Biogeochemistry (2022) 159:87–102
https://doi.org/10.1007/s10533-022-00916-w

Temporal patterns in sediment, carbon, and nutrient burial in ponds associated with changing agricultural tillage

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Received: 8 October 2021 / Accepted: 28 February 2022 / Published online: 17 March 2022

Abstract Lakes and ponds play a disproportionate role in retaining sediment, carbon, nitrogen, and phosphorus, potentially mitigating negative environmental effects. However, how sequestration rates change over a pond's lifetime, and how rates are affected by watershed land use practices remains poorly characterized. In this study, we quantified sediment, carbon, nitrogen, and phosphorus burial rates, and the values of these ecosystem services, in three ponds. The ponds were 19–25 years in age (as of 2019), and their watersheds experienced a shift in the early 1990s to conservation tillage. We found that sediment burial rates decreased over time within these ponds (establishment to 2006, vs. 2006 to 2019), consistent with reduced soil erosion rates associated with conservation tillage. However, patterns in carbon, nitrogen, and phosphorus burial rates were not as clear; almost half of the elemental burial rates we quantified increased over time. We suggest that this may be due to increased importance of in-pond processes, such as in situ primary production and subsequent organic matter sedimentation, as the ponds age. Finally, we estimated the ecosystem service value of sediment, carbon, and nutrient retention by these ponds. We estimate that these three ponds provided ecosystem services equal to approximately 360,083 US$ over their lifetimes through burial of sediment, carbon, nitrogen, and phosphorus. Our results show that small retention ponds can provide considerable environmental and economic value by trapping and retaining sediments and nutrients.

Keywords Carbon · Nitrogen · Phosphorus · Ponds · Conservation tillage · Burial rates · Ecosystem services

Responsible Editor: Fiona Soper.

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Introduction

Lakes and ponds are globally important carbon (C), nitrogen (N), and phosphorus (P) sinks (Jossette et al. 1999; Harrison et al. 2009; Tranvik et al. 2009; Maranger et al. 2018; Rosentreter et al. 2021), despite occupying only 3% of the earth's land surface (Downing et al. 2006). Though small ponds occupy a relatively small percentage of land, they are numerous, particularly in rural areas. For example, in southwestern Ohio and the broader Midwestern USA, small ponds are a common feature on farms and exurban developments (Chumchal et al. 2016; Davis et al. 2021; Swartz and Miller 2021). Further, these and other regions are undergoing changes in agricultural practices, with an increasing proportion of crops grown using conservation tillage practices (Claassen et al. 2018), generally defined as tillage methods that leave ≥ 30% of the soil covered with crop residue. This disturbs soil less than "conventional" tillage, and is employed to reduce soil erosion (Claassen et al. 2018; Cusser et al. 2020). However, how this shift in tillage practice influences sediment, C, N, and P sequestration in downstream ponds is poorly characterized. Because ponds are potentially critical for sediment, C, N, and P sequestration, it is important to quantify how burial rates in these ecosystems change over time as agricultural practices shift.

Inland lakes and ponds have annual burial rates of organic C that are much higher, per unit area, than ocean sediments (Dean and Gorham 1998; USGS 1999). Furthermore, constructed impoundments (such as retention ponds and reservoirs) have higher sediment and organic C burial rates than natural lakes (Downing et al. 2008; Mendonça et al. 2017). High C burial rates have been attributed to relatively high inputs of allochthonous (often soil) C, in addition to settling and subsequent burial of organic C produced by autochthonous primary production. High primary production rates are typical of eutrophic ponds in agricultural areas, and burial of autochthonous C can contribute greatly to total C burial in sediments (Withers et al. 2014; Lignell 1990; Downing et al. 2008; Tranvik et al. 2009; Leach et al. 2018). Ponds may thus be effective C sinks on a global scale (Tranvik et al. 2009; Maranger et al. 2018), and burial rates may be particularly high if bottom water anoxia increases burial efficiency (Sobek et al. 2009; Hamre et al. 2018).

Lentic ecosystems are also hotspots for N removal via both N burial and denitrification, playing a disproportionate role (relative to their area) in N sequestration both at watershed and global scales (Harrison et al. 2009). In particular, reservoirs occupy only 6% of the global lentic surface area (which includes lakes plus reservoirs), but they retain approximately 33% of N removed by lentic systems via burial and denitrification (Harrison et al. 2009). Phosphorus is also buried within pond sediments (O'Connell et al. 2020). Because P tends to attach to soil particles, a large proportion of P runoff to water bodies is in particulate form, particularly in agricultural landscapes with significant soil erosion (USGS 1999; Stow et al. 2015). Organic P associated with settling phytoplankton and decaying macrophytes can also be buried at high rates, especially under eutrophic conditions (O'Connell et al. 2020). Though P retention in lakes and ponds may be insignificant in comparison to retention at the watershed scale, particularly in soils, it is nonetheless important to quantify as it can significantly impact water quality (Bennett et al. 1999; Schindler et al. 2016). Thus, P buried in pond sediments decreases the amount of P transported downstream, but this 'legacy' P in sediments can fuel eutrophication within the pond itself if sediment P is released to overlying water (for example, under anoxic conditions), a potentially long-term problem which may generate a positive feedback loop contributing to persistent eutrophication (Carpenter 2005; Wurtsbaugh et al. 2019).

Burial of sediments and elements are important ecosystem services, because excess transport of these constituents downstream or to the atmosphere as greenhouse gasses has potentially negative environmental effects. Burial in sediments reduces C emitted to the atmosphere as CO_2 or methane, where it can contribute to global climate warming (Allen et al. 2018). Burial also reduces C transport to downstream ecosystems, ultimately reducing ocean acidification (Weiss et al. 2018). Because both N and P loading in lakes, rivers, and coastal areas can fuel harmful algal blooms (Wurtsbaugh et al. 2019), the burial of these elements (and denitrification of N) can reduce eutrophication in downstream ecosystems. This in turn helps to reduce harmful algal blooms (HABs) that can lead to anoxia and toxin production that negatively impacts fisheries, drinking water, and recreation (Anderson et al. 2002). However, lentic systems can also be large sources of greenhouse gasses to the atmosphere, including CO_2, methane, and nitrous oxide (Deemer and Holgerson 2021; Tranvik et al. 2009; Allen et al. 2018; Beaulieu et al. 2019, 2020; Rosentreter et al. 2021). These emissions could offset some of the value lentic systems provide by burying sediments and elements.

Conservation tillage has been adopted in many agricultural areas to reduce soil erosion (Claassen et al. 2018; Cusser et al. 2020), but little is known about how this impacts the burial of sediment and elements in aquatic ecosystems. In our study watershed in rural southwestern Ohio, conservation tillage increased from about 15% of cropland in 1990 to about 65% in 2000, and has been relatively stable since (Fig. 1) (Renwick et al. 2018). Between 1995 and 2000, three small sediment retention ponds were built in this watershed in an effort to slow the sedimentation rates in Acton Lake, a downstream reservoir in Hueston Woods State Park that provides recreation and other ecosystem services such as climate regulation and nutrient transformation and retention (Kelly et al. 2018; Renwick et al. 2018). Sediment, C, N, and P burial rates were quantified in these three ponds in 2006 (Knoll et al. 2014). Though conservation tillage has been relatively stable in the watershed since 2000, studies show that there is often a lag in its effects on sediment, C, N, and P burial in downstream ecosystems (Wohl 2015; Wohl et al. 2015; Cusser et al. 2020). This phenomenon of legacy sediments is caused by a variety of anthropogenic activities, including changes in tillage practices, and has been documented for over a century (Marsh 1864). Indeed,

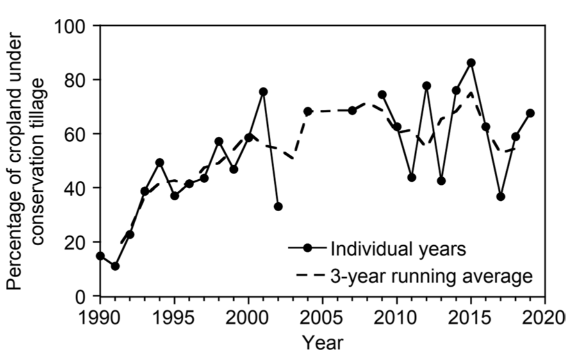

Fig. 1 Conservation tillage trends in the Acton Lake (Upper Four Mile Creek) watershed from 1990 to 2019. The solid line corresponds to individual years, while the dashed line represents the 3-year running average

concentrations of suspended sediments in streams draining this watershed, standardized for streamflow, have decreased substantially since 1994 and continue to decrease (Renwick et al. 2018). Therefore, we quantified the burial rates in these ponds again in 2019 to compare burial rates to those measured in 2006 and to test the general hypothesis that burial rates are changing in conjunction with increased conservation tillage in the watershed, which primarily occurred in the 1990s. Specifically, we predict that: (1) sediment burial rates will decrease over time in retention ponds whose landscapes have transitioned to primarily conservation tillage practices, due to lowered soil erosion rates; and (2) C, N, and P burial rates will also decrease in conjunction with reduced sediment burial rates. Alternatively, C, N, and P burial rates may not correlate well with sediment burial rate, as elemental burial rates also depend on in-pond processes such as primary production and subsequent burial of autotroph biomass (Downing et al. 2008; Nurnberg 1988). We also quantified the ecosystem services these ponds provide by way of sediment, C, N, and P burial, and compared this value to their construction cost to determine their approximate net worth.

Methods

Study sites

Our study sites consisted of three ponds (Table 1) located within < 1 km of each other in the Acton Lake watershed (also referred to as the Upper Four Mile Creek Watershed; USDA 1992) in southwestern Ohio (Fig. 2). These ponds are identified as Four Mile Ponds 9, 10 and 14 by USDA (1992), Renwick et al. (2005) and Knoll et al. (2014). The ponds are relatively shallow (0–3 m) and generally well mixed

Table 1 Descriptions of the study ponds

Pond	Pond surface area (m^2)	Age (years)	Watershed area (km^2)
9	36,811	24	3.18
10	24,222	19	1.37
14	9324	21	1.14

Age is as of 2019, when sampling occurred

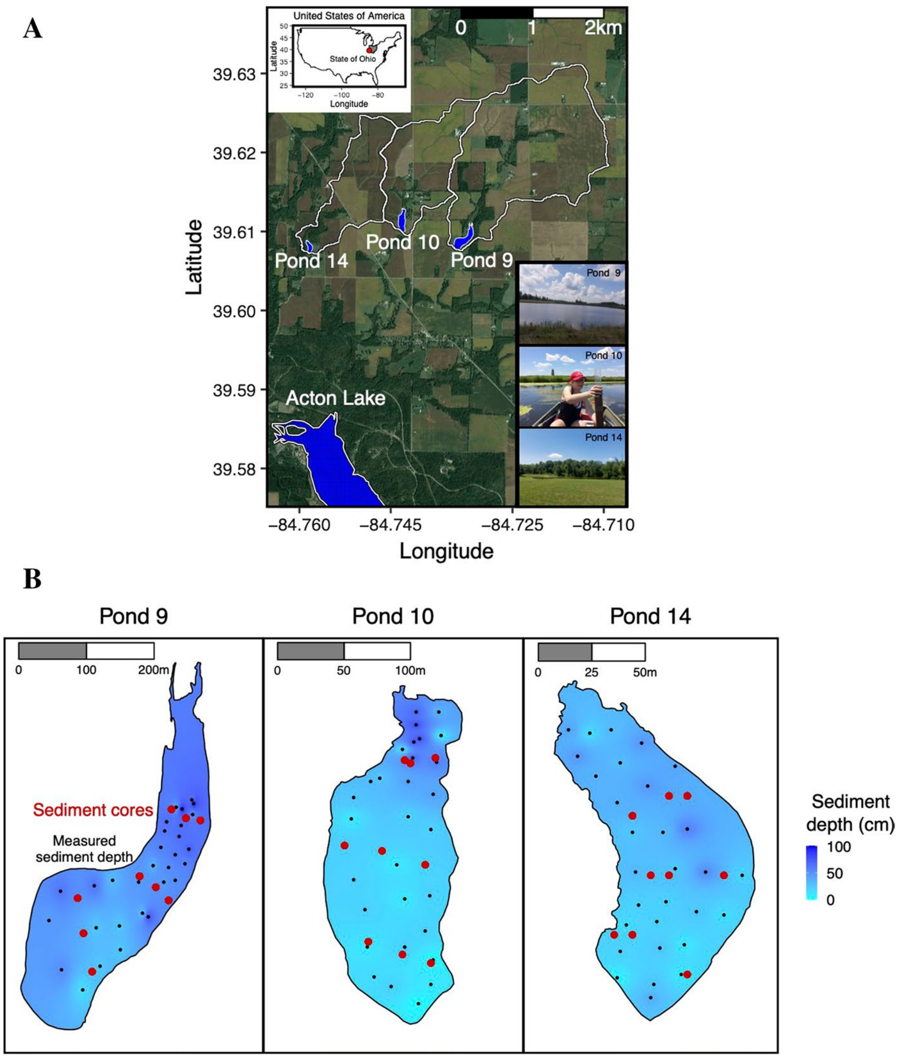

Fig. 2 **A** Locations of ponds and their corresponding watersheds, Acton Lake, and pictures of the ponds taken on the day of sampling. Pond and lake areas are shaded in blue while the white lines indicate the ponds' watershed boundaries. A map of the United States of America is given as a reference to the ponds' locations. **B** Locations of sediment cores (red) and depth measurements (black), as well as sediment depth (shades of blue), for each pond

throughout the year. They are permanently wetted year-round, and their sediments do not experience seasonal desiccation. They are fed by small zero to first order streams that experience occasional intermittency in surface flow. Ponds 9 and 14 are underlain by Russell-Xenia association soils, which are moderately- to well-drained and covered by a moderately thick layer of silty material (USDA 2005). The substrate beneath Pond 10 is also mostly Russell-Xenia association soils, but part of the pond lies on Fincastle-Brookston association soils; these are poorly drained deep soils, but are also covered by a moderately thick layer of silty material (USDA 2005). The watershed's tillage methods have been relatively stable since 2000, and the primary crops grown include soybean and corn (Renwick et al. 2018). Our methods for sediment collection and processing generally followed those of Knoll et al. (2014), who quantified burial rates in these ponds through 2006. However, our sampling was more spatially intensive. We collected nine intact sediment cores, and measured sediment thickness at 30 locations, within each pond.

Tillage practices

Tillage practices have been quantified in the Upper Four Mile Creek (UFMC) Watershed (above Acton Lake) annually since 1990 (except in 2003, 2005 and 2006). In late May-early June, we visually surveyed sites for crop type (soy, corn, etc.) and the type of tillage practice (conventional, conservation no-till, and conservation mulch-till) at fixed locations that were constant each year (Renwick et al. 2018). The data we present here are those for the UFMC Watershed as a whole, but several survey points are located in the watersheds of our study ponds, and tillage practices and crop types are similar throughout the UFMC Watershed.

Sediment depth and core collection

All of our sampling took place over the course of 2 months in the summer of 2019. We measured sediment thickness in each pond in a general grid pattern, collecting more measurements where sediment depth changed rapidly (Fig. 2). This approach allowed us to account for potential spatial variability to accurately characterize whole-pond estimates of sediment and element burial. Sediment depth was measured with a 2.5 cm diameter graduated pole with a pointed end (Renwick et al. 2005; Knoll et al. 2014). The pole was plunged through the water until the top layer of sediment was detected. The probe was then pushed through the sediment while counting off 2-inch (5.08 cm) increments until the much harder, pre-pond terrestrial soil was detected, and the depth between these was noted. The difference in depth was recorded to the nearest inch (2.54 cm) and used as our estimate of sediment thickness (Renwick et al. 2005; Knoll et al. 2014).

We collected nine sediment cores from each pond, in a 3 × 3 grid, with three cores each taken longitudinally near the stream inflow, middle of the pond, and near the outflow (dam) (Fig. 2). In each of these three transects, the three longitudinal cores were collected at roughly equal distance across the width of the pond. Cores were collected with a 4.76-cm diameter gravity corer (Wildco, Yulee, FL, USA). They were capped and sealed in the field, and kept on ice until transported to the lab (~ 15 km) within 5 h after collection. In the lab, cores were kept upright overnight at 10℃ to allow settling of any surface materials re-suspended during extraction and transport. The next morning, we siphoned excess water from the top of the cores without disturbing the sediments, and froze them at − 20℃. Once frozen, we extruded each core as an intact cylinder and sectioned them with a heated handsaw. Cores were sectioned at 4 cm intervals, except for the surface section which was cut into a slice representing the top 0–2 cm.

Loss on ignition (LOI) and dry bulk density (DBD)

Our analytical methods were similar to those of Knoll et al. (2014). Briefly, each sectioned core slice was oven dried at 60 °C in a pre-weighed, acid-washed plastic cup until a constant mass was reached. Dry bulk density (DBD, g dry mass cm^{-3}) was quantified for each slice using section dry mass and volume. LOI was quantified for each slice as an indicator of organic matter content. For LOI, subsamples of each slice were weighed, ashed for four hours at 550℃, then weighed again. The difference in mass pre-and post-ignition was converted to LOI as percentage of total dry mass lost.

Carbon, nitrogen, and phosphorus quantification

We quantified total C concentration for all sediment core slices using a 10–15 mg subsample of dried and homogenized sediment. Additionally, we quantified the relative fraction of organic and inorganic C on a subset of sediment core slices. For this subset, we selected the top slice of every core in all ponds, and all of the slices from one single core from Pond 10. To quantify inorganic C concentration, we ashed samples at 550 ℃ for 4 h to remove organic C. We also corrected inorganic C for LOI [Proportion inorganic carbon = (mass inorganic carbon)/(mass ashed/(1 − LOI))]. Organic C was estimated as the difference between the total C and the ashed inorganic C in the samples. N content was quantified on un-ashed, dried, and homogenized subsamples from every sediment core slice. All C and N samples were analyzed with a CE Elantech Flash 2000 CHN analyzer (Lakewood, NJ, USA).

We determined phosphorus content on a 3–6 mg subsample from each sediment core slice. Dried and homogenized sediment was weighed on foil and transferred to a glass vial. The foil was reweighed to account for any sediment remaining on the foil. Samples were ashed in their vials for four hours at 550 ℃. We also ashed a subset of samples for 1 h at 550 ℃, following the method used in the previous study (Knoll et al. 2014), to ensure that P concentrations were not different for different ashing times (we found no significant difference; average CV = 0.056). After ashing, samples were digested with HCl and analyzed for soluble reactive phosphorus (SRP) using colorimetric methods on a Lachat QC 8000 auto-analyzer (Milwaukee, WI, USA).

Burial rates of sediment, C, N, and P

We calculated total pond sediment mass and burial rates using two different approaches. Our first method is the same as that used by Knoll et al. (2014), henceforth referred to as the "simple mean" method, which allows us to directly compare our study and theirs. The second method used a spatially explicit approach that accounted for heterogeneity in sediment depth and distribution within the ponds, henceforth referred to as the "spatially explicit" method.

For the simple mean method, we estimated total sediment volume by multiplying pond area by the mean of the 30 discrete sediment depth measurements. Sediment volume was then multiplied by mean DBD to estimate total sediment dry mass. Sediment, C, N, and P burial rates from pond establishment to 2019 for each pond were calculated from averaging concentrations (element mass per g dry mass of sediment) in all slices in each pond. These elemental means were then multiplied by total sediment dry mass, then divided by the age (years) and area (m^2) of the pond to obtain annual areal burial rates. To obtain sediment burial rates, we divided total sediment dry mass in the pond by the age of the pond in years (kg $year^{-1}$) over the life of the pond, from pond establishment to 2019.

For the spatially explicit method, we estimated whole-pond sediment depth using an inverse distance weighted (IDW) interpolation from the 30 discrete sediment depth points. IDW interpolations were fit with the "idw" function in the R package "gstat" (Pebesma 2004; Gräler et al. 2016; R Core Team 2020). We then calculated the area represented by each cross-sectional slice (broken into 2 cm increments over the entire depth). The depth of each slice (2 cm) was multiplied by its corresponding area to find volume for that stratum. This was multiplied by the average DBD for that depth to estimate the dry sediment mass represented by that stratum. Dry mass was summed for the entire sediment depth to estimate total dry sediment mass in each pond. For deeper slices where we did not have DBD measurements, we used the average of the deepest 4–5 DBD measurements we had for that pond. This approach is justified as the deepest 4–5 DBD measurements were relatively consistent (Fig. 3).

Burial rates using the spatially explicit method were then calculated by averaging the C, N, or P (proportion of dry mass) in each layer (for example, in layers 0–2 cm, 2–6 cm, etc.) for each pond. Because some cores did not reach the deepest sediments, the values for deeper layers were estimated as the mean of the bottom 4 overlying layers for which we had data. This assumes that concentrations do not change below the lowest layer measured, which is consistent with findings from Acton Lake (downstream from our study ponds) as well as other reservoirs in Ohio (Vanni et al. 2011). Because the proportion of organic C was known only for a subset of sediment core slices

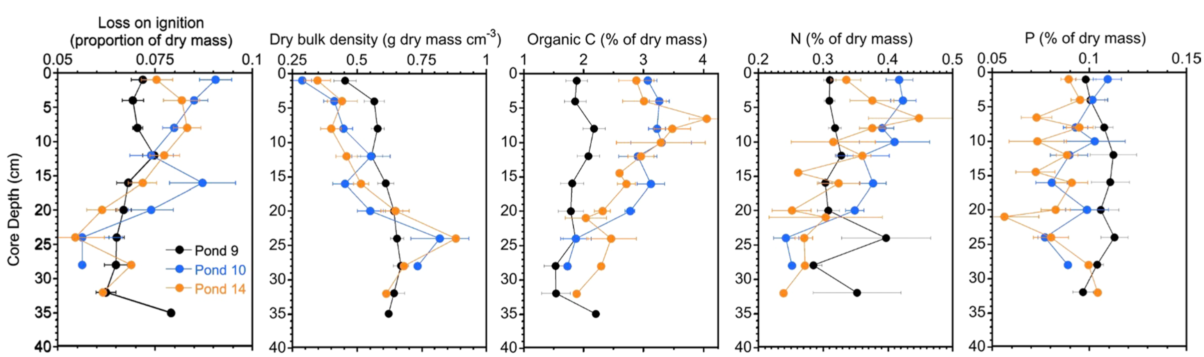

Fig. 3 Average loss on ignition, dry bulk density, and organic C, N and P concentrations in sediments, as a function of sediment depth in the three ponds. If no error bar is included, only one core had data for that depth. The black line represents Pond 9 data, the blue line represents Pond 10 data, and the orange line represents Pond 14 data

(as described above), we estimated organic C in each layer as follows. Using the organic C measurements which were measured on the one entire core, the proportion organic C to total C was calculated for each sediment layer. That average proportion was multiplied by each layer's total C to obtain organic C for that layer. C, N, and P values (proportion of dry mass) for each layer were then multiplied by the sediment mass of that layer to find the mass of C, N, and P (g) for each layer, and these were then summed to find a total C, N, and P mass for each pond. These values were then divided by age of the pond (years) and area (m^2) to find annual areal burial rates. We recognize that estimating organic C in this manner may introduce some error compared to if we directly estimated organic C on all slices. However, we feel this approach is justified because organic C and LOI were positively correlated in the slice for which we analyzed both constituents ($r^2 = 0.449$, $n = 33$, $P < 0.01$) and LOI showed the same pattern with depth in all three ponds (see Fig. 3). Furthermore, other studies have estimated organic C concentration using LOI (e.g., Downing et al. 2008).

We report values from both burial rate methods. To compare burial rates between periods, total sediment, C, N, and P masses (per pond) found in 2006 by Knoll et al. (2014) were subtracted from their respective masses found in 2019 (using the simple mean method), to quantify the mass of sediment, C, N, and P accumulated between 2006 and 2019. These were then divided by the area of the pond and converted to average annual areal rates.

Effectiveness of ponds in trapping sediments and elements

We also estimated the percentage of sediment, N, and P loads to Acton Lake that these ponds trapped annually and over their lifetimes. Methods for estimating loads to Acton Lake, which we have quantified every year since 1994, are described in Renwick et al. (2018) and Kelly et al. (2018). Briefly, loads were calculated from stream discharge, monitored continuously, and concentrations of suspended sediment, total N, and total P are obtained from high-frequency sampling, on three of Acton Lake's inflow streams that collectively drain 86% of the lake's watershed (Renwick et al. 2018). These sites are located below our three study ponds in the Acton Lake watershed (Fig. 1). We do not have estimates of total or organic C loads to the lake. To calculate the proportion of Acton sediment, N, and P loads trapped by the ponds, we divided annual burial rates of these constituents in the ponds by annual loads to the lake. Because the ponds vary in age, for each pond we used annual loads over that pond's lifetime (through 2017, the most recent year of loading data available). We also compared these proportions to the proportion of the lake's watershed that drains into our study ponds; this serves as a measure of the effectiveness of the ponds in trapping constituents.

Ecosystem services provided by sediment and element retention

We estimated the value of ecosystem services provided by sediment and element retention in the three ponds. All values are expressed in 2019 US$. For sediment, we multiplied the proportion of Acton Lake sediment load trapped (buried) by the three ponds (proportion sediment trapped, or Sed_{prop}), by the estimated costs of sedimentation and flooding in the lake's watershed. In the early 1990s, damage caused by flooding and the resulting sedimentation in the Acton watershed was estimated to cost 381,400 US$ annually. Assuming this was based on 1992 US$, converting it to 2019 dollars yields an annual cost of 694,993 US$. However, the 1992 cost estimate was based on a sedimentation rate in Acton Lake measured from 1979 to 1987. A more recent estimate, from 1987 to 2001, yielded a sedimentation rate that was only 26% that of the earlier rate, a decline attributed to improved land use practices (Renwick et al. 2005). Assuming that the cost of sedimentation damage is proportional to sediment load, we estimated the cost of flooding and sedimentation in the Acton Lake watershed to be 694,993 US$ * 0.26 or 180,698 US$ $year^{-1}$. The original cost estimate included the costs of both flooding plus sedimentation, and we do not know the precise relative costs of these two processes. However, the USDA (1992) report indicates that most of this cost is due to sedimentation. Therefore, we then multiplied Sed_{prop} by 180,698 US$ to obtain the annual value of sediment trapping by the ponds.

To estimate the value of the C and N burial in the ponds, we used the 'social costs' of these elements (defined as the value of the cumulative, worldwide impact one additional ton of that element has after emission), again adjusting to their value to 2019 dollars. Values for the social cost of carbon vary, but the most widely used value is 31 US$ per metric ton of CO_2 (Nordhaus 2017; Bradbury et al 2021); this represents the economic damage that would result from emission of one ton of CO_2 into the atmosphere. We converted this to a cost of 114 US$ per ton of C (because C represents 27.3% of the mass of a CO_2 molecule). To estimate the social cost of nitrogen, we converted Keeler et al.'s (2016) estimate of 2620 US$ per ton N based on 2010 US$ to 2019 dollars, which yields a cost 3,070 US$ per metric ton N.

To estimate the value of P trapped by the ponds, we used the cost of removing P, based on a water quality credit trading program developed for the Great Miami River Watershed, within which our study ponds' watersheds (and the Acton Lake watershed) reside (Miami Valley Conservancy District 2017). This value is based on the cost of implementing best management practices to reduce non-point P loading to water bodies from agricultural areas, and was estimated to be 2381 US$ per ton P based on 2005 dollars; we converted that to 2019 dollars to yield a cost of $3,117 per ton P. For C, N, and P, we multiplied these cost values ($ per ton) by the annual burial rates to obtain the value provided by these ponds by trapping these elements.

The ecosystem services we include here correspond to the Common International Classification of Ecosystem Services (CICES) classes 2.1.2.1 (Filtration/sequestration/storage/accumulation by ecosystems) because we are directly quantifying sediment, C, N, and P sequestration in the pond sediments; 2.3.4.2 (Chemical condition of freshwaters) because sequestration of sediment, N, and P in pond sediments should benefit downstream waters; and 2.3.5.1 (Global climate regulation by reduction of greenhouse gas concentrations) because C and N sequestered in pond sediments should reduce the fluxes of CO_2, CH_4 and N_2O to the atmosphere (Czúcz et al. 2018).

Results

Sediment nutrient concentrations

Mean sediment N and P concentrations (proportion of dry mass) were similar among the ponds, differing by < 20% between the lowest and highest value (Table 2). However, mean organic C concentration varied > 50% among ponds, mainly because it was lower in Pond 9 than in the other two ponds (Table 2). Organic C concentration decreased with sediment depth in all three ponds, and N concentration decreased with depth in Ponds 10 and 14 (Fig. 3). In Pond 9, N decreased slightly with depth from 0 to ~ 22 cm, below which it varied irregularly. Depth profiles for P were more complex; in Ponds 10 and 14, P concentration decreased with depth from the sediment surface to ~ 20–25 cm and then increased (Fig. 3). In contrast, Pond 9 showed the opposite pattern, but with more subtle changes in concentration (Fig. 3). Below ~ 25 cm, all three ponds had somewhat similar P concentrations of ~ 0.1% dry mass.

LOI and DBD

Trends in mass LOI and DBD reflect trends in the relative contribution of organic matter. Average LOI ranged from a mean of 6.87% in Pond 9 to 7.99% in Pond 10 (Table 2) and generally decreased with depth as expected (Fig. 3). Mean sediment DBD, on the other hand, generally increased with depth (Fig. 3) and ranged from a mean of 0.47 (Pond 10) to 0.59 (Pond 9) g cm^{-3} (Table 2). LOI was higher near the outflow of each pond, while DBD was generally highest near the inflow (data not shown).

Table 2 Average values for the three ponds' sediment, based on sampling in 2019

Pond	Dry bulk density (g cm^{-3})	Loss on ignition (% dry mass)	Organic C (% dry mass)	N (% dry mass)	P (% dry mass)
9	0.59 (±0.12)	6.87 (±0.782)	1.88 (±0.537)	0.321 (±0.0839)	0.105 (±0.022)
10	0.47 (±0.18)	7.99 (±1.45)	3.01 (±0.452)	0.379 (±0.0781)	0.095 (±0.022)
14	0.50 (±0.22)	7.44 (±1.36)	2.84 (±0.961)	0.335 (±0.0831)	0.087 (±0.017)

Values are calculated using a similar approach to the simple mean method, as described in the Methods section

Burial rates of sediment, C, N, and P

The two methods (simple mean vs. spatially explicit) for estimating burial rates from date of establishment to 2019 (e-2019) yielded very similar rates (Fig. 4). Annual sediment burial rates varied substantially among the ponds (Table 3), but among-pond patterns were similar for all three time periods. That is, sediment burial rates were lowest in Pond 14, and highest in Pond 9 in all time periods (Fig. 4). Annual C, N, and P burial rates (g m^{-2} $year^{-1}$) from e-2019, and from 2006 to 2019 varied among ponds similarly to sediment burial rates, i.e., all rates were lowest in Pond 14 and highest in Pond 9 (Fig. 4). However, elemental burial rates from e-2006 did not vary similarly to sediment burial rates. Pond 10 had the lowest C, N, and P burial rates for the e-2006 period, even though its sediment burial rate was intermediate to that of the other two ponds (Fig. 4). However, for the e-2006 period, Pond 14 had the highest C burial rate, Pond 9 had the highest N burial rate, and P burial rates were similar among all ponds (Fig. 4). Burial rates were very similar with the simple mean and spatially explicit methods (Fig. 4).

Comparison of burial rates in the two time periods

As predicted, annual sediment burial rates were lower from 2006 to 2019 than from date of establishment to 2006 in all three ponds (Fig. 4). However, elemental burial rates did not always follow this temporal pattern. Of the 9 elemental burial rates (3 elements × 3 ponds), 5 were lower after 2006 than before 2006 (Fig. 4). In two ponds (9 and 14) organic C and N burial rates were lower after 2006 than before. P burial rates were lower in Pond 14 post-2006 than pre-2006, but in the other two ponds P burial rates actually increased after 2006, and the increase was particularly pronounced in Pond 9 (Fig. 4). In Pond 10, burial rates of all three elements were higher after 2006 than before 2006, although differences were relatively slight.

Effectiveness of sediment, N, and P burial in ponds

The three ponds collectively trapped and buried 4.01, 0.24 and 2.70% of the loads of suspended sediment, total N and total P delivered to Acton Lake, respectively. The ponds' three watersheds collectively

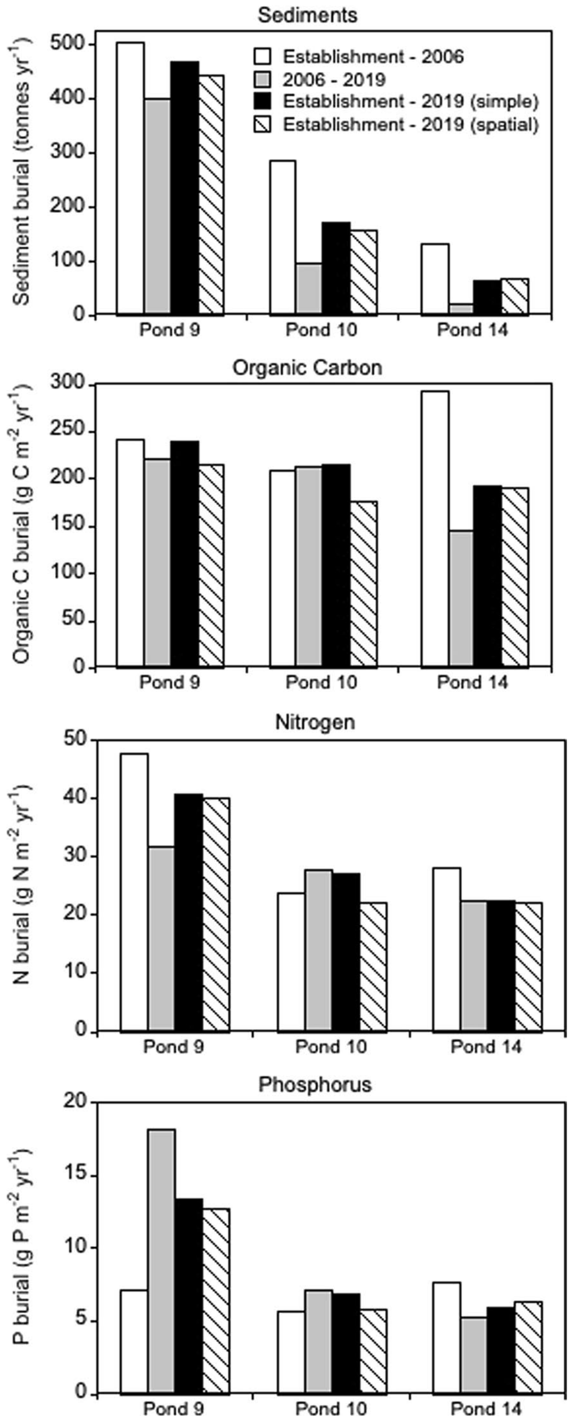

Fig. 4 Burial rates for sediment and carbon, nitrogen, and phosphorus in the sediments of the three ponds. Rates are presented for the time of establishment (pond construction) to 2006 (white bar), 2006 to 2019 (gray bar), establishment to 2019, calculated with the simple mean method (black bar), and establishment to 2019, calculated using the spatially explicit method (striped bar)

Table 3 Burial rates for the three ponds from establishment (year of construction) to 2019

Pond	Sediment burial (MT $year^{-1}$)		Organic carbon burial (g m^{-2} $year^{-1}$)		Nitrogen burial (g m^{-2} $year^{-1}$)		Phosphorus burial (g m^{-2} $year^{-1}$)	
	Simple mean	Spatially explicit	Simple mean	Spatially explicit	Simple mean	Spatially explicit	Simple mean	Spatially explicit
9	468	446	239	214	40.7	39.9	13.4	12.7
10	173	158	216	176	27.1	22.1	6.8	5.8
14	63	68	192	190	22.6	22.2	5.9	6.3

Both methods used (simple mean and spatially explicit) are represented

represent 2.15% of the lake's watershed. Thus, the ponds were effective traps of P, and in particular sediment, relative to the watershed areas they drain. However, they were quite ineffective at trapping N, at least when expressed as the proportion of total N load.

Ecosystem services provided by sediment and element retention

Assuming the ponds reduce sedimentation by 4.01%, and that sedimentation damages in the Acton watershed costs about 180,698 US\$ $year^{-1}$ annually, the ponds save 7246 US\$ (2019 US\$) per year in sedimentation damages. Thus, when accounting for each pond's age and annual sediment burial rate, by trapping sediment the three ponds combined have saved approximately 142,625 US\$ since they were established. By burying C, N, and P, the three ponds together provide ecosystem services equal to 1,693 US\$ $year^{-1}$ for C, 7023 US\$ $year^{-1}$ for N, and 2150 US\$ $year^{-1}$ for P. Over the ponds' lifetimes, these services equate to 32,437 US\$ for C, 140,605 US\$ for N, and 44,416 US\$ for P. Summing up the values of all four ecosystem services (sediment, C, N, and P burial), the ponds have provided a value equal to 360,083 US\$ since they were constructed (Fig. 5), with pond 9 contributing by far the most value (272,137 US\$) compared to Pond 10 or Pond 14 (49,359 and 38,588 US\$, respectively). The relatively high value of Pond 9 is probably because it is the oldest and largest pond, and has the largest watershed (Table 1).

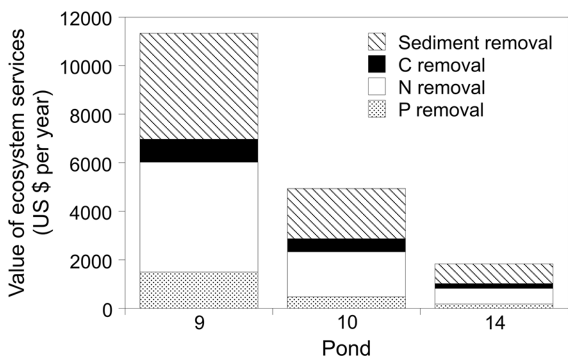

Fig. 5 The ecosystem service values of sediment, C, N, and P sequestration in US\$ per year for each pond

Discussion

Sediment burial rates

Data from all three ponds are consistent with the hypothesis that sediment burial rates decrease over time in association with sustained conservation tillage. Sediment burial rates from 2006 to 2019 were lower in all three ponds (on average 57% lower) than the rates from establishment to 2006 (Fig. 4). The magnitude of sediment burial decreased with decreasing pond and watershed size (Table 1; Fig. 4). While we cannot be sure that the lower sediment burial rates after 2006 are due to sustained conservation tillage, suspended sediment concentrations in streams draining into Acton Lake downstream of our study ponds (Renwick et al. 2018), as well as sediment loads to Acton Lake (Kelly et al. 2018), have been declining since the mid-1990s. These trends, as well as those in other watersheds experiencing pronounced increases in conservation tillage (Tiessen et al. 2010; Baker et al. 2014), suggest that conservation tillage reduces sediment delivery to water bodies by reducing soil erosion. Our results suggest that this is manifested by lower sediment burial rates in ponds draining such watersheds.

Further evidence that sediment inputs have decreased over time derives from the prediction that these ponds would fill with sediment within 25 years of establishment (USDA 1992), which has not occurred for any of the ponds. For example, Pond 9, the oldest of our study ponds, was built in 1995, and therefore was predicted to fill entirely with sediment by 2020. However, when we sampled in 2019, at nearly the end of its predicted lifetime, pond 9 was not close to being completely filled with sediment. Similarly, the other two ponds do not appear to be anywhere close to being filled in. Mean sediment thicknesses at the time of our 2019 sampling were 0.51, 0.29 and 0.28 m in Ponds 9, 10 and 14, respectively. Although we did not measure water depth in the ponds, based on our observations when measuring sediment thickness, all ponds have a water depth of at least 1 m. Thus, the ponds do not appear to be close to filling in with sediment, and sediment burial rates are lower than predicted in the early 1990s, before conservation tillage became widespread in this watershed. As was mentioned above, in the Four Mile Creek (Acton Lake) watershed, conservation tillage increased significantly from 1990 to 2000 and has remained approximately constant since this time (Renwick et al. 2018) (Fig. 1). Our study ponds were built in 1995, 1998 and 2000. Thus, the shift to conservation tillage was underway when these ponds were established. However, one would expect the effects of this land management shift on pond burial rates to lag behind management implementation, because many effects of conservation tillage, at the plot scale, take decades to play out (Cusser et al. 2020). Specifically, within this watershed, suspended sediment concentrations in streams draining into Acton Lake have continued to decrease through 2014 (Renwick et al. 2018) and subsequent years (MJ Vanni and BP Grudzinski, in prep.). Furthermore, it can take years for ‘legacy’ sediments in streams to be transported downstream and thus for stream sediment concentrations to reach a steady-state (Wohl 2015; Wohl et al. 2015). Therefore, we would expect that sediment delivery to these ponds may still be declining in response to the increase in conservation tillage.

Elemental burial rates

Trends in elemental burial rates were much more complex than sediment burial rates. In particular, only 5 of 9 elemental burial rates were lower after 2006 than before 2006. However, for each of the three elements, burial rate increased with pond and watershed size (Table 1; Fig. 4). In a proximate sense, the reason why elemental burial rates did not follow the same temporal trends as sediment burial rate is because most (7 of 9) elemental concentrations were higher after 2006 than before (Table 2 and Knoll et al. 2014). Sediment P concentrations were much higher (by $\geq 81\%$) in all three ponds, and in Ponds 10 and 14, C and N concentrations increased by $> 50\%$ and $\geq 75\%$, respectively. However, in Pond 9 organic C concentration was essentially unchanged ($< 1\%$ difference) while N concentration was $\sim 14\%$ lower in 2019 compared to 2006 (Knoll et al. 2014). Although 7 of 9 elemental concentrations increased over time, all of these values are within the ranges reported for these elements by Knoll et al. (2014) for sediment concentrations in 13 ponds in southwestern Ohio (our three study ponds plus 10 others).

One likely reason for the temporal increase in elemental concentrations is that as the ponds age and watershed inputs decline due to reduced soil erosion, in-pond processes should contribute relatively more to sediment elemental composition. In particular, relatively increased primary production by pond algae and macrophytes should increase the concentration of organic matter and nutrients in the sediments (Downing et al. 2008). The general decrease in concentrations of all three elements with depth (Fig. 3) probably reflects a greater contribution of deposition of autochthonous production in surficial sediments relative to inputs of soil from the watershed, which would have higher concentrations of inorganic materials and lower concentrations of organic C, N and P. The exception was Pond 9, which showed decreased organic C with depth but more complex patterns for N and P (Fig. 3). As organic matter produced by algae and macrophytes increases over time, this matter increasingly dominates pond sediment composition.

In streams draining into Acton Lake (downstream of the ponds, just before streams enter the northern end of the lake), sediment-bound particulate P and N concentrations are declining over time, but the concentration of soluble reactive P has actually been increasing over the past decade (Renwick et al. 2018; Kelly et al 2019). These trends are consistent with those seen in tributaries of Lake Erie that have experienced similar increases in conservation tillage in their watersheds, an effect attributed to saturation of surface soil with dissolved P after many years of conservation tillage (Joosse and Baker 2011; Baker and Richards 2002; Renwick et al. 2018). Because P is often the limiting nutrient for aquatic primary producers, it is possible that primary production in our study ponds has been increasing or at least has not declined over time, providing a steady flux of autochthonous organic matter to the sediments. We do not have current data on pond productivity, but one pond (Pond 10) was nearly completely covered with free floating, rooted non-emergent, and rooted emergent macrophytes, and all ponds had some level of macrophyte growth. Chlorophyll measurements from 2010 were indicative of eutrophic conditions, with values between 64 and 70 µg chlorophyll L^{-1} (LB Knoll, unpublished data). Furthermore, the a temporal decrease in C:N burial ratios also support the idea of increased primary production (described below).

Sediment elemental composition varies considerably among lakes, but surface sediment C and P concentrations tend to be higher in natural lakes than in reservoirs (Dean and Gorham 1998; Nurnberg 1988). Presumably, this is at least partially due to larger watershed area: lake area ratios in reservoirs, which results in a greater influence of watershed inputs (primarily eroded soil in agricultural watersheds) in reservoirs compared to natural lakes. Watershed inputs, which are rich in inorganic materials such as silt and clay, may 'dilute' autochthonously produced organic matter being deposited onto sediments. The watershed area: pond area ratios for our study ponds range from 56 to 122, which are larger than those for most natural lakes. Therefore, our study ponds likely receive considerable inputs of materials from their watersheds, but as the ponds age, these inputs would contribute relatively less compared to autochthonous production, and sediment C, N and P concentrations would thus increase over time.

Sediment elemental concentrations are also affected by biogeochemical processes occurring within the sediments. For example, microbial respiration, methanogenesis, and denitrification convert C and/or N to gaseous forms, which can all decrease the concentrations of C and/or N in the sediments. In contrast, P may be trapped more tightly in sediments, particularly if overlying water is oxic. This would result in greater retention of P relative to C and N over time. Indeed, in Acton Lake and other reservoirs, the N:P ratio of buried sediments is much lower than the N:P of stream inputs, or N:P in the lake water column (Vanni et al. 2011). Alternatively, nitrogen fixation by cyanobacteria in the ponds could serve as a potentially significant N input, though for the same reasons explained above, denitrification is likely the dominant of these two processes. Additional evidence that elemental burial is strongly influenced by autochthonous production in our study ponds is the C:N ratio of burial, which varied from 6.3 to 10.0 (molar) across the three ponds (obtained by dividing C burial by N burial from e-2019). These C:N ratios indicate a dominance of autochthonous sources (Meyer and Ishiwatari 1993). Furthermore, burial C:N decreased with pond age (6.3, 9.3, and 10.0 in Ponds 9, 10, and 14, respectively), which is what one would expect if autochthonous processes are increasingly important with pond age.

Thus, a combination of decreased watershed inputs, increased deposition of materials derived from autochthonous production, and greater retention of P compared to C and N can potentially explain the patterns we observed in sediment and elemental burial. The greatest deviation between temporal trends in sediment burial rates vs. elemental burial rates we observed was for P in Pond 9. Since Pond 9 is the oldest of all three ponds sampled, we hypothesize that the processes outlined above have occurred to a fuller extent in Pond 9 than in the other two ponds.

The location of these ponds (and other ponds that were planned but never constructed) were chosen because they reside in watersheds with highly erodible soils (USDA 1992). Therefore, their effectiveness at trapping sediments was expected. Although the ponds' watersheds drain 2.15% of Acton Lake's watershed, the ponds trap 4.01% of the sediment load to the lake. They are also effective at trapping P (2.7% of the total P load), but are ineffective at trapping N (only 0.24% of the total N load to the lake). The relatively low efficiency at trapping N, compared to P and sediments, is probably because most of the N load is nitrate while about half of the P load is in particulate form (Vanni et al. 2001; Kelly et al. 2018). During high stream flow, which is when most nutrients and sediments are delivered to the ponds, most nitrate is probably not taken up and is exported downstream. However, denitrification probably also contributes to the low N retention efficiency (Harrison et al. 2009), and is also a valuable ecosystem service.

Ecosystem services provided by sediment and element retention

These ponds were originally established to reduce sedimentation rates in downstream Acton Lake (USDA 1992). Thus, we were interested in their effectiveness in trapping sediment and the ecosystem services they provide by trapping sediments and elements. Our estimates suggest that over the lifetimes of these ponds so far, these ecosystem services have been worth 360,083 2019 US$. We do not know the exact costs of constructing these ponds, but they were estimated to be 56,700; 50,100 and 33,700 US$ (1991 US$) for Ponds 9, 10 and 14, respectively (USDA 1992), for a total of 140,500 US$, which equates to 263,728 in 2019 US$. Therefore, based only on C, N, P, and sediment burial, the ponds have provided ecosystem services that exceed their costs by nearly $100,000, through 2019. However, because autochthonous production likely contributes to elemental burial in the ponds, it is possible that our calculations overestimate the worth of these ponds in terms of the trapping of nutrients that would have otherwise been transported downstream or emitted to the atmosphere. Regardless, though, trapping of either autochthonous or allochthonous C, N, or P removes it (at least temporarily) from the global cycle and is thus a service. Furthermore, the values we estimated do not include other ecosystem services such as denitrification, recreation, conservation of biodiversity, flood protection, pollination, habitat provision, microclimate regulation, and other services. Conversely, pond construction may have reduced the value of some ecosystem services provided by the streams that were dammed to form the ponds, for example migration of fish and maintenance of natural flow in downstream reaches. In addition, the ponds occupy land that could otherwise be arable, and they could be sources of CO_2, methane and nitrous oxide, which contribute to climate warming; these fluxes would thus be considered ecosystem disservices. While an extended discussion of the consequences of these ecosystem services and disservices is beyond the scope of this paper, it is clear that the ponds at least provide a valuable service by trapping sediments and elements, which could be something of great importance to consider as agriculture and landscapes evolve.

Conclusion

Our sediment burial results, when compared to the results of Knoll et al. (2014) and combined with data on streams entering Acton Lake (Renwick et al. 2018; Kelly et al. 2018, 2019), are consistent with the hypothesis that the shift to conservation tillage in the watershed contributed to reduced sediment burial rates from 2006 to 2019 compared to the pre-2006. While temporal trends in C, N, and P burial rates are not as clear, and therefore do not support the hypothesis that conservation tillage also reduces C, N, and P burial rates, this is likely because in-pond processes countered any effects that conservation tillage may have had. Future research is needed to quantify the long-term effects of sustained conservation tillage on pond burial rates, as this agricultural practice is now widespread and likely will continue to be so (Claassen et al. 2018). It is also clear that ponds can provide valuable ecosystem services by trapping sediments and elements, but future studies should assess the full range of ecosystem services and disservices provided by ponds in agricultural landscapes.

Acknowledgements We thank the property owners for allowing us access to their ponds, and BJ Price from the Preble County Soil and Water Conservation District for coordinating site access. We also thank Tera Ratliff of the Miami University Center for Aquatic & Watershed Sciences (CAWS) for her assistance in C, N and P sample analyses, Josh Tivin for his help in processing samples, Bartosz Grudzinski for his aid in the field, and Miami University's Ecology Research Center for boat access. This paper benefited from comments on a prior draft by María González, Carrie Ann Sharitt, Elizabeth Gallagher, Heather Luken, Katie Busch, and Amy Weber. Financial support for this project was provided by National Science Foundation (NSF) Grants 1255159 and 1930655, and MNR was supported by a Research Experiences for Undergraduates (REU) supplement to NSF Grant 1255159 during summer 2019.

Author contributions MNR, TJW, and MJV developed the study questions, experimental design, and conducted the field and laboratory analyses. LBK provided data from cores collected in 2006. All authors contributed to the manuscript preparation and revision. All authors have read and approved the final manuscript draft.

Funding Financial support for this project was provided by National Science Foundation (NSF) Grants 1255159 and 1930655, and MNR was supported by an Research Experiences for Undergraduates (REU) supplement to NSF Grant 1255159 during summer 2019.

Data availability Data are not yet provided. However, our data will be permanently archived on EDI along with other data from our Acton Lake/watershed project after publication. The link to access this is as follows: https://portal.edirepository.org/nis/mapbrowse?packageid=edi.256.1.

Code availability Not applicable.

Declarations

Conflict of interest Not applicable. The authors have no relevant financial or non-financial interests to disclose.

Ethical approval Not applicable.

Consent to participate Not applicable.

Consent for publication Not applicable.

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