High net CO$_2$ and CH$_4$ release at a eutrophic shallow lake on a formerly drained fen

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Abstract

Drained peatlands often act as carbon dioxide (CO$_2$) hotspots. Raising the groundwater table is expected to reduce their CO$_2$ contribution to the atmosphere and revitalize their function as carbon (C) sink in the long term. Without strict water management rewetting often results in partial flooding and the formation of spatially heterogeneous, nutrient-rich shallow lakes. Uncertainties remain as to when the intended effect of rewetting is achieved, as this specific ecosystem type has hardly been investigated in terms of greenhouse gas exchange (GHG) exchange. In most cases, methane (CH$_4$) emissions increase under anoxic conditions due to a higher water table and in terms of global warming potential (GWP) outperform the shift towards CO$_2$ uptake, at least in the short-term.

Based on eddy covariance measurements we studied the ecosystem–atmosphere exchange of CH$_4$ and CO$_2$ (NEE) at a shallow lake situated on a former fen grassland in Northeast (NE) Germany. The lake evolved shortly after flooding, 9 years previous to our investigation period. The ecosystem consists of two main surface types: open water (inhabited by submerged and floating vegetation) and emergent vegetation (particularly including the eulittoral zone of the lake, dominated by $Typha$ latifolia). To determine the individual contribution of the two main surface types to the net CO$_2$ and CH$_4$ exchange of the whole lake ecosystem, we combined footprint analysis with CH$_4$ modelling and NEE partitioning.

The CH$_4$ and CO$_2$ dynamics were strikingly different between open water and emergent vegetation. Net CH$_4$ emissions from the open water area were around 4-fold higher than from emergent vegetation stands, accounting for 53 and 13 g CH$_4$ m$^{-2}$ a$^{-1}$, respectively. In addition, both surface types were net
Introduction

Peatland ecosystems play an important role in global greenhouse gas (GHG) cycles, although they cover only about 3% of the earth’s surface (Frolking et al. 2011). Peat growth depends on the proportion of carbon (C) sequestration and release. Pristine peatlands act as long-term C sinks and are near-neutral (slightly cooling) regarding their global warming potential (GWP; Frolking et al. 2011), dependent on rates of C sequestration and methane (CH₄) emissions. However, many peatlands worldwide are used e.g. for agriculture, as are more than 85% of the peatlands in Germany and the Netherlands (Silvius et al. 2008). Drainage is associated with shrinkage and internal phosphor fertilisation of the peat (Zak et al. 2008). Moreover, the hydrology of the area as well as physical and chemical peat characteristics are changing (Holden et al. 2004, Zak et al. 2008). Above all, drained and intensively managed peatlands are known as strong sources of carbon dioxide (CO₂; e.g. Joosten et al. 2010, Hatala et al. 2012, Beetz et al. 2013). On the other hand, lowering the water table is typically accompanied with decreasing CH₄ emissions (Roulet et al. 1993). Emission factors of 1.6 g CH₄ m⁻² a⁻¹ and 2235 g CO₂ m⁻² a⁻¹ were assigned to temperate deep-drained nutrient-rich grassland in the 2013 wetland supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories (IPCC 2014).

In the last decades rewetting of peatlands attracted attention in order to stop soil degradation, reduce CO₂ emissions and to recover their functions as C and nutrient sink and ecological habitat (Zak et al. 2015). Large rewetting projects were initiated, e.g. the Mire Restoration Program of the federal state of Mecklenburg-West Pomerania in Northeast (NE) Germany (Berg et al. 2000) starting in 2000 and involving 20 000 ha of formerly drained peatlands, thereby especially fens (Zerbe et al. 2013) e.g. in the Peene river catchment. However, uncertainties remain as to when the intended effects of rewetting are achieved. Only few studies exist on the temporal development of GHG emissions of rewetted fens, especially on longer time scales. Augustin and Joosten (2007) discuss three very different states following peatland rewetting based on observations at Belarusian mires, though without specifying the individual lengths of the phases. Broad agreement exists concerning the CH₄ hot spot characteristic of newly rewetted peatlands (e.g. Meyer et al. 2001, Hahn-Schöffl et al. 2011,
Knox et al. 2015). However, a rapid recovery of the net CO$_2$ sink function is not consistently reported (e.g. Wilson et al. 2007).

Peatlands develop a pronounced microtopography after drainage and subsequent subsidence. Rewetting e.g. in the Peene river catchment resulted in the formation of large-scale shallow lakes in the lower parts of the fens, with water depths usually below 1 m (Zak et al. 2015, Steffenhagen et al. 2012). These new ecosystems are nutrient-rich and most often strikingly different from natural peatlands. They experience a rapid secondary plant succession (Zak et al. 2015). Helophytes are expected to progressively enter the open water body over the time leading to the terrestrialisation of the shallow lake and in the best case peat formation. However, this new ecosystem type and its progressive transformation have hardly been investigated in terms of GHG dynamics. The ecosystem-inherent spatial heterogeneity suggests complex patterns of GHG emissions due to distinct GHG source or sink characteristics of the involved surface types (generally open water and the littoral zone) resulting in measurement challenges. Site-specific heterogeneity implicitly has to be considered for the evaluation of ecosystem scale flux measurements (e.g. Barcza et al. 2009, Hendriks et al. 2010, Hatala Matthes et al. 2014). The importance of small open water bodies in wetlands as considerable GHG sources was highlighted in previous studies (e.g. by Schrier-Uijl et al. 2011, Zhu et al. 2012, IPCC 2014) and in case of CH$_4$ even for landscape-scale budgets e.g. by Repo et al. (2007). In addition, the littoral zone of lakes is often found to be a CH$_4$ hot spot (Juutinen et al. 2003, Wang et al. 2006) with a contribution of up to 90 % to the whole-lake CH$_4$ release (Smith and Lewis 1992), albeit depending on the lake size (Bastviken et al. 2004) and plant community. Rõõm et al. (2014) measured the largest CH$_4$ (and CO$_2$) emissions of a temperate eutrophic lake at the helophyte zone within the littoral.

The objectives of this study are 1) to investigate the ecosystem-atmosphere exchange of CH$_4$ and CO$_2$ (NEE) of a nutrient-rich lake ecosystem emerged at a former fen grassland and 2) particularly infer the individual GHG dynamics of the main surface types within the ecosystem and quantify their contribution to the annual exchange rates. Therefore, we applied the eddy covariance technique from May 2013 to May 2014 and used an analytical footprint model to downscale the spatially integrated, half-hourly fluxes to the main surface types “open water” and “emergent vegetation”. The resulting source area (i.e. spatial origin of the flux) fractions were then included in a temperature response (CH$_4$) and NEE partitioning model (CO$_2$) in order to quantify the source strength of the two surface types.
2 Material and methods

2.1 Study site

The study site “Polder Zarnekow” is a rewetted, rich fen (minerotrophic peatland) located in the Peene river valley (Mecklenburg-West Pomerania, NE Germany, 53°52.5’ N 12°53.3’ E, see Fig. 1), with less than 0.5 m a.s.l. elevation. It is part of the Terrestrial Environmental Observatories Network (TERENO). The temperate climate is characterised by a long-term mean annual air temperature and mean annual precipitation of 8.7 °C and 584 mm, respectively (German Weather Service, meteorological station Teterow, 24 km SW of the study site; reference period 1981–2010). The geomorphological character of the area is predominantly a result of the Weichselian glaciation as the last period of the Pleistocene (Steffenhagen et al. 2012). The fen developed with continuous percolating groundwater flow (Succow 2001). Peat depth partially reaches 10 m (Hahn-Schöfl et al. 2011). Drainage was initialized in the 18th century and strongly intensified between 1960 and 1990 within an extensive melioration program (Höper et al. 2008). The decline of the water table to > 1 m below surface and subsequent decomposition and mineralisation of the peat (especially in the upper 30 cm, Hahn-Schöfl et al. 2011) caused phosphor fertilisation (Zak et al. 2008) and soil subsidence to levels below that of adjacent freshwater bodies (Steffenhagen et al. 2012, Zerbe et al. 2013). The latter simplified the rewetting process which was initiated in winter 2004/2005 by opening the dikes. In consequence of flooding the drained fen was converted into a spatially heterogeneous site of emergent vegetation (on temporarily inundated soil) and permanent open water areas. In this study we focus on a eutrophic shallow lake (open water body about 7.5 ha) as part of the rewetted area, with water depths ranging from 0.1 to 0.7 m. During the study period the open water body of the lake was inhabited by submerged and floating macrophytes, particularly Ceratophyllum demersum, Lemna minor, Spirodela polyrhiza (Steffenhagen et al. 2012) and Polygonum amphibium, which rather corresponds to the sublittoral zone in a typical lake zonation. Ceratophyllum and Lemna sp. were already reported to colonise the lake in the second year of rewetting (Hahn-Schöfl et al. 2011). Phalaris arundinacea, that dominated the fen before rewetting, died off in the first year of inundation (Hahn-Schöfl et al. 2011) and has been limited to the non-inundated periphery of the ecosystem. Helophytes (e.g. Glyceria, Typha) started the colonisation of lake margins and other temporarily inundated areas in the third year of rewetting. The eulittoral zone of the lake is now dominated by Typha latifolia stands gradually colonising the open water in the last years. Emergent vegetation stands also include sedges as Carex gracilis (Steffenhagen et al. 2012). At the bottom of the shallow lake an up to 30 cm thick layer of organic sediment evolved, initially fed by fresh plant material of the former vegetation and...
since then continuously replenished by recent aquatic plants and helophytes after die-back (Hahn-124 Schöfl et al. 2011).

### 2.2 Eddy covariance and additional measurements

We conducted eddy covariance (EC) measurements of CO₂ and CH₄ exchange on a tower placed on a stationary platform at the NE edge of the shallow lake (see Fig. 1). Thereby we ensured to frequently catch the signal from both the open water body and the *Typha latifolia* dominated belt of the shallow lake (eulittoral zone). We defined an area of interest (AOI) in order to focus on an ecosystem dominated by a shallow lake and to avoid a possible impact of the farm and grassland to the north of the shallow lake. The EC measurement setup included: an ultrasonic anemometer for the 3D wind vector \((u, v, w)\) and sonic temperature (HS-50, Gill, Lymington, Hampshire, UK), an enclosed-path infrared gas analyser (IRGA) and an open-path IRGA for CO₂/H₂O and CH₄ concentrations, respectively (LI-7200 and LI-7700, LI-COR Biogeosciences, Lincoln NE, USA). Flowrate was about 10-11 l min⁻¹.

Measurement height was on average 2.63 m above the water surface at the position of the tower, depending on the water level. We recorded raw turbulence and concentration data with a LI-7550 digital data logger system (LI-COR Biogeosciences, Lincoln NE, USA) at 20 Hz in half-hourly files. The dataset is shown in Coordinated Universal Time (UTC), which is 1 hour behind local time (LT).

We further equipped the tower with instrumentation for net radiation, air temperature/humidity, 2D wind direction and speed, incoming and reflected photosynthetic photon flux density (PPFD/PPFDr) and water level. Additional measurements in close proximity to the tower included precipitation, soil heat flux as well as soil and water temperature. Soil temperature was measured below the water column in depths of 10 cm, 20 cm, 30 cm, 40 cm and 50 cm and water temperature at the sediment-water interface. All non-eddy covariance-related measurements were logged as 1 min averages/sums (precipitation). Gaps were filled with measurements of the Leibniz Centre for Agricultural Landscape Research (ZALF, Müncheberg, Germany) at the same platform and a nearby climate station (Climate station Karlshof, GFZ German Research Centre for Geosciences, 14 km distance from study site, Itzerott 2015).

A water density gradient was calculated based on the temperature at the water surface and at the sediment-water interface. The water surface temperature was calculated based on the Stefan-Boltzmann law (see e.g. Foken et al. 2008):

\[
T_w = \sqrt[4]{\frac{I}{k_w \sigma_S B}}
\]  

(1)
where \( T_w \) is the water surface temperature (K), \( I \) is the long-wave outgoing radiation (W m\(^{-2}\)), \( \varepsilon_w \) is the infrared emissivity of water (0.960) and \( \sigma_{SB} \) is the Stefan–Boltzmann constant (5.67·10\(^{-8}\) W m\(^{-2}\) K\(^{-4}\)). We calculated the density of the air-saturated water at the water surface and the sediment-water interface according to Bignell (1983):

\[
\rho_{as} = \rho_{af} - 0.004612 + 0.000106 \times T
\]

where \( \rho_{as} \) is the density of the respective air-saturated water (k m\(^{-3}\)), \( \rho_{af} \) is the density of the respective air-free water (k m\(^{-3}\); see Wagner and Prüß 2002) at atmospheric pressure (1013 hPa) and \( T \) is the respective water temperature (°C). The gradient of the two water densities (air-saturated) \( \frac{\Delta \rho}{\Delta z} \) was calculated as difference of the water density (air-saturated) at the sediment-water interface and the surface water density (air-saturated), divided by the distance (m) between the two basic temperature measurements. Changes of the distance due to the fluctuating water level were considered. Positive and negative gradients indicate periods of stratification and thermally induced convective mixing of the water column, respectively.

### 2.3 Flux computation and further processing

For this analysis we used data from 14 May 2013 to 14 May 2014. We calculated half-hourly fluxes of CO\(_2\) and CH\(_4\) based on the covariances between the respective scalar concentration and the vertical wind velocity using the processing package EddyPro 5.2.0 (LI-COR, Lincoln, Nebraska, USA). Sonic temperature was corrected for humidity effects according to van Dijk et al. (2004). Artificial data spikes were removed from the 20 Hz data following Vickers and Mahrt 1997. We used the planar fit method (Finnigan et al. 2003, Wilczak et al. 2001) for axis rotation and defined the sector borders according to Siebicke et al. (2012). Block averaging was used to detrend turbulent fluctuations. For time lag compensation we applied covariance maximization (Fan et al. 1990). Spectral losses due to crosswind and vertical instrument separation were corrected according to Horst and Lenschow (2009). The methods of Moncrieff et al. (2004) and Fratini et al. (2012) were used for the correction of high-pass filtering and low-pass filtering effects, respectively. For fluctuations of CH\(_4\) density we corrected changes in air density according to Webb et al. (1980), considering LI-7700-specific spectroscopic effects (McDermitt et al. 2011). According to the micrometeorological sign convention, positive values represent fluxes from the ecosystem into the atmosphere (emission) and negative values fluxes from the atmosphere into the ecosystem (ecosystem uptake).
2.4 Quality assurance

We filtered the averaged fluxes according to their quality as follows (see Table 1):

- We rejected fluxes with quality flag 2 (QC 2, bad quality) based on the 0-1-2 system of Mauder and Foken (2004).
- CH$_4$ fluxes were skipped if the signal strength (RSSI) was below the threshold of 14 %. This threshold was estimated according to Dengel et al. (2011).
- Fluxes with friction velocity ($u^*$) < 0.12 m s$^{-1}$ and > 0.76 m s$^{-1}$ were not included due to considerably high fluxes beyond these thresholds, which were estimated similar to the procedure described in Aubinet et al. (2012) based on binned $u^*$ classes. The storage term was calculated as described in Béziat et al. (2009).
- Unreasonably high positive and negative fluxes (0.2 %/99.8 % percentile) were discarded from the CO$_2$ and CH$_4$ flux dataset.

Quality control (apart from EddyPro internal steps) and the subsequent processing steps were performed with the free software environment R (R Core Team 2012).

2.5 Footprint modelling

We applied footprint analysis to determine the source area including the fractions of the surface types of each quality-controlled half-hourly flux using a footprint calculation procedure following Göckede et al. (2004). The source area functions were calculated based on the analytical footprint model of Kormann and Meixner (2001). Roughness length and vegetation height were estimated with an iterative algorithm (see also Barcza et al. 2009). Based on an aerial image (GoogleEarth, http://earth.google.com/) the surface of our study site was classified into two main types and implemented in a land cover grid: “open water” including in particular the open waterbody of the shallow lake with 0.1 to 0.7 m water depth and “emergent vegetation” with a height up to 2 m and including the eulittoral zone of the shallow lake dominated by Typha latifolia. The cumulative annual footprint climatology was calculated following Chen et al. (2011). Fluxes were excluded where footprint information was not available or more than 20 % of the source area was outside the AOI (see Fig. 1 and Table 1). The fractional coverage within the AOI ($A_i$) was 21.7 % for open water.

Quasi-continuous source area information for the two surface types were achieved by gapfilling the results of the footprint model with the means of the source area fractions of the surface types ($\Omega_i$) for 1°-wind direction-intervals, separately for stable and unstable conditions. In case the sum of the $\Omega_i$ was
less than 100%, when the source area exceeded the set borders, we assigned the remaining contribution percentages to emergent vegetation, as the area beyond the borders is dominated by emergent vegetation rather than open water.

2.6 Gapfilling

An enhanced lookup table (LUT) approach proposed by Reichstein et al. (2005), available as web tool based on the R package REddyProc (http://www.bgc-jena.mpg.de/REddyProc/brew/REddyProc.rhtml) was applied for gapfilling and partitioning of NEE measurements (LUT\textsubscript{CO2nofoot}), with air temperature as temperature variable. For the gapfilling of CH\textsubscript{4} measurements non-linear regression (NLR) was applied (NLR\textsubscript{CH4nofoot}):

\[ F_{\text{CH}_{4}} = \exp(a + b_1 \cdot X_1 + \ldots + b_j \cdot X_j) \]  

where \(a\) and \(b_1\ldots b_j\) are fitting parameters and \(X_1\ldots X_j\) are environmental parameters. Several environmental parameters, which were reported to be correlated with \(\text{CH}_4\) flux on different time scales, were tested to find the best bi- or multivariate NLR model for the ecosystem \(\text{CH}_4\) flux: pressure change, \(u^*\), PAR, air temperature, soil heat flux, soil/peat temperature in different heights and waterlevel. Only fluxes of the best quality (QC 0) were used to fit the NLR model and the LUT.

2.7 Calculation of the annual CO\textsubscript{2} and CH\textsubscript{4} budget and the global warming potential (GWP)

We used the continuous flux datasets derived from gapfilling for the calculation of annual CO\textsubscript{2} and CH\textsubscript{4} budgets. The ecosystem GHG balance was calculated by summation of the net ecosystem exchange of CO\textsubscript{2} and CH\textsubscript{4} using the global warming potential (GWP) of each gas at the 100-year time horizon (IPCC, 2013). According to the IPCC AR5 (IPCC, 2013) CH\textsubscript{4} has a 28-fold global warming potential compared to CO\textsubscript{2} (without inclusion of climate-carbon feedbacks).

The uncertainty of the annual estimates was calculated as the square root of the sum of the squared random error (measurement uncertainty) and gapfilling error within the one-year observation period (see e.g. Hommeltenberg et al. 2014, Shoemaker et al. 2015). An estimation of the random uncertainty due to the stochastic nature of turbulent sampling according to Finkelstein and Sims (2001) is implemented in EddyPro 5.2.0. In case of the LUT approach the gapfilling error (standard error) was calculated from the standard deviation of the fluxes used for gapfilling, provided by the web tool. For budgets based on the NLR approach we used the residual standard error of the NLR model as gapfilling error (following Shoemaker et al. 2001).
2.8 Estimation of surface type fluxes

To estimate the specific surface type fluxes, we combined footprint analysis with NEE partitioning (using NLR) to assign gross primary production (GPP) and ecosystem respiration ($R_{eco}$) to the two main surface types (NLRCO2box). $R_{eco}$ and GPP were modelled as sum of the two surface type fluxes weighted by $\Omega_i$ (analogous to Forbrich et al. 2011). Night-time $R_{eco}$ (global radiation < 10 W m$^{-2}$) was estimated by the exponential temperature response model of Lloyd and Taylor (1994) assuming that night-time NEE represents the night time $R_{eco}$:

$$R_{eco} = \sum_{i=1}^{2} \Omega_i \cdot (R_{refi} \cdot \exp\left(\frac{1}{T_{refi}} - \frac{1}{T_{airi}}\right))$$

(4)

where $R_{eco}$ is the half-hourly measured ecosystem respiration ($\mu$mol m$^{-2}$s$^{-1}$), $\Omega_i$ is the source area fraction of the respective surface type, $R_{refi}$ is the respiration rate at the reference temperature $T_{refi}$ (283.15 K), $E_0$ defines the temperature sensitivity, $T_0$ is the starting temperature constant (227.13 K) and $T_{airi}$ the mean air temperature during the flux measurement. The model parameters achieved for night time $R_{eco}$ were applied for the modelling of day-time $R_{eco}$. GPP was calculated by subtracting daytime $R_{eco}$ from the measured NEE. GPP was further modelled using a rectangular, hyperbolic light response equation based on the Michaelis–Menten kinetic (see e.g. Falge et al. 2001):

$$GPP = \sum_{i=1}^{2} \Omega_i \cdot \left(\frac{GP_{maxi} \cdot x_i \cdot PAR}{x_i \cdot PAR + GP_{maxi}}\right)$$

(5)

where $GPP$ is the calculated gross primary production ($\mu$mol m$^{-2}$s$^{-1}$), $\Omega_i$ is the source area fraction of the respective surface type, $GP_{maxi}$ is the maximum C fixation rate at infinite photon flux density of the photosynthetic active radiation $PAR$ ($\mu$mol m$^{-2}$s$^{-1}$), $x$ is the light use efficiency (mol CO$_2$ mol$^{-1}$ photons). We calculated one parameter set for $R_{eco}$ and GPP per day based on a moving window of 28 days (method NLR$_{total}$). In order to avoid over-parameterization we introduced fixed values of 150 for $E_0$ and -0.03 and -0.01 for $x$ of emergent vegetation and water bodies, respectively, to get reasonable parameter values for $R_{refi}$ and $GP_{maxi}$. We excluded parameter sets for $R_{eco}$ or GPP, if one of the two $R_{refi}$ and $GP_{max}$ parameter values was insignificant (p-value $\geq$ 0.05), negative or zero. In addition, the 1 %/99 % percentiles of $GP_{maxi}$ were excluded. These gaps within the parameter set were filled by linear interpolation. Gaps remain in $R_{eco}$ and GPP time series due to gaps in the environmental variables. Gaps up to 3 hours length were filled by linear interpolation. Larger gaps were filled with the mean of the flux during the same time of the day before and after the gap. Due to the moving window approach, we could not estimate model parameters for the first and last 14 days of our study period. Instead, we applied the first and last estimated parameter set, respectively. Modelled GPP and $R_{eco}$ were summed up to half-hourly NEE fluxes and used for alternative NEE gapfilling.
As for NEE we expect different CH$_4$ emission rates of the two surface types. Thus, we extended the NLR model (NLR$_{\text{CH4foot}}$) in a way that the CH$_4$ flux is the sum of the two surface type fluxes weighted by $\Omega_i$:

$$F_{\text{CH}_4} = \sum_{i=1}^{2} \Omega_i \cdot \exp(a_i + b_{1i} \cdot X_1 + \ldots + b_{ji} \cdot X_j) \quad (6)$$

where $\Omega_i$ is the source area fraction of the respective surface type. Considering the principle of parsimony, we combined up to three parameters besides the contribution of the surface types. Remaining gaps were filled by interpolation. Surface type CO$_2$ and CH$_4$ fluxes were derived based on the fitted NLR parameters.

We calculated the annual budgets of CO$_2$ and CH$_4$ for the EC source area, the surface types (assuming source area fraction of 100 % for the respective surface type) and the AOI, the latter following Forbrich et al. (2011) by applying Eq. 4 and Eq. 5 for CO$_2$ as well as Eq. 6 for CH$_4$ with the fitted parameters, but $A_i$ instead of $\Omega_i$ as weighting surface type contribution. The gapfilling error for the NLR$_{\text{CO2foot}}$ model was based on the residual standard error of both $R_{\text{eco}}$ and GPP.

3 Results

3.1 Environmental conditions and fluxes of CO$_2$ and CH$_4$

Mean annual air temperature and annual precipitation for the study period were 10.1 °C and 416.5 mm, respectively, indicating an unusual dry and warm measurement period compared to the long-term average. The summer 2013 was among the 10 warmest since the beginning of the measurements in 1881 (German Weather Service DWD). From June to August monthly averaged air temperature was 0.2 up to 0.9 °C higher and precipitation was 9.1 up to 38.1 mm less than the long-term averages. The open water area of the shallow lake was densely vegetated with submerged and floating macrophytes. A summertime algae slick accumulated in the NE part of the shallow lake. Winter 2013/2014 was characterised by exceptionally mild temperatures and very sparse precipitation. However, a short cold period (see Fig. 2) resulted in ice cover on the shallow lake between 21 January and 16 February 2014. The water level of the shallow lake fluctuated between 0.36 and 0.77 m (at the position of the sensor) and had its minimum at the end of August/beginning of September and its maximum in January. We observed the exposure of normally inundated soil surface at emergent vegetation stands during the dry period in summer 2013.

Both CO$_2$ and CH$_4$ flux measurement time series showed a clear seasonal trend with median CO$_2$ flux of 0.57 $\mu$mol m$^{-2}$ s$^{-1}$ and a median CH$_4$ flux of 0.02 $\mu$mol m$^{-2}$ s$^{-1}$. CH$_4$ emissions peaked in mid-August
2013 with 0.57 μmol m$^{-2}$ s$^{-1}$. The highest net CO$_2$ uptake (-15.34 μmol m$^{-2}$ s$^{-1}$) and release (21.04 μmol m$^{-2}$ s$^{-1}$) were both observed in June 2013. A diurnal cycle of CO$_2$ fluxes with peak uptake around midday and peak release around midnight was obvious until November 2013 and beginning in March 2014 (see Fig. 3). To investigate the potential presence of a diurnal cycle of CH$_4$ fluxes we normalized the mean half-hourly CH$_4$ fluxes per month with the respective median of the half-hourly fluxes of the specific month (minimum five 30 min fluxes per day; method modified from Rinne et al. 2007). We found a clear diurnal cycle of CH$_4$ fluxes from June to September 2013 and starting again in March 2014 (April and May not shown as the sensor was dismantled) with daily peaks during night time (around midnight until early morning). The water density gradient indicates thermally induced convective mixing of the whole water column during the night (around midnight until early morning) from May until October 2013 and from February to May 2014. In May 2014 the diurnal pattern of the water density gradient was less pronounced than in May 2013.

### 3.2 Gapfilling performance and annual budgeting of CO$_2$, CH$_4$, C and GWP

The LUT$_{CO2nofoot}$ approach explained 74 % of the variance in NEE (see Table 2). Median NEE accounted for 1.9 g CO$_2$ m$^{-2}$ d$^{-1}$. The annual budget of gapfilled NEE (LUT$_{CO2nofoot}$) between 14 May 2013 and 14 May 2014 was 524.5 ± 5.6 g CO$_2$ m$^{-2}$ (see Table 3), characterising the site as strong CO$_2$ source with moderate rates of R$_{eco}$ and GPP. We found a surprising CO$_2$ release strength during summer 2013, where already at the end of June daily R$_{eco}$ often exceeded GPP. The highest CO$_2$ emission and uptake rates of 24.8 g CO$_2$ m$^{-2}$ d$^{-1}$ and -27.9 g CO$_2$ m$^{-2}$ d$^{-1}$ were both observed in the beginning of July 2013 (see Fig. 2). July 2013 accounted for 23.2 % and 25.8 % of the annual R$_{eco}$ and GPP, respectively. In addition, net CO$_2$ release outside the growing season (definition of the growing season following Lund et al. 2010; until 19 November 2013 and starting 26 February 2014) was 203.7 g CO$_2$ m$^{-2}$ with a median of 2.2 g CO$_2$ m$^{-2}$ d$^{-1}$.

The environmental variable giving the best NLR model for CH$_4$ was soil temperature in 10 cm depth ($T_{s10}$):

$$F_{CH_4} = \exp(-7.224 + 0.313 \cdot T_{s10})$$

(7)

The model described 79 % of the variance in CH$_4$ flux (see Table 2). Including additional environmental variables to the regression function did not increase the model performance significantly. Cumulative CH$_4$ emissions were 40.5 ± 0.2 g CH$_4$ m$^{-2}$ a$^{-1}$ (see Table 3). Median CH$_4$ emissions were 41.9 mg m$^{-2}$ d$^{-1}$, peaked at the end of July 2013 with 0.6415 g CH$_4$ m$^{-2}$ d$^{-1}$ and were at the minimum in January 2014 (see Fig. 2). The month with the highest proportion of annual CH$_4$ emissions was August 2013 (27.3 %).
Non-growing season \( \text{CH}_4 \) fluxes only accounted for a small proportion within the annual budget, about 337 ± 0.8 g \( \text{CH}_4 \) m\(^{-2}\) a\(^{-1}\).

The site was an effective C and GHG source, accounting for 173.4 ± 1.7 g C m\(^{-2}\) a\(^{-1}\) and 1658.5 ± 11.2 g CO\(_2\)-Eq. m\(^{-2}\) a\(^{-1}\) for the EC source area (see Fig. 4). The proportion of CO\(_2\) in the C and GWP budget was 82.5 % and 31.6 %, respectively. Components of the annual net C balance other than CO\(_2\) and \( \text{CH}_4 \) fluxes, e.g. dissolved C, are not considered in this study. Our uncertainty estimates are within the range of similar studies (e.g. Shoemaker et al. 2015).

### 3.3 Source area composition and spatial heterogeneity of CO\(_2\) and \( \text{CH}_4 \) exchange

Footprint analysis revealed the peak contribution in an average distance of 18 m from the tower and mainly from the open water area of the shallow lake (see Fig. 5). Open water covered on average 62.5 % of the EC source area. The two surface types showed different emission rates in terms of higher \( \text{CH}_4 \) fluxes and lower NEE rates with increasing \( \Omega \text{water} \) (see Fig. 6). Within the NLR\( \text{CO}_2\)foot approach both surface types were denoted as sources of CO\(_2\), but with about 4-fold stronger rates of GPP, \( \text{R}_{\text{eco}} \) and NEE for emergent vegetation compared to open water (see Fig. 7 and Table 3). The approach yielded a similar cumulative annual NEE for the whole EC source area including both surface types as the LUT\( \text{CO}_2\)nofoot approach, but lower component fluxes (GPP and \( \text{R}_{\text{eco}} \)). As for CO\(_2\), we implemented \( \Omega \) as weighting factors within the NLR model for \( \text{CH}_4 \) (NLR\( \text{CH}_4\)foot) to get the surface type specific fluxes of \( \text{CH}_4 \) and fitted the parameters as follows:

\[
F_{\text{CH}_4} = \Omega_{\text{veg}} \cdot \exp(-10.076 + 0.415 \cdot T_{310}) + \Omega_{\text{water}} \cdot \exp(-6.449 + 0.286 \cdot T_{310}) 
\]

Open water accounted for more than 4-fold higher emissions than the vegetated areas (see Fig. 7 and Table 3). The NLR\( \text{CH}_4\)foot approach revealed a similar annual \( \text{CH}_4 \) budget as the NLR\( \text{CH}_4\)nofoot approach.

Annual budgets of CO\(_2\) (844 g CO\(_2\) m\(^{-2}\) a\(^{-1}\)) and \( \text{CH}_4 \) (22 g \( \text{CH}_4 \) m\(^{-2}\) a\(^{-1}\)) for the AOI differed strongly from the budgets for the EC source area due to the contrasting emission rates of open water and emergent vegetation (see Table 3) and different fractional coverages of the surface types within the AOI and the EC source area. This resulted in a higher C loss (246.5 g C m\(^{-2}\) a\(^{-1}\)) and a lower GWP (1452.9 g CO\(_2\)-Eq. m\(^{-2}\) a\(^{-1}\)) than for the EC source area. In the following we will primarily discuss the budgets of the EC source area and the surface types.
4 Discussion

4.1 Diurnal variability of CH\textsubscript{4} emissions

In terms of its daily cycle, CH\textsubscript{4} exchange between wetland ecosystems and the atmosphere is not
generalisable, but rather dependent on the spatial characteristics of the wetland and thus, the impact
of the individual CH\textsubscript{4} emission pathways (diffusion, ebullition, plant-mediated transport). Our
measurements showed a diurnal cycle of CH\textsubscript{4} exchange from June to September 2013 and in March
2014, with the strongest emissions during night, as reported for shallow lakes (e.g. Podgrajsek et al.
2014) and wetland sites with a considerable fraction of open water (e.g. Godwin et al. 2013, Koebisch
et al. 2015). In comparison, wetland CH\textsubscript{4} emissions were also reported to show daily maxima at day-
time (e.g. Morrisey et al. 1993, Hendriks et al. 2010, Hatala Matthes et al. 2014), especially at sites with
high abundance of vascular plants. No diurnal pattern (e.g. Rinne et al. 2007, Forbrich et al. 2011,
Herbst et al. 2011) occurred especially at sites without large open water areas (Godwin et al. 2013).

We assume the process of convective mixing of the water column (e.g. Godwin et al. 2013, Poindexter
and Variano 2013, Podgrajsek et al. 2014, Sahlée et al. 2014, Koebisch et al. 2015) to be crucial for the
diurnal pattern of CH\textsubscript{4} emissions at our study site. This is indicated by the concurrent timing of
convective mixing and daily peak CH\textsubscript{4} emissions and a generally high fractional source area coverage
of the open water, which shows higher rates of CH\textsubscript{4} release than emergent vegetation. Furthermore,
closed chamber measurements likewise show night-time peak emissions on the shallow lake in
summer 2013 (Hoffmann et al. 2015). During the day, CH\textsubscript{4} is trapped in the lower (anoxic) layers of the
thermally stratified water column. Due to the heat release of the surface water to the atmosphere in
the night the surface water cools down, initiating convective mixing of the water column down to the
bottom. Diffusion is enhanced due to the buoyancy-induced turbulence, the associated increased gas
transfer velocity at the air-water interface (Eugster et al. 2003, MacIntyre et al. 2010, Podgrajsek et al.
2014) as well as the transport of CH\textsubscript{4} enriched bottom water to the surface (Godwin et al. 2013,
Podgrajsek et al. 2014). In addition, ebullition can be triggered by turbulence due to convective mixing
(Podgrajsek et al. 2014, Read et al. 2012). The daily pattern of the open water CH\textsubscript{4} release might
superimpose the reverse diurnal cycle of plant-mediated transport with peak emissions during day-
time, as the release of methane is dependent on the stomatal conductance of the plants (e.g. Morrisey
et al. 1993). This pathway is limited to plants with aerenchymatic tissue like \textit{Typha latifolia}, which
dominates the eulittoral zone at our study site. CH\textsubscript{4} is transported from the soil to the atmosphere,
bypassing potential oxidation zones above the rhizosphere (chimney effect). Unusually for wetland
plants (Torn and Chapin 1993), complete stomatal closure during night was observed for *Typha latifolia* (Chanton et al. 1993).

### 4.2 Annual CH$_4$ emissions

The CH$_4$ emissions of our studied ecosystem were within the range of other temperate fen sites rewetted for several years (up to 63 g CH$_4$ m$^{-2}$ a$^{-1}$; e.g. Hendriks et al. 2007, Wilson et al. 2008, Günther et al. 2013, Schrier-Uijl et al. 2014). This rate corresponds to twice the emission factor of 21.6 g CH$_4$ m$^{-2}$ a$^{-1}$, that was assigned to rewetted temperate rich organic soils, which is in turn more than twice the rate of the nutrient-poor complement (IPCC 2014). In contrast, newly rewetted fens emit its multiple. In the first year after flooding, Hahn et al. (2015) observed an average net release of 260 g CH$_4$ m$^{-2}$ a$^{-1}$, which is 186 times higher than before flooding, at a fen site in NE Germany. Two years later the CH$_4$ emissions were significantly lower (40 g CH$_4$ m$^{-2}$ per growing season; Koebsch et al. 2015). However, natural fens release most often less CH$_4$ than the majority of rewetted fens (e.g. Bubier et al. 1993, Nilsson et al. 2001), with some exceptions (e.g. Huttunen et al. 2003).

The two main surface types open water and emergent vegetation differed substantially in their CH$_4$ exchange rates. Open water contributed overproportionally to the measured ecosystem fluxes and showed higher CH$_4$ release rates (52.6 g CH$_4$ m$^{-2}$ a$^{-1}$) than the emergent vegetation stands (13.2 g CH$_4$ m$^{-2}$ a$^{-1}$). However, closed-chamber measurements at the shallow lake show an even higher long-term average annual CH$_4$ release rate (206 g CH$_4$ m$^{-2}$ a$^{-1}$) since rewetting with large interannual variability and occasionally extreme high release rates (up to 400 g CH$_4$ m$^{-2}$ a 1; Casares et al., in prep.).

We assume the permanent high inundation and high productivity due to eutrophic conditions, feeding the organic mud deposited at the bottom of the open water body (which is typically for shallow lakes in rewetted fens), to be of particular importance for high CH$_4$ emissions as substrate for decomposition. The mud initially evolved as a mixture of sand and easily decomposable labile plant litter from reed canary grass, which died-off after flooding and produced a large C pool for CH$_4$ production (Hahn-Schöffl et al 2011). During an incubation experiment with substrate from our study site Hahn-Schöffl et al. (2011) observed that the new sediment layer has very high specific rates of anaerobic CH$_4$ (and CO$_2$) production. In addition, Zak et al. (2015) emphasised the impact of litter quality and reported a very high CH$_4$ production potential for litter of *Ceratophyllum demersum*, which dominates the biomass in the open water at our study site. Due to the eutrophic character of the lake and associated high productivity within the open water body and in the eulittoral zone, high amounts of fresh labile organic matter continuously replenish the mud layer and thus the C pool. As the C balance (CO$_2$ and CH$_4$) seems to be extremely unbalanced, we further assume lateral input of
allochthonous organic matter into the NE “bay” of the shallow lake, which is the area with the peak contribution of our EC derived fluxes, especially during strong winds. The importance of fresh labile organic matter provided by the die-back of the former vegetation as driving force for high CH₄ emissions was also discussed in Hahn et al. (2015). They measured the highest CH₄ emissions in sedge stands suffering from strongest die-back. For comparison annual budgets of CO₂ and CH₄ for other nutrient-rich lentic freshwater ecosystems in terms of pristine, anthropogenically influenced and transient ecosystems are listed in Table 4. Studies on nutrient-rich lakes generally revealed lower CH₄ release for open water. In contrast, beaver ponds were partially reported to emit similar rates of CH₄. Similarly to our study site beaver ponds are at least in the beginning disbalanced ecosystems due to a rapidly increased water level with associated suffering and finally the die-back of former vegetation, which is not adapted to higher water levels. A large C pool for CH₄ production develops. However, even for a beaver pond existing more than 30 years CH₄ emissions still account for 40 g CH₄ m⁻² a⁻¹ (Yavitt et al. 1992).

Annual CH₄ emissions of the surface type emergent vegetation were about 4-fold lower than for open water. This might be the result of increased CH₄ oxidation in the soil, as plants with aerenchymatic tissue release oxygen into the rhizosphere, in reverse to the emission of CH₄ into the atmosphere (Bhullar et al. 2013). Minke et al. (2015) highlight the difference in net CH₄ release for typical helophyte stands with moderate emissions for Typha dominated sites. Besides the effect of the gas transport within plants, lower water and sediment temperatures due to shading by the emergent vegetation might yield lower CH₄ production than for open water. Furthermore, the soil of emergent vegetation stands is generally only temporarily and partly inundated and the water table decreased additionally during the unusual warm and dry summer 2013, probably resulting in a lower rate of anaerobic decomposition to CH₄. This in turn might be a reason, that in comparison to other sites dominated by Typha (rewetted wetlands, lake shores and freshwater marshes; see Table 4) the emergent vegetation at our site is at the lower limit of reported CH₄ release rates and best comparable to closed chamber measurements of Typha latifolia microsites at another rewetted fen site in NE Germany ( Günther et al. 2015).

4.3 Annual net CO₂ release

We observed high annual net release of CO₂ during the observation period, which is rather uncommon for fens several years after rewetting (e.g. Hendriks et al. 2007, Schrier-Uijl et al. 2014, Knox et al. 2015). Surprisingly, net CO₂ budgets were similar to those of drained and degraded peatlands (e.g. Hatala et al. 2012, Schrier-Uijl et al. 2014). Both surface types acted as net sources, with emergent
vegetation (750 g CO$_2$ m$^{-2}$ a$^{-1}$) showing a distinctively higher net budget as well as GPP and $R_{eco}$ rates than open water (158 g CO$_2$ m$^{-2}$ a$^{-1}$). Only few NEE rates are published for the open water body of eutrophic shallow lakes. Ducharme-Riel et al. (2015) report 224 g CO$_2$ m$^{-2}$ a$^{-1}$ as annual NEE of a eutrophic lake in Canada (see Table 4). According to Kortelainen et al. (2006) Finnish lakes, which are mainly small and shallow, continuously emit CO$_2$ during the ice-free period, positively correlated with their trophic state.

Our study revealed a high annual net CO$_2$ release for emergent vegetation, which is in the wide range of NEE rates for *Typha* sites reported in other studies, including both net CO$_2$ sources and sinks (see Table 5). GPP and $R_{eco}$ are generally high (especially at rewetted fen sites; both component fluxes most often $>$ 3000 g CO$_2$ m$^{-2}$ a$^{-1}$), characterising *Typha* stands as high turnover sites, usually resulting in net CO$_2$ uptake. In contrast, $R_{eco}$ and GPP rates at our study site are in the lower part of the reported range. We assume the continuously high $R_{eco}$ rates during winter 2013/2014, contributing to the high annual net CO$_2$ emissions, to be the result of mild and dry meteorological conditions. In summer 2013, $R_{eco}$ exceeded GPP already in late June, indicating a significant contribution of heterotrophic respiration to the CO$_2$ production. We cannot completely exclude a misestimation of the CO$_2$ exchange during midsummer due to longer data gaps. However, unusual warm and dry conditions and associated water table lowering during summer 2013 might have triggered a shift from anaerobic to aerobic decomposition. This includes the exposed organic mud at former shallowly inundated soil of emergent vegetation stands, e.g. at the edge of the lake. Besides CH$_4$, Hahn-Schöffl et al. (2011) showed that the new sediment layer at the bottom of inundated areas exhibits very high rates of anaerobic CO$_2$ production. The effect of water table lowering at *Typha* sites due to dry conditions is also shown by Günther et al. (2015) and Chu et al. (2015): relative increase of $R_{eco}$ rates, resulting in net CO$_2$ release. This might be of special interest in terms of climate change, as a temperature increase and significantly less precipitation in summer are expected for NE Germany. In addition, a considerable increase of microbial activity and thus, generally increased decomposition due to high temperatures might be of importance. Allochthonous organic matter import into the NE bay due to lateral transport, as discussed for CH$_4$, might have further enhanced decomposition (e.g. Chu et al. 2015).

### 4.4 Global warming potential and the impact of spatial heterogeneity

The lake ecosystem is characterised by a high GWP 9 years after rewetting, mainly driven by high CH$_4$ emissions. Based on our results the site can hardly be classified into any phase following peatland rewetting discussed by Augustin and Joosten (2007). The slow development and shift of the ecosystem to a C sink with reduced climate impact might be the result of the exceptional characteristics
represented by eutrophic conditions and lateral transport of organic matter within the open water body. The trophic status of water and sediment is an important factor regulating GHG emissions, as shown by Schrier-Uijl et al. (2011) for lakes and drainage ditches in wetlands. However, the unusual meteorological conditions during our study period might have caused a comparable low GWP compared to previous years due to lower CH$_4$ emissions at the expense of high net CO$_2$ release. In comparison, e.g. Schrier-Uijl et al. (2014) report C uptake and a GHG sink function of a fen 10 years after rewetting with water levels below or at the soil surface. In a study by Knox et al. (2015) a wetland with mean water level above the soil surface was characterised by a near-neutral climate impact after 15 years of rewetting, where continued high CH$_4$ emissions were compensated by strong net CO$_2$ uptake. In the course of rewetting the water table is recommended to be held at or just below the soil surface to prevent inundation and thus, the formation of organic mud (Couwenberg et al. 2011, Joosten et al. 2012, Zak et al. 2015).

We calculated the “true” fluxes of CO$_2$ and CH$_4$ for the AOI by weighting the non-linear regression functions for the two surface types with their fractional coverage inside the AOI. The inferred C budget and global warming potential differs considerably from that of the EC source area, highlighting the strikingly different emission rates of open water versus emergent vegetation. Thus, footprint analysis providing the fractional coverage of the main surface types is imperative for the interpretation of ecosystem flux measurements as provided by the EC technique at such a spatially heterogeneous site. In addition, for an interannual comparison of EC derived budgets for such sites it is necessary to define a fixed AOI, as the cumulative footprint climatology (representing the EC source area) changes interannually. Inter-site comparisons (e.g. with other shallow lakes evolved during fen rewetting) are challenging with regard to the site-specific spatial heterogeneity.

5 Conclusions

This study contributes to the understanding of eutrophic shallow lakes as a challenging ecosystem often evolving during fen rewetting in NE Germany. Within the study period the ecosystem was a strong source of CH$_4$ and CO$_2$. Both open water and emergent vegetation, particularly including the eulittoral zone, were net emitters of CH$_4$ and CO$_2$, but with strikingly different release rates. This illustrates the importance of footprint analysis for the interpretation of the EC measurements on a rewetted site with distinct spatial heterogeneity. Our results show that the intended effects of rewetting in terms of CO$_2$ emission reduction and C sink recovery are not yet achieved at this site. The negative climate impact of the lake is dominated by considerable CH$_4$ release, particularly from the open water section. In combination with the high net CO$_2$ release the C budget seems to be extremely
unbalanced. Measurements of lateral transport of organic substrate within the open water body and a full C budget could give indication on a potential allochthonous input into the NE bay. Furthermore, the effect of unusual meteorological conditions need further investigation. A comparison with existing chamber measurements at the open water body for the same time period will be helpful for the evaluation of our measurements and estimation for the surface type fluxes. The site is continuously changing, with *Typha latifolia* progressively entering the open water body in the course of terrestrialisation, probably resulting in peat formation and C uptake once the shallow lake is replenished by organic sediments. Therefore, long-term measurements are necessary to evaluate the impact of future ecosystem development on GHG emissions. Moreover, statements for the climate impact of rewetted fens can only be provided by inclusion of additional sites varying in groundwater table and vegetation type. We assume that shallow lakes represent a special case with regard to the GHG dynamics and climate impact, with exceptionally high CH4 release and occasionally high net CO2 emissions. Inundation involves the risk of unpredictable and long-term high CH4 emissions, especially in case of nutrient-rich conditions, that counteract the actually intended lowering of the climate impact of drained and degraded fens. We strongly recommend to consider this risk in future rewetting projects and support the call of Lamers et al. (2015) for the need of well-conceived restoration management instead of the trial-and-error approach, whereon restoration of wetland ecosystem services was based on for a long time.

Acknowledgements

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References


Table 1: Data loss and final data coverage during the observation period. Percentage of CO$_2$ and CH$_4$ flux data lost by power and instrument failure and maintenance as well as quality control and footprint analysis.

<table>
<thead>
<tr>
<th>Filter criteria</th>
<th>Percentage of data [%]</th>
<th>CO$_2$</th>
<th>CH$_4$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Power and instrument failure, maintenance</td>
<td>15.0</td>
<td>46.4</td>
<td></td>
</tr>
<tr>
<td>Absence of sensor</td>
<td>-</td>
<td>11.2</td>
<td></td>
</tr>
<tr>
<td>QC 2</td>
<td>7.5</td>
<td>2.0</td>
<td></td>
</tr>
<tr>
<td>RSSI</td>
<td>-</td>
<td>2.1</td>
<td></td>
</tr>
<tr>
<td>u*</td>
<td>18.6</td>
<td>8.8</td>
<td></td>
</tr>
<tr>
<td>Unreasonably high fluxes</td>
<td>0.2</td>
<td>0.1</td>
<td></td>
</tr>
<tr>
<td>No footprint information/footprint &gt; 20 % outside the AOI</td>
<td>13.2</td>
<td>6.5</td>
<td></td>
</tr>
<tr>
<td>Final data coverage</td>
<td>45.5</td>
<td>22.9</td>
<td></td>
</tr>
</tbody>
</table>
Table 2: Gapfilling model performance was estimated according to Moffat et al. (2007) with several measures ($n_{CO_2} = 6193$, $n_{CH_4} = 3386$, fluxes of best quality QC 0): the adjusted coefficient of determination $R^2_{adj}$ for phase correlation (significant in all cases, p-value < 2.2e-16), the absolute root mean square index (RMSE$_{abs}$) and the mean absolute error (MAE) for the magnitude and distribution of individual errors, as well as the bias error (BE) for the bias of the annual sums.

<table>
<thead>
<tr>
<th>Method</th>
<th>$R^2_{adj}$</th>
<th>RMSE$_{abs}$ (mg m$^{-2}$ 30min$^{-1}$)</th>
<th>MAE (mg m$^{-2}$ 30min$^{-1}$)</th>
<th>BE (g m$^{-2}$ a$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>LUT$_CO_2$nofoot</td>
<td>0.74</td>
<td>104.35</td>
<td>24.05</td>
<td>13.14</td>
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<tr>
<td>NLR$_CO_2$foot</td>
<td>0.66</td>
<td>119.10</td>
<td>27.51</td>
<td>-2.12</td>
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<tr>
<td>NLR$_CH_4$nofoot</td>
<td>0.79</td>
<td>1.36</td>
<td>0.83</td>
<td>-3.34</td>
</tr>
<tr>
<td>NLR$_CH_4$foot</td>
<td>0.81</td>
<td>1.28</td>
<td>0.78</td>
<td>-2.54</td>
</tr>
</tbody>
</table>
Table 3: Annual balances of CO$_2$ and CH$_4$ derived by different methods for the whole EC source area, the area of interest (AOI) and the two surface types: LUT approach without footprint consideration (LUT$_{CO2nofoot}$), NLR approach without (NLR$_{CH4nofoot}$) and with (NLR$_{CH4foot}$, NLR$_{CO2foot}$) footprint consideration. Uncertainty was calculated as square root of the sum of squared random uncertainty (measurement uncertainty) and gapfilling uncertainty.

<table>
<thead>
<tr>
<th>Source area</th>
<th>Flux (g m$^{-2}$ a$^{-1}$)</th>
<th>Method</th>
<th>CO$_2$</th>
<th>CH$_4$</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>CO$_2$foot</td>
<td>LUTCO2nofoot</td>
<td>NLR$_{CO2foot}$</td>
</tr>
<tr>
<td>Whole EC source area</td>
<td>NEE</td>
<td>524.5 ± 5.6</td>
<td>531.4 ± 13.0</td>
<td></td>
</tr>
<tr>
<td></td>
<td>GPP</td>
<td>-2380.5 ± 5.6</td>
<td>-2122.1 ± 16.7</td>
<td></td>
</tr>
<tr>
<td></td>
<td>$R_{eco}$</td>
<td>2863.6 ± 5.6</td>
<td>2603.6 ± 8.4</td>
<td>40.5 ± 0.2</td>
</tr>
<tr>
<td>AOI</td>
<td>NEE</td>
<td>843.5 ± 13.0</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>GPP</td>
<td>-3192.2 ± 16.7</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>$R_{eco}$</td>
<td>4035.7 ± 8.4</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Emergent vegetation</td>
<td>NEE</td>
<td>750.3 ± 13.0</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>GPP</td>
<td>-4076.8 ± 16.7</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>$R_{eco}$</td>
<td>4827.2 ± 8.4</td>
<td></td>
<td></td>
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<tr>
<td>Open water</td>
<td>NEE</td>
<td>158.2 ± 13.0</td>
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<tr>
<td></td>
<td>GPP</td>
<td>-1021.5 ± 16.7</td>
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<tr>
<td></td>
<td>$R_{eco}$</td>
<td>1179.7 ± 8.4</td>
<td></td>
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<tr>
<td></td>
<td>CH$_4$</td>
<td>52.6 ± 0.2</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Table 4: NEE and net CH₄ exchange at open water sites. The letters in parentheses indicate seasonal (S; May to October) and annual (A) budgets. Positive water level indicates inundated conditions. GHG flux measurement methods are denoted as: CH = chambers, CO = concentration profiles, TR = gas traps.

<table>
<thead>
<tr>
<th>Reference</th>
<th>Location, ecosystem type</th>
<th>Dominant plant species</th>
<th>Study year</th>
<th>Average water depth (m)</th>
<th>NEE CH₄ (g CH₄ m⁻² a⁻¹)</th>
<th>CH₄ (g CO₂ m⁻² a⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Huttunen et al. (2003), CH</td>
<td>Lake Positampi, Finland: hypertrophic lake</td>
<td>Utricularia spp., Potamogeton spp.</td>
<td>1997</td>
<td>3.2</td>
<td>16 (A)</td>
<td>13 (A)</td>
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<tr>
<td>Ducharme-Beil et al. (2015), CO</td>
<td>Brand-de-Sie, Quebec: hypertrophic lake</td>
<td></td>
<td>2007-2008</td>
<td>3.2</td>
<td>224 (A)</td>
<td></td>
</tr>
<tr>
<td>Wang et al. (2006), CH</td>
<td>Beijing, China: hypertrophic lake</td>
<td></td>
<td>2003-2004</td>
<td>0.5 to 1.8</td>
<td>3 (A)</td>
<td>4 (A)</td>
</tr>
<tr>
<td>Hendriks et al. (2007), CH</td>
<td>Horstermeer, The Netherlands: eutrophic ditches</td>
<td></td>
<td>2004-2006</td>
<td>&gt; 0</td>
<td>&gt; 0</td>
<td>&gt; 0</td>
</tr>
<tr>
<td>Waddington and Day (2007), CH</td>
<td>Bois-des-Beauvais, Quebec: ponds</td>
<td></td>
<td>2000-2002</td>
<td>&gt; 0</td>
<td>&gt; 0</td>
<td>&gt; 0</td>
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<td>Naiman et al. (1991), CH</td>
<td>Kabetogama Peninsula, Minnesota: submergent aquatic plants</td>
<td></td>
<td>1988</td>
<td>0.2 to 0.4</td>
<td>14 (A)</td>
<td>2.9 (S)</td>
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<tr>
<td>Roulet et al. (1992), CH</td>
<td>Low forest region, Ontario: beaver ponds</td>
<td></td>
<td>1990</td>
<td>≤ 2</td>
<td>4 (A)</td>
<td>0.3 (S)</td>
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<td>Yavitt et al. (1992), CH</td>
<td>Clay Belt, Ontario: beaver ponds</td>
<td></td>
<td>1991</td>
<td>0.5 to 1.5</td>
<td>44 (A)</td>
<td></td>
</tr>
<tr>
<td>Bubier et al. (1993), CH</td>
<td>Gay Belt, Ontario: beaver ponds</td>
<td></td>
<td>1990</td>
<td>≤ 2</td>
<td>34 (A)</td>
<td>20 (A)</td>
</tr>
<tr>
<td>Yavitt et al. (1992), CH</td>
<td>Clay Belt, Ontario: beaver ponds</td>
<td></td>
<td>1990</td>
<td>≤ 2</td>
<td>34 (A)</td>
<td>20 (A)</td>
</tr>
</tbody>
</table>
### Table 5: Annual (A)/seasonal (S) NEE, GPP, R eco and net CH$_4$ exchange at Typha sites. Positive water level indicates inundated soil. GHG flux measurement methods are denoted as: CH = chambers, EC = eddy covariance.

<table>
<thead>
<tr>
<th>Reference</th>
<th>Domain, ecosystem type</th>
<th>Dominant plant species</th>
<th>Study year</th>
<th>Mean water level (m)</th>
<th>NEE (g CO$_2$ m$^{-2}$ a$^{-1}$)</th>
<th>GPP (g CO$_2$ m$^{-2}$ a$^{-1}$)</th>
<th>R eco (g CO$_2$ m$^{-2}$ a$^{-1}$)</th>
<th>CH$_4$ (g CH$_4$ m$^{-2}$ a$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Kankaala et al. (2004), CH</td>
<td>Lake Vesijärvi, Finland: - inner cattail-reed zone</td>
<td>Phragmites australis, Typha latifolia</td>
<td>1997</td>
<td>&lt; 0.1 to &gt; 0.2</td>
<td>51 (S)</td>
<td>43 (S), 6 (SF)</td>
<td>30 (S)</td>
<td>23 (S), 7 (SF)</td>
</tr>
<tr>
<td></td>
<td>- outer cattail-reed zone</td>
<td>Phragmites australis, Typha latifolia</td>
<td>1998</td>
<td>&lt; 0.1 to &gt; 0.2</td>
<td>23 (SF)</td>
<td>21 (SF)</td>
<td>7 (SF)</td>
<td></td>
</tr>
<tr>
<td>Chu et al. (2015), EC</td>
<td>Lake Erie, Freshwater marsh</td>
<td>Typha angustifolia, Nymphaea odorata</td>
<td>2011-2012</td>
<td>0.3 to 0.6</td>
<td>-289 (A)</td>
<td>-33.38 (A)</td>
<td>3049 (A)</td>
<td>58 (A)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>2013</td>
<td>0.3 to 0.6</td>
<td>109 (A)</td>
<td>34.90 (A)</td>
<td>3599 (A)</td>
<td>76 (A)</td>
</tr>
<tr>
<td>Bonneville et al. (2008), EC</td>
<td>Mer Bleue, Canada, freshwater marsh</td>
<td>Typha angustifolia</td>
<td>2005-2009</td>
<td>winter + summer</td>
<td>0.37</td>
<td>51 (S)</td>
<td>84 (S)</td>
<td>30 (S)</td>
</tr>
<tr>
<td>Strachan et al. (2015), NEE: EC, CH4: CH</td>
<td>Virginia, freshwater marsh</td>
<td>Typha latifolia</td>
<td>1992-1993</td>
<td>0.5 to 0.2</td>
<td>-288 (A)</td>
<td>-3.7 (A)</td>
<td>30 (A)</td>
<td>21 (A)</td>
</tr>
<tr>
<td>Whiting and Chanton (2001), CH</td>
<td>Florida, lake shore</td>
<td>Typha latifolia</td>
<td>1992-1993</td>
<td>0.5 to 0.2</td>
<td>-3568 (A)</td>
<td>-26.66 (A)</td>
<td>3006 (A)</td>
<td>70 (A)</td>
</tr>
<tr>
<td>Rocha and Goulden (2008), EC</td>
<td>San Joaquin Freshwater Marsh Reserve, California: - freshwater marsh</td>
<td>Typha latifolia</td>
<td>1999</td>
<td>winter +, midsummer</td>
<td>-967 (A)</td>
<td>-30.45 (A)</td>
<td>2078 (A)</td>
<td>170 (A)</td>
</tr>
<tr>
<td>Knox et al. (2015), EC</td>
<td>- wetland (rewetted 2010)</td>
<td>Schoenoplectus acutus, Typha spp.</td>
<td>2012</td>
<td>0.26</td>
<td>-1455 (A)</td>
<td>-55.19 (A)</td>
<td>4064 (A)</td>
<td>52 (A)</td>
</tr>
<tr>
<td></td>
<td>- wetland (rewetted 1997)</td>
<td>Schoenoplectus acutus, Typha spp.</td>
<td>2011-2012</td>
<td>0.51</td>
<td>388 (A)</td>
<td>-21 (A)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Petrescu et al. (2015), EC</td>
<td>- wetland (rewetted 2010)</td>
<td>Typha latifolia, Hydrocharis morsus-ranae</td>
<td>2010-2011</td>
<td>1</td>
<td>553 (A)</td>
<td>-28.25 (A)</td>
<td>3375 (A)</td>
<td>80 (A)</td>
</tr>
<tr>
<td></td>
<td>- wetland (rewetted 1985)</td>
<td>Typha latifolia, Hydrocharis morsus-ranae</td>
<td>2011-2012</td>
<td>0.7</td>
<td>-414 (A)</td>
<td>-39.80 (A)</td>
<td>3566 (A)</td>
<td>91 (A)</td>
</tr>
<tr>
<td>Minke et al. (2015), CH</td>
<td>Giel'čykaŭ Kašyl, Belarus, fen (rewetted 1985)</td>
<td>Typha latifolia</td>
<td>2011</td>
<td>0.02</td>
<td>-156 (A)</td>
<td>-13 (A)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Günther et al. (2015), CH</td>
<td>Trebeltal, Germany, fen (rewetted 1997)</td>
<td>Typha latifolia</td>
<td>2012</td>
<td>-0.09</td>
<td>545 (A)</td>
<td>4 (A)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wilson et al. (2007, 2008), CH</td>
<td>Turrum, Ireland, cutaway bog (rewetted 1991)</td>
<td>Typha latifolia</td>
<td>2002-2003</td>
<td>0.07</td>
<td>975 (A)</td>
<td>-32.72 (A)</td>
<td>4064 (A)</td>
<td>39 (A)</td>
</tr>
</tbody>
</table>

1 open water period
2 winter
Figure 1: a) Polder Zarnekow is situated in NE Germany within the Peene River valley; map source and copyright: https://commons.wikimedia.org/wiki/File:Germanymap2.png (modified). b) Footprint climatology calculated according to Chen et al. (2011) on a Landsat image (6 Jun 2013, source: Google Earth). White lines represent the isopleths of the cumulative annual footprint climatology, where the area within the 95 isopleth indicates 95 % contribution to the annual flux. The white dot denotes the tower position. The yellow box indicates the area of interest (AOI) as a filter criterion to focus on fluxes of the shallow lake and to avoid the possible impact of a farm and grassland to the north of the shallow lake. If the half-hourly flux source area exceeded the AOI by more than 20 % the flux was discarded. The site is characterised by two main surface types: open water and emergent vegetation.
Figure 2: Temporal variability of environmental variables and ecosystem CO$_2$ and CH$_4$ exchange. Seasonal course a) of water level (Wlevel), cumulative precipitation (Cum. Precip) and air temperature ($T_{air}$), b) the daily CH$_4$ flux (gapfilled, NLR$_{CH4\text{nofoot}}$) and c) the daily NEE (gapfilled LUT$_{CO2\text{nofoot}}$) and component fluxes (modelled $R_{eco}$ and GPP, LUT$_{CO2\text{nofoot}}$).
Figure 3: Average diurnal cycle of a) CO$_2$ flux, b) CH$_4$ flux and c) the water density gradient per month. The numbers at the x-axis denote midnight (0) and midday (12). Midnight is also illustrated with a dashed line. Black and grey lines represent the mean and the range, respectively. The CH$_4$ fluxes are normalized with the monthly median of the half-hourly fluxes. Positive CO$_2$ fluxes represent the dominance of R$_{eco}$ against GPP, negative fluxes the dominance of GPP against R$_{eco}$. The period of ice-cover was excluded from the calculation of the temperature gradient. A density gradient equal to or below zero indicates thermally induced convective mixing down to the bottom of the open water body of the shallow lake, positive gradients instead thermal stratification.
Figure 4: Cumulative GWP budgets of CO$_2$ (based on LUT$_{CO2\text{nofoot}}$), CH$_4$ (based on NLR$_{CH4\text{nofoot}}$) for the EC source area and the sum of both during the observation period.
Figure 5: Source area fraction $\Omega_i$ of the two main surface types in dependence on the wind direction (2°-bins).
Figure 6: Impact of the fractional coverage of open water ($\Omega_{\text{water}}$) within the EC source area on the measured fluxes of CO$_2$ and CH$_4$. The variability of CO$_2$ flux rates decreased with increasing $\Omega_{\text{water}}$, whereas the variability of the CH$_4$ flux increased.
Figure 7: Daily CH$_4$, NEE and component fluxes (Reco and GPP) for the surface types: a) daily CH$_4$ flux of open water and emergent vegetation, b) daily NEE and component fluxes for open water, c) daily NEE and component fluxes for emergent vegetation, derived by NLR with the source area fractions of the surface types ($\Omega_i$) as weighting factors (NLR$_{CH4foot}$, NLR$_{CO2foot}$).