GLOBAL CHANGE BIOLOGY
BIOENERGY

GCB Bioenergy (2016) **8**, 969–980, doi: 10.1111/gcbb.12334

Carbon balance of rewetted and drained peat soils used for biomass production: a mesocosm study

SANDHYA KARKI, LARS ELSGAARD, TANKA P. KANDEL and POUL ERIK LÆRKE
Department of Agroecology, Aarhus University, Blichers Allé 20, DK-8830, Tjele, Denmark

Abstract

Rewetting of drained peatlands has been recommended to reduce CO_2 emissions and to restore the carbon sink function of peatlands. Recently, the combination of rewetting and biomass production (paludiculture) has gained interest as a possible land use option in peatlands for obtaining such benefits of lower CO_2 emissions without losing agricultural land. This study quantified the carbon balance (CO_2, CH_4 and harvested biomass C) of rewetted and drained peat soils under intensively managed reed canary grass (RCG) cultivation. Mesocosms were maintained at five different groundwater levels (GWLs), that is 0, 10, 20 cm below the soil surface, representing rewetted peat soils, and 30 and 40 cm below the soil surface, representing drained peat soils. Net ecosystem exchange (NEE) of CO_2 and CH_4 emissions was measured during the growing period of RCG (May to September) using transparent and opaque closed chamber methods. The average dry biomass yield was significantly lower from rewetted peat soils (12 Mg ha^{-1}) than drained peat soils (15 Mg ha^{-1}). Also, CO_2 fluxes of gross primary production (GPP) and ecosystem respiration (ER) from rewetted peat soils were significantly lower than from drained peat soils, but net uptake of CO_2 was higher from rewetted peat soils. Cumulative CH_4 emissions were negligible (0.01 g CH_4 m^{-2}) from drained peat soils but were significantly higher (4.9 g CH_4 m^{-2}) from rewetted peat soils during measurement period (01 May–15 September 2013). The extrapolated annual C balance was 0.03 and 0.68 kg C m^{-2} from rewetted and drained peat soils, respectively, indicating that rewetting and paludiculture can reduce the loss of carbon from peatlands.

Keywords: ecosystem respiration, gross primary production, groundwater level, methane, net ecosystem exchange, paludiculture, reed canary grass

Received 14 July 2015; revised version received 4 December 2015 and accepted 11 December 2015

Introduction

Natural peatlands are important ecosystems in the global carbon cycle as they sequester and store atmospheric carbon for thousands of years (Gorham, 1991; Yu *et al.*, 2011). Water-logged conditions in natural peatlands facilitate accumulation of partially decomposed plant residues, which results in a steadily increasing reservoir of carbon. From an atmospheric perspective, natural peatlands are net sinks of carbon dioxide (CO_2), but net sources of methane (CH_4) (Frolking *et al.*, 2011). Drainage as a prerequisite for agricultural crop production reduces CH_4 emissions, but changes the peatland from a net sink to large source of CO_2 due to aerobic peat mineralization (Maljanen *et al.*, 2010). It is estimated that about 1 Pg year^{-1} of CO_2 is emitted from drained peatlands globally, which is equivalent to 10% of the CO_2 emissions from the entire agriculture, forestry and other land use sector (IPCC, 2014b). In recent years, rewetting of formerly drained peatlands has been a major focus for reducing CO_2 emissions and restoring the carbon sink function (Tanneberger & Wichtmann, 2011; Joosten *et al.*, 2012; Wilson *et al.*, 2013). In this context, rewetting of drained peatlands has also been included as a potential target for climate change mitigation in the Kyoto protocol (IPCC, 2014a).

To combine the advantages of rewetting and agricultural biomass production, paludiculture has been suggested as a promising option to reduce anthropogenic CO_2 emissions from peatlands (Wichtmann & Tanneberger, 2011; Joosten *et al.*, 2012). However, rewetting and paludiculture risks increasing emissions of CH_4 due to increased microbial CH_4 production and plant-mediated transport of CH_4 produced in the anaerobic soil environment (Tuittila *et al.*, 2000; Waddington & Day, 2007; Wilson *et al.*, 2009). Indeed, large and fluctuating CH_4 emissions have been measured after re-establishment of vegetation in rewetted peatlands notably in response to varying environmental factors such as precipitation, groundwater level (GWL) and temperature (Waddington & Day, 2007; Wilson *et al.*, 2009; Günther *et al.*, 2015; Vanselow-Algan *et al.*, 2015). Karki *et al.* (2014) also found an increase in CH_4 emissions in

Correspondence: Sandhya Karki, tel. +45 8715 4785, fax +45 8715 4798, e-mail: sandhya.karki@agro.au.dk

rewetted peat soil with reed canary grass (RCG; *Phalaris arundinacea* L.) grown as a bioenergy crop, but indicated that the increase in CH_4 emissions had lower impact on the global warming potential (GWP) than the net reduction in CO_2 emissions achieved by rewetting. However, the plant CO_2 uptake (gross primary production, GPP) was not measured by Karki *et al.* (2014) but was tentatively estimated from aboveground biomass yield assuming fixed ratios of root to shoot and GPP to NPP (net primary production). In general, there is lack of data on greenhouse gas emissions from paludiculture where biomass crops are established and harvested for bioenergy production (IPCC, 2014a).

This study was designed to quantify the net ecosystem exchange (NEE) of CO_2, the CH_4 emission, and the net ecosystem carbon balance (NECB) of peat soil managed at different GWL, where NECB also included carbon export with the harvested crop. The perennial grass RCG, which is suitable for cultivation in Nordic peatlands, was intensively managed with two times fertilization and two times harvesting within 1 year, as this may improve the quantity and quality of RCG biomass for biogas production (Kandel *et al.*, 2013d). NEE of CO_2 and CH_4 emissions was measured during the RCG growing season from May to September 2013, using transparent and opaque closed chambers applied to intact soil mesocosms with controlled GWL at 0, 10, 20, 30 and 40 cm below the soil surface. For an integrated data synthesis, comparison was made between management systems operationally representing rewetted (0–20 cm GWL) and drained (30–40 cm GWL) peatland cropping system.

Materials and methods

Study site and experimental set-up

Soil cores for mesocosm studies were collected from a fen peatland in the Nørre Å river valley, Denmark (56°44′N, 9°68′E). The peatland was drained to a depth of 60–70 cm, starting in the early 20th century, and has been used for agricultural purposes since then. The site was cultivated with spring barely continuously during the 1970s and early 1980s. In 1985, the cropping system was changed to a 4-year rotation where spring barley was undersown with grass. After harvesting of spring barley, grass remained in the field for 3 years. Until 2008, about 200 kg N ha^{-1} $year^{-1}$ was applied either as mineral fertilizer or cattle slurry. The site was cultivated with RCG since 2009 (Kandel *et al.*, 2013b). The upper peat layer (0–20 cm) was highly decomposed (H9 on von Post scale) with bulk density of 0.29 g m^{-3}, total organic carbon content of 37.8% and total nitrogen content of 3.2% (Karki *et al.*, 2014). Peat depth at the study site was >1 m.

Intact soil cores were collected and handled as described by Karki *et al.* (2014). Briefly, in May 2012, intact soil cores (n = 25) were collected by inserting PVC pipes (depth, 60 cm; diameter, 30 cm) into the soil. The soil cores were retrieved and transported to semi-field facilities at AU-Foulum (Karki *et al.*, 2014). The bottom of each PVC pipe was covered with nylon net to secure the soil, but allow for free water movement, and the pipes were installed in plastic cylinders (height, 70 cm; diameter, 37 cm). The cylinders were filled with gravel at the bottom 10 cm, and the space between the PVC pipes and cylinder walls was filled with sand. The whole set-up was then installed in a trench with the soil surface at ground level and exposed to natural fluctuations in temperature and precipitation. The areas between the cylinders were insulated with mineral wool fibre mats.

The 25 mesocosms were randomly divided into five groups. The GWL of each group was adjusted to 0, 10, 20, 30 and 40 cm below soil surface by fitting a plastic tube (diameter, 1 cm) to the base of the cylinders and adjusting the other end of the plastic tube at the height corresponding to the GWL treatment. Demineralized water was supplied to the sand-filled interspace between the PVC pipes and the cylinder walls every day for 1 h by a drip irrigation system.

Reed canary grass seeds were sown in all mesocosms in June 2012 after uprooting of the initial biomass. This was performed due to poor regrowth of RCG both under mesocosm and field conditions during spring 2012. RCG was harvested in October 2012 and after regrowth in spring 2013, each mesocosm was fertilized (30 April) with mineral NPK corresponding to 80 kg N ha^{-1}, 13 kg P ha^{-1} and 77 kg K ha^{-1}. During this study period from 01 May to 15 September 2013, RCG was harvested (first harvest) on 27 June and then fertilized again with same amount of fertilizer as applied in spring. Final harvesting (second harvest) was carried out on 12 September 2013.

Biomass measurement

Canopy development of biomass was monitored through the nondestructive measurement of ratio vegetation index (RVI) which combines information on amounts and photosynthetic activity of green biomass. Measurements were taken using a RapidScan CS-45 handheld crop sensor (Holland Scientific, Lincoln, NE, USA). The sensor measured the canopy reflectance at the red (670 nm) and infrared regions (780 nm) and RVI was calculated as the ratio of infrared to red reflectance (Christensen, 1992). The height of the sensors was adjusted to capture the light reflectance from each mesocosm. RVI measurements were taken at weekly to fortnightly intervals on the same days as CO_2 flux measurements. A value of 2.4 was deducted from RVI measurement as a background reflectance measured in bare soil conditions (Yuan *et al.*, 2016).

Dry weight of total aboveground biomass from each mesocosm was quantified after first and second harvest by oven-drying at 60°C to constant weight. Spontaneous vegetation (notably marsh foxtail, *Alopecurus geniculatus*) was obvious in the mesocosms, and species composition from each mesocosm was determined on dry weight basis after the first harvest.

Environmental parameters and dissolved organic carbon

Air temperature (2 m) and precipitation data were obtained from a climate station at AU-Foulum ca. 1 km from the semi-field facility. Further, one of the five mesocosms at each GWL treatment was instrumented with a soil temperature probe (5 cm depth), a piezometer pipe (length, 65 cm; diameter, 2 cm) and a time domain reflectometry (TDR) soil moisture probe (Thomsen *et al*., 2007). Soil moisture probes of different length were used: 5 cm for 0 cm GWL, 10 cm for 10 cm GWL and 20 cm for 20–40 cm GWL. Soil temperature was measured automatically every hour, whereas soil moisture and GWL (in piezometers) were measured manually at fortnightly intervals on every CH_4 gas sampling occasions. For practical reasons, CO_2 and CH_4 fluxes were measured from four of the five replicates at each GWL treatment, thus excluding the instrumented replicates (i.e. $n = 4$ for gas flux measurements).

For analysis of dissolved organic carbon (DOC), 20 mL of soil water was sampled monthly from the piezometers and filtered through a 0.45-μm nylon membrane filter (SNY 4525, Frisenette, Denmark). Concentrations of DOC were determined with a Shimadzu TOC-V CPH/CPN analyser (Shimadzu Corp., Kyoto, Japan). Before analysis, the water samples were acidified to pH 2–3 and degassed to remove any carbonates.

CO_2 fluxes

Fluxes of CO_2 were determined with a transparent Plexiglas chamber (Plexiglas XT; RIAS A/S, Roskilde, Denmark) of 50 cm height and outer diameter of 30 cm. The chamber had similar control systems as used by Elsgaard *et al.* (2012), but with some modifications such as the size (to fit directly to RCG mesocosms) and a cooling system with peltier elements (Karki, 2015). The chamber was equipped with a quantum sensor (LI-190SL; Li-Cor Inc., Lincoln, NE, USA) to record photosynthetically active radiation (PAR), a fan (Sunon KDE, 12V; Conrad Electronic, Hirschau, Germany) for continuous mixing of chamber air and a vent for pressure equilibration. Four temperature sensors (Pt100; RS Components A/S, Copenhagen, Denmark) were fitted to the chamber: three inside the chamber at different height and one outside. Two additional fans were attached to the cooling side of the peltier element (inside the chamber); these fans started automatically when the temperature difference between inside (average of three sensors) and outside temperature was greater than 0.2°C and continued until the difference was below 0.2°C. Concentrations of CO_2 and H_2O inside the chamber were measured in-line with a with a Li-840 infrared gas analyser (Li-Cor Inc.) which was connected to the chamber by inlet and outlet tubing (inner diameter, 4 mm). A Campbell datalogger (CR-850; Campbell Scientific, Logan, UT, USA) was used to log CO_2 and H_2O data as well as PAR and air temperature (inside and outside chamber) at 1-s intervals.

The NEE of CO_2 was measured by placing the chamber on collars that were permanently mounted on the mesocosms. Rubber seals were glued to the flange of the collars and the base of the chamber to ensure air tightness during measurement. Each measurement lasted for 2 min. Immediately after the flux measurement, the chamber was vented until the CO_2 concentration inside the chamber reached the ambient level. The chamber was then repositioned on the collar and ecosystem respiration (ER) was measured after blocking PAR completely with a white opaque cover. An extension of Plexiglas with same dimensions as the top chamber was used when the vegetation height exceeded the chamber height. The extension was equipped with a fan to ensure proper air mixing. CO_2 fluxes were measured between 10:00 and 14:00 at weekly to fortnightly intervals. The measuring order of the mesocosms was randomized on different measurement dates. The measurements were taken both on sunny and cloudy days covering a wide range of PAR conditions with photon fluxes of 300–1800 μmol m^{-2} s^{-1}.

Fluxes of CO_2 were calculated by linear or exponential regression using an updated version of the MATLAB routine by Kutzbach *et al.* (2007) applying a water vapour correction algorithm. Before the flux calculation, each CO_2 concentration curve was visually inspected for irregularities. Occasionally such irregularities were observed during NEE measurements due to sudden changes in PAR caused by clouds. If possible, a flux was calculated from the remaining part of the flux curve (minimum 40 data points) after discarding the irregular part. Fluxes were accepted only if they passed the quality control criteria as specified by Elsgaard *et al.* (2012). More than 90% of the fluxes passed the criteria.

CH_4 fluxes

Methane flux measurements were taken with dark chambers equipped with a fan and a vent, and having similar dimensions as the transparent chamber. Measurements were carried out between 10:00 and 12:00 at fortnightly intervals. Four gas samples (10 mL) were drawn at regular interval from the chamber headspace with polypropylene syringes during a chamber deployment time of 45 min. The gas samples were transferred to evacuated 6-mL Exetainers. Gas samples were analysed by FID on an Agilent 7890 gas chromatograph with a CTC Combi-PAL automatic sample injection system (Agilent, Nærum, Denmark). Fluxes ($n = 250$) were calculated using the HMR method (Pedersen *et al*., 2010). The HMR method applies a nonlinear model if possible and otherwise suggests a linear model or no flux. The nonlinear model is a regression-based extension of the Hutchinson and Mosier model (Hutchinson & Mosier, 1981). For the present data, 37% of the CH_4 fluxes were analysed by nonlinear regression and 63% were analysed by linear regression according to the best model fit.

Cumulative CO_2 and CH_4 fluxes

Cumulative fluxes of CO_2 were derived from modelling of GPP and ER separately. GPP was estimated from consecutive flux measurements of NEE and ER as GPP = NEE-ER (where GPP and ER are defined as negative and positive entities, respectively). GPP for the whole growing season was modelled in one step for each GWL treatment according to a rectangular hyperbolic saturation curve (Thornley & Johnson, 1990) extended with RVI as a proxy of foliar biomass (Kandel *et al*., 2013a):

$$GPP = \frac{\beta \times RVI \times \alpha \times PAR}{(\beta \times RVI) + (\alpha \times PAR)} \quad (1)$$

where β is a scaling parameter of RVI and α is the apparent quantum yield (μg CO_2 per μmol photon). The model parameters were estimated by nonlinear regression in SIGMAPLOT 11 (Systat Software, Chicago, IL, USA). The estimated model parameters were used to extrapolate GPP for the entire growing season using continuous hourly PAR measurements from the climate station and linearly interpolated RVI data.

ER was modelled with a response function of temperature and biomass based on Mäkiranta *et al.* (2010) but using RVI in place of leaf area index:

$$ER = R_{10} \times \exp\left(E_0\left(\frac{1}{10 - T_0} - \frac{1}{T - T_0}\right)\right) + (\beta \times RVI) \quad (2)$$

where R_{10} is reference respiration at 10°C, E_0 is the ecosystem sensitivity coefficient, T_0 is a theoretical zero respiration temperature, here fixed to −46.02°C (Lloyd & Taylor, 1994), T is the average of air and soil temperature at 5 cm depth (°C), and β is a scaling parameter of RVI. Cumulative ER for the entire growing season was extrapolated for each GWL treatment using a similar approach as for GPP but based on hourly temperature and linearly interpolated RVI data. As for the derivation of model parameters, T in the extrapolations was the average of air and soil temperature (5 cm depth).

Cumulative NEE of CO_2 was obtained as the sum of cumulative emissions of GPP and ER.

Uncertainty in modelling of cumulative GPP and ER fluxes was derived from the maximum and minimum fluxes based on standard errors associated with each of the model parameters as specified by Kandel *et al.* (2013b). This approach produced a slightly asymmetrical interval around the mean cumulative estimates, and the largest deviation from the mean was used as resulting standard error estimate. Uncertainty in cumulative NEE was calculated by propagating the errors of cumulative GPP and ER.

Nash–Sutcliffe modelling efficiencies (ME) of GPP and ER fluxes were calculated as:

$$ME = 1 - \frac{\sum(Mes_i - Mod_i)^2}{\sum(Mes_i - \overline{Mes})^2} \quad (3)$$

where Mes_i and Mod_i are measured and modelled values, respectively, and $\overline{Mes}$ is the mean of measured values (Nash & Sutcliffe, 1970).

Cumulative CH_4 fluxes were calculated by linear interpolation over the sampling dates (Petersen *et al.*, 2012). Cumulative CH_4 fluxes were calculated for each individual mesocosm and then averaged (n = 4) for each GWL treatment.

NECB at each GWL was calculated by summing the NEE of CO_2-C, CH_4-C and harvested biomass C. For evaluation in units of GWP (CO_2 equivalents), CH_4 emissions were converted to CO_2 equivalents by multiplying with 28 (Myhre *et al.*, 2013).

Estimation of annual fluxes

The present monitoring period was the RCG growing period from May to September 2013. Annual GPP and ER were estimated using the presently derived model parameters and hourly PAR, RVI and temperature data from 16 September 2012 to 15 September 2013, where environmental data from September 2012 to May 2013 were taken from Karki *et al.* (2014) who studied greenhouse gas emissions from rewetted RCG mesocosms. In the study by Karki *et al.* (2014), RVI was measured with a SpectroSense2+ fitted with SKR1800 sensors (Skye Instruments, Powys, UK). RVI values obtained by RapidScan (present technique) and SpectroSense2+ in May and June 2013 were linearly correlated (r = 0.95, P < 0.001) and the SpectroSense2+ data could thus be converted into RapidScan data. The annual CH_4 emission was calculated by summing the currently measured emissions and the emissions from September 2012 to May 2013 as published by Karki *et al.* (2014).

Statistical analyses

Gas fluxes and RVI data were analysed with a linear mixed model using the R package *nlme* (Pinheiro *et al.*, 2013) and function *lme* in R version 3.0.2 (R Core Team, 2013):

$$Y_{ijk} = \mu + a_i + b_j + ab_{ij} + D_k + e_{ijk} \quad (4)$$

where μ is the general mean, a_i, and b_j are the fixed effects of GWL (i) and date (j), and ab_{ij} is the two-way interaction between GWL and date (ij). D_k is the random effect associated with each experimental unit (k) and e_{ijk} is the random error. Prior to analysis, CH_4 flux data were log-transformed after addition of a constant (0.2 mg CH_4 m^{-2} h^{-1}) to obtain normal distribution (P > 0.05, Shapiro–Wilk test) and variance homogeneity (uncorrelated residuals and fitted values). Data were treated as repeated measurements with application of the autocorrelation structure CorAR1 (Maxwell & Delaney, 2004). A similar model was run to determine statistical difference between CO_2 and CH_4 fluxes from the two management systems operationally defined as rewetted (0–20 cm GWL) and drained (30–40 cm GWL). Apart from the mixed model, correlation analyses were carried out to relate flux data to PAR, temperature and RVI using linear regression analysis in SIGMAPLOT 11 (Systat Software).

Differences of mean biomass yield between the GWL treatments were analysed using one-way ANOVA. Central tendencies are reported as mean ± standard deviation (SD) or mean ± standard error (SE) as indicated. Significance of all tests was accepted at P < 0.05.

Results

Environmental conditions and DOC

The daily mean air temperature during the measurement period varied from 6 to 22°C with an average of 14.2°C and cumulative precipitation was 294 mm; this was similar to the long-term average (1970–2000) of 13.9°C and 298 mm in the study area for the period from May to September. In the mesocosms, the daily mean soil temperature at depth of 5 cm varied from 7 to 21°C (Fig. 1) with seasonal averages (May to September) of 15.4 to 15.8°C for the five GWL treatments;

lowest temperatures were at 0 cm GWL and highest were at 40 cm GWL. Measured GWL in piezometers at each treatment showed some seasonal fluctuation, but with mean GWL aligning with the manipulated, nominal GWL (Fig. 2a). At the 0 cm GWL treatment, there was always visible water at the soil surface. Volumetric soil water contents also reflected the nominal GWL, although they were rather similar for the 20 and 30 cm GWL treatments (Fig. 2b). The concentrations of DOC remained more or less constant over the measurement period, with much higher levels at 0 cm GWL (288 ± 43 mg C L^{-1}) than at 40 cm GWL (46 ± 17 mg C L^{-1}); DOC concentrations at 10, 20 and 30 cm GWL were at an intermediate level, ranging from 124 ± 23 mg C L^{-1} to 147 ± 11 mg C L^{-1} (mean ± SD, n = 5).

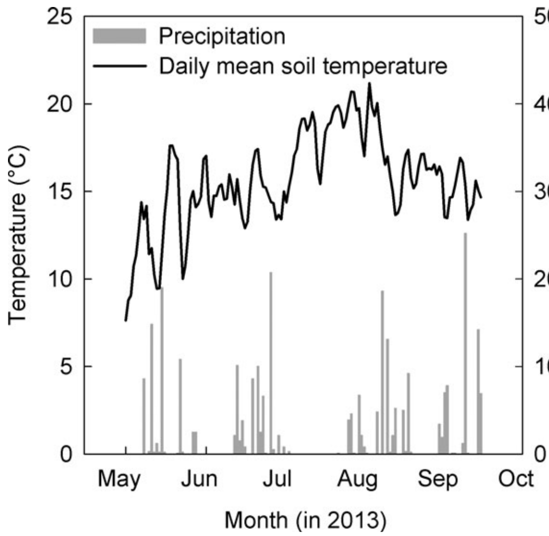

Fig. 1 Daily precipitation and mean daily soil temperature at 5 cm depth in the mesocosms. Data represent all five groundwater-level treatments during the study period from May to September 2013.

Biomass production

Average biomass yield across the five GWL treatments was 7.4 ± 0.5 Mg dry matter ha^{-1} at the first harvest and 5.6 ± 0.4 Mg dry matter ha^{-1} at the second harvest (mean ± SE, n = 5). Total yields showed a tendency of increase with lowering of the GWL; however, this was not statistically significant (Fig. 3a). Testing only the difference between the two operationally defined treatments (rewetted and drained) indicated that the total biomass yield was 25% lower ($P < 0.01$) for the rewetted treatments than for the drained treatments.

The vegetation in all mesocosms comprised three distinct species: RCG, marsh foxtail and *Poa* sp. Yet, whereas the same species occurred, the relative species composition differed between the rewetted and the drained treatments; marsh foxtail was the dominant species under rewetted conditions, whereas RCG was more dominant under drained conditions (Fig. 3b).

The dynamics of biomass development indicated by RVI was similar across the five GWL treatments (Fig. 4a). RVI peaked in late May and again in mid-July following the first harvest event. Decrease in RVI during June (before first harvest) was attributed to plant senescence.

Dynamics of CO_2 and CH_4 fluxes

As tested with the linear mixed model, there was no significant difference in GPP between the different

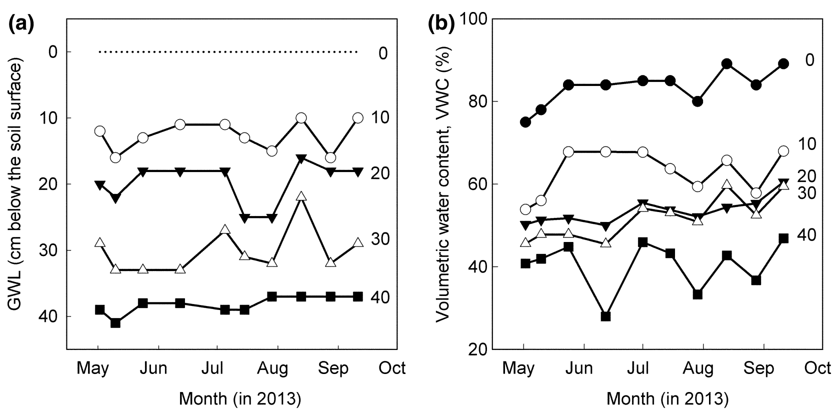

Fig. 2 Dynamics of (a) measured groundwater level (GWL) and (b) volumetric water content at the five nominal GWL during the study period from May to September 2013. Numbers next to lines indicate the different GWL treatments (cm below soil surface); dotted line indicates that water was always visible at the surface of the 0 cm GWL treatment. Volumetric water content was measured at 0–5 cm depth for 0 cm GWL, at 0–10 cm depth for 10 cm GWL and at 0–20 cm depth for 20, 30 and 40 cm GWL.

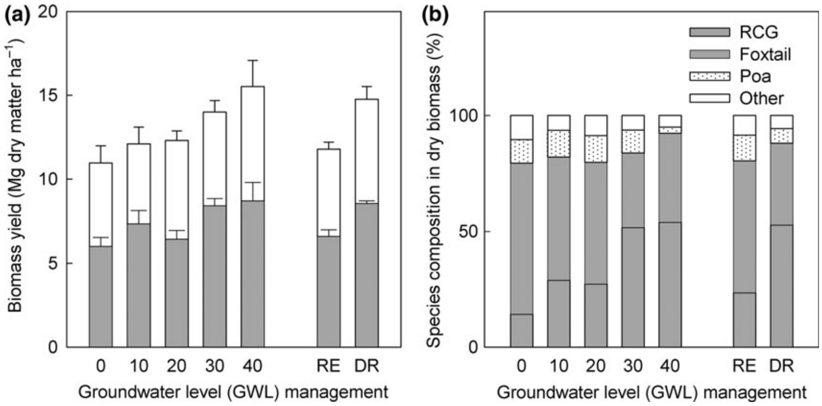

Fig. 3 (a) Biomass yield from first (grey bars) and second (white bars) harvest at different groundwater-level (GWL) managements; total height of the bars denotes the combined yield of the two harvest events. (b) Relative species composition (biomass) at first harvest at different GWL managements (RCG, reed canary grass). Biomass yields are shown as mean ± standard error ($n = 5$). RE represents the rewetted system (mean of data at 0–20 cm GWL) and DR represents the drained system (mean of data at 30–40 cm GWL).

GWL treatments (Fig. 4b). The GPP fluxes showed distinct seasonality correlating with PAR ($r = 0.78$–0.88, $P < 0.001$) and RVI ($r = 0.58$–0.73, $P < 0.001$). However, the depicted GPP and NEE dynamics were at some (cloudy) measurement dates constrained by low PAR (Fig. 4a and b). Indeed, rather than reflecting seasonal GPP and NEE dynamics, the measurement protocol was designed to allow a one-step GPP modelling including both PAR and RVI as dynamic variables.

ER showed significant difference among the GWL treatments ($P < 0.001$), and ER was lowest at 0 cm GWL and highest at 40 cm GWL (Fig. 4c). Seasonal variation in ER (Fig. 4c) was correlated with RVI ($r = 0.52$–0.69, $P < 0.001$) and soil temperature ($r = 0.60$–0.72, $P < 0.001$).

Methane fluxes decreased significantly with lowering the GWL ($P < 0.001$). CH_4 fluxes varied between 0.8 and 6.5 mg CH_4 m^{-2} h^{-1} at 0 cm GWL and between 0.2 and 3.0 mg CH_4 m^{-2} h^{-1} at 10 cm GWL (Fig. 4e); at deeper GWL, the fluxes varied between negligible net emissions and uptake of 0.3 to −0.2 mg CH_4 m^{-2} h^{-1}. The seasonal pattern of methane fluxes at 0 cm GWL was correlated with soil temperature ($r = 0.58$, $P < 0.001$) and RVI ($r = 0.37$, $P < 0.05$).

Cumulative fluxes and carbon balance

Estimated parameters for the GPP and ER models (Eqns 1 and 2) are shown in Table 1. Modelling efficiencies of the GPP and ER models were 85–88% and 46–61%, respectively (Table 1). The cumulative CO_2 uptake (GPP) was lower in rewetted (−5.6 kg m^{-2}) than in drained (−6.7 kg m^{-2}) treatments (Fig. 5a). Cumulative ER decreased consistently when GWL was raised (Fig. 5b). The decrease in ER was 34% when comparing the rewetted and drained management systems. The resulting cumulative NEE was negative from all GWL treatments signifying a net uptake of CO_2 from the atmosphere during the study period (Fig. 5c). The net uptake of CO_2 was 89% higher in rewetted than in drained management systems.

Cumulative CH_4 emissions showed a strong increase when the water table was raised from 20 cm to 10 cm and 0 cm; at lower GWL (drained treatments), cumulative CH_4 fluxes were negligible (Fig. 5d).

Integrating the NEE of CO_2-C, the CH_4-C flux and the export of harvested biomass C showed that the drained management system had a positive NECB and acted as a C source during the study period (Fig. 5e). Yet, for the rewetted management system the NECB was on average neutral. Thus, at 0 cm GWL the system was a net sink of C, but at deeper GWL, the system was a net source of C.

In terms of GWP, the integrated C flux results were similar to NECB as C fluxes in CO_2 and biomass were much larger than CH_4 fluxes for all GWL treatments.

Similar patterns of C balances were obtained for extrapolated fluxes on the annual scale as on the seasonal scale (Table 2). Thus, in terms of NECB the drained system was an annual net source of carbon (0.68 kg C m^{-2} year^{-1}), whereas the rewetted system was virtually carbon neutral (0.03 kg C m^{-2} year^{-1}).

Discussion

The present study quantified the carbon balance of RCG cultivation at different GWL representing rewetted (0, 10 and 20 cm GWL) and drained (30 and 40 cm GWL) management systems. The operational distinction between these two management systems was to some

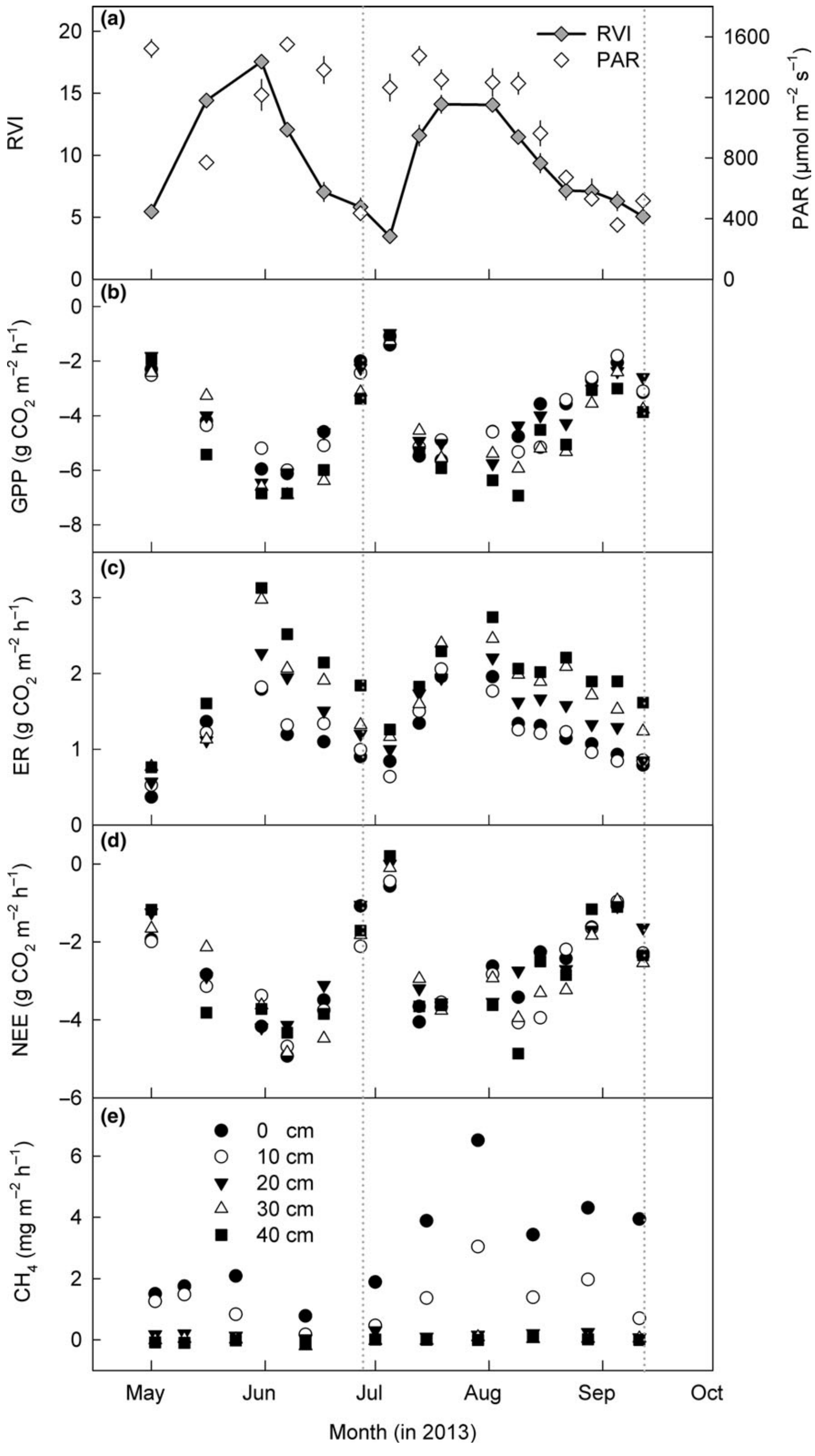

Fig. 4 Vegetation status and carbon fluxes for soil mesocosms managed at different groundwater levels during the study period from May to September 2013. (a) Ratio vegetation index (RVI) and average photosynthetically active radiation (PAR) during CO_2 flux measurements across all GWL. Data are presented as mean ± standard error (n = 4); (b, c, d) measured CO_2 fluxes of gross primary production (GPP), ecosystem respiration (ER) and net ecosystem exchange (NEE), respectively; (e) CH_4 fluxes. Data shown are mean of four mesocosms. Dates of first and second harvesting are indicated by dotted lines. Symbols for the GWL treatments (cm below soil surface) are shown in panel e.

Table 1 Parameter estimates for GPP and ER models (Eqns 1 and 2) fitted to data obtained at each groundwater level (GWL). Uncertainties shown in parentheses are standard errors of the parameter estimates. Also, the Nash–Sutcliffe modelling efficiencies (ME) and coefficients of determination (R^2) of the nonlinear regressions ($P < 0.001$) are shown

	Groundwater level (GWL), depth below soil surface				
Model parameters	0 cm	10 cm	20 cm	30 cm	40 cm
GPP model (Eqn 1)					
α	−3.3 (0.5)	−2.0 (0.2)	−2.3 (0.3)	−3.6 (0.6)	−3.2 (0.5)
β	−166 (14)	−294 (41)	−217 (28)	−207 (21)	−214 (22)
ME	0.87	0.85	0.88	0.85	0.86
R^2	0.90	0.85	0.90	0.86	0.86
ER model (Eqn 2)					
R_{10}	62 (22)	105 (22)	162 (28)	164 (39)	258 (40)
E_0	506 (92)	381 (77)	315 (64)	386 (77)	254 (56)
β	12.5 (2.7)	11.8 (2.6)	11.3 (3.3)	14.9 (4.2)	11.4 (4.1)
ME	0.61	0.57	0.54	0.55	0.46
R^2	0.64	0.58	0.54	0.50	0.45

extent arbitrary, but followed the delineation by IPCC (2014a), which considered mean GWL of ≤30 cm as a suitable proxy for rewetted sites and mean GWL of ≥30 cm as deeply drained systems. Likewise, Couwenberg & Fritz (2012) operationally distinguished between wet (≤20 cm GWL) and dry (>20 cm GWL) peat soil ecosystems in relation to CH_4 emissions.

With our experimental set-up, we controlled the GWL throughout the measurement period as documented by the GWL and soil moisture measurements. Monitoring of environmental variables was achieved by instrumentation of one of five replicate mesocosms at each GWL, assuming that the measured variables were representative for all five mesocosm replicates. Karki *et al.* (2015) previously found that ER, CH_4 and N_2O fluxes from instrumented and noninstrumented mesocosms (0–20 cm GWL) were similar which substantiated the assumption of similar environmental conditions among the mesocosm replicates.

Biomass production

Reed canary grass is considered as a suitable crop for biomass production in paludiculture due to its ability to grow under anoxic soil conditions (Wichtmann & Tanneberger, 2011). GWL and soil moisture content in our study were within a range that has previously been reported not to affect the productivity of RCG (Coops *et al.*, 1996; Ge *et al.*, 2012). However, in our study, RCG showed weaker biomass development at rewetted than at drained management systems. It was remarkable that biomass of RCG was displaced by marsh foxtail under rewetted conditions during the second year of RCG establishment. Both of these plants prefer wet conditions and they have similar Ellenberg humidity preferences (Hill *et al.*, 1999). The currently seeded RCG was the cultivar 'Bamse', but information on differences in GWL preferences among RCG cultivars is scarce. Yet, the present study underlines the importance of selecting suitable crops/genotypes that may thrive well under the targeted soil and climatic conditions for paludiculture. Although the shift in species composition may have contributed to slightly decreased biomass yield, the idea of paludiculture was not compromised as meadow grasses such as marsh foxtail may also be suitable for anaerobic digestion as these grasses have similar biogas potential as RCG (Raju *et al.*, 2011; Kandel *et al.*, 2013c).

CO_2 and CH_4 fluxes

Both ER and CO_2 uptake (numeric GPP) were lower under rewetted than under drained conditions. Decrease in ER under rewetted conditions aligns with the decrease in depth of the oxic soil layer and the volume of peat where aerobic (rather than anaerobic) decomposition takes place (Blodau *et al.*, 2004; Dinsmore *et al.*, 2009; Karki *et al.*, 2014). Lower GPP in the rewetted treatments could be due to lower biomass production. The increasing proportion of marsh foxtail in rewetted treatments could also have different GPP than RCG. Yet, the experimental set-up did not allow distinguishing between GPP of different plant species.

The growing season NEE of CO_2 in rewetted mesocosms was between −2.6 and −0.9 kg CO_2 m^{-2} which represented a higher net CO_2 uptake than the growing season fluxes of −0.5 to −0.1 kg CO_2 m^{-2} reported from other rewetted peatlands with different natural vegetation succession (Tuittila *et al.*, 1999; Soini *et al.*, 2010; Waddington *et al.*, 2010; Strack *et al.*, 2014). It is uncertain to compare the results from this study, where the

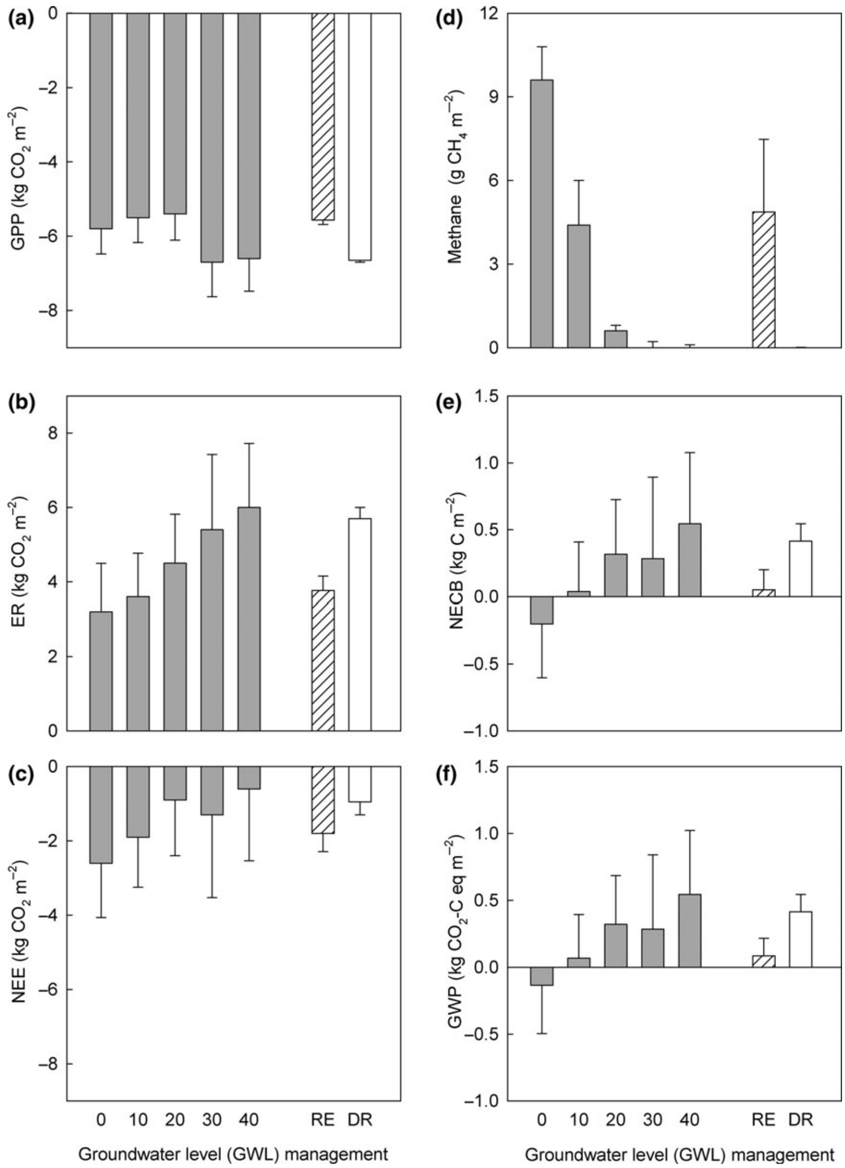

Fig. 5 Cumulative fluxes of (a) gross primary production, GPP, (b) ecosystem respiration, ER, (c) net ecosystem exchange, NEE and (d) CH_4 from May to September 2013 at different groundwater levels (GWLs). Also shown are (e) the net ecosystem carbon balance (NECB) and (f) the global warming potential (GWP) of carbon fluxes. RE represents the rewetted system (mean of data at 0–20 cm GWL) and DR represents the drained system (mean of data at 30–40 cm GWL). Data for individual GWL are shown as mean and standard error (SE) based on the SE of model parameters. Data for RE and DR systems are shown as mean and SE of data from 0 to 20 cm GWL (n = 3) and 30–40 cm GWL (n = 2), respectively.

crop was intensively managed on cultivated peatlands, with results from other rewetted study sites based on former peat extraction sites, natural vegetation succession or with extensive management practices. However, comparable annual NEE was reported for a RCG-dominated wetland in Denmark (−2.3 kg CO_2 m^{-2}; Askaer *et al.*, 2011) and a rewetted peatland in the Netherlands dominated with RCG and other grasses like velvet grass, reeds and forbs (−1.8 kg CO_2 m^{-2}; Schrier-Uijl *et al.*, 2014). Comparable CO_2 fluxes were also measured during the RCG growing season at the cultivated field in the Nørre Å river valley (where our mesocosms were collected) where similar crop management was applied in the first year after rewetting (T. Kandel, unpublished results). Also the growing season NEE of CO_2 from drained peat soils (−1.0 kg CO_2 m^{-2}, Fig. 5c) was similar to growing season NEE (−0.9 kg CO_2 m^{-2}) obtained at the RCG field site where our mesocosms were collected and where the average GWL was 43 cm during the growing season (Kandel *et al.*, 2013b).

Table 2 Annual estimates (16 September 2012 to 15 September 2013) of gross primary production (GPP), ecosystem respiration (ER), net ecosystem exchange (NEE), methane emission (CH_4) and net ecosystem carbon balance (NECB) at different groundwater-level (GWL) treatments (cm below soil surface). RE represents the rewetted system (mean of data at 0–20 cm GWL) and DR represents the drained system (mean of data at 30–40 cm GWL)

	Annual estimates of carbon fluxes				
GWL treatment	GPP (kg CO_2 m^{-2})	ER (kg CO_2 m^{-2})	NEE (kg CO_2 m^{-2})	CH_4 (g CH_4 m^{-2})	NECB (kg C m^{-2})
0	−7.3	4.1	−3.2	11.8	−0.36
10	−7.0	5.0	−2.0	6.2	0.01
20	−7.1	6.6	−0.5	1.8	0.43
30	−8.5	7.4	−1.1	0.6	0.35
40	−8.1	9.2	1.1	0.0	1.00
RE	−7.1	5.2	−1.9	6.6	0.03
DR	−8.3	8.3	0.0	0.3	0.68

Rewetting and colonization of rewetted peat soils by aerenchymatous plants species may generally lead to increase in CH_4 emissions (Tuittila *et al.*, 2000; Waddington & Day, 2007; Wilson *et al.*, 2009; Couwenberg & Fritz, 2012; IPCC, 2014a). The mean cumulative growing season emission of 4.9 g CH_4 m^{-2} obtained from rewetted soils in our experiment was similar to growing season CH_4 emissions (4.3–4.9 g CH_4 m^{-2}) reported from rewetted peatlands in boreal regions (Waddington & Day, 2007; Strack *et al.*, 2014). Also the estimated annual emission of 6.6 g CH_4 m^{-2} from our rewetted mesocosms falls within the range of annual emissions of 2–63 g CH_4 m^{-2} that have previously been observed from rewetted peatlands (Waddington & Day, 2007; Wilson *et al.*, 2009, 2013; Couwenberg & Fritz, 2012; Cooper *et al.*, 2014; IPCC, 2014a; Günther *et al.*, 2015). Large spatial and temporal variation may exist in CH_4 emissions due to dependence of CH_4 fluxes on site-specific GWL (Couwenberg *et al.*, 2011), vegetation types (Günther *et al.*, 2015), time after rewetting (Tuittila *et al.*, 2000; Waddington & Day, 2007) and weather conditions (Günther *et al.*, 2015). In our study, some underestimation of the CH_4 emissions could be possible as opaque, rather than transparent chambers were used for the flux measurements. Thus, notably for plant species with convective internal gas transport (unlike RCG) light-mediated processes may contribute to the CH_4 emissions (Günther *et al.*, 2014; Minke *et al.*, 2014).

Carbon balances

Net ecosystem exchange of CO_2 and biomass export were the most important components of the total carbon balance at all GWL as also suggested by a number of studies in drained peatlands (Shurpali *et al.*, 2010; Elsgaard *et al.*, 2012; Renou-Wilson *et al.*, 2014) and rewetted peatlands (Schrier-Uijl *et al.*, 2014; Günther *et al.*, 2015). Thus, although CH_4 emissions increased after rewetting, the increase in CH_4 emissions was more than offset by the decrease in ER even after considering the higher GWP of CH_4. The annual release of CH_4–C from rewetted soils was only 1% of net annual uptake of CO_2-C.

The growing season NEE results indicated a net uptake of CO_2, but including biomass export in the carbon balance changed the ecosystem from net sink of atmospheric CO_2 to net loss of soil carbon, except at the 0 cm GWL treatment, where the ecosystem was a net sink of soil carbon. A similar role of including exported biomass in the carbon balance has been seen in previous studies (Elsgaard *et al.*, 2012; Renou-Wilson *et al.*, 2014). However, the exported biomass can be used for bioenergy production mitigating fossil fuel CO_2 release to the atmosphere. One option is biogas production and Kandel *et al.* (2013d) showed that RCG biomass under similar management as in the current study produced 339 m^3 CH_4 Mg^{-1} DM. Assuming an energy yield of 0.036 GJ per 1 m^3 of CH_4 and that 57 kg CO_2 is released when 1 GJ of energy is produced from natural gas (Møller *et al.*, 2008), 0.8 and 1.0 kg CO_2 m^{-2} can be saved from the rewetted and drained management systems, respectively when natural gas is displaced by biogas produced from biomass.

The estimated annual NECB was close to being neutral in the rewetted management system, thus saving ca. 6.5 Mg C ha^{-1} $year^{-1}$ as compared to the drained management system (Table 2). This suggests that rewetting and cultivation with bioenergy crops can be a valid option to reduce the carbon emissions from drained peatland. The total saving of C was higher than total savings of 5 Mg C ha^{-1} $year^{-1}$ reported by Günther *et al.* (2015) from a low input paludiculture system (one cut in a year and no fertilization). Regina *et al.* (2015) suggested that raising the GWL from a drainage depth of 70 to 30 cm, and cultivating suitable crops like RCG, was a good mitigation option for reducing carbon emissions from cultivated peat soils. Our results extend this potential, showing that rewetting of peat soils to 0–20 cm is more efficient in carbon reduction as compared to maintaining GWL at 30 cm depth. Thus, as indicated here, peatlands could be turned into a sink of carbon if the GWL can be maintained near to the soil surface.

In addition to gaseous and biomass C fluxes, the total carbon balance of peatlands also depends on waterborne carbon fluxes which were not taken into account in this study (IPCC, 2014a; Renou-Wilson *et al*., 2014). Yet, DOC concentrations were measured and found to be higher from rewetted than from drained management systems. The concentration of DOC may increase under anoxic conditions due to less efficient anaerobic (than aerobic) decomposition leading to higher concentrations of water-soluble intermediate metabolites (Mulholland *et al*., 1990; Kalbitz *et al*., 2003), due to decrease in DOC adsorption (Kaiser & Zech, 1997) and due to slower conversion of released DOC to CO_2 (Moore & Dalva, 2001). However, an increase in DOC concentration may not necessary increase the total DOC fluxes from the rewetted system as the hydrological changes made during the rewetting would lower the discharge and thereby total DOC export (Strack & Zuback, 2013). Further studies are needed to specifically address the role of waterborne C fluxes for the total carbon balance at different GWL managements.

Although our results may support paludiculture as climate smart agriculture, it is emphasized that the present results were based on only one growing season during the second year after RCG establishment in mesocosms at (almost) constant GWL throughout the growing season. During the early succession stage after rewetting, higher carbon uptake can be possible due to increase in microbial and plant biomass pools (Wilson *et al*., 2013). Over a longer time period, the rate of CO_2 sequestration may decrease and reach an equilibrium point (Yli-Petäys *et al*., 2007). In addition, CH_4 emissions may also increase after some years due to re-establishment of microbial communities and activities which were altered when the peatlands were drained (Juottonen *et al*., 2012). Above all, the carbon balance is highly sensitive to weather conditions that potentially could change the ecosystem from sink to source of carbon (Roulet *et al*., 2007; Nilsson *et al*., 2008; Koehler *et al*., 2011; Herbst *et al*., 2013; Wilson *et al*., 2013). Hence, long-term studies are required for more robust estimates of the carbon balance from rewetted peatlands under field conditions.

Acknowledgements

The study was partly supported by the Danish Centre for Food and Agriculture (DCA) as a part of the BioBase programme. The authors would like to thank Morten Skov for the construction of the transparent chamber and Bodil Stensgaard, Finn Henning Christensen, Holger Bak, Jørgen M. Nielsen and Stig T. Rasmussen for their technical supports.

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