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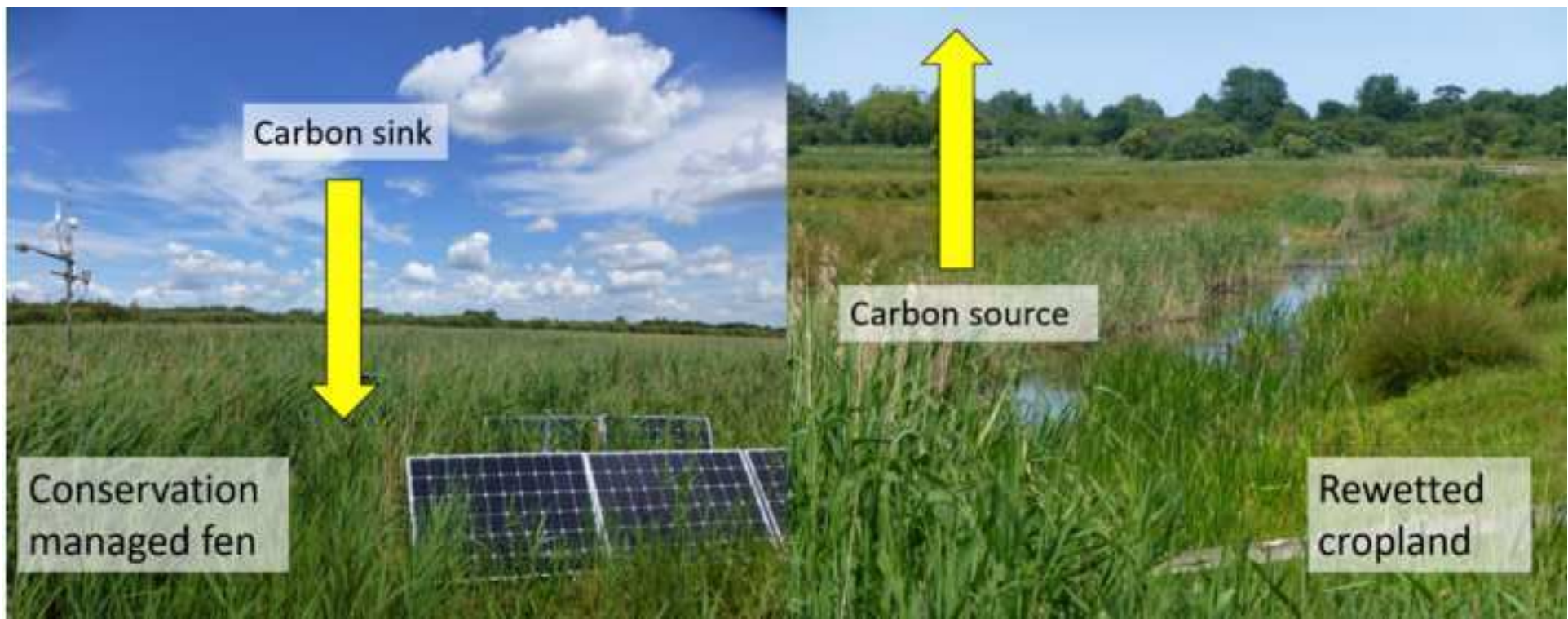
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Highlights (for review)

- 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 $-104 \text{ g C m}^{-2} \text{ yr}^{-1}$. In contrast, the grassland was a C source of $133 \text{ g C m}^{-2} \text{ yr}^{-1}$, with gaseous carbon dioxide (CO₂) emissions being the main loss pathway, due to low water tables exposing the soil profile in summer. At each site, ditch

32 emissions of CO₂ were moderately large (22 and 37 g C m⁻² yr⁻¹), whilst ditch methane (CH₄)
33 emissions (0.2 and 1.8 g C m⁻² yr⁻¹) made a negligible contribution to the NECB, but are important
34 when considering the ecosystem GHG balance in terms of CO₂ equivalents. Excluding dissolved
35 inorganic carbon (DIC), fluvial C losses were 6 g C m⁻² yr⁻¹ for the conservation-managed fen and 12 g
36 C m⁻² yr⁻¹ for the former cropland, and were dominated by dissolved organic carbon (DOC). The
37 small fluvial C loss is the result of both sites being hydrologically isolated from the surrounding
38 agricultural landscapes. Although the partially re-wetted cropland was still acting as a net C source,
39 our estimates suggest that seasonal rewetting has reduced net annual C losses to ~20% of their
40 former cropland values. Maintaining high water tables year round would potentially further reduce
41 C losses, and shallow inundation might allow the return of wetland species such as *Phragmites* and
42 *Typha*, perhaps as floating rafts.

43

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

46

47 “*The height of a man in the life of a man.*” – old East Anglian saying describing peat losses due to
48 subsidence.

49

50 **1. Introduction**

51 Globally, approximately 300,000 km² (~7 %) of peatlands are used for agriculture
52 (International Peat Society, 2008), including extensive areas of lowland peat that have been drained
53 and converted to intensive arable use, and are now important areas for food production (Joosten
54 and Clarke, 2002). Drainage generally disrupts the natural functioning of the peatland carbon (C)
55 store, leading to increased emissions of nitrous oxide (N₂O) (Haddaway *et al.*, 2014) and carbon
56 dioxide (CO₂), as extensive peat losses occur due to this oxidation (Hooijer *et al.*, 2012). The most
57 recent report of the Intergovernmental Panel on Climate Change (IPCC) emphasises the importance

58 of CO₂ emissions from oxidation of cultivated peatlands (primarily in Europe and Southeast Asia)
59 (Smith *et al.*, 2014), and it has been estimated that greenhouse gas (GHG) emissions from drained
60 and burned peatlands account for 5% of anthropogenic emissions (Global Peatlands Initiative, 2017).

61 As concern for peatland C stocks has grown, there has been an increased emphasis on
62 restoring and rewetting bogs and fens that have been disturbed by human activities (e.g. Wilson *et*
63 *al.*, 2016), and the potential global importance of such work on GHG emissions was recognised in the
64 development of a reporting methodology for wetland drainage and rewetting in the IPCC Wetland
65 Supplement (IPCC, 2014). The Paris Climate Agreement commits nations to limiting climatic warming
66 to less than 2°C (Rogelj *et al.*, 2016). This commitment will require zero net CO₂ emissions by 2050
67 (Matthews and Caldeira, 2008) which, because all realistic future scenarios involve some level of
68 continued fossil fuel emission, will necessitate the development of compensating measures which
69 remove CO₂ from the atmosphere; i.e. negative emissions. The recent “4 per 1000” initiative
70 proposes that a significant fraction of this target could be achieved through enhanced sequestration
71 of C into soils (Minasny *et al.*, 2017). Wetlands, in particular, have been highlighted as being key in
72 delivering “natural climate solutions” due to their potential to accumulate and retain C (Griscom *et*
73 *al.*, 2017). For lowland agricultural fens, it has been shown that restoration can reduce oxidation-
74 induced losses of peat and, in some cases, lead to the re-establishment of their function as a C sink
75 (Knox *et al.*, 2014).

76 However, it can be difficult to restore agricultural land back to a properly functioning fen
77 ecosystem (Stroh *et al.*, 2013). Reasons for such difficulty include extensive peat loss through
78 oxidation and the compaction of remaining peat, loss of local seed banks, heavily modified drainage
79 systems and previous addition of silt to the peatland via warping (the agricultural practice of
80 diverting mineral-rich river water onto peat soils to deposit sediment) (Smart *et al.*, 1986). Where
81 ‘complete’ restoration is impossible, it may nevertheless be feasible to convert agricultural land to
82 semi-natural fen meadows, which will still bring associated increases in biodiversity and ecosystem
83 services (Klimkowska *et al.*, 2010), and may also reduce rates of C loss (Hendriks *et al.*, 2007). In

84 much of Europe, including parts of UK and the Netherlands, the target ecosystem for fen restoration
85 is a semi-natural environment involving ongoing water-level and vegetation management (Klötzli
86 and Grootjans, 2001), for example to maintain or enhance plant species richness (Menichino *et al.*,
87 2016). However, sometimes it may be that constraints in water availability result in unexpected
88 vegetation shifts, often in undesired directions, which may limit the success of restoration attempts
89 (Klötzli and Grootjans, 2001).

90 Knowledge gaps still remain on the effects of agricultural fen restoration on C and nutrient
91 cycling, and on how the functioning of these restored ecosystems compares to conservation-
92 managed fens that have never been under agricultural production. For instance, Tiemeyer *et al.*
93 (2016) found that CO₂ emissions increased with deeper water tables in drained peat grasslands, but
94 could not model CO₂ fluxes across multiple sites solely as a function of water table, and suggested
95 that additional factors such as drought stress could result in lower emissions (because CO₂ fluxes
96 from respiration are limited by both very dry and very wet soil conditions). Contrary to this, it has
97 sometimes been shown that drained grasslands can be CO₂ sinks, and could act as C stores
98 depending on management practices; e.g. large amounts of biomass removal could counteract a
99 terrestrial CO₂ sink and result in a C source (Renou-Wilson *et al.* 2014). However, methane (CH₄) can
100 still be emitted by drained soils (Hendriks *et al.*, 2007, Henneberg *et al.*, 2015), with implications for
101 C and GHG budgets.

102 The East Anglian fens are the largest and most intensively modified area of lowland peat in
103 the UK. In their original form they occupied approximately 150,000 ha (Burton and Hodgson, 1987),
104 but drainage and agricultural conversion has resulted in just 12,600 ha of deep peat remaining,
105 which now stores an estimated 41 Tg of C (Holman, 2009). Of this remaining peat, approximately
106 800 ha exists as undrained fen, in four separate nature reserves. The aphorism quoted above,
107 assuming a man of 170 cm living for sixty years, would result in a peat loss of 2.8 cm per year. This
108 figure falls within the range of 0.27-3.09 cm per year (mean = 1.37 cm) for the region reported by
109 Richardson and Smith (1977). As a consequence of peat subsidence, much of the land in the region

110 is now below mean sea level, and a complex series of ditches, embankments, sluices and pumps
111 control the area's hydrology. Although the region is of national significance for the production of
112 arable and horticultural crops, several projects are now underway to return areas of agricultural land
113 to wetland.

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

128

129 **2. Materials and methods**

130 *2.1. Field sites*

131 Both field sites are part of the Wicken Fen National Nature Reserve which is owned and
132 managed by the National Trust, a conservation organisation. Mean annual air temperature from an
133 automatic weather station (AWS) on site was 9.3 °C in 2013, 10.9 °C in 2014 and 10.3 °C in 2015.
134 Missing data from the AWS precludes the calculation of site-specific annual rainfall totals, but
135 rainfall was 648 mm, 765 mm, and 641 mm in 2013, 2014 and 2015 at another lowland site 27 km

136 away (Evans *et al.*, 2016a). Rainfall was measured on the study site in 2015 using a manual rain
137 gauge, with an annual total of 643 mm.

138 The conservation-managed site is referred to as the Wicken Sedge Fen (52.31° N, 0.28° E,
139 area = 61 ha, 2-3 m above sea level). It lies on a surviving area of deep peat and contains large areas
140 of reed bed that are cut on approximately a three year rotation. Sedge Fen has never been
141 agriculturally drained, and has been under active conservation management since 1899, thereby
142 making it the oldest nature reserve in the UK. The dominant plant species present are saw sedge
143 *Cladium mariscus* and common reed *Phragmites australis*, with abundant reed canary grass *Phalaris*
144 *arundinacea*, and some purple small-reed *Calamagrostis canescens* (Eades, 2016). A network of
145 ditches cross the site, which are used for water level management rather than drainage; the ditches
146 are not permanently connected to the wider river network in the region, but water may be
147 transferred onto the fen from the adjacent river (named Wicken Lode). The fen has no defined
148 outflow, i.e. it is not drained by a stream. Much of the perimeter is bunded to minimise water loss
149 to the surrounding agricultural landscape, which is at a lower elevation as a result of subsidence, and
150 it is assumed that water losses occur laterally as surface/subsurface flow. A dense ‘aquitard’ layer
151 has been identified within the peat column that reduces water movement downwards to the deep
152 groundwater (Boreham, 2017). As long ago as 1908, concerns were raised that the fen was
153 becoming drier (Yapp, 1908). However, there has been more recent concern that water tables at
154 Sedge Fen are declining compared to historical measurements (McCartney and de la Hera, 2004), so
155 in 2011 a wind pump was installed to pump mineral-rich river water onto the site. Water is pumped
156 onto site for the period November-March due to restrictions imposed by the Environment Agency (a
157 government body responsible for environmental protection), whereby water distribution is
158 prioritised for agricultural use. The catchment of the river (Wicken Lode/New River) upstream from
159 the field sites is 27.6 km² (McCartney and de la Hera, 2004) and is principally arable farmland.

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

162 19th century for agriculture, resulting in extensive peat loss and subsidence. It has been under
163 conservation management since 1994. Restoration activities have included cessation of arable
164 cultivation, enclosure of the fen in a waterproof membrane in 1994 to retain water inputs, re-
165 seeding of the fen in 1995 with an unknown “grass mixture” (Saltmarsh, 2000), and the excavation
166 of several scrapes to create seasonal standing water bodies. Like Sedge Fen, Baker’s Fen is
167 hydrologically isolated, as it is not connected to the surrounding network of rivers or to
168 groundwater. Current hydrological management involves transferring river water onto site during
169 November-March, and the site is extensively grazed by Highland cattle and Konik ponies. A network
170 of ditches cross the site, and water is released from the fen on an occasional, *ad-hoc* basis using a
171 sluice in the western corner. Bakers Fen supports a species-poor damp grassland, reflecting its past
172 management history. Dry areas are a mesotrophic grassland community dominated by *Agrostis*
173 *stolonifera* with *Arrhenatherum elatius*, *Cirsium arvense*, *Dactylis glomerata* and *Holcus lanatus*.
174 Wetter areas, such as the scrapes, are rush pasture communities, featuring hard rush (*Juncus*
175 *inflexus*) and *Agrostis stolonifera* (Eades, 2016).

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

181

182 2.2. Soil properties

183 Peat depth was measured at 24 locations at each site using a Dutch auger. The
184 measurements were taken in an area of 18 m x 18 m, centred on the flux towers (see section 2.3).
185 Two soil cores were taken at each site using a Dutch auger from the areas of deepest peat that we
186 measured within the sampled area. The cores were sectioned in the field at pre-determined
187 intervals and the samples brought back to the laboratory. The sections chosen were 5 cm increments

188 to 20 cm depth, and then 10 cm increments to the base. All sections of the cores were analysed for
189 dry bulk density, and selected samples were analysed for pH, mineral content, and C and N
190 elemental content. Samples were dried at 105°C for 16 hours and checked for no further mass loss,
191 and their bulk density measured prior to further processing. Three sets of sub-samples were then
192 taken. One set of sub-samples was analysed for pH by placing each sample in 0.01 M CaCl₂ at a mass
193 to volume ratio of 1:10. The second set of sub-samples were ashed at 550°C for 4 hours and the
194 residual mass recorded as the mineral content. The third set were used for C and N elemental
195 analysis: triplicate samples were milled to a sub-mm powder using a 6770 Freezer/Mill (Spex,
196 Metuchen, USA). The ground samples were then analysed on an ECS 4010 elemental combustion
197 system with a pneumatic autosampler (Costech, Santa Clarita, USA), using acetanilide standards. All
198 samples were corrected for their measured ash content and expressed as their molar ratio.

199

200 *2.3. CO₂ eddy-covariance fluxes*

201 Both sites were instrumented with open-path eddy covariance (EC) flux towers to measure
202 ecosystem-scale CO₂ fluxes. The instrumentation comprised a Solent R3 sonic anemometer (Gill
203 Instrument Ltd. Lymington, UK) at Sedge Fen, and a CSAT3 sonic anemometer (Campbell Scientific
204 Inc. Logan Utah, USA) at Baker's Fen for measurements of the three components of atmospheric
205 turbulence and sonic temperature. An LI7500A open path analyser (LI-COR Biosciences, Lincoln,
206 Nebraska, USA) was used to measure concentrations of atmospheric water vapour and CO₂ as well as
207 barometric pressure at both fens. At both sites, EC data were scanned at 20 Hz and logged using a
208 LI-COR LI7550 Analyser Interface Unit (LI-COR Biosciences, Lincoln, Nebraska, USA). EC systems were
209 installed at central locations within the two sites to maximise each particular land use within the
210 tower fetch under prevailing south-westerly wind conditions. Measurements were made at heights
211 above the ground surface of 3.9 m and 2.3 m at Sedge Fen and Bakers Fen, respectively.

212 A range of ancillary meteorological and soil physical measurements were made at each flux
213 tower. The net radiation and its incoming and outgoing short- and longwave components were

214 measured using CNR1 net radiometers (Kipp and Zonen BV, Delft, The Netherlands). Soil heat fluxes
215 were measured at a depth of 5 cm below the soil surface using HFP01 soil heat flux plates (Hukseflux
216 Thermal Sensors BV, Delft, The Netherlands). Air temperature and relative humidity were measured
217 with HMP45 probes (Vaisala, Vantaa, Finland) installed at 2 m above the ground surface.

218 Raw (20 Hz) EC data were post-processed using EddyPRO® Flux Calculation Software (LI-COR
219 Biosciences, Lincoln, Nebraska, USA). Thirty minute flux densities were computed as the mean
220 covariance between the vertical wind speed and atmospheric scalar quantities (e.g. H₂O, CO₂). Fluxes
221 were calculated using block averaging and by applying standardized procedures and corrections.

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

227

228 2.4. CO₂ and CH₄ static chamber fluxes

229 Static chambers to measure GHG fluxes were used on a total of 31 occasions at Sedge Fen,
230 and 37 occasions at Baker's Fen, with a higher frequency in summer (every 2-3 weeks) than winter
231 (every 4-8 weeks). All winter sampling occurred on snow-free days, and snowfall is rare in the
232 region. Sampling started in May 2013 at Baker's Fen, but was delayed until August 2013 at Sedge
233 Fen due to flooding making collar installation difficult. At each site, six polyvinyl chloride collars (20
234 cm high, 60 cm by 60 cm, inserted approximately 10 cm into the ground) were installed for CH₄ flux
235 measurements. At Sedge Fen all collars were located in the same area, with three collars being
236 dominated by *Phragmites australis* and three by *Cladium mariscus*. At Baker's Fen three collars were
237 sited within an *Agrostis stolonifera*-dominated dry mesotrophic grassland community, and three
238 within a *Juncus inflexus*-dominated rush pasture (see section 2.1). The *Agrostis* and *Juncus* sets of

239 collars were sited 80 m apart. At each study site two ditch locations were selected for floating
240 chamber measurements of CH₄ and CO₂.

241 To take flux measurements, transparent acrylic chambers were attached to the collars.
242 Stackable intermediate chamber sections were used when vegetation was tall. Silicone sponge was
243 used to create seals between chamber sections and collars. Small fans were used in all chamber
244 sections to facilitate internal mixing. At Sedge Fen the water table was frequently near, or above,
245 the peat surface, so boardwalk was used to minimise disturbance during sampling. CH₄
246 concentrations were measured in real time in the field using an Ultraportable Greenhouse Gas
247 Analyzer (Los Gatos Research, San Jose, USA). Changes in CH₄ concentrations were observed using a
248 tablet computer, and flux chambers were deployed until a linear flux, or clear zero flux, was
249 observed, which was typically 1-5 minutes.

250 Chamber fluxes were calculated according to Green *et al.* (2018), assuming a linear
251 relationship between chamber deployment time and mass change in CH₄. It has been common
252 practice to only include flux data where the R² of this relationship is above a certain value, but,
253 traditionally, fluxes have been calculated using several (~5) discrete gas samples analysed by gas
254 chromatograph in the lab. However, we measured fluxes in real time in the field, with a sampling
255 frequency of 1 Hz, thereby giving a much clearer picture of the behaviour of CH₄ emissions.
256 Removing measurements with a low R² could lead to the exclusion of small but noisy fluxes, thus
257 biasing the dataset towards higher fluxes. We therefore included all fluxes with a significant (≤ 0.05)
258 *p* value.

259 An attempt was made at calculating annual CH₄ fluxes for terrestrial collars, and CH₄ and CO₂
260 ditch fluxes, using the method of Green *et al.* (2018). A variety of environmental variables were
261 trialled in the models, including air temperature, soil temperature, water table depth, irradiance,
262 and temperature sum index (ETI). The temperature and irradiance data were taken from flux
263 towers. No satisfactory model fits were obtained. As such, annual CH₄ fluxes and ditch CO₂ fluxes
264 were estimated between measurement dates: days without measurements were assumed to have

265 the same flux as that recorded on the nearest day with a measurement (i.e. an approach equivalent
266 to linear interpolation; Green and Baird (2017)).

267 We weighted the flux of ditch CH₄ and CO₂ using the method of Evans *et al.* (2016), whereby
268 the annual flux expressed per unit of ditch surface is multiplied by the proportion of the fen
269 occupied by ditches (Frac_{ditch}). Frac_{ditch} was calculated using aerial photography and was 0.014 and
270 0.017 for Sedge Fen and Baker's Fen respectively.

271

272 2.5. Water sampling and analysis

273 Water sampling took place on 42 occasions, with a higher frequency in summer (every 2-3
274 weeks) than winter (every 4-5 weeks). Water sampling took place at four different ditch locations
275 on each site. Additionally, one sample was taken from Wicken Lode; the river that is used as a water
276 supply to transfer water onto both sites. At each sampling point, water was collected in a 60 ml
277 Nalgene® and a 500 ml Nalgene® bottle, and a sample for dissolved GHG analysis was collected in a
278 12 ml borosilicate glass vial, using the headspace method (Hope *et al.*, 2004). Air pressure and
279 temperature were recorded at the time of sampling using a C4141 thermo-hygro-barometer
280 (Commeter, Roznov pod Radhostem, Czech Republic) and water temperature was measured with a
281 SuperFast Thermapen (ETI, Worthing, UK). After collection, samples were returned to the laboratory
282 and stored in the dark at 4°C until analysis, typically within one week.

283 Electrical conductivity (EC) and pH were measured on the 60 ml sample with an Orion VERSA
284 STAR (Thermo Scientific, Waltham, USA). The sample was then filtered at 0.45 µm. DOC (measured
285 as non-purgeable organic carbon) and dissolved inorganic carbon (DIC) were measured on the
286 filtered samples using a TOC analyser (Shimadzu, Kyoto, Japan). Nitrate was measured using an ELIT
287 8021 ion-selective electrode (NICO 2000, Harrow, UK) and appropriate standards (range 1-100 mg l⁻¹).
288

289 The 500 ml sample was used to measure particulate organic carbon (POC). For each sample,
290 500 ml of deionised water was passed through a 0.7 µm Whatman GF/F filter which was then

291 combusted at 500°C for five hours, and weighed. 500 ml of sample was then filtered using the same
292 filter, which was oven-dried at 105°C for five hours and weighed to give an estimate of suspended
293 sediment. The filter was then placed in a furnace at 375°C overnight, and weighed a final time to
294 provide an estimate of particulate organic matter (POM). POC was then calculated from POM using
295 the regression equation of Ball *et al.* (1964).

296 The 12 ml headspace sample was analysed for CH₄ and CO₂ using the Ultraportable
297 Greenhouse Gas Analyzer equipped with a sampling loop following Baird *et al.* (2010). For this, gas is
298 continuously circulated in a loop through the inlet and outlet of the analyser, until the concentration
299 stabilises. The headspace sample of dissolved gas is then injected into the loop and the
300 concentration noted. Using the equations from Baird *et al.* (2010), it is then possible to calculate the
301 true concentration of the dissolved gas.

302

303 2.6. Hydrology

304 Due to the sensitivity of Sedge Fen, there were considerable restrictions on hydrological
305 instrumentation imposed by the landowner. However, a previous detailed analysis of the hydrology
306 of the site concluded that the main control over water-table depths was the balance between
307 rainfall and evapotranspiration, with other losses being comparatively minor (McCartney *et al.*,
308 2001). At Sedge Fen the water table was measured next to the collars using a dipwell fitted with an
309 Orpheus Mini (OTT, Kempten, Germany) pressure transducer with a 1 cm resolution, logging every
310 hour. At Baker's Fen, a dipwell was positioned next to each set of collars. Each dipwell was fitted
311 with a Level TROLL 500 (In-Situ, Fort Collins, USA) with an accuracy of 0.35 cm or better, and a
312 resolution of 0.035 cm, logging every 15 minutes. An additional 10 manually-recorded dipwells were
313 installed across the fen with measurements being taken approximately monthly. A third Level TROLL
314 500 was deployed directly into the ditch near the sluice outflow at Baker's Fen, to measure ditch
315 water level. Ditch water level was also manually measured at locations where water samples were
316 collected.

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

322

$$323 \quad P + Q_{in} + G_{in} = ET + Q_{out} + G_{out} + \Delta s \quad \text{Equation 1}$$

324

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

341

342 **3. Results**

343 3.1. Soil properties

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

347

348 3.2. Hydrology

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

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

367

368 3.3. Water chemistry and fluvial carbon losses

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

378 Due to the low water fluxes from each fen, aquatic C losses were small (Fig. 3). For Sedge
379 Fen the majority of aquatic C flux was in the form of DIC (mean annual flux $16.4 \text{ g C m}^{-2} \text{ yr}^{-1}$) with a
380 mean annual DOC flux of $4.1 \text{ g C m}^{-2} \text{ yr}^{-1}$, and POC of $0.4 \text{ g C m}^{-2} \text{ yr}^{-1}$. The dissolved CO_2 flux leaving
381 the site via the ditch network was estimated to be $1.23 \text{ g C m}^{-2} \text{ yr}^{-1}$, and dissolved CH_4 exports were
382 negligible ($< 0.01 \text{ g C m}^{-2} \text{ yr}^{-1}$). Aquatic C fluxes for Baker's Fen followed a similar pattern, and were
383 also dominated by DIC (mean annual fluxes $27.1 \text{ g C m}^{-2} \text{ yr}^{-1}$). Exports of DOC were $8.8 \text{ g C m}^{-2} \text{ yr}^{-1}$,
384 POC $1.1 \text{ g C m}^{-2} \text{ yr}^{-1}$, dissolved CO_2 $1.8 \text{ g C m}^{-2} \text{ yr}^{-1}$ and dissolved CH_4 $0.01 \text{ g C m}^{-2} \text{ yr}^{-1}$. Fluxes of C onto
385 both sites via managed inputs of river water were dominated by DIC (Fig. 3), with inputs of
386 $\text{DOC} + \text{POC} + \text{CO}_2 + \text{CH}_4$ summing to 1.5 and $1.6 \text{ g C m}^{-2} \text{ yr}^{-1}$ for Sedge Fen and Baker's Fen respectively.

387

388 3.4. CO_2 eddy-covariance fluxes

389 The cumulative CO_2 budget for Sedge Fen indicates that the fen is functioning as a sink,
390 although the uncertainty range falls above zero for the merged 2013-2014 year. For the annual
391 period from July 2013 to June 2014 NEE was $-55 \pm 112 \text{ g C m}^{-2} \text{ yr}^{-1}$, whilst for 2015 it was $-183 \pm 98 \text{ g}$
392 $\text{C m}^{-2} \text{ yr}^{-1}$ (Fig. 4), giving a mean of $-119 \text{ g C m}^{-2} \text{ yr}^{-1}$. Water table and meteorological data suggest
393 that the drought-induced drawdown in 2013 was particularly extreme, and data from 2007-2015
394 show that severe water table drawdown occurred three times during this period. Therefore, if the

395 data were weighted assuming that the 2013-14 value was representative of one year in three, and
396 that the flux measured in 2015 was representative of two years in three, the estimated mean annual
397 NEE would be $-140 \text{ g C m}^{-2} \text{ yr}^{-1}$. Baker's Fen was a consistent source of CO_2 , with NEE values of $157 \pm$
398 111 , 83 ± 107 and $130 \pm 91 \text{ g C m}^{-2} \text{ yr}^{-1}$ for 2013, 2014 and 2015 respectively; giving a mean (and SD)
399 of $123 \pm 37 \text{ g C m}^{-2} \text{ yr}^{-1}$ (Fig. 4).

400

401 3.5. CO_2 and CH_4 static chamber fluxes

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

410 CH_4 fluxes at Sedge Fen were close to zero in 2013, coinciding with severe water table
411 drawdown, but large emissions were observed in 2014 and 2015 (Fig. 5). Overall estimated mean
412 annual CH_4 emissions were $11.9 \text{ g C m}^{-2} \text{ yr}^{-1}$ for *Phragmites*, and $5.6 \text{ g C m}^{-2} \text{ yr}^{-1}$ for *Cladium*. Assuming
413 an equal mix of communities across the site gives a mean CH_4 emission of $8.75 \text{ g C m}^{-2} \text{ yr}^{-1}$. Ditch
414 emissions of CH_4 and CO_2 were larger at Sedge Fen, with the highest fluxes occurring during spring
415 and summer (Fig. 6). Annual ditch CO_2 flux was $2610 \text{ g C m}^{-2} \text{ yr}^{-1}$. Adjusting this value to the total
416 ditch area of the site gives an emission of $36.6 \text{ g C m}^{-2} \text{ yr}^{-1}$ for the entire fen. The estimated annual
417 mean CH_4 emission was $125 \text{ g C m}^{-2} \text{ yr}^{-1}$, and when adjusted to the total ditch area gives was 1.76 g C
418 $\text{m}^{-2} \text{ yr}^{-1}$.

419

420 3.6. Annual carbon balances

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

$$423 \text{NECB} = \text{NEE} + \text{CH}_4_{\text{ditch}} + \text{CH}_4_{\text{terrestrial}} + \text{DOC} + \text{POC} + \text{CH}_4_{\text{diss}} + \text{CO}_2_{\text{diss}} \quad \text{Equation 2}$$

424 where NECB is the net ecosystem carbon balance, NEE is net ecosystem exchange measured by flux
425 tower (and therefore includes ditch CO₂ fluxes), CH₄_{ditch} is CH₄ emission from the ditches measured
426 by static chamber, DOC and POC are the respective net fluvial fluxes, and CH₄_{diss} and CO₂_{diss} are the
427 respective net lateral fluxes of dissolved GHGs. The calculated NECBs show that Sedge Fen is a C
428 sink, whilst Baker's Fen is a C source (Table 3).

429

430 **4. Discussion**

431 *4.1. Soil properties*

432 The management histories of the two sites are reflected in the soil properties. Past use as
433 cropland has resulted in extensive subsidence at Baker's Fen; peat depth and C content are both
434 low, and bulk density and mineral content are very high. Peat depth at Sedge Fen reaches almost 4
435 m, whilst on Baker's Fen it is under 0.5 m, suggesting that over 3 m of peat has been lost due to
436 conversion to cropland (Table 1). Estimated subsidence rates (based on NECB) at Baker's Fen are
437 0.06 cm yr⁻¹, compared to 0.44 and 0.62 cm yr⁻¹ at nearby arable sites on shallow and deep peat
438 respectively (Evans *et al.*, 2016a). It therefore appears that rewetting has reduced subsidence rates.
439 Nevertheless, considering the present soil conditions, it may no longer be appropriate to consider
440 the soil a peat, although it remains organic-rich and conforms to both the definition of a 'wasted
441 peat' (Natural England, 2010), and of 'organic soil' (IPCC, 2006). Similar soils have been shown to
442 retain many of the biochemical functions of deeper peats, including ongoing CO₂ emissions when
443 exposed to drainage (Tiemeyer *et al.*, 2016). It has been calculated that, assuming current loss rates,
444 all peat will be lost from Baker's Fen in 400 years (Evans *et al.*, 2016a). In contrast, Sedge Fen has
445 very deep peat and remains a relatively large store of C.

446

447 4.2. Hydrology

448 Hydrological monitoring clearly demonstrates the challenges of keeping both sites wet (Fig.
449 1). The sites are small fragments of non-arable land in an otherwise agricultural region, and are
450 hydrologically disconnected from the surrounding rivers and from groundwater. Furthermore, both
451 sites have been historically modified to varying degrees. Water is transferred from the adjacent
452 river to irrigate the fens and, although the period of fen irrigation is limited due to regional demands
453 for water to irrigate crops, the transferred amount is an important component of the water balance
454 at each site. When water inputs cease during summer, both sites dry out, as evapotranspiration
455 exceeds precipitation. The severe and prolonged water table drawdown that occurred at Sedge Fen
456 in 2013 is unlike the hydrological dynamics of intact fen systems with natural hydrological function,
457 where the water table typically resides close to the surface year round (e.g. Chimner and Cooper,
458 2003). Before drainage, the wider fenland region would have been a wetland mix of floodplain fen
459 and open water, with numerous dendritic river channels (Malone, no date). Drawdown at Baker's
460 Fen was also severe and prolonged, and occurred in all years; every summer the water table fell
461 below the level of the loggers, indicating that the entire soil profile was aerated. Once water is
462 transferred onto site in autumn, rewetting occurs within days. This 'bimodal' pattern of seasonal
463 water table variation is unlikely to be conducive to the full restoration of wetland vegetation species,
464 a conclusion supported by Stroh *et al.* (2013). They suggested that, even if suitable hydrological
465 conditions and a propagule source were established, the site would still not be able to support a
466 species-rich wetland flora. However, if seasonal restrictions on fen irrigation were removed it might
467 be feasible to keep the fen inundated throughout summer, with the possibility that *Phragmites* and
468 *Typha* might colonise the site, perhaps initially as floating rafts (Money *et al.*, 2009).

469

470 4.3. Water chemistry and fluvial carbon losses

471 At Sedge Fen DOC concentrations displayed small fluctuations, though with no clear
472 seasonal pattern (Fig. S13). Concentrations were lowest after the dry summer of 2013. The ditch

473 water levels at Sedge Fen were relatively stable, and the low concentrations in summer 2013 are
474 likely to be due to reduced production/mobility of DOC in the peat (Clark *et al.* 2005), or increased
475 DOC degradation in the ditches (Moody *et al.*, 2013). In contrast, concentrations at Baker's Fen
476 displayed pronounced seasonal fluctuations, with peaks in spring/summer, and troughs in
477 autumn/winter (Fig. SI3). Considering that the troughs coincided with autumn addition of low-DOC
478 river water into the ditches it seems likely that the primary control on DOC concentrations is evapo-
479 concentration in summer, followed by dilution in winter (Waiser, 2006). At Baker's Fen ditch water
480 levels became very low in summer (some ditches dried out completely), and a negative correlation
481 between ditch depth and DOC concentration was found (Fig. SI4). The influence of transferring river
482 water onto Baker's Fen is also evident in the increases in ditch nitrate concentration that were
483 observed in November 2013 and 2014 (Fig. SI6). There were fluxes of DOC and POC (and dissolved
484 GHGs) onto both fens during the addition of river water. These represented 27% and 16% of total
485 fluvial C losses at Sedge Fen and Baker's Fen respectively. The lower fluvial inputs, when compared
486 to fluvial losses, are due to the low DOC and POC concentrations in the river (Table 2, Fig. SI3, SI5),
487 and the relatively small contribution of inputs of river water to the hydrological budgets.

488 DIC concentrations were high for both sites (Table 2, Fig. SI2). Since DIC in fen runoff is
489 generally derived from weathering of carbonate or siliceous minerals, rather than peat, this flux
490 cannot strictly be considered part of the peatland C balance (Evans *et al.*, 2016b). Additionally, since
491 DIC in the drainage network will remain in a dissociated form due to the high pH of the water, little
492 of this flux can be expected to be evaded as CO₂, or therefore to contribute to overall GHG emissions
493 from the fen.

494 At Sedge Fen exports of aquatic C generally occurred on a restricted seasonal basis, due to
495 the limited water discharge from the fen, but were more frequent at Baker's Fen where water
496 discharge was greater (Fig. 2, 3). DOC was the principal component of aquatic fluxes, and was
497 responsible for ~73% of fluvial C exported. However, our estimated DOC exports (4.1 and 8.8 g C m⁻²
498 yr⁻¹ from Sedge Fen and Baker's Fen respectively) are close to fluxes reported from some temperate

499 and boreal fens in Scandinavia and Canada ($5 \text{ g C m}^{-2} \text{ yr}^{-1}$, Strack *et al.*, 2008; $3.7 \text{ g C m}^{-2} \text{ yr}^{-1}$, Juutinen
500 *et al.*, 2013) and German drained peat grasslands ($5.2 \text{ g C m}^{-2} \text{ yr}^{-1}$, Tiemeyer and Kahle, 2014), and
501 are low compared to values from UK raised bogs ($25 \text{ g C m}^{-2} \text{ yr}^{-1}$, Dinsmore *et al.*, 2010) and blanket
502 bogs ($33 \text{ g C m}^{-2} \text{ yr}^{-1}$, Worrall *et al.*, 2003). Other semi-natural and agricultural fens in the same
503 region (East Anglia, UK), measured concurrently with our study, also had low DOC fluxes ($4.1 - 7.9 \text{ g}$
504 $\text{C m}^{-2} \text{ yr}^{-1}$), whilst semi-natural fens and peat grasslands in wetter parts of the UK had higher fluxes
505 (Somerset; $10-22 \text{ g C m}^{-2} \text{ yr}^{-1}$, Anglesey; $18-31 \text{ g C m}^{-2} \text{ yr}^{-1}$) (Evans *et al.*, 2016a). This reflects the fact
506 that hydrological regime has a strong control on such small and isolated fens; that water losses, and
507 therefore fluvial C losses, will be greater in systems where the difference between precipitation and
508 evapotranspiration is larger. POC fluxes were small, comprising 6.8% ($0.4 \text{ g C m}^{-2} \text{ yr}^{-1}$) and 9.2% (1.1
509 $\text{g C m}^{-2} \text{ yr}^{-1}$) of fluvial C flux at Sedge Fen and Baker's Fen, respectively. In upland peatlands, POC
510 export can sometimes equal that of DOC, particularly if erosional features are present (Pawson *et al.*,
511 2012). The low rainfall levels, lack of overland flow and consequently low rates of fluvial erosion in
512 most lowland peatlands reduce the importance of POC to the fluvial C budget. Most estimates of
513 POC have been for upland blanket bogs, but Olefeldt and Roulet (2012) reported fluxes of 1.1 and
514 $3.6 \text{ g C m}^{-2} \text{ yr}^{-1}$ for fen outflows in a subarctic peatland complex in Sweden.

515 For dissolved GHGs, we assumed that lateral (dissolved) fluxes were separate from vertical
516 (gaseous) fluxes. When water outputs from the fens occur, dissolved GHGs will be exported out of
517 the system into rivers, and may be emitted off-site. Dissolved CO_2 was $\sim 18\%$ of total fluvial C export,
518 larger than the POC flux, whilst CH_4 (which has a low solubility in water) made a negligible
519 contribution ($\leq 0.1\%$). Our dissolved CO_2 fluxes of $1.2 \text{ g C m}^{-2} \text{ yr}^{-1}$ and $1.8 \text{ g C m}^{-2} \text{ yr}^{-1}$ are similar to
520 that reported from UK a raised bog ($1.3 \text{ g C m}^{-2} \text{ yr}^{-1}$, Dinsmore *et al.*, 2010) but smaller than those
521 from blanket bogs ($3.8 \text{ g C m}^{-2} \text{ yr}^{-1}$, Worrall *et al.*, 2003) and drained Irish grasslands ($2.4-4.4 \text{ g C m}^{-2}$
522 yr^{-1} , Barry *et al.*, 2014). It is likely that this is because slow water movement in ditches results in the
523 majority of aquatic CO_2 being lost on-site as gaseous fluxes, rather than exported off-site fluvially. It
524 should be noted that many C-balance studies do not measure dissolved GHGs and POC, and instead

525 focus solely on DOC (e.g. Roulet *et al.*, 2007). The total aquatic C fluxes for our sites were 5.72 and
526 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
527 (Table 3). Therefore, if we had neglected to measure POC and dissolved GHGs, 25-28% of fluvial C
528 exports would be missing from the total budget.

529

530 4.4. CO₂ and CH₄ fluxes

531 Eddy covariance flux tower measurements showed that Sedge Fen was a large CO₂ sink (Fig.
532 4). Although significant periods of water table drawdown occur at Sedge Fen, the fen is also
533 seasonally inundated with standing water, and the plant species present have the potential to form
534 peat under waterlogged conditions. NEE was in the same range as reported values from northern
535 bogs and fens (Yu, 2012). In contrast, flux tower measurements for Baker's Fen suggest that the site
536 was a net source of CO₂ (Fig. 4). Systematic reviews have shown that drained peatlands have higher
537 rates of ecosystem respiration (Haddaway *et al.*, 2014), and Baker's Fen had suffered serious soil loss
538 and compaction before the restoration activity was conducted, and still experiences consistent and
539 pronounced water table drawdown in summer (Fig. 1). It is therefore unsurprising that the site is a
540 source of CO₂. Grasslands on drained organic soils can act as net CO₂ sinks (e.g. Renou-Wilson *et al.*,
541 2014), but it seems probable that a higher water table would need to be instated for this to occur at
542 Baker's Fen (Wilson *et al.*, 2016).

543 Net CH₄ fluxes were approximately zero at Baker's Fen, with areas of *Agrostis* acting as small
544 sinks for the majority of the time (Fig. 5). Areas of *Juncus* were often small sinks of CH₄, but
545 emissions were occasionally observed, with the overall effect being that *Juncus* patches were net
546 sources. Low emissions from organic grasslands would be expected due to the low water table and
547 organic matter content (Tiemeyer *et al.*, 2016), and emissions will be further mitigated by low CH₄
548 diffusion due to drainage-induced increases in soil bulk density (Nykänen *et al.*, 1998). The observed
549 emissions could be due to CH₄ transport through aerenchymatous tissue in *Juncus* plants
550 (Henneberg *et al.*, 2012); *Juncus* clumps have sometimes been observed to act as point-source

551 emissions of CH₄ in drained peatlands (Henneberg *et al.*, 2015). Equally, the presence of *Juncus* may
552 simply indicate that these collars were situated in wetter areas of the site where CH₄ emissions were
553 more likely to occur. At Sedge Fen, CH₄ emissions were low in 2013 when the largest water table
554 drawdown occurred, but much larger in 2014 and 2015 (Fig. 5), as would be expected from a site
555 with deep peat, wetland vegetation, and seasonal inundation.

556 Ditch emissions of CH₄ were low from Baker's Fen (Fig. 6). However, as the terrestrial
557 component of the fen was CH₄ neutral, the ditches resulted in the fen acting as a small net source of
558 CH₄. The annual flux (per unit ditch water surface) of 8.9 g C m⁻² yr⁻¹ is low compared to other
559 reports from grasslands, which span 40-75 g C m⁻² yr⁻¹ (Evans *et al.*, 2016b). Whilst our estimate is
560 based on just two floating chambers, a more spatially intensive campaign in 2015 (replicated
561 seasonally) produced a similar estimate for the site of 13.7 g C m⁻² yr⁻¹ (Peacock *et al.*, 2017). The
562 relatively low ditch flux is explicable if ditch CH₄ fluxes are driven by inputs from the soil, as Rasilo *et*
563 *al.* (2017) found for small boreal streams. The extreme peat oxidation, low organic content of the
564 soil, and low water tables at Baker's Fen are unlikely to favour methanogenesis in the soil, as well as
565 resulting in a large zone where methanotrophy can occur (Yavitt *et al.*, 1997). In contrast to this,
566 emissions were substantial at Sedge Fen, at 125 g C m⁻² yr⁻¹, making them equivalent to 20% of
567 terrestrial CH₄ fluxes. Although CH₄ is only a minor component of the NECB, CH₄ fluxes from
568 terrestrial vegetation and ditches are important from a climate perspective due to the higher global
569 warming potential of CH₄ (IPCC, 2006).

570 Annual ditch fluxes of CO₂ were larger at Sedge Fen: 2610 g C m⁻² yr⁻¹ compared to 1245 g C
571 m⁻² yr⁻¹ at Baker's Fen (Fig. 6). Although we are unsure of why fluxes from Baker's Fen are lower, it
572 could be that the low organic content of the soil, alongside other changes in soil properties (Table 1),
573 resulted in reduced respiration of organic matter and therefore lower emissions. Alternatively, it
574 could be an artefact of the low number of spatial replicates at each site. Whilst some have found
575 that ditches do not contribute any significant amount to net CO₂ emissions in cutaway peatlands
576 (Sundh *et al.*, 2000) or peatland grasslands (Best and Jacobs, 1997), others have reported large

577 fluxes from ditches in peatland grassland and reedbeds and from agricultural ditches (Schrier-Uijl *et*
578 *al.*, 2011). Our relatively high measured CO₂ fluxes are potentially important, especially at Sedge
579 Fen; when weighted by ditch area the annual flux is 36.6 g C m⁻² yr⁻¹, which has the effect of
580 somewhat reducing the net CO₂ uptake of the fen. However, this calculation may be an artefact of
581 having only two floating chamber locations. A spatially intensive campaign repeated four times in
582 2015 gave an annual flux of 413 g C m⁻² yr⁻¹ (Peacock *et al.*, 2017) which is more in keeping with
583 literature values. Using this number would give an area-weighted flux of 5.8 g C m⁻² yr⁻¹; i.e.,
584 offsetting considerably less of terrestrial CO₂ uptake.

585

586 4.5. Annual carbon balance and implications for rewetting

587 Despite being subjected to occasional, extreme water table drawdown events, Sedge Fen
588 remains a considerable overall C sink of -104 g C m⁻² yr⁻¹ (Table 3). This is relatively high when
589 compared to published measurements from UK bogs (e.g. -28 g C m⁻² yr⁻¹, Helfter *et al.*, 2015; -15.4 g
590 C m⁻² yr⁻¹, Worrall *et al.*, 2003; -56 g C m⁻² yr⁻¹, Worrall *et al.*, 2009), acidic Scandinavian peatlands
591 (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
592 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-
593 natural *Cladium* and *Phragmites* fen, but lower than -281 g C m⁻² yr⁻¹ from a nutrient-rich *Phragmites*
594 fen (both in the UK) (Evans *et al.*, 2016a). It seems probable that these large values are due to the
595 high productivity of tall fen vegetation (Wheeler and Shaw, 1991), which in turn is due to the
596 favourable climatic and chemical conditions for growth in lowland fens when compared to
597 upland/northern peatlands. When combined with favourable hydrological conditions there is,
598 therefore, a greater potential for relatively rapid C accumulation. However, it is important to
599 consider that peatlands can switch dramatically from C sources to sinks (Roulet *et al.*, 2007), and a
600 longer period of monitoring would be needed to see whether this is the case at Sedge Fen.

601 In contrast, the substantial net C loss from Bakers Fen (NECB 133 g C m⁻² yr⁻¹, Table 3)
602 suggests that peat loss is continuing at this site despite the restoration measures undertaken. Beetz

603 *et al.* (2013) reported NECBs of -147 and 88 g C m⁻² yr⁻¹ for a rewetted peat grassland over two years,
604 and suggested that the difference was due to a mowing event in October of the second year.
605 However, the water table was higher at their site compared to Baker's Fen, with a mean depth of
606 approximately 25 cm. The absence of a return of wetland vegetation and C sink at Baker's Fen, even
607 after ~20 years of restoration is perhaps not surprising. Moreno-Mateos *et al.* (2012) showed in
608 their meta-analysis of 621 global wetland sites that C storage and accumulation of soil organic
609 matter remain lower in restored sites compared to reference sites, even on 50-100 year time scales.
610 They hypothesised that restored wetlands may shift to stable states that differ from their original
611 condition.

612 It therefore seems likely that Baker's Fen will not begin to sequester more C than it loses
613 unless management is changed. The most effective option would be to transfer more water onto
614 the fen throughout the year, but this would be at the expense of agricultural water needs in the
615 region. The site will continue to behave like a seasonally-inundated wetland without a year-round
616 higher water table. Tiemeyer *et al.* (2016) suggest that mean water-table depth needs to be less
617 than 20 cm to constrain CO₂ losses due to decomposition, but at Bakers Fen it was 46 cm in 2014
618 and 55 cm in 2015. Other research suggests that because the soil properties have been altered to
619 such a degree the reestablishment of original wetland vegetation would remain difficult (Stroh *et al.*,
620 2013). However, as noted in section 4.2, it might be that prolonged inundation could lead to the
621 development of floating rafts of wetland plant species (Money *et al.*, 2009). Nevertheless, it is worth
622 noting that the NECB of other croplands in the region is 693 and 773 g C m⁻² yr⁻¹ (Evans *et al.*, 2016a).
623 Rewetting has therefore potentially suppressed C losses to ~20% of their former value. Similarly,
624 research from Finland has shown that CO₂ fluxes from abandoned agricultural peatlands is
625 considerably less than fluxes from arable peatlands (Maljanen *et al.*, 2007). As well as reducing C
626 losses, the rewetting of Baker's Fen has provided a buffer zone to Sedge Fen, increased biodiversity,
627 and provided a recreational environment for visitors (Peh *et al.*, 2014).

628

629 4.6. Concluding remarks

630 Global GHG emissions caused by draining peatlands for cropland are 630 Tg CO₂e yr⁻¹
631 (Carlson *et al.*, 2017), and there is continued interest in peatland restoration as a potential climate
632 mitigation measure (Griscom *et al.*, 2017). The rewetting of peat-based croplands offers a viable
633 way to substantially reduce GHG and C losses, with the emission reductions from rewetted grassland
634 and cropland being in the region of 20 t CO₂e ha⁻¹ yr⁻¹ (Bonn *et al.*, 2014). If C losses are simply
635 slowed, rather than being reversed, the entire volume of peat will still eventually be lost to the
636 atmosphere; however, this nevertheless represents a reduction in GHG emissions (equivalent to
637 reducing fossil fuel combustion) in the medium term. Considering the national and global
638 importance of drained organic soils for food production, there are significant socio-economic
639 barriers to the re-wetting of cultivated peatlands, including issues relating to national food security
640 and risks of ‘leakage’ if GHG emissions associated with food production are simply transferred from
641 one location or form to another. Paludiculture (high water table agriculture supporting both
642 economic returns and peat formation) has been suggested as an optimal future use for currently
643 drained peatlands (Wichtmann and Joosten, 2007), but remains both technically and economically
644 challenging to implement at the large scale. In the short to medium term, therefore, it is likely that
645 measures to reduce drainage-related GHG emissions from peatland remaining under cultivation (so
646 called “responsible peatland management”; Wijedasa *et al.* (2016)), including transitions from deep-
647 drained to shallow-drained cropland or grasslands, may provide the most effective means of
648 reducing GHG emissions from these regions.

649

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658

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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 ($\mu\text{S cm}^{-1}$)	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 CO₂ 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
<i>Gaseous C</i>		
NEE	-119	123
Ditch CO ₂	36.6*	21.6*
Ditch CH ₄	1.8	0.15
Terrestrial CH ₄	8.8	0
<i>Aquatic C losses</i>		
DOC	4.1	8.83
POC	0.39	1.08
Dissolved CO ₂	1.23	1.81
Dissolved CH ₄	0.003	0.013
<i>Aquatic C inputs</i>		
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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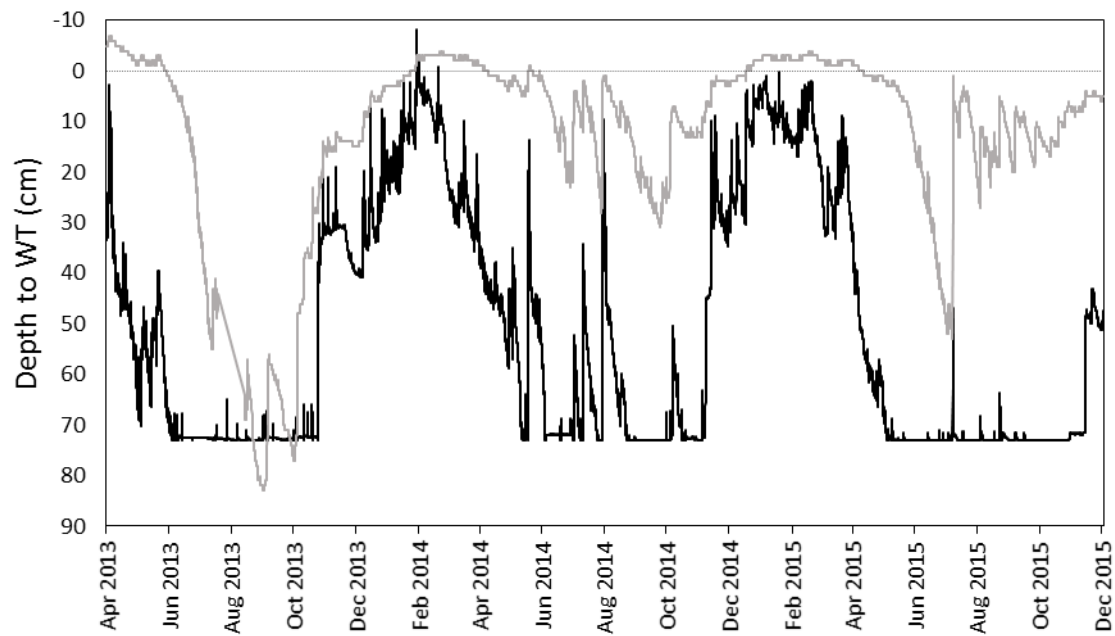


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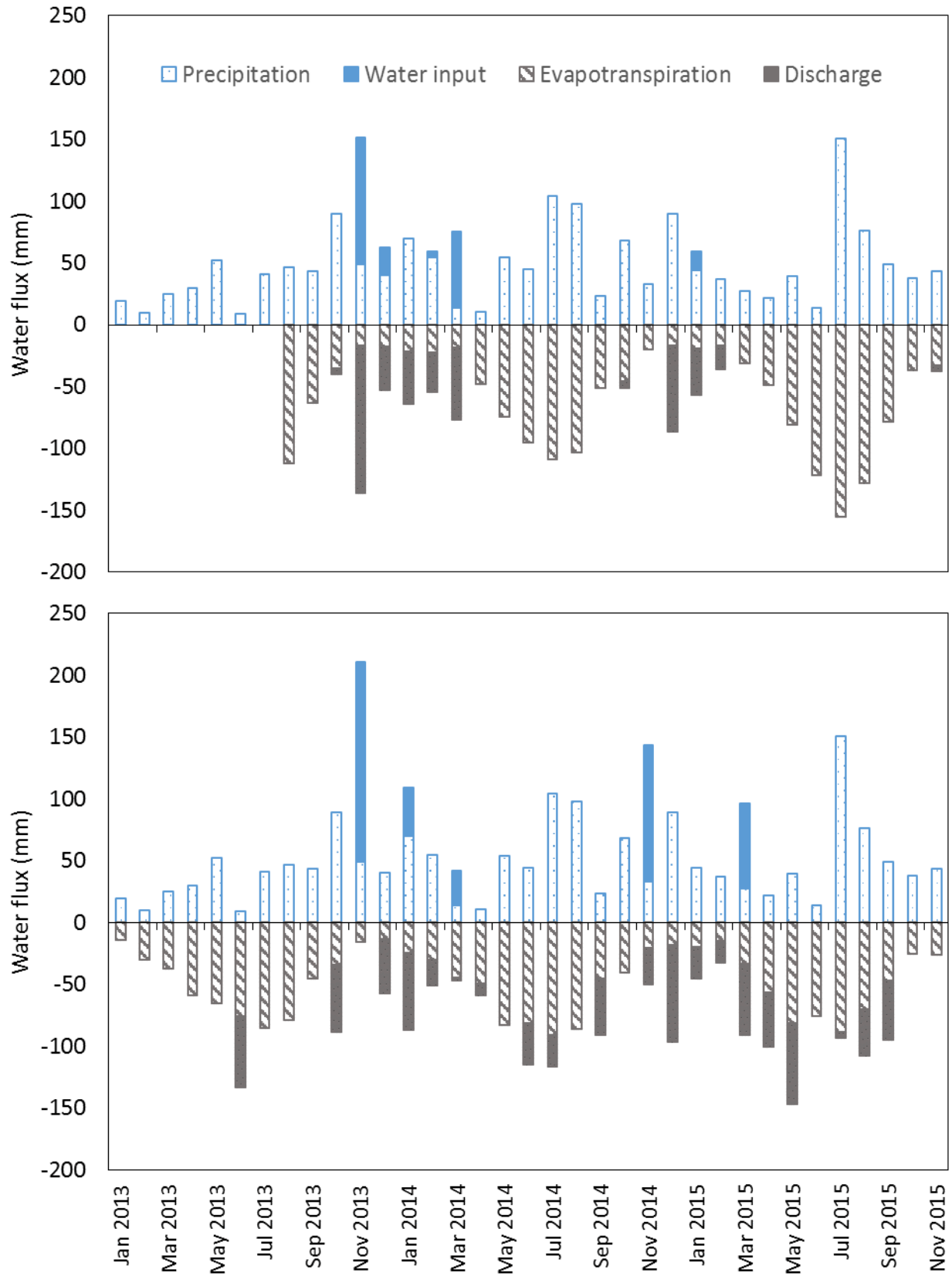
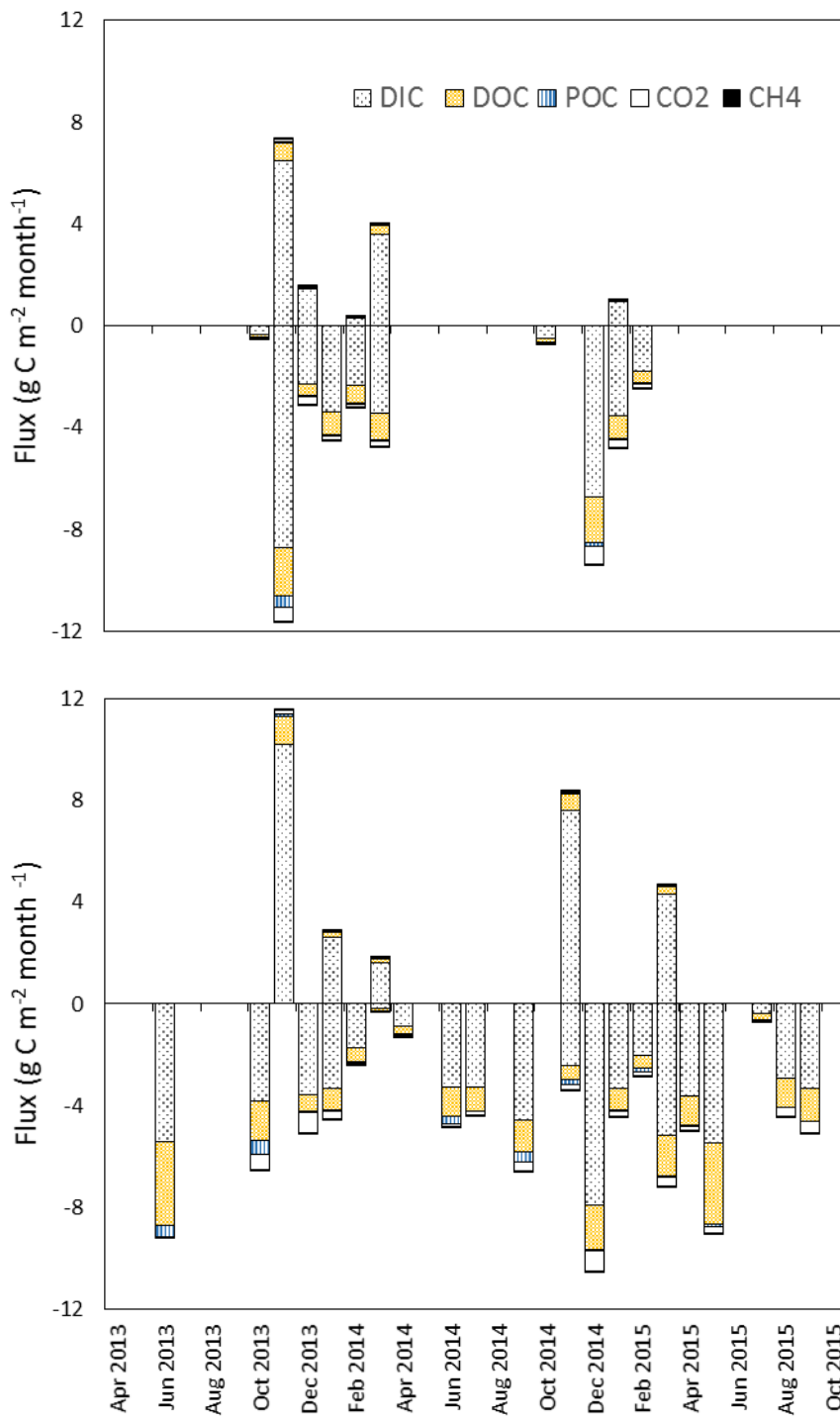


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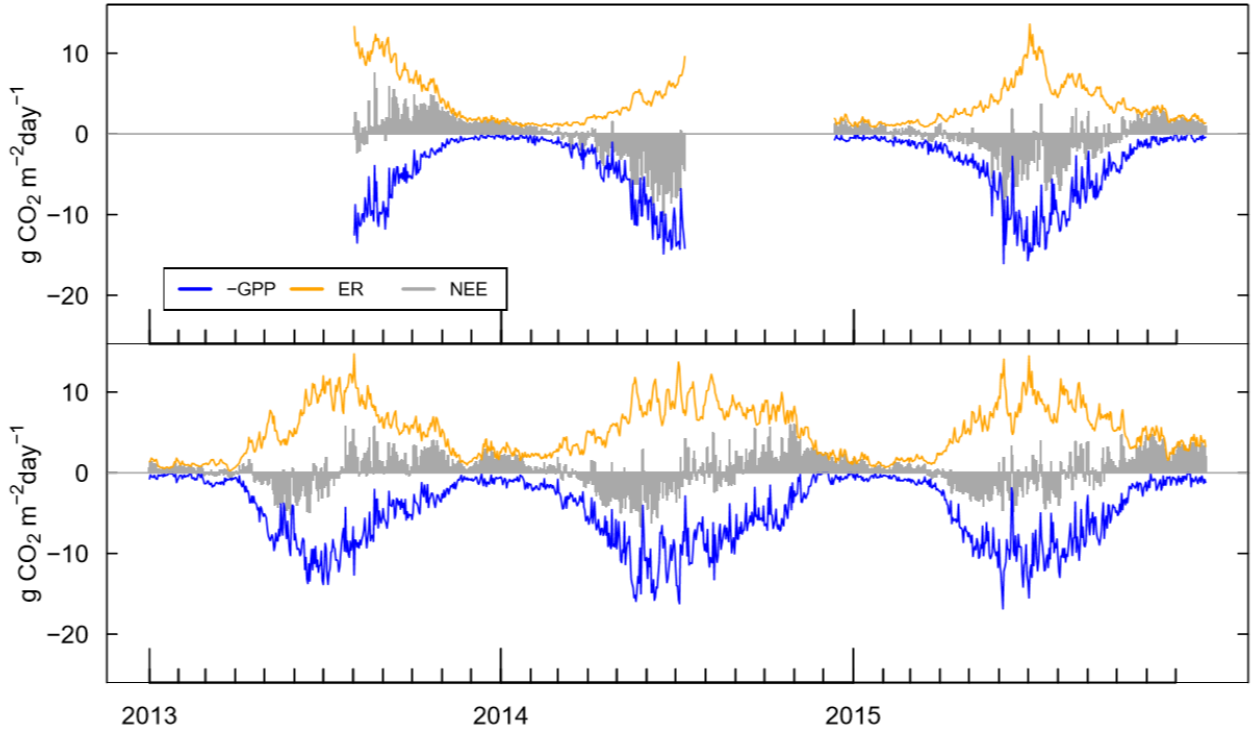
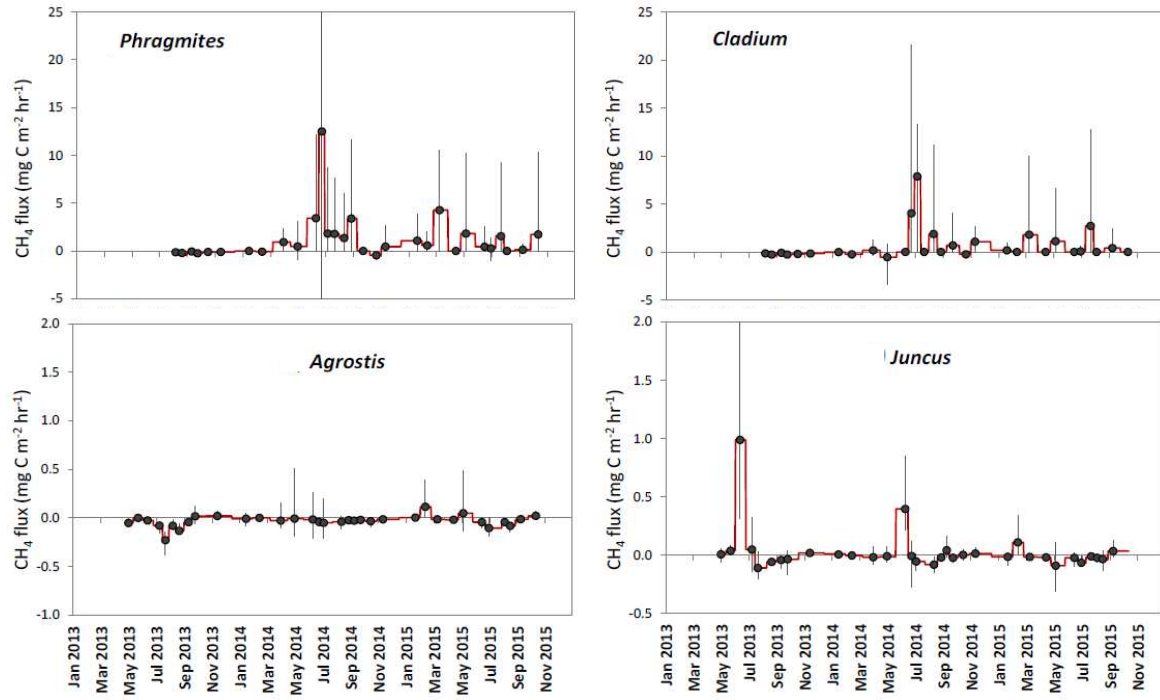
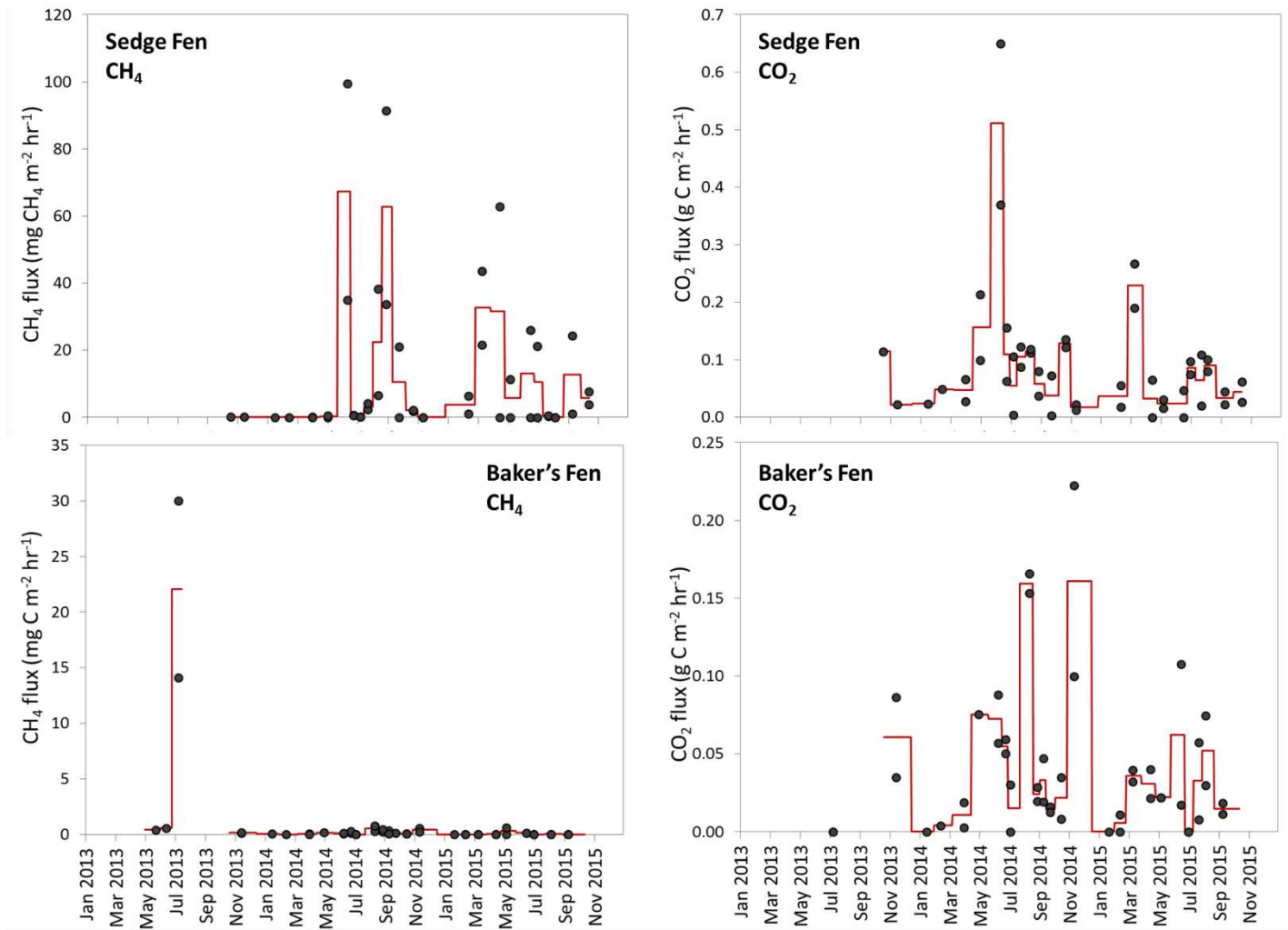


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Figure 1. Water tables for Sedge Fen (grey) and Baker's Fen (black). Negative values indicate water levels above the peat surface (horizontal line at 0 cm), i.e. flooding. For Baker's Fen, the logger in the dipwell was located at approximately 73 cm 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 CO₂ budgets for Sedge Fen (top) and Baker's Fen (bottom), showing gap-filled NEE, GPP and ER.

Figure 5. Measured CH₄ 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. CH₄ and CO₂ 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.