Florida’s urban stormwater ponds are net sources of carbon to the atmosphere despite increased carbon burial over time

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Stormwater ponds are engineered ecosystems designed for flood control and sediment retention in urban watersheds. They are the most commonly used stormwater control measure in the USA, but their biogeochemical processes and impacts are often overlooked. Here, we assessed the potential impact of stormwater ponds on regional carbon cycling by coupling carbon burial rates and fluxes of carbon dioxide and methane gases in five sites over an age gradient of 14–34 years. Carbon burial increased logarithmically with site age, ranging from 22 to 217 g carbon m⁻² y⁻¹, while, median floating chamber diffusive gas fluxes were 1290 g carbon dioxide m⁻² y⁻¹ and 5 g methane m⁻² y⁻¹, which, when combined as carbon dioxide equivalents, equates to 2900 g carbon dioxide eq m⁻² y⁻¹. Comparing carbon burial to gas flux reveals that stormwater ponds can be net carbon sources and need to be considered for regional and global carbon models.
Small lakes and ponds (<0.01 km²) have emerged as important water bodies for the processing and transportation of carbon (C)⁴⁻⁵. Of an estimated 0.583 Pg C yr⁻¹ emitted from lakes and ponds globally, ~24% is from waterbodies <0.01 km² despite only accounting for ~15% of total pond and lake surface area. Similarly, small lentic artificial freshwaters (e.g., ponds, reservoirs, ditches, infiltration basins) are estimated to store large quantities of organic carbon (OC) in sediments globally, more than natural ecosystems and up to 4-times as much as the world’s oceans annually⁶.

In urban ecosystems, permanently wet stormwater ponds (SWPs) are suspected to play a large role in C processing due to characteristics similar to those of other small lentic systems. For example, both SWPs and natural ponds have individually small sizes (commonly <0.01 km²)⁵⁻⁶, shallow water columns, high sediment and perimeter to water volume ratios, and frequent mixing. These pond features contribute to rapid accumulation of C, high internal productivity during the growing season, frequent heterotrophy, and high C gas evasion, all to a disproportionately high terrestrial C loads, SWPs can be highly internally productive, and intensive water quality and sediment management practices (i.e., dredging, algaecide application).

SWPs are anthropized ecosystems constructed to provide specific ecosystem services such as flood management and pollutant removal by capturing stormwater runoff from the surrounding watershed via drainage infrastructure (e.g., gutters, curbs, roads, drains). This runoff can carry high loads of biologically significant materials such as dissolved and particulate organic matter with varying lability, C, and essential nutrients (nitrogen (N), phosphorus (P))⁶,⁷. Further, SWPs are ubiquitous across urban landscapes, representing the most common stormwater control measure in the U.S⁸. In Florida, they are estimated to cover an area of 672 km²⁵ and are prolific in eastern U.S coastal communities⁹,¹⁰. Despite this widespread presence, SWPs are overlooked and underestimated and little is known about their role in regional C cycling or their net benefit to society (services vs. disservices). Still, the growing body of research on artificial waters and urban ponds has shown that they are capable of emitting substantial quantities of carbon dioxide (CO₂) and methane (CH₄) greenhouse gases (GHGs) and potentially more so than natural ecosystems, up to 2.5x more compared to natural ponds <10,000 m² in area¹¹, despite high C burial rates¹².

As urbanization and SWPs replace natural uplands and downstream receiving waters (e.g., wetlands, streams, ponds), changing the movement of water, solutes, and energy in the landscape, there is an increasing need to understand the capacity of SWPs to store C in sediments and exchange C gas with the atmosphere. Studies of urban ponds or impoundments around the globe have reported burial rates ranging from 8.7 g C m⁻² y⁻¹ in urban SWPs (South Carolina, USA)⁹ to 2120 g organic C (OC) m⁻² y⁻¹ in eutrophic impoundments¹³. In addition to receiving high terrestrial C loads, SWPs can be highly internally productive, with a significant proportion of the OC pool being autochthonously produced⁴,⁶,¹⁴. Despite high rates of C accumulation in sediments, small ponds are typically net C sources, driven by CO₂ and CH₄ emission to the atmosphere. For example, in terms of CO₂ equivalents, 96% of artificial ponds and ditches from seven countries were net sources of GHGs to the atmosphere¹¹. Similarly, an investigation of 77 small agricultural impoundments (Victoria, Australia) showed that they accounted for 3.4x higher CO₂ equivalent fluxes than temperate reservoirs while having only 0.94 times the comparative area¹⁵. Mean CO₂ equivalent fluxes have been reported as high as 9.2 g CO₂ eq m⁻² d⁻¹ in SWPs, with 38% from diffusive CO₂ fluxes and 62% diffusive + ebullitive CH₄ fluxes (The Netherlands)¹⁶. However, C fluxes from small ponds are highly variable¹⁷ and urban ponds may be particularly heterogeneous due to variation in pond management, urban infrastructure, and watershed activities. While many studies have separately estimated GHG emissions or C burial in artificial ponds, rarely have studies combined both to compare net C budgets (see ref. ¹⁶,¹⁸), which can allude to their net benefit to society. Understanding the drivers of C source/sink dynamics in SWPs can enhance our ability to manage these ecosystems to reduce the negative impacts stormwater control measures might pose on downstream ecosystems and to regional climate change.

Ponds and lakes can undergo successional development as sediments accumulate over time, which can be associated with successional changes to pond biogeochemistry. For example, sediment organic matter, summer nitrogen limitation, sediment oxygen demand, and net N₂-fixation rates have been shown to increase with increasing SWP age¹⁹. Similarly, C burial rates were considerably lower in new (3 year old) artificial ponds compared to mature (18–20 year old) artificial ponds¹². Ecosystem age has also recently been found to be a strong predictor in empirical models (G-res tool) for the diffusive emission of C gases from reservoirs, especially at higher temperatures.²⁰

Observing the temporal C source/sink dynamics of SWPs and patterns over a pond age gradient can provide insight into how these inland urban waters behave over short (annual) and long (decadal) timelines. In this study, we quantify rates of C burial as well as CO₂ and CH₄ gas fluxes in five SWPs spanning a 20-year age gradient (14, 15, 18, 23, and 34 years old; ponds identified based on pond age hereafter) in southwestern Florida. We measured gas fluxes using a floating chamber method over 10 months (June 2019 – March 2020) and compared fluxes against pond age, measured environmental factors, and C burial rates to identify potential drivers of gas fluxes and estimate the net C flux of urban SWPs. Our results show that subtropical urban SWPs can sequester large quantities of C in their sediments. As ponds matured in age, C burial increased and C emissions (CO₂-C+C+CH₄-C) decreased, suggesting that ponds may shift toward a net C sink as they age.

Results and discussion

Carbon burial rates increased with pond age. Areal and annual C burial rates were calculated for four to six sediment cores extracted per pond in June 2019 using total OC (TOC), dry weight, and piston corer dimensions. Mean TOC content in whole cores of ponds ranged from 1.7 to 26.5% (Table 1 and Supplementary Fig. 1). The areal OC stock ranged from 306 ± 8.2 g OC m⁻² in Pond 14 (14-year-old pond) to 7380 ± 56.1 g OC m⁻² in Pond 34 (34-year-old pond), increasing linearly with increasing pond age (R² = 0.99, p < 0.001, Fig. 1a and Table 1). Scaling this OC stock by pond age (assuming constant burial rates over time) yields estimated annual burial rates ranging from 218 ± 7.5 g OC m⁻² y⁻¹ in Pond 14 to 217 ± 50.2 g OC m⁻² y⁻¹ in Pond 34, with annual rates increasing logarithmically with pond age (R² = 0.93, p < 0.01, Fig. 1b). These OC burial rates, especially in the 23- and 34-year-old ponds (141 and 217 g OC m⁻² y⁻¹, respectively), are comparable to or exceed previously reported rates from a range of aquatic ecosystems (Supplementary Table 1) including critical blue C sinks such as mangrove forests, salt marshes, and seagrasses.²¹ The OC burial values reported here are also similar to reported rates in other SWPs and constructed systems⁹,¹⁶,²²,²³ but remain below results for some impoundments around the globe (Supplementary Table 1).

The relationship between C burial and pond age suggests that internal sediment and/or aquatic properties allow ponds to become increasingly stable environments for C deposition and
storage. Burial rates were not related to pond morphology (perimeter, area, depth, area:perimeter). SWPs are unique ecosystems that receive a dramatic quantity of sediment from impervious and piped catchments. It is possible that over time and as more particulate material is added to the sediment surface, sediment thickness and anaerobic conditions beneath newly deposited material increase, slowing down the rate of OM decomposition. A higher degree of recalcitrant OM can also be left behind, increasing the pool of stable OM more resistant to degradation. Further, the addition of SWPs to newly developed

| Lat, long | Pond 14 | Pond 15 | Pond 18 | Pond 23 | Pond 34 |
|-----------|---------|---------|---------|---------|---------|
|           | 27.43, -82.39 | 27.43, -82.40 | 27.43, -82.41 | 27.43, -82.42 | 27.43, -82.44 |
| Perimeter (m) | 307 | 374 | 707 | 469 | 477 |
| Area (m²) | 6220 | 8850 | 18,800 | 7900 | 13,100 |
| Depth (m)a | 7.2 | 4.7 | 5.6 | 3.4 | 1.8 |
| Total nitrogen (mg L⁻¹) | 1.2 ± 0.07 | 0.97 ± 0.03 | 1.1 ± 0.04 | 0.91 ± 0.03 | 1.3 ± 0.06 |
| Dissolved organic carbon (mg L⁻¹)b | 11.2 ± 0.5 | 8.1 ± 0.4 | 9.3 ± 0.4 | 12.1 ± 0.5 | 16.8 ± 0.9 |
| Surface dissolved oxygen (%) | 66.0 ± 4.4 | 91.6 ± 2.9 | 101.7 ± 3.0 | 105.0 ± 2.7 | 106.0 ± 8.1 |
| Benthic dissolved oxygen (%) | 6.7 ± 1.1 | 39.9 ± 4.0 | 35.1 ± 5.1 | 32.7 ± 3.8 | 35.2 ± 4.2 |
| Surface pH | 7.39 ± 0.04 | 7.46 ± 0.06 | 7.67 ± 0.06 | 8.33 ± 0.07 | 8.33 ± 0.15 |
| Benthic pH | 6.97 ± 0.03 | 6.99 ± 0.04 | 7.06 ± 0.05 | 7.43 ± 0.05 | 7.74 ± 0.12 |
| Core total organic C | 6.9 ± 4.0 | 9.3 ± 2.7 | 6.4 ± 0.7 | 4.8 ± 0.8 | 13.7 ± 2.2 |
| Core dry weight (g) | 20.4 ± 10.0 | 28.1 ± 12.0 | 79.7 ± 11.0 | 191 ± 57.6 | 116 ± 17.6 |
| Areal OC stock (g OC m⁻²) | 306 ± 115 | 928 ± 402 | 2,270 ± 572 | 3,230 ± 245 | 7,380 ± 1,910 |
| OC burial rate (g OC m⁻² yr⁻¹) | 21.8 ± 7.6 | 61.9 ± 24.2 | 126 ± 29.1 | 141 ± 9.8 | 217 ± 50.3 |
| CO₂ flux (mg m⁻² d⁻¹) | 13,300 ± 2462 | 8700 ± 1530 | 4040 ± 1020 | 1240 ± 767 | 2460 ± 1750 |
| CH₄ flux (mg m⁻² d⁻¹) | 129 ± 82.2 | 29.3 ± 4.1 | 21.5 ± 11.1 | 14.6 ± 3.9 | 44.3 ± 15.9 |

*a Mean depth of center most location of pond over the study period.

*b Mean values from sampling events between October 2019 and March 2020.

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Fig. 1 Sediment organic carbon stocks and burial rates in stormwater ponds. Organic carbon stock (g OC m⁻², a) and the estimated rate of burial (g OC m⁻² yr⁻¹, b) increase with pond age. Individual core values (n = 4–6 per pond) are plotted as small gray points and the larger black points represent the pond mean, which was used for regressions. Both linear and logarithmic regressions are statistically significant (p < 0.01).
lakes is associated with the addition of new landscaping vegetation to the urban watershed, such as trees which can increasingly contribute leaf litter to ponds as they mature over time, suggesting that allochthonous C inputs may increase concurrent with in-pond conditions favorable for C burial. Natural factors such as precipitation, length of the growing season, and littoral vegetation also affect C accumulation in urban SWPs. In addition to allochthonous inputs, SWPs can support highly productive algal communities and phytoplankton turnover represent an internal OC input. Littoral vegetation may provide a similar autochthonous OC source, but sites in this study contained no littoral vegetation and also had not been dredged (a common management practice). Despite the apparent relationship between age and C burial, our sample size is small, and this relationship may be spurious. A prior study found no relationship between sediment OC and age in an analysis of 45 aquaculture ponds, but the aquaculture activities may have offset natural successional dynamics. Furthermore, an experimental study of small ponds found that new ponds (3 years old) exhibit significantly less C burial in sediments than mature ponds. Nonetheless, in the process of preventing urban sediment from moving into downstream waters, constructed SWPs have emerged as significant reservoirs of sediment C, as shown here and in other anthropized ecosystems (reservoirs, impoundments, SWPs) or small ponds. In Florida, there are at least 76,000 SWPs. Given the role of SWPs in storing sediment C and their ubiquity on the landscape, these anthropized ecosystems represent a large pool of C in urban aquatic sediments. Using the mean OC burial rates in the youngest and oldest pond combined with the estimated surface area covered by SWPs in Florida (627 km²), we estimate that SWPs in Florida (627 km²) emit between 52 and 204 Tg CO₂-equivalents annually (or 12–50 Tg C annually).

Ponds were a source of CO₂ over the vast majority of the study period excluding the two oldest ponds, which reported negative flux values during 6 of 17 sampling events each (35%), reaching as low as −507 mg CO₂-C m⁻² d⁻¹ (Pond 23) and −3730 g CO₂-C m⁻² d⁻¹ (Pond 34, Supplementary Fig. 2). Ponds 15 and 18 only exhibited negative CO₂ gas flux one time each. These results suggest that at some points in the year, older ponds may switch from net heterotrophy to net autotrophy. A gradual decrease in mean CO₂ flux with increasing pond age can be seen in Supplementary Fig. 3, but the same pattern is not evident with CH₄ fluxes. Similarly, a survey on hydroelectric reservoirs around the globe (n = 85) found that CO₂ and CH₄ emissions were inversely related to reservoir age, a pattern that was also found on a temporal study of a single hydroelectric reservoir (CO₂ only). The CO₂ fluxes from this study were similar to or higher than other aquatic ecosystems, CO₂ fluxes were similar or higher, whereas CH₄ fluxes were typically lower than other urban studies (Supplementary Table 2). Eubullition is considered an important pathway for methane emission in lentic ecosystems with anoxic sediments. Because we did not measure ebulition our values for overall CH₄ flux are likely underestimated. Small patches of bubbles were occasionally captured underneath our flux chamber, causing the internal concentration of CH₄ to spike to ~130,000 ppb. For reference, the global mean atmospheric CH₄ concentration for 2019 was 1870 ppb. There was not a similar spike in CO₂ concentrations when bubbles were captured.

Linear mixed-effects models (LMM) were used to estimate the relationship between gas fluxes and environmental variables measured in this study (Supplementary Table 3). For both CO₂ and CH₄ models, site (five levels) and sampling date (seventeen levels) were set as random effects and pond age was a similar fixed effect. Including the variation from random effects, 68% of the variation in CO₂ fluxes was explained by inverse relationships with surface % DO, pH, pond age, and the interaction between %DO and pH. In addition to the negative relationship between CO₂ and pH and the observed decrease in mean CO₂ flux with age (Supplementary Fig. 3), an associated observation was the slight increase in mean pH with pond age, up to 10.4 in Pond 34 (Fig. 2 and Supplementary Fig. 4). Surface %DO and pH were strongly
related to CO₂ in the LMM, and when combined can be considered indicators of primary production. Photosynthetic fixation of CO₂ from the water column results in an increase in pH and DO. Concurrently, the chemical form of CO₂ in the water column (and its concentration) is controlled by pH, speciating into carbonic acid, bicarbonate, and carbonate at higher pH values. According to the kinetics of this reaction, free CO₂ in freshwater becomes depleted at pH > 8.3. Of the 14 observations of negative fluxes, all but one occurred above pH 8.3 (Fig. 2). So, while primary production contributes to a decrease in CO₂ emissions, the increased pH can exert a compounding effect, rendering high pH ponds and lakes to be weak CO₂ sources, similar to observations in in other urban ponds13, saline lakes14, and in agricultural reservoirs15. Other abiotic factors may contribute to an increase in pH. Florida is known to have shallow carbonate bedrock which, coupled with the weathering of urban concrete infrastructure (calcium hydroxide, portlandite, calcium silicate hydrates), can contribute carbonates and calcium salts, increasing alkalinity in aquatic ecosystems16,17. Additionally, in high CaCO₃ systems, inputs of CO₂ can contribute to increased alkalinity over time via calcium release. Thus, ponds have the potential to become CO₂ sinks over time as a result of increasing eutrophication and accumulation of natural- and anthropogenically sourced alkaline ions increasing pH.

Compared to CO₂, a lower amount of the variation in diffusive CH₄ flux was explained by variables in this study. With accounted variance from random effects, 51% of CH₄ flux variation was explained by an inverse relationship to benthic %DO and positive relationships to benthic temperature and log-transformed CO₂ fluxes (Supplementary Table 3). There was no relationship between CH₄ flux and site age. The bottom of most pond water columns remained hypoxic throughout the study period with 65% of the data below 3.0 mg/L DO, creating conditions beneficial for anaerobic methanogenesis, which can produce CH₄ alone (by hydrogenotrophic methanogens) or CH₄ and CO₂ (by acetoclastic and methylotrophic methanogens). The positive relationship with water column DO was also indicated during the extreme methanogenesis during DO depletion (0–5% DO through the water column) in Pond 14. Finally, sedimentary methanogens typically become more active at higher temperatures as a result of increasing use of higher redox potential electron acceptors as well as a higher input of primary production-derived substrates that fuel their metabolic activity36,37.

Younger ponds are net sources of C to the atmosphere. We estimated a C balance for each pond by comparing median organic C burial to median C gas flux (CO₂-C + CH₄-C) (Fig. 3). The combination of C gas flux (C export) and burial (C storage) reveals that three of the five ponds were, on average, net sources of C to the atmosphere. However, the median burial values fall within the 5th–95th quantile range for C efflux in Pond 23 and 34, the net burial ponds (Fig. 3 and Table 1). It should be noted that these estimates compare rate measurements determined at separate time scales (seconds vs multi-decadal). As pond age increased, the difference between C gas efflux and C storage in sediments decreased, implying that ponds may become more C-neutral as they age. The degree of this net benefit in older ponds may change if methane ebullition estimates had been included, as others have found that urban pond ebullition accounted for up to 50% of total CH₄ emission18. Previous studies have identified increasing CH₄ emission with ecosystem age18 and increasing trophic level39, which can be associated with pond succession, converting ponds into net sources in terms of CO₂-equivalents due to increasing CH₄ contributions. Regardless of pond age, this study provides evidence that SWPs, which are constructed to provide ecosystem services, may be providing a disservice by acting as net contributors to GHG emissions, despite storing substantial quantities of C in sediments. Other studies of small ponds support this assertion. For example, both Peacock et al. (2019) and Ollivier et al. (2019) observed release of GHGs from urban and small agricultural ponds at rates high enough to call for their inclusion in global carbon budgets.

C burial and C gas fluxes have been compared in other anthropized ecosystems. In a 20-year old urban pond (the Netherlands) the annual flux of CO₂-C and CH₄-C combined, 391 g C m⁻² yr⁻¹, was substantially higher than the pond’s rate of burial at 29 g C m⁻² yr⁻¹. The aforementioned pond exhibited a lower burial rate and annual emission than Pond 18 (and younger sites) in our study (burial = 126 g C m⁻² yr⁻¹, emission = 429 g C m⁻² yr⁻¹). In a 4-year-old (time since flooding) hydroelectric reservoir, daily estimates of flux and burial were 2.3 g C m⁻² d⁻¹ emission vs. 0.1 g C m⁻² d⁻¹ burial18.

Our results suggest that after urban ponds are constructed, they emit large proportions of C inputs from the landscape and potentially increase storage efficiency over time. We suspect that when C enters urban SWPs, less C is buried in younger ponds compared to older ponds, as seen in burial studies on aging artificial ponds12. Younger ponds may exhibit higher C mineralization rates which simultaneously increases emissions while reducing burial. However, identified trends can be drastically altered depending on how ponds are managed, suggesting management actions can be used to modulate pond C storage and C emission. For example, increasing urban pond depth is associated to reduced CH₄ release29, likely due to increased water column contributions to organic matter oxidation prior to sediment settling and CH₄ oxidation, and ponds or lakes larger in area exhibit lower C emissions per unit area compared to smaller ponds2. Other suggested properties for anthropized ecosystem C management include water column stratification, water retention time, water source inputs, and landscape position34. It is important to note that drivers of C emission can differ between natural and anthropized ecosystems such that CH₄
emissions from reservoirs globally were correlated to productivity and fluxes increased with increasing area\(^40\). In contrast, natural lakes were better explained by morphometry and fluxes decreased with increasing area\(^40\).

Methods for pond maintenance include sediment dredging and muck removal, algacide application, aeration systems (fountains, bubblers), fish stocking, and vegetation management\(^41\). Although ponds included in this study have not been dredged, dredging could reset the ‘effective pond age’. Dredging may not effectively reset a pond, however, as sediment resuspension that occurs in the process has caused lakes to relapse into a eutrophic state because of the reintroduced availability of sediment nutrients to the water column\(^42\). The application of aquatic pesticides (i.e., chemical treatments for algal blooms and invasive macrophytes) in SWPs is likely to impact C cycling such that substantial amounts of labile OC are rapidly contributed to the sediment surface, influencing sediment oxygen demand and the respiration of CO\(_2\) and CH\(_4\). Fish are added to ponds for recreational use as well as mosquito and algal management, and can have varying effects on CO\(_2\) equivalents fluxes. For example, by grazing on zooplankton that consume both phytoplankton and CH\(_4\)-oxidizing bacteria, fish can elicit a trophic cascade that causes an increase in algal productivity and CH\(_4\) flux and reduced CO\(_2\) flux\(^39\). However, fish can also graze on predators to CH\(_4\)-oxidizing bacteria, decreasing CH\(_4\) flux\(^43\). The presence of vegetation may also be beneficial for reducing GHG emission, as seen in previous studies\(^44\) and is responsible for increasing C burial rates\(^22\). Finally, aeration systems are commonly used to prevent anoxia and could therefore reduce CH\(_4\) emissions, as shown in experimental studies on hypolimnion DO manipulations\(^45\).

The number of small constructed ponds was found to increase 18-fold from 1937 to 2005 in the Brandywine watershed of central Pennsylvania and northern Delaware, and land use change associated with urbanization will likely continue to increase the numbers of small constructed ponds regionally and globally\(^46\). As ubiquitous anthropized aquatic ecosystems such as SWPs continue to be constructed in urban landscapes and have emerged as significant contributors to GHGs and C storage, more work is required to assess their net C footprint, patterns with ecosystem age over a larger number of sites, and biogeochemical responses to management strategies. An improved knowledge of how management actions interact with natural ecological processes is critical for understanding the role of SWPs and other small, anthropogenic aquatic ecosystems in the global C cycle.

### Sediment OC burial

C accumulation rates were estimated in five ponds over an age gradient to establish C storage within a pond and to identify differences in burial over time. Six sediment cores were collected from each pond. Cores were taken in evenly distributed areas in the center region of the pond away from littoral shelves and banks. Cores were extracted during May and June 2019 using a modified piston corer with a clear sleeve 89.7 cm in height and opening area of 22.88 cm\(^2\). The layer of deposited material is defined as all organic sediment found above the distinct pond sand-fill material, which was visually identified in each core through the clear coring sleeves. Due to the difficulty of coring from sand underlain sediments (falling apart before they could be fully collected), some cores were compromised and so replicates range from four to six cores per pond. This mud-sand contact defines ‘time zero’ when the pond was constructed, and material began to deposit. We assume that all mud present has been deposited since the time of construction. Once cores were extracted, all deposited sediment material, excluding sand fill, was extruded and collected. The collected sediment of a core was then homogenized. Sediments were frozen for 24 h and freeze dried to remove moisture. Once dried, each core was weighed for dry weight. Three samples of dried sediments were analyzed for total organic carbon (TOC). The third core sample was preserved to represent the amount of organic C found in each core. TOC was quantified for each core by the UF Wetland Biogeochemistry Lab using a Shimadzu TOC5050A Total Organic Carbon Analyzer. Using a catalytic combustion oxidation method, sediment samples are combusted and analyzed for total carbon (TC) and total inorganic carbon (TIC); TOC is calculated as the difference between the two. Results are reported as g OC per 100 g dry sediment concentrations, or % OC.

Organic carbon stock (g OC m\(^{-2}\)) and burial rates (g OC m\(^{-2} y^{-1}\)) were calculated and represented using the following equations. Below the second considers the cumulative pond age to estimate a burial rate of g OC m\(^{-2} y^{-1}\). (Eq. 2). Equation 1 calculated the mean C content per core (g OC per 100 g dry sediment) times the core dry weight, divided by the sleeve opening area (m\(^2\)) and aged by the pond age (years) as a multiplier in the denominator.

\[
\text{Burial rate (g OC m}^{-2}y^{-1}\text{)} = \frac{\text{TOC} \times \text{Core dry weight (g)}}{\text{Sleeve opening area (m}^2\text{)} \times \text{Pond Age (yrs)}}
\]

Floating chamber fluxes of CO\(_2\) and CH\(_4\)

To examine fluxes of CO\(_2\) and CH\(_4\) from SWPs to the atmosphere we employed a floating chamber method where the vertical rate of flux from the air-water interface was estimated as the change in gas concentration within the chamber over time and cross-sectional area\(^\text{37}\). The use of floating chambers is increasingly employed in gas flux studies for its affordability and ease of use, which involves short deployments, and simple operation. Upon deployment, results are only representative of single measurement points and can be used for qualitative assessments. CO\(_2\) and CH\(_4\) fluxes were measured biweekly from June 2019 to May 2020 mid-day between 10:00 a.m. and 2:00 p.m. Gas measurements were collected using a LI-COR Smart Chamber as the enclosed floating component attached to a LI-COR LI-7810 CO\(_2/\)CH\(_4/\)H\(_2\)O Trace Gas analyzer that utilizes a non-dispersive infrared method. Detection ranges is 0 to 100 ppm CH\(_4\) (0.25ppb precision) and 0 to 10,000 ppm CO\(_2\) (1.5 ppm precision).

The Smart Chamber (floating chamber) is equipped with an air temperature thermostat (accuracy ± 0.5 °C between 0 and 70 °C) and a barometric pressure sensor (accuracy ± 0.4 KPa between 50 and 110 KPa). It has a volume of 4244.1 cm\(^3\) and attaches to a polyethylene collar 11.43 cm in height with an opening area of 317.8 cm\(^2\) exposed to the water surface. The collar is fixed onto a buoyant Styrofoam lid with a hole cut in the center, allowing ~14 cm of the collar to extend into the water. The submerged collar walls improve flux accuracy, as the absence of submerged wall extension has reported fluxes 3-5 times greater than when walls are present\(^\text{38}\).

Ventilation to ambient air is required for the chamber pressure equilibrate with ambient air pressure. The chamber features patented pressure vent technology, characterized as a radially symmetrical vent that allows the chamber to maintain a pressure inside that is equal to that of the ambient air pressure outside under low and high wind conditions regardless of wind direction. This improves the ability of measurements to represent fluxes under natural conditions. Additionally, to account for decreased gas diffusion gradients upon chamber enclosure, an
exponential function is used to determine initial slope conditions as soon as the chamber is closed to represent fluxes under ambient conditions. When ebullitive bubbling of sediment gases were caught under the chamber during a measurement, the chamber was reset and measurements re-done to maintain consistency in chamber exponential or linear slopes used to calculate fluxes. Supplementary Fig. 5 shows R² values for CO₂ and CH₄ chamber flux measurements. For CH₄ measurements, 82% of individual measurements reported R² of 0.95 and above, and only two observations were below 0.50. CO₂ slope R² were more variable, where only 66% of values were above 0.80. We kept all measurements with significant exponential or linear regressions (p < 0.05), removing thirteen non-significant CO₂ measurements and ten for CH₄.

On each sampling date gas fluxes were measured at three locations within the pond. The mean of these values was used to represent whole pond fluxes. At each point, the chamber was allowed to rest on the water’s surface for a few minutes before taking measurements. Measurements in the closed chamber were taken for 200–300 s. As gas accumulates in the chamber the sample air is mixed by both the hemispherical shape of the chamber and the position of the inlet and outlet tubing. Sample air is pumped into the trace gas analyzer by a 1.2-m-long tube and detected for CO₂, CH₄, and H₂O by optical feedback – cavity-enhanced absorption spectroscopy. Gas accumulation within a chamber often exhibits exponential increases, therefore an exponential fit of the concentration measured over time is used to determine slope. We compared the R² of exponential fits to linear fits in case there were occasions where a linear fit was more appropriate. For CO₂, there were thirteen out of 236 observations where a linear fit was used and ten out of 239 for CH₄. The equation used here to estimate flux rates is a slight modification of that by Duc et al. (2013)⁵⁹ (Eq. 3):

\[ F = \frac{dX}{dRTA} \]

Where \( F \) represents the rate of gas flux at the air-water interface (µmol [C gas] m⁻² s⁻¹), \( X \) is the slope of the exponential fit to the change in concentration within the chamber (µmol [C gas] m⁻² s⁻¹) from the moment of closure (µmol m⁻³ s⁻¹), \( T \) is ambient pressure (atm), \( V \) is the volume of the Smart Chamber (mL), \( A \) is the area of chamber coverage (m²), \( R \) the universal gas constant (82.056 mL atm mol⁻¹ K⁻¹), and \( T \) is ambient temperature (K). \( b \) is the molecular weight of either CO₂ or CH₄ (µmol) and converts µmol m⁻² s⁻¹ to g m⁻² s⁻¹.

**Statistics** All statistical analyses were conducted in the statistical software RStudio (version 1.2.5) and R (version 3.6.4). Significance in all analyses were results that reported p-values below 0.05. Because of the large sample size in data observations, we relied on histograms and QQ plots to determine univariate normality. In the case of a variable containing non-normally distributed data, log transformation was applied. In the case of CH₄, normality was achieved after log-transformation. The same was not true for CO₂, which was left untransformed for further statistical analyses. Multicollinearity was assessed by computing pairwise Pearson correlation coefficients between specific outcome variables using the cor.test() or cor.mat() function and homogeneity of variance was assessed using Levene’s test (levene.test() function), all from the R package car.⁶⁰

To assess the existence of trends in C burial with ponds of differing ages, a criterion (AICc). Surface and benthic DO were correlated to CH₄ fluxes. Multicollinearity was assessed using Levene’s test against one another and comparing linear models with CH₄ flux that did and did not include an interaction between surface and benthic % DO.

**Data availability** Data collected for this study is available as .csv files and have been submitted to The Institutional Repository at the University of Florida (IR@UF) as of 7 July 2021 and can be viewed at the following link: https://ufdc.ufl.edu/IR00011476/00001.

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**Author contributions**

correspondent – A.H.G., methodology – all authors, data curation – A.H.G., software, validation, and visualization – A.H.G., formal analysis – A.H.G., writing original draft – A.H.G. and M.G.L., writing, reviewing, and editing – all authors, project administration and supervision – M.G.L., funding acquisition – M.G.L. A.H.G designed the study, collected data, conducted formal data analysis and data visualization, and led the writing of the manuscript. M.G.L contributed to data interpretation, writing and editing of the manuscript, and project supervision. A.J.R contributed to data analysis and interpretation, and manuscript editing. J.D.H contributed to data analysis and interpretation. J.M.S contributed to methodology and data interpretation.

**Competing interests**

All authors declare no competing interests.

**Additional information**

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