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Hydrophysical and Hydrochemical Controls of Cyanobacterial Blooms in Coursey Pond, Delaware (USA)

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Abstract

Noxious cyanobacterial blooms are common in many ponds in the mid-Atlantic Coastal Plain. In Delaware, blooms normally occur between July and October, yet no in-depth analyses of the causes and predictors exist. A study using commercially available, high-frequency, continuous, and automated biogeochemical sensors at Coursey Pond, Delaware, a pond known for perennial summer blooms, was conducted to investigate how hydrophysical and hydrochemical conditions affect bloom dynamics. Cyanobacterial abundance (based on the in vivo phycocyanin fluorescence and phycocyanin/chlorophyll fluorescence ratios) increases during periods of high water temperatures (up to 32°C), low discharge through the pond (mean hydraulic residence time $\geq$5 d) with evaporative concentration of dissolved solids, and decreasing NO_3^- concentrations (reaching <0.1 mg L^{-1}, the detection limit). These conditions lead to the uptake and depletion of bioavailable N in the pond surface waters and provide a competitive advantage for temperature-tolerant and N-fixing cyanobacteria. Irrigation water use within the watershed can exceed pond discharge during drier summer months, enhancing bloom-forming conditions. Bloom intensity varies due to storms but persists until mid-October to mid-November when temperatures cool, daylength decreases, and discharge and NO_3^- concentration recovers. Changes in these easily monitored physical and chemical parameters can serve to anticipate the onset, intensity, persistence, and the eventual dissipation of cyanobacterial blooms at Coursey Pond and similar ponds elsewhere. Therefore, the use of sensors and high-frequency data has the potential to assist in forecasting and management of blooms.

Core Ideas

- Cyanobacteria blooms are common in small flow-through ponds in the mid-Atlantic Coastal Plain.
- Available biogeochemical sensors can characterize bloom onset, intensity, and dissipation.
- Residence time, temperature, and NO_3^- concentration are robust predictors of bloom intensity.
- Storms temporarily disrupt blooms by flushing, but blooms reestablish once stormflow ceases.
- Irrigation within the watershed increases pond residence time, thus promoting blooms.

Noxious and potentially toxic cyanobacterial blooms in fresh, estuarine, and marine waters have increased significantly in recent years and now occur in a wide range of settings worldwide (Paerl and Otten, 2013; Kudela et al., 2015). Bloom species are able to exploit the physical and biogeochemical conditions in eutrophic ponds and lakes during the summer due to their ability to tolerate higher temperatures and low dissolved nutrient availability, and their ability to take advantage of a wider array of nutrient sources (Paerl and Huisman, 2008; Paerl and Otten, 2013).

There are concerns about health impacts of cyanobacterial blooms as a number of bloom-forming species and strains can produce toxic compounds (Stewart and Shaw, 2008). However, limited monitoring resources and no reports of cyanobacterial-related illnesses have limited the ability of environmental agencies and researchers to directly monitor cyanobacteria populations and their toxins. Alternative monitoring strategies are needed to provide early warnings of bloom onset that would allow agencies to focus limited resources on monitoring of blooms and bloom toxicity once the blooms occur.

Cyanobacteria blooms can also discolor surface waters and occasionally form unsightly and malodorous floating "scums" that limit light penetration in ponds, discourage recreational use, promote development of anoxia in bottom waters below the blooms, and affect the abundance and survival of other pond populations (Anderson et al., 2002; Heisler et al., 2008).

Fourteen of Delaware's 43 state-managed ponds have historically suffered from seasonal or episodic algal blooms that include noxious and potentially toxic cyanobacterial species, such as *Microcystis* spp. and *Dolichospermum* spp. (formerly planktonic *Anabaena* spp.; Humphries et al., 2008; Wacklin et al., 2009). Enzyme-linked immunosorbent assay testing found measureable but low concentrations of the toxin microcystin in a few Delaware ponds in the past, but surveys of pond users found no evidence of direct toxic impacts on people, pets, or wildlife (Humphries et al., 2008). Since 2008, additional ponds have been impacted by cyanobacteria, and the persistence of blooms in some ponds has increased (Robin M. Tyler, personal communication, 2015).

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Abbreviations: Chl, chlorophyll a; DO_{sat}, dissolved oxygen saturation; MHRT, mean hydraulic residence time; Pc, phycocyanin; SC, specific conductivity.

In nearby New Jersey, there are 1775 similar Coastal Plain ponds (NJ Department of Environmental Protection Water Assessment Team, 2004), with many of these ponds experiencing water quality impairments similar to those in Delaware (NJ Department of Environmental Protection, 2015) and some having documented cyanobacterial outbreaks (http://www.state.nj.us/dep/wms/bfbm/cyanoHABevents.html). In fact, most US states have reported one or more cyanobacterial bloom since 1983 (Lopez et al., 2008), often in smaller and less well-monitored ponds and lakes.

Although it is difficult to eliminate or control blooms, it is increasingly possible to anticipate the occurrence of blooms using remote sensing (Downing et al., 2001; Vincent et al., 2004), in situ sensors designed for continuous monitoring, and/or appropriate modeling to mitigate the ecological and recreational impacts of individual bloom events (Baben et al., 2008; Francy et al., 2015). Operational bloom forecasts for the Gulf of Mexico and Lake Erie are already available from the NOAA (https://tidesandcurrents.noaa.gov/hab_info.html) where blooms damage fishing, tourism, and water supplies. Efforts to extend similar forecasting capabilities to additional ecosystems that are similarly susceptible to cyanobacterial blooms and potential bloom toxicity (e.g., https://www.epa.gov/water-research/cyanobacteria-assessment-network-cyan) are not yet completed, and the spatial resolution of satellite or aircraft-mounted sensors is too coarse to effectively monitor pigment concentrations as a proxy for toxicity in smaller ecosystems.

In this study, we used data from a continuous, automated, and high-frequency biogeochemical sensor package at the outlet of Coursey Pond, Delaware (Fig. 1), to develop a better understanding of the hydrophysical and hydrochemical factors that may be controlling the onset, intensity, and dissipation of cyanobacterial blooms. If hydrophysical and hydrochemical forcing are coupled to biological responses, it may be possible to anticipate the onset and dissipation of algal blooms at Coursey Pond and/or elsewhere based on a few easily measured or estimated hydrophysical (stage height, discharge, irrigation rates, and/or water temperature) and hydrochemical (NO_3^- concentrations) parameters. We also tested whether the direct measurement of in vivo phycocyanin (Pc) and chlorophyll a (Chl) concentrations, and in vivo Pc/Chl ratios can serve as indicators of evolving bloom intensity and upcoming needs for active monitoring, management, and possibly mitigation.

Environmental Setting

The Murderkill River drains a small Coastal Plain watershed (26,936 ha) in Kent County, Delaware, to Delaware Bay. Coursey Pond (Fig. 1a) is a small (243 ha), shallow (mean depth = 0.9 m), historical millpond, similar to many others in the mid-Atlantic region, and the last nontidal pond on the main stem of the Murderkill River. This hydrologic setting has some similarities to lowland tropical impounded rivers (Bormans et al., 2004) but is quite different from those of large lakes (Chafin et al., 2014; Mantzouki et al., 2016; McCarthy et al., 2016) and coastal oceans (Baben et al., 2008) that also have documented problems with cyanobacteria blooms. Coursey Pond is fed by flow from the mainstem of the Murderkill River, groundwater discharge, surface discharge from one small tributary, and rainfall (Fig. 1b). Just downstream of Coursey Pond dam, the Murderkill River is tidal, but usually fresh. Coursey Pond is at the downstream end of a predominantly agricultural (58%) and forested (29%) watershed of 6125 ha, with only a small (11%) portion of urban or suburban land cover (Delaware FirstMap, 2016). Corn (*Zea mays* L.) and soybeans [*Glycine max* (L.) Merr.] dominate agricultural production in the heavily irrigated watershed (DDA-NASS, 2016).

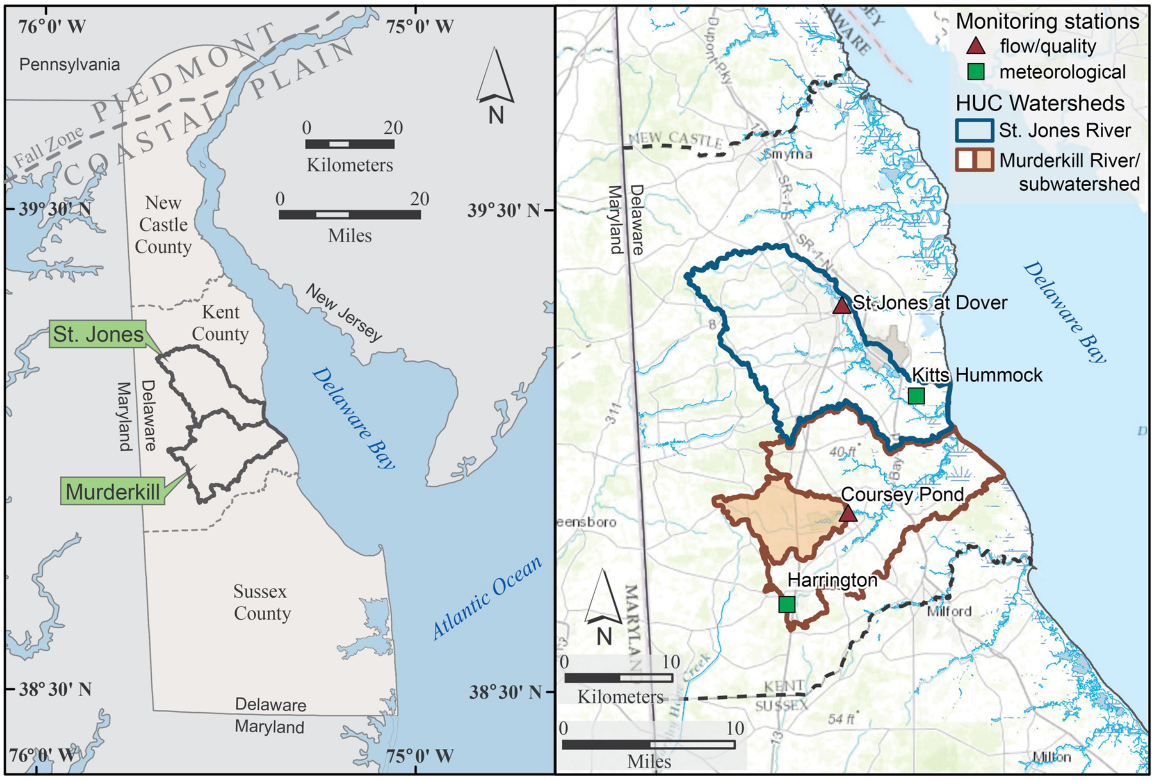

Fig. 1. Location map showing watersheds of the Murderkill and St. Jones Rivers, Coursey Pond, and Delaware Bay. HUC, hydrologic unit code.

Groundwater discharge from the shallow Columbia aquifer dominates the hydrology of Coastal Plain watersheds in Delaware, with baseflow making up >70% of total discharge (Johnston, 1976). As such, groundwater quality and local, primarily agricultural land-use practices heavily influence the biogeochemistry of Coastal Plain streams and ponds. Given the quartzose sand composition of the Columbia aquifer, the relatively shallow depth to groundwater (Martin and Andres, 2008), and the dominance of agricultural land uses, the limited groundwater quality data in the watershed (https://waterdata.usgs.gov/de/nwis/qw) show the expected elevated concentrations of NO_3^- and total dissolved solids but very little NH_4^+ (Debrewer et al., 2005; Ator, 2008).

Results of decades of monthly sampling done between 1998 and 2017 by the Delaware Department of Natural Resources and Environmental Control at Coursey Pond (Water Quality Portal Station ID: 21DELAWQ_WQX-206451) show the pond is enriched in N and dissolved P and highly eutrophic. Average Chl concentrations collected during June through September range between 50 and 135 μg L^{-1}. Total monthly mean dissolved inorganic N concentrations are greatest between December and April, with an annual range between 2 and 4 mg L^{-1}. Pond N speciation between October and June reflects the groundwater input signal with NO_3^-–N 40 times more than that of NH_4^+–N. During July through September NO_3^-–N to NH_4^+–N ratio ranges between 7 and 20 and, together with the lesser NH_4^+–N and NO_3^-–N concentrations, indicates complex N dynamics in the pond. To our knowledge, there are no measurements of denitrification or sediment N-regeneration rates at the pond. Monthly average dissolved P concentrations during June through September range from 0.014 to 0.022 mg L^{-1}, concentrations that are not limiting to phytoplankton growth. Water Quality Portal data indicate that nearly all turbidity, and thus particulate P, in the pond in summer months is due to algal biomass, likely because low flow velocities and long mean hydraulic residence times (MHRTs) are sufficient to induce sedimentation of mineral particles.

Materials and Methods

A YSI EXO2 multiparameter sonde was deployed at a fixed vertical position (between 0.25- and 0.5-m depth, depending on stage height) adjacent to the outflow weir of Coursey Pond for the determination of conductivity, temperature, pH, dissolved O_2 (optical sensor), and nephelometric turbidity. In vivo fluorescence, determined by a dual-wavelength EXO2 sensor, was used as a relative indicator of Chl and Pc concentrations and bloom intensity. Due to turbidity, temperature, and scattering interferences associated with these in vivo fluorescence measurements, the ratio of in vivo Pc to in vivo Chl (Pc/Chl) was used to characterize the relative abundance of cyanobacteria in this study. Other investigators have studied and discussed the relationships between in vivo fluorescence, the actual concentrations of Chl and Pc, the structure of the phytoplankton community, and the difficulties of using these measurements as quantitative indicators of plankton abundance (Chang et al., 2012; Zamyadi et al. 2012; Bowling et al., 2016; Hodges et al., 2017).

The YSI sensors were calibrated in the laboratory before, during, and after deployments (at intervals of 2–6 wk) using standards and procedures recommended by the manufacturer. We used manufacturer-supplied software to convert fluorescence to pigment concentration units (μg L^{-1}).

An s::can spectro::lyser (s::can Messtechnik) with a 5-mm optical path and the s::can "global river" algorithm provided by the manufacturer were used for the automated, in situ determination of NO_3^- concentrations. The instrument sensor was deployed at a fixed vertical position that varied between 0.4 and 0.7 m below the water surface, depending on the stage height. Occasional grab samples were field filtered, placed on ice, returned to the laboratory, and stored at 4°C until analysis of NO_3^- by an automated version of the Cd reduction–diazo-blue method (Grasshoff et al., 1999), for comparison with the automated field measurements of NO_3^-. Due to long-term drift, fouling, and the limitations of the manufacturer's "global river" calibration, NO_3^- concentrations close to the instrument detection level (0.1 mg L^{-1}) are suspect, but higher concentrations and variation in concentrations with time are reliable and consistent with other onsite and laboratory measurements.

Comparison of 24 pairs of NO_3^- laboratory analyses of samples with paired instrumental in situ measurements shows an r^2 of 0.86 and a slope of 1.03, demonstrating that the "global river" algorithm produces reasonable and unbiased estimates at the Coursey Pond site. We recognize that partial least squares regression of spectral and laboratory data typically improves the correlation between instrument- and laboratory-measured NO_3^- (Etheridge et al., 2014; Vaughan et al., 2017). However, as our focus in this study was on changes in NO_3^- concentrations, such an analysis was not required.

Depth profiles of dissolved O_2, temperature, conductivity, and pH were measured with a YSI Model 556 in the months of June through September on multiple dates during the study. Unfiltered samples were obtained within 1 m of the sensor station with a pole-mounted, submersible pump at depths ranging from the surface to 15 cm above the pond bottom at 15-cm intervals and measured in an airtight, flow-through cell. Profile data were also collected by mounting the YSI Model 556 sonde and cable to the pole and measuring from the surface to 15 cm above the pond bottom at 15-cm intervals. No differences were found between the two methods.

Stage height was determined using an In-Situ LevelTroll 500 data logger at the pond discharge weir. Pond discharge was determined using USGS methods (Turnipseed and Sauer, 2010) from the relationship between stage height, measured at the pond weir, and discharge determined 80 m downstream of the pond using a Teledyne Stream-Pro Acoustic Doppler Current Profiler and WinRiver II Software (Teledyne RD Instruments, 2018). The period of flow observation at Coursey Pond began in June 2014. Unit discharge data from Coursey Pond were similar to those from nearby long-term USGS Station 01483700 (St. Jones River; 39.09494° N, 75.31087° W) in the immediately adjacent watershed (Fig. 1). Mean hydraulic residence time was calculated using the pond volume (186,857 m^3, determined by acoustic survey and assumed to be constant over the period of this study) divided by discharge (determined from stage height and the site rating curve.) Discharge statistics were evaluated using the

WHAT software package (Purdue Engineering, https://engineering.purdue.edu/mapserve/WHAT/).

Groundwater levels and temperatures were measured by the Delaware Geological Survey in a nearby well (Delaware Geological Survey identifier Ld15-21; 38.99845° N, 75.51853° W) by manual tape (Solinst) roughly every 3 mo (from September 2000 until August 2014) and by automated logging (every 15 min), using a pressure and temperature logger (In-Situ LevelTroll 300) since August 2014. This well is screened across the water table and the level–temperature sensors are located ~1 m below the water table. The processed water level data (barometrically compensated and corrected for drift) and additional information about groundwater level measurement methods and data protocols are available from http://www.dgs.udel.edu/water-resources.

The Department of Natural Resources and Environmental Control Water Supply Section (Dover, DE; Anne Mundel, personal communication, 2017) provided data on irrigation water use and irrigation systems in the Coursey Pond watershed. The available data are monthly total irrigation amounts estimated and reported by owners of individual wells. Unfortunately, total irrigation water use is poorly constrained, as most individual irrigators do not report use and rates. Those that do report irrigation do so semiquantitatively according to the elapsed time of irrigation and pumping rates estimated at the time of well construction. Locations and areas served by irrigation systems were extracted from Pope et al. (2013). Irrigated areas and annual unit irrigation demands from Levin and Zarriello (2013) are used to estimate watershed irrigation rates for low-demand and high-demand years.

Whole water samples were collected monthly during spring and summer 2015 and 2016, as described above, and at the same depth as the EXO2 sensors for phytoplankton and cyanobacterial characterization (only during spring and summer 2015) and were transported back to the laboratory on ice. Samples were preserved with a modified formaldehyde solution (40% formaldehyde, 20% liquid detergent, and 1.3 M $CuSO_4$; ASTM International, 2012) before microscopic examination. An EVOS FL Auto microscope (Thermo Fisher) was used to identify the phytoplankton community to the genus level (Prescott, 1978) and estimate relative abundances of the principal phytoplankton populations. No direct cell counts were made during this study, and, therefore, phytoplankton abundance is qualitative and not quantitative.

Results and Discussion

Hydrophysics

The average annual discharge from Coursey Pond in 2015 and 2016 was ~0.82 $m^3\ s^{-1}$, but daily discharge ranged from a minimum of 0.059 $m^3\ s^{-1}$ to a maximum of 24.9 $m^3\ s^{-1}$ (Table 1). Storm hydrographs are relatively flashy, typically returning to baseflow conditions within 3 d (Fig. 2 and 3). The largest source of flow into Coursey Pond is from the upstream Murderkill River. In comparison, the one small tributary, direct groundwater discharge, and direct precipitation are small and represent <10% of the water budget of Coursey Pond. Peak daily discharge rates are associated with recent rainfall, particularly during the winter months. Minimum flows occur in the late summer.

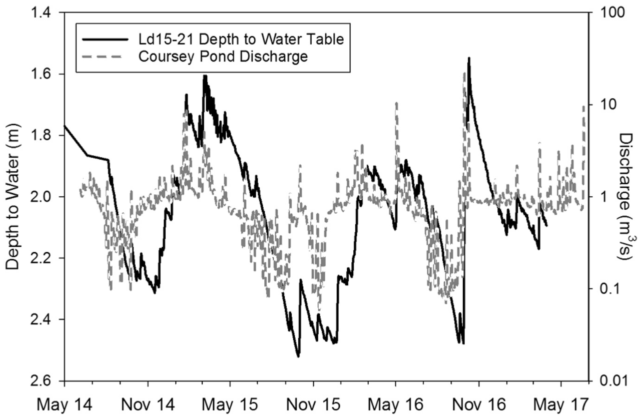

Fig. 2. Groundwater levels and discharge from Coursey Pond. Discharge is more variable than the groundwater levels due to rainfall and runoff.

In the Murderkill River watershed, the water table is present in the surficial Columbia aquifer. The water table at the nearby Ld15-21 monitoring well shows strong seasonal patterns with maximum water elevations in the winter, minima in the summer, and annual ranges near 0.9 m (Fig. 2). This seasonal pattern is consistent with agricultural irrigation and net evapotranspiration during the summer and net recharge during the winter. During our study, daily mean groundwater temperatures in this well averaged 14.4°C and ranged between about 9.4 and 18.7°C, with annual minima and maxima lagging surface water temperature by 2 to 3 mo.

Groundwater serves irrigation demands in the Coursey Pond watershed (Delaware Department of Natural Resources and Environmental Control, unpublished data, 2017). Of the 65 irrigation systems in the watershed (based on aerial photography; Pope et al., 2013), only 15 reported water-use data (Delaware Department of Natural Resources and Environmental Control, unpublished data, 2017), and thus a significant fraction of the total irrigation in the watershed goes unreported. Reported irrigation rates for the period of 2005 to 2016 indicate that irrigation in the watershed occurs primarily between May and October, with maximum rates occurring during June, July, and August. The maximum reported irrigation rates (0.0093 $m^3\ s^{-1}$) for the Coursey Pond watershed are about one order of magnitude smaller than summer baseflow discharge (lowest quartile) from Coursey Pond. Estimates of irrigation water use based on the irrigated area in the watershed (1150 ha) identified by Pope et al. (2013) are much larger and similar in magnitude to the lowest quartile of stream discharge rates (Table 1), ranging from 0.14 $m^3\ s^{-1}$ for low-demand years to 0.25 $m^3\ s^{-1}$ for high-demand years. Thus, during high-demand years when streamflows would also be reduced by dry weather, irrigation may increase MHRT in Coursey Pond by a day to weeks.

Table 1. Daily discharge statistics for the Murderkill River at Coursey Pond and the equivalent mean hydraulic residence times (MHRTs) for Coursey Pond.

Parameter	Min.	5th	25th	Median	Mean	75th	95th	Max.
Discharge ($m^3\ s^{-1}$)	0.059	0.103	0.577	0.928	0.816	1.01	1.85	24.9
Equivalent MHRT (d)	36.7 (max.)	8.81	3.74	2.65	2.33	2.15	1.07	0.087 (min.)

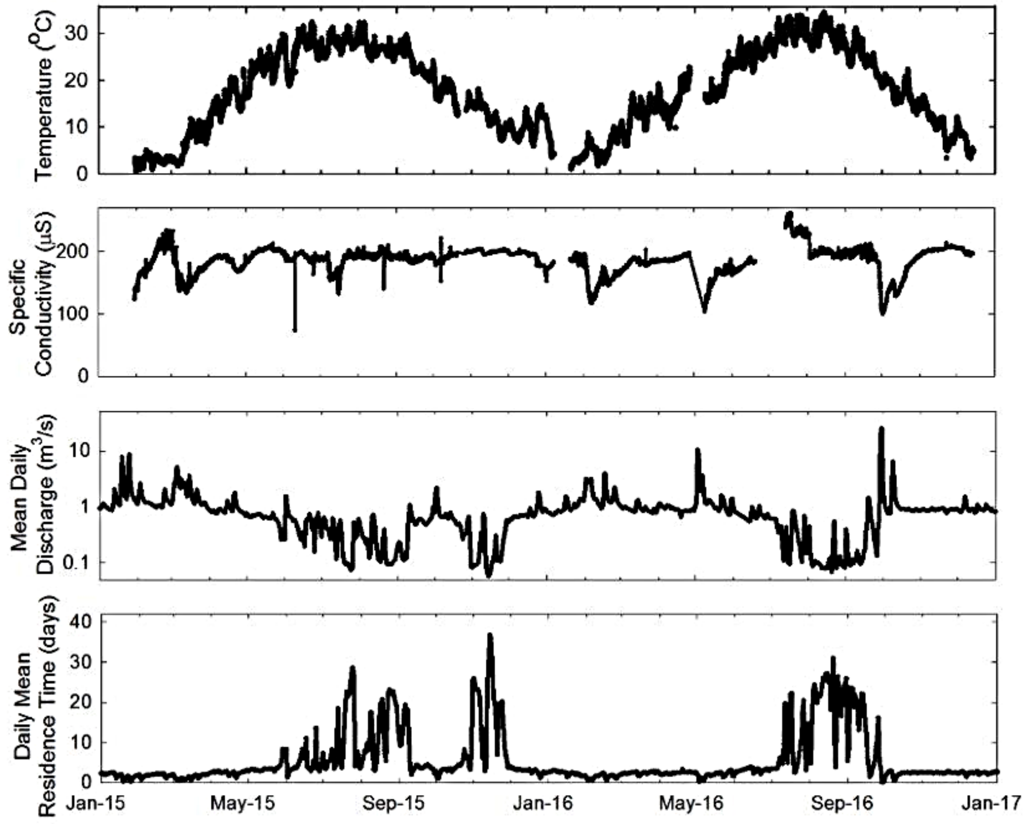

Fig. 3. Surface water temperature, specific conductivity, mean daily discharge, and daily mean hydraulic residence time.

Water temperatures in Coursey Pond range from lows near 2°C (late January to early February) to highs near 35°C (late August and early September). Groundwater temperature variations are significantly muted in comparison with those for surface water. At the monitoring site, there is no evidence of expected stratification of temperature, density, or dissolved O_2 with depth; this may be due to very shallow pond depths, effects of persistent, wind-driven mixing, and rapid flushing in response to stormflow events.

Ecology

During March, April, and May when pond discharge is high and temperatures are moderate, the pond supports a productive macrophyte population dominated by spatterdock (*Nuphar* spp.) and water lily (*Nymphaea* spp.) (http://www.dnrec.delaware.gov/fw/fisheries/pages/delawarepondbooklet.aspx). Our observations during April and May found a phytoplankton population dominated by centric and pennate diatoms and green algae with background levels of dinoflagellates and cyanobacteria. By late June, the macrophyte community is limited to the shallow waters along the shoreline, perhaps due to increasing water temperature, turbidity, or competition for light with the phytoplankton community. Our observations found that by late June to early July, the phytoplankton community is increasingly dominated by cyanobacteria, first *Microcystis* spp., and by mid- to late July a mixture of *Microcytis* spp. and *Dolichospermum* spp. with some remaining diatoms. After mid- to late July, the filamentous cyanobacteria *Dolichospermum* spp. and *Oscillatoria* spp. dominate the phytoplankton community with some remaining *Microcytis* spp. and diatoms.

Hydrochemistry and Biogeochemistry

Seasonal variations in discharge, water temperature, specific conductivity (SC), NO_3^- and pigment concentrations, and diel pH and dissolved O_2 saturation (DO_{sat}) fluctuations within the pond reflect the balance between allochthonous groundwater and surface water sources and autochthonous biological uptake and regeneration (Fig. 4). Daily DO_{sat} maxima frequently exceeded 200% on cloud-free days during June through September, but DO_{sat} minima rarely fall below 40%. During the study period, mean SC in Coursey Pond was 185 μS and varied between 100 and 235 μS (Fig. 3), values that are within the previously reported range for Coursey Pond (Water Quality Portal Station ID: 21DELAWQ_WQX-206451) and regional groundwater (Debrewer et al., 2005; Ator, 2008). Variations in SC clearly reflect the importance of hydrologic forcing of gross pond chemistry. Consistent with previous studies in other small watersheds in Delaware (Ullman et al., 2002; Volk et al., 2006; Andres et al., 2007), the lowest SCs occurred during and immediately after rainfall events and reflected the dilution of water in the pond by direct precipitation, quickflow groundwater, and overland runoff. The highest observed SCs in Coursey Pond during this study occurred in summer 2016 during a 12-wk period of below-normal precipitation and almost certainly represent evaporative concentration of dissolved salts in the pond.

The highest NO_3^- concentrations occurred in the winter and early spring (Fig. 4) when groundwater discharge rates were greatest and biological uptake rates were at their minimum. The lowest concentrations (less than the 0.1-mg L^{-1} detection limit) occurred in the late summer and early fall when groundwater discharge rates were at their minimum and biological uptake rates were at their maximum.

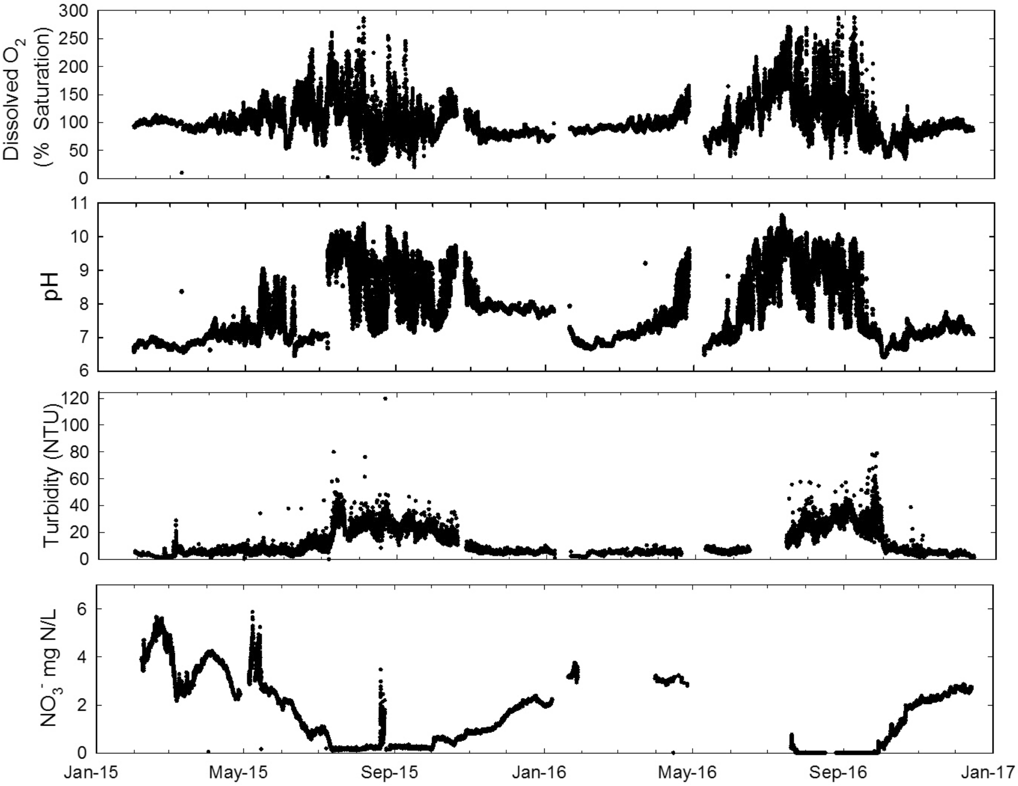

Fig. 4. Dissolved O_2 saturation percentage, pH, and turbidity and NO_3^- concentrations. Turbidity is measured in nephelometric turbidity units (NTUs).

Summertime blooms developed relatively rapidly and were characterized by substantial and reproducible changes in the hydrophysical and hydrochemical characteristics of the pond. For example, in both July 2015 and July 2016, there were periods of ~1 wk during which there were systematic decreases in discharge and NO_3^- concentrations coincident with increases in SC, MHRT, in vivo Pc, and Pc/Chl and increasing amplitudes of diel PC, DO_{sat}, and pH (Fig. 5–7). Linear regression analysis of 2016 data illustrates the trends present in both years. In this discussion, correlations are considered significant at the $p = 0.05$ level or better. Significant negative correlations between instantaneous (e.g., individual sensor measurements) NO_3^- and time ($r^2 = 0.76$) and instantaneous discharge and time ($r^2 = 0.91$), concurrent with significant positive correlations between instantaneous Pc ($r^2 = 0.46$), Pc/Chl ($r^2 = 0.56$), and time, show increasing cyanobacterial abundance while NO_3^- and flow are decreasing.

The impact of decreasing flow through the pond on hydrophysics and biogeochemistry is striking. There was a significant negative correlation between discharge and temperature ($r^2 = 0.51$) and a positive correlation between discharge and NO_3^- ($r^2 = 0.7$). Diel fluctuations of temperature caused a poorer fitting but significant positive correlation between flow and instantaneous temperature ($r^2 = 0.36$). However, there was a greater fit between daily means of these variables ($r^2 = 0.76$).

The relationships between hydrophysical and biological variables were strongly influenced by diel variability. Diel fluctuations in Pc and Chl concentrations, Pc/Chl ratio, and temperature were out of phase between 6 and 9 h and, as a result, there were weak to insignificant correlations between both the instantaneous and daily mean values of these variables. Diel fluctuations in Pc and Chl concentrations also reduced the fit of correlations of both instantaneous and daily mean values with flow.

Interestingly, daily means of flow and Pc/Chl ratio were positively correlated ($r^2 = 0.99$). This trend developed during a period of bloom intensification, indicating that small increases in flow during otherwise low-flow conditions favor increased production of Pc. The Pc/Chl ratio may be a better indicator of additional ecological processes, such as competition for light or nutrients between cyanobacterial taxa, competition between cyanobacteria and other planktonic primary producers, or predation by viruses and zooplankton (Simis et al., 2005). The physiological status of a cyanobacterial bloom may also independently alter the intracellular concentrations of these photosynthetic pigments in response to environmental conditions (MacIntyre et al., 2002; Descy et al., 2009). Further simultaneous monitoring of cyanobacterial pigments, community ecology, and hydrophysical, hydrochemical, and ecological forcing on a variety of timescales is needed to test these hypotheses.

To summarize, decreased flow (increased MHRT) leads to warmer water, greater abundance of cyanobacteria, greater productivity, and uptake of dissolved N from the water column. Relative abundance data indicate that the population changes from one that included *Microcystis* spp., a genus that does not fix N, to a population that includes N-fixing cyanobacteria, principally *Dolichospermum* spp. and *Oscillatoria* spp.

Once cyanobacteria blooms are established in the pond, however, daily variations in DO_{sat}, NO_3^- concentrations, and pH are related to the changing daily balance between photosynthesis and respiration, in response to daylight, temperatures, and the stoichiometries of uptake and regeneration. The large

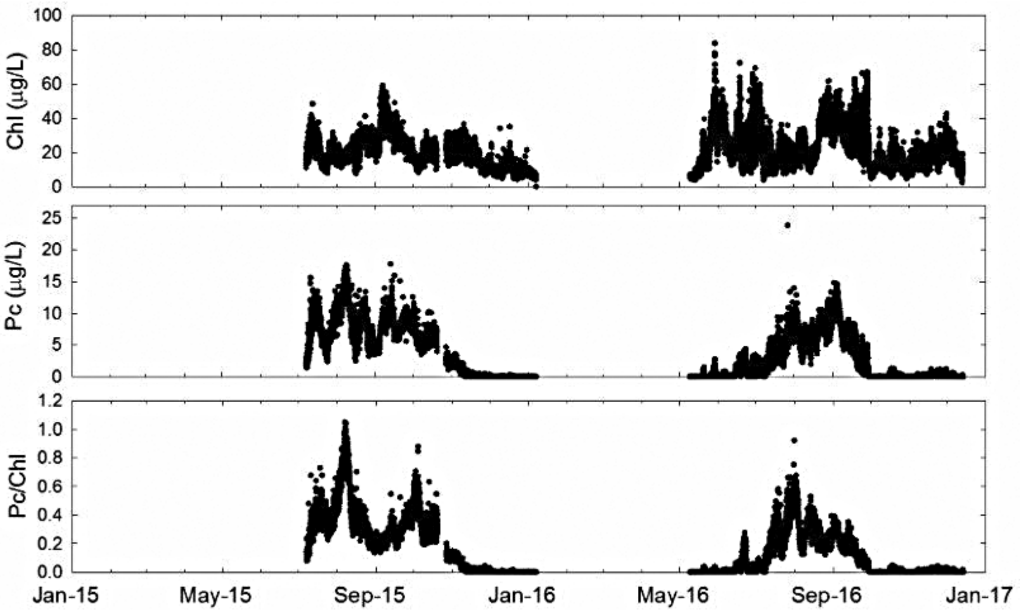

Fig. 5. Relative in vivo fluorescence of chlorophyll a (Chl) and phycocyanin (Pc), ratio of in vivo Chla fluorescence to in vivo Pc fluorescence, and residence time.

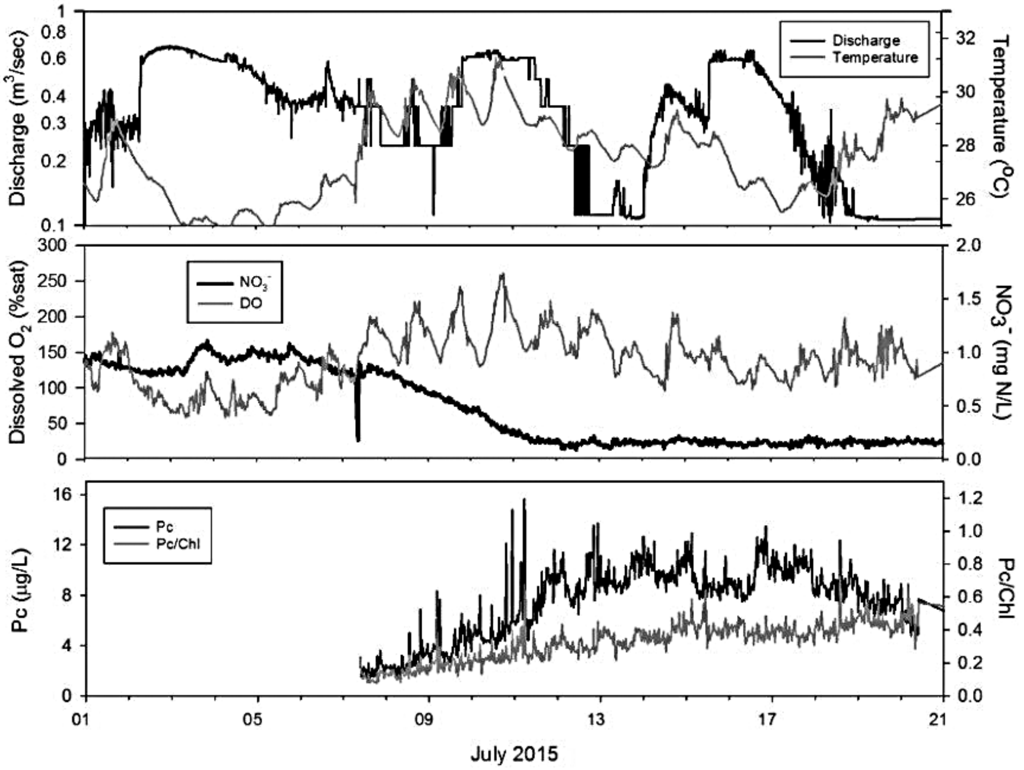

Fig. 6. Development of the cyanobacterial bloom in summer 2015 and its relationship to decrease in NO_3^- availability, discharge, temperature, dissolved O_2, NO_3^-, phycocyanin (Pc), and Pc to chlorophyll a (Pc/Chl) ratio.

daily variation in these parameters during summers is primarily an indicator of overall pond eutrophication and productivity. In the winter, the daily variations are still observed, but magnitudes are much reduced, due to lower temperature, shorter daylength, and the resulting lower rates of primary production and respiration.

Decreased MHRT (increased discharge) after summertime storms appears to temporarily disrupt cyanobacteria blooms, as evidenced by increases in NO_3^- concentrations, muted diel ranges of pH and DO_{sat}, and decreases in vivo fluorescence (Fig. 6 and 7). In July 2016, after establishment of a bloom, there was a major rainfall event (25.6 mm [1.02 in.]; Delaware Environmental Observing System, http://www.deos.udel.edu) that resulted in a short-term disruption of the cyanobacterial bloom followed by a recovery (Fig. 7).

Cyanobacteria blooms dissipate in the early fall (Fig. 8) in response to decreasing temperatures and insolation, and competition with other planktonic species that are better adapted to the cooler temperatures and are also able to take advantage of increased inputs of NO_3^- from watershed sources and, possibly, regeneration within the pond (Paerl and Otten, 2013). Both Pc and the Pc/Chl ratios decrease from typical bloom levels to trace levels over a few weeks. As the bloom dissipates, the diel variation in DO_{sat} drops significantly, indicating decreasing rates of primary production and respiration. Nitrate concentrations in the pond that were negligible during the bloom rise to levels more similar to the input waters typically observed during January through March. Evidence that storms may contribute to the final dissipation of the bloom was seen in 2016 (29–30 October, Fig. 8).

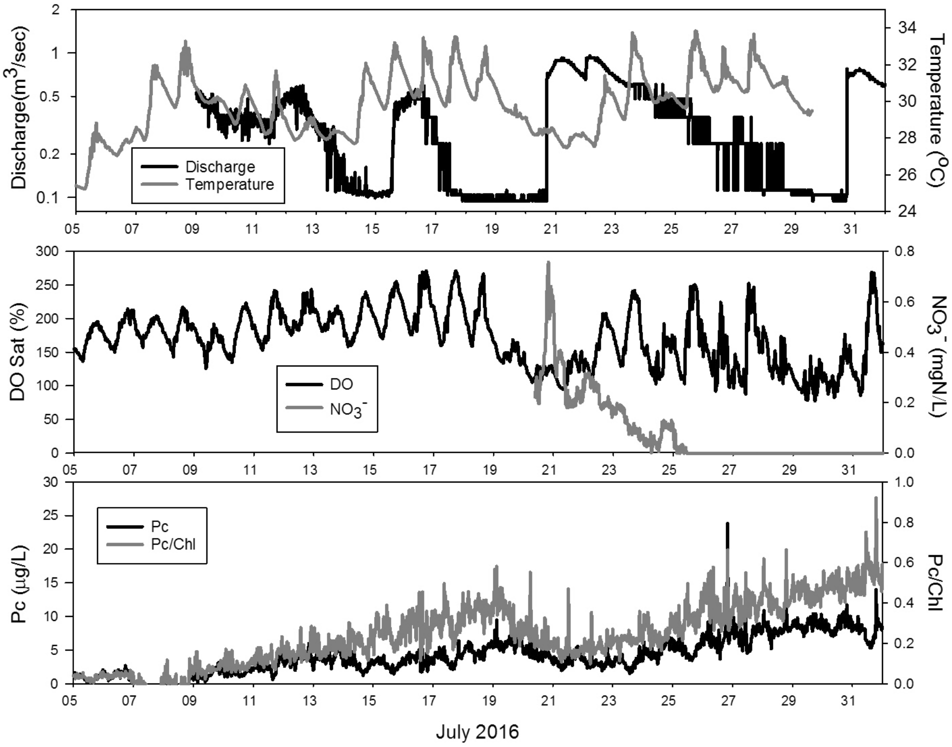

Fig. 7. Hydrophysical and biogeochemical conditions during development and temporary disruption of a bloom in summer 2016. Pc, phycocyanin; Chl, chlorophyll a; DO, dissolved O_2; DO Sat, dissolved O_2 saturation.

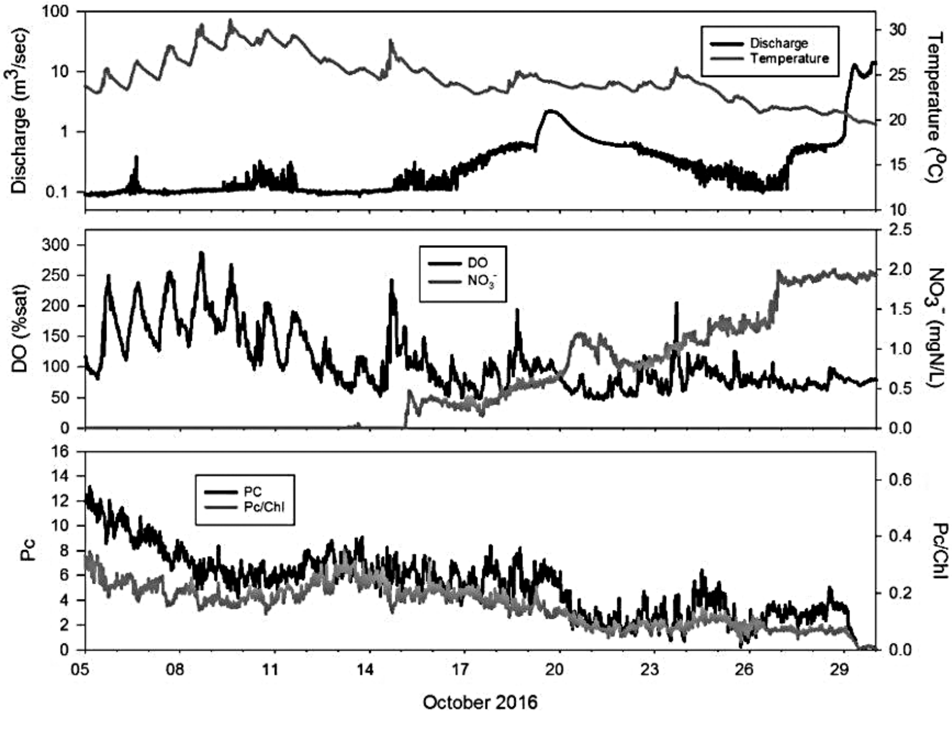

Fig. 8. Hydrophysical and biogeochemical conditions during dissipation of cyanobacteria bloom in fall 2016. Pc, phycocyanin; Chl, chlorophyll a; DO, dissolved O_2; DO Sat, dissolved O_2 saturation.

Implications for Monitoring and Management of Blooms

The present work indicates that a number of easily measured hydrophysical and hydrochemical parameters can serve as indicators and potential predictors of bloom onset. These include increases in water temperature, in vivo fluorescence of photosynthetic pigments, MHRT, and decreasing NO_3^- concentrations. All of these parameters are easily monitored or estimated using modern continuous sensors similar to those used in this study. While not all ponds will follow patterns identical to those seen in Coursey Pond, it is likely that most ponds that do show similar seasonal patterns in flow, temperature, and NO_3^- concentration will also show similar patterns of the onset and dissipation of cyanobacterial blooms, reflecting the common dependence of these patterns on watershed hydrology.

As irrigation practices within the watershed are likely reducing discharge, and therefore bloom dynamics, it is interesting to consider the prevention, mitigation, and control strategies that managers might use to moderate or minimize cyanobacterial blooms in Coursey Pond and its upstream tributary, Killens Pond (Ibelings et al., 2016). Readily available irrigation management tools such as the Delaware Irrigation Management System (http://dims.deos.udel.edu/) can reduce and spread the water pumping over longer times, a practice that could lead to levels of discharge, temperature, and NO_3^- concentrations in the ponds that would favor alternative phytoplankton species (and perhaps macrophytes) capable of outcompeting the bloom species.

An alternative approach to supply water for irrigation while maintaining flow rates in the upper watershed is to extract groundwater from lower in the watershed for use higher in the watershed. Changing the timing or locations of water supplies would require new resources for the infrastructure and the energy needed to the move the water, and new administrative and financial systems to share the groundwater resource. Other than the hydrological options, there are other control and treatment options, including within-pond chemical (Jančula and Maršálek, 2011) and physical (Visser et al., 2016) treatment systems that would likely be more expensive to build, operate, and maintain over the long term (www.epa.gov/nutrient-policy-data/control-and-treatment).

Conclusions

The Coursey Pond results are consistent with a systematic planktonic response to hydrophysical and hydrochemical forcing from its watershed. The early- to late-summer cyanobacterial blooms occur as the pond switches from a predominantly flow-through system (MHRT = 1–5 d) to a more stagnant closed system due to the reduction of discharge from the watershed (MHRT = 5–37 d, Fig. 3). Irrigation pumping likely exacerbates seasonal low-flow conditions, especially during years of high irrigation demand. The combination of high temperatures, reduced amounts of bioavailable N, and abundant dissolved and particulate P also favor cyanobacteria, including *Microcystis* spp., *Dolichospermum* spp., and *Oscillatoria* spp. (Beversdorf et al., 2013; Paerl and Otten, 2013), and particularly favor the cyanobacteria that are capable of fixing atmospheric N_2.

Seasonal pond hydrology is predictable, and therefore the onset, evolution, and dissipation of blooms are predictable. A better understanding of the relationships among watershed hydrology, pond biogeochemistry, and planktonic ecology in bloom-susceptible ponds, derived at least in part on long-term and high-frequency monitoring, will ultimately provide the basis for better management of noxious and potentially harmful blooms in small ponds of the mid-Atlantic region of the United States, and perhaps similar settings worldwide.

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