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Research article

Quantifying organic carbon storage in temperate pond sediments

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ABSTRACT

Ponds may hold significant stocks of organic carbon in their sediments and pond creation may offer a practical application for land managers to increase carbon storage. However, ponds are overlooked in global carbon budgets. Their potential significance is suggested by the abundance of ponds throughout terrestrial biomes and their high carbon burial rates, but we lack measures of sediment carbon stocks from typical ponds. We sampled sediment from lowland temperate ponds in north east England comparing carbon stocks from ponds categorised by surrounding land use, or dominant vegetation, or drying regime, along with measures of variation within ponds. Sediment carbon varied considerably between ponds. This variation was more important than any systematic variation between pond types grouped by land use, vegetation or drying, or any variation within an individual pond. Our estimates of pond sediment organic carbon give measures that are higher than from soils in widespread habitats such as temperate grassland and woodland, suggesting that ponds are significant for carbon budgets in their own right. Ponds are relatively easy to create, are ubiquitous throughout temperate biomes and can be fitted in amongst other land uses; our results show that pond creation would be a useful and practical application to boost carbon sequestration in temperate landscapes.

1. Introduction

Ponds are a part of the plumbing for the global carbon cycle, the freshwater ecosystems, from large rivers and lakes to small ponds and wetlands, responsible for transporting significant amounts of carbon (Cole et al., 2007). Tranvik et al.'s (2018) review of carbon in freshwaters noted the progress from studies of individual systems, to a holistic view of freshwaters as "collectors and reactors", active as transporters, sources and sinks. Much of the recent work on carbon fluxes in ponds was prompted by Downing's re-evaluation of their potential importance (Downing et al., 2006, 2008; Downing 2010). Downing combined evidence for the intensity of ponds' geochemical processing with data suggesting that we underestimated the number and area of small ponds, proposing that as a result, they play an "unexpectedly major role" in the global carbon cycle. The potential importance of small ponds in the carbon cycle was spotlighted by estimates of the numbers and areas of these habitats that suggested they made up a significant but unrecognised proportion of lentic habitats, notably Downing et al.'s. (2008) estimate of 277, 400, 000 ponds of <0.001 km^2. More recent data suggest numbers like this are over-estimates (Seekell et al., 2013; Verpoorter et al., 2014) but the contribution of small ponds could still be significant. For example Holgerson and Raymond (2016) estimated that ponds of <0.001 km^2 have a global area of up to 861,578 km^2, 8.6% of the total area of lake and pond habitats, but could be responsible for 15.1% of CO_2 and 40% of diffused CH_4 emissions from these inland waters.

Contemporary studies have highlighted the important but variable role of ponds for carbon fluxes. Holgerson and Raymond (2016) suggest that ponds may be important sources of Carbon to the atmosphere, supported by evidence from boreal and arctic pools, (Abnizova et al., 2012; Wik et al., 2016; Kuhn et al., 2018), a role likely to increase as climate change warms these higher latitudes (Wik et al., 2016). Conversely, Taylor et al. (2019) estimated carbon burial rates from temperate lowland ponds that were higher than other terrestrial habitats, although Gilbert et al. (2016), working on the same ponds, showed very rapid switches from being net sinks to net source as the ponds dried. Autochthonous and allochthonous inputs can be important in temperate, temporary ponds (Dalu et al., 2016), the precise balance likely to vary with pond hydrology (Abnizar et al., 2014) and succession (Taylor et al., 2019).

For example Rubbo et al. (2016) demonstrated the importance of terrestrial inputs from leaf litter in a temporary forest pond, whilst Àvila

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et al. (2016) showed the dominance of autochthonous production in temporary ponds in south east Europe.

The evidence generally supports ponds' role as a source of C to the atmosphere (Torgersen and Branco, 2008), however, much remains uncertain for small, ephemeral systems (Marcé et al., 2019).

Whilst examples of flux measurements for ponds are increasing, the actual Sediment Carbon Stock (here-after SCS) currently stored in pond sediments remains largely unknown. Gilbert et al. (2014) found higher SCS in the sediments of lowland agricultural ponds compared to the surrounding soils, and other studies highlight ponds' high C burial rates (Mulholland and Elwood, 1982; Dean and Gorham, 1998; Downing, 2010; Taylor et al., 2019). Ponds are overlooked in land-use policies promoted for climate change mitigation. This absence of pond SCS from landscape carbon budgets is due in part to an absence of data; ponds are an overlooked habitat (Céréghino et al., 2014; Biggs et al., 2017), despite their global ubiquity in terrestrial habitats (Jeffries 2012, 2016). Our aim was to quantify SCS in temperate ponds. We hypothesised that SCS in ponds would vary with three key variables that drive the variations in ecosystem functions among ponds; (1) Surrounding land-use, (2) vegetation, and (3) permanence/drying regime. All three of these variables have the potential to be managed to maximise carbon stocks held by ponds but their importance is not understood.

Land-use has been identified as an important driver of variation in pond biodiversity in studies from around the world, both the nature of the landscape surrounding ponds and also direct impacts resulting from land-use taking place within the pond such as livestock access. For example in Japan different land uses impacted farm pond eutrophication (Usio et al., 2017), in Texan playa pools (seasonal ponds), decreasing invertebrate diversity was associated with landscape homogenisation (Hall et al., 2004) and in Belgian farm ponds plant community complexity declined with increasing trampling from livestock (DeClerck et al., 2006). Some pond types can survive in very intensive agricultural landscapes. For example, Bissels et al. (2005) and Altenfelder et al. (2016) highlight the shallow pools of arable fields in French and German lowlands, which may benefit from ploughing to maintain the disturbed ground inundation plant communities.

Our second focus was the vegetation in ponds. The overall amount of plant biomass growing in a wetland is known to affect CH_4 fluxes although results vary, some showing a positive correlation (Christensen et al., 2003), others negative (Koelbener et al., 2010). Living plants may be an important route for emissions (Kelker and Chanton, 1997) or produce different root exudates enhancing or diminishing carbon availability. Plant detritus may inhibit metabolising of carbon due, for example, to phenolics, or vary in robustness, for example, the substantial biomass of many reeds and rushes versus the slight and fragile remains of ephemeral herbs (Dunn et al., 2016).

Drying out is the third factor we investigated and can cause rapid changes to greenhouse gas (GHG) fluxes. Gilbert et al. (2016) measured CO_2 fluxes from small ponds as they transitioned from wetted to dry, which resulted in a switch from being sinks to sources within days. Drying out increases CO_2 efflux as sediments are exposed to the air (Fromin et al., 2010; Catalan et al., 2014; Martinsen et al., 2019). Marcé et al.'s (2019) review of CO_2 and CH_4 emissions from dried out inland waters suggests that CO_2 efflux is generally increased by drying, whilst CH_4 emissions are reduced, this reduction attributed to the reduced anoxia and lack of ebullition.

Another key factor impacting the inclusion of SCS in carbon budgets is the limited knowledge of how SCS varies within individual ponds. Ponds may contain localised focal points for sediment deposition, a trend often seen in impoundments (Pittman et al., 2013; Shotbolt et al., 2005; Vanni et al., 2011). Equally, given the seasonal nature in size or permanence of small ponds, it is likely that sediment that stays submerged for longer periods may be subject to longer periods of sediment anoxia, favourable for higher C preservation. Furthermore, shallow systems are likely to be subject to high levels of disturbance from bioturbation or agricultural activities (e.g. grazing cattle or farm machinery). Given these likely variations, this study also comprises a sampling regime designed to investigate variations in SCS within individual ponds and explore optimal sampling densities for small ponds to inform future studies.

Understanding the inter-pond variation (differences of SCS *among a group* of ponds at a local scale: all the ponds are within 10 km of each other in the same biogeographic unit, the South Northumberland Coastal Plain) and intra-pond variation (spatial differences in SCS *within an individual* pond) is crucial to enable upscaling studies to regional, national, and global estimates, their successful integration into carbon budgets and use for carbon mitigation. Our study specifically targets small, lowland ponds, a habitat found throughout the temperate biomes, to provide robust estimates of SCS and how these vary with land use, plant communities and drying regime to inform carbon budgets and the practical management of ponds for carbon sequestration. We measure SCS variation in sediment cores across 40 ponds, representing a range of land uses, plant communities and drying regimes. In addition, we measure the intra-pond variation of SCS by taking 10 cores from one pond in each different land-use category (arable, pasture, dune slack and naturalistic).

2. Materials and methods

2.1. Study region

The subject ponds were in north east England, on the coastal plain at Druridge Bay, Northumberland. The ponds in this region have been the focus of multiple studies, for example on their ecological importance, history, permanence and relationship to regional weather (Jeffries, 1998, 2012, 2016), and their geochemical processes and potential importance in the carbon cycle (Gilbert et al., 2014, 2016; Taylor et al., 2019).

There are over 130 ponds along Druridge Bay including a few farm ponds dating to at least the 1860s as well as ponds dug for nature conservation, subsidence ponds over coal seams, dune slacks and flooded World War 2 defences. The majority are <1000 m^2, and <1 m deep, their wetted areas fluctuating markedly with local rainfall. Jeffries (2012, 2016) characterises the Druridge pondscape in detail.

2.2. Pond selection to explore inter-variations

This study sampled 40 ponds across Druridge Bay, selected to cover a range of land uses, vegetation, and permanence, and also depths and area. Their morphology, setting and wildlife are typical of lowland ponds in the UK, and of the temperate biomes in general (Jeffries et al., 2016).

Our primary purpose was to quantify the inter-variation in SCS among a range of pond types, using three approaches to divide the ponds into differing groups: (1) adjacent land-use, (2) plant communities and, (3) drying regime. A map of the ponds is given in Supplement 1 and a summary of each of the ponds samples is given in Supplement 2.

2.2.1. Land use

The four land-use groups in this study refer to the terrain immediately surrounding the ponds: arable fields used for commercial crops such as cereals, permanent pasture used for grazing sheep or cattle, sand dune slacks with some brackish influence, and the final type being deeper, mostly permanent ponds surrounded by a buffer of wetland vegetation and supporting plant communities typical of ponds in the region (hereafter arable, pasture, dune and naturalistic. Supplement 3 details the characteristic plants from ponds in each land use, Supplement 4 shows a variety of ponds in situ). For example, ponds in the arable fields are routinely ploughed and planted with commercial crops most years, lack any surrounding buffer and usually dry out to leave exposed soil, whilst the naturalistic ponds are heavily vegetated and unmanaged. Species richness was similar in ponds from arable, dune and naturalistic land use ponds, but lower in pasture. The similarities may seem surprising but the arable ponds benefit from a combination of inundation species such *Juncus buffonius* combined with weeds of disturbed ground whilst the naturalistic ponds are often dominated by emergents and lacked submerged taxa perhaps because of their shallow, emergent-choked nature.

All 40 ponds rely primarily on rainfall, although the arable and pasture ponds are sensitive to changes in rainfall over 3–4 weeks, whilst the naturalistic and dune ponds are more buffered and respond, in terms of depth and extent, to variation over 4–5 months (Jeffries, 2016).

The 40 ponds comprise ten each from the four land-uses, which had been unchanged for at least 40 years.

2.2.2. Plant communities

All 40 ponds were included in botanical surveys of eighty ponds along Druridge Bay conducted between 2012 and 2015 (Jeffries, 2016). The survey followed the strategy of the UK National Pond Survey (Pond Action, 1998); all macrophyte plant within a pond's outer margin defined by the maximum winter water level were recorded between June and early September. Macrophyte abundance was recorded using the Domin scale, a 1–10 categorical scale each category representing a range of % cover used for the UK's National Vegetation Classification survey (Rodwell, 1995). Plants were identified using Stace (1997), including microscopic examination, for example of *Epilobium* seeds, except Starwort, *Callitriche* species, checked against Lansdown (2008). We also included bare ground as a category.

The ponds' plant communities were classified using TWINSPAN, run on CAP 3.1, taking this to four groups. The four groups coincide broadly, though not exclusively, with land use, notably one set dominated by grassy pasture ponds and another by inundation weed communities which contained all the arable ponds (Supplement 5). Whilst this is not surprising given the role of land-use in shaping pond biodiversity, it does mean that plant group and land-use type partly confound each other.

2.2.3. Drying regime

All of the 40 ponds had been monitored over several years to assess their vulnerability to drying out, thirty five of them as part of a specific study of their responses to local weather variation over three years (Jeffries, 2016). Drying regime information was based on visits to the ponds carried out every six weeks from November 2010 to November 2013, fifteen visits in total: 2012 was an unusually wet summer so that sites did not dry in the summer. The remaining five ponds were visited regularly as part of other surveys over several years. Ponds were recorded as dried out if they had no water above the surface of the sediment. Ponds were allocated to one of three categories: never known to dry out ($n = 6$), dried out occasionally over the survey period ($n = 13$) or dry out annually ($n = 21$). Whilst the assessment of drying regime of the ponds did not directly coincide with the period of the sediment sampling, it was of sufficient duration and closeness in time to be strongly indicative of the pond behaviour and, importantly, indicative of the geochemical conditions which the sediments were subjected to in the years prior to sampling.

2.3. Intra pond variations of SCS

SCS sampling took place over the period April to December 2014. Four ponds were chosen, one from each land use type, to examine variations in SCS distribution within individual ponds. From each pond ten cores were collected in a systematic grid pattern across the full pond topography, depth profile, and susceptibility to drying edges; 10 cores gives 45 pair-wise comparisons of cores from within each pond, each of which were compared to test for significant variation of SCS within a pond.

2.4. Sediment coring

Our study utilised a bespoke metal corer, designed specifically to allow extraction of cores from ponds with markedly differing sediment consistencies. Constructed of a high polish chromium-vanadium steel cylinder (core diameter 48 mm, length 500 mm), with a sharpened cutting edge around the bottom rim, this corer gives a fine, clean cut with minimal micro-crevices to facilitate extraction of the cored material. An internal plunger allowed the cored material to be extruded in the field. Sediment depth in ponds is seldom reported but published figures of 8.9 cm (Gilbert et al., 2014), 11 cm (Nicolet et al., 2004), 28 cm (DeClerck et al., 2006), 27 cm (Tsais et al., 2011), suggest approximately 20 cm is a typical depth. For our study, for individual ponds, core depths ranged from 9.2 to 33 cm (average, 16.9) for the inter-pond comparison cores and 10–33 cm (average 18.3 cm) for the intra-pond comparison cores (Supplement 2).

Upon extrusion, the core was dissected into ~1 cm thick slices along the whole length of the core, wrapped in tin foil, placed in a paper sample bag and stored in refrigeration prior to analysis.

All cores were taken by the same person to minimise any variation in technique and were taken between April and December 2014, during which period all ponds held some standing water at the time of sampling. For inter pond comparisons cores were taken in the centre of the pond within this wetted area, or in some cases in slightly shallower water where the depth was too great to allow safe wading access.

2.5. Sediment analysis

Data were recorded from each sample slice from each core. Firstly, the moisture content and Dry Bulk Density (DBD) of each sample slice were measured, followed by % carbon analysis by total elemental analysis (TEA). Moisture content and DBD are inversely related: they give a measure of the density of the sediment layer that has been laid down, allowing us to see how this varies with depth and location. DBD and % carbon are both required to calculate the carbon density (mg C cm^{-3}) in a sample, as detailed below. The carbon density can be scaled up to SCS (Kg C m^{-2}) for a specified depth of sediment, typically 10 cm, as used in this study.

For moisture content (%) all samples were weighed within 24 h of coring then placed in a drying cabinet for 7 day at ~ 40 °C until a constant weight was achieved, to give the dry weight of each sample. Soil moisture was calculated using equation (1).

$$moisture\ content\ (\%) = \frac{wet\ weight - dry\ weight}{wet\ weight} * 100 \qquad \text{Eqn 1}$$

Dry Bulk density (DBD; g cm^{-3}), was calculated using the dry weight and known volume for each dissected 1 cm section using equation (2).

$$DBD = \frac{dry\ weight_{sediment\ section}}{Volume_{sediment\ section}} \qquad 2$$

Dried samples were ground and sieved and ~5 mg processed for total carbon analysis via TEA (Thermo Scientific FLASH, 2000 Series Organic Elemental Analyser; oven temperature = 980 °C; run time = 360 s). Samples were placed in a carousel for automated analysis, with a program that every tenth sample was run in triplicate to calculate the precision of analysis (% relative standard deviation, RSD = 7.81%), followed by a blank to monitor the Limit of Detection and Limit of Quantification (0.46% C and 1.43% C respectively). These QC checks also allowed us to identify any analysis sequences that had encountered instrumental issues or malfunctions. To calculate the mass of carbon per 1 cm section (carbon density, mg C cm^{-3}) we combined the % C concentration from the TEA with the DBD using equation (3). The same form of equation (3) was used to calculate the nitrogen density

$$Carbon\ density(mg\ C\ cm^{-3}) = \frac{\%\ C}{100} \times DBD_{sediment\ section}(g\ cm^{-3}) \times 1000$$

Eqn 3

2.6. Statistical analysis

2.6.1. Intra-pond differences

Variations in the density of C (mg C cm^{-3}) among the sediment cores were tested separately for each pond, using Linear Mixed Models. Each of the ten cores from a pond were included as factors, and as a random effect since individual cores may show different trends, along with depth as a covariate with a repeat measures design (AR1). Differences between individual cores were tested using Bonferonni post-hoc comparisons. Data were normalised using ln transformation. Differences in dry bulk density, % moisture and % carbon were also tested, DBD using the same mixed models design as the carbon density. The % moisture and % carbon were compared using the Kruskal-Wallace test as the data could not be effectively normalised.

2.6.2. Inter-pond differences

Differences in the density of C (mg C cm^{-3}) and DBD between ponds were tested using Linear Mixed Models. Land-use, plant community and drying were included as fixed factors, with individual ponds as random effects and depth as a covariable with a repeat measures design (AR1) and ln transformed data. Because of the confounding of plant community within land-use categories, a full model with both could not be created. Plant communities and land use were tested in separate models in combination with drying regime and depth. Differences between individual factors were tested using Bonferonni post-hoc comparisons. Data were normalised using ln transformation. Differences of % moisture and % carbon between ponds were tested separately for land-use, plant community and drying regime using the Kruskal-Wallace test.

All statistics were run on SPSS 24, with the significance level at $P = 0.05$.

2.6.3. Decision tree analysis

To explore the interaction between plant community and wetting/drying cycles as explanatory factors for variation in carbon stocks, we used a Decision Tree Analysis (SPSS), which is a non-parametric multivariate statistical technique that has been used for a range of environmental applications (Baker et al., 2006; Elnaggar and Noller, 2010). The analysis was carried out on carbon density (mg C cm^{-3}) values from 616 samples, with permanence (always, sometimes or never dries) and plant community (diverse, ephemeral, grassy or reeds) as independent variables. The Chi-squared Automatic Interaction Detection (CHAID) branching method was used with 95% confidence levels.

3. Results and discussion

3.1. Intra-pond variation

Our primary purpose was to quantify variations in SCS among a diversity of lowland ponds, testing intra-pond variation and, comparing carbon density between ponds of different types defined by land use, plant communities and drying pattern, to inform carbon budget models and pond management for carbon capture. Variability in C distribution across small ponds is to be expected, perhaps due to localised focal points for sediment deposition (Pittman et al., 2013; Shotbolt et al., 2005; Vanni et al., 2011) or seasonal fluctuations to the wetted area creating gradients of anoxia resulting in differential C accumulation.

Mean density of carbon (mg C cm^{-3}) for the ten sampled cores within each of the four ponds selected for the intra-pond comparison is shown in Fig. 1 and Table 1.

From the results of the Linear Mixed Model, among the ten replicate cores collected from the four individual ponds, significant differences in density of C (mg cm^{-3}) were observed between some of the cores in only two of the ponds. In the arable field pond one core was significantly different from five of the other nine. In the naturalistic pond out of the 45 possible pair-wise comparisons of cores there were 10 that gave significant differences resulting from three cores being significantly different to another group of six (Table 1 summarises these significant differences between individual cores). Thus, the results show some, but limited, intra pond variation in density of C.

The range of carbon density from the intra-pond replicates (10.5–59.0 mg C cm^{-3}) is broadly similar to the overall range observed from single cores in the 40 pond survey, (12–123 mg C cm^{-3}), only two of which had > 73 mg C cm^{-3}. At the 95% CI, the margins of error among replicate cores were all <11% of the mean for each pond (mean

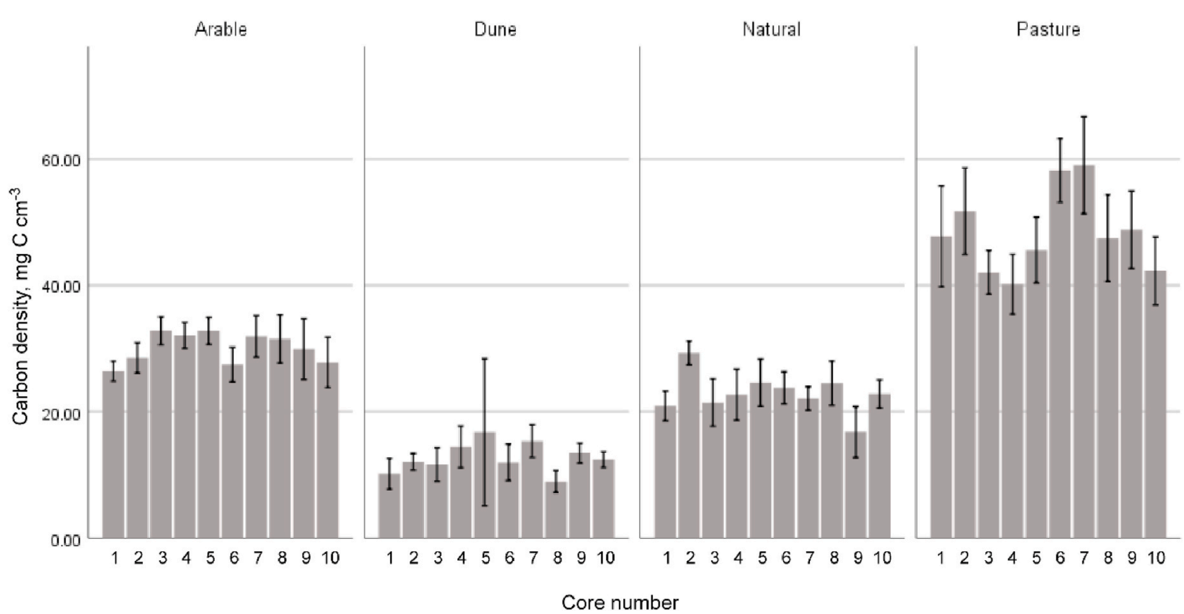

Fig. 1. Within-pond carbon density variation. Carbon density from 10 cores taken from four ponds, with the carbon density reported as the mean mg C cm^{-3} for the whole column length (column depth information is given in Supplement1; error bars represent ± 1 sd). The numbers along the x-axis refer to the ten cores from each pond. For comparison to Fig. 2, the four ponds used for the intra pond sampling were 30 (naturalistic), 38 (arable), 29c (pasture) and CPP1 (dune slack). Carbon density is the mean concentration along the core.

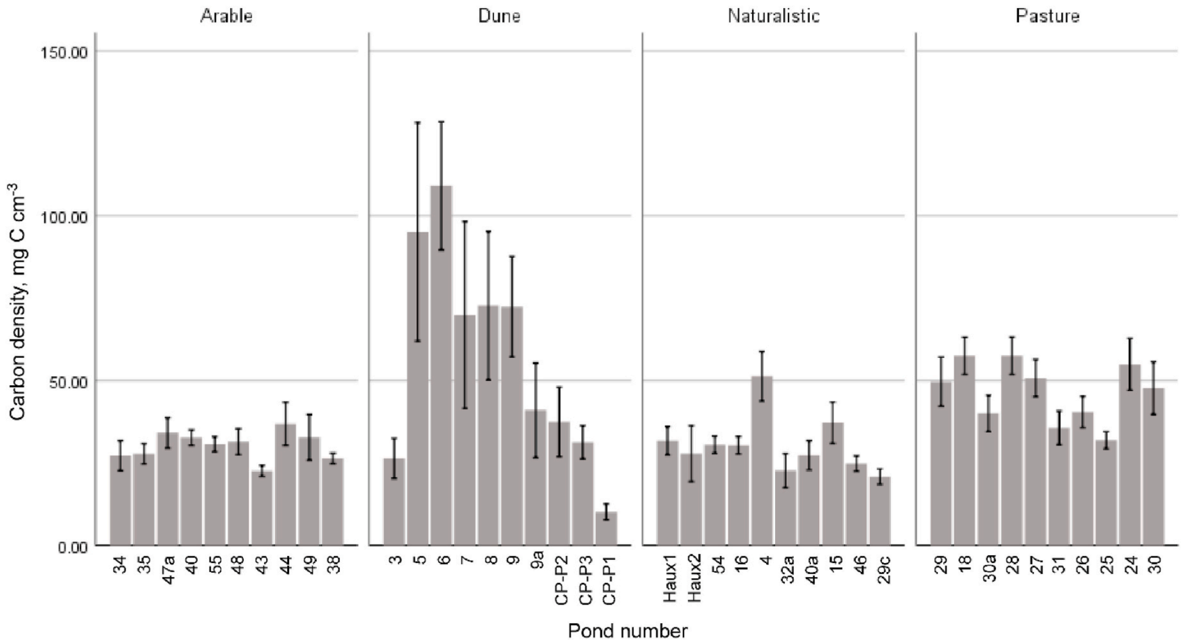

Fig. 2. Between pond carbon density variation. Carbon density from 10 cores taken from forty ponds, ten ponds from each of four land uses, with the carbon density reported as the mean mg C cm^{-3} for the whole column length (column depth information is given in Supplement1; error bars represent ± 1 sd). The names and numbers along the x-axis refer to the 40 individual ponds.

Table 1
Intra pond core summary, mean ± one standard deviation for carbon density, carbon %, bulk density of sediment and moisture % from the four ponds. "Intra core differences?" indicates if significant differences were found between some of the 10 cores from within a pond. The carbon densities (mean ± standard deviation) are shown for the significantly different cores from ponds 30 and 38. Different superscripts indicate the significant differences.

	Pond: land use and pond ID			
	Pasture, 29c	Dune slack, CPP1	Naturalistic, 30	Arable, 38
Carbon density (mg C cm^{-3})	23.00 ± 7.13	12.48 ± 6.47	48.59 ± 10.98	30.15 ± 6.21
Carbon, %	3.70 ± 3.17	1.09 ± 0.99	6.28 ± 3.68	3.14 ± 0.87
Dry Bulk density, g cm^{-3}	0.86 ± 0.36	1.32 ± 0.34	0.95 ± 0.33	0.98 ± 0.12
Moisture, %	43.60 ± 15.6	27.13 ± 8.1	40.24 ± 13.04	29.99 ± 3.97
Intra core differences?	ns	ns	$F = 8.09$, df 1, 9, $P < 0.001$	$F = 4.05$, df 1,9, $P < 0.001$
Significant differences between cores				
Pond 30, core numbers	2, 6, 7	9	1, 3, 4, 5, 8, 10	
Carbon density (mg C cm^{-3})	56.03 ± 10.89^{a}	48.78 b ± 10.60^{a,b}	44.31 ± 8.45^{b}	
Pond 38, core numbers	1	2, 6, 9, 10	3, 4, 5, 7, 8	
Carbon density (mg C cm^{-3})	26.41 ± 3.27^{a}	28.43 ± 6.86^{a,b}	32.19 ± 5.47^{b}	

= 8.4%, range = 6.1–10.8%) i.e., when calculating carbon stocks from individual sediment cores we can assume, with 95% confidence, that the estimated C stock is representative of the sediment distribution across the pond within 8.4%.

The low margins of error for replicate cores stated above indicate a good level of reproducibility in our sampling of these sediments. In an analysis of replicate cores from impoundments ~100,000 m^2, Pittman et al. (2013) found that a 25% precision could be gained from 10 cores, and while the ponds in this study are ~ 1–2 orders of magnitude smaller, the mean % RSD was 16%. Whilst the areas of the ponds were considerably varied (range = 366–6675 m^2) no relationship was observed between precision and sampling densities, suggesting that sediment distribution is equally varied among ponds of differing size.

Lack of detailed studies regarding the heterogeneity of sediment C distributions within systems is one of the major factors leading to poorly constrained C stock estimates within small water bodies such as ponds. This study highlights that whilst C concentrations (% C) may vary when compared among replicate cores from individual ponds, when calculating the carbon the margin of error in estimations is comparatively low, with C density estimations from individual sediment cores being relatively representative of sediments across the pond. To include pond SCS in carbon budgets, it is more important to sample as many ponds as possible to capture the variation between ponds, rather than take more samples from fewer ponds. These factors support the single core sampling strategy used in the 40 pond survey and the validity of single core, extensive surveys across as many ponds as possible for future work.

3.2. Inter-pond variation

3.2.1. Relationship between carbon density and land use, plant community type and drying regime

The data for all cores from the forty ponds, categorised by land use, are shown in Fig. 2. The mean data for carbon density (mg C cm^{-3}) and C concentrations (% C), sediment DBD and moisture % are summarised for the ponds categorised by surrounding land use (Table 2), drying regime (Table 3) and vegetation type (Table 4) and Fig. 3.

Mean carbon density varied with land use, from 29.92 mg C cm^{-3} in the naturalistic ponds to 51.24 mg C cm^{-3} in the dune sites (Fig. 3a). Carbon density was higher in the sediments of ponds that dried out every year, 49.69 mg C cm^{-3} versus ponds that only dry in some years or never, at 32.52 mg C cm^{-3} and 30.69 mg C cm^{-3} respectively (Fig. 3c). When the ponds were classified by vegetation types, the highest density was found in ponds with a diverse mixed sward of wetland flora, 48.8 mg C cm^{-3} (Fig. 3b).

Table 2
Summary, mean ± one standard deviation for carbon density, carbon %, bulk density of sediment and moisture % in the four categories of land use. Significant differences are indicated by different superscripts: carbon density and bulk density results from the linear mixed models; % moisture and % carbon from Kruskal Wallace test.

	Land use			
	Naturalistic	Arable	Pasture	Dune
Carbon density (mg C cm^{-3})	29.92 ± 11.24	30.46 ± 7.24	44.82 ± 13.58	51.24 ± 39.19
Carbon, %	4.40 ± 4.46^{a}	2.89 ± 0.90^{a}	5.63 ± 3.27^{b}	9.12 ± 8.79^{b}
Dry Bulk density, g cm^{-3}	0.99 ± 0.45	1.12 ± 0.33	0.96 ± 0.32	0.94 ± 0.48
Moisture, %	39.3 ± 18.7^{a,b}	33.1 ± 8.2^{a}	40.3 ± 13.0^{b}	42.7 ± 20.5^{b}

Table 3
Summary, mean ± one standard deviation for carbon density, carbon %, bulk density of sediment and moisture % in the three categories of drying regime. Significant differences are indicated by different superscripts: carbon density and bulk density results from the linear mixed models; % moisture and % carbon from Kruskal Wallace test.

	Dry period: do the ponds dry out annually?		
	Never	Sometimes	Always
Carbon density (mg C $cm^{-3)}$	30.69 ± 12.76^{a}	32.52 ± 14.27^{a}	49.69 ± 30.99^{b}
Carbon, %	4.10 ± 2.89^{a}	3.89 ± 3.80^{a}	7.88 ± 7.43^{b}
Dry Bulk density cm^{-3}	0.96 ± 0.42	1.09 ± 0.38	0.94 ± 0.43
Moisture, %	39.16 ± 17.01^{a,b}	35.88 ± 13.82^{a}	41.80 ± 17.98^{b}

Table 4
Summary, mean ± one standard deviation for carbon density, carbon %, bulk density of sediment and moisture % in the four categories of vegetation type. Significant differences are indicated by different superscripts: carbon density and bulk density results from the linear mixed models; % moisture and % carbon from Kruskal Wallace test.

	Vegetation			
	1, reeds	2, diverse	3, grassy	4, ephemeral
Carbon density (mg C cm^{-3})	28.46 ± 16.70^{a}	48.80 ± 32.93^{b}	38.48 ± 12.04^{a,b}	30.46 ± 7.27^{a}
Carbon, %	5.29 ± 6.24^{a,c}	7.97 ± 7.41^{b}	4.42 ± 3.16^{c}	2.89 ± 0.90^{a,d}
Dry Bulk density cm^{-3}	1.02 ± 0.50	0.89 ± 0.44	1.06 ± 0.30	1.12 ± 0.32
Moisture, %	40.10 ± 21.75^{a,c}	43.65 ± 18.45^{b}	35.92 ± 12.41^{c,d}	33.07 ± 8.18^{d}

3.2.2. Outcomes of the Linear Mixed Models

Summary outcomes for the models of carbon density are shown in Table 5. A combination of plant community type, drying regime and depth provide the best model based on Akaike's Information Criterion (AIC). Despite the differences in mean carbon density between ponds from the four different land uses, land use was not significant, unless used as the sole factor, because of marked variation of carbon density between ponds within each category. Carbon density is significantly higher in the ponds that dry up every year compared to those that only dry some years or not at all (Fig. 3b, Table 3). Carbon is also higher in those ponds with diverse wetland flora forming a dense cover (Fig. 3c, Table 4).

The design of the inter-pond sampling anticipated finding marked differences among pond types. In particular, we hypothesised significant differences between all four land-uses given the striking differences in

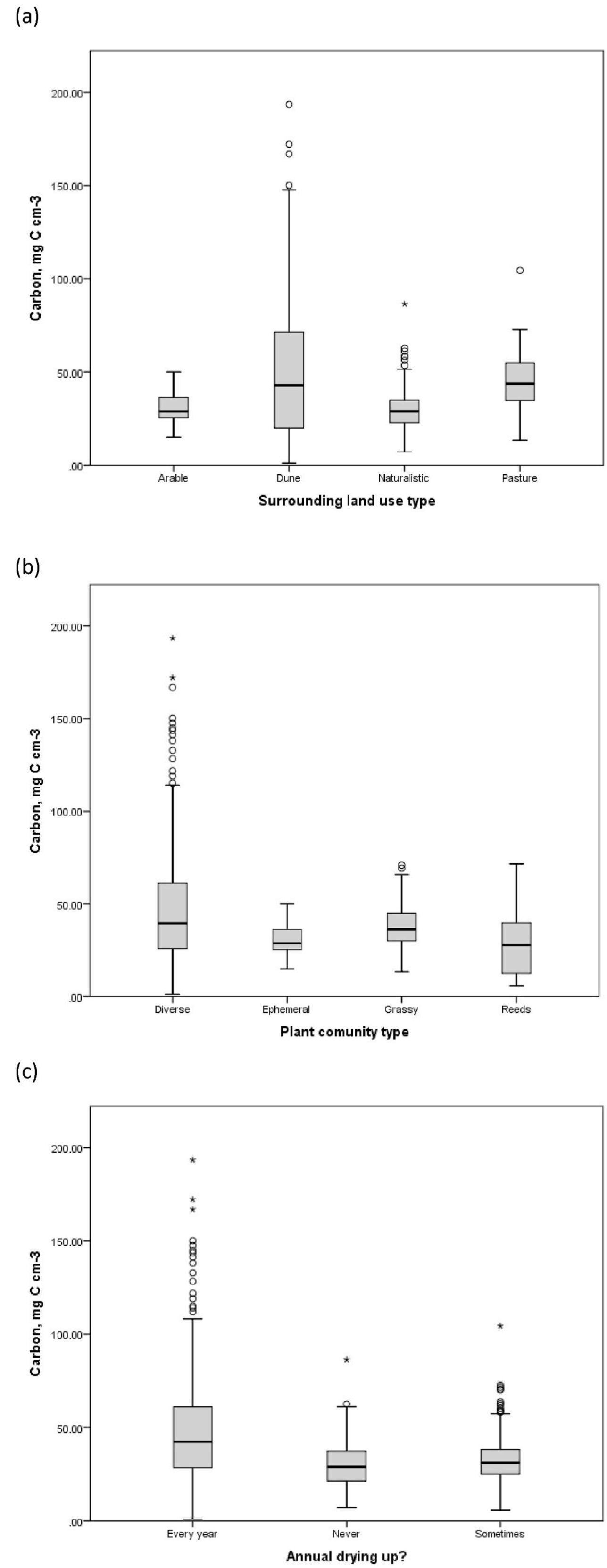

Fig. 3. Organic carbon density, mg C cm^{-3}, in pond sediments from (a) ponds in amongst four land-uses, (b) different plant communities dominated by reeds, diverse herbs and grasses, grass or ephemeral weed species, (c) ponds with different drying regimes. The thick horizontal bars show the median and the box, the inter-quartile range, IQR. Circles and ★are outliers 1.5 or 3 times the IQR beyond the upper quartile.

Table 5
Summary of General Linear Mixed Models of carbon density, mg C cm^{-3} ranked by AIC. The variables included in each model are shown, along with their significance, in same order as the variables were included in the models and overall AIC and number of parameters.

Variables included in model	Significant	AIC, total parameters
Plant community, permanence, depth	P < 0.01, P < 0.01, P < 0.05	617.30, 10
Land use, permanence	ns, P < 0.01	627.30, 9
Permanence	P < 0.01,	628.18, 6
Land use, permanence, depth	ns, ns, P < 0.01	629.40, 10
Plant community, depth	P < 0.05, P < 0.01	630.48, 8
Permanence, depth	P < 0.05, P < 0.05	630.57, 7
Land use	P < 0.05	630.94, 7
Land use, depth	P < 0.05, P < 0.01	631.75, 8
Depth	P < 0.01	635.74, 5

the morphology, biodiversity, and context of the landscapes, from arable field ponds which were ploughed and dried out every year to the naturalistic ponds which were unmanaged and sheltered within protective buffers of wetland and rank grassland. However, there were no systematic differences in carbon density between the ponds from the four different land uses. Our hypothesis that marked differences in land-use would result in very different carbon stocks is not supported. Instead, the carbon density in the sediments of individual ponds within each of the land use appears to be as heterogeneous as the well-established heterogeneity of pond wildlife at the landscape scale (Davies et al., 2008). In summary, pond sediments contain high densities of carbon compared to many other land-uses such as grassland or forestry, and all types of ponds may have this potential.

However, the results of the Linear Mixed Models do support the potential role of both plant community and wetting/drying cycles as explanatory factors for variation in carbon stocks, although it should be noted that plant community and land use could not be used together in models. Ponds with diverse plant communities and also ponds that dry out every year were associated with significantly higher carbon density than for other community types and ponds that either don't dry out, or do so sometimes. To explore further the interaction between these two factors, we used a SPSS decision tree classification analysis. The results in Fig. 4 show that permanence is the most important factor in determining carbon density, with those ponds that dry out each year having 57% more carbon (49.7 vs 31.7 mg C cm^{-3}) than those that either dry out occasionally or not at all. In addition, of the ponds that always dry out, those with diverse plant communities have higher carbon densities than other plant community types, with an average of 69.8 mg C cm^{-3}, or 76% higher than the average for all ponds. The next highest carbon density was in ponds with 'grassy' plant communities (47.3 mg C cm^{-3}, or 19.7% higher than the average for all ponds). Such information may be of use in designing landscape management strategies that maximise carbon densities in ponds that are constructed for the purpose of carbon sequestration. In the following sections we discuss the role of the rhizosphere in sediment carbon accumulation and the likely influence of community type and drying regime as an explanation for the results shown in Fig. 4.

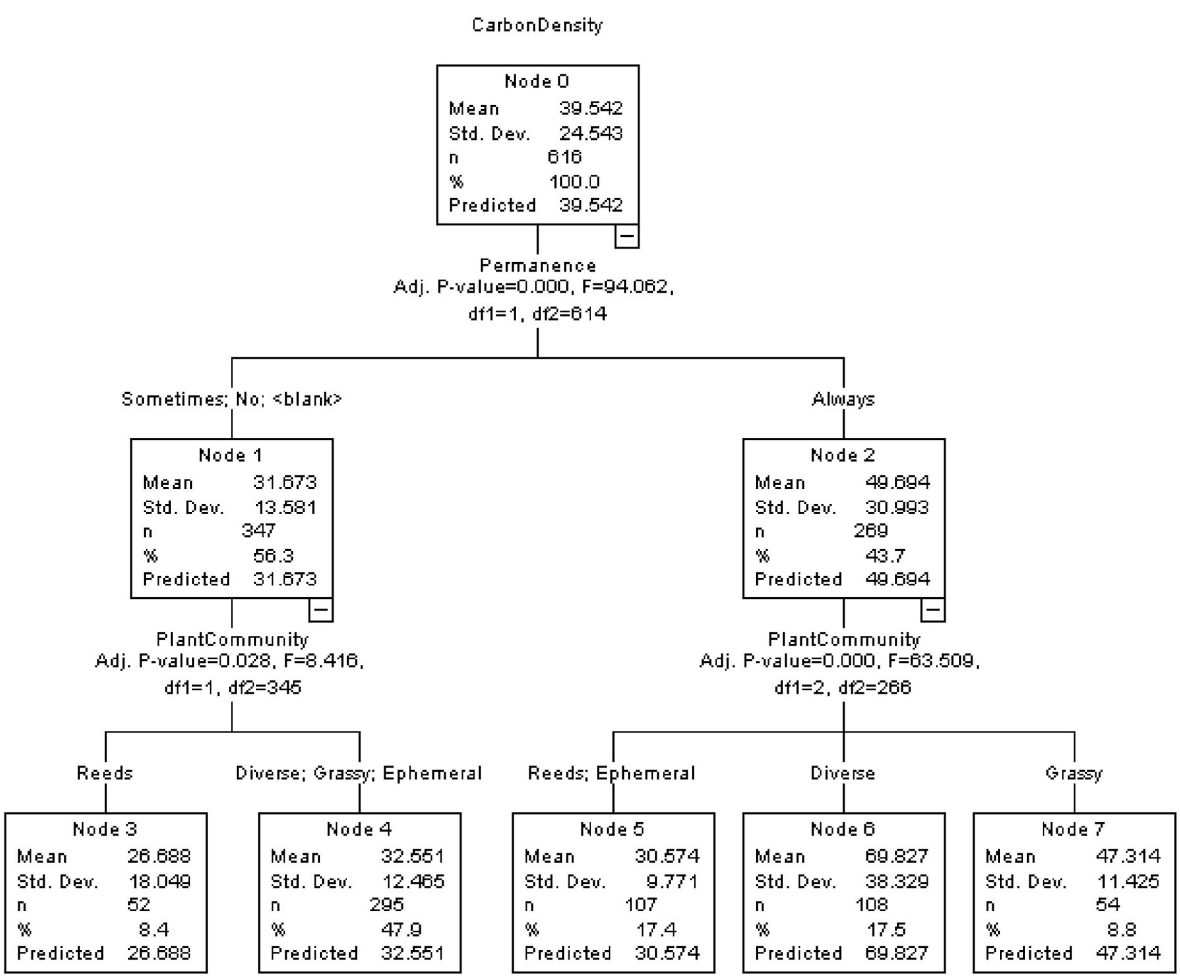

Fig. 4. Decision tree analysis for carbon density (mg C cm^{-3}), based on permanence (always, sometimes or never dries) and plant community (diverse, ephemeral, grassy or reeds) as independent variables. The CHAID branching method was used with 95% confidence levels. Mean and predicted are the same in this model.

3.2.3. The significance of drying regime and plant community type

The much higher carbon density observed in ponds that both dry out and have a diverse plant community can be explained by either greater carbon inputs, reduced carbon outputs or a combination of both, though the underlying ecological, microbiological and chemical factors are complex. Moreover, because ponds with highest carbon densities experience both wet and dry periods, processes that apply to both soil and submerged sediments should be considered when identifying the most likely contributing factors. The drying regime may have a direct impact on carbon storage, for example, there is evidence that cycles of wetting and drying, can stimulate the production of root exudates (Atere et al., 2017).

For the effect of plant communities on carbon storage, overall, we found that those communities associated with the highest carbon stocks were those with a thick sward, either of diverse herbs, grasses and a few larger rushes ("type 2, diverse") and sedges or predominantly grasses ("type 3, grassy"), with species such as flote grass (*Glyceria fluitans*), soft rush (*Juncus effusus*) or bistort (*Persicaria amphibia*). The ephemeral inundation weed community of community "type 4, ephemeral" included many annuals that rapidly decay away, such as cud weed (*Filago vulgaris*) and pineapple mayweed (*Matricaria discoidea*), unlikely to leave substantial organic remains and the bare mud of these ponds is not visibly rich in plant debris. Carbon density was also lower in the "type 1, reeds" ponds, characterised by common reed (*Phragmites australis*).

To further investigate which aspects of the plant community might explain the higher C density shown from the decision tree analysis for "type 2, diverse" ponds that also dry out, we have identified, in Table 6, the plant species that were either uniquely associated with the 'diverse' plant communities or were present in a greater proportion of 'diverse' ponds, compared to the other community types (the species in Table 6 represented 30% by cover of all plant types present in 'diverse' ponds compared to only 6.5% in other community types). Of these, it is notable that several are members of genera that have been shown in the literature to have a high content of polyphenols, including *Ranunculus* (Neag et al., 2017), *Vicia*, (Orhan et al., 2009), *Equisetum* (Graefe and Veit, 1999) and *Epibolium* (Tóth et al., 2009). Plant biomass that is rich in polyphenols is known to show some resistance to microbial degradation, particularly bacterial degraders, which may be affected by the antimicrobial properties of some polyphenols (Yakimovich et al., 2018). Polyphenols can also leach into aquatic environments from plants in surrounding land (Cieślewicz 2014). In such circumstances, fungi, which are able to degrade polyphenols, may benefit from reduced competition. A microflora dominated by fungi will give rise to reduced CO_2 emissions compared to bacteria, which are considered to be the "drivers of more active decomposition" (Yakimovich et al., 2018). Thus, high polyphenol content in plant detritus is likely to contribute to the stabilisation of organic carbon in pond sediments.

Another major input of C into pond sediments, and one that likely to benefit from a diverse plant community type, is the production of root exudates, which is linked to the rhizosphere priming effect (Shahzad et al., 2015). The root zone of aquatic plants typically extends to 20–30 cm (Bowden, 1987), and has the potential to transform the sediment environment through the supply of (labile) organic carbon as root exudates and root detritus, as well as oxygen (Bais et al., 2006; Kotas et al., 2019; Shahzad et al., 2015). The introduction of root exudates such as organic acids, sugars, amino acids and phenolics, as well, as polysaccharides and proteins (Bais et al., 2006), benefits the plant through the stimulation of the microflora to release nutrients from the stored organic carbon (Bais et al., 2006; Fontaine et al., 2011; Shahzad et al., 2015). Root exudates can also be directly incorporated into stable carbon stocks if deposited as aggregates with inorganic soil components (Atere et al., 2017). Cycles of wetting and drying, can stimulate the production of root exudates (Atere et al., 2017), which may be a factor in the increased carbon densities for ponds that dry out.

Table 6
Species specifically associated with ponds that have 'diverse' plant communities, compared to ponds that have predominantly 'reeds', 'ephemeral' and 'grassy' plant communities. The comparison is made only for ponds that dry out each year.

Alisma plantago-aquatica	*Ranunculus lingua*
Capsella bursa-pastoris	*Ranunculus scleratatus*
Elytrigia repens	*Rumex crispus*
Epilobium hirsutum	*Salix* spp.
Equisitum fluviatile	*Solanum dulcamara*
Filipendula ulmara	*Sonchus* sp.
Iris pseudacorus	*Sparganium erectum*
Juncus buffonius	*Tripleurospermum inodorum*
Juncus conglomeratus	*Typha latifolia*
Large unid sedge	*Vicia cracca*

The effects of root exudate production on the microbial flora may be complex and vary with time. For example, in our recent work on some small ponds of exactly known age, also at Druridge Bay (Taylor et al., 2019), lower C burial rates were correlated with abundant *Juncus articulatus*. A possible explanation is that the labile root exudates this species is known to promote microbial activity which decomposes organic matter (Dunn et al., 2016). Nevertheless, Kotas et al. (2019) demonstrated that for sedge wetlands, whilst bacteria were the initial beneficiaries of ^{13}C labelled exudates, fungi were the longer-term recipients. Fungi have been shown to have a major role in the decomposition of detrital matter, having the advantage of hyphal growth that can extend deep into the stored organic matter. Fungi also have the extracellular enzymes that can degrade the more recalcitrant organic matter fraction such as lignocellulose, though at slower rates than for bacterial degradation (Fontaine et al., 2011; Kotas et al., 2019).

3.2.4. The possible role of nitrogen fixation

For several of the plant species listed in Table 6, there are literature examples of either the same species or species within the same genera having rhizosphere associations with nitrogen-fixing bacteria. These include *Equisitum* (Andersson and Lundegårdh, 1999), *Iris* (Chung et al., 2015), *Juncus* (Tjepkema and Evans, 1976), *Sonchus* (Hong et al., 2009), *Typha* (Biesboer, 1984) and *Viccia* (Van Cauwenberghe et al., 2014). There is evidence from soils under trees whose rhizosphere was associated with nitrogen-fixing bacteria that the older (humified) carbon stocks are conserved in comparison to soil under non-nitrogen fixing trees (Binkley 2005; Resh et al., 2002). Whilst these are very different environments to the ponds in the present study (though these ponds do dry out), the effect may hold more generally and could be an important factor in conserving sediment carbon.

The discussion in this and the previous section highlights the importance of factors that affect the ecological balance of the soil microflora and the role this has on nutrient cycling and decomposition rates of organic matter (Fontaine et al., 2011). The rich and varied rhizosphere that arises from diverse plant communities will have an important function in determining this ecological balance, and therefore on carbon storage.

Our results suggest that the rhizosphere may be a key but overlooked driver of carbon storage in pond sediment, in need of investigation to maximise the effectiveness of pond creation and management for carbon storage.

3.2.5. Pond age

The age of the pond might also affect carbon accumulation. The precise age of the study ponds is not known accurately enough for all ponds to include it in the models. Most of the ponds were at least 40 years old, created by subsidence over old coal mines (Jeffries, 2012), although probably even older. One of the natural ponds is known to be over 100 years old and another, the youngest, at 19 years when sampled. New ponds may show a lag time of two or three years before carbon accumulation becomes substantive (Taylor et al., 2019), but all the ponds in this study are considerably older. The results do not suggest that the amount of time ponds have had to accumulate carbon may be a confounding factor in our data.

3.3. Overall carbon stocks

Our results highlight the need to quantify carbon density rather than just a percentage. The high % C in dune ponds was compensated by low bulk density so that the overall carbon density was lowest in these sites. Conversely, the low % of carbon the arable field ponds, which when dry would be exposed, baked and cracked mud, barely different from the surrounding soil that had not been inundated, was compensated by a high bulk density so that overall carbon density was relatively high.

In Table 7, we present the scaled-up sediment carbon stock for each land use type and for all ponds. The ponds' mean sediment carbon (4.18 $\pm$ 2.21 kg C $m^{-2}{}_{< 10\ cm}$) is in the midrange of values reported for habitats of the UK (range = 2.9–5.9 kg C m^{-2}; calculated as < 10 cm from values reported in Countryside Survey, 2007), being higher than those of coastal margins, agricultural land, grassland, and woodland, yet lower than wetlands, bogs, and fens, marshes and swamps. Given the relative youth of the ponds, the amount of SCS compared to many other habitats and the ease of pond creation within heterogeneous landscapes, the results show that ponds have the potential to be an important tool in the mitigation of C emissions.

4. Conclusions

The carbon buried in pond sediments is higher, volume for volume, than many other terrestrial habitats. The striking outcome of the survey is that individual ponds show considerable variation in carbon stocks, sufficient to obscure many systematic differences that might be expected due to land use, vegetation, or drying out. However, this result also suggests that a global estimate combining the data from all ponds is useful regardless of categorisations such as land use. The combined measure of sediment carbon stocks is 4.18 $\pm$ 2.12 kg C m^{-2}, over the top 10 cm, a first estimate for typical lowland temperate ponds. Variation within individual ponds is less than between ponds: future studies should maximise the number of ponds sampled to capture inter-pond variation, one core per pond.

The results suggest that the drying regime and vegetation of ponds deserve more detailed investigation as potential drivers of carbon accumulation. Such considerations will be important if ponds are constructed to capture and hold carbon to maximise their effectiveness as carbon sinks. The relative ease of pond creation suggests their potential as an application to help maximise carbon sequestration at the landscape scale. Recent proposals for landscape rewilding have explicitly included ponds as a means of carbon sequestration (Rewilding Britain, 2019), and our study shows that ponds can indeed play a significant role.

Credit roles statement

Gilbert: Conceptualisation, Data curation, Formal analysis. original writing, review, Taylor: Data curation, review, Cooke: Conceptualisation, Data curation, review, Deary: Conceptualisation, Supervision, original writing, review, Jeffries: Conceptualisation, Supervision, Formal analysis, original writing, review

Data repository

The data set is lodged with the Mendeley repository: Jeffries et al. (2019), "Druridge Bay (Northumberland, UK) pond sediment carbon core data for 40 ponds", Mendeley Data, V1, https://doi.org/10.17632/wmbhzhdr6b.1.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Table 7
Scaling up pond sediment carbon stocks (SCS) to Kg C m^{-2}. The core data have been used to estimate the carbon stock in a 1 m^2 area of sediment over a depth of 10 cm, including a global figure for all ponds combined.

	Pond type				
	Naturalistic	Arable	Pasture	Dune	All ponds
SCS, mean $\pm$ SD kg C m^{-2}	3.01 $\pm$ 0.77	3.04 $\pm$ 0.48	4.74 $\pm$ 0.70	5.92 $\pm$ 3.25	4.18 $\pm$ 2.12
Min, Max, kg C m^{-2}	2.14, 4.90	2.26, 3.54	3.43, 5.98	1.12, 11.81	1.12, 11.81

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2020.111698.

References

Abnixar, A., Young, K.L., Lafrenière, M.J., 2014. Pond hydrology and dissolved carbon dynamics at Polar Bear pass wetland, Bathurst Island, Nunavut, Canada. Ecohydrology 7, 73–90.
Abnizova, A., Siemens, J., Langer, M., Boike, J., 2012. Small ponds with major impact: the relevance of ponds and lakes in permafrost landscapes to carbon dioxide emissions. Global Biogeochem. Cycles 26. https://doi.org/10.1029/2011GB004237, 2012.
Altenfelder, S., Kollmann, J., Albrecht, H., 2016. Effects of farming practice on populations of threatened amphibious plant species in temporarily flooded arable fields: implications for conservation management. Agric. Ecosyst. Environ. 222, 30–37.
Andersson, T.N., Lundegårdh, B., 1999. Growth of field horsetail (*Equisetum Arvense*) under low light and low nitrogen conditions. Weed Sci. 47, 41–46.
Àvila, N., López-Flores, R., Boix, D., Gascón, S., Quintana, X.D., 2016. Environmental factors affecting the balance of autotrophs versus heterotrophs in the microbial foodwebs of temporary ponds. Hydrobiologia 782, 127–143.
Bais, H.P., Weir, T.L., Perry, L.G., Gilroy, S., Vivan, J.M., 2006. The role of root exudates in rhizosphere interactions with plantB and other organisms. Annu. Rev. Plant Biol. 57, 233–266.
Baker, C., Lawrence, R., Montagne, C., Patten, D., 2006. Mapping wetlands and riparian areas using Landsat ETM+ imagery and decision-tree-based models. Wetlands 26, 465.
Biesboer, D.D., 1984. Nitrogen fixation associated with natural and cultivated stands of *Typha latifolia* l. (Typhaceae). Am. J. Bot. 71, 505–511.
Biggs, J., Fumetti, S. von, Kelly-Quinn, M., 2017. The importance of small water bodies for biodiversity and ecosystems services: implications for policy makers. Hydrobiologia 793, 3–39.
Binkley, D., 2005. How nitrogen-fixing trees change soil carbon. In: Binkley, D., Menyailo, O. (Eds.), Tree Species Effects on Soils: Implications for Global Change. Springer, Dordrecht, pp. 155–164.
Bissels, S., Donath, T.W., Hölze, l.N., Otte, A., 2005. Ephemeral wetland vegetation in irregularly flooded arable fields along the northern Upper Rhine: the importance of persistent seedbanks. Phytocoenologia 35, 469–488.
Bowden, W.B., 1987. The biogeochemistry of nitrogen in freshwater wetlands. Biogeochemistry 4, 313–348.
Catalán, N., von Schiller, D., Marcé, R., Koschorreck, M., Gomez-Gnere, L., Obrador, B., 2014. Carbon dioxide efflux druing the flooding phase of temporary ponds. Limnética 33, 349–360.
Céréghino, R., Boix, D., Cauchie, H.-M., Martens, K., Oertli, B., 2014. The ecological role of ponds a changing world. Hydrobiologia 723, 1–6.
Christensen, T.R., Panikov, N., Mastepanove, M., Joabsson, A., Stewart, A., Őquist, M., Sommerkorn, M., Reynaud, S., Svensson, B., 2003. Biotic controls on CO_2 and CH4 exchange in wetlands – a closed environment study. Biogeochemistry 64, 337–354.
Chung, E.J., Park, T.S., Kim, K.H., Jeon, C.O., Lee, H.-I., Chang, W.-S., Aslam, Z., Chung, Y.R., 2015. *Nitrospirillum irinus* sp. Nov., a diazotrophic bacterium isolated from the rhizosphere soil of *Iris* and amended description of the genus *Nitrospirillum*. Anton. Leeuw. Int. J. G. 108, 721–729.
Cieślewicz, J., 2014. Polyphenolic compounds in lacustrine sediments. Pol. J. Environ. Stud. 23, 1965–1973.

Cole, J.J., Prairie, Y.T., Caraco, N.F., McDowell, W.H., Tranvik, L.J., Streigl, 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, 171–184.

Dalu, T., Weyl, O.L.F., Froneman, P.W., Wasserman, R.J., 2016. Trophic interactions in an austral temperate ephemeral pond inferred using stable isotope analysis. Hydrobiologia 768, 81–94.

Davies, B.R., Biggs, J., Williams, P., Whitfield, M., Nicolet, P., Sear, D., Bray, S., Maund, S., 2008. Comparative biodiversity of aquatic habitats in the European agricultural landscape. Agric. Ecosyst. Environ. 125, 1–8.

Dean, W., Gorham, E., 1998. Magnitude and significance of carbon burial in lakes, reservoirs and peatlands. Geology 26, 535–538.

DeClerck, S., De Bie, T., Ercken, D., Hampel, H., Schijver, s S., Van Wichelen, J., Gillard, V., Mandiki, R., Losson, B., Bauwnes, D., Keijers, S., Vyverman, W., Goddeeris, B., De Meester, L., Brendonck, L., Martens, K., 2006. Ecological characteristics of small farm ponds: associations with land use practices at multiple spatial scales. Biol. Conserv. 131, 523–532.

Downing, J.A., 2010. Emerging global role of small lakes and ponds: little things mean a lot. Limnética 29, 9–24.

Downing, J.A., Prairie, Y.T., Cole, J.J., Duarte, C.M., Tranvik, L.J., Streigl, R.G., McDowell, W.H., Kortelainene, P., Caraco, N.F., Melack, J.M., Middelburg, J.J., 2006. The global abundance and size distribution of lakes, ponds, and impoundments. Limnol. Oceanogr. 51, 2388–2397.

Downing, J.A., Cole, J.J., Middleburg, J.J., Strieg, l R.G., Duarte, C.M., Kortelainen, P., Praoirie, Y.Y., Laube, K.A., 2008. Sediment organic carbon burial in agriculturally eutrophic impoundments over the last century. Global Biogeochem. Cycles 22, 1–10.

Dunn, C., Jones, T.G., Roberts, S., Freeman, C., 2016. Plant species effects on carbon storage capabilities of a blanket bog complex. Wetlands 36, 47–58.

Elnaggar, A.A., Noller, J.S., 2010. Application of remote-sensing data and decision-tree analysis to mapping salt-affected soils over large areas. Rem. Sens. 2, 151–165.

Fontaine, S., Hénault, C., Aamor, A., Bdioui, N., Bloor, J., Maire, V., Mary, B., revailott, S., Maron, P.A., 2011. Fungi mediate long term sequestration of carbon and nitrogen in soil through their priming effect. Soil Biol. Biochem. 43, 86–96.

Fromin, N., Pinay, G., Montuelle, B., Landais, D., Ourcival, J.M., Joffre, R., Lensi, R., 2010. Impact of seasonal sediment dessication and rewetting on microbial processes involved in greenhouse gas emissions. Ecohydrology 3, 339–348.

Gilbert, P.J., Taylor, S., Cooke, D.A., Deary, M., Jeffries, M.J., 2014. Variations in sediment organic carbon among different types of small natural ponds along Druridge Bay, Northumberland. Inland Waters 4, 57–64.

Gilbert, P.J., Cooke, D.A., Deary, M., Taylor, S., Jeffries, M.J., 2016. Quantifying rapid spatial and temporal variations of CO_2 fluxes from small, lowland freshwater ponds. Hydrobiologia 793, 83–93.

Graefe, E., Veit, M., 1999. Urinary metabolites of flavonoids and hydroxycinnamic acids in humans after application of a crude extract from equisetum arvense. Phytomedicine 6, 239–246.

Hall, D.L., Willig, M.R., Moorhead, D.L., Sites, R.W., Fish, E.B., Molhagen, T.R., 2004. Aquatic macroinvertebrate diversity of playa wetlands : the role of landscape and island biogeographic characteristic. Wetlands 24, 77–91.

Holgerson, M.A., Raymond, P.A., 2016. Large contribution to inland waters CO_2 and CH_4 emissions from very small ponds. Nat. Geosci. https://doi.org/10.1038/NGEO2654.

Hong, Y.-Y., Ma, Y.-C., Zhou, Y.-G., Gao, F., Liu, H.-C., Chen, S.-F., 2009. *Paenibacillus sonchi* sp. Nov., a nitrogen-fixing species isolated from the rhizosphere of *Sonchus oleraceus*. Int. J. Syst. Evol. Microbiol. 59, 2656–2661.

Jeffries, M.J., 1998. Pond macrophyte assemblages, biodisparity and spatial distribution of ponds in the Northumberland coastal plain, UK. Aquat. Conserv. 8, 657–667.

Jeffries, M.J., 2012. Ponds and the importance of their history: an audit of pond numbers, turnover and the relationship between the origins of ponds and their contemporary plant communities in south east Northumberland, UK. Hydrobiologia 689, 11–12.

Jeffries, M.J., 2016. Flood, drought and the inter annual variation to the number and size of ponds and small wetlands in an English lowland landscape over three years of weather extremes. Hydrobiologia 768, 255–272.

Jeffries, M.J., Epele, L., Studinski, J.M., Vad, C.F., 2016. Invertebrates in temporary wetland ponds of the temperate biomes. In: Batzer, D., Boix, D. (Eds.), Invertebrates in Freshwater Wetlands. An International Perspective on Their Ecology. Springer, Switzerland, pp. 105–140.

Jeffries, Michael, Gilbert, Pete, Deary, Mike, Taylor, Scott, 2019. Druridge Bay (Northumberland, UK) Pond Sediment Carbon Core Data for 40 Ponds. Mendeley Data. https://doi.org/10.17632/wmbhzhdr6b.1 vol. 1.

Kelker, D., Chanton, J., 1997. The effect of clipping on methane emissions from *Carex*. Biogeochemistry 39, 37–44.

Koelbener, A., Strom, L., Edwards, P.J., Venterink, H., 2010. Plant species from mesotrophic wetlands cause relatively high methane emissions from peat spoil. Plant Soil 326, 147–158.

Kotas, P., Edwards, K., Jandová, K., Kaštovská, E., 2019. Interaction of fertilization and soil water status determine c partitioning in a sedge wetland. Soil Biol. Biochem. 135, 85–94.

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. Nature, Scientific Reports 8 article 9535.

Lansdown, R.V., 2008. Water starworts Callitriche of Europe. In: BSBI Handbook No 11. BSBI, London.

Marcé, R., Obrador, B., Gómez-Gener, L., Catalón, N., Koschorereck, M., Arce, M.I., Singer, G., von Schiller, D., 2019. Emissions from dry land waters are a blind spot in the global carbon cycle. Earth Sci. Rev. 188, 240–248.

Martinsen, K.T., Kragh, T., Sand-Jenssen, K., 2019. Carbon dioxide fluxes of air exposed sediments and dessicating ponds. Biogeochemistry doi.org/10.1007/s10533-019-00579-0.

Mulholland, P.J., Elwood, J.W., 1982. The role of lake reservoir sedimentss as sinks in the perturbed global carbon cycle. Tellus 34, 490–499.

Neag, T., Toma, C.-C., Olah, N., Ardelean, A., 2017. Polyphenols profile and antioxidant activity of some Romanian *Ranunculus* species. Studia Ubb. Chemia 62, 75–88.

Nicolet, P., Biggs, J., Fox, G., Hodson, M.J., Reynolds, C., Whitfield, M., Williams, P., 2004. The wetland plant and macroinvertebrate assemblages of temporary ponds in England and Wales. Biol. Conserv. 120, 261–278.

Orhan, I., Kartal, M., Abu-Asaker, M., Şenol, F.S., Yilmaz, G., Şener, B., 2009. Free radical scavenging properties and phenolic characterization of some edible plants. Food Chem. 114, 276–281.

Pittman, B., Jones, J.R., Millspaigh, J.J., Kremer, R.J., Downing, J.A., 2013. Sediment organic carbon distribution in 4 small northern Missouri impoundments: implications for sampling and carbon sequestration. Inland Waters 3, 39–46.

Pond Action, 1998. A Guide to the Methods of the National Pond Survey. Pond Conservation, Oxford.

Resh, S.C., Binkley, D., Parrotta, J.A., 2002. Greater soil carbon sequestration under nitrogen-fixing trees compared with eucalyptus species. Ecosystems 5, 217–231.

Rewilding Britain, 2019. Rewilding and Climate Breakdown. How Restoring Nature Can Help Decarbonise the UK. Rewilding Britain. https://www.rewildingbritain.org.uk/.

Rodwell, J.S., 1995. British Plant Communities. Volume 4. Aquatic Communities. Swamps and Tall Herb Fens. Cambridge University Press, Cambridge.

Rubbo, M.J., Cole, J.J., Kiesecker, J.M., 2016. Terrestrial subsidies of organic carbon support net ecosystem production in temporary forest ponds: evidence from an ecosystem experiment. Ecosystems 9, 1170–1176.

Seekell, D.A., Pace, M.L., Travnik, L.J., Verpoorter, C., 2013. A fractal based approach to lake-size distributions. Geophys. Res. Lett. 40, 517–521.

Shahzad, T., Chenu, C., Genet, P., Barot, S., Perveen, N., Mougin, C., et al., 2015. Contribution of exudates, arbuscular mycorrhizal fungi and litter depositions to the rhizosphere priming effect induced by grassland species. Soil Biol. Biochem. 80, 146–155.

Shotbolt, L.A., Thomas, a.D., Hutchinson, S.M., 2005. The use of reservoir sediments as environmental archives of catchment inputs and atmospheric pollution. Prog. Phys. Geogr. 29, 337–361.

Stace, C., 1997. New Flora of the British Isles, second ed. Cambridge University Press, Cambridge.

Taylor, S., Gilbert, P.J., Cooke, D.A., Deary, M.E., Jeffries, M.J., 2019. High carbon burial rates by small ponds in the landscape. Front. Ecol. Environ. 17, 25–31.

Tjepkema, J., Evans, H., 1976. Nitrogen fixation associated with *Juncus balticus* and other plants of Oregon wetlands. Soil Biol. Biochem. 8, 505–509.

Torgerson, T., Branco, B., 2008. Carbon and oxygen fluxes from a small pond to the atmosphere: temporal variability and the CO2/O2 imbalance. Water Resour. Res. 44 https://doi.org/10.1029/2006WR005634, 2008.

Tóth, B.H., Blazics, B., Kéry, Á., 2009. Polyphenol composition and antioxidant capacity of *Epilobium* species. J. Pharmaceut. Biomed49 26–31.

Tranvik, L.J., Cole, J.J., Prairie, Y.T., 2018. The study of carbon in inland waters – from isolated ecosystems to players in the global carbon cycle. Limnol. Oceanography Letters 3, 41–48.

Tsais, J.-S., Venne, L.S., McMurry, S.T., Smith, L.M., 2011. Local and landscape influences on plant communities in playa wetlands. J. Appl. Ecol. 49, 174–181.

Usio, N., Nakagawa, M., Aoki, T., Higuchi, S., Kadono, Y., Akasaka, M., Takamura, N., 2017. Effects of land use on trophic states and multi-taxonomic diversity in Japanese farm ponds. Agric. Ecosyst. Environ. 247, 205–215.

Van Cauwenberghe, J., Verstraete, B., Lemaire, B., Lievens, B., Michiels, J., Honnay, O., 2014. Population structure of root nodulating rhizobium leguminosarum in vicia cracca populations at local to regional geographic scales. Syst. Appl. Microbiol. 37, 613–621.

Vanni, M.J., Rensick, W.H., Bowling, A.M., Horgan, M.J., Christian, A.D., 2011. Nutrient stoichiometry of linekd catchment-lake systems along a gradient of land use. Freshw. Biol. 56, 791–811.

Verpoorter, C., Kutser, T., Seekell, D.A., Travnik, L.J., 2014. A global inventory of lakes based on high-resolution satellite images. Geophys. Res. Lett. 41, 6396–6402.

Wik, M., Varner, R.K., Anthony, K.W., MacIntyre, S., Bastviek, D., 2016. Climate-sensitive northern lakes and ponds are critical components of methane release. Nat. Geosci. 9, 99–106.

Yakimovich, K.M., Emilson, E.J., Carson, M.A., Tanentzap, A.J., Basiliko, N., Mykytczuk, N., 2018. Plant litter type dictates microbial communities responsible for greenhouse gas production in amended lake sediments. Front. Microbiol. 9 https://doi.org/10.3389/fmicb.2018.02662.