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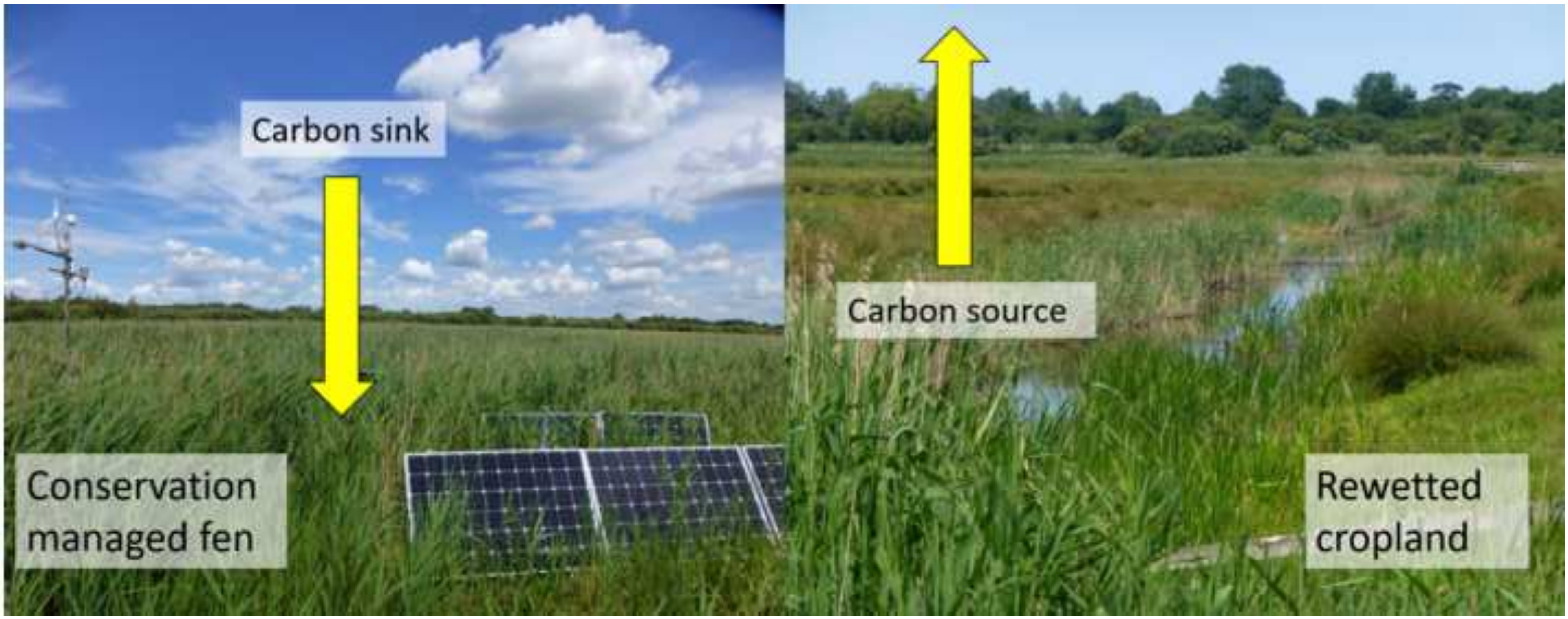
Graphical Abstract



Highlights (for review)

- We measure the full C balance of a rewetted cropland and a semi-natural fen
- For both sites, net ecosystem exchange was the largest component of the C budget
- Fluvial C losses were small at both sites
- The semi-natural fen was a C sink, the rewetted fen a C source
- Higher water tables are needed to reduce C losses in rewetted croplands

The full carbon balance of a rewetted cropland fen and a conservation-managed fen

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Abstract

On a global scale, the release of greenhouse gases (GHG) from peatland drainage and cultivation are believed to account for ~5% of estimated anthropogenic GHG emissions. Drainage generally leads to peat subsidence and extensive soil loss, resulting in a diminishing store of soil carbon (C). This is a challenge for maintaining drainage-based agriculture, as such practices will eventually lead to the loss of organic soils that arable cultivation depends on. The conversion of croplands on peat to semi-natural grasslands, alongside raising water tables, is one possible way to reduce the loss of these valuable C stores. Here, we report the net ecosystem carbon balances (NECB) of two lowland peatlands in East Anglia, south-east UK. One site is a relic conservation-managed fen on deep peat, subject to active hydrological management to maintain water levels, and dominated by *Cladium* and *Phragmites* sedge and reed beds, whilst the other is a former cropland that has been converted to seasonally-inundated grazed grassland. Despite occasionally experiencing severe water table drawdown, the conservation-managed fen was a strong C sink of -104 g C m^{-2} yr^{-1}. In contrast, the grassland was a C source of 133 g C m^{-2} yr^{-1}, with gaseous carbon dioxide (CO_2) emissions being the main loss pathway, due to low water tables exposing the soil profile in summer. At each site, ditch emissions of CO_2 were moderately large (22 and 37 g C m^{-2} yr^{-1}), whilst ditch methane (CH_4) emissions (0.2 and 1.8 g C m^{-2} yr^{-1}) made a negligible contribution to the NECB, but are important when considering the ecosystem GHG balance in terms of CO_2 equivalents. Excluding dissolved inorganic carbon (DIC), fluvial C losses were 6 g C m^{-2} yr^{-1} for the conservation-managed fen and 12 g C m^{-2} yr^{-1} for the former cropland, and were dominated by dissolved organic carbon (DOC). The small fluvial C loss is the result of both sites being hydrologically isolated from the surrounding agricultural landscapes. Although the partially re-wetted cropland was still acting as a net C source, our estimates suggest that seasonal rewetting has reduced net annual C losses to ~20% of their former cropland values. Maintaining high water tables year round would potentially further reduce C losses, and shallow inundation might allow the return of wetland species such as *Phragmites* and *Typha*, perhaps as floating rafts.

Keywords: peatland, net ecosystem carbon balance, greenhouse gas, dissolved organic carbon, restoration, drainage

"The height of a man in the life of a man." – old East Anglian saying describing peat losses due to subsidence.

1. Introduction

Globally, approximately 300,000 km^2 (~7 %) of peatlands are used for agriculture (International Peat Society, 2008), including extensive areas of lowland peat that have been drained and converted to intensive arable use, and are now important areas for food production (Joosten and Clarke, 2002). Drainage generally disrupts the natural functioning of the peatland carbon (C) store, leading to increased emissions of nitrous oxide (N_2O) (Haddaway *et al*., 2014) and carbon dioxide (CO_2), as extensive peat losses occur due to this oxidation (Hooijer *et al*., 2012). The most recent report of the Intergovernmental Panel on Climate Change (IPCC) emphasises the importance of CO_2 emissions from oxidation of cultivated peatlands (primarily in Europe and Southeast Asia) (Smith *et al*., 2014), and it has been estimated that greenhouse gas (GHG) emissions from drained and burned peatlands account for 5% of anthropogenic emissions (Global Peatlands Initiative, 2017).

As concern for peatland C stocks has grown, there has been an increased emphasis on restoring and rewetting bogs and fens that have been disturbed by human activities (e.g. Wilson *et al*., 2016), and the potential global importance of such work on GHG emissions was recognised in the development of a reporting methodology for wetland drainage and rewetting in the IPCC Wetland Supplement (IPCC, 2014). The Paris Climate Agreement commits nations to limiting climatic warming to less than 2°C (Rogelj *et al*., 2016). This commitment will require zero net CO_2 emissions by 2050 (Matthews and Caldeira, 2008) which, because all realistic future scenarios involve some level of continued fossil fuel emission, will necessitate the development of compensating measures which remove CO_2 from the atmosphere; i.e. negative emissions. The recent “4 per 1000” initiative proposes that a significant fraction of this target could be achieved through enhanced sequestration of C into soils (Minasny *et al*., 2017). Wetlands, in particular, have been highlighted as being key in delivering “natural climate solutions” due to their potential to accumulate and retain C (Griscom *et al*., 2017). For lowland agricultural fens, it has been shown that restoration can reduce oxidation-induced losses of peat and, in some cases, lead to the re-establishment of their function as a C sink (Knox *et al*., 2014).

However, it can be difficult to restore agricultural land back to a properly functioning fen ecosystem (Stroh *et al*., 2013). Reasons for such difficulty include extensive peat loss through oxidation and the compaction of remaining peat, loss of local seed banks, heavily modified drainage systems and previous addition of silt to the peatland via warping (the agricultural practice of diverting mineral-rich river water onto peat soils to deposit sediment) (Smart *et al*., 1986). Where ‘complete’ restoration is impossible, it may nevertheless be feasible to convert agricultural land to semi-natural fen meadows, which will still bring associated increases in biodiversity and ecosystem services (Klimkowska *et al*., 2010), and may also reduce rates of C loss (Hendriks *et al*., 2007). In much of Europe, including parts of UK and the Netherlands, the target ecosystem for fen restoration is a semi-natural environment involving ongoing water-level and vegetation management (Klötzli and Grootjans, 2001), for example to maintain or enhance plant species richness (Menichino *et al*., 2016). However, sometimes it may be that constraints in water availability result in unexpected vegetation shifts, often in undesired directions, which may limit the success of restoration attempts (Klötzli and Grootjans, 2001).

Knowledge gaps still remain on the effects of agricultural fen restoration on C and nutrient cycling, and on how the functioning of these restored ecosystems compares to conservation-managed fens that have never been under agricultural production. For instance, Tiemeyer *et al*. (2016) found that CO_2 emissions increased with deeper water tables in drained peat grasslands, but could not model CO_2 fluxes across multiple sites solely as a function of water table, and suggested that additional factors such as drought stress could result in lower emissions (because CO_2 fluxes from respiration are limited by both very dry and very wet soil conditions). Contrary to this, it has sometimes been shown that drained grasslands can be CO_2 sinks, and could act as C stores depending on management practices; e.g. large amounts of biomass removal could counteract a terrestrial CO_2 sink and result in a C source (Renou-Wilson *et al*. 2014). However, methane (CH_4) can still be emitted by drained soils (Hendriks *et al*., 2007, Henneberg *et al*., 2015), with implications for C and GHG budgets.

The East Anglian fens are the largest and most intensively modified area of lowland peat in the UK. In their original form they occupied approximately 150,000 ha (Burton and Hodgson, 1987), but drainage and agricultural conversion has resulted in just 12,600 ha of deep peat remaining, which now stores an estimated 41 Tg of C (Holman, 2009). Of this remaining peat, approximately 800 ha exists as undrained fen, in four separate nature reserves. The aphorism quoted above, assuming a man of 170 cm living for sixty years, would result in a peat loss of 2.8 cm per year. This figure falls within the range of 0.27-3.09 cm per year (mean = 1.37 cm) for the region reported by Richardson and Smith (1977). As a consequence of peat subsidence, much of the land in the region is now below mean sea level, and a complex series of ditches, embankments, sluices and pumps control the area's hydrology. Although the region is of national significance for the production of arable and horticultural crops, several projects are now underway to return areas of agricultural land to wetland.

To understand how fen management affects hydrology and C cycling, we established an intensive field measurement programme spanning three growing seasons at two adjacent sites; one a conservation-managed fen on deep peat, and one a former cropland on shallow peat that has been converted to seasonally-inundated grazed meadow grassland. The conservation-managed fen is an example of the target ecosystem for successful rewetting in the region, whilst the former cropland represents an ecosystem that has been removed from agricultural production and set on a restoration trajectory towards a semi-natural status. Two different projects (the Wicken Fen Vision and the Great Fen Project) within the region currently aim to restore a combined total of 9000 ha of wetlands, primarily by taking agricultural land out of production (Peh *et al*., 2014). In addition to C sequestration, these projects aim to deliver ecosystem services such as flood protection, nature-based recreation, grazing provision and increased biodiversity (Peh *et al*., 2014). We therefore measured both gaseous C exchanges and fluvial C losses, thereby enabling complete C budgets to be calculated, thus determining: 1) whether the conservation-managed fen is a net C sink and; 2) what effect conversion to grassland has had on the C budget of the former arable fen.

2. Materials and methods

2.1. Field sites

Both field sites are part of the Wicken Fen National Nature Reserve which is owned and managed by the National Trust, a conservation organisation. Mean annual air temperature from an automatic weather station (AWS) on site was 9.3 ˚C in 2013, 10.9 ˚C in 2014 and 10.3 ˚C in 2015. Missing data from the AWS precludes the calculation of site-specific annual rainfall totals, but rainfall was 648 mm, 765 mm, and 641 mm in 2013, 2014 and 2015 at another lowland site 27 km away (Evans *et al*., 2016a). Rainfall was measured on the study site in 2015 using a manual rain gauge, with an annual total of 643 mm.

The conservation-managed site is referred to as the Wicken Sedge Fen (52.31° N, 0.28° E, area = 61 ha, 2-3 m above sea level). It lies on a surviving area of deep peat and contains large areas of reed bed that are cut on approximately a three year rotation. Sedge Fen has never been agriculturally drained, and has been under active conservation management since 1899, thereby making it the oldest nature reserve in the UK. The dominant plant species present are saw sedge *Cladium mariscus* and common reed *Phragmites australis,* with abundant reed canary grass *Phalaris arundinacea,* and some purple small-reed *Calamagrostis canescens* (Eades, 2016). A network of ditches cross the site, which are used for water level management rather than drainage; the ditches are not permanently connected to the wider river network in the region, but water may be transferred onto the fen from the adjacent river (named Wicken Lode). The fen has no defined outflow, i.e. it is not drained by a stream. Much of the perimeter is bunded to minimise water loss to the surrounding agricultural landscape, which is at a lower elevation as a result of subsidence, and it is assumed that water losses occur laterally as surface/subsurface flow. A dense 'aquitard' layer has been identified within the peat column that reduces water movement downwards to the deep groundwater (Boreham, 2017). As long ago as 1908, concerns were raised that the fen was becoming drier (Yapp, 1908). However, there has been more recent concern that water tables at Sedge Fen are declining compared to historical measurements (McCartney and de la Hera, 2004), so in 2011 a wind pump was installed to pump mineral-rich river water onto the site. Water is pumped onto site for the period November-March due to restrictions imposed by the Environment Agency (a government body responsible for environmental protection), whereby water distribution is prioritised for agricultural use. The catchment of the river (Wicken Lode/New River) upstream from the field sites is 27.6 km^2 (McCartney and de la Hera, 2004) and is principally arable farmland.

The former cropland is known as Baker's Fen (52.30° N, 0.29° E, area = 56 ha, 0-1 m above sea level), which is approximately 200 m away from Sedge Fen. Bakers Fen was drained in the mid-

19th century for agriculture, resulting in extensive peat loss and subsidence. It has been under conservation management since 1994. Restoration activities have included cessation of arable cultivation, enclosure of the fen in a waterproof membrane in 1994 to retain water inputs, re-seeding of the fen in 1995 with an unknown “grass mixture” (Saltmarsh, 2000), and the excavation of several scrapes to create seasonal standing water bodies. Like Sedge Fen, Baker’s Fen is hydrologically isolated, as it is not connected to the surrounding network of rivers or to groundwater. Current hydrological management involves transferring river water onto site during November-March, and the site is extensively grazed by Highland cattle and Konik ponies. A network of ditches cross the site, and water is released from the fen on an occasional, *ad-hoc* basis using a sluice in the western corner. Bakers Fen supports a species-poor damp grassland, reflecting its past management history. Dry areas are a mesotrophic grassland community dominated by *Agrostis stolonifera* with *Arrhenatherum elatius, Cirsium arvense, Dactylis glomerata* and *Holcus lanatus*. Wetter areas, such as the scrapes, are rush pasture communities, featuring hard rush (*Juncus inflexus*) and *Agrostis stolonifera* (Eades, 2016).

Fieldwork started in April 2013 and finished in October 2015 and comprised repeated measurements of fluvial and gaseous GHGs, water tables, and water chemistry. Soil properties were measured on one occasion in April 2013. Both field sites were visited during each sampling trip. Typically this meant that sites would be sampled on consecutive days or, sometimes, on the same day.

2.2. Soil properties

Peat depth was measured at 24 locations at each site using a Dutch auger. The measurements were taken in an area of 18 m x 18 m, centred on the flux towers (see section 2.3). Two soil cores were taken at each site using a Dutch auger from the areas of deepest peat that we measured within the sampled area. The cores were sectioned in the field at pre-determined intervals and the samples brought back to the laboratory. The sections chosen were 5 cm increments to 20 cm depth, and then 10 cm increments to the base. All sections of the cores were analysed for dry bulk density, and selected samples were analysed for pH, mineral content, and C and N elemental content. Samples were dried at 105°C for 16 hours and checked for no further mass loss, and their bulk density measured prior to further processing. Three sets of sub-samples were then taken. One set of sub-samples was analysed for pH by placing each sample in 0.01 M $CaCl_2$ at a mass to volume ratio of 1:10. The second set of sub-samples were ashed at 550°C for 4 hours and the residual mass recorded as the mineral content. The third set were used for C and N elemental analysis: triplicate samples were milled to a sub-mm powder using a 6770 Freezer/Mill (Spex, Metuchen, USA). The ground samples were then analysed on an ECS 4010 elemental combustion system with a pneumatic autosampler (Costech, Santa Clarita, USA), using acetanilide standards. All samples were corrected for their measured ash content and expressed as their molar ratio.

2.3. CO_2 eddy-covariance fluxes

Both sites were instrumented with open-path eddy covariance (EC) flux towers to measure ecosystem-scale CO_2 fluxes. The instrumentation comprised a Solent R3 sonic anemometer (Gill Instrument Ltd. Lymington, UK) at Sedge Fen, and a CSAT3 sonic anemometer (Campbell Scientific Inc. Logan Utah, USA) at Baker's Fen for measurements of the three components of atmospheric turbulence and sonic temperature. An LI7500A open path analyser (LI-COR Biosciences, Lincoln, Nebraska, USA) was used to measure concentrations of atmospheric water vapour and CO_2 as well as barometric pressure at both fens. At both sites, EC data were scanned at 20 Hz and logged using a LI-COR LI7550 Analyser Interface Unit (LI-COR Biosciences, Lincoln, Nebraska, USA). EC systems were installed at central locations within the two sites to maximise each particular land use within the tower fetch under prevailing south-westerly wind conditions. Measurements were made at heights above the ground surface of 3.9 m and 2.3 m at Sedge Fen and Bakers Fen, respectively.

A range of ancillary meteorological and soil physical measurements were made at each flux tower. The net radiation and its incoming and outgoing short- and longwave components were measured using CNR1 net radiometers (Kipp and Zonen BV, Delft, The Netherlands). Soil heat fluxes were measured at a depth of 5 cm below the soil surface using HFP01 soil heat flux plates (Hukseflux Thermal Sensors BV, Delft, The Netherlands). Air temperature and relative humidity were measured with HMP45 probes (Vaisala, Vantaa, Finland) installed at 2 m above the ground surface.

Raw (20 Hz) EC data were post-processed using EddyPRO® Flux Calculation Software (LI-COR Biosciences, Lincoln, Nebraska, USA). Thirty minute flux densities were computed as the mean covariance between the vertical wind speed and atmospheric scalar quantities (e.g. H_2O, CO_2). Fluxes were calculated using block averaging and by applying standardized procedures and corrections.

An extensive data loss occurred at Sedge Fen during the latter half of 2014. Because of this, we calculated Sedge Fen NEE as an annual period from July 2013 to June 2014, and as the full year for 2015. For Baker's Fen, annual NEE could be calculated for the full years of 2013, 2014 and 2015. More information concerning the eddy-covariance methods can be found in the supplementary information.

2.4. CO_2 and CH_4 static chamber fluxes

Static chambers to measure GHG fluxes were used on a total of 31 occasions at Sedge Fen, and 37 occasions at Baker's Fen, with a higher frequency in summer (every 2-3 weeks) than winter (every 4-8 weeks). All winter sampling occurred on snow-free days, and snowfall is rare in the region. Sampling started in May 2013 at Baker's Fen, but was delayed until August 2013 at Sedge Fen due to flooding making collar installation difficult. At each site, six polyvinyl chloride collars (20 cm high, 60 cm by 60 cm, inserted approximately 10 cm into the ground) were installed for CH_4 flux measurements. At Sedge Fen all collars were located in the same area, with three collars being dominated by *Phragmites australis* and three by *Cladium mariscus*. At Baker's Fen three collars were sited within an *Agrostis stolonifera*-dominated dry mesotrophic grassland community, and three within a *Juncus inflexus*-dominated rush pasture (see section 2.1). The *Agrostis* and *Juncus* sets of collars were sited 80 m apart. At each study site two ditch locations were selected for floating chamber measurements of CH_4 and CO_2.

To take flux measurements, transparent acrylic chambers were attached to the collars. Stackable intermediate chamber sections were used when vegetation was tall. Silicone sponge was used to create seals between chamber sections and collars. Small fans were used in all chamber sections to facilitate internal mixing. At Sedge Fen the water table was frequently near, or above, the peat surface, so boardwalk was used to minimise disturbance during sampling. CH_4 concentrations were measured in real time in the field using an Ultraportable Greenhouse Gas Analyzer (Los Gatos Research, San Jose, USA). Changes in CH_4 concentrations were observed using a tablet computer, and flux chambers were deployed until a linear flux, or clear zero flux, was observed, which was typically 1-5 minutes.

Chamber fluxes were calculated according to Green *et al*. (2018), assuming a linear relationship between chamber deployment time and mass change in CH_4. It has been common practice to only include flux data where the R^2 of this relationship is above a certain value, but, traditionally, fluxes have been calculated using several (~5) discrete gas samples analysed by gas chromatograph in the lab. However, we measured fluxes in real time in the field, with a sampling frequency of 1 Hz, thereby giving a much clearer picture of the behaviour of CH_4 emissions. Removing measurements with a low R^2 could lead to the exclusion of small but noisy fluxes, thus biasing the dataset towards higher fluxes. We therefore included all fluxes with a significant (≤ 0.05) *p* value.

An attempt was made at calculating annual CH_4 fluxes for terrestrial collars, and CH_4 and CO_2 ditch fluxes, using the method of Green *et al*. (2018). A variety of environmental variables were trialled in the models, including air temperature, soil temperature, water table depth, irradiance, and temperature sum index (ETI). The temperature and irradiance data were taken from flux towers. No satisfactory model fits were obtained. As such, annual CH_4 fluxes and ditch CO_2 fluxes were estimated between measurement dates: days without measurements were assumed to have the same flux as that recorded on the nearest day with a measurement (i.e. an approach equivalent to linear interpolation; Green and Baird (2017)).

We weighted the flux of ditch CH_4 and CO_2 using the method of Evans *et al*. (2016), whereby the annual flux expressed per unit of ditch surface is multiplied by the proportion of the fen occupied by ditches ($Frac_{ditch}$). $Frac_{ditch}$ was calculated using aerial photography and was 0.014 and 0.017 for Sedge Fen and Baker's Fen respectively.

2.5. Water sampling and analysis

Water sampling took place on 42 occasions, with a higher frequency in summer (every 2-3 weeks) than winter (every 4-5 weeks). Water sampling took place at four different ditch locations on each site. Additionally, one sample was taken from Wicken Lode; the river that is used as a water supply to transfer water onto both sites. At each sampling point, water was collected in a 60 ml Nalgene® and a 500 ml Nalgene® bottle, and a sample for dissolved GHG analysis was collected in a 12 ml borosilicate glass vial, using the headspace method (Hope *et al*., 2004). Air pressure and temperature were recorded at the time of sampling using a C4141 thermo-hygro-barometer (Commeter, Roznov pod Radhostem, Czech Republic) and water temperature was measured with a SuperFast Thermapen (ETI, Worthing, UK). After collection, samples were returned to the laboratory and stored in the dark at 4°C until analysis, typically within one week.

Electrical conductivity (EC) and pH were measured on the 60 ml sample with an Orion VERSA STAR (Thermo Scientific, Waltham, USA). The sample was then filtered at 0.45 µm. DOC (measured as non-purgeable organic carbon) and dissolved inorganic carbon (DIC) were measured on the filtered samples using a TOC analyser (Shimadzu, Kyoto, Japan). Nitrate was measured using an ELIT 8021 ion-selective electrode (NICO 2000, Harrow, UK) and appropriate standards (range 1-100 mg l^{-1}).

The 500 ml sample was used to measure particulate organic carbon (POC). For each sample, 500 ml of deionised water was passed through a 0.7 µm Whatman GF/F filter which was then combusted at 500°C for five hours, and weighed. 500 ml of sample was then filtered using the same filter, which was oven-dried at 105°C for five hours and weighed to give an estimate of suspended sediment. The filter was then placed in a furnace at 375°C overnight, and weighed a final time to provide an estimate of particulate organic matter (POM). POC was then calculated from POM using the regression equation of Ball *et al*. (1964).

The 12 ml headspace sample was analysed for CH_4 and CO_2 using the Ultraportable Greenhouse Gas Analyzer equipped with a sampling loop following Baird *et al*. (2010). For this, gas is continuously circulated in a loop through the inlet and outlet of the analyser, until the concentration stabilises. The headspace sample of dissolved gas is then injected into the loop and the concentration noted. Using the equations from Baird *et al*. (2010), it is then possible to calculate the true concentration of the dissolved gas.

2.6. Hydrology

Due to the sensitivity of Sedge Fen, there were considerable restrictions on hydrological instrumentation imposed by the landowner. However, a previous detailed analysis of the hydrology of the site concluded that the main control over water-table depths was the balance between rainfall and evapotranspiration, with other losses being comparatively minor (McCartney *et al*., 2001). At Sedge Fen the water table was measured next to the collars using a dipwell fitted with an Orpheus Mini (OTT, Kempten, Germany) pressure transducer with a 1 cm resolution, logging every hour. At Baker's Fen, a dipwell was positioned next to each set of collars. Each dipwell was fitted with a Level TROLL 500 (In-Situ, Fort Collins, USA) with an accuracy of 0.35 cm or better, and a resolution of 0.035 cm, logging every 15 minutes. An additional 10 manually-recorded dipwells were installed across the fen with measurements being taken approximately monthly. A third Level TROLL 500 was deployed directly into the ditch near the sluice outflow at Baker's Fen, to measure ditch water level. Ditch water level was also manually measured at locations where water samples were collected.

Data on pumped river water volumes were provided by the site owners. Rainfall data were provided by an AWS on Baker's Fen, or from a site 27 km away when data were missing from Baker's Fen (see section 2.1). Evapotranspiration measurements were calculated from flux tower measurements made on both sites. By using a mass balance approach, the output of water at each site could then be calculated as the sum of inputs plus/minus any changes in water storage:

$$P + Q_{in} + G_{in} = ET + Q_{out} + G_{out} + \Delta s \qquad \text{Equation 1}$$

where P is precipitation, Q_{in} is river water transferred onto site, Q_{out} is discharge, G_{in} and G_{out} are groundwater flows in and out, ET is evapotranspiration, and Δs is change in water storage. Due to the hydrologically isolated nature of both sites (see 2.1) G_{in} and G_{out} were considered negligible (McCartney *et al.*, 2001). The term Δs was calculated using the automated dipwells and a specific yield estimate for each site. For Sedge Fen a specific yield of 0.12 was used based on previous measurements at the site (McCartney et al., 2001). For Baker's Fen a specific yield of 0.36 was calculated using P, Qin, ET and changes in WT height. Water outputs and inputs were then combined with water chemistry data to calculate aquatic C losses and gains on a mean monthly basis. In some instances multiple samples had been collected in one calendar month with no samples collected in the previous or next calendar months. For these cases, if one sample was collected in the first or last few days of that month this sample was instead taken to represent the previous or next calendar month. To estimate mean annual aquatic C fluxes, a mean for each calendar month was calculated for all fluxes obtained for that month during the study period, and the twelve monthly means summed to give the annual flux; i.e. all data collected in, e.g. January, was combined, regardless of the year in which it was collected. This approach avoided seasonal bias that could result from the disparity in summer and winter sampling frequencies.

3. Results

3.1. Soil properties

There were clear differences observed in the physical and chemical soil properties of the two fens (Table 1). Although the peat at Baker's Fen had a high bulk density, peat depths were very low, resulting in a much lower C stock than Sedge Fen.

3.2. Hydrology

The water table at Sedge Fen was closer to the peat surface than at Baker's Fen, but both sites experienced considerable water table drawdown in summer (Fig. 1). For Sedge Fen, this drawdown was particularly pronounced in 2013, when the water table decreased to 83 cm below the surface. In 2014 and 2015 water tables at Sedge Fen fell to low points of approximately 30 cm and 50 cm. Water levels were above the surface in the winter/spring period, indicating site flooding. At Baker's Fen, water tables fell below the level of the logged dipwell at 73 cm (i.e. below the entire peat layer) every summer, and this was also the case in the manual dipwells (Fig. SI1). In 2014 and, to a lesser extent, in 2015, this drying out was punctuated by rainfall events that raised the water table for short periods of time. Water tables rose quickly in autumn following the transfer of water onto the site, so that the depth to water table was < 35 cm in November 2013 and 2014, and eventually < 5 cm in January. When referenced to a common datum, water levels within the monitored ditch at the southwest of the site were routinely lower than the water table (Fig. SI1).

Water discharge at both sites principally occurred during the winter months when rainfall, or rainfall plus water inputs, exceeded evapotranspiration. At Sedge Fen discharge occurred from October to March (Fig. 2), and the annual water flux (calculated as the sum of monthly means for all years) was 192 mm yr^{-1}. Water discharge at Baker's Fen occurred primarily during October to February (Fig. 2), but also during some summer months when excess summer rainfall resulted in discharge. The total water flux from Baker's Fen was 315 mm yr^{-1}.

3.3. Water chemistry and fluvial carbon losses

Mean pH was similar for both fens and river water. In contrast, there was a clear difference in EC and DOC concentration in the order of Baker's Fen > Sedge Fen > river (Table 2). DIC concentrations were high at both sites, and lower in the river (Fig. SI2). DOC concentrations in the ditches of Sedge Fen were relatively stable but fluctuated at Baker's Fen (Fig. SI3). There was a weak but significant (R^2 = 0.29, p < 0.001, n = 195, Fig. SI4) negative relationship between ditch water level and DOC concentration. POC concentrations were extremely low in the river but variable for both fens, with highest concentrations being observed during dry summer conditions (Fig. SI5). Nitrate concentrations were high in the river but lower at Baker's Fen and Sedge Fen, and showed a seasonal pattern, with peaks each winter (Fig. SI6).

Due to the low water fluxes from each fen, aquatic C losses were small (Fig. 3). For Sedge Fen the majority of aquatic C flux was in the form of DIC (mean annual flux 16.4 g C m^{-2} yr^{-1}) with a mean annual DOC flux of 4.1 g C m^{-2} yr^{-1}, and POC of 0.4 g C m^{-2} yr^{-1}. The dissolved CO_2 flux leaving the site via the ditch network was estimated to be 1.23 g C m^{-2} yr^{-1}, and dissolved CH_4 exports were negligible (< 0.01 g C m^{-2} yr^{-1}). Aquatic C fluxes for Baker's Fen followed a similar pattern, and were also dominated by DIC (mean annual fluxes 27.1 g C m^{-2} yr^{-1}). Exports of DOC were 8.8 g C m^{-2} yr^{-1}, POC 1.1 g C m^{-2} yr^{-1}, dissolved CO_2 1.8 g C m^{-2} yr^{-1} and dissolved CH_4 0.01 g C m^{-2} yr^{-1}. Fluxes of C onto both sites via managed inputs of river water were dominated by DIC (Fig. 3), with inputs of DOC+POC+CO_2+CH_4 summing to 1.5 and 1.6 g C m^{-2} yr^{-1} for Sedge Fen and Baker's Fen respectively.

3.4. CO_2 eddy-covariance fluxes

The cumulative CO_2 budget for Sedge Fen indicates that the fen is functioning as a sink, although the uncertainty range falls above zero for the merged 2013-2014 year. For the annual period from July 2013 to June 2014 NEE was -55 ± 112 g C m^{-2} yr^{-1}, whilst for 2015 it was -183 ± 98 g C m^{-2} yr^{-1} (Fig. 4), giving a mean of -119 g g C m^{-2} yr^{-1}. Water table and meteorological data suggest that the drought-induced drawdown in 2013 was particularly extreme, and data from 2007-2015 show that severe water table drawdown occurred three times during this period. Therefore, if the data were weighted assuming that the 2013-14 value was representative of one year in three, and that the flux measured in 2015 was representative of two years in three, the estimated mean annual NEE would be -140 g C m^{-2} yr^{-1}. Baker's Fen was a consistent source of CO_2, with NEE values of 157 ± 111, 83 ± 107 and 130 ± 91 g C m^{-2} yr^{-1} for 2013, 2014 and 2015 respectively; giving a mean (and SD) of 123 ± 37 g C m^{-2} yr^{-1} (Fig. 4).

3.5. CO_2 and CH_4 static chamber fluxes

CH_4 fluxes were small at Baker's Fen; *Juncus* collars emitted a net mean of 0.25 g C m^{-2} yr^{-1}, whilst *Agrostis* collars were a mean net sink of -0.22 g C m^{-2} yr^{-1} (Fig. 5). Assuming an equal mix of communities across the site as a whole would result in an approximate value of zero for net CH_4 flux. Ditches emitted CO_2 (Fig. 6) with an annual flux of 1245 g C m^{-2} yr^{-1}. Adjusting this value to the total ditch area of Baker's Fen gave an emission of 21.6 g C m^{-2} yr^{-1} for the entire fen. Ditch CH_4 emissions were generally small (Fig. 6), although a large pulse of CH_4 was measured in summer 2013. The estimated annual mean CH_4 emission was 8.9 g C m^{-2} yr^{-1}, and when adjusted to the total ditch area gave a value of 0.15 g C m^{-2} yr^{-1}, making Baker's Fen a small net source of CH_4.

CH_4 fluxes at Sedge Fen were close to zero in 2013, coinciding with severe water table drawdown, but large emissions were observed in 2014 and 2015 (Fig. 5). Overall estimated mean annual CH_4 emissions were 11.9 g C m^{-2} yr^{-1} for *Phragmites*, and 5.6 g C m^{-2} yr^{-1} for *Cladium*. Assuming an equal mix of communities across the site gives a mean CH_4 emission of 8.75 g C m^{-2} yr^{-1}. Ditch emissions of CH_4 and CO_2 were larger at Sedge Fen, with the highest fluxes occurring during spring and summer (Fig. 6). Annual ditch CO_2 flux was 2610 g C m^{-2} yr^{-1}. Adjusting this value to the total ditch area of the site gives an emission of 36.6 g C m^{-2} yr^{-1} for the entire fen. The estimated annual mean CH_4 emission was 125 g C m^{-2} yr^{-1}, and when adjusted to the total ditch area gives was 1.76 g C m^{-2} yr^{-1}.

3.6. Annual carbon balances

From the above results, we calculated the annual C balances for both sites, using the equation:

$$NECB = NEE + CH_{4\ ditch} + CH_{4\ terrestrial} + DOC + POC + CH_{4\ diss} + CO_{2\ diss} \qquad \text{Equation 2}$$

where NECB is the net ecosystem carbon balance, NEE is net ecosystem exchange measured by flux tower (and therefore includes ditch CO_2 fluxes), $CH_{4\ ditch}$ is CH_4 emission from the ditches measured by static chamber, DOC and POC are the respective net fluvial fluxes, and $CH_{4\ diss}$ and $CO_{2\ diss}$ are the respective net lateral fluxes of dissolved GHGs. The calculated NECBs show that Sedge Fen is a C sink, whilst Baker's Fen is a C source (Table 3).

4. Discussion

4.1. Soil properties

The management histories of the two sites are reflected in the soil properties. Past use as cropland has resulted in extensive subsidence at Baker's Fen; peat depth and C content are both low, and bulk density and mineral content are very high. Peat depth at Sedge Fen reaches almost 4 m, whilst on Baker's Fen it is under 0.5 m, suggesting that over 3 m of peat has been lost due to conversion to cropland (Table 1). Estimated subsidence rates (based on NECB) at Baker's Fen are 0.06 cm yr^{-1}, compared to 0.44 and 0.62 cm yr^{-1} at nearby arable sites on shallow and deep peat respectively (Evans *et al*., 2016a). It therefore appears that rewetting has reduced subsidence rates. Nevertheless, considering the present soil conditions, it may no longer be appropriate to consider the soil a peat, although it remains organic-rich and conforms to both the definition of a 'wasted peat' (Natural England, 2010), and of 'organic soil' (IPCC, 2006). Similar soils have been shown to retain many of the biochemical functions of deeper peats, including ongoing CO_2 emissions when exposed to drainage (Tiemeyer *et al*., 2016). It has been calculated that, assuming current loss rates, all peat will be lost from Baker's Fen in 400 years (Evans *et al*., 2016a). In contrast, Sedge Fen has very deep peat and remains a relatively large store of C.

4.2. Hydrology

Hydrological monitoring clearly demonstrates the challenges of keeping both sites wet (Fig. 1). The sites are small fragments of non-arable land in an otherwise agricultural region, and are hydrologically disconnected from the surrounding rivers and from groundwater. Furthermore, both sites have been historically modified to varying degrees. Water is transferred from the adjacent river to irrigate the fens and, although the period of fen irrigation is limited due to regional demands for water to irrigate crops, the transferred amount is an important component of the water balance at each site. When water inputs cease during summer, both sites dry out, as evapotranspiration exceeds precipitation. The severe and prolonged water table drawdown that occurred at Sedge Fen in 2013 is unlike the hydrological dynamics of intact fen systems with natural hydrological function, where the water table typically resides close to the surface year round (e.g. Chimner and Cooper, 2003). Before drainage, the wider fenland region would have been a wetland mix of floodplain fen and open water, with numerous dendritic river channels (Malone, no date). Drawdown at Baker's Fen was also severe and prolonged, and occurred in all years; every summer the water table fell below the level of the loggers, indicating that the entire soil profile was aerated. Once water is transferred onto site in autumn, rewetting occurs within days. This 'bimodal' pattern of seasonal water table variation is unlikely to be conducive to the full restoration of wetland vegetation species, a conclusion supported by Stroh *et al*. (2013). They suggested that, even if suitable hydrological conditions and a propagule source were established, the site would still not be able to support a species-rich wetland flora. However, if seasonal restrictions on fen irrigation were removed it might be feasible to keep the fen inundated throughout summer, with the possibility that *Phragmites* and *Typha* might colonise the site, perhaps initially as floating rafts (Money *et al*., 2009).

4.3. Water chemistry and fluvial carbon losses

At Sedge Fen DOC concentrations displayed small fluctuations, though with no clear seasonal pattern (Fig. SI3). Concentrations were lowest after the dry summer of 2013. The ditch water levels at Sedge Fen were relatively stable, and the low concentrations in summer 2013 are likely to be due to reduced production/mobility of DOC in the peat (Clark *et al*. 2005), or increased DOC degradation in the ditches (Moody *et al*., 2013). In contrast, concentrations at Baker's Fen displayed pronounced seasonal fluctuations, with peaks in spring/summer, and troughs in autumn/winter (Fig. SI3). Considering that the troughs coincided with autumn addition of low-DOC river water into the ditches it seems likely that the primary control on DOC concentrations is evapo-concentration in summer, followed by dilution in winter (Waiser, 2006). At Baker's Fen ditch water levels became very low in summer (some ditches dried out completely), and a negative correlation between ditch depth and DOC concentration was found (Fig. SI4). The influence of transferring river water onto Baker's Fen is also evident in the increases in ditch nitrate concentration that were observed in November 2013 and 2014 (Fig. SI6). There were fluxes of DOC and POC (and dissolved GHGs) onto both fens during the addition of river water. These represented 27% and 16% of total fluvial C losses at Sedge Fen and Baker's Fen respectively. The lower fluvial inputs, when compared to fluvial losses, are due to the low DOC and POC concentrations in the river (Table 2, Fig. SI3, SI5), and the relatively small contribution of inputs of river water to the hydrological budgets.

DIC concentrations were high for both sites (Table 2, Fig. SI2). Since DIC in fen runoff is generally derived from weathering of carbonate or siliceous minerals, rather than peat, this flux cannot strictly be considered part of the peatland C balance (Evans *et al*., 2016b). Additionally, since DIC in the drainage network will remain in a dissociated form due to the high pH of the water, little of this flux can be expected to be evaded as CO_2, or therefore to contribute to overall GHG emissions from the fen.

At Sedge Fen exports of aquatic C generally occurred on a restricted seasonal basis, due to the limited water discharge from the fen, but were more frequent at Baker's Fen where water discharge was greater (Fig. 2, 3). DOC was the principal component of aquatic fluxes, and was responsible for ~73% of fluvial C exported. However, our estimated DOC exports (4.1 and 8.8 g C m^{-2} yr^{-1} from Sedge Fen and Baker's Fen respectively) are close to fluxes reported from some temperate and boreal fens in Scandinavia and Canada (5 g C m^{-2} yr^{-1}, Strack *et al*., 2008; 3.7 g C m^{-2} yr^{-1}, Juutinen *et al*., 2013) and German drained peat grasslands (5.2 g C m^{-2} yr^{-1}, Tiemeyer and Kahle, 2014), and are low compared to values from UK raised bogs (25 g C m^{-2} yr^{-1}, Dinsmore *et al*., 2010) and blanket bogs (33 g C m^{-2} yr^{-1}, Worrall *et al*., 2003). Other semi-natural and agricultural fens in the same region (East Anglia, UK), measured concurrently with our study, also had low DOC fluxes (4.1 – 7.9 g C m^{-2} yr^{-1}), whilst semi-natural fens and peat grasslands in wetter parts of the UK had higher fluxes (Somerset; 10-22 g C m^{-2} yr^{-1}, Anglesey; 18-31 g C m^{-2} yr^{-1}) (Evans *et al*., 2016a). This reflects the fact that hydrological regime has a strong control on such small and isolated fens; that water losses, and therefore fluvial C losses, will be greater in systems where the difference between precipitation and evapotranspiration is larger. POC fluxes were small, comprising 6.8% (0.4 g C m^{-2} yr^{-1}) and 9.2% (1.1 g C m^{-2} yr^{-1}) of fluvial C flux at Sedge Fen and Baker's Fen, respectively. In upland peatlands, POC export can sometimes equal that of DOC, particularly if erosional features are present (Pawson *et* al., 2012). The low rainfall levels, lack of overland flow and consequently low rates of fluvial erosion in most lowland peatlands reduce the importance of POC to the fluvial C budget. Most estimates of POC have been for upland blanket bogs, but Olefeldt and Roulet (2012) reported fluxes of 1.1 and 3.6 g C m^{-2} yr^{-1} for fen outflows in a subarctic peatland complex in Sweden.

For dissolved GHGs, we assumed that lateral (dissolved) fluxes were separate from vertical (gaseous) fluxes. When water outputs from the fens occur, dissolved GHGs will be exported out of the system into rivers, and may be emitted off-site. Dissolved CO_2 was ~18% of total fluvial C export, larger than the POC flux, whilst CH_4 (which has a low solubility in water) made a negligible contribution (≤ 0.1%). Our dissolved CO_2 fluxes of 1.2 g C m^{-2} yr^{-1} and 1.8 g C m^{-2} yr^{-1} are similar to that reported from UK a raised bog (1.3 g C m^{-2} yr^{-1}, Dinsmore *et al*., 2010) but smaller than those from blanket bogs (3.8 g C m^{-2} yr^{-1}, Worrall *et al*., 2003) and drained Irish grasslands (2.4-4.4 g C m^{-2} yr^{-1}, Barry *et al*., 2014). It is likely that this is because slow water movement in ditches results in the majority of aquatic CO_2 being lost on-site as gaseous fluxes, rather than exported off-site fluvially. It should be noted that many C-balance studies do not measure dissolved GHGs and POC, and instead focus solely on DOC (e.g. Roulet *et al*., 2007). The total aquatic C fluxes for our sites were 5.72 and 11.73 g C m^{-2} yr^{-1} for, with the total losses of POC + dissolved GHGs being 1.62 and 2.90 g C m^{-2} yr^{-1} (Table 3). Therefore, if we had neglected to measure POC and dissolved GHGs, 25-28% of fluvial C exports would be missing from the total budget.

4.4. CO_2 and CH_4 fluxes

Eddy covariance flux tower measurements showed that Sedge Fen was a large CO_2 sink (Fig. 4). Although significant periods of water table drawdown occur at Sedge Fen, the fen is also seasonally inundated with standing water, and the plant species present have the potential to form peat under waterlogged conditions. NEE was in the same range as reported values from northern bogs and fens (Yu, 2012). In contrast, flux tower measurements for Baker's Fen suggest that the site was a net source of CO_2 (Fig. 4). Systematic reviews have shown that drained peatlands have higher rates of ecosystem respiration (Haddaway *et al*., 2014), and Baker's Fen had suffered serious soil loss and compaction before the restoration activity was conducted, and still experiences consistent and pronounced water table drawdown in summer (Fig. 1). It is therefore unsurprising that the site is a source of CO_2. Grasslands on drained organic soils can act as net CO_2 sinks (e.g. Renou-Wilson *et al*., 2014), but it seems probable that a higher water table would need to be instated for this to occur at Baker's Fen (Wilson *et al*., 2016).

Net CH_4 fluxes were approximately zero at Baker's Fen, with areas of *Agrostis* acting as small sinks for the majority of the time (Fig. 5). Areas of *Juncus* were often small sinks of CH_4, but emissions were occasionally observed, with the overall effect being that *Juncus* patches were net sources. Low emissions from organic grasslands would be expected due to the low water table and organic matter content (Tiemeyer *et al*., 2016), and emissions will be further mitigated by low CH_4 diffusion due to drainage-induced increases in soil bulk density (Nykänen *et al*., 1998). The observed emissions could be due to CH_4 transport through aerenchymatous tissue in *Juncus* plants (Henneberg *et al*., 2012); *Juncus* clumps have sometimes been observed to act as point-source emissions of CH_4 in drained peatlands (Henneberg *et al*., 2015). Equally, the presence of *Juncus* may simply indicate that these collars were situated in wetter areas of the site where CH_4 emissions were more likely to occur. At Sedge Fen, CH_4 emissions were low in 2013 when the largest water table drawdown occurred, but much larger in 2014 and 2015 (Fig. 5), as would be expected from a site with deep peat, wetland vegetation, and seasonal inundation.

Ditch emissions of CH_4 were low from Baker's Fen (Fig. 6). However, as the terrestrial component of the fen was CH_4 neutral, the ditches resulted in the fen acting as a small net source of CH_4. The annual flux (per unit ditch water surface) of 8.9 g C m^{-2} yr^{-1} is low compared to other reports from grasslands, which span 40-75 g C m^{-2} yr^{-1} (Evans *et al*., 2016b). Whilst our estimate is based on just two floating chambers, a more spatially intensive campaign in 2015 (replicated seasonally) produced a similar estimate for the site of 13.7 g C m^{-2} yr^{-1} (Peacock *et al*., 2017). The relatively low ditch flux is explicable if ditch CH_4 fluxes are driven by inputs from the soil, as Rasilo *et al*. (2017) found for small boreal streams. The extreme peat oxidation, low organic content of the soil, and low water tables at Baker's Fen are unlikely to favour methanogenesis in the soil, as well as resulting in a large zone where methanotrophy can occur (Yavitt *et al*., 1997). In contrast to this, emissions were substantial at Sedge Fen, at 125 g C m^{-2} yr^{-1}, making them equivalent to 20% of terrestrial CH_4 fluxes. Although CH_4 is only a minor component of the NECB, CH_4 fluxes from terrestrial vegetation and ditches are important from a climate perspective due to the higher global warming potential of CH_4 (IPCC, 2006).

Annual ditch fluxes of CO_2 were larger at Sedge Fen: 2610 g C m^{-2} yr^{-1} compared to 1245 g C m^{-2} yr^{-1} at Baker's Fen (Fig. 6). Although we are unsure of why fluxes from Baker's Fen are lower, it could be that the low organic content of the soil, alongside other changes in soil properties (Table 1), resulted in reduced respiration of organic matter and therefore lower emissions. Alternatively, it could be an artefact of the low number of spatial replicates at each site. Whilst some have found that ditches do not contribute any significant amount to net CO_2 emissions in cutaway peatlands (Sundh *et al*., 2000) or peatland grasslands (Best and Jacobs, 1997), others have reported large fluxes from ditches in peatland grassland and reedbeds and from agricultural ditches (Schrier-Uijl *et al*., 2011). Our relatively high measured CO_2 fluxes are potentially important, especially at Sedge Fen; when weighted by ditch area the annual flux is 36.6 g C m^{-2} yr^{-1}, which has the effect of somewhat reducing the net CO_2 uptake of the fen. However, this calculation may be an artefact of having only two floating chamber locations. A spatially intensive campaign repeated four times in 2015 gave an annual flux of 413 g C m^{-2} yr^{-1} (Peacock *et al*., 2017) which is more in keeping with literature values. Using this number would give an area-weighted flux of 5.8 g C m^{-2} yr^{-1}; i.e., offsetting considerably less of terrestrial CO_2 uptake.

4.5. Annual carbon balance and implications for rewetting

Despite being subjected to occasional, extreme water table drawdown events, Sedge Fen remains a considerable overall C sink of -104 g C m^{-2} yr^{-1} (Table 3). This is relatively high when compared to published measurements from UK bogs (e.g. -28 g C m^{-2} yr^{-1}, Helfter *et al*., 2015; -15.4 g C m^{-2} yr^{-1}, Worrall *et al*., 2003; -56 g C m^{-2} yr^{-1}, Worrall *et al*., 2009), acidic Scandinavian peatlands (e.g. -20 to -56 g C m^{-2} yr^{-1}, Nilsson *et al*., 2008, Olefeldt *et al*., 2012) and Canadian bogs (-89 to +13.5 g C m^{-2} yr^{-1}, Roulet *et al*., 2007). Instead, it is similar to the value of -102 g C m^{-2} yr^{-1} from a semi-natural *Cladium* and *Phragmites* fen, but lower than -281 g C m^{-2} yr^{-1} from a nutrient-rich *Phragmites* fen (both in the UK) (Evans *et al*., 2016a). It seems probable that these large values are due to the high productivity of tall fen vegetation (Wheeler and Shaw, 1991), which in turn is due to the favourable climatic and chemical conditions for growth in lowland fens when compared to upland/northern peatlands. When combined with favourable hydrological conditions there is, therefore, a greater potential for relatively rapid C accumulation. However, it is important to consider that peatlands can switch dramatically from C sources to sinks (Roulet *et al*., 2007), and a longer period of monitoring would be needed to see whether this is the case at Sedge Fen.

In contrast, the substantial net C loss from Bakers Fen (NECB 133 g C m^{-2} yr^{-1}, Table 3) suggests that peat loss is continuing at this site despite the restoration measures undertaken. Beetz *et al*. (2013) reported NECBs of -147 and 88 g C m^{-2} yr^{-1} for a rewetted peat grassland over two years, and suggested that the difference was due to a mowing event in October of the second year. However, the water table was higher at their site compared to Baker's Fen, with a mean depth of approximately 25 cm. The absence of a return of wetland vegetation and C sink at Baker's Fen, even after ~20 years of restoration is perhaps not surprising. Moreno-Mateos *et al*. (2012) showed in their meta-analysis of 621 global wetland sites that C storage and accumulation of soil organic matter remain lower in restored sites compared to reference sites, even on 50-100 year time scales. They hypothesised that restored wetlands may shift to stable states that differ from their original condition.

It therefore seems likely that Baker's Fen will not begin to sequester more C than it loses unless management is changed. The most effective option would be to transfer more water onto the fen throughout the year, but this would be at the expense of agricultural water needs in the region. The site will continue to behave like a seasonally-inundated wetland without a year-round higher water table. Tiemeyer *et al*. (2016) suggest that mean water-table depth needs to be less than 20 cm to constrain CO_2 losses due to decomposition, but at Bakers Fen it was 46 cm in 2014 and 55 cm in 2015. Other research suggests that because the soil properties have been altered to such a degree the reestablishment of original wetland vegetation would remain difficult (Stroh *et al*., 2013). However, as noted in section 4.2, it might be that prolonged inundation could lead to the development of floating rafts of wetland plant species (Money *et al*., 2009). Nevertheless, it is worth noting that the NECB of other croplands in the region is 693 and 773 g C m^{-2} yr^{-1} (Evans *et al*., 2016a). Rewetting has therefore potentially suppressed C losses to ~20% of their former value. Similarly, research from Finland has shown that CO_2 fluxes from abandoned agricultural peatlands is considerably less than fluxes from arable peatlands (Maljanen *et al*., 2007). As well as reducing C losses, the rewetting of Baker's Fen has provided a buffer zone to Sedge Fen, increased biodiversity, and provided a recreational environment for visitors (Peh *et al*., 2014).

4.6. Concluding remarks

Global GHG emissions caused by draining peatlands for cropland are 630 Tg CO_2e yr^{-1} (Carlson *et al*., 2017), and there is continued interest in peatland restoration as a potential climate mitigation measure (Griscom *et al*., 2017). The rewetting of peat-based croplands offers a viable way to substantially reduce GHG and C losses, with the emission reductions from rewetted grassland and cropland being in the region of 20 t CO_2e ha^{-1} yr^{-1} (Bonn *et al*., 2014). If C losses are simply slowed, rather than being reversed, the entire volume of peat will still eventually be lost to the atmosphere; however, this nevertheless represents a reduction in GHG emissions (equivalent to reducing fossil fuel combustion) in the medium term. Considering the national and global importance of drained organic soils for food production, there are significant socio-economic barriers to the re-wetting of cultivated peatlands, including issues relating to national food security and risks of 'leakage' if GHG emissions associated with food production are simply transferred from one location or form to another. Paludiculture (high water table agriculture supporting both economic returns and peat formation) has been suggested as an optimal future use for currently drained peatlands (Wichtmann and Joosten, 2007), but remains both technically and economically challenging to implement at the large scale. In the short to medium term, therefore, it is likely that measures to reduce drainage-related GHG emissions from peatland remaining under cultivation (so called "responsible peatland management"; Wijedasa *et al*. (2016)), including transitions from deep-drained to shallow-drained cropland or grasslands, may provide the most effective means of reducing GHG emissions from these regions.

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Tables

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Table 1. Soil properties for the two fens. Bulk density, pH, mineral %, carbon % and C/N are means for the top 50 cm of peat at Sedge Fen, and for the entire 40 cm at Baker's Fen. Full profile C stock estimates are based on measured %C and bulk density values to the maximum coring depth.

	Peat depth (cm)	Bulk density (g cm^{-1})	pH	Mineral (%)	C (%)	C/N	Full profile C stock (t C ha^{-1})
Sedge Fen	380	0.37	7.54	52.2	32.0	15.8	2820
Baker's Fen	40	1.06	7.10	65.7	22.3	19.7	610

Table 2. Water chemistry determinands for the two fens and river, presented as means and standard errors (in parentheses). POC concentrations are reported as medians with interquartile range (in parentheses) due to the abnormally high values in the dry summer of 2013 that would skew the mean (see Fig. SI5).

	pH	EC (µS cm^{-1})	DOC (mg l^{-1})	POC (mg l^{-1})
Baker's Fen	7.53 (0.02)	1426 (32)	34.5 (1.6)	1.84 (6.21)
Sedge Fen	7.63 (0.02)	901 (15)	22.8 (0.8)	0.62 (1.14)
River	7.79 (0.02)	790 (11)	5.3 (0.2)	0.29 (0.48)

	DIC (mg l^{-1})	NO_3^- (mg l^{-1})	C-CO_2 (mg l^{-1})	C-CH_4 (mg l^{-1})
Baker's Fen	88 (2.1)	11.6 (1.1)	2.82 (0.17)	0.086 (0.022)
Sedge Fen	80.7 (1.3)	15.1 (1.1)	2.32 (0.23)	0.115 (0.041)
River	60.5 (1.5)	40.2 (1.8)	0.97 (0.03)	0.017 (0.002)

Table 3. Net ecosystem carbon budget for each site. DIC was not included (see section 4.3). Note that ditch CO_2 flux (*) is included for information, but is not included in the total NECB as this flux is also measured by the flux tower.

Flux (g C m^{-2} y^{-1})	Wicken Fen	Baker's Fen
Gaseous C		
NEE	-119	123
Ditch CO_2	36.6*	21.6*
Ditch CH_4	1.8	0.15
Terrestrial CH_4	8.8	0
Aquatic C losses		
DOC	4.1	8.83
POC	0.39	1.08
Dissolved CO_2	1.23	1.81
Dissolved CH_4	0.003	0.013
Aquatic C inputs		
DOC	-1.25	-1.6
POC	-0.09	-0.09
Dissolved CO_2	-0.2	-0.23
Dissolved CH_4	-0.003	-0.003
NECB	-104.2	133.0

Figure
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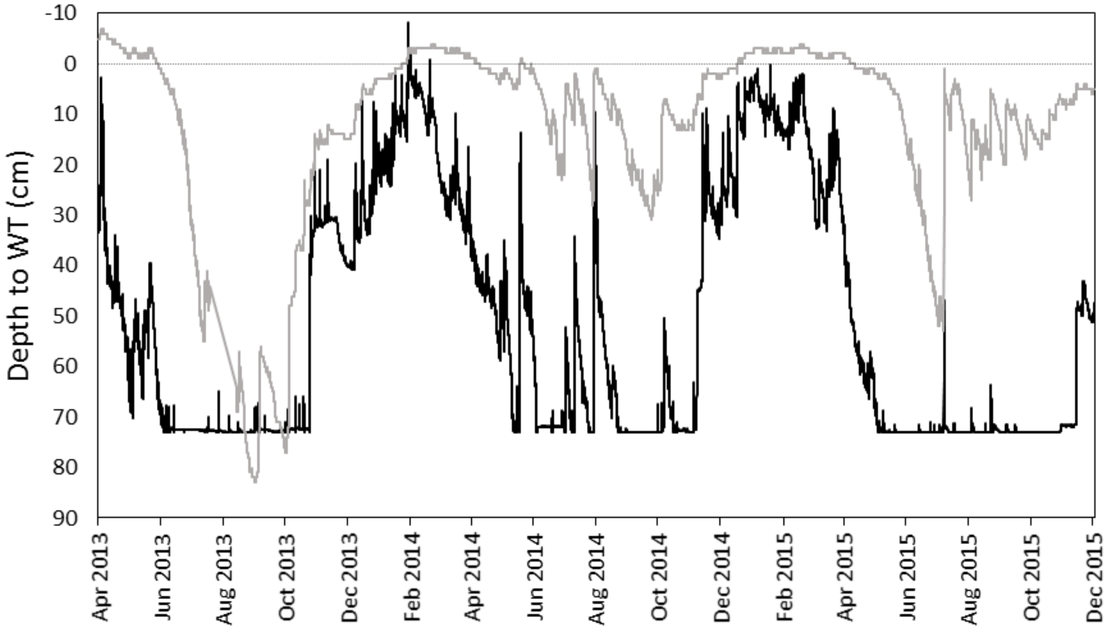

Figure

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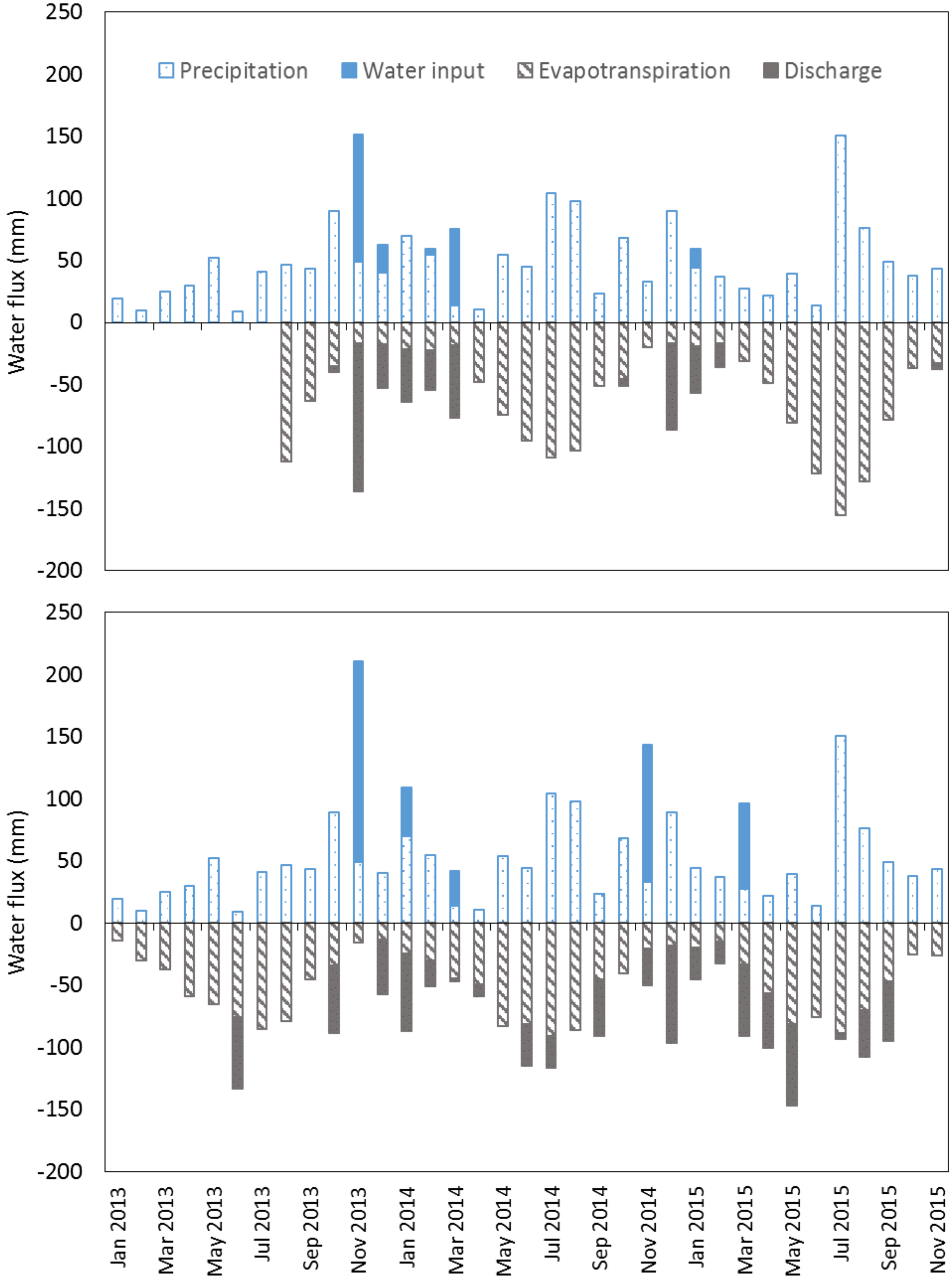

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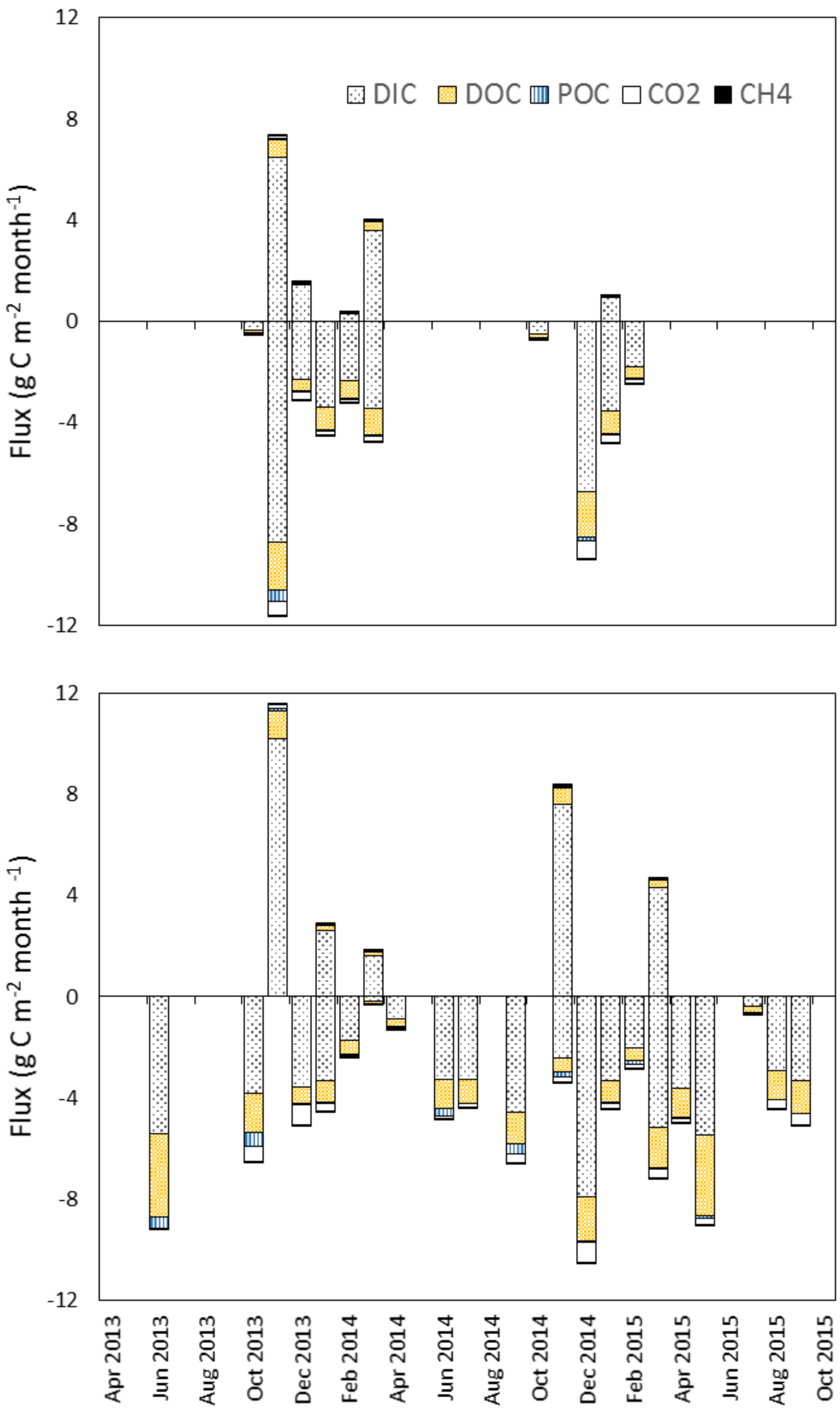

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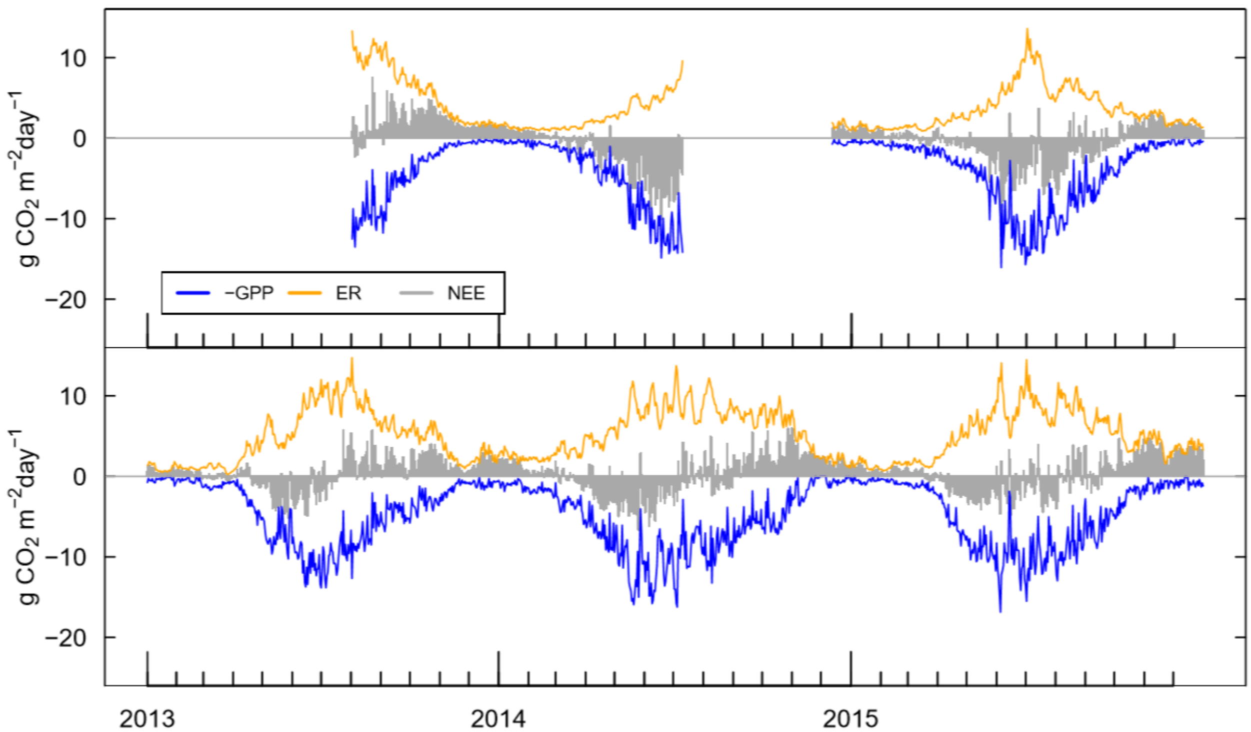

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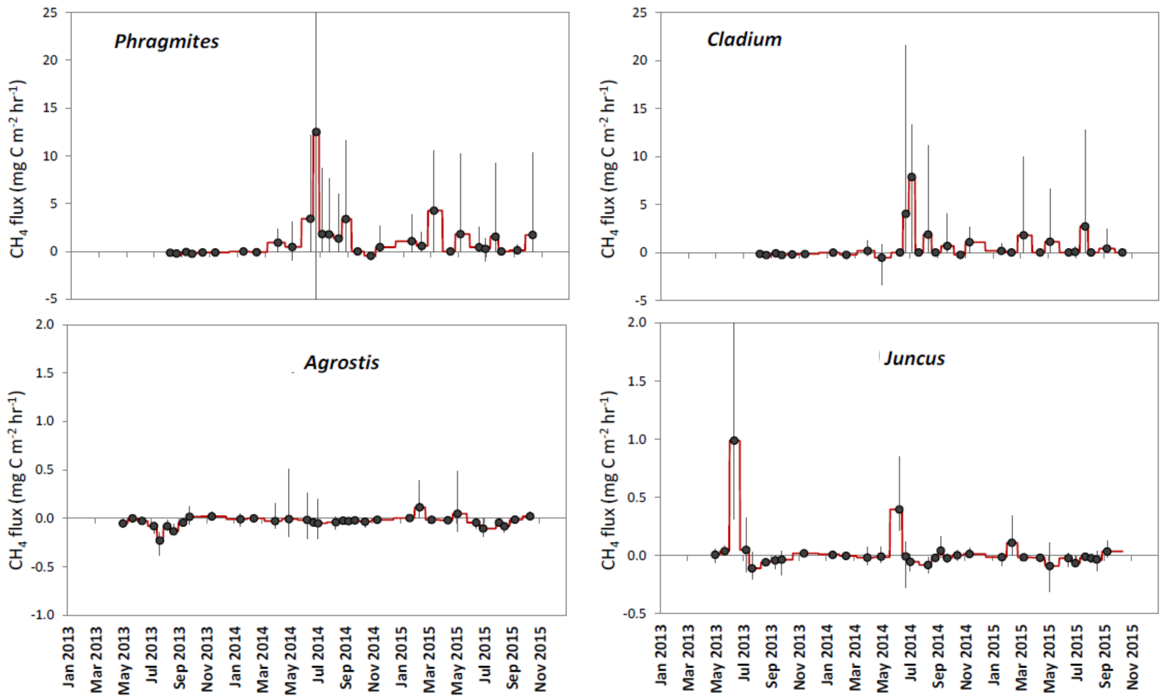

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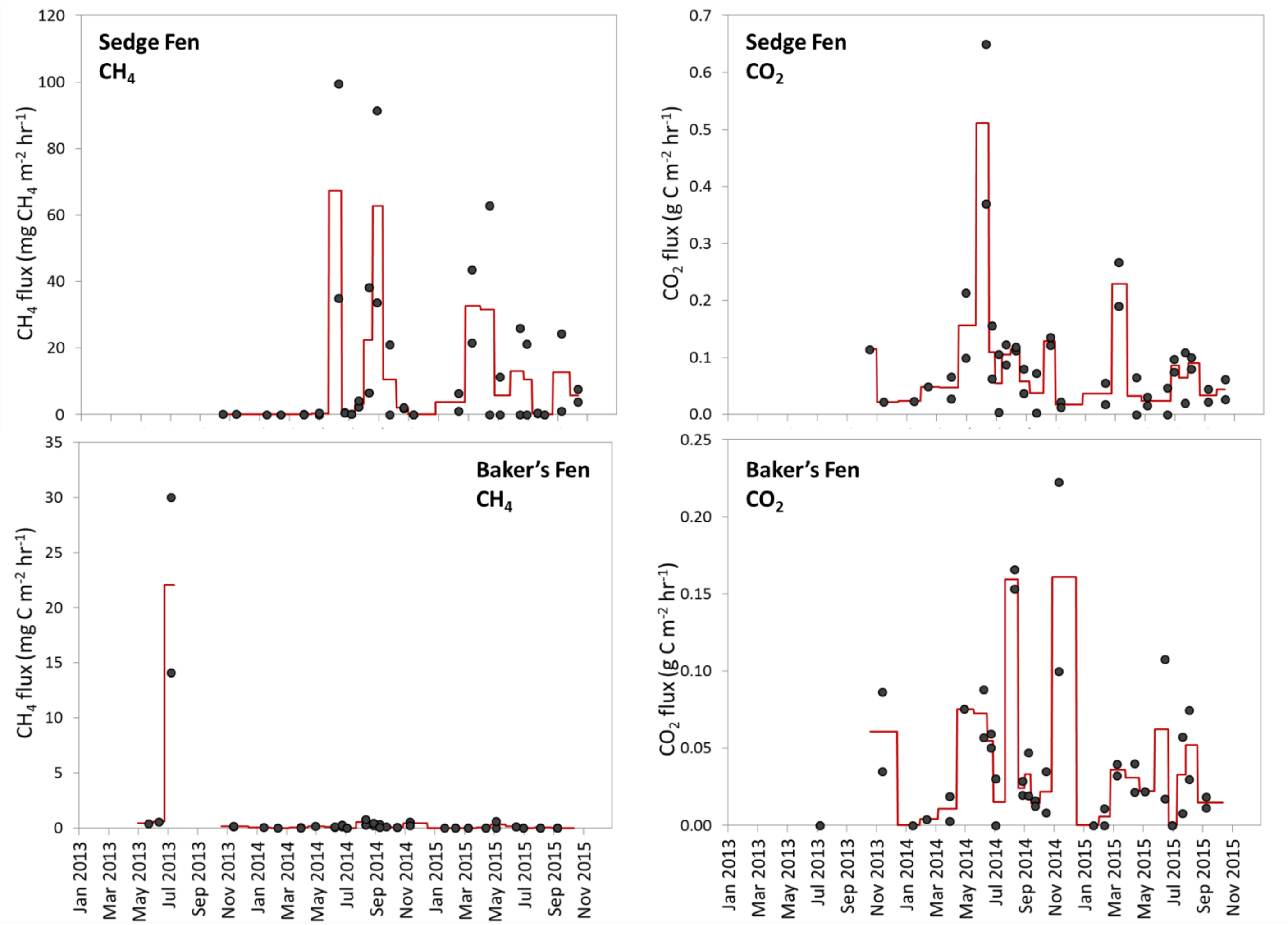

Figure 1. Water tables for Sedge Fen (grey) and Baker's Fen (black). Negative values indicate water levels above the peat surface (horizontal line at 0 cm), i.e. flooding. For Baker's Fen, the logger in the dipwell was located at approximately 73 cm depth which was the lowest point in the soil profile; i.e. if this WT depth was reached the dipwell was dry.

Figure 2. Monthly hydrological budgets for Sedge Fen (top) and Baker's Fen (bottom). Note that evapotranspiration and discharge were not determined Jan-July 2013 for Sedge Fen, and abstraction data were not available for November 2015.

Figure 3. Monthly aquatic carbon fluxes for Sedge Fen (top) and Baker's Fen (bottom). Positive numbers are fluxes into the fens, occurring when river water is transferred onto site. Negative numbers are discharge leaving the fens. All zero values indicate no flux.

Figure 4. Daily eddy covariance CO_2 budgets for Sedge Fen (top) and Baker's Fen (bottom), showing gap-filled NEE, GPP and ER.

Figure 5. Measured CH_4 fluxes for Sedge Fen (top: *Phragmites*- and *Cladium*-dominated communities) and for Baker's Fen (bottom: *Agrostis*- and *Juncus*-dominated vegetation communities). Points show mean observations on each measurement date, and error bars show range of measured values on that date. Red lines show estimated fluxes. Note the difference in *y* axes scales between the sites.

Figure 6. CH_4 and CO_2 fluxes measured in ditches at Sedge Fen (top) and Baker's Fen (bottom). Observations are represented by circles, red line shows estimated fluxes. Data were not collected at Baker's Fen during late summer 2013 as ditches dried out at this time. Note varying scales on the *y* axes.