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ORIGINAL PUBLICATIONS

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IV. Leppelt T, Dechow R, Gebbert S, Freibauer A, Lohila A, Augustin J, Drösler M, Fiedler S, Glatzel S, Höper H, **Järveoja J**, Lærke PE, Maljanen M, Mander Ü, Mäkiranta P, Minkkinen K, Ojanen P, Regina K, Strömgren M (2014) Nitrous oxide emission budgets and land-use-driven hotspots for organic soils in Europe. *Biogeosciences*, 11, 6595–6612.

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The participation of the author in preparing the listed publications was as follows:

Publication I: The author contributed to developing the experimental design, was partly responsible for the field data collection and data processing, contributed to data analysis and to the writing of the manuscript.

Publication II: The author contributed to developing the experimental design, was partly responsible for the field data collection, processed and analyzed the data, wrote the manuscript.

Publication III: The author contributed to developing of the experimental design, was partly responsible for the field data collection, processed and analyzed the data, wrote the manuscript.

Publication IV: The author participated in data collection, synthesized and provided data.

ABSTRACT

Natural peatlands are an important component of the global carbon (C) cycle storing > 25% of the global soil C pool and providing a small but persistent sink for atmospheric carbon dioxide (CO_2). Within the past century, however, large peatland areas have been drained and exploited for various purposes, including peat extraction for fuel and horticultural use. After cessation of peat extraction activities, enhanced CO_2 and nitrous oxide (N_2O) emissions occur while emissions of methane (CH_4) commonly decrease due to increased aeration of the surface peat layer. Altogether these greenhouse gas (GHG) fluxes may have a large impact on atmospheric GHG concentrations and global climate. Thus, there is a need for after-use strategies that mitigate the GHG emissions from these degraded peat soils. Currently, however, knowledge about the impact of different after-use options and associated management effects on the annual C and GHG balances of abandoned peat extraction areas is limited.

This dissertation investigated the impact of bioenergy crop cultivation and peatland restoration on the GHG exchanges from abandoned peat extraction areas. For this purpose, GHG fluxes (including CO_2, CH_4 and N_2O) were quantified using the closed chamber technique in fertilized and nonfertilized reed canary grass (RCG; *Phalaris arundinacea*) cultivations, restored peatlands with high and low water table level (WTL) and in abandoned bare peat (BP) soil. Above- and belowground biomass production as well as vegetation cover were estimated by destructive sampling, soil coring and vegetation inventory. Various environmental variables were measured to identify the main abiotic controls of the individual fluxes. In addition, N_2O flux data from 109 sites with organic soils across temperate and boreal Europe were synthesized and combined with a modeling approach to estimate the European N_2O budget and its main drivers.

Net C uptake and negative GHG balances of −6.0 and −3.9 t CO_2 eq ha^{-1} yr^{-1} were observed in the fertilized and nonfertilized RCG treatments, respectively, in the cool and wet year 2010 (Publication I – Mander *et al.*, 2012), whereas net C losses and net GHG emissions of 3.6 and 7.9 t CO_2 eq ha^{-1} yr^{-1} occurred in the same treatments, respectively, in the warm and dry year 2014 (Publication II – Järveoja *et al.*, 2015). In comparison, net C losses and positive GHG balances of 2.5 and 6.6 t CO_2 eq ha^{-1} yr^{-1} were observed at the BP treatment in 2010 and 2014, respectively. Overall, these results suggest that RCG cultivation may provide an effective method for mitigating the net C and GHG emissions from abandoned peat extraction areas. However, these findings also highlight the strong impact of climatic conditions on the C and GHG balances of RCG cultivations on drained organic soils.

Furthermore, greater net C uptake and lower net GHG emissions observed in fertilized relative to nonfertilized RCG cultivations suggest that fertilization may increase the climate benefit potential of RCG cultivations through enhancing biomass production and net CO_2 uptake which largely exceeded the

increase in soil N_2O emissions following fertilization. Net CO_2 exchange dominated the C and GHG balances in all treatments while the contributions of CH_4, N_2O and dissolved organic carbon fluxes remained relatively small (1–6%). Thus, when converting drained peatlands 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 since its benefits are likely to exceed the associated potentially negative effects from increased CH_4 and N_2O emissions.

Net C losses and positive GHG balances of 4.1, 3.8 and 10.2 t CO_2 eq ha^{-1} yr^{-1} were observed in restored treatments with high and low WTL and BP, respectively (Publication III – Järveoja *et al.*, Submitted). This demonstrates that restoration may effectively mitigate the negative climate impacts of drained peat soils. Changes in the C and GHG balances following restoration of the peat extraction area were mainly due to a large reduction in heterotrophic respiration which advocates raising the WTL as an effective method to reduce the aerobic organic matter decomposition commonly occurring in drained peatlands. Furthermore, raising the WTL resulted in significantly reduced N_2O emissions whereas the effect on the CH_4 fluxes was negligible in both restored treatments compared to the abandoned BP site. The results further suggests that, although differences in the re-established WTL baselines affected vegetation composition and plant-related CO_2 fluxes, the impact on the net C and GHG balances was limited three years following restoration of the peat extraction area.

The N_2O flux data synthesis showed that N_2O emissions from organic soils across Europe were predominantly driven by human management effects on the WTL, while climatic parameters played a secondary role (Publication IV – Leppelt *et al.*, 2014). The total European N_2O budget for organic soils was estimated at 149.5 Gg N yr^{-1} to which peat extraction areas contributed a total of 0.1 Gg N yr^{-1}. This suggests that, due to their small area coverage, peat extraction areas have little impact on the European N_2O budget when compared to other land use types such as croplands and grasslands.

Overall, this dissertation concludes that both bioenergy crop cultivation and peatland restoration may provide effective methods for mitigating the negative climate impact of abandoned peat extractions areas. The choice of after-use is, however, in addition to its atmospheric impact dependent on several other factors and therefore ultimately site-specific. Future research on bioenergy crop production needs to address alternative management options (e.g. water table management) to ensure sustainable yields and climate benefits in bioenergy cultivations on drained organic soils. Furthermore, long-term observations are needed to improve our understanding of the impacts of bioenergy cultivation and peatland restoration on the ecosystem C and GHG balances over longer time scales. This knowledge will also improve predictions of ecosystem responses to changes in future management strategies and climatic conditions.

1. INTRODUCTION

Peatland ecosystems are terrestrial water-logged environments that have accumulated vast amounts of carbon (C) in the form of peat, i.e. partially decomposed organic material, since the end of the last glacial period about 10 000 years ago (Frolking *et al.*, 2001; Laine *et al.*, 2006). Most of the world's peatlands (i.e. > 80%) are located in the northern hemisphere covering large areas in Europe, Russia and North-America (Joosten & Clarke, 2002; Vasander *et al.*, 2003; Rydin & Jeglum, 2006). Although they cover only ~3% of the global land area, northern peatlands have been estimated to store about a third of the global soil C pool (Gorham, 1991; Turunen *et al.*, 2002). Moreover, recent estimates suggest that peatlands in their natural state continue to act as small but persistent contemporary C sinks with mean annual uptake rates of ~15–30 g C m^{-2} yr^{-1} (Roulet *et al.*, 2007; Nilsson *et al.*, 2008; Koehler *et al.*, 2011). Carbon accumulation in northern peatland ecosystems occurs mainly due to the slow decomposition rate of organic matter under water-logged and thus poorly aerated conditions (Clymo, 1984). The C sink strength, however, is strongly dependent on climatic conditions and may vary among years even within the same peatland. Recent studies show that climate anomalies such as drought or heat wave events associated with lower water table levels (WTLs) may severely reduce or even reverse the C sink function of peatlands (Shurpali *et al.*, 1995; Alm *et al.*, 1999; Lafleur *et al.*, 2003; Lund *et al.*, 2010; Peichl *et al.*, 2014). Natural and anthropogenic disturbances that alter the hydrological and biogeochemical conditions in peatland ecosystems may significantly affect ecosystem functioning and the balance between production and decomposition processes which in turn may therefore have severe implications for the global C cycle and climate (Limpens *et al.*, 2008; Maljanen *et al.*, 2010; Fenner & Freeman, 2011; Charman *et al.*, 2013).

1.1. The component fluxes of the peatland carbon and greenhouse gas balances

The main component of the peatland C balance is the net ecosystem carbon dioxide (CO_2) exchange (NEE) which is determined by the photosynthetic uptake of CO_2 during plant production and the CO_2 losses that occur due to plant respiration and the microbial decomposition of dead organic matter. In addition to the net CO_2 exchange, the microbial production and oxidation of methane (CH_4) in the anaerobic and aerobic peat layers, respectively, and the subsequent net CH_4 exchange represent another important component of the peatland C balance. Due to the water-logged conditions, most of the peat layer is anoxic and natural peatlands therefore commonly act as major sources of CH_4 to the atmosphere (Harriss *et al.*, 1985; Lai, 2009). Furthermore, the lateral export of dissolved organic carbon (DOC) with groundwater leaching may contribute significantly to the C balance (Roulet *et al.*, 2007; Nilsson *et al.*, 2008; Koehler *et al.*, 2011). Thus, the net peatland C balance (i.e. the C sink-source strength) is the product of CO_2, CH_4 and DOC fluxes and therefore sensitive to changes in environmental conditions and disturbance which may affect either one of these component fluxes.

Apart from their contribution to the peatland C balance, CO_2 and CH_4 also act as potent greenhouse gases (GHGs) and affect the global climate through their radiative forcing. Since the global warming potential (expressed in CO_2 equivalents over a 100 year time frame) of CH_4 is 34 times greater relative to CO_2 (IPCC, 2013), the importance of the CH_4 exchange is much more pronounced within the climate context relative to its contribution to the C balance. In addition, nitrous oxide (N_2O) is a third major GHG which is both produced and consumed during microbial processes in soils. In most cases, the production of N_2O largely exceeds its consumption which may lead to considerable emissions to the atmosphere. Compared to other ecosystems (e.g. croplands, grasslands), N_2O emissions are commonly small in natural peatlands (Martikainen *et al.*, 1993; Regina *et al.*, 1996; Silvan *et al.*, 2005; Roobroeck *et al.*, 2010). However, since the global warming potential of N_2O is 298 times greater than that of CO_2 (IPCC, 2013), even small changes in N_2O emissions could have severe impacts on the GHG balance of peatland ecosystems.

1.2. Human use of peatlands

Within the past century, a large fraction of natural peatlands has been exploited for various economic purposes including agriculture, forestry and peat extraction. In northern regions, human exploitation has altered 50×10^6 ha of peatlands so severely that peat accumulation has stopped entirely (Lappalainen, 1996; Joosten & Clarke, 2002; Strack, 2008). The largest share of losses, both in absolute and relative terms to its original peatland extent, has been suffered in Europe showing clearly that an abundance of natural peatlands is no guarantee of their long-term survival (Joosten & Clarke, 2002). The total area of peat-covered land in Estonia is 1×10^6 ha which corresponds to ~22% of the country's mainland territory (Orru & Orru, 2008). Thus, Estonia is considered to be one of the most peatland-rich countries in the world. Recent estimates, however, show that at present only 5.5% (245 000 ha) of the total peatland area still remains in its natural state while the remainder has been drained or influenced by drainage to the extent that no longer allows peat accumulation (Paal & Leibak, 2011).

Conventional peatland utilization requires drainage to lower the WTL. This is commonly achieved by establishing a network of drainage ditches across the peatland. To facilitate agricultural and forestry use of peatlands, drainage is essential for regulating the soil oxygen and water conditions in order to meet the growth requirements of the cultivated crops and to improve forest productivity

(Laine *et al.*, 2006). In case of commercial peat extraction, however, drainage is crucial to initiate the drying process of surface peat and accommodate heavy peat harvesting machinery (Charman, 2002). While drainage is a fundamental prerequisite for principally any type of peatland utilization, lowering the WTL and aerating the peat also inevitably leads to peatland degradation due to peat oxidation, shrinkage and compaction as well as to decreased hydraulic conductivity (Waddington *et al.*, 2002) which has increasingly negative implications for the management of drained peatlands.

Out of the various uses of peatlands, the level of disturbance imposed on the ecosystem can be considered to be highest in the case of peat extraction since the peatland is severely degraded after cessation of extraction activities. In comparison to forestry or agricultural use, the vegetation and thus plant production is entirely eliminated as a result of peat harvesting operations (Frilander *et al.*, 1996). Furthermore, in addition to initial drainage, peat extraction also requires progressively increasing the drainage depth as peat harvest continues. In contrast to other peatland uses, peat extraction also encompasses mechanical stripping and export of the accumulated peat deposit. The removal of peat material may be limited to the uppermost, less decomposed peat layer in the case of horticultural use of peat, or may extend to the entire peat layer if peat is harvested also for the purpose of energy production. As a result, a major negative long-term consequence of commercial peat extraction is that, following the cessation of peat harvesting activities, vast areas of abandoned and degraded bare peat soils remain.

Within northern regions, a total of approximately 5×10^6 ha of natural peatlands have been used for peat extraction (Joosten & Clarke, 2002). In Estonia, peat is the third most important domestic fuel resource and therefore its use for heating purposes has a long history (Paal & Leibak, 2011). Currently, peat is being extracted for industrial purposes on about 19 574 ha (Orru & Orru, 2008). Moreover, given the extent of exploitable peat resources, it has been estimated that commercial peat mining at the current levels of $0.3–1.5\times10^6$ t of dry peat yr^{-1} could potentially continue for several hundred years (Orru & Orru, 2008). The total area of abandoned peat extraction sites in Estonia is currently 9371 ha and is expected to double over the coming decades as further depletion of resources and cessation of ongoing peat extraction will occur (Ramst & Orru, 2009). Given the current extent and potential future expansion of these abandoned peat extraction areas, there is a growing interest and need to understand how GHG emissions from these degraded peat soils contribute to regional and national carbon and greenhouse gas budgets.

1.3. Greenhouse gas emissions from drained and abandoned peat extraction areas

Greenhouse gas emissions from peatlands are mainly determined by the properties of the remaining peat such as pH, temperature, C substrate quality, nutrient availability as well as water and oxygen contents (Regina *et al.*, 1996; Basiliko *et al.*, 2007; Limpens *et al.*, 2008; Leifeld *et al.*, 2012; Bragazza *et al.*, 2013). Generally, increased soil aeration associated with lower WTLs stimulates the decomposition of the exposed peat layer causing large CO_2 emission to the atmosphere (Silvola *et al.*, 1996; Waddington *et al.*, 2002; Basiliko *et al.*, 2007). Peatland drainage and extraction operations have been shown to increase CO_2 emissions to the atmosphere by as much as 400% with oxidation rates remaining high potentially for decades after the peat extraction ceases (Silvola *et al.*, 1996; Waddington *et al.*, 2002; Waddington & McNeil, 2002). Moreover, the decrease in soil moisture and greater substrate supply due to increased mineralization rates commonly result in enhanced N_2O production and emission from abandoned peat extraction areas relative to natural peatlands (Martikainen *et al.*, 1993; Regina *et al.*, 1996; Maljanen *et al.*, 2010). On the other hand, the reduction of the waterlogged anaerobic zone following drainage usually leads to a decrease in CH_4 production and emission in drained peatlands (Sundh *et al.*, 2000; Tuittila *et al.*, 2000).

In addition to these soil biogeochemical controls, climatic factors such as air temperature and precipitation patterns might further affect the magnitudes and temporal patterns of these GHG fluxes (Shurpali *et al.*, 1995; Lafleur *et al.*, 2003; Roulet *et al.*, 2007; Limpens *et al.*, 2008). However, while concerns about potential GHG emissions from abandoned peat extraction areas have been raised in previous studies (Sundh *et al.*, 2000; Waddington *et al.*, 2002; Salm *et al.*, 2012), the current understanding of the complex interactions between the various controls and GHG fluxes as well as data on annual ecosystem C and GHG balances is still limited. Specifically, high N_2O emissions from drained organic soils may have great importance at the national level in countries which contain a large share of drained peatlands, yet measurements of this potent GHG are not included in many current studies of GHG budgets. Thus, the future expansion of peat extraction activities into pristine areas will result in a growing demand for developing appropriate after-use strategies that have the potential for mitigating the GHG emissions from abandoned peat extraction areas (Tuittila *et al.*, 2000; Maljanen *et al.*, 2010).

1.4. After-use options for mitigating carbon and greenhouse gas emissions from abandoned peat extraction areas: bioenergy crop cultivation and peatland restoration

The main after-use options for abandoned peat extraction areas encompass forestry, agriculture, berry plantations, bioenergy crop cultivation and peatland restoration. Ultimately, the choice of after-use is determined by a combination of site-specific factors which include the condition of the drainage network, the properties of the residual peat layer (e.g. pH, thickness, nutrient status, degree of decomposition), properties of the mineral soil as well as site accessibility and socio-economic interests of various land owners (i.e. private and state) (McNally, 1995). Due to concerns about rising GHG concentrations in the atmosphere and its effect on the global climate, another important factor that may influence the choice of after-use form is its potential for mitigating GHG emissions.

Among the different after-use options, cultivation of dedicated bioenergy crops has been suggested as a promising alternative to increase the proportion of renewable energy supply while creating a sink for atmospheric CO_2 (Lemus & Lal, 2005; Don *et al.*, 2012). Specifically, bioenergy crop cultivation enhances the uptake of CO_2 from the atmosphere during plant photosynthesis and its storage in above- and belowground biomass and soil. In addition, using biomass as an alternative energy source results in reduced CO_2 emissions from fossil fuel burning. In most bioenergy cropping systems, however, fertilizer is applied to maximize biomass production which may cause high N_2O emissions (Maljanen *et al.*, 2010; 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 emissions due to fertilization is, however, limited and its implication for the GHG balance of bioenergy cultivations therefore still highly uncertain. Among various bioenergy crop species, the perennial reed canary grass (RCG; *Phalaris arundinacea*), has been proposed as the most suitable bioenergy crop on organic soils in the Nordic countries due to its tolerance to low temperatures and short growing seasons (Venendaal *et al.*, 1997; Lewandowski *et al.*, 2003). Moreover, RCG is also considered as one of the highest yielding cool-season grasses (Wrobel *et al.*, 2009) with a tendency to also allocate significant amounts of biomass to belowground organs (i.e. roots and rhizomes) (Xiong & Kätterer, 2010). Few studies in Northern regions have previously indicated that RCG cultivations on drained organic soils may provide a net CO_2 sink on the annual scale (Shurpali *et al.*, 2009, 2010; Karki *et al.*, 2015) without causing significant emissions of CH_4 and N_2O (Hyvönen *et al.*, 2009; Kandel *et al.*, 2013a; Karki *et al.*, 2014, 2015). In contrast, other studies reported that RCG cultivations act as CO_2 sources during the growing season (Kandel *et al.*, 2013b; Karki *et al.*, 2014). One reason for contrasting findings on the C sink-source strength of RCG systems might be the impact of climatic conditions (Shurpali *et al.*, 2009). Thus, much uncertainty remains to date with regards to the potential of RCG cultivations for mitigating GHG emissions from abandoned peat extraction areas.

Apart from bioenergy crop cultivation, restoration towards natural peatland ecosystems with resumed long-term peat accumulation is an after-use option that is both desirable from the ecological perspective (Rochefort & Lode, 2006; Lamers *et al.*, 2015) and potentially beneficial with regards to mitigating GHG emissions from drained organic soils (Tuittila *et al.*, 1999, 2000; Graf & Rochefort, 2009; Waddington *et al.*, 2010; Strack & Zuback, 2013). Peatland restoration includes the active re-introduction of natural peatland vegetation communities (i.e. fragments of moss and vascular companion species) and raising the WTL to create favorable conditions for the development of a peatland ecosystem. As a result, peatland restoration commonly results in enhanced CO_2 uptake by the re-established vegetation and decreased CO_2 losses due to reduced aerobic decomposition of organic matter (Tuittila *et al.*, 1999; Waddington & Warner, 2001; Maljanen *et al.*, 2010). On the other hand, however, the presence of vegetation (through substrate supply and aerenchymatic CH_4 transport) and anoxic conditions due to higher WTLs may increase the production and emission of CH_4 from restored peatlands (Tuittila *et al.*, 2004; Waddington & Day, 2007; Vanselow-Algan *et al.*, 2015). The net C balance of restored peatlands is therefore highly sensitive to vegetation and WTL dynamics (Tuittila *et al.*, 2004; Strack & Zuback, 2013). Studies of restoration projects using the moss layer transfer method or rewetting have reported successful results in terms of peatland vegetation recovery and the re-establishment of the C sink function (Graf & Rochefort, 2009; Waddington *et al.*, 2010). However, estimates for the time required until the restored peatland regains its C sink function vary between 10 to 50 years (Bortoluzzi *et al.*, 2006; Waddington *et al.*, 2010). Moreover, most C balance estimates are currently limited to the growing season (Tuittila *et al.*, 1999, 2000, 2004; Waddington *et al.*, 2010; Samaritani *et al.*, 2011; Strack *et al.*, 2014) while ignoring the additional C losses that occur during the non-growing season period (Yli-Petäys *et al.*, 2007; Strack & Zuback, 2013). In addition, the potential of peatland restoration in reducing N_2O emissions relative to drained organic soil has not been studied to date. Thus, the current knowledge of annual C and GHG budgets of restored peatlands is limited and further research is needed to better evaluate the potential of restoration for mitigating the negative climate effects of abandoned peat extraction areas.

1.5. Objectives

The goal of this dissertation was to investigate the impact of bioenergy crop (i.e. reed canary grass) cultivation and peatland restoration on the GHG exchanges (including CO_2, CH_4 and N_2O) in abandoned peat extraction areas.

The main objectives were:

1. To determine the magnitudes, seasonal dynamics and controls of GHG fluxes in fertilized and nonfertilized reed canary grass cultivations on a former peat extraction area compared to abandoned bare peat soil (Publications I and II)
2. To derive and compare annual C and GHG balances of fertilized and nonfertilized reed canary grass cultivations relative to those of abandoned bare peat soil (Publications I and II)
3. To examine the magnitudes, seasonal dynamics and controls of GHG fluxes in a former peat extraction area restored with high and low water table levels compared to abandoned bare peat soil (Publication III)
4. To estimate and compare annual C and GHG balances of peatland restoration with high and low water table levels relative to those of abandoned bare peat soil (Publication III)
5. To explore the controls and budgets of N_2O fluxes from organic soils in Europe (Publication IV)

The main hypotheses were:

1. The C and GHG balances of reed canary grass cultivation will be improved (i.e. greater net uptake or less emission of C and GHGs) relative to the abandoned peat extraction area due to enhanced plant CO_2 uptake
2. Fertilization of reed canary grass cultivation will enhance the C sink strength and improve the GHG balance relative to nonfertilized cultivation due to its beneficial effects on the plant CO_2 uptake
3. The C and GHG balances of the restored peatland will be improved (i.e. greater net uptake or less emission of C and GHGs) relative to the abandoned peat extraction area due the decreased peat mineralization following raising of the WTL
4. Restoration with high water table level will result in improved C and GHG balances relative to restoration with low water table level due to a greater reduction in peat mineralization and greater water availability enhancing gross primary production

2. MATERIAL AND METHODS

2.1. Study sites

The impact of bioenergy crop cultivation on GHG fluxes (Publications I and II) was investigated at the largest Estonian peat extraction area in Lavassaare (58°34′20″N, 24°23′15″E) which is situated in western Estonia (Figure 1). 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 1960's and lasted until 2006. The remaining peat deposit is ~0.6–1.2 m deep and consists of well-mineralized *Phragmites-Carex* peat. In 2007, 18 abandoned peat extraction strips were tilled and sown with seeds of the Estonian-bred reed canary grass variety 'Pedja'. Twelve of these strips were fertilized with different fertilizer types and rates, while six strips remained nonfertilized.

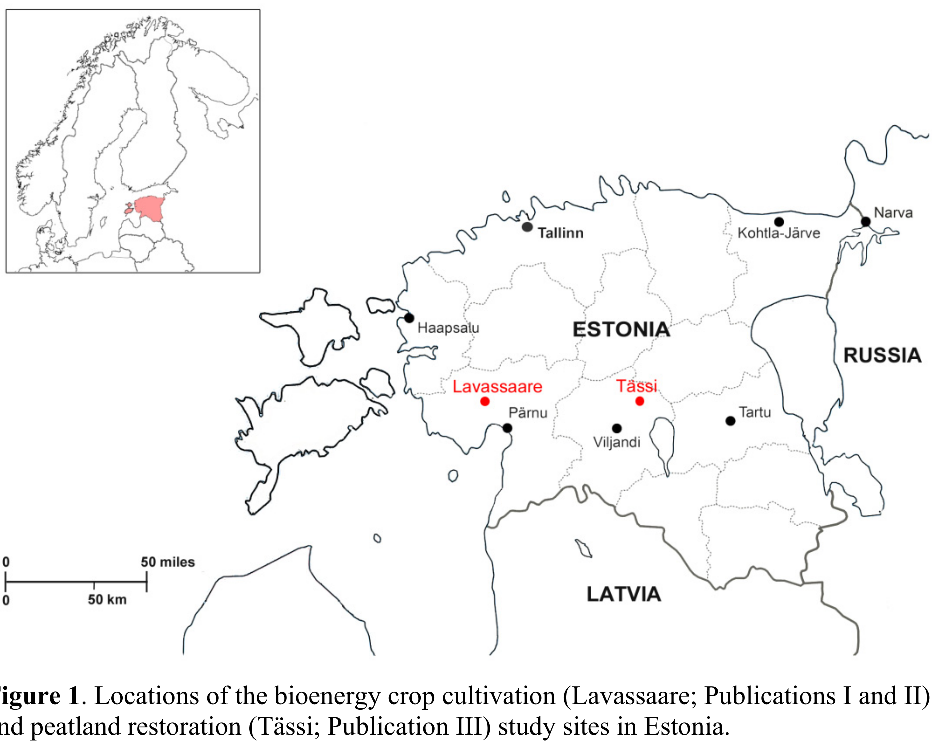

Figure 1. Locations of the bioenergy crop cultivation (Lavassaare; Publications I and II) and peatland restoration (Tässi; Publication III) study sites in Estonia.

In 2010 the following three strips were selected within the abandoned peat extraction area: fertilized RCG (RCG-F), nonfertilized RCG (RCG-C) and abandoned bare peat (BP) (Publication I). The fertilized RCG strip received

76 kg N, 88 kg P and 43 kg K ha^{-1} in 2007 and 82 kg N, 94 kg P and 46 kg K ha^{-1} in 2008 as a combination of mineral and organic fertilizers. In addition, a nearby natural raised bog (NB) and a cultivated fen meadow (FM) were included as reference sites. The NB had 3 m of peat deposits, of which the upper 1.3 m layer consisted of non-mineralized *Sphagnum fuscum* peat. The FM had a 0.7 m deep highly mineralized fen peat layer, and its grass cover was dominated by *Elytrigia repens* and *Urtica dioica*. The main soil properties for each treatment are summarized in Table 1.

Table 1. Main topsoil properties at the Lavassaare study site in 2010 (Publication I); numbers in parenthesis indicate standard error. RCG-F, fertilized reed canary grass cultivation; RCG-C, nonfertilized reed canary grass cultivation; BP, bare peat; NB, natural bog; FM, fen meadow.

Soil property	RCG-F	RCG-C	BP	NB	FM
C (%)	51.0 (3.0)	49.0 (1.5)	50.0 (1.0)	49.0 (1.0)	16.0 (2.0)
N (%)	2.6 (0.04)	2.7 (0.1)	2.3 (0.1)	1.5 (0.1)	1.1 (0.08)
C:N	19.6	18.1	21.7	32.7	14.5
Total P (mg g^{-1})	0.52 (0.1)	0.30 (0.05)	0.24 (0.05)	0.38 (0.1)	0.54 (0.1)

In 2014, a new experimental set up was established within the Lavassaare peat extraction area (Publication II). Two cultivated strips and two adjacent bare peat strips were selected for a replicated study design. In each cultivated strip, a fertilized and a nonfertilized plot (2.5×10 m) were established. Thus, the study included two replicate plots for each of the treatments: fertilized RCG (RCG-F), nonfertilized RCG (RCG-C) and bare peat (BP). The fertilized plots received 72 kg N, 18 kg P and 36 kg K ha^{-1} of mineral fertilizer once per year since 2012. The main soil properties for each treatment are summarized in Table 2.

Table 2. Main topsoil properties at the Lavassaare study site in 2014 (Publication II); numbers in parenthesis indicate standard error. RCG-F, fertilized reed canary grass cultivation; RCG-C, nonfertilized reed canary grass cultivation; BP, bare peat.

Soil property	RCG-F	RCG-C	BP
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
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)
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)

The impact of peatland restoration on GHG fluxes (Publication III) was investigated in the Tässi peat extraction area (58°32′16″N, 25°51′43″E) located in central Estonia (Figure 1). The long-term mean (1981–2010) annual temperature and precipitation in the region are 5.8 °C and 764 mm, respectively (Estonian Weather Service). Peat extraction in the peatland started in late 1960's and today peat is harvested for horticultural purposes on about 264 ha. The study was carried out on a 4.5 ha area which included an abandoned bare peat area set aside from peat extraction in the early 1980's as well as an area of 0.24 ha within the abandoned site which was restored with *Sphagnum* moss in April 2012 to initiate the development of a natural bog. Restoration was carried out following a slightly modified protocol of the moss layer transfer technique (Quinty & Rochefort, 2003) which has since 1990's been widely used in North-America to restore bogs after peat extraction. The two main restoration steps included raising the WTL in the peatland by damming the drainage ditches and spreading *Sphagnum* and vascular plant fragments collected from a nearby (10 km) donor site (Soosaare bog). Prior to re-introducing the vegetation fragments, the restoration site was divided into wetter and drier sections by lowering the peat surface by 10 cm for approximately one third of the area. This resulted in restoration treatments with high (Res-H) and low (Res-L) water table levels. In addition, an unrestored bare peat site (BP) was included in the study as a reference. Two replicate plots (20×20 m) were established for each of the Res-H, Res-L and BP treatments. The main soil properties for each treatment are summarized in Table 3.

Table 3. Main topsoil properties at the Tässi study site in 2014 (Publication III); numbers in parenthesis indicate standard error. Res-H, restoration with high water table level; Res-L, restoration with low water table level; BP, bare peat.

Soil property	Res-H	Res-L	BP
C (%)	49.0 (0.6)	50.0 (0.3)	48.0 (0.6)
N (%)	0.61 (0.04)	0.76 (0.05)	0.85 (0.04)
C:N	80.3	65.8	56.5
Total P (mg g^{-1})	0.21 (0.03)	0.23 (0.02)	0.36 (0.03)
K (mg g^{-1})	0.16 (0.007)	0.21 (0.003)	0.09 (0.004)
pH	3.97 (0.07)	3.93 (0.07)	3.90 (0.06)
Bulk density (g cm^{-3})	0.08 (0.002)	0.09 (0.003)	0.13 (0.004)

Data for the synthesis and upscaling of N_2O fluxes from European organic soils (Publication IV) was obtained from 109 sites spread across the main organic soil regions of temperate and boreal Europe. The majority of measurements came from central Europe (Germany, Netherlands) and from northern European countries like Finland, Sweden and Estonia.

2.2. Environmental variables

During each GHG flux sampling campaign (Publications I–III), manual measurements of environmental variables were conducted. Soil temperatures (Ts) at four different depths (10, 20, 30 and 40 cm) were recorded by a hand-held temperature logger (Comet Systems Ltd.). Manual WTL measurements were taken inside groundwater observation wells (Ø 7.5 cm, 1.0 m long PVC pipes perforated and sealed in the lower end) installed at each sampling location. Furthermore, groundwater temperature, pH, redox potential, dissolved oxygen content, electrical conductivity as well as nitrate (NO_3^-) and ammonium (NH_4^+) concentrations were measured in the same observation wells using YSI Professional Plus handheld instruments (YSI Inc.). In addition, volumetric soil water content (VWC; depth 0–5 cm) was measured using a handheld soil moisture sensor (model GS3, Decagon Devices Inc.) (Publications II and III).

Air temperature, precipitation and radiation data were obtained from nearby meteorological stations of the Estonian Weather Service (Publications I–III). In 2014, automated meteorological stations were set up at the Lavassaare (Publication II) and Tässi (Publication III) sites to continuously monitor on-site air temperature (Ta; model 107, Campbell Scientific Inc.), photosynthetically active radiation (PAR; model LI-190SL, LI-COR Inc.) and precipitation (PPT; tipping bucket model 52202, R. M. Young Company). Soil temperature (depths of 5 and 30 cm) was measured with temperature probes (model 107, Campbell Scientific Inc.) and VWC (depth 5 cm) with water content reflectometers (model CS615, Campbell Scientific Inc.). All data were collected in 1 min intervals and stored as 10 min averages on a CR1000 datalogger (Campbell Scientific Inc.). Continuous 30 min records of the WTL relative to the soil surface were obtained with submerged HOBO Water Level Loggers (Onset Computer Corporation) placed inside perforated 1.0 m long PVC pipes (Ø 5 cm; sealed in the lower end).

In addition, composite soil samples (0–10 or 0–20 cm depth; 3 replicates) were taken at both of the study sites, Lavassaare and Tässi, and analyzed for total carbon, total nitrogen, phosphorous, potassium, calcium and sulphur concentrations at the Tartu Laboratory of the Estonian Environmental Research Centre using the standard methods (APHA, 1989) (Publications I–III). Additional samples were taken from 0–10 cm depth to determine soil pH as well as bulk density (Publications II and III). Also, water samples were taken at each flux sampling location from groundwater wells or plate lysimeters and analyzed for dissolved organic carbon (DOC), NO_3^- and NH_4^+ concentrations (Publications I and II).

2.3. Vegetation measurements

At the Lavassaare site, above- and belowground biomass pools in the cultivated RCG treatments were estimated with destructive sampling and soil coring, respectively, in September 2010 and April 2011 (Publication I) as well as in April and September 2014 (Publication II). Harvested plant material was analyzed for total C and N concentrations at the Tartu Laboratory of the Estonian Environmental Research Centre.

Annual aboveground net primary production was calculated by multiplying the harvested biomass (dry weight) with its C concentration (Publications I and II). Annual belowground net primary production was calculated by multiplying the change in root and rhizome biomass (dry weight) with its C concentration (Publications I and II). The change in belowground biomass was obtained by multiplying the estimated average root and rhizome biomass pools with treatment-specific turnover rates calculated from Xiong & Kätterer (2010) (Publication I) or with the maximum-minimum method (McClaugherty *et al.*, 1982) as the difference between the estimated maximum (September sampling) and minimum (April sampling) belowground biomass pools (Publication II).

In 2014, the temporal development of vegetation was quantified using a greenness index based on digital repeat photography (Publication II). At the Tässi site, vegetation cover and species composition of the restored treatments was determined inside each of the flux measurement collars by vegetation inventory in late spring 2014 (Publication III).

2.4. Greenhouse gas flux measurements

Fluxes of CO_2, CH_4 and N_2O were measured using the closed chamber technique (Hutchinson & Livingston, 1993) (Publications I–III). Measurements were conducted in weekly to monthly intervals from May 2010 to May 2011 (Publication I) and from January to December 2014 (Publication II) at the Lavassaare site and from March 2014 to February 2015 at the Tässi site (Publication III). At each sampling location, a collar (Ø 50 cm) with a water-filled ring for air-tight sealing was permanently installed to a soil depth of 10 cm. For measurements of ecosystem respiration (RE), CH_4 and N_2O fluxes, opaque PVC chambers (h 50 cm, V 65 L) were placed on the collars. During the 1-hour deployment period, 3–4 air samples were manually drawn into pre-evacuated (0.3 mbar) glass bottles using a syringe. These air samples were analyzed for their CO_2, CH_4 and N_2O concentrations using a Shimadzu GC-2014 gas chromatograph combined with a Loftfield automatic sample injection system (Loftfield *et al.*, 1997).

In 2014, the net ecosystem CO_2 exchange (NEE) was measured using a transparent Plexiglas chamber (95% transparency; h 50 cm, V 65 L) combined with a portable infra-red gas analyzer (EGM-4, PP Systems, Hitchin, UK). The chamber was equipped with a sensor to measure PAR and Ta (TRP-2, PP

Systems, Hitchin, UK) inside the chamber. After every NEE measurement, RE was determined from a subsequent measurement during which the transparent chamber was covered with an opaque and light reflective shroud. Gross primary production (GPP) was derived from the difference between NEE and RE (i.e. GPP = NEE – RE). In addition, an estimate of net primary production (NPP) was derived from the difference between NEE and heterotrophic respiration (Rh; see below) (i.e. NPP = NEE – Rh). In the vegetation-free BP treatment, GPP as well as NPP were assumed to be zero and NEE subsequently equaled RE.

In 2014, heterotrophic respiration (Rh) was estimated at the Lavassaare and Tässi sites on trenched plots where all living vegetation was removed (Publications II and III). For this purpose, separate PVC collars (Ø 17.5 cm) and a second set of instrumentation including an opaque chamber (h 30 cm, V 0.065 L) combined with an EGM-4 infra-red gas analyzer was used. Autotrophic respiration (Ra) was derived for the vegetated treatments from the difference between the measured RE and Rh fluxes (i.e. Ra = RE – Rh).

Fluxes of CO_2, CH_4 and N_2O were calculated from the linear change in gas concentrations in the chamber headspace over time corrected for changes in air density using the ideal gas law (Eq. 1):

$$F = S \times \frac{p \times V \times M \times t}{R \times Ta \times A} \qquad \text{(Eq. 1)}$$

where F is the measured flux (i.e. CO_2 in mg CO_2-C m^{-2} h^{-1}, CH_4 in µg CH_4-C m^{-2} h^{-1} or N_2O in µg N_2O-N m^{-2} h^{-1}), S is the linear slope fitted to the concentration change over time (CO_2 in ppm, N_2O and CH_4 in ppb), p is the air pressure, V is the chamber headspace volume, M is the molar mass of the gas (44.01 g mol^{-1} for CO_2, 16.04 g mol^{-1} for CH_4 and 44.01 g mol^{-1} for N_2O), t is the chamber deployment time, R is the universal gas constant of 8.3143 (J mol^{-1} K^{-1}), Ta is the mean headspace air temperature during the measurement and A is the flux collar area. To ensure high quality flux data, CO_2, CH_4 and N_2O fluxes were accepted only if the R^2 values of the linear fits were > 0.90, 0.80 and 0.80, respectively. The atmospheric sign convention was used in which positive (e.g. RE) and negative (e.g. GPP and NPP) fluxes represent emissions to and uptake from the atmosphere, respectively.

2.5. Annual carbon and greenhouse gas balances

Annual sums of CO_2 (Publication I), CH_4 and N_2O (Publications I–III) were derived from scaling their mean (or median) fluxes to the annual scale. In 2014, empirical non-linear regression models were built based on environmental variables and the measured CO_2 fluxes to model hourly CO_2 fluxes and to obtain cumulative sums over the growing season and annual time scales (Publications II and III).

Models for predicting gross primary production included a hyperbolic light response term combined with either a term accounting for vegetation effects (Eq. 2) (Publication II) following Kandel *et al.* (2013a) or water table level effects (Eq. 3) (Publication III) following Tuittila *et al.* (2004):

$$GPP = \frac{\alpha \times A_{max} \times PAR \times gcc_{norm}}{\alpha \times PAR + A_{max} \times gcc_{norm}} \quad \text{(Eq. 2)}$$

$$GPP = \frac{\alpha \times A_{max} \times PAR}{\alpha \times PAR + A_{max}} \times exp\left[-0.5 \times \left(\frac{WTL - WTL_{opt}}{WTL_{tol}}\right)^2\right] \quad \text{(Eq. 3)}$$

where *GPP* is gross primary production (mg 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 C µmol $photon^{-1}$), A_{max} is maximum photosynthesis at light saturation (mg C m^{-2} h^{-1}), gcc_{norm} is the collar-specific chromatic greenness index normalized to scale between 0 and 1, *WTL* is the water table level (cm), WTL_{opt} is the WTL at which maximum photosynthetic activity occurs and WTL_{tol} is the tolerance, i.e. the width of the Gaussian response curve of GPP to WTL.

In the RCG treatments, ecosystem respiration was modeled using an exponential relationship to air temperature accounting for the additional effects from changes in vegetation biomass (Eq. 4) (Publication II) following Kandel *et al.* (2013a), while in the BP and restored treatments, the model was based on an exponential relationship to air temperature only (Eq. 5) (Publications II and III) following Lloyd & Taylor (1994):

$$RE = R_0 \times exp^{(b \times Ta)} + (\beta \times gcc_{norm}) \times exp^{(b \times Ta)} \quad \text{(Eq. 4)}$$

$$RE = R_0 \times exp^{(b \times Ta)} \quad \text{(Eq. 5)}$$

where *RE* is ecosystem respiration (mg C m^{-2} h^{-1}), *Ta* is air temperature (°C), R_0 is the soil respiration (mg C m^{-2} h^{-1}) at 0 °C, *b* is the sensitivity of respiration to Ta 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 Ta, 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 the annual GPP and RE estimates resulted in the annual NEE in RCG cultivation and peatland restoration treatments (Eq. 6):

$$NEE = GPP + RE \quad \text{(Eq. 6)}$$

The annual GHG balances were estimated by converting the cumulative fluxes to CO_2 equivalents (CO_2 eq) using their respective global warming potentials (GWP, over a 100-year timeframe). The GWP of 298 was used for N_2O in Publications I–III (IPCC, 2007, 2013). For CH_4, the GPW of 25 (IPCC, 2007) was used in Publication I while the GWP of 34 (IPCC, 2013) was used in Publications II and III.

2.6. Upscaling and predicting spatial N_2O emission patterns to European organic soils

In Publication IV, an empirical fuzzy logic modeling approach was used to predict N_2O fluxes based on non-linear responses with the main driving parameters across various European organic soils. The distribution of organic soils within Europe was based on previous work by Montanarella *et al*. (2006). Separate models were developed for different land-use types including forest, cropland, grassland, natural peatland and peat extraction based on a total of 659 annual N_2O measurements. Using meteorological variables, mean WTL and soil parameters as input variables, these models were applied to land-use cover maps (CORINE land cover, CLC; Historic Land Dynamics Assessment, HILDA) to upscale N_2O emissions from organic soils to the European level for each land-use type. The Europe-wide annual N_2O emissions were then estimated as the sum of the emissions from cropland, grassland, forest, peat extraction and natural sites on organic soils. Fluxes above the 90th quantile of the flux distribution within each land use category were defined as hotspot emissions for the particular land use.

2.7. Statistical analysis

Normal distribution of the data was evaluated using the Kolmogorov–Smirnov, Lilliefors, and Shapiro–Wilk tests. Statistical differences ($P < 0.05$; unless stated otherwise) among three treatment means of the various GHG fluxes, environmental variables and biomass pools were determined with the non-parametric Kruskal–Wallis ANOVA or Friedman one-way analysis of variance (ANOVA) combined with a Bonferroni post-hoc comparison. Differences between two treatment means were assessed using the Wilcoxon's matched-pairs test. Spearman's rank order or Pearson's correlations were used to investigate the correlations of abiotic and biotic controls with the GHG fluxes. All calculations and statistics were computed using the softwares STATISTICA 7.1 (StatSoft Inc., USA) and Matlab (Matlab Student version, 2013a, Mathworks, USA).

3. RESULTS AND DISCUSSION

3.1. Carbon and greenhouse gas fluxes from abandoned peat extraction areas: impact of reed canary grass cultivation (Publications I and II)

3.1.1. Climatic conditions

The annual mean air temperature and total precipitation from May 2010 to April 2011 (Publication I) were 5.6 and 911 mm, respectively. Given the 30-year long-term climate normals of 6.3 °C and 745 mm for the region, the study period represented a cooler year with above-normal precipitation. The water table levels during the growing season remained on average at approximately 30 cm below the peat surface.

In contrast, the annual mean air temperature and total precipitation from January to December 2014 (Publication II) were 6.9 °C and 525 mm, respectively, which indicates relatively warmer and considerably drier conditions. Moreover, the growing season included two warm and dry periods (mid-May to mid-June and early July to early August) during which the soil moisture and water table levels were greatly reduced in all treatments.

3.1.2. Biomass production

The aboveground biomass production based on destructive harvesting in September 2010 was estimated at 1390 and 796 g m^{-2} in the fertilized and nonfertilized RCG treatments (Publication I). Meanwhile, the belowground biomass pools were 935 and 724 g m^{-2}. In the warm and dry year 2014 aboveground biomass production in fertilized and nonfertilized RCG cultivations were estimated at 234 and 42 g m^{-2}, respectively, while the belowground biomass pools were 646 and 416 g m^{-2} (Publication II).

The between-year comparison suggests that both above- and belowground biomass pools were considerably reduced during the dry year 2014 compared to the wet year 2010. This highlights the sensitivity of biomass production to climatic conditions in RCG cultivations. However, in both studied years above- and belowground biomass pools were significantly greater in the fertilized compared to the nonfertilized RCG cultivations. Thus, fertilization greatly increased biomass production in the wet as well as in the dry year.

Overall, the yields in 2010 were at the top end whereas yields in 2014 were at the bottom end of the range of 200 to 1400 g m^{-2} and 100 to 1100 g m^{-2} previously reported for fertilized and nonfertilized RCG cultivations, respectively (Shurpali *et al.*, 2010; Heinsoo *et al.*, 2011; Kandel *et al.*, 2013a; Karki *et al.*, 2014). The low yields in 2014 were likely due to water stress constraining plant growth during an exceptionally dry summer, which highlights the importance of climatic conditions for the biomass production in RCG cultivations. These 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.

Fertilizer effects on plant growth and soil nutrient status might not only affect the total biomass production but also its allocation into above- and belowground components (Xiong & Kätterer, 2010). For instance, greater above- to belowground biomass ratio in the fertilized compared to the nonfertilized RCG cultivations observed in both years (Publications I and II) 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 belowground NPP, increased C input to the soil may still occur in fertilized compared to nonfertilized RCG cultivations.

3.1.3. Carbon and greenhouse gas fluxes: seasonal dynamics and controls

Carbon dioxide

The annual mean midday NEE in the fertilized RCG was significantly lower (i.e. indicating greater net CO_2 uptake) ($P < 0.01$) than in the nonfertilized RCG and BP treatments in 2014 (Publication II) (Figure 2) suggesting that a significant reduction in the CO_2 emissions from the abandoned peat extraction area was only achieved in the fertilized RCG treatment during the warm and dry year. The seasonal dynamics of NEE showed a negative NEE (i.e. CO_2 uptake) of up to −162 mg C m^{-2} h^{-1} between May and September in the fertilized RCG site, whereas NEE in the nonfertilized RCG site remained close to zero during the early growing season (May and June) and switched to CO_2 emissions during the late growing season (July and August) (Figure 4a in Publication II). Small midday net CO_2 uptake, however, also occurred in the nonfertilized RCG treatment after intermittent rainfall at the end of June which indicates that also nonfertilized RCG cultivations might sequester CO_2 given sufficient water supply. This is supported by a Danish study reporting that both fertilized and nonfertilized RCG cultivations with VWC > 55% provided midday net CO_2 uptake for the entire growing season (Kandel *et al.*, 2013a). Furthermore, the climatic effect on NEE was also found in a RCG cultivation in eastern Finland where daily net CO_2 uptake rates decreased by about half in dry compared to wet years (Shurpali *et al.*, 2009). The combined findings from this and previous studies strongly indicate that soil water availability is a major control of the CO_2 sink-source strength of RCG cultivations on drained peat soils.

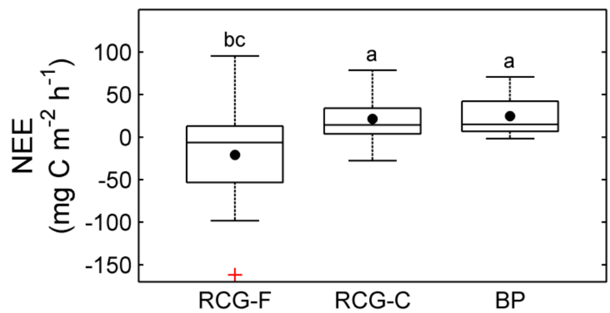

Figure 2. Box plot for net ecosystem CO_2 exchange (NEE) in 2014 for three different sites: fertilized RCG (RCG-F), nonfertilized RCG (RCG-C) and bare peat (BP). The horizontal lines and dots inside the boxes are the medians and means, respectively, the edges of the box are the 25th and 75th percentiles, 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.

The cultivation of the abandoned peat extraction area with RCG also significantly affected RE. In both years (2010 and 2014), the mean midday RE was lowest in BP (10 and 25 mg C m^{-2} h^{-1}) and highest in the fertilized RCG treatment (97 and 89 mg C m^{-2} h^{-1}). Meanwhile RE in the nonfertilized RCG (68 and 60 mg C m^{-2} h^{-1}) was significantly higher than in BP and lower than in the fertilized RCG (Figure 3a,b). The observed differences in RE between the cultivated RCG treatments and BP were likely due to the additional respiration losses from vegetation, root turnover and labile C substrate input in the RCG treatments. Similarly, greater vegetation biomass and growth might explain the higher RE in the fertilized compared to the nonfertilized RCG cultivation. In both years, the seasonal patterns of midday RE followed closely that of air temperature (Figure 5a in Publication I, Figure 4b in Publication II). Overall, RE rates in these three treatments were comparable to CO_2 emissions reported from cultivated and abandoned peat extraction areas in Sweden and Finland (Sundh *et al.*, 2000; Shurpali *et al.*, 2008, 2009; Maljanen *et al.*, 2010).

In comparison to the RCG and BP treatments, the midday RE in the natural bog was on average 37 mg C m^{-2} h^{-1} (Figure 3a) and ranged between 10 and 100 mg C m^{-2} h^{-1} (Figure 5a in Publication I). Thus, RE in NB was similar to BP and the nonfertilized RCG cultivation but significantly lower than in the fertilized RCG treatment. In contrast, considerably higher mean RE of 209 mg C m^{-2} h^{-1} (Figure 3a) and peak rates of > 500 mg C m^{-2} h^{-1} (Figure 5a in Publication I) were found in FM. The high RE in FM was likely caused by the combination of enhanced mineralization due to highly aerobic conditions (intensive drainage was established in the 1950's) as well as large autotrophic respiration from vigorously growing herbaceous vegetation. Overall, the contrasting results for RE observed among the various treatments highlight the potential large impact of human use and management on the peatland C cycle.

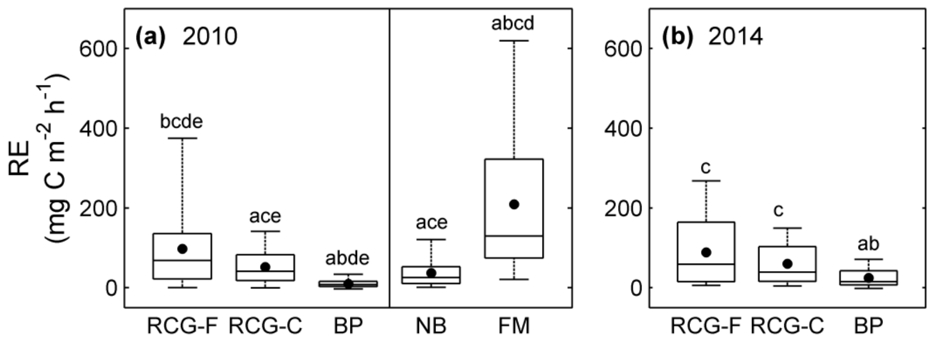

Figure 3. Box plots for ecosystem respiration (RE) in 2010 (panel a) and 2014 (panel b) for five different sites: fertilized RCG (RCG-F), nonfertilized RCG (RCG-C), bare peat (BP), natural bog (NB) and fen meadow (FM); see Figure 2 for a description of the box plot features.

In all studied sites, the main control of the seasonal RE dynamics was the soil temperature (Spearman Rank Correlation, $\rho = 0.74$–0.99). In addition, RE was also correlated with WTL in NB (Spearman Rank Correlation, $\rho = 0.79$) and the nonfertilized RCG treatment (Spearman Rank Correlation, $\rho = 0.95$). Soil temperature and water availability have been previously reported to control RE by affecting plant growth and associated autotrophic respiration as well as by influencing the rates of microbial decomposition (Bubier *et al.*, 2003; Alm *et al.*, 2007; Kløve *et al.*, 2010). However, effects from soil temperature and WTL might also counterbalance each other or be masked by other factors which affect RE (e.g. soil pH and nutrient availability). For instance, overriding effects from other environmental variables might explain the absence of the WTL control in the fertilized RCG and BP sites.

The mean midday GPP and NPP in 2014 (Publication II) were significantly lower (i.e. suggesting greater production) ($P < 0.01$) in the fertilized (−185 and −105 mg C m^{-2} h^{-1}) than in the nonfertilized RCG cultivation (−62 and −28 mg C m^{-2} h^{-1}) (Figure 4a,b), demonstrating the large impact of fertilization on plant production in RCG cultivations. The results further suggest that the greater midday net CO_2 uptake in the fertilized cultivation was due to variations in GPP since the increase in GPP (by 69%) was larger than the increase in RE (by 37%) relative to the nonfertilized RCG. Similarly, GPP was also reported as main driver for inter-annual variations in NEE during wet and dry years in a Finnish RCG cultivation (Shurpali *et al.*, 2009). Thus, ensuring optimum growing conditions through adequate water supply is essential not only for achieving economically sustainable yields but also for maximizing the CO_2 sequestration potential in RCG cultivations established on drained organic soils.

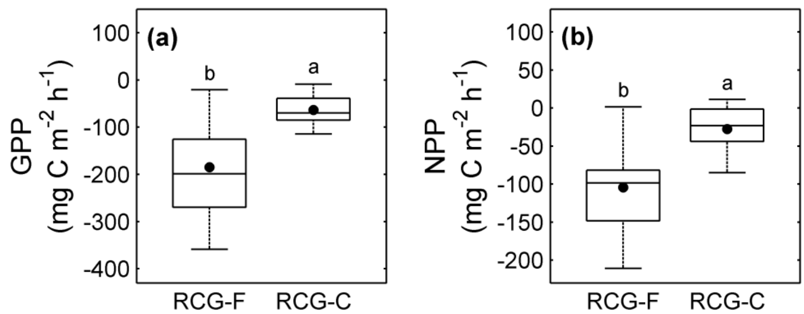

Figure 4. Box plots for growing season gross primary production (GPP; panel a) and net primary production (NPP; panel b) in 2014 for two different sites: fertilized RCG (RCG-F) and nonfertilized RCG (RCG-C); see Figure 2 for a description of the box plot features.

Due to its positive effect on plant growth, fertilization also resulted in significantly higher mean growing season Ra in the fertilized RCG compared to the nonfertilized RCG cultivation (Publication II) (Figure 5a). In contrast, no significant difference was observed in Rh between the fertilized and nonfertilized RCG cultivations (Figure 5b), indicating that fertilization had no significant effect on microbial respiration. The effect of fertilization on mineralization is controversial with several previous studies reporting no effect or a decrease in mineralization following fertilization (e.g. Fog, 1988; Aerts & Toet, 1997). Aerts & Toet (1997) suggested that the decrease in mineralization observed in some cases is primarily related to the indirect effects of fertilization on soil pH. Thus, the difference in the RE partitioning into its components Rh and Ra between the fertilized (Rh < Ra) and nonfertilized RCG (Rh > Ra) cultivations was the result of enhanced plant growth due to fertilization and the subsequent increase of Ra in the fertilized treatment.

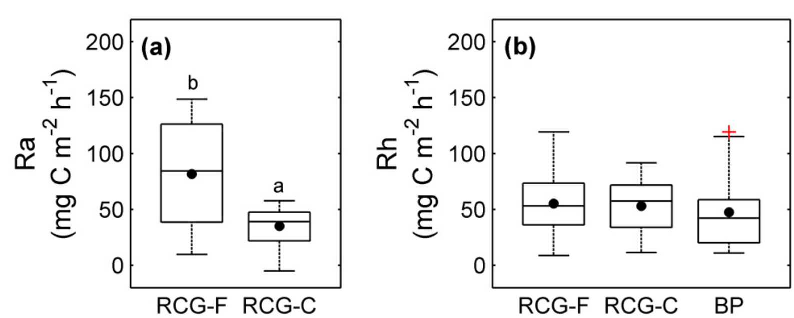

Figure 5. Box plots for growing season autotrophic (Ra; panel a) and heterotrophic respiration (Rh; panel b) in 2014 for three different sites: fertilized RCG (RCG-F), nonfertilized RCG (RCG-C) and bare peat (BP); see Figure 2 for a description of the box plot features.

Meanwhile, the comparison of Rh between RCG and BP treatments suggests no significant impact of cultivation when Rh is averaged over the entire growing season (Figure 5b). Nevertheless, during the warmest summer period (July to August), Rh was consistently higher ($P < 0.01$) in RCG-F and RCG-C than in BP (Figure 6a in Publication II). This underlines the risk of increased mineralization of organic matter in drained peat soils following cultivation and its negative implications for the C and GHG balances previously highlighted in several studies (Maljanen *et al.*, 2010; Schrier-Uijl *et al.*, 2014). Thus, a substantial additional C input from plant CO_2 uptake would be required to outbalance these CO_2 losses from enhanced mineralization following cultivation of organic soils.

The results also suggest that fertilization caused a decrease in the mean contribution of Rh to RE in the RCG treatments since, averaged over all sampling dates, Rh accounted for only 42% of RE in the fertilized RCG treatment but 62% in the nonfertilized RCG treatment. The contribution of Rh to RE in the fertilized RCG site 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 (Riutta *et al.*, 2007a; Biasi *et al.*, 2012).

Methane

CH_4 emission in the growing season 2014 (Figure 7a in Publication II) occurred in the range of 0.01 to 9.3 μg C m^{-2} h^{-1} in both RCG and the bare peat treatments which is comparable to the values observed in 2010 (Figure 5b in Publication I). Between mid-June and early September 2014, the mean CH_4 emission from RCG treatments was approximately 1.5 times higher than in BP ($P = 0.052$). The annual mean CH_4 exchanges, however, showed no significant differences among the three treatments in neither of the years (Figure 6a,b). Overall, annual CH_4 emissions of < 0.02 g C m^{-2} yr^{-1} from the RCG treatments 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 3.1 g C m^{-2} yr^{-1} observed in cultivated cutaway peatlands (Hyvönen *et al.*, 2009; Karki *et al.*, 2015). These low CH_4 emissions were likely the result of the lowered 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 a cutaway peatland in which the WTL remained mostly close to the surface (i.e. within 10 cm) (Wilson *et al.*, 2009). In addition, generally lower CH_4 emission from the Lavassaare sites may also be due to the high sulfur concentrations in peat which may inhibit methanogenesis due to the increased competition for acetate and hydrogen from sulfate reducing bacteria (Deppe *et al.*, 2010). Thus, besides the depth of the WTL, peat chemistry may act as an additional important control of CH_4 emissions from cultivated and abandoned peat extraction areas.

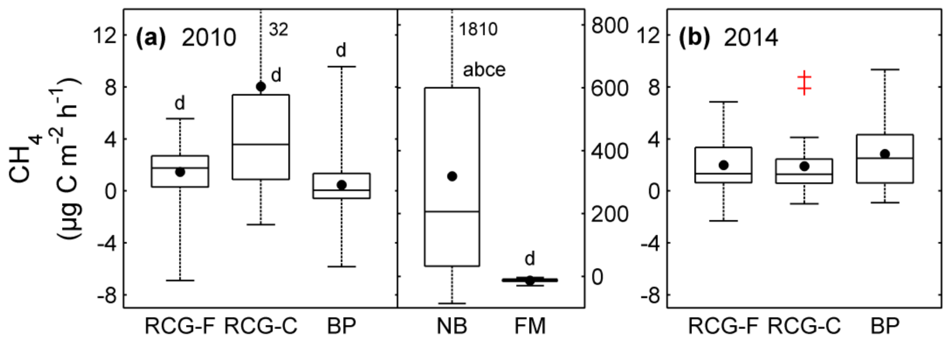

Figure 6. Box plots for methane (CH_4) fluxes in 2010 (panel a) and 2014 (panel b) for five different sites: fertilized RCG (RCG-F), nonfertilized RCG (RCG-C), bare peat (BP), natural bog (NB) and fen meadow (FM). The secondary y-axis in panel (a) applies to the NB and FM treatments; see Figure 2 for a description of the box plot features.

The main control of CH_4 fluxes in the fertilized RCG, nonfertilized RCG and BP treatments in 2010 was the depth of the WTL (Spearman Rank Correlation, $\rho = -0.77$, $\rho = -0.87$ and $\rho = -0.87$, respectively). In 2014, the negative correlation between CH_4 fluxes and WTL could not be observed due to the exceptionally dry conditions during the study year which resulted in very low CH_4 fluxes and WTLs below the depth of the peat layer. In the NB, the CH_4 flux was correlated with Ts (Spearman Rank Correlation, $\rho = 0.86$) suggesting that temperature controls on vegetation growth and associated CH_4 transport through aerenchymatic stems may have played a greater role in determining the flux dynamics than variations in the WTL.

Nitrous oxide

No regular seasonal patterns in N_2O emissions were found in any of the studied treatments in 2010 (Figure 5c in Publication I). Overall, median fluxes of −0.07, −0.29 and 0.97 µg N m^{-2} h^{-1} were observed in the fertilized RCG, nonfertilized RCG and BP treatments, respectively, suggesting that the cultivation of the abandoned peat extraction area with RCG resulted in decreased N_2O fluxes (Figure 7a). The negative N_2O flux observed in the RCG sites could be due to microbial N_2O uptake in soil microsites where N_2O can be rapidly transformed to dinitrogen, although the phenomenon of negative N_2O flux has not yet been clarified (Chapuis-Lardy *et al.*, 2007). Among all studied treatments, the highest median N_2O flux was observed in the FM (9.6 µg N m^{-2} h^{-1}) which was likely due to the combination of a low C:N ratio (14.5) as a result of intensified peat mineralization (Klemedtsson *et al.*, 2005) and a low WTL (Martikainen *et al.*, 1993).

The main controls of N_2O fluxes commonly include soil moisture, temperature as well as contents of nitrate and organic carbon (Tiedje *et al.*, 1983; Dobbie & Smith, 2003). Out of these factors, Ts explained best the variations in N_2O fluxes in the fertilized RCG (Spearman Rank Correlation, ρ = 0.80), nonfertilized RCG (Spearman Rank Correlation, ρ = 0.71) and the FM (Spearman Rank Correlation, ρ = 0.70) sites. Thus, temperature constraint on microbial activity and substrate supply were likely the limiting factors for N_2O production in these sites.

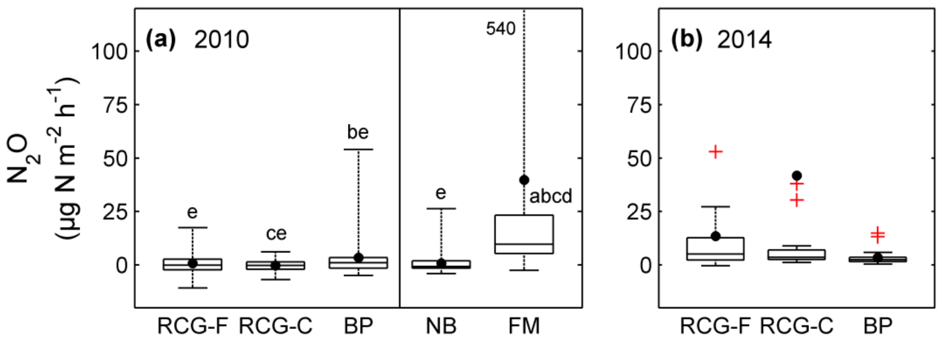

Figure 7. Box plots for nitrous oxide (N_2O) fluxes in 2010 (panel a) and 2014 (panel b) for five different sites: fertilized RCG (RCG-F), nonfertilized RCG (RCG-C), bare peat (BP), natural bog (NB) and fen meadow (FM); see Figure 2 for a description of the box plot features; note that in 2014 outliers 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 2014 (Publication II), N_2O fluxes were within the range of −0.4 to 25 µg N m^{-2} h^{-1} for most of the year, with the exception of large emission peaks of up to 420 µg N m^{-2} h^{-1} in the RCG cultivations (Figure 7b in Publication II). These peak emissions coincided with large rainfall events occurring just prior to sampling dates. This indicates that annual N_2O emission might be greater during wetter years with more frequent rainfall events (e.g. Dobbie & Smith, 2003). In contrast to the small uptake observed in 2010, annual N_2O emissions of 0.03 to 0.07 g N_2O m^{-2} yr^{-1} across all treatments were observed in 2014. However, these annual N_2O emissions are still low compared to the range of 0.2 to 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). Meanwhile, the magnitudes of N_2O emissions in 2014 are comparable with the 0.1 and 0.01 g N_2O m^{-2} yr^{-1} reported for Finnish RCG and BP sites (Hyvönen *et al.*, 2009). It is further noteworthy that the annual median N_2O exchanges were not significantly different among the RCG and BP treatments (Figure 7b), which suggests that N fertilizer application did not result in considerably increased N_2O emissions. The N_2O emission factor (i.e. the % of N fertilizer lost as N_2O) was estimated at 0.63% in the fertilized RCG cultivation which is considerably lower compared to the default emission factor of 1% suggested in the IPCC methodology (De Klein *et al.*, 2006) for the accounting of GHG emissions due to N fertilizer application in agriculture.

Dissolved organic carbon

The mean DOC concentrations were 17, 16 and 16 mg L^{-1} resulting in an annual DOC export of 4.2, 4.2 and 4.1 g C m^{-2} yr^{-1} in the fertilized RCG, nonfertilized RCG and BP treatments, respectively (Publication II). The annual DOC export 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 to 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 a relatively small contribution to the full C balance (~2–4%) during the dry year of 2014, the DOC export might increase under management and climate scenarios that alter soil hydrology and runoff (Freeman *et al.*, 2004).

3.1.4. Annual carbon and greenhouse gas balances

Carbon balance

Combining the annual CO_2 and CH_4 exchanges (and the DOC export in 2014) suggests that RCG cultivations on abandoned peat extraction areas may act as substantial C sinks as well as C sources in different years. A net C uptake of −163 and −91 g C m^{-2} yr^{-1} was observed in the fertilized and nonfertilized RCG treatments in 2010 (Publication I), whereas a net C loss of 96 and 215 g C m^{-2} yr^{-1} occurred in the same treatments, respectively in 2014 (Publication II) (Table 4). Similarly, the net C loss of 68 g C m^{-2} yr^{-1} in 2010 was smaller than the 180 g C m^{-2} yr^{-1} in 2014 at the BP treatment. The main reason for these contrasting results on the C sink-source strength is likely the difference in climatic conditions during the study years. Specifically, the results suggest a switch in the RCG treatments from a CO_2 sink in the wet year of 2010 with above-normal precipitation (911 mm) to a CO_2 source during the dry year (525 mm) of 2014. 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.

Table 4. Annual carbon (C) (g C m^{-2} yr^{-1}) and greenhouse gas (GHG) (t CO_2 eq ha^{-1} yr^{-1}) balances in 2010 (Publication I) and 2014 (Publication II). The total C balance in 2010 is the sum of the annual CO_2 and CH_4 fluxes, while in 2014 the C balance includes CO_2, CH_4 and DOC fluxes. The total GHG balance is the sum of annual CO_2, CH_4 and N_2O fluxes. Negative and positive balances represent net uptake and emission, respectively. RCG-F, fertilized reed canary grass cultivation; RCG-C, nonfertilized reed canary grass cultivation; BP, bare peat.

	RCG-F		RCG-C		BP	
	2010	2014	2010	2014	2010	2014
Total C balance	−163	96	−91	215	68	180
Total GHG balance	−6.0	3.6	−3.9	7.9	2.5	6.6

Greenhouse gas balance

The sum of the CO_2, CH_4 and N_2O exchanges suggested negative GHG balances of −6.0 and −3.9 t CO_2 eq ha^{-1} yr^{-1} for the fertilized RCG and nonfertilized RCG treatments, and a positive GHG balance of 2.5 t CO_2 eq ha^{-1} yr^{-1} for the BP treatment in 2010 (Publication I) (Table 4). In comparison, the fertilized RCG and nonfertilized RCG treatments had positive GHG balances of 3.6 and 7.9 t CO_2 eq ha^{-1} yr^{-1} in 2014, while the GHG balance of BP increased to 6.6 t CO_2 eq ha^{-1} yr^{-1} (Publication II). The GHG balances of the RCG cultivations were determined by the net CO_2 exchange whereas the combined contribution of CH_4 and N_2O emission to the GHG balance was small (< 6%) in both years. Similarly, other studies found a relatively small contribution of CH_4 and N_2O to the GHG balance of cultivated organic soils (Hyvönen *et al.*, 2009; Shurpali *et al.*, 2010; Karki *et al.*, 2015). Management practices need to be therefore carefully evaluated with respect to their direct and indirect impacts on the ecosystem CO_2 exchange.

Furthermore, the lower GHG balance in the fertilized relative to the nonfertilized RCG treatment in both years suggests that the increase in biomass production and net CO_2 uptake largely exceeded the increase in N_2O emissions following moderate fertilization. Moreover, the GHG balance of the fertilized RCG treatment was significantly reduced relative to that of BP in both years, whereas the nonfertilized RCG treatment mitigated the GHG balance of abandoned BP site only during the wet year 2010. Thus, moderate fertilization could be a beneficial management practice to sustain and maximize yields and climate benefits in RCG cultivations 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 must be considered when evaluating optimum fertilizer rates (Smith & Slater, 2010; Verhoeven & Setter, 2010; Don *et al.*, 2012).

It has been previously reported that RCG cultivation may not only mitigate GHG emissions from drained organic soils but even provide a net GHG sink (e.g. Shurpali *et al.*, 2010). Similarly, negative GHG balances (i.e. net GHG sink) were also observed in the wet year 2010 for both fertilized and non-fertilized RCG cultivations at the Lavassaare site. In contrast, however, both RCG treatments had positive GHG balances in the dry year 2014. This highlights the impact of climatic conditions on the GHG sink-source strength of RCG cultivations on organic soils. However, previous studies indicated that a negative GHG balance could be maintained by cultivating RCG in agricultural systems with elevated WTLs and sufficient soil water availability (Karki *et al.*, 2014; Schrier-Uijl *et al.*, 2014). Although rewetting of drained organic soils might increase CH_4 emissions, these increases have been estimated to be modest (Tuittila *et al.*, 2000; Wilson *et al.*, 2009), and are therefore unlikely to compromise the benefits gained from enhanced plant growth and CO_2 uptake due to sufficient water supply. Thus, management strategies for RCG cultivation need to ensure optimum plant growth through raised WTLs and nutrient supply to maximize the net ecosystem CO_2 uptake since its benefits are likely to exceed the associated potentially negative effects from increased CH_4 and N_2O emissions.

3.2. Carbon and greenhouse gas fluxes from abandoned peat extraction areas: impact of peatland restoration (Publication III)

3.2.1. Climatic conditions

The annual mean air temperature and total precipitation from March 2014 to February 2015 were 7.2 °C and 784 mm, respectively, which suggests warmer conditions with normal wetness when compared to the long-term climate normal (5.8 °C and 764 mm). The mean WTL in the high WTL restoration treatment (Res-H) and low WTL restoration treatment (Res-L) were −24 and −31 cm, respectively, resulting in a mean annual difference of 7 cm. The mean WTL in the bare peat (BP) treatment was −46 cm. Thus, restoration activities including ditch blocking and lowering of the peat surface were effective in raising the WTL by about 15–20 cm relative to the BP treatment.

3.2.2. Vegetation cover

As a result of the WTL difference, contrasting vegetation communities developed in the two restored treatments within three years following restoration. Specifically, a greater bryophyte cover of 63% (primarily *Sphagnum* spp.) was found in the wetter Res-H treatment. In contrast, the lower WTL in Res-L resulted in a lower bryophyte cover of 44% but greater abundancy of vascular plants. These differences in vegetation composition can be explained by the functional characteristics of these two plant groups. For instance, bryophytes rely on capillary forces for acquiring water and thus require wetter conditions (Rydin, 1985) as provided in Res-H. In contrast, the extended zone of aeration due to the lower WTL was likely more favorable for vascular plant roots in Res-L. Apart from having roots to absorb water and nutrients from the soil, vascular plants also differ from bryophytes by having leaf stomata to regulate water transport and CO_2 exchange (Turner *et al.*, 1985; Schulze *et al.*, 1994). Thus, these differences in the vegetation communities due to contrasting WTL baselines may have important implications for the biogeochemical cycles and GHG fluxes in restored peat extraction areas (Weltzin *et al.*, 2000).

3.2.3. Carbon and greenhouse gas fluxes: seasonal dynamics and controls

Carbon dioxide

The differences in WTL and vegetation composition showed a strong impact on plant production in the two restored treatments. Specifically, variations in GPP and NPP among individual flux measurement collars (i.e. indicating spatial variability) were significantly correlated to bryophyte but not to vascular plant cover in Res-H, whereas significant correlations to vascular plant but not to bryophyte cover were observed in Res-L (Table 5). Moreover, both midday GPP and NPP were lower (i.e. representing greater production) in Res-L than in Res-H throughout most of the growing season (Figure 2c,d in Publication III). Overall, the growing season mean GPP of −65.5 mg C m^{-2} h^{-1} in Res-L was significantly lower than that of −49.3 mg C m^{-2} h^{-1} in Res-H (Table 6). The higher GPP in Res-L was likely due to the greater vascular plant cover compared to Res-H, since vascular plants reach higher photosynthesis rates at higher light levels compared to mosses (Bubier *et al.*, 2003; Riutta *et al.*, 2007a). Similarly, Strack & Zuback (2013) reported a strong correlation between vascular plant cover and GPP in a restored peatland in Canada. In return, the greater GPP also explains the higher Ra observed in Res-L compared to Res-H. Thus, these results highlight the implications of hydrological differences and the associated vegetation development on plant-related CO_2 fluxes.

Table 5. Correlation coefficients of vegetation (bryophytes and vascular plants) cover (%) with gross primary production (GPP) and net primary production (NPP) in restoration treatments with high (Res-H) and low (Res-L) water table level. Total vegetation represents the sum of bryophyte and vascular plant cover; significant correlations are marked with asterisks (* indicates $P < 0.05$ and ** indicates $P < 0.01$).

	Res-H		Res-L	
Vegetation cover	GPP	NPP	GPP	NPP
Bryophytes	-0.95^{**}	-0.84^{*}	-0.81^{*}	−0.70
Vascular plants	−0.76	−0.68	-0.97^{**}	-0.93^{**}
Total vegetation	-0.95^{**}	-0.84^{*}	-0.84^{*}	−0.75

Midday RE in Res-H and Res-L reached maximum values of 74 and 96 mg C m^{-2} h^{-1} during early July, respectively, while peak RE of 104 mg C m^{-2} h^{-1} occurred in early August in BP (Figure 2b in Publication III). The annual mean midday RE was significantly lower in Res-H and Res-L than in BP (Table 6). Soil temperature at 10 cm depth was the abiotic variable that best explained variations in RE (R^2 = 0.79, 0.84 and 0.81 in Res-H, Res-L and BP, respectively). The lower RE in the restored treatments relative to BP was the result of a considerable reduction in Rh which showed maximum rates of up to 61, 73 and 104 mg C m^{-2} h^{-1} in Res-H, Res-L and BP, respectively (Figure 2e in Publication III). This suggests that the raised WTL following restoration effectively reduced the potential for aerobic peat decomposition commonly occurring in drained peatlands (Silvola *et al.*, 1996; Frolking *et al.*, 2001; Whiting & Chanton, 2001). In comparison, Ra in the restored treatments reached maximum values of up to 27 and 36 mg C m^{-2} h^{-1} in Res-H and Res-L, respectively, and was on average significantly higher (by about two times) in Res-L than in Res-H (Figure 2f in Publication III). Overall, the additional Ra from the growing vegetation was negligible compared to the large reduction in Rh in the restored treatments relative to BP (Table 6). Strack & Zuback (2013) also found significantly lower Rh and RE in the restored compared to an unrestored site in Canada 10 years following peatland restoration. Thus, these results demonstrate the effectiveness of raising the WTL in reducing peat decomposition and CO_2 emissions from drained organic soils.

Table 6. Means of measured CO_2 fluxes (mg C m^{-2} h^{-1}) including net ecosystem exchange (NEE), ecosystem respiration (RE), gross primary production (GPP), net primary production (NPP), autotrophic respiration (Ra) and heterotrophic respiration (Rh) as well as means of measured methane (CH_4; µg C m^{-2} h^{-1}) and nitrous oxide (N_2O; µg N m^{-2} h^{-1}) fluxes in restoration treatments with high (Res-H) and low (Res-L) water table level and bare peat (BP). Negative and positive fluxes represent uptake and emission, respectively. Numbers in parenthesis indicate standard error; different letters indicate significant (P < 0.05) differences among treatments. (Publication III)

Component flux	Res-H	Res-L	BP
NEE	0.57 (4.9)^{c}	−2.82 (4.9)^{c}	44.9 (8.2)^{ab}
RE	29.9 (5.1)^{c}	35.1 (6.4)^{c}	44.9 (8.2)^{ab}
GPP*	−49.3 (7.4)^{a}	−65.5 (7.3)^{b}	n.a.
NPP*	−41.5 (5.3)	−48.1 (4.2)	n.a.
Ra*	7.9 (2.6)^{a}	16.2 (3.4)^{b}	n.a.
Rh*	37.0 (5.1)^{c}	38.5 (5.9)^{c}	71.2 (8.4)^{ab}
CH_4	23.0 (10.7)	10.9 (6.1)	14.7 (3.7)
N_2O	−0.12 (0.25)^{c}	2.13 (1.29)^{c}	27.1 (9.1)^{ab}

n.a., not applicable
* Growing season mean (May 1 to October 31)

Nevertheless, despite the significant effects of the re-established WTL baseline on vegetation development and the associated CO_2 component fluxes (i.e. RE and GPP), the mean midday net CO_2 exchange of the two restored treatments was not significantly different (Table 6). During the early (i.e. June) and late (i.e. mid-August to September) summer, net CO_2 uptake of up to −42 and −41 mg C m^{-2} h^{-1} occurred in both Res-H and Res-L, respectively, whereas net CO_2 emissions of up to 36 and 27 mg C m^{-2} h^{-1} were observed during the warm and dry month of July in the same treatments (Figure 2a in Publication III). In contrast to Res-H and Res-L, NEE remained positive in BP and followed the seasonal pattern of air temperature with maximum emission rates of 104 mg C m^{-2} h^{-1}. As a result, the mean midday NEE in BP was significantly higher than in the two restored treatments (Table 6). The NEE rates observed in this study were comparable to those reported from other restored an unrestored peatlands in Canada and Finland (Tuittila *et al.*, 1999; Waddington *et al.*, 2010; Strack & Zuback, 2013). Overall, differences in the re-established WTL baseline had no significant effect on the CO_2 sink-source strength three years following restoration of the abandoned peat extraction area. However, further divergence in the vegetation composition might result in contrasting net CO_2 balances over longer time spans (Weltzin *et al.*, 2000; Yli-Petäys *et al.*, 2007; Samaritani *et al.*, 2011; Vanselow-Algan *et al.*, 2015).

Methane

CH_4 fluxes were in the range of −13 to 60 µg C m^{-2} h^{-1} in all three treatments (Figure 3a in Publication III) which is comparable with CH_4 fluxes reported from other restored and unrestored peatlands in Canada and Finland (Tuittila *et al.*, 2000; Waddington & Day, 2007; Strack & Zuback, 2013). However, occasional peak CH_4 emission of up to 170 and 92 µg C m^{-2} h^{-1} occurred in Res-H and Res-L, respectively. Overall, the annual mean CH_4 exchange was about two times greater in Res-H than in Res-L, however, the differences among the three treatments were not statistically significant (Table 6). Moreover, the CH_4 exchange did not show any significant relationships with vegetation cover or any abiotic variable for any of the three treatments. Given that both WTL and vegetation dynamics have been previously highlighted as major controls on the CH_4 exchange in natural, restored and abandoned peatlands (Bubier, 1995; Frenzel & Karofeld, 2000; Tuittila *et al.*, 2000; Riutta *et al.*, 2007b; Waddington & Day, 2007; Strack *et al.*, 2014), it was surprising to observe this lack of controls and similar CH_4 emissions among the contrasting Res-H, Res-L and BP treatments. Most likely, similar CH_4 emissions in Res-H and Res-L were the result of opposing effects counterbalancing the production and consumption of CH_4. For instance, enhanced anaerobic CH_4 production due to higher WTL in Res-H could have been partly compensated by greater CH_4 oxidation within or immediately below the more developed moss layer (Frenzel & Karofeld, 2000; Basiliko *et al.*, 2004; Larmola *et al.*, 2010). In Res-L on the other hand, greater vascular plant substrate supply might have sustained substantial CH_4 production despite a reduction of the anaerobic zone (Tuittila *et al.*, 2000; Weltzin *et al.*, 2000). Further noteworthy is that, while very few aerenchymatic sedge species (e.g. *Eriophorum* spp.) were established at the time of this study, a future increase in the sedge cover is likely to occur (Tuittila *et al.*, 2000; Weltzin *et al.*, 2000; Vanselow-Algan *et al.*, 2015) which could considerably increase the CH_4 emission in the restored treatments over longer time spans. Nevertheless, this study suggests a limited effect of contrasting WTL baselines and vegetation establishment on the CH_4 emissions during the initial few years following peatland restoration.

Nitrous oxide

N_2O fluxes in Res-H and Res-L were commonly low and remained within the range of −2.8 to 25 μg N m^{-2} h^{-1} for most of the year. Similarly, low N_2O emissions have been reported for natural peatlands (Martikainen *et al.*, 1993) and drained organic soils (Maljanen *et al.*, 2010), however, no other study has estimated N_2O emissions from a restored peatland to date. In comparison, high peak N_2O emissions of 66 to 133 μg N m^{-2} h^{-1} occurred in BP after summer rainfall events (Figure 3b in Publication III). This might be due to the increase in soil moisture and the concurrent decrease in the soil oxygen content following rainfall events which may trigger N_2O flux peaks (Dobbie & Smith, 2003). Averaged over all sampling dates, the mean N_2O exchange was not different between the two restored treatments (−0.12 μg N m^{-2} h^{-1} in Res-H and 2.13 μg N m^{-2} h^{-1} in Res-L). However, mean N_2O exchanges in the restored treatments were significantly lower (by 1–2 magnitudes) compared to the 27.1 μg N m^{-2} h^{-1} in BP (Table 6).

Among all investigated controls, N_2O fluxes correlated best with VWC measured at 0–5 cm soil depth in Res-L ($R^2 = 0.60$) and in BP ($R^2 = 0.39$) but not in Res-H (Figure 8). Soil moisture and WTL effects on the soil oxygen status have been previously identified as the main control on N_2O emissions from pristine and drained peatlands (Firestone & Davidson, 1989; Martikainen *et al.*, 1993; Klemedtsson *et al.*, 2005). In addition, substrate supply (i.e. C and inorganic N) is a key prerequisite for N_2O production (Firestone & Davidson, 1989). In our study, similar N_2O fluxes in the two restored treatments therefore suggest that the differences in WTL, soil moisture and substrate supply from mineralization of organic matter were too small to affect the magnitudes of N_2O emission three years following restoration with different WTL baselines. On the other hand, the enhanced anaerobic conditions due to higher WTLs as well as lower soil N concentrations due to reduced mineralization and enhanced plant N uptake might explain the lower N_2O emissions in the restored Res-H and Res-L treatments relative to BP. Thus, peatland restoration has a large potential for reducing the N_2O emissions commonly occurring in drained abandoned peatlands (Maljanen *et al.*, 2010).

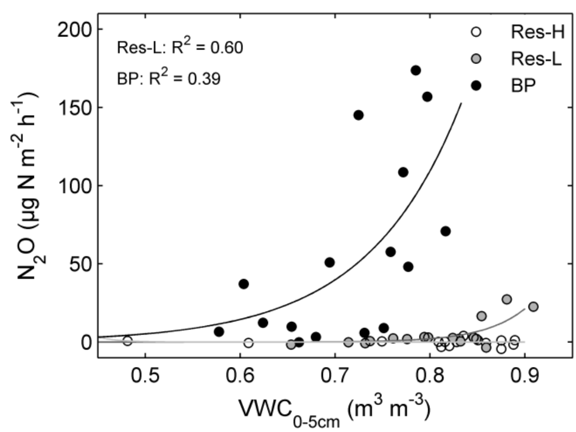

Figure 8. Response of nitrous oxide (N_2O) fluxes to changes in volumetric water content (VWC) measured at 0–5 cm soil depth during the growing season 2014 in restoration treatments with high (Res-H) and low (Res-L) water table level and bare peat (BP). (Publication III)

3.2.4. Annual carbon and greenhouse gas balances

Annual net CO_2 exchanges based on model estimates were 111, 103 and 268 g C m^{-2} yr^{-1} in Res-H, Res-L and BP, respectively (Table 7). The growing season net CO_2 loss (i.e. NEE) represented 45 and 37% of the annual net CO_2 loss in Res-H and Res-L, respectively, while it accounted for 67% in BP. This highlights the importance of accounting for the considerable non-growing season emissions when evaluating the CO_2 sink potential of restored peatlands. The additional C losses via CH_4 emissions were < 0.2 g C m^{-2} yr^{-1} in all treatments. In total, all treatments acted as net C sources, however, the annual C balance in the restored Res-H and Res-L treatments was considerably lower than in the unrestored BP. These results indicate that the CO_2 uptake by the re-established vegetation was not able to compensate for the C losses via respiration and CH_4 emissions three years following restoration.

Several studies have previously reported estimates for the growing season C sink-source strength of restored peatlands, with contrasting findings owing to different restoration techniques, environmental conditions during the study year and time passed since the initiation of the restoration (Tuittila *et al.*, 1999; Bortoluzzi *et al.*, 2006; Yli-Petäys *et al.*, 2007; Waddington *et al.*, 2010; Samaritani *et al.*, 2011; Strack *et al.*, 2014). For instance, restored peatlands in Finland (Tuittila *et al.*, 1999) and Canada (Waddington *et al.*, 2010; Strack *et al.*, 2014) were C sinks during the growing season three to six years after restoration. In contrast, other studies suggested that several decades may be required before restored peatlands resume their functioning as C sinks (Yli-Petäys *et al.*, 2007; Samaritani *et al.*, 2011). Initiating the restoration by raising the WTL in combination with re-introduction of peatland vegetation might, however, reduce the time required for the ecosystem to return to being a C sink similar to that of a natural peatland (Tuittila *et al.*, 2004; Waddington *et al.*, 2010). Overall, this study highlights that while growing season studies can provide important information on processes governing the fluxes, it is necessary to quantify and compare full annual budgets to better evaluate the climate benefits of peatland restoration relative to abandoned peatland areas (and other after-use options, e.g. afforestation or energy crop cultivation).

Table 7. Annual sums of the carbon (C) balance components (g C m^{-2} yr^{-1}) including net ecosystem CO_2 exchange (NEE) and methane (CH_4) fluxes as well as of the greenhouse gas (GHG) balance components (t CO_2 eq ha^{-1} yr^{-1}) including NEE, CH_4 and nitrous oxide (N_2O) exchanges (using global warming potentials of 34 and 298 for CH_4 and N_2O, respectively) in restoration treatments with high (Res-H) and low (Res-L) water table level and bare peat (BP).

	Res-H	Res-L	BP
C balance components			
NEE	110.6	102.7	267.8
CH_4	0.19	0.12	0.14
Total C balance	110.8	102.8	268.0
GHG balance components			
NEE	4.05	3.76	9.82
CH_4	0.09	0.05	0.06
N_2O	0.004	0.020	0.332
Total GHG balance	4.14	3.83	10.21

The total GHG balances, including the net CO_2 exchange as well as CH_4 and N_2O emissions expressed as CO_2 eq, were 4.1, 3.8 and 10.2 t CO_2 eq ha^{-1} yr^{-1} in Res-H, Res-L and BP, respectively (Table 7). The similarity in the GHG balances of the two restored treatments Res-H and Res-L suggests that the differences in the mean WTL had a limited effect on the GHG balance within few years following restoration of the peat extraction area. In comparison, the difference between the GHG balances in restored and BP treatments was considerable, suggesting a reduction in the GHG balance of the restored treatments by about half relative to BP. This reduction was mainly due to lower annual CO_2 emissions (i.e. lower NEE) in the restored treatments compared to BP as a result of increased WTLs and vegetation development. In addition, annual N_2O emissions were also significantly reduced in the restored treatments, although, compared to the differences in the CO_2 balance, the impact of the reduction in N_2O emissions on the GHG balance was relatively small. Another important finding was that the GHG balance was driven by the net CO_2 exchange (96 to 98%) in all three treatments. In contrast, 30 years following rewetting of a German bog, high CH_4 emissions were reported as the main component of the GHG balance (Vanselow-Algan *et al.*, 2015). The same study also reported GHG balances ranging from 25–53 t CO_2 eq ha^{-1} yr^{-1} which are considerably higher compared to our study. This indicates that the GHG balances of restored peatlands may vary greatly over longer time spans. Moreover, this also suggests that the GHG balance of peatland restoration with differing WTL baselines is likely to further diverge over time due to contrasting trajectories in vegetation development and changes in soil biogeochemistry (e.g. pH, nutrient contents and soil moisture dynamics) (Yli-Petäys *et al.*, 2007; Vanselow-Algan *et al.*, 2015). Nevertheless, this study demonstrates that peatland restoration may provide an effective method to mitigate the negative climate impacts of abandoned peat extraction areas.

3.3. N_2O emission from organic soils in Europe (Publication IV)

Across the different land use types on organic soils, highest N_2O fluxes generally occurred in the cropland and grassland sites, whereas natural and rewetted organic soils featured low emissions on average (Figure 9a). Annual N_2O fluxes from active and abandoned peat extraction areas ranged between −0.01 to 3.69 g N m^{-2} yr^{-1} with a mean of 0.47 g N m^{-2} yr^{-1}. Fluxes from forest sites were generally lower than the emissions from croplands, grasslands and peat extraction areas. Furthermore, the mean annual WTLs for different land use categories (Figure 9b) were significantly correlated to N_2O fluxes with a correlation coefficient of r = 0.32 ($P < 0.05$). The sensitivity of N_2O emissions to mean annual WTLs across the various land use classes indicates that WTL management is one of the most effective ways to mitigate N_2O emissions from organic soils.

Overall, N_2O emissions from organic soils were predominantly driven by human management effects on the WTL, while climatic parameters played a secondary role. In addition, soil properties such as the C:N ratio, pH and bulk density further modify the response of N_2O emissions from organic soils to human management. N_2O fluxes from peat extraction areas were best explained by topsoil bulk density, annual precipitation and winter temperature. The positive correlation with bulk density suggests that highest N_2O fluxes occur in highly compacted and degraded peat soils. Thus, variations in management intensity affecting the bulk density in peat extraction sites may have a large impact on annual N_2O emissions. In addition, peak N_2O fluxes can occur immediately after rainfall events (Dobbie & Smith, 2003). Therefore, high annual precipitation amounts can increase the probability of such N_2O peak flux events in drained organic soils. Furthermore, N_2O emissions increased with rising winter air temperatures up to maximum values around 0 °C. Thus, future changes in precipitation patterns and winter temperature might directly affect N_2O emissions from peat extraction areas.

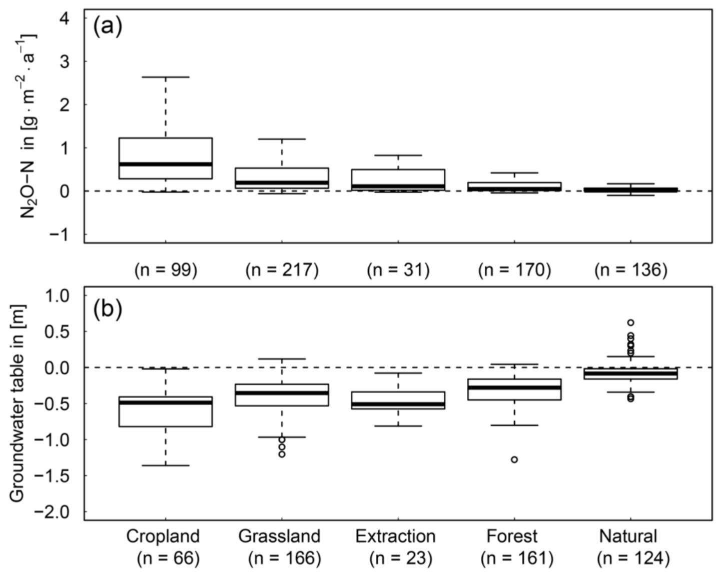

Figure 9. Box plots for nitrous oxide (N_2O) fluxes (panel a) and mean annual groundwater table (panel b) for five different land use categories: cropland, grassland, peat extraction, forest and natural sites. N_2O fluxes are shown without outliers and *n* indicates the number of measurements per category. (Publication IV)

The total European N_2O budget for organic soils was 149.5 Gg N yr^{-1} based on the fuzzy model estimate (Table 4 in Publication IV). This estimate was almost twice as high compared to the IPCC default methodology which is based on constant emission factors. This suggests that N_2O emissions from organic soils may be even more significant than previously estimated. Due to their small area coverage, peat extraction sites emitted in total only 0.1 Gg N yr^{-1} suggesting that peat extraction contributes little to the European N_2O budget compared to other land use types (e.g. croplands). Furthermore, the strongest hotspots for N_2O emissions were observed in acidic croplands such as in Denmark or Poland and in intensively fertilized grasslands such as in the Netherlands or Germany. This finding gives important information regarding where to focus N_2O mitigation since croplands and grasslands represent the main sources of N_2O emissions per area.

4. CONCLUSIONS

The results presented in this dissertation demonstrate the impact of reed canary grass cultivation and peatland restoration on the dynamics of the individual CO_2, CH_4 and N_2O fluxes as well as on the resulting total C and GHG balances of abandoned peat extraction areas.

A greater net C uptake and lower net GHG emissions in RCG cultivations relative to bare peat soil were observed in both Publications I and II which suggests that RCG cultivation may provide an effective method for mitigating the net C and GHG emissions from abandoned peat extraction areas. However, the C and GHG sink-source strength of RCG cultivations may vary between a sink in cool and wet years (Publication I) and a source in warm and dry years (Publication II). The between-year difference was related to contrasting amounts of precipitation which was identified as the major control of above- and belowground biomass production and thus of the C and GHG sink-source strength. These findings highlight the strong impact of climatic conditions on the C and GHG balances of RCG cultivations on drained organic soils.

Greater net C uptake and lower net GHG emissions observed in fertilized relative to nonfertilized RCG cultivations suggest that fertilization increased the climate benefit potential of RCG cultivations. This increase resulted from enhanced biomass production and net CO_2 uptake which largely exceeded the increase in soil N_2O emissions (in CO_2 equivalents) following moderate fertilization. Thus, fertilization could be a beneficial management practice to maximize biomass yields 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 must be considered when evaluating optimum fertilizer rates.

The net CO_2 exchange determined both the C and the GHG balances in fertilized and nonfertilized RCG cultivations. In comparison, the contributions of CH_4, N_2O and DOC fluxes to the full C and GHG balances were relatively small (1–6%). Management practices in drained organic soils need to be therefore carefully evaluated with respect to their direct and indirect impacts on the net ecosystem CO_2 exchange. Thus, when converting abandoned peat extraction areas into RCG cultivations, management strategies need to ensure optimum plant growth through raised WTLs and sufficient nutrient supply to maximize the net ecosystem CO_2 uptake since its benefits are likely to considerably exceed the associated potentially negative effects from increased CH_4 and N_2O emissions.

The net C and GHG emissions in the restored treatments were reduced by approximately half relative to those in the abandoned bare peat site three years following restoration (Publication III). This demonstrates that peatland restoration may effectively mitigate the negative climate impacts of drained peat soils. Changes in the C and GHG balances following restoration of the peat extraction area were mainly due to a considerable reduction in peat mineralization which advocates raising the WTL as an effective method to reduce the aerobic organic matter decomposition commonly occurring in drained peatlands. Furthermore, raised WTLs in the restored treatments resulted in significantly reduced N_2O emissions whereas the effect on the CH_4 fluxes was negligible compared to the bare peat site.

Water table level differences following peatland restoration (wetter and drier treatments) had a strong impact on vegetation community development. Furthermore, the difference in vegetation cover and composition was identified as the main control of within- and between-site variations in plant production and respiration. Thus, variations in the re-established WTL baselines may have important implications for plant-related CO_2 fluxes in restored peatlands. In contrast, differing WTL baselines had minor effects on the net CO_2 exchange due to the concurrent changes in plant production and ecosystem respiration fluxes in wet and dry treatments. Moreover, the similar CH_4 and N_2O exchanges in the two restored treatments suggest that the difference in mean WTLs had a limited effect on the C and GHG balances three years following restoration.

Overall, both bioenergy crop cultivation and peatland restoration may serve as effective methods for mitigating the negative climate impact of abandoned peat extractions areas. The ultimate choice of after-use option will, however, further depend on a combination of site-specific factors and socio-economic interests of the land owner. Given the observed sensitivity of the C and GHG balances to climatic conditions, future research needs to address alternative management options to ensure sustainable yields and climate benefits in RCG cultivations on drained organic soils. Furthermore, understanding the long-term (i.e. decades) impact of bioenergy cultivation and peatland restoration on the C and GHG balances is required since it will improve predictions of ecosystem responses to changes in future management strategies and climatic conditions.

REFERENCES

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.

Alm J, Schulman L, Walden J, Nykänen H, Martikainen PJ, Silvola J (1999) Carbon Balance of a Boreal Bog during a Year with an Exceptionally Dry Summer. *Ecology*, **80**, 161–174.

Alm J, Shurpali NJ, Minkkinen K, Aro L, Hytonen J, Laurila T, Lohila A, Maljanen M, Martikainen PJ, Makiranta P, Penttila T, Saarnio S, Silvan N, Tuittila ES, Laine J (2007) Emission factors and their uncertainty for the exchange of CO_2, CH_4 and N_2O in Finnish managed peatlands. *Boreal Environment Research*, **12**, 191–209.

Basiliko N, Knowles R, Moore TR (2004) Roles of moss species and habitat in methane consumption potential in a northern peatland. *Wetlands*, **24**, 178–185.

Basiliko N, Blodau C, Roehm C, Bengtson P, Moore TR (2007) Regulation of Decomposition and Methane Dynamics across Natural, Commercially Mined, and Restored Northern Peatlands. *Ecosystems*, **10**, 1148–1165.

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.

Bortoluzzi E, Epron D, Siegenthaler A, Gilbert D, Buttler A (2006) Carbon balance of a European mountain bog at contrasting stages of regeneration. *New Phytologist*, **172**, 708–718.

Bragazza L, Parisod J, Buttler A, Bardgett RD (2013) Biogeochemical plant-soil microbe feedback in response to climate warming in peatlands. *Nature Climate Change*, **3**, 273–277.

Bubier JL (1995) The Relationship of Vegetation to Methane Emission and Hydrochemical Gradients in Northern Peatlands. *The Journal of Ecology*, **83**, 403.

Bubier J, Crill P, Mosedale A, Frolking S, Linder E (2003) Peatland responses to varying interannual moisture conditions as measured by automatic CO_2 chambers. *Global Biogeochemical Cycles*, **17**, 1066.

Chapuis-Lardy L, Wrage N, Metay A, Chotte J-L, Bernoux M (2007) Soils, a sink for N_2O? A review. *Global Change Biology*, **13**, 1–17.

Charman D (2002) *Peatlands and Environmental Change*. John Wiley & Sons Ltd, Chichester, 301 pp.

Charman DJ, Beilman DW, Blaauw M, Booth RK, Brewer S, Chambers FM, Christen JA, Gallego-Sala A, Harrison SP, Hughes PDM, Jackson ST, Korhola A, Mauquoy D, Mitchell FJG, Prentice IC, van der Linden M, De Vleeschouwer F, Yu ZC, Alm J, Bauer IE, Corish YMC, Garneau M, Hohl V, Huang Y, Karofeld E, Le Roux G, Loisel J, Moschen R, Nichols JE, Nieminen TM, MacDonald GM, Phadtare NR, Rausch N, Sillasoo U, Swindles GT, Tuittila ES, Ukonmaanaho L, Valiranta M, van Bellen S, van Geel B, Vitt DH, Zhao Y (2013) Climate-related changes in peatland carbon accumulation during the last millennium. *Biogeosciences*, **10**, 929–944.

Clymo RS (1984) The Limits to Peat Bog Growth. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences*, **303**, 605–654.

De Klein C, Novoa RSA, Ogle S, Smith KA, Rochette P, Wirth TC, McConkey BG, Mosier A, Rypdal K, Walsh M, Williams SA (2006) N_2O emissions from managed

soils, and CO_2 emissions from lime and urea application (Chapter 11). In: 2006 IPCC Guidelines for National Greenhouse Gas Inventories.
Deppe M, Knorr K-H, McKnight DM, Blodau C (2010) Effects of short-term drying and irrigation on CO_2 and CH_4 production and emission from mesocosms of a northern bog and an alpine fen. *Biogeochemistry*, **100**, 89–103.
Dobbie KE, Smith KA (2003) Nitrous oxide emission factors for agricultural soils in Great Britain: the impact of soil water-filled pore space and other controlling variables. *Global Change Biology*, **9**, 204–218.
Don A, Osborne B, Hastings A, Skiba U, Carter MS, Drewer J, Flessa H, Freibauer A, Hyvonen N, Jones MB, Lanigan GJ, Mander Ü, Monti A, Djomo SN, Valentine J, Walter K, Zegada-Lizarazu W, Zenone T (2012) Land-use change to bioenergy production in Europe: implications for the greenhouse gas balance and soil carbon. *Global Change Biology – Bioenergy*, **4**, 372–391.
Fenner N, Freeman C (2011) Drought-induced carbon loss in peatlands. *Nature Geoscience*, **4**, 895–900.
Firestone M, Davidson E (1989) *Microbiological Basis of NO and N_2O Production and Consumption in Soil*, Vol. 47 (eds. Andreae M, Schimel D). John Wiley & Sons Ltd, Chichester.
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, Kang H, Dowrick DJ, Reynolds B, Lock MA, Sleep D, Hughes S, Hudson J (2004) Export of dissolved organic carbon from peatlands under elevated carbon dioxide levels. *Nature*, **430**, 195–198.
Frenzel P, Karofeld E (2000) CH_4 emission from a hollow-ridge complex in a raised bog: The role of CH_4 production and oxidation. *Biogeochemistry*, **51**, 91–112.
Frilander P, Leinonen A, Alakangas E (1996) Peat production technology. In: *Peatlands in Finland* (ed. Vasander, H), pp. 99–106. Finnish Peatland Society, Helsinki.
Frolking S, Roulet NT, Moore TR, Richard PJH, Lavoie M, Muller SD (2001) Modeling Northern Peatland Decomposition and Peat Accumulation. *Ecosystems*, **4**, 479–498.
Gorham E (1991) Northern Peatlands: Role in the Carbon Cycle and Probable Responses to Climatic Warming. *Ecological Applications*, **1**, 182–195.
Graf M, Rochefort L (2009) Examining the Peat-Accumulating Potential of Fen Vegetation in the Context of Fen Restoration of Harvested Peatlands. *Ecoscience*, **16**, 158–166.
Harriss RC, Gorham E, Sebacher DI, Bartlett KB, Flebbe PA (1985) Methane flux from northern peatlands. *Nature*, **315**, 652–654.
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.
Hutchinson GL, Livingston GP (1993) Use of Chamber Systems to Measure Trace Gas Fluxes. In: *Agricultural Ecosystem Effects on Trace Gases and Global Climate Change, ASA Special Publication No. 55* (eds. Rolston DE, Duxbury JM, Harper LA, Mosier AR). American Society of Agronomy, Crop Science Society of America, and Soil Science Society of America.
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 (2007) Climate Change 2007: The physical science basis. *Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change* (eds: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, et al.), Cambridge University Press, Cambridge, UK and New York, NY, USA, 996 pp.

IPCC (2013) The physical science basis. *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, 1535 pp.

Järveoja J, Peichl M, Maddison M, Soosaar K, Vellak K, Karofeld E, Teemusk A, Mander Ü (Submitted) Impact of water table level on annual carbon and greenhouse gas balances of a restored peat extraction area. *Biogeosciences*.

Järveoja J, Peichl M, Maddison M, Teemusk A, Mander Ü (2015) Full carbon and greenhouse gas balances of fertilized and nonfertilized reed canary grass cultivations on an abandoned peat extraction area in a dry year. *Global Change Biology – Bioenergy*, doi: 10.1111/gcbb.12308.

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, 304 pp.

Kandel TP, Elsgaard L, Karki S, Lærke PE (2013a) 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.

Kandel TP, Elsgaard L, Lærke PE (2013b) Measurement and modelling of CO_2 flux from a drained fen peatland cultivated with reed canary grass and spring barley. *Global Change Biology – Bioenergy*, **5**, 548–561.

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

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.

Koehler A-K, Sottocornola M, Kiely G (2011) How strong is the current carbon sequestration of an Atlantic blanket bog? *Global Change Biology*, **17**, 309–319.

Lafleur PM, Roulet NT, Bubier JL, Frolking S, Moore TR (2003) Interannual variability in the peatland-atmosphere carbon dioxide exchange at an ombrotrophic bog. *Global Biogeochemical Cycles*, **17**, 1036.

Lai DYF (2009) Methane Dynamics in Northern Peatlands: A Review. *Pedosphere*, **19**, 409–421.

Laine J, Laiho R, Minkkinen K, Vasander H (2006) Forestry and boreal peatlands. In: *Boreal Peatland Ecosystems*, Vol. 188 (eds. Wieder RK, Vitt DH), pp. 331–358, Springer-Verlag Berlin Heidelberg.

Lamers LPM, Vile MA, Grootjans AP, Acreman MC, van Diggelen R, Evans MG, Richardson CJ, Rochefort L, Kooijman AM, Roelofs JGM, Smolders AJP (2015) Ecological restoration of rich fens in Europe and North America: from trial and error to an evidence-based approach. *Biological Reviews of the Cambridge Philosophical Society*, **90**, 182–203.

Lappalainen E (1996) *Global Peat Resources.* International Peat Society, Finland.

Larmola T, Tuittila E-S, Tiirola M, Nykanen H, Martikainen PJ, Yrjala K, Tuomivirta T, Fritze H (2010) The role of *Sphagnum* mosses in the methane cycling of a boreal mire. *Ecology*, **91**, 2356–2365.

Leifeld J, Steffens M, Galego-Sala A (2012) Sensitivity of peatland carbon loss to organic matter quality. *Geophysical Research Letters*, **39**, L14704.

Lemus R, Lal R (2005) Bioenergy Crops and Carbon Sequestration. *Critical Reviews in Plant Sciences*, **24**, 1–21.

Leppelt T, Dechow R, Gebbert S, Freibauer A, Lohila A, Augustin J, Drösler M, Fiedler S, Glatzel S, Höper H, Järveoja J, Laerke PE, Maljanen M, Mander Ü, Mäkiranta P, Minkkinen K, Ojanen P, Regina K, Strömgren M (2014) Nitrous oxide emission budgets and land-use-driven hotspots for organic soils in Europe. *Biogeosciences*, **11**, 6595–6612.

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.

Limpens J, Berendse F, Blodau C, Canadell JG, Freeman C, Holden J, Roulet N, Rydin H, Schaepman-Strub G (2008) Peatlands and the carbon cycle: from local processes to global implications – a synthesis. *Biogeosciences*, **5**, 1475–1491.

Lloyd J, Taylor JA (1994) On the Temperature Dependence of Soil Respiration. *Functional Ecology*, **8**, 315–323.

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.

Lund M, Lafleur PM, Roulet NT, Lindroth A, Christensen TR, Aurela M, Chojnicki BH, Flanagan LB, Humphreys ER, Laurila T, Oechel WC, Olejnik J, Rinne J, Schubert P, Nilsson MB (2010) Variability in exchange of CO_2 across 12 northern peatland and tundra sites. *Global Change Biology*, **16**, 2436–2448.

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. *Global Change Biology – Bioenergy*, **4**, 462–474.

Martikainen PJ, Nykänen H, Crill P, Silvola J (1993) Effect of a lowered water table on nitrous oxide fluxes from northern peatlands. *Nature*, **366**, 51–53.

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.

McNally G (1995) The utilisation of industrial cutaway peatland. The key factors influencing the various afteruse options. In: *Proceedings of the Conference on the Peat Industry and the Environment*, pp. 84–86. Tallinn, Estonia.
Montanarella L, Jones R, Hiederer R (2006) The distribution of peatland in Europe. *Mires and Peatland*, **1**.
Nilsson M, Sagerfors J, Buffam I, Laudon H, Eriksson T, Grelle A, Klemedtsson L, Weslien P, Lindroth A (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.
Orru M, Orru H (2008) Sustainable use of Estonian peat reserves and environmental challenges. *Estonian Journal of Earth Sciences*, **57**, 87–93.
Paal J, Leibak E (2011) *Estonian Mires: Inventory of Habitats*. Publication of the project "Estonian Mires Inventory completion for maintaining biodiversity". Regio, Tartu.
Peichl M, Öquist M, Ottosson Löfvenius M, Ilstedt U, Sagerfors J, Grelle A, Lindroth A, Nilsson MB (2014) A 12-year record reveals pre-growing season temperature and water table level threshold effects on the net carbon dioxide exchange in a boreal fen. *Environmental Research Letters*, **9**, 055006.
Quinty F, Rochefort L (2003) *Peatland Resoration Guide: Second Edition*, Canadian Sphagnum Peat Moss Association and New Brunswick Department of Natural Resources and Energy.
Ramst R, Orru M (2009) Recultivation of abandoned peat extraction areas in Estonia. In: *Estonian Combustible Natural Resources and Wastes,* Estonian Biofuels Association, Tallinn. In Estonian, abstract in English.
Regina K, Nykänen H, Silvola J, Martikainen PJ (1996) Fluxes of nitrous oxide from boreal peatlands as affected by peatland type, water table level and nitrification capacity. *Biogeochemistry*, **35**, 401–418.
Riutta T, Laine J, Tuittila E-S (2007a) Sensitivity of CO_2 Exchange of Fen Ecosystem Components to Water Level Variation. *Ecosystems*, **10**, 718–733.
Riutta T, Laine J, Aurela M, Rinne J, Vesala T, Laurila T, Haapanala S, Pihlatie M, Tuittila ES (2007b) Spatial variation in plant community functions regulates carbon gas dynamics in a boreal fen ecosystem. *Tellus B*, **59**.
Rochefort L, Lode E (2006) Restoration of Degraded Boreal Peatlands. In: *Boreal Peatland Ecosystems* (eds. Wieder K, Vitt D), pp. 381–423. Springer Berlin Heidelberg.
Roobroeck D, Butterbach-Bahl K, Brüggemann N, Boeckx P (2010) Dinitrogen and nitrous oxide exchanges from an undrained monolith fen: short-term responses following nitrate addition. *European Journal of Soil Science*, **61**, 662–670.
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.
Rydin H (1985) Effect of Water Level on Desiccation of Sphagnum in Relation to Surrounding Sphagna. *Oikos*, **45**, 374–379.
Rydin H, Jeglum JK (2006) *The Biology of Peatlands*. Oxford University Press, 400 pp.
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.
Samaritani E, Siegenthaler A, Yli-Petäys M, Buttler A, Christin P-A, Mitchell EAD (2011) Seasonal Net Ecosystem Carbon Exchange of a Regenerating Cutaway Bog:

How Long Does it Take to Restore the C-Sequestration Function? *Restoration Ecology*, **19**, 480–489.
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.
Schulze E, Kelliher FM, Korner C, Lloyd J, Leuning R (1994) Relationships among Maximum Stomatal Conductance, Ecosystem Surface Conductance, Carbon Assimilation Rate, and Plant Nitrogen Nutrition: A Global Ecology Scaling Exercise. *Annual Review of Ecology and Systematics*, **25**, 629–662.
Shurpali NJ, Verma SB, Kim J, Arkebauer TJ (1995) Carbon dioxide exchange in a peatland ecosystem. *Journal of Geophysical Research: Atmospheres*, **100**, 14319–14326.
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 B*, **60**, 200–209.
Shurpali NJ, Hyvönen NP, Huttunen JT, Clement RJ, Reichstein M, Nykänen H, Biasi C, Martikainen PJ (2009) Cultivation of a perennial grass for bioenergy on a boreal organic soil – carbon sink or source? *Global Change Biology – Bioenergy*, **1**, 35–50.
Shurpali NJ, Strandman H, Kilpeläinen A, Huttunen J, Hyvönen N, Biasi C, Kellomäki S, Martikainen PJ (2010) Atmospheric impact of bioenergy based on perennial crop (reed canary grass, *Phalaris arundinaceae*, L.) cultivation on a drained boreal organic soil. *Global Change Biology – Bioenergy*, **2**, 130–138.
Silvan N, Tuittila E-S, Kitunen V, Vasander H, Laine J (2005) Nitrate uptake by *Eriophorum vaginatum* controls N_2O production in a restored peatland. *Soil Biology and Biochemistry*, **37**, 1519–1526.
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. *Global Change Biology – Bioenergy*, **2**, 169–179.
Strack M (2008) *Peatlands and climate change*. IPS, International Peat Society.
Strack M, Zuback YCA (2013) Annual carbon balance of a peatland 10 yr following restoration. *Biogeosciences*, **10**, 2885–2896.
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.
Strack M, Keith AM, Xu B (2014) Growing season carbon dioxide and methane exchange at a restored peatland on the Western Boreal Plain. *Ecological Engineering*, **64**, 231–239.
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.
Tiedje JM, Sexstone AJ, Myrold DD, Robinson JA (1983) Denitrification: ecological niches, competition and survival. *Antonie van Leeuwenhoek*, **48**, 569–583.
Tuittila E-S, Komulainen V-M, Vasander H, Laine J (1999) Restored cut-away peatland as a sink for atmospheric CO_2. *Oecologia*, **120**, 563–574.
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.

Tuittila ES, Vasander H, Laine J (2004) Sensitivity of C sequestration in reintroduced *Sphagnum* to water-level variation in a cutaway peatland. *Restoration Ecology*, **12**, 483–493.

Turner NC, Schulze E-D, Gollan T (1985) The responses of stomata and leaf gas exchange to vapour pressure deficits and soil water content. *Oecologia*, **65**, 348–355.

Turunen J, Tomppo E, Tolonen K, Reinikainen A (2002) Estimating carbon accumulation rates of undrained mires in Finland – application to boreal and subarctic regions. *The Holocene*, **12**, 69–80.

Vanselow-Algan M, Schmidt SR, Greven M, Fiencke C, Kutzbach L, Pfeiffer E-M (2015) High methane emissions dominate annual greenhouse gas balances 30 years after bog rewetting. *Biogeosciences*, **12**, 2809–2842.

Vasander H, Tuittila E-S, Lode E, Lundin L, Ilomets M, Sallantaus T, Heikkilä R, Pitkänen M-L, Laine J (2003) Status and restoration of peatlands in northern Europe. *Wetlands Ecology and Management*, **11**, 51–63.

Venendaal R, Jørgensen U, Foster CA (1997) European energy crops: A synthesis. *Biomass and Bioenergy*, **13**, 147–185.

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

Waddington JM, Day SM (2007) Methane emissions from a peatland following restoration. *Journal of Geophysical Research: Biogeosciences*, **112**, G03018.

Waddington JM, McNeil P (2002) Peat oxidation in an abandoned cutover peatland. *Canadian Journal of Soil Science*, **82**, 279–286.

Waddington JM, Warner KD (2001) Atmospheric CO_2 sequestration in restored mined peatlands. *Ecoscience*, **8**, 359–368.

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

Waddington JM, Strack M, Greenwood MJ (2010) Toward restoring the net carbon sink function of degraded peatlands: Short-term response in CO_2 exchange to ecosystem-scale restoration. *Journal of Geophysical Research: Biogeosciences*, **115**, G01008.

Weltzin JF, Pastor J, Harth C, Bridgham SD, Updegraff K, Chapin CT (2000) Response of bog and fen plant communities to warming and water-table manipulations. *Ecology*, **81**, 3464–3478.

Whiting GJ, Chanton JP (2001) Greenhouse carbon balance of wetlands: methane emission versus carbon sequestration. *Tellus B*, **53**.

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.

Wrobel C, Coulman BE, Smith DL (2009) The potential use of reed canarygrass (*Phalaris arundinacea* L.) as a biofuel crop. *Acta Agriculturae Scandinavica, Section B – Soil & Plant Science*, **59**, 1–18.

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.

Yli-Petäys M, Laine J, Vasander H, Tuittila E-S (2007) Carbon gas exchange of a re-vegetated cut-away peatland five decades after abandonment. *Boreal Environment Research*, **12**, 177–190.

SUMMARY IN ESTONIAN

Kasvuhoonegaaside CO_2, CH_4 ja N_2O vood päiderooga rekultiveeritud ja turbasamblaga taastatud jääkturbasoodest

Soode olulisus on tänapäeval laialdaselt tunnustatud ning paljudes riikides on nende kaitseks või taastamiseks rakendatud rohkelt meetmeid (Paal & Leibak, 2011). Loodusliku sooökosüsteemi tähtsaim funktsionaalne iseärasus on turba teke ja ladestumine (Wieder *et al.*, 2006). Turba järk-järgulise ladestumise käigus toimub orgaanilise süsiniku akumuleerumine ja seega on looduslikud turbaalad, vaatamata nende küllaltki väikesele katvusele (umbes 3% maakera maismaa pinnast), ühed olulisemad süsiniku reservuaarid ja globaalse kliima reguleerijad (Joosten & Clarke, 2002). Ainuüksi põhjapoolkeral asuvate soode süsinikuvaru hinnatakse olevat 270 kuni 450 Pg C, mis moodustab enam kui 25% maailma muldade süsinikuvarust (Gorham, 1991; Turunen *et al.*, 2002).

Eesti on üks sooderikkamaid piirkondi maailmas – Eesti turbaalade kogupindala on ligikaudu 1×10^6 ha, mis moodustab ~22% riigi territooriumist (Orru & Orru, 2008). Soode laialdasest kasutusest tingituna on nende pindala aga kiirelt vähenenud ning arvatakse vaid umbkaudu 5.5% on jätkuvalt säilinud looduslikus seisundis (Paal & Leibak, 2011). Soid on kasutatud väga mitmesugustel eesmärkidel, sealhulgas põllumajanduses ja metsanduses. Lisaks on turvas üheks tähtsaimaks maavaraks Eestis ja seega on turbasoid ulatuslikult ka kaevandatud. Aktiivselt kaevandatavate turbaväljade pindala Eestis on praegu ~20 000 ha. Lisaks on turba kaevandamine lõpetatud ~10 000 hektaril, mis tähendab, et kokku on Eestis turbakaevandamisega rikutud alasid ligikaudu 30 000 ha (Orru & Orru, 2008; Paal & Leibak, 2011).

Mahajäetud lõpuni ammendamata jäänud turbatootmisalad ehk jääkturbasood on olulised kasvuhoonegaaside (KHGde) allikad (Maljanen *et al.*, 2010). Turbasoode kuivendamise ja kaevandamise tulemusel kiireneb ladestunud orgaanilise aine mineraliseerumine, mis põhjustab suurenenud süsihappegaasi (CO_2) emissiooni atmosfääri. Metaani (CH_4) emissioon on üldjuhul madalam kui looduslikes soodes, kuid võib ka mahajäetud aladel olla märkimisväärne (Salm *et al.*, 2012), samal ajal kui kuivendusjärgsed dilämmastikoksiidi (N_2O) vood on varieeruvad ja sõltuvad paljuski turba lämmastikusisaldusest. Kasvuhoonegaaside emissiooni leevendamiseks tuleks mahajäetud kaevandusaladel turvas kas lõpuni kaevandada või siis taastada need alad viisil, mis KHGde emissiooni vähendaks.

Üheks võimalikuks mahajäetud turbakaevandusalade korrastamise viisiks on nende alade kasutamine energiakultuuride, nt päideroo (*Phalaris arundinacea*) kasvatamiseks. Soomes ja Taanis läbi viidud uuringud päideroo kultiveerimisest jääkturbasoodes näitavad, et KHGde emissioon neil aladel võib märkimisväärselt kahaneda (Hyvönen *et al.*, 2009; Shurpali *et al.*, 2009; Karki *et al.*, 2015). Teisalt on võimalik ka loodusliku soo-suunalise taimestumise soodustamine veetaseme tõstmise ja turbasambla (*Sphagnum* spp.) fragmentide külvamise kaudu, mis on KHGde emissiooni kahandamise osas seni edukaid tulemusi näidanud eelkõige Põhja-Ameerikas ja ka Soomes (Tuittila *et al*., 1999; Waddington *et al*., 2010; Strack *et al*., 2014). Olemasolevad andmed põhinevad aga vaid mõnedes riikides läbi viidud uuringutel, mis sageli on keskendunud vaid kasvuperioodiaegsete KHGde voogude mõõtmisele ning seega on aastaseid süsiniku ja KHGde bilansse kajastavaid uuringuid kirjanduses seni jätkuvalt vähe.

Käesoleva doktoritöö peamiseks eesmärgiks oli uurida päiderooga rekultiveerimise ja turbasamblaga taastamise mõju mahajäetud turbatootmisalade kasvuhoonegaaside (CO_2, CH_4 ja N_2O) voogudele. Uurimistööd teostati päiderooga taimestatud (väetatud ja väetamata uurimisalad) endisel turbakaevandusalal Lavassaares ning turbasamblaga taastatud (kõrge ja madala veetasemega uurimisalad) endisel turbakaevandusalal Tässis. Kasvuhoonegaaside voogusid mõõdeti aastaringselt suletud kambri meetodil kahe- kuni neljanädalase sammuga. Päideroo biomassi produktsiooni hindamiseks koguti vegetatsiooniperioodi alguses ja lõpus maapealse ja maa-aluse biomassi proovid. Turbasamblaga taastatud aladel teostati taimkatteanalüüs. Täiendavalt mõõdeti igal proovivõtul erinevaid gaasivoogude dünaamikat reguleerivaid keskkonnaparameetreid, nt mullaniiskust ja -temperatuuri, veetaset jne. Lisaks teostati 109 erineva orgaanilisel mullal paikneva ala N_2O voogude põhjal modelleerimispõhine analüüs Euroopa N_2O bilansi ja seda peamiselt mõjutavate tegurite leidmiseks.

Päideroo biomassi produktsioon varieerus erinevate uurimisaastate lõikes märkimisväärselt. Sademeterohkel 2010. aastal (911 mm) oli maapealse biomassi saagikus väetatud ja väetamata aladel vastavalt 14 ja 8 t ha^{-1} (Publikatsioon I – Mander *et al*., 2012), mis on kõrgem Soomes leitud saagikuse andmetest (kuni 5 t ha^{-1}) (Shurpali *et al*., 2009), kuid võrreldav varasemalt Taanis saadud tulemustega (kuni 16 t ha^{-1}) (Kandel *et al*., 2013a; Karki *et al*., 2014). Seevastu sademetevaesel 2014. aastal (525 mm) oli biomassi saagikus mõlemal alal < 3 t ha^{-1} (Publikatsioon II – Järveoja *et al*., 2015). Kuna päideroo saagi suurus on olulisel määral mõjutatud ilmastikutingimustest, siis võib eeldada, et aastatevahelise suure saagierinevuse ning kuival 2014. aastal saadud madala saagikuse peamiseks põhjuseks oli põuane suvi ja sellest tingitud ebasoodsad kasvutingimused. Selle põhjal võib järeldada, et kuivendatud orgaanilistel muldadel on jätkusuutliku biomassi tootmise potentsiaal kuivadel aastatel täiendava veetaseme reguleerimise võimaluseta limiteeritud.

Väetatud ja väetamata päideroo uurimisalade aastased CO_2 bilansid olid märjal 2010. aastal negatiivsed, mis tähendab, et CO_2 sidumine maapealsesse ja maa-alusesse biomassi ületas orgaanilise aine lagundamisel tekkiva CO_2 emissiooni ning alad olid kokkuvõttes olulised süsiniku sidujad. Leitud KHGde bilansid, kus lisaks CO_2 vahetusele arvestati CO_2 ekvivalentidesse (CO_2 ekv) ümberarvutatuna ka CH_4 ja N_2O voogusid (CH_4 = 34 CO_2 ekv ja N_2O = 298 CO_2 ekv), olid samuti negatiivsed (vastavalt −6.0 ja −3.9 t CO_2 ekv ha^{-1} a^{-1}), samal ajal kui mahajäetud freesturbaala KHGde bilanss oli positiivne (2.5 t CO_2 ekv ha^{-1} a^{-1}). Kuival 2014. aastal olid kõik uurimisalad väiksemast CO_2 sidumisest tingituna aga süsiniku ja KHGde allikad, väetatud ja väetamata päideroo alade ja freesturba KHGde bilansid olid vastavalt 3.6, 7.9 ja 6.6 t CO_2 ekv ha^{-1} a^{-1}. Need tulemused ühtivad Soomes läbiviidud uuringutega, mis leidsid samuti, et CO_2 sidumise intensiivsus on kuival aastal märkimisväärselt madalam (Shurpali *et al.*, 2009). See näitab, et päiderooga taimestamine võib olla efektiivne meetod turbatootmisaladelt lähtuva KHGde emissiooni leevendamiseks, kuid olenevalt klimaatilistest tingimustest võivad päiderooga taimestatud turbatootmisalad olla kokkuvõttes nii süsiniku ja KHGde allikad kui ka sidujad.

Päiderooga taimestatud uurimisalade KHG bilansside omavahelisel võrdlemisel on näha, et väetatud ala bilanss oli nii märjal kui ka kuival uurimisaastal väetamata ala bilansist oluliselt madalam. See näitab, et mõõdukas väetamine võib päideroo kasvualade summaarset kasvuhooneefekti tekitavat mõju vähendada, kuna selle positiivne mõju biomassi produktsioonile ja seeläbi ka CO_2 sidumise intensiivsusele ületab mitmekordselt väetamisega kaasnevaid võimalikke negatiivseid mõjusid suurenenud N_2O emissioonist. Veelgi enam, väetatud päideroo ala KHGde bilanss oli freesturbaala omaga võrreldes madalam nii märjal kui ka kuival aastal, samal ajal kui väetamata uurimisala leevendas KHGde emissiooni mahajäetud alaga võrreldes ainult märjal aastal. Töö tulemused näitasid ka seda, et CO_2 vahetus oli peamiseks süsiniku ja KHGde bilanssi mõjutavaks teguriks, samal ajal kui CH_4, N_2O ja lahustunud orgaanilise süsiniku voogude mõju kogubilanssidele oli väike (1–6% olenevalt uurimisalast). Sarnaseid tulemusi on leitud ka varasemalt (Hyvönen *et al.*, 2009; Karki *et al.*, 2015) ning seega tuleks päideroo biomassi saagikuse ja seeläbi ka CO_2 sidumise maksimaalseks suurendamiseks panustada eelkõige piisava vee- ja taimetoitainete kättesaadavuse tagamisele.

Veetaseme tõstmine ning turbasambla fragmentide külvamine mõjutas olulisel määral uurimisalade taimestiku katvust. Kõrge veetasemega (aasta keskmine veetase −24 cm) taastatud ala taimestiku katvus oli 3 aastat peale taastamistööde teostamist 63%, samas kui madala veetasemega (−31 cm) ala taimestiku katvus oli 52%. Taastamisjärgne veetaseme erinevus mõjutas oluliselt ka kujunenud taimestiku liigilist koosseisu. Kõrge veetasemega taastatud ala taimestik koosnes peamiselt (62%) sammaltaimedest, millest omakorda turbasammalde katvus moodustas 61%. Madala veetasemega taastatud ala taimkate koosnes aga 44% ulatuses sammaltaimedest ning 14% ulatuses soontaimedest. Seega on taastamisjärgselt saavutatud keskmine veetase olulise tähtsusega, kuna erinevused taimestiku katvuses ning liigilises koosseisus võivad olulisel määral mõjutada ka süsiniku ja KHGde bilanssi (Weltzin *et al.*, 2000).

Turbasamblaga taastatud uurimisalade aastased KHGde bilansid olid 3 aastat peale taastamistööde teostamist positiivsed, seda nii kõrge kui madala veetasemega taastatud uurimisaladel (vastavalt 4.1 ja 3.8 t CO_2 ekv ha^{-1} a^{-1}) (Publikatsioon III – Järveoja *et al.*, Submitted). Mahajäetud turbatootmisala (aasta keskmine veetase −46 cm) KHGde bilanss oli samal ajal võrdluseks

10.2 t CO_2 ekv ha^{-1} a^{-1}. Antud tulemused näitavad, et turbasoode taastamine võib olla väga efektiivseks meetodiks kuivendatud turvasmuldade negatiivse klimaatilise mõju vähendamiseks – taastatud alade KHGde bilansid olid mahajäetud freesturbaalaga võrreldes ligikaudu 2 korda madalamad. Muutused süsiniku ja KHGde bilansis olid peamiselt tingitud märkimisväärselt vähenenud turba mineraliseerumisest, mis näitab, et veerežiimi taastamine ja veetaseme tõstmine on väga efektiivne meetod kuivendatud turbasoodes toimuva aeroobse orgaanilise aine lagunemise vähendamiseks. Lisaks vähendas veetaseme tõstmine ka mitme suurusjärgu võrra N_2O emissioone, samal ajal kui mõju CH_4 voogudele oli freesturbaalaga võrreldes ebaoluline nii kõrge kui ka madala veetasemega taastatud uurimisaladel. Töö tulemused näitasid ka seda, et kuigi veetaseme erinevused mõjutasid oluliselt kujunenud taimede liigilist koosseisu ja taimestiku katvust ning seeläbi ka taimedega seonduvaid CO_2 voogusid, siis selle mõju süsiniku ja KHGde bilansile oli 3 aastat peale taastamist väike.

Üle-Euroopaline analüüs näitas, et N_2O vood orgaanilistelt muldadelt on peamiselt seotud veetaset mõjutava inimtegevusega, samal ajal kui klimaatiliste tegurite mõju oli teisese tähtsusega. Euroopa orgaaniliste muldade N_2O kogubilanss oli 149.5 Gg N a^{-1}, millest mahajäetud ja aktiivsed turbatootmisalad moodustasid vaid 0.1 Gg N a^{-1} (Publikatsioon IV – Leppelt *et al.*, 2014). Turbatootmisalade väike panus N_2O bilanssi on peamiselt tingitud nende väikesest pindalast ning seega võib öelda, et nende mõju Euroopa N_2O bilansile on muude maakasutusklassidega (haritavad maad ja rohumaad) võrreldes väike.

Doktoritöö tulemused näitasid, et nii päiderooga taimestamine kui ka turbasamblaga taastamine alandasid mahajäetud turbatootmisalade KHGde bilanssi, mis tähendab, et nende alade summaarne kasvuhooneefekti tekitav mõju oli endiste turbakaevandusaladega võrreldes väiksem. Seega võib kokkuvõtvalt järeldada, et mõlemad meetodid võivad olla sobivad alternatiivseks jääkturbasoode kasutusvõimaluseks. Jääksoode edasise kasutuse planeerimisel tuleb lisaks KHGde bilansile ja atmosfäärsele mõjule arvestada aga ka mitmete muude teguritega ning lõplik valik on paratamatult ala-spetsiifiline. Lisaks on oluline korrastamise ja taastamise järgse pikema-ajalise seire ja mõõtmiste teostamine, et tagada erinevate meetoditega ja erinevatel eesmärkidel tehtud tööde tulemuslikkuse hindamine.

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PUBLICATIONS

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2013	Summer school on 'Measurement Methods in Environmental Biology' (Estonia, Järvselja)

Conference and workshop presentations

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2013	Workshop on 'Net Ecosystem Carbon Balance (NECB) measurements at Fennoscandian mires': Poster presentation 'Carbon balance of an abandoned peat extraction area cultivated with reed canary grass' (Sweden, Bålsta)
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Scholarships for academic training

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Publications

Järveoja J, Peichl M, Maddison M, Soosaar K, Vellak K, Karofeld E, Teemusk A, Mander Ü (201x) Impact of water table level on annual carbon and greenhouse gas balances of a restored peat extraction area. *Biogeosciences* (Submitted)

Järveoja J, Peichl M, Maddison M, Teemusk A, Mander Ü (2015) Full carbon and greenhouse gas balances of fertilized and nonfertilized reed canary grass cultivations on an abandoned peat extraction area in a dry year. *Global Change Biology – Bioenergy*, doi:10.1111/gcbb.12308

Kasak K, Mander Ü, Truu J, Truu M, **Järveoja J,** Maddison M, Teemusk A (2015) Alternative filter material removes phosphorus and mitigates greenhouse gas emission in horizontal subsurface flow filters for wastewater treatment. *Ecological Engineering*, 77: 242–249, doi:10.1016/j.ecoleng.2015.01.038

Leppelt T, Dechow R, Gebbert S, Freibauer A, Lohila A, Augustin J, Drösler M, Fiedler S, Glatzel S, Höper H, **Järveoja J,** Lærke PE, Maljanen M, Mander Ü, Mäkiranta P, Minkkinen K, Ojanen P, Regina K, Strömgren M (2014)

Nitrous oxide emission budgets and land-use-driven hotspots for organic soils in Europe. *Biogeosciences*, 11, 6595–6612, doi:10.5194/bg-11-6595-2014

Järveoja J, Maddison M, Teemusk A, Mander Ü (2014) Kasvuhoonegaaside emissiooni kahandamine Lavassaare jääkturbasoos energiaheina *Phalaris arundinacea* kasvatamisel. Tammiksaar E, Pae T, Mander Ü (Toim.) Publicationes Instituti Geographici Universitatis Tartuensis (342–351), Tartu: Eesti Ülikoolide Kirjastus

Kasak K, **Järveoja J,** Maddison M, Truu M, Mander Ü (2014) Süsiniku- ja lämmastikugaaside vood Prantsuse Guajaana troopilistel turbaaladel. Tammiksaar E, Pae T, Mander Ü (Toim.) Publicationes Instituti Geographici Universitatis Tartuensis (296–306), Tartu: Eesti Ülikoolide Kirjastus

Soosaar K, Mander Ü, Lõhmus K, Uri V, Rannik K, Ostonen I, Uuemaa E, **Järveoja J,** Läänelaid A, Maddison M, Muhel M, Teemusk A (2014) Ökosüsteemi süsinikdioksiidi bilanss, mullahingamine ja netoprimaar-produktsioon Soontaga boreaalses männikus. Tammiksaar E, Pae T, Mander Ü (Toim.) Publicationes Instituti Geographici Universitatis Tartuensis (352–362), Tartu: Eesti Ülikoolide Kirjastus

Järveoja J, Laht J, Maddison M, Soosaar K, Ostonen I, Mander Ü (2013) Mitigation of greenhouse gas emissions on an abandoned peat extraction area by growing reed canary grass: Life-cycle assessment. *Regional Environmental Change*, 13(4): 781–795, doi:10.1007/s10113-012-0355-9

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. *Global Change Biology – Bioenergy*, 4: 462–474, doi:10.1111/j.1757-1707.2011.01138.x

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Haridus

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1993–2005	Kohtla-Järve Järve Gümnaasium

Teenistuskäik

2012–	Tartu Ülikool, Loodus- ja tehnoloogiateaduskond, Ökoloogia ja Maateaduste Instituut, Geograafia osakond; loodusgeograafia ja maastikuökoloogia spetsialist

Erialane enesetäiendamine

2015	Välisvisiit uurimistöö läbiviimiseks Rootsi põllumajandusülikooli (Rootsi, Umeå)
2014	Seminar 'Eddy Covariance training workshop' (Austria, Viin)
2014	Seminar 'ICOS-NEON GHG data training workshop' (Prantsusmaa, Haute-Provence)
2013	Suvekool 'Flux measurement techniques for non CO_2 GHG: methods, sensors, databases and modelling' (Poola, Mierzecin)
2013	Suvekool 'Challenges in measurements of greenhouse gases and their interpretation' (Soome, Hyytiälä)
2013	Suvekool 'Measurement Methods in Environmental Biology' (Eesti, Järvselja)

Konverentsi- ja seminariettekanded

2015	Keskkonnamuutustele kohanemise tippkeskuse (ENVIRON) konverents: Posterettekanne 'Impact of water table level on annual carbon and greenhouse gas balances of a restored peat extraction area' (Eesti, Tartu)
2013	Seminar 'Net Ecosystem Carbon Balance (NECB) measurements at Fennoscandian mires': Posterettekanne 'Carbon balance of an

	abandoned peat extraction area cultivated with reed canary grass' (Rootsi, Bålsta)
2012	European Geosciences Union (EGU) General Assembly: Suuline ettekanne 'Mitigation of greenhouse gas emission on abandoned peatlands by growing reed canary grass' (Austria, Viin)

Stipendiumid

2015	Sihtasutus Archimedes, DoRa Prorammi 6 stipendium – Doktorantide õppe- ja teadustöö välismaal
2015	Maateaduste ja Ökoloogia doktorikool – Kompleksekspeditsioonil osalemine (Maroko)
2014	Euroopa Teadusfond – Välissõidustipendium seminaril osalemiseks
2013	Euroopa Teadusfond – Välissõidustipendium suvekoolis osalemiseks
2013	Maateaduste ja Ökoloogia doktorikool – Kompleksekspeditsioonil osalemine (Prantsuse Guajaana)
2012	Sihtasutus Archimedes, Kristjan Jaagu välissõidustipendium konverentsil osalemiseks

Publikatsioonid

Järveoja J, Peichl M, Maddison M, Soosaar K, Vellak K, Karofeld E, Teemusk A, Mander Ü (201x) Impact of water table level on annual carbon and greenhouse gas balances of a restored peat extraction area. *Biogeosciences* (Submitted)

Järveoja J, Peichl M, Maddison M, Teemusk A, Mander Ü (2015) Full carbon and greenhouse gas balances of fertilized and nonfertilized reed canary grass cultivations on an abandoned peat extraction area in a dry year. *Global Change Biology – Bioenergy*, doi:10.1111/gcbb.12308

Kasak K, Mander Ü, Truu J, Truu M, **Järveoja J,** Maddison M, Teemusk A (2015) Alternative filter material removes phosphorus and mitigates greenhouse gas emission in horizontal subsurface flow filters for wastewater treatment. *Ecological Engineering*, 77: 242–249, doi:10.1016/j.ecoleng.2015.01.038

Leppelt T, Dechow R, Gebbert S, Freibauer A, Lohila A, Augustin J, Drösler M, Fiedler S, Glatzel S, Höper H, **Järveoja J,** Lærke PE, Maljanen M, Mander Ü, Mäkiranta P, Minkkinen K, Ojanen P, Regina K, Strömgren M (2014) Nitrous oxide emission budgets and land-use-driven hotspots for organic soils in Europe. *Biogeosciences*, 11, 6595–6612, doi:10.5194/bg-11-6595-2014

Järveoja J, Maddison M, Teemusk A, Mander Ü (2014) Kasvuhoonegaaside emissiooni kahandamine Lavassaare jääkturbasoos energiaheina *Phalaris arundinacea* kasvatamisel. Tammiksaar E, Pae T, Mander Ü (Toim.) Publicationes Instituti Geographici Universitatis Tartuensis (342–351), Tartu: Eesti Ülikoolide Kirjastus

Kasak K, **Järveoja J,** Maddison M, Truu M, Mander Ü (2014) Süsiniku- ja lämmastikugaaside vood Prantsuse Guajaana troopilistel turbaaladel. Tammiksaar E, Pae T, Mander Ü (Toim.) Publicationes Instituti Geographici Universitatis Tartuensis (296–306), Tartu: Eesti Ülikoolide Kirjastus

Soosaar K, Mander Ü, Lõhmus K, Uri V, Rannik K, Ostonen I, Uuemaa E, **Järveoja J,** Läänelaid A, Maddison M, Muhel M, Teemusk A (2014) Ökosüsteemi süsinikdioksiidi bilanss, mullahingamine ja netoprimaarproduktsioon Soontaga boreaalses männikus. Tammiksaar E, Pae T, Mander Ü (Toim.) Publicationes Instituti Geographici Universitatis Tartuensis (352–362), Tartu: Eesti Ülikoolide Kirjastus

Järveoja J, Laht J, Maddison M, Soosaar K, Ostonen I, Mander Ü (2013) Mitigation of greenhouse gas emissions on an abandoned peat extraction area by growing reed canary grass: Life-cycle assessment. *Regional Environmental Change*, 13(4): 781–795, doi:10.1007/s10113-012-0355-9

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. *Global Change Biology – Bioenergy*, 4: 462–474, doi:10.1111/j.1757-1707.2011.01138.x

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