Full carbon and greenhouse gas balances of fertilized and nonfertilized reed canary grass cultivations on an abandoned peat extraction area in a dry year

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Abstract

Bioenergy crop cultivation on former peat extraction areas is a potential after-use option that provides a source of renewable energy while mitigating climate change through enhanced carbon (C) sequestration. This study investigated the full C and greenhouse gas (GHG) balances of fertilized (RCG-F) and nonfertilized (RCG-C) reed canary grass (RCG; Phalaris arundinacea) cultivation compared to bare peat (BP) soil within an abandoned peat extraction area in western Estonia during a dry year. Vegetation sampling, static chamber and lysimeter measurements were carried out to estimate above- and belowground biomass production and allocation, fluxes of carbon dioxide (CO2), methane (CH4) and nitrous oxide (N2O) in cultivated strips and drainage ditches as well as the dissolved organic carbon (DOC) export, respectively. Heterotrophic respiration was determined from vegetation-free trenched plots. Fertilization increased the above- to belowground biomass production ratio and the autotrophic to heterotrophic respiration ratio. The full C balance (incl. CO2, CH4 and DOC fluxes from strips and ditches) was 96, 215 and 180 g C m⁻² yr⁻¹ in RCG-F, RCG-C and BP, respectively, suggesting that all treatments acted as C sources during the dry year. The C balance was driven by variations in the net CO2 exchange, whereas the combined contribution of CH4 and DOC fluxes was <5%. The GHG balances were 3.6, 7.9 and 6.6 t CO2 eq ha⁻¹ yr⁻¹ in RCG-F, RCG-C and BP, respectively. The CO2 exchange was also the dominant component of the GHG balance, while the contributions of CH4 and N2O were <1% and 1–6%, respectively. Overall, this study suggests that maximizing plant growth and the associated CO2 uptake through adequate water and nutrient supply is a key prerequisite for ensuring sustainable high yields and climate benefits in RCG cultivations established on organic soils following drainage and peat extraction.

Keywords: bioenergy, biomass production, carbon dioxide, carbon sequestration, dissolved organic carbon, land management, methane, nitrous oxide, organic soils, Phalaris arundinacea

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Introduction

Commercial peat extraction for energy production and horticultural use is an important industry in many countries within northern Europe (e.g., Finland, Sweden, Ireland, Estonia, Belarus) and other parts of the world (i.e., USA, Canada, Russia, Indonesia) (Waddington et al., 2002; Lemus & Lal, 2005; Couwenberg et al., 2011; Don et al., 2012). Peat extraction has taken place on more than 50 000 km² of northern peatland areas which accounts for 10% of the total loss of natural peatlands due to human use (Joosten & Clarke, 2002). Peat extraction commonly be sustained over several decades depending on the depth of the peat deposit but eventually ceases, resulting in large abandoned areas of drained and degraded peat soils. Furthermore, due to the expansion of peat extraction activities into pristine peatlands and the vast areas of current peat extraction sites being annually abandoned, the extent of these degraded areas will likely increase in the future, resulting in a growing demand for developing appropriate after-use strategies (Tuittila et al., 2000; Maljanen et al., 2010). In addition, the large carbon dioxide (CO2) emission from these drained organic soils is a major concern from the climate change perspective (Gorham, 1991; Waddington et al., 2002; Mäkiranta et al., 2007; Salm et al., 2012), which further amplifies the need for adequate management strategies.
Among different after-use options, cultivating dedicated high-yielding energy crops (e.g., RCG, *Phalaris arundinacea*) on abandoned peat soils has been suggested as a promising strategy to increase the proportion of renewable energy supply while creating a sink for atmospheric CO₂ without competing for productive agricultural land required for food crop production (Lewandowski *et al.*, 2003; Lemus & Lal, 2005; Shurpali *et al.*, 2010; Don *et al.*, 2012). Furthermore, cultivation of perennial crops has the advantage of reducing the need for regular tillage commonly associated with annual crop cultivation, which could reduce the associated C losses and therefore increase the soil C sequestration potential (Adler *et al.*, 2007; Jones *et al.*, 2015). Although RCG grows well on most kinds of soils, highest biomass is commonly reached on wet and humus-rich soils (e.g., in abandoned peat extraction areas) (Don *et al.*, 2012). In addition, RCG is adapted to short growing seasons and low temperatures. Thus, the potential of RCG cultivation is especially high in Northern Europe given the climatic conditions and vast occurrence of organic soils.

In most bioenergy cropping systems, fertilizer is applied to maximize biomass production (Maljanen *et al.*, 2010; Don *et al.*, 2012). Increased plant growth and nutrient supply may, however, also alter the above- and belowground allocation of plant biomass and trigger structural changes in the belowground biomass due to its contrasting effects on root and rhizome production (Kätterer & Andréén, 1999; Xiong & Kätterer, 2010; Kinmonth-Schultz & Kim, 2011; Jones *et al.*, 2015). Furthermore, enhanced plant growth and changes in biomass allocation following fertilization may also modify the partitioning of the ecosystem CO₂ exchange into its component fluxes of gross and net primary production as well as soil heterotrophic and plant autotrophic respiration (Shurpali *et al.*, 2008; Kandel *et al.*, 2013). To date, detailed information of fertilization effects on biomass allocation and CO₂ balance partitioning is limited for perennial cropping systems; however, this knowledge is crucial to confidently predict variations in annual yields and the C cycle of bioenergy cropping systems on abandoned organic soils under future climate change and management scenarios (Strand *et al.*, 2008; Gong *et al.*, 2014; Jones *et al.*, 2015).

Within the climate change context, it is imperative to understand fertilization effects on not only biomass production but also on the full ecosystem C balance which is determined by the net CO₂ exchange of vegetation and soil as well as by non-CO₂ carbon fluxes such as the exchange of methane (CH₄) and the aquatic export of dissolved organic carbon (DOC). Furthermore, due to the relatively small area coverage by drainage ditches, most studies focus on the C balance of the cultivated strips ignoring potentially high emissions from ditches (Sundh *et al.*, 2000; Schrier-Uijl *et al.*, 2010; Hyvönen *et al.*, 2013). Although individual studies have previously investigated the patterns and magnitudes of the different C balance components (Sundh *et al.*, 2000; Shurpali *et al.*, 2009; Strack *et al.*, 2011; Hyvönen *et al.*, 2013), a comprehensive assessment of all major C fluxes from both strips and ditches and their contrasting contributions to the full annual C balance in fertilized and nonfertilized RCG cultivations on organic soil is currently lacking.

A major drawback of nitrogen (N) fertilizer application is that it may cause high emissions of the greenhouse gas (GHG) nitrous oxide (N₂O) (Crutzen *et al.*, 2008). Considering that N₂O has a much larger (by 298 times) global warming potential compared to CO₂ (IPCC, 2013), the positive fertilizer effects on plant growth and C sequestration could be partly or entirely offset when considering the GHG balance of bioenergy cultivations (Don *et al.*, 2012). A recently proposed strategy for reducing N₂O emissions is to substitute annual crops with perennial crops, such as RCG, which have a lower N-demand and higher N-use efficiency and hence emit 40–99% less N₂O compared to conventional annual crops (Don *et al.*, 2012). To date, the number of studies investigating the trade-off between the increased CO₂ uptake due to stimulated plant growth and the enhanced N₂O emission following fertilization is, however, limited and its implication for the GHG balance of bioenergy cultivations therefore highly uncertain.

To address these knowledge gaps, this study investigated the full C and GHG balance in fertilized and nonfertilized RCG cultivations compared to bare peat soil in an abandoned peat extraction area in western Estonia. The study addressed the following specific questions:

1. How does fertilization affect the above- and belowground biomass production and allocation
2. How does the net ecosystem CO₂ exchange (NEE) and its partitioning into production and respiration component fluxes differ among the fertilized RCG, nonfertilized RCG and bare peat treatments
3. What are the full C and GHG balances and the contributions of their respective component fluxes from cultivated strips and drainage ditches in the fertilized RCG, nonfertilized RCG and bare peat treatments

**Material and methods**

**Study site**

The study site (58°34′20″N, 24°23′15″E; Halinga Parish, Pärnu County) is located within the largest Estonian peat extraction area, Lavassaare, which is situated in the northern part of the
Pärnu Lowland. The region has a temperate climate with a 30-year (1981–2010) mean annual temperature of 6.3 °C and annual precipitation of 746 mm (Estonian Weather Service). The peat extraction area is divided into 20 m wide strips separated by 1 m wide drainage ditches. Commercial peat extraction at the site started in the 1960s and lasted until 2006. In 2007, the abandoned strips were tilled and sown with seeds of the Estonian-bred RCG variety ‘Pedja’. Since the pH_ec of the remaining peat was above 5.0 (Table 1), no liming was carried out. The remaining peat deposit is approximately 0.45–0.60 m deep and consists of well-mineralized Phragmites–Carex peat with a degree of humification of H7 according to the von Post scale. The main soil properties are summarized in Table 1. No fertilization or biomass harvest was carried out between the seeding in 2007 and spring 2012.

**Experimental design**

The study was designed as a replicated field experiment with six experimental plots (2.5 × 10 m) of which four were located within the cultivated strips and two within the abandoned bare peat strips. The cultivated plots consisted of two fertilized plots and two adjacent nonfertilized control plots. The fertilized plots received 72 kg N, 18 kg P and 36 kg K of mineral fertilizer per hectare once per year during the early growing season since 2012. Thus, the experiment consisted of two replicate plots for each of the three treatments: reed canary grass cultivation with fertilization (RCG-F), reed canary grass cultivation control (RCG-C) and bare peat (BP).

**Above- and belowground biomass and net primary production**

Above- and belowground biomass stocks were measured in the cultivated RCG-F and RCG-C plots. Aboveground biomass was harvested on five subplots (50 × 50 cm) within each plot in September 2014. Belowground biomass (roots and rhizomes) was determined in April and September from five soil cores per plot taken to a depth of 30 cm using an 8.5 cm diameter corer. Each soil core was divided into three 10 cm sections (0–10, 10–20 and 20–30 cm). After manually washing each core over a 0.5 mm mesh sieve to remove the bulk soil, roots and rhizomes were manually picked with tweezers from the residual soil. Both above- and belowground biomass was oven-dried at 70 °C to a constant weight and analyzed for C concentrations at the Tartu Laboratory of the Estonian Environmental Research Centre.

Annual aboveground net primary production (ANPP) was calculated by multiplying the harvested biomass with its C concentration. The annual belowground net primary production (BNPP) was estimated with the maximum–minimum method (McClaugherty et al., 1982) as the difference between the maximum (September sampling) and minimum (April sampling) belowground biomass stocks. The sum of ANPP and BNPP resulted in the annual total net primary production (NPP_b) estimate based on biomass sampling (Eqn 1).

\[
NPP_b = ANPP + BNPP
\]

**Vegetation greenness index**

To estimate plant development, we derived a vegetation greenness index from digital images using repeat photography (Sonnenfag et al., 2012; Peichl et al., 2015). A Wingscapes TimelapseCam 8.0 camera (model WSCT01; Wingscapes, Calera, AL, USA) was installed on a vertical pole at 3 m height above the ground surface viewing the experimental plots with a downward looking viewing angle of 15° from a southerly direction. The white balance was set to ‘sunlight’. The camera was programmed to take images at half-hourly intervals from mid-April to late October 2014. The vegetation greenness index was derived from the green chromatic coordinate (\(g_G\)) (Eqn 2):

\[
g_G = \frac{G}{R + G + B}
\]

where \(R\), \(G\) and \(B\) are the digital numbers (0–255) of the red, green and blue image channels. The RGB digital numbers were calculated for each pixel and averaged over selected regions of interest (ROI) representing fertilized and nonfertilized RCG cultivations as well as bare peat areas, respectively. Three-day mean \(g_G\) time series were then created by assigning the 90th percentile to all values within a 3-days window to the center day of a discrete (nonoverlapping) moving window following Sonnenfag et al. (2012). To estimate the collar-specific \(g_G\) images were also taken from above each collar on July 19 and analyzed for their \(g_G\). The time series of the mean \(g_G\) in RCG-F and RCG-C were then multiplied with the relative difference in \(g_G\) among collars to obtain collar-specific \(g_G\) estimates for the growing season as input for the regression models described below.

**Meteorological and soil environmental measurements**

During every sampling event, soil volumetric water content (VWC) was measured on each plot in two depths (0–5 and
15–20 cm) using a handheld Decagon GS3 soil moisture sensor (Decagon Devices Inc., Pullman, WA, USA). In addition, soil temperature ($T_s$) was recorded on each plot in four different depths (10, 20, 30 and 40 cm) by a handheld Comet S0141 temperature logger with Pt1000TG8 sensors (Comet Systems Ltd., Rožnov pod Radhoštěm, Czech Republic). At each plot, continuous 30 min records of water table level (WTL) position relative to the soil surface were obtained with submerged HOBO Water Level Loggers (model U20-001-01; Onset Computer Corporation, Bourne, MA, USA). These automatic WTL records were calibrated with manual WTL measurements taken at the same locations.

A meteorological station was installed on-site in June to continuously measure air temperature ($T_a$) using a shielded temperature sensor (model CS 107; Campbell Scientific Inc., Logan, UT, USA) as well as photosynthetically active radiation (PAR) and precipitation (PPT) using a LI-190SL Quantum Sensor (LI-COR Inc., Lincoln, NE, USA) and a Young 52202 tipping bucket (R. M. Young Company, Traverse City, MI, USA), respectively. All meteorological instruments were mounted on a pole at 1.2 m height above the ground. Soil temperature, $T_s$ (model CS 107; Campbell Scientific Inc.), was recorded at 5 and 30 cm depths. The soil volumetric water content, VWC (model CS615; Campbell Scientific Inc.), was measured at 10 cm depth. One sensor for each of the $T_a$, $T_s$, and VWC measurements was placed in one of the RCG-F, RCG-C and BP plots. All automated meteorological and soil environmental data were collected in 1 min intervals and stored as 10 min averages on a CR1000 datalogger (Campbell Scientific Inc., Logan, UT, USA) as well as photosynthetically active radiation (PAR) and precipitation (PPT) from the nearby (~20 km away) Pärnu meteorological station (Estonian Weather Service) for 2014. Global radiation was converted to PAR, pressure and relative humidity inside the chamber were monitored with a portable infrared gas analyzer (IRGA, EGM-4; PP Systems) connected to the chamber in a closed loop through 1.5 m inlet and outlet tubing (inner Ø 0.3 cm, flow rate 350 ml min$^{-1}$). Measurements were conducted in a random plot order between 10:00 and 14:00 to avoid diurnal effects on the fluxes. Gross primary production (GPP) was derived from the difference between NEE and RE (Eqn 3):

$$\text{GPP} = \text{NEE} - \text{RE}$$  \hspace{1cm} (3)

In addition, hourly net primary production rates based on the CO2 flux measurements (NPP$^\text{r}$) were derived for each sampling date from the difference between NEE and heterotrophic respiration (Rh; see below) (Eqn 4):

$$\text{NPP}^\text{r} = \text{NEE} - \text{Rh}$$  \hspace{1cm} (4)

Due to the absence of vegetation, NEE and RE at the BP plots were represented by the static chamber measurements described below.

Heterotrophic and autotrophic respiration measurements

Heterotrophic respiration (Rh) was measured concurrently with NEE and RE fluxes on trenched plots (A 0.07 m$^2$) which were established in the RCG-F and RCG-C plots in late March 2014. Within each plot, three subplots were trenched to 0.5 m depth and a water-permeable cloth was inserted vertically to prevent lateral root in-growth. All living plants were clipped and removed from inside these trenched plots and the plots were kept vegetation-free for the remaining year. One PVC collar (Ø 17.5 cm) was permanently installed to 10 cm depth in each of the trenched plots. During the measurement, an opaque chamber (h 30 cm, V 0.065 L) equipped with a low-speed fan was placed onto the collar and the CO2 concentration as well as headspace $T_a$ was measured every 4.8 s with the EGM-4 IRGA during 3 min. Autotrophic respiration (Ra) was derived as the difference between the measured RE and Rh (Eqn 5):

$$\text{Ra} = \text{RE} - \text{Rh}$$  \hspace{1cm} (5)

Relative humidity was also measured in the BP plots using the same collars and chamber. However, trenching and vegetation removal were not necessary, and as Rh measurements represent RE in vegetation-free ecosystems, Ra was not determined in the BP plots.

CH$\text{}_4$, N$_2$O and nongrowing season CO2 flux measurements

Methane (CH$\text{}_4$) and nitrous oxide (N$_2$O) fluxes were measured weekly to biweekly during the 2014 growing season (May 1 to October 31) and once per month outside the growing season with the closed static chamber method at the same measurement locations (i.e., same collars) of the NEE and RE.
measurements. During each 1-h chamber deployment period, four evenly timed (0, 0.33, 0.66 and 1 h) air samples were drawn from the chamber headspace (h 50 cm, V 65 L; white opaque PVC chambers) with polypropylene syringes through a plastic tube into pre-evacuated (0.3 mbar) 100 mL glass bottles. The air samples were analyzed within a week for CH₄ and N₂O concentrations using a Shimadzu GC-2014 gas chromatograph (GC) combined with a Loftfield automatic sample injection system (Loftfield et al., 1997), a flame ionization detector (FID) and an electron capture detector (ECD). To obtain RE estimates during the nongrowing season months of January to April, the same air samples were also analyzed for their CO₂ concentrations on the same GC using the ECD detector. These RE measurements were also assumed to represent NEE from January to April in all treatments.

Ditch CO₂, CH₄ and N₂O flux measurements

To account for the spatial variation between fluxes from strips and drainage ditches, CO₂, CH₄ and N₂O fluxes were measured within the same peat extraction area from drainage ditches bordering RCG-F, RCG-C and BP strips at a monthly interval from June to December 2011 (with an annual PPT of 826 mm, 2011 was wetter than 2014). Three collars were permanently installed at the bottom of each ditch. The measurements were conducted using same chamber and measurement protocol as described above for the static chamber measurements in the strips.

Flux calculation and quality control

Fluxes of CO₂, CH₄ and N₂O were calculated from the change in gas concentrations in the chamber headspace volume corrected for air density using the ideal gas law (Eqn 6):

\[ F_{\text{dyn,stat}} = S \times \frac{p \times V \times M \times t}{R \times T_a \times A} \]  

(6)

where \( F_{\text{dyn}} \) and \( F_{\text{stat}} \) are fluxes measured by the dynamic chamber (i.e., CO₂ in mg CO₂-C m⁻² h⁻¹) and the static chamber method (i.e., CH₄ in μg CH₄-C m⁻² h⁻¹, N₂O in μg N₂O-N m⁻² h⁻¹ and CO₂ in mg CO₂-C m⁻² h⁻¹), respectively, S is the linear slope fitted to the concentration change over time (ppm s⁻¹ for the dynamic and ppm h⁻¹ or ppb h⁻¹ for the static chamber method), p is the air pressure (measured by the EGM-4 instrument or approximated by a constant value of 1013 kPa in the dynamic and static chamber methods, respectively), V is chamber headspace volume, M is the molar mass of the gas, R is the universal gas constant of 8.3143 (J mol⁻¹ K⁻¹), \( T_a \) is the mean headspace air temperature during the measurement (°K), A is the collar area, and t converts the time unit from seconds to hour (i.e., \( t = 3600 \) for the dynamic chamber method and \( t = 1 \) for the static chamber method). In the dynamic chamber method, \( S \) was the slope with the best \( R^2 \) from the individual slopes determined for windows of 25 measurement points (i.e., 2 min) moving stepwise (with one-point increments) over the measurement period after discarding the first two measurement points (i.e., applying a 9.6 s ‘dead band’). In the static chamber method, \( S \) was calculated over all four data points. The headspace volume was corrected for changes in effective chamber height due to frost heave (resulting in uplifting of the collars) or snow/ice buildup.

All dynamic chamber CO₂ fluxes with a coefficient of determination \( R^2 \geq 0.90 \) were accepted as good fluxes. However, as low fluxes generally result in a lower \( R^2 \) (which is especially critical for NEE measurements), fluxes with \( S \leq \pm 0.15 \) ppm s⁻¹ were always accepted. The \( S \) threshold was determined based on a regression between \( S \) and \( R^2 \) values. For static chamber measurements, the \( R^2 \) threshold for accepting CO₂, CH₄ and N₂O fluxes was 0.90 \( (P < 0.05) \), 0.80 \( (P < 0.1) \) and 0.80 \( (P < 0.1) \), respectively, except no filtering criteria was used when the maximum difference in the concentration values was less than the gas-specific GC detection limit (i.e., <20 ppm for CO₂, <20 ppb for CH₄ and <20 ppb for N₂O).

This study used the atmospheric sign convention in which positive (e.g., RE) and negative (e.g., GPP and NPP) fluxes represent emission and uptake, respectively.

Model development for estimating annual CO₂, CH₄ and N₂O fluxes

Nonlinear regression models following Kandel et al. (2013) were used to estimate annual RE and GPP fluxes based on \( T_a \), PAR and vegetation development. Specifically, GPP fluxes from each collar were fitted to PAR inside the chamber using a hyperbolic function with an additional parameter describing the seasonal changes in vegetation biomass (expressed by the collar-specific \( g_{cc} \) estimates) (Eqn 7):

\[ \text{GPP} = \frac{a \times A_{max} \times \text{PAR} \times g_{cc_{max}}}{a \times \text{PAR} \times A_{max} \times g_{cc_{norm}}} \]  

(7)

where GPP is gross primary production (mg CO₂-C m⁻² h⁻¹), PAR is the photosynthetically active radiation (μmol m⁻² s⁻¹) inside the chamber, \( a \) is the light-use efficiency of photosynthesis (i.e., the initial slope of the light response curve, mg CO₂-C μmol photons⁻¹), \( A_{max} \) is maximum photosynthesis at light saturation (mg CO₂-C m⁻² h⁻¹), and \( g_{cc_{norm}} \) is the collar-specific chromatic greenness index normalized to scale between 0 and 1.

RE fluxes were fitted to headspace \( T_a \) accounting for effects from vegetation biomass using an exponential function (Eqn 8):

\[ \text{RE} = R_{0} \times \exp^{(b \times T_{a})} + (\beta \times g_{cc_{norm}}) \times \exp^{(b \times T_{a})} \]  

(8)

where RE is ecosystem respiration (mg CO₂-C m⁻² h⁻¹), \( T_a \) is air temperature (°C), \( R_{0} \) is the soil respiration (mg CO₂-C m⁻² h⁻¹) at 0°C, \( b \) is the sensitivity of respiration to \( T_a \) and \( \beta \) is a scaling parameter representing the contribution of plant respiration to ecosystem respiration. Using the respective model coefficients, hourly GPP and RE were modeled for the entire year using hourly \( T_{a} \) PAR and \( g_{cc} \) as input variables. Annual GPP and RE were then estimated from the cumulative sums of these modeled estimates. The balance between annual GPP and RE estimates resulted in the annual NEE in RCG-F and RCG-C.

In the BP plots, RE was modeled based on an exponential relationship to \( T_a \) only (Eqn 9):
The cumulative RE model estimates also represented annual NEE at the BP treatment. The GPP and RE model parameters for the different treatments are summarized in Table 2. Due to weak relationships with environmental variables, the annual CH₄ exchange was estimated by linear interpolation. For the different treatments are summarized in Table 2. The cumulative RE model estimates also represented annual NEE at the BP treatment. The GPP and RE model parameters for the different treatments are summarized in Table 2. The annual GHG balances were estimated by converting the cumulative and ditch fluxes to CH₄ and N₂O equivalents (CO₂ eq) using the global warming potentials (GWP, over a 100-year time frame including carbon–climate feedbacks) of 34 and 298 for CH₄ and N₂O, respectively (IPCC, 2013).

### Concentrations and fluxes of dissolved organic carbon

Starting in February 2014, dissolved organic carbon (DOC) concentrations were determined for water samples taken at each flux sampling location from groundwater wells (perforated PVC pipes; Ø 7.5 cm) which collected soil solution from the soil surface down to 50 cm depth. From late June onward, DOC concentrations were estimated at 30 cm soil depth (below the main rooting zone) on every flux sampling date using stainless steel plate lysimeters with a collecting area of 625 cm² (Uri et al., 2011). All water samples were analyzed for their DOC concentrations within 1 day after collection. The DOC export was calculated by multiplying the DOC concentration with the monthly precipitation based on previous literature estimates (Klöve et al., 2010; Hyvänen et al., 2013). To assess the sensitivity of the annual DOC export to the choice of this assumed leaching rate, DOC export was also estimated using rates of 25% and 75% of the monthly precipitation to provide a minimum–maximum range which very likely encompasses the true leaching rate.

### Statistical analysis

Collar flux data were averaged for each plot before conducting further statistical analysis to avoid pseudoreplication. The non-parametric Friedman one-way analysis of variance (ANOVA) by ranks test for dependent samples was used to account for repeated measurements in time when testing for treatment effects (i.e., fertilized RCG, nonfertilized RCG and bare peat) on the growing season or annual means of the various component fluxes. This analysis was followed by a Bonferroni post hoc comparison to determine significant differences among treatment means. The Mann–Whitney U-test was used when comparing only the fertilized and nonfertilized treatments for significant effects (i.e., on GPP, NPP and Ra fluxes and biomass pools). The significance level was \( P < 0.05 \) unless stated otherwise. All calculations and statistics were computed using the MATLAB software (MATLAB Student version, 2013a; Mathworks, Natick, MA, USA).

### Results

#### Environmental conditions

The annual mean \( T_a \) and total PPT for the study year 2014 were 6.9 °C and 525 mm, respectively, which indicates warmer and drier conditions relative to the long-term climate normal (6.3 °C and 745 mm). PAR and \( T_a \) peaked in the first week of July and in the first week of August, respectively (Fig. 1a, b). The growing season included two warm and dry periods: one from mid-May to mid-June and the other from early July to early August. Total rainfall during these two periods was only 22.8 and 37.7 mm, respectively (Fig. 1c). The dry summer was interrupted by intermittent rainfall events in mid- to late June and eventually terminated by 2 weeks of heavy rainfall (179.5 mm) from early to late August.

Soil temperatures at 5 and 10 cm depths were similar among the three treatments throughout the year (Fig. 1d). The WTL, however, was higher in RCG-F and RCG-C compared to BP for most of the measurement period and dropped below the depth of the peat layer (i.e., <45 cm) in all treatments during the two dry and warm summer periods (mid-May to mid-June and early July to early August) (Fig. 1e). The intermittent rainfall in late June and the onset of heavy rainfall in early August were reflected by rapid increases in the WTL.

\[
RE = R_0 \times \exp^{(b \times T_a)}
\]
Vegetation greenness index

The $g_{cc}$ was higher in RCG-F than in RCG-C throughout the growing season (Fig. 2). Its temporal patterns suggest that plant growth started in mid-May and that full canopy development was reached by the second week of June in both RCG-F and RCG-C. Furthermore, the dry period in late June coincided with a temporary reduction in $g_{cc}$. The start of the senescence period in early August, as indicated by the decline in $g_{cc}$, coincided with decreasing $T_a$ and $T_s$. The $g_{cc}$ eventually decreased to its pregrowing season values after the harvest cut on September 3. The $g_{cc}$ did not show any seasonal patterns in BP, except for small fluctuations related to illumination noise and color changes due to soil moisture variations.

Above- and belowground biomass production and allocation

At the time of harvest, mean aboveground biomass ($\pm$ standard error) in RCG-F and RCG-C was $234 \pm 19$ and $42 \pm 6 \text{ g m}^{-2}$, respectively (Fig. 3). The belowground biomass increased from $536 \pm 17$ and $364 \pm 22 \text{ g m}^{-2}$ in April to $646 \pm 23$ and $416 \pm 29 \text{ g m}^{-2}$ in September in RCG-F and RCG-C, respectively, and was significantly greater in RCG-F than in RCG-C on both sampling dates (Fig. 3). Both root and rhizome biomass decreased significantly with soil depth. The upper 0–10 cm layer contained $57\%$ and $58\%$ of total belowground biomass, $92\%$ and $93\%$ of rhizome biomass and $46\%$ and $43\%$ of the root biomass in RCG-F and RCG-C, respectively. While root biomass in RCG-F was significantly greater than in RCG-C for all soil depth classes and sampling dates, rhizome biomass was significantly
greater (on both sampling dates) in the upper 10 cm layer only in RCG-F compared to RCG-C. The root to rhizome ratio was 2.84 and 2.31 in RCG-F and RCG-C, respectively.

The C concentration of harvested aboveground biomass was 48% and 47%, while it was 50% for belowground biomass in both RCG-F and RCG-C, respectively, without any significant difference between the April and September sampling dates.

Both above- and belowground NPP derived from biomass sampling were greater in RCG-F than in RCG-C (Table 3). The ratio of above- to belowground NPP decreased from 2.1 in RCG-F to 0.8 in RCG-C, respectively.

**Seasonal net ecosystem CO₂ exchange, respiration and production**

A negative midday NEE, that is, CO₂ uptake, was observed in RCG-F from late May to the end of September, with a maximum uptake rate of 162 mg C m⁻² h⁻¹ in early July (Fig. 4a). In comparison, midday NEE remained close to zero during the early growing season (May and June) and switched to positive values suggesting CO₂ emission of up to 77 mg C m⁻² h⁻¹ during the late growing season (July and August) in RCG-C. Both RCG-F and RCG-C were small CO₂ sources for most of the nongrowing season. Continuous midday CO₂ emission occurred throughout the year in BP, reaching a maximum rate of 71 mg C m⁻² h⁻¹ in early July. The annual mean midday NEE was significantly lower in RCG-F than in RCG-C and BP (Fig. 5a).

During the growing season, midday RE was highest in RCG-F and lowest in BP, reaching peak values of 268, 149 and 71 mg C m⁻² h⁻¹ during late July in RCG-F, RCG-C and BP, respectively (Fig. 4b). The annual mean midday RE was significantly higher in RCG-F than in BP (Fig. 5b).

Midday GPP was consistently greater (i.e., more negative) in RCG-F than in RCG-C during the growing season (Fig. 4c). In RCG-F, GPP peaked with 359 mg C m⁻² h⁻¹ simultaneously with Tₚ in late July whereas in RCG-C, values remained in the range of 50 to 110 mg C m⁻² h⁻¹ throughout most of the growing season following the pattern of gₚ. The seasonal patterns in midday NPP were similar to those of GPP, reaching maximum rates of −211 and −85 mg C m⁻² h⁻¹ in RCG-F and RCG-C, respectively (Fig. 4d). The mean midday GPP and NPP were significantly lower (i.e., suggesting greater production) in RCG-F than in RCG-C (Fig. 5c, d).

**Heterotrophic and autotrophic respiration**

Similar seasonal patterns of Rh were observed in RCG-F and RCG-C, with maximum rates of 119 and 92 mg C m⁻² h⁻¹, respectively, occurring in late July (Fig. 6a).
In RCG-F and RCG-C, Rh was higher \((P < 0.01)\) from early July to mid-August than in BP where maximum Rh rates of 77 mg C m\(^{-2}\) h\(^{-1}\) were observed (Fig. 6a). In comparison, Ra was consistently higher in RCG-F than in RCG-C throughout the growing season (Fig. 6b). Maximum Ra during July was approximately 2.5 times higher in RCG-F \((-150 \text{ mg C m}^{-2} \text{ h}^{-1}\) than in RCG-C \((-60 \text{ mg C m}^{-2} \text{ h}^{-1}\). Except for one sampling date (May 30), the Ra to Rh ratio was always \(\geq 1\) in RCG-F while it was \(\leq 1\) in RCG-C for most of the growing season (Fig. 6c). The mean growing season Rh was not significantly different between RCG-F and RCG-C (Fig. 5e). In contrast, mean Ra was significantly higher in RCG-F than in RCG-C (Fig. 5f). Averaged over all sampling dates, Rh accounted for 42% and 62% of RE in RCG-F and RCG-C, respectively.

### CH\(_4\) and N\(_2\)O exchanges

Throughout the growing season, CH\(_4\) emission occurred in the range of 0.01 to 9.3 \(\mu\)g C m\(^{-2}\) h\(^{-1}\) in all three treatments (Fig. 7a). Between mid-June and early September, the mean CH\(_4\) emission was approximately 1.5 times higher in BP than in RCG-F and RCG-C \((P = 0.052)\). During the nongrowing season, the CH\(_4\) exchange was close to zero with small rates \((-2.3 \text{ to } 3.7 \mu\text{g C m}^{-2} \text{ h}^{-1}\) of both uptake and emission occurring at individual collars and sampling dates, with no differences among the three treatments. The annual mean CH\(_4\) exchange was

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**Table 3** The full carbon (C) balance (g C m\(^{-2}\) yr\(^{-1}\) and the annual sums of its components: ecosystem respiration (RE), gross primary production (GPP), above- and belowground net primary production (ANPP, BNPP), total net primary production (NPP\(_{\text{B}}\); based on biomass sampling), net ecosystem CO\(_2\) exchange (NEE) and methane (CH\(_4\)) export of dissolved organic carbon (DOC) and the ditch CO\(_2\) and CH\(_4\) fluxes for reed canary grass with fertilization (RCG-F), reed canary grass control (RCG-C) and bare peat (BP) treatments. Negative and positive fluxes represent C uptake and emission, respectively.

| C flux component | RCG-F | RCG-C | BP |
|------------------|-------|-------|----|
| Strips           |       |       |    |
| RE               | 512   | 326   | 170|
| GPP              | -433  | -125  | n.a.|
| ANPP             | -115  | -20   | n.a.|
| BNPP             | -55   | -26   | n.a.|
| NPP\(_{\text{B}}\) | -170  | -46   | n.a.|
| NEE              | 79    | 201   | 170*|
| CH\(_4\)         | 0.014 | 0.018 | 0.020|
| DOC export       | 4.2   | 4.2   | 4.1 |
| Ditches          |       |       |    |
| CO\(_2\)         | 360   | 415   | 298 |
| CH\(_4\)         | 0.134 | 0.120 | 0.063|
| Total C balance† | 96    | 215   | 180 |

n.a., not applicable.

*GPP for BP was assumed to be zero.

†The total C balance is the sum of area-weighted (strip width = 20 m; ditch width = 1 m) fluxes of NEE, CH\(_4\) and DOC as well as the ditch CO\(_2\) and CH\(_4\) fluxes.

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![Fig. 4](image-url) (a) Net ecosystem CO\(_2\) exchange (NEE), (b) ecosystem respiration (RE), (c) gross primary production (GPP) and (d) net primary production (NPP\(_{\text{B}}\)) for reed canary grass with fertilization (RCG-F), reed canary grass control (RCG-C) and bare peat (BP) treatments; error bars indicate standard error; the horizontal dotted line in (a) visualizes the zero line above and below which emission and uptake occur, respectively.

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not significantly different among the three treatments (Fig. 5g).

In all three treatments, N₂O fluxes were within the range of −0.4 to 25 µg N₂O-N m⁻² h⁻¹ for most of the year, with the exception of large emission peaks of up to 420 µg N₂O-N m⁻² h⁻¹ in RCG-F on July 4 and September 3 (Fig. 7b). On September 3, an N₂O emission peak of 172 µg N₂O-N m⁻² h⁻¹ was also observed in RCG-C. These peak emission events on July 4 and September 3 coincided with large rainfall events occurring just prior to both sampling dates (compare with Fig. 1c). The annual means and medians of the N₂O exchange were not significantly different among the three treatments (Fig. 5h). The N₂O emission factor (i.e., the % of N fertilizer lost as N₂O) was 0.63% in RCG-F.

Ditch emissions of CO₂, CH₄ and N₂O
Mean CO₂, CH₄ and N₂O emissions from the drainage ditches were not significantly different among treatments (Fig. 8a–c). Ditch CO₂ emissions accounted for >99% of the total ditch C flux (Table 3) and >98% of the total GHG flux from ditches (Table 4) in all three treatments.

DOC concentrations and export
Concentrations of DOC ranged within 10–19 mg L⁻¹ in the nongrowing season months and increased during the early growing season to a maximum of 25 mg L⁻¹ (Fig. 9). No soil solution samples could be retrieved from the dried out soil during the two dry summer periods. Averaged over all sampling dates and depths, the mean DOC concentrations were 17, 16 and 16 mg L⁻¹ in RCG-F, RCG-C and BP, respectively, with no significant differences among treatments (see inset figure in Fig. 9). Assuming the leaching rate to be 50% of the annual precipitation, the annual DOC export was estimated at 4.2, 4.2 and 4.1 g C m⁻² yr⁻¹ in RCG-F, RCG-C and BP, respectively (Table 3). In comparison,
the annual DOC export was 2.1, 2.1 and 2.1 g C m\(^{-2}\) yr\(^{-1}\) assuming a leaching rate of 25% and 6.3, 6.2 and 6.2 g C m\(^{-2}\) yr\(^{-1}\) assuming a leaching rate of 75% in RCG-F, RCG-C and BP, respectively.

The full carbon balance

The model estimates of cumulative annual RE were 512, 326 and 170 g C m\(^{-2}\) yr\(^{-1}\) in RCG-F, RCG-C and BP, respectively, and −433 and −125 g C m\(^{-2}\) yr\(^{-1}\) for GPP in RCG-F and RCG-C, respectively (Table 3). The carbon-use efficiencies (i.e., the ratio of NPP to GPP) were 0.39 and 0.37 in RCG-F and RCG-C, respectively. The cumulative NEE based on model estimates suggested annual CO\(_2\) emission of 79, 201 and 170 g C m\(^{-2}\) yr\(^{-1}\) in RCG-F, RCG-C and BP, respectively (Table 3).

The cumulative CH\(_4\) exchange resulted in annual CH\(_4\) emission of 0.01, 0.02 and 0.02 g C m\(^{-2}\) yr\(^{-1}\) in RCG-F, RCG-C and BP, respectively (Table 3). The
The greenhouse gas balance

Combining the area-weighted annual CO$_2$, CH$_4$ and N$_2$O exchanges from the strips and drainage ditches resulted in total GHG balances of 3.6, 7.9 and 6.6 t CO$_2$ eq ha$^{-1}$ yr$^{-1}$ in RCG-F, RCG-C and BP, respectively (Table 4). The contribution of the combined ditch CO$_2$, CH$_4$ and N$_2$O emissions to the total GHG balance in RCG-F (18%) was two times greater than in RCG-C (9%) and BP (8%). The sum of CH$_4$ and N$_2$O fluxes from both strips and ditches accounted for 5.9, 1.8 and 1.6% of the total GHG balance in RCG-F, RCG-C and BP, respectively. The N$_2$O to biomass yield ratio (i.e., the ratio of the N$_2$O flux to yield in CO$_2$ eq) increased from 0.05 in RCG-F to 0.21 in RCG-C.

Discussion

Above- and belowground biomass production and allocation

The yields in both RCG-F (2.3 t ha$^{-1}$) and RCG-C (0.4 t ha$^{-1}$) were at the bottom end of the range of 2.0 to 13.9 t ha$^{-1}$ and 1.0 to 11.0 t ha$^{-1}$ previously reported for fertilized and nonfertilized RCG cultivations, respectively (Shurpali et al., 2010; Heinsoo et al., 2011; Mander et al., 2012; Kandel et al., 2013; Karki et al., 2014). The low yields in this study were likely due to water stress constraining plant growth during an exceptionally dry summer. Our findings therefore suggest that RCG cultivation on abandoned peat extraction areas has limited potential for economically sustainable biomass production during dry years without proper WTL management.

While aboveground biomass is harvested and exported from the system, a significant fraction of belowground biomass C is permanently incorporated into the soil C pool and BNPP is therefore important with regard to long-term C sequestration (Xiong & Kätterer, 2010). Specifically, RCG cultivations have been previously highlighted as systems with a higher potential for C input into the soil compared to annual crops and nonrhizomatous perennial leys due to their expansive rootstocks and high root turnover rates (Hansson & Andren, 1986; Xiong & Kätterer, 2010; Don et al., 2012). In our study, BNPP (20–56 g C m$^{-2}$ yr$^{-1}$) was at the lower end of the 80–235 g C m$^{-2}$ yr$^{-1}$ reported for irrigated RCG cultivations in Sweden (Kätterer & Andren, 1999) and a RCG cultivation in the same abandoned peat extraction area during wet years (Mander et al., 2012). Thus, BNPP and its contribution to soil C sequestration in RCG cultivation systems might be considerably reduced during dry years.

Fertilizer effects on plant growth and soil nutrient status might affect not only the total biomass production

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Table 4 The greenhouse gas (GHG) balance (t CO$_2$ eq ha$^{-1}$ yr$^{-1}$) and its component fluxes net ecosystem CO$_2$ exchange (NEE), methane (CH$_4$) and nitrous oxide (N$_2$O) within the strips and carbon dioxide (CO$_2$), CH$_4$ and N$_2$O emissions from the ditches adjusted for their global warming potentials (34 and 298 for CH$_4$ and N$_2$O, respectively) for reed canary grass with fertilization (RCG-F), reed canary grass control (RCG-C) and bare peat (BP) treatments.

| GHG flux component | RCG-F | RCG-C | BP |
|--------------------|-------|-------|----|
| NEE                | 2.9   | 7.4   | 6.2|
| CH$_4$             | 0.006 | 0.008 | 0.009|
| N$_2$O             | 0.21  | 0.15  | 0.10|
| CO$_2$             | 13.2  | 15.2  | 10.9|
| CH$_4$             | 0.061 | 0.055 | 0.029|
| N$_2$O             | 0.04  | 0.07  | 0.17|
| Total GHG balance* | 3.6   | 7.9   | 6.6|

*The total GHG balance is the sum of area-weighted (strip width = 20 m; ditch width = 1 m) fluxes of CO$_2$, CH$_4$ and N$_2$O in strips and ditches.

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Fig. 9 Concentrations of dissolved organic carbon (DOC) in soil solution (0–50 cm depth) for reed canary grass with fertilization (RCG-F), reed canary grass control (RCG-C) and bare peat (BP) treatments; error bars indicate standard error. The inset figure shows the treatment means from all sampling dates (see Fig. 5 for a description of the box plot features).
but also its allocation into above- and belowground components (Xiong & Kätterer, 2010). For instance, the greater above- to belowground biomass ratio in RCG-F than in RCG-C suggests that fertilization resulted in greater biomass yields available for bioenergy production, however, at the cost of C allocation and long-term storage belowground. Nevertheless, given the greater absolute magnitudes of BNPP, increased C input to the soil may still occur in fertilized compared to nonfertilized RCG cultivations.

Belowground biomass dynamics have been proposed as one of the least understood aspects of plant functioning (Strand et al., 2008). Specifically, a good understanding of the partitioning of belowground biomass into roots and rhizomes is imperative as it has important implications for nutrient uptake and storage and provides information on plant survival strategies (Don et al., 2012; Jones et al., 2015). The increase in the root to rhizome biomass ratio observed in our study suggests that fertilized RCG plants invest less into storage organs, that is, rhizomes, under nutrient-rich conditions. As roots and rhizomes have different turnover times and longevity (Xiong & Kätterer, 2010), structural changes in belowground biomass due to fertilization may affect the potential for long-term soil C storage. Our study therefore highlights the need to consider fertilization effects on belowground biomass structure and allocation patterns, for example, in process-based models, to improve predictions of C sequestration under future climate and management scenarios (Strand et al., 2008; Shurpali et al., 2010; Gong et al., 2014).

Seasonal net ecosystem CO$_2$ exchange, respiration and production

Midday net CO$_2$ uptake occurred only in RCG-F whereas RCG-C and BP were consistent net CO$_2$ sources even during the midday. However, midday net CO$_2$ uptake also occurred at RCG-C after intermittent rainfall at the end of June which indicates that nonfertilized RCG cultivations might also sequester CO$_2$ given sufficient water supply. In comparison, both fertilized and nonfertilized RCG cultivations with VWC $> 55\%$ provided midday net CO$_2$ uptake for the entire growing season in a Danish study (Kandel et al., 2013). Daily net CO$_2$ uptake rates also decreased by about half from wet to dry years in a RCG cultivation in eastern Finland (Shurpali et al., 2009). The combined findings from this and other studies indicate that soil water availability is a major control of the CO$_2$ sink–source strength of RCG cultivations on drained peat soils.

Understanding the controls and seasonal patterns of the NEE component fluxes GPP, NPP and RE is essential to explain the variations in NEE. Overall, the independent estimates of annual NPP and modeled GPP based on biomass and flux sampling, respectively, agreed reasonably well in both RCG-F and RCG-C given that NPP commonly represents between 39% and 58% of GPP (Waring et al., 1998; Vicca et al., 2012). The enhanced potential for GPP and NPP due to fertilization resulted in a more pronounced response to abiotic conditions (e.g., $T_a$ and PPT) in RCG-F than in RCG-C which may explain why the timing of peak production was decoupled from that of plant development and instead more closely related to that of $T_a$ in RCG-F. In contrast, GPP and NPP in RCG-C were considerably lower and their seasonal patterns followed closely that of the plant development. Thus, GPP and NPP might be controlled primarily by abiotic controls in fertilized RCG cultivations whereas reduced photosynthetic capacity might be the main constraint on plant production in nonfertilized (i.e., low productive) cultivations.

The greater midday net CO$_2$ uptake in RCG-F relative to RCG-C was due to variations in GPP as the increase in GPP (by 69%) was larger than the increase in RE (by 37%) in RCG-F relative to RCG-C. Similarly, GPP was also reported as the main driver for interannual variations in NEE during wet and dry years in a Finnish RCG cultivation (Shurpali et al., 2009). This suggests that ensuring optimum growing conditions is essential not only for achieving economically sustainable yields but also to maximize the CO$_2$ sequestration potential in RCG cultivations.

Heterotrophic and autotrophic respiration

The difference in the RE partitioning into its components Rh and Ra between RCG-C (Rh > Ra) and RCG-F (Rh < Ra) was the result of enhanced plant growth due to fertilization and the subsequent increase of Ra in RCG-F. Meanwhile, fertilization had no significant effect on Rh. Previous studies reported a decrease or no effect on mineralization rates following fertilization, with the contrasting findings primarily related to the indirect effects of fertilization on soil pH (e.g., Fog, 1988; Aerts & Toet, 1997). Overall, the contribution of Rh to RE in RCG-F (42%) was similar to the 45% reported for a fertilized RCG cultivation in Finland (Shurpali et al., 2008) but lower than the 55–75% observed in other drained and natural peatlands (Silvola et al., 1996; Riutta et al., 2007; Biasi et al., 2012). Thus, fertilization of RCG systems may reduce the relative contribution of Rh to RE which has important implications for the response of RE to management and climate impacts as the two respiratory component fluxes Ra and Rh respond to different controls, that is, to biotic vs. abiotic variables, respectively.

During the warmest summer period (July to August), Rh was consistently higher in RCG-F and RCG-C than
in the bare peat soil. Increased mineralization of organic matter in drained peat soils following cultivation and its negative implications for the C and GHG balances have been previously highlighted in several studies (Kasimir-Klemedtsson et al., 1997; Drösler et al., 2008; Maljanen et al., 2010; Schrier-Uijl et al., 2014). Although Rh rates were not significantly different among treatments over the entire measurement period, we estimated that the cumulative CO\(_2\) loss due to mineralization during the warmest summer period (July and August) was 50 and 43 g C m\(^{-2}\) yr\(^{-1}\) greater in RCG-F and RCG-C, respectively, relative to BP. Thus, a substantial additional C input from plant CO\(_2\) uptake is required to outbalance these CO\(_2\) losses due to enhanced mineralization in cultivated organic soils.

**The full carbon balance**

Our finding that both RCG-F and RCG-C as well as the bare peat soil were considerable annual CO\(_2\) sources is in contrast to previous work conducted in the same abandoned peat extraction area in which both fertilized and nonfertilized RCG cultivations were CO\(_2\) sinks during a year with above-normal precipitation (911 826 mm) (Mander et al., 2012). This indicates a switch from a CO\(_2\) sink in a previous wetter year to a CO\(_2\) source during the dry year in the current study. Similarly, the CO\(_2\) sink strength of a fertilized RCG cultivation established on organic soil in Finland substantially decreased from \(-127\) and \(-211\) g C m\(^{-2}\) yr\(^{-1}\) during two wet years to \(-9\) and \(-52\) g C m\(^{-2}\) yr\(^{-1}\) in two dry years (Shurpali et al., 2009). Thus, these results highlight the risk that future increases in drought frequency (IPCC, 2013) might considerably reduce the potential of RCG cultivations for C sequestration.

Furthermore, while fertilized and nonfertilized RCG systems in our study and in another study in Denmark (Kandel et al., 2013) were large CO\(_2\) sources, the Finnish RCG cultivation remained a CO\(_2\) sink even during dry years (Shurpali et al., 2009). These contrasting CO\(_2\) sink–sink strength potentials could be related to differences in the annual mean \(T_a\) (6.9 and 7.3 for this study and the Danish site vs. 3.7 °C for the Finnish site, respectively) and its effect on Rh. For instance, annual Rh from BP in the current study (170 g C m\(^{-2}\) yr\(^{-1}\)) was about twice that of bare peat soil at the Finnish site (72 g C m\(^{-2}\) yr\(^{-1}\)) during a dry year (Shurpali et al., 2008). Thus, a latitudinal effect from the positive correlation between \(T_a\) and Rh might determine the CO\(_2\) sink–source strength of RCG cultivation during dry years with reduced plant CO\(_2\) uptake.

The annual CH\(_4\) emissions of \(<0.02\) g C m\(^{-2}\) yr\(^{-1}\) from RCG-F and RCG-C were much smaller compared to the ranges of 3 to 14 g C m\(^{-2}\) yr\(^{-1}\) reported for pristine peatlands (Roulet et al., 2007; Nilsson et al., 2008) and of 0.5 to 1.5 g C m\(^{-2}\) yr\(^{-1}\) observed in restored or cultivated cutaway peatlands (Tuittila et al., 2000; Hyvönen et al., 2009). These low CH\(_4\) emissions were likely the result of an exceptionally low WTL which reduced the potential for anaerobic CH\(_4\) production. In comparison, CH\(_4\) emissions of 18 to 31 g C m\(^{-2}\) yr\(^{-1}\) were observed in an Irish RCG cultivation on cutaway peatland in which the WTL remained mostly close to the surface (i.e., within 10 cm) (Wilson et al., 2009). Thus, while the contribution of the CH\(_4\) exchange to the full C balance was negligible in these dry RCG systems, cultivation techniques which raise the WTL to sustain high biomass yields during dry years, for example, paludiculture (Wichtmann & Schäfer, 2007; Karki et al., 2014), might considerably increase the potential for CH\(_4\) emissions in cultivated organic soils.

The annual DOC export (4.2 g C m\(^{-2}\) yr\(^{-1}\)) at this site was slightly lower than the 5.7 and 6.2 g C m\(^{-2}\) yr\(^{-1}\) reported from a Finnish RCG cultivation in an abandoned peat extraction area (Hyvönen et al., 2013) and from a Canadian cutover peatland (Strack et al., 2011), respectively. Together, these studies suggest that the DOC export from cultivated peatlands is considerably lower in comparison with the 12–15 g C m\(^{-2}\) yr\(^{-1}\) reported for natural peatlands (Roulet et al., 2007; Nilsson et al., 2008; Koehler et al., 2009). Nevertheless, despite the relatively small contribution to the full C balance (−2.4%) during the dry year in this study, DOC export might increase under management and climate scenarios that alter soil hydrology and runoff (Freeman et al., 2004).

After incorporating all major C fluxes of CO\(_2\), CH\(_4\) and DOC from strips and ditches, all three treatments represented net C sources. Although the net C emission was lowest in RCG-F, its source strength would further increase when including the additional C loss via the export of harvested biomass (115 g C m\(^{-2}\) yr\(^{-1}\)). Meanwhile, RCG-C was a considerably greater C source than RCG-F and BP, regardless of whether or not the additional export of harvested biomass was accounted for. This indicates that fertilized RCG cultivations may to some extent mitigate the negative effects on the full C balance commonly observed following cultivation of drained peat soils (Maljanen et al., 2010).

Ditch CO\(_2\) and CH\(_4\) emissions are rarely included in C balance estimates due to the small area coverage by ditches (Maljanen et al., 2010). Previous studies, however, reported considerable CH\(_4\) emissions specifically from water-logged drainage ditches in abandoned peat extraction areas (Maljanen et al., 2010; Hyvönen et al., 2013). In contrast, the CO\(_2\) flux constituted the dominant ditch C emission component in this study likely as the

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result of the exceptionally dry conditions at this study site at which ditches dry out completely during the summer months. Overall, ditch emissions accounted for 8–18% of the full C balance although ditches represented <5% of the area in this study. Thus, the combined CO₂ and CH₄ emissions from ditches might represent a significant component of the full C balance not only in wet but also in dry cropping systems.

The greenhouse gas balance

N₂O emissions from the strips were low (0.03–0.07 g N₂O m⁻² yr⁻¹) in all treatments when compared to the range of 0.2–5.5 g N₂O m⁻² yr⁻¹ reported for agricultural systems (Klemmedtsson et al., 2005; Maljanen et al., 2010; Don et al., 2012). However, high peak fluxes were observed after rainfall which indicates that annual N₂O emission might be greater during wetter years with more frequent rainfall events. Furthermore, fertilizer application did not result in an immediate increase in N₂O emissions although it likely enhanced the potential for the high peak N₂O emissions occurring after the heavy rainfall in late June. Overall, the combined contribution of N₂O and CH₄ emission from strips and ditches to the GHG balance was small (<6%) during the dry year in this study. To some extent, this low contribution might be due to using the median when extrapolating N₂O fluxes to the annual scale. Using the mean would have increased the contribution of N₂O (3%, 6% and 30%, respectively, for BP, RCG-C and RCG-F) as the mean is highly sensitive to the few extremely large peak emission rates measured in RCG-F and RCG-C. Given the highly non-normal distribution of N₂O fluxes, the choice of the median over the mean appeared therefore more reasonable. Nevertheless, the large uncertainty related to upsampling of N₂O fluxes in agricultural systems remains problematic and can only be reduced through measurements with high temporal (i.e., hourly to daily) resolution. In support of our findings, other studies also found a relatively small contribution of N₂O and CH₄ to the GHG balance of cultivated organic soils (Hyvönen et al., 2009; Shurpali et al., 2010; Mander et al., 2012; Karki et al., 2015a). We therefore conclude that the net CO₂ exchange determines both the C and GHG balances in RCG cultivations on organic soils. Management practices need to be therefore carefully evaluated with respect to their direct and indirect impacts on the ecosystem CO₂ exchange.

The recently proposed management strategy of cultivating perennial bioenergy crops with low N-demand, such as RCG, to reduce N₂O emissions compared to conventional crop cultivation (Don et al., 2012) is supported by the low N-emission factor (0.63) observed in this current study. However, in contrast to other perennial cropping systems (e.g., Miscanthus) which may produce high yields and greater climate benefits without fertilization (Struliu et al., 2011), the lower N₂O to yield ratio in the fertilized compared to the nonfertilized RCG system in our study suggests that the increase in biomass production and net CO₂ uptake largely exceed the increase in N₂O emissions (in CO₂ eq) following moderate fertilization, even after accounting for the additional emissions of about 0.5 t CO₂ eq ha⁻¹ yr⁻¹ occurring during fertilizer production, transport and application (Jarvooja et al., 2013). Thus, moderate fertilization could still be a beneficial management practice to maximize yield and climate benefits of RCG cultivation given the limited land resources available for reaching national bioenergy production targets. Nevertheless, other aspects such as economic constraints, effects on combustion quality and ecological concerns (e.g., groundwater eutrophication) must be considered when evaluating optimum fertilizer rates (Smith & Slater, 2010; Verhoeven & Setter, 2010; Don et al., 2012).

In contrast to previous studies suggesting that RCG cultivations provide negative GHG balances and thus mitigate the GHG emissions from drained organic soils (Shurpali et al., 2010; Mander et al., 2012), both the fertilized and nonfertilized RCG systems had positive GHG balances in the current study due to the exceptionally dry conditions during the studied year. However, previous studies indicate that a negative GHG balance could be achieved by cultivating RCG in agricultural systems with elevated WTL and sufficient soil water availability (Kätterer & André, 1999; Freibauer et al., 2004; Schrier-Uijl et al., 2014; Karki et al., 2015b). Although raising the WTL in drained organic soils might result in increased CH₄ emissions, these increases have been estimated to be modest (Komulainen et al., 1998; Tuittila et al., 2000; Wilson et al., 2009; Karki et al., 2014), and are therefore unlikely to compromise the benefits gained from increased plant growth and CO₂ uptake due to sufficient water supply. Thus, we conclude that, when converting abandoned peat extraction areas into RCG cultivations, management strategies need to ensure optimum plant growth through adequate water and nutrient supply to maximize the net ecosystem CO₂ uptake as its benefits are likely to considerably exceed the associated potentially negative effects from increased CH₄ and N₂O emissions.

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