.

243 **Potential NH_4^+ uptake**

244 Potential NH_4^+ uptake rates were not influenced by depth across station ($F_{9,231}$ = 1.38; p =
245 0.196), month ($F_{7,233}$ = 6.07; p = 0.129), or year ($F_{3,237}$ = 11.8; p = 0.572), so surface and bottom

246 rates were averaged. Light NH_4^+ uptake closest to the Maumee River mouth (WE6 and WE2)
247 were greater than at WE4 or WE13 ($F_{3,327} = 29.2$; $p < 0.001$), but rates at WE6 and WE2, and
248 WE4 and WE13, were not different from each other (Fig. 2a). Light NH_4^+ uptake rates were
249 different across months ($F_{6,324} = 12.8$; $p < 0.001$), with highest rates in August 2015 (Fig. 2b).
250 Disregarding August 2015, the effect of month was still robust, but August was no longer
251 different from June, July, or September ($F_{6,306} = 5.24$; $p < 0.001$; Fig. 3a). Neither spatial
252 (station) nor yearly patterns changed from adjusting linear model input. Across all months
253 (excluding August 2015), light NH_4^+ uptake was different year-to-year ($F_{2,328} = 58.8$; $p < 0.001$).
254 Uptake rates were greater in 2015 than 2016 or 2017, but light NH_4^+ uptake rates in 2016 and
255 2017 were not different from each other (Fig. 3b).
256 Dark NH_4^+ uptake rates were not different between WE2 and WE6 or WE4, but rates at
257 all of these stations were higher than WE13 ($F_{3,327} = 13.1$; $p < 0.001$; Fig. 3a). Dark NH_4^+ uptake
258 did not differ between months ($F_{6,306} = 1.14$; $p = 0.338$; Fig. 4a) and were also greater in 2015
259 than 2016 or 2017 ($F_{2,328} = 32.2$; $p < 0.001$; Fig. 3b).
260
261

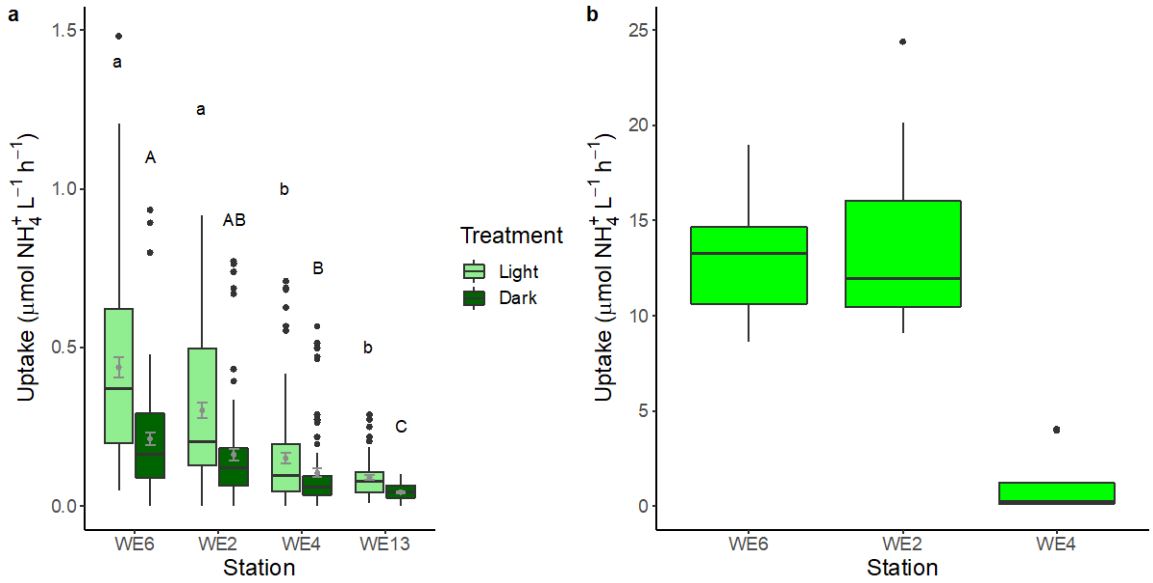

Figure 2. Potential NH_4^+ uptake rates (μmol L^{-1} h^{-1}) (a) in light and dark incubations across all sampling events by station, and (b) August 10, 2015 (no light/dark due to abbreviated incubation; note difference in y-axis scale). Means (± SE) are overlaid on each boxplot in panel (a) in gray. Letters reflect differences in uptake rates between stations (Tukey's HSD post-hoc tests) within each treatment (lowercase letters for Light, uppercase letters for Dark). *n* for each station: WE6 = 90, WE2 = 95, WE4 = 87, WE13 = 60.

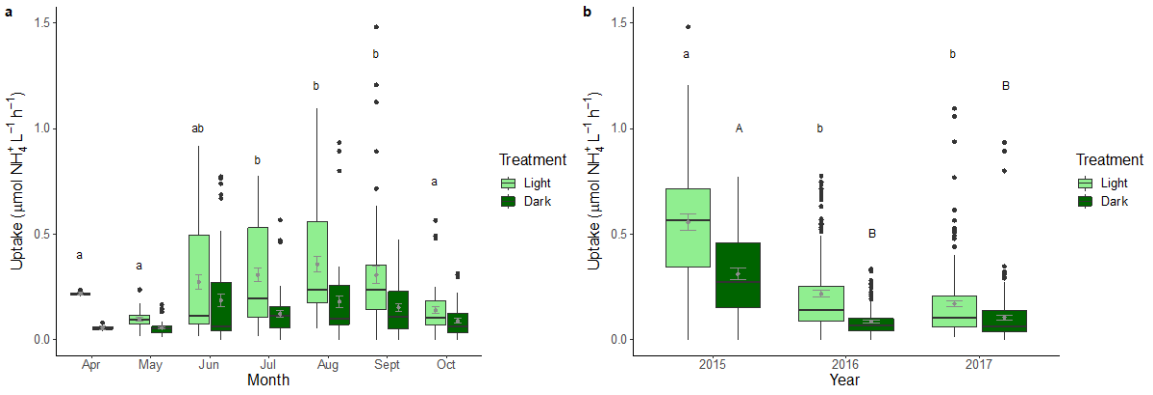

Figure 3. (a) Potential NH_4^+ uptake rates (μmol L^{-1} h^{-1}) in the light (light green) and dark (dark green) for each month across all three years (August 2015 excluded). Means (± SE) are overlaid

on each boxplot in gray. Letters indicate differences in uptake rates between months (Tukey's HSD post-hoc tests) in the light treatment; there were no differences in dark rates across months. *n* for each month: April = 6, May = 36, June = 66, July = 57, August = 66, September = 53, October = 48. (b) Potential NH_4^+ uptake rates (μmol L^{-1} h^{-1}) in the light (light green) and dark (dark green) across all three years (August 2015 excluded). Means (± SE) are indicated on each boxplot in gray. Letters indicate differences in uptake rates between years (Tukey's HSD post-hoc tests); yearly differences in each treatment (Light and Dark) were the same. *n* for each year: 2015 = 71, 2016 = 117, 2017 = 144).

Excluding August, light and dark NH_4^+ uptake rates in 2015 ranged from undetectable to 1.48 μmol N L^{-1} h^{-1} and undetectable to 0.772 μmol N L^{-1} h^{-1} (Figs. 3b, S1), respectively. The greatest rates (light = 20.2 μmol N L^{-1} h^{-1}, dark = 24.4 μmol N L^{-1} h^{-1}) were observed in August at WE6 and WE2. NH_4^+ uptake rates were generally higher at these stations than at WE4 (Fig. S1). In 2016, light and dark NH_4^+ uptake rates ranged from undetectable to 0.775 μmol N L^{-1} h^{-1} and undetectable to 0.334 μmol N L^{-1} h^{-1}, respectively (Figs. 3b, S2). Rates in 2016 peaked at WE6 in July and August and WE2 in August. By October, rates had nearly returned to those observed in May and June (Fig. S2). In 2017, light and dark NH_4^+ uptake ranged from 0.011–1.10 μmol N L^{-1} h^{-1} and undetectable to 0.934 μmol N L^{-1} h^{-1}, respectively (Figs. 3b, S3). Rates at WE6 peaked in August, which were the highest 2017 rates except WE2 in June (Fig. S3).

Across the entire dataset, light and dark NH_4^+ uptake rates were correlated with many physicochemical parameters, including several nutrient forms (TN, particulate organic N), both biomass parameters, and microcystin concentrations (Table S2).

For the entire dataset, light exceeded dark NH_4^+ uptake at all stations ($F_{7,654} = 21.4$; $p < 0.001$), in June–October ($F_{13,648} = 9.61$; $p < 0.001$), and within each year ($F_{5, 656} = 40.6$; $p < 0.001$). There were instances (usually WE4 and WE13) where dark NH_4^+ uptake rates were comparable to or exceeded light rates, particularly in July (Figs. S1–S3).

NH_4^+ regeneration

Light and dark NH_4^+ regeneration rates were not different ($F_{1,660} = 0.637$; $p = 0.425$), so they were averaged together for each incubation. Across all sampling dates, NH_4^+ regeneration exhibited differences by station ($F_{3,327} = 11.8$; $p < 0.001$) and was greatest at WE6 and WE2, but rates at WE6 and WE2 were not different from each other (Fig. 4a). NH_4^+ regeneration rates were also different when evaluated by month ($F_{6,306} = 5.86$; $p < 0.001$), with highest regeneration rates in June and July (Fig. 5a). NH_4^+ regeneration rates also varied across years ($F_{2,328} = 38.5$; $p < 0.001$), with 2015 rates (even excluding August 2015) greater than 2016 or 2017, while the latter two years were not different from each other (Fig. 5b).

NH_4^+ regeneration rates peaked in August 2015 (13.0 μmol N L^{-1} h^{-1}; Fig. 4b), ranging from undetectable to 0.650 μmol N L^{-1} h^{-1} across the rest of 2015 (Figs. 5b, S4). NH_4^+ regeneration rates in 2016 ranged from undetectable to 0.282 μmol N L^{-1} h^{-1}, peaking at WE6 in July and WE2 in August, before declining in September (Figs. 5b, S5). In 2017, rates ranged from undetectable to 0.289 μmol N L^{-1} h^{-1} and remained low until mid-summer, reaching a maximum at WE6 in August (Fig. 5b, S6).

NH_4^+ regeneration rates were negatively correlated with Secchi depth and positively related to light and dark NH_4^+ uptake, water and air temperature, conductivity, turbidity, total suspended solids, phycocyanin, chlorophyll *a*, particulate microcystins, total P, particulate organic N, and TN (Table S2).

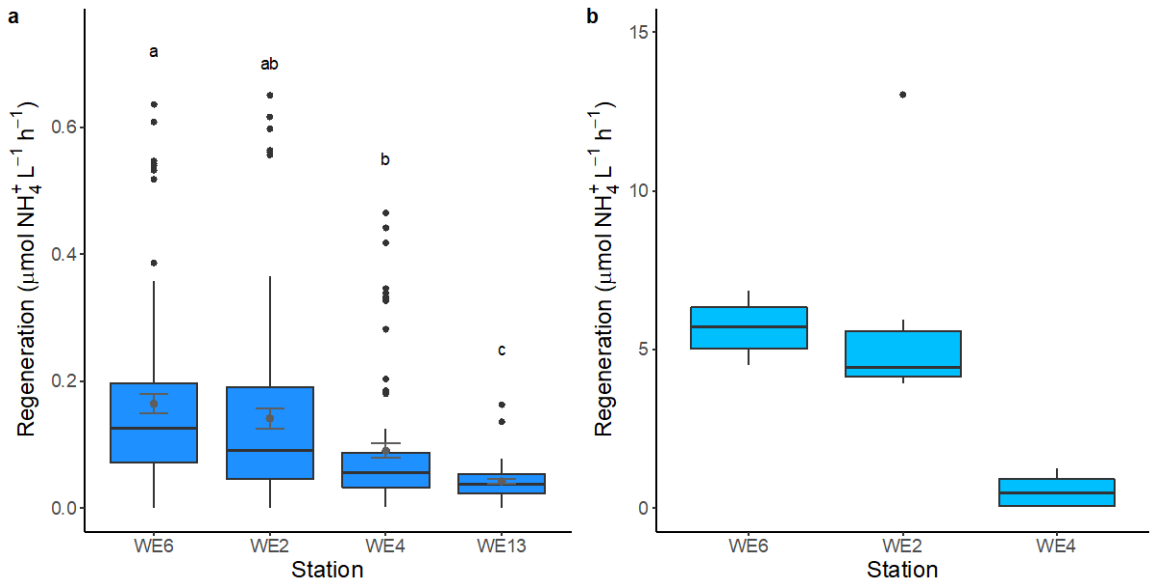

Figure 4. NH_4^+ regeneration rates (μmol L^{-1} h^{-1}) (a) across all months and years by sampling station; and (b) from August 14, 2015 (note difference in y-axis scale). Means (± SE) are overlaid on each boxplot in panel (a) in gray. Letters indicate differences in regeneration rates between sites (Tukey's HSD post-hoc tests). *n* for each station: WE6 = 90, WE2 = 95, WE4 = 87, WE13 = 60.

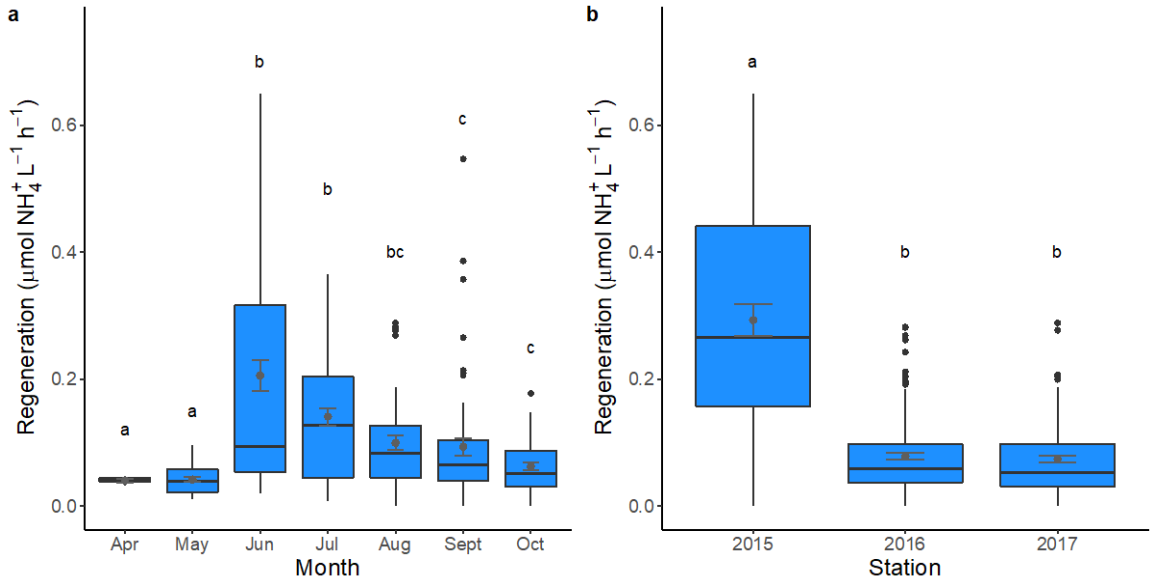

Figure 5. (a) NH_4^+ regeneration rates (μmol L^{-1} h^{-1}) by month across all sampling years (excluding August 2015). *n* for each month: April = 6, May = 36, June = 66, July = 57, August = 66, September = 53, October = 48. (b) NH_4^+ regeneration rates (μmol L^{-1} h^{-1}) by year (excluding August 2015). *n* for each year: 2015 = 71, 2016 = 117, 2017 = 144. Means (± SE) are overlaid on each boxplot in gray. Letters indicate differences in regeneration rates between (a) months or (b) years (Tukey's HSD post-hoc tests).

The capacity of water column NH_4^+ regeneration to support community NH_4^+ demand in western Lake Erie exhibited similar patterns in each year, peaking in June (82.4–124%) and decreasing through August (Fig. S7a), when TN loads from the Maumee River watershed decreased (Fig. S7b). Water column nutrient concentrations (Table 1) and external TN loads (Fig. S7b) were low during peak bloom months (August and September), while NH_4^+ regeneration continued at rates supporting ~20–60% of community NH_4^+ demand, providing an internal N load not accounted for in external loading or water column concentration measurements.

Discussion

This study (2015–2017) quantified NH_4^+ cycling rates in western Lake Erie along a spatial gradient from the primary nutrient source (Maumee River) into the main basin and assessed the role of internal NH_4^+ dynamics in supporting cyanoHABs. Our results explain the persistence of high biomass and toxin concentrations despite N depletion.

Water column NH_4^+ cycling

We did not observe depth-driven differences for any rate measurement. Western Lake Erie is shallow and well-mixed, and *Microcystis* can regulate buoyancy and migrate vertically in the water column (Wallace et al. 2000), perhaps explaining this observation. While extreme for Lake Erie, NH_4^+ regeneration and potential uptake rates measured in August 2015 were similar to maxima reported during a cyanoHAB in Lake Smith, Alaska (15.4 μmol L^{-1} h^{-1}; Gu and Alexander 1993). At WE6 in August 2015, extracted phycocyanin and chlorophyll *a* values from surface scums were 1,700 and 1,900 μg L^{-1}, respectively (Table 3), and NH_4^+ concentrations were near-undetectable (Table 1). NH_4^+ regeneration and potential uptake rates measured at a station <1 km from WE6 five days prior (Fig. S8) were nearly two orders of magnitude less than our highest values, but approximately half of potential NH_4^+ uptake rates at WE6 in August 2016 and 2017, reflecting spatial and temporal variability in bloom biomass (e.g., Chaffin et al. 2021).

Potential NH_4^+ uptake rates were greater in the light than dark, indicating expected photoautotrophic activity when light and NH_4^+ were available. Potential NH_4^+ uptake rates across both light and dark treatments were similar to, but sometimes exceeded, rates reported in other eutrophic systems, including Lake Champlain (Missisquoi Bay; McCarthy et al. 2013) and Lake Okeechobee (James et al. 2011), and were often much greater than those reported in Lake

Balaton (Présing et al. 2001), Lake Michigan (Gardner et al. 2004), Lake Superior (Kumar et al. 2008), and several orders of magnitude greater than those previously reported in Lake Erie (Murphy 1980). In addition to differences in methodology, this large discrepancy is almost certainly explained by greatly reduced phytoplankton biomass following aggressive nutrient load mitigation strategies in the years prior to the study (Watson et al. 2016). Greater light NH_4^+ uptake rate maxima were reported in hypereutrophic Lake Taihu (Hampel et al. 2018), as well as in *Planktothrix*-dominated Sandusky Bay in Lake Erie (Hampel et al. 2019a), Lake Okeechobee (Hampel et al. 2019b), and subarctic Smith Lake in Alaska (Gu and Alexander 1993).

Maximal NH_4^+ regeneration rates in western Lake Erie were similar to those reported for Lake Okeechobee (James et al. 2011) and Missisquoi Bay (Lake Champlain; McCarthy et al. 2013), as well as an Amazon floodplain lake (Morrissey and Fisher 1988). NH_4^+ regeneration rate maxima in western Lake Erie were two orders of magnitude greater than those from Lake Michigan (Gardner et al. 2004), up to an order of magnitude less than those in Sandusky Bay (Hampel et al. 2019a) and Lake Okeechobee (Hampel et al. 2019b), and 3–5 times less than those from Lake Taihu (McCarthy et al. 2007; Jiang et al. 2019) and Petit Saut Lake (French Guyana; Collos et al. 2001). In general, NH_4^+ regeneration and potential uptake rates in this study generally fell within the range of those reported for other eutrophic lakes and measured using similar methods.

The hypothesis that community NH_4^+ uptake and turnover rates would decrease with distance from the Maumee River inflow was supported by the results. WE4 and WE13 had lower rates than the westernmost stations, indicating less demand and recycling further from the nutrient source. We also predicted greater NH_4^+ uptake rates during peak bloom months, and although there was no difference between summer months (June–September) once August 2015

393 values were disregarded, positive relationships between light NH_4^+ uptake, temperature,
394 phycocyanin, and chlorophyll *a* support this prediction.
395 Contrary to our prediction, NH_4^+ regeneration could not support potential NH_4^+ demand
396 during peak bloom periods (Fig. S7). This result agrees with findings from Lake Champlain
397 (McCarthy et al. 2013) but contrasts with *Microcystis*-dominated Lake Taihu, where summer
398 NH_4^+ regeneration rates could support 100% of community NH_4^+ demand (Hampel et al. 2018).
399 Unlike Lake Erie, NH_4^+ is detectable in Taihu in August, and the external N load is much greater
400 (Hampel et al. 2018). High NH_4^+ regeneration rates we report here represent a large, continuous
401 NH_4^+ supply to support cyanoHAB biomass and toxin production, especially during peak
402 biomass months (August and September), when cyanoHABs in Lake Erie are N-stressed
403 (Chaffin et al. 2018). N stress also decreases intracellular concentrations of N-rich microcystins
404 (Van de Waal et al. 2014). Our data suggest that toxin production during this period required
405 regenerated NH_4^+. Sediment-water interface N effluxes also supply bioavailable N to supplement
406 water column NH_4^+ regeneration and external loading (Présing et al. 2008; McCarthy et al.
407 2016), as shown recently for the same study area of Lake Erie (Boedecker et al. 2020).
408 Throughout the study, the 2015 bloom was the most severe in terms of biomass and areal
409 coverage, followed by 2017 and 2016 (NOAA; Wynne et al. 2021; further description in
410 Supplemental Materials). Even excluding August 2015 rates, 2015 exhibited higher potential
411 NH_4^+ uptake rates than other years. N concentrations and cyanobacteria biomass were inversely
412 related (Table S2(3?)), but neither biomass proxy was related to o-PO_4^{3-} concentration. This
413 pattern reflects seasonal cyanoHAB maxima, where bioavailable N became depleted by non-N-
414 fixing *Microcystis* (Table 1). High TKN:TN ratios in the Maumee River load coincide with
415 bloom formation in mid-July, several months after spring P loads (Newell et al. 2019).

Microcystis-dominated blooms in the western basin originated in the mid-1990s (Steffen et al. 2014), coinciding with the increased proportion of TKN (Newell et al. 2019), while increased dissolved reactive P loads did not occur until after 2002 (Jarvie et al. 2017). Combined with reported over-estimation of Maumee River dissolved reactive P loads (River and Richardson 2019), the focus on dissolved reactive P loads as the main driver of cyanoHABs in Lake Erie (e.g., Annex 4 of the Great Lakes Water Quality Agreement, IJC) is insufficient.

Scaling up: NH_4^+ regeneration in western Lake Erie

CyanoHAB biomass and toxin production is sustained following N depletion by recycled NH_4^+ (McCarthy et al. 2007; Gardner et al. 2017; Hampel et al. 2018; this study). NH_4^+ regeneration rates were scaled by delineating the western basin into zones by station and calculating the volume of each station area (see Boedecker et al. 2020). NH_4^+ regenerated at each station was then compared to incoming TN loads from the Maumee River (NCWQR Heidelberg tributary monitoring program, https://ncwqr.org/monitoring/, last accessed 10/17/2020; Richards et al. 2010).

In 2015, NH_4^+ regeneration during the sampling season added NH_4^+ equivalent to 76% of the annual TN load and 160% of the TN load during the sampling period (June–September). Including August 2015, the internal NH_4^+ regeneration load increases to 2.2 times the annual TN load and 4.5 times the TN load during the sampling season. In 2016 and 2017, regenerated NH_4^+ contributed NH_4^+ equivalent to 60% of the annual TN load and 1.3–1.5 times the seasonal TN load (Table 4). When monthly NH_4^+ regeneration rates were summed to calendar year (using April 2016 rates as a proxy for winter months), NH_4^+ regeneration could supply 240–250% of the annual TN load in 2016 and 2017. This relative contribution (~2×) is similar to estimates for

Missisquoi Bay (McCarthy et al. 2013), Lake Taihu (Hampel et al. 2018), and Sandusky Bay (~77%; Hampel et al. 2019).

Sediments also contribute to water column N availability in aquatic ecosystems (An and Gardner 2002; McCarthy et al. 2016). Recent work at many of the same stations as the present study reported that western basin sediments release NH_4^+ and urea at rates equivalent to 4–11% of the annual TN load (Boedecker et al. 2020). Combined, water column and sediments (NH_4^+ release ~11% of water column NH_4^+ regeneration) represent a continuous supply of internal N when external N loads are minimal, cyanobacterial biomass is high, and water column N concentrations are depleted.

Table 4. Extrapolated NH_4^+ regeneration in the water column vs. annual (a) or sampling season (b) TN load (in metric tons). 2015 sampling season June–September, 2016 April–October, and 2017 May–October. Regenerated NH_4^+ values in 2015 include those generated from very high rates in August.

a	Regenerated NH_4^+	TN Annual Load	% of annual load
2015	8.15×10^4	3.79×10^4	215
2016	1.64×10^4	2.72×10^4	60.3
2017	2.40×10^4	3.87×10^4	62.0

b	Regenerated NH_4^+	TN Sampling Season Load	% of seasonal load
2015	8.15×10^4	1.80×10^4	452
2016	1.64×10^4	1.05×10^4	156
2017	2.40×10^4	1.85×10^4	130

Despite high spring nutrient loading from the Maumee River, cyanoHABs do not develop in western Lake Erie until mid- or late-July, corresponding with higher water temperatures and proportion of chemically reduced N forms in the external N load (Newell et al. 2019). The ratio

of microcystin to phycocyanin peaked when regenerated NH_4^+ was (up to 50 times) greater than the Maumee River TN load from the previous week($F_{2,46} = 2.70$; $p = 0.020$, Fig. S9). This observation suggests a 'sweet spot', where NH_4^+ regeneration supports biomass and toxin production, and biomass maintenance is favored over toxin production (Harke and Gobler 2013). When NH_4^+ regeneration exceeded weekly TN loading by > 50 times (late August and September), toxin concentrations in biomass decreased and were not different from those when external loads were higher than internal loading (Fig. S9).

Although microcystins contain, on average, 10 N per molecule (Honkanen et al. 1990), there were no strong relationships between any N species and toxin concentrations in the present study. Given the high rates of NH_4^+ regeneration reported here, snapshot sampling to determine DIN concentrations do not accurately represent N availability. However, NH_4^+ regeneration and potential uptake rates are good proxies for N demand and bioavailability (Gardner et al. 2017). Thus, water column NH_4^+ cycling rates, previously missing from the literature for this and most other freshwater systems, may represent key parameters to consider when developing models to predict toxin concentrations in cyanoHAB impacted systems.

Conclusions

This study contributes to the growing body of literature highlighting the importance of NH_4^+ and chemically reduced N in eutrophic lakes affected by seasonal cyanoHABs, especially those comprised of non-N-fixing, toxin-producing taxa (e.g., *Microcystis*). Water column NH_4^+ regeneration rates met or exceeded external N loads in the western basin and help explain why non-N-fixing cyanoHABs proliferate despite minimal external N loading. These results reinforce the need to manage external N loading to mitigate growth and toxin production of non-N-fixing cyanobacteria (e.g., McCarthy et al. 2013, 2016; Gobler et al. 2016; Newell et al. 2019). External

nutrient loads stimulate biomass and toxin production, which becomes the substrate for NH_4^+ regeneration when cells are grazed and/or remineralized in the water column (Gardner and Lee 1975; McCarthy et al. 2013) and sediments (McCarthy et al. 2016; Boedecker et al. 2020). Thus, while in-system turnover should be accounted for in management and modeling applications, reducing external nutrient loads from watersheds remains the key to mitigating the global proliferation of contemporary cyanoHABs that cannot fix atmospheric N and produce N-rich toxins.

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