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Final report on project SP1210: *Lowland peatland systems in England and Wales – evaluating greenhouse gas fluxes and carbon balances*

Final Report, 2017



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SUMMARY AND KEY FINDINGS

Lowland peatlands represent one of the most carbon-rich ecosystems in the UK. As a result of widespread habitat modification and drainage to support agriculture and peat extraction, they have been converted from natural carbon sinks into major carbon sources, and are now amongst the largest sources of greenhouse gas (GHG) emissions from the UK land-use sector. Despite this, they have previously received relatively little policy attention, and measures to reduce GHG emissions either through re-wetting and restoration or improved management of agricultural land remain at a relatively early stage. In part, this has stemmed from a lack of reliable measurements on the carbon and GHG balance of UK lowland peatlands. This project aimed to address this evidence gap via an unprecedented programme of consistent, multi-year field measurements at a total of 15 lowland peatland sites in England and Wales, ranging from conservation-managed 'near-natural' ecosystems to intensively managed agricultural and extraction sites. The use of standardised measurement and data analysis protocols allowed the magnitude of GHG emissions and removals by peatlands to be quantified across this heterogeneous dataset, and for controlling factors to be identified. The network of seven flux towers established during the project is believed to be unique on peatlands globally, and has provided new insights into the processes that control GHG fluxes in lowland peatlands. The work undertaken is intended to support the future development and implementation of agricultural management and restoration measures aimed at reducing the contribution of these important ecosystems to UK GHG emissions.

Key findings:

1. Results from the project confirm that lowland peats in England and Wales are major sources of UK GHG emissions. Out of 15 sites studied, 11 were losing carbon (including all sites under cropland and grassland) and 13 were net GHG emission sources.
2. Conservation-managed lowland fens appear to be among the most effective carbon sinks per unit area in England and Wales, whereas lowland peats under intensive arable agriculture in England are probably the UK's largest land-use derived source of carbon dioxide (CO₂) emissions.
3. The overriding control on CO₂ emissions from lowland peatlands is mean water-table depth; for every 10 cm increase in water-table depth, CO₂ emissions increase by around 4 t CO₂-eq ha⁻¹ yr⁻¹ (Figure S1a)
4. Methane (CH₄) emissions from peatlands show an opposing trend, with no emissions recorded when mean water tables were below 25 cm depth, but an increase of around 0.2 t CO₂-eq ha⁻¹ yr⁻¹ for every 1 cm increase in water table above this threshold (Figure S1b). This appears to continue where water tables rise above the surface, making inundated sites potentially major sources of CH₄ emissions.
5. Methane emissions from ditches were highly variable in time and space, but made a significant contribution to overall emissions at some sites, particularly those with larger areas of open water.
6. Aquatic carbon fluxes made a smaller but significant contribution to overall rates of carbon loss, with dissolved organic carbon (DOC) making the largest contribution. Rates of DOC loss were affected by site type and location, tending to be highest from raised bogs in high-rainfall areas, and lowest from fens in low-rainfall areas. We also found evidence that DOC loss in any given location was increased by drainage, consistent with the IPCC methodology.
7. Nitrous oxide (N₂O) emissions contributed an additional 20 to 50% to total GHG emissions at two intensive agricultural sites where fluxes were measured. At unfertilised sites emissions can be assumed minor. Emissions of N₂O from re-wetted former agricultural land remain a knowledge gap.

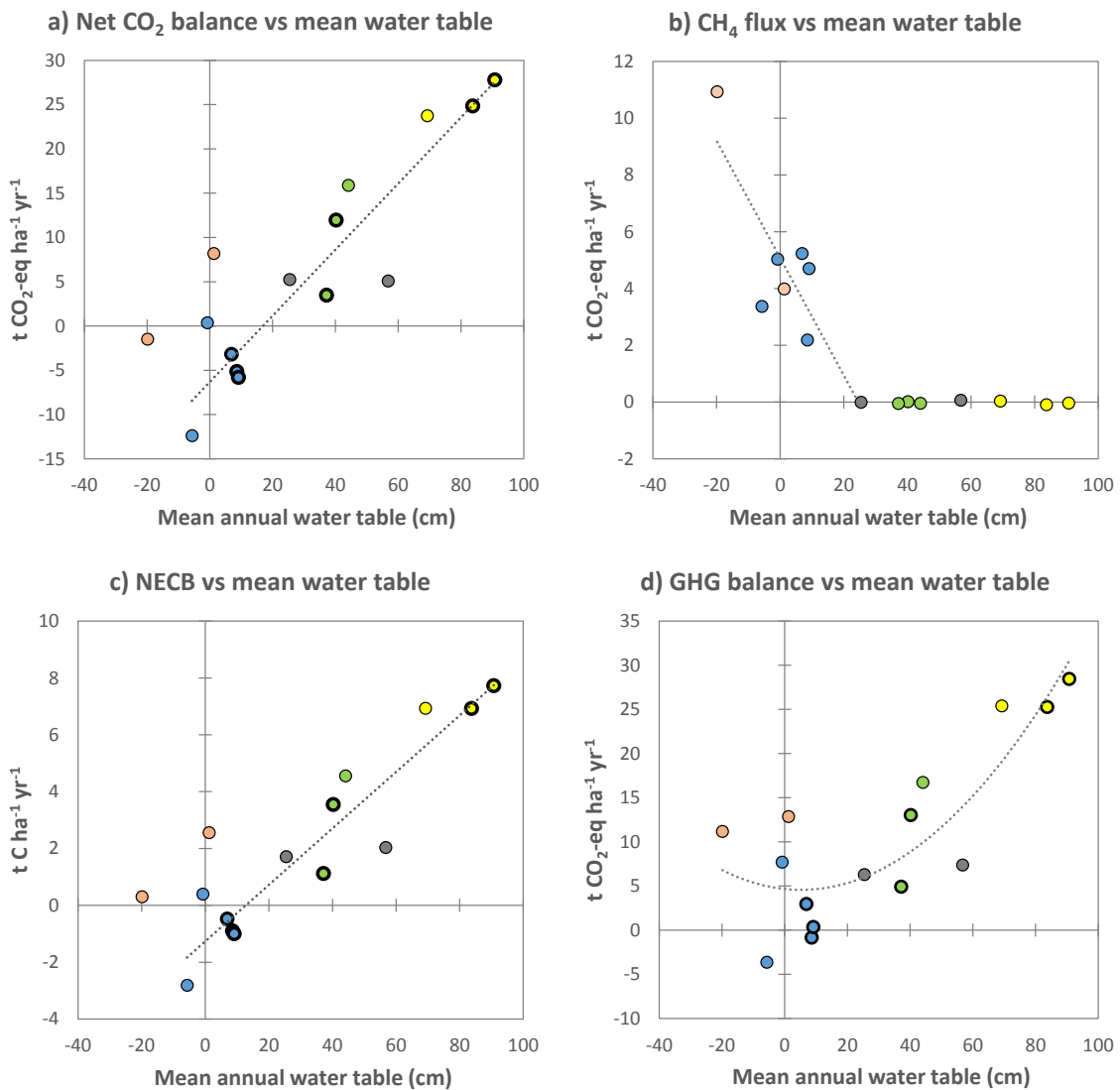


Figure S1. Observed relationships between a) net CO₂ flux; b) terrestrial CH₄ flux, c) Net Ecosystem Carbon Balance and d) overall GHG balance (excluding N₂O) and mean measured water-table depth for all study sites. Sites are colour-coded as: blue = conservation-managed fen; orange = re-wetted raised bog; grey = extraction site; green = grassland; yellow = arable. See Figure 4.5 and accompanying text for further details.

8. The overall GHG balance (based on CO₂, CH₄ and aquatic C fluxes, but not including N₂O) also showed a strong but non-linear relationship with mean water table, with peatlands becoming large net emissions sources when sites were drained (up to a maximum measured rate of around 30 t CO₂-eq ha⁻¹ yr⁻¹ at the most deeply drained sites) or when inundated (over 10 t CO₂-eq ha⁻¹ yr⁻¹ at the most waterlogged site) but approaching GHG-neutral when mean water tables were in the region of 0 to 10 cm below the surface (Figure S1d)
9. The results indicate that subsidence of drained peatlands is significant and ongoing in many areas, and that a significant fraction of this subsidence is due to the oxidative loss of peat. As a result, the remaining 'lifetime' of some drained peatlands may be less than 100 years.

Implications for policy, management and climate-change mitigation:

1. The study confirms that drainage-based conventional agriculture on lowland fen and raised bog peatlands is a major (and probably the largest) source of land-use GHG emissions per unit area in the UK, and thus a high priority for future climate-change mitigation activity. Even if emissions and peat loss from these areas cannot be halted entirely (i.e. truly 'sustainable' agricultural management may be unachievable),

major emissions reductions appear achievable through ‘responsible’ management of agricultural peatlands. Despite falling short of the aspiration for peatlands to make an active contribution to carbon sequestration, as in other areas such as upland blanket bogs, such ‘avoided emissions’ are directly analogous to reducing rates of fossil fuel combustion, and potentially significant. Reducing emissions from cultivated peatlands would almost certainly make a larger contribution to reducing total emissions than blanket bog restoration on a per hectare basis, and could generate significant emissions savings at a UK scale if financial and policy incentives to support improved agricultural management of lowland peatlands were put in place.

2. Given the over-riding influence of mean water-table depth on the rate of CO₂ emission, any mitigation measure that enables productive agricultural activity to continue under higher water tables can be expected to deliver a significant climate mitigation benefit. Since CO₂ losses from arable sites were found to continue through the winter, when fields were typically bare or weed-covered, measures to increase water levels at this time (‘winter re-wetting’) could also deliver significant emissions reductions.
3. We did not find clear evidence that crop selection (e.g. horticultural versus cereal crops) had a major impact on annual CO₂ emissions from agricultural sites, however there was clear evidence that farming operations leading to soil disturbance triggered short-term pulses of CO₂ release, and that any differences in water level management associated with different crops will exert a strong influence on overall CO₂ emissions. Changes in farm management practices that minimise soil disturbance, and/or changes in crop selection (particularly to enable higher water tables to be maintained) may therefore present opportunities to reduce GHG emissions.
4. Measures to limit the use of nitrogen fertilisers are likely to reduce N₂O emissions from agricultural peatlands, with significant benefits for total GHG emissions. Experimental testing of the nitrification inhibitor DCD produced variable results, but suggested that it has the potential to reduce the rate of N₂O production relative to N application in some circumstances, for example if used in combination with organic nitrogen rather than nitrate-based fertilisers.
5. For grasslands, maintenance of shallower water tables should also reduce CO₂ emissions. Reduction or cessation of biomass removal via hay cropping could also, if some of this biomass is incorporated into the peat, reduce rates of carbon loss. However, this approach would need to be tested, as biomass remaining on site could simply decompose *in situ*, or even lead to accelerated decomposition of the peat through ‘priming’.
6. Conservation-managed fens contribute to carbon sequestration within the land-use sector. Although this is partly offset by CH₄ emissions, they nevertheless make the lowest overall contribution to GHG emissions of any lowland peat type studied. There is some evidence to suggest that more productive tall fens, such as managed reedbeds, may confer greater climate mitigation benefits than wetter short fens, perhaps even acting as net GHG sinks under optimal management. However, any potential GHG benefits of changing fen management would need to be set against possible detrimental effects on species diversity resulting from increased dominance of tall fen species.
7. Re-wetting of agricultural land and restoration to natural fen or raised bog vegetation provides clear potential climate mitigation benefits if successfully implemented. However, it may not be enough simply to halt or reduce agricultural activities, or to raise water tables; both re-wetted former extraction sites appeared to be stronger net GHG sources than nearby areas remaining under bare peat, in large part due to CH₄ emissions. Whilst it may be necessary to inundate a site in the short term in order to re-establish key species such as *Sphagnum* mosses, in the longer term it would be preferable in terms of the GHG balance to aim for a mean water table just below the peat surface.
8. The extensive ‘wasted’ peats of East Anglia and elsewhere present potentially large, but as yet poorly quantified, opportunities for climate mitigation. Results from Bakers Fen suggest that conversion from arable to conservation grassland may have helped to stem CO₂ losses, but that more intensive intervention is required if sites such as this are to re-establish their original hydrological and carbon sink function. As for conservation-managed fens, any measures to reduce GHG emissions by re-wetting agricultural land need to be taken in the wider context of the other functions these landscapes perform, in particular their role in food production; consideration should be given to the risk that restoration of

agricultural land in one area could simply displace farming activity, and the associated GHG emissions, to another area. If effectively implemented, however, re-establishment of functioning wetland systems over some parts of the large areas that they once occupied could generate significant climate mitigation benefits, as well as possible co-benefits in terms of biodiversity, flood mitigation and landscape.

Implications for UK GHG emissions inventory reporting:

1. The data collected during this project provide the first complete set of carbon and GHG budgets for UK lowland peatlands, and greatly increase the total number of UK peatland sites with full emissions data. As such they provide a robust basis for the development of Tier 2 emission factors for UK inventory reporting.
2. Measurements from cropland sites support the current (high) IPCC Tier 1 emission factors for CO₂, confirming that drained organic soils under arable and horticultural cultivation are important emission sources.
3. Tier 1 CO₂ emission factors for managed grassland are also supported by data from the project, although results were more consistent with the IPCC's 'shallow-drained' rather than 'deep-drained' category, and a lower Tier 2 EF may be justified for conservation managed grasslands where higher water tables are maintained.
4. Emissions of CO₂ from peat extraction sites appear to be lower than the IPCC's Tier 1 default value, and a smaller Tier 2 value should therefore be developed. This finding is supported by a wider assessment of emissions from peat extraction sites in Ireland and Scotland.
5. The IPCC's Tier 1 CO₂ emission factor for re-wetted fen, which suggests that these act as substantial CO₂ sources, should not be used for UK conservation-managed fens, which results from the project demonstrate are mostly acting as strong CO₂ sinks. A new Tier 2 emission factor for this category should be developed.
6. Data from two re-wetted raised bog sites on former extraction sites suggested that neither has (yet) recovered its natural carbon sink function, resulting in higher GHG emissions than indicated by the IPCC Tier 1 values. However, more data are needed to develop a Tier 2 emission factor for this category.
7. Methane emissions from project sites were broadly compatible with Tier 1 values for the different land-use categories, but with some significant areas of divergence. Emissions from conservation-managed fens, although consistently large, were less than the Tier 1 default value, and a lower Tier 2 value should therefore be developed. Very high CH₄ emissions from one permanently waterlogged site suggest that separate Tier 2 emission factors are needed for re-wetted sites where mean water levels have been restored to near-surface levels, and those that have become inundated.
8. Drainage ditch CH₄ emissions were lower than Tier 1 default values in a number of categories, but given their high spatial and temporal variability current data are not yet sufficient to allow the development of Tier 2 values. Emissions of CH₄ from larger areas of artificial open water ('flooded lands') have the potential to be very large, both in peatlands and elsewhere, but are not currently captured in the UK inventory.
9. The methodology and emission factors developed by the IPCC to account for 'off-site' emissions of CO₂ due to aquatic carbon leaching were broadly supported by the results of the project, and confirm that dissolved organic carbon (DOC) makes up the largest fraction of peat-derived aquatic carbon leaching. However a lower Tier 2 emission factor for DOC should be developed for fen peats in lower rainfall areas.
10. The strong linear relationships observed between emissions of CO₂, CH₄ and mean water table present an opportunity to move towards a simple, empirically-based Tier 3 approach to GHG emissions reporting for UK lowland peats. This would require activity data in the form of either direct measurements of water table, or indirect 'proxy' information such as vegetation composition as an indicator of water-table status.

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1. INTRODUCTION

1.1. Background

1.1.1. Rationale of study

This study was commissioned by Defra in 2011, in order to address an identified gap in knowledge regarding the carbon and greenhouse gas (GHG) balance of lowland peatlands in England and Wales. Lowland peat soils occupy a relatively small proportion of the overall peatland area in England and Wales compared to upland blanket bogs (at a UK level, lowland raised bogs and fens occupy around 15% of the total peat area; JNCC, 2011). However, some lowland peats are very deep, and they therefore store large amounts of carbon (C). Furthermore, lowland peats in England and Wales have historically been subject to a disproportionately intense degree of land-use pressure, as a result of their proximity to populations and, following drainage, their agricultural potential. All industrial peat extraction occurs in lowland areas, and very large areas remain under conventional drained agriculture, representing a significant fraction of the UK's productive arable land. Previous work has suggested that, despite their limited extent, the UK's lowland peatlands may account for around 50% of total GHG emissions from peat (Worrall et al., 2012). However, data to substantiate this suggestion are required.

Until recently, the vast majority of restoration activity, policy attention and research has been focused on upland blanket bogs. While blanket bogs are of great importance to the UK environment and its carbon stores given their great extent, the intensity of land-use pressures and the resulting magnitude of GHG emissions per unit area tends to be greater for modified lowland peatlands. Furthermore, the transferability of GHG flux measurements and wider scientific understanding from blanket bogs to lowland raised bogs and fens is doubtful given the differences between them in terms of vegetation, hydrological function, topography, nutrient status, climate and management. A review for JNCC by Evans et al. (2011) identified the scarcity of C/GHG research sites in lowland peats as a key evidence gap, which included a complete absence of measurement sites on peat under arable cultivation. The difficulty of quantifying C and GHG emissions for lowland peats is increased by their greater heterogeneity in terms of both typology and management, as well as their fragmented nature across England and Wales. Because of their importance for a wide range of ecosystem services (notably provisioning services, but also cultural services such as access to natural landscapes in otherwise often highly developed regions, and regulating services such as flood control in some areas; Bonn et al., 2010), the role of lowland peats in climate regulation must be weighed against these other ecosystem services to enable appropriate management decisions. On the other hand, ongoing peat oxidation under drainage-based agriculture will ultimately, and inevitably, lead to the exhaustion of the peat, further subsidence (increasing pumping costs and flood risk) and declining agricultural yields, thus consideration needs to be given to the long-term as well as the present-day economic value of lowland peat landscapes. This requires accurate estimates of net C and GHG fluxes as a function of peat type and management, at a range of sites sufficient to support upscaling. Reporting of these fluxes within the UK's Land Use, Land Use Change and Forestry (LULUCF) emissions inventory, in line with the recent IPCC Wetland Supplement (IPCC, 2014) requires accurate estimation of emissions factors (EFs) for all modified peatlands.

1.1.2. Types and extent of lowland peat in England and Wales

Lowland peats can be broadly classified into two categories, fens and raised bogs. Fens (peats receiving at least some minerotrophic water input) occupy the largest area, an estimated 958 km² in England (Natural England, 2010) and 209 km² in Wales (Blackstock et al., 2010). Fens are naturally heterogeneous in terms of their morphology, acidity and nutrient levels. In England, many fens have been drained and cultivated for intensive agriculture, giving rise to some of the highest-value arable land in the UK. In the Fens of Lincolnshire, Cambridgeshire and Norfolk, this has resulted in rates of peat loss in the order of 1-2 cm yr⁻¹ (Richardson and Smith, 1977; Burton and Hodgson, 1984) and to the development of large areas of 'wasted' peat, where a large part of the original peat layer has been lost, or mixed with underlying mineral substrate; this wasted peat has been estimated to occupy an area of 1922 km² across England, primarily within East Anglia but also on the fringes of other large peatland areas such as the Somerset Levels (Natural England, 2010). Areas such as the Somerset Levels and Norfolk Broads have been less affected by arable agriculture, but large areas have been modified, to varying degrees, by drainage for grazing. Overall, it is estimated that

39% of fen peat in England is under cultivation, and that 22% of the remaining deep fen peat is under improved grassland. In Wales, topogenous fens (formed due to restricted drainage) are spread widely across the lowlands, often in small patches due to a combination of topographic constraints and past modification. Soligenous fens (sloping sites maintained by lateral water inputs from groundwater or seepage) are mainly located within upland areas. Of a total area of 66 km² of lowland fen in Wales, a considerable proportion remains in a fairly unmodified condition (Blackstock et al., 2010). However, large areas of fen in both England and Wales have been indirectly affected by nutrient enrichment, either from adjacent farmland or via river flooding. Baird et al. (2009) concluded that of the fens, topogenous fens, specifically basin and floodplain fens, are of the greatest overall significance for C storage and GHG fluxes at a UK level.

Raised bogs are rain-fed peats, which develop in poorly drained locations (e.g. fens, floodplains or former lakes) and form shallow domes. They are naturally acid and nutrient poor, and under natural conditions are commonly dominated by peat-forming *Sphagnum* species. Raised bogs occupy an estimated 357 km² in England (Natural England, 2010), although much of this area is actually within the uplands; Baird et al. (2009) estimated the total surviving area of UK lowland raised bog at just 60 km². However much larger areas of lowland raised bog existed in the past, but have been drained and converted to agriculture or other land-use. For example, Bragg et al. (1984) estimated that only 1% of the original 'mosslands' of Northwest England still remain under semi-natural bog vegetation. In England, 16% of the raised bog area is estimated to be affected by extraction, including large areas of raised bog in Northern England, and parts of the Somerset Levels. Around 15% of English raised bogs are estimated to have been converted to improved grassland. Within Wales, lowland raised bogs occupy a current area of around 18 km², less than half of their original estimated extent, with the remainder having been lost through drainage and conversion to grassland (Blackstock et al., 2010). Only 10 km² of the surviving raised bog area is classed as unmodified, and much of this is concentrated within two large raised bog complexes at Cors Fochno and Cors Caron in Mid Wales. Other lowland raised bogs in Wales have been modified by drainage and grazing.

1.1.3 Study design

The study described in this report aimed to address existing data and knowledge gaps through a comprehensive and integrated programme of measurements at a large number of representative sites across multiple lowland peat regions of England and Wales, ranging from conservation-managed fens and raised bogs under semi-natural vegetation through sites under extensive and intensive agricultural grassland management to highly drained and modified arable and peat extraction sites. Although the impacts of restoration and land-use change are of clear scientific and policy relevance, the aim of the current study was to establish a robust set of 'baseline' fluxes for a suite of sites under stable long-term management, as the basis for calculating 'emission factors' for different land-use categories suitable for use in UK emissions reporting. At each site, attempts were made to include all potentially significant components of the C and GHG budgets, including carbon dioxide (CO₂) and methane (CH₄) fluxes between land surface and the atmosphere; emissions from drainage ditches; carbon additions and removals in biomass (e.g. harvested crops); and aquatic carbon fluxes of organic and inorganic carbon. Nitrous oxide (N₂O) emissions were measured at a subset of sites under intensive agricultural management. The application of project-wide measurement, analytical and data processing protocols ensured that results were comparable between sites. Detailed hydrological monitoring, peat core analyses and vegetation surveys were carried out in order to characterise the sites, and to provide explanatory data so that the factors controlling differences in C/GHG balance between sites and over time could be identified. Measurements were made over a period of three years at the majority of sites, to enable year-to-year variations in weather conditions and site management to be taken into account.

1.2. Site overview

In total, fifteen field sites were studied during the project. These sites are located across six of the largest and most important lowland peat complexes of England and Wales, including both raised bogs and fens, and spanning a representative range of land-use from wetland conservation management to intensive agriculture and peat extraction. Ten of the sites are on fen peat, in the East Anglian Fens, Anglesey Fens,

Somerset Levels and Norfolk Broads, of which five sites are under semi-natural fen vegetation, three of which are considered to be in a relatively favourable condition with a low nutrient status (coded LN) and two to be in a less favourable condition with high nutrient status (HN). Two sites are under conservation-managed extensive grassland (EG) and one under intensive grassland (IG). Of the two fen sites under intensive arable cultivation one is located on deep peat (DA) and one on shallow peat (SA). The five remaining sites are on areas of lowland raised bog in the Manchester Mosses and at Thorne Moors, part of the Humberhead Levels. Within each region, one site is located on a peat extraction site (EX) and one on a re-wetted former extraction site (RW). At the Manchester Mosses, an additional deep peat site under arable cultivation was included. The full list of site names and characteristics, together with the site codes which are used throughout this report, are shown in Table 1.1. Site locations are shown in Figure 1.1; more detailed site descriptions and regional maps are provided in Section 2.

Table 1.1. Measurement locations, site types and codes. The first two letters of the site code refer to the study region, and the last two describe the site characteristics (see text above for details).

Region	Peat type	Site type	Site name	Site code
Primary sites				
East Anglian Fens	Fen	Low-nutrient semi-natural	Wicken Fen	EF-LN
	Fen	Extensive grassland	Bakers Fen	EF-EG
	Fen	Deep peat arable	Rosedene Farm	EF-DA
	Fen	Shallow peat arable	Redmere Farm	EF-SA
Manchester Mosses	Raised bog	Re-wetted semi-natural	Astley Moss	MM-RW
	Raised bog	Peat extraction	Little Woolden Moss	MM-EX
	Raised bog	Deep peat arable	Little Woolden Moss	MM-DA
Anglesey Fens	Fen	Low-nutrient semi-natural	Cors Erddreiniog	AF-LN
	Fen	High-nutrient semi-natural	Cors Erddreiniog	AF-HN
Somerset Levels	Fen	Extensive grassland	Tadham Moor	SL-EG
	Fen	Intensive grassland	Tadham Moor	SL-IG
Secondary sites				
Norfolk Broads	Fen	Low-nutrient semi-natural	Sutton Fen	NB-LN
	Fen	High-nutrient semi-natural	Strumpshaw Fen	NB-HN
Thorne Moors	Raised bog	Peat extraction	Thorne Moors	TM-EX
	Raised bog	Re-wetted semi-natural	Thorne Moors	TM-RW

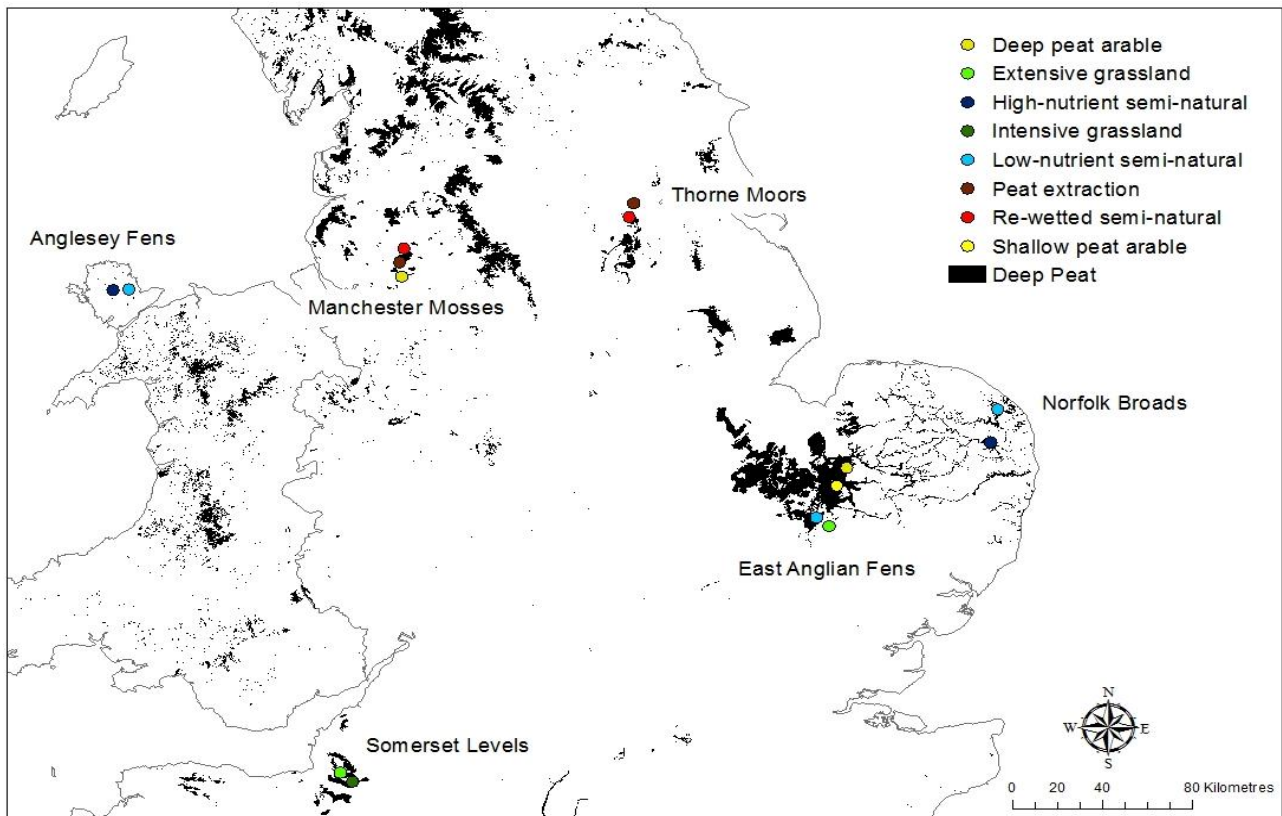


Figure 1.1. Study site locations (note that location markers have been moved where necessary to avoid overlapping symbols) overlaid on the Natural England (NE) peat map for England (derived from British Geological Survey (BGS), Cranfield University and NE mapping data) and the unified peat map of Wales (derived from BGS and NRW mapping data). The peat map shows both upland and lowland peat, as well as areas of former deep peat that have become ‘wasted’ as a result of agricultural activity.

For financial reasons, it was not possible to instrument or measure all sites to the same extent, as a result of which 11 of the sites were designated ‘primary sites’, all of which were monitored over a three year period. At seven of these sites, we were able either to use existing or to install new eddy covariance flux towers, creating what is thought to be a globally unique peatland measurement infrastructure. The remaining four ‘secondary’ sites (in the Norfolk Broads and at Thorne Moors) were monitored for a shorter period and for a more restricted range of measurements. One other originally proposed site, Cors Fochno in Mid Wales, was omitted due to budget restrictions, one consequence of which is that the project lacks a true ‘near-natural’ raised bog reference site. However, Cors Fochno has been the subject of previous flux studies (e.g. Baird et al., 2010; Stamp et al., 2013) and is subject to ongoing flux measurements on both the intact raised bog and adjacent drained grassland (Emma Brown, unpublished data; Manchester Metropolitan University, unpublished data). In addition, other semi-natural UK raised bogs such as Auchencorth Moss and Flanders Moss in Scotland have been the subject of intensive flux studies, and thus provide effective reference data for this project.

Summary physical and chemical characteristics (standardised to the top 50 cm of the peat layer) are shown in Table 1.2. These data highlight the wide range of conditions encapsulated within the measurement network, with mean peat depth ranging from < 0.5 m to around 4 m, bulk density from 0.06 to 1.06 g cm⁻³, carbon content from 22 to 49%, and C/N ratio from 14 to 47 g g⁻¹. Plotting core properties with sites ranked according to bulk density (Figure 1.2) highlights a broad condition gradient among the sites, with the raised bog (TM and MM) sites, Norfolk Broads (NB) and Anglesey Fen (AF) sites having the lowest bulk density, and highest carbon contents, and the four East Anglian Fen (EF) sites having the highest bulk density and low carbon contents. The raised bog sites are – as would be expected – the most nutrient-poor, having high C/N ratios even at the arable MM-DA site. The two Somerset Levels (SL) sites are consistently in the middle of the range, and high C/N ratios here suggest they were at least intermediate between raised bog and fen prior to their drainage and cultivation. The arable sites tend to have higher mineral contents and bulk densities, and

lower carbon contents and C/N ratios, than less intensively managed sites in the same regions, although the deep arable EF-DA site retains a relatively high carbon content and the MM-DA site has a surprisingly high C/N ratio, suggesting that the peat here is still quite nutrient-poor. The EF-EG extensive grassland site clearly shows the legacy of its historical use for arable farming, having the highest bulk density, shallowest peat and lowest carbon content of any site. Similar – although subtler – differences in bulk density, carbon and mineral content are apparent between the two ‘high nutrient’ and ‘low nutrient’ fen sites (AF-LN and AF-HN, NB-LN and NB-HN) supporting the selection of these sites to represent more and less disturbed examples of surviving fen.

Full profile C stock estimates shown in Table 1.2 also show a wide range of variation. The lowest (although still substantial) C stocks were measured in the shallow peat EF-EG, and the current or former extraction sites MM-EX and TM-RW. Surprisingly, the arable EF-DA site had the highest estimated C stock, although this estimate is uncertain as the profile data show evidence of intermixing of organic and mineral soils. The intact fen sites have C stocks ranging from 1,370 to 2,820 t C ha⁻¹ yr⁻¹.

More detailed descriptions of individual sites are provided in Section 2.

Table 1.2. Summary physical and chemical characteristics of measurement sites.

Site	Peat depth cm	Mean peat properties, 0-50 cm				Full profile	
		Bulk density g cm ⁻³	pH	Mineral %	C %	C/N g g ⁻¹	C stock t C ha ⁻¹
EF-LN	380	0.37	7.54	52.2	32.0	15.8	2,820
EF-EG	< 50	1.06	7.10	65.7	22.3	19.7	610
EF-SA	75	0.62	6.84	60.9	30.8	16.4	1,500
EF-DA	200	0.50	6.70	26.4	43.6	15.0	3,880
MM-RW	380	0.14	2.66	5.4	47.1	32.0	2,530
MM-EX	180	0.24	2.96	7.5	50.0	35.3	810
MM-DA	100	0.32	5.80	31.9	36.2	43.9	1,290
AF-LN	280	0.17	5.48	13.4	45.3	17.5	1,370
AF-HN	315	0.11	4.99	20.2	41.8	14.7	2,170
SL-EG	160	0.34	5.62	25.4	39.6	30.4	1,650
SL-IG	200	0.27	4.47	15.5	42.7	28.6	1,980
NB-HN	> 300	0.12	ND	41.1	32.1	13.4	1,830
NB-LN	> 300	0.07	ND	25.9	43.5	14.5	1,900
TM-RW	370	0.06	ND	ND	49.4	46.5	940

Note that some peat depths are based on only 2-3 cores, and therefore should only be treated as indicative (additional depth probe measurements were made at other sites). Because it was not possible to carry out all analyses on all sampled horizons, measurements of carbon and mineral content may not be completely comparable in all cases. Full profile C stock estimates are based on measured %C and bulk density values to the maximum coring depth, which at shallower sites was the base of the peat. At deeper sites, the C stock of lower layers was based on mean %C and bulk density values from overlying horizons. Measurements of pH were not made on cores collected from ‘secondary’ sites.

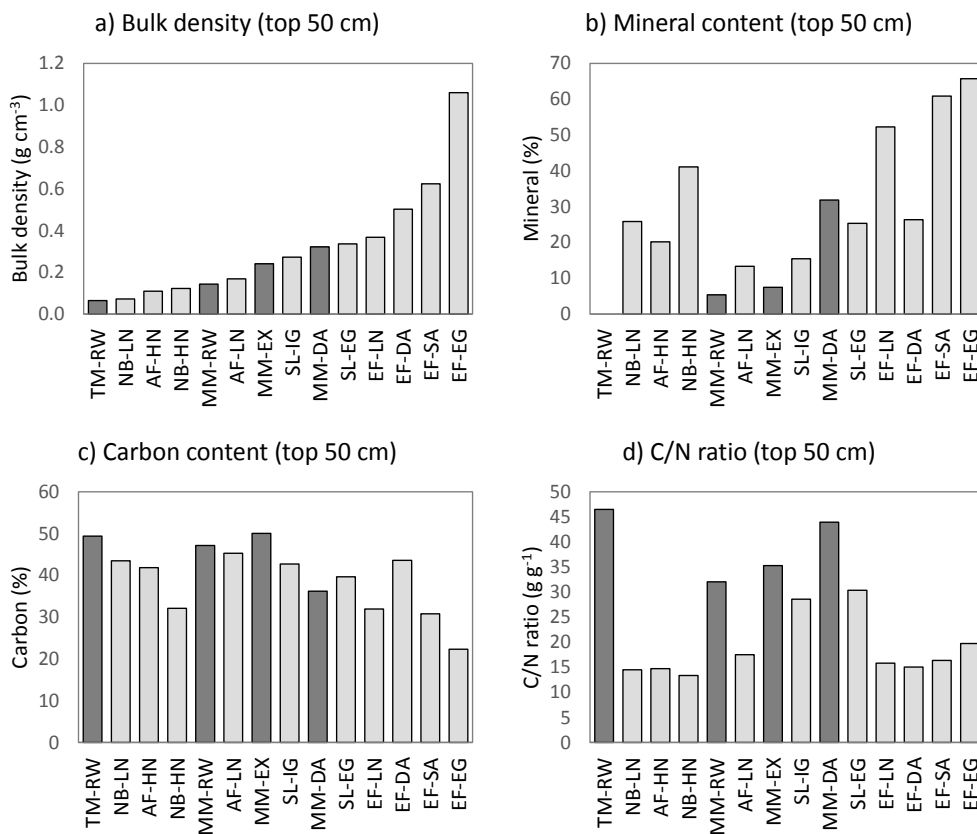


Figure 1.2. Characteristics of near-surface (0-50 cm) peat at all study sites with core data, ranked from low to high bulk density. Sites on raised bog peat are shown in dark grey, sites on fen peat in light grey.

1.3. Method summary

1.3.1. Meteorological measurements

Automatic weather stations (AWS) were installed at or close to all of the main lowland peatland measurement sites. In the majority of cases, AWS were operated as part of the project (e.g. alongside eddy covariance instrumentation) although in some cases meteorological data were obtained from external sources (e.g. MM and TM sites). All AWS were equipped to continuously monitor the main meteorological variables, namely incoming shortwave radiation (or photosynthetically active radiation), air temperature and relative humidity, wind speed and direction, and precipitation. At some sites, additional meteorological (e.g. net radiation and its components) and soil physics (e.g. soil temperature, soil moisture, soil heat flux) sensors were also installed. As far as practicable, air temperature and relative humidity were measured at 1.5 to 2 m above the land surface. Wind speed and direction were measured at varying heights, depending on site conditions and the maximum height of the vegetation (e.g. crop) canopy present. Precipitation was monitored at ground level using tipping bucket rain gauges installed in unobstructed areas. The frequency at which meteorological measurements were recorded varied across sites, ranging from fifteen minutes (e.g. EF-SA and SL-EG) to thirty minutes (most sites) to one hour (TM and MM sites). In this report, meteorological measurements are reported as monthly and annual values. In the case of mean air temperature, monthly values have been calculated as the mean of all data logged over a given month or year. Monthly and annual temperatures are reported in degrees Celsius (°C). Incoming short wave radiation data were measured in units of Watts per meter square ($W m^{-2}$) but are presented in mega Joules per meter square month ($MJ m^{-2} month^{-1}$). Monthly and annual precipitation sums are provided in mm per time period (e.g. $mm month^{-1}$).

1.3.2. Hydrological measurements

Hydrological measurements were made for a number of reasons: i) in order to calculate the amount of water flowing out of a site and (in combination with aquatic carbon concentration measurements) to calculate the total aquatic carbon loss from the site; ii) to provide contextual data, such as water-table depths, in order to help explain variations in carbon fluxes (gaseous and aquatic) between sites; and iii) to help explain (and hence model) variations in gaseous carbon fluxes over time at each site. In contrast to upland blanket bogs, which are characterised by large water fluxes, measurable topographic gradients and gaugeable outflows, lowland peats are typically rather flat, often with no clear single outflow. Most sites have highly modified drainage patterns with interconnecting ditches that are used to transfer water off or in some cases onto the site, as well as water level control structures such as sluices or pumped drainage. Direct gauging of flows in these systems is therefore generally extremely challenging. However, by taking a water mass balance approach (e.g. Gilman, 1994) at these sites we are still able to determine the aquatic losses. In simple terms, the water output from a site will equal the sum of water input plus or minus any change in storage, i.e:

$$P_{\text{net}} + Q_{\text{in}} + G_{\text{in}} = ET + Q_{\text{out}} + G_{\text{out}} + \Delta s$$

where P_{net} is precipitation which reaches the ground, Q_{in} and Q_{out} are surface flows in and out, G_{in} and G_{out} are groundwater flows in and out, ET is evapotranspiration, and Δs is change in water storage.

P_{net} was measured at all sites. ET is continuously and directly measured by the flux towers at all four EF sites, the two AF sites and SL-EG (which was also considered representative of the nearby SL-IG). For the MM sites no direct measurements of ET were possible due to the theft and vandalism of the flux tower that was briefly deployed at MM-EX. As a result, two alternative methods were used to estimate potential evapotranspiration (PET) using the Penman-Monteith (Allen et al., 1998) and Thornthwaite (1948) methods.

Changes in storage, Δs , were estimated via the automated dipwells at all sites, which recorded the height of the water-table every 15 minutes. While these dipwell records provide the water-table change over time they do not directly provide the change in water storage because the specific yield for each site is also required, i.e. how much the water-table changes per unit input (rainfall) assuming no other water losses from the peat. A number of representative specific yield values were taken from the literature; for Wicken Fen (EF-LN) a specific yield of 0.12 was used based on previous measurements at the site, elsewhere a default peat specific yield of 0.2 was used.

Surface water discharges into most sites (Q_{in}) were considered to be negligible. For a number of sites, attempts were made to measure Q_{out} via automated ditch or stream level readings; however the complex ditch management and subdued hydraulic gradients made standard approaches such as the use of v-notch weirs virtually impossible. Two sites (MM-RW and EF-LN) are surrounded by bunds which restrict surface water flows from the sites. Groundwater flows into and out of the sites were not measured directly, however the underlying geology and drainage configuration of most sites limits any interactions with groundwater. Where groundwater flows were potentially significant (notably at AF-LN, which has a small external catchment upstream of the fen itself) piezometer nests were installed, and manual measurements of saturated hydraulic conductivity (K) were used to determine water flux rates.

Further details of water flux calculations for each site are given in Appendix 1.

1.3.3. Eddy covariance gas flux measurements

Eddy covariance flux towers were operated at seven of the project sites, utilising a number of existing flux towers operated by CEH and the University of Leicester, and adding additional sites during the project with equipment support from Bangor University, Durham University and a NERC Urgency grant awarded the University of Leicester. The spatial density of eddy covariance (EC) measurement sites on peat achieved during the project is believed to be globally unique.

EC measurements are based on sampling the vertical component of atmospheric turbulence and the concentration(s) of atmospheric scalars of interest (e.g. water vapour, CO_2 , CH_4) using fast response instrumentation (typically 20 Hz). In this project, the three components of atmospheric turbulence were measured using either a Campbell Scientific Inc. (Logan, Utah, USA) CSAT3 (at AF-HN, AF-LN, EF-DA, EF-EG) or

a Gill Instruments Ltd (Lymington, UK) Solent R3 (EF-SA, EF-LN, SL-EG) sonic anemometer. All EC network sites employed the same fast response, open-path gas analyser (LI-7500 or LI-500A, LI-COR Biosciences, Lincoln, Nebraska, US) for measurements of atmospheric concentrations of water vapour and CO₂. Fast (20 Hz) data were logged on CR3000 Measurement and Control Systems (Campbell Scientific Inc. Logan, Utah, USA) at most sites, and using LI7550 data loggers (LI-COR Biosciences, Lincoln, Nebraska, USA) at EF-EG and EF-LN. Data were submitted to a central project database at CEH Wallingford.

High frequency (20 Hz) EC data were processed using EddyPRO[®] flux Calculation Software (LI-COR Biosciences, Lincoln, Nebraska, USA) following widely adopted flux calculation and correction protocols to ensure consistency across sites. All data were quality checked using standardised tests for outlier removal (Papale et al., 2006), technical quality (Foken et al., 2004) and spatial representativeness (Kormann and Meixner, 2001). The measured net ecosystem CO₂ exchange (NEE) was partitioned into estimates of the component fluxes of gross primary production (GPP) and total ecosystem respiration (ER) using the standardised method of the global Fluxnet community (Reichstein et al., 2005; Reichstein et al., 2016). This partitioning enables a more complete understanding of the processes underlying the net surface-atmosphere CO₂ exchange. Uncertainties in daily and annual CO₂ fluxes were calculated for the EC data based on random sampling errors (Finklinstein & Simms, 2001) and uncertainties introduced by data gap-filling (Reichstein et al., 2005; Reichstein et al., 2016). In this method, a standard deviation is derived for each measured and gap-filled data point. Uncertainties in daily estimates were calculated as the sum of squares of the standard deviations. Annual uncertainties were calculated as the cumulative sum of the daily uncertainties.

1.3.4. Static chamber gas flux measurements

At all of the sites closed chambers were used to measure fluxes of CH₄ and CO₂ at multiple (usually six) locations. Details of the operation of such chambers may be found in Denmead (2008) and Baird et al. (2009). Chambers were specially constructed for the project and used 60 x 60 cm collars onto which the chamber could be placed. These collars were left permanently in place wherever possible to minimise disturbance, but for managed arable sites it was necessary to remove and re-install the collars between measurements to avoid them being damaged during routine farming activities. For tall fen vegetation we used 'stackable chambers' with extensions that allowed the chamber to be elongated. Initially, separate tests for CH₄ flux and CO₂ flux were conducted. For CO₂, both light and dark chamber tests were used in which within-chamber CO₂ concentrations were measured using a portable infrared gas analyser (IRGA). Light chamber tests provide a measure of net ecosystem CO₂ exchange (NEE), while dark tests provide an estimate of ecosystem respiration (ER) (see Green et al., 2016). From NEE and ER the gross primary productivity (GPP) can be calculated. For CH₄ fluxes, dark tests were used and gas samples removed from the chamber for later analysis using gas chromatography. Later in the project, most teams had access to a Los Gatos Inc. Ultra-portable Greenhouse Gas Analyser which measures simultaneously, in field, both CO₂ and CH₄ concentrations; therefore, separate chamber tests for the two gases were not needed.

Fluxes were estimated for each chamber test using the same protocols and spreadsheets as used on the Defra SP1202 project (Green et al., 2016). The calculations in the spreadsheet convert increases or decreases in the concentration of the target gas within the chamber into a flux. Formally, the flux is expressed as a flux density: the mass of gas released or taken up by a unit area of peatland in a unit of time. These flux estimates are 'snapshot' measurements and do not provide information on the overall flux over longer time periods, such as a year, which is needed when compiling carbon inventories and emissions factors for sites. To obtain annualised fluxes we used a variety of approaches. For CH₄, we typically took a weighted mean of measured values, taking into account the variations in sampling frequency through the year. For CO₂ we used various models to simulate ER and GPP. We followed a very similar approach to that used in Defra SP1202 (see Green et al. (2016) for more details). Essentially, flux measurements are used to build empirical models in which fluxes are related to environmental variables such as water-table position, air or soil temperature, and solar irradiance. These models were then used to scale the instantaneous flux measurements up to annual integrated values using the high-frequency measurements of these environmental variables, as described in Section 1.3.1.

All flux chamber measurements, flux calculations and flux models were developed using standard project protocols that were adopted by each site group, to maximise consistency between sites. These protocols follow best practice from the research literature and are described in more detail in Appendix 2.

1.3.5. Aquatic carbon fluxes

Concentrations of all significant components of the aquatic carbon flux were measured at each site. Samples were collected on the same dates as the static chamber measurements. Samples were collected from ditches draining the sites where present. At sites where ditches were not present (MM-RW) or contained a mixture of water from different sources (MM-DA) water samples were collected from dipwells. Water samples were sampled for dissolved organic carbon (DOC) and dissolved inorganic carbon (DIC), and (where samples were collected from ditches) for particulate organic carbon (POC). Dissolved carbon dioxide ($\text{CO}_{2(g)}$), methane ($\text{CH}_{4(g)}$) and nitrous oxide ($\text{N}_2\text{O}_{(g)}$) were analysed on headspace samples. Details of analytical methods used are given in Appendix 1.

Routine water quality sampling data were combined with the hydrological budgets above to produce aquatic carbon budgets for DOC, DIC and POC. Where more than one sample was collected per month mean monthly averages were calculated. However, where there were multiple samples in one month and no sample was collected in the previous or next month 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 month. Where the monthly hydrological budget indicated that no water was lost from the site, the monthly carbon flux was considered to be zero. Fluxes were expressed in g C m^{-2} for each month. To estimate mean annual aquatic carbon 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. This approach overcame problems with missing data from some months when concentration and/or water flux data were not available, and avoided seasonal bias in flux calculations.

1.3.6. Site characterisation

At least two peat cores were collected from all 'primary' sites, and subsequently from one of the Thorne Moors sites (TM-RW) and both Norfolk Broad sites, according to the same protocols. Each core was subdivided into 5-10 cm depth increments, and subsamples were analysed for bulk density, pH, carbon, nitrogen and mineral (ash) content. The data were used to estimate whole-profile carbon stocks. Peat depth was measured at additional points at each site using a probe, and where necessary the core data (carbon and bulk density) were extrapolated to deeper horizons to provide a full depth carbon stock estimate.

Vegetation surveys were carried out at sites under semi-natural vegetation by a vegetation surveyor, Phil Eades. Surveys were carried out in 2013 and 2015, to establish whether any successional changes in vegetation composition had occurred at any of the sites, or at specific flux measurement locations. All measurement sites were assigned to a category in the National Vegetation Classification (NVC; Rodwell *et al.*, 2000), species lists created, and plant communities mapped across each site. These surveys are described in full in Appendix 3. At sites under agricultural management, carbon removals in harvested biomass were quantified through a combination of direct measurements, information from farm managers and literature data. In some cases it was also possible to estimate biomass inputs associated with the use of plug plants in horticulture.

2. FIELD MEASUREMENTS

2.1. East Anglian Fens

The East Anglian Fens (Figure 2.1.1) are the largest and most intensively modified area of lowland peat in the UK. The Fens were first subject to large-scale drainage in the 17th century, which became more extensive and effective from the late 18th century following the installation of industrial pumped drainage systems. Drainage has created a large area of highly productive farmland, which remains one of the UK's most important agricultural regions, much of which is used for high-grade horticulture and arable production. However, drainage also triggered widespread and ongoing peat 'wastage' due to a combination of compaction and oxidation. Subsidence at the Holme Post, which was installed in 1851, following drainage of the adjacent Whittlesey Mere, is now in the region of 4 m (Waltham, 2000). Given that the Holme Post stands in an area of woodland, subsidence in areas under active arable cultivation is likely to have been greater, probably exceeding 5 m. Based on a number of previous studies, Holman (2009) estimated that the mean ongoing rate of drained peat subsidence in the region was 1.5 cm yr⁻¹, with the highest rates associated with areas of remaining deep peat and greatest degree of drainage. The original extent of peat in the Fens is likely to have been in the region of 150,000 ha, much of it 5 m or more deep, but as a result of wastage this area was estimated to have been reduced to 24,000 ha of peat > 40 cm by the 1980s (Burton and Hodgson, 1987), of which less than half was more than 1 m deep. Extrapolating from these data based on measured subsidence rates, Holman (2009) estimated that just 16,500 ha of peat > 40 cm depth remain. Thus the majority of the original peat area of the fens has now become 'wasted' (Figure 2.1.1), such that the original peat has been reduced to a thin (and less agriculturally productive) residual layer of mixed organic and mineral soil, known locally as 'skirtland'. As a consequence of peat subsidence, large areas of the Fens are now below sea-level, and are protected from inundation by a network of ditches, pumps, embanked channels and sluices. Some small areas of semi-natural fen vegetation remain, but have to be maintained through active water management. Other areas of deep peat survive under grassland in 'washlands', strips of deep peat between or adjacent to rivers that are seasonally flooded as part of the regional water management. Some areas of cultivated deep or wasted peat have undergone (or are undergoing) re-wetting and restoration to fen or extensive grassland vegetation.

Holman (2009) estimated that ongoing oxidation of East Anglian Fen peat is generating a total emission of 0.4 Tg CO₂-C yr⁻¹, equivalent to around 0.3% of the UK's total CO₂ emissions. This equates, however, to an emission per unit area of remaining peat of 84 t CO₂ ha⁻¹ yr⁻¹, considerably higher than the IPCC's Tier 1 default emission factor for temperate peat under cropland of 29 t CO₂ ha⁻¹ yr⁻¹. On the other hand, the estimate of Holman (2009) does not, as the author notes, include any estimate of continuing CO₂ emissions from the extensive area of 'wasted' peat. Initial inclusion of lowland peat drainage in the UK LULUCF inventory, based on emission factors of 47 t CO₂ ha⁻¹ yr⁻¹ for deep peat and 4 t CO₂ ha⁻¹ yr⁻¹ for thin peat, suggest that cultivated peats in the East Anglian Fens are the UK's largest land-use related CO₂ emissions source (Hallsworth and Moxley, 2013).

Since the East Anglian sites are fairly widely separated, the established automatic weather station (AWS) at EF-LN was augmented by temperature loggers and rain gauges (as well as eddy covariance meteorological instrumentation) at EF-DA and EF-SA. The remaining site (EF-EG) is within 1 km of EF-LN, and meteorological data from EF-LN were therefore considered representative for both locations.

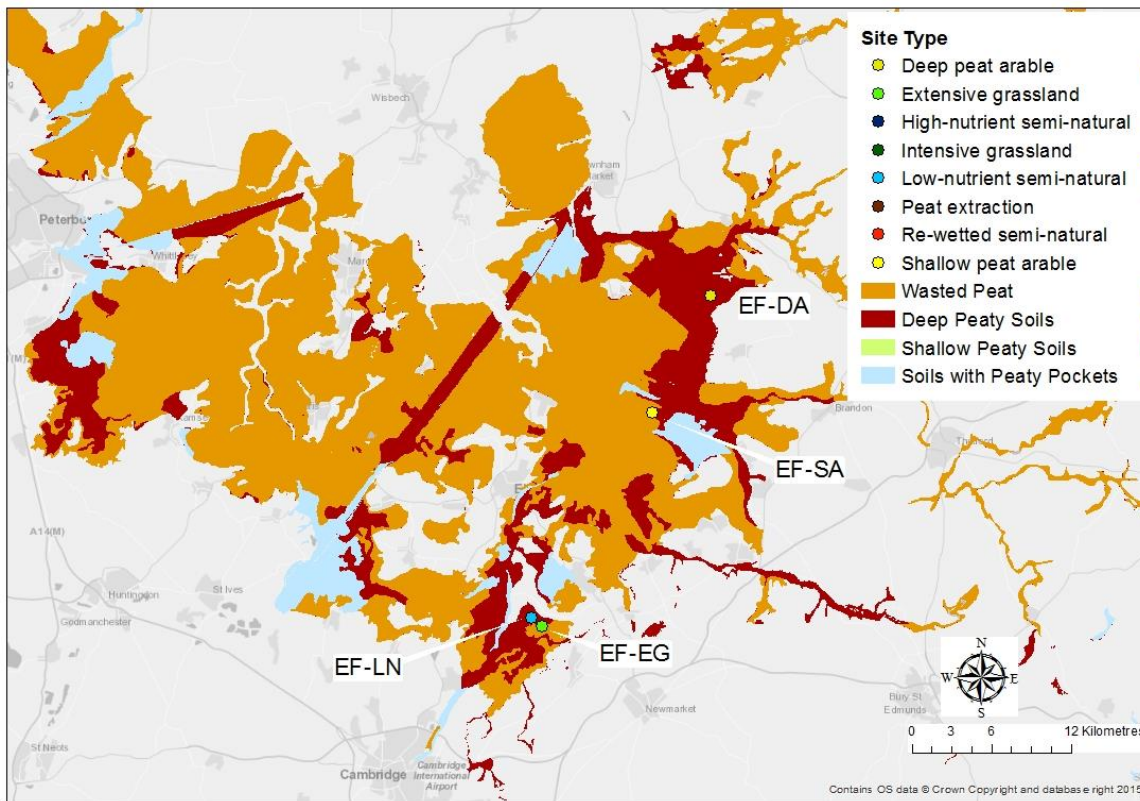


Figure 2.1.1. East Anglian Fens study area

2.1.1. Wicken Sedge Fen – low nutrient fen (EF-LN)

Part of the National Trust Wicken Fen Nature Reserve, Wicken Sedge Fen is the largest of the only four surviving fragments of the East Anglia Fens remaining under natural tall fen vegetation. The fen, which is managed by the National Trust, has an exceptionally rich flora and fauna, is a SSSI and conserves habitat and rare species. The sedge is harvested regularly in a rotational cropping scheme every 3-4 years, continuing the traditional low impact use. Although sedge harvesting represents a net carbon offtake from the fen, no harvesting took place in the vicinity of the flux tower during the project period, so biomass fluxes were not included in carbon balance calculations. The site is under shallow water-table management, with protective bunds to maintain water levels (the surrounding agricultural land is at a significantly lower elevation) and a network of ditches that are used to transport water into the fen (McCartney et al., 2001). As the fen is now isolated from the surrounding landscape, mineral-rich water is pumped onto the site from an adjacent channel (Monk's Lode). However the site remains vulnerable to significant water-table drawdown during dry summer periods. Wicken Fen has been the subject of detailed ecological research for over a century.

The study area used for this project has been mapped as National Vegetation Classification (NVC) Class 24c, tall-herb fen community. Dominant species are saw sedge *Cladium mariscus* and common reed *Phragmites australis*, with abundant reed canary grass, *Phalaris arundinacea*, and some purple small-reed *Calamagrostis canescens* (for a more detailed botanical description of this and all other semi-natural sites see Appendix 3). Peat core data (Figure 2.1.2) show that peat at the site remains deep (around 4 m) but suggest that bulk density is quite high relative to other natural sites, particularly towards the surface (*cf.* Anglesey Fens in Table 1.2 and Figure 1.2). The upper peat layer also appears to be nitrogen-enriched, which may be due to nutrient-rich river water having been transferred onto the site via the ditch network. The mean pH of near-surface peat (7.5) is the highest of any of the sites studied, and it is possible that the very high measured mineral content in the upper 10 cm is due to calcium carbonate precipitation.

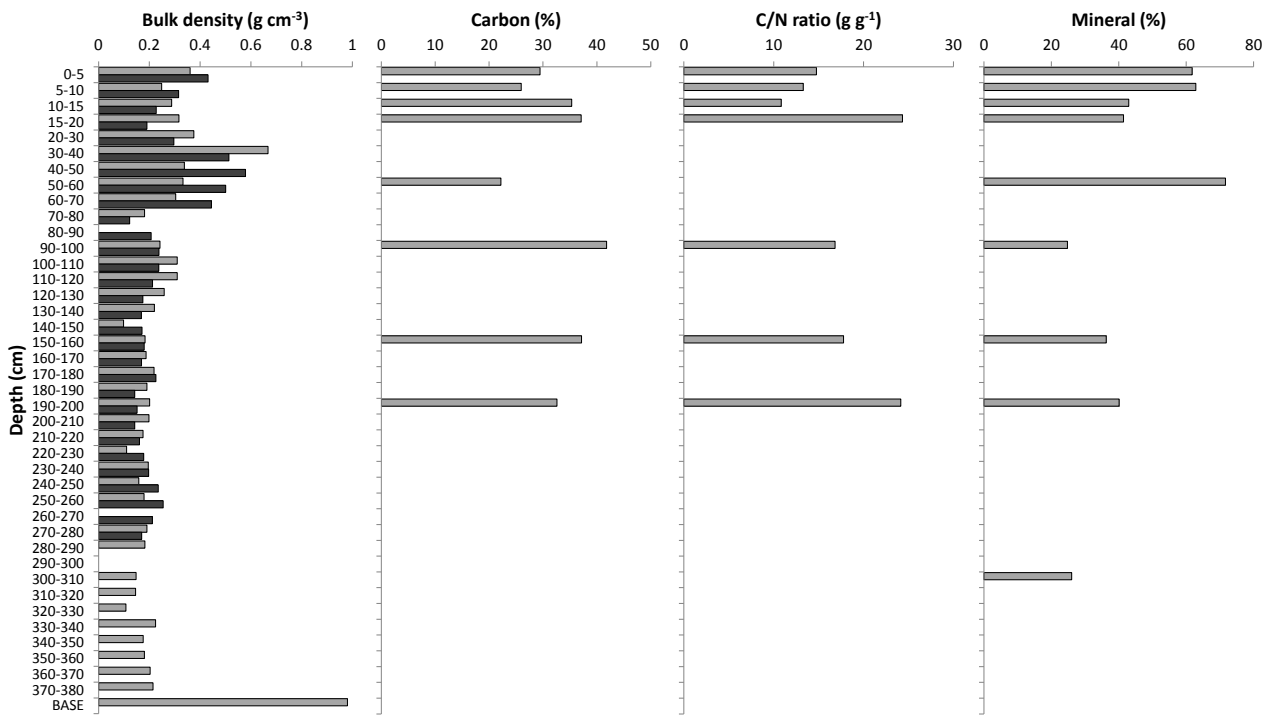


Figure 2.1.2. Peat core data, EF-LN. Data from two cores analysed from the site are represented by dark and light grey-shaded bars.

2.1.1.1. Meteorology

Figure 2.1.3 shows the seasonal change in monthly values of the main meteorological variables (incoming short wave radiation, air temperature and precipitation) measured at EF-LN for each month that measurements are available. Missing data from the EF-LN site were filled using data from the nearby EF-EG. Missing incoming shortwave radiation (SW_{in}) data were filled using measurements made at EF-DA. Wind flow is predominately from the southwest at this location (Figure 2.1.4), although the site experiences wind from all compass directions over the course of the year.

Total monthly SW_{in} (top panel in Figure 2.1.3) peaked in June (2014) or July (2013 and 2015) with monthly minima during December (all years). The maximum total monthly SW_{in} recorded for the 2013 to 2015 period was $359 \text{ kW m}^{-2} \text{ month}^{-1}$ in July 2013. Mean annual air temperature was $9.3 \text{ }^\circ\text{C}$ in 2013, $10.9 \text{ }^\circ\text{C}$ in 2014 and $10.3 \text{ }^\circ\text{C}$ in 2015. Monthly mean air temperatures (middle panel in Figure 2.1.3) between January and June were lowest during 2013 and highest during 2014. Maximum monthly temperatures were $16\text{--}18 \text{ }^\circ\text{C}$, in July–August, while monthly minima were $2.5 \text{ }^\circ\text{C}$ in March 2013, $4.9 \text{ }^\circ\text{C}$ in December 2014, and $3.6 \text{ }^\circ\text{C}$ in March 2015. The early autumn period (September and October) was warmest in 2014 and coolest during 2015. December 2015 was the warmest December month in the instrumental record, with a mean temperature of $10.2 \text{ }^\circ\text{C}$.

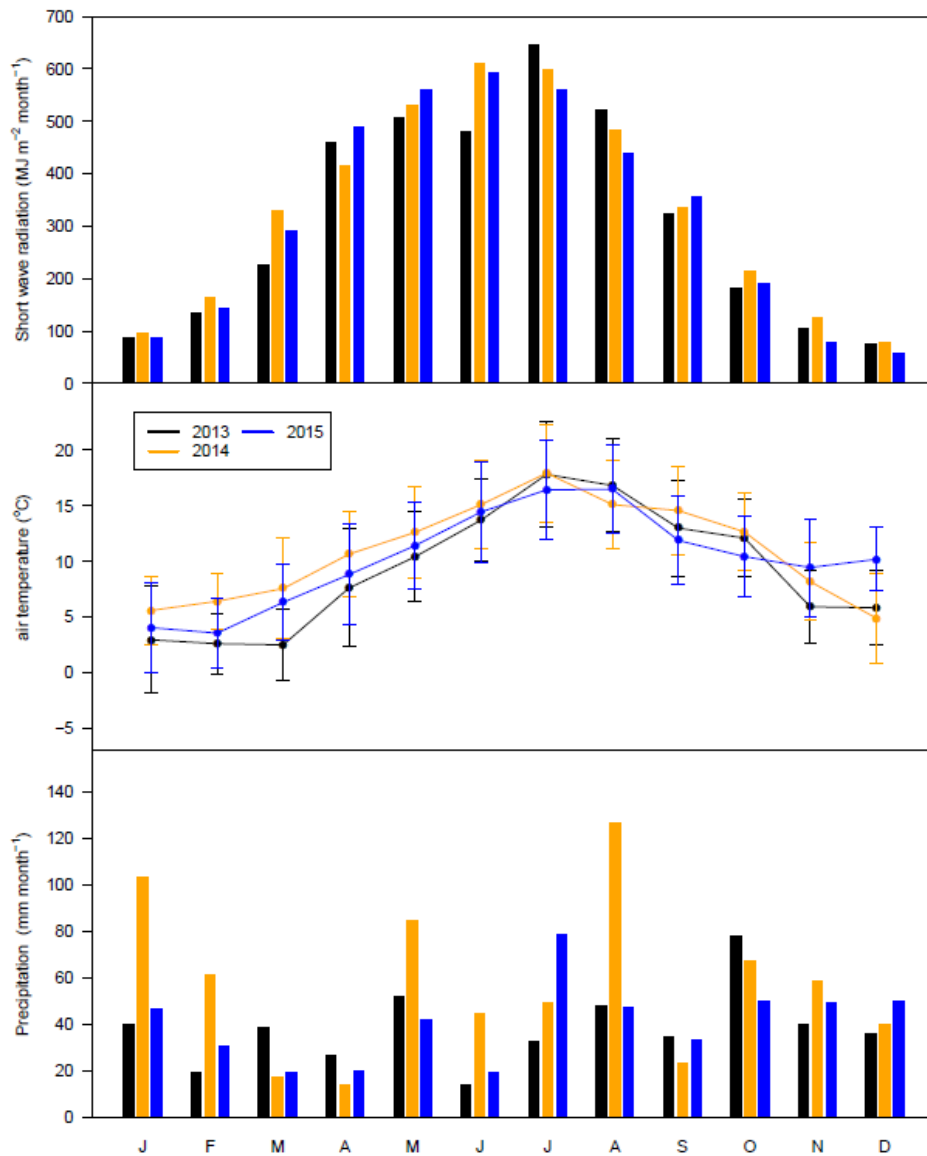


Figure 2.1.3. Total monthly incoming solar radiation (top) mean monthly air temperature (middle) and total monthly precipitation (lower) for the EF-LN and EF-EG (Wicken Fen National Nature Reserve) sites. Error bars on the temperature plot show one standard deviation of the mean.

A number of periods of missing data at the Wicken Fen sites preclude the calculation of accurate annual and monthly rainfall totals. The most reliable (SP1210) rain gauge in the region is located at EF-DA which recorded annual precipitation sums of 648 mm yr⁻¹ in 2013, 765 mm yr⁻¹ in 2014 and 641 mm yr⁻¹ in 2015 (see below). In the lower panel of Figure 2.1.3, monthly precipitation sums are shown for the Met Office NIAB site in Cambridge (approximately 15 km from Wicken Fen). These data show particularly wet periods in January, May and August 2014, whereas conditions in 2013 were particularly dry during the June to August growing season.

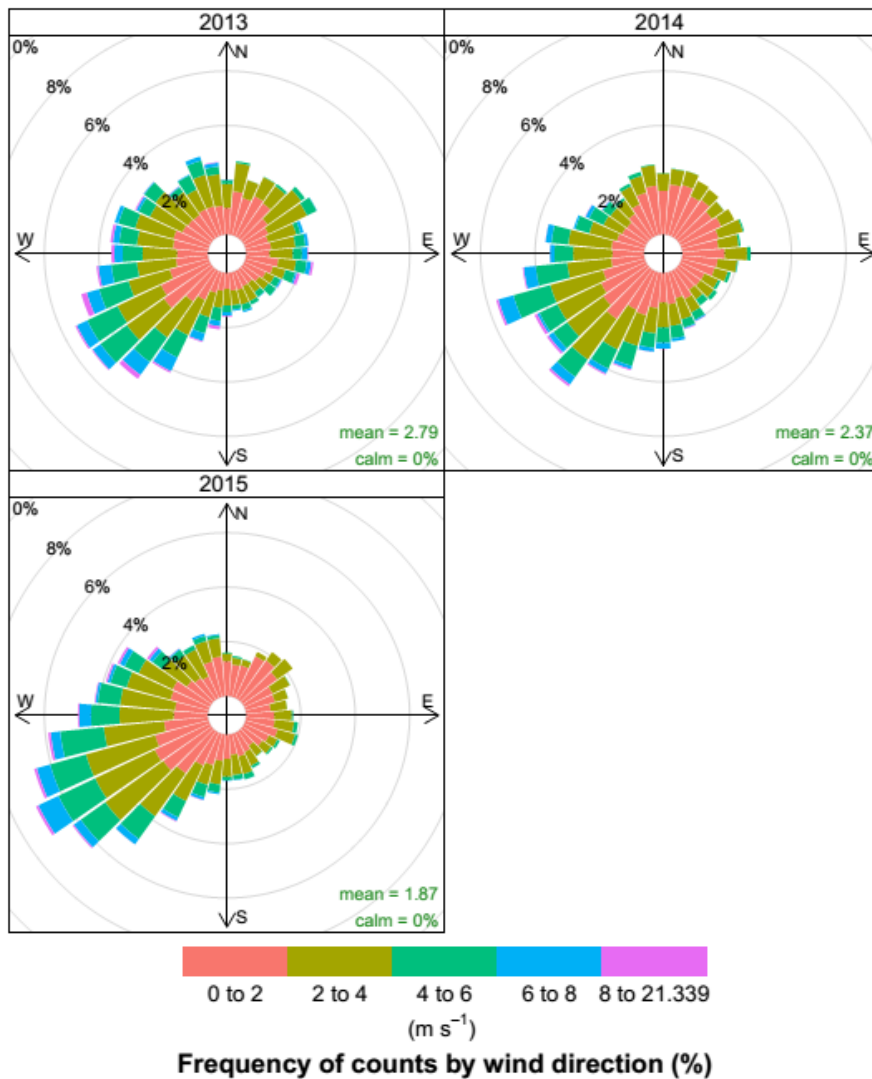


Figure 2.1.4. Wind rose plots showing wind direction and wind speed at the EF-EG (Wicken Fen National Nature Reserve) site.

2.1.1.2. Hydrology

Automated water-table logger data for the EF-LN site (Figure 2.1.5) were provided by the National Trust, based on loggers installed in existing dipwells (permission to install additional dipwells at the study site was not granted, in order to limit disturbance to the peat, therefore a comparison with manual water-table readings was not possible). The two loggers show very similar temporal variations, but very large between-year differences. The start of 2013 was characterised by an extended period of inundation (leading to delays in the initiation of some other measurements, as site access was restricted), which was followed by major drawdown of the water-table to approximately 80 cm depth by early autumn. This degree of drawdown was far greater than anything observed at any of the other intact (AF and NB) fen sites, and reflects the difficulty of maintaining water levels in the hydrologically isolated Wicken Fen. In 2014 and 2015, hydrological variations were less pronounced, with shorter periods of winter inundation and less severe summer drawdown; in 2014, water-tables did not fall below 30 cm at any point, and in 2015 this occurred only during a relatively short period during July.

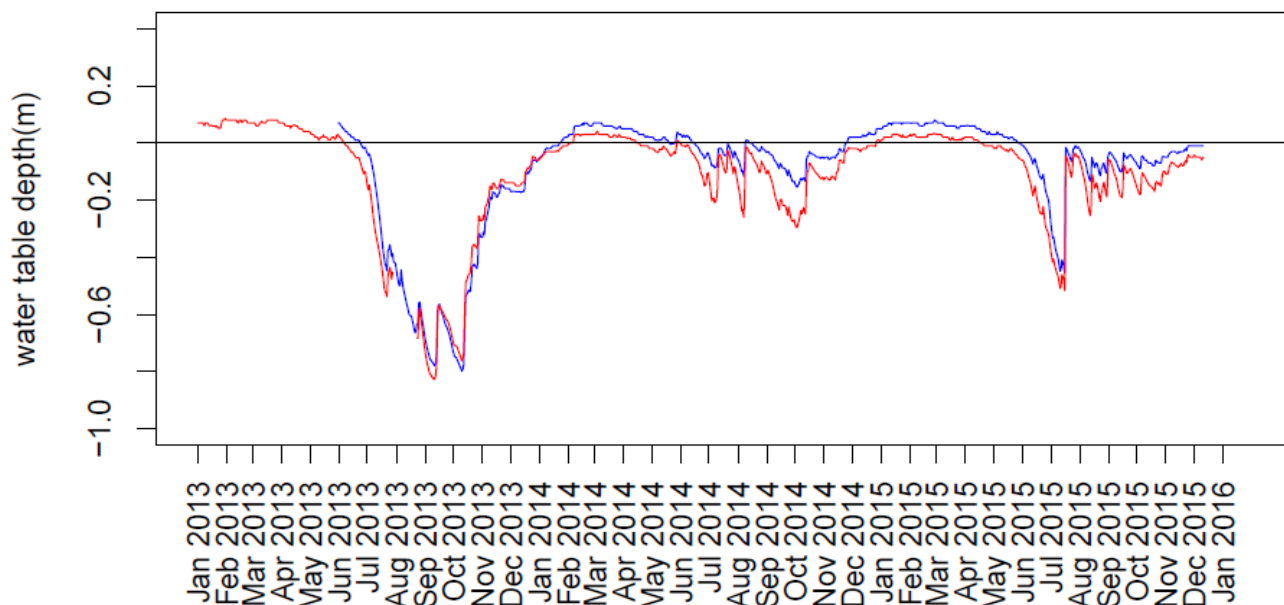


Figure 2.1.5 Continuously monitored water-table data from two dipwells at EF-LN.

Given the nature of site management and restrictions on hydrological instrumentation, few data were available from which to calculate a hydrological budget for the site. However, McCartney and de la Hera (2004) previously carried out detailed analysis of the hydrology of the site, and concluded that the main control over water-table depths was the balance between rainfall and evapotranspiration, with other losses being minor. For the current project, evapotranspiration (ET) data were taken from the flux tower at EF-LN, with gap-filling where necessary (for summer 2014) based on the relationship between ET at EF-LN and the nearby EF-EG flux tower. Water inputs to the site were known. The overall hydrological balance of the site (Figure 2.1.6) indicates that water discharge from the site is limited to the winter period, from October-November to February-March, with evapotranspiration dominating at all other times. Estimated mean annual discharge for the period over which a hydrological budget could be constructed (calculated as the sum of month means) was just 162 mm yr⁻¹, the lowest in the network. For further details of the hydrological analyses and full results see Appendix 1.

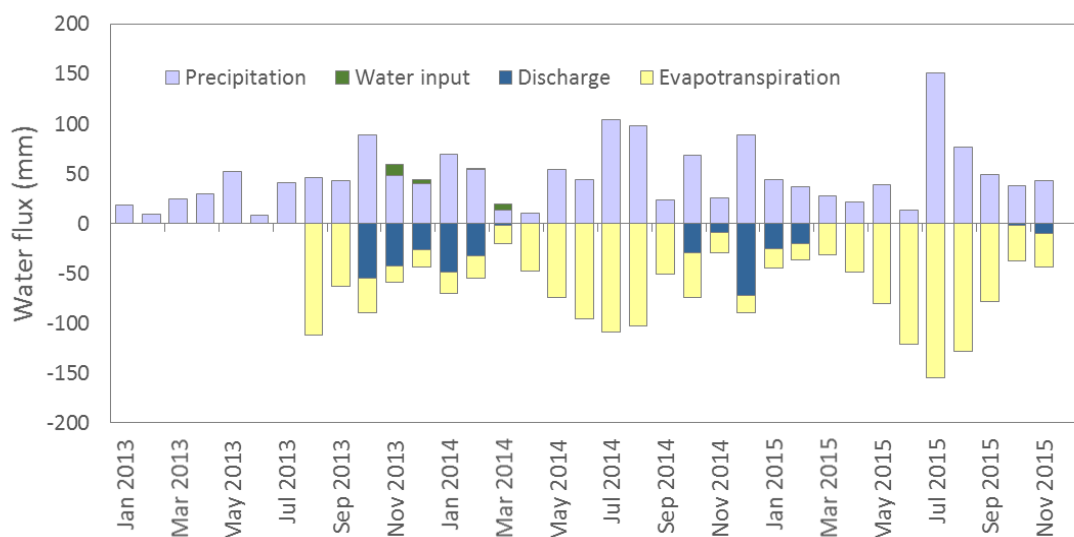


Figure 2.1.6 Monthly hydrological budgets for EF-LN. Note that ET and discharge were not determined for Jan-Jul 2013.

2.1.1.3. Eddy covariance gas fluxes

A ‘fingerprint’ plot showing measured (upper panel) and measured plus gap-filled (lower panel) net ecosystem CO₂ exchange (NEE) for the EF-LN site is shown in Figure 2.1.7. These fingerprints illustrate diurnal and seasonal changes in (30 minute) CO₂ flux densities, as well as the temporal distribution of data-gaps (upper panel) and the performance of the method used to fill missing values. In these and subsequent plots, positive values (yellow to red) indicate periods when the ecosystem was a net source for atmospheric CO₂ (e.g. at night and in winter) and negative values (blue) represent times when the ecosystem was functioning as a net CO₂ sink (e.g. daytime in summer). Data in these figures are presented in units of $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$. Short periods of missing data (e.g. a few values) in the top plot of Figure 2.1.7 (and subsequent fingerprint plots) are mainly due to data exclusion following the application of data quality control (QC) procedures (see Appendix 2). Longer periods of data loss (e.g. whole days and longer) are typically due to system power outages or instrument failures.

Data coverage at EF-LN has been intermittent since the start of the project (lower panel) due to site access restrictions (early 2013, when the site was flooded) followed by persistent technical issues in 2014. These technical issues had been resolved by the end of 2014 and a complete annual cycle (with only short data gaps) was recorded at EF-LN during 2015. Although it is possible to gap-fill the long (over one month) data gaps at EF-LN (lower panel) it should be noted that this process increased the uncertainties in time-integrated (e.g. daily, annual) estimates of the net CO₂ balance.

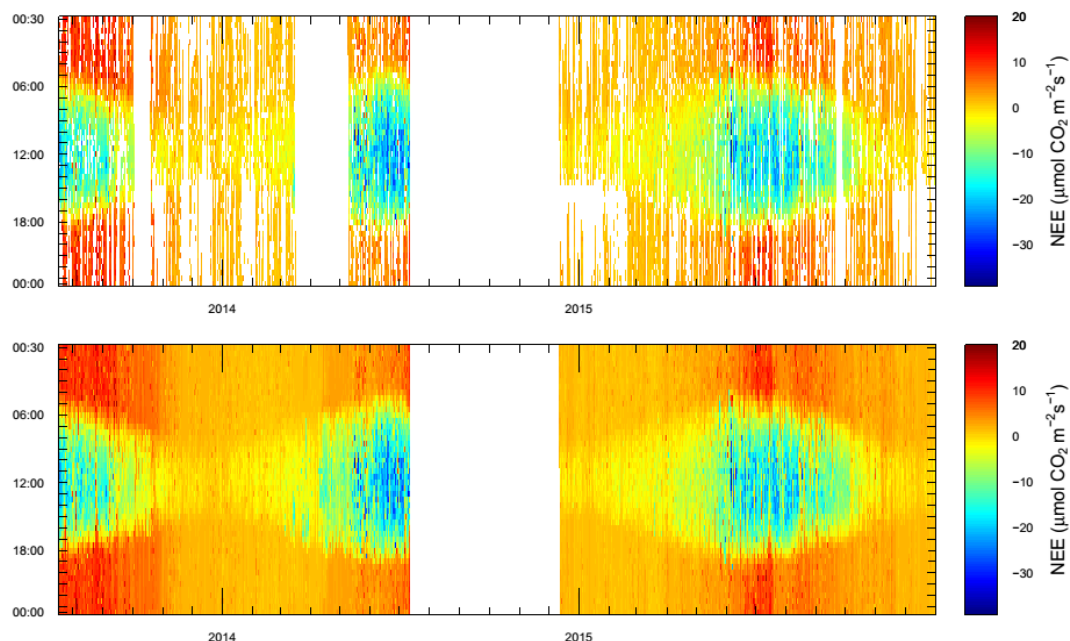


Figure 2.1.7. Fingerprint plot of measured (top panel) and measured and gap-filled (lower panel) net ecosystem carbon dioxide exchange (NEE) for the EF-LN (Wicken Sedge Fen) site. Units are in $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$.

Monthly mean diurnal cycles (MDCs) are a means of comparing seasonal and between-year differences in CO₂ fluxes on a side-by-side basis. They are generated by taking the average of measured fluxes at each time interval for all days within that month, and thus provide an indication of the average rate of CO₂ uptake or emission over the diurnal cycle during that month. Monthly MDCs of NEE measured at the EF-LN site are shown in Figure 2.1.8 for all months of the main growing season with acceptable data coverage (note that October 2013 has been omitted due to low data coverage – see Figure 2.1.7). In these plots, each data point represents the mean of thirty-minute values measured (i.e. not gap-filled) at the same time of day over the course of each month (i.e. for the 48 thirty-minute intervals in each day). As with the fingerprint plots (Figure 2.1.7), average diurnal cycles are presented in $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$. At sites with permanent vegetation like EF-LN, the MDCs illustrate the seasonal evolution of the amplitude of NEE in response to weather conditions and ecosystem phenology.

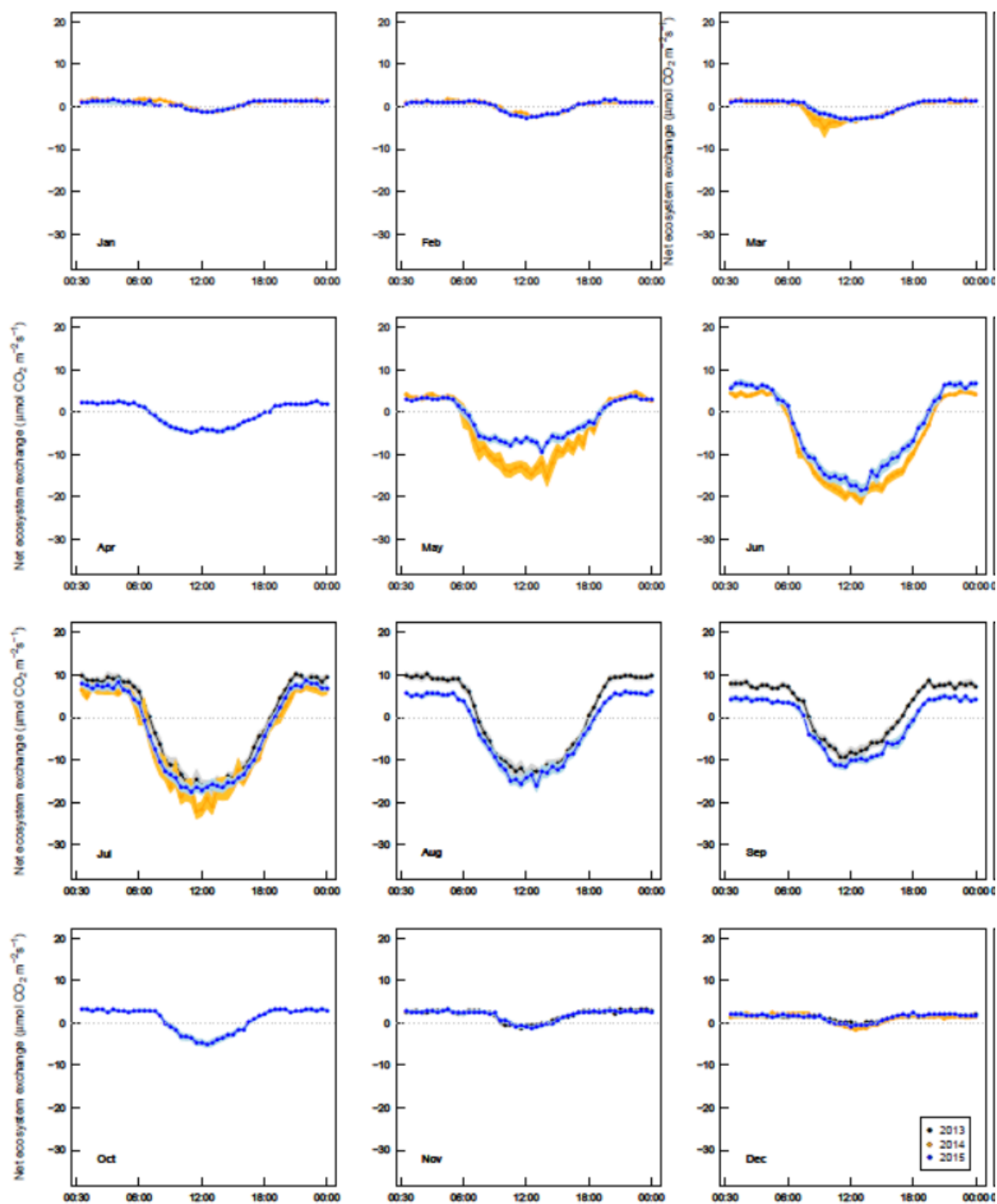


Figure 2.1.8. Mean diurnal cycles of CO₂ flux for each month that eddy covariance measurements were made for the EF-LN site (Wicken Sedge Fen). Data points are the mean of data measured at the same time of day within each month. Shaded areas show one standard error of the mean. Monthly diurnal averages were calculated using measured (not gap-filled) data. Months with poor data coverage have been omitted.

At EF-LN, the only calendar month with complete data coverage in all years was July (Figure 2.1.8) and between-year comparisons at EF-LN are based on differences compared to the complete 2015 dataset. The warmer conditions of May and June 2014 were associated with higher rates of CO₂ uptake (e.g. more negative) compared to the same months of 2015. Maximum CO₂ uptake rates at EF-LN were observed during July in 2013 ($-17.12 \pm 0.79 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) and 2014 ($-22.14 \pm 1.69 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) and during June in 2015 ($18.54 \pm 1.33 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$). The highest (monthly average) rates of nocturnal CO₂ efflux was observed in during August in 2014 at $9.55 \pm 0.15 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$. The period of maximum water-level drawdown during the summer months of 2013 had a clear influence on CO₂ fluxes at this site. Net CO₂ uptake rates were less negative in 2013 compared to 2015 when water levels were closer to the surface, whereas nocturnal CO₂

losses were notably higher. For the two full one-year measurement periods for which CO₂ balances could be determined, EF-LN acted as a net sink, with an NEE of -55 (± 105) g C m⁻² yr⁻¹ from July 2013 to July 2014, and -183 (± 98) g C m⁻² yr⁻¹ for the 2015 calendar year. Annual C balance data are analysed in detail in Section 4.

Methane (CH₄) fluxes are also measured by eddy covariance at EF-LN by the University of Leicester. Since most other EC sites in the network did not have CH₄ sensors, a standard project protocol QC and gap-filling of CH₄ fluxes based on eddy covariance measurements has not been developed, and data are not included in the current report. However CH₄ data from eddy covariance and chamber measurements made during the project, along with eddy covariance estimates of evapotranspiration, have recently been included in a manuscript (Kaduk et al., submitted) examining the role of water-table variation on emissions. This analysis suggests that total CH₄ emissions from the EF-LN site are in the region of 3.8 to 4.0 g CH₄ m⁻² yr⁻¹.

2.1.1.4. Static chamber gas fluxes

Static chamber measurements at EF-LN started in August 2013, after flooding restricted access to the site in the early part of 2013, and continued to the end of 2015. Measurements were made in the two dominant vegetation communities at the site, *Phragmites australis* and *Cladium mariscus*, with 3 replicate collars in each community. Measured and modelled fluxes are shown in Figure 2.1.9. The observed chamber data show the expected seasonality in both gross primary productivity (GPP) and ecosystem respiration (ER), with daytime measured GPP greatly exceeding ER during the summer growing season for both vegetation types. Peak values of both GPP and ER were observed in the dry summer of 2013. Modelled fluxes for EF-LN were based on air temperature for ER, and photosynthetically active radiation (PAR) plus temperature for GPP. Compared to some other sites, model fits were relatively poor, with the ER model in particular failing to capture summer maxima or winter minima. The GPP model was somewhat more successful in capturing daytime peak rates of photosynthesis. Modelled NEE values suggest that the site acted as a fairly consistent sink for CO₂ in the growing seasons of all three measurement years, with near-zero fluxes during winter. Differences between vegetation communities were minor. Overall, assuming a 50/50 cover of *Phragmites* and *Cladium* across the site, modelled NEE for the three measurement years was -153, -149 and -155 g C m⁻² yr⁻¹ for 2013, 2014 and 2015 respectively, with an overall mean of -152 g C m⁻² yr⁻¹. The chamber modelled NEE for 2015 compares reasonably well with the value of -183 g C m⁻² yr⁻¹ obtained from the flux tower for that year, supporting the conclusion that the site is acting as a fairly strong net CO₂ sink.

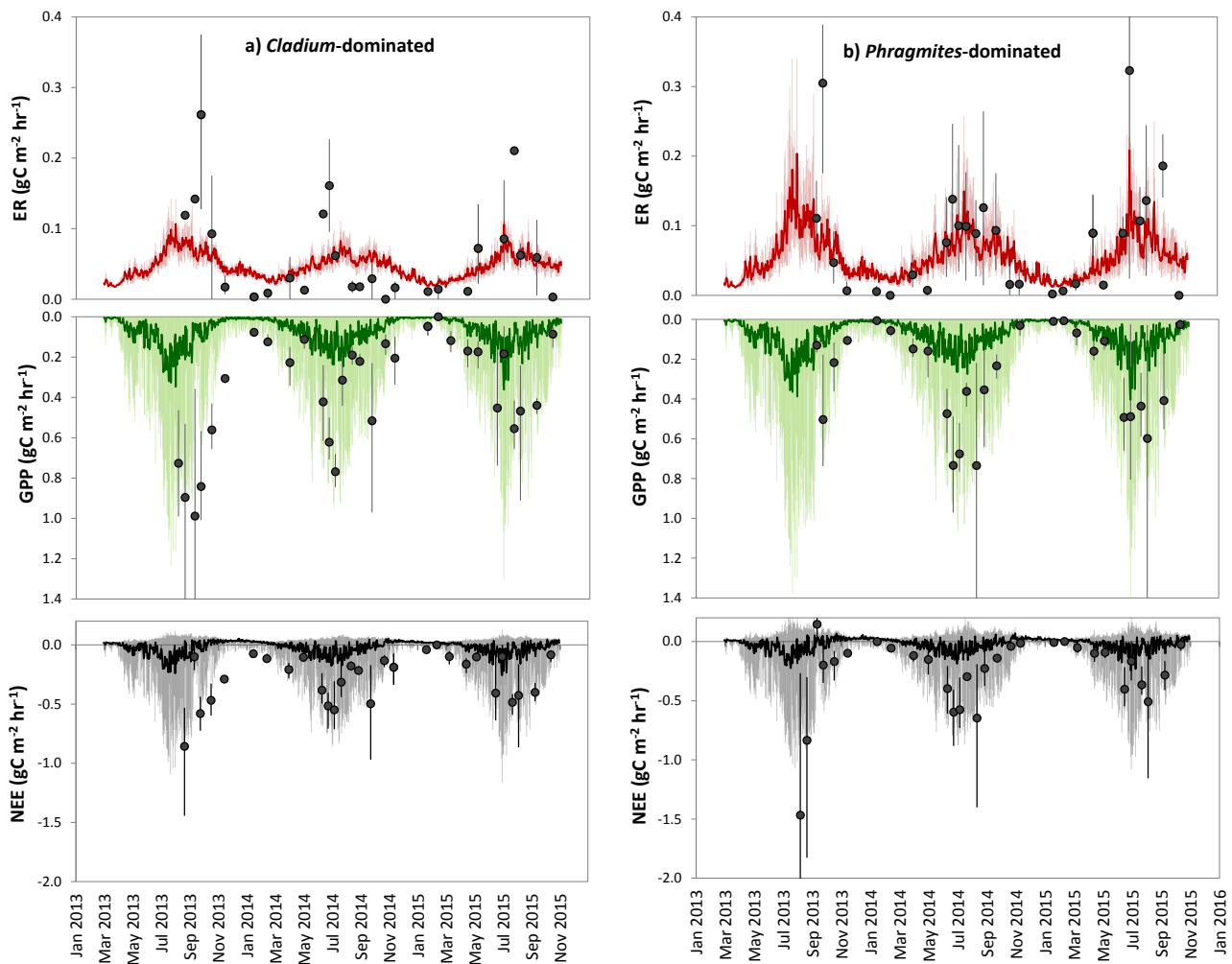


Figure 2.1.9. Modelled and observed CO_2 fluxes (ER = ecosystem respiration, GPP = gross primary productivity, NEE = net ecosystem exchange) based on static chamber measurements in two vegetation communities at EF-LN. Continuous lines show modelled daily mean fluxes, shading shows modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date. Note that static chamber measurements were generally taken around the middle of the day, so tend to be representative of peak (rather than daily mean) rates of photosynthesis and respiration.

Methane fluxes for the two vegetation communities are shown in Figure 2.1.10. Fluxes were uniformly close to zero during 2013, when the severe water-table drawdown event occurred, but significant emissions were measured in 2014 and 2015. Despite some evidence of seasonality and of a relationship to water table depth, CH_4 fluxes showed high short-term temporal and spatial variability, and a robust empirical flux model could not be developed. Therefore, fluxes were interpolated between measurement dates as shown in Figure 2.1.10. Overall estimated mean annual CH_4 emissions were $11.9 \text{ g C m}^{-2} \text{ yr}^{-1}$ for the *Phragmites* community, and $5.6 \text{ g C m}^{-2} \text{ yr}^{-1}$ for the *Cladium* community. The difference in fluxes between communities appears to be fairly consistent over the monitoring period. Assuming a 50/50 mix of communities across the site as a whole gives a mean CH_4 emission of $8.75 \text{ g C m}^{-2} \text{ yr}^{-1}$.

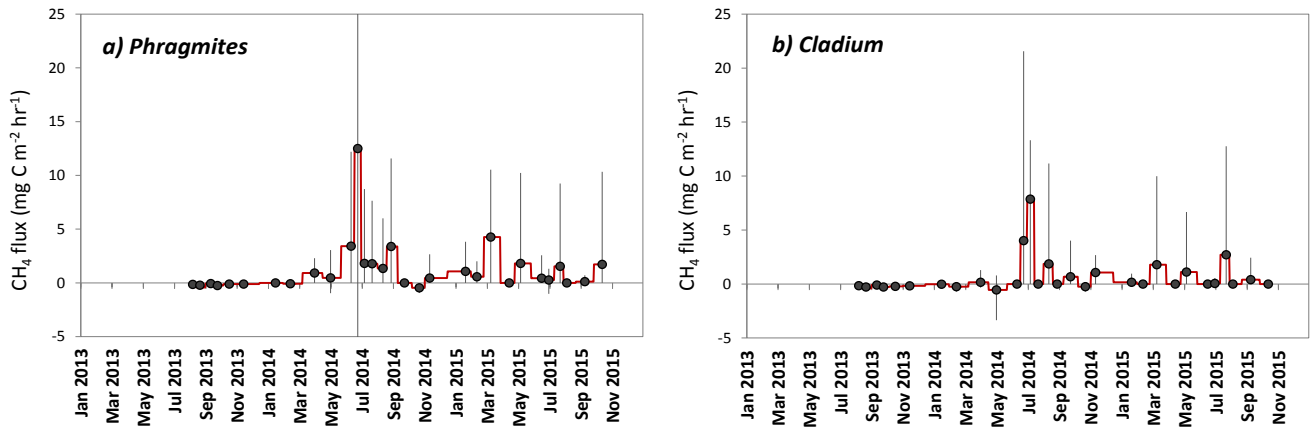


Figure 2.1.10. Measured CH_4 fluxes for the Phragmites- and Cladium-dominated vegetation communities at EF-LN. Points show mean observations on each measurement date, and error bars show range of measured values on that date. Red lines show interpolated fluxes (fluxes on days without measurements assumed to be equal to those obtained on the closest day with measurements).

Average annual measured CH_4 and CO_2 fluxes from ditches at EF-LN are shown in Figure 2.1.11. Both showed high temporal variation, but were consistently positive (i.e. net emission) with higher fluxes occurring during spring and summer. The annual mean estimated CH_4 emission per unit ditch area was $125 \text{ g C m}^{-2} \text{ yr}^{-1}$. Based on the mapped ditch network across the site, the fractional area of ditch surface across the fen, $Frac_{ditch}$, was estimated to be 0.014, giving an area-weighted ditch CH_4 for the Sedge Fen as a whole of $1.76 \text{ g C m}^{-2} \text{ yr}^{-1}$. For CO_2 , the mean emission from the ditches was $2610 \text{ g C m}^{-2} \text{ yr}^{-1}$, implying an area-weighted emission of $36.6 \text{ g C m}^{-2} \text{ yr}^{-1}$. Thus, it appears that CO_2 emissions from the drainage ditches at EF-LN could offset around 20% of CO_2 uptake by the terrestrial part of the fen (based on the terrestrial uptake rate modelled from the static chamber data; ditch emissions should be captured within the flux tower measurements).

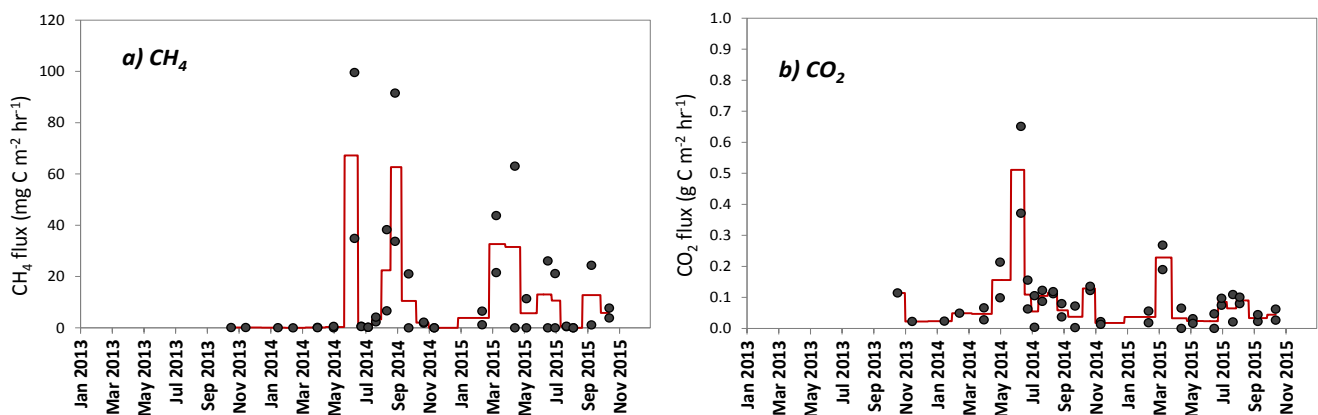


Figure 2.1.11. Observed CH_4 and CO_2 fluxes measured in ditches at EF-LN. Observations are represented by circles, red line shows interpolated fluxes.

2.1.1.5. Aquatic carbon fluxes

Measured ditch concentrations of DOC and DIC (the two largest aquatic carbon components) are shown for EF-LN in Figure 2.1.12. Monthly mean DOC concentrations ranged from around 15 to 30 mg l^{-1} , but without any clear pattern. The lowest concentrations occurred during and after the dry summer of 2013, suggesting reduced production of DOC from the peat, or increased degradation of DOC in the ditch network, at this time. It is also likely that the active hydrological management of Wicken Sedge Fen (i.e. pumping of water from adjacent rivers onto the fen) is affecting DOC concentrations at this site, reducing concentrations in the vicinity of the pump (M. Peacock, unpublished data). Concentrations of DIC were consistently high (around 80 mg l^{-1}), and measured pH was correspondingly high (mean 7.68, data not shown). Note that DIC export from fens is unlikely to represent carbon loss from the peat, since most of the flux is likely to be derive from mineral weathering processes (see Section 4.3.2.3)

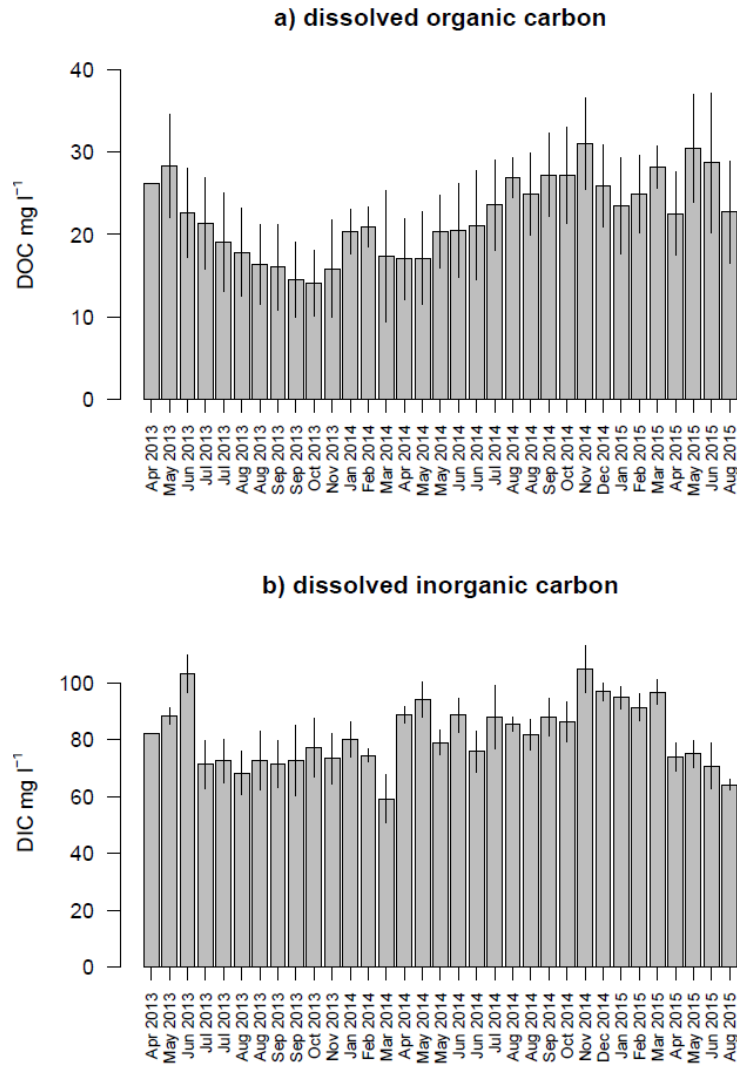
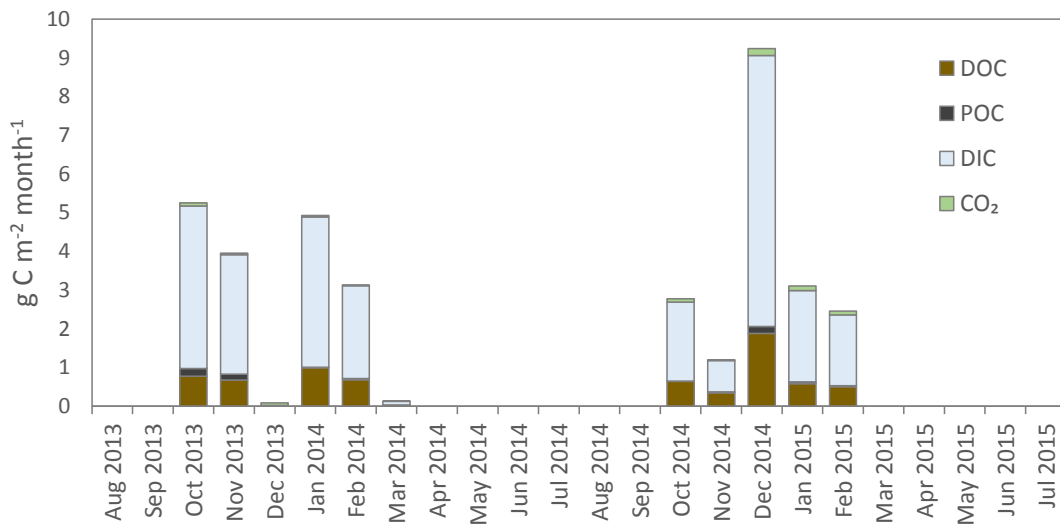


Figure 2.1.12. Mean and standard error of ditch dissolved organic and inorganic carbon concentrations, EF-LN. Note that x axis simply records sampling dates and is therefore not a true time axis.



Aquatic carbon fluxes are shown in Figure 2.1.13. Due to the lack of water flow out of the site during summer, and low discharge in general (Figure 2.1.6) aquatic carbon fluxes from EF-LN were small and sporadic. Most of the C exported from the fen was DIC (estimated mean annual flux $17.4 \text{ g C m}^{-2} \text{ yr}^{-1}$) with a mean annual DOC flux of just $4.5 \text{ g C m}^{-2} \text{ yr}^{-1}$, and POC of $0.4 \text{ g C m}^{-2} \text{ yr}^{-1}$. The dissolved CO_2 flux leaving the site via the ditch network (assumed to be degassed downstream) was also low, at $0.4 \text{ g C m}^{-2} \text{ yr}^{-1}$, and dissolved CH_4 concentrations were negligible ($< 0.01 \text{ g C m}^{-2} \text{ yr}^{-1}$).

2.1.2. Bakers Fen – extensive grassland (EF-EG)

Bakers Fen forms part of the Wicken Fen Nature Reserve, located close to the natural Sedge Fen site and also managed by the National Trust. Bakers Fen was drained in the mid-19th century, and was used for the intensive production of cereals and row crops in the later part of the 20th century, resulting in a high level of peat wastage. The site was taken out of arable production in 1994, and restored to conservation-managed grassland, with grazing by Konik ponies and Highland cattle. Bakers Fen retains a functional ditch network which is used to support water management. Water-tables are raised in autumn by flooding water onto the site from the adjacent (and higher, as a result of peat wastage) Monk's Lode. There is an altitudinal gradient across the site from north east to south west, so that water flows across the site and out in to a deep ditch along the southern-western edge of the site. Man-made hollows mean that parts of the site remain flooded until late spring. In summer, water supply is limited to rainfall and the water-table falls below the base of the organic horizon. Although the site is subject to low-intensity livestock grazing, net biomass input and output fluxes are thought to be negligible.

Bakers Fen supports a species-poor damp grassland, reflecting its past management history. Drier areas are considered to most closely resemble NVC class MG1a, a mesotrophic grassland community dominated by *Agrostis stolonifera* with *Arrhenatherum elatius*, *Cirsium arvense*, *Dactylis glomerata* and *Holcus lanatus* in drier areas. Other areas fall within the MG10b rush pasture community, with clumps of hard rush (*Juncus inflexus*) growing within a matrix of *Agrostis stolonifera*. Ditch margins are largely occupied by *Phragmites australis* (for more information see Appendix 3).

Peat core data (Figure 2.1.14) confirm that the organic horizon at the site is very shallow (around 20-50 cm), mineral- and nutrient-enriched, with a high bulk density. The soil appears typical of the wasted (skirtland) peats that now cover a large part of the Fens, and it is debatable whether it still qualifies as a histosol, although the carbon stock is still substantial (6100 t ha^{-1} ; Table 1.2). On the other hand, the proximity of the site to Wicken Fen clearly demonstrates that it once supported deep peat, and the current management of the site is aimed at restoring some degree of wetland function.

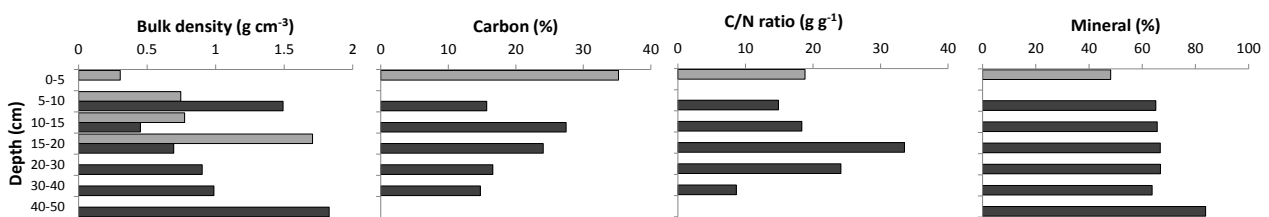


Figure 2.1.14. Peat core data, EF-EG. Data from two cores analysed from the site are represented by dark and light grey-shaded bars.

Meteorological conditions at EF-EG are represented by the data presented above for EF-LN.

2.1.2.1. Hydrology

Water-table data for EF-EG are shown in Figure 2.1.15. Differences between the two logger records primarily reflect variations in surface topography and resulting organic horizon depth, rather than differences in the absolute elevation of the water-table between sites. Apparent periods of steady water-table are an artefact and occurred when the water table fell below the deepest recordable depth of the sensors, and indicated that the entire peat layer had become aerated. This occurred in all three summers, mostly notably during

2013, coinciding with the severe water-table drawdown at EF-LN. In 2014, and to a lesser extent 2015, the summer drawdown period was interrupted by short periods of partial peat re-wetting following rain events. Ditches respond rapidly to rainfall, indicating flow into the ditch, and there is little evidence to suggest that ditch levels help to maintain the depth of the water-table during the summer months; the manual and automated ditch level records indicate that the water level within the ditch is often below the water-table height within the field.

The autumn flooding of water onto the site is evident in near-instantaneous water-table rises in all three years, following which water levels remain close to (and occasionally above) the surface throughout the winter. This near-bimodal pattern of water-table variation suggests that the site now functions more like a seasonally inundated wetland than a typical fen peatland.

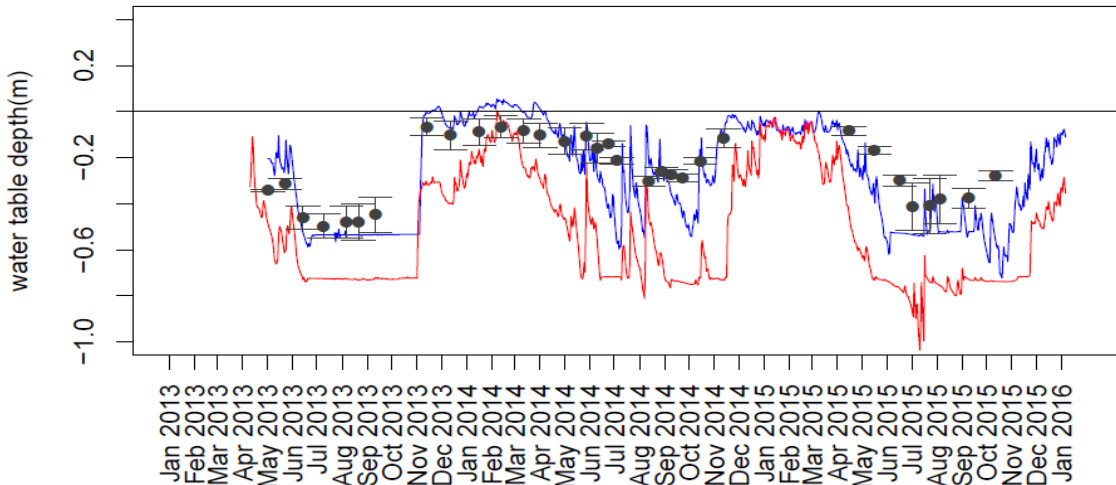


Figure 2.1.15. Continuously monitored and manually measured water-table data, EF-EG. Blue and red lines show data from different dipwells at the site.

Monthly water balance data for EF-EG are shown in Figure 2.1.16. Precipitation was measured on site, along with ET from the flux tower. Given the highly modified hydrological characteristics of the site, groundwater inputs and outputs were assumed to be zero. Water transfers onto the site were recorded, and any precipitation not lost via ET was assumed to be lost as surface water discharge via the ditch network (which was regulated via a sluice at the lower boundary of the site). The site has similar water balance characteristics to EF-LN, with runoff limited to the winter months when water levels are raised. Autumn water transfers onto the site appear to add a relatively small amount to the total water input. Mean annual discharge from EF-EG was estimated to be 180 mm yr⁻¹.

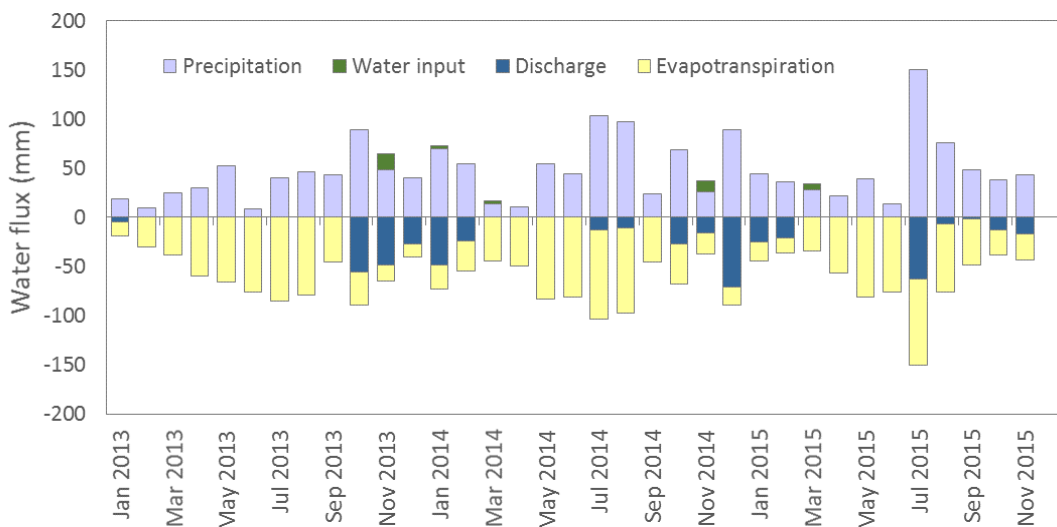


Figure 2.1.16. Monthly hydrological budgets for EF-EG.

2.1.2.2. Eddy covariance gas fluxes

Measured and gap-filled fingerprint plots for the EF-EG site are shown in Figure 2.1.17. Data capture (after QC) at EF-EG was good during the main growing season but less good during the winter months (upper panel in Figure 2.1.17). A significant data gap (total data loss for longer than one complete month) occurred at EF-EG from early January to mid-February 2014 following theft of the batteries. No further extended periods of complete data loss (of more than a few days) occurred for the remainder of 2014 and 2015, although problems with electrical power supply resulted in the loss of data during nocturnal periods from January 2013 to March in 2013 and between mid-November 2014 until the end of February 2015. As these data gaps occurred during non-growing season months when CO₂ fluxes were at a seasonal low, the filling of these data gaps is less likely to introduce significant uncertainty in terms of the annual CO₂-C budget compared with data losses during the season of maximum plant growth.

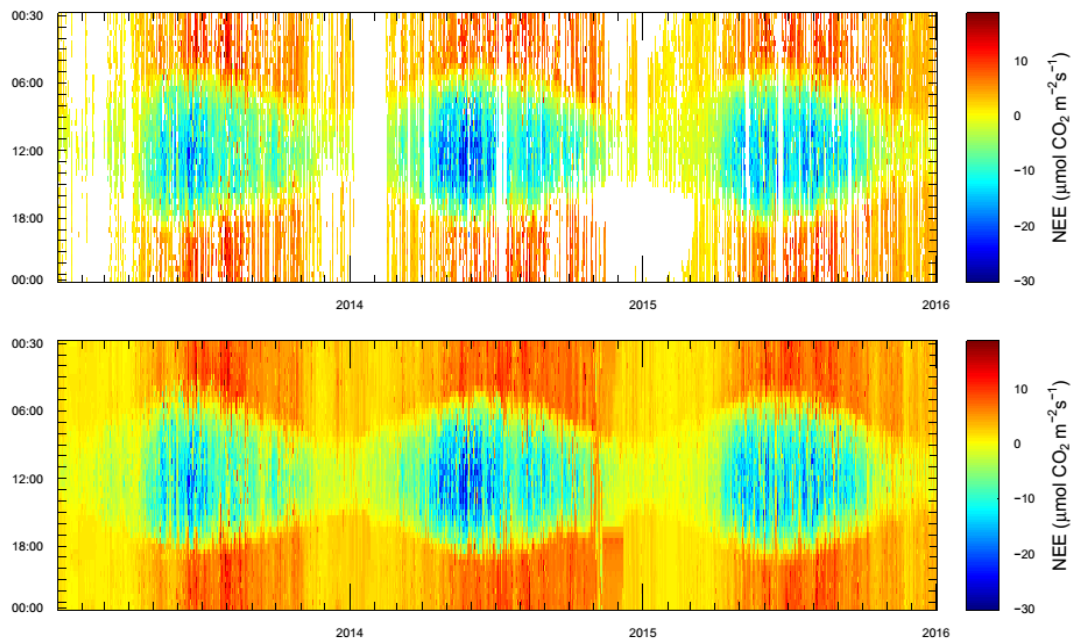


Figure 2.1.17. Fingerprint plot of measured (top panel) and measured and gap-filled (lower panel) net ecosystem carbon dioxide exchange for the EF-EG (Bakers Fen) site. Units are in $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$.

The measurements at EF-EG now include three complete annual cycles. The seasonal pattern of NEE is characteristic of sites with permanent vegetation cover and low-intensity management, with the lowest fluxes observed during winter (typically positive) and the largest (positive and negative) values in the summer months. The monthly mean diurnal patterns reveal large between-year differences in CO₂ fluxes at EF-EG (Figure 2.1.18). The amplitude of the mean diurnal cycles during the spring months appears to be controlled by differences in air temperature, with spring 2014 (warmest) having the largest amplitude and spring 2013 the smallest amplitude. Maximum CO₂ uptake rates occurred early in the growing season in all years, peaking during June in 2013 ($-14.48 \pm 0.69 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$), and during May in 2014 ($-18.0 \pm 0.96 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) and 2015 ($-13.84 \pm 0.8 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$). Net daily CO₂ uptake also peaked at this time. Nocturnal CO₂ efflux rates peaked later in the year, at between 7 to 8 $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$, in July (2013) or August (2014 and 2015). Diurnal CO₂ uptake was generally more negative between July and September 2015 compared to preceding years, although nocturnal release rates were similar. The amplitude of the diurnal cycle was higher during October 2014 than for the same months of other years, most likely related to the warm autumn period of 2014. A remarkable difference in net CO₂ uptake and release rates was observed during November, with more positive fluxes measured during the warm late autumn period of 2015.

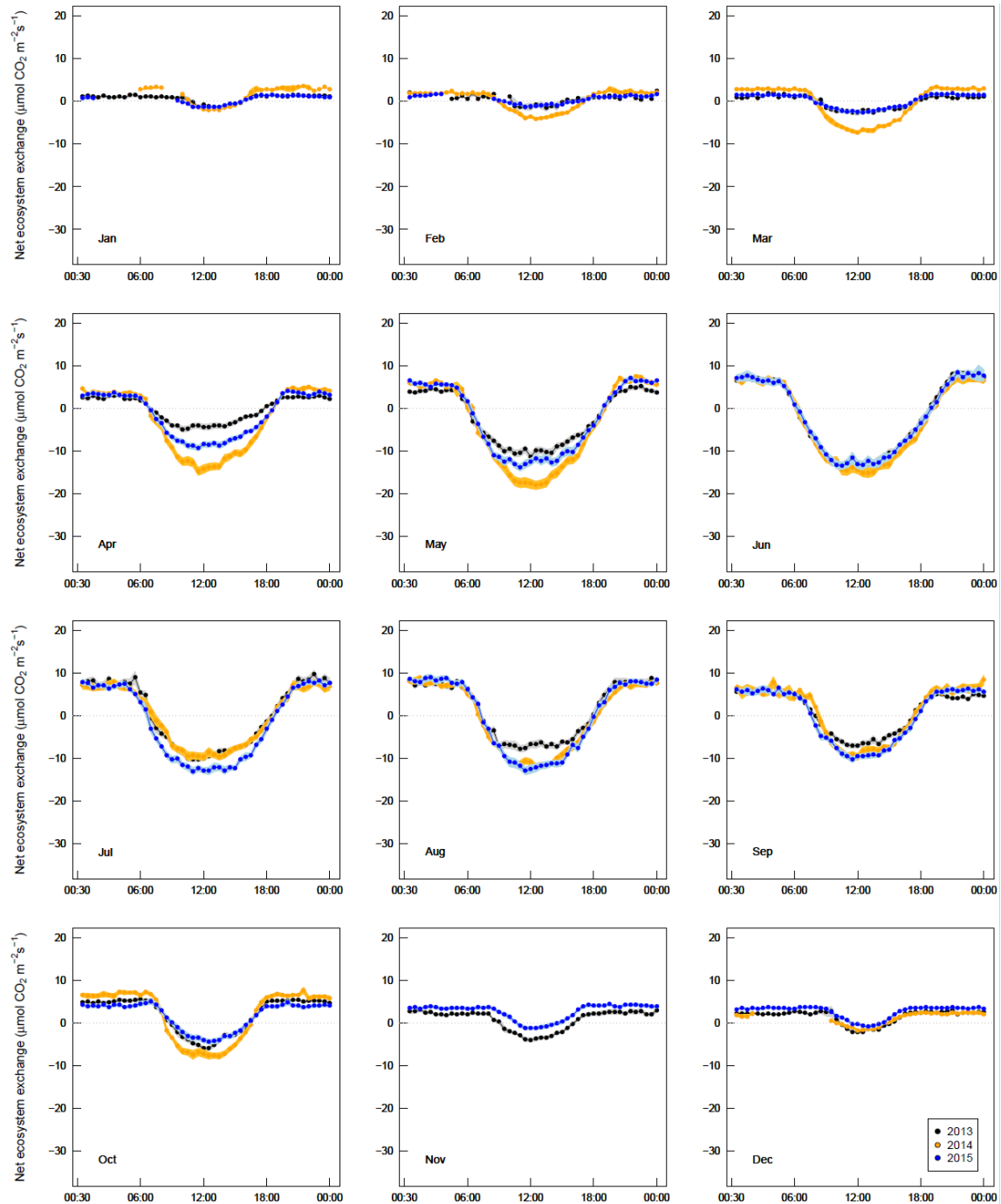


Figure 2.1.18. Mean diurnal cycles of CO₂ flux for each month that measurements were made at the EF-EG site (Bakers Fen). Data points are the mean of data measured at the same time of day during each month. Shaded areas show one standard error of the mean. Monthly diurnal averages were calculated using measured (not gap-filled) data. Months with poor data coverage (e.g. November 2013) have been omitted.

EF-EG was a net source of CO₂ emissions in all three years of measurement, with estimated NEE of +157 (\pm 111) g C m⁻² yr⁻¹ in 2013, +83 (\pm 39) g C m⁻² yr⁻¹ in 2014, and +130 (\pm 29) g C m⁻² yr⁻¹ in 2015. Annual CO₂ budgets are discussed in Section 4.

2.1.2.3. Static chamber gas fluxes

Static chamber flux measurements were made at EF-EG from May 2013 to August 2015, from areas within the *Agrostis stolonifera*-dominated dry mesotrophic grassland community, and from the *Juncus inflexus*-dominated rush pasture. Modelled ER was based on an exponential relationship with air temperature for both communities, while GPP was modelled as a function of PAR and air temperature. Observed fluxes

(Figure 2.1.19) suggest that summer ER peaks were higher in 2013 and 2014 than in 2015, with near-zero fluxes in the intervening winter periods. Peak measured GPP values exceeded maximum measured ER, although as all measurements were made during daytime this result is expected. Most measured NEE values were negative (i.e. net CO₂ uptake) although a few small positive fluxes were observed in winter. Model fits were comparatively poor for ER in both communities, with peak growing season respiration rates under-predicted, and minimum winter values over-predicted. Growing season peak daytime GPP values were also under-predicted, although the GPP model performed relatively well during the remainder of the year. Modelled NEE was negative throughout the growing season, with small positive fluxes predicted during winter. Modelled net CO₂ uptake was apparently greater in the grassland community compared to the rush pasture. Annual modelled NEE was similar for all three measurement years, with a site mean (assuming 50/50 cover of grassland and rush pasture) of -244 g C m⁻² yr⁻¹. This is clearly at odds with the net emission of CO₂ indicated by the flux tower data for all three years. The reason for this mismatch is unclear; it is possible that the static chambers failed to capture fluxes from drier areas within the grassland that are losing carbon, and/or wetter areas where seasonal ponding limits CO₂ uptake. Given the relatively weak model fits obtained, it is also possible that the static chamber data are simply not providing a good estimate of annual net CO₂ fluxes. In particular, the fitted models were unable to reproduce maximum rates of observed ER during hot summer periods, and may therefore have under-estimated overall CO₂ losses from the site.

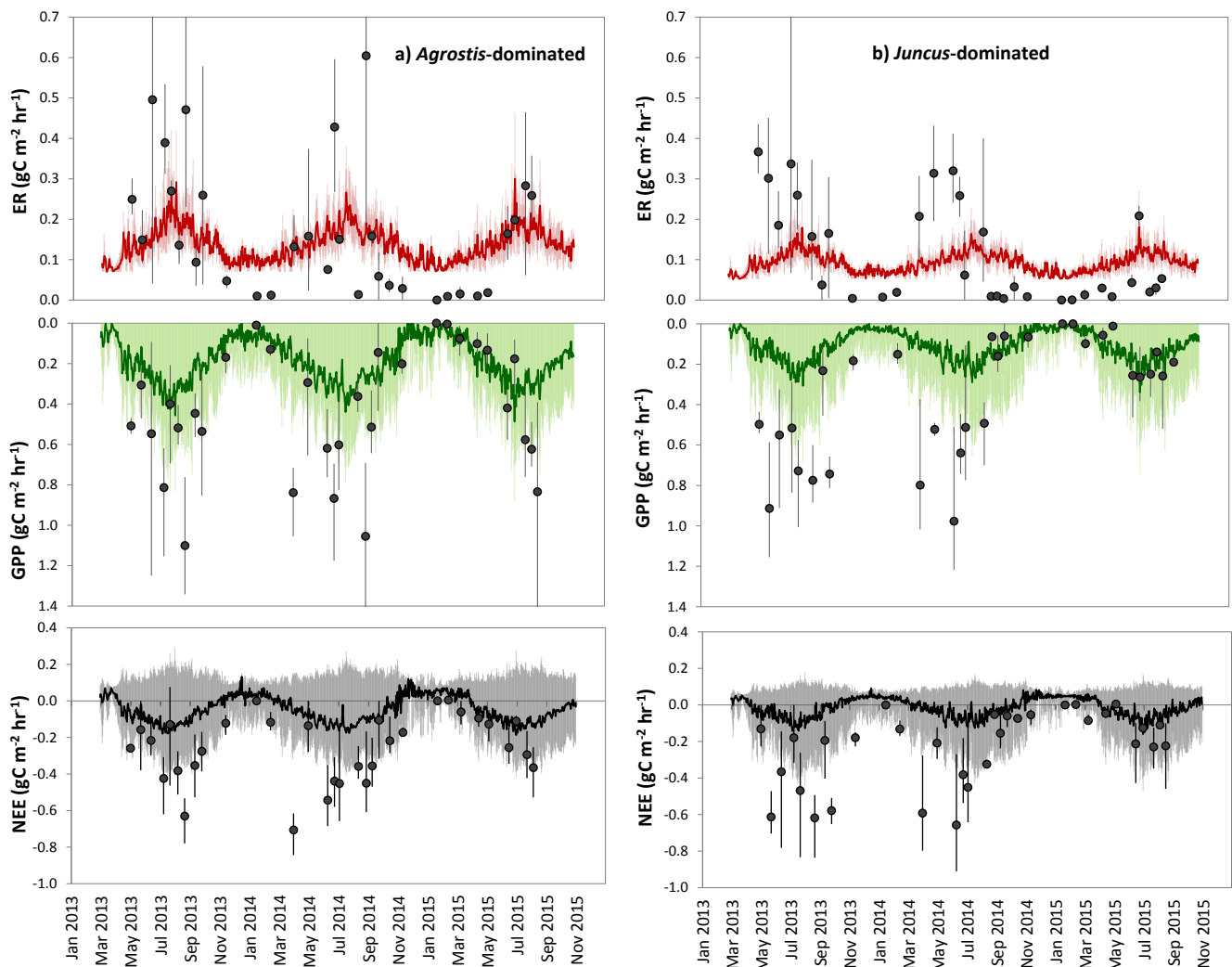


Figure 2.1.19. Modelled and observed CO₂ fluxes (ER = ecosystem respiration, GPP = gross primary productivity, NEE = net ecosystem exchange) based on static chamber measurements in two vegetation communities at EF-EG. Continuous lines show modelled daily mean fluxes, shading shows modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date. Note that static chamber measurements were generally taken around the middle of the day, so tend to be representative of peak (rather than daily mean) rates of photosynthesis and respiration.

Methane fluxes at EF-EG (Figure 2.1.20) were much lower than at the adjacent EF-LN, with the grass areas showing marginal net uptake (estimated annual mean flux $-0.22 \text{ g C m}^{-2} \text{ yr}^{-1}$) and the *Juncus* rush pasture showing a similar net emission (annual mean flux $+0.25 \text{ g C m}^{-2} \text{ yr}^{-1}$). Based on a 50/50 mixture of the two vegetation communities, this suggests an approximately zero net flux of CH_4 across the land-air interface.

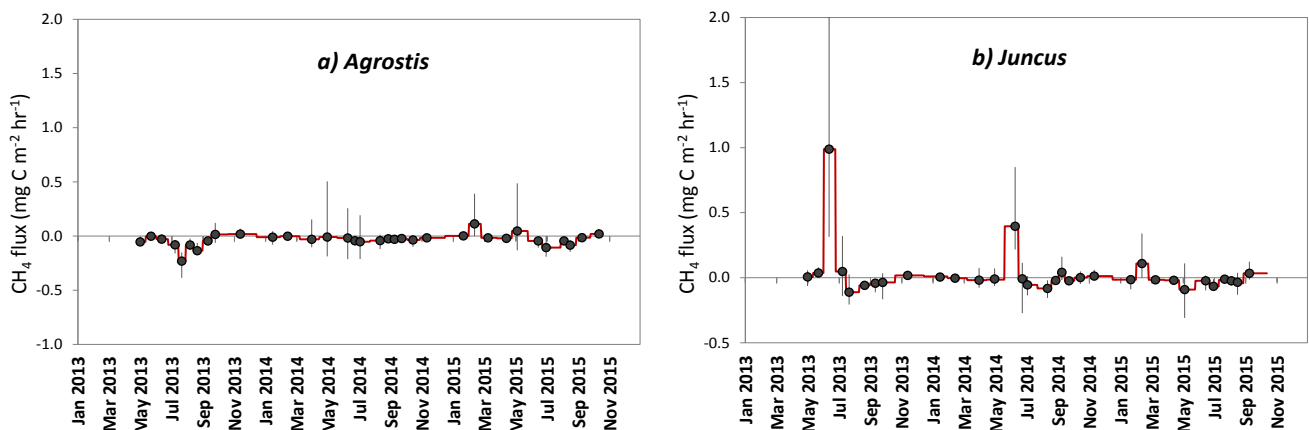


Figure 2.1.20. Measured CH_4 fluxes for the *Agrostis*- and *Juncus*-dominated vegetation communities at EF-EG. Points show mean observations on each measurement date, and error bars show range of measured values on that date. Red lines show interpolated fluxes.

Floating chamber measurements at EF-EG (Figure 2.1.21) showed very low CH_4 fluxes on all occasions except for a very large pulse of emissions in July 2013 which (since it was repeated in two measurements) is assumed to be real. Ditch CO_2 fluxes showed less extreme variability, but nevertheless fluctuated unpredictably over time, with some tendency towards higher emissions in summer. The estimated annual mean CH_4 emission was $8.9 \text{ g C m}^{-2} \text{ yr}^{-1}$ per unit ditch surface area, although this is clearly strongly influenced by the single high flux measurement. Adjusted to the area of Bakers Fen as a whole, with an estimated $Frac_{\text{ditch}}$ of 0.017, the area-weighted ditch emission was $0.15 \text{ g C m}^{-2} \text{ yr}^{-1}$. Although this is sufficient to make the site as a whole a net source of CH_4 , the total flux remains very small. Additional measurements made on a campaign basis across EF-EG showed locally higher CH_4 emissions from other areas of the ditch network, suggesting that this flux could have been under-estimated at the field scale. For CO_2 , the estimated annual flux per unit ditch area was $1245 \text{ g C m}^{-2} \text{ yr}^{-1}$, giving an area-adjusted ditch emission for the whole fen of $21.6 \text{ g C m}^{-2} \text{ yr}^{-1}$. This suggests that ditch emissions make a substantial contribution to CO_2 emissions from the site, although this flux is clearly insufficient to explain the mismatch between annual CO_2 fluxes estimated from static chamber measurements (which exclude ditch emissions) and those estimated from eddy covariance (which include ditch emissions).

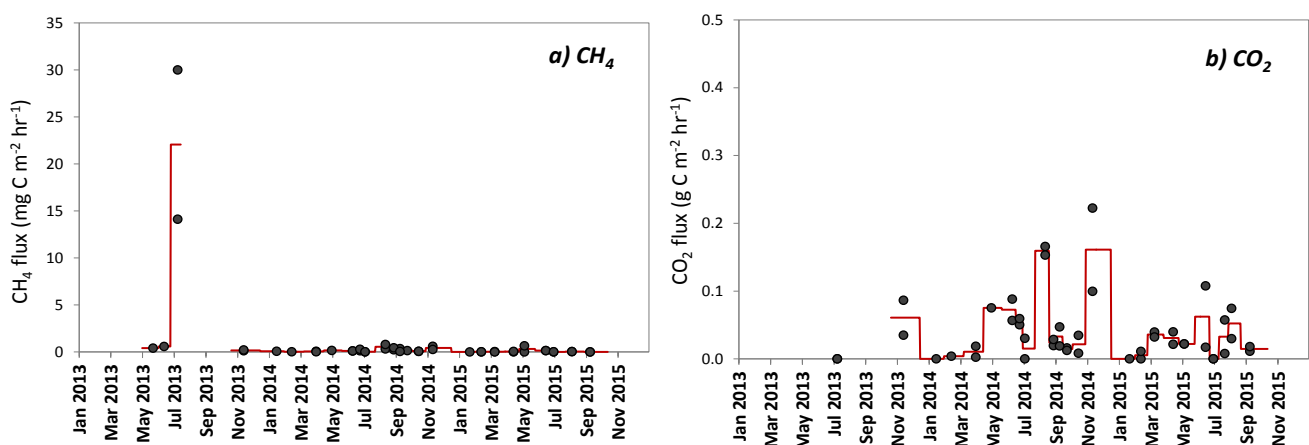


Figure 2.1.21. Observed CH_4 and CO_2 fluxes measured in ditches at EF-EG. Observations are represented by circles, red line shows interpolated fluxes. Data were not collected during late summer 2013 as the ditch dried out at this time.

2.1.2.4. Aquatic carbon fluxes

Ditch water DOC concentrations at EF-EG were highly variable (Figure 2.1.22). Maximum concentrations were recorded in the early summer of 2013 (when ditch levels were very low) and minimum concentrations in the subsequent autumn, associated with the transfer of low-DOC river water onto the site. A similar, but more subdued, pattern of variation was observed in the following years. Concentrations of DIC were fairly high and highly variable, but lacked any clear seasonal pattern. As at EF-LN, these data suggest that hydrological management activities dominate over natural seasonality in controlling variations in aquatic carbon concentrations.

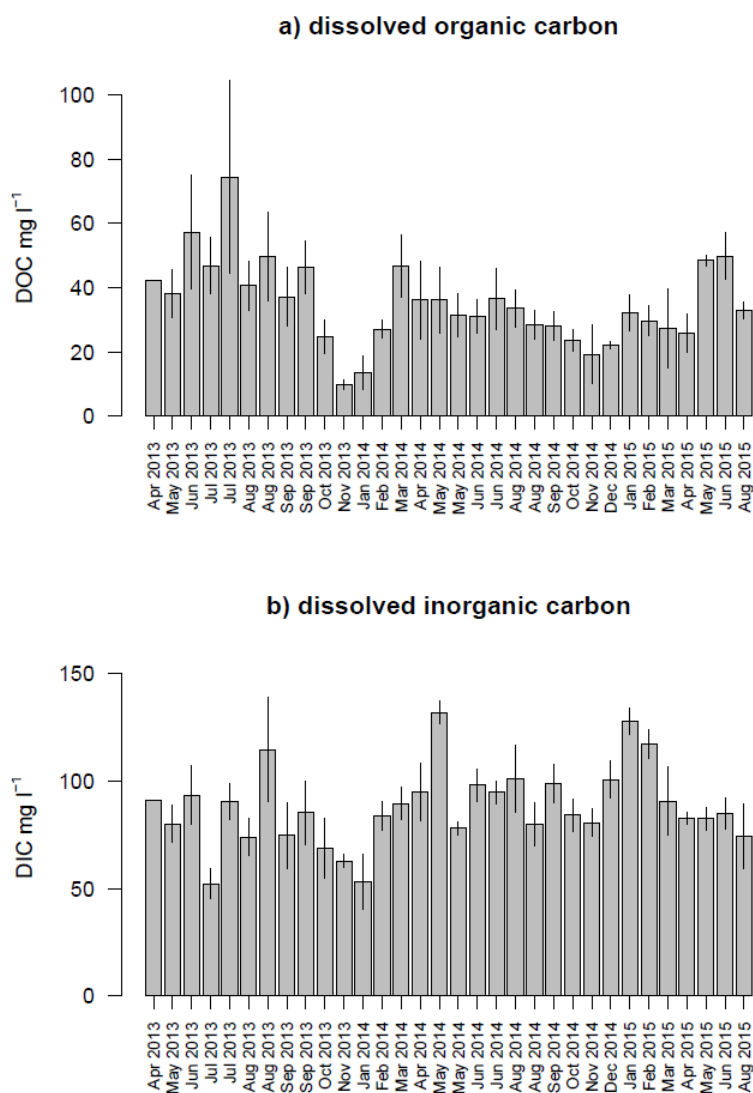


Figure 2.1.22. Mean and standard error of ditch dissolved organic and inorganic carbon concentrations, EF-EG. Note that x axis simply records sampling dates and is therefore not a true time axis.

Monthly aquatic carbon fluxes were calculated for EF-EG from April 2013 to July 2015 (Figure 2.1.23). Fluxes were restricted to the October – February period when water export occurred via the ditch network, and was dominated by DIC; estimated mean annual fluxes were 20.1 g C m⁻² yr⁻¹ for DIC, 5.7 g C m⁻² yr⁻¹ for DOC, 0.5 g C m⁻² yr⁻¹ for dissolved CO₂, 0.7 g C m⁻² yr⁻¹ for POC and 0.02 g C m⁻² yr⁻¹ for dissolved CH₄.

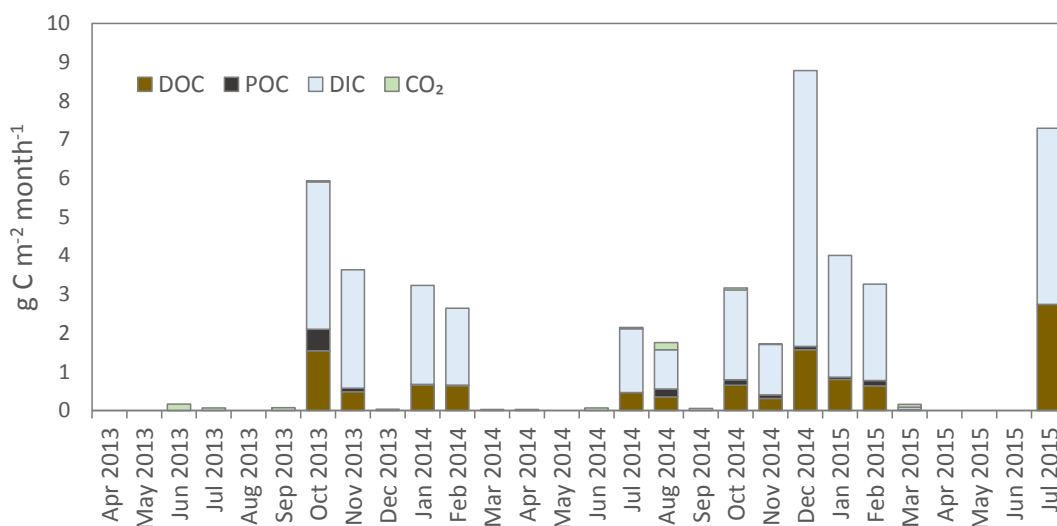


Figure 2.1.23. Estimated monthly aquatic carbon fluxes, EF-EG. Data missing for December 2013; all other zero values indicate no flux.

2.1.3. Rosedene Farm – arable on deep peat (EF-DA)

Rosedene Farm is located around 30 km northeast of the two Wicken Fen sites. The farm lies within one of the largest remaining areas of deep peat in the Fens (Figure 2.1.1), in an area which was estimated by Holman (2009) to have amongst the highest ongoing subsidence rates in the Fens. The site is now owned and farmed by G's Fresh. It was drained and placed under cultivation after the Second World War, with a dense drain network surrounding small fields which are primarily used to grow salad crops. Regularly spaced pipes within the fields are used for subsurface drainage, and can also be used for irrigation by raising ditch water levels. Water-tables are actively managed to maintain optimal moisture levels for crop growth during the growing season (typically water-table depths > 50 cm) and are also maintained at deep levels during winter to provide flood storage capacity and enable ditch maintenance. Fields are precision-levelled and subject to a range of agricultural practices including ploughing and disking post-harvest. The soil level at the site is thought to have fallen by approximately 1 m since agricultural production commenced but is still deep peat (around 2-3 m depth). Tree shelter belts around the fields are used to try to limit wind erosion.

During the study period, the study field at EF-DA was used to grow lettuce crops in 2012 and 2014, leeks in 2013, and celery in 2015. Lettuce and celery were grown from plug plants, which are grown in horticultural peat which thus adds biomass carbon to the field, albeit in the form of peat that has been extracted from sites elsewhere. Both annual carbon inputs and carbon offtake in harvested crops have been estimated for the site (A. Cumming, pers. comm.) and included in the site carbon balances (see Section 4). Outside of the crop growth period, the field is left fallow and develops a cover of agricultural weeds.

Core data (Figure 2.1.24) confirm that EF-DA retains many of the properties characteristic of a deep peat, including a high carbon content and fairly low mineral content. On the other hand, bulk density is high (0.5 g cm⁻³ in the upper 50 cm) and the C/N ratio is very low at 15 g g⁻¹, reflecting the effects of agricultural machinery and fertilisation respectively. Below a metre, mineral content and bulk density both increase markedly, and one core showed a layer of very carbon-poor soil at around 1.5 m depth. However the data indicate that around 2m of peat remains present at the site.

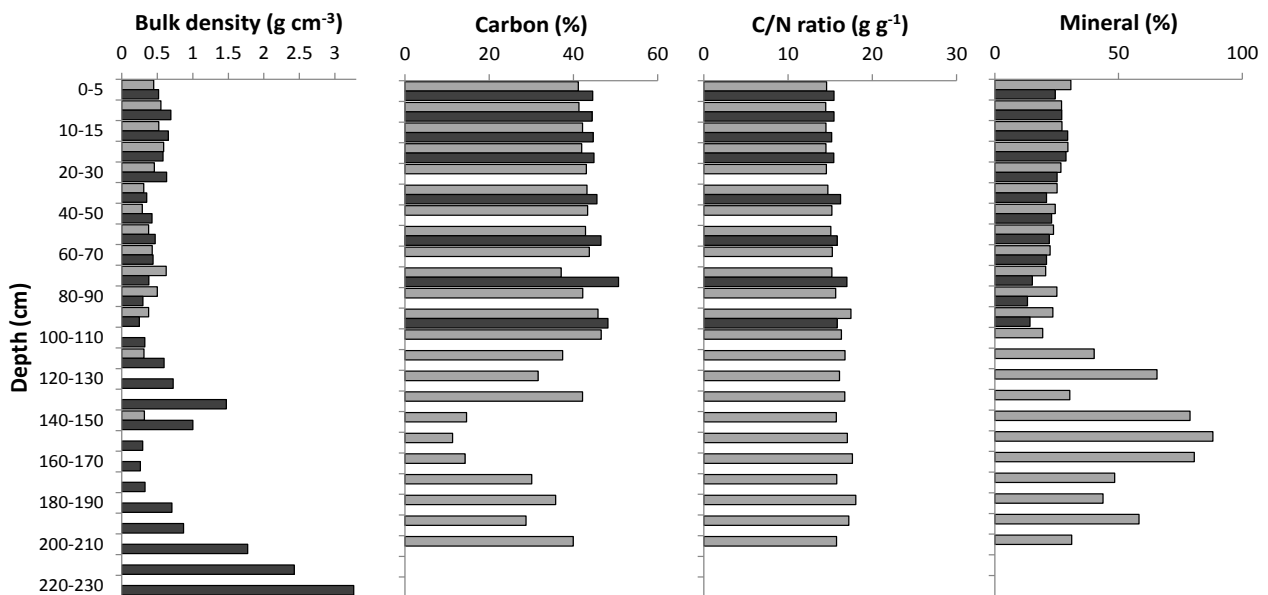


Figure 2.1.24. Peat core data, EF-DA. Data from two cores analysed from the site are represented by dark and light grey-shaded bars.

2.1.3.1. Meteorology

Monthly values of the main meteorological variables measured at EF-DA are shown in Figure 2.1.25. Any gaps in the SW_{in} and air temperature records were filled using linear relationships derived from the EF-SA site located approximately 11 km southeast of EF-DA. SW_{in} was only recorded at EF-DA from November 2013 onwards; however, photosynthetic photon flux density (PPFD, also called photosynthetically active radiation or PAR) was measured from the start of the period of record at this site. To enable comparison with other network sites, SW_{in} was estimated using a linear relationship between thirty minute measurements of SW_{in} and PPFD ($SW_{in} = 0.49 \cdot PPFD + 2.96$, $r^2 = 0.99$). Precipitation measurements at EF-DA are the most reliable of the four sites in East Anglia. Short periods of missing precipitation data were filled using a secondary rain gauge that was installed at the site. The EF-DA site experiences winds from all compass directions but with the southerly and south-westerly being the prevailing wind sectors (Figure 2.1.26).

The seasonal patterns of the main meteorological variables at EF-DA were similar to those measured at the Wicken Fen sites (Figure 2.1.3) as would be expected for sites located within the same region. Maximum total monthly SW_{in} was 278 kW m^{-2} in August 2012, 372 kW m^{-2} in July 2013, 349 kW m^{-2} in July 2014 and 333 kW m^{-2} in June 2015. Large between year differences in SW_{in} were observed in March (highest in 2014; lowest in 2014) and April (highest in 2015; lowest in 2014) and for July and August (lowest in 2012; highest in 2013).

Mean annual air temperatures at EF-DA for 2013, 2014 and 2015 were $9.6 \text{ }^\circ\text{C}$, $11.2 \text{ }^\circ\text{C}$ and $10.5 \text{ }^\circ\text{C}$, respectively. The early part of 2013 was noticeably colder than in other years, whilst 2014 was warmer. Peak summer temperatures were slightly lower in 2015, whilst the November-December period of this year was considerably warmer than average.

Total precipitation between 27th June 2012 and the end of December 2015 was estimated at 2545 mm. Total annual precipitation was 648 mm yr^{-1} in 2013, 765 mm yr^{-1} in 2014 and 641 mm yr^{-1} in 2015. Monthly precipitation varied considerably between years, but (as at EF-LN) there were a number of very wet months in 2014, whereas the June-July period of 2013 was particularly dry.

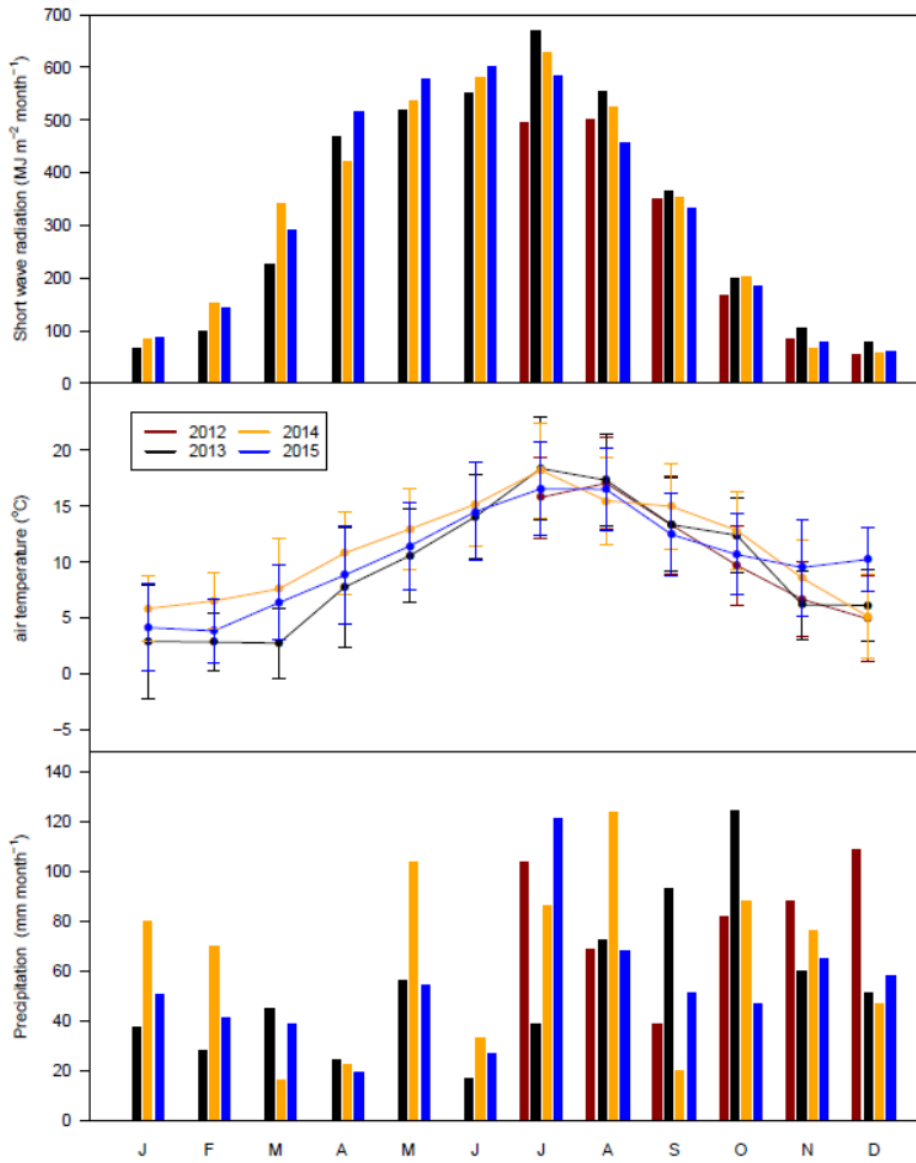


Figure 2.1.25. Total monthly incoming solar radiation (top) mean monthly air temperature (middle) and total monthly precipitation (lower) for the EF-DA (Rosdene Farm) site. Error bars on the temperature plot show one standard deviation of the mean.

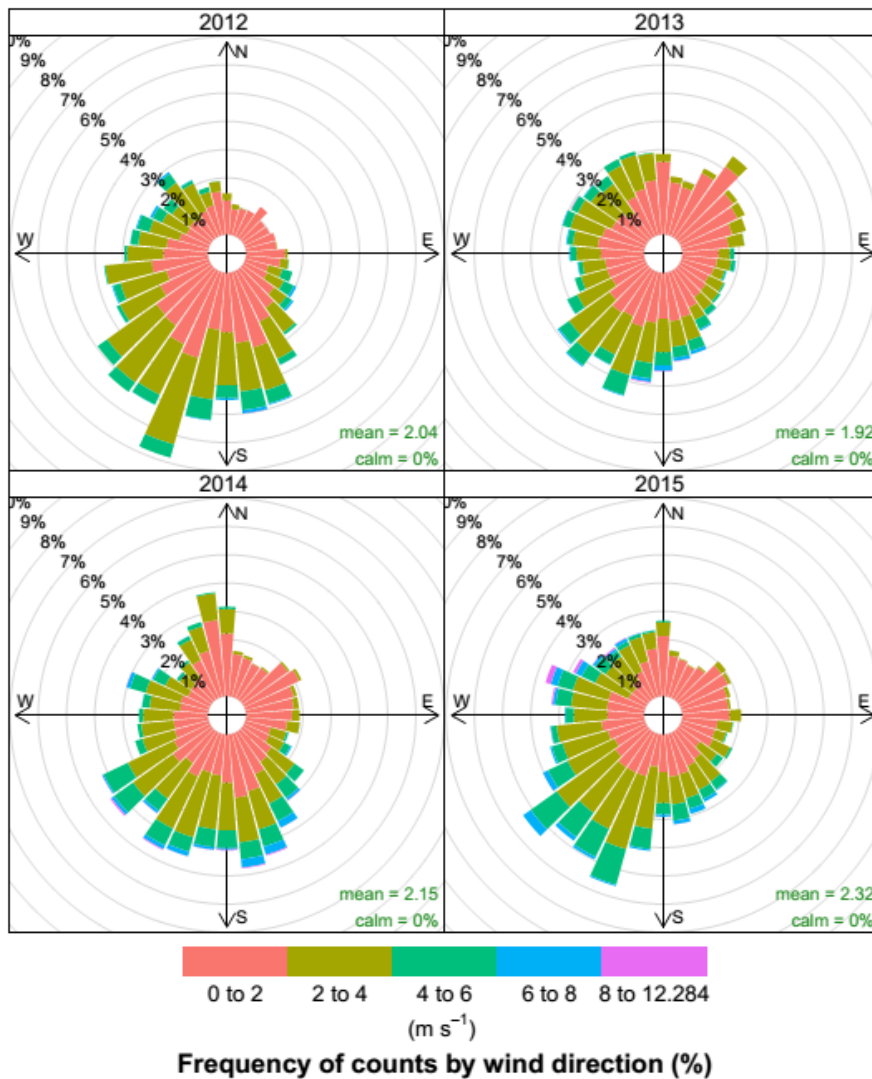


Figure 2.1.26. Wind rose plots showing wind direction and wind speed at the EF-DA (Rosdene Farm) site. Note that the wind rose for 2012 is based on data collected between 21st June and the end of the year.

2.1.3.2. Hydrology

Due to regular field operations within the arable EF-DA site, a single water level recorder was deployed close to the field boundary (established in late 2013), augmented by manual dipwell measurements which began in early 2013 (Figure 2.1.27). The data highlight the continuous effects of drainage at this highly managed arable site, with water tables rarely rising to within 50 cm of the ground surface, and frequently falling below the base of lowest recordable depth of 1 m. Water tables within the field are regulated by ditch levels, with ditch levels maintained above the water table during spring and summer, so that water flows onto the site via the subsurface drain network. As a consequence no aquatic losses occur during these periods as ET also exceeds rainfall during the growing season. Following heavy rainfall events the water table can rise above the ditch level, before falling. Observed peaks in water level were generally short-lived, indicating rapid loss of water via the subsurface drains.

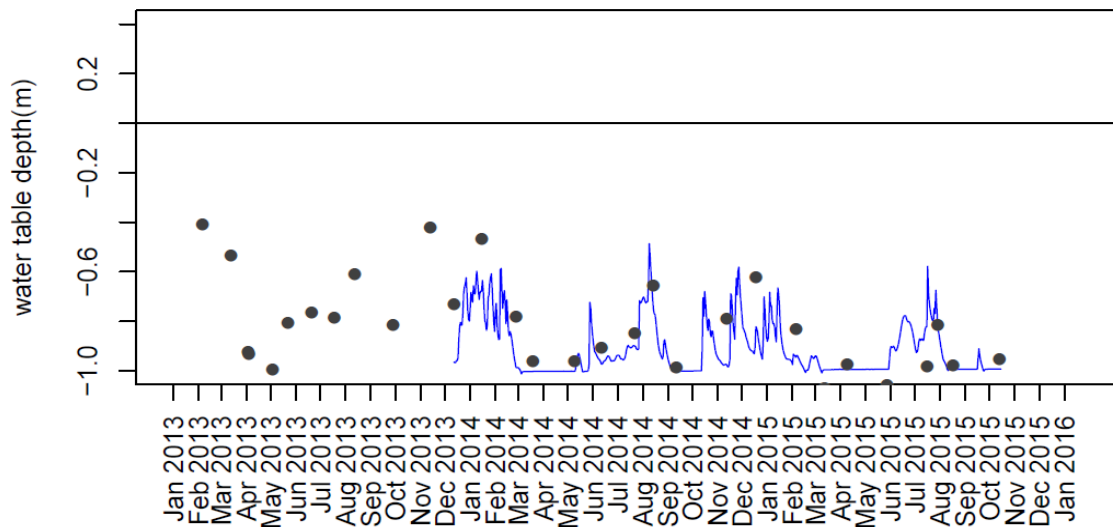


Figure 2.1.27. Continuously monitored and manually measured water-table data, EF-DA.

When the ditches were maintaining the water table, it was assumed that no discharge occurred from the site. When the ditch levels were below the water table it was assumed that water loss could occur when rainfall exceeded evapotranspiration, allowing for changes in storage within the soil as measured by the automated water-table logger. The change in storage was calculated by multiplying the fall in the water-table depth by the specific yield. Water export from the site (Figure 2.1.28) occurred discontinuously from late summer until early spring, with periods of no flux occurring in months during the growing season when rainfall was low. Mean annual water flux from the site was estimated at 240 mm yr^{-1} , higher than in the other East Anglian Fen sites.

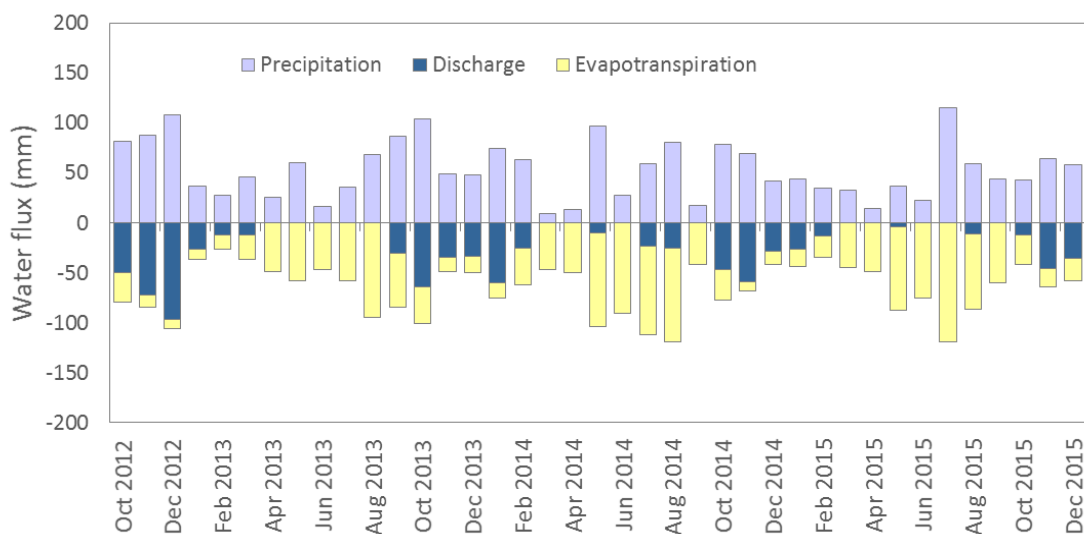


Figure 2.1.28. Monthly hydrological budgets for EF-DA.

2.1.3.3. Eddy covariance gas fluxes

Eddy covariance flux measurements at the EF-DA flux site commenced in June 2012 with support from a NERC Urgency Grant to the University of Leicester aiming to understand the impacts of drought conditions on fen peat soils in East Anglia (Morrison et al., 2013). Measurements at this site are continuing as part of a University of Leicester PhD studentship (Alexander Cumming), and are provided as in-kind support to the project. Near-continuous flux measurements have been recorded at EF-DA since June 2012; however, EC data coverage after QC is generally lower (circa 48% over the measurement period) than most other sites in

the SP1210 network (Figure 2.1.29, upper panel) with a non-representative source area being the most common cause for data rejection (in other words, when the measurement footprint lay outside the study field as a result of the relatively variable wind direction at this site illustrated by Figure 2.1.26). Despite this, the distribution of missing values is such that the longest gaps occurred outside of the cropping season when fluxes were smallest (e.g. during cold conditions at the start of 2013). A near-complete meteorological record at this site further improves the performance of the data gap-filling method used.

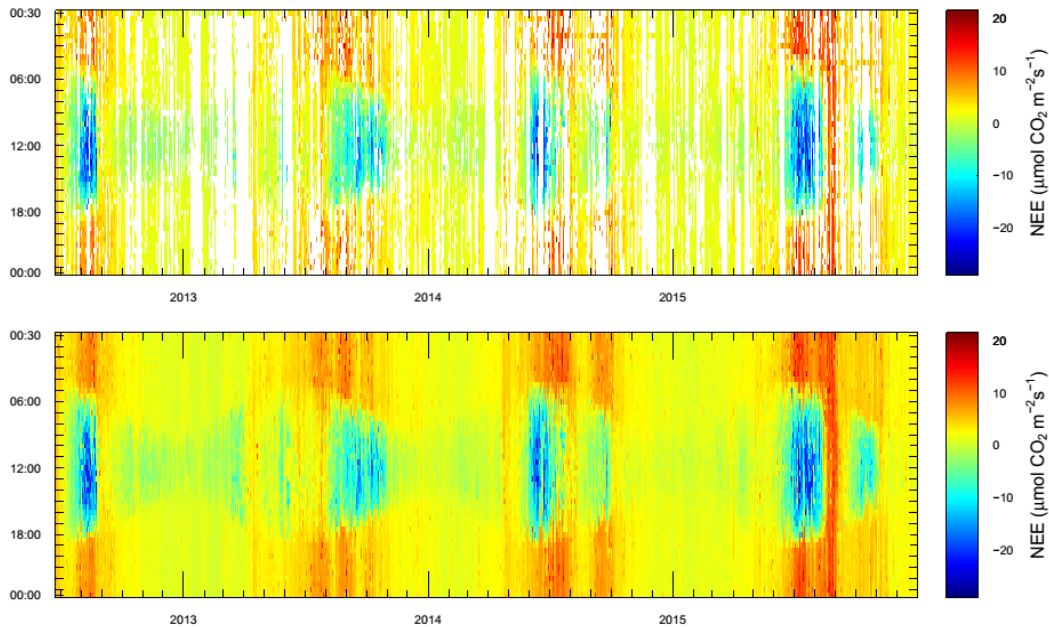


Figure 2.1.29. Fingerprint plot of measured (top panel) and measured and gap-filled (lower panel) net ecosystem CO₂ exchange for the EF-DA (Rosdene Farm) site. Units are in $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$.

The measurements at EF-DA have now captured CO₂ fluxes during four cropping seasons and intervening fallow periods (Figures 2.1.29, 2.1.30), given three full years of flux measurements from 2013-2015. Iceberg lettuce was grown at the site in 2012 and 2014, with leek and celery crops produced in 2013 and 2014 respectively. The crop growth periods when CO₂ uptake was occurring are clearly visible in Figure 2.1.29 as short periods of negative daytime NEE (light blue areas). Differences in the timing, strength and duration of CO₂ uptake for these different crops is clearly evident in the figures, leading to large between-month and between-year differences during the summer period in particular (Figure 2.1.30).

In 2012, lettuce was planted in June resulting in a period of (relatively) strong daytime uptake and night-time release of CO₂ in August (maximum rates of $-13.11 \pm 1.93 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ and $6.79 \pm 0.14 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$, respectively). This was followed by lower CO₂ flux densities for the remainder of the year with the diurnal pattern reflecting the development of a secondary plant cover. The lettuce crop was planted earlier in 2014 (May) compared to 2012. This resulted in the largest negative CO₂ fluxes for this site in June (maximum uptake of $-17.99 \pm 0.94 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$), followed by the highest night-time CO₂ effluxes for that year ($9.46 \pm 0.23 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) after harvesting in July. The seasonal pattern of NEE for the leek crop in 2013 was characterised by (predominantly) net losses of CO₂ between April and July (for both day and night) when the near-surface peat was warm (i.e. high soil respiration rates) and the canopy had not fully developed, exposing bare peat to oxidation. This was followed by a late season period characterised by net (daytime) uptake of CO₂ (maximum uptake of $-9.9 \pm 0.86 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ in September) as the crop started to mature from August until harvest in early November (Figure 2.1.30). The celery crop (2015) attained maximum net uptake rates ($-16.74 \pm 1.21 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) in July, and was associated with the highest observed monthly average nocturnal CO₂ emission rate for this site ($10.08 \pm 0.17 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$) in August after the crop was harvested. In both 2014 and 2015 (and to a lesser extent 2012) a secondary period of daytime CO₂ uptake occurred, due to the development and photosynthesis of a secondary cover of agricultural weeds. However daytime CO₂ uptake was negligible during most winter months, whereas night-time (and sometimes daytime) CO₂ emissions occurred over the majority of the year when crops were not present. As a result, the EF-

DA site was a large and consistent source of CO₂ emissions in all measurement years, with NEE ranging from +724 (± 82) g C m⁻² yr⁻¹ in 2014 to +783 (± 84) g C m⁻² yr⁻¹ in 2015 (see Section 4).

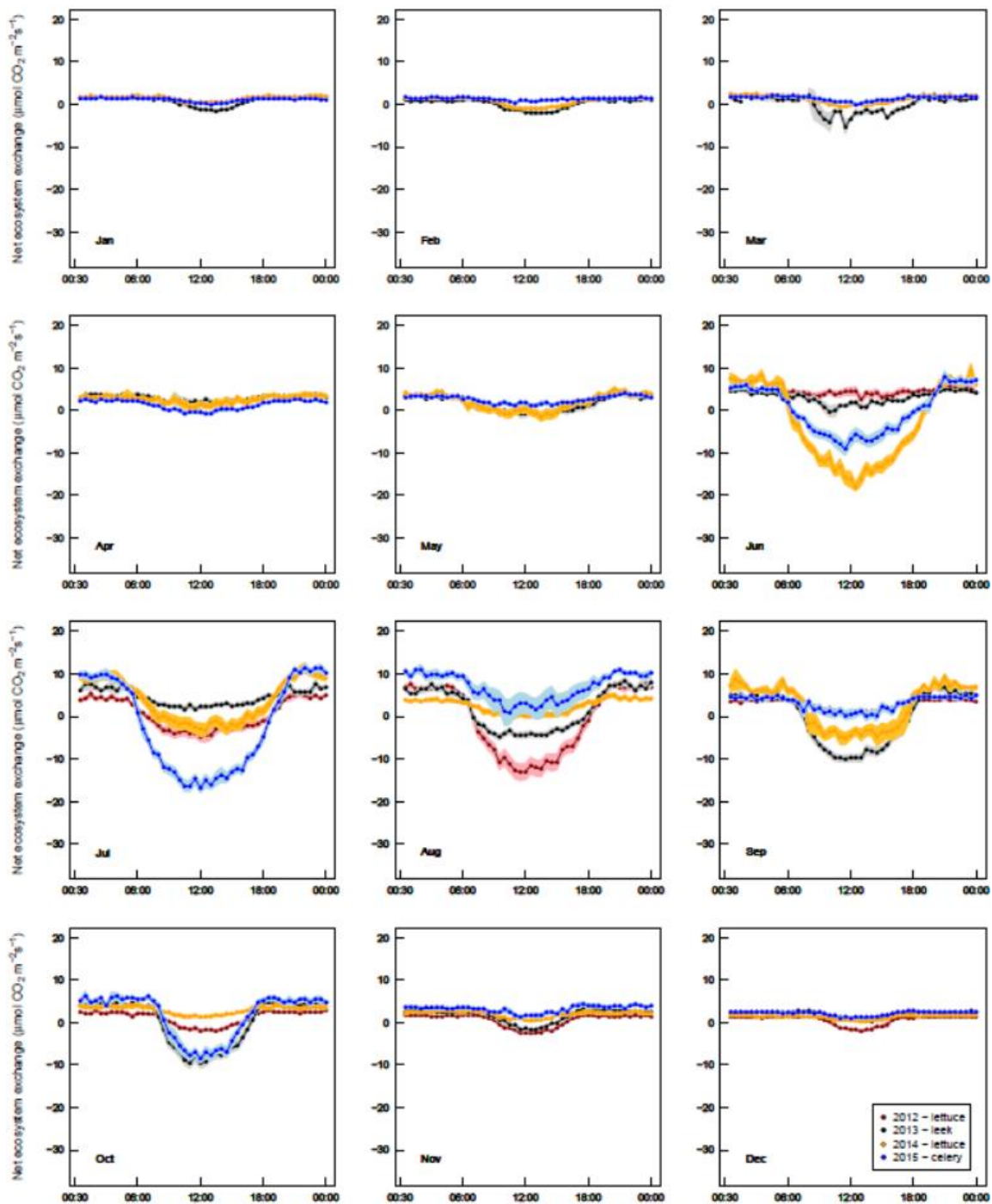


Figure 2.1.30. Mean diurnal cycles in each month that measurements were made at the EF-DA (Rosdene Farm) site. Data points are the mean of data measured at the same time of day during each month. Shaded areas show one standard error of the mean. Monthly diurnal averages calculated using measured (not gap-filled) data. Note that the cycle of agricultural land management does not follow calendar months and monthly time periods were selected for illustrative purposes and to enable comparability with other sites.

2.1.3.4. Static chamber gas fluxes

Land-atmosphere fluxes of CO₂ and CH₄ were measured using static chambers at EF-DA from March 2013 to October 2015. Six replicate collars were deployed within the crop on each sampling visit and left overnight before measurements were made on subsequent days, then removed between visits to avoid damage by agricultural machinery. The mean and range of observed dark chamber (ecosystem respiration, ER) and light chamber (net ecosystem exchange, NEE) measurements across the replicate collars are shown in Figure 2.1.31. Due to the intensive horticultural management of the site, with crops present for only a small part of the year, and different crops present in different years, it was not possible to develop an empirical model of NEE based on climate variables based on the static chamber data. However, the static chamber measurements showed a generally good fit to the eddy covariance data, with short periods of intensive CO₂ uptake during crop growth periods interrupting long periods of net CO₂ emission for the remainder of the record. These results provide support for the NEE estimates obtained from the flux tower. Furthermore, ER showed a strong exponential relationship with air temperature, which was the basis for the modelling fluxes shown in the upper panel of Figure 2.1.31. Based on this model, annual ER losses of CO₂ for EF-DA were similar in all three years (1286, 1387 and 1223 g C m⁻² yr⁻¹ in 2013, 2014 and 2015 respectively) with a three-year mean of 1299 g C m⁻² yr⁻¹. At an arable site such as EF-DA, the measured ER effectively includes heterotrophic respiration from the peat, autotrophic respiration from the plants (when present) and additional heterotrophic respiration of any crop residues left on site. Given that plant productivity and crop residue inputs are likely to have been relatively small at EF-DA, it is likely that the large majority of ER at EF-DA results from peat decomposition, and therefore that the ER estimate obtained from the static chamber data represents an upper limit for possible peat CO₂ emissions. Conversely, NEE measured by the flux towers incorporates uptake into above-ground biomass which is then removed in crop harvest (and subsequently converted to CO₂) and thus represents a lower limit for CO₂ emissions. On this basis, the mean ER value of 1299 g C m⁻² yr⁻¹ obtained from the static chambers, and the mean NEE of 691 g C m⁻² yr⁻¹ obtained from the flux tower, appear consistent.

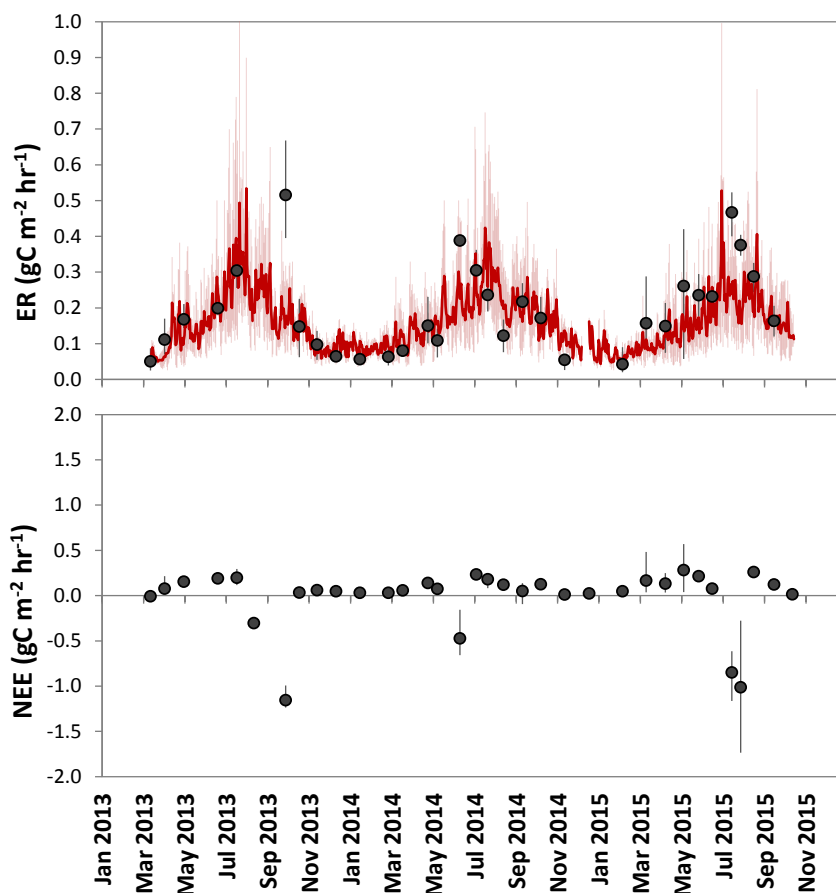


Figure 2.1.31. Modelled and observed ecosystem respiration (ER) and observed net ecosystem exchange (NEE) CO₂ fluxes at EF-DA. Continuous lines in ER plot show modelled daily mean fluxes, shading shows

modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date. NEE is strongly controlled by agricultural activities and was not therefore modelled based on climate variables.

The drained peat at EF-DA was a small but consistent net sink for CH₄. At this site it was possible to model CH₄ uptake as a function of air temperature; observed and modelled fluxes are shown in Figure 2.1.32. Overall, the model simulations suggest that the land surface at EF-DA was a net sink of 0.13 g C m⁻² yr⁻¹ for CH₄ over the three measurement years, with relatively little between-year variation.

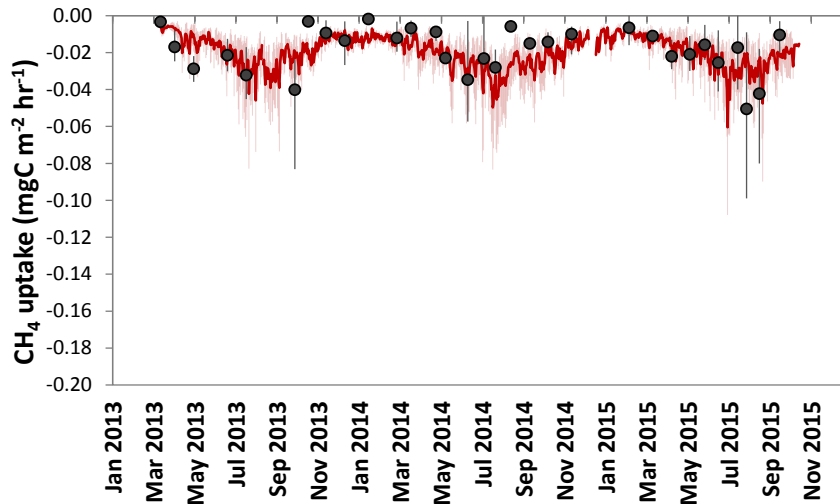


Figure 2.1.32. Modelled and observed CH₄ fluxes at EF-DA. Continuous lines show modelled daily mean fluxes, shading shows modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date.

Emissions were measured in two drainage ditches on either side of the field at EF-DA; measured fluxes are shown in Figure 2.1.33. One of the ditches (dark circles) was connected to larger drainage channels at both ends, was observed to have flowing water, and had generally low CH₄ emissions. The other ditch was blocked by a track, and tended to have stagnant water. This ditch had periodically very high CH₄ emissions in all three summer periods. Over the full period, and interpolating mean measured fluxes for the two ditches, gave a mean ditch CH₄ flux of 33 g C m⁻² yr⁻¹. The estimated fractional ditch area (Frac_{ditch}) for the field at EF-DA is 0.016, giving an area-weighted CH₄ emission from the ditches of 0.52 g C m⁻² yr⁻¹ for the site as a whole. Thus, CH₄ emissions from the ditches more than offset CH₄ uptake by the field, giving a small overall CH₄ emission from the site. Fluxes of CO₂ from the ditches were small, but mostly positive; the estimated annual flux interpolated from the measurements was 225 g C m⁻² yr⁻¹, lower than the measured emission from the field, and giving an area-weighted ditch CO₂ emission of 3.6 g C m⁻² yr⁻¹.

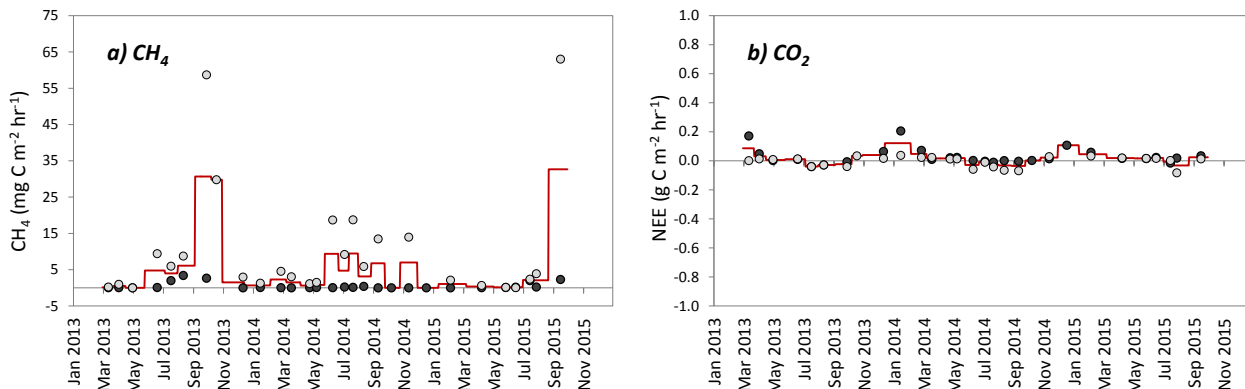


Figure 2.1.33. Observed CH₄ and CO₂ fluxes measured in ditches at EF-DA. Observations from two measurement locations are shown by dark and light circles. Red line shows interpolated fluxes.

2.1.3.5. Aquatic carbon fluxes

Ditch water DOC and DIC concentration data are shown in Figure 2.1.34. Compared to the other EF sites, DOC concentrations at EF-DA were exceptionally variable, with several very large peaks ($> 250 \text{ mg l}^{-1}$ in December 2013, $> 100 \text{ mg l}^{-1}$ in September 2014 and for a more extended period from February to March 2015). High concentrations in 2014 and 2015 coincided with periods of low or zero water flux from the site (see below) and low water-table (Figure 2.1.27) so they are associated with stagnant, high-DOC water in the ditches rather than a large carbon export flux. DIC concentrations and pH (mean 7.61) were reasonably stable, and similar to the other EF sites.

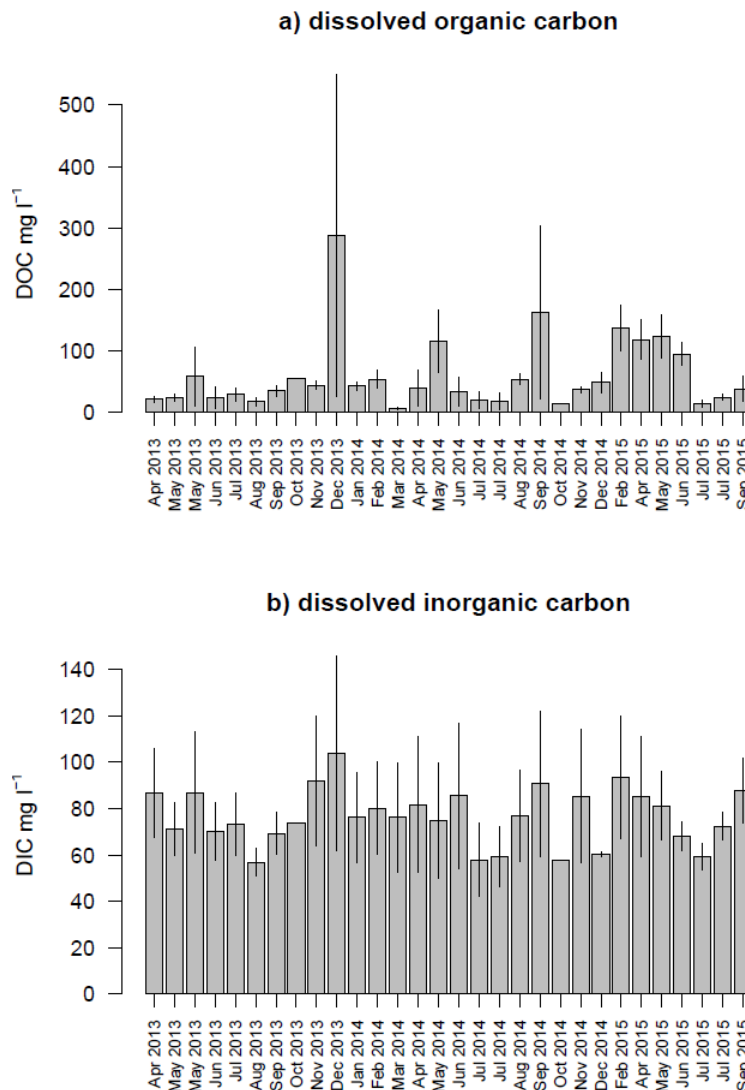


Figure 2.1.34. Mean and standard error of ditch dissolved organic and inorganic carbon concentrations, EF-DA. Note that x axis simply records sampling dates and is therefore not a true time axis.

Aquatic carbon fluxes (Figure 2.1.35) were associated with periods of water loss from the site, with peak DOC and DIC exports in winter. Estimated annual mean DOC flux was $7.9 \text{ g C m}^{-2} \text{ yr}^{-1}$, which is 76% higher than the closest 'reference' site at EF-LN, although still fairly low compared to a typical upland bog, largely due to the low water fluxes. The annual mean DIC flux was $14.2 \text{ g C m}^{-2} \text{ yr}^{-1}$, lower than the values for EF-LN and EF-EG, while the measured POC flux was similar, at $0.3 \text{ g C m}^{-2} \text{ yr}^{-1}$. Compared to the semi-natural EF sites, dissolved CO_2 made a larger contribution to the total aquatic carbon export, with an annual mean flux of $2.2 \text{ g C m}^{-2} \text{ yr}^{-1}$, the highest of any fen site. Dissolved CH_4 fluxes were also the highest recorded at a fen

site, with a mean of 0.12 g C m⁻² yr⁻¹; this appears consistent with the relatively high vertical CH₄ fluxes from the drainage ditches measured by floating chambers.

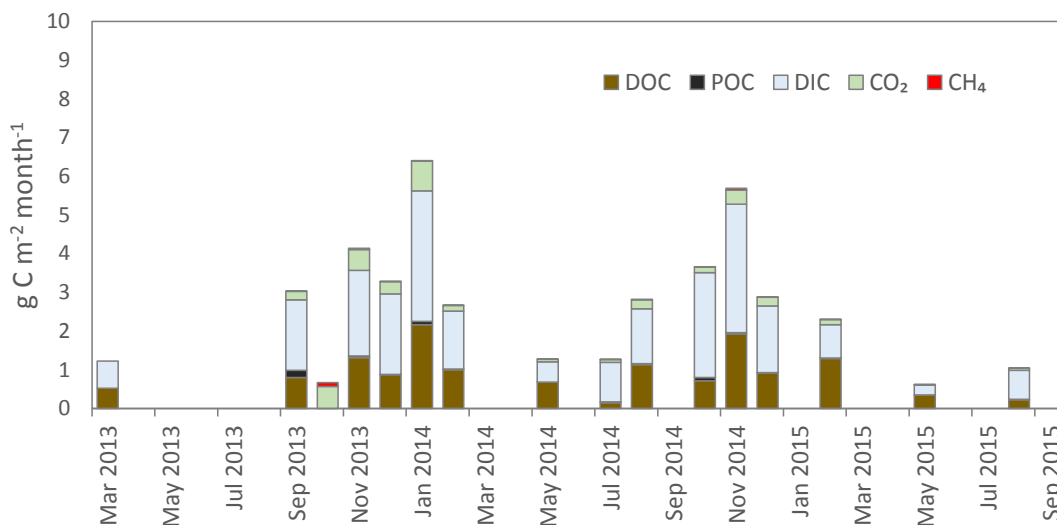


Figure 2.1.35. Estimated monthly aquatic carbon fluxes, EF-DA. Data missing for March 2013 (CO₂ only), October 2013 and March 2015 (DOC, DIC and POC) and January 2015 (all determinands); other zero values indicate no flux.

2.1.4. Redmere Farm – arable on shallow peat (EF-SA)

Redmere Farm, which is also owned by G’s Fresh, is around 10 km Southeast of Rosedene Farm. It has been drained and intensively farmed for a longer period than at EF-DA, and the soil surface is believed to have fallen by more than 2 m since agricultural production commenced. The landscape at EF-SA is more open than at EF-DA, with larger fields, a consequently lower drainage ditch density, and no tree shelter belts. As at EF-DA, a network of subsurface pipes is used to maintain drainage across the site, but irrigation is applied above-ground. The study field was used to grow a wheat crop in 2013, two lettuce crops in 2014, and a maize crop in 2015. Biomass inputs and offtakes have been estimated from a combination of biomass sampling from the site (maize), data from EF-DA (lettuce) and published yield estimates (wheat).

Core data from EF-SA (Figure 2.1.36) indicate that the bulk density of the near-surface peat is higher than at EF-DA (0.62 vs 0.50 g cm⁻³), carbon content lower (31 vs 44%) and mineral content approximately double. The C/N ratios are similar. The depth of peat at EF-SA is around 75 cm, with very high mineral contents and bulk densities indicating a true mineral soil below this depth, although carbon content begins to decline below 30 cm depth. As such, the site can be considered (as also suggested by the location map, Figure 2.1.1) to be close to the transition from ‘shallow’ peat to ‘wasted’ peat.

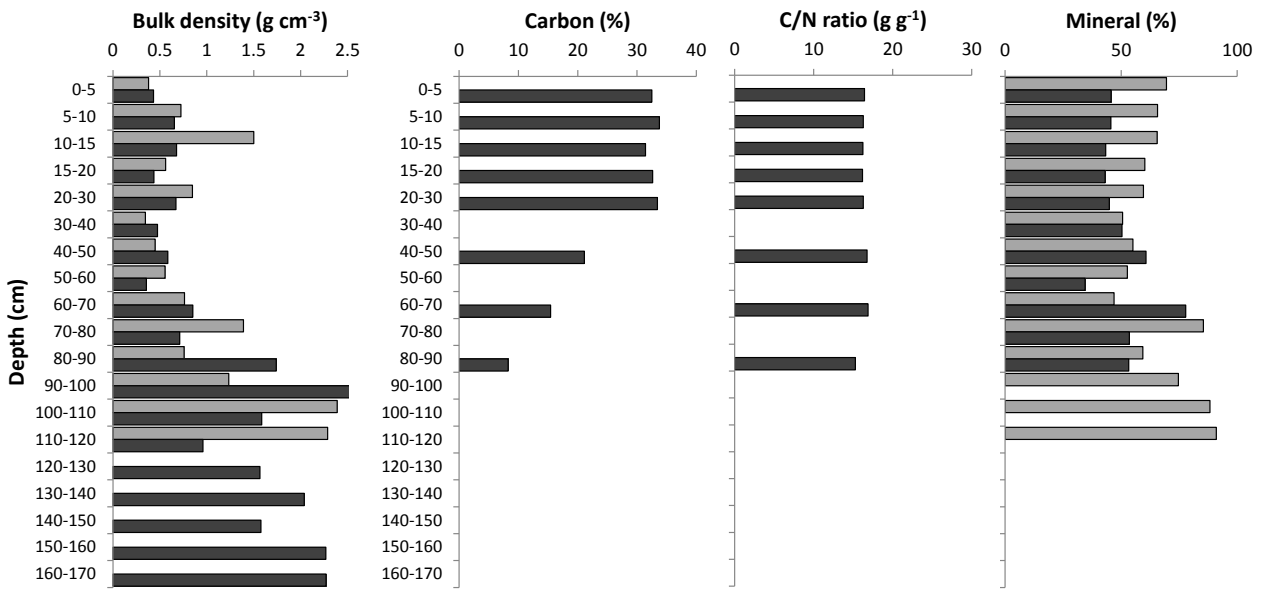


Figure 2.1.36. Peat core data, EF-SA. Data from two cores analysed from the site are represented by dark and light grey-shaded bars.

2.1.4.1. Meteorology

Figure 2.1.37 shows monthly SW_{in} totals and monthly mean air temperature measured at EF-SA. Monthly precipitation sums have been omitted due to periods of missing data. As EF-SA is located close to EF-DA (approximately 11 km), the monthly and annual precipitation sums presented above for EF-DA can be taken as being representative for EF-SA. Total monthly SW_{in} and mean air temperature show the same seasonal patterns to other sites in East Anglia and so only a brief summary of these variables is provided here. As at other EF sites, wind flow comes from all compass directions with the prevailing wind direction from the southwest. Higher mean wind speeds at this location compared to EF-DA (compare Figures 2.1.37 and 2.1.26) partly reflect a higher measurement height at EF-SA, and may also be associated with the lack of tree shelter belts in the surrounding farmland.

Monthly totals of SW_{in} are shown in the top panel of Figure 2.1.37. As for EF-DA, SW_{in} peaked in July in 2013 (360 k W m^{-2}) and 2014 (328 k W m^{-2}), and in June in 2015 (325 k W m^{-2}). Notable between year differences in SW_{in} were observed for March (highest in 2014; lowest in 2013) and April (highest in 2015; lowest in 2014) and for July and August (lowest in 2012; highest in 2013). Mean annual air temperature measured at EF-SA was $9.4 \text{ }^{\circ}\text{C}$ in 2013, $11.2 \text{ }^{\circ}\text{C}$ in 2014 and $10.7 \text{ }^{\circ}\text{C}$ in 2015.

Noteworthy between year differences in monthly average air temperatures were the January to June period (lowest in 2013 and highest in 2014), differences during September and October (warmest in 2014, lowest in 2015) and the large difference for December. In 2015, the mean December air temperature was $10.4 \text{ }^{\circ}\text{C}$, compared to 4.8 , 5.9 and $5.3 \text{ }^{\circ}\text{C}$ in 2012-2014.

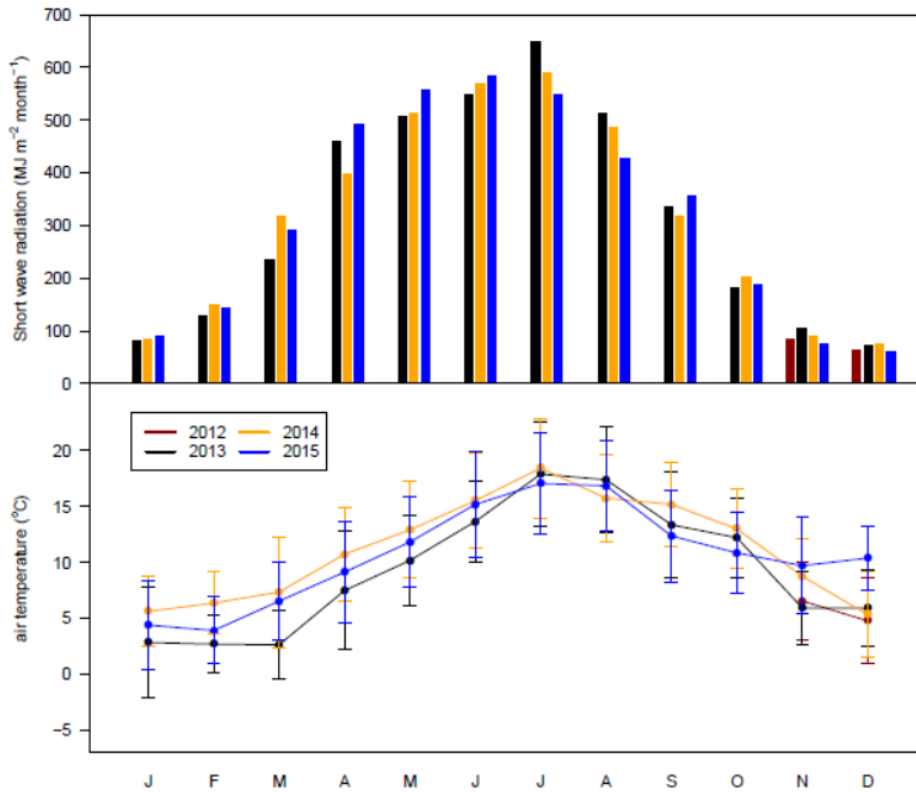


Figure 2.1.37. Total monthly incoming solar radiation (top) and mean monthly air temperature (lower) for the EF-SA site (Redmere Farm). Error bars on the temperature plot show one standard deviation of the mean.

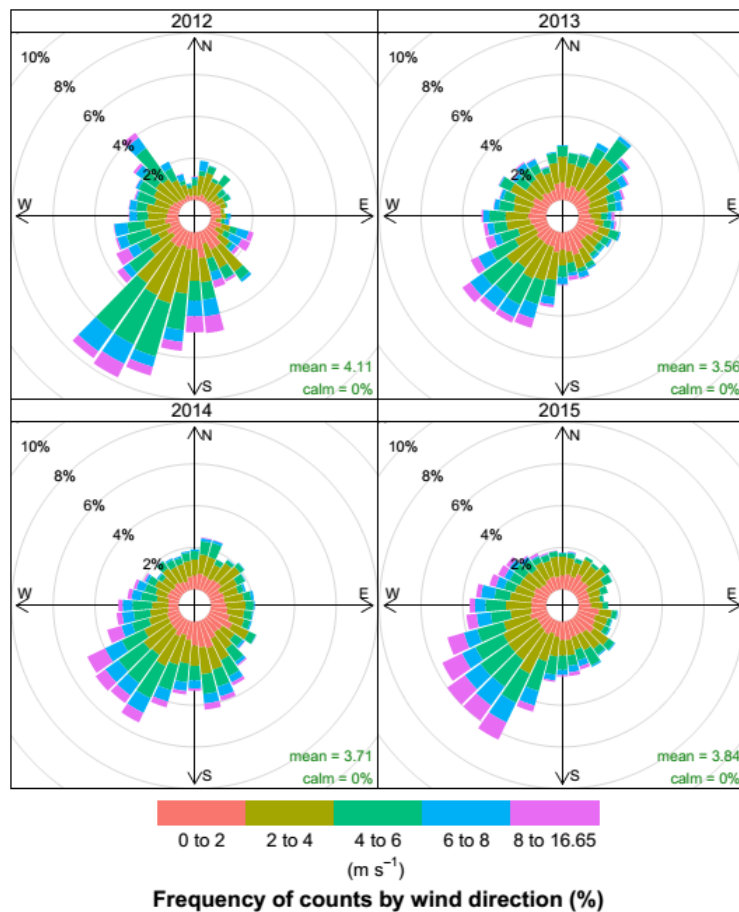


Figure 2.1.38. Wind rose plots showing wind direction and wind speed at the EF-SA (Redmere Farm) site.

2.1.4.2. Hydrology

As at EF-DA, a single water-table logger was installed at the edge of the EF-SA site in December 2013 (Figure 2.1.39). Since that time, water-table levels have fluctuated between 60 and 100 cm below the surface, although (unlike the EF-DA site) the water table rarely fell below the 1 m base of the dipwells. This may simply result from the shallower depth of peat (and hence shallower ditches) at EF-SA compared to EF-DA. The temporal variability in water level was, however, broadly similar to that at EF-DA, with a sustained period of slightly higher water table in winter, and more sporadically raised water levels during the growing season. Water table data show substantial water table drawdown throughout the record, with slightly higher levels associated with winter wet periods and summer irrigation period. At most times, the water table was higher than the ditch level, implying flow of water out of the peat and into the ditch network. Water losses from the site (Figure 2.1.40) are thought to have occurred from around October to March in all winter periods, with sporadic losses also occurring during the summer growing season. These may partly have been associated with spray irrigation; this water input was not quantified, although as water was extracted from the ditches it is difficult to determine the extent to which this represented 'new' water input to the site. Mean discharge from the site was estimated at 199 mm yr⁻¹.

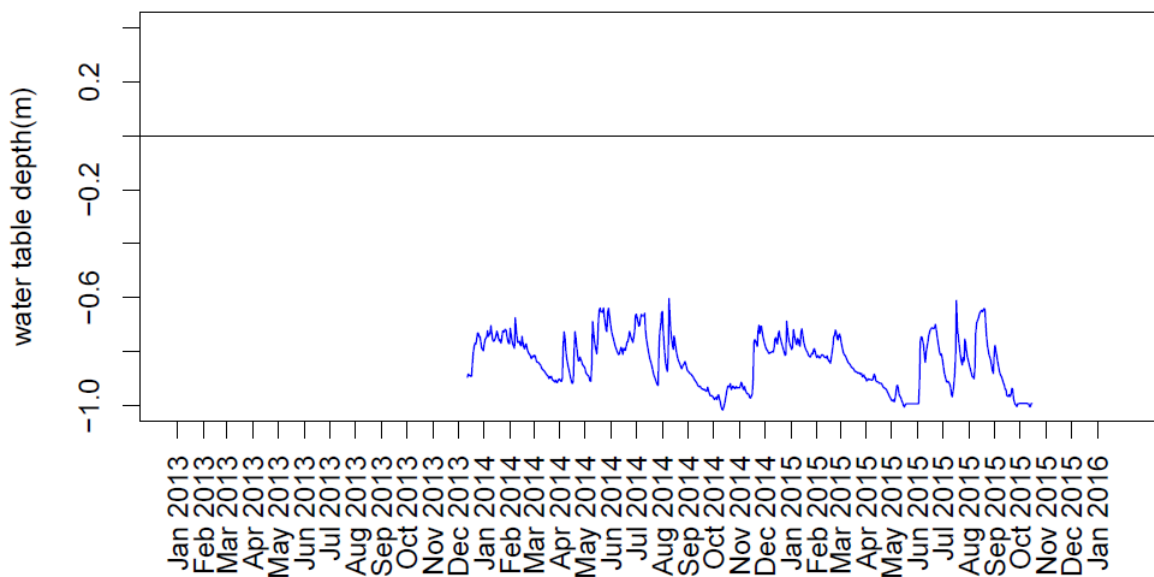


Figure 2.1.39. Continuously monitored water-table data, EF-SA.

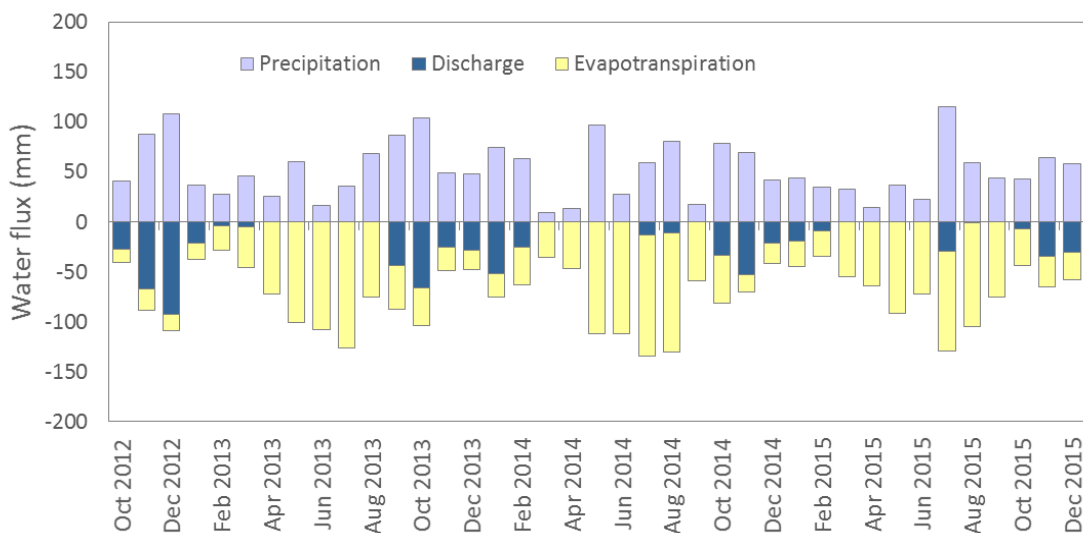


Figure 2.1.40. Monthly hydrological budgets for EF-SA.

2.1.4.3. Eddy covariance gas fluxes

Fingerprint plots of measured and gap-filled NEE for the EF-SA site are presented in Figure 2.1.41. Data capture (after QC) at EF-SA has been good since measurements commenced in October 2012. A few short periods of system failure were experienced due to issues with electrical power, most notably for nocturnal periods at the end of 2013 (upper panel of Figure 2.1.41). The EC measurements at EF-SA have captured CO₂ fluxes during a wheat (*Triticum aestivum*) crop in 2013, two Iceberg lettuce (*Lactuca sativa*) crops in May–June and August–September in 2014, and a corn (*Zea Maize*) crop in 2015. The different crops can be clearly identified as short periods of negative NEE (blue) against times of net CO₂ loss (orange and red). Differences in agricultural land management between years clearly had a large influence on measured CO₂ fluxes at this site, as can also be seen in the monthly mean diurnal cycles (Figure 2.1.42).

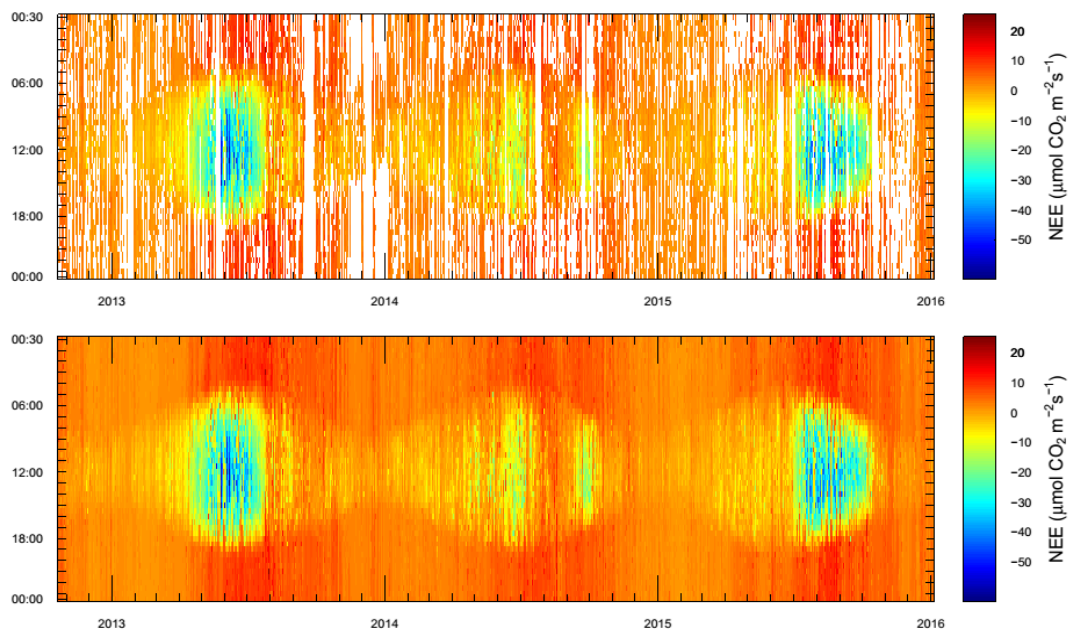


Figure 2.1.41. Fingerprint plot of measured (top panel) and measured and gap-filled (lower panel) net ecosystem carbon dioxide exchange for the EF-SA (Redmere Farm) site. Units are in $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$.

The mean diurnal cycles shown in Figure 2.1.42 illustrate large between-year differences in NEE for the different crops at EF-SA. In particular, net CO₂ uptake and release rates were highest at EF-SA during the spring and early summer for the wheat crop in 2013, whereas the maize crop in 2015 had the highest net uptake rates between July and October and the highest nocturnal losses between August and November. The highest monthly average daytime (uptake) CO₂ flux values for the wheat (maximum of $-31.92 \pm 2.09 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ in June 2013) and maize (maximum of $-30.1 \pm 1.84 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ in August 2015) crops are almost three times higher than those measured for lettuce in 2014 (maximum of $-11.96 \pm 1.03 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ in June 2014) and are the highest (most negative) CO₂ flux densities measured at any of the sites in the project flux tower network. Maximum nocturnal CO₂ losses were also higher for the wheat (9.43 ± 0.1 and $9.53 \pm 0.17 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ in July and August 2013, respectively) and maize ($9.98 \pm 0.16 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ in August 2015) than for the lettuce crops (maximum of $8.26 \pm 0.23 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ in July) in 2014, which can likely be explained (in part) by higher rates of maintenance respiration (CO₂ release from plants) of wheat and maize compared to lettuce.

As a result of the large differences in observed CO₂ uptake by the cereal versus horticultural crops, NEE at EF-SA was highly variable between years, with a measure NEE of $-20 \text{ g C m}^{-2} \text{ yr}^{-1}$ in 2013 (wheat), $+678 \text{ g C m}^{-2} \text{ yr}^{-1}$ in 2014 (lettuce) and $+90 \text{ g C m}^{-2} \text{ yr}^{-1}$ in 2015 (maize). These NEE values should not be interpreted directly as measures of net carbon balance, because a large part of the photosynthetic uptake by the cereal crops was subsequently removed during the harvest. However, the relatively high degree of soil disturbance associated with the production of two lettuce crops, shorter periods of crop cover and larger amount of the

soil surface that remains exposed to solar radiation (and wind-related loss) compared to the more closed canopies of cereal crops, could have led to higher rates of overall C loss. These issues are discussed further in Section 4.

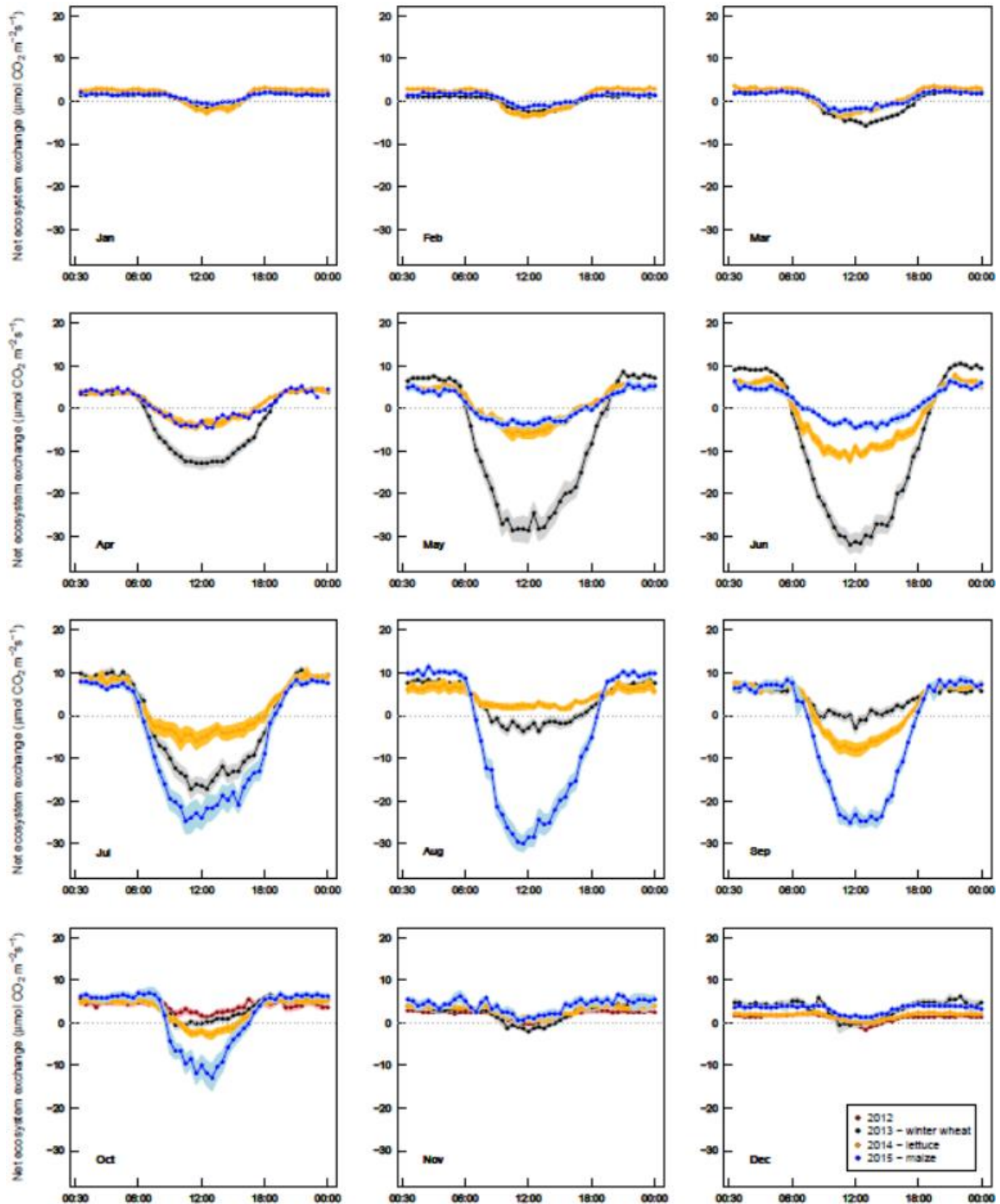


Figure 2.1.42. Mean diurnal cycles for each month that measurements were available for the EF-SA (Redmere Farm) site. Data points are the mean of data measured at the same time of day during each month. Shaded areas show one standard error of the mean. Monthly diurnal averages calculated using measured (not gap-filled) data. Units are in $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$. Note that the cycle of agricultural land management does not follow calendar months and monthly time periods were selected for illustrate purposes and comparability with other sites.

2.1.4.4. Static chamber gas fluxes

Static chamber measurements from EF-SA, made from March 2013 to October 2015, and following the same sampling design described above for EF-DA, are shown in Figure 2.1.43. Again, the intensive agricultural management of the site precluded the development of an empirical model to describe GPP or therefore NEE, but ER was successfully modelled as an exponential function of air temperature. Large deviations between modelled and observed values (positive outlier observations in lower panel of Figure 2.1.43) coincided with periods of intensive wheat and maize growth (when autotrophic respiration by the growing crops was contributing to observed ER, but not captured by the simple temperature model) and a single measurement point in November 2014 that occurred directly after the field had been disked, which clearly led to a very large short-term pulse of CO₂ emissions. Outside of crop growth periods, the fit between flux tower and static chamber modelled ER was very good. The static chamber-based model therefore appears to provide a good indication of the underlying rate of heterotrophic respiration of the peat at EF-SA. Modelled annual fluxes are 1549, 1702 and 1515 g C m⁻² yr⁻¹ for 2013, 2014 and 2015, giving a three-year mean flux of 1589 g C m⁻² yr⁻¹. This is 22% higher than the equivalent flux at EF-DA, despite EF-DA being on deeper and more organic-matter rich peat.

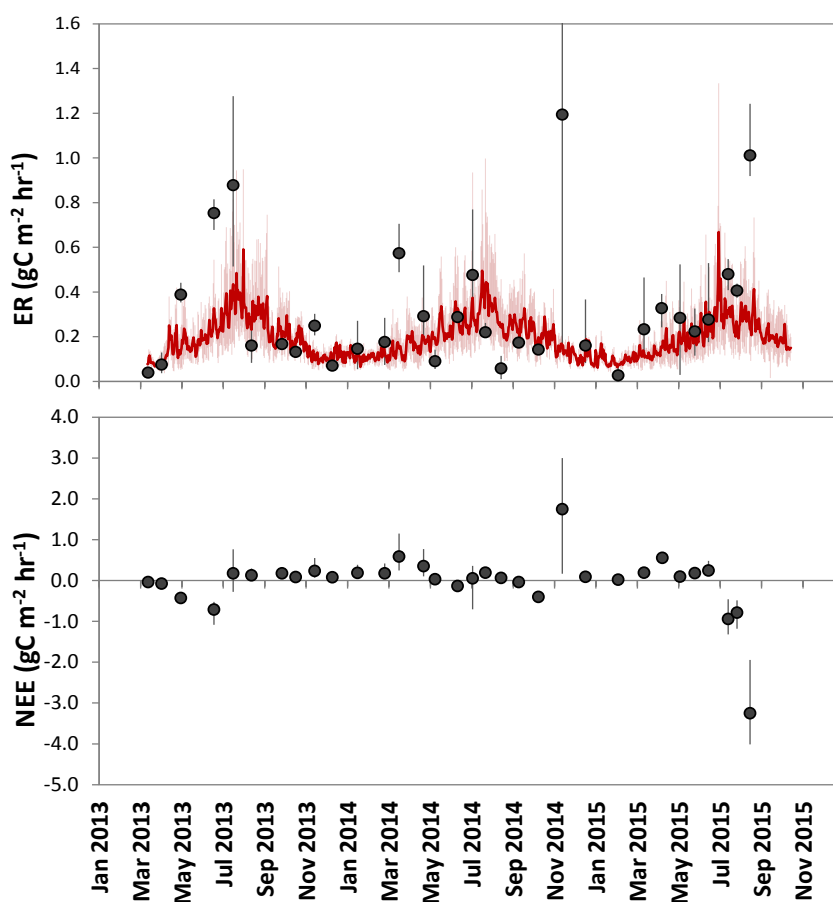


Figure 2.1.43. Modelled and observed ecosystem respiration (ER) and observed net ecosystem exchange (NEE) CO₂ fluxes at EF-SA. Continuous lines in ER plot show modelled daily mean fluxes, shading shows modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date. NEE is strongly controlled by agricultural activities and was not therefore modelled based on climate variables.

The soil at EF-SA was a consistent net sink for CH₄, which showed a positive correlation with air temperature and could therefore be modelled (Figure 2.1.44). Over the three year period, net CH₄ uptake to the field was 0.29 g C m⁻² yr⁻¹, larger than that at EF-DA. Additionally, emissions of CH₄ from the ditches adjacent to EF-SA (Figure 2.1.45) were much lower, with a mean for the study period of 0.84 g C m⁻² yr⁻¹. The larger field at EF-SA also resulted in a much lower *Frac*_{ditch}, of just 0.006, as a result of which the area-weighted ditch

emissions for the site as a whole were only $0.01 \text{ g C m}^{-2} \text{ yr}^{-1}$. The reasons for the much lower ditch CH_4 emission compared to EF-DA are uncertain, but may be related to differences in the characteristics of the ditches, which are incised into underlying mineral soil at EF-DA, contain little vegetation, contain relatively acidic water (see below) and are also affected by visible deposits of iron oxide. The ditches were consistent net sources of CO_2 , with a calculated annual mean of $652 \text{ g C m}^{-2} \text{ yr}^{-1}$ for the ditch area, similar to the rate of CO_2 emission from the adjacent field, and giving an area-weighted ditch emission for the site as a whole of $3.9 \text{ g C m}^{-2} \text{ yr}^{-1}$.

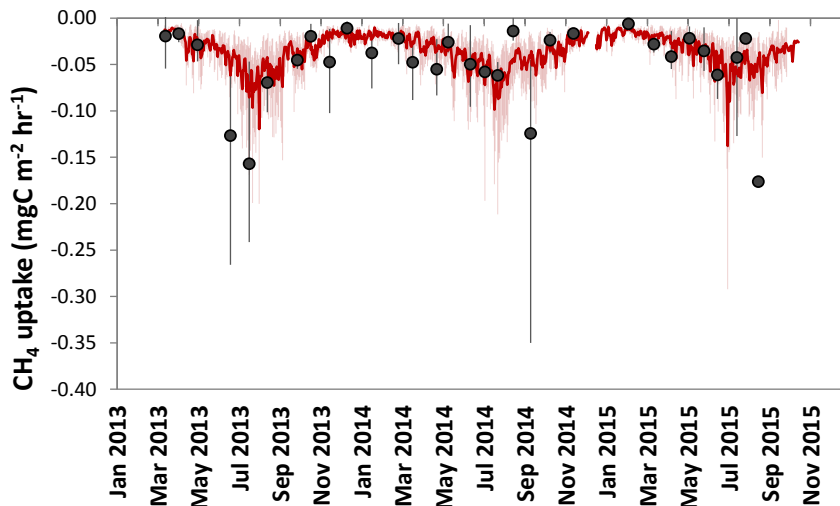


Figure 2.1.44. Modelled and observed CH_4 fluxes at EF-SA. Continuous lines show modelled daily mean fluxes, shading shows modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date.

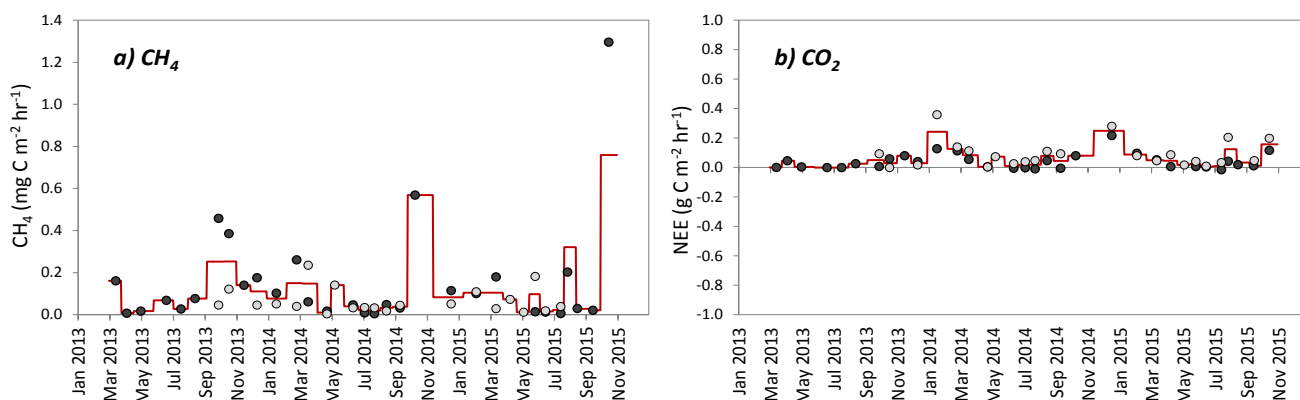


Figure 2.1.45. Observed CH_4 and CO_2 fluxes measured in ditches at EF-SA. Observations from two measurement locations are shown by dark and light circles. Red line shows interpolated fluxes.

2.1.4.5. Aquatic carbon fluxes

Ditch water data for EF-SA (Figure 2.1.46) shows some striking temporal variations, with both DIC and DOC both dropping to near-zero concentrations in January-March 2014, which coincided with a dramatic decline in pH from a peak of > 7 in summer to a minimum below 4. These changes appear to be attributable to the oxidation of reduced sulphur compounds in the peat as a result of drainage, which can lead to sulphate leaching, acidification, degassing of DIC as CO_2 , and suppression of DOC solubility (e.g. Evans et al., 2014). These processes are characteristic of 'acid sulphate' soils where drainage exposes reduced sulphur compounds to oxidation, and can have significant detrimental impacts on agricultural productivity. However, this phenomenon did not recur during the remainder of the measurement period, and DOC concentrations in particular were considerably higher during 2015 than in 2013-14.

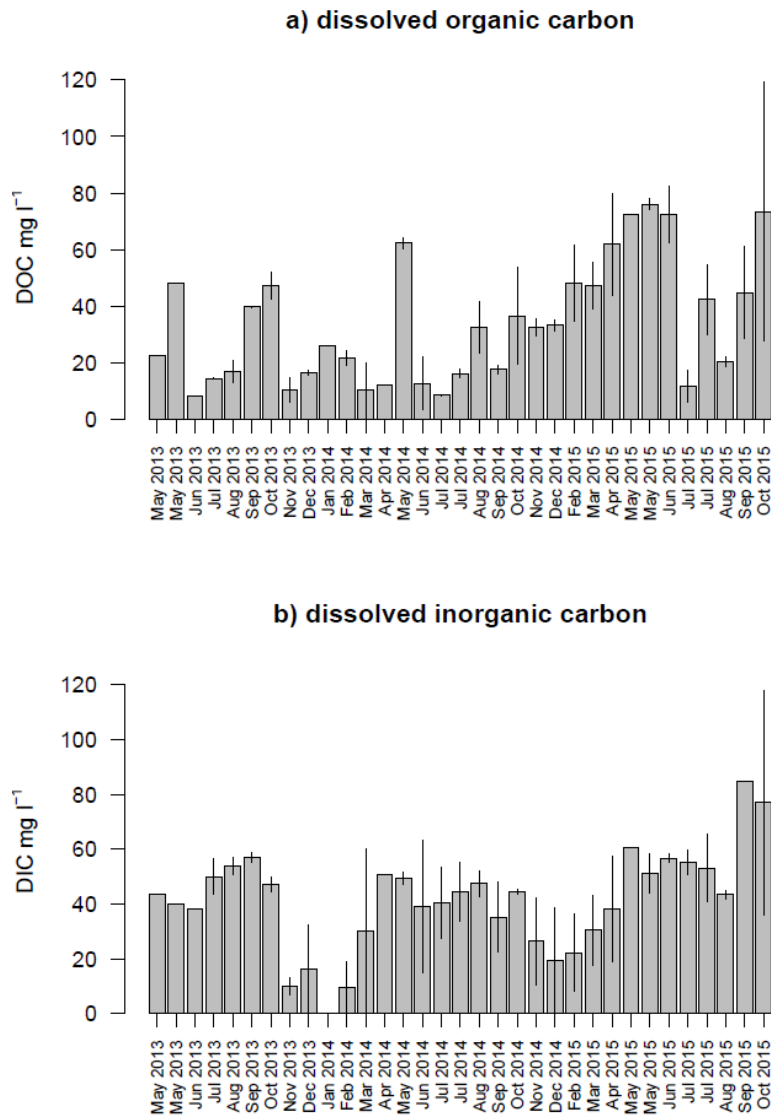


Figure 2.1.46. Mean and standard error of ditch dissolved organic and inorganic carbon concentrations, EF-SA. Note that x axis simply records sampling dates and is therefore not a true time axis.

Estimated aquatic carbon fluxes from EF-SA (Figure 2.1.47) show similar seasonal and short-term variability to the other EF sites, with peak fluxes in autumn and winter. Export of both DOC and DIC were highest in September-October 2013, and a significant pulse of POC loss was also measured at this time. Overall, DIC fluxes were smaller than at the other EF sites, with an estimated annual mean flux of $3.9 \text{ g C m}^{-2} \text{ yr}^{-1}$. The DOC flux was $6.6 \text{ g C m}^{-2} \text{ yr}^{-1}$, 46% higher than at EF-LN but lower than at EF-DA. The mean POC flux was $2.2 \text{ g C m}^{-2} \text{ yr}^{-1}$, which was higher than the other EF sites, although quite strongly influenced by the September 2013 pulse. Dissolved CO_2 fluxes were very similar to EF-DA, with a mean of $2.1 \text{ g C m}^{-2} \text{ yr}^{-1}$, but dissolved CH_4 fluxes were (as for the vertical floating chamber ditch fluxes) much lower (mean $0.02 \text{ g C m}^{-2} \text{ yr}^{-1}$).

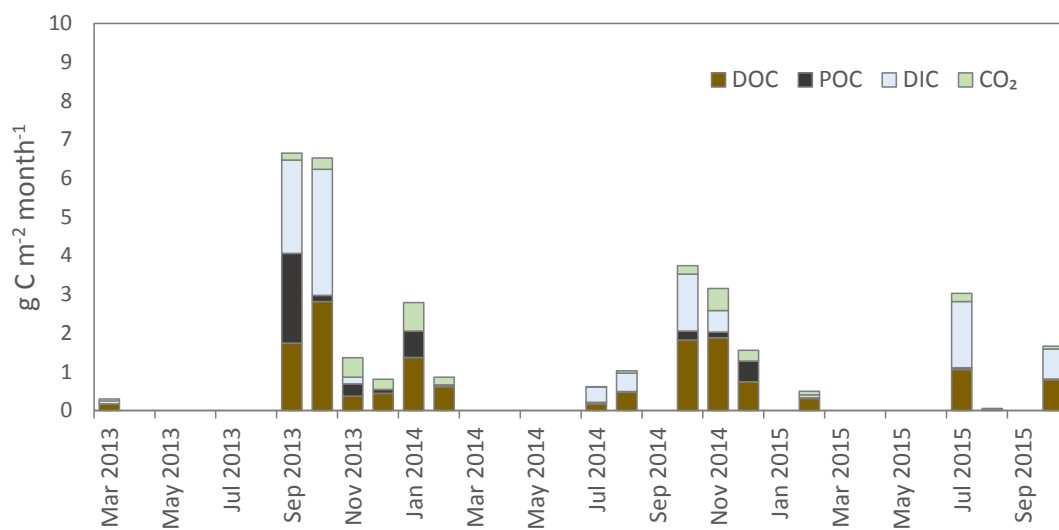


Figure 2.1.47. Estimated monthly aquatic carbon fluxes, EF-SA. Data missing for April 2013 and January 2015; all other zero values indicate no flux.

2.2. Manchester Mosses

The Manchester Mosses form part of the Lancashire Mosslands, which together held a large proportion of the UK's original lowland raised bog. Bragg et al. (1984) estimated that, in the mid-1800s, the total lowland mire area in Lancashire was around 4300 ha, with a further 1800 ha in South Cumbria. The Lancashire Wildlife Trust have estimated that the original area of raised bog across Lancashire, Greater Manchester and North Merseyside may have exceeded 28,000 ha (Chris Miller, pers. comm.), of which 392 ha retains some form of bog habitat, and a similar area is now under peat extraction. The Chat Moss complex, within which all of the study sites are located (Figure 2.2.1), is thought to have originally been the second largest area of lowland raised bog in the UK, with an original extent of around 3570 ha. In the early 18th century, Daniel Defoe noted that: "the surface, at a distance, looks black and dirty, and is indeed frightful to think of, for it will bear neither horse or man", adding that "We saw it in some places eight or nine foot thick, and the water that drains from it look'd clear, but of a deep brown, like stale beer. What nature meant by such a useless production, 'tis hard to imagine" (Defoe, 1727). In 1829, George Stephenson's Liverpool and Manchester railway was built across the bog by 'floating' the line on a layer of branches. At around this time, the large-scale drainage and conversion of Chat Moss, and of the other Lancashire Mosses began, with the majority of the land converted to productive farmland. Between 1895 and 1923, Chat Moss was also used as a disposal site for human waste from the growing population of Manchester. By the late 1970s, Bragg et al. (1984) estimated that over 99.5% of the original bog area had been lost, and the remaining raised bog area occupies 115 ha whilst active or former extraction sites cover 281 ha. The area of the Manchester Mosses under farmland is now used mainly to grow arable crops and for turf production. Significant areas are also under woodland.

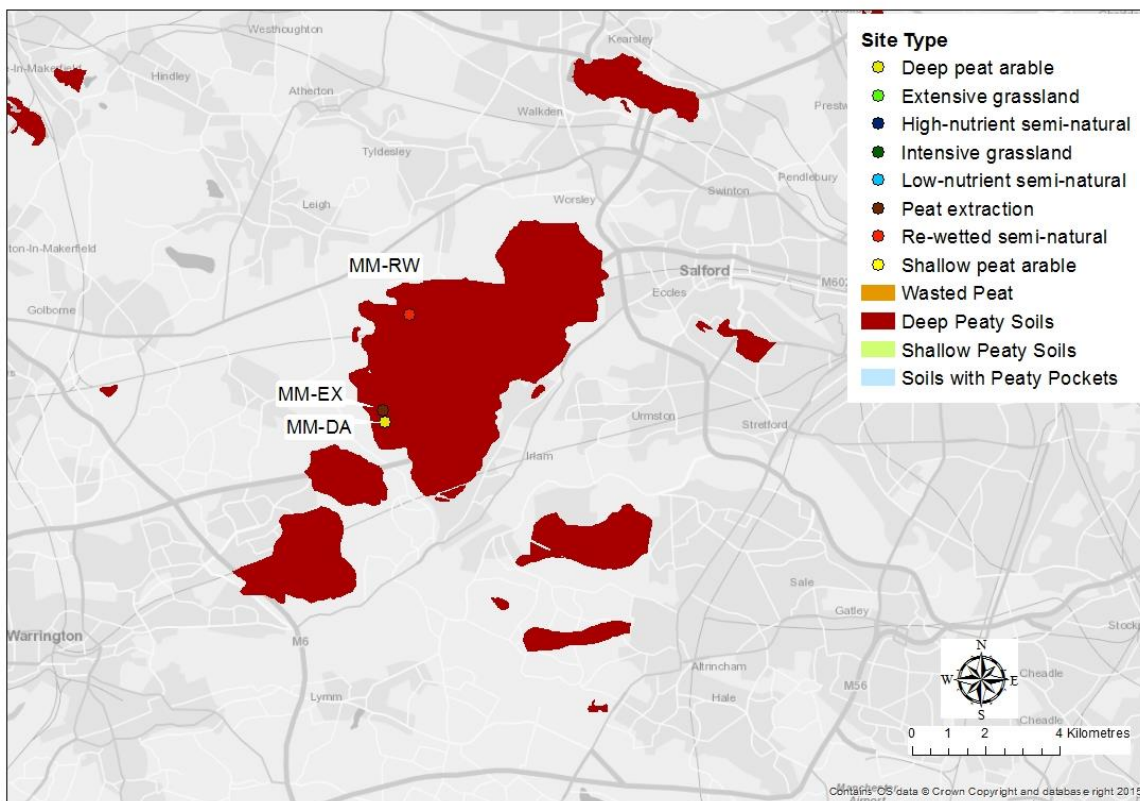


Figure 2.2.1. Manchester Mosses study area.

Given their close proximity, meteorology for all three Manchester Mosses sites was represented by a single AWS at the MM-RW site. Monthly plots of the main meteorological variables are shown in Figure 2.2.2 (data

supplied by E.ON, collected as part of a terrestrial ecosystem monitoring project on the site). Wind flow at this location comes from all directions excluding the east (Figure 2.2.3). The prevailing wind direction is from the south and southwest.

Photosynthetically active radiation (PAR) showed large seasonal and between-year variability at the MM sites. Monthly averaged PAR values peaked at 432 $\mu\text{mol photons m}^{-2} \text{s}^{-1}$, 405 $\mu\text{mol photons m}^{-2} \text{s}^{-1}$ and 405 $\mu\text{mol photons m}^{-2} \text{s}^{-1}$ in 2013, 2014 and 2015, respectively (top panel). The largest differences in monthly averaged PAR were observed for April (highest in 2015, lowest in 2013) and July (highest in 2013 and lowest in 2015), and September (highest in 2015 lowest in 2013) and October (highest in 2015).

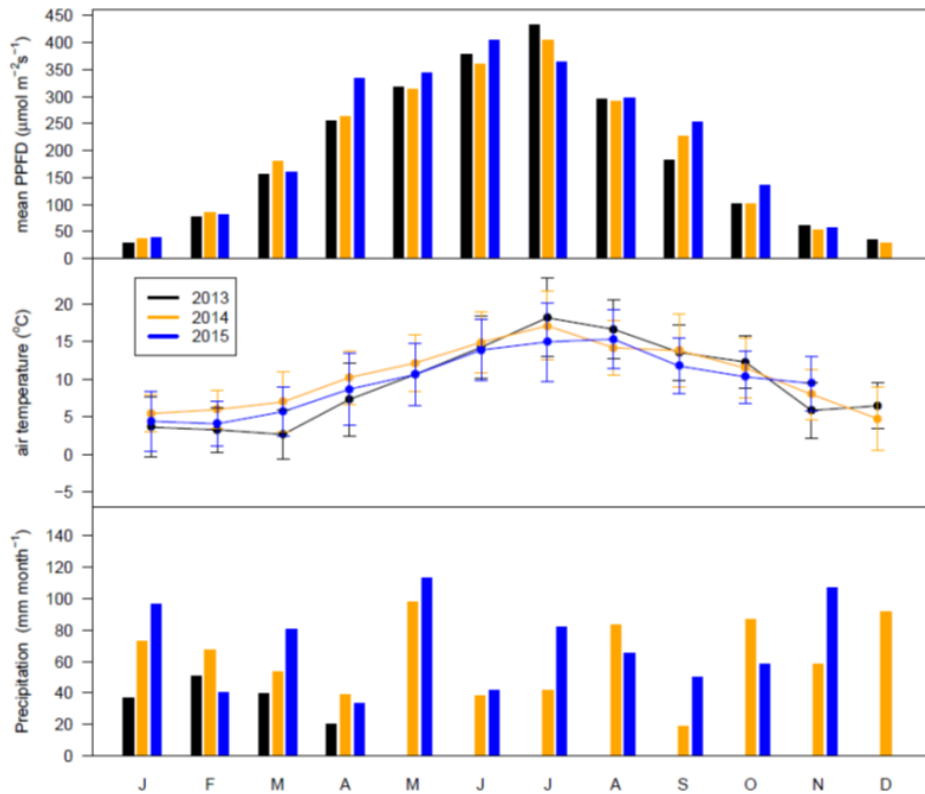


Figure 2.2.2. Total monthly incoming photosynthetic photon flux density (PAR, top) mean monthly air temperature (middle) and total monthly precipitation (lower) for the MM-RW site (Astley Moss). Error bars on the temperature plot show one standard deviation of the mean. Data supplied by E.ON.

Mean annual air temperature was 9.6 °C for 2013 and 10.4 °C in 2014. Missing data for December 2015 preclude calculation of the mean annual temperature for 2015 (middle panel). Notable between-year differences in air temperature were the lower summer peak air temperatures in 2015 compared to 2013 and 2014, as well as the warmer conditions of the January to May period during early 2014 compared to other years. July was the warmest month of 2013 and 2014, whereas August was the warmest month of 2015.

The annual precipitation total for 2013 was 748 mm yr⁻¹ at this location. The precipitation total between January and November 2015 was 767 mm. The wettest months at MM-RW were May and December in 2014, and May and November (106.94 mm month⁻¹) in 2015. The driest months were September in 2014 and April in 2015.

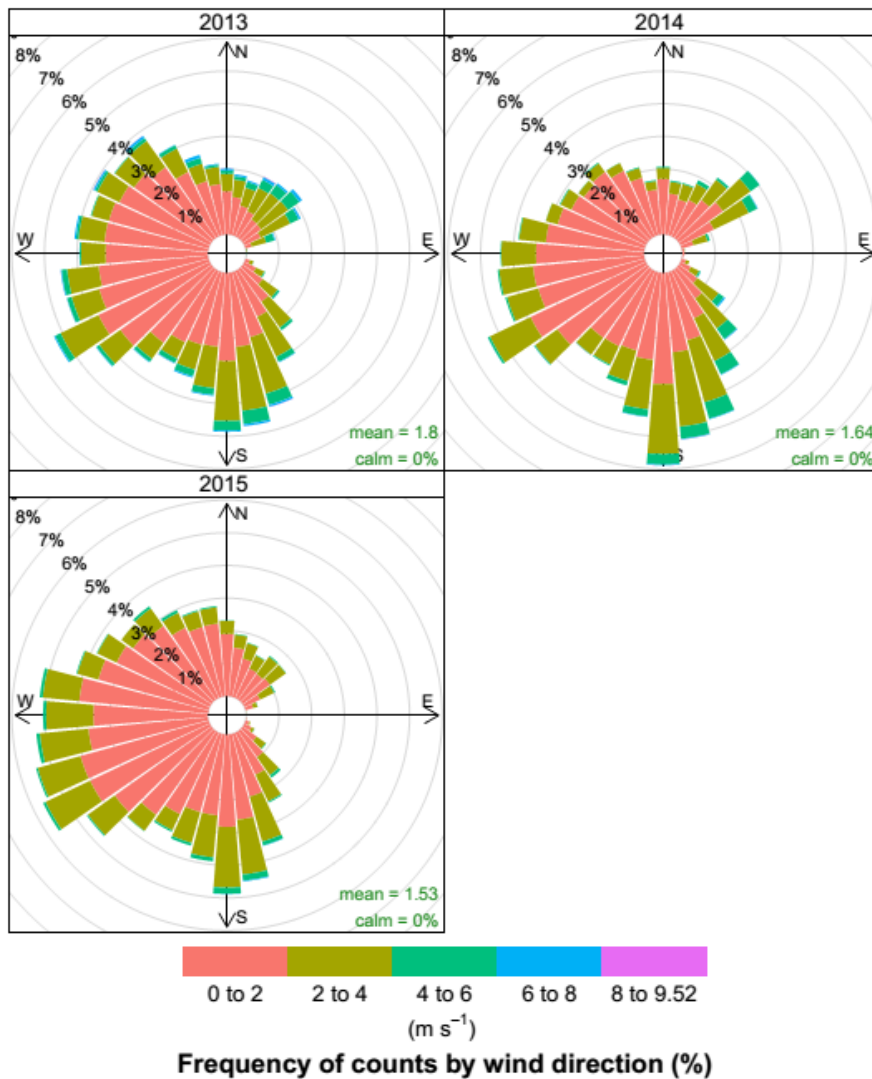


Figure 2.2.3. Wind rose plots showing wind direction and wind speed at the MM-RW (Astley Moss) site.

2.2.1. Astley Moss – re-wetted raised bog (MM-RW)

Astley Moss is the largest, and one of the best preserved, surviving fragments of the Manchester Mosses, with an area of 35 ha. It forms part of a 92 ha SSSI/SAC and is managed by the Lancashire Wildlife Trust (LWT). The area was formerly used for peat cutting and retains a ridged topography. It has been re-wetted incrementally through the creation of peat bunds, some of which are plastic-lined, and the study area was re-wetted in 2009-10. The use of bunds effectively isolates the bog from surrounding drained and subsidence-affected farmland, and has led to the partial inundation of the site. The re-wetted area which was previously dominated by *Molinia caerulea*, now supports a tussocky vegetation community of *Molinia caerulea* plus *Eriophorum angustifolium*, *Eriophorum vaginatum*, *Sphagnum fimbriatum* and *Sphagnum subnitens* (NVC classes M19a/M20). Wetter ‘lawn’ areas support a *Sphagnum cuspidatum* bog pool community (NVC M2), whilst drier areas (including the bund) support a damp heath community (NVC H9e) dominated by *Molinia caerulea*, *Calluna vulgaris* and *Betula pubescens* saplings (see Appendix 3). Scrub and birch woodland occur around the site, but some tree-clearance has taken place as part of ongoing conservation work.

The peat at MM-RW remains fairly deep (around 1.5 to 3 m) and retains the characteristics of a bog peat, with low bulk density, low mineral content, high carbon content and high C/N ratio (Figure 2.2.4). The peat is highly acid (mean 0-50 cm pH 2.66) which is the lowest of any of the study sites (Table 2.2)

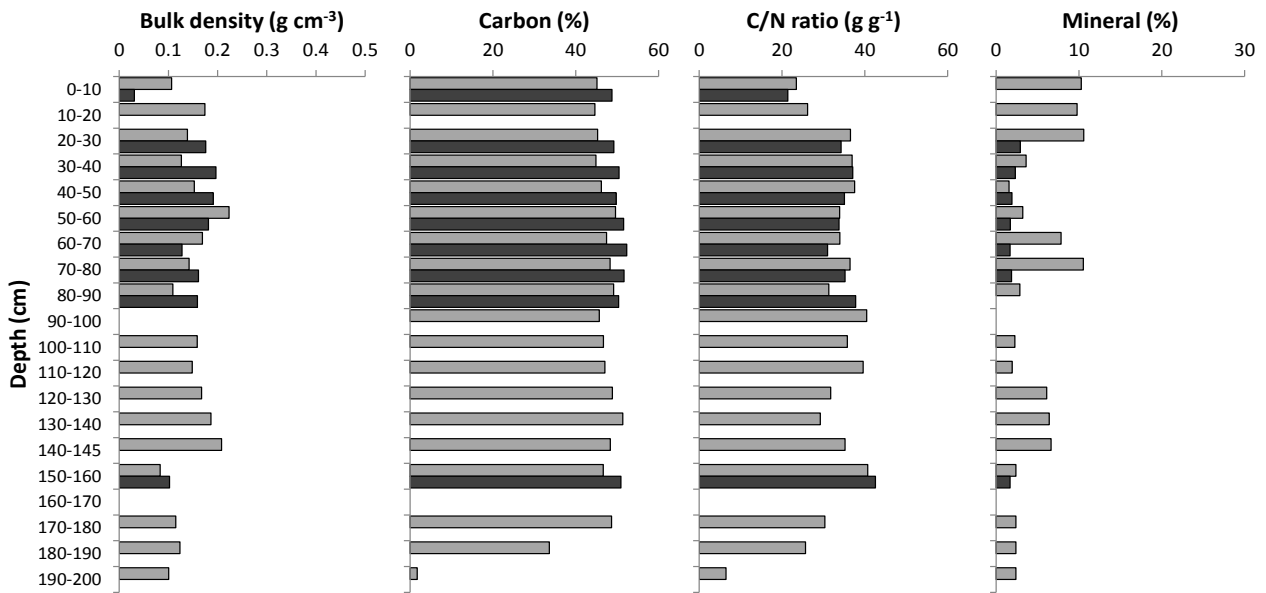


Figure 2.2.4. Peat core data, MM-RW. Data from two cores analysed from the site are represented by dark and light grey-shaded bars.

2.2.1.1. Hydrology

Continuously logged and manually recorded water-table data are shown for two loggers within the MM-RW site in Figure 2.2.5. All data are from dipwells located within the bund. There is very good agreement between automated and manual measurements, and the data indicate that the site has been continuously inundated over the two years of monitoring. Water levels at the continuously monitored dipwells peak at around 20-30 cm above the surface during winter periods, and fall to around 0-10 cm above the surface during summer. In effect, the bund at MM-RW has created a ‘bucket’, whereby 1 mm of precipitation directly raises water levels inside the bund by 1 mm. Water is lost from the site via a mixture of ‘open water’ evaporation and some seepage into groundwater or through leaks in the bund. Topographic data indicate that at no point does the water level exceed the height of the bund.

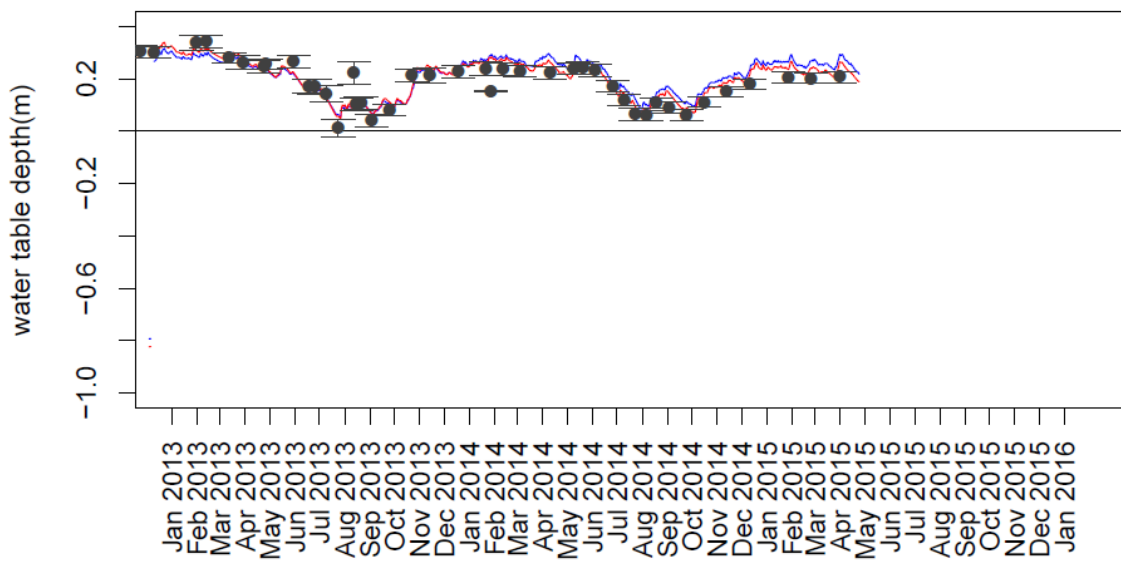


Figure 2.2.5. Continuously monitored and manually measured water-table data for MM-RW. Measurements at this site ended in May 2015.

Monthly water balances (Figure 2.2.6) were calculated for MM-RW from a combination of measured rainfall, water level logger data, meteorological data from the E.ON AWS, and monthly average potential evapotranspiration data from the CEH Climate, Hydrology and Ecology Support System (CHESS) converted to an estimate of open water evaporation using the Thornthwaite method (see Appendix 1 for details). Annual rainfall totals for 2013 and 2014 both exceeded ET totals, with 2014 experiencing approximately 80 mm more rainfall than 2013. Between spring and the beginning of autumn ET routinely exceeded incoming rainfall, implying that aquatic losses were minimal. During 2013 this resulted in no water loss from the start of April through to the end of September. The pattern differed slightly for 2014 when rainfall totals in both May and August exceeded evapotranspiration. This between-year difference was reflected by the automated dipwell data (Figure 2.2.5) which showed water levels falling closer to the ground surface, and over a longer period, in summer 2013 versus summer 2014. Manual dipwell data from across the site indicate that, whilst water levels remained continuously above the surface across most areas of MM-RW, some areas did become dry at the surface during summer. Water losses via seepage below the bund occurred between October to March, and the mean annual water loss was estimated at 280 mm yr⁻¹.

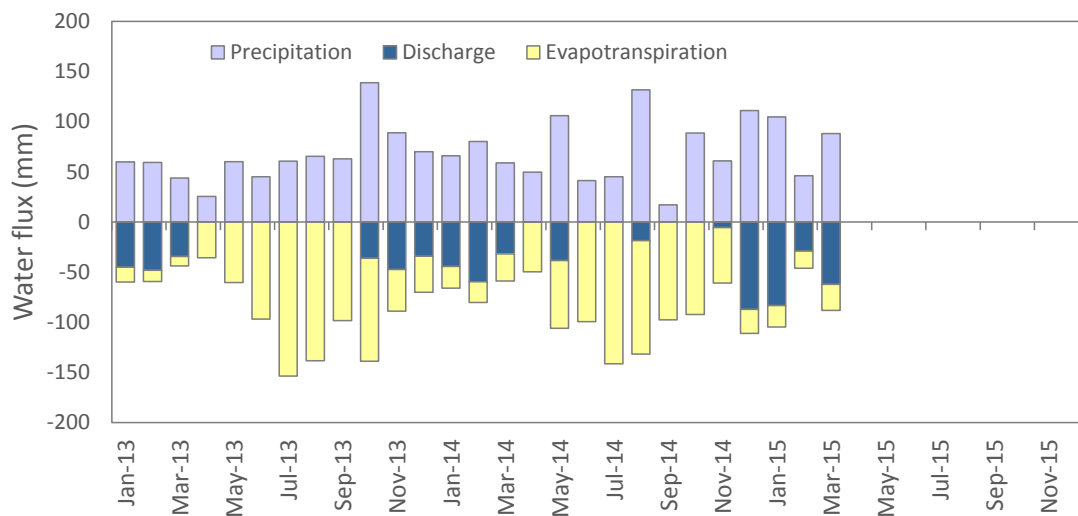


Figure 2.2.6 Monthly hydrological budgets for MM-RW, January 2013 to March 2015.

2.2.1.2. Static chamber gas fluxes

Static chamber gas fluxes were measured in the three main vegetation communities present across the site, namely *Molinia*-dominated drier areas, transitional areas with a mixed cover of tussocky *Molinia* and *Eriophorum*, and wetter *Sphagnum* lawns. Measured (and hence modelled) CO₂ fluxes at the mixed *Molinia/Eriophorum* community were consistently intermediate between the dry *Molinia* and wet *Sphagnum* areas, therefore only data from the latter two datasets are shown (Figure 2.2.7). These show clearly differing behaviour, with the *Molinia* areas having much higher rates of photosynthesis, and correspondingly higher ecosystem respiration, with both fluxes showing clear seasonal cycles (and similar fluxes) in both measurement years. Modelled NEE in the *Molinia*-dominated areas was weakly positive in winter, with net CO₂ drawdown during the growing season. Modelled annual mean GPP for this vegetation type in 2013-14 was 1265 g C m⁻² yr⁻¹, with a mean ER of 884 g C m⁻² yr⁻¹, suggesting that it acted as a strong net sink for CO₂ of -381 g C m⁻² yr⁻¹. In contrast, the *Sphagnum* lawn areas showed comparatively weak photosynthesis (Figure 2.2.7b; 2013-14 mean 360 g C m⁻² yr⁻¹) and similarly low ecosystem respiration (mean 301 g C m⁻² yr⁻¹) giving a modest mean net CO₂ sink of -59 g C m⁻² yr⁻¹. The *Molinia/Eriophorum* tussock area had a mean modelled GPP of 448 g C m⁻² yr⁻¹, and a mean ER of 554 g C m⁻² yr⁻¹, which would imply that these areas were acting as net CO₂ sources of 106 g C m⁻² yr⁻¹ on an annual basis. Model fits for all three communities at MM-RW were comparatively poor, especially for GPP (Appendix 2) so although differences in GPP and ER were fairly clear between vegetation types (Figure 2.2.7), any comparison of NEE values between vegetation types should be made with caution. Assuming a 20/50/30 split of dry, intermediate and wet areas across the broader site would suggest a mean site-wide NEE of -41 g C m⁻² yr⁻¹. However, given the

large apparent differences in NEE between vegetation communities, as well as the uncertainties in the flux models noted above, this calculation is sensitive to the relative areas assigned to each community, and it is consequently highly uncertain whether the site as a whole is acting as a net CO₂ sink or a net source. It is also worth noting that the presence of *Molinia* at this site pre-dates the rewetting of the site, and its persistence under elevated water levels may have resulted in a net CO₂ balance that is atypical of the somewhat dryer conditions under which it more commonly occurs.

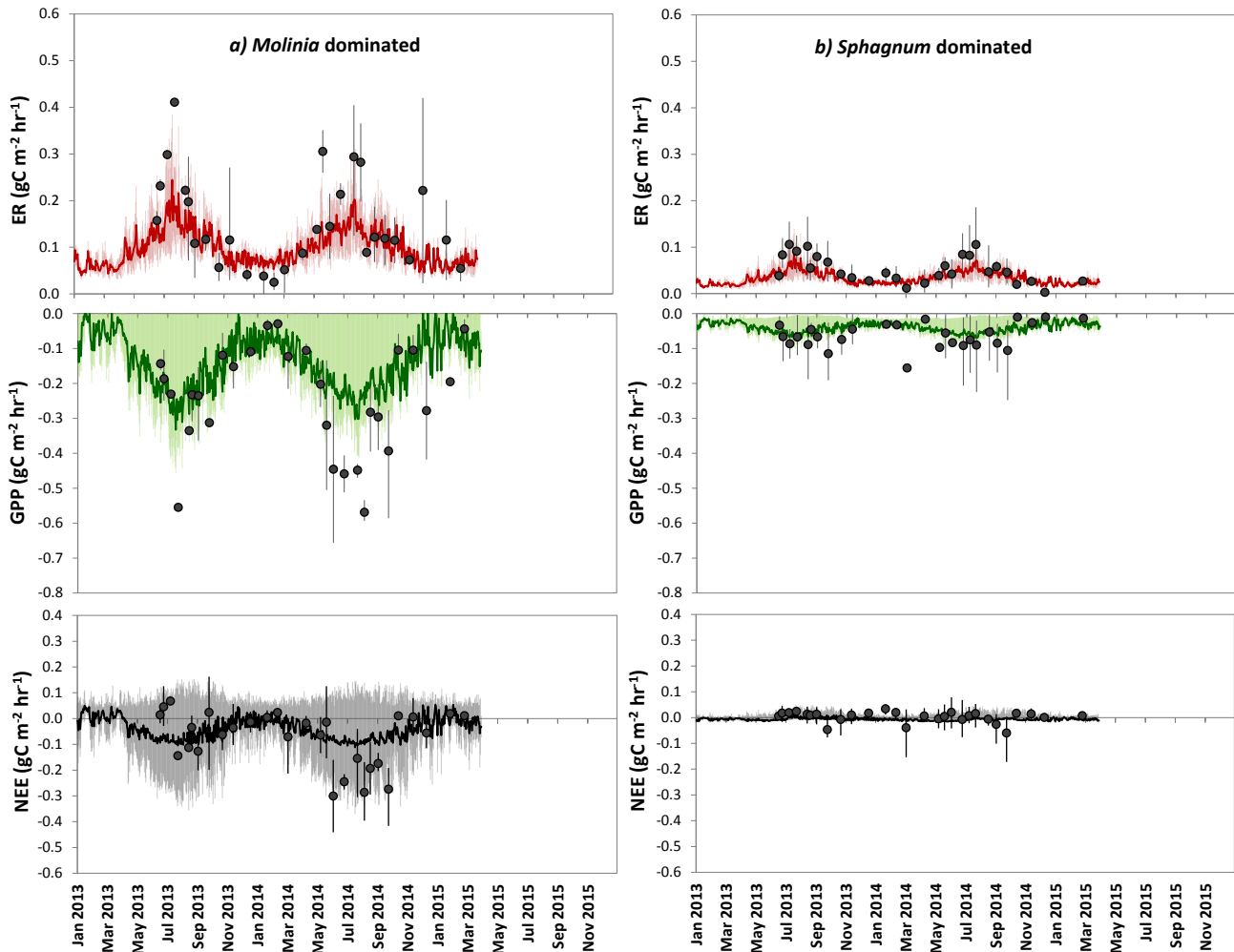


Figure 2.2.7. Modelled and observed CO₂ fluxes (ER = ecosystem respiration, GPP = gross primary productivity, NEE = net ecosystem exchange) based on static chamber measurements for two of the three vegetation communities measured at MM-RW. Continuous lines show modelled daily mean fluxes, shading shows modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date. Note that static chamber measurements were generally taken around the middle of the day and therefore reflect peak rates of photosynthesis and respiration rather than daily means.

Methane fluxes at MM-RW (Figure 2.2.8) were very high. Some seasonality was observed, but emissions in 2014 were much higher than in 2013, and no empirical model could be fitted to the data. Therefore, an interpolation method was used to estimate annual fluxes. Fluxes from all three areas were quite similar, with mean emissions of 31.6 g C m⁻² yr⁻¹ from the drier *Molinia* areas, 31.8 g C m⁻² yr⁻¹ from the intermediate *Molinia/Eriophorum* area (data not shown), and 35.1 g C m⁻² yr⁻¹ from the wetter *Sphagnum* lawns, giving a site-wide mean CH₄ emission of 32.8 g C m⁻² yr⁻¹, the highest of any site in the network. It is worth noting that this flux is significant not only for the GHG balance of MM-RW, but also for its C balance, almost cancelling out the estimated CO₂ uptake.

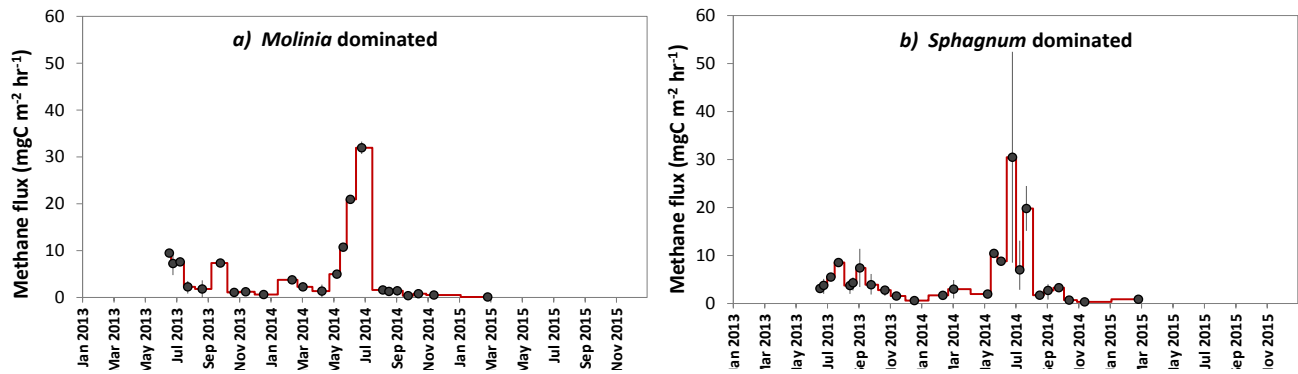


Figure 2.2.8. Measured CH_4 fluxes at MM-RW. Points show mean observations on each measurement date, and error bars show range of measured values on that date. Red lines show interpolated fluxes.

2.2.1.3. Aquatic carbon fluxes

Water chemical analysis at MM-RW was undertaken on samples collected from 12 dipwells, as there were no ditches within the bunded area. DOC data (Figure 2.2.9) were extremely high, ranging from 70 mg l^{-1} in winter to a summer maximum of over 300 mg l^{-1} in 2013. Peak concentrations were lower in the wetter summer of 2014, although some seasonality was still apparent. Peat pore waters at the site were highly acidic (mean pH 3.83) reflecting the acidic character of the bog peat of the Manchester Mosses (see Table 2.2), and DIC concentrations were therefore assumed negligible. POC concentrations were not measured on the dipwell samples, because these are not indicative of POC loss from the peat; at MM-RW, where all water loss occurs via seepage, POC losses can be assumed to be negligible.

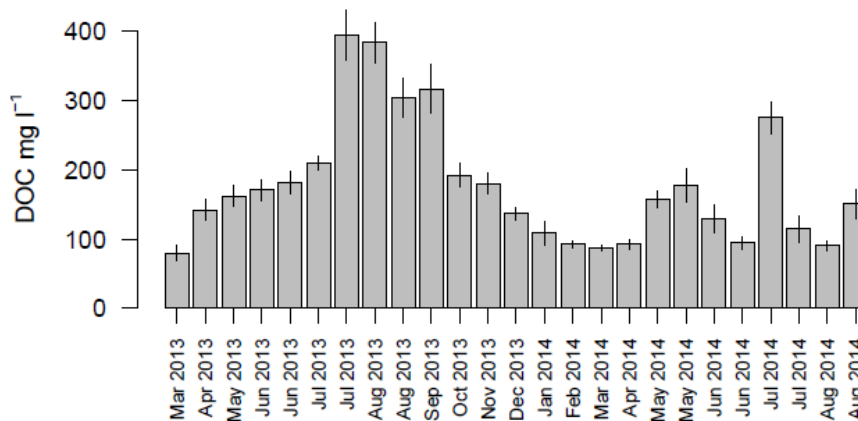


Figure 2.2.9. Mean and standard error of porewater dissolved organic carbon concentrations, MM-RW. Note that x axis simply records sampling dates and is therefore not a true time axis.

Aquatic C fluxes are shown in Figure 2.2.10 (DIC and POC were assumed to be negligible, as noted above). For the period over which fluxes could be calculated, DOC losses peaked in October 2013, when autumn water discharge from the site began, but concentrations were still high, then declined through to spring 2014. Overall DOC losses from the site were large, estimated at $34.6 \text{ g C m}^{-2} \text{ yr}^{-1}$. Dissolved CO_2 fluxes were generally fairly low (estimated annual flux $1.8 \text{ g C m}^{-2} \text{ yr}^{-1}$), whilst dissolved CH_4 fluxes were (by far) the highest measured at any site, with an estimated annual mean of $1.6 \text{ g C m}^{-2} \text{ yr}^{-1}$. Whilst this high flux appears to be very consistent with the measured CH_4 emissions from the peat surface, it is almost entirely associated with the very high dissolved CH_4 concentration measured in March 2014. This could suggest that there is episodic release of CH_4 from the site to the surrounding drainage network, or it could represent an anomalous measurement, for example associated with vertical transport of CH_4 from the water column to the atmosphere (i.e. an 'on-site' emission rather than a lateral 'off-site' emission).

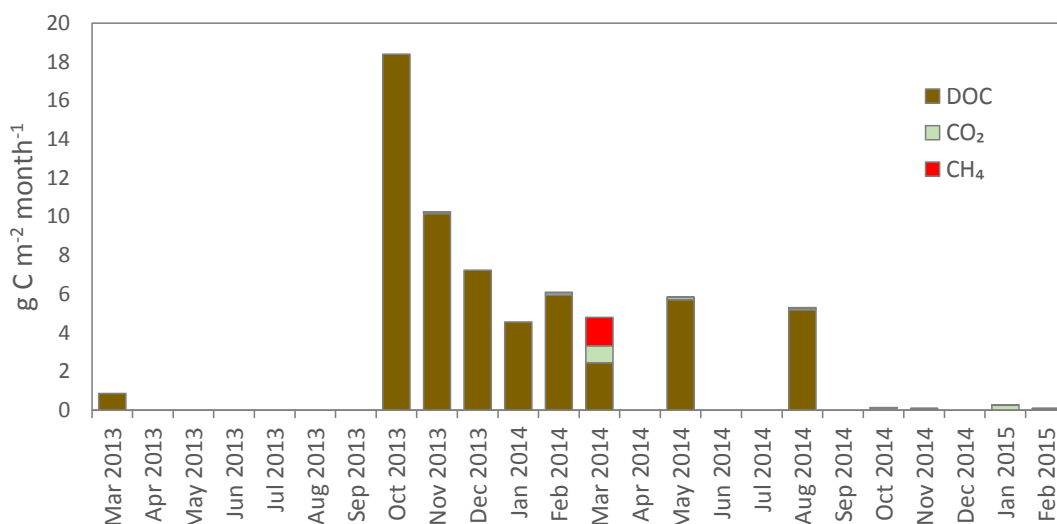


Figure 2.2.10. Estimated monthly aquatic carbon fluxes (DOC, CO₂ and CH₄ only), MM-RW, from March 2013 to August 2014. DOC measurements were made March 2013 to August 2014 (zero values during this period indicate no flux). Dissolved gas fluxes were measured in November 2013, February to November 2014, and January to February 2015.

2.2.2. Little Woollen Moss – extraction site (MM-EX)

Little Woollen Moss, 3 km south of Astley Moss, includes a large (107 ha) active industrial peat extraction site. At the outset of the project, the site was under commercial operation, but in 2012 it was purchased by the Lancashire Wildlife Trust. Some peat extraction is, however, still continuing on 35 ha of the site, under the existing licence. The peat extraction area comprises bare peat surfaces with a dense network of parallel (< 50 m spaced) drainage ditches throughout the site. The location used for field measurements is close to the edge of the peat extraction area, in part of the site where much of the peat has already been removed, and which is not subject to active extraction. Apart from some limited plant growth along the ditch edges, the peat surface was largely bare throughout the study.

Peat cores were collected at MM-EX from a location close to the flux measurement area (dark bars in Figure 2.2.11) and from another part of the extraction site where a greater depth of peat is still present (light bars). The data show that the surviving peat at both locations is compositionally similar to that at MM-RW, with a very low pH, high %C and high C/N ratio. The bulk density is higher at MM-EX than at MM-RW (0.24 vs 0.14 g cm⁻³), which could result from the transit of peat-harvesting machinery across the site, or may reflect intrinsically higher peat density at the base of the peat profile, which has effectively been truncated by peat extraction at the measurement site (compare data from shallow and deep cores in Figure 2.2.11). In the core collected from the measurement site, there is a sharp transition from organic to mineral soil at 60 cm. At the other sampling location, a similar transition occurs at 260 cm, thus implying (assuming the same initial peat depth at both locations) that around 2 m of peat has been extracted in the measurement area compared to other parts of the extraction site.

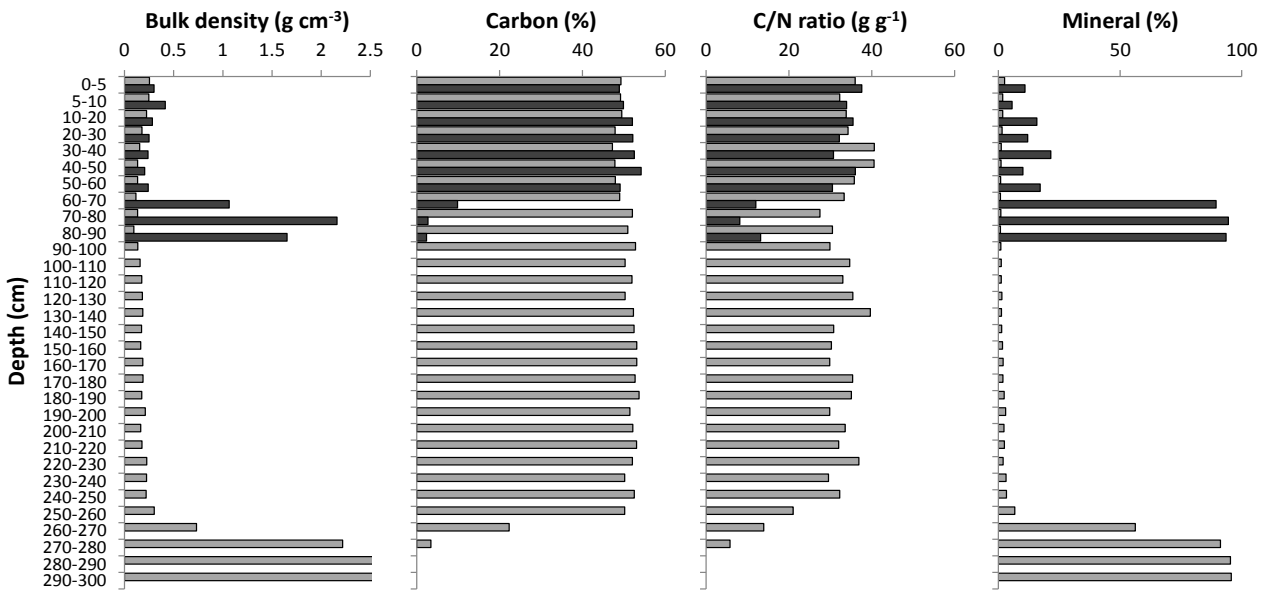


Figure 2.2.11. Peat core data, MM-EX. Data shown are from cores collected from an area of heavily extracted peat (dark bars) and one where less extraction has occurred (light bars).

2.2.2.1. Hydrology

Water-table data for MM-EX are shown in Figure 2.2.12. In marked contrast to MM-RW, water tables are continuously below the peat surface, with a baseline depth of around 60-80 cm, and exhibit very high short-term variability linked to individual rainfall events. The rapid water-table recessions observed after rainfall are likely to be associated with peat degradation and large topographic gradients between inter-ditch areas and the ditch water levels across the extraction site. During both 2013 and 2014 deep water tables were observed from March onwards, before recovering to shallower depths in October. During late autumn and winter the manual water-table data showed water tables were often close to the surface, but the automated data continued to show high short-term variability even during winter. Mean water-table depths varied from around 70 cm in summer to 20 cm in winter.

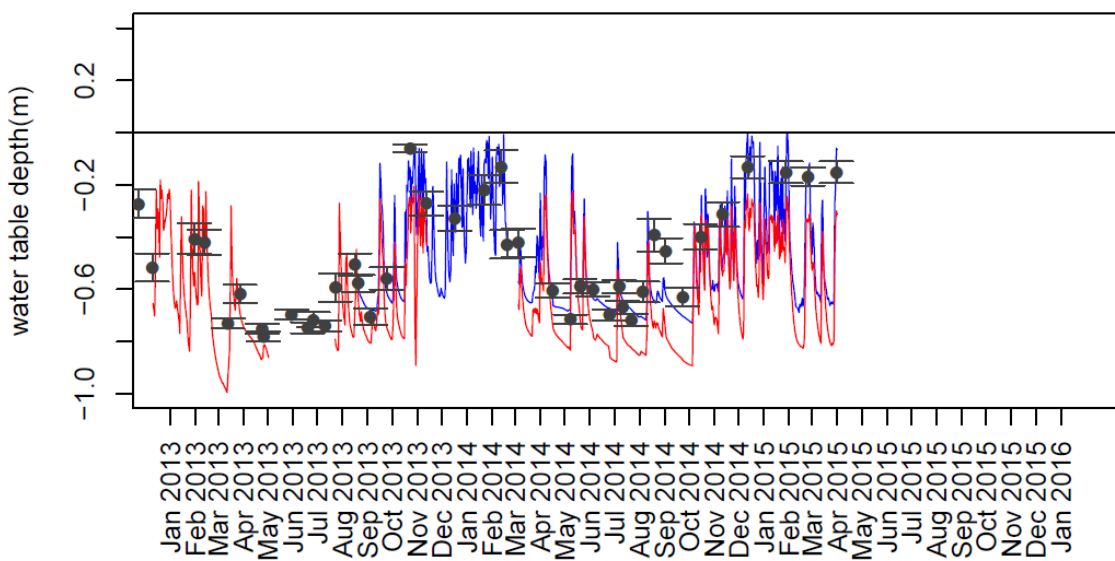


Figure 2.2.12. Continuously monitored and manually measured water-table data for MM-EX. Measurements at this site ended in May 2015.

The water balance of MM-EX was calculated from measured precipitation, with evapotranspiration estimated from the CHESSE-based estimates for open water calculated for MM-RW, adjusted from open water to bare peat assuming a ratio of 1.35 between open water and bare peat (Scarlett, 2015; see Appendix 1). A v-notch weir was installed in the ditch draining the site, and used to gauge flow from the 212 m² catchment. However, very little flow over the weir was recorded, indicating that most water was leaving the site via deeper subsurface flow. The monthly water balance for MM-EX (Figure 2.2.13) suggests that, due to the permeability of the peat and comparatively low ET from the unvegetated peat surface, water is lost from the site throughout the year, with the exception of dry summer months. Total annual water discharge from the site was estimated at 494 mm yr⁻¹, 76% higher than that from MM-RW.

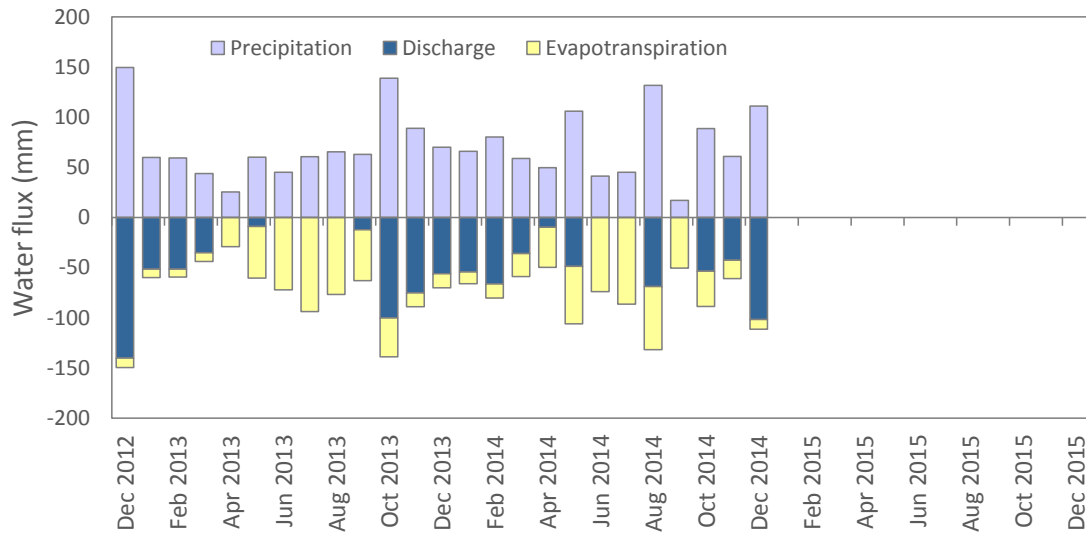


Figure 2.2.13. Monthly hydrological budgets for MM-EX, December 2012 to December 2014.

2.2.2.2. Static chamber gas fluxes

Static chamber CO₂ measurements were made at MM-EX from June 2013 to March 2015 (Figure 2.2.14). As would be expected, photosynthetic uptake at this largely unvegetated site was negligible, therefore respiration was assumed to equate to net CO₂ loss from the peat. Respiration fluxes were consistently very low (compare y axis on Figure 2.2.14 to equivalent data from other sites), and were rather weakly related to air temperature as shown by the low variability of the modelled flux. The estimated mean annual respiration rate (and thus also the estimated rate of CO₂ emission) was 138 g C m⁻² yr⁻¹.

Measured CH₄ fluxes at MM-EX are shown in Figure 2.2.15. For most of the measurement period CH₄ fluxes were near-zero or positive, but on two occasions (November 2013 and June 2014) some CH₄ uptake was observed. On an annual basis, the estimated CH₄ emission from the peat at MM-EX was 0.18 g C m⁻² yr⁻¹. Gas fluxes from the ditch network within the extraction site were not measured. Given the high ditch density at the site there is the potential for CH₄ fluxes from the water surface to add significantly to overall site emissions.

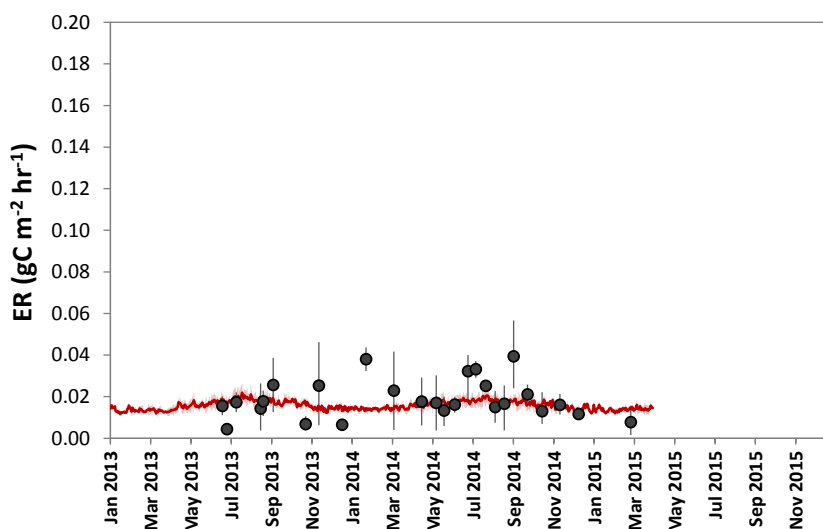


Figure 2.2.14. Modelled and observed ecosystem respiration (ER) based on static chamber measurements at MM-EX. Continuous line shows modelled daily mean fluxes, shading shows modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date.

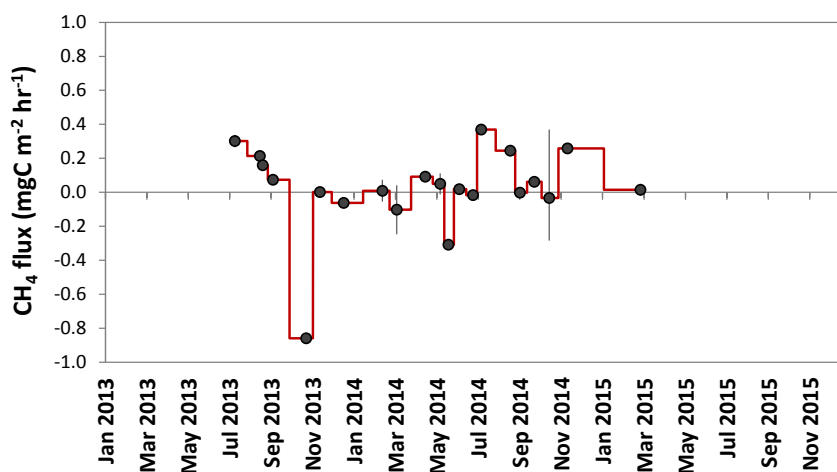


Figure 2.2.15. Measured CH_4 fluxes at MM-EX. Points show mean observations on each measurement date, and error bars show range of measured values on that date. Red lines show interpolated fluxes.

2.2.2.3. Aquatic carbon fluxes

Water chemistry was measured at three locations in each of the ditches bounding the flux measurement site. As at MM-RW, pH was sufficiently low (mean 4.21 in porewater, 3.76 in ditch water) that there was no DIC present. The range and temporal pattern of DOC concentrations (Figure 2.2.16) at MM-EX was similar to MM-RW, despite the obvious differences in both vegetation and hydrology, with maximum concentrations in the summer of 2013, and smaller peaks in May and July 2014. Since the ditches at MM-EX rarely flowed, only a few POC samples were collected, which were insufficient to quantify fluxes (data not shown). The flux of DOC from the site was temporally variable (Figure 2.2.17) with high fluxes in months when high concentrations coincided with significant flows. Over the period of measurement, the estimated annual mean flux was very high, at $58.4 \text{ g C m}^{-2} \text{ yr}^{-1}$, the highest of all the sites for which fluxes could be calculated, and 70% higher than the flux from MM-RW. The dissolved CO_2 flux was much lower, with an annual mean of $2.2 \text{ g C m}^{-2} \text{ yr}^{-1}$, and the CH_4 flux was negligible ($< 0.01 \text{ g C m}^{-2} \text{ yr}^{-1}$).

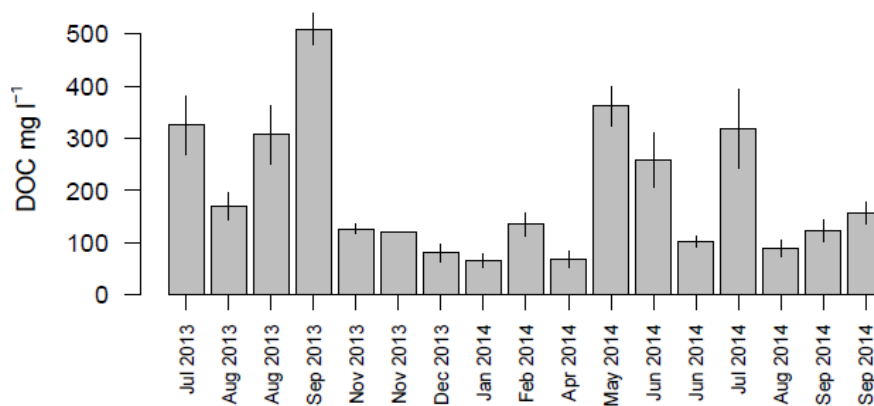


Figure 2.2.16. Mean and standard error of porewater dissolved organic carbon concentrations, MM-EX. Note that x axis simply records sampling dates and is therefore not a true time axis.

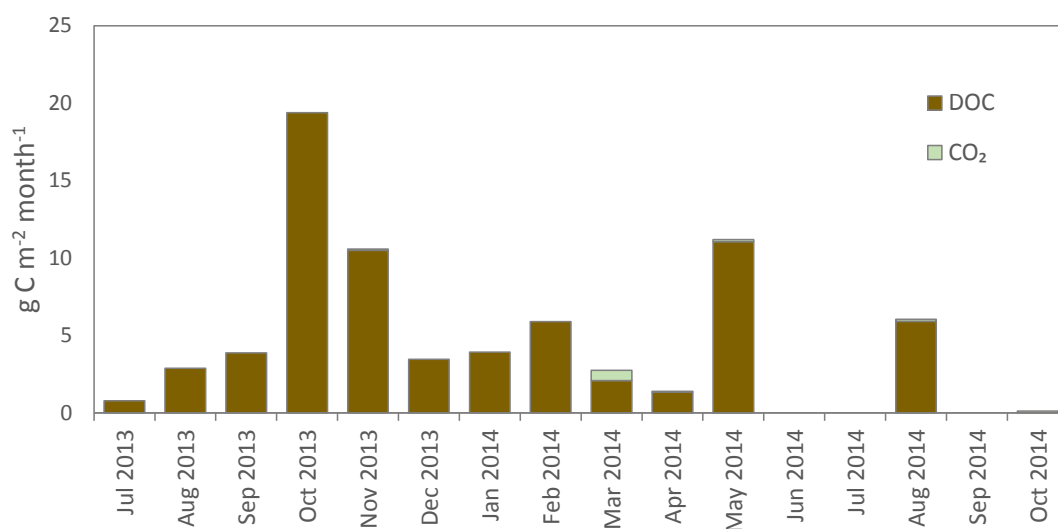


Figure 2.2.17. Estimated monthly DOC and dissolved CO₂ fluxes, MM-EX, from July 2013 to October 2014. No DOC data for October 2014; other zero values indicate no flux.

2.2.3. Little Woolden Moss – arable on deep peat (MM-DA)

The arable field at Little Woolden Moss is directly adjacent to the extraction site MM-EX. The extraction area bounds the northern edge of the field, and active drainage ditches form the eastern and southern boundaries. One partially collapsed pipe drain was noted connecting the MM-DA site to an adjacent field, but the intensity of subsurface drainage, and of agricultural management more generally, was considerably less than at the EF-DA or EF-SA sites. The field is ploughed, fertilised and reseeded annually, and was used to grow wheat in all measurement years, although ley grass was noted on the site prior to the onset of measurements in 2012. Biomass offtake in the wheat harvest was estimated from available yield estimates.

A single peat core collected from MM-DA (Figure 2.2.18) suggests that farming activity has strongly affected the upper 25 cm of peat, which has a higher bulk density and mineral content, and lower %C and C/N ratio, than either the underlying peat or the nearby MM-RW and MM-EX sites. The pH of the upper peat layer is also markedly higher than at the other MM sites (5.8 vs < 3). On the other hand, the peat at MM-DA is less compacted, less mineral- and less fertiliser-enriched than the East Anglian arable sites, and has a lower pH. This partly reflects pre-existing differences in peat type (bog vs fen), but also suggests a generally lower intensity of agricultural activity. The sharp transition to peat properties more comparable to those of an undisturbed raised bog (including very high C/N ratios) below 25 cm suggests that the impacts of ploughing, fertilisation and other activities has largely been limited to the surface layer.

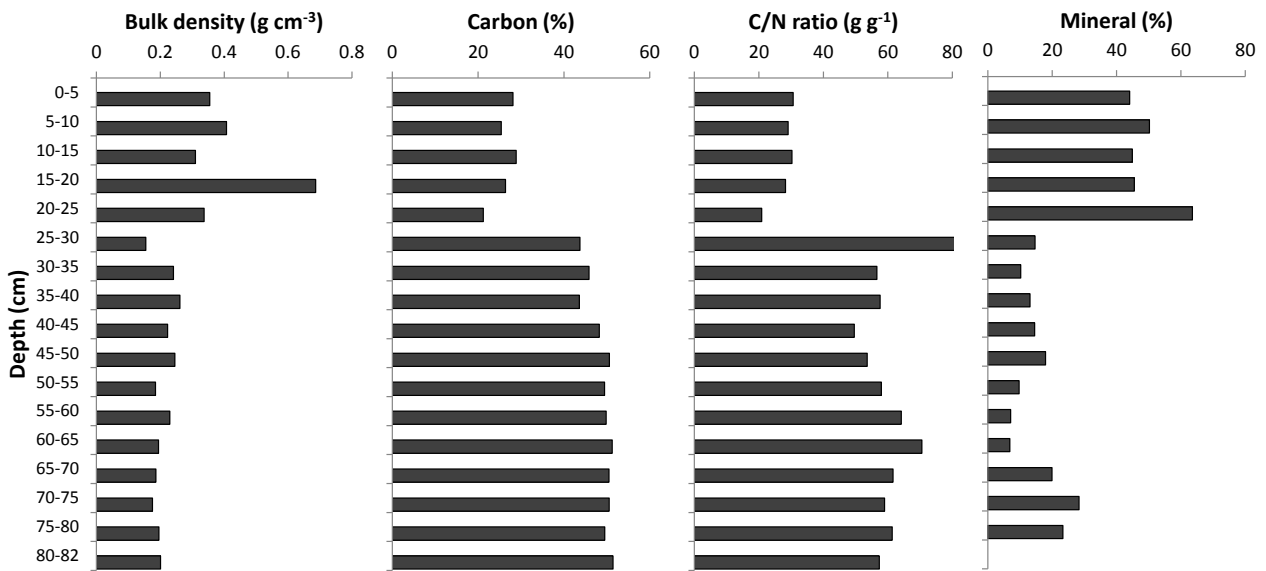


Figure 2.2.18. Peat core data, MM-DA.

2.2.3.1. Hydrology

Continuous and manual water-table measurements for MM-DA are shown in Figure 2.2.19. The two dipwell records show good agreement over the period of common measurements, until summer 2014, but following equipment theft in August 2013 only manual records were obtained. The data show a dramatic and sustained decline in water table through the 2013 growing season, from around 20 cm depth in March to nearly 1 m in August. The field re-wetted rapidly in October 2013, but water tables remained low, at around 30-40 cm during the winter of 2013-14. Drawdown was slightly less rapid during the 2014 growing season, prior to the end of the instrumental record. Subsequent manual data suggest that water tables recovered to levels similar to those at the start of the record during the winter of 2014-15. It is worth noting that summer water-table drawdown relative to the peat surface was greater at MM-DA than at the adjacent MM-EX, although this is possibly related to the level of the ground surface (lower in the cutover site).

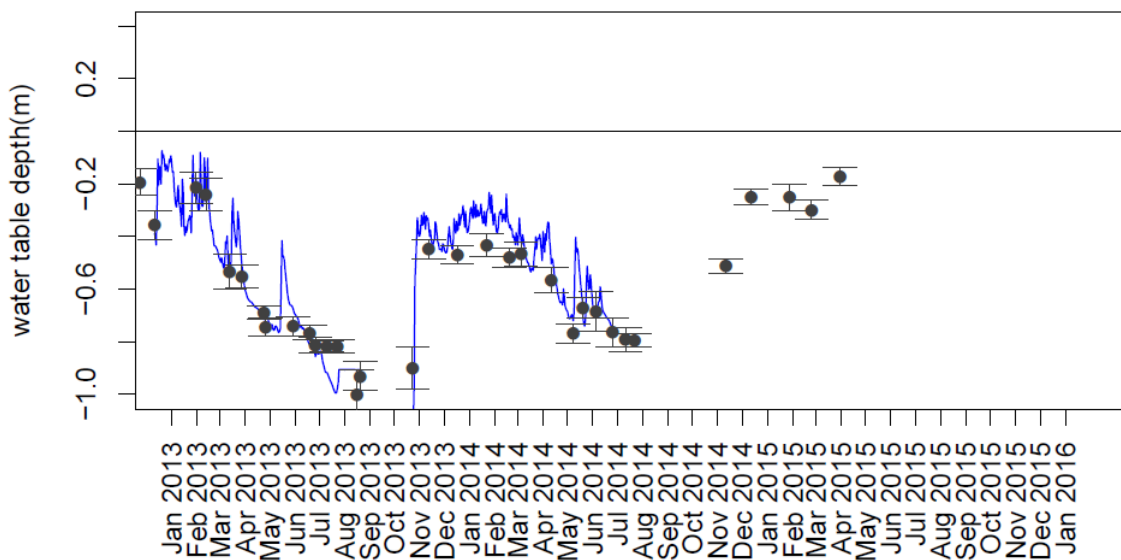


Figure 2.2.19. Continuously monitored and manually measured water-table data for MM-DA. Continuous measurements ended at this site due to vandalism and theft of equipment in 2014, after which only manual measurements were obtained. Measurements at the site ended in April 2015.

The agricultural management of MM-DA made it difficult to accurately calculate a hydrological budget for the site. The removal of equipment during summer months to allow farming activities to take place means that data are often missing. Evapotranspiration was estimated from the CHESSE dataset, assuming a well-watered field, and monthly water balances were calculated for June 2013 to December 2014 (Figure 2.2.20). Typically, ET exceeded rainfall during spring and summer, but some discharge was predicted for the high-rainfall months of May and August 2014. The lack of discharge for the 2013 growing season is consistent with the water-table drawdown noted above, and the May 2014 wet period is reflected in a modest water-table recovery. Estimated annual discharge from MM-DA was 384 mm yr⁻¹, intermediate between the other Manchester Mosses sites.

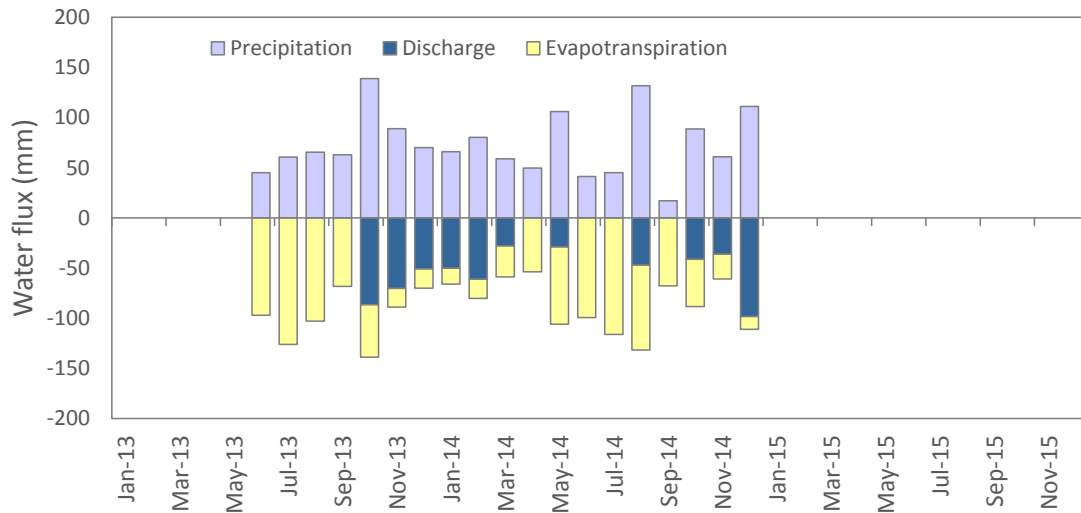


Figure 2.2.20. Monthly hydrological budgets for MM-DA.

2.2.3.2. Static chamber gas fluxes

Static chamber data (ER and NEE) for MM-DA are shown in Figure 2.2.21. Respiration rates were well-reproduced by an exponential air temperature model, which captured summer peaks, winter minima and some of the observed differences between the 2013 and 2014 measurement years. Measured and observed ER were very similar to those obtained for the two East Anglian arable sites, EF-DA and EF-SA (compare Figure 2.2.21 to Figures 2.1.31 and 2.1.43). Given the agricultural management of the site it was not possible to develop an empirical model of GPP, however NEE was almost continuously positive (lower panel of Figure 2.2.21), indicating that the field was losing carbon even during periods when the wheat crop was growing. As at the East Anglian arable sites, the majority of the CO₂ flux measured by the dark chambers can be assumed to represent heterotrophic respiration of the peat, and thus CO₂ emission. Over the modelled period, estimated annual mean ER was 1151 g C m⁻² yr⁻¹.

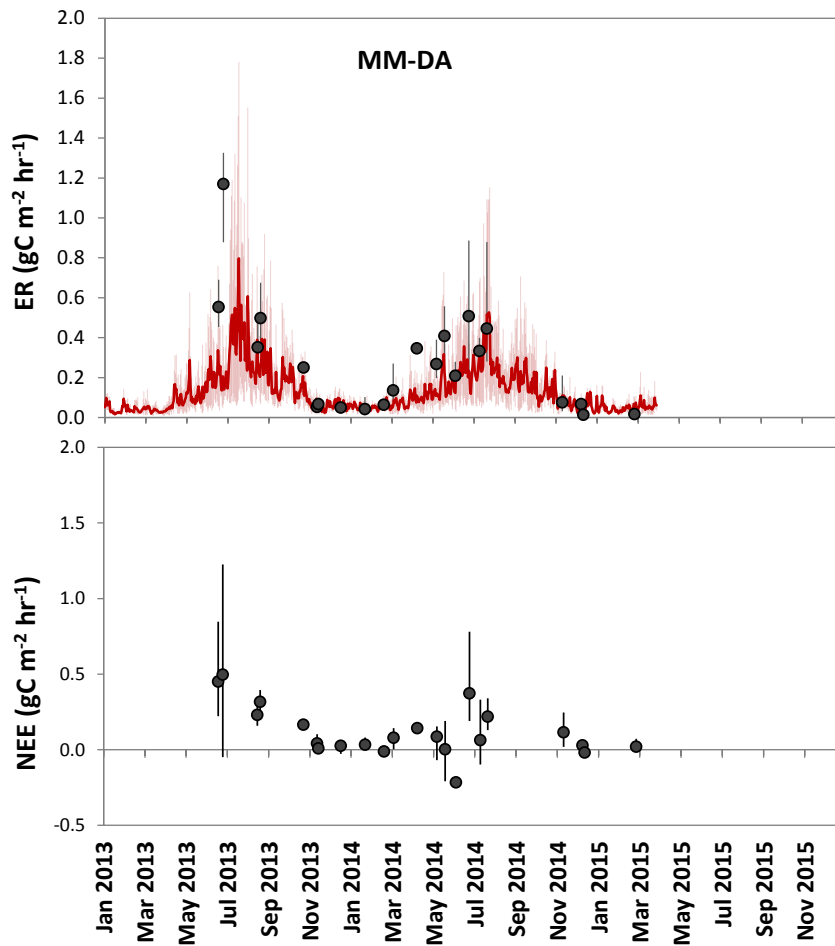


Figure 2.2.21. Modelled and observed CO₂ fluxes (ER = ecosystem respiration, NEE = net ecosystem exchange) based on static chamber measurements at MM-DA. Continuous lines show modelled daily mean fluxes, shading shows modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date. Note that static chamber measurements were generally taken around the middle of the day and therefore reflect peak rates of photosynthesis and respiration rather than daily means.

Methane fluxes at MM-DA were very small, and fluctuated between net uptake and net emission (Figure 2.2.28). Over the measurement period the net flux was marginally positive, with an estimated annual mean of 0.1 g C m⁻² yr⁻¹. Ditch fluxes were not measured at MM-DA.

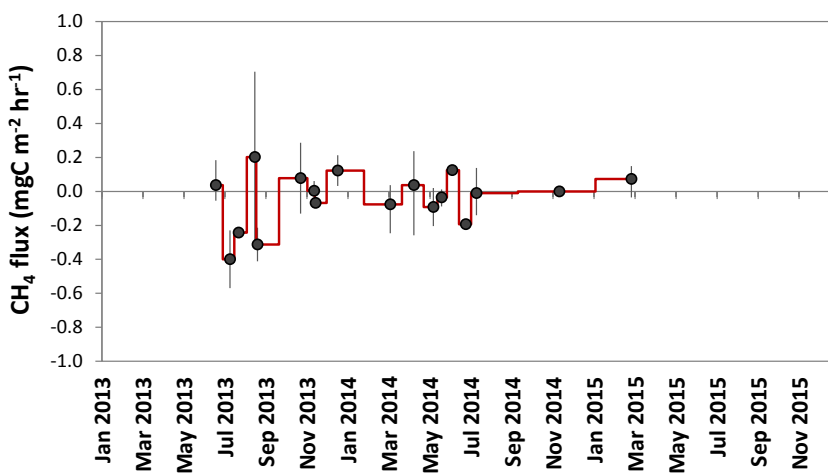


Figure 2.2.28. Measured CH₄ fluxes at MM-DA. Points show mean observations on each measurement date, and error bars show range of measured values on that date. Red lines show interpolated fluxes.

2.2.3.3. Aquatic carbon fluxes

Concentrations of DOC at MM-DA (Figure 2.2.29) were similar to the other Manchester sites, with similar (very high) summer maxima, and winter minima. This could be considered somewhat surprising given the major differences in drainage, vegetation and management activities between the sites. However the pH of pore water at MM-DA is notably higher than that of the other Manchester Mosses sites, with an average of 6.1, compared to < 4.2 at the other sites. This coincides with the much higher pH of near-surface peat (5.8 vs < 3, Table 1.2) and is assumed to be the result of agricultural practices such as fertiliser or lime addition. The high pH value suggests that there may be some DIC in the water, although this was not measured.

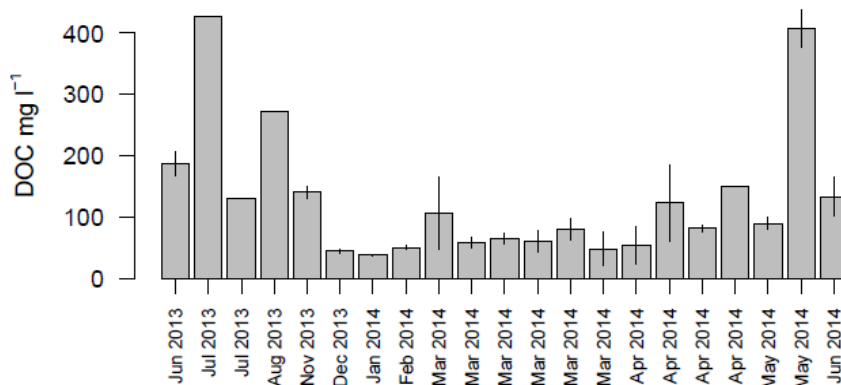


Figure 2.2.29. Mean and standard error of porewater dissolved organic carbon concentrations, MM-DA. Note that x axis simply records sampling dates and is therefore not a true time axis.

Monthly DOC fluxes could only be calculated for a total of 11 months, during a 13 month period from June 2013 to June 2014 (Figure 2.2.30). These limited data show export of DOC from November 2013 to March 2014, with a further (large) DOC flux during the wet May 2014. In the absence of any measured fluxes for the September-October period, fluxes for these months were predicted from a linear regression of fluxes for months where flux estimates were obtained for both MM-DA and MM-EX, which showed a strong correlation between the adjacent sites ($R^2 = 0.84$, $p < 0.001$). This gave an overall estimate of annual DOC flux from MM-EX of $39.2 \text{ g C m}^{-2} \text{ yr}^{-1}$, which is similar to that at MM-RW, but considerably lower than at MM-EX. Dissolved gas concentrations were not measured at this site.

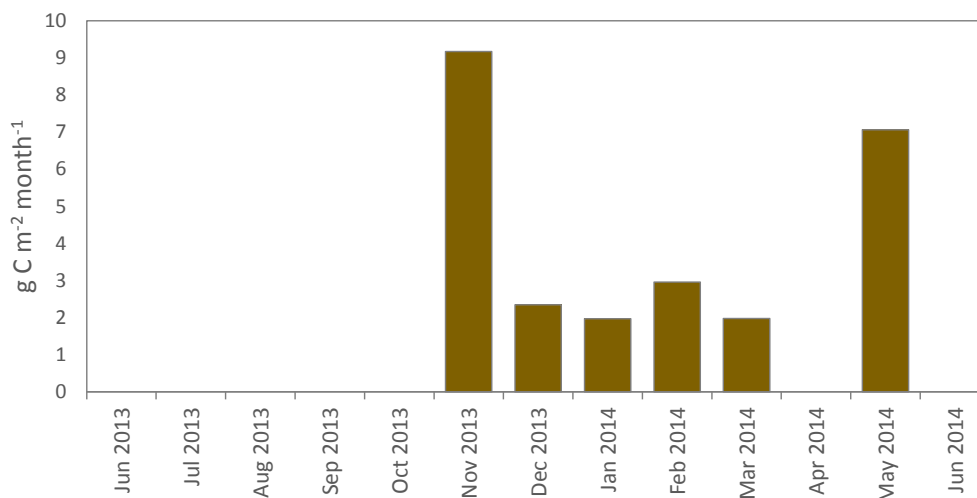


Figure 2.2.30. Estimated monthly DOC fluxes, MM-DA, from June 2013 to June 2014. Fluxes could not be calculated for September-October 2013; other zero values indicate no flux.

2.3. Anglesey Fens

Wales holds some of the largest areas of fen peat in the UK outside East Anglia, of which a number of relatively well-preserved sites are located on the Isle of Anglesey and the Llŷn Peninsula in Northwest Wales. The two regions together hold around 750 ha of peat, including some of the best examples of alkaline and calcareous fen in the UK, and are designated as a Ramsar site and as Special Areas of Conservation. Cors Erddreiniog, on Anglesey, holds the largest single area (250 ha) of peat in the region, and is owned and managed as a National Nature Reserve by Natural Resources Wales (NRW). It is a valley-head rich fen, one of a number of similar systems in the northeast part of Anglesey (Figure 2.3.1). Much of the site is underlain by deep fen peat, some extending to over 5 m depth, which in places alternates with bands of marl. The site was historically affected by peat-cutting, which led to the loss of a core area of rain-fed bog, and more recently has been impacted by drainage and inflow of nutrient-enriched water from surrounding farmland. Cors Erddreiniog was a major focus for NRW's Anglesey and Llŷn Fens EU LIFE project (NRW, 2016), with restoration activities at the site including mowing and burning of biomass to open areas up to low-growing species, turf stripping to return nutrient-enriched pasture to fen vegetation, and the use of constructed wetlands and channels to remove agricultural nutrients from springs entering the fen, and increase diffuse water flow onto the site. Water levels are maintained close to the mire surface via a sluice on the main outflow, which drains a network of ditches that extend across the fen. Two sites within Cors Erddreiniog were included in the study, identified by NRW as being in favourable and unfavourable status in terms of the plant communities present. Both sites are subject to low intensity grazing by ponies, and have remained under stable management throughout the project.

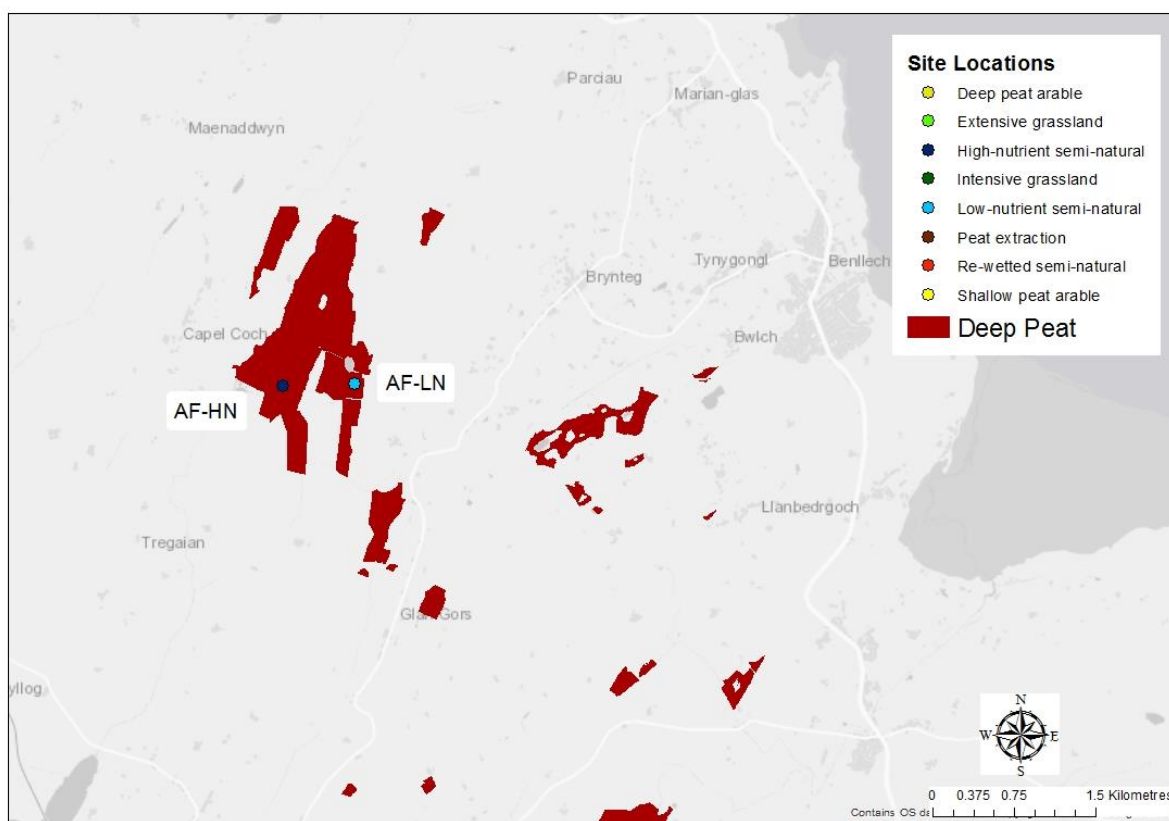


Figure 2.3.1. Anglesey Fens study area.

Monthly meteorological data (Figure 2.3.2) for the two Anglesey Fen sites were provided from an AWS located on the eastern edge of the AF-HN site, which is operated by NRW. Where possible, missing data from this AWS were filled using data collected by the flux towers. No other external source of SW_{in} or air temperature data was available for the AF site and filling of short gaps was performed using the mean of available measurements for the same time of day within a time window of seven days. In Figure 2.3.2, SW_{in} and air temperature values have been excluded for months during which the equivalent of more than three days in any month data required gap-filling. Wind rose diagrams for AF-HN (Figure 2.3.3) show that wind

flow is bimodal at this location, with prevailing winds arriving from the southwest and the northeast. This pattern most likely reflects the local topography and proximity to the coast, as well as the possible influence of an area of woodland to the east of the AWS.

Mean annual air temperatures in 2013, 2014 and 2015 were 9.2 °C, 10.3 °C and 9.5 °C respectively. Monthly average air temperatures showed broadly similar seasonal trends to other network sites. Large between-year differences (lower panel) were observed between January and June with the warmest conditions recorded during the first half of 2014, and cooler conditions in 2013. 2015 had the lowest September and October temperatures but the warmest December of the measurement period (9.5 °C).

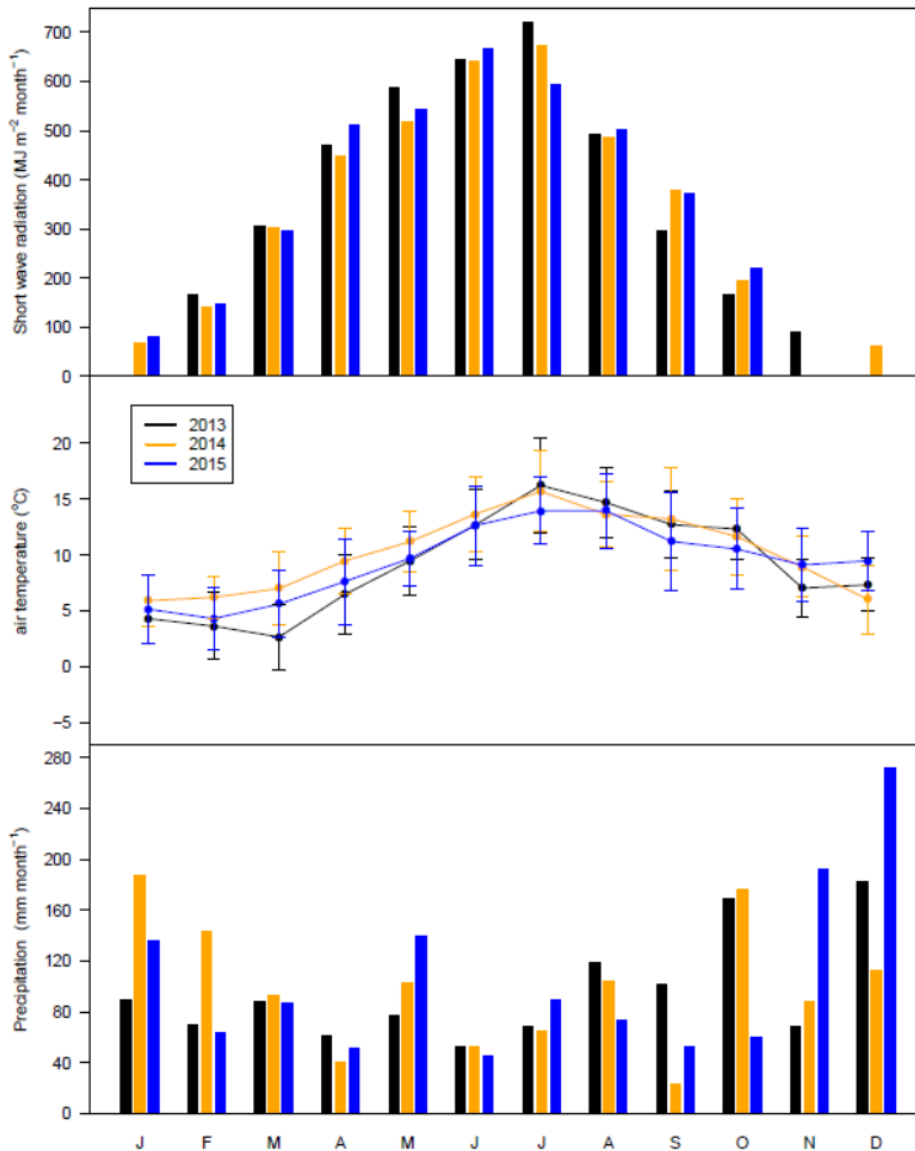


Figure 2.3.2. Total monthly incoming solar radiation (top) mean monthly air temperature (middle) and total monthly precipitation (lower) for the AF-HN (Cors Erddreiniog) site. Error bars on the temperature plot show one standard deviation of the mean. Data supplied by National Resources Wales.

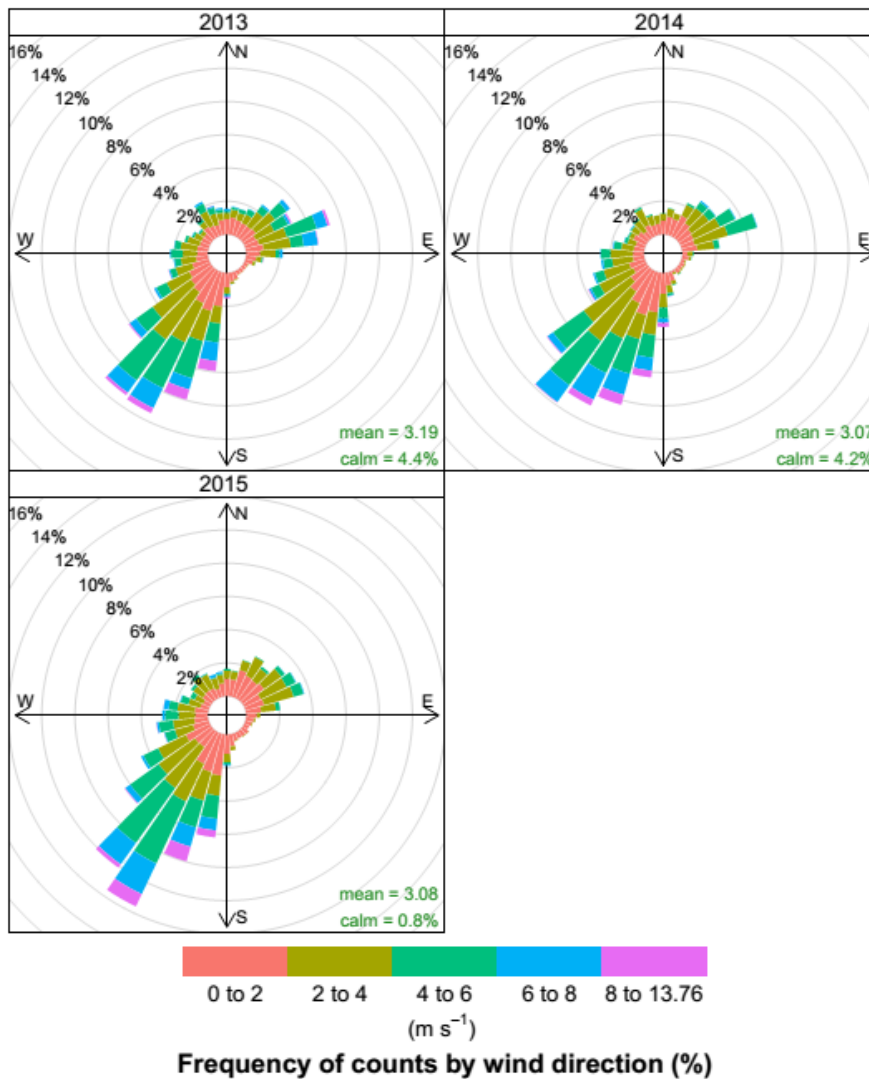


Figure 2.3.3. Wind rose plots showing wind direction and wind speed at the AF-HN (Cors Erddreiniog) site.

The Anglesey Fen sites are the wettest in the measurement network; annual precipitation sums were estimated at 1439 mm yr⁻¹ in 2013, 1189 mm yr⁻¹ in 2014 and 1264 mm yr⁻¹ in 2015. The AF sites were characterised by large seasonal and between-year variability in rainfall. Monthly precipitation was in excess of 100 mm for a number of months in 2013 (August, September, October and December), 2014 (January, February, May, August, October, December) and 2015 (January, May, November, December). November and December 2015 were the two wettest months of the measurement period, with a combined precipitation of 465 mm.

2.3.1. Cors Erddreiniog – Low Nutrient site (AF-LN)

The ‘low nutrient’ site at Cors Erddreiniog is located on the eastern edge of the fen, in an area considered by NRW to be of a high habitat quality. Unlike the majority of the other sites AF-LN, has a small external catchment comprising farmland and some woodland, on limestone geology, with an estimated area of around 2.8 ha (the fen itself has an area of around 3.5 ha). Base-rich water from the hillslope enters the fen via springs along the eastern boundary of the peat. Other springs upwell directly into the fen through the peat. Some spring water traverses the wetland area via ditches, but these have been partly blocked in order to encourage diffuse water flow across the ground surface. The overall effect of the additional water supply from the hillslope is to maintain high water levels at the site throughout much of the year.

The AF-LN site is close to a marl-forming lake, Llyn yr Wyth Eidion. It is located with an area of NVC class M22 *Juncus subnodulosus*–*Cirsium palustre* fen-meadow. The low canopy height allows growth of a range of low-growing fen specialist species, including brown mosses.

Peat core data (Figure 2.3.4) show a fairly uniform vertical peat profile to a depth of 2.5 m. The peat has a relatively low bulk density and mineral content, and a high carbon content. The C/N ratio of the upper peat is fairly low (17.5 g g⁻¹), reflecting the relatively high nitrogen content of peat-forming fen species compared to bog species, and pH is intermediate at around 5.5.

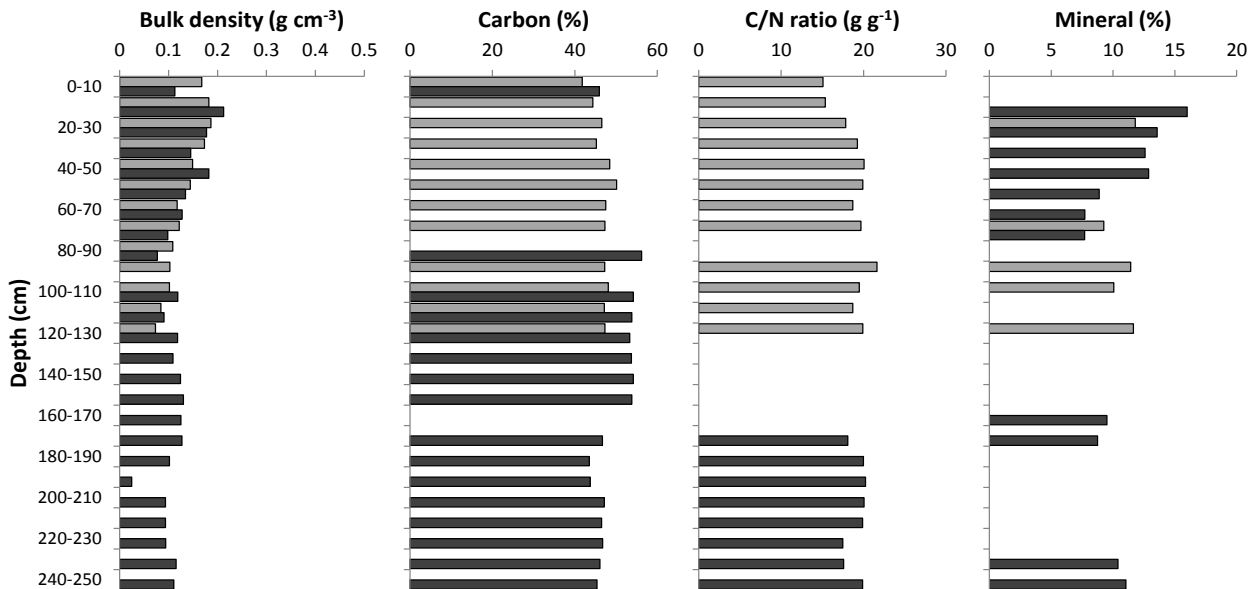


Figure 2.3.4. Peat core data, AF-LN. Data from two cores analysed from the site are represented by dark and light grey-shaded bars (note that not all analyses were run for all core increments).

2.3.1.1. Hydrology

The short fen at AF-LN is characterised by very shallow water tables (Figure 2.3.5), with frequent localised inundation in areas of upwelling groundwater, and water tables within 10 cm of the surface at other sites during most of the study period. Some water-table drawdown occurred in all three summer periods, but this rarely caused water levels to fall more than 30 cm below the surface. The most substantial (but short-lived) period of water-table drawdown was recorded during July 2013, with less pronounced but more extended periods of drawdown in the summers of 2014 and 2015. Overall, the AF-LN site was one of the wettest sites included in the study, but with less evidence of sustained or widespread inundation compared to the wettest site, MM-RW. The AF-LN site could, therefore, arguably be considered to have the least disturbed hydrological regime of any of the sites.

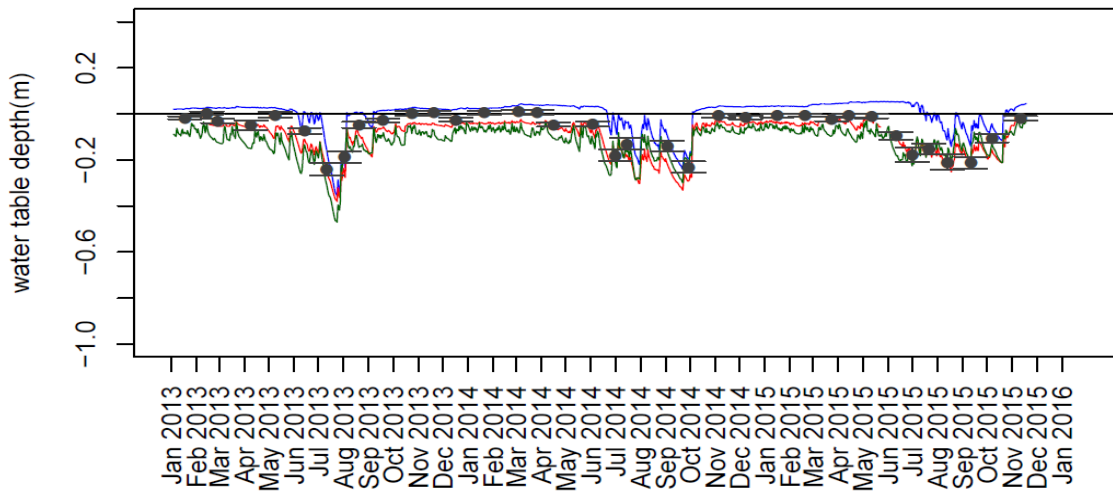


Figure 2.3.5. Continuously monitored and manually measured water-table data, AF-LN.

As noted above, the water supply to AF-LN is significantly augmented by flow from the hillslope subcatchment, which makes up over one third of the total catchment area. To characterise the hydrological complexity of this site, a grid of 12 piezometer nests was installed across the study area, each comprising piezometers measuring hydraulic head at 40, 80, 120 and 160 cm depth, corrected to a local datum. These data were used to determine local flow direction, which showed an east-west gradient in water table across the site as water flowed from the base of the hillslope towards the main drain on the west side of the fen. Hydraulic conductivity (K) was also measured for each piezometer depth at each site; these data were then combined in order to calculate the discharge for each piezometer site and depth (see Appendix 1), and consequently to calculate i) the flow into the fen from the hillslope (Q_{in}); and ii) flow out of the fen into the main drainage ditch (Q_{out}). However because Q_{out} was found to be typically lower than Q_{in} , it was assumed that some water was also being lost towards the lake on the northern edge of the fen; this interpretation was also supported by the hydraulic gradient data (see Appendix 1) suggesting some flow from south to north as well as east to west. Since water tables were continuously high, there was very little variation in water storage within the fen itself.

Two methods were used to estimate flow from AF-LN. The first assumed that water input from the hillslope subcatchment to the fen would equate to precipitation minus evapotranspiration for this area (i.e. that all excess precipitation to this area would enter the fen). The second method took measured Q_{in} values from the piezometers at the eastern edge of the fen as the water input. The water balance of the fen itself was then calculated as $Q_{out} = P_{net} + Q_{in} - ET$ (see Section 1.3.2). The first method gave an estimated discharge of 1151 mm yr^{-1} , whereas the second method gave a discharge of 814 mm yr^{-1} . Note that these discharge values are not equivalent to normal areal discharge values which are typically compared only to rainfall entering the system. Instead, in addition to rainfall, they include inflows to the site, where the catchment area of the inflow region is not included in the areal averaging of runoff depth. As the second method was based on direct field measurements and there was also a possibility that some of the upslope flow could in reality be diverted away from the site by nearby ditches before it reached the site, we applied the second method, although data produced using both methods are provided in Appendix 1.

The monthly water balance for AF-LN based on the second method is shown in Figure 2.3.6. Over the measurement period (from October 2013 to November 2015) lateral water inputs from the hillslope contributed an estimated 15% of water input to the fen. Water was lost from the fen in all measurement months except June and September 2014, and June 2015. Peak discharge rates occurred during wet winter months. Even with the lower estimate of 814 mm yr^{-1} , AF-LN had the highest mean discharge of any of the study sites. It is worth noting the hydrological characteristics of AF-LN, in which direct precipitation inputs are augmented by lateral water flows, are those that would have occurred at all fen sites prior to human modification.

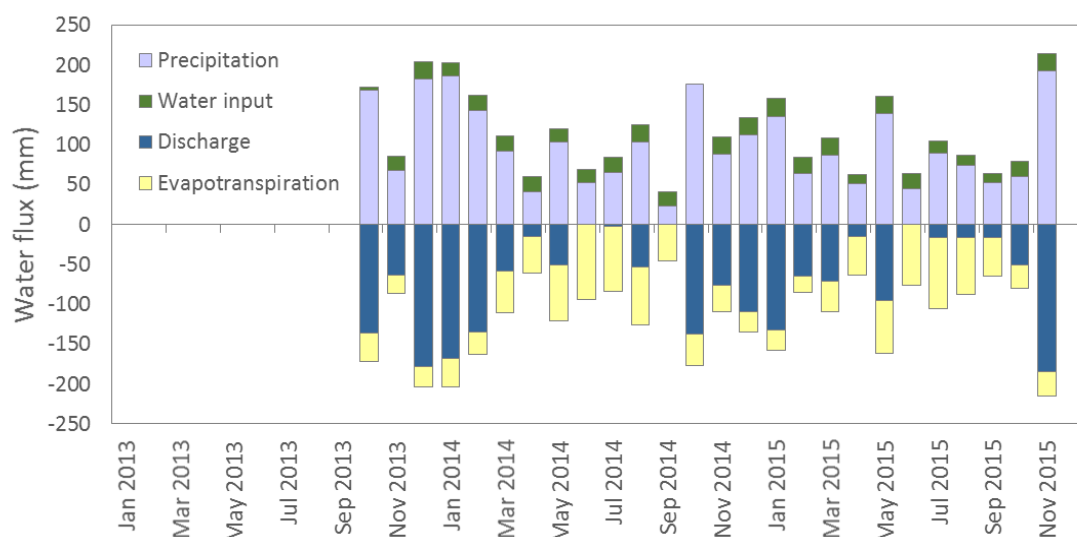


Figure 2.3.6. Monthly hydrological budgets for AF-LN.

2.3.1.2. Eddy covariance gas fluxes

A flux tower was installed at AF-LN in October 2014, following theft and vandalism of part of the system after it was originally deployed at MM-EX. The AF-LN site has been producing flux data since mid-November (following some initial power supply issues in October) and has captured fluxes for most of 2015. The longest data gaps at AF-LN (top panel in Figure 2.3.7) are related to limitations in the power supply (November 2014 and 2015) and a technical issue with the gas analyser (late April to early May 2015). As at other flux measurement sites, these data gaps were filled using the standard methodology. The filling of data gaps during the main growing season increases the uncertainty in time-integrated CO₂ budgets.

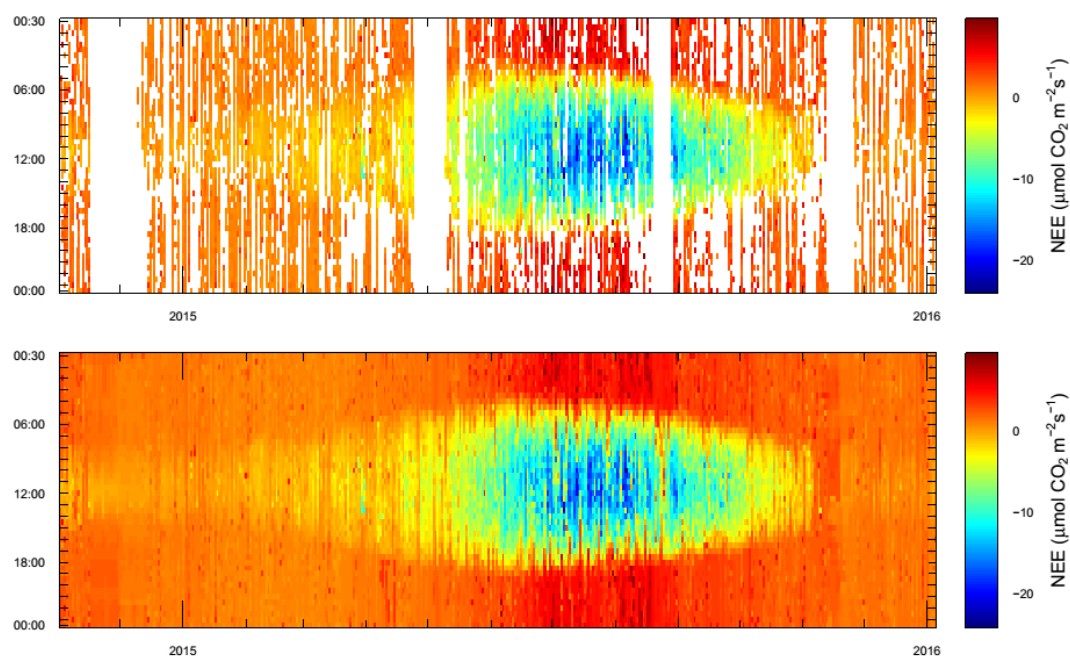


Figure 2.3.7. Fingerprint plot of measured (top panel) and measured and gap-filled (lower panel) net ecosystem carbon dioxide exchange for the AF-LN (Cors Erddreiniog – low nutrient) site. Units are in $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$.

The seasonal pattern of NEE at AF-LN is similar to the other sites with permanent vegetation cover. Monthly mean diurnal cycles for the AF-LN site are shown in Figure 2.3.8. The amplitude of the average diurnal cycles

increased relatively slowly during the spring months, particularly compared to the managed grassland sites (EF-EG and SL-EG). This probably reflects the phenology of the fen vegetation at the site, which attains peak growth rates later in the year than drier grassland species. The highest rates of CO₂ uptake occurred in July at $-14.80 \pm 0.63 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ with maximum nocturnal CO₂ loss rates of $5.66 \pm 0.18 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ in August. These net CO₂ uptake and release rates are amongst the lowest observed across the lowland peat flux tower network, and reflect the low productivity of the short fen vegetation at the site. Uptake and loss fluxes of CO₂ then declined rapidly into the autumn, and were close to zero through the winter.

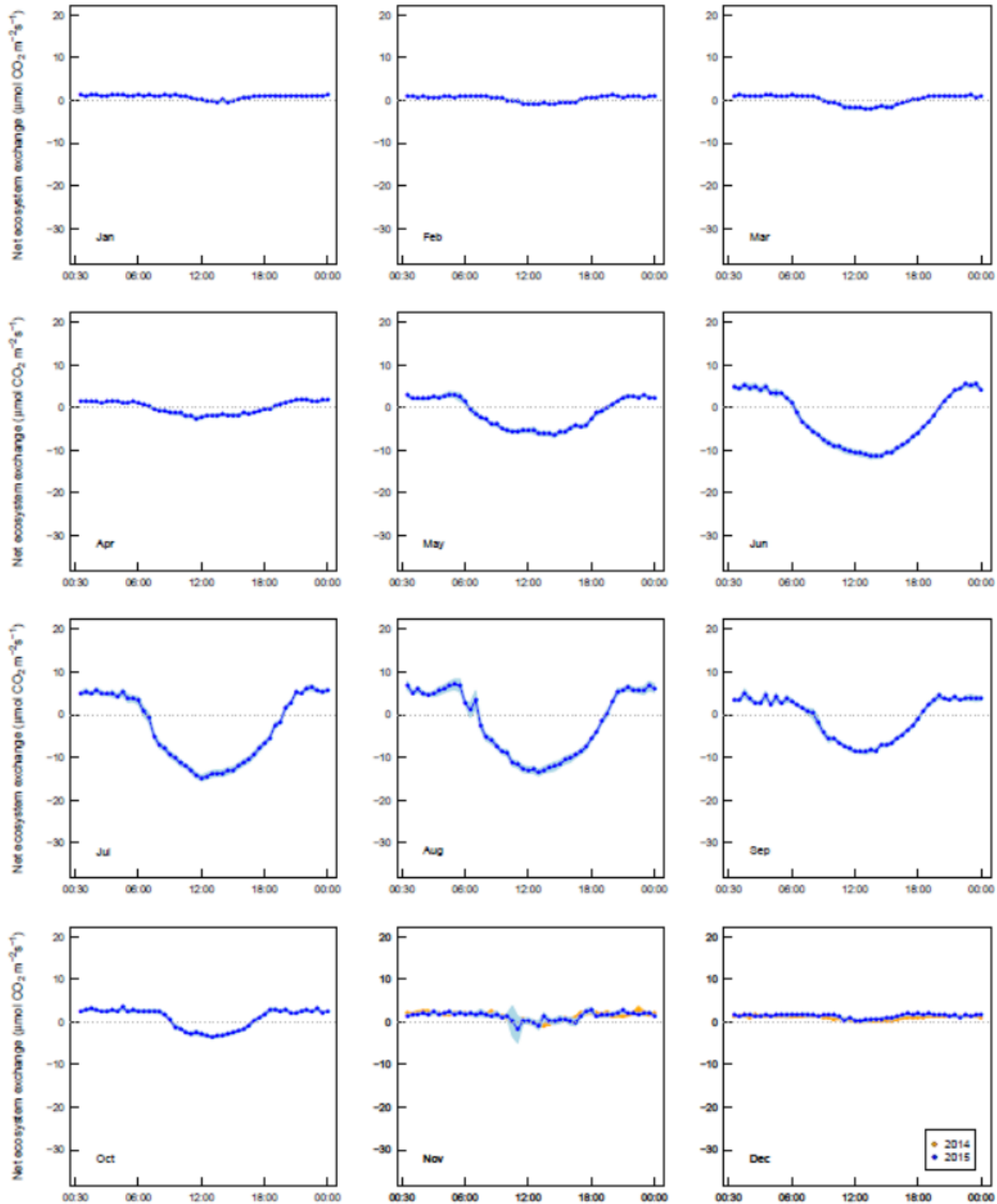


Figure 2.3.8. Mean diurnal cycles of net ecosystem CO₂ exchange at the AF-LN (Cors Erddreiniog – low nutrient) site for months that eddy covariance measurements were available. Data are grouped by meteorological seasons. Shaded areas show one standard error of the mean. Monthly diurnal averages calculated using measured (not gap-filled) data.

Despite the low productivity of the AF-LN site, it was nevertheless a significant net sink for CO₂, with an NEE of $-87 \pm 69 \text{ g C m}^{-2} \text{ yr}^{-1}$ in 2015, because respiration rates were also extremely low (see Section 4). This is consistent with the shallow average water table depth at the site, noted above.

2.3.1.3. Static chamber gas fluxes

Static chamber measurements were made at AF-LN from April 2013 to November 2015, and these data have been used to model daily CO₂ fluxes for the three full calendar years. Measurements were made on 3 replicate collars located within each of two contrasting vegetation types, comprising bryophytes (brown mosses) in wetter areas and vascular plants dominated by *Juncus subnodulosus* (blunt flowered rush) in slightly drier areas. For both vegetation communities, ER was modelled as an exponential function of soil temperature, and GPP as a function of PAR and ETI (see Section 1.3.4). Full information on model performance and fitted parameter values are shown in Appendix 2. Both the measured data and fitted models show relatively small CO₂ fluxes in the bryophyte-dominated areas, with negligible GPP, RE and NEE during winter, and modest net CO₂ uptake during the growing season (Figure 2.3.9a). Modelled annual NEE values for the study period were $-88 \text{ g C m}^{-2} \text{ yr}^{-1}$ in 2013, $-171 \text{ g C m}^{-2} \text{ yr}^{-1}$ in 2014, and $-135 \text{ g C m}^{-2} \text{ yr}^{-1}$ in 2015. The weaker uptake in 2013 resulted from the period of increased ER during July 2013 which coincided with relatively intense water-table drawdown (Figure 2.3.5), and led to the bryophyte area briefly becoming a (modelled) net CO₂ source during the peak of the growing season.

For the *Juncus*-dominated areas (Figure 2.3.9b) both GPP and ER tended to be larger throughout the year, and showed a strong inverse relationship. Both GPP and ER were larger in the relatively dry 2013 growing season, leading to relatively strong net uptake of CO₂ in this year (2013 annual mean NEE $-156 \text{ g C m}^{-2} \text{ yr}^{-1}$). In 2014, modelled CO₂ uptake peaked at lower levels but was more sustained, giving an annual mean NEE of $-191 \text{ g C m}^{-2} \text{ yr}^{-1}$, whilst in the wetter 2015 (and in contrast to the bryophyte areas) uptake was considerably lower, at $-82 \text{ g C m}^{-2} \text{ yr}^{-1}$. For the three years as a whole, and assuming a 50/50 mixture of the two vegetation communities across the site, mean NEE was $-137 \text{ g C m}^{-2} \text{ yr}^{-1}$. For 2015, when the flux tower was in operation, mean chamber-based NEE was $-108 \text{ g C m}^{-2} \text{ yr}^{-1}$, which is close to the flux-tower based estimate of $-87 \text{ g C m}^{-2} \text{ yr}^{-1}$ above.

Methane fluxes at AF-LN (Figure 2.3.10) were consistently positive for both vegetation communities, and showed clear seasonal cycles with relatively little short-term variability during most of the measurement period. In neither vegetation community was it possible to develop a robust model of CH₄ fluxes as a function of measured meteorological or hydrological variables, therefore fluxes were interpolated between points as shown by the red lines in the figures. Very high fluxes were observed on the first and third visits, in May and July 2013, from both sites. Although these large fluxes occurred on relatively warm days, water tables were low and similar conditions in subsequent years did not generate similarly high emissions. Therefore it is considered likely that these fluxes were the result of initial site disturbance following collar installation. Omitting the two sets of very high values from the start of the measurement period would give mean annual emissions of $21.4 \text{ g C m}^{-2} \text{ yr}^{-1}$ from the bryophyte-dominated areas, and $10.0 \text{ g C m}^{-2} \text{ yr}^{-1}$ from the *Juncus* community (if these measurements were retained, the calculated means were 25.3 and $17.1 \text{ g C m}^{-2} \text{ yr}^{-1}$ respectively). The difference in flux between the two vegetation communities was highly consistent throughout the study, with the wetter bryophyte-dominated areas clearly acting as stronger CH₄ sources. Assuming a 50/50 mixture of the two plant communities, and taking the lower flux estimates above (without the two outlying early values) would suggest a mean annual emission from AF-LN of $15.7 \text{ g C m}^{-2} \text{ yr}^{-1}$.

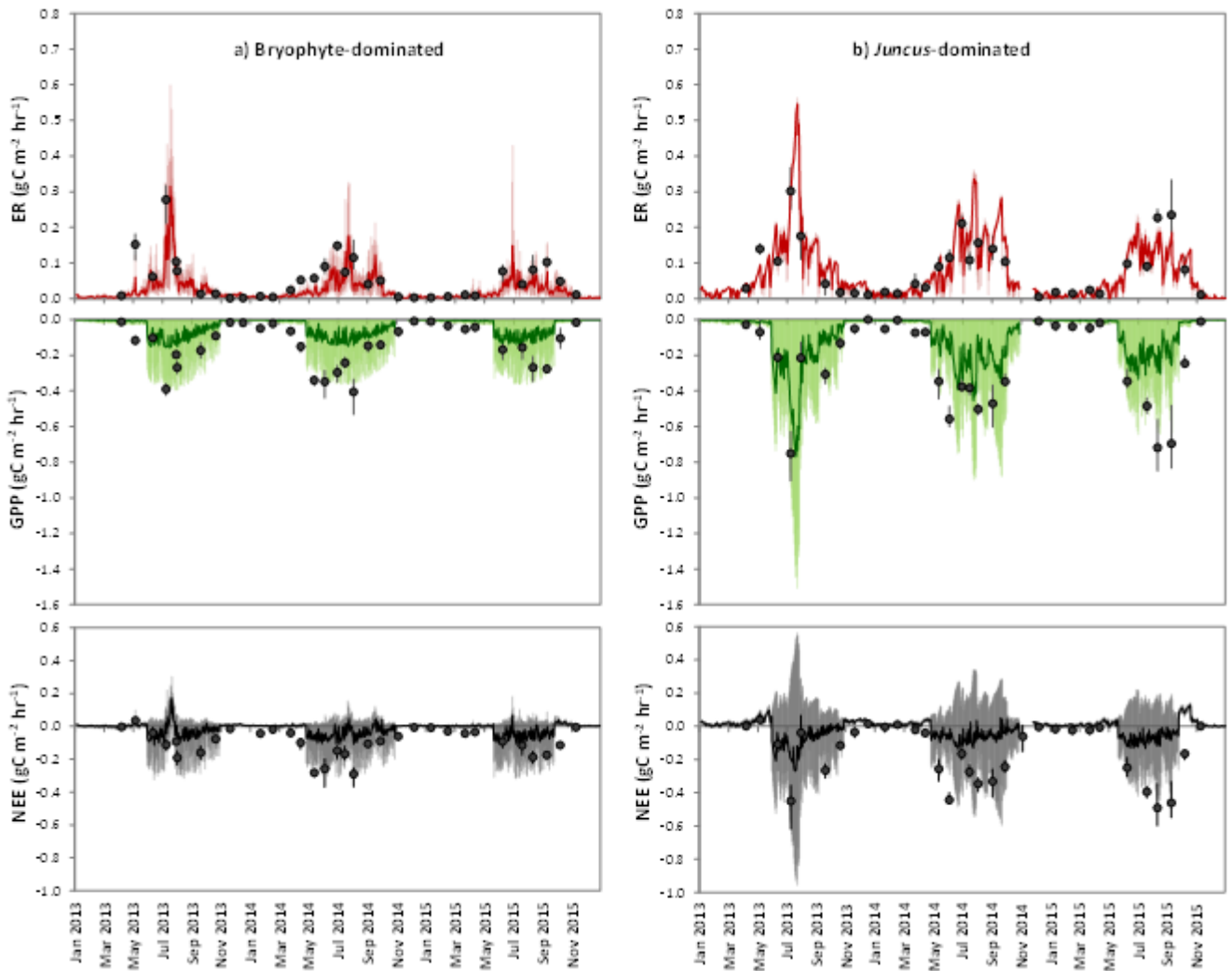


Figure 2.3.9. Modelled and observed CO_2 fluxes (ER = ecosystem respiration, GPP = gross primary productivity, NEE = net ecosystem exchange) based on static chamber measurements in two vegetation communities at AF-LN. Continuous lines show modelled daily mean fluxes, shading shows modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date. Note that static chamber measurements were generally taken around the middle of the day, so tend to be representative of peak (rather than daily mean) rates of photosynthesis and respiration.

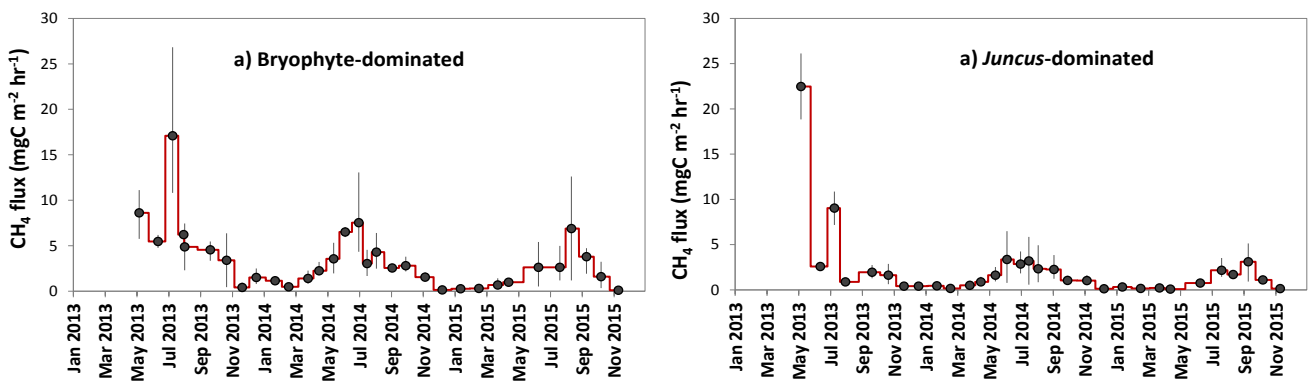


Figure 2.3.10. Measured CH_4 fluxes for the bryophyte and *Juncus*-dominated vegetation communities at AF-LN. Points show mean observations on each measurement date, and error bars show range of measured values on that date. Red line shows interpolated fluxes

Floating chamber data (Figure 2.3.11) indicate that from the ditch adjacent to AF-LN was a consistent net source of CH₄ emissions, although these were much higher in the summer of 2014 than 2015. A period of CO₂ uptake was recorded in early summer 2014, but the ditch was a net source of CO₂ for the remainder of the measurement period. Overall, the ditch was an annual net source of both CO₂ and CH₄ emissions, with mean annual net fluxes (per unit ditch surface area) of 354 and 12.2 g C m⁻² yr⁻¹ respectively. Based on a *Frac*_{ditch} value for the site of 0.01, these fluxes represented area-weighted emissions of 3.5 g C m⁻² yr⁻¹ for CO₂, and 0.12 g C m⁻² yr⁻¹ for CH₄.

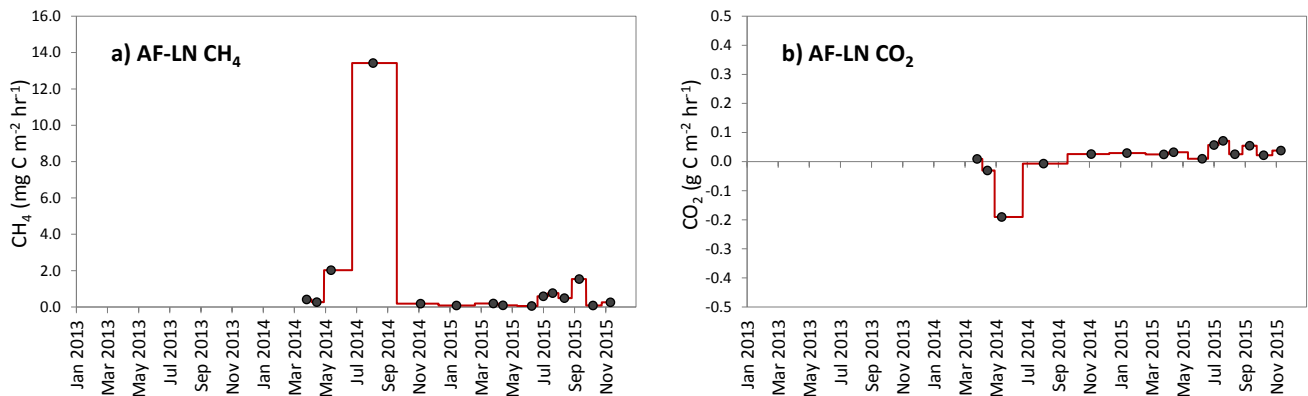


Figure 2.3.11. Observed CH₄ and CO₂ fluxes measured in the ditch at AF-LN. Red line shows interpolated fluxes.

2.3.1.3. Aquatic carbon fluxes

The ditch at AF-LN has fairly high DIC concentrations, along with a high pH (mean 7.61), as would be expected in a good condition alkaline fen (Figure 2.3.12). DOC concentrations were comparatively low, and during some winter periods fell to very low levels. This may be associated with inputs of low-DOC groundwater from the adjacent hillslope, either directly via springs at the edge of the fen (one of which discharges into the sampling ditch) or through upwelling beneath the peat. Periods of higher DOC concentrations, including a sustained period of concentrations around 60 mg l⁻¹ in the first half of 2015, suggest that at these times ditch water was largely derived from the fen itself, rather than upstream sources.

Monthly aquatic carbon fluxes for AF-LN are shown in Figure 2.3.13. As water losses are near-continuous from this site, aquatic carbon fluxes are also significant in most months. The flux is dominated by DIC, reflecting base-rich groundwater inputs, and fairly high overall given the large water flux from the site (estimated annual mean 31.4 g C m⁻² yr⁻¹). DOC fluxes were high in some months, particularly during the period of high DOC concentrations in early 2015, and the overall estimated annual flux was 17.9 g C m⁻² yr⁻¹, comparatively high for a fen site and of similar magnitude to the DOC flux from a blanket bog. The POC flux was minor (estimated annual mean 1.3 g C m⁻² yr⁻¹). The annual mean dissolved CO₂ flux was 1.3 g C m⁻² yr⁻¹, and the dissolved CH₄ flux was comparatively high at 0.05 g C m⁻² yr⁻¹. Note that all aquatic C fluxes from AF-LN would be higher if the alternative estimate of water discharge was applied.

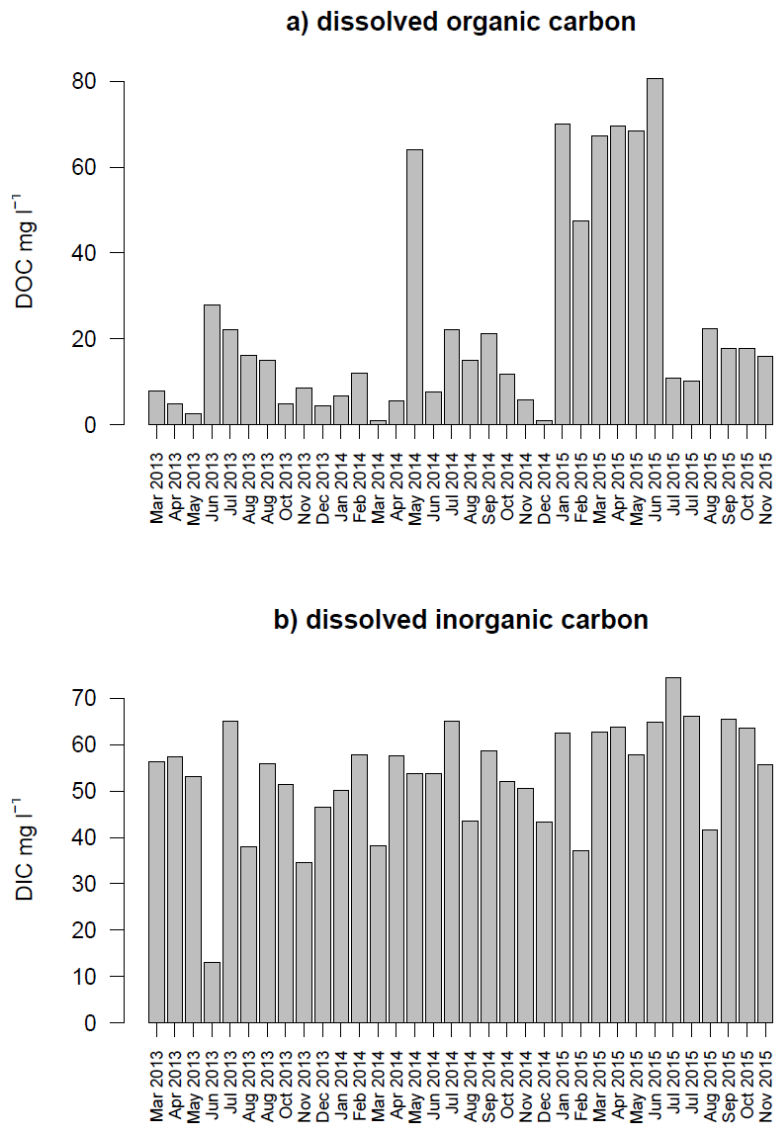


Figure 2.3.12. Mean and standard error of ditch dissolved organic and inorganic carbon concentrations, AF-LN. Note that x axis simply records sampling dates and is therefore not a true time axis.

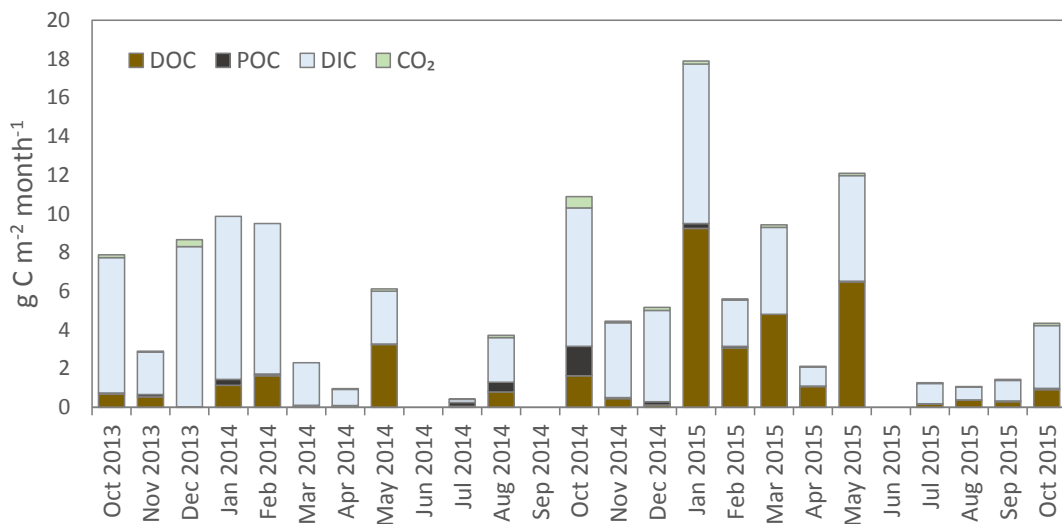


Figure 2.3.13. Estimated monthly aquatic carbon fluxes, AF-LN. DOC flux data missing for December 2013, POC flux data missing for March 2015, CO₂ fluxes missing January to March 2014; all other zero values indicate no flux.

2.3.2. Cors Erddreiniog – high nutrient site (AF-HN)

The ‘high nutrient’ site at Cors Erddreiniog is located in the southern part of the fen, and is considered by NRW to be relatively botanically impoverished, with stands of tall *Cladium mariscus* and *Phragmites australis* interspersed with *Molinia caerulea*, covering the site. This dominance of tall fen vegetation is thought to be indicative of nutrient enrichment, exacerbated by a lack of management by grazing or cutting in the recent past. The study site is classified as species poor M25 *Molinia caerulea*-*Potentilla erecta* mire. It is located in the central part of the fen, and surrounded by a network of perimeter ditches which, although now maintained at a high water level, are likely to reduce lateral near-surface inputs of base-rich water. It is thought that the area was historically subject to hand-cutting of surface peat, and some adjacent areas of the fen were affected by past cultivation, which may have transferred nutrients onto the site.

Peat core profile data (Figure 2.3.14) are fairly similar to the cores collected at the AF-LN site, but with a slightly higher mineral content and lower near-surface C/N ratio (14.7 versus 17.5 g g⁻¹), with particularly low values closer to the surface which appear consistent with its classification as a ‘high nutrient’ fen. Near-surface peat pH is lower than at AF-LN (5.0 vs 5.5), reflecting its relative isolation from base-rich groundwater inputs. Peat depth at AF-HN (around 3.2 m) is slightly greater than at AF-LN (2.8 m).

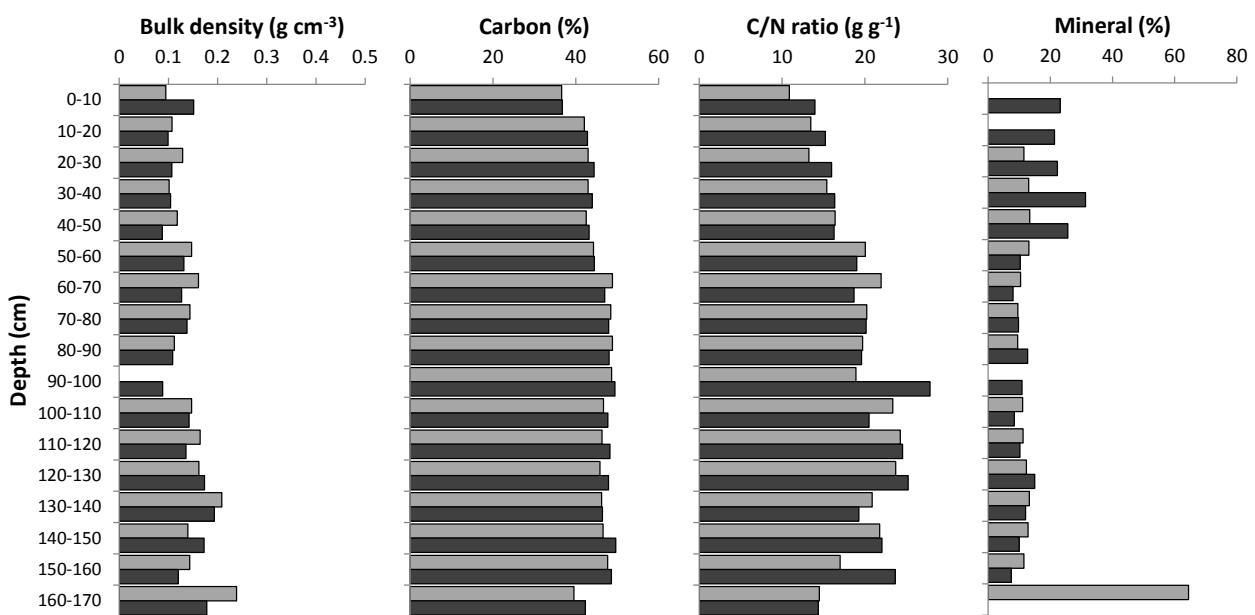


Figure 2.3.14. Peat core data, AF-HN. Data from two cores analysed from the site are represented by dark and light grey-shaded bars.

2.3.2.1. Hydrology

As at AF-LN, water tables remained generally close to the surface at AF-HN (Figure 2.3.15). The higher short-term variability in water level observed at AF-HN can be explained by its hydrological characteristics: whereas the AF-LN site is supplied by flow from the adjacent hillslope via groundwater, and is therefore slow to respond to rainfall events, AF-HN is surrounded by ditches and thus less hydrologically connected to regional groundwater flows. This means that water levels rise rapidly in response to individual rain events, before falling again as water is discharged to the surrounding ditch network. This situation resembles that observed at the arable and extraction sites, although as water levels are maintained at a high level by NRW the magnitude of fluctuations is comparatively small. Of the three continuously monitored dipwells, two recorded water levels were consistently at or just below the ground surface, whilst one showed frequent short periods of inundation (red trace in Figure 2.3.15). As at AF-LN, the AF-HN site experienced a short period of water-table drawdown in summer 2013, and more sustained periods of drawdown in the summers of 2014 and 2015. At all other times of year the site was largely saturated.

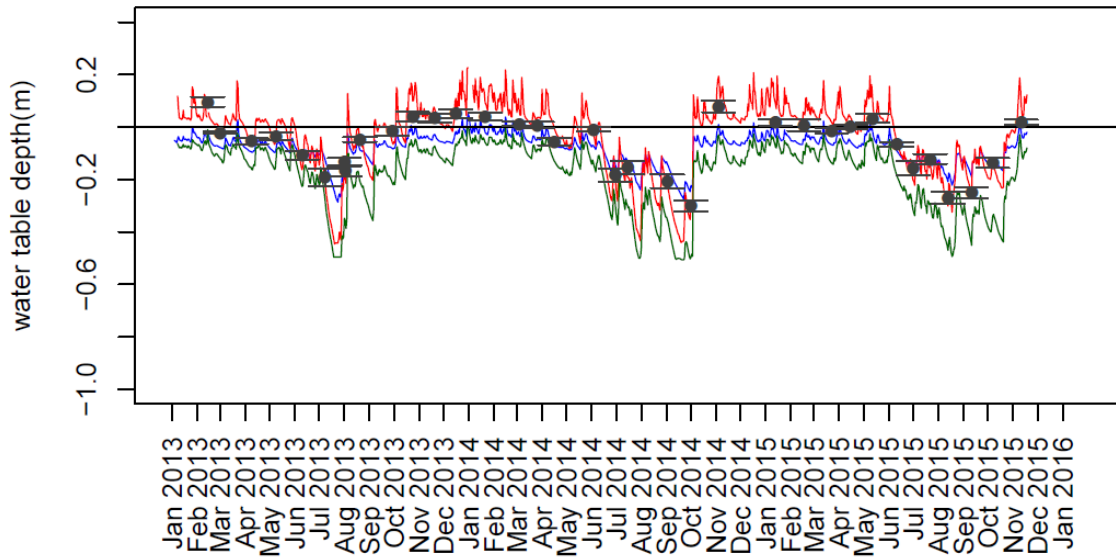


Figure 2.3.15. Continuously monitored and manually measured water-table data, AF-HN.

Given the relative hydrological isolation of AF-HN, lateral water inputs to the site were assumed to be negligible. The hydrological budget of the site is therefore dependent on the balance of precipitation and evapotranspiration, with any excess water assumed to be lost as Q_{out} to the ditches. Since both P_{net} and ET (from the flux tower) were measured on site, the water balance for AF-HN is considered to be fairly well constrained. During the majority of the measurement period, P_{net} exceeded ET, so discharge occurred, with the largest water losses in high rainfall autumn and winter months (Figure 2.3.16). Zero or near-zero flows occurred during the warm and dry summer months of June-July and September 2014, and June-September 2015, coinciding with the periods of water-table drawdown shown in Figure 2.3.15. The annual water discharge from AF-HN was estimated at 665 mm yr^{-1} , somewhat lower than AF-LN due to the lack of lateral water input and close to the areal mean discharge value at the nearby NRW gauging station of 602 mm yr^{-1} (<http://nrfa.ceh.ac.uk/data/station/info/102001>).

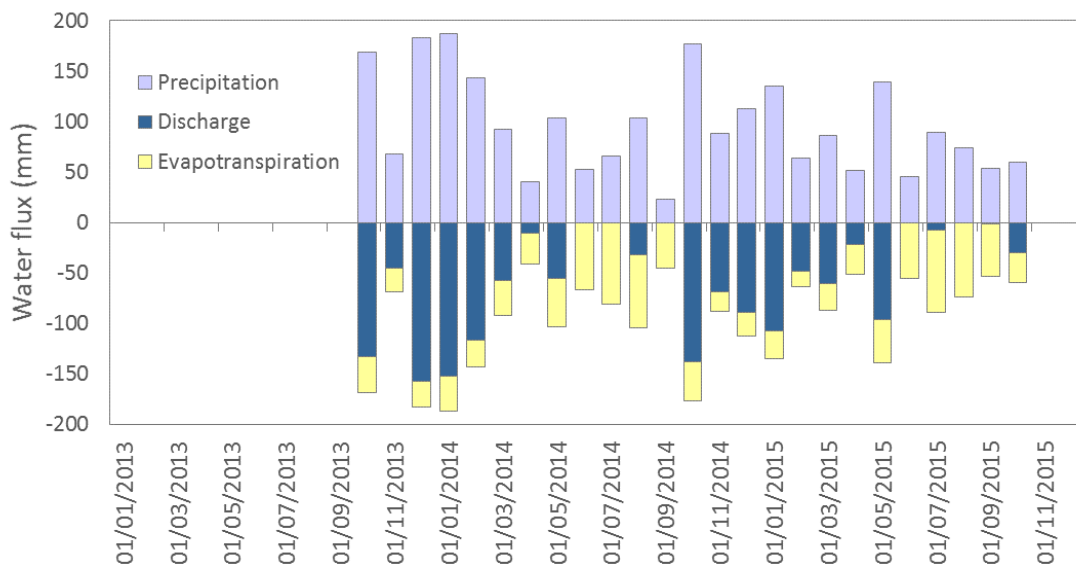


Figure 2.3.16. Monthly hydrological budgets for AF-HN

2.3.2.2. Eddy covariance gas fluxes

Fingerprint plots of NEE for AF-HN are shown in Figure 2.3.17. The eddy covariance system has been operational since October 2013 but has experienced a number of technical problems over this time period. Limitations to the (solar) electrical power supply system during the winter months and a technical problem with a datalogger resulted in the near-complete loss of data between mid-December 2013 and early April 2014. Electrical problems led to a further loss of data between mid-December 2014 and late January in 2015, and from 11th November 2015 onwards. Despite these data losses outside of the main growth period, data capture was high for the 2014 and 2015 growing seasons. As for EF-LN, although it is possible to fill the long data gaps using the standard gap-filling method, the application of gap-filling methods over a long data gap will be associated with large uncertainties for the period that is filled and derived annual sums. However, the subsequent capture of most of the period of peak growth and senescence in 2014 and 2015 provides a reasonable basis for estimating annual CO₂ fluxes.

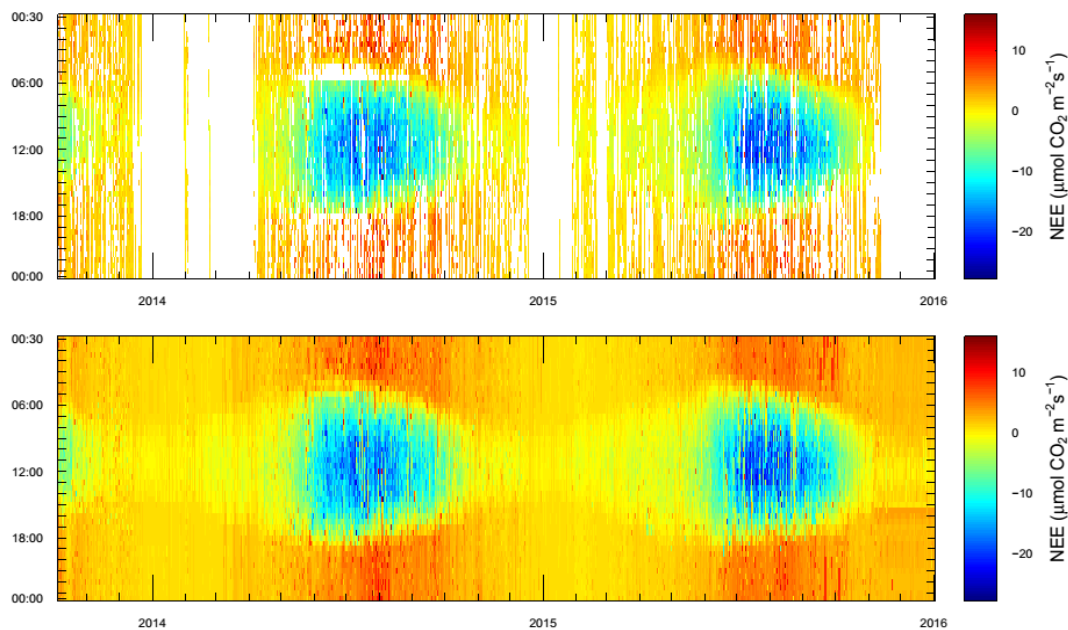


Figure 2.3.17. Fingerprint plot of measured (top panel) and measured and gap-filled (lower panel) net ecosystem carbon dioxide exchange for the AF-HN (Cors Erddreiniog – high nutrient) site. Units are in $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$.

The AF-HN site has a similar vegetation composition to the EF-LN (Wicken Fen) site and therefore shows a broadly similar pattern in terms of seasonal NEE (compare Figures 2.3.17-2.3.18 with Figures 2.1.7-2.1.8). There are also close similarities in measured fluxes between AF-HN and the adjacent AF-LN during the 2015 common measurement year, suggesting similar phenology between the short- and tall-fen vegetation types. Similar to other sites with semi-natural vegetation cover, the amplitude of the diurnal averaged CO₂ exchange increased from April to a seasonal maximum in July, then declined steadily through late summer and autumn. Maximum daytime uptake rates were highest at AF-HN during the warm spring conditions in 2014, as observed at other network sites (see e.g. EF-EG). Peak season (July) estimates of maximum net CO₂ uptake and nighttime CO₂ efflux were $-16.63 \pm 0.56 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ and $5.83 \pm 0.15 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ in 2014 and $-18.8 \pm 0.53 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ and $4.88 \pm 0.12 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ in 2015. Maximum rates of CO₂ uptake are slightly less negative than those measured at EF-LN, whereas the maximum CO₂ efflux rates are less positive. Conversely, maximum uptake and loss rates were larger than at AF-LN during the common measurement period. Net CO₂ uptake rates were higher between July and September in 2015 compared to 2014, whereas nocturnal CO₂ efflux rates were slightly less positive (August) or similar for the two years. With the exception of slightly larger daytime fluxes in October 2013 relative to October 2014, mean CO₂ flux densities were similar during the autumn and early winter months of 2013, 2014 and 2015 (Figure 2.3.18).

Over the two full years of flux measurement, AF-HN was a strong sink for CO₂, with measured NEE of -176 (± 84) g C m⁻² yr⁻¹ in 2014, and -139 (± 76) g C m⁻² yr⁻¹ in 2015. For 2015, when both flux towers were operating in parallel, the tall fen AF-HN thus drew down 60% more CO₂ than the short fen AF-LN.

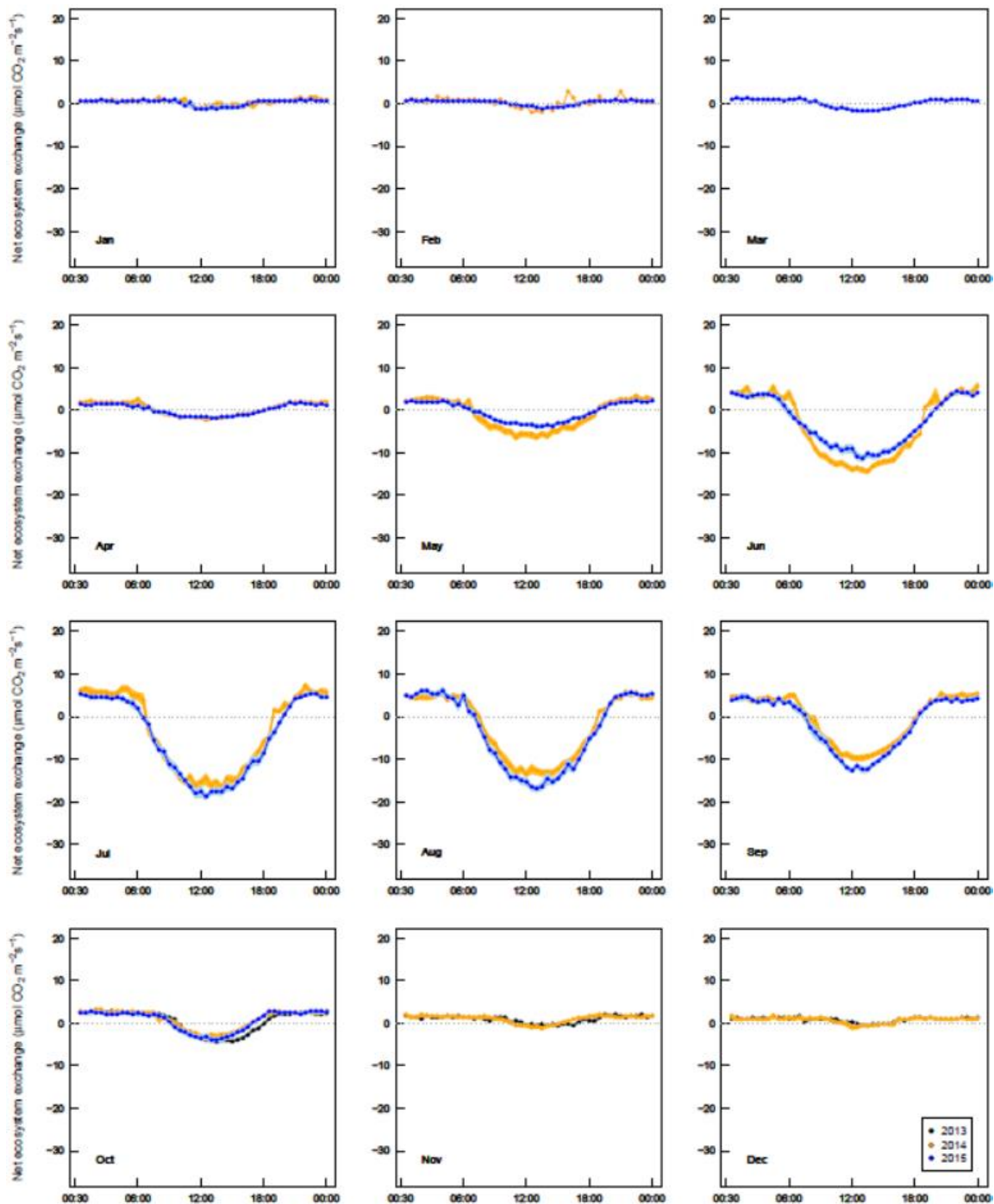


Figure 2.3.18. Mean diurnal cycles for each month of the growing season that measurements were available for the AF-HN (Cors Erddreiniog – high nutrient) site. Data points are the mean of data measured at the same time of day during each month. Shaded areas show one standard error of the mean. Monthly diurnal averages calculated using measured (not gap-filled) data.

2.3.2.3. Static chamber gas fluxes

Static chamber measurements were made at AF-HN within the two major vegetation communities, dominated by *Phragmites australis* and *Cladium mariscus* respectively. Measured and modelled CO₂ fluxes are shown in Figure 2.3.19. For the *Phragmites*, which senesces in winter, GPP was modelled as a function of

photosynthetically active radiation (PAR), soil temperature and ETI (see Appendix 2) which effectively shuts down photosynthetic activity when temperatures are low, consistent with near-zero measured GPP during the winter period. For *Cladium*, which remains green throughout the year, some photosynthetic uptake was measured during winter, and the best fit was obtained with a model including PAR and soil temperature only. For both plant communities, ER was modelled as an exponential function of temperature. Model fits were generally reasonable, although the *Cladium* ER and GPP models both slightly under-predicted winter fluxes. Modelled NEE values (lower panels of Figure 2.3.19) suggest that both plants were net sinks for CO₂ on a daily basis throughout the summer, with negligible net fluxes in winter. Over the full study period, estimated mean annual GPP was 1070 g C m⁻² yr⁻¹ in the *Phragmites*, and 1404 g C m⁻² yr⁻¹ in the *Cladium*. Mean estimated annual ER was 929 g C m⁻² yr⁻¹ in the *Phragmites*, and 820 g C m⁻² yr⁻¹ in the *Cladium*, giving modelled net CO₂ uptake (NEE of -140 and -583 g C m⁻² yr⁻¹ in the respective communities). On the basis that the site is around 75% covered by *Phragmites*, and 25% by *Cladium*, this gives a site mean NEE of -251 g C m⁻² yr⁻¹. This is somewhat larger than the annual CO₂ uptake values obtained by the flux tower (see above), but of a similar order, and the methods would converge further if a higher estimate of proportional *Phragmites* cover were used. Overall, the two methods provide consistent evidence that AF-HN is acting as a strong sink for CO₂.

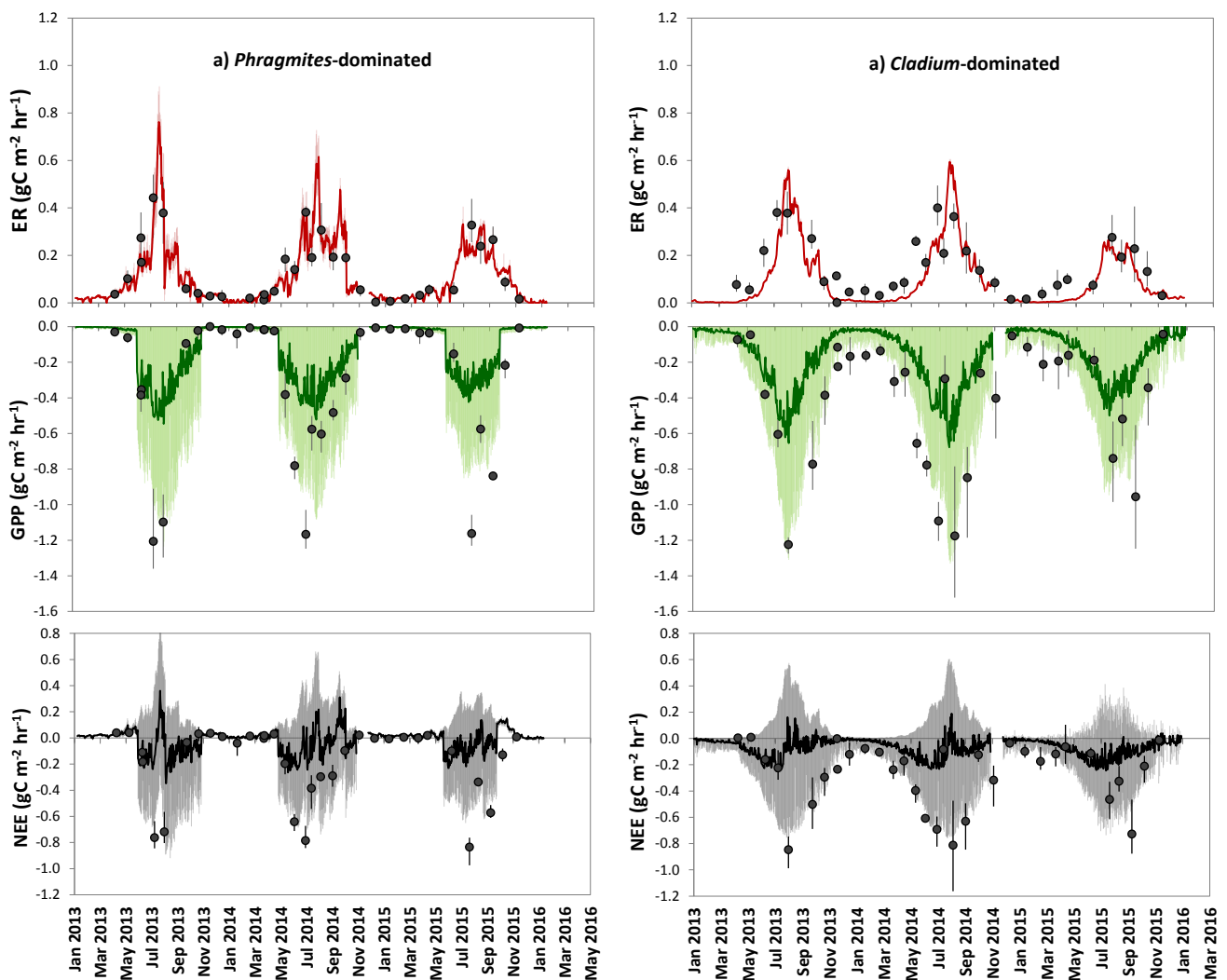


Figure 2.3.19. Modelled and observed CO₂ fluxes (ER = ecosystem respiration, GPP = gross primary productivity, NEE = net ecosystem exchange) based on static chamber measurements in two vegetation communities at AF-HN. Continuous lines show modelled daily mean fluxes, shading shows modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date. Note that static chamber measurements were generally taken around the middle of the day, so tend to be representative of peak (rather than daily mean) rates of photosynthesis and respiration.

Measured static chamber CH_4 data are shown in Figure 2.3.20. As at AF-LN, it was not possible to fit an empirical model to the data, and fluxes were therefore interpolated between sampling points and annual means calculated. The data showed a reasonably consistent seasonal pattern for *Phragmites* in particular, although with generally higher fluxes in summer 2014 compared to 2013 or 2015. Fluxes were consistently lower in the *Cladium* areas. Estimated mean annual fluxes were $16.6 \text{ g C m}^{-2} \text{ yr}^{-1}$ for the *Phragmites*, and $6.8 \text{ g C m}^{-2} \text{ yr}^{-1}$ for the *Cladium*. Based on a 75%/25% cover of these vegetation communities, the mean CH_4 emission from AF-HN is estimated as $14.1 \text{ g C m}^{-2} \text{ yr}^{-1}$.

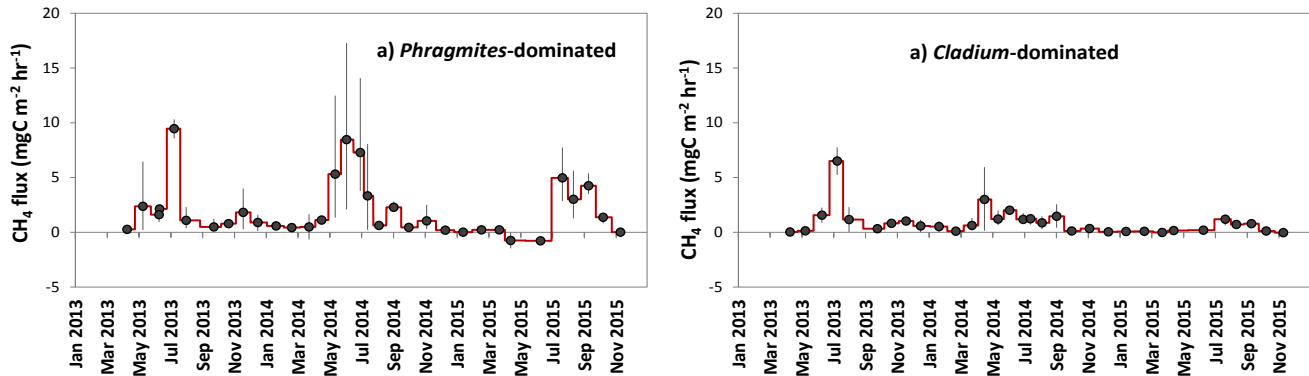


Figure 2.3.20. Measured CH_4 fluxes for the *Phragmites*- and *Cladium*-dominated vegetation communities at AF-HN. Points show mean observations on each measurement date, and error bars show range of measured values on that date. Red line shows interpolated fluxes.

Floating chamber measurements were made in a ditches adjacent to AF-HN (Figure 2.3.21). Measurements were very limited because the ditch dried up during summer, hence it was not possible to evaluate temporal dynamics, and the associated flux estimates are highly uncertain. For CH_4 , the ditch appears to be a consistent source, in the region of $23 \text{ g C m}^{-2} \text{ yr}^{-1}$. Assuming a F_{ditch} value of 0.01, this translates to an areal mean emission from the site of $0.23 \text{ g C m}^{-2} \text{ yr}^{-1}$. The mean CO_2 flux varied either side of zero, with no clear indication of net CO_2 uptake or removal.

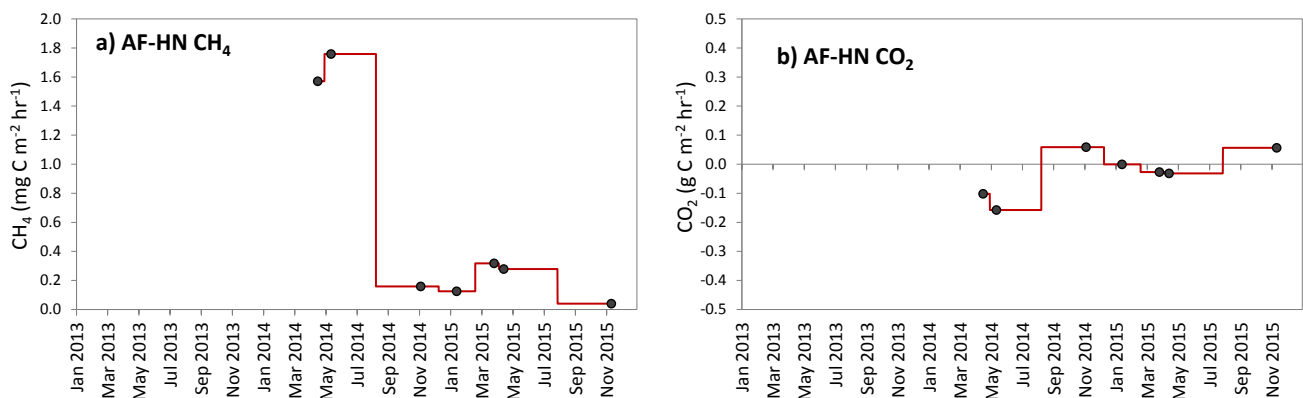


Figure 2.3.21. Observed CH_4 and CO_2 fluxes measured in ditches at AF-HN. Red line shows interpolated fluxes.

2.3.2.4. Aquatic carbon fluxes

Compared to AF-LN, mean DOC concentrations at AF-HN were normally higher, whereas DIC concentrations (and pH, with a mean of 7.13) were normally lower (Figure 2.3.22). These differences are all consistent with the isolation of the AF-HN site from groundwater inputs, in part due to its location towards the middle of the fen, and in part to the isolating effects of the surrounding ditch network. However the period of elevated DOC concentrations observed at AF-LN during early 2015 was much less evident at AF-HN. Concentrations of DIC peaked in summer during all three sampling years, with the highest concentrations recorded in 2013.

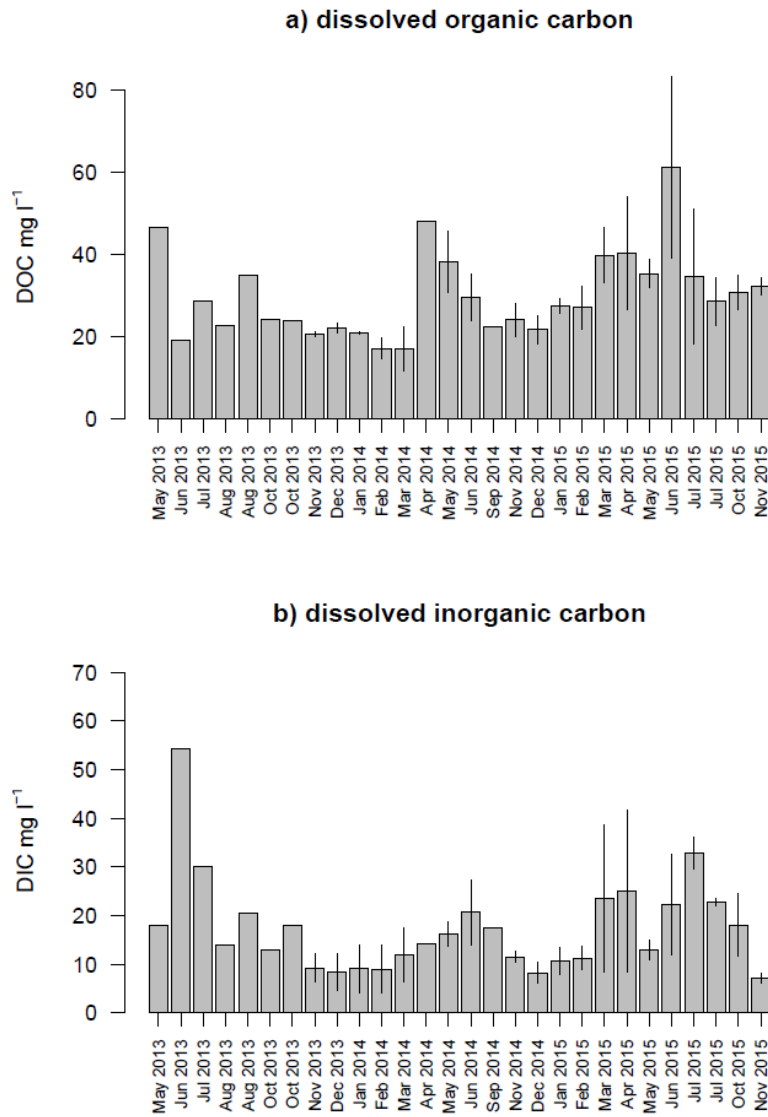


Figure 2.3.22. Mean and standard error of ditch dissolved organic and inorganic carbon concentrations, AF-HN. Note that x axis simply records sampling dates and is therefore not a true time axis.

Aquatic C fluxes from AF-HN are shown in Figure 2.3.23. Due to the combination of lower groundwater inputs and lower flows, DIC fluxes were less than half those from AF-LN (estimated annual mean 15.0 g C m⁻² yr⁻¹). On the other hand, DOC fluxes were considerably higher (annual mean 31.9 g C m⁻² yr⁻¹) due to the higher DOC concentrations at AF-HN, giving the highest DOC export of any fen site in the study. POC fluxes were also the highest recorded at any site, with high fluxes in some winter months giving an estimated annual mean of 8.6 g C m⁻² yr⁻¹. Dissolved CO₂ fluxes were smaller, and similar to AF-LN (mean 1.3 g C m⁻² yr⁻¹), and dissolved CH₄ fluxes were negligible (0.01 g C m⁻² yr⁻¹).

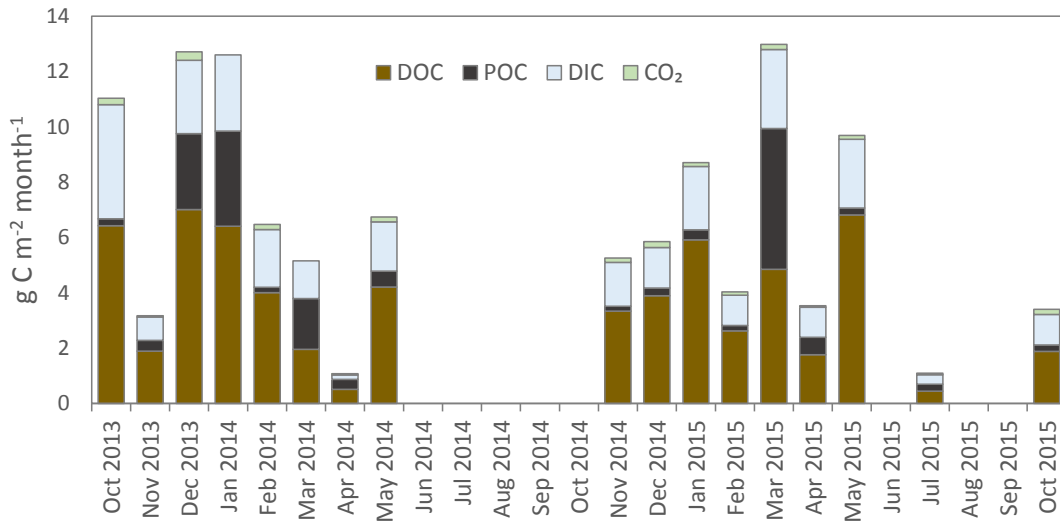


Figure 2.3.23. Estimated monthly aquatic carbon fluxes, AF-HN. Data missing for August and October 2014, and September 2015; other zero values indicate no flux.

2.4. Somerset Levels

The Somerset Levels and Moors Environmentally Sensitive Area (ESA) extends over 277 km² of the central Somerset lowlands, bounded by the Mendips to the north, low limestone escarpments to the east, the Blackdown Hills to the south and the Quantock Hills to the west. The moors are an extensive very low-lying basin peat, with a few remnants of raised bog, surrounded by alluvial silt and clay. The peat areas lie within a number of large valleys which flow northwest towards the Bristol Channel, separated by a series of low ridges. The peat is overlain in places by a varying thickness of riverine clay. Although less extensively drained than the East Anglian Fens, water levels are heavily managed throughout the Somerset Levels by a dense network of ditches with pumped drainage into the river network. Brunning (2001) reported subsidence rates of peat under pasture in the Somerset Levels of 0.44 to 0.79 cm yr⁻¹, and estimated that some areas of raised bog have lost 4 m in elevation since the mediaeval period. Much of the northern part of the Levels is mapped as wasted peat (Figure 2.4.1). However, relatively little of the area is under arable cultivation, with much of the peatland area used for pasture at varying levels of intensity. The Somerset Levels as a whole comprise the largest lowland grazing marsh system in Britain. Around 260 ha of the Levels are currently used for peat extraction. Both study sites are located within the Brue Valley, at the northern edge of the remaining area of deep peat (Figure 2.4.1).

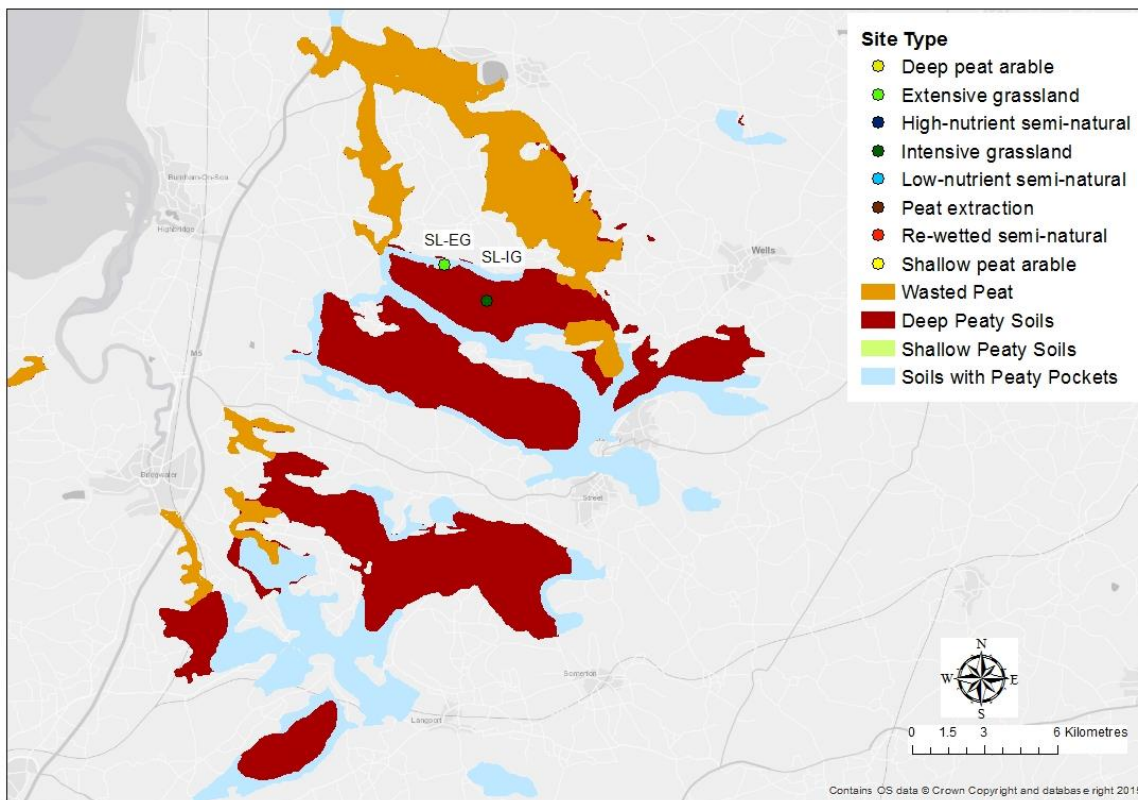


Figure 2.4.1. Somerset Levels study area.

Meteorology for both SL sites is represented by an AWS at the SL-EG site. Figure 2.4.2 shows monthly values for the primary meteorological variables measured at SL-EG between November 2012 and December 2015. Precipitation measurements were unreliable at SL-EG between October 2012 and October 2014 and required gap-filling using external data. Measurements of precipitation became more reliable after a COSMOS-UK station (see: <http://cosmos.ceh.ac.uk/node/430>) was installed at the site on 14th October 2014. Monthly precipitation totals for the period prior to this were obtained from the CEH GEAR (Gridded Estimates of Areal Rainfall) dataset (Keller et al., 2015; Tanguy et al., 2015) which currently extends to December 2014. Wind rose plots are shown in Figure 2.4.3. Wind flow at the SL-EG site was strongly bimodal, with prevailing easterly and westerly winds broadly aligned with the orientation of the Brue Valley.

Total monthly SW_{in} (top panel in Figure 2.4.2) reached a seasonal (and measurement period) maximum in July 2013 (394 kW m^{-2}). Maximum SW_{in} values were similar for June and July 2014, and in June 2015. Notable between-year differences in monthly SW_{in} were observed between January and April 2013 (lowest during 2013), for July and August (lowest values during 2015), and during September (highest values during 2015). SW_{in} was lower in November and December 2012 than for the corresponding months of the subsequent years.

Mean annual air temperatures at SL-EG were $10.0 \text{ }^\circ\text{C}$, $11.1 \text{ }^\circ\text{C}$ and $10.6 \text{ }^\circ\text{C}$ for 2013, 2014 and 2015, respectively. Seasonal trends in monthly mean air temperatures (middle panel in Figure 2.4.2) showed broadly similar patterns to other network sites. As elsewhere, notable between-year differences are the January to June period (coolest in 2013; warmest in 2014), during summer and early autumn (coolest during 2015) and early winter (warmest in December 2015).

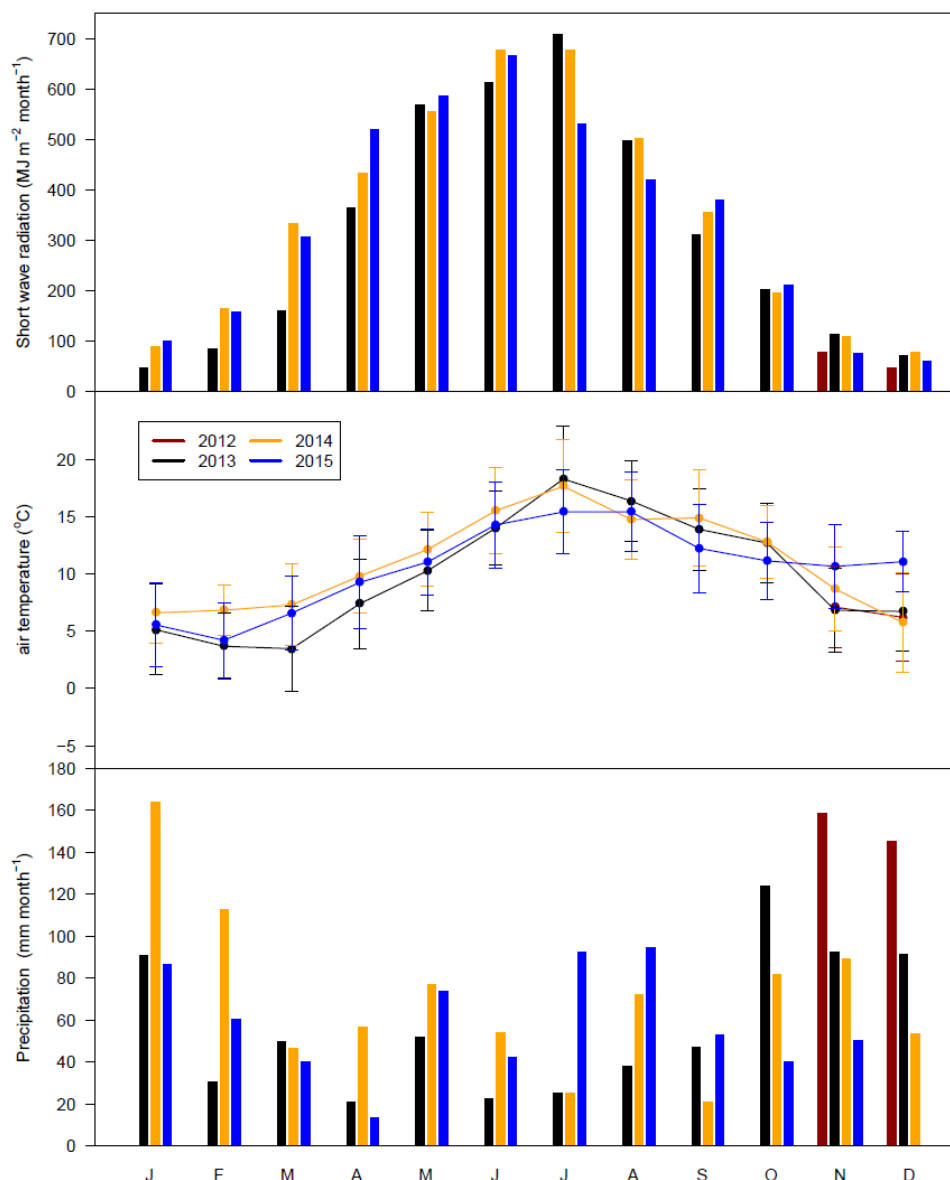


Figure 2.4.2. Total monthly incoming solar radiation (top) mean monthly air temperature (middle) and total monthly precipitation (lower) for the SL-EG (Tadham Moor) site. Error bars on the temperature plot show one standard deviation of the mean.

Total annual rainfall at SL-EG was estimated at 685 mm yr⁻¹ in 2013 and 854 mm yr⁻¹ in 2014 (both years based on the CEH GEAR dataset). Missing data in December 2015 precludes an annual value for this year; however, the precipitation sum was 648 mm between January and the end of November. Monthly precipitation showed large seasonal and between-year variability. The months of November and December in 2012, October 2013 and January - February in 2014 received over 100 mm of rainfall, with the latter period resulting in severe flooding across the Somerset Levels. By contrast, a number of months received low monthly precipitation totals. The driest months for each year were April in 2013 and 2015, and September in 2014. A prolonged dry period was observed between February and September 2013 with all months receiving less than 52 mm month⁻¹.

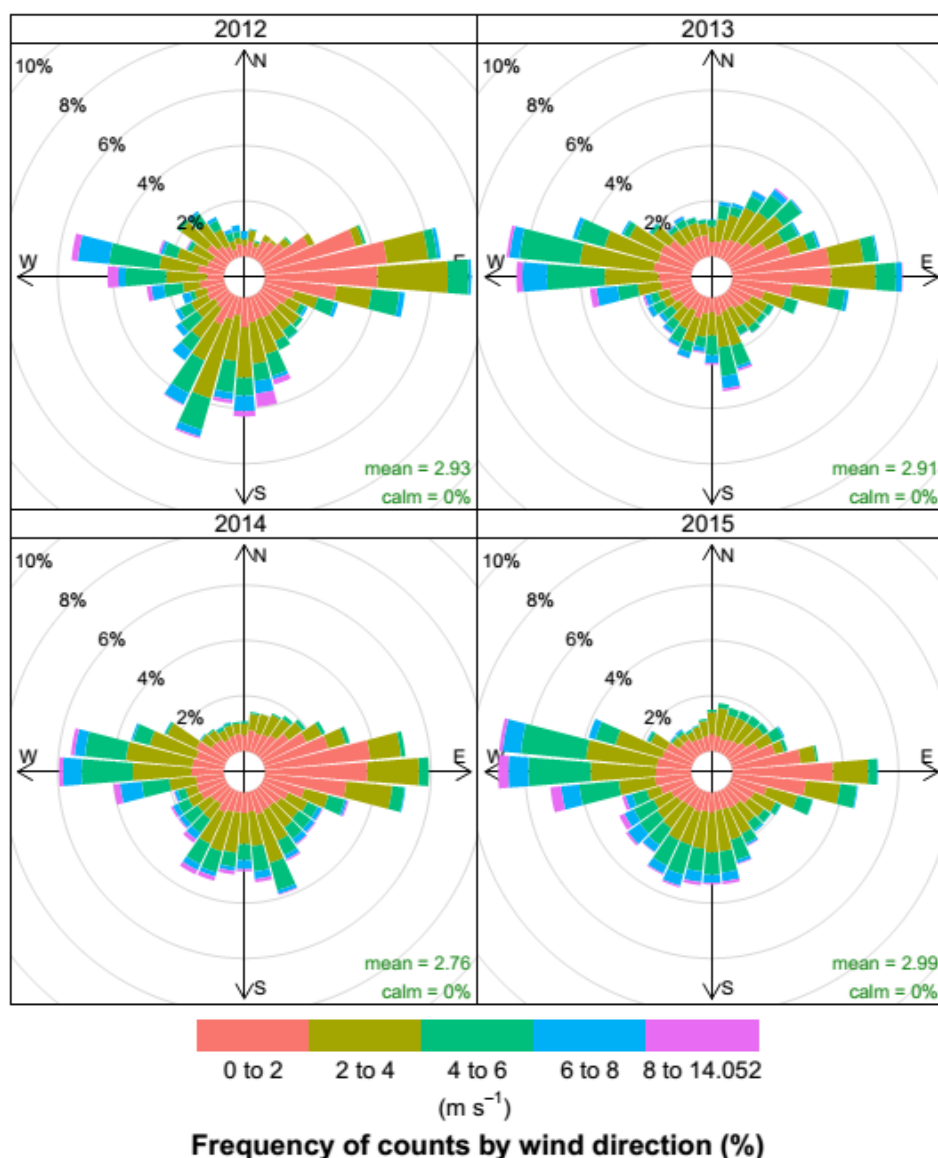


Figure 2.4.3. Wind rose plots showing wind direction and wind speed at the SL-EG (Tadham Moor) site.

2.4.1. Somerset Levels – Tadham Moor extensive grassland (SL-EG)

Tadham and Tealham Moors SSSI comprises an area of 20 ha of deep loamy peat of the Altcar 1 soil series, overlaid in some areas by acid fibrous peat of the Turbary Moor series, in the floodplain of the River Brue. The area is intersected by an extensive network of drainage ditches, in which the water level is controlled by pumping stations. The site is extensively managed to maintain semi-natural grassland, with hay cutting in July followed by aftermath grazing with cattle, and receives no inorganic fertiliser inputs. The ditch network is used to maintain water levels in summer, while free drainage occurs during winter. The site is flooded annually by over-topping of the River Brue. Previous research at SL-EG has included estimation of an annual carbon balance, which included several years of CO₂ eddy covariance measurements (Lloyd, 2006), and assessment of the effects of inorganic fertilisers on flower-rich hay meadows. The field in which measurements were made supports a diverse grass-, sedge-, rush- and forb-rich wet grassland vegetation. The vegetation conforms most closely to NVC class MG8 (*Cynosurus cristatus* - *Caltha palustris*) grassland, which is characteristic of periodically inundated land that has been traditionally managed as a water meadow (See Appendix 3).

Peat cores collected from SL-EG (Figure 2.4.4) show that the peat has fairly uniform properties to a depth of at least 150 cm, although there is evidence of slight mineral-enrichment of the upper 40 cm. The bulk

density of the upper 50 cm of peat is in the mid-range of the study sites (0.34 g cm^{-3}), as is the pH (5.6). The C/N ratio of the peat is fairly high, at around 30 g g^{-1} .

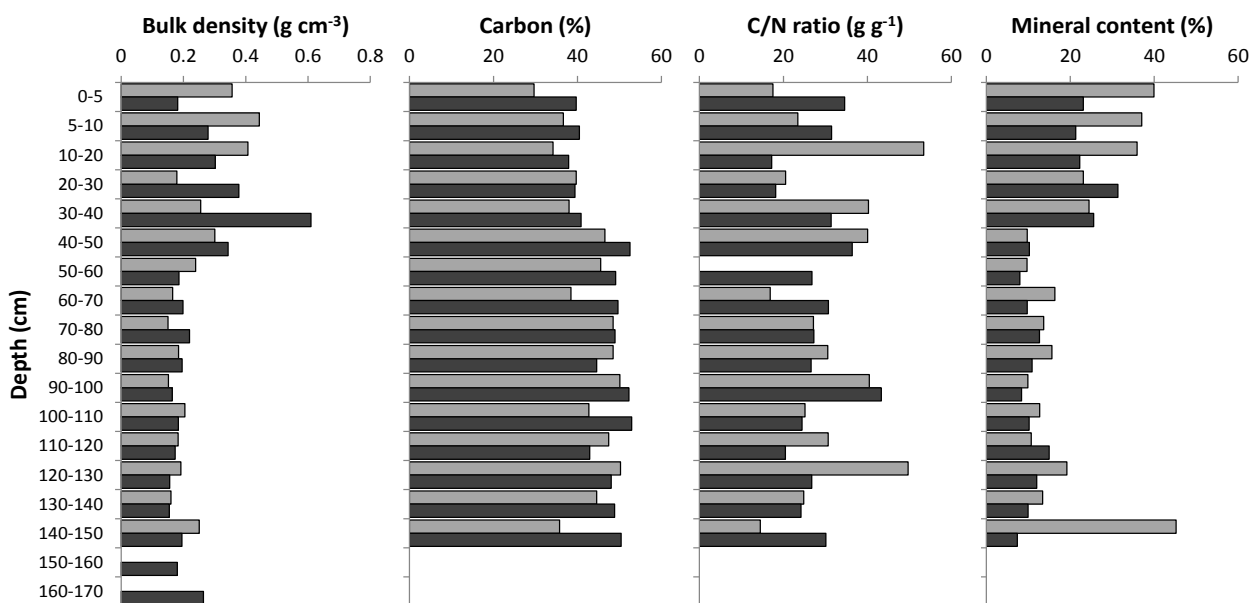


Figure 2.4.4. Peat core data, SL-EG. Data from two cores analysed from the site are represented by dark and light grey-shaded bars.

2.4.1.1. Hydrology

Water-table data for SL-EG (Figure 2.4.5) show very strong seasonality, with water levels close to the surface during winter, and progressive drawdown from early spring through to late summer, reflecting the hydrological management of this wet meadow ecosystem. Topographic data and three dipwell transects across the site indicate that there is a moderate east west gradient in the water table, with a typically convex shape indicating that water tables were closest to the surface in the middle of the site. As noted above, the water table tended to be higher than the level of the ditch network during much of the year, but fell below ditch levels during summer, particularly in 2013. The severe flooding that affected the Somerset Levels in early 2014 caused complete inundation of the site. The logger data show that at the peak of the flooding the field was around 25 cm underwater, at which time water levels actually became tidal. The greatest degree of water-table drawdown (to a depth of 70 cm) took place in summer 2013, with somewhat higher levels maintained in subsequent summers. Water tables approached the surface during the winter of 2014-15, but no flooding took place, possibly because ditch levels were being maintained at lower levels following the floods.

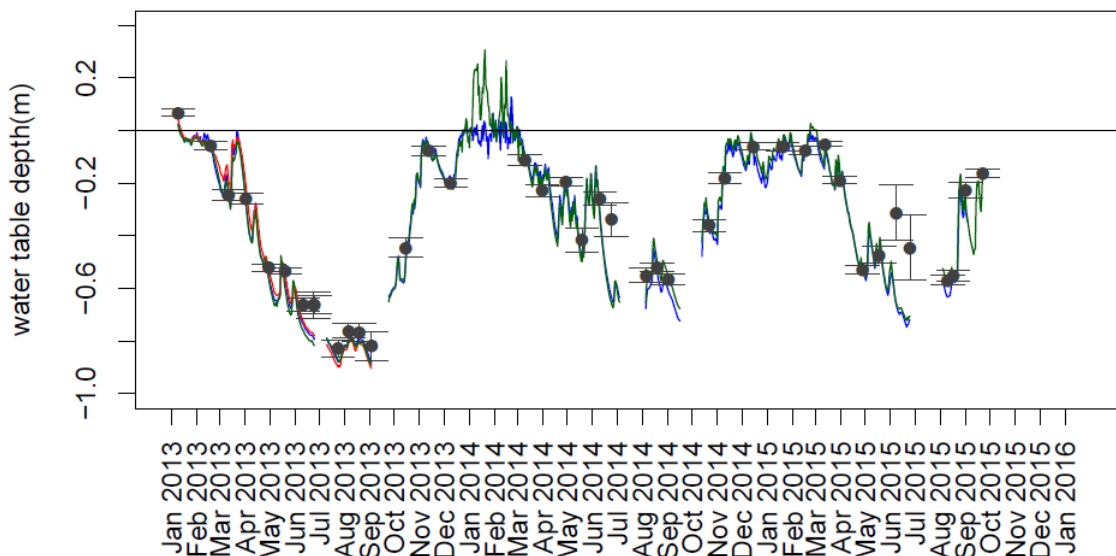


Figure 2.4.5. Continuously monitored and manually measured water-table data, SL-EG.

The monthly water budget of SL-EG is shown in Figure 2.4.6. Precipitation was measured on site (see above), and evapotranspiration was derived from the flux tower. Given the position of the peat on impermeable marine clays, and the surrounding ditch network, groundwater inputs were assumed to be zero. The observed variations in water table meant that a substantial volume of water was transferred to and from storage during the year, for which a specific yield of 0.2 was used based on previous hydrological analysis of the site by Stratford and Acreman (2014). During summer periods when ditch levels were higher than the water table, water was assumed to be moving onto the site, with all water losses occurring via evapotranspiration, i.e. no lateral discharge. When water tables were higher than ditch levels, discharge could occur. This was the case in all winter periods, with the highest discharge rates occurring in the periods when flooding took place, in November-December 2012 and December 2013-February 2014, and smaller discharges in the winter of 2014-15 when flooding did not occur. Note that it was not possible to quantify the amount of water transferred laterally onto or off the site during the periods of large-scale floodplain inundation. The average estimated water loss from SL-EG was 350 mm yr⁻¹.

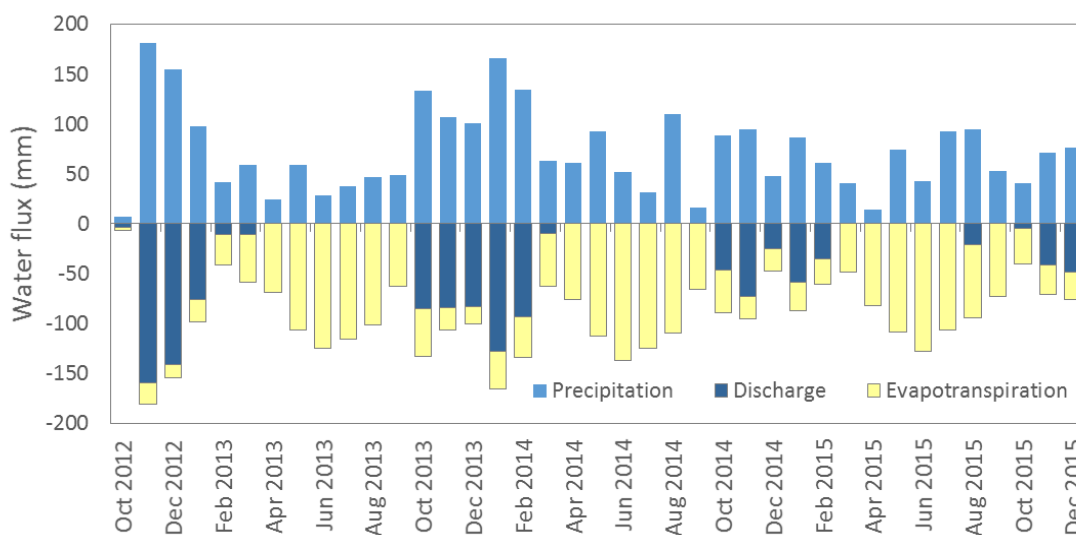


Figure 2.4.6. Monthly hydrological budgets for SL-EG

2.4.1.2. Eddy covariance gas fluxes

The flux tower at SL-EG has been operational since late October 2012 (Figure 2.4.7). A number of gaps in the flux record have occurred at this site due to power outages and instrument failures (top panel). Notably, a

data gap caused by the loss of electrical power to a gas analyser resulted in the loss of flux data between mid-November 2013 and mid-January 2014 (although meteorological data were captured during this time period). As this data gap occurred outside of the main growth season at this site, the data gap was filled using the standard gap-filling method (Reichstein et al., 2005; Reichstein & Moffat, 2015). The gap-filling was performed so that the part of the data gap in each year was filled using data collected during that calendar year (e.g. from mid-November to 31st December 2014, and from 1st January 2014 to mid-January 2014). It is acknowledged that this approach increases the uncertainty in the annual CO₂-C flux estimates, but enables annual CO₂-C budgets to be derived for the complete calendar years of 2013 and 2014.

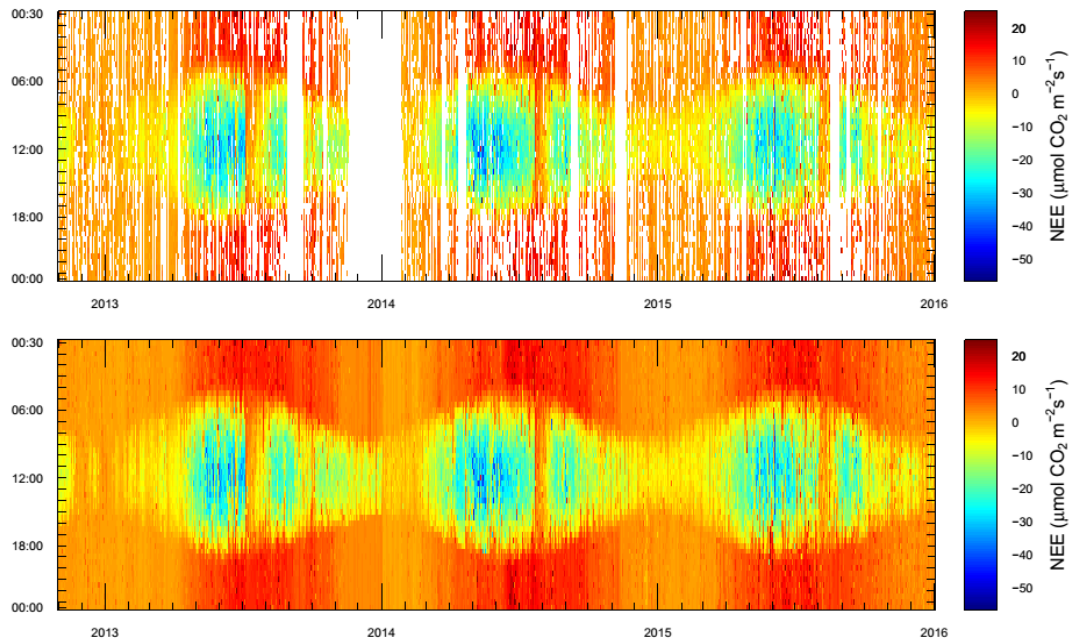


Figure 2.4.7. Fingerprint plot of measured (top panel) and measured and gap-filled (lower panel) net ecosystem carbon dioxide exchange for the SL-EG (Tadham Moor) site. Units are in $\mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$.

The seasonal pattern of NEE at SL-EG is similar to the other managed grassland (EF-EG) sites in the lowland peatland flux tower network. In contrast to EF-EG, where biomass is managed by grazing alone, the SL-EG site is cut for hay each summer. In all three growing seasons, the impact of the hay cut is evident in midsummer (as a distinct ‘red stripe’ in Figure 2.4.7) when daytime fluxes become positive in the weeks following the cut, before becoming progressively more negative again as photosynthesis increases with the regrowth of the grass.

The mean diurnal cycles (MDCs) for SL-EG (Figure 2.4.8) reveal large seasonal and between-year differences in NEE. In particular, and similar to the EF-EG site, the amplitude of the monthly mean diurnal cycles was larger during the warm spring and early summer months of 2013 and 2014 compared to the colder conditions of the preceding spring, with the highest uptake rates measured during 2014. Maximum net CO₂ uptake rates at SL-EG were second in magnitude only to the productive wheat and maize crops at EF-SA at $-28.63 \pm 1.26 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ in June 2013, $-27.44 \pm 1.68 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ in May 2014 and $-24.53 \pm 1.26 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ in May 2015. The maximum (monthly average) nocturnal CO₂ efflux rates at SL-EG are the highest of any of the sites in the network at $12.62 \pm 0.16 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$, $13.61 \pm 0.22 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ and $13.07 \pm 0.31 \mu\text{mol CO}_2 \text{ m}^{-2} \text{ s}^{-1}$ in 2013, 2014 and 2015, respectively. These maximum monthly average nocturnal CO₂ emissions were observed in June in 2013, and during July in 2014 and 2015. The influence of the hay cut is evident in Figure 2.4.8 as reduced rates of CO₂ uptake during the midsummer period (the exact timing of which varied between years), as is the influence of regrowth on the mean diurnal pattern during late summer and early autumn. In 2015, the amplitude of the mean diurnal pattern was lower during August

than in other years. Average net CO₂ uptake was higher during 2015 than during 2013 and 2014, although nocturnal losses were lower.

Over the full monitoring period, SL-EG had the highest annual values of both GPP and ER in all three years, implying a very high overall rate of carbon cycling under the productive permanent grassland vegetation. Nevertheless, the site was a net source of CO₂ in all years, from +22 g C m⁻² yr⁻¹ in 2014 to +149 g C m⁻² yr⁻¹ in 2015 (see Section 4.1).

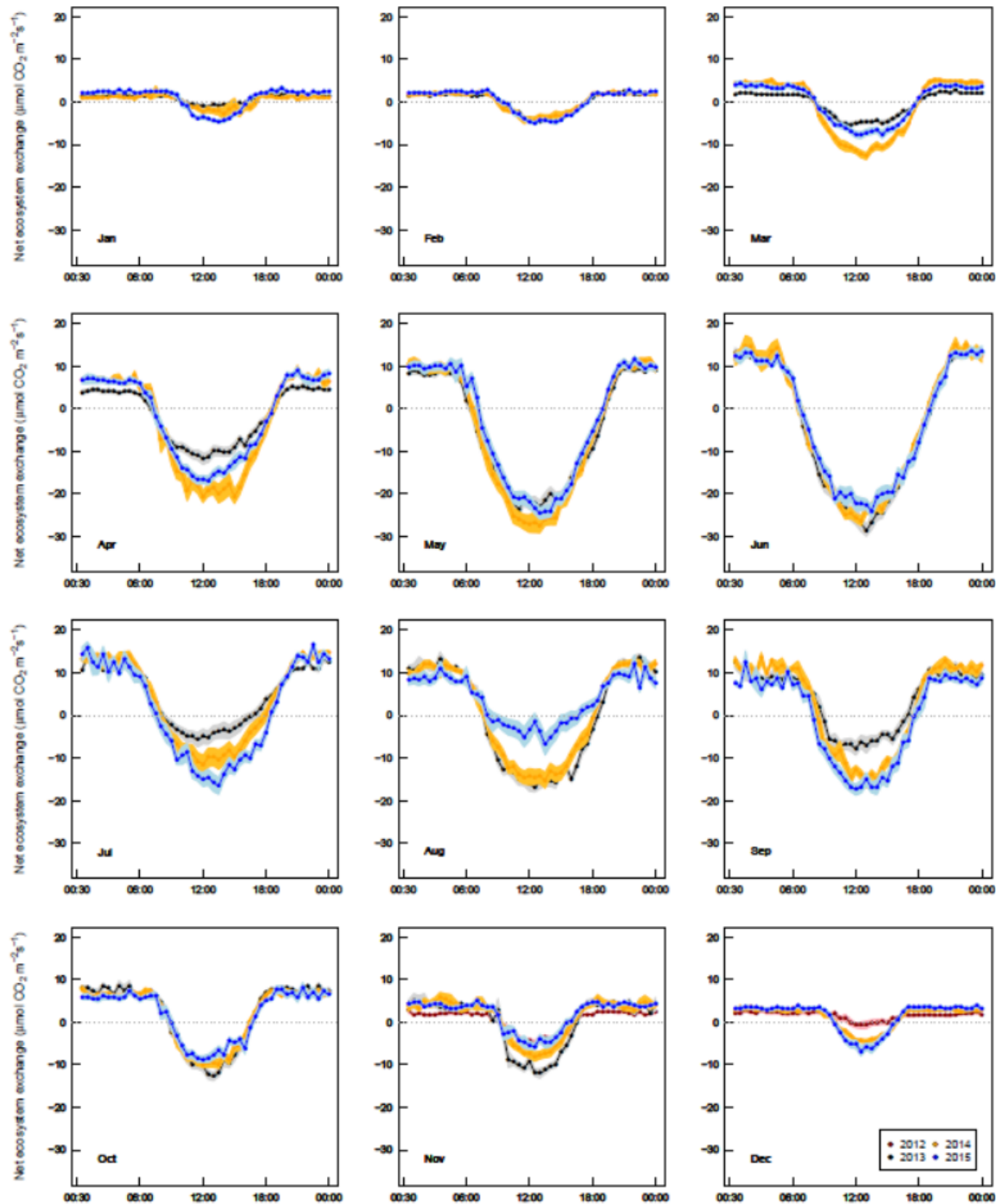


Figure 2.4.8. Mean diurnal cycles for each month that measurements were available for the SL-EG (Tadham Moor) site. Data points are the mean of data measured at the same time of day during each month. Shaded areas show one standard error of the mean. Monthly diurnal averages calculated using measured (not gap-filled) data.

2.4.1.3. Static chamber gas fluxes

Measured and modelled static chamber CO₂ fluxes for SL-EG are shown in Figure 2.4.9. Due to the active management of the site it was not possible to fully reflect seasonal changes in grass growth and associated respiration through the use of simple models based on climate variables because (as shown in the flux tower data above) photosynthesis rates dropped dramatically following the summer hay cut. As a result, the fitted model tended to under-predict GPP (and ER, which includes the autotrophic component associated with grass growth) in the early part of the summer. From late summer onwards, on the other hand, ER appears to be over-predicted. The annual mean GPP of 1253 g C m⁻² yr⁻¹ derived from the modelled static chamber data was less than half the GPP derived from the flux tower dataset (2610 g C m⁻² yr⁻¹) as a result of this. The chamber-derived ER of 2040 g C m⁻² yr⁻¹ was closer to the flux tower mean of 2700 g C m⁻² yr⁻¹, and as a result the chamber-derived NEE was +787 g C m⁻² yr⁻¹. This very high value is not considered to be realistic for the site, and the flux tower CO₂ data were therefore used in preference.

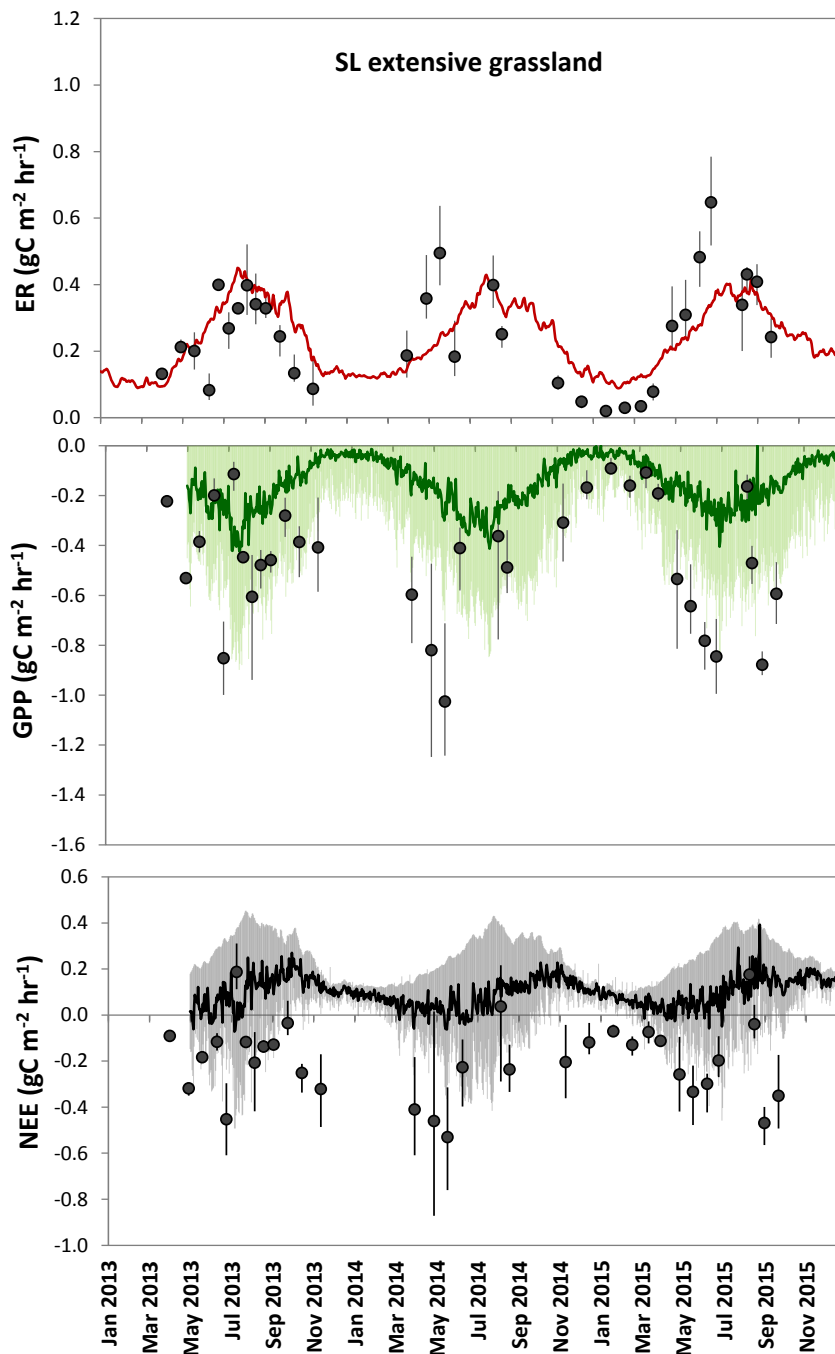


Figure 2.4.9. Modelled and observed CO₂ fluxes (ER = ecosystem respiration, GPP = gross primary productivity, NEE = net ecosystem exchange) based on static chamber measurements from the field at SL-EG. Continuous lines show modelled daily mean fluxes, shading shows modelled diurnal range, points show mean

observations on each measurement date, and error bars show range of measured values on that date. Note that static chamber measurements were generally taken around the middle of the day, so tend to be representative of peak (rather than daily mean) rates of photosynthesis and respiration.

Land surface CH₄ fluxes at SL-EG were predominantly small and negative (Figure 2.4.10), although occasional spikes of CH₄ were recorded from individual collars, typically in late summer. Overall the field was estimated to be acting as a net CH₄ sink of -0.17 g C m⁻² yr⁻¹. In contrast, the ditches at SL-EG were substantial sources of CH₄, with very high flux rates recorded in all three growing seasons (Figure 2.4.11). The estimated annual CH₄ emission per unit ditch surface area was 112 g C m⁻² yr⁻¹. The fractional ditch area of the site was estimated to be 0.027, giving an area-adjusted ditch CH₄ emission for the site of 2.99 g C m⁻² yr⁻¹. For CO₂, the vegetated ditches tended to act as CO₂ sinks during summer and CO₂ sources during winter. Overall, the ditch was estimated to be a net CO₂ source of 116 g C m⁻² yr⁻¹, giving an area-adjusted emission for the site of 3.1 g C m⁻² yr⁻¹.

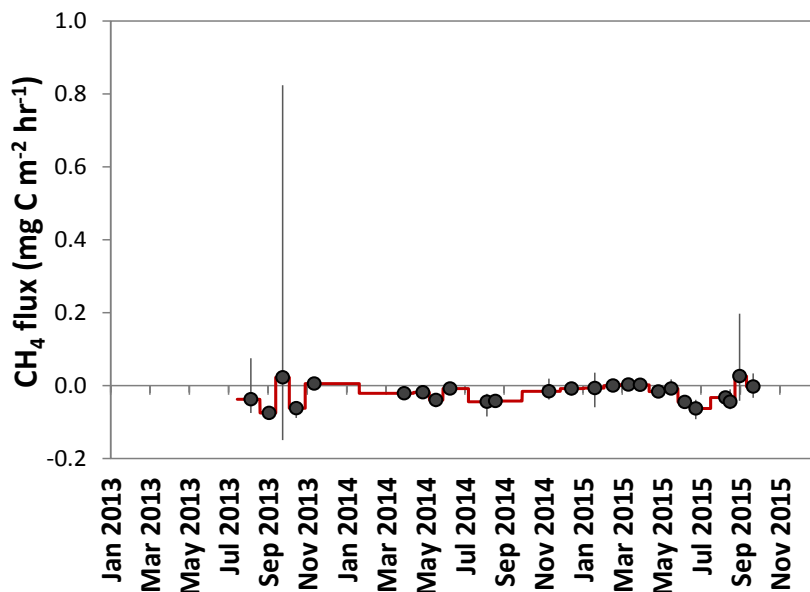


Figure 2.4.10. Measured CH₄ fluxes from the field at SL-EG. Points show mean observations on each measurement date, and error bars show range of measured values on that date. Red lines show interpolated fluxes.

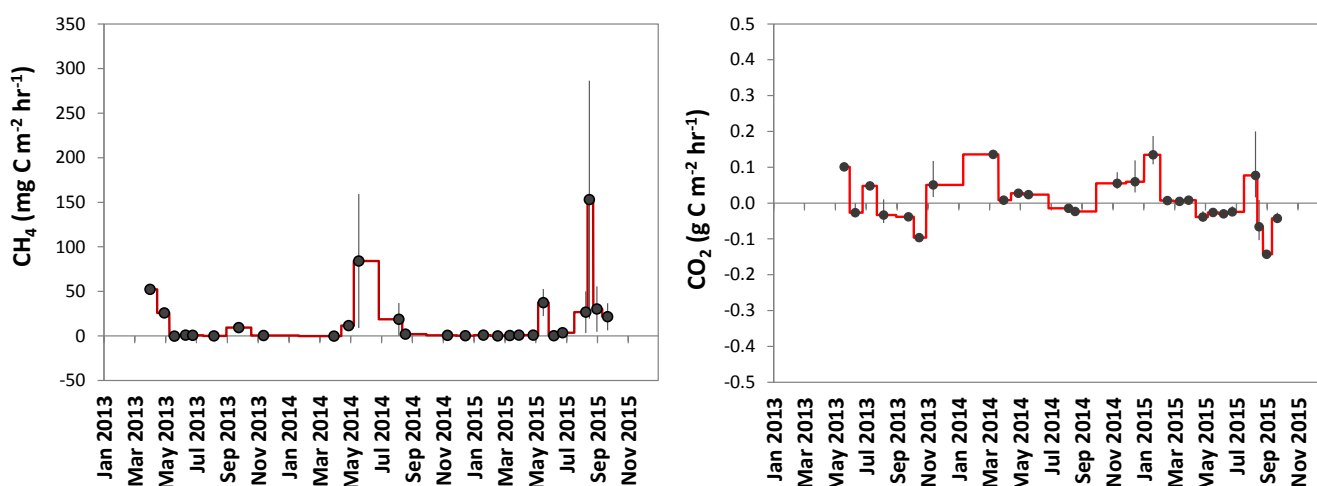


Figure 2.4.11 Observed CH₄ and CO₂ fluxes measured in ditches at SL-EG. Red line shows interpolated fluxes.

2.4.1.4. Aquatic carbon fluxes

Mean DOC and DIC concentration data from SL-EG are shown in Figure 2.4.12. With the exception of a few sampling dates in the early part of the measurement period, DIC was fairly stable at around 40-70 mg l⁻¹. Mean pH was high, at 7.59. In contrast to DIC, DOC concentrations were highly variable, ranging from near-zero values (typically in mid-summer) to peaks of 40-70 mg l⁻¹ in winter and spring. Minimum values coincided with periods when ditch levels were higher than water tables, i.e. when low-DOC water from the river was flowing through the ditch network and into the peat, whereas maximum values were observed during winter periods when the water-table was higher than the ditch level, water was flowing out of the peatland, and ditch water was therefore largely composed of peat leachate. This seasonal pattern is markedly different to that observed at most undisturbed peatlands, where DOC concentrations peak in mid-summer, and highlights the highly seasonal nature of water (and hence aquatic carbon) export from the Somerset Levels peatlands.

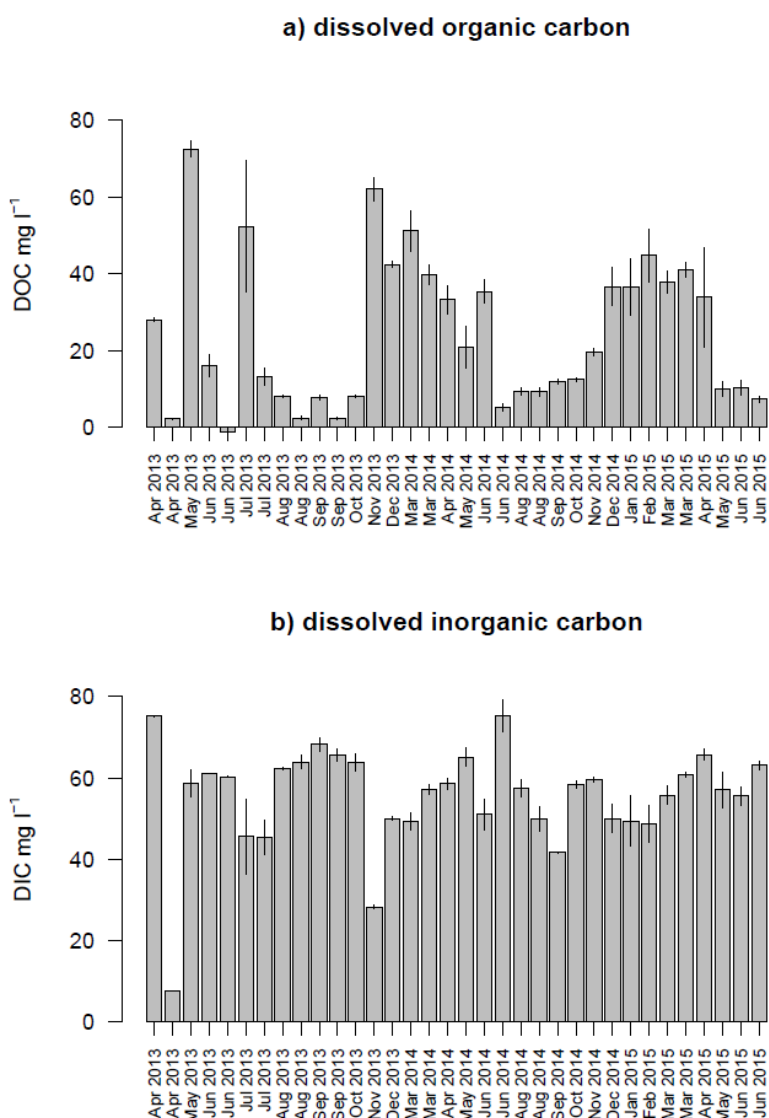


Figure 2.4.12. Mean and standard error of ditch dissolved organic and inorganic carbon concentrations, SL-EG. Note that x axis simply records sampling dates and is therefore not a true time axis.

As a result of the bi-directional seasonal flow regime at SL-EG, carbon export from the site was limited to the wet winter months (Figure 2.4.13). The estimated annual mean DOC flux from the site was 10.2 g C m⁻² yr⁻¹, with a slightly higher DIC flux of 15.4 g C m⁻² yr⁻¹. The estimated dissolved CO₂ flux was low (0.8 g C m⁻² yr⁻¹), and fluxes of POC (0.3 g C m⁻² yr⁻¹) and dissolved CH₄ (0.03 g C m⁻² yr⁻¹) were negligible.

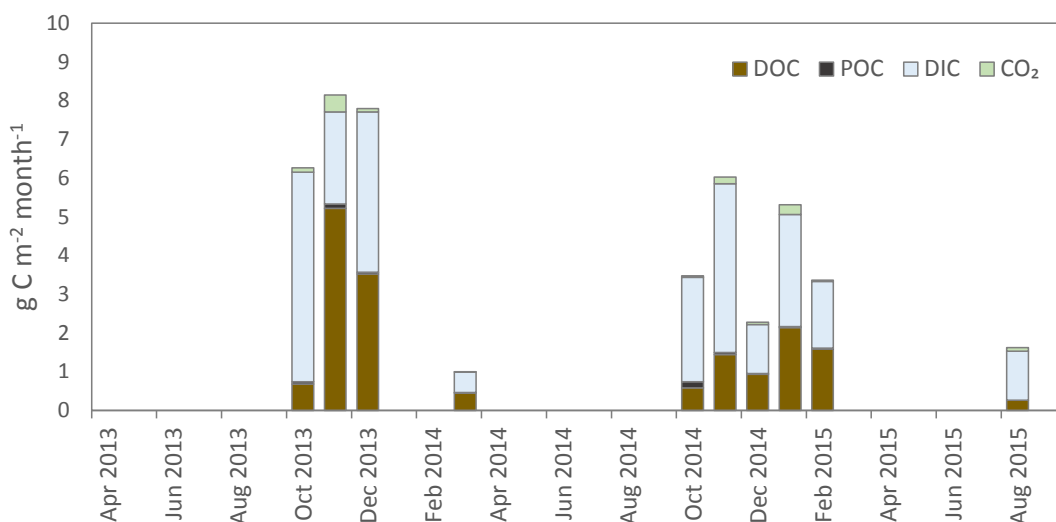


Figure 2.4.13. Estimated monthly aquatic carbon fluxes, SL-EG. Data are missing for January-February 2014 (when the site was flooded and therefore inaccessible); all other zero values indicate no flux.

2.4.1. Somerset Levels – Tadham Moor intensive grassland (SL-IG)

The intensive grassland site is located around 2 km southeast of the SL-EG site, in a small area of improved farmland adjacent to a road that crosses the central part of the Brue Valley deep peat. The site is managed as an annually reseeded meadow for cattle production, involving annual applications of inorganic fertilisers, silage cutting and more intensive grazing than at SL-EG. The study field was classified as a MG7a *Lolium perenne-Trifolium repens* (rye-grass and clover) ley (see Appendix 3). Some areas of bare ground were observed during vegetation surveys, allowing pioneer species to establish. The field is bordered by ditches, with a shallower ‘terminal’ ditch running into the middle of the field to enhance drainage having been dug shortly before the start of the project. In 2015, the western part of the field was turned over to maize production, which necessitated the transfer of some gas flux measurement collars to the eastern part of the field which remained under grassland.

The characteristics of peat cores collected from SL-IG (Figure 2.4.14) are very similar to those from SL-EG (Figure 2.4.4). Surprisingly however, SL-IG has slightly lower bulk density and mineral content in the upper 50 cm, despite being more intensively cultivated. The C/N ratio of peat at the surface is lower than at SL-EG, however, which may be a result of fertiliser addition, and the peat pH is markedly lower (4.5 vs 5.6). Both of the cores collected at SL-IG show a sharp transition to mineral soil at 170 cm depth.

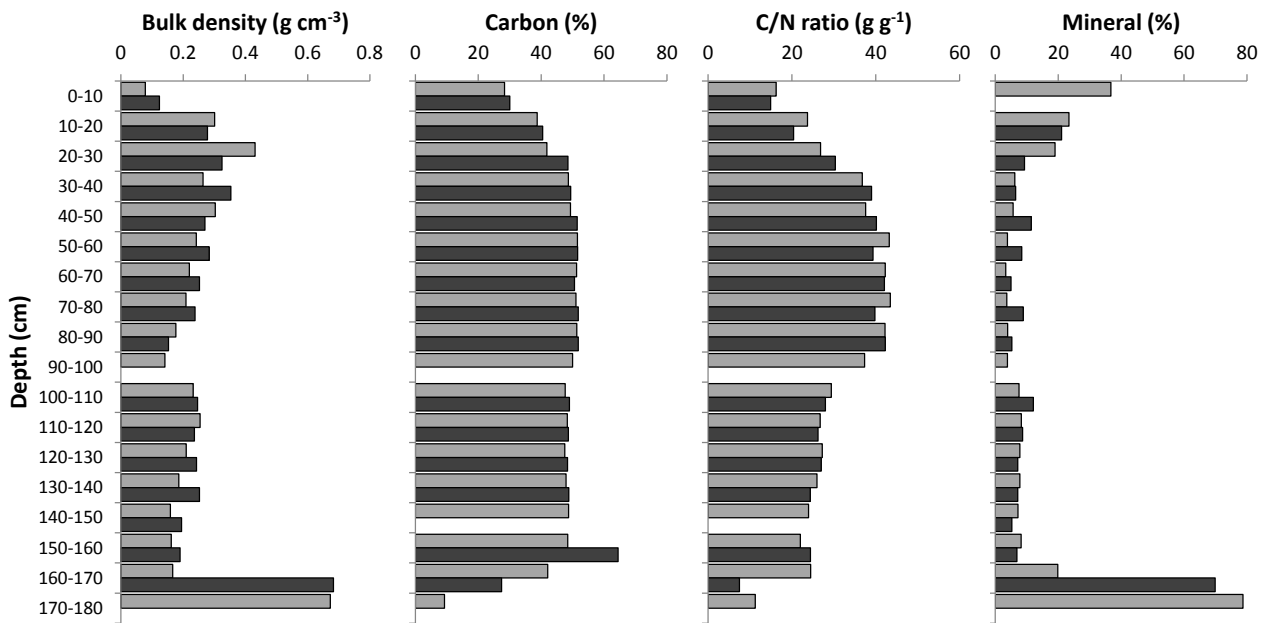


Figure 2.4.14. Peat core data, SL-IG. Data from two cores analysed from the site are represented by dark and light grey-shaded bars.

2.4.2.1. Hydrology

Water-table data from SL-IG (Figure 2.4.15) show very similar temporal patterns to those of the nearby SL-EG, reflecting the hydrological connection of the two sites within the same river basin management district. As at SL-EG, the water table was higher than perimeter ditch levels during winter, and lower during summer. Mean water levels nevertheless did differ somewhat between the two sites, primarily during winter when water tables remained around 10-20 cm below the surface in the winters of 2012-13 and 2014-15, and only briefly rose above the surface during the floods of early 2014. The rates and magnitude of water-table drawdown during summer were very similar for the two sites in all years, with slightly greater drawdown in 2013. The water level in the in-field ditch at SL-IG was typically quite low, with similar baseline levels throughout the year, and limited response to rain events. Instead the in-field ditch levels appear to be strongly influenced by ET, with significant drawdowns occurring in both 2013 and 2014. Generally higher levels in 2015 may have partly been due to the revegetation of this ditch that took place during the study period.

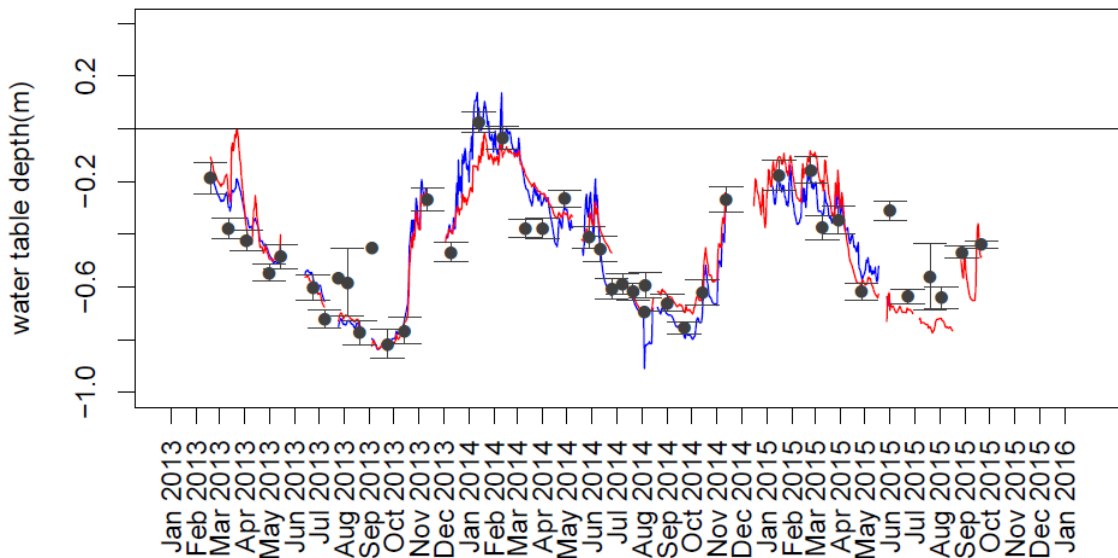


Figure 2.4.15. Continuously monitored and manually measured water-table data, SL-IG.

The modelled water balance for SL-IG was identical to that for SL-EG (shown in Figure 2.4.6), as it was based on the same P_{net} and ET data (both measured at SL-EG). It is possible that the shorter average sward height in the improved grass of SL-IG has a slightly lower associated ET, and that somewhat lower ditch levels allowed greater discharge, but these effects were not quantified.

2.4.2.2. Static chamber gas fluxes

Measured and modelled CO_2 fluxes are shown in Figure 2.4.16. Despite the agricultural management of the site the fit between modelled and observed GPP and ER appears generally good. The estimate of annual ER derived from the chamber data ($2718 \text{ g C m}^{-2} \text{ yr}^{-1}$) is very similar to the mean values obtained from the flux tower at SL-EG, but higher than the estimate derived from chamber measurements at that site. The estimated annual GPP for SL-IG was $1567 \text{ g C m}^{-2} \text{ yr}^{-1}$ which, if correct, would imply that the site is generating very high rates of CO_2 emission ($\text{NEE } +1151 \text{ g C m}^{-2} \text{ yr}^{-1}$), with almost continuous CO_2 loss throughout the year (lower panel of Figure 2.4.16). However, the estimation of GPP at the managed grassland SL-IG is subject to the same problems as at SL-EG, due to inevitable changes in the relationship between environmental variables and photosynthesis associated with annual planting and harvesting of the grass crop, and for this reason the NEE value obtained by via method is considered unreliable.

In order to try to obtain a more realistic estimation of the NEE of SL-IG, we took data from the flux tower at SL-EG, and applied an adjustment factor based on the two sets of parallel chamber measurements from the adjacent sites. Based on the fitted chamber models, annual GPP and ER were both 1.33 times higher at SL-IG than at SL-EG. Applying these ratios to the annual GPP and ER values for SL-EG obtained from the flux tower would suggest that the 'true' values for SL-IG are 3474 and 3600 $\text{g C m}^{-2} \text{ yr}^{-1}$ respectively, giving an estimated NEE of $+126 \text{ g C m}^{-2} \text{ yr}^{-1}$. Clearly there is a high uncertainty associated with this estimate.

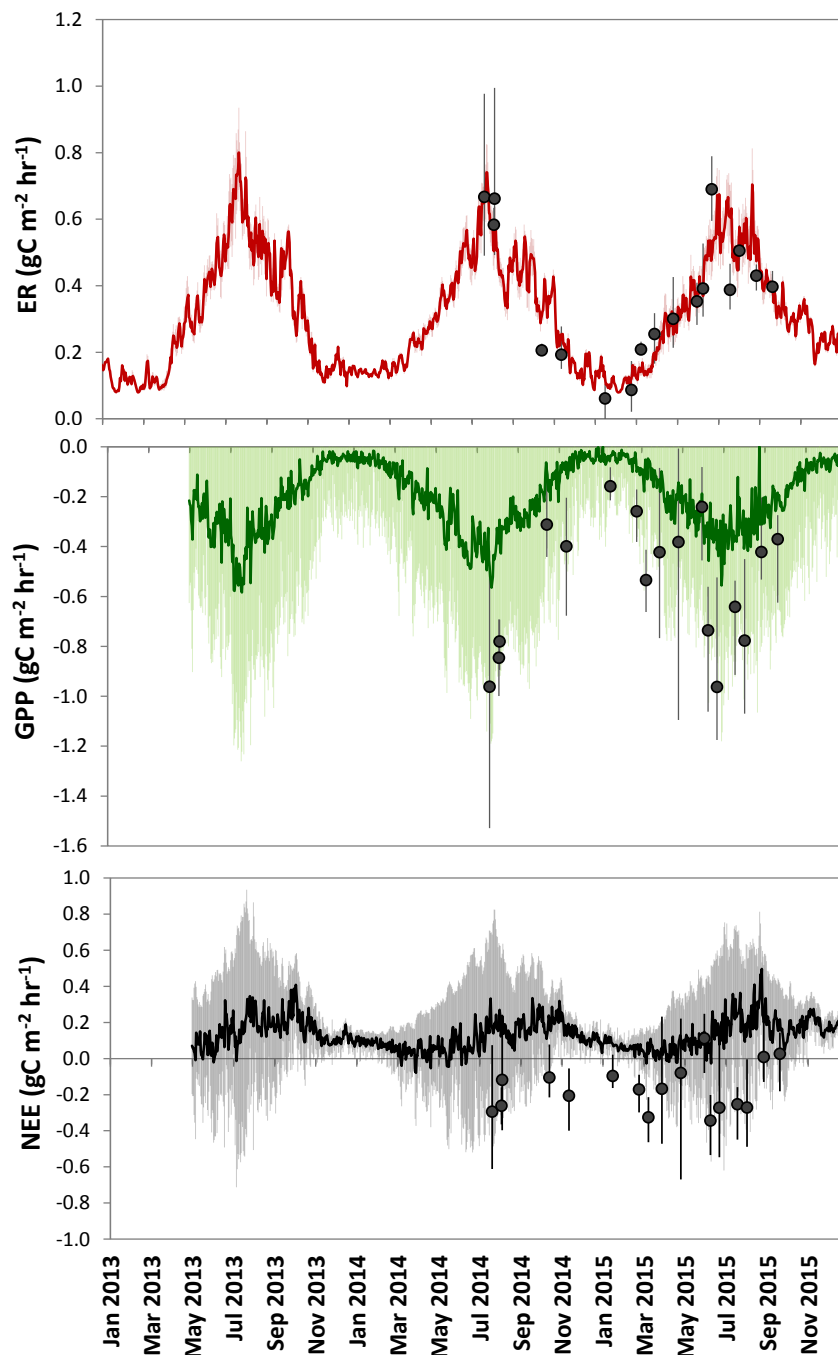


Figure 2.4.16. Modelled and observed CO_2 fluxes (ER = ecosystem respiration, GPP = gross primary productivity, NEE = net ecosystem exchange) based on static chamber measurements in the field at SL-IG. Continuous lines show modelled daily mean fluxes, shading shows modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date. Note that static chamber measurements were generally taken around the middle of the day, so tend to be representative of peak (rather than daily mean) rates of photosynthesis and respiration.

Methane fluxes for the field at SL-IG (Figure 2.4.17) were very small and predominantly negative. The estimated annual CH_4 flux across the land surface was $-0.15 \text{ g C m}^{-2} \text{ yr}^{-1}$. Ditch CH_4 fluxes were highly variable (Figure 2.4.18), with generally low values interspersed with occasional very large emissions during the growing season. The annual mean CH_4 flux per unit ditch area was $37 \text{ g C m}^{-2} \text{ yr}^{-1}$. We estimated the fractional ditch area for the field as 0.016, giving an area-weighted ditch CH_4 emission for the site as a whole of $0.59 \text{ g C m}^{-2} \text{ yr}^{-1}$. Measured ditch CO_2 fluxes were more often negative than positive, indicating net uptake by the vegetation within the ditch. The weighted mean CO_2 flux per unit ditch area was $-420 \text{ g C m}^{-2} \text{ yr}^{-1}$, implying an area-weighted ditch CO_2 flux for the site of $-6.7 \text{ g C m}^{-2} \text{ yr}^{-1}$. However since all ditch fluxes were measured during the day, this will clearly be an over-estimate of the true CO_2 uptake by the ditch. Since the

data were insufficient to produce a full light-response model of ditch CO₂ fluxes, we simply assumed that the annual ditch NEE is half of the measured daytime value. This is clearly a simplification, however as the resulting flux is small (-3.4 g C m⁻² yr⁻¹) the impact of this assumption on the overall C and GHG balance of the site is negligible.

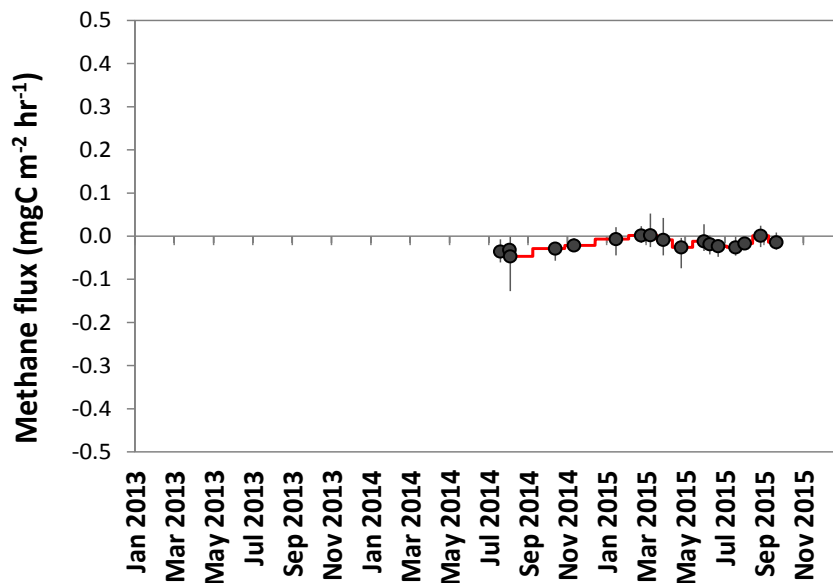


Figure 2.4.17. Measured CH₄ fluxes for the field at SL-IG. Points show mean observations on each measurement date, and error bars show range of measured values on that date. Red lines show interpolated fluxes.

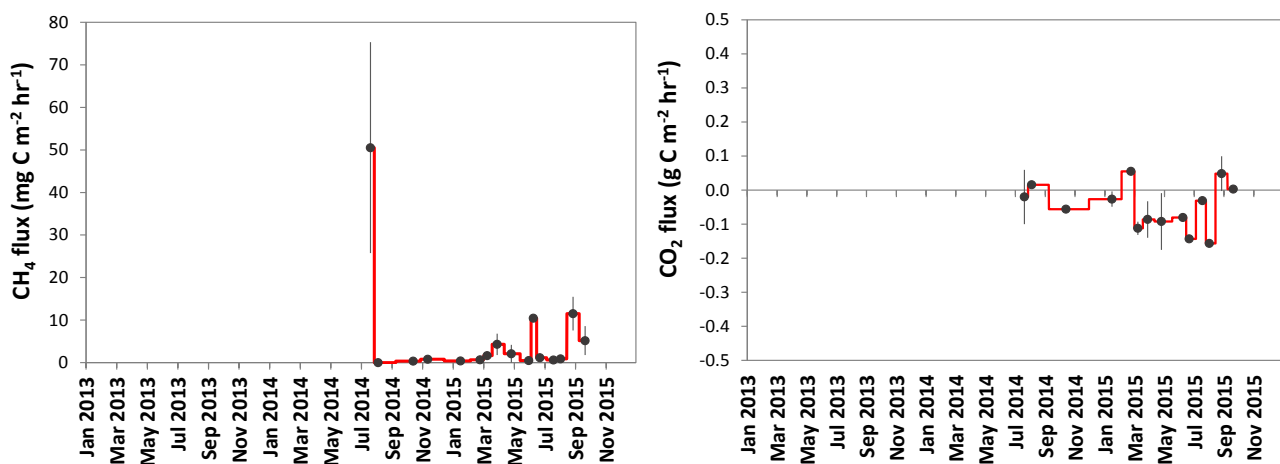


Figure 2.4.18. Observed CH₄ and CO₂ fluxes measured in ditches at SL-IG. Red line shows interpolated fluxes

2.4.2.3. Aquatic carbon fluxes

Concentrations of DOC and DIC in ditches at SL-IG (Figure 2.4.19) bear some similarities to those at SL-EG, but with some notable differences. In particular, DOC concentrations at SL-EG were markedly higher on almost all sampling occasions. Concentrations rarely fell below 20 mg l⁻¹, and frequently exceeded 80 mg l⁻¹. For dates on which DOC samples were obtained from both sites, mean concentrations at SL-IG were 2.3 times higher than those at SL-EG. The largest concentrations occurred in the spring of 2014, during the warm conditions that followed the floods in February. It is thus possible that the floods caused increased mobilisation of DOC from the intensive grassland field, however the same response was not evident at the extensive grassland SL-EG. The higher minimum DOC values during summer probably indicate a smaller input of river water to the ditches around SL-IG, reflecting the more conventional drainage management of SL-IG versus the higher water level requirements of conservation management at SL-EG. DIC concentrations at the

two sites were similar, and between-year differences were small. However mean pH was lower at SL-IG (6.92 versus 7.59), which can in part be attributed to the higher DOC concentrations.

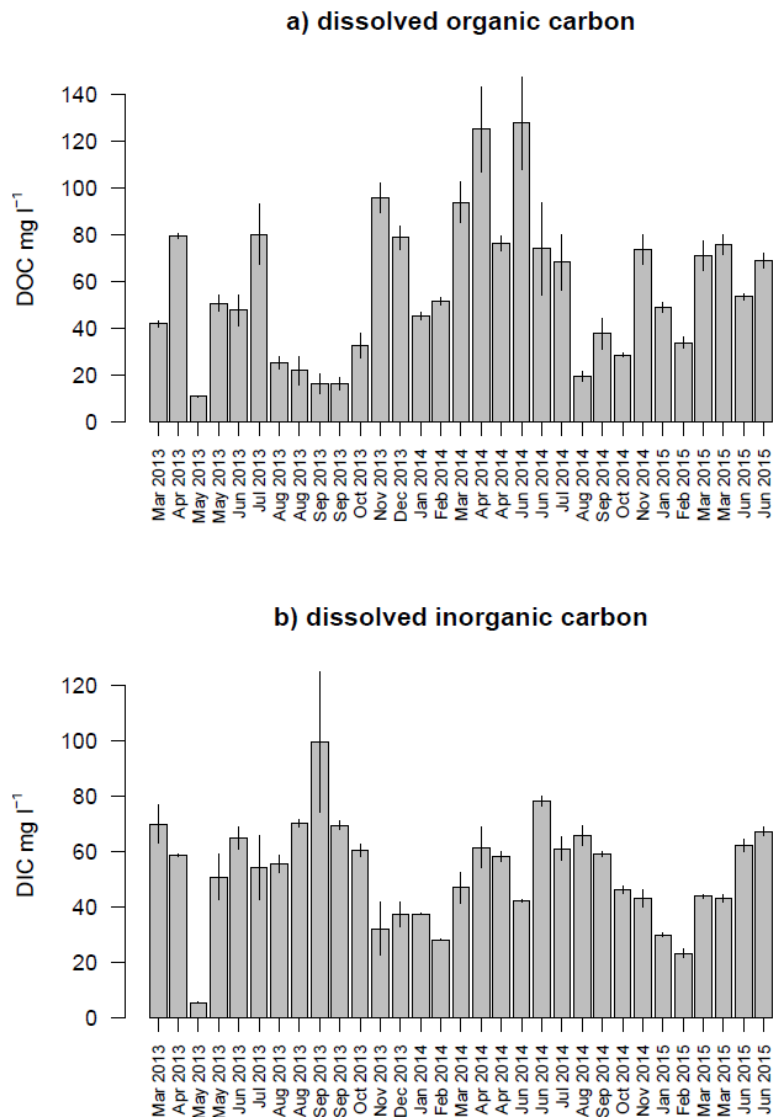


Figure 2.4.19. Mean and standard error of ditch dissolved organic and inorganic carbon concentrations, SL-IG. Note that x axis simply records sampling dates and is therefore not a true time axis.

Monthly aquatic carbon fluxes are shown in Figure 2.4.20. Given that modelled water fluxes for the SL-IG and SL-EG were identical, differences in carbon flux between sites are associated with differences in concentration, as well as differences in missing months between sites (notably during the January-February 2014 floods when SL-IG remained accessible but SL-EG was not). The higher mean DOC concentrations at SL-IG are reflected in higher fluxes, with an estimated mean annual export of 21.8 g C m⁻² yr⁻¹ for the measurement period. The estimated annual DIC flux was 12.2 g C m⁻² yr⁻¹, and the POC flux was 1.2 g C m⁻² yr⁻¹. The annual mean dissolved CO₂ flux was 7.1 g C m⁻² yr⁻¹, more than double that at SL-EG, but dissolved CH₄ was lower (0.01 g C m⁻² yr⁻¹).

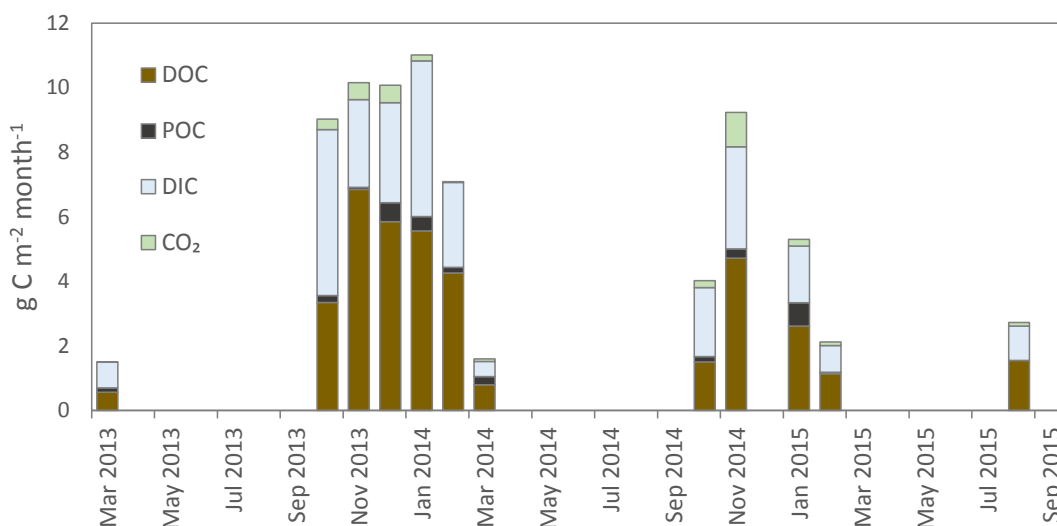


Figure 2.4.20. Estimated monthly aquatic carbon fluxes, SL-IG. Data missing for December 2014; all other zero values indicate no flux.

2.5. Norfolk Broads

The Norfolk Broads support a large proportion of the UK's total area of fen and reed bed (39 km²), much of which is floodplain fen of a high carbon density (exceeding 2000 t/ha and 10m depth close to rivers) associated with the five main Norfolk river systems (Figure 2.5.1). Most of the remaining deep peat is located in the lower valleys and adjacent to the coast, with shallower 'wasted' peats prevalent in the more intensively cultivated upper floodplains. The 'broads' (shallow areas of open water characteristic of the region) were mostly formed by flooding of pits created during peat digging (post-879 AD). Rapid infilling of some broads occurred between 1800 and 1950 AD, owing to high nutrient loading. Large areas of fen deemed unsuitable for drainage and land-use change still exist, while other areas invaded by willow and alder scrub have been restored in the past 60 years. Water levels are managed using embankments, ditch networks and water control structures, including pumps. The peat surface may be inundated through the winter/spring, but evapotranspiration leads to a gradual fall in water-tables over the summer/autumn. Over-bank flooding from adjacent rivers may occur during periods of high precipitation.

As noted earlier, the two Norfolk Broads sites were 'secondary' sites in the network, which meant that it was not possible to make a full suite of measurements for the full project period. However, with the support of a PhD student, Kieran Stanley, it was possible to make an intensive set of static chamber flux and water chemistry measurements, supported by continuous water-table monitoring, spanning both the 2012 and 2013 growing seasons. Peat cores were also collected from the site and analysed according to the methodology used at other project sites.

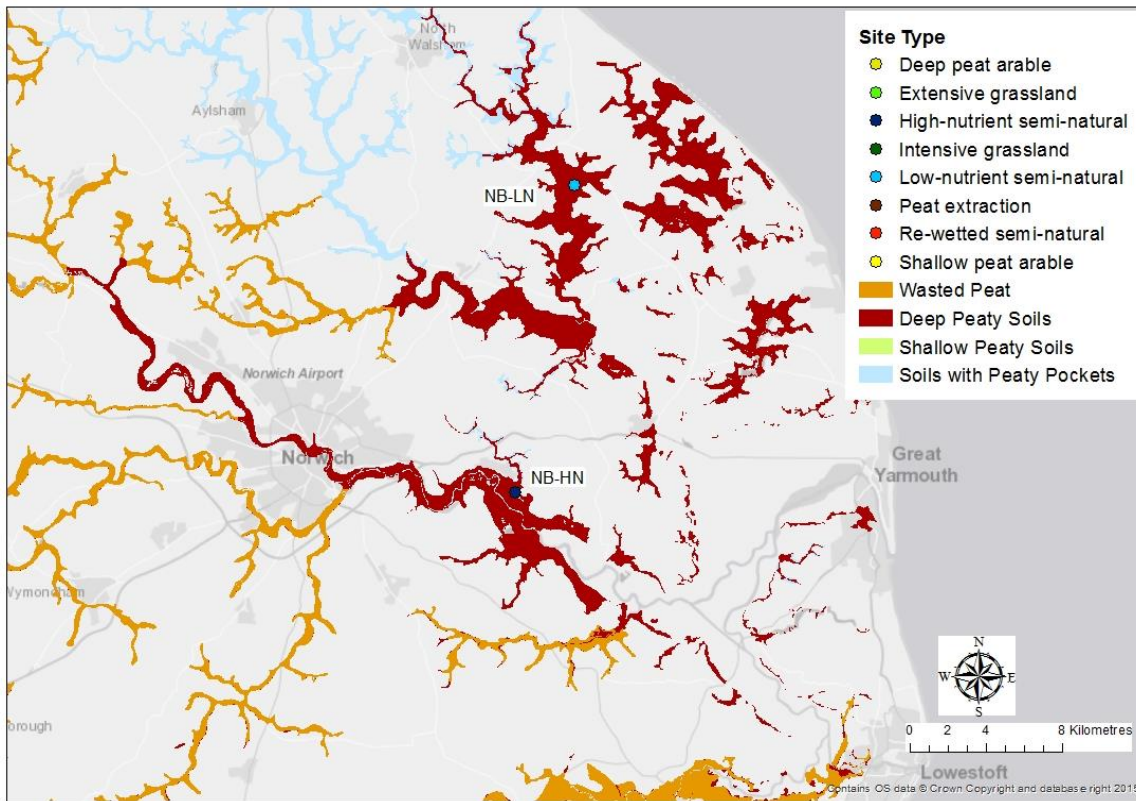


Figure 2.5.1. Norfolk Broads study area.

Meteorological measurements made at both of the Norfolk Broad study sites are shown in Figure 2.5.2. Incomplete data coverage prevents calculation of annual mean values. Monthly mean air temperatures (middle panels) were almost identical at the two sites, with maxima of around 17 °C in July-August of both years, and minima of around 2.5 °C in March 2013 (note no data for Jan-Mar 2012). The largest between-year differences in monthly average air temperatures were observed for May (warmest in 2012) and July (warmest in 2013).

Monthly precipitation sums (lower panels) showed high seasonal and between year variability, and slightly greater differences between sites. The wettest month of the measurement period was June 2012 (100 mm and 132 mm recorded at NB-HN and NB-LN respectively). All months with overlapping data coverage were wetter in 2012 than in 2013, with the exception of May. The driest months were May in 2012 and July in 2013, which has exceptionally low rainfall (< 7 mm) at both sites.

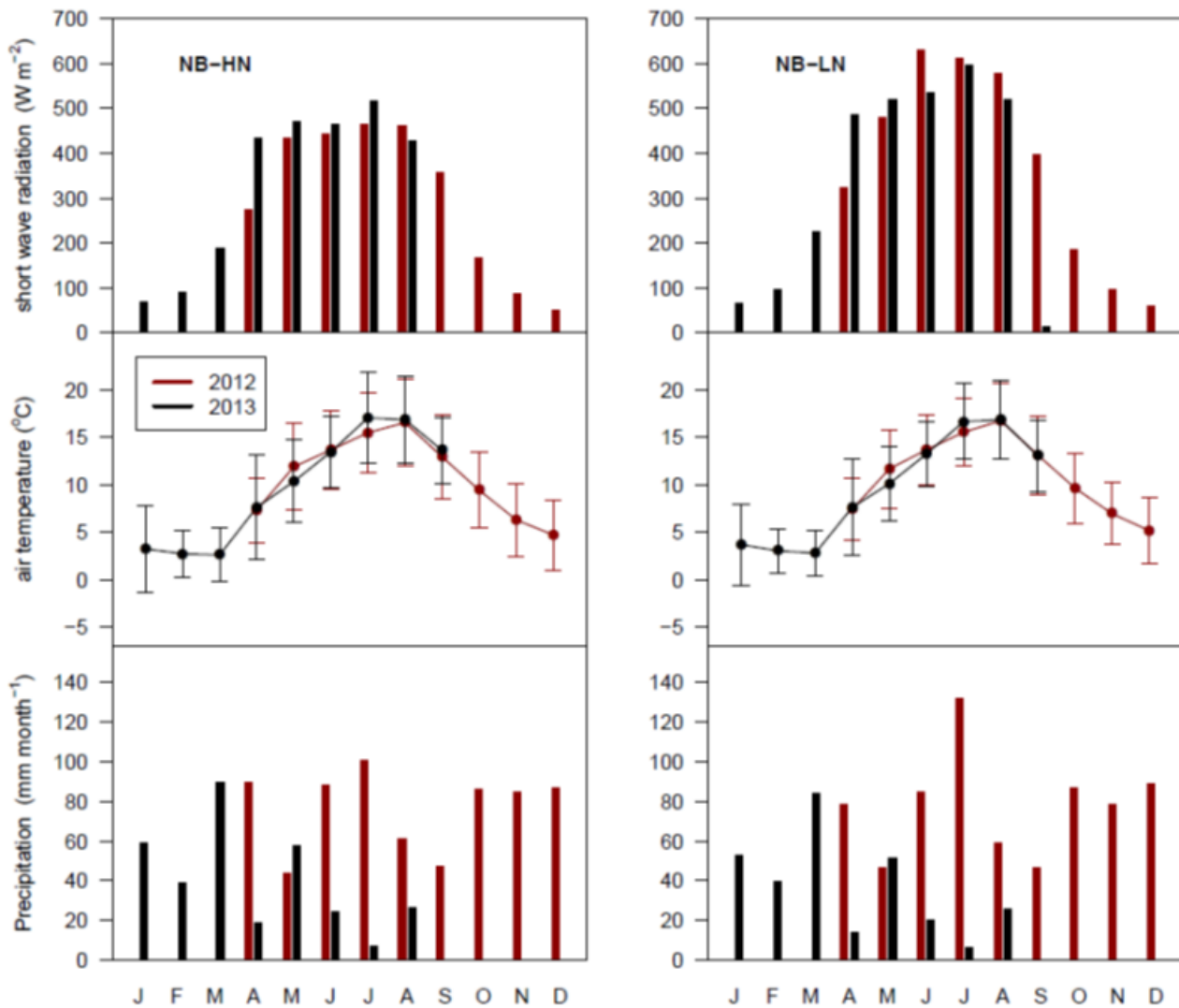


Figure 2.5.2. Mean monthly short-wave radiation (top), air temperature (middle) and total monthly precipitation (lower) for the NB-HN (Strumpshaw Fen) and NB-LN (Sutton Fen) sites. Error bars on the temperature plot show one standard deviation of the mean.

2.5.1. Norfolk Broads – Sutton low-nutrient fen (NB-LN)

The 192 ha Sutton Fen reserve, located within the Ant catchment in the Northern Broads, has been under RSPB ownership and management since 2006. In the preceding decade, willow scrub was removed to reinstate fen. Prior to this very little active management of the fen occurred, apart from some commercial saw sedge (*Cladium mariscus*) cutting. Prior to the two World Wars most of the site would have been cut to harvest saw sedge or for marsh hay. Present-day vegetation is dominated by a mix of *Phragmites australis*, *Cladium mariscus*, other sedge species and brown mosses. Reed harvesting took place two years before the start of measurements, and no further harvesting was taken during the measurement period. Water levels are managed to maintain wet conditions between November and March, in some cases leading to inundation of the site. Between April and July water levels drop naturally to a point where the water table is around 30 cm below the fen surface to facilitate harvesting of sedge and reed without damaging the peat. Water chemistry is slightly basic, and nutrient levels are among the lowest in the Broads.

The Norfolk Broads sites, along with Thorne Moors (below) were identified as ‘secondary’ sites at the outset of the project, which were subject to a shorter and less comprehensive measurement programme. The two Norfolk sites were the first to be established during the project, and were originally funded for one year of measurements. This was extended to include a second growing season, and additional measurements

including collection of peat cores, through a combination of in-kind support from QMUL via a PhD student project, and additional funding from Defra to support fieldwork expenses. The field programme was completed in September 2013.

Three peat cores collected from the site, to a depth of 3 m, show consistently low peat bulk density and high carbon content (Figure 2.5.3), consistent with the low-intensity historical use of the site described above. Fairly low C/N ratios reflect the role of (low C/N) sedge species in peat formation, whilst the moderately high mineral content may be attributable to overbank flooding from the river during high flow periods.

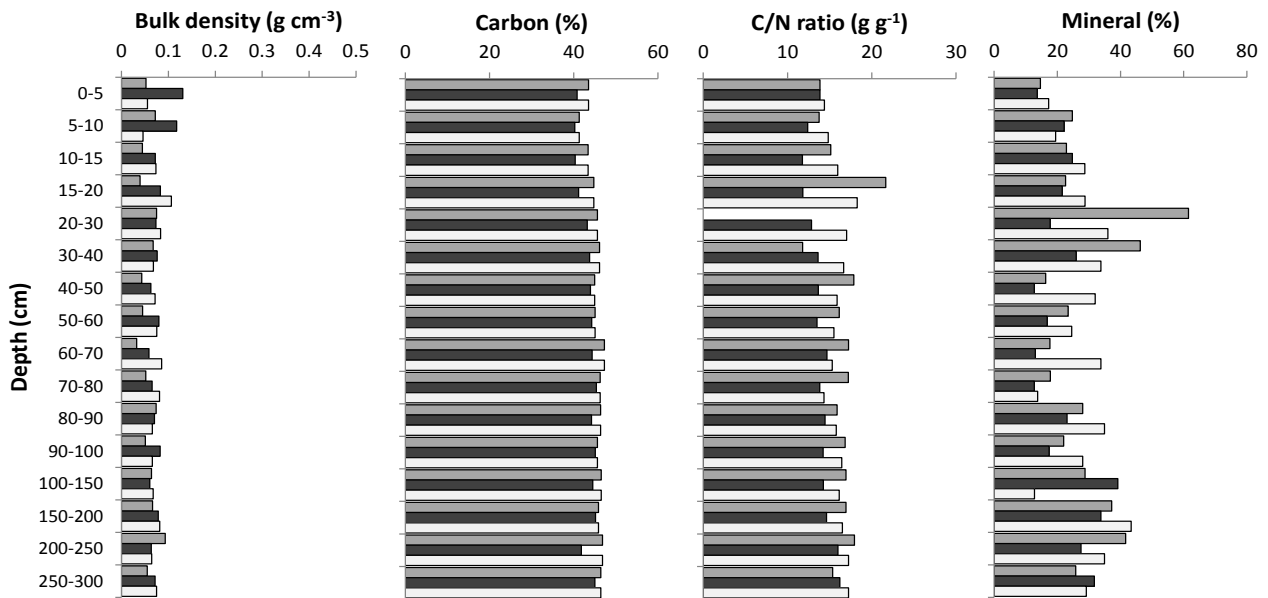


Figure 2.5.3. Peat core data, NB-LN. Data are shown from three cores collected at the site (dark, mid and light grey shading)

2.5.1.1. Hydrology

Full water balances were not generated for the Norfolk Broad sites. However, continuous and manually-recorded water tables were measured at both sites, and provide information on the hydrological characteristics of the sites. The data for NB-LN (Figure 2.5.4) show that the site was inundated for much of the 18-month measurement period, with only a brief period of water-table drawdown in the late summer of 2012, and more protracted and substantial drawdown during the summer of 2013 (coinciding with the more severe water table drawdown in the nearby EF sites, notably EF-LN). During the intervening winter of 2012-13, water levels were frequently 20 cm above the surface, and showed a high degree of short-term variability indicative of tidal variation, suggesting that water levels in the fen had become hydrologically connected to the adjacent river. Although a site-specific estimate of water flux from the site could not be calculated, data from the nearby EA gauging station on the River Ant at Honing Lock (<http://nrfa.ceh.ac.uk/data/station/info/34008>) suggest a mean areal discharge for the broader catchment of 214 mm yr⁻¹.

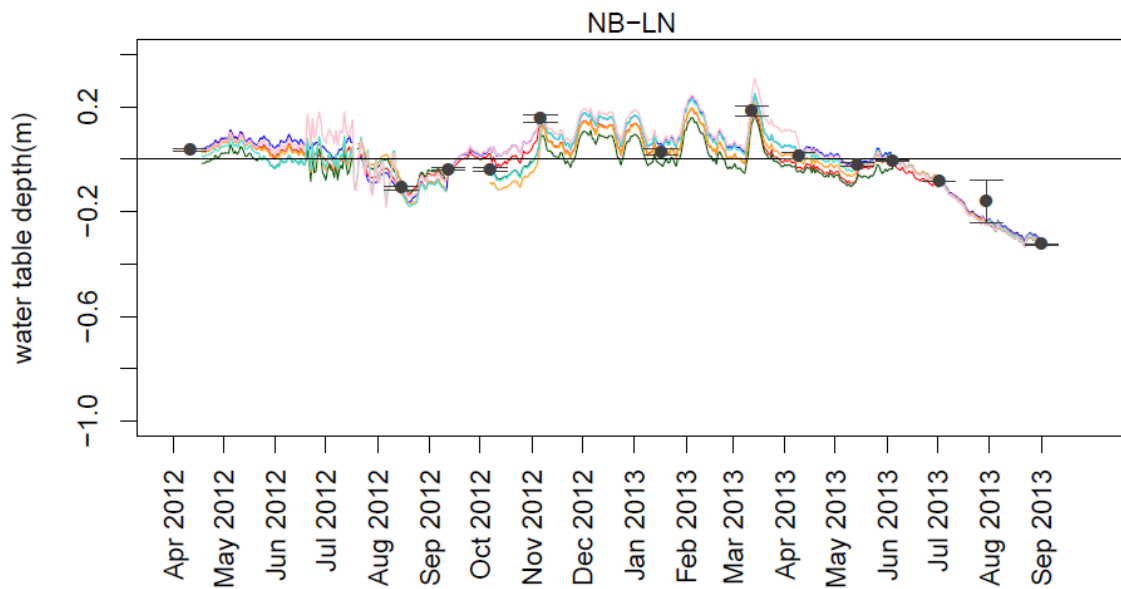


Figure 2.5.4. Manually measured water-table data for NB-LN

2.5.1.2. Static chamber gas fluxes

Static chamber gas flux measurements were made at NB-LN from June 2012 to September 2013. Modelled CO₂ fluxes from the *Phragmites*-dominated vegetation show a reasonable fit to measurements (Figure 2.5.5). Over the 15 month study period, which included two growing seasons, mean annually-adjusted GPP was 712 g C m⁻² yr⁻¹, whilst RE was 722 g C m⁻² yr⁻¹. This suggests that, despite most daytime NEE measurements showing net CO₂ uptake, the site was close to being balance in terms of its annual CO₂ exchange, with an estimated mean annual NEE of +10 g C m⁻² yr⁻¹.

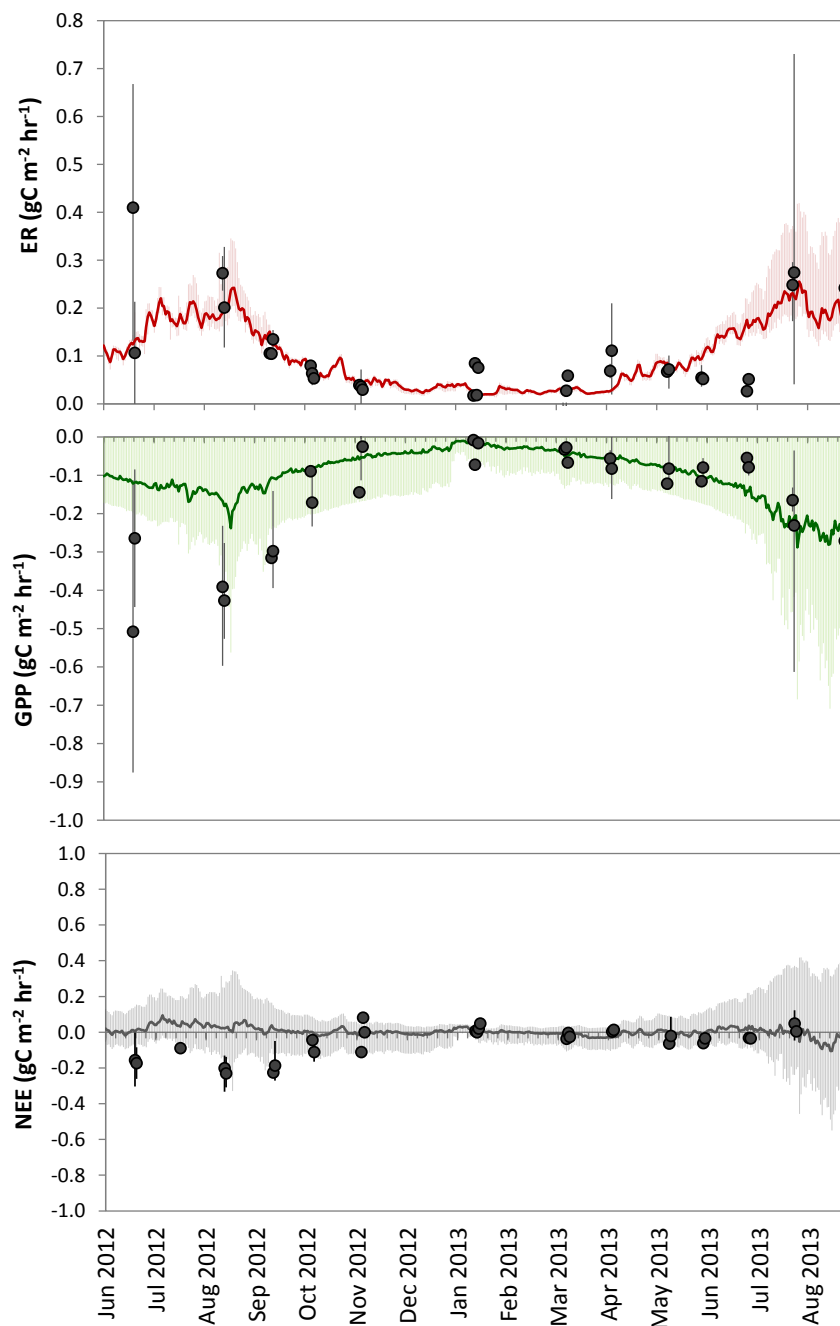


Figure 2.5.5. Modelled and observed CO_2 fluxes (ER = ecosystem respiration, GPP = gross primary productivity, NEE = net ecosystem exchange) based on static chamber measurements at NB-LN. Continuous lines show modelled daily mean fluxes, shading shows modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date. Note that static chamber measurements were generally taken around the middle of the day, so tend to be representative of peak (rather than daily mean) rates of photosynthesis and respiration.

In contrast to the majority of other sites, it was possible to fit an empirical model of CH_4 fluxes at NB-LN (Figure 2.5.6). Modelled fluxes were largely positive, and fairly large during the growing seasons, with higher estimated emissions during the wetter 2012 growing season compared to 2013. Fluxes were small but positive throughout the winter. Overall, the estimated annual mean CH_4 emission from NB-LN was high, at $15.1 \text{ g C m}^{-2} \text{ yr}^{-1}$.

Static chamber measurements from two ditches at NB-LN are shown in Figure 2.5.7. Measured CH_4 fluxes were high during both summer periods, with particularly large fluxes observed from both ditches in September 2013. Over the period of measurement, the estimated annual CH_4 emission per unit ditch area was $289 \text{ g C m}^{-2} \text{ yr}^{-1}$. The estimated proportion of ditch surface across the site, $\text{Frac}_{\text{ditch}}$, is 0.021, giving an

area-adjusted ditch CH_4 flux for the site of $6.0 \text{ g C m}^{-2} \text{ yr}^{-1}$. This adds almost 50% to the land surface CH_4 emission. Ditch CO_2 fluxes varied from positive to negative, with a small annual net emission over the study period of $60 \text{ g C m}^{-2} \text{ yr}^{-1}$ per unit ditch area, which equates to an emission of $1.2 \text{ g C m}^{-2} \text{ yr}^{-1}$ on a site-wide basis.

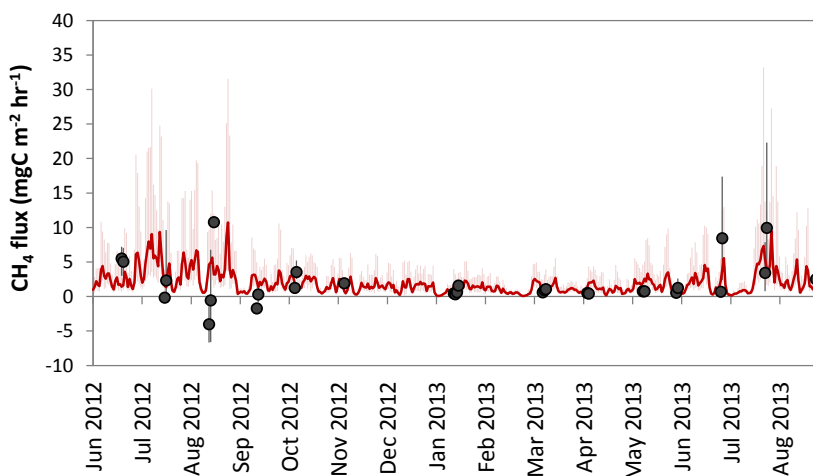


Figure 2.5.6. Modelled and observed CH_4 fluxes at NB-LN. Continuous lines show modelled daily mean fluxes, shading shows modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date.

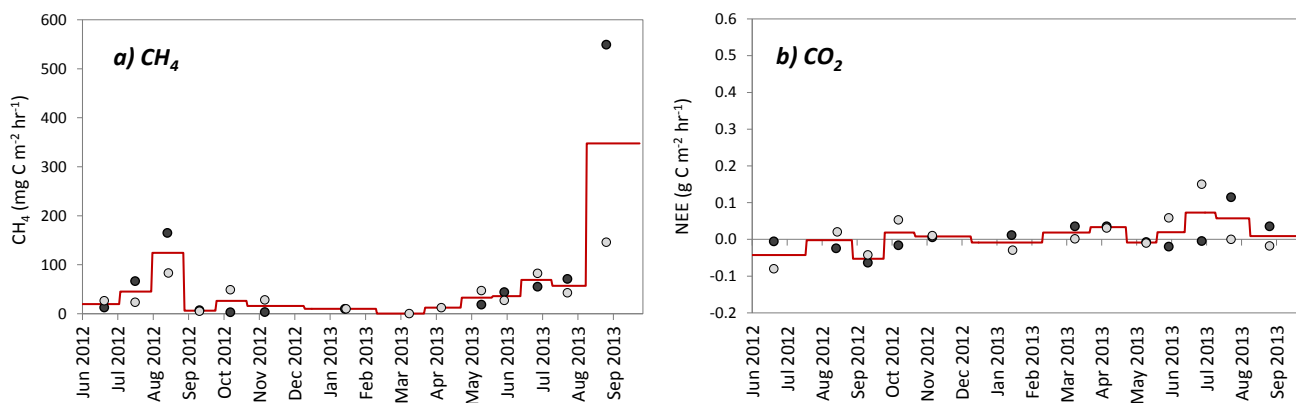


Figure 2.5.7. Observed CH_4 and CO_2 fluxes measured in ditches at NB-LN. Observations from two ditches are represented by grey and black circles, red line shows interpolated fluxes.

2.5.1.3. Aquatic carbon fluxes

Concentrations of DOC and DIC at NB-LN are shown in Figure 2.5.8. Apart from the first sampling date, DOC concentrations were consistently below 40 mg l^{-1} , with an estimated annual mean for the study period of 32 mg l^{-1} . DIC concentrations were highly variable, with high values between November and June, and lower values from June to September, coinciding with periods of lower water-table. The estimated annual mean concentration was 40 mg l^{-1} . While full aquatic carbon fluxes could not be calculated, multiplying mean concentrations by mean annual flows based on the EA gauging station gives indicative flux estimates of around $6.7 \text{ g C m}^{-2} \text{ yr}^{-1}$ for DOC, and $8.6 \text{ g C m}^{-2} \text{ yr}^{-1}$ for DIC. Given the highly discontinuous nature of water flux from sites in other regions, these estimates must be considered highly approximate, however it seems reasonable to infer that aquatic carbon losses from NB-LN are comparatively minor.

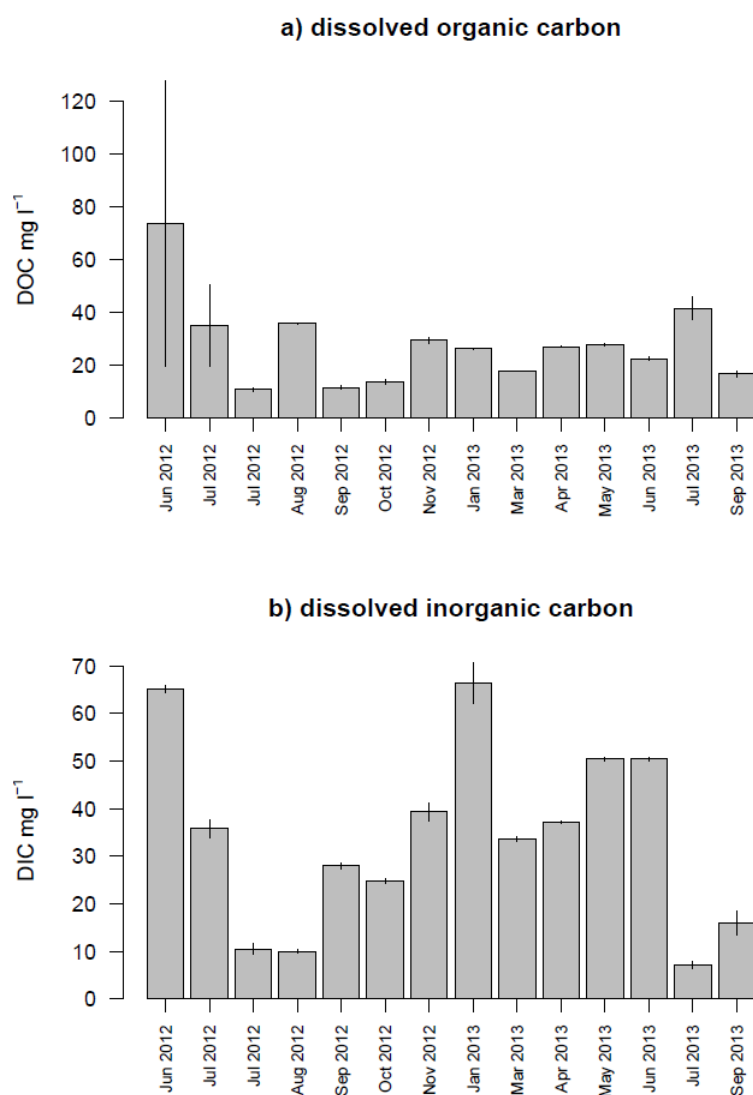


Figure 2.5.8. Mean and standard error of ditch dissolved organic and inorganic carbon concentrations, NB-LN. Note that x axis simply records sampling dates and is therefore not a true time axis.

2.5.2. Norfolk Broads – Strumpshaw high-nutrient fen (NB-HN)

Strumpshaw Fen is a similar size to Sutton Fen (197 ha) and is located adjacent to the River Yare downstream of Norwich. It has been managed by the RSPB since 1975. Since 1978, all free connections between the ditch network and the river have been dammed, and a control structure installed to facilitate management of water levels within the fen. Since 1987, groundwater abstractions from the underlying chalk aquifer have been compensated by pumping. The peat surface is inundated from late autumn/early winter until late spring/early summer, and water levels fall to about 45 cm depth below the surface in summer. Inputs of river water, which contain high levels of nitrogen and phosphorus, occur during over-bank flood events and when the control structure is opened. As a result, the upper layer of peat is considered to be nutrient-enriched (SurrIDGE et al., 2007). The vegetation over much of the fen is dominated by *Phragmites australis*. Since the RSPB took over management of the site, new areas of shallow open water have been created, and large areas of scrub have been cleared. As at NB-LN, harvesting took place two years before the start of measurements, with no harvesting during the measurement period.

The peat core data from NB-HN (Figure 2.5.9) show broadly similar characteristics to NB-LN, but with some higher bulk density layers near the surface, which are associated with lower carbon contents and higher mineral contents. This could reflect historically more intensive agricultural use of the site, and/or greater inputs of mineral material due to over-bank flooding. Compared to NB-LN, the top 50 cm of NB-HN have

higher bulk density (0.12 vs 0.07 g cm⁻³), lower carbon content (32% vs 44%), higher mineral content (42% vs 26%) and a slightly lower C/N ratio (13.4 vs 14.5 g g⁻¹).

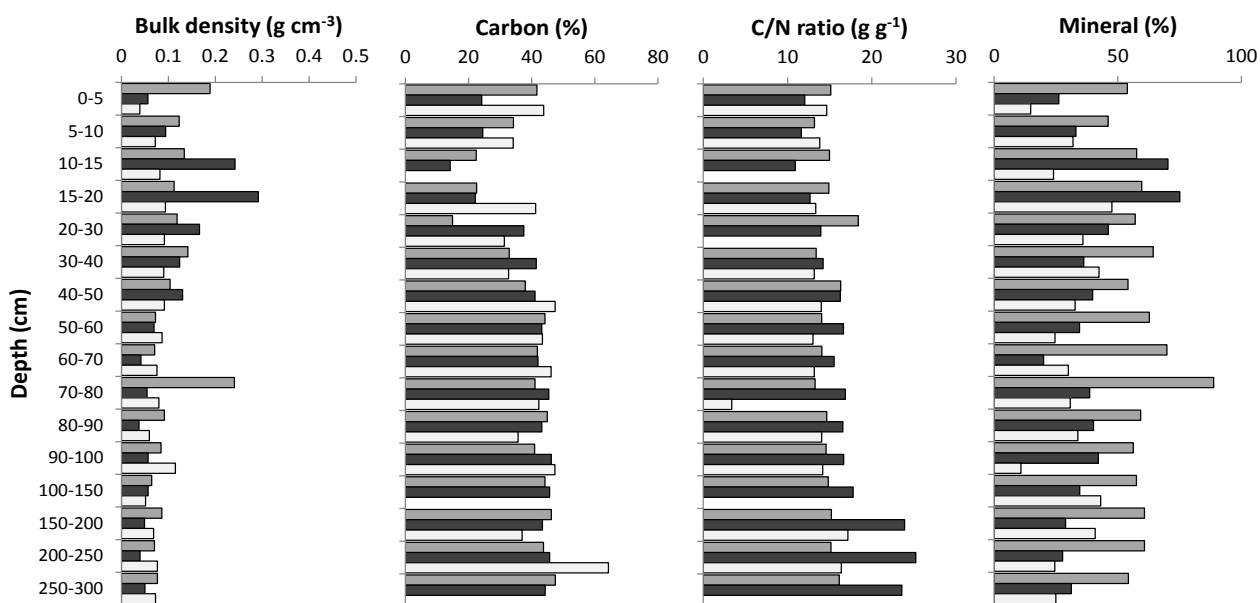


Figure 2.5.9. Peat core data, NB-HN. Data are shown from three cores collected at the site (dark, mid and light grey shading)

2.5.2.1. Hydrology

Water-table data for NB-HN (Figure 2.5.10) show a similar seasonal and between-year pattern to NB-LN, with very little water-table drawdown in the summer of 2012, and somewhat greater drawdown in 2013. The winter 2012-13 period was characterised by continuous flooding of the site, with two short periods of very high water levels in early 2013 when river water flooded the fen. The nearest available EA gauging station (on the Yare at Colney, upstream of Norwich; <http://nrfa.ceh.ac.uk/data/station/info/34001>) suggest that average runoff for the wider area is around 200 mm yr⁻¹.

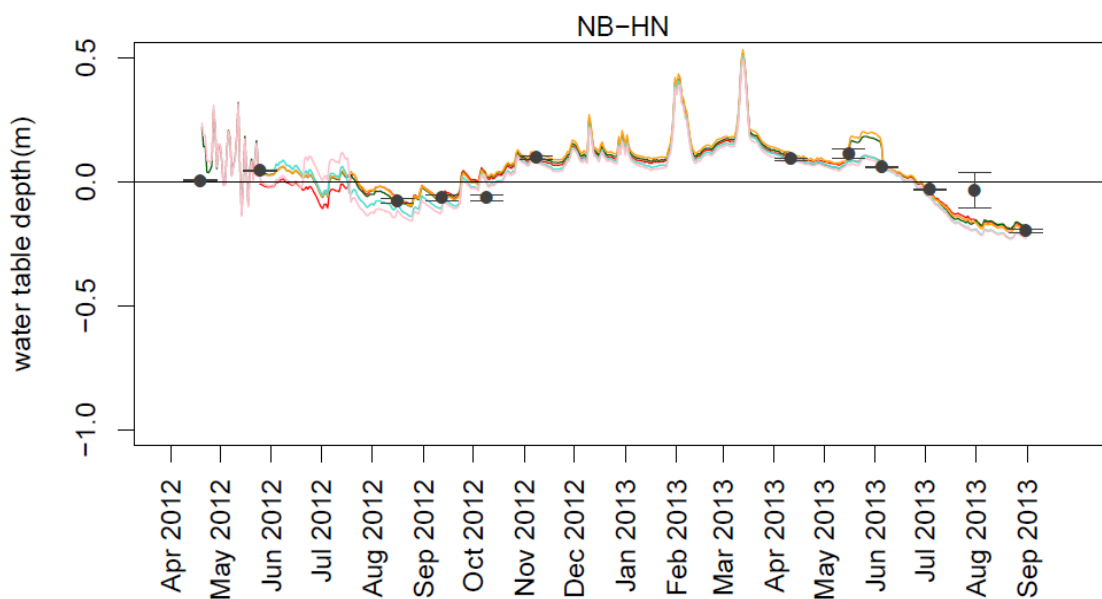


Figure 2.5.10. Continuous and manually measured water-table data for NB-HN

2.5.2.2. Static chamber gas fluxes

Static chamber gas flux measurements were made at NB-HN from June 2012 to September 2013, as at NB-HN. Fitted models for CO₂ fluxes from the *Phragmites*-dominated vegetation (Figure 2.5.11) suggest strong seasonal variation in ER, but somewhat less pronounced variations in GPP.

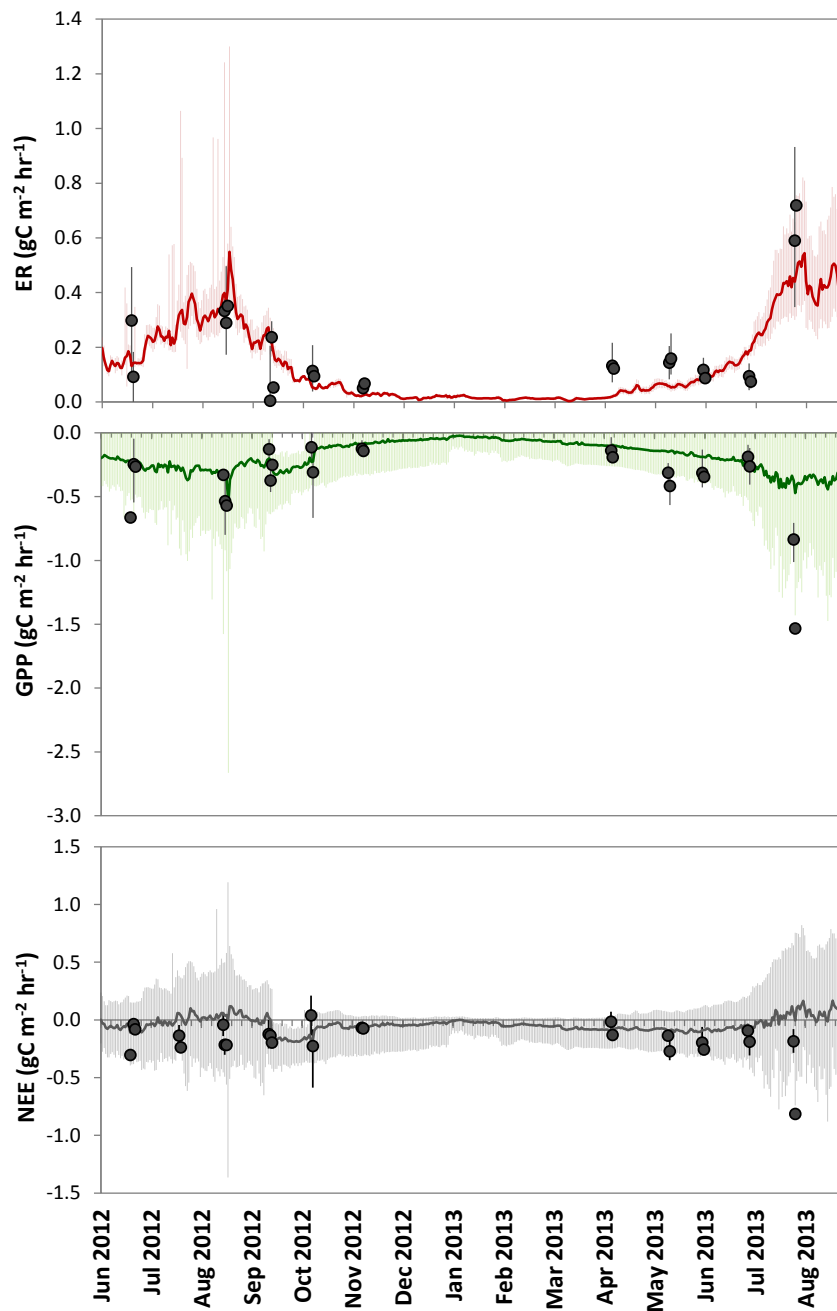


Figure 2.5.11. Modelled and observed CO₂ fluxes (ER = ecosystem respiration, GPP = gross primary productivity, NEE = net ecosystem exchange) based on static chamber measurements at NB-HN. Continuous lines show modelled daily mean fluxes, shading shows modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date. Note that static chamber measurements were generally taken around the middle of the day, so tend to be representative of peak (rather than daily mean) rates of photosynthesis and respiration.

Estimated annual ER and GPP for the study period were 1334 and 901 g C m⁻² yr⁻¹ respectively, giving a very large apparent net CO₂ uptake (annual NEE -434 g C m⁻² yr⁻¹). If correct, this would represent the strongest rate of net uptake of any of the study sites, and is in marked contrast to the near-zero NEE of NB-LN. However because a significant part of the reserve is covered by open water (which is a net CO₂ source; see below) the actual rate of CO₂ uptake by the terrestrial ecosystem is somewhat lower when considered on a

site-wide based. Nevertheless, these results suggest that, as in the Anglesey Fen sites, the more nutrient-enriched, *Phragmites*-dominated tall fen is acting as a stronger CO₂ sink (into the biomass and/or peat) than the lower-nutrient short fen sites in the same region. Some caution is however clearly required in interpreting the results of flux models based on a relatively small number of static chamber measurements at the 'secondary' Norfolk Broad sites.

As at NB-LN, an empirical flux model was fitted to the CH₄ data for NB-HN (Figure 2.5.12), although the scatter in measurements was relatively high, and in some cases net CH₄ uptake was observed. From the model, the net annual emission of CH₄ from NB-HN was 13.0 g C m⁻² yr⁻¹, which is very similar to the estimated emissions from NB-LN, and also the Anglesey Fen sites.

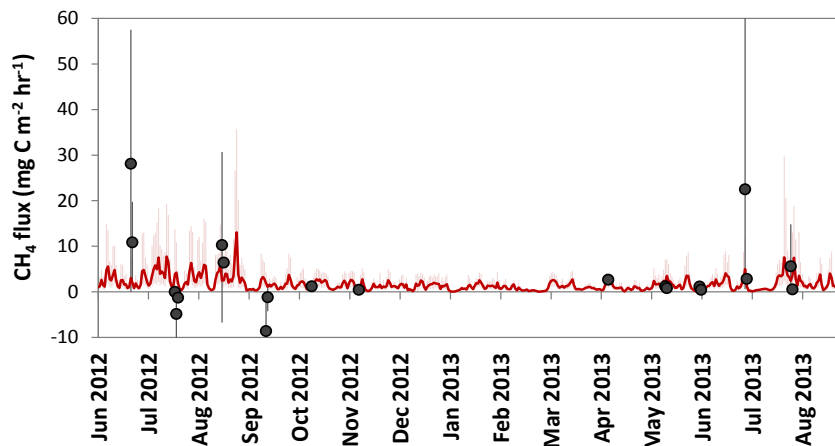


Figure 2.5.12. Modelled and observed CH₄ fluxes at NB-HN. Continuous lines show modelled daily mean fluxes, shading shows modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date.

Floating chamber data from two ditches at NB-HN (Figure 2.5.13) show a fairly consistent seasonal pattern of CH₄ emissions, with large fluxes between May and September, and near-zero values (albeit based on a limited number of measurements) at other times. Interpolation of these data gives a CH₄ emission per unit ditch area of 12 g C m⁻² yr⁻¹, which is less than 5% of the emissions measured at NB-HN. However, the Strumpshaw Fen reserve which comprises the NB-HN site contains large areas of shallow open water, integrated with the ditch network, and if included in the overall water surface area from the site along with the ditch area this leads to a Frac_{ditch} of 0.22, i.e. nearly a quarter of the reserve is open water. If these areas are all assumed to emit CH₄ at the same rate as the measurement ditches, this would lead to a total area-adjusted open water CH₄ emission for the site of 2.74 g C m⁻² yr⁻¹, which is much more similar to the open water emission at NB-LN. On the other hand, because the mean emission per unit ditch area is very similar to the mean emission per unit land area (12.3 g C m⁻² yr⁻¹) the inclusion of open water CH₄ emissions makes little difference to the total emissions from the site as a whole; indeed, if the larger water bodies were emitting less CH₄ than the measurement ditches this would actually lead to a reduction in the overall site CH₄ emission relative to the terrestrial flux alone. The actual magnitude of open water CH₄ emissions for these sites is unknown; natural water bodies were excluded from the IPCC Wetland Supplement, and methods for reporting emissions from 'flooded lands' (i.e. man-made water bodies) other than ditches have not yet been developed. For CO₂, the net flux varies from negative to positive over time, but on average the ditches were a net annual source of 129 g C m⁻² yr⁻¹ ditch surface area, which translates to a site-wide areal flux of 28.7 g C m⁻² yr⁻¹.

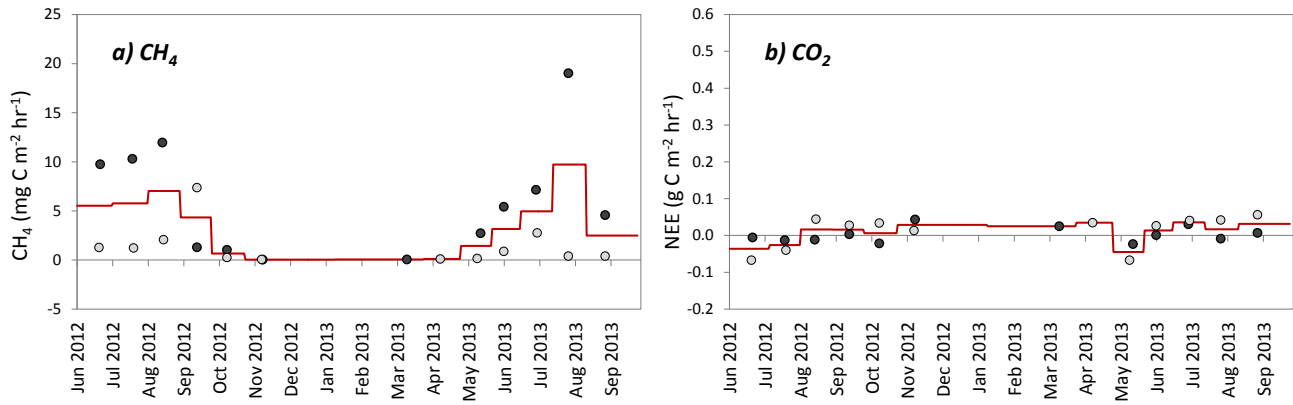


Figure 2.5.13. Observed CH_4 and CO_2 fluxes measured in ditches at NB-HN. Observations from two ditches are represented by grey and black circles, red line shows interpolated fluxes.

2.5.2.3. Aquatic carbon fluxes

Measured ditch water concentrations at NB-HN (Figure 2.5.14) show somewhat contrasting temporal variations in DOC concentrations compared to NB-LN, with short-lived peak concentrations in July of both years, and low concentrations (mostly $< 20 \text{ mg l}^{-1}$) for the rest of the measurement period, and an estimated annual mean of 21 mg l^{-1} , lower than NB-LN. DIC concentrations were high ($> 50 \text{ mg l}^{-1}$) for most of the measurement period, with a mean of 73 mg l^{-1} , but fell to lower levels when water-tables fell in the summer of 2013. Based on mean annual estimated concentrations and mean areal runoff from the EA gauging station, DOC and DIC fluxes are estimated to be in the region of 4.1 and $14.6 \text{ g C m}^{-2} \text{ yr}^{-1}$ respectively. Again these estimates are highly approximate, but suggest that the DOC export from NB-HN is fairly minor, and the DIC flux somewhat larger.

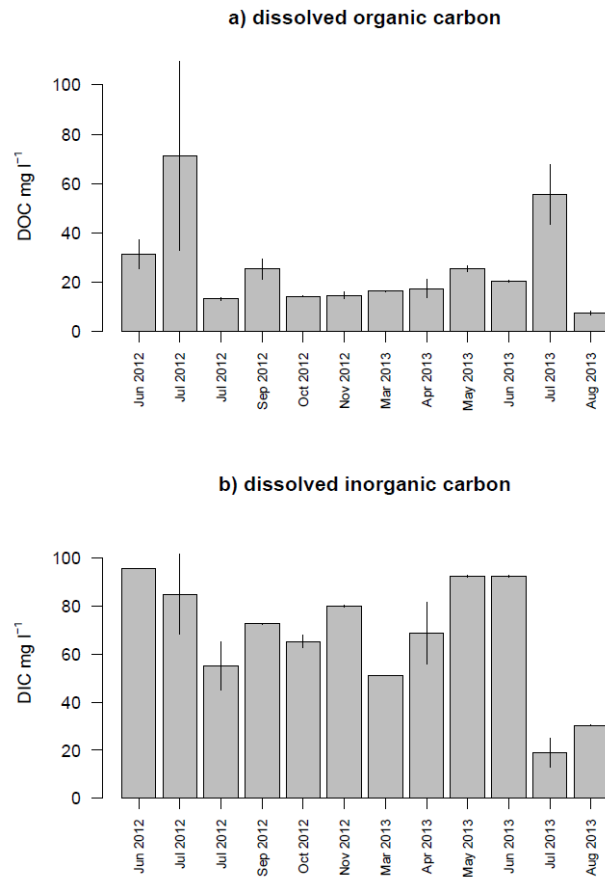


Figure 2.5.14. Mean and standard error of ditch dissolved organic and inorganic carbon concentrations, NB-HN. Note that x axis simply records sampling dates and is therefore not a true time axis.

2.6. Thorne Moors

Thorne Moors, and the nearby Hatfield Moors, are large lowland raised bogs that lie within the Humberhead Levels of South Yorkshire (Figure 2.6.1). With a combined area of 1900 ha, they represent one of the largest remaining areas of lowland raised bog in England. Large areas of peat surrounding the Moors have been lost through drainage for agriculture, as well as ‘warping’, a process whereby silt-rich river water was flooded onto the peat to enhance fertility. Peat block cutting has been carried out in the Humberhead Levels since the 14th century, and in the late 19th century peat from the Moors was exported to London to provide bedding material for animals (Caulfield, 1991). This process was mechanised in the 1960s to produce peat for horticulture, and a dense network of drainage ditches installed. From the 1970s onwards, parts of the Moor were acquired by conservation organisations, but milled peat extraction continued across large parts of the site until 2004. Subsequently, Thorne Moor has been managed by Natural England as a National Nature Reserve, and is also designated as a SSSI, SAC and SPA. Initial restoration of the site involved partial blocking of ditches and formation of water-retaining compartments using bunds. However, a functional ditch network remains, and some water is pumped off the site during winter to avoid flooding of surrounding farmland. Much of the extracted peat surface has now revegetated, with a high cover of hare’s tail cotton grass, *Eriophorum vaginatum*, and some re-establishment of *Sphagnum*, but areas of bare peat still remain.

Along with the Norfolk Broads, the two Thorne Moors sites were ‘secondary’ study sites within the network, with a reduced and shorter period of sample collection carried out with the support of a PhD student, Gemma Dooling at the University of Leeds. This included static chamber measurements, water chemistry sampling and manual water-table monitoring. Peat cores were collected at TM-RW.

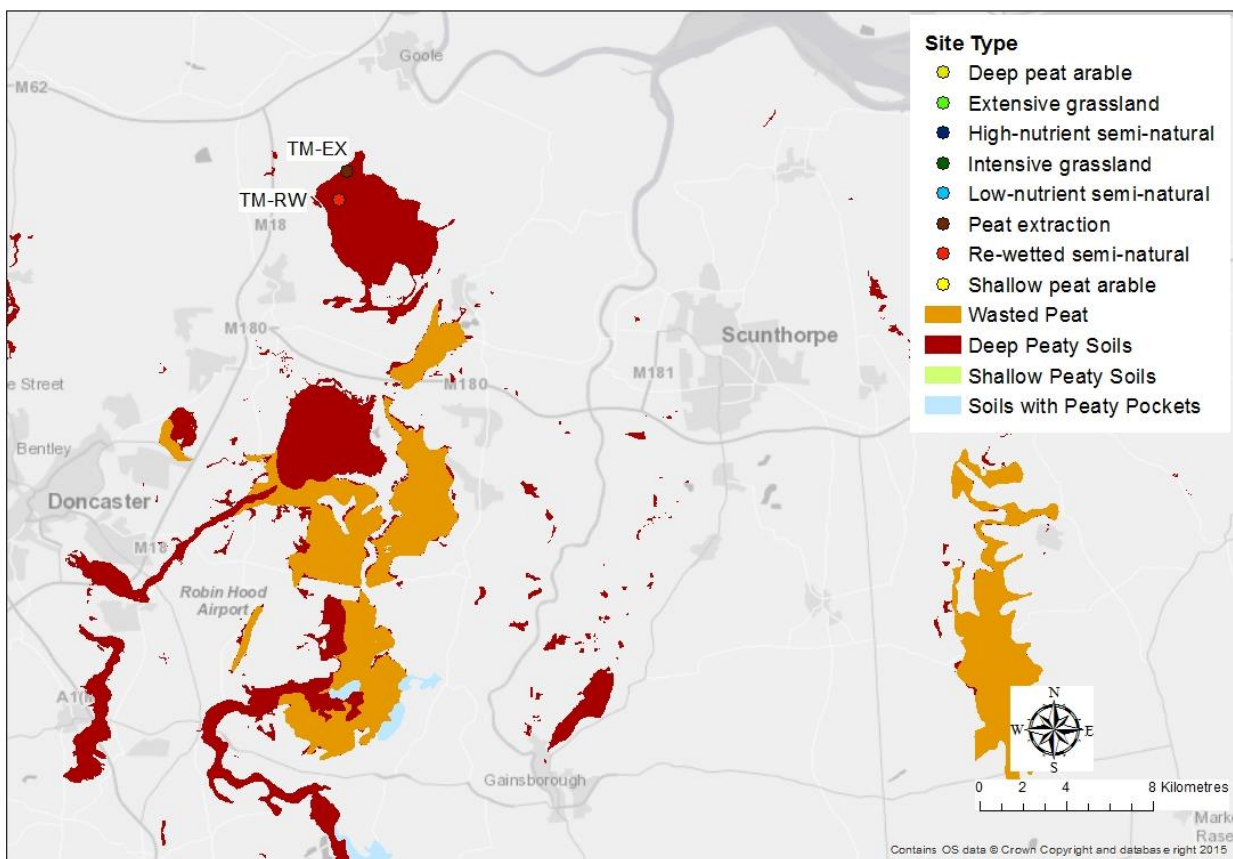


Figure 2.6.1. Thorne Moors and the wider Southern Humberhead Levels study area.

Figure 2.6.2 shows monthly meteorological data for Thorne Moors, based on a single AWS which is close to both study sites. Project measurements at Thorne Moors ended in 2014, so meteorological data are shown for the earlier period only. Data from the AWS at the site have gap-filled using data from two ancillary data sources (see Figure 2.6.2 caption). Total incoming solar radiation at this location peaked at 154 and 176 kW m⁻² in July 2013 and 2014, respectively. These peak values are lower than the maximum monthly irradiance

measured at other sites in the network (particularly the EF and SL-EG sites). With the exception of July, total solar radiation for the months with data coverage was slightly higher in 2013 than 2014.

Similar to other sites, mean air temperature was lower during spring 2013 than for the corresponding months of 2014. August and October were warmer in 2013 than 2014. Minimum and maximum average monthly air temperatures were 1.8 °C in March 2013, and 17.4 °C in July 2013. Missing data preclude the calculation of mean annual temperatures at this site. Monthly precipitation totals for the months with data coverage (lower panel of Figure 2.6.2) were below 70 mm month⁻¹ with the exception of October 2014 (154 mm month⁻¹) and November 2014 (176 mm month⁻¹).

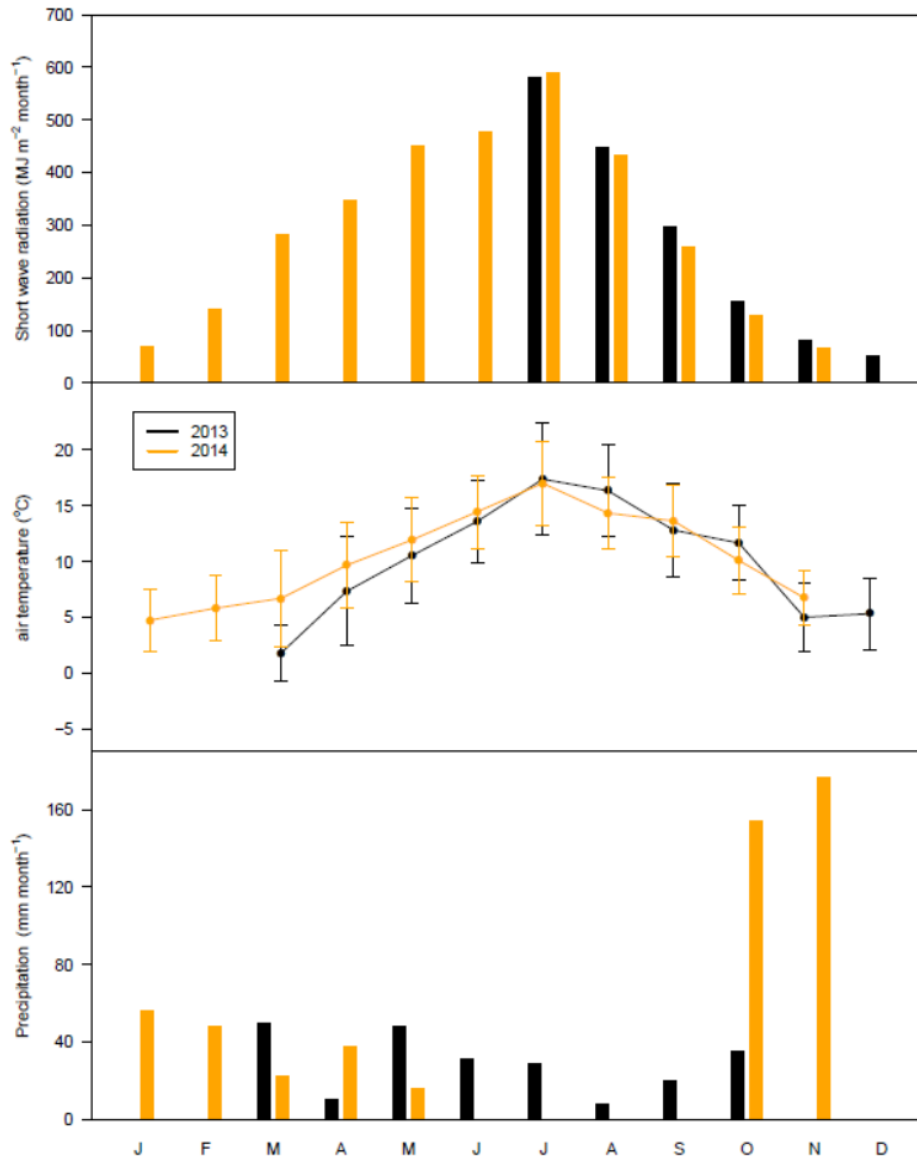


Figure 2.6.2. Total monthly incoming solar radiation (top), air temperature (middle) and total monthly precipitation (lower) for the Thorne Moors site. Error bars on the temperature plot show one standard deviation of the mean. Data have been gap-filled using data from two other automated weather stations located at Thorne Moor (data supplied by Kevin Brown, E.ON) and Top House Farm (data courtesy of James Hinchliffe).

2.6.1. Thorne Moors – Extraction site (TM-EX)

The extraction site is located in an area of former industrial peat cutting. Although the measurement site has been abandoned since the early 2000s, and is located within the wider area of re-wetted bog, it lies in an area where ditch blocking has been less effective in raising water tables, which remain below the surface year-round. As a result, and in the absence of active measures to re-establish vegetation cover, the peat surface remains bare, and is thus considered to remain broadly representative of an extraction site. Data from other peat extraction sites in the UK and Ireland (Wilson et al., 2014) suggest that GHG fluxes from active and abandoned industrial extraction sites are not significantly different, at least until active restoration measures are undertaken.

2.6.1.1. Hydrology

Manual water-table measurements for TM-EX are shown in Figure 2.6.3, for the period from June 2013 to September 2014. Although it is not possible to assess short-term variability in the absence of continuous logger data, the manual data suggest that mean water levels at the site have remained relatively stable at between 10 and 40 cm. This contrasts with the highly variable water-table observed at the active MM-EX extraction site (Figure 2.2.12), probably because the TM-EX site is part of an abandoned former extraction site which (as it lies within the larger, re-wetted Thorne Moors site) is no longer effectively drained. Nevertheless, water-tables remain consistently below the peat surface, indicating that some residual drainage effects are still present.

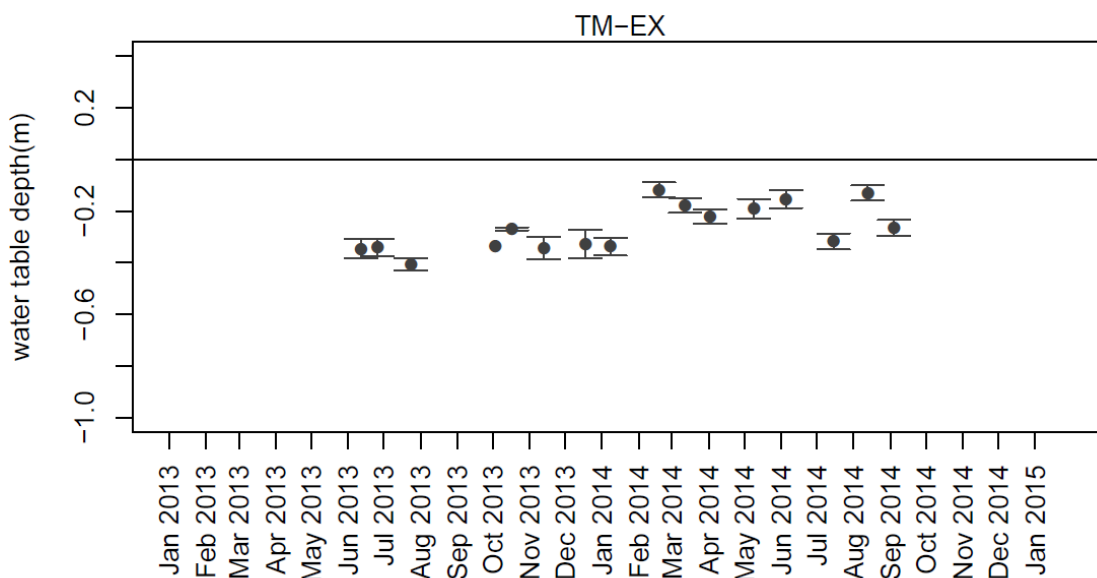


Figure 2.6.3. Manually measured water-table data for TM-EX

2.6.1.2. Static chamber gas fluxes

As the TM-EX site is unvegetated, GPP was zero and the rate of CO₂ emission from the peat surface is represented by the measured rate of ER (Figure 2.6.4). Respiration rates were consistently low compared to most vegetated sites. Observed fluxes were consistently higher than modelled daily means, which can partly be attributed to higher daytime temperatures when measurements were made, however it does appear in this case that the model may have under-estimated rates of CO₂ loss, especially during winter. The estimated annual mean CO₂ emission from TM-EX was 143 g C m⁻² yr⁻¹, which was very similar to the estimated value for MM-EX.

Measured CH₄ fluxes for TM-EX were consistently small, and predominantly negative (Figure 2.6.5). Estimated annual mean CH₄ uptake was -0.04 g C m⁻² yr⁻¹.

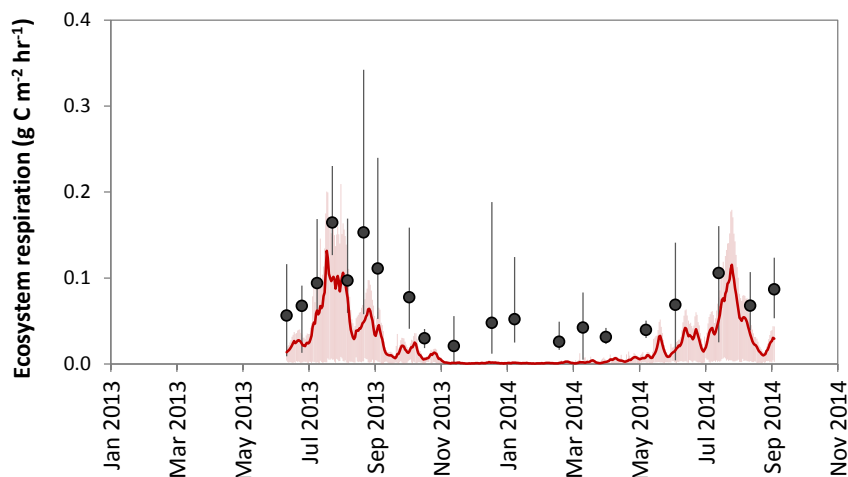


Figure 2.6.4. Modelled and observed ecosystem respiration at TM-EX. Continuous lines show modelled daily mean fluxes, shading shows modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date. Note that static chamber measurements were generally taken around the middle of the day, so tend to be representative of peak (rather than daily mean) respiration rates.

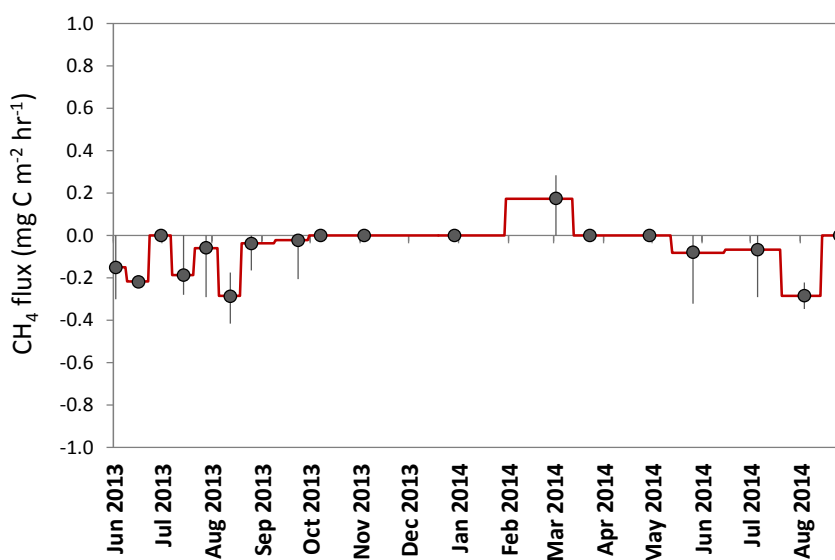


Figure 2.6.5. Measured CH_4 fluxes for TM-EX. Points show mean observations on each measurement date, and error bars show range of measured values on that date. Red lines show interpolated fluxes.

2.6.1.3. Aquatic carbon fluxes

Only four (pore)water samples were collected from TM-EX, during the summer of 2014. These all showed very high DOC concentrations (range 200 – 250 mg l^{-1}) and very low DIC concentrations (range 1 – 6 mg l^{-1}), the latter reflecting the low pH of the raised bog. While it was not possible to calculate either mean annual concentrations or flux estimates from such a limited dataset, the DOC concentrations at TM-EX were quite similar to those measured at MM-EX during the summer period (Figure 2.2.29). This suggests that the two extraction sites are functioning similarly in terms of DOC production. However mean river discharges in the Humberhead Levels (e.g. Torne at Auckley, 210 mm yr^{-1} ; <http://nrfa.ceh.ac.uk/data/station/info/28050>) are considerably lower than those in the vicinity of the Manchester Mosses (e.g. Glaze Brook at Little Woolden Hall, 897 mm yr^{-1}) which would suggest that aquatic carbon losses from the Thorne Moors sites are probably lower than the very high values obtained for the Manchester Mosses sites, despite similarities in peat type and management.

2.6.2. Thorne Moors – Re-wetted site (TM-RW)

The re-wetted site at Thorne Moors was previously subject to industrial peat extraction, but has been undergoing restoration for more than a decade. Water levels have been raised via ditch blocking, and the water-table remains at or slightly above the peat surface for much of the year, although in contrast to the MM-RW site some water-table drawdown occurs during dry summer periods. The study area has revegetated, but remains dominated by *Eriophorum vaginatum*.

Peat core data (Figure 2.6.6) show that the site retains the physical and chemical characteristics of a raised bog, with the lowest measured bulk density values of any site in the network, and high carbon content which (although this was not measured directly) suggests a negligible mineral content. The peat C/N ratio was somewhat reduced in the upper 25 cm, but very high below that depth.

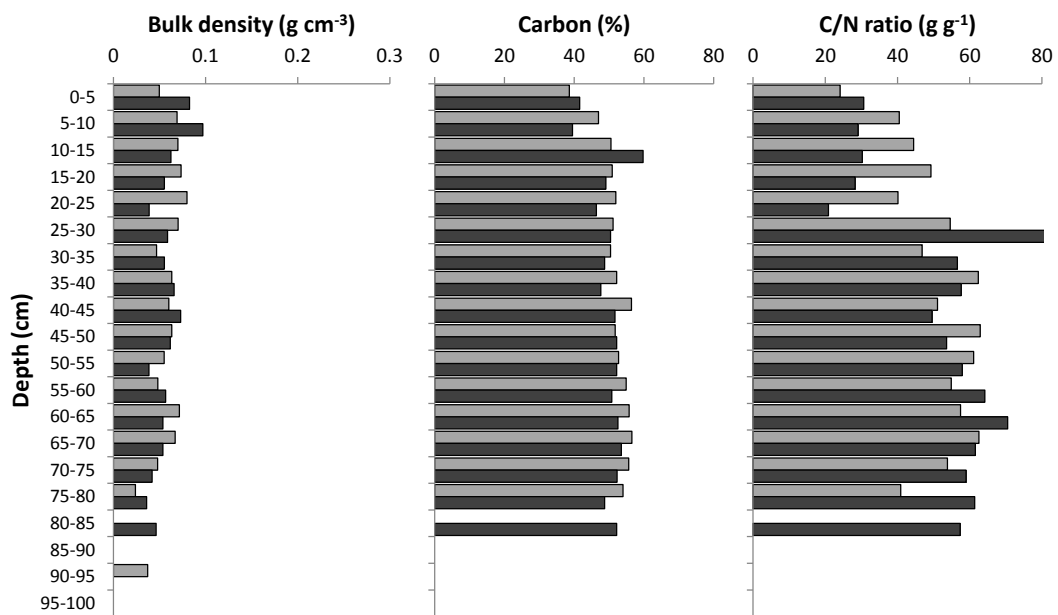


Figure 2.2.6. Peat core data, TM-RW. Data from two cores analysed from the site are represented by dark and light grey-shaded bars.

2.6.2.1. Hydrology

Manual water-table measurements at TM-RW from June 2013 to September 2014 are shown in Figure 2.6.7. The data show that water tables are at or slightly above the surface for much of the year, with a sustained period of inundation during the winter period. Water-table drawdown occurred during the summer of 2013, to a maximum depth of around 30 cm, whereas the site remained wet throughout 2014.

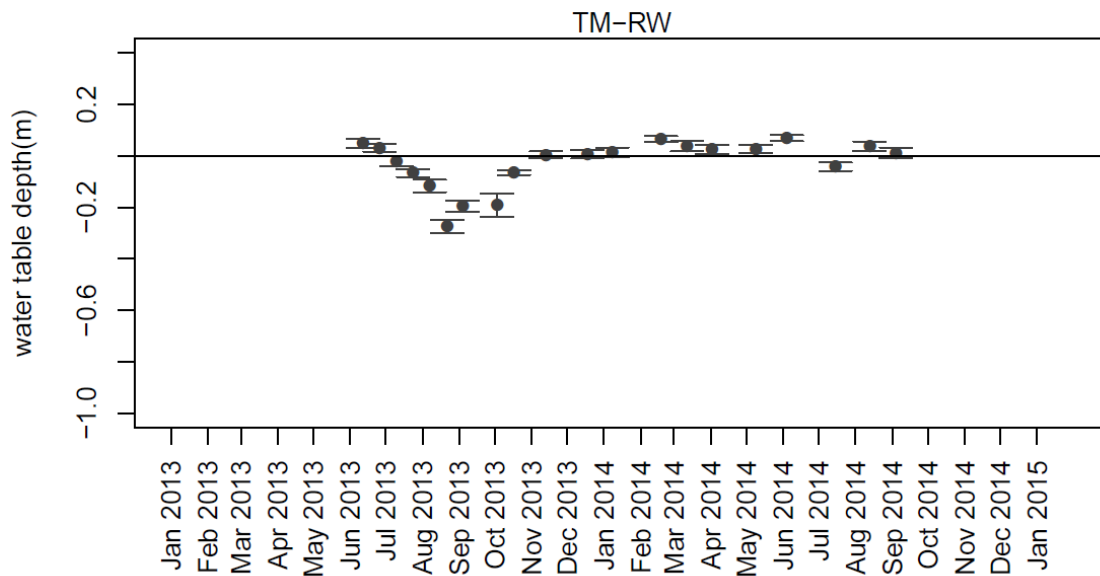


Figure 2.6.7. Manually measured water-table data for TM-RW

2.6.2.2. Static chamber gas fluxes

Measured and modelled static chamber CO₂ fluxes at TM-RW are shown in Figure 2.6.8. The fitted models for both ER and GPP appear generally good, and gave annual mean flux estimates of 773 g C m⁻² yr⁻¹ for GPP, and 996 g C m⁻² yr⁻¹ for ER. Modelled daily NEE showed greater deviation from (daytime) measured values, with net CO₂ exchange oscillating either side of zero during much of the growing season. There was a tendency for the site to become a net CO₂ source during hot summer periods. Overall, based on the balance of annual GPP and ER estimates, it appears that TM-RW remains a net CO₂ source despite re-wetting, with an NEE of +223 g C m⁻² yr⁻¹.

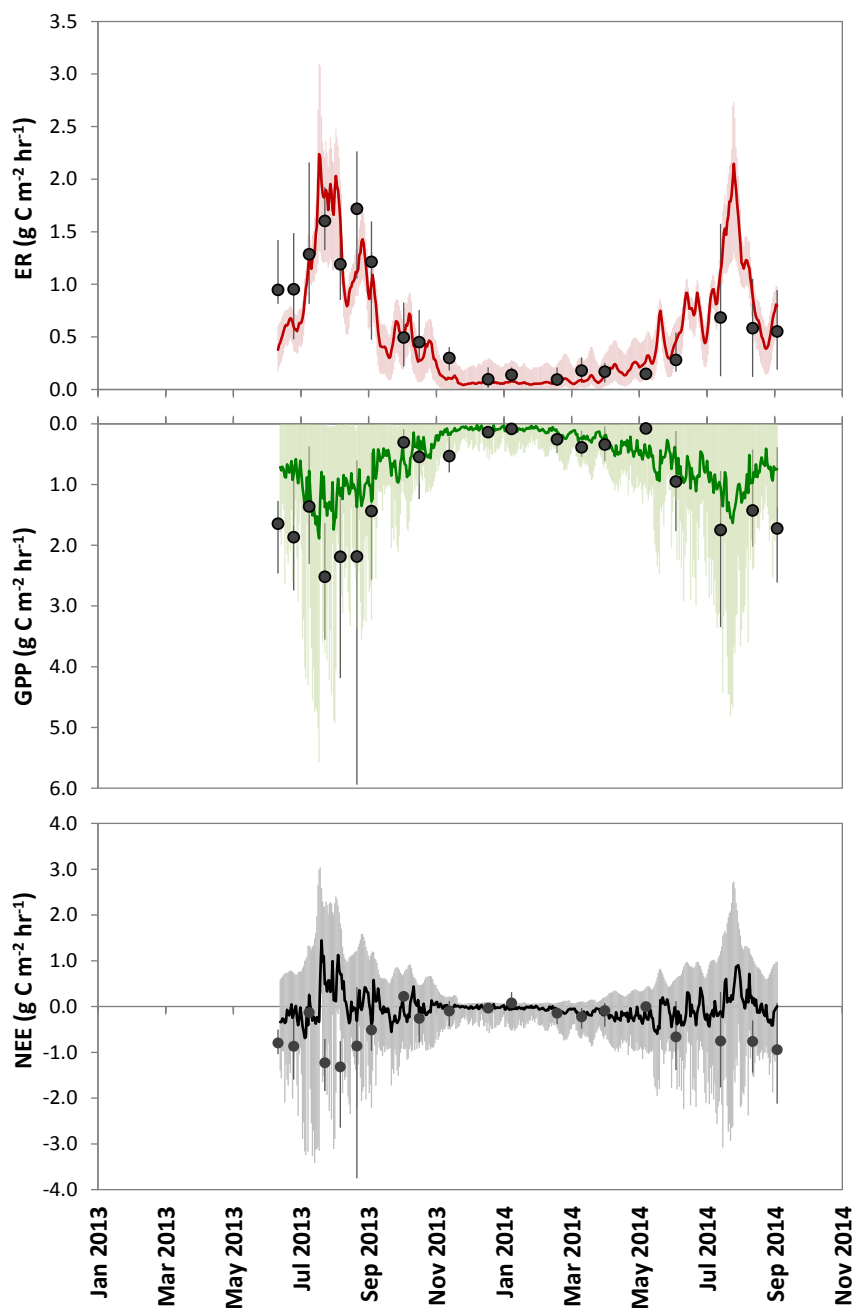


Figure 2.6.8. Modelled and observed CO_2 fluxes (ER = ecosystem respiration, GPP = gross primary productivity, NEE = net ecosystem exchange) based on static chamber measurements at TM-RW. Continuous lines show modelled daily mean fluxes, shading shows modelled diurnal range, points show mean observations on each measurement date, and error bars show range of measured values on that date. Note that static chamber measurements were generally taken around the middle of the day, so tend to be representative of peak (rather than daily mean) rates of photosynthesis and respiration.

Methane flux data from the peat surface at TM-RW are shown in Figure 2.6.9 (ditch fluxes were not measured). Although temporal and small-scale (between-chamber) variations were considerable, mean fluxes were positive on all but two sampling dates, and mostly fell within the range $1\text{--}4 \text{ g C m}^{-2} \text{ hr}^{-1}$. Over the measurement period, the estimated annual mean emission for CH_4 was $11.9 \text{ g C m}^{-2} \text{ yr}^{-1}$. This emission rate is much lower than the $32.8 \text{ g C m}^{-2} \text{ yr}^{-1}$ recorded at the permanently inundated MM-RW, but fairly similar to that recorded at most of the conservation-managed fen sites. It is also fairly similar to CH_4 fluxes estimated for a re-wetted upland blanket bog where *Eriophorum* colonisation of blocked ditches had occurred ($8.8 \text{ g C m}^{-2} \text{ yr}^{-1}$; Cooper et al., 2014), although higher than typical fluxes from undrained bogs (e.g. $4.6 \text{ g C m}^{-2} \text{ yr}^{-1}$ in the same study).

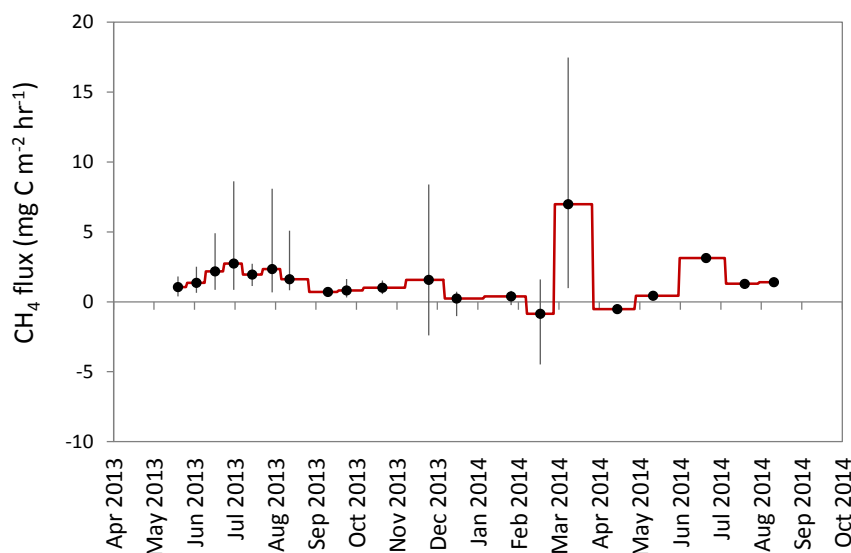


Figure 2.6.9. Measured CH₄ fluxes for TM-RW. Points show mean observations on each measurement date, and error bars show range of measured values on that date. Red lines show interpolated fluxes.

2.6.2.3. Aquatic carbon fluxes

Four porewater samples were collected from TM-RW in the summer of 2014. As for TM-EX, DOC concentrations during this period were very high (190 to 240 mg l⁻¹) and DIC concentrations very low (< 4 mg l⁻¹). Data were insufficient to calculate mean concentrations or fluxes, but again suggest that DOC export may be significant from the site, but that fluxes were probably not as extreme as those obtained for MM-RW.

3. N₂O EMISSIONS FROM PEAT SOILS UNDER AGRICULTURAL MANAGEMENT

3.1. Reporting in the Agriculture sector of the GHG inventory

The drainage and management of organic soils (Histosols) is assumed to result in N mineralisation which will subsequently give rise to direct emissions of N₂O, an estimate of which is required for annual reporting in the Agriculture sector of the UK GHG inventory. To provide this annual estimate, the default emission factor of 8 kg N₂O-N ha⁻¹ from the IPCC 2006 Guidelines is assumed. This default emission factor is based on relatively few data (Klemmedtsson et al., 1999) with no accounting for potential influencing factors including climate (temperature, rainfall), soil composition (C to N ratio) and soil management (e.g. tilled lowland peat soils might be expected to have far higher mineralisation rates than drained grassland).

Estimates of emissions are also reported for direct (arising from nitrification and denitrification processes) and indirect (secondary nitrification/denitrification following deposition of volatilised N and/or leaching of N) N₂O emissions from soil N amendments associated with agricultural production including fertiliser, grazing returns and managed livestock manures. The UK is currently in the process of revising the direct N₂O emission factors for these N amendments based on a large experimental and review program (Defra AC0116 and AC0114, e.g. Bell et al., 2015a; Bell et al., 2016), but measurements were made on mineral soils with no peat soil sites included in that study. With the ambition to relate emission factors to soil and climatic factors, information relating to emissions from N amendments to peat soils under agricultural management is a current knowledge gap.

Nitrification inhibitors offer potential to mitigate N₂O emissions when included with N amendments to soils (de Klein and Eckard, 2008). However, the efficacy of nitrification inhibitors at reducing emissions may be influenced by factors including soil temperature, soil texture and rainfall (e.g. Bronson et al., 1989; Kelliher et al., 2008; McGeough et al., 2016; Shepherd et al., 2014). In a study across six sites in England, Misselbrook et

al. (2014) reported mean reduction efficiencies when using the nitrification inhibitor dicyandiamide (DCD) of 39, 69, 70 and 56% when used with ammonium nitrate fertiliser, urea fertiliser, cattle urine and cattle slurry, respectively. Further UK assessments of the inclusion of DCD with N amendments to soils were conducted as part of Defra project AC0116, giving broadly similar results. McGeough et al. (2016) reported an influence of soil texture and soil organic matter content on DCD effectiveness, based on laboratory studies using the same soils as in the AC0116 field experiment with finer textured soils and those with a higher organic matter content associated with a lower emission reduction efficiency. Again, however, no peat soils were included in the study so the effectiveness of DCD in reducing direct N₂O emissions from N amendments to peat soils under agricultural management is unknown and if to be included in the national GHG emission inventory would be based on data from studies on mineral soils.

The objectives of this specific task within the project were, therefore, to provide some limited preliminary country-specific data on: 1) direct N₂O emissions deriving from mineralisation of lowland peat soils under agricultural management; 2) direct N₂O emissions from fertiliser N amendments to lowland peat soils; and 3) the effectiveness of the nitrification inhibitor DCD in reducing direct N₂O emissions from fertiliser N amendments to lowland peat soils. These preliminary data can then be used to assess whether currently used parameters are appropriate.

3.2. Methods

3.2.1 Sites and experimental design

Measurements were conducted at the three sites under intensive agricultural management i.e. EF-DA, MM-DA and SL-IG. At the EF-DA and SL-IG sites, randomised block design small plot experiments were established, but constraints imposed by the commercial farmer at the MM-DA site precluded this and so at MM-DA measurements were made using some of the chambers already installed for the GHG flux measurements as described in Section 1.3.4.

At EF-DA, flux measurements were made following establishment of a commercial potato crop on 3rd May 2014. Nine small plots (3 x 4m) were established, arranged in a randomised block design with three replicates of three treatments: 1) Control: no N fertiliser but otherwise managed as the rest of the field; 2) AN: ammonium nitrate applied at a rate of 60 kg ha⁻¹ N following commercial practice; 3) AN+DCD: ammonium nitrate and DCD applied (2% solution at 10 kg ha⁻¹) at a total N rate of 60 kg ha⁻¹. Half of the plot area was used for gas flux measurements, with five static chambers per plot, and half used for soil sampling for mineral N analysis. Flux measurements were made on 14 occasions following fertiliser application over a period of 40 d after which the farmer required the static chambers to be removed from the field. Soil samples were taken weekly for mineral N analysis.

A limited number of flux measurements were made at MM-DA in spring 2014 following fertiliser application to a winter wheat crop. Ammonium nitrate fertiliser was applied to the whole field on 23rd March. Six static chambers were used to measure N₂O fluxes, with DCD being added to three of the chambers directly after fertiliser application as a 2% solution at a rate equivalent to 10 kg ha⁻¹. Flux measurements were made on days 0, 1, 2, 3, 4, 7, 9, 45 and 67 after fertiliser application. No corresponding soil mineral N measurements were made for this site.

At SL-IG, a randomised block design experiment was established on a silage grass field, with nine small plots (3 x 6m) comprising three replicates of three treatments: 1) Control: no N fertiliser but otherwise managed as the rest of the field; 2) ASN: ammonium sulphate nitrate applied at a rate of 140 kg ha⁻¹ N; 3) AN+DCD: ammonium nitrate and DCD applied (2% solution at 10 kg ha⁻¹) at a total N rate of 140 kg ha⁻¹. Fertiliser applications were split across two application timings, with 70 kg N ha⁻¹ being applied on 24th March and again on 9th May. As for the EF-DA site, half of the plot area was used for flux measurements, with five static chambers per plot, and half used for soil sampling for mineral N analysis. Flux measurements were made on 32 occasions following the first fertiliser application over a 204-d period, with more frequent measurements immediately following fertiliser applications. There were two further sampling occasions for the control plots only on day 232 and 287. Soil samples were taken weekly for mineral N analysis for the first 4 weeks after the first fertiliser application and then approximately monthly thereafter.

3.2.2 Nitrous oxide flux measurements

Nitrous oxide flux measurements were made using the static chamber technique, employing five chambers per plot (except at MM-DA) to account for spatial variability, with more frequent sampling in the period immediately following fertiliser application. On each sampling occasion, lids were placed on the chambers which were closed for a period of 40 minutes. Ten ambient air samples were taken, the mean N₂O concentration of which was assumed to equate to the initial concentration in each chamber (t_0) and a single sample was taken from each chamber at 40 minutes after closure (t_{40}). For three chambers on each occasion, samples were taken at t_0 , t_{20} , t_{40} and t_{60} to verify that the increase in headspace N₂O concentration was linear over the 40-minute closure period. Sampling was conducted between 10 am and 12 pm, with fluxes measured during this time assumed to be representative of the mean daily flux. This method has been proven to be sufficiently robust for the purposes of these experiments (Chadwick et al., 2014). Gas samples were stored in pre-evacuated vials and N₂O concentration measured using a gas chromatograph fitted with an electron capture detector. Mean daily flux (F , g N₂O-N ha⁻¹ d⁻¹) was determined for each chamber according to:

$$F = \rho \frac{V(C_{t40} - C_{t0})}{At} \times \frac{273}{(T + 273)} \times (60 \times 24 \times 10)$$

where ρ (mg m⁻³) is the density of N₂O, V (m³) the volume of the chamber, C_{t40} and C_{t0} (ppmv) the chamber headspace N₂O concentrations at t_{40} and t_0 , A (m²) the surface area covered by the chamber, t (minutes) the chamber closure time and T (°C) the ambient air temperature at the time of sampling.

The mean flux of the five chambers per plot was used as the representative plot daily flux for each sampling occasion and cumulative emissions were derived for the overall experimental period by linear interpolation between measurement occasions. Nitrous oxide emission factors for the fertiliser treatments were derived as the net cumulative emission (i.e. cumulative emission of the fertiliser treatment minus that of the control treatment) expressed as a percentage of the N applied.

3.2.3 Soil measurements

At each flux sampling occasion at the EF-DA and SL-IG sites, representative soil samples (0-10 cm depth) were taken from the experimental area for determination of gravimetric moisture content by drying for 24 h at 80 °C. Water-filled pore space was then derived from the gravimetric moisture content, site bulk density and soil particle density. Less frequently, soil samples were taken per plot (0-10 cm depth) for determination of soil ammonium-N and nitrate-N concentrations following extraction with 2 M KCl.

3.3. Results

3.3.1 Rosedene Farm – arable on deep peat (EF-DA)

Nitrous oxide fluxes were low throughout the measurement period for the Control and AN+DCD treatments, although both showed a small peak event 13 d after treatment application, related to increasing temperature (Figure 3.3.1). Flux from the AN treatment was also low initially, but reached a significantly greater peak after 13 d of 66 g N₂O-N ha⁻¹ d⁻¹ than either the control or AN+DCD treatments. Fluxes were not significantly different between treatments at 40-d after treatment application at which time measurements were discontinued.

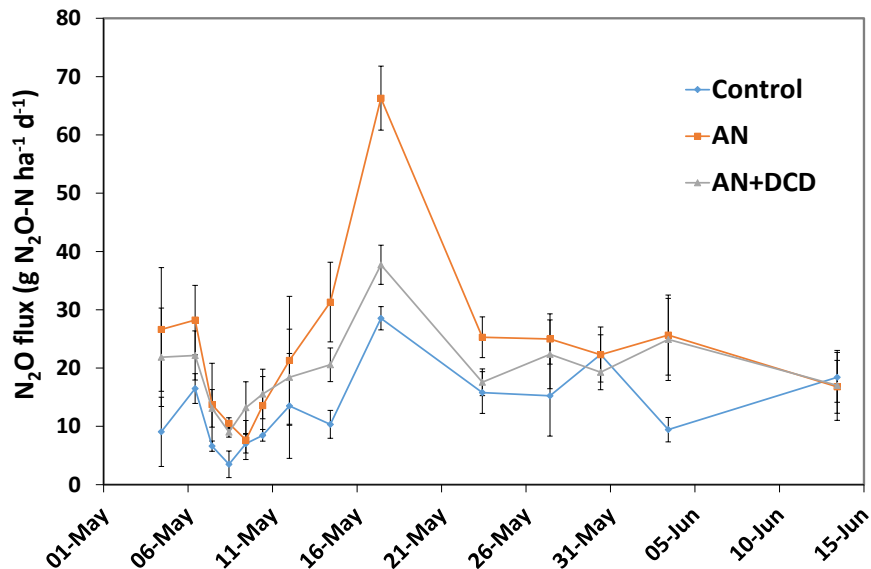


Figure 3.3.1. Measured N₂O flux for the control, fertiliser (AN) and fertiliser plus nitrification inhibitor (AN+DCD) treatments at the EF-DA site. Data are mean values (n=3) with error bars showing ±1 standard error of the mean.

Soil water filled pore space (WFPS) was low (<25%) throughout the measurement period, suggesting nitrification rather than denitrification as the predominant source of N₂O. There was some evidence of a delay in nitrification for the AN+DCD treatment as soil ammonium-N content declined less rapidly for that treatment than the AN alone.

Cumulative N₂O emissions over the 40-d measurement period were 627, 1118 and 884 g N₂O-N ha⁻¹ for the control, AN and AN + DCD treatments, respectively. There were significant differences in cumulative flux between treatments (P=0.025) with that from AN being significantly greater than from the control, but AN+DCD was not significantly different from either the control or AN treatment. Emission factors for the AN and AN+DCD treatments were 0.82 and 0.43%, respectively, thus DCD giving a numerical (but not statistically significant) reduction in emission of approximately 50%.

3.3.2 Little Woolden Moss - arable on deep peat (MM-DA)

Emissions measured at the MM-DA site following fertiliser application were very low (<10 g N₂O-N ha⁻¹ d⁻¹) for all measurement occasions, although we cannot rule out the possibility of a peak in emission during the period between 2nd April and 8th May when no measurements were made (Figure 3.3.2). With no direct comparison with a control treatment, it was not possible to derive net cumulative emission estimates of emission factors for the fertiliser treatments. There was some evidence of a reduction in emission rates with the inclusion of DCD with the fertiliser, but at such low emission rates an effect of the inhibitor would not necessarily be expected to be observed.

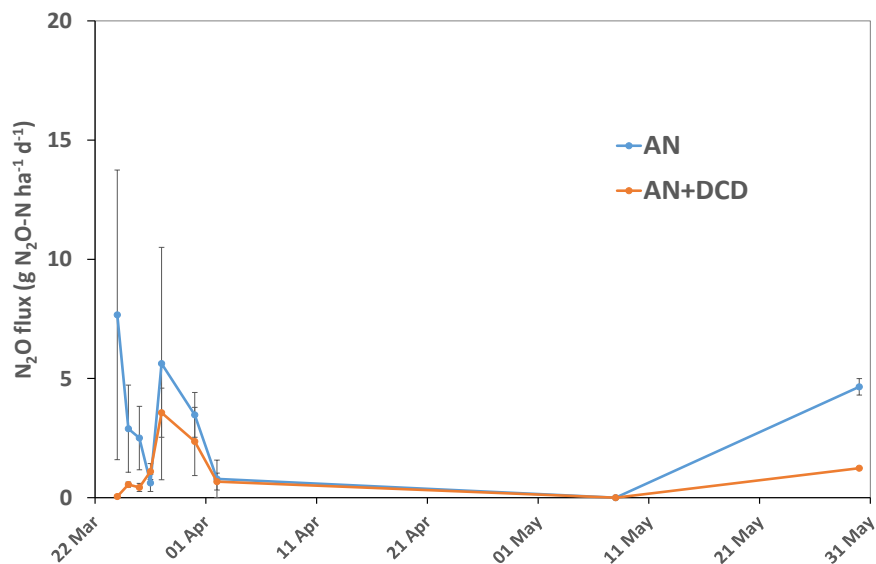


Figure 3.3.2. Measured N₂O flux for the fertiliser (AN) and fertiliser plus nitrification inhibitor (AN+DCD) treatments at site MM-DA. Data are mean values (n=3) with error bars showing ± 1 standard error of the mean.

3.3.3 Tadham Moor – intensive grassland (SL-IG)

Peak N₂O fluxes were greater at the SL-IG site than the other two sites for the control and fertiliser amendment treatments (Figure 3.3.3). Soil water content was much greater for the SL-IG site than the EF-DA site throughout the measurement period, being fairly constant at approximately 50% WFPS between March and early June and then declining rapidly to a low of 28% followed by a steady increase over the remainder of the year. There was large variability in measured flux rates and the differences between treatments on given sampling dates were often not statistically significant ($P > 0.05$). Fluxes were numerically greater from the fertiliser treatments than the control following the first fertiliser application but fluxes from all treatments declined to similar values just prior to the second fertiliser application. Following the second application there was an immediate increase in fluxes from both fertiliser treatments, with only limited evidence of any reduction effect of the DCD, which then declined over the next 4-6 weeks until they were no different from the control.

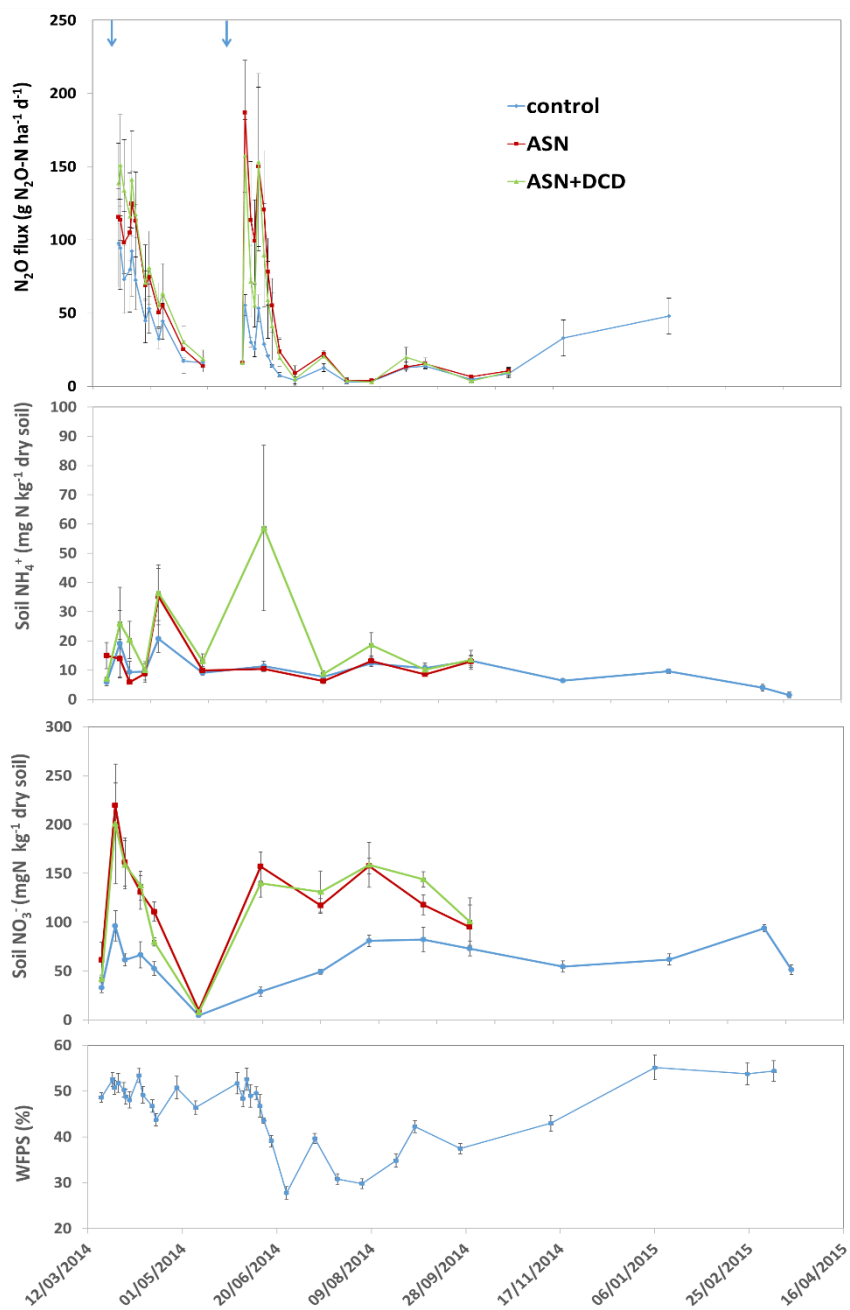


Figure 3.3.3. Measured N₂O flux, soil ammonium-N and nitrate-N contents and water-filled pore space for the control, fertiliser (ASN) and fertiliser plus nitrification inhibitor (ASN+DCD) treatments at the SL-IG site. Data are mean values ($n=3$) with error bars showing ± 1 standard error of the mean.

Soil nitrate-N concentrations increased markedly following each fertiliser application, declining rapidly following the first application to a level similar to the control immediately prior to the second application. Following the second application the soil nitrate-N concentrations remained elevated above those of the control until measurements ceased on the fertiliser treated plots. There was no evidence from the measured soil nitrate-N concentrations of any nitrification inhibitory effect of the added DCD, although soil ammonium-N content was much greater in the ASN+DCD than ASN treatment for the first soil sampling occasion following the second fertiliser application.

Cumulative N₂O emissions over the 204-d period from which measurements were made from all treatments were not significantly different between treatments ($P=0.251$), despite a large numerical difference between the control and fertiliser treatments with values of 3830, 6272 and 6216 g N₂O-N ha⁻¹ for control, ASN and ASN+DCD, respectively. Emission factors were effectively the same for AN and ASN (i.e. the DCD had no effect in reducing emissions) at 1.7% of N applied. Cumulative emission from the control treatment over the 287-d measurement period was estimated as 6676 g N₂O-N ha⁻¹.

3.4. Discussion

Mean N₂O flux from the control plots, which might be considered to equate with the IPCC 2006 Guidelines emission source 'managed histosols', was 15.68 and 23.26 g N₂O-N ha⁻¹ d⁻¹ for EF-DA and SL-IG, respectively. Assuming these to be representative of the mean daily flux across the whole year, this would equate with annual emission factors of 5.72 and 8.49 kg N₂O-N ha⁻¹. The assumption of annual representativeness is not valid for the EF-DA site where measurements were only made for a 40-day period during summer. Even for the SL-IG site, where measurements were made over 287 days, the frequency of measurement was greater for the spring and summer months. With the high variability for the SL-IG site and the short measurement period for the EF-DA site it is also not possible to draw any conclusions regarding the impact of land management (tillage vs. grassland) on the magnitude of the emission factor. To derive a country-specific emission factor, a more balanced measurement approach across the year would be recommended or, if resources allow, semi-continuous measurements using a technique such as Eddy Covariance for example. However, the data from these two sites lend support to the continued use of the IPCC 2006 Guidelines default emission factor of 8 kg N₂O-N ha⁻¹ for this emission source in the UK GHG inventory.

Emission factors relating to fertiliser N amendments to lowland peat soils were derived as 0.82 and 1.7% of applied N for the EF-DA and SL-IG sites, respectively. The higher value for SL-IG was most likely because of the higher soil WFPS at that site. The value for EF-DA was derived from a single fertiliser application in May and a subsequent 40-d measurement period which suggested that emissions from fertiliser amendments had peaked and returned to background levels by then. While IPCC 2000 Good Practice Guidance recommends that emission measurements should be made over a full year for the derivation of emission factors, a number of UK studies have reported that the main emission occurs during the first 1-3 months after N application (e.g. Bell et al., 2015; Misselbrook et al., 2014). However, the possibility cannot be ruled out that further emissions due to the N application may have occurred after the measurements were stopped at the EF-DA site in particular. Also, depending on crop rotation and management, there are often several N applications to the same field during the year. The emission factor derived from the measurements at the EF-DA site cannot therefore be considered as representative of an annual emission factor for the site/crop management, but do provide useful information to be combined with other data relating to the factors influencing N₂O emissions following fertiliser N application. The emission factor of 1.7% of applied N from the SL-IG site can be considered as representative of the management for the site/year. It is greater than the IPCC 2006 Guidelines default value of 1%, although within the uncertainty range given for that default emission factor of 0.3-3%. A single site/year value is insufficient data from which to develop an emission factor specific to lowland peat soils; however, this value is at the upper end of the range of emission factors derived from experiments within Defra AC0116 and associated projects. The UK has developed country-specific emission factors for fertiliser applications to land based on these experimental data as 0.48 and 0.60% for urea-based fertilisers and 1.29 and 0.79% for other N fertiliser types applied to grassland and arable land, respectively (UK National Inventory Report, Annex, 2016). The data from the present study can be usefully combined with this growing UK dataset to explore potential relationships between annual emission factors and soil and climatic factors, but further measurements from peat soils would be useful in this respect.

The use of the nitrification inhibitor DCD had contrasting effects. At MM-DA emissions were too low for a realistic assessment, but there was a suggestion that DCD resulted in lower daily N₂O fluxes. At EF-DA the DCD resulted in a non-significant 50% reduction in cumulative N₂O emission whereas at SL-IG there was no effective reduction in emission. Other UK studies have shown a range in the effectiveness of DCD across different (mineral) soil types, N sources and land management (Barneze et al., 2015; Bell et al., 2015a; Bell et al., 2015b; Misselbrook et al., 2014) and while there is good evidence from laboratory studies of the influence of temperature and soil texture (McGeough et al., 2016) it is still difficult to predict effectiveness when used under field conditions. The effectiveness of DCD will be greater when used with urea fertiliser, where all of the N will go through the nitrification process than with AN or ASN where half of the N content is already present as nitrate. Misselbrook et al. (2014) reported a mean reduction efficiency from UK field experiments across cropping and grassland on mineral soils of 69 and 39% for urea and AN, respectively. Further measurements are required to develop a mechanistic understanding, but the results of this study demonstrate that there is potential for DCD to mitigate N₂O emissions from N amendments to peat soils.

4. SYNTHESIS OF RESULTS

4.1. Eddy covariance gas fluxes

The project has supported the operation of eddy covariance (EC) flux towers at seven of the study sites. These are considered to provide the most reliable estimates of CO₂ fluxes in the project, and are therefore considered separately here.

In total, the flux towers generated more than seventeen site years of EC data within the project period (Figures 4.1 and 4.2). Three sites began to generate data during 2012, and all sites produced data to the end of 2015. This is in part due to successful operation of the flux towers in the network over the last year as well as more effective collation of available data. All available EC data (to the end of 2015) have been processed, quality checked and used to calculate net ecosystem CO₂ exchange (NEE), gross primary production (GPP) and total ecosystem respiration (ER) as described in Section 1.3.3. The following section describes the seasonal change in accumulated daily NEE (i.e. the net land-atmosphere CO₂ balance) and its component fluxes (GPP and ER) and, where possible, provides estimates of annual and cumulative CO₂ budgets.

Seasonal changes in daily NEE, GPP and ER (in mass units of g CO₂-C m⁻² day⁻¹) are shown in Figure 4.1 for all measurement periods at all seven sites. Estimates of daily GPP are shown using negative values to more effectively illustrate the opposing influences of the assimilatory (GPP) and respiratory (ER) fluxes on the net ecosystem CO₂ exchange (NEE). As with previous plots, the sign convention for NEE is positive when the site is acting as a net source for atmospheric CO₂, and negative when it is acting as a net CO₂ sink. Cumulative estimates of GPP, ER and NEE, as well as any lateral inputs and outputs of C in biomass (e.g. in harvest) are summarised in Table 4.1 for the different time periods. Figure 4.2 shows accumulated NEE for each full calendar year. At EF-LN, where data were obtained from the second half of 2013 and the first half of 2014, these data have been combined to provide a single ‘flux year’.

Table 4.1. Annual estimates of net ecosystem CO₂ exchange for flux tower sites. Lateral fluxes of C in imported and harvested biomass have been included where significant. GPP is gross primary production, ER is ecosystem respiration, NEE is net ecosystem exchange, C imports are in the form of seeds and peat plugs, C exports are harvested biomass, NEP is net ecosystem production (NEE and the net balance between imported and exported C). Sites are grouped by vegetation type. All data are expressed in g C m⁻² yr⁻¹. For information on derivation of uncertainty ranges see Section 1.3.3.

Site	Time period	GPP	ER	NEE	C import	C export	NEP
EF-LN	07/2013 to 07/2014	1667	1595	-72 ± 112	--	--	-72 ± 112
	2015	1511	1328	-183 ± 96	--	--	-183 ± 98
AF-HN	2014	1211	1035	-176 ± 84	--	--	-176 ± 84
	2015	1166	1027	-139 ± 76	--	--	-139 ± 76
AF-LN	2015	1085	998	-87 ± 69	--	--	-87 ± 69
EF-EG	2013	1257	1684	157 ± 111	--	--	157 ± 111
	2014	2016	2099	83 ± 107	--	--	83 ± 107
	2015	1725	1855	130 ± 91	--	--	130 ± 91
SL-EG	2013	2437	2551	114 ± 147	--	203	317 ± 147
	2014	2800	2822	22 ± 179	--	203	225 ± 179
	2015	2588	2737	149 ± 188	--	203	351 ± 188
EF-DA	2013	826	1428	602 ± 82	--	127	730 ± 82
	2014	863	1596	734 ± 79	46	36	724 ± 79
	2015	966	1702	736 ± 84	49	99	783 ± 84
EF-SA	2013	2053	2033	-20 ± 132	-	635	615 ± 132
	2014	1218	1896	678 ± 128	104	90	664 ± 128
	2015	1843	1932	90 ± 140	-	597	687 ± 140

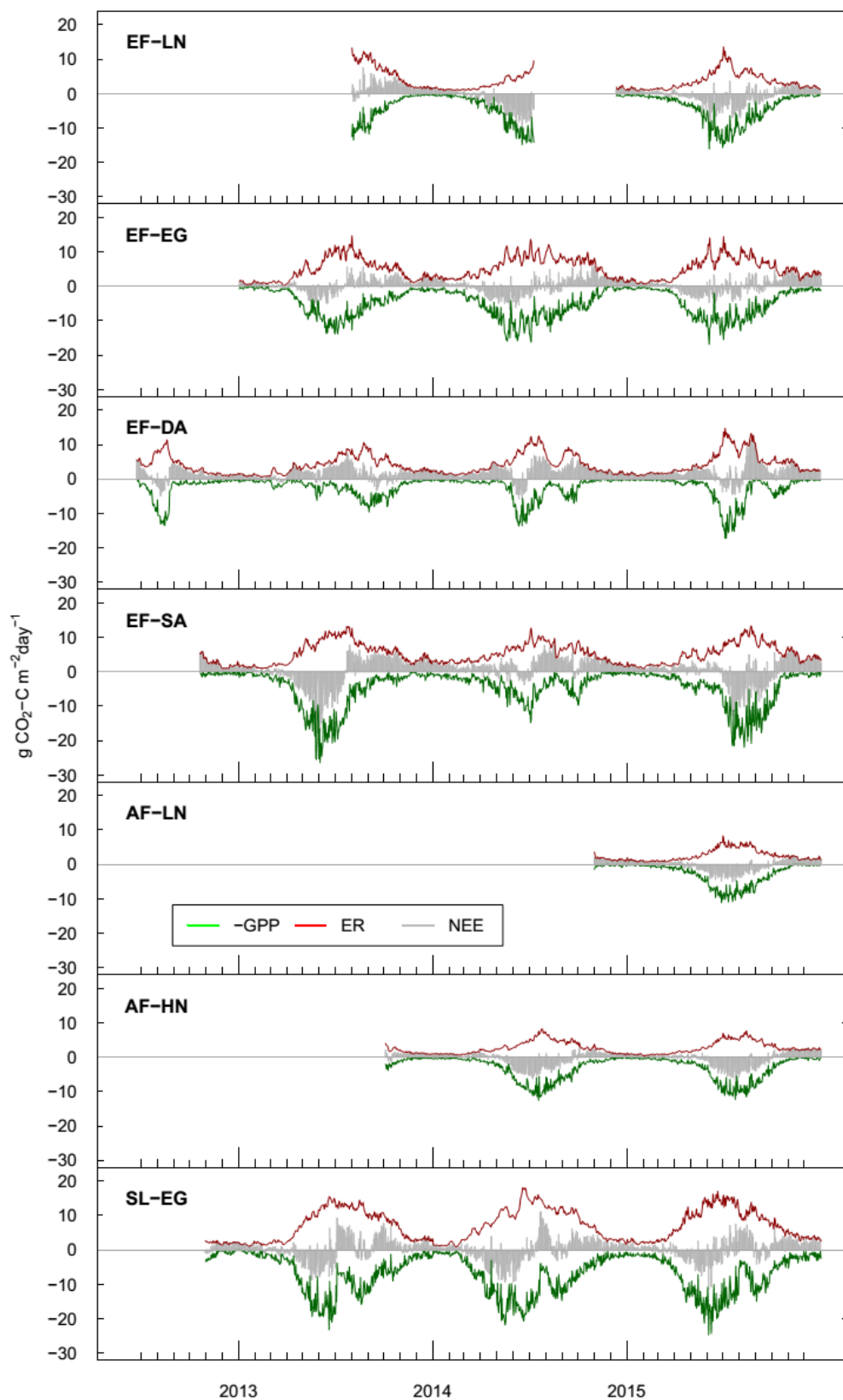


Figure 4.1. Daily carbon dioxide budgets for eddy covariance flux measurement sites. Figures show estimates of (gap-filled) net ecosystem CO₂ exchange and estimates of gross primary production (GPP) and total ecosystem respiration (ER). Note that all sites are shown on the same y-axis scale.

4.1.1. Semi-natural fens

Cumulative CO₂ budgets estimated for the three sites with permanent (semi-natural) fen vegetation (EF-LN, AF-LN and AF-HN) suggest these sites are all functioning as strong net sinks for CO₂, with the full uncertainty range for cumulative NEE falling below zero in all sites and years except for the (merged) 2013-14 year at EF-LN (Figure 4.2, Table 4.1). As noted in Section 2.1.1.3, this period included a period of severe water-table drawdown, during the peak of which (in July 2013) the site switched from a normal pattern of summer net CO₂ uptake to net CO₂ emission. Although GPP was higher in the drought-affected 2013-14 measurement year than in 2015, ER was higher by a greater amount (Table 4.1), presumably in part because exposure of a considerable depth (around 70 cm, Figure 2.5) of peat to aerobic conditions caused an increase in peat decomposition rates. As a result, NEE for the 2013-14 measurement year (-55 g C m⁻² yr⁻¹) was much lower than NEE for the more hydrologically stable 2015 (-183 g C m⁻² yr⁻¹). Taking a simple mean of the two one-year periods would suggest a mean NEE for EF-LN of -119 g C m⁻² yr⁻¹. However, water-table data from the project period suggests that the summer of 2013 was hydrologically extreme, and therefore unlikely to be representative of long-term mean uptake. On the other hand, the hydrologically isolated condition of Wicken Fen, which is now elevated above the surrounding subsidence-affected landscape, protected by bunds and supplied with water from adjacent rivers, suggest that it is more vulnerable to water-table drawdown than sites that remain hydrologically connected to the surrounding landscape, including the AF and NB sites. Data from the nearby Bakers Fen suggest that severe water-table drawdown events occurred three times from 2007 to 2015, i.e. one year in three. Weighting the data from EF-LN on this basis suggests a mean NEE of -140 g C m⁻² yr⁻¹. In this respect it is also worth noting that previous (incomplete) years of eddy covariance measurements at EF-LN in 2009 and 2010 (both relatively dry years) showed evidence of net CO₂ release in 2009 and only a weak sink in 2010 (R. Morrison, unpublished data) further supporting the conclusion that the measured CO₂ sink in 2015 was larger than average.

Of the two AF fen sites, AF-HN functioned as a strong CO₂ sink in both full measurement years (-176 and -139 g C m⁻² yr⁻¹ in 2014 and 2015 respectively), giving a site mean NEE of -158 g C m⁻² yr⁻¹. This is very similar to the value derived for EF-LN above, although it is worth noting that average GPP for the two full years at AF-HN was approximately 25% lower than at EF-LN, and that ER was lower by a similar amount. This may be due to generally higher temperatures and solar radiation levels in East Anglia compared to Anglesey (Table 2.2). It is possible that productivity at EF-LN is enhanced by inputs of nutrient-enriched river water to the fen, although clear differences in peat nutrient status between the sites were not evident (Table 2.2). Given that the vegetation characteristics of AF-HN and EF-LN (tall *Phragmites australis* and *Cladium mariscus*) were rather similar, whereas the vegetation at AF-LN (short fen species) was different, we infer that the vegetation community present may be a better predictor of its CO₂ balance than nutrient status, which was used *a priori* to classify the natural fen sites.

Compared to the tall fen AF-HN, the short fen AF-LN site functioned as a weaker CO₂ sink (-87 g C m⁻² yr⁻¹) during the common 2015 measurement year. Given the very strong similarities in the temporal pattern of both gross and net CO₂ fluxes during this period (Figure 4.1) it seems reasonable to conclude that this represents a genuine contrast in the long-term rates at which the two adjacent sites sequester CO₂. Since the 2015 ER values for the two sites were very similar (Table 4.1) differences in CO₂ sequestration appear to be largely attributable to the greater productivity of the tall fen species at AF-HN (the presence of which presumably reflects the nutrient status of the site) compared to the slower-growing short fen species at AF-LN. This difference arose despite AF-LN acting as a net sink for a longer period (161 days in total in 2015) compared to AF-HN (150 days).

Given that the area within the measurement footprint of all three fen flux sites was subject to minimal management during the study period (limited to low-intensity conservation grazing by ponies at the Anglesey Fen site), biomass offtake was assumed to be negligible. Therefore the mean NEE values obtained for each site can be considered to equate to the net ecosystem productivity (NEP) of the site. Over longer periods, however, it is reasonable to assume that biomass removal will occur (at least at the EF-LN site, which is subject to periodic reed cutting) which would have the effect of removing some of the carbon sequestered over the preceding period. On that basis, the C balance estimates obtained for these sites may somewhat over-estimate the true long-term rate of CO₂ sequestration by conservation-managed fens.

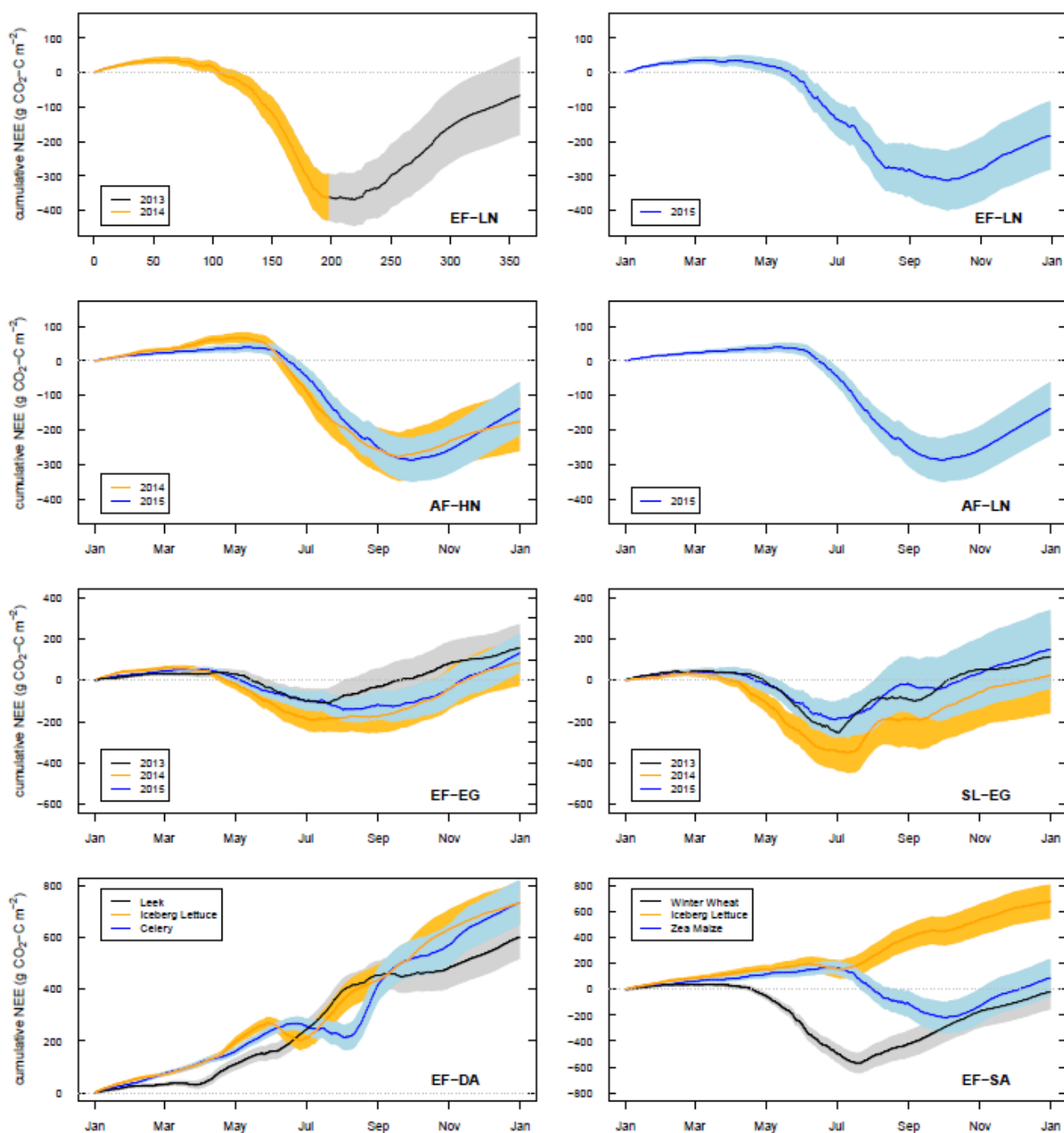


Figure 4.2. Cumulative annual CO_2 fluxes for eddy covariance flux measurement sites for full calendar years. Note the different y-axis scaling on different subplots. Sites have been grouped in rows by vegetation type. Note that the upper left panel shows cumulative CO_2 at EF-LN from 16th July 2013 and 9th July 2014. The 2014 and 2013 periods have been reversed in the figure in order to show the seasonal development of cumulative fluxes from January to December, for consistency with the other plots. Consequently the plot does now show a ‘true’ year (and has therefore been plotted separately from the continuous 2015 measurement period, upper right panel) although the overall cumulative flux is correct for the July 2013 – July 2014 period.

4.1.2. Managed grasslands

Daily and cumulative CO₂ budgets at the two managed grasslands with flux towers, EF-EG and SL-EG, reveal large within-year, between-year and between-site differences in CO₂ fluxes (Figures 4.1 and 4.2, Table 4.1). Three years of data from these sites indicate that both are functioning as net (*in situ*, i.e. before accounting for indirect fluxes) sources of CO₂, although uncertainty ranges extend either side of zero in all years at SL-EG, and one year at EF-EG, so the possibility that SL-EG in particular is acting as a slight *in situ* CO₂ sink cannot be ruled out. The deep peat, hay-cropped grassland SL-EG was the most productive flux tower site in all three years of operation, with an average GPP of 2608 g C m⁻² yr⁻¹ (Figure 4.1, Table 4.1). The shallow peat, conservation-grazed EF-EG was far less productive (mean GPP 1666 g C m⁻² yr⁻¹ for the same period), although it had a correspondingly lower mean ecosystem respiration rate (1869 g C m⁻² yr⁻¹ vs 2703 g C m⁻² yr⁻¹ at SL-EG). Both sites acted as net CO₂ sinks in the early part of the growing season (Figure 4.1) but transitioned to net sources later in the summer, notably following the mid-summer hay harvest at SL-EG. The warm summer of 2014 was associated with the highest annual GPP and smallest net *in situ* CO₂ emission at both sites, suggesting that between year variations in productivity (rather than respiration) determine the magnitude of direct CO₂ emissions. Perhaps surprisingly, there were no indications that severe flooding of the Somerset Levels in the winter of 2013-14, which led to a severe and prolonged period of inundation at SL-EG, had an adverse impact on the productivity of the C balance of the site; in fact, higher rates of CO₂ uptake were observed during the warm spring and early summer period of 2014, directly after the flooding, than in the corresponding periods of either 2013 or 2015.

Over the three years of EC data from the two grassland sites, EF-EG had a mean NEE of 123 g C m⁻² yr⁻¹, whereas SL-EG had a mean NEE of 95 g C m⁻² yr⁻¹. At EF-EG, where grazing-related C offtake is believed to be negligible, the NEE is probably a reasonable indicator of the net ecosystem productivity (NEP) of the site. At SL-EG, annual biomass offtake in the annual summer hay crop was estimated during a previous study (Lloyd, 2006) to be 203 g C m⁻² yr⁻¹, with an additional 25 g C m⁻² yr⁻² associated with cattle live weight gain. We collected additional data on hay offtake for 2016, which gave a slightly lower value of 153 g m⁻² yr⁻¹. Combining these data (assuming the same cattle offtake per year) gave a mean biomass removal flux of 203 g C m⁻² yr⁻¹, which was applied to all three years of the study. This gave annual NEP values of 225 to 352 g C m⁻² yr⁻¹, with a mean of 298 g C m⁻² yr⁻¹ over the three year (2013-2015) time period. In the earlier flux study at this site, Lloyd (2006) reported an NEE (for 2002) of -169 g C m⁻² yr⁻¹, much lower than we observed in this study, giving a smaller (but still positive) NEP of +59 g C m⁻² yr⁻¹. For EF-EG, previous flux tower measurements made at the site in 2010 (Morrison et al., 2013) gave an estimated NEE of 21 g C m⁻² yr⁻¹; again this is lower than in any of the years of the current study.

4.1.3. Arable farmland

The two arable sites, EF-DA and EF-SA, are the most dynamic in terms of their daily CO₂ fluxes, with the least consistent seasonal patterns (Figure 4.1). This is due to the major effects of agricultural land-management activities on the peat and vegetation at these locations. The deep peat EF-DA site was a net (daily) source for CO₂ during most of the measurement period, with only brief periods of small net CO₂ uptake when crops were growing in each year. In contrast to the sites with semi-natural and permanent vegetation cover, where net CO₂ uptake typically occurred for 100-150 days per year, net CO₂ uptake was only observed at EF-DA for an average of 38 days per year between 2013 and 2015. As a consequence of the short crop-growing periods, and the relatively low biomass yield of the horticultural crops grown in all study years (see below), the mean annual GPP of EF-DA was just 885 g C m⁻² yr⁻¹, making it the least productive of all the flux tower sites (the next lowest, AF-LN, had a GPP for 2015 of 1085 g C m⁻² yr⁻¹).

Seasonal changes in GPP at this site are related both to cropping periods, and to the development of secondary weed communities during fallow periods after crops have been harvested. The highest respiration rates, and largest net CO₂ emissions, are associated with biomass management events (e.g. planting, ploughing, disking, weed spraying), when disturbance of the peat causes high heterotrophic respiration rates. The highest net losses for this location were measured in August and September 2014 (Figure 4.1), when the field was bare of vegetation and air and soil temperatures were high. However, slow but sustained CO₂ emission also occurred during all winter periods when GPP was near-zero, but respiration (associated

with ongoing peat decomposition) remained around 1-2 g C m⁻² day⁻¹. Comparing NEE for each winter (October-March) period with the subsequent summer (April to September) period, net emissions during the winter period were 123 g C m⁻² in the winter of 2012-13 (23% of total gaseous CO₂ emissions for that 12 month period), 257 g C m⁻² in winter 2013-14 (38% of emissions for that year) and 320 g C m⁻² in winter 2014-15 (41% of emissions). These results highlight the substantial contribution of the winter fallow period to CO₂ emissions (and thus carbon loss and subsidence) at EF-DA, and suggest that mitigation measures that helped to reduce CO₂ respiration rates outside of the growing season could provide a significant mitigation benefit.

As noted in Section 2.1.3.3, variations in the timing and magnitude of CO₂ emissions between years were associated with the different crops grown on the site (leek, lettuce and celery) but overall net emissions for the three full measurement years (2013-2015) were fairly stable, ranging from 602 to 736 g C m⁻² yr⁻¹ (mean 691 g C m⁻² yr⁻¹).

The shallow peat arable site, EF-SA, had the largest between-year differences in daily CO₂ exchange (Figure 4.1). The site acted as a strong net sink for atmospheric CO₂ during the growth of the wheat crop in spring and early summer in 2013 (net CO₂ uptake on 137 days) and between July and October during the maize production period in 2015 (net CO₂ uptake on 105 days). By contrast, the number of days with net CO₂ uptake was much lower for the two lettuce crops grown in 2014 (53 days of net CO₂ uptake in total) and peak GPP and CO₂ uptake rates were far lower than in either the wheat or maize crop years (Figure 4.1). As for EF-DA, the largest net emissions of CO₂ at EF-SA were associated with days following land management events (primarily harvesting at this site) and low but sustained CO₂ emission was observed during the fallow winter period, when GPP was minimal. The cumulative annual (*in situ*) NEE at EF-SA ranged from a small net sink to a small net source (Figure 4.2). The best estimate of annual NEE was -20 g C m⁻² yr⁻¹ for 2013, +678 g C m⁻² yr⁻¹ for 2014 and +90 g C m⁻² yr⁻¹ for 2015 (Table 4.1), giving a mean for the three years of +249 g C m⁻² yr⁻¹. Flux partitioning produced a mean GPP estimate of 1705 g C m⁻² yr⁻¹ (much higher than at EF-DA), with much higher values observed in wheat and maize crop years than in the lettuce crop year (Table 4.1). Mean ecosystem respiration for the full period was similarly higher than EF-DA, at 1954 g C m⁻² yr⁻¹.

As for the SL-EG grassland site, full assessment of the CO₂-C budget (or net ecosystem production – NEP) for the arable sites requires consideration of lateral biomass C fluxes. Lateral fluxes at these sites include C imports in seeds, seedlings and peat plugs, and exports in harvested crop products and other biomass (e.g. straw). At EF-DA, in years with lettuce crops, C imports in the form of peat plugs and lettuce seedlings (58 g C m⁻²) actually *exceeded* exports in harvested biomass, by 14 g C m⁻² in 2012 and 10 g C m⁻² in 2014 (Table 4.1). Celery crop production at the site in 2015 resulted in net biomass export of around 50 g C m⁻² in 2015, whilst the leek crop in 2013 (overlooking C inputs in seeds) generated a biomass export of 127 g C m⁻². When these lateral C fluxes are accounted for, estimates of NEP for EF-DA (i.e. NEP) were estimated at a fairly uniform 729 g C m⁻² yr⁻¹ in 2013, 724 g C m⁻² yr⁻¹ in 2014, and 783 g C m⁻² yr⁻¹ in 2015 (mean 746 g C m⁻² yr⁻¹). Over the full 3½ years of EC measurements, we estimate that EF-DA has lost a total of 2.3 kg C m⁻² (equivalent to 23 t ha⁻¹ of peat C loss and 84.5 t ha⁻¹ of CO₂ emissions), before accounting for any aquatic or wind-driven carbon losses.

For EF-SA, direct estimates of biomass inputs and outputs were not obtained for the first two years. In 2013, the lateral export of C in wheat and straw was estimated at 400 g C m⁻² and 235 g C m⁻², respectively, using data for the East Anglian region collated by ADAS on behalf of Defra (Sarah Wynn, personal communication). For 2014, we assumed that both lettuce crops produced during the growing season were associated with the same net input of biomass as measured at EF-DA, i.e. a total net input of 24 g C m⁻². For the maize crop in 2015, the lateral biomass flux was estimated at 597 g C m⁻² on the basis of destructive biomass sampling (n = 3 plants), from which dry weights were measured and converted to a carbon flux assuming a 50% carbon content and a measured density of 10 maize plants per m². We assumed that only the cobs were removed from the site, with stems being left on the field. These calculations gave NEP estimates of 615 ± 132 g C m⁻² yr⁻¹ for 2013, 654 ± 128 g C m⁻² yr⁻¹ for 2014, and 686 ± 140 g C m⁻² yr⁻¹ during 2015 (three-year mean 652 g C m⁻² yr⁻¹).

The results from EF-DA and EF-SA highlight the dominant influence of farming operations on the timing of CO₂ emissions from organic soils under arable cultivation. Any activity which disturbs the soil can be expected to accelerate CO₂ loss, and extended periods of bare peat exposure, accompanied by continued

drainage, when fields are left fallow lead to slow but steady CO₂ emissions outside of the growing season, which make a significant contribution to annual losses. This appears to be particularly pronounced for horticultural crops, which are only on the field for a relatively small part of the year. Even when these crops are growing, their CO₂ uptake is low, and continued exposure of bare peat between plants exposes the peat to warming by solar radiation and therefore to accelerated decomposition. Our data suggest that cereal crops such as wheat and maize, which produce a more complete canopy cover and have a higher rate of productivity, generate much higher rates of CO₂ uptake during the growth period. However, this C uptake appears to be largely accumulated in above-ground biomass, with the result that removal of this biomass during crop harvest removes very large amounts of C. Once these fluxes were taken into account, estimated net CO₂ losses in 2013 (wheat) and 2015 (maize) were almost identical to those recorded in 2014 under lettuce (Figure 4.3). Whilst the exact amount of carbon removed in harvested biomass is uncertain, the evidence of these results is that – despite large observed differences in NEE between years – the overall impact of different crop types on net ecosystem C balance, and thus overall CO₂ emissions, may be comparatively small.

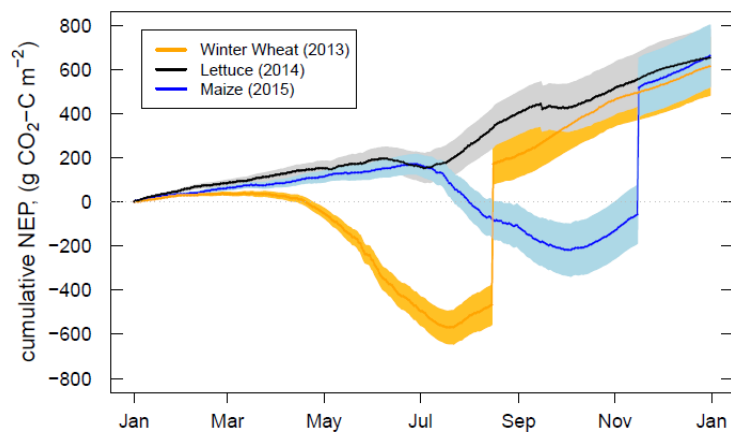


Figure 4.3. Cumulative annual CO₂ fluxes at EF-SA (Redmere Farm) taking into account the estimated biomass removal associated with wheat and maize crop harvests in 2013 and 2015 respectively.

Overall, once the effect of crop biomass removal was taken into account, we observed only a small, albeit fairly consistent, difference in the rate of CO₂ loss between the deep peat EF-DA and shallow peat EF-SA sites (mean annual NEP 746 and 678 g C m⁻² yr⁻¹ respectively). This suggests that rates of C loss from the shallower, denser and more mineral-enriched peat at EF-SA have not slowed relative to the deeper peat EF-DA, although it should be emphasised that the ‘shallow’ peat at EF-SA remains considerably deeper than the thin ‘wasted’ peats (represented in this project by EF-EG), which cover large parts of East Anglia and which are largely still under arable cultivation. Further measurements of CO₂ balance in arable ‘wasted’ peats are needed in order to establish the point at which CO₂ emissions decrease relative to those from deep peat.

4.2. Static chamber gas fluxes

As noted above, eddy covariance flux towers were considered to provide the most reliable estimates of CO₂ flux, and were used in preference wherever possible. However, manual chambers were the only method available for quantifying CH₄ fluxes, and also provided CO₂ flux estimates for a larger number of sites. Best estimates of CO₂ and CH₄ fluxes derived from static chamber measurements are shown in Table 4.2. All GPP and ER estimates were derived from hourly modelled flux values as described in the previous section, and NEE was derived as a balance of these gross fluxes. Methane fluxes to/from the land surface was based on interpolated measurements in most cases, but were modelled at a subset of sites. Ditch CH₄ fluxes were interpolated from measurements in all cases.

Table 4.2. Annual estimates of gross and net CO₂ fluxes, and terrestrial and ditch CH₄ fluxes, derived from static chamber measurements. All fluxes expressed in g C m⁻² yr⁻¹.

Site	Time period	GPP	ER	NEE	CH ₄ (land)	CH ₄ (ditch) Per m ² ditch	CH ₄ (ditch) Per m ² site
EF-LN	2013 to 2015	544	391	-152	8.8	125	1.8
EF-EG	2013 to 2015	1135	891	-244	0.0	9	0.2
EF-DA	2013 to 2015	ND	1299	ND	-0.1	33	0.5
EF-SA	2013 to 2015	ND	1589	ND	-0.3	1	0.0
MM-RW	2013 to 2014	584	544	-41	32.8	-	-
MM-EX	2014 to 2014	0	138	138	0.2	-	-
MM-DA	2015 to 2014	ND	1151	ND	0.1	-	-
AF-LN	2013 to 2015	657	480	-177	15.7	12	0.1
AF-HN	2013 to 2015	1153	902	-251	14.1	23	0.2
SL-EG	2013 to 2015	1253	2040	787	-0.2	112	3.0
SL-IG	2013 to 2015	1669	2717	1048	-0.1	37	0.6
NB-LN	06/2012 to 08/2013	712	722	10	15.1	289	6.1
NB-HN	06/2012 to 08/2013	1334	901	-434	13.0	12	2.5
TM-RW	06/2013 to 09/2014	1129	996	-132	11.9	-	-
TM-EX	06/2013 to 09/2014	0	143	143	0.0	-	-

4.2.1. CO₂ fluxes

Annual estimates of GPP based on static chamber measurements ranges from zero (bare peat extraction sites) to 1669 g C m⁻² yr⁻¹ (SL-IG). At the three arable sites it was not possible to construct annual models of GPP from chamber data due to the over-riding influence of farming operations on the timing and magnitude of crop growth. These issues also affect the three grassland sites to varying degrees, due to the influence of hay cutting (as shown in the flux tower data for SL-EG) and grazing. Therefore, the GPP models are considered to be most reliable for sites under semi-natural vegetation with low management intensity, where plant growth rates are primarily controlled by environmental conditions. A comparison of GPP estimates derived from the two flux measurements methods at sites with flux towers (Figure 4.4a) shows that chamber-based methods gave lower estimates of GPP in four out of five cases, with the greatest discrepancy associated (as expected) with the managed grassland SL-EG. For ecosystem respiration (Figure 4.4b) there was also a tendency for the chamber-based method to give lower annual fluxes than the eddy covariance method, although the correlation between the two sets of flux estimates was fairly high ($R^2 = 0.69$). If we assume that the eddy covariance method provides the best estimate of CO₂ fluxes when available (whilst also acknowledging that this method also has associated limitations and uncertainties) it would appear that there is a general tendency for models based on chamber measurements to underestimate gross CO₂ fluxes. This could be due to the difficulty of modelling true rates of gross CO₂ fluxes (particularly photosynthesis) based on a limited number of explanatory variables, which make it difficult to accurately predict peak rates. It is also possible that there is a 'chamber effect' on plant growth, whereby

disturbance of roots and water movement by the installed collars, and of the site as a whole during field measurements, may have reduced primary productivity and associated autotrophic respiration.

For NEE, however, the apparent biases in the two estimates of gross fluxes appeared to cancel out for the three natural fen sites (Figure 4.4c), giving a fairly close comparison between chamber and eddy covariance based annual fluxes. It is thus possible that - despite the possible influence of field measurements on plant growth – the impact of these activities on the overall rate of plant litter input to the peat, and heterotrophic respiration from the peat, were less affected. For the two managed grassland sites, on the other hand, there were large discrepancies between NEE derived from the two methods. For these sites, and for the reasons discussed above, and we conclude that the direct application of a chamber based modelling approach is unlikely to provide a robust estimate of annual CO₂ exchange. The final derivation of CO₂ balance estimates for these and other sites is described below in Section 4.4.

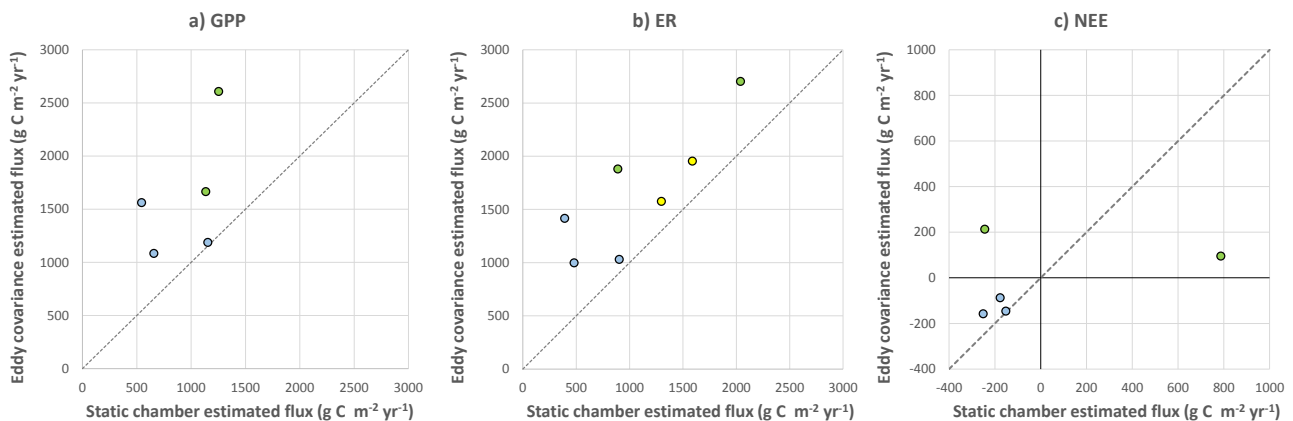


Figure 4.4. Comparison of gross and net CO₂ fluxes per site estimated by eddy covariance and static chamber methods. Sites are colour-coded as: blue = conservation-managed fen; green = grassland; yellow = arable. Note that GPP (and therefore NEE) could not be estimated directly from static chamber data at arable sites due to the over-riding influence of agricultural activities on rates of plant CO₂ uptake.

4.2.2. CH₄ fluxes

Terrestrial fluxes of CH₄ ranged from very small net uptake (minimum -0.3 g C m⁻² yr⁻¹) to very high rates of emission (maximum 32.8 g C m⁻² yr⁻¹). There were clear relationships between land-use and associated drainage status, with all of the arable, grassland and extraction sites showing negligible fluxes (range -0.3 to +0.2 g C m⁻² yr⁻¹), suggesting that these drained land-use categories do not emit CH₄ directly from the peat surface. All semi-natural fen sites, on the other hand, had substantial CH₄ emissions (range 8.8 to 15.7 g C m⁻² yr⁻¹). These emission rates are of sufficient magnitude to exert a large influence on the overall GHG balance of the ecosystem. The highest rate of ‘terrestrial’ CH₄ emission was observed at the permanently inundated re-wetted raised bog, MM-RW. The other re-wetted raised bog, TM-RW, is discontinuously inundated, and CH₄ emissions were lower (11.9 g C m⁻² yr⁻¹) and more similar to the conservation-managed fens. Controls on terrestrial CH₄ emissions are examined in more detail in Section 4.5.

Emissions of CH₄ from drainage ditches were highly variable between sites and over time, and even between ditches at the same site (see e.g. EF-DA). This high degree of small-scale spatial heterogeneity was further demonstrated by high-resolution campaign measurements made at the East Anglian sites (Peacock et al., 2017), and is also evidence across the range of published studies that were used to derive the IPCC Tier 1 EF for this component of the GHG balance (Evans et al., 2016), suggesting that more detailed measurements are needed to accurately quantify this flux. However, the data obtained from the project clearly highlight the potential importance of ditches as sources of CH₄ emissions across a range of land-use types, with fluxes per unit ditch area ranging from 1 to 289 g C m⁻² yr⁻¹ at sites where measurements were made. The significance of these fluxes depends also on the extent of ditches (and other open water bodies) within the peatland landscape which was typically in the range 1 to 3% at most of the study sites, but may be higher in locations with larger or denser ditch networks (e.g. extraction sites, some wet meadow systems) or in areas where larger areas of open water either occur naturally or have been created (e.g. to support bird populations) in

conservation-managed areas. A notable example of this is the NB-HN site, Strumpshaw Fen, where 21% of the fen area comprises shallow open water bodies. Measurements of CH₄ emissions from these larger water bodies was not undertaken within the project, and thus remains a research gap. The role of infrequent but potentially large ebullition (bubble) emissions of CH₄ from areas of standing water or surface inundation was also not quantified, and represents a potential missing term in our emissions estimates. Overall, ditch CH₄ fluxes (where measured) contributed from 1% to 100% of all CH₄ emissions, with a maximum area-adjusted rate of 6.1 g C m⁻² yr⁻¹ at NB-LN. In drained arable and grassland sites, ditches were the only important source of CH₄ emissions, and (although data were too limited to draw firm conclusions) appeared to be highest in deep, stagnant ditches, or those potentially receiving labile organic matter inputs from animal wastes. If this is correct, there may be potential to mitigate ditch CH₄ emissions through changes in water management or controls on organic matter/and or nutrient inputs to the ditch network. Clearly, these results demonstrate that, despite the absence of CH₄ emissions directly from the peat surface, it cannot be assumed that agricultural peatland landscapes cease to act as CH₄ sources following drainage.

4.3. Water and aquatic carbon fluxes

4.3.1. Water fluxes

Estimated annual water fluxes for sites where full hydrological budgets were constructed are shown in Table 4.3. The hydrology of the study sites was highly variable, reflecting both intrinsic climate gradients (particularly declining annual rainfall from west to east) and the high degree to which the natural hydrology of most sites has been altered through management. Although fen peats develop in topographic positions where they are supplied by lateral input of groundwater or river water, only one site, AF-LN, is thought to maintain a significant external water supply via natural runoff from the adjacent limestone hillslope. Water supply to most of the remaining fens has been modified by a combination of ditch networks, embankment of rivers, subsidence and water level management such as pumping. The four East Anglian Fen sites are thought to have small external water inputs via pumping of water onto the site (EF-LN) or gravity fed water transfer via the ditch network (EF-EG, EF-DA and EF-SA). The three raised bog sites in the Manchester Mosses are also strongly modified through drainage (MM-EX, MM-DA) or bunding (MM-RW). Evapotranspiration (ET) also varied between sites, although to a far lesser extent than precipitation. High ET values were generally observed at sites with abundant water supply and/or continuous vegetation cover (AF-LN, AF-HN, MM-RW, SL-EG). Lower ET values were estimated for EF-EG, which is subject to severe seasonal drying, and at the EF arable sites with discontinuous vegetation cover. The lowest ET estimate was obtained for the bare peat MM-EX. These management-related differences in ET exerted a clear influence on rates of water loss from different sites within each study region, with arable and extraction sites having higher discharge rates (note that a separate water balance was not derived for SL-IG, so a comparison between intensive and extensive grassland could not be made).

Table 4.3. Average annual water fluxes (in mm) for all primary sites with full water balances.

Site	Site type	Precipitation	Other water input	Actual evapotranspiration	Discharge
EF-LN	Near-natural fen	624	21	483	162
EF-EG	Extensive grass on shallow fen peat	572	21	413	180
EF-DA	Arable on deep fen peat	627	30	420	237
EF-SA	Arable on shallow fen peat	617	42	460	199
MM-RW	Re-wetted raised bog	837	0	557	280
MM-EX	Extraction site	839	0	345	494
MM-DA	Arable on deep raised bog peat	875	0	526	349
AF-LN	Near-natural fen	1175	215	577	814
AF-HN	Near-natural fen	1137	0	472	665
SL-EG	Extensive grass on fen peat	852	0	504	347
SL-IG*	Extensive grass on fen peat	852	0	504	347

*Note water balance for SL-IG is assumed to be the same as that calculated for SL-EG.

4.3.2. Aquatic carbon fluxes

Arithmetic mean concentrations of measured aquatic carbon forms for all the sites where measurements were made are shown in Table 4.4, and the calculated mean annual fluxes in Table 4.5. A complete set of measurements were made for the EF, AF and SL sites. For the MM sites only DOC was measured routinely, although given the acidity of the raised bog peats in this area it is likely that DIC concentrations were low. For the NB sites a full set of concentrations measurements were made, but in the absence of full hydrological data from this 'secondary' study area it was only possible to estimate aquatic C fluxes based on mean discharge data from nearby gauging stations. For the TM secondary sites, data were insufficient to allow annual concentration or flux estimates to be made. Individual components of the aquatic carbon flux are discussed below, after which the overall contribution of aquatic carbon fluxes to carbon loss from lowland peats is evaluated.

Table 4.4. Mean annual concentrations of all measured aquatic carbon forms, by site.

Site	Site type	Water flux (mm yr ⁻¹)	Flow-adjusted mean concentrations (mg l ⁻¹)				
			DOC	POC	DIC	CO ₂ (g)	CH ₄ (g)
EF-LN	Near-natural fen	162	27.7	2.2	107.1	2.5	0.04
EF-EG	Extensive grassland, shallow fen peat	180	31.5	4.0	111.4	2.0	0.09
EF-DA	Arable, deep fen peat	240	32.9	1.4	59.2	6.1	0.27
EF-SA	Arable, shallow fen peat	199	32.9	11.1	19.7	6.2	0.02
MM-RW	Re-wetted raised bog	383	139.3	-	0*	1.6	5.65
MM-EX	Extraction site	555	120.0	-	0*	6.7	0.01
MM-DA	Arable, deep raised bog peat	349	100.6	-	0*	-	-
AF-LN	Near-natural fen	814	22.0	1.6	38.6	2.1	0.02
AF-HN	Near-natural fen	665	48.1	13.0	22.6	3.5	0.06
SL-EG	Extensive grassland, fen peat	350	29.2	0.8	44.1	5.0	0.07
SL-IG	Extensive grassland, fen peat	350	62.2	3.4	34.8	4.8	0.26
NB-LN	Near-natural fen	214	31.5	-	40.1	3.4	0.20
NB-HN	Near-natural fen	200	20.6	-	72.8	6.0	0.40

*Flux not measured, but assumed to be zero due to low pH (this assumption may be invalid for MM-DA which had a higher pH, see Table 1.2)

4.3.2.1. Dissolved organic carbon

Concentrations of DOC were generally high, with all sites having mean concentrations greater than 20 mg l⁻¹. Fluxes of DOC were more variable than concentrations due to the wide variation in discharge from the different sites discussed above. As a result, DOC fluxes ranged from just 4 g C m⁻² yr⁻¹ up to 67 g C m⁻² yr⁻¹. The sites clearly fell into two groups according to peat type, with the three raised bog sites of the Manchester Mosses having exceptionally high mean DOC concentrations (> 100 mg l⁻¹) and fluxes (≥ 35 g C m⁻² yr⁻¹). Similarly high DOC concentrations were also observed in the limited number of samples collected at Thorne Moors, suggesting a consistent pattern of high DOC loss from all raised bogs, regardless of management. These fluxes compare to typical values ranging from 20-30 g C m⁻² yr⁻¹ for UK upland blanket bogs (e.g. Dawson et al., 2004; Billett et al., 2010; Dinsmore et al., 2010). Amongst the Manchester Mosses sites, mean DOC concentrations were highest at the re-wetted MM-RW, but DOC flux was 70% higher from the MM-EX extraction than at MM-RW due to the larger annual water flux. At the arable MM-DA site, the increase in water flux relative to MM-RW was smaller, and was partly offset by lower measured DOC concentrations, so the DOC flux was only 13% higher.

Concentrations of DOC at the fen sites were consistently lower (range 23 to 49 mg l⁻¹) and fluxes ranged from 4 to 37 g C m⁻² yr⁻¹. As might be expected, the magnitude of the annual DOC flux was correlated with annual water flux (R² = 0.58, excluding Norfolk Broads sites, for which water fluxes are uncertain), with all sites in the drier East Anglian Fens and Norfolk Broads study regions having fluxes < 8 g C m⁻² yr⁻¹, and all sites in the

wetter Anglesey Fens and Somerset Levels having fluxes $> 10 \text{ g C m}^{-2} \text{ yr}^{-1}$. With each region, however, we observed variations in DOC concentrations and fluxes that appeared to be consistent with the extent of management impact. In the East Anglian Fens, both concentrations and fluxes of DOC were lowest at the relatively undisturbed EF-LN (with similarly low values for the two Norfolk Broads sites, which may provide a better near-natural reference for this region). Concentrations of DOC at the arable and grassland sites in the East Anglian Fens were 14 to 19% higher than at EF-LN, and fluxes were 27 to 76% higher. These values are similar to the 'Tier 1' increase in DOC loss resulting from peat drainage derived from literature data for the IPCC Wetland Supplement (IPCC 2014; Evans et al., 2016). However, the very low baseline DOC flux from this region implies that DOC losses remain a minor contributor to total carbon loss from sites in this region, regardless of management. For the Somerset Levels, we did not have a true 'near-natural' baseline, however the two study sites do provide a good paired comparison between conservation-managed extensive grassland with high water-table management (SL-EG) and conventionally drained intensive grassland (SL-IG). Both mean DOC concentrations (62 versus 29 mg l^{-1}) and fluxes (22 versus 10 $\text{g C m}^{-2} \text{ yr}^{-1}$) are more than twice as high at SL-IG compared to SL-EG, suggesting a strong management effect. Similarly, at the Anglesey Fens DOC concentrations at the relatively tall-fen AF-HN site are more than double those at the short-fen AF-LN. Despite larger water fluxes from the latter, mean DOC fluxes from AF-HN are around 80% higher than at AF-LN. Strikingly, DOC losses at the two AF sites equate to an almost identical 20% of the mean NEE measured by the respective flux towers.

Table 4.5. Estimated annual aquatic carbon fluxes, and total aquatic GHG flux by site (see text for derivation of GHG flux)

Site	Site type	Annual fluxes ($\text{g C m}^{-2} \text{ yr}^{-1}$)					GHG flux ($\text{t CO}_{2\text{eq}}$ $\text{ha}^{-1} \text{ yr}^{-1}$)
		DOC	POC	DIC	CO_2 (g)	CH_4 (g)	
EF-LN	Near-natural fen	4.5	0.4	17.4	0.4	0.00	0.17
EF-EG	Extensive grassland on shallow fen peat	5.7	0.7	20.1	0.5	0.01	0.23
EF-DA	Arable on deep fen peat	7.9	0.3	14.2	2.2	0.12	0.39
EF-SA	Arable on shallow fen peat	6.6	2.2	3.9	2.1	0.02	0.36
MM-RW	Re-wetted raised bog	53.4	-	0*	1.8	1.55	1.74
MM-EX	Extraction site	66.6	-	0*	2.3	0.00	2.01
MM-DA	Arable on deep raised bog peat	35.1	-	0*	-	-	1.29
AF-LN	Near-natural fen	17.9	1.3	31.4	1.2	0.05	0.68
AF-HN	Near-natural fen	31.9	8.6	15.0	1.3	0.01	1.33
SL-EG	Extensive grassland on fen peat	10.2	0.3	15.4	0.8	0.01	0.38
SL-IG	Extensive grassland on fen peat	21.8	1.2	12.2	1.9	0.01	0.82
NB-LN†	Near-natural fen	6.7	-	8.6	0.7	0.04	0.26
NB-HN†	Near-natural fen	4.1	-	14.6	1.2	0.08	0.21

*DIC not determined at MM sites but assumed to be zero due to low pH of water, GHG flux based on measured fluxes only. †Indicative fluxes based on mean annual discharge of nearest EA gauging station and mean annual concentration. Dashes indicate sites where POC fluxes were not determined.

4.3.2.2. Particulate organic carbon

Fluxes of POC from the study sites were typically much lower than those of DOC or DIC. At all sites other than AF-HN, mean fluxes were $< 2.5 \text{ g C m}^{-2} \text{ yr}^{-1}$, and although slightly higher fluxes were observed at some of the more intensively managed sites (e.g. EF-SA, SL-IG) this pattern was not entirely consistent (e.g. EF-DA had low measured POC loss). The higher POC flux at AF-HN resulted from a fairly small number of high measured POC concentrations in winter samples, when water flows from the site were relatively high, but cannot be clearly attributed to any local management factors. Overall, it appears that aquatic POC fluxes from the lowland peat sites included in the study make only a very minor contribution to overall carbon losses. However, it is worth noting that the filters available for DOC and POC determination (material passing through a $0.45 \mu\text{m}$ filter for the 'dissolved' fraction, and material captures by a $0.7 \mu\text{m}$ for the 'particulate' fraction) both fail to capture an intermediate 'colloidal' fraction (0.45 to $0.7 \mu\text{m}$) which could represent an

additional carbon loss from the system. Given the episodic nature of erosional losses, we also cannot rule out the possibility that larger POC losses did occur (e.g. from bare arable peats) during intensive rain events that were not captured by the relatively low-frequency sampling programme. Furthermore, POC losses from the two extraction sites, MM-EX and TM-EX, were not measured, but are likely to have been more significant. Finally, we were unable to quantify wind-borne erosion of peat as part of the project, but visual observations at both the active extraction site MM-EX and the arable sites EF-DA and EF-SA strongly suggest that these losses may be large, particularly during dry periods.

4.3.2.3. Dissolved inorganic carbon

Losses of DIC were assumed to be negligible from raised bog peats, with the possible exception of the cultivated MM-DA where topsoil pH was higher (Table 1.2). At the fen sites, DIC fluxes were relatively stable with most sites falling between 10 and 20 g C m⁻² yr⁻¹. Higher values were observed at sites receiving large inputs of base-rich groundwater, either from their catchments (AF-LN) or via water transfers (EF-LN, EF-EG). The markedly lower value recorded at EF-SA may be attributable to the exposure of underlying sulphur-rich marine sediments beneath the shallow peat (so called 'acid sulphate soils') which release sulphuric acid via oxidation when drained. This process is believed to have caused a number of very low observed pH values in ditch water at this site.

It is important to note that most DIC in runoff from fens is likely to be derived from weathering of carbonate minerals or from weathering of siliceous minerals by dissolved CO₂ in rainwater. In either case, the carbon contained in this DIC does not derive from the peat itself, and is not therefore part of the carbon balance of the system. On the other hand, some DIC may be produced as a result of the interaction between CO₂ or organic acids produced from the peat and underlying minerals, in which case this DIC flux would contribute to carbon losses from the peat. Although some attempts have been made to partition DIC export from peatlands into mineral and respiration sources (e.g. Worrall et al., 2005), we do not have sufficient data to carry out a similar separation here. As a first approximation, it appears reasonable to conclude that there is little opportunity for DOC produced in the fen systems studied here to interact with mineral soils before entering the ditch network, and therefore that most if not all of the DIC exported from these sites derives from incoming groundwater or river water, and is thus independent of the peatland carbon cycle. Furthermore, it is unlikely that DIC transported out of (or through) the peatland will subsequently be converted to CO₂, because the high observed pH of drainage waters from these sites means that DIC in runoff will remain in a dissociated form, as bicarbonate (HCO₃⁻) or carbonate (CO₃²⁻). To the extent that this DIC is ultimately be transported to the ocean, and precipitated into carbonate sediments, it will not make a contribution to GHG emissions from the peatland.

4.3.2.4. Dissolved gases

Dissolved gases were measured in the fen sites. Concentrations of dissolved CO₂ ranged from 1.6 to 6.8 mg C l⁻¹. There was a clear tendency towards higher concentrations at more intensively managed, with the two EF arable sites, and the SL intensive grassland site, all having CO₂(g) concentrations at least two times higher than the semi-natural reference sites in the same regions. The tall fen AF-HN site also had higher CO₂(g) concentrations than the short fen AF-LN. Differences in fluxes were even more pronounced at the EF sites, with the two arable sites having five times higher CO₂ fluxes than the semi-natural reference sites in that region. These results suggest that drainage and intensification of land-use may have caused greater loss of gaseous CO₂ in drainage waters, in addition to direct CO₂ emission to the atmosphere. The elevated gaseous CO₂ concentration in drainage waters likely reflects higher respiration rates within the aerated peat matrix, and can thus be considered an additional pathway of peat CO₂ loss.

For the purposes of this study we assumed that the vertical emissions of CO₂ from the surface of the ditches, as measured by floating chambers, were separate from the lateral flux of these gases captured by headspace sampling. Whilst this clearly would not be the case under stagnant conditions, when dissolved gases may be transmitted directly to the atmosphere, the approach used assumed no lateral aquatic carbon flux when discharge from the site was not occurring, so the risk of double-counting emissions should be low. When discharge is occurring, dissolved gases in the water column will be transported out of the site via the ditch network into downstream river systems, and gas concentrations in excess of those in the atmosphere (i.e.

'supersaturated') are likely to be emitted, contributing to 'off-site' GHG emissions. An exception to this situation can occur if water pH increases downstream before this equilibration process has taken place, which will increase the solubility of CO₂, leading to dissociation to bicarbonate and the incorporation of CO₂ into stable DIC pool. This situation may arise downstream of raised bogs, where acidic CO₂-rich water mixes with more alkaline runoff from surrounding mineral soils.

Based on these assumptions, estimated C loss fluxes in dissolved CO₂ from the study sites ranged from 0.4 to 2.3 g C m⁻² yr⁻¹. The sites with the highest fluxes were predominantly those with a high degree of management disturbance (EF-DA, EF-SA, MM-EX, SL-IG) as well as the re-wetted MM-RW. However, these fluxes are on average more than an order of magnitude lower than the DOC flux, and thus make a relatively small contribution to the total rate of aquatic C loss.

For dissolved CH₄, measured concentrations and resulting fluxes were consistently very low, even when the global warming potential of CH₄ is taken into account, for all sites except MM-RW. The very high aquatic CH₄ flux obtained for this site is based on a porewater samples, and in particular on one very high measured dissolved CH₄ concentration, and is therefore rather doubtful as noted earlier, although consistent with high CH₄ emissions measured via static chambers at this site. Among the remaining sites, the highest implied CH₄ export flux, 0.16 g C m⁻² yr⁻¹, was measured at EF-DA, coinciding with the largest vertical ditch CH₄ emission measured by floating chambers. In the case of CH₄, the assumption that any dissolved gas transported off-site in discharge will ultimately be degassed is questionable, because it is possible that methane oxidising bacteria will remove it from the water column before degassing can occur. However, the calculated fluxes are so small that (with the exception of MM-RW) the implications for overall site GHG balance are negligible regardless of the assumptions made.

4.3.3. Contribution of aquatic carbon fluxes to total C fluxes and GHG emissions

For the eight fen sites where full aquatic carbon flux measurements were made, the range of total carbon export ranged from 15 to 58 g C m⁻² yr⁻¹. On average, DIC was the largest flux (mean 52%, range 26-77%), followed by DOC (mean 38%, range 20-59%). The other fluxes were all smaller; the dissolved CO₂ flux contributed a mean of 5% (range 2-14%) and POC mean 5%, range 1-15%). The dissolved CH₄ flux accounted for ≤ 0.5% of aquatic carbon export in all fen sites. At the three MM raised bog sites, DOC fluxes alone generated a carbon flux of 35 to 58 g C m⁻² yr⁻¹. Dissolved CO₂ generated a much smaller estimated flux of around 2 g C m⁻² yr⁻¹ at the two sites where it was measured (MM-EX and MM-RW). The dissolved CH₄ flux was negligible at MM-EX, but apparently much larger at MM-RW (1.6 g C m⁻² yr⁻¹). If correct, this would make a significant contribution to total GHG emissions, however as noted earlier this figure was highly influenced by a single high-concentration sample and may not represent a true 'off-site' GHG flux.

Comparison of the magnitude of aquatic C and gaseous C fluxes (derived by eddy covariance) requires some caution, because NEE is positive at some sites and negative at others. At the two AF sites, which are both CO₂ sinks, the aquatic C flux is over half of the NEE, although a significant component of this is associated with DIC which is considered to derive largely from outside the fen, and to be unreactive in the aquatic system. Excluding DIC, the 'reactive' aquatic C flux is 24% of NEE at AF-LN, and 26% at AF-HN, with DOC making up the majority of the flux (20% of NEE at both sites). These results suggest that aquatic C represents a significant fraction of the overall C balance of these sites. In contrast, the aquatic C flux equates to only 15% of CO₂ uptake at EF-LN, of which just 3% is in reactive forms. At the two extensive grassland sites, which are both moderate net CO₂ sources, aquatic C adds a further 22-28% to the overall C loss from the system, of which 6-12% is in reactive forms, which again represents a substantive component of the C budget. Finally, at the EF arable sites, which are both large gaseous CO₂ sources, aquatic carbon represents only a small additional pathway of C loss (4-6% based on total aquatic C flux, 2-4% based on the reactive component).

To calculate the GHG emissions associated with aquatic C loss, we converted each aquatic C fraction to CO₂ equivalents in line with IPCC methodology. For DOC, we assumed 90% conversion CO₂ based on the IPCC (2014) and for POC a 70% conversion factor was used, based on previous Defra-funded research (Evans et al., 2013). As discussed above, we assumed that all dissolved CO₂ and CH₄ would be degassed (and applied a 100 year GWP of 25 for CH₄) whilst DIC was assumed to remain in solution (i.e. a conversion factor of zero). On this basis, the calculated GHG emissions for aquatic carbon among the eight fen sites with full aquatic C budgets ranged from 0.17 t CO_{2eq} ha⁻¹ yr⁻¹ at EF-LN to 1.33 t CO_{2eq} ha⁻¹ yr⁻¹ at AF-HN. At the MM raised bog

sites, although not all aquatic C fluxes were quantified, the contribution of the measured fluxes (overwhelmingly DOC) to GHG emissions was relatively high, ranging from 1.29 to 2.01 t CO₂_{eq} ha⁻¹ yr⁻¹. Across all sites, relationships between land-use intensity and aquatic C loss were not straightforward due to the complexity of hydrological management at some sites, and the over-riding influence of differences in rainfall and discharge between regions. However, as discussed above, we did find evidence of management impacts on DOC export in particular, and this is reflected in differences in overall GHG export in a number of cases, notably SL-IG vs SL-EG and EF-DA and EF-SA versus EF-LN and EF-EG. A similar situation would arise for MM-EX and MM-DA versus MM-RW if the single high dissolved CH₄ flux is omitted. Overall, however, the contribution of aquatic C to overall GHG emissions was small relative to CO₂-related emissions from dry sites, and relative to CH₄ emissions from wet sites.

4.4. Full site carbon and GHG budgets

Full site carbon and GHG budgets were constructed based on the best available data for each location, as follows:

- 1) CO₂ fluxes were based on eddy covariance NEE data wherever these were available, using the mean of full year fluxes as shown in Table 4.1 (for EF-LN a weighted mean was used, as described in Section 4.1). Any lateral carbon removal in harvested biomass at agricultural sites was included in the CO₂ balance (Table 4.1).
- 2) Where eddy covariance data were unavailable, CO₂ fluxes were taken directly from modelled chamber GPP and ER fluxes for conservation-managed sites (MM-RW, NB-LN, NB-HN, TM-RW), and from ER only for bare peat extraction sites (MM-EX, TM-EX). For the remaining agriculturally managed sites (MM-DA and SL-IG) a more complex approach was required (see below).
- 3) Terrestrial CH₄ fluxes, and ditch CO₂ and CH₄ fluxes where available, were taken from static chamber measurements where available (Table 4.2). At the re-wetted sites (MM-RW and TM-RW) fluxes from any remaining ditches were assumed to be the same as those for the terrestrial area. For the two extraction sites (MM-EX and TM-EX) and for the remaining arable site (MM-DA) we assumed that ditch fluxes were equal to the average measured flux at other arable sites.
- 4) Aquatic DOC fluxes were taken directly from the values given in Table 4.5, for all sites except those at TM. At these sites, fluxes were estimated (based on similar measured DOC concentrations at the TM and MM sites) by dividing measured DOC fluxes from the corresponding MM site by the water fluxes for that site (Table 4.4), then multiplying by the estimated annual runoff at Thorne Moors (see Section 2.6.1.3).
- 5) Dissolved CO₂ fluxes were taken from Table 4.5 where available, or from the most analogous sites where unavailable (MM-RW for TM-RW, MM-EX for TM-EX, mean of EF-DA and EF-SA for MM-DA).
- 6) Dissolved CH₄ and POC fluxes were taken from Table 4.5 where available, and assumed to be zero otherwise. DIC fluxes were not included in the calculation for the reasons given in Section 4.3.2.3.

For the two agriculturally managed sites without flux towers, it was not possible to take data from the static chambers directly, because GPP could not be reliably modelled from environmental variables alone. We therefore used data from comparable sites with flux towers to derive a best estimate of the net ecosystem CO₂ balance including biomass offtake (net ecosystem productivity, NEP).

MM-DA: To estimate the NEE of this site, we first adjusted the chamber-based estimate of ER (1151 g C m⁻² yr⁻¹) based on the ratio of eddy covariance to chamber-derived ER at the East Anglian arable sites. This was necessary because, although the chamber- and eddy covariance-based models of ER were very similar for most of the measurement period, the chamber-based models failed to capture the increase in autotrophic respiration that occurred during periods of rapid crop growth. At EF-DA and EF-SA the ratios of EC to chamber-derived ER were almost identical at 1.21 and 1.23 respectively, therefore a ratio of 1.22 was applied to the data from MM-DA, giving an adjusted ER of 1405 g C m⁻² yr⁻¹. Secondly, we calculated the ratios of NEP to ER for each measurement of the six flux measurement years at EF-DA and EF-SA, which gave a mean of 46% and a range of 30% to 69%. This is effectively an estimate of the ratio of heterotrophic

respiration (i.e. CO₂ loss) to total respiration (incorporating autotrophic plant respiration), comparable to the approach used in the 'gain-loss' method of CO₂ accounting for organic soils (IPCC, 2014). Applying this ratio to the adjusted ER value for MM-DA gives an estimated NEP of 647 g C m⁻² yr⁻¹. Assuming a similar rate of biomass offtake in the wheat crop at MM-DA to that at EF-SA (635 g C m⁻² yr⁻¹), would suggest that NEE at MM-DA is close to zero.

SL-IG: For this managed grassland site, the same procedure as above was applied, using the nearby SL-EG as the reference flux tower site (EF-EG was not used because, in addition to being much more distant, it is also dissimilar in terms of both peat properties and the absence of biomass harvesting). Based on the SL-EG data, the ratio of eddy covariance to chamber-derived ER was 1.33, and the mean ratio of NEP to ER for the three years of flux measurements was 12% (range 9% to 14%), suggesting that autotrophic rather than heterotrophic respiration dominates the total flux at this site. Applying these values to the chamber-based ER estimate of 2717 g C m⁻² yr⁻¹ gives an estimated NEP of 433 g C m⁻² yr⁻¹ for this intensive grassland site. If we assume the same rate of biomass offtake at SL-IG as at SL-EG (203 g C m⁻² yr⁻¹) this would give an NEE for the site of +230 g C m⁻² yr⁻¹. In reality it is possible that the grass harvest at SL-IG is higher than at SL-EG, given the intensive management and fertilisation of the site, giving a smaller NEE. However, different assumptions about biomass removal rates would not affect the calculation of NEP, or therefore of the overall C or GHG balance of the site.

Final estimates of the carbon and GHG balances of all 15 study sites are shown in Tables 4.6 and 4.7. The results suggest that the net ecosystem carbon balance (NECB) of the sites ranges from -281 g C m⁻² yr⁻¹ to +773 g C m⁻² yr⁻¹. The most important influence on NECB is clearly the combination of NEE and (where present at agricultural sites) biomass offtake. However other fluxes have a significant influence on NECB at a subset of sites, notably CH₄ emissions at conservation-managed fens and raised bogs, and DOC export at raised bogs. The climate forcing impact of the sites also ranges from negative to positive, from a net GHG sink of 3.6 t CO₂-eq ha⁻¹ yr⁻¹ to a net source of 28.5 t CO₂-eq ha⁻¹ yr⁻¹ (Table 4.7) The GHG balance of the sites is dominated by the sum of NEE and biomass removal at drained agricultural sites, and the balance of CO₂ uptake versus CH₄ emission at the conservation-managed and re-wetted sites.

Note that the GHG balances presented in Table 4.7 do not include N₂O fluxes, because these were not measured at all sites. However, the data collected as part of N₂O study described in Section 3 gave tentative annual fluxes from EF-DA and SL-IG of 5.72 and 8.49 kg N₂O-N ha⁻¹ yr⁻¹ respectively which (based on a 100 year GWP of 298) would equate to an additional GHG emission of 5.36 and 7.95 t CO₂-eq ha⁻¹ yr⁻¹ respectively. If included in the GHG balance below, this would increase emissions from SL-IG by 48%, and those from EF-DA by 18%. It seems reasonable to assume that N₂O emissions from EF-SA would be similar to those from EF-DA. However, at the other arable site included in the N₂O study, MM-DA, emissions remained very low even after fertiliser addition. At the remaining unfertilised sites, we assumed that N₂O emissions were negligible. Whilst we were not currently able to fully test the validity of this assumption, the ditch dissolved N₂O data did provide some evidence of the potential for N₂O production. At fertilised agricultural sites (EF-SA, EF-DA, SL-IG) ditch N₂O concentrations were periodically high, consistent with measured N₂O emissions from the peat surface at EF-DA and SL-IG. At conservation-managed sites with no evidence of agricultural nitrogen enrichment (e.g. based on peat core data), dissolved N₂O concentrations were consistently below ambient atmospheric concentrations, which suggests that (if anything) these sites could be acting as net N₂O sinks. However at the two Wicken Fen sites (EF-LN and EF-EG), both of which are periodically irrigated with nutrient-enriched river water, ditch N₂O concentrations were periodically elevated, which suggests that these sites could be acting as emission sources. Ditch fluxes at these sites have been investigated in further detail in Peacock et al. (2017).

The environmental factors controlling variations in GHG fluxes between sites are considered in Section 4.5, and the implications for UK lowland peat emission factors in Section 6.

Table 4.6. Full estimated carbon balance of all study sites (all data in $\text{g C m}^{-2} \text{yr}^{-1}$). Sites are listed in approximate order of land-use intensity.

Flux ($\text{g C m}^{-2} \text{yr}^{-1}$)	AF-LN	NB-LN	EF-LN	AF-HN	NB-HN	MM-RW	TM-RW	MM-EX	TM-EX	EF-EG	SL-EG	SL-IG	MM-DA	EF-DA	EF-SA
NEE	-87.0	10.0	-140.0	-158.0	-337.7	-41.0	223.0	138.0	142.6	123.0	95.0	229.5	12.1	691.0	249.0
Biomass	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	203.2	203.2	635.0	67.3	429.0
CO ₂ (ditch)	2.8	1.2	36.6	2.2	28.7	0.0	0.0	3.8	3.8	21.6	3.1	-3.4	3.8	3.6	3.9
CH ₄ (land)	15.7	15.1	6.6	14.1	10.1	32.8	11.9	0.2	0.0	0.0	-0.2	-0.1	0.1	-0.1	-0.3
CH ₄ (ditch)	0.4	6.0	1.8	0.1	12.3	0.0	0.0	0.3	0.3	0.2	3.0	0.6	0.3	0.5	0.0
DOC	17.9	6.7	4.5	31.9	4.1	34.6	19.0	58.4	22.1	5.7	10.2	21.8	39.2	7.9	6.6
POC	1.3	0.0	0.4	8.6	0.0	0.0	0.0	0.0	0.0	0.7	0.3	1.2	0.0	0.3	2.2
CO ₂ (dissolved)	1.2	0.7	0.4	1.3	1.2	1.8	1.8	2.3	2.2	0.5	0.8	1.9	2.2	2.2	2.1
CH ₄ (dissolved)	0.1	0.0	0.0	0.0	0.1	1.6	0.0	0.0	0.1	0.0	0.0	0.0	0.1	0.1	0.0
NECB	-47.6	39.7	-89.8	-99.8	-281.1	29.8	255.7	202.9	170.9	151.7	315.4	454.7	692.6	772.8	692.5

Note that biomass offtakes in the form of extracted peat were not quantified for the active MM-EX extraction site (although no harvesting took place at the measurement location itself). Active extraction has ceased at TM-EX.

Table 4.7. Greenhouse gas balance of all study sites (all data in $\text{t CO}_2\text{-eq ha}^{-1} \text{yr}^{-1}$). Calculations are based on a 100 year global warming potential of 25 for CH₄, a DOC to CO₂ conversion rate of 0.9 and a POC to CO₂ conversion rate of 0.7. Sites are listed in approximate order of land-use intensity. N₂O fluxes were not measured at all sites, and are therefore omitted from the table.

Emission/removal ($\text{t CO}_2\text{-eq ha}^{-1} \text{yr}^{-1}$)	AF-LN	NB-LN	EF-LN	AF-HN	NB-HN	MM-RW	TM-RW	MM-EX	TM-EX	EF-EG	SL-EG	SL-IG	MM-DA	EF-DA	EF-SA
NEE	-3.19	0.37	-5.13	-5.79	-12.38	-1.50	8.18	5.06	5.23	4.51	3.48	8.42	0.44	25.34	9.13
Biomass	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	7.45	7.45	23.28	2.47	15.73
CO ₂ (ditch)	0.10	0.04	1.34	0.08	1.05	0.00	0.00	0.14	0.14	0.79	0.11	-0.12	0.14	0.13	0.14
CH ₄ (land)	5.23	5.03	2.18	4.70	3.37	10.93	3.98	0.06	-0.01	0.01	-0.06	-0.05	0.03	-0.04	-0.10
CH ₄ (ditch)	0.13	2.00	0.59	0.03	4.11	0.00	0.00	0.09	0.09	0.05	1.00	0.20	0.09	0.17	0.00
DOC	0.59	0.22	0.15	1.05	0.14	1.14	0.63	1.93	0.73	0.19	0.34	0.72	1.29	0.26	0.22
POC	0.03	0.00	0.01	0.22	0.00	0.00	0.00	0.00	0.00	0.02	0.01	0.03	0.00	0.01	0.06
CO ₂ (dissolved)	0.04	0.03	0.01	0.05	0.04	0.07	0.07	0.08	0.08	0.02	0.03	0.07	0.08	0.08	0.08
CH ₄ (dissolved)	0.02	0.01	0.00	0.00	0.03	0.53	0.00	0.00	0.02	0.00	0.00	0.00	0.02	0.04	0.01
GHG balance	2.96	7.70	-0.85	0.35	-3.64	11.17	12.85	7.36	6.27	5.59	12.36	16.71	23.38	28.45	25.27

4.5. Controls on carbon fluxes

4.5.1. Between-year variations in carbon fluxes

Between-year variations in different components of the carbon balance can occur due to changes in meteorological conditions, or to changes in agricultural management. In semi-natural peatlands, NEE is typically the largest and most variable component of the C balance; long-running flux tower studies have shown that natural sites can transition from strong to weak net sinks, or even to sources, of CO₂ in different years depending on factors such as winter severity, length of growing season, duration of cloud cover and extent of summer water-table drawdown (Roulet et al., 2007; Helfter et al., 2015; Peichl et al. 2014; McVeigh et al., 2014). Compared to these studies, which are based on long-term eddy covariance data, our three years

of flux tower measurements are generally not sufficient to draw clear conclusions regarding the magnitude or drivers of variability in C fluxes, however some general observations can be made.

Firstly, at the conservation-managed fens with more than one year of data (EF-LN, AF-HN) we observed greater variability in NEE between years at EF-LN, commensurate with the greater degree of water-table variability observed at this site, with much weaker net CO₂ uptake in the 2013-14 measurement year, which included a severe drought event. As discussed earlier, the severity of water-table drawdown at EF-LN, although partly explained by the lower and more variable rainfall in East Anglia compared to Anglesey, appears to have been exacerbated by the difficulty of maintaining water levels at this hydrologically isolated site, which is surrounded by drained and subsided agricultural land. Although more data would be needed to confirm these observations, we infer that conservation-managed fens in more heavily modified landscapes may be more susceptible to between-year variations in weather conditions, and therefore to act as weaker CO₂ sinks over longer periods.

At the two managed grassland flux tower sites, EF-EG and SL-EG, NEE was higher (more positive) in 2013 and 2015 compared to 2014. As noted above, the summer of 2014 was relatively warm, and it appears that reduced CO₂ emissions in this year were attributable to greater photosynthetic uptake during the more productive growing season. Since water table drawdown was no greater in 2014 than in other years at these sites (see Figures 2.1.15, 2.4.5) it appears that warmer temperatures were not necessarily associated with greater aeration or decomposition of the peat, and thus that the enhancement of GPP outweighed any increase in ER. However, in the absence of annual biomass offtake data for SL-EG we cannot rule out the possibility that some or all of the increase primary production was subsequently removed from the site in the hay harvest. Perhaps surprisingly, we did not find any evidence that the severe flooding of the Somerset Levels in early 2014 had any impact on C balances.

For the arable sites, it is clear from the eddy covariance data that the major driver of between-year variations in NEE is crop selection, which influences both the magnitude and timing of CO₂ uptake and release as discussed above. However, when harvested biomass removal is taken into account, it becomes apparent that lower/negative NEE values associated with cereal crops at EF-SA are not indicative of a reduced rate of CO₂ emission; indeed it appears that crop type may have little or no overall impact on rates of CO₂ emission once harvested biomass is taken into account. At this stage, we do not have sufficient data to rule out an effect of crop type (or to determine the possible impacts of meteorological factors) on rates of CO₂ loss from arable sites. In particular, it seems likely that differences in the timing and nature of soil disturbance, extent and duration of exposed peat surface, and water level requirements associated with different arable and horticultural crops could – directly or indirectly – have an influence on annual rates of CO₂ loss from arable sites. The specific influence of water table is addressed below.

4.5.2. Spatial controls on site carbon fluxes

Given the highly variable management and associated vegetation cover of the study sites, we anticipated a high degree of heterogeneity in observed fluxes, and that it might be difficult to relate these fluxes to any easily identifiable site condition metric. However, analysis of the data shows that all of the net CO₂ flux (including biomass offtake), the terrestrial CH₄ flux, the NECB and the overall GHG balance of the sites can, to a remarkable degree, be explained by differences in mean measured water table (Figure 4.5). Although other explanatory variables were investigated, such as the nitrogen stock of peat above the mean water table, which was recently proposed as a strong predictor of GHG fluxes from peat under grassland in Germany (Tiemeyer et al., 2016), no other measured variable provided an equivalent degree of explanatory power.

The net CO₂ flux of all semi-natural and drained sites was linearly correlated with mean water-table depth ($R^2 = 0.87$, $p < 0.001$, $n = 13$). The regression equation (shown in Figure 4.5a) has an intercept $-6.3 \text{ t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$, which is the implied CO₂ sink when the water table is at the peat surface, and a coefficient of 0.37, implying that each 10 cm lowering of mean water table will increase CO₂ emissions by $3.7 \text{ t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$. The intersection with the x-axis suggests that lowland peatlands will act as net sinks of CO₂ when the water table is within 17 cm of the surface. It is notable that all seven flux tower sites (shown in bold in the figure) fall close to the regression line, adding further confidence to the relationship obtained. It is also interesting to note that, for a large dataset of (predominantly German and Dutch) sites, Couwenberg et al. (2011) obtained a similar linear relationship (to a maximum depth of 50 cm) with an intercept of $-4.8 \text{ t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$, and an

even steeper gradient of $0.75 \text{ t CO}_2 \text{ ha}^{-1} \text{ yr}^{-1}$. However, if longer-term subsidence data presented by Couwenberg et al. were included, a gradient closer to that observed in our dataset would be obtained. Note that the two re-wetted sites in our study were omitted from the regression analysis, as they were clear positive outliers; again, Couwenberg et al. also omitted an outlier from an inundated site. The implication of these observations is that, despite re-wetting, the two former extraction sites included in our study have so far failed to re-establish the CO_2 sink function that might be expected given their current water table.

A very similar relationship was observed between NECB and mean water table ($R^2 = 0.88$, $p < 0.001$, $n = 13$) with a gradient of $0.10 \text{ t C ha}^{-1} \text{ yr}^{-1}$. The x-axis intercept is at 13 cm, indicating the maximum water-table depth at which a peatland can be expected to act as an overall carbon sink. Again, the re-wetted bog sites were clear outliers and were excluded from the regression.

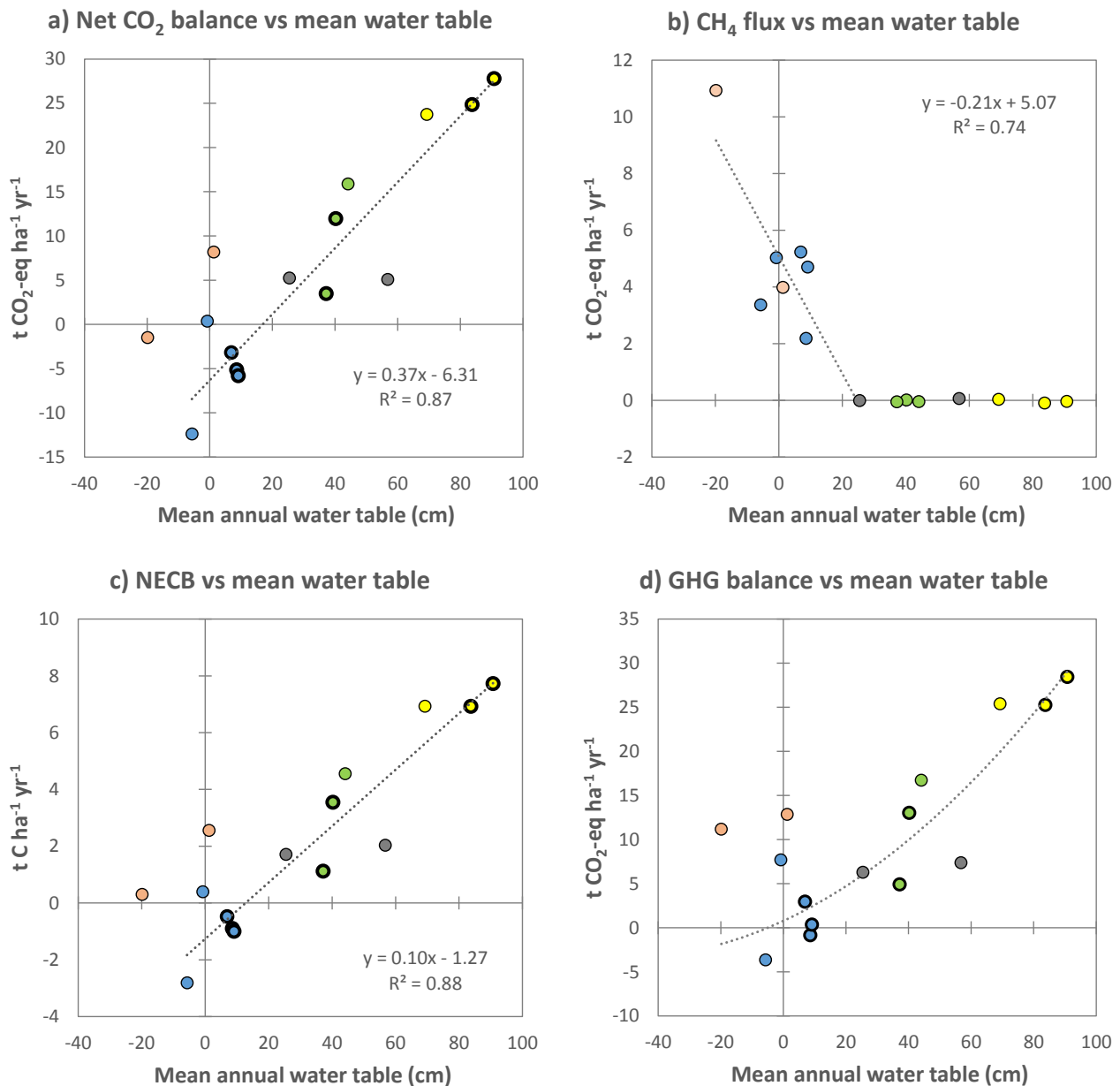


Figure 4.5. Observed relationships between a) net CO_2 flux (including biomass offtake; b) terrestrial CH_4 flux, c) Net Ecosystem Carbon Balance and d) GHG balance (excluding N_2O) and mean measured water-table depth for all study sites. Sites are colour-coded as: blue = conservation-managed fen; orange = re-wetted raised bog; grey = extraction site; green = grassland; yellow = arable. Sites at which eddy covariance data were used to derive CO_2 fluxes are shown with bold outlines. Linear regression lines in a) and c), and quadratic regression in d) are fitted to all points except re-wetted raised bog.

For CH₄, we observed an approximately linear relationship between CH₄ emission and mean water-table depth, where this was within 25 cm of the surface ($R^2 = 0.74$, $p = 0.006$, $n = 8$). The gradient of this relationship was $0.21 \text{ t CO}_2\text{-eq ha}^{-1} \text{ yr}^{-1}$, which represents the increase in CH₄ emission associated with a 1 cm rise in mean water table above this threshold. The inundated MM-RW site falls fairly close to the regression line, which suggests that the very high emissions measured here are indeed attributable to the continuous presence of standing water. At water-table depths > 25 cm, terrestrial CH₄ fluxes were consistently near zero. This threshold is very similar to that obtained in previous studies of peatland CH₄ emissions (e.g. Levy et al., 2012; Couwenberg et al., 2011). Note that we were unable to identify any reliable predictors of ditch CH₄ emission.

The GHG balance of each site was calculated without including N₂O, as we did not have measured values for all sites, and again the two re-wetted sites were excluded. Given the opposing relationships between CO₂, CH₄ and mean water table, we fitted a non-linear (quadratic) relationship between the non-N₂O GHG balance of the study sites and water table (Figure 4.5d; $R^2 = 0.72$, $p < 0.001$, $n = 13$). Whilst a linear model could also be fitted to these data, this was considered less realistic because it would imply that increasingly inundated sites would be increasingly strong GHG sinks, which is unrealistic.

Due to the magnitude of CH₄ emissions from wetter sites, only two of the sites (EF-LN and NB-HN) were found to be net GHG sinks, and the chamber-derived CO₂ flux estimates from NB-HN must be considered relatively uncertain. However, these data do suggest that GHG emissions from lowland peats can be reduced to near-zero values as water-table rises to within around 5 cm below the surface. This finding is similar to that obtained from a collation of German flux data by Drösler et al. (2013; see Figure 41 of their report) which suggested that GHG emissions are minimal when water table is just below the peat surface, but substantially higher when sites are inundated (as at MM-RW). Whilst N₂O fluxes were omitted from these calculations, their inclusion would be expected to reinforce the observed relationship; measured rates of N₂O emission at EF-DA and SL-IG were significant (5.36 and $7.95 \text{ t CO}_2\text{-eq ha}^{-1} \text{ yr}^{-1}$ respectively), whereas previous work has consistently shown negligible N₂O emissions from unfertilised, high water table sites.

4.5.3 Carbon stocks, subsidence and peat accumulation

By combining the estimates of NECB derived from the flux measurements with the peat profile carbon stock measurements, we were able to derive indicative values of peat accumulation or subsidence rates for each of the sites. In order to convert carbon fluxes to changes in peat depth we assumed a 50% carbon content of organic matter. At sites that were gaining carbon, we assumed that peat would accumulate with a bulk density equivalent to that measured in the upper 50 cm of the peat profile. These calculations also assume that the CO₂ being sequestered at these sites will not subsequently be removed in above-ground biomass, e.g. due to reed harvesting. For sites that were losing carbon, we estimated the rate of subsidence that would occur if only oxidative losses were occurring (again, the bulk density of the upper 50 cm was used to convert loss of organic matter into a change in peat volume) and also the total subsidence rate that would arise if around 50% were due to oxidation and 50% to compaction (see Couwenberg et al., 2010 and references therein). Finally we extrapolated current rates of carbon loss forward in order to estimate the number of years it would take for all of the remaining peat to be lost, assuming that this rate remains constant over time. The results of this analysis are shown in Table 4.8.

Excluding the NB sites, which appear to be outliers, the remaining conservation-managed fens are estimated to be accumulating peat at a rate of around 0.05 to 0.20 cm yr^{-1} . All of the other sites are losing carbon (including the re-wetted sites) and are therefore subject to some degree of subsidence. This is lowest at MM-RW (which is close to being in balance in relation to carbon) and at EF-EG, where thin remaining peat layer is highly compacted and therefore slow to lose further volume.

Table 4.8. Estimates peat subsidence/accumulation rates for all study sites

Site	NECB	Peat Accumulation	Subsidence rate (cm yr ⁻¹)		Peat 'lifetime'
	g C m ⁻² yr ⁻¹	rate (cm yr ⁻¹)	Oxidative	Total	(years)
AF-LN	-48	0.06			
NB-LN	40		0.11	0.22	>1000
EF-LN	-90	0.05			
AF-HN	-100	0.18			
NB-HN	-281	0.46			
MM-RW	30		0.04	0.08	>1000
TM-RW	256		0.79	1.58	367
MM-EX*	203		0.17	0.34	399
TM-EX*	171		0.53	1.05	549
EF-EG	152		0.03	0.06	403
SL-EG	315		0.19	0.38	523
SL-IG	430		0.32	0.63	461
MM-DA	693		0.43	0.86	186
EF-DA	773		0.31	0.62	503
EF-SA	693		0.22	0.44	217

*Note that subsidence rates and resulting 'lifetimes' for the two extraction sites assume no further peat extraction (which was the case at both sites at the time measurements were made).

Perhaps surprisingly, the highest estimates rates of subsidence were obtained for the two TM sites, as a result of their very low bulk density (Table 1.2) combined with apparently ongoing carbon loss, even at the re-wetted site. At the five agricultural sites on peat > 50 cm deep (SL-EG, SL-IG, MM-DA, EF-DA and EF-SA) estimated total subsidence rates ranged from 0.38 to 0.86 cm yr⁻¹. The subsidence rates for the East Anglian arable sites are somewhat lower than the 1.5 cm yr⁻¹ estimated by Holman et al. (2009), but reinforce the evidence of this and previous studies that ongoing subsidence remains an issue of major concern for this region, much of which is already below sea-level and prone to flooding. It is also worth noting that our subsidence estimates do not take account of wind-borne erosional losses, which may be considerable in this area due to the low rainfall and frequent exposure and disturbance of bare peat during arable cultivation. For the Somerset Levels grassland sites, our subsidence estimates are very similar to those made by Brunning (2001) for peat under pasture (0.44 to 0.79 cm yr⁻¹). Additionally, subsidence rates for the extraction sites do not take into account any ongoing peat extraction activity, which would greatly reduce the remaining lifetimes of sites in active use.

Our estimates of the remaining 'lifetime' of the peat at sites with significant rates of carbon loss should be viewed as no more than illustrative. However, they suggest that many of these peat deposits will cease to exist within a few centuries should current rates of carbon loss continue. The sites with the shortest remaining lifetime are MM-DA and EF-SA, both of which are predicted to lose all of their remaining peat within around 200 years. Given that this is the time taken to lose *all* remaining organic matter at current rates (i.e. not the time to transition from deep to 'wasted' peat), and that again this estimate does not take account of wind erosion, the actual survival time of deep peat at these sites may be considerably shorter, perhaps less than a century.

5. EVIDENCE REVIEW

The systematic review of the effects of lowland peat management on GHG fluxes was accepted and published in Environmental Evidence, the official journal of the Collaboration for Environmental Evidence (<http://www.environmentalevidencejournal.org/content/pdf/2047-2382-3-5.pdf>) in March 2014. Over 26,000 articles were identified from searches, and screening of obtainable full texts resulted in the inclusion of 93 relevant articles (110 independent studies). Critical appraisal excluded 39 of these studies, leaving 71 to proceed to synthesis. The update of the review is now underway and will include the results of the primary research collected as part of this project, along with new published studies from the literature that have appeared during the intervening period.

A new protocol for the update was written and submitted for open access peer review in March 2016 to Peerage for Science. Literature searches for primary studies published since the original systematic review (from October 2011, performed in January 2016) yielded 3,382 articles. Of these, 721 were screened at abstract level and 303 at full text, resulting in 78 relevant articles to go through to critical appraisal screening. This is now in progress and will shortly lead to meta-analysis of a subset of comparable study results. We expect to submit the updated review for publication by mid-2016.

From the project's primary research data we have intervention-comparator pairs from each of the six research areas to add to the studies found within the published literature. These are:

1. East Anglian Fens - arable-vs-grassland
2. Manchester Mosses - rewetted-vs-extracted
3. Anglesey fens - low nutrient semi-natural-vs-high nutrient semi-natural
4. Somerset levels - extensive-vs-intensive grassland
5. Norfolk Broads - low nutrient semi-natural-vs-high nutrient semi-natural
6. Thorne Moors - rewetted-vs-extracted

The number of published studies relevant to lowland peats that met the requirements for the original systematic review meta-analysis (i.e. robust measurement of fluxes over a sufficient period from spatially replicated intervention and comparison sites) was very limited, and even fewer studies included comprehensive GHG flux measurements. This update will assess the impact of the new data on the inferences that were drawn.

6. TOWARDS TIER 2 EMISSION FACTORS FOR UK LOWLAND PEATS

A primary aim of this project was to support the development of Tier 2 emission factors for British managed lowland peatlands. Subsequently a project was commissioned by DECC to carry out full GHG accounting for UK peatlands, which will generate a full set of EFs for managed lowland peats based on all available published data. The data presented here represent a major contribution to this assessment. However, as we do not wish to pre-empt or duplicate the DECC study we have not attempted to produce a separate set of EFs here. Instead, we directly compare the emission values obtained from the study with the equivalent IPCC Tier 1 EFs from the Wetland Supplement (IPCC, 2014) for each of the land-use classes covered. The aims of this comparison are firstly to provide a cross-check of our results against the previously published work that underpins the IPCC EFs, and secondly to evaluate whether the data suggest that emissions from UK lowland peatlands deviate substantially from the Tier 1 EFs derived from studies undertaken on peatlands over a wider geographic area (northern temperate peatlands).

Figure 6.1 shows a comparison of IPCC Tier 1 emission factors and the equivalent measured values for all study sites, listed in approximate order of management intensity as in Section 4. Although there are clearly similarities between the two sets of emissions estimates, some significant discrepancies are also evident. These are discussed in relation to each land-use category in the following sections.

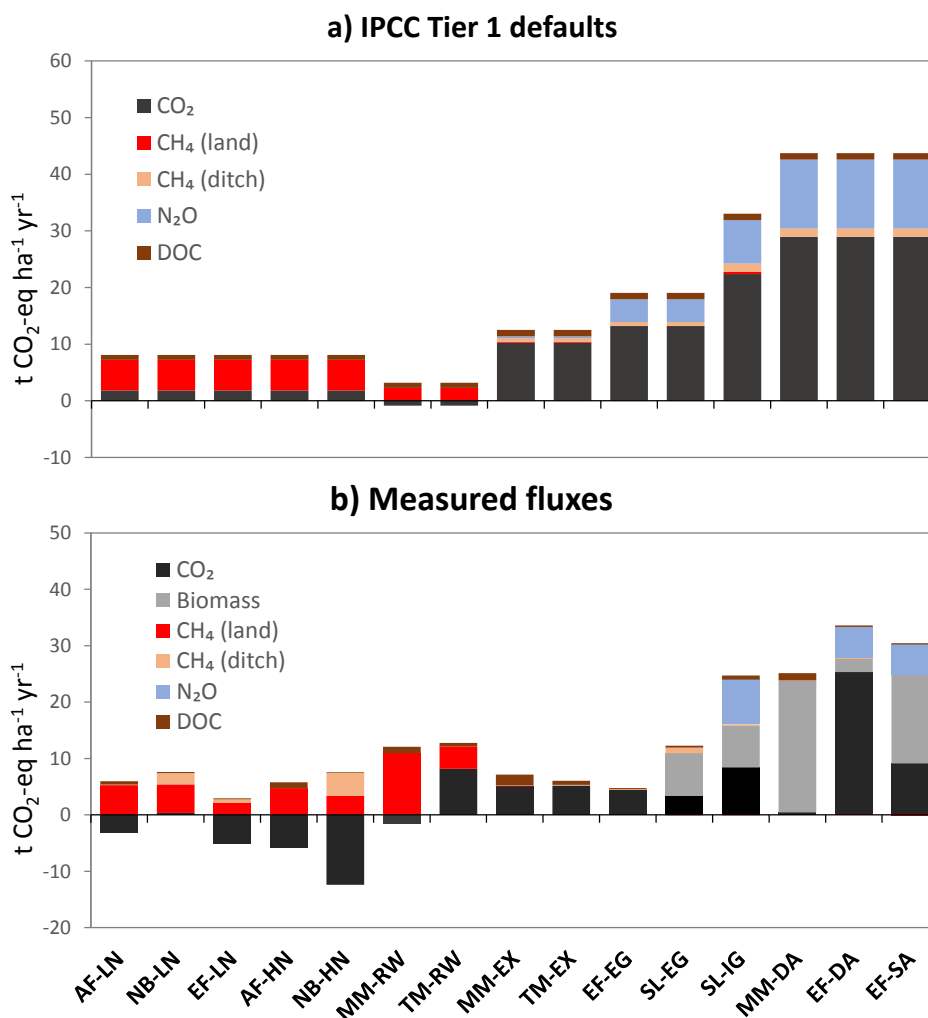


Figure 6.1. Comparison of IPCC Tier 1 default emission factor and measured fluxes for all study sites. Note that biomass fluxes shown in b) are incorporated in the total CO₂ flux shown in a), and that N₂O fluxes were only measured for limited periods at SL-IG and EF-DA and are therefore approximate; the N₂O flux at EF-SA was assumed to be equal to that at EF-DA. Note that biomass offtake fluxes are implicit in the IPCC CO₂ emission factors, but shown separately to the direct CO₂ flux (NEE) in the measured data.

6.1. Conservation-managed fens

The study encompassed five conservation-managed fens, in the East Anglian Fens (EF-LN), Anglesey Fens (AF-LN, AF-HN) and Norfolk Broads (NB-LN, NB-HN). Table 6.1 shows the mean and range of each of the main measured C fluxes for the five sites, relative to the equivalent IPCC Tier 1 EFs. Note that (in this and subsequent tables) the other aquatic C fluxes have been omitted as they either tended to be smaller (POC, dissolved CO₂ and CH₄) or were not considered directly associated with peat C loss (DIC). Furthermore consideration of these fluxes was limited to an appendix of the IPCC Wetland Supplement, as areas requiring further methodological development, so no Tier 1 EFs are available for comparison.

In Table 6.1, it is important to note that the IPCC Tier 1 EFs were derived for re-wetted fens, rather than near-natural systems directly. However, many of the measured data used to derive these EFs were taken from studies undertaken in near-natural systems, after a statistical analysis suggested that mean fluxes did not differ significantly between near-natural and re-wetted sites (IPCC, 2014, Chapter 3). Chapter 3 of the wetland supplement did not specifically address emissions from ditches within re-wetted fens, which were assumed to have the same emission rates as the re-wetted peat surface.

Table 6.1. Measured fluxes vs IPCC default emission factors for conservation-managed fens

Flux	Units	Emission factors		Comparison project vs IPCC Tier 1 EFs
		IPCC Tier 1	Project data	
CO ₂ (land)	t CO ₂ -C ha ⁻¹ yr ⁻¹	+0.50 (-0.71 to +1.71)	-1.43 (-3.38 to +0.10)	Change from source to sink
DOC [†]	t CO ₂ -C ha ⁻¹ yr ⁻¹	+0.24 (+0.14 to +0.36)	+0.11 (+0.03 to +0.26)	Measured ≤ Tier 1
CH ₄ (land)	kg CH ₄ -C ha ⁻¹ yr ⁻¹	+216 (0 to +856)	+129 (+66 to +157)	Measured ≤ Tier 1
CH ₄ (ditch)*	kg CH ₄ -C ha ⁻¹ yr ⁻¹	+216 (0 to +856)*	+92 (+12 to +289)	Measured ≤ Tier 1

*Implicit values in IPCC Wetland Supplement; ditches assumed to emit CH₄ at same rate as rewetted terrestrial areas.

†DOC converted to off-site GHG emission according to IPCC methodology assuming 90% conversion to CO₂.

For CO₂ fluxes, data from the project clearly deviated from the Tier 1 value, which gave a net emission of +0.5 t C ha⁻¹ yr⁻¹. This flux is clearly problematic, in the sense that natural fens must have been net CO₂ sinks over the majority of their history in order for peat to have formed. A recent study by Wilson et al. (2016) further examined and updated the data used to derive this EF in the Wetland Supplement, and concluded that the net emission might result from the inclusion of recently and/or incompletely rewetted sites in the analysis. Our data provide a significantly more positive assessment, suggesting that conservation-managed fens in England and Wales are mostly acting as strong net CO₂ sinks. If we exclude the two values obtained from the Norfolk Broads (which were based on a short period of chamber measurements) the implied EF for CO₂ for the other three sites (all eddy-covariance based) narrows to -1.58 to -0.87 t C ha⁻¹ yr⁻¹. This net uptake is slightly offset by net CO₂ emissions from the areas of open water, and would also be reduced by reed harvesting at sites where this takes place; we did not include an estimate of biomass offtake in the current assessment as no reed harvesting took place at any site during the measurement period. However, over a harvest rotation this would likely reduce the amount of CO₂ sequestered into the fen peat (as opposed to the standing biomass). In the absence of harvesting, measured annual CO₂-C uptake should equate to net long-term C sequestration into the peat, assuming that vegetation biomass reaches a steady state. The DOC loss flux across the study sites was generally somewhat lower than the IPCC default value (which was mainly based on bogs), but there was considerable overlap between observed ranges.

For CH₄, our measured fluxes from the peat surface were around 40% lower than the Tier 1 default, although all measurements fall within the (extremely wide) range of fluxes used to generate the Tier 1 value. Given the comparatively narrow range of flux estimates obtained from the five study sites, all of which gave mean CH₄ emissions below the Tier 1 default estimate, there would appear to be a good case for applying a lower Tier 2 EF for CH₄ emissions from UK conservation-managed peatlands. This may however not be the case for recently re-wetted fens, which were not included in our study but which might be expected to have higher emissions. Emissions of CH₄ from drainage ditches at the study sites were also on average less than the Tier 1

default value, and did not differ strongly from the land-surface emissions, suggesting that a single CH₄ EF may be appropriate for this land-use category.

Finally, we examined whether there was any evidence that the nutrient/vegetation status of conservation-managed fens was having an effect on GHG emissions, by disaggregating the two paired ‘LN’ and ‘HN’ sites in Anglesey and Norfolk (Table 6.2). As previously discussed, it is debatable whether these sites are differentiated by nutrient status *per se*, or by vegetation height (which although thought to be linked to nutrient status may also be influenced by other factors such as site management). The EF-LN site, Wicken Fen, was initially classed as ‘low nutrient’, but is characterised by similar tall fen vegetation to the two ‘HN’ sites, and additionally we found clear evidence of nutrient enrichment in the surface peat, presumably due to the use of eutrophic river water to maintain water levels. Given the difficulty of assigning this site to a class, as well as the absence of a direct comparator site, we therefore excluded it from the analysis. Subject to the strong caveat that we only had two sites in each category, the results do suggest that nutrient-enriched/tall fens act as stronger CO₂ sinks and smaller CH₄ sources, compared to nutrient-poor/short fens. The close proximity and common measurement period of the two AF flux towers adds some confidence to the first of these interpretations. These findings also appear to be consistent with observed differences in vegetation, with the two ‘HN’ sites having taller and more productive (albeit more homogenous) *Phragmites* and *Cladium* cover than the lower-growing, more botanically diverse ‘LN’ sites. The AF-LN site also had a higher mean water table than AF-HN, which could explain the higher CH₄ emissions, although water table differences between the NB sites were less evident. Overall, it appears that there may be a case for disaggregating Tier 2 EFs for conservation-managed fens based on factors such as nutrient status, water level and/or vegetation type, however additional measurements would be needed to support this.

Table 6.2. Measured emissions from ‘paired’ high and low nutrient status conservation-managed fens

Flux	Units	Measured emissions		Comparison high vs low nutrient
		Low nutrient	High nutrient	
CO ₂ (land)	t CO ₂ -C ha ⁻¹ yr ⁻¹	-0.39 (-0.87 to +0.10)	-2.48 (-3.38 to -1.58)	HN stronger C sink
DOC*	t CO ₂ -C ha ⁻¹ yr ⁻¹	+0.10 (+0.05 to +0.14)	+0.15 (+0.03 to +0.26)	No clear difference
CH ₄ (land)	kg CH ₄ -C ha ⁻¹ yr ⁻¹	+154 (+151 to +157)	+136 (+130 to +141)	LN > HN
CH ₄ (ditch)	kg CH ₄ -C ha ⁻¹ yr ⁻¹	+151 (+12 to +289)	+17 (+12 to +23)	LN > HN

*DOC converted to off-site GHG emission according to IPCC methodology assuming 90% conversion to CO₂.

6.2. Managed grasslands

The project included three sites under grassland management. Although all three sites were on fen peat, they were quite heterogeneous in terms of peat depth and management with EF-EG a former arable site thin peat subject to conservation grazing management, SL-EG a grazed and hay-cropped wet meadow on deep peat, and SL-IG a more intensively managed grassland located nearby on similar deep peat. The first two sites, both of which had flux towers operating throughout the project, would appear to fall into the IPCC ‘shallow drained, nutrient-rich’ grassland category, and indeed older flux data from SL-EG (Lloyd, 2006) were used in the derivation of the Tier 1 EF for this category. Although the more intensively managed SL-IG might be considered closer to the ‘deep drained, nutrient-rich’ IPCC category, given the very close similarity in measured water tables between SL-EG and SL-IG (means 40 and 44 cm respectively) we have compared flux measurements from all three sites to the ‘shallow drained’ category (Table 6.3).

The mean and range of measured CO₂ emissions from the three grassland sites was broadly similar to the Tier 1 default value, and associated confidence interval. If only the two extensive grassland sites (i.e. site with eddy covariance tower data) were included, the mean emission of 2.11 t CO₂-C ha⁻¹ yr⁻¹ (range 1.23 – 2.98 t CO₂-C ha⁻¹ yr⁻¹) would be close to the lower confidence interval of the Tier 1 value, which might justify the use of a lower Tier 2 EF for grasslands under conservation management (and a correspondingly higher value for grasslands under conventional agricultural management). Measured DOC fluxes were consistently

lower than the Tier 1 default values which (as for arable sites) is probably related to differences between the lowland fen sites included in the study and the upland bog sites used to derive the Tier 1 value.

For CH₄, project data clearly showed negligible net fluxes across the land surface, and a zero Tier 2 EF for grasslands on peat would therefore appear to be appropriate. Although significant CH₄ emissions were observed from ditches at two of the three study sites, the mean flux was much lower than the Tier 1 default value. This default is largely reliant on measurements made in Dutch grasslands, many of which have high livestock densities, which could contribute to the high rates of ditch CH₄ production, whereas livestock were either only present infrequently (SL sites) or at low densities (EF-EG) in our study sites. Again, therefore, a lower Tier 2 EF may be justified for UK grasslands.

Table 6.3. Measured fluxes vs IPCC default emission factors for managed grassland on peat*

Flux	Units	Emission factors		Comparison project vs IPCC Tier 1 EFs
		IPCC Tier 1	Project data	
CO ₂ (land)	t CO ₂ -C ha ⁻¹ yr ⁻¹	+3.6 (+1.8 to +5.4)	+2.85 (+1.23 to +4.33)	Measured ≈ Tier 1
DOC [†]	t CO ₂ -C ha ⁻¹ yr ⁻¹	+0.31 (+0.19 to +0.46)	+0.10 (+0.05 to +0.18)	Measured < Tier 1
CH ₄ (land)	kg CH ₄ -C ha ⁻¹ yr ⁻¹	+39 (-3 to +81)	-1 (-2 to 0)	Measured < Tier 1
CH ₄ (ditch)	kg CH ₄ -C ha ⁻¹ yr ⁻¹	+527 (+285 to +769)	+53 (+9 to +112)	Measured < Tier 1

*IPCC emission factors for shallow-drained grassland on fen peat; [†]DOC converted to off-site GHG emission according to IPCC methodology assuming 90% conversion to CO₂.

6.3. Cropland

The Wetland Supplement provides a single set of EFs for cropland on temperate peat. The three cropland sites studied in the project (EF-DA and EF-SA on deep and shallow fen peat respectively, and MM-DA on deep bog peat) were therefore compared to these EFs (Table 6.4). The two EF sites had full flux tower data, whilst chamber data from MM-DA were used to estimate the net CO₂ balance of this site as described above. Note that all three CO₂ flux estimates take account of C removal in harvested biomass. Ditch CO₂ and CH₄ fluxes were based on data from the EF sites only.

For CO₂ emissions, data from the project show reasonable consistency with the IPCC Tier 1 default values; the mean from the project is slightly lower than the Tier 1 value, but all measured values lie within the confidence interval of the IPCC value. If only the two sites with flux towers are considered (i.e. MM-DA is excluded) this has little influence on the mean flux CO₂ flux (mean 7.18, range 6.78 to 7.58 t CO₂-C ha⁻¹ yr⁻¹). It should be noted that some older data from EF-DA (Morrison et al., 2013) were used in the calculation of the Tier 1 values, although as this provided only one of 39 data points its influence on the resulting value would have been marginal. On this basis, it appears reasonable to conclude that the current Tier 1 EF for cropland is appropriate for the UK, although the addition of new data from the project (and omission of data from less UK-relevant locations) could provide a more robust Tier 2 EF, with a narrower uncertainty range.

Table 6.4. Measured fluxes vs IPCC default emission factors for cropland on peat

Flux	Units	Emission factors		Comparison project vs IPCC Tier 1 EFs
		IPCC Tier 1	Project data	
CO ₂ (land)	t CO ₂ -C ha ⁻¹ yr ⁻¹	+7.9 (+6.5 to +9.4)	+6.94 (+6.47 to +7.58)	Measured ≈ Tier 1
DOC*	t CO ₂ -C ha ⁻¹ yr ⁻¹	+0.31 (+0.19 to +0.46)	+0.14 (+0.05 to +0.32)	Measured ≤ Tier 1
CH ₄ (land)	kg CH ₄ -C ha ⁻¹ yr ⁻¹	+6 (+2 to +14)	-1.1 (-2.9 to +1.0)	Measured < Tier 1
CH ₄ (ditch)	kg CH ₄ -C ha ⁻¹ yr ⁻¹	+542 (+102 to +981)	+17 (+1 to +33)	Measured < Tier 1

*DOC converted to off-site GHG emission according to IPCC methodology assuming 90% conversion to CO₂.

Measured DOC losses from cropland sites were on average lower than the Tier 1 default values, but clearly varied between the two EF fen sites (mean 0.07 t C ha⁻¹ yr⁻¹) and the MM raised bog site (0.32 t C ha⁻¹ yr⁻¹). The existing Tier 1 default therefore appears suitable for cropland on raised bog, but a much lower Tier 2

value would be more appropriate for cropland on fen peat. For the land surface CH₄ flux, two of the three sites recorded net CH₄ uptake, and the third was only marginal net source, suggesting that the Tier 1 default EF (although small) may be too high. Similarly, we observed very much lower CH₄ fluxes from ditches draining the two EF sites than the Tier 1 EF for ditch emissions. This is not particularly surprising, because the Tier 1 value for cropland was based on measurements from intensively managed grassland systems, in the absence of any data from cropland sites. It is likely that high CH₄ emissions from ditches draining grasslands may be enhanced by high inputs of labile organic matter from animal wastes, and our results suggest that a new, lower Tier 2 EF for ditches in cropland should be developed.

6.4. Extraction sites on raised bog

The project included two peat extraction sites on raised bog, one of which (MM-EX) was a 'primary' site located within an active extraction site, and the other (TM-EX) was a 'secondary' site in an abandoned extraction site for which a shorter (18 month) run of data were collected. In both cases, the peat surface was bare and, in the absence of any photosynthetic uptake by plants, CO₂ emissions could be obtained directly from chamber-based respiration measurements. It is essential to note, however, that our calculations only incorporate CO₂ emissions from the *in situ* peat, and therefore exclude 'off-site' CO₂ emissions from harvested peat used for horticulture or energy in active extraction areas, which may be very large. Based on figures for Ireland reported in Wilson et al. (2015), typical rates of peat removal from extraction areas are around 80 t ha⁻¹ yr⁻¹. Assuming a 50% carbon content and complete oxidation of the extracted peat this equates to an indirect emission of 40 t CO₂-C ha⁻¹ yr⁻¹.

The comparison of measured and Tier 1 default emissions values (Table 6.5) shows considerable differences, most notably for (direct) CO₂ emissions which were around half the Tier 1 value for both study sites. This finding is supported by a wider analysis that was undertaken during the project, based on six industrial and three domestic peat extraction sites on bog peat in Ireland and the UK (which included early data from MM-EX, but not TM-EX) which gave a mean CO₂ emission of 1.68 t C ha⁻¹ yr⁻¹ (Wilson et al., 2015). This study concluded that the discrepancy between CO₂ emissions measured at sites in the British Isles and the Tier 1 default were attributable to differences in peat quality, as peat extraction sites in the UK and Ireland are generally extracted down to highly decomposed deeper peat layers, whereas many of those in the IPCC dataset (particularly from Canada) are from areas in which shallow peat is still being actively extracted and regularly scarified. On this basis, a lower Tier 2 EF for direct CO₂ emissions from UK extraction sites appears appropriate. Data from our sites, and from Wilson et al. (2015), did not show any clear difference in emissions between active and abandoned sites (where vegetation had not recolonised), or (in the case of the Wilson et al. study) between industrial and domestic extraction sites. The results also demonstrate five-fold lower direct CO₂ emissions from extraction sites compared to arable sites, despite their apparent similarities in terms of drainage and bare peat exposure. We interpret this difference to be a consequence of the higher levels of microbial decomposition in peat receiving high rates of labile organic matter input from growing crops, which have been shown to 'prime' decomposition of the native peat organic matter, as well as possible effects of fertiliser addition. In comparison, residual peat in extraction sites tends to be highly decomposed, nutrient poor, and in the absence of labile organic matter input from vegetation appears to decompose more slowly.

Although full DOC flux data were only available from one site, MM-EX, this flux was above the 95% confidence interval of the Tier 1 EF for DOC loss (which is generic across all drained peatlands) and suggests that the true rate of DOC loss from extraction sites may be higher. For the TM-EX site, measured DOC concentrations were very similar to MM-EX, however likely lower water fluxes (inferred from nearby EA gauging stations) at Thorne Moors when compared to the wetter Manchester Mosses suggest that the DOC flux here may also be lower. Therefore, the current Tier 1 EF for DOC may remain appropriate for extraction sites on raised bog, at least until further data are obtained. Emissions of CH₄ from both sites were below the confidence interval of the Tier 1 default EF, suggesting that a lower Tier 2 value might be justified. Emissions of CH₄ from ditches were not measured, so the Tier 1 value remains applicable for this flux. Additionally, neither waterborne nor airborne POC losses were quantified, but both may be large as a result of the exposure and scarification of bare peat surfaces.

Table 6.5. Measured fluxes vs IPCC default emission factors for extraction sites on raised bog

Flux	Units	Emission factors		Comparison project vs IPCC Tier 1 EFs
		IPCC Tier 1	Project data	
CO ₂ (land)*	t CO ₂ -C ha ⁻¹ yr ⁻¹	+2.8 (+1.1 to +4.2)	+1.40 (+1.38 to +1.43)	Measured < Tier 1
DOC†	t CO ₂ -C ha ⁻¹ yr ⁻¹	+0.31 (+0.19 to +0.46)	+0.47	Measured > Tier 1
CH ₄ (land)	kg CH ₄ -C ha ⁻¹ yr ⁻¹	+6 (+2 to +14)	+0.7 (-0.4 to +1.8)	Measured < Tier 1
CH ₄ (ditch)	kg CH ₄ -C ha ⁻¹ yr ⁻¹	+542 (+102 to +981)	Not measured	

*Note that the CO₂ flux only represents direct oxidation of in situ peat, and not the (much larger) off-site emission of CO₂ associated with removal and subsequent oxidation of peat at active extraction sites. †DOC converted to off-site GHG emission according to IPCC methodology assuming 90% conversion to CO₂.

6.5. Re-wetted raised bog

The project included two previously re-wetted former extraction sites on raised bog, MM-RW (a primary site) and TM-RW (a secondary site). The corresponding Tier 1 EFs are taken from the rewetted bog category in Chapter 3 of the IPCC Wetland Supplement, which does not differentiate according to former land-use, and was also to a significant extent based on data from near-natural reference sites. Aquatic C flux measurements were only made at MM-RW, and were based on porewater measurements in the absence of either ditches or a surface outflow from the site; ditch emissions were therefore also not measured.

Measured CO₂ fluxes at the two re-wetted study sites were very different, with MM-RW acting as a small net CO₂ sink and TM-RW as a larger net source. The former lies within the confidence interval of the Tier 1 EF for re-wetted bog, but the latter is much higher (Table 6.6). The reasons for this discrepancy are unclear, but since the Tier 1 EF is based on a large number of studies, many of which should be fairly analogous to the UK, there does not appear to be a strong case for altering the existing Tier 1 EF.

Measured and Tier 1 default emissions associated with DOC export were very similar, suggesting that the current Tier 1 value for re-wetted bog is applicable to these systems. On the other hand, measured CH₄ emissions from both of the study sites were higher than the Tier 1 default value, albeit within the wide confidence interval for the Tier 1 EF. A recent update of the CH₄ EF for re-wetted bog by Wilson et al. (2016), taking into account newly published data, actually suggested marginally lower values for this category, however our data clearly suggest that, at least for some re-wetted sites such as MM-RW, emissions may be considerably higher. Based on the water table data from MM-RW, it appears that the site is currently continuously inundated, a situation which would not be expected to occur naturally, and which has probably contributed to the very high CH₄ emission from this site. Similar work in Germany (e.g. Hahn-Schöfl et al., 2011; Vaneslow-Algan et al., 2014) has shown that CH₄ emissions increase with increasing degree of surface inundation. Although it would be difficult to justify a different Tier 2 EF for re-wetted bogs where inundation has occurred, the results of the study do suggest that a higher EF might be appropriate in this instance, at least during the early period of re-wetting prior to the re-establishment of a more natural vegetation cover and hydrological function, and/or the optimisation of water management to avoid prolonged waterlogging.

Table 6.6. Measured fluxes vs IPCC default emission factors for re-wetted raised bogs

Flux	Units	Emission factors		Comparison project vs IPCC Tier 1 EFs
		IPCC Tier 1	Project data	
CO ₂ (land)	t CO ₂ -C ha ⁻¹ yr ⁻¹	-0.23 (-0.59 to -0.09)	+0.91 (-0.41 to +2.23)	Measured highly variable
CO ₂ (ditch)*	t CO ₂ -C ha ⁻¹ yr ⁻¹	-0.23 (-0.59 to -0.09)	Not measured	
DOC†	t CO ₂ -C ha ⁻¹ yr ⁻¹	+0.24 (+0.14 to +0.36)	+0.28	Measured ≈ Tier 1
CH ₄ (land)	kg CH ₄ -C ha ⁻¹ yr ⁻¹	+92 (+3 to +445)	+224 (+119 to +328)	Measured ≥ Tier 1
CH ₄ (ditch)*	kg CH ₄ -C ha ⁻¹ yr ⁻¹	+92 (+3 to +445)	Not measured	

*Implicit values in IPCC Wetland Supplement; ditches assumed to emit CO₂ and CH₄ at same rate as rewetted terrestrial areas. †DOC converted to off-site GHG emission according to IPCC methodology assuming 90% conversion to CO₂.

7. SUGGESTIONS FOR FUTURE WORK

(Note that key findings are given at the start of the report)

7.1. Sustainable management and climate mitigation in cultivated peatlands

This project has shown that the cultivation of lowland peatlands leads to very high rates of GHG emission. However, given the economic, cultural and food security significance of areas such as the East Anglian Fens, it is unrealistic to expect all agricultural activities on these areas to cease. The project summary provides some suggestions for potential mitigation activities that could be undertaken on cultivated peat. However, the effectiveness of many of these measures remains untested or uncertain. Suggested future research activities in this area include the following:

- 1) Experimental evaluation of the impacts of changing water-table regimes within conventional arable/horticultural management. This should include field or plot scale trials of the impacts of different annual and/or seasonal drainage regimes (such as winter re-wetting) on rates of CO₂ loss, together with an assessment of any associated impacts on crop yields. This work should if possible take place in the vicinity of one or more of the established flux towers on arable land, to allow results to be scaled up to annual fluxes and placed in the context of the full long-term GHG balance.
- 2) Experimental field-scale assessment of a range of other possible management options to reduce GHG emissions from croplands. These could include: i) the effects of changing crop type (e.g. cereal versus salad crops, crop varieties with a higher moisture tolerance or lower need for soil disturbance); ii) introduction of 'wetland' winter cover crops that with the potential to add recalcitrant organic matter to soil during fallow periods; iii) changes in agricultural management practices to reduce soil disturbance in order to minimise decomposition and wind-related peat loss; iv) further investigation of the impacts of reduced or altered fertiliser regimes, and use of nitrification inhibitors to reduce N₂O emissions. Again these experiments could take place alongside established flux towers, and in collaboration with farmers to ensure that any mitigation measures developed are practically and economically viable.
- 3) A similar (ideally parallel) set of experiments on managed grasslands, to evaluate the impacts of changes in water level and grassland management activities such as changing from annual to permanent grassland, altering the amount of form of fertiliser addition, and changes in the nature and timing of grazing and harvesting. As for arable sites this work should if possible be undertaken at a plot or field scale, alongside existing flux towers.

7.2. Achieving climate mitigation benefits from lowland peat restoration

Re-wetting and restoration of lowland fens and raised bogs has occurred in many areas of England and Wales, and is likely to increase in extent in future through large-scale actions such as the Great Fen Project. The evidence of this project is that restoration measures to date have not always been effective in reinstating either the hydrological or the carbon sink function of sites that have been degraded by decades to centuries of agricultural wastage and peat extraction. Achieving effective climate mitigation through restoration of these areas therefore requires a strong evidential basis and effective field trials. Suggested future work includes the following:

- 1) An experimental assessment of optimal water-table conditions to maximise the GHG benefits of re-wetting at both raised bog and fen sites. This will need to take account of the practical requirements of restoring keystone species (such as *Sphagnum* in raised bogs) which may require a transient period of wetter conditions, but also the need to avoid CH₄-transporting species such as cotton grasses (*Eriophorum*) becoming dominant.
- 2) Practical work to identify effective interventions that would enable natural fen species to re-establish on degraded former agricultural sites, particularly those on thin 'wasted' peat such as Bakers Fen which may not revert to a true wetland ecosystem without active interventions such as water level management, nutrient removal and species re-introductions. This activity would be best

led by conservation agencies or NGOs, but would benefit from scientifically rigorous underpinning measurements and monitoring.

- 3) Further investigation of the potential for long-term productive use of peatlands through high water table wetland agriculture ('paludiculture'), such as *Sphagnum* cultivation for horticulture (effectively an alternative to horticultural peat) on former extraction sites, and the use of wetland species for bioenergy production. Since any productive use of peatlands involves biomass removal, the implications of these activities for the carbon and GHG balance needs to be assessed.
- 4) Establishment of new flux towers at transitional sites. The current project was focused primarily on establishing baseline rates of GHG fluxes at sites under long-term stable management. The network of flux towers that has been established, most of which are continuing to operate, provides a powerful baseline against which to assess the impacts of land-use change and restoration. If one or more flux towers could be established at sites that are going to be restored in future would, alongside these baseline sites, provide a highly robust before-after, control-intervention (BACI) design that would enable the transient and longer-term impacts of lowland peat restoration to be rigorously quantified.

7.3 Optimising carbon sequestration by conservation-managed lowland peatlands

Results from the project demonstrate that conservation-managed peatlands contribute to carbon sequestration and may, if optimally managed, act as net GHG sinks. Further research to support this optimisation could include:

- 1) Investigation of the influence of vegetation type and productivity on the net ecosystem carbon balance of, and CH₄ emissions from, fen peatlands. At present, management regimes are largely directed towards maximising botanical or bird species diversity, but it may be possible to reconcile these objectives with reduced CH₄ loss or increased peat accumulation rate through altered management of water levels, cutting and burning regimes, or grazing.
- 2) Specific activities relating to the extent, configuration and water levels in open water features, including ditches and pools, within conservation-managed fens. In many restoration projects, large areas of open water (over 20% of the entire fen area at one of our study sites) are created in order to provide habitat for wetland bird species. These areas - effectively 'flooded lands' in the IPCC terminology - are potentially major hotspots of CH₄ emission, which could cancel out any carbon sequestration benefits derived from the terrestrial area of the peatland. Whilst the current project measured emissions from drainage ditches, the magnitude of CH₄ fluxes from larger open water features in fen peatlands remains unquantified. A focused measurement programme to quantify emissions from open waters would be beneficial both in determining realistic emission rates for GHG accounting, and in identifying measures to minimise these emissions through controls on nutrient and organic matter concentrations and improved ecohydrological design of new restoration sites.
- 3) Research on ways to improve the hydrological integrity of conservation-managed fens, particularly in relation to the maintenance of water tables during the summer, and avoidance of extended periods of inundation during wet periods. Both issues are particularly challenging at hydrologically isolated sites such as Wicken Fen, where severe water table drawdown events during dry summers significantly reduce the CO₂ sink strength of the peat.

7.4 Addressing data gaps for Tier 2 reporting of GHG emissions from lowland peats

We believe that this project represents the most comprehensive attempt ever made to quantify the full carbon and GHG budget of a large, coordinated set of peatland sites. Nevertheless, we identified a number of gaps in both the suite of measurements made and the representativeness of the study sites that could have a significant bearing on overall GHG accounting for lowland peatlands. Key gaps were as follows:

- 1) The study lacked a truly representative example of conventional agricultural management on 'wasted' peat, as the EF-SA site still retained around 80 cm of peat, whilst the wasted peat at the EF-EG site had been converted to conservation grassland. The results from EF-SA suggest that shallow

peats continue to emit CO₂ at a high rate, and core data from EF-EG demonstrate that there is still a large stock of carbon in wasted peats that could be lost as CO₂. Our expectation is therefore that wasted peats under arable agriculture will continue to be large and (given their spatial extent) important emission sources, but this remains to be confirmed through direct flux measurements.

- 2) Woodland and scrub encroachment onto lowland fen and raised bog is a common issue, and a significant amount of restoration activity is devoted to their removal. At present, there are no data on the effects of these activities on carbon or GHG balances, and hence their impact cannot be quantified in national emissions inventories. Quantification of the effects of changing scrub or woodland cover would, given the height of the canopy, require the installation of tall flux towers.
- 3) Wind-related loss of bare peat soils was observed during the project in both arable and extraction sites. Although exploratory attempts to measure wind-borne losses were made, it was not possible to obtain reliable estimates of the magnitude of this flux, which may represent a significant additional pathway of carbon loss. Further work is therefore needed both to develop and implement robust methods to measure wind-borne peat loss, and also to determine the likely fate of this material (i.e. oxidation versus reincorporation in soil organic matter elsewhere).
- 4) Quantification of CH₄ emissions from both wet terrestrial areas and ditches remains a significant source of uncertainty in current GHG budgets, due to the very high observed temporal and spatial variability in this flux. To date it has not been possible to generate reliable estimates of CH₄ emissions by eddy covariance, although this should be possible in future. Flux estimates would also be improved by deployment of larger numbers of funnels or floating chamber systems to measure episodic ebullition fluxes, and the use of autochambers would allow diurnal cycles in CH₄ emission (which, given that all measurements were made during the day could have led to an over-estimation of emissions) to be quantified.

7.5 Building a UK peatland GHG measurement network

This project was, at least in part, born out of a review for JNCC that identified GHG emissions from lowland peatlands as a major evidence gap in the range of peat measurement and research activity being undertaken across the UK. This report also identified the need to establish highly instrumented and fully characterised long-term measurement sites in order to quantify the spatial and temporal controls on peatland carbon and GHG fluxes, and to provide baseline data and 'platforms' for experimental research to determine the impacts of land-use and land-management change, as well as climate change, on the UK's peatlands. By establishing seven fully instrumented flux tower sites, this project has made significant strides towards achieving this objective, and along with the four CEH Carbon Catchments and other flux tower sites on blanket bog operated by the James Hutton Institute, the UK now possesses perhaps the most spatially extensive network of flux measurement sites on peat anywhere in the world.

Having established these instrumented sites, the ongoing cost of maintaining automated measurements such as flux towers, weather stations and water level sensors is comparatively low, and the development of telemetry and automated data processing systems presents the opportunity to collect, process, analyse and report flux data in near real time. As part of a new UK-wide flux network, CEH has committed to support three of the flux tower sites beyond the end of the current project, and the University of Leicester also plans to maintain the three sites they currently operate. However, these activities by individual organisations are inevitably subject to future uncertainty as organisational budgets and priorities change, and there remains a need for coordination and integration of data generated by different groups. The development of a centralised, securely funded long-term network of peatland flux sites would add significant value to existing research activities, increase our understanding of the controls on this nationally important GHG emissions source, and provide strategic underpinning for government policies and practical measures in relation to the responsible management and restoration of the UK's peatland resource. Such a national network could also be connected to a global network of long-term peatland monitoring sites that is being rolled out under an initiative called PeatDataHub, led by the University of Leeds. This would help ensure UK-specific GHG inventories and emissions factors could be clearly differentiated from and contextualised within international datasets.

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