Received: 15 May 2021 | Accepted: 11 June 2021

DOI: 10.1111/gcb.15762

PRIMARY RESEARCH ARTICLE

Global Change Biology WILEY

Small artificial waterbodies are widespread and persistent emitters of methane and carbon dioxide

M. Peacock[1] | J. Audet[2] | D. Bastviken[3] | S. Cook[4] | C. D. Evans[1,5] | A. Grinham[6] | M. A. Holgerson[7] | L. Högbom[8,9] | A. E. Pickard[10] | P. Zieliński[11] | M. N. Futter[1]

[1]Department of Aquatic Sciences and Assessment, Swedish University of Agricultural Sciences, Uppsala, Sweden

[2]Department of Bioscience, Aarhus University, Silkeborg, Denmark

[3]Department of Thematic Studies - Environmental Change, Linköping University, Linköping, Sweden

[4]School of Biosciences, Division of Agricultural and Environmental Science, University of Nottingham, Loughborough, UK

[5]UK Centre for Ecology & Hydrology, Environment Centre Wales, Bangor, Gwynedd, UK

[6]School of Civil Engineering, The University of Queensland, Brisbane, Qld, Australia

[7]Department of Ecology and Evolutionary Biology, Cornell University, Ithaca, NY, USA

[8]Skogforsk, Uppsala, Sweden

[9]Department of Forest Ecology and Management, SLU, Umeå, Sweden

[10]UK Centre for Ecology and Hydrology, Edinburgh, UK

[11]Department of Environmental Protection, Faculty of Biology, University of Bialystok, Białystok, Poland

Correspondence
M. Peacock, Department of Aquatic Sciences and Assessment, Swedish University of Agricultural Sciences, 756 51 Uppsala, Sweden.
Email: michael.peacock@slu.se

Funding information
European Research Council, Grant/Award Number: 725546; FORMAS, Grant/Award Number: 2018-01794 and 2020-00950; NERC, Grant/Award Number: NE/N018087/1

Abstract

Inland waters play an active role in the global carbon cycle and emit large volumes of the greenhouse gases (GHGs), methane (CH_4) and carbon dioxide (CO_2). A considerable body of research has improved emissions estimates from lakes, reservoirs and rivers but recent attention has been drawn to the importance of small, artificial waterbodies as poorly quantified but potentially important emission hotspots. Of particular interest are emissions from drainage ditches and constructed ponds. These waterbody types are prevalent in many landscapes and their cumulative surface areas can be substantial. Furthermore, GHG emissions from constructed waterbodies are anthropogenic in origin and form part of national emissions reporting, whereas emissions from natural waterbodies do not (according to Intergovernmental Panel on Climate Change guidelines). Here, we present GHG data from two complementary studies covering a range of land uses. In the first, we measured emissions from nine ponds and seven ditches over a full year. Annual emissions varied considerably: 0.1–44.3 g CH_4 m^{-2} $year^{-1}$ and −36–4421 g CO_2 m^{-2} $year^{-1}$. In the second, we measured GHG concentrations in 96 ponds and 64 ditches across seven countries, covering subtropical, temperate and sub-arctic biomes. When CH_4 emissions were converted to CO_2 equivalents, 93% of waterbodies were GHG sources. In both studies, GHGs were

positively related to nutrient status (C, N, P), and pond GHG concentrations were highest in smallest waterbodies. Ditch and pond emissions were larger per unit area when compared to equivalent natural systems (streams, natural ponds). We show that GHG emissions from natural systems should not be used as proxies for those from artificial waterbodies, and that artificial waterbodies have the potential to make a substantial but largely unquantified contribution to emissions from the Agriculture, Forestry and Other Land Use sector, and the global carbon cycle.

KEYWORDS
ditch, drainage, greenhouse gas, inland waters, pond, stream

1 | INTRODUCTION

> There are miles and miles of ditches which have no perceptible current [...] ditches in which vegetation appears to run wild [...]. Add to these fish ponds, duck ponds, horse ponds, ponds upon commons, moors, heaths, or beside the village green. Wherever there is an expanse of water, not even larger than could be covered with a blanket, or a ditch no wider than can be straddled over, if there is no perceptible flow or current then we claim it for the definition of 'still waters', and extend over it our jurisdiction.
>
> The introductory text, written ~130 years ago, to a naturalist's guidebook titled 'Ponds and Ditches', that covers plants, algae, protozoa and invertebrates (Cooke, 1892).

Inland waters (lakes, rivers, reservoirs and streams) are active components of the global carbon cycle; they transport and transform terrestrially derived organic and inorganic carbon, capture carbon dioxide (CO_2) through aquatic primary production, emit CO_2 via mineralization and degassing, bury carbon in their sediments and have the potential to emit large quantities of methane (CH_4) produced in anaerobic sediments (Cole et al., 2007). The magnitude of CO_2 and CH_4 evasion from inland waters on a global scale is uncertain, with recent estimates ranging from 1.0 to 3.9 Pg C $year^{-1}$ (Ciais et al., 2013; Drake et al., 2018). Despite this uncertainty, these emissions are large enough to offset a considerable proportion of the terrestrial carbon sink (Bastviken et al., 2011; Raymond et al., 2013), and thus, there is continued interest in identifying the sources and quantifying the magnitude of inland water carbon evasion.

Small waterbodies and especially 'still waters' have historically been excluded from GHG flux studies, but their global abundance and high potential for biogeochemical cycling indicate that they could be an important component of the global carbon budget (Downing, 2010; Verpoorter et al., 2014). In support of this idea, a synthesis of data from 427 lakes and primarily natural ponds found inverse relationships between waterbody size and both CH_4 and CO_2, with GHG concentrations greatest in ponds (Holgerson & Raymond, 2016). Although ponds are smaller than lakes, the distinction between ponds and lakes can be fuzzy, and definitions for what exactly constitutes a pond are numerous and sometimes contradictory (Biggs et al., 2005). Small waterbodies likely have higher GHG concentrations due to their physical characteristics: High edge effects can lead to proportionally more terrestrial carbon inputs relative to larger systems, frequent water mixing means sediment respiration impacts more of the water column and shallow depth favours emission pathways (e.g. ebullition) that limit the potential mitigation by CH_4 oxidation (Holgerson & Raymond, 2016). Since Holgerson and Raymond's (2016) synthesis was published, additional studies have found high GHG fluxes from vernal pools (Kifner et al., 2018), thaw ponds (Kuhn et al., 2018) and artificial ponds (Gorsky et al., 2019; Grinham et al., 2018; Martinez-Cruz et al., 2017; Ollivier et al., 2019a; Peacock et al., 2019; Webb, Leavitt, et al., 2019). A new synthesis further supports the inverse lake size–GHG flux relationship: 37% of total lentic CH_4 emissions (diffusive + ebullitive) came from waterbodies <0.001 km^2 in size (Rosentreter et al., 2021). These recent studies have also suggested that artificial ponds may have even higher GHG emissions per m^2 than natural ones.

Recent studies on GHG flux from natural and artificial ponds suggest that another small and artificial waterbody type may also have high carbon emissions: ditches. Ditches share some of the same physical characteristics that drive high GHG flux in ponds: They are generally shallow, have high terrestrial inputs relative to aquatic area, are common landscape features and their cumulative length can be greater than that of streams and rivers (e.g. for Great Britain, total ditch length is more than double that of streams and rivers; Brown et al., 2006). Yet ditches are largely unexplored ecosystems (Koschorreck et al., 2020), and recent evidence suggests that they can have high CH_4 emissions, with implications for landscape to global-scale CH_4 budgets (Evans et al., 2016; Koschorreck et al., 2020; Peacock et al., 2021).

Variations in pond and ditch GHG emissions, when compared to natural aquatic systems, may be largely attributed to human-induced alterations to biogeochemical cycling. For instance, artificial waterbodies are often subjected to extensive hydrological management (regulated inflows and outflows, water abstraction, etc.), which affects retention times and water sources, in turn affecting aquatic

biogeochemistry (Clifford & Heffernan, 2018). Ongoing managements such as dredging or clearing (Nieminen et al., 2018), and historic land uses such as industry or agriculture (Blaszczak et al., 2018) will also have effects on biogeochemistry and carbon evasion (Singh et al., 2000). The geographic distribution of natural and artificial waterbodies also differs; notably many natural ponds occur in glaciated terrain in areas of relatively low-intensity land use, and may be oligotrophic, whereas many artificial ponds are often eutrophic because they are embedded within agricultural (e.g. farm ponds) and urban environments (e.g. engineered storm water ponds which are designed to receive high-nutrient run-off). Land use is highly relevant, because a positive relationship between eutrophication, particularly phosphorus concentration, and CH_4 emissions has been widely documented for aquatic ecosystems (Bastviken et al., 2004; Beaulieu et al., 2019; Juutinen et al., 2009). Shallow water bodies with low flows can also (in the absence of regular dredging and maintenance) develop a high cover of emergent macrophytes, increasing CH_4 transport from sediment to the atmosphere (Oliveira-Junior et al., 2018), or dense floating mats of vegetation that can lead to anoxia within the water column (Kosten et al., 2016).

Several recent studies have provided evidence for the importance of small, artificial waterbodies in the global carbon cycle. Grinham et al. (2018) found that artificial ponds in the state of Queensland emit the equivalent of 10% of the GHG emissions from the state's land use, land use change and forestry (LULUCF) sector emissions. For ditches, two syntheses (one of peatland studies, and one of all studies) found that they act as hotspots for CH_4 emission and, despite their limited areal extent, have the potential to offset carbon uptake by the terrestrial portion of the ecosystem (Evans et al., 2016; Peacock et al., 2021). Although other studies of carbon emissions from artificial ponds and ditches exist, they are often limited with either no temporal replication (i.e. multiple waterbodies are only sampled once; Ollivier et al., 2019a; Panneer Selvam et al., 2014; Peacock et al., 2019; Webb, Leavitt, et al., 2019) or no spatial replication (i.e. one waterbody is sampled repeatedly over time; Gao et al., 2013; Liu et al., 2017; Natchimuthu et al., 2014; van Bergen et al., 2019). Another shortcoming is that measurements of CH_4 are often limited to diffusive fluxes, with ebullition being excluded (e.g. Audet et al., 2020; Ollivier et al., 2019a; Peacock et al., 2019; Yu et al., 2017). The recently published refinement to the 2006 IPCC Guidelines (IPCC, 2019) explicitly addresses the potential importance of CH_4 (but not CO_2) emissions from small, artificial waterbodies and their possible role in climatic warming, whilst also pointing to a lack of data, particularly from ditches on non-peat soils, and from artificial ponds (excluding aquaculture, where emissions are better documented; Yuan et al., 2019). The estimated emission factors were 18.3 g CH_4 m^{-2} $year^{-1}$ for artificial ponds and 41.6 g CH_4 m^{-2} $year^{-1}$ for ditches (IPCC, 2019). No disaggregation of emission factors by climate zone, nutrient status or morphology was possible given the limited data available.

Here, we address these knowledge gaps by measuring CH_4 and CO_2 emissions from artificial ponds and ditches, using two approaches:

1. Intensive monitoring, whereby fluxes of CH_4 and CO_2 from seven ditches and nine ponds under different land uses and soil types were measured repeatedly over a 1 year period.
2. An extensive survey, whereby over the course of 1 year, 64 ditches and 96 ponds, spanning seven countries and a range of ecoregions, were sampled on one occasion only.

For both approaches, we supplemented our CH_4 and CO_2 measurements with analyses of water chemistry to determine whether factors such as nutrient status were implicated in controlling GHG evasion.

2 | METHODS

2.1 | Intensive monitoring

Fieldwork took place in April–December 2018 in Uppsala County, Sweden, during the ice-free period. The region is hemiboreal, and for 2018, the mean annual temperature was 8.0℃ and annual precipitation was 510 mm (Figure S1). Seven ditches and nine ponds were monitored in three areas (Figure S2), and individual waterbodies spanned a range of land use (settlement, forest, cropland), nutrient status (total phosphorus 8–2200 μg L^{-1}, dissolved organic carbon 4–90 mg L^{-1}) and morphology (mean depth 4–78 cm, pond area 40–4000 m^2; Table 1, Table SI1). These waterbodies were chosen to represent the broad diversity in soil type, land use, size and water chemistry encountered in this region.

Measurements of GHG fluxes took place on 11 occasions, every 2–5 weeks, and were more frequent during the summer. Generally, sampling took place over two consecutive days, the exception being 4 July 2018 when all sites were visited during one day. Sampling was conducted during daylight hours, typically 9:00–15:00, and on each occasion individual ditches and ponds were visited in a different order so as to reduce any systematic bias due to time of day.

Fluxes of CH_4 and CO_2 were measured in the field using a floating chamber design adapted from Bastviken et al. (2015). Chambers had circular bases with a diameter of 31.5 cm, and a total volume of 9.56 L. Chambers were covered in aluminium foil to reflect sunlight and minimize heating effects. Water flow rates in waterbodies were minimal/absent, and chambers were not anchored and therefore free to drift, as recommended by Lorke et al. (2015). The chamber was connected in a closed loop via two plastic tubes to a Picarro GasScouter G4301 which uses cavity ring-down spectroscopy to measure concentrations of CH_4 and CO_2 in real time. The chamber was deployed until a linear increase, or no change, was observed in GHG concentrations, which was typically 1–5 min (Ollivier et al., 2019a; Peacock et al., 2017). On a few occasions where flux data were excessively noisy, the chamber was removed from the water, air ventilated and the measurement repeated until a clear flux, or no change, was observed. During the course of the year, some waterbodies dried out; this occurred nine times at site 13 (a forest ditch), and once each at sites 9 (pond) and 10 (ditch). For these 11

TABLE 1 Information on the seven ditches and nine ponds (for the intensive monitoring) sampled across the three areas of Uppsala showing waterbody type, pond area or ditch depth (*D*) and width (*W*) and annual GHG fluxes

Site	Type	Soil	Land use	Area (m^2) $D \times W$ (m)	CH_4 (g CH_4 m^{-2} $year^{-1}$)	CO_2 (g CO_2 m^{-2} $year^{-1}$)
1	Pond	Clay	Settlement	139	0.11	15.3
2	Pond	Clay	Settlement	1331	44.34	540.4
3	Pond	Clay	Settlement	1297	1.8	1.0
4	Pond	Clay	Settlement	2588	3.07	−36.4
5	Pond	Clay	Settlement	4152	0.48	129.4
6	Ditch	Clay	Settlement	2.5 × 5	0.53	328.1
7	Ditch	Clay	Settlement	1.1 × 3.5	8.51	264.4
8	Pond	Clay	Settlement	114	32.51	718.2
9	Pond	Fen peat	Felled forest	40	0.94	1138
10	Ditch	Fen peat	Felled forest	0.7 × 1.6	0.1	1658
11	Ditch	Fen peat	Forest	0.6 × 1.6	8.14	4420.5
12	Ditch	Clay	Cropland	0.8 × 2.9	25.59	1723
13	Ditch	Sandy till	Forest	0.8 × 2.4	1.53	1465.6
14	Ditch	Clay	Cropland	1.1 × 2.6	3.56	795.6
15	Pond	Fine sand	Cropland	240	1.91	549.3
16	Pond	Clay	Cropland	252	2.79	628.4

Note: Soil types were derived from Geological Survey of Sweden maps. All sampling sites were located approximately 10–35 m above sea level.

occasions, a flux measurement was taken by carefully setting the chamber onto the base of the ditch and proceeding as above. At site 13, the mean R^2 from linear regressions between CO_2 and chamber deployment time was 0.71, which is relatively high and suggests a good seal between sediment and chamber. Forest ditches in this region frequently dry out and, despite this drying period, site 13 did not show anomalously low or high annual emissions (Table 1).

Fluxes were calculated by performing linear regression between chamber deployment time and GHG concentration, and were corrected to air temperature and pressure. We used the screening process of Peacock et al. (2017) that assumes all non-significant ($p > 0.05$) regressions are zero fluxes, but accepts significant fluxes that have low R^2 values. This approach means that small, noisy, non-zero fluxes are retained. Although these fluxes make little difference to the annual flux, they are useful when evaluating the GHG source/sink status of these water bodies. Across 16 sites and 11 sampling occasions, 176 individual flux measurements were made. For CH_4, five flux measurements were not significant and therefore assumed to be below detection limits and, for the accepted fluxes, the mean R^2 was 0.80. For CO_2, 17 fluxes were not significant and assumed to be zero, and the mean R^2 was 0.77 for the accepted fluxes. Annual diffusive fluxes were calculated using linear interpolation between sampling dates. Although our experimental approach was designed to measure diffusive fluxes, ebullition events were frequently observed, particularly during summer. On some of these occasions, the chamber captured ebullition events, visible as sudden increases in CH_4 concentration. For these, it was possible to calculate ebullition fluxes by dividing the mass of CH_4 released during the bubble event by the total deployment time of the chamber (Gogo et al., 2011; Vermaat et al., 2011). Due to the short chamber deployment time, these can be considered minimum ebullition fluxes. If ebullition events were captured by the chamber, the chamber was ventilated and replaced on the waterbody to measure a diffusive flux. In our analysis, we mostly consider diffusive fluxes, and ebullitive fluxes are presented and discussed separately.

Water depth was measured at each waterbody, and measurements of pH, electrical conductivity (EC), dissolved O_2 and water temperature were made using a Hanna Instruments Multiparameter Meter Hi 9829. Additionally, from the second sampling trip onwards, water samples were collected (no samples were collected if ditches were dry) for analyses of ammonium (NH_4), nitrite + nitrate (NO_2 + NO_3, from herein abbreviated as NO_3), total phosphorus (TP) and dissolved organic carbon (DOC) at the SWEDAC-accredited Geochemical Laboratory at the Swedish University of Agricultural Sciences, Uppsala. Additional water samples were collected and filtered at 0.45 μm using pre-rinsed cellulose acetate filters. Measurements of light absorbance were then made using a 1 cm path length cuvette and an Avantes AvaLight DH-S-BAL light source. Proxies for dissolved organic matter (DOM) composition were calculated, that is, specific ultraviolet absorbance (SUVA; absorbance at 254 nm normalized by DOC concentration; Weishaar et al., 2003), the spectral slopes at 275–295 nm ($S_{275-295}$) and 350–400 nm ($S_{350-400}$), the ratio of these slopes (SR; Helms et al., 2008) and the E2:E3 ratio (Peuravuori & Pihlaja, 1997). These metrics are frequently used to provide information about the composition of the bulk DOM pool (e.g. molecular weight, aromaticity).

2.2 | Extensive survey

Between 10 May 2018 and 16 May 2019, 160 individual waterbodies ditches (n = 64) and ponds (n = 96) were sampled in seven countries covering subtropical, temperate and sub-arctic biomes: Australia, the United Kingdom, Denmark, Poland, Belarus, Sweden and Norway (Figures S3–S9). Our aim was to sample artificial waterbodies. However, without knowing detailed histories of all sites, there is the possibility that our analysis includes heavily modified natural waterbodies. Each individual waterbody was only sampled once. Collated by season, samples were collected during winter = 16, spring = 31, summer = 73, autumn = 40. For this, we assumed northern (N) and southern (S) hemisphere seasons were the following months: winter, N = 12–2, S = 6–8; spring, N = 3–5, S = 9–11; summer, N = 6–8, S = 12–2; autumn, N = 9–11, S = 3–5.

Sampling methods were standardized for consistency. Field measurements were made of water and air temperature, and the surrounding land use was noted. Land use types included cropland (n = 18), grassland (n = 31), unmanaged land and forest (n = 26) and urban/settlement (n = 85). For ponds, area was estimated using Google Earth. At each waterbody, a dissolved gas sample was taken in a pre-evacuated 12 ml Labco Exetainer using the headspace method (Hope et al., 2004) with a ratio of 1:1 of ambient air to water samples. Headspace gas samples were extracted in the field and shipped to SLU Uppsala for analysis of dissolved CH_4 and CO_2 using a Picarro GasScouter equipped with a sampling loop (Wilkinson et al., 2018) and CH_4 and CO_2 were converted to dissolved concentrations according to Henry's law (Weiss, 1974; Wiesenburg & Guinasso, 1979). Fluxes were calculated using dissolved gas concentration, water temperature and gas exchange velocities of 0.36 m day^{-1} for ditches and ponds <1000 m^2 (n = 39), 0.48 m day^{-1} for ponds 1000–10,000 m^2 (n = 50) and 0.57 m day^{-1} for waterbodies >10,000 m^2 (n = 7, our largest sampled waterbody was 60,650 m^2; Holgerson & Raymond, 2016).

A water sample was collected from each ditch or pond in a 12 ml glass Labco Exetainer or 100 ml high-density polyethylene bottle. Analyses of pH and EC were made using a Metrohm 691 and a WTW Cond 3310 respectively. The sample was then filtered and analysed for DOM composition as in Section 2.1. The calculated metrics were $S_{275-295}$, $S_{350-400}$, SR and E2:E3. DOC concentration was calculated using absorbance at 270 nm and 350 nm, according to a published model (Carter et al., 2012; Tipping et al., 2009). All analyses were performed at SLU Uppsala. In contrast to the intensive monitoring, samples from the extensive survey were not analysed for TP, NH_4 or NO_3.

2.3 | Statistics

Linear mixed effect models were used to explore linkages in the intensive data set between CH_4 and CO_2 fluxes and selected environmental variables, as these models are particularly suitable to examine temporal patterns in data sets from different sites (Zuur et al., 2009). The aim of the analysis was to test the effect of selected potential drivers of GHG fluxes. Waterbody type (pond or ditch), water depth, water temperature, pH, EC, O_2, NH_4, NO_3, TP, DOC, SUVA, $S_{275-295}$, $S_{350-400}$, SR and E2:E3 were added as fixed effects in the models, and the sampling location was added as a random effect (Table SI2). Flux measurements with no associated water chemistry sample were not included in the analysis.

To analyse the extensive data set, linear models were used to explore potential correlations between CH_4 and CO_2 concentrations and waterbody type (pond or ditch), land use, water depth, water temperature, pH, EC, DOC, $S_{275-295}$, $S_{350-400}$, SR and E2:E3 (Table SI3). Starting with a full model containing all the above-listed variables, we created a fully reduced model using AIC-based stepwise regression (R; stepAIC package) to determine the most parsimonious model explaining variation in CH_4 or CO_2 concentrations. We ran models on the full extensive data set and also split the data by pond or ditch and re-ran the models in order to see if the same drivers were significant for both waterbody types.

All models were checked for normality and homogeneity of variance by visual inspection of plots of residuals against fitted values (Zuur et al., 2009). The significance of the models was assessed by comparison with a null model using the likelihood ratio. The potential predictor variables were checked for multicollinearity using the variance inflation factor (VIF) values (VIF < 10 indicating low risk of multicollinearity). The statistical analyses were performed using the open source statistical software R version 3.4.4 for Windows (R Development Core Team, 2020), with the package 'nlme' and the function 'lme' therein (Pinheiro et al., 2012) for the mixed models, and function 'lm' and 'stepAIC' from the package MASS (Venables & Ripley, 2002).

We used Mann–Whitney and Kruskal–Wallis tests to check for differences in levels of GHGs and other biogeochemical determinants between categories (e.g. season, pond vs. ditch, organic vs. mineral soil, etc.). In order to adjust for multiple tests, we used the false discovery rate (FDR) to adjust p values; this reduces the chance of type I errors while retaining greater power when compared to a Bonferroni adjustment (Benjamini & Hochberg, 1995). Where significant (p < 0.05) differences were found, we also calculated effect sizes (d) as:

$$d = (\text{mean of treatment } A - \text{mean of treatment } B) / \text{standard deviation}.$$

Effect sizes were: 0.1–0.19 = very small, 0.2–0.49 = small, 0.5–0.79 = medium and >0.8 = large (Cohen, 1988; Sawilowsky, 2009).

Finally, we investigated potential differences in GHG emissions from artificial and natural waterbodies. For this, we compared the CO_2 and CH_4 concentrations from our extensive survey (artificial waterbodies) with those synthesized by Holgerson and Raymond (2016) (almost all natural ponds). For this comparison, we used the pond area groupings from Holgerson and Raymond (2016) of <1000 m^2, 1000–10,000 m^2 and 10,000–100,000 m^2. As a comparison with ditches, we took the mean CH_4 concentration for streams from a global synthesis (table 1 in Stanley et al., 2016).

3 | RESULTS

3.1 | Intensive monitoring

Overall mean daily fluxes were 41.3 ± 17.2 mg CH_4 m^{-2} day^{-1} and 3643 ± 482 mg CO_2 m^{-2} day^{-1} from the 11 sampling campaigns. Slight seasonal patterns were observed for both GHGs, and variation among sites within each month was frequently substantial. For CH_4, mean fluxes from ditches and ponds were greatest during the summer months (June–August) and into September for ditches (Figure 1a). Means were considerably larger than medians, showing the skewness of the data set which is reasonable when ebullition constitutes a large share of the flux, but both mean and median fluxes were lowest during April and October–December. In comparison, mean and median CO_2 fluxes were more similar, with higher averages and greater variation in fluxes from ditches than ponds (Figure 1b). Maximum mean and median CO_2 fluxes occurred during summer. Negative fluxes (i.e. CO_2 uptake) were observed on occasion, and this was particularly pronounced for ponds during the start of August. Calculated mean annual fluxes for all waterbodies were 8.5 g CH_4 m^{-2} $year^{-1}$ and 900 g CO_2 m^{-2} $year^{-1}$. Fluxes were 0.1–25.6 g CH_4 m^{-2} $year^{-1}$ and 264–4421 g CO_2 m^{-2} $year^{-1}$ for ditches and 0.1–44.3 g CH_4 m^{-2} $year^{-1}$ and −36–1138 g CO_2 m^{-2} $year^{-1}$ for ponds (Table 1).

Linear mixed effect models showed significant effects of water temperature (positive effect), NH_4 (positive) and O_2 (negative) on CH_4 fluxes (Figure 2a–c; Table SI2). Waterbody type was not a significant driver in this model. Fluxes of CO_2 were inversely related to pH and negligible when pH was above 8 (Figure 2d; presumably due to carbonate equilibrium leading to a dominance of carbonates over CO_2 at high pH) and were positively related to DOC (Figure 2e; Table SI2). The models found no empirical relationship between TP and GHG emissions but, when grouped by TP concentration, fluxes were significantly greater in waterbodies with higher TP levels (Figure 3a,b), and the same pattern was found for NH_4 (Figure 3c,d).

On 33 occasions at 10 sites, bubble events were caught by the static chamber, allowing ebullitive fluxes to be calculated. The mean ebullition flux was 257 mg CH_4 m^{-2} day^{-1} (SE = 103 mg CH_4 m^{-2} day^{-1}). No ebullition events were captured during the final two sampling campaigns (November and December), and only five events were captured at water temperatures <12°C, despite 58 (approximately one-third of all) chamber measurements taking place on ditches/ponds below this temperature. Although there was no significant correlation between water temperature and ebullition, there was a strong, positive relationship (R^2 = 0.66) between the mean air temperature during each individual sampling campaign and the percentage of sites where ebullition events occurred during chamber deployments (Figure S12).

3.2 | Extensive survey

Considerable variation was evident in the chemistry and physical characteristics of the sampled ponds and ditches (Figure 4). On average, ponds were significantly more alkaline (Figure 4e). Ditches had higher EC although the effect size was small (Figure 4f), and ditches

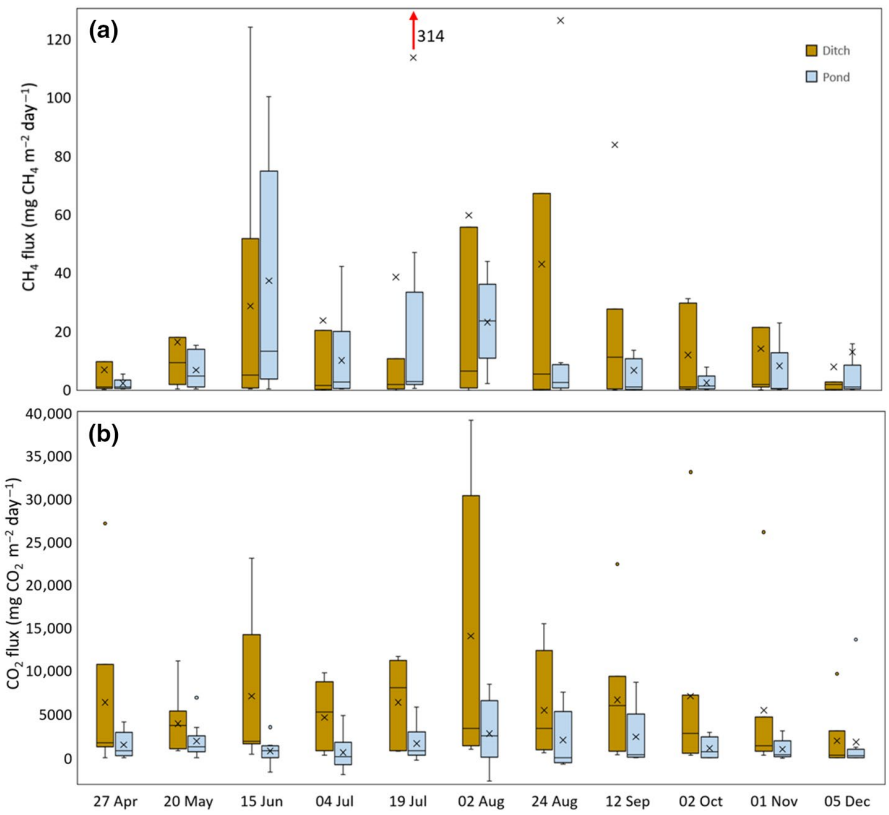

FIGURE 1 Box plot of daily CH_4 (a) and CO_2 (b) fluxes from the seven ditches and nine ponds in the intensive monitoring. Boxes represent medians and interquartile range (IQR), whiskers mark minimum and maximum values, excluding outliers (calculated as box limits ±1.5 x IQR). Also shown are mean fluxes (x) and outliers (o) for CO_2 fluxes. Note that outliers for CH_4 flux extend far beyond the figure boundaries so for clarity are not shown but are included in Figure S10. The mean CH_4 flux for ponds on 19 July is outside the boundaries of the panel and marked by the red arrow

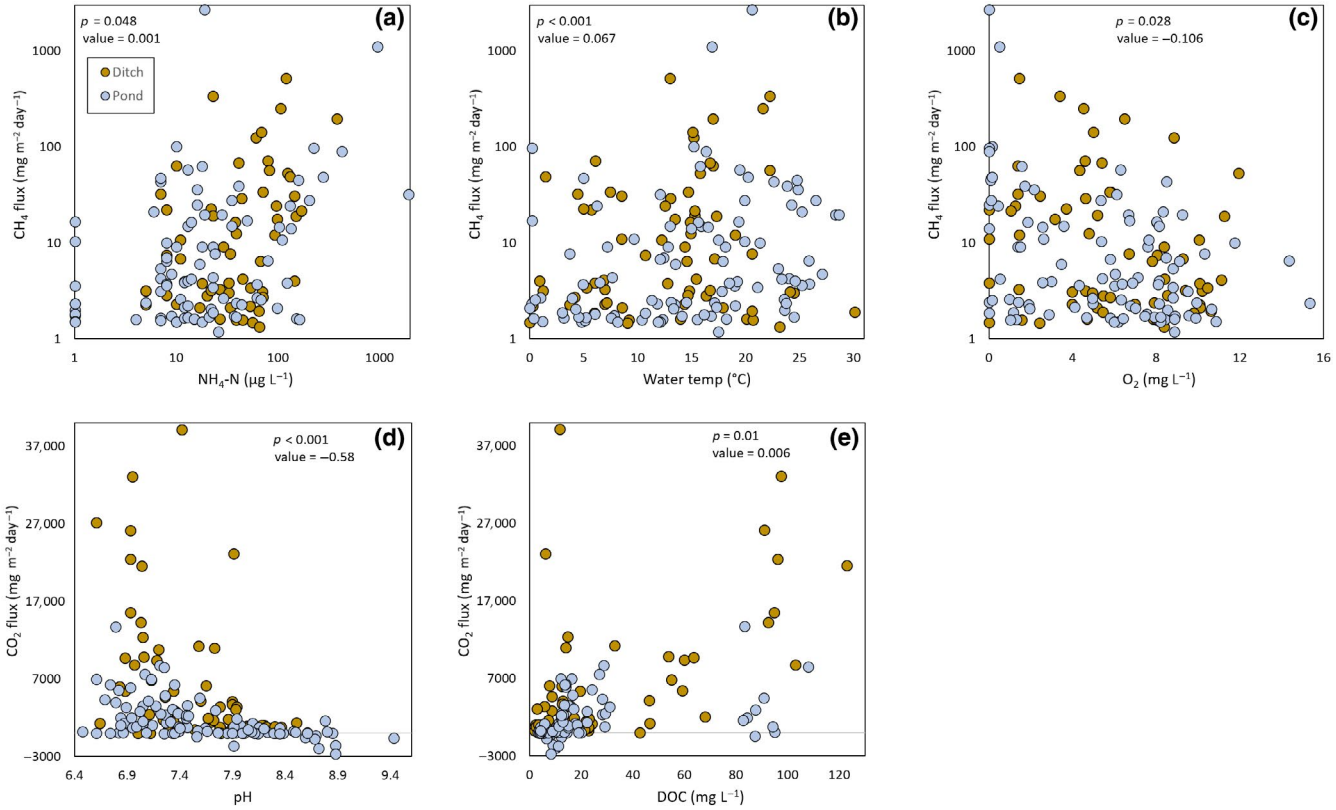

FIGURE 2 Scatter plots of GHG fluxes and environmental/biogeochemical variables (a–e) that were found to be significantly related to fluxes in the linear mixed effect models for the intensive monitoring. For each plot, the significance level and value of the relationship are shown. For graphical presentation of log-transformed data only (i.e. not for statistical testing), the following transformations were used: A value of 1.5 was added to daily CH_4 fluxes so all fluxes were positive; a value of 1 was added to NH_4-N concentrations to remove zero values. Flux measurements with no associated water chemistry sample are not included in the analysis

had almost twice the DOC concentration of ponds (22 vs. 12 mg L^{-1}, Figure 4g), but no difference in DOM quality as measured by $S_{275-295}$, $S_{350-400}$, SR and E2:E3 was identified (Figure 4h–k).

For all 160 waterbodies, mean GHG concentrations were 160.1 ± 62 µg C L^{-1} and 3.68 ± 0.4 mg C L^{-1} for CH_4 and CO_2, respectively, and a weak, positive correlation was found between the GHGs (Figure S13). These concentrations result in estimated mean (±SE) fluxes of 91.7 (±48) mg CH_4 m^{-2} day^{-1} and 3920 (±570) mg CO_2 m^{-2} day^{-1}. When converted to partial pressures, 17 samples had pCO_2 below atmospheric.

Mann–Whitney tests showed that there was no difference in dissolved CH_4 between ditches and ponds (Figure 4b), but that CO_2 was significantly higher in ditches (Figure 4a); this finding was also reflected in the linear models, where no significant effect of waterbody type was found for CH_4 but was found for CO_2 ($p < 0.001$). Mean concentrations of GHGs were significantly higher (by a factor of 2.1 for both GHGs) in waterbodies situated on peat soils compared to mineral soils (Table 2). There was no effect of season on CH_4 concentration, but differences were found for CO_2 (Table SI4), where mean summer concentrations were 2.79 (±0.56) mg C L^{-1} compared to 4.43 (±0.55) mg C L^{-1} for all other seasons ($p = 0.001$, $d = 0.35$, effect size = small). There was no significant difference between land uses for CH_4 but CO_2 concentrations were significantly lower for waterbodies in settlements when compared to grasslands and unmanaged land (Table 2).

The linear models found two significant relationships between CH_4 and various environmental variables, although the explanatory power was weak (adjusted $R^2 \leq 0.15$, Table SI3; Figure S14). CH_4 was significantly positively related to DOC (both for the entire data set and for the pond samples only) and negatively related to slope ratio (SR). The strength of the models was slightly higher for CO_2 (adjusted $R^2 \leq 0.31$, Table SI3). For the full data set, a negative relationship was found between pH and CO_2 concentrations, whilst a positive effect of EC was found (Figure S14). A negative effect of water temperature was the only significant variable for the ditch data set, whilst negative effects of pH and SR were found for the pond data set.

A comparison of our data with that synthesized from natural ponds and streams (Figure 5) showed that concentrations of both CH_4 and CO_2 were higher in artificial ponds for all pond sizes, except for the largest size class (10,000–100,000 m^2) where CO_2 concentrations were approximately the same. Concentrations of both GHGs decreased with increasing pond size.

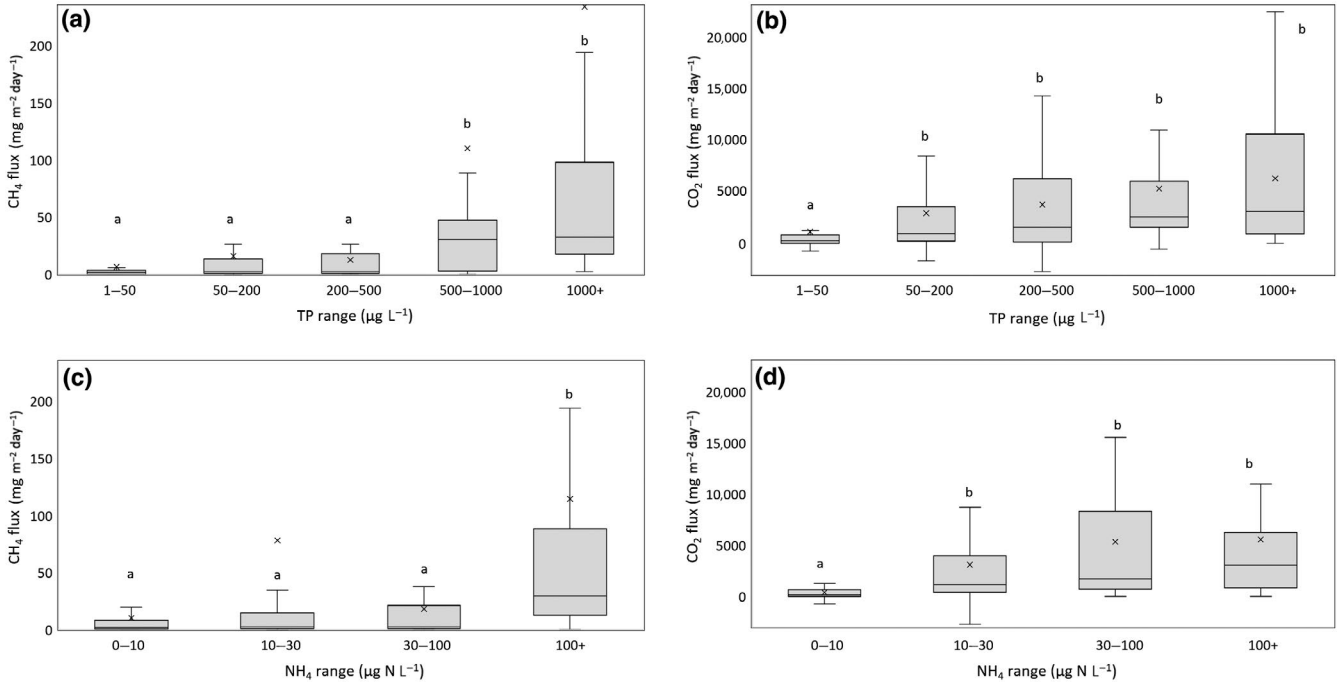

FIGURE 3 Boxplots of GHG fluxes grouped by TP (a, b) and NH_4 (c, d) concentration ranges, for the intensive monitoring. Boxes represent medians and IQR, whiskers mark minimum and maximum values, excluding outliers (calculated as box limits ±1.5 x IQR). Also shown are mean fluxes (x). Note that outliers extend far beyond the figure boundaries so for clarity are not shown but are included in Figure S11. Letters mark significant differences

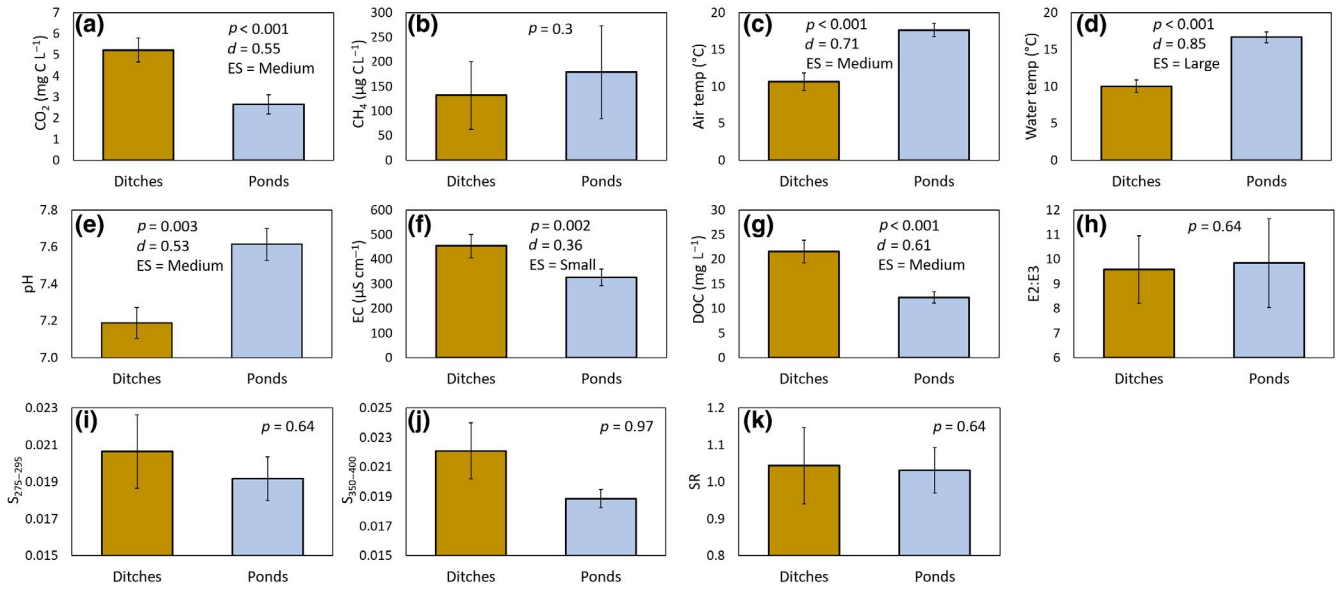

FIGURE 4 Means (± standard errors) for GHG concentrations (a, b), environmental variables (c, d), basic water chemistry (e, f) and DOC (g) and DOM (h–k) for ponds (n = 96) and ditches (n = 64) from the extensive survey. For each figure, the p value is shown and, if the result is significant, Cohen's d and the corresponding effect size are also given

4 | DISCUSSION

Our intensive monitoring and extensive survey give complementary evidence that small, artificial waterbodies are widespread and persistent emitters of CH_4 and CO_2 to the atmosphere. We find that nutrient concentrations (C, N and P) appear to exert a stimulatory effect on the concentrations and fluxes of both GHGs. Importantly, our results suggest that GHG emissions are considerably higher from ditches and artificial ponds when compared to their natural counterparts (streams and natural ponds respectively). We also show how, even when occupying only small total areas, ditches and artificial ponds can exert large effects on landscape-scale GHG balances. Below, we discuss our findings and their implications in detail.

TABLE 2 Mean and standard errors (SE) of GHGs from the extensive survey grouped by soil type and surrounded land use

		CO_2 (mg C L^{-1})		CH_4 (µg C L^{-1})	
Soil type	***n***	**Mean**	**SE**	**Mean**	**SE**
Peat	30	6.39	0.94	283	145
Mineral	130	3.06	0.38	132	69.6
Land use	***n***	**Mean**	**SE**	**Mean**	**SE**
Unmanaged land and forest	26	4.72	0.84	95.6	40.2
Cropland	18	3.36	0.64	17.5	8.1
Grassland	31	6.45	1.47	540	310
Settlement	85	2.42	0.27	71.8	23.9

Note: Soil type had a significant effect on CO_2 concentrations ($p < 0.001$, $d = 0.71$, effect size = medium) and CH_4 concentrations ($p = 0.041$, $d = 0.19$, effect size = very small). For land use, significant differences were found for CO_2 between settlement and unmanaged land and forest ($p = 0.003$, $d = 0.73$, effect size = medium) and settlement and grassland ($p = 0.003$, $d = 0.8$, effect size = large). No significant effect of land use was found for CH_4 ($p = 0.054$).

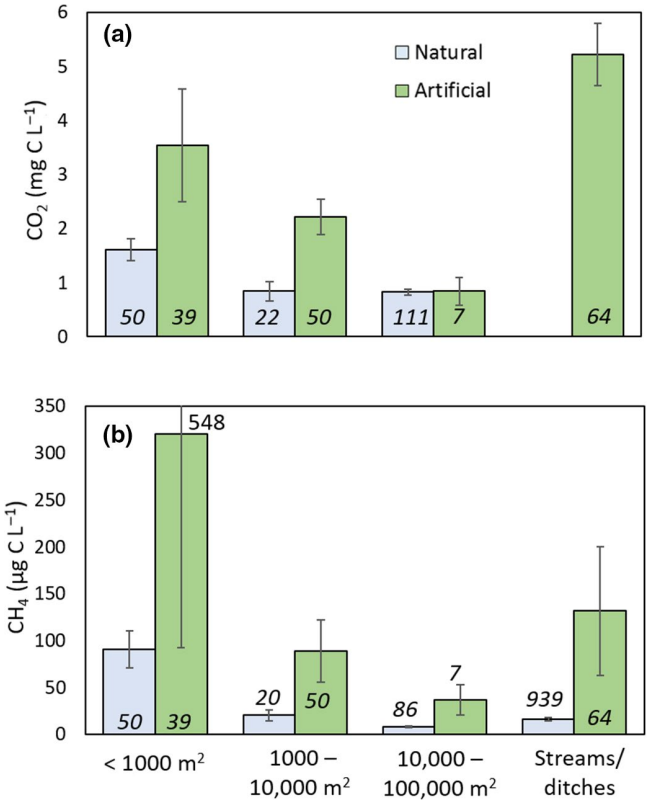

FIGURE 5 Comparisons of mean (± standard errors) dissolved CH_4 (a) and CO_2 (b) concentrations between artificial waterbodies (data from our extensive survey; green) and natural ponds (data from Holgerson & Raymond, 2016; blue) and streams (data from Stanley et al., 2016; blue). Pond GHGs are grouped by pond area. Italic numbers are sample sizes

4.1 | Drivers of GHG concentrations and fluxes

For both the intensive monitoring and extensive survey, we found significant relationships between GHG fluxes/concentrations and other variables. Negative relationships were found between pH and CO_2 fluxes/concentrations, and values of both were closer to zero at pH > 8, demonstrating the importance of carbonate buffering on CO_2 dynamics (Stets et al., 2017). Fluxes of CO_2 from the intensive monitoring also correlated positively with DOC concentration; such relationships have been observed previously for lakes (Hope et al., 1996; Sobek et al., 2003). High DOC concentrations can be considered a proxy for organic-rich soils such as peatlands, where high concentrations of pore water CH_4, CO_2 and DOC are transferred laterally into drainage waters and will fuel aquatic CO_2 evasion (Rasilo et al., 2017). An empirical relationship between DOC and CO_2 was not found in the extensive survey, but mean CO_2 and CH_4 concentrations in waterbodies on peat soils were double those on mineral soils, and there was a significant positive effect of DOC on CH_4 concentration.

For CH_4 flux, we observed a negative effect of O_2 concentration, and a positive effect of water temperature, reflecting the role that these factors play in regulating methanogenesis and methanotrophy (Segers, 1988). We also found a positive relationship between CH_4 fluxes and NH_4. We assume that NH_4 is essentially a signal of agricultural/urban run-off, which will deliver high inputs of N, P and labile organic matter into waterbodies thus fuelling CH_4 production. There were significantly higher CH_4 fluxes in waterbodies with the largest TP concentrations, in keeping with a wide body of literature showing higher emissions in nutrient-rich waterbodies including ditches and artificial ponds (Audet et al., 2020; Beaulieu et al., 2019; Herrero Ortega et al., 2019; Ollivier et al., 2019a; Peacock et al., 2017, 2019; Webb, Hayes, et al., 2019). However, high TP concentrations did not guarantee high fluxes, and this is in agreement with a recent synthesis of ditch CH_4 emissions (Peacock et al., 2021). We assume that high nutrient systems have the potential to emit large quantities of CH_4 provided that other factors are not limiting; for example, a eutrophic ditch in the boreal zone may have low fluxes, due to low temperatures inhibiting methanogenesis. Interestingly, fluxes of CO_2 were also smallest when concentrations of TP and NH_4 were lowest. It is often the case that higher aquatic nutrient levels lead

to lower CO_2 due to in situ primary production, but DelSontro et al. (2018) showed that TP and CO_2 were positively correlated in small waterbodies and suggested that this was due to TP being a proxy for terrestrial inputs of CO_2 and organic carbon.

For the extensive survey, GHG concentrations did not necessarily respond to the same drivers when the data set was split by waterbody type. The aforementioned relationships between DOC and CH_4, and pH and CO_2 were evident in the combined data set and in the pond data set but were not significant when ditches were considered in isolation. Indeed, we found no significant drivers of CH_4 concentrations for the ditch data set. This suggests that the mechanisms of GHG production and emission in ponds and ditches are not the same. With a limited data set, it is difficult to draw further conclusions, but one plausible mechanism is the difference in hydrology. Although rates are generally slow, ditches do often flow, in contrast to ponds which are more stagnant, and this may have implications for the delivery of nutrients or the addition of pollutants into waterbodies. Water level fluctuations may also be greater in ditches, and these have been implicated in controlling GHG dynamics previously (Peacock et al., 2017). Interestingly, for the pond data set, we found negative relationships between slope ratio (SR) and both CH_4 and CO_2, suggesting that GHG concentrations are related to DOM composition; a finding reported previously (Bodmer et al., 2016; Peacock et al., 2017; Turner et al., 2016). Without detailed molecular characterization, we cannot know the precise mechanistic link between DOM and GHGs, but this warrants further investigation.

Finally, we note that all our measurements were conducted during daytime. For the intensive monitoring, on each sampling date waterbodies were sampled in a different order so as not to introduce any systematic bias, and so comparisons between waterbodies are therefore likely to be robust. However, the lack of night-time data could introduce uncertainty into our annual flux calculations. For instance, substantially larger night-time fluxes and concentrations of CH_4 and CO_2 have been observed in arable ditches, urban ponds and channelized streams (Deng et al., 2020; Harrison et al., 2005; Natchimuthu et al., 2014) which are often attributed to variations in photosynthesis and O_2 concentrations. However, some pond and lake studies have found lower CH_4 fluxes during night-time due to decreased wind speeds reducing gas exchange (Natchimuthu et al., 2014; Sieczko et al., 2020) and others have found no clear diel pattern in pond CH_4 or CO_2 (Miller et al., 2019), especially during winter sampling (Ollivier et al., 2019b). No diel measurements were made during our study and so we cannot be sure to what extent, if any, our annual fluxes have been under- or overestimated, but night-time measurements should be a priority for future work.

4.2 | Fluxes from intensive monitoring

Over the measured annual cycle, our results show that all ditches and ponds were net emitters of CH_4 and, except one pond, all were net emitters of CO_2. Highest diffusive fluxes for both GHGs were observed during the warmer summer months, in agreement with other ditch (Peacock et al., 2017; Van den Pol-van Dasselaar et al., 1999) and pond studies (e.g. Baker-Blocker et al., 1997; Natchimuthu et al., 2014; Ollivier et al., 2019b). The mean daily diffusive flux of CH_4 from all waterbodies (41.3 mg CH_4 m^{-2} day^{-1}) was similar to that from other artificial ditches and ponds in temperate zones (20–90 mg CH_4 m^{-2} day^{-1}, Audet et al., 2020; McPhillips et al., 2016; McPhillips & Walter, 2015; Peacock et al., 2019; Stadmark & Leonardson, 2005; van Bergen et al., 2019). Considering that diffusive fluxes from subtropical zones are generally higher (50–350 mg CH_4 m^{-2} day^{-1}, Gorsky et al., 2019; Ollivier et al., 2019a; Panneer Selvam et al., 2014; Purvaja & Ramesh, 2001), and that we found a positive relationship between water temperature and CH_4 flux, it is likely that mean annual temperature is an important controller on the magnitude of CH_4 evasion. Our mean daily flux of CO_2 from ditches and ponds was 3640 mg CO_2 m^{-2} day^{-1}; the same magnitude as fluxes from subtropical ponds (1070–2950 mg CO_2 m^{-2} day^{-1}, Gorsky et al., 2019; Ollivier et al., 2019a; Panneer Selvam et al., 2014), grassed temperate urban ditches (8600 mg CO_2 m^{-2} day^{-1}, McPhillips et al., 2016) and temperate urban ponds (2300–3480 mg CO_2 m^{-2} day^{-1}, Audet et al., 2020; van Bergen et al., 2019).

The lowest annual diffusive CH_4 flux (0.1 g CH_4 m^{-2} $year^{-1}$) was from the peatland ditch in felled forest, and the pond in the same felled forest also had a low flux (0.94 g CH_4 m^{-2} $year^{-1}$). These fluxes are low when compared to literature values for ditches (mean = 69 g CH_4 m^{-2} $year^{-1}$, Evans et al., 2016). Both the pond and ditch dried out for one sampling date, and mean water depths were, respectively, 10 and 4 cm. Similarly low fluxes of 0.1 and 1.3 g CH_4 m^{-2} $year^{-1}$ have been reported elsewhere from peatland ditches that dry out (Bravo et al., 2018; Järveoja et al., 2016). Dry sediments will allow the formation of oxidized potential electron acceptors and some of the highest NO_3 concentrations were measured at these sites. The presence of these alternate electron acceptors likely hampers CH_4 production and promotes CH_4 oxidation.

Although limited in extent, our ebullition data suggest that this is an important pathway of CH_4 release in both ponds and ditches. The mean flux of all observed bubble events was 257 mg CH_4 m^{-2} day^{-1}, which is higher than the mean diffusive flux, although our ebullition data are limited to 33 observations from 10 sites. Our ebullition flux is similar to that from temperate artificial ponds (270–470 mg CH_4 m^{-2} day^{-1}, Herrero Ortega et al., 2019; van Bergen et al., 2019) and ditches (140 mg CH_4 m^{-2} day^{-1}, Vermaat et al., 2011), but smaller than that from subtropical ponds (450 mg CH_4 m^{-2} day^{-1}, Grinham et al., 2018). Aben et al. (2017) found that temperature exerted a strong positive effect on ebullition, and thus, the contribution of bubbles to emissions is likely to be greatest in summer, particularly in eutrophic waterbodies (Davidson et al., 2018). Our data support this, with the occurrence of ebullition events being strongly related to air temperature. Long-term chamber deployments or bubble traps would be necessary to quantify the annual bubble flux. This should be a priority considering that some authors have found that ebullition is a dominant pathway for CH_4 release in ponds and ditches (Grinham et al.,

2018; Herrero Ortega et al., 2019; Panneer Selvam et al., 2014; van Bergen et al., 2019; Vermaat et al., 2011), although there are exceptions; for example, zero/low ebullitive fluxes reported in boreal peatland ditches (Minkkinen et al., 1997). Nevertheless, the exclusion of ebullitive fluxes from our annual emissions means that they may be significantly underestimated.

4.3 | Concentrations/fluxes from extensive survey

CO_2 concentrations were significantly higher in ditches than ponds, but there was no significant difference in CH_4 concentrations. When converted to fluxes, emissions were similar from the intensive monitoring and the extensive survey: Respective mean fluxes ± SEs were 41 (±17) and 92 (±48) mg CH_4 m^{-2} day^{-1}, and 3640 (±480) and 3920 (±570) mg CO_2 m^{-2} day^{-1}. This suggests that our intensively monitored sites may be reasonably representative of artificial waterbodies across larger geographical scales. Here, we compare our international data to that from other large-scale national/global syntheses. Concentrations of CH_4 in ditches were an order of magnitude higher than concentrations in global streams (Stanley et al., 2016) and, across the three size classes, CH_4 concentrations in our artificial ponds were 3.5–4.5 times larger than those in global natural ponds (Holgerson & Raymond, 2016). Ditch CO_2 concentrations were double those in Swedish streams (Wallin et al., 2018), and artificial pond CO_2 concentrations were approximately 2.5 times larger than those from natural ponds, except at the largest size class (10,000–100,000 m^2) where there was no difference (Holgerson & Raymond, 2016). We therefore conclude that the processes leading to GHG production tend to be intensified in artificial ponds when compared to natural waterbodies. This can be due to a variety of reasons; for example, hydrological management such as water abstraction has the potential to alter water retention times and dredging or clearing will disrupt natural processes. Ditches and ponds are often situated in agricultural or urban environments, where they may receive more diffuse or point source nutrients inputs, whereas a higher proportion of streams and ponds are located in natural areas. Although not measured here, GHG emissions from heavily modified natural waterbodies (e.g. polluted ponds in agricultural landscapes) are also likely to be large. However, regardless of the extent of anthropogenic eutrophication, emissions from these waterbodies are not included in IPCC inventories. The exception is if a natural waterbody is dammed to increase its surface area; all emissions from such a waterbody must be accounted for.

We acknowledge that because waterbodies were only sampled once, our extensive survey results ignore the seasonal variation found in our intensive monitoring (e.g. higher GHG flux in summer). Because 45% of extensive survey samples were taken during summer, results may be skewed towards higher CH_4 concentrations due to elevated temperatures. However, for the extensive survey, no seasonal differences were found for CH_4, and concentrations of CO_2 were significantly lower in summer when compared to spring and autumn (Table SI4).

We found that 89% of our sampled waterbodies emitted CO_2. In comparison, other surveys have reported that 52%–90% of artificial ponds studied emitted CO_2 (Ollivier et al., 2019a; Peacock et al., 2019; Webb, Leavitt, et al., 2019). After converting CH_4 emissions to CO_2 equivalents, using the sustained-flux global warming potential (SGWP) over 100 years (=45, Neubauer & Megonigal, 2015), the percentage of waterbodies acting as a GHG source increased to 93% (i.e. only 12 waterbodies were GHG sinks). Finally, the inclusion of N_2O (not measured here) could further change the source versus sink behaviour of the studied waterbodies. N_2O emissions are not always large from ditches and ponds (McPhillips & Walter, 2015; Ollivier et al., 2019b), and uptake is sometimes observed (Audet et al., 2020; Webb et al., 2021; Webb, Hayes, et al., 2019), but N_2O is a powerful GHG that can be fuelled by elevated nutrient concentrations and emitted by ditches and ponds (Audet et al., 2017; Peacock et al., 2017; Reay et al., 2003; Webb et al., 2021). Given the high SGWP for N_2O (=270), such emissions could outweigh the climatic benefits of any CO_2 sink behaviour.

4.4 | Implications

Our findings show that small artificial waterbodies play an active role in carbon cycling and are overwhelmingly net emitters of CO_2 and CH_4 to the atmosphere. This finding holds true for ditches and ponds across different climate zones and land uses; indeed, we found no difference in CH_4 concentrations between different land uses, suggesting that site-specific characteristics are more important than generalized management regimes. A primary implication of this finding is that these small waterbodies have the potential to influence field-scale, regional, national and even global carbon and GHG budgets. At the field scale, and with the exception of mires (which can emit CH_4 even when ditched), terrestrial soils are generally small CH_4 sinks, with uptake by temperate grasslands, croplands and forests in the region of 0.2–1.5 mg CH_4 m^{-2} day^{-1} (Dutaur & Verchot, 2007; Glatzel & Stahr, 2001; Saggar et al., 2008; Wang et al., 2014). The mean CH_4 emission for forest and agricultural ditches and ponds from our combined data sets is 82 mg m^{-2} day^{-1}. Assuming the aforementioned terrestrial soil uptake, the field ceases being a net CH_4 sink when the fraction of it occupied by ditches ($Frac_{ditch}$, sensu Evans et al., 2016) or ponds is 0.002–0.018 (0.2%–1.8%); values well within real world examples (Peacock et al., 2021). In heavily ditched landscapes, the cumulative effect of ditch emissions can be large; for example, it has been estimated that ditches are responsible for 16% of national CH_4 emissions in the Netherlands (Koschorreck et al., 2020) and 9% in Finland (Peacock et al., 2021).

Ponds can also occupy cumulatively large areas, and in some landscapes, their total surface area is approximately the same as that of larger reservoirs (Ollivier et al., 2019a). The effect of this is that GHG emissions from artificial ponds are non-trivial contributors to regional and national budgets (Grinham et al., 2018; Ollivier et al., 2019a). Furthermore, in agreement with measurements from natural ponds (Holgerson & Raymond, 2016), we found an inverse

relationship between emissions of CH_4 and CO_2 and pond area. Smaller ponds are difficult to accurately map (Sivanpillai & Miller, 2010), and therefore, uncertainty remains in upscaled estimates, from the regional to the global scale. Similar uncertainty exists for mapping ditches, rendering upscaled ditch GHG emissions uncertain (Peacock et al., 2021), although recent developments in methods appear promising (e.g. Connolly & Holden, 2017). Accurately upscaling GHG emissions from ponds and ditches are important because, unlike emissions from natural lakes and streams, fluxes from artificial waterbodies must be considered anthropogenic in nature (IPCC, 2019). A comparison of our data with literature values suggests that emissions from artificial waterbodies are on average four times larger than those from analogous natural waterbodies, and we therefore argue that emissions from natural systems cannot be used as proxies for those from artificial waterbodies. Current IPCC emission factors for ditches and ponds are uncertain, with respective 95% confidence intervals of 26–67 and 12–23 g CH_4 m^{-2} $year^{-1}$. At present, there are insufficient data to disaggregate emission factors by climate zone or land use, which makes GHG accounting for these systems difficult. We therefore advocate that multi-year, multi-site measurements from ditches and ponds are needed across a range of climate zones and land uses to fully understand their role in climatic warming.

ACKNOWLEDGEMENTS

No specific grant funded this work. DB was supported by the European Research Council (ERC grant 725546) and FORMAS (grant 2018-01794). MP acknowledges funding from Formas (grant 2020-00950). AP acknowledges funding from NERC (LOCATE—Land Ocean Carbon Transfer project, grant NE/N018087/1). Data are available at https://doi.org/10.6084/m9.figshare.14784852. We thank the Geochemical Laboratory at the Swedish University of Agricultural Sciences for water chemistry analysis. We thank Ekatierina Bielova at the Department of Ecology, Faculty of Biology and Ecology, Yanka Kupala State University of Grodno, Belarus for sampling assistance. We thank two anonymous reviewers for their constructive comments that helped to improve the manuscript.

DATA AVAILABILITY STATEMENT

The data from this paper are available at https://doi.org/10.6084/m9.figshare.14784852.

ORCID

M. Peacock https://orcid.org/0000-0002-3086-2854
J. Audet https://orcid.org/0000-0001-5839-8793
C. D. Evans https://orcid.org/0000-0002-7052-354X
P. Zieliński https://orcid.org/0000-0002-1833-950X

REFERENCES

Aben, R. C., Barros, N., van Donk, E., Frenken, T., Hilt, S., Kazanjian, G., Lamers, L. P. M., Peeters, E. T. H. M., Roelofs, J. G. M., de Senerpont Domis, L. N., Stephan, S., Velthuis, M., Van de Waal, D. B., Wik, M., Thornton, B. F., Wilkinson, J., DelSontro, T., & Kosten, S. (2017). Cross continental increase in methane ebullition under climate change. *Nature Communications*, *8*, 1682. https://doi.org/10.1038/s41467-017-01535-y

Audet, J., Carstensen, M. V., Hoffmann, C. C., Lavaux, L., Thiemer, K., & Davidson, T. A. (2020). Greenhouse gas emissions from urban ponds in Denmark. *Inland Waters*, *10*(3), 373–385. https://doi.org/10.1080/20442041.2020.1730680

Audet, J., Wallin, M. B., Kyllmar, K., Andersson, S., & Bishop, K. (2017). Nitrous oxide emissions from streams in a Swedish agricultural catchment. *Agriculture, Ecosystems & Environment*, *236*, 295–303. https://doi.org/10.1016/j.agee.2016.12.012

Baker-Blocker, A., Donahue, T. M., & Mancy, K. H. (1997). Methane flux from wetland areas. *Tellus*, *29*(3), 245–250.

Bastviken, D., Cole, J., Pace, M., & Tranvik, L. (2004). Methane emissions from lakes: Dependence of lake characteristics, two regional assessments, and a global estimate. *Global Biogeochemical Cycles*, *18*, GB4009. https://doi.org/10.1029/2004GB002238

Bastviken, D., Sundgren, I., Natchimuthu, S., Reyier, H., & Gålfalk, M. (2015). Cost-efficient approaches to measure carbon dioxide (CO_2) fluxes and concentrations in terrestrial and aquatic environments using mini loggers. *Biogeosciences*, *12*(12), 3849–3859.

Bastviken, D., Tranvik, L. J., Downing, J. A., Crill, P. M., & Enrich-Prast, A. (2011). Freshwater methane emissions offset the continental carbon sink. *Science*, *331*(6013), 50. https://doi.org/10.1126/science.1196808

Beaulieu, J. J., DelSontro, T., & Downing, J. A. (2019). Eutrophication will increase methane emissions from lakes and impoundments during the 21st century. *Nature Communications*, *10*(1), 1–5. https://doi.org/10.1038/s41467-019-09100-5

Benjamini, Y., & Hochberg, Y. (1995). Controlling the false discovery rate: A practical and powerful approach to multiple testing. *Journal of the Royal Statistical Society: Series B (Methodological)*, *57*, 289–300. https://doi.org/10.1111/j.2517-6161.1995.tb02031.x

Biggs, J., Williams, P., Whitfield, M., Nicolet, P., & Weatherby, A. (2005). 15 years of pond assessment in Britain: Results and lessons learned from the work of Pond Conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *15*(6), 693–714. https://doi.org/10.1002/aqc.745

Blaszczak, J. R., Steele, M. K., Badgley, B. D., Heffernan, J. B., Hobbie, S. E., Morse, J. L., Rivers, E. N., Hall, S. J., Neill, C., Pataki, D. E., Groffman, P. M., & Bernhardt, E. S. (2018). Sediment chemistry of urban stormwater ponds and controls on denitrification. *Ecosphere*, *9*(6), e02318. https://doi.org/10.1002/ecs2.2318

Bodmer, P., Heinz, M., Pusch, M., Singer, G., & Premke, K. (2016). Carbon dynamics and their link to dissolved organic matter quality across contrasting stream ecosystems. *Science of the Total Environment*, *553*, 574–586. https://doi.org/10.1016/j.scitotenv.2016.02.095

Bravo, T. G., Rochefort, L., & Strack, M. (2018). The impact of a black spruce (*Picea mariana*) plantation on carbon exchange in a cutover peatland in Western Canada. *Canadian Journal of Forest Research*, *48*, 388–398.

Brown, C. D., Turner, N., Hollis, J., Bellamy, P., Biggs, J., Williams, P., Arnold, D., Pepper, T., & Maund, S. (2006). Morphological and physico-chemical properties of British aquatic habitats potentially exposed to pesticides. *Agriculture, Ecosystems & Environment*, *113*(1–4), 307–319. https://doi.org/10.1016/j.agee.2005.10.015

Carter, H. T., Tipping, E., Koprivnjak, J. F., Miller, M. P., Cookson, B., & Hamilton-Taylor, J. (2012). Freshwater DOM quantity and quality from a two-component model of UV absorbance. *Water Research*, *46*(14), 4532–4542. https://doi.org/10.1016/j.watres.2012.05.021

Ciais, P., Sabine, C., Bala, G., Bopp, L., Brovkin, V., Canadell, J., Chhabra, A., DeFries, R., Galloway, J., Heimann, M., Jones, C., Le Quéré, C., Myneni, R. B., Piao, S., & Thornton, P. (2013). Carbon and other biogeochemical cycles. In T. F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex, & P. M. Midgley

(Eds.), *Climate change 2013: The physical science basis. Contribution of working group I to the fifth assessment report of the Intergovernmental Panel on Climate Change* (pp. 465–570). Cambridge University Press.

Clifford, C. C., & Heffernan, J. B. (2018). Artificial aquatic ecosystems. *Water*, *10*, 1096. https://doi.org/10.3390/w10081096

Cohen, J. (1988). *Statistical power analysis for the behavioral sciences*. Routledge.

Cole, J. J., Prairie, Y. T., Caraco, N. F., McDowell, W. H., Tranvik, L. J., Striegl, R. G., Duarte, C. M., Kortelainen, P., Downing, J. A., Middelburg, J. J., & Melack, J. (2007). Plumbing the global carbon cycle: Integrating inland waters into the terrestrial carbon budget. *Ecosystems*, *10*(1), 172–185. https://doi.org/10.1007/s10021-006-9013-8

Connolly, J., & Holden, N. M. (2017). Detecting peatland drains with object based image analysis and Geoeye-1 imagery. *Carbon Balance and Management*, *12*(1), 7.

Cooke, M. C. (1892). *Natural history rambles ponds and ditches*. Society for Promoting Christian Knowledge.

Davidson, T. A., Audet, J., Jeppesen, E., Landkildehus, F., Lauridsen, T. L., Søndergaard, M., & Syväranta, J. (2018). Synergy between nutrients and warming enhances methane ebullition from experimental lakes. *Nature Climate Change*, *8*, 156–160. https://doi.org/10.1038/s41558-017-0063-z

DelSontro, T., Beaulieu, J. J., & Downing, J. A. (2018). Greenhouse gas emissions from lakes and impoundments: Upscaling in the face of global change. *Limnology and Oceanography Letters*, *3*(3), 64–75. https://doi.org/10.1002/lol2.10073

Deng, O., Li, X. I., Deng, L., Zhang, S., Gao, X., Lan, T., Zhou, W., Tian, D., Xiao, Y., Yang, J., Ou, D., & Luo, L. (2020). Emission of CO2 and CH4 from a multi-ditches system in rice cultivation region: Flux, temporal-spatial variation and effect factors. *Journal of Environmental Management*, *270*, 110918. https://doi.org/10.1016/j.jenvman.2020.110918

Downing, J. A. (2010). Emerging global role of small lakes and ponds. *Limnetica*, *29*, 9–24.

Drake, T. W., Raymond, P. A., & Spencer, R. G. (2018). Terrestrial carbon inputs to inland waters: A current synthesis of estimates and uncertainty. *Limnology and Oceanography Letters*, *3*(3), 132–142. https://doi.org/10.1002/lol2.10055

Dutaur, L., & Verchot, L. V. (2007). A global inventory of the soil CH_4 sink. *Global Biogeochemical Cycles*, *21*(4).

Evans, C. D., Renou-Wilson, F., & Strack, M. (2016). The role of waterborne carbon in the greenhouse gas balance of drained and re-wetted peatlands. *Aquatic Sciences*, *78*(3), 573–590. https://doi.org/10.1007/s00027-015-0447-y

Gao, Y., Liu, X., Yi, N., Wang, Y., Guo, J., Zhang, Z., & Yan, S. (2013). Estimation of N_2 and N_2O ebullition from eutrophic water using an improved bubble trap device. *Ecological Engineering*, *57*, 403–412. https://doi.org/10.1016/j.ecoleng.2013.04.020

Glatzel, S., & Stahr, K. (2001). Methane and nitrous oxide exchange in differently fertilised grassland in southern Germany. *Plant and Soil*, *231*(1), 21–35.

Gogo, S., Guimbaud, C., Laggoun-Défarge, F., Catoire, V., & Robert, C. (2011). In situ quantification of CH_4 bubbling events from a peat soil using a new infrared laser spectrometer. *Journal of Soils and Sediments*, *11*(4), 545–551. https://doi.org/10.1007/s11368-011-0338-3

Gorsky, A. L., Racanelli, G. A., Belvin, A. C., & Chambers, R. M. (2019). Greenhouse gas flux from stormwater ponds in southeastern Virginia (USA). *Anthropocene*, *28*, 100218. https://doi.org/10.1016/j.ancene.2019.100218

Grinham, A., Albert, S., Deering, N., Dunbabin, M., Bastviken, D., Sherman, B., Lovelock, C. E., & Evans, C. D. (2018). The importance of small artificial water bodies as sources of methane emissions in Queensland, Australia. *Hydrology and Earth System Sciences*, *22*(10), 5281–5298. https://doi.org/10.5194/hess-22-5281-2018

Harrison, J. A., Matson, P. A., & Fendorf, S. E. (2005). Effects of a diel oxygen cycle on nitrogen transformations and greenhouse gas emissions in a eutrophied subtropical stream. *Aquatic Sciences*, *67*, 308–315. https://doi.org/10.1007/BF02506134

Helms, J. R., Stubbins, A., Ritchie, J. D., Minor, E. C., Kieber, D. J., & Mopper, K. (2008). Absorption spectral slopes and slope ratios as indicators of molecular weight, source, and photobleaching of chromophoric dissolved organic matter. *Limnology and Oceanography*, *53*(3), 955–969. https://doi.org/10.4319/lo.2008.53.3.0955

Herrero Ortega, S., Romero González-Quijano, C., Casper, P., Singer, G. A., & Gessner, M. O. (2019). Methane emissions from contrasting urban freshwaters: Rates, drivers, and a whole-city footprint. *Global Change Biology*, *25*, 4234–4243. https://doi.org/10.1111/gcb.14799

Holgerson, M. A., & Raymond, P. A. (2016). Large contribution to inland water CO_2 and CH_4 emissions from very small ponds. *Nature Geoscience*, *9*(3), 222–226. https://doi.org/10.1038/ngeo2654

Hope, D., Kratz, T. K., & Riera, J. L. (1996). Relationship between pCO2 and dissolved organic carbon in northern Wisconsin lakes. *Journal of Environmental Quality*, *25*(6), 1442–1445.

Hope, D., Palmer, S. M., Billett, M. F., & Dawson, J. J. (2004). Variations in dissolved CO_2 and CH_4 in a first-order stream and catchment: An investigation of soil–stream linkages. *Hydrological Processes*, *18*(17), 3255–3275. https://doi.org/10.1002/hyp.5657

IPCC. (2019). Chapter 7: Wetlands. In C. E. Lovelock, C. Evans, N. Barros, Y. Prairie, J. Alm, D. Bastviken, J. J. Beaulieu, M. Garneau, A. Harby, J. Harrison, D. Pare, H. L. Raadal, B. Sherman, C. Zhang, S. M. Ogle, A. Grinham, B. Deemer, M. Aurelio dos Santos, S. Kosten, … V. Stepanenko (Eds.), *Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 4: Agriculture, forestry and other land use (AFOLU)* (pp. 7.1–7.52). Author.

Järveoja, J., Peichl, M., Maddison, M., Teemusk, A., & Mander, Ü. (2016). Full carbon and greenhouse gas balances of fertilized and nonfertilized reed canary grass cultivations on an abandoned peat extraction area in a dry year. *Global Change Biology Bioenergy*, *8*, 952–968. https://doi.org/10.1111/gcbb.12308

Juutinen, S., Rantakari, M., Kortelainen, P., Huttunen, J. T., Larmola, T., Alm, J., Silvola, J., & Martikainen, P. J. (2009). Methane dynamics in different boreal lake types. *Biogeosciences*, *6*, 209–223. https://doi.org/10.5194/bg-6-209-2009

Kifner, L. H., Calhoun, A. J., Norton, S. A., Hoffmann, K. E., & Amirbahman, A. (2018). Methane and carbon dioxide dynamics within four vernal pools in Maine, USA. *Biogeochemistry*, *139*(3), 275–291. https://doi.org/10.1007/s10533-018-0467-5

Koschorreck, M., Downing, A. S., Hejzlar, J., Marcé, R., Laas, A., Arndt, W. G., Keller, P. S., Smolders, A. J., van Dijk, G., & Kosten, S. (2020). Hidden treasures: Human-made aquatic ecosystems harbour unexplored opportunities. *Ambio*, *49*(2), 531–540. https://doi.org/10.1007/s13280-019-01199-6

Kosten, S., Piñeiro, M., de Goede, E., de Klein, J., Lamers, L. P., & Ettwig, K. (2016). Fate of methane in aquatic systems dominated by free-floating plants. *Water Research*, *104*, 200–207. https://doi.org/10.1016/j.watres.2016.07.054

Kuhn, M., Lundin, E. J., Giesler, R., Johansson, M., & Karlsson, J. (2018). Emissions from thaw ponds largely offset the carbon sink of northern permafrost wetlands. *Scientific Reports*, *8*, 9535. https://doi.org/10.1038/s41598-018-27770-x

Liu, X., Gao, Y., Zhang, Z., Luo, J., & Yan, S. (2017). Sediment-water methane flux in a eutrophic pond and primary influential factors at different time scales. *Water*, *9*(8), 601. https://doi.org/10.3390/w9080601

Lorke, A., Bodmer, P., Noss, C., Alshboul, Z., Koschorreck, M., Somlai-Haase, C., Bastviken, D., Flury, S., Mcginnis, D. F., Maeck, A., & Müller, D. (2015). Drifting versus anchored flux chambers for measuring greenhouse gas emissions from running waters. *Biogeosciences*, *12*(23), 7013–7024.

Martinez-Cruz, K., Gonzalez-Valencia, R., Sepulveda-Jauregui, A., Plascencia-Hernandez, F., Belmonte-Izquierdo, Y., & Thalasso, F. (2017). Methane emission from aquatic ecosystems of Mexico City. *Aquatic Sciences*, *79*(1), 159–169. https://doi.org/10.1007/s00027-016-0487-y

McPhillips, L. E., Groffman, P. M., Schneider, R. L., & Walter, M. T. (2016). Nutrient cycling in grassed roadside ditches and lawns in a suburban watershed. *Journal of Environmental Quality*, *45*, 1901–1909. https://doi.org/10.2134/jeq2016.05.0178

McPhillips, L., & Walter, M. T. (2015). Hydrologic conditions drive denitrification and greenhouse gas emissions in stormwater detention basins. *Ecological Engineering*, *85*, 67–75. https://doi.org/10.1016/j.ecoleng.2015.10.018

Miller, B. L., Chen, H., He, Y., Yuan, X., & Holtgrieve, G. W. (2019). Magnitudes and drivers of greenhouse gas fluxes in floodplain ponds during drawdown and inundation by the Three Gorges Reservoir. *Journal of Geophysical Research: Biogeosciences*, *124*, 2499–2517. https://doi.org/10.1029/2018JG004701

Minkkinen, K., Laine, J., Nykänen, H., & Martikainen, P. J. (1997). Importance of drainage ditches in emissions of methane from mires drained for forestry. *Canadian Journal of Forest Research*, *27*, 949–952. https://doi.org/10.1139/x97-016

Natchimuthu, S., Paneer Selvam, B., & Bastviken, D. (2014). Influence of weather variables on methane and carbon dioxide flux from a shallow pond. *Biogeochemistry*, *119*, 403–413. https://doi.org/10.1007/s10533-014-9976-z

Neubauer, S. C., & Megonigal, J. P. (2015). Moving beyond global warming potentials to quantify the climatic role of ecosystems. *Ecosystems*, *18*(6), 1000–1013. https://doi.org/10.1007/s10021-015-9879-4

Nieminen, M., Palviainen, M., Sarkkola, S., Laurén, A., Marttila, H., & Finér, L. (2018). A synthesis of the impacts of ditch network maintenance on the quantity and quality of runoff from drained boreal peatland forests. *Ambio*, *47*(5), 523–534. https://doi.org/10.1007/s13280-017-0966-y

Oliveira-Junior, E. S., Tang, Y., van den Berg, S. J., Cardoso, S. J., Lamers, L. P., & Kosten, S. (2018). The impact of water hyacinth (*Eichhornia crassipes*) on greenhouse gas emission and nutrient mobilization depends on rooting and plant coverage. *Aquatic Botany*, *145*, 1–9. https://doi.org/10.1016/j.aquabot.2017.11.005

Ollivier, Q. R., Maher, D. T., Pitfield, C., & Macreadie, P. I. (2019a). Punching above their weight: Large release of greenhouse gases from small agricultural dams. *Global Change Biology*, *25*(2), 721–732. https://doi.org/10.1111/gcb.14477

Ollivier, Q. R., Maher, D. T., Pitfield, C., & Macreadie, P. I. (2019b). Winter emissions of CO_2, CH_4, and N_2O from temperate agricultural dams: Fluxes, sources, and processes. *Ecosphere*, *10*(11), e02914.

Panneer Selvam, B., Natchimuthu, S., Arunachalam, L., & Bastviken, D. (2014). Methane and carbon dioxide emissions from inland waters in India–implications for large scale greenhouse gas balances. *Global Change Biology*, *20*(11), 3397–3407. https://doi.org/10.1111/gcb.12575

Peacock, M., Audet, J., Bastviken, D., Futter, M. N., Gauci, V., Grinham, A. R., Harrison, J. A., Kent, M. S., Kosten, S., Lovelock, C. E., Veraart, A. J., & Evans, C. D. (2021). Global importance of methane emissions from drainage ditches and canals. *Environmental Research Letters*. https://doi.org/10.1088/1748-9326/abeb3.

Peacock, M., Audet, J., Jordan, S., Smeds, J., & Wallin, M. B. (2019). Greenhouse gas emissions from urban ponds are driven by nutrient status and hydrology. *Ecosphere*, *10*, e02643. https://doi.org/10.1002/ecs2.2643

Peacock, M., Ridley, L. M., Evans, C. D., & Gauci, V. (2017). Management effects on greenhouse gas dynamics in fen ditches. *Science of the Total Environment*, *578*, 601–612. https://doi.org/10.1016/j.scitotenv.2016.11.005

Peuravuori, J., & Pihlaja, K. (1997). Molecular size distribution and spectroscopic properties of aquatic humic substances. *Analytica Chimica Acta*, *337*, 133–149. https://doi.org/10.1016/S0003-2670(96)00412-6

Pinheiro, J., Bates, D., Debroy, S., Sarkar, D., R Development Core Team. (2012). nlme: Linear and nonlinear mixed effects models. R package.

Purvaja, R., & Ramesh, R. (2001). Natural and anthropogenic methane emission from coastal wetlands of South India. *Environmental Management*, *27*(4), 547–557. https://doi.org/10.1007/s002670010169

R Development Core Team. (2020). *R: A language and environment for statistical computing*. https://www.R-project.org/

Rasilo, T., Hutchins, R. H., Ruiz-González, C., & Del Giorgio, P. A. (2017). Transport and transformation of soil-derived CO_2, CH_4 and DOC sustain CO_2 supersaturation in small boreal streams. *Science of the Total Environment*, *579*, 902–912. https://doi.org/10.1016/j.scitotenv.2016.10.187

Raymond, P. A., Hartmann, J., Lauerwald, R., Sobek, S., McDonald, C., Hoover, M., Butman, D., Striegl, R., Mayorga, E., Humborg, C., & Kortelainen, P. (2013). Global carbon dioxide emissions from inland waters. *Nature*, *503*(7476), 355–359.

Reay, D. S., Smith, K. A., & Edwards, A. C. (2003). Nitrous oxide emission from agricultural drainage waters. *Global Change Biology*, *9*(2), 195–203. https://doi.org/10.1046/j.1365-2486.2003.00584.x

Rosentreter, J. A., Borges, A. V., Deemer, B. R., Holgerson, M. A., Liu, S., Song, C., Melack, J., Raymond, P. A., Duarte, C. M., Allen, G. H., Olefeldt, D., Poulter, B., Battin, T. I., & Eyre, B. (2021). Half of global methane emissions come from highly variable aquatic ecosystem sources. *Nature Geoscience*, *14*, 225–230. https://doi.org/10.1038/s41561-021-00715-2

Saggar, S., Tate, K. R., Giltrap, D. L., & Singh, J. (2008). Soil-atmosphere exchange of nitrous oxide and methane in New Zealand terrestrial ecosystems and their mitigation options: A review. *Plant and Soil*, *309*(1–2), 25–42. https://doi.org/10.1007/s11104-007-9421-3

Sawilowsky, S. S. (2009). New effect size rules of thumb. *Journal of Modern Applied Statistical Methods*, *8*, 467–474. https://doi.org/10.22237/jmasm/1257035100

Segers, R. (1998). Methane production and methane consumption: A review of processes underlying wetland methane fluxes. *Biogeochemistry*, *41*(1), 23–51.

Sieczko, A. K., Duc, N. T., Schenk, J., Pajala, G., Rudberg, D., Sawakuchi, H. O., & Bastviken, D. (2020). Diel variability of methane emissions from lakes. *Proceedings of the National Academy of Sciences of the United States of America*, *117*(35), 21488–21494. https://doi.org/10.1073/pnas.2006024117

Singh, S. N., Kulshreshtha, K., & Agnihotri, S. (2000). Seasonal dynamics of methane emission from wetlands. *Chemosphere-Global Change Science*, *2*(1), 39–46. https://doi.org/10.1016/S1465-9972(99)00046-X

Sivanpillai, R., & Miller, S. N. (2010). Improvements in mapping water bodies using ASTER data. *Ecological Informatics*, *5*(1), 73–78. https://doi.org/10.1016/j.ecoinf.2009.09.013

Sobek, S., Algesten, G., Bergström, A. K., Jansson, M., & Tranvik, L. J. (2003). The catchment and climate regulation of pCO_2 in boreal lakes. *Global Change Biology*, *9*(4), 630–641.

Stadmark, J., & Leonardson, L. (2005). Emissions of greenhouse gases from ponds constructed for nitrogen removal. *Ecological Engineering*, *25*, 542–551. https://doi.org/10.1016/j.ecoleng.2005.07.004

Stanley, E. H., Casson, N. J., Christel, S. T., Crawford, J. T., Loken, L. C., & Oliver, S. K. (2016). The ecology of methane in streams and rivers: Patterns, controls, and global significance. *Ecological Monographs*, *86*(2), 146–171. https://doi.org/10.1890/15-1027

Stets, E. G., Butman, D., McDonald, C. P., Stackpoole, S. M., DeGrandpre, M. D., & Striegl, R. G. (2017). Carbonate buffering and metabolic controls on carbon dioxide in rivers. *Global Biogeochemical Cycles*, *31*(4), 663–677. https://doi.org/10.1002/2016GB005578

Tipping, E., Corbishley, H. T., Koprivnjak, J. F., Lapworth, D. J., Miller, M. P., Vincent, C. D., & Hamilton-Taylor, J. (2009). Quantification

of natural DOM from UV absorption at two wavelengths. *Environmental Chemistry*, *6*(6), 472–476. https://doi.org/10.1071/EN09090

Turner, T. E., Billett, M. F., Baird, A. J., Chapman, P. J., Dinsmore, K. J., & Holden, J. (2016). Regional variation in the biogeochemical and physical characteristics of natural peatland pools. *Science of the Total Environment*, *545*, 84–94. https://doi.org/10.1016/j.scitotenv.2015.12.101

van Bergen, T. J. H. M., Barros, N., Mendonça, R., Aben, R. C. H., Althuizen, I. H. J., Huszar, V., Lamers, L. P. M., Lürling, M., Roland, F., & Kosten, S. (2019). Seasonal and diel variation in greenhouse gas emissions from an urban pond and its major drivers. *Limnology and Oceanography*, *64*(5), 2129–2139. https://doi.org/10.1002/lno.11173

Van den Pol-van Dasselaar, A., Van Beusichem, M. L., & Oenema, O. (1999). Methane emissions from wet grasslands on peat soil in a nature preserve. *Biogeochemistry*, *44*(2), 205–220. https://doi.org/10.1007/BF00992979

Venables, W. N., & Ripley, B. D. (2002). *Modern applied statistics with S*. Springer.

Vermaat, J. E., Hellmann, F., Dias, A. T., Hoorens, B., van Logtestijn, R. S., & Aerts, R. (2011). Greenhouse gas fluxes from Dutch peatland water bodies: Importance of the surrounding landscape. *Wetlands*, *31*(3), 493–498. https://doi.org/10.1007/s13157-011-0170-y

Verpoorter, C., Kutser, T., Seekell, D. A., & Tranvik, L. J. (2014). A global inventory of lakes based on high-resolution satellite imagery. *Geophysical Research Letters*, *41*(18), 6396–6402. https://doi.org/10.1002/2014GL060641

Wallin, M. B., Campeau, A., Audet, J., Bastviken, D., Bishop, K., Kokic, J., Laudon, H., Lundin, E., Löfgren, S., Natchimuthu, S., Sobek, S., Teutschbein, C., Weyhenmeyer, G. A., & Grabs, T. (2018). Carbon dioxide and methane emissions of Swedish low-order streams—A national estimate and lessons learnt from more than a decade of observations. *Limnology and Oceanography Letters*, *3*(3), 156–167. https://doi.org/10.1002/lol2.10061

Wang, Y., Chen, H., Zhu, Q., Peng, C., Wu, N., Yang, G., Zhu, D., Tian, J., Tian, L., Kang, X., He, Y., Gao, Y., & Zhao, X. (2014). Soil methane uptake by grasslands and forests in China. *Soil Biology and Biochemistry*, *74*, 70–81. https://doi.org/10.1016/j.soilbio.2014.02.023

Webb, J. R., Clough, T. J., & Quayle, W. C. (2021). A review of indirect N_2O emission factors from artificial agricultural waters. *Environmental Research Letters*. https://doi.org/10.1088/1748-9326/abed0

Webb, J. R., Hayes, N. M., Simpson, G. L., Leavitt, P. R., Baulch, H. M., & Finlay, K. (2019). Widespread nitrous oxide undersaturation in farm waterbodies creates an unexpected greenhouse gas sink. *Proceedings of the National Academy of Sciences of the United States of America*, *116*(20), 9814–9819. https://doi.org/10.1073/pnas.1820389116

Webb, J. R., Leavitt, P. R., Simpson, G. L., Baulch, H. M., Haig, H. A., Hodder, K. R., & Finlay, K. (2019). Regulation of carbon dioxide and methane in small agricultural reservoirs: Optimizing potential for greenhouse gas uptake. *Biogeosciences*, *16*(21), 4211–4227. https://doi.org/10.5194/bg-16-4211-2019

Weishaar, J. L., Aiken, G. R., Bergamaschi, B. A., Fram, M. S., Fujii, R., & Mopper, K. (2003). Evaluation of specific ultraviolet absorbance as an indicator of the chemical composition and reactivity of dissolved organic carbon. *Environmental Science & Technology*, *37*(20), 4702–4708. https://doi.org/10.1021/es030360x

Weiss, R. (1974). Carbon dioxide in water and seawater: The solubility of a non-ideal gas. *Marine Chemistry*, *2*(3), 203–215. https://doi.org/10.1016/0304-4203(74)90015-2

Wiesenburg, D. A., & Guinasso, N. L. Jr (1979). Equilibrium solubilities of methane, carbon monoxide, and hydrogen in water and sea water. *Journal of Chemical and Engineering Data*, *24*(4), 356–360. https://doi.org/10.1021/je60083a006

Wilkinson, J., Bors, C., Burgis, F., Lorke, A., & Bodmer, P. (2018). Measuring CO_2 and CH_4 with a portable gas analyzer: Closed-loop operation, optimization and assessment. *PLoS One*, *13*(4), e0193973. https://doi.org/10.1371/journal.pone.0193973

Yu, Z., Wang, D., Li, Y., Deng, H., Hu, B., Ye, M., Zhou, X., Da, L., Chen, Z., & Xu, S. (2017). Carbon dioxide and methane dynamics in a human-dominated lowland coastal river network (Shanghai, China). *Journal of Geophysical Research: Biogeosciences*, *122*(7), 1738–1758. https://doi.org/10.1002/2017JG003798

Yuan, J., Xiang, J., Liu, D., Kang, H., He, T., Kim, S., Lin, Y., Freeman, C., & Ding, W. (2019). Rapid growth in greenhouse gas emissions from the adoption of industrial-scale aquaculture. *Nature Climate Change*, *9*, 318–322. https://doi.org/10.1038/s41558-019-0425-9

Zuur, A. F., Leno, E. N., Walker, N., Saveliev, A. A., & Smith, G. M. (2009). *Mixed effects models and extensions in ecology with R*. Springer.

SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section.

How to cite this article: Peacock, M., Audet, J., Bastviken, D., Cook, S., Evans, C. D., Grinham, A., Holgerson, M. A., Högbom, L., Pickard, A. E., Zieliński, P., & Futter, M. N. (2021). Small artificial waterbodies are widespread and persistent emitters of methane and carbon dioxide. *Global Change Biology*, 27, 5109–5123. https://doi.org/10.1111/gcb.15762