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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.

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Small lakes and ponds (<0.01 km^2) have emerged as important water bodies for the processing and transportation of carbon (C)[1–4]. Of an estimated 0.583 Pg C yr^{-1} emitted from lakes and ponds globally, ~24% is from waterbodies <0.01 km^2 despite only accounting for ~15% of total pond and lake surface area[2]. 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 annually[1].

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 km^2 [2,5]), 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 higher degree than larger water bodies[1–4]. However, SWPs may differ from natural ponds in C cycling properties due to their differences in hydrology (engineered in and outflow structures), elevated allochthonous resource inputs from urban landscapes, 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.S[8]. In Florida, they are estimated to cover an area of 672 km^2 [5] and are prolific in eastern U.S coastal communities[9,10]. Despite this widespread presence, SWPs are overlooked and under-researched 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 (CO_2) and methane (CH_4) greenhouse gases (GHGs) and potentially more so than natural ecosystems, up to 2.5x more compared to natural ponds <10,000 m^2 in area[11], despite high C burial rates[12].

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 impoundments[13]. In addition to receiving high terrestrial C loads, SWPs can be highly internally productive, with a significant proportion of the OC pool being autochthonously produced[4,6,14]. Despite high rates of C accumulation in sediments, small ponds are typically net C sources, driven by CO_2 and CH_4 emission to the atmosphere. For example, in terms of CO_2 equivalents, 96% of artificial ponds and ditches from seven countries were net sources of GHGs to the atmosphere[11]. Similarly, an investigation of 77 small agricultural impoundments (Victoria, Australia) showed that they accounted for 3.4x higher CO_2 equivalent fluxes than temperate reservoirs while having only 0.94 times the comparative area[15]. Mean CO_2 equivalent fluxes have been reported as high as 9.2 g CO_2 eq m^{-2} d^{-1} in SWPs, with 38% from diffusive CO_2 fluxes and 62% diffusive + ebullitive CH_4 fluxes (The Netherlands)[16]. However, C fluxes from small ponds are highly variable[17] and urban ponds may be particularly heterogenous 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 N_2-fixation rates have been shown to increase with increasing SWP age[19]. Similarly, C burial rates were considerably lower in new (3 year old) artificial ponds compared to mature (18–20 year old) artificial ponds[12]. 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 CO_2 and CH_4 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 (CO_2–C + CH_4–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 ($R^2 = 0.99$, $p < 0.001$, 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 21.8 ± 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 ($R^2 = 0.93$, $p < 0.01$, 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 systems[9,16,22,23], 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

Table 1 Pond morphology, water chemistry, and C burial data (mean ± standard error).

Lat, long	Pond 14 27.43, -82.39	Pond 15 27.43, -82.40	Pond 18 27.42, -82.41	Pond 23 27.43, -82.42	Pond 34 27.43, -82.44
Perimeter (m)	307	374	707	469	477
Area (m^2)	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
CO_2 flux ($mg\,m^{-2}\,d^{-1}$)	13,300 ± 2462	8700 ± 1530	4040 ± 1020	1240 ± 767	2460 ± 1750
CH_4 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.

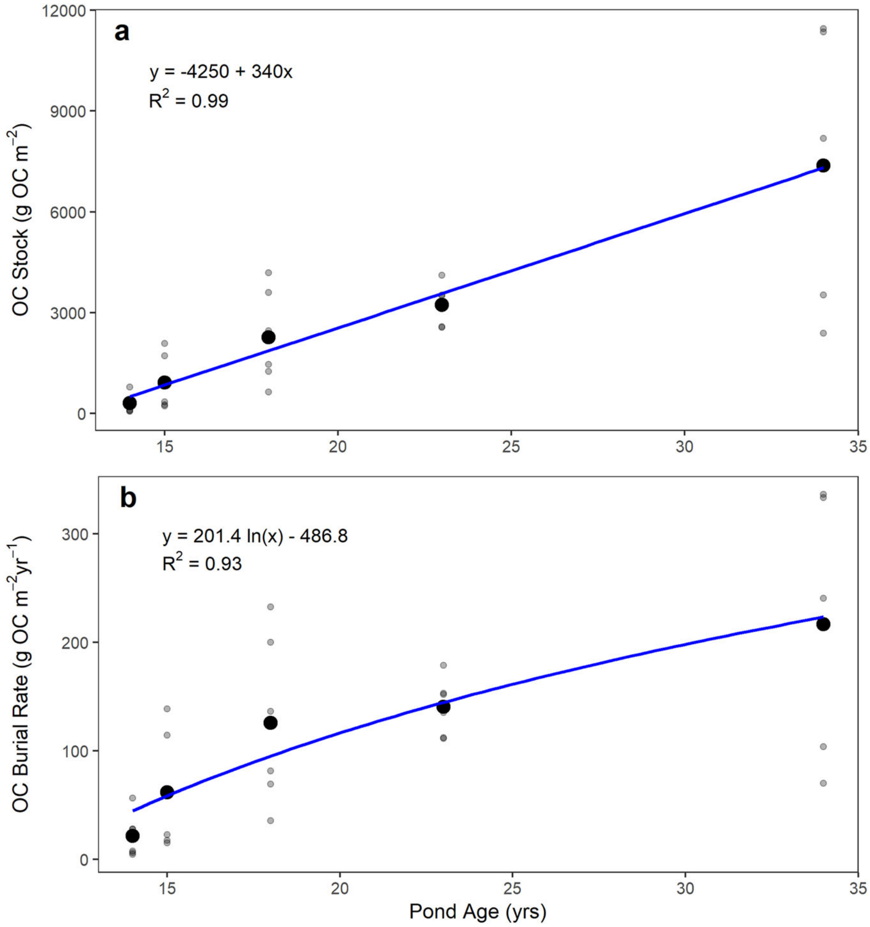

Fig. 1 Sediment organic carbon stocks and burial rates in stormwater ponds. Organic carbon stock ($g\,OC\,m^{-2}$, **a**) and the estimated rate of burial ($g\,OC\,m^{-2}\,y^{-1}$, **b**) increase with pond age. Individual core values ($n = 4$–6 per pond) are plotted as small gray points and the larger black points represent the pond mean, which was used for regressions. Both linear and logarithmic regressions are statistically significant ($p < 0.01$).

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 landscapes 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[9,22]. In addition to allochthonous inputs, SWPs can support highly productive algal communities and phytoplankton turnover represent an internal OC input[24]. 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[25], 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[12]. 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[3,9,17,22,23].

In Florida, there are at least 76,000 SWPs[5]. 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 km^2)[5], we estimate that SWPs in Florida can store between 0.2 and 4.6 Tg C annually. The upper end of this estimate would be enough to offset 1.4% of the global CO_2 evasion from inland lakes and reservoirs[26].

Carbon gas fluxes are controlled by physical and chemical factors within ponds. The mean floating chamber flux of CO_2 over the study period (biweekly measurements June 2019 to March 2020) was 5940 ± 857 mg CO_2 m^{-2} d^{11} with a range of −13600 to 34300 mg CO_2 m^{-2} d^{-1} (Supplementary Fig. 2). CH_4 fluxes were consistently positive but more variable and lower than CO_2 fluxes with a mean of 47.8 ± 17.1 mg CH_4 m^{-2} d^{-1} and range of 0.12–1400 mg CH_4 m^{-2} d^{-1} (Supplementary Fig. 3). This maximum CH_4 flux was accompanied by an unexplained water quality change that coincided with cloudy white water and DO levels depressed to 0–5% throughout the water column. Using median values, annual fluxes equate to 1290 g CO_2 m^{-2} yr^{-1} and 5.0 g CH_4 m^{-2} yr^{-1}. The median estimate for CH_4 is far below the 2019 Intergovernmental Panel on Climate Change estimate for manmade freshwater ponds (18.3 g CH_4 m^{-2} yr^{-1}), although the mean CH_4 flux in this study is on par at 17.4 g CH_4 m^{-2} yr^{-1} [27]. On a C mass basis, the daily mean flux of C gases combined (CO_2-C + CH_4-C) equates to 1636 mg C m^{-2} d^{-1} (CO_2 making up 98%), and an annual flux of 324 g C m^{-2} yr^{-1} (using the median). Diffusive CH_4 contributed a small percentage of total pond C fluxes in this study. Converting CH_4 to CO_2 equivalents with a sustained-flux global warming potential of 45[28] and summing with CO_2 fluxes results in a mean of 8090 mg CO_2 eq m^{-2} d^{-1}, of which CH_4 makes up 27%, and annual median of 1660 g CO_2 eq m^{-2} yr^{-1}. This is a smaller fraction of CH_4 contribution to overall CO_2 equivalent fluxes compared to other urban SWPs such as 94% of 28.2 g CO_2 eq m^{-2} d^{-1} (excluding ebullitive emission)[29] and 62% of 9.2 g CO_2 eq m^{-2} d^{-1} (including ebullitive emissions)[16]. Using the median CO_2-equivalent fluxes in the oldest and youngest pond combined with the estimated surface area covered by SWPs in Florida (627 km^2)[5], we estimate that SWPs in Florida can emit between 52 and 204 Tg CO_2-equivalents annually (or 12–50 Tg C annually).

Ponds were a source of CO_2 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 CO_2-C m^{-2} d^{-1} (Pond 23) and −3730 g CO_2-C m^{-2} d^{-1} (Pond 34, Supplementary Fig. 2). Ponds 15 and 18 only exhibited negative CO_2 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 CO_2 flux with increasing pond age can be seen in Supplementary Fig. 3, but the same pattern is not evident with CH_4 fluxes. Similarly, a survey on hydroelectric reservoirs around the globe ($n = 85$) found that CO_2 and CH_4 emissions were inversely related to reservoir age[30] a pattern that was also found on a temporal study of a single hydroelectric reservoir (CO_2 only)[18].

The CO_2 fluxes from this study were similar to or higher than other aquatic ecosystems, CO_2 fluxes were similar or higher, whereas CH_4 fluxes were typically lower than other urban studies (Supplementary Table 2). Ebullition is considered an important pathway for methane emission in lentic ecosystems with anoxic sediments[31]. Because we did not measure ebullition our values for overall CH_4 flux are likely underestimated. Small patches of bubbles were occasionally captured underneath our flux chamber, causing the internal concentration of CH_4 to spike to ~130,000 ppb. For reference, the global mean atmospheric CH_4 concentration for 2019 was 1870 ppb[32]. There was not a similar spike in CO_2 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 CO_2 and CH_4 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 CO_2 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 CO_2 and pH and the observed decrease in mean CO_2 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

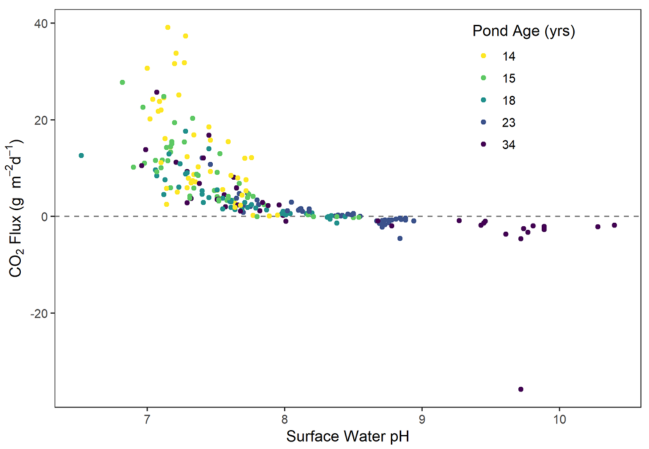

Fig. 2 Carbon dioxide fluxes in stormwater ponds as a function of surface water pH. CO_2 fluxes decrease with increasing surface water pH. In agreement with CO_2-carbonic acid equilibrium, a majority of CO_2 fluxes result in negative values at pH > 8.3, the point at which water column CO_2 is nearly all converted to carbonate or bicarbonate. The dotted line denotes zero CO_2 flux.

related to CO_2 in the LMM, and when combined can be considered indicators of primary production. Photosynthetic fixation of CO_2 from the water column results in an increase in pH and DO. Concurrently, the chemical form of CO_2 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 CO_2 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 CO_2 emissions, the increased pH can exert a compounding effect, rendering high pH ponds and lakes to be weak CO_2 sources, similar to observations in in other urban ponds[11], saline lakes[33], and in agricultural reservoirs[34]. 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 ecosystems[35,36]. Additionally, in high $CaCO_3$ systems, inputs of CO_2 can contribute to increased alkalinity over time via calcium release. Thus, ponds have the potential to become CO_2 sinks over time as a result of increasing eutrophication and accumulation of natural- and anthropogenically sourced alkaline ions increasing pH.

Compared to CO_2, a lower amount of the variation in diffusive CH_4 flux was explained by variables in this study. With accounted variance from random effects, 51% of CH_4 flux variation was explained by an inverse relationship to benthic %DO and positive relationships to benthic temperature and log-transformed CO_2 fluxes (Supplementary Table 3). There was no relationship between CH_4 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 CH_4 alone (by hydrogenotrophic methanogens) or CH_4 and CO_2 (by acetoclastic and methylotrophic methanogens). The positive relationship between CH_4 and CO_2 could be due to this simultaneous production, or due to increased production of CH_4 using CO_2 and H_2 as substrates as a result of its increased availability (by hydrogenotrophic methanogens). CO_2 may also be attributed to aerobic or anaerobic CH_4-oxidation. The negative relationship with water column DO was also indicated during the extreme methane emission 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 activity[37,38].

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 (CO_2-C + CH_4-C flux) (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 CH_4 emission[16]. Previous studies have identified increasing CH_4 emission with ecosystem age[18]

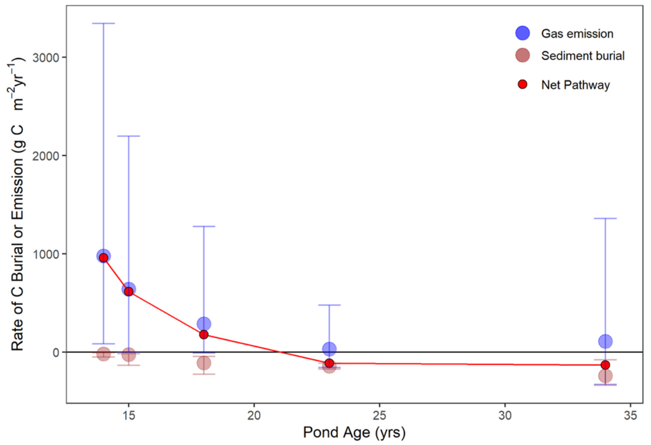

Fig. 3 Net carbon flux in stormwater ponds based on sediment burial and gaseous emission. A comparison of median C flux (blue circles, CO_2-C + CH_4-C) as positive values and median OC burial rate (brown circles) as negative values. The net C flux (g C m^{-2} y^1) in stormwater ponds is shown by the red circles and line and represents the difference between gas flux and burial. Bars on points indicate the 5th and 95th quantiles.

and increasing trophic level[39], which can be associated with pond succession, converting ponds into net sources in terms of CO_2-equivalents due to increasing CH_4 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 CO_2-C and CH_4-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} burial[18].

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 ponds[12]. 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 CH_4 release[29], likely due to increased water column contributions to organic matter oxidation prior to sediment settling and CH_4 oxidation, and ponds or lakes larger in area exhibit lower C emissions per unit area compared to smaller ponds[2]. Other suggested properties for anthropized ecosystem C management include water column stratification, water retention time, water source inputs, and landscape position[34]. It is important to note that drivers of C emission can differ between natural and anthropized ecosystems such that CH_4

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, algaecide 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 recreation as well as mosquito and algal consumption, and can have varying effects on CO_2-equivalent fluxes. For example, by grazing on zooplankton that consume both algae 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 hypolimnetic 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.

Methods

Study site. The study ponds are in Manatee County, Florida, which is part of the Tampa Bay metropolitan area. It has a subtropical climate, with mean annual temperature of 22.8 °C and mean annual rainfall of 140 cm, 60% of which occurs during a distinct annual wet season from May to September. To assess C processing in pond systems, sites selected were permanently wet ponds and had no littoral or emergent vegetation. Submersed macrophytes were not accounted for in this study. Additionally, each pond was surrounded by homes with similar landscape conditions (i.e., ornamental plants, trees, turf grass) and contained no aeration systems. The only exception is Pond 18, which has drainage connection to a residential golf course in addition to the residential neighborhood. A potentially important difference between each pond is the number of inflow and outflow structures, which may impact a pond's total incoming water volume from either surface runoff or other ponds that drain into it. All ponds in this study had one outflow structure and at least one inflow structure, but these were not fully accounted for.

The age of ponds was determined using multiple sources such as the Manatee County Property Appraisal and Google Earth. Google Earth was used to observe aerial photos that date back to 1995. For gap years and ponds older than 1995, ages were determined using aerial photos and housing information on the appraisal website, with the assumption that ponds were constructed at the same time as the houses around it.

Water properties. C cycling processes can be affected by various water properties, such as temperature, pH, salinity, conductivity, and dissolved oxygen (DO). Along with each floating chamber measurement, we recorded these water quality parameters from both surface and bottom water column locations (benthic samples) at three locations evenly distributed in open water at least five meters from littoral shelves. Measurements were taken with a YSI ProDSS sonde. Probe calibration was conducted on the day of each sampling event. Surface and bottom measurements were recorded within 30–60 cm of the water's surface and above the sediment surface, respectively. Samples for dissolved organic carbon and total dissolved nitrogen were filtered with a 0.45 μm filter and analyzed on a Shimadzu TOC-L Total Organic Carbon Analyzer coupled with a TNM-L Total Nitrogen Module.

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 de fined as all organic sediment muck 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 muck-sand contact defines 'time zero' when the pond was constructed, and material began to deposit. We assume that all muck 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 subsamples of dried sediments were analyzed for total organic carbon (TOC). The three subsamples were averaged 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 two equations below. The second considers the cumulative pond age to estimate a burial rate of g OC m^{-2} yr^{-1} (Eq. 2). Equation 1 is calculated as the mean C content per core (g C per 100 g dry sediment) times the core dry weight, divided by the core sleeve opening area (m^2). Equation 2 is the same as Eq. 1 with the addition of pond age (years) as a multiplier in the denominator.

$$\text{Burial rate}\left(\text{g OC m}^{-2}\right) = \frac{\text{TOC}\left(\text{g}\,C\,per\,100\,\text{g dry sediment}\right) * \text{Core dry weight}\,(g)}{\text{Sleeve opening area}\,(\text{m}^2)} \tag{1}$$

$$\text{Burial rate}\left(\text{g OC m}^{-2}\text{yr}^{-1}\right) = \frac{\text{TOC}\left(\text{g}\,C\,per\,100\text{g dry sediment}\right) * \text{Core dry weight (g)}}{\text{Sleeve opening area}\,(\text{m}^2) * \text{Pond Age (yrs)}} \tag{2}$$

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 based on the change in gas concentration within the chamber over time and cross-sectional area[47]. 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 but can be used for quantifying spatial variability. CO_2 and CH_4 gas 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 $CO_2/CH_4/H_2O$ Trace Gas analyzer that utilizes a non-dispersive infrared method. Detection ranges were 0–100 ppm CH_4 (0.25ppb precision) and 0–10,000 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 wall extensions are present[48].

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 enclosement, 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 R^2 values for CO_2 and CH_4 chamber flux measurements. For CH_4 measurements, 82% of individual measurements reported R^2 of 0.95 and above, and only two observations were below 0.50. CO_2 slope R^2 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 CO_2 measurements and ten for CH_4.

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 CO_2, CH_4, and H_2O 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 R^2 of exponential fits to linear fits in case there were occasions where a linear fit was more appropriate. For CO_2, there were thirteen out of 236 observations where a linear fit was used and ten out of 239 for CH_4. The equation used here to estimate flux rates is a slight modification of that by Duc et al. (2013)[49](Eq. 3):

$$F = \frac{dX}{dt} \frac{p_a Vb}{RTA} \tag{3}$$

Where F represents the rate of gas flux at the air-water interface (μmol [C gas] $m^{-2}\,s^{-1}$), is the slope of the exponential fit to the change in concentration within the chamber (dC) over time (dt) from the moment of closure ($\mu mol\,mol^{-1}\,s^{-1}$). p_a is ambient pressure (atm), V is the volume of the Smart Chamber (mL), A is the area of chamber coverage (m^2), 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 CO_2 or CH_4 ($g\,\mu mol^{-1}$) and converts $\mu mol\,m^{-2}\,s^{-1}$ to $g\,C\,m^2\,s^{-1}$.

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 CH_4, normality was achieved after log-transformation. The same was not true for CO_2 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 variances was assessed using Levene's test (levene_test() function), all from the R package *rstatix*[50].

To assess the existence of trends in C burial with ponds of differing ages, a linear and logarithmic regression was conducted on aerial rates ($g\,C\,m^{-2}$) and rates using pond age ($g\,C\,m^{-2}\,yr^{-1}$), respectively. In both cases, regressions were calculated as burial rate as a function of pond age (yrs). A linear mixed-effects model (LMM) was conducted for each gas to summarize the full effect of correlated variables using the lmer() function of the R package *lme4*[51]. In both the CO_2 and CH_4 LMM, pond site and sampling date were set as random effect. Correlated variables for each respective gas were initially determined using Pearson correlation coefficients and then included as fixed effects. Slightly modified versions of each CO_2 and CH_4 LMM were compared using the Second-order Akaike information criterion (AICc). Surface and benthic DO were correlated to CH_4 fluxes in the LMM and were determined to be acting independently on fluxes by reviewing their plots against one another and comparing linear models with CH_4 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

conceptualization – 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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