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Tackling uncertainties of GHG emissions from managed temperate peatlands

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Zusammenfassung

Moore sind wichtige Elemente im globalen Treibhausgas-(THG)-Kreislauf. Die Intensivierung der landwirtschaftlichen Nutzung der Moore durch den Menschen hat den Verlust des Torfes über die letzten Jahrhunderte beschleunigt. Dies hat zu einem weit verbreiteten Moorschwund in Norddeutschland geführt und die ökonmisch sinnvolle Moornutzung fast unmöglich gemacht. Darüber hinaus hat im Zuge des Klimawandels die Zahl der Extremwettererscheinungen zugenommen, was die Belastbarkeit der Ökosysteme an ihre Grenzen stoßen lässt. Zur Vermeidung des Klimawandels wurde in den letzten Jahren eine Vielzahl von alternative Moornutzungsstrategien diskutiert, welche eine standortangepasste Nutzung durch den Menschen nicht kategorisch ausschließen. Die Methoden zur Bestimmung der THG-Bilanzen von Mooren variieren jedoch stark und unterliegen keiner standardisierten Struktur was die Zuverlässigkeit der Abschätzung des Klimaeffekts verschiedener Moornutzungsoptionen verkompliziert. Die zunehmende Witterungsvariabilität, die zahlreichen Managementoptionen und das Fehlen standardisierter Gasmessmethoden sind daher die wahrscheinlichsten Gründe für die großen Unsicherheiten bei der Bestimmung von THG-Emissionen bewirtschafteter Moore der temperaten Klimate. In diesem Kontext behandelt diese Arbeit vier Fragen: (*i*) Ist naturschutzgerechte Grünlandnutzung von drainierten Niedermooren vor dem Hintergrund von Extremwintern und hohen Lachgas-(N_2O)-Emissionen ausreichend um das Klima zu schützen? (*ii*) Ist langfristige Wiedervernässung eine passende Klimaschutzmaßnahme vor dem Hintergrund zunehmender sommerlicher Starkregenereignisse und durch Süßwasserüberstau erhöhte Methan-(CH_4)-Emissionen? (*iii*) Ändert die Ernte aerenchymatischer Röhrichtpflanzen als standortangepasste Moorvegetation die THG-Bilanz von wiedervernässten Niedermooren? (*iv*) Verursachen unterschiedliche methodische Herangehensweisen von Kohlenstoffdioxid-(CO_2)-Austauschstudien divergierende Kohlenstoffbilanzen, die die Vergleiche zwischen den Studien erschweren? Die Messungen dieser Arbeit wurden auf verschiedenen Standorten unter typischer Witterung Nordostdeutschlands durchgeführt: auf einem drainierten Niedermoor unter naturschutzgerechter Grünlandnutzung während des extrem kalten und schneereichen Winters 2009/2010; auf einem wiedervernässten Niedermoor unter ungestörten und extensive genutzten Bedingungen über einen Zeitraum von zwei Jahren (März 2011 bis März 2013); auf einem Mineralboden mit intensiver Schnittnutzung von Futtergrasmischungen über einen Zeitraum von einem Jahr (November 2013 bis November 2014).

Naturschutzgerechte Grünlandnutzung ist keine ausreichende Maßnahme um THG-Emissionen drainierter Niedermoore zu reduzieren, da selbst unter extrem kalter Witterung viel CO_2 und N_2O emittiert wird. Im Gegensatz dazu wird durch langfristige Wiedervernässungsmaßnahmen (> 15 Jahre) die Belastbarkeit von Niedermoorökosystemen bei zunehmenden sommerlichen Starkregenereignissen und damit einhergehendem Süßwasserüberstau ausreichend sein um THG-Emissionen auf das Niveau natürlicher Moorökosysteme zu reduzieren. Obwohl die THG-Emissionen der beiden untersuchten

Niedermoore aufgrund der aufgetretenen, extremen Witterungsereignisse als konservative Schätzungen angesehen werden können, sind diese aufgrund des generellen hydrologischen Zustands der beiden Niedermoore (drainiert/wiedervernässt) klar voneinander zu unterscheiden. Daher ist die Wasserversorgung selbst unter extremer Klimavariabilität von höchster Bedeutung um hohe THG-Emissionen aus Niedermooren zu vermeiden. Die Ernte aerenchymatischer Röhrichtpflanzen führt im Gegensatz dazu nicht zu veränderten THG-Emissionen und ist gegenüber dem Wasserhaushalt und der Witterungsvariabilität zumindest innerhalb der ersten Jahre von geringerer Bedeutung. Große Mengen an THG-Einsparungen werden daher durch die Wiedervernässung selbst erreicht. Nichtsdestotrotz ist der langfristige Effekt wiederholter Biomasseentnahme wiedervernässter Niedermoore mit standortangepasster Vegetation auf THG-Bilanzen nur durch Dauerfeldversuche zu klären. Darüber hinaus entsteht ein hohes Maß an Unsicherheit bei der Bestimmung verlässlicher THG-Bilanzen durch den Mangel an standardisierten Rahmenbedingungen zur Bestimmung des CO_2-Austauschs mit manuellen Hauben. Daher betont diese Arbeit die Notwendigkeit einheitlicher THG-Bestimmungsmethoden sowie Dauerfeldversuche auf Moorböden zu entwickeln und zu etablieren um die Unsicherheiten der THG-Bilanzen, die durch Klimavariabilität, Klimawandel und verschiedene Moornutzungsoptionen entstehen, bewältigen zu können.

Abstract

Peatlands are an important element of the global cycle of greenhouse gases (GHG). The intensification of agricultural land use on peatlands during the last centuries has accelerated peat loss over time which led to the limits of economically reasonable peatland use and to extensive peatland losses in northern Germany. As a consequence of climate change, extreme weather anomalies have also become more frequent pushing the limits of ecosystem resilience. A variety of alternative peatland use options have been discussed as climate change mitigation measures in recent years. However, the methodologies of assessing GHG balances from peatlands vary and are lacking a standardised framework, further complicating reliable estimates of GHG balances of these systems. The combination of increasing climate variability, the numerous management practices and a lack of standardised methods may therefore be responsible for the large uncertainties in GHG emissions from managed temperate peatlands. Against this background, this thesis tackles four major questions: (*i*) Is nature conservation management of a drained fen a sufficient climate protection measure in the light of extreme winters and nitrous oxide (N_2O) emissions? (*ii*) Is long-term rewetting of fens a suitable measure for climate protection in the light of increased methane (CH_4) emissions and summer freshwater flooding? (*iii*) Does harvest of emergent macrophytes as paludiculture crops alter the GHG balance of a rewetted temperate fen? (*iv*) Do methodological differences of carbon dioxide (CO_2) exchange studies with manual chambers produce diverging carbon (C) balances that further complicate inter-study comparisons? Measurements were conducted in north-eastern Germany: (a) on a drained fen under nature conservation management during the extreme winter 2009/2010; (b) on a rewetted fen under undisturbed and harvested conditions over two years (March 2011 to March 2013); (c) on a mineral soil with a high frequence of harvesting forage crops over one year (November 2013 to November 2014).

Nature conservation management is not sufficient to reduce GHG emissions from a drained fen due to relatively high CO_2 and N_2O emissions even under extremely cold conditions. In contrast, long-term rewetting (> 15 years) likely reduces GHG emissions from formerly drained fens and leads to sufficient ecosystem resilience even under extreme summer rain and freshwater flooding. Although the GHG emissions of both peatlands can be seen as conservative estimates because they were obtained during climatic extreme events, differences in GHG balances are most likely explained by their different hydrological state (drained/rewetted). Therefore, even under extreme climate variability, water supply of a peatland is of major importance to mitigate high GHG emissions. In contrast, the short-term effect of harvesting emergent macrophytes from rewetted fens is of minor importance compared to water supply and climate variability. Large amounts of GHG emissions may hence be avoided through the rewetting itself. Nevertheless, assessing the long-term effect of repeated biomass removal requires long-term field trials. In addition, the lack of a standardised framework adds another source of uncertainty to the determination of GHG emissions with manual chambers. Therefore, this

thesis emphasizes the need for standardised methods determining GHG emissions and long-term field trials to tackle uncertainties of GHG emssions of temperate fens arising from climate variability, climate change and various management options.

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List of publications and author contributions

I. Winter emissions of carbon dioxide, methane and nitrous oxide from a minerotrophic fen under nature conservation management in north-east Germany, Chapter 2 (Huth et al. 2012)

Huth, V	Study design, field work, data analysis, writing, editing
Jurasinski, G	Study design, data analysis, editing
Glatzel, S	Study design, editing

Huth V, Jurasinski G, Glatzel S (2012) Winter emissions of carbon dioxide, methane and nitrous oxide from a minerotrophic fen under nature conservation management in north-east Germany. *Mires and Peat*, **10**, Article 04, 1–13.

II. The effect of an exceptionally wet summer on methane effluxes from a 15-year re-wetted fen in north-east Germany, Chapter 3 (Huth et al. 2013)

Huth, V	Study design, field work, data analysis, writing, editing
Günther, A	Study design, field work, editing
Jurasinski, G	Study design, data analysis, editing
Glatzel, S	Study design, editing

Huth V, Günther A, Jurasinski G & Glatzel S (2013) The effect of an exceptionally wet summer on methane effluxes from a 15-year re-wetted fen in north-east Germany. *Mires and Peat*, **13**, Article 02, 1–7.

III. The effect of biomass harvesting on greenhouse gas emissions from a rewetted temperate fen, Chapter 4 (Günther et al. 2015)

Günther, A	Study design, field work, data analysis, writing, editing
Huth, V	Study design, field work, data analysis, writing, editing
Jurasinski, G	Study design, editing
Glatzel S	Study design, editing

Günther A, Huth V, Jurasinski G & Glatzel S (2015) The effect of biomass harvesting on greenhouse gas emissions from a rewetted temperate fen. *Global Change Biology Bioenergy*, **7**, 1092–1106.

IV. Divergent NEE balances from manual chamber CO_2 fluxes by different measurement and modelling approaches, Chapter 5 (Huth et al., submitted)

Huth, V	Study design, field work, data analysis, writing, editing
Vaidya, S	Study design, field work, data analysis
Hoffmann, M	Study design, data analysis, editing
Jurisch, N	Study design, field work, editing
Günther, A	Data analysis, editing
Gundlach, L	Study design
Hagemann, U	Study design, editing
Elsgaard, L	Editing
Augustin, J	Study design, editing

Huth V, Vaidya S, Hoffmann M, Jurisch N, Günther A, Gundlach L, Hagemann U, Elsgaard L & Augustin J (submitted) Divergent NEE balances from manual chamber CO_2 fluxes by different measurement and modelling approaches. *Agricultural and Forest Meteorology*.

1 Introduction

1.1 BACKGROUND

1.1.1 Climate change

The lifetime of men is only the blink of an eye compared to the timescale on which climate change happens. Yet, we tend to attribute extreme weather anomalies to the increase in global temperature since the pre-industrial era (Crowley 2000), although a single event does not make a statistical trend. However, with increasing global temperatures the occurrence of extreme weather anomalies, especially hot summer anomalies, has become more likely (Hansen et al. 2012) showing, that there is some truth in our perception of climate change and that the current climate change happens so rapidly that mankind is able to experience it. In fact, the speed and magnitude of change in temperatures of the northern hemisphere during the last 50 years is unique during the past 1300 years (Jansen et al. 2007). While 41–64 % of the temperature variability of the pre-industrial era may be explained by natural factors such as solar activity and volcanism these fail to explain more than 25 % of the temperature increase of the 20th century (Crowley 2000). Instead, the rapid concentration increase of the long-lived greenhouse gases (GHG) carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) since the industrial revolution and their estimated impact on radiative forcing offers a good fit for the temperature increase of the 20th century (Crowley 2000) making an anthropogenic contribution very likely (Hegerl et al. 2007, Jansen et al. 2007). In fact, with high confidence climate change has recently been recognized to have more negative than positive impact on crop yields and with very high confidence weather extremes related to climate change revealed vulnerability of ecosystems and human systems to climate variability (Field et al. 2014)

Imagine, the earth is a pot, the sun is a cooker, our climate is the boiling water and the greenhouse gases are the lid. The more they close the system, the more energy is stored inside and the more likely big bubbles occur. These are the extreme weather anomalies we perceive to have become more frequent.

1.1.2 Peatland/Ice-Age-Hypothesis

So, to understand what makes man-made climate change different from natural climate change a short introduction to the glacial/interglacial periods are needed here. Evidence for a glaciation already prior to the Cambrian (500 mill. years ago) was given by Harland (1964). The Milankovitch cycles (Milankovitch 1930, Meinardus 1944) describe the periodic change of the Earth's orbital parameters eccentricity (100 kyr), tilt (41 kyr) and precession (21 kyr) and their influence on the radiative budget of the Earth's surface. These changes are the common explanation for the glacial/interglacial periods.

Fitting these periods best, the periodic change of the Earth's orbital eccentricity has been recognized as the most important parameter. This 100 kyr interval is evident in the Vostok ice core showing the periodic cooling and heating of the Earth during the last 160 kyr (Jouzel et al. 1987, Figure 1.1).

However, a common criticism of the Milankovitch cycles is that they are too weak to explain the amplitude of the temperature changes between glacial/interglacial periods (Rind et al. 1989, Berger et al. 2005, Franzén et al 2012). Therefore, other mechanisms must amplify the periodic heating/cooling. For this reason Klinger (1991) and Franzén (1994) proposed an alternative to the common theory of the Milankovitch cycles which is where peatlands come into play. These authors hypothesized that the growth of temperate/boreal peatlands over the time of the warm interglacial sequesters so much carbon (C), that at a critical low value of CO_2 concentration (which was around 240 ppm at the beginning of the last Weichselian Ice Age) induces a new glacial period, whereas the C, which is released thereafter through erosion and decomposition induces a new interglacial.

That there is a relation of temperature development and atmospheric CO_2 and CH_4 concentration is also evident from the Vostok ice core (Barnola et al. 1987, Chappellaz et al. 1990, Figure1.1). In addition, large parts of the Earth's land surface have moved north of 45° N(S) during the last 400 mill. years (Donn & Shaw 1977) which fits well to the first evidences of glaciations (Harland 1964) and which is a fundamental factor in temperate/boreal peatland genesis. Therefore, without anthropogenic effects, the potential temperate/boreal peatland CO_2 sink might reach up to 2 ppm yr^{-1} during the current interglacial (Franzén et al. 2012). With the anthropogenic release of GHGs and the disturbance of the natural temperate/boreal peatlands however, we might just be at the beginning of an incomparable interglacial epoch (Franzén et al. 1994).

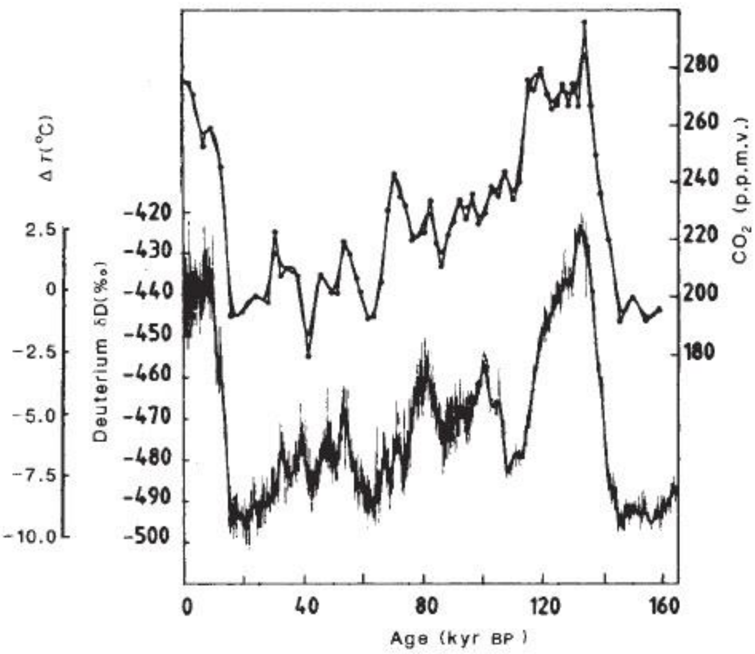

Figure 1.1: CO_2 concentrations ('best estimates') and smoothed values (spline function) in p.p.m.v. plotted against age in the Vostok record (upper curves) and atmospheric temperature change derived

(Jouzel et al. 1987) from the isotopic profile (lower curve). The deuterium scale corresponds to values after correction for deuterium changes of oceanic water. Adapted from Barnola et al. (1987).

1.2 UNCERTAINTY FACTORS OF C BALANCES AND GHGS FROM TEMPERATE PEATLANDS

Land-use intensification has been recognized as one of the anthropogenic drivers that have descreased global soil and biomass C stocks leading to increased GHG emissions (Leifeld et al. 2014, Le Quéré et al. 2015). However, up to today the quantification of GHG balances for peatlands is characterized by high uncertainties (Kirschke et al. 2013) mainly resulting from a variety of land uses, an increased climate variability, and methodological deficits (Luo et al. 2015). The following chapters therefore outline temperate peatland use history (Chapter 1.2.1), phenomena of climate change in north-east Germany (Chapter 1.2.2), climate change mitigation strategies (Chapter 1.2.3), and recent methodological advances in determing C balances (Chapter 1.2.4). Against this background, the research objectives will be extracted (Chapter 1.3)

1.2.1 Anthropogenic disturbances and degradation of temperate peatlands

1.2.1.1 History

Since the end of the last glacial mankind has been an integrative part of the northern hemisphere and central Europe which is evident from settlements neighbouring peatlands already during the Paleolithic period (Succow & Jeschke 1986, Hayen 1990). During the Neolithic Revolution when growing settlements and an agricultural lifestyle initiated deforestation of larger landscapes, anthropogenic disturbances of temperate peatlands occurred both peat forming as a consequence of an altered evapotranspiration especially in north-western Europe as well as peat exploiting through the first signs of peat extraction (Succow & Jeschke 1986). Alongside extensive pastures and peat fuel, north-east German paludification mires have been extracted for e. g. bog iron and lime by Germanic tribes already since the 4th century BC (Succow & Jescke 1986, Schopp-Guth 1999, Succow 2001). An intensification of drainage, peat extraction and grassland use systems since the industrial revolution led to the most significant alterations (Schopp-Guth 1999, Succow 2001, Figure 1.2), an acceleration of C-loss rates of temperate peatlands (Leifeld et al. 2014), and an extensive loss of peatland ecosystems (Couwenberg & Joosten 2001). In the following the main anthropogenic disturbances of temperate peatlands will therefore be introduced.

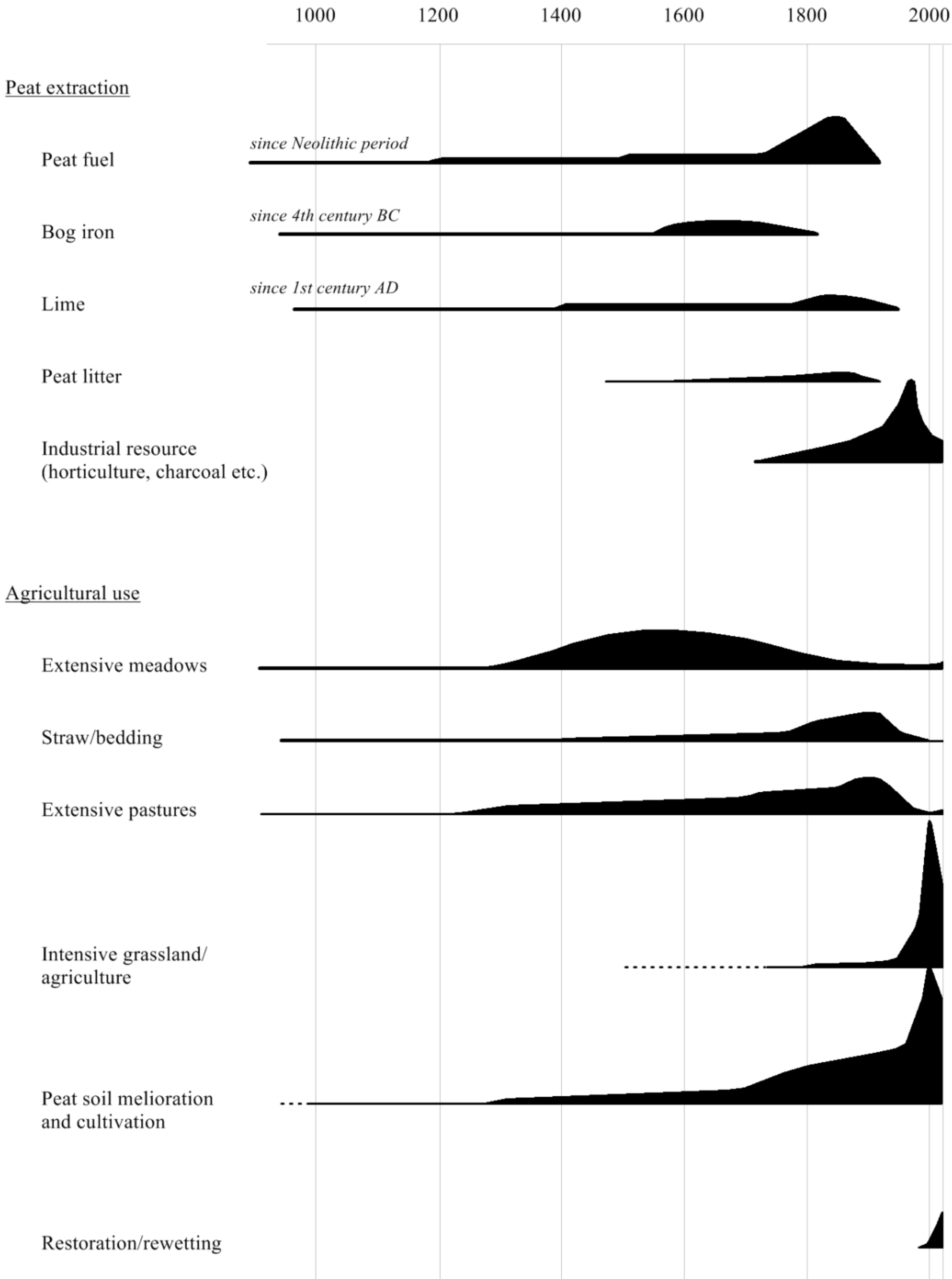

Figure 1.2: Historical development of significant types of peatland use in Germany over the last millenium. Black areas indicate their relative importance over time and illustrate an estimation of area of peat soil exploitation through the respective land use type. Redrawn, adapted and extended from Schopp-Guth (1999).

1.2.1.2 Pastures and meadows

Extensive grassland use as pastures or summer-cut-meadow is believed to have been a consequence of settlements close to peatlands already during the Neolithic period (Schopp-Guth 1999). In north-east Germany a sparse pasture use of sedge reeds is likely from the 6th century AD (Succow 2001). First drainages allowed wet meadows and pastures on minerotrophic fens during the 13th century (Schopp-Guth 1999). As a consequence of peasant clearance and the Thirty Years' War grassland use was widely abandoned during the 16th/17th century. The 1765 edict for 'Moorkultur' by Friedrich II re-installed peat grassland use in north-east Germany (Succow 2001). 18th century grassland use mainly consisted of wet meadows, changing to grassland communities characterized by *Molinia* spp. SCHRANK due to nutrient extraction until mineral fertilization allowed two or more harvests, high quality grassland sowing was developed, and yields of 80 dt ha^{-1} and more became possible (Succow 2001). Therefore, extensive grassland use has become relatively scarce in north-east Germany until the early 1990s (< 12 % in Mecklenburg Western Pomerania, Anonymous 2009). With the abandonment of intensive grassland use and the fulfillment of nature protection goals however, extensive peat grassland use has become more frequent by 2000 (up to 20 %) but is decreasing again (Anonymous 2009).

1.2.1.3 Peat extraction

Peat extraction for peat fuel is likely to have taken place during the Neolithic period (Schopp-Guth 1999). Pliny the Elder described peat extraction for fuel by Germanic tribes of the North Sea marshes 2000 years ago when it has been the alternative to wood during times of scarce forests, and therefore being a major peat use type mainly in north-western Europe (von Bülow 1925). In north-eastern Germany peat was extracted for bog iron and lime usually at the bottom of the peat layer since about 2000 years (Succow & Jeschke 1986). An occasional extraction for peat fuel combustion began around the 13th century. The saline of (Bad) Sülze was fired with peat from 1553 on. Peat fuel had been the major energy supply during the 19th century until it was replaced by brown coal at the end of the 19th century. Peat fuel is now prohibited by law in Germany (Succow 2001). Peat as an industrial resource for e. g. horticulture has become the most important reason for peat extraction in the 20th century in north-western Germany, where 25 % of the peat bogs still are under extraction (Joosten & Couwenberg 2001).

1.2.1.4 Drainage

The first peatland drainage measures were often run through the Cistercian Order during the 13th century (Schopp-Guth 1999). As a consequence of peasant clearance and the Thirty Years' War they were lost in north-east Germany during the 16th/17th century. At the same time the so-called 'Fehnkultur' spread from Holland to the extensive peat bogs of north-western Germany, where navigable canals were built leading to deep drainage. In the following, peat was extracted and the peat residues were mixed with sand allowing agricultural land use (Succow & Jeschke 1986, Schopp-Guth

1999). In 1718, drainage works in the minerotrophic fen ‘Havelländisches Luch’ began followed by an edict for ‘Moorkultur’ enacted by Friedrich II in 1765 re-installing peatland management measures in north-east Germany (Succow 2001). From ca. 1965 on, intensive peatland melioration works began fundamentally altering the hydrological state of ca. 99 % of north-east German peatlands (Arbeitsgruppe Moorboden 1965, 1967, Succow 2001). As a consequence of a wide abandonment of intensive agricultural and grassland use, extensive peatland rewetting measures have taken place since the 1990s (e. g. > 10 % of peatland area in Mecklenburg Western-Pomerania between 2000 and 2008, Anonymous 2009).

1.2.1.5 Settlements

Due to the difficulties in crossing, peatlands have offered a natural protection for human settlements throughout history. Settlements close to peatlands are evident from residues of domestic animals and hunting/fishing tools already during the Paleo- and Mesolithic periods (Succow & Jeschke 1986, Hayen 1990). During the Neolithic colonization a widespread wooden road network especially on north-west German peat bogs can be proven through findings in the respective peat layers (Hayen 1990). These networks had to be restored over and over again due to peat bog growth, but were used until the Middle Ages. When the so-called ‘Fehnkultur’ evolved from Holland to the extensive peat bogs of north-western Germany a new era of peat colonization started (Succow & Jeschke 1986), fundamentally altering temperate peatlands of the north west through significant subsidence rates of peat (Schothorst 1977). Nowadays, a large part of north-west German peatlands is under settlement use.

1.2.1.6 Forestry

It is likely that trees have been cut from virgin, forested peatlands during times of forest shortage of the Middle Ages. In treeless areas of Ireland, blanket bogs have even been exploited for wood fossils until the 19th century, which is evident from the extensive Neolithic site named ‘Céide Fields’ in Co. Mayo. However, even in Finland –the country with the largest area of forestry on drained peatlands in the world– very little tree harvesting has been carried out on virgin peatlands (Päivänen 1996). In contrast, draining peatlands to increase tree growth is known since the mid-19th century and has become a significant land use type in the northern boreal countries (Päivänen 1996). In Germany however, forestry only plays a minor role with ca. 10 % of the drained peatlands (Päivänen 1996, Joosten & Couwernberg 2001).

1.2.1.7 Intensive grassland use

Intensive grassland use has been a major land use type of minerotrophic fens formed by paludification, especially in north-eastern Germany (Okruszko 1996). When mineral fertilization allowed two or more harvests, high quality grassland sowing was developed, and highly efficient drainages were built in the 1960s hay yields of 120–140 dt ha^{-1} became possible (Succow & Jeschke 1986, Succow 2001). Due to the replacement of herbaceous plant communities within a short time and the need for re-ploughing and re-sowing, the fen-peat soils became heavily humified within a couple of decades (Okruszko 1996). Until the 1990s this type of peatland use was the most frequent, occupying more than 50 % of the north-eastern German peatlands (Lenschow 2001, Anonymous 2009). Due to peat subsidence and C loss this land use type has become inefficient and is more and more abandoned especially on the minerotrophic fens of north-eastern Germany with now about 30 % of the north-eastern German peatlands (Anonymous 2009).

1.2.1.8 Cropland

Transforming peatlands into arable soils has mainly been a practice of the large peat bog areas in the Netherlands and north-western Germany (Okruszko 1996). Alongside the above introduced 'Fehnkultur' (see Chapter 1.2.1.4 'Drainage'), first attempts at agricultural land use were made by peat bog surface burning ('Moorbrandkultur') with a buckwheat-buckwheat-oat-pickpurse-rye crop rotation and a 30-year-fallow in the late Middle Ages until it was prohibited by law in 1923 in Germany (Succow & Jeschke 1986, Okruszko 1996). Due to poor short-term benefits of the 'Moorbrandkultur' practices to supply the peat bogs with additional nutrients were soon developed. These involved spraying sand, clay (usually from the mineral ground of the peat profile), and organic fertilisers (mainly manure) on to the peat surface. This 'Sandmischkultur' has been so successful, that croplands on peat soils are still in use today (17 % of peatlands in Lower Saxony, Flessa 2012, oral presentation; 11 % of peatlands in Mecklenburg-Western Pomerania, Lenschow 2001, Anonymous 2009), varied and intensified according to the agro-technical development of the 20th century and usually cropped with cereals or maize (Succow & Jeschke 1986, Okruzsko 1996).

1.2.2 Phenomena of climate change in north-east Germany

A commonly used scenario to predict climate change assumes a future world of rapid economic and demographic growth peaking in the mid of the 21st century aligned with the introduction of new and more efficient technologies where the energy system is balanced across all energy sources (A1B, Carter et al. 2001). Climate predictions using the regional climate model WETTREG2010 based on scenario A1B for north-east Germany predict a mean annual temperature increase of more than 3 °C compared to the reference period (1971–2000) until the end of the 21st century with emphasis of warmer winters and hotter summers (Figure 1.3, Bauwe et al. 2015). Annual precipitation sums are predicted to remain similar with a shift from summer to winter precipitation and an increasing drought stress due to the temperature increase (Figure 1.3, Bauwe et al. 2015).

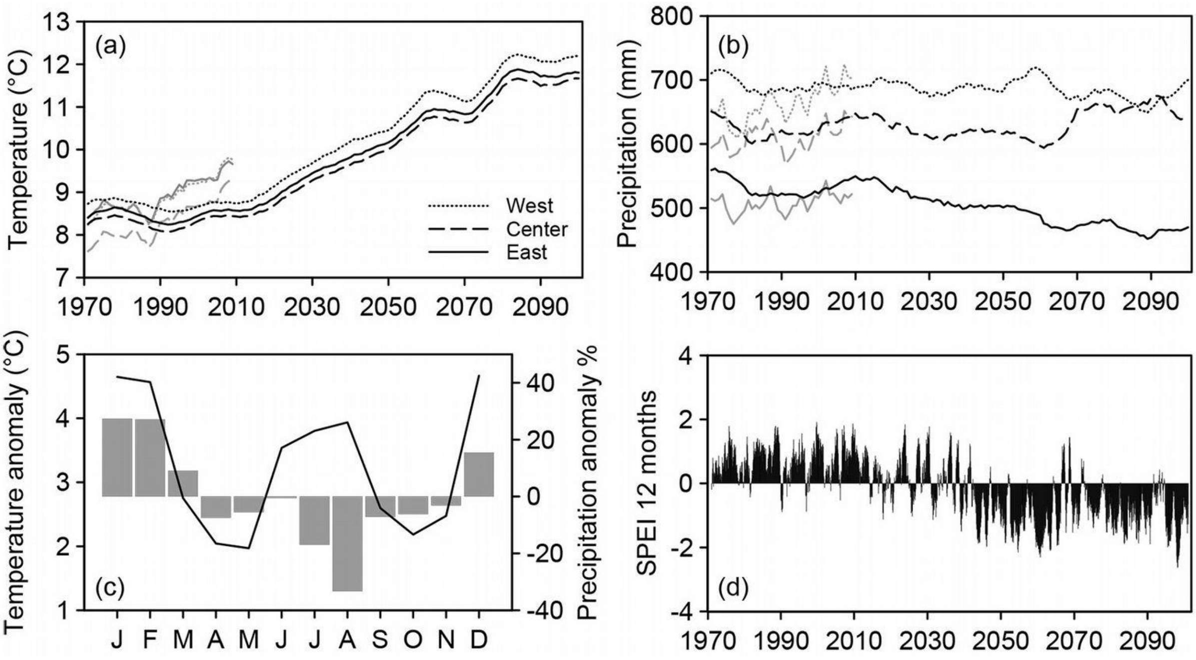

Figure 1.3: Climate projections according to the A1B emission scenario (balanced emphasis on all energy sources, Carter et al. 2001) of the regional climate model WETTREG2010 for the period 1971–2100. (a) Mean annual temperature and (b) annual precipitation with corresponding observed values for the period 1971–2010 (gray lines). (c) Monthly deviation of temperature and precipitation in the distant future (2071–2100) from the reference period (1971–2000) averaged over the study sites. (d) Twelve-months standardized precipitation-evaporation index (SPEI) for the central Mecklenburg-Western Pomerania. Adapted from Bauwe et al. (2015).

The phenomenon behind increasing global temperatures is the increasing likelihood of the occurrence of extreme weather anomalies (Hansen et al. 2012). While the increase of the frequency, duration and intensity of heat waves over Europe can be robustly predicted, other predictions such as the change in annual precipitation sums in northern Europe are of higher uncertainty (Beniston et al. 2007). The mean annual temperature in north-east Germany has increased especially over the last 30 years whereas a significant change in annual sums of precipitation is absent (Figure 1.4). However, this does not contradict stronger precipitation events, since longer lasting heat waves would need to be compensated with stronger rain fall during comparably shorter times. Therefore, extreme weather anomalies are among the highest uncertainty factors for a robust determination of the GHG emissions of managed temperate peatlands and will very likely become even more important in the future.

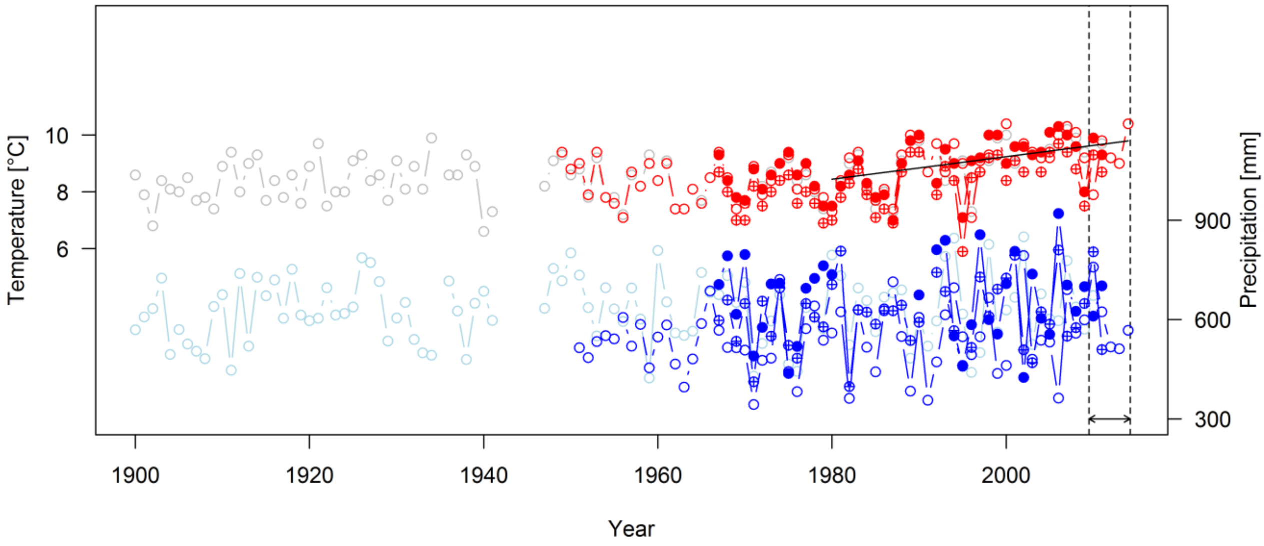

Figure 1.4: Long-term annual mean temperatures and annual sums of precipitation for the north-east german climate stations 'Schwerin' (1900–2012, grey/lightblue circles), 'Müncheberg' (1949–2015, red/blue circles), 'Boizenburg' (1967–2012, red/blue dots), and 'Groß Lüsewitz' (1967–2012, red/blue crossed circles) covering the study region of this thesis. Dashed lines mark the overall study period. Black line shows significant temperature increase in 'Müncheberg' from 1980–2015 at $P > 0.01$.

1.2.3 Climate change mitigation strategies

1.2.3.1 Nature conservation management

Nature conservation management has been implemented in the early 1990s when large areas of former intensively used peatlands became uneconomic. The initial intention of this type of land management has been to fulfil the requirements of the EU Habitats Directive and Birds Directive forming the EU wide habitat protection network "Natura 2000", with emphasis on habitats having their main distribution in Europe. Nature conservation management includes abandonment of mineral fertilisation and a reduced cutting frequency or summer meadow usually aligned with the needs of the protected species. More than 50.000 ha of the peatland area in north-east Germany have therefore been under nature conservation management by 1996 (derived from Roth et al. 2001), slowly decreasing since then with ca. 24.000 ha in 2010 (Anonymous 2011). Although GHG emission inventories have not been considered for this type of peatland management in the first place drained eutrophic fens under agricultural use have been seen as sites with a higher potential of large CO_2 and N_2O emissions than drained but unfertilised nutrient poor sites (Regina et al. 1996). Due to a lower management intensity and in the following a generally lower microbial activity, this type of peatland management may result in lowered GHG emissions and therefore contribute to climate change mitigation.

1.2.3.2 Rewetting

The focus of nature-conservation management of peatlands has been the abandonment of intensive agriculture. Therefore their intensive water management had often remained unchanged (Roth et al. 2001). Since large parts of the peat body undergo strong mineralisation and humification during drainage (Mundel 1976, Roth et al. 2001, Leifeld et al. 2014), effective abiotic peatland conservation to prevent peat loss is in need of a year-round water regime close to the ground surface and may not be reached with the abandonment of intensive agriculture and nature conservation management alone. By the early 1990s less than 3 % of the peatlands of Mecklenburg-Western Pomerania were classified as near-natural (Lenschow 2001), in Germany just about 1 % (Couwenberg & Joosten 2001). In addition, peatland habitats having their main distribution in Europe such as alder carrs or reed stands may only be protected through the restoration of the typical water regime re-creating river basin and flood retention areas as intended by the Water Framework Directive. Therefore peatland rewetting especially of thick fen mires has become an essential part in terms of resource protection from the mid-1990s on (Erwin 2009). By 2008 nearly 40.000 ha (ca. 13 %) of the north-eastern peatlands have been rewetted now being classified as “proper C sinks” (Anonymous 2009).

1.2.3.3 Paludiculture

The goal of peat body protection through water levels close to ground surface excludes conventional meadow and pasture use since rewetted peat soils are weak and therefore not trafficable with conventional agro-engines (Roth et al. 2001). Options of adding value from peatlands in rural areas may be found in historical peatland use. Minerotrophic fens of the southern Baltic region are naturally climaxed with extensive reed stands traditionally used for thatched roof tops. Other options of reed use may be heat isolation mats as frost or wind protection, or as a building material such as lightweight panels etc. (Rodewald-Rodescu 1974). Recent studies also suggest that reeds have potential as an energy source for combustion or biofuel with calorific values of ca. 14 MJ kg^{-1} for *Phragmites* (Köbbing et al. 2013) or briquettes/pellets with ca. 18 MJ kg^{-1} for *Typha* (Cicek et al. 2006). Therefore, paludiculture (lat. “palus” – swamp, marsh), the cultivation of wet soils, has recently been acknowledged as a land use option on rewetted organic soils in the IPCC Wetlands supplement (Blain et al. 2014).

1.2.4 Recent methodological advances of the manual closed chamber method

Manual closed chambers typically are used for plant, soil or treatment comparisons of GHG emissions on small scales and, in principle, allow for analysis of spatial variability of gas fluxes (Rochette et al. 1991, Davidson et al. 2002). For this thesis, the manual closed chamber method is without any alternative since it is the only chamber based method that is flexible, adaptable, time- and cost-efficient enough to answer extensive treatment comparisons (e.g. differing hydrological regimes, differing peatland vegetation, differing management options) and their effect on GHG emissions of typically managed peatlands in north-east Germany.

However, the comparably few field measurements that are achieved with manual closed chamber systems (as a result of weekly to monthly measurement intervals) are often critized to produce large temporal gaps usually representing > 99 % of the study period. In terms of CO_2 exchange which usually constitutes the main compound of the C cycle of managed peatlands, temporal dynamics may be reconstructed through modelling ecosystem respiration (R_{ECO}) and gross primary production (GPP). However, due to the large temporal gaps, modelling CO_2 exchange from manual closed chamber measurements is the most challenging gap-filling scenario (Gomez-Casanovas et al. 2013). In addition, several deductive or empirical approaches of gap filling have been used in different studies, and data acquisition to obtain the primary data of R_{ECO} and GPP also differed (e.g. Whiting et al. 1992, Bubier et al. 1998, Beetz et al. 2013, Günther et al. 2015). Where no consensus exists on a single standardised framework, a literature survey shows that at least two distinct approaches were commonly applied in the past, which differ fundamentally in at least three methodological levels related to (*i*) the CO_2 exchange measurements, (*ii*) the R_{ECO} data pooling, and (*iii*) the empirical GPP modelling.

In the older approach, CO_2 fluxes are measured under mid-day conditions (typically 10 a.m. to 2 p.m.) with the use of shading to obtain responses to low PAR conditions (mid-day measurements). This method is often used in combination with estimating GPP fluxes directly (direct GPP) by subtracting opaque chamber R_{ECO} fluxes from preceding transparent (or shaded) chamber fluxes representing the net ecosystem exchange (NEE) of CO_2 at different PAR (Whiting et al. 1992, Carroll & Crill 1997, Elsgaard et al. 2012). For R_{ECO} gap filling, relationships between R_{ECO} and temperature are typically derived by pooling and analysing R_{ECO} data over meterological or plant physiological seasons (Drösler 2005) or even the entire study period (Alm et al. 1997, Yli-Petäys 2007, Elsgaard et al. 2012, season-wise R_{ECO} modelling). Since R_{ECO} is both soil and plant dependent, R_{ECO} modelling parameters may change during different plant phenological stages, especially before and after harvest events. Therefore, the inclusion of vegetation proxies (Burrows et al. 2005, Kandel et al. 2013) or plant-dependent dynamic model parameters over the course of the study (Reichstein et al. 2005) has been suggested to improve seasonal R_{ECO} modelling.

A very recent methodological advance of the manual chamber studies consists of a modified approach for data acquisition and temporal gap filling (Beetz et al. 2013, Leiber-Sauheitl et al. 2014, Hoffmann et al. 2015). Following this methodology, CO_2 fluxes are measured over the diurnal range of PAR and air/soil temperatures during the time from sunrise to early afternoon (sunrise measurements). The subsequent modelling relies on empirical R_{ECO} models that are based on data from individual campaigns (campaign-wise modelling) to account for changing plant phenological stages. GPP fluxes are obtained by substracting modelled R_{ECO} values from the NEE measurements (indirect GPP) rather than using the proximate measured R_{ECO} fluxes. Compared with the mid-day measurement approach, the sunrise approach is associated with a workload approximately twice as high due to the increased number of measurements (Beetz et al. 2013, Leiber-Sauheitl et al. 2014, Hoffmann et al. 2015). The idea behind the sunrise measuements is to minimize the possible bias from favourable diurnal measurement conditions where CO_2 fluxes at low PAR are measured under typical air/soil temperatures. Yet, up to today it remained untested if systematic differences in CO_2 balances arise from studies with these three profound methodological differences.

1.3 AIMS AND STRUCTURE

The intensification of agricultural land use on peatlands has accelerated the peat loss over time (Leifeld et al. 2014). This acceleration led to the limits of economically reasonable peatland use and to extensive peatland losses in northern Germany (ca. 10–20 %, Joosten & Couwenberg 2001). In addition, extreme weather anomalies have become more frequent pushing the limits of ecosystem resilience (Hansen et al. 2012). As a part of climate change mitigation and adaptation strategies a variety of alternative peatland use options have been discussed in recent years (Erwin 2009). Both increasing climate variability and the numerous management practices may therefore be responsible for the large uncertainties in e. g. CH_4 emissions from peatlands (Kirschke et al. 2013). Therefore, in the chapters 2–4 this work aims at giving insight of alternative land use options on peatlands from a climate perspective. Using fortnightly GHG data from a drained minerotrophic fen under nature conservation management during the extraordinarily cold winter 2009/2010 and from a rewetted minerotrophic fen covering the extraordinarily wet summer 2011, the impacts of extreme weather anomalies are estimated. In addition, different types of emergent macrophytes possibly suitable for paludiculture (see Chapter 1.2.3.3) were harvested testing its effect on the GHG balance of a rewetted fen. The need for an estimation of the effect of land management types and intensity on GHG emissions from temperate peatlands has recently become more attention (e.g. Schrier-Uijl et al. 2010b, Drösler et al. 2013, Beetz et al. 2013, Renou-Wilson et al. 2014). This thesis therefore has the following objectives:

- to evaluate, if nature conservation management of fens is suitable in terms of climate protection in the light of extreme winters and N_2O emissions (Chapter 2, Huth et al. 2012)
- to estimate the effect of long-term rewetting of a fen on climate protection in the light of increased CH_4 emissions and summer freshwater flooding (Chapter 3, Huth et al. 2013)
- to examine whether removing emergent macrophytes as paludiculture crops alter its GHG balance with emphasis on CO_2 exchange and C balance (Chapter 4, Günther et al. 2015)

Other sources of uncertainty derive from numerous methodological approaches of GHG exchange estimation which may vary even within relatively simple methods such as the manual closed chamber method. Since the above used manual chamber approach (Günther et al. 2015), that is relatively new in terms of CO_2 data collection and modelling and has been published for the first time by Beetz et al.

(2013) standardization of these methods is a prevailing topic (Hoffmann et al. 2015). Therefore, Chapter 5 presents a methodological study that has been carried out helping to evaluate the robustness of the methods used in Chapter 4 in comparison to balance estimation of older studies (e.g. Alm et al. 1997, Bubier et al. 1998). Its objective is:

- to compare fundamental differences of CO_2 exchange studies, namely mid-day vs. sunrise chamber measurements, campaign-wise vs. seasonal-wise vs. cluster-wise ecosystem respiration modelling, and direct vs. indirect GPP estimation (Chapter 5, Huth et al., submitted).

In Chapter 6 the results derived in the preceding chapters will be synthesized, its weaknesses discussed, concepts for future GHG studies be given and its implications on site-adapted peatland management in terms of GHG balances will be shown.

2 Winter emissions of carbon dioxide, methane and nitrous oxide from a minerotrophic fen under nature conservation management in north-east Germany[1]

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SUMMARY

Drained peatlands are known to be important sources of carbon dioxide (CO_2) and nitrous oxide (N_2O). While CO_2 emissions occur mainly during the growing season, large N_2O emissions may occur during the non-growing season as well. Peatland re-wetting may be an effective measure to prevent those emissions. However, recent research shows that re-wetted peatlands may release large amounts of methane (CH_4) during the years immediately after re-wetting whereas abandonment of intensive grassland on drained peat soils possibly leads to low nutrient supply and thus to small greenhouse gas (GHG) emissions. Here we examine the role of extensification practices (such as abandonment of mineral fertilisation, reduced cutting frequency and a cattle-free winter period) on GHG emissions from a temperate peatland during winter. From November 2009 to March 2010 GHG measurements were made on a minerotrophic fen five years after intensive grassland use was abandoned. During the measurement period CO_2 and N_2O emissions amounted to 4.4 t ha^{-1} and 2.6 t ha^{-1} CO_2-equivalent, whilst CH_4 emissions were negligible. Altogether the site emitted 7 t ha^{-1} CO_2-equivalent, of which 37 % was N_2O, even though the winter 2009/2010 was extraordinarily cold. Thus, extensification of grassland use alone may not be sufficient to reduce GHG emissions from temperate peatlands.

KEY WORDS

peatland, greenhouse gas emissions, extensive grassland

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2.1 INTRODUCTION

The greenhouse gases (GHG) carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) contribute 63 %, 18 % and 6 % respectively to the global anthropogenic radiative forcing (Forster et al. 2007). In addition, N_2O plays an important role in the stratospheric ozone chemistry because it is an important source for nitric oxide (NO) and nitrogen dioxide (NO_2) in the upper atmosphere (Crutzen 1970). Owing to its relative stability it will remain the major ozone-depleting substance throughout the 21st century (Ravishankara et al. 2009).

Peatlands are known to be an important element of the global GHG cycles (Frolking et al. 2006). Although covering only 3 % of the world's land-surface area (Lappalainen 1996), they store 20 % to 30 % of the soils' C and N reserves (Martikainen et al. 1993). Natural peatlands are regarded as long-term sinks of C, converting atmospheric CO_2 to growing peat whilst emitting significant amounts of CH_4. Their net climatic impact is estimated to be a slight warming (if the effects of CH_4 emissions exceed those of carbon sequestration) or a slight cooling (if sequestration exceeds CH_4 emission), and depend on the time since peatland formation (Frolking et al. 2006). In contrast, drained peatlands act as a source of carbon and nitrogen, emitting CO_2 and N_2O from decomposing peat. Their net climate impact is a strong warming, with CO_2 effluxes up to 50 t ha^{-1} a^{-1} and N_2O effluxes up to 60 kg ha^{-1} a^{-1} (Couwenberg et al. 2011). Therefore, recently, re-wetting has been used as a measure to restore the peatlands' function as a C and N sink, whilst also having high value in terms of nature conservation. However, associated studies indicate, that flooding of eutrophic drained fens may cause strong CH_4 emission peaks, possibly counteracting the reduction of CO_2 and N_2O emissions (Höper et al. 2008, Wilson et al. 2009, Glatzel et al. 2011). It is suggested that these enhanced emissions may be caused by anaerobic consumption of organic litter formed by plants that died back after flooding rather than by anaerobic consumption of peat (Hahn-Schöfl et al. 2011).

Nevertheless, drained eutrophic fens under agricultural use have a great potential to emit large effluxes of GHGs. Their potential to emit N_2O is significantly greater than that of virgin fens or drained, but nutrient-poor peatlands (Regina et al. 1996). N_2O emissions are usually driven by a combination of several factors. Background emissions are controlled by long-term site-specific conditions such as nutrient status (e.g. C/N-quotient, Maljanen et al. 2009), hydrological characteristics (Freeman et al. 1993, Martikainen et al. 1993), vegetation type (Glatzel et al. 2008) and soil temperature (Röver et al. 1998). Event-based emission peaks are induced by short-term changes of site-specific conditions, such as freeze-thaw cycles (Teepe et al. 2000), fertiliser application (Ruser et al. 2001) and heavy rain. Hence, the annual release of N_2O can be very erratic, with a very large temporal and spatial variability (Flessa et al. 1995).

Despite this strong variability, non-growing-season effluxes may contribute 40–80 % of the annual emission of nitrous oxide on boreal minerotrophic peatlands (e.g. Alm et al. 1999, Regina et al. 2004, Maljanen et al. 2009) and temperate mineral soils (e.g. Flessa et al. 1995, Röver et al. 1998, Teepe et al. 2000). In contrast, studies of winter nitrous oxide emissions from temperate peatlands are scarce. Those existing either find a winter contribution of about 50 % (van Beek et al. 2010) or N_2O effluxes too close to the detection limit to be further analysed (Hendriks et al. 2007).

In contrast to nitrous oxide, there are many reports of winter carbon dioxide and methane emissions from temperate and boreal wetlands. These emissions are significantly smaller outside the growing season (Dise 1992, Melloh & Crill 1996, Alm et al. 1999, Panikov & Dedysh 2000, Hao et al. 2006, Hendriks et al. 2007, van Beek et al. 2010) contributing 10–40 % (CO_2) or about 10 % (CH_4) depending on site-specific climatic conditions during the winter.

In Mecklenburg-Western-Pomerania (north-eastern Germany) an area of 245,152 ha is covered by peatlands (Zauft et al. 2010) which accounts for 10.6 % of the federal state’s land surface. By the early 1990s, 99 % of these peatlands were drained for agricultural use (Gelbrecht et al. 2001). It is now an integral part of Mecklenburg-Western-Pomerania’s environmental policy to protect and restore its peatlands. An important goal is the reduction of GHG emission and the calculation of its monetary values (Federal Environment Agency 2007). Several peatlands in this region are now part of National Parks, Biosphere Reserves or Nature Protection Zones. Some have already been re-wetted, on others intensive agriculture is abandoned but extensive agricultural land use is still common.

Drained eutrophic fens are a strong source for CO_2 and N_2O, but re-wetting them may cause large emissions of CH_4. Nature conservation management guidelines include the abandonment of mineral fertiliser, a reduced cutting frequency and a cattle-free winter period. Since increasing wetness in winter may alter the anoxic layer of the soil and since cool conditions reduce microbial metabolism, we expect small net release rates of CO_2 and CH_4. Although winter N_2O effluxes may be large on temperate mineral soils (e. g. Flessa et al. 1995, Röver et al. 1998, Teepe et al. 2000), we expect small N_2O effluxes because land use abandonment has been shown to reduce N_2O effluxes due to the lack of inorganic fertiliser (Hendriks et al. 2007).

In the work reported here we measured the effects of extensive agricultural use on GHG effluxes from a drained eutrophic fen.

2.2 MATERIAL AND METHODS

2.2.1 Site description

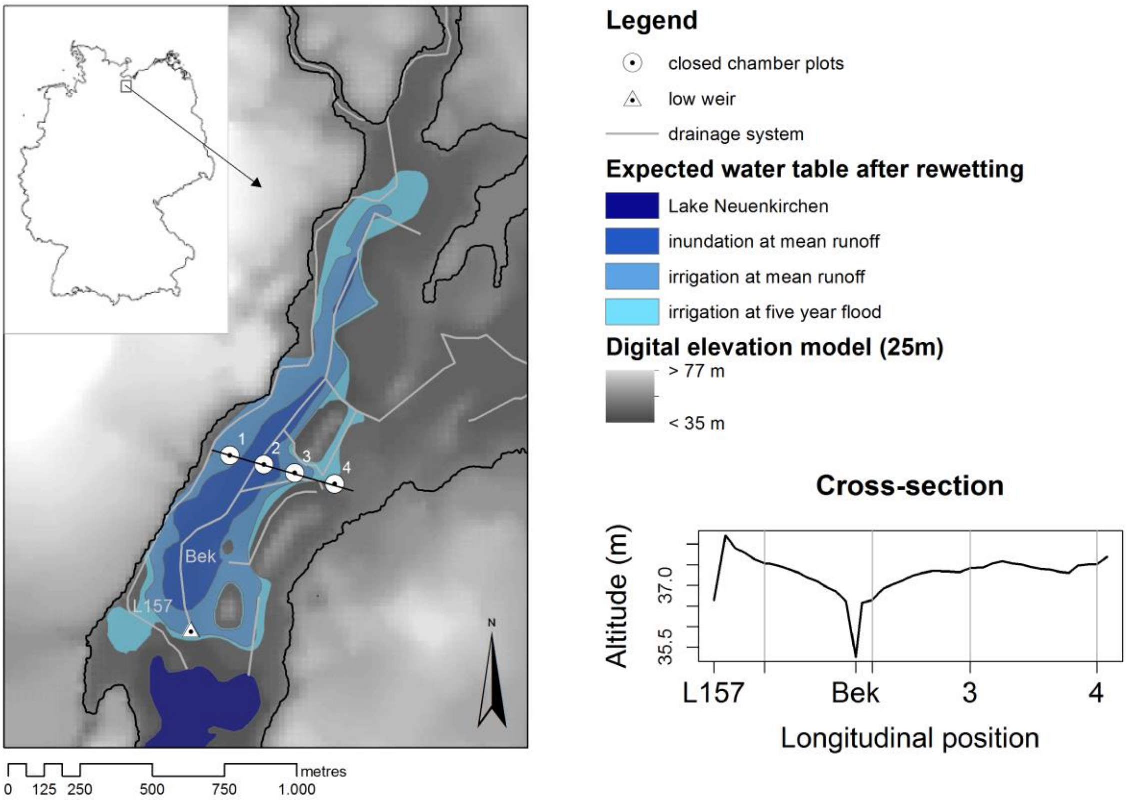

Figure 2.1: Map of the study site “Neuenkirchener Niederung” showing four measurement plots along the transect, the drainage system, expected water conditions after re-wetting, and landscape topography (digital elevation model with a raster resolution of 25 m). ‘L157’ identifies the main ditch. The inset shows the location in north-east Germany.

The study site (108 ha) is part of the Biosphere Reserve “Schaalsee”, in a small river valley (“Neuenkirchener Niederung”) near Lake Schaalsee, 60 km east of Hamburg (Mecklenburg-Western Pomerania, northern Germany; 53°36’ N, 10°59’ E, see Figure 2.1). The regional climate is temperate with a maritime influence. Climatic data were derived from a 1 × 1 km grid provided by the German Weather Service. The data were extrapolated using data from nearby stations and a digital elevation model, and have been shown to correlate well with measured values (Müller-Westermeier 1995). Mean annual air temperature is 9.0 °C. The long-term mean temperature in January is 0.2 °C, and the average snow cover period is 5.9 days. Annual precipitation is 711 mm with an annual climatic water balance of +134 mm (Hippke, pers. comm.).

The fen is part of a river valley mire system that originated from a glacigenic tunnel valley. As its hydrology is dominated by groundwater flow, it is a percolation mire (Joosten & Succow 2001). The Bek, a dredged and channelled river, flows through the mire system. The surrounding area of the study site is characterised by agricultural land use on stagnosols originating from glacial till. These soils are typically decalcified down to 30–50 cm (Dann & Ratzke 2004) with pH 4–6 (Reuter 1962).

The study site was moderately drained during the 19th century. During the second half of the 20th century much deeper ditches were made (1.5–2.2 m), profoundly altering the water balance of the system. These ditches guide the main part of the outflow around the fen through a major ditch (L157), that formerly ran into the Bek (Figure 2.1).

From the 1970s until 2004 the study site was intensively used as grassland. Since then, the area has been managed under nature conservation guidelines, including abandonment of mineral fertilisation, reduced cutting frequency and a cattle-free winter period. Typical plant-species of intensively used temperate grasslands are still dominant (such as *Alopecurus pratensis* L., and *Poa trivialis* L). Further species with large abundance or coverage are *Poa pratensis* L., *Taraxacum officinale* L., *Ranunculus repens* L. and *Holcus lanatus* L. indicating an ample nutrient supply from the soil (Ellenberg & Leuschner 2010). The plant community can be classified as *Molinio-Arrhenatheretea* Tx. 1937. The fen peat reaches more than 5 m deep, but the upper layer of peat (1 m) is strongly decomposed (H10–H8, von Post scale). At depths where decomposition is not too strong, peat originating from alder can be found at the western and at the eastern boundaries of the fen, whereas in the central parts the peat originates from reed and sedges (Table 2.1). Re-wetting of the mire system is planned but not yet implemented.

Table 2.1: Characteristics of Plots 1 to 4: degree of decomposition (von-Post scale); organic carbon (OC) proportion of dry mass [g / g] estimated from a 1 m peat core; C/N quotient estimated from cores (n = 5) of the uppermost 0.3 m.

Plot	1	2	3	4
Identified macrofossils	Alder	Reed	Reed/Sedges	Alder
Degree of decomposition	H8–H10	H8–H10	H9–H10	H8–H10
OC	0.67	0.68	0.57	0.41
C/N	12.3	13.3	12.4	11.8

2.2.2 Study design

In October 2009, four GHG measurement plots were established along a transect crossing the “Neuenkirchener Niederung” from west to east. The plots were chosen to span the expected water conditions caused by the planned re-wetting of the mire (Figure 2.1). Plot 2 represents the wettest and

Plot 4 the driest conditions, while Plots 1 and 3 will have intermediate water levels. In 2009, the area lying west of the Bek (Plot 1) was grazed by young male beef cattle. The area east of the Bek (Plots 2, 3, 4) was cut twice the same year. We measured gas fluxes in removable chambers set on collars sunk permanently into the peat surface. We could not install these collars until the cattle had been removed from the plots, only three weeks before measurements began. The good fit of CO_2 effluxes *versus* soil temperature (see Figure 2.5) indicates, however, that any altered respiratory activity (due to root damage caused by installation of the collars) was negligible. Because of grazing, mowing, and the protected status of the fen, no system of boardwalks could be laid; but the peat was dense and strongly humified, so ebullition events caused by physical disturbance when the chambers were approached were neither expected nor observed. The absence of stepwise linear increases in concentration, that would have been a sign of ebullition events (Chanton & Whiting 1995), supports this conclusion.

Each plot consisted of a triangle of collar locations (about 5 m apart). Since the nature of N_2O effluxes is erratic, an unfeasibly large number of collars would be needed to achieve statistically significant differences of N_2O effluxes between treatments (Folorunso & Rolston 1984). Thus, our approach was a compromise between covering spatial variability and being able to make all the necessary measurements during the short daylight hours in winter. The circular PVC collars (30 cm outside diameter, 20 cm tall, 8 mm thick) were inserted in slots cut in the peat with a knife (depth between 5 and 10 cm). GHG effluxes were estimated from concentration measurements using the non-steady-state chamber method (Livingston & Hutchinson 1995). Sampling was carried out every two weeks from November 2009 until March 2010, with additional sampling during freeze-thaw events. Overall, the measurement period lasted 129 days.

For each sampling, opaque PVC-chambers (diameter 30 cm, height 30 cm) were carefully placed on top of the collars. The collar-chamber edge was sealed with a grooved ring from the inside and taped from the outside. Snow within the collar was not removed as snow removal is known to alter GHG effluxes (Maljanen et al. 2003, 2009). Four gas samples were taken (one every 15 minutes) with evacuated gas flasks (100 ml) that were attached to the chambers with a short (< 5 cm) silicone rubber tube. The samples were analysed by gas chromatography (Perkin Elmer Auto System) within a week for concentration of CO_2, N_2O and CH_4 using an Electron Capture Detector (ECD) and a Flame Ionization Detector (FID). The precision of analysis was about 10 vpb for methane, 70 vpb for nitrous oxide and 10 vpm for carbon dioxide.

Furthermore, we measured soil and air temperature, depth of water table, soil water, and nitrate concentration during each sampling period. High-resolution meteorological data (air temperature, precipitation, relative humidity, air pressure) covering the measurement period were collected by a weather station in Zarrentin, 7 km south-east of the study site. Peat characteristics and a height/depth profile along the transect were recorded once.

2.2.3 Data analysis

Gas efflux rates were estimated from the chamber concentration data using a prototype version of the R package “flux” (Jurasinski & Koebsch 2011). We used it to obtain the best linear fit to any three points out of the four possible groups of three (abc, abd, acd, bcd). The parameters of the model with the best linear fit (greatest R^2) were then used to obtain the change in concentration in the chamber headspace over the sampling time (d*c*/d*t*). When none of the models had $R^2 \geq 0.8$ the resulting efflux was discarded. The gas effluxes were calculated according to Fick’s first law and the assumption that diffusion is the single process of gas accumulation in the chamber headspace. Thus, the efflux rate f (mg m^{-2} h^{-1} for CO_2, µg m^{-2} h^{-1} for CH_4 and N_2O) was calculated from the molar mass M (g mol^{-1}) of the gas, the air pressure p (Pa), the chamber volume V (m^3), the gas constant R (m^3 Pa K^{-1} mol^{-1}), the chamber temperature T (K), the surface area A (m^2), and the concentration change over the sampling time d*c*/d*t* (vpm h^{-1} for CO_2, vpb h^{-1} for CH_4 and N_2O) as follows:

$$f = 10^3 \frac{MpV}{RTA} \frac{dc}{dt} \tag{2.1}$$

Plot-wise efflux was calculated as the mean efflux of the three chambers. Estimation of total efflux rates during the sampling period was made by integrating the area under the efflux curves. To calculate the global warming potential (in CO_2-equivalents) we used the 100 year time horizon given by Forster et al. (2007) with 25 CO_2-eq for CH_4 and 298 CO_2-eq for N_2O.

Differences of efflux and of environmental variables among the four plots were tested for significance using the pair-wise Wilcoxon rank test with Bonferroni adjustment of P values, because the data within single plots were not normally distributed in all cases. Generalised linear models were constructed to explain the variability within plots, and mixed effect models were built to explain the variability between plots. The best model was found by step-wise deletion of non-significant parameters (Crawley 2005). All statistical analyses were performed with R 2.12.0 (R Core Development Team 2011).

2.3 RESULTS

2.3.1 Winter 2009/2010 environmental characteristics

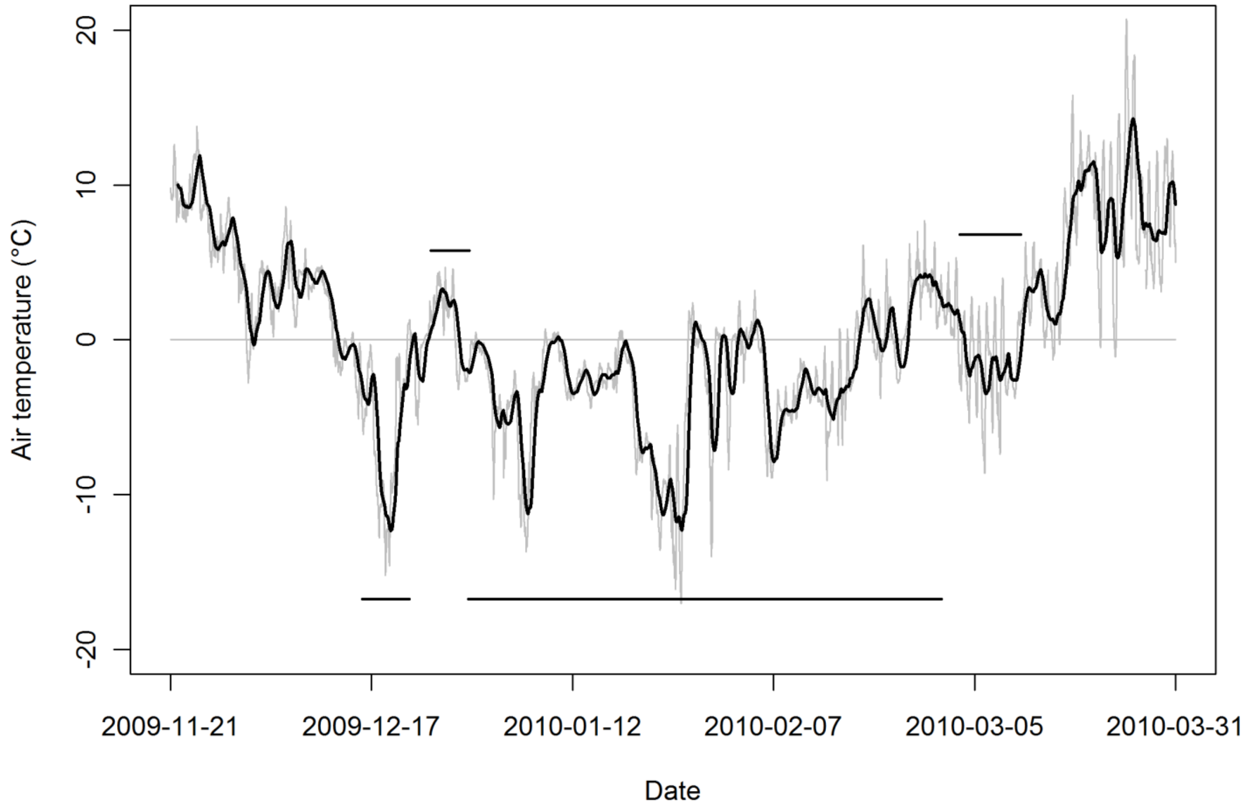

Figure 2.2: Air temperature at the weather station in Zarrentin, 7 km south-west of the study site during the winter 2009/2010 (thick line = daily running mean, thin line = 10-minute running mean). Upper horizontal bars (black) indicate thawing (December) and freeze-thaw cycles (March), lower horizontal bars (black) indicate snow-cover periods.

The winter 2009/2010 was the harshest for 30 years in Mecklenburg-Western Pomerania. The monthly mean temperatures of December, January and February fell below the long-term mean temperatures (1970–2000) by 1.6, 4.4 and 1.4 °C, respectively. The lowest air temperature (-17 °C) was measured around 26th January. The long-term average snow-cover period of 6 days per winter was exceeded by more than 60 days (Figure 2.2). The snow cover reached a maximum thickness of about 40 cm after heavy snowfall at the end of January and lasted until the melting period one month later. During this period, the snow cover isolated the soil from fluctuating air temperature. Therefore, soil temperature remained constantly around 0 °C and the eastern area of the fen (Plots 2, 3, 4) continued to be unfrozen during the snow-cover period. In contrast, a strong freeze-thaw cycle occurred just when the mire became snow free again. At that time, the air temperature varied greatly between day and night ranging from +4 °C to -10 °C (Figure 2.2).

The water table depths during the measurement period differed slightly among the four plots according to the expected water table conditions after a possible re-wetting (Figure 2.1, Figure 2.4). The depth of water table at Plot 2 was significantly greater than at Plot 4 (according to Wilcoxon-rank-test at $P < 0.05$). Water table depths at Plots 1 and 3 did not differ significantly from each other or from Plots 2 and 4 during the measurement period. Furthermore, parts of the study area (Plot 2, similar to "inundation at mean runoff", Figure 2.1) were inundated by meltwater for two weeks from the beginning to the middle of March 2010. The soil temperature was similar in all four plots throughout the investigation period.

2.3.2 Gas effluxes

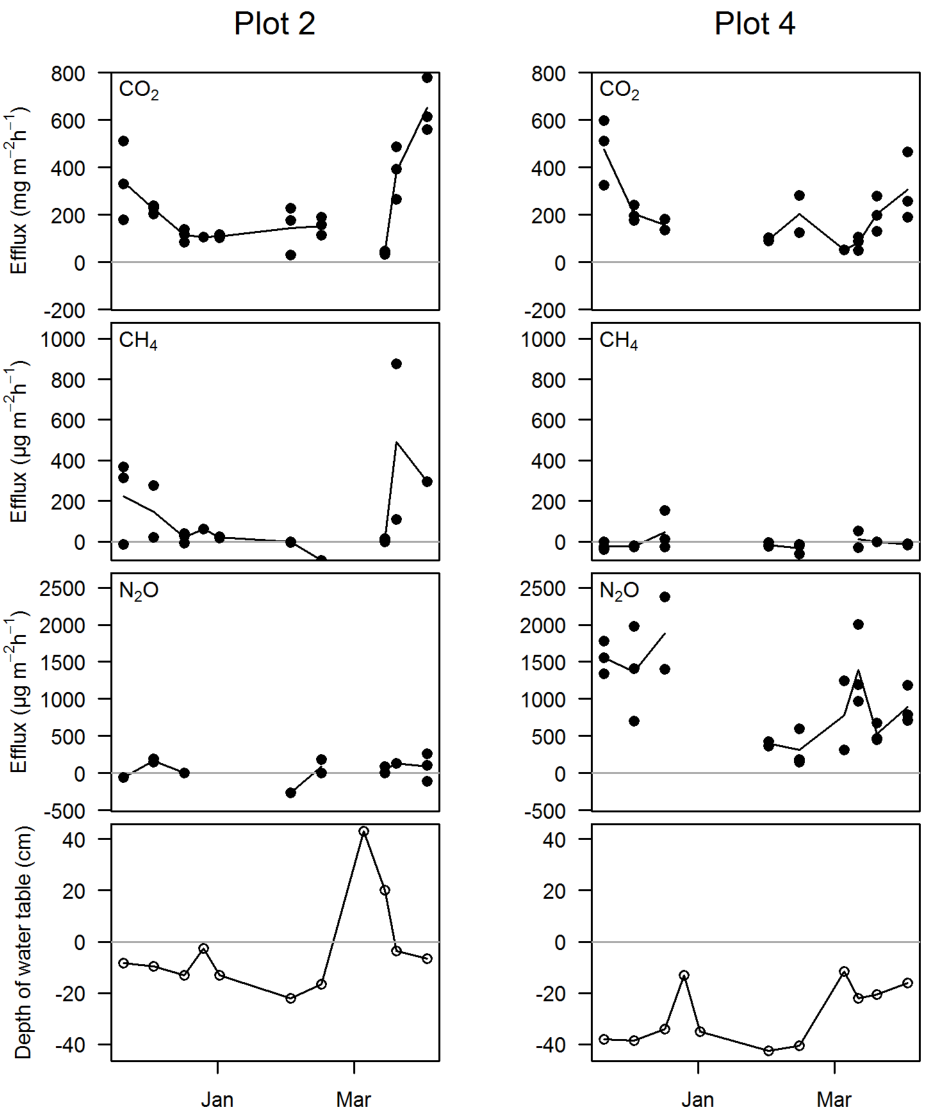

Figure 2.3: Gas effluxes and depth of water table (bottom row) of Plots 2 and 4 during the measurement period. These data span the hydrological range of the transect (Plots 1 and 3 have intermediate water level conditions, see Figure 2.4). For gas measurements, dots indicate single measurements and black lines indicate curves of mean effluxes. Zero line (grey) of water table graphs marks the soil surface. Note the differences in scales.

Greenhouse gas effluxes occurred during the whole measurement period even when the ground was snow-covered (Figure 2.3). The greatest effluxes of CO_2 and N_2O were recorded in November and March during periods with relatively high temperatures and without snow cover. At plot scale, GHG effluxes usually approximated a normal distribution except for CO_2 effluxes at Plot 2 (Table 2.2).

Table 2.2: Characteristics at four plots of gas effluxes during the 129-day measurement period (November 2009 to March 2010). Mean and median effluxes and coefficients of variation (CV) are given. Non-normally distributed effluxes (according to Shapiro-Wilk test at $P < 0.05$) in *italics*.

	CO_2			CH_4			N_2O		
	Mean	Median	CV	Mean	Median	CV	Mean	Median	CV
Plot	mg m^{-2} h^{-1}			µg m^{-2} h^{-1}			µg m^{-2} h^{-1}		
1	180	188	0.71	37	0	1.60	31	-28	3.92
2	*227*	*150*	*0.81*	118	42	1.49	24	47	5.45
3	132	124	0.99	11	11	1.37	26	10	9.58
4	198	203	0.66	-6	-14	-4.21	1013	896	0.55

N_2O effluxes were similar at Plots 1, 2 and 3 (-380 to 420 µg m^{-2} h^{-1}) but significantly greater at Plot 4 (310 to 1900 µg m^{-2} h^{-1}, Figure 2.4). On 10th March an N_2O efflux peak was observed at Plot 4 reaching 1400 µg m^{-2} h^{-1}. It followed a week-long freeze-thaw cycle that reached minimum temperatures of -10 °C at night and maximum temperatures of +4 °C during daytime.

CO_2 effluxes were similar at all four plots, ranging from a small uptake of -160 mg m^{-2} h^{-1} to 650 mg m^{-2} h^{-1}. CH_4 emissions were near zero throughout the measurement period except for Plot 2 at the end of November and for Plots 1 and 2 at the end of March after the snow cover period. CH_4 effluxes ranged from -90 to 490 µg m^{-2} h^{-1} but did not differ significantly among plots (Figure 2.4).

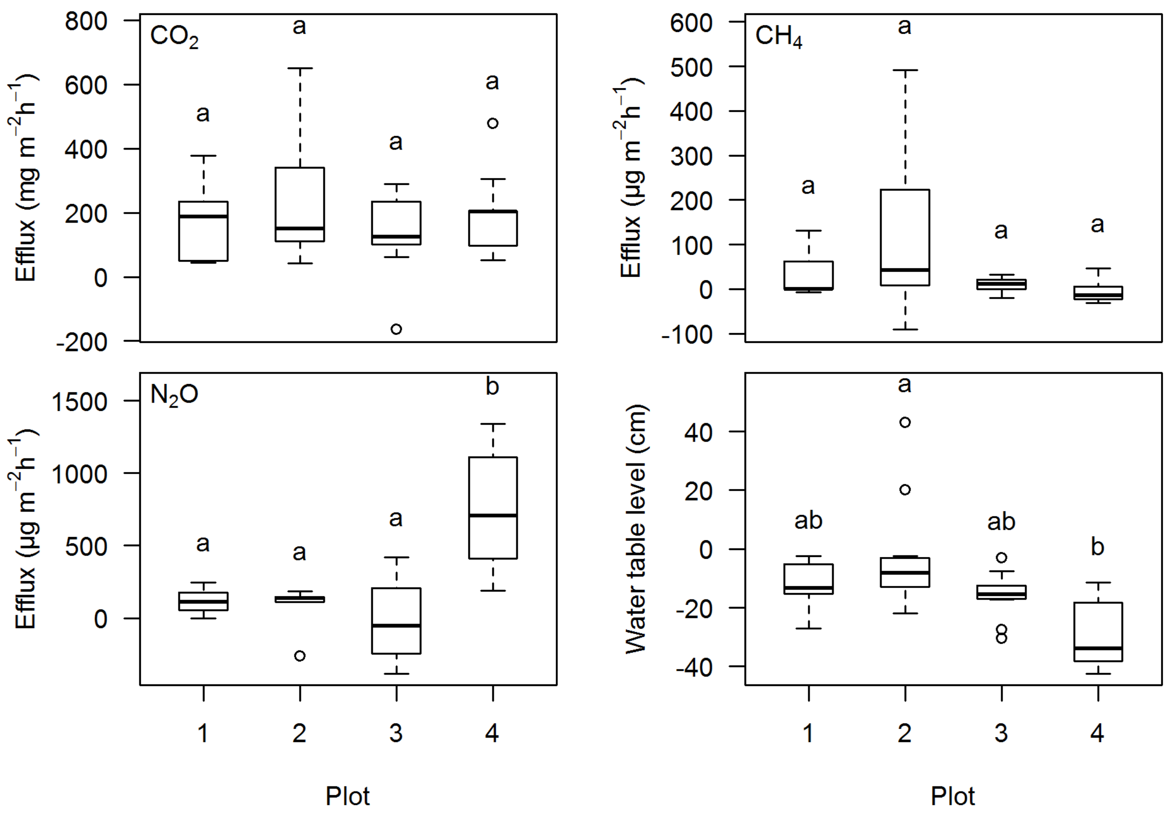

Figure 2.4: Plot-wise distribution of greenhouse gas effluxes and water table levels (relative to ground surface) during the measurement period. Thick line: median; box extent: interquartile range; whisker extent: marks the data lying within 1.5 times interquartlie range. Different lower case letters indicate significant differences between the plots at $P < 0.05$, according to pair-wise Wilcoxon-rank-test with Bonferroni *P*-value adjustment. Note the small number of samples ($n < 12$) for each plot.

CO_2, CH_4 and N_2O effluxes contributed 63 %, 0.2 % and 37 %, respectively, of the accumulated greenhouse gas emissions of the study site (Table 2.3). N_2O played the most important role (64 %) at Plot 4, whereas at Plot 2 N_2O was taken up during the measurement period. Plot 4 had the greatest release of GHG, emitting more than 14 t ha^{-1} CO_2-equivalents, whilst Plot 3 had the smallest (3.5 t ha^{-1}). On average, the study site emitted 7.0 t ha^{-1} CO_2-equivalents during the winter of 2009/2010.

Table 2.3: Total emissions from four plots of carbon dioxide, methane and nitrous oxide (t ha^{-1} CO_2-equivalents), contributions of CO_2, CH_4, N_2O (in %, *italics*) and mean depth of water table during the measurement period.

	all Plots	Plot 1	Plot 2	Plot 3	Plot 4
CO_2	4.4	3.9	5.7	2.9	5.2
	63	*85*	*102*	*84*	*36*

CH_4	0.02	0.02	0.04	0.01	0.00
	0	*1*	*1*	*0*	*0*
N_2O	2.6	0.7	-0.2	0.6	9.2
	37	*14*	*-3*	*16*	*64*
Mean depth of water table (cm)	-15	-12	-3	-18	-28

2.3.3 Environmental controls

Carbon dioxide and methane effluxes do not differ among the plots (Figure 2.4). Therefore generalised linear regression models were built by using environmental data covering the whole measurement period (Figure 2.5). Carbon dioxide efflux, *Reco* (mg m^{-2} h^{-1}), in relation to soil temperature T_{soil}, can be described best (among the models tested) by:

$$Reco\ [mg/(m^2 \cdot h)] = R10^{E0(1\ /\ (283.15\ K\ -\ 227.13\ K)\ -\ 1\ /\ (Tsoil\ [K]\ -\ 227.13\ K)))} \qquad (2.2)$$

($R^2 = 0.55$, $P < 0.001$), following the suggestion of Lloyd & Taylor (1994), with $R10$ = 429.7 mg m^{-2} h^{-1} being the reference efflux at 10 °C and $E0$ = 392.1 K.

Methane effluxes of the study site during the winter season can be best described by the following regression equation:

$$y = 288.6\ \mu g\ m^{-2} h^{-1} - 91.3\ \mu g\ m^{-2} h^{-1} \cdot \ln(bx) \qquad (2.3)$$

($R^2 = 0.38$, $P < 0.001$) in which y is the CH_4 efflux (µg m^{-2} h^{-1}), x is the depth of water table (cm), and b =1 cm^{-1}.

In contrast, nitrous oxide effluxes do differ between the plots (Figure 2.4). Therefore the model was built by using parameters that discriminate the plots in space. The best model found is:

$$z = 129.4\ \mu g\ m^{-2} h^{-1} + 48.6\ \mu g\ m^{-2} h^{-1} \cdot bx - 74.3\ \mu g\ m^{-2} h^{-1} \cdot bx{:}w \qquad (2.4)$$

($P < 0.05$) in which z is the N_2O efflux (µg m^{-2} h^{-1}), x is the mean depth of water table (cm), w is organic carbon content (g g^{-1}), and b =1 cm^{-1}, with *bx:w* being the interaction between depth of water table by organic carbon content.

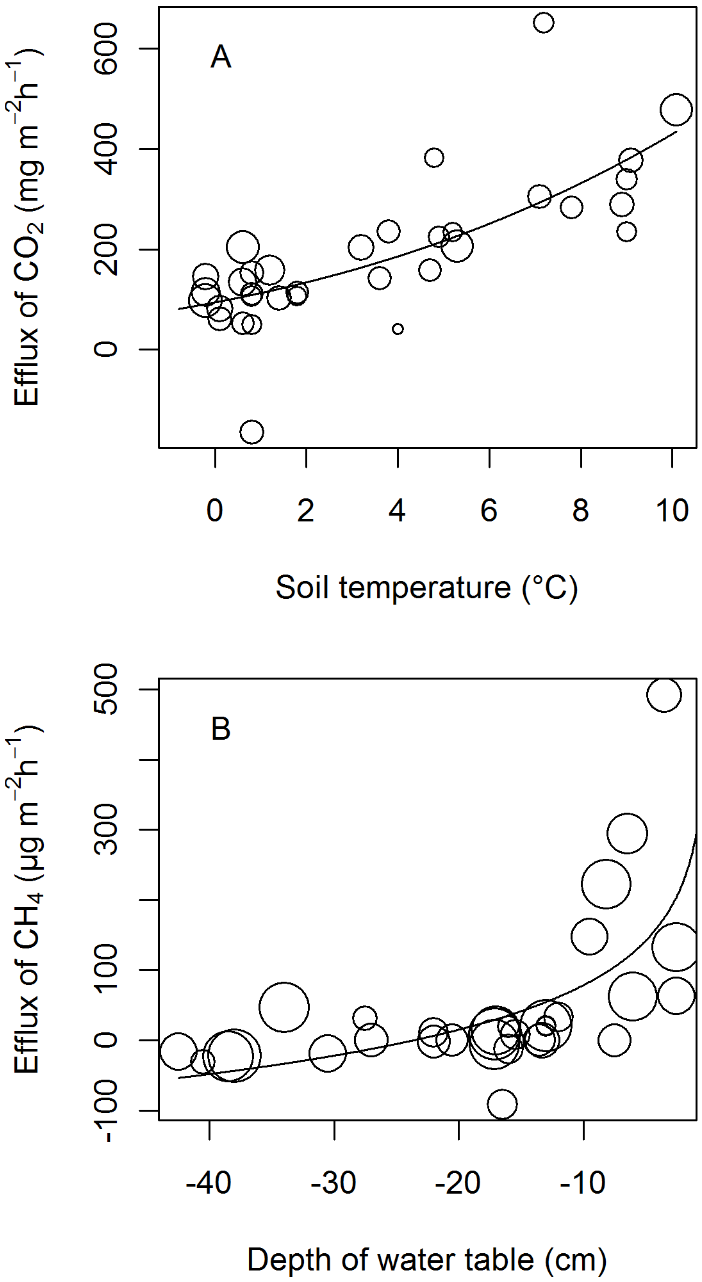

Figure 2.5: Relations of CO_2- ($R^2 = 0.55$, $P < 0.001$) and CH_4-fluxes ($R^2 = 0.38$, $P < 0.001$) to the relevant ecosystem controls. A circle represents the mean of plot-wise efflux measurements ($n = 3$) with circle diameter being proportional to a third variable, namely depth of water table in (A) and air temperature in (B).

2.4 DISCUSSION

The radiative forcing of GHG emissions from the Neuenkirchener Niederung during the winter 2009/2010 was mainly caused by carbon dioxide (63 %) and nitrous oxide emissions (37 %). Methane emissions played only a minor role.

Compared with other studies, the global warming potential (GWP) of winter N_2O emissions (2.6 t ha^{-1} CO_2-equivalents) was similar to the GWP of annual N_2O emissions reported from a drained fen in Finland (2–4 t ha^{-1}, Nykänen et al. 1995) and to the GWP of winter N_2O production of organic grassland soils in the Netherlands (~2.8 t ha^{-1} a^{-1}, derived from van Beek et al. 2010). Given the extraordinarily cold winter of 2009/2010 with its long period of snow cover, and that the greatest nitrous oxide emissions occurred in late autumn (November) and early spring (March), our results resemble an emission pattern typical for a boreal climate as described by Alm et al. (1999) rather than an emission pattern for a temperate climate where the largest N_2O emissions may occur during the winter months of December to February (Flessa et al 1995, Teepe et al. 2000). For this reason, the N_2O emissions measured in this study seem likely to have been smaller than during a climatically normal winter.

The snow cover during winter 2009/2010 prevented the soil from becoming deeply frozen. In such cases, nitrous oxide effluxes during freeze-and-thaw events are typically smaller than effluxes from formerly deeply frozen soils (Maljanen et al. 2009). In contrast, effluxes during the snow cover period are typically greater if the soil is not deeply frozen. For this reason, the distribution of N_2O effluxes in this study may not be as skewed as reported by others (Flessa et al. 1995). Hence, the timing of snow cover development and frost during the beginning of winter is an important factor that controls N_2O emissions in the non-growing season (Maljanen et al. 2003). Nevertheless, an N_2O efflux peak of 1,400 µg m^{-2} h^{-1} at Plot 4 was detected after one week of intensive freezing and thawing of snow-free soil. Pihlatie et al. (2010) reported a time span of one week after freezing and thawing to be the period of greatest N_2O release during such a freeze-thaw cycle.

Although the land use of the study site was extensified five years before the measurements were carried out and despite the extraordinarily cold winter, the N_2O emissions of the Neuenkirchener Niederung are still as large as reported for intensively used grassland on organic soils (Velthof & Oenema 1995, Augustin et al. 1998). Some authors show that fertilization is not necessarily needed on peatlands to produce N_2O effluxes as much as 10,000 µg m^{-2} h^{-1} (Maljanen et al. 2009) or 56.4 kg ha^{-1} a^{-1} N_2O-N (Flessa et al. 1998). Depth of water table and organic carbon content may be much more important drivers of N_2O emissions in our case (Equation 2.4). These were shown to be important controls in several other studies (e.g. Freeman et al. 1993, Martikainen et al. 1993, Maljanen et al. 2009).

CO_2 efflux can be modelled with our data following Lloyd & Taylor (1994) and is thus driven by soil temperature. Although soil temperature varied only within an interval of 10 °C during the measurement period, a clear dependence of CO_2 efflux on soil temperature was observed (Equation 2.2). Since grassland use has been constant for decades and the study site is either used for hay production or cattle grazing, C uptake of the soil during the growing season is improbable. Therefore, the CO_2 balance of the system can be described by CO_2 emissions alone showing that the study site acts as a source for CO_2. The winter CO_2 emissions from the study site amount to 4.4 t ha^{-1}, which is closer to winter CO_2 emissions of a boreal minerotrophic fen (2.5 t ha^{-1}, derived from Alm et al. 1999) than to winter CO_2 emissions of an abandoned maritime peat meadow (11.0 t ha^{-1}, derived from Hendriks et al. 2007), again indicating the impact of the extraordinarily cold winter 2009/2010. In contrast, the CO_2 balance from virgin fens is generally close to 0 or even negative (Frolking et al. 2006) as their hydrology inhibits oxic decomposition, and carbon is sequestered.

Winter methane effluxes were close to the detection limit and contributed less than 0.2 % (0.02 t ha^{-1} CO_2-equivalents) to the total winter emissions of the study site. Annual methane effluxes from similar fen sites in Finland (~0.04 t ha^{-1}, derived from Nykänen et al. 1995) are in the same order of magnitude. In contrast, annual methane emissions from virgin fen sites are two orders of magnitude greater (8–15 t ha^{-1} CO_2-equivalents, derived from Dise 1992, Nykänen et al. 1995, Melloh & Crill 1996), but N_2O emissions from such virgin sites are typically close to the detection limit.

To get an idea about the magnitude of the emission potential from the study site we assumed that winter carbon dioxide emissions contribute 25 % (23 %, Alm et al. 1999, 34 %, derived from Hendriks et al. 2007), winter methane emissions contribute 10 % (Dise 1992, Alm et al. 1999) and winter nitrous oxide emissions contribute 50 % (40–80 %, Alm et al. 1999, Maljanen et al. 2009, van Beek et al. 2010) of the annual emissions. Therefore, the annual GHG emissions of the study site are estimated at 18 (CO_2), 0.2, (CH_4) and 5 (N_2O) t ha^{-1} CO_2-equivalents, which would total about 23 t ha^{-1}, with the winter contributing 30 %. For this reason, our assumption of small net GHG emissions during winter must be rejected. This indicates that winter GHG emissions from temperate peatlands should be taken into account when comparing the GHG emission potentials of extensively used and re-wetted mires.

In addition, both young male beef cattle and dairy cattle emit methane. Given that one lactating cow (weight 650 kg, milk yield 6500 kg a^{-1}, CH_4-emission 135 kg a^{-1}, Jentsch et al. 2009) needs about 0.8 ha of extensively used grassland, the net annual methane emissions of the study site would increase to about 4 t ha^{-1} CO_2-equivalents, and the annual GHG emissions to 27 t ha^{-1}. Therefore, extensification of land use without re-wetting might not reduce the GHG emissions of the Neuenkirchener Niederung. According to our findings, aiming to reduce the GWP of peatlands is not a suitable objective. Only a permanent and effective rise of the water table to a level close to the ground surface will lead to permanently reduced CO_2 and N_2O emissions.

However, re-wetting of drained fens may cause raised CH_4 emissions that possibly counteract the reduced CO_2 and N_2O emissions (Höper et al. 2008, Wilson et al. 2009, Glatzel et al. 2011). Field studies of GHGs from these dynamic and young ecosystems are few, and seldom cover long time spans. On the one hand, increased methane emissions after re-wetting might originate from dead organic compounds of inundated grassland plants such as *Phalaris arundinacea* L. (Hahn-Schöfl et al.

2011). On the other hand, typical wetland plants such as *Typha* spp. or *Phragmites australis* (Cav.) Trin. ex Steud. are absent directly after re-wetting, but they are known to be important vectors for CH_4 emissions in virgin peatlands (Chanton et al. 1993, van der Nat et al. 1998). Thus, it can be assumed that these large CH_4 emissions prevail after re-wetting as long as the plant composition of the ecosystem is shifting towards a new equilibrium state reflecting altered hydrological conditions of the habitat. On a raised bog in north-east Germany, Bönsel & Sonneck (2011) showed that this shift in plant composition after re-wetting takes at least a decade. Therefore, we propose that future research on GHG emissions from peatlands should focus

1) on study sites with a different stages of re-wetting, in order to develop a time series model of CH_4 emission dynamics after re-wetting; and
2) on the estimation of CH_4 emission potentials of different vegetation types from drained peatlands, in order to better predict the possible CH_4 output of inundated grassland plant communities.

When these points are addressed, comparisons of GWPs of drained and re-wetted peatlands should be more accurate and reliable than comparisons of annual GHG budgets derived from single- or two-year measurement campaigns.

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3 The effect of an exceptionally wet summer on methane effluxes from a 15-year re-wetted fen in north-east Germany[2]

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SUMMARY

Re-wetting minerotrophic fens has become an important strategy to mitigate climate change in Germany. However, recent studies report raised methane (CH_4) effluxes during the first years after flooding. A minerotrophic fen in north-east Germany that was re-wetted 15 years ago was exposed to exceptionally heavy rainfall and freshwater flooding in August 2011. We measured CH_4 effluxes from wetland vegetation stands dominated by *Phragmites australis* (Cav.) Trin. ex Steud., *Typha latifolia* L. and *Carex acutiformis* Ehrh., using the closed-chamber method, fortnightly from March 2011 to March 2012 with extra sampling during the flooding. The respective annual effluxes of CH_4 (mean ± 1 standard error) in these three types were 18.5 ± 1.3, 21.1 ± 1.2 and 47.5 ± 5.0 g m^{-2} a^{-1}, with the August effluxes contributing 40 %, 50 % and 10 % of the annual effluxes. Despite the freshwater flooding in August, annual CH_4 effluxes from the 15-year re-wetted fen are similar to those reported from pristine fens. These results are promising, because they indicate that, although CH_4 effluxes are elevated after re-wetting, they may return to values typical for pristine fens after 15 years. Hence, re-wetting can achieve the purpose of reducing greenhouse gas effluxes from drained minerotrophic fens.

KEY WORDS

emergent macrophytes, heavy rainfall, freshwater flooding, peatland

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The effect of an exceptionally wet summer on methane effluxes from a 15-year re-wetted fen in north-east Germany

3.1 INTRODUCTION

The greenhouse gas (GHG) methane (CH_4) is an important driver of anthropogenic radiative forcing (Forster et al. 2007). While agricultural activities have increased the global atmospheric CH_4 concentration during the last century, pristine peatlands are natural sources of CH_4 (Dise 1992, Melloh & Crill 1996). The reported climatic impact of pristine peatlands is a slight warming or a slight cooling depending on the balance of carbon (C) sequestration and CH_4 effluxes (Frolking et al. 2006). However, about 99 % of north-east Germany's fens had been intensively drained by 1990 (Gelbrecht et al. 2001).While natural peatlands are regarded as C sinks (Frolking et al. 2006), drainage has converted them to C sources due to high carbon dioxide (CO_2) emissions (Couwenberg et al. 2011). Drained fens in north-east Germany have been being re-wetted since the mid-1990s to restore their ecosystem functions including (primarily) the habitat function for rare biota and the C sequestration function as an important climate change mitigation strategy (Erwin 2009).

However, recent studies indicate that re-wetting of drained fens may cause large CH_4 effluxes if, as is common, the ecosystem is flooded (i.e. water table ranging above the ground surface), which counteracts the reduction of CO_2 and nitrous oxide (N_2O) effluxes (Augustin & Chojnicki 2008, Höper et al. 2008, Glatzel et al. 2011). Litter from dead and drowned vegetation has been identified as a source of increased CH_4 effluxes (Hahn-Schöfl et al. 2011). Assessment of the success of measures to mitigate GHG release depends on knowledge about the duration of these increased CH_4 effluxes after re-wetting. This assessment is complicated by differences in CH_4 effluxes among plant species (Chanton et al. 1993, Couwenberg & Fritz 2012) and intra-annual differences in precipitation with resulting fluctuations in ground water table.

The Trebel valley mire in north-east Germany is a complex of minerotrophic fens and one raised bog. The plant composition already reflects the altered hydrology caused by re-wetting 15 years ago. Since then, the average water table of the mire system has remained within 0–20 cm below ground surface (Bönsel & Sonneck 2011).

The exceptionally wet summer of 2011 gave us the opportunity to measure the effect on CH_4 effluxes of a natural prolonged flooding with freshwater under high summer temperatures. The three types of vegetation available in the fen parts of the mire system were stands dominated by *Phragmites australis* (Cav.) Trin. ex Steud., *Typha latifolia* L. and *Carex acutiformis* Ehrh. As these vegetation stands are better adapted to inundation than the grassland communities of drained fens, we hypothesise that, despite the summer flood, CH_4 effluxes from the 15-year re-wetted fen are similar to those reported for pristine fens in similar climate zones.

3.2 MATERIAL AND METHODS

3.2.1 Site description

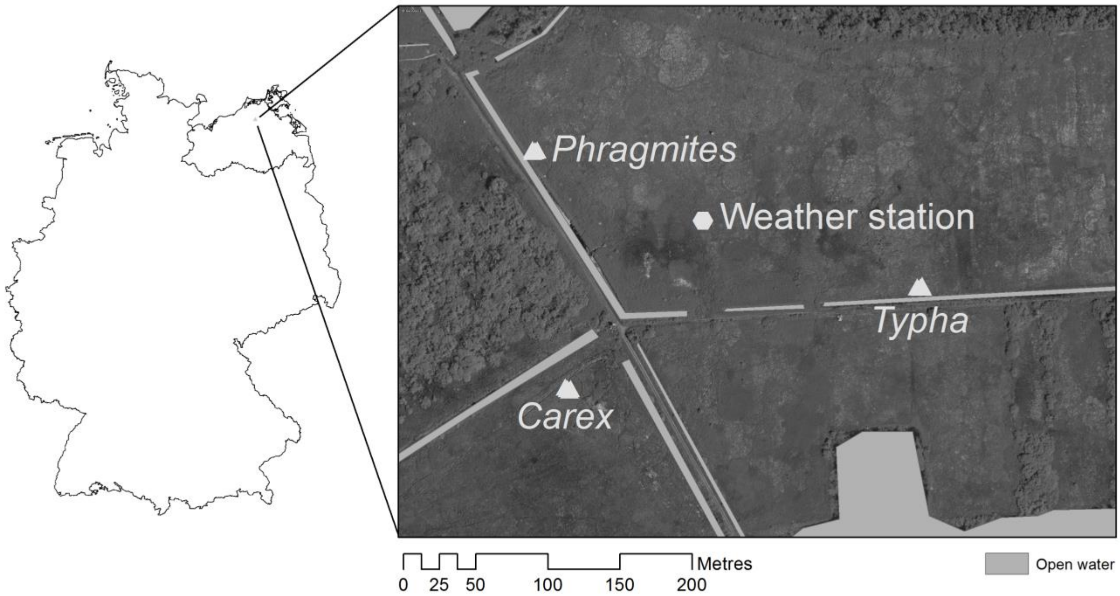

Figure 3.1: Maps showing (left) the location of the study site in north-east Germany, and (right) the study site (54°06‘ N, 12°44‘ E) at larger scale. Measurement plots (triangles) were located in *Phragmites*, *Typha* and *Carex* stands.

The study site is in the Trebel valley mire system close to the town of Tribsees (54°06‘ N, 12°44‘ E, Figure 3.1) within a re-wetted area of more than 3000 ha. The climate is humid with a continental influence, mean annual (1991–2010) air temperature 9.1 °C and mean annual (1981–2010) precipitation 626 mm (both calculated from the available German Weather Service data). For August, mean temperature is 19.1 °C, mean precipitation is 64 mm and net precipitation is slightly positive at +50 to +100 mm (Klämt & Schwanitz 2001). The fen is a typical percolation mire of the southern Baltic region. The peats are mainly of reed and sedge origin with thickness in the range 4–6 m. Deep (~1.5 m) drainage ditches allowed high-intensity grassland use until the fen was re-wetted in 1997. Since that time the water table has remained close to the ground surface (Bönsel & Sonneck 2012) and the vegetation of the study site has shown a typical shift from a managed grassland with plants such as *Agrostis* spp. L., *Alopecurus geniculatus* L. and *Phleum pratense* L. to a re-wetted state with reeds (*P. australis* and *Phalaris arundinacea* L.) and stands dominated by *Carex* spp. L. or *Typha* spp. L. (Bönsel & Sonneck 2011) where the only remaining land use is hunting/shooting.

3.2.2 Study design

In November 2010, three GHG measurement plots were established in vegetation stands dominated by *P. australis*, *T. latifolia* and *C. acutiformis*, respectively (Figure 3.1). Every plot consisted of three measurement locations arranged at ~2 m intervals along a boardwalk. The measurement locations were marked with permanent collars (h = 20 cm, d = 63 cm) sunk into the peat surface to a depth of 5–10 cm. We installed boardwalks and collars four months before the first measurements were made. in In mid-July 2011, we installed collars at two additional locations close to the boardwalk in each vegetation stand, and we carried out additional measurements during August 2011. We did not cut the vegetation, as cutting may alter internal gas transport in convective and diffusive plants (van der Nat & Middelburg 2000, Ding et al. 2004).

We sampled CH_4 effluxes fortnightly throughout the year with flexible, height adjustable, opaque, dynamic closed chambers (d = 63 cm) that could cover plants up to 2.3 m tall. The chamber material was flexible thermoplastic polyurethane, white outside and black inside, which is normally used in agriculture for covering silage. The chamber wall was stabilised with rings of plastic tubing to ensure full volume when installed in the field. The chambers were sealed onto the collars by pressing the lowest (or two lowest) ring(s) (depending on vegetation height) around the collars. During (rare) strong winds we added a tension belt. We tested the chamber for air tightness in the lab and found no significant concentration loss during a two-hour enclosure time. Five gas samples per location were taken at ten-minute intervals with evacuated gas flasks (12 ml Exetainer, Labco Ltd.) that were attached to the chamber with a short (<2 cm) silicone rubber tube. The average temperature change inside the chambers during the 40-minute deployment time was <2 °C. To account for diurnal variability, sampling was carried out at randomised times of day (Ding et al. 2004), but usually between 8 a.m. and 4 p.m. Gas samples were analysed for CH_4 concentration in the laboratory, within one week, using a gas chromatograph (Shimadzu Auto System). We collected water samples from the plots during the fortnightly gas measurements. The water table in each vegetation stand was recorded hourly by automated loggers (Solinst, Canada) submerged in dipwells. Other environmental variables were logged hourly by a weather station (F&C, Germany) located in the middle of the study site (Figure 3.1).

3.2.3 Data analysis

R 2.15.0 (R Development Core Team 2012) was used for all statistical analyses. Mean values ± 1 standard error are given. The R package "flux" was used to derive effluxes from the chamber concentration data (Jurasinski et al. 2012). The parameters of the model with the best linear fit (greatest R^2) were used to obtain the change in concentration in the chamber headspace over the sampling time (dc/dt) using four out of five concentration values. When none of the models had

$R^2 \geq 0.9$ the associated efflux was discarded. Effluxes lying beyond four standard deviations of the mean of effluxes from each vegetation stand were regarded as outliers and excluded from further analyses.

We estimated annual effluxes with a Monte Carlo repeated sampling procedure using the function auc.mc in the R package "flux". This function linearly integrates the effluxes over the covered time period many times, each time leaving out a specified number of measurements (jackknife method); we used (n – 2) measurements in all cases. From the resulting distribution of seasonal efflux estimates *per* sampling location we calculated the mean (best estimate) and the standard deviation (providing an estimate of the error introduced by temporal variation in sampling and by missing high/low effluxes during regular sampling campaigns). The best estimates *per* sample location were averaged to get the reported annual efflux values. The reported standard errors were calculated from the propagated standard deviations of the best estimates (law of error propagation). To extract the effect of the flood, we estimated whole-year effluxes (a) including and (b) excluding all August 2011 measurements.

Because the single-plot data were not normally distributed in all cases, differences in efflux and environmental variables amongst the four plots were tested for significance using the pair-wise Wilcoxon rank test with Bonferroni adjustment of P values. Generalised linear models of water table *vs*. CH_4 efflux were constructed to explain variability within the vegetation stands.

3.3 RESULTS

In Mecklenburg-Western Pomerania, the summer of 2011 was the wettest in the last 30 years. Total precipitation during July and August was 392 mm, which is three times the long-term mean. As a result, the Trebel valley was flooded in August with the water table 10 cm (*Phragmites*), 20 cm (*Typha*), and 40 cm (*Carex*) above the peat surface. In contrast, the median water table for the whole study period was -7.7 cm (*Phragmites*) (i.e. below the surface), 5.7 cm (*Typha*), and 3.7 cm (*Carex*) above the peat surface. Mean temperatures in 2011 were 8.9 °C for the study year and 17.0 °C for August, i.e cooler than the annual (9.1 °C) and August (19.1 °C) long-term means. Analyses of fen water samples revealed mesotrophic conditions (total N <1 mg l^{-1}, Table 3.1). The peat is strongly decomposed (H9–H10 on the von Post scale) with low SOC and C/N quotient, less pronounced at the *Carex* stands (Table 3.1).

Out of 311 flux measurements, 110 were discarded on the basis of the efflux estimation criteria and six were eliminated by outlier detection. At the *Phragmites* and *Typha* plots, CH_4 effluxes remained below 5 mg m^{-2} h^{-1} during most of the measurement period, but rose to 15–20 mg m^{-2} h^{-1} in August–September (Figure 3.2). The effluxes from these two vegetation types did not differ significantly from one another during the measurement year. CH_4 effluxes at the *Carex* plot were significantly higher ($P < 0.01$) and ranged from 5–20 mg m^{-2} h^{-1} during the growing season (May–November).

Table 3.1: Plot characteristics under the tree vegetation types. For peat: degree of decomposition (von Post scale), Soil (peat) organic carbon (SOC, C / soil, g / g) and C/N quotient estimated from cores (*n* = 5) of the uppermost 30 cm of peat. For fen water: nitrate and ammonium concentrations (µmol L^{-1}), pH and electric conductivity (EC, µS) during the year of measurements (*n* = 22 for each vegetation type); means ± one standard deviation are given.

Plot		*P. australis*	*T. latifolia*	*C. acutiformis*
Peat	Decomp.	H9–H10	H9–H10	H9–H10
	SOC	0.3 ± 0.07	0.3 ± 0.05	0.4 ± 0.02
	C/N	11.3 ± 0.0	11.4 ± 0.1	13.0 ± 0.4
Fen water	nitrate	21.7 ± 8.9	35.9 ± 15.3	12.6 ± 5.4
	ammonium	8.4 ± 4.7	47.4 ± 47.6	15.2 ± 6.8
	pH	8.3 ± 0.4	8.2 ± 0.3	7.9 ± 0.3
	EC	534 ± 58	732 ± 177	484 ± 56

The annual CH_4 efflux estimates (mean ± one standard error, derived by the jackknife method) of the *Phragmites*, *Typha* and *Carex* plots were 18.5 ± 1.3, 21.1 ± 1.2 and 47.5 ± 5.0 g m^{-2} a^{-1}, respectively (18.4, 21.0 and 48.0 g m^{-2} a^{-1} when integrating all measurements). To test the influence of the August flooding on the annual efflux estimates, we re-estimated the annual CH_4 effluxes leaving out the August CH_4 measurements (Figure 3.2), which yielded the results 11.4 ± 0.6, 11.0 ± 1.0 and 41.2 ± 8.0 g m^{-2} a^{-1}. The August measurements were thus responsible for 40 %, 50 %, and 10 % of the annual CH_4 effluxes from the *Phragmites*, *Typha* and *Carex* plots, respectively.

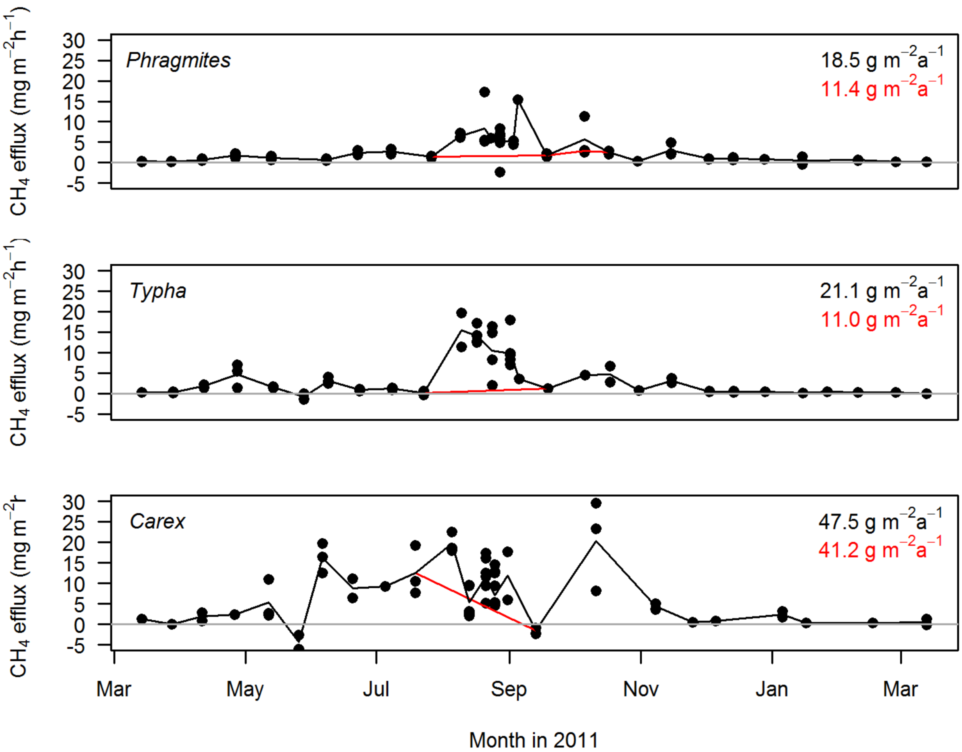

Figure 3.2: Course of CH_4 efflux from the three types of vegetation during the measurement period in 2011. Dots represent single measurements, black lines connect daily means of all measurements, red lines connect the daily means flanking the August measurements. Annual effluxes at the top right of each graph (black: all measurements, red: August measurements excluded) were calculated by the jackknife method described in the Methods section. Note that the last measurement before the start of August was on 23rd July, and the first measurement after the end of August was on 13th September; the red lines for *Typha* and *Carex* stands span this period. The red line for the *Phragmites* stand continues until 11th October because one large efflux measurement in September was excluded by outlier detection after the August measurements were removed from the dataset.

3.4 DISCUSSION

The *Carex* stand showed the highest effluxes observed at the study site. *Carex* transports oxygen (O_2) to the rhizosphere by diffusion (Ding et al. 2004). In contrast, *Phragmites* and *Typha* are able to transport O_2 to the rhizosphere by convective flow (Brix et al. 1992), which may result in effective oxidation of CH_4 before it is released to the atmosphere when the O_2 demand of the soil exceeds the O_2 supply by the plants (Fritz et al. 2011). Since the study site was flooded in summer and saturated during the rest of the year a strong O_2 demand during the study period was likely.

CH_4 effluxes from the study site (18–48 g m^{-2} a^{-1}) are similar to CH_4 effluxes from boreal pristine fens (12–66 g m^{-2} a^{-1}, Dise 1992; 27–63 g m^{-2} a^{-1}, Martikainen et al. 1995; 20–35 g m^{-2} a^{-1}, Nykänen et al. 1995) and temperate pristine fens (55–120 g m^{-2} a^{-1}, Melloh & Crill 1996). In contrast, CH_4 effluxes from other fens in north-east Germany that were re-wetted less than five years before CH_4 measurements are an order of magnitude greater when the ecosystem is flooded (up to 493 g m^{-2} a^{-1}, derived from Augustin & Chojnicki 2008; up to 267 g m^{-2} a^{-1}, Höper et al. 2008; 147 g m^{-2} a^{-1}, derived from Glatzel et al. 2011). Our findings suggest that CH_4 effluxes will become similar to those from pristine fens 15 years after re-wetting, even with a summer flood,.

Water table is recognised as a major control for both annual (Fiedler & Sommer 2000, Couwenberg et al. 2011) and seasonal CH_4 effluxes (Hargreaves & Fowler 1998, Jungkunst et al. 2008, Huth et al. 2012). Generalised linear modelling of water table *vs*. CH_4 effluxes explained 12 % ($P < 0.01$, *Phragmites*), 42 % ($P < 0.001$, *Typha*) and 11 % ($P < 0.05$, *Carex*) of the overall CH_4 variability (R^2 values) in this study. Hence, the impact of freshwater flooding on CH_4 effluxes at the *Phragmites* and *Carex* plots was small, whereas it explained almost half of the CH_4 efflux variability at the *Typha* plot. The peat is similar among the measurement locations (Table 3.1), but the *Typha* stand has a larger fraction of easily decomposable grassland type understorey (e.g. *Poa trivialis*, L., unpublished results) than the *Phragmites* and *Carex* stands. Therefore, the difference in water table response may indicate that, in contrast to *Phragmites* and *Carex* stands, the *Typha* stand is not yet fully established.

This study highlightsthe finding that, if the water table is close to the ground surface (Bönsel & Sonneck 2012), and the shift in vegetation composition already reflects the altered hydrology (Bönsel & Sonneck 2011), as it does in the Trebel valley mire, then the CH_4 effluxes from re-wetted fens become similar to those from pristine fens even under extremely wet conditions. Therefore, our results are very promising because they show that re-wetting projects can reduce GHG effluxes within 15 years.

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4 The effect of biomass harvesting on greenhouse gas emissions from a rewetted temperate fen[3]

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SUMMARY

The growing demand for bioenergy increases pressure on peatlands. The novel strategy of wet peatlands agriculture (paludiculture) may permit the production of bioenergy from biomass while avoiding large greenhouse gas emissions as occur during conventional crop cultivation on drained peat soils. Herein, we present the first greenhouse gas balances of a simulated paludiculture to assess its suitability as a biomass source from a climatic perspective. In a rewetted peatland, we performed closed-chamber measurements of carbon dioxide, methane, and nitrous oxide exchange in stands of the potential crops *Phragmites australis*, *Typha latifolia*, and *Carex acutiformis* for two consecutive years. To simulate harvest, the biomass of half of the measurement spots was removed once per year. Carbon dioxide exchange was close to neutral in all tested stands. The effect of biomass harvest on the carbon dioxide exchange differed between the two years. During the first and second year, methane emissions were 13–63 g m^{-2} a^{-1} and 2–5 g m^{-2} a^{-1}, respectively. Nitrous oxide emissions lay below our detection limit. Net greenhouse gas balances in the study plots were close to being climate neutral during both years except for the *Carex* stand, which was a source of greenhouse gases in the first year (in CO_2-equivalents: 18 t ha^{-1} a^{-1}). Fifteen years after rewetting the net greenhouse gas balance of the study site was similar to those of pristine fens. In addition, we did not find a significant short-term effect of biomass harvest on net greenhouse gas balances. In our ecosystem, ~17 t ha^{-1} a^{-1} of CO_2-equivalent emissions are saved by rewetting compared to a drained state. Applying this figure to the fen area in northern Germany, emission savings of 2.8–8.5 Mt a^{-1} CO_2-equivalents could possibly be achieved by rewetting; this excludes additional savings by fossil fuel replacement.

KEY WORDS

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The effect of biomass harvesting on greenhouse gas emissions from a rewetted temperate fen

carbon dioxide, carex, methane, minerotrophic mire, paludiculture, phragmites, typha

4.1 INTRODUCTION

The late-20th century increase in global temperatures is likely attributable to the growing anthropogenic production and release of greenhouse gases (GHGs; Crowley 2000; Myhre et al. 2013). The GHGs carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) alone contribute 64 %, 17 % and 6 %, respectively, to the anthropogenic global radiative forcing (derived from Myhre et al. 2013).

One strategy to reduce CO_2 emissions to the atmosphere is to replace energy derived from fossil fuels with bioenergy from biomass. As a result, the gross production of bioenergy almost doubled in the OECD countries from 1990 to 2011 (International Energy Agency Statistics 2012). In the European Union alone an additional 18–21 million hectares of land will need to be converted to energy crop production to fulfil the goal of a self-imposed bioenergy share by 2020 (Özdemir et al. 2009). To avoid conflicts with food security, energy crop production will likely be restricted to marginal or abandoned agricultural land (Don et al. 2012). These marginal lands include a large fraction of formerly cultivated peatlands, which have been taken out of agricultural use due to decreasing yields caused by soil degradation (Joosten 2006). Consequently, the overall growth in energy crop cultivation affects organic soils (Shurpali et al. 2009; Röder & Grützmacher 2012) and will continue to do so.

Peatlands are an integral part of the global GHG cycle and must be accounted for under the Kyoto Protocol (United Nations Framework Convention on Climate Change 2012). Pristine peatlands store significant amounts of carbon (C) through the incomplete decomposition of organic matter (Clymo et al. 1998; Frolking et al. 2006). To establish conditions suitable for agricultural use, peatlands have been and are still being deeply drained (with water tables >60 cm below ground surface). Drained peatlands release large amounts of CO_2 (up to 50 t ha^{-1} a^{-1}; Couwenberg et al. 2011). They may furthermore become large sources of N_2O (up to 60 kg ha^{-1} a^{-1}; Couwenberg et al. 2011), a GHG 265 times more potent than CO_2 over the 100-year horizon (without climate-carbon feedbacks; Myhre et al. 2013). For these reasons, drained peatlands represent significant net GHG sources (Maljanen et al. 2010; Couwenberg et al. 2011). The increase in CO_2 emissions from peat soils following drainage often more than compensates for the reduction in CO_2 emissions due to bioenergy production (Couwenberg 2007; Minkkinen et al. 2008). Thus, one of the major goals in the cultivation of energy crops is counteracted.

If formerly drained peatlands are rewetted (by raising water tables close to ground surface), the potential for large CO_2 and N_2O emissions is significantly reduced (Hendriks et al. 2007; Couwenberg et al. 2011). Therefore, the rewetting of drained peatlands has become an important mitigation strategy for climate change since the mid-1990s (Erwin 2009). Undrained peatlands are globally significant sources of CH_4 (Nykänen et al. 1995; Melloh & Crill 1996). However, the net climatic effect of C sequestration vs. CH_4 emission in pristine peatlands is a slight warming or cooling (Frolking et al. 2006; Franzén et al. 2012).

In central Europe, most peatlands have been drained for agricultural reasons (Germany 86 %, Poland 70 %; Joosten & Couwenberg 2001). A large fraction of these peatlands are thick minerotrophic fens with peat depths of up to 12 m, representing significant C stores (Zauft et al. 2010). These extensive peat bodies now face the risk of oxidation, which is why the concept of paludiculture evolved in this region as an attempt to reduce GHG emissions.

Paludiculture is the agricultural use of wet and rewetted peatlands; it has been discussed as a solution to the problem of GHG emissions from drained peatlands because it combines the production of biomass for bioenergy while retaining the peat body (Wichtmann & Tanneberger 2011; Don et al. 2012; Joosten et al. 2012). This approach may avoid large emissions of CO_2 and N_2O. As a consequence, paludiculture is acknowledged in the IPCC Wetlands supplement as a possible land-use option on rewetted organic soils (Blain et al. 2014).

Suitable plants for biomass production on rewetted temperate fens include *Phragmites australis* (Cav.) Trin. ex Steud. (common reed), *Typha latifolia* L. (common bulrush) and *Carex acutiformis* Ehrh. (lesser pond sedge, Wichtmann & Tanneberger 2011). Köbbing et al. (2013) suggest that winter-harvested reed, with moisture contents of 15–20 %, has economic potential as an industrial raw material (e.g. for roof thatching, construction and gardening, paper and pulp) and as an energy source (combustion, biofuel). The calorific values of potential crops are for example 14–15 MJ kg^{-1} for *Phragmites* (10 % water content; Köbbing et al. 2013; Wichtmann et al. 2014), 18.2 MJ kg^{-1} on average for *Typha* (Briquettes, Pellets; Cicek et al. 2006), and 18.2–18.7 MJ kg^{-1} for *Carex* (Grzelak et al. 2011). With an oil equivalent of 42.5 MJ kg^{-1}, ~0.4 kg of fossil fuels may therefore be replaced by 1 kg of biomass from temperate fens (derived from Köbbing et al. 2013).

All of these crops have aerenchymatic tissues that allow them to grow in water-logged conditions. Aerenchymatic plants often play an important role in regulating CH_4 emissions (Whiting & Chanton 1996; Afreen et al. 2007). Up to 90 % of the CH_4 emissions from peatlands may pass through aerenchymatic plants (van der Nat & Middelburg 2000; Askaer et al. 2011). If these plants are harvested, CH_4 emissions will likely be affected. However, cutting experiments show inconsistent results; some studies suggest that CH_4 emissions from aerenchymatic plants decrease when these plants are cut (van der Nat & Middelburg 2000; Duan et al. 2006), whereas other studies only find an effect when plants are cut below the water surface (Ding et al. 2004; Juutinen et al. 2004).

Cutting removes only the aboveground part of the plant, which is not essential for peat formation in fens since it rapidly decomposes under natural (no-use) conditions (Schulz et al. 2011; Wichtmann & Tanneberger 2011). Cutting has been shown to increase aboveground biomass production in *Phragmites* (Granéli 1989; Ostendorp 1999) and *Carex* (Güsewell et al. 1998). However, no data on the effect of biomass harvesting on the net GHG balance of wet temperate fens are available (Blain et al. 2014). In addition, literature on the GHG balances in temperate fens several (>5) years after rewetting is scarce (Hendriks et al. 2007).

Here, we present the first GHG balances for a simulated paludiculture system of the potential crops *P. australis*, *T. latifolia*, and *C. acutiformis*, in order to assess their suitability as source of biomass for bioenergy production from a climatic perspective. In addition, we show the emission regime of a temperate fen 15 years after rewetting to draw conclusions about the potential of rewetting as a mitigation measure against climate change.

4.2 MATERIAL AND METHODS

4.2.1 Study site

The study site is located in the Trebel river valley mire complex in north-eastern Germany (54°06‘ N, 12°44‘ E, Figure 4.1). The site has a humid climate with a continental influence, a mean annual air temperature of 9.1 °C (reference period 1991–2010, German Weather Service), and an annual precipitation of 626 mm (reference period 1981–2010, German Weather Service). The mean August temperature is 19.1 °C, and the mean August precipitation is 64 mm. The climatic water balance is slightly positive with +50 to +100 mm a^{-1} (Klämt & Schwanitz 2001). The fen is a typical percolation mire of the southern Baltic region (Succow 2001). The peats are mainly of reed and sedge origin with depths ranging from 4 to 6 m. Since the late 1960s, deep drainage ditches (~1.5 m) have allowed high-intensity grassland use until the fen was rewetted in 1997, bringing and keeping the water table close to the ground surface (Bönsel & Sonneck 2012). Since then, the vegetation in the study site developed from a managed grassland with plants like *Agrostis* spp. L. (bentgrass), *Alopecurus geniculatus* L. (marsh foxtail) and *Phleum pratense* L. (timothy-grass) to a rewetted state with reeds of *P. australis, Carex* spp. L., or *Typha* spp. L. (Bönsel & Sonneck 2011).

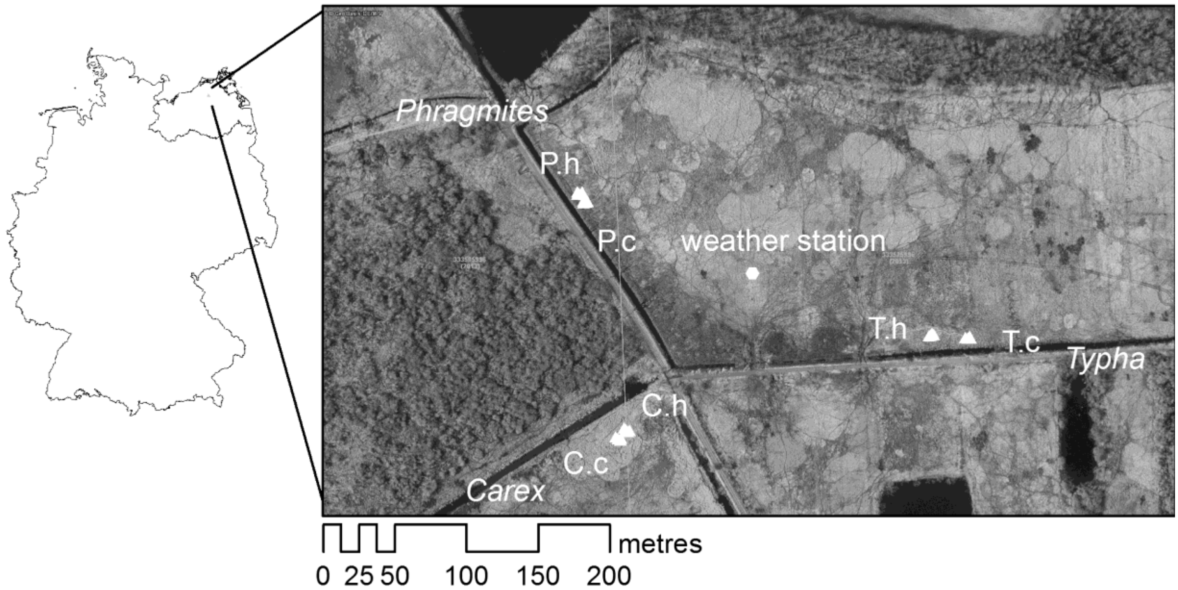

Figure 4.1: Left: location of the study site in north-eastern Germany. Right: locations of the weather station (hexagon) and the measurement plots (triangles) in *Phragmites-*, *Typha-* and *Carex*-dominated stands within the study site. P.h, T.h and C.h are the harvested plots and P.c, T.c, and C.c are the control plots.

4.2.2 Study plots

We selected one naturally grown dominant stand for each of the three macrophyte species *P. australis*, *T. latifolia* and *C. acutiformis.* In each plant stand, we installed six circular collars (h = 20 cm, d = 63 cm, insertion depth = 10 cm) arranged in two groups of three in November, 2010. The collars remained in the field for the entire investigation period (Figure 4.1) and were accessible by boardwalks to minimize disturbance during sampling. To simulate biomass harvest, the biomass inside and surrounding one group of three collars in each dominant stand (ca. 5 × 7 m) was cut ~10 cm above the ground surface and removed once per year. The other group of three collars in each plant stand (controls) were left uncut during the measurement period. Therefore, the setup was a compromise between covering spatial variability of GHG exchange and the necessary spatial aggregation needed for the large number of repetitions during measurements of CO_2 exchange (see section *Greenhouse gas measurements and sampling procedure*). In addition, we wanted to simulate conditions in a paludiculture field where large harvested areas lead to altered micrometeorological conditions, and the potential crop plants form large connected gas spaces via their rhizomes. Fertilizer was neither applied on the control nor on the harvested plots. In the remaining text, the harvested plots of each stand are referred to as P.h, T.h and C.h (for *Phragmites*, *Typha*, and *Carex*, respectively) whereas the control plots are referred to as P.c, T.c and C.c (see Figure 4.1).

Data were recorded from March, 2011 to March, 2013. In the following, the time period from 14 March, 2011 to 13 March, 2012 will be referred to as ‘year 1’, and the period from 14 March, 2012 to 13 March, 2013 as ‘year 2’.

4.2.3 Environmental, soil and biomass parameters

A weather station was installed close to the measurement plots (Figure 4.1) to record hourly data on soil temperatures at 2, 5 and 10 cm depths, air temperature, photosynthetic photon flux density (PPFD) and rainfall. Groundwater pipes were installed at every plot to record water table readings every 30 minutes using automated loggers (Levellogger Gold Junior M5, Solinst, Canada). Water samples were taken biweekly from the same pipes for analyses of nitrate and ammonium contents, pH as well as electric conductivity (EC). Soil organic carbon (SOC) and C/N ratio were estimated from peat cores that were sampled at each vegetation stand (n = 5 per stand) once during the study period in November, 2011. We recorded vegetation composition for each study plot to monitor vegetation changes following biomass harvesting in July, 2011, 2012, and 2013. *Carex* is resilient to summer mowing, whereas *Phragmites* and *Typha* are not (Briemle & Ellenberg 1994). Therefore, we harvested *Carex* biomass in July and the *Phragmites* and *Typha* biomass in January. To establish the experiment, we also removed *Phragmites* and *Typha* biomass from the harvested plots directly before the start of the measurements in March, 2011. Biomass was removed by manually cutting P.h and T.h on 9 March, 2011, 30 January, 2012, 29 January, 2013, and C.h on 16 July, 2011 and 27 July, 2012. The harvested biomass was dried in the lab for 24 h at 65 °C and analysed for C content by an element analyser (vario EL, Elementar Analysensysteme, Germany).

4.2.4 Greenhouse gas measurements and sampling procedure

We used flexible, height-adjustable, closed chambers (enclosed volume of ~0.6 m^3, d = 63 cm) for all flux estimations (see also Günther et al. 2014a). For CO_2 measurements, the chambers consisted of a transparent polycarbonate lid connected to transparent polyurethane walls and were equipped with fans and a temperature sensor. During measurements of net ecosystem exchange (NEE), the chamber lid was set up on three poles and the flexible chamber wall was gently lowered to the ground where it was attached to the collar. For dark measurements of ecosystem respiration (R_{eco}), the chamber was covered with an opaque polyurethane hood. NEE and R_{eco} of each dominant stand were determined every six weeks during the vegetation period. Outside the vegetation period, only R_{eco} was measured on each collar once every six weeks. On measurement days, we conducted multiple measurements of NEE at different PPFD levels in one collar, starting at sunrise and ending around midday when PPFD was no longer rising. R_{eco} measurements were conducted over a range of soil temperatures two to three times per measurement day. During the measurements, CO_2 concentrations (in ppm) in the chamber headspace were monitored with a portable infrared gas analyser (EGM-4, PP Systems, USA). At the same time, PPFD was monitored with a quantum sensor (Indium Sensor, Germany). All sensor readings were recorded in ~1.5 s intervals. Temperatures were monitored inside and outside the chamber, as well as at 2, 5, and 10 cm beneath the soil surface (F & C, Germany); temperatures were recorded simultaneously with the CO_2 data. During measurements of NEE (using transparent chambers), the chambers were cooled by freezer packs installed near the fans. Despite this precaution, temperatures rose more than 2 °C from the beginning of the measurement in 16 % of measurements. NEE and R_{eco} measurements lasted between 120 and 300 s.

We conducted CH_4 and N_2O measurements biweekly throughout the year with chambers similar to those described above, but manufactured from opaque, thermoplastic polyurethane (walls) and polyvinyl chloride (lid). Apart from that, CH_4 (and N_2O) chamber installation and handling was similar to the CO_2 measurements. Sampling usually occurred between 8 a.m. and 4 p.m., and sampling order was randomized to account for diurnal variability. Five gas samples per chamber placement were taken every ten minutes using evacuated glass containers (V = 12 ml, Labco, UK) during a closure time of 40 minutes. Within one week after field sampling, gas samples were analysed for CH_4 and N_2O concentration with a gas chromatograph (Shimadzu, Japan) equipped with a flame ionization detector (FID) and electron capture detector (ECD).

Statistical analyses were performed in R 3.0.1 (R Core Team 2013). For estimating gas fluxes and fitting of NEE- and R_{eco}-models, we used the R package 'flux' (Jurasinski et al. 2013). All means are given as ± one standard error. Because the data were not normally distributed in all cases, the differences in gas fluxes and environmental parameters among the plots were tested for significance using the Wilcoxon signed-rank-test with Bonferroni adjustment of P values. We follow the atmospheric sign convention, where positive values represent gas fluxes from the soil to the atmosphere.

4.2.5 Modelling CO_2 exchange

CO_2 exchange rates were estimated based on linear regressions of the change in concentration values over time (aiming for maximization of R^2). The function fluxx of the 'flux' package detects the most linear part of the concentration change vs. time graph whilst omitting outliers and non-linear parts. We executed the function so that at least 50 % of the data points were retained. These settings result in a minimum number of 40 concentration measurements for each flux estimation (the minimum closure time of 120 s corresponds to 80 concentration measurements). CO_2 fluxes (f, g m^{-2} h^{-1}) were then calculated using formula 4.1,

$$f = \frac{MpV}{RTA} \cdot \frac{\mathrm{d}c}{\mathrm{d}t} 10^6 \tag{4.1}$$

where M is the molar mass of CO_2 (g mol^{-1}), p is air pressure (Pa), V is chamber volume (m^3), R is the gas constant (m^3 Pa K^{-1} mol^{-1}), T is temperature inside the chamber (K), A is ground surface area (m^2) and dc/dt is concentration change over time (ppm h^{-1}).

We modelled CO_2 exchange in each plot by combining NEE measurements from all three spots. NEE integrates two processes, gross primary production (GPP) and ecosystem respiration (R_{eco}):

$$NEE = GPP + R_{eco} \tag{4.2}$$

R_{eco} was modelled by optimizing the parameters R_{ref} and $E0$ of formula 4.3,

$$R_{eco} = R_{ref} \cdot e^{E0\left[\frac{1}{T_{ref}-T_0}-\frac{1}{T-T_0}\right]} \tag{4.3}$$

where R_{ref} is the CO_2 release (g m^{-2} h^{-1}) from respiration at the reference temperature T_{ref} (283.5 K), *E0* is the activation energy (K), *T0* is the temperature constant at the start of biological process (237.48 K) and *T* is the temperature during the measurements (Lloyd & Taylor 1994). During modelling, *T* could either be the air temperature or one of the soil temperatures at 2, 5, or 10 cm depth, depending on the best fit (lowest Akaike Information Criterion). We modelled R_{eco} over an entire year of measurement per vegetation and treatment. In C.h, where biomass was harvested in summer, we fitted two models per year (one before and one after the harvest).

With the fitted values for R_{ref} and *E0,* we modelled R_{eco} during the NEE measurements using logger temperature readings of the best fit temperature and subtracted it from the measured NEE fluxes. The resulting estimated GPP CO_2 fluxes (g m^{-2} s^{-1}) were then modelled against PPFD using formula 4.4,

$$GPP = \frac{GP_{max} \cdot \alpha PPFD}{GP_{max} + \alpha PPFD} \tag{4.4}$$

where α is the initial slope of the regression and GP_{max} (g m^{-2} s^{-1}) is the boundary value of primary CO_2 production at infinitely high PPFD.

We used the R_{eco} and GPP models to predict hourly fluxes with data from the weather station to obtain annual CO_2 exchange values. To do so, we used the respective temperature data to calculate hourly rates of $R_{eco,}$ and the PPFD data to calculate hourly rates of GPP. After biomass removal, the GPP model parameters α and GP_{max} were set to -0.0001. Directly before biomass removal, we assumed the model parameters of the preceding model. To interpolate GPP data between two measurement days, we initially calculated GPP rates using the (unchanged) model parameters of both measurement days. This left us with two GPP rates for each hour between the measurement days. We assumed that closer in time to one of the measurement days, the actual GPP rate would be better described by the model derived for that day than by the model derived for the more distant measurement day. Therefore, we weighed the two GPP rates according to temporal proximity to the model, which was used for their individual calculation. During this procedure, the values received large weights close to the measurement day for which the model was calculated, and small weights for the time when the adjacent model was calculated. Using these weights, we calculated hourly weighted means to get the final hourly GPP rates. The resulting hourly R_{eco} and GPP values were used to calculate hourly NEE values, which were summed up to obtain the annual CO_2 exchange.

We determined modelling errors (E_{mod}) by evaluating the difference between the observed (NEE_{obs}) and modelled NEE values (NEE_{mod}) following formula 4.5 (Aurela et al. 2002; Adkinson et al. 2011):

$$E_{mod} = \sqrt{\frac{(NEE_{obs} - NEE_{mod})^2}{(n-1) \cdot n}} \tag{4.5}$$

This implies that no single error terms are determined for GPP and R_{eco}. No consensus exists about how to propagate such modelling errors when estimating the uncertainty in the extrapolated annual balances (Kandel et al. 2013). Here, we decided to calculate modelling errors for each measurement date and linearly interpolate them between subsequent dates to get hourly NEE errors. The hourly values were then summed to yield annual errors.

4.2.6 *CH_4 and N_2O flux estimation*

To estimate fluxes of CH_4 and N_2O, we determined the change in concentration in the chamber headspace over the sampling time using the best linear fit (smallest normalized root mean square error, NRMSE) based on at least four out of five concentration measurements. When all of the models had $R^2 < 0.9$, the resulting flux was discarded (resulting in an exclusion of 394 out of the total 954 measurements). By this procedure, all fluxes remaining in the data set approximately met the level of significance ($P < 0.05$). Following Hütsch (2001), who found no evidence in the literature for uptake rates greater than -0.3 mg m^{-2} h^{-1}, even from ecosystems with a much higher potential for net CH_4 uptake (aerobic soils), we discarded all fluxes ($n = 5$) indicating a CH_4 uptake greater than -0.5 mg m^{-2} h^{-1} (conservative estimate). Additionally, fluxes lying beyond the upper 2.5 % tail of the log-norm-distribution of fluxes for each vegetation stand and treatment were regarded as outliers and excluded from further analyses ($n = 33$).

Different integration approaches are found in the literature for the determination of annual CH_4 balances from closed-chamber data (e.g. linear interpolation and integration using vegetation models). The determination of uncertainties is also variable (Kandel et al. 2013). Since our data did not allow for statistical modelling using environmental parameters, we used a combination of bootstrap and jackknife methods for the estimation of annual balances. First, we mixed up the three spatial repetitions of each plot by randomly picking one of the available fluxes for each measurement day. We repeated this procedure to yield a total of 100 annual CH_4 flux data sets (bootstrap method). For each of these data sets, we estimated CH_4 balances using the function auc.mc in the R package ‘flux’. This function repeatedly performs a linear integration of the fluxes over the covered time period, omitting one measurement date each time (jackknife method). We calculated the jackknife error $SE_{JK}(T_n)$ for each of the 100 data sets using the formula,

$$SE_{JK}(T_n) = \sqrt{\frac{(n-1)}{n} \times \sum(\bar{T}_{n-1} - T_{n-1})^2} \tag{4.6}$$

where n is the number of measurements in the original data set, T_{n-1} is the estimated balance of each subsample, and $\bar{T}_{n-1}$ is the mean of all estimated jackknife balances (Köhler et al. 2007). The reported annual balances represent the means of all jackknife estimates from the 100 data sets, and the errors of the annual balances represent the means of all 100 jackknife errors. This approach allows us to report robust annual estimates, including estimates of the uncertainty associated with linear interpolation during the integration of fluxes over a year.

The global warming potentials (GWP) of CH_4 and N_2O emissions were calculated using a 100-year time horizon as 28 CO_2-equivalents for CH_4 and 265 CO_2-equivalents for N_2O (Myhre et al. 2013). The errors in the net GHG and C balances were obtained by integrating the individual errors following the law of error propagation.

4.3 RESULTS

4.3.1 Environmental, soil, and biomass parameters

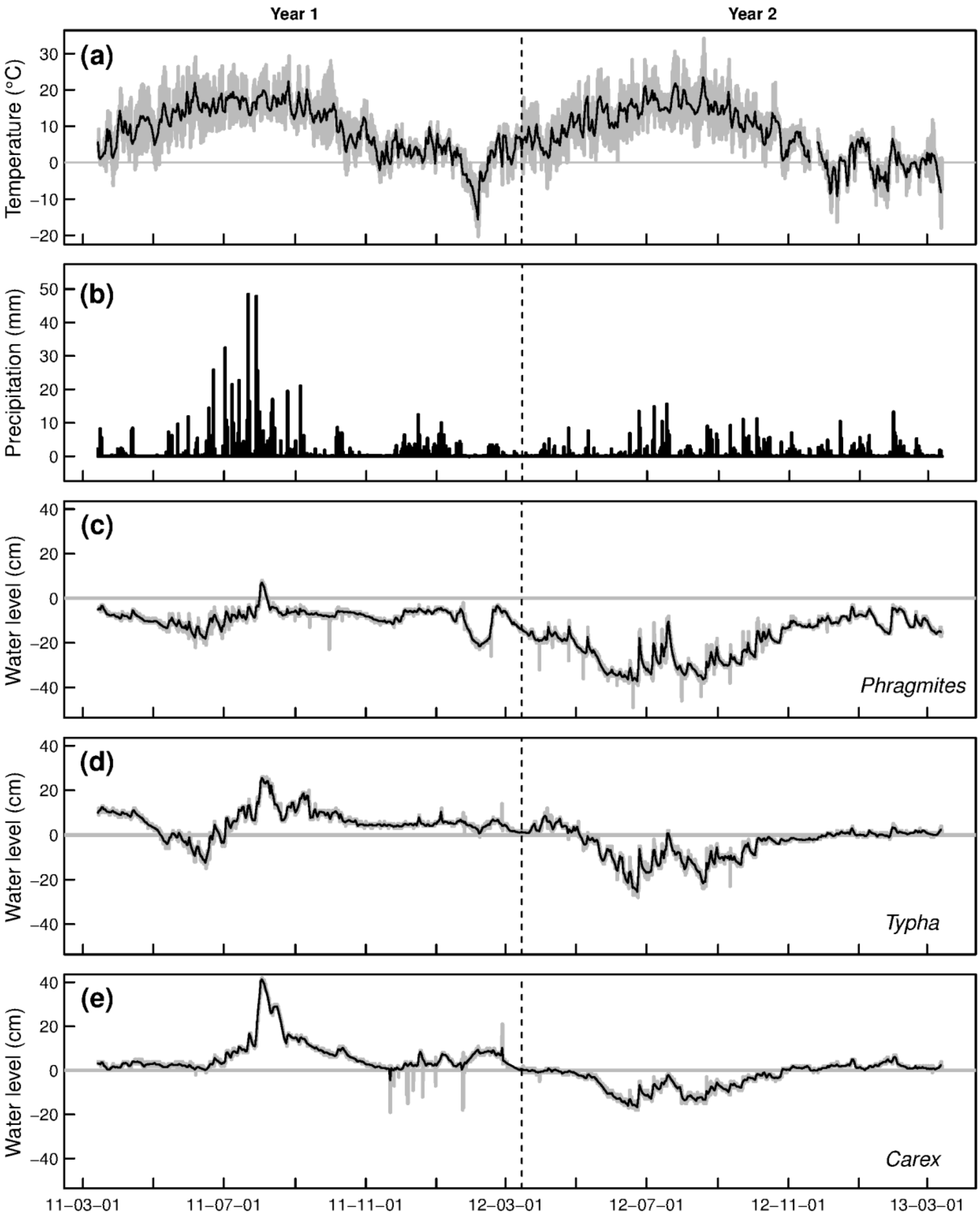

Figure 4.2: (a) Air temperature (grey: hourly means, black: daily means), (b) daily precipitation, and water table relative to the ground surface (grey: half hourly means, black: daily means) within the dominant stands *Phragmites* (c), *Typha* (d) and *Carex* (e) during the two study years. Note that during one week in November, 2012, the weather station failed to record data.

The two measurement years strongly differed with respect to precipitation (Figure 4.2). Year 1 was characterized by high precipitation, especially during the summer months (473 mm from June to August, 2011 vs. 201 mm long term mean), resulting in inundation in parts of the study area in August. Total precipitation during year 1 was 821 mm, and the mean annual water table was 2 cm above the ground surface (Table 1). In contrast, year 2 was rather dry with a total precipitation of 481 mm (168 mm from June to August 2012) and a mean annual water table of -9 cm relative to ground surface. Both years were cooler than the long term mean of 9.1 °C (year 1: 9.0 °C, year 2: 8.0 °C). The analysed soil parameters indicate a high degree of decomposition (C/N ratio 11–13, H9–H10 von Post scale, see Table 4.1). Fen water samples show mesotrophic conditions in all investigated stands during the first year and eutrophic conditions during the second year (Table 4.1). The *Carex* stand was poorest in nutrients for both years.

The composition of plant species was similar between the harvested and control plots at the beginning of the study and did not change considerably over the study period. *C. acutiformis* was most dominant (Table 4.1), with only scarce occurrences of understory plants such as *Equisetum fluviatile* L. (water horsetail) or *Solanum dulcamara* L. (bittersweet). The *P. australis* plots were intermingled predominantly with *Urtica dioica* L. s. l. (common nettle) and *Poa trivialis* L. s. l. (rough meadow-grass). The *T. latifolia* plots were the most species-rich, showing a mosaic of plants including *Agrostis stolonifera* L. (creeping bent), *Mentha aquatica* L. (water mint), *E. fluviatile*, and *Poa palustris* L. (swamp meadow-grass).

Biomass yields halved from year 1 to year 2 in *Phragmites* and *Carex* stands. In contrast, biomass yields in the *Typha* stand were higher in year 2 (Table 4.1).

Table 4.1: Plot Characteristics: dominant species with percentage cover (mean of three vegetation surveys); degree of decomposition (von Post scale), soil organic carbon (SOC, g C g^{-1} soil) and C/N-ratio estimated from peat cores (n = 5) of upper 30 cm; nitrate (NO_3^-) and ammonium (NH_4^+) concentrations (µmol l^{-1}); pH and EC (µS) of the fen pore water during the measurement period (year 1: n = 22, year 2: n = 21), mean water table (WL, cm relative to soil surface) and biomass (g m^{-2} a^{-1}, P.h and T.h – determined in January, C.h – determined in July).

			P.h	P.c	T.h	T.c	C.h	C.c
Vegetation	Year 1 and 2	Dominant species	*Phragmites australis*		*Typha latifolia*		*Carex acutiformis*	
		Cover [%]	60	70	50	40	90	90
Soil	Year 1 and 2	Decomposition	H9–H10		H9–H10		H9–H10	
		SOC	0.3 ± 0.01		0.3 ± 0.02		0.4 ± 0.01	
		C/N	11.3 ± 0.1		11.4 ± 0.3		13.0 ± 0.2	
Water	Year 1	NO_3^-	42.9 ± 3.7	21.7 ± 1.9	51.9 ± 1.5	35.9 ± 3.5	22.4 ± 2.9	12.6 ± 1.2
		NH_4^+	4.1 ± 0.9	8.4 ± 1.0	47.2 ± 7.8	47.4 ± 10.9	8.6 ± 2.0	15.2 ± 1.6
		pH	8.3 ± 0.1	8.3 ± 0.1	7.9 ± 0.1	8.2 ± 0.1	7.9 ± 0.1	7.9 ± 0.1
		EC	566 ± 49	534 ± 13	734 ± 27	732 ± 41	500 ± 83	484 ± 13
		Mean WL	0	-9	3	6	10	5
	Year 2	NO_3^-	282.1 ± 17.5	206.1 ± 13.4	265.2 ± 12.6	271.1 ± 13.5	177.5 ± 10.6	163.8 ± 10.0
		NH_4^+	2.7 ± 0.9	1.3 ± 0.1	5.4 ± 2.8	6.4 ± 1.4	11.0 ± 9.0	2.7 ± 1.4
		pH	7.9 ± 0.1	7.8 ± 0.1	8.0 ± 0.0	8.1 ± 0.0	8.0 ± 0.0	8.0 ± 0.0
		EC	401 ± 12	430 ± 14	697 ± 45	708 ± 47	397 ± 34	368 ± 15
		Mean WL	-20	-19	-6	-4	-1	-3
Biomass	Year 1	DM	702 ± 152	-	402 ± 67	-	854 ± 115	-
		C	324 ± 70	-	176 ± 29	-	367 ± 49	-
	Year 2	DM	372 ± 66	-	664 ± 154	-	345 ± 24	-
		C	169 ± 31	-	291 ± 65	-	146 ± 10	-

4.3.2 CO_2 exchange

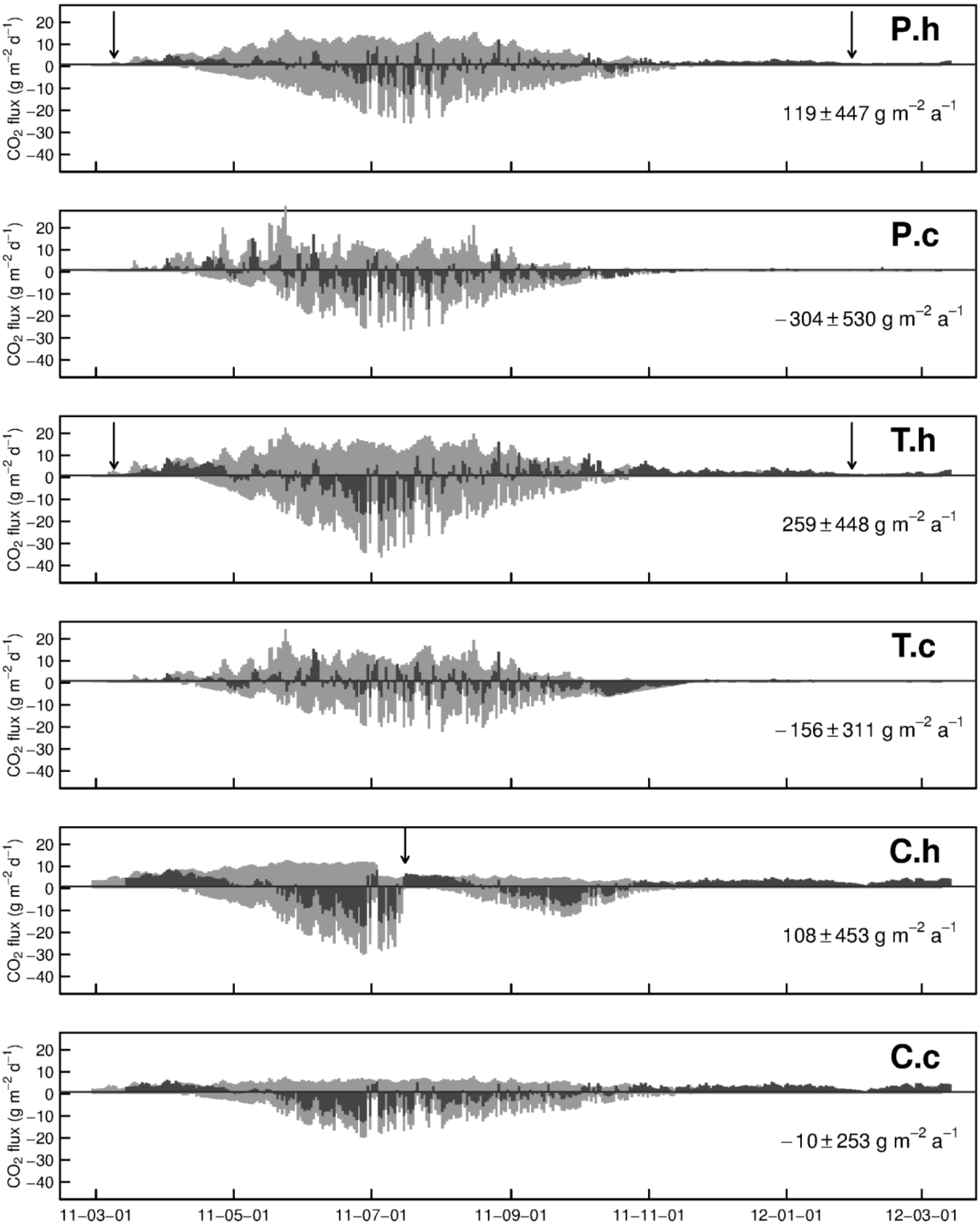

Figure 4.3: Modelled daily values of R_{eco} (light grey, values >0 g m^{-2} d^{-1}), GPP (light grey, values < 0 g m^{-2} d^{-1}) and NEE (dark grey), together with annual NEE (g m^{-2} a^{-1}) from harvested (P.h, T.h, C.h) and control plots (P.c, T.c, C.c) during year 1. Arrows show the dates of biomass harvesting.

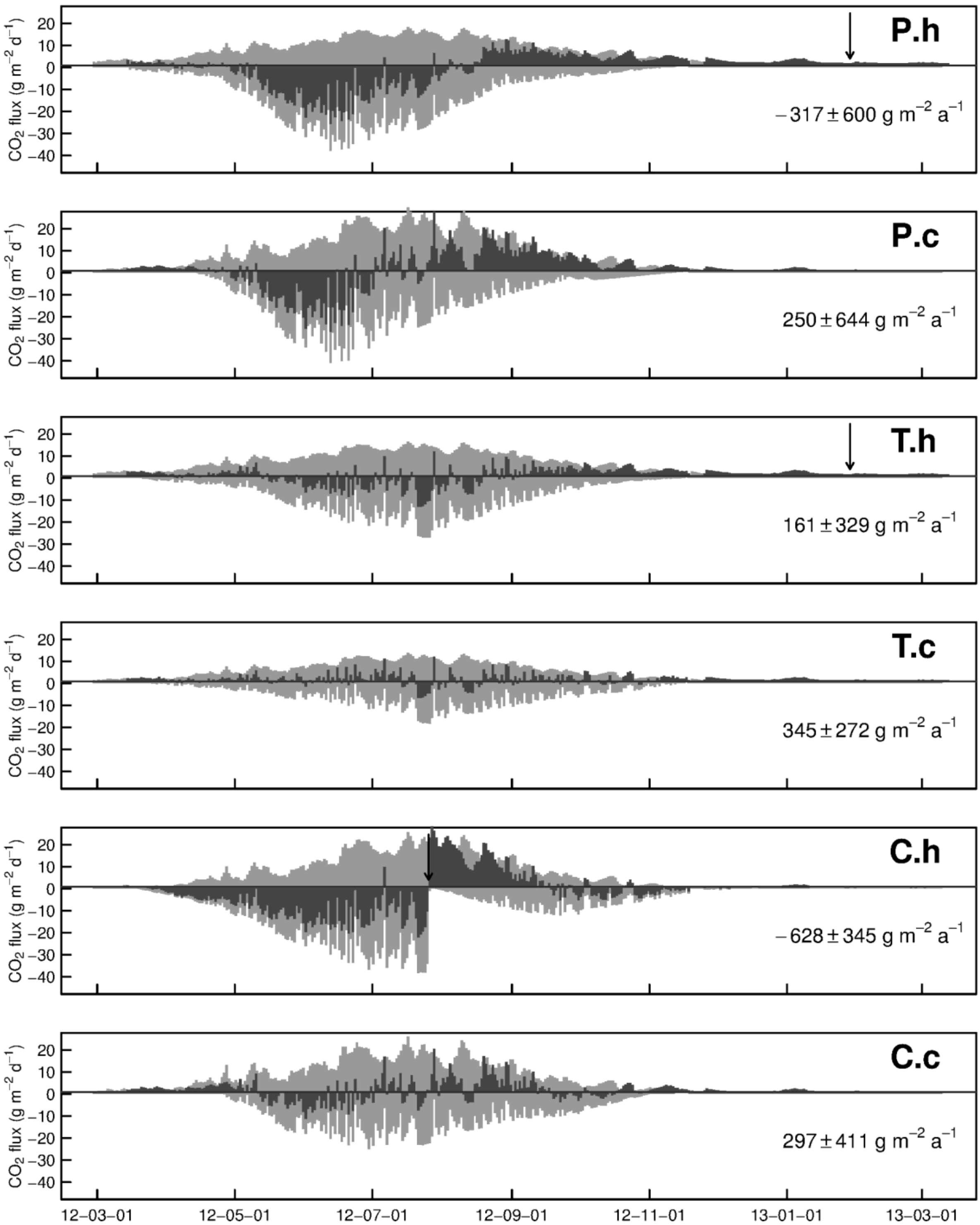

Figure 4.4: Modelled daily values of R_{eco} (light grey, values >0 g m^{-2} d^{-1}), GPP (light grey, values <0 g m^{-2} d^{-1}) and NEE (dark grey), together with annual NEE (g m^{-2} a^{-1}) from harvested (P.h, T.h, C.h) and control plots (P.c, T.c, C.c) during year 2. Arrows show the dates of biomass harvesting.

Modelled rates of GPP and R_{eco} showed seasonality in both years (Figures 4.3 and 4.4). Outside the vegetation period, no CO_2 was taken up by the plants. Similarly, R_{eco} was reduced to only 0.05 ±

0.05 g m^{-2} h^{-1} CO_2 during winter (December to February) of year 1 (June to August: 0.5 ± 0.2 g m^{-2} h^{-1}) and to 0.03 ± 0.02 g m^{-2} h^{-1} during winter of year 2 (June to August: 0.6 ± 0.2 g m^{-2} h^{-1}).

During year 1, the overall rates of R_{eco} for the harvested treatments were larger than those of the controls, whereas the harvested treatments showed higher photosynthetic activity than the controls (Figure 4.5). As NEE represents the (small) difference between GPP and R_{eco}, the combination of the uncertainties in GPP and R_{eco} balances leads to large relative errors in the annual NEE balances. As a result, the direction of the annual NEE flux could not be ultimately determined. Nonetheless, during year 1, all harvested treatments irrespective of dominant vegetation appeared to be small net CO_2 sources, while the controls were small net CO_2 sinks. The largest CO_2 uptake occurred in P.c (-304 ± 530 g m^{-2} a^{-1}), while T.h was the largest CO_2 source (259 ± 448 g m^{-2} a^{-1}, Table 4.2).

During year 2, the overall rates of R_{eco} were not affected by harvesting, while the harvested treatments again showed higher photosynthetic activity than the controls (Figure 4.5). Therefore, the harvested sites generally sequestered slightly more CO_2 than the controls in year 2. Nevertheless, only two treatments seemed to be net CO_2 sinks (P.h and C.h). The largest CO_2 uptake occurred in C.h (-628 ± 345 g m^{-2} a^{-1}), while T.c was the strongest CO_2-source (345 ± 272 g m^{-2} a^{-1}).

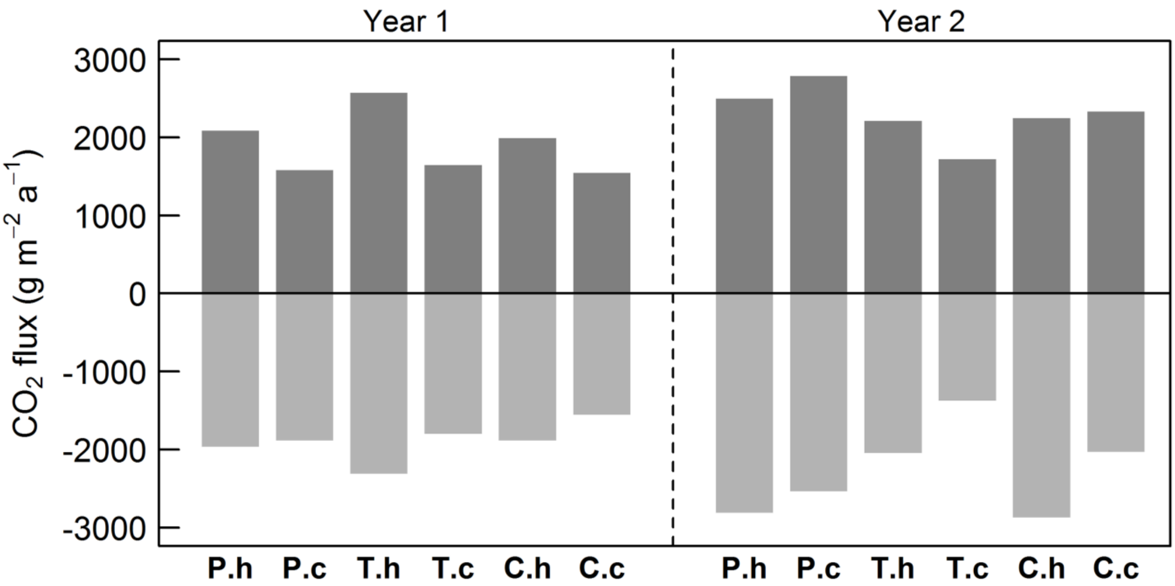

Figure 4.5: Annual balances of R_{eco} (dark grey, values >0 g m^{-2} a^{-1}) and GPP (light grey, values <0 g m^{-2} a^{-1}) for the six treatments during both years. Error bars are not shown because annual errors were only calculated for total NEE and not separately for R_{eco} and GPP.

4.3.3 *CH_4 and N_2O fluxes*

During year 1, fluxes showed distinct seasonality with winter fluxes (December – February) being smaller than summer fluxes (June – August, Figure 4.6). C.h exhibited the maximum valid CH_4 flux (after excluding outliers) on 26 April, 2011, with a value of 31 mg m^{-2} h^{-1}. The largest annual CH_4

emission in year 1 occurred at C.c (63 ± 16 g m^{-2} a^{-1}). CH_4 fluxes in the *Carex* stands were significantly higher than in the *Phragmites* and *Typha* stands ($P < 0.001$).

During year 2, CH_4 fluxes showed no distinct seasonality and ranged only from -1 mg m^{-2} h^{-1} (C.h 26 October, 2012) to 2 mg m^{-2} h^{-1} (C.c, 28 August, 2012, Figure 4.7). In year 2, the largest annual CH_4 emissions were determined for C.h (5 ± 1 g m^{-2} a^{-1}). CH_4 fluxes in the *Phragmites* stands were significantly lower than in the *Typha* and *Carex* stands ($P < 0.001$). Harvesting did not have a significant effect on annual CH_4 balances in either year.

N_2O fluxes were below the detection limit throughout the investigation period. Additional measurements at the harvested plots with small chambers (d = 30 cm, h = 30 cm) employed to improve measurement precision (due to a smaller volume-area-ratio) also failed to find significant N_2O fluxes. Since N_2O fluxes were not significantly different from zero, they were skipped in all further analyses.

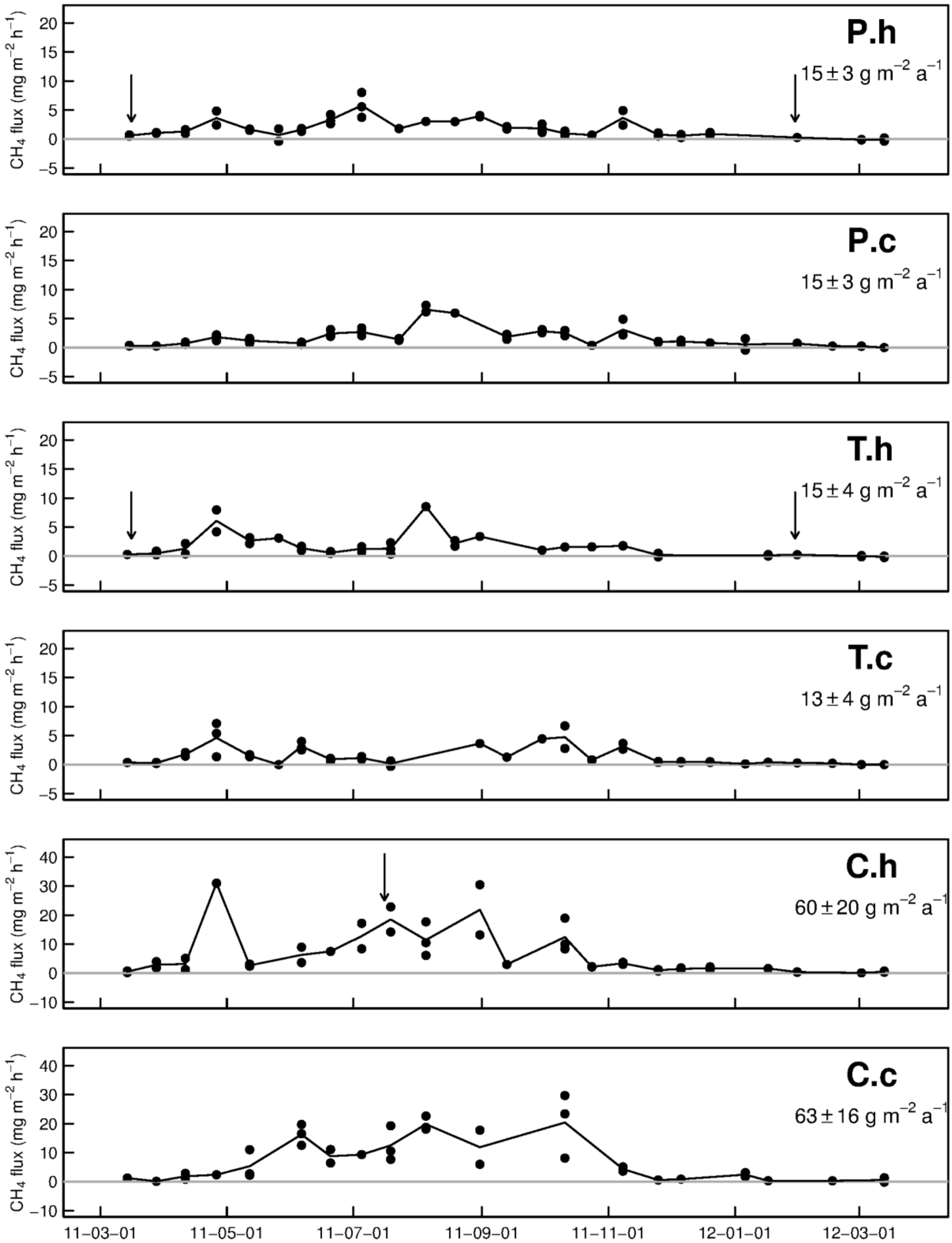

Figure 4.6: CH_4 fluxes during year 1 with annual emission estimates (g m^{-2} a^{-1}) for *Phragmites*, *Typha* and *Carex* stands in harvested (P.h, T.h, C.h) and control plots (P.c, T.c, C.c). Points represent single measurements and black lines represent courses of daily means. Arrows show the dates of biomass harvesting. Note the difference in scale between the C.h and C.c graphs.

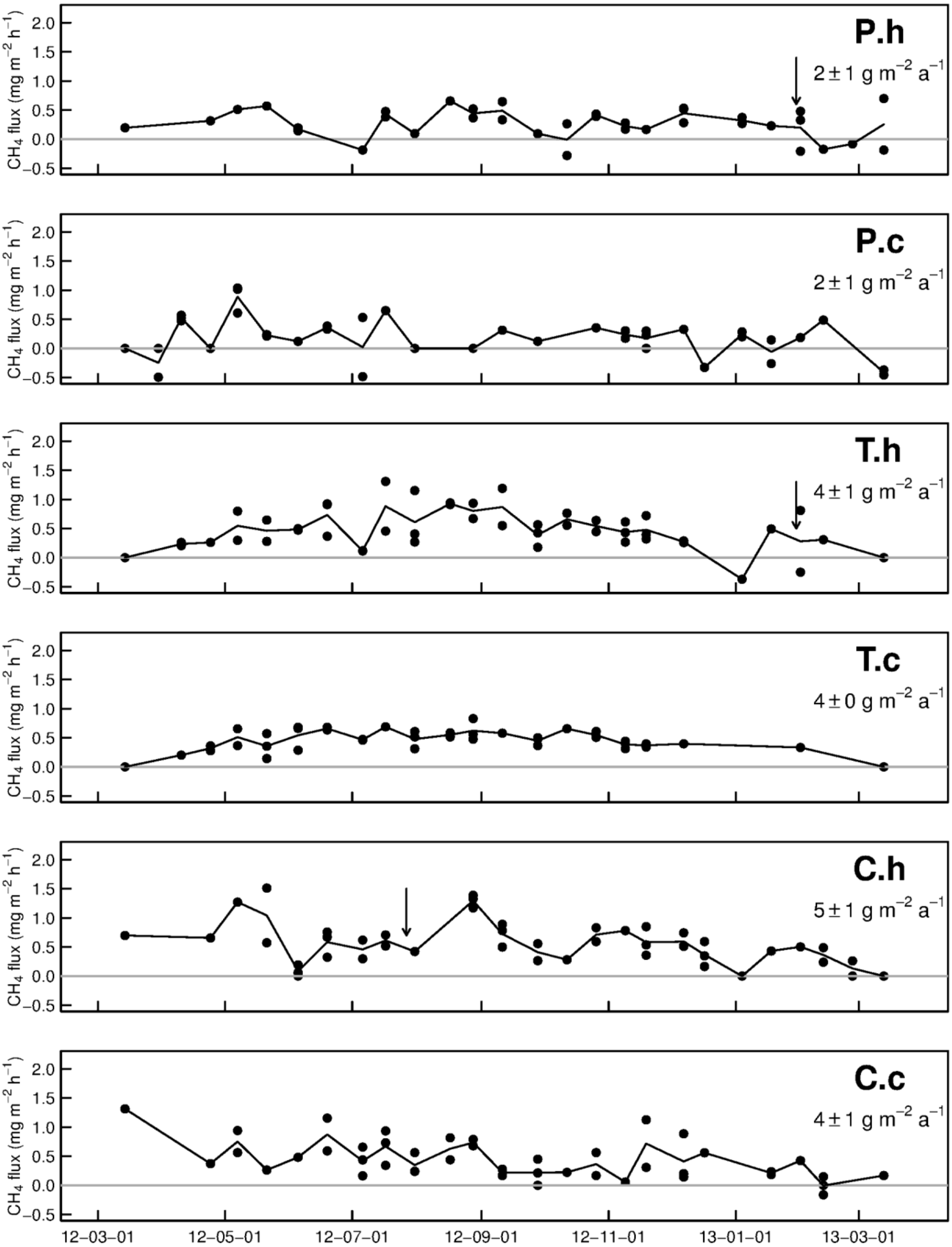

Figure 4.7: CH_4 fluxes during year 2 with annual emission estimates (g m^{-2} a^{-1}) for *Phragmites*, *Typha* and *Carex* stands in harvested (P.h, T.h, C.h) and control plots (P.c, T.c, C.c). Points represent single measurements and black lines represent courses of daily means. Arrows show the dates of biomass harvesting. Note that the y-axis scale is ten times smaller than in Figure 4.6.

4.3.4 Net GHG and C balance

Table 4.2: CO_2 and CH_4 emissions, net GHG and C balance, and harvested biomass (P.h and T.h – determined in January, C.h – determined in July) of the measurement plots in year 1. Given values are ± one standard error.

		P.h	P.c	T.h	T.c	C.h	C.c
CO_2	CO_2	119 ± 447	-304 ± 530	259 ± 448	-156 ± 311	108 ± 453	-10 ± 235
	CO_2-C	32 ± 122	-83 ± 145	71 ± 122	-43 ± 85	29 ± 124	-3 ± 64
CH_4	CH_4	15 ± 3	15 ± 3	15 ± 4	13 ± 4	59 ± 20	63 ± 16
	CH_4-C	11 ± 2	11 ± 2	11 ± 3	10 ± 2	44 ± 15	47 ± 12
	CO_2-eq	423 ± 73	417 ± 76	406 ± 106	358 ± 98	1660 ± 560	1770 ± 448
Sum GHGs	CO_2-eq	542 ± 453	113 ± 535	665 ± 460	202 ± 326	1768 ± 720	1760 ± 506
	C	44 ± 122	-72 ± 145	82 ± 122	-33 ± 85	74 ± 124	45 ± 65
	CO_2-eq ($t\ ha^{-1}\ a^{-1}$)	5 ± 5	1 ± 5	7 ± 5	2 ± 3	18 ± 7	18 ± 5
Biomass	C	324 ± 70	-	176 ± 29	-	367 ± 49	-
Sum C	C	368 ± 141	-72 ± 145	257 ± 126	-33 ± 85	441 ± 134	45 ± 65

During year 1, the *Phragmites* and *Typha* plots were small net GHG sources, whereas the *Carex* plots were relatively large GHG sources due to high CH_4 emissions (Table 4.2). Both *Carex* plots had the largest GHG emissions (C.h: 18 ± 7 t ha^{-1} a^{-1} CO_2-eq; C.c: 18 ± 5 t ha^{-1} a^{-1} CO_2-eq). Harvesting led to a slightly larger net release of GHGs from the *Phragmites* and *Typha* stands, but had no effect in the *Carex* stand. During year 2, P.h and C.h were net GHG sinks, whereas the other plots were small GHG sources. Harvesting caused the net release of GHGs from the *Typha* stand to be slightly smaller, and even resulted in a slight net uptake of GHGs in the *Phragmites* and *Carex* stands during year 2 (Table 4.3).

The C balances of the control plots were slightly lower than those of the harvested plots in year 1, whereas sequestration did not differ between harvested and control plots in year 2

Table 4.3: CO_2 and CH_4 emissions, net GHG and C balance, and harvested biomass (P.h and T.h – determined in January, C.h – determined in July) of the measurement plots in year 2. Given values are ± one standard error.

		P.h	P.c	T.h	T.c	C.h	C.c
CO_2	CO_2	-317 ± 600	250 ± 644	161 ± 329	345 ± 272	-628 ± 345	297 ± 411
	CO_2-C	-86 ± 164	68± 176	44 ± 90	94 ± 74	-171 ± 94	81 ± 112

CH_4	CH_4	2 ± 1	2 ± 1	3 ± 1	4 ± 0	5 ± 1	4 ± 1
	CH_4-C	2 ± 0	1 ± 1	2 ± 1	3 ± 0	4 ± 1	3 ± 1
	CO_2-eq	62 ± 17	42 ± 22	70 ± 20	101 ± 8	134 ± 25	109 ± 28
Sum GHGs	CO_2-eq	-255 ± 600	292 ± 644	231 ± 330	446 ± 272	-494 ± 346	406 ± 412
	C	-85 ± 164	69 ± 94	46 ± 90	97 ± 74	-168 ± 94	84 ± 112
	CO_2-eq (t ha^{-1} a^{-1})	-3 ± 6	3 ± 6	2 ±3	4 ± 3	-5 ± 3	4 ± 4
Biomass	C	169 ± 31	-	291 ± 65	-	146 ± 10	-
Sum C	C	84 ± 167	69 ± 94	337 ± 111	97 ± 74	-22 ± 95	84 ± 112

4.4 DISCUSSION

4.4.1 *CO_2 exchange*

With rates of CO_2 exchange ranging from an uptake of 6 t ha^{-1} a^{-1} (C.h, year 2) to a release of 3 t ha^{-1} a^{-1} (T.c, year 2), the study site represents a typical peatland with water tables less than 20 cm below the ground surface (Hendriks et al. 2007; Couwenberg et al. 2011; Blain et al. 2014). In contrast, deeply drained peatlands are typically strong CO_2 sources (18–50 t ha^{-1} a^{-1}: Maljanen et al. 2010; Couwenberg et al. 2011). Rewetting, which was carried out 15 years before our measurements started, seems to have effectively stopped peat oxidation and reduced CO_2 emissions by at least 15 t ha^{-1} a^{-1} compared to a drained state. Even during the relatively dry year 2, water tables remained at sufficiently high levels to prevent high rates of peat oxidation. Rates of CO_2 exchange between the three tested vegetation stands were not significantly different even though an especially high peat-forming potential has been suggested for *Phragmites* stands (Hartmann 1999; Richert et al. 2000). However, the comparatively low overall water table of the *Phragmites* stand and the resulting higher oxidation potential may explain our results.

During year 1, the harvested plots of all vegetation stands were minor CO_2 sources while the control plots were minor CO_2 sinks. This effect was reversed during year 2, when harvested plots sequestered more CO_2 than the controls. In both years, all harvested plots showed higher rates of GPP compared to the control plots. Rates of R_{eco} exhibited the same pattern in year 1, but not in year 2. Due to water saturation in year 1, it is likely that aerobic peat respiration played only a minor role; therefore, the higher rates of R_{eco} in year 1 may reflect increased plant stand respiration (and higher overall plant stand activity) due to the harvest events. Since year 2 was rather dry, peat respiration may have played a substantial role in ecosystem respiration, masking the effect seen in year 1.

Higher biomass production following the cutting of vegetation has been previously described: winter harvest increases culm density and standing crop in *Phragmites* (Granéli 1989; Ostendorp

1999), and *Carex* spp. shows higher biomass during the first four years of harvest (Güsewell et al. 1998). We found higher rates of GPP in the harvested plots; however, this did not result in higher biomass yields. A possible increase in biomass yields may have been masked by effects of experiment establishment, especially in the *Carex* stand, because values of the first harvest included the old biomass of previous years in addition to the biomass of year 1. Also, it is possible that more biomass was allocated to belowground plant parts in year 2 in order to replace root losses from the very wet year 1. Either way, overcompensation effects have only been observed during the first years of harvest and seem to be reversed in the long-term (Güsewell et al. 1998).

Overall, the amount of harvested biomass of *Carex* and *Phragmites* found in this study lies within the range reported in the literature (630-692 g m^{-2} a^{-1} DM for *C. acutiformis* – Bernard et al. 1988, 500-1000 g m^{-2} a^{-1} DM for *P. australis* – Granéli 1989, all values determined in summer). In contrast, our biomass yields in *Typha* (176 ± 29 and 291 ± 65 g m^{-2} a^{-1} DM) were slightly lower than literature values (330–1380 g m^{-2} a^{-1} DM, determined in winter – Maddison et al. 2009). However, biomass yields and, consequently, CO_2 balances of *Typha* stands are known to differ significantly between years (Maddison et al. 2009). Thus, the long-term effect of harvesting on the CO_2 balances in *Typha* stands cannot be assessed during only two years of study.

4.4.2 CH_4 and N_2O fluxes

Year 1 was very rainy; the precipitation from June to August alone (473 mm) equalled the complete precipitation of year 2 (481 mm). It is well known that the water table is an important factor in controlling CH_4 emissions (Couwenberg et al. 2011). Thus, the difference in precipitation may have been responsible for the 5-times (*Typha*), 10-times (*Phragmites*) and 15-times (*Carex*) higher CH_4 emissions in year 1 than in year 2.

During year 1, CH_4 emissions were highest in the *Carex* stands, and they were significantly higher than those in the *Phragmites* and *Typha* stands. *Carex* transports O_2 to the rhizosphere via diffusion only (Ding et al. 2004). In contrast, *Phragmites* and *Typha* are able to transport O_2 to the rhizosphere via convective flow (Brix et al. 1992), which may effectively oxidize CH_4 before it is released to the atmosphere when the O_2 demand of the soil exceeds the O_2 supply of the plants (Fritz et al. 2011). Since the study site was flooded in summer 2011 and saturated during the rest of year 1, there was likely a strong O_2 demand during this year.

During year 2, the lower CH_4 emissions can be ranked according to the mean annual water tables of the study plots (*Phragmites* < *Typha* < *Carex*, $P < 0.05$), following the typical correlation between CH_4 emissions and water table (Hargreaves & Fowler 1998; Huth et al. 2012).

Overall, the CH_4 emissions in the study site during year 1 (13–63 g m^{-2} a^{-1}) were within the range reported for undrained temperate organic soils (up to 70 g m^{-2} a^{-1}, Blain et al. 2014) and pristine fens (44–66 g m^{-2} a^{-1} – Dise et al. 1993, 55–120 g m^{-2} a^{-1} – Melloh & Crill 1996), whereas the CH_4

emissions in year 2 (2–5 g m^{-2} a^{-1}) were much smaller. The lower emissions during year 2 may have been caused by the drought during the summer of 2012, which likely resulted in aerobic decomposition and oxidation of produced methane.

Although some experiments reveal lowered CH_4 emissions directly after cutting (van der Nat & Middelburg 2000; Duan et al. 2006), one cut per year apparently does not significantly alter the annual CH_4 balances. The variability of CH_4 emissions in this study depends on differing weather conditions between the years (Günther et al. 2014a) and on vegetation. In the long term, cutting aerenchymatic plants may alter the redox equilibrium of the peat soil, affecting the balance of CH_4 production and oxidation. In general, plants of eutrophic fens (such as *P. australis* and *C. acutiformis*) have a lower CH_4 emission potential than plants of mesotrophic fens (Koelbener et al. 2010). Therefore, the long-term CH_4 emission potential of paludiculture likely depends on the nutrient state of the peat soil and the stability of the paludiculture plant stands.

N_2O emissions were not different from zero throughout the investigation period, which is similar to the results reported for a rewetted peat meadow in the Netherlands (Hendriks et al. 2007) and also typical of pristine fens (Martikainen et al. 1993; Nykänen et al. 1995). Therefore, the net GHG balance in the study site is determined by the CO_2 and CH_4 balance alone.

4.4.3 Net GHG balance and global implications

The net GHG balances of the study plots were close to climate neutral in both years except for the *Carex* stands in year 1 (18 t ha^{-1} a^{-1} CO_2-equivalents), which was clearly due to high CH_4 emissions, mainly during the exceptionally wet summer (Huth et al. 2013). Climate neutral GHG exchange was also found in an abandoned and rewetted peat meadow with water tables similar to the ones in our study area (Hendriks et al. 2007). In addition, the net GHG balances reported here were similar to the ones of pristine fens (Drewer et al. 2010); therefore, we conclude that rewetting was successful in re-establishing a natural GHG exchange regime in the study area. These results indicate that paludiculture is preferable to conventional agriculture on drained peatlands from a climatic perspective because ~17 t ha^{-1} a^{-1} in CO_2-equivalent emissions were saved (conservative estimate) in our system compared to the drained state (with CO_2-equivalent emissions of ~22–29 t ha^{-1} a^{-1}; Maljanen et al. 2010; Drösler et al. 2014).

Pristine fens provide the largest contribution to the CH_4 emissions from European peatlands (Saarnio et al. 2009). At the same time, the vegetation of temperate pristine fens is often dominated by aerenchymatic plant species known to considerably influence CH_4 emissions (Whiting & Chanton 1996; Afreen et al. 2007). Any short-term effect of biomass harvesting on CH_4 emissions can therefore be expected to be most distinct in ecosystems like those studied here; however, we did not observe such an effect.

The successful rewetting of degraded fens in Central Europe is difficult due to relatively low mean annual precipitation; hence, there is a need for a recharge of water from a catchment greater than the fen itself (Richert et al. 2000). However, percolation mires of the southern Baltic region, which constitute the major fen type in this area (approx. 50 %, Zauft et al. 2010), are usually fed through the larger catchments of rivers (Succow 2001). In addition, the technical implementation of paludiculture remains challenging. Whereas harvesting techniques for specialised products like reed for roof thatching are well established, efficient machines for harvesting energy biomass on wet peat soils are still under development (Komulainen et al. 2008; Wichtmann et al. 2014). Successful rewetting depends on multiple factors, and the technical implementation of paludiculture remains challenging. Nevertheless, the following outlines the potential of GHG emission reduction by rewetting and paludiculture:

Recent estimates of the potential area suitable for paludiculture on temperate fens in northern Germany vary from 20 % to 60 % of the overall temperate fen area (830,000 ha – Wichtmann 2003; Wichtmann et al. 2014). Assuming an emission reduction of ~17 t ha^{-1} a^{-1} (this study), the GHG mitigation potential of paludiculture (CO_2-equivalents) from temperate fens in northern Germany would range between 2.8 and 8.5 Mt a^{-1}. This reduction is equivalent to 4–12 % of the total emissions from German agriculture in 2011 (United Nations Framework Convention on Climate Change 2014). If we additionally assume a winter yield of 7 t ha^{-1} a^{-1} reed and the possibility of fossil fuel replacement by 0.4 t t^{-1} (derived from Köbbing et al. 2013) another 7 t ha^{-1} a^{-1} of emissions savings could be achieved. This number is similar to fossil fuel replacement by bioenergy production from maize (6.3 t ha^{-1} a^{-1}; Felten et al. 2013). In total, paludiculture in northern Germany could lead to GHG savings of approximately 24 t ha^{-1} a^{-1} CO_2-equivalents.

In Poland, Lithuania and Belarus 910,000 ha, 160,000 ha (derived from Joosten & Couwenberg 2001) and 925,000 ha (derived from Tanovitskaya 2011) of peatlands, respectively, have been drained for agriculture. Assuming the same mitigation potential from paludiculture as in Germany, the GHG savings (CO_2-equivalents) would sum to 3.1–9.3 (Poland), 0.5–1.6 (Lithuania), and 3.1–9.4 Mt a^{-1} (Belarus), accounting for 9–27 % (Poland), 11–33 % (Lithuania), and 13–40 % (Belarus) of the countries' GHG emissions from agriculture (United Nations Framework Convention on Climate Change 2014). Hence paludiculture has the potential to play a substantial role in mitigating GHG emissions from Central European agriculture.

In this study, we established a low-intensity paludiculture system (one cut per year, no fertilization) that is preferable to conventional agriculture from a climatic perspective because high CO_2 and N_2O emissions from peat oxidation are largely inhibited and CH_4 emissions remain at levels typical of pristine temperate fens. However, to increase yields and revenue for future applications, paludiculture will possibly be extended to higher-intensity cultivation including a higher harvesting frequency (two harvests per year) and fertilization. Such systems will probably behave differently in terms of GHG emissions, warranting further investigations.

We did not find a negative short-term effect of biomass harvest on CH_4 emissions and consequently on the net GHG balance. Therefore, we conclude that paludiculture is a sustainable alternative to traditional agricultural use on drained peatlands with high peat oxidation rates as long as harvesting frequency does not alter the equilibrium state of the ecosystem. However, the long-term effects of cutting on annual GHG balances in wet peatlands are unknown, and future studies are needed to assess the sustainability of GHG savings from paludiculture for bioenergy production.

In conclusion, paludiculture appears to provide solutions for both the growing demand for agricultural area for bioenergy and for the need to reduce GHG emissions.

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5 Divergent NEE balances from manual chamber CO_2 fluxes by different measurement and modelling approaches[4]

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SUMMARY

Manual closed chamber measurements are a common method for comparing ecosystem carbon dioxide (CO_2) balances of plants, sites or treatments in a wide range of terrestrial ecosystems. However, differences regarding both data acquisition and gap filling of manual closed chamber data are large in the existing literature, complicating inter-study comparisons. The aim of this study was to compare common approaches for quantifying CO_2 exchange at three methodological levels. (*i*) Two different ways of obtaining CO_2 fluxes; one via measurements from sunrise to noon (sunrise approach) and one via measurements during mid-day applying netted coverages (mid-day approach) to obtain a span of light conditions for measurements of net ecosystem exchange (NEE) with transparent chambers. (*ii*) Three different ways of pooling ecosystem respiration (R_{ECO}) fluxes for empirical modelling of R_{ECO}, i.e., campaign-wise (single measurement day R_{ECO} models), season-wise (one R_{ECO} model for the entire study period), and cluster-wise (two R_{ECO} models representing a low and a high vegetation status). (*iii*) Two different ways of deriving fluxes of gross primary production (GPP) from substracting either adjacent measured R_{ECO} fluxes (direct GPP modelling) from NEE fluxes or subtracting empirically modelled R_{ECO} fluxes from NEE fluxes (indirect GPP modelling). Measurements were done during 2013–2014 (one year) at a lucerne-clover-grass field in NE Germany. Across the different combinations of data acquisition and gap filling, the NEE balances of the agricultural field varied strongly. Balances derived from mid-day measurements were particularly sensitive to further gap filling strategies (-200 to 425 g CO_2-C m^{-2}). The campaign-wise R_{ECO}

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modelling approach was very sensitive to daily data acquisition (-101 to 425 g CO_2-C m^{-2}) and is therefore only advisable if the diurnal ranges of CO_2 fluxes and environmental parameters (e.g., photosynthetically active radiation and temperature) are sufficiently covered. The direct GPP modelling approach probably introduces an R_{ECO} data bias into the NEE modelling due to favoured times of diurnal data collection even when temperatures can be kept equal between transparent and opaque chamber measurements. NEE balances were most similar and closest to literature when derived from sunrise measurements and indirect GPP modelling (-101 to -131 g CO_2-C m^{-2}). Based on our case study, we recommend sunrise measurements with indirect GPP modelling and R_{ECO} pooling from adjacent measurement campaigns as long as pooling over, e.g., harvest events and other significant changes in plant development can be avoided. If the sunrise approach is not feasible, e.g., for extensive treatment comparisons, data pooling accounting for plant development is necessary. This can be achieved by the current clustering approach or by including more elaborate indices of plant development. Overall, the large variation in NEE balances resulting from different data acquisition or gap-filling strategies indicates that these decisions should be taken very carefully and, preferrably, a standard approach needs to be developed.

KEY WORDS

dynamic closed chambers, ecosystem respiration, net ecosystem exchange, lucerne, clover, ryegrass

Divergent NEE balances from manual chamber CO2 fluxes by different measurement and modelling approaches

5.1 INTRODUCTION

Manual closed chambers are commonly used to quantify terrestrial sources and sinks of CO_2 (Elsgaard et al. 2012, Beetz et al. 2013, Günther et al. 2015). Measurements are done at a high spatial resolution and in principle allow for plot-scale comparisons of, e.g., crops, sites or treatments (Rochette et al. 1991, Davidson et al. 2002). Yet, data acquisition is intermittent, often at weekly to monthly intervals, leaving large temporal gaps that usually account for >99 % of the study period. For this reason, the parent processes of ecosystem respiration (R_{ECO}) and gross primary production (GPP) are commonly modelled against continous environmental variables to reconstruct temporal CO_2 dynamics and to estimate CO_2 balances over longer periods of time. However, the comparably few field measurements in relation to the large temporal gaps generate a risk of low accuracy and high bias error on a temporal scale (Gomez-Casanovas et al. 2013), thus leading to potential deficits when estimating net CO_2 balances.

Gap-filling strategies of closed chamber R_{ECO} and GPP are based on systematic relationships between CO_2 fluxes and environmental parameters, such as temperature, photosynthetically active radiation (PAR), and green plant biomass (Burrows et al. 2005, Kandel et al. 2013, Renou-Wilson et al. 2014). However, several deductive or empirical approaches of gap filling have been used in different studies, and data acquisition to obtain the primary data of R_{ECO} and GPP also differs (e.g. Whiting et al. 1992, Bubier et al. 1998, Beetz et al. 2013, Günther et al. 2015). No consensus exists on a single standardised framework, but a literature survey shows that at least two distinct approaches are commonly applied, which differ fundamentally in at least three methodological levels related to (*i*) the CO_2 exchange measurements, (*ii*) the R_{ECO} data pooling, and (*iii*) the empirical GPP modelling.

In one approach, CO_2 fluxes are measured under mid-day conditions (typically 10 a.m. to 2 p.m.) with the use of shading to obtain responses to low PAR conditions (here referred to as mid-day measurements). This method is often used in combination with estimating GPP fluxes directly (here referred to as direct GPP) by subtracting opaque chamber R_{ECO} fluxes from preceding transparent (or shaded) chamber fluxes representing the net ecosystem exchange (NEE) of CO_2 at different PAR (Whiting et al. 1992, Carroll & Crill 1997, Elsgaard et al. 2012). For R_{ECO} gap filling, relationships between R_{ECO} and temperature are typically derived by pooling and analysing R_{ECO} data over meterological or plant physiological seasons (Drösler 2005) or even the entire study period (Alm et al. 1997, Yli-Petäys 2007, Elsgaard et al. 2012); here referred to as season-wise R_{ECO} modelling. Since R_{ECO} is both soil and plant dependent, R_{ECO} modelling parameters may change during different plant phenological stages, especially before and after harvest events. Therefore, the inclusion of vegetation proxies (Burrows et al. 2005, Kandel et al. 2013) or plant-dependent dynamic model parameters over the course of the study (Reichstein et al. 2005) has been suggested to improve seasonal R_{ECO} modelling. For GPP gap filling, GPP models are either derived for different plant phenological stages of the vegetation (Elsgaard et al. 2012) or may include a vegetation proxy to account for vegetation development over the study period (Alm et al. 1997, Yli-Petäys et al. 2007, Renou-Wilson et al. 2014). The typical combined approach of mid-day measurements, season-wise R_{ECO} modelling, and direct GPP estimation is here in short referred to as M/S/D.

A number of recent studies present a modified approach for data acquisition and temporal gap filling (Beetz et al. 2013, Leiber-Sauheitl et al. 2014, Hoffmann et al. 2015). Following this methodology, CO_2 fluxes are measured over the diurnal range of PAR and air/soil temperatures during the time from sunrise to early afternoon (hereafter referred to as sunrise measurements). The subsequent modelling relies on empirical R_{ECO} models that are based on data from individual campaigns (hereafter referred to as campaign-wise modelling) to account for changing plant phenological stages. GPP fluxes are obtained by substracting modelled R_{ECO} values from the NEE measurements (hereafter referred to as indirect GPP) rather than using the proximate measured R_{ECO} fluxes. In short this aproach is here referred to as S/C/I, i.e., sunrise measurements, campaign-wise R_{ECO} modelling and indirect GPP estimation. Compared with the mid-day measurement approach, the sunrise approach is associated with a workload approximately twice as high due to the increased number of measurements (Beetz et al. 2013, Leiber-Sauheitl et al. 2014, Hoffmann et al. 2015).

In summary, the resulting CO_2 balances derived from manual chamber measurements may be affected by the methodologies of (*i*) mid-day vs. sunrise measurements, (*ii*) campaign-wise vs. season-wise R_{ECO} modelling, and (*iii*) direct vs. indirect estimation of GPP fluxes. The two common approaches (M/S/D and S/C/I) use recurring combinations of these profoundly different methodological approaches, but other combinations have been used as well (Günther et al. 2015). Yet, it has remained untested to what extent each choice would produce systematic differences in resulting CO_2 balances. For this reason the present study was designed to:

(i) simultaneously apply both measurement approaches (sunrise vs. mid-day) on a lucerne/clover-grass field in NE Germany;

(ii) subsequently model R_{ECO} with campaign-wise, season-wise and cluster-wise data (where the cluster-wise modelling represents a simplified attempt to account for plant phenological stages by including green plant height as a variable to better separate the R_{ECO} data for improved R_{ECO} modelling);

(iii) compare the two procedures of using either direct or indirect GPP estimation.

We were thus able to quantify the effects of the individual factors on NEE modelling by comparing all 2 × 3 × 2 combinations of time of measurement × R_{ECO} pooling × GPP estimation. We tested the effect of each factor on the flux results by means of leave-one-campaign out cross-validation, and via the variation of NEE balances induced by the individual factor. Finally, we also compared the two common approaches, M/S/D and S/C/I.

5.2 MATERIAL AND METHODS

5.2.1 Study site

The study site (62 m a.s.l.) is part of a long-term research field at the Leibniz Centre for Agricultural Landscape Research (ZALF) located approximately 40 km east of Berlin, Germany (52°31' N, 14°07' E). The climate is sub-humid with a strong continental influence. The long-term (1992–2010) mean annual temperature is 9 °C and mean annual precipitation is 560 mm (German Weather Service 2015). July is on average the warmest month (18.9 °C) and January is the coldest (-0.2 °C). The driest and wettest months are on average February (31 mm) and June (73 mm), respectively. The soil of the study site was formed on Pleistocene parent material and is classified as a Haplic Albeluvisol (Schindler et al. 2010). The soil type is a sandy loam with 5 % clay, 0.6 % organic C, a bulk density of 1.45 g cm^{-3} and a field capacity of 12.5 % (volumetric water content).

During the study period, the research field was cultivated with a mixture of lucerne (*Medicago sativa* L.), red clover (*Trifolium pratense* L.), and perennial ryegrass (*Lolium perenne* L.) as part of a crop rotation experiment (Sauer et al. 2015). The management practice included ploughing, and fertilization rates and harvest frequencies were adopted from the surrounding crop rotation experiment (Table 5.1).

Table 5.1: Timeline of management events at the lucerne/grass-clover field. Dates of establishment of the experimental plots as well as first and final dates of flux measurements are shown. The preceding crop, triticale, was harvested on 24th June 2013.

Date	Management	Rate of fertilizer and seed application (per ha)
2013-06-28	Ploughing	
2013-07-03	Basic fertilisation	48 kg P, 229 kg K, 13 kg Mg, 55 kg S
2013-07-08	Sowing	15 kg lucerne (cv. La Bella Campagnola), 3 kg red clover (cv. Milvus), 10 kg perennial ryegrass (cv. Arsenal)
2013-07-15	N-fertilisation	40 kg N, 4.4 kg S
2013-09-10	Pre-experimental cutting	
2013-10-13	Establishment of plots	
2013-11-26	First flux measurements	
2014-03-20	N-fertilisation	60 kg N, 6.6 kg S
2014-05-14	1st harvest	
2014-05-16	N-fertilisation	50 kg N, 5.5 kg S

2014-05-16	Basic fertilisation	48 kg P, 229 kg K, 13 kg Mg, 55 kg S
2014-07-02	2nd harvest	
2014-07-08	N-fertilisation	50 kg N, 11 kg S
2014-08-05	3rd harvest	
2014-08-11	N-fertilisation	60 kg N, 14 kg S
2014-10-09	4th harvest	
2014-11-04	Final flux measurements	

5.2.2 Study design and CO_2 flux measurements

Six weeks prior to the first flux measurements, six measurement plots were set up ~1 m apart by inserting PVC collars (0.75 m × 0.75 m × 0.15 m) permanently into the soil to a depth of ~5 cm. The CO_2 exchange was determined using both measurement approaches (sunrise and mid-day) on the same day at intervals of three to four weeks from 26th November 2013 to 4th November 2014. In total, 19 measurement campaigns were conducted during the measurement year.

Manual closed chambers (0.75 m × 0.75 m × 0.5 m) were used in non-steady-state flow-through mode (Livingston & Hutchinson 1995) to estimate R_{ECO} (opaque chambers) and NEE of CO_2 (transparent chambers). Chambers were constructed and instrumented as described by Drösler (2005). CO_2 concentrations in the chamber headspace were determined with an infrared gas analyzer at 3-s-intervals (LI-820, LI-COR, USA) during a measurement period of 180 s after placing the chamber on the collar. During measurements, the gas analyzer was connected to the chamber via two rubber tubes (inner diameter, 3 mm) in permanent flow-through mode. PAR was monitored with quantum sensors (Skye, UK) installed within the transparent chamber. Similarly, air temperatures were recorded inside and outside of the chamber and soil temperatures were measured manually in 2, 5, and 10 cm soil depth at a position between the third and fourth collar. During flux measurements, two fans (diameter, 12 cm) ensured thorough mixing of air in the chamber headspace (Drösler 2005).

Sunrise measurements started just before sunrise with opaque chamber measurements on all six collars, followed by alternating rounds of transparent and opaque chamber measurements. After each measurement, the chamber was vented for a short time to restore ambient CO_2 concentrations (~400 ppm). Data collection continued until 2 p.m., when the soil temperature at 2 cm depth reached its daily maximum. Following this procedure, the main drivers of CO_2 exchange, i.e., PAR and soil temperature, were captured at their maximal diurnal amplitude. With this approach two full rounds in winter and up to six full rounds in summer could be completed before 10 a.m. (where one full round represents a set of R_{ECO} and NEE measurements on all six plots).

Mid-day measurements were included from around 10 a.m. Following the regular NEE measurements with the transparent chamber (100 % PAR), the measurements were repeated after applying one (~50 % PAR), and two layers (~25 % PAR) of PAR-reducing netted coverage fitted to the size of a metal frame placed around the transparent chamber, and finally with opaque chambers (0 % PAR). From 10 a.m. until 2 p.m. it was possible to accomplish three full rounds of measurements at all four PAR levels. The fully transparent and opaque chamber measurements conducted from 10 a.m. to 2 p.m. (typically 18 + 18 flux measurements for each campaign) were included in both the sunrise and mid-day measurement approaches.

5.2.3 Environmental parameters

A climate station installed on the research field ~100 m apart from the measurement plots recorded half-hourly values of PAR (2 m height), air temperature (2 m height), relative humidity (0.2 m height), and precipitation (1 m height). A second climate station monitored half-hourly values of soil temperatures at 2, 5, and 10 cm depth at a distance of 5 m from the measurement plots. Plant height was measured at five locations within each collar during the CO_2 campaigns. Aboveground biomass was manually harvested from all collars at ~15 cm height four times during the study year in accordance with the harvest events of the surrounding field experiment (Table 5.1).

5.2.4 CO_2 flux calculation

CO_2 flux estimation was done in R 3.0.2 (R Core Team 2013) using the data processing script presented by Hoffmann et al. (2015). The *user defined* flux estimation settings used in this study were:

- 5 % death-band at the beginning and end of each measurement period
- Linear regression of gas concentrations over chamber deployment time
- Minimum moving window size of seven consecutive concentration measurements, i.e., 21-s-periods
- Significance of regression slope at $P \leq 0.1$
- Maximum temperature change of ± 0.75 K during the identified flux period
- Maximum PAR fluctuation of ± 10 % of the average during the identified flux period

Using these settings, CO_2 fluxes (F, µmol C m^{-2} s^{-1}) were calculated according to:

$$F = \frac{pV}{RTA} \cdot \frac{\mathrm{d}c}{\mathrm{d}t} \tag{5.1}$$

where p is the air pressure (101,300 Pa), V is the chamber volume, R is the gas constant (8.314 m^3 Pa K^{-1} mol^{-1}), T is the temperature inside the chamber (K), A is the inside collar surface area (0.5625 m^2) and dc/dt is the CO_2 concentration change over time (ppm C s^{-1}) derived from the linear regression. Only fluxes meeting all criteria of the above mentioned *user defined* settings were accepted for further analysis. Fluxes are reported using the atmospheric sign convention, where positive values represent gas fluxes from the soil to the atmosphere.

5.2.5 *Modelling CO_2 exchange*

We modelled CO_2 exchange of the study site according to Hoffmann et al. (2015) by combining NEE and R_{ECO} fluxes from all six plots. R_{ECO} was modelled by optimizing the parameters R_{10} and E_0 of an empirical Arrhenius type respiration model (Lloyd & Taylor, 1994):

$$R_{ECO} = R_{10} \cdot e^{E_0\left[\frac{1}{283.15-T_0} - \frac{1}{T-T_0}\right]} \tag{5.2}$$

where R_{10} is the respiration (µmol C m^{-2} s^{-1}) at a reference temperature of 10 °C (283.15 K), E_0 is a temperature sensitivity parameter (K), T (K) is the temperature during the measurements and T_0 is a notional zero-respiration temperature (here fixed to 227.13 K). T could either be the air temperature or one of the soil temperatures at 2, 5, or 10 cm depth, depending on the best fit (lowest Akaike Information Criterion). The level of significance during R_{ECO} model parameterisation was $P \leq 0.1$. If no significant model could be fitted, we used the respective mean R_{ECO} of the input data for the further calculations.

For the campaign-wise modelling approach, R_{ECO} was modelled over single measurement days (19 models). For the season-wise modelling approach only one R_{ECO} model was parameterised for each measurement approach (sunrise and mid-day). For the cluster-wise modelling approach, R_{ECO} fluxes from the study period were clustered according to green plant height and temperature using k-means clustering (Hartigan & Wong 1979). This resulted in a low-vegetation-stage cluster of R_{ECO} fluxes (non-growing season and post-harvest-measurements) and a high-vegetation-stage cluster of R_{ECO} fluxes (growing season and pre-harvest-measurements, Figure 5.A.1). For each of the two clusters, an individual R_{ECO} model was parameterised and applied at the dates corresponding to the respective vegetation stage (low-vegetation-stage vs. high-vegetation-stage). Thus, two alternating R_{ECO} models were used in the cluster-wise modelling approach.

GPP fluxes (µmol C m^{-2} s^{-1}) were either calculated from the measured NEE and R_{ECO} fluxes of the same round and plot (direct GPP) or estimated from the corresponding modelled R_{ECO} fluxes, based on the model parameters (R_{10} and E_0) and the temperatures during NEE measurements (indirect GPP). GPP fluxes were then fitted against PAR (µmol photons m^{-2} s^{-1}) for each campaign according to Hoffmann et al. (2015) using Eq. 5.3:

$$GPP = \frac{GP_{max} * \alpha * PAR}{\alpha * PAR + GP_{max}} \tag{5.3}$$

where α is the initial slope of the light response curve (µmol C $µmol^{-1}$ photons) and GP_{max} (µmol C m^{-2} s^{-1}) is the asymptotic value of primary CO_2 uptake at infinitely high PAR. The user defined level of significance of GPP model parameterisation was $P \leq 0.1$. For non-significant models, we used the respective mean GPP flux of the modelling input data for the further modelling procedure (see Hoffmann et al. 2015).

5.2.6 Gap filling and NEE balances

For gap filling, we used the parameters of the R_{ECO} models (campaign-wise, season-wise, cluster-wise) and the GPP models (direct vs. indirect) to predict half-hourly CO_2 fluxes using continous temperature and PAR data from the weather station as driving variables. To calculate annual CO_2 exchange we used the procedure of Günther et al. (2015). Thus, for the time steps between two measurement days, half-hourly GPP and R_{ECO} fluxes were first calculated according to the model parameters for each of the respective measurement days. This resulted in two parallel fluxes at each time step, and a weighted flux was then calculated according to temporal proximity of the time step to the parent measurement days. Finally, the resulting half-hourly GPP and R_{ECO} fluxes were used to calculate half-hourly NEE values (g C m^{-2}), which were summed up to obtain the net CO_2 exchange of the study period.

Estimating the uncertainty of modelled R_{ECO}, GPP and NEE fluxes is challenging (Beetz et al. 2013) and no consensus exists about propagating modelling errors for annual balances (Kandel et al. 2013). Here we used the error prediction algorithm suggested by Hoffmann et al. (2015) for reasons of consistency. This method uses a number of bootstrapping steps to estimate uncertainties deriving from flux measurements and modelling of R_{ECO} and GPP. The total error of the NEE modelling was then derived following the law of error propagation (see Hoffmann et al. 2015 for details).

5.2.7 Flux validation and role of harvest events

In order to evaluate the sensitivity of the 2 × 3 × 2 approaches towards seasonal flux dynamics and harvest events, the above described modelling and gap-filling procedure was repeated with (1) sequentially omitting one measurement campaign with only the first and the last campaign fixed, and (2) sequentially omitting one measurement campaign with the first and last campaign as well as the 8 pre- and post-harvest campaigns fixed (i.e., 4 + 4 campaigns). The derived modelled R_{ECO} and NEE fluxes were then validated against the omitted measured R_{ECO} and NEE fluxes.

5.3 RESULTS

5.3.1 Environmental conditions and biomass parameters

The measurement year 2013–2014 was warmer (10.4 °C vs. 9.0 °C) and slightly wetter (596 mm vs. 560 mm) than the long-term means. The lucerne/clover-grass mix was cut six weeks before this study started on 26th November 2013 (Table 5.1). During the winter 2013–2014, the grassland mixture remained at a low vegetation height of ≤20 cm, but then grew more than 20 cm in height during spring

2014. At harvest, the plants were cut at about 15 cm above the soil surface and then regrew to a height of 40–60 cm within three weeks. Harvested biomass was removed except for the fourth harvest where biomass was left on site as a mulch fertilisation treatment in accordance with the treatment of the surrounding research field. After the fourth harvest event, the vegetation remained at a height of ~20 cm.

5.3.2 *Measured CO_2 fluxes*

The sunrise measurements resulted in 734–740 valid NEE and 642–647 valid R_{ECO} fluxes (93–98 % of measurements). The exact number of fluxes depended on how the differently pooled R_{ECO} input data (campaign-wise, season-wise, and cluster-wise) met the flux quality criteria. Similarly, the mid-day measurements resulted in 1003–1016 valid NEE and 322–331 valid R_{ECO} fluxes (97–100 % of measurements). During the whole study period, the two measurement approaches shared approximately 322–335 NEE fluxes (100 % PAR) and all 322–331 R_{ECO} fluxes from 10 a.m. to 2 p.m. The highest net CO_2 uptake (-24 µmol C m^{-2} s^{-1}) occured on 6th June 2014 at a time of both high PAR and plant height (Figure 5.1). Highest CO_2 release (R_{ECO}) was generally found in August and September (~19 µmol C m^{-2} s^{-1}).

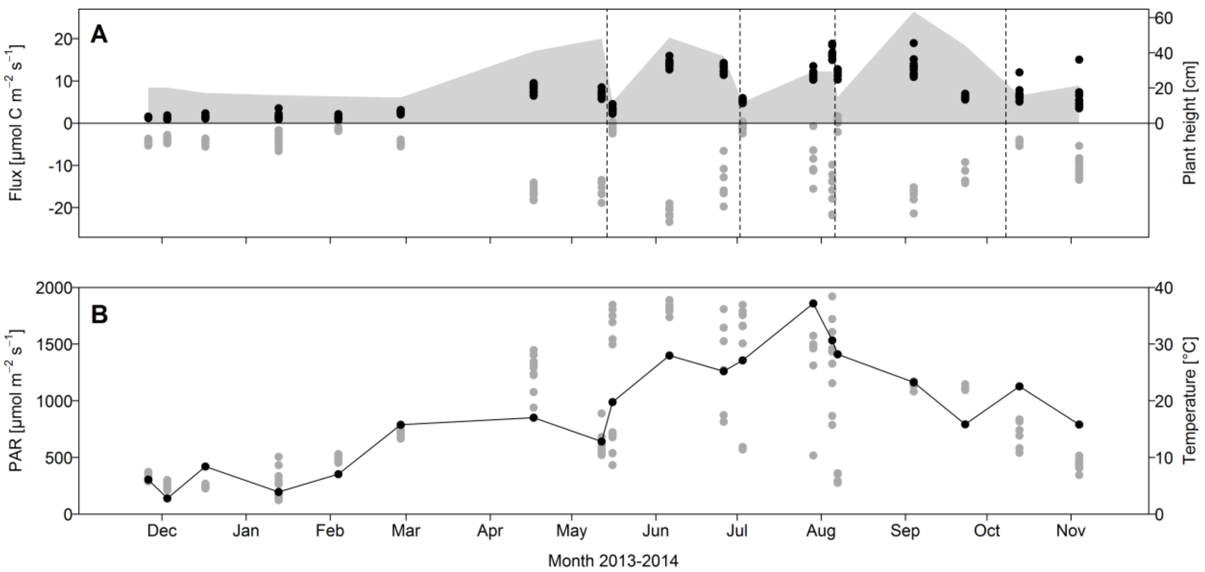

Figure 5.1: Overview of data collected at noon (11 a.m. to 1 p.m.) during the 19 field campaigns of the study period. Panel A: R_{ECO} fluxes (black dots, $n = 209$) and non-shaded NEE fluxes (grey dots, $n = 174$), together with mean plant height (filled grey area). Individual fluxes are shown for all six collars; plant height was linearly interpolated between the measurement campaigns. Dashed lines indicate harvest events. Panel B: Mean measured PAR at each of the six plots inside the transparent chambers (grey dots) and mean outside temperature (black dots and line).

5.3.3 *R_{ECO}, GPP and NEE modelling*

5.3.3.1 Mid-day vs. sunrise measurements

R_{ECO} models based on mid-day data generally had lower reference fluxes (R_{10}) and higher ecosystem temperature sensitivities (E_0) than the corresponding R_{ECO} models based on sunrise data (Table 5.2). In addition, temperature ranges of the mid-day R_{ECO} models covered on average 37 %, 91 %, and 76 % (campaign-, season-, and cluster-wise) of the respective sunrise approach R_{ECO} models. GP_{max} model parameters indicated generally higher CO_2 uptake for the direct GPP approach when derived from mid-day rather than from sunrise measurements (Table 5.3) and generally lower uptake for the indirect GPP approach (Table 5.4). For the initial light response parameter (α) the slopes tended to be steeper for mid-day than for sunrise measurements.

5.3.3.2 Campaign-wise vs. season-wise vs. cluster-wise R_{ECO} pooling

Significant R_{ECO} models could be fitted for all season- and cluster-wise modelling approaches irrespective of the underlying measurement approach. However, for some campaign-wise modelling approaches, model parameterisation failed and mean daily R_{ECO} fluxes were used, most often in combination with data collected by the mid-day approach (Table 5.2). Further, E_0 of the campaign-wise R_{ECO} models were lower than E_0 of the season- and cluster-wise models. GPP model parameterisation failed for some campaigns most often related to the campaign-wise R_{ECO} modelling approach with indirect GPP (Table 5.4). This was mainly caused by too narrow ranges of temperature or PAR obtained during individual measurement days (campaign-wise modelling).

The 1:1 agreement between modelled and measured R_{ECO} fluxes (calibration) decreased in the order of campaign-wise > cluster-wise > season-wise R_{ECO} modelling both for the sunrise and the mid-day measurement approach (Figure 5.A.2, Table 5.5). The 1:1 agreement with one omitted campaign (validation) was best for the cluster-wise R_{ECO} modelling (Table 5.5, Figure 5.A.5).

5.3.3.3 Direct vs. indirect GPP estimation

Irrespective of the underlying measurement or R_{ECO} pooling approach, GP_{max} model parameters generally indicated lower uptake for the direct GPP approach (Table 5.3, 5.4). Accordingly, the daily mean GPP fluxes of direct GPP were 1.4 ± 0.2 µmol C m^{-2} s^{-1} higher (lower uptake) than the daily mean GPP fluxes of indirect GPP (mean ± standard error, n = 114).

The agreement of calibration was always higher for direct than for indirect GPP (Figures 5.A.3, 5.A.4, Table 5.5), whereas validation with fixed pre- and post-harvest events was always higher for indirect GPP (Figures 5.A.6, 5.A.7, Table 5.5) with the exception of the mid-day/campaign/indirect approach showing the weakest overall performance. In addition, the modelled NEE values of the six direct GPP approaches were systematically lower than the modelled NEE of the six indirect GPP approaches (lower offsets to measured NEE), also signifying a lower net CO_2 release (Figure 5.A.6).

5.3.3.4 M/S/D vs. S/C/I

The combined M/S/D approach had a lower calibration agreement of R_{ECO} fluxes compared to the combined S/C/I approach (Figure 5.2). Similarly, NEE calibration (modelled vs. measured non-shaded NEE fluxes) was better for the S/C/I approach compared to the M/S/D approach (Figure 5.3).

In contrast, the leave-one-out validation of R_{ECO} fluxes was higher for the M/S/D approach compared to the S/C/I approach; when the pre- and post-harvest measurement campaigns were fixed both approaches were similar (Figure 5.2, Table 5.5). Leave-one-out NEE validation was of poor quality for all combined approaches. In contrast, NEE flux validation with fixed pre- and post-harvest measurements performed nearly as good as NEE flux calibration both for all approaches (Figure 5.3, Table 5.5, Figures 5.A.6, 5.A.7).

Table 5.2: Fitted R_{ECO} parameters R_{10} (µmol C m^{-2} s^{-1}) and E_0 (K) for the 2 × 3 approaches. Significant fits ($P < 0.1$) were used as obtained with either inside (T_{in}) or outside (T_{out}) chamber air temperature. If no significant model could be fitted to the data, mean R_{ECO} fluxes (mean, µmol C m^{-2} s^{-1}) were used. Note that parameters of the season-wise R_{ECO} models apply to the whole study period and that parameters of the two cluster-wise R_{ECO} models alternate over time.

<table>
<tr><th rowspan="3">Date</th><th colspan="12">Mid-day measurements with modelling:</th><th colspan="12">Sunrise measurements with modelling:</th></tr>
<tr><th colspan="4">campaign-wise</th><th colspan="4">season-wise</th><th colspan="4">cluster-wise</th><th colspan="4">campaign-wise</th><th colspan="4">season-wise</th><th colspan="4">cluster-wise</th></tr>
<tr><th>fit</th><th>R_{10}</th><th>E_0</th><th>mean R_{ECO}</th><th>fit</th><th>R_{10}</th><th>E_0</th><th>mean R_{ECO}</th><th>fit</th><th>R_{10}</th><th>E_0</th><th>mean R_{ECO}</th><th>fit</th><th>R_{10}</th><th>E_0</th><th>mean R_{ECO}</th><th>fit</th><th>R_{10}</th><th>E_0</th><th>mean R_{ECO}</th><th>fit</th><th>R_{10}</th><th>E_0</th><th>mean R_{ECO}</th></tr>
<tr><td>2013-11-26</td><td>T_{in}</td><td>2.7</td><td>353</td><td>1.4</td><td rowspan="19">T_{in}</td><td rowspan="19">4.3</td><td rowspan="19">237</td><td rowspan="19">6.6</td><td rowspan="6">T_{in}</td><td rowspan="6">2.4</td><td rowspan="6">286</td><td rowspan="6">3.3</td><td>T_{in}</td><td>1.4</td><td>58</td><td>1.2</td><td rowspan="19">T_{in}</td><td rowspan="19">5.2</td><td rowspan="19">202</td><td rowspan="19">6.3</td><td rowspan="6">T_{in}</td><td rowspan="6">3.0</td><td rowspan="6">243</td><td rowspan="6">3.2</td></tr>
<tr><td>2013-12-03</td><td>T_{in}</td><td>1.8</td><td>150</td><td>1.1</td><td>T_{in}</td><td>1.5</td><td>77</td><td>1.1</td></tr>
<tr><td>2013-12-17</td><td>T_{in}</td><td>1.7</td><td>186</td><td>1.4</td><td>T_{in}</td><td>1.6</td><td>147</td><td>1.2</td></tr>
<tr><td>2014-01-13</td><td>T_{out}</td><td>2.0</td><td>222</td><td>1.2</td><td>AVG</td><td>-</td><td>-</td><td>1.4</td></tr>
<tr><td>2014-02-04</td><td>T_{in}</td><td>1.8</td><td>322</td><td>1.1</td><td>T_{in}</td><td>1.1</td><td>67</td><td>0.9</td></tr>
<tr><td>2014-02-27</td><td>T_{out}</td><td>2.0</td><td>150</td><td>2.4</td><td>T_{in}</td><td>2.2</td><td>87</td><td>2.1</td></tr>
<tr><td>2014-04-17</td><td>T_{in}</td><td>6.2</td><td>174</td><td>8.1</td><td rowspan="2">T_{in}</td><td rowspan="2">7.8</td><td rowspan="2">130</td><td rowspan="2">11.3</td><td>T_{in}</td><td>6.5</td><td>145</td><td>5.7</td><td rowspan="2">T_{in}</td><td rowspan="2">7.7</td><td rowspan="2">141</td><td rowspan="2">9.6</td></tr>
<tr><td>2014-05-12</td><td>mean</td><td>-</td><td>-</td><td>7.6</td><td>AVG</td><td>-</td><td>-</td><td>7.9</td></tr>
<tr><td>2014-05-16</td><td>mean</td><td>-</td><td>-</td><td>2.9</td><td>T_{in}</td><td>2.4</td><td>286</td><td>3.3</td><td>T_{out}</td><td>2.7</td><td>58</td><td>2.9</td><td>T_{in}</td><td>3.0</td><td>243</td><td>3.2</td></tr>
<tr><td>2014-06-06</td><td>mean</td><td>-</td><td>-</td><td>13.7</td><td rowspan="2">T_{in}</td><td rowspan="2">7.8</td><td rowspan="2">130</td><td rowspan="2">11.3</td><td>T_{in}</td><td>7.5</td><td>184</td><td>9.8</td><td rowspan="2">T_{in}</td><td rowspan="2">7.7</td><td rowspan="2">141</td><td rowspan="2">9.6</td></tr>
<tr><td>2014-06-26</td><td>mean</td><td>-</td><td>-</td><td>13.0</td><td>T_{in}</td><td>8.7</td><td>145</td><td>11.0</td></tr>
<tr><td>2014-07-03</td><td>T_{in}</td><td>2.1</td><td>219</td><td>5.2</td><td>T_{in}</td><td>2.4</td><td>286</td><td>3.3</td><td>T_{in}</td><td>3.8</td><td>80</td><td>4.5</td><td>T_{in}</td><td>3.0</td><td>243</td><td>3.2</td></tr>
<tr><td>2014-07-24</td><td>T_{out}</td><td>8.5</td><td>50</td><td>11.1</td><td rowspan="2">T_{in}</td><td rowspan="2">7.8</td><td rowspan="2">130</td><td rowspan="2">11.3</td><td>T_{in}</td><td>9.0</td><td>44</td><td>10.5</td><td rowspan="2">T_{in}</td><td rowspan="2">7.7</td><td rowspan="2">141</td><td rowspan="2">9.6</td></tr>
<tr><td>2014-08-05</td><td>T_{in}</td><td>9.1</td><td>144</td><td>15.8</td><td>T_{in}</td><td>10.3</td><td>107</td><td>14.4</td></tr>
<tr><td>2014-08-07</td><td>mean</td><td>-</td><td>-</td><td>11.3</td><td>T_{in}</td><td>2.4</td><td>286</td><td>3.3</td><td>T_{in}</td><td>5.6</td><td>182</td><td>9.7</td><td>T_{in}</td><td>3.0</td><td>243</td><td>3.2</td></tr>
<tr><td>2014-09-04</td><td>T_{in}</td><td>7.8</td><td>172</td><td>12.8</td><td rowspan="2">T_{in}</td><td rowspan="2">7.8</td><td rowspan="2">130</td><td rowspan="2">11.3</td><td>T_{in}</td><td>8.4</td><td>149</td><td>11.1</td><td rowspan="2">T_{in}</td><td rowspan="2">7.7</td><td rowspan="2">141</td><td rowspan="2">9.6</td></tr>
<tr><td>2014-09-23</td><td>mean</td><td>-</td><td>-</td><td>5.8</td><td>T_{in}</td><td>5.2</td><td>83</td><td>5.4</td></tr>
<tr><td>2014-10-13</td><td>T_{in}</td><td>4.9</td><td>91</td><td>6.2</td><td rowspan="2">T_{in}</td><td rowspan="2">2.4</td><td rowspan="2">286</td><td rowspan="2">3.3</td><td>T_{in}</td><td>5.4</td><td>59</td><td>6.0</td><td rowspan="2">T_{in}</td><td rowspan="2">3.0</td><td rowspan="2">243</td><td rowspan="2">3.2</td></tr>
<tr><td>2014-11-04</td><td>mean</td><td>-</td><td>-</td><td>5.0</td><td>T_{in}</td><td>4.5</td><td>118</td><td>4.8</td></tr>
</table>

Table 5.3: Direct modeling of GPP: fitted GPP parameters α (µmol C µmol^{-1} photons) and GP_{max} (µmol C m^{-2} s^{-1}) together with mean direct (NEE – measured R_{ECO}) GPP fluxes (µmol C m^{-2} s^{-1}) for the two measurement approaches. Significant fits were used at $P \leq 0.1$, else mean GPP fluxes were used. Note that GPP modelling with direct GPP was independent of R_{ECO} pooling and modelling.

Date	Mid-day measurements			Sunrise measurements		
	α	GP_{max}	mean GPP	α	GP_{max}	mean GPP
2013-11-26	-	-	-4.6	-	-	-3.7
2013-12-03	-0.04	-9.7	-3.0	-0.01	-2.9	-2.8
2013-12-17	-	-	-3.4	-0.03	-25.2	-3.6
2014-01-13	-0.05	-8.4	-2.5	-0.05	-10.9	-3.1
2014-02-04	-	-	-1.9	-	-	-1.5
2014-02-27	-0.02	-13.1	-4.5	-0.02	-13.7	-4.9
2014-04-17	-0.07	-33.5	-15.2	-0.06	-32.6	-14.6
2014-05-12	-0.08	-50.4	-10.3	-0.21	-23.1	-14.3
2014-05-16	-0.02	-4.3	-2.5	-0.01	-5.8	-3.3
2014-06-06	-0.08	-46.6	-22.3	-0.09	-44.8	-20.2
2014-06-26	-0.07	-38.1	-15.3	-0.09	-33.8	-18.5
2014-07-03	-0.02	-8.2	-4.0	-0.04	-6.6	-4.2
2014-07-24	-0.04	-33.4	-13.5	-0.03	-45.0	-13.9
2014-08-05	-0.06	-48.8	-17.2	-0.08	-44.2	-19.1
2014-08-07	-0.04	-22.7	-8.5	-0.07	-20.3	-10.5
2014-09-04	-0.10	-42.9	-18.1	-0.26	-30.5	-19.1
2014-09-23	-0.09	-25.0	-12.4	-0.11	-22.9	-12.1
2014-10-13	-0.05	-17.3	-6.4	-0.08	-14.3	-8.9
2014-11-04	-0.07	-29.8	-8.1	-0.10	-23.8	-12.2

Table 5.4: Indirect modelling of GPP fluxes: fitted GPP parameters α (µmol C µmol^{-1} photons) and GP_{max} (µmol C m^{-2} s^{-1}) together with mean indirect (NEE – modelled R_{ECO}) GPP fluxes (µmol C m^{-2} s^{-1}) for the 2 × 3 approaches (measurements × modelling). Significant fits were used at $P \leq 0.1$, else mean GPP fluxes were used. Note that GPP modelling with indirect GPP was dependent of R_{ECO} pooling.

Date	Mid-day measurements with modelling:									Sunrise measurements with modelling								
	campaign-wise			season-wise			cluster-wise			campaign-wise			season-wise			cluster-wise		
	α	GP_{max}	mean GPP	α	GP_{max}	mean GPP	α	GP_{max}	mean GPP	α	GP_{max}	mean GPP	α	GP_{max}	mean GPP	α	GP_{max}	mean GPP
2013-11-26	-0.08	-9.1	-5.2	-0.13	-10.2	-6.7	-	-	-6.5	-0.02	-68.3	-3.7	-0.03	-73.8	-6.1	-	-	-5.9
2013-12-03	-0.04	-10.0	-3.1	-	-	-8.5	-0.04	-10.3	-3.1	-	-	-2.7	-	-	-7.9	-	-	-4.9
2013-12-17	-0.05	-12.5	-3.4	-0.11	-11.6	-5.5	-	-	-5.3	-0.03	-23.8	-3.6	-0.07	-20.3	-6.1	-0.04	-31.6	-4.4
2014-01-13	-0.04	-13.0	-2.4	-0.08	-12.1	-3.8	-0.04	-14.6	-2.5	-0.04	-11.8	-3.0	-0.07	-15.0	-4.7	-0.04	-16.8	-3.2
2014-02-04	-0.03	-4.4	-2.4	-	-	-7.4	-0.04	-4.7	-2.9	-0.01	-16.1	-1.6	-0.02	-20.6	-4.4	-	-	-3.7
2014-02-27	-0.02	-13.1	-4.5	-0.07	-15.7	-8.7	-0.04	-14.9	-6.1	-0.02	-14.3	-5.1	-0.03	-38.3	-8.5	-0.02	-61.6	-6.2
2014-04-17	-0.08	-36.9	-17.1	-0.06	-37.4	-15.2	-0.09	-36.7	-18.2	-0.06	-39.6	-16.6	-0.05	-44.8	-15.8	-0.07	-41.1	-18.0
2014-05-12	-	-	-10.4	-0.06	-78.9	-9.1	-0.10	-48.3	-12.1	-0.10	-38.4	-15.5	-0.08	-47.4	-14.2	-0.11	-43.7	-16.6
2014-05-16	-0.01	-5.9	-2.6	-0.09	-14.2	-9.9	-0.04	-11.4	-6.5	-0.02	-5.7	-3.4	-0.04	-18.4	-9.8	-0.02	-15.2	-6.5
2014-06-06	-0.08	-46.5	-22.1	-0.07	-51.2	-22.0	-0.08	-49.3	-23.1	-0.10	-53.5	-23.2	-0.08	-50.0	-20.0	-0.10	-46.7	-21.9
2014-06-26	-	-	-15.3	-0.05	-47.5	-13.4	-0.06	-42.9	-15.4	-0.09	-41.2	-20.6	-0.06	-42.3	-17.7	-0.08	-38.9	-19.0
2014-07-03	-0.02	-9.9	-4.9	-0.12	-16.4	-12.0	-0.05	-13.1	-8.1	-0.02	-8.6	-4.1	-0.05	-20.8	-9.1	-0.02	-17.8	-6.0
2014-07-24	-0.04	-33.4	-13.4	-0.09	-37.5	-19.6	-0.09	-35.3	-19.0	-0.03	-44.5	-14.3	-0.03	-92.5	-17.6	-0.04	-64.1	-18.2
2014-08-05	-0.08	-51.7	-20.0	-0.04	-60.7	-15.2	-0.05	-54.5	-16.3	-0.08	-50.4	-20.1	-0.05	-64.4	-19.8	-0.06	-54.6	-18.9
2014-08-07	-	-	-8.7	-0.05	-27.8	-10.6	-0.02	-35.4	-6.7	-0.06	-26.0	-11.9	-0.06	-27.6	-12.0	-0.03	-28.2	-8.9
2014-09-04	-0.10	-47.2	-19.2	-0.06	-49.5	-14.7	-0.09	-45.4	-17.2	-0.13	-43.9	-21.3	-0.09	-43.0	-19.1	-0.12	-41.8	-20.0
2014-09-23	-0.09	-24.8	-12.4	-0.10	-27.9	-13.9	-0.15	-29.2	-17.1	-0.11	-23.6	-12.4	-0.10	-29.3	-13.9	-0.14	-29.9	-16.2
2014-10-13	-0.04	-19.6	-6.4	-0.08	-20.7	-9.1	-0.04	-23.7	-6.0	-0.06	-15.3	-8.1	-0.06	-27.1	-10.7	-0.03	-27.9	-7.6
2014-11-04	-	-	-8.2	-0.09	-36.1	-9.8	-0.05	-53.0	-7.4	-0.08	-32.6	-12.2	-0.08	-41.0	-13.9	-0.06	-43.9	-11.1

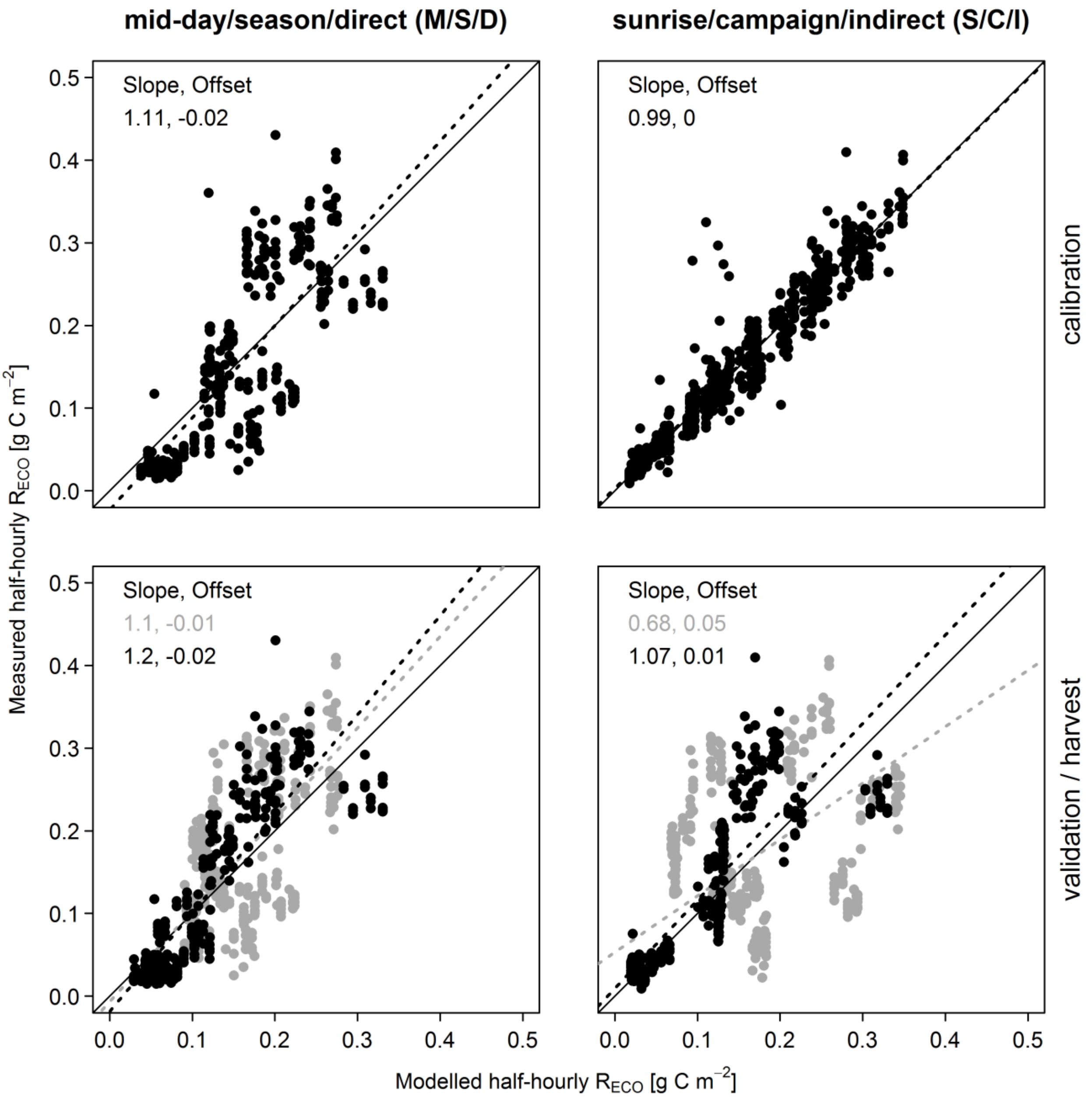

Figure 5.2: Calibration (upper panels) and leave-one-campaign-out relationships (lower panels) between modelled and measured half-hourly R_{ECO} fluxes (g C m^{-2}) for the two most common approaches (M/S/D and S/C/I) for the period from November 2013 to November 2014. The solid black line shows the 1:1 agreement between modelled and measured fluxes; dashed lines show the linear fit of the actual relationship. In the lower panels grey colours represent analyses with fixed start and final campaigns and black colours represent analyses with additionally fixed pre- and post-harvest campaigns.

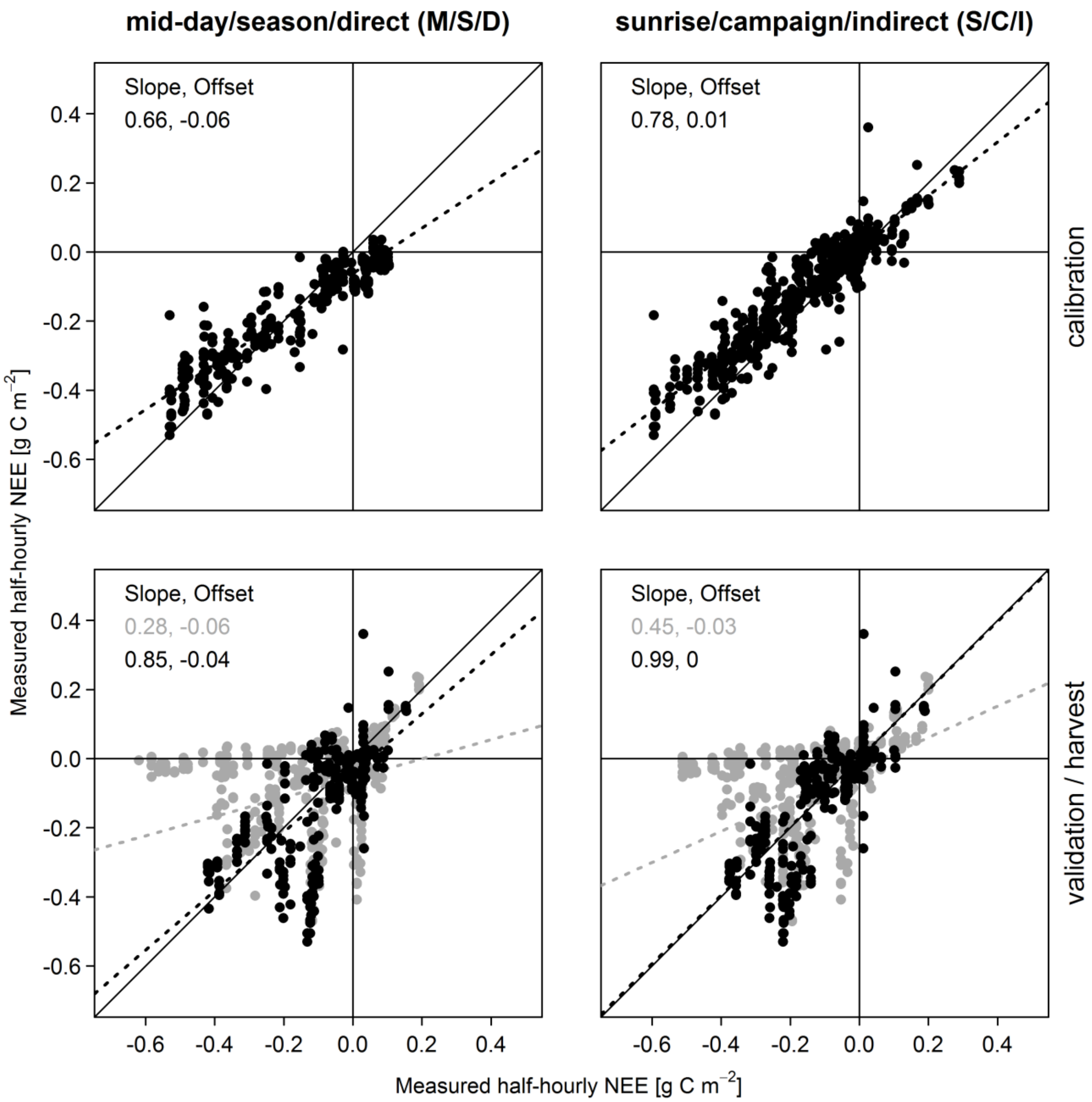

Figure 5.3: Calibration (upper panels) and leave-one-campaign-out relationships (lower panels) between modelled and measured half-hourly NEE fluxes (g C m^{-2}) for the two most common approaches (M/S/D and S/C/I) for the period from November 2013 to November 2014. The solid black line shows the 1:1 agreement between modelled and measured fluxes; dashed lines show the linear fit of the actual relationship. In the lower panels grey colours represents analyses with fixed start and final campaigns and black colours represent analyses with additionally fixed pre- and post-harvest campaigns.

5.3.4 *Cumulative R_{ECO} and GPP fluxes and net CO_2 balances*

5.3.4.1 Mid-day vs. sunrise measurements

In general, cumulative R_{ECO} fluxes varied by less than 10 % and were not affected by either measurement approach (Table 5.6). The ranges of cumulative R_{ECO} fluxes were higher for the sunrise than for the mid-day measurement approaches (149 g C m^{-2} vs. 86 g C m^{-2}). Cumulative GPP fluxes tended to indicate lower uptake for mid-day measurements and were most sensitive with the mid-day measurements and indirect GPP modelling with a range of 539 g C m^{-2}. NEE balances and seasonal flux dynamics (Figures 5.4, 5.5) were not systematically affected by the measurement approaches.

5.3.4.2 Campaign- vs. season- vs. cluster-wise R_{ECO} pooling

The variation of cumulative R_{ECO}, GPP and NEE fluxes generally decreased in the order of campaign-wise > season-wise > cluster-wise R_{ECO} modelling approaches (Table 5.6). While cluster-wise R_{ECO} modelling reduced the variation of cumulative R_{ECO} fluxes to less than 3 % (51 g C m^{-2}) the variation of the respective cumulative GPP and NEE fluxes remained relatively high (317 g C m^{-2} and 362 g C m^{-2}, respectively). In general, the combination of sunrise/indirect and mid-day/direct approaches each yielded relatively robust NEE balances regardless of the R_{ECO} pooling. Yet, the three R_{ECO} modelling approaches showed distinct differences in reflecting the temporal dynamics during the study period (Figures 5.4, 5.5): Campaign-wise R_{ECO} modelling approaches resulted in the highest seasonal amplitude of daily fluxes, followed by the cluster-wise approaches, and with somewhat dampened CO_2 dynamics for the season-wise approaches (Figures 5.4, 5.5).

5.3.4.3 Direct vs. indirect GPP estimation

Cumulative R_{ECO} fluxes were independent of the GPP modelling approach. Therefore, NEE balances were only dependent on differences in cumulative GPP fluxes, which were higher (= lower uptake) by approx. 150–300 g C m^{-2} when modelled directly compared to indirect modelling except for the mid-day/campaign/indirect approach (Table 5.6). In addition, the temporal GPP curves of the direct GPP approaches showed similar dynamics as the sunrise/campaign-wise and sunrise/cluster-wise approaches with indirect GPP. However, the maximum uptake of the direct GPP approaches was substantially smaller especially during times of high productivity in the summer months (Tables 5.3, 5.4, Figures 5.4, 5.5).

5.3.4.4 M/S/D vs. S/C/I

Over the study period, cumulative R_{ECO} and GPP fluxes of the M/S/D and the S/C/I approach differed substantially, leading to NEE balances that diverged in the net direction of CO_2 exchange (171 ± 14 g C m^{-2} and -101 ± 17 g C m^{-2}, respectively). The smallest range of NEE balances with varying R_{ECO} modelling was found for the sunrise/indirect (-101 to -131 g C m^{-2}) and mid-day/direct approaches (171 to 258 g C m^{-2}, Table 5), whereas the largest range was found for the mid-day/indirect approaches (-200 to 425 g C m^{-2}). The latter resulted mainly from the large deviation of the mid-day/campaign/indirect approach (425 ± 55 g C m^{-2}).

The cumulative flux curves of the mid-day/cluster, sunrise/campaign and sunrise/cluster approaches with indirect GPP evolved similarly with a small net CO_2 release during the non-growing season that was overcompensated by a high uptake before the first and second harvest events (Figure 5.5). Thereafter, the system remained relatively CO_2 neutral. In contrast, seasonal modelling approaches showed a higher net CO_2 release during the non-growing season and a higher CO_2 uptake before the second harvest event (Figures 5.4, 5.5). Thereafter, the development was somewhat erratic leading to the larger variation in net balances. The mid-day/campaign/indirect approach differed both during the non-growing season and during the growing season from all the other approaches leading to the exceptionally high overall net CO_2 release of 425 ± 55 g C m^{-2} (Figure 5.5A).

Table 5.5: Nash-Sutcliffe model efficiency (NSE, %) of model calibration (all campaigns, 'Cal'), leave-one-campaign-out cross-validation (*n* = 17, 'Val') and leave-one-campaign-out cross-validation with fixed pre- and post-harvest measurements (*n* = 9, 'Val Fix') for the 2 × 3 × 2 approaches of data acquisition, R_{ECO} modelling and GPP estimation.

Data acquisition	R_{ECO} modelling	GPP estimation	NSE (%) for R_{ECO}			NSE (%) for NEE		
			Cal	Val	Val Fix	Cal	Val	Val Fix
Mid-day	Campaign	Direct	95	31	67	83	-24	50
Mid-day	Campaign	Indirect				26	-27	-10
Mid-day	Season	Direct	62	61	76	62	-59	43
Mid-day	Season	Indirect				59	-52	53
Mid-day	Cluster	Direct	81	31	86	77	-24	-23
Mid-day	Cluster	Indirect				64	-19	60
Sunrise	Campaign	Direct	94	30	71	81	8	54
Sunrise	Campaign	Indirect				73	-20	62
Sunrise	Season	Direct	58	57	69	61	-29	42
Sunrise	Season	Indirect				58	-38	56
Sunrise	Cluster	Direct	76	30	81	72	-5	43
Sunrise	Cluster	Indirect				62	-38	59

Table 5.6: Cumulative R_{ECO}, GPP and NEE fluxes (g C m^{-2}) of the 2 × 3 × 2 approaches over the study period. Balance ± error estimation follow Hoffmann et al. (2015).

Data acquisition	R_{ECO} modelling	GPP estimation	Cumulative CO_2 fluxes (g C m^{-2})		
			R_{ECO}	GPP	NEE
Mid-day	Campaign	Direct	2204 ± 54	-1947 ± 7	258 ± 55
Mid-day	Campaign	Indirect	2204 ± 54	-1779 ± 6	425 ± 55
Mid-day	Season	Direct	2118 ± 11	-1947 ± 7	171 ± 14
Mid-day	Season	Indirect	2118 ± 11	-2318 ± 8	-200 ± 14
Mid-day	Cluster	Direct	2179 ± 13	-1947 ± 7	232 ± 15
Mid-day	Cluster	Indirect	2179 ± 13	-2264 ± 7	-86 ± 14
Sunrise	Campaign	Direct	2052 ± 15	-1990 ± 10	62 ± 18
Sunrise	Campaign	Indirect	2052 ± 15	-2153 ± 9	-101 ± 17
Sunrise	Season	Direct	2201 ± 5	-1990 ± 10	211 ± 11
Sunrise	Season	Indirect	2201 ± 5	-2323 ± 12	-122 ± 13
Sunrise	Cluster	Direct	2128 ± 5	-1990 ± 10	138 ± 11
Sunrise	Cluster	Indirect	2128 ± 5	-2259 ± 10	-131 ± 11

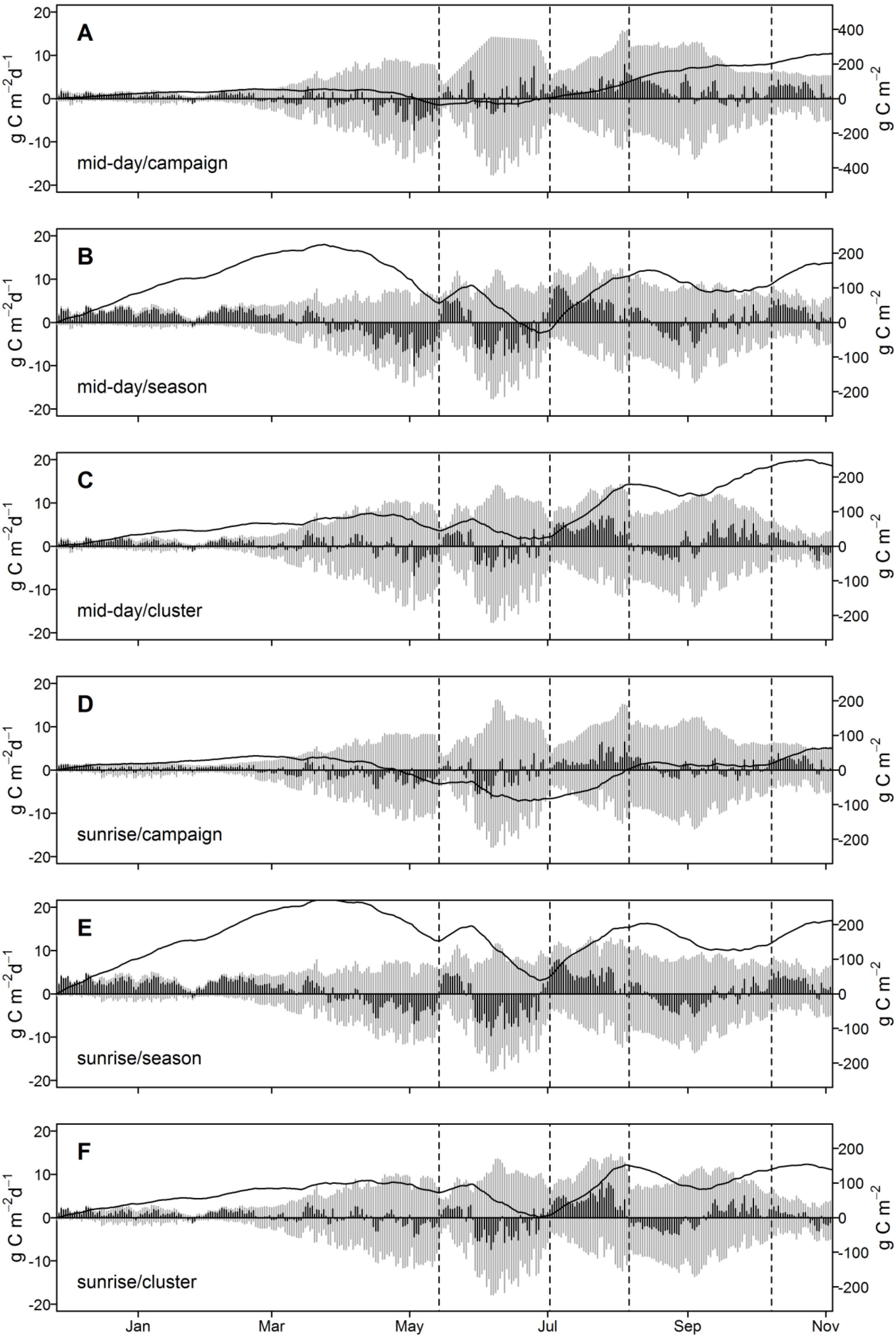
A
mid-day/campaign
B
mid-day/season
C
mid-day/cluster
D
sunrise/campaign
E
sunrise/season
F
sunrise/cluster
g C m⁻²d⁻¹
g C m⁻²
Jan
Mar
May
Jul
Sep
Nov

Figure 5.4: CO_2 exchange of the 2 × 3 approaches (measurement × modelling) with direct GPP during the study period. Positive grey bars show modelled daily R_{ECO} fluxes, negative grey bars show modelled daily GPP fluxes, black bars show the resulting modelled daily NEE fluxes (all in g C m^{-2} d^{-1}). The cumulative NEE (g m^{-2}) is shown by the solid black line (right y-axes). Vertical dashed lines mark days of harvest. Note that the scale of the y-axis for the cumulative NEE curve of panel A is two times larger than those of the other panels.

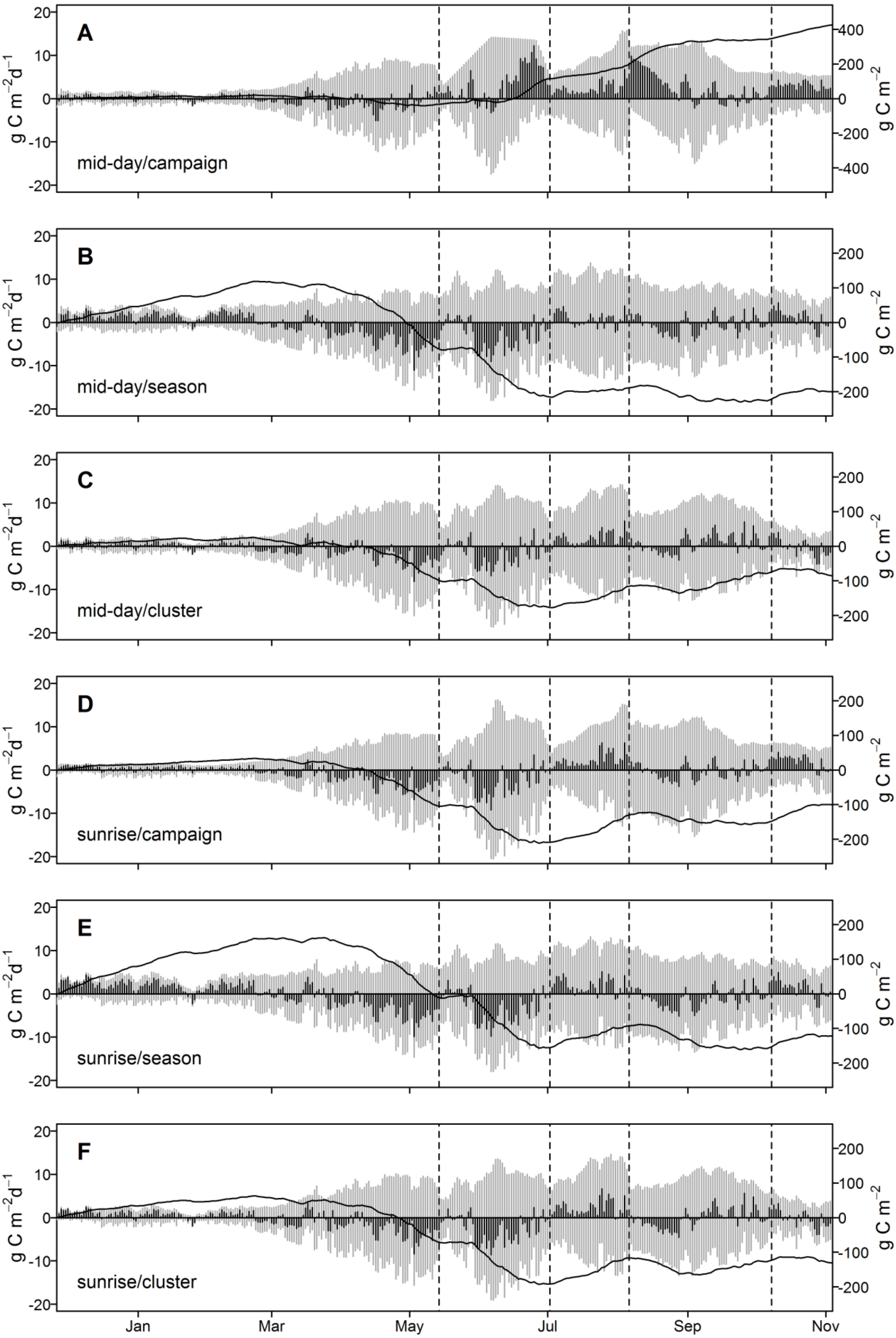
A
mid-day/campaign
B
mid-day/season
C
mid-day/cluster
D
sunrise/campaign
E
sunrise/season
F
sunrise/cluster
g C m⁻²d⁻¹
g C m⁻²
Jan
Mar
May
Jul
Sep
Nov

Figure 5.5: CO_2 exchange of the 2 × 3 approaches (measurement × modelling) with indirect GPP during the study period. Positive grey bars show modelled daily R_{ECO} fluxes, negative grey bars show modelled daily GPP fluxes, black bars show the resulting modelled daily NEE fluxes (all in g C m^{-2} d^{-1}). The cumulative NEE (g m^{-2}) is shown by the solid black line (right y-axes). Vertical dashed lines mark days of harvest. Note that the scale of the y-axis for the cumulative NEE curve of panel A is two times larger than those of the other panels.

5.4 DISCUSSION & CONCLUSIONS

5.4.1 Mid-day vs. sunrise measurements

Both the data sets themselves and the derived R_{ECO} and GPP model parameters substantially differ between the mid-day and sunrise measurements. In addition, the temperature ranges of the mid-day R_{ECO} models are smaller than those from the sunrise data. This supports the conceptual idea behind the sunrise measurements, i.e. to reduce the possible bias of mid-day data collection when CO_2 fluxes at low PAR are measured under atypical air/soil temperatures. Both the respiration and photosynthesis of higher plants are temperature dependent (Berry & Björkmann 1980, Marcolla et al. 2011), and uncoupling diurnal temperature variability from diurnal PAR dynamics may bias the different data sets towards over- or underestimation of R_{ECO} and GPP through temperature differences. However, we did not find a systematic shift in the overall cumulative R_{ECO}, GPP and NEE between mid-day and sunrise data, indicating that the variability induced by decisions on the subsequent gap-filling procedures generally exceeds the possible differences induced by the data sets.

5.4.2 Campaign-wise vs. season-wise vs. cluster-wise R_{ECO} pooling

Apart from a small number of fluxes (<10), the data sets for the differently pooled R_{ECO} models were the same. The campaign-wise approaches resulted in the most flexible representation of R_{ECO} flux dynamics (Figure 5.A.2). This was partly true also for the NEE modelling, where only the mid-day/campaign/indirect approach failed due to extensive interpolation of mean daily R_{ECO} and GPP fluxes in the summer months (Tables 5.2, 5.4) resulting in the overall highest net release of CO_2.

In contrast, the season-wise approaches were robust when omitting entire campaigns (validation), suggesting that the net CO_2 exchange modelled over a longer period of time is less biased towards single measurements. However, all of the 2 × 3 × 2 approaches perform poorly during leave-one-campaign-out-validation due to the four harvest events. Accordingly, the modelling was strongly improved by keeping the pre- and post-harvest measurements fixed in the validation (sensitivity) analyses (Figure 5.A.6 and 5.A.7, Table 5.5). Since in this case validation performs nearly as well as model calibration, we suggest that manual chamber data should only be pooled within periods between cutting or other major plant physiological changes (Beetz et al. 2013, Günther et al. 2015, Hoffmann et al. 2015).

In the studied summer-active agro-ecosystem, the ecosystem sensitivity (E_0) of the R_{ECO} models was higher when derived from long-term data than from short-term data (season-/cluster-wise vs. campaign-wise modelling), substantiating the results of Reichstein et al. (2005). This may imply that the application of long-term sensitivity parameters derived from seasonally pooled R_{ECO} data for the extrapolation of day- to night-time (i.e. to lower temperatures) may lead to an underestimation of night-time R_{ECO} fluxes and thereby also to an underestimation of cumulative R_{ECO} fluxes. However, the variation in cumulative R_{ECO} of this study cannot be explained by the R_{ECO} pooling approach. Therefore, a higher E_0 from seasonal R_{ECO} modelling does not necessarily lead to an underestimation of night-time R_{ECO} resulting in a lower CO_2 release.

The variation of cumulative R_{ECO}, GPP and NEE is highest for the campaign-wise R_{ECO} modelling approaches, mainly because empirical R_{ECO} and GPP modelling based on weather station data fails more often during single campaigns (due to the comparatively low number of flux measurements). This indicates that the campaign-wise modelling approach is very sensitive to diurnal data collection, and is probably only applicable if diurnal NEE and R_{ECO} flux variation is sufficiently covered (sunrise approaches). This problem can be overcome by adequately pooling flux data to the respective plant phenological stages (cluster-wise approaches). Although the improvement is small compared to the overall variability and therefore may not be significant this result fits well to the recommendations of other studies suggesting the use of vegetation proxies to account for plant phenological stages for improved R_{ECO} and NEE modelling (Burrows et al. 2005, Kandel et al. 2013, Renou-Wilson et al. 2014).

5.4.3 Direct vs. indirect GPP estimation

GP_{max} model parameters, mean daily GPP fluxes, NEE flux validation and cumulative GPP fluxes all signify a lower photosynthetic uptake for the direct GPP than for the indirect GPP approach in our study. The mode of obtaining GPP fluxes from NEE measurements (either directly by using measured R_{ECO} or indirectly by using modelled R_{ECO}) is part of a basic dilemma: using proximate measured R_{ECO} fluxes avoids propagating R_{ECO} modelling errors into GPP models, but on the other hand, potential temperature deviations between transparent and opaque chamber measurements (especially in the summer months) may lead to an underestimation of GPP when R_{ECO} fluxes of the somewhat cooler opaque chamber measurements are assumed for the warmer transparent chamber. Thus, an effective temperature control is needed to ensure near constant temperature conditions both within flux measurements (Elsgaard et al. 2012) and between transparent and opaque chamber measurements of the same measurement round. In cases with inadequate chamber temperature control, indirect GPP is assumed to overcome this possible source of error through modelled R_{ECO} fluxes using the actual temperatures during the transparent chamber measurements (Leiber-Sauheitl et al. 2014, Hoffmann et al. 2015). In addition, we assume that indirect GPP also overcomes a possible bias of the R_{ECO} flux data set (due to favoured times of the day) since the bias is negatively propagated into the GPP flux data set of the NEE modelling (due to GPP = NEE – R_{ECO}). Therefore, the strong variation between GPP direct and GPP indirect of our study likely shows how strongly temperature deviations can affect NEE balances of manual chamber measurements collected at unevenly distributed times of the day. In this respect hysteresis in CO_2 flux responses to temperature could present an additional source of error in the modelling of manual closed chamber data (Phillips et al. 2011).

5.4.4 M/S/D vs. S/C/I and factor interactions

Although the cumulative R_{ECO} and GPP fluxes vary by about 160 g C m^{-2} for the sunrise/indirect approaches, the respective NEE balances vary by only 30 g C m^{-2}. This suggests that also the variation of the R_{ECO} pooling is propagated, yet 'mirrored' to the GPP side of the net CO_2 balance. Since the variation of the NEE balances is very high for the mid-day/indirect approaches, we assume that using indirect GPP may only be applied with sunrise measurements which more adequately cover diurnal R_{ECO} flux and temperature ranges.

The two most common approaches in the literature to derive NEE balances with manual closed chambers (M/S/D and S/C/I) differ from each other both in the reflection of R_{ECO} and NEE flux dynamics as well as in resulting NEE balances. This produces a serious problem for meta-analyses and inter-study comparisons. Here, we suppose that the S/C/I approach is likely closer to the true NEE of our system, since studies from agro-ecosystems with forage crops (e.g., ryegrass and lucerne) generally estimate an NEE in the range of -100 to -400 g C m^{-2} (Byrne et al. 2005, Bolinder et al. 2012, Gilmanov et al. 2014).

5.5 RECOMMENDATIONS FOR A CONSISTENT MANUAL CHAMBER APPROACH

Our results suggest that it is not necessarily the divergent data sets (mid-day vs. sunrise measurements) but rather the following decisions made for gap filling that may produce divergent NEE balances derived from manual closed chambers. Therefore, the following recommendations are given:

1. When campaign-wise modelling fails, the use of average fluxes for interpolation should be avoided. Instead, it is more advisable to pool data from adjacent campaigns (Beetz et al. 2013), to use a moving-window approach of neighbouring campaigns to obtain significant R_{ECO} models (Hoffmann et al., submitted), or to use seasonal R_{ECO} data corrected for plant phenological stages (e.g., Burrows et al. 2005, Kandel et al. 2013). Nonetheless, the pooling of data across harvest events must be avoided.
2. In case of a non-effective temperature control also between transparent and opaque chamber measurements (like in this study), we recommend using the sunrise measurements with indirect GPP (using modelled R_{ECO}) in combination with R_{ECO} models pooled according to plant phenological stage (cluster-wise). Alternatively, other vegetation indices can be included in future studies aiming to obtain annual net CO_2 balances with manual closed chambers.
3. If the mid-day approach is used (e.g., for logistical reasons), cluster-wise R_{ECO} modelling may reduce R_{ECO} uncertainty arising with vegetation development and harvest events in addition to reducing R_{ECO} model sensitivity towards individual measurement campaigns. However, we

assume that it may be even more important to propagate potentially biased R_{ECO} data sets (e.g., from high temperature mid-day measurements or divergences between transparent and opaque chambers) to the GPP side of the gap filling to avoid this bias in the gap-filled net CO_2 balances. The efficiency of an effective temperature control to avoid this bias has yet to be tested in future studies.

Finally, we emphasize the importance of further developing consistent standards of data acquisition and gap-filling procedures to facilitate meta analysis of manual chamber data.

ACKNOWLEDGEMENTS

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APPENDIX

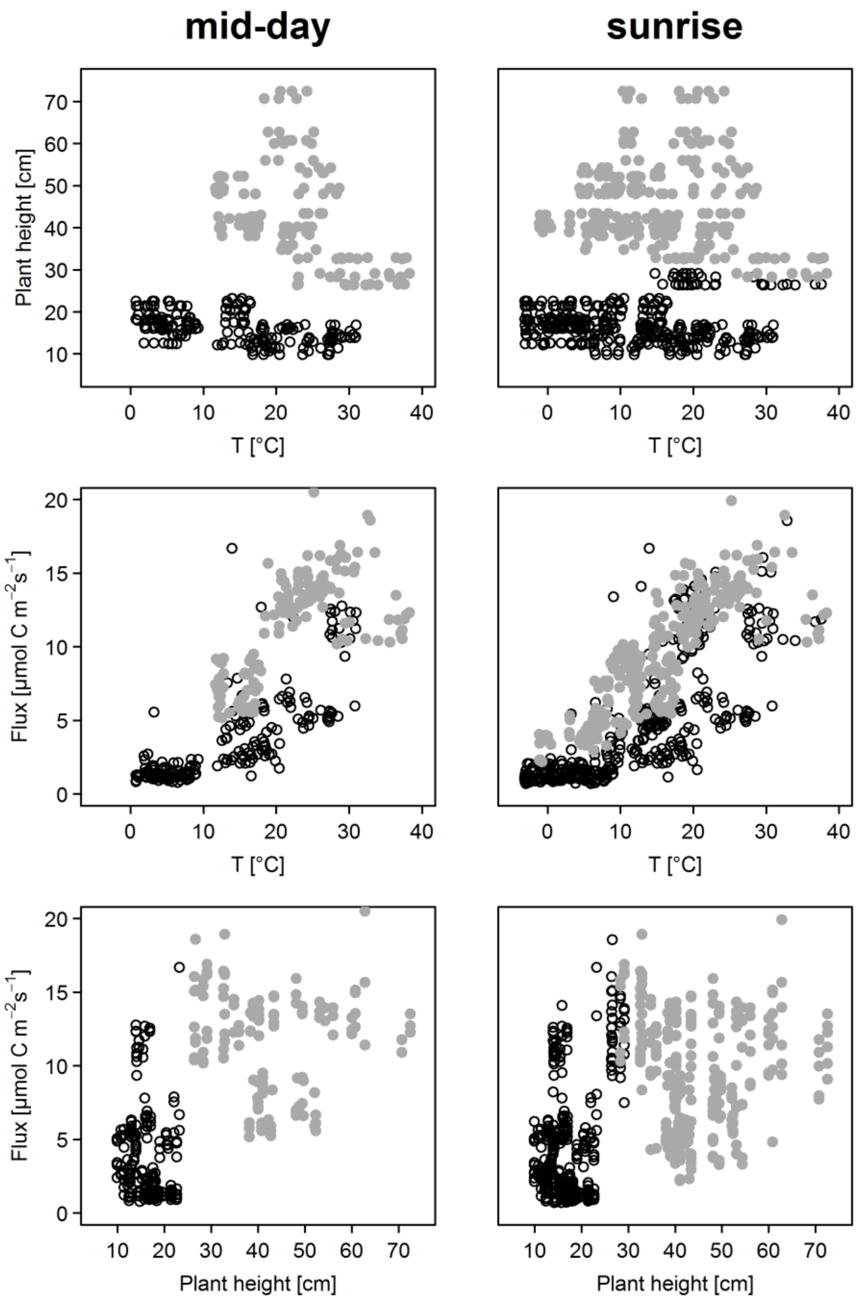

Figure 5.A.1: Cluster analyses of the mid-day and sunrise R_{ECO} fluxes data set. Black circles depict the cluster of R_{ECO} fluxes of the non-growing season and post-harvest measurements (= low vegetation stage), grey dots depict the cluster of R_{ECO} fluxes of the growing season before harvest events (= high vegetation stage).

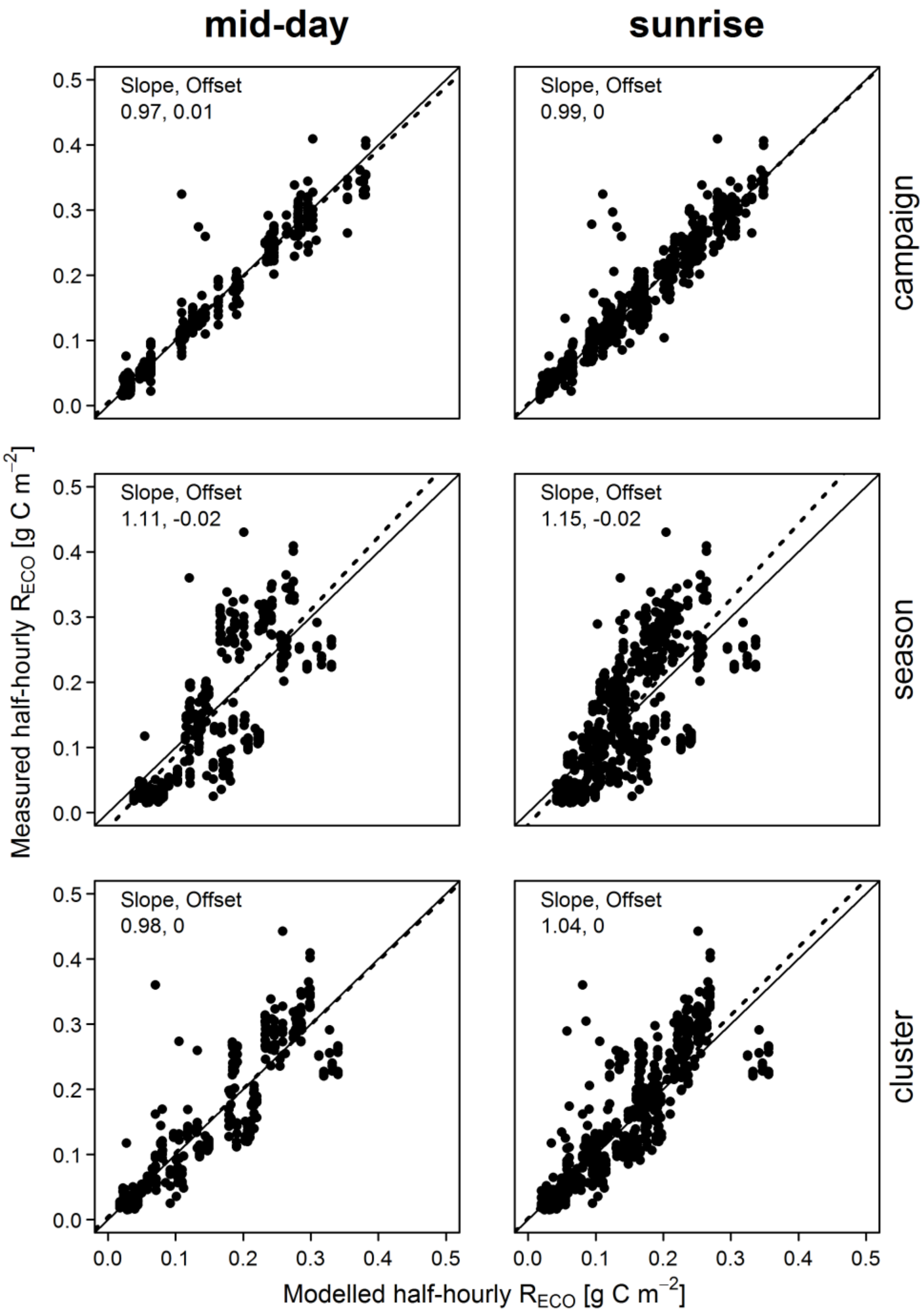

Figure 5.A.2: Calibration between modelled and measured half-hourly R_{ECO} fluxes (g C m^{-2}) for the 2 × 3 approaches during the study period from November 2013 to November 2014. The solid black line shows the 1:1 agreement, the dashed line the linear fit of the actual relationship. Note that R_{ECO} models remain unchanged for the different GPP modelling approaches.

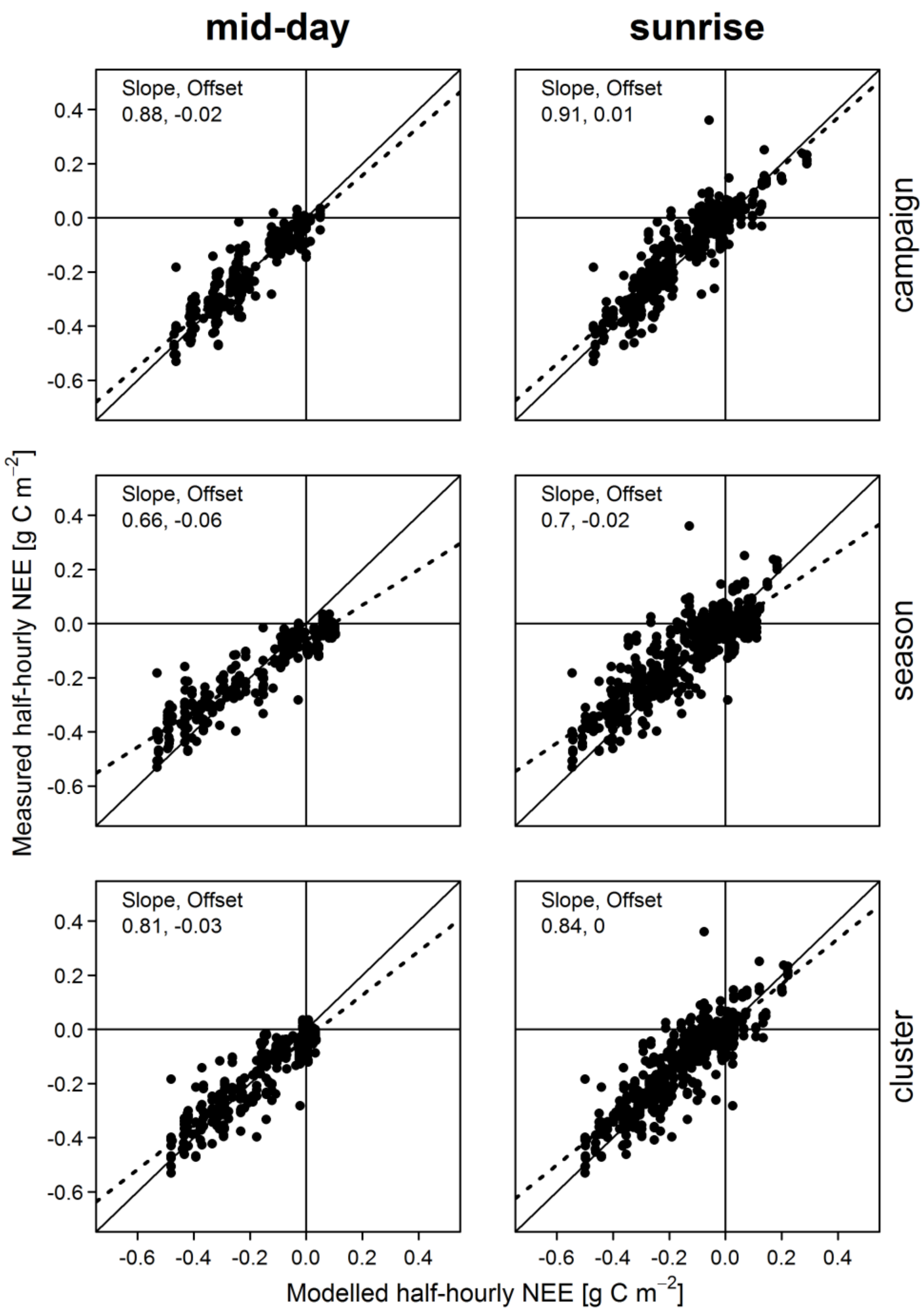

Figure 5.A.3: Calibration between modelled and measured half-hourly NEE fluxes (g C m^{-2}) for the 2 × 3 approaches with direct GPP during the study period from November 2013 to November 2014. The solid black line shows the 1:1 agreement, the dashed line the linear fit of the actual relationship.

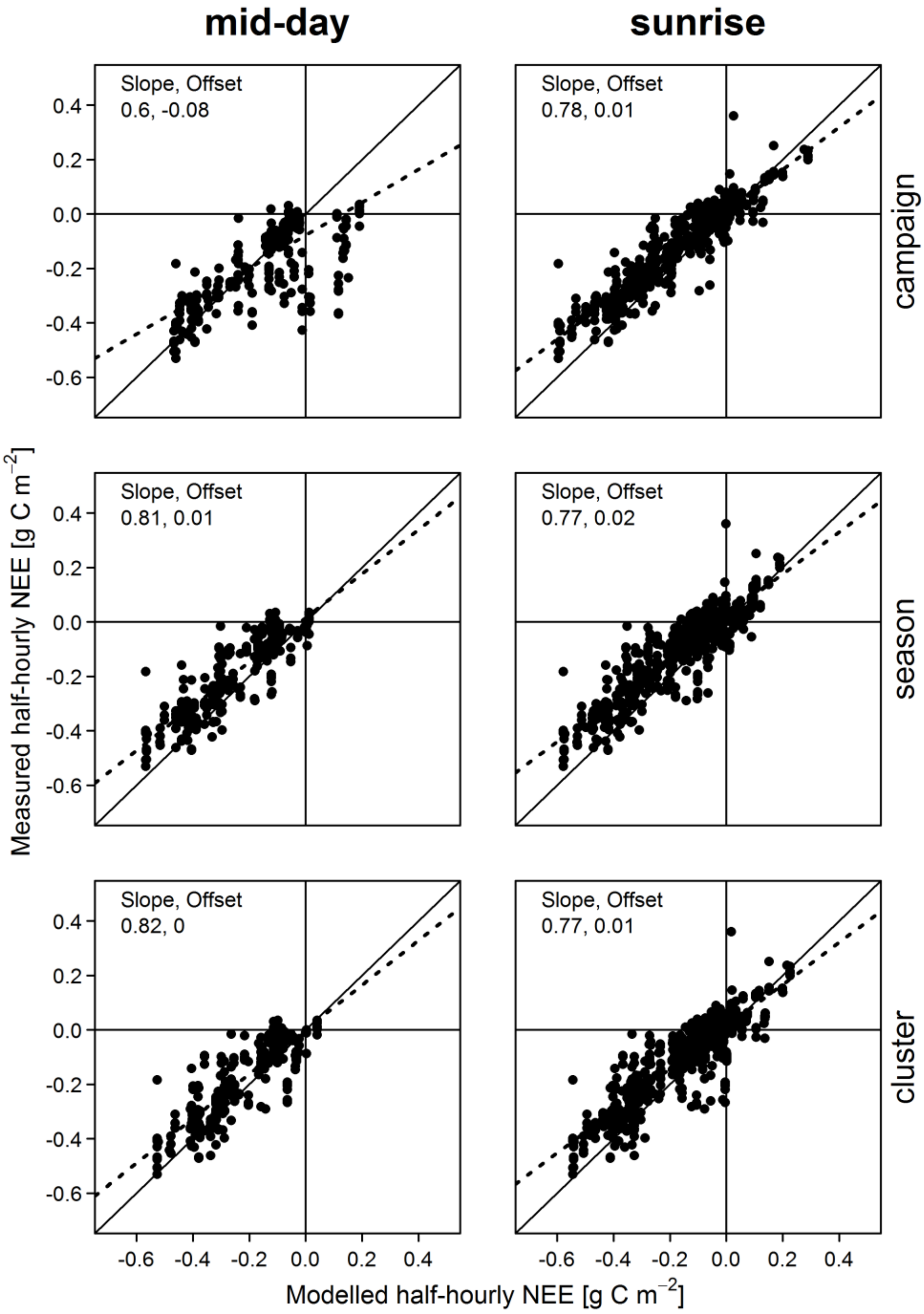

Figure 5.A.4: Calibration between modelled and measured half-hourly NEE fluxes (g C m^{-2}) for the 2 × 3 approaches with indirect GPP during the study period from November 2013 to November 2014. The solid black line shows the 1:1 agreement, the dashed line the linear fit of the actual relationship.

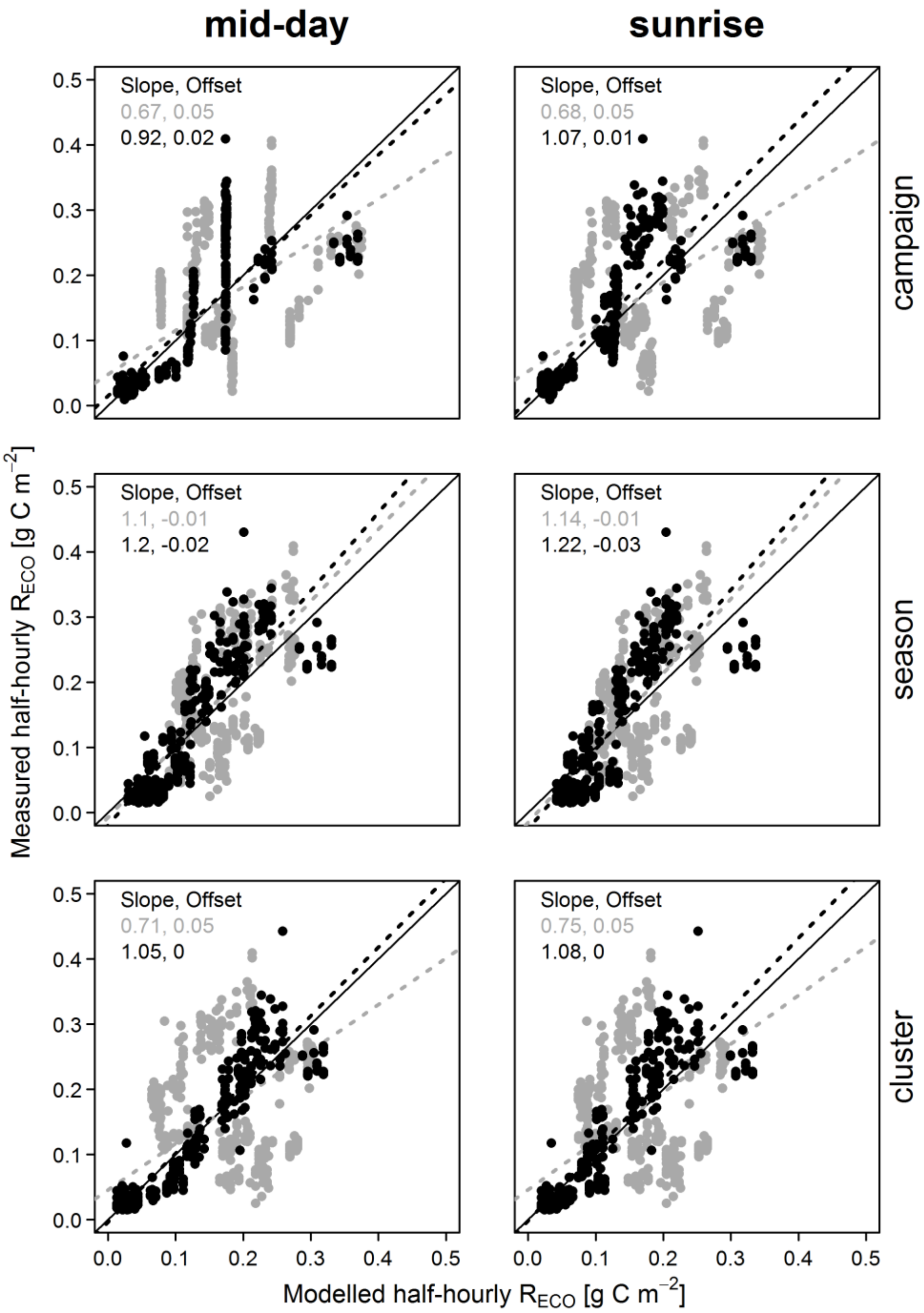

Figure 5.A.5: Leave-one-campaign-out validation between modelled and measured half-hourly R_{ECO} fluxes (g C m^{-2}) for the 2 × 3 approaches during the study period from November 2013 to November 2014. The solid black line shows the 1:1 agreement between modelled and measured fluxes; the dashed line shows the linear fit of the actual relationship. Grey colours represent analyses with fixed start and final campaigns and black colours those with additionally fixed pre- and post-harvest campaigns. Note that R_{ECO} models remain unchanged for the different GPP modelling approaches.

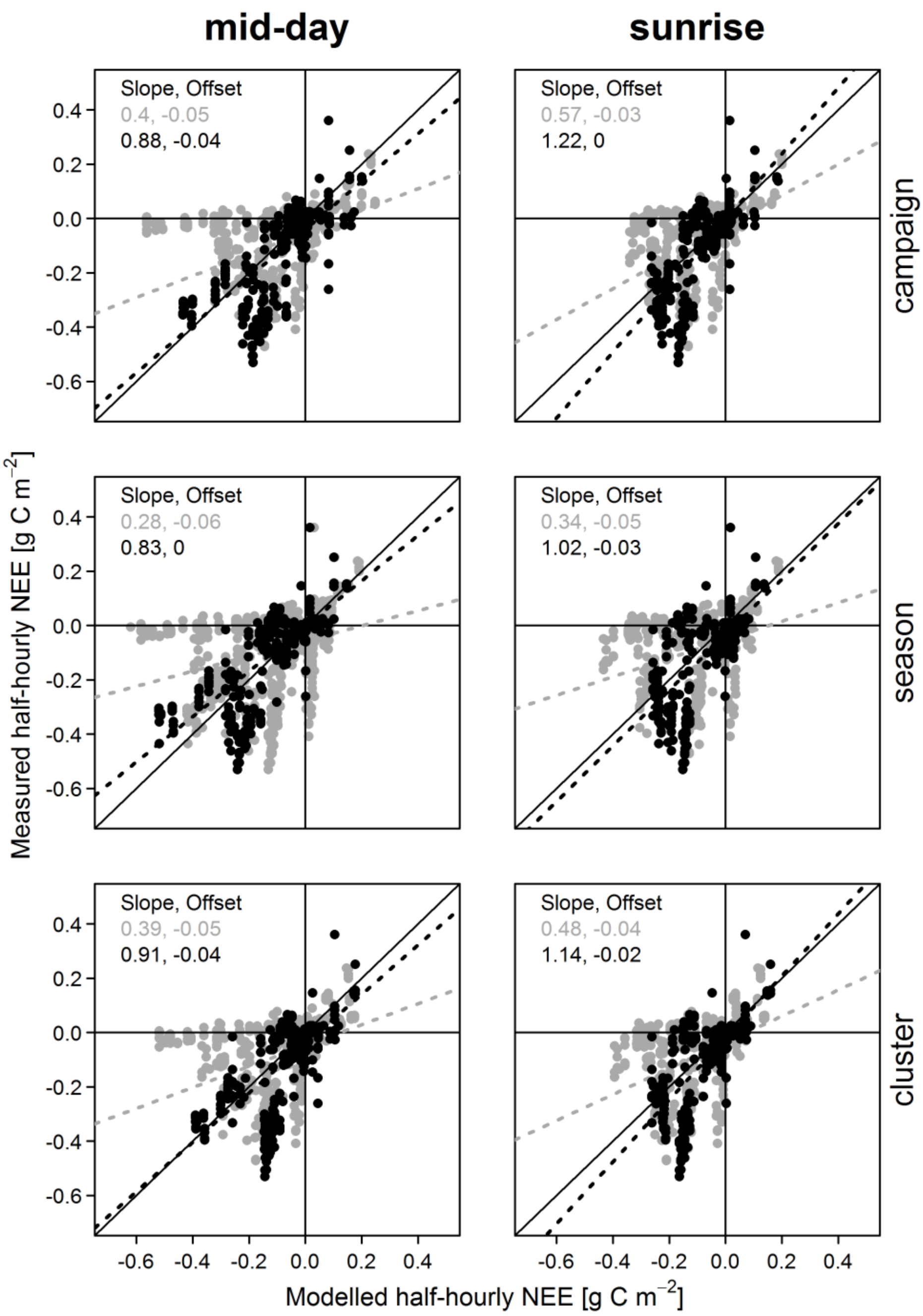

Figure 5.A.6: Leave-one-campaign-out validation between modelled and measured half-hourly NEE fluxes (g C m^{-2}) for the 2 × 3 approaches with direct GPP during the study period from November 2013 to November 2014. The solid black line shows the 1:1 agreement between modelled and measured fluxes; the dashed line shows the linear fit of the actual relationship. Grey colours represent analyses with fixed start and final campaigns and black colours those with additionally fixed pre- and post-harvest campaigns.

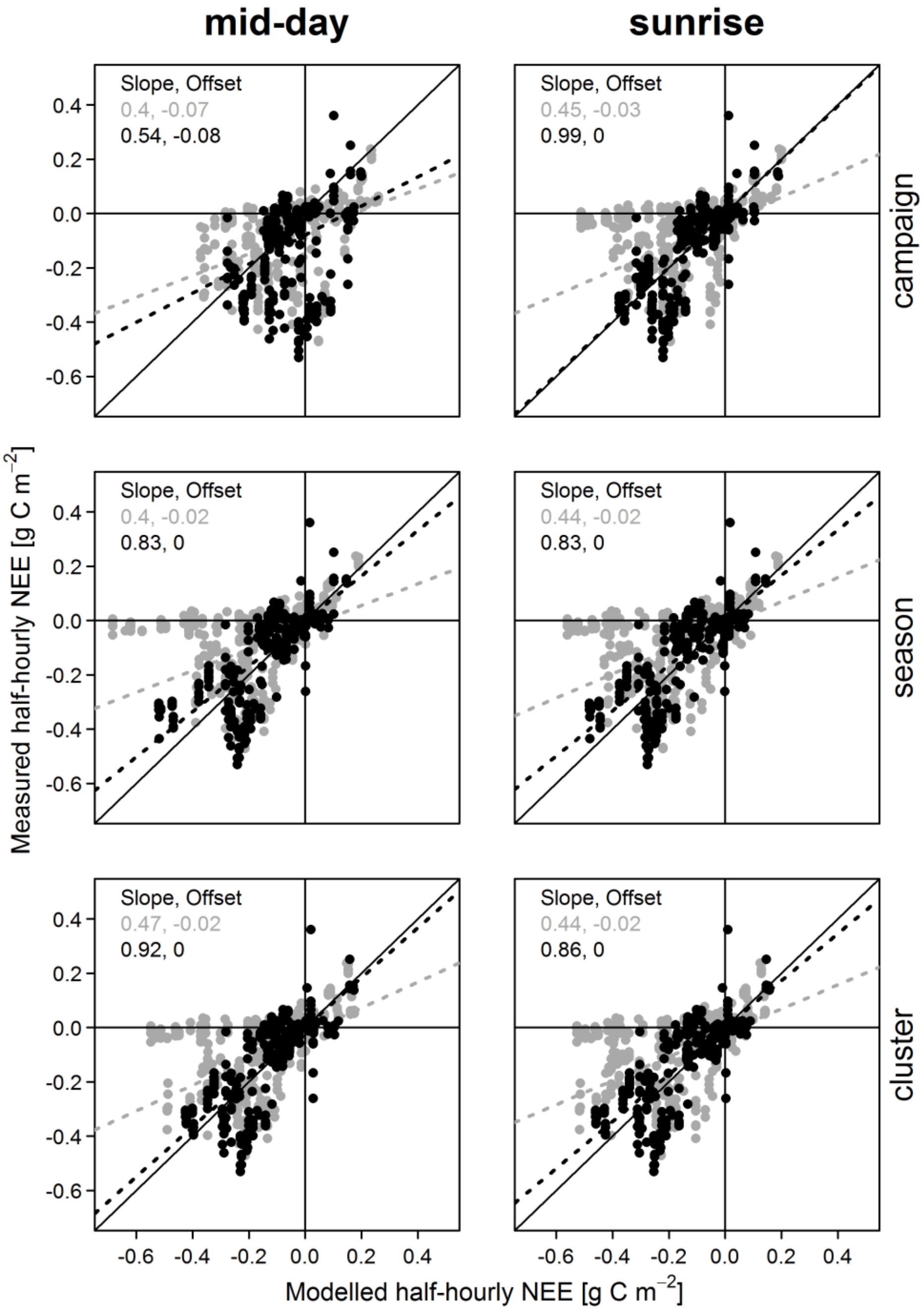

Figure 5.A.7: Leave-one-campaign-out-relationships between modelled and measured half-hourly NEE fluxes (g C m^{-2}) for the 2 × 3 approaches with indirect GPP during the study period from November 2013 to November 2014. The solid black line shows the 1:1 agreement between modelled and measured fluxes; the dashed line shows the linear fit of the actual relationship. Grey colours represent analyses with fixed start and final campaigns and black colours those with additionally fixed pre- and post-harvest campaigns.

6 Tackling uncertainties

SUMMARY

In the following, the limits and conclusions of the chapters 2–5 will be synthesized. First, reasons for decisions of chapter-specific manual closed chamber designs and their relevance on the uncertainty of the derived results will be discussed. Second, the limitation of the applied manual closed chamber methods in the context of temporal and spatial variability of GHG emissions will be outlined. Hereby, the reliability of the C balance of Chapter 4 will be given special emphasis in the context of the recently introduced advances in the closed-chamber method that were intensively tested in Chapter 5. Third, from the applied study designs of the preceding chapters an idealised study design will be discussed to answer questions of peatland management effects on GHG emissions excluding the uncertainties deriving from spatiotemporal limitations. Finally, an outlook on the implications for peatland management and future research will be given.

6.1 DECISIONS REGARDING CLOSED CHAMBER DESIGN

Closed chambers constitute a compromise between the ability to capture GHG fluxes and the risk of disturbing them where every measuerment aims to balance disturbance and effectiveness against each other. Therefore there is no ideal chamber but optimized chambers for particular questions and environments (Günther 2015). Taking major constraints deriving from e.g. chamber design (Pumpanen et al. 2004), atmospheric turbulences (Lai et al. 2012), chamber deployment (Koskinen et al. 2014), or physical environmental changes during closure time (Drösler 2005, Langensiepen et al. 2012) into account, the chamber method is able to minimize the disturbance of the system through the measurement and estimates reasonable GHG fluxes on the field-plot scale (Livingston & Hutchinson 1995, Davidson et al. 2002).

For reasons of different ecosystem and vegetation types, the closed chambers used in Chapter 2, Chapters 3/4 and Chapter 5 differed from each other (Table 6.1). A lower V:A-ratio leads to an increase of concentration change at a fixed flux, in return meaning that lower fluxes can be detected with the same measurement precision at a lower V:A-ratio (Livingston & Hutchinson 1995) which is particularly important for the generally low N_2O fluxes (Figure 6.1). For this reason, the small chamber was chosen for the study at the Neuenkirchener Niederung that focussed on capturing winter N_2O fluxes (Huth et al. 2012). Additionally, the *Molinia* type grassland was cut and grazed before winter and thus average plant height was lower than 20 cm.

Table 6.1: Specifications of the chamber types used in chapters 2–5.

	Chapter 2	Chapters 3 & 4	Chapter 5
Reference	Huth et al. (2012)	Huth et al. (2013), Günther et al. (2015)	Huth et al. (submitted)
Vegetation	Molinia grassland	Reeds/sedges	Forage crops
Maximum vegetation height [m]	0.20	2.30	0.60
Collar/chamber shape	round	round	square
Chamber wall	rigid	flexible	rigid
Basal area [m^2]	0.07	0.31	0.56
~ Aboveground collar height [m]	0.10	0.10	0.10
Chamber height [m]	0.30	1.60–2.30	0.50
Volume [m^3]	0.03	0.53–0.75	0.34
V:A-ratio [m]	0.40	1.70–2.40	0.60
Description	‘small’	‘flexibe’	‘box-shaped’

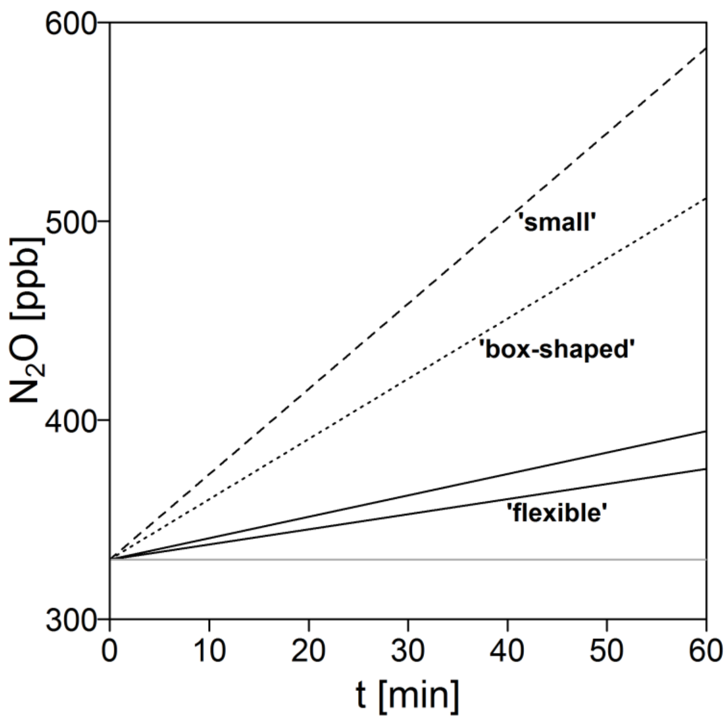

Figure 6.1: Estimated N_2O concentration change during typical chamber-closure times at a fixed efflux of 200 µg m^{-2} h^{-1} for the differening V:A-ratios of the ‘small’ (dashed line, V:A-ratio: 0.4 m), ‘flexible’ (black lines, V:A-ratios: 1.7–2.4 m), and ‘box-shaped’ chambers (dotted line: V:A-ratio 0.6 m). Grey line marks an ambient N_2O concentration of 330 ppb.

In contrast, the reeds and sedges of the Trebel valley mire complex (Huth et al. 2013, Günther et al. 2015) which became dominant after rewetting reached plant heights of more than 2 m. Since they are able to directly (*Phragmites*, *Typha*) or indirectly (*Carex*) aerate their root layer through aerenchymatic tissues cutting may alter their gas transport mechanisms (Armstrong & Armstrong 1991, Brix et al. 1992, Arkebauer et al. 2001, Günther et al. 2014b). Therefore, it was necessary to develop special height adjustable flexible chambers to omit folding or cutting of the vegetation at the control sites. Since these chambers were newly developed, we tested the airtightness of this field setup in the laboratory prior to measurements in February 2011 and found that GHG concentrations in the chamber headspace did not decrease over a priod of 140 min (Figure 6.2). Furthermore, we tested if the flexibility of the chamber wall led to pressure pulses in the headspace when the chamber was subjected to wind and found that maximum pressure fluctuations were less than 1 Pa both during chamber deployment and during strong wind gusts in the field (data not shown). The high V:A-ratio of the flexible chamber increased the minimum detectable GHG flux on the one hand, but is advantageous in terms of minimizing saturation effects over chambers with lower V:A-ratios when large CO_2 and CH_4 fluxes as in Chapters 3/4 occur. Recent research also debates, that chamber transparency may influence GHG exchange of ecosystems with aerenchymatic plants (Günther et al. 2014b, Minke et al. 2014, Günther 2015) and should also be considered in decisions regarding closed chamber design in these ecosystems.

The chambers used at the agricultural research field site (Huth et al. submitted) are described in detail by Drösler (2005). In this case, the large box-shaped chambers were preferable over chambers of other dimensions since the lucerne-clover-grass mix maxed at a plant height of less than 0.6 m. Similarly to the Chapters 3/4, the focus of Chapter 5 was on relatively large CO_2 exchange fluxes. For this reason, the V:A-ratio did not have to be as small as for the study described in Chapter 2.

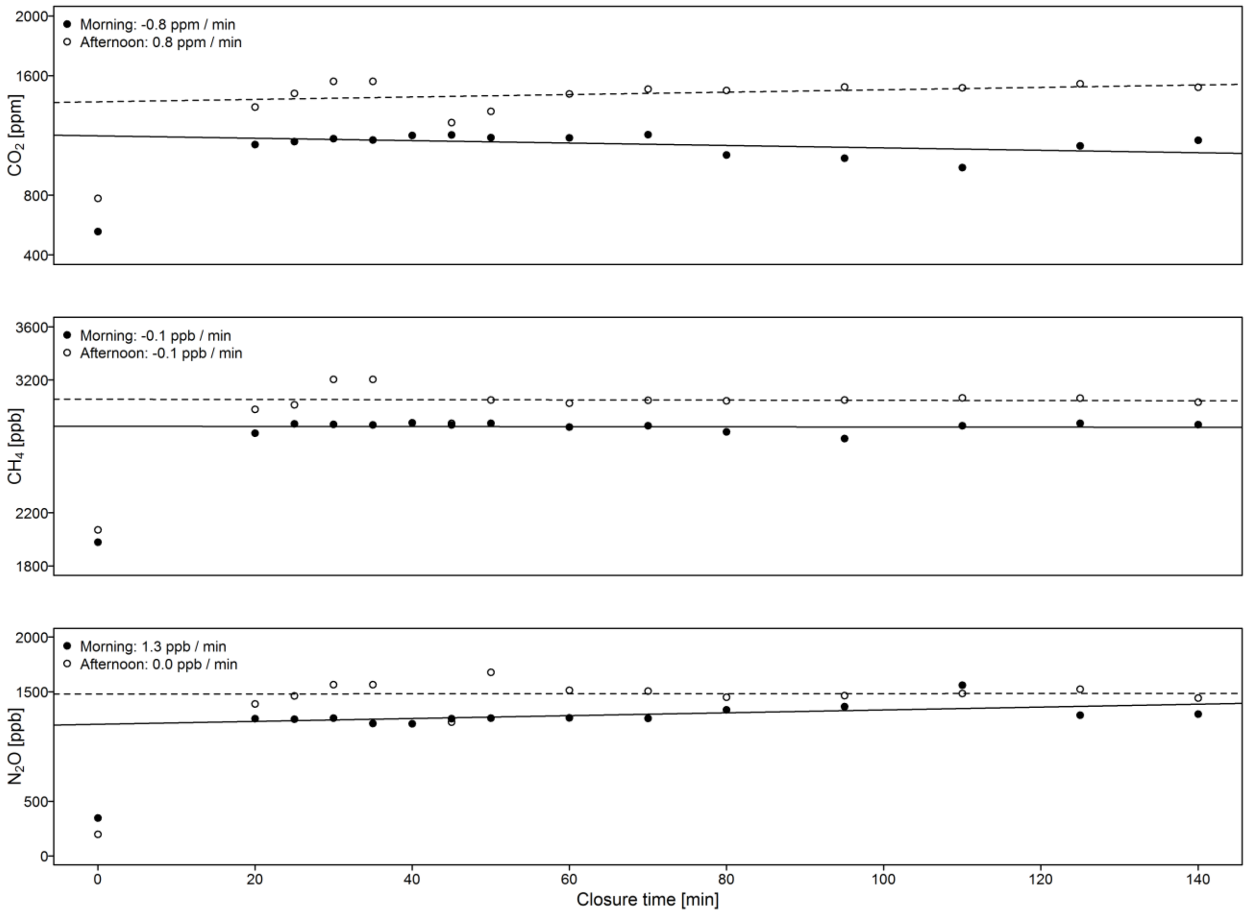

Figure 6.2: Laboratory tests for airtightness of the height-adjustable, flexible chambers at full height (~2.3 m) in the morning and afternoon of 28th February 2011. An initial biogas pulse (~40 % CH_4) was induced into the chamber setup and gas samples were taken between 20 and 140 min closure time thereafter. Concentration changes over closure time were not significant at $P < 0.05$.

6.2 TEMPORAL GHG VARIABILITY

6.2.1 *Diurnal variability*

Diurnal variability of GHG fluxes is - for obvious reasons of changing environmental conditions such as PAR, air/soil temperatures etc. - mainly an issue of CO_2 exchange and CH_4 effluxes. In contrast, N_2O effluxes usually behave erratic and are rather determined by rapid changes in environmental conditions such as fluctuating water level, nitrate availability (through fertilisation), freeze-thaw cycles etc. (Flessa et al.

1995). Their magnitude may change on even smaller than diurnal time scales and are generally not favored by day- or nighttime conditions.

Diurnal CO_2 exchange is in large parts determined by the opposing photosynthesis and respiration fluxes both closely following the changing environmental conditions during day- and nighttime. This variability is usually addressed by different approaches of modelling CO_2 exchange in detail discussed in Chapter 5 (Huth et al. submitted). In Chapter 4, this variability has been addressed through sunrise to noon measurements maximising diurnal variation in environmental conditions with limited measurement resources. In Chapter 5, this 'sunrise' measurement approach in combination with indirect GPP modelling (measured NEE – modelled R_{ECO}) has been shown to be robust towards differing data analyses assumptions (Huth et al. submitted). This means that the variability of diurnal CO_2 exchange in Chapter 4 has been most adequately addressed with the - in terms of measurement frequency and favorable sampling conditions - limited manual closed-chamber method (Gomez-Casanovas et al. 2013).

Diurnal CH_4 variability of wetlands, however, may also be determined through various convective gas transport mechanisms of wetland species such as *Phragmites* and *Typha* (Armstrong & Armstrong 1991, Brix et al. 1992, Arkebauer et al. 2001, Günther et al. 2014b). The shape and magnitude of diurnal CH_4 efflux variability is therefore plant specific with e.g., mid-morning efflux peaks in *Typha* stands (Chanton et al. 1993, Whiting & Chanton 1996) or PAR following effluxes in *Phragmites* stands (Brix et al. 2001, Minke et al. 2014). That parts of the system studied in Chapters 3/4 (Huth et al. 2013, Günther et al. 2015) possess a daytime-dependent magnitude of CH_4 fluxes has been shown by Günther et al. (2014b, Figure 6.3). Therefore the CH_4 sampling order of the three different plant stands studied in Chapters 3/4 was randomised in between "office hours" (9 a.m. to 5 p.m.). However, even these "office hours" have been shown to overestimate CH_4 balances systematically since CH_4 effluxes are usually higher during the day- than the nighttime (Juutinen et al. 2004). The duration and the magnitude of pronounced diel CH_4 emission patterns in near-natural peatland ecosystems is site specific and may be of relevance only during the short growing season (Morin et al. 2014, Koebsch et al. 2015). This suggests that the CH_4 emissions of Chapters 3/4 may be seen as a conservative estimate and that, ideally, CH_4 sampling should be carried out covering full diurnal ranges of air/soil temperatures and PAR similarly to CO_2 sampling whenever possible.

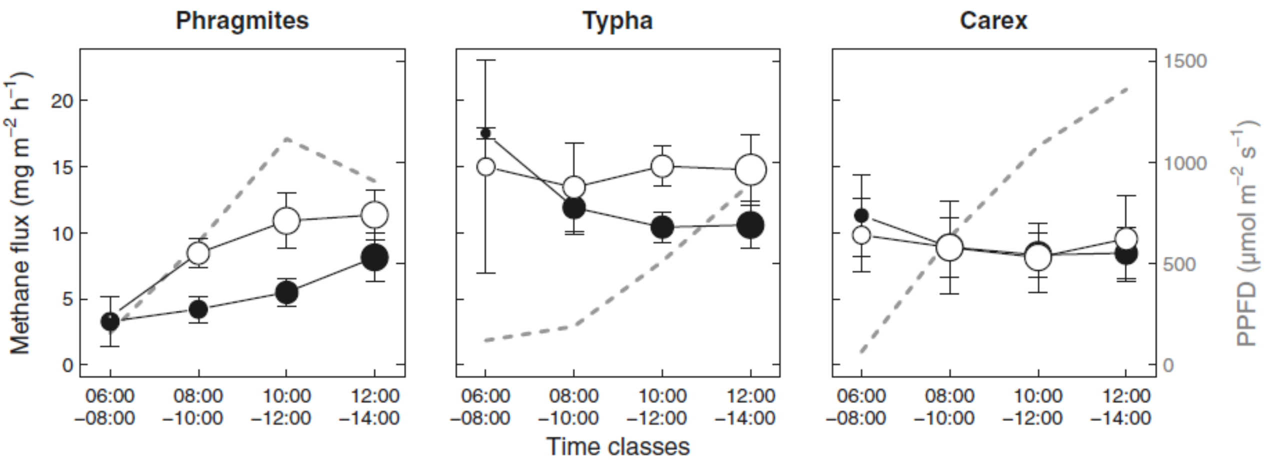

Figure 6.3: Emission patterns of methane fluxes estimated with opaque (black circles) and transparent (white circles) chambers together with mean PPFDs (dashed line) in the three vegetation stands of Chapters 3/4. Circles mark the average value of all measurements that fall within a given time class; error bars denote standard errors of the mean; circle size shows the number of observations included in the mean (n = 1-6). Adopted from and described in detail by Günther et al. (2014b).

6.2.2 Seasonal variability

A distinct seasonality between growing and non-growing season is usually a matter of GHG fluxes that are directly (e.g. photosynthesis, plant respiration) or indirectly (e.g. soil respiration, CH_4 effluxes) dependent on plant-growth and/or substrate availability. Especially photosynthesis and/or the combined ecosystem respiration, which are usually obtained via Michaelis-Menten kinetics (1913) and the Lloyd & Taylor (1994) respiration model are often either corrected for plant growth (Alm et al. 1997, Burrows et al. 2005, Yli-Petäys et al. 2007) or modelled numerous times at different plant-phenological stages (Beetz et al. 2013, Günther et al. 2015, Hoffmann et al. 2015). The quality of the reflection of seasonal CO_2 exchange variation is therefore dependent on an adequate reflection of plant development through the measurements.

In Chapter 4, photosynthesis and respiration modelling was done at approximately six weeks intervals due to limited resources (one CO_2 logger system vs. 18 measurement locations) which may have been insufficient especially during spring shoot growth and senescence. Chapter 5 has shown, that lacking or failing of measurements during the growing season may alter the annual CO_2 balance towards a lower uptake suggesting, that the estimated CO_2 uptake of plant stands studied in Chapter 4 (Table 4.2, 4.3, Figure 6.5) may have been underestimated. Although it remains highly speculative, the Trebel valley mire system studied in Chapter 4 may indeed have been a C sink during the study period 2011-2013.

In contrast, CH_4 and N_2O effluxes in the Chapters 2-4 were derived from measurements at fortnightly intervals. Seasonal CH_4 flux variation was evident from year one of the study period in Chapters 3/4 but absent from year two (Figure 3.2, 4.6, 4.7, 6.6) suggesting that fresh litter decomposition during the growing and late growing season (Koebsch et al. 2013b, Huth et al. 2013, Günther et al. 2015) plays the major role in seasonal CH_4 variability as long as water levels are high enough to ensure an anaerobic environment close to the peat surface. In addition, the fresh water floods following snow melt (Huth et al. 2012) and heavy summer rain (Huth et al. 2013) lasted longer than two weeks indicating that in peatland depressions such as the Neuenkirchener Niederung and the Trebel valley mire the fortnightly interval of GHG sampling may have been sufficient to cover longer lasting extreme events. In terms of N_2O effluxes from peatlands and freeze-thaw-cycling, however, follow up measurements within one week thereafter as carried out in Chapter 2 seem to be necessary to cover the magnitude of freeze-thaw N_2O fluxes (Pihlatie et al. 2010, Huth et al. 2012).

6.2.3 *Inter-annual variability*

Due to the limitation of discontinuous data acquisition and the inherent problem of adequately reflecting temporal GHG dynamics (Günther et al. 2014a, Huth et al., submitted) the majority of the closed-chamber GHG studies focus on treatment comparison experiments and short-term flux dynamics. For this reason long-term GHG time series with closed chambers are rare and seldom exceed five years of study (e.g. Smemo & Yavitt 2006), although climate is acknowledged as a main driver of GHG exchange from natural and anthropogenic disturbed ecosystems (Moore et al. 1998, Frolking et al. 2006). Therefore, the majority of long-term GHG studies (>5 years) are done using the eddy covariance technique most often focusing on CO_2 exchange (Hollinger et al. 2004, Marcolla et al. 2011, Peichl et al. 2011) and less frequently on CH_4 monitoring (e.g., Roulet et al. 2007). To improve long-term GHG monitoring on the ecosystem scale a number of initiatives have started in recent years (e.g. ICOS – Integrated Carbon Observation System, TERENO – Terrestrial Environmental Observatories). However, the fact that the signal of long-term climate variability may outrange soil, plant or treatment effect signals on smaller spatial scales has not only recently got more attention (Chen et al. 2013, Günther et al. 2014a, Figure 6.6), it would also need to integrate both the closed-chamber and the eddy-covariance techniques if the true differences between inter-annual and inter-site variability are to be compared with each other.

The intention of Chapter 2 was the estimation of the potential of winter N_2O fluxes from a drained but unfertilized minerotrophic fen with an average of six days of snow-cover and occasional freeze-thaw cycles during the period of December and February during which large N_2O emissions may occur in temperate climatic regions (Flessa et al. 1995, Teepe et al. 2000). The winter 2009/2010 was extraordinarily in 30 years with a snow-cover period of more than 60 days. The pattern of GHG emissions of the study site Neuenkirchener Niederung was therefore rather typical for a boreal peatland (Alm et al. 1999). In addition, the annual release of N_2O can be very erratic, with a very large temporal and spatial variability (Flessa et al. 1995) and is likely even higher under fertilized temperate fens (Kroon et al. 2010). For these reasons the general conclusions of Chapter 2 are limited to the fact that even under harsh conditions GHG fluxes are higher than expected from pristine fens and nature conservation management alone may not be a suitable measure to reduce GHG emissions from drained peatlands (Huth et al. 2012, Figure 6.5).

In contrast, Chapters 3/4 show that even with the 100-year summer freshwater flood the GHG emissions of the Trebel valley mire remain typical for pristine fens (Dise et al. 1993, Drewer et al. 2010, Huth et al. 2013, Günther et al. 2015). This indicates, that the long-term shift in vegetation composition after rewetting may induce enough resilience of peatland ecosystems in terms of extraordinarily high GHG emissions, that are usually observed within the first years of flooding (Höper et al. 2008, Hahn-Schöfl et al. 2011, Hahn et al. 2015). However, the *Typha* stand had a significant amount of grassland type understorey (Günther 2015, A1 Vegetation data) that formed an easily decomposable litter layer (Figure 6.4). This may have led to the significant CH_4 efflux vs. water table response ($R^2 = 42$ %, Huth et al. 2013) suggesting, that even after 15 years of rewetting the *Typha* stand has not yet been fully established compared to the *Phragmites* and *Carex* stands. Vegetation development after rewetting temperate minerotrophic fens is known to last more than a decade (Timmermann et al. 2011, Bönsel & Sonneck 2011) which limits our conclusions of the effect of *Typha* cropping on GHG emissions to a case study of a highly dynamic state.

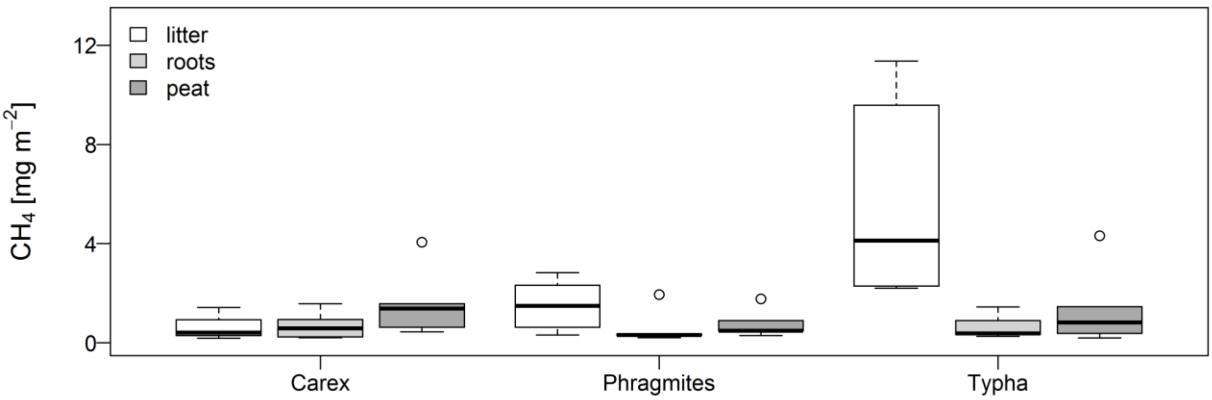

Figure 6.4: Cumulated CH_4 emissions of a 12-day laboratory incubation experiment (20 g soil sample, 200 ml water, 600 ml Erlenmeyer flask) of three dominant vegetation stands (*Carex*, *Phragmites*, *Typha*) in three soil depths ('litter': 0-10 cm, 'roots': 10-20 cm, 'peat': 20-30 cm) with five iterations (3 x 3 x 5, $n = 45$). Analysis of variance resulted in both 'plant' and 'soil depth' as significant factors for the CH_4 emission potential at $P < 0.05$.

Both the Neuenkirchener Niederung and the Trebel valley mire were randomly subjected to extraordinary climate events leading to rather conservative GHG emission estimates on the lower end for the drained study site due to very low winter temperatures (Huth et al. 2012) and on the upper end for the rewetted study site due to the freshwater flooding under high summer temperatures in the late growing season of 2011 (Huth et al. 2013, Günther et al. 2015). Yet, it is still possible to distinguish between them in terms of GHG relevance (Figure 6.5). This may indicate that peatland water management may even outrange the response to climate variability and therefore plays the major role for GHG emission reduction (Couwenberg et al. 2011). Although the overall responses of the studied ecosystems fell into a dry and warm period in the long-term (Chapter 1.2.2, Figure 1.4), the extreme events occurring during the study period were unusual cold and strong snowfall in 2009/2010, heavy summer rain in 2011, and an extraordinarily warm year 2013/2014, however. Therefore, general conclusions of the effects of drainage, nature-conservation, rewetting and paludiculture cannot be drawn for the long-term.

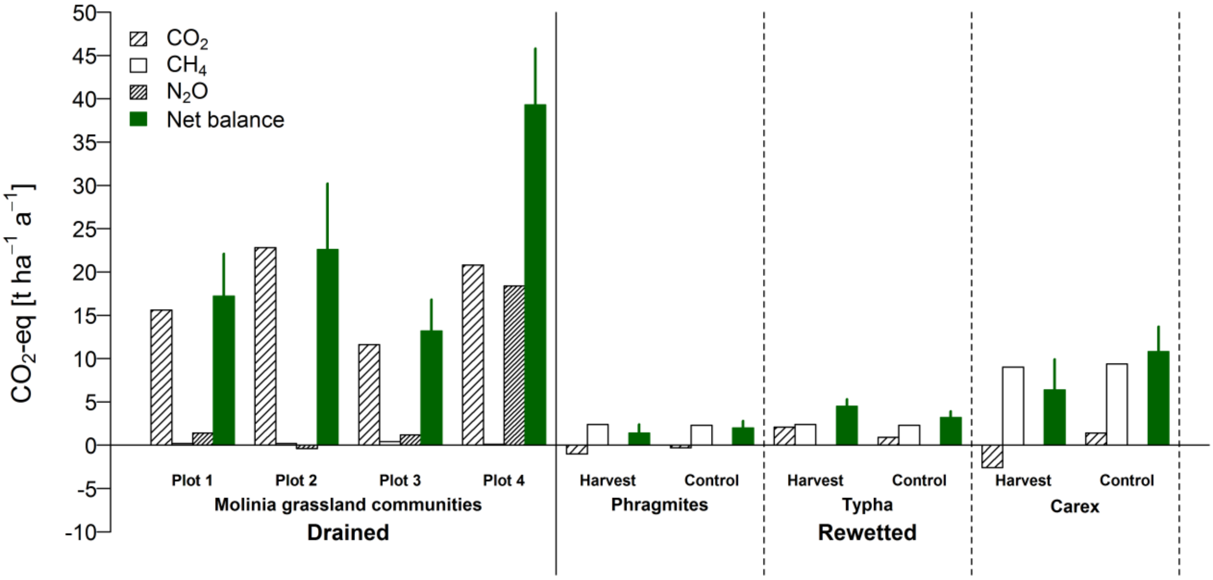

Figure 6.5: Estimated annual CO_2 exchanges, CH_4 and N_2O emissions and GHG balances (in t CO_2-eq ha^{-1} a^{-1}) of the study sites Neuenkirchener Niederung (‘Drained’, derived from winter 2009/10 measurements and literature) and the Trebel valley mire complex (‘Rewetted’, derived and averaged from 03/2011-03/2013 measurements).

Although various studies have related a generally small climate-driven inter-annual variability of CO_2 exchange of grassland-type ecosystems to a counteraction between R_{ECO} and GPP (Peichl et al. 2011, Marcolla et al. 2011, Koebsch et al. 2013a), the short-term the effect (<3 years) of extensive paludiculture use of reeds and sedges on CO_2 exchange and CH_4 emissions is even smaller (Günther et al. 2015). In addition, wet grassland type vegetation is known to adapt differently to short-term and long-term harvesting according to their mowing capability (Granéli 1989; Ostendorp 1999, Güsewell et al. 1998). For these reasons, only long-term monitoring may elucidate the effects of different types and intensities of management practices on the GHG cycling of paludicultural ecosystems. However, long-term monitoring of the effect of typical agricultural land use systems of the temperate climate region exist for C and N cycling on mineral soils (e.g. Körschens et al. 2012) but are absent from organic soils (>20 % SOM) where the longest existing N manipulation experiment is from Mer Bleue, Canada (11–16 years, Juutinen et al. 2016). Since these soils store especially large amounts of C and N compared to their relative share of land surface area, long-term monitoring on the biogeochemical and GHG cycling of peatlands in the temperate and boreal climate regions is an highly overlooked necessity and should be subjected to research in the future.

6.3 SPATIAL GHG VARIABILITY

Chamber-based methods of quantifying GHG exchanges are usually used for plant, soil or treatment comparisons (Elsgaard et al. 2012, Beetz et al. 2013, Günther et al. 2015, Fiedler et al. 2015) and, in principle, allow for an adequate reflection of spatial variability of gas fluxes (Rochette et al. 1991, Davidson et al. 2002, Koch et al. 2014). However, since the nature of GHG fluxes is erratic, an unfeasibly large number of collars would be needed to achieve statistically significant differences of GHG fluxes between treatments (Folorunso & Rolston 1984, Rochette et al. 1991). The spatial design of Chapter 2 had the intention to reflect the different hydrological conditions through the four plots from the higher edges to the lower central parts of the Neuenkirchener Niederung and to quantify spatial variability on the plot scale through the three replicate collars per plot (Huth et al. 2012). Although the drainage ditches within the Neuenkirchener Niederung only shared a minor land cover, they may have been a significant but not identified source for CH_4 as other studies have shown (Schrier-Uijl et al. 2010b, Cooper et al. 2014). In addition, spatial variability also varies in time which e.g. Günther et al. (2014a) have shown in the peatland complex studied in Chapters 3/4 where the expected site-specific differences of CH_4 effluxes between vegetation stands were especially pronounced during the generally high emission level of year one but nearly absent in year two (Figure 6.6). Similar mechanics apply for CO_2 exchange (mostly through soil respiration) and N_2O effluxes on various time scales showing that the spatial variability of GHG fluxes can hardly be uncoupled from their temporal variability. In addition, organic soils are also highly diverse on the landscape level as are management practices which are usually coupled to water management and site characteristics (Beetz et al. 2013, Drösler et al. 2013, Renou-Wilson et al. 2014). Therefore, adequate reflection of spatial variability of GHGs on plot, site and landscape level remains a major limitation for the emission factor estimation with closed chambers from peatland ecosystems.

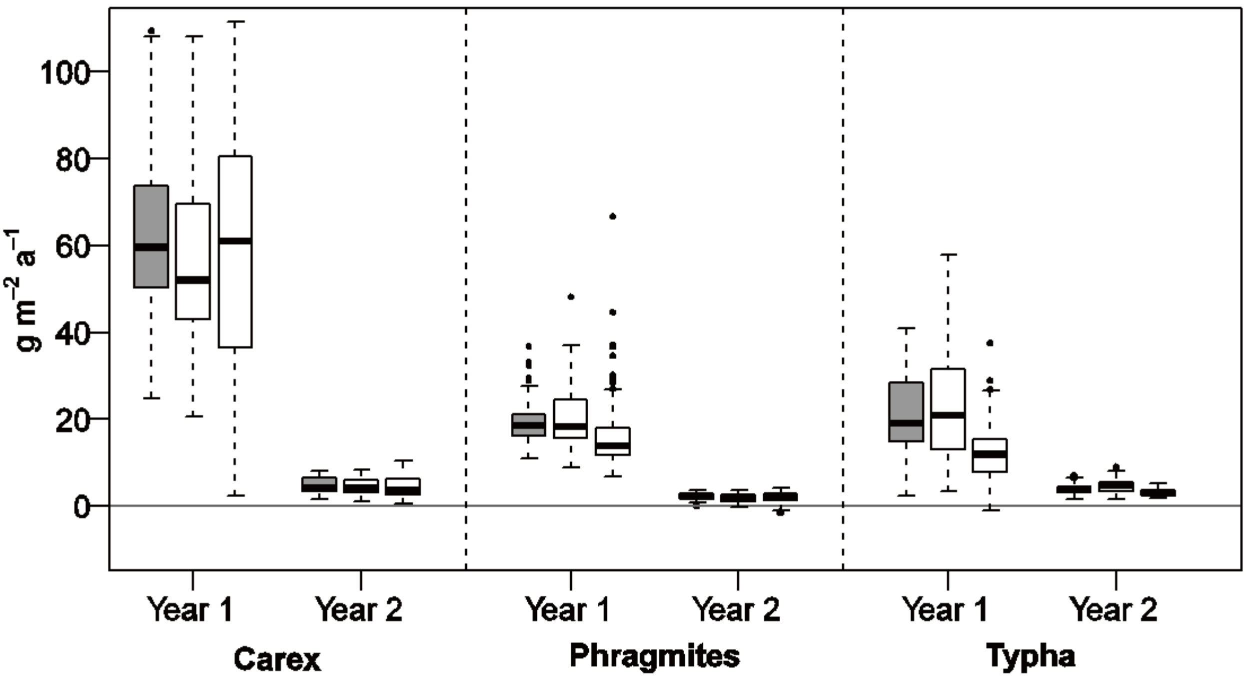

Figure 6.6: Annual CH_4 emissions estimates (g m^{-2} a^{-1}) of the dominant uncut vegetation stands of the study site Trebel valley (see Chapters 3/4) during the two years. Annual budgets were determined using all measurements dates (grey boxes, biweekly data) and half of the measurement dates (white boxes, monthly data). The boxes represent the combined data from all emission estimates of one plant stand during a slightly adjusted Monte Carlo procedure used in Chapter 3 (Huth et al. 2013) and Chapter 4 (Günther et al. 2015). *n* = 600. Adapted from and described in detail by Günther et al. (2014a).

6.4 IMPLICATIONS FOR FUTURE STUDY DESIGN

6.4.1 Extracting the major research questions

From the spatiotemporal limitations of GHG studies discussed in Chapters 6.2 and 6.3 the following chapter debates an outline of an idealised study design to derive reliable GHG balances of anthropogenic disturbed peatlands. Summarising the preceding chapters, the need for long-term GHG monitoring is given for three major reasons (including suggested duration of monitoring):

1. Climate variability (30 years),
2. Plant succession dynamics and ecosystem shifts after rewetting (20 years), and
3. Long-term management effects on biomass productivity of potential paludiculture crops and inherently on the C and N cycle of these systems (20 years).

Since GHG monitoring is time consuming and field equipment is expensive a continuous monitoring over 30 years with closed chambers may be unrealistic. Assuming that rewetting and/or paludicultural peatland use have their strongest impact after implementation (Augustin & Joosten 2007, Figure 6.7) inter-annual sampling frequency should be highest at the beginning of the measure slowly decreasing over time. However, with the background of increasing climatic extreme events and their impact on inter-annual variability sampling periods should be clustered and last two consecutive years at least.

Specifically studying effects of climate change (e.g. increasing temperature, increased CO_2, extended floods and droughts and their interactions) on the GHG exchange and biomass productivity of a temperate fen would need a field lysimeter design covered with greenhouses to constantly control the microclimate of the treatments, hence extensive technological equipment and maintenance. However, the combined climatic effect during a long-term experiment may as well be determined by the monitoring of environmental parameters through the field weather stations. Therefore, an isolated ‘Climate’ experiment will not be outlined in the following. Instead, the focus remains on potential experiments covering plant succession dynamics (‘Rewetting’) and the long-term effects of biomass harvesting (‘Paludiculture’).

Plant succesion after rewetting lasts decades and the spatial extent of possible paludiculture crops over time is hardly predictable. In contrast the estimation of management effects on wetland crops needs homogenous plant stands at the site level to allow for an adequate factorial block design. Therefore the effects of ‘Rewetting’ and ‘Paludiculture’ on GHGs and C cycling have to be addressed in spatially differently designed long-term monitoring experiments. An exemplary 30 years monitoring of GHGs for the ficitive long-term experiments ‘Rewetting’ and ‘Paludiculture’ is given in Table 6.3.

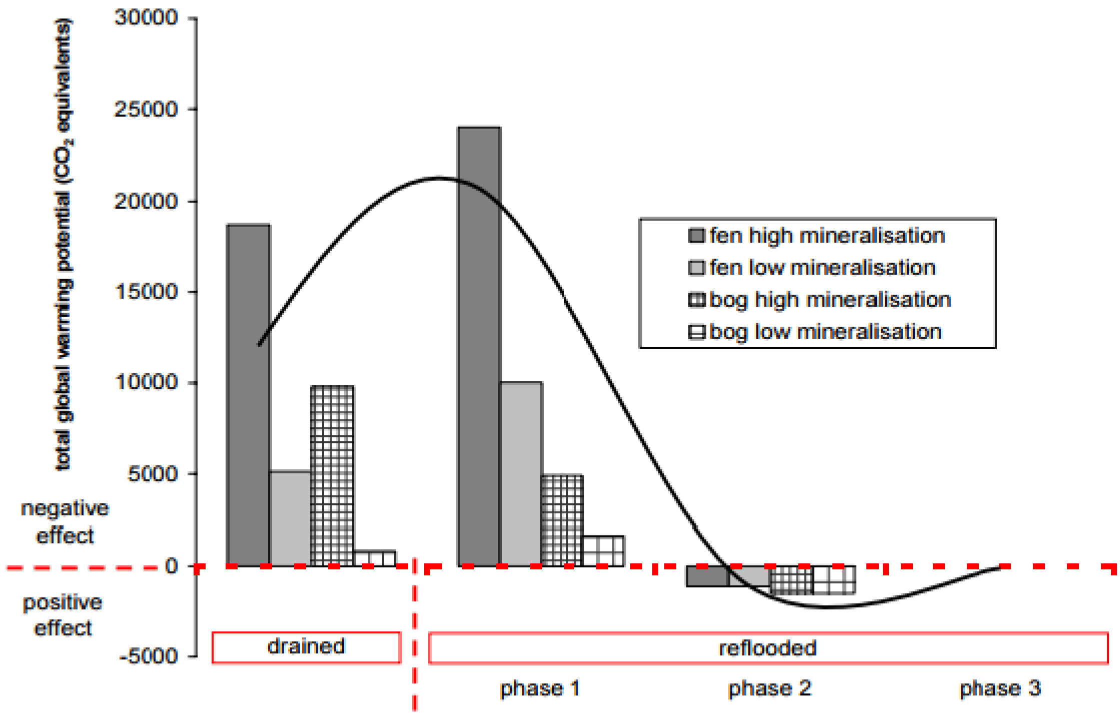

Figure 6.7: Estimated changes in total global warming potential of the greenhouse gas release from Belarusian mires after rewetting (in kg CO_2-eq ha^{-1} a^{-1}). Adapted from Augustin & Joosten 2007.

6.4.2 *The 'Rewetting' experiment*

A number of studies have integrated the advantages of continuous eddy covariance measurements and the spatial resolution obtained through closed chambers on boreal and temperate peatlands for different GHGs (Sachs et al. 2010, Schrier-Uijl et al. 2010a, Forbrich et al. 2011). However, studies running both methods parallel over longer times are rare (Moore et al. 2011). Up to date, the design applied in recent years in the Hütelmoor, a coastal brackish fen in north-east Germany (Koebsch et al. 2013a, Koebsch et al. 2013b, Koch et al. 2014, Koebsch et al. 2015, Hahn et al. 2015) is unique both in spatial coverage of CH_4 monitoring of different plant types and in temporal coverage of closed chamber (CH_4) and eddy covariance measurements (CO_2 and CH_4, now running the sixth year). It was originally designed for the extrapolation of CH_4 closed-chamber measurements to the ecosystem scale with vegetation data and eddy covariance measurements (Jurasinski, oral communication). A fundamental altering in the water supply in winter 2009/10 led to a quasi-permanent flooding and a destabilisation of the ecosystem (Hahn et al. 2015) counteracting the initial research goals with erratic GHG emissions. Due to its original intention it is not yet fully adapted to long-term monitoring, since CO_2 exchange measurements with closed chambers and profile-based C storage estimations are lacking. However, the main design consiting of an eddy covariance tower surrounded by 33 closed-chamber measurements organised in 11 clusters and 64 vegetation sampling points seem to be promising also for long-term vegetation development following rewetting ('Rewetting' experiment) and its effect on GHGs and C cycling (Figure 6.8).

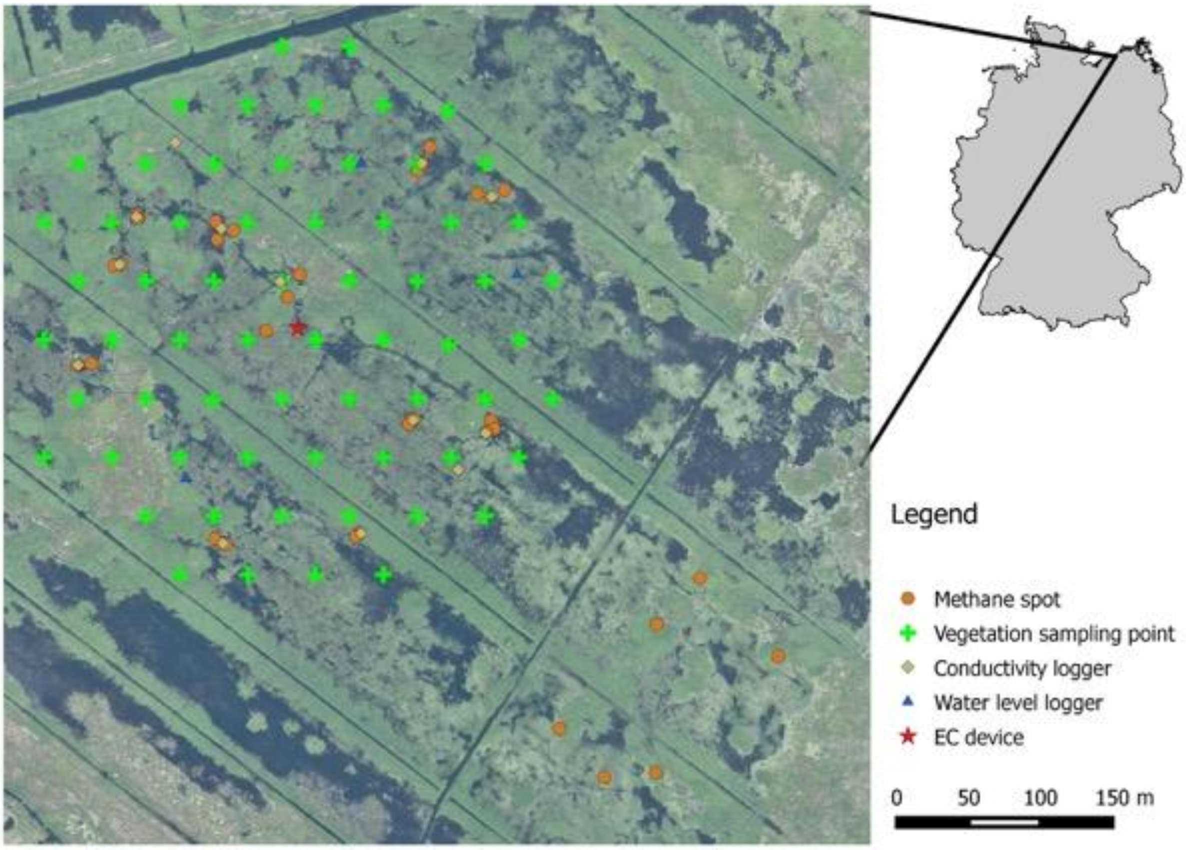

Figure 6.8: Study area 'Hütelmoor' with sampling locations (see legend). The spots of the project for GHG measurements are arranged in eleven clusters of three featuring different distances between spots within. Adapted from Jurasinski (oral communication).

An alternative approach to quantify GHG and ecosystem dynamics following rewetting could be a space-for-time substitution study. In north-east Germany, nearly all of the former percolation mires that formed nearly 50 % of the peatland types, have been deeply drained at the end of the 1960s undergoing intensive grassland management until the 1990s. From the mid-1990s rewetting projects have been implemented on numerous peatlands of similar genesis and land-use history with only little climatic deviation. Therefore it may be possible, to find a sufficiently high number (e.g. $n = 5$) of formerly deep-drained, minerotrophic fens that have been rewetted during the last 20 years. With this approach, the effect of 'Rewetting' on GHG emissions from temperate fens may tentatively be studied within three years.

6.4.3 *The 'Paludiculture' experiment*

The suggested design of 'Paludiculture' is in need of homogenous site characteristics and plant stands that are in a climaxed state (e.g. the extensive reed stands of the Peene river mouth, NE Germany). An exemplary factorial block design of *Phragmites* harvesting and fertilization of different intensities is given in Table 6.2. Since the blocks are much smaller (25 m^2) than usual footprints of eddy covariance measurements (10,000 m^2), only chamber-based methods are able to catch the relatively small differences of C balances between the management treatments. In addition, reed stands are permanent pastures simplifying the block design due to a lacking crop rotation. Single closed-chamber measurements and topsoil parameter determination can therefore be carried out withing each parcel. The suggested parcel size is 5 x 5 m with an edge effect buffer of 2 m. Since reeds have a relatively low mowing capability (Briemle & Ellenberg 1994), the overall experiment may consist of two harvest frequencies (1 / 2 harvests per year) and three fertilisation treatments (0 / 90 / 180 kg N ha^{-1}) plus an untreated control (unharvested / unfertilised). Therefore the seven treatments with four replications each could be carried at an overall pasture of 51 x 30 m.

Table 6.2: Randomised block design of a fictive 'Paludiculture' experiment on a homogenous *Phragmites* stand (n = 4). Treatments 1 – unharvested / unfertilized, 2 – 1 x harvest / unfertilized, 3 – 1 x harvest / 90 kg N, 4 – 1 x harvest / 180 kg N, 5 – 2 x harvest / unfertilizied, 6 – 2 x harvest / 90 kg N, 7 – 2 x harvest / 180 kg N.

A	6	3	5	4	2	7	1
B	4	3	6	5	7	1	2
C	7	4	2	1	5	3	6
D	2	5	1	4	6	7	3

A space-for-time substitution study for the 'Paludiculture' research question similar to the 'Rewetting' question will most likely not be applicable for the following reason: Apart from isolated regions in the northern countries (South Sweden, South Finland, Estonia & Latvia), at Lake Neusiedl (Austria) and in the Danube Delta (Romania, Ukraine) reed harvesting on temperate peatlands has become relatively scarce in Europe (Köbbing et al. 2013). Therefore, it will be unlikely that differences in the effect of management intensities on GHG emissions can be observed on temperate peatlands with similar genesis and land-use history under a little varying climate.

6.4.4 Measurement details

Closed-chamber measurements should be carried out over two days approximately every four weeks with one additional measurement campaign during spring shoot growth and one during late summer senescence (= fortnightly interval, 14 measurement campaigns during one year) depending on weather quality (sunshine, no rain). To cover diurnal variability, CH_4 measurements should be carried out with a portable CH_4 logger system (e.g. UGGA, Los Gatos Research, USA) once at sunrise and once in the afternoon during the growing season with a randomized measurement point sampling order. Depending on the minimal detectable flux of the logger system, CH_4 measurement time could be reduced down to 3 min per point. In this case, one portable logger system would be enough to cover all measurement points. CO_2 exchange measurements should be carried out the day after with quasi-continuous sunrise to noon measurements (see Chapter 5). If every 2 of the 11 sampling point clusters of the ‘Rewetting’ experiment are close to each other, the CO_2 measurements could be achieved with 6 portable CO_2 logger systems (e.g. LI-840A, LI-COR, USA) and 6 persons (= 1 logger for 6 measurement points). The CO_2 exchange measurements of ‘Paludiculture’ experiment could be achieved with 7 CO_2 logger systems and 7 persons (= 1 logger for 4 measurement points) during one measurement day or with 4 / 4 during two consecutive measurement days. An extensive boardwalk system would be needed in any case.

Eddy-covariance measurements and water-table monitoring ideally should run continuously. Water table for ‘Rewetting’ should be monitored at every cluster (= 11 groundwater pipes) for ‘Paludiculture’ in every parcel (= 16 groundwater pipes) with automated loggers (e.g. Levellogger Gold Junior M5, Solinst, Canada).

Vegetation development should be monitored by measuring plant height of dominant species and leaf area index (e.g. SunScan Canopy Analysis System, Delta-T Devices, UK) during every closed-chamber measurement campaign. Additionally, plant species composition should be determined at the end of July every year using the scale for cover-abundance estimation (Reichelt & Wilmanns 1973). Additionally, aerial images (color composite) should be taken with a high resolution (e.g., with a drone down to 3 cm) every year at the end of July for the ‘Rewetting’ experiment to cover long-term plant development on the ecosystem scale. Biomass harvesting for the ‘Paludiculture’ experiment should be done in the whole parcel and the GHG measurement collars separately during the desired time of harvest (e.g. winter for *Phragmites*, *Typha*). Dry matter, C and N content should be determined to estimate their export from the system.

Topsoil properties such as soil organic matter, C, N, P should be taken annually from every cluster/parcel at the end of the non-growing season to account for long-term nutrient state changes. The profile based C storage estimation should be carried out at six georeferenced vegetation points that are exemplary for the initial composition of dominant species of the ‘Rewetting’ experiment and within each treatment of the ‘Paludiculture’ experiment. Peat profiles should be taken at the start of the experiments and every 15 years thereafter. Sampling depth has to reach the mineral soil to account for bulk density changes and changing decay rates of the acrotelm during plant succession and nutrient states changes. With this approach, profile based C sequestration estimates can be compared to short-term C exchange of the closed chamber measurements (Roulet et al. 2007). Finally, the re-installment of long-term C accumulation rates of ‘Rewetting’ and ‘Paludiculture’ may be evaluated.

Table 6.3: Long-term monitoring scheme of the effects of ‘Rewetting’ and ‘Paludiculture’ on GHGs and C cycling.

		Year																													
		1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30
Rewetting/plant succession	Closed-chamber measurements	x	x	x	x	x				x	x	x				x	x	x					x	x						x	x
	Profile-based C-storage estimation	x														x															x
	Eddy covariance	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
	Water-table monitoring	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
	Vegetation monitoring	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
	Remote sensing	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
Paludiculture	Closed-chamber measurements	x	x	x	x	x				x	x	x				x	x	x					x	x						x	x
	Profile-based C-storage estimation	x														x															x
	Water-table monitoring	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
	Vegetation monitoring	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
	Biomass harvesting / sampling	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
	Topsoil C and N sampling	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x

6.5 IMPLICATIONS FOR PEATLAND USE

So far, there are no long-term monitoring experiments on organic soils. Therefore all general conclusions for peatland-use options on GHG emissions can momentarily be drawn via meta analyses only. Plant species composition and water table have been discussed as the main drivers of GHG exchange from peatland ecosystems (Dias et al. 2010, Couwenberg et al. 2011). That peatland water management may even outrange the response of GHG emissions to climate variability is also supported by the results of this thesis (Chapter 6.2.3, Figure 6.5). However, the reliability of these proxies as tools for an adequate prediction of GHG emissions is limited to systems with a quasi-equilibrium state. Some recently rewetted ecosystems show strong deviations from the generally used explanatory concepts of GHG emissions (Hahn-Schöfl et al. 2011, Hahn et al. 2015). In addition, these proxies do not account for the large variation in land-use and management options applied in drained and rewetted peatlands, which has just recently got more attention (Beetz et al. 2013, Drösler et al. 2013, Renou-Wilson et al. 2014, Günther et al. 2015). Therefore, proxies like plant species composition and water table may be used as a general management tool, when integrating the climatic effect of peatland rewetting over larger areas and over longer time scales while keeping in mind that the true site- and time-specific GHG emissions may largely deviate from the predicted GHG emissions.

6.6 OUTLOOK

Land-use and management options of formerly intensively used minerotrophic fens such as nature conservation management, rewetting and paludiculture have been shown to behave differently in terms of GHG emissions. Aside strong inter-annual GHG variability and extreme weather events a clear distinction between a drained and a rewetted peatland has confirmed water level to be of major importance for GHG emissions even on short time scales.

Increasing frequency of extreme weather events due to climate change and their effect on the resilience of peatland ecosystems towards GHG emissions and C balance will become of interest in the future. However, it has been shown, that even under a 100-year summer freshwater flooding, typical minerotrophic fens such as the Trebel valley behave similar compared to pristine fens in terms of GHG emissions after 15 years of rewetting. A space-for-time substitution experiment on a variety of river valley fens in the southern Baltic region would elucidate the success ratio of this effect on a larger spatial scale. Long-term experiments will be the only option however, to cover a higher frequency of extreme weather anomalies.

Additionally, the use of typical peatlands crops such as *Phragmites*, *Typha* and *Carex* on the GHG and C balance has been tested in a rewetted minerotrophic fen. Harvesting emergent macrophytes does not add to the climate effect of these ecosystems in the short term. If technical solutions allow the commercial harvesting of the macrophytes of these ecosystems paludiculture may be a peat body protective and climate cooling peatland use option compared to conventional peatland use. However, long-term paludicultural research on rewetted peat soils has to be installed when questions about harvesting effects and rewetting on C sequestration shall be answered.

Tackling uncertainties

Although the chamber-based methods of this study to derive short-term GHG balances have only been applied in recent years, it was shown that it is possible to adequately address short-term GHG dynamics. However, a standardised framework on GHG measurements and modelling has yet to be developed. Apart from this, future GHG research most likely has to address long-term dynamics of inter-annual climate variability, climate change, ecosystem shifts and C sequestration on various spatial scales. Similar to long-term climate station networks, long-term GHG monitoring networks over representative biogeographical regions have to be installed to reduce the high uncertainty associated with the terrestrial C sources and sinks.

This thesis aimed to address uncertainty factors for determining GHG balances of temperate peatlands. Although land use options are various and climate extremes have been recorded during the studies of this thesis, the great importance of water supply and hydrology on GHG balancing has been supported. In terms of the effect of climate change and land use GHG emissions from temperate peatlands this thesis extracted and highlighted the need for long-term field trials. Compared to the time scales on which peatland formation, C sequestration and climate change occur, however, long-term field trials may answer these questions withing the short lifetime of man. Although climate change has accelerated during the last century, mankind may therefore still be able to reduce its impact it has on our earth, leaving behind an intact world for future generations.

References

Adkinson AC, Syed KH & Flanagan LB (2011) Contrasting responses of growing season ecosystem CO_2 exchange to variation in temperature and water table depth in two peatlands in northern Alberta, Canada. *Journal of Geophysical Research*, **116**, Article G01004.

Afreen F, Zobayed S, Armstrong J & Armstrong W (2007) Pressure gradients along whole culms and leaf sheaths, and other aspects of humidity-induced gas transport in *Phragmites australis*. *Journal of Experimental Botany*, **58**, 1651–1662.

Alm J, Saarnio S, Nykänen H, Silvola J & Martikainen PJ (1999) Winter CO_2, CH_4 and N_2O fluxes on some natural and drained boreal peatlands. *Biogeochemistry*, **44**, 164–186.

Alm J, Talanov A, Saarnio S, Silvola J, Ikkonen E, Aaltonen H, Nykänen H & Martikainen PJ (1997) Reconstruction of the carbon balance for microsites in a boreal oligotrophic pine fen, Finland. *Oecologia*, **110**, 423–431.

Anonymous (2009) Konzept zum Schutz und zur Nutzung der Moore (Concept for the protection and usage of peatlands). Ministerium für Landwirtschaft, Umwelt und Verbraucherschutz (eds), Schwerin, Germany, 109 pages.

Anonymous (2011) Agrarbericht des Landes Mecklenburg-Vorpommern 2011 (Agricultural report oft he federal state Mecklenburg Western-Pomerania 2011). Ministerium für Landwirtschaft, Umwelt und Verbraucherschutz (eds), Schwerin, Germany, 111 pages.

Arbeitsgruppe Moorboden (1965, 1967) Standortgutachten Moorgebiet „Trebelniederung bei Triebsees“. Institut für Grünland und Moorforschung Paulinenaue (eds), Berlin, Germany.

Arkebauer TJ, Chanton JP, Verma SB & Kim J (2001) Field measurements of internal pressurization in *Phragmites australis* (Poaceae) and implications for regulation of methane emissions in a midlatitude prairie wetland. *American Journal of Botany*, **88**, 653–658.

Armstrong J & Armstrong W (1991) A convective through-flow of gases in *Phragmites australis* (Cav.) Trin. Ex Steud. *New Phytologist* **114**, 121–128.

Askaer L, Elberling B, Friborg T, Jørgensen CJ & Hansen BU (2011) Plant-mediated CH_4 transport and C gas dynamics quantified in-situ in a *Phalaris arundinacea*-dominant wetland. *Plant and Soil*, **343**, 287–301.

Aurela M, Laurila T & Tuovinen J (2002) Annual CO_2 balance of a subarctic fen in northern Europe: Importance of the wintertime efflux. *Journal of Geophysical Research*, **107**, Article 4607.

Augustin J & Chojnicki B (2008) Austausch von klimarelevanten Spurengasen, Klimawirkung und Kohlenstoffdynamik in den ersten Jahren nach der Wiedervernässung von degradiertem Niedermoorgrünland. In: Gelbrecht J, Zak D & Augustin J (eds) *Phosphor- und Kohlenstoff- Dynamik und Vegetationsentwicklung in wiedervernässten Mooren des Peenetals in Mecklenburg-Vorpommern*. Leibniz-Institut für Gewässerökologie und Binnenfischerei, Berlin, pp. 50–67.

Augustin J & Joosten H (2007) Peatland rewetting and the greenhouse effect. *IMCG newsletter*, **2007/3**, 29–30.

References

Augustin J, Merbach W & Rogasik J (1998) Factors influencing nitrous oxide and methane emissions from minerotrophic fens in northeast Germany. *Biology and Fertility of Soils*, **28**, 1–4.

Barnola JM, Raynaud D, Korotkevich YS, Lorius C (1987) Vostok ice core: A 160,000 year record of atmospheric CO_2. *Nature*, **329**, 408–414.

Bauwe A, Jurasinski G, Scharnweber T, Schröder C & Lennartz B (2015) Impact of climate change on tree-ring growth of Scots pine, common beech and pedunculate oak in northeastern Gemany. *iForest* (early view), doi: 10.3832/ifor1421-008.

Beetz S, Liebersbach H, Glatzel S, Jurasinski G, Buczko U & Höper H (2013) Effects of land use intensity on the full greenhouse gas balance in an Atlantic peat bog. *Biogeosciences*, **10**, 1067–1082.

Beniston M, Stephenson DB, Christensen OB et al. (2007) Future extreme events in European climate: an exploration of regional climate model projections. *Climatic Change*, **81**, 71–95.

Berger A, Mélice JL, Loutre MF (2005) On the origin of the 100-kyr cycles in the astronomical forcing. *Paleooceanography*, **20**, PA4019.

Bernard JM, Solander D & Kvét J (1988) Production and nutrient dynamics in *Carex* wetlands. *Aquatic Botany*, **30**, 125–147.

Berry J & Björkman O (1980) Photosynthetic response and adaptation to temperature in higher plants. *Annual Review of Plant Physiology*, **31**, 491–543.

Blain D, Murdiyarso D, Couwenberg J, Nagata O, Renou-Wilson F, Sirin A, Strack M, Tuittila E-S, Wilson D, Evans CD, Fukuda M & Parish F (2014) Rewetted organic soils. In: Hiraishi T, Krug T, Tanabe K, Srivastava N, Jamsranjav B, Fukuda M & Troxler T (eds.) *2013 Supplement to the 2006 Guidelines for National Greenhouse Gas Inventories: Wetlands*. Geneva, Switzerland, **Chapter 3**, pp. 3.1–3.43.

Bönsel A & Sonneck A (2011) Effects of a hydrological protection zone on the restoration of a raised bog: a case study from Northeast-Germany 1997–2008. *Wetlands Ecology and Management*, **19**, 183–194.

Bönsel A & Sonneck A (2012) Development of ombrotrophic raised bogs in North-east Germany 17 years after the adoption of a protective program. *Wetlands Ecology and Management*, **20**, 503–520.

Bolinder MA, Kätterer T, Andrén O & Parent LE (2012) Estimating carbon inputs to soil in forage-based crop rotations and modelling the effects on soil carbon dynamics in a Swedish long-term field experiment. *Canadian Journal of Soil Science*, **92**, 821–833.

Briemle G & Ellenberg H (1994) Zur Mahdverträglichkeit von Grünlandpflanzen. Möglichkeiten der praktischen Anwendung von Zeigerwerten. *Natur und Landschaft*, **69**, 139–147.

Brix H, Sorrell BK & Lorenzen B (2001) Are *Phragmites*-dominated wetlands a net source or net sink of greenhouse gases? *Aquatic Botany*, **69**, 313–324.

Brix H, Sorrell BK & Orr PT (1992) Internal pressurization and convective gas flow in some emergent freshwater macrophytes. *Limnology and Oceanography*, **37**, 1420–1433.

Bubier JL, Crill PM, Moore TR, Savage K & Varner RK (1998) Seasonal patterns and controls on net ecosystem CO_2 exchange in a boreal peatland complex. *Global Biogeochemical Cycles*, **12**, 4, 703–714.

References

Burrows EH, Bubier JL, Mosedale A, Cobb GW & Crill PM (2005) Net ecosystem exchange of carbon dioxide in a temperate poor fen: a comparison of automated and manual chamber techniques. *Biogeochemistry*, **76**, 21–45.

Byrne KA, Kiely G & Leahy P (2005) CO_2 fluxes in adjacent new and permanent temperate grasslands. *Agricultural and Forest Meteorology*, **135**, 82–92.

Carroll P & Crill P (1997) Carbon balance of a temperate poor fen. *Global Biogeochemical Cycles*, **11**, 349–356.

Carter TR, La Rovere EL, Jones RN, Leemans R, Mearns LO, Nakicenovic N, Pittock AB, Semenov SM, Skea J, Gromov S, Jordan AJ, Khan SR, Koukhta A, Lorenzoni I, Posch M, Tsyban AV, Velichko A & Zeng N (2001) Developing and Applying Scenarios. In: McCarthy JJ, Canziani OF, Leary NA, Dokken DJ & White KS (eds) *Climate Change 2001: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Third Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, UK and New York, USA, **Chapter 3**, pp. 145–190.

Chanton JP & Whiting GJ (1995) Trace gas exchange in freshwater and coastal marine environments: ebullition and transport by plants. In: Matson PA & Harriss RC. (eds) *Biogenic trace gases: measuring emissions from soil and water*. Blackwell Science Ltd., UK, **Chapter 4**, pp. 98–125.

Chanton JP, Whiting GJ, Happel JD & Gerard D (1993) Contrasting rates and diurnal patterns of methane emission from emergent aquatic macrophytes. *Aquatic Botany*, **46**, 111–128.

Chappellaz J, Barnola JM, Raynaud D, Korotkevich YS & Lorius C (1990) Ice-core record of atmospheric methane over the past 160 000 years. *Nature*, **345**, 127–131.

Chen H, Wu N, Wang YF, Zhu D, Zhu QA, Yang G, Gao YH, Fang XQ, Wang X & Peng CH (2013) Inter-Annual Variations of Methane Emission from an Open Fen on the Qinghai-Tibetan Plateau: A Three-Year Study. *PLoS ONE*, **8**, Article e53878, 1–8.

Cicek N, Lambert S, Venema HD, Snelgrove KR, Bibeau EL & Grosshans R (2006) Nutrient removal and bio-energy production from Netley-Libau Marsh at Lake Winnipeg through annual biomass harvesting. *Biomass and Bioenergy*, **30**, 529–536.

Clymo RS, Turunen J & Tolonen (1998) Carbon accumulation in peatland. *Oikos*, **81**, 368–388.

Cooper MDA, Evans CD, Zielinski P, Levy PE, Gray A, Peacock M, Norris D, Fenner N & Freeman C (2014) Infilled Ditches are Hotspots of Landscape Methane Flux Following Peatland Re-wetting. *Ecosystems*, **17**, 1227–1241.

Couwenberg J (2007) Biomass energy crops on peatlands: on emissions and perversions. *IMCG Newsletter*, **2007/3**, 12–14.

Couwenberg J & Fritz C (2012) Towards developing IPCC methane ‘emission factors’ for peatlands (organic soils). *Mires and Peat*, **10**, Article 03, 1–17.

Couwenberg J & Joosten H (2001) Das Beispiel Deutschland. In: Succow M & Joosten H (eds) *Landschaftsökologische Moorkunde*, Schweizerbart, Stuttgart, **Chapter 8.2.2**, pp. 409–411.

Couwenberg J, Thiele A, Tanneberger F, Augusting J, Bärisch S, Dubovik D, Liashchynskaya N, Michaelis D, Minke M, Skuratovich A & Joosten H (2011) Assessing greenhouse gas emissions from peatlands using vegetation as a proxy. *Hydrobiologia*, **674**, 67–89.

Crawley MJ (2005) *Contrasts*. In: Crawley MJ (ed) *Statistics – An Introduction using R*. Wiley, Chichester, **Chapter 12**, pp. 209–226.

Crowley T (2000) Causes of climate change over the past 1000 years. *Science*, **289**, 270–277.

Crutzen PJ (1970) The influence of nitrogen oxides on the atmospheric ozone content. *Quarterly Journal of the Royal Meteorological Society,* **96**, 320–325.

Dann T & Ratzke U (2004) Böden. In: Katzung G (ed) *Geologie von Mecklenburg-Vorpommern*. Schweizerbart, Stuttgart, **Chapter 6.12**, pp. 489–508.

Davidson EA, Savage K, Verchot LV & Navarro R (2002) Minimizing artifacts and biases in chamber-based measurements of soil respiration. *Agricultural and Forest Meteorology*, **113**, 21–37.

Dias A, Hoorens B, van Logtestijn R, Vermaat J & Aerts R (2010) Plant species composition can be used as a proxy to predict methane emissions in peatland ecosystems after land-use changes. *Ecosystems*, **13**, 526–538.

Ding W, Cai Z & Tsuruta H (2004) Diel variation in methane emissions from the stands of *Carex lasiocarpa* and *Deyeuxia angustifolia* in a cool temperate freshwater marsh. *Atmospheric Environment*, **38**, 181–188.

Dise NB (1992) Winter fluxes of methane from Minnesota peatlands. *Biogeochemistry,* **17**, 71–83.

Dise NB, Gorham E & Verry ES (1993) Environmental factors controlling methane emissions from peatlands in northern Minnesota. *Journal of Geophysical Research*, **98**, 10583–10594.

Don A, Osborne B, Hastings A, Skiba U, Carter MS, Drewer J, Flessa H, Freibauer A, Hyvönen N, Jones MB, Lanigan GJ, Mander Ü, Monti A, Njakou Djomo S, 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. *GCB Bioenergy*, **4**, 372–391.

Donn WL & Shaw DM (1977) Model of climate evolution based on continental drift and polar wandering. *Geological Society of America Bulletin*, **88**, 390–396.

Drewer J, Lohila A, Aurela M, Laurila T, Minkinnen K, Penttilä T, Dinsmore KJ, McKenzie RM, Helfter C, Flechard C, Sutton MA & Skiba UM (2010) Comparison of greenhouse gas fluxes and nitrogen budgets from an ombotrophic bog in Scotland and a minerotrophic sedge fen in Finland. *European Journal of Soil Science*, **61**, 640–650.

Drösler M (2005) Trace gas exchange and climatic relevance of bog ecosystems. PhD Thesis, Southern Germany.

Drösler M, Adelmann W, Augustin J, Bergmann L, Beyer C, Chojnicki B, Förster C, Freibauer A, Giebels M, Görlitz S, Höper H, Kantelhardt J, Liebersbach H, Hahn-Schöfl M, Minke M, Petschow U, Pfadenhauer J, Schaller L, Schägner P, Sommer M, Thuille A & Wehrhan M (2013) Klimaschutz durch Moorschutz – Schlussbericht des Vorhabens "Klimaschutz – Moornutzungsstrategien" 2006–2010, Freising, Germany, 191 pages.

References

Drösler M, Verchot L, Freibauer A, Pan G, Evans CD, Bourbonniere RA, Alm JP, Page S, Agus F, Hergoualc'h K, Couwenberg J, Jauhiainen J, Sabiham S & Wang C (2014) Drained inland organic soils. In: Hiraishi T, Krug T, Tanabe K, Srivastava N, Jamsranjav B, Fukuda M & Troxler T (eds) *2013 Supplement to the 2006 Guidelines for National Greenhouse Gas Inventories: Wetlands*. Geneva, Switzerland, **Chapter 2**, pp. 2.1–2.79.

Duan X, Wang X & Ouyang Z (2006) Plant-mediated CH_4 emission from a *Phragmites*-dominated wetland in an arid region, China. *Journal of Freshwater Ecology*, **21**, 139–145.

Ellenberg H & Leuschner C (2010) Zeigerwerte der Pflanzen Mitteleuropas. In: Ellenberg H & Leuschner C (eds) *Vegetation Mitteleuropas mit den Alpen*, **Chapter 27**, 109 pages. Available via http://www.utb-mehr-wissen.de

Elsgaard L, Görres C-M, Hoffmann CC, Blicher-Mathiesen G, Schelde K & Petersen SO (2012) Net ecosystem exchange of CO_2 and carbon balance for eight temperate organic soils under agricultural management. *Agriculture, Ecosystems and Environment*, **162**, 52–67.

Erwin KL (2009) Wetlands and global climate change: the role of wetland restoration in a changing world. *Wetlands Ecology and Management*, **17**, 71–84.

Federal Environment Agency (2007) Regional climate changes: Recent findings – WETTREG: A statistical regionalization model, 27 pages. Available via http://www.umweltdaten.de/publikationen/fpdf-l/3543.pdf

Felten D, Fröba N, Fries J & Emmerling C (2013) Energy balances and greenhouse gas-mitigation potentials of bioenergy cropping systems (Miscanthus, rapeseed, and maize) based on farming conditions in Western Germany. *Renewable Energy*, **55**, 160–174.

Fiedler SR, Buczko U, Jurasinski G & Glatzel S (2015) Soil respiration after tillage under different fertilizer treatments – implications for modeling and balancing. *Soil & Tillage Research*, **150**, 30–42.

Fiedler S & Sommer M (2000) Methane emissions, groundwater levels and redox potentials of common wetland soils in a temperate-humid climate. *Global Biogeochemical Cycles*, **14**, 1081–1093.

Field CB, Barros VR, Dokken DJ, Mach KJ, Mastrandrea MD, Bilir TE, Chatterjee M, Ebi KL, Estrada YO, Genova RC, Girma B, Kissel ES, Levy AN, MacCracken S, Mastrandrea PR & White LL (2014) *Climate Change 2014: Impacts, Adaptation, and Vulnerability – Summary for Policymakers*. Cambridge University Press, Cambridge, UK and New York, NY, USA, 32 pages.

Flessa H, Dörsch P & Beese F (1995) Seasonal variation of N_2O and CH_4 fluxes in differently managed arable soils in southern Germany. *Journal of Geophysical Research*, **100**, D11, 23115–23124.

Flessa H, Wild U, Klemisch M & Pfadenhauer J (1998) Nitrous oxide and methane fluxes from organic soils under agriculture. *European Journal of Soil Science*, **49**, 327–335.

Folorunso OA & Rolston DE (1984) Spatial variability of Field-Measured Denitrification Gas Fluxes. *Soil Science Society of America Journal*, **48**, 1214–1219.

Forbrich I, Kutzbach L, Wille C, Becker T, Wu J & Wilmking M (2011) Cross-evaluation of measurements of peatland methane emissions on microform and ecosystem scales using high-resolution landcover classification and source weight modelling. *Agricultural and Forest Meteorology*, **151**, 864–874.

Forster P, Ramaswamy V, Artaxo P, Berntsen T, Betts R, Fahey DH, Haywood J, Lean J, Lowe DC, Myhre G, Nganga J, Prinn R, Raga G, Schulz M & Van Dorland R (2007) Changes in Atmospheric Constituents and in Radiative Forcing. In: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M & Miller HL (eds) *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, UK and New York, USA, **Chapter 2**, pp. 130–234.

Franzén LG (1994) Are Wetlands the Key to the Ice-age Cycle Enigma? *Ambio*, **23**, 300–308.

Franzén LG, Lindberg F, Viklander V & Walther A (2012) The potential peatland extent and carbon sink in Sweden, as related to the Peatland / Ice Age Hyopthesis. *Mires and Peat*, **10**, Article 08, 1–19.

Freeman C, Lock M & Reynolds B (1993) Fluxes of CO_2, CH_4 and N_2O from a Welsh peatland following simulation of water table draw-down: Potential feedback to climatic change. *Biogeochemistry*, **19**. 51–60.

Fritz C, Pancotto VA, Elzenga JTM, Visser EJW, Grootjans AP, Pol A, Iturraspe R, Roelofs JGM & Smolders AJP (2011) Zero methane emission bogs: extreme rhizosphere oxygenation by cushion plants in Patagonia. *New Phytologist*, **190**, 398–408.

Frolking S, Roulet NT & Fuglestvedt J (2006) How northern peatlands influence the Earth's radiative budget: Sustained methane emission versus sustained carbon sequestration. *Journal of Geophysical Research*, **111**, Article G01008.

Gelbrecht J, Koppisch D & Lengsfeld H (2001) Nordostdeutsche Niedermoore als Akkumulationsräume. In: Succow M & Joosten H (eds) *Landschaftsökologische Moorkunde*. E. Schweizerbart, Stuttgart, pp. 38–40.

Gilmanov TG, Baker JM, Bernacchi CJ, Billesbach DP, Burba GG, Castro S, Chen J, Eugster W, Fischer M L, Gamon JA, Gebremedhin MT, Glenn AJ, Griffis TJ, Hatfield JL, Heuer MW, Howard DM, Leclerc MY, Loescher HW, Marloie O, Meyers TP, Olioso A, Philipps RL, Prueger JH, Skinner RH, Suyker A E, Tenuta M & Wylie BK (2014) Productivity and Carbon Dioxide Exchange of Leguminous Crops: Estimates from Flux Tower Measurements. *Crop Ecology & Physiology*, **106**, 545–559.

Glatzel S, Forbrich I, Krüger C, Lemke S & Gerold G (2008) Small scale controls of greenhouse gas release under elevated N deposition rates in a restoring peat bog in NW Germany. *Biogeosciences*, **5**, 925–935.

Glatzel S, Koebsch F, Beetz S, Hahn J, Richter P & Jurasinski G (2011) Maßnahmen zur Minderung der Treibhausgasfreisetzung aus Mooren im Mittleren Mecklenburg. *Telma*, **Beiheft 4**, 85–105.

Gomez-Casanovas N, Anderson-Teixeira K, Zeri M, Bernacchi CJ & DeLucia EH (2013) Gap filling strategies and error in estimating annual soil respiration. *Global Change Biology*, **19**, 1941–1952.

Granéli W (1989) Influence of standing litter on shoot production in reed, Phragmites *australis* (Cav.) Trin. ex Steudel. *Aquatic Botany*, **35**, 99–109.

Günther A (2015) Greenhouse gas exchange and its relation to emergent vegetation in a temperate fen. PhD Thesis, Rostock, Germany.

Günther A, Huth V, Jurasinski G & Glatzel S (2014a) Scale-dependent temporal variation in determining the methane balance of a temperate fen. *Greenhouse Gas Measurement and Management*, **4**, 41–48.

References

Günther A, Huth V, Jurasinski G & Glatzel S (2015) The effect of biomass harvesting on greenhouse gas emissions from a rewetted temperate fen. *Global Change Biology Bioenergy*, **7**, 1092–1106.

Günther A, Jurasinski J, Huth V & Glatzel S (2014b) Opaque closed chambers underestimate methane fluxes of *Phragmites australis* (Cav.) Trin ex. Steud. *Environmental Monitoring and Assessment*, **186**, 2151–2158.

Güsewell S, Buttler A & Klötzli F (1998) Short-term and long-term effects of mowing on the vegetation of two calcareous fens. *Journal of Vegetation Science*, **9**, 861–872.

Hahn J, Köhler S, Glatzel S & Jurasinski G (2015) Methane Exchange in a Coastal Fen in the First Year after Flooding – A Systems Shift. *PLoS ONE*, **10**, 1–25.

Hahn-Schöfl M, Zak D, Minke M, Gelbrecht J, Augustin J & Freibauer A (2011) Organic sediment formed during inundation of a degraded fen grassland emits large fluxes of CH_4 and CO_2. *Biogeosciences*, **8**, 1539–1550.

Hansen J, Sato M & Ruedy R (2012) Perception of climate change. *Proceedings of the National Academy of Sciences of the United States of America*, **109**, E2415–E2423.

Hao QJ, Wang YS, Song CC & Huang Y (2006) Contribution of winter fluxes to the annual CH_4, CO_2 and N_2O emissions from freshwater marshes in the Sanjiang Plain. *Journal of Environmental Sciences*, **18**, 270–275.

Hargreaves KJ & Fowler D (1998) Quantifying the effects of water table and soil temperature on the emission of methane from peat wetland at the field scale. *Atmospheric Environment*, **32**, 3275–3282.

Harland WB (1964) Critical evidence for a great Infra-Cambrian glaciation. *Geologische Rundschau*, **54**, 45–61.

Hartigan JA & Wong MA (1979) A K-means clustering algorithm. *Applied Statistics*, **28**, 100–108.

Hartmann M (1999) Species dependent root decomposition in rewetted soils. *Plant and Soil*, **213**, 93–98.

Hayen H (1990) Moorarchäologie. In: Göttlich K (ed) *Moor- und Torfkunde*, **Chapter 3.5**, pp. 156–174.

Hegerl GC, Zwiers FW, Braconnot P, Gillet NP, Luo Y, Orsini JAM, Nicholls N, Penner JE & Stott PA (2007) Understanding and attributing climate change. In: Solomon S, Qin D, Manning M, Chen M, Marquis M, Averyt KB, Tignor M & Miller HL (eds) *Climate Change 2007. The physical science basis. Contribution of working group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, UK and New York, USA, **Chapter 9**, pp. 663–746

Hendriks DM, van Huissteden J, Dolman AJ & van der Molen MK (2007) The full greenhouse gas balance of an abandoned peat meadow. *Biogeosciences*, **4**, 411–424.

Hippke M (personal communication) Water balance of the Neuenkirchener Niederung. Biosphere Reserve Schaalsee.

Hoffmann M, Jurisch N, Alba JG, Borraz EA, Schmidt M, Rogasik H, Rieckh H, Verch G, Sommer M & Augustin J (submitted) Using automatic chambers to detect SOC stock dynamics with a high temporal and spatial resolution: A case study at the carboZALF experimental trial. *Agricultural and Forest Meteorology*.

Hoffmann M, Jurisch N, Borraz EA, Hagemann U, Drösler M, Sommer M & Augustin J (2015) Automated modeling of ecosystem CO_2 fluxes based on periodic closed chamber measurements: A standardized conceptual and practical approach. *Agricultural and Forest Meteorology*, **200**, 30–45.

Hollinger DY, Aber J, Dail B, Davidson EA, Goltz SM, Hughes H, Leclerc MY, Lee JT, Richardson AD, Rodrigues C, Scotts NA, Achuatavarier D & Walsh J (2004) Spatial and temporal variability in forest–atmosphere CO_2 exchange. *Global Change Biology*, **10**, 1689–1706.

Höper H, Augustin J, Cagampan JP, Drösler M, Lundin L, Moors E, Vasander H, Waddington JM & Wilson D (2008) Restoration of peatlands and greenhouse gas balances. In: Strack M (ed) *Peatlands and Climate Change*. International Peat Society, Saarijärvi, 184–212.

Huth V, Günther A, Jurasinski G & Glatzel S (2013) The effect of an exceptionally wet summer on methane effluxes from a 15-year re-wetted fen in north-east Germany. *Mires and Peat*, **13**, Article 02, 1–7.

Huth V, Jurasinski G & Glatzel S (2012) Winter emissions of carbon dioxide, methane and nitrous oxide from a minerotrophic fen under nature conservation management in north-east Germany. *Mires and Peat*, **10**, Article 04, 1–13.

Huth V, Vaidya S, Hoffmann M, Jurisch N, Günther A, Gundlach L, Hagemann U, Elsgaard L & Augustin J (submitted) Divergent NEE balances from manual chamber CO_2 fluxes by different measurement and modelling approaches. *Agricultural and Forest Meteorology*.

Hütsch BW (2001) Methane oxidation in non-flooded soils as affected by crop production — invited paper. *European Journal of Agronomy*, **14**, 237–260.

International Energy Agency (IEA) Statistics (2012) Renewables Information. Organization for Economic Co-operation and Development OECD, Paris, France.

Jansen E, Overpeck J, Briffa KR, Duplessy J-C, Joos F, Masson-Delmotte V, Olago D, Otto-Bliesner B, Peltier WR, Rahmstorf S, Rengaswamy R, Raynaud D, Rind D, Solomina O, Villalba R & Zhang D (2007) Palaeoclimate. In: Solomon S, Qin D, Manning M, Chen M, Marquis M, Averyt KB, Tignor M & Miller HL (eds) *Climate Change 2007. The physical science basis. Contribution of working group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, UK and New York, USA, **Chapter 6**, pp. 433–498.

Jentsch W, Piatkowski B, Schweigel M & Derno M (2009) Quantitative results for methane production of cattle in Germany. *Archiv Tierzucht*, **52**, 587–592.

Joosten H (2006) Moorschutz in Europa: Restauration und Klimarelevanz. In: *Moore in der Regionalentwicklung*. BUND Landesverband Niedersachsen, Wagenfeld/Ströhen, 35–43.

Joosten H & Couwenberg J (2001) Das Beispiel Europa. In: Succow M & Joosten H (eds) *Landschaftsökologische Moorkunde.* Schweizerbart, Stuttgart, **Chapter 8.2.1**, pp. 406–408.

Joosten H & Succow M (2001) Hydrogenetische Moortypen. In: Succow M & Joosten H (eds) *Landschaftsökologische Moorkunde.* Schweizerbart, Stuttgart, pp. 234–240.

Joosten H, Tapio-Biström M & Tol S (2012) Peatlands. Guidance for climate change mitigation through conservation, rehabilitation and sustainable use. Food and agriculture organization of the United nations (FAO) and Wetlands International, Rome, Italy.

References

Jouzel J, Lorius C, Petit JR, Genthon C, Barkov NI, Kotlyakov VM & Petrov VM (1987) Vostok ice core: a continuous isotope temperature record over the last climatic cycle (160,000 years). *Nature*, **329**, 403–408.

Jungkunst HF, Flessa H, Scherber C & Fiedler S (2008) Groundwater level controls CO_2, N_2O and CH_4 fluxes of three different hydromorphic soil types of a temperate forest ecosystem. *Soil Biology and Biochemistry*, **40**, 2047–2054.

Jurasinski G & Koebsch F (2011) flux: Flux rate calculation from dynamic closed chamber measurements. R package version 0.1-4, Rostock. Available via http://cran.r-project.org/web/packages/flux

Jurasinski G, Koebsch F & Hagemann U (2012) flux: Estimate gas flux rates using dynamic closed chamber data. R package version 0.2-1, Rostock. Available via http://CRAN.R-project.org/package=flux

Jurasinski G, Koebsch F, Hagemann U & Günther A (2013) flux: Flux rate calculation from dynamic closed chamber measurements. R package version 0.2-2. Available via http://CRAN.R-project.org/package=flux

Juutinen S, Alm J, Larmola T, Saarnio S, Martikainen PJ & Silvola J (2004) Stand-specific diurnal dynamics of CH_4 fluxes in boreal lakes: Patterns and controls. *Journal of Geophysical Research*, **109**, Article D19313, 1–11.

Juutinen S, Bubier J, Larmola T, Humphreys E, Arnkil S, Roy C & Moore T (2016) Nutrient load can lead to enhanced CH_4 fluxes through the changes in vegetation, peat surface elevation and water table depth in ombrotrophic bog. *European Geosciences Union General Assembly, Vienna, Austria,17-22 April*. *Geophysical Research Abstracts*, **18**, EGU2016-12050.

Kandel TP, Elsgaard L & Lærke PE (2013) 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.

Kirschke S, Bousquet P, Ciais P, Saunois M, Canadell JG, Dlugokencky EJ, Bergamaschi P, Bergmann D, Blake DR, Bruhwiler L, Cameron-Smith P, Castaldi S, Chevallier F, Feng L, Fraser A, Heimann M, Hodson EL, Houweling S, Josse B, Fraser PJ, Krummel PB, Lamarque J-F, Langenfelds RL, Le Quéré C, Naik V, O'Doherty S, Palmer PI, Pison I, Plummer D, Poulter B, Prinn RG et al. (2013) Three decades of global methane sources and sinks. *Nature Geoscience*, **6**, 813–823.

Klämt A & Schwanitz D (2001) Hydrologischer Atlas von Deutschland. Karte der mittleren jährlichen klimatischen Wasserbilanz. Bundesministerium für Umwelt, Naturschutz und Reaktorsicherheit (BMU), **Chapter 2.14**, p. 116.

Klinger LF (1991) Peatland formation and Ice Ages: A possible Gaian mechanism related to community succession. In: Schneider SH & Boston PJ (eds) *Scientists on Gaia*, Cambridge, Massachusetts, London, England, pp. 247–255.

Köbbing JF, Thevs N & Zerbe S (2013) The utilisation of reed (*Phragmites australis*): a review. *Mires and Peat*, **13**, Article 01, 1–14.

Koch S, Jurasinski G, Koebsch F, Koch M & Glatzel S (2014) Spatial Variability of Annual Estimates of Methane Emissions in a *Phragmites Australis* (Cav.) Trin. Ex Steud. Dominated Restored Coastal Brackish Fen. *Wetlands*, **34**, 593–602.

References

Koebsch F, Glatzel S, Hofmann J, Forbrich I & Jurasinski G (2013a) CO_2 exchange of a temperate fen during the conversion from moderately rewetting to flooding. *Journal of Geophysical Research: Biogeosciences*, **118**, 940–950.

Koebsch F, Glatzel S & Jurasinski G (2013b) Vegetation controls methane emissions in a coastal backish fen. *Wetlands Ecology and Management*, **21**, 323–337.

Koebsch F, Jurasinski G, Koch M, Hofmann J & Glatzel S (2015) Controls for multi-scale temporal variation in ecosystem methane exchange during the growing season of a permanently inundated fen. *Agricultural and Forest Meteorology*, **204**, 94–105.

Koelbener A, Ström L, Edwards PJ & Olde Venterink H (2010) Plant species from mesotrophic wetlands cause relatively high methane emissions from peat soil. *Plant and Soil*, **326**, 147–158.

Köhler W, Schachtel G & Voleske P (2007) Biostatistik. Eine Einführung für Biologen und Agrarwissenschaftler. Springer, Berlin, Heidelberg, New York, 288 pages.

Körschens M, Albert E, Armbruster M, Barkusky D, Baumecker M, Behle-Schalk L, Bischoff R, Čergan Z, Ellmer F, Herbst F, Hoffmann S, Hofmann B, Kismanyoky T, Kubat J, Kunzova E, Lopez-Fando C, Merbach I, Merbach W, Pardor MT, Rogasik J, Rühlmann J, Spiegel H, Schulz E, Tajnsek A, Toth Z, Wegener H & Zorn W (2012) Effect of mineral and organic fertilization on crop yield, nitrogen uptake, carbon and nitrogen balances, as well as soil organic carbon content and dynamics: results from 20 European long-term field experiments of the twenty-first century. *Archives of Agronomy and Soil Science*, **59**, 1017–1040.

Koskinen M, Minkkinen K, Ojanen P, Kämäräinen M, Laurila T & Lohila A (2014) Measurements of CO_2 exchange with an automated chamber system throughout the year: challenges in measuring night-time respiration on porous peat soil. *Biogeosciences*, **11**, 347–363.

Kroon PS, Schrier-Uijl AP, Hensen A, Veenendaal EM & Jonker HJJ (2010) Annual balances of CH_4 and N_2O from a managed fen meadow using eddy covariance flux measurements. *European Journal of Soil Science*, **61**, 773–784.

Lai DYF, Roulet NT, Humphreys ER, Moore TR & Dalva M (2012) The effect of atmospheric turbulence and chamber deployment period on autochamber CO_2 and CH_4 flux measurements in an ombrotrophic peatland. *Biogeosciences*, **9**, 3305–3322.

Langensiepen M, Kupisch M, van Wijk MT & Ewert F (2012) Analyzing transient closed chamber effects on canopy gas exchange for optimizing flux calculation timing. *Agricultural and Forest Meteorology*, **164**, 61–70.

Lappalainen E (ed) (1996) *Global Peat Resources*. International Peat Society, Jyväskylä, 368 pages.

Leiber-Sauheitl K, Fuß R, Voigt C & Freibauer A (2014) High CO_2 fluxes from grassland on histic Gleysol along soil carbon and drainage gradients. *Biogeosciences*, **11**, 749–761.

Leifeld J, Bader C, Borraz E, Hoffmann M, Giebels M, Sommer M & Augustin J (2014) Are C-loss rates constant over time? The additive value of soil profile based and flux budget approach. *Biogeosciences Discussions*, **11**, 12341–12373.

References

Lenschow U (2001) Das Beispiel Mecklenburg-Vorpommern. In: Succow M & Joosten H (eds) *Landschaftsökologische Moorkunde*, Schweizerbart, Stuttgart, **Chapter 8.2.3**, pp. 411–415.

Le Quéré C, Moriarty R, Andrew RM, Peters GP, Ciais P, Friedlingstein P, Jones SD, Sitch S, Tans P, Arneth A, Boden TA, Bopp L, Bozec Y, Canadell JG, Chini LP, Chevallier F, Cosca CE, Harris I, Hoppema M, Houghton RA, House JI, Jain AK, Johannessen T, Kato E, Keeling RF, Kitidis V et al. (2015) Global carbon budget 2014. *Earth System Science Data*, **7**, 47–85.

Livingston GP & Hutchinson GL (1995) Enclosure-based measurement of trace gas exchange: applications and sources of error. In: Matson PA & Harriss RC (eds) *Biogenic trace gases – Measuring emissions from soil and water*. Blackwell Science Ltd., UK, **Chapter 2**, pp. 14–51.

Lloyd J & Taylor JA (1994) On the temperature dependence of soil respiration. *Functional Ecology*, **8**, 315–323.

Luo Y, Keenan TF & Smith M (2015) Predictability of the terrestrial carbon cycle. *Global Change Biology*, **21**, 1737–1751.

Maddison M, Mauring T, Remm K, Lesta M & Mander Ü (2009) Dynamics of *Typha latifolia* L. populations in treatment wetlands in Estonia. *Ecological Engineering*, **35**, 258–264.

Maljanen M, Liikanen A, Silvola J & Martikainen PJ (2003) Nitrous oxide emissions from boreal organic soil under different land-use. *Soil Biology & Biochemistry*, **35**, 689–700.

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.

Maljanen M, Virkajärvi P, Hytönen J, Öquist M, Sparrman T & Martikainen PJ (2009) Nitrous oxide production in boreal soils with variable organic matter content at low temperature – snow manipulation experiment. *Biogeosciences*, **6**, 2461–2473.

Marcolla B, Cescatti A, Manca G, Zorer R, Cavagna M, Fiora A, Gianelle D, Rodeghiero M, Sottocornola M & Zampedri R (2011) Climatic controls and ecosystem responses drive the inter-annual variability of the net ecosystem exchange of an alpine meadow. *Agricultural and Forest Meteorology*, **151**, 1233–1243.

Martikainen PJ, Nykänen H, Alm J & Silvola J (1995) Change in fluxes of carbon dioxide, methane and nitrous oxide due to forest drainage of mire sites of different trophy. *Plant and Soil*, **168-169**, 571–577.

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

Meinardus W (1944) Zum Kanon der Erdbestrahlung. *Geologische Rundschau*, **34**, 748–762.

Melloh RA & Crill PM (1996) Winter methane dynamics in a temperate peatland. *Global Biogeochemical Cycles*, **10**, 247–254.

Michaelis L & Menten ML (1913) Die Kinetik der Invertinwirkung. *Biochemische Zeitschrift*, **49**, 333–369.

Milankovitch M (1930) Mathematische Klimalehre und astronomische Theorie der Klimaschwankungen. In: Köppen WP, Geiger R (eds) *Handbuch der Klimatologie*, **Band I**, Teil A, Berlin.

References

Minke M, Augustin J, Hagemann U & Joosten H (2014) Similar methane fluxes measured by transparent and opaque chambes point at belowground connectivity of *Phragmites australis* beyond the chamber footprint. *Aquatic Botany*, **113**, 63–71.

Minkkinen K, Laine J & Penttilä T (2008) Greenhouse gas emissions from forestry drained peatlands and contributing environmental factors. In: Sarkkola S (ed) *Greenhouse impacts of the use of peat and peatlands in Finland.* Research programme final report, Ministry of Agriculture and Forestry, Helsinki, Finland, pp 22–26.

Moore TR, De Young A, Bubier JL, Humphreys ER, Lafleur PM & Roulet NT (2011) A Multi-Year Record of Methane Flux at the Mer Bleue Bog, Southern Canada. *Ecosystems*, **14**, 646–657.

Moore TR, Roulet NT & Waddington JM (1998) Uncertainty in predicting the effect of climatic change on the carbon cycling of Canadian peatlands. *Climatic Change*, **40**, 229–245.

Morin T, Bohrer G, Naor-Azrieli L, Mesi S, Kenny W, Mitsch W & Schäfer K (2014) The seasonal and diurnal dynamics of methane flux at a created urban wetland. *Ecological Engineering*, **72**, 74–83.

Müller-Westermeier G (1995) Numerisches Verfahren zur Erstellung klimatologischer Karten (A numeric method creating climatologic maps). *Berichte des Deutschen Wetterdienstes (Reports of the German Weather Service)*, **193**, 1–17.

Mundel G (1976) Untersuchungen zur Torfmineralisation in Niedermooren (Studies on peat mineralization of minerotrophic fens). *Archiv Acker-, Pflanzenbau und Bodenkunde*, **20**, 669–679.

Myhre G, Shindell D, Breon F-M, Collins W, Fuglestvedt J, Huang J, Koch D, Lamarque J-F, Lee D, Mendoza B, Nakajima T, Robock A, Stephens G, Takemura T & Zhang H (2013) Anthropogenic and natural radiative forcing. In: Stocker TF, Qin D, Plattner G-K, Tignor MMB, Allen SK, Boschung J, Nauels A, Xia Y, Bex V, Midgley PM (eds) *Climate change 2013. The physical science basis. Contribution of working group I to the fifth assessment report of the Intergovernmental Panel on Climate Change*. Cambridge University Press, Cambridge, UK and New York, USA, **Chapter 8**, pp. 659–740.

Nykänen H, Alm J, Lång K, Silvola J & Martikainen PJ (1995) Emissions of CH_4, N_2O and CO_2 from a virgin fen and a fen drained for grassland in Finland. *Journal of Biogeography*, **22**, 351–357.

Okruszko H (1996) Agricultural use of peatlands. In: Lappalainen E (ed) *Global peat resources*. International Peat Society, Jyskä, Finland, pp. 303–309.

Ostendorp W (1999) Management impacts on stand structure of lakeshore *Phragmites* reeds. *International Review of Hydrobiology*, **84**, 33–47.

Özdemir ED, Härdtlein M & Eltrop L (2009) Land substitution effects of biofuel side products and implications on the land area requirement for EU 2020 biofuel targets. *Energy Policy*, **37**, 2986–2996.

Päivänen J (1996) Forestry use of peatlands. In: Lappalainen E (ed) *Global peat resources*. International Peat Society, Jyskä, Finland, pp. 311–314.

Panikov NS & Dedysh SN (2000) Cold season CH_4 and CO_2 emission from boreal peat bogs (West Siberia): Winter fluxes and thaw activation dynamics. *Global Biogeochemical Cycles*, **14**, 1071–1080.

Peichl M, Leahy P & Kiely G (2011) Six-year Stable Annual Uptake of Carbon Dioxide in Intensively Managed Humid Temperate Grassland. *Ecosystems*, **14**, 112–126.

References

Phillips CL, Nickerson N, Risk D & Bond BJ (2011) Interpreting diel hysteresis between soil respiration and temperature. *Global Change Biology*, **17**, 515–527.

Pihlatie MK, Kiese R, Brüggemann N, Butterbach-Bahl K, Kieloaho A-J, Laurila T, Lohila A, Mammarella I, Minkkinen K, Penttilä T, Schönborn J & Vesala T (2010) Greenhouse gas fluxes in a drained peatland forest during spring frost-thaw event. *Biogeosciences*, **7**, 1715–1727.

Pumpanen J, Kolari P, Ilvesniemi H et al. (2004) Comparison of different chamber techniques for measuring soil CO_2 efflux. *Agricultural and Forest Meteorology*, **123**, 159–176.

R Development Core Team (2011) R: A language and environment for statistical computing. *R Foundation for Statistical Computing*. Vienna, Austria. Available via http://www.R-project.org/

R Development Core Team (2012) R: A language and environment for statistical computing. *R Foundation for Statistical Computing*. Vienna, Austria. Available via http://www.R-project.org/

R Core Team (2013) R: A language and environment for statistical computing. *R Foundation for Statistical Computing*, Vienna, Austria. Available via http://www.R-project.org/

Ravishankara AR, Daniel SJ & Portmann RW (2009) Nitrous Oxide (N_2O): The Dominant Ozone-Depleting Substance Emitted in the 21st Century. *Science*, **326**, 123–125.

Regina K, Nykänen H, Silvola J & Martikainen PJ (1996) Fluxes of nitrous oxide from boreal peatlands as affected by peatland type, water table and nitrification capacity. *Biogeochemistry*, **35**, 401–418.

Regina K, Syväsalo E, Hannukkala A & Esala M (2004) Fluxes of N_2O from farmed peat soils in Finland. *European Journal of Soil Science*, **55**, 591–599.

Reichelt G & Wilmanns O (1973) Vegetationsgeographie. Westermann, Braunschweig, 210 pages.

Reichstein M, Falge E, Baldocchi D, Papale D, Aubinet M, Berbiger P, Bernhofer C, Buchmann N, Gilmanov T, Granier A, Grünwald T, Havránková K, Ilvesniemi H, Janous D, Knohl A, Laurila T, Lohila A, Loustau D, Metteucci G, Meyers T, Miglietta F, Ourcival J-M, Pumpanen J, Rambal S, Rotenberg E, Sanz M, Tenhunen J, Seufert G, Vaccari F, Vesala T, Yakir D & Valentini R (2005) On the separation of net ecosystem exchange into assimilation and ecosystem respiration: review and improved algorithm. *Global Change Biology*, **11**, 1424–1439.

Renou-Wilson F, Barry C, Müller C & Wilson D (2014) The impacts of drainage, nutrient status and management practice on the full carbon balance of grasslands on organic soils in a maritime temperate zone. *Biogeosciences*, **11**, 4361–4379.

Reuter G (1962) Tendenzen d. Bodenentwicklung im Küstenbezirk Mecklenburgs. *Wissenschaftliche Abhandlungen Nr. 49*, Deutsche Akademie d Landwirtschaftswissenschaften zu Berlin.

Richert M, Dietrich O, Koppisch D & Roth S (2000) The influence of rewetting on vegetation development and decomposition in a degraded fen. *Restoration Ecology*, **8**, 186–195.

Rind D, Peteet D & Kukla G (1989) Can Milankovitch orbital variations initiate the growth of ice sheets in a General Circulation Model? *Journal of Geophysical Research*, **94**, 12851–12871.

Rochette P, Desjardins RL & Pattey E (1991) Spatial and temporal variability of soil respiration in agricultural fields. *Canadian Journal of Soil Science*, **71**, 189–196.

Röder N & Grützmacher F (2012) Emissionen aus landwirtschaftlich genutzten Mooren. Vermeidungskosten und Anpassungsbedarf. *Natur und Landschaft*, **87**, 56–61.

Rodewald-Rodescu L (1974) Das Schilfrohr (Common reed). *Die Binnengewässer*, **27**, Schweizerbart, Stuttgart, 302 pages.

Roth S, Koppisch D, Wichtmann W & Zeitz J (2001) „Moorschonende Grünlandnutzung“ – Erste Erfahrungen auf nordostdeutschen Niedermooren. In: Succow M & Joosten H (eds) *Landschaftsökologische Moorkunde*, Schweizerbart, Stuttgart, **Chapter 9.2.1**, pp. 472–480.

Roulet NT, Lafleurs 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.

Röver M, Heinemeyer O & Kaiser E-A (1998) Microbial induced nitrous oxide emissions from an arable soil during winter. *Soil Biology & Biochemistry*, **30**, 1859–1865.

Ruser R, Flessa H, Schilling R, Beese F & Munch JC (2001) Effect of crop-specific field management and N fertilization on N_2O emissions from a fine-loamy soil. *Nutrient Cycling in Agroecosystems*, **59**, 177–191.

Saarnio S, Winiwarter W & Leitão J (2009) Methane release from wetlands and watercourses in Europe. *Atmospheric Environment*, **43**, 1421–1429.

Sachs T, Giebels M, Boike J & Kutzbach L (2010) Environmental controls on CH_4 emission from polygonal tundra on the microsite scale in the Lena river delta, Siberia. *Global Change Biology*, **16**, 3096–3110.

Sauer T, Reckling M & Hufnagel J (2015) Einfluss von Bodenbearbeitung und Beregnung auf pflanzenbauliche Parameter einer diversen Energiepflanzen-Fruchtfolge im Vergleich zu Mais-Selbstfolge. In: Kage H, Sieling K & Francke-Weltmann L (eds) *Multifunktionale Agrarlandschaften – Pflanzenbaulicher Anspruch, Biodiversität, Ökosystemdienstleistungen: Kurzfassungen der Vorträge und Poster*; 58. Jahrestagung der Gesellschaft für Pflanzenbauwissenschaften, 22–24th September, Braunschweig, Göttingen, Germany, pp. 21–22.

Schindler U, Müller L, Dannowski R, Barkusky D & Francis G (2010) Long-Term Measurements to Quantify the Impact of Arable Management Practices on Deep Seepage and Nitrate Leaching. In: Müller F, Baessler C, Schubert H & Klotz S (eds) *Long-Term Ecological Research*, **Chapter 17**, pp. 243–252.

Schopp-Guth A (1999) Renaturierung von Moorlandschaften. Naturschutzfachliche Anforderungen aus bundesweiter Sicht. Bundesamt für Naturschutz, Bonn-Bad Godesberg, Germany, 219 pages.

Schothorst CJ (1977) Subsidence of low moor peat soils in the Western Netherlands. *Geoderma*, **17**, 265–291.

Schrier-Uijl AP, Kroon PS, Hensen A, Leffelaar PA, Berendse F & Veenendaal EM (2010a) Comparison of chamber and eddy covariance-based CO_2 and CH_4 emission estimates in a heterogeneous grass ecosystem on peat. *Agricultural and Forest Meteorology*, **150**, 825–831.

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

Schulz K, Timmermann T, Steffenhagen P, Zerbe S & Succow M (2011) The effect of flooding on carbon and nutrient standing stocks of helophyte biomass in rewetted fens. *Hydrobiologia*, **674**, 25–40.

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.

Smemo KA & Yavitt JB (2006) A multi-year perspective on methane cycling in a shallow peat fen in central New York state, USA. *Wetlands*, **26**, 20–29.

Succow M & Jeschke (1986) Moore in der Landschaft. Deutsch, Thun & Frankfurt/Main, Switzerland & Germany, 268 pages.

Succow M (2001) Kurzer Abriß zur Nutzungsgeschichte mitteleuropäischer Moore. In: Succow M & Joosten H (eds) *Landschaftsökologische Moorkunde*, Schweizerbart, Stuttgart, **Chapter 8.1**, pp. 404–406.

Tanovitskaya N (2011) Land use of drained peatlands. In: Tanneberger F & Wichtmann W (eds) *Carbon credits from peatland rewetting. Climate - biodiversity - land use,* Schweizerbart, Stuttgart, pp. 7–9.

Teepe R, Brumme R & Beese F (2000) Nitrous oxide emissions from frozen soils under agricultural, fallow and forest land. *Soil Biology & Biochemistry*, **32**, 1807–1810.

Timmermann T, Steffenhagen P, Schulz K, Zerbe S & Frick A (2011) Vegetationsentwicklung von wiedervernässten Flusstalmooren Mecklenburg-Vorpommerns. *Telma*, **Beiheft 4**, 159–182.

United Nations Framework Convention on Climate Change (2012) Report of the Conference of the Parties serving as the meeting of the Parties to the Kyoto Protocol on its seventh session, held in Durban from 28 November to 11 December 2011. Part Two: Action taken by the Conference of the Parties serving as the meeting of the Parties to the Kyoto Protocol at its seventh session. Available via http://unfccc.int/resource/docs/2011/cmp7/eng/10a01.pdf.

United Nations Framework Convention on Climate Change (2014) GHG emission profiles for Annex I Parties and major groups. Available via www.unfccc.int/ghg_data/ghg_data_unfccc/ghg_profiles/items/4625.php. Accessed 20 Feb 2014.

van Beek CL, Pleijter M, Jacobs CMJ, Velthof GL, Groeningen JW & Kuikman PJ (2010) Emissions of N_2O from fertilized and grazed grassland on organic soil in relation to groundwater level. *Nutrient Cycling in Agroecosystems*, **86**, 331–340.

van der Nat FW & Middelburg JJ (2000) Methane emission from tidal freshwater marshes. *Biogeochemistry*, **49**, 103–121.

van der Nat F W, Middelburg JJ, van Meteren D & Wielemakers A. (1998) Diel methane emission patterns from *Scirpus lacustris* and *Phragmites australis*. *Biogeochemistry* **41**, 1–22.

Velthof GL & Oenema O (1995) Nitrous oxide fluxes from grassland in the Netherlands: II. Effects of soil type, nitrogen fertilizer application and grazing. *European Journal of Soil Science*, **46**, 541–549.

von Bülow K (1925) Moorkunde. Walter de Gruyter & Co., Berlin & Leipzig, Germany, 142 pages.

Whiting GJ & Chanton JP (1996) Control of the diurnal pattern of methane emission from emergent aquatic macrophytes by gas transport mechanisms. *Aquatic Botany*, **54**, 237–253.

Whiting GJ, Bartlett DS, Fan S, Bakwin PS & Wofsy SC (1992) Biosphere/Atmosphere CO_2 Exchange in Tundra Ecosystems: Community Characteristics and Relationships With Multispectral Surface Reflectance. *Journal of Geophysical Research*, **97**, 16,671–16,680.

Wichtmann W (2003) Verwertung von Biomasse von Niederungsstandorten. In: Timmermann T, Wichtmann W, Succow M & Billwitz K (eds) *Alternative Nutzungsformen für Moorstandorte in Mecklenburg-Vorpommern*, pp. 43–54.

Wichtmann W, Schröder C & Joosten H (2014) Paludiculture – cultivation of wet peatlands. Climate protection – biodiversity – regional economic benefits. Schweizerbart, Stuttgart, Germany, 272 pages.

Wichtmann W & Tanneberger F (2011) Land use options for rewetted peatlands. In: Tanneberger F & Wichtmann W (eds) *Carbon credits from peatland rewetting. Climate - biodiversity - land use*, Schweizerbart, Stuttgart, pp. 107–132.

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.

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.

Zauft M, Fell H, Glaßer F, Rosskopf N & Zeitz J (2010) Carbon storage in the peatlands of Mecklenburg-Western Pomerania, north-east Germany. *Mires and Peat*, **6**, Article 04, 1–12.

Appendices

A.1 CURRICULUM VITAE

Publication record

Peer-reviewed articles

Günther A, **Huth V**, Jurasinski G & Glatzel S (2014a) Scale-dependent temporal variation in determining the methane balance of a temperate fen. *Greenhouse Gas Measurement and Management*, **4**, 41–48.

Günther A, **Huth V**, Jurasinski G & Glatzel S (2015) The effect of biomass harvesting on greenhouse gas emissions from a rewetted temperate fen. *Global Change Biology Bioenergy*, **7**, 1092–1106.

Günther A, Jurasinski J, **Huth V** & Glatzel S (2014b) Opaque closed chambers underestimate methane fluxes of *Phragmites australis* (Cav.) Trin ex. Steud. *Environmental Monitoring and Assessment*, **186**, 2151–2158.

Huth V, Günther A, Jurasinski G & Glatzel S (2013) The effect of an exceptionally wet summer on methane effluxes from a 15-year re-wetted fen in north-east Germany. *Mires and Peat*, **13**, Article 02, 1–7.

Huth V, Jurasinski G & Glatzel S (2012) Winter emissions of carbon dioxide, methane and nitrous oxide from a minerotrophic fen under nature conservation management in north-east Germany. *Mires and Peat*, **10**, Article 04, 1–13.

Huth V, Vaidya S, Hoffmann M, Jurisch N, Günther A, Gundlach L, Hagemann U, Elsgaard L & Augustin J (submitted) Divergent NEE balances from manual chamber CO_2 fluxes by different measurement and modelling approaches. *Agricultural and Forest Meteorology*.

Book chapters

Jurasinski G, Günther A, **Huth V**, Couwenberg J & Glatzel S (2016) Greenhouse gas emissions. In: Wichtmann W, Schröder C & Joosten H (eds) *Paludiculture – productive use of wet peatlands. Climate protection – biodiversity – regional economic benefits*. Schweizerbart, Stuttgart, **Chapter 5.1**, pp. 79–93.

Conference abstracts

Andres M, **Huth V** & Augustin J (2014) Influence of soil tillage system on CO2, N2O and CH4 exchange, climate impact, and C/N budgets of winter oilseed rape in NE Germany. *8th International Symposium on Ecosystem Behaviour – BIOGEOMON 2014, Bayreuth, Germany, 13-17 July, Book of abstracts*, Poster presentation P 3.1, 118. http://www.bayceer.uni-bayreuth.de/biogeomon2014/

de Mol F, Babenschneider S, **Huth V**, von Redwitz C (2010) Observing invertebrate seed predators by infrared photography. *Proc. of the 15th Symposium of the European Weed Research Society, Kaposvár, Hungary, 12-15 July*, 183.

Günther A, **Huth V**, Jurasinski G, Albrecht K & Glatzel S (2015) Paludiculture as a chance for peatland and climate: the greenhouse gas balance of biomass production on two rewetted peatlands does not differ from the natural state. *European Geosciences Union General Assembly, Vienna, Austria,12-17 April*. *Geophysical Research Abstracts*, **17**, EGU2015-10972.

Günther A, **Huth V**, Jurasinski G & Glatzel S (2014) The effect of biomass harvesting on greenhouse gas emissions from a rewetted temperate fen. *8th International Symposium on Ecosystem Behaviour – BIOGEOMON 2014, Bayreuth, Germany, 13-17 July, Book of abstracts*, Oral presentation O 6.2, 197. http://www.bayceer.uni-bayreuth.de/biogeomon2014/

Günther A, **Huth V**, Jurasinski G & Glatzel S (2013a) Impact of biomass harvesting on greenhouse gas balances of a rewetted fen. *International Conference on the Utilization of Emergent Wetland Plants – Reed as a Renewable Resource, Greifswald, Germany, 14-16 February. Book of Abstracts*, 79–80 . http://www.paludiculture.uni-greifswald.de/en/projekte/rrr2013/index.php

Günther A, **Huth V**, Jurasinski G & Glatzel S (2013b) Nasse Nutzung als Chance für Moor und Klima – Treibhausgasmessungen im Projekt "Vorpommern Initiative für Paludikultur". *BfN-Skripten*, **332**, 65.

Günther A, Jurasinski G, **Huth V** & Glatzel S (2012) Opaque chambers bias methane measurements of convective plants. *14th International Peat Congress, Stockholm, Sweden, 3-8 June*, Extended abstract No. 18, 1–5.

Huth V, Günther A, Augustin J, Borrraz E, Ellerbrock R, Giebels M, Hierold W, Hoffmann M, Jurasinski G, Rosskopf N, Sommer M, Zeitz J & Glatzel S (2013) Treibhausgas-Emissionen aus wiedervernässten Niedermooren Nordostdeutschlands. *Mitteilungen der DBG (Exkursionsführer zur Jahrestagung der Deutschen Bodenkundlichen Gesellschaft in Rostock, 07-12 September)*, **116**, 50–64. http://www.dbges.de/wb/pages/publikationen/mitteilungen-der-dbg-exkursionsfuehrer.php

Huth V, Günther A, Jurasinski G & Glatzel S (2013) Klimaschutz durch Moorschutz – Treibhausgasforschung im Verbundprojekt „Vorpommern Initiative für Paludikultur (VIP)“. *BfN-Skripten*, **335**, 23–28.

Huth V, Günther A, Jurasinski G & Glatzel S (2013) On the influence of a summer freshwater flooding on methane emissions from a 15-year re-wetted fen. *International Conference on the Utilization of Emergent Wetland Plants – Reed as a Renewable Resource, Greifswald, Germany, 14-16 February. Book of Abstracts*, 79. http://www.paludiculture.uni-greifswald.de/en/projekte/rrr2013/index.php

Huth V, Jurasinski G, Günther A & Glatzel S (2012) Winter greenhouse gas emissions of a minerotrophic fen under nature conservation management in north-east Germany. *14th International Peat Congress, Stockholm, Sweden, 3-8 June*, Extended abstract No. 146, 1–6.

Huth V, Mansur N, Grenzdörffer G, Leinweber P (2009) *Integration spektroskopischer Daten zur Abschätzung des Humuszustandes von Ackerböden.* In: Bill R, Korduan P, Theuvsen L & Morgenstern M (eds) Anforderungen an die Agrarinformatik durch Globalisierung und Klimaveränderung, *Rostock, Germany, 9-10 March*, 65–69.

Huth V, Oehmke C, Günther A & Jurasinski G (2013) The Trebel valley. *International Conference on the Utilization of Emergent Wetland Plants – Reed as a Renewable Resource, Greifswald, Germany, 14-*

16 February. *Book of Abstracts*, 18. http://www.paludiculture.uni-greifswald.de/en/projekte/rrr2013/index.php

Huth V, Vaidya S, Hoffmann M, Jurisch N, Günther A, Gundlach L, Hagemann U, Elsgaard L & Augustin J (2016) Towards a consistent approach of measuring and modelling CO_2 exchange with manual chambers. *European Geosciences Union General Assembly, Vienna, Austria,17-22 April*. *Geophysical Research Abstracts*, **18**, EGU2016-14904.

Jurasinski G, Günther A, **Huth V**, Glatzel S & Couwenberg J (2013) GHG emissions from peatlands: Effects of rewetting and land use. *International Conference on the Utilization of Emergent Wetland Plants – Reed as a Renewable Resource, Greifswald, Germany, 14-16 February*. *Book of Abstracts*, 59. http://www.paludiculture.uni-greifswald.de/en/projekte/rrr2013/index.php

Moffat AM, **Huth V**, Brümmer C, Herbst M, Andres M & Augustin J (2015) Einfluss von Raps auf den Bodenkohlenstoffhaushalt. *Jahrestagung der Deutschen Bodenkundlichen Gesellschaft, München, Germany, 05-10 September*, Oral presentation V16, https://www.dbges.de/wb/pages/jahrestagung-2015.php.

Moffat AM, Herbst M, **Huth V**, Andres M & Augustin J (2015) Impact of rapeseed cropping on the soil carbon balance. *European Geosciences Union General Assembly, Vienna, Austria,12-17 April*. *Geophysical Research Abstracts*, **17**, EGU2015-11548.

Rostock, 11.11.2016

A.2 SELBSTSTÄNDIGKEITSERKLÄRUNG

Hiermit erkläre ich, dass ich diese Arbeit selbstständig verfasst und keine anderen als die dabei angegebenen Hilfsmittel benutzt habe.

Rostock, 11.11.2016

Vytas Huth

Theses / Thesen zur Dissertation

TACKLING UNCERTAINTIES OF GHG EMISSIONS FROM MANAGED TEMPERATE PEATLANDS

submitted by Vytas Huth

I. RATIONALE AND RESEARCH OBJECTIVES

- The intensification of agricultural land use on peatlands has accelerated the peat loss over time leading to the limits of economically reasonable peatland use
- As a consequence of climate change, extreme weather anomalies have also become more frequent pushing the limits of ecosystem resilience and fluctuation of GHG emissions
- As a part of climate change mitigation strategies a variety of alternative peatland use options have been discussed in recent years, e.g, nature conservation management, rewetting and paludiculture
- The methodologies of assessing GHG balances from peatlands vary and are lacking a standardised framework further complicating reliable estimates of GHG balances of these systems.
- This thesis addresses the following questions: (*i*) Is nature conservation management of drained minerotrophic fens a sufficient measure in terms of climate protection in the light of extreme winters and N_2O emissions? (*ii*) Is long-term rewetting of mineroptrophic fens a suitable measure for climate protection in the light of increased CH_4 emissions and summer freshwater flooding? (*iii*) Is the harvest of emergent macrophytes as paludiculture crops altering the GHG balance of rewetted temperate fens? (*iv*) Do fundamental methodological differences of CO_2 exchange studies with manual chambers produce diverging carbon balances that further complicate inter-study comparisons?

II. METHODS

- GHGs were measured usually fortnightly using manual closed chambers under north-eastern german climate
- on a drained fen under nature conservation management during the extreme winter 2009/2010
- on a rewetted fen under undisturbed and harvested conditions over two years (March 2011 to March 2013)
- on a mineral soil with a high frequence of harvesting forage crops over one year (November 2013 to November 2014)

III. MAIN RESULTS

- Nature conservation management is not sufficient to reduce GHG emissions from drained minerotrophic fens even under extremely cold conditions due to relatively high CO_2 and N_2O emissions
- Long-term rewetting (>15 years) likely reduces GHG emissions from formerly drained minerotrophic fens and is able to induce enough ecosystem resilience even under extreme summer rain and freshwater flooding
- Harvesting emergent macrophytes from rewetted minerotrophic has no effect on the GHG balances and is of minor importance compared to water supply and climate variability in the short-term
- C balances diverge depending on different measurement and modelling approaches with manual chambers complicating inter-study comparisons

IV. CONCLUSIONS AND OUTLOOK

- Even under extreme climate variability, the water supply of a peatland is of major importance to mitigate high GHG emissions
- The effect of rewetting and repeated biomass removal will need long-term field trials to determine their relative importance compared to climate variability and climate change
- An important source of uncertainty when determing GHG emisssions is the lack of standardised methods to detect C balances with manual chambers
- Therefore, this thesis emphasizes the need for a standardised framework to determine GHG emissions and long-term field trials to tackle uncertainties of GHG emssions of temperate fens arising from climate variability, climate change and various management options

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“There is never enough time to count all the stars that you want.” Credit: Vytas Huth. The centre of the Milky Way taken near Krakow am See, Germany. October, 2015. Winner of the EGU Photo Contest 2016. Inspired by the wisdom of Bill Watterson.