GLOBAL CHANGE BIOLOGY
BIOENERGY

GCB Bioenergy (2016) **8**, 952–968, doi: 10.1111/gcbb.12308

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

JÄRVI JÄRVEOJA[1], MATTHIAS PEICHL[2], MARTIN MADDISON[1], ALAR TEEMUSK[1] and ÜLO MANDER[1,3]

[1]*Department of Geography, Institute of Ecology and Earth Sciences, University of Tartu, 46 Vanemuise St, Tartu 51014, Estonia,* [2]*Department of Forest Ecology and Management, Swedish University of Agricultural Sciences Skogsmarksgränd 1, 90183 Umeå, Sweden,* [3]*Hydrosystems and Bioprocesses Research Unit, National Research Institute of Science and Technology for Environment and Agriculture (Irstea), 1 rue Pierre-Gilles de Gennes CS 10030, F92761 Antony Cedex, France*

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 (CO_2), methane (CH_4) and nitrous oxide (N_2O) 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. CO_2, CH_4 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 CO_2 exchange, whereas the combined contribution of CH_4 and DOC fluxes was <5%. The GHG balances were 3.6, 7.9 and 6.6 t CO_2 eq ha^{-1} yr^{-1} in RCG-F, RCG-C and BP, respectively. The CO_2 exchange was also the dominant component of the GHG balance, while the contributions of CH_4 and N_2O were <1% and 1–6%, respectively. Overall, this study suggests that maximizing plant growth and the associated CO_2 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*

Received 12 July 2015; accepted 20 August 2015

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 km^2 of northern peatland areas which accounts for 10% of the total loss of natural peatlands due to human use (Joosten & Clarke, 2002). Peat extraction can 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 (CO_2) 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.

Correspondence: Järvi Järveoja, tel. +372 56674523, fax +372 7375825, e-mail: Jarvi.Jarveoja@ut.ee

© 2015 The Authors. *Global Change Biology Bioenergy* Published by John Wiley & Sons Ltd.
This is an open access article under the terms of the Creative Commons Attribution License, which permits use, distribution and reproduction in any medium, provided the original work is properly cited.

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 CO_2 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 & Andrén, 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 CO_2 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 CO_2 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 CO_2 exchange of vegetation and soil as well as by non-CO_2 carbon fluxes such as the exchange of methane (CH_4) 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 (N_2O) (Crutzen *et al*., 2008). Considering that N_2O has a much larger (by 298 times) global warming potential compared to CO_2 (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 N_2O 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 N_2O compared to conventional annual crops (Don *et al*., 2012). To date, the number of studies investigating the trade-off between the increased CO_2 uptake due to stimulated plant growth and the enhanced N_2O 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 CO_2 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_{KCL} 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.

Table 1 Mean values of the soil chemical and physical properties in 0–20 cm depth for reed canary grass with fertilization (RCG-F), reed canary grass control (RCG-C) and bare peat (BP) treatments; numbers in parentheses indicate standard error

Soil property*	RCG-F	RCG-C	BP
pH	5.15 (0.02)	5.11 (0.04)	5.47 (0.17)
Bulk density (g cm^{-3})	0.17 (0.01)	0.18 (0.01)	0.19 (0.01)
C (%)	46.5 (0.7)	45.5 (0.6)	44.3 (0.9)
N (%)	2.8 (0.1)	2.9 (0.1)	2.4 (0.1)
C : N	16.6	16.0	18.6
Soil C stock (kg C m^{-2})	15.6	16.6	16.4
Total P (mg g^{-1})	0.32 (0.01)	0.32 (0.01)	0.25 (0.01)
K (mg g^{-1})	0.26 (0.01)	0.12 (0.02)	0.09 (0.02)
Ca (mg g^{-1})	0.69 (0.03)	1.03 (0.03)	0.29 (0.02)
NH_4-N (mg L^{-1})	<1.5	<1.5	<1.5
NO_3-N (mg L^{-1})	85.0 (24.4)	33.0 (3.7)	49.5 (4.8)

*All variables were measured in 2014, except for C % and N % which were measured in 2012.

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).

$$NPP_B = ANPP + BNPP \quad (1)$$

Vegetation greenness index

To estimate plant development, we derived a vegetation greenness index from digital images using repeat photography (Sonnentag *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_{cc}) (Eqn 2):

$$g_{cc} = \frac{G}{R + G + B} \quad (2)$$

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_{cc} 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 Sonnentag *et al.* (2012). To estimate the collar-specific g_{cc}, images were also taken from above each collar on July 19 and analyzed for their g_{cc}. The time series of the mean g_{cc} in RCG-F and RCG-C were then multiplied with the relative difference in g_{cc} among collars to obtain collar-specific g_{cc} 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_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.). On-site meteorological measurements were complemented by hourly data of T_a, global radiation and PPT from the nearby (~20 km away) Pärnu meteorological station (Estonian Weather Service) for 2014. Global radiation was converted to PAR based on its linear regression relationship to on-site PAR during periods when both measurements were available.

Net ecosystem CO_2 exchange, respiration and production measurements

Net ecosystem CO_2 exchange (NEE) and ecosystem respiration (RE) on the RCG-F and RCG-C plots were measured using the closed dynamic chamber method at a weekly to biweekly interval from May to December 2014. Three PVC collars (Ø 50 cm) with a water-filled ring for airtight sealing were permanently installed into the soil to a depth of 10 cm within each plot. For NEE measurements, a transparent (95% transparency) Plexiglas chamber (h 50 cm, V 65 L) was used. The chamber was equipped with a TRP-2 probe (PP Systems, Hitchin, UK) which measured PAR and T_a inside the chamber. An additional sensor was placed on the chamber outside to record the ambient T_a. The chamber was climate-controlled via internal and external metal thermoelectric cooling ribs powered by a rechargeable 12-V battery. In addition, frozen cooling packs were fixed inside the chamber to further limit the T_a increase inside the chamber during measurements. The headspace air was continuously mixed by a low-speed fan. Immediately after each NEE measurement, RE measurements were conducted on the same collar with the chamber covered by an opaque, light reflecting shroud that blocked 100% of the incoming PAR.

During each chamber deployment period (3 and 4 min for RE and NEE measurements, respectively), CO_2 concentration, PAR, T_a, 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):

$$\mathrm{GPP} = \mathrm{NEE} - \mathrm{RE} \quad (3)$$

In addition, hourly net primary production rates based on the CO_2 flux measurements (NPP_F) were derived for each sampling date from the difference between NEE and heterotrophic respiration (Rh; see below) (Eqn 4):

$$\mathrm{NPP}_F = \mathrm{NEE} - \mathrm{Rh} \quad (4)$$

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 CO_2 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):

$$\mathrm{Ra} = \mathrm{RE} - \mathrm{Rh} \quad (5)$$

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_4, N_2O and nongrowing season CO_2 flux measurements

Methane (CH_4) and nitrous oxide (N_2O) 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 CH_4 and N_2O 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 CO_2 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 CO_2, CH_4 and N_2O flux measurements

To account for the spatial variation between fluxes from strips and drainage ditches, CO_2, CH_4 and N_2O 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 CO_2, CH_4 and N_2O were calculated from the change in gas concentrations in the chamber headspace volume corrected for air density using the ideal gas law (Eqn 6):

$$F_{\mathrm{dyn,stat}} = S \times \frac{p \times V \times M \times t}{R \times T_a \times A} \quad (6)$$

where F_{dyn} and F_{stat} are fluxes measured by the dynamic chamber (i.e., CO_2 in mg CO_2-C m^{-2} h^{-1}) and the static chamber method (i.e., CH_4 in μg CH_4-C m^{-2} h^{-1}, N_2O in μg N_2O-N m^{-2} h^{-1} and CO_2 in mg CO_2-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 CO_2 fluxes with a coefficient of determination (R^2) ≥ 0.90 ($P < 0.001$) 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 CO_2, CH_4 and N_2O fluxes was 0.90 ($P < 0.05$), 0.80 ($P < 0.1$) and 0.80 ($P < 0.1$), respectively, except no filtering criterion was used when the maximum difference in the concentration values was less than the gas-specific GC detection limit (i.e., <20 ppm for CO_2, <20 ppb for CH_4 and <20 ppb for N_2O). 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 CO_2, CH_4 and N_2O 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):

$$\mathrm{GPP} = \frac{a \times A_{\max} \times \mathrm{PAR} \times g_{cc_{\mathrm{norm}}}}{a \times \mathrm{PAR} \times A_{\max} \times g_{cc_{\mathrm{norm}}}} \quad (7)$$

where GPP is gross primary production (mg CO_2-C m^{-2} h^{-1}), PAR is the photosynthetically active radiation (μmol m^{-2} s^{-1}) inside the chamber, α is the light-use efficiency of photosynthesis (i.e., the initial slope of the light response curve, mg CO_2-C μmol photons^{-1}), A_{max} is maximum photosynthesis at light saturation (mg CO_2-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):

$$\mathrm{RE} = R_0 \times \exp^{(b \times T_a)} + (\beta \times g_{cc_{\mathrm{norm}}}) \times \exp^{(b \times T_a)} \quad (8)$$

where RE is ecosystem respiration (mg CO_2-C m^{-2} h^{-1}), T_a is air temperature (°C), R_0 is the soil respiration (mg CO_2-C m^{-2} h^{-1}) at 0 °C, b is the sensitivity of respiration to T_a, and β 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):

$$RE = R_0 \times \exp^{(b \times T_a)} \quad (9)$$

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 CH_4 exchange was estimated by linear interpolation. The annual N_2O exchange, however, was calculated by scaling the median of the measured fluxes to an annual sum as the occurrence of episodic high peak fluxes would have caused annual sums to be overestimated in the linear interpolation method. Annual ditch emissions were estimated by scaling the mean CO_2 and CH_4 fluxes as well as the median N_2O flux to the entire year. The annual GHG balances were estimated by converting the cumulative strip and ditch fluxes to CO_2 equivalents (CO_2 eq) using the global warming potentials (GWP, over a 100-year time frame including carbon–climate feedbacks) of 34 and 298 for CH_4 and N_2O, 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 cm^2 (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 water leaching rate which was assumed to be 50% of the monthly precipitation based on previous literature estimates (Kløve *et al*., 2010; Hyvönen *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.

Table 2 Parameters for the gross primary production (GPP) (Eqn 7) and ecosystem respiration (RE) (Eqns 8 and 9) models for reed canary grass with fertilization (RCG-F), reed canary grass control (RCG-C) and bare peat (BP) treatments; α is the quantum-use efficiency of photosynthesis (mg CO_2-C μmol photons); A_{max} is the maximum rate of photosynthesis at light saturation (mg CO_2-C m^{-2} h^{-1}); R_0 is the soil respiration (mg CO_2-C m^{-2} h^{-1}) at 0 °C; b is the sensitivity of respiration to air temperature; and β is a scaling parameter representing the contribution of plant respiration to ecosystem respiration; numbers in parentheses indicate standard error; Adj. R^2 = adjusted R^2

Model properties	RCG-F	RCG-C	BP
GPP model			
α	−1.21 (0.73)	−0.33 (0.22)	n.a.
A_{max}	−386 (56)	−120 (25)	n.a.
Adj. R^2	0.71	0.67	n.a.
RE model			
R_0	11.9 (8.8)	12.8 (5.5)	13.0 (3.6)
b	0.045 (0.008)	0.035 (0.007)	0.045 (0.010)
β	61 (22)	37 (12.2)	n.a.
Adj. R^2	0.90	0.89	0.61

n.a., not applicable.

Statistical analysis

Collar flux data were averaged for each plot before conducting further statistical analysis to avoid pseudoreplication. The nonparametric 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.

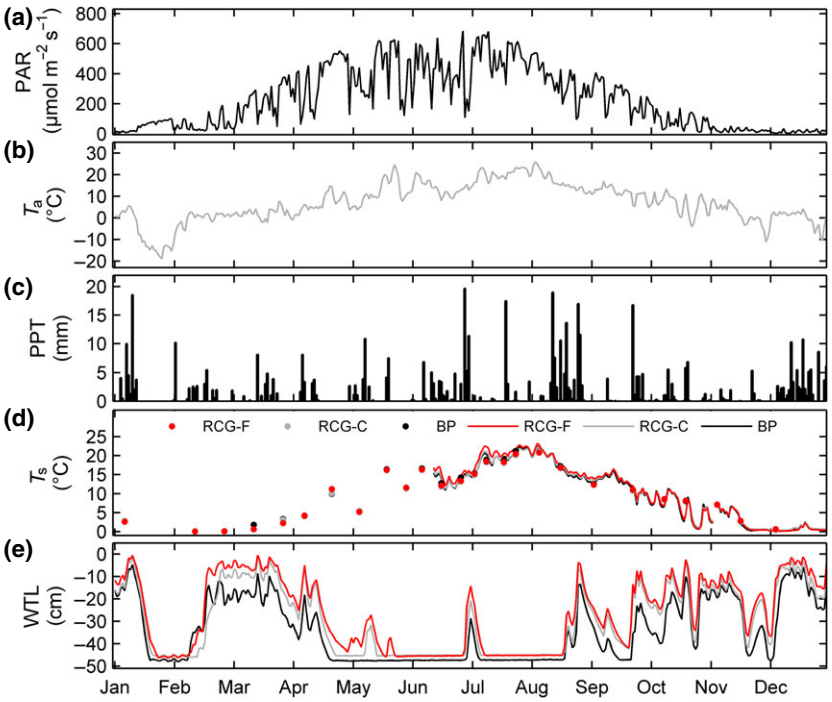


Fig. 1 Seasonal patterns of (a) photosynthetically active radiation (PAR), (b) air temperature (T_a), (c) precipitation (PPT), (d) soil temperature (T_s) at 5 and 10 cm depths (dots and lines represent manual and automated measurements, respectively) and (e) water table level (WTL) for reed canary grass with fertilization (RCG-F), reed canary grass control (RCG-C) and bare peat (BP) treatments in 2014. Data for PAR, T_a and PPT were taken from the Pärnu meteorological station for January 1 to June 11 and measured at the study site from June 12 to December 31.

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.

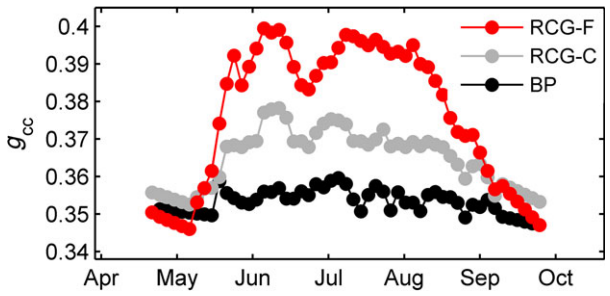


Fig. 2 Temporal pattern of 3-day means of the vegetation greenness index (g_{cc}) for reed canary grass with fertilization (RCG-F), reed canary grass control (RCG-C) and bare peat (BP) treatments in 2014.

Above- and belowground biomass production and allocation

At the time of harvest, mean aboveground biomass (± standard error) in RCG-F and RCG-C was 234 ± 19 and 42 ± 6 g m^{-2}, respectively (Fig. 3). The belowground biomass increased from 536 ± 17 and 364 ± 22 g m^{-2} in April to 646 ± 23 and 416 ± 29 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

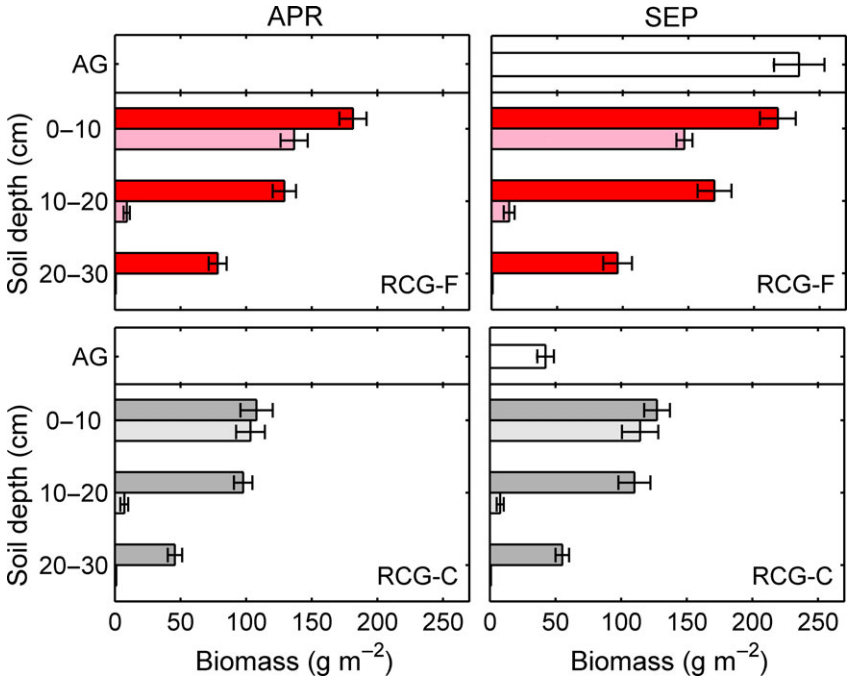


Fig. 3 Aboveground (AG) biomass (white bars) in September and the vertical distribution (0–30 cm depth) of belowground root (dark red and gray bars) and rhizome (light red and gray bars) biomass (dry weight) as a mean (±standard error) of April and September sampling dates in reed canary grass with fertilization (RCG-F) and reed canary grass control (RCG-C) treatments.

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_B derived from biomass sampling were greater in RCG-F (−115 and −55 g C m^{-2} yr^{-1}) than in RCG-C (−20 and −26 g C m^{-2} yr^{-1}) (Table 3). The ratio of above- to belowground NPP_B decreased from 2.1 in RCG-F to 0.8 in RCG-C, respectively.

Seasonal net ecosystem CO_2 exchange, respiration and production

A negative midday NEE, that is, CO_2 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} noted 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 CO_2 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 CO_2 sources for most of the nongrowing season. Continuous midday CO_2 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 T_a 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 g_{cc}. 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).

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_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_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.

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 mg C m^{-2} h^{-1}) than in RCG-C (~60 mg C m^{-2} h^{-1}). Except for one sampling date (May 30), the Ra to Rh ratio was always ≥1 in RCG-F while it was ≤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_2O exchanges

Throughout the growing season, CH_4 emission occurred in the range of 0.01 to 9.3 μg 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 to 3.7 μg C m^2 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

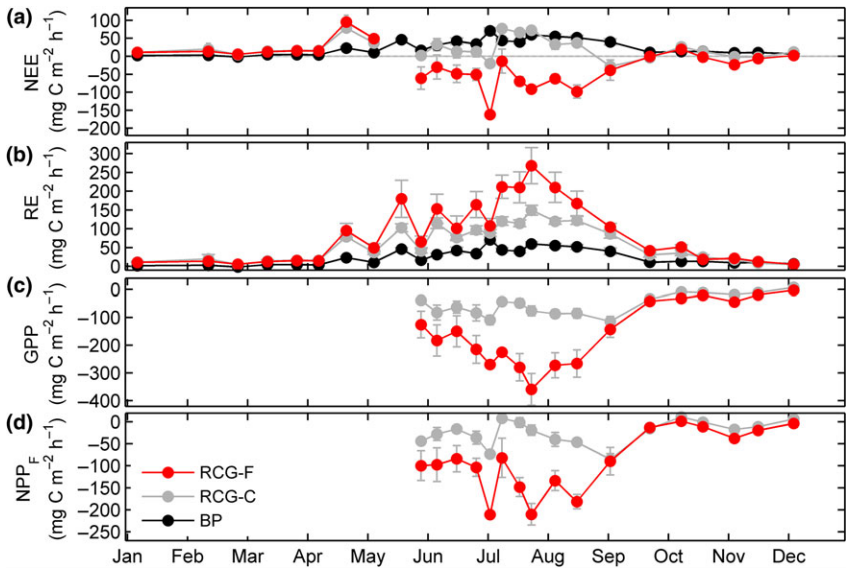


Fig. 4 (a) Net ecosystem CO_2 exchange (NEE), (b) ecosystem respiration (RE), (c) gross primary production (GPP) and (d) net primary production (NPP_F) for reed canary grass with fertilization (RCG-F), reed canary grass control (RCG-C) and bare peat (BP) treatments; error bars indicate standard error; the horizontal dotted line in (a) visualizes the zero line above and below which emission and uptake occur, respectively.

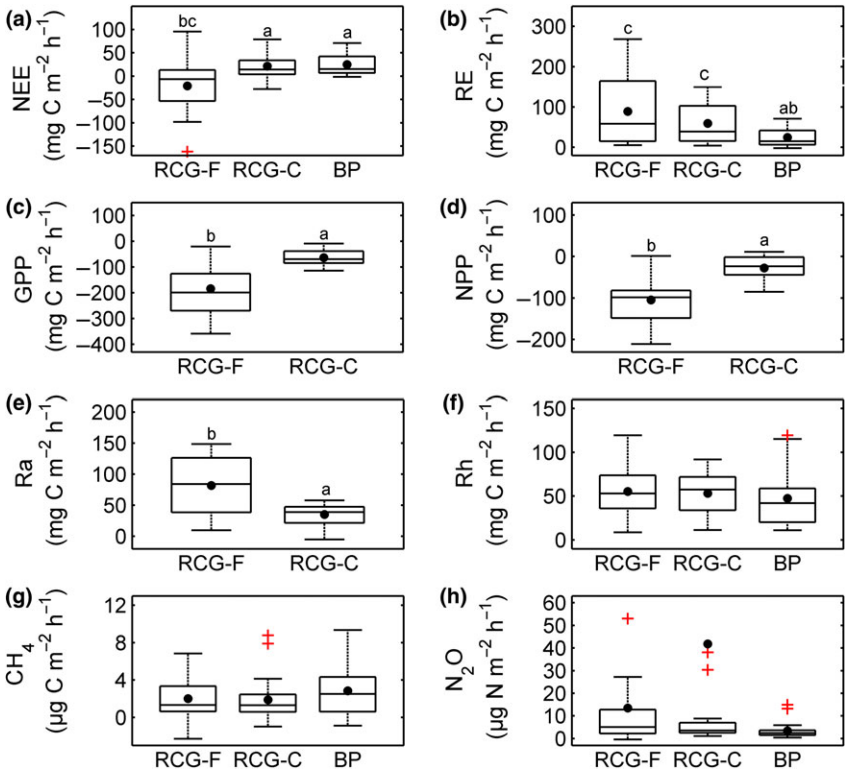


Fig. 5 Box plots for (a) net ecosystem CO_2 exchange (NEE), (b) ecosystem respiration (RE), (c) gross primary production (GPP), (d) net primary production (NPP_F; derived from flux measurements), (e) autotrophic respiration (Ra), (f) heterotrophic respiration (Rh), (g) methane (CH_4) and (h) nitrous oxide (N_2O) fluxes for reed canary grass with fertilization (RCG-F), reed canary grass control (RCG-C) and bare peat (BP) treatments; boxplots for NEE, RE, CH_4 and N_2O are based on annual data; boxplots for GPP, NPP, Ra and Rh are based on growing season data. Note that outliers of 391 and 421 µg N m-2 h-1 for RCG-F and 171 µg N m-2 h-1 for RCG-C are not shown in panel (h). The central line and dot are the median and mean, respectively, the edges of the box are the 25th and 75th percentiles, the whiskers extend to the most extreme data points which are not considered outliers, red cross symbols indicate outliers defined as data points exceeding a standard deviation of 2.7, and different letters indicate significant ($P < 0.05$) differences among treatments.

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

In all three treatments, N_2O fluxes were within the range of −0.4 to 25 μg N_2O-N m^{-2} h^{-1} for most of the year, with the exception of large emission peaks of up to 420 μg N_2O-N m^{-2} h^{-1} in RCG-F on July 4 and September 3 (Fig. 7b). On September 3, an N_2O emission peak of 172 μg N_2O-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 N_2O exchange were not significantly different among the three treatments (Fig. 5h). The N_2O emission factor (i.e., the % of N fertilizer lost as N_2O) was 0.63% in RCG-F.

Ditch emissions of CO_2, CH_4 and N_2O

Mean CO_2, CH_4 and N_2O emissions from the drainage ditches were not significantly different among treatments (Fig. 8a–c). Ditch CO_2 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,

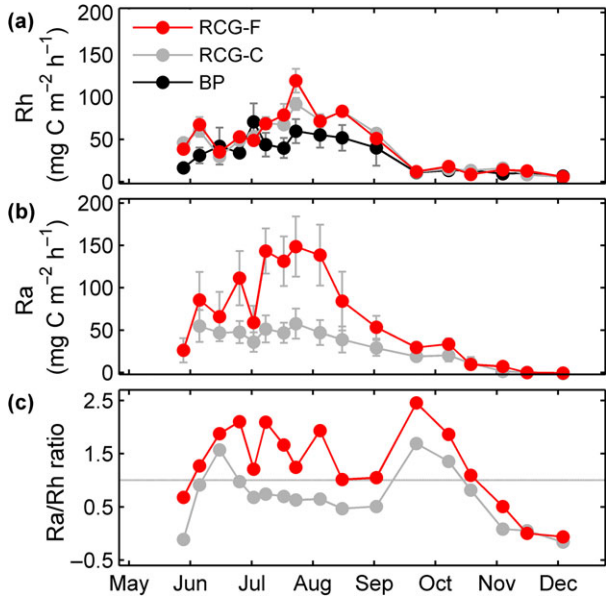


Fig. 6 (a) Heterotrophic respiration (Rh), (b) autotrophic respiration (Ra) and (c) the Ra/Rh ratio for reed canary grass with fertilization (RCG-F), reed canary grass control (RCG-C) and bare peat (BP) treatments; error bars indicate standard error; the horizontal dotted line in (c) indicates Ra/Rh = 1 (i.e., Ra = Rh).

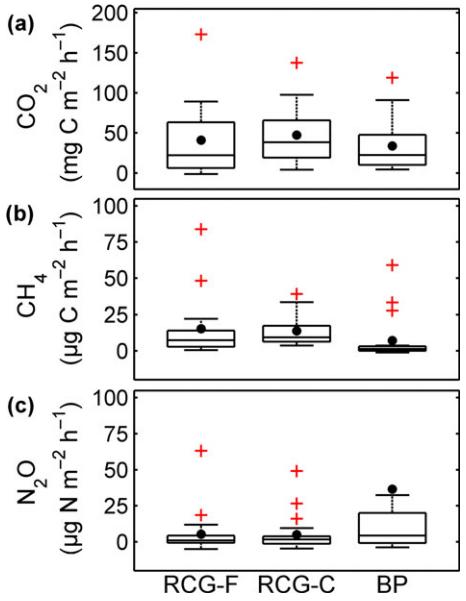


Fig. 8 Box plots for ditch emissions of (a) carbon dioxide (CO_2), (b) methane (CH_4) and (c) nitrous oxide (N_2O) for reed canary grass with fertilization (RCG-F), reed canary grass control (RCG-C) and bare peat (BP) treatments (see Fig. 5 for a description of the box plot features).

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_B 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

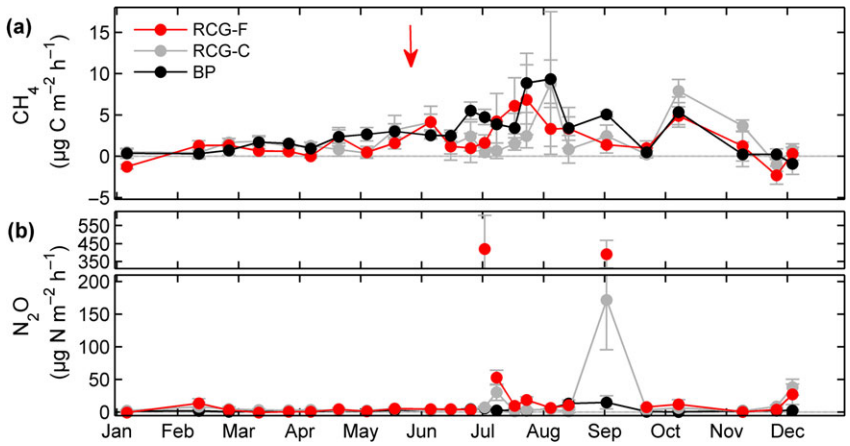


Fig. 7 Exchanges of (a) methane (CH_4) and (b) nitrous oxide (N_2O) for reed canary grass with fertilization (RCG-F), reed canary grass control (RCG-C) and bare peat (BP) treatments; error bars indicate standard error; arrow indicates timing of fertilizer application; the horizontal dotted line visualizes the zero line above and below which emission and uptake occur, respectively.

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_2O) within the strips and carbon dioxide (CO_2), CH_4 and N_2O emissions from the ditches adjusted for their global warming potentials (34 and 298 for CH_4 and N_2O, 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
Strips			
NEE	2.9	7.4	6.2
CH_4	0.006	0.008	0.009
N_2O	0.21	0.15	0.10
Ditches			
CO_2	13.2	15.2	10.9
CH_4	0.061	0.055	0.029
N_2O	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_2O in strips and ditches.

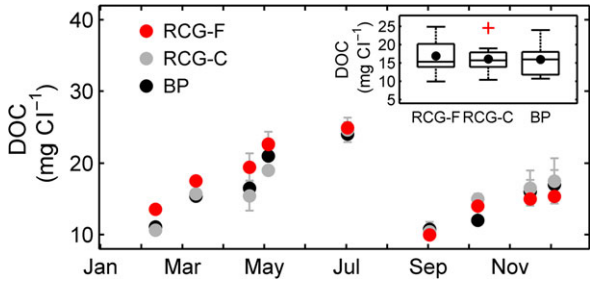


Fig. 9 Concentrations of dissolved organic carbon (DOC) in soil solution (0–50 cm depth) for reed canary grass with fertilization (RCG-F), reed canary grass control (RCG-C) and bare peat (BP) treatments; error bars indicate standard error. The inset figure shows the treatment means from all sampling dates (see Fig. 5 for a description of the box plot features).

cumulative CO_2 and CH_4 fluxes from the ditches were 360, 415 and 298 g C m^{-2} yr^{-1} and 0.13, 0.12 and 0.06 g C m^{-2} yr^{-1} in RCG-F, RCG-C and BP, respectively (Table 3). Combining the area-weighted annual CO_2 and CH_4 exchanges from the 20 m wide strips and the 1 m wide drainage ditches with the DOC export resulted in full C balances of 96, 215 and 180 g C m^{-2} yr^{-1} in RCG-F, RCG-C and BP, respectively (Table 3). The relative contribution of DOC export to the full C balance was 4.1%, 1.9% and 2.2% in RCG-F, RCG-C and BP, respectively, while the relative contribution of the CH_4 exchange was <1% for all treatments.

The greenhouse gas balance

Combining the area-weighted annual CO_2, CH_4 and N_2O 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_2O 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_2O 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_2O to biomass yield ratio (i.e., the ratio of the N_2O 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 & Andrén, 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

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 source–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

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 CO_2 and CH_4 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

N_2O emissions from the strips were low (0.03–0.07 g N_2O m^{-2} yr^{-1}) in all treatments when compared to the range of 0.2–5.5 g N_2O m^{-2} yr^{-1} reported for agricultural systems (Klemedtsson *et al*., 2005; Maljanen *et al*., 2010; Don *et al*., 2012). However, high peak fluxes were observed after rainfall which indicates that annual N_2O emission might be greater during wetter years with more frequent rainfall events. Furthermore, fertilizer application did not result in an immediate increase in N_2O emissions although it likely enhanced the potential for the high peak N_2O emissions occurring after the heavy rainfall in late June. Overall, the combined contribution of N_2O and CH_4 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 N_2O fluxes to the annual scale. Using the mean would have increased the contribution of N_2O (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 N_2O fluxes, the choice of the median over the mean appeared therefore more reasonable. Nevertheless, the large uncertainty related to upscaling of N_2O 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 N_2O and CH_4 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 CO_2 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 CO_2 exchange.

The recently proposed management strategy of cultivating perennial bioenergy crops with low N-demand, such as RCG, to reduce N_2O 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 (Strullu *et al*., 2011), the lower N_2O to yield ratio in the fertilized compared to the nonfertilized RCG system in our study suggests that the increase in biomass production and net CO_2 uptake largely exceed the increase in N_2O emissions (in CO_2 eq) following moderate fertilization, even after accounting for the additional emissions of about 0.5 t CO_2 eq ha^{-1} yr^{-1} occurring during fertilizer production, transport and application (Järveoja *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 & Andrén, 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 CH_4 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 CO_2 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 CO_2 uptake as its benefits are likely to considerably exceed the associated potentially negative effects from increased CH_4 and N_2O emissions.

Acknowledgements

This study was supported by the European Regional Development Fund through ENVIRON (Centre of Excellence in Environmental Adaption), by the Ministry of Education and Research of the Republic of Estonia (grant IUT2-16) and by the Estonian Environmental Observatory Biosphere-Atmosphere Science and Development Programme: BioAtmos (KESTA,

SLOOM12022T). We would like to thank Dr. Ivika Ostonen for comments and discussions, Dr. Marika Truu for participation in setting up the experiment and Dr. Veiko Uri for providing the plate lysimeters. We are grateful to Mae Uri, Kristi Jänesmägi, Kaarel Kukk and Kevin Kesküla for their help in the laboratory. We also thank the three anonymous reviewers for their constructive comments on the original version of the manuscript.

References

Adler PR, Grosso SJD, Parton WJ (2007) Life-cycle assessment of net greenhouse-gas flux for bioenergy cropping systems. *Ecological Applications*, **17**, 675–691.

Aerts R, Toet S (1997) Nutritional controls on carbon dioxide and methane emission from *Carex*-dominated peat soils. *Soil Biology and Biochemistry*, **29**, 1683–1690.

Biasi C, Pitkamaki AS, Tavi NM, Koponen HT, Martikainen PJ (2012) An isotope approach based on C-13 pulse-chase labelling vs. the root trenching method to separate heterotrophic and autotrophic respiration in cultivated peatlands. *Boreal Environment Research*, **17**, 184–192.

Couwenberg J, Thiele A, Tanneberger F *et al.* (2011) Assessing greenhouse gas emissions from peatlands using vegetation as a proxy. *Hydrobiologia*, **674**, 67–89.

Crutzen PJ, Mosier AR, Smith KA, Winiwarter W (2008) N_2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. *Atmospheric Chemistry and Physics*, **8**, 389–395.

Don A, Osborne B, Hastings A *et al.* (2012) Land-use change to bioenergy production in Europe: implications for the greenhouse gas balance and soil carbon. *GCB Bioenergy*, **4**, 372–391.

Drösler M, Freibauer A, Christensen TR, Friborg T (2008) Observations and status of peatland greenhouse gas emissions in Europe. In: *The Continental-Scale Greenhouse Gas Balance of Europe* (eds Dolman AJ, Valentini R, Freibauer A), pp. 243–261. Springer, New York.

Fog K (1988) The effect of added nitrogen on the rate of decomposition of organic matter. *Biological Reviews*, **63**, 433–462.

Freeman C, Fenner N, Ostle NJ *et al.* (2004) Export of dissolved organic carbon from peatlands under elevated carbon dioxide levels. *Nature*, **430**, 195–198.

Freibauer A, Rounsevell MDA, Smith P, Verhagen J (2004) Carbon sequestration in the agricultural soils of Europe. *Geoderma*, **122**, 1–23.

Gong J, Kellomäki S, Shurpali NJ *et al.* (2014) Climatic sensitivity of the CO_2 flux in a cutaway boreal peatland cultivated with a perennial bioenergy crop (*Phalaris arundinaceae*, L.): beyond diplotelmic modeling. *Agricultural and Forest Meteorology*, **198–199**, 232–249.

Gorham E (1991) Northern peatlands: role in the carbon cycle and probable responses to climatic warming. *Ecological Applications*, **1**, 182–195.

Hansson A-C, Andren O (1986) Below-ground plant production in a perennial grass ley (*Festuca pratensis* Huds.) assessed with different methods. *Journal of Applied Ecology*, **23**, 657–666.

Heinsoo K, Hein K, Melts I, Holm B, Ivask M (2011) Reed canary grass yield and fuel quality in Estonian farmers' fields. *Biomass and Bioenergy*, **35**, 617–625.

Hyvönen NP, Huttunen JT, Shurpali NJ, Tavi NM, Repo ME, Martikainen PJ (2009) Fluxes of nitrous oxide and methane on an abandoned peat extraction site: effect of reed canary grass cultivation. *Bioresource Technology*, **100**, 4723–4730.

Hyvönen NP, Huttunen JT, Shurpali NJ, Lind SE, Marushchak ME, Heitto L, Martikainen PJ (2013) The role of drainage ditches in greenhouse gas emissions and surface leaching losses from a cutaway peatland cultivated with a perennial bioenergy crop. *Boreal Environment Research*, **18**, 109–126.

IPCC (2013) The physical science basis. In: *Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (eds Stocker TF, Qin D, Plattner G-K, Tignor M, Allen SK, Boschung J, Nauels A, Xia Y, Bex V, Midgley PM). Cambridge University Press, Cambridge.

Järveoja J, Laht J, Maddison M, Soosaar K, Ostonen I, Mander Ü (2013) Mitigation of greenhouse gas emissions from an abandoned Baltic peat extraction area by growing reed canary grass: life-cycle assessment. *Regional Environmental Change*, **13**, 781–795.

Jones MB, Finnan J, Hodkinson TR (2015) Morphological and physiological traits for higher biomass production in perennial rhizomatous grasses grown on marginal land. *GCB Bioenergy*, **7**, 375–385.

Joosten H, Clarke D (2002) *Wise Use of Mires and Peatlands: Background and Principles Including a Framework for Decision-Making*. International Mire Conservation Group and Internatonal Peat Society, Saarijärvi, Finland.

Kandel TP, Elsgaard L, Karki S, Lærke PE (2013) Biomass yield and greenhouse gas emissions from a drained fen peatland cultivated with reed canary grass under different harvest and fertilizer regimes. *BioEnergy Research*, **6**, 883–895.

Karki S, Elsgaard L, Audet J, Lærke PE (2014) Mitigation of greenhouse gas emissions from reed canary grass in paludiculture: effect of groundwater level. *Plant and Soil*, **383**, 217–230.

Karki S, Elsgaard L, Kandel TP, Laerke PE (2015a) Full GHG balance of a drained fen peatland cropped to spring barley and reed canary grass using comparative assessment of CO_2 fluxes. *Environmental Monitoring and Assessment*, **187**, 62.

Karki S, Elsgaard L, Lærke PE (2015b) Effect of reed canary grass cultivation on greenhouse gas emission from peat soil at controlled rewetting. *Biogeosciences*, **12**, 595–606.

Kasimir-Klemedtsson Å, Klemedtsson L, Berglund K, Martikainen P, Silvola J, Oenema O (1997) Greenhouse gas emissions from farmed organic soils: a review. *Soil Use and Management*, **13**, 245–250.

Kätterer T, Andrén O (1999) Growth dynamics of reed canarygrass (*Phalaris arundinacea* L.) and its allocation of biomass and nitrogen below ground in a field receiving daily irrigation and fertilisation. *Nutrient Cycling in Agroecosystems*, **54**, 21–29.

Kinmonth-Schultz H, Kim S (2011) Carbon gain, allocation, and storage in rhizomes in response to elevated atmospheric carbon dioxide and nutrient supply in a perennial C 3 grass, *Phalaris arundinacea*. *Functional Plant Biology*, **38**, 797–807.

Klemedtsson L, Von Arnold K, Weslien P, Gundersen P (2005) Soil CN ratio as a scalar parameter to predict nitrous oxide emissions. *Global Change Biology*, **11**, 1142–1147.

Kløve B, Sveistrup TE, Hauge A (2010) Leaching of nutrients and emission of greenhouse gases from peatland cultivation at Bodin, Northern Norway. *Geoderma*, **154**, 219–232.

Koehler A-K, Murphy K, Kiely G, Sottocornola M (2009) Seasonal variation of DOC concentration and annual loss of DOC from an Atlantic blanket bog in South Western Ireland. *Biogeochemistry*, **95**, 231–242.

Komulainen V-M, Nykänen H, Martikainen PJ, Laine J (1998) Short-term effect of restoration on vegetation change and methane emissions from peatlands drained for forestry in southern Finland. *Canadian Journal of Forest Research*, **28**, 402–411.

Lemus R, Lal R (2005) Bioenergy crops and carbon sequestration. *Critical Reviews in Plant Sciences*, **24**, 1–21.

Lewandowski I, Scurlock JMO, Lindvall E, Christou M (2003) The development and current status of perennial rhizomatous grasses as energy crops in the US and Europe. *Biomass and Bioenergy*, **25**, 335–361.

Loftfield N, Flessa H, Augustin J, Beese F (1997) Automated gas chromatographic system for rapid analysis of the atmospheric trace gases methane, carbon dioxide, and nitrous oxide. *Journal of Environment Quality*, **26**, 560.

Mäkiranta P, Hytönen J, Lasse A *et al.* (2007) Soil greenhouse gas emissions from afforested organic soil croplands and cutaway peatlands. *Boreal Environment*, **12**, 159–175.

Maljanen M, Sigurdsson BD, Guðmundsson J, Óskarsson H, Huttunen JT, Martikainen PJ (2010) Greenhouse gas balances of managed peatlands in the Nordic countries – present knowledge and gaps. *Biogeosciences*, **7**, 2711–2738.

Mander Ü, Järveoja J, Maddison M, Soosaar K, Aavola R, Ostonen I, Salm J-O (2012) Reed canary grass cultivation mitigates greenhouse gas emissions from abandoned peat extraction areas. *GCB Bioenergy*, **4**, 462–474.

McClaugherty CA, Aber JD, Melillo JM (1982) The role of fine roots in the organic matter and nitrogen budgets of two forested ecosystems. *Ecology*, **63**, 1481–1490.

Nilsson M, Sagerfors J, Buffam I *et al.* (2008) Contemporary carbon accumulation in a boreal oligotrophic minerogenic mire – a significant sink after accounting for all C-fluxes. *Global Change Biology*, **14**, 2317–2332.

Peichl M, Sonnentag O, Nilsson MB (2015) Bringing color into the picture: using digital repeat photography to investigate phenology controls of the carbon dioxide exchange in a boreal mire. *Ecosystems*, **18**, 115–131.

Riutta T, Laine J, Tuittila E-S (2007) Sensitivity of CO2 exchange of fen ecosystem components to water level variation. *Ecosystems*, **10**, 718–733.

Roulet NT, Lafleur PM, Richard PJH, Moore TR, Humphreys ER, Bubier J (2007) Contemporary carbon balance and late Holocene carbon accumulation in a northern peatland. *Global Change Biology*, **13**, 397–411.

Salm J-O, Maddison M, Tammik S, Soosaar K, Truu J, Mander Ü (2012) Emissions of CO_2, CH_4 and N_2O from undisturbed, drained and mined peatlands in Estonia. *Hydrobiologia*, **692**, 41–55.

Schrier-Uijl AP, Kroon PS, Leffelaar PA, van Huissteden JC, Berendse F, Veenendaal EM (2010) Methane emissions in two drained peat agro-ecosystems with high and low agricultural intensity. *Plant and Soil*, **329**, 509–520.

Schrier-Uijl AP, Kroon PS, Hendriks DMD, Hensen A, Van Huissteden J, Berendse F, Veenendaal EM (2014) Agricultural peatlands: towards a greenhouse gas sink – a synthesis of a Dutch landscape study. *Biogeosciences*, **11**, 4559–4576.

Shurpali NJ, Hyvönen NP, Huttunen JT, Biasi C, Nykänen H, Pekkarinen N, Martikainen PJ (2008) Bare soil and reed canary grass ecosystem respiration in peat extraction sites in Eastern Finland. *Tellus Series B*, **60**, 200–209.

Shurpali NJ, Hyvönen NP, Huttunen JT *et al.* (2009) Cultivation of a perennial grass for bioenergy on a boreal organic soil – carbon sink or source? *GCB Bioenergy*, **1**, 35–50.

Shurpali NJ, Strandman H, Kilpeläinen A *et al.* (2010) Atmospheric impact of bioenergy based on perennial crop (reed canary grass, *Phalaris arundinaceae*, L.) cultivation on a drained boreal organic soil. *GCB Bioenergy*, **2**, 130–138.

Silvola J, Alm J, Ahlholm U, Nykanen H, Martikainen PJ (1996) CO_2 fluxes from peat in boreal mires under varying temperature and moisture conditions. *Journal of Ecology*, **84**, 219–228.

Smith R, Slater FM (2010) The effects of organic and inorganic fertilizer applications to *Miscanthus×giganteus*, *Arundo donax* and *Phalaris arundinacea*, when grown as energy crops in Wales, UK. *GCB Bioenergy*, **2**, 169–179.

Sonnentag O, Hufkens K, Teshera-Sterne C *et al.* (2012) Digital repeat photography for phenological research in forest ecosystems. *Agricultural and Forest Meteorology*, **152**, 159–177.

Strack M, Tóth K, Bourbonniere R, Waddington JM (2011) Dissolved organic carbon production and runoff quality following peatland extraction and restoration. *Ecological Engineering*, **37**, 1998–2008.

Strand AE, Pritchard SG, McCormack ML, Davis MA, Oren R (2008) Irreconcilable differences: fine-root life spans and soil carbon persistence. *Science*, **319**, 456–458.

Strullu L, Cadoux S, Preudhomme M, Jeuffroy M-H, Beaudoin N (2011) Biomass production and nitrogen accumulation and remobilisation by *Miscanthus×giganteus* as influenced by nitrogen stocks in belowground organs. *Field Crops Research*, **121**, 381–391.

Sundh I, Nilsson M, Mikkela C, Granberg G, Svensson BH (2000) Fluxes of methane and carbon dioxide on peat-mining areas in Sweden. *Ambio*, **29**, 499–503.

Tuittila E-S, Komulainen V-M, Vasander H, Nykänen H, Martikainen PJ, Laine J (2000) Methane dynamics of a restored cut-away peatland. *Global Change Biology*, **6**, 569–581.

Uri V, Löhmus K, Mander Ü *et al.* (2011) Long-term effects on the nitrogen budget of a short-rotation grey alder (*Alnus incana* (L.) Moench) forest on abandoned agricultural land. *Ecological Engineering*, **37**, 920–930.

Verhoeven JTA, Setter TL (2010) Agricultural use of wetlands: opportunities and limitations. *Annals of Botany*, **105**, 155–163.

Vicca S, Luyssaert S, Peñuelas J *et al.* (2012) Fertile forests produce biomass more efficiently. *Ecology Letters*, **15**, 520–526.

Waddington JM, Warner KD, Kennedy GW (2002) Cutover peatlands: a persistent source of atmospheric CO_2. *Global Biogeochemical Cycles*, **16**, 1–7.

Waring RH, Landsberg JJ, Williams M (1998) Net primary production of forests: a constant fraction of gross primary production? *Tree Physiology*, **18**, 129–134.

Wichtmann W, Schäfer A (2007) Alternative management options for degraded fens – utilisation of biomass from rewetted peatlands.

Wilson D, Alm J, Laine J, Byrne KA, Farrell EP, Tuittila E-S (2009) Rewetting of cutaway peatlands: are we re-creating hot spots of methane emissions? *Restoration Ecology*, **17**, 796–806.

Xiong S, Kätterer T (2010) Carbon-allocation dynamics in reed canary grass as affected by soil type and fertilization rates in northern Sweden. *Acta Agriculturae Scandinavica, Section B — Soil & Plant Science*, **60**, 24–32.