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Biosphere-Atmosphere Exchange of CO_2 and CH_4 over Natural and Drained Bog Forest Ecosystems in Southern Germany

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List of Abbreviations and Symbols

absRMSE	Absolute relative root mean square error
BE	Bias
BMELV	Federal Ministry of Food, Agriculture and Consumer Protection
C	Carbon
c	Concentration of a scalar c
Ca	Calcium
CET	Central European Time
CH_4	Methane
C/N	Carbon – Nitrogen ratio
CO_2	Carbon dioxide
C_{org}	Organic Carbon
CWD	Coarse woody debris
D	Molecular diffusion
DM	Dry matter
DOC	Dissolved organic carbon
EC	Eddy covariance
E_0	Activation energy describes the temperature sensitivity of respiratory fluxes , expressed in Kelvin
F_c	Eddy flux of a scalar c
Fe	Iron
Foot70	Reach of the 70% footprint isoline
G	Soil heat flux
GEP	Gross ecosystem production (GPP > 0 denotes carbon uptake)
GHG	Greenhouse gas
GPP	Gross primary production (GPP < 0 denotes carbon uptake)
GPP_{max}	Maximum carbon uptake at infinitive PAR
GWP	Global warming potential
H	Sensible heat flux
HD	Humification degree
H_2O	Water
ICOS	Integrated Carbon Observation System

ITC	Integral turbulence characteristics
L	Monin Obukov length
LE	Latent heat flux
LULUCF	Land use, land use changes and forestry
LUT	Look-up table
MAE	Mean absolute error
MDV	Mean diurnal variation
n	Number
N	Nitrogen
NBB	Net biome balance
NEMP	Net ecosystem methane production
NEE	Net ecosystem exchange (uptake is negative, release is positive)
NEP	Net ecosystem production (uptake is positive, release is negative)
NLR	Non-linear regression
N_o	Number of observed half-hourly fluxes
NPP	Net ecosystem productivity
N_2O	Nitrous oxide
P	Phosphor
PAI	Plant area index
PAR	Photosynthetic active radiation
PPFD	Photosynthetic active photon flux density
PS	Photosynthesis
S_c	Source/ sink of component c
R_a	Autotrophic respiration
R_{eco}	Ecosystem respiration
$R_{eco, day}$	Daytime ecosystem respiration
$R_{eco, night}$	Ecosystem respiration at night
relRMSE	Relative root mean square error
R_{glob}	Global radiation
rH	Relative humidity
R_h	Heterotrophic respiration
RMSE	Root mean square error
R_{net}	Net radiation
R_{ref}	Respiration at a reference temperature of 10 °C

RSSI	Received Signal Strength Indicator
R^2	Coefficient of determination
S	Sulfur
T_{air}	Air temperature in °C
TERENO	Terrestrial Enviromental Observatories
TI	Thünen-Institute
T_{ref}	Reference temperature of 10°C
T_{soil}	Soil temperature
T_0	Constant temperature of 223.8 K declared by Lloyd and Taylor (1994)
u	Horizontal wind component
u_*	Friction velocity
UNFCC	United Nations Framework Convention on Climate Change
v	Lateral wind component
VPD	Vapor pressure deficit
VWC	Volume water content
w	Vertical wind component
WT	Water table
x	Direction of the mean wind
x	Measured flux
y	Direction of the lateral wind
YBP	Years before present
z	Direction of the vertical wind
z_m	Measurement height
α	Apparent quantum yield
ε	Total random uncertainty
ε_o	Random uncertainty of observed fluxes
ε_p	Random uncertainty of gap-filled fluxes
ε_t	Random uncertainty of turbulent sampling
δ	Systematic uncertainty
σ_w	Variance of the vertical wind speed

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Abstract

The main prerequisite for peatland development is a surplus of water, which leads to anoxic soil conditions. Poorly aerated soil conditions result in suppressed organic matter decomposition, which induce small but continuous carbon storage for sometimes thousands of years. On the other hand, reduced availability of oxygen naturally favors methane (CH_4) production, albeit at comparatively small emission rates. However, CH_4 is a potent greenhouse gas with a global warming potential that is 25-times stronger (on the century scale), than that of CO_2. Thus even small CH_4 releases often cancel moderate CO_2 uptake. For the full greenhouse gas balance at nutrient poor bog ecosystems, nitrous oxide (N_2O) can be generally neglected.

Small changes in abiotic parameters, due to climate- or land use change, lead to considerable changes in the carbon cycle. For example, large peatland areas were drained in the last century for peat-cutting, or agricultural- and forestry use. Drainage enhances the availability of oxygen in the soil. On the one hand, this leads to reduced CH_4 production, but on the other hand, large amounts of carbon stored in the peat are oxidized to carbon dioxide (CO_2) and released to the atmosphere. Hence, ecosystems that naturally serve as carbon sinks can turn into carbon sources after drainage.

This study focuses on the CO_2 exchange of a natural bog forest and of a bog drained for forestry, both located in the pre-alpine region of southern Germany. The net ecosystem exchange of CO_2 (NEE) has been investigated for two years (July 2010 to June 2012), using the eddy covariance technique. Additionally, at the natural bog site the methane exchange was analyzed for one annual cycle (July 2012-June 2013), to quantify the contribution of methane to the carbon budget and to the global warming potential of the natural ecosystem.

As there is a distance of only ten kilometers between the research sites, they share the same geological history and are generally exposed to the same climate and weather conditions. In contrast, the two measurement sites differ by land use history and current land use: at the natural site Schechenfilz a bog-pine forest (*Pinus mugo ssp. rotundata* (Link)) grows on an undisturbed (> 5 m) thick peat layer, while in Mooseurach a planted spruce forest *(Picea abies* (L.) KARST) grows on drained and degraded peat (<4 m).

Unexpectedly, the investigation of two annual cycles indicates much stronger CO_2 uptake of the drained forested bog ecosystem than of the natural bog-pine forest (Mooseurach: -130 ±31, and -300 ±66 g C m^{-2} a^{-1}; compared to Schechenfilz: -53 ±25, and -73 ±38 g C m^{-2} a^{-1} for 2010/2011 and 2011/2012, respectively). The strong net CO_2 uptake at the drained site can be explained by the high gross primary productivity of the spruces that offsets the relatively strong ecosystem respiration. The larger productivity of the spruces can be attributed to the larger plant area index (PAI) of the spruce site. However, even though current flux measurements indicate strong CO_2 uptake of the drained spruce forest,

on a longer time scale the site turns out to be a strong net CO_2 source, if the whole life-cycle is considered. We determined the difference between carbon fixation by the spruces and the carbon loss from peat degradation due to drainage since forest planting. The estimate resulted in a strong carbon release of +134 t C ha^{-1} within the last 44 years. This means that the spruces would need to fix carbon for another 100 years, at the current rate, to compensate the peat loss of former years. In contrast, the natural bog-pine ecosystem has likely been a small but stable carbon sink for decades, which our results suggest is relatively robust regarding short-term changes of environmental factors. Furthermore, a study of three different gap-filling methods show that the annual NEE-estimate of the natural bog-pine site is less affected by the type of gap-filling method (±18 g C m^{-2} a^{-1}), than the NEE-estimate of the drained spruce forest (±250 g C m^{-2} a^{-1}).

One measurement year (July 2012-June 2013) of methane exchange at the natural site Schechenfilz indicates small rates of net ecosystem methane production (NEMP) of +5.3 ±0.34 g C m^{-2} a^{-1}, resulting in a small greenhouse gas sink of -50 ±74 g [CO_2- eq.] m^{-2} a^{-1}. Air temperature was identified as the environmental parameter showing the highest correlation with methane production, except for periods with low water table (< -0.2 m). Three different gap-filling methods were applied to find the most appropriate approach: the mean daily variation approach (MDV), a look up table (LUT) with various environmental parameters and an exponential Arrhenius-type regression function between methane flux and air temperature (NLR). It turned out that for the data of the present study the LUT provides the best result for the gap-filling of half-hourly CH_4 fluxes. By increasing the number of parameters in the LUT, the CH_4 flux prediction could be considerably improved. Except for dry periods, day to day variations could be reproduced very well by the NLR method, but results for sub-daily fluctuations were poor. The choice of gap-filling method affects the annual methane budget estimate by at most ±0.5 g C m^{-2} a^{-1}, or about 10% of the annual flux.

To conclude, the annual carbon uptake is currently larger at the drained site Mooseurach, particularly if the methane emissions of the natural ecosystem Schechenfilz are taken into account. However, if the whole life-cycle of the spruces is considered at the drained site, that ecosystem was found to be a distinct net carbon source, because the trees cannot offset the net carbon loss, due to peat-soil mineralization, over the early years of spruce growth within one spruce rotation period.

Zusammenfassung

Moorökosysteme zeichnen sich neben einem hohen Bodenkohlenstoffgehalt durch ein meist wassergesättigtes Bodenmilieu aus. Durch die sauerstoffarmen Bedingungen, ist der mikrobielle Abbau von organischem Material gehemmt. Als Folge kann das von der Vegetation aufgenommene Kohlenstoffdioxid (CO_2) teilweise in Form von Torf über Jahrtausende hinweg gespeichert werden. Daher gelten Moore generell als natürliche Kohlenstoff-Senken. Andererseits sind die meist wassergesättigten Moorböden natürliche Quellen für Methan (CH_4). Im Vergleich zu CO_2 ist Methan ein 25-mal stärkeres Treibhausgas (bei einem Zeithorizont von 100 Jahren), sodass auch nur geringe Emissionen sich stark auf die gesamte Treibhausgasbilanz auswirken können. Änderungen abiotischer Faktoren, wie z.B. Temperaturerhöhung oder die Senkung des Grundwasserspiegels, die durch Klima- oder Landnutzungsänderungen herbeigeführt werden können, führen zu großen Veränderungen im Kohlenstoffkreislauf. Durch beispielsweise Moordrainage (z.B. für Torfgewinnung oder landwirtschaftliche Nutzung) steht umgehend Sauerstoff im Boden zur Verfügung und der eingelagerte Kohlenstoff kann in relativ kurzer Zeit oxidiert und in Form von CO_2 an die Atmosphäre freigegeben werden. Somit können vormals als CO_2-Senken fungierende Ökosysteme zu einer oft bedeutenden CO_2-Quelle werden. Anderseits führt die Verfügbarkeit von Sauerstoff dazu, dass Methan kaum mehr gebildet und emittiert wird. Der Austausch von Lachgas (N_2O) wird für die Treibhausgasbilanz von naturnahen nährstoffarmen Hochmoorböden als nicht relevant betrachtet.

Für die Quantifizierung der Treibhausgasbilanz der bisher unterrepräsentierten Moorwälder wird in der vorliegenden Studie der CO_2-Austausch mit der Eddy Kovarianz Methode über zwei verschiedenen Moorwaldökosystemen für einen Zeitraum von zwei Jahren untersucht (Juli 2010 bis Juni 2012). Die Standorte im süddeutschen Alpenvorland kennzeichnen die beiden Extreme der Moorwaldnutzung in den gemäßigten Breiten: Zum einen ein natürlicher Moorkiefernwald und zum anderen ein drainierter Fichtenforst auf Hochmoor. Im Anschluss wurde für ein Jahr (Juli 2012 bis Juni 2013) am natürlichen Standort der Beitrag von Methan an der Kohlenstoff- und Treibhausgasbilanz, ebenfalls mit Hilfe von Eddy Kovarianz Messungen, ermittelt. Durch die geringe Entfernung von nur 10 km Luftlinie unterlag die Moorbildung beider Ökosysteme den gleichen geologischen Voraussetzungen, außerdem sind beide Standorte denselben Klima- und Wetterbedingungen ausgesetzt. Allerdings unterscheiden sie sich deutlich in ihrer aktuellen und historischen Landnutzung. Der natürliche Standort Schechenfilz ist gekennzeichnet durch einen natürlichen im Durchschnitt 2 m hohen Moorkiefernwald (*Pinus mugo ssp. rotundata* (Link)), der auf einem ungestörten gut 5 m mächtigen Torfkörper wächst. Im Gegensatz dazu, ist der Torfkörper (heute < 4 m) am

Standort Mooseurach vor etwa 100 Jahren dräniert und mit heute 21 m hohen Fichten (*Picea abies* (L.) KARST) aufgeforstet worden.

Unsere zweijährige Untersuchung zum CO_2-Austausch zeigt, dass der drainierte Fichtenstandort aktuell eine größere CO_2 Senke darstellt als der natürliche Moorkiefernwald (Mooseurach: -130 ±31 und -300 ±66 g C m^{-2} a^{-1} gegenüber Schechenfilz: -53 ±28 und -72 ±38 g C m^{-2} a^{-1}). Die starke Photosyntheseleistung durch den hohen Pflanzenflächenindex (PAI) der Fichten kompensiert die relativ hohen Respirationswerte am drainierten Standort und bewirkt so netto eine CO_2-Aufnahme. Für eine bewertbare Kohlenstoffbilanz muss allerdings der gesamte Zeitraum seit Fichtenpflanzung vor 44 Jahren betrachtet werden. Schätzungen des Kohlenstoffverlustes aus dem Boden nach der Drainage gegenüber der Nettoaufnahme der Fichten zeigen, dass das dränierte Moorwald-Ökosystem netto ca. +134 t C ha^{-1} verloren hat. Das bedeutet, der Wald müsste noch etwa weitere 100 Jahre mit der aktuellen Aufnahmeleistung (-1.57 t ha^{-1}) Kohlenstoff speichern, um den Netto-Verlust an Kohlenstoff der ersten Jahrzehnte auszugleichen. Potenzielle Auswirkungen oder Konsequenzen des Holzexportes aus dem Ökosystem sind in der Bilanzierung nicht berücksichtigt.

Zusätzlich reagiert am dränierten Standort die CO_2-Bilanz vergleichsweise sensibel auf Veränderungen von Umweltparametern, wie z.B. Temperatur und Niederschlag. Im Gegensatz dazu ist der Jahresverlauf des CO_2-Austauschs am natürlichen Standorte verhältnismäßig stabil, auch länger anhaltende warme und trockene Witterungsbedingungen können meistens, z.B. durch die hohe Wasserhaltekraft des Bodens, kompensiert werden. Des Weiteren zeigt sich durch eine Studie zum Einfluss unterschiedlicher „Gap-filling-Methoden“, dass der Einfluss auf die Variabilität des CO_2-Austauschs am natürlichen Waldstandort Schechenfilz deutlich geringer ist (±18 g C m^{-2} a^{-1}), wohingegen die Wahl der „Gap-filling-Methode“ am dränierten Forststandort große Auswirkungen auf die jährliche CO_2-Bilanz hat (±250 g C m^{-2} a^{-1}).

Für eine Gesamttreibhausgas-Bilanz am natürlichen Standort ist eine Abschätzung der Methanemissionen notwendig. Die Bilanz von einem Jahr CH_4-Messung zeigt eine geringe jährliche Methanemission von +5.3 ±0.34 g C m^{-2} a^{-1}, woraus sich zusammen mit der CO_2 Aufnahme in diesem Jahr (-62 ±20 g C m^{-2} a^{-1}) eine geringe Treibhausgassenke von -50 ±74 g [CO_2 eq.] m^{-2} a^{-1} ergibt. Außerdem konnte gezeigt werden, dass kurzfristige Änderungen des Methanaustausches meistens am besten mit der Lufttemperatur korrelieren. Wenn allerdings der Grundwasserstand unter -0.2 m sinkt, verschwindet die Korrelation mit Luft- und Bodentemperatur und die Methanemissionen sind deutlich reduziert. Zusätzliche Untersuchungen von drei verschiedenen „Gap-filling“ Methoden („Mean daily variation“ (MDV), „Look-up-table“ (LUT) und „Non-linear regression“ (NLR)) zeigen, dass die Jahresbilanz verhältnismäßig robust auf die Anwendung unterschiedlicher Methoden reagiert. Insgesamt aber führte die „Look-up-table Methode“ zu der besten Bestimmung des halbstündigen Methanaustausches.

Zusammenfassend wird gezeigt, dass aktuell die gemessene netto Kohlenstoff-Aufnahme am dränierten Standort größer ist, insbesondere wenn die Methanemissionen des natürlichen Ökosystems miteinbezogen werden. Andererseits stellt sich heraus, dass im Vergleich der dränierte Moorwald langfristig, z.B. für die Dauer einer Rotationsperiode, eine wesentlich stärkere Kohlenstoffquelle ist.

1. Introduction

1.1 Motivation

Peatlands can be found all over the world except in arid and semi-arid regions; however, most of them develop in the cold humid latitudes of the northern hemisphere (Fig. 1). Nevertheless, they are relatively rare ecosystems with a global land surface coverage of only 3% (Rydin et al., 2013).

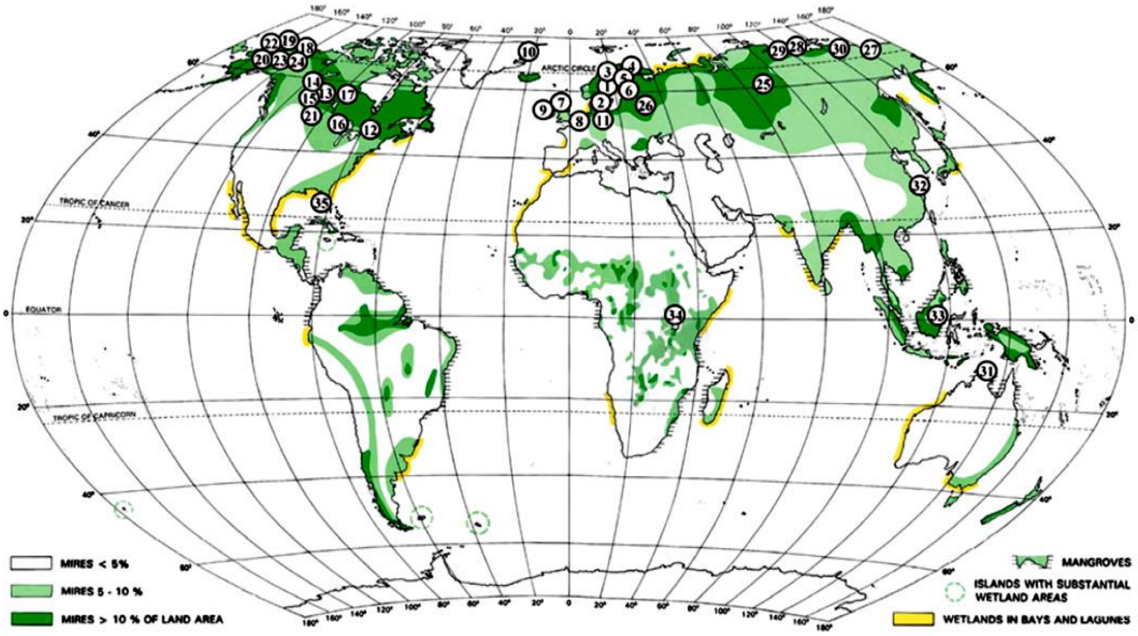

Fig. 1 Global distribution of mires (Lappalainen, 1996) together with the locations of present and earlier eddy covariance measurement sites (numbered circles, information to the sites can be found in Laurila et al. (2012))

However, peatlands play a major role with respect to the global carbon balance, and thus for the greenhouse effect. Since the last ice age (~10,000 YBP) peatlands have cumulatively stored between 270 and 455 Pg C (Gorham, 1991; Turunen et al., 2002), which corresponds to 20-30% of the world's estimated soil C pool (Gorham, 1991) and to 50% of the atmospheric carbon (Gorham, 1991; Houghton et. al., 1990). Pristine peatlands are the only natural ecosystems that can accumulate carbon continuously and over periods of thousands of years (Clymo, 1984). However, this carbon storage pool is threatened, as many natural peatlands are disturbed by human interference, such as peat cutting and land use change for agricultural (e.g., Alm et al., 1999b; Drösler et al., 2008), or forestry use (e.g., Lindroth et al., 1998; Meyer et al., 2013). In addition, the carbon storage potential of peatlands is threatened by climate change induced drought, as lower water tables lead to marked CO_2 emissions in peatland ecosystems (e.g., Alm et al., 1999b; Arneth et al., 2002; Aurela et al., 2007). Peatland drainage leads to water table drawdown and the formerly anaerobic soil layer becomes aerobic. Hence, carbon stored in the peat layer for thousands of years can be oxidized and released as carbon dioxide (CO_2) to the atmosphere within a relatively short time. Drained peatlands are thus expected to lose their sink function and become a carbon source.

To estimate the total global warming potential (GWP) of peatland ecosystems, further greenhouse gases (GHG) are important, depending on water table depth and nutrient supply. The anoxic soil conditions of pristine peatlands commonly support emissions of methane. Methane is a 25-times stronger greenhouse gas than the reference gas carbon dioxide, considering a time horizon of 100 years. It is responsible for about 20% of current climate forcing (IPCC, 2007). Wetlands, both natural as well as anthropogenically influenced ones, are estimated to contribute to one quarter of the total annual methane emissions (Lelieveld et al., 1998). Methane emissions exhibit high spatial variations among peatland types (Lai, 2009) and various land use forms (Byrne et. al., 2004; Hendriks et al., 2010). Generally, natural peatlands are seen to affect the radiative forcing of the atmosphere in two opposite ways: cooling induced by CO_2 uptake from the atmosphere (acting on a timescale of millennia) and warming induced by CH_4 emissions on a timescale of decades (Frolking et al., 2006; Laurila et al., 2012).

Nitrous oxide (N_2O) is the third most important greenhouse gas that has to be taken into account for the estimate of the global warming potential of peatlands ecosystems. Nitrous oxide is a very strong greenhouse gas; its century scale global warming potential is 298-times that of CO_2 (IPCC, 2007). Unmanaged peatlands commonly have negligible emissions of N_2O or can even act as weak sinks (e.g., Martikainen et al., 1993; Minkkinen et al., 2002; Regina et al., 1996). However, if peatlands are drained and fertilized, N_2O can be emitted at significant rates (e.g., Drösler, 2005; Maljanen et al., 2010).

Within the last 20 years, the determination of greenhouse gas exchange of peatlands has received increasing attention, by science as well as by politics in the course of the obligations of the Kyoto-protocol. To date, a number of studies have investigated the CO_2 exchange of peatlands and the environmental factors that control it by eddy covariance (e.g., Aurela et al., 2009; Lafleur et al., 2005; Sottocornola and Kiely, 2005), or chamber measurements (e.g., Bubier et al., 2003a; Drösler, 2005; Goulden and Crill, 1997). The resulting carbon budgets depend on land use and peatland type. Generally ombrotrophic bogs are stronger carbon sinks than more nutrient-rich fens (Byrne et. al., 2004), while intensive land use (e.g., cropland) leads to stronger CO_2 emissions than extensive land use (Byrne et. al., 2004; Drösler et al., 2008). Most studies to date have focused on the greenhouse gas exchange of natural and agriculturally used peatlands (e.g., Alm et al., 1999a; Aurela et al., 2007; Aurela et al., 2009; Bubier et al., 2003b; Hendriks et al., 2007; Lafleur et al., 2003; Sottocornola and Kiely, 2005). However, studies about peatland forests are still rare; e.g., Maljanen et al. (2010) emphasize the insufficient knowledge about the full greenhouse gas exchange of peatland forests, even in the boreal climate zone. So far, there is no agreement whether peatland drainage for forestry generally leads to net gain or net loss of carbon. A number of studies about the long-term net carbon exchange of drained peatland forests have been carried out in Great Britain and Scandinavia. Most of them are based on extrapolated chamber measurements and model approaches. These studies found that the loss of peat carbon is mainly compensated by the carbon fixation of the trees (e.g., Hargreaves et al., 2003; Minkkinen et al., 2002; Ojanen et al., 2013). On the other hand, Armentano and Menges (1986) noted that, in temperate regions, regional mean losses of soil carbon are about 10 times larger than in boreal regions. Cannell et al. (1993) concluded that carbon losses larger than +300 g C m^{-2} a^{-1} cannot be compensated by the carbon uptake of the trees in drained and afforested peatlands in Britain, and that peatland afforestation may lead to net loss of carbon.

Some short-term studies conclude that peatland ecosystems drained for forestry are net CO_2 sinks (Lohila et al., 2011; Meyer et al., 2013; von Arnold et al., 2005). However, eddy co-variance measurements over a boreal drained forest in central Sweden show considerable net CO_2 loss over two measurement years (Lindroth et al., 1998), while other studies determined annual net CO_2 exchange close to zero (Dunn et al., 2007; Lohila et al., 2007). The recently published “Wetlands Supplement to the IPCC 2006 Guidelines for National Greenhouse Gas Inventories” (Drösler et. al., 2013) considered drained and afforested peat soils as carbon dioxide sources, and in addition the report quantified that temperate ecosystems are stronger greenhouse gas sources than boreal peat soils. Overall, the results of these studies are fraught by large uncertainties, and most are based on measurements that were extremely extrapolated in time and space. Furthermore, the reported carbon budgets are difficult to compare, because the carbon balances were determined by several different approaches (e.g., eddy covariance method, chamber measurements, and biometric estimates of carbon stock changes). In addition, studied peatland sites were highly variable in terms of their nutrient supply, drainage system, tree population, and their climate conditions (e.g., temperatures, precipitation, and snow coverage), all of which affect the cycling of carbon through these ecosystems.

Methane exchange of peatlands has been intensively measured in the last two decades. Several measurement approaches, such as static chambers (e.g., Beetz et al., 2013; Drösler, 2005; Moore and Knowles, 1990; Moore et al., 2011), automatic chambers (e.g., Bäckstrand et al., 2010; Bubier et al., 2003a), flux gradient techniques (e.g., Edwards et al., 2001; Miyata et al., 2000; Simpson et al., 1997), and eddy covariance measurements (e.g., Baldocchi et al., 2012; Herbst et al., 2011; Rinne et al., 2007) have been used to estimate the net ecosystem methane production (NEMP) over different peatland types. Although, there is much experience of CO_2 and water vapor-exchange measurements by eddy covariance technique, some details of determining annual CH_4 balances by eddy covariance are still rarely discussed. For example, the effect of different gap-filling methods for methane fluxes has not been reported yet, to our knowledge.

The current thesis intends to contribute to a better understanding of the carbon exchange of two poorly investigated ecosystem types. It compares the CO_2 exchange of a natural bog-pine ecosystem and a managed spruce forest that mark the extremes of bog-forest management, for a period of two years. The net ecosystem methane exchange has been measured at the natural site for 15 months. This study is one of the first eddy covariance based annual methane exchange estimates over a natural bog-pine ecosystem outside the boreal zone

1.2 Peatland ecosystems in general

Peatlands are areas, with or without vegetation, with a naturally accumulated peat layer at the surface. Following Joosten and Clarke (2002), peat is defined as accumulating material (bottom-up growing sedimentation) consisting of at least 30% (dry mass) of dead and humified plant material. Due to water-logged soil conditions and therefore low oxygen availability this material cannot be entirely decomposed. The structure of plant material is still recognizable (Succow et al., 2001). Commonly we have to distinguish between two different peat layers in profile: the acrotelm and the catotelm. The acrotelm is the upper layer where water table fluctuations occur. This zone is characterized by a content of high organic matter that is poorly decomposed with high specific yield and porosity. The pore structure leads to a particularly large water storage capacity (Strack et al., 2008). The catotelm is the lower and deeper layer of peat that is permanently water saturated and contains highly decomposed organic material. Since catotelmic peat is degraded and contains smaller pores, less water can be drained by gravity, resulting in greater water retention. Consequently, the structural differences between acrotelm and catotelm are important in determining the storage of water in bog systems (e.g., Romanov, 1968). The high water storage capacity of the acrotelm and its ability to shrink and swell act as regulatory functions that minimize water table fluctuations, maintaining the water table close to the surface (Ingram, 1983).

Peatlands have been variously defined, depending on country and scientific discipline. Generally peatlands are supposed to have a minimum peat thickness between 20 and 70 cm (Montanarella et al., 2006). Peat is included in soil classification systems under names such as ‘peat soils’, ‘muck soils’, ‘bog soils’, and ‘organic soils’, but the name that is used most commonly at international level is “Histosol” (FAO, 1998).

Depending on trophic status and water source, peatlands can be classified in fens and bogs. Bogs are ombrotrophic sites, implying that they are served solely by precipitation. Because of the bottom-up growing of the peat-mosses, called *sphagnum*, bogs are isolated from laterally moving groundwater. Thus bogs are typically nutrient poor and very acidic ecosystems. Sphagnum is growing on its own dead material, leading bogs to rise above the surrounding landscape surface. The existence of peat mosses is an important prerequisite for peat growth.

Fens on the other hand are minerotrophic peatlands that are still connected to groundwater and thus have a better nutrient availability. They are typically more or less pH-neutral. While the dominating species in bogs are peat-mosses, fens are dominated by sedges (*Carex spec.*) (Keddy, 2010; Lai, 2009).

Peatland ecosystems, including bog and fens, can be found almost anywhere, but mostly in regions where precipitation exceeds evapotranspiration. Thus most of the peatlands are located in the cold and humid latitudes of the northern hemisphere, e.g., they cover large areas in Russia, North America and Europe (International Peat Society, 2008). Undisturbed peatlands are the only natural ecosystems that can accumulate carbon continuously and over periods of thousands of years (Clymo, 1984). Present peatland ecosystems of the northern hemisphere are typically the result of about 10,000 years of net carbon accumulation. For example, Tolonen and Turunen (1996) estimated an average annual carbon-accumulation rate of Finnish mires between 15-30 g C m^{-2} a^{-1}.

Apart from their function as long-term carbon storage, peatlands are important for water retention. Despite its service functions, the global peatland coverage is decreasing. The essential water-logged soil conditions for peatland development make peatlands particularly vulnerable. For any cultivation, peatlands have to be drained. Thus, large peatland areas have been drained for agriculture, peat cutting or forestry. Due to drainage, the soil is no longer water logged, all peat-beneficial factors are disturbed, and the special fauna and flora of peatlands, which is particularly adapted to wet and nutrient poor conditions, is replaced by other species.

For a range of reasons, like landscape conversion or biodiversity protection, peatland restoration is of growing importance in Europe and North America, and is likely to remain relevant for the next half century (International Peat Society, 2008). The long-term effect to the greenhouse gas balance is not finally understood, but the common opinion is that after more than 10 years after rewetting, the greenhouse gas exchange will be close to pre-drainage conditions (International Peat Society, 2008).

1.3 Peatlands and their carbon cycle

Generally, in undisturbed peatland ecosystems the decomposition of organic matter is suppressed by anoxic and often low temperature soil conditions. In consequence, the net primary production exceeds ecosystem respiration. Therefore, peatlands have the potential to act as effective long-term carbon sinks. However, the accumulation of carbon results mainly from inhibited decomposition, rather than high photosynthetic uptake rates. Even so only 2-16% of the net ecosystem production of the peatland vegetation ends up to be stored in peat (Paavilainen and Paivanen, 1994). Thus, the largest part of the annually produced biomass will pass though the short carbon cycle and is decomposed again. Peat is therefore very slow growing, on average 1 mm per year. However, according to the IPCC (2007), peat is declared, somewhat confusing, “as renewable but fossil fuel”.

The carbon budget of peatlands has three main components: the exchange of carbon dioxide (CO_2) and methane (CH_4) between atmosphere and biosphere and the stream loss of dissolved organic carbon (DOC). The CO_2 fluxes are expected to be the largest part of the carbon cycle (e.g., Moore et al., 1998).

The CO_2 exchange is often expressed as net ecosystem exchange (NEE) which can be divided in the two components: ecosystem respiration (R_{eco}) and gross primary production (GPP). Ecosystem respiration summarizes heterotrophic soil respiration (R_h) of microorganisms and autotrophic plant respiration (R_a), the loss of carbon by internal plant metabolism, while the GPP is the total carbon uptake due to photosynthesis of the vegetation (e.g., Drösler, 2005).

Thus,

$$NEE = GPP + R_{eco} \quad . \tag{1}$$

We use the common micrometeorological convention where negative NEE means carbon uptake and positive NEE carbon release by the ecosystem. Thus GPP is denoted to be negative here and R_{eco} positive. GPP and NEE as determined form eddy covariance measurements (see Section 2.3), are equivalent to gross ecosystem production (GEP) and net ecosystem production (NEP), except for the change of sign.

Further definitions are used to express single processes or total balances following Schulze et al. (2000) (Fig. 2).

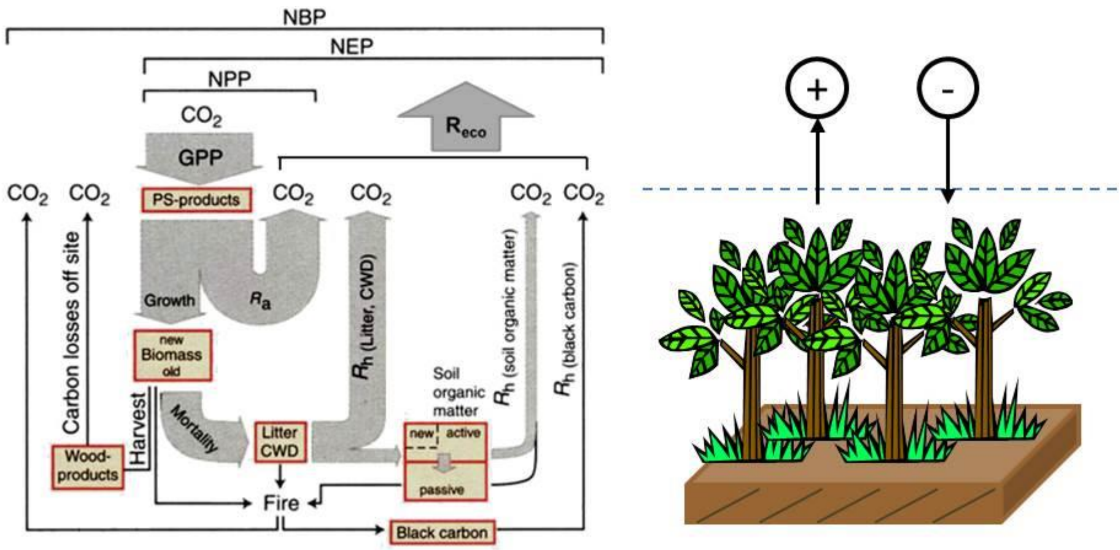

Fig. 2 Left: modified from Schulze et al. (2000): overview of main terms describing carbon balances of an ecosystem, arrows indicate fluxes, and boxes indicate pools. The size of the boxes represents differences in carbon distribution in terrestrial ecosystems. CWD= coarse woody debris, PS Photosynthesis. Right: sign convention for exchange fluxes, e.g., CO_2 or CH_4.

1.3.1 Carbon exchange in natural peatlands

A schematic overview of CO_2 and CH_4 exchange in natural peatlands is given in Fig. 3.

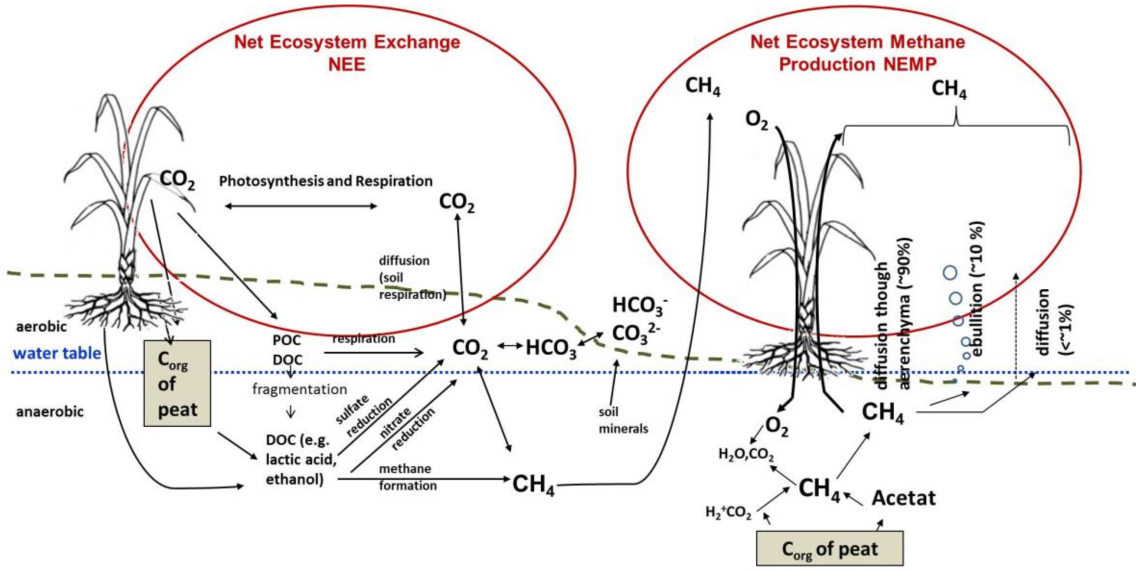

Fig. 3 Carbon dioxide and methane cycle in undisturbed peatland ecosystems, based on Mitch and Gosselink (1993) (left) and Keine (1991) (right)

Carbon dioxide (CO_2)

Carbon enters the peatland ecosystem particularly by CO_2 fixation via photosynthesis. A small part is transported via precipitation, and in fens additionally via the lateral groundwater flow which can be an important source of dissolved organic- or inorganic carbon. After plant death, rapidly decomposable carbon compounds are oxidized to CO_2 and released to the atmosphere. Persistent carbon compounds, like lignin are preserved, converted to humic substances, and stored as peat. If soil conditions are aerobic, all degradable carbon compounds of dead biomass are eventually respirated as CO_2 to the atmosphere (Succow et al., 2001).

Methane (CH_4)

Within the anaerobic zone, organic carbon compounds are fermented by microorganisms to e.g., acetate (CH_3COOH), hydrogen (H_2) or ethanol (C_2H_6O). These products can be degraded by other microorganisms by nitrate or sulfate reduction (Fig. 3). These conversions may lead to CO_2 release as well, but anaerobic bacteria (methanogenic microbes, Arechea) can convert acetate as well as CO_2 to methane (Eq. 2, 3).

By the two main processes of methane generation acetotrophic methanogens use acetate as a substrate to produce methane and carbon dioxide gases (e.g., Lai, 2009; Whalen, 2005):

$$CH_3COOH \rightarrow CH_4 + CO_2, \qquad \textbf{(2)}$$

and hydrogenotrophic methanogens reduce CO_2 using H_2 gas as an electron donor:

$$4H_2 + CO_2 \rightarrow CH_4 + 2H_2O\ . \qquad \textbf{(3)}$$

Acetate and hydrogen are generated by the decomposition of organic matter. The access of organic matter to methanogens is often the limited factor of methane emission (Coles and Yavitt, 2002). Methane gas is released from peat via three main pathways, namely diffusion, ebullition and plant mediated transport (e.g., Lai, 2009). Overall, the most important pathway is plant transport. Many wetland plants possess aerenchymous tissue as an adaptation to rooting in the water-logged soils, that allows oxygen to reach the root zone (Couwenberg, 2012). In the other direction along the aerenchym, methane can bypass the near surface aerobic soil layer without oxidation to CO_2. Ebullition is a process that releases CH_4 formed as gas bubbles into the atmosphere. This release is often associated with changes in water level (Strack et al., 2005), barometric pressure (Comas et al., 2008; Tokida et al., 2007b), temperature (Beckmann et al., 2004) or mechanical stress (Goodrich et al., 2011). Also thaw periods frequently lead to ebullition events (Hargreaves et al., 2001; Tokida et al., 2007a). Finally, compared to the other pathways, the diffusive efflux of methane is of minor importance (Lai, 2009).

In spite of such natural methane emissions from undisturbed peatlands, over the time span of several centuries, undisturbed peatlands exert a net cooling effect on the global radiation balance, because the effect of removing long-lived (~50-100 years) atmospheric CO_2 ultimately surpasses that of releasing short-lived (~10-15 years) CH_4 (Frolking and Roulet, 2007).

Dissolved organic carbon (DOC)

Export of dissolved organic carbon by stream- or groundwater-flow is estimated to be the smallest part of the entire carbon budget of peatlands. For example, Buffam et al. (2007) reported a DOC export of 2-10 g C m^{-2} a^{-1} for 15 Swedish peatland sites. However, we still know relatively little about what controls the transport of DOC and particulate matter through stream loss in peatlands (Limpens et al., 2008). Drösler et al. (2013) recently reviewed an annual DOC-loss from natural peatland sites ranging between +4 and +63 g C m^{-2} a^{-1}. Highest loss rates were found for tropical peatlands, while the DOC-loss at boreal sites was in the lower range. For temperate peatland the reported average loss rate was +25 ±17 g C m^{-2} a^{-1}.

1.3.2 Carbon exchange in drained peatlands

The carbon balance of drained peatlands is different from pristine peatlands. Due to the newly aerobic soil conditions, the carbon stored in peat can be oxidized and released to the atmosphere within a comparatively short time. Peatlands can thus become a strong source of CO_2. Depending on nutrient status, also emissions of the powerful greenhouse gas N_2O may increase. But on the other hand, methane emissions may cease or be greatly reduced, because the organic carbon can be oxidized to CO_2 (Fig. 4).

The strength and composition of greenhouse gas emissions depend on environmental conditions, land use and nutrient supply. Numerous studies have been carried out to investigate the impact of different cultivations und different peatland classes all over the world, to quantify the greenhouse gas balance after drainage (among others: Alm et al., 1999a; Drösler, 2005; Lohila et al., 2011; Veenendaal et al., 2007).

In particular, the long-term effect of peatland restoration is not completely understood. Hence, it is unknown which method of peatland restoration is the most effective, but it is generally accepted that CO_2 and N_2O emissions are reduced rapidly, while CH_4 emissions could become large. Often, the vegetation density of drained peatlands is larger, compared to natural ones. If the site is not cut before rewetting, species which are generally not well adapted to high levels of soil moisture will die. Thus, a lot of organic matter is available to be decomposed. Due to anaerobic conditions in the soil, the carbon may be converted to CH_4 and emitted the atmosphere.

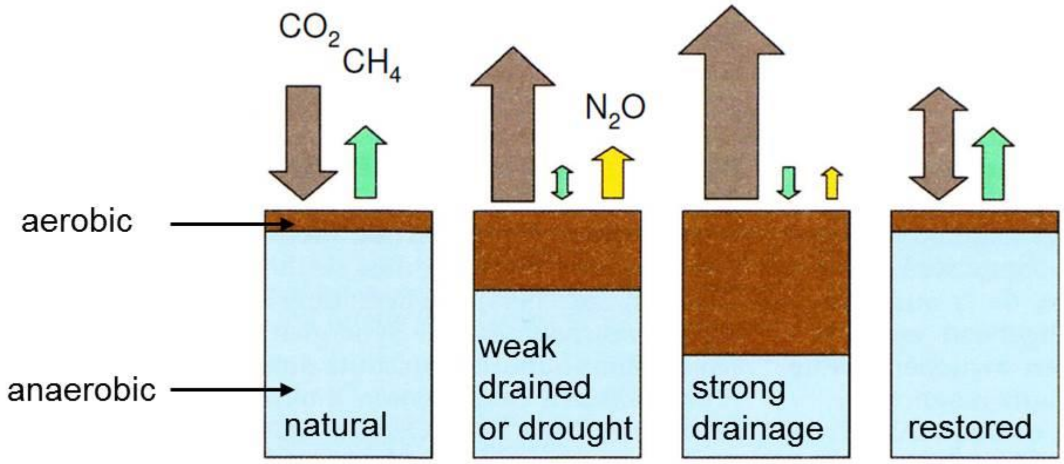

Fig. 4 Schematic representation of greenhouse gas exchange dependent on water level (modified after Drösler et al. (2008)). Size of arrows represents the strength of trace gas exchange. Downwards direct are uptake and upwards directed arrows are emission.

A lot of different components influence the estimation of the greenhouse gas balance of peatland ecosystems. The water level as well as nutrient supply and vegetation composition are likely important. Because of the varying radiative forcing of the different greenhouse gases CO_2, CH_4 and N_2O, their individual global warming potentials (GWP) should be considered.

1.3.3 Global warming potential (GWP)

According to the IPCC (2007), the global warming potential (GWP) is a simplified index of the relative radiative effect of greenhouse gases compared to CO_2 (CO_2-equivalents, expressed as g C [CO_2-eq] m^{-2} a^{-1}). The GWP is based on a number of factors, including among others the radiative efficiency and the decay rate. This index can be used for estimating the potential future impacts of emissions upon the climate system (e.g., Drösler et al., 2008). Global warming potentials are used to express radiative forcing of a gas over a defined time period. For reporting under the UNFCCC and for biological sink assessments under the Kyoto Protocol, global warming potentials values are typically used for the 100-year time horizon (Tab. 1).

Tab. 1 Peatland relevant greenhouse gases and their global warming potential depending on time horizon (IPCC, 2007).

Gas	**Global warming potential**		
	20 years	100 years	500 years
CO_2	1	1	1
CH_4	72	25	7.6
N_2O	289	298	153

The global warming potential used in this study based on IPCC (2007). The balance of global warming potentials can be used to express the sites-specific climate impact of greenhouse gas emissions and uptake, respectively. The GWP-balance can be written as:

$$GWP\text{-}balance = F_{CO_2} \times GWP_{CO_2} + F_{CH_4} \times GWP_{CH_4} + F_{N_2O} \times GWP_{N_2O} \,. \qquad \textbf{(4)}$$

We have to note that the calculation of GWP is based on the molar mass of the whole CO_2 (g CO_2 m^{-2} a^{-1}), CH_4 (g CH_4 m^{-2} a^{-1}) and N_2O (g N_2O m^{-2} a^{-1}) molecules, including the molar mass of oxygen and water, while the balances of NEE and CH_4 (NEMP) considers only the carbon content (g C m^{-2} a^{-1}). The analyses of this thesis consider solely the GWP of the century scale.

1.4 Peatlands and climate policy

Peatlands are hotspots of GHG emission, even in industrialized countries like Germany. In Germany drained peatlands are estimated to account for about 2.3-4.5% of the total anthropogenic greenhouse gas emissions (Drösler et al., 2008). They are the largest single source beside the industrial sector and need to be reported with state-specific emissions factors (Tier 2 of IPCC reporting rules), depending for example on climate conditions, soil and vegetation type. Currently, such emission factors are not available for many land use und soil types (NIR, 2011).

Germany, like many other states, has to report annually its anthropogenic greenhouse gas emissions according to the UN Framework Convention on Climate Change (UNFCCC). Accounting of anthropogenic emissions of peatlands is done in different chapters of the Kyoto protocol: chapter 4 (agriculture) include N_2O emissions from organic soils under cropland and grassland. In chapter 5 (land use changes and forestry (LULUCF)) C-emissions of afforestation, deforestation and forest management on organic soils are accounted for.

Germany selected in 2006 from the Art 3.4 of the Kyoto Protocol (UNFCCC, 1998) the activity of forest management. Under that coverage, activities to reduce the emissions of organic soils under forests can be accounted for and consequently financed. Therefore solid data are needed to assess the sink strength of forest ecosystems on organic soils and the effect of management activities.

It is estimated that "climate-friendly" restoration of drained peatlands in Germany could offset emissions by up to 35 million tons of CO_2 equivalents per year (Freibauer et al., 2009). In addition to the important benefit to biodiversity conservation, peatland restoration is thus expected to have enormous climate change mitigation potential. For a better assessment of this potential, several regional and international projects are ongoing or have been completed recently; e.g., GHG-Europe (Task 2.1 Peatland synthesis) or the German cooperative project “Organic soils” funded by TI, in which the present study is participating.

1.5 Related projects

This thesis is related to, and in part supported by, four different projects; the so-called TI-project “Organische Böden” (Organic Soils): cooperative project; Climate reporting "Organic Soils" -assessment and development of methods, activity-data and emission-factors for the climate reporting under LULUCF/ AFOLU (funded by TI BMELV; 2009-2012; Subproject: Eddy-Covariance (I); the so called LfU-Project: “Effect of peatland restoration on climate change-Assessment of mitigation potential of peatland restoration in Bavaria within framework of Klip2020” (II), the TERENO infrastructure activity (III) and the ICOS-ecosystems long-term project (IV).

I. TI-Project: “Organic Soils” (2009-2012)

The scope of this project is the determination of country-specific emission factors depending on climate region, soil type and land use, which need to be reported in the national greenhouse gas inventories with country-specific emission factors as well as with a complete set of activity data (e.g., management, soil type or groundwater level).

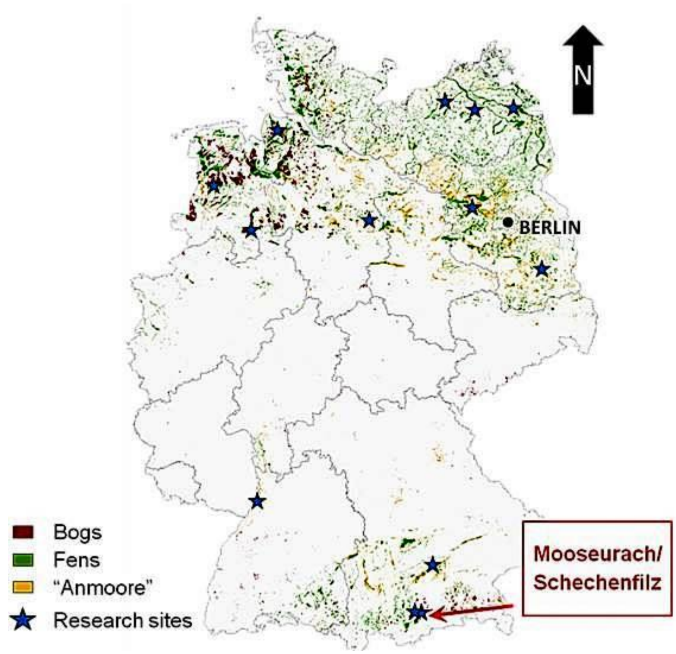

Fig. 5 Map of German histosols, based on: GÜK200 (1:200,000), source: Bundesanstalt für Geowissenschaften und Rohstoffe. Blue stars: 14 research sites in the project “Organic Soils”, brown areas are bogs, green ones are fens and yellow areas are “Anmoore” (mineral soil with an organic dry matter content of 15-30%).

The exchange of greenhouse gases (CO_2, CH_4 and N_2O) was investigated at 14 peatland sites in Germany (Fig. 5), the study sites differed in land use, management and peatland type. The GHG exchange was determined by different techniques: manual chambers, automatic chambers and the eddy-covariance measurement method. Additionally, all important meteorological, vegetational and hydrological driving factors have been recorded. For further information, see: www.organische-boeden.de, last access: 19.09.2013.

II. LfU-Project: "Effect of peatland restoration on climate change-Assessment of mitigation potential of peatland restoration in Bavaria within framework of Klip2020" (2008-2011)

This project aims to assess the climate impact of already restored peatlands in Bavaria by investigation of greenhouse gas exchange with focus on underrepresented land use types, like peatland forests or landscape conservation sites and their temporal development. Good practice guidelines should be developed that optimize the climate change mitigation potential from peatland restoration. For further information, see: www.lfu.bayern.de, last access: 19.09.2013.

III. Terrestrial Environmental Observatories (TERENO) (2008-ongoing)

The infrastructure activity TERENO is an interdisciplinary long-term project that aims to establish an observation platform linking terrestrial observatories in different regions. Four different regional observatories (Fig. 6) are established in Germany that generate continuous data sets covering spatial and temporal scales.

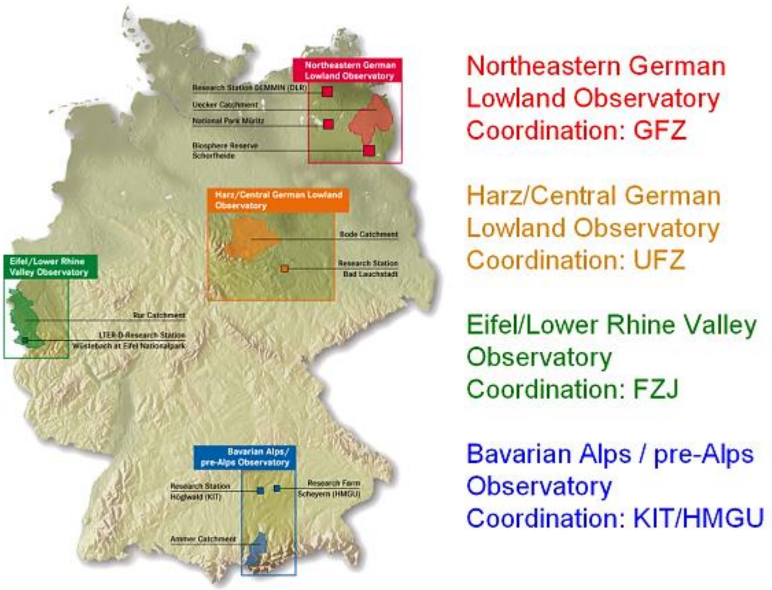

Fig. 6 Location of the four observatories in the terrestrial observatory network TERENO. The Observatories spread over Germany and cover different sites and catchments of different ecosystems.

These datasets help to improve our knowledge of the complex biosphere-hydrosphere-atmosphere interactions and to detect and analyze the impact of Global Change parameters on these interactions as well as to develop, improve and validate biosphere-hydrosphere-atmosphere model systems. Further information is available on www.tereno.net, last access: 02.06.2014.

IV. European Project: "Integrated Carbon Observation System (ICOS)" (2012-ongoing)

The objective of the ICOS-network (Integrated Carbon Observation System) is to observe and understand the current state of carbon exchange and predict the future behavior of climate change, carbon cycle and greenhouse gas emissions. Therefore, ICOS provides a European long-term (20+ years) carbon observations network system. Carbon fluxes should be tracked by monitoring ecosystems (40-60 sites), the atmosphere (30 sites) and the oceans (Fig. 7).

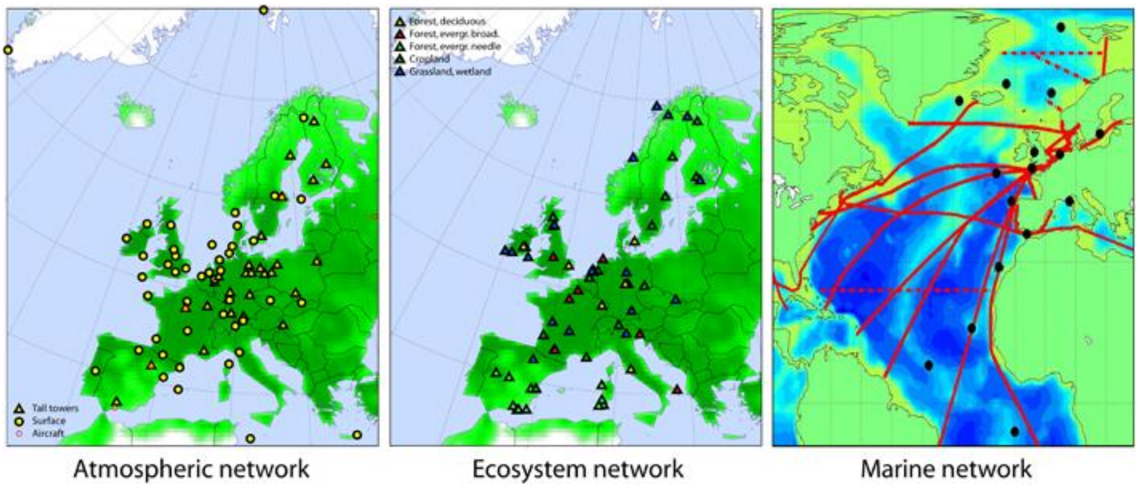

Fig. 7 Sites monitoring: the atmosphere (left); different ecosystems (middle) and the oceans (right), source: ICOS www.icos-infrastructure.eu, last access: 27.11.2013.

Further information is available on the project web page www.icos-infrastructure.eu, last access: 27.11.2013.

1.6 Objectives

The present research aims at investigating the carbon exchange of a drained and a natural bog forest of the temperate climate zone, based on eddy covariance flux measurements. This study focuses on three key issues:

The first issue addresses the influence of environmental forcing on the variability of net ecosystem exchange of CO_2 (NEE) at the drained site, compared to that at the natural site. The carbon exchange of peatlands is hypothesized to be very sensitive to changes in environmental forcing, e.g., due to climate change. Thus, the question arises whether the NEE at the drained site or at the natural site is more stable in terms of different environmental conditions. For the natural site even the environmental influence on the net methane productivity and the parameters affecting the methane exchange is an important issue. It should be examined, whether temperature, water table depth or even other parameters, like footprint dimension or photosynthetic active radiation explain most of the short-term variation of methane.

The second issue focuses on the influence of different gap-filling methods and on estimates of the annual budgets of NEE at both sites, and of net ecosystem methane productivity (NEMP) at the natural site. As commonly about 50% of eddy covariance flux data need to be rejected during quality control, creating data gaps, it is expected that the type of gap-filling procedure used strongly affects the estimate of annual net carbon exchange at both sites.

Finally, the study focuses on the comparison of the annual NEE at the drained and the natural site. It should be clarified, whether the carbon fixation of the spruces at the drained ecosystem can compensate the expected carbon loss from the drained peat. The objective is to quantify and discuss the annual net CO_2 balance at both sites for the observation period, and to evaluate the results with a view to the long-term carbon balance of peat-bog ecosystems. In addition, significant methane emissions are expected at the natural site. These should be quantified and discussed: how large is the contribution of expected methane emissions to the net carbon- and greenhouse gas balance at the natural site?

2 Materials and Methods

2.1 Research sites

Measurements took place at two bog forests located within the pre-alpine region in southern Germany, approximately 40 km south of Munich. The lakes and peatlands in this pre-alpine region were formed on the basal moraine after the retreat of the Isar-piedmont glacier (Isar-Vorlandgletscher) at the end of the last ice age. Peatlands in this region extend over the lowlands in an area south of lakes Ammer and Starnberg that reaches far beyond the two present study sites (Fig. 8c).

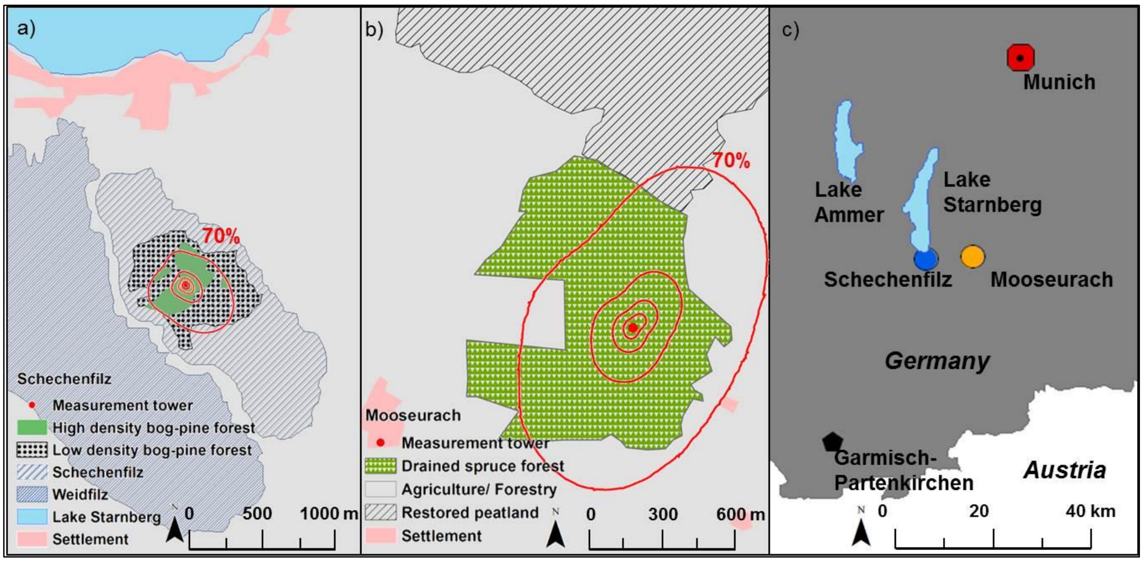

Fig. 8 a) Schematic map of Schechenfilz site. The target area is the bog-pine forest (high and low density), b) schematic map of Mooseurach site (the target area is the drained spruce forest). Red lines indicate the isolines (10%, 30%, 50% and 70%) of the mean flux footprint area of the year 2011; c) location of the research sites in southern Bavaria, Germany. Note the different scales of each map.

The natural peatland Schechenfilz is close to the southern end of Lake Starnberg, near the town of Seeshaupt (Fig. 8a, c). The drained peatland forest Mooseurach (Fig. 8b,c) is located 10 km to the east. The sites are part of the pre-alpine bog zone; they share the same geological history and are exposed to the same general weather conditions. The climate at both sites can be characterized as cool-temperate and humid (e.g., Kottek et al., 2006). At the sites the air temperature is on average 8.6 °C and the annual precipitation 1127 mm (Tab. 2), with the maximum (40%) in summer, in June and July (Data of German Weather Service (DWD)). However, the two bogs sites are clearly distinct by their land use and land use history.

Tab. 2 Long-term averages of meteorological parameters (sum of precipitation, mean of air temperature, sum of sunshine duration), provided by the nearby (distance of 8.5 km and 10.5 km to Schechenfilz and Mooseurach, respectively) climate-station "Attenkam", run by the German Weather Service (DWD).

Period	Station (Elevation m a.s.l.)	Precipi-tation mm a^{-1}	Air temperature °C	RH %	Sunshine duration h
1971-2011	DWD (672)	1127	8.6	77.9	1817
2010	DWD (672)	1209	7.6	80.1	1645
2011	DWD (672)	1092	9.6	78.4	2114
2012	DWD (672)	1141	8.7	80.2	1902

2.1.1 Schechenfilz

The ombrotrophic bog Schechenfilz (111 ha) is located about 3 km to the southeast of Seeshaupt. The Schechenfilz is part of the conservation area "Osterseen" that is also part of the Europe-wide "Habitats Directive" (Fig. 9).

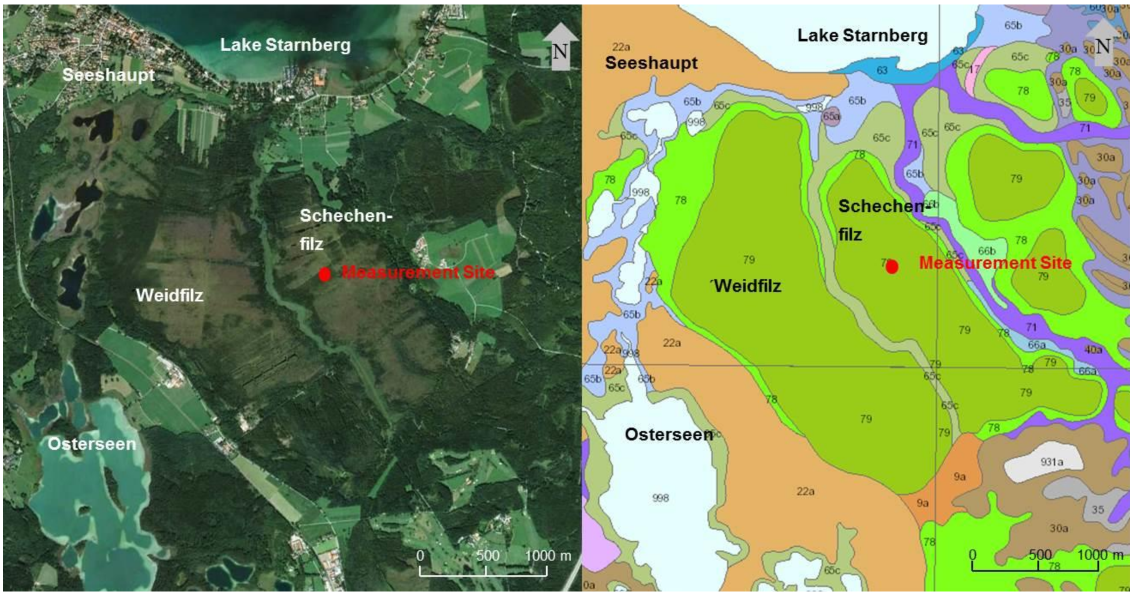

Fig. 9 Left: aerial image of Schechenfilz, (based on Google Earth); Right: soil map section (Key: 79: organic soil (bog) 78: organic soil (fen) 65c: "Anmoor" (gley soil, loamy, calcareous), 65 b: Gleyic-brown earth (loamy, calcareous), based on Übersichtsbodenkarte (ÜBK25), source: Bayerisches Landesamt für Umwelt. Both images represent the same area.

The Schechenfilz bog was affected by peat-cutting until the 1950s. The northern part was prepared for peat-cutting by drainage ditches. These ditches were refilled for peatland restoration in 2001. However, the central part of the bog, where the studies take place, is still pristine and reaches a thickness of up to 6 m (shown by test drillings). The Schechenfilz is structured into open sedge areas, heather meadows and wooded areas as illustrated in Fig. 10. The study site is situated in the wooded area, dominated by a small bog-pine (*Pinus mugo spp. Rotundata*) that is adapted to wetlands.

Fig. 10 Left: picture shows sedges and heathers in front of the bog-pine forest edge and the Alps in the background; right: bog-pine forest and the two eddy covariance measurement towers (source: IMK-IFU).

Since the study focuses on carbon exchange, and because vegetation, peat conditions (Tab. 3), as well as water table depth (i.e. the most important site factors for carbon exchange) corresponds to typical pristine bog characteristics, we consider the status of this site to be natural, in spite of previous human activity on the peatland. At the Schechenfilz site the tree population shows a high variability in stocking density, age and canopy height. The age of population ranges between young saplings up to 150 year old trees. The canopy height is on average 2 m, with a maximum tree height of 5 m. The estimated tree biomass at this site is estimated at 30 t ha^{-1} (S. Röhling, personal communication, 2012) and the average plant area index (PAI) of the bog-pines is 2.3 ±0.8 (see Section 2.2). The ground layer vegetation is dominantly formed by peat mosses (*Sphagnum spec*.) with additions of heather (*Calluna vulgaris* (L.)), bog bilberry shrubs (*Vaccinium uliginosum* (L.s.l)), and several species of the sedge-family (*Cyperacaea*, mainly *Eriophorum vaginatum* (L.), hare's-tail cottongrass). As in all such bog forests, the distribution of vegetation is quite heterogeneous, but the heterogeneity occurs at scales much smaller than the expected flux measurement footprint (see Section 2.6.1).

Dominant tree species: *Pinus mugo ssp. rotundata* (bog-pine)

Pinus mugo spp. rotundata (LINK) is a small, shrub-like, rare bog-pine that belongs to the group of *Pinus mugo* (*Turra sensu lato*) (Oberdorfer, 1990). *Pinus mugo spp. rotundata* is endemic to central Europe. Its range of distribution is limited to the Alps and pre-Alps, Bavarian Forest, Black Forest and the mountain region "Erzgebirge" (Jalas et al., 1973). These bog-pines grow mostly on water-logged, acidic and nutrient poor peat-soils: areas where these slow growing and light demanding species can resist the competitive pressure of other trees (Schmid et al., 1995). They grow at the limit of survivability of wood (concerning nutrient supply, acidity and water-logged conditions). The photosynthetic activity of these bog-pines is sensitive to cold temperatures, which leads to a shorter growing activity compared to other coniferous species e.g., spruces (von Sengbusch, 2002).

Wind conditions

In Schechenfilz the daily wind conditions (Fig. 11) are strongly influenced by the lake breeze system (e.g., Hupfer et al., 2005) of the Lake Starnberg to the north of the research site (see Fig. 9). Thus, during daytime, the wind blows mainly from the lake Starnberg with a prevailing direction between 340 ° and 350 °. At night, when the land surface cools down faster than the water of lake Starnberg the wind turns to a prevailing wind direction of around 135 ° at night. The mean annual horizontal wind speed amounts to 1.3 m s^{-1}.

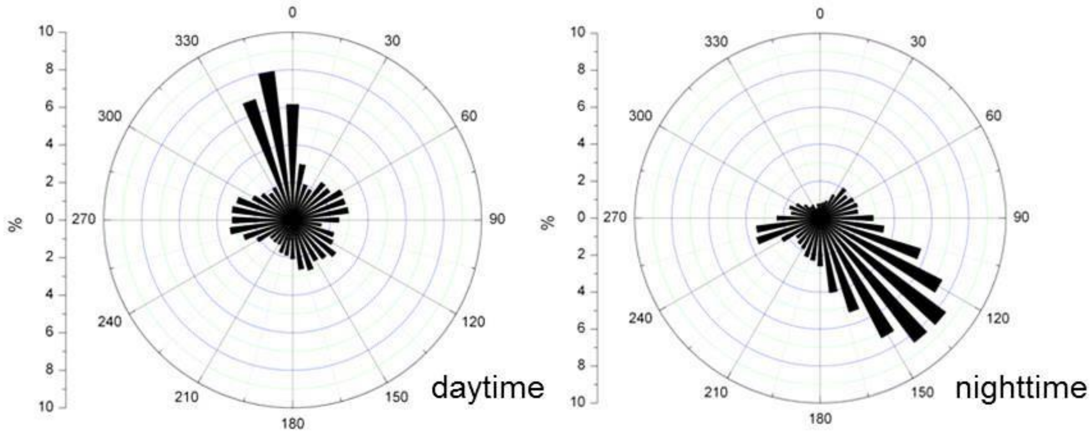

Fig. 11 Wind roses illustrate the prevailing wind direction at the Schechenfilz site during daytime (global radiation > 20 W m^{-2}; left) and during nighttime (global radiation < 20 W m^{2}; right); wind data were measured at the top of a 6 m high tower. Data presented are from the year 2011.

Soil conditions

The analyses of one soil profile indicate pristine peat conditions (N. Rosskopf, personal communication, 2012). Overall, the peat is very acidic, indicated by the low pH-value of about 4 (Tab. 3). The upper 12 cm are only temporarily water-saturated, and plants are only very weakly decomposed, so that their structure is easily discernible. The underlying continuously water-saturated layer extends to a thickness of almost 5 m (from 0.12-5.10 m below surface level). Physical and chemical analyses of a soil profile show a high carbon content of about 50% and pH-values of about 4, indicating the acid environment typical for peat bogs. The C/N ratio (45 ±13) varies within the different layers but shows a consistently low nutrient supply.

Tab. 3 Soil characteristics and element concentrations of the different peat layers at Schechenfilz: HD: Humification degree (after von Post ,1922) C_{org}: organic carbon, N: nitrogen, S: sulfur, Ca: calcium, Fe: iron, P: phosphor, DM = dry matter (N. Rosskopf, personal communication, 2012).

Layer	**HD**	**C_{org}**	**N**	**S**	**Ca**	**Fe**	**P**	**pH**	**C_{org}/N**	**Dry density**
cm	--	**%/ DM**	**%/ DM**	**%/ DM**	**g/kg DM**	**g/kg DM**	**g/kg DM**	--	--	**g/cm^3**
0-2	2	47.5	1.3	0.2	3.0	1.0	0.55	3.9	37.5	0.09
2-12	1	45.2	1.1	0.2	2.1	1.1	0.46	3.9	42.0	0.09
12-110	7	55.6	1.5	0.2	0.7	0.3	0.34	4.0	37.4	0.08
110-220	8	55.1	1.1	0.1	1.3	0.4	0.23	4.2	52.7	--
220-370	7	55.3	1.1	0.1	4.0	0.3	0.24	3.5	52.6	--
370-410	8	52.4	0.8	0.1	2.4	0.3	0.18	4.0	68.8	--
410-480	6	54.5	2.3	0.4	9.1	1.0	0.21	3.5	23.5	--
480-510	6	37.9	1.9	0.6	10.6	5.7	0.61	3.9	19.6	--
510-518+	--	1.2	0.1	0.0	52.3	10.8	0.39	3.8	11.7	--

2.1.2 Mooseurach

The peatland in Mooseurach (70 ha) is part of the Weidfilz wetland (250 ha) located to the west and southern west of Königsdorf (Fig. 12).

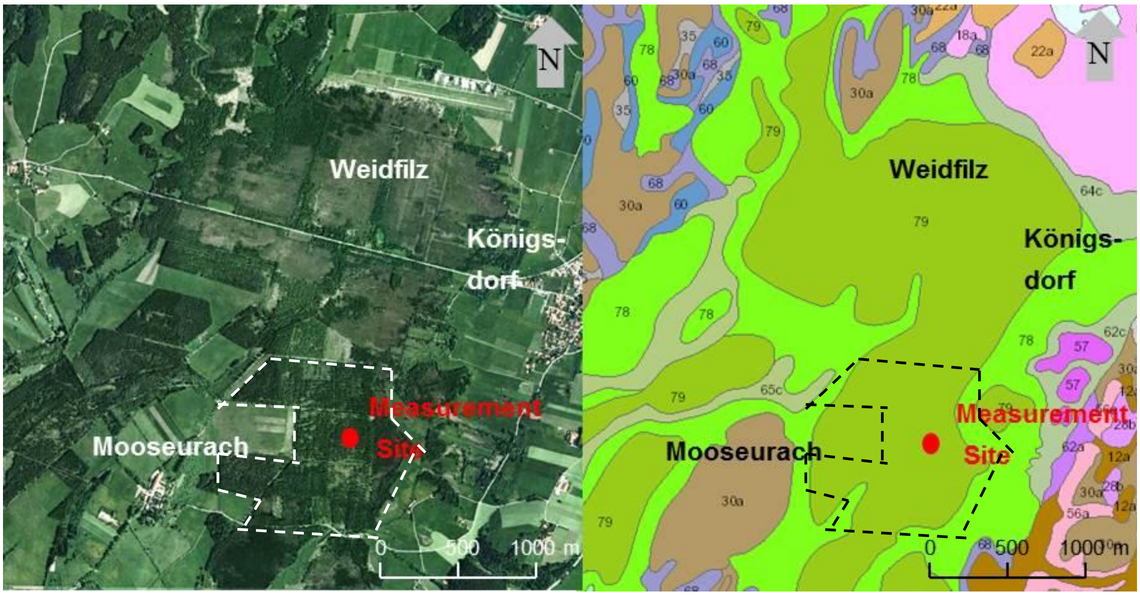

Fig. 12 Left: aerial image (based on Google Earth) of Mooseurach (target area is white/black framed); right: soil map section (Key: 79: organic soil (bog) 78: organic soil (fen) 65c: “Anmoor" (gley soil, loamy, calcareous), 30a: *Browen earth* (gravel loam, *Moraine)*, based on: Übersichtsbodenkarte (ÜBK25), source: Bayerisches Landesamt für Umwelt. Both images present the same map section.

The area was drained at the beginning of the 20th century. Initially, the research site was used for agriculture. However, due to unfavorable agricultural site conditions, such as nutrient deficiency and a still relatively high water table, agricultural use was discontinued after only a few years. Subsequently, the area was used as grassland and pastureland for a few decades. In the 1960s forestry became more important. The research site was afforested in 1967; the dominant species is Norway spruce (*Picea abies* (L.) KARST), supplemented by a small number of Scots pines (*Pinus sylvestris (L.)*). At the time of our measurements the trees were 44-45 years old and, on average, 21 m high (Fig. 13). The tree biomass of the spruces amounts to 185 t ha^{-1} (S. Röhling, personal communication, 2012), and their PAI is about 5.9 ±2.0 (see Section 2.2). The stem density ranges between 500 and 600 stems ha^{-1}. The understory vegetation consists of mosses (mostly *Sphagnum spec.*), fern, and occasional young spruces, forming a mostly closed vegetation sub-canopy layer.

After the dry summer of 2003 and the high water levels of 2009 the vitality of the spruces has been compromised. Additionally, they have been attacked by bark beetles and the ensuring weakening has made them susceptible to storm events. The periphery of the site is already characterized by several small wind throws. It is planned that the forest will be harvested within the next 10-15 years (H. Neustifter, Heimo, Amt für Ernährung, Landwirtschaft und Forsten Weilheim, personal communication, 2013).

Fig. 13 Left: picture shows the view to sub-canopy layer of the spruce forest in Mooseurach; right: view from the measurement tower above the canopy, in western direction (source: IMK-IFU).

Since 1992 same parts of the Weidfilz wetland have been under restoration, using different methods. Active restoration of the study area is not intended, but the currently still active drainage system (1 m depth, spacing of 10-15 m) is no longer maintained.

Dominant tree species *Picea abies* (Norway spruce)

The Norway spruce is a very common and fast growing coniferous tree. It is one of the economically most important tree species in northern Europe, planted within and outside its natural range (mountain regions up to 2100 m. a.s.l., German low mountain range and Scandinavian and Baltic lowland (> 100-200 m. a.s.l.). Norway spruces prefer cold and humid climate (Stinglwagner et al., 2005). Further, they prefer a moist, acidic, well-drained soil in full sun, but can adapt to just about any poor soil. Norway spruces growing on well aerated soils root up to 2 m below the surface. However, if the soil is only poorly aerated, the roots are typically limited to the top soil layer (20-30 cm). Due to this shallow rooting such spruces are particularly susceptible to damage by storms (Stinglwagner et al., 2005).

Wind condition

At Mooseurach, the prevailing wind direction is south-westerly. In addition, winds from north-eastern directions occur frequently (Fig. 14), corresponding to the topographic alignment of airflow.

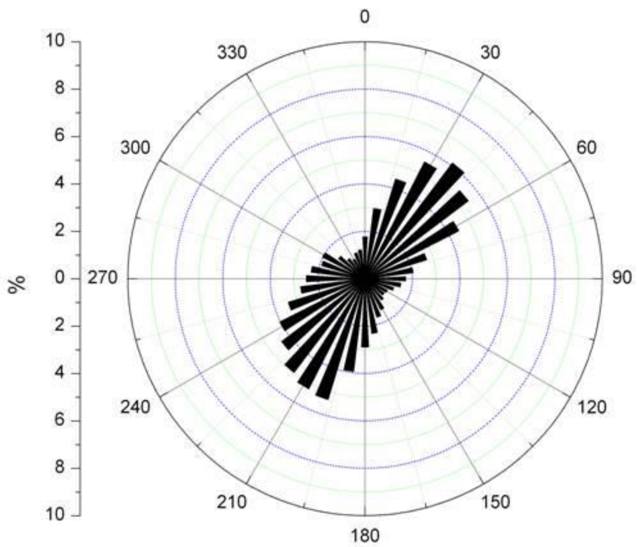

Fig. 14 Wind rose, presenting prevailing wind direction in Mooseurach, data of the year 2011.

Neither seasonal nor daily patterns of wind direction could be identified at Mooseurach. The mean horizontal wind speed is 1.5 m s^{-1}. This is only minimally higher than at Schechenfilz, likely because the wind components were measured at 30 meters height, about 7 meters above the canopy, at Mooseurach.

Soil conditions

The peat is moderately drained, and although the drainage system is no longer maintained, it is still effective. Analyses of one soil profile at Mooseurach (N. Rosskopf, personal communication, 2012, Tab. 4) demonstrate the effects of drainage: in the upper 20 cm humification of the peat is well advanced and plant structures are no longer identifiable (humification degree 10 out of 10, after von Post (1922)). Between 20 and 35 cm beneath the surface the peat is occasionally water-saturated in the course of the year, and below this is an almost 3 m thick continuously water-saturated peat layer. Below 3.4 m the soil material is mineral.

Tab. 4 Soil characteristics and element concentrations of the different peat layers at Mooseurach: HD: Humification degree (after von Post ,1922) C_{org}: organic carbon, N: nitrogen, S: sulfur, Ca: calcium, Fe: iron, P: phosphor, DM = dry matter (N. Rosskopf, personal communication, 2012).

Layer	HD	C_{org}	N	S	Ca	Fe	P	pH	C_{org}/ N	Dry density
cm	--	%/ DM	%/ DM	%/ DM	g/kg DM	g/kg DM	g/kg DM	--	--	g/cm^3
0-20	10	49.7	2.3	0.30	4.6	4.15	0.99	4.2	22.2	0.15
20-35	7	53.5	2.1	0.22	3.0	0.39	0.50	3.9	25.6	0.10
35-100	8	55.3	1.6	0.21	1.4	0.58	0.27	5.0	34.9	--
100-250	8	54.6	1.6	0.13	3.4	0.31	0.28	3.6	33.3	0.14
250-340	8	52.6	1.5	0.28	21.0	1.08	0.25	4.2	36.4	0.10
340-350+	--	6.4	0.5	0.11	305.4	0.63	0.06	6.2	13.3	--

It should be noted that the fraction of available nitrogen (N) is greater at Mooseurach than at Schechenfilz, implying a better availability of nutrients, which as indicated by the lower C/N ratio (33.7 ±3.5, Tab. 4). The drained site is also characterized by stronger humification and mineralization of the peat. A comparison of soil analyses between the two sites is presented in Fig. 15.

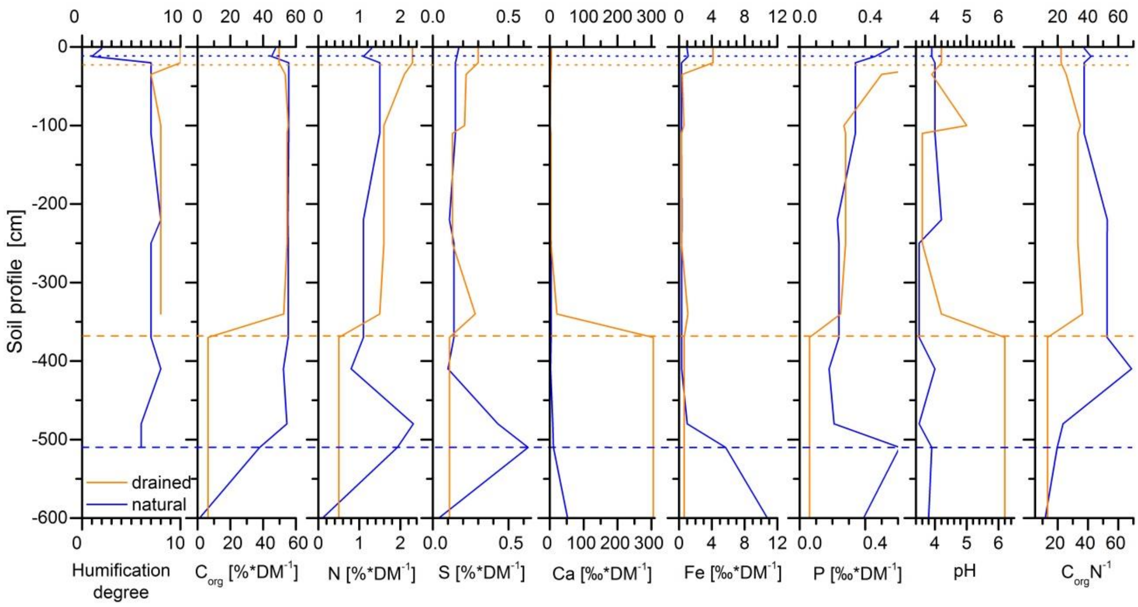

Fig. 15 Soil characteristics and element concentrations of the different peat layer at the natural and the drained site (C_{org}: organic carbon, N: nitrogen, S: sulfur, Ca: calcium, Fe: iron, P: phosphor, DM = dry matter). Dotted line indicate threshold between frequently water saturated layer and permanent water saturated layer. Dashed line indicate threshold between peat and mineral soil layer. Data are based on the analysis of one soil profile at each site.

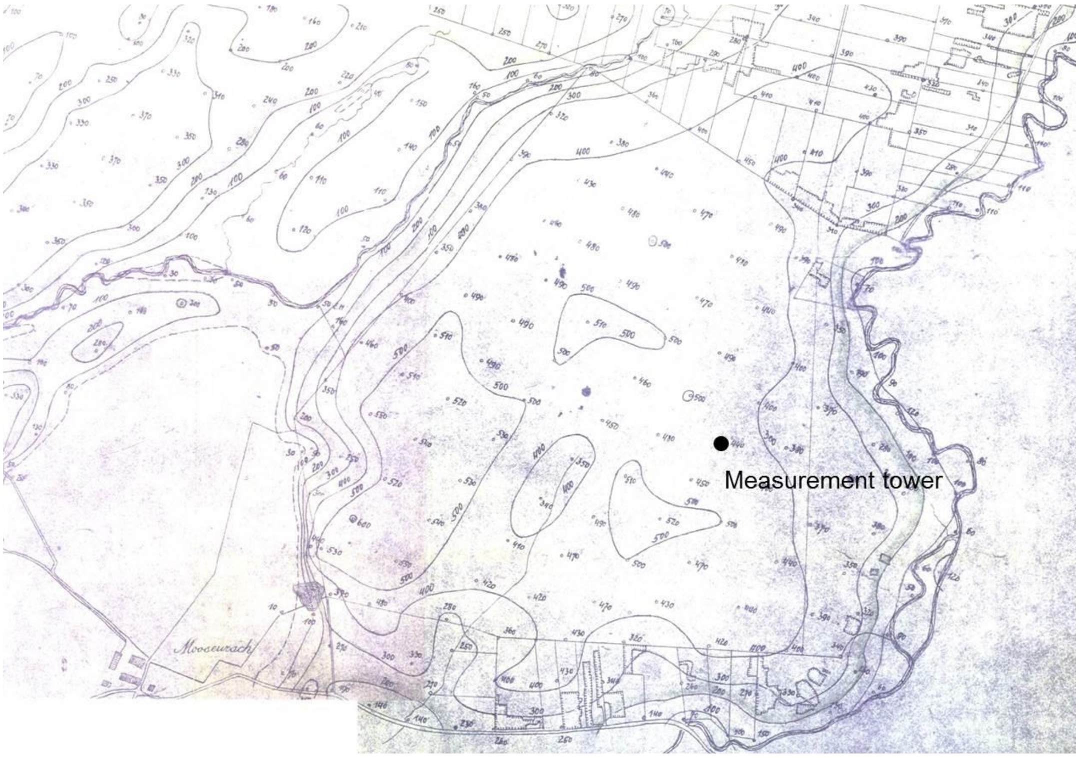

Fig. 16 Historical survey of the peat thickness in Mooseurach; author: unknown, source: Bayerische Landesanstalt für Landwirtschaft (LfL), compiled in 1930s, scale 1:5000.

Based on a gridded stratigraphic survey of peat thickness from the end of the 1930s (Fig. 16, spacing 75 m, data courtesy by District Government of Upper Bavaria), including about 70 measurements in the target area (see Fig. 12), the former peat thickness at the location of the current soil measurements was 4.4 m. This is about one meter thicker than today, illustrating the peat loss in the last 70-80 years.

2.2 Instrumentation

CO_2 exchange was measured at the two study sites using the eddy covariance technique (e.g., Aubinet et al., 2000; Baldocchi et al., 1988; Foken et al., 2012a). Due to differences in vegetation height, a 30 m radio-antenna-type tower was installed in the 21 m high drained spruce forest at Mooseurach and a 6 m tower in the bog-pine forest (mean canopy height about 2 m) at Schechenfilz (Fig. 17).

Fig. 17 Left: 30 m measurement tower in Mooseurach; right: 6 m measurement tower in Schechenfilz (source: IMK-IFU).

The towers were equipped with eddy covariance systems complemented by instruments to measure relevant auxiliary parameters. A 3-D sonic anemometer (CSAT-3, Campbell Scientific, Inc., Logan, Utah, USA) was used at each site and mounted almost at the top of the tower. The aerodynamic measurement height is 4.34 m about the zero-plain displacement level at the natural bog-pine site, and 15.6 m at the drained spruce forest. Carbon dioxide and water vapor were measured at the natural site by an open path infrared gas analyzer (IRGA, LI7500, Li-Cor, Inc., Lincoln, Nebraska, USA), while at the drained site a closed path infrared gas analyzer (LI7200, Li-Cor) was installed. The intake tube was a 1 m insulated steel line with 3/8’’ (9.525 mm) inner diameter and a flow rate of 15 L min^{-1}. The measurement principle of LI7200 is based on the absolute Non-Dispersive-Infrared (NDIR) design of the LI7500 (Burba et al., 2010b), which leads to a good agreement between the two measurement devices (see also Appendix A3). The closed path system has a considerable advantage, because rain drops and fog do not compromise the measurements of the close path system, which leads to higher data coverage (see Section 2.5).

Tab. 5 Instrumentation at the Mooseurach site and Schechenfilz site 1) Campbell Scientific, Inc., Logan, Utah, USA; 2) LI-Cor, Inc., Lincoln, Nebraska, USA; 3) Kipp and Zonen, Delft, Netherlands; 4) Vaislala, Helsinki, Finland; 5) Hukseflux, Delft, Netherlands; 6) Delta-T, Cambridge, UK; 7) Schlumberger Water Services, Delft, Netherlands)

Instrument	Mooseurach		Schechenfilz	
	Model	**Location/ Position**	**Model**	**Location/ Position**
3-D sonic anemometer	CSAT-3[1)]	29.5 m	CSAT -3[1)]	5.68 m
Infrared CO_2/H_2O analyzer	LI-7200[2)]	29.5 m	LI-7500[2)]	5.68 m
Net radiometer	CNR-4[3)]	30 m	NR-01[5)]	6 m
PAR sensor	LI-190SL[2)]	30 m, 2 m	LI-190SL[2)]	6 m
Relative humidity/ Air temperature probe	HMP45C[4)]	30 m, 2 m	HMP45C[4)]	6 m
Infrared radiometer	IR100[1)]	30 m, 2 m	IR100[1)]	4 m
PAR sensor line	LI191SL[2)]	0.5 m	--	--
Sunshine pyranometer	SPN01	30 m	--	--
Soil temperature profile	STP01SC[5)]	3 x 0.02, 0.05, 0.1, 0.2, 0.5 m below surface	STP01SC[5)] T-107[1)]	3 x 0.02, 0.05, 0.1, 0.2, 0.5 m below surface 0.1
Soil water content	CS616[1)]	3 x, 30 cm profile	CS616[1)]	3 x, 30 cm profile
Soil heat-flux plates	HFP01CS[5)]	3 x 0.05 m below surface	HFP01[5)]	1x 0.05 m below surface
Tipping bucket raingauge	52202[1)]	22 m	52202[1)]	1m
Diver gauges	Mini diver[7)]	4 pieces	Mini diver[7)]	8 pieces
Plant area index (PAI)	SunScan SS1[6)]	100 individual measurements	SunScan SS1	100 individual measurements
3-D sonic anemometer[)]	--	--	CSAT -3[1)]	6.67m
Infrared CO_2/H_2O analyzer	--	--	LI-7200[2)]	6.67 m
Laser methane analyzer	--	--	LI-7700[2)]	6.67 m

At the top of each tower, air temperature and relative humidity were measured by the HMP45C (Vaisala, Helsinki, Finland), photosynthetic active photon flux density (PPFD) by the LI 190SL (Li-Cor), the four components of the net radiation by the heated and ventilated CNR4 (Kipp and Zonen, Delft, The Netherlands). Precipitation was detected by a heated tipping bucket raingauge 52202 (Campbell Scientific) mounted about 1 m above the canopy at each site. Ground water table fluctuations were measured continuously by several mini-diver gauges (Schlumberger Water Services, Delft, The Netherlands; eight gauges at the natural and four at the drained site), distributed over the core of the bog, including the footprint area. The soil water content was detected by three water content reflectometers CS616 (Campbell Scientific), averaging the water content over the top 30 cm. For measurements of volumetric water content in organic soils the calibration coefficients of the measurement devices have to be adapted. We adopted the coefficients according to Yoshikawa et al. (2004) for organic soils consisting of dead sphagnum material. The surface radiation-temperature was measured by an infrared remote sensor IR100 (Campbell Scientific). The PAI of the trees at each site is the mean of 100 individual measurements with the SunScan Canopy Analysis System SS1 (Delta-T, Cambridge, UK). At the natural site the soil temperature at 0.1 m depth was measured by thermistor a temperature probe (type T107,Campbell Scientific) and in the drained soil at Mooseurach by soil temperature profiles STP01SC (Hukseflux, Delft, The Netherlands) in five different depths (0.02, 0.05, 0.1, 0.2, 0.5 m).

The meteorological and soil parameters were detected every 10 seconds, averaged and stored every 10 minutes to CR3000 or CR1000 (Campbell Scientific) data loggers. The instrumentation for eddy covariance measurements and associated meteorological parameters differed only slightly between the sites. All instruments for eddy covariance and environmental parameters and their location or position are listed in Tab. 5.

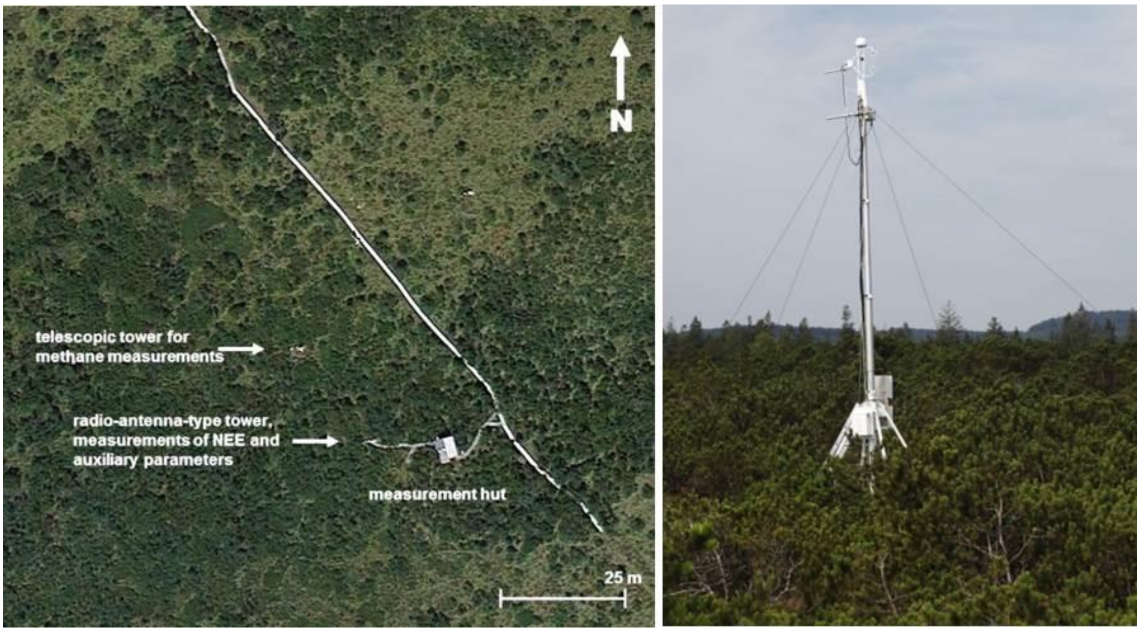

Fig. 18 Left: measurement site Schechenfilz, based on Google Maps, record date 18.08.2012. Right: telescopic mast with cloth path measurement system for CO_2 and water vapor and an open path laser for methane-flux measurements (source: IMK-IFU).

At Schechenfilz the measurement setup was complemented by methane-flux measurements, starting in July 2012 (Fig. 18). We equipped a second 6 m guyed telescopic-tower, 35 m to the northeast of the already existing CO_2 tower, where we measured methane fluxes via the open-path laser by Li-Cor (LI7700). The wind vectors were measured with CSAT-3 as well. Additionally we installed a second H_2O/CO_2 closed path analyzer (LI7200) at Schechenfilz to get the full carbon balance at the telescopic tower.

2.3 Measurement method

2.3.1 Eddy covariance method

The eddy covariance (EC) method is used for the determination of vertical turbulent fluxes of heat, momentum or trace gases (e.g., CO_2) between the biosphere and atmosphere. The eddy covariance method is described in detail is several publications, e.g., Aubinet et al. (2000), Arya (2001), Baldocchi et al. (1988), Stull (1988), and Foken (2008).

Generally, the vertical turbulent transport of a scalar *c* of a certain averaging period (over-bar) can be expressed as

$$F_c = \overline{wc}, \tag{5}$$

where:

F_c: is the vertical turbulent flux of the scalar *c* (e.g., CO_2) [e.g., µmol m^2 s^{-1}]
w: is the vertical wind component [m s^{-1}] and
c: is the concentration of the scalar [e.g., µmol m^{-3}].

Thus, a flux is the value of a scalar per unit area and per unit time. On the basis of the Reynolds decomposition the vertical wind speed of turbulent elements and the concentration of any scalar of this turbulent element can be divided in turbulent (e.g., c') and mean (e.g., $\overline{c}$) part (e.g., Arya, 2001; Baldocchi et al., 1988; Foken et al., 2012a) as presented in Fig. 19.

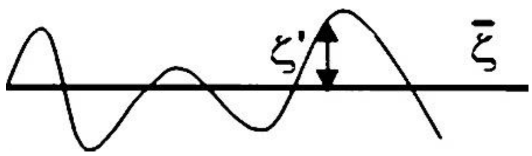

Fig. 19 Schematic presentation of Reynolds decomposition of the value ζ (Foken, 2008)

On the basis of the Reynolds decomposition the vertical exchange fluxes can be written as

$$F_c = \overline{(\overline{w}+w')(\overline{c}+c')}; \tag{6}$$

by transformation and multiplying, we get Eq. 7:

$$F_c = \overline{w}\,\overline{c} + \overline{w'}\overline{c} + \overline{w}\,\overline{c'} + \overline{w'c'}. \tag{7}$$

Equation 7 can be simplified because, by definition $\overline{w'}$ and $\overline{c'}$ are equal to zero, make the terms $\overline{w'}\overline{c}$ and $\overline{w}\,\overline{c'}$ negligible. In addition, there is no vertical mass movement in the air, thus the mean vertical wind speed $\overline{w}$ has to be zero over a suitable time interval (e.g., 30 min), such that $\overline{w}\,\overline{c}$ vanishes as well. The remaining eddy flux $\overline{w'c'}$ involves the mean of a product of two fluctuating components and rarely equals zero (Moncrieff et al., 1997).

The eddy-flux of a certain averaging period of any scalar c can also be expressed as:

$$F_c = \overline{w'c'} = \frac{1}{n}\sum_{i=1}^{n}\left[\left(c_i - \overline{c}\right)\cdot\left(w_i - \overline{w}\right)\right], \qquad \textbf{(8)}$$

where the number n depends on the measurement frequency (usually 10-20 Hz) and length of the averaging period (commonly 30-60 min).

For a valid application of the eddy covariance method (i.e. that $\overline{w'}\equiv0$, $\overline{c'}\equiv0$, and $\overline{w}\equiv0$), some assumptions need to be considered (e.g., Burba et al., 2010a; Foken, 2008; Stull, 1988).

- The atmospheric turbulence has to be well developed.
- Local steady state conditions of the vertical wind component w and the scalar c during the 30-60 minutes averaging interval (Foken and Wichura, 1996).
- Horizontal homogeneity of the underlying surface ensuring that all sensors are located within the same footprint (horizontal exchange processes can thus be neglected and only the vertical transport has to be considered)
- Taylor hypothesis (turbulence elements pass the sensors as “frozen” elements, i.e. the horizontal wind speed translates the turbulence elements as a function of time to their corresponding measurement in space (Stull, 1988)).
- Air density, flow convergence and divergence are negligible.
- The use of fast-response sensors to quantify the respective scalar (temporal resolution of 10-20 Hz). Using slower sensors, the turbulent components are filtered by the instrument’s response, resulting in incorrect fluxes (Stull 1988).

The EC method is widely used to estimate biosphere-atmosphere exchange fluxes of momentum, heat, water vapor, CO_2, methane and other trace gases. The method grew in popularity in the last decades due to some important advantages (Baldocchi, 2003): the EC method enables the investigation of the net ecosystem exchange (NEE) of a whole ecosystem, without influencing the studied site. Additionally, the technique is capable of measuring trace gas exchange across a range of time scales of a few hours up to decades (e.g., Baldocchi and Wilson, 2001; Dunn et al., 2007; Gough et al., 2008). The measurement of the turbulent fluxes by the eddy covariance method is possible for all mass-specific air components, if an adequate measurement instrument is available. In recent years, appropriate instruments have been developed for air constituents like CO_2, water vapor and methane.

2.3.2 Differential conservation equation

The conservation equation of a scalar describes the atmospheric transport processes of heat and trace gases. Following e.g., Aubinet et al. (2000) the conservation equation of a scalar *c* is:

$$\frac{\partial c}{\partial t}+u\frac{\partial c}{\partial x}+v\frac{\partial c}{\partial y}+w\frac{\partial c}{\partial z}=Q_c+D \quad , \tag{9}$$

where	u:	horizontal wind component
	v:	lateral wind component
	w:	vertical wind component
	x:	the mean wind direction
	y:	the lateral wind direction
	z:	the vertical wind direction
	Q_c:	the source/ sink strength of c and
	D:	the molecular diffusion.

After the application of the Reynolds decomposition of u, v, w, and c, the integration along z between the surface and the measurement height (z_m), and the assumption of no horizontal divergence of the eddy flux (Aubinet et al., 2000) the conservation equation can be written as

$$\underbrace{\int_0^{z_m} Q_c\, dz}_{\mathrm{I}}=\underbrace{\overline{w'c'}}_{\mathrm{II}}+\underbrace{\int_0^{z_m} Q_c \frac{\partial \overline{c}}{\partial t}dz}_{\mathrm{III}}+\underbrace{\int_0^{z_m} \overline{u}\,\frac{\partial \overline{c}}{\partial x}dz}_{\mathrm{IV}}+\underbrace{\int_0^{z_m} \overline{w}\,\frac{\partial \overline{c}}{\partial z}}_{\mathrm{V}} \quad . \tag{10}$$

The term on the left hand side (I) represents the source-/sink-strength of a scalar (e.g., the NEE for CO_2 or the NEMP for CH_4). The second term (II) is the eddy flux at the measured height (e.g., F_c) and term three (III) represents the storage term (S_c). The last two terms (IV and V) are fluxes generated by horizontal and vertical advection (V_c). All terms are schematically represented in Fig. 20.

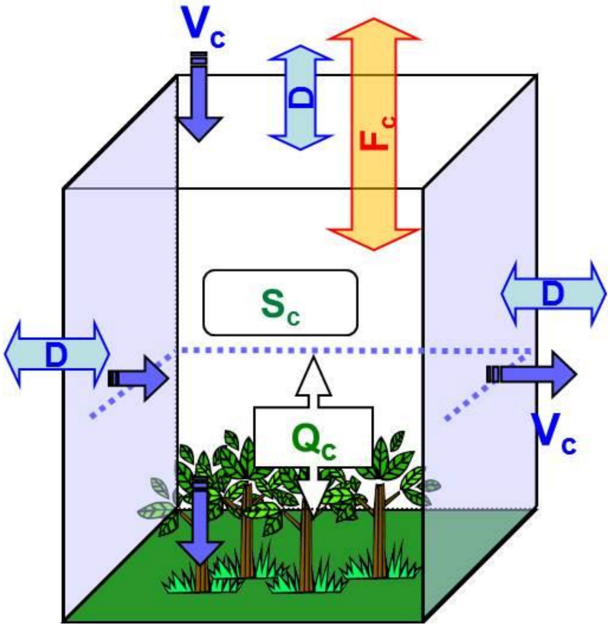

Fig. 20 Schematic representation of transport processes in a control volume over a forest site (modified after H.P. Schmid (unpublished)). V_c presents vertical and horizontal advection, D is molecular diffusion, S_c is the storage of *c* (CO_2), Q_c presents the source and sink of c and F_c is the turbulent exchange flux of c.

Overall the conversation equation of CO_2 can be simplified as

$$NEE = F_c + S_c + V_c \,. \qquad \textbf{(11)}$$

In theory, eddy covariance measurements require homogeneous terrain, thus the advection term should be zero. However, in real world, and in particular over tall vegetation, the advection term could be very important during calm nights (Lee, 1998). To date, it is not possible to quantify advection, but regular occurring mismatch between NEE and F_c and S_c in Eq. 11, as well as the lack of energy balance closure at many sites (e.g., Stoy et al., 2013) point to a non-negligible advection term. In contrast, the storage term cancels for timescales larger than a few days (e.g., Aubinet et al., 2012). Therefore, the NEE calculation is based mostly only on the covariance between the fluctuations of the vertical wind velocity and the concentration of the scalar (Eq. 8).

2.4 Flux calculation and processing of CO_2 and CH_4 fluxes

Calculation of half-hourly CO_2 fluxes for the CO_2 site comparison (July 2010-June 2012) have been performed with the software package TK 3.1 (Mauder and Foken, 2011). The CO_2 and CH_4 fluxes between July 2012 and June 2013 were performed by the data processing package EddyPro, version 4.1 (http://www.licor.com/env/products/eddy_covariance/software.html, last access: 23.11.2013). Both involve all important and commonly applied proceeding steps. Following corrections and methods were used:

- Time-lag compensation by covariance maximization
- Spike detection of raw data after Vickers and Mahrt (1997) based on Højstrup (1993); we used a window size of 15 values, a group of more than four spikes are not flagged as those, values exceeding mean ±4.5σ are spikes.
- Compensation of density fluctuations for open path measurement systems (WPL-adjustment after Webb et al. (1980)). Fluxes based on the enclosed-path IRGA LI7200 were derived from mixing ratios (relative to dry air) making WPL adjustment obsolete.
- In TK3.1 corrections of spectral loss were performed (high frequency) following Moore (1986) using the spectral model by Kaimal (1972) and Højstrup (1981). Eddy-Pro performed the spectral correction after Moncrieff et al. (2004) (low frequency) and Moncrieff et al. (1997) (high-frequency).
- To ensure that the mean vertical wind speed is zero we used the planar fit method for coordinate transformation after Wilczak et al. (2001) for the site comparison between 2010-2012. One regression plain was determined semiannually to ensure zero mean vertical wind speed for this time period at each site. In the following period (2012-2013) we used the double rotation method for the methane study. By this method the vertical wind velocity is nullified for each half hour value (Kaimal et al., 1994).
- Quality assessment, applying tests for steady state conditions and well-developed turbulence (integral turbulence characteristics) after Foken and Wichura (1996) in the version proposed by (Foken et al., 2004).

2.4.1 Footprint

Eddy covariance is a micrometeorological point measurement method, typically carried out on the top of a tower at a certain distance above the canopy, which depends on canopy height (see Section 2.2). The surface source area of the measured trace gases is called footprint (Fig. 21). The size and location of the footprint area depends on several factors, which are mostly the horizontal wind speed and wind direction, the stability of the atmosphere (e.g., Monin Obukov Length (L), the friction velocity (u_*)), and the measurement height above canopy (e.g., Schmid, 2002).

For the patchy site Mooseurach, the analytical footprint model of Kormann and Meixner (2001) was used as a rejection criterion to identify fluxes, where less than 70% of the weighted footprint lies inside the target area (Section 2.6.1).

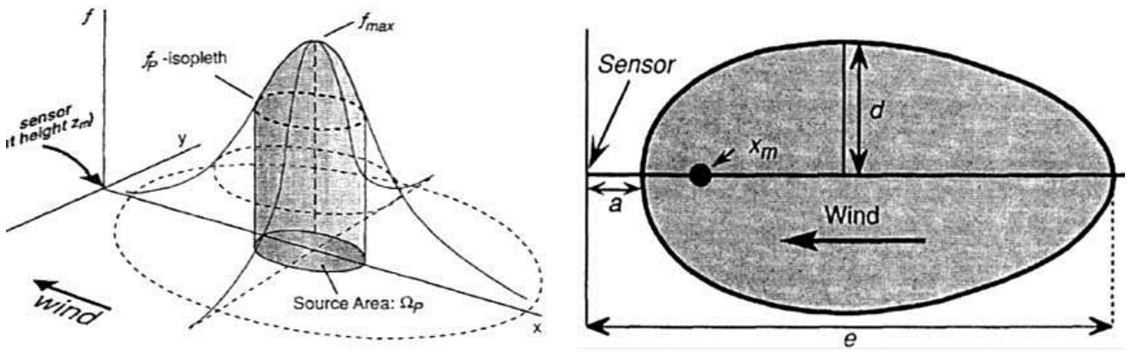

Fig. 21 Left: source area and its relation to the footprint- or source weight function. The source weight is small for small separation distances. It will rise to a maximum with increasing distance and then fall off again to all sides as the separation is further increased. Horizontally homogeneous turbulence is assumed, with the mean wind direction parallel but counter to the x-axis direction. Right: characteristic dimensions of the source area. xm: maximum source location (upwind distance of the surface element with the maximal influence on a given sensor); a: near end; e: far end; and d: maximum lateral half-width of the source area (from Schmid (1994))

Stable atmospheric conditions and low wind speed lead to a large footprint up to several kilometers long. During unstable conditions, the footprint is much smaller and the maximum of the source function is larger. Particularly, flux measurements in patchy, inhomogeneous terrain need to consider the footprint dimensions, in order to identify measurements that are influenced by flux contributions outside the target area (Schmid, 1997).

2.5 Data coverage

2.5.1 Data coverage of CO_2 fluxes for two years site comparison (2010-2012)

Eddy covariance measurements allow the detection of continuous time series although data gaps occur for various reasons. Short gaps occurred sporadically, due to sensor malfunction, in particular the open path CO_2/H_2O analyzer (Schechenfilz) is prove to such gaps during wet and foggy conditions. At Mooseurach, four long data gaps occurred on April 12 -April 19, 2011; July 10-July 25, 2011; November 15 - November 26, 2011, and February 23-March 27, 2012. In Schechenfilz three long gaps happened on October 23 -October 26, 2010; April 28 – May 04, 2011, and May 22- May 27, 2011. These longer data gaps were caused by power failure or problems of data storage. The raw data coverage of half-hourly flux measurements was 91% at Mooseurach and 71% at Schechenfilz. Subsequently, these data were screened to ensure good quality according to three rejection criteria (Section 2.6).

2.5.2 Data coverage of CH_4 fluxes

Methane and CO_2 fluxes were continuously measured at the natural bog-pine site Schechenfilz over 15 months between July 2012 and September 2013. A failure of the sonic anemometer caused a data gap of seven weeks between May 15, 2013 and July 04, 2013.

Measurement failures, due to a blocked signal path, are identified by a “Received Signal Strength Indicator (RSSI)” that was less than 20%. Data coverage of methane fluxes is about 64% before the application of quality assurance rejection criteria.

2.6 Quality assurance

2.6.1 Footprint criterion

The analytical footprint model by Kormann and Meixner (2001) was used to estimate how well the measured fluxes reflected sources and sinks of the bog forest (Fig. 22). If less than 70% of the 30-min flux value originated from the area of interest, the data were excluded from further analysis.

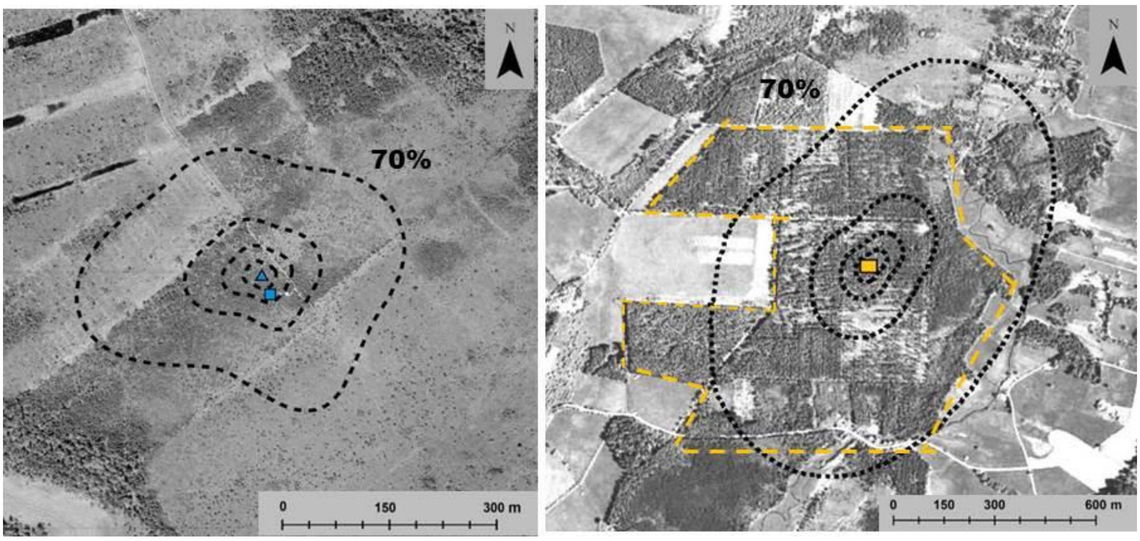

Fig. 22 Aerial images of the Schechenfilz site (left) and Mooseurach (right) (based on Google Maps). Left: dark gray: bog-pines; light gray: open bog area (grassland (sedges-family) and dwarf shrubs); blue square marks the main eddy covariance tower (CO_2 and H_2O fluxes and auxiliary environmental parameters (Section 3.1). The blue triangle marks a second tower for the CH_4 study (CH_4, CO_2 and H_2O, Section 3.3). Right: orange dashed line: target area; orange square marks the measurement tower position. Orange square marks the eddy covariance tower for the CO_2 comparison study. In both images: black dashed lines: denote footprint isolines (10%, 30%, 50% and 70%, estimated according to Kormann and Meixner (2001)) of one year (2011).

In Mooseurach 21% of the flux measurements did not originate from the area of interest. At the Schechenfilz site, the footprint is smaller due to lower measurement and canopy height, and additionally the forest edge is smoother due to highly variable tree density. The site is very heterogeneous with not definable target area (Fig. 8a). For these reasons we decide to neglect the footprint criterion for the Schechenfilz site.

2.6.2 Turbulence criterion (u_*-criterion)

An important error source of the determination of NEE or NEMP by eddy covariance is the underestimation of nighttime fluxes, due to low turbulent atmospheric mixing. During such situations, emitted CO_2 or CH_4 can accumulate near the surface and be advected away without reaching the measurement system. The friction velocity (u_*) was used to determine time periods, when flux measurements were likely unreliable as a result of low turbulent conditions. We determined a lower u_*-limit of 0.225 m s^{-1} and of 0.125 m s^{-1} in Mooseurach and Schechenfilz, respectively (Fig. 23), following Goulden et al. (1996). The lower limit of friction velocity depends on surface roughness and therefore on vegetation type.

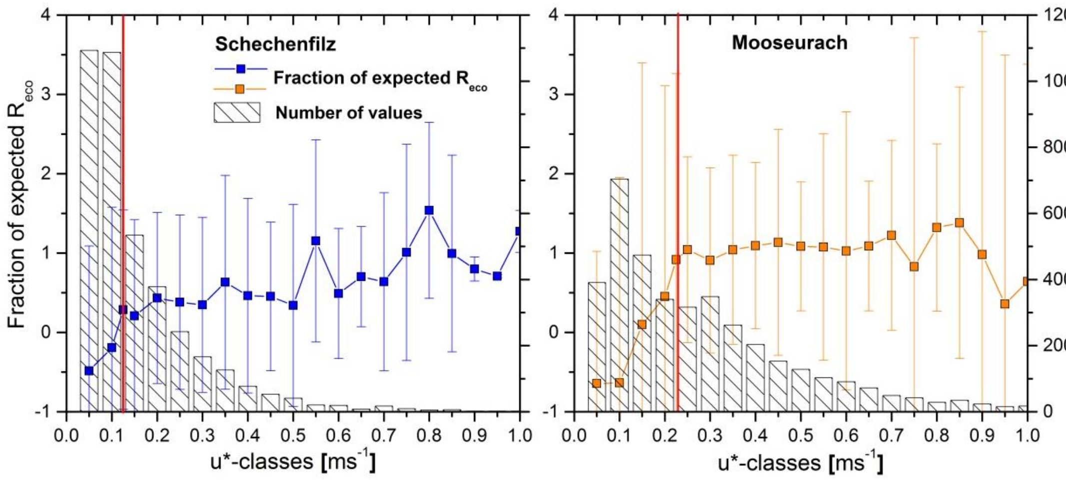

Fig. 23 Binned values of the fraction of expected nighttime ecosystem respiration versus friction velocity. The expected ecosystem respiration is modeled by temperature dependence. Red line indicates u_*-threshold, whereby fluxes are independent of friction velocity. Bars are the number of values within the u_*-bins (data exemplary shown for 2011). Vertical bars denote the standard deviation of each bin.

Fluxes measured at fiction velocities larger than the threshold are independent of this turbulence parameter, because the atmospheric mixing is sufficient, implying that most of the trace gas exchange reaches the measurement height. Seasonal variation of the u_*-threshold is expected for ecosystems which have a changing canopy height throughout the year (Aubinet et al., 2012), e.g., for crop or grassland sites. We did not find any variation of the u_*-threshold within the measurement period at either site. Thus, the determined thresholds were applied to the whole dataset. At Schechenfilz, the same u_*-threshold was adopted on the methane fluxes. At both sites, about 30% of the flux data (CO_2 and CH_4) had to be rejected by this criterion, particularly at night.

2.6.3 Outlier test

Finally, an outlier test was applied to filter out unrealistic singular values. Outliers are due e.g., bio-physical (e.g., fast changes in turbulence conditions) or instrumental reasons (water drops in the sonic anemometer or open-path IRGA, Papale et al. (2006)). Commonly, such outliers do not affect the amount of the annual net exchange but can compromise the quality of the gap-filling models. We separated the dataset into daytime (> 20 W m^{-2} global radiation) and nighttime values (< 20 W m^{-2} global radiation) and used a moving window with a width of ±14 half-hourly values. For each moving window we calculated the average and the 99% confidence interval of this average. Each 30-min flux value within the window had to be within the 99% confidence interval, otherwise the value was excluded from further calculations. This conservative outlier test rejected only 1% of the data at Mooseurach and 2% at Schechenfilz. Considering missing values (28% and 9% the natural and the drained site, respectively) and data rejection based on strict criteria (32% and 52%) leads to a high quality dataset covering about 40% of all date at both sites.

For methane gap-filling and studies of relations between methane and environmental parameters, we considered a range between -0.07 and +0.13 µmol m^{-2} s^{-1} (Fig. 24) which covers more than 99% of all CH_4 fluxes.

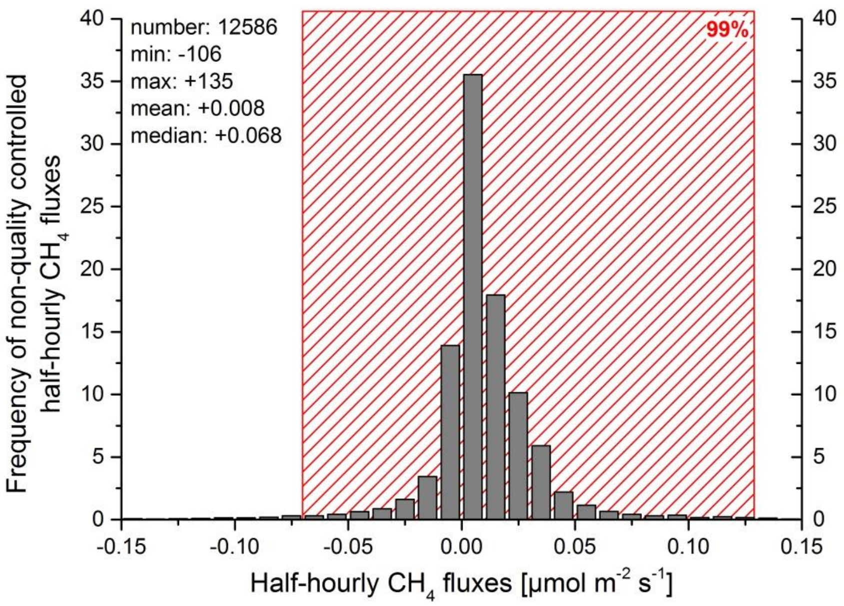

Fig. 24 Range and frequency of half hourly non-quality controlled CH_4 fluxes, whereas 99% of all fluxes range between -0.07 and +0.13 µmol m^{-2} s^{-1} (red hatched box). CH_4 fluxes are bins of 0.01 µmol $m^{-2}s^{-1}$. After quality control most negative fluxes were rejected.

Following the implementations of all criteria described above, 38% of all methane data are remaining at Schechenfilz site. This percentage is comparable to other CH_4-studies, using the eddy-covariance method (e.g., Olson et al., 2013; Rinne et al., 2007)

2.7 Gap-filling methods

For analyses of seasonal or annual patters of NEMP and NEE, or differences in the CO_2 balance between both sites, continuous sets of flux values are needed. Many different approaches and methods have been published (e.g., Falge et al., 2001; Hui et al., 2004; Moffat et al., 2007; Papale, 2012; Reichstein et al., 2005b; Ruppert et al., 2006). In this study, we consider three different empirical gap-filling methods for CO_2: the examined gap-filling methods for CO_2 are: look-up table (LUT), mean daily variation (MDV) and non-linear regression (NLR), based on the publications of Falge et al. (2001) and Reichstein et al. (2005b). The same methods were tested for CH_4-gap-filling, but the NLR and LUT method had to be adjusted, due to different environmental drivers.

Additionally, continuous time series of the drivers of CO_2 and CH_4 are needed to fill or analyze the NEE and NEMP data. Gaps within the meteorological parameters up to 2 hours were filled by linear regression. For longer periods we used the strong linear correlation between the meteorological parameters of the other site (Section 3.1.1, Fig. 25).

2.7.1 Mean daily variation (MDV)

Using the mean daily variation method, missing half hourly fluxes are replaced by the averaged value of the adjacent days at exactly that time of day. This technique assumes a temporal autocorrelation of the fluxes (Falge et al., 2001). To gap-fill the data for centered moving windows of 14 days, mean daily variations were established to fill gaps within each period. For longer gaps the MDV is not suitable, since for longer periods non-linear dependence of environmental drivers could introduce large uncertainties and errors. The MDV method does not require any independent drives and is thus the only method applicable when all auxiliary environmental data are missing (Papale, 2012). This method can be applied for different scalars (e.g., CH_4 or CO_2) in the same way, but it is not transferable between sites and time periods.

2.7.2 Look-up table (LUT)

Look-up tables are site specific; they provide non-parametric estimates of NEE or NEMP based on environmental conditions, in particular temperatures and global radiation for each missing observation (Falge et al., 2001; Moffat et al., 2007). Standard look-up tables consist of fixed periods over a year with corresponding fixed intervals for the variables (Falge et al., 2001). We used a “modified look-up table”, using a moving window of different sizes. For NEE we used an online tool described by Reichstein et al. (2005b), (http://www.bgc-jena.mpg.de/~MDIwork/eddyproc/method.php, last access 15.07.2013). It uses marginal distribution sampling which is a combination of MDV and look up table. In this case similar meteorological conditions are sampled in the temporal vicinity of the gap to be filled. This “moving look-up table technique” is able to exploit the temporal autocorrelation structure of the fluxes.

In the algorithm three different situations are identified (Reichstein et al., 2005a): first, only data of direct interest, e.g., CO_2 fluxes or latent heat flux are missing; second, air temperature and vapor pressure deficit (VPD) are also not available; third, additionally radiation values are missing. In the first case missing values are replaced by the average value under similar meteorological conditions (complies when global radiation (R_{glob}), air temperature (T_{air}) and VPD do not deviate by more than 50 W m^{-2}, 2.5 °C, and 5.0 hPa, respectively). A time window of 7 days is used, however if no similar meteorological conditions could be detected, then the window is increased to 14 days. In the second case similar meteorological conditions can be proved only on the radiation deviation which is not allowed to be larger than 50 W m^{-2}, the window size will be not increased. In the third case missing values are replaced by the mean diurnal variation.

Similar to the LUT for NEE, the LUT for NEMP samples for similar meteorological conditions in the temporal vicinity of the gap to be filled. The parameters in the LUT were air temperature (2 °C bins), photosynthetic active photon flux density (PPFD) (100 µmol m^{-2} s^{-1} bins), water table depth (0.02 m bins) and the upwind reach of the 70% footprint isoline (50 m bins). The flux footprint was estimated following Kormann and Meixner (2001). The window size for the LUT was initially set to 7 days. If no LUT estimates could be identified for this window size, the window was increased successively up to 21 days. In the few cases where an LUT estimate was still not possible, gaps were filled by linear. This method was used for the gap-filling of half-hourly methane fluxes (see Section 3.3.3).

2.7.3 Non-linear regression (NLR)

For the gap-filling of CO_2 fluxes we used a non-linear regression method according to Falge et al. (2001) and Moffat et al. (2007) to model the GPP and R_{eco} components individually.

The respiration model is based on an Arrhenius-type exponential relation between nighttime CO_2 fluxes and temperature. Nighttime data were again identified using a global radiation (R_{glob}) threshold of 20 W m^{-2} (Reichstein et al., 2005b), and the temperatures used were those that provided the best fit, and consequently the lowest uncertainty of gap-filling, at each site. The temperatures thus selected were the soil temperature at a depth of 0.1 m at the natural bog-pine forest and the radiative soil-surface temperature at the drained spruce forest. The relation used follows Lloyd and Taylor (1994) :

$$R_{\text{eco, night}}=R_{\text{ref}}\times\exp^{\left[E_0\left(\frac{1}{T_{ref}-T_0}-\frac{1}{T-T_0}\right)\right]}, \qquad \textbf{(12)}$$

where R_{ref} is defined as the ecosystem respiration at a reference temperature of 283.15 K (T_{ref}), E_0 is a fitting parameter called the activation energy in K, T_0 is a constant temperature of 227.13 K and T (in K) the measured half-hourly temperatures providing the best fit. To ensure realistic relationships, we fitted the data over a wide temperature range, choosing a fitting period of 6 months. As the respiration-temperature relationship showed no response to the phenology of the vegetation, the fitting period is taken to be of acceptable length.

During the daytime (global radiation > 20 Wm^{-2}), the respiration ($R_{\text{eco, day}}$) was estimated by the same fitting parameters, determined by the nighttime respiration-temperature-relation (Eq. 12). Note that here $R_{\text{eco, day}}$ indicates ecosystem respiration determined by the nighttime relation, but using daytime temperature only. The GPP was modeled with a rectangular hyperbolic Michaelis-Menten-type function (Falge et al., 2001):

$$NEE=\frac{\alpha'\times PPFD\times GPP_{max}}{(GPP_{max}+\alpha')\times PPFD}+R_{eco}, \qquad \textbf{(13)}$$

where α´ is the apparent quantum yield, interpreted as the ecosystem light use efficiency (µmol m^{-2} s^{-1}/µmol m^{-2} s^{-1}), in this case expressed as the carbon uptake per photon of photosynthetic active photon flux density (PPFD). The fitting parameter GPP_{max} is the maximum carbon fixation rate at unlimited PPFD. The annual growing cycle of vegetation activity is strongly related to radiation and temperature variation. Based on this scheme, GPP was calculated for several sub-periods, whose lengths depended on the mean daily soil temperature at 0.1 m depth (13 sub-periods for data of Mooseurach (July 2010- June 2012), and 20 sub-periods for data of Schechenfilz (July 2010- Sep. 2013)). During transition periods in spring and autumn we used a temperature range of 2 °C, and during more stable periods in midsummer and winter 4 °C.

In the methane dataset, an exceptional long data gap of 7 weeks (see Section 2.5.2) was gap-filled by an Arrhenius-type exponential relation (non-linear regression, NLR) between mean daily air temperature and mean daily methane flux (F_{CH_4} [µmol m^{-2} s^{-1}]=0.00679× exp (0.06725×T_{air} [°C])). This was done similarly by Rinne et al. (2007).

3 Results and Discussion

3.1 Comparison of the CO_2 exchange of a drained and a natural bog forest

In this section we compared the CO_2 exchange of the natural bog-pine ecosystem and the drained spruce forest. The key issue to be addressed centers on the magnitude of the CO_2 budgets at the two sites and the critical factors that accounted for the differences between them. The results of this section are published in *Biogeosciences* (Hommeltenberg et al., 2014).

3.1.1 Meteorological conditions

The site comparison examines the assumption that meteorological drivers of CO_2 exchange like temperatures, precipitation, vapor pressure deficit (VDP) and global radiation are almost equal at both peatland sites.

Although the parameters were measured at different heights, 30 m above the surface in Mooseurach and 6 m above the surface in Schechenfilz, all parameters highly correlate (R^2 between 0.95 and 0.98, Fig. 25). The slope of the linear fit of air temperature between the sites is 1 and the offset of 1.45 °C is very small. Comparison plots of global radiation and radiative surface temperature show a larger scatter. Differences in surface temperature are due to site-specific properties, e.g., the larger tree density in Mooseurach which leads to a stronger shading of the surface and thus to a smaller range of surface temperature compared to the more exposed surface at Schechenfilz. However, overall the half-hourly variations of meteorological drivers are almost the same. Therefore, we used air temperature, radiation and precipitation values of only one site for the meteorological characterization of the two measurement years. The presented series of air temperature in Fig. 26 were detected at Schechenfilz because of the lower measurement height of 6 m and therefore better comparability with the standardized meteorological measurement height of air temperature of 2 m (e.g., as specified by the DWD “German Weather Service”). Global radiation and precipitation values were detected by the measurement system in Mooseurach. Due to the more effective heating and ventilation system of the net radiometer (model: CNR4, Campbell Scientific), we expect better data quality from this measurement (see Section 2.2). Precipitation values were also taken from Mooseurach because of much better data coverage.

Long-term measurements (1971-2011) of air temperature are available from the nearby (distance of 8.5 km and 10.5 km to Schechenfilz and Mooseurach, respectively) weather-station "Attenkam", run by the German Weather Service (DWD) (Tab. 2).

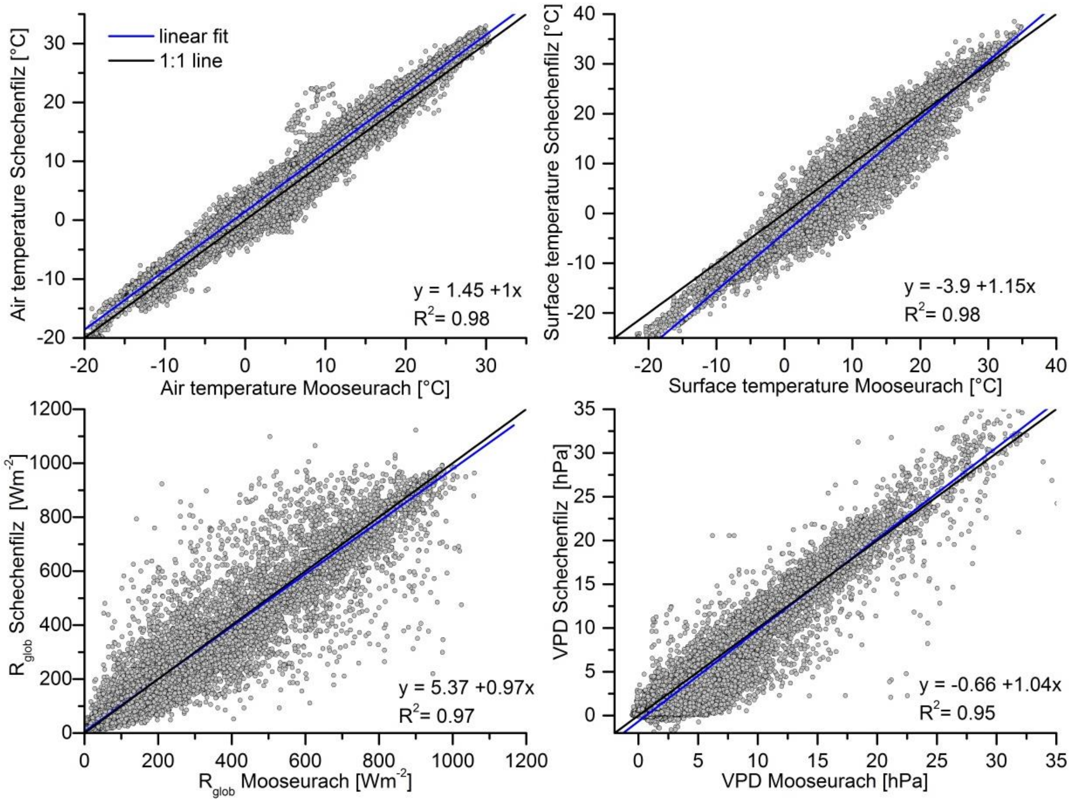

Fig. 25 Linear regressions of air temperature, radiative surface temperature, global radiation (R_{glob}) and water vapor pressure deficit (VPD) at the Mooseurach and Schechenfilz sites. Data of the two-year measurement period are presented.

However, site-related environmental factors such as soil temperature and water table (WT) depth were different (Fig. 26c-e). Following Couwenberg's (2011) classification, the natural bog Schechenfilz belongs to the class of wet peatlands, where the mean annual water table depth during the two years measurement period is above -0.2 m (-0.06 ±0.04 m). At the drained bog Mooseurach, the mean annual water table is just below -0.2 m (-0.21 ±0.08 m), thus placing Mooseurach into the category of dry peatland site. The water table was consistently higher at the natural site; in addition, water content was more stable than at the drained bog Mooseurach. Due to its intact drainage system, the water table at Mooseurach reacted more rapidly to low precipitation periods. The water holding capacity of the pristine soil is much larger than that of the drained soil (Fig. 26d/e).

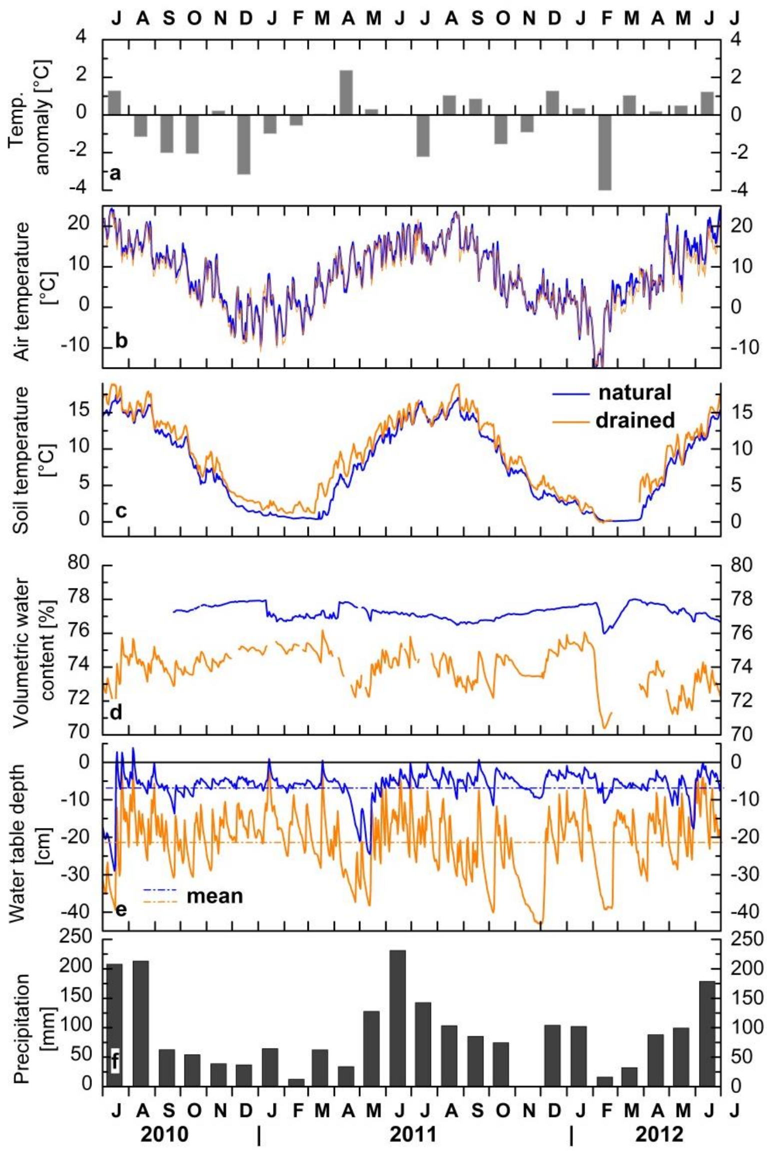

Fig. 26 Time series of daily means (b, c, d, e) and monthly values (a, f) of environmental parameters from July 2010 to June 2012. The temperature anomaly is based on 40 years of long-term data, provided by the German Weather Service (DWD). The air temperatures were measured at 6 m and 30 m height in Schechenfilz (natural) and in Mooseurach (drained), respectively. Soil temperatures were measured in 0.1 m depth. The volumetric water content is the average of the 30 cm upper peat layer. Over the two measurement years, the mean water table depth was -0.21 ±0.08 m the drained site and -0.06 ±0.04 m at the natural site.

Overall, the first measurement year was slightly cooler and wetter than the second. Within the two years, five periods of considerable water table drawdown were recorded, more prominently at the drained ecosystem (Fig. 26e); at the beginning of July 2010, during the warm and sunny spring of 2011, during the warm and relatively dry period in August and September 2011, and in November 2011 when there was no precipitation and it was sunny (more than 50% longer sunshine duration in November 2011 (142 h) compared to the long-term (92 h)), but about 1 °C cooler than the long-term average. The dry period in November led to the strongest water table drawdown (average water table in November 2011: -0.37 cm below the surface) at the drained site. Another strong water table drawdown was recorded during the extraordinary cold period in January and February 2012. At this time, the monthly air temperature was about 4 °C lower than the long-term average and we measured half hourly temperature extremes of below -20 °C. Unfortunately, soil temperatures were below the operation range of the sensors; hence the extent of the detected strong water table drawdown in February 2012 remains uncertain and will not be considered in further analyses.

3.1.2 Factors affecting CO_2 exchange

The individual component fluxes, R_{eco} and GPP, which together form NEE, depend on different environmental factors that vary throughout the year.

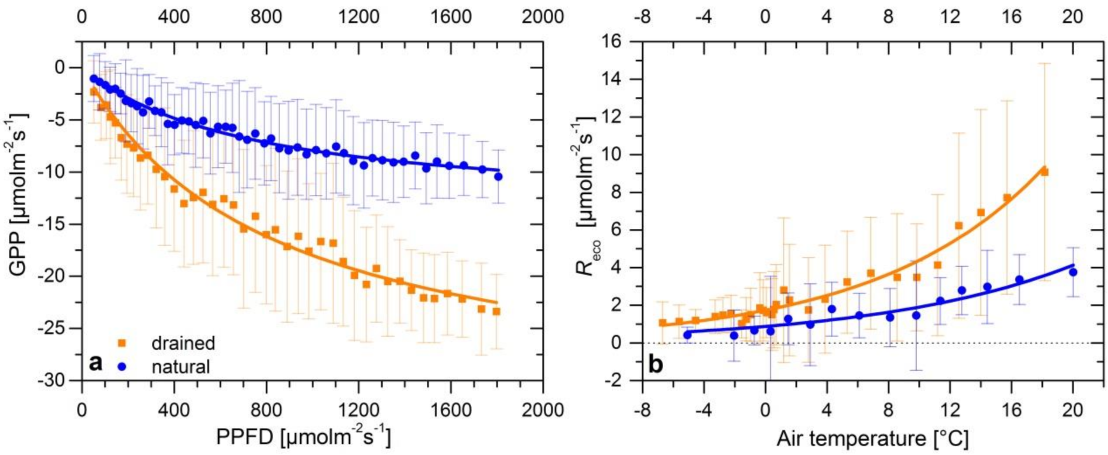

Fig. 27 a) Gross primary production (GPP: daytime fluxes-R_{eco}) versus photosynthetic active photon flux density (PPFD); b) Ecosystem respiration (R_{eco}, nighttime fluxes) versus air temperature. Each point is the average of 100 non-gap-filled half-hourly measurements. The bars denote the standard deviations. Plots show binned data from the whole year 2011. Air temperature and PPFD were measured at the top of each tower (6 m at the natural site and 30 m at the drained site).

GPP is mainly controlled by photosynthetically active photon flux density (PPFD). As expected, we detected a strong hyperbolic relationship between GPP and PPFD (Fig. 27a) at both sites. Light use efficiency (see Section 2.7.3), as well as maximum GPP (i.e. at infinite PPFD), was larger at Mooseurach (drained) than at Schechenfilz (natural), resulting in a two times larger carbon uptake at the drained site. However, the higher carbon fixation rate is probably attributable to the larger biomass at the drained site (see Section 3.1.3).

Respiration increases with increasing temperature (Fig. 27b). For the whole temperature range, respiration was greater at the drained than at the natural forest. At the reference temperature of 10°C, the respiration of the drained ecosystem was about two-times larger.

Furthermore, we tested for dependences between changes in volumetric water content (VWC) and nighttime respiration (Fig. 28) for Mooseurach.

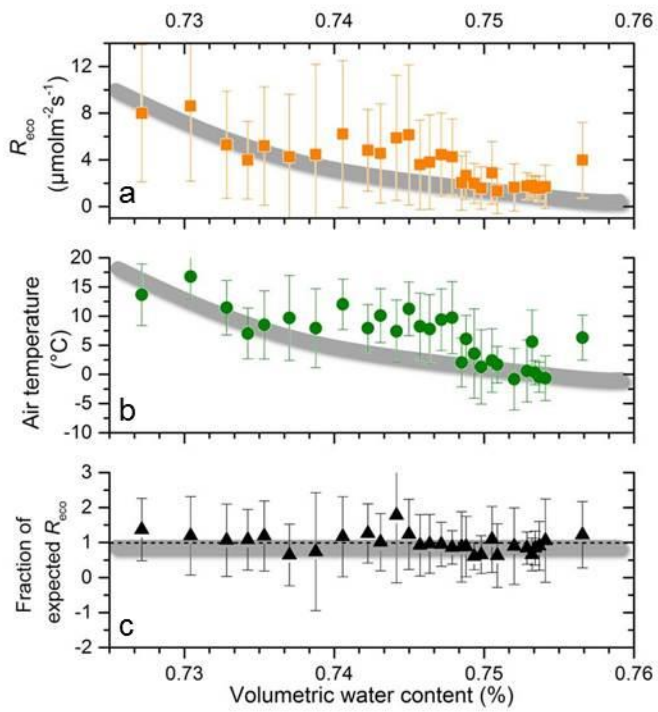

Fig. 28 Relation between VWC and R_{eco} (non-gap-filled), air temperature and the fraction of expected R_{eco} (measured R_{eco}/ modeled R_{eco} by temperature relation). Each point is the average of 100 non-gap-filled half-hourly measurements. The vertical bars denote the standard deviations. The grey line indicates an exponential dependence (non-linear regression). The dependence disappears when R_{eco} is normalized by the temperature dependent Arrhenius model for R_{eco} lowest panel) Plots show binned data from 2011 at Mooseurach.

The slope of R_{eco} in Fig. 28a suggests a moderate exponential relation between R_{eco} and volumetric water content. However, VWC is also linked to air temperature. After normalization of R_{eco} with air temperature using the Arrhenius relations of Fig. 27b, to exclude the influence of air temperature on R_{eco}, the dependence between R_{eco} and water content disappeared Fig. 28c. The same was found for to the data from the natural site (not shown). Persistent differences in VWC resulted in marked differences in respiration at the two sites, but short-term fluctuations could be explained by the air temperature dependence.

3.1.3 Annual CO_2 exchange

Annual budgets of NEE, GPP and R_{eco} were calculated for the observation periods from July 2010 to June 2011, and from July 2011 to June 2012. The estimation of annual budgets is based on continuous datasets. For gap-filling we followed the procedure described in Section 2.7.3.

Despite similarities in weather conditions and geological origin, the carbon budgets of the drained and the natural peatland were considerably different. The individual budgets of GPP and R_{eco} for the whole annual cycle show that respiration, as well as GPP, were approximately two times larger at Mooseurach (drained) than at Schechenfilz (natural), Fig. 29.

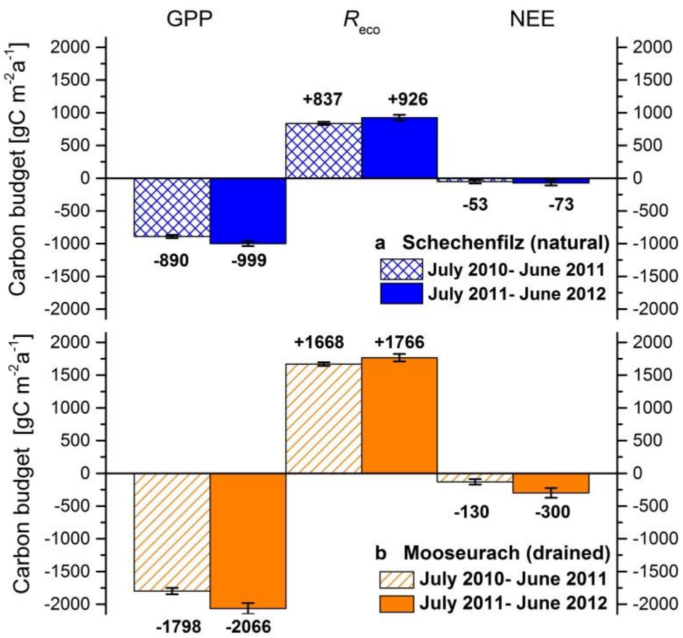

Fig. 29 Annual sums of NEE, R_{eco} and GPP at (a) the natural site Schechenfilz and (b) the drained site Mooseurach for the two measurement periods. Vertical bars indicate the uncertainty of the budgets (see Section 3.1.6)

However, after normalizing the GPP with the corresponding PAI, such that GPP_{norm}=GPP/PAI (PAI was 5.9 at the drained site and 2.3 at the natural site), we found very similar light use efficiencies and GPP_{norm} of a similar magnitude (-387 and -434 g C m^{-2} a^{-1} at the natural and -305 and -350 g C m^{-2} a^{-1} at the drained forest) at both sites and for both analyzed periods, respectively. Note: GPP_{norm} has the same units as PAI is dimensionless. Following the notion that more biomass usually produces more respiration, we normalized R_{eco} with PAI, analogously to GPP_{norm}. This resulted in similar magnitudes of normalized emission rates as well ($R_{eco,norm}$ was +283 and +299 g C m^{-2} a^{-1} at the drained forest and +364 and +403 g C m^{-2} a^{-1} at the natural pine-bog, respectively) between the sites for both analyzed periods. Nevertheless, respiration processes are very complex and cannot be solely attributed to PAI.

The NEE indicates stronger CO_2 uptake at the drained site. At both sites the uptake was smaller in the first, slightly wetter and colder measurement period from July 2010 to June 2011 (-130 ±31 g C m^{-2} a^{-1} at the drained, and -53 ±28 g C m^{-2} a^{-1} at the natural site, see Section 3.1.6 for methods to determine uncertainty) than in the second measurement period July 2011 to June 2012 (-300 ±66 g C m^{-2} a^{-1} at the drained and -73 ±38 g C m^{-2} a^{-1} at the natural site). Depending on the start of the annual averaging period, the annual NEE is highly variable. This is more pronounced at the drained spruce forest Mooseurach (range between -80 and -300 g C m^{-2} a^{-1}, Fig. 30a). At the natural site the range of NEE is noticeably smaller; it ranges between -33 and -73 g C m^{-2} a^{-1}. On average, the observed CO_2 uptake at the drained site (-157 ±36 g C m^{-2} a^{-1}) was three times larger than that at the natural site (-55 ±23 g C m^{-2} a^{-1}, Fig. 30b).

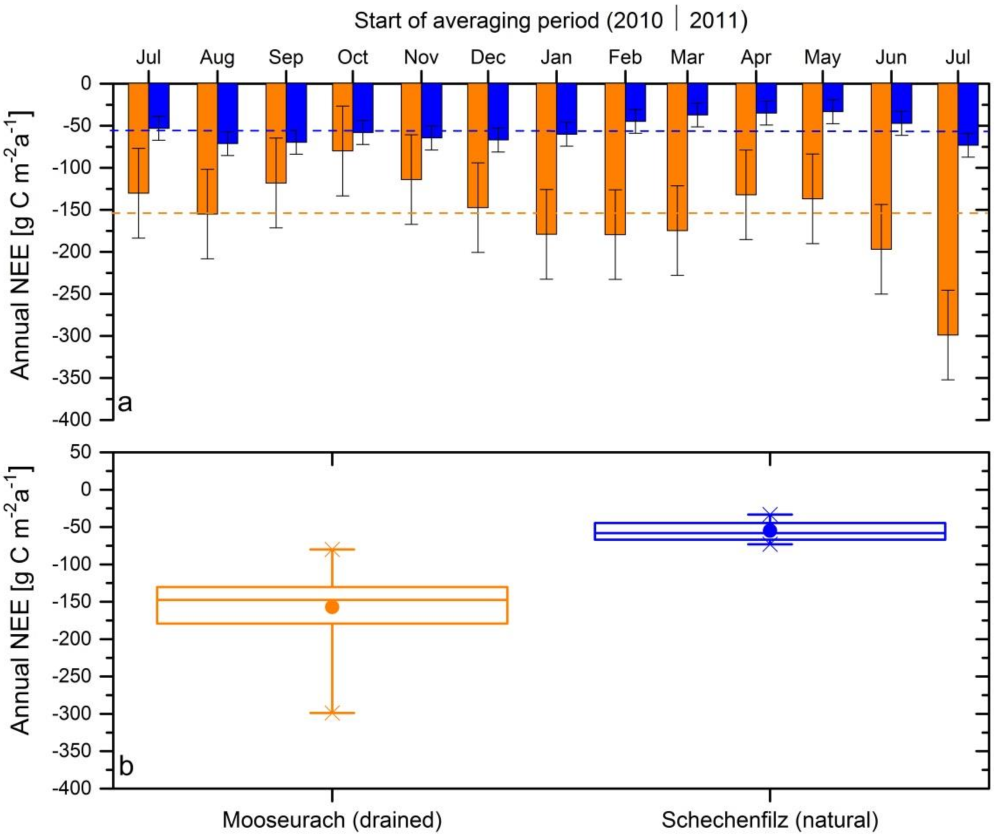

Fig. 30 a) Aannual NEE depending on start of averaging period, dashed line illustrate the mean 13 different averaging periods, the error bars present the standard deviation b) box plot illustrate the range (—) of detected annual sums of NEE, their mean ●. The box indicates the 25th, 75th and 50th percentile.

The mean annual net CO_2 uptake at the natural bog-pine site Schechenfilz is very similar to annual NEE budgets of other temperate and boreal natural, non-forested bog sites reported in the literature. For example, Lund et al. (2007) determined an NEE of -21 g C m^{-2} a^{-1} for a southern Swedish temperate bog site. In a temperate Canadian bog Lafleur et al. (2003) found an NEE ranging between +10 and -76 g C m^{-2} a^{-1}, depending on snow coverage in winter and water availability during the growing season. For an Irish blanket bog, an NEE of a very similar range (-49 and -61 g C m^{-2} a^{-1}) was reported by Sottocornola and Kiely (2005). This comparison implies that the presence of the bog-pines does not enhance the annual CO_2 uptake compared to non-forested bog sites, with their corresponding larger coverage of grass species.

Studies of the CO_2 exchange of drained and afforested peatland sites are rare, and the reported annual budgets are highly variable (see also Section 1.1). Mostly, drained peatland forests are reported to be strong annual net CO_2 sinks (> -150 g C m^{-2} a^{-1}). At some peatland forest sites the net CO_2 sink is entirely attributable to the carbon accumulation of the trees (Hargreaves et al., 2003; Meyer et al., 2013; von Arnold et al., 2005), but at other sites even the soil is a net CO_2 sink (Lohila et al., 2011; Minkkinen et al., 2002). However, for two drained peatland forests, of different age and different nutrient supply, Dunn et al. (2007) and Lohila et al. (2007) found an annual NEE close to zero (+84 and -58 g C m^{-2} a^{-1}). Measurements over two years over a boreal drained forest in central Sweden even indicate annual net CO_2 release (+128 g C m^{-2} a^{-1}, Lindroth et al. (1998)). Thus a comparison of our results with the carbon budgets from other sites is difficult, because site specific factors (e.g., nutrient supply, annual mean temperature, tree stock) are very different.

3.1.4 Seasonal variation of CO_2 exchange

Over the whole two year measurement period, the total net CO_2 uptake by the drained ecosystem at Mooseurach was -429 ±73 g C m^{-2} and -126 ±45 g C m^{-2} by the natural ecosystem at Schechenfilz (Fig. 31). Seasonal and short-term patterns were very similar at the two sites; the differences in the cumulative CO_2 exchange curves are mostly a result of the generally larger component fluxes (GPP and R_{eco}) at the drained spruce forest Mooseurach.

In spring, both ecosystems were strong CO_2 sinks in both years, in spite of the considerable water table drawdown in March and April 2011. The strong and consistent CO_2 uptake in spring is due to the phase-shift in the annual cycle of soil temperatures which are still low in spring and thus limit soil respiration while high radiation levels lead to moderately high photosynthetic activity (Dunn et al., 2007; Griffis et al., 2003). However, the carbon uptake rate was markedly stronger at the drained site.

The start of the net uptake season at the natural Schechenfilz site was very similar in spring 2011 and 2012 (mid-March), but due to a four-week data gap, the beginning of the growing season at the drained forest Mooseurach could not be detected precisely in 2012.

During dry and warm conditions in summer (July 2010 and August to September 2011), we observed reduced CO_2 uptake, which again was more pronounced at the drained site (Fig. 31). At this time soil respiration reaches its maximum because of highest soil temperatures. In addition, the photoperiod shortens and in spite of the sunny conditions, vegetation senescence starts, leading to lower photosynthetic activity. In this period, the CO_2 exchange at the drained site differs considerably between the two years. In 2010 we observed continuous CO_2 uptake until early October, while in 2011 the CO_2 uptake was discontinuous during the warm and dry period between mid-August and mid-October.

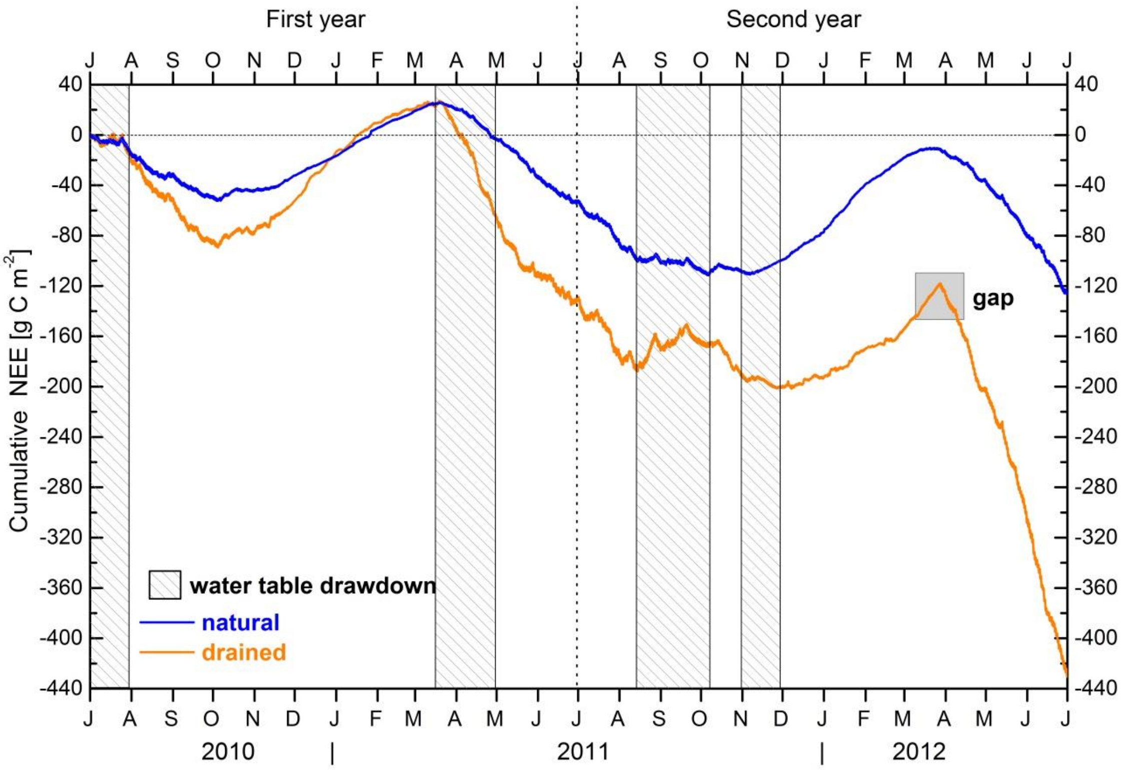

Fig. 31 Cumulative NEE at both sites over the whole measurement period (01 July 2010 to 30 June 2012). The red box marks a long data gap at Mooseurach due to power failure. The dotted vertical line shows the end of the first annual cycle. The horizontal dashed line highlights the zero line; negative slope of NEE represent carbon uptake and positive values carbon release. For clarity, uncertainty intervals are not shown here.

Finally, during the unusual sunny and rainfree weather conditions in November 2011 (compare Fig. 26) we observed an extended secondary net uptake period, while the natural bog-pine system stopped carbon uptake in October, similarly to the previous year (Fig. 31). The different NEE response of the two sites can be attributed to differences in tree physiology. The light-dependent photosynthesis of bog-pines is highly sensitive to low temperatures, whereas the photosynthesis of the Norway spruces is more robust in low temperatures (von Sengbusch, 2002). Thus, the extended period of sunshine at the natural bog-pine site Schechenfilz in November 2011 had no notable influence, as the bog-pines had already ceased photosynthetic activity in response to the drop in temperature. In contrast, photosynthesis by the spruces in Mooseurach continued in spite of the relatively low temperatures. Moreover, the combination of a strong water table drawdown (which makes growing conditions more favorable for the spruces), and low soil temperatures (which reduce soil respiration) further enhanced the net carbon uptake at the drained site in late autumn 2011. Overall, despite noticeable water table drawdown at the drained site, the spruces were apparently never water limited. This is additionally indicated by the independence between CO_2 exchange and soil moisture (Fig. 28).

A comparison of the pattern of cumulative NEE over the two years of measurements illustrates the different response of the CO_2 exchange of the drained and the natural peatland ecosystem to changing environmental factors. At the natural bog pine site, the annual cycles of cumulative NEE were very similar for both years, despite differences in (e.g.) drought periods between the two years. At this site the overall CO_2 exchange is more in balance, due to the low growing activity of the bog pines on the one hand, and the suppressed soil emissions caused by high soil water level on the other hand. In contrast, at least the respirative parts of CO_2 exchange at the drained spruce site are sensitive to changing environmental factors, like periods of increased temperature and water table drawdown. However, whether warm and dry anomaly periods increase or reduce net carbon uptake at the drained site depends on soil temperature and spruce phenology and thus on the season.

3.1.5 Long-term carbon balance

The results of the two years of eddy covariance measurement presented in this study indicate stronger CO_2 uptake of the drained bog forest than of the natural bog forest. Currently, the 44 year old spruces almost reach their maximum productivity (Gower et al., 1996), and can compensate the soil carbon efflux. However, this two year measurement period can only reflect the current stage of the observed ecosystems. For meaningful comparisons between peatland forests and full evaluation of the climate impact of different land uses requires a longer-term perspective, and in addition the determination of methane fluxes. However, we don’t have any reliable information on methane for this time period. Thus, we attempt to assess roughly the biome carbon balance in its long-term context based on CO_2 only. For an unbiased interpretation of the flux measurements at the drained site, we need to validate the impact of drainage and spruce afforestation on the long-term carbon balance. Therefore, peat loss-induced carbon emissions, as well as the net carbon fixation within the spruce life-cycle have to be considered.

Our estimation of the net peat loss is based on survey data from the 1930s which determined a peat thickness of about 4.4 m (Fig. 16). Thus, the peat thickness was reduced approximately by one meter down to today’s 3.4 m thickness over the last 70-80 years, resulting in a potential loss-rate of roughly 1.25-1.4 cm a^{-1}. The oxidative contribution to the overall subsidence is estimated to be almost 50%, the rest is attributed to peat compaction and consolidation (Armentano and Menges, 1986; Gronlund et al., 2008; Leifeld et al., 2011a). Assuming a constant carbon content of 49.7% (Fig. 15, Tab 4) and a dry bulk density of 0.15 g cm^{-3} (N. Rosskopf, personal communication, 2012) of the first 20 cm top soil layer in the last decades, we estimate a potential mean annual carbon loss to the atmosphere of about +500 g C m^{-2} a^{-1}(see Appendix A2). This rough annual carbon loss estimate is based on survey data from the 1930s, and therefore includes a period that started before the site was afforested. Thus, the potential annual peat carbon loss rate at our drained site is perhaps somewhat overestimated, as the carbon loss of peatland used for pasture is potentially larger than of peatlands used for forestry. Our estimate matches the annual carbon loss estimate of a drained mountain bog in the Swiss Alps, drained 119 years ago (Rogiers et al., 2008). This estimate of Rogiers et al. (2008) is based on differences in ash content after combustion of the peat-profile. A similar approach was used by Leifeld et al. (2011b) who estimated mean carbon loss rates ranging from +140 to +490 g C m^{-2} a^{-1} for two drained pre-alpine mountain bogs. Kluge et al. (2008) modeled a larger mean annual peat-carbon loss of about +700 g C m^{-2} a^{-1} for an agricultural peatland in northeast Germany. Reported soil emission rates of forested peatlands span a wide range, including our rough estimate. For example Braekke (1987) determined a carbon loss of a Norwegian boreal drained Scots pine ecosystem of +250 g C m^{-2} a^{-1}. However, in a later study Breakke and Finer (1991) note that about +100 g C m^{-2} a^{-1} are lost in the years immediately after drainage. The smaller emission rate could be explained by a 50% lower bulk density compared to the drained forest in Mooseurach, albeit the volume of peat loss was about two times larger in the Scots pine ecosystem. Simola et al. (2012) estimated a mean carbon loss of +150 g C m^{-2} a^{-1} at 37 drained forestry peatland sites in central Finland by re-sampling the peat stratum after about 30 years. Between the 37 sites the annual soil carbon flux varied strongly between -200 to +800 g C m^{-2} a^{-1}. They found no apparent relation between the amount of soil carbon flux and site fertility, or post-drainage timber growth. Minkkinen el al. (2007) measured heterotrophic soil respiration fluxes of a similar range between +248 and +515 g C m^{-2} a^{-1} at three boreal forestry-drained peatlands. However, in contrast to Simola et al. (2012), they point out that the carbon loss increased with increasing nutrient supply. Furthermore, the strongest carbon loss from soil peat was measured at the most northern site, which is likely attributed to differences in site factors such as substrate quality, nutrient status, and hydrology. On the other hand, Armentano and Menges (1986) note that the carbon loss in temperate climate zones is about 10 times larger than in the boreal zone. In summary, the carbon loss estimates from drained and afforested peat are highly variable and likely depend on mean annual temperature and nutrient supply. Both, temperature and nutrient supply are relative high at the drained bog forest in Mooseurach, supporting a relative large carbon loss estimate.

The current standing biomass of the 44 years old forest, above and below ground, was determined by biometry and common allometric relations as 86 t C ha^{-1} (S. Röhling, personal communication, 2012). In comparison, the estimated potential net carbon loss from peat degradation is approximately +220 t C ha^{-1} within the same period of 44 years, resulting in a total net emission of +134 t C ha^{-1}. These quantities should be considered as rough estimates to evaluate current measurement results in their long-term context. Currently, the eddy covariance measurements indicate an average annual net CO_2 uptake of -157 ±36 g C m^{-2} a^{-1} (=-1.57 t C ha^{-1} a^{-1}). Thus, in this scenario the forest would need almost another 100 years of carbon assimilation at the current rate, to offset the net carbon loss of the last 44 years. Because the expected life-cycle of the spruces at the drained site is only about 60 years, we must conclude that, in the long-term, the drained bog forest is a strong net overall CO_2 source even if the current eddy-covariance measurements indicate net CO_2 uptake.

Based on published results of methane exchange of drained peatland forests, the contribution of methane to the long-term carbon balance is seen to be negligible. For example, Lohila et al. (2011) measured minor methane uptake of -0.09 g C m^{-2} a^{-1} in a forestry-drained boreal peatland in southern Finland. Other studies indicate low methane release between 0 and +0.12 g C m^{-2} a^{-1} in temperate and drained coniferous peatland forests of comparable age (von Arnold et al. 2005, Yamulki et al. 2013).

At the natural bog site Schechenfilz the peat layer as well as the water level was not significantly affected by human interference in the past, and the soil conditions are still pristine. Furthermore, the forest is no plantation, and the age structure of the trees covers a wide range. Hence, it is likely that the carbon exchange of the natural bog forest is close to the long-term balance with a small net accumulation rate. However, due to the high water level at natural bog sites we have to expect methane emissions which would reduce the carbon uptake budget and due to their larger global warming potential considerably reduce the climate mitigation effect.

According to the literature review of Saarnio et al. (2007) the methane budget of boreal ombrotrophic mires ranges between +1 and +16 g C m^{-2} a^{-1}. For example, Rinne et al. (2007) determined an annual net methane budget of +9.4 g C m^{-2} a^{-1} for an ombrotrophic fen site in southern Finland . At the bog-pine site Schechenfilz the groundwater level is never above the surface, which would enhance methane production. Moreover, the coverage of sedges, which are known to serve as conduits for the methane fluxes from the soil, is low in comparison to non-forested bog-sites. Therefore, we expect only small emission rates which likely not offset the climate mitigation potential of the carbon uptake. Thus the natural bog-pine site was a small but stable net carbon sink during the observation period.

3.1.6 Uncertainty of the annual NEE

We distinguish between random (ε) and systematic errors or bias (δ). The systematic errors need to be further differentiated into full systematic bias and selective bias (Moncrieff et al., 1996). Selective bias affects solely carbon uptake and carbon release and increases separately, when the measurement period is extended. Causes of systematic errors include underestimation of nighttime respiration, high frequency loss, variations of footprint size and orientation (Massman and Lee, 2002; Richardson et al., 2012; Schmid and Lloyd, 1999). Generally, systematic errors cannot be identified by statistical analysis, but known systematic errors are often minimized by quality control strategies such as the u_*-criterion to eliminate periods with non-turbulent fluxes, corrections of spectral loss (Moncrieff et al., 1997; Moncrieff et al., 2004), or the rejection of observations not representing the target area. However, we are not able to prevent all systematic errors, particularly those whose origin or behavior are not known sufficiently. Concerning the site comparison we can assume that the impact of systematic errors is relatively small, as flux calculation, post-processing (rejection criteria) and gap-filling were conducted in the same way (with the exception of the footprint analysis at the natural site). Thus, any bias due to data processing is expected to be very similar at both sites. However, two different measurement devices were used for H_2O and CO_2 flux measurements (open-path (LI7500, Li-Cor) at the natural site and closed-path (LI7200, Li-Cor) at the drained site), which may leads to a biased site comparison. Based on a study of Burba et al. (2010b) and Burba et al. (2009), CO_2 and latent heat fluxes measured by LI7500 and LI7200 show a very good agreement. The slope of the linear regression between latent heat- and CO_2 fluxes measured with this two devices for several months is almost one, and the coefficient of determination is better than 0.96 (not shown). Thus the expected bias is negligible and is not addressed in the uncertainty estimation for the site comparison.

Random errors cannot be corrected for, but identified by statistical analysis and they often decrease with extended datasets. We estimated the gap-filling uncertainty using 10,000 bootstrap samples for R_{eco} and GPP estimation, resulting in a gap-filling uncertainty of ±30.7 g C m^{-2} a^{-1} in the first measurement year and ±65.9 g C m^{-2} a^{-1} in the second year at the drained site. The gap-filling uncertainty based on data from the natural site is ±27.8 g Cm^{-2} a^{-1} and ±38.1 g Cm^{-2} a^{-1} for the two periods.

Random errors of measured fluxes are caused by footprint variability, errors of turbulence sampling and instrument errors (Dragoni et al., 2007). The errors of turbulence sampling can be determined by the variance of the covariance, including auto- and cross-covariance terms, following Finkelstein and Sims (2001). This error estimation is implemented in some eddy covariance flux processing software packages, e.g., TK3 (Mauder et al., 2013).

The mean error of turbulence sampling is thus determined at ±2.6 g C m^{-2} a^{-1} in the first, and ±3.1 g Cm^{-2} a^{-1} in the second year at the drained, and ±1.5 g Cm^{-2} a^{-1} and ±2.5 g Cm^{-2} a^{-1} at the natural site. The uncertainty due to instrument errors was not be estimated accurately in the present project, but based on comparative estimates in the literature (e.g., Dragoni et al., 2007) we expect them to be small in comparison to the gap-filling error.

The random error of measured and gap-filled fluxes were combined by quadrative error propagation (e.g., Richardson et al., 2012; Richardson and Hollinger, 2007), resulting in a negligible contribution of the error induced by turbulence sampling. Thus random uncertainty of the cumulative NEE at the drained forest in Mooseurach was determined as ±30.7 g Cm^{-2} a^{-1} in the first and ±65.6 g Cm^{-2} a^{-1} in the second year, and amounted to 40% and 22% of the total annual NEE. At the natural site Schechenfilz, the random uncertainty of annual NEE is estimated at ±27.8 g Cm^{-2} a^{-1} and ±38.1 g Cm^{-2} a^{-1}, or 47% and 52% of the total budgets.

Even if an overall uncertainty of about 50% were assumed at both sites, the annual CO_2 uptake is invariably stronger at the drained site.

3.1.7 Summary and conclusions

Eddy covariance measurements of NEE over two complete annual cycles (July 2010-June 2012) indicate a stronger uptake of CO_2 at a drained spruce forest ecosystem at Mooseurach, compared to a natural bog-pine site at Schechenfilz (-130 ±31 and -300 ±66 g Cm^{-2} a^{-1} in Mooseurach and -53 ±28 and -73 ±38 g Cm^{-2} a^{-1} in Schechenfilz, respectively). Due to the small distance of 10 km between the sites, differences in the CO_2 exchange can be attributed solely to site-specific factors, such as land use history, soil conditions and vegetation composition, rather than to different atmospheric conditions.

At the drained ecosystem, the budgets of both component fluxes, GPP and R_{eco}, are about twice as large as at the natural bog-pine site. The stronger GPP at the drained site can be attributed mostly to the larger PAI. Furthermore, the response of the CO_2 flux at the drained spruce forest is much more sensitive to environmental forcing and therefore any variability of the driving factors has a greater impact on the NEE. Whether water table drawdown promotes or reduces net carbon uptake depends on the state of phenology of the spruces and the level of soil respiration, which is strongly related to soil temperature and thus to the season.

The present eddy covariance measurements represent only the current state of forest and soil conditions, but CO_2 exchange studies of managed peat forest ecosystems need to consider the different stages of ecosystem development. A rough estimate of carbon loss due to drainage and peat degradation resulted in a potential annual carbon loss of +500 g C m^{-2} a^{-1} for the previous 70-80 years. Comparing carbon fixation by the spruces and carbon loss from the peat since forest planting, 44 years ago, we roughly determined a total net carbon loss of +134 t C ha^{-1}. Consequently, the spruce forest in Mooseurach would need to assimilate carbon for a further 100 years at the present rate to compensate the carbon loss of peat degradation in former years.

In contrast, the natural bog forest in Schechenfilz is a robust and stable but small CO_2 sink. We thus have to conclude that the natural bog-pine forest is a more effective CO_2 sink in the long-term, in spite of lower net uptake rate during the observation period. Thus, this study serves to illustrate the potential climate benefits provided by peatland restoration.

3.2 Modification of annual NEE due to data rejection and gap-filling

In this section, we aim to analyze the influence of increasing rigor of data rejection criteria and the impact of different gap-filling methods on short-term variability and on annual sums of NEE. This study is based on data from 2011 of both peatland sites Mooseurach and Schechenfilz.

We increased the rigor of rejection criteria by successively applying the footprint-criterion, u_*-criterion and outlier-test (Section 2.5.1); however, the footprint criterion was solely applied to Mooseurach data. To compare the impact on annual balances, the data were gap-filled by the NLR-method (Section 2.7.3). In the next step, the divergence of NEE induced by the chosen gap-filling method was investigated. We applied three different standard methods: non-linear regression (NLR), mean daily variation (MDV) and look-up table (LUT), following Falge et al. (2001) and Reichstein et al. (2005) (Section 2.7).

3.2.1 Influence of data rejection on annual NEE

Data of Schechenfilz (data coverage is 72% in 2011) is affected by more gaps than the data of Mooseurach (data coverage 87%). Data rejection, to ensure quality controlled half-hourly fluxes, leads to further reduction of the data volume. In Mooseurach, the application of the footprint criterion is advisable because of the limited forest expansion (Fig. 22b). This criterion reduces the data volume by about 20%, whereby most of the rejected data was discarded at night. This is due to the regularly occurring stable atmospheric stratification at night, leading to a theoretical infinite large footprint area. Thus, in many cases, data that is rejected by footprint criterion would be also rejected by a turbulence criterion (e.g., u_* threshold). By choosing the strictest studied rejection criterion, 56% of all data are rejected from the Mooseurach dataset and 41% from the Schechenfilz data. Overall, only 16% of the nighttime data at Mooseurach and 12% at Schechenfilz remains. The outlier test has only a very small influence on the data volume. A detailed list of rejection criteria and their influence is given in Tab. 6.

As data rejection is mainly subjected to fluxes which are measured during low atmospheric mixing conditions, thus mainly small respiration flux values are affected (Fig. 32a). Therefore, the data rejection leads to a large modification of the dataset, where mostly fluxes between ±5 $\mu molm^{-2}\ s^{-1}$ are discarded. However, the application of the outlier test leads to a smoothing effect, as this criterion particularly affects large fluxes (Fig. 32b).

Tab. 6 Influence of data rejection on data coverage, presented separately for the entire dataset (total), and for day- and nighttime fluxes only at both sites. NEE_x, where x is the substitute for the subscript orig: all available flux data; f: application of footprint criterion (only at Mooseurach); U: application of u_*-criterion; o: outlier test.

Mooseurach	**data_total**		**data_night**		**data_day**		**annual NEE**
	N	**%**	**N**	**%**	**N**	**%**	**g C $m^{-2}a^{-1}$**
NEE_orig	15294	87	8483	87	6811	87	-469
NEE_f	12016	69	5619	58	6397	82	-301
NEE_fU	6738	38	2147	22	4591	59	-170
NEE_fUo	6589	38	2064	21	4525	58	-179

Schechenfilz	**data_total**		**data_night**		**data_day**		**annual NEE**
	N	**%**	**N**	**%**	**N**	**%**	**g C $m^{-2}a^{-1}$**
NEE_orig	12590	72	6041	63	6549	83	-213
NEE_U	7338	42	1956	20	5382	68	-79
NEE_Uo	7062	40	1771	18	5291	67	-60

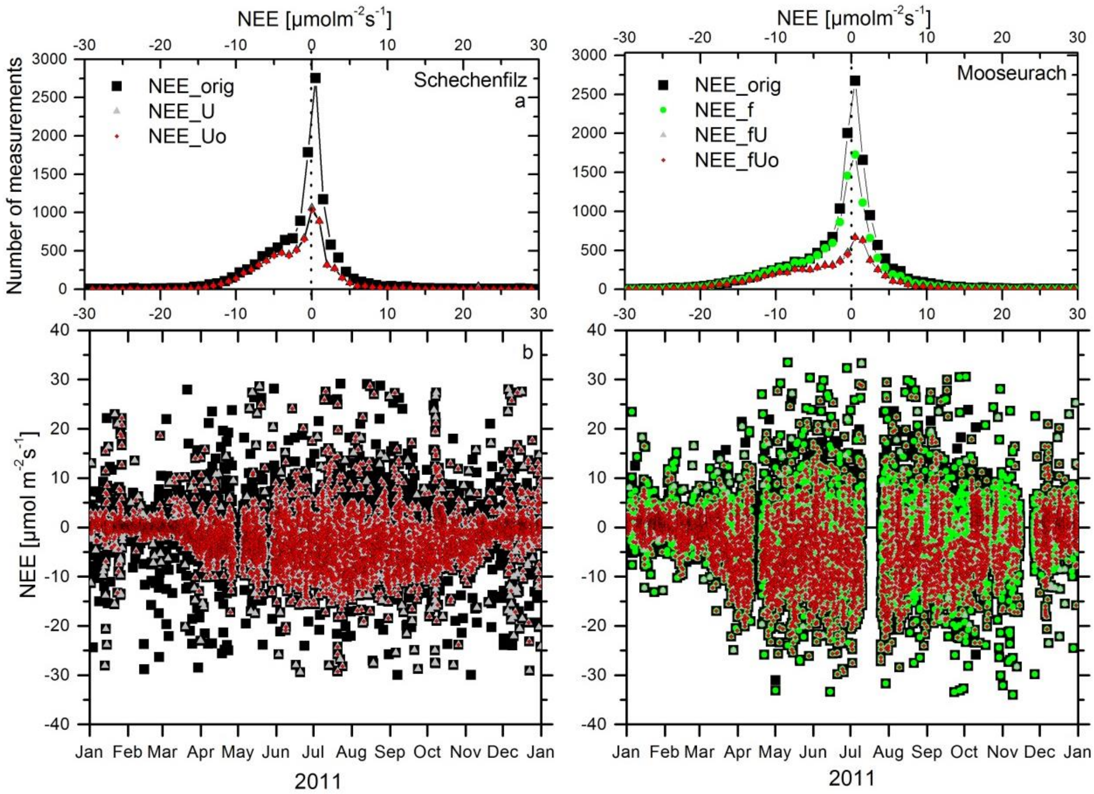

Fig. 32 Impact of data rejection (u_*-criterion (U), footprint (f) and outlier test (o)) on flux distribution and scatter of NEE.

The estimated cumulative annual NEE varies strongly, depending on data rejection criteria. At the natural site Schechenfilz, the annual NEE ranges between -214 and -63 g C $m^{-2}a^{-1}$

(±151 g C $m^{-2}a^{-1}$) and at the drained afforested site at Mooseurach even between -470 and -170 g C $m^{-2}a^{-1}$ (±300 g C $m^{-2}a^{-1}$, Fig. 33).

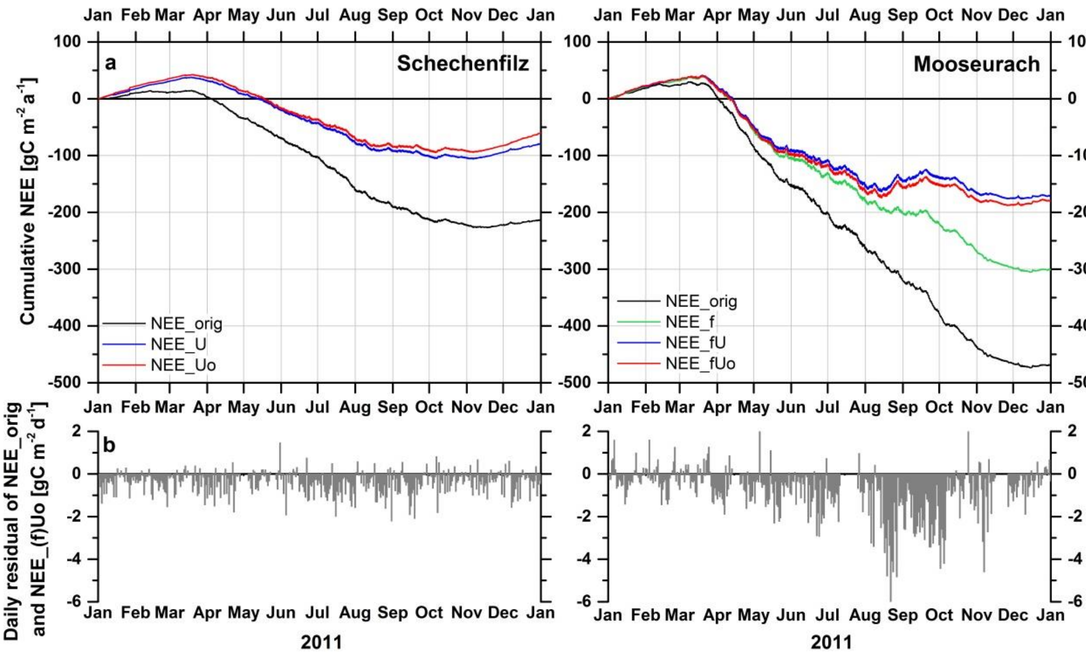

Fig. 33 a) Annual NEE, influenced by increasing rigor of rejection criteria to ensure good data quality. For Mooseurach, the footprint criterion, u*-threshold and the outlier test were successively applied. In Schechenfilz, we applied only the u*-and the outlier criterion. To compare annual budgets, data gaps were filled by non-linear regression. b) daily residual of NEE determined without data rejection (NEE_orig) and with the strongest data rejection criteria.

At both sites, the NEE-data underlying no quality check (NEE_orig) result in a distinctly stronger CO_2 uptake, compared to the NEE based on quality checked data. Most of the differences occur during the growing season, when flux amplitudes are large (Fig. 33b). At Mooseurach, the course of the NEE is very similar during winter, but differs strongly during the growing season. The largest divergence occurred in the period of mid-August to mid-September 2011 (Fig. 33b), when soil temperature and thus gap-filled respiration terms, were distinctly larger. The NEE uptake is reduced by one-third, when solely the footprint criterion (NEE_f) was applied at Mooseurach site. At both sites, the application of a u*-threshold leads to significantly reduced CO_2 uptake, compared to the budgets estimated without this rejection criterion. Overall, the uptake is reduced by more than 60%, the annual CO_2 uptake is smaller than -200 g C $m^{-2}a^{-1}$ at Mooseurach and smaller than -100 g C $m^{-2}a^{-1}$ at Schechenfilz. At both sites, the u*-criterion is a very effective tool to detect fluxes which are measured during insufficient atmospheric mixing conditions. Similar CO_2 uptake reduction due to flux-corrections during insufficient turbulent mixing is reported by many authors (e.g., Aubinet et al., 2000; Goulden et al., 1996; Schmid et al., 2003). Most studies quantify a systematic reduction of annual NEE of 50 to 100 g C $m^{-2}a^{-1}$ (Falge et al., 2001; Goulden et al., 1996; Hui et al., 2004; Ruppert et al., 2006). This is similar to the reduced CO_2 uptake at the natural site Schechenfilz but less than the decreased NEE balance we found for the drained spruce forest at Mooseurach. Poor atmospheric mixing occurs mostly at night, which leads to a selective underestimation of respiratory fluxes (Moncrieff et al., 1996), because insufficient turbulence supports advective drainage flows which could not be measured. With this horizontal flow, CO_2 is transported out of the target area without reaching the measurement height and thus without being recorded. Consequently, only a fraction of CO_2 emissions could be measured and the fluxes are underestimated. Thus, the annual CO_2 uptake decreases markedly when the underestimated fluxes are rejected. Particularly, ecosystems that have large storage terms, e.g., forest ecosystems like in Mooseurach, are strongly affected by drainage flows, which results in an underestimation of CO_2 emissions. However, the CO_2 stored in the canopy air could be flushed out by turbulence as soon as it restarts (e.g., Goulden et al., 1996; Massman and Lee, 2002; Moncrieff et al., 1996) and could be counted double if data were gap-filled during the low u_* period only. It is generally accepted that for timescales larger than a few days, the storage term cancels and is therefore negligible for long term NEE studies. Nevertheless, handling of the CO_2 storage and the systematic nighttime underestimation would benefit from measuring CO_2 storage-change below the measurement height. However no conclusively reliable method to achieve this goal has been developed to date. Thus we decided to address this problem by quality control strategies (and data rejection, rather than correction terms.

We applied the currently most widely used u_*-criterion to circumvent the problem of underestimation of nighttime fluxes, which is a very effective tool to detect low turbulence data at our sites (Fig. 23). However, this method is still debatable (Aubinet et al., 2012) and the correction is affected by several drawbacks and has to be applied with care. Despite different u_*-thresholds, caused by different canopy heights, the response to the application of u_*-correction is similar at both sites. We also tested the influence of quality flagging after Foken and Wichura (1996), which includes a measure for turbulent development (ITC-test: integral turbulence characteristic). However, the u_*-correction is more effective than the flag system. Even if a strong flag threshold of 3 is implemented, the u_*-criterion leads to an additional reduction of the data volume of 9% (Schechenfilz) and 13% (Mooseurach) which corresponds to almost 1600 and 2230 half-hourly flux values for the year 2011, respectively. The application of further rejection criteria has less effect to the magnitude of annual NEE.

Rejected data have to be gap-filled based on fluxes that passed the rejection criteria. Thus, it has to be assumed that fluxes measured during poor atmospheric mixing can be inferred by gap-filling from measurements made in windy conditions, which is not proven (Papale et al., 2006). Furthermore, as most of the uncertainty of NEE can be attributed to the gap-filling (Dragoni et al. (2007), see Section 3.1.6), the question arises, how much data rejection is too much? Generally, it is recommended to apply the strongest filtering for data used for detailed studies, like gap-filling model evaluation. Less strict criteria can be used for long-term studies (Foken, 2008; Mauder and Foken, 2011). Filtering out the natural fluctuation of turbulent fluxes and increasing the number and length of gaps should be avoided. To find the optimal balance is very difficult and requires data handling experience at each site, covering individual responses to specific periods and conditions. Our findings go along with Falge et al. (2001) who conclude that the accuracy of gap-filling depends strongly on the pre-treatments of data used to develop the gap-filling algorithm, particularly if a u_*-correction is implemented or not. This emphasizes the importance of well performed data post-processing to avoid uncertainties in further progress of NEE determination. Despite the need of generalized post-processing criteria for better site comparisons and investigations of interannual datasets, the final decision about data rejection or acceptance has to be done site specifically. However, at the two peatland sites, increasing strictness of rejection criteria has a similar effect on the relative deviation of annual NEE.

3.2.2 Influence of gap-filling on annual NEE

The study on the influence on NEE, induced by the choice of gap-filling method (NLR, MDV, LUT; see Section 2.7), is based on data that passed the strictest rejection criteria. Results show that NEE estimated for the natural peatland forest Schechenfilz is less affected by the choice of gap-filing method than the NEE estimate for the drained site Mooseurach (Fig. 34). In Schechenfilz, the annual NEE budgets differ only by ±18 g C $m^{-2}a^{-1}$ depending on the method, whereas in Mooseurach the range of the annual NEE is about ±250 g C $m^{-2}a^{-1}$, including a change of sign (+68 to -180 g C $m^{-2}a^{-1}$ Fig. 34b). Thus, the quantification of the CO_2 emissions and the global warming potential of this site is strongly dependent on the selected gap-filling method, underpinning the importance of a sensible discussion of the applied gap-filling procedure.

Because meteorological conditions were the same at both sites (see Section 3.1.1), we conclude that the influence of the gap-filling method depends on the site specific conditions. We found that differences of NEE induced by the applied gap-filling method increase with increasing amplitude of CO_2 fluxes. This is consistent with the studies of Richardson and Hollinger (2007) and Dragoni et al. (2007) who detect a similar relation between flux amplitude and gap-filling uncertainty. At the drained spruce site Mooseurach the rates of CO_2 uptake and CO_2 release are much larger due to high primary productivity of the spruces and high respiration fluxes from the drained soil (Section 3.1), leading to larger amplitudes of CO_2 exchange. At Schechenfilz, the primary production of the slow growing bog-pines is relatively small and the decomposition processes in the soil are suppressed due to water saturated and anoxic soil conditions. Therefore, the amplitudes of the CO_2 exchange are relatively small at the natural site.

At the Schechenfilz site, NEE gap-filled by the NLR and LUT-methods indicate a very similar course during the whole year. The difference between the two time series of NEE is generally close to zero with a mean of +0.25 g C m^{-2} d^{-1}, whereas the maximum differences are between -2.6 and +3.3 g C m^{-2} d^{-1} (Fig. 34b). Although differences between NEE gap-filled by the NLR and MDV methods are much larger (maximum difference -6.8 and +7.8 g C m^{-2} d^{-1}), the mean of the residuals equals zero. Additionally, we did not find any dependence between divergences in the NEE and the presence of long data gaps at the Schechenfilz site.

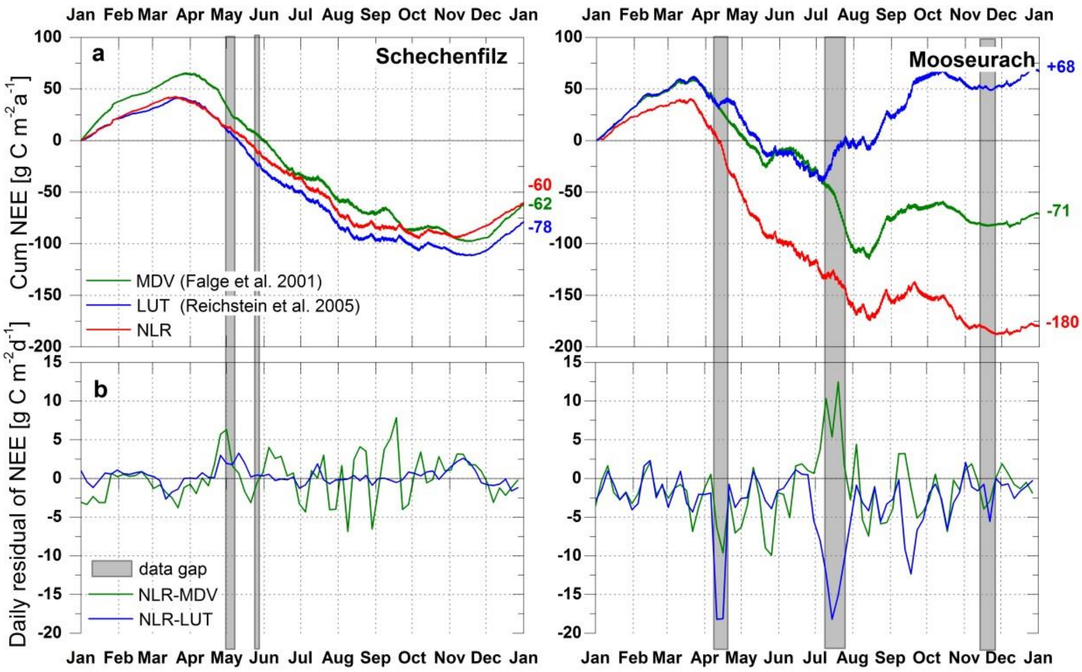

Fig. 34 NEE calculated with different gap-filling methods; data passed u_*-criterion, outlier test and data from Mooseurach additionally passed the footprint criterion. a) daily residuals of NLR and MDV and LUT-method, respectively; b) comparison of the annual course of NEE, generated by the three different methods. Grey periods mark by long data gaps (> 3 days).

In Mooseurach, the residuals of NEE detected by different methods present a generally larger range of variation. The maximum differences of NEE gap-filled by NLR and MDV-approach are -10 and +12.5 g C m^{-2} d^{-1}, whereas the differences of NEE gap-filled by NLR and LUT-approach are -18 and +2.3 g C m^{-2} d^{-1}. At the drained site, the occurrence of long data-gaps turns out to have a strong impact on divergences in the NEE, due to the chosen gap-filling approach. The largest differences occur during long data gaps in April and July 2011, where the application of the LUT method leads to predominantly CO_2 release in spring and in summer. Applying the MDV method leads to stronger CO_2 uptake in April and stronger release in July compared to the NEE resulting from the NLR method. As in the previous section (3.2.1), large differences in the NEE values occur in the period of mid-August to mid-September 2011 although data coverage was on usual level in this period. At this time, soil temperature as well as soil respiration fluxes reach their maximum. At both sites, the residuals between NEE estimated by different methods indicate larger variation during the growing season than during wintertime.

Generally, long data gaps resulted in the largest gap-filling uncertainty (e.g., Falge et al., 2001; Richardson and Hollinger, 2007), particularly when they occur during transition periods (Richardson and Hollinger, 2007). At Mooseurach, large data gaps strongly affect the estimate of NEE, even during summer when the GPP is more consistent.

Overall, the NLR method leads to the smoothest course of NEE, the short-term variability of NEE determined by LUT- or MDV-methods is much larger. The LUT-method overestimates the contribution of respiration and particularly long data gaps have a strong impact and lead to a non-comprehensible course of NEE. For example, the LUT-method estimates strong carbon release during the long data gaps in April and July, which cannot be attributed to environmental conditions and is not confirmed by the other methods. Additionally, NEE gap-filled by LUT method is the only one that does not indicate CO_2 uptake in the outstanding sunny November 2011 (Section 3.1.1).

However, the findings of Moffat et al. (2007) and Falge et al. (2001), who tested 15 and 3 different gap-filling methods respectively, indicate that the uncertainty related to the selection of the gap-filling method is small. Moffat et al. (2007) estimated a mean influence of the gap-filling on the annual sum of ±25 g C $m^{-2}a^{-1}$ for 10 benchmark datasets of 6 European forests. Their findings go along with the NEE sensitivity towards gap-filling that was determined at the natural bog-pine forest Schechenfilz, but does not apply for the drained site Mooseurach where the NEE variation is 10 times larger.

To determine the gap-filling method providing the best performance, 7 and 14 day long artificial data gaps were generated (June 01, 2011 to June 07, and June 14, respectively). Therefore, we chose a period during the growing season, in which relatively good coverage of quality controlled data was achieved (52% (14 days gap) to 62% (6 days gap), applies to both sites). The comparison between observed and predicted half hourly fluxes is based on statistical parameters (Fig. 35).

NEE predicted by the NLR-method provided the best performance for both sites and gap sizes. The coefficients of determination (R^2) indicate equal or better fit between measured and predicted half-hourly fluxes and the relative root mean square error (relRMSE) is always smaller compared to the parameters provided by the other two methods. The central range of the NEE is well represented but, due to the lower data coverage in the extreme ranges of CO_2 exchange, large uptake or release values were predicted not very well by this method. Consequently, the bad capture of extreme values leads to a smoothing effect.

The LUT-method also resulted in good performance of predicted NEE. Using this method, the influence of outliers is stronger while the smoothing effect is smaller (Fig. 35).

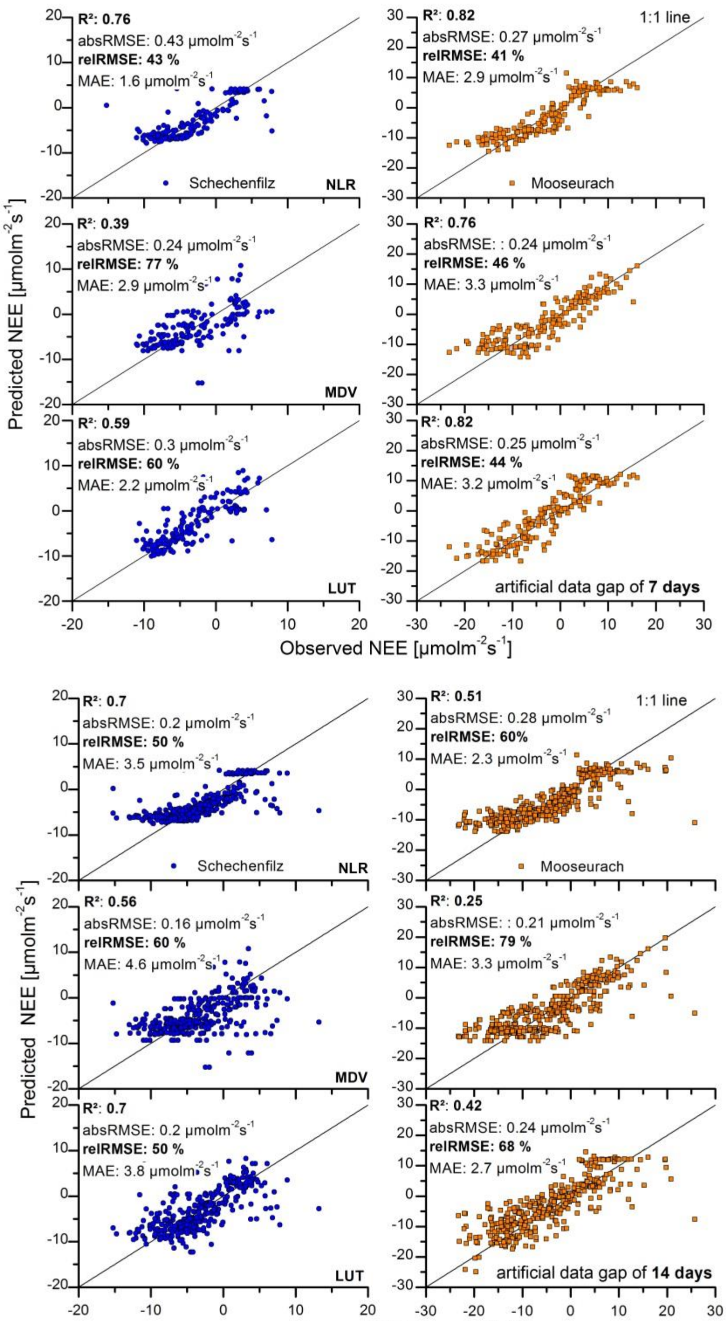

Fig. 35 Scatter plots of observed vs. fluxes predicted half-hourly NEE, gap-filled by three different methods. Top: artificial data gap of 7 days (June 01, 2011-June 07, 2011) and bottom: artificial data gap of 14 days (June 01, 2011-June 14, 2011). Statistical parameters are: coefficient of determination (R^2), absolute and relative root mean square error (absRMSE and relRMSE) and mean absolute error (MAE).

The mean daily variation method leads to the worst result, which could be explained by the large influence of extreme values. In particular, during the night when data coverage is small, single fluxes will be repeated several times, depending on the size of the moving window. Thus, this method is also very sensitive to outliers. Application of the MDV method leads to the largest root-mean square errors (RMSE) and mean-absolute-errors (MAE) for both sites and both gap lengths. Therefore, we adopted the NLR-method to estimate the NEE at both sites. This method was adopted for the estimation of annual NEE at many sites (e.g., Arneth et al., 2002; Lindroth et al., 2007; Lund et al., 2007; Schmid et al., 2003). Additionally, Moffat et al. (2007) and Falge et al. (2001) found a comparatively good performance of the NLR-method in their gap-filling comparison studies.

3.2.3 Summary and conclusions

Based on an eddy covariance dataset from the year 2011, we studied the influence of data rejection and different gap-filling approaches on NEE at both sites. Application of the u_*-criterion, where data measured at low turbulent conditions are discarded, the annual net CO_2 uptake is reduced by more than 60% at both sites. Data rejection leads to strong modification of the dataset - mostly nighttime respiratory fluxes are rejected. Data rejection affects the data of both sites proportionally in the same way. In contrast, the influence of gap-filling procedures (non-linear regression (NLR, look-up table (LUT), mean daily variation (MDV)) on NEE is very different between the sites. The annual NEE only varies between $\pm 18\ g\ C\ m^{-2}a^{-1}$ at Schechenfilz but between $\pm 250\ g\ C\ m^{-2}a^{-1}$ at Mooseurach, including a change of sign. The large variation at Mooseurach is due to the large amplitudes of CO_2 exchange in the drained spruce forest. At Schechenfilz, fluctuations of CO_2 exchange are much smaller; this is probably also an explanation for the smaller impact of long data gaps on the NEE prediction.

Data gaps, which have to be filled, are usually not randomly distributed throughout the course of day and year. Due to the reduced data coverage during the night, the respiration term is affected more strongly by uncertainties in comparison to the GPP term. At drained peatland sites, the amplitudes of soil respiration are very large and the uncertainty induced by gap-filling is larger than at natural sites. Depending on site conditions, the influence of the gap-filling method on the variation and uncertainty of the annual budget can exceed the influence of all the other corrections and measurement uncertainties (Section 3.1.6). In spite of large differences between ecosystems, more and more scientists plead the need for standardization of post-processing procedure to improve the comparability of carbon budgets across biomes and for underpinning interannual variability (Dragoni et al., 2007; Moffat et al., 2007; Rebmann et al., 2005; Ruppert et al., 2006). A common consensus is important for site-comparability, particularly in flux networks like ICOS, TERENO and FLUXNET. However, the CO_2 balance of some ecosystems is very sensitive to poorly adapted gap-filling procedures. Therefore, plausibility should be verified in detail by one who knows the ecosystem and local conditions.

3.3 Methane exchange at the natural bog-pine site Schechenfilz

In this section, we present the CH_4 and CO_2 exchange, measured over 15 months between July 2012 and September 2013 at the natural, temperate bog-pine site Schechenfilz. The study determined the most relevant environmental parameters that influence short-term CH_4 exchange. Furthermore, we compared different gap-filling methods, to get the best correlation between measured and modeled CH_4 fluxes at our site. The comparison included three different approaches: the mean daily variation (MDV) method, the look-up table (LUT) method and a non-linear regression between methane and air temperature (NLR), in analogy to the CO_2 gap-filling comparison methods discussed by Falge et al. (2001). The results of this section are submitted as a research paper "Ecosystem scale methane fluxes in a natural temperate bog-pine forest in southern Germany" in Agriculture and Forest Meteorology (in review).

3.3.1 Environmental factors controlling CH_4 exchange

During the observation period (July 01, 2012- June 30, 2013), the air temperature was only +0.1 °C warmer than the long-term average (1971-1990), indicated by data of the nearby climate station "Attenkam" (distance to Schechenfilz 8.5 km, 47°52'.40"N; 11°21'46"E; 672 m a.s.l.) of the German weather service (DWD). In July and August 2012 and 2013, and in April 2013 temperatures were clearly above the long-term average, while the cold temperatures in February and March 2013 likely led to a delayed start of spring (Fig. 36b, c). In the observation period, the water table depth was usually close to the surface, and fluctuated only slightly around the mean level of -0.05 m. In summer 2012, only two short water table drawdown events (water level below-0.12 m) of less than 6 days were observed. However, in July and August 2013, the water table dropped markedly below a level of -0.12 m for more than 6 weeks.

The annual courses of the methane flux and of the environmental parameters (Fig. 36) indicate that the annual variation of methane emissions was in phase with air and soil temperature, as well as with the variation of photosynthetic active photon flux density (PPFD), except for the markedly long water table drawdown event in July and August 2013 (Fig. 36e). In this period, CH_4 fluxes were considerably smaller than in the previous year, despite relatively warm soil- and air temperatures. The consistently positive mean daily methane flux (blue dots in Fig. 36; positive flux values indicate emission, and negative values indicate uptake), marks this site as a methane source throughout the year (Fig. 36a).

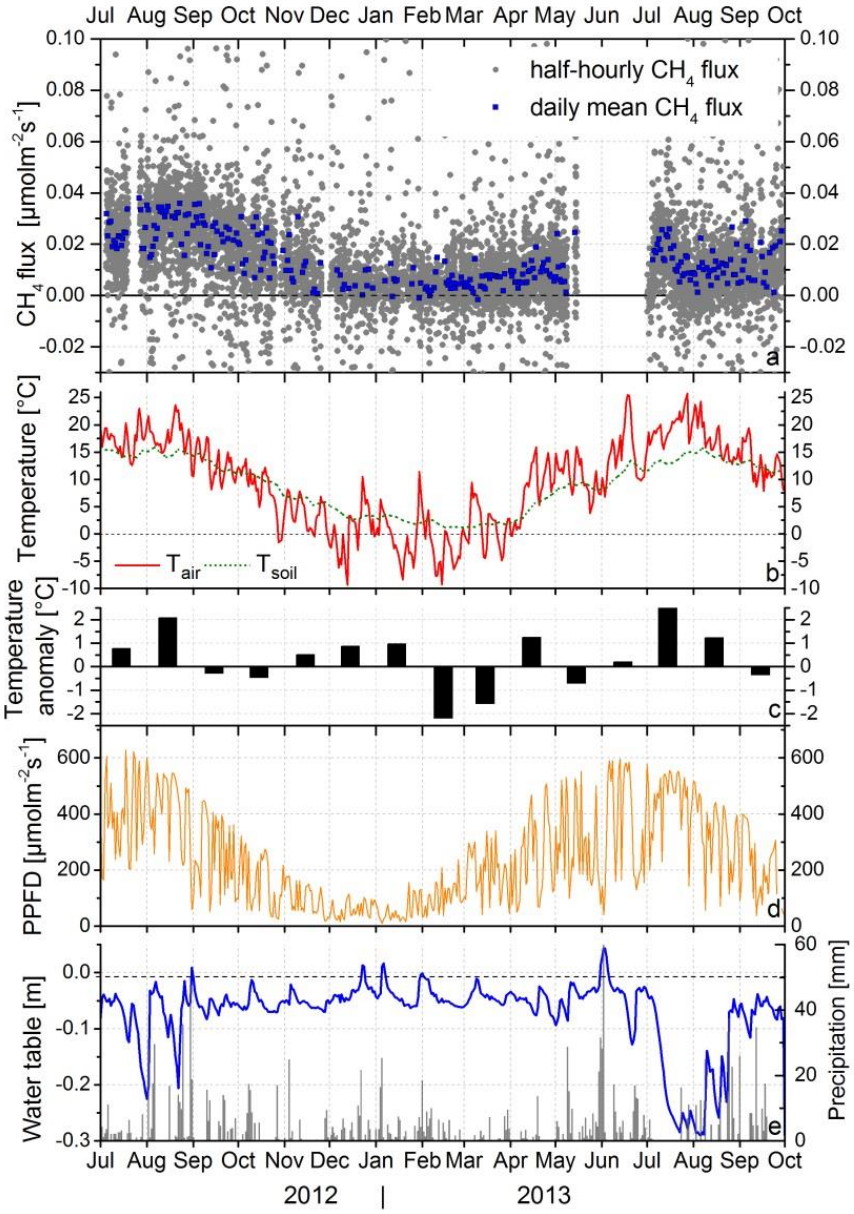

Fig. 36 Fifteen months of a) eddy covariance measured half-hourly CH_4 fluxes (gray dots), and daily mean of the CH_4 exchange (blue squares). For daily mean fluxes, only days with a quality controlled data coverage of higher than 33% were considered. The large data gap from May 15 – July 4 was caused by instrument failure; b) daily mean of air temperature (red solid line) and soil temperature (green dotted/dashed line) at 10 cm depth; c) air temperature anomaly, based on 30 years long term data, provided by the German weather service (DWD); d) daily mean of photosynthetic active photon flux density (PPFD); e) daily mean of water table depth (blue line) and daily sum of precipitation (grey bars).

Figure 37 illustrates the influence of different environmental parameters on half hourly methane fluxes. At Schechenfilz, we found increasing methane emissions with increasing soil temperature (T_{soil}), measured -0.1 m below the surface. However, the positive correlation of methane emission and air temperature (T_{air}) was slightly stronger (Fig. 37a). Methane production is a microbiologically induced process in the anaerobic peat layer (e.g., Lai, 2009;

Whalen, 2005). Thus, methane emissions are expected to increase with an Arrhenius-type exponential dependence on soil temperature. This dependence has been reported by many studies (e.g., Hargreaves et al., 2001; Herbst et al., 2011; Jackowicz-Korczynski et al., 2010; Kim et al., 1998; Rinne et al., 2007; Song et al., 2009). Unfortunately, T_{soil} measurements are very local in space, and likely not representative for the flux footprint of EC-measurements. In contrast, T_{air} is typically more representative for larger scales, such as the footprint area. Because of its higher correlation with CH_4 fluxes, we used T_{air} as the parameter of choice for gap-filling purposes (see Section 3.3.3). However, we do not intend to imply a more direct functional relation between CH_4 flux and T_{air} by this.

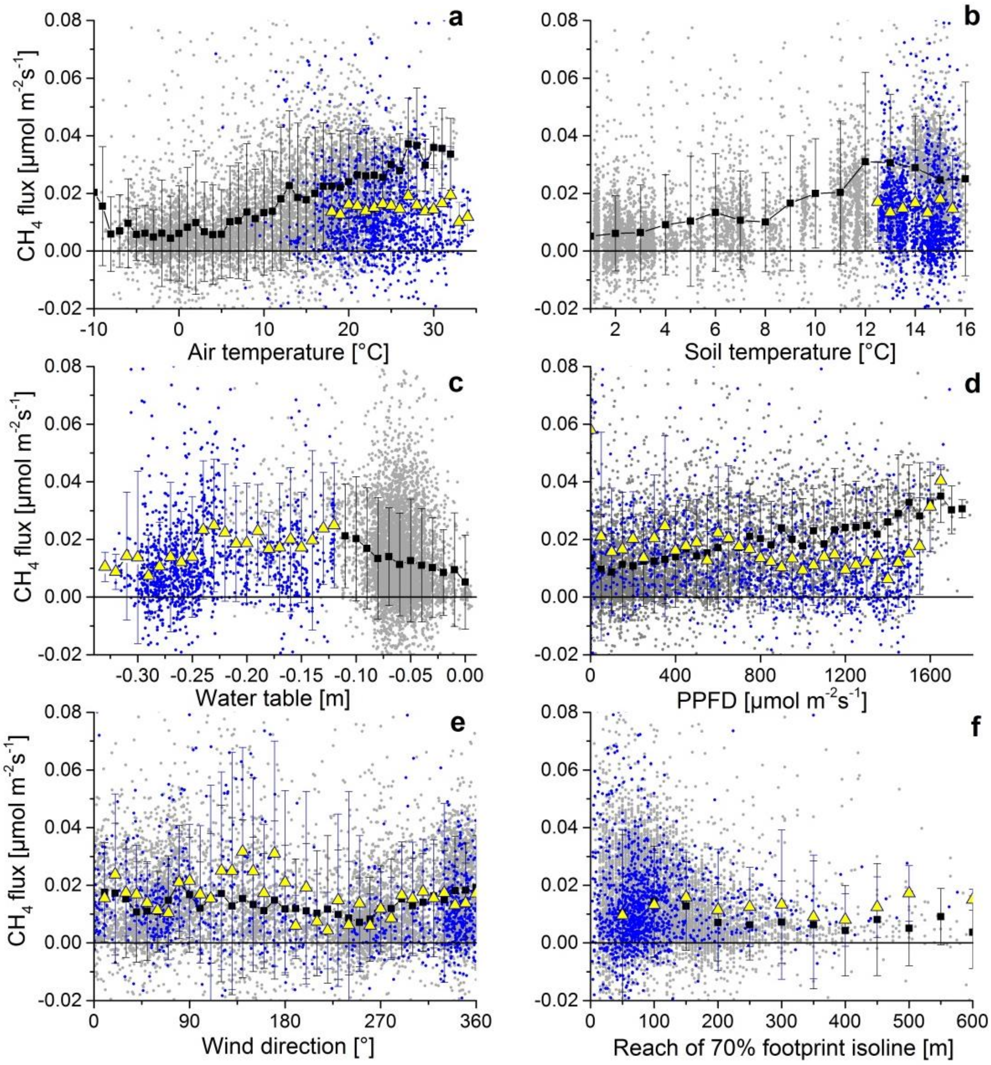

Fig. 37 Scatter plots of half‑hourly methane fluxes (gray dots) and environmental parameters: a) air temperature; b) soil temperature at 10 cm depth; c) water table depth; d) PPFD; e) wind direction; f) reach of the 70% flux footprint isoline. Presented CH_4 flux data are screened for RSSI, outliers and the u*-criterion (see Section 2.6). Blue dots represent CH_4 fluxes detected at water table depths < -0.12 m. Black squares are the binned averages of all methane fluxes, and yellow triangles are those at water tables < -0.12 m; vertical bars are the standard deviation of the range in each bin.

The binned averages (black squares) of T_{air} and T_{soil} vs. methane flux in Fig. 37a,b indicate that the dependence of methane on temperature apparently vanished at high temperature levels. The binned methane emissions decreased for T_{air} exceeding 27 °C and T_{soil} exceeding 12 °C. Other studies also found poorly defined or changing CH_4 flux-temperature relations over the course of the year (e.g., Herbst et al., 2011; Rinne et al., 2007). Rinne et al. (2007) have attributed the lack of temperature dependence in mid- to late summer to limited substrate supply, following (e.g.) Bergman et al. (1998), Chasar et al. (2000) or Wagner et al. (2005). In contrast, the lack of dependence at Schechenfilz is likely attributable to water table (WT) drawdown during warm periods. The emission of methane requires water tables to be close to the surface, because anoxic soil conditions cause methanotrophic processes to be suppressed, while methanogenic processes gain importance (Langeveld et al., 1997). This principle has been confirmed by many studies that found increasing CH_4 emissions with rising WT (e.g., Limpens et al., 2008; Sturtevant et al., 2012; Teh et al., 2011). Fluxes observed during low and relatively low WT conditions (< -0.12 m, blue dots; yellow triangles are bin-averages) did not show any clear dependence on environmental parameters (Fig. 37a-f), other than the WT threshold. One reason for this lack of dependence could be that, at low water levels, methane production and -oxidation can both occur at similar magnitudes, which may mask any CH_4-temperature relation in the net emission (Svensson and Rosswall, 1984). In a review, Drösler et al. (2008) found a critical water table depth for net CH_4 emission of -0.1 m for European peatlands. At Schechenfilz, we found reduced methane emissions when the water table fell below -0.2 m, which actually only happened during the 6-week event in summer 2013. In moderately dry conditions, with WTs between -0.2 and -0.12 m, methane emissions remained almost constant on a comparatively high level. At even greater WT depths (> -0.12 m), binned methane emissions appeared to decrease with rising water level (Fig. 37c), but the scatter is very large. Other studies, (e.g.) by Juszczak and Augustin (2013) or Rinne et al. (2007), also found that very high water levels are usually associated with cool weather conditions, and thus the negative CH_4 flux trend at high WTs in Fig. 37c is likely attributable to this temperature effect. To remove the temperature effect on the methane emissions, we normalized the observed mean daily methane fluxes by emissions calculated by the T_{air}-relation for wet conditions (water level > -0.2 m, Section 3.3.3, Fig. 40a), following Rinne et al. 2007. We found a very weak positive correlation (R^2=0.07, not shown) between water level and normalized mean daily methane flux for relatively dry conditions at water levels below -0.12 m. No such correlation was apparent in wet conditions (water level > -0.12 m). Thus, it appears that, as long as the lack of oxygen in the wet soil inhibits CH_4 oxidation, temperature showed the highest correlation with methane production. In contrast, when the water table drops below -0.12 m, the net methane emission is influenced by water table fluctuations, likely through oxidation effects in the unsaturated layer. Overall, small and short-term fluctuations of water table depth had no conclusive impact on the amount of methane emissions at the natural bog-pine site

Schechenfilz, but a persistently low WT (below -0.2 m for several days to weeks) markedly reduced the magnitude of CH_4 emissions. However, such water table drawdown events contribute to less than 10% of the observation period, and consequently, temperature fluctuations explained most of the CH_4 flux variability at Schechenfilz.

For methane exchange during wet soil conditions, we additionally found an almost linear increase of CH_4 emissions with increasing photosynthetic photon flux density PPFD (Fig. 37d), pointing to a potential link with photosynthetic assimilation. Generally, CH_4 release to the atmosphere, as well as the CH_4 production rates, are thought to be controlled by substrate supply (e.g., root exudates) to the methanogenic microorganisms in the soil, and, on longer timescales, are thus indeed coupled to the net primary production of the vegetation (Bergman et al., 2000; Friborg et al., 2000; Nykanen et al., 2003; Whiting and Chanton, 1993). Additionally, vascular plants facilitate the transport of CH_4 from the soil to the surface, and bypass CH_4 oxidation in the upper, mostly aerobic soil layer. However, the response of CH_4 exchange to these processes is lagged, and is likely apparent only for seasonal or longer variations (Fig. 38a). Because periods of high PPFD usually coincide with high temperature, the apparent correlation between PPFD and CH_4 is probably attributable to the correlation of both with air temperature. The fact that temperature–normalized CH_4 flux shows negligible correlation with PPFD confirms this notion (not shown).

Spatial inhomogeneity, due to the limited expanse of the bog-pine forest (Fig. 22a), did also not seem to affect the CH_4 flux measurements, as indicated by the flux footprint analyses (Fig. 37e, f). Variations of the flux footprint size did not notably influence the magnitude of the methane flux, neither during daytime, nor at night. However, we found that, the larger the upwind reach of the footprint, the smaller the scatter in the calculated flux. Consequently, the uniformity of the flux signal increased with increasing footprint size. This finding implies that the eddy covariance measurements represent sufficient spatial averages to account for small-scale heterogeneity at the site (Fig. 22a).

3.3.2 Daily and seasonal variation of CH_4 and CO_2

Carbon dioxide and methane exchanges both exhibited annual and daily courses (Fig. 38), but the variations of methane were less pronounced. In summer we found up to three times larger daytime methane emissions compared to those at night. Even though the scatter of the nighttime fluxes was very large, a Wilcoxon signed ranks test showed that nighttime and daytime fluxes were significantly different from each other in the summer period, but not in other seasons. In autumn and spring (transition months: April, May, October and November) the daily cycle is hardly present and vanishes in winter. In these seasons no significant difference between nighttime and daytime CH_4 fluxes could be found. Consistent with their temperature dependence, wintertime methane emission rates were considerably smaller than during summer, spring and autumn months. The mean daily methane emission was +21 mg C m^{-2} d^{-1} in summer, +12 mg C m^{-2} d^{-1} during the transition periods and almost +6 mg C m^{-2} d^{-1} in the winter months. The winter efflux of methane at Schechenfilz accounted for 16% of the annual NEMP. This proportion is similar to reported values from other sites, ranging between 9 - 22% (Alm et al., 1999a; Olson et al., 2013; Rinne et al., 2007; Saarnio et al., 2007).

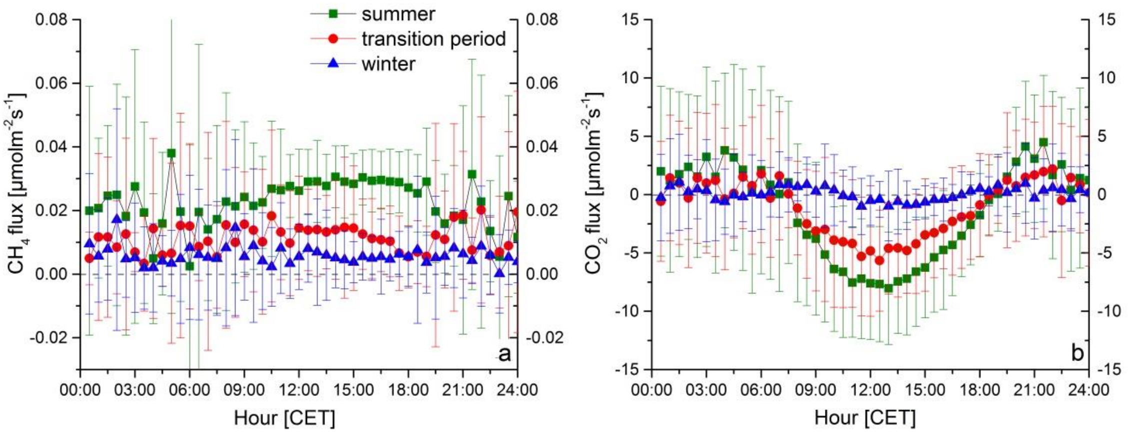

Fig. 38 Mean daily variation of a) CH_4 and b) CO2 fluxes, individually represented for summer months (July-Sept 2012: green squares), transition periods (Oct., Nov, Apr., May: red dots) and the winter months (Dec.-Mar.: blue triangles). Vertical bars denote the standard deviation of the half-hourly values around the daily mean.

Similar seasonal variations are shown in several studies (e.g., Hargreaves et al., 2001; Herbst et al., 2011; Long et al., 2010; Rinne et al., 2007; Wilson et al., 2009), but daily cycles are less frequently reported (Kim et al., 1998; Kowalska et al., 2013; Long et al., 2010; Suyker et al., 1996), and if so, only during warmer seasons like in the present study. The absence of the daily cycle at some sites is explainable by the notion that temperatures at deeper soil layers, with only minimal daily amplitudes, are likely the main drivers for methane production (Rinne et al., 2007).

3.3.3 Gap-filling strategies for annual sums of CH_4 (NEMP)

To estimate annual budgets, continuous data sets are needed. However, after quality control rejections, remaining data coverage of methane fluxes is only 36% (see Section 2.5.2), which is similar to comparable eddy covariance studies (e.g., Olson et al., 2013; Rinne et al., 2007). The high quality data coverage during daytime was 63%, but only 21% at night. Therefore, the three discussed gap-filling methods (LUT, MDV and NLR, Section 2.7) show a much better correlation between daytime observed and gap-filled fluxes than at night.

Tab. 7 Statistics of the gap-fill model comparison. Results are shown for the LUT, MDV and NLR-methods (see Section 2.4); the latter is analyzed for half-hourly and mean daily CH_4 fluxes. The LUT method was run with several parameter combinations of air temperature (T_{air}), photosynthetic active photon flux density (PPFD), water table depth (WT), and the reach of the 70% footprint isoline (Foot70). Statistical parameters presented are: absolute root mean square error (absRMSE), relative root mean square error (relRMSE) of the mean CH_4 flux and the coefficient of determination (R^2).

Method	**NEMP [g C m^{-2} a^{-1}]**	**absRMSE [µmol m^{-2} s^{-1}]**	**relRMSE**	**R^2**
LUT_T_{air}	5.6	0.016	0.68	0.36
LUT_T_{air}_WT	5.6	0.016	0.67	0.31
LUT_T_{air}_PPFD	5.4	0.015	0.62	0.38
LUT_T_{air}_Foot70	5.4	0.015	0.63	0.38
LUT_T_{air}_PPFD_WT_Foot70	**5.3**	**0.012**	**0.52**	**0.49**
MDV	4.8	0.015	0.62	0.4
NLR_half hourly	4.8	0.018	0.75	0.16
NLR_daily	**4.9**	**0.008**	**0.52**	**0.43**

The LUT method was tested with various parameters. Because air temperature (T_{air}) correlates strongest with CH_4 exchange at Schechenfilz, we adopted T_{air} as the leading parameter in the LUT. Tab. 7 shows that the relative root mean square error (relRMSE), absolute root mean square error (absRMSE) and the coefficient of determination (R^2) of the LUT are only slightly improved by including a single additional parameter: photosynthetic photon flux density (PPFD), water table depth (WT), or even the distance of the 70% footprint isoline (Foot70). However, when all four parameters were included, the gap-filling could be considerably improved. For example, the relRMSE improved by up to 16% and the R^2 by 13%. By enhancing the number of parameters, the resulting annual net methane productivity (NEMP) decreased slightly by 0.3 g C m^{-2} a^{-1}. Although air temperature turned out to correlate best with CH_4 at the natural bog-pine site Schechenfilz, the temperature dependence alone was not sufficient to explain the observed short-term variability of methane.

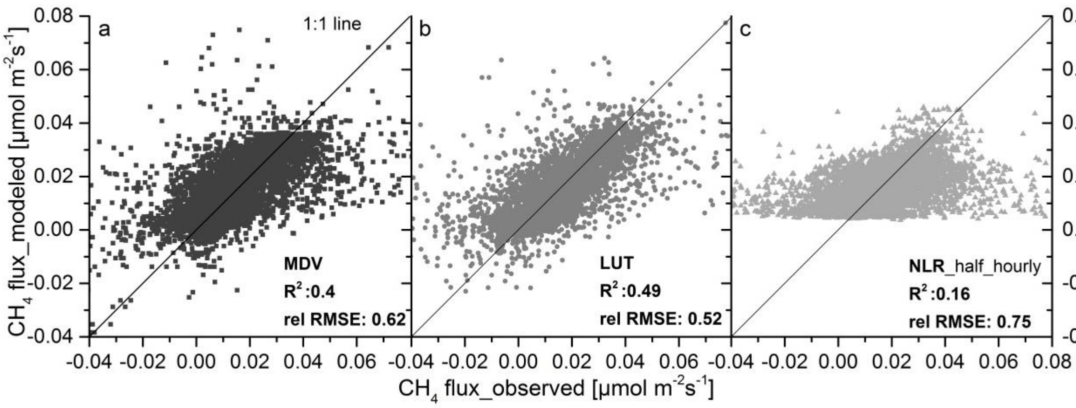

Fig. 39 Comparison of measured and predicted 30-min-CH_4 fluxes. For modeling we evaluated three different gap-filling methods a) the mean daily variation (MDV); b) a look up-table (LUT), reference parameters are T_{air}, PPFD, water table depth and the reach of the 70% footprint isoline and c) a non-linear regression (NLR) between the half-hourly methane flux and air temperature. The black line is the 1:1 line.

With the other methods examined (MDV, NLR) the annual NEMP estimate became 10% smaller. However, these latter methods exhibit a much poorer performance than with the LUT, which scored highest. The NLR_half-hourly method performed worst, indicated by the small R^2 and the large relRMSE (see also Fig. 39). Error estimation by bootstrapping, using 10,000 random samples (see Section 3.3.5), determined a gap-filling uncertainty of 13.8% for the NLR, 6.7% for the MDV, and 6.0% for LUT method, confirming the relatively good performance of the LUT method and the much poorer result of the NLR approach. Thus, the LUT-approach was adopted as the most appropriate gap-filling method for the Schechenfilz dataset.

The half-hourly methane fluxes are strongly affected by the choice of gap-filling method, while the annual NEMP variation amounts only to ±0.5 g C m^{-2} a^{-1} or 10% (Tab. 7), indicating that the annual sums are comparatively conservative.

Long data gap

Because the data gap of seven weeks between 15 May and 04 July 2013 (Section 2.5.2) was too long for gap-filling based on half-hourly fluxes with the LUT method, we applied the NLR_daily method, describing an exponential dependence of daily total methane flux on daily mean air temperature. Similar daily gap-filling methods were used in many other studies (e.g., Hargreaves et al., 2001; Hendriks et al., 2007; Herbst et al., 2011; Rinne et al., 2007; Schrier-Uijl et al., 2010). Statistical parameters resulting from the comparison between modeled and observed values (Tab. 7) indicate a good performance of the NLR_daily approach. However, for the statistical model evaluation we excluded periods with low water table, since these cannot be represented by the NLR_daily model (Fig. 40). Fortunately, such low WT conditions did not occur during the long data gap, where this method was applied. Including the water table depth as a second fitting parameter in the NLR_daily did not improve the result (not shown).

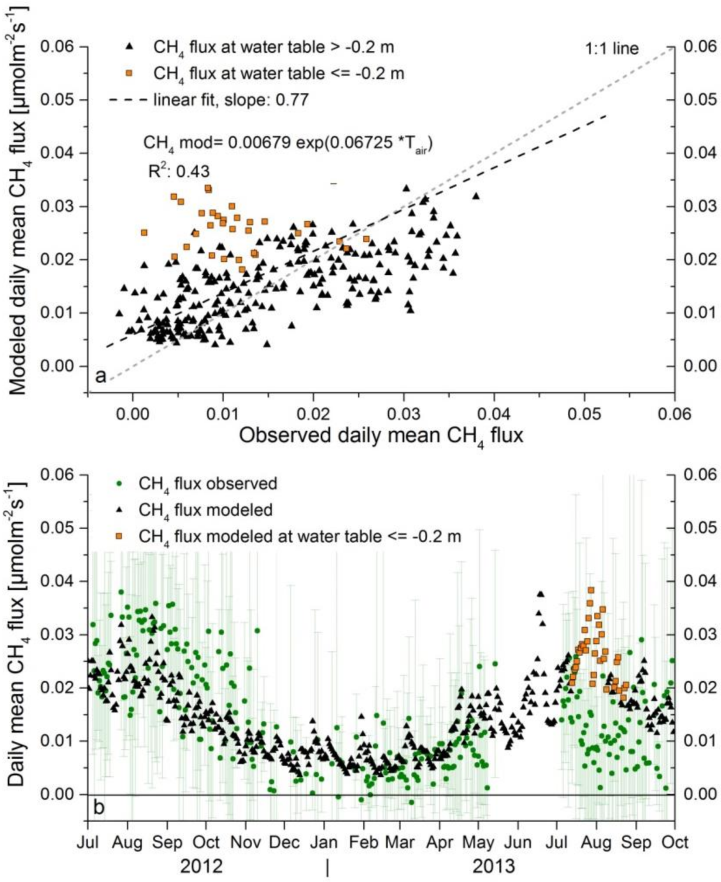

Fig. 40 a) Comparison of measured and predicted mean daily CH_4 fluxes based on an Arrhenius-type exponential model. Orange squares: mean daily methane fluxes at low water table depths of < -0.2 m; these fluxes were not considered in the model-evaluation; b) course of measured (green dots) and modeled (black triangles, orange squares) mean daily methane fluxes. Vertical bars denote the standard deviation of the half-hourly values around the daily mean.

3.3.4 Annual balances of CO_2, CH_4 and GWP

After gap-filling, we estimated an annual (July 2012-June 2013) carbon loss as methane of +5.3 ±0.34 g C m^{-2} a^{-1} (equivalent to +7.0 ±0.47 g CH_4 m^{-2}a^{-1}). The contribution of the methane emissions to the global warming potential of the 100-yr time horizon is expressed as CO_2 equivalent. Considering a CH_4 global warming potential that is 25 times stronger than that of CO_2 (IPCC, 2007), the annual global warming potential of methane amounted to +176 ±11.7 g [CO_2 eq.] m^{-2} a^{-1} (Tab. 8). For the same period, the annual CO_2 uptake was estimated at -62 ±20 g C m^{-2} a^{-1}, which is equivalent to a global warming potential of -226 ±73 g [CO_2 eq.] m^{-2} a^{-1}. N_2O fluxes can be neglected at nutrient poor, natural peatland sites such as Schechenfilz (Laine et al., 1996; Laurila et al., 2012). Thus we estimated the total greenhouse gas balance from the global warming potentials of CO_2 and CH_4, resulting in -50 ±74 g [CO_2 eq.] m^{-2} a^{-1}.

Tab. 8 Annual balances of CO_2 and CH_4 expressed as mass C, mass CH_4 and mass CO_2 respectively as well as their global warming potential with 100-yr. time horizon (GWP_{100}). Additionally presented are the total carbon balance and the total GWP balance.

	CH_4	**CO_2**	**Total**
Carbon‑balance [g C m^{-2} a^{-1}]	+5.3 ±0.34	-62 ±20	-56.5 ±20
[in gCH_4 m^{-2} a^{-1} and g CO_2 m^{-2} a^{-1}]	(+7.0 ±0.45)	(-226 ±73)	--
GWP_{100}‑balance [g [CO_2 eq.] m^{-2} a^{-1}]	+176 ±11.7	-226 ±73	-50 ±74

Since negative radiative forcing (cooling), caused by the CO_2 uptake, offsets the global warming potential from methane emissions, the natural bog-pine site Schechenfilz turned out to be a small net sink of greenhouse gases. Although this result is not statistically significant due to the relatively large uncertainty of the GWP balance, our results indicate that the site is more likely a greenhouse gas sink than a source. Obviously, this result is valid only for a time horizon up to 100 years. For longer time scales the GWP of CH_4 decreases relative to CO_2 (e.g., Frolking et al., 2006), and thus, the greenhouse gas sink functioning of peat bogs like Schechenfilz increases with longer time horizons.

In contrast, at many natural or restored peatland sites, the annual CH_4 emissions, with their higher global warming potential, appear to easily offset the annual CO_2 uptake (e.g., Bäckstrand et al., 2010; Beetz et al., 2013; Drösler et al., 2008; Olson et al., 2013; Rinne et al., 2007; Saarnio et al., 2007)

The annual course of the GWP (Fig. 41c) shows that the bog pine ecosystem was a greenhouse gas source in the winter months and a sink in the growing season. However, this sink functioning is apparently sensitive to environmental forcing. For example, during the low water table period in 2013 of about six weeks (July 13 to August 24) the ecosystem temporally switched to a weak greenhouse gas source (+8 ±6 g [CO_2 eq.] m^{-2}), while it was a sink in the same period of the previous year (-93 ±42 g [CO_2 eq.] m^{-2}). Due to lower water levels in 2013 (14 cm lower than in the same period in 2012) methane emissions were reduced by one-third. On the other hand, respiration fluxes increased during the same low water table period due to the more aerated soil conditions, leading to 86% less net CO_2 uptake compared to 2012 (Fig. 41a). However, we note that the net ecosystem methane production (NEMP) at the bog-pine site Schechenfilz was at the lower limit of reported annual methane exchange from other natural or rewetted peatland sites (Tab. 9). According to the literature review of Saarnio et al. (2007), the NEMP of boreal ombrotrophic mires ranges between less than +1 and +16 g C m^{-2} a^{-1}.

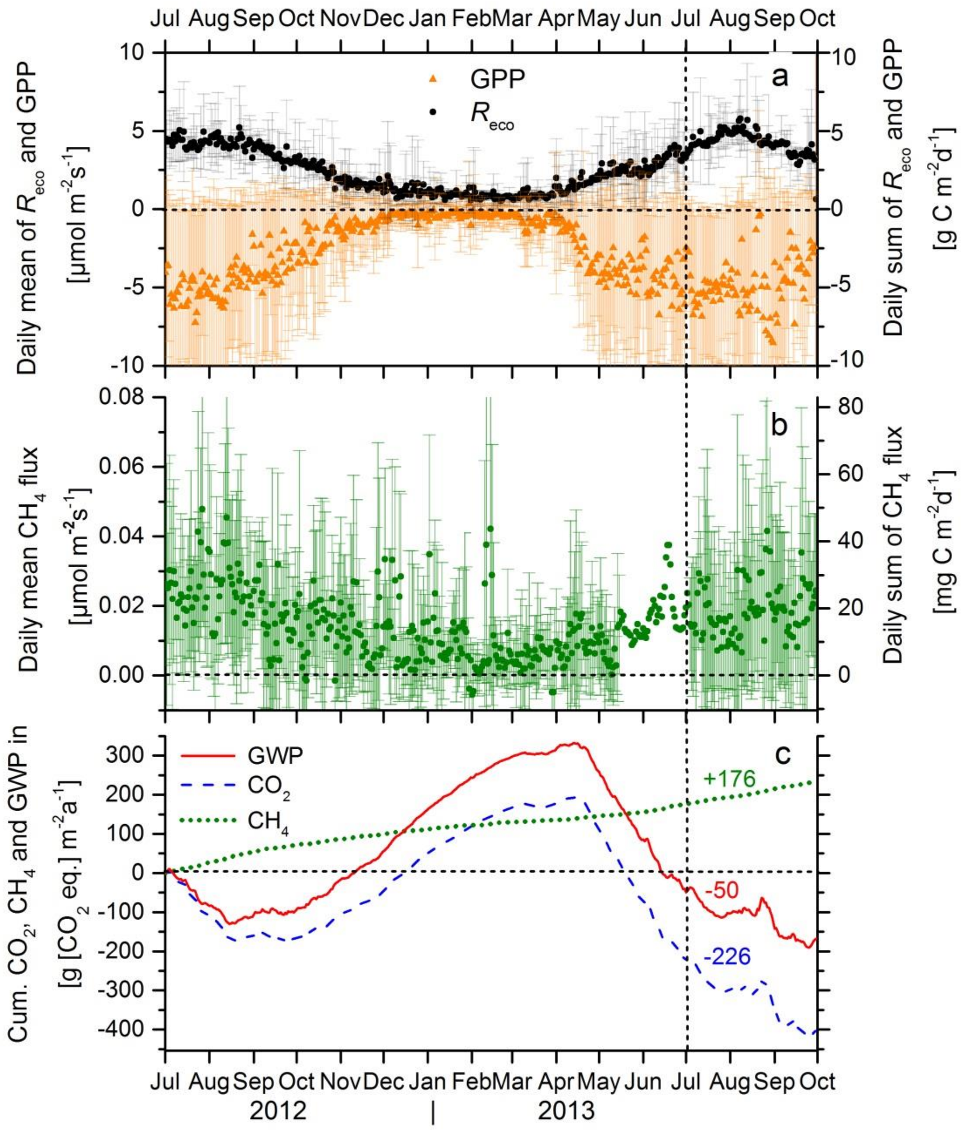

Fig. 41 a) Daily mean and sum of half-hourly estimated (measured and gap-filled) R_{eco} (black dots) and GPP fluxes (orange triangles); b) daily mean and sum of half-hourly measured and gap-filled CH_4 fluxes; c) cumulative budgets of CO_2 (blue, dashed), CH_4 (green, dotted) and the sum of their GWP (red, solid) of the one-year (period delimited by the vertical dashed line). Vertical bars denote the standard deviation of the half-hourly values around the daily mean.

Reported methane budgets of wooded bog sites are rare. Nykänen et al. (1998) detected differences in the NEMP between pine-bogs and treeless bogs by chamber measurements at two Finnish boreal sites. They found that NEMP is generally lower at wooded plots, due to mostly lower water table depths in forested bogs. At four bog-pine plots with slightly lower water tables than Schechenfilz, they estimated a similar mean NEMP of +6.1 ±1.2 g C m^{-2} a^{-1}.

Tab. 9 Review of annual net methane exchange at other pristine or restored peatland ecosystems, measured by eddy covariance method.

Ecosystem type	Location	NEMP [g C m^{-2} a^{-1}]	NEE [g C m^{-2} a^{-1}]	Reference
Temperate nutrient poor fen	northern Minnesota, USA	+11.8, +12.2, +24.9	-38.6, -27.7, -39.5	Olson et al. (2013)
Boreal fen	southern Finland	+9.4	-42.5	Rinne et al. (2007)
Boreal aapa mire	Finnish Lapland	+5.5	--	Hargreaves et al. (2001)
Subarctic mire	Sweden	+18.4, +22.1	--	Jackowicz-Korczynski et al. (2010)
Arctic polygonal tundra	Siberia, Russia	+3.2	-17.3	Wille et al. (2008)

At Schechenfilz, the CH_4 budget accounted for 9% of the net annual carbon budget, which was small compared to other peatlands, in which the estimated contributions range between 14 and 39% (Olson et al., 2013; Rinne et al., 2007; Wille et al., 2008). The larger methane to CO_2 ratios are also attributable to slightly smaller CO_2 uptake at these sites (Tab. 9). At Schechenfilz, the NEE reported in the present study, together with measurements at the same site in the years 2010 to 2012 (Hommeltenberg et al., 2014) indicate a fairly stable CO_2 uptake, ranging between -53 ±28 and -73 ±38 g Cm $^{-2}$ a $^{-1}$, which is similar to the magnitude of annual NEE at other temperate, natural but open bog sites (e.g., Lafleur et al., 2003; Lund et al., 2007; Sottocornola and Kiely, 2005).

3.3.5 Uncertainty of the annual net CH_4 exchange

Generally, measurement errors could be classified in two groups, namely random (εε) and systematic error (δ) (Moncrieff et al., 1996). The observed flux (x) of a particular quantity $\hat{x}$ incorporates both random and systematic error (δ) ($x=\hat{x}+\varepsilon+\delta$). For fluxes based on eddy covariance measurements, systematic errors include those due to non-ideal conditions, instrument errors (e.g., resulting from calibration or design errors), and systematic processing errors (e.g., detrending and coordinate rotation) (Richardson et al., 2012). To minimize known systematic errors, several correction terms or data exclusion criteria were established, including the u_* criterion to exclude conditions with insufficient turbulence, corrections for spectral loss (Moncrieff et al., 1997; Moncrieff et al., 2004), and others (see Section 2.4 and 2.6.2). However, we were not able to prevent all systematic errors, particularly those with unknown behavior. Evidence of one type of systematic error is the lack of observed energy balance closure by about 28% at Schechenfilz site. The weak energy balance closure is likely attributable to the large heterogeneity of the bog-forest site Schechenfilz. Lake Starnberg (19.5 x 4.7 km) is located just two kilometers north of the site. This distinct heterogeneity may induce secondary circulations that are not captured by the eddy-covariance system. However, evidence of systematic errors in energy fluxes are not readily transferable to fluxes of scalars (Mauder et al., 2013), and corrections for such potential systematic errors are not recommended (Foken et al., 2012b; Richardson et al., 2012) and are not performed in this study.

Random uncertainties are stochastic, thus it is generally impossible to correct for them. To estimate the random uncertainty, we have to distinguish between the random uncertainty of observed fluxes (ε_o) and of gap-filled fluxes (ε_p). Causes for random uncertainty in observed fluxes include random footprint variability, uncertainty of turbulence sampling, and random instrument uncertainty (e.g., Dragoni et al., 2007; Richardson et al., 2012). Following Finkelstein and Sims (2001), the uncertainty due to sampling of turbulence (ε_t) can be determined by the variance of the covariance of the variables *c* and *w*, including auto- and cross-covariance terms (Eq.2), where *n* is the number of samples in the averaging interval (i.e., 18,000 for 30 min of 10 Hz data), *m* is the number of samples sufficiently large to capture the integral timescale (e.g., 200 for 20 sec), and *h* varies from –*m* to *m* in the summation.

$$\varepsilon_{\mathrm{t}} = \sqrt{1/n \sum_{h=-m}^{m} (\overline{w'w'_h} \cdot \overline{c'c'_h} + \overline{w'c'_h} \cdot \overline{c'w'_h})} \ . \qquad \textbf{(14)}$$

This uncertainty estimation is implemented in the eddy covariance flux processing package EddyPro, version 4.1 (http://www.licor.com/env/products/eddy_covariance/software.html, last access: 23.11.2013). In this way the uncertainty resulting from the stochastic nature of turbulent sampling (ε_t) is estimated at ±3.6 g C m^{-2} a^{-1} for CO_2 and ±0.0104g C m^{-2} a^{-1} for CH_4, based on 9302 and 5874 observed half-hourly fluxes, respectively.

Because we have the two-tower arrangement for CO_2 measurements at Schechenfilz site (see Section 2.2) we have the possibility to estimate the total random uncertainty of observed fluxes (ε_o), including the turbulence sampling uncertainty, instrument uncertainty and the effects of flux footprint variability, following Hollinger and Richardson (2005). A comparison of CO_2 fluxes from the two tower points to some systematic differences, likely including a location bias (Schmid and Lloyd, 1999). The coefficient of determination (R^2) of the linear correlation between the CO_2 fluxes observed at the two towers is about 0.7, considering uncertainty in both axes. The slope of the linear fit is 1.24 and the intercept is -0.17 µmol m^{-2} s^{-1}).

Random uncertainty can be determined by calculating the standard deviation of the difference between half-hourly fluxes of both towers (x_1-x_2) and be expressed as:

$$\varepsilon_o = 1/\sqrt{2} \cdot \sigma(x_{1,i} - x_{2,i}) \quad \mathbf{(15)}$$

This approach resulted in an annual random uncertainty of the observed CO_2 fluxes of ±7.6 g C m^{-2} a^{-1} for CO_2. Since methane is measured only at one tower, this approach is not appropriate to estimate the methane uncertainty. However, we note the random error related to the turbulence process (ε_t) accounts for 47% of the total measurement error (ε_o) of CO_2 fluxes.

Assuming a similar flux footprint variability of CO_2 and CH_4, and a similar random uncertainty resulting from the instruments, we transferred this ratio to the uncertainty of CH_4. Based on this assumption, the annual random error of the observed methane fluxes was estimated at ±0.023 g C m^{-2} a^{-1} or 0.5% of the annual flux. However, we are aware that this value is probably an underestimation, because the signal to noise ratio of the methane analyzer was about nine times smaller than the ratio of the CO_2 analyzer, leading to a larger random instrument uncertainty for observed annual methane emissions. Furthermore, the position of the flux-footprint was the same for both trace gases, but the distribution of sources and sinks were probably different. Nevertheless, because of a lack of better knowledge we used this uncertainty for further analysis.

The random error of the predicted fluxes (ε_p) was estimated by bootstrapping, using 10,000 randomly sampled values from a normal distribution, with the mean taken from the LUT for half-hourly methane fluxes, and from the NLR for daily methane and CO_2 fluxes. The sample range is limited by the standard deviation we obtained from the LUT and NLR, respectively. This simulation resulted in a component uncertainty of CH_4 exchange of ±0.32 g C m^{-2} for the LUT modeling, and ±0.017 g C m^{-2} for the seven weeks data-gap, filled by the NLR_daily approach. The uncertainty of predicted fluxes was estimated at ±0.337 g C m^{-2} or 6.4% for annual NEMP, and ±18 g C m^{-2} a^{-1} or 29% for annual NEE.

The combined uncertainty of observed (ε_o) and predicted fluxes (ε_p) accumulates in quadrature (e.g., Richardson et al., 2012; Richardson and Hollinger, 2007):

$$\varepsilon = \sqrt{\varepsilon_o^2 + \varepsilon_p^2} \quad \mathbf{(16)}$$

This combined uncertainty estimate was +5.3 ±0.338 g C m^{-2} a^{-1} or 6.4% for annual NEMP, and -62 ±20 g C m^{-2} a^{-1} or 32% for annual NEE.

Observed methane fluxes at our site are almost entirely directed towards the atmosphere and changes in sign are rare. In contrast, the distinct daily and seasonally cycle, as well as regular changes of CO_2 release and uptake (positive or negative sign), explain the larger relative uncertainties of the net CO_2 exchange in comparison to those of CH_4.

3.3.6 Summary and conclusions

We determined an annual CH_4 efflux of +5.3 ±0.34 g Cm^{-2} a^{-1} and a CO_2 uptake of -62 ±20 g Cm^{-2} a^{-1}, resulting in a global warming potential balance of -50 ±74 g [CO_2 eq.] m^{-2} a^{-1} for the one-year measurement period July 2012-June 2013 at a natural temperate bog-pine site in southern Germany. Thus, the site turned out to be a small net greenhouse gas sink over this measurement year.

About 90% of the time, methane exchange was mainly influenced by temperature variations. However, at water tables below -0.2 m, the magnitude of methane emissions was reduced, and fluctuations were independent of temperature variations.

For the determination of the annual net ecosystem methane production (NEMP), the influence of three different gap-filling methods was investigated. The application of the non-linear regression method (NLR) resulted in very poor fits between predicted and observed half-hourly CH_4 fluxes. Results indicate better fits for fluxes modeled by mean daily variation (MDV) or a look-up table (LUT) approach. The LUT was tested for a combination of four parameters: air temperature, photosynthetic active photon flux density, water table depth and the reach of the 70% flux-footprint isoline. An increasing number of parameters improve the flux prediction by up to 16%. Since the uncertainty induced by gap-filling is smallest using the LUT-approach, this method was adopted here. However, to gap-fill an exceptionally large data gap of seven weeks, we applied an exponential Arrhenius-type function between daily methane flux and daily air temperature (NLR_daily). We found that daily averages were predicted very well by this method, except for dry periods with low water tables.

The annual mean air temperature over the observation period was almost equal to the long-term average. As the air temperature correlates best with methane exchange, the observed NEMP is likely within the range of expected annual methane emissions, but future observations, conducted as part of the European ICOS program will shed more light on this.

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4 General Discussion

The two research sites in this study are representative of managed and natural bog forest ecosystems in the pre-alpine region of southern Germany. Two years of CO_2 exchange measurements (July 2010-June 2012), by the eddy-covariance technique at these sites indicate a relatively strong annual net CO_2 uptake of -130 ±31 and -300 ±66 g C m^{-2} a^{-1} at the drained spruce forest site Mooseurach. The CO_2 uptake at the natural bog-pine site Schechenfilz, measured for more than three years (July 2010-September 2013), indicate stable but about three times lower annual net uptake, ranging between -53 ±25 and -73 ±38 g C m^{-2} a^{-1}. However, the ecosystem respiration is about two times larger at the drained site as well. Overall, the stronger net CO_2 uptake at the drained site is likely attributable to the larger photosynthetic activity of the faster growing and much taller spruces, compared to the small, slow-growing bog-pines. This is underpinned by the almost three times larger plant area index and the more than 6 times larger tree biomass of the spruces at the drained site compared to the natural site. The spruces are currently about 44 years old and are almost at their maximum productivity (Gower et al., 1996).

Annual CO_2 budgets of drained and afforested peatland sites reported in the literature are highly variable. Mostly, drained peatland forests are found to be strong annual net CO_2 sinks (> -150 g C m^{-2} a^{-1}). At some peatland forest sites the net CO_2 sink is entirely attributable to the carbon accumulation of the trees (Hargreaves et al., 2003; Meyer et al., 2013; von Arnold et al., 2005), but at other sites even the soil is a net CO_2 sink (Lohila et al., 2011; Minkkinen et al., 2002). Simola et al. (2012) estimated a mean carbon loss of +150 g C m^{-2} a^{-1} at 37 drained forestry peatland sites in central Finland by re-sampling the peat stratum after about 30 years. At these sites, the annual soil carbon flux varied strongly between -200 to +800 g C m^{-2} a^{-1}, but the amount of soil carbon loss was apparently independent of site fertility, or post-drainage timber growth. A direct comparison of our results with the carbon budgets from other sites is difficult, because site specific factors (e.g., nutrient supply, annual mean temperature, trees stock) are very different. Maljanen et al. (2010) note that the NEE in forestry-drained peatlands is highly dependent on the age of the tree stand as well as the weather conditions. However, reliable data providing a more detailed understanding are still very scarce.

The annual net CO_2 uptake rate at the natural bog-pine site Schechenfilz is in a similar range as non-forested bog and fen sites in boreal or temperate regions (Lafleur et al., 2003; Lund et al., 2007; Rinne et al., 2007; Sottocornola and Kiely, 2005). Apparently, the slow-growing bog-pines did not enhance the net CO_2 uptake at Schechenfilz site compared to non-forested natural peatlands.

At natural bog sites CH_4 emissions are very important for the overall greenhouse gas (GHG) balance. One year (July 2012-June 2013) of eddy covariance measurements of methane exchange at the natural site indicated an annual net release of +5.3 ±0.34 g C m^{-2} a^{-1} (of CH_4) equivalent to a GWP of +177 ±11 g [CO_2- eq.] m^{-2} a^{-1} for a 100-year reference time scale. The net CO_2 uptake in the same period was -62 ±20 g C m^{-2} a^{-1} (=-226 ±73 g [CO_2- eq.] m^{-2} a^{-1}). Thus, the annual methane exchange is of minor importance in terms of the net carbon balance. However, due to the stronger global warming potential of methane, even such relatively low emissions reduce the climate mitigation effect of the CO_2 uptake by about 80%. The sum results in an almost neutral GWP-balance of -50 ±74 g [CO_2- eq.] m^{-2} a^{-1}. This result can be considered the total greenhouse gas balance at the natural site, because N_2O fluxes are general negligible at such nutrient poor pristine bog sites (e.g., Martikainen et al., 1993; Minkkinen et al., 2002; Regina et al., 1996). Methane emissions at the present bog-pine site are in the lower range of values from pristine sites in the literature (e.g., Olson et al., 2013; Rinne et al., 2007; Saarnio et al., 2007). This low value is likely attributable to the relatively sparse coverage of sedges, whose aerenchymous tissues serve as a bypass for methane through the aerated soil layer (e.g., Lai, 2009; Whalen, 2005). Furthermore, flooded periods which are thought to lead to enhanced methane emissions, never occurred in the three observation years at the bog-pine site Schechenfilz.

At drained and afforested peatland sites, both methane and N_2O exchange play only a minor role in the overall GHG balance (Lohila et al., 2011; Meyer et al., 2013; Yamulki et al., 2013). Chamber measurements within the flux footprint at our drained forest site Mooseurach (run by the University of Applied Sciences Weihenstephan Triesdorf (HSWT)) indicated methane uptake and minor N_2O release rates (C. Förster, personal communication, 2012) that are so small, to make them irrelevant, compared to the large CO_2 uptake rates.

For the sake of completeness, it should be mentioned that the loss of carbon as dissolved organic carbon (DOC) in the run-off from the bogs is not considered in this study at either site, but usually the loss of DOC can be expected to be only a small fraction of the GHG exchange balance (e.g., Buffam et al., 2007; Olsrud and Christensen, 2004).

In a survey study Drösler et al. (2008) found generally almost climate neutral GWP balances for natural, restored, but also afforested European peatlands. This is in contrast to the strong uptake we observed for Mooseurach. Overall, the GHG exchange at the drained site indicated an annual scale climate change mitigation potential that was about 10 times larger than at the natural ecosystem. It should be noted that chamber and eddy covariance measurements can only reflect the current state of ecosystems, usually without considering potential effects and consequences of harvesting. However, at natural sites, ecosystem renewal is a continuous process, with morbidity and regrowth of plant individuals, rather than a successional transition, where entire stands or whole ecosystems are affected at once. Thus, soil effluxes from the pristine peat, and gross primary production of the natural bog-pine ecosystem are likely stable over several years.

In contrast, particularly in managed monocultures with nearly uniform age-structure carbon balances are highly variable over the rotation period, and at least one whole life-cycle of the vegetation community should be considered for the GHG-exchange evaluation. Taking into account the carbon loss from the peat and the carbon accumulation by the spruces since forest planting 44 years ago, we estimated a net loss of +134 t C ha^{-1} for this period (see Section 3.1.5). According to the local forestry department (Amt für Ernährung, Landwirtschaft und Forsten, Weilheim), the expected rotation period of the spruces in the research area is about 60 years (personal communication, H. Neustifter, 2013). Assuming an annual net CO_2 uptake at the current rate (-1.57 t ha^{-1}, Section 3.1.3, Fig. 30) for a further 16 years, the net loss would be reduced by -25 t ha^{-1} to a total loss of +109 t ha^{-1} for the expected rotation period of 60 years of the spruces in Mooseurach. Based on this notion, we roughly estimate a life-cycle averaged annual carbon loss of +182 g C m^{-2} a^{-1} (+1.82 t ha^{-1} a^{-1}), which is equivalent to a GWP of +660 g [CO_2- eq.] m^{-2} a^{-1}. The long-term carbon loss rate is only a rough estimate, but important for an unbiased discussion of the flux measurements that presents only a “snap-shot” of the current state of the ecosystem.

For illustration, the roughly estimated life-cycle averaged annual CO_2 release of the drained forest in Mooseurach (70 ha) is comparable to the CO_2 emissions of 180 cars (compact car, e.g., VW Golf: 129 CO_2 km^{-1} each driving about 20,000 km per year). In the German pre-alpine region, about 1/3 (25,470 ha; source: Geobasisdaten, Bayerische Vermessungsverwaltung) of all peatlands are peat-forests, and they are mostly artificially drained and managed ecosystems. Up-scaling of the estimated long-term net carbon emission of Mooseurach to the entire pre-alpine peat-forest area results in an annual carbon loss amounting to 46,355 t C a^{-1}. This rate is comparable to the CO_2 emissions of 65,000 compact cars with a kilometer performance of 20,000 km a^{-1}. The extrapolation of site specific carbon balances to larger areas is fraught by large uncertainties, as estimated carbon balances of drained peat forests are site specific and strongly depend on drainage, forest-management, nutrient status and vegetation. However, this scenario illustrates the great potential of carbon mitigation by peatland restoration. The “climate friendly” restoration of Germans bogs and fens (1.3×10^6 ha) theoretically could offset GHG emissions by up to 35 million tons CO_2 equivalents per year (Freibauer et al., 2009). However, “climate friendly” peatland restoration is a hotly disputed issue. Restoration can take several years to decades, until an equilibrium of GHG-exchange is approached and formerly drained peatlands reach a balanced state again. Rewetting of drained peatlands initially induce methane emissions that are often much larger than those of natural, undisturbed mires (Laine et al., 2007; Wilson et al., 2009). For example, a German pre-alpine bog was still a GHG source 12 years after restoration (Drösler, 2005). On the other hand, an abandoned agricultural peat meadow in the Netherlands was a small GHG sink after only 10 years of restoration, in spite of relatively strong CH_4 emissions (Hendriks et al., 2007), and a southern Finnish cut-away peatland was found to have returned to a CO_2 sink after 20 years of rewetting (Tuittila et al., 1999). The only generalization possible seems to be that the time required and overall effects on GHG balance is highly site specific. As peatland restoration may initially induce methane emission hotspots, several studies even plead that peatland afforestation could be an attractive scenario (e.g., Byrne and Farrell, 2005; Hargreaves et al., 2003; Wilson et al., 2009), if climate cooling und carbon mitigation is the primary objective. This is contrary to our findings. Also Byrne et al. (2004) note that the fact that forestry peatlands emit less GHGs than undisturbed bogs or fens has to be viewed with caution, because underling studies often suggest a mild drainage only, at which CH_4 emissions are reduced but peat formation still goes on. In addition, drained and afforested peat-soils are generally considered as CO_2 sources in the recently published wetland-supplement of the IPCC 2006 (Drösler et. al., 2013). They reported a CO_{2-} emission-factor for temperate forestry drained peat of +260 g C m^{-2} a^{-1}. Furthermore, undisturbed peatlands are the only natural ecosystems that can accumulate carbon continuously and over periods of thousands of years (Clymo, 1984).

Aside from the benefit of carbon storage, it should be mentioned that natural peatlands have some more ecosystem services. The most important are flood prevention, improvement of water quality, enhancement of peatland typical biodiversity; peatlands are important resting places for migratory birds and favorite recreation areas. As stipulated in Art. 30 II of the German national regulation on environmental protection (§ 30 BNatSchG) all existing peatlands are legally protected against destruction and human interventions. On the other hand, about 95% of all Germany peatlands are threatened by human interference, and they are one of the most endangered habitats in Germany.

Even a part from the detrimental effects of land use change, peatlands are extremely vulnerable to climate change (International Peat Society, 2008). For example, Bellamy et al. (2005) examined the carbon losses from soils across England and Wales between 1973 and 2003. They found that peat soils lost more carbon than other soil types. This was attributed to the high sensitivity of peat soils to increasing temperatures and changes in rainfall distribution and frequency.

Climate change is expected to cause rising air temperature and a reduction of annual precipitation in southern Bavaria (IPCC, 2007). These changes are likely to persistently reduce the water table depth, which has an impact on CO_2 and CH_4 exchange. Lower water table will cause an aerobic milieu in the upper soil-layer, and thus methane will probably no longer be emitted. However, the available oxygen in the soil leads to soil-carbon oxidation, and therefore greater CO_2 emissions. The CO_2 emissions rate is additionally increased by rising temperatures, which overall lead to peat loss in the long-term. On the other hand, dryer and more aerated soil conditions may give rise to more productive plant species, leading to higher gross primary productivity and thus stronger carbon fixation. In this scenario the impact of climate change on the GHG exchange of natural peatlands is similar to the effect of drainage. However, consequences of climate change could be diverse, thus the true impact on the GHG exchange of peatlands remains uncertain.

To confirm the present results and to constrain their uncertainty, longer measurement periods are required, especially at the drained site where the NEE is very sensitive to environmental forcing (Section 3.1.4) and the applied data gap-filling procedures (Section 3.2.2). Ongoing measurements would open opportunities for several interesting studies: due to the ceased maintenance of the drainage system, and the therefore expected rising water level, the vegetation composition and soil conditions will likely change at Mooseurach within the next years or decades. Observing current changes of NEE due to harvesting and natural mortality of the spruces would be a unique opportunity to enhance the knowledge of NEE during proceeding peatland forest rewetting. Conducting chamber measurements on the bare soil, would provide more information about the heterotrophic respiration from the degrading peat, and may improve the validation of the long-term carbon loss estimate from the peat. At Schechenfilz, the two-tower arrangement is perfectly suited to improve the uncertainty estimate of net ecosystem methane exchange.

Both sites are earmarked as sites in the ICOS-network (see Section 1.5IV), with the objective of long-term observations of at least 20 years. Thus, at both sites many opportunities will arise for future investigations of scientific and climate policy relevance.

5 Summary and Conclusions

This thesis is one of the first direct comparison studies of net CO_2 exchange (NEE) at a natural (Schechenfilz) and a drained (Mooseurach) bog-forest ecosystem of the temperate climate zone. The natural site is characterized by a bog-pine forest, growing on a pristine peat layer. The drained site is afforested with spruces. Both sites are located in the pre-alpine region of southern Germany. Due to the small distance of only 10 km between the sites, atmospheric conditions are very similar. Thus differences of the CO_2 exchange can be largely attributed to site-specific factors, such as soil conditions, vegetation composition, and management. To discuss the full greenhouse gas balance at the natural site, CO_2 measurements were continued and supplemented by CH_4 measurements for 15 months, after the two year CO_2-site comparison.

Influence of environmental parameters on CO_2 and CH_4 exchange:

The CO_2 exchange at the drained spruce forest site Mooseurach is highly sensitive to the variability of environmental forcing. Particularly changes in temperature and precipitation frequency affect the CO_2 exchange. Whether dry or warm periods support or reduce net carbon uptake depends on the state of phenology of the spruces, and on the soil temperature (Section 3.1.2 and 3.1.4).

At the natural bog-pine site Schechenfilz, the annual course of NEE is very similar in the three measurement years. The impact of dry periods on NEE is small, because the high water retention of the bog acts as a buffer. Only one period of markedly strong water table drawdown (five times lower than the average of three years, due to three rainless weeks in midsummer 2013), lead to a considerably reduced CO_2 uptake compared to other years (Section 3.1.2 and 3.1.4).

The methane emissions at the natural site usually increase with rising air temperature, except for periods when the water table depth drops below -0.12 m. In such periods methane emissions are considerably reduced, and independent of further rising temperature (Section 3.3.1). This is likely due to oxidation of methane in the unsaturated layer, before it is release to the atmosphere.

Variance of NEE and NEMP due to gap-filling method:

The NEE of CO_2 is strongly influenced by the chosen gap-filling method at Mooseurach (± 250 g C $m^{-2}a^{-1}$), but almost independent of the gap-filling method at Schechenfilz (± 18 g C $m^{-2}a^{-1}$). The smallest uncertainty was achieved with the non-linear regression method at both sites. This method was adopted for the determination of annual NEE (Section 3.2.2).

The net ecosystem methane production (NEMP) at Schechenfilz varies by about 10%, depending on gap-filling method. The best prediction of half-hourly methane flux was achieved by a look-up table method. However, a long data gap of 7 weeks was gap-filled by an Arrhenius-type exponential relation that depends on daily mean air temperature. This gap-filling strategy fits very well, except for extraordinary water table drawdown event in summer 2013 (water table < -0.2 m) (Section 3.3.3).

Budgets of NEE and NEMP:

The results of the two years of eddy covariance measurement presented in this study indicate stronger CO_2 uptake of the drained bog forest than of the natural bog forest. Currently, the 44 year old spruces are thought to be almost at their maximum productivity (Gower et al., 1996), and can compensate the soil carbon efflux. Including methane fluxes to the full greenhouse gas (GHG) balance at the natural site, the CO_2 uptake induced climate change mitigation potential of the natural site is reduced by about 80%. As N_2O fluxes are generally negligible at nutrient-poor bog sites, we thus estimated an overall GHG balance of -50 g C [CO_2-eq.] m^{-2} a^{-1} for the one-year full observation period with CH_4. At the drained site, neither CH_4 uptake nor N_2O exchange was measured, but in the face of the strong CO_2 uptake at this site, these other greenhouse gases likely have only a minor contribution to the overall GHG balance (Section 3.3.4).

However, this two year measurement period can only reflect the current stage of the observed ecosystems. For meaningful comparisons between peatland forests and full evaluation of the climate impact of different land uses requires a longer-term perspective. A rough long-term carbon estimate resulted in an average annual net carbon release of +134 g C m^{-2} a^{-1} at the drained spruce forest site, representing a considerable emission source. To compensate the carbon loss from the early years of the spruce plantation, the spruces would have to assimilate carbon for another 100 years at the current rate. In comparison, the natural bog ecosystem is seen to be a minor, but comparatively robust GHG sink (Section 3.1.5).

6 Appendices

A1 Data of forest growth modeling (Spruce forest at Mooseurach)

Estimated carbon and biomass increase of the last 20 years (status as of 2009) the spruces at the drained site. Data were provided by Steffi Röhling (Technical University of Munich, Center of Life and Food Sciences Weihenstephan, Chair for Forest Growth and Yield), and based on biometry and common allometric relations.

Tab. 10 Estimated carbon and biomass increase of the spruces at the drained site, based on biometry and common allometric relations (S. Röhling, personal communication).

Carbon	**Above surface**	**Below surface**
	t_{DM} ha^{-1}	**t_{DM} ha^{-1}**
1988	34.1	6.7
1989	41.8	8.4
1990	48.9	9.9
1991	55.1	11.3
1992	59.6	12.3
1993	64.3	13.3
1994	68.2	14.2
1995	72.7	15.2
1996	77.8	16.4
1997	84.1	17.9
1998	89.9	19.2
1999	95.2	20.4
2000	101.2	21.8
2001	105.8	22.9
2002	112.1	24.4
2003	115.0	25.0
2004	117.3	25.6
2005	121.3	26.5
2006	126.6	27.8
2007	134.8	29.7
2008	140.8	31.2
$I_{B\ 1989-2008}$	**7.83 t ha^{-1} a^{-1}**	

Carbon	**Above surface**	**Below surface**
	t_{DM} ha^{-1}	**t_{DM} ha^{-1}**
1988	16.9	3.4
1989	20.7	4.2
1990	24.3	5.0
1991	27.3	5.6
1992	29.6	6.1
1993	31.9	6.7
1994	33.9	7.1
1995	36.1	7.6
1996	38.6	8.2
1997	41.8	8.9
1998	44.7	9.6
1999	47.3	10.2
2000	50.3	10.9
2001	52.6	11.4
2002	55.7	12.2
2003	57.1	12.5
2004	58.3	12.8
2005	60.3	13.3
2006	62.9	13.9
2007	67.0	14.9
2008	70.0	15.6
$I_{C1989-2008}$	**3.30 t C ha^{-1} a^{-1}**	

Total_biomass (status quo 44 years)	**172 t ha^{-1} a^{-1}**

Total_carbon (status quo 44 years)	**86 t C ha^{-1} a^{-1}**

A2 Estimate of carbon loss due to peat degradation at Mooseurach

- 1 m peat loss in the last 70-80 years (Fig. 16): leads to a subsidence rate of 1.25-1.43 cm a^{-1}
- Following Leifeld (2011a) 50% of the peat subsidence is due to carbon oxidation

→ Thus, the peat loss due to carbon oxidation can be estimated at 0.63-0.71 cm a^{-1}

- Assuming similar soil conditions (bulk density, carbon content) of the top-soil layer in the last decades, the carbon loss can be estimated as illustrated in Tab. 11.

Tab. 11 Estimate of carbon loss at the drained site Mooseurach. Data of C_{org} and dry bulk density of the top soil layer were provided by N. Rosskopf (Humboldt University Berlin, Faculty of Agriculture and Horticulture, Department of Crop and Animal Sciences)

Depth	C_{org}	Dry bulk density	Proportion of carbon (dry matter)	Proportion of carbon (dry matter)	Annual carbon loss, based on annual volume loss of 0.63 cm a^{-1}	Carbon loss since forest planting (44 years ago)
	[%/dry matter]	g cm^{-3}	g cm^{-3}	t C ha^{-1}	t C ha^{-1}	t C ha^{-1}
0-20	**49.73**	**0.154**	**0.077**	**770**	**4.9**	**216**

→Annual C-loss from the peat 5 t C ha^{-1} (500 g C m^{-2} a^{-1})

Data were provided by Niko Rosskopf (Humboldt University Berlin, Faculty of Agriculture and Horticulture, Department of Crop and Animal Sciences).

A3 Comparison between CO_2 and latent heat fluxes (LE), measured by open (LI-7500) and cloth path (LI-7200) infrared gas analyzers

For CO_2 and water vapor measurements, two different measurement instruments were used at Mooseurach (LI-7200, closed-path) and Schechenfilz (Li-7500, open path). The measurement principle of LI7200 is based on the absolute Non-Dispersive-Infrared (NDIR) design of the LI7500 (Burba et al., 2010b), which leads to a good agreement between the two measurement devices (Burba et al. 2009). Since 2012 both instruments are implemented at Schechenfilz, separated by 35 m. The agreement between the measurements (CO_2 fluxes and latent heat fluxes (LE)) is illustrated in Fig. 42 and Fig. 43.

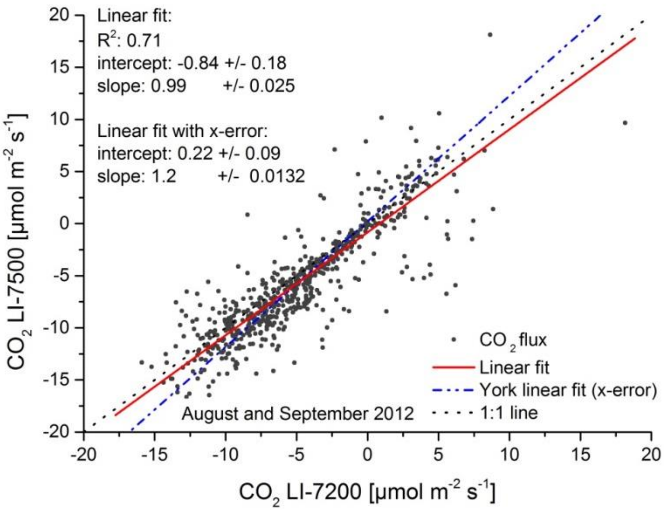

Fig. 42 CO_2 flux measured by LI7200 vs. LI 7500. Quality controlled data of August and September 2013 measured at the heterogeneous bog-pine site Schechenfilz. Measurement devices were separated by 35m.

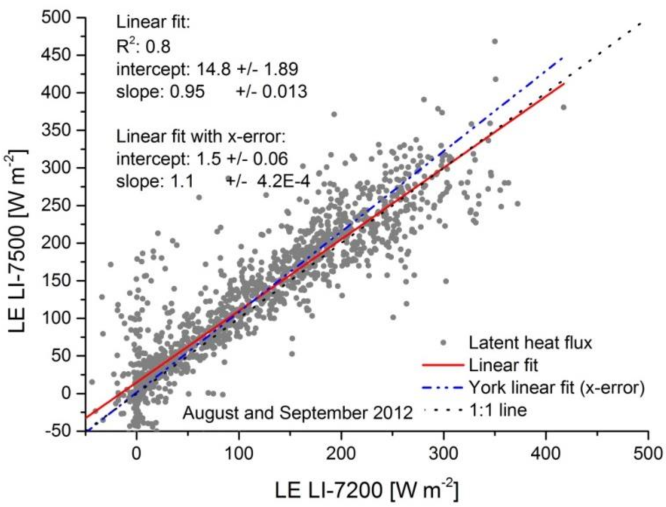

Fig. 43 Latent heat flux (LE) measured by LI7200 vs. LI 7500. Quality controlled data of August and September 2013 measured at the heterogeneous bog-pine site Schechenfilz. Measurement devices were separated by 35m.

A4 Energy balance closure

The energy balance closure was determined by Equation 17:

$$R_{net}\text{-}G= H+LE. \quad \textbf{(17)}$$

The energy budget of the four key components: net radiation (R_{net}), soil heat flux (G), sensible heat flux (H) and latent heat flux (LE) should sum to zero as required for conservation of energy.

We determined an energy balance closure of 75% at the drained spruce forest at Mooseurach, and a closure of 72% at the natural bog-pine site Schechenfilz (Fig. 44).

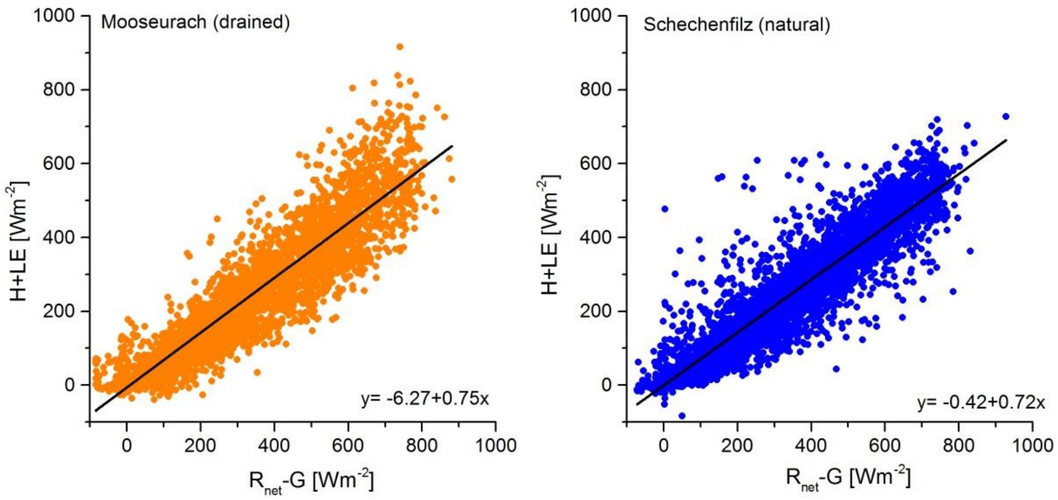

Fig. 44 Annual energy balance closure at the drained site Mooseurach (left) and the natural site Schechenfilz (right) over the year 2011. R_{net} is the net radiation, G the soil heat flux, H the sensible heat flux and LE the latent heat flux.

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§ 30 BNatSchG (Bundesnaturschutzgesetz) n.F. (neue Fassung) in der ab dem 01.03.2010 geltenden Fassung durch B. v. 26.07.2010 BGBl. I S. 970.

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