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Greenhouse gas emissions from urban ponds are driven by nutrient status and hydrology

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Abstract. Inland waters emit significant quantities of greenhouse gases (GHGs) such as methane (CH_4) and carbon dioxide (CO_2) to the atmosphere. On a global scale, these emissions are large enough that their contribution to climate change is now recognized by the Intergovernmental Panel on Climate Change. Much of the past focus on GHG emissions from inland waters has focused on lakes, reservoirs, and rivers, and the role of small, artificial waterbodies such as ponds has been overlooked. To investigate the spatial variation in GHG fluxes from artificial ponds, we conducted a synoptic survey of forty urban ponds in a Swedish city. We measured dissolved concentrations of CH_4 and CO_2, and made complementary measurements of water chemistry. We found that CH_4 concentrations were greatest in high-nutrient ponds (measured as total phosphorus and total organic carbon). For CO_2, higher concentrations were associated with silicon and calcium, suggesting that groundwater inputs lead to elevated CO_2. When converted to diffusive GHG fluxes, mean emissions were 30.3 mg $CH_4 \cdot m^{-2} \cdot d^{-1}$ and 752 mg $CO_2 \cdot m^{-2} \cdot d^{-1}$. Although these fluxes are moderately high on an areal basis, upscaling them to all Swedish urban ponds gives an emission of 8336 t CO_2eq/yr (±1689) equivalent to 0.1% of Swedish agricultural GHG emissions. Artificial ponds could be important GHG sources in countries with larger proportions of urban land.

Key words: artificial; carbon dioxide; greenhouse gas; methane; pond; urban.

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INTRODUCTION

The global importance of inland waters in the cycling of carbon and greenhouse gases (GHGs) is now well recognized (Cole et al. 2007). Lakes, rivers, streams, reservoirs, and ponds emit large amounts of carbon dioxide (CO_2) and methane (CH_4; Bastviken et al. 2004, Holgerson and Raymond 2016, DelSontro et al. 2018). The emissions of these GHGs can occur diffusively from the water surface, and also by ebullition, which is the pathway by which CH_4 is emitted directly to the atmosphere via the sporadic release of CH_4-containing bubbles from the sediment. Inland waters also contain dissolved and particulate organic carbon, which can be degraded to release GHGs (Tranvik et al. 2009, Evans et al. 2017), or buried in sediments, thus acting as a carbon (C) sink (Sobek et al. 2009).

On a global scale, the area of small artificial ponds is similar to that of large reservoirs, due to the extremely large number of small ponds (Downing 2010). Additionally, it has been suggested that C and GHG processing in small waterbodies is particularly intense due to the specific physical properties of ponds, such as shallow water and frequent mixing (Downing 2010, Holgerson and Raymond 2016). In support of this, Holgerson and Raymond (2016) estimated that very small waterbodies (with areas <0.001 km^2) contribute 41% of all diffusive CH_4 emissions from lakes and ponds, despite comprising just 8.6% of total global lake and pond area. However, an investigation of their data suggests that only one of the sites in their synthesis is artificial: the Swedish urban pond from Natchimuthu et al. (2014) where relatively high summertime CH_4 fluxes were measured. A comparison of natural and artificial ponds in India by Singh et al. (2000) noted lower CH_4 fluxes in artificial ponds. This was because the artificial ponds had lower nutrient concentrations, less sediment, and sometimes dried up. For restored wetlands, Jordan et al. (2016) found higher CO_2 emissions but similar CH_4 emissions from artificial ponds compared to adjacent artificial shallow lakes. Two recent Australian studies provided compelling evidence for the importance of artificial ponds. Grinham et al. (2018) measured CH_4 fluxes from artificial waterbodies including farm ponds and urban lakes, and found higher diffusive emissions when compared to larger waterbodies in the same region. Similarly, Ollivier et al. (2018) found higher CH_4 emissions from agricultural ponds compared to reservoirs. Considering the above, it can therefore be expected that artificial ponds release disproportionately large amounts of GHGs, although at present there is a general lack of data from these systems, and it is limited to certain land use types and countries.

Artificial ponds can fall under a range of classes, from those that have been created anew on previously terrestrial land, to wetlands that have been modified into ponds (although even the exact definition of what constitutes a pond varies between researchers; Hassall 2014). Nevertheless, Clifford and Heffernan (2018) point to a range of reasons as to why artificial ponds may function differently when compared to natural ones. For instance, artificial ponds will generally be relatively young, giving a reduced time for ecological and biogeochemical processes to develop. Additionally, such ponds are purposefully designed, usually with human-orientated, not ecologically orientated, aims which will likely result in a different set of ecosystem functions. Artificial ponds may also feature vastly different hydrology; inflows and outflows may be regulated, thus changing retention times and the source of water inputs, and the ponds may be used for water abstraction (Lawrence and Breen 1998). Considering the above, there is reason to assume that biogeochemical cycling may differ between natural and artificial ponds.

In order to determine variability in GHG emissions between ponds, we conducted a synoptic sampling campaign of 40 artificial urban ponds where concentrations of CH_4 and CO_2 and complementary water chemistry determinands were measured. Our aim was to determine the spatial, rather than temporal, variation in pond GHG concentrations, because this temporal variation has previously been shown to be large in artificial waterbodies, due to differences in surrounding land use (Vermaat et al. 2011). Additionally, we measured a range of nutrient concentrations and physical pond attributes, in an effort to elucidate the mechanisms driving GHGs.

METHODS

Sampling took place in the Swedish city of Uppsala on 30 May and 1 June 2018. The area has a humid continental climate, with annual precipitation of 668 mm and mean annual temperature of 6.6°C in 2017. Total precipitation and mean temperature for the 30 d preceding sampling were 16.3 mm and 14.6°C (Swedish Meteorological and Hydrological Institute [SMHI] https://www.smhi.se/klimatdata). Using aerial photography, we selected 40 ponds for sampling, all of which fell within a circular area of 78.5 km^2 (diameter of ~10 km). Pond sizes were measured using aerial photography and were grouped into area classes <0.001 km^2 (n = 27) or >0.001 km^2 (n = 13). All ponds were artificial and had been constructed for various purposes: ornamentation (e.g., parks) n = 13; farm ponds n = 2; quarry/industry use n = 2; water regulation (e.g., functioning as stormwater ponds or reducing nutrient and metal exports); n = 23. Ponds were either within the built-up area of the city itself or adjacent to major roads and highways, and therefore, their immediate catchments all had some degree of urban influence, but agriculture and forestry land uses will also have influenced some ponds to a lesser extent. The natural soil type at the sampled ponds is glacial and postglacial clay, but it is highly possible that the ponds soils have been amended or managed.

For each of the two sampling days, ponds were sampled between 10:00 and 16:00. Respective sunrise and sunset times were 3:43 and 21:51, and 3:40 and 21:55, for both sampling days, thus sampling took place during the middle of the day. SMHI data show that the air temperature varied by 1.2°C and 3.7°C during the sampling hours on days 1 and 2, respectively. No adjustments were made to account for diel fluctuations. At the edge of each pond, water temperature and dissolved O_2 were measured in situ using a Hanna Instruments Multiparameter Meter Hi 9829 (Woonsocket, Rhode Island, USA), and individual samples of surface water were taken for GHGs and water chemistry. Samples for dissolved CH_4 and CO_2 analysis were collected using the headspace method (Hope et al. 2004); 30 mL of pond water and 30 mL of ambient air were shaken together in a 60-mL syringe for 60 s, and the headspace was transferred to a 12-mL pre-evacuated glass vial. Water samples were collected for analyses of pH, EC, and nutrient concentrations in high-density polyethylene bottles and for analysis of dissolved organic matter (DOM) quality in glass vials.

Dissolved organic matter composition was measured as absorbance using an Avantes Ava-Light DH-S-BAL (Apeldoorn, The Netherlands) connected to a cuvette holder. Samples were filtered through pre-rinsed 0.45-μm cellulose acetate filters and analyzed 3/5 d after collection. Spectral scans were performed between 180 and 1100 nm, at increments of ~0.3 nm. We calculated the specific ultraviolet absorbance (SUVA) at 254 nm ($SUVA_{254}$) and 280 nm ($SUVA_{280}$) as a measure of DOM aromaticity (Weishaar et al. 2003). We also calculated the spectral slope at 275–295 nm ($S_{275–295}$) as an indicator of molecular weight by taking the slope of the log-transformed spectra, as in Helms et al. (2008). For many of the samples, absorbance was zero in the 350- to 400-nm region; therefore, we did not calculate $S_{350–400}$. Finally, we calculated E2:E3 as the absorbance ratio 250:365 nm, as a measure of aromaticity and molecular weight (Peuravuori and Pihlaja 1997). Headspace CH_4 and CO_2 concentrations were measured on a Picarro GasScouter equipped with a sampling loop (Baird et al. 2010) and converted to dissolved concentrations according to Henry's law (Weiss 1974, Wiesenburg and Guinasso 1979). The following analyses of water samples were performed by the SWEDAC-accredited Geochemical Laboratory at the Swedish University of Agricultural Sciences (for detailed information, refer to SLU 2018, Fölster et al. 2014): pH, EC, alkalinity, ammonium (NH_4), calcium (Ca), chloride, fluoride, magnesium, nitrite + nitrate, phosphate, potassium, silicon (Si), sodium, sulfate, total nitrogen, total organic carbon (TOC), and total phosphorus (TP).

We used two approaches to explore the relationships between dissolved GHGs and other variables. Firstly, we used Spearman's rank correlation coefficients (r_S) to test for monotonic relationships between dissolved GHGs and other variables ($n = 24$) which comprised pond area, water temperature, dissolved O_2, pH, EC, DOM composition, and nutrient/metal concentrations. For this, we used the false discovery rate (FDR) to adjust for multiple comparisons which reduces the chance of type I errors while retaining greater power when compared to a Bonferroni adjustment (Benjamini and Hochberg 1995, Pike 2010). Secondly, we conducted partial least squares regression (PLS) to explore how dissolved GHGs were related to the other biogeochemical variables. This approach has shown to be useful in exploring dissolved carbon dynamics elsewhere (Wallin et al. 2010). We used Mann–Whitney tests to check for differences in GHGs between very small ponds (areas <0.001 km^2) and larger ponds, and to test for differences between pond use. Partial least squares regression was conducted using SIMCA 14, and all other statistical analyses were performed in SPSS Statistics 24 (IBM, Armonk, New York, USA). For all tests, results were significant when P or FDR-adjusted P was <0.05.

To provide an estimate of pond GHG emissions, we calculated fluxes of CH_4 and CO_2 using the method of Holgerson and Raymond (2016). This uses aquatic GHG concentration, water temperature, and a gas exchange velocity (k_{600}), which is assigned as 0.36 m/d for ponds <0.001 km^2 and 0.48 m/d for ponds >0.001 km^2. Gas saturation is factored into the calculation using atmospheric concentrations of 410 ppm for CO_2 and 1.86 ppm for CH_4; thus, pond fluxes can be negative if ponds are under-saturated in

GHGs. Fluxes for each pond were then multiplied by pond area to give a flux for the total area of each pond. We then assumed an ice-free season of 275 d and multiplied the flux per pond by this value to give an annual flux. This assumes that the concentrations measured during our campaign represent an annual average, which is unlikely to be true. Nevertheless, pond GHG concentrations will be higher in summer and lower in winter, thus our spring time sampling is useful as a first approximation of potential fluxes. CH_4 fluxes were converted to CO_2 equivalents using a 100-yr global warming potential of 34 (IPCC 2013).

RESULTS

Concentrations of CH_4 and CO_2

Dissolved CH_4–C ranged 1.73–325 μg/L with a median of 26.0 μg/L. Significant (FDR-corrected) correlations were found with TOC ($r_S = 0.53$, $P = 0.009$) and TP ($r_S = 0.46$, $P = 0.033$; Fig. 1).

Dissolved CO_2–C ranged 0.10–8.13 mg/L with a median of 1.35 mg/L. Significant (FDR-corrected) correlations were found with pH ($r_S = -0.85$, $P < 0.001$), alkalinity ($r_S = 0.64$, $P < 0.001$), Ca ($r_S = 0.68$, $P < 0.001$), Si ($r_S = 0.46$, $P = 0.012$), NH_4 ($r_S = 0.49$, $P = 0.009$), $SUVA_{254}$ ($r_S = 0.42$, $P = 0.022$), and $SUVA_{280}$ ($r_S = 0.43$, $P = 0.022$; Fig. 2). We discounted pH and alkalinity from further analysis as these variables will be influenced by, rather than influence, CO_2. We also discounted $SUVA_{254}$ as this was highly correlated with $SUVA_{280}$ ($r_S = 0.98$, $P < 0.001$).

After removing pH and alkalinity as predictors, the PLS analysis found that the 22 measures of pond biogeochemistry could explain 41% of the variation in dissolved CH_4 and CO_2, although the predictive power of the model was poor (Fig. 3). The PLS shows that dissolved CH_4 and CO_2 are generally unrelated and reinforces the importance of the significantly correlated variables (Figs. 1, 2) for each GHG: that TOC and TP group near CH_4, and Ca, SI, NH_4, and SUVA group near CO_2.

When grouped according to pond area, we found no difference in dissolved GHGs between very small (<0.001 km^2) ponds and larger ponds (Fig. 4). We also grouped ponds according to the two dominant functions, ornamental ponds and water regulation ponds, and found no significant difference between pond groupings for both CH_4 and CO_2 (Fig. 4; note that quarry/industry ponds and farm ponds were excluded entirely from this comparison).

Fluxes of CH_4 and CO_2

For CO_2, 28 ponds were net emitters and 12 ponds consumed CO_2, whereas all ponds were net emitters of CH_4 (Appendix S1: Fig. S2). Estimated daily mean fluxes were 30.3 mg $CH_4 \cdot m^{-2} \cdot d^{-1}$ (ranging from 0.4 to 174 mg $CH_4 \cdot m^{-2} \cdot d^{-1}$)

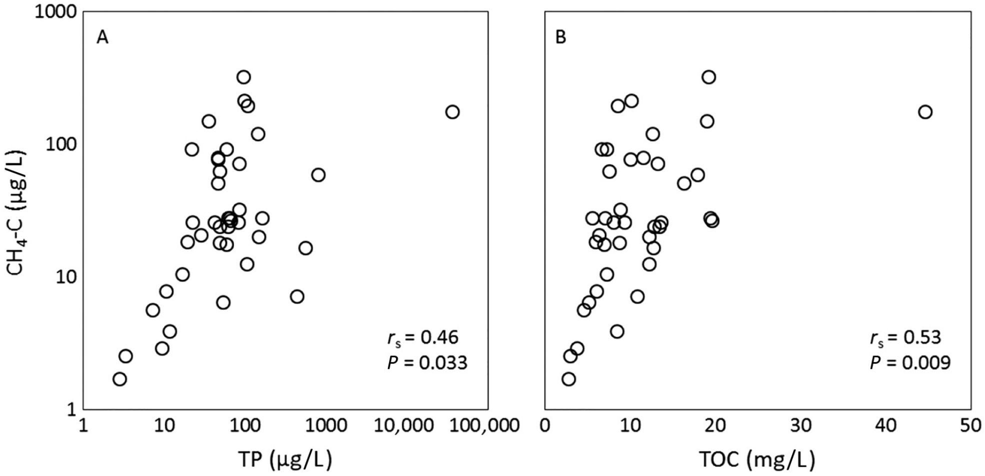

Fig. 1. Scatter plots showing significant correlations for all ponds ($n = 40$) between log-scaled dissolved CH_4 and (A) log-scaled total phosphorus (TP), $r_S = 0.46$, $P = 0.033$, and (B) total organic carbon (TOC), $r_S = 0.53$, $P = 0.009$.

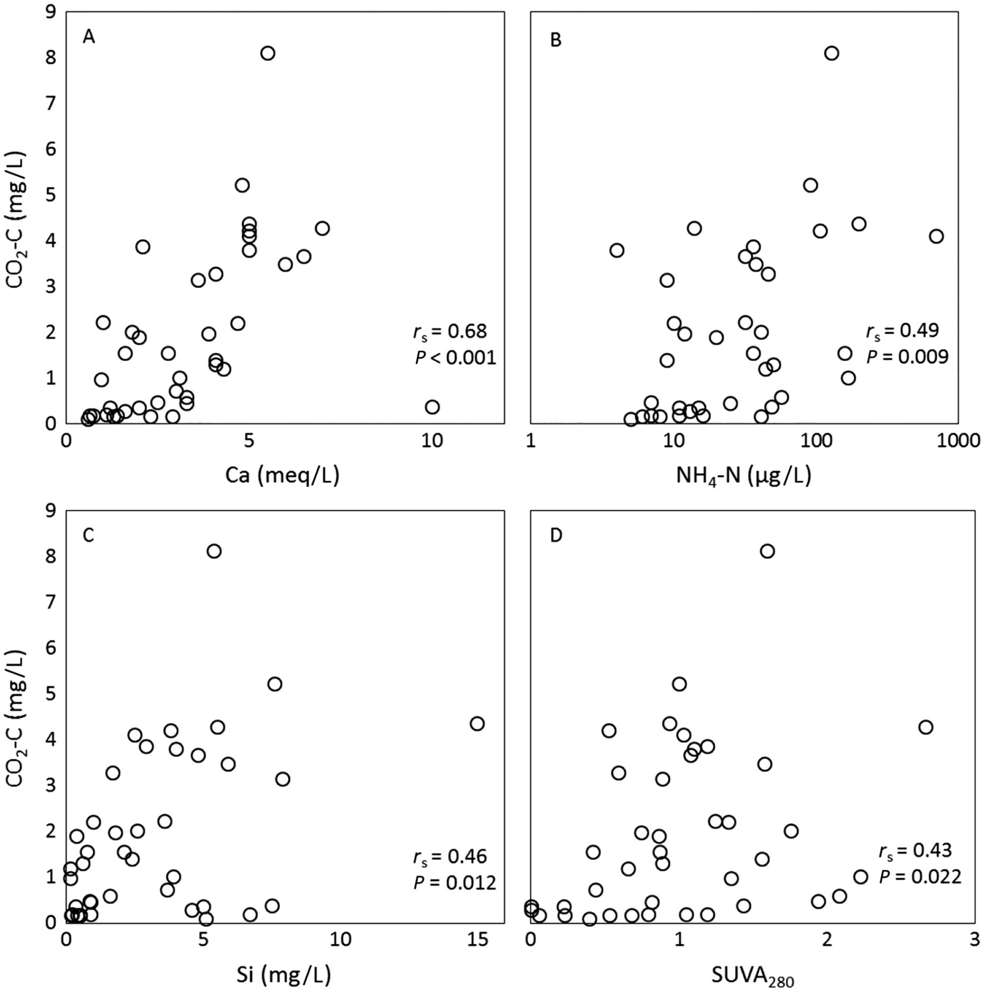

Fig. 2. Scatter plots showing significant correlations for all ponds ($n = 40$) between dissolved CO_2 and (A) calcium (Ca), $r_S = 0.68$, $P < 0.001$, (B) log-scaled ammonium (NH_4–N), $r_S = 0.49$, $P = 0.009$, (C) silicon (Si), $r_S = 0.46$, $P = 0.012$, and (D) $SUVA_{280}$, $r_S = 0.43$, $P = 0.022$.

and 752 mg $CO_2 \cdot m^{-2} \cdot d^{-1}$ (ranging from −187 to 3449 mg $CO_2 \cdot m^{-2} \cdot d^{-1}$). These translate into mean annual fluxes of 8.33 g $CH_4 \cdot m^{-2} \cdot yr^{-1}$ and 206 g $CO_2 \cdot m^{-2} \cdot yr^{-1}$.

DISCUSSION

Concentrations and fluxes of CH_4

We found considerable variation in concentrations of dissolved CH_4 and CO_2 in 40 urban ponds. Our median of CH_4–C of 26.0 μg/L is similar to the mean reported for a Swedish urban pond (of 20.8 μg/L, Natchimuthu et al. 2014). When compared to other aquatic systems, dissolved CH_4 is an order of magnitude higher than the global median for CH_4 concentrations in streams and rivers of 3.0 μg/L (Stanley et al. 2016) and higher than the median of 6.7 μg/L for Swedish streams (Wallin et al. 2018). Similar to findings from lakes (Bastviken et al. 2004, Juutinen et al. 2009), we report significant positive correlations between dissolved CH_4 and both TP and TOC, suggesting that nutrient status is an important driver of CH_4 emissions.

Our calculated mean daily flux of 30.3 mg $CH_4 \cdot m^{-2} \cdot d^{-1}$ is of the same magnitude as other literature values from temperate artificial ponds: for instance, ~80 mg $CH_4 \cdot m^{-2} \cdot d^{-1}$ for Swedish

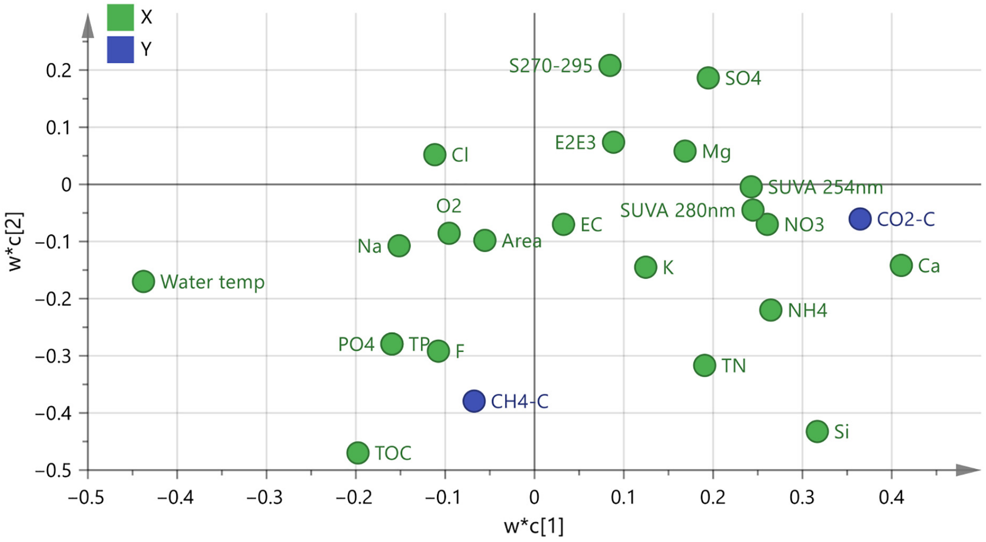

Fig. 3. Partial least squares regression score plot using CH_4–C and CO_2–C as response variables (Y), and pond biogeochemistry and area as predicting variables (X) for all 40 ponds. $R^2 = 0.41$, $Q^2 = 0.11$.

agricultural ponds (Stadmark and Leonardson 2005), and 88 mg $CH_4 \cdot m^{-2} \cdot d^{-1}$ for wet stormwater basins in the United States (McPhillips and Walter 2015), as well as 80 mg $CH_4 \cdot m^{-2} \cdot d^{-1}$ for urban streams draining U.S. stormwater ponds (Smith et al. 2017). Higher average fluxes have been recorded in tropical and sub-tropical ponds, for example, 287 mg $CH_4 \cdot m^{-2} \cdot d^{-1}$ in India (Selvam et al. 2014) and 115–453 mg $CH_4 \cdot m^{-2} \cdot d^{-1}$ in Australia (Grinham et al. 2018, Ollivier et al. 2018). It is important to note that we did not measure ebullition which could be a major pathway for CH_4 release in small, artificial waterbodies (Vermaat et al. 2011, Aben et al. 2017, Grinham et al. 2018), and thus, our fluxes are likely to be an underestimate.

Concentrations and fluxes of CO_2

Median dissolved CO_2–C concentration was 1.35 mg/L, a value that is similar to the averages of 1.7 mg/L from Swedish streams (Wallin et al. 2018) and of 1.61 mg/L from small natural ponds (Holgerson and Raymond 2016). Significant positive correlations were found between dissolved CO_2 and, in order of correlation strength, Ca, NH_4, Si, and SUVA (both $SUVA_{254}$ and $SUVA_{280}$). We suggest that the Ca and Si are both signatures of groundwater (Maxe 2001) and that groundwater inputs are, at least for some ponds, partly driving CO_2 saturation; such effects have been observed for small lakes (Perkins et al. 2015). Positive correlations between dissolved CO_2 and SUVA have been observed for small peatland pools and ditches (Turner et al. 2016, Peacock et al. 2017), and could suggest that DOM with higher molecular weight or increasing aromaticity is more bioavailable to aquatic microbial communities (Tranvik 1990). In support of this, recent work at the molecular level has highlighted that not all aromatic DOM is resistant to degradation (Mostovaya et al. 2017). Finally, positive relationships between aquatic CO_2 and NH_4 have been reported elsewhere (Schrier-Uijl et al. 2011, Yu et al. 2017). Dai et al. (2008) hypothesize that higher concentrations of NH_4 stimulate nitrification, thus leading to consumption of dissolved O_2, with resulting increases in CO_2 production (the authors suggest that by combining the two oxidation processes that constitute nitrification, one mole of NH_4 would yield 1.9 moles of free CO_2).

The calculated mean daily flux of CO_2 was 752 mg $CO_2 \cdot m^{-2} \cdot d^{-1}$. This is larger than the flux from a Swedish urban pond (48.4 mg $CO_2 \cdot m^{-2} \cdot d^{-1}$, Natchimuthu et al. 2014) but considerably lower than that from tropical and sub-tropical ponds, for example, ~3000 mg $CO_2 \cdot m^{-2} \cdot d^{-1}$ in India (Selvam et al. 2014) and ~1100 mg $CO_2 \cdot m^{-2} \cdot d^{-1}$ in Australia (Ollivier et al. 2018). It is important to consider that our daily CO_2 flux

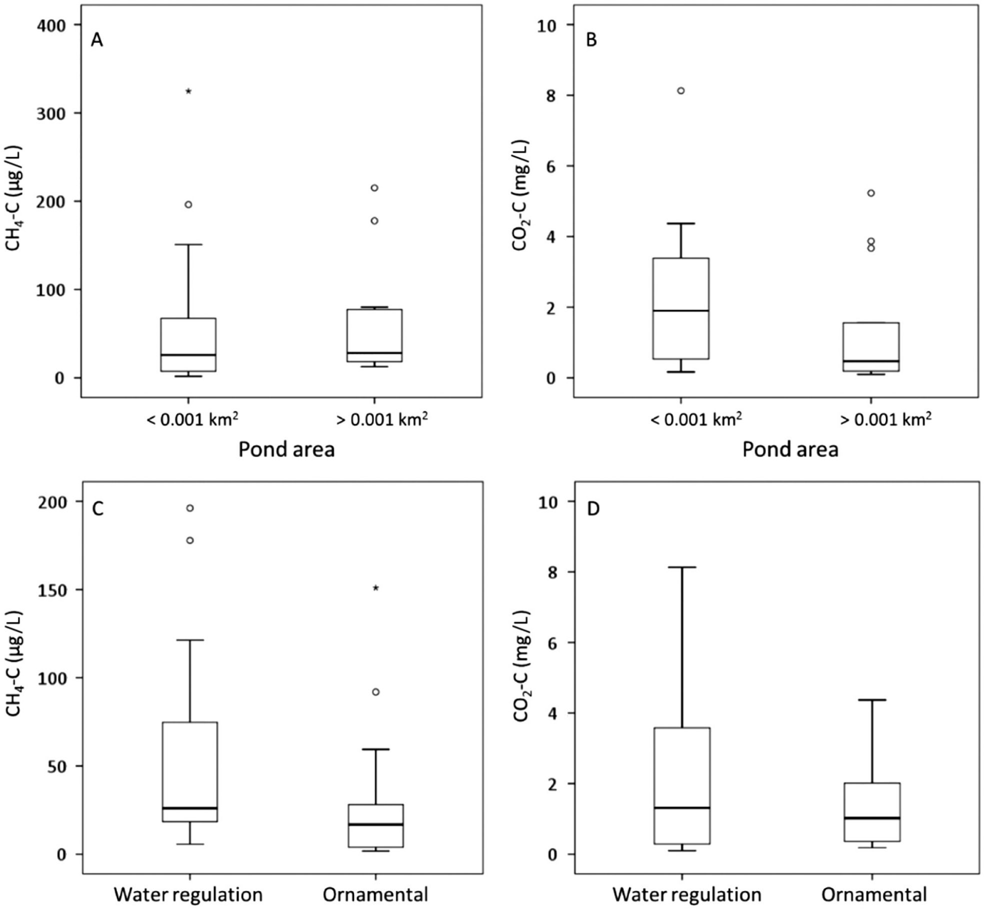

Fig. 4. Dissolved concentrations of CH_4 (A) and CO_2 (B) grouped by pond area. n = 27 for ponds <0.001 km^2 and n = 13 for ponds >0.001 km^2. Dissolved concentrations of CH_4 (C) and CO_2 (D) grouped by the two dominant pond categories: water regulation (n = 23) and ornamental (n = 13). For each panel, the difference in greenhouse gas concentration between treatments (pond area or pond type) is not significant (Mann–Whitney).

calculation may be biased by our daytime-only sampling; 12 of the ponds were found to be sinks for CO_2, presumably due to eutrophication driving high rates of primary production and thus a high photosynthetic CO_2 uptake (Pacheco et al. 2014). CO_2 concentrations will likely follow a diel cycle (Maberly 1996), and it is possible that night time emissions may offset the uptake that occurs during the day.

Implications

Our synoptic sampling campaign demonstrates that GHG emissions from artificial, urban ponds are driven, in part, by nutrient status and hydrology, namely the input of CO_2-rich groundwater. However, all statistical fits were relatively modest, suggesting that untangling the controls on pond GHGs is not straightforward and that further drivers remain to be identified. Unidentified drivers could relate to pond sediment properties and associated microbial communities (Duc et al. 2010), water depth which could affect water column methanotrophy (McEnroe et al. 2009), and eutrophication status (Balmer and Downing 2011).

We found no effect of pond size on dissolved GHGs, in contrast to Holgerson and Raymond (2016) who found that very small (natural) ponds had the highest concentrations. Additionally, and in contrast to the results of Grinham et al. (2018), we found no differences in GHGs between pond category/function (ornamental or water regulation). These differences can perhaps be attributed to the wider landscape beyond the ponds; Holgerson and Raymond (2016) focused on ponds in natural ecosystems, while Grinham et al. examined farm ponds and urban waterbodies. Because our ponds were all within an urban environment, we hypothesize that anthropogenic influences local to each pond (e.g., point sources of nutrients) override any effect of pond physicality (i.e., area) or function. This hypothesis is supported by sediment data from 64 urban stormwater ponds in the United States, where neither nutrient nor pollutant concentrations varied according to the proportion of urban land cover (Blaszczak et al. 2018). The authors suggested that factors such as pond age, the presence of legacy pollutants, retention time, groundwater exchange, and management regime (e.g., use of fertilizers nearby and use of algaecide) could instead control the biogeochemistry of individual ponds.

When expressed as annual fluxes, we find that urban ponds are important sources of CH_4, with annual emissions of 8.3 g $CH_4 \cdot m^{-2} \cdot yr^{-1}$, similar to that reported from a eutrophic stormwater pond (7.3 g $CH_4 \cdot m^{-2} \cdot yr^{-1}$; Martinez-Cruz et al. 2017). This equates to approximately half the mean emissions from northern peatlands (95% confidence level: 10–21 g $CH_4 \cdot m^{-2} \cdot yr^{-1}$, Abdalla et al. 2016) which are key components of the global CH_4 cycle. Similarly, our (potentially underestimated) CO_2 fluxes (752 mg $CO_2 \cdot m^{-2} \cdot d^{-1}$) are only slightly lower than mean global areal emissions from lakes (791 mg $CO_2 \cdot m^{-2} \cdot d^{-1}$) and reservoirs (1209 mg $CO_2 \cdot m^{-2} \cdot d^{-1}$), which are important GHG sources (Deemer et al. 2016). We therefore echo other recent sentiments that small, artificial waterbodies may have been overlooked as significant sources of GHGs (Grinham et al. 2018, Ollivier et al. 2018). There are numerous uncertainties surrounding the detection and mapping of artificial ponds on a global scale, but when their total number and cumulative area are summed, the area they occupy is similar to that occupied by large lakes (Downing 2010). Furthermore, when considering small vs. large waterbodies, it is vital to recognize the proportionally larger littoral zone and aquatic–terrestrial interface for ponds and small lakes, and their associated high rates of biogeochemical processing (Winslow et al. 2014).

To elucidate the importance of urban ponds on a national scale, we further upscaled our measurements as follows. We summed the total area of ponds (0.07 km^2) in the Uppsala urban area (49 km^2) to calculate the proportion of urban area occupied by ponds (0.14%). The total area of urban land in Sweden is 11,822 km^2 (Statistics Sweden 2018) which, assuming a constant proportion of ponds, gives an estimated total pond area of 17 km^2. Applying our annual fluxes to this area gives a total emission of 8336 t CO_2eq/yr (with a standard error of 1689 t CO_2eq/yr). For context, this is equivalent to 0.1% of Swedish agricultural GHG emissions (Statistics Sweden 2017). Our data therefore suggest that artificial urban ponds are important sources of GHGs on a local scale, but not on a national scale, due to the fact that urban areas in Sweden occupy a small proportion of the entire country. The caveat also remains that our calculations exclude CH_4 ebullition and include only diffusive fluxes.

Due to the continued global expansion of urban areas (Seto et al. 2011), the number of small ponds such as those studied here (water regulation, ornamental) will continue to increase. Understanding their climatic impacts and investigating ways to mitigate their GHG emissions is therefore an emerging requirement for future research. For example, Moore and Hunt (2012) noted that the presence of emergent macrophytes was a signifier of pond sediment carbon accumulation. Simple changes, such as stocking urban ponds with appropriate vegetation species, could therefore lead to meaningful changes in aquatic biogeochemistry. Ideally, any future pond management for GHG mitigation will aim to deliver ecosystem service co-benefits such as increased biodiversity and opportunities for recreation (McGuckin and Brown 1995), thus enhancing the role that these blue–green infrastructures play in the urban environment.

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