TECHNISCHE UNIVERSITÄT MÜNCHEN

Lehrstuhl für Renaturierungsökologie

In Kooperation mit

Fachgebiet für Vegetationsökologie

Hochschule Weihenstephan-Triesdorf

Assessment and modeling of greenhouse gas emissions (CO_2, CH_4, N_2O) from different land-use types and management practices on drained fen peatlands and associated organic soils – case study Freisinger Moos

Tim Eickenscheidt

Vollständiger Abdruck der von der Fakultät Wissenschaftszentrum Weihenstephan für Ernährung, Landnutzung und Umwelt der Technischen Universität München zur Erlangung des akademischen Grades eines

Doktors der Naturwissenschaften

genehmigten Dissertation.

Vorsitzender: Univ.-Prof. Dr. J. Schnyder

Prüfer der Dissertation:

           1. Univ.-Prof. Dr. J. Kollmann

           2. Prof. Dr. M. Drösler (Hochschule Weihenstephan-Triesdorf)

           3. Hon.-Prof. Dr. J. Augustin (Martin-Luther-Universität Halle-Wittenberg)

Die Dissertation wurde am 15.04.2015 bei der Technischen Universität München eingereicht und durch die Fakultät Wissenschaftszentrum Weihenstephan für Ernährung, Landnutzung und Umwelt am 17.07.2015 angenommen.

Summary

In the last century, drainage and intensification of agriculture turned German peatlands into hot-spots for greenhouse gas (GHG) emissions. Particularly peatlands converted into croplands and intensive grasslands have been regarded as the main producers of carbon dioxide (CO_2) and nitrous oxide (N_2O). Despite high research efforts in the last decade, there is still a high uncertainty in defining reliable emission factors (EF) for the climate reporting according to the Intergovernmental Panel on Climate Change (IPCC) guidelines. This is due to the fact that the few observed GHG fluxes on these soils show a large temporal and spatial variability, and the identification of histosols is challenging since several national and international classification systems exist for organic soils. Most studies have focused on typical drained peatlands, which are very rich in organic carbon (C_{org}), whereas former peatlands and associated organic soils containing <12% C_{org} were neglected, although it is unknown whether they emit less GHG's compared to typical drained peatlands. This issue was examined in the first of three studies (**PUBLICATION I.**). This study addressed the questions as to what extent the soil organic carbon (SOC) content affects the GHG release from different land-use types and management practices on drained organic soils. For this purpose, two soil types with contrasting SOC content were chosen, i.e. a mollic Gleysol (9.4–10.9% C_{org} in 0–10 cm soil depth) and a sapric Histosol (16.1–17.2% C_{org} in 0–10 cm soil depth). On both soils, four treatments were established once, two croplands (maize or oat) and two intensive grasslands (fertilized with cattle slurry or biogas digestate). GHG fluxes (CO_2, N_2O and methane (CH_4)) were measured and modeled over a period of two years. No differences were found in the estimated net CO_2 balances or global warming potentials (GWP) between the two soil types. Thus, like typical drained peatlands also drained mollic Gleysols can be considered as hot-spot for GHG emissions, provided that they are intensively managed as cropland or grassland. Estimated GWPs in this study, calculated for a time horizon of 100 years, revealed a very high release of GHG's from all treatments investigated ranging from 1837 to 7095 g $CO_{2\ eq.}$ m^{-2} yr^{-1}. These findings clearly revealed that primarily the management and not the SOC content nor the land-use type itself is responsible for the magnitude of GHG emissions from intensive cropping on drained organic soils.

In addition to the estimation of total GHG emissions examined in **PUBLICATION I.**, the second study (**PUBLICATION II.**) focused in more detail on the short-term effects of biogas digestate and cattle slurry application on the previously mentioned intensive grassland sites, regarding gaseous emissions, mineral nitrogen use efficiency (NUE_{min}) and grass yields. It is well known that organic fertilization can enhance N_2O production particularly on organic soils. Nevertheless, despite the strong increase in the number of biogas plants in Germany, caused by the change in German energy policy, only a few studies have investigated the effect of biogas digestate application on the formation of gaseous nitrogen and in fact no studies exist which consider organic soils. The objective of **PUBLICATION II.** were to answer the questions i) how does biogas digestate and cattle slurry application affect N_2O and CH_4 fluxes as well as the NUE_{min} and grass yield, and ii) to what extent different soil organic matter (SOM) and nitrogen (N) contents influence on production of N_2O. Furthermore, ammonia (NH_3) volatilization was determined in the course of one fertilizer application event to obtain first clues with respect to the effects of soil and fertilizer types. The results of the second study (**PUBLICATION II.**) showed that, compared with cattle slurry, the application of biogas digestate significantly enhanced N_2O and NH_3 emissions and additionally increased the plant N-uptake and NUE_{min}, leading to significantly higher grass yields. This could partly be attributed to the higher ammonium (NH_4^+) content of the digestate and presumably also to a priming effect caused by increased SOM mineralization triggered by digestate application. Furthermore, the results clearly revealed that N_2O fluxes significantly increase with increasing SOM content, probably due to the more favourable soil conditions for denitrification. CH_4 emissions were found to be negligible at the grassland sites, irrespective of fertilization treatment due to aerobic conditions in the upper soil layer as result of drainage. In conclusion, both studies (**PUBLICATION I.** and **PUBLICATION II.**) revealed a very high release of GHG's, demonstrating the environmentally harmful agricultural use of drained organic soils and the urgent need for the rapid implementation of GHG-mitigation strategies and restoration measures.

Beside croplands and intensive grasslands, also black alder (*Alnus glutinosa* (L.) Gaertn.) forests on drained peat soils have been reported to be hot-spots for high N_2O losses, which has been attributed to alternating water tables of peatlands as well as to the incorporation of high amounts of easily decomposable organic nitrogen into the ecosystem by symbiotic dinitrogen (N_2)-fixation of alder trees. However, black alder forests represent the natural vegetation on moist and wet soils, often with high SOC

contents and afforestation in combination with water level management is considered to be one mitigation strategy for the reduction of GHG emissions. In this context, the third study (**PUBLICATION III.**) addressed the questions i) as to what extent drainage enhances the emissions of N_2O from black alder forests and ii) how N turnover processes and physical factors influence the production of N_2O and total denitrification rate. This study was conducted in a drained black alder forest with variable groundwater tables at a southern German fen peatland. Additionally, a laboratory incubation experiment was carried out to investigate GHG and N_2 fluxes under different soil temperature and moisture conditions. The study demonstrated that drainage considerably increased net N mineralization and N turnover processes and shifted the dominant process from net ammonification to net nitrification, resulting in nitrate leaching. Observed N_2O emissions in the field remained low at the drained sites, but the incubation experiment revealed the potential for high N_2O emissions under changing soil physical conditions at the drained alder sites. Temporally water-saturated conditions were found to enhance N_2 emissions, whereas N_2O emissions remained unchanged. In contrast, the permanently high groundwater level at the undrained site prevented net nitrification and resulted in negligible gaseous N losses. In conclusion, N mineralization and N turnover processes in black alder forests are very sensitive to alternating groundwater tables and have a high potential for high N_2O emissions if the soil is not permanently water saturated. The rough estimation of GWPs made in this study highlights the needs for further investigations, particularly regarding the net CO_2 exchange, to verify to what extent water-saturated black alder forests can serve as mitigation strategy for the reduction of GHG emissions.

Zusammenfassung

Entwässerung und Intensivierung der Landwirtschaft haben im letzten Jahrhundert dazu geführt, dass sich deutsche Moore zu „hot-spots" von Treibhausgas-Emissionen (THG) entwickelten. Vor allem Moore, die in Ackerland und intensiv genutztes Grünland umgewandelt wurden, zählen zu den Hauptverursachern von Kohlenstoffdioxid- (CO_2) und Distickstoffmonoxid-Emissionen (N_2O). Trotz enormer Forschungsanstrengungen im letzten Jahrzehnt, besteht nach wie vor hohe Unsicherheit in der Definition zuverlässiger Emissionsfaktoren (EF) für die Klimaberichterstattung, entsprechend den Vorgaben des Intergovernmental Panel on Climate Change (IPCC). Dies ist einerseits auf die Tatsache zurückzuführen, dass die geringe Anzahl, der auf diesen Böden erhobenen THG-Flüsse eine sehr hohe zeitliche und räumliche Variabilität aufweist und andererseits eine Abgrenzung von Histosolen aufgrund der verschiedenen nationalen und internationalen Klassifikationssysteme für organische Böden schwierig ist. Die meisten Untersuchungen konzentrierten sich bisher auf typische entwässerte Moore, die einen hohen Gehalt an organischem Kohlenstoff (C_{org}) aufweisen, wohingegen rezente Moore und mit Mooren vergesellschaftete organische Böden mit C_{org}-Gehalten von weniger als 12% C_{org} vernachlässigt wurden, obwohl nicht geklärt ist, ob diese Böden im Vergleich zu typischen entwässerten Moore geringere THG-Emissionen verursachen. Diese Fragestellung wurde in der ersten von drei Studien aufgegriffen (**Publikation I.**). In dieser Studie wurde untersucht, wie sich der Gehalt an organischem Bodenkohlenstoff (SOC) auf die Freisetzung von THGs aus entwässerten organischen Böden mit unterschiedlichen Landnutzungen und Bewirtschaftungsformen auswirkt. Hierfür wurden zwei, sich in ihrem SOC-Gehalt unterscheidende Böden ausgewählt, zum einen ein mollic Gleysol (9.4–10.9% C_{org} in 0–10 cm Bodentiefe) sowie ein sapric Histosol (16.1–17.2% C_{org} in 0–10 cm Bodentiefe). Auf beiden Böden wurden jeweils vier Untersuchungsvarianten, bestehend aus zwei Acker- (mit Mais oder Hafer) und zwei intensiven Grünlandnutzungen (Düngung mit Rindergülle oder Biogasgärrest) eingerichtet. Über einen Zeitraum von zwei Jahren wurden THG-Flüsse (CO_2, N_2O und Methan (CH_4)) gemessen und modelliert. Es konnten keine Unterschiede zwischen den zwei Bodentypen bezüglich der Netto-CO_2-Bilanzen oder des Globalen Erwärmungspotenitals (GWP) nachgewiesen werden. Demzufolge können auch entwässerte und intensiv als Acker oder Grünland genutzte Anmoore, ebenso wie

typische entwässerte Moore als „hot-spots" für THG-Emissionen angesehen werden. Die abgeschätzten GWPs, die für einen Zeitraum von 100 Jahren berechnet wurden, ergaben eine sehr hohe Freisetzung von THGs für alle Untersuchungsvarianten im Bereich von 1837 bis 7095 g $CO_{2\ eq.}$ m^{-2} a^{-1}. Die Ergebnisse deuten darauf hin, dass hauptsächlich die Bewirtschaftung und nicht der SOC-Gehalt oder der Landnutzungstyp an sich für die Höhe der THG-Emissionen aus landwirtschaftlich intensiv genutzten und entwässerten organischen Böden verantwortlich ist.

Neben der Abschätzung der THG-Bilanzen (**Publikation I.**), konzentriert sich die zweite Studie im Detail auf die kurzfristigen Auswirkungen der Ausbringung von Biogasgärresten und Rindergülle in Bezug auf gasförmige Emissionen, mineralische Stickstoffausnutzungs-Effizienz (NUE_{min}) und Graserträge, auf den zuvor erwähnten intensiv bewirtschafteten Grünlandflächen. Es ist hinlänglich bekannt, dass organische Düngung vor allem auf organischen Böden zu verstärkter Bildung von N_2O führen kann. Trotz des starken Anstiegs der Anzahl von Biogasanlagen im Zuge der energiepolitischen Veränderungen in Deutschland, existieren bisher nur wenige Studien über die Auswirkungen von Biogasgärrest-Applikationen auf die Bildung von gasförmigem Stickstoff, wobei organische Böden nicht berücksichtigt wurden.

Ziel der zweiten Publikation war die Beantwortung der Fragen i) in wieweit die Ausbringung von Biogasgärrest und Rindergülle die N_2O- und CH_4-Flüsse, sowie die NUE_{min} und den Grasertrag beeinflussen, und ii) welchen Einfluss unterschiedliche Bodenhumus- (SOM) und Bodenstickstoffgehalte (N) auf die Entstehung von N_2O haben. Zudem wurde im Zuge eines Düngeereignisses die Ammoniakverflüchtigung (NH_3) bestimmt, um die Auswirkung der unterschiedlichen Bodentypen und der Düngerarten abschätzen zu können. In der **zweiten Studie (Publikation II.)** wurde gezeigt, dass die Ausbringung von Biogasgärrest im Vergleich zur Düngung mit Rindergülle zu einer signifikanten Erhöhung der N_2O- und NH_3-Emissionen sowie der Stickstoffaufnahme und NUE_{min} der Pflanzen führte, was wiederum eine signifikante Steigerung der Graserträge zur Folge hatte. Dies könnte zum Teil auf den höheren Ammoniumgehalt (NH_4^+) im Biogasgärrest zurückzuführen sein sowie möglicherweise auf einen „Priming-Effekt", ausgelöst durch eine verstärkte SOM-Mineralisation als Folge der Gärrest-Düngung. Darüber hinaus zeigen die Ergebnisse deutlich, dass die N_2O-Flüsse mit zunehmendem SOM-Gehalt signifikant ansteigen, was wahrscheinlich auf günstigere Bedingungen für Denitrifikation im Boden zurückzuführen ist. Zudem hat sich gezeigt, dass die CH_4-Emissionen der Grünlandflächen, unabhängig von der

Düngevariante, aufgrund der aeroben Bodenverhältnisse infolge der Entwässerung zu vernachlässigen sind. Zusammenfassend lässt sich sagen, dass in beiden Studien (**Publikation I.** und **II.**) eine sehr hohe Freisetzung von THGs nachgewiesen wurde, was die geringe Nachhaltigkeit der Bewirtschaftung entwässerter organischer Böden veranschaulicht und die dringende Notwendigkeit für die Umsetzung von THG-Minderungsstrategien und Renaturierungsmaßnahmen verdeutlicht.

Neben Ackerland und intensivem Grünland werden auch Schwarzerlenwälder (*Alnus glutinosa* (L.) Gaertn.) auf entwässerten Moorböden, als „hot-spots" für hohe N_2O-Verluste angesehen, was auf die stark schwankenden Grundwasserflurabstände von Mooren sowie den Eintrag großer Mengen leicht verfügbaren organischen Stickstoffs durch die symbiotische Distickstoff(N_2)-Fixierung von Schwarzerlen zurückgeführt wird. Schwarzerlenwälder repräsentieren jedoch die natürliche Vegetation auf feuchten und nassen Böden, häufig mit hohem SOM-Gehalt und die Aufforstung in Kombination mit einem Grundwassermanagement gilt als eine Strategie zur Reduzierung der THG-Emissionen. In diesem Zusammenhang setzt sich die dritte Studie (**Publikation III.**) mit den Fragen auseinander, i) inwieweit die Entwässerung von Schwarzerlenwäldern zu einem Anstieg der N_2O-Emissionen führt und ii) wie Stickstoffumsetzungsprozesse und physikalische Einflussfaktoren die Bildung von N_2O und die Gesamt-Denitrifikationsraten beeinflussen. Diese Studie wurde in einem drainierten Schwarzerlenbestand in einem süddeutschen Niedermoor mit unterschiedlichen Grundwasserflurabständen durchgeführt. Zudem wurde im Rahmen eines Laborinkubationsexperiments der Einfluss von Bodentemperatur und -feuchte auf die THG- und N_2-Flüsse untersucht. Die Studie zeigte, dass die Netto-N-Mineralisierung und die N-Umsetzungsprozesse durch Entwässerung deutlich anstiegen und die vorherrschende Netto-Ammonifikation zugunsten der Netto-Nitrifikation verschoben wurde, was eine erhöhte Auswaschung von Nitrat zur Folge hatte. Obwohl im Feld auf den entwässerten Standorten nur niedrige N_2O-Austauschraten beobachtet wurden, konnte im Rahmen des Inkubationsexperiments nachgewiesen werden, dass veränderte bodenphysikalische Bedingungen im entwässerten Erlenwald zu hohen N_2O Emissionen führen können. Temporär wassergesättigte Bedingungen verursachten einen Anstieg der N_2-Emissionen, wohingegen kein Einfluss auf die N_2O-Emissionen beobachtet wurde. Im Gegensatz dazu verhinderte der konstant hohe Grundwasserstand des nicht drainierten Untersuchungsstandorts die Netto-Nitrifikation was in vernachlässigbaren gasförmigen N-Verlusten resultierte. Zusammenfassend

kann festgestellt werden, dass die N-Mineralisierung und die N-Umsetzungsprozesse in Schwarzerlenwälder sehr sensitiv auf sich ändernde Grundwasserflurabstände reagieren und bei nicht dauerhaft gewährleisteter Wassersättigung des Bodens ein hohes N_2O-Bildungspotenzial bergen. Die in dieser Studie vorgenommene grobe Abschätzung des GWPs macht deutlich, dass weitere Untersuchungen vor allem bezüglich des Nett-CO_2-Austauschs erforderlich sind, um zu überprüfen inwieweit Erlenwälder eine Option zur Verminderung von THG-Emissionen darstellen.

Table of Contents

List of Figures

List of Tables

List of Abbreviations and Acronyms

AFOLU	Agriculture, Forestry and Other Land Uses
AIC	Akaike Information Criterion
Ar	Argon
C	Carbon
$CaCl_2$	Calcium dichloride
CH_4	Methane
CO_2	Carbon dioxide
C_{org}	Organic carbon
C_{tot}	Total soil carbon
DM	Dry matter
DTM	Dräger-Tube Method
E_0	Activation energy
ECD	Electron capture detector
EF	Emission factor
FID	Flame ionization detector
GC	Gas chromatograph
GHG	Greenhouse gases
GPP	Gross primary production
GPP_{2000}	Gross primary production at PAR 2000
GW	Groundwater
GWP	Global Warming Potential
He	Helium
IPCC	Intergovernmental Panel on Climate Change
LULUCF	Land Use, Land-Use Change and Forestry
MIT	Microbial mineralization-immobilization turnover
N	Nitrogen
N_2	Dinitrogen
N_2O	Nitrous oxide
NECB	Net Ecosystem Carbon Balance
NEE	Net ecosystem exchange of CO_2
N_{dep}	Nitrogen deposition
Nr	Reactive nitrogen (Nr includes inorganic reduced forms of N (e.g., NH_3^-, NH_4^+), inorganic oxidized forms (e.g., NO_x, HNO_3, N_2O, NO_3^-), and organic compounds (e.g., urea, amines, proteins, nucleic acids)).
NH_3	Ammonia
NH_4^+	Ammonium
NH_2OH	Hydroxylamine
N_{org}/R-NH_2	Monomeric organically bound N forms
N_{min}	Mineral soil nitrogen
NNM	Net nitrogen mineralization
NO	Nitric oxide
NO_2^-	Nitrite
NO_2	Nitrogen dioxide
NO_3^-	Nitrate
NO_X	Nitric oxide + nitrogen dioxide
NPP	Net primary production
NSE	Nash-Sutcliffe efficiency

N_{tot}	Total soil nitrogen
NUE	Nitrogen use efficiency
NUE_{min}	Mineral nitrogen use efficiency
O_2	Molecular oxygen
O_3	Ozone
PAR	Photon flux density of the photosynthetic active radiation
PBIAS	Percent bias
Q_{10}	Temperature quotient
r	Pearson's correlation coefficient
R_a	Autotrophic respiration
R_{ECO}	Ecosystem respiration
RF	Radiative Forcing
R_h	Heterotrophic respiration
R_{ref}	Respiration at T_{ref}
RSR	Ratio of the root mean square error to the standard deviation of measured data
SD	Standard Deviation
SE	Standard Error
SOC	Soil organic carbon
SOM	Soil organic matter
$ST_{2/5/10}$	Soil temperature in 2, 5 or 10 cm soil depths
T_0	Temperature constant for the start of biological processes: 227.13 [K]
T_{air}	Air temperature in 20 cm above soil surface
TCD	Thermal conductivity detector
T_{ref}	Reference temperature: 283.15 [K]
UNFCC	United Nation Framework Convention on Climate Change
VDS	Volatile dry solid
WFPS	Water filled pore space
α	Light use efficiency

List of Publications and Contributions

Research papers as first author

I. Eickenscheidt, T., Heinichen, J., and Drösler, M.: The greenhouse gas balance of a drained fen peatland is mainly controlled by land-use rather than soil organic carbon content, Biogeosciences Discuss (submitted for publication in Biogeosciences).

II. Eickenscheidt, T., Freibauer, A.; Heinichen, J., Augustin, J., and Drösler, M.: Short-term effects of biogas digestate and cattle slurry application on greenhouse gas emissions affected by N availability from grasslands on drained fen peatlands and associated organic soils, Biogeosciences, 11, 6187–6207, doi:10.5194/bg-11-6187-2014, 2014

III. Eickenscheidt, T., Heinichen, J., Augustin, J., Freibauer, A., and Drösler, M.: Nitrogen mineralization and gaseous nitrogen losses from waterlogged and drained organic soils in a black alder (*Alnus glutinosa* (L.) Gaertn.) forest, Biogeosciences, 11, 2961–2976, doi:10.5194/bg-11-2961-2014, 2014.

The dissertation is based on the above named publications.

Research papers as co-author

IV. Metzger, C., Jansson, P.-E., Lohila, A., Aurela, M., Eickenscheidt, T., Belelli-Marchesini, L., Dinsmore, K. J., Drewer, J., van Huissteden, J., and Drösler, M.: CO_2 fluxes and ecosystem dynamics at five European treeless peatlands – merging data and process oriented modeling, Biogeosciences., 12, 125–146, doi:10.5194/bg-12-125-2015, 2015.

V. Metzger, C., Heinichen, J., Eickenscheidt, T., and Drösler, M.: Impact of land use intensity on the relationships between vegetation indices (NDVI, LAI), photosynthesis and biomass on temperate grassland fens, Grass and Forage Science (submitted).

The contributions to the publications were as follows

General contribution to publication I.–III.

General contribution was the construction and further development of measurement equipment, installation and maintenance of measurement sites and climate stations, implementation and management of all GHG (CO_2, CH_4, N_2O) measurements, and the soil sampling and sample preparation for the determination of soil properties (carbon and nitrogen content, bulk density) and mineral nitrogen ($N_{min} = NH_4^+$-N + NO_3^--N). All steps were carried out in close agreement with M. Drösler.

Specific contribution to publication I.–III.

I. Specific contribution comprised the topic development in close agreement with M. Drösler, data preparation and calculations, modeling of CO_2, statistical analyses, and the preparation of the manuscript.

II. Here, the contribution mainly included the topic development in agreement with M. Drösler and the advice of J. Augustin and A. Freibauer, additional measurements of NH_3, data preparation and calculations, statistical analyses, and the main preparation of the manuscript.

III. Contribution involved topic development in agreement with M. Drösler and J. Augustin, soil sampling and sample preparation for the incubation procedure and for the determination of net nitrogen mineralization, assistance in N_2, N_2O, CH_4 and CO_2 measurements in the laboratory of the Institute of Landscape Matter Dynamics, Leibniz Centre for Agricultural Landscape Research (ZALF) as well as data preparation and calculations, statistical analyses, and the main preparation of the manuscript.

Contribution as co-author

IV. The contributions covered the supply of data sets from the Freisinger Moos (sites FsA and FsB) for the time period 03/2009–01/2012, the installation and maintenance of measurement sites and climate stations, the implementation and management of all CO_2 measurements, soil sampling and sample preparation, data preparation and calculations as well as the modeling of CO_2.

V. Here, contributions comprised supply of the GPP data sets from the sites E3a, E3b, I1 and I2, the installation and maintenance of measurement sites and climate stations, the implementation and management of all CO_2 measurements, data preparation and calculations as well as the modeling of GPP.

1 GENERAL INTRODUCTION

1.1 BACKGROUND

In the last century, drainage and intensification of agriculture turned European peatlands to hot-spots for greenhouse gas (GHG) emissions (Drösler et al., 2008a). According to estimates, peatlands in Germany account for approximately 98% of soil derived net CO_2 emissions, which corresponds to approximately 5.1% of the national GHG emissions, although peatlands cover only 5.1% of the land surface area (Drösler et al., 2011). Thus drained peatlands represent the largest source of GHG emissions in the Agriculture, Forestry and Other Land Uses (AFOLU) sector and in addition the largest single source for GHG emissions outside the energy sector in Germany (Drösler et al., 2011). Hence, according to the Intergovernmental Panel on Climate Change (IPCC) guidelines, drained peatlands are identified as key category which leads to the fact that Germany is obligated to calculate the annual GHG emission inventory on the basis of national specific emission factors (EF). However, the national climate reporting in the Land Use, Land-Use Change and Forestry (LULUCF) sector as well as in the AFOLU sector is a challenging task for organic soils. This is partly because reliable measurements of GHGs from temperate drained peatlands are rare and also because observed GHG fluxes show a large temporal and spatial variability. Furthermore, the definition of histosols is complex and several national and international classification systems exist for organic soils. According to the United Nations Framework Conventions on Climate Change (UNFCCC) and the Kyoto protocol, the reporting of GHG emissions from a key category demands an increased requirement on the data base (Tier 2 or Tier 3 methods). Since such an extensive dataset was not available for the German climate reporting, a large research project was started in the year 2008 by the Thünen Institute, Federal Research Institute for Rural Areas, Forestry and Fisheries to fill the gaps of knowledge. The present PhD thesis was part of this joint research project, called "Organic soils"; Acquisition and development of methods, activity data and emission factors for the climate reporting under LULUCF/AFOLU.

1.2 PEATLANDS – DEFINITION, GENESIS AND THEIR GLOBAL DISTRIBUTION

Peatlands comprise very special kinds of transitional ecosystems with habitats between uplands and water, covered with a tree and/or moss layer (Rydin and Jeglum, 2006; Limpens et al., 2008). Peatland ecosystems are characterized by a waterlogged, anoxic soil layer (catotelm), in which organic matter from adapted plants accumulates and finally leads to the formation of peat, owing to incomplete decomposition (Limpens et al., 2008). However, depending on hydrological and geomorphological building conditions the upper peat layer can temporarily be water unsaturated (acrotelm), leading to a shift in the depth of the anoxic-oxic boundary layer during the year.

Peatland ecosystems can be found all over the world except for arid and permanently ice-covered areas (Rydin and Jeglum, 2006). They develop mostly in regions or altitudes where precipitation exceeds evapotranspiration (Göttlich, 1990), but groundwater-driven peatlands can develop as well in generally drier surroundings, as long as the water saturation of the soil is sufficient for peat formation. Globally, peatland cover about 4 million km^2 which corresponds to approximately 3% of the earth's land surface (Rydin and Jeglum, 2006). Approximately 75–85% of worldwide peatlands are situated in the boreal and subarctic regions of the northern hemisphere, particularly in Russia, Canada and the USA (Fig. 1). The remaining peatlands are found in the subtropical and tropical regions (10–15%), particularly in Indonesia, Malaysia and Papa New Guinea and approximately 10% in the temperate regions (Joosten, 2004; Limpens et al., 2008; Yu et al., 2010; Frolking et al., 2011).

Based on their water origin and trophic status, peatlands are mostly classified in fens, bogs and transition peatlands (Rydin and Jeglum, 2006). Fens are minerotrophic peatlands, which can further be divided in topogenous (basin) or soligenous (sloping) fens (Rydin and Jeglum, 2006). The term minerotrophic indicates that the water is originated from groundwater or from surface inflow from surrounding mineral soils, leading to a higher nutrient availability as compared to bogs and approximately neutral pH. Bogs are ombrotrophic peatlands, isolated from groundwater or other mineral-rich soil water, receiving their water and nutrients solely from precipitation and airborne dust (Rydin and Jeglum, 2006). Thus, bogs are typically nutrient poor and strongly acid, usually having pH values lower than four. The shape of bogs varies from convex (raised bogs), to flat with potential sloping. Bogs and fens are often not sharply separated and

several transition forms of peatlands can occur. The different nutrient supply and acid status of fens and bogs significantly affect the peat-forming vegetation, leading to characteristic plant communities. Fens are mostly dominated by sedges (e.g. *Carex elata*, *C. rostrata*, *C. lasiocarpa*, *C. elongata*) but depending on development stage, water table height and nutrient supply also brown mosses (e.g. *Drepanocladus revolvens*), grasses (e.g. *Molina caerulea*), trees (*Alnus*, *Salix*) and several other groups with a grass-like morphology such as cotton grasses (e.g. *Eriophorum angustifolium*), rushes (e.g. *Schoenus ferrugineus*) and reed (e.g. *Phragmites australis*, *Thypha latifolia*) can be found (Göttlich, 1990; Rydin and Jeglum, 2006). Bogs however, are mainly dominated by peat mosses of the genus *Sphagnum*. In dependence of the water table height, important species are *S. magellanicum* and *S. rubellum* (in dryer areas), *S. cuspidatum* and *S. dusenii* (typically in flooded hollows) and several others like *S. fuscum*, *S. palustre*, *S. majus*, *S. subsecundum*. (Göttlich, 1990; Rydin and Jeglum, 2006). Additionally, some few other species like *Eriophorum vaginatum*, *Trichophorum caespitosum*, *Oxycoccus palustris*, *Andromeda polifoila*, *Drosera rotundifolia*, *Erica tetralix* are also characteristically for bogs (Göttlich, 1990).
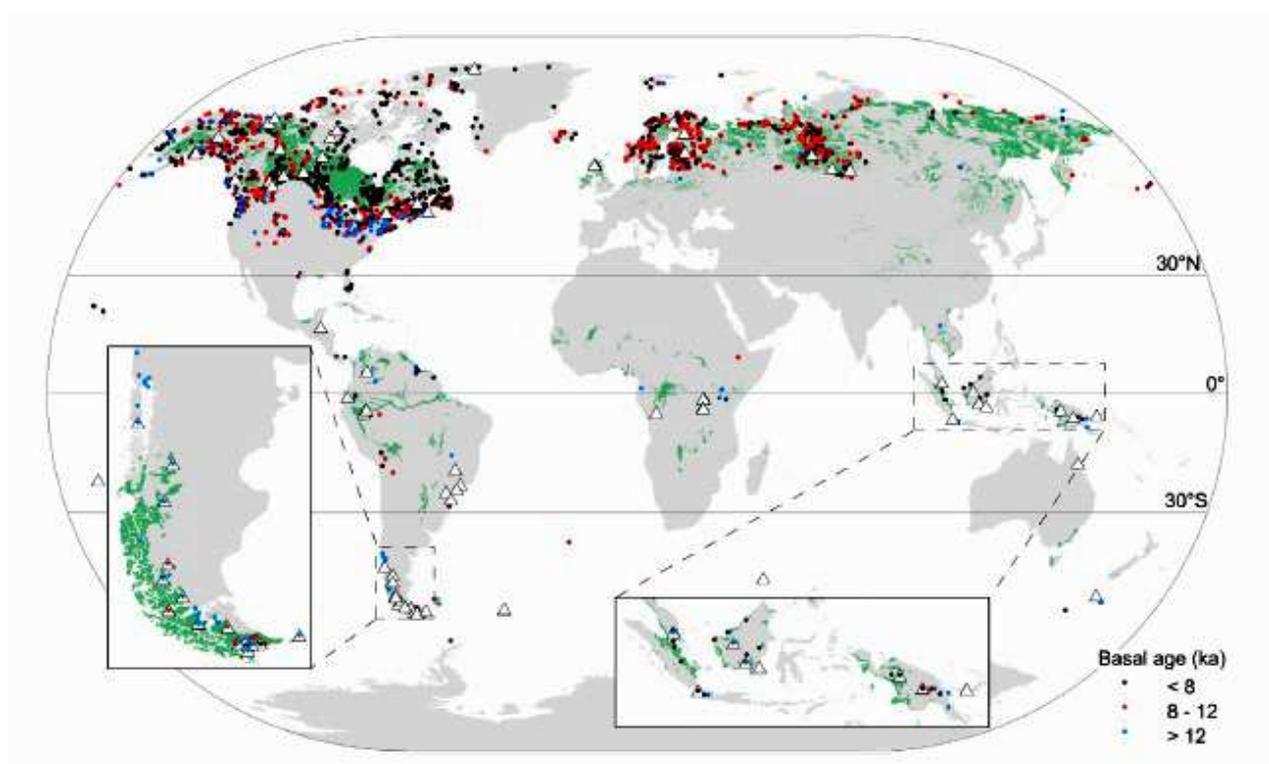
Figure 1 Global peatland distribution (green areas). Dots showing peatland study sites with basal peat age (colors showing the ages of peatland initiation: black <8 ka (ka =1000 yr), red 8□12 ka, and blue >12 ka) (Source: Yu et al., 2010).

Peatlands or peats also appear in several national soil classification systems e.g. as Moore (Germany), Organosols (Australia), Organossolos (Brazil), Organic order (Canada), and Histosols (United States of America), of which the US classification system has formed the basis for the effective international standard (Couwenberg, 2011). However, the definition of organic soils/histosols is complex and not entirely consistent (Couwenberg, 2011). Most definitions refers to the thicknesses of soil layers, their organic matter content, their water origin, underlying material, clay content as well as the mean duration of the annual period of water saturation (Couwenberg, 2011). For the climate reporting under LULUCF/AFOLU, the 2006 IPCC Guidelines (Annex 3A.5, Chapter 3 in Volume 4) and the Wetlands Supplement (2014a) largely follow the definition of organic soils (Histosols) by the Food and Agriculture Organization (FAO). Therefore, a soil is defined as an organic soil when fulfilling criteria 1 and 2, or 1 and 3 of those listed below:

> "*1. Thickness of organic horizon greater than or equal to 10 cm. A horizon of less than 20 cm must have 12 percent or more organic carbon when mixed to a depth of 20 cm.*
>
> *2. Soils that are never saturated with water for more than a few days must contain more than 20 percent organic carbon by weight (i.e., about 35 percent organic matter).*
>
> *3. Soils are subject to water saturation episodes and have either*
>
> > *a. at least 12 percent organic carbon by weight (i.e., about 20 percent organic matter) if the soil has no clay; or*
> >
> > *b. at least 18 percent organic carbon by weight (i.e., about 30 percent organic matter) if the soil has 60% or more clay; or*
> >
> > *c. an intermediate proportional amount of organic carbon for intermediate amounts of clay.*"

In contrast to the FAO definition of Histosols, the IPCC definition (2006) does not include the criterion of >40 cm thickness of the organic horizon to allow for deviating, often historically derived, country-specific definitions of organic soils (IPCC, 2014a). Thus the IPCC definition of organic soils is broader than the definition of histosols in the FAO (1998) or in the World Reference Base for soil resources (WRB, 2006). However, the interpretation of the IPCC (2006) deviates also from most European definitions of peat (Couwenberg, 2011), providing a challenge for the climate reporting under the UNFCC.

For example the German classification system (KA5; Ad-hoc-AG Boden, 2005) usually requires a thickness of more than 30 cm. Furthermore, a distinction is made between soil horizons with ≥30% SOM content (called organic horizon) and those containing 15–30% SOM (called anmoor horizon). Particularly at the boundary between mineral and organic soils, the conversion from SOM to C_{org} contents leads to uncertainties due to the difference in conversion factors for mineral soils (1.72) and peat soils (2.00) according to the KA5 (Tiemeyer et al., 2013). Depending on the used conversion factor the upper limit of IPCC requirement is between 21 and 24% SOM by weight (Tiemeyer et al., 2013). Thus the IPCC guidelines consider only a fraction of the entire area, defined as anmoor, whereas other parts are not taken into account for the climate reporting of drained organic soils.

1.3 CARBON CYCLE IN PEATLANDS AND ITS HUMAN ALTERATION

Incomplete decomposition due to anoxic and sometimes acid conditions, coupled with low litter quality, has led to significant C accumulation in peatlands, particularly since the last ice age about 10,000 years ago (Rydin and Jeglum, 2006). Estimated carbon stocks for peatlands differ widely, mainly due to uncertainties for peat depth, bulk density and mean C content (Turunen et al., 2002). However, it is assumed that peatlands globally store about 400–612 Gt C, of which 270–547 Gt have accumulated in the northern hemisphere, 42–52 Gt have been stored in tropical latitudes and approximately 15 Gt have accumulated in the southern hemisphere, particularly in Patagonia (Lappalainen, 1996; Turunen et al., 2002; Hooijer et al., 2006 & 2010; Tarnocai et al., 2009; Page et al., 2011; Yu et al., 2010 & 2011). Thus, the global peatland C pool size equals the amount of organic C contained in the global living biomass (~560 Gt) and corresponds to 50–80% of the CO_2-C held in the atmosphere (Lal, 2004). Moreover, peatlands store approximately 26–39% or 16–25% of the global soil C pool estimated to a depth of one or two meters, respectively (Gorham, 1991; Batjes, 1992; Eswaran et al., 1995). Generally, the SOC provides the energy essential to biological processes and its associated nutrients (e.g. N, P, S) can contribute to the resilience of plant/soil systems to disturbances (Baldock, 2007). Moreover, the quality and quantity of SOC influence several physical and chemical properties, including peat structure, water retention, thermal properties and the cation exchange and buffer capacities of peatlands (Scheffer and Schachtschabel, 2002). The substantial accumulation of organic C and the long-term storage function of peatlands have markedly altered the global C cycle and also

significantly contributed to global cooling during the Holocene (see chapter 1.5) (Frolking and Roulet, 2007; Limpens et al., 2008). However, in the last century human-induced activities (e.g. drainage, land-use change) and associated consequences (e.g. increased nitrogen deposition) have massively altered peatland C cycle on a global scale. Many ecosystem components and processes are involved in peatlands C cycle. The most important flows of the C cycles in a natural and a drained and cultivated fen peatland are depicted in Figure 2.
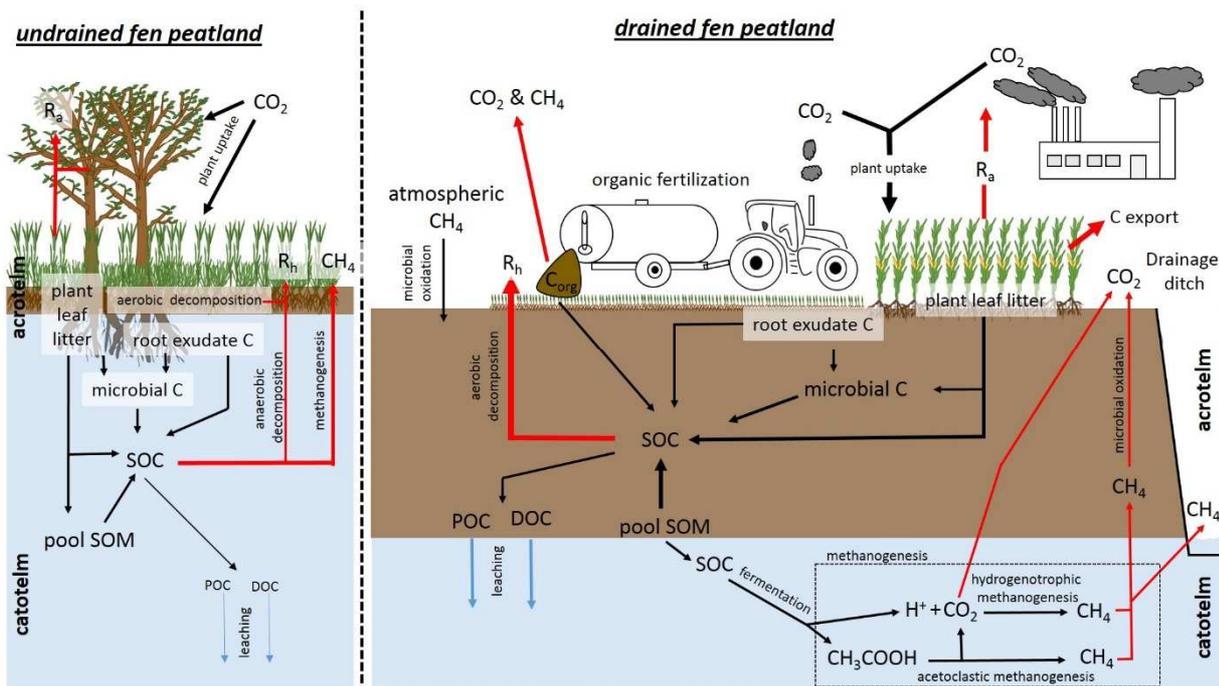
Figure 2 Simplified carbon cycle of a pristine and drained fen peatland.

1.3.1 Net primary production and C input into peatlands

Carbon input in pristine peatlands is mainly controlled by the net primary productivity (NPP), originating from photosynthesis of autotrophic plants (Nilsson and Öquist, 2009). During photosynthesis, radiation energy is absorbed and transformed into chemically bounded energy, as basically shown in Equation 1 (Larcher, 2001).

$$6CO_2 + 6H_2O + \text{radiation energy} \rightarrow C_6H_{12}O_6 + 6O_2 \tag{1}$$
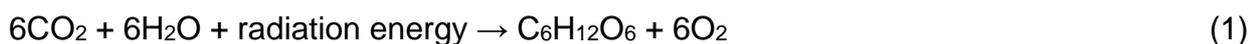
The total amount of CO_2 fixed in the process of photosynthesis is called gross primary production (GPP). GPP is principally a function of incident photosynthetically active radiation (PAR), which includes the wavelength range of 380–710 nm (Frolking et al., 1998). While GPP results in the formation of energy containing compounds (e.g. sugars, starch), several metabolic processes in the plant consume them, resulting in the continuous release of CO_2, which is referred as autotrophic respiration (R_a; reverse

function of Eq.1). The amount of C converted to CO_2 by R_a is estimated to be half of the C fixed by GPP (Larcher, 2001). The sum of GPP (positive flow) and R_a (negative flow) refers to as net primary production (NPP). Several studies indicate that pristine peatlands had both significantly lower productivity at high PAR levels (gross primary production at PAR infinite (GP_{max}) range from 5.2 to 12.4 μmol m^{-2} s^{-1}) and a lower apparent quantum yield at low light (α values range from 0.017 to 0.027 μmol CO_2 μmol^{-1} PAR) compared to croplands and forest ecosystems on mineral soils (GP_{max} up to 82.9 μmol m^{-2} s^{-1} and α up to 0.056 μmol CO_2 μmol^{-1} PAR; Frolking et al., 1998). According to Blodau (2002), the NPP of peatlands range between 5 g DM m^{-2} yr^{-1} in northern regions and more than 1000 g DM m^{-2} yr^{-1} in the tropics with most common values between 100 and 400 g DM m^{-2} yr^{-1}. This is in accordance with Gorham (1995) and Schulze et al. (2010), who reported NPP values of about 230 g C m^{-2} yr^{-1} and 295 g C m^{-2} yr^{-1}, respectively, for naturally peatlands. Although NPP is relatively low in pristine peatlands, at least in Nordic regions (Frolking et al., 1998), the imbalance between NPP and decomposition is strong enough to cause a significant SOM accumulation (Bragazza et al., 2009). However, when peatlands are drained and intensively cultivated, NPP significantly increases as a result of cultivated field-crops, especially those developed to produce high biomass yields in a short time. According to Schulze et al. (2010), NPP of European grassland and cropland averaged 750 g C m^{-2} yr^{-1} and 550 g C m^{-2} yr^{-1}, respectively. However, depending on crop, dry matter yields of up to 1650 g DM m^{-2} yr^{-1} for winter wheat and up to 2500 g DM m^{-2} yr^{-1} for silo maize were reported for Germany (Aigner et al., 2012).

In contrast to natural peatlands, cultivated peatlands often additionally receive C through the application of organic fertilizers. However, the effects of this input on the C balance of the ecosystem is difficult to estimate, since organic fertilizers vary widely in their physical and chemical composition. Also, annual application rates differ in their quantities and frequencies between different land-use types, land-use intensities and regions. In order to obtain a balanced humus balance, C application rates of about 40–60 g C m^{-2} yr^{-1} are required for mineral soils according to the European cross-compliance regulations (Gutser et al., 2010), thus reflecting approximately the average C input on agricultural land in Europe.

1.3.2 Carbon decomposition, accumulation and losses in peatlands

SOM decomposition is controlled by several abiotic and biotic factors. Generally, decomposition refers to the breakdown of organic matter, ending with the conversion

into inorganic substances (mineralization), and is primarily driven by bacteria, invertebrate and fungi (Rydin and Jeglum, 2006). The total CO_2 release from soil during decomposition is called heterotrophic respiration (R_h). The sum of R_a and R_h is refered to as ecosystem respiration (R_{ECO}). According to Cicerone and Oremland (1988), R_{ECO} is responsible for returning a substantial part of up to 90% of C entering the plant/soil system by GPP to the atmosphere.

In peatlands, the high water level and the resulting anoxic soil conditions, at least in the catotelm, is considered as the key factor for the low decomposition rates of organic matter. Litter-bag studies under aerobic and anaerobic conditions revealed that the percentage of mass losses from vascular plant litter and *Sphagnum* litter was on average 2.3 times higher under aerobic conditions (Bragazza et al., 2009). Also in laboratory studies it was shown that the ratio of aerobic/anaerobic CO_2 production averaged 2.5 to 2.8 in peatlands (Moore and Dalva, 1993; Updegraff et al., 1995). Generally, the slowness of anaerobic decay is caused by the fact that anaerobic transformation processes provide less energy than aerobic processes (Succow and Josten, 2012). Beside anoxic soil conditions, decomposition rate is mainly constrained by plant litter recalcitrance, low temperatures and acid soil conditions (Bragazza et al., 2009). At an ecosystem scale, water level and pH mostly control the vegetation composition. In turn the vegetation composition determines the C balance through its effects on NPP and litter quality (Limpens et al., 2008). For example, the residues from many peatland adapted plants (e.g. sedges, *Sphagnum* species, alder, etc.) are often of low quality with high contents of lignin or lignin-like polymeric phenolic substances (Bragazza et al., 2009). Apart from the macromolecular organic composition of the peat forming vegetation, the SOM quality of peat strongly depends on hydrological and geomorphological building conditions during peat formation (Heller and Zeitz, 2012).

In most peatlands, the degradation of organic matter is totally dominated by fungi and bacteria (Nilsson and Öquist, 2009), whereas fungi numerically dominate in the acrotelm (Artz, 2009). Since microbes are unable to take up particulate organic matter (POM), the first step of mineralization involves the breakdown of solid, insoluble biopolymers into soluble components (hydrolyzation; Nilsson and Öquist, 2009). In the oxic zone, the dominant decomposition pathway is via aerobic, exo-enzym-driven decomposition (Artz, 2009). Oxygen has two main functions, it serves as terminal e^- acceptor during respiration and it acts as a direct oxidant of organic molecules (Nilsson and Öquist, 2009). However, when biomass residues (e.g. litter, roots, root exudates, faunal fecal

input) are deposited on or in the aerobic peat layer, a rapid mineralization starts, accompanied by the release of CO_2. First of all, simple organic compounds (e.g. sugar, starches, amino acids) and monomeric molecules are quickly metabolized and serve as energy source, followed by the turnover of carbohydrate polymers, amorphous cellulose and hemicellulose. Substances like lignin, cutins and tannins are quite resistant and thus the decomposition occurs much more slowly leading to a selective accumulation.

With increasing anoxic conditions the proportion of aerobic respiration decreases whereas fermentative processes and methanogenesis gain in importance (Artz, 2009). Under states of anoxia, only easily degradable substrates undergoe a slow, incomplete decomposition with lactic acid, ethanol and acetate as intermediate products, accompanied by the release of CO_2 (called anaerobic respiration). In contrast, the breakdown of complex substances like lignin is almost impossible and only low molecular weight lignin components and lignin precursors are attacked by the organisms or their exo-enzymes. As a result, lignin and other aromatic compounds accumulate in the catotelm, are humified and gradually form peat (Artz, 2009; Scheffer and Schachtschabel, 2002). In natural peatlands or in the catotelm of drained peatlands, CH_4 is produced either through the reduction of acetate (acetoclastic methanogenesis), with the simultaneous release of CO_2, or through the reduction of CO_2 with H_2 (hydrogenotrophic methanogenesis; Artz, 2009). Acetoclastic methanogenesis is considered as the terminal step of anaerobic decomposition in peatlands and it is assumed that around two thirds of global CH_4 production originate from this process (Artz, 2009). Once CH_4 is produced, it can be released to the atmosphere via three main pathways, diffusion in the soil towards the surface, ebullition (release of bubbles) and plant-mediated transport (Wahlen, 2005). Once CH_4 passes through an oxic layer at the soil surface, aerobic methanotrophs, particularly alpha- and gamma-proteobacteria take it up and oxidize it to CO_2. Thus the amount of CH_4 that reaches the atmosphere depends mainly on the thickness of the acrotelm. Nevertheless, aerenchymatic vascular transport of CH_4 from an anaerobic layer to the atmosphere, bypassing the oxic zone at the soil surface, can account for up to 90% of total CH_4 emissions (Grosse et al., 1992; Svensson and Sundh, 1992; Wahlen, 2005). In turn, also O_2 can be transported within the plant into an anoxic zone, leading to the oxidation of CH_4 in close vicinity of the root (Whalen, 2005). Recently, there has been growing evidence that beside aerobic CH_4 oxidation also anaerobic CH_4 oxidation (AMO) can play a major role in the C-cycle of natural peatlands (e.g. Gupta et al., 2013). Up to date the mechanisms of AMO are not

well understood (Valentine and Reeburgh, 2000), but evidence suggests that AMO is coupled to NO_3^-, SO_4^{3-} or Fe^{3+} reduction as well as to "reverse methanogenesis" carried out by methanogens (Segers, 1998; Gupta et al., 2013). In the last decade, several investigations have been carried out to determine CH_4 fluxes from peatlands all over the world. Observed CH_4 fluxes range from –1.3 to 644 mg CH_4 m^{-2} d^{-1} for northern peatlands and from 21 to 912 mg CH_4 m^{-2} d^{-1} for temperate and subtropical peatlands (Whalen, 2005), demonstrating the high temporal and spatial variability. However, annual CH_4 emission for drier bogs commonly averaged <1 g C m^{-2} yr^{-1} (Christinsen et al., 2004; Roulet et al., 2007), while wetter bogs averaged 5–8 g C m^{-2} yr^{-1} (Laine et al., 2007), whereas wetter fens covered by sedges can have annual CH_4 emissions exceeding 15 g C m^{-2} yr^{-1} (Limpens et al., 2008).

With increasing peat depth the organic matter gets more and more recalcitrant and thus the anaerobic decomposition rates decrease (Rydin and Jeglum, 2006). Although anaerobic decay rates reach only a hundredth or even a thousandth of aerobic decay rates in the acrotelm, the C lost due to anaerobic decay integrated over the whole peat profile can be substantial and could finally balance the amount of NPP (Rydin and Jeglum, 2006). When this happen, the peatland no longer exhibits net accumulation of organic substrates as it has reached an equilibrium of litter input and decomposition, setting a limit to peat growth (Rydin and Jeglum, 2006). However, in natural peatlands, observed net ecosystem exchange (NEE = GPP – R_{ECO}) rates of CO_2 ranged from small losses of 3 g C m^{-2} yr^{-1} to large uptakes of –90 g C m^{-2} yr^{-1}, with multiple-year averages between –20 and –60 g C m^{-2} yr^{-1} (Alm et al., 1997; Bryne et al., 2004; Limpens et al., 2008).

When peatlands are drained, aerobic decomposition processes speed up as a result of higher O_2 availability in the upper soil profile. Simultaneously, anaerobic decomposition distinctly decreases and peatlands turn from a significant CH_4 source to a small sink of atmospheric CH_4 (Crill et al., 1994; Flessa et al., 1998). Water table manipulation experiments show that heterotrophic CO_2 emission rates are 4 to 6 times larger when the water table is lowered to a depth of –40 to –50 cm below the soil surface compared to water table heights close to the peat surface (Moore and Knowles, 1989; Moore and Dalva, 1993). However, the release of C varies strongly, especially as a result of differences in the type and intensity of land-use (Fell et al., 2014). In particular management activities such as tillage and fertilization are known to enhance the microbial SOM decomposition directly and by increasing the amount of easily

decomposable organic substances they possibly cause a priming effect, i.e. the further acceleration of SOM decomposition (Kandel et al., 2013). Thus, highest CO_2 emissions from drained organic soils in boreal and temperate zones were observed from croplands and grasslands. According to the IPCC (2014a), observed NEE rates ranged from 650 to 940 g C m^{-2} yr^{-1} for boreal and temperate croplands and from 500 to 730 g C m^{-2} yr^{-1} for temperate nutrient-rich grasslands.

Beside CO_2–C losses of particulate organic carbon (POC) and dissolved organic carbon (DOC) significantly increase after drainage and cultivation of organic soils (Nilsson and Öquist, 2009). Generally, DOC is formed via hydrolyzation of POM (Nilsson and Öquist, 2009) and commonly is the largest component of total waterborne C losses from peatlands (IPCC, 2014a). Even in natural peatlands, measured DOC fluxes range from 4 to 63 g C m^{-2} yr^{-1} in boreal and tropical regions, which clearly shows that this flux is of the same magnitude as the rate of long-term C accumulation and thus can control whether these ecosystems are carbon sinks or sources (IPCC, 2014a). Moreover, after drainage DOC fluxes of up to 118% compared to natural sites were reported (IPCC, 2014a). Nevertheless, when peatlands are drained and cultivated the crop export via harvesting is quantitatively the second most important pathway of C losses after CO_2 emissions and corresponds to nearly the amount of NPP.

1.4 NITROGEN CYCLE IN PEATLANDS AND ITS HUMAN ALTERATION

Since the last deglaciation, huge amounts of organic matter have accumulated in peatlands, leading to considerable organic N stocks (McNeill and Unkovich, 2007). The accumulation of N in peatlands is caused by several abiotic and biotic factors, most of all the low rate of soil organic matter mineralization and the low gaseous and hydrologic N losses (Limpens et al., 2006). According to estimates, 9–16% of the world's soil nitrogen is stored in peatlands (Limpens et al., 2006), particularly in the northern latitudes. The N availability controls many aspects of ecosystem productivity (Vile et al., 2014), since N is an essential macronutrient for all organisms, being a key component of plant amino acids, nucleic acids and chlorophyll and it is also usually quantitatively the most required element for plants from the soil (McNeill and Unkovich, 2007). As in plant dry matter, N is also one of the most abundant non-C elements in peat (Rydin and Jeglum, 2006).

In the last century, human activities have massively accelerated the N cycle on a global scale (Vitousek et al., 1997; Galloway et al., 2008). One key factor was the

invention of the Haber-Bosch process (industrial reduction of N_2 to $\rightarrow NH_3$), which made possible the industrial production of mineral N fertilizers on a large scale (Gruber and Galloway, 2008). Simultaneously, fossil fuel combustion increased the emission of N oxides into the atmosphere and agricultural productivity enhanced the biological N fixation by cropping increasing areas with legume crops (Vitousek et al., 1997; Jenkinson, 2001). According to Galloway (1995), every year approximately 160 million metric tonnes N are incorporated in terrestrial ecosystems, of which 61% are derived from industrial N_2 fixation, mainly as fertilizers, and 14% from combustion of fossils fuels. As a consequence, peatland ecosystems have been exposed to increased N deposition (N_{dep}) for several decades, particularly in Central Europe. Generally, natural peatlands, particularly bogs, are considered as N-limited ecosystems, in which the rates of N input and output are very small compared with the total storage of N (Ambus and Zechmeister-Boltenstern, 2007). However, drainage and the increased use of N-fertilizers to maintain soil/peat productivity as well as the widespread increase in anthropogenic N_{dep} have drastically changed the N-cycle of peatlands in the last century. Figure 2 presents a simplified scheme of the N cycling in a pristine and human-disturbed fen peatland, showing inputs, internal fluxes, storages and outputs.
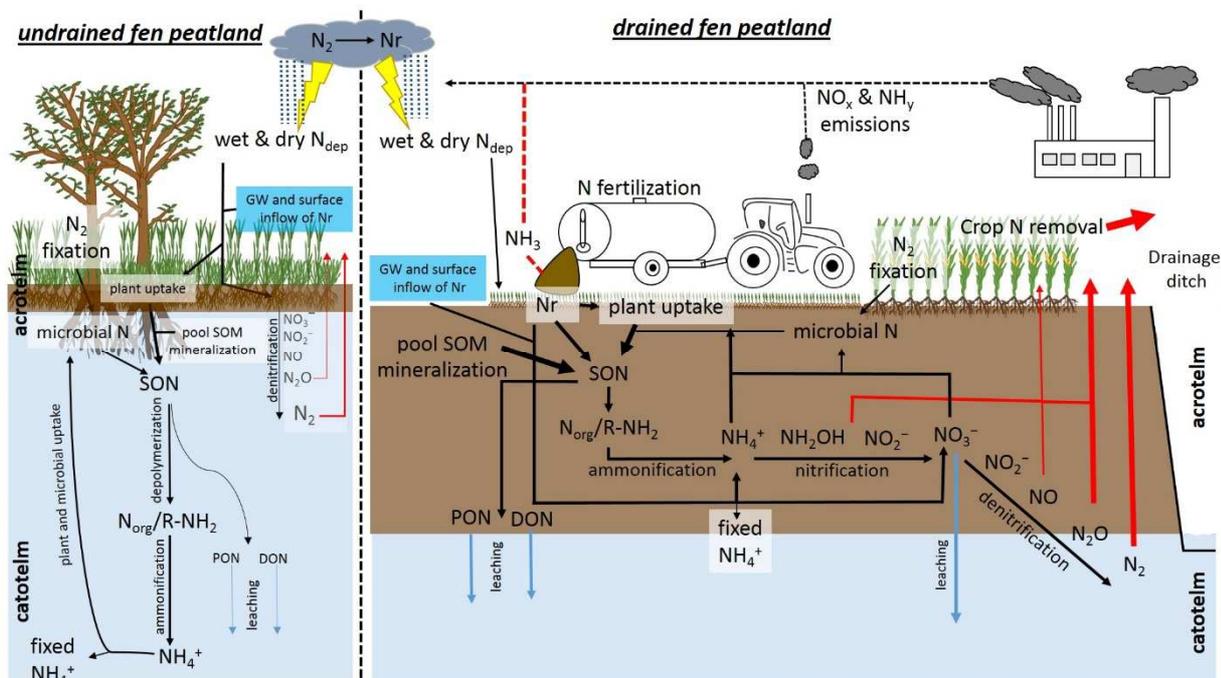
Figure 3 Simplified nitrogen cycle of a pristine and drained fen peatland.

1.4.1 Input pathways of N in peatland ecosystems

Natural bogs solely receive their N by the biological fixation of atmospheric N_2 and through wet and dry deposition of reactive N (Nr), whereas pristine fen peatlands

additionally receive Nr through groundwater and surface inflow. In the present manuscript, Nr includes all forms of N that are principally available for biological assimilation or transformation except of the relative inert dinitrogen (N_2), which makes up approximately 78% of the atmosphere (Galloway, 1998). Only a few genera of specialized microorganisms (diazotrophs) are able to incorporate atmospheric N_2 into organic molecules through the reduction of N_2 to $2NH_3$, using the enzyme complex *nitrogenase* (McNeill and Unkovich, 2007). The reduction of N_2 is a very energy demanding process (consuming 16 ATP), which requires between 8 and >100 g C per gram N (Vitousek et al., 2002). In peatlands, N-fixing microorganisms exist as free-living forms or in complex symbioses with plants, e.g. autotrophic Cyanobacteria as well as proteobacterial methanotrophs in association with *Sphagnum* (Vile et al., 2014) or Actinomycete (*Frankia*) within root-nodules of *Alder* trees (Rydin and Jeglum, 2006; McNeill and Unkovich, 2007). Beside the biological N fixation, the conversion of atmospheric N_2 to Nr additionally occurs naturally through lightning ($N_2 \rightarrow NO$), leading to wet and dry deposition of Nr. Natural N deposition rates are in the range of < 6 kg N ha^{-1} yr^{-1} (Pitcairn and Fowler, 1995; Koopmans et al., 1998) depending on latitude. In contrast, the annual N supply through biological N_2-fixation ranges between 0.3–62 kg N ha^{-1} for bogs and 1.6–115 kg N ha^{-1} for fen peatlands (Dittert, 1992; Succow and Joosten, 2012; Vile et al., 2014). Thus, it is assumed that the biological N_2-fixation represented the dominant input of new N into most pristine ecosystems in pre-industrial times (Vitousek et al., 2002; Vile et al., 2014). However, in the last decades anthropogenic N_{dep} has drastically increased by a factor of >10 (Galloway et al., 2004). In Germany, prevailing total N_{dep} rates in open land are estimated to be on average 26 kg N ha^{-1} yr^{-1} (Wochele and Kiese, 2010). In contrast, critical N loads have been estimated to constitute 5–10 kg N ha^{-1} yr^{-1} (Bobbink and Roelofs, 1995; UNECE, 2004; Bobbink et al., 2010). It is well known that particularly NH_3 deposition causes soil acidification and eutrophication of ecosystems (e.g., Fangmeier et al., 1994; Galloway, 1995, 2001; Smith et al., 1999). Mainly N limited ecosystems like ombrotrophic bogs are highly sensitive to enhanced N inputs, since their characteristic plants and microbes are adapted to conditions of restricted N availability (Bobbink et al., 1998; Sheppard et al., 2013). Field studies of increased N-deposition on peatlands have shown changes in species composition, particularly the loss of *Sphagnum* mosses (Heijmans et al. 2001; Tomassen et al. 2004; Bubier et al. 2007). Such crucial changes in species composition

can have important implications for key processes within the peatland and peatland services (Sheppard et al., 2013).

Beside biological N_2 fixation, N_{dep} and inflow of Nr with groundwater (GW) and surface, fertilization represent quantitatively the dominant input of N in drained and cultivated peatlands. However, amount and form of N applied to cultivated peatlands to maintain productivity widely differ among geographical regions, land-use types and management practices.

1.4.2   Nitrogen mineralization and immobilization

Generally, the SOM represents the largest N pool in the soil, but the organically bound N forms are mostly unavailable to plants, since they acquire nitrogen in inorganic forms (e.g. as NH_4^+, NO_2^- or NO_3^-). Some plants, mainly found in SOM-rich environments in arctic, boreal and alpine regions are able to take up simple organic N forms, like amino acids and proteins (Chapin, 1995; Lipson and Monson, 1998). During SOM decomposition, mostly heterotrophic bacteria (both aerobic and anaerobic) and fungi breakdown polymeric organic N compounds firstly into simpler monomeric N compounds (collectively abbreviated as N_{org}-NH_2) and finally to ammonia (NH_3) or ammonium (NH_4^+); a process called ammonification (Rydin and Jeglum, 2006; McNeill and Unkovich, 2007). In natural peatlands, anoxic conditions predominate in the soil profile since the diffusion of oxygen in water is about 10^4 times slower than that in air, and thus the microbial and chemical demand for oxygen distinctly exceeds its supply (Cleemput van et al., 2007). Generally, anaerobicity slows down decomposition processes and thus also ammonification rates (Vepraskas and Faulkner, 2001). However, SOM decomposition is a complex process and its rates depend on several abiotic and biotic factors, of which the subsequent supply of newly added plant and microbial residues is a key element since this alters the stage of decay and the degree of recalcitrance of the SOM (McNeill and Unkovich, 2007). Under aerobic conditions, once NH_4^+ is released into the soil it will be quickly oxidized by chemolithoautotrophic and very likely by heterotrophic bacteria; a process called nitrification (Ferguson et al., 2007). In this process, the chemolithoautotrophic *Nitrosomonas* firstly convert NH_4^+ (or NH_3) into hydroxylamine (NH_2OH) using the enzyme ammonia mono-oxygenase and then, in a second step, NH_2OH is oxidized to nitrite (NO_2^-) (Ferguson et al., 2007). The further oxidation of NO_2^- to nitrate (NO_3^-) takes place in species of the chemolithoautotrophic *Nitrobacter*. However, due to the very low cell numbers in the soil and the difficulty to cultivate sufficient quantities of cells in the lab to investigate e.g. vesicular membranes

and purified proteins, the biochemistry and molecular biology of chemolithoautotrophic and heterotrophic nitrification is still poorly understood (Ferguson et al., 2007). In natural peatlands, nitrification is usually hampered mainly by anoxic conditions and in bogs additionally by low pH (McNeill and Unkovich, 2007). Nevertheless, O_2 can be transported with the surface- or groundwater inflow in fen peatlands, or by diffusion in the uppermost centimeters of the peat layer, partly creating an aerobic zone and appropriate conditions for nitrification (Cleemput van et al., 2007). Also, peatland-adapted plants with aerenchymatic tissue (e.g. *Carex*, *Phragmites*, *Typha*) can promote an oxic zone in close vicinity to their roots, resulting in the formation of NO_3^- via nitrification as schematically shown in Fig. 4 (Rydin and Jeglum, 2006; Cleemput van et al., 2007). However, these transformation processes only play a minor role for the N cycle of pristine peatlands and were therefore not included in Fig. 3.

After drainage and cultivation of peatlands, N-mineralization and -transformation processes speed up and the concentrations of inorganic N forms, particularly NO_3^-, distinctly increase (Rydin and Jeglum, 2006). Moreover, the application of mineral or organic fertilizers enhances the soil N pool and further stimulates SOM mineralization, leading to additional inorganic N (Gutser et al., 2005; Jones et al., 2007). The annual N supply provided by aerobic peat mineralization can vary between 70 and 570 kg N ha^{-1} yr^{-1} for intensively used crop- or grasslands, whereas under anaerobic conditions, net N mineralization rates are relatively low with approximately 20 kg N ha^{-1} yr^{-1} (Schothorst, 1977; Göttlich, 1990; Flessa et al., 1998; Sonneveld and Lantinga, 2011; Leppelt et al., 2014).

Figure 4 Simplified scheme of plant induced interactions of aerobic and anaerobic zones around a macrophyte root (modified after Cleemput van et al., 2007).

Plant uptake of N depends on the concentration of NH_4^+, NO_2^- and NO_3^- in the soil solution, the physiological plant stage, root distribution and soil water content (Mc Neil and Unkovich, 2007). Since in acid, anoxic peat NH_4^+ is the main form of plant-available

N, many peatland-adapted plants prefer NH_4^+ as N source (Succow and Joosten, 2012). Also, some plants are not able to utilize NO_3^- because they lack the enzyme *nitroreductase*, which is necessary for the reduction of NO_3^- to NH_4^+ (nitrate assimilation) (Succow and Joosten, 2012), and are thus restricted to NH_4^+. To avoid NH_4^+ toxicity, absorbed or reduced NH_4^+ is rapidly assimilated (ammonium assimilation) into glutamine ($C_5H_{10}N_2O_3$), which is readily transported, stored or converted into other amino acids within the plant (Burgiere et al., 1997; Mc Neill and Unkovich, 2007). Generally, plants and microbes compete for inorganic N and once N is immobilized in microbial biomass it can only become plant available by microbe death. The continuous transfer of mineral N into organic material via the incorporation of N into microbial biomass and the release of N during microbial decomposition of SOM or by microbial death is called the microbial mineralization-immobilization turnover (MIT) (Jansson and Persson, 1982; McNeill and Unkovich, 2007). The ratio of immobilization to mineralisation upon decomposition mainly depends on the C/N ratio of the organic substrate; C/N ratios >25 require additional N from the inorganic pool for decomposition of the organic material while C/N ratio <10 lead to a release of N (Göttlich, 1990).

1.4.3 Nitrogen absorption and leaching

When NH_4^+, NO_2^- or NO_3^- is transferred into the soil solution, a portion of the N is taken up by plants and microbes, whereas some N can partially be fixed at the humus surface, is lost in gaseous forms to the atmosphere or is leached to the groundwater (McNeill and Unkovich, 2007). Since organic particle surfaces in peatlands are mostly negatively charged, NH_4^+ is readily absorbed while the anion NO_3^- can relatively easily be leached into the groundwater (McNeill and Unovich, 2007). Furthermore drainage was found to significantly increase leaching of particulate organic N (PON) and dissolved organic N (DON) into the ground- and surface water (total leaching losses of up to 150 kg N ha^{-1} yr^{-1}), whereas leaching losses of PON and DON were found to be of minor importance in natural peatlands (3–6 kg N ha^{-1} yr^{-1}) (Göttlich, 1990).

1.4.4 Gaseous N production processes

In peatland soils several abiotic and biotic processes produce gaseous N compounds (NH_3, NO_x ($NO + NO_2$), N_2O or N_2) which can rapidly lost to the atmosphere. Commonly, microbial denitrification and nitrification are cited as the main sources of gaseous N forms, but this is just a simplification, because microbial metabolic pathways provide a wealth of processes that form gaseous N (Butterbach-Bahl et al., 2013). The reduction

of ionic oxidized nitrogen forms (NO_3^-, NO_2^-) to the gaseous oxides NO_x and N_2O, which may themselves be further reduced to the inert atmospheric N_2, is called biological denitrification (Knowles, 1982). Denitrification presupposes restricted O_2 availability and the presence of easily degradable C-sources (Ambus and Zechmeister-Boltenstern, 2007). The stepwise reduction is catalyzed by several enzymes, mostly in heterotrophic bacteria and fungi which are able to use N oxides as terminal electron acceptors and the organic C as electron donors (McNeil and Unkovich, 2007). Which gas (NO_x, N_2O or N_2) will finally be released to the atmosphere depends inter alia on the degree of anaerobicity, but as well on the organism group carrying out denitrification. For example the fungal denitrification chain is often ending with the formation of N_2O (Philipoot et al., 2011). It was found that with increasing anoxic conditions, the percentage of N_2 increases until it becomes the major gas evolved (Rolston et al., 1978; Davidson, 1991). Moreover, high contents of available NO_3^- inhibit N_2O reductase activity, the enzyme responsible for the conversion of N_2O to N_2 (Blackmer and Bremner, 1978; Firestone et al., 1979; Weier et al., 1993; Regina et al., 1996). Apart from biological denitrification, in acidic soils (pH < 5) with high concentrations of NO_2^-, various purely abiotic reactions can lead to the formation of NO, N_2O and N_2, generally summarized as chemodenitrification (Nelson, 1982; van Cleemput, 1998). However, the importance of chemodenitrification to total gaseous N losses has rarely been studied and further investigations, particularly for drained bog ecosystems, are necessary.

Heterotrophic and autotrophic bacteria and archaea are known to produce NO and N_2O in the course of nitrification. According to Butterbach-Bahl et al. (2013), there are two direct pathways in which N gases can be produced. Firstly, the chemical decomposition of NH_2OH can lead to N_2O and perhaps N_2, with NO as precursor (Wrage et al., 2005; Frame and Casciotti, 2010), and secondly, N_2O can be produced during nitrifier-denitrification (Poth and Focht, 1985; Butterbach-Bahl et al.; 2013). Nitrifier-denitrification describes the oxidation of NH_4^+ (or NH_3) to NO_2^- and its subsequent stepwise reduction to NO, N_2O and perhaps N_2 by the same autotrophic ammonia oxidizers. This process requires conditions of high N availability but low organic C and O_2 availability (Poth and Focht, 1985; Colliver and Stephenson, 2000; Braker and Conrad, 2011). In environments where aerobic and anaerobic microsites exist in close proximity (e.g. drained peatlands or other hydromorphic soils) nitrification and denitrification can occur in a concerted action, called "coupled nitrification-denitrification" (Wrage et al., 2001; Butterbach-Bahl et al.; 2013). Several further processes (e.g. Co-

denitrification with NO or N_2O, dissimilatory nitrate reduction to ammonium (DNRA), anaerobic ammonia oxidation (ANAMMOX)) can lead to the production of nitrogenous gases (Butterbach-Bahl et al.; 2013). However, the source of N gas production in peatlands is often uncertain because within the peat profile different physical and chemical conditions exist and thus several processes producing N gases can run simultaneously and within close proximity (Davidson et al., 1986; Butterbach-Bahl et al., 2013). Generally, gaseous N losses from pristine peatlands were found to be close to zero (Kasimir-Klemendtsson et al., 1997; Flessa et al., 1998; Couwenberg et al., 2011) since all processes leading to N emissions presuppose at least product parts of nitrification. Nevertheless, beside N deposition, small amounts of oxidized N can enter the anaerobic peat layer through GW or surface water inflow in natural fen peatlands, resulting mainly in N_2 or partially N_2O emissions due to denitrification. In contrast, drainage and fertilization have turned peatlands into a significant source of gaseous N emissions. Several studies reported N_2O emissions from drained organic soils ranging from 0 to 61 kg N_2O-N ha^{-1} yr^{-1} for arable lands (Kasimir-Klemendtsson et al., 1997; Augustin et al., 1998a; Flessa et al., 1998; Petersen et al., 2012; Drösler et al., 2013) and ranging from 1.15 to 41 kg N_2O-N ha^{-1} yr^{-1} for grasslands (Velthof et al., 1996; Augustin et al., 1998a; Flessa et al., 1997 and 1998; van Beek et al., 2010 and 2011; Petersen et al., 2012; Beetz et al., 2013; Drösler et al., 2013). Additionally, organic fertilization is known to produce high NH_3 emissions particularly after splash plate application on grassland (Rubæk et al., 1996; Wulf et al., 2002). Depending on climatic factors and manure composition, up to 90% of applied NH_4^+-N can be lost as NH_3 in a few hours (Vandre et al., 1997). Since NH_3 is a highly reactive, alkaline gas it is often deposited within a radius of 1–2 km from the source (Sheppard et al., 2013). However, in dependence of the water table depth, it is assumed that highest gaseous N losses occur in the form of N_2 in drained peatlands. Mainly due to analytical difficulties in the determination of microbial produced N_2, to date the total denitrification potential from peatlands has rarely been estimated. First results revealed annual N_2 emissions ranging between 15 and 650 kg N_2-N ha^{-1} yr^{-1} (Teiter and Mander, 2005; Wray and Bayley, 2007; Mander et al., 2008; Roobroeck et al., 2010; Soosaar et al., 2011), confirming the importance of gaseous N losses through denitrification in the case of peatland disturbance. Beside gaseous losses and leaching of N, highest N losses originate from harvesting crops at intensively used drained peatlands (Göttlich, 1990).

1.5 PEATLANDS AND THE CLIMATE CHANGE

1.5.1 Greenhouse effect and climate change

The earth-atmosphere system is heated by a net absorption of solar short-wave radiation (Bowman, 1990). In an equilibrium state this is balanced by a net emission of long-wave radiation from the earth-atmosphere to space (Bowman, 1990; Jain, 1993). Certain atmospheric gases (called greenhouse gases; GHG) are relatively transparent to the incoming short-wave radiation, but are absorbing the outgoing long-wave (infrared) radiation and re-radiate it back to the earth's surface and space, leading to a net warming of the earth-atmosphere, which is called "natural greenhouse effect" (Bowman, 1990; Jain, 1993). Several natural and anthropogenic drivers (e.g. GHGs such as CO_2, CH_4, N_2O, halocarbons; aerosols such as mineral dust, black carbon, SO_4^{2-}, NO_3^-; change in albedo and solar irradiance) affect the earth's energy budget (IPCC, 2013). The natural greenhouse effect ensures that life is possible on earth since it leads to an increase of global mean surface temperature of about 33°C (IPCC, 1994). The strength of a driver to change the energy flux is quantified as radiative forcing (RF) in units of watts per square meter (W m^{-2}), and is a measure of how the energy balance of the earth-atmosphere system (incoming and outgoing energy) is altered by this driver (IPCC, 2013). The RF of the climate system is dominated by the long-lived greenhouse gases (GHGs) (IPCC, 2007). Greenhouse gases differ in their ability to absorb infrared radiation and also have different turnover-times and pathways of chemical breakdown in the atmosphere. Therefore, GHGs contribute to different extents to the warming of the global climate system (IPCC, 2007). To take this into account, the warming influence of a long-lived GHG may be expressed as Global Warming Potential (GWP), which is based on the time-integrated RF of CO_2 (GWP = 1; IPCC, 2007). Table 1 shows relevant GHGs from peatlands and their GWPs for different time horizons.

Table 1 Peatland relevant GHGs and their lifetimes, radiative efficiencies and direct GWPs relative to CO_2 (according to Forster et al., 2007).

Industrial Designation or Common Name	Chemical Formula	Lifetime [years]	Radiative Efficiency [W m^{-2} ppb^{-1}]	Global Warming Potential for Given Time Horizon		
				20-yr	100-yr	500-yr
Carbon dioxide	CO_2	a	1.4×10^{-5}	1	1	1
Methane	CH_4	12[b]	3.7×10^{-4}	72	25	7.6
Nitrous oxide	N_2O	114	3.03×10^{-3}	289	298	153

a The CO_2 response function is based on the revised version of the Bern Carbon cycle model using a background CO_2 concentration value of 378 ppm. The decay of a pulse of CO_2 with time t is given by

$$a_0 + \sum_{i=1}^{3} a_i \cdot e^{-t/\tau_i}$$

Where $a_0 = 0.217$, $a_1 = 0.259$, $a_2 = 0.338$, $a_3 = 0.186$, $\tau_1 = 172.9$ years, $\tau_2 = 18.51$ years, and $\tau_3 = 1.186$ years.

b The perturbation lifetime for CH_4 is 12 years. The GWP for CH_4 includes indirect effects from enhancements of ozone and stratospheric water vapor.

Global atmospheric concentrations of CO_2, CH_4 and N_2O have increased markedly since 1750, mainly due to human activities (Figure 5; IPCC, 2007). In April 2014, for the first time monthly concentrations of CO_2 in the atmosphere topped 400 parts per million (ppm) throughout the northern hemisphere (WMO, 2014). Also atmospheric CH_4 and N_2O concentrations reached new records of about 1819 parts per billion (ppb) and 325.1 ppb in 2012 (WMO, 2014), and thus exceeded the pre-industrial levels by about 150%, and 20%, respectively (IPCC, 2013). The entire human-induced emissions of GHG had led to a total anthropogenic RF of 2.29 W m^{-2} for 2011 relative to 1750 (IPCC, 2013). Because of that the global climate system has changed. For example, the averaged combined land and ocean surface temperature has increased by 0.85 °C between the years 1880 and 2012 (IPCC, 2013). Additionally, a shift in precipitation as well as extreme weather and climate events (e.g. increase in heat waves and heavy precipitation events) and a decrease in the Greenland and Antarctic ice sheets, with the consequence of a rising sea level have been observed (IPCC, 2013). The future global and regional climate changes are unknown. However, despite their uncertainty all climate models predict a further reinforcement of the changes observed in the last decades. In dependence of the chosen model scenarios of anthropogenic forcing, the simulations predict an increase of global mean surface temperatures for 2081–2100 relative to 1986–2005 of 0.3–1.7°C or 2.6–4.8°C in the case of the RCP2.6 or RCP8.5 scenario, respectively (see Fig. 6; ICPP, 2013).
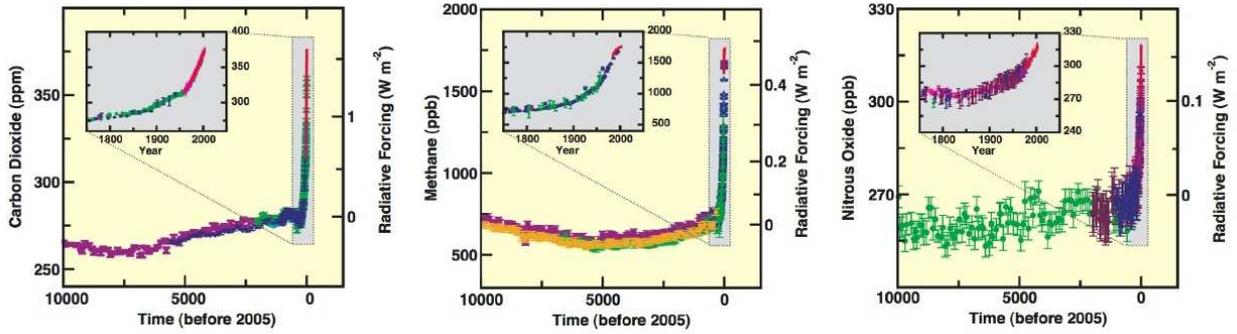
Figure 5 Atmospheric concentrations of the three important greenhouse gases CO_2, CH_4 and N_2O over the last 10,000 years. Increases since about 1750 (inset panels) are attributed to human activities in the industrial era (source: IPCC, 2007).
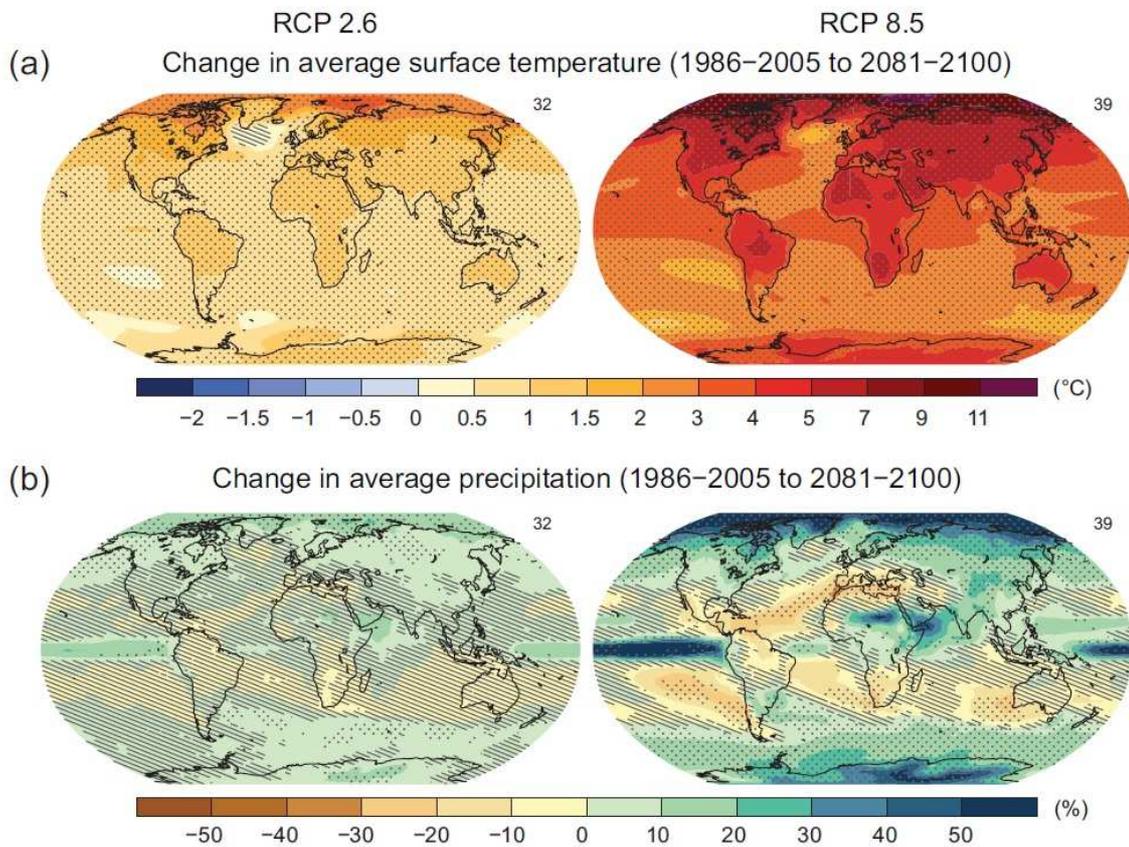
Figure 6 Maps of CMIP5 multi-model mean results for the scenarios RCP2.6 and RCP8.5 in 2081–2100 of (a) annual mean surface temperature change, (b) average percent change in annual mean precipitation, For the Fifth Assessment Report of IPCC, the scientific community has defined a set of four new scenarios, denoted Representative Concentration Pathways (RCPs, see Glossary). They are identified by their approximate total radiative forcing in year 2100 relative to 1750: 2.6 W m^{-2} for RCP2.6, 4.5 W m^{-2} for RCP4.5, 6.0 W m^{-2} for RCP6.0, and 8.5 W m^{-2} for RCP8.5. (source: IPCC, 2013)

1.5.2 Contribution of peatlands to climate change – Past, Present and Future perspectives

Globally, peatlands store approximately 30% and 16% of the world's soil C and N, although they cover less than 3% of the earth's land surface (Yu et al., 2010 & 2011; Limpens et al., 2006). Based on a simple conversion from change in carbon pool to atmospheric CO_2 (1 ppmv = 2.123 Gt C), by taking into account the compensation effect of ocean outgassing, Charman et al. (2013) estimated that the uptake of CO_2 by peatlands reduced atmospheric CO_2 concentration by 7–12 ppm over a 1000 yr period. Thus the capability of pristine peatlands to act as a long-term sink for C and N, have significantly contributed to global cooling since the last ice age (~10,000 years) (Frolking and Roulet, 2007; Limpens et al., 2008). As extensively discussed in section 1.3 and 1.4, today peatlands are undergoing major transformations due to human alteration (e.g. drainage, cultivation, peat extraction and burning) and thereby the radiative forcing (RF) caused by peatland-atmosphere exchange has shifted from a net cooling to a net warming.

In 2010, total anthropogenic GHG emissions reached 49 ± 4.5 Gt $CO_{2\ eq.}$ yr^{-1} and were the highest between 2000 and 2010 (IPCC, 2014b). Carbon dioxide is the most important GHG accounting for 76% of total anthropogenic GHG emissions in 2010 (IPCC, 2014b). Its concentrations have increased primarily due to emissions from fossil fuel combustion (38.2 Gt $CO_{2\ eq.}$ yr^{-1}) and secondarily due to emissions from land-use change (deforestation, agricultural soil and nutrient management, livestock farming), which accounts for approximately 10–12 Gt $CO_{2\ eq.}$ yr^{-1} (IPCC, 2014b). From 1750 to 2011, CO_2 emissions from deforestation and other land use change are estimated to have released 180 Gt C (Hooijeret al., 2010). According to the Food and Agriculture Organization of the United Nations (FAO), globally 250,000 km^2 of peatland area are estimated to be drained, leading to approximately total annual GHG emissions of 0.9 Gt $CO_{2\ eq.}$ yr^{-1} in 2010 (FAOSTAT, 2013). Furthermore, Joosten (2010) estimated worldwide 500,000 km^2 of degraded peatland, leading to annual CO_2 emissions of approximately 1.30 Gt CO_2 yr^{-1} (excl. extracted peat and fires) in 2008, of which 0.5 Gt are emitted from Indonesia, 0.174 Gt from the EU (27 countries) and 0.161 Gt from Russia (see Fig. 7). Including peat fire and peat extraction, Joosten (2010) assumed that degraded peatlands (0.3% of earth's land surface) may exceed 2 Gt CO_2 yr^{-1} globally, which corresponds to 5.4% of global anthropogenic CO_2 emissions. Nevertheless, under

the present climate, northern peatlands still act as small C sinks of up to 60 g C m^{-2} yr^{-1} (Lafleur et al., 2003; Roulet et al., 2007; Nilsson et al., 2008; Sagerfors et al., 2008).
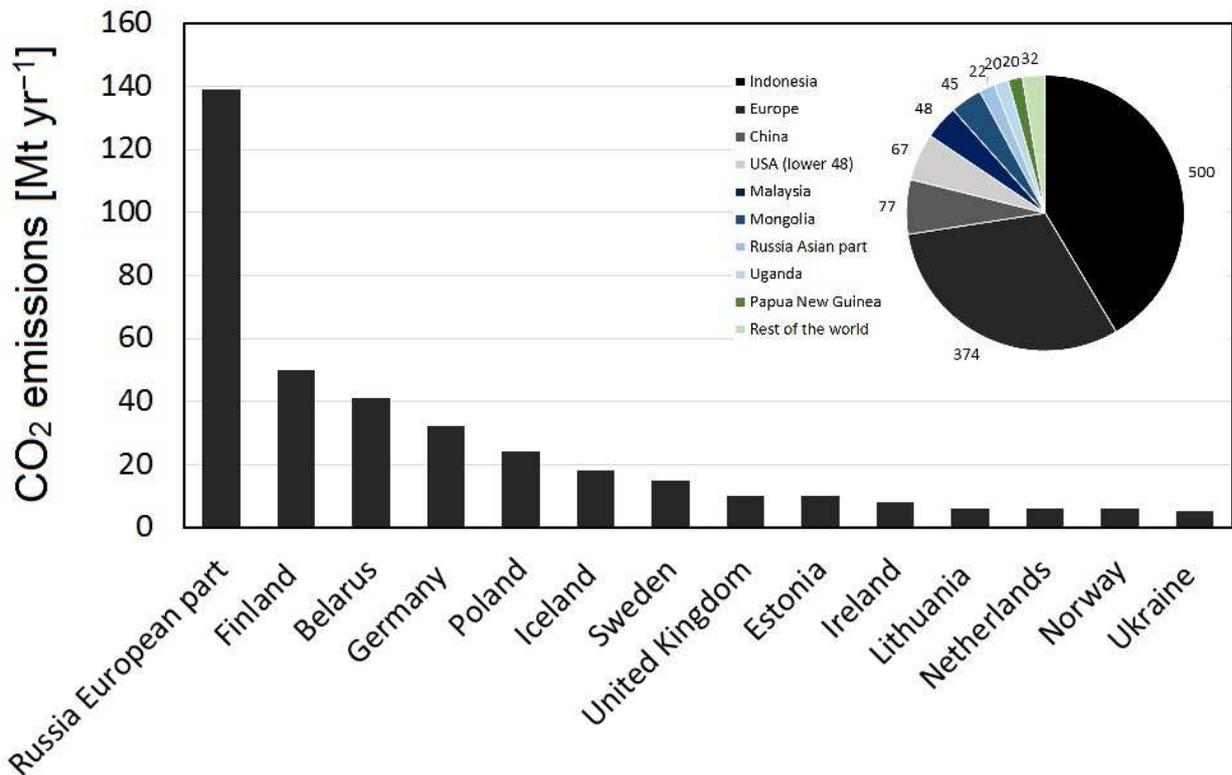
Figure 7 Estimated CO$_2$ emissions in Mt (10^6 t) from degrading peat for the year 2008. Bar chart shows European countries; pie chart shows total global emissions in Mt (Joosten, 2010).

Beside the uptake capacity of CO$_2$ since the last glaciation, peatland development has likewise significantly increased atmospheric CH$_4$ concentrations as is documented in atmospheric records from ice cores (Blodau, 2002; Yu, 2010). Methane is the second most important anthropogenic GHG (16% or 7.8 ± 1.6 Gt CO$_{2\ eq.}$ yr^{-1}) and its increase has primarily been caused by human activities related to agriculture, natural gas distribution and landfills (IPCC, 2014b). However, CH$_4$ is also released from natural ecosystems, of which pristine peatlands and other wetlands are the largest sources with estimated emissions of 55–150 Tg CH$_4$ yr^{-1} (Watson et al., 2000). According to Rydin and Jeglum (2006), northern peatlands are responsible for 6–9% of the global CH$_4$ emissions. However, as previously pointed out, CH$_4$ emissions from drained organic soils (e.g., drained peat soils and soils developed in wet conditions) are assumed to be negligible and, depending on drainage status, they may even become small atmospheric CH$_4$ sink (IPCC, 2014a).

N$_2$O is the third most important GHG accounting for 6.2% of total anthropogenic GHG, which corresponds to 3.1 ± 1.9 Gt CO$_{2\ eq.}$ yr^{-1}. It is emitted into the atmosphere from

both natural (approx. 60%) and anthropogenic sources (approx. 40%), including oceans, soils, biomass burning, fertilizer use, and various industrial processes (IPCC, 2007; IPCC, 2014b). In particular, cultivated organic soils are considered as hotspots of N_2O emissions (Kasimir-Klemedtsson et al., 1997; Freibauer et al., 2004; Leppelt et al., 2014). According to Leppelt et al. (2014), direct N_2O emissions from organic soils in Europe amount up to 149.5 Gg N_2O-N yr^{-1}, which represents 13% of total European N_2O emissions reported in the European Union (EU) greenhouse gas inventory of 2011. Additionally, peatland cultivation can lead to NH_3 as well as NO_X ($NO + NO_2$) emissions. Besides acidification and eutrophication, NH_3 deposition promotes the formation of N_2O at the immission sites and therefore NH_3 is considered as indirect GHG (Moiser, 2001). However, the contribution of peatland cultivation to NH_3 emissions is still unclear, but overall agriculture is responsible for 95.3% of anthropogenic NH_3 emissions in Germany (Haenel et al., 2010). Due to high reactivity, NO and NO_2 are considered as Near-Term Climate Forcers (NTCFs) since they are involved in a sequence of photochemical reactions, which may lead to the formation of the GHG ozone (O_3) and hydroxyl radicals ($OH\cdot$) in the troposphere (IPCC, 2007). Additionally, NO_X also causes RF through NO_3^- formation, and via reaction with CH_4 it increases stratospheric water vapour (IPCC, 2013).

As previously mentioned, current climate scenarios partly predict a drastic increase in global warming and further aspects of the climate. This actually triggered questions as to what extent this will affect peatlands globally. Principally, higher temperatures are assumed to increase peat decay, causing a positive feedback on climate warming (Charman et al., 2013). As depicted in Fig. 6a, boreal and arctic regions are predicted to warm faster and more strongly than temperate latitudes (Treat et al., 2013). Also precipitation is predicted to significantly increase in this region (Fig. 6b; IPCC, 2014b). Worldwide, 75–85% of all peatlands are situated in the northern hemisphere, storing 270–547 Gt C (Turunen et al., 2002; Frolking et al., 2011; Charman et al., 2013). It is expected that increased temperatures cause permafrost melting as well as increased thawing of the perennially-frozen peat in the arctic and northern boreal region, leading to water-saturated conditions. This, together with higher temperatures, will probably result in anaerobic decomposition, leading to the production of CH_4 (Tarnocai, 2006; Arneth et al., 2010 Charman et al., 2013). In the RCP8.5 climate scenario, thawing of frozen peat and permafrost is assessed to release 50 to 250 Gt C as CO_2 and CH_4 to the atmosphere, causing a positive feedback on climate change (IPCC, 2013). In other

latitudes (e.g. south boreal wetland belt, temperate and tropical regions), higher temperatures will cause higher evapo-transpiration leading to drought conditions. Under such conditions, peatlands can become a net source of CO_2 and also N_2O due to enhanced decomposition (Tarnocai, 2006; Hooijer et al., 2010; Sulman et al., 2010). Moreover, dryer conditions enhance the risk of wild- and peat fires, particularly in the tropic regions, further promoting CO_2 emissions (Turetsky et al., 2004; Limpens et al., 2008). In contrast, several authors (e.g. Tarnocai, 2006; Arneth et al., 2010; Charman et al., 2013) suggested that in the mid and high arctic latitudes, higher temperatures, longer growing seasons, and higher atmospheric CO_2 could stimulate photosynthesis leading to increasing biomass production and thus promote peat accumulation, and may provide a negative climate feedback in a warmer future (Yu et al., 2011). However, climate change as well as feedbacks in vegetation type shifts and land-use changes (e.g. drainage of peatlands) are difficult to predict with highly uncertain projections, limiting our ability to draw conclusions. Nevertheless, the impact of climate change will be considerable and trigger feedback mechanisms that might further accelerate climate change (Houghton and Woodwell, 1989; Davidson and Janssens, 2006).

1.6 OBJECTIVES

The present PhD thesis was part of the joint research project "Organic soils"; Acquisition and development of methods, activity data and emission factors for the climate reporting under LULUCF/AFOLU. In the course of this joint research project, several open questions were to be answered to fulfil the requirements for the German climate reporting according to the extended methodology of the Tier 2 and Tier 3 approach of the IPCC guidelines. According to Drösler et al. (2008b) the main goals of the joint research project were:

1) Geo-referenced assessment of organic soils according to the IPCC guidelines to draw up a dynamic land register of organic soils in Germany.
2) Assessment of national EF for different soil-types (including soils containing 15–30% SOM), climate regions, land-use types (including forests) and management practices to reduce the uncertainty derived from the high spatial and temporal variability in GHG fluxes.
3) Development, adjustment, parametrization and validation of models for the calculation of GHGs, derived from drained organic soils.

4) Assessment and development of methods and parameters for the nationwide regionalization of investigated data sets.

5) Quantification of uncertainties in determined activity data sets and emission calculation methods, as well as the estimation of the overall uncertainties from the national climate reporting for drained organic soils.

The objectives of the present thesis focused on **three key issues**, which fall within the scope of point 2 of the above mentioned goals of the joint research project.

The **first key issue** (**study I**) addressed the question as to what extent the soil organic carbon (SOC) content affects the GHG release of drained organic soils under arable land and intensively used grassland in South Germany. Both land-use types have been regarded as the main producers of CO_2 and N_2O, due to tillage and fertilization which additionally enhance microbial SOM decomposition and N turnover at drained organic soils (Kandel et al., 2013). Beside management practices, several other physical and chemical factors control the intensity of mineralization processes of which the SOC content and quality counts as one key element that may control the release of GHG emissions. It can be assumed that at a similar aeration status and temperature, mineralization processes are more intensive at peatlands containing higher amounts of SOC. However, most studies have focused on typical drained peatlands, rich in organic carbon (C_{org}), whereas former peatlands and associated organic soils containing <12% C_{org} were neglected by scientific research, although it is unknown whether they emit less GHG's compared to typical drained peatlands.

Following hypotheses were examined:

(i)   Greenhouse gas emissions significantly increase with increasing SOC content in drained organic soils.

(ii)   Greenhouse gas emissions from arable soils exceed GHG emissions from intensive managed grassland soils.

The first key issue provides the basis for **Publication I**.

The **second key issue** (**study II**) focuses on the effects of different organic fertilizers (cattle slurry vs. biogas digestate), on N_2O and CH_4 fluxes for two types of organic soils with contrasting SOC and N contents. Furthermore, the question was targeted as to what extent biogas digestate and cattle slurry application affect N availability and grass yield.

The background of this was that Germany has become the largest biogas producing country in the world (Weiland, 2010). The strong increase in the number of biogas plants has caused a land-use change towards agro-biomass production and additionally raised land-use intensity to satisfy the huge demand for fermentative substrates (Don et al., 2011). Today, also drained peatlands are increasingly used to satisfy the huge demand for fermentative substrates (e.g., energy crops, grass silage) and the remaining nutrient-rich residues are returned to the peatlands to maintain nutrient supply and grass yield. Generally, drained organic soils are considered as hot spots for N_2O emissions and organic fertilization is known to increase N_2O emissions from managed grasslands. Digested products are considered to be more recalcitrant than fresh slurry and some authors (e.g. Clemens and Huschka, 2001; Oenema et al., 2005; Möller and Stinner, 2009) assume that microbial degradation is slower, resulting in fewer anoxic microsites and reduced N_2O emissions compared with fresh slurry. However, the effect of biogas digestate application on drained organic soils has not been studied before.

Following hypotheses were examined:

(i)   More N_2O is emitted after biogas digestate than after slurry application because of higher NH_4^+–N concentrations in the substrate and, thus, higher NH_4^+–N amounts when using equal volumetric application rates. The more recalcitrant nature of the carbon in the biogas digestate does not matter for GHG formation in high organic carbon soils.

(ii) N_2O emissions increase with increasing soil C_{org} and N content due to more favorable conditions for denitrification after organic fertilizer application.

(iii) Biogas digestate leads to a significantly higher grass yield and N-use efficiency compared to cattle slurry due to the higher N availability of the digestate.

In addition, NH_3 volatilization was determined in the course of one fertilization event to obtain first clues with respect to the effects of soil and fertilizer types on this gas. The second key issue provides the basis for **Publication II**.

The **third key issue** (**study III**) addressed the questions as to what extent drainage enhances the emissions of N_2O from black alder (Alnus glutinosa (L.) Gaertn.) forests and as to how N turnover processes and physical factors influence the production of N_2O and total denitrification rates. Unlike most other wetlands, black alder forests on peatland represent non N-limited ecosystems. To date, there are only a few studies on

N turnover and N transformation processes in black alder forests on peatlands, but first results indicate that these ecosystems show high N-transfer rates between the atmosphere, the alder trees and the soil (Augustin, 2003). High N_2O emissions from alder forests on drained organic soils are attributed to alternating water tables and to the incorporation of high amounts of easily decomposable N into the ecosystem by symbiotic N_2-fixation of the alder trees. The high magnitude of observed N_2O emissions from black alder forests, which exceed by far the IPCC default emission factor, requires in-depth process understanding of N_2O formation, not least because of the attempt to reduce the climate impact of drained fen peatland by rewetting and afforestation with black alder trees (Schäfer and Joosten, 2005; Wichtmann and Joosten, 2007).

Following hypotheses were examined:

(i)   In drained black alder forests high N_2O emissions occur due to accelerated N turnover and N mineralization.

(ii) N_2O losses are replaced by N_2 losses during periods of temporarily high water levels. In contrast, in permanently waterlogged peat soils NO_3^- limitation results in negligible N_2O and N_2 losses.

The third key issue provides the basis for **Publication III**.

2 METHODS FOR OBSERVATION AND QUANTIFICATION OF TRACE GAS EMISSIONS

In the last two decades, a range of different measurement techniques, highly precise sensors and analyzing units as well as various mathematical models have been developed to determine and quantify trace gas exchange between landscapes and the atmosphere. Figure 8 summarize the used measurement techniques of the present work. All applied methods are extensively explained in the three papers attached to this thesis, but various other measurement approaches are available and thus should be briefly compared and discussed. From the outset, it should be emphasized that each technique has its advantages and disadvantages. There is no system that is superior in all situations; instead each has its own "niche". Table 2 gives an overview of different approaches for measuring trace gas exchange between landscapes and the atmosphere, presenting their advantages and disadvantages.

Figure 8 Different Measurement systems and analyzing units for the determination of soil gas fluxes which were used in the present study. a) Closed dynamic manual chamber system for the determination of NEE (transparent chamber) or R_{ECO} (opaque chamber, not shown) coupled to an b) infrared gas analyzer (IRGA, LI-820); c) Closed static manual chamber system for the determination of N_2O and CH_4 fluxes. Gas samples were analyzed via a GC d) equipped with an FID and an ECD; e) Open dynamic chamber method ("Dräger-Tube Method" for the determination of NH_3 volatilization, NH_3 concentrations were analyzed via ammonia indicator tubes f); g) Helium-oxygen flow-through the chamber incubation system at the laboratory of the Institute of Landscape Matter Dynamics, Leibniz Centre for Agricultural Landscape Research (ZALF; Müncheberg, Germany) for the simultaneously determination of N_2, N_2O, CO_2 and CH_4 fluxes in an controlled environment, h) concentration of N_2 were analyzed by a micro-GC equipped with a TCD. Concentrations of trace gases were analyzed by a GC equipped with an FID and an ECD.

To date, chambers are the most common tool for measuring trace gases (Denmead, 2008). Their operating principle is simple, mostly inexpensive, normally requires no

electricity supply in the field and allows process studies and experiments with numerously treatments and very small gas fluxes (up to 100 times smaller compared to micrometeorological applications; Denmead, 2008). However, the chamber technique also has disadvantages owing to the effects of the measurement itself on environmental variables (e.g. temperature, soil evaporation rate, relative humidity, plant damage, disturbance of diffusion gradient, inhibition of the flux through a buildup in head space concentration), to the limited coverage of the soil surface, requiring lots of replications, and perhaps, most important, to the temporal resolution of gas flux measurements (Butterbach-Bahl, 2013). The later can greatly impoved by the use of automatic chamber systems, with the consequenc of higher demands for infrastructure and higher costs. Where practicable, flow-through chambers are to be preferred because these enable controlling the increase in gas concentration by adjustment of the volume flow rate (Denmead, 2008). However, this technique requires highly precise analyzing units to detect even slight flux rates reliably and it is also more complicated in its technical implementation. In the last few years, high effort has been made to construct fast (less than 1Hz), highly precise and sensitive operating analyzers, mostly based on laser technology (e.g. tunable diode laser (TDL) or quantum cascade laser (QCL)) for the micrometeorological detection of trace gases (Denmead, 2008; Butterbach-Bahl, 2013). In combination with chamber techniques (fast-Box method), they allow a significant reduction in the closure time of chambers, reducing unwanted impacts on environmental parameters to a minimum, and they further enable the investigation of spatial variability by quickly changing chamber positions (Butterbach-Bahl, 2013). Additionally, an instant verification of the measurement quality is given through the on-line concentration detection. However, the big disadvantage of these techniques is the enormous cost.

Micrometeorological applications are mostly related to eddy covariance, eddy accumulation and flux-gradient methods. They are very suitable for the continuous detection of integrated trace gas fluxes from large landscapes with homogeneous surfaces, without any restrictions on gas exchange (Denmead, 2008). Usually, gas concentration and wind speed in the air are measured with fast response instruments (>10 Hz) and average values are calculated over sampling periods of 15 min to 1 h (Denmead, 2008). Thus, temporal and spatial variability are considered but the spatial variability is unquantifiable, since typically no replications exist for micrometeorological approaches. From the above-mentioned three methods, the eddy covariance approach is the micrometeorologist's preferred approach because it gives the vertical flux at the

point of measurement. The disadvantages of micrometeorological approaches are diverse and include dealing with the effects of simultaneous fluxes of heat and water vapour, problems occurring from pre-conditioning air samples before measuring their gas concentration, problems with lags between measuring vertical wind speeds and gas concentrations, correct mounting heights in relation to sensor path lengths, flow distortion from neighboring instruments and support structures, and others. However, the main issue results from the fact that the measuring principle relies on theories of turbulent transport in the atmosphere, so that this method can only be applied when weather conditions are favorable. This applies equally to the backward Lagrangian stochastic dispersion technique (bLs). Moreover, micrometeorological techniques are very cost-intensive and often require high demands for the infrastructure.

Mass balance techniques are very suitable for small and well-defined areas, irrespective of homogeneous or heterogeneous source distributions and thus fill the gap between chamber and micrometeorological approaches (Denmead, 2008). Mass balance methods use a vertical gas concentration profile in combination with the vertical windspeed gradient (Hensen, 2011). They have a simple theoretical basis and require no special form of the wind profile or corrections for atmospheric stability like micrometeorological approaches (Denmead, 2008). Moreover, there is no need for very fast sensors as is the case for eddy covariance measurements, which lowers the cost. However, mass balance methods require a small background concentration and therefore are only suitable for determining emissions of NH_3 and N_2O in fertilizer applications studies or CH_4 emissions from small-scale grazing systems (Denmead, 2008; Hensen, 2011). However, some uncertainty remains about the magnitude of the diffusive flux which necessitates mathematical corrections or the implementation of fast response instrumentation (Denmead, 2008).

The backward Lagrangian stochastic dispersion techniques (bLs) represents a suitable tool for measuring gas emissions from small, well-defined source areas of any shape (Denmead, 2008). The bLs method neither alters the gas exchange nor ambient surface conditions (e.g. surface temperature, wind speed; Gao et al., 2009). As the mass balance method, the bLs is particularly appropriate for determining gas fluxes with high temporal resolution from treated fields and intensive animal production systems with small background concentrations. Its main advantage is that gas fluxes can be determined from just one concentration measurement, providing that the background concentration is known. This keeps the installation costs lower compared to mass

balance methods (Denmead, 2008). However, gas measurements based on laser or FTIR systems will often cause the corresponding costs. The main disadvantage of the bLs technique derives from the complex parameterization of turbulent transport, so that these methods can only be applied when weather conditions are favorable as mentioned above.

Table 2 Advantages and disadvantages of different approaches for measuring trace gas fluxes between landscapes and the atmosphere. The extent to which a system is suitable or not can range between ++ to ––, in which ++ stands for very suitable or low and –– stands for unsuitable or high (extended from Drösler, 2008b).

	Chamber techniques[1]		Micrometeorological techniques	Mass balance methods[3]	Backward Lagrangian stochastic dispersion technique
	manual systems	automatic systems	Eddy covariance[2]		
Undisturbed gas exchange	+/––	+/––	++	++	++
Detecting the spatial variability in gas exchange	+/–	–	–	–	–
Detecting the temporal variability in gas exchange	–	++	++	++	++
Suitable for — homogeneous, large-scale	––	––	++	+/–	+/–
Suitable for — Heterogeneous, small scale	+	+	–	++	++
Suitable for — process studies and experiments	++	++	––	–	–
Sensitivity of gas fluxes	++/–	++/–	+/–	–	–
Detectable gases	All trace gases*	All trace gases*	CO_2, NH_3, N_2O, CH_4	CH_4, N_2O, NH_3	CH_4, N_2O, NH_3
Workload	––	++	++	++	++
Straightforward application	++	+	–	+	–
Demands on infrastructure including power supply	++/–	–	––	–	–
Costs	++/–**	–/––**	–/––**	+/––**	+/–**

[1] including open and closed systems; closed systems contain static and dynamic gas detection.
[2] including eddy accumulation, relaxed eddy accumulation and flux-gradient methods.
[3] including open and closed systems with adaption of open systems to circular source areas.
* e.g. CO_2, CH_4, N_2O, NH_3, and several more; dissection in subcomponents and production processes possible.
** In dependence of used analyzing unit and detection precision.

3 GENERAL DISCUSSION AND OUTLOOK

3.1 IMPACT OF SOC AND MANAGEMENT

In the present thesis, it is hypothesized that drainage of peatlands, rich in SOC, leads to significantly higher GHG emissions compared to organic soils containing lower SOC contents. With exception of N_2O fluxes, this assumption could not be confirmed by the present study. Instead, the **first study** gave evidence that the GHG balance of a drained and intensively cropped organic soil is mainly affected by management (e.g. fertilization, tillage, crop target) rather than SOC content or land-use type.

Despite significantly higher annual sums of GPP, calculated sums of NEE were significantly higher at the grassland treatments compared to treatments of the arable land, which was caused by distinctly increased R_{ECO} at these plots (Table 3, **Publication I.**). With the exception of the maize plot at the C_{medium} site in the year 2011, all treatments showed positive NEE balances in both years investigated, as expected for drained organic soils and as commonly reported in the literature. (e.g. Maljanen et al. 2001; Grønlund et al., 2006 and 2008; Maljanen et al., 2010; Elsgaard et al., 2012; Beetz et al., 2013; Drösler et al., 2013). NEE rates observed in **study I** ranged between -6 ± 546 and 1707 ± 619 g CO_2-C m^{-2} yr^{-1} at the arable land and between 1354 ± 740 and 1823 ± 851 g CO_2-C m^{-2} yr^{-1} at the grassland. Generally, the annual CO_2 emissions observed were in the upper range or partly higher compared to literature values (e.g. Maljanen et al., 2001, 2007, 2010; Grønlund et al., 2008; Höper et al., 2008; Leifeld et al., 2011; Elsgaard et al., 2012; Beetz et al., 2013; Drösler et al., 2013; Leifeld et al., 2014; Renou-Wilson et al., 2014) and also partly exceed the IPCC default emission factors up to threefold. However, in respect to the SOC content, the comparison of the determined CO_2 or CH_4 fluxes is difficult because no further studies exist concerning intensively used organic soils with SOC contents comparable to the C_{medium} sites. Nevertheless, our findings were corroborated in the framework of the joint research project "organic soils" (e.g. Tiemeyer et al., 2014; M. Drösler, personal communication 2014) and are in line with investigations from Leiber-Sauheitl et al. (2014), who reported that CO_2 fluxes from extensive grasslands on organic soils were not related to different SOM contents. It can be assumed that not only the SOC content itself acts as a key factor controlling the CO_2 release, but the quality of SOC and particularly the proportion of SOC which is exposed to aerobic mineralization. However, in accordance to investigations made by Leiber-

Sauheitl et al. (2014) and Tiemeyer et al. (2014), no relation was found between the effective C stock and the C flux components (Publication I, Figure 10), demonstrating the complexity of SOM decomposition. Nevertheless, the separation of the dataset according to land-use types revealed comparable effective C stocks at the grassland sites, probably explaining the equal CO_2 loss rates despite different SOC contents. Recent studies have shown that in particular the SOM quality and its labile and more recalcitrant fractions act as key variables affecting the decomposability of the SOM and thus control CO_2 fluxes from peatlands (Byrne and Farrell, 2005; Heller and Zeitz, 2012; Leifeld et al., 2012). In the present study, the SOM quality was not examined, but all sites showed equal narrow C/N ratios, indicating high organic matter quality at all sites investigated, perhaps explaining the comparable CO_2 emission patterns.

Despite different SOC and groundwater levels, no differences in CH_4 fluxes were found between the two soil types investigated (**Publication I.** and **II.**). In contrast to literature (e.g. Maljanen et al., 2010) CH_4 emissions of the arable land significantly exceeded those of the grassland (**Publication I.**), whereas neither fertilization with cattle slurry nor with biogas digestate led to significantly different CH_4 emission patterns, despite of distinctly different fertilizer composition. Generally, the position of the groundwater table is considered as the key factor, regulating methanogenic and methanotrophic processes in peatlands (Whalen, 2005). Thus, it can be assumed that the intensive drainage effectively inhibited microbial methanogenesis at both soil types investigated. However, annual CH_4 exchange rates were low in the present study, ranging between -0.10 and 0.51 g C m^{-2} yr^{-1} at the arable plots and between -0.11 and -0.02 g C m^{-2} yr^{-1} at the grassland plots. However, in accordance with several other authors (e.g. Maljanen et al., 2010; Petersen et al., 2012; Schäfer et al., 2012; Drösler et al., 2013; Renou-Wilson et al., 2014), **study I** and **II** demonstrate that CH_4 exchange was only of minor importance, considering total GHG emissions.

In line with our hypotheses examined in **study I** and **study II** significantly higher N_2O emissions were found at the C_{high} sites compared to the C_{medium} sites, regardless of land-use type and management practices. Likewise, significantly higher NO_3^- contents and higher GW levels were observed at these sites. From the calculated net C losses in the **first study** it can be concluded, that high amounts of N must be released in the course of SOM degradation. As revealed by the **second study**, an equal amount of available N resulted in a higher proportion of N_2O losses at the sapric histosol compared to the molic gleysol (**Publication II.,** Figure 5b), which was attributed to the probably more

favorable soil conditions for denitrification. Nevertheless, the source of N_2O production in soils is often uncertain since aerobic and anaerobic microsites occur within close proximity and thus different biotic and abiotic N_2O production processes can run simultaneously (Davidson et al., 1986; Butterbach-Bahl et al., 2013).

In the framework of the **second study**, fertilization was found to significantly increase N_2O emissions from the grasslands investigated. However, the **second study** demonstrated that NH_3 volatilization is quantitatively the most important N loss from surface application (Flessa and Beese, 2000), particularly when slurries contain high concentrations of NH_4^+ and had high pH values. In line with our expectations, the **second study** revealed that the application of biogas digestate significantly enhances annual N_2O emissions as well as biomass production and NUE compared to the application of cattle slurry. Calculated harvest-related N_2O emissions averaged 0.20 kg N_2O-N t^{-1} DM^{-1} at the biogas treatments compared to 0.17 kg N_2O-N t^{-1} DM^{-1} at the cattle slurry treatments, demonstrating that the higher grass yield could not compensate for the significantly higher N_2O emissions at these treatments. However, contrary to our hypothesis, the significantly higher N_2O emissions as well as biomass production from the digestate treatments cannot solely be explained by the higher content of available N in the biogas digestate, since the amount of applied NH_4^+–N in the substrate was not markedly different. Nevertheless, the much higher N-uptake at the biogas treatments (**Publication II.**, Fig. 5a) indicated that the application of this fertilizer resulted in a markedly higher N availability probably promoting N_2O and biomass production. Beside a assumed slightly enhanced SOM mineralization, caused by differences in fertilizer composition (e.g. C/N ratio, pH), it is conceivable, that the significantly increased biomass production at the biogas treatments itself is related to the increased N_2O emissions, due to the stimulating effect of plant roots (i.e. root exudates) on denitrification activity (Klemedtsson et al., 1987; Bakken, 1988). In addition, **study I** revealed significantly higher R_{ECO} rates at the grassland site compared to the arable sites, corroborating the assumption that, firstly fertilization probably increase SOM mineralization (Gutser et al. 2005; Jones et al., 2007), and secondly, a higher GPP may increase the transport of photosynthates to the plant rhizosphere, favoring bacterial metabolism through increased root exudates (Mounier et al., 2004; Henry et al., 2008; Sey et al., 2010). However, in line with Velthof et al. (2003) the **second study** confirms our assumption that the more recalcitrant nature of the carbon in the biogas digestate is not predominantly responsible for GHG formation in high organic carbon soils, since

sufficient metabolizable C is generally widely available in those soils. Nevertheless, N_2O fluxes from the unfertilized arable treatments significantly exceeded those of the grassland sites, which was also reported by Maljanen et al. (2007 and 2010) and Petersen et al. (2012) and was additionally confirmed by Leppelt et al. (2014) for European organic soils. (**Publication I.**, Table 3). Determined N_{min} contents in the soil at the arable sites were significantly lower compared to the grassland sites, but Silvan et al. (2005) assumed that the temporary absent of plants resulted in a higher N availability for microbial denitrification and thus in higher N_2O fluxes at croplands. Moreover, GW levels were found to be significantly higher at the arable sites compared to the grassland sites, further favoring denitrificative N losses.

Estimated net climate effects (GWPs) ranged between 1837 and 4562 g $CO_{2eq.}$ m^{-2} yr^{-1} at the arable land and between 5884 and 7095 g $CO_{2eq.}$ m^{-2} yr^{-1} at the grassland (**Publication I.**, Table 6), revealing a very high release of greenhouse gases from all treatments. According to the IPCC (2014a), the emission factor for total GWP from temperate drained cropland averaged 3508 g $CO_{2eq.}$ m^{-2} yr^{-1} with a 95% confidence interval of 2769 to 4292 g $CO_{2eq.}$ m^{-2} yr^{-1}, whereas the GWP of deep drained nutrient-rich grassland averaged 2662 g $CO_{2eq.}$ m^{-2} yr^{-1} with a 95% confidence interval of 2070 to 3266 g $CO_{2eq.}$ m^{-2} yr^{-1}. Thus, for the grassland investigated, estimated GWPs are markedly higher compared to the default emission factor of the Tier 1 approach and, in addition, are located far outside of the 95% confidence interval. In contrast, estimated GWPs of the arable land cover the entire span of the 95% confidence interval, with slightly higher values at the upper confidence limit and distinctly lower values at the lower confidence limit. However, Kasimir-Klemendtsson et al. (1997) reported distinctly higher net CO_2 exchange rates ranging from 800 to 11500 g CO_2 m^{-2} yr^{-1} for farmed organic soils, demonstrating the high bandwidth of measured CO_2-balances. Nevertheless, the question arises, to what extent model as well as measurement uncertainties triggered the magnitude of estimated GWPs. As mentioned above, N_2O and CH_4 contributed only negligible to total GWP but we cannot rule out that we may have missed at least high N_2O emission events, due to our regular measurement intervals. It is well known that peak emissions significantly alter total GHG balances, as was also confirmed in the present study. However, the main uncertainty in **study I** derived from R_{ECO} as well as GPP model accuracy constrained by partly insufficient parameterisation. Particularly, management activities like ploughing at the arable sites as well as harvesting and fertilization at the grassland sites were partly not sufficiently accurate covered by

additional measurements. In addition, despite high model accuracy, the calculated PBIAS revealed a slightly model overestimation bias for R_{ECO} and NEE for both years (R_{ECO} only in 2010). Thus, modeled R_{ECO} and calculated NEE rates should be considered as a conservative estimation. Nevertheless, the comparison of GWPs relative to each other, regarding the effect of SOC contents or within land-use types, is unaffected of potential bias in the absolute magnitude of GWP.

With the exception of Denmark, organic soil containing <12% C_{org} were so far neglected in national GHG emission inventories. Since GHG emissions are not expected to be zero from organic soils containing 6–12% C_{org}, Denmark introduced a fixed emission factor, half as much as for drained peatlands (Nielsen et al., 2012). However, despite being subject to uncertainties, **study I** clearly reveal that the GHG emission potential of soils intermediate between mineral and organic soils can be as high or partly higher as for drained peatlands under intensive agricultural use. Therefore, we suggest establishing a further category in the wetland supplement (IPCC, 2014a) which provides emission factors for different land-use types on former drained peatlands or associated organic soils, containing <12% C_{org}, in order to avoid a significant underestimation of GHG emissions in the LULUCF/AFOLU sector.

3.2 GHG POLICY

To mitigate GHG emissions, the European Commission set out a long-term strategy to increase the proportion of renewable energy (e.g. biomass, wind power, photovoltaic) from 9% in 2010 to 21% of total energy consumption in 2020 (EU, 2009; Schleupner and Schneider, 2010; Don et al., 2011). As one measure, heat and power from biogas plants can substitute for fossil fuels and thus is considered to mitigate GHG emissions (Weiland, 2010; Don et al., 2011). However, bioenergy is not necessarily C neutral, since GHG release during crop production may reduce or completely counterbalance CO_2 savings of the substituted fossil fuels (Don et al., 2011). For illustration, assuming an average CH_4 yield of 310 m^3 CH_4 t VDS^{-1} (VDS = Volatile dry solids, VDS is equivalent to 90% DM content; Gerin, 2008; Steffen, 1998 cited in Smyth et al., 2009) and an average CH_4 content of 55% in the biogas (Smyth et al., 2009), the gross energy production of the grasslands investigated (**study I** and **II**) is calculated as 70–77 GJ ha^{-1}, which correspond to 19–21 MWh primary energy per hectare for the year 2011. This estimated value is in good agreement with Rösch et al., (2009) who reported that approximately 19 MWh primary energy per hectare could be substituted through wet

fermentation of grass silage. Thus considering an emission factor of 0.315 kg CO_2 kwh^{-1} for a fossil fuel mix of natural gas, hard coal and brown coal (Icha, 2013), approximately 6.1–6.7 t CO_2 ha^{-1} yr^{-1} of fossil fuels can be substituted through the production of biogas from the grasslands investigated. In contrast, approximately 70 t $CO_{2\ eq.}$ ha^{-1} yr^{-1} are released during grass silage production at these sites, clearly demonstrating the partially contradictory development in the implementation of GHG mitigation strategies.

Beside of grassland used for biogas production, **study I** revealed that all land-use and management forms investigated, show a very high release of GHGs, demonstrating the unsustainable agricultural use of drained organic soils and the current need for the implementation of GHG mitigation strategies and peatland restoration measures. Nevertheless, the enhanced bioenergy demand increases the value of land and thus, the marginal costs for peatland restoration measures (Schleupner and Schneider, 2010). As an alternative to abandoned rewetted peatlands, new management strategies, collectively referred to as paludiculture, are currently being tested to verify whether they can satisfy climate, ecologically as well as economic benefits.

In order to achieve a reduction in peat mineralization and to mitigate greenhouse gases, the enhancement of the water table height in combination with reduced land-use intensity are generally considered as the key components in peatland restoration measures (Höper et al., 2008). In line with this, the **third study** demonstrated the effect of anaerobicity on decomposition and turnover processes. At the undrained site NNM rates observed were significantly lower (84 kg N ha^{-1} yr^{-1}) compared to the drained sites (519–653 kg N ha^{-1} yr^{-1}). However, despite of distinctly different C and N contents, significantly different NNM rates between the two drained sites were not found, corroborating the findings from **study I** and **II**.

Generally, NO_3^- availability is considered as the main driver of N_2O emissions from drained histosols (Klemedtsson et al., 2005), but despite of significant differences in NNM rates and NO_3^- contents in the soil of the three sites, no differences were found regarding field N_2O fluxes. Generally, annual N_2O emissions observed in the **third study** were very low and ranged from 0.51 ± 0.07 kg N ha^{-1} yr^{-1} at the undrained site to 0.97 ± 0.13 kg N ha^{-1} yr^{-1} at the drained sites. Different studies reported considerably higher values for N_2O emissions from drained black alder stands, varying from 5 to 75 kg N ha^{-1} yr^{-1} (Mogge et al., 1998; Augustin, 1998b; Brumme et al., 1999; Merbach et al., 2001; Hefting et al., 2003; von Arnold et al., 2005). This contrasts our hypothesis that in drained black alder forests high N_2O emissions occur due to accelerated N turnover and

transformation processes. However, in contrast to the field observations significant differences in the N_2O fluxes between the two investigated sites were found in the incubation experiment at both soil moisture contents. Thus, the question arises which key factors are responsible for N_2O production in the alder forest investigated. As expected, at the undrained site anoxic conditions prevent net nitrification for most of the time in the year, as was confirmed by the finding that ammonification entirely controls NNM and observed N_{min} contents were solely dominated by NH_4^+ at this site. In line with this observation, N_2O and N_2 flux pattern observed in the incubation experiment demonstrated that denitrification was limited by NO_3^- availability, confirming our hypothesis that NO_3^- limitation in permanently waterlogged peat soils results in negligible N_2O and N_2 losses.

In contrast, the very high NNM rates at the drained sites revealed that NH_4^+ produced becomes immediately oxidized to NO_3^-. Nevertheless, no relationship was found between field N_2O fluxes and NO_3^- availability, and less than 0.2% of nitrified N was emitted as N_2O–N at these sites. However, the **third study** gave evidence that field N_2O fluxes obviously are triggered by temperature and the GW level. This relationship becomes even more apparent regarding the results from the incubation experiment. At both soil moisture contents, increasing temperature resulted in significantly enhanced N_2O fluxes. Observed exchange rates at 100% WFPS increase from 37.3 ± 4.5 µg N_2O-N $m^{-2} h^{-1}$ at 0℃ to 115.2 ± 22.6 µg N_2O-N $m^{-2} h^{-1}$ at 15℃, followed by a decline to 80.0 ± 29.4 µg N_2O-N $m^{-2} h^{-1}$ at 25℃, whereas at 70% WFPS, N_2O fluxes increase from 19.1 ± 3.1 µg N $m^{-2} h^{-1}$ at 0℃ to 132.4 ± 31.7 µg N $m^{-2} h^{-1}$ at 25 ℃. Furthermore, considering soil core samples from the drained site separately, it was found that N_2O fluxes at the first three temperature levels at 70% WFPS were lower compared to samples at 100% WFPS, but differences were not significant. However, overall N_2O fluxes were comparable between 100% and 70/83% WFPS, but total denitrification significantly increased, which is in line with studies by Scholefield et al. (1997). The low proportion of N_2O emissions of total N release from denitrification indicated that in the present alder stand losses of N_2O are only of minor importance compared to N_2 losses at water saturated conditions. Moreover, the observed decline of N_2O fluxes at water-saturated conditions and 25℃ with simultaneously exponential ly increasing or at least constant N_2 fluxes supports our hypothesis that N_2O release is displaced by N_2 losses during periods of temporarily high water levels. However, the field observations never reached the point in which the decreasing $N_2O:N_2$ ratio overcompensated the increasing denitrification

rate. The up to four times higher N_2O emissions measured in the incubation experiment at 70% WFPS compared to the field fluxes revealed the potential of high N_2O emissions under changing soil physical conditions. This assumption was confirmed by continued N_2O gas flux measurements for the time period 2012–2013 at these sites (unpublished data), where annual N_2O emission of up to 2.65 ± 0.23 kg N ha^{-1} yr^{-1} and 2.62 ± 0.85 kg N ha^{-1} yr^{-1} were observed at the undrained and drained sites, respectively. The equal N_2O emissions, despite of distinctly different C and N contents, observed at the drained alder sites contrasts the findings of **study I** and **II**. Enhanced N_2O emissions at the C_{org} high sites in study **I** and **II** are related to more favorable conditions for denitrification. However, high net nitrification rates in combination with low N_2O and N_2 fluxes indicate that, for most of the year, aerobic or only slightly anaerobic soil conditions are predominant, preventing denitrification. Determined bulk densities and higher porosity additionally supports the assumption of a better aeration and thus a faster O_2 diffusion into deeper soil layers of the alder forest compared to the intensively used grassland and cropland.

Besides N_2O, particularly CO_2 emissions dominate the GHG balances from drained organic soils as shown in **study I** and additionally confirmed by several other studies (e.g. Grønlund et al., 2006; Maljanen et al., 2010; Leiber-Sauheitl et al., 2014). In contrast, in pristine or restored peatlands with high groundwater tables, mainly CH_4 emissions are responsible for the climate impact. Unpublished data of determined CH_4 fluxes from the alder forest investigated, revealed annual CH_4 exchange rates of 8.93 ± 1.58 (U), -0.27 ± 0.01 (D–1) and -0.20 ± 0.01 g CH_4-C m^{-2} yr^{-1} (D–2), for the year 2011. So far there is no report in the literature of an entirely measured greenhouse gas budget from an alder forest on a fen peatland. Particularly the estimation of net CO_2 fluxes displays a high uncertainty due to a lack of data. All published gas fluxes from previous studies based on the dark chamber measurement technique, which does not allow a conclusion about the net CO_2 exchange of a forest ecosystem. Some studies estimate the GWP using a combination of compiled biomass-data, measured flux data and assumptions based on literature values (Augustin, 2003; von Arnold et al., 2005; Mander et al., 2008). The reported net GHG fluxes range between -736.3 and 1113 g CO_2-$C_{equ.}$ m^{-2} yr^{-1} for drained alder stands and between -469.6 and 487 g CO_2-$C_{equ.}$ m^{-2} yr^{-1} for undrained alder forests (Augustin, 2003; von Arnold et al., 2005; Mander et al., 2008). In order to assess the climate impact of the alder forest investigated, a rough estimation of net GHG fluxes was carried out and described in detail in Appendix 7.4. Calculated

GWP_{100} values indicated that both sites slightly contributed to global warming with GWP_{100} values of about 72–76 g CO_2-$C_{equ.}$ m^{-2} yr^{-1} for the undrained site and about 98–127 g CO_2-$C_{equ.}$ m^{-2} yr^{-1} for the drained sites. The estimated GWP_{100} values are in the order of magnitude reported for semi-natural or restored fen peatlands in Germany (Drösler et al., 2012). These finding support the assumption that black alder forests can serve as one mitigation strategy to reduce GHG's from drained fen peatlands, providing groundwater level enhancement and perhaps a partial forest-use restriction. However, further investigations particularly for NEE are required to improve the accuracy of greenhouse gas budgets from alder forests on fen peatlands.

4 CONCLUSION

The present PhD thesis shed new lights on GHG emissions from drained organic soils not fulfilling the definition of organic soils according to the IPCC guidelines for the national climate reporting in the LULUCF/AFOLU sector. Two years of GHG measurements clearly revealed that drained mollic Gleysols can also be considered as hotspots for GHG emissions likewise SOC-rich sapric Histosols, provided that they are intensively managed as arable land or grassland. The results gave evidence that the GHG balance, of a drained and intensively cropped organic soil is mainly affected by management rather than SOC content or land-use type. The study indicated that N_2O emissions increase with increasing C_{org} content, whereas CH_4 exchanges were of only minor importance at the intensively drained and cropped organic soils. However, CO_2 emissions dominated total GHG balances of all treatments to nearly 100%. Estimated GWPs partly more than doubled the emission factor of the Tier 1 approach of the IPCC regardless of the SOC content in the topsoil. Thus, to avoid a significant underestimation of GHG emissions in the LULUCF/AFOLU sector, there is a corresponding need to extent the IPCC guidelines for drained inland organic soils in respect to former peatlands and associated organic soils containing <12% C_{org}. However, further investigations are needed to define the lower boundary for C_{org} limits.

Furthermore, the effects of biogas digestate application on short-term N_2O and CH_4 emissions as well as on biomass production and NUE from grasslands on drained organic soils were discussed for the first time. It was found that N_2O fluxes were significantly higher after digestate than after cattle slurry fertilization, which probably could be attributed to a higher N availability in the soil caused by increased SOM mineralization or other sources of labile carbon triggered by digestate application. Moreover, the study demonstrated that the higher grass yield could not compensate for the significantly higher N_2O emissions at the biogas digestate treatments. Generally, the apparent NUE_{min} indicated that a frequent but low dosage application of fertilizer and quick N uptake by plants avoid conditions favorable for high N_2O emissions. However, the observed linear increase in cumulative N_2O emissions with increasing NH_4^+ fertilization and increasing groundwater table reveals the importance of site adapted N fertilization and the avoidance of N surpluses during agricultural use of C_{org} rich grasslands.

In contrast to the intensively cropped drained organic soils, different SOC and N contents did not affect N_2O emissions in a drained black alder forest. Regardless of waterlogging or drained conditions, it could be shown that despite considerably increased NNM and different N turnover processes N_2O production remained low at least in the year 2011. In general, a permanently high groundwater level was found to prevent net nitrification resulting in negligible gaseous N losses. In contrast, drainage shifted N transformation processes from net ammonification to net nitrification. The conducted helium-incubation experiment showed that temporarily water-saturated conditions mainly increased N_2 fluxes. Nevertheless, the laboratory study gave evidence that changing soil physical conditions can result in enhanced N_2O emissions at the drained alder sites. Continuous N_2O measurements in two subsequent years confirmed the findings from the incubation experiment, but it has to be pointed out that N_2O emissions also markedly increased at the waterlogged site. A rough estimated GHG balance revealed that black alder forests, which represent the natural vegetation of fen peatlands, can serve as one mitigation strategy to reduce GHGs from drained fen peatlands, providing groundwater level enhancement and perhaps a partial forest-use restriction.

In conclusion, the observed very high release of GHGs demonstrated the unsustainable agricultural use of drained organic soils and the current need for rapid implementation of mitigation strategies and restoration measures.

5 REFERENCES

Ad-Hoc AG Boden: Bodenkundliche Kartieranleitung, 5. Aufl., Schweizerbart'sche Verlagsbuchhandlung, Hannover, 438 pp., 2005.

Aigner, A., Biertümpel, A., Conrad, M., Deiglmayr, K., Diepolder, M., Eder, B., Eder, J., Formowitz, B., Fritz, M., Gehring, K., Hartmann, A., Hartmann, S., Heiermann, M., Herrmann, C., Hofmann, D, Idler, C., Kawasch, M., Kornatz, P., Lichti, F., Nehring, A., Salzeder, G., Schaffner, S., Sticksel, E., Stockmann, F., Vetter, A., Vollrath, B., von Buttlar, C., Wendland, M., Willms, M., Wolf, G., Zeise, K.: Energiepflanzen für Biogasanlagen, Fachagentur Nachwachsende Rohstoffe e.V. (FNR), Bestell-Nr. 558, 1. Auflage, 116 pp., 2012.

Alm, J., Talanov, A., Saarnio, S., Silvola, J., Ikkonen, E., Aaltonen, H., Nykanen, H., and Martikainen, P.J.: Reconstruction of the carbon balance for microsites in a boreal oligotrophic pine fen, Finland, Oecologia, 110, 423–431, 1997.

Ambus, P., and Zechmeister-Boltenstern, S.: Denitrification and N-Cycling in Forest Ecosystems, In: Biology of the Nitrogen Cycle, Bothe, H., Ferguson, S.J., and Newton, W.E., (eds.), Elsevier B.V., 2007.

Arneth, A., Harrison, S.P., Zaehle, S., Tsigaridis, K., Menon, S., Bartlein, P.J., Feichter, J., Korhola, A., Kulmala, M., O'Donnell, D., Schurgers, G., Sorvari, S., and Vesala, T.: Terrestrial biogeochemical feedbacks in the climate system, Nature Geoscience, 3, 525–532, 2010.

Arnold von, K., Nilsson, M., Hånell, B., Weslien, P., and Klemedtsson, L.: Fluxes of CO_2, CH_4 and N_2O from drained organic soils in deciduous forests, Soil Biology and Biochemistry, 37, 1059–1071, 2005.

Artz, R.R.E.: Microbial community structure and carbon substrate use in northern peatlands, In: Carbon Cycling in Northern Peatlands, Baird, A.,J., Belyea, L.R., Comas, X., Reeve, A.S., and Slater, L.D. (eds.), Geophysical Monograph, 184, American Geophysical Union, Washington, DC, 2009.

Augustin, J., Merbach, W., Steffens, L., and Snelinski, B.: Nitrous Oxide Fluxes of Disturbed Minerotrophic Peatlands, Agribiol. Res., 51 (1), 47–57, 1998a.

Augustin, J., Merbach, W., and Rogasik, J.: Factors influencing nitrous oxide and methane emissions from minerotrophic fens in northeast Germany, Biol. Fertil. Soils, 28, 1–4, 1998b.

Augustin, J.: Erlenstandorte als Quelle und Senke klimarelevater Spurengase. (Alder forests as a source and sink of Greenhouse gases), In: Die Schwarz-Erle (Alnus glutinosa [L.] GAERTN.) im nordostdeutschen Tiefland, Eberswalder Forstliche Schriftenreihe, 17, 46–59, 2003.

Baird, A.J., Comas, X., Slater, L.D., Belyea, L.R., and Reeve, A.S.: Understanding Carbon Cycling in Northern Peatlands: Recent Developments and Future Prospects, In: Carbon Cycling in Northern Peatlands, Baird, A.,J., Belyea, L.R., Comas, X., Reeve, A.S., and Slater, L.D. (eds.), Geophysical Monograph, 184, American Geophysical Union, Washington, DC, 2009.

Bakken, L.R.: Denitrification under different cultivated plants: effects of soil moisture tension, nitrate concentration, and photosynthetic activity, Biol. Fertil. Soils, 6, 271–278, 1988.

Baldock, J.A.: Composition and Cycling of Organic Carbon in Soil, in Nutrient Cycling in Terrestrial Ecosystems, Eds. Marschner, R., Rengel, Z., Soil Biology, Volume 10, 397 pp., 2007.

Batjes, N.H.: Total carbon and nitrogen in the soils of the world, European Journal of Soil Sciences, 47, 151–163, 1996.

Beetz, S., Liebersbach, H., Glatzel, S., Jurasinski, G., Buczko, U., and Höper, H.: Effects of land-use intensity on the full greenhouse gas balance in an Atlantic peat bog, Biogeosciences, 10, 1067–1082, doi:10.5194/bg-10-1067-2013, 2013.

Berndes, G. Hoogwilk, M., and van den Broek, R.: The contribution of biomass in the future global energy supply: a review of 17 studies. Biomass Bioenergy 25, 1–28, 2003.

Blackmer, A. M. and Bremner, J. M.: Inhibitory effect of nitrate on reduction of N_2O to N_2 by soil microorganisms, Soil Biol. Biochem., 10, 187–191, 1978.

Blodau, C.: Carbon cycling in peatlands – A review of processes and controls, Environ. Rev., 10, 111–134, 2002.

Bobbink, R., and Roelofs, J.G.M.: Nitrogen critical loads for natural and seminatural ecosystems: the empirical approach. Water Air Soil Pollut. 85, 2413–2418, 1995.

Bobbink, R., Hornung, M., and Roelofs, J.G.: The effects of airborne nitrogen pollutants on species diversity in natural and semi-natural European vegetation, J. Ecol., 86, 717–738, 1998.

Bobbink, R., Hicks, K., Galloway, J., Spranger, T., Alkemade, R., Ashmore, M., Bustamante, M., Cinderby, S., Davidson, E., Dentener, F., Emmett, B.A., Erisman, J.W., Fenn, M., Gilliam, F.S., Nordin, A., Pardo, L., and de Vries, W.: Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. Ecol. Appl., 20, 30–59, 2010.

Bowman, J.: The greenhouse effect, Land Use Policy, 7 (2), 101–108, 1990.

Braker, G., and Conrad, R.: Diversity, structure, and size of N_2O-producing microbial communities in soils - what matters for their functioning?, Adv. Appl. Microbiol., 75, 33–70, 2011.

Bragazza, L., Buttler, A., Siegenthaler, A., and Mitchell, E.A.D.: Plant litter decomposition and nutrient release in peatlands, In: Carbon Cycling in Northern Peatlands, Baird, A.,J., Belyea, L.R., Comas, X., Reeve, A.S., and Slater, L.D. (eds.), Geophysical Monograph, 184, American Geophysical Union, Washington, DC, 2009.

Brumme, R., Borken, W., and Finke, S.: Hierarchical control on nitrous oxide emission in forest ecosystems, Global Biogeochem. Cy., 13 (4), 1137–1148, 1999.

Bubier, J., Moore, T., and Bledzki, L.: Effects of nutrient addition on vegetation and carbon cycling in an ombrotrophic bog, Glob. Change Biol., 13 (6), 1168–1186, 2007.

Butterbach-Bahl, K., Baggs, E.M., Dannenmann, M., Kiese, R., Zechmeister-Boltenstern, S.: Nitrous oxide emissions from soils: how well do we understand the processes and their controls?, Phil trans R Soc B, 368:20130122, 2013.

Brugiere, N., Suzuki, A., and Hirel, B.: Ammonium assimilation, In: Nitrogen assimilation by plants; physiological, biochemical and molecular aspects, Morot-Gaudry J-F (ed.), Science Publishers, Enfield, NH, 71–94, 1997.

Byrne, K. A., Chojnicki, B., Christensen, T. R., Drösler, M., Freibauer, A., Friborg, T., Frolking, S., Lindroth, A., Mailhammer, J., Malmer, N., Selin, P., Turunen, J., Valentini, R., and Zetterberg, L.: EU peatlands: Current carbon stocks and trace gas fluxes, Carbo-Europe-GHG Concerted Action-Synthesis of the European Greenhouse Gas Budget, Report, 4, 2004.

Byrne, K.A., and Farrell, E.P.: The effect of afforestation on soil carbon dioxide emissions in blanket peatland in Ireland, Forestry, 78, 217–227, doi:10.1093/forestry/cpi020, 2005.

Chapin, F.S.: New cog in the nitrogen cycle, Nature, 377, 199–200, 1995.

Charman, D. J., Beilman, D. W., Blaauw, M., Booth, R. K., Brewer, S., Chambers, F. M., Christen, J. A., Gallego-Sala, A., Harrison, S. P., Hughes, P. D. M., Jackson, S. T., Korhola, A., Mauquoy, D., Mitchell, F. J. G., Prentice, I. C., van der Linden, M., De Vleeschouwer, F., Yu, Z. C., Alm, J., Bauer, I. E., Corish, Y. M. C., Garneau, M., Hohl, V., Huang, Y., Karofeld, E., Le Roux, G., Loisel, J., Moschen, R., Nichols, J. E., Nieminen, T. M., MacDonald, G. M., Phadtare, N. R., Rausch, N., Sillasoo, Ü., Swindles, G. T., Tuittila, E.-S., Ukonmaanaho, L., Väliranta, M., van Bellen, S., van Geel, B., Vitt, D. H., and Zhao, Y.: Climate-related changes in peatland carbon accumulation during the last millennium, Biogeosciences, 10, 929-944, doi:10.5194/bg-10-929-2013, 2013.

Christensen, T.R., Johansson, T.R., Akerman, H.J., Mastepanov, M., Malmer, N., Friborg, T., Crill, P., and Svensson, B.H.: Thawing sub-arctic permafrost: Effects on vegetation and methane emissions, Geophys. Res. Lett., 31, L04501, doi:10.1029/2003GL018680, 2004.

Cicerone, R.J. and Oremland, R.S.: Biogeochemical aspects of atmospheric methane, Global Biogeochemical Cycles, 2, 288–327, 1988.

Cleemput van, O., Boeckx, P., Lindgren, P.-E., and Tonderski, K.: Denitrification in Wetlands, In: Biology of the Nitrogen Cycle, Bothe, H., Ferguson, S.J., and Newton, W.E., (eds.), Elsevier B.V., 2007.

Clemens, J. and Huschka, A.: The effect of biological oxygen demand of cattle slurry and soil moisture on nitrous oxide emissions, Nutr. Cycling Agroecosyst. 59, 193–198, 2001.

Colliver, B.B. and Stephenson, T.: Production of nitrogen oxide and dinitrogen oxide by autotrophic nitrifiers, Biotechnology Advances, 18, 219–232, 2000.

Couwenberg, J.: Greenhouse gas emissions from managed peat soils: is the IPCC reporting guidance realistic, Mires and Peat, 8, 1–10, 2011.

Crill, P.M., Martikainen, P.J., Nykänen, H., and Silvola, J.: Temperature and fertilization effects on methane oxidation in a drained peatland soil, Soil Biol. Biochem., 26(10), 1331–1339, 1994.

Davidson, E.A., Swank, W.T., and Perry, T.O.: Distinguishing between nitrification and denitrification as sources of gaseous nitrogen production in soil, Applied and Environmental Microbiology, 52 (6), 1280–1286, 1986.

Davidson, E. A.: Fluxes of nitrous and nitric oxide from terrestrial ecosystems. In: Microbial Production and Consumption of Greenhouse Gases: Methane, Nitrogen Oxides and Halomethanes, edited by: Rogers, J. E. and Whitman, W. P., American Society for Microbiology, Washington, D.C., 219–235, 1991.

Davidson, E.A., and Janssens, I.A.: Temperature sensitivity of soil carbon decomposition and feedbacks to climate change, Nature, 440, 165–173, 2006.

Denmead, O.T.: Approaches to measuring fluxes of methane and nitrous oxide between landscapes and the atmosphere, Plant Soil, 309, 5–24, 2008.

Dittert, K.: Die stickstofffixierende Schwarzerle- Frankia-Symbiose in einem Erlenbruch der Bornhöveder Seenkette, EcoSys. Suppl., 5, 1–98, 1992.

Don, A., Osborne, B., Hastings, A., Skiba, U., Carter, M.S., Drewer, J., Flessa, H., Freibauer, A., Hyvönen, N., Jones, M.B., Lanigan, G.J., Mander, Ü., Monti, A., Djomo, S.N., Valentine, J., Walter, K., Zegada-Lizarazu, W., and Zenone, T.: Land-use change to bioenergy production in Europe: implications for the greenhouse gas balance and soil carbon, GCB Bioenergy, doi: 10.1111/j.1757-1707.2011.01116.x, 2011.

Drösler, M., Freibauer, A., Christensen, T. and Friborg, T.: Observation and status of peatland greenhouse gas emission in Europe, In: Dolman, H., Valentini, R. &

Freibauer, A. (eds) The Continental-Scale Greenhouse Gas Balance of Europe. Ecological Studies, 203, 237–255, 2008a.

Drösler, M., Fiedler, S., Zeitz, J., Höper, H., Glatzel, S., Sommer, M., Augustin, J., Pretzsch, H., Rötzer, T., and Edom, F.: Verbundvorhaben: Klimaberichterstattung „organische Böden" – Ermittlung und Bereitstellung von Methoden, Aktivitätsdaten und Emissionsfaktoren für die Klimaberichterstattung LU-LUCS/AFOLU. Projektskizze zur Begleitforschung, 2008b.

Drösler, M., Freibauer, A., Adelmann, W., Augustin, J., Bergman, L., Beyer, C., Chojnicki, B., Förster, C., Giebels, M., Görlitz, S., Höper, H., Kantelhardt, J., Liebersbach, H., Hahn-Schöfl, M., Minke, M., Petschow, U., Pfadenhauer, J., Schaller, L., Schägner,P., Sommer, M., Thuille, A., and Wehrhan, M.: Klimaschutz durch Moorschutz in der Praxis, Arbeitsbericht aus dem vTI-Institut für Agrarrelevante Klimaforschung, p. 21, available at http://www.vti.bund.de/de/startseite/institute/ak/ publikationen.html, 2011.

Drösler, M., Augustin, J., Bergmann, L., Förster, C., Fuchs, D., Hermann, J.M., Kantelhardt, J., Kapfer, A., Krüger, G., Schaller, L., Sommer, M., Schweiger, M., Steffenhagen, P., Tiemeyer, B., and Wehrhan, M.: Beitrag ausgewählter Schutzgebiete zum Klimaschutz und dessen monetäre Bewertung. Abschlussbericht des gleichnamigen F+E-Vorhabens (FKZ 3509850500), BfN-Skripten, 328, 152 pp., 2012.

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

Elsgaard, L., Gorres, C.-M., Hoffmann, C. C., Blicher-Mathiesen, G., Schelde, K., and Petersen, S. O.: Net ecosystem exchange of CO_2 and carbon balance for eight temperate organic soils under agricultural management, Agr. Ecosyst. Environ., 162, 52–67, 2012.

Eswaran, H., van den Berg, E., Reich, P., and Kimbel, J.: Global soil carbon resources. In: Soils and Global Change, Lewis Publishers, Boca Raton, Fl (1995), 27–43, 1995.

EU: Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC, O. J. o. t. E. Union (ed Union OjotE), EU, Brussels, 2009.

Fangmeier, A., Hadwiger-Fangmeier, A., van der Eerden, L., and Jäger, H.J.: Effects of atmospheric ammonia on vegetation – a review, Environmental pollution, 86, 43–82, 1994.

FAO, Food and Agriculture Organization of the United Nations: World Reference Base for Soil Resources, World Soil Resources Reports, 84, 88 pp., 1998.

FAOSTAT, 2013. Statistics Division of the Food and Agriculture Organization of the United Nations. Rome, Italy: FAOSTAT. http://faostat.fao.org. Viewed Feb 2015.

Fell, H., Roßkopf, N., and Zeitz, J.: Impact oft he spatial resolution of soils data on climate reporting for organic soils using the example of Germany, Mires and Peat, 14, 1–9, 2014.

Ferguson, S.J., Richardson, D.J., and Spanning. R.J.M.: Biochemistry and Molecular Biology of Nitrification, In: Biology of the Nitrogen Cycle, Bothe, H., Ferguson, S.J., and Newton, W.E., (eds.), Elsevier B.V., 2007.

Firestone, M. K., Smith, M. S., Firestone, R. B., and Tiedje, J. M.: The influence of nitrate, nitrite, and oxygen on the composition of the gaseous products of denitrification in soil, Soil Sci. Soc. Am. J., 43, 1140–1144, 1979.

Flessa, H., Wild, U., Klemisch, M., and Pfadenhauer, J.: C- und N-Stoffflüsse auf Torfstichsimulationsflächen im Donaumoos, Z. f. Kulturtechnik und Landentwicklung, 38, 11–17, 1997.

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

Flessa, H., and Beese, F.: Laboratory Estimates of Trace Gas Emissions following Surface Application and Injection of Cattle Slurry, J. Environ. Qual., 29, 262–268, 2000.

Forster, P., V. Ramaswamy, P. Artaxo, T. Berntsen, R. Betts, D.W. Fahey, J. Haywood, J. Lean, D.C. Lowe, G. Myhre, J. Nganga, R. Prinn, G. Raga, M. Schulz and R. Van Dorland, 2007: Changes in Atmospheric Constituents and in Radiative Forcing. In: Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M.Tignor and H.L. Miller (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 2007.

Frame, C.H., and Casciotti, K.L.: Biogeochemical controls and isotopic signatures of nitrous oxide production by a marine ammonia-oxidizing bacterium, Biogeosciences, 7, 2695–2709, 2010.

Freibauer, A., Rounsevell, M.D.A., Smith, P., and Verhagen, J.: Carbon sequestration in the agricultural soils of Europe, Geoderma, 122, 1–23, 2004.

Frolking S., Bubier, J.L., Moore, T.R., Ball, T., Bellisario, L.M., Bharwaj, A., Carroll, P., Crill, P.M., Lafleur, P.M., McCaughey, J.H., Roulet, N.T., Suyker, A.E., Verma, S.B., Waddington, J.M., and Whiting, G.M.: Relationship netween ecosystem productivity and photosynthetically active radiation for northern peatlands, Global Biogeochemical Cycles, 12 (1), 115–126, 1998.

Frolking, S. and Roulet, N. T.: Holocene radiative forcing impact of northern peatland carbon accumulation and methane emissions, Glob. Change Biol., 13, 1079–1088, 2007.

Frolking, S., Talbot, J., Jones, M.C., Treat, C.C., Kauffman, B., Tuittila, E.S., and Roulet, N.: Peatlands in the Earth's 21st century climate system, Environ. Rev., 19, 371–396, 2011.

Galloway, J.N.: Acid deposition: Perspectives in time and space, Water, air and Soil Pollution, 85, 15–24, 1995.

Galloway, J.N.: The global nitrogen cycle: changes and consequences, Environmental Pollution, 102, 15–24, 1998.

Galloway, J.N.: Acidification of the world: natural and anthropogenic, Journal of Water, Air, and Soil Pollution, 130, 17–24, 2001.

Galloway, J.N., Cowling, E.B., Seitzinger, S.P., and Socolow, R.H.: Reactive Nitrogen: Too Much of a Good Thing?, Ambio, 31 (2), 60–63, 2002.

Galloway, J.N., Dentener, F.J., Capone, D.G., Boyer, E.W., Howarth, R.W., Seitzinger, S.P., Asner, G.P., Cleveland, C.C., Green, P.A., Holland, E.A., Karl, D.M., Michaels, A.F., Porter, J.H., Townsend, A.R., and Vorosmarty, C.J.: Nitrogen cycles: past, present, and future, Biogeochemistry, 70, 153–226, 2004.

Galloway, J.N., Townsend, A.R., Erisman, J.W., Bekunda, M., Cai, Z.C., Freney, J.R., Martinelli, L.A., Seitzinger, S.P., and Sutton, M.A.: Transformation of the nitrogen

cycle: Recent trends, questions, and potential solutions, Science, 320(5878), 889–892, 2008.

Gao, Z., Desjardins, R.L., and Flesch, T.K.: Comparison of a simplified micrometeorological mass difference technique and an inverse dispersion technique for estimating methane emissions from small area sources, Agricultural and forest Meteorology, 149, 891–898, 2009.

Gerin PA, Vliegen F, and Jossart J-M.: Energy and CO_2 balance of maize and grass as energy crops for anaerobic digestion, Bioresource Technology, 99 (7), 2620–2627, 2008.

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

Gorham, E.: The biogeochemistry of northern peatlands and its possible responses to global warming, in: Biotic Feedbacks in The Global Climate System: Will the Warming Feed the Warming?, Woodwell, G.M., and Mackenzie, F.T. (ed.), Oxford Univ. Press, New York, 169–187, 1995.

Göttlich, K.: Moor- und Torfkunde, E. Schweizbart'sche Verlagsbuchhandlung (Nägele u. Obermiller) Stuttgart, 3. Auflage, 520 pp., 1990.

Grønlund, A., Sveistrup, T.E., Søvik, A.K., Rasse, D.P., and Kløve, B.: Degradation of cultivated peat soils in northern Norway based on field scale CO_2, N_2O and CH_4 emission measurements, Arch Agron. Soil Sci., 52, 149–159, 2006.

Grønlund, A., Hauge, A., Hovde, A., and Rasse, D.A.: Carbon loss estimates from cultivated peat soils in Norway: a comparison of three methods, Nutr. Cycl. Agroecosyst., 81, 157–167, 2008.

Grosse, W., Frye, J., and Lattermann, S.: Root aeration in wetland trees by pressurized gas transport, Tree Physiology, 10, 285–295, 1992.

Grote, R., and Pretzsch, H.: A Model for Individual Tree Development Based on Physiological Processes, Plant Biology, 4 (2), 167–180, 2002.

Gruber, N., and Galloway, J.N.: An Earth-system perspective of the global nitrogen cycle, Nature, 451 (7176), 293–296, 2008.

Gupta, V., Smemo, K.A., Yavitt, J.B., Fowle, D., Branfireun, B., and Basiliko, N.: Stable isopes reveal widespread anaerobic methane oxidation across latitude and peatland type, Environ. Sci. technol., 47, 8273–8279, 2013.

Gutser, R., Ebertseder, Th., Weber, A., Schraml, M., and Schmidhalter, U.: Short-term and residual availability of nitrogen after long-term application of organic fertilizers on arable land, J. Plant Nutr. Soil Sci., 168, 439–446, 2005.

Gutser, R., Ebertseder, T., Schraml, M., von Tucher, S., and Schmidhalter, U.: Stickstoffeffiziente und umweltschonende organische Düngung, in Emissionen landwirtschaftlich genutzter Böden, KTBL-Schrift, 483, 384 pp., 2010.

Haenel, H.D., Freibauer, A., Rösemann, C., Poddey, E., Gensior, A., Menden, B.M., and Döhler, H.: Emissionen landwirtschaftlich genutzter Böden im Rahmen der deutschen Klimaberichterstattung, in Emissionen landwirtschaftklich genutzter Böden, KTBL-Schrift, 483, 384 pp., 2010.

Hefting, M.M., Bobbink, R., and Caluwe, H.: Nitrous oxide emission and denitrification in chronically nitrate-loaded riparian buffer zones, J. Environ. Qual., 32, 1194–1203, 2003.

Heijmans, M.P.D., Berendse, F., Arp, W.J., Masselink, A.B.K., Klees, H., de Visser, W., and van Breemen, N.: Effects of elevated carbon dioxide and increased nitrogen deposition on bog vegetation in the Netherlands, J. Ecol., 89(2), 268–279, 2001.

Heller, C., and Zeitz, J.: Stability of soil organic matter in two northeastern German fen soils: the influence of site and soil development, J. Soils Sediments, 12, 1231–1240, DOI 10.1007/s11368-012-0500-6, 2012.

Henry, S., Texier, S., Hallet, S., Bru, D., Dambreville, C., Chèneby, D., Bizouard, F., Germon, J.C., and Philippot, L.: Disentangling the rhizosphere effect on nitrate reducers and denitrifiers: insight into the role of root exudates, Environ. Microbiol., 10, 3082–3092, 2008.

Hensen, A.: Methods for observation and quantification of trace gas emissions from diffuse sources, PhD thesis, VU Amsterdam, 144 pp., 2011.

Hooijer, A., Silvius, M., Woesten, H., and Page, S.: Peat-CO_2: Assessment of CO_2 emissions from drained peatlands in SE Asia, Delft Hydraulics report Q3943, 41 pp., 2006.

Hooijer, A., Page, S., Canadell, J.G., Silvius, M., Kwadijk, J., Wösten, H., and Jauhiainen, J.: Current and future CO_2 emissions from drained peatlands in Southeast Asia, Biogeosciences, 7, 1505–1514, 2010.

Höper, H., Augustin, J., Cagampan, J. P., Drösler, M., Lundin, L., Moors, E. J., Vasander, H., Waddington, J. M., and Wilson, D.: Restoration of peatlands and greenhouse gas balances, in: Peatlands and Climate Change. edited by: Strack, D. M., International Peat Society, Jyvaskyla, 182–210, 2008.

Houghton, J.T., and Woodwell, G.M.: Global climate change, Sci. Am., 260, 36–44, 1989.

Icha, P.: Entwicklung der spezifischen Kohlendioxid-Emissionen des deutschen Strommix in den Jahren 1990 bis 2012, Umweltbundesamt, Climate Change 07/2013, http://www.uba.de/uba-info-medien/4488.html, p. 18, 2013.

IPCC: Climate Change 1994: Radiative Forcing of Climate Change and An Evaluation oft he IPCC IS92, Emission Scenarios, J.T. Houghton, L.G. Meira Filho, J. Bruce, Hoesung Lee, B.A. Callander, E. Haites, N. Harris and K. Maskell (Eds.), Cambridge University Press, UK, 339 pp., 1994.

IPCC: 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Prepared by the National Greenhouse Gas Inventories Programme, Eggleston H.S., Buendia L., Miwa K., Ngara T. and Tanabe K. (eds). Published: IGES, Japan, 2006.

IPCC: Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, 2007, Solomon, S., D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor and H.L. Miller (eds.), Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA., 2007.

IPCC: Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 1535 pp., 2013.

IPCC: 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands, Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M. and Troxler, T.G. (eds). Published: IPCC, Switzerland, 2014a.

IPCC: Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Edenhofer, O., R. Pichs-Madruga, Y. Sokona, E. Farahani, S. Kadner, K. Seyboth, A. Adler, I. Baum, S. Brunner, P. Eickemeier, B. Kriemann, J. Savolainen, S. Schlömer, C. von Stechow, T. Zwickel and J.C. Minx (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, 2014b.

Jain, P.C.: Greenhouse effect and climate change: Scientific basis and overview, Renewable Energy, 3 (4/5), 403–420, 1993.

Jansson, S., and Persson, J.: Mineralization and immobilization of soil nitrogen, In: Nitrogen in agricultural soils, Stevenson, F.J. (ed.), ASA/CSSA/SSSA, Madison, 229–252, 1982.

Jenkinson, D.S., Adams, D.E., and Wild, A.: Model estimates of CO_2 emissions from soil in response to global warming, Nature, 351, 304–306, 1991.

Jenkinson, D.S.: The impact of humans on the nitrogen cycle, with focus on temperate arable agriculture, Plant Soil, 228, 3–15, 2001.

Jones, S.K., Rees, R.M., Skiba, U.M., and Ball, B.C.: Influence of organic and mineral N fertiliser on N_2O fluxes from temperate grassland, Agriculture, Ecosystems and Environment, 121, 74–83, 2007.

Joosten, H.: The IMCG Global Peatland Database. www.imcg.net/gpd/gpd.htm, 2004.

Joosten, H.: The global peatland CO_2 picture. Peatland status and drainage related emissions in all countries of the world, report, Wetlands Int., Ede, Netherlands, 36 pp., 2010.

Kandel, T.P., Elsgaard, L., Karki, S., and Læerke, P.E.: Biomass yield and greenhouse gas emissions from a drained fen peatland cultivated with reed canary grass under different harvest and fertilizer regimes, Bioenerg. Res., 6, 883–895, DOI 10.1007/s12155-013-9316-5, 2013.

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

Klemedtsson, L., Svensson, BO.H., and Rosswall, T.: Dinitrogen and nitrous oxide produced by denitrification and nitrification in soil with and without barley plants, Plant and Soil, 99, 303–319, 1987.

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

Kiehl, J.T., and Trenberth, K.E.: Earth's annual global mean energy budget, Bulletin of the American Meteorological Society, 78 (2), 197–208, 1997.

Knowles, R.: Denitrification, Microbiol. Rev., 46, 43–70, 1982.

Koopmans, C.J., Tietema, A., and Verstraten, J.M.: Effects of reduced N deposition on litter decomposition and N cycling in two N saturated forests in the Netherlands, Soil Biol. Biochem., 30 (2), 141–151, 1998.

Lafleur, P.M., Roulet, N. T., Bubier, J.L., Frolking, S., and Moore, T.R.: Interannual variability in the peatland-atmosphere carbon dioxide exchange at an ombrotrophic bog, Global Biogeochemical Cycles, 17 (2), doi:10.1029/2002GB001983, 2003.

Lal, R.: Soil carbon sequestration to mitigate climate change, Geoderma, 123 (1–2), 1–22, 2004.

Laine, A., Wilson, D., Kiely, G., and Byrne, K.A.: Methane flux dynamics in an Irish lowland blanket bog, Plant Soil, 299, 181–193, 2007.

Lappalainen, E.: General review on world peatland and peat resources. In Lappalainen, E., editor, Global peat resources, Helsinki: International Peat Society and Geological Survey of Finland, 53–56, 1996.

Larcher, W.: Ökophysiologie der Pflanzen: Leben, Leistung und Streßbewältigung der Pflanzen in ihrer Umwelt, Verlag Eugen Ulmer GmbH & Co., 6. Neubearb. Aufl., 408 pp., 2001.

Leiber-Sauheitl, K., Fuß, R., Voigt, C., and Freibauer, A.: High CO_2 fluxes from grassland on histic Gleysol along soil carbon and drainage gradients, Biogeosciences, 11, 749–761, doi:10.5194/bg-11-749-2014, 2014.

Leifeld, J., Müller, M., and Fuhrer, J.: Peatland subsidence and carbon loss from drained temperate fens, Soil Use Manage., 27, 170–176, 2011.

Leifeld, J., Steffens, M., and Galego-Sala, A.: Sensitivity of peatland carbon loss to organic matter quality, Geophys. Res. Lett., 39, L14704, doi:10.1029/2012GL051856, 2012.

Leifeld, J., Bader, C., Borraz, E., Hoffmann, M., Giebels, M., Sommer, M., and Augustin, J.: Are C-loss rates from drained peatlands constant over time? The additive value of soil profile based and flux budget approach, Biogeosciences Discuss., 11, 12341–12373, 2014.

Leppelt, T. Dechow, R., Gebbert, S., Freibauer, A., Lohila, A., Augustin, J., Drösler, M., Fiedler, S., Glatzel, S., Höper, H., Järveoja, J., Lærke, P.E., Maljanen, M., Mander, Ü., Mäkiranta, P., Minkkinen, K., Ojanen, P., Regina, K., and Strömgren, M.: Nitrous oxide emission hotspots from organic soils in Europe, Biogeosciences Discuss., 11, 9135–9182, 2014.

Lipson, D., and Monson, R.: Plant-microbe competition for amino acids in the alpine tundra: effect of freeze-thaw and dry-wet events, Oecologia, 113, 406–414, 1998.

Limpens, J., Heijmans, M.M.P.D., Berendse, F.: Nitrogen in boreal peatlands, In: Boreal Peatland Ecosystems, Wieder, R.K., Vitt, D.H. (eds.), Ecological Studies Series, 188, Springer-Verlag, Berlin, 2006.

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

Maljanen, M., Hytönen, J., and Martikainen, P. J.: Fluxes of N_2O, CH_4 and CO_2 on afforested boreal agricultural soils, Plant Soil, 231, 113–121, 2001.

Maljanen, M., Hytönen, J., Mäkiranta, P., Alm, J., Minkkinen, K., Laine, J. and Martikainen, P.J.: Greenhouse gas emissions from cultivated and abandoned organic arable lands in Finland, Boreal Environment Research, 12, 133–144, 2007.

Maljanen, M., Sigurdsson, B. D., Guðmundsson, J., Óskarsson, H., Huttunen, J. T., and Martikainen, P. J.: Greenhouse gas balances of managed peatlands in the Nordic countries – present knowledge and gaps, Biogeosciences, 7, 2711–2738, doi:10.5194/bg-7-2711-2010, 2010.

Mander, Ü., Lõhmus, K., Teiter, S., Mauring, T., Nurk, K., and Augustin, J.: Gaseous fluxes in the nitrogen and carbon budgets of subsurface flow constructed wetlands, Sci. Total Environ, 404, 343–353, 2008.

McNeill, A., and Unkovich, M.: The nitrogen Cycle in Terrestrial Ecosystems, in Nutrient Cycling in Terrestrial Ecosystems, Eds. Marschner, R., Rengel, Z., Soil Biology, Volume 10, 397 pp., 2007.

Merbach, W., Augustin, J., and Gans, W.: Nitrous oxide emissions from fen mires in dependence of anthropogenic activities, Journal of Applied Botany, 75, 118–123, 2001.

Mogge, B., Kaiser, E.A., and Munch, J.C.: Nitrous oxide emissions and denitrification N-losses from forest soils in the Bornhöved lake region (northern Germany), Soil Biol. Biochem., 30 (6), 703–710, 1998.

Möller, K., and Stinner, W.: Effects of different manuring systems with and without biogas digestion on soil mineral nitrogen content and gaseous nitrogen losses (ammonia, nitroux oxide), Europ. J. Agronomy, 30, 1–16, 2009.

Moiser, A.R.: Exchange of gaseous nitrogen compounds between agricultural systems and the atmosphere. Plant Soil, 228, 17–27, 2001.

Moore, T.R., and Knowles, R.: The influence of water table on methane and carbon dioxide emissions from peatland soils, Can. J. Soil Sci., 69, 33–38, 1989.

Moore, T.R., and Dalva, M.: The influence of temperature and water table position on carbon dioxide and methane emissions from laboratory columns of peatland soils, J. Soil Sciences, 44, 651–664, 1993.

Mounier, E., Hallet, S., Chèneby, D., Benizri, E., Gruet, Y., Nguyen, C., Piutti, S., Robin, C., Slezack-Deschaumes, S., Martin-Laurent, F., Germon, J.C., and Philippot, L.: Influence of maize mucilage on the diversity and activity of the denitrifying community, Environ. Microbiol., 6, 301–312, 2004.

Nelson, D.W.: Gaseous loss of nitrogen other than through denitrification. In: Nitrogen in agricultural soils, Stevenson FJ (ed.), ASA-CSSA-SSSA, Madison, 327–363, 1982.

Nielsen, O.-K., Mikkelsen, M.H., Hoffmann, L., Gyldenkærne, S., Winther, M., Nielsen, M., Fauser, P., Thomsen, M., Plejdrup, M.S., Albrektsen, R., Hjelgaard, K., Bruun, H.G., Johannsen, V.K., Nord-Larsen, T., Bastrup-Birk, A., Vesterdal, L., Møller, I.S., Rasmussen, E., Arfaoui, K., Baunbæk, L. & Hansen, M.G.: Denmark's National Inventory Report 2012. Emission Inventories 1990-2010 - Submitted under the United Nations Framework Convention on Climate Change and the Kyoto Protocol. Aarhus University, DCE – Danish Centre for Environment and Energy, Scientific Report from DCE – Danish Centre for Environment and Energy No. 19 http://www.dmu.dk/Pub/SR19.pdf, 1168 pp., 2012.

Nilsson, M., Sagerfors, J. Buffam, I. Laudon, H. Eriksson, T. Grelle, A. Klemedtsson, L. Weslien, P. and Lindroth, A.: Contemporary carbon accumulation in a boreal oligotrophic minerogenic mire - A significant sink after accounting for all C-fluxes, Global Change Biol., 14 (10), 2317–2332, 2008.

Nilsson, M., and Öquist, M.: Partitioning litter mass loss into carbon dioxide and methane in peatland ecosystem, In: Carbon Cycling in Northern Peatlands, Baird, A.,J., Belyea, L.R., Comas, X., Reeve, A.S., and Slater, L.D. (eds.), Geophysical Monograph, 184, American Geophysical Union, Washington, DC, 2009.

NIR 2010: National Emission Inventory Report (NIR) 2010 for 2008 – Calculation of Emissions from German Agriculture, Ed. Haenel, H.D., vTI Agriculture and Forestry Research, Special Issue 334, 428 pp., 2010.

Oenema, O., Wrage, N., Velthof, G.L., van Groenigen, J.W., Dolfing, J., and Kuikman, P.J.: Trends in global nitrous oxide emissions from animal production systems, Nutrient Cycling in Agroecosystems, 72, 51–65, 2005.

Page, S.E., Rieley, J.O., and Banks, C.J.: Global and regional importance of the tropical peatland carbon pool, Global Change Biol., 17 (2), 798–818, 2011.

Petersen, S.O., Hoffmann, C.C., Schäfer, C.-M., Bilcher-Mathiesen, G., Elsgaard, L., Kristensen, K., Larsen, S.E., Torp, S.B., and Greve, M.H.: Annual emissions of CH_4 and N_2O, and ecosystem respiration, from eight organic soils in Western Denmark managed by agriculture, Biogeosciences, 9, 403–422, 2012.

Philipott, L., Andert, J., Jones, C.M., Bru, D. & Hallin, S.: Importance of denitrifiers lacking the genes encoding the nitrous oxide reductase for N_2O emissions from soil. Global Change Biol., 17, 1497–1504, 2011.

Pitcairn, C.E. R., and Fowler, D.: Deposition of fixed atmospheric nitrogen and foliar nitrogen content of Bryophytes and Calluna vulgaris (L.) Hull, Environmental Pollution, 88, 193205, 1995.

Poth, M., and Focht, D.D.: N-15 kinetic analysis of N_2O production by Nitrosomonas europaea – an examination of nitrifier denitrification, Appl. Environm. Microbiol., 49, 1134–1141, 1985.

Pretzsch, H., Biber, P., and Dursky, J.: The single tree-based stand simulator SILVA: construction, application and evaluation, Forest Ecology and Management, 162, 3–21, 2002.

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

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

Rolston, D. E., Hoffman, D. L., and Toy, D.W.: Field measurements of denitrification: 1. Flux of N_2 and N_2O, Soil Sci. Soc. Am. J., 42, 863–869, 1978.

Roobroeck, D., Butterbach-Bahl, K., Brüggemann, N., and Boeckx, P.: Dinitrogen and nitrous oxide exchanges from an undrained monolith fen: short-term responses following nitrate addition, Eur. J. Soil Sci., 61, 662–670, 2010.

Roulet, N.T., Lafleur, P.M., Richard, P.J.H., Moore, T.R., Humphreys, E.R., and Bubier, J.: Contemporary carbon balance and late holocene carbon accumulation in a northern peatland, Glob. Change Biol., 13, 397–411, 2007.

Rösch, C., Skarka, J., Raab, K., and Stelzer, V.: Energy production from grassland – Assessing the sustainability of different process chains under German conditions, Biomass and Bioenergy, 33, 689–700, 2009.

Rubæk, G.H., Henriksen, K., Petersen, J., Rasmussen, B., and Sommer, S.G.: Effects of application technique and anaerobic digestion on gaseous loss from animal slurry applied to ryegrass (*Lolium perenne*), J. Agric. Sci. Camb., 126, 481–492, 1996.

Rydin, H., and Jeglum, J.K.: The Biology of Peatlands, with contributions from Hooijer, A., Clarkson, B.R., Clarkson, B.D., Mauquoy, D., and Bennett, K.D., The Biology of Habitats series, Oxford University Press, 343 pp., 2006.

Sagerfors, J., Lindroth, A., Grelle, A., Klemedtsson, L., Weslien, P., and Nilsson, M.: Annual CO_2 exchange between a nutrient-poor, minerotrophic, boreal mire and the atmosphere, Journal of Geophysical Research, 113 (G1), http://dx.doi.org/10.1029/2006JG000306, 2008.

Schäfer, A., and Joosten, H., (eds.): Erlenaufforstung auf wiedervernässten Niedermooren. (Alnus afforestation on restored fen peatlands), Greifswald: Institute for Sustainable Development of Landscapes of the Earth, 68 pp., 2005.

Schäfer, C.M., Elsgaard, L., Hoffmann, C.C., and Petersen, S.O.: Seasonal methane dynamics in three temperate grasslands on peat, Plant Soil, 357, 339–353, 2012.

Scheffer, F., and Schachtschabel, P.: Lehrbuch der Bodenkunde. Neu bearbeitet und erweitert von Blume, H. P. & Brümmer, G. W. & Schwertmann, U. & Horn, R. & Kögler-Knabner, I. & Stahr, K. & Auerswald, K. & Beyer, L. & Hartmann, A. & Litz, N. & Scheinost, A. & Stanjek, H. & Welp, G. & Wilke, B. M., Spektrum Akademischer Verlag, Heidelberg, 2002, 15, 593 pp., 2002.

Schleupner, C., and Schneider, U.A.: Effects of bioenergie policies and targests on European wetland restoration options, Environmental Science & Policy, 13, 721–732, 2010.

Scholefield, D., Hawkins, J.M.B., and Jackson, S.M.: Use of a flowing helium atmosphere incubation technique to measure the effects of denitrification controls

applied to intact cores of a clay soil, Soil Biol. Biochem., 29 (9–10), 1337–1344, 1997.

Schothorst, C.J.: Subsidence of low moor peat soils in the Western Netherlands, Geoderma, 17, 265–291, 1977.

Schulze, E.D., Ciais, P., Luyssaert, S., Schrumpf, M., Janssens, I.A., Thiruchittampalam, B., Theloke, J., Saurat, M., Bringezu, S., Lelieveld, J., Lohila, A., Rebmann, C., Jung, M., Bastviken, D., Abril, G., Grassi, G., Leip, A., Freibauer, A., Kutsch, W., Don, A., Nieschulze, J., Boerner, A., Gash, J.H., and Dolman, A.J.: The European carbon balance. Part 4: integration of carbon and other trace-gas fluxes, Glob. Change Biol., 16 (5), 1451–1469, 2010.

Segers, R.: Methane production and methane consumption: a review of processes underlying wetland methane fluxes, Biogeochemistry, 41, 23–51, 1998.

Sey, B.K., Manceur, A.M., Wahlen, J.K., Gregorich, E.G, and Rochette, P.: Root-derived respiration and nitrous oxide production as affected by crop phenology and nitrogen fertilization, Plant Soil, 326, 369–379, 2010.

Sheppard, L.J., Leith, I.D., Leeson, S.R., van Dijk, N., Field, C., and Levy, P.: Fate of N in a peatland, Whim bog: immobilisation in the vegetation and peat, leakage into pore water and losses as N_2O depend on the form of N, Biogeosciences, 10, 149–160, 2013.

Silvan, N., Tuittila, E.-S., Kitunen, V., Vasander, H., and Laine, J.: Nitrate uptake by *Eriophorum vaginatum* controls N_2O production in a restored peatland, Soil Biol. Biochem., 37, 1519–1526, 2005.

Smith, V.H., Tilman, G.D., and Nekola, J.C.: Eutrophication: impacts of excess nutrient input on freshwater, marine, and terrestrial ecosystems, Environmental Pollution, 100, 179–196, 1999.

Smyth, B.M., Murphy, J.D., and O'Brien, C.M.: What is the energy balance of grass biomethane in Ireland and other temperate northern European climates?, Renewable and Suitable Energy Reviews, 13, 2349–2360, 2009.

Sonneveld, M.P.W., and Lantinga, E.A.: The contribution of mineralization to grassland N uptake on peatland soils with anthropogenic A horizons, Plant Soil; 340, 357–368, 2011.

Soosaar, K., Mander, Ü., Maddison, M., Kanal, A., Kull, A., Lõhmus, K., Truu, J., and Augustin, J.: Dynamics of gaseous nitrogen and carbon fluxes in riparian alder forests, Ecol. Eng., 37, 40–53, 2011.

Steffen R, Szolar O, and Braun R.: Feedstocks for anaerobic digestion, Vienna: Institute for Agrobiotechnology Tulln, University of Agricultural Sciences; 1998.

Succow, M., and Joosten, H.: Landschaftsökologische Moorkunde, Unveränderter Nachdruck der 2. Auflage, E.Schweizerbart'sche Verlagsbuchhandlung, Stuttgart, Germany, 622 pp., 2012.

Sulman, B.N., Desai, A.R., Saliendra, N.Z., Lafleur, P.M., Flanagan, L.B., Sonnentag, O., Mackay, D.S., Barr, A.G., and Kamp, G.V.D.: CO_2 fluxes at northern fens and bogs have opposite responses to inter-annual fluctuations in water table, Geophys. Res. Lett., 37, L19702, doi:10.1029/2010GL044018, 2010.

Svensson, B.H., and Sundh, I.: Factors affecting methane production in peat soils. Suo., 43, 183–190, 1992.

Tarnocai, C.: The effect of climate change on carbon in Canadian peatlands, Global and Planetary Change, 53, 222–232, 2006.

Tarnocai, C., Canadell, J.G., Schuur, E.A.G., Kuhry, P., Mazhitova, G., and Zimov, S.: Soil organic carbon pools in the northern circumpolar permafrost region, Global Biogeochem. Cycles, 23, 1–11 2009.

Teiter, S. and Mander, Ü.: Emission of N_2O, N_2, CH_4 and CO_2 from constructed wetlands for wastewater treatment and from riparian buffer zones, Ecol. Eng., 25, 528–541, 2005.

Tiemeyer, B., Freibauer,A., Drösler, M., Albiac-Borraz, E., Augustin J., Bechtold, M., Beetz, S., Belting, S., Bernrieder, M., Beyer, C., Eberl, J., Eickenscheidt, T., Fell, H., Fiedler, S., Förster, C., Frahm, E., Frank, S., Giebels, M., Glatzel, S., Grünwald, T., Heinichen, J., Hoffmann, M., Hommeltenberg, J., Höper, H., Laggner, A., Leiber-Sauheitl, K., Leppelt, T., Metzger, C., Peichl-Brak, M., Röhling, S., Rosskopf, N., Rötzer, T., Sommer, M., Wehrhan, M., Werle, P., and Zeitz, J.: Klimarelevanz von Mooren und Anmooren in Deutschland: Ergebnisse aus dem Verbundprojekt "Organische Böden in der Emissionsberichterstattung", Thünen Working Paper, No. 15, http://nbn-resolving.de/urn:nbn:de:gbv:253-201311-dn052806-7, 2013.

Tiemeyer, B., Borraz, E.A, Augustin, J., Bechtold, M., Beetz, S., Beyer, C., Eickenscheidt, T., Drösler, M., Förster C., Freibauer, A., Giebels, M., Glatzel, S., Heinichen, J., Hoffmann, M., Höper, H., Leiber-Sauheitl, K., Rosskopf, N., and Zeitz, J.: Greenhouse gas budgets for grasslands on peatlands and other organic soils, Geophysical Research Abstracts, Vol. 16, EGU2014-14825, 2014, EGU General Assembly, 2014.

Tomassen, H.B.M., Smolders, A.J.P., Limpens, J., Lamers, L.P.M., and Roelofs, J.G.M.: Expansion of invasive species on ombrotrophic bogs: desiccation or high N deposition?, J. Appl. Ecol., 41 (1), 139–150, 2004.

Treat, C.C., Wisser, D., Marchenko, S., and Frolking, S.: Modelling the effects of climate change and disturbance on permafrost stability in northern organic soils, Mires and Peat, 12 (2), 1–17, 2013.

Turetsky, M. R., Amiro, B. D., Bosch, E., and Bhatti, J. S.: Historical burn area in western Canadian peatlands and its relationship to fire weather indices, Glob. Biogeochem. Cycles, 18 (4), GBD4014, 2004.

Turunen, J., Tomppo, E., Tolonen, K., and Reinikainen, A.: Estimating carbon accumulation rates of undrained mires in Finland – application to boreal and subarctic regions, The Holocene, 12 (1), 69–80, 2002.

UNECE: Manual on Methodologies and Criteria for Modeling and Mapping Critical Loads and Levels and Air Pollution Effects, Risks and Trends, 52/04. Federal Environmental Agency, Germany, 2004.

Updegraff, K., Pastor, J., Bridgham, S.D., and Johnston, C.A.: Environmental and substrate controls over carbon and nitrogen mineralization in northern wetlands, Ecol. Appl., 5, 151–163, 1995.

Valentine, D.L., and Reeburgh, W.S.: New perspectives on anaerobic methane oxidation, Environmental Microbiology, 2 (5), 477–484, 2000.

Van Beek, C.L., Pleijter, M., Jacobs, C.M.J., Velthof, G.L., van Groenigen, J.W., and Kuikman, P.J.: Emissions of N_2O from fertilized and grazed grassland on organic soil in relation to groundwater level, Nutr. Cycl. Agroecosyst., 86, 331–340, 2010.

Van Beek, C.L., Pleijter, M., and Kuikman, P.J.: Nitrous oxide emissions from fertilized and unfertilized grasslands on peat soil, Nutr. Cycl. Agroecosyst., 89, 453–461, 2011.

Van Cleemput, O.: Subsoils: chemo- and biological denitrification, N_2O and N_2 emissions, Nutrient Cycl. Agroecosystems, 52, 187–194, 1998.

Van Cleemput, O., Boeckx, P., Lindgren, P.-E., and Tonderski, K.: Denitrification in wetlands. In: Biology of the Nitrogen Cycle, Bothe, H., Ferguson, S.J., and Newton, W.E., (eds.), Elsevier B.V., 2007.

Vandré, R., Clemens, J., Goldbach, H., and Kaupenjohann, M.: NH_3 and N_2O Emissions after Landspreading of Slurry as Influenced by Application Technique and Dry Matter-Reduction. I. NH_3 Emissions, Z. Pflanzenernähr. Bodenk., 160, 303–307, 1997.

Velthof, G.L., Brader, A.B., and Oenema, O.: Seasonal variations in nitrous oxide losses from managed grasslands in the Netherlands, Plant and Soil, 181, 263–274, 1996.

Velthof, G.L., Kuikman, P.J., and Oenema, O.: Nitrous oxide emission from animal manures applied to soil under controlled conditions, Biol. Fertil. Soils, 37, 221–230, 2003.

Venterea, R.T.: Simplified method for Quantifying theoretical underestimation of chamber-based trace gas fluxes, J. Environ. Qual., 39, 126–135, 2010.

Vepraskas M.J., and Faulkner, S.P.: Redox Chemistry of Hydric soils, in: Wetland Soils: Genesis, Hydrology, Landscapes, and Classification J.L., Richardson and M.J., Vepraskas (eds.), Lewis Publisher, Boca Raton, 84–105, 2001.

Vitousek, P.M., Aber, J.D., Howarth, R.W., Likens, G.E., Matson, P.A., Schindler, D.W., Schlesinger, W.H., and Tilman, D.G.: Human alteration of the global nitrogen cycle: sources and consequences, Ecol. Appl., 7 (3), 737–750, 1997.

Vitousek, P.M., Cassman, K., Cleveland, C., Crews, T., Field, C.B., Grimm, N.B., Howarth, R.W., Marino, R., Martinelli, L., Rastetter, E.B., and Sprent, J.I.: Towards an ecological understanding of biological nitrogen fixation. Biogeochemistry 57, 1–45. (doi:10.1023/ A:1015798428743), 2002.

Vile, M.A., Kelman Wieder, R., Živković, T., Scott, K.D., Vitt, D.H., Hartsock, J.A., Iosue, C.L., Quinn, J.C., Petix, M., Fillingim, H.M., Popma, J.M.A., Dynarski, K.A., Jackman, T.R., Albright, C.M., and Wykoff, D.D.: N_2-fixation by methanotrophs sustains carbon and nitrogen accumulation in pristine peatlands, Biogeochemistry, 121, 317–328, 2014.

Watson, R.T., Noble, I.R., Bolin, B., Ravindranath, N.H., Verardo, D.J. and Dokken, D.J.: Special Report of the IPCC on Land Use, Land-Use Change, and Forestry, Cambridge University Press, UK, 375 pp., 2000.

Weier, K. L., Doran, J. W., Power, J. F., and Walters, D. T.: Denitrification and dinitrogen/nitrous oxide ratio as affected by soil water, available carbon, and nitrate, Soil Sci. Soc. Am. J., 57, 66–72, 1993.

Weiland, P.: Biogas production: current state and perspectives, Appl. Microbiol. Biotechnol., 85, 849–860, 2010.

Whalen, S.C.: Biogeochemistry of methane exchange between natural wetlands and the atmosphere, Environmental Engineering Science, 22 (1), 73–94, 2005.

Wichtmann, W., and Joosten, H.: Paludiculture: peat formation and renewable resources from rewetted peatlands, In: IMCG-Newsletter 3/2007, 24–28, 2007.

Wochele, S. and Kiese, R.: Modellierung und Kartierung räumlich differenzierter Wirkungen von Stickstoffeinträgen in Ökosysteme im Rahmen der UNECE-Luftreinhaltekonvention - Teilbericht I: Simulationen ökosystemarer Stoffumsetzungen und Stoffausträge aus Waldökosystemen in Deutschland unter Berücksichtigung geänderter Stoffeinträge und Klimabedingungen, Umweltbundesamt, Dessau, 115 pp., 2010.

World Meteorological Organization (WMO): Global Atmosphere Watch Programme: http://www.wmo.int/pages/mediacentre/press_releases/pr_980_en.html, 11-2014.

Wrage, N., Velthof, G.L., Van Beusichem, M.L., and Oenema, O.: The role of nitrifier denitrification in the production of nitrous oxide, Soil Biol. Biochem., 33, 1723–1732, 2001.

Wrage, N., Van Groeningen, J.W., Oenema, O. & Baggs, E.M.: A novel dualisotope labelling method for distinguishing between soil sources of N_2O, Rapid Comm. Mass Spectrom., 19, 3298–3306, 2005.

Wray, H. E. and Bayley, S. E.: Denitrification rates in marsh frings and fens in two boreal peatlands in Alberta, Canada, Wetlands, 27, 1036–1045, 2007.

WRB, 2006 – IUSS Working Group: World Reference Base for Soil Resources 2006, 2nd edition, World Soil Resources Reports No. 103. Rome. 2006.

Wulf, S., Maeting, M., and Clemens, J.: Application Technique and Slurry Co-Fermentation Effects on Ammonia, Nitrous Oxide, and Methane Emissions after Spreading: II Greenhouse Gas Emissions, J. Environ. Qual., 31, 1795–1801, 2002.

Yu, Z., Loisel, J., Brosseau, D.P., Beilman, D.W., and Hunt, S.: Global peatland dynamics since the Last Glacial Maximum, Geophysical Research Letters, 37, 1–5, 2010.

Yu, Z., Beilman, D.W., Frolking, S., MacDonald, G.M., Roulet, N.T., Camill, P., and Charman, D.J.: Peatland and Their Role in the Global Carbon Cycle, EOS, 92 (12), 97–108.

6 ACKNOWLEDGMENT

I would never have been able to finish my dissertation without the help and support of the kind people around me. For only some of whom it is possible to express my gratitude here. To all the others who made this thesis possible and were omitted, I would like to thank them for their individual support and helpfulness.

My special thanks go to my supervisor Prof. Dr. Matthias Drösler for providing me an interesting and challenging research topic, his efforts to guarantee a continuous financial funding during the last six years, for his help in scientific questions and especially for leaving room for independent work and ideas, which he always unreservedly supported.

I would wish to express my special gratitude to Prof. Dr. Johannes Kollmann for acting as referee for this thesis.

I wish to express my special thanks to Prof. Dr. Jürgen Augustin for acting as co-referee of this thesis, for the opportunity to conduct the helium-incubation experiment at the laboratory of the ZALF and his support and help especially during the manuscript preparation and review process. His scientific knowledge contribute a lot to the success of this work.

Further I thank Prof. Dr. Hans Schnyder for acting as chairman of the examination committee.

I would like to thank Priv.-Doz. Dr. Harald Albrecht for stepping in to act as supervisor after the retirement of Prof. Dr. Jörg Pfadenhauer.

Most importantly, I want to express my gratitude with all my heart to my sister Dr. Nadine Eickenscheidt for her tireless support and help in all scientific questions, critical review of all manuscripts and belief in me. Without you, I would not have finished my PhD thesis.

I am very grateful to Jan Heinichen, despite the high work load regarding his own PhD project, he gave tireless support during all the years of excessive field sampling and laboratory work, as well as during data analysis and manuscript preparations.

Furthermore, I like to emphasize the support of Annette Freibauer, especially her motivating and impressing way for writing scientific papers and her profound scientific knowledge, which distinctly improve the quality of the first two publications.

Special thanks deserve all my colleagues from the chairs of Vegetation ecology and Restoration ecology namely: Jan, Gawan, Jörg, Shom, Michael, Alicia, Martina, Christoph, Christine, Marika, Julia, Wolfram, Ingrid Kapps and Michaela Bücherl for their

7 APPENDIX

7.1 PUBLICATION I.

Eickenscheidt, T., Heinichen, J., and Drösler, M.: The greenhouse gas balance of a drained fen peatland is mainly controlled by land-use rather than soil organic carbon content, Biogeosciences Discuss. (submitted for publication in Biogeosciences).

The greenhouse gas balance of a drained fen peatland is mainly controlled by land-use rather than soil organic carbon content.

T. Eickenscheidt[1,2], J. Heinichen[1,2], M. Drösler[1]

[1] {University of Applied Sciences Weihenstephan-Triesdorf, Chair of Vegetation Ecology, Weihenstephaner Berg 4, 85354 Freising, Germany}

[2] {Technische Universität München, Chair of Restoration Ecology, Emil-Ramann-Str. 6, 85354 Freising, Germany}

Correspondence to: T. Eickenscheidt (tim.eickenscheidt@hswt.de)

Abstract

Drained organic soils are considered as hotspots for greenhouse gas (GHG) emissions. Particularly arable lands and intensively used grasslands have been regarded as the main producers of carbon dioxide (CO_2) and nitrous oxide (N_2O). However, GHG balances of former peatlands and associated organic soils not considered as peatland according to the definition of the Intergovernmental Panel on Climate Change (IPCC) have not been investigated so far. Therefore, our study addressed the question to what extent the soil organic carbon (SOC) content affects the GHG release of drained organic soils under two different land-use types (arable land and intensively used grassland). Both land-use types were established on a mollic Gleysol (named C_{medium}) as well as on a sapric Histosol (named C_{high}). The two soil types significantly differed in their SOC contents in the topsoil (C_{medium}: 9.4–10.9% SOC; C_{high}: 16.1–17.2% SOC). We determined GHG fluxes (CO_2, N_2O and methane (CH_4)) over a period of 2 years. The daily and annual net ecosystem exchange (NEE) of CO_2 was determined with the closed dynamic chamber technique and by modeling the ecosystem respiration (R_{ECO}) and the gross primary production (GPP). N_2O and CH_4 were determined by the close chamber technique. Estimated NEE of CO_2 significantly differed between the two land-use types with lower NEE values (–6 to 1707 g CO_2-C m^{-2} yr^{-1}) at the arable sites and higher values (1354 to 1823 g CO_2-C m^{-2} yr^{-1}) at the grassland sites. No effect on NEE was found regarding the SOC content. Significantly higher annual N_2O exchange rates were observed at the arable sites (0.23–0.86 g N m^{-2} yr^{-1}) compared to the grassland sites

34 (0.12–0.31 g N m^{-2} yr^{-1}). Furthermore, N$_2$O fluxes from the C$_{high}$ sites significantly

35 exceeded those of the C$_{medium}$ sites. CH$_4$ fluxes were found to be close to zero at all plots.

36 Estimated global warming potential, calculated for a time horizon of 100 years (GWP$_{100}$)

37 revealed a very high release of GHGs from all plots ranging from 1837 to 7095 g CO$_{2\ eq.}$

38 m^{-2} yr^{-1}. Calculated global warming potential (GWP) values did not differ between soil

39 types and partly exceeded the IPCC default emission factors of the Tier 1 approach by

40 far. However, despite being subject to high uncertainties, the results clearly highlight the

41 importance to adjust the IPCC guidelines for organic soils not falling under the definition,

42 to avoid a significant underestimation of GHG emissions in the corresponding sectors of

43 the national climate reporting. Furthermore, the present results revealed that mainly the

44 land-use including the management and not the SOC content is responsible for the

45 height of GHG exchange from intensive farming on drained organic soils.

46

47 **1.Introduction**

48 Natural peatlands act as a sink for atmospheric carbon dioxide (CO$_2$) and as a source

49 for methane (CH$_4$) (Blodau, 2002; Whalen, 2005; Drösler et al., 2008). The net climate

50 effect of natural peatlands regarding the greenhouse gas (GHG) fluxes, however, is

51 close to zero (Drösler et al. 2008). In the last century, drainage and intensification of

52 agriculture turned European peatlands to hot spots for GHG emissions (Drösler et al.,

53 2008). Increased CO$_2$ and nitrous oxide (N$_2$O) emissions have been observed from

54 drained peatlands as a result of enhanced decomposition of organic matter (Martikainen

55 et al., 1993, Silvola et al.,1996). The mentioned gases (CO$_2$, CH$_4$ and N$_2$O) act as

56 climatic relevant greenhouse gases (IPCC, 2007). Additionally, N$_2$O contributes to the

57 chemical destruction of stratospheric ozone (Crutzen, 1979).

58 Through the ratification of several international agreements on climate protection (e.g.

59 UNFCCC 1992, Kyoto protocol 1997 – specified by the Bonn Agreements and

60 Marrakesh Accords, several EU decisions) Germany is obliged to publish annual

61 national greenhouse gas emissions inventories according to the Intergovernmental

62 Panel on Climate Change (IPCC) guidelines. However, the national climate reporting in

63 the Land-use, Land-Use Change and Forestry (LULUCF) sector as well in the

64 Agriculture, Forestry and Other Land-uses (AFOLU) sector is challenging for organic

65 soils. This is mainly because reliable measurements of GHGs from temperate drained

66 peatlands are rare and observed GHG fluxes show a large temporal and spatial

67 variability ranging from −2 to 31 t CO$_2$–C ha^{-1} yr^{-1} and 2 to 38 kg N$_2$O–N ha^{-1} yr^{-1} (IPCC,

68   2014). Furthermore, the definition of histosols is complex (Couwenberg, 2011) and

69   several national and international classification systems exist for organic soils. For the

70   climate reporting under LULUCF/AFOLU, the IPCC guidelines require at least ≥10 cm

71   thickness of the soil/peat layer and a C_{org} content of ≥12% in case of a soil thickness

72   ≤20 cm for peat soils. Thus, the IPCC definition of peat soils is broader than the definition

73   of histosols in the world reference base for soil resources (WRB, 2008). In the German

74   classification system (KA5) (Ad-hoc-AG Boden, 2005) a distinction is made between soil

75   horizons with ≥30% soil organic matter (SOM) content (called organic horizon) and those,

76   containing 15–30% SOM (called anmoor horizon). Particularly at the boundary between

77   mineral and organic soils, the conversion from SOM to C_{org} leads to uncertainties due to

78   different conversion factors which are commonly used for mineral soils and peat soils

79   according to the KA5 (Tiemeyer et al., 2013). Depending on the conversion factor (1.72

80   for mineral soils or 2 for peat soils), the maximum limit of IPCC requirement is between

81   21% and 24% SOM (Tiemeyer et al., 2013). Up to date, soils which are, by definition in

82   the transition between mineral and organic soils were mostly neglected in the national

83   GHG inventory of most countries (Leiber-Sauheitl et al., 2014). In the Danish

84   greenhouse gas inventory, for example, GHG emissions from very thin and shallow

85   organic soils, which do not meet the definition of organic soils according to the IPCC,

86   were additionally considered. Due to a lack of information about the release of GHG

87   emissions of those soils, a fixed emission factor, half as much as for typical organic soils

88   (>12% C_{org}), has been introduced in Denmark for soils containing 6–12% organic carbon

89   (Nielsen et al., 2012).

90   According to estimates, peatlands in Germany account for approximately 5.1% of the

91   national GHG emissions although they only account for 5.1% of the total area (NIR, 2010;

92   Drösler et al., 2011). Drained peatlands even represent the largest single source for

93   GHG emissions outside the energy sector in Germany (Drösler et al., 2011; NIR, 2010).

94   Hence, according to the IPCC guidelines, drained peatlands are identified as key

95   category which leads to the fact that Germany is obligated to calculate the annual GHG

96   emission inventory on the basis of national specific emission factors (EF; Tier 2 or Tier

97   3 methods). The main reason for the critical climate balance is caused by the fact that

98   more than two-thirds of the German peatlands are intensively used as grassland or

99   arable land (Drösler et al., 2008). Both land-use types have been regarded as the main

100 producers of CO_2 and N_2O from farmed organic soils (Kasimir-Klemedtsson et al., 1997;

101 Kroeze et al., 1999; Drösler et al., 2008; International Peat Society, 2008). Highest GHG

102   emissions from drained organic soils were related to management activities such as

103   tillage and fertilization which enhance microbial SOM decomposition and nitrogen

104   turnover (Kandel et al., 2013). Beside management practices, several other physical and

105   chemical factors control the intensity of mineralization processes (Heller and Zeitz, 2012)

106   in which soil temperature and soil moisture are considered to be the primary regulators

107   for CO_2 emissions from soils (Silvola et al. 1996; Maljanen et al., 2001; Hardie et al.,

108   2011). However, recent studies have shown that in particular the SOM quality and its

109   labile and more recalcitrant fractions act as key variables affecting the decomposability

110   of SOM and thus control CO_2 fluxes from peatlands (Byrne and Farrell, 2005; Heller and

111   Zeitz, 2012; Leifeld et al., 2012). Beside the macromolecular organic composition (e.g.

112   polysaccharides, lignin, aliphatic biopolymers) of the peat forming vegetation, the SOM

113   quality of peat strongly depends on hydrological and geomorphological building

114   conditions during peat formation (Heller and Zeitz, 2012). Additionally, peat and SOM

115   quality is strongly affected by human impact which leads to peat shrinking, secondary

116   decomposition and mineralization (Heller and Zeitz, 2012). It can be assumed that with

117   increasing peat humification, aggregation and organo-mineral association gain in

118   importance in the SOM stabilization and thus decrease the CO_2 emissions from peat

119   soils which are by definition in the transition between mineral soils and peat compared

120   to peat soils with higher SOM content. However, those soils were mostly neglected and

121   to our knowledge no GHG studies exist for such soil types under intensive agricultural

122   land-use.

123   The objective of this study was to quantify GHG emissions from arable lands and

124   grasslands on two types of drained organic soils with different C_{org} contents in South

125   Germany. We hypothesize: i) that GHG emissions significantly increase with increasing

126   SOC content in the soil and ii) that GHG emissions from arable soils exceed GHG

127   emissions from intensive managed grassland soils.

128

129   **2.Material and methods**

130   **3.Study area and experimental design**

131   The study was conducted at a drained fen peatland 30 km north-east of Munich

132   (Freisinger Moos, 48°21'N, 11°41'E; 450 m a.s.l.). Since 1914 the Freisinger Moos (FSM)

133   was systematically drained for intensive cultivation (Zehlius-Eckert et al., 2003). Today

134   about 40% of the whole area is used as grassland and 20% as arable land (Schober et

135   al., 2008).

136   According to the climate station in Weihenstephan, located 10 km northeast of the study

137   sites, the 30-years mean annual temperature was 7.5 ℃ and the mean annual

138   precipitation was 787 mm (1961–1990). Annual atmospheric N deposition amounted to

139   6.22 and 7.20 kg N ha^{-1} yr^{-1} in 2010 and 2011. Data of N deposition was collected by

140   the Bavarian State Institute of Forestry at a German Level II monitoring plot (Forest

141   Intensive Monitoring Programme of the UNECE), located in 7 km distance to the

142   investigated sites.

143   In October 2009, we selected two adjacent areas, one used as intensive grassland and

144   the other as arable land. Both areas are characterized by a distinct gradient in their soil

145   organic carbon (SOC) content in the top soil (Table 1), which increases from southeast

146   to northwest. In March 2010 the arable land was split into two equal halves to simulate

147   two different crop rotations (maize (*Zea mays*) and oat (*Avena sativa*); see Table 2)

148   along the SOC gradient (named A1 and A2). At the grassland area a similar design was

149   conducted to investigate the effect of two different organic fertilizers (named G1,

150   fertilized with cattle slurry and G2, fertilized with biogas digestate). Within these areas

151   we selected two sites with maximum different SOC contents per land-use (Fig. 1).

152   According to the WRB (2006), soil types at the sites were classified as mollic Gleysol

153   (named C_{medium}) or as sapric Histosol (named C_{high}) (N. Roßkopf personal

154   communication, 2010). At each site two plots were selected according to the

155   management type (Fig. 1). A detailed description of the experimental design of the

156   grassland sites and the chemical and physical composition of the applied fertilizers is

157   given in Eickenscheidt et al. (2014b). The arable land was managed according to organic

158   farming criteria but without any fertilization during the investigated period.

159   At each plot, three PVC-collars for GHG measurements (inside dimension 75 x 75 cm)

160   were permanently inserted 10 cm into the soil with a distance of 1.5–2 m to each other.

161   In case of management activities, collars were removed for a short period at the arable

162   land. To prevent oscillations of the peat through movements during the measurements,

163   boardwalks were installed. In March 2010, climate stations were centrally set up

164   between two identical land-use and soil types for the continously recording (every 0.5

165   hour) of air temperature (T_{air}) and humidity at 20 cm above soil surface, soil temperatures

166   at the depth –2, –5 and –10 cm ($ST_{2, 5, 10}$) and soil moisture content at –5 cm depth. In

167   addition, two further climate stations, additionally equipped with air temperature in 200

168   cm above soil surface and photosynthetic active radiation (PAR) sensors were operated

169   in close proximity to the investigated areas. For measuring the groundwater table, plastic

170   perforated tubes (JK-casings DN 50, 60 mm diameter, 1 m length) were inserted close

171   to each collar for plot-specific measurements of groundwater (GW) tables during gas

172   flux measurements at the grassland plots. At the arable land only three tubes were

173   inserted between the two plots of the same soil type. In April 2010, we equipped one

174   tube per plot or, in case of the arable land one tube per soil type, with a water level

175   logger (Type MiniDiver, Schlumberger water services), which recorded the water tables

176   every 15 minutes. Additionally to the recorded data, plot-specific soil temperatures in

177   three soil depths (−2, −5 and −10 cm) were determined with penetration thermometers

178   at the beginning and end of each gas flux measurement.

179

180   **4.Crop yield, soil sampling and laboratory analyses**

181   Crop yield was determined by harvesting the biomass inside the PVC-collars with a

182   scissor at each harvesting event (same cutting height as the farmers) (Table 2). To

183   determine the annual crop yield, grass samples were oven dried at 60°C for 48 hours

184   and phytomasses of each harvesting event per year were summed. To determine the

185   total carbon (C_{tot}) and total nitrogen (N_{tot}) content, total phytomasses was milled (0.5 mm)

186   and a pooled and homogenized sample from each PVC-collar and harvesting event was

187   analysed by the AGROLAB Labor GmbH (Bruckberg, Germany).

188   Mineral N ($N_{min} = NH_4^+-N + NO_3^--N$) contents of each plot were determined according

189   to VDLUFA (1997). Samples were taken during every CH_4/N_2O gas flux measurement.

190   For the determination of C_{tot} and organic carbon (C_{org}), a mixed soil sample of nine

191   individual samples was collected close to each collar at two soil depths (0–10, 10–20

192   cm) using a 3 cm diameter auger. After drying for 72 hours at 40 °C, soil samples were

193   sieved to 2 mm to remove stones and living roots. Analyses were conducted at the

194   Division of Soil Science and Site Science (Humbold Universität zu Berlin, Germany). For

195   the determination of bulk density and porosity, three undisturbed core cutter samples

196   (100 cm³) were randomly taken at four depths (0–5, 5–10, 10–15, 15–20 cm) for each

197   plot.

198

199   **5.GHG measurements**

200   We measured fluxes of N_2O and CH_4 every second week from December 2009 to

201   January 2012 using the static manual chamber method (Livingston and Hutchinson;

202   1995). In periods when the vegetation grew higher than the chamber height was (0.5 m),

203   extensions were used between the collar and chamber (white, opaque, volume varied

204 between 309 and 1236 L). A detailed description of chamber dimension and

205 configuration is given in Drösler (2005). N_2O and CH_4 gas flux rates were calculated

206 from the linear change in gas concentration over time (four gas samples; 60 min

207 enclosure time, 120 min in case of two or more extensions) considering chamber air

208 temperature and atmospheric pressure. Gas fluxes were accepted when the linear

209 regression was significant ($P \leq 0.05$). In case of small N_2O or CH_4 fluxes, fluxes were

210 also accepted if the coefficient of determination was ≥ 0.90 and the regression slope

211 was between -1 and 1 ppb min^{-1}. The cumulative annual mean exchange rate was

212 calculated by linear interpolation between the measurement dates. To minimize diurnal

213 variation in the flux pattern, N_2O and CH_4 sampling was always carried out between 9.00

214 a.m. and 11.30 a.m. We removed the gas fluxes measured in 2010 from the data set

215 due to errors in the gas chromatography analysis and due to long vial storage. A detailed

216 description of gas sampling and gas chromatograph settings is given in Eickenscheidt

217 et al. (2014a and 2014b).

218 For CO_2 flux measurements we used the closed dynamic manual chamber system which

219 was described in detail by Drösler (2005) and Elsgaard et al. (2012). CO_2 measurement

220 campaigns took place in irregular time intervals depending on weather conditions,

221 management activities and the phenological stage of plants (Table S1–S8 in the

222 Supplement). Measurement campaigns always started one hour before sunrise and

223 lasted till late afternoon to cover the full range of the photosynthetic active radiation (PAR)

224 and air and soil temperature. Opaque and transparent chambers (same dimension as

225 for N_2O and CH_4 measurements) were alternately used at each of the three collars per

226 plot during the time course of a measurement campaign to obtain the ecosystem

227 respiration (R_{ECO}) and the net ecosystem exchange (NEE). In total up to 55 NEE

228 measurements and 33 R_{ECO} measurements were conducted per measurement day and

229 plot (Table S1–S8 in the Supplement). As for N_2O and CH_4 measurements, extensions

230 were installed between the collar and chamber in case of vegetation growing higher than

231 the chamber height was (transparent or opaque, volume varied between 309 and 1236

232 L). Chambers were connected to an infrared gas analyser (IRGA, LI-820, LI-COR, USA),

233 which continuously determined the CO_2 headspace concentration. In the case that

234 extensions were used, chamber air from each level of an extension (every 0.5 m) was

235 sucked and merged to guarantee a reliable mixture signal from inside the chamber.

236 Additionally, three fens (SUNON® Super Silence MAGLev®-Lüfter) continuously

237 operated during the measurement to ensure a constant mixing of the chamber air.

238 Chamber enclosure time was 120 s for transparent chambers and 240 s for opaque
239 chambers, respectively. The CO_2 concentration, air temperature from inside the
240 chamber and site specific PAR was recorded every 5 s with a data logger (GP1 Data
241 logger, Delta-T Devices, UK). To prevent heating of the air in the transparent chambers,
242 freezer packs (1–10 pieces) were positioned in the air stream of the fens at the inner
243 surface of the PVC collar (Drösler, 2005; Beetz et al., 2013). Single measurements
244 where the PAR changed more than 15% of the starting value or the temperature inside
245 the chamber increased more than 1.5 °C compared to the outside air temperature were
246 discarded and measurement was repeated (Leiber-Sauheitl et al., 2014). CO_2 gas fluxes
247 were calculated by linear regression. Non significant gas fluxes ($P \geq 0.05$) with slopes
248 close to zero or zero (equilibrium between GPP and R_{ECO}) were not discarded (Alm et
249 al., 2007; Leiber-Sauheitl et al., 2014). For NEE flux calculation, a minimum time interval
250 of 25 s was used, whereas for R_{ECO} fluxes a minimum interval of 60 s was applied.
251

252 **6. Modeling of CO_2 net ecosystem exchange**
253 The net ecosystem exchange (NEE) of CO_2 is defined as the product of the gross
254 primary production (GPP) and the ecosystem respiration (R_{ECO}) (Chapin et al., 2006).
255

256 $$NEE = GPP + R_{ECO} \tag{1}$$

257

258 In the present study we followed the atmospheric sign convention in which a positive
259 NEE is defined as a net flux of CO_2 to the atmosphere (Elsgaard et al., 2012).
260

261 **7. Modeling of ecosystem respiration**
262 The measured R_{ECO} fluxes are the sum of autotrophic (R_a) and heterotrophic (R_h)
263 respiration. Both compartments are mainly controlled by temperature (Lloyd and Taylor,
264 1994; Tjoelker et al., 2001). For each measurement campaign and plot the dependency
265 between R_{ECO} and temperature was modeled according to Lloyd and Taylor (1994) who
266 developed an Arrhenius type relationship to predict soil respiration rates (Eq. 2).

267 $$R_{ECO} = R_{ref} \cdot e^{E_0 \cdot \left(\frac{1}{T_{ref} - T_0} - \frac{1}{T - T_0} \right)} \tag{2}$$

268 R_{ECO} ecosystem respiration [mg CO_2-C m^{-2} h^{-1}]
269 R_{ref} respiration at the reference temperature [mg CO_2-C m^{-2} h^{-1}]
270 E_0 activation energy [K]
271 T_{ref} reference temperature: 283.15 [K]

272   *T₀*   temperature constant for the start of biological processes: 227.13 [K]
273   *T*   air or soil temperature [K]
274

275   In response to the phenological stage of the plants, management activities or changing

276   soil moisture conditions, the applied temperature as explanatory variable could change

277   during the year. Therefore, the R_ECO model was fitted to the appropriate temperature

278   type (air temperature in 20 cm or soil temperature in –2, –5 or –10 cm) which showed

279   the best explanatory power for R_ECO. In case that the temperature span was too small

280   for model building (e.g. winter time, snow cover) or a significant relationship between the

281   temperature and R_ECO could not be observed (e.g. after ploughing), an average CO_2 flux

282   was calculated for the measurement campaign. Annul sums of R_ECO were calculated by

283   summing 0.5 hourly R_ECO fluxes recalculated from Eq. (2), based on the linear

284   interpolated parameters R_{ref} and E_0 of two consecutive measurement campaigns and the

285   continuous time series of air and soil temperatures (Elsgaard et al., 2012). In case of

286   management events (e.g. harvesting, plugging, etc.) or snow cover, R_{ref} and E_0 were kept

287   constant from the previous measurement campaign until the management date. After

288   the management, parameters were taken from the subsequent measurement campaign

289   (Leiber-Sauheitl et al., 2013). However, in case of harvesting at the grassland plots,

290   estimated parameters were linearly interpolated over this period. Estimated parameters

291   and used temperatures for R_ECO are shown in Table S1 to S8.

292

293   **8.Modeling of gross primary production**

294   We estimated GPP as the product of measured NEE minus modeled R_ECO at the same

295   time step, since it is not possible to determine GPP through measurements. The

296   relationship between GPP and PAR was modeled by a Michaelis-Menten type

297   rectangular hyperbolic function proposed by Falge et al (2001) (Eq. 3).

298    $$GPP = \frac{\alpha \cdot PAR}{\left(1 - \left(\frac{PAR}{2000}\right) + \left(\frac{\alpha \cdot PAR}{GPP_{2000}}\right)\right)} \qquad (3)$$

299   *GPP*   gross primary production [mg CO_2-C m^{-2} h^{-1}]
300   α   initial slope of the curve; light use efficiency [mg CO_2-C m^{-2} h^{-1}/μmol m^{-2} s^{-1}]
301   *PAR*   photon flux density of the photosynthetic active radiation [μmol m^{-2} s^{-1}]
302   *GPP₂₀₀₀*   gross primary production at PAR 2000 [mg CO_2-C m^{-2} h^{-1}]
303

304   Prior to modeling GPP, we corrected the plot specific PAR values since the acrylic glass

305   of the transparent chambers absorbed at least 5% of the incoming radiation (PS-plasitc,

306   Eching, Germany) (Leiber-Sauheitl et al., 2014). Annual sums of *GPP* were calculated

307   based on the linear interpolation of α and *GPP$_{2000}$* between two consecutive

308   measurement campaigns and the continuous time series of the PAR (Drösler, 2005;

309   Elsgaard et al., 2012). In case of management events (e.g. harvesting, plugging, etc.) α

310   and *GPP$_{2000}$* were kept constant from the preceding measurement until the management

311   time and were set to zero at the 0.5 hour time step during the working process.

312   Thereafter, parameters were immediately linearly interpolated from the subsequent

313   measurement campaign for the grassland plots. For the arable land plots, parameter

314   interpolation started after the establishment of the seed. Estimated parameters are

315   shown in Table S1 to S8 in the Supplement.

316

317   **9. Model evaluation and uncertainties analysis**

318   For R$_{ECO}$ and NEE model evaluation, we used Pearson's correlation coefficient (*r*), Nash-

319   Sutcliffe efficiency (NSE) (Nash and Sutcliffe, 1970), percent bias (PBIAS) and the ratio

320   of the root mean square error to the standard deviation of measured data (RSR) (Moriasi

321   et al., 2007). According to Moriasi et al. (2007) model simulation can be judged as

322   satisfactory if NSE > 0.50 and RSR ≤ 0.70. For PBIAS, the optimal value is 0.0, with low-

323   magnitude values indicating accurate model simulation. Additionally, positive PBIAS

324   values indicate model underestimation bias, and negative values indicate model

325   overestimation bias (Gupta et al., 1999; Moriasi et al., 2007). To account for the

326   uncertainties in annual R$_{ECO}$ and annual GPP modeling, annual sums from the upper

327   and lower limits of the determined parameters (R$_{ref}$, E$_0$, α, GPP$_{2000}$), based on their

328   standard errors (SE) were estimated (Drösler, 2005; Elsgaard et al., 2012). However,

329   quantifying total model uncertainties is challenging because of the multiple sources of

330   errors (Beetz et al., 2013) and due to a lack of independent data for gap-filling verification.

331   The main uncertainty in the present study may derive from management activities where

332   no additional measurements were conducted and parameters were kept constant (e.g.

333   *R$_{ref}$* and *E$_0$* at the grassland) or set to zero (e.g. α and *GPP$_{2000}$* at the grassland).

334

335   **10. Estimation of NECB and GWP**

336   A simple net ecosystem carbon balance (NECB) was calculated for each plot based on

337   the NEE, the carbon export of harvested phytomass, the carbon input through organic

338   fertilizer application and the cumulative annual CH_4 exchange (Elsgaard et al., 2012;

339   Beetz et al., 2013).

340   To assess the global warming potential (GWP) from the different plots the net emissions

341   of carbon equivalents of NECB and N_2O were summed according to Beetz et al (2013).

342   For the conversion of CH_4 and N_2O to CO_2 equivalents, radiative forcing factors of 25

343   and 298 were used (Forster et al., 2007).

344

345   **11. Statistical analyses**

346   Statistical analyses were conducted using R 3.0.1 (R Development Core Team, 2013).

347   The assumption of normality of residuals was tested using the Lilliefors or Shapiro-Wilk

348   test and by plotting the Quantile-Quantile plots. Homogeneity of variances of residuals

349   was checked using the Levene or Breusch-Pagan test and by plotting the residuals

350   against the fitted values. Where necessary, data were box-cox transformed prior to

351   analyses. For the comparison of cumulative modeled GPP, R_{ECO} and NEE as well as for

352   annual yields and N_{min} values we used a two-factorial ANOVA with land-use and soil

353   type as fixed effects (including an interaction term in the model), neglecting the individual

354   plot specific standard error for modeled CO_2 values. Non-significant terms were removed

355   from the model structure. In case of significant differences among the means, we used

356   Tukey's honest significant differences (TukeyHSD). For GW level we used the

357   nonparametric Kruskal-Wallis Rank Sum test and the non-parametric Pairwise Wilcoxon

358   Rank Sum test with Bonferroni correction for multiple comparisons. For testing two

359   independent sample means regarding the two investigated years 2010 and 2011, we

360   use the Welch two sample t-test (C_{org} contents, bulk density, yields) or the non

361   parametric Mann-Whitney U-test (for N_{min}). Due to temporal pseudoreplication of time

362   series data (N_2O, CH_4 field measurements) we applied linear mixed effects models

363   (Crawley 2007; Hahn-Schöfl et al., 2011; Eickenscheidt et al., 2014a and 2014b). For

364   N_2O fluxes we set up a basic model with land-use type and soil type as fixed effects and

365   the spatial replication (individual plot) nested in time as random effect. We extended the

366   basic model by a variance function due to observed heteroscedasticity. Furthermore,

367   N_2O fluxes showed significant serial correlation. To take this into account, a first-order

368   temporal autoregressive function was included in the model. Autocorrelation was tested

369   using the Durbin-Watson test and by plotting the empirical autocorrelation structure. The

370   model extension was proved by the Akaike Information Criterion (AIC). For multiple

371   comparisons we conducted Tukey contrasts using the General Linear Hypotheses

372   function from the "multcomp" package (Hothorn et al., 2013). CH_4 fluxes did not satisfy

373   the necessary requirements for the linear mixed effects model therefore CH_4 analysis

374 were restricted to the nonparametric Mann-Whitney U-test. We accepted significant

375 differences if $P \leq 0.05$. Results in the text are given as means ± 1 standard deviation.

376

12.Results

13.Environmental variables

379 Temperatures between the two investigated land-use types and soil types did not differ

380 considerably. In 2010 and 2011, air temperature in 20 cm height ranged from −17.5 to

381 39.5℃ with an annual mean of 8.6℃ at both grassla nd sites in 2011. Soil temperature

382 in −2 cm soil depth averaged 10.3℃ at the GC_{medium} site and 10.5℃ at the GC_{high} site

383 in 2011. At the arable land air temperature in 20 cm height ranged from −15.0 to 39.5℃

384 in 2010 and 2011, with an annual mean of 8.8℃ AC_{medium} and 8.7℃ AC_{high} in 2011. Soil

385 temperature in −2 cm soil depth averaged 10.1℃ at both arable land sites in 2011.

386 Longer periods of snow cover occurred in the period 1st of January to 12th of March 2010,

387 28th of November 2010 to 10th of January 2011 and from 24th of January to 5th of

388 February 2011 (see also Fig. 4 and Fig. 5). In 2011, the annual sum of PAR was 17%

389 higher compared to the year 2010. Annual precipitation amounted to 850 mm (2010)

390 and 841 mm (2011) in the investigated period, which was slightly above the 30-years

391 mean of the period 1961–1990. Mean annual groundwater levels of the C_{high} sites were

392 significantly higher (all $P < 0.001$) compared to the C_{medium} sites in 2010 and 2011

393 (Table1). Furthermore the GW level at the arable sites were significantly higher (all $P <$

394 0.001) compared to the grassland sites in both investigated years. Longer periods of

395 flooding and water saturation were only observed at the AC_{high} sites for the period from

396 1st to 17th June 2010.

397

14.Soil properties and mineral nitrogen contents

399 Total organic carbon contents and bulk density in the 0–10 cm and 10–20 cm soil layers

400 significantly (all $P < 0.01$) differed between the two soil types investigated (Table 1). At

401 the grassland sites pH values in the 0–20 cm soil layer were approximately one decade

402 lower compared to the arable land (Table 1). Observed C/N ratios at the soil depth 0–20

403 cm were between 10 and 12 (Table 1), indicating nitrogen-rich conditions at all plots.

404 Extractable N_{min} contents of the soils ranged from 1 to 178 mg N kg^{-1} at the arable sites

405 and from 2 to 115 mg N kg^{-1} at the grassland sites (Fig. 2, Fig 3). In both years, the N_{min}

406 contents at the grassland sites significantly ($P < 0.001$) exceeded those from the arable

407 site (Fig. 3). Furthermore the N_{min} contents of the C_{high} sites were significantly ($P < 0.01$)

408   higher compared to the C_{medium} sites (Fig. 3), but this was not valid considering the arable

409   land separately. Slightly higher N_{min} contents were found at the soil depth 10–20 cm

410   compared to the soil depth 0–10 cm, but differences were only significant for the

411   grassland sites ($P < 0.05$). In both years, N_{min} was mainly dominated by NO_3^-, whereas

412   NH_4^+ was only of minor importance. However, at the AC_{high} sites the proportion of NO_3^-

413   in the soil depth 0–10 cm was lower (approximately 80%) compared to the AC_{medium} sites

414   (approximately 97%), whereas at the grassland sites no differences were found between

415   the two soil types investigated (91-95%).

416

417   **15. Biomass yield**

418   The mean annual crop yield ranged from 1.2 ± 0.5 to 10.2 ± 1.6 t DM ha^{-1} yr^{-1} at the

419   arable land and from 6.2 ± 0.7 to 13.1 ± 2.9 t DM ha^{-1} yr^{-1} at the grassland in 2010 and

420   2011 (see also Eickenscheidt et al., 2014b) (Table 3). For both land-use types the crop

421   yield was significatly ($P < 0.01$) lower in the year 2010 compared to the year 2011 (73%

422   at the A sites and 52% at the G sites). However, it has to be taken into consideration

423   that at the grassland sites three instead of two cuts were carried out in 2011. At the

424   arable land a longer period with partially flooding and high water saturation damaged or

425   partly killed the maiz seedlings as well as the oat plants in June 2010, especially at the

426   C_{high} sites. Furthermore, in 2010 the entire plants were harvested and used as silo maize

427   or oat corn and straw respectively, whereas in 2011 only the grains were harvested

428   regarding both management practices and the remaining plants were left on the field. In

429   both years investigated, the yield from the grassland sites significantly exceeded those

430   from the arable land (all $P < 0.001$), whereas no significant differences were found

431   between the two soil types observed.

432

433   **16. CO_2 fluxes**

434   The modeling showed that the air temperature in 20 cm above soil surface and soil

435   temperature in –2 cm are the main drivers of R_{ECO} in the present study, while soil

436   temperatures in –5 cm and –10 cm soil depth mostly showed distinctly weaker

437   correlations (Table S1-S8 in the Supplement). At the arable land, 88% of the calculated

438   models based on T_{air}, and only 12% on ST_2, whereas at the grassland sites 54% of the

439   models based on T_{air} and 39% on ST_2. Model evaluation statistics from observed R_{ECO}

440   versus modeled R_{ECO} generally revealed a good model performance with a slight

441   tendency of model overestimation bias for the year 2010 (mean PBIAS –2.39).

442   Pearson's correlations coefficients for observed R_{ECO} versus modeled R_{ECO} ranged

443   between 0.89 and 0.98, NSE values ranged from 0.70 to 0.97 and RSR values were ≤

444   0.55 (Table 4, Fig. S3–S4 in the Supplement). According to the annual temperature trend,

445   R_{ECO} showed a clear seasonality with maximum flux rates during the summer time. In

446   2010, highest daily R_{ECO} fluxes of up to 41 g CO_2-C m^{-2} d^{-1} were modeled at the

447    $A2C_{medium}$ (oat) and $G1C_{medium}$ plot, whereas in 2011, distinctly lower maximum daily

448   R_{ECO} fluxes of up to 28 g CO_2-C m^{-2} d^{-1} and 32 g CO_2-C m^{-2} d^{-1} were modeled for the

449    $A2C_{high}$ (maize) plot and the $G2C_{high}$ plot, respectively (Fig. 4 and 5). At the grassland

450   sites, annual sums of modeled R_{ECO} ranged from 3521 ± 1041 to 4316 ± 562 g CO_2-C

451   m^{-2} yr^{-1}, which was significantly ($P < 0.001$) higher compared to the arable sites where

452   R_{ECO} ranged from 2012 ± 284 to 2992 ± 230 g CO_2-C m^{-2} yr^{-1} (Table 3, Fig. 6a).

453   Differences in R_{ECO} between the two soil types investigated were only small and not

454   significantly different (Fig.6a).

455   Like R_{ECO}, GPP showed a clear seasonal trend with increasing CO_2 uptake capacity with

456   increasing PAR intensity in summer time. In 2010, highest maximum daily GPP of up to

457   −25 g CO_2-C m^{-2} d^{-1} were modeled for the arable land (maize, C_{medium}) and up to −20 g

458   CO_2-C m^{-2} d^{-1} for the grassland ($G2C_{high}$), whereas in 2011, distinctly higher GPP values

459   up to −35 g CO_2-C m^{-2} d^{-1} were modeled for both maize plots and up to −28 g CO_2-C

460   m^{-2} d^{-1} for the $G2C_{high}$ plot (Fig. 4 and 5). At the grassland sites annual sums of GPP

461   ranged between −2093 ± 152 and −2962 ± 178 g CO_2-C m^{-2} yr^{-1}, which was significantly

462   ($P < 0.01$) higher compared to the arable sites where GPP ranged between −873 ± 110

463   and −2360 ± 237 g CO_2-C m^{-2} yr^{-1} (Table 3, Fig 6b). Differences in GPP between the

464   two soil types were not significant.

465   Calculated NEE were in good agreement with observed NEE (Fig. S1 and S2 in the

466   Supplement). Nevertheless, the calculated percent bias revealed a tendency of model

467   overestimation for both years (mean PBIAS −7.5 in 2010 and −6.1 in 2011). Pearson's

468   correlations coefficients for observed NEE versus calculated NEE ranged from 0.79 to

469   0.98, NSE values ranged from 0.61 to 0.96 (Table 5). The mean RSR values was 0.36.

470   Annual NEE significantly ($P < 0.01$) differed between the two land-use types with lower

471   NEE values at the arable sites, ranging from −6 ± 546 to 1707 ± 619 g CO_2-C m^{-2} yr^{-1},

472   compared to the grassland sites were NEE ranged from 1354 ± 740 to 1823 ± 851 g

473   CO_2-C m^{-2} yr^{-1} (Table 3, Fig 6c). Differences between the two soil types were not

474   significant for NEE.

475

17.N$_2$O and CH$_4$ fluxes

Nitrous oxide emissions were generally low at all plots (Fig 7). N$_2$O fluxes rarely exceeded 50 µg N m^{-2} h^{-1}. However, single N$_2$O peaks with maximum flux rates of up to 2832 µg N m^{-2} h^{-1} were detected at the 3rd June at both maize plots as well as at the 6th of September at both oat plots with maximum flux rates of up to 289 µg N m^{-2} h^{-1}. At the grassland sites, highest N$_2$O fluxes of up to 992 µg N m^{-2} h^{-1} were found immediately after fertilizer application (see Eickenscheidt et al., 2014b). In general, N$_2$O fluxes from the arable sites were significantly ($P < 0.01$) higher compared to the grassland sites (Fig. 8a). Furthermore, N$_2$O fluxes from the C$_{high}$ sites significantly ($P < 0.05$) exceeded N$_2$O fluxes from the C$_{medium}$ sites, but this was not valid considering the arable land separately. Significant differences within the land-use types, regarding N$_2$O flux rates, were only found between the grassland plots, where the application of biogas digestate significantly ($P < 0.01$) enhanced the N$_2$O fluxes compared to the application of cattle slurry (see Eickenscheidt et al., 2014b). At the arable land distinctly different N$_2$O flux rates between maize and oat were not found, but the single peak emissions observed led to significantly ($P < 0.01$) higher annual cumulative N$_2$O emissions at the maize plots (Table 3, Fig 8a). N$_2$O peaks accounted for 75% and 87% of the annual N$_2$O balances at the maize plots, whereas at the oat plots peaks account for 63% and 54% of the annual N$_2$O sums (at C$_{medium}$ and C$_{high}$, respectively). Annual cumulative N$_2$O emissions ranged from 0.12 ± 0.01 g N m^{-2} yr^{-1} (G1C$_{medium}$) to 0.86 ± 0.21 g N m^{-2} yr^{-1} (A2C$_{high}$) (Table 3).

Most of the time, all sites showed a weak uptake of CH$_4$ or zero fluxes. CH$_4$ peaks up to 173 µg C m^{-2} h^{-1} were occasionally found immediately after fertilization at the G1 sites (see Eickenscheidt et al., 2014b). Moreover, a high CH$_4$ peak event of up to 2177 µg C m^{-2} h^{-1} occurred on the 14th of July 2011 at the oat plots. Generally, CH$_4$ fluxes of the arable sites significantly ($P < 0.01$) exceeded CH$_4$ fluxes of the grassland sites, whereas no differences were found between the two soil types investigated (Fig 7, and 8b). Significantly different CH$_4$ fluxes within the land-use types could not be observed regarding the annual fluxes in 2011. However, considering the annual cumulative exchange rates, CH$_4$ emissions of the oat plots significantly ($P < 0.05$) exceeded those of the maize plots. The observed weak CH$_4$ emissions or uptakes amounted to cumulative annual CH$_4$ exchange rates ranging between –0.11 ± 0.05 g C m^{-2} yr^{-1} (G2C$_{medium}$) and 0.51 ± 0.17 g C m^{-2} yr^{-1} (A1C$_{medium}$) (Table 3). However, as previously mentioned for N$_2$O, the single CH$_4$ peak event observed at the arable sites entirely

510 controls the cumulative sum of CH_4 and turns the plots from a sink into a source of CH_4.
511

18.NECB and GWP

513 Including the C export from harvested phytomass, C import from fertilization and CH_4
514 exchange to NEE, calculated NECB ranged from 451 ± 617 to 1894 ± 872 g C m^{-2} yr^{-1}.
515 Estimated GWP's ranged from 1837 ± 2293 to 7095 ± 3243 g $CO_{2eq.}$ m^{-2} yr^{-1}, revealing
516 a very high release of greenhouse gases from all plots (Table 6). However, CO_2
517 dominated the GWP of all plot to nearly 100% (range between 97–99% and for maize
518 86–90%), whereas the contribution of N_2O and CH_4 were almost negligible, with
519 exception of the maize plots.
520

19.Discussion

20.Magnitude of GHG fluxes

523 The observed annual CO_2 emissions were in the upper range or partly higher than CO_2
524 exchange rates reported in the literature from temperate drained arable lands (e.g.
525 Maljanen et al., 2001 and 2007; Grønlund et al., 2008; Höper et al., 2008; Maljanen et
526 al., 2010; Leifeld et al., 2011; Elsgaard et al., 2012; Drösler et al., 2013) and grasslands
527 (e.g. Maljanen et al., 2001; Grønlund et al., 2006 and 2008; Maljanen et al., 2010;
528 Elsgaard et al., 2012; Beetz et al., 2013; Drösler et al., 2013; Leifeld et al., 2014; Renou-
529 Wilson et al., 2014). No differences in the CO_2 release of the C_{medium} and C_{high} sites were
530 found in the current study, and no information about CO_2 fluxes of comparable soils to
531 those of the C_{medium} sites were available in the literature. Observed CO_2 emissions from
532 the arable land were in the range or partly doubled (4.51–12.04 t CO_2-C ha yr^{-1}) the
533 IPCC default emission factor from the Tier 1 approach for drained boreal and temperate
534 arable lands (7.9 t CO_2-C ha yr^{-1}; IPCC, 2014) whereas more than three times higher
535 CO_2 emissions were observed at the grassland sites (15.81–18.94 t CO_2-C ha yr^{-1})
536 compared to the IPCC default emission factor for deep-drained temperate grasslands
537 (6.1 t CO_2-C ha yr^{-1}; IPCC, 2014). However, comparison of CO_2 exchange rates is
538 difficult since annual variability is very high. For example Leifeld et al. (2014) reported
539 that the NECB of a temperate grassland in Germany ranged from 0.98 to 19.46 t C ha^{-1}
540 yr^{-1}, with a five year mean of 9.06 ± 6.64 t C ha^{-1} yr^{-1}. In this study the highest value
541 was observed for the period 2010 to 2011 which was in good agreement with the values
542 estimated by us during this period. The finding is also in line with Kasimir-Klemendtsson
543 et al. (1997), who reported net CO_2 exchange rates ranging from 2.2 to 31.3 t C ha^{-1} yr^{-1}

544   for farmed organic soils, demonstrating the high bandwidth of measured CO_2-balances.

545   Observed cumulative annual N_2O emissions were distinctly lower than the default

546   emission factor from the Tier 1 approach for boreal and temperate, drained arable land

547   (13 kg N_2O-N ha^{-1} yr^{-1}; IPCC. 2014) and for temperate deep drained, nutrient rich

548   grassland (8.2 kg N_2O-N ha^{-1} yr^{-1}; IPCC. 2014). In line with this, several other authors

549   reported much higher N_2O emissions from organic soils ranging from 0 to 61 kg N_2O-N

550   ha^{-1} yr^{-1} for arable lands (Kasimir-Klemendtsson et al., 1997; Augustin et al., 1998;

551   Flessa et al., 1998; Petersen et al., 2012; Drösler et al., 2013) and ranging from 1.15 to

552   41 kg N_2O-N ha^{-1} yr^{-1} for grasslands (Velthof et al., 1996; Augustin et al., 1998; Flessa

553   et al., 1997 and 1998; van Beek et al., 2010 and 2011; Kroon et al., 2010; Petersen et

554   al., 2012; Beetz et al., 2013; Drösler et al., 2013).

555   As expected, observed CH_4 fluxes from all plots were low, which is in line with generally

556   low groundwater levels and the absence of aerenchymous plant species which can

557   transport CH_4 from an anaerobic layer to the atmosphere, bypassing the oxic zone at

558   the soil surface (Grosse et al., 1992; Svensson & Sundh, 1992; Wahlen, 2005).

559   Cumulative annual CH_4 emissions or uptakes were in the range reported for other deep

560   drained arable lands and grasslands (Maljanen et al., 2010; Petersen et al., 2012; Beetz

561   et al., 2013; Drösler et al., 2013; Renou-Wilson et al., 2014) and fit also well with the

562   IPCC default emission factor for boreal and temperate drained arable land (0 kg CH_4

563   ha^{-1} yr^{-1}; IPCC, 2014). A distinctly higher emission factor however is given by the IPCC

564   for a temperate deep-drained, nutrient-rich grassland (16 kg CH_4 ha^{-1} yr^{-1}; IPCC, 2014)

565   compared to our estimations.

566

567   **21.Uncertainties in GHG fluxes and modeling**

568   Several factors probably influenced the accuracy of estimated CO_2 exchange rates.

569   Firstly, the R$_{ECO}$ model based only on temperature changes disregarding the effect of

570   soil moisture or GW level. Thus changing soil moisture contents or GW levels between

571   two consecutive measurements campaigns were neglected since we assume a linear

572   change in derived model parameters (see also Beetz et al., 2013 and Leiber-Sauheitl et

573   al., 2014). Secondly, management activities like ploughing at the arable sites probably

574   produced peak CO_2 emissions, which we may have missed. Additionally, it can be

575   assumed that after harvesting at the grassland sites, R$_{ECO}$ decreased due to the reduced

576   phytomass. However, additional measurement campaigns to capture this effect did not

577   take place in the current study and no corresponding data were found in the literature.

578   Furthermore, it is well known that the application of organic fertilizers produced short-
579   term CO_2 emission peaks, which were also not sufficiently detected. However, both
580   sources of errors may even have an opposite effect. Thirdly for GPP, linearly
581   interpolation of parameters produced some uncertainties since it can be assumed that
582   plant growth after cutting did not increase linearly (Horrocks and Valentine, 1999; Beetz
583   et al., 2013). However with the available data set, it was not possible to quantify the error
584   by the used interpolation approach of parameters since the dataset was too small for
585   cross validation and no additional measurements for an independent model validation
586   were conducted. In addition, despite high model accuracy, the calculated PBIAS
587   revealed a slightly model overestimation bias for R_{ECO} and NEE for both years (R_{ECO}
588   only in 2010). Thus, modeled R_{ECO} and calculated NEE rates should be considered as
589   a conservative estimation. However, modeled values fit well with values reported in the
590   literature (see Figure 9).
591   Several studies indicated that dissolved organic C can significantly contribute to
592   terrestrial C balances (e.g. Worrall et al., 2009; Dinsmore et al., 2010 Renou-Wilson et
593   al., 2014). Thus, for the calculation of NECB from drained organic soils, fluvial C losses
594   should additionally be considered in future investigations.
595   Observed N_2O fluxes showed a high temporal variability with long periods of low
596   background emissions and a few high peaks, mainly after management activities.
597   Measurement frequency was increased after fertilization at the grassland plots for at
598   least two weeks (see Eickenscheidt et al., 2014b) but due to our regular measurement
599   intervals in the remaining year we cannot rule out that we may have missed high N_2O
600   events driven by changing climate conditions (e.g. drying–rain or freeze–thaw events)
601   and/or management activities, particularly at the arable sites. N_2O peaks are known to
602   last a couple of days up to several weeks (Stolk et al., 2011). Due to our measurement
603   intervals and interpolation approach, observed N_2O and CH_4 peaks distinctly altered the
604   cumulative annual budgets, increasing the overall uncertainties in estimated GHG
605   emissions. However, for future investigations in GHG emissions we strongly advocate
606   the combined use of automatic and manual chamber systems to maintain a higher
607   accuracy of data.
608
609
610   **22. Soil organic carbon effects**
611   With exception of N_2O, significantly different GHG emissions between the two soil types

612   investigated were not found in the present study, although significantly different SOC

613   contents in the upper soil horizon were detected. The observation is in strong contrast

614   to our hypothesis that GHG emissions significantly increase with increasing SOM

615   content (hypothesis i).

616   Regarding CO_2 fluxes, the current findings are however in line with investigations from

617   Leiber-Sauheitl et al. (2014), who reported that CO_2 emissions were not related to

618   different SOM contents in the upper horizon of an extensive grassland in North Germany.

619   Contrary, Veenendaal et al. (2007) and Renou-Wilson et al. (2014) assumed that their

620   different estimated respiration rates for grassland sites were driven by different

621   SOC/SOM contents. However, it can be assumed that not only the SOM content itself

622   acts as a key factor controlling the CO_2 release, but the proportion of SOM which is

623   exposed to mineralization, which in turn is driven by drainage depth. Therefore we

624   calculated the effective C stock as the fraction of aerated carbon in the soil profile

625   according to Leiber-Sauheitl et al. (2014) (Fig. 10). No relationship was found between

626   the effective C stock and the C flux components (expressed as NECB), which was also

627   reported by Leiber-Sauheitl et al. (2014) and Tiemeyer et al. (2014). However, Fig. 10

628   shows that at the grassland sites, C stocks available for mineralization processes are

629   comparable (40–45 kg C m^{-2}), probably explaining the equal CO_2 loss rates from this

630   land-use type. Temperature and soil moisture are considered to be the primary

631   regulators for CO_2 emissions from soils (Silvola et al. 1996; Maljanen et al., 2001; Hardie

632   et al., 2011), since they directly affect microbial activity and the rate of enzymatic

633   processes (Michaelis and Menten, 1913; Tietema et al., 1992). In the present study,

634   temperatures are found to be equal at all sites due to their close proximity, whereas the

635   soil moisture contents significantly differed between the C_{high} and C_{medium} sites mainly

636   according to the GW oscillation. It is well known that the water level height has a strong

637   influence on CO_2 emissions from peatlands as it directly affects the oxygen availability

638   for microbial activity as was reported in several studies (e.g. Silvola et al., 1996; Berglund

639   and Berglund, 2011; Renou-Wilson et al., 2014; Leiber-Sauheitl et al., 2014). Beside

640   abiotic factors substrate chemistry, in particular the SOM quality and its labile and more

641   recalcitrant fractions, are considered to act as key variables affecting the

642   decomposability of SOM and thus controlling CO_2 fluxes from peatlands (Byrne and

643   Farrell, 2005; Heller and Zeitz, 2012; Leifeld et al., 2012). For example, Leifeld et al.

644   (2012) showed that the soil respiration rate of a disturbed temperate peatland was

645   strongly controlled by its polysaccharides content, particularly the O-alkyl-C content was

646 found to be a useful proxy for respiration rates. SOM quality was not examined in our
647 study, but both soil types at all plots investigated exhibited highly decomposed organic
648 material (H10, according to Von Post's humification scale; N. Roßkopf personal
649 communication, 2013). This is typical for organic soils which have been drained and
650 intensively managed for a long time, and is in line with Leifeld et al. (2012), who found
651 that organic matter quality declines with ongoing decomposition, resulting in low
652 polysaccharides contents and a lower sensitivity for heterotrophic metabolism.
653 Nevertheless, observed NECB revealed very high C loss rates from the SOC pool.
654 Leifeld et al. (2014) suggested that intensive management, drainage and changed
655 climate drivers accelerate peat decomposition today, and therefore outweighed
656 declining peat quality. Additionally, Reiche et al. (2010) reported that the degree of
657 humification is not suitable for the prediction of CO_2 and CH_4 fluxes from anaerobic
658 decomposition, which stands in contrast to assumptions made by Glatzel et al. (2004).
659 However, observed equal narrow C/N ratios (10–12) in the upper soil reveal firstly a high
660 organic matter quality, easily to mineralize, and secondly comparable SOM qualities at
661 all plots, probably explaining why no significantly different C loss rates between the two
662 different soil types were found in the present study.
663 In line with CO_2, CH_4 fluxes were also not different between the two soil types
664 investigated, but this can mainly be attributed to the intensive drainage and thus soil
665 aeration, which effectively inhibited microbial methanogenesis at the C_{medium} and C_{high}
666 sites. It is known that the availability and quality of organic substrates influences the
667 amount of produced CH_4. Nevertheless, several studies indicate that high CH_4 fluxes in
668 bogs are mainly controlled by labile organic substrates such as root exudates or plant
669 litter and not by bulk peat (Minchin & McNaughton, 1984; Chanton et al., 1995; Bridgham
670 et al., 1998; Wahlen, 2005; Hahn-Schöfl et al., 2011).
671 In contrast to CO_2 and CH_4 fluxes, N_2O fluxes from the C_{high} sites significantly exceeded
672 N_2O fluxes from the C_{medium} sites. This can probably be attributed to the more favorable
673 soil conditions for denitrification, supported by higher N_{min} contents and higher
674 groundwater levels at these sites (Eickenscheidt et al., 2014b). In both years N_{min} was
675 mainly dominated by NO_3^-, demonstrating that net nitrification entirely controls net
676 nitrogen mineralization at all plots. Thus, nitrification provided the substrate for
677 denitrification and additionally, may itself have contributed to N_2O production. In general,
678 N_2O production processes are various and can occur simultaneously within close
679 proximity (Davidson et al., 1986; Butterbach-Bahl et al., 2013). Both nitrification as well

680   as denitrification depending on the availability of labile organic compounds as C and/or

681   energy source (Butterbach-Bahl et al., 2013). However, for denitrification the actual

682   regulation by C is currently not yet understood (Baggs and Philippot, 2011), but it can

683   be assumed that sufficient metabolizable C was widely available at all plots investigated.

684

685   **23.Land-use and management effects**

686   At peatlands GW level and land-use type are closely linked. From a meta-analysis of 53

687   German peatlands Tiemeyer et al. (2013) found that the mean annual GW level was

688   lower for arable land than for intensive grassland with median GW levels of

689   approximately −70 and −37 cm above soil surface. The GW levels observed in our study

690   were on average lower at the arable land and higher at the grassland compared with the

691   average of the meta-analysis. In general, intensive farming at peatlands presupposes

692   low GW levels, since most of the arable crops are not adapted to low oxygen contents

693   in the rhizosphere as could be seen in the present study, were the temporarily high GW

694   level caused plant damage and yield losses at the arable sites in 2010. The effect of

695   reduced biomass productivity due to high GW levels which inhibited photosynthesis by

696   slowing the rate of gas diffusion through the vegetation (Lohila, 2008) was also reported

697   by Renou-Wilson et al. (2014). Both annual sums of GPP as well as yields were in good

698   agreement with those reported from other peatlands as can be seen in Figure 9.

699   Statistical analysis revealed significantly higher yields at the grassland sites compared

700   to the arable sites, but it has to be taken into account that at the arable sites only the

701   grains were harvested in 2011 and up to 3.84 t DM ha^{-1} and 9.05 t DM ha^{-1} remained

702   on the field regarding the oat and maize plots, respectively. Due to the continuous plant

703   cover over the whole year at the grassland plots annual sums of GPP were significantly

704   higher at these plots compared to the arable plots in 2010 as well as in 2011.

705   As GPP, modeled annual sums of R_{ECO} significantly differed between the two land-use

706   types with distinctly higher R_{ECO} values at the grassland sites. As mentioned above, R_{ECO}

707   is strongly controlled by temperature since it stimulates both R_a and R_h, as can be seen

708   in the pronounced seasonality of R_{ECO}. From the model fits it can be concluded that the

709   more frequent model adaptation with T_{air} (88%) reveals a higher share of R_a at the arable

710   site compared to the grassland sites. At the later, approximately 40% of the R_{ECO} models

711   based on ST_2, perhaps demonstrating a more balanced ratio of R_a and R_h. Nevertheless,

712   the proportion of the different respiration compartments of R_{ECO} is unknown, but Silvola

713   et al. (1996) reported that root-derived respiration from grasslands established on

714   peatland accounted for 35–45% of total soil respiration. Furthermore, Maljanen et al.

715   (2001) found that root-associated respiration on grasslands was distinctly higher

716   compared to arable lands. However, the significantly higher R_{ECO} at the grassland sites

717   can firstly perhaps be related to the higher biomass production at these sites, because

718   a higher GPP also results in higher above- and below-ground autotrophic respiration

719   (Leiber-Sauheitl et al., 2014; Renou-Wilson et al., 2014). Moreover, the increased

720   transport of photosynthates to the plant rhizosphere due to the higher GPP may favor

721   bacterial metabolism through increased root exudates (Mounier et al., 2004; Henry et

722   al., 2008; Sey et al., 2010), additionally enhancing R_h. Secondly, the organic fertilizer

723   application at the grassland plots stimulates microbial growth and thus SOM

724   mineralization (Gutser et al., 2005; Jones et al., 2007). Additionally, a large part of the

725   C from the organic fertilizer will quickly be metabolized to CO_2 (Vuichard et al., 2007).

726   Several authors (see e.g. Dao, 1998; Maljanen et al., 2010) reported that regularly

727   ploughed and fertilized arable lands are larger sources of CO_2 than non-tilled arable land

728   soils or grasslands, due to aerating and mixing of crop residues into the soil. However,

729   in the current study the effect of management is difficult to capture.

730   Despite of higher modeled GPP values, the distinctly higher modeled R_{ECO} values led to

731   significantly higher calculated NEE values at the grassland sites compared to the arable

732   sites. With the exception of the maize plot at the C_{medium} site in the year 2011, all plots

733   show positive NEE balances in both years investigated, as expected for drained organic

734   soils and as commonly reported in the literature (e.g. Maljanen et al., 2001; Grønlund et

735   al., 2006 and 2008; Maljanen et al., 2010; Elsgaard et al., 2012; Beetz et al., 2013;

736   Drösler et al., 2013). However, the huge CO_2 uptake capacity during the short growth

737   period of the maize plants, compensates for the soil CO_2 release due to microbial

738   decomposition of organic matter at least in the year 2010. Nevertheless, as seen in the

739   NECB, the C export also reversed the maize cultivation on the C_{medium} site to a C source.

740   Previous studies of annual NEE from maize on organic soils are rare in literature, but

741   our results are in line with Drösler et al. (2013) who reported NEE values ranging from

742   -216.2 to 443.8 g C m^{-2} yr^{-1}. As mentioned above, it has to be take into account that in

743   the year 2011 only the grains were harvested at all arable plots. Assuming that silage

744   maize would have been produced instead of maize grains or the straw was additionally

745   harvested at the oat plots, NECB would partly be doubled and more comparable to

746   calculated grassland values.

747   According to Maljanen et al. (2010) the better aeration of regularly ploughed arable land

748   leads to a larger sink of atmospheric CH_4 compared to permanent grasslands. This

749   contrasted our results, where the CH_4 fluxes from the arable plots significantly exceeded

750   CH_4 fluxes from the grassland plots. However, all measured CH_4 fluxes were very low

751   and CH_4 emissions and uptakes were almost negligible in the NECB of the plots, as was

752   also reported by several other authors for drained organic soils (e.g. Maljanen et al.,

753   2010; Petersen et al., 2012; Schäfer et al., 2012; Drösler et al., 2013; Renou-Wilson et

754   al., 2014). Moreover, the C import through fertilization contributed only marginally (3–

755   14%) to the NECB of the grassland plots.

756   In the course of the present study, fertilization was found to enhance N_2O fluxes at the

757   grassland sites, where the application of biogas digestate led to significantly higher N_2O

758   emissions compared to cattle slurry application (for further discussion see Eickenscheidt

759   et al., 2014b). From a meta-study of European organic soils Leppelt et al. (2014) found

760   that the amount of N fertilizer was directly linked to N_2O fluxes from grasslands, whereas

761   no significant relationship between N fertilization and N_2O fluxes from arable lands were

762   found. Nevertheless, N_2O fluxes from the arable plots significantly exceeded those of

763   the grassland sites, as was also reported by Maljanen et al. (2007 and 2010) and

764   Petersen et al. (2012) and additionally confirmed by Leppelt et al. (2014) for European

765   organic soils. Observed N_2O peaks at the arable sites can be related to harvesting and/or

766   several consecutive tillage steps (e.g. ploughing, milling, mattocking) in the previous

767   weeks. This is in line with Silvan et al. (2005) who supposed that higher N_2O fluxes from

768   arable lands are related to the higher N availability for microbial denitrification in the

769   absence of plants. No fertilizer was applied at the arable plots, which is also reflected in

770   the significantly lower N_{min} contents and perhaps higher pH values compared to the

771   grassland plots. However, it is well known that drainage and intensive management

772   enhanced the degradation of SOM and thus stimulates net nitrogen mineralization and

773   nitrogen transformation processes (Kasimir Klemedtsson et al., 1997; Freibauer et al.,

774   2004; Goldberg et al., 2010). Several authors reported an annual N supply through peat

775   mineralization of approximately 70–425 kg N ha^{-1} yr^{-1} (Schothorst, 1977; Flessa et al.,

776   1998; Sonneveld and Lantinga, 2011; Leppelt et al., 2014). Taking into account the

777   calculated soil carbon losses and plot specific C/N ratios of the upper soil/peat layer,

778   estimated SOM mineralization leads to an annual N supply of approximately 451–1720

779   kg N ha^{-1} yr^{-1}. This estimation seems very high but regardless of the high uncertainties

780   it clearly indicates that sufficient N must be available for nitrification and denitrification,

781   independent of fertilizer application as previously assumed by Leppelt et al. (2014).

782   Furthermore, the admixture of *Vicia sativa* or *Vicia faba minor*, both N_2 fixing
783   leguminoses further increase the soil N_{min} pool of the arable sites through the release of
784   N-rich root exudates (Rochette et al. 2004; Sey et al., 2010) as well as their incorporation
785   into the soil, albeit to an unknown extent.

786   In conclusion, taking together estimated GHG emissions, calculated GWPs clearly differ
787   between the two land-use types investigated with distinctly higher GWP's observed at
788   the grassland plots compared to the arable land. However, all plots show a very high
789   release of GHGs, demonstrating the unsustainable agricultural use of drained organic
790   soils and the current need for the implementation of mitigation strategies and restoration
791   measures. We hypothesized that GHG emissions from arable soils exceed GHG
792   emissions from intensively managed grassland soils. The contrary was found in the
793   present study; therefore we have to reject hypothesis ii. However, from the present
794   results it can be concluded that mainly the management and not the land-use type itself
795   or the SOC content is responsible for the amount of released GHGs from intensive
796   farming on drained organic soils.

797

798   **24.Implications for the climate reporting under LULUCF/AFOLU**

799   For the climate reporting under LULUCF/AFOLU, the IPCC guidelines consider GHG
800   emissions from peat soils having at least ≥10 cm thickness of the soil/peat layer and a
801   C_{org} content of ≥12% in case of a soil thickness ≤20 cm. However, the intensive
802   cultivation of organic soils leads to a continuous decrease in the amount of SOM and
803   thus the area of soils which fulfil the requirements of the IPCC guidelines for organic
804   soils rapidly declined in the last decades. For example Nielsen et al. (2012) reported an
805   average annual decrease of organic soils of approximately 1400 ha in Denmark since
806   1975. The remaining soils often contain >6% C_{org} and not the required >12% (Nielsen et
807   al., 2012). Contrary to mineral soils or natural peatlands in equilibrium, Nielsen et al.
808   (2012) assume that drained and managed soils having >6% C_{org} will evidently lose
809   carbon until a new equilibrium is reached. Since no data was available in literature for
810   those soils, Nielsen et al. (2012) decided to allocate an fixed emission factor half of the
811   amount of what was measured for soils having >12% C_{org} to account for these losses in
812   the Danish greenhouse gas inventory. However, despite being subject to high
813   uncertainties, our results reveal that the GHG emission potential of soils intermediate
814   between mineral and organic soils can be as high or partly higher as for typical drained
815   organic soils under intensive agricultural use. This is in line with observations from

816   Leiber-Sauheitl et al. (2013) for extensive grasslands. To avoid a significant
817   underestimation of GHG emissions in the LULUCF/AFOLU sector, there is a
818   corresponding need to adjust the IPCC guidelines for drained inland organic soils
819   accordingly. The new 2013 Supplement to the IPCC guidelines for national GHG
820   inventories on wetlands distinguishes several emission factors for different land-use
821   types, climate regions, nutrient status and drainage intensities (IPCC, 2014). We
822   suggest establishing a further category which provides emission factors for different
823   land-use types on former drained peatlands or associated organic soils, which do not
824   fulfil the necessary requirements of typical organic soils but also contain high amounts
825   of C_{org}. To define reliable emissions factors for those soils further investigations
826   regarding their potential to release GHGs are needed. Furthermore, it has to be clarified
827   to what extent the composition of the SOM is responsible for the magnitude of GHG
828   release from drained organic soils.

829

25. Conclusion

831   This study presents estimations of GHG fluxes from arable lands and intensive
832   grasslands on sapric Histosol and mollic Gleysol, which significantly differed in their SOC
833   content in the top soil. The present results clearly revealed that like typical drained
834   peatlands also drained mollic Gleysols can be considered as hotspots for GHG
835   emissions, provided that they are intensively managed as arable land or grassland.
836   However, observed GHG fluxes revealed a very high sensitivity against changing key
837   factors like climate variables (e.g. temperature, precipitation) and management.
838   Estimated GHG emission factors partly more than doubled the emission factor of the
839   Tier 1 approach of the IPCC independent of the SOC content in the topsoil. Thus former
840   peatlands and associated organic soils, containing <12% C_{org} should be integrated in
841   the national GHG emission inventories to avoid a significant underestimation in the
842   climate reporting. Moreover there is a current need to adjust the IPCC guidelines for
843   drained inland organic soils accordingly. Besides climate reporting, the observed very
844   high release of GHGs demonstrates the unsustainable agricultural use of drained
845   organic soils and the current need for rapid implementation of mitigation strategies and
846   restoration measures.

847

26. Acknowledgements

864 # 27.References

865 Ad-Hoc AG Boden: Bodenkundliche Kartieranleitung, 5. Aufl., Schweizerbart'sche
866 Verlagsbuchhandlung, Hannover, 438 pp., 2005.
867

868 Alm, J., Shurpali, N. J., Tuittila, E.-S., Laurila, T., Maljanen, M., Saarnio, S., and
869 Minkkinen, K.: Methods for determining emission factors for the use of peat and
870 peatlands – flux measurements and modeling, Boreal Environ. Res., 12, 85–100, 2007.
871

872 Augustin, J., Merbach, W., Steffens, L., and Snelinski, B.: Nitrous Oxide Fluxes of
873 Disturbed Minerotrophic Peatlands, Agribiol. Res., 51 (1), 47–57, 1998.
874

875 Baggs, E.M., and Philippot, L.: Nitrous oxide production in the terrestrial environment,
876 Nitrogen Cycl. In Bacteria: Molecular Analysis, 211–232, Caister Academic Press,
877 England, 2011.
878

879 Beek van, C.L., Pleijter, M., Jacobs, C.M.J., Velthof, G.L., van Groenigen, J.W., and
880 Kuikman, P.J.: Emissions of N_2O from fertilized and grazed grassland on organic soil in
881 relation to groundwater level, Nutr. Cycl. Agroecosyst., 86, 331–340, 2010.
882

883 Beek van, C.L., Pleijter, M., and Kuikman, P.J.: Nitrous oxide emissions from fertilized
884 and unfertilized grasslands on peat soil, Nutr. Cycl. Agroecosyst., 89, 453–461, 2011.
885

886 Beetz, S., Liebersbach, H., Glatzel, S., Jurasinski, G., Buczko, U., and Höper, H.: Effects
887 of land-use intensity on the full greenhouse gas balance in an Atlantic peat bog,
888 Biogeosciences, 10, 1067–1082, doi:10.5194/bg-10-1067-2013, 2013.
889

890 Berglund, Ö., and Berglund, K.: Influence of water table level and soil properties on
891 emissions of greenhouse gases from cultivated peat soil, Soil Biology and Biochemistry,
892 43(5), 923–931, 2011.
893

894 Blodau, C.: Carbon cycling in peatlands – A review of processes and controls, Environ.
895 Rev., 10, 111–134, 2002.
896
897 Bridgham, S.D., Updegraff, K., and Pastor, J.: Carbon, nitrogen, and phosphorus
898 mineralization in northern wetlands, Ecology, 79(5), 1545–1561, 1998,
899
900 Butterbach-Bahl, K., Baggs, E.M., Dannenmann, M., Kiese, R., Zechmeister-Boltenstern,
901 S.: Nitrous oxide emissions from soils: how well do we understand the processes and
902 their controls?, Phil trans R Soc B, 368:20130122, 2013.
903
904 Byrne, K.A., and Farrell, E.P.: The effect of afforestation on soil carbon dioxide
905 emissions   in   blanket   peatland   in   Ireland,   Forestry,   78,   217–227,
906 doi:10.1093/forestry/cpi020, 2005.
907
908 Byrne, K. A., Chojnicki, B., Christensen, T. R., Drösler, M., Freibauer, A., Friborg, T.,
909 Frolking, S., Lindroth, A., Mailhammer, J., Malmer, N., Selin, P., Turunen, J., Valentini,
910 R., and Zetterberg, L.: EU peatlands: Current carbon stocks and trace gas fluxes, Carbo-
911 Europe-GHG Concerted Action-Synthesis of the European Greenhouse Gas Budget,
912 Report, 4, 2004.
913
914 Chanton, J.P., Bauer, J.E., Glaser, P.A., Siegel, D.I., Kelley, C.A., Tyler, S.C.,
915 Romanowicz, E.H., and Lazrus, A.: Radiocarbon evidence for the substrates supporting
916 methane formation within northern Minnesota peatlands, Geochimica et Cosmochimica
917 Acta, 59(17), 3663–3668, 1995.
918
919 Chapin, F. S., Woodwell, G. M., Randerson, J. T., Rastetter, E. B., Lovett, G. M.,
920 Baldocchi, D. D., Clark, D. A., Harmon, M. E., Schimel, D. S., Valentini, R., Wirth, C.,
921 Aber, J. D., Cole, J. J., Goulden, M. L., Harden, J. W., Heimann, M., Howarth, R. W.,
922 Matson, P. A., McGuire, A. D., Melillo, J. M., Mooney, H. A., Neff, J. C., Houghton, R. A.,
923 Pace, M. L., Ryan, M. G., Running, S. W., Sala, O. E., Schlesinger, W. H., and Schulze,
924 E.–D.: Reconciling Carbon-cycle Concepts, Terminology, and Methods, Ecosystems, 9,
925 1041–1050, 2006.
926
927 Couwenberg, J.: Greenhouse gas emissions from managed peat soils: is the IPCC
928 reporting guidance realistic, Mires and Peat, 8, 1–10, 2011.
929
930 Crutzen, P.J.: the role of NO and NO_2 in the chemistry of the troposphere and
931 stratosphere, Annu. Rev. Earth Planet. Sci., 7 (1), 443–472, 1979.
932
933 Crawley, M. J.: The R Book, John Wiley and Sons Ltd, Chichester, p. 942, 2007.
934
935 Dao, T.H.: Tillage and crop residue effects on carbon dioxide evolution and carbon
936 storage in a paleustoll, Soil Science Society of America Journal, 62, 250–256, 1998.
937
938 Davidson, E.A., Swank, W.T., and Perry, T.O.: Distinguishing between nitrification and
939 denitrification   as   sources   of   gaseous   nitrogen   production   in   soil,   Applied   and
940 Environmental Microbiology, 52 (6), 1280–1286, 1986.
941
942 Dinsmore, K. J., Billet, M. F., Skiba, U. M., Rees, R. M., Drewer, J., and Helfter, C.: Role
943 of the aquatic pathway in the carbon and greenhouse gas budgets of a peatland
944 catchment, Glob. Change Biol., 16, 2750–2762, 2010.

945

946 Drösler, M.: Trace gas exchange and climatic relevance of bog ecosystems, Southern
947 Germany, PhD thesis, Chair of Vegetation Ecology, Department of Ecology, Technical
948 University Munich, p. 179, 2005.

949

950 Drösler, M., Freibauer, A., Christensen, T. and Friborg, T.: Observation and status of
951 peatland greenhouse gas emission in Europe, In: Dolman, H., Valentini, R. & Freibauer,
952 A. (eds) The Continental-Scale Greenhouse Gas Balance of Europe. Ecological Studies,
953 203, 237–255, 2008.

954

955 Drösler, M., Freibauer, A., Adelmann, W., Augustin, J., Bergman, L., Beyer, C., Chojnicki,
956 B., Förster, C., Giebels, M., Görlitz, S., Höper, H., Kantelhardt, J., Liebersbach, H.,
957 Hahn-Schöfl, M., Minke, M., Petschow, U., Pfadenhauer, J., Schaller, L., Schägner,P.,
958 Sommer, M., Thuille, A., and Wehrhan, M.: Klimaschutz durch Moorschutz in der Praxis,
959 Arbeitsbericht aus dem vTI-Institut für Agrarrelevante Klimaforschung, p. 21, available
960 at http://www.vti.bund.de/de/startseite/institute/ak/ publikationen.html, 2011.

961

962 Drösler, M., Adelmann, W., Augustin, J., Bergmann, L., Beyer, C., Chojnicki, B., Förster,
963 C., Freibauer, A., Giebels, M., Görlitz, S., Höper, H., Kantelhardt, J., Liebersbach, H.,
964 Hahn-Schöfl, M., Minke, M., Petschow, U., Pfadenhauer, J., Schaller, L., Schägner, P.,
965 Sommer, M., Thuille, A., and Wehrhan, M.: Klimaschutz durch Moorschutz.
966 Schlussbericht des Vorhabens "Klimaschutz – Moorschutzstrategien", 2006–2010, 2013.

967

968 Eickenscheidt, T., Heinichen, J., Augustin, J., Freibauer, A., and Drösler, M.: Nitrogen
969 mineralization and gaseous nitrogen losses from waterlogged and drained organic soils
970 in a black alder (*Alnus glutinosa* (L.) Gaertn.) forest, Biogeosciences, 11, 2961–2976,
971 2014a.

972

973 Eickenscheidt, T., Freibauer, A., Heinichen, J., Augustin, J., and Drösler, M.: Short-term
974 effects of biogas digestate and cattle slurry application on greenhouse gas emissions
975 affected by N availability from grasslands on drained fen peatlands and associated
976 organic soils, Biogeosciences, 11, 6187–6207, 2014b.

977

978 Elsgaard, L., Gorres, C.-M., Hoffmann, C. C., Blicher-Mathiesen, G., Schelde, K., and
979 Petersen, S. O.: Net ecosystem exchange of CO_2 and carbon balance for eight
980 temperate organic soils under agricultural management, Agr. Ecosyst. Environ., 162,
981 52–67, 2012.

982

983 Falge, E., Baldocchi, D., Olson, R., Anthoni, P., Aubinet, M., Bernhofer, C., Burba, G.,
984 Ceulemans, R., Clement, R., Dolman, H., Granier, A., Gross, P., Grunwald, T., Hollinger,
985 D., Jensen, N. O., Katul, G., Keronen, P., Kowalski, A., Lai, C. T., Law, B. E., Meyers,
986 T., Moncrieff, H., Moors, E., Munger, J. W., Pilegaard, K., Rannik, U., Rebmann, C.,
987 Suyker, A., Tenhunen, J., Tu, K., Verma, S., Vesala, T., Wilson, K., and Wofsy, S.: Gap
988 filling strategies for defensible annual sums of net ecosystem exchange, Agr. Forest
989 Meteorol., 107, 43–69, 2001.

990

991 Flessa, H., Wild, U., Klemisch, M., and Pfadenhauer, J.: C- und N-Stoffflüsse auf
992 Torfstichsimulationsflächen im Donaumoos, Z. f. Kulturtechnik und Landentwicklung, 38,
993 11–17, 1997.

994

995 Flessa, H., Wild, U., Klemisch, M., and Pfadenhauer, J.: Nitrous oxide and methane

996   fluxes from organic soils under agriculture, European Journal of Soil Science, 49, 327–
997   335. 1998.
998
999   Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D. W., Haywood,
1000   J., Lean, J., Lowe, D. C., Myhre, G., Nganga, J., Prinn, R., Raga, G., Schulz, M., and
1001   Van Dorland, R.: Changes in Atmospheric Constituents and in Radiative Forcing, in:
1002   Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to
1003   the Fourth Assessment Report of the Intergovernmental Panel on Climate Change,
1004   edited by: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K. B., Tignor,
1005   M., and Miller, H. L., Cambridge University Press, Cambridge, United Kingdom and New
1006   York, NY, USA, 2007.
1007
1008   Freibauer, A., Rounsevell, M.D.A., Smith, P., and Verhagen, J.: Carbon sequestration in
1009   the agricultural soils of Europe, Geoderma, 122, 1–23, 2004.
1010
1011   Glatzel, S., Basiliko, N., and Moore, T.: Carbon dioxide and methane production
1012   potential of peats from natural, harvested and restored sites, Eastern Québec, Canada,
1013   Wetlands,24(2), 261–267, 2004.
1014
1015   Goldberg, S.D., Knorr, K.H., Blodau, C., Lischeid, G., and Gebauer, G.: Impact of
1016   altering the water table height of an acidic fen on N_2O and NO fluxes and soil
1017   concentrations, Global Change Biol., 16, 220–233, 2010.
1018
1019   Grønlund, A., Sveistrup, T.E., Søvik, A.K., Rasse, D.P., and Kløve, B.: Degradation of
1020   cultivated peat soils in northern Norway based on field scale CO_2, N_2O and CH_4 emission
1021   measurements, Arch Agron. Soil Sci., 52, 149–159, 2006.
1022
1023   Grønlund, A., Hauge, A., Hovde, A., and Rasse, D.A.: Carbon loss estimates from
1024   cultivated peat soils in Norway: a comparison of three methods, Nutr. Cycl. Agroecosyst.,
1025   81, 157–167, 2008.
1026
1027   Grosse, W., Frye, J., and Lattermann, S.: Root aeration in wetland trees by pressurized
1028   gas transport, Tree Physiology, 10 285–295, 1992.
1029
1030   Gupta, H.V., Sorooshian, S., and Yapo, P.O.: Status of automatic calibration for
1031   hydrologic models: Comparison with multilevel expert calibration, J. Hydrologic Eng.,
1032   4(2), 135–143, 1999.
1033
1034   Gutser, R., Ebertseder, Th., Weber, A., Schraml, M., and Schmidhalter, U.: Short-term
1035   and residual availability of nitrogen after long-term application of organic fertilizers on
1036   arable land, J. Plant Nutr. Soil Sci., 168, 439–446, 2005.
1037
1038   Hahn-Schöfl, M., Zak, D., Minke, M., Gelbrecht, J., Augustin, J., and Freibauer, A.:
1039   Organic sediment formed during inundation of a degraded fen grassland emits large
1040   fluxes of CH_4 and CO_2, Biogesciences, 8, 1539–1550, 2011.
1041
1042   Hardie, S.M.L., Garnett, M.H., Fallick, A.E., Rowland, A.P., Ostle, N.J., and Flowers,
1043   T.H.: Abiotic drivers and their interactive effect on the flux and carbon isotope (^{14}C and
1044 δ^{13}C) composition of peat-respired CO_2, Soil Biol. Biochem., 43, 2432–2440,
1045   doi:10.1016/j.soilbio.2011.08.010, 2011.
1046

1047 Heller, C., and Zeitz, J.: Stability of soil organic matter in two northeastern German fen
1048 soils: the influence of site and soil development, J. Soils Sediments, 12, 1231–1240,
1049 DOI 10.1007/s11368-012-0500-6, 2012.

1051 Henry, S., Texier, S., Hallet, S., Bru, D., Dambreville, C., Chèneby, D., Bizouard, F.,
1052 Germon, J.C., and Philippot, L.: Disentangling the rhizosphere effect on nitrate reducers
1053 and denitrifiers: insight into the role of root exudates, Environ. Microbiol., 10, 3082–3092,
1054 2008.

1056 Hothorn, T., Bretz, F., Westfall, P., Heiberger, R.M., and Schuetzenmeister, A.:
1057 Simultaneous Inference in General Parametric Models: R package version 1.2-17, 2013.

1059 Höper, H., Augustin, J., Cagampan, J. P., Drösler, M., Lundin, L., Moors, E. J., Vasander,
1060 H., Waddington, J. M., and Wilson, D.: Restoration of peatlands and greenhouse gas
1061 balances, in: Peatlands and Climate Change. edited by: Strack, D. M., International Peat
1062 Society, Jyvaskyla, 182–210, 2008.

1064 Horrocks, R.D., and Valentine, J.F.: Harvested forages, Academic Press, 1999.

1066 International Peat Society: Peatlands and climate change, Ed. Strack, M., Publish by the
1067 International Peat Society, p. 227, 2008.

1069 IPCC: Changes in atmospheric constituents and in radiative forcing, in: Climate Change
1070 2007: The physical science basis. Contribution of Working Group I to the Fourth
1071 Assessment Report of the Intergovernmental Panel on Climate Change, edited by:
1072 Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K. B., Tignor, M., and
1073 Miller, H. L., Cambridge University Press, Cambridge, UK and New York, USA, 2007.

1075 IPCC, 2014: 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse
1076 Gas Inventories: Wetlands, editd by: Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N.,
1077 Baasansuren, J., Fukuda, M., and Troxler, T. G., IPCC, Switzerland, 2014.

1079 Jones, S.K., Rees, R.M., Skiba, U.M., and Ball, B.C.: Influence of organic and mineral N
1080 fertiliser on N_2O fluxes from temperate grassland, Agriculture, Ecosystems and
1081 Environment, 121, 74–83, 2007.

1083 Kandel, T.P., Elsgaard, L., Karki, S., and Læerke, P.E.: Biomass yield and greenhouse
1084 gas emissions from a drained fen peatland cultivated with reed canary grass under
1085 different harvest and fertilizer regimes, Bioenerg. Res., 6, 883–895, DOI
1086 10.1007/s12155-013-9316-5, 2013.

1088 Kasimir-Klemedtsson, Å., Klemedtsson, L., Berglund, K., Martikainen, P., Silvola, J., and
1089 Oenema, O.: Greenhouse gas emissions from farmed organic soils: a review, Soil Use
1090 and Management, 13, 245–250, 1997.

1092 Kroeze, C., Mosier, A., and Bouwman, L.:1999. Closing the global N_2O budget: A
1093 retrospective analysis 1500 – 1994, Global Biogeochemical Cycles, 13 (1), 1–8, 1999.

1095 Krull, E.S., Baldock, J.A., and Skjemstad, J.O: Importance of mechanisms and
1096 processes of the stabilisation of soil organic matter for modeling carbon turnover,
1097 Functional Plant biology, 30, 207–222, 2003.

1098
1099   Leiber-Sauheitl, K., Fuß, R., Voigt, C., and Freibauer, A.: High CO_2 fluxes from grassland
1100   on histic Gleysol along soil carbon and drainage gradients, Biogeosciences, 11, 749–
1101   761, doi:10.5194/bg-11-749-2014, 2014.

1102
1103   Leifeld, J., Müller, M., and Fuhrer, J.: Peatland subsidence and carbon loss from drained
1104   temperate fens, Soil Use Manage., 27, 170–176, 2011.

1105
1106   Leifeld, J., Steffens, M., and Galego-Sala, A.: Sensitivity of peatland carbon loss to
1107   organic matter quality, Geophys. Res. Lett., 39, L14704, doi:10.1029/2012GL051856,
1108   2012.

1109
1110   Leifeld, J., Bader, C., Borraz, E., Hoffmann, M., Giebels, M., Sommer, M., and Augustin,
1111   J.: Are C-loss rates from drained peatlands constant over time? The additive value of
1112   soil profile based and flux budget approach, Biogeosciences Discuss., 11, 12341–12373,
1113   2014.

1114
1115   Leppelt, T. Dechow, R., Gebbert, S., Freibauer, A., Lohila, A., Augustin, J., Drösler, M.,
1116   Fiedler, S., Glatzel, S., Höper, H., Järveoja, J., Lærke, P.E., Maljanen, M., Mander, Ü.,
1117   Mäkiranta, P., Minkkinen, K., Ojanen, P., Regina, K., and Strömgren, M.: Nitrous oxide
1118   emission hotspots from organic soils in Europe, Biogeosciences Discuss., 11, 9135–
1119   9182, 2014.

1120
1121   Livingston, G.P., and Hutchinson, G.L.: Enclosure-based measurement of trace gas
1122   exchange: application and sources of error, In: Matson, PA, Harriss, RC, (eds.), Biogenic
1123   Trace Gases: Measuring Emissions from Soil and Water. Blackwell Science, Cambridge,
1124   14–50, 1995.

1125
1126   Lloyd, J. and Taylor, J. A.: On the temperature dependence of soil respiration, Funct.
1127   Ecol., 8, 315–323, 1994.

1128
1129   Lohila, A.: Carbon dioxide exchange on cultivated and afforested boreal peatlands,
1130   Finnish Meteorological Institute Contributions, 73, 1–47, 2008.

1131
1132   Lützow, v.M., Kögel-Knabner, I., Ekschmitt, K., Matzner, E., Guggenberger, G.,
1133   Marschner, B., and Flessa, H.: Stabilization of organic matter in temperate soils:
1134   mechanisms and their relevance under different soil conditions – a review, European
1135   Journal of Soil Science, 57, 426–445, doi: 10.1111/j.1365-2389.2006.00809.x, 2006.

1136
1137   Maljanen, M., Hytönen, J., and Martikainen, P. J.: Fluxes of N_2O, CH_4 and CO_2 on
1138   afforested boreal agricultural soils, Plant Soil, 231, 113–121, 2001.

1139
1140   Maljanen, M., Hytönen, J., Mäkiranta, P., Alm, J., Minkkinen, K., Laine, J. and
1141   Martikainen, P.J.: Greenhouse gas emissions from cultivated and abandoned organic
1142   arable lands in Finland, Boreal Environment Research, 12, 133–144, 2007.

1143
1144   Maljanen, M., Sigurdsson, B. D., Guðmundsson, J., Óskarsson, H., Huttunen, J. T., and
1145   Martikainen, P. J.: Greenhouse gas balances of managed peatlands in the Nordic
1146   countries – present knowledge and gaps, Biogeosciences, 7, 2711–2738,
1147   doi:10.5194/bg-7-2711-2010, 2010.

1148

1149 Martikainen, P.J., Nykänen, H., Crill, P., and Silvola, J.: Effect of a lowered water table
1150 on nitrous oxide fluxes from northern peatlands, Nature, 366, 51–53, 1993.
1151
1152 Michaelis, L. and Menten, M. L.: Die Kinetik der Invertinwirkung, Biochem. Z., 49, 333–
1153 369, 1913.
1154
1155 Minchin, P.E.H., and McNaughton, G.S.: Exudation of recently fixed carbon by non-
1156 sterile roots, Journal of Experimental Botany, 35(150), 74–82, 1984.
1157
1158 Moriasi, D.N, Arnold, J.G., Van Liew, M.W., Bingner, R.L., Harmel, R.D., and Veith, T.L.:
1159 Model evaluation guidelines for systematic quantification of accuracy in watershed
1160 simulations, American Society of Agricultural and Biological Engineers, 50(3), 885–900,
1161 2007.
1162
1163 Mounier, E., Hallet, S., Chèneby, D., Benizri, E., Gruet, Y., Nguyen, C., Piutti, S., Robin,
1164 C., Slezack-Deschaumes, S., Martin-Laurent, F., Germon, J.C., and Philippot, L.:
1165 Influence of maize mucilage on the diversity and activity of the denitrifying community,
1166 Environ. Microbiol., 6, 301–312, 2004.
1167
1168 Nash, J.E., and Sutcliffe, J.V.: River flow forecasting through conceptual models: Part 1.
1169 A discussion of principles, J. Hydrology, 10(3), 282–290, 1970.
1170
1171 Nielsen, O.-K., Mikkelsen, M.H., Hoffmann, L., Gyldenkærne, S., Winther, M., Nielsen,
1172 M., Fauser, P., Thomsen, M., Plejdrup, M.S., Albrektsen, R., Hjelgaard, K., Bruun, H.G.,
1173 Johannsen, V.K., Nord-Larsen, T., Bastrup-Birk, A., Vesterdal, L., Møller, I.S.,
1174 Rasmussen, E., Arfaoui, K., Baunbæk, L. & Hansen, M.G.: Denmark's National Inventory
1175 Report 2012. Emission Inventories 1990-2010 - Submitted under the United Nations
1176 Framework Convention on Climate Change and the Kyoto Protocol. Aarhus University,
1177 DCE – Danish Centre for Environment and Energy, 1168 pp. Scientific Report from DCE
1178 – Danish Centre for Environment and Energy No. 19 http://www.dmu.dk/Pub/SR19.pdf,
1179 2012.
1180
1181 NIR 2010: National Emission Inventory Report (NIR) 2010 for 2008 – Calculation of
1182 Emissions from German Agriculture, Ed. Haenel, H.D., vTI Agriculture and Forestry
1183 Research, Special Issue 334, p. 428, 2010.
1184
1185 Petersen, S.O., Hoffmann, C.C., Schäfer, C.-M., Bilcher-Mathiesen, G., Elsgaard, L.,
1186 Kristensen, K., Larsen, S.E., Torp, S.B., and Greve, M.H.: Annual emissions of CH_4 and
1187 N_2O, and ecosystem respiration, from eight organic soils in Western Denmark managed
1188 by agriculture, Biogeosciences, 9, 403–422, 2012.
1189
1190 R Development Core Team: R: A language and environment for statistical computing, R
1191 Foundation for Statistical Computing, Vienna, Austria, ISBN 3-900051-07-0, available at:
1192 http://www.R-project.org, 2013.
1193
1194 Reiche, M., Gleixner, G., and Küsel, K.: Effect of peat quality on microbial greenhouse
1195 gas formation in an acidic fen, Biogeosciences, 7, 187–198, 2010.
1196
1197 Renou-Wilson, F., Barry, C., Müller, C., and Wilson, D.: The impacts of drainage, nutrient
1198 status and management practice on the full carbon balance of grasslands on organic
1199 soils in a maritime temperate zone, Biogeosciences, 11, 4361–4379, 2014.

1200
1201   Rochette, P., Angers, D.A., Belanger, G., Chantigny, M.H., Prevost, D., and Levesque,
1202   G.: Emissions of N_2O from alfalfa and soybean crops in eastern Canada, Soil Sci. Soc.
1203   Am. J., 68, 493–506, 2004.
1204
1205   Schäfer, C. M., Elsgaard, L., Hoffmann, C. C., and Petersen, S. O.: Seasonal methane
1206   dynamics in three temperate grasslands on peat, Plant Soil, 357, 339–353, 2012.
1207
1208   Schober, H.M., Stein, Ch., and Prösl, K.-H: Interkommunales Flächenmanagement
1209   Schlussbericht, LEADER+-Projekt Freisinger Moos, p. 56, 2008.
1210
1211   Schothorst, C.J.: Subsidence of low moor peat soils in the Western Netherlands,
1212   Geoderma, 17, 265–291, 1977.
1213
1214   Sey, B.K., Manceur, A.M., Wahlen, J.K., Gregorich, E.G, and Rochette, P.: Root-derived
1215   respiration and nitrous oxide production as affected by crop phenology and nitrogen
1216   fertilization, Plant Soil, 326, 369–379, 2010.
1217
1218   Silvan, N., Tuittila, E.-S., Kitunen, V., Vasander, H., and Laine, J.: Nitrate uptake by
1219   *Eriophorum vaginatum* controls N_2O production in a restored peatland, Soil Biol.
1220   Biochem., 37, 1519–1526, 2005.
1221
1222   Silvola, J., Alm, J., Ahlolm, U., Nykänen, H., and Martikainen, P.J.: CO_2 fluxes from peat
1223   in boreal mires under varying temperature and moisture conditions, Journal of Ecology,
1224   84, 219–228, 1996.
1225
1226   Sonneveld, M.P.W., and Lantinga, E.A.: The contribution of mineralization to grassland
1227   N uptake on peatland soils with anthropogenic A horizons, Plant Soil; 340, 357–368,
1228   2011.
1229
1230   Stolk, P.C., Hendriks, R.F.A., Jacobs, C.M.J., Moors, E.J., and Kabat, P.: Modeling the
1231   effect of aggregates on N_2O emission from denitrification in an agricultural peat soil,
1232   Biogeosciences, 8, 2649–2663, 2011.
1233
1234   Svensson, B.H., and Sundh, I.: Factors affecting methane production in peat soils. Suo.,
1235   43, 183–190, 1992.
1236
1237   Tiemeyer, B., Freibauer,A., Drösler, M., Albiac-Borraz, E., Augustin J., Bechtold, M.,
1238   Beetz, S., Belting, S., Bernrieder, M., Beyer, C., Eberl, J., Eickenscheidt, T., Fell, H.,
1239   Fiedler, S., Förster, C., Frahm, E., Frank, S., Giebels, M., Glatzel, S., Grünwald, T.,
1240   Heinichen, J., Hoffmann, M., Hommeltenberg, J., Höper, H., Laggner, A., Leiber-Sauheitl,
1241   K., Leppelt, T., Metzger, C., Peichl-Brak, M., Röhling, S., Rosskopf, N., Rötzer, T.,
1242   Sommer, M., Wehrhan, M., Werle, P., and Zeitz, J.: Klimarelevanz von Mooren und
1243   Anmooren in Deutschland: Ergebnisse aus dem Verbundprojekt "Organische Böden in
1244   der Emissionsberichterstattung", Thünen Working Paper, No. 15, http://nbn-
1245   resolving.de/urn:nbn:de:gbv:253-201311-dn052806-7, 2013.
1246
1247   Tiemeyer, B., Borraz, E.A, Augustin, J., Bechtold, M., Beetz, S., Beyer, C., Eickenscheidt,
1248   T., Drösler, M., Förster C., Freibauer, A., Giebels, M., Glatzel, S., Heinichen, J.,
1249   Hoffmann, M., Höper, H., Leiber-Sauheitl, K., Rosskopf, N., and Zeitz, J.: Greenhouse
1250   gas budgets for grasslands on peatlands and other organic soils, Geophysical Research

Abstracts, Vol. 16, EGU2014-14825, 2014, EGU General Assembly, 2014.

Tietema, A., Warmerdam, B., Lenting, E., and Riemer, L.: Abiotic factors regulating nitrogen transformations in the organic layer of acid forest soils: Moisture and pH, Plant and Soil, 147, 69–78, 1992.

Tjoelker, M.G., Oleksyn, J., and Reich, P.B.: Modeling respiration of vegetation: evidence for a general temperature-dependent Q_{10}, Global Change Biology, 7, 223–230, 2001.

Veenendaal, E.M., Kolle, O., Leffelaar, P.A., Schrier-Uijl, A.P., Huissteden van, J., Walsem van, J., Möller, F., and Berendse, F.: CO_2 exchange and carbon balance in two grassland sites on eutrophic drained peat soils, Biogeosciences, 4, 1027–1040, 2007.

Velthof, G.L., Brader, A.B., and Oenema, O.: Seasonal variations in nitrous oxide losses from managed grasslands in the Netherlands, Plant and Soil, 181, 263–274, 1996.

VDLUFA: Bestimmung von mineralischem (Nitrat-)Stickstoff in Bodenprofilen (Nmin-Labormethode), in: Methodenbuch Teil 2, VDLUFA, Speyer, Germany, 1997.

Vuichard, N., Soussana, J.-F., Ciais, P., Viovy, N., Ammann, C., Calanca, P., Clifton-Brown, J., Fuhrer, J., Jones, M., and Martin, C.: Estimating the greenhouse gas fluxes of European grasslands with a process-based model: 1. Model evaluation from in situ measurements, Global Biogeochem. Cycles, 21, GB1004, doi:10.1029/2005GB002611, 2007.

Whalen, S.C.: Biogeochemistry of methane exchange between natural wetlands and the atmosphere, Environmental Engineering Science, 22 (1), 73–94, 2005.

Worrall, F., Burt, T. P., Rowson, J. G., Warburton, J., and Adamson, J. K.: The multi-annual carbon budget of a peat-covered catchment, Sci. Total Environ., 407, 4084–4094, 2009.

WRB, 2006 – IUSS Working Group: World Reference Base for Soil Resources 2006, 2nd edition, World Soil Resources Reports No. 103. Rome. 2006.

Zehlius-Eckert, W., Schwaiger, H., and Beckmann, A.: Monitoring und Erfolgskontrolle im Freisinger Moos, Bayer. Akad.f. Naturschutz u. Landschaftspflege - Laufen /Salzach 2003, Laufener Seminarbeitr., 1/03, 147–170, 2003.

Table 1 Physical and chemical properties of the investigated plots.

Site	Soil type	Organic carbon [%]		C/N ratio	pH (CaCl$_2$)	Bulk density [g cm^{-3}]		Mean GW level above surface [cm]	
		0–10 cm	10–20 cm	0–20 cm	0–20 cm	0–10 cm	10–20 cm	2010	2011
A1C$_{medium}$	molic Gleysol	9.6 ± 0.1	9.3 ± 0.2	10	5.24	0.72 ± 0.03	0.85 ± 0.08	−56 (−86/0)	−67 (−86/−4)
A1C$_{high}$	sapric Histosol	16.9 ± 0.2	17.2 ± 0.2	12	5.61	0.63 ± 0.05	0.67 ± 0.04	−45 (−90/7)	−49 (−76/5)
A2C$_{medium}$	molic Gleysol	9.4 ± 0.0	9.2 ± 0.1	10	5.24	0.83 ± 0.06	0.90 ± 0.06	−56 (−86/0)	−67 (−86/−4)
A2C$_{high}$	sapric Histosol	16.1 ± 0.9	16.8 ±0.2	12	5.61	0.67 ± 0.11	0.77 ± 0.08	−45 (−90/7)	−49 (−76/5)
G1C$_{medium}$	molic Gleysol	10.5 ± 0.2	9.4 ± 0.1	10	4.10	0.71 ± 0.09	0.90 ± 0.06	−65 (−91/−2)	−72 (−92/0)
G1C$_{high}$	sapric Histosol	17.2 ± 0.0	16.7 ± 0.1	11	4.24	0.53 ± 0.09	0.64 ± 0.05	−45 (−64/−1)	−52 (−66/−3)
G2C$_{medium}$	molic Gleysol	10.9 ± 0.2	10.1 ± 0.1	10	4.10	0.81 ± 0.09	0.88 ± 0.03	−63 (−92/0)	−72 (−97/0)
G2C$_{high}$	sapric Histosol	16.4 ± 0.1	15.6 ± 0.1	11	4.24	0.57 ± 0.08	0.67 ± 0.03	−45 (−67/−1)	−50 (−65/−3)

Values present means ± SD
Values in brackets are minimum and maximum values
A, arable land; G, grassland;

Table 2 Date and type of conducted management events.

Date	Julian day	Management events		
		A1	A2	G
2009-09-24	–	seed sowing (Secale cereale)	seed sowing (Secale cereale)	–
2010-03-26	85	–	–	levelling
2010-03-30	89	–	plowing & seed sowing (Avena sativa + 15% Vicia faba minor)	–
2010-04-07	97	–	–	rolling
2010-04-13	103	–	harrowing	–
2010-04-28	118	plowing	–	–
2010-04-30	120	seed sowing (Zea mays)	–	–
2010-05-24	144	grubbering	–	harvesting
2010-06-11	162	grubbering	–	–
2010-06-14	165	–	–	manuring [20 m³ ha^{-1}]
2010-07-06	187	grubbering & hilling	–	–
2010-08-20	232	–	–	harvesting
2010-08-22	234	–	harvesting	–
2010-08-25	237	–	–	manuring [20 m³ ha^{-1}]
2010-08-28	240	–	milling	–
2010-09-04	247	–	–	–
2010-09-23	266	–	–	herbicide against common sorrel (Rumex acetosa)
2010-10-15	288	harvesting	–	–
2010-10-30	303	mulching	–	–
2011-03-16	440	–	–	levelling
2011-04-01	456	plowing & seed sowing (Avena sativa + 20% Vicia sativa)	–	–
2011-04-18	473	–	plowing	–
2011-04-26	481	–	grubbering + seed sowing (Zea mays)	–
2011-04-30	485	harrowing	harrowing	–
2011-05-08	493	–	harrowing	–
2011-05-19	504	–	mattocks	–
2011-05-23	508	–	–	harvesting
2011-05-27	512	–	–	manuring [25 m³ ha^{-1}]
2011-06-14	530	–	hilling	–
2011-08-01	578	–	–	harvesting
2011-08-16	593	harvesting	–	–
2011-08-18	595	milling	–	–
2011-08-27	604	plowing & seed sowing (Secale cereale)	–	–
2011-09-13	621	–	–	harvesting
2011-09-22	630	–	–	manuring [20 m³ ha^{-1}]
2011-09-28	636	–	harvesting	–

Table 3 Cumulative R_{ECO}, GPP, NEE, CH_4 and N_2O exchange rates as well as C import through fertilizer and C export due to crop/grass yield.

Plot/year	R_{ECO} [g C m^{-2} yr^{-1}]	GPP [g C m^{-2} yr^{-1}]	NEE [g C m^{-2} yr^{-1}]	Fertilizer input* [g C m^{-2} yr^{-1}]	Crop yield* [g C m^{-2} yr^{-1}]	CH_4* [g C m^{-2} yr^{-1}]	N_2O* [g N m^{-2} yr^{-1}]
A1C$_{medium}$ / 10	2473 ± 272	−1454 ± 114	1019 ± 386	–	193 ± 53	–	–
A1C$_{medium}$ / 11	2992 ± 230	−1862 ± 126	1130 ± 356	–	74 ± 8	0.51 ± 0.17	0.27 ± 0.01
A1C$_{high}$ / 10	2012 ± 284	−873 ± 110	1139 ± 394	–	58 ± 23	–	–
A1C$_{high}$ / 11	2117 ± 123	−1302 ± 77	815 ± 200	–	135 ± 7	0.22 ± 0.04	0.23 ± 0.05
A2C$_{medium}$ / 10	2704 ± 544	−1449 ± 103	1255 ± 647	–	227 ± 27	–	–
A2C$_{medium}$ / 11	2354 ± 309	−2360 ± 237	−6 ± 546	–	457 ± 71	−0.03 ± 0.05	0.39 ± 0.06
A2C$_{high}$ / 10	2907 ± 482	−1200 ± 137	1707 ± 619	–	145 ± 19	–	–
A2C$_{high}$ / 11	2538 ± 329	−2188 ± 253	350 ± 582	–	330 ± 79	−0.10 ± 0.07	0.86 ± 0.21
G1C$_{medium}$ / 10	3954 ± 671	−2131 ± 180	1823 ± 851	126	297 ± 32	–	–
G1C$_{medium}$ / 11	4099 ± 300	−2414 ± 195	1685 ± 495	267	344 ± 63	−0.06 ± 0.09	0.12 ± 0.01
G1C$_{high}$ / 10	3736 ± 491	−2152 ± 140	1584 ± 631	126	325 ± 41	–	–
G1C$_{high}$ / 11	4026 ± 707	−2633 ± 138	1393 ± 845	267	455 ± 41	−0.07 ± 0.02	0.18 ± 0.02
G2C$_{medium}$ / 10	3683 ± 453	−2131 ± 213	1552 ± 666	76	342 ± 39	–	–
G2C$_{medium}$ / 11	4265 ± 379	−2880 ± 177	1385 ± 556	53	543 ± 58	−0.11 ± 0.05	0.19 ± 0.02
G2C$_{high}$ / 10	3521 ± 1041	−2093 ± 152	1428 ± 1193	76	380 ± 43	–	–
G2C$_{high}$ / 11	4316 ± 562	−2962 ± 178	1354 ± 740	53	593 ± 132	−0.02 ± 0.02	0.31 ± 0.09

* Data from grassland plots derived from Eickenscheidt et al. (2014).
A, arable land; G, grassland; 10, year 2010; 11, year 2011.

Table 4 Model evaluation statistics from observed R_{ECO} versus modeled R_{ECO}. r = Pearson's correlation coefficient, NSE = Nash-Sutcliffe efficiency, PBIAS = percent bias, RSR = ratio of the root mean square error to the standard deviation of measured data.

Site	2010				2011			
	r	NSE	PBIAS	RSR	r	NSE	PBIAS	RSR
A1C$_{medium}$	0.90	0.70	-7.93	0.55	0.98	0.95	-0.17	0.22
A1C$_{high}$	0.98	0.96	0.44	0.19	0.98	0.97	1.79	0.18
A2C$_{medium}$	0.93	0.81	-5.68	0.44	0.94	0.89	-0.23	0.33
A2C$_{high}$	0.96	0.92	2.60	0.29	0.98	0.96	0.00	0.20
G1C$_{medium}$	0.96	0.93	1.54	0.27	0.95	0.91	-2.40	0.31
G1C$_{high}$	0.89	0.75	-6.27	0.50	0.97	0.95	0.03	0.23
G2C$_{medium}$	0.93	0.86	0.80	0.37	0.98	0.96	0.06	0.19
G2C$_{high}$	0.93	0.82	-4.65	0.42	0.97	0.94	0.92	0.25

Table 5 Model evaluation statistics from observed NEE versus modeled NEE. r = Pearson's correlation coefficient, NSE = Nash-Sutcliffe efficiency, PBIAS = percent bias, RSR = ratio of the root mean square error to the standard deviation of measured data.

Site	2010					2011			
	r	NSE	PBIAS	RSR		r	NSE	PBIAS	RSR
A1C$_{medium}$	0.94	0.87	-11.84	0.36		0.97	0.93	1.41	0.26
A1C$_{high}$	0.94	0.88	-7.94	0.35		0.98	0.96	-4.94	0.21
A2C$_{medium}$	0.85	0.72	3.03	0.53		0.96	0.92	-3.64	0.28
A2C$_{high}$	0.79	0.61	3.63	0.63		0.96	0.91	-9.56	0.29
G1C$_{medium}$	0.90	0.80	-10.98	0.45		0.92	0.84	-10.47	0.40
G1C$_{high}$	0.91	0.82	-12.07	0.43		0.94	0.88	-10.04	0.35
G2C$_{medium}$	0.95	0.89	-13.23	0.33		0.96	0.92	-5.43	0.28
G2C$_{high}$	0.94	0.87	-10.71	0.36		0.94	0.89	-6.22	0.34

Table 6 Estimated global warming potential for a time horizon of 100 years.

Site/periode	GWP$_{100}$ NEE$_{corrected}$* [g CO$_{2 eq.}$ m^{-2} yr^{-1}]	GWP$_{100}$ CH$_4$ [g CO$_{2 eq.}$ m^{-2} yr^{-1}]	GWP$_{100}$ N$_2$O [g CO$_{2 eq.}$ m^{-2} yr^{-1}]	GWP$_{100}$ balance [g CO$_{2 eq.}$ m^{-2} yr^{-1}]
A1C$_{medium}$ / 11	4419 ± 1336	16.96 ± 5.65	126.32 ± 4.68	4562 ± 1346
A1C$_{high}$ / 11	3487 ± 760	7.32 ± 1.33	107.61 ± 23.39	3601 ± 785
A2C$_{medium}$ / 11	1655 ± 2264	-1.00 ± 1.33	182.47 ± 28.07	1837 ± 2293
A2C$_{high}$ / 11	2496 ± 2426	-3.33 ± 1.66	402.36 ± 98.25	2895 ± 2526
G1C$_{medium}$ / 11	6467 ± 2048	-2.00 ± 2.99	56.14 ± 4.68	6521 ± 2056
G1C$_{high}$ / 11	5802 ± 3252	-2.33 ± 0.67	84.21 ± 9.36	5884 ± 3262
G2C$_{medium}$ / 11	6881 ± 2253	-3.66 ± 1.66	88.89 ± 9.36	6967 ± 2264
G2C$_{high}$ / 11	6951 ± 3200	-0.67 ± 0.67	145.04 ± 42.11	7095 ± 3243

* Corrected for C export and C import
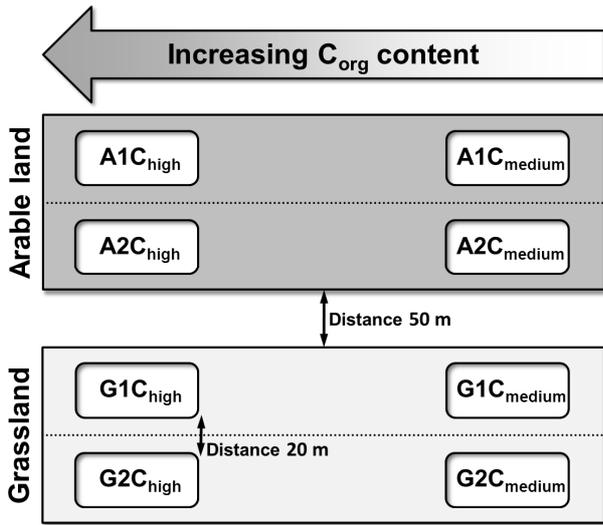
Fig. 1 Schema of the experimental design.
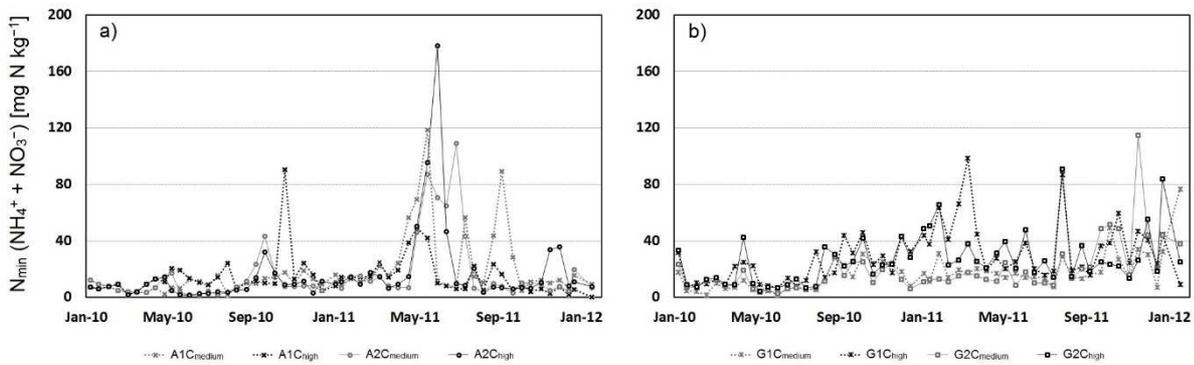
Fig. 2 Mineral nitrogen contents [mg N kg^{-1}] for the arable land a) and the grassland b) of the soil depth 0–10 cm for the years 2010 and 2011.
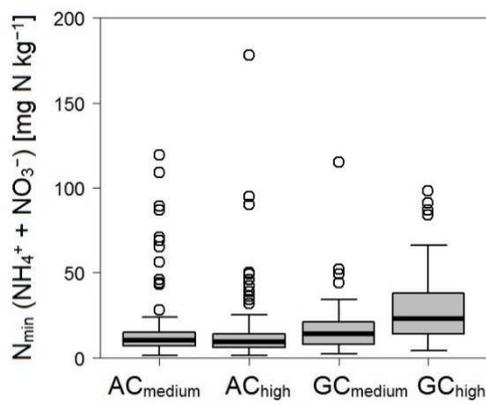
Fig. 3 Box plots of mineral nitrogen contents [mg N kg^{-1}] of the soil depth 0–10 cm (A = arable land, G = grassland).
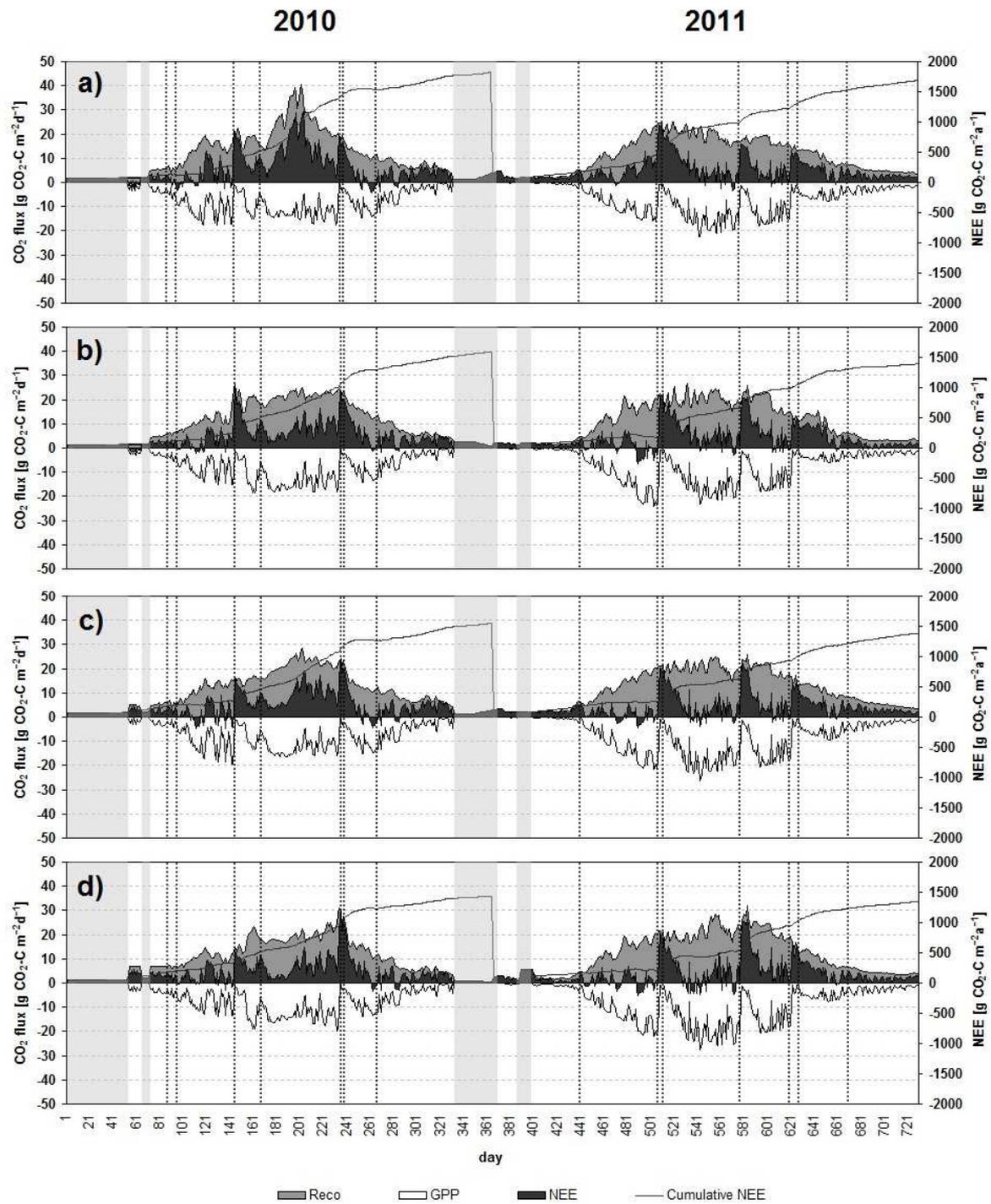
Fig. 4 Time series of modeled CO_2 fluxes [g CO_2-C m^{-2} d^{-1}; g CO_2-C m^{-2} yr^{-1}] for each site in 2010 and 2011; a) grassland, cattle slurry, C_{medium}; b) grassland cattle slurry, C_{high}; c) grassland biogas digestate C_{medium}; d) grassland, biogas digestate, C_{high}. Grey bars mark the period with snow cover. Dashed lines indicate management activities (see table 2).
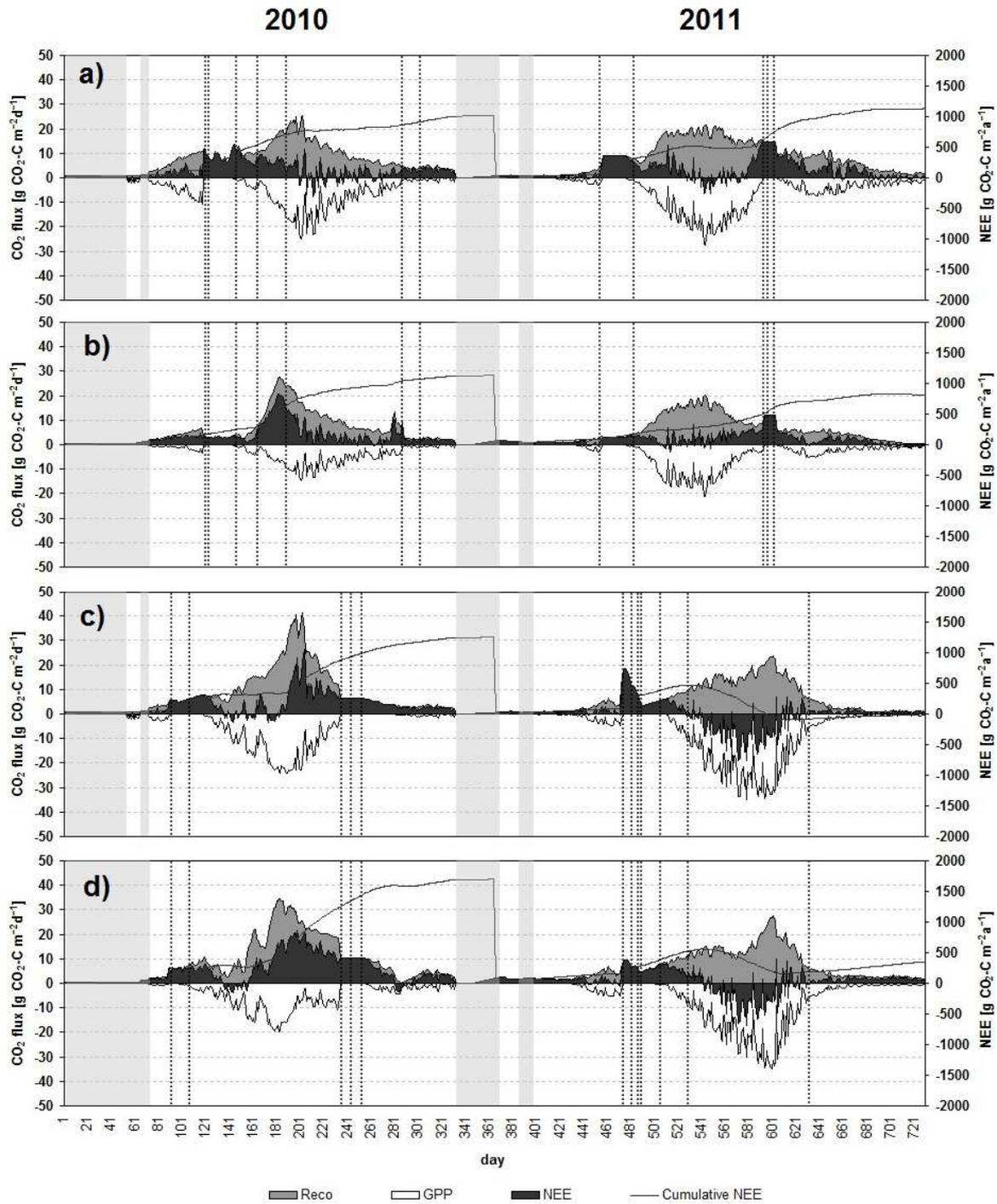
Fig. 5 Time series of modeled CO_2 fluxes [g CO_2-C m^{-2} d^{-1}; g CO_2-C m^{-2} yr^{-1}] for each site in 2010 and 2011; a) arable land, 2010 maize, 2011 oat, C_{medium}; b) arable land, 2010 maize, 2011 oat, C_{high}; c) arable land, 2010 oat, 2011 maize, C_{medium}; d) arable land, 2010 oat, 2011 maize, C_{high}. Grey bars mark the period with snow cover. Dashed lines indicate management activities (see table 2).
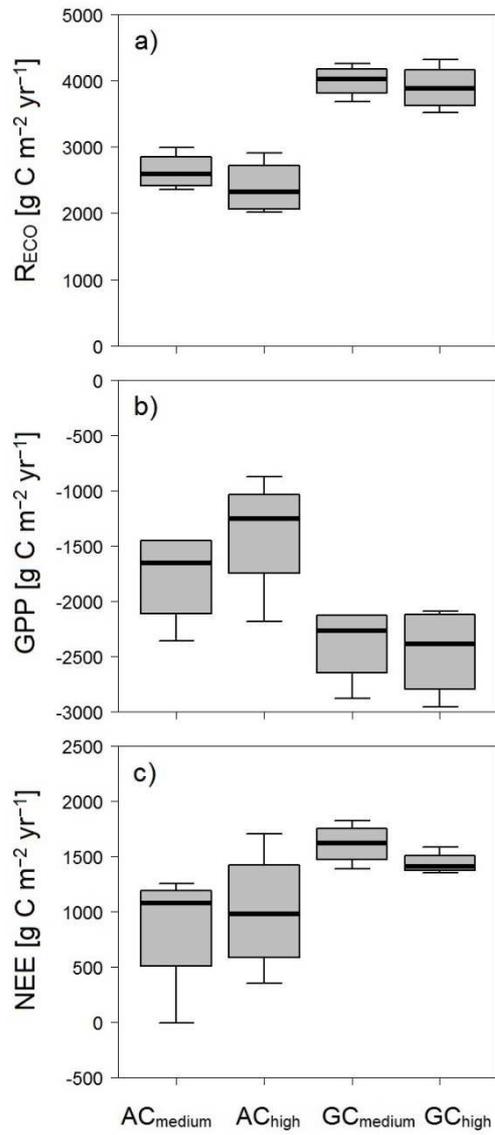
Fig. 6 Box plots of R_{ECO} (a), GPP (b) and NEE (c) for the two soil types and land-use types.
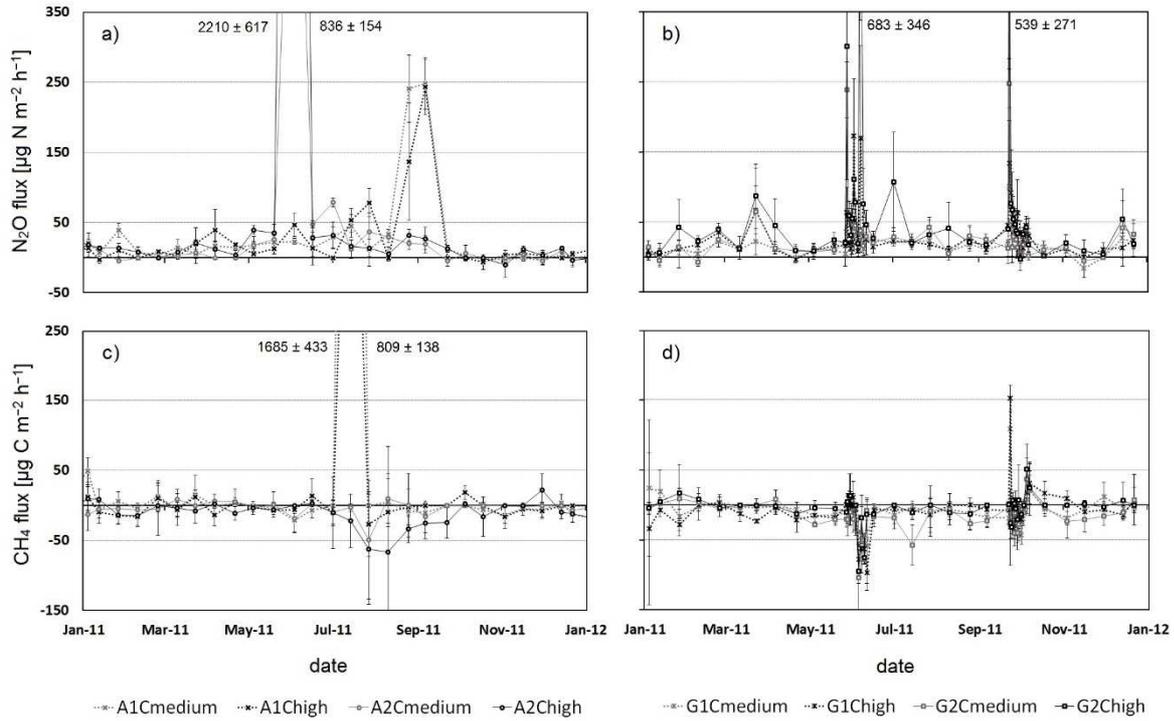
Fig. 7 Time series of measured N$_2$O fluxes (a, arable land; b, grassland) and CH$_4$ fluxes (c, arable land; d, grassland) for the year 2011.
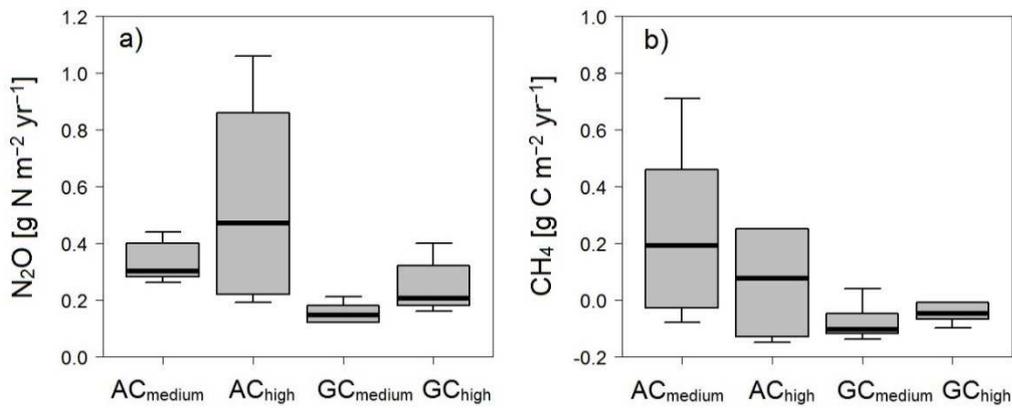
Fig. 8 Box plots of cumulative annual N$_2$O emissions (a), and cumulative annual CH$_4$ emissions for the two soil types and land-use types.
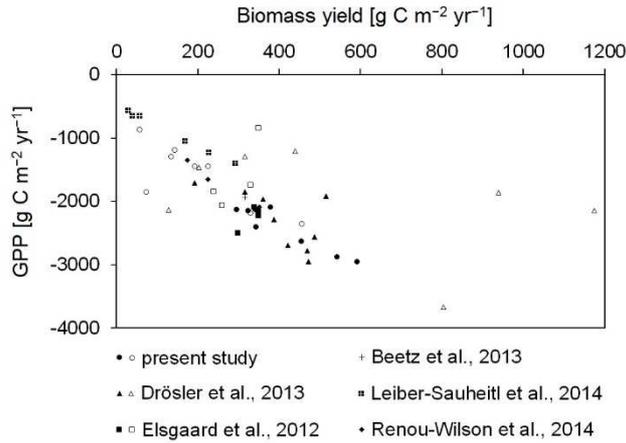
Fig. 9 Relationship of GPP and biomass export from temperate peatlands. Filled symbols represents grassland sites (intensive and extensive), unfilled symbols represents arable lands.
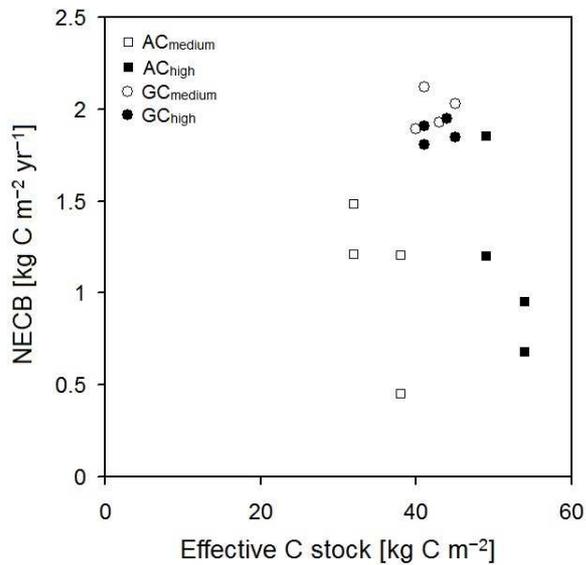
Fig. 10 NECB plotted against the effective C stock, which is defined as the fraction of aerated carbon in the soil profile (according to Leiber-Sauheitl et al., 2014) (calculated NECB did not include CH_4 losses).

SUPPLEMENT TO PUBLICATION I.
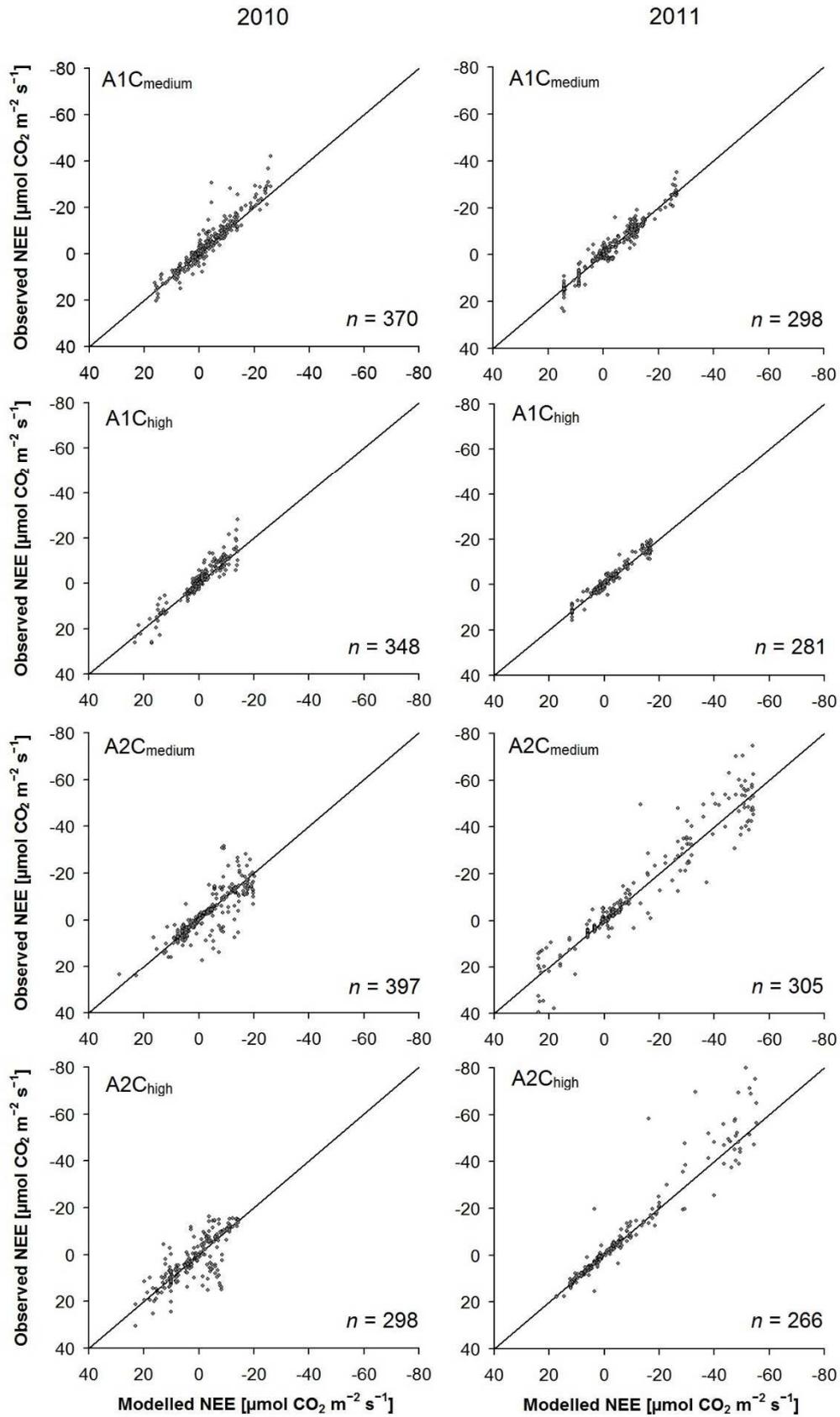
Fig. S1 Observed vs. modelled plots of NEE for the arable sites. Solid lines indicate the 1:1 line for a perfect fit.
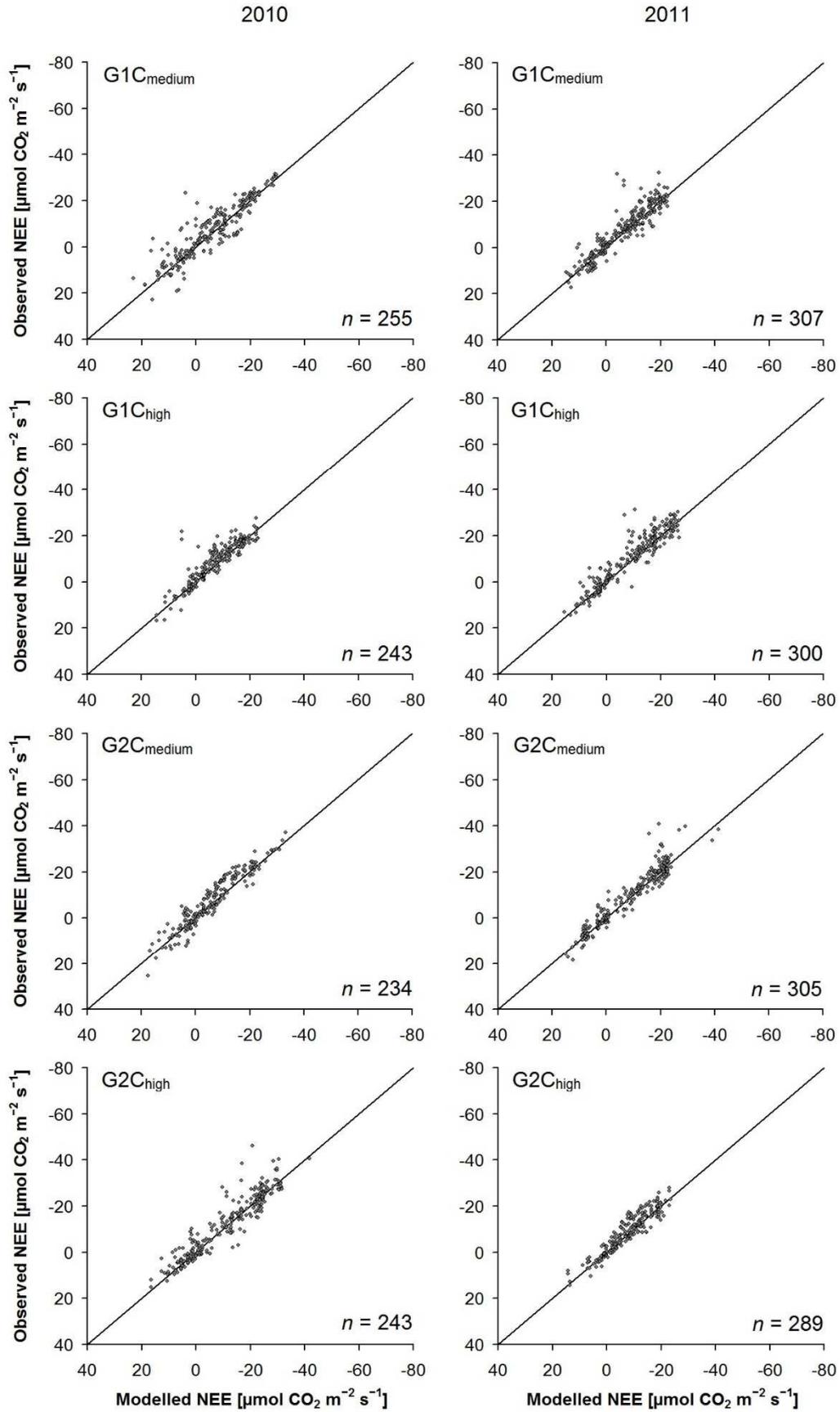
Fig. S2 Observed vs. modelled plots of NEE for the grassland sites. Solid lines indicate the 1:1 line for a perfect fit.

Appendix

2010   2011
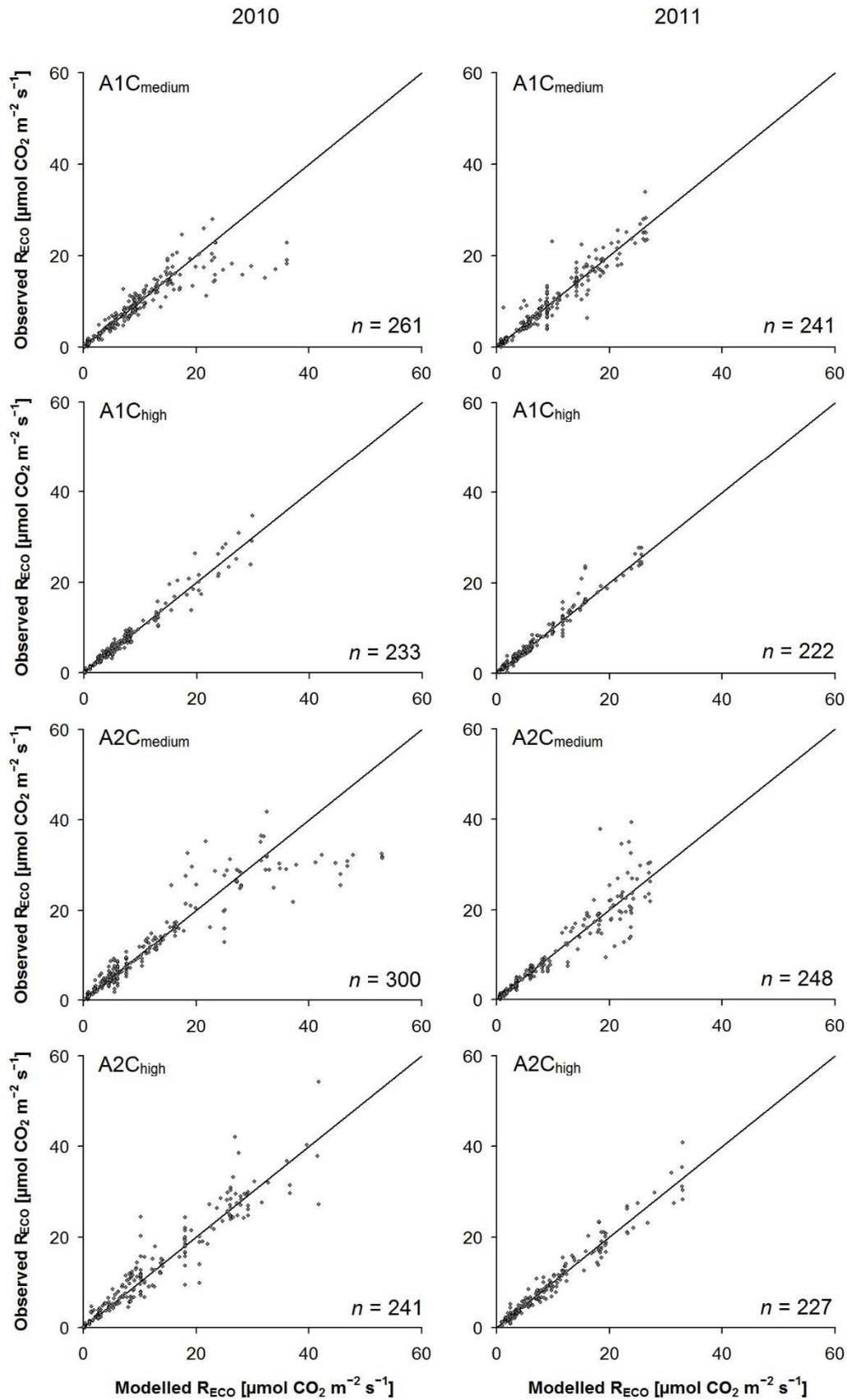
Fig. S3 Observed vs. modelled plots of R_{ECO} for the arable sites. Solid lines indicate the 1:1 line for a perfect fit.
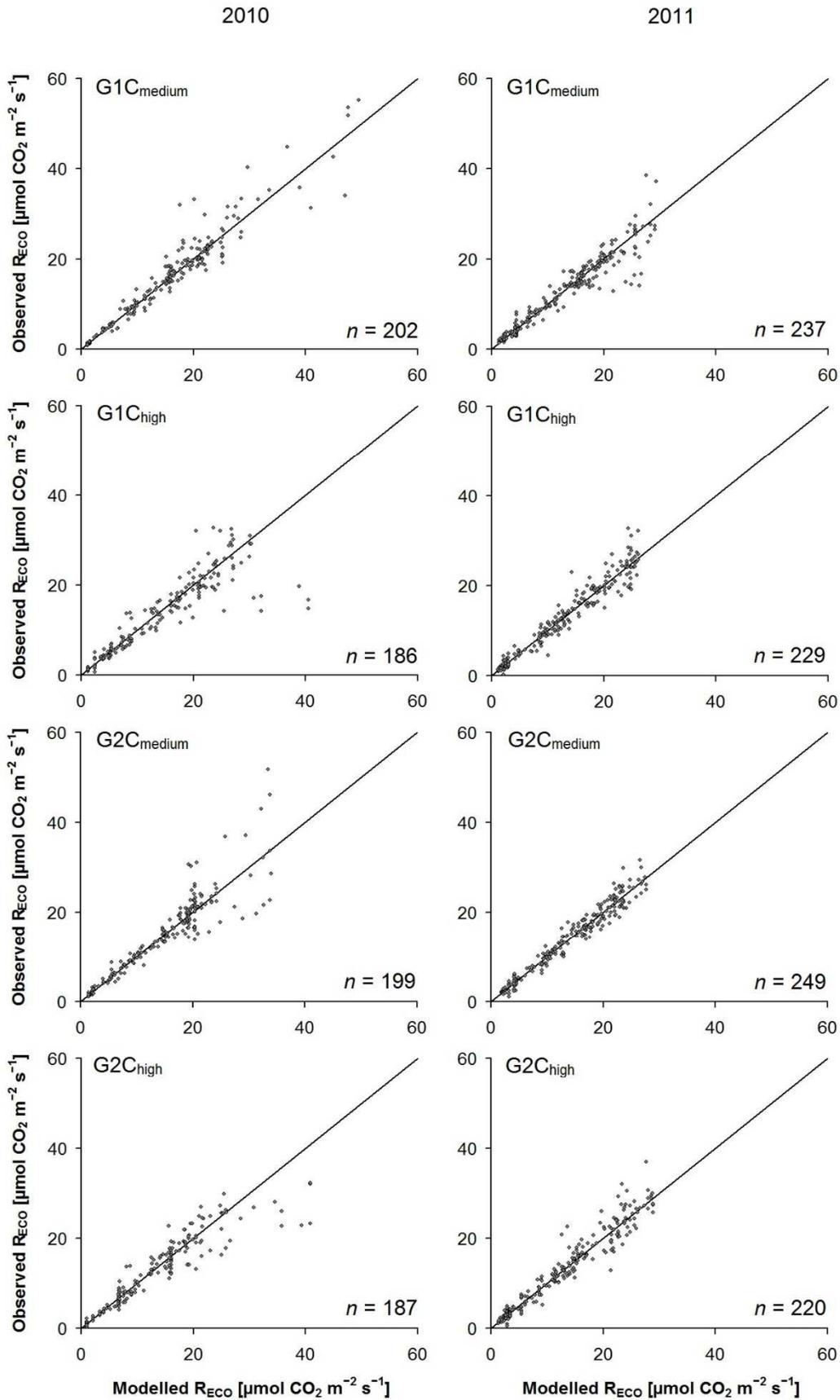
Fig. S4 Observed vs. modelled plots of R_{ECO} for the grassland sites. Solid lines indicate the 1:1 line for a perfect fit.

Table S1 Model parameters of the A1C$_{medium}$ site.

Site	Date	R_{ref}	E_0	SE R_{ref}	SE E_0	lower R_{ref}	upper R_{ref}	lower E_0	upper E_0	n	model based on	α	GPP_{2000}	SE α	SE GPP_{2000}	lower α	upper α	lower GPP_{2000}	upper GPP_{2000}	n	daily mean R_{ECO} [µmol CO$_2$ m^{-2} s^{-1}]	SE R_{ECO}	lower R_{ECO}	upper R_{ECO}	n
A1C$_{medium}$	2010-02-09	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.78	0.18	0.44	1.11	9
A1C$_{medium}$	2010-02-25	–	–	–	–	–	–	–	–	–	–	-0.01	-6.91	0.00	0.15	-0.01	-0.01	-7.22	-6.61	36	0.92	0.12	0.69	1.15	15
A1C$_{medium}$	2010-03-08	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1.11	0.09	0.95	1.28	9
A1C$_{medium}$	2010-03-24	3.43	138.31	0.06	7.64	3.32	3.55	122.73	153.89	33	T_{air}	-0.01	-10.44	0.00	0.25	-0.01	-0.01	-10.94	-9.93	55	–	–	–	–	–
A1C$_{medium}$	2010-04-07	7.99	119.04	0.26	19.17	7.43	8.55	78.41	159.67	18	ST_2	-0.02	-21.89	0.00	0.73	-0.02	-0.01	-23.37	-20.42	44	–	–	–	–	–
A1C$_{medium}$	2010-04-28	9.62	103.31	0.40	17.69	8.77	10.47	65.60	141.02	17	ST_2	-0.03	-28.78	0.00	1.00	-0.03	-0.02	-30.84	-26.73	26	–	–	–	–	–
A1C$_{medium}$	2010-05-23	6.73	231.96	0.71	35.92	5.27	8.19	157.65	306.26	25	ST_2	–	–	–	–	–	–	–	–	–	–	–	–	–	–
A1C$_{medium}$	2010-06-10	–	–	–	–	–	–	–	–	–	–	-0.01	-11.00	0.00	0.54	-0.01	-0.01	-12.11	-9.89	27	9.15	0.27	8.63	9.67	17
A1C$_{medium}$	2010-07-02	12.59	110.77	1.24	23.54	10.02	15.16	61.82	159.72	23	T_{air}	-0.03	-29.42	0.01	1.78	-0.04	-0.02	-33.03	-25.81	36	–	–	–	–	–
A1C$_{medium}$	2010-07-22	11.34	218.93	0.64	23.03	10.03	12.65	171.89	265.97	32	T_{air}	-0.06	-67.47	0.01	1.68	-0.07	-0.05	-70.94	-64.01	27	–	–	–	–	–
A1C$_{medium}$	2010-08-21	7.43	143.40	0.54	16.74	6.28	8.58	107.90	178.89	18	T_{air}	-0.04	-42.18	0.01	1.27	-0.05	-0.03	-44.83	-39.53	22	–	–	–	–	–
A1C$_{medium}$	2010-09-12	6.28	135.16	0.48	21.87	5.25	7.31	88.27	182.06	16	T_{air}	-0.01	-33.80	0.00	1.92	-0.02	-0.01	-37.80	-29.80	22	–	–	–	–	–
A1C$_{medium}$	2010-10-04	4.62	202.38	0.18	14.43	4.23	5.01	171.21	233.55	15	T_{air}	-0.05	-20.85	0.01	1.44	-0.07	-0.02	-23.98	-17.72	14	–	–	–	–	–
A1C$_{medium}$	2010-10-13	6.10	175.57	0.54	46.50	4.83	7.37	65.62	285.52	9	T_{air}	-0.01	-13.33	0.00	7.28	-0.02	0.00	-32.05	5.39	7	–	–	–	–	–
A1C$_{medium}$	2010-10-29	4.70	104.79	0.36	36.38	3.93	5.47	26.77	182.81	16	T_{air}	-0.01	-18.61	0.00	4.97	-0.01	0.00	-29.14	-8.08	18	–	–	–	–	–
A1C$_{medium}$	2010-12-13	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.20	0.06	0.09	0.31	9
A1C$_{medium}$	2011-01-05	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.82	0.14	0.56	1.09	9
A1C$_{medium}$	2011-01-27	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.75	0.02	0.72	0.79	9
A1C$_{medium}$	2011-02-07	1.28	151.19	0.09	41.66	1.09	1.48	61.20	241.19	15	T_{air}	–	–	–	–	–	–	–	–	–	–	–	–	–	–
A1C$_{medium}$	2011-03-08	1.07	32.45	0.04	10.60	0.99	1.15	10.09	54.81	19	T_{air}	–	–	–	–	–	–	–	–	–	–	–	–	–	–
A1C$_{medium}$	2011-03-29	4.16	196.91	0.18	25.69	3.78	4.53	142.93	250.89	20	T_{air}	-0.01	-14.78	0.00	1.31	-0.01	0.00	-17.53	-12.04	22	–	–	–	–	–
A1C$_{medium}$	2011-04-20	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	8.92	0.19	7.91	9.94	23
A1C$_{medium}$	2011-05-06	5.90	79.18	0.24	15.44	5.39	6.41	47.06	111.29	23	T_{air}	-0.01	-8.27	0.00	0.55	-0.02	-0.01	-9.39	-7.15	36	–	–	–	–	–
A1C$_{medium}$	2011-05-25	14.57	76.98	1.02	19.05	12.41	16.72	36.59	117.36	18	ST_2	-0.03	-34.06	0.01	1.44	-0.04	-0.02	-36.99	-31.12	32	–	–	–	–	–
A1C$_{medium}$	2011-06-28	14.88	124.68	1.12	19.41	12.56	17.20	84.43	164.93	24	T_{air}	-0.16	-52.10	0.02	0.84	-0.20	-0.12	-53.81	-50.39	30	–	–	–	–	–
A1C$_{medium}$	2011-08-02	10.82	133.18	0.67	18.45	9.41	12.23	94.26	172.10	19	T_{air}	-0.15	-29.02	0.03	0.79	-0.21	-0.08	-30.64	-27.40	30	–	–	–	–	–
A1C$_{medium}$	2011-08-24	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	14.22	0.20	13.32	15.11	19
A1C$_{medium}$	2011-09-24	7.30	177.34	0.30	20.51	6.66	7.94	133.64	221.05	17	T_{air}	-0.02	-28.26	0.00	2.35	-0.03	-0.02	-33.25	-23.28	18	–	–	–	–	–
A1C$_{medium}$	2011-10-16	9.69	207.95	0.37	22.25	8.90	10.49	160.52	255.38	17	ST_2	-0.03	-33.29	0.00	1.48	-0.03	-0.02	-36.37	-30.20	23	–	–	–	–	–
A1C$_{medium}$	2011-12-15	2.42	168.70	0.20	31.97	1.95	2.89	93.11	244.28	9	T_{air}	-0.04	-16.40	0.00	3.16	-0.05	-0.03	-24.13	-8.66	8	–	–	–	–	–
A1C$_{medium}$	2012-01-12	4.53	302.37	0.55	59.54	3.28	5.78	167.69	437.06	11	T_{air}	-0.02	-21.94	0.01	7.93	-0.03	-0.01	-39.21	-4.66	14	–	–	–	–	–

Table S2 Model parameters of the A1C$_{high}$ site.

Site	Date	R$_{ref}$	E$_0$	SE R$_{ref}$	SE E$_0$	lower R$_{ref}$	upper R$_{ref}$	lower E$_0$	upper E$_0$	n	model based on	α	GPP$_{2000}$	SE α	SE GPP$_{2000}$	lower α	upper α	lower GPP$_{2000}$	upper GPP$_{2000}$	n	daily mean R$_{ECO}$ [μmol CO$_2$ m^{-2} s^{-1}]	SE R$_{ECO}$	lower R$_{ECO}$	upper R$_{ECO}$	n
A1C$_{high}$	2010-02-09	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.30	0.14	0.05	0.56	9
A1C$_{high}$	2010-02-25	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.15	0.04	0.07	0.23	9
A1C$_{high}$	2010-03-08	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1.07	0.09	0.90	1.25	9
A1C$_{high}$	2010-03-24	2.49	76.95	0.07	13.88	2.34	2.64	48.53	105.38	30	T_{air}	0.00	-2.46	0.00	0.12	-0.01	0.00	-2.71	-2.21	44					
A1C$_{high}$	2010-04-06	4.59	81.94	0.20	23.27	4.16	5.02	32.85	131.03	20	T_{air}	0.00	-3.57	0.00	0.18	-0.01	0.00	-3.94	-3.19	30					
A1C$_{high}$	2010-04-28	5.84	155.65	0.34	33.27	5.11	6.57	84.29	227.01	16	ST_5	-0.01	-8.94	0.00	0.51	-0.01	0.00	-10.00	-7.89	24					
A1C$_{high}$	2010-05-23	2.95	106.80	0.23	27.68	2.45	3.45	47.82	165.79	17	T_{air}	–	–	–	–	–	–	–	–	–					
A1C$_{high}$	2010-06-10	1.39	404.93	0.19	36.87	0.98	1.79	327.15	482.70	19	ST_2	-0.01	-9.42	0.00	0.30	-0.01	-0.01	-10.04	-8.80	29					
A1C$_{high}$	2010-07-02	23.46	42.87	1.83	21.10	19.33	27.59	4.86	90.60	11	T_{air}	-0.02	-17.50	0.01	1.43	-0.02	0.00	-20.50	-14.50	20					
A1C$_{high}$	2010-07-21	12.61	88.94	1.14	21.86	10.20	15.02	42.83	135.05	19	T_{air}	-0.04	-37.67	0.01	1.76	-0.04	-0.02	-41.28	-34.06	30					
A1C$_{high}$	2010-08-21	6.66	146.95	0.47	18.09	5.65	7.67	107.88	186.02	15	T_{air}	-0.04	-24.30	0.01	1.27	-0.05	-0.02	-26.99	-21.60	18					
A1C$_{high}$	2010-09-12	5.73	81.71	0.33	17.09	5.01	6.44	44.79	118.63	15	T_{air}	-0.01	-20.57	0.00	1.09	-0.02	-0.01	-22.80	-18.34	30					
A1C$_{high}$	2010-10-04	4.54	203.09	0.39	36.90	3.67	5.40	120.88	285.29	12	T_{air}	-0.03	-20.40	0.01	1.25	-0.05	-0.02	-23.12	-17.69	14					
A1C$_{high}$	2010-10-13	14.00	577.38	2.83	138.77	7.48	20.51	257.36	897.39	10	ST_2	-0.02	-15.84	0.01	8.05	-0.04	0.00	-33.23	1.56	15					
A1C$_{high}$	2010-10-29	2.76	87.44	0.07	12.46	2.60	2.92	60.30	114.59	14	T_{air}	-0.01	-10.84	0.00	3.14	-0.01	0.00	-17.57	-4.10	16					
A1C$_{high}$	2010-12-13	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.07	0.03	0.03	0.12	9
A1C$_{high}$	2011-01-05	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1.85	0.33	1.23	2.47	9
A1C$_{high}$	2011-01-27	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.47	0.06	0.42	0.52	9
A1C$_{high}$	2011-02-07	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1.20	0.09	1.03	1.37	15
A1C$_{high}$	2011-03-08	1.20	64.75	0.04	13.73	1.11	1.29	35.10	94.41	15	T_{air}	–	–	–	–	–	–	–	–	–					
A1C$_{high}$	2011-03-29	3.06	250.34	0.19	35.10	2.65	3.47	175.52	325.16	17	T_{air}	-0.01	-10.48	0.00	0.46	-0.01	-0.01	-11.45	-9.51	21					
A1C$_{high}$	2011-04-20	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	2.98	0.11	2.76	3.20	24
A1C$_{high}$	2011-05-06	4.33	167.57	0.18	20.60	3.96	4.70	124.11	211.03	19	ST_5	-0.01	-5.85	0.00	0.27	-0.01	0.00	-6.39	-5.31	42					
A1C$_{high}$	2011-05-25	12.13	68.75	0.46	11.91	11.08	13.17	41.81	95.69	11	T_{air}	-0.08	-33.08	0.01	0.71	-0.10	-0.06	-34.53	-31.62	31					
A1C$_{high}$	2011-06-28	13.41	150.82	0.49	10.04	12.37	14.45	129.64	171.99	19	T_{air}	-0.10	-41.08	0.01	0.82	-0.12	-0.07	-42.80	-39.37	23					
A1C$_{high}$	2011-08-02	6.49	96.67	0.34	13.66	5.77	7.21	67.98	125.36	20	T_{air}	-0.01	-7.74	0.00	0.26	-0.02	-0.01	-8.28	-7.21	22					
A1C$_{high}$	2011-08-24	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	11.69	0.10	10.86	12.52	17
A1C$_{high}$	2011-09-24	4.43	137.92	0.27	28.27	3.85	5.01	77.29	198.55	16	T_{air}	-0.03	-17.13	0.00	0.71	-0.04	-0.02	-18.64	-15.63	19					
A1C$_{high}$	2011-10-16	5.71	86.06	0.14	12.31	5.42	6.00	59.82	112.29	17	T_{air}	-0.01	-14.30	0.00	0.89	-0.02	-0.01	-16.16	-12.43	21					
A1C$_{high}$	2011-12-15	–	–	–	–	–	–	–	–	–	–	-0.03	-12.11	0.00	2.21	-0.03	-0.02	-17.78	-6.43	11	0.41	0.05	0.32	0.49	9
A1C$_{high}$	2012-01-12	1.03	301.93	0.11	68.91	0.79	1.27	153.06	450.81	15	T_{air}	-0.01	-11.43	0.00	3.50	-0.01	0.00	-18.86	-4.01	18	–	–	–	–	–

Table S3 Model parameters of the A2C$_{medium}$ site.

Site	Date	R_{ref}	E_0	SE R_{ref}	SE E_0	lower R_{ref}	upper R_{ref}	lower E_0	upper E_0	n	model based on	α	GPP_{2000}	SE α	SE GPP_{2000}	lower α	upper α	lower GPP_{2000}	upper GPP_{2000}	n	daily mean R_{eco} [µmol CO$_2$ m^{-2} s^{-1}]	SE R_{eco}	lower R_{eco}	upper R_{eco}	n
A2C$_{medium}$	2010-02-09	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.78	0.18	0.44	1.11	9
A2C$_{medium}$	2010-02-25	–	–	–	–	–	–	–	–	–	–	-0.01	-6.91	0.00	0.15	-0.01	-0.01	-7.22	-6.61	36	0.92	0.12	0.69	1.15	15
A2C$_{medium}$	2010-03-08	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1.11	0.09	0.95	1.28	9
A2C$_{medium}$	2010-03-24	3.43	138.31	0.06	7.64	3.32	3.55	122.73	153.89	33	T_{air}	-0.01	-10.44	0.00	0.25	-0.01	-0.01	-10.94	-9.93	55	–	–	–	–	–
A2C$_{medium}$	2010-04-06	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	5.48	0.55	4.44	6.53	15
A2C$_{medium}$	2010-04-07	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	4.53	0.30	3.96	5.09	19
A2C$_{medium}$	2010-04-28	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	7.68	0.33	7.04	8.32	18
A2C$_{medium}$	2010-05-23	4.94	239.51	0.27	18.32	4.39	5.49	201.62	277.40	25	T_{air}	-0.06	-24.05	0.00	0.25	-0.07	-0.05	-24.57	-23.54	38	–	–	–	–	–
A2C$_{medium}$	2010-06-10	12.65	48.50	0.72	13.26	11.10	14.21	20.06	76.95	16	T_{air}	-0.08	-34.35	0.02	1.44	-0.12	-0.04	-37.31	-31.38	26	–	–	–	–	–
A2C$_{medium}$	2010-07-02	15.39	105.33	1.14	15.77	13.00	17.78	72.21	138.45	20	T_{air}	-0.15	-47.21	0.03	1.56	-0.20	-0.09	-50.37	-44.04	36	–	–	–	–	–
A2C$_{medium}$	2010-07-22	13.58	328.79	2.34	71.82	8.48	18.68	172.29	485.28	14	ST_2	-0.08	-54.20	0.01	1.64	-0.10	-0.05	-57.70	-50.70	17	–	–	–	–	–
A2C$_{medium}$	2010-08-21	9.82	55.31	0.90	22.33	7.92	11.72	7.98	102.65	18	T_{air}	-0.02	-12.72	0.00	0.40	-0.03	-0.01	-13.57	-11.87	20	–	–	–	–	–
A2C$_{medium}$	2010-09-12	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	6.13	0.40	5.37	6.90	16
A2C$_{medium}$	2010-10-04	3.35	114.22	0.23	26.44	2.86	3.84	56.61	171.84	14	T_{air}	-0.01	-9.40	0.00	1.31	-0.01	0.00	-12.31	-6.48	12	–	–	–	–	–
A2C$_{medium}$	2010-10-29	3.51	109.14	0.10	14.11	3.29	3.72	79.07	139.21	17	T_{air}	–	–	–	–	–	–	–	–	–	–	–	–	–	–
A2C$_{medium}$	2010-12-13	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.17	0.01	0.15	0.19	9
A2C$_{medium}$	2011-01-05	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.70	0.16	0.40	1.00	9
A2C$_{medium}$	2011-01-27	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1.02	0.04	0.95	1.09	9
A2C$_{medium}$	2011-02-07	1.27	119.11	0.09	39.12	1.07	1.47	34.59	203.63	15	T_{air}	0.00	-1.50	0.00	0.67	0.00	0.00	-2.87	-0.13	34	–	–	–	–	–
A2C$_{medium}$	2011-03-08	1.63	110.71	0.05	13.91	1.34	1.75	47.28	163.19	19	T_{air}	–	–	–	–	–	–	–	–	–	–	–	–	–	–
A2C$_{medium}$	2011-03-29	4.30	237.53	0.17	24.85	3.94	4.66	185.33	289.74	20	T_{air}	-0.01	-15.21	0.00	1.28	-0.01	-0.01	-17.85	-12.57	26	–	–	–	–	–
A2C$_{medium}$	2011-04-20	17.78	81.14	2.43	44.53	12.71	22.84	-11.75	174.02	22	T_{air}	–	–	–	–	–	–	–	–	–	–	–	–	–	–
A2C$_{medium}$	2011-05-06	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	3.39	0.07	3.08	3.70	23
A2C$_{medium}$	2011-05-25	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	6.06	0.15	5.70	6.41	22
A2C$_{medium}$	2011-06-28	9.05	151.48	0.74	20.79	7.51	10.58	108.36	194.59	24	T_{air}	-0.06	-50.00	0.01	1.61	-0.08	-0.04	-53.29	-46.71	30	–	–	–	–	–
A2C$_{medium}$	2011-08-02	9.67	215.74	1.08	31.17	7.40	11.93	150.50	280.99	21	T_{air}	-0.14	-79.49	0.03	3.32	-0.21	-0.07	-86.28	-72.69	30	–	–	–	–	–
A2C$_{medium}$	2011-08-24	15.56	116.88	1.89	30.61	11.58	19.55	52.29	181.46	19	T_{air}	-0.12	-85.43	0.02	4.04	-0.17	-0.08	-93.67	-77.19	33	–	–	–	–	–
A2C$_{medium}$	2011-09-24	4.95	218.83	0.18	17.84	4.56	5.33	180.82	256.85	17	T_{air}	-0.01	-26.59	0.00	3.01	-0.02	-0.01	-32.89	-20.30	21	–	–	–	–	–
A2C$_{medium}$	2011-10-16	2.65	136.11	0.05	14.55	2.54	2.76	105.41	166.81	19	T_{air}	-0.01	-7.35	0.00	0.80	-0.02	0.00	-9.02	-5.68	23	–	–	–	–	–
A2C$_{medium}$	2011-12-15	1.91	202.88	0.18	39.64	1.47	2.35	109.14	296.62	9	T_{air}	-0.01	-4.35	0.00	1.50	-0.02	-0.01	-8.02	-0.68	8	–	–	–	–	–
A2C$_{medium}$	2012-01-12	2.35	256.47	0.22	44.32	1.84	2.85	156.21	356.73	11	T_{air}	-0.01	-6.71	0.00	0.47	-0.01	-0.01	-7.75	-5.68	14	–	–	–	–	–

Table S4 Model parameters of the A2C$_{high}$ site.

Site	Date	R_{ref}	E_0	Std.Error R_{ref}	Std.Error E_0	lower R_{ref}	upper R_{ref}	lower E_0	upper E_0	n	model based on	α	GPP_{2000}	Std.Error α	Std.Error GPP_{2000}	lower α	upper α	lower GPP_{2000}	upper GPP_{2000}	n	daily mean R_{CO_2} [$\mu mol\ CO_2\ m^{-2}\ s^{-1}$]	Std.Error R_{CO_2}	lower R_{CO_2}	upper R_{CO_2}	n
A2C$_{high}$	2010-02-09	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.30	0.14	0.05	0.56	9
A2C$_{high}$	2010-02-25	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.15	0.04	0.07	0.23	9
A2C$_{high}$	2010-03-08	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1.07	0.09	0.90	1.25	9
A2C$_{high}$	2010-03-24	2.49	76.95	0.07	13.88	2.34	2.64	48.53	105.38	30	T_{air}	0.00	-2.46	0.00	0.12	-0.01	0.00	-2.71	-2.21	44	–	–	–	–	–
A2C$_{high}$	2010-04-06	7.03	144.14	0.56	52.92	5.85	8.21	32.50	255.78	19	T_{air}	–	–	–	–	–	–	–	–	–	–	–	–	–	–
A2C$_{high}$	2010-04-28	8.78	113.36	0.83	33.95	6.97	10.60	39.39	187.32	14	T_{air}	-0.01	-8.95	0.00	0.61	-0.02	-0.01	-10.24	-7.65	17	–	–	–	–	–
A2C$_{high}$	2010-05-23	2.42	397.23	0.39	46.67	1.60	3.24	300.17	494.29	23	T_{air}	-0.04	-22.39	0.00	0.34	-0.04	-0.03	-23.08	-21.70	30	–	–	–	–	–
A2C$_{high}$	2010-06-10	3.81	480.26	0.51	35.58	2.73	4.89	405.19	555.33	19	ST_2	-0.07	-31.57	0.01	0.75	-0.09	-0.05	-33.11	-30.02	28	–	–	–	–	–
A2C$_{high}$	2010-07-02	13.84	278.17	2.24	51.41	9.14	18.55	170.16	386.19	20	ST_2	-0.08	-42.19	0.01	1.45	-0.11	-0.05	-45.15	-39.23	32	–	–	–	–	–
A2C$_{high}$	2010-07-21	20.09	61.88	1.20	14.67	17.55	22.64	30.77	92.99	18	T_{air}	-0.01	-19.72	0.00	1.97	-0.02	-0.01	-23.79	-15.64	25	–	–	–	–	–
A2C$_{high}$	2010-08-21	–	–	–	–	–	–	–	–	–	–	-0.09	-23.80	0.04	2.05	-0.18	0.01	-28.44	-19.16	11	18.05	1.07	16.19	19.92	14
A2C$_{high}$	2010-09-12	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	10.17	0.68	8.88	11.46	14
A2C$_{high}$	2010-10-04	4.94	277.42	0.90	77.83	2.96	6.91	106.12	448.72	13	T_{air}	-0.02	-19.00	0.00	1.64	-0.03	-0.01	22.56	-15.43	14	–	–	–	–	–
A2C$_{high}$	2010-10-13	7.38	87.46	0.69	41.44	5.76	9.01	10.52	185.44	9	T_{air}	–	–	–	–	–	–	–	–	–	–	–	–	–	–
A2C$_{high}$	2010-10-29	5.12	127.75	0.25	26.17	4.57	5.67	70.73	184.78	14	T_{air}	-0.01	-30.81	0.00	11.22	-0.02	-0.01	-55.04	-6.58	15	–	–	–	–	–
A2C$_{high}$	2010-12-13	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.04	0.01	0.01	0.06	9
A2C$_{high}$	2011-01-05	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	2.42	0.53	1.45	3.40	9
A2C$_{high}$	2011-01-27	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1.96	0.47	1.65	2.26	9
A2C$_{high}$	2011-02-07	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1.55	0.07	1.43	1.68	15
A2C$_{high}$	2011-03-08	2.62	114.38	0.13	17.76	2.34	2.90	75.68	153.08	14	T_{air}	0.00	-0.70	0.00	0.11	0.00	0.00	-0.95	-0.45	13	–	–	–	–	–
A2C$_{high}$	2011-03-29	4.63	234.03	0.38	44.50	3.82	5.43	139.69	328.37	18	T_{air}	-0.02	-15.01	0.00	0.70	-0.02	-0.01	-16.46	-13.56	23	–	–	–	–	–
A2C$_{high}$	2011-04-20	8.78	94.88	0.40	14.88	7.96	9.61	64.18	125.58	26	T_{air}	–	–	–	–	–	–	–	–	–	–	–	–	–	–
A2C$_{high}$	2011-05-06	4.69	69.13	0.17	13.36	4.34	5.05	40.94	97.32	19	T_{air}	–	–	–	–	–	–	–	–	–	–	–	–	–	–
A2C$_{high}$	2011-05-25	6.91	84.23	0.18	9.25	6.53	7.30	64.61	103.85	18	T_{air}	0.00	-1.25	0.00	0.14	-0.01	0.00	-1.54	-0.97	31	–	–	–	–	–
A2C$_{high}$	2011-06-28	9.92	155.21	0.57	15.63	8.71	11.13	122.24	188.18	19	T_{air}	-0.04	-39.75	0.01	1.17	-0.06	-0.03	-33.97	-42.21	20	–	–	–	–	–
A2C$_{high}$	2011-08-02	8.32	177.09	1.21	35.68	5.77	10.86	102.13	252.04	20	T_{air}	-0.11	-69.31	0.03	2.86	-0.17	-0.05	-75.28	-63.34	22	–	–	–	–	–
A2C$_{high}$	2011-08-24	17.70	120.30	2.27	29.64	12.80	22.59	56.27	184.33	15	T_{air}	-0.10	-97.68	0.03	7.84	-0.16	-0.05	-113.87	-81.50	26	–	–	–	–	–
A2C$_{high}$	2011-09-24	4.90	203.68	0.45	39.86	3.94	5.85	118.71	288.64	17	T_{air}	-0.02	-23.07	0.00	2.18	-0.03	-0.01	-27.68	-18.47	19	–	–	–	–	–
A2C$_{high}$	2011-10-16	3.47	108.54	0.10	17.64	3.26	3.67	71.31	145.76	18	T_{air}	-0.01	-9.84	0.00	0.93	-0.01	-0.01	-11.78	-7.90	22	–	–	–	–	–
A2C$_{high}$	2011-12-15	3.50	98.86	0.36	41.42	2.66	4.35	0.92	196.81	9	T_{air}	-0.03	-3.51	0.01	0.75	-0.06	0.00	-5.44	-1.58	9	–	–	–	–	–
A2C$_{high}$	2012-01-12	1.80	144.11	0.11	29.54	1.55	2.05	80.29	207.93	15	T_{air}	-0.01	-8.57	0.00	2.28	-0.01	0.00	-13.40	-3.74	18	–	–	–	–	–

Table S5 Model parameters of the G1C$_{medium}$ site.

Site	Date	R_{ref}	E_0	SE R_{ref}	SE E_0	lower R_{ref}	upper R_{ref}	lower E_0	upper E_0	n	model based on	α	GPP_{2000}	SE α	SE GPP_{2000}	lower α	upper α	lower GPP_{2000}	upper GPP_{2000}	n	daily mean R_{ECO} [µmol CO$_2$ m^{-2} s^{-1}]	R_{ECO}	lower R_{ECO}	upper R_{ECO}	n
G1C$_{medium}$	2010-02-09	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1.59	0.06	1.48	1.70	9
G1C$_{medium}$	2010-03-08	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1.49	0.03	1.44	1.54	9
G1C$_{medium}$	2010-03-24	5.56	51.60	0.13	10.16	5.26	5.86	27.59	75.62	9	T_{air}	-0.01	-17.44	0.00	3.41	-0.02	-0.01	-24.86	-10.02	14	–	–	–	–	–
G1C$_{medium}$	2010-04-06	6.96	445.87	0.35	76.92	6.19	7.72	279.70	612.04	15	ST_2	-0.01	-36.01	0.00	5.56	-0.01	0.00	-47.61	-24.41	22	–	–	–	–	–
G1C$_{medium}$	2010-04-28	15.08	181.18	0.90	33.49	13.16	16.99	109.80	252.56	17	ST_2	-0.05	-43.75	0.00	1.19	-0.06	-0.04	-46.26	-41.24	20	–	–	–	–	–
G1C$_{medium}$	2010-05-24	10.99	301.17	0.51	29.03	9.92	12.06	239.63	362.70	18	ST_2	-0.07	-47.56	0.00	0.56	-0.08	-0.06	-48.73	-46.38	22	–	–	–	–	–
G1C$_{medium}$	2010-06-09	11.29	154.84	1.05	26.62	9.10	13.48	99.13	210.56	21	ST_2	-0.07	-31.18	0.01	0.79	-0.09	-0.05	-32.81	-29.56	28	–	–	–	–	–
G1C$_{medium}$	2010-07-07	5.76	467.71	1.28	74.30	3.05	8.47	310.95	624.47	19	ST_5	-0.04	-31.64	0.01	1.37	-0.06	-0.02	-34.51	-28.77	21	–	–	–	–	–
G1C$_{medium}$	2010-07-21	15.52	275.53	2.15	41.54	10.99	20.05	188.26	362.80	20	ST_2	-0.05	-36.69	0.02	2.80	-0.08	-0.02	-42.45	-30.93	27	–	–	–	–	–
G1C$_{medium}$	2010-08-22	15.85	72.27	0.52	7.49	14.73	16.97	56.20	88.34	16	T_{air}	-0.09	-43.62	0.01	1.23	-0.12	-0.06	-46.19	-41.04	21	–	–	–	–	–
G1C$_{medium}$	2010-09-11	9.99	146.57	0.40	14.49	9.14	10.84	116.01	177.13	19	T_{air}	-0.10	-40.60	0.01	0.70	-0.12	-0.08	-42.05	-39.15	26	–	–	–	–	–
G1C$_{medium}$	2010-10-29	7.76	135.15	0.44	33.85	6.81	8.70	62.02	208.28	15	T_{air}	-0.02	-39.42	0.00	7.00	-0.03	-0.01	-54.27	-24.57	18	–	–	–	–	–
G1C$_{medium}$	2010-12-16	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1.05	0.06	0.94	1.17	9
G1C$_{medium}$	2011-01-05	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	4.47	0.61	3.35	5.59	9
G1C$_{medium}$	2011-01-27	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	2.15	0.12	1.94	2.37	9
G1C$_{medium}$	2011-02-07	2.45	88.93	0.11	22.55	2.20	2.69	39.80	138.06	14	T_{air}	0.00	-1.86	0.00	0.95	-0.01	0.00	-4.30	0.57	7	–	–	–	–	–
G1C$_{medium}$	2011-03-09	6.23	275.39	1.00	52.83	4.13	8.34	164.39	386.38	20	ST_2	0.00	-4.13	0.00	0.32	-0.01	0.00	-4.80	-3.46	23	–	–	–	–	–
G1C$_{medium}$	2011-03-29	7.86	273.58	0.32	36.32	7.19	8.53	196.94	350.21	19	ST_2	-0.02	-20.96	0.00	0.71	-0.02	-0.02	-22.42	-19.50	28	–	–	–	–	–
G1C$_{medium}$	2011-04-20	12.07	108.27	0.54	16.87	10.95	13.19	72.97	143.58	21	T_{air}	-0.04	-31.53	0.01	1.33	-0.06	-0.03	-34.26	-28.80	30	–	–	–	–	–
G1C$_{medium}$	2011-05-10	17.04	139.74	0.59	14.65	15.81	18.28	109.08	170.39	21	ST_2	-0.04	-37.61	0.01	1.32	-0.05	-0.03	-40.31	-34.91	31	–	–	–	–	–
G1C$_{medium}$	2011-06-21	7.30	428.58	1.09	89.72	4.64	9.96	209.05	648.11	12	ST_{10}	-0.08	-45.56	0.01	1.71	-0.11	-0.06	-49.22	-41.90	16	–	–	–	–	–
G1C$_{medium}$	2011-06-28	15.68	132.75	1.68	27.44	12.17	19.19	75.31	190.19	21	T_{air}	-0.08	-45.52	0.01	0.98	-0.10	-0.06	-47.53	-43.51	30	–	–	–	–	–
G1C$_{medium}$	2011-07-26	13.64	100.63	0.63	14.36	12.35	14.93	70.98	130.27	26	T_{air}	-0.09	-41.98	0.01	0.91	-0.11	-0.07	-43.82	-40.14	35	–	–	–	–	–
G1C$_{medium}$	2011-08-17	15.17	77.85	0.57	10.18	13.97	16.38	56.36	99.34	19	T_{air}	-0.08	-42.61	0.01	0.93	-0.10	-0.06	-44.51	-40.71	29	–	–	–	–	–
G1C$_{medium}$	2011-09-16	13.34	52.50	0.30	7.79	12.70	13.98	35.99	69.01	18	T_{air}	-0.03	-11.37	0.01	1.05	-0.05	0.00	-13.55	-9.19	22	–	–	–	–	–
G1C$_{medium}$	2011-10-16	9.60	177.01	0.20	19.33	9.17	10.02	136.22	217.80	19	ST_2	-0.03	-40.11	0.00	2.74	-0.04	-0.02	-45.83	-34.39	22	–	–	–	–	–
G1C$_{medium}$	2011-12-15	–	–	–	–	–	–	–	–	–	–	-0.07	-10.65	0.03	3.29	-0.15	0.01	-18.71	-2.58	8	4.19	0.31	3.61	4.77	9
G1C$_{medium}$	2012-01-12	10.80	447.23	2.23	73.24	5.95	15.66	287.65	606.81	14	ST_2	-0.02	-15.99	0.01	8.09	-0.03	0.00	-33.63	1.64	14	–	–	–	–	–

Table S6 Model parameters of the G1C$_{high}$ site.

Site	Date	R_{ref}	E_0	Std.Err R_{ref}	Std.Err E_0	lower R_{ref}	upper R_{ref}	lower E_0	upper E_0	n	model based on	a	GPP_{max}	Std.Err a	Std.Err GPP_{2000}	lower a	upper a	lower GPP_{2000}	upper GPP_{2000}	n	daily mean R_{ECO} [µmol CO$_2$ m^{-2} s^{-1}]	Std.Err R_{ECO}	lower R_{ECO}	upper R_{ECO}	n
G1C$_{high}$	2010-02-09	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1.26	0.88	1.02	1.50	6
G1C$_{high}$	2010-03-08	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1.21	0.09	1.05	1.38	9
G1C$_{high}$	2010-03-24	4.87	40.00	0.20	13.63	4.45	5.28	10.95	69.06	17	T_{air}	-0.02	-8.91	0.01	0.47	-0.03	-0.01	-9.88	-7.94	27	–	–	–	–	–
G1C$_{high}$	2010-04-06	7.75	110.35	0.61	38.27	6.46	9.05	28.78	191.92	17	T_{air}	-0.02	-25.30	0.00	1.27	-0.02	-0.01	-27.94	-22.65	22	–	–	–	–	–
G1C$_{high}$	2010-04-28	11.39	63.78	0.53	16.12	10.26	12.53	29.20	98.36	16	T_{air}	-0.03	-36.67	0.01	1.48	-0.04	-0.02	-39.79	-33.55	20	–	–	–	–	–
G1C$_{high}$	2010-05-24	8.94	351.02	0.72	52.71	7.36	10.51	235.00	467.04	13	ST_2	-0.04	-51.43	0.00	1.31	-0.05	-0.03	-54.17	-48.68	20	–	–	–	–	–
G1C$_{high}$	2010-06-09	15.58	91.65	1.92	33.23	11.54	19.61	21.83	161.48	20	T_{air}	-0.10	-36.43	0.01	0.66	-0.12	-0.08	-37.78	-35.08	27	–	–	–	–	–
G1C$_{high}$	2010-07-07	15.79	85.17	0.99	20.65	13.65	17.93	40.56	129.78	15	T_{air}	-0.06	-39.26	0.01	0.98	-0.07	-0.04	-41.32	-37.20	21	–	–	–	–	–
G1C$_{high}$	2010-07-21	16.78	107.93	1.38	18.93	13.69	19.86	65.75	150.10	12	T_{air}	-0.07	-38.41	0.01	1.40	-0.09	-0.04	-41.43	-35.39	15	–	–	–	–	–
G1C$_{high}$	2010-08-22	20.25	58.29	1.73	19.61	16.57	23.94	16.49	100.09	17	T_{air}	-0.08	-46.19	0.01	1.45	-0.11	-0.06	-49.20	-43.18	23	–	–	–	–	–
G1C$_{high}$	2010-09-11	13.45	116.83	0.54	14.92	12.29	14.60	85.04	148.63	17	T_{air}	-0.10	-39.34	0.01	0.55	-0.11	-0.08	-40.47	-38.22	29	–	–	–	–	–
G1C$_{high}$	2010-10-29	6.24	111.28	0.19	14.32	5.83	6.64	80.34	142.21	15	T_{air}	-0.02	-36.79	0.00	4.99	-0.03	-0.02	-47.56	-26.02	15	–	–	–	–	–
G1C$_{high}$	2010-12-16	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	2.45	0.46	1.60	3.30	9
G1C$_{high}$	2011-01-05	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	2.13	0.45	1.35	2.91	9
G1C$_{high}$	2011-01-27	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	2.16	0.18	1.82	2.50	9
G1C$_{high}$	2011-02-07	2.23	113.41	0.14	32.98	1.93	2.52	43.49	183.34	18	T_{air}	0.00	-5.11	0.00	0.73	0.00	0.00	-6.68	-3.54	15	–	–	–	–	–
G1C$_{high}$	2011-03-08	2.45	82.00	0.16	21.30	2.10	2.80	36.61	127.40	17	T_{air}	0.00	-4.85	0.00	0.59	0.00	0.00	-6.10	-3.61	19	–	–	–	–	–
G1C$_{high}$	2011-03-29	8.98	159.99	0.42	25.77	8.10	9.86	105.61	214.37	19	T_{air}	-0.03	-25.07	0.00	0.85	-0.04	-0.02	-26.83	-23.31	24	–	–	–	–	–
G1C$_{high}$	2011-04-20	12.36	339.75	0.72	53.59	10.83	13.88	227.16	452.35	20	ST_2	-0.04	-40.50	0.00	1.27	-0.05	-0.03	-43.11	-37.89	27	–	–	–	–	–
G1C$_{high}$	2011-05-10	15.68	112.87	1.13	21.06	13.32	18.05	68.94	156.80	22	T_{air}	-0.10	-51.21	0.01	0.80	-0.11	-0.08	-52.85	-49.58	29	–	–	–	–	–
G1C$_{high}$	2011-06-21	7.49	416.41	1.63	91.65	3.87	11.12	212.20	620.62	12	ST_2	-0.07	-48.31	0.01	1.71	-0.09	-0.06	-51.96	-44.66	17	–	–	–	–	–
G1C$_{high}$	2011-06-28	17.60	83.62	0.88	12.42	15.78	19.41	57.86	109.39	24	T_{air}	-0.13	-43.58	0.02	0.78	-0.16	-0.10	-45.18	-41.99	32	–	–	–	–	–
G1C$_{high}$	2011-07-26	9.48	355.18	0.68	39.94	8.05	10.90	271.57	438.78	21	ST_2	-0.09	-43.69	0.01	0.79	-0.11	-0.08	-45.29	-42.08	35	–	–	–	–	–
G1C$_{high}$	2011-08-18	11.24	219.60	1.16	34.20	8.80	13.68	147.74	291.46	20	ST_2	-0.09	-42.79	0.01	0.68	-0.11	-0.07	-44.19	-41.39	28	–	–	–	–	–
G1C$_{high}$	2011-09-16	7.45	285.14	0.49	48.61	6.42	8.48	183.02	387.26	20	ST_5	-0.01	-9.15	0.01	1.06	-0.03	0.00	-11.35	-6.95	24	–	–	–	–	–
G1C$_{high}$	2011-10-16	8.24	437.52	0.60	113.76	6.85	9.62	175.19	699.84	10	ST_5	-0.03	-27.09	0.00	1.05	-0.03	-0.02	-29.37	-24.81	14	–	–	–	–	–
G1C$_{high}$	2011-12-15	–	–	–	–	–	–	–	–	–	–	-0.06	-15.15	0.01	3.28	-0.08	-0.04	22.91	-7.39	9	2.98	0.31	2.40	3.55	9
G1C$_{high}$	2012-01-12	13.31	525.53	3.73	127.91	5.18	21.45	246.82	804.23	14	ST_2	-0.03	-23.67	0.00	1.76	-0.03	-0.02	-27.51	-19.82	14	–	–	–	–	–

Table S7 Model parameters of the G2C_medium site.

Site	Date	R_ref	E_0	Std.Err R_ref	Std.Err E_0	lower R_ref	upper R_ref	lower E_0	upper E_0	n	model based on	α	GPP_2000	Std.Err α	Std.Err GPP_2000	lower α	upper α	lower GPP_2000	upper GPP_2000	n	daily mean R_ECO [µmol CO₂ m⁻² s⁻¹]	Std.Err R_ECO	lower R_ECO	upper R_ECO	n
G2C_medium	2010-02-09	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1.53	0.09	1.36	1.71	6
G2C_medium	2010-03-08	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	2.14	0.16	1.85	2.43	9
G2C_medium	2010-03-24	5.65	72.15	0.12	11.65	5.36	5.94	44.61	99.69	9	T_{air}	-0.01	-16.82	0.00	3.81	-0.01	-0.01	-25.06	-8.58	15	–	–	–	–	–
G2C_medium	2010-04-06	7.19	419.60	0.26	47.88	6.63	7.74	318.09	521.11	18	ST_2	-0.01	-34.75	0.00	4.77	-0.01	0.00	-44.69	-24.80	22	–	–	–	–	–
G2C_medium	2010-04-28	11.73	191.24	0.95	28.69	9.71	13.74	130.09	252.39	17	T_{air}	-0.04	-45.59	0.01	3.03	-0.05	-0.02	-51.99	-39.20	19	–	–	–	–	–
G2C_medium	2010-05-24	13.14	93.72	0.52	11.74	12.03	14.25	68.84	118.61	18	T_{air}	-0.08	-52.31	0.01	1.03	-0.10	-0.07	-54.44	-50.17	22	–	–	–	–	–
G2C_medium	2010-06-09	11.75	109.05	1.14	25.24	9.31	14.18	54.91	163.19	16	T_{air}	-0.07	-26.29	0.01	0.79	-0.09	-0.04	-27.93	-24.66	25	–	–	–	–	–
G2C_medium	2010-07-07	–	–	–	–	–	–	–	–	–	–	-0.14	-31.20	0.04	1.30	-0.22	-0.06	-33.93	-28.46	20	20.37	0.86	18.72	22.01	18
G2C_medium	2010-07-21	19.26	112.06	1.64	21.33	14.99	23.53	56.20	167.93	20	T_{air}	-0.05	-32.41	0.02	4.16	-0.09	0.00	-41.02	-23.80	25	–	–	–	–	–
G2C_medium	2010-08-22	11.75	273.86	1.02	37.89	9.59	13.91	193.53	354.19	18	ST_5	-0.10	-42.30	0.01	0.80	-0.12	-0.08	-43.97	-40.63	21	–	–	–	–	–
G2C_medium	2010-09-11	10.70	106.79	0.41	14.14	9.84	11.57	76.96	136.63	19	T_{air}	-0.12	-39.73	0.01	0.52	-0.14	-0.10	-40.81	-38.64	26	–	–	–	–	–
G2C_medium	2010-10-29	8.51	222.39	0.48	40.84	7.48	9.55	134.17	310.61	15	ST_2	-0.03	-42.97	0.00	5.23	-0.03	-0.02	-54.01	-31.94	19	–	–	–	–	–
G2C_medium	2010-12-16	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1.44	0.12	1.22	1.66	9
G2C_medium	2011-01-05	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	3.26	0.40	2.52	4.01	9
G2C_medium	2011-01-27	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	2.30	0.09	2.14	2.46	9
G2C_medium	2011-02-07	2.31	29.03	0.05	9.46	2.20	2.42	8.20	49.86	13	T_{air}	0.00	-3.29	0.00	2.12	-0.01	0.00	-7.88	1.29	15	–	–	–	–	–
G2C_medium	2011-03-09	9.46	387.53	2.50	87.27	4.16	14.76	202.53	572.54	18	ST_2	0.00	-3.58	0.00	0.69	-0.01	0.00	-5.02	-2.14	23	–	–	–	–	–
G2C_medium	2011-03-29	7.33	265.06	0.18	21.50	6.96	7.71	220.06	310.05	21	ST_2	-0.02	-25.22	0.00	0.89	-0.03	-0.02	-27.04	-23.40	28	–	–	–	–	–
G2C_medium	2011-04-20	11.59	171.79	0.39	19.77	10.78	12.40	130.55	213.03	22	ST_2	-0.04	-42.66	0.00	1.46	-0.05	-0.03	-45.65	-39.66	29	–	–	–	–	–
G2C_medium	2011-05-10	17.16	73.71	0.67	11.51	15.78	18.54	49.90	97.53	25	T_{air}	-0.08	-45.86	0.01	1.39	-0.11	-0.06	-48.71	-43.00	31	–	–	–	–	–
G2C_medium	2011-06-21	11.94	216.47	0.91	29.21	9.95	13.93	152.83	280.10	14	ST_2	-0.10	-62.80	0.01	2.04	-0.12	-0.07	-67.21	-58.40	15	–	–	–	–	–
G2C_medium	2011-06-28	17.53	95.90	0.82	12.36	15.83	19.22	70.26	121.53	24	T_{air}	-0.13	-48.98	0.01	0.91	-0.16	-0.10	-50.84	-47.12	32	–	–	–	–	–
G2C_medium	2011-07-26	9.43	340.97	0.56	28.40	8.28	10.58	282.35	399.59	26	ST_2	-0.11	-41.73	0.01	0.69	-0.13	-0.09	-43.14	-40.31	35	–	–	–	–	–
G2C_medium	2011-08-17	14.31	126.80	1.45	29.90	11.27	17.35	63.99	189.61	20	ST_2	-0.10	-47.49	0.01	0.57	-0.12	-0.09	-48.66	-46.31	26	–	–	–	–	–
G2C_medium	2011-09-16	13.49	102.46	0.44	10.71	12.57	14.40	79.96	124.96	20	T_{air}	-0.01	-14.22	0.00	1.23	-0.02	-0.01	-16.79	-11.65	22	–	–	–	–	–
G2C_medium	2011-10-16	10.69	71.95	0.13	6.59	10.41	10.97	58.04	85.86	19	T_{air}	-0.05	-38.23	0.01	1.73	-0.07	-0.04	-41.83	-34.62	22	–	–	–	–	–
G2C_medium	2011-12-15	–	–	–	–	–	–	–	–	–	–	-0.08	-10.96	0.03	2.57	-0.14	-0.01	-17.26	-4.67	8	4.15	0.42	3.36	4.93	9
G2C_medium	2012-01-12	3.97	153.98	0.67	64.04	2.33	5.62	-2.72	310.67	8	T_{air}	-0.03	-14.57	0.00	1.33	-0.03	-0.02	-18.80	-10.33	5	–	–	–	–	–

Table S8 Model parameters of the G2C$_{high}$ site.

Site	Date	R_{ref}	E_0	SE R_{ref}	SE E_0	lower R_{ref}	upper R_{ref}	lower E_0	upper E_0	n	model based on	α	GPP_{2000}	SE α	SE GPP_{2000}	lower α	upper α	lower GPP_{2000}	upper GPP_{2000}	n	daily mean R_{ECO} [µmol CO$_2$ m^{-2} s^{-1}]	SE R_{ECO}	lower R_{ECO}	upper R_{ECO}	n
G2C$_{high}$	2010-02-09	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1.14	0.06	1.03	1.26	6
G2C$_{high}$	2010-03-08	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	2.05	0.12	1.84	2.27	9
G2C$_{high}$	2010-03-24	–	–	–	–	–	–	–	–	–	–	-0.03	-10.99	0.01	0.63	-0.05	-0.01	-12.29	-9.69	27	6.73	0.28	6.19	7.27	21
G2C$_{high}$	2010-04-06	7.75	110.35	0.61	38.27	6.46	9.05	28.78	191.92	17	T_{air}	-0.02	-25.30	0.00	1.27	-0.02	-0.01	-27.94	-22.65	22	–	–	–	–	–
G2C$_{high}$	2010-04-28	10.72	133.27	0.45	25.92	9.74	11.69	77.69	188.85	16	ST_2	-0.02	-38.83	0.00	1.90	-0.03	-0.01	-42.86	-34.80	18	–	–	–	–	–
G2C$_{high}$	2010-05-24	8.93	182.53	0.98	28.95	6.75	11.11	118.03	247.04	12	T_{air}	-0.05	-39.66	0.00	0.98	-0.06	-0.04	-41.73	-37.60	20	–	–	–	–	–
G2C$_{high}$	2010-06-09	12.92	165.80	2.65	66.35	7.34	18.50	26.40	305.20	20	ST_2	-0.09	-38.11	0.01	1.15	-0.12	-0.06	-40.49	-35.74	25	–	–	–	–	–
G2C$_{high}$	2010-07-07	–	–	–	–	–	–	–	–	–	–	-0.09	-29.82	0.01	0.55	-0.11	-0.07	-30.99	-28.66	18	16.00	0.56	14.89	17.10	15
G2C$_{high}$	2010-07-21	9.76	232.70	1.90	63.17	5.47	14.06	89.79	375.61	11	ST_2	-0.06	-30.86	0.01	0.93	-0.07	-0.04	-32.79	-28.93	25	–	–	–	–	–
G2C$_{high}$	2010-08-22	10.06	404.98	3.24	144.78	3.15	16.97	96.40	713.57	17	ST_2	-0.09	-51.92	0.01	1.60	-0.12	-0.07	-55.24	-48.59	23	–	–	–	–	–
G2C$_{high}$	2010-09-11	10.09	325.19	1.06	61.72	7.86	12.32	194.98	455.41	19	ST_2	-0.10	-36.65	0.01	1.01	-0.13	-0.07	-38.72	-34.57	29	–	–	–	–	–
G2C$_{high}$	2010-10-29	6.32	125.94	0.22	17.65	5.84	6.80	87.82	164.06	15	T_{air}	-0.03	-38.30	0.00	4.25	-0.03	-0.02	-47.49	-29.11	15	–	–	–	–	–
G2C$_{high}$	2010-12-16	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	0.99	0.20	0.63	1.36	9
G2C$_{high}$	2011-01-05	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	2.89	0.50	1.97	3.81	9
G2C$_{high}$	2011-01-27	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	5.39	0.59	4.31	6.48	9
G2C$_{high}$	2011-02-07	2.67	111.98	0.15	31.07	2.35	2.99	46.11	177.85	18	T_{air}	-0.01	-5.60	0.00	0.85	-0.01	0.00	-7.37	-3.83	23	–	–	–	–	–
G2C$_{high}$	2011-03-08	2.58	67.33	0.23	25.69	2.07	3.09	11.84	122.82	15	T_{air}	-0.01	-5.60	0.00	0.85	-0.01	0.00	-7.37	-3.83	23	–	–	–	–	–
G2C$_{high}$	2011-03-29	9.00	162.98	0.37	24.02	8.22	9.79	111.79	214.16	17	T_{air}	-0.04	-27.65	0.01	1.53	-0.05	-0.02	-30.82	-24.48	24	–	–	–	–	–
G2C$_{high}$	2011-04-20	12.16	273.72	0.54	41.96	11.00	13.33	183.07	364.37	15	ST_2	-0.04	-40.82	0.01	1.66	-0.05	-0.03	-44.28	-37.37	23	–	–	–	–	–
G2C$_{high}$	2011-05-10	12.93	196.63	0.76	25.47	11.33	14.54	142.88	250.38	19	ST_2	-0.08	-48.03	0.01	0.95	-0.09	-0.06	-49.98	-46.08	27	–	–	–	–	–
G2C$_{high}$	2011-06-21	14.10	120.68	1.37	28.61	11.09	17.10	57.70	183.65	13	T_{air}	-0.09	-67.43	0.01	2.65	-0.12	-0.07	-73.07	-61.79	17	–	–	–	–	–
G2C$_{high}$	2011-06-28	16.82	115.08	1.18	17.50	14.36	19.28	78.69	151.47	23	T_{air}	-0.11	-52.91	0.01	0.93	-0.14	-0.09	-54.81	-51.01	31	–	–	–	–	–
G2C$_{high}$	2011-07-26	9.20	460.15	0.98	57.13	7.16	11.25	340.58	579.71	21	ST_2	-0.10	-53.96	0.01	0.86	-0.12	-0.09	-55.72	-52.21	36	–	–	–	–	–
G2C$_{high}$	2011-08-18	17.83	78.70	1.54	21.46	14.56	21.10	33.20	124.19	18	T_{air}	-0.10	-49.63	0.01	1.08	-0.12	-0.07	-51.85	-47.41	27	–	–	–	–	–
G2C$_{high}$	2011-09-16	8.55	345.34	0.55	47.23	7.39	9.70	245.21	445.46	18	ST_5	-0.01	-14.66	0.00	1.41	-0.02	-0.01	-17.59	-11.74	23	–	–	–	–	–
G2C$_{high}$	2011-10-16	7.96	145.79	0.29	38.84	7.33	8.59	61.16	230.42	14	ST_2	-0.03	-33.96	0.00	1.67	-0.04	-0.03	-37.49	-30.43	19	–	–	–	–	–
G2C$_{high}$	2011-12-15	–	–	–	–	–	–	–	–	–	–	-0.06	-20.27	0.01	3.78	-0.07	-0.04	-29.20	-11.35	9	3.40	0.15	3.12	3.68	9
G2C$_{high}$	2012-01-12	7.19	248.93	1.31	76.03	4.30	10.08	81.58	416.28	13	T_{air}	-0.03	-18.45	0.00	1.58	-0.04	-0.02	-21.89	-15.00	14	–	–	–	–	–

7.2 PUBLICATION II.

Eickenscheidt, T., Freibauer, A.; Heinichen, J., Augustin, J., and Drösler, M.: Short-term effects of biogas digestate and cattle slurry application on greenhouse gas emissions affected by N availability from grasslands on drained fen peatlands and associated organic soils, Biogeosciences, 11, 6187–6207, doi:10.5194/bg-11-6187-2014, 2014.

Biogeosciences

Open Access

Short-term effects of biogas digestate and cattle slurry application on greenhouse gas emissions affected by N availability from grasslands on drained fen peatlands and associated organic soils

T. Eickenscheidt[1], **A. Freibauer**[2], **J. Heinichen**[1], **J. Augustin**[3], **and M. Drösler**[1]

[1]University of Applied Sciences Weihenstephan-Triesdorf, Chair of Vegetation Ecology, Weihenstephaner Berg 4, 85354 Freising, Germany
[2]Thünen Institute of Climate-Smart Agriculture, Bundesallee 50, 38116 Braunschweig, Germany
[3]Leibniz Centre for Agricultural Landscape Research e.V., Institute of Landscape Matter Dynamics, Eberswalder Straße 84, 15374 Müncheberg, Germany

Correspondence to: T. Eickenscheidt (tim.eickenscheidt@hswt.de)

Abstract. A change in German energy policy has resulted in a strong increase in the number of biogas plants in Germany. As a consequence, huge amounts of nutrient-rich residues, the by-products of the fermentative process, are used as organic fertilizers. Drained peatlands are increasingly used to satisfy the huge demand for fermentative substrates (e.g., energy crops, grass silage) and the digestate is returned to the peatlands. However, drained organic soils are considered as hot spots for nitrous oxide (N_2O) emissions and organic fertilization is additionally known to increase N_2O emissions from managed grasslands. Our study addressed the questions (a) to what extent biogas digestate and cattle slurry application increase N_2O and methane (CH_4) fluxes as well as the mineral nitrogen use efficiency (NUE_{min}) and grass yield, and (b) how different soil organic matter contents (SOMs) and nitrogen contents promote the production of N_2O. In addition NH_3 volatilization was determined at one application event to obtain first clues with respect to the effects of soil and fertilizer types. The study was conducted at two sites within a grassland parcel, which differed in their soil organic carbon (SOC) and N contents. At each site (named C_{org}-medium and C_{org}-high) three plots were established: one was fertilized five times with biogas digestate, one with cattle slurry, and the third served as control plot. On each plot, fluxes of N_2O and CH_4 were measured on three replicates over 2 years using the closed chamber method. For NH_3 measurements we used the calibrated dynamic chamber method. On an annual basis, the application of biogas digestate significantly enhanced the N_2O fluxes compared to the application of cattle slurry and additionally increased the plant N-uptake and NUE_{min}. Furthermore, N_2O fluxes from the C_{org}-high treatments significantly exceeded N_2O fluxes from the C_{org}-medium treatments. Annual cumulative emissions ranged from 0.91 ± 0.49 to 3.14 ± 0.91 kg N ha^{-1} yr^{-1}. Significantly different CH_4 fluxes between the investigated treatments or the different soil types were not observed. Cumulative annual CH_4 exchange rates varied between -0.21 ± 0.19 and -1.06 ± 0.46 kg C ha^{-1} yr^{-1}. Significantly higher NH_3 losses, NUE_{min} and grass yields from treatments fertilized with biogas digestate compared to those fertilized with cattle slurry were observed. The total NH_3 losses following the splash plate application were 18.17 kg N ha^{-1} for the digestate treatments and 3.48 kg N ha^{-1} for the slurry treatments (36 and 15 % of applied NH_4^+–N). The observed linear increase of 16 days' cumulative N_2O–N exchange or annual N_2O emissions, with mean groundwater level and ammonium application rate, reveals the importance of site-adapted N fertilization and the avoidance of N surpluses in C_{org}-rich grasslands.

1 Introduction

Germany has become the largest biogas producing country in the world since a change in German energy policy and the enactment of the German Renewable Energy Act (Weiland, 2010). At the end of 2012, more than 7500 agricultural biogas plants were operating in Germany (Fachverband Biogas, 2014). Heat and power from biogas substitute for fossil fuels and therefore can reduce greenhouse gas (GHG) emissions

(Weiland, 2010; Don et al., 2012). The strong increase in the number of biogas plants caused a land-use change towards agro-biomass production and additionally raised land-use intensity to satisfy the huge demand for fermentative substrates (Don et al., 2012). In 2011, grass silage accounted for 9 % of the total renewable resources for biogas production (DBFZ, 2012) and was the second most important fermentation substrate after maize silage.

During the fermentative process high amounts of nutrient-rich residues are produced as a by-product. Today, this new form of organic fertilizer is used instead of mineral fertilizers or animal slurries to maintain soil fertility and productivity. Several studies reported a significant increase in nitrous oxide (N_2O) emissions due to the application of nitrogen fertilizers (e.g., Bouwman, 1996; Chadwick et al., 2000; Rodhe et al., 2006; Ruser, 2010). Liquid organic fertilizers such as animal slurry add easily degradable organic carbon (Christensen, 1983) and moisture, both further favoring N_2O losses through denitrification (Clayton et al., 1997). Enhanced N_2O emissions are of major concern due to the fact that N_2O acts as a radiative forcing GHG (IPCC, 2007) and contributes to the chemical destruction of stratospheric ozone (Crutzen, 1979). In Germany, about 78 % of N_2O emissions originate from the agricultural sector (Umweltbundesamt, 2014). In particular, organic soils (e.g., drained peat soils and soils developed in wet conditions) are considered as hotspots of GHG emissions including N_2O, due to the very high mineralization rates of degrading peat (Kasimir-Klemedtsson et al., 1997; Freibauer et al., 2004; Klemedtsson et al., 2005; Goldberg et al., 2010) and to soil moisture conditions that favor anaerobic microsites. According to Maljanen et al. (2010), N_2O emissions from drained organic soils under agricultural use were on average four times as high as those from mineral soils. The few field studies of organic fertilization effects on annual N_2O emissions from drained organic grassland soils revealed very high N_2O emissions of up to $41.0\,kg\,N\,ha^{-1}\,yr^{-1}$ (Velthof et al., 1996).

In Germany, 50 % of the drained peatlands are used as grasslands (Drösler et al., 2011), particularly in the smallholder structure of south Germany. Grassland soils in Europe and Germany produce more N_2O per unit of fertilizer-N than croplands, and emission factors further increase with soil organic carbon (SOC) and nitrogen content (Freibauer and Kaltschmitt, 2003; Dechow and Freibauer, 2011). Moreover, agricultural soils in the southern part of Germany emit about three times as much of the applied N as N_2O as soils in the rest of Germany, which is attributed to the more frequent frost–thaw cycles and enhanced precipitation rates (Jungkunst et al., 2006; Dechow and Freibauer, 2011). Thus, grasslands on organic soils in South Germany represent a widespread high-risk situation for high N_2O emissions after cattle slurry or biogas digestate application, which has not, to our knowledge, been studied before.

Biogas digestate is depleted in easily degradable C compounds and in organic dry matter content compared to fresh slurry due to anaerobic digestion (Möller and Stinner, 2009). The pH value and the ammonium (NH_4^+) content as well the $NH_4^+ : N_{org}$ ratio are higher than in fresh slurry (Wulf et al., 2002b; Möller and Stinner, 2009). Since digested products are more recalcitrant than fresh slurry it could be assumed that microbial degradation is slow, resulting in fewer anoxic microsites and reduced N_2O emissions compared with fresh slurry application (Clemens and Huschka, 2001; Oenema et al., 2005; Möller and Stinner, 2009). However, the few available field and laboratory experiments are contradictory regarding the effect of biogas digestate application on N_2O emissions (e.g., Clemens and Huschka, 2001; Wulf et al., 2002b; Clemens et al., 2006; Senbayram et al., 2009; Sänger et al., 2010), and very few studies exist for grasslands (e.g., Wulf et al., 2002b, Clemens et al., 2006).

The different properties of biogas digestate and cattle slurry (e.g., higher NH_4^+–N concentrations, narrower C : N ratio, higher pH values) directly affect N transformation processes, plant N availability, and thus grass yield. Furthermore it can be assumed that the plant N-uptake and the N_2O emissions are closely interconnected since N-uptake can be considered as a proxy for N availability, affecting N gaseous losses as well. Currently, the effect of anaerobic digestates on crop growth after surface application under field conditions is contradictory, since some authors reported higher crop yields compared to undigested slurries (e.g., Odlare, 2005 cited in Möller and Müller, 2012) whereas others found no effects (e.g., Möller et al., 2008). Only a few studies exist for grassland but it seems that fertilization with biogas digestates positively affects grass yields, but only in single years (Elsässer et al., 1995; Rubæk et al., 1996; Möller et al., 2008; Möller and Müller, 2012).

As well as N_2O, slurry application leads to short-term methane (CH_4) and ammonia (NH_3) emissions. Methane acts as a strong GHG, whereas NH_3 is considered an indirect GHG through ammonia deposition which could promote the formation of N_2O (Moiser, 2001). Moreover, NH_3 deposition causes soil acidification and eutrophication of ecosystems (e.g., Fangmeier et al., 1994; Galloway, 1995, 2001; Smith et al., 1999). In Germany, agriculture is responsible for 95.3 % of anthropogenic NH_3 emissions (Haenel et al., 2010). Particularly high NH_4^+ contents and high pH values, which are typically for the biogas digestate, promote accelerated NH_3 volatilization (Quakernack et al., 2011). High NH_3 emissions particularly occur after splash plate application on grassland (Rubæk et al., 1996; Wulf et al., 2002a), which is still common practice in the smallholder farms of South Germany.

The objective of this study was to quantify short-term N_2O and CH_4 emissions after application of biogas digestate and cattle slurry on grassland on two types of organic soils in South Germany, which differed in their SOC and N contents. Additionally, to what extent biogas digestate and cattle slurry application affect N availability and grass yield should

T. Eickenscheidt et al.: Short-term effects of biogas digestate and cattle slurry application   **6189**

Table 1. Summary of the grassland management for both vegetation periods.

Date	Management event
26 Mar 2010	leveling
7 Apr 2010	rolling
24 May 2010	mowing
14 Jun 2010	manuring [20 $\mathrm{m^3\,ha^{-1}}$]
20 Aug 2010	mowing
25 Aug 2010	manuring [20 $\mathrm{m^3\,ha^{-1}}$]
23 Sep 2010	herbicide against common sorrel (*Rumex acetosa*)
16 Mar 2011	leveling
23 May 2011	mowing
27 May 2011	manuring [25 $\mathrm{m^3\,ha^{-1}}$]
1 Aug 2011	mowing
13 Sep 2011	mowing
22 Sep 2011	manuring [20 $\mathrm{m^3\,ha^{-1}}$]
4 Nov 2011	manuring [25 $\mathrm{m^3\,ha^{-1}}$]

be tested. Furthermore, NH_3 volatilization was determined at one application event to obtain first clues with respect to the effects of soil and fertilizer types. We hypothesize the following: (a) more N_2O is emitted after biogas digestate than after slurry application because of higher NH_4^+–N concentrations in the substrate and, thus, higher NH_4^+–N amounts when using equal volumetric application rates. The more recalcitrant nature of the carbon in the biogas digestate does not matter for GHG formation in high organic carbon soils. (b) N_2O emissions increase with increasing soil C_{org} and N content due to more favorable conditions for denitrification after organic fertilizer application. (c) Biogas digestate leads to a significantly higher grass yield and N-use efficiency compared to cattle slurry due to the higher N availability of the digestate.

2   Materials and methods

2.1   Study site

The study was conducted on a permanent grassland at a drained fen peatland 30 km northeast of Munich (Freisinger Moos, 48°21′ N, 11°41′ E; 450 m a.s.l.). The dominant species were *Poa trivialis*, *Poa pratensis*, *Festuca pratensis*, *Dactylis glomerata*, and *Alopecurus pratensis*. The grassland was mown two and three times in 2010 and 2011 respectively, as is the usual practice in this region. A summary of the complete grassland management over both vegetation periods can be found in Table 1. The grass was used as silage or hay for cattle or as substrate for biogas plants. According to the climate station in Weihenstephan, located 10 km northeast of the site, the 30-year mean annual temperature was 7.5 °C and the mean annual precipitation was 787 mm (1961–1990). Annual atmospheric N deposition amounted to 6.22 and 7.20 kg N $\mathrm{ha^{-1}\,yr^{-1}}$, with NH_4^+–N : NO_3^-–N ra-

tios of 46 : 54 and 49 : 51 in 2010 and 2011. Data of N deposition were collected by the Bavarian State Institute of Forestry at a German Level II monitoring area (Forest Intensive Monitoring Programme of the UNECE), located at 7 km distance from the investigated grassland. In October 2009 we selected two sites within the grassland parcel, which differed in their SOC contents in the topsoil (Table 2). According to the WRB (2006), soil types were classified as mollic Gleysol (named C_{org}-medium) and as sapric Histosol (named C_{org}-high; N. Roßkopf, personal communication, 2013).

2.2   Experimental design

At each site of the grassland parcel, three adjacent plots (plot dimension 12 × 12 m) were selected. At one plot biogas digestate and at another plot cattle slurry were applied; the third plot served as a control (without fertilization). Centrally at each plot, three PVC-collars for GHG measurements (inside dimension 75 × 75 cm; 0.5625 $\mathrm{m^2}$) were permanently inserted 10 cm into the soil with a distance of 1.5 m to each other. To prevent oscillations of the peat through movements during the measurements, boardwalks were installed. At each site a climate station was set up in March 2010 for the continuous recording (every 0.5 h; CR200X Datalogger, Campbell Scientific) of air temperature and humidity at 20 cm above soil surface (CS215-L, Campbell Scientific) and soil temperatures at depths of −2, −5 and −10 cm (109-L, Campbell Scientific). For NH_3 measurements, sensors for wind speed and wind direction (Kleinwindsensor, Thies Clima) in 2 m height were additionally integrated from May to July 2011, with a logging frequency of 5 s (GP1, Delta-T Devices). For measuring the groundwater table, plastic perforated tubes (JK-casings DN 50, 60 mm diameter, 1 m length) were inserted close to each collar to obtain individual groundwater tables for all repetitions during each gas flux measurement. In April 2010, we equipped one tube per plot with a water level logger (Type MiniDiver, Schlumberger water services), which logged the water tables every 15 min. Additionally to the recorded data, plot-specific soil temperatures at three soil depths (−2, −5 and −10 cm) were determined with penetration thermometers at the beginning and end of each gas flux measurement.

In 2010 and 2011, organic fertilizers were applied via splash plate (swiveling slurry spreader for biogas digestate; gooseneck scatterer for cattle slurry) on 14 June 2010, 25 August 2010, 27 May 2011, 22 September 2011, and 4 November 2011 by the landowners (see Table 1). The surface application technique via splash plate is the most common application technique in the smallholder structure of the region. The organic fertilizer was applied on the basis of equal volumetric rates per application event (20 or 25 $\mathrm{m^3\,ha^{-1}}$). This method is typical for farming practices, but produces diverging N application rates per event between slurry and digestate based on NH_4^+ or N_{tot} applications. It is known that the splash plate application technique can result in very uneven

Table 2. Soil properties of the study site.

	Sampling depth	C_{org}-medium	C_{org}-high	n
Soil type (WRB, 2006)[1]		mollic Gleysol	sapric Histosol	
Soil type (German classification KA5)		GMq	KV-KM	
Peat depth [cm][1]		80	70	1
pH value[2]		4.1	4.2	
Total nitrogen [%][2]		1.0	1.5	
Organic carbon [%]	0–10 cm	10.3 ± 0.2	17.0 ± 0.1	9
	10–20 cm	9.3 ± 0.2	16.3 ± 0.2	9
Bulk density [g cm^{-3}]	0–10 cm	0.79 ± 0.02	0.54 ± 0.02	18
	10–20 cm	0.90 ± 0.01	0.64 ± 0.01	18
Porosity [%]	0–10 cm	71 ± 1	78 ± 1	18
	10–20 cm	67 ± 1	72 ± 0	18

Values present means $\pm$ SE. [1] World Reference Base for Soil Resources. [2] Relative to the upper horizon (C_{org}-medium 0–20 cm; C_{org}-high 0–15 cm); N. Roßkopf, personal communication, 2013.

spreading regarding the application rate and/or the evenness. Both chosen spreading devices are known for the higher precision in their application evenness compared to conventional splash plates (approximately 15 and 18–27 % application variability for swiveling slurry spreader and gooseneck scatterer and up to 47 % for conventional splash plate; Frick, 1999). In the present study, the application of an equal volumetric slurry rate was controlled via the barrel content and the tractor speed. At all plots, the tractor lane was 1 m in front of the collars, which were placed in a row with a distance of 1.5 m to each other. Both spreading systems had a spreading width of 12 m and no overlapping zones occurred. Nevertheless we cannot give any estimation of the precision of application evenness that was actually achieved.

The physical and chemical composition of the slurries and digestates varied between the four application events (Table 3). Composition of organic fertilizers was analyzed from 1 L samples which were taken from the slurry tank in the field. Slurries were immediately frozen at $-20\,°C$ until analysis was conducted by the AGROLAB Labor GmbH (Bruckberg, Germany). Due to technical problems at the first application event, cattle slurry was applied by watering cans on the ground of the collars and on a 120 m^2 adjacent area. To ensure an equal volumetric amount of organic fertilizer a 1×1 m grid, fabricated from cords, was previously installed. The same method was used at the fourth application event for the digestate.

2.3 N$_2$O and CH$_4$ flux measurements

As a background, we measured fluxes of N$_2$O and CH$_4$ every second week from January 2010 to January 2012 using the static manual chamber method (volume 309 L) (Livingston

and Hutchinson; 1995). We removed, however, the gas fluxes measured in 2010 from the data set due to errors in the gas chromatography analysis and due to long vial storage. Intensive measurement campaigns were performed after the four fertilization events on 14 June 2010, 25 August 2010, 27 May 2011, and 22 September 2011. Immediately after fertilization flux measurements were carried out daily for a week and on every second day for another 8–9 days. To minimize diurnal variation in the flux pattern, sampling was always carried out between 09.00 and 11.30 a.m. A detailed description of chamber dimensions and configuration is given in Drösler (2005). Four gas samples were taken at regular time intervals after chamber closure (enclosure time 60 min). The samples were collected in 20 mL glass vials, each sealed with a butyl rubber septum. The vials were flushed with chamber air for 30 s using a portable micro pump (KNF Neuberger GmbH, NMP015B), so that the air in the vials was exchanged 32 times. In addition the pump was used to build up an overpressure of approximately 550 mbar to protect the sample against fluctuations in atmospheric pressure during storage. Gas analyses were carried out with a gas chromatograph (Perkin and Elmer, Clarus 400 GC respectively Clarus 480 GC) equipped with a headspace auto sampler (Perkin & Elmer, TurboMatrix 110), a PoraPack 80/100 mesh column, an electron capture detector (ECD) for N$_2$O (ECD temperature 380 °C) and a flame ionization detector (FID) for CH$_4$ analyses (FID temperature 310 °C). Gas samples from the first fertilization event (14–30 June 2010) were immediately analyzed at the Max Planck Institute for Biogeochemistry in Jena, whereas samples from the second fertilization event (25 August–10 September 2010) were analyzed at the Thünen Institute in Braunschweig with a Varian CP-3800 GC-FID/-

T. Eickenscheidt et al.: Short-term effects of biogas digestate and cattle slurry application   **6191**

Table 3. Physical and chemical properties of the applied digestates and slurries.

	1. Application (14 Jun 2010)	2. Application (25 Aug 2010)	3. Application (27 May 2011)	4. Application (22 Sep 2011)	5. Application (4 Nov 2011)
	Cattle slurry				
Fertilizer quantity [$m^3\,ha^{-1}$]	20	20	25	20	20
Total carbon [$kg\,ha^{-1}$]	579	676	798	797	1073
Organic carbon [$kg\,ha^{-1}$]	410	573	655	706	960
Total nitrogen [$kg\,ha^{-1}$]	47	64	70	85	97
NO_3^- [$kg\,N\,ha^{-1}$]	0	0	0	0	0
NH_4^+ [$kg\,N\,ha^{-1}$]	20	28	23	33	38
NH_4^+ : N_{tot} ratio	0.42	0.44	0.33	0.38	0.39
C : N ratio	12	11	11	9	11
pH ($CaCl_2$)	–	–	6.8	7.0	7.0
Dry matter content [%]	5	7	7	9	10
	Biogas digestate				
Fertilizer quantity [$m^3\,ha^{-1}$]	20	20	25	20	20
Total carbon [$kg\,ha^{-1}$]	384	373	167	184	178
Organic carbon [$kg\,ha^{-1}$]	306	337	148	161	178
Total nitrogen [$kg\,ha^{-1}$]	49	52	78	35	61
NO_3^- [$kg\,N\,ha^{-1}$]	0	0	0	0	0
NH_4^+ [$kg\,N\,ha^{-1}$]	22	28	51	17	40
NH_4^+ : N_{tot} ratio	0.45	0.53	0.65	0.49	0.66
C : N ratio	8	7	2	5	3
pH ($CaCl_2$)	–	–	7.7	7.4	7.7
Dry matter content [%]	4	4	2	2	3

ECD using a headspace autosampler (QUMA Elektronik & Analytik GmbH, Germany) and similar conditions. Gas flux rates were calculated from the linear change in gas concentration over time considering chamber air temperature and atmospheric pressure. Gas fluxes were accepted when the linear regression was significant ($P \leq 0.05$). In case of small N_2O or CH_4 fluxes, fluxes were also accepted if the coefficient of determination was ≥ 0.90 and the regression slope was between -1 and $1\,ppb\,min^{-1}$. The cumulative annual mean exchange rate was calculated by linear interpolation between the measurement dates.

2.4 NH_3 flux measurements

Ammonia volatilization was measured at the third organic fertilizer application event on 27 May 2011. Measurements were performed immediately after fertilizer application and thereafter at irregular time intervals of a few hours (in total 96 measurements). For NH_3 measurements we used the calibrated dynamic chamber method ("Dräger-Tube Method"; DTM) which was described in detail by Pacholski et al. (2006). One day before application, eight stainless steel rings (104 cm^2) were inserted into the upper soil (3 cm) at each plot, of which four were grouped close together. Ambient air was sucked with a defined flow rate (1 L min^{-1}) through four connected (via Teflon tubes) conical stainless

steel chambers to an ammonia indicator tube (Drägerwerk AG, Lübeck, Germany). The NH_3 volume concentration was corrected for air temperature and air pressure (Pacholski et al., 2006). To prevent overestimation of NH_3 volatilization through NH_3 enriched ambient air from surrounding area, ammonia concentration from the control treatments was subtracted from the fertilized treatments prior to NH_3 flux calculation. Different studies report a distinct underestimation of up to one order of magnitude of NH_3 fluxes determined by the DTM, mainly due to the low air exchange rate in the chambers (Roelcke et al., 2002; Pacholski et al., 2006). To avoid underestimation of cumulative NH_3–N losses determined by the DTM, Pacholski et al. (2006) developed the following calibration formula to correct the NH_3 fluxes:

$$\ln(NH_3\,flux_{IHF}) = 0.444 \cdot \ln(NH_3\,flux_{DTM}) + 0.590 \cdot \ln(v_{2m}), \qquad (1)$$

where $NH_3\,flux_{IHF}$ is NH_3 flux measured by the integrated horizontal flux method (kg N $ha^{-1}\,h^{-1}$); $NH_3 flux_{DTM}$ is NH_3 flux measured by the DTM (kg N $ha^{-1}\,h^{-1}$); v_{2m} is wind speed at 2 m height (m s^{-1}). Quakernack et al. (2011) compared the DTM method with the frequently used micrometeorological method, concluding that the corrected DTM method also allows quantitative NH_3-loss measurements. The total cumulative NH_3 volatilization was estimated by curve fitting and integration of the area obtained by the fit-

ted curve between time zero and the time point where the NH_3 flux was zero.

2.5   Grass yield, apparent N use efficiency, and N-balances

The annual yield was determined by harvesting the grass inside the PVC-collars with a pair of scissors at each mowing event (same cutting height as the farmer, at about 5 cm; sample area $= 0.5625\,m^2$). There was no visible disturbance from trace gas measurements in the collars. The grassland parcel showed a strong spatial heterogeneity in grass yield so that only sampling inside the collars allowed grass yield and N uptake to be related with N_2O emissions. Mowing events took place on 24 May 2010, 20 August 2010, 23 May 2011, 1 August 2011, and 13 September 2011 (see Table 1). To determine the dry mass (DM), grass samples were oven dried at 60 °C for 48 h. To determine the total carbon (C_{tot}) and total nitrogen (N_{tot}) concentrations of plant biomass, dried grass samples were milled (0.5 mm) and mixed subsamples were analyzed according to DIN ISO 10694 and DIN ISO 13878 by the AGROLAB Labor GmbH (Bruckberg, Germany). The apparent N_{tot} or N_{min} use efficiency (NUE, NUE_{min}) was calculated as follows:

$$NUE\,or\,NUE_{min} = \left(\frac{N\,uptake_{treatment} - N\,uptake_{control}}{total\,N\,applied}\right)\cdot 100\,\%,\quad (2)$$

where $N\,uptake_{treatment}$ is the amount of N taken up by the plants in the fertilized treatments, $N\,uptake_{control}$ is the amount of N taken up by the plants in the unfertilized control, and "total N applied" is the amount of N_{tot} or N_{min} applied, corrected by NH_3–N losses. Emission factors for NH_3 were taken from the German national GHG inventory (see Haenel et al., 2014, Table 4.6 – broadcast, grassland, emission factor (EF) of $0.60\,kg\,kg^{-1}$ related to applied NH_4^+–N), whereas for the third application event our own estimated values based on our measurements were used.

Based on the measured gaseous N fluxes and the N uptake by plants and soil N_{min} contents, a simple N balance was calculated:

$$N\,balance = \left(N\,applied + \left(N\,min_{t_2} - N\,min_{t_1}\right) + N_{dep}\right)\quad (3)$$
$$- \left(N\,uptake + N_2O_{cum} + NH_{3cum}\right),$$

where "N applied" is the amount of N_{tot} applied, $N\,min_{t_1}$ and $N\,min_{t_2}$ are the amounts of N_{min} at time 1 (6 April 2011; early April represents the beginning of the vegetation period in 2011) and time 2 (18 October 2011; end of October represents the end of the vegetation period in 2011) for the soil depth 0–20 cm, N_{dep} is the annual atmospheric N deposition, "N uptake" is the amount of N taken up by the plants (quantified in harvested biomass), N_2O_{cum} is the amount of the annual cumulative N_2O–N losses, and NH_{3cum} is the amount of the annual cumulative NH_3–N losses.

2.6   Soil sampling and laboratory analyses

For the determination of mineral N ($N_{min} = NH_4^+$–N $+ NO_3^-$–N) contents, one mixed soil sample consisting of nine individual samples was collected at two soil depths (0–10, 10–20 cm) at each plot during every gas flux measurement. Samples were immediately cooled and stored in an ice box before analyses. Mineral N was extracted after shaking 40 g of fresh soil with 160 mL $CaCl_2$ (0.0125 M) for 1 hour. The extracts were filtered through a 4–7 µm filter paper (Whatman 595 1/2) and the first 20 mL of the extract were discarded. The solution was frozen at −20 °C until analysis, which was conducted by the AGROLAB Labor GmbH (Bruckberg, Germany). A subsample of 20–30 g was used to determine the gravimetric water content, which was taken into account for the calculation of mineral N concentrations. For determination of C_{tot} and organic carbon (C_{org}) a mixed soil sample of nine individual samples was collected close to each collar at two soil depths (0–10, 10–20 cm) using a 3 cm diameter auger. After drying for 72 h at 40 °C, soil samples were sieved to 2 mm to remove stones and living roots. Analyses were conducted at the Division of Soil Science and Site Science (Humbold Universität zu Berlin, Germany). For the determination of bulk density and porosity, three undisturbed core cutter samples ($100\,cm^3$) were randomly taken at four depths (0–5, 5–10, 10–15, 15–20 cm) for each plot.

2.7   Statistical analysis

Statistical analyses were conducted using R 2.12.1 (R Development Core Team, 2010). We used analysis of variance (ANOVA; for grass yield, 16 days' cumulative N_2O emissions and treatment NO_3^- comparison) or the nonparametric Kruskal–Wallis rank sum test (for GW level) to compare means of samples. In case of significant differences among the means, we used Tukey's honest significant differences (Tukey HSD) or the non-parametric pairwise Wilcoxon rank sum test with Bonferroni correction for multiple comparisons. For testing two independent sample means, we use the Welch two-sample t test (for soil type NO_3^- comparison in 2010) or the nonparametric Mann–Whitney U test (for soil type NO_3^- comparison in 2011). For time series data (N_2O, CH_4 field measurements) we applied linear mixed effects models (Crawley, 2007; Eickenscheidt et al., 2011; Hahn-Schöfl et al., 2011). We set up a basic model with soil type and fertilizer treatment as fixed effects and the spatial replication (individual collar) nested in time as random effect. Non-significant terms were removed from the fixed structure. We extended the basic model by a variance function when heteroscedasticity was observed. In case of significant serial correlation in data, a moving average or a first-order temporal autoregressive function was included in the model. Autocorrelation was tested using the Durbin–Watson test and by plotting the empirical autocorrelation structure. The model extension was proved by the Akaike information criterion

T. Eickenscheidt et al.: Short-term effects of biogas digestate and cattle slurry application   **6193**
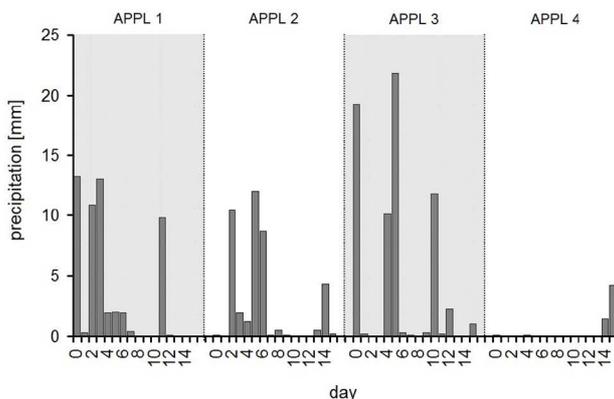
Figure 1. Daily sums of precipitation following the organic fertilizer application events (APPL).

(AIC). For multiple comparisons we conducted Tukey contrasts using the General Linear Hypotheses function from the "multcomp" package (Hothorn et al., 2013).

The assumption of normality of residuals was tested using the Lilliefors or Shapiro–Wilk test and the quantile–quantile plots. Homogeneity of variances of residuals was checked using the Levene or Breusch–Pagan test and by plotting the residuals against the fitted values. Where necessary, data were box-cox transformed prior to analyses. We used simple and multiple linear or non-linear regression models to explain N_2O, CH_4 and NH_3 fluxes. We accepted significant differences if $P \leq 0.05$. Results in the text are given as means ± 1 SD.

3   Results

3.1   Environmental drivers

Temperatures between the two investigated soil types did not differ. In 2010 and 2011, air temperature at 20 cm height ranged from -17.5 to $39.5\,^\circ C$, with an annual mean of $8.6\,^\circ C$ in 2011 at both investigated areas. Soil temperature in -2 cm soil depth averaged $10.3\,^\circ C$ at the C_{org}-medium site and was slightly higher with $10.5\,^\circ C$ at the C_{org}-high site in 2011. Air temperature at 20 cm height 15 or 16 days after fertilization averaged 16.0, 13.1, 15.4, and $11.5\,^\circ C$ for application events 1 to 4 at both investigated soil types. Soil temperature in -2 cm soil depth was approximately $2\,^\circ C$ above the mean air temperature in the same periods at both soil types. In 2010 and 2011 annual precipitation was 850 and 841 mm, which was slightly above the 30-year mean of the period 1961–1990. Figure 1 shows the precipitation following the fertilizer application. With the exception of the third application event, no rainfall occurred during the application of the organic fertilizers. However, precipitation during and after the third application event was weak and amounted to 3 mm in the time span between 16:00 and 00:00 LT.

All treatments showed similar dynamics in their annual hydrographs (Fig. 2a), but mean annual groundwater levels of the C_{org}-high treatments were significantly higher (all $P < 0.001$) compared to the C_{org}-medium treatments in 2010 and 2011 (Table 4). Mean groundwater levels following the fertilizer applications are shown in Table 4.

3.2   N input and N availability

The amount of N applied was 111 and $252\,kg\,N\,ha^{-1}$ for slurry treatments and 101 and $174\,kg\,N\,ha^{-1}$ for digestate treatments in 2010 and 2011, respectively. However, due to the distinctly higher NH_4^+–$N : N_{tot}$ ratio of the biogas digestate, total NH_4^+–N input was comparable to (2010) or slightly higher than (15 % more NH_4^+–N; 2011) that for the slurry treatments (Table 3). Additional physical and chemical properties of the slurry and digestate are shown in Table 3.

The extractable N_{min} contents of the soils were dominated by NO_3^- whereas NH_4^+ was only of minor importance, especially at the C_{org}-medium treatments (Fig. 2b and c). The NO_3^- content was significantly higher ($P < 0.001$) at the C_{org}-high treatments than at the C_{org}-medium treatments in 0–10 cm soil depth in both years and in 10–20 cm soil depth in 2010 ($P < 0.01$) (Table 4). With the exception of the first application event, all fertilization events increased the NO_3^- contents of the soil for a short period (Fig. 2c, Table 4). However, only in 2011 did the fertilized sites show significantly ($P < 0.01$) higher NO_3^- contents compared to the control treatments, but differences between digestate and slurry were generally not significant (except for 0–10 cm soil depth at the C_{org}-medium site; Table 4).

3.3 N_2O emissions

N_2O fluxes were generally low for all treatments (Fig. 2d). Background emissions rarely exceeded $50\,\mu g\,N\,m^{-2}\,h^{-1}$. Highest N_2O fluxes were found immediately after fertilizer application (Figs. 2d and 3), sometimes followed by a second phase of higher emissions after 6 to 12 days. In the case of the C_{org}-medium treatments N_2O fluxes returned to background emission level within 3 to 7 days, whereas the C_{org}-high treatments had longer lasting increased N_2O emissions, particularly for the digestate treatment.

Short-term (16 days) N_2O fluxes of fertilized treatments significantly ($P < 0.01$) exceeded N_2O fluxes of control treatments at all fertilization events. However, only in one out of four fertilization events were short-term N_2O fluxes significantly ($P < 0.001$) higher for the digestate treatments compared to the slurry treatments. Additionally, significantly ($P < 0.001$) higher short-term N_2O fluxes were observed at the C_{org}-high treatments compared to the C_{org}-medium treatments in 2011, but the opposite was observed at the second fertilization event in 2010.

However, due to the high variability and the partially fast return to the background emission level, short-term (16 days)
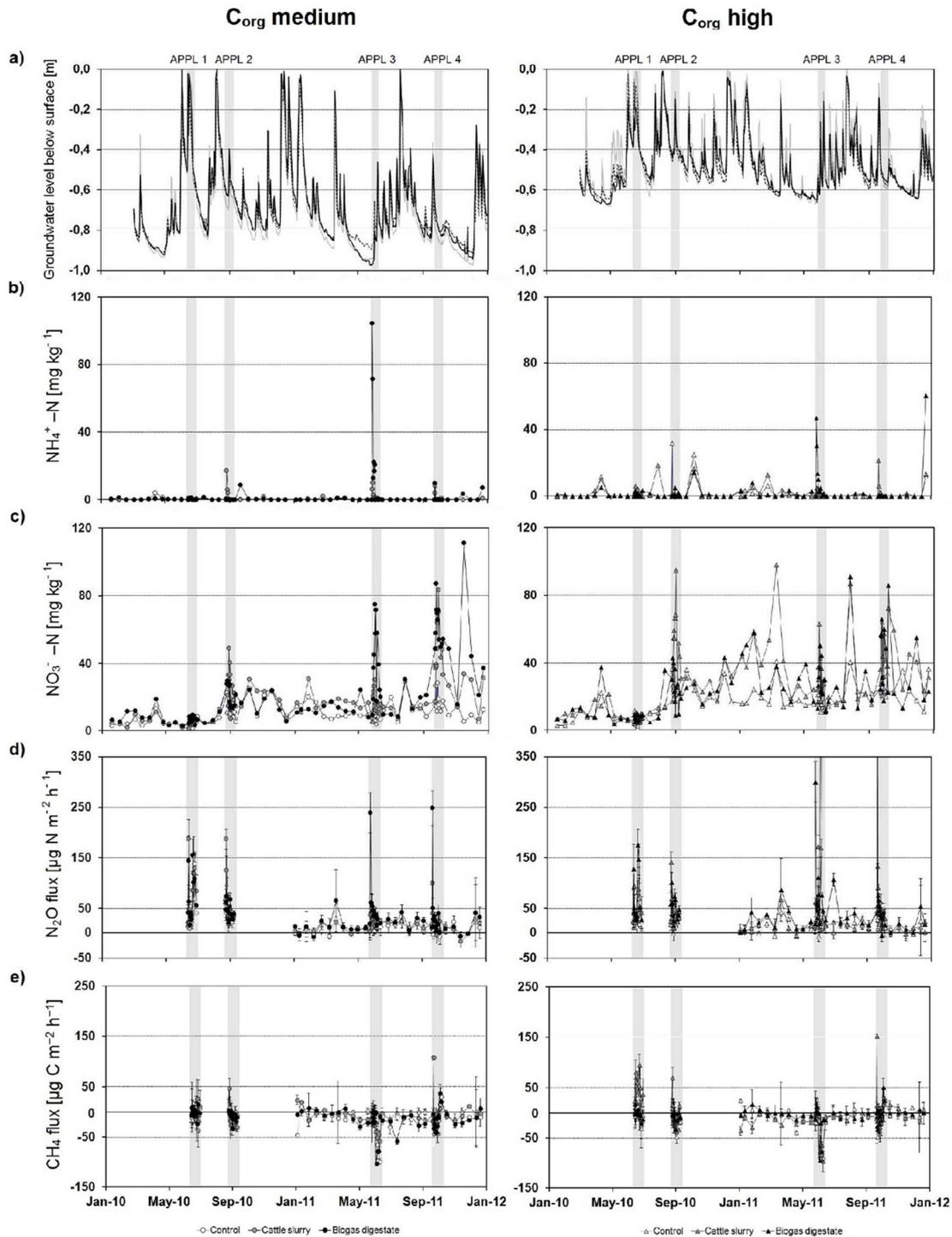
Figure 2. Variation in groundwater level (**a**), extractable NH_4^+ (**b**) and NO_3^- (**c**) contents for the soil depth 0–10 cm, N_2O (**d**) and CH_4 fluxes (**e**) (mean $\pm$ SD, $n = 3$) of the C_{org}-medium and C_{org}-high sites from January 2010 to January 2012. Gray bars mark the 15 or 16 days period of intensive gas flux measurements following organic fertilizer application.

T. Eickenscheidt et al.: Short-term effects of biogas digestate and cattle slurry application   **6195**

Table 4. Mean (minimum/maximum) groundwater level (GW), NO_3^- and NH_4^+ content in the soil following organic fertilizer application and for the investigated years 2010 and 2011.

	Sampling depth [cm]	C_{org}-medium			C_{org}-high			n
		Control	Cattle slurry	Biogas digestate	Control	Cattle slurry	Biogas digestate	
				1 Application (14 Jun–30 Jun 2010)				
GW level [cm]		−32 (−62/−2)	−39 (−60/−5)	−31 (−58/−2)	−21 (−46/−1)	−26 (−45/−7)	−33 (−45/−19)	
NO_3^- [mg N kg^{-1}]	0–10	5 (1/9)	5 (1/7)	7 (3/10)	6 (1/12)	8 (3/11)	8 (5/10)	12
	10–20	9 (6/12)	9 (5/13)	11 (8/15)	11 (8/15)	12 (7/15)	14 (6/19)	12
NH_4^+ [mg N kg^{-1}]	0–10	0 (0/1)	0 (0/1)	0 (0/1)	1 (0/2)	1 (0/6)	1 (0/3)	12
	10–20	0 (0/1)	0 (0/1)	0 (0/1)	0 (0/2)	1 (0/11)	1 (0/2)	12
				2 Application (25 Aug–10 Sep 2010)				
GW level [cm]		−64 (−70/−49)	−58 (−63/−42)	−57 (−63/−40)	−36 (−40/−37)	−40 (−46/−22)	−37 (−43/−15)	
NO_3^- [mg N kg^{-1}]	0–10	13 (6/23)	27 (7/49)	21 (14/30)	25 (17/37)	50 (17/95)	25 (9/43)	12
	10–20	22 (17/28)	28 (19/37)	27 (17/38)	31 (26/35)	34 (11/45)	31 (12/48)	12
NH_4^+ [mg N kg^{-1}]	0–10	0 (0/0)	2 (0/17)	0 (0/0)	3 (0/32)	0 (0/1)	1 (0/5)	12
	10–20	1 (0/10)	0 (0/0)	0 (0/1)	0 (0/1)	0 (0/1)	1 (0/3)	12
				3 Application (27 May–11 Jun 2011)				
GW level [cm]		−82 (−94/−57)	−76 (−89/−52)	−80 (−97/−46)	−41 (−60/−11)	−47 (−62/−16)	−49 (−62/−16)	
NO_3^- [mg N kg^{-1}]	0–10	9 (4/17)	17 (5/30)	40 (10/75)	17 (11/26)	29 (12/63)	29 (11/50)	12
	10–20	17 (11/22)	31 (18/44)	45 (18/75)	24 (18/40)	28 (18/40)	45 (21/148)	12
NH_4^+ [mg N kg^{-1}]	0–10	0 (0/2)	2 (0/10)	21 (0/104)	0 (0/1)	1 (0/5)	10 (0/47)	12
	10–20	0 (0/1)	1 (0/2)	6 (0/26)	1 (0/2)	1 (0/3)	4 (0/12)	12
				4 Application (22 Sep–7 Oct 2011)				
GW level [cm]		−83 (−87/−72)	−77 (−81/−70)	−76 (−83/−58)	−54 (−60/−33)	−55 (−58/−46)	−53 (−57/−41)	
NO_3^- [mg N kg^{-1}]	0–10	18 (12/28)	48 (18/83)	62 (49/87)	23 (20/34)	43 (28/73)	45 (18/86)	12
	10–20	32 (18/46)	50 (21/79)	53 (35/66)	24 (20/30)	30 (22/39)	38 (23/86)	12
NH_4^+ [mg N kg^{-1}]	0–10	0 (0/0)	1 (0/8)	1 (0/10)	0 (0/0)	3 (0/21)	0 (0/0)	12
	10–20	0 (0/0)	0 (0/0)	0 (0/1)	0 (0/1)	1 (0/3)	0 (0/0)	12
				2010				
GW level [cm]*		−67 (−94/−2)	−65 (−91/−2)	−63 (−92/0)	−41 (−68/2)	−45 (−64/−1)	−45 (−67/−1)	
NO_3^- [mg N kg^{-1}]	0–10	9 (1/26)	14 (1/49)	12 (3/30)	15 (1/37)	24 (3/95)	17 (4/43)	45
	10–20	14 (5/34)	17 (5/38)	17 (4/38)	19 (7/47)	23 (6/64)	21 (6/49)	45
NH_4^+ [mg N kg^{-1}]	0–10	0 (0/4)	1 (0/17)	0 (0/9)	2 (0/32)	2 (0/19)	1 (0/14)	45
	10–20	0 (0/10)	0 (0/1)	0 (0/1)	0 (0/8)	1 (0/11)	1 (0/5)	45
				2011				
GW level [cm]		−76 (−98/−3)	−72 (−92/0)	−72 (−97/0)	−47 (−67/1)	−52 (−66/−3)	−50 (−65/−3)	
NO_3^- [mg N kg^{-1}]	0–10	13 (4/31)	25 (5/83)	36 (8/111)	21 (11/41)	36 (12/98)	34 (11/91)	47
	10–20	24 (9/46)	34 (15/79)	40 (18/120)	27 (14/52)	33 (18/78)	37 (10/148)	47
NH_4^+ [mg N kg^{-1}]	0–10	0 (0/2)	1 (0/10)	6 (0/104)	1 (0/12)	2 (0/21)	4 (0/60)	47
	10–20	0 (0/3)	0 (0/4)	2 (0/26)	1 (0/7)	1 (0/27)	2 (0/12)	47

cumulative N_2O emissions were not significantly different from the control treatments in 2010 (Fig. 4), but for 2011 short-term cumulative N_2O emissions had a clear trend in the order digestate > slurry > control (although not significant in one case).

On an annual basis organic fertilization led to significantly ($P < 0.001$) higher N_2O fluxes compared to unfertilized treatments. Additionally, the application of biogas digestate significantly ($P < 0.01$) enhanced the N_2O fluxes compared to the application of cattle slurry. Furthermore, N_2O fluxes from the C_{org}-high treatments significantly ($P < 0.001$) exceeded N_2O fluxes from the C_{org}-medium treat-

ments. Annual cumulative emissions ranged from 0.91 ± 0.49 kg N ha^{-1} yr^{-1} (control treatment, C_{org}-medium site) to 3.14 ± 0.91 kg N ha^{-1} yr^{-1} (digestate treatment, C_{org}-high site; Table 5). Calculated EFs based on the amount of N_{tot} ranged from 0.12 to 0.23 for the slurry treatments and from 0.55 to 1.13 for the digestate treatments (Table 5).

Observed collar-specific cumulative annual N_2O fluxes were strongly related to collar-specific annual plant N-uptake (Fig. 5a and b). Of the temporal and spatial variation in the 16 days' cumulative N_2O–N exchange rates, 53 % was explained by the amounts of applied NH_4^+–N and the mean groundwater levels below surface during the same time
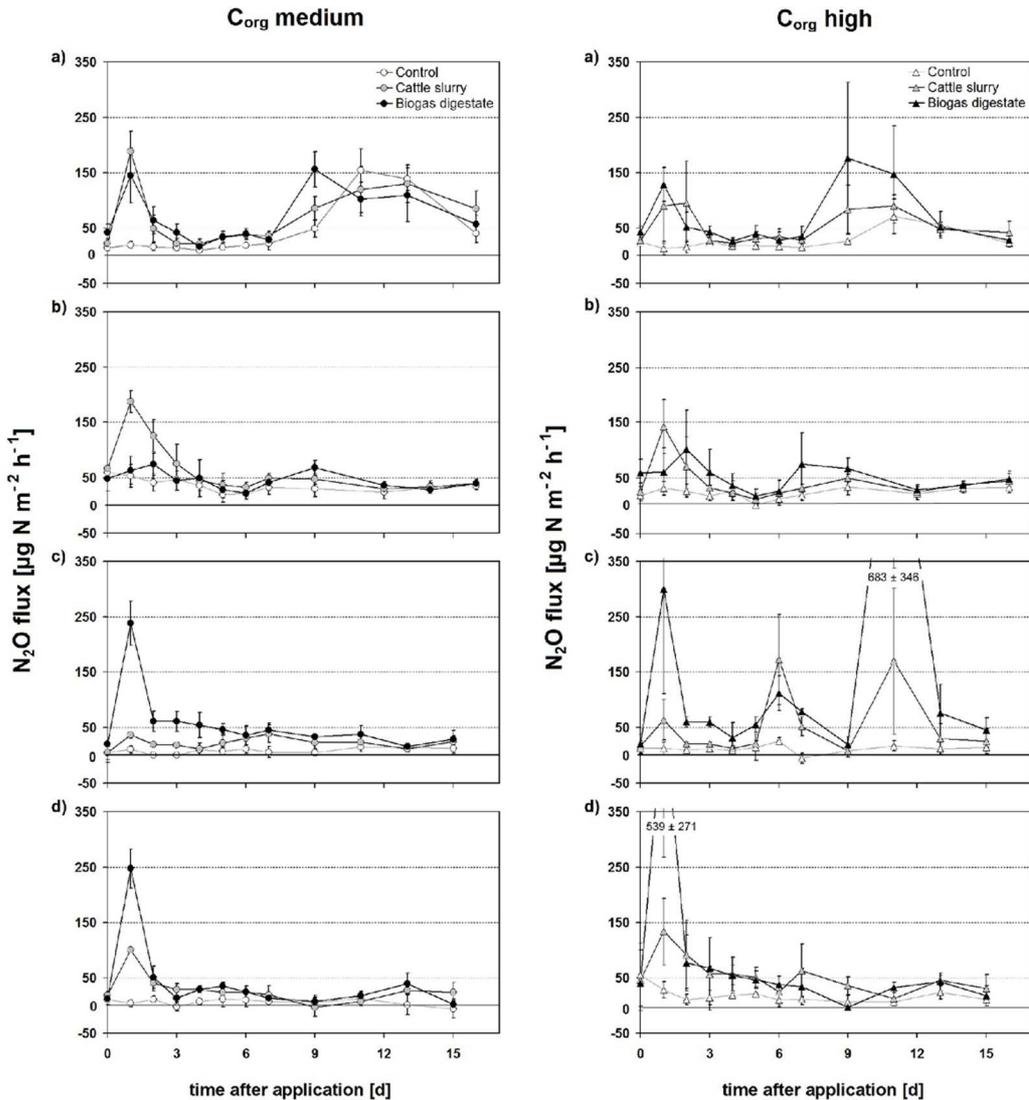
Figure 3. Mean ($\pm$ SD, $n = 3$) N$_2$O fluxes following organic fertilizer application events **(a)** 14–30 June 2010; **(b)** 25 August–10 September 2010; **(c)** 27 May–11 June 2011 and **(d)** 22 September–7 October 2011.

(Fig. 6). A similar trend was observed for the annual cumulative N$_2$O emissions, but regression analysis was not possible due to the small sample size ($n = 6$).

3.4 CH$_4$ emissions

Most of the time, CH$_4$ emissions could not be detected (Fig. 2e). Occasionally CH$_4$ peaks were only found immediately after cattle slurry application. However, with the exception of the slurry treatment of the C$_{org}$-high site at the first application event, the organic fertilization did not result in significantly different short-term (15 or 16 days) CH$_4$ fluxes between the treatments or the investigated soil types. The observed weak CH$_4$ emissions or uptakes amounted to cumulative annual CH$_4$ exchange rates of -0.21 ± 0.19

to -1.06 ± 0.46 kg C ha^{-1} yr^{-1}. Significantly different CH$_4$ fluxes between the investigated treatments or the different soil types could not be observed regarding the annual fluxes in 2011.

3.5 NH$_3$ volatilization

Highest NH$_3$ losses were observed immediately after fertilization (Fig. 7). During the first 24 h, 64 and 100 % of total NH$_3$ losses occurred at the digestate and slurry treatments, respectively. Since differences in the response of NH$_3$ volatilization were not significant, treatment data were pooled by soil type prior to regression analysis. The total NH$_3$ loss following application was 18.17 kg N ha^{-1} for the digestate treatments and 3.48 kg N ha^{-1} for the slurry

T. Eickenscheidt et al.: Short-term effects of biogas digestate and cattle slurry application 6197
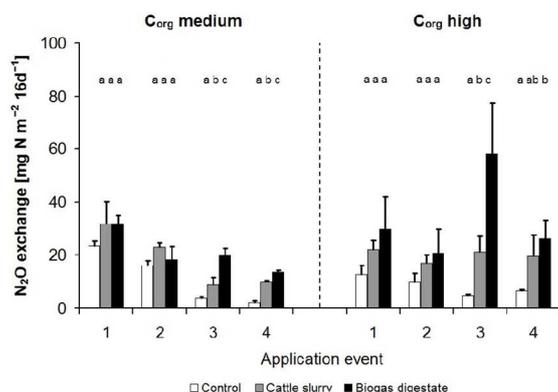
Figure 4. Cumulative N_2O exchange during 16 days following organic fertilizer application. Bars indicate mean values + SD ($n = 3$). Means with the same letter indicated no significant differences between treatments at a single application event for sites C_{org}-medium and C_{org}-high respectively (ANOVA, Tukey HSD test at $P \leq 0.05$).
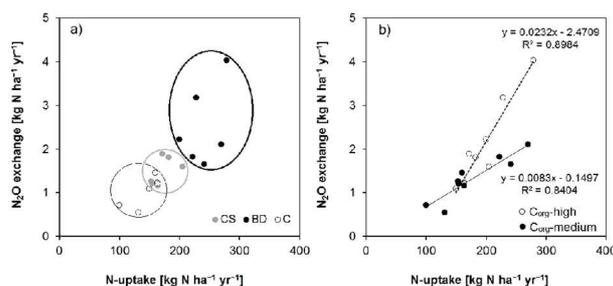
Figure 5. Relationship between cumulative annual N_2O emissions and annual plant N uptake for the treatments (**a**) and for the investigated soil types (**b**). Dots represent mean annual values of each PVC collar; CS, cattle slurry; BD, biogas digestate; C, control.
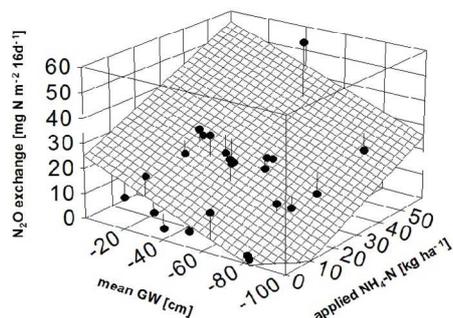
Figure 6. Relationship of plot-wise mean 16 days' cumulative N_2O–N emissions of the four application events (y) to mean groundwater level ($x1$) and the amount of applied NH_4–N ($x2$). The regression equation is $y = 24.98$ (± 4.98) $+ x1 \cdot 0.30 (\pm 0.09) + x2 \cdot 0.51 (\pm 0.11)$; R^2 adj. $= 0.53$, $P < 0.001$, d$f = 21$. Solid lines indicate the deviation of measured data from the model surface.

Table 5. Calculated EFs for the year 2011 and for single application events (16 days; Appl. 1–Appl. 4). The EF is based on the amount of total nitrogen (N_{tot}) without consideration of NH_3–N losses.

	Control	Cattle slurry	Biogas digestate
		C_{org}-medium	
N_2O exchange [kg N ha^{-1} yr^{-1}]	0.91 ± 0.49	1.21 ± 0.05	1.86 ± 0.23
Annual EF		0.12	0.55
EF Appl. 1		0.18	0.17
EF Appl. 2		0.11	0.05
EF Appl. 3		0.08	0.21
EF Appl. 4		0.09	0.33
		C_{org}-high	
N_2O exchange [kg N ha^{-1} yr^{-1}]	1.18 ± 0.07	1.77 ± 0.15	3.14 ± 0.91
Annual EF		0.23	1.13
EF Appl. 1		0.20	0.35
EF Appl. 2		0.11	0.21
EF Appl. 3		0.23	0.68
EF Appl. 4		0.15	0.56

treatments. The relative N loss was 36 and 15 % of applied NH_4^+–N, or 23 and 5 % of total applied N for the digestate and slurry treatments, respectively.

3.6 Grass yield, apparent N use efficiency and estimated N balances

In 2010 and 2011, the mean annual grass yield ranged from 4.5 (control C_{org}-medium) to 13.1 t DM ha^{-1} yr^{-1} (digestate C_{org}-high; Table 6). In both years the mean annual grass yield from the digestate treatments was significantly ($P < 0.05$) higher compared to the slurry treatments. This pattern was also found in the annual plant N-uptake, which showed a clear partitioning between the treatments investigated (Fig. 5a). Additionally, the mean annual grass yield from the C_{org}-high treatments exceeded yields from the C_{org}-medium treatments of both years, but differences were not significant.

The application of biogas digestate markedly increased apparent NUE and NUE$_{min}$ compared to cattle slurry treatments (Table 6). NUE values were on average 115 ± 136 % for biogas digestate treatments and 24 ± 23 % for cattle slurry. NUE$_{min}$ values were always > 100 % for biogas digestate treatments, whereas for cattle slurry NUE$_{min}$ values averaged 74 ± 68 % in 2011 but were > 100 % in 2010. Besides fertilizer type effects, higher NUE and NUE$_{min}$ were observed at the C_{org}-medium treatments compared to the C_{org}-high treatments.

The estimated N balances revealed N surpluses of up to 70 kg N ha^{-1} yr^{-1} for cattle slurry treatments but deficits of

6198   T. Eickenscheidt et al.: Short-term effects of biogas digestate and cattle slurry application
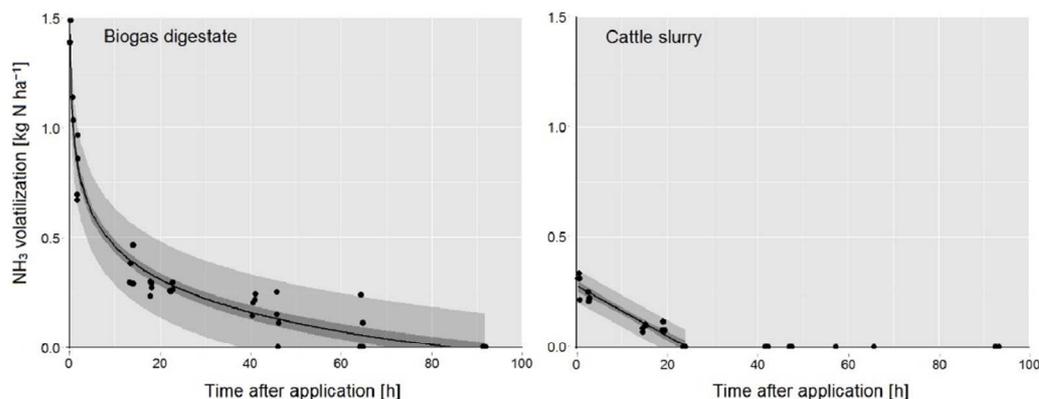
Figure 7. Ammonia (NH_3) volatilization following organic fertilizer application at event 3 (27 May 2011). Dots present single NH_3 measurements for a time period of 94 h. Black lines show the estimated NH_3 volatilization with 95 % confidence band (dark gray) and 95 % prediction band (light gray). The model function for biogas digestate is $y = -0.2619\,(\pm 0.025) \cdot \ln(x) + 0.9605\,(\pm 0.008)$; $r^2 = 0.96$; $P < 0.0001$; the model function for cattle slurry is $y = 0.2818\,(\pm 0.012) - 0.0114(x)\,(\pm 0.001)$; $r^2 = 0.92$; $P < 0.0001$.

Table 6. Grass yields, N uptake and N use efficiency for the years 2010 and 2011.

Treatment	Cutting date	Fertilization date	N content plant [%]	DM [t ha^{-1} yr^{-1}]	N uptake [kg N ha^{-1}]	N applied [kg N ha^{-1}][b]	N_{min} applied [kg N ha^{-1}][b]	N use efficiency [%]	N_{min} use efficiency [%]
Control C_{org}-medium	24 May 2010	–	2.04[a]	2.52	51	–	–	–	–
Control C_{org}-high	24 May 2010	–	2.14	2.93	63	–	–	–	–
Cattle slurry C_{org}-medium	24 May 2010	N.A.	2.37	3.19	76	–	–	–	–
Cattle slurry C_{org}-high	24 May 2010	N.A.	2.14	3.58	77	–	–	–	–
Biogas digestate C_{org}-medium	24 May 2010	N.A.	2.04	4.17	85	–	–	–	–
Biogas digestate C_{org}-high	24 May 2010	N.A.	2.27	4.39	100	–	–	–	–
Control C_{org}-medium	20 Aug 2010	–	2.03	2.02	41	–	–	–	–
Control C_{org}-high	20 Aug 2010	–	2.00	2.63	53	–	–	–	–
Cattle slurry C_{org}-medium	20 Aug 2010	14 Jun 2010	2.19	3.06	67	45	17	58	153
Cattle slurry C_{org}-high	20 Aug 2010	14 Jun 2010	1.93	3.23	62	45	17	22	57
Biogas digestate C_{org}-medium	20 Aug 2010	14 Jun 2010	2.03	2.99	61	38	14	52	140
Biogas digestate C_{org}-high	20 Aug 2010	14 Jun 2010	2.00	3.51	70	38	14	47	125
Control C_{org}-medium	23 May 2011	–	1.96	2.66	52	–	–	–	–
Control C_{org}-high	23 May 2011	–	1.70	3.82	65	–	–	–	–
Cattle slurry C_{org}-medium	23 May 2011	25 Aug 2010	2.01	2.58	52	61	24	0	0
Cattle slurry C_{org}-high	23 May 2011	25 Aug 2010	1.70	4.20	71	61	24	11	27
Biogas digestate C_{org}-medium	23 May 2011	25 Aug 2010	1.96	3.97	78	40	18	64	144
Biogas digestate C_{org}-high	23 May 2011	25 Aug 2010	1.83	4.54	83	40	18	45	101
Control C_{org}-medium	1 Aug 2011	–	1.71	2.06	35	–	–	–	–
Control C_{org}-high	1 Aug 2011	–	1.48	2.88	43	–	–	–	–
Cattle slurry C_{org}-medium	1 Aug 2011	27 May 2011	1.71	2.73	47	67	20	17	58
Cattle slurry C_{org}-high	1 Aug 2011	27 May 2011	1.51	3.19	48	67	20	8	28
Biogas digestate C_{org}-medium	1 Aug 2011	27 May 2011	1.78	4.88	87	60	33	86	158
Biogas digestate C_{org}-high	1 Aug 2011	27 May 2011	1.48	5.34	79	60	33	61	112
Control C_{org}-medium	13 Sep 2011	–	2.53	1.71	43	–	–	–	–
Control C_{org}-high	13 Sep 2011	–	2.26	2.27	51	–	–	–	–
Cattle slurry C_{org}-medium	13 Sep 2011	27 May 2011	2.57	2.28	59	(55)[c]	(8)[c]	28	189
Cattle slurry C_{org}-high	13 Sep 2011	27 May 2011	2.53	2.64	67	(61)[c]	(14)[c]	25	110
Biogas digestate C_{org}-medium	13 Sep 2011	27 May 2011	2.53	3.15	80	(8)[c]	(0)[c]	436	–
Biogas digestate C_{org}-high	13 Sep 2011	27 May 2011	2.26	3.25	74	(24)[c]	(0)[c]	94	–

[a] N contents from control treatments were estimated from fertilized treatments. [b] Applied N_{tot} and N_{min} were corrected by NH_3–N losses (23 and 5 % from N_{tot}, or 36 and 15 % from N_{min} for biogas digestate and cattle slurry, respectively). [c] Hypothetically remaining N_{tot} and N_{min} from the application event 3 (27 May 2011). N.A. = not available.

T. Eickenscheidt et al.: Short-term effects of biogas digestate and cattle slurry application   **6199**

Table 7. Estimated nitrogen balance for the year 2011.

Treatment	N applied [kg N ha^{-1} yr^{-1}]	$N_{min}t_1$[a] [kg N ha^{-1}]	$N_{min}t_2$[a] [kg N ha^{-1}]	N deposition [kg N ha^{-1} yr^{-1}]	N uptake [kg N ha^{-1} yr^{-1}]	N_2O [kg N ha^{-1} yr^{-1}]	NH_3[b] [kg N ha^{-1} yr^{-1}]	N balance [kg N ha^{-1} yr^{-1}]
Control C_{org}-medium	0	27.5	29.4	7.2	130	0.9	0.0	−122.4
Control C_{org}-high	0	22.8	27.7	7.2	159	1.2	0.0	−148.0
Cattle slurry C_{org}-medium	252	35.7	51.2	7.2	157	1.2	46.1	70.4
Cattle slurry C_{org}-high	252	27.3	68.1	7.2	186	1.8	46.1	66.1
Biogas digestate C_{org}-medium	174	29.8	83.3	7.2	244	1.9	52.4	−64.1
Biogas digestate C_{org}-high	174	26.2	28.4	7.2	236	3.1	52.4	−108.1

[a] Reference date for t_1 is 6 April 2011 and for t_2 18 October 2011. [b] NH_3–N losses at the fourth and fifth application event were estimated based on EF taken from the German national greenhouse gas inventory (EF = 0.6 related to applied NH_4^+; Haenel et al., 2014).

up to 108 kg N ha^{-1} yr^{-1} for biogas digestate treatments, for the year 2011 (Table 7).

4 Discussion

4.1 Drainage and fertilizer effects on N-availability and N-transformation

Mineral nitrogen contents were consistently higher at the C_{org}-high treatments than at the C_{org}-medium treatments, in line with the considerably higher amount of SOC and N at this site. It is well known that drainage enhances the degradation of soil organic matter (SOM) and thus stimulates net nitrogen mineralization and N transformation processes (Kasimir Klemedtsson et al., 1997; Freibauer et al., 2004; Klemedtsson et al., 2005; Goldberg et al., 2010). Various studies reported an annual N supply through peat mineralization of 70 to 292 kg N ha^{-1} yr^{-1} (Schothorst, 1977; Flessa et al., 1998; Sonneveld and Lantinga, 2011). It can be assumed that at a comparable aeration status and temperature, mineralization processes are more intensive at peatlands that were recently drained (Hacin et al., 2001; Renger et al., 2002; Sonneveld and Lantinga, 2011) or contain higher amounts of SOM.

As expected from the literature, the biogas digestates differed in their physical and chemical properties from the cattle slurries. In the present study, the biogas digestates had narrower C : N ratios (see also, e.g., Tambone et al., 2009), higher pH values (see also Wulf et al., 2002b and Quakernack et al., 2011), wider NH_4^+ : N_{tot} ratios and thus relatively higher NH_4^+ contents than the cattle slurries (see also Möller and Stinner 2009). The amounts of NH_4^+–N were not distinctly different between the applied organic fertilizers but in 2011 biogas treatments received 15 % more NH_4^+–N compared to cattle slurry treatments.

We observed an unexpected small change in the NH_4^+ content of the soil immediately after fertilizer application, which can be attributed to different reasons. Firstly, the fertilizers partly remained on the plant canopy after splash plate application and therefore soil contact and infiltration were limited (Quakernack et al., 2011). Secondly, a significant fraction of NH_4^+ from the organic fertilizer was lost in a few hours after splash plate application via NH_3 volatilization. Thirdly,

and probably most importantly, in well aerated soils applied NH_4^+ underwent rapid nitrification, as was indicated by the increasing soil NO_3^- contents in the upper soil layer after fertilizer application. In general, the continuously observed absent or low NH_4^+ contents with simultaneously high extractable NO_3^- in the soil indicate that net nitrification entirely controlled net nitrogen mineralization at all treatments of the investigated study sites. Nitrification requires sufficient oxygen (O_2) availability in the soil (Davidson et al., 1986), hence we can assume well aerated soil conditions, at least in the upper soil layer, for most of the time at the study sites.

Several studies (e.g., Gutser et al., 2005; Jones et al., 2007) reported that the infiltration of organic fertilizer may enhance the soil N pool and further stimulates SOM mineralization, leading to additional N_{min}. This becomes evident in the observed significantly higher NO_3^- contents of the fertilized treatments compared to the unfertilized control treatments, especially in the 0–10 cm soil layer. However, significant differences in the N_{min} contents between the two investigated organic fertilizers were not found in 2010 and 2011. This may be due to the fact that the N uptake by plants at the digestate treatments was on average 27 % higher and that marked differences in the amount of N_{tot} and NH_4^+ of the applied organic fertilizers were only observed in the second study year.

To maintain soil fertility and yield and to reduce harmful side effects (e.g., N_2O losses, NO_3^- leaching), site-adapted fertilization is necessary. The estimated negative N balances for biogas treatments are in line with Andres et al. (2013), who reported that positive N balances could only be achieved when the amount of applied digestate contains more than 200 kg N ha^{-1} yr^{-1}. However, the strong negative N balances of the control treatments reveal that large amounts of up to 148 kg N ha^{-1} yr^{-1} originate from peat mineralization, demonstrating the unsustainable agricultural use of drained peatlands. Assuming that the fertilized treatments received equal amounts of N from peat mineralization, all N balances of these treatments were strongly positive. N surpluses as estimated for the cattle slurry treatments enhance the soil N pool, but the gradual release of N at a non-predictable stage from the soil N pool carries the risk of leaching or gaseous losses (Amon et al., 2006). Particularly in wintertime, high amounts of available NO_3^- in the soil, as observed especially at the fertilized treatments of the C_{org}-high sites, carry the

risk of N leaching due to the reduced N demand by plant uptake and by the microbial community during this period (Merino et al., 2002; Sänger et al., 2010).

4.2 Fertilizer effect on N-use efficiency and grass yield

In line with investigations from Schils et al. (2008) most of the applied and produced N_{min} was probably rapidly absorbed by the grassland as the soil N_{min} content usually decreased within a few days after fertilizer application (Fig. 2b and c). This becomes evident also in the apparent NUE_{min}, especially from biogas digestate treatments. A significant effect of biogas digestate on crop yields and apparent NUE_{min} as observed in the present study was also reported from pot experiments (e.g., de Boer, 2008; Möller and Müller, 2012), but not for field applications without incorporation of the digestate into the soil (Möller and Müller, 2012). According to de Boer (2008), the higher NUE_{min} at digestate treatments can be attributed to the wider NH_4^+/N_{tot} ratio as well as to the narrower C/N ratio of the applied digestate. Thus more N was immediately available for plant growth after fertilization (Amon et al., 2006; Sänger et al., 2010), whereas the lower C/N ratio reduced the potential for microbial immobilization of applied N (Velthof et al., 2003, de Boer, 2008). We hypothesized that the application of biogas digestate leads to a significantly higher grass yield and N-use efficiency compared to the application of cattle slurry due to the higher N availability of the digestate. This could partly be confirmed, but the much higher grass yields from biogas digestate treatments cannot solely be explained by differences in applied NH_4^+, since differences were small. However, the much higher N-uptake at the digestate treatments (Fig. 5a) indicated that much more N must have been available at these treatments. Many studies have shown that the utilization of N derived from organic fertilizer is relatively small in the year of application, due to the slow release of organically bound N (Jensen et al., 2000; Sørensen and Amato, 2002; Gutser et al., 2005). The consistently higher NUE_{min} of > 100 % at the digestate treatments indicates that some organic N derived from the fertilizer or from the SOM pool has been mineralized (Gunnarsson et al., 2010). Since the digestate is considered as more recalcitrant (Clemens and Huschka, 2001; Oenema et al., 2005; Möller and Stinner, 2009), it can be assumed that the digestate enhanced SOM mineralization more than cattle slurry, or that N mineralized from SOM or fertilizer had a larger share in the uptake by the plants due to lower competition of microbial immobilization as was reported by Gutser et al. (2010). Probably the assumed higher SOM mineralization at the digestate treatments could partly be related to a priming effect since the higher biomass production probably caused a higher release of root exudates, containing easily available C and N that enhanced microbial activity (Mounier et al., 2004; Henry et al., 2008), promoting SOM mineralization. However, further investigations

are needed to prove this explanation. The lower NUE at the C_{org}-high treatments compared to C_{org}-medium treatments reveals that plants are more independent of N input by fertilizer with increasing SOM at drained fen peatlands due to the extra N_{min} derived from enhanced mineralization processes, as mentioned above.

4.3 Fertilizer- and site-induced N_2O emissions

The observed annual N_2O emissions were markedly lower than the actual default emission factor from the Tier 1 approach for temperate, deep drained, nutrient-rich grassland of $8.2\,kg\,N_2O–N\,ha^{-1}\,yr^{-1}$ (IPCC, 2014) and at the lower end of literature values from other organic soils. Studies from Germany reported much higher N_2O emissions, ranging from 1.15 to $19.8\,kg\,N\,ha^{-1}\,yr^{-1}$ (Flessa et al., 1997; Augustin et al., 1998; Flessa et al., 1998; Beetz et al., 2013). Also investigations from other European countries showed that much higher N_2O emissions can be released from grasslands on drained peatlands. For example, Velthof et al. (1996) and van Beek et al. (2010, 2011) reported N_2O emissions ranging from 4.2 to $41.0\,kg\,N\,ha^{-1}\,yr^{-1}$ for the Netherlands, whereas at boreal regions N_2O emissions of up to $9\,kg\,N\,ha^{-1}\,yr^{-1}$ were measured (Nykänen et al., 1995; Maljanen et al., 2004; Regina et al., 2004). The observed N_2O emissions were also in the range of those reported from grasslands on mineral soils in Germany, summarized by Jungkunst et al. (2006). In line with our results, Flessa et al. (1998) also found that N_2O losses from peat soils are not always larger than from nearby mineral soils, but in contrast, Maljanen et al. (2010) found on average four times higher N_2O emissions from cultivated organic soils than from mineral soils. The N_2O emissions from the C_{org}-high site significantly exceeded those from the C_{org}-medium site in all treatments, which was in line with higher N_{min} contents and higher groundwater levels. Additionally, the observed markedly stronger increase of cumulative annual N_2O emissions with increasing N-uptake by plants (Fig. 5b) reveals that with increasing N availability a higher proportion is lost as N_2O at the C_{org}-high treatments compared to the C_{org}-medium treatments. This probably could be attributed to the more favorable soil conditions for denitrification, due to higher C and N mineralization rates and alternating groundwater levels, promoting anaerobicity (Koops et al., 1996). Moreover, as mentioned above, net nitrification entirely controls net nitrogen mineralization, promoting also N_2O losses, but probably to a lesser extent. However, the source of N_2O production in soils is often uncertain because aerobic and anaerobic microsites can occur within close proximity and thus nitrification and denitrification as well as other abiotic processes producing N_2O (e.g., nitrifier–denitrification, coupled nitrification–denitrification) can run simultaneously (Davidson et al., 1986; Butterbach-Bahl et al., 2013). Despite surprisingly low N_2O emission levels, we confirmed our hypothesis that N_2O emissions increase with increasing

soil C_{org} content, probably due to more favorable conditions for denitrification.

The observed background emissions on the two organic soils correspond well to those on mineral agricultural soils (Bouwman, 1996). However, calculated emission factors as a percentage of applied N without consideration of the NH_3 losses were lower for all treatments than the IPCC default value. Several other studies also reported emission factors < 1 % of applied N (Chadwick et al., 2000; Velthof et al., 2003; Clemens et al., 2006; Jones at al., 2007; Möller and Stinner, 2009), but so far not for organic soils. Indeed, N_2O studies on organic soils rarely differentiate between fertilizer and soil derived N sources by unfertilized control plots as we do in this study. In line with Möller and Stinner (2009), the application of biogas digestate resulted in a markedly higher percentage of N_2O produced from applied N compared to cattle slurry, yet at a low level.

One reason of generally low N_2O emissions observed in the present study could be the small number of frost–thaw cycles in 2011. In general frost–thaw cycles are considered to favor high N_2O emissions (Flessa et al., 1998; Jungkunst et al., 2006) but these observations seem to be more pronounced for croplands than for grasslands in Germany (Dechow and Freibauer, 2011). Denitrification activity is strongly related to the NO_3^- content close to the groundwater level (van Beek et al., 2004). Given the high NO_3^- contents, in particular in the C_{org}-high soil, the evidence for fast nitrification and high net nitrogen mineralization, we argue that frequent but low dosage application of fertilizer and quick N uptake by plants avoid conditions favorable for high N_2O emissions. Moreover, through the splash plate application technique high amounts of NH_4^+ were rapidly lost as NH_3, and therefore reduced the proportion of immediately available N for nitrification and denitrification.

As expected from the literature, highest N_2O fluxes were found immediately after fertilizer application. The initial N_2O peak could mainly be attributed to the denitrification of available soil NO_3^-, presumably due to the more favorable conditions for denitrification through the addition of easily degradable organic C and water (Comfort et al., 1990; Chadwick et al., 2000; Velthof et al., 2003). Additionally, a probably smaller part of initial N_2O could be ascribed to the rapid nitrification (Chadwick et al., 2000) or to nitrifier denitrification of slurry NH_4^+. In contrast, the partially observed second N_2O peak, mostly found a week after fertilizer application, can be attributed to the denitrification of mineralized and nitrified organic components of fertilizer N (Velthof et al., 2003).

Several authors proposed that the more recalcitrant digestate might reduce the rate of microbial degradation and oxygen consumption in the soil, resulting in reduced N_2O emissions through less anaerobic soil conditions (Clemens and Huschka, 2001; Oenema et al., 2005; Möller and Stinner, 2009). In contrast, our study on organic soils found significantly higher N_2O emissions from the digestate treatments compared to the slurry treatments. Higher N_2O emissions derived from biogas digestates were also reported by a few other authors (e.g. Senbayram et al., 2009; Sänger et al., 2010), whereas Clemens et al. (2006) found no differences between untreated and digested slurry.

It can be assumed that in drained organic soils, as in the present study, sufficient metabolizable C is generally widely available in the upper soil profile (e.g. van Beek al., 2004). Thus, as hypothesized, labile carbon is not limiting on organic soils. This was in line with Velthof et al. (2003), who supposed that the application of available C with the organic fertilizer has a larger effect on denitrification activity at soils with a lower C_{org} content compared to C_{org} rich soils. However, contrary to our hypothesis, the significantly higher N_2O emissions from the digestate treatments cannot solely be explained by the higher content of available N in the biogas digestate, since the amount of applied NH_4^+–N in the substrate was not markedly different. Nevertheless, as mentioned above, the much higher N-uptake at the biogas treatments (Fig. 5a) indicates that the application of this fertilizer resulted in a markedly higher N availability, promoting N_2O production. It could be assumed that the high pH and the lower C : N ratio of the biogas digestate obviously slightly enhanced SOM mineralization, probably due to increased microbial activity compared to cattle slurry fertilizer, leading to extra N for nitrification and denitrification. Moreover, as proposed above, it is also conceivable that the higher biomass production at these treatments itself is related to the increased N_2O emissions, due to the stimulating effect of plant roots on denitrification activity (Klemedtsson et al., 1987; Bakken, 1988). Considering that increasing biomass production means increasing root growth and activity, it could be assumed that exudation of easily available organic C and N (Hailer and Stolp, 1985), as well as the O_2 demand due to root respiration, is higher for the digestate treatments, promoting more anaerobic microsites and thus denitrification compared to cattle slurry (Erich et al., 1984; Klemedtsson et al., 1987). Furthermore, several authors have suggested that root exudates may increase bacterial metabolism (Klemedtsson, 1987; Mounier et al., 2004; Henry et al., 2008), further lowering the oxygen concentration and thus increasing denitrification (Klemedtsson et al., 1987). However, the enhanced biomass production simultaneously should have depleted mineral N in the soil and thus reduced available N for nitrification and denitrification processes. Obviously, despite the negative apparent N balance of the biogas digestate treatment, there was no real nitrogen competition between plants and microbes. However, further investigations are required to prove whether digestates enhanced SOM mineralization or to what extent increased biomass production favors N_2O emissions and which N pathways and processes are involved.

The observed linear increase in the cumulative N_2O–N emissions during the first 16 days or annual N_2O emissions,

due to a higher mean groundwater level and a higher application rate of NH_4^+–N, reveals the importance of site adapted N fertilization and the avoidance of N surpluses during agricultural use of C_{org}-rich grasslands. The same was postulated for mineral soils by Ruser (2010).

4.4 Fertilizer- and site-induced CH_4 emissions

The observed consumption rates of CH_4 were in the range of CH_4 uptakes reported by Flessa et al. (1998) for two different meadows in a southern German fen peatland. Slightly higher CH_4 emissions of up to $1.46\,kg\,CH_4$–C $ha^{-1}\,yr^{-1}$ were reported from Beetz et al. (2013) for a drained intensive grassland in northern Germany and from Nykänen et al. (1995) for a drained grassland in Finland. It is known that drainage turns peatlands from a significant source back to a sink of CH_4 (Crill et al., 1994). In peatlands the position of the groundwater table is considered the key factor regulating methanogenic and methanotrophic processes (Whalen, 2005). In line with this, Flessa et al. (1998) showed that the consumption rate of CH_4 increased with lowering of the groundwater level. Nevertheless, significant differences in the amount of the annual CH_4 uptake capacity between the two study sites C_{org}-medium and C_{org}-high could not be seen, although distinct differences in the groundwater table were observed.

The occasionally observed CH_4 peak emissions were only found immediately after cattle slurry application. This was in line with several other studies that reported short-term CH_4 emissions immediately after organic fertilizer application, due probably to volatilization of dissolved CH_4 from the applied substrate (Sommer et al., 1996; Chadwick et al., 2000; Wulf et al., 2002b; Jones et al., 2005; Amon et al., 2006). The longer lasting CH_4 emissions observed after the first application event at the slurry treatment of the C_{org}-high site might result from the degradation of volatile fatty acids by methanogenic bacteria (Chadwick et al., 2000; Wulf et al., 2002b). Furthermore, the high groundwater level promotes the formation of CH_4 during this time period. However, we could not find any significantly differences in the short term or annual CH_4 emissions between the two investigated fertilizers. According to Chadwick et al. (2000) more than 90 % of total CH_4 emissions occur during the first 24 h following fertilizer application. Therefore, we must assume that we have missed most of the fertilizer-induced CH_4 emissions. However, all studies from literature confirm the minor importance of CH_4 emissions from applied organic fertilizers in the GHG balance of agricultural grasslands (Wulf et al., 2002b; Amon et al., 2006; Dietrich et al., 2012).

4.5 N-losses by NH_3 volatilization

The NH_3 losses measured after splash plate application at the third application event followed the typical pattern of lost ammonia (Clemens et al., 2006), particularly at the digestate treatments. Significantly higher NH_3 losses from treatments

fertilized with biogas digestate were observed compared to those fertilized with cattle slurry. This is in line with several other studies (Messner, 1988, Döhler and Haring, 1989 (cited in Döhler and Horlacher, 2010); Amon et al., 2006; Möller and Stinner, 2009; Pacholski et al., 2010; Ni et al., 2011), whereas Pain et al. (1990), Rubæk et al. (1996), Wulf et al. (2002a) and Clemens et al. (2006) found no differences between anaerobic digested slurries and other animal slurries. However, it has to be taken into account that the present results are based only on measurements from a single application event. The observed relative N losses of 36 % of applied NH_4^+–N at the biogas digestate treatments were in the range reported for liquid slurries and digestates applied via surface application, whereas the significantly lower relative N losses (15 %) at the cattle slurry treatments stand in strong contrast to those reported in the literature (e.g., Döhler and Haring, 1989 (cited in Döhler and Horlacher, 2010); Smith et al., 2000; Wulf et al., 2002a; Chantigny et al., 2004). However, compared to the EF of 60 % used in the German national GHG inventory, both estimated NH_3 loss rates were rather low (Haenel et al., 2014). It can be assumed that the higher concentration of NH_4^+ (NH_4^+/N_{tot} ratio 0.65 vs. 0.33) and the markedly higher pH value (7.7 vs. 6.8) of the applied digestate compared to the cattle slurry caused the observed differences in the current study, since temperature and wind speed were equal. According to Sommer and Hutchings (2001) a change in the pH value from 7.7 to 8.0 will double the emission. However, the factors controlling the rise in pH are complex (Sommer and Husted, 1995 cited in Sommer and Hutchings, 2001) and the pH value was not determined after fertilization in the present study. Several authors propose that a lower dry matter content of slurries favors the infiltration into the soil with a subsequent faster decrease of NH_3 losses (Sommer et al., 1996; Ni et al., 2011). There over a limited range (slurry DM of 2–5 %), NH_3 losses increase by approximately 6 % for every 1 % DM content (Smith et al., 2000). Although the observed dry matter content of the biogas digestates was very low and at the lower end of values reported in the literature (e.g., Gutser et al., 2005; Möller et al., 2008; Quarkernack et al., 2011), no corresponding effect was found in the present study, as was also reported by Möller and Stinner (2009). According to Döhler and Horlacher (2010) and Smith et al. (2000), water-saturated grassland soils as well as very dry grassland soils high in organic matter lead to higher NH_3-losses due to the reduced infiltration of slurries. Thus it could be assumed that the infiltration of the slurries was possibly hampered in the current study, removing the effect of the different DM contents, due to the strong rain event that took place before the fertilizer application. The cattle slurry in our experiment had very favorable characteristics for crust formation (high DM content, grass silage diet; Smith et al., 2007). Warm weather also supported crust formation after application of cattle slurry, which can effectively inhibit NH_3 exchange with the atmosphere and has been proposed as an NH_3 mitigation measure for slurry

storage (Smith et al., 2007). The emission pattern observed in our study on soil with limited infiltration capacity supports the effectiveness of crusts for low NH_3 losses. Additionally, at low dosage applications a large part of the organic fertilizer remained on the plant canopy and thus soil contact and infiltration was limited after spreading. We conclude that this was also the main reason why no significant differences in the pattern of NH_3 volatilization between the soil types were found in the present study. Nevertheless, the markedly lower relative N losses from cattle slurry compared to literature values could not be explained in this way, but NH_3 volatilization reported in literature showed a high variability in respect to climatic and soil conditions, slurry composition, and application technique.

The observed relative N losses of 15–36 % of applied NH_4^+–N demonstrates that NH_3 volatilization is quantitatively the most important N loss from slurry application, as was also proposed by Flessa and Beese (2000). Besides the negative effects of eutrophication and acidification of ecosystems (e.g., Fangmeier et al., 1994; Galloway, 1995; Smith et al., 1999; Galloway, 2001), marked NH_3 volatilization decreases N fertilizer use efficiency. One of the most effective measures to reduce NH_3 emissions from grassland is the incorporation of slurry (Rodhe et al., 2006). However, several studies reported a considerable increase of GHGs, mainly N_2O, after injection of slurries and biogas digestates (Dosch and Gutser, 1996; Flessa and Beese, 2000; Wulf et al., 2002b). To date no study has examined the effect of the injection technique on organic soils.

5   Conclusions

We studied N_2O, CH_4, and NH_3 fluxes after splash plate application of biogas digestate and cattle slurry in a region known for its risk of high N_2O and NH_3 emissions and we were the first to study digestate application on high organic carbon soils with 10 to 17 % C_{org} content in the topsoil. To our surprise, N_2O emissions and EFs were lower than generally observed on mineral soils in the vicinity of the sites. We attributed the low N_2O emissions to a mild winter without clear freeze–thaw cycles, but maybe also to frequent application with low dosage of N, which was quickly taken up by the grass vegetation, as was indicated by the apparent NUE_{min}. N_2O emissions increased with C_{org} content and fertilization. As hypothesized, N_2O levels were markedly higher after digestate than after slurry fertilization, which probably could be attributed to a priming effect caused by increased SOM mineralization or other sources of labile carbon triggered by digestate application. Due to the deep drainage, CH_4 emissions were of minor importance and were independent of fertilizer type. Estimated N balances were negative for the control and the digestate treatments, but strongly positive in all cases when the net N supply from SOM mineralization was considered. The observed linear increase in cumulative N_2O

emissions with increasing NH_4^+ fertilization and increasing groundwater table reveals the importance of site adapted N fertilization and the avoidance of N surpluses during agricultural use of C_{org}-rich grasslands.

Edited by: T. Laurila

References

Amon, B., Kryvoruchko, G. M., and Amon, T.: Greenhouse gas and ammonia emission abatement by slurry treatment, International Congress Series, 1293, 295–298, 2006.

Andres, M., Hagemann, U., Seidel, A., and Augustin, J.: Gasförmige N-Verluste (N_2O, NH_3) und N-Bilanzsalden beim Anbau von Energiemais mit Gärrestapplikation, in: Böden – Lebensgrundlage und Verantwortung, Jahrestagung der DBG, Symposium AG Bodengase, 7–12 September 2013, Rostock, 2013 (in German).

Augustin, J., Merbach, W., Steffens, L., and Snelinski, B.: Nitrous Oxide Fluxes of Disturbed Minerotrophic Peatlands, Agribiol. Res., 51 , 47–57, 1998.

Bakken, L. R.: Denitrification under different cultivated plants: effects of soil moisture tension, nitrate concentration, and photosynthetic activity, Biol. Fertil. Soils, 6, 271–278, 1988.

Beetz, S., Liebersbach, H., Glatzel, S., Jurasinski, G., Buczko, U., and Höper, H.: Effects of land use intensity on the full greenhouse gas balance in an Atlantic peat bog, Biogeosciences, 10, 1067–1082, doi:10.5194/bg-10-1067-2013, 2013.

Bouwman, A. F.: Direct emissions of nitrous oxide from agricultural soils, Nutr. Cycl. Agroecosys, 45, 53–70, 1996.

Butterbach-Bahl, K., Baggs, E. M., Dannenmann, M., Kiese, R., and Zechmeister-Boltenstern, S.: Nitrous oxide emissions from soils: how well do we understand the processes and their controls?, Philos. T. R. Soc. B., 368, 1621, 2013.

Chadwick, D. R., Pain, B. F., and Brookman, S. K. E.: Nitrous oxide and Methane Emissions following Application of Animal Manures to Grassland, J. Environ. Qual., 29, 277–287, 2000.

Chantigny, M. H., Rochette, P., Angers, D. A., Massé, D., and Côté, D.: Ammonia volatilization and selected soil characteristics fol-

lowing application of anaerobically digested pig slurry, Soil Sci. Soc. Am. J., 68, 306–312, 2004.

Christensen, S.: Nitrous oxide emission from a soil under permanent grass: Seasonal and diurnal fluctuations as influenced by manuring and fertilisation, Soil Biol. Biochem., 15, 531–536, 1983.

Clayton, H., McTaggart, I. P., Parker, J., and Swan, L.: Nitrous oxide emissions from fertilised grassland: A 2-year study of the effects of N fertiliser form and environmental conditions, Biol Fertil Soils, 25, 252–260, 1997.

Clemens, J. and Huschka, A.: The effect of biological oxygen demand of cattle slurry and soil moisture on nitrous oxide emissions, Nutr. Cycling Agroecosyst., 59, 193–198, 2001.

Clemens, J., Trimborn, M., Weiland, P., and Amon, B: Mitigation of greenhouse gas emissions by anerobic digestion of cattle slurry, Agr. Ecosyst. Environ., 112, 171–177, 2006.

Comfort, S. D., Kelling, K. A., Keeney, D. R., and Converse, J. C.: Nitrous oxide production from injected dairy manure, Soil Sci. Soc. Am., 1, 421–427, 1990.

Crawley, M. J.: The R Book, John Wiley and Sons Ltd, Chichester, p. 942, 2007.

Crill, P. M., Martikainen, P. J., Nykänen, H., and Silvola, J.: Temperature and fertilization effects on methane oxidation in a drained peatland soil, Soil Biol. Biochem., 26, 1331–1339, 1994.

Crutzen, P. J.: The role of NO and NO_2 in the chemistry of the troposphere and stratosphere, Annu. Rev. Earth Pl. Sc., 7, 443–472, 1979.

Davidson, E. A., Swank, W. T., and Perry, T. O.: Distinguishing between nitrification and denitrification as sources of gaseous nitrogen production in soil, Appl. Environ. Microb., 52, 1280–1286, 1986.

DBFZ (Deutsches BiomasseForschungsZentrum gemeinnützige GmbH): Monitoring zur Wirkung des Erneuerbare-Energien-Gesetz (EEG) auf die Entwicklung der Stromerzeugung aus Biomasse – Endbericht zur EEG-Periode 2009 bis 2011, p. 120, 2012 (in German).

De Boer, H. C.: Co-digestion of animal slurry can increase short-term nitrogen recovery by crops, J. Environ. Qual., 37, 1968–1973, 2008.

Dechow, R. and Freibauer, A.: Assesment of German nitrous oxide emissions using empirical modelling approaches, J. Nutr. Cycl. Agroecosyst., 91, 235–254, 2011.

Dietrich, B., Finnan, J., Frost, P., Gilkinson, S., and Müller, C.: The extent of methane (CH_4) emissions after fertilisation of grassland with digestate, Biol. Fertil. Soils, 48, 981–985, 2012.

Don, A., Osborne, B., Hastings, A., Skiba, U., Carter, M. S., Drewer, J., Flessa, H., Freibauer, A., Hyvönen, N., Jones, M. B., Lanigan, G. J., Mander, Ü., Monti, A., Djomo, S. N., Valentine, J., Walter, K., Zegada-Lizarazu, W., and Zenone, T.: Land-use change to bioenergy production in Europe: implications for the greenhouse gas balance and soil carbon, GCB Bioenergy, 4, 372–391, doi:10.1111/j.1757-1707.2011.01116.x, 2012.

Dosch, P. and Gutser, R.: Reducing N losses (NH_3, N_2O, N_2) and immobilization from slurry through optimized application techniques, Fer. Res., 43, 165–171, 1996.

Döhler, H. and Horlacher, D.: Ammoniakemissionen organischer Düngemittel, in Emissionen landwirtschaftlich genutzter Böden, KTBL-Schrift, 483, p. 384, 2010 (in German).

Drösler, M.: Trace gas exchange and climatic relevance of bog ecosystems, Southern Germany, PhD thesis, Chair of Vegetation Ecology, Department of Ecology, Technical University Munich, p. 179, 2005.

Drösler, M., Freibauer, A., Adelmann, W., Augustin, J., Bergman, L., Beyer, C., Chojnicki, B., Förster, C., Giebels, M., Görlitz, S., Höper, H., Kantelhardt, J., Liebersbach, H., Hahn-Schöfl, M., Minke, M., Petschow, U., Pfadenhauer, J., Schaller, L., Schägner, P., Sommer, M., Thuille, A., and Wehrhan, M.: Klimaschutz durch Moorschutz in der Praxis – Ergebnisse aus dem BMBF-Verbundprojekt "Klimaschutz – Moornutzungsstrategien" 2006–2010, 2011 (in German).

Eickenscheidt, N., Brumme, R., and Veldkamp, E.: Direct contribution of nitrogen deposition to nitrous oxide emissions in a temperate beech and spruce forest – a ^{15}N tracer study, Biogeosciences, 8, 621–635, doi:10.5194/bg-8-621-2011, 2011.

Elsässer, R. M., Kunz, H. G., and Briemle, G.: Unterschiedliche technische Behandlung von Gülle und deren Auswirkungen auf intensiv genutztes Grünland, J. Agron. Crop. Sci., 174, 253–264, 1995 (in German).

Erich, M. S., Bekerie, A., and Duxbury, J. M.: Activities of denitrifying enzymes in freshly sampled soils, Soil Sci., 138, 25–32, 1984.

Fachverband Biogas e.V.: Branchenzahlen – Prognosen 2013/2014, Entwicklung des jährlichen Zubaus von neuen Biogasanlagen in Deutschland, available at: http://www.biogas.org/edcom/webfvb.nsf/id/DE_Branchenzahlen, last access: November 2013.

Fangmeier, A., Hadwiger-Fangmeier, A., van der Eerden, L., and Jäger, H. J.: Effects of atmospheric ammonia on vegetation – a review, Environ. Pollu., 86, 43–82, 1994.

Flessa, H. and Beese, F.: Laboratory Estimates of Trace Gas Emissions following Surface Application and Injection of Cattle Slurry, J. Environ. Qual., 29, 262–268, 2000.

Flessa, H., Wild, U., Klemisch, M., and Pfadenhauer, J.: C- und N-Stoffflüsse auf Torfstichsimulationsflächen im Donaumoos, Z. f. Kulturtechnik und Landentwicklung, 38, 11–17, 1997 (in German).

Flessa, H., Wild, U., Klemisch, M., and Pfadenhauer, J.: Nitrous oxide and methane fluxes from organic soils under agriculture, Eur. J. Soil Sci., 49, 327–335. 1998.

Freibauer, A. and Kaltschmitt, M.: Nitrous oxide emissions from agricultural mineral soils in Europe – controls and models, Biogeochemistry, 63, 93–115, 2003.

Freibauer, A., Rounsevell, M. D. A., Smith, P., and Verhagen, J.: Carbon sequestration in the agricultural soils of Europe, Geoderma, 122, 1–23, 2004.

Frick, R.: Verteilgeräte an Güllefässern – Grosse Unterschiede bezüglich Arbeitsbreite und Verteilgenauigkeit, FAT Berichte, 531, 1–37, 1999 (in German).

Galloway, J. N.: Acid deposition: Perspectives in time and space, J. Water, Air Soil Poll., 85, 15–24, 1995.

Galloway, J. N.: Acidification of the world: natural and anthropogenic, J. Water, Air, Soil Poll., 130, 17–24, 2001.

Goldberg, S. D., Knorr, K. H., Blodau, C., Lischeid, G., and Gebauer, G.: Impact of altering the water table height of an acidic fen on N_2O and NO fluxes and soil concentrations, Global Change Biol., 16, 220–233, 2010.

Gunnarsson, A., Bengtsson, F., and Caspersen, S.: Use efficiency of nitrogen from biodigested plant material by ryegrass, J. Plant Nutr. Soil Sci., 173, 113–119, 2010.

Gutser, R., Ebertseder, Th., Weber, A., Schraml, M., and Schmidhalter, U.: Short-term and residual availability of nitrogen after long-term application of organic fertilizers on arable land, J. Plant Nutr. Soil Sci., 168, 439–446, 2005.

Gutser, R., Ebertseder, T., Schraml, M., Tucher von, S., and Schmidhalter, U.: Stickstoffeffiziente und umweltschonende organische Düngung, in Emissionen landwirtschaftlich genutzter Böden, KTBLSchrift, 483, p. 384, 2010 (in German).

Hacin, J., Cop, J., and Mahne, I.: Nitrogen mineralization in marsh meadows in relation to soil organic matter content and watertable level, J. Plant Nutr. Soil Sci., 164, 503–509, 2001.

Haenel, H. D., Freibauer, A., Rösemann, C., Poddey, E., Gensior, A., Menden, B. M., and Döhler, H.: Emissionen landwirtschaftlich genutzter Böden im Rahmen der deutschen Klimaberichterstattung, in Emissionen landwirtschaftlich genutzter Böden, KTBL-Schrift, 483, p. 384, 2010 (in German).

Haenel, H. D., Rösemann, C., Dämmgen, U., Poddey, E., Freibauer, A., Wulf, S., Eurich-Menden, B., Döhler, H., Schreiner, C., Bauer, B., and Osterburg, B.: Calculations of gaseous and particulate emissions from German agriculture 1990–2012, Johann Heinrich von Thünen-Institut, Braunschweig, 348 pp., Thünen Rep 17, 2014.

Hahn-Schöfl, M., Zak, D., Minke, M., Gelbrecht, J., Augustin, J., and Freibauer, A.: Organic sediment formed during inundation of a degraded fen grassland emits large fluxes of CH_4 and CO_2, Biogeosciences, 8, 1539–1550, doi:10.5194/bg-8-1539-2011, 2011.

Hailer, T. and Stolp, H.: Quantitative estimation of root exudation of maize plants, Plant Soil, 86, 207–216, 1985.

Henry, S., Texier, S., Hallet, S., Bru, D., Dambreville, C., Chèneby, D., Bizouard, F., Germon, J. C., and Philippot, L.: Disentangling the rhizosphere effect on nitrate reducers and denitrifiers: insight into the role of root exudates, Environ. Microbiol., 10, 3082–3092, 2008.

Hothorn, T., Bretz, F., Westfall, P., Heiberger, R. M., and Schuetzenmeister, A.: Simultaneous Inference in General Parametric Models: R package version 1.2-17, 2013.

IPCC: Changes in atmospheric constituents and in radiative forcing, in: Climate Change 2007: The physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, edited by: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K. B., Tignor, M., and Miller, H. L., Cambridge University Press, Cambridge, UK and New York, USA, 2007.

IPCC: 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands, edited by: Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M. and Troxler, T. G., Published: IPCC, Switzerland, 2014.

Jensen, L. S., Pedersen, I. S., Hansen, T. B., and Nielsen, N. E.: Turnover and fate of ^{15}N-labelled cattle slurry ammonium-N applied in the autumn to winter wheat, Eur. J. Agron., 12, 23–35, 2000.

Jones, S. K., Rees, R. M., Skiba, U. M., and Ball, B. C.: Greenhouse gas emissions from managed grassland, Global Planet. Change, 47, 201–211, 2005.

Jones, S. K., Rees, R. M., Skiba, U. M., and Ball, B. C.: Influence of organic and mineral N fertiliser on N_2O fluxes from temperate grassland, Agriculture, Ecosystems Environ., 121, 74–83, 2007.

Jungkunst, H. F., Freibauer, A., Neufeldt, H., and Bareth, G.: Nitrous oxide emissions from agricultural land use in Germany – a synthesis of available annual field data, J. Plant Nutr. Soil Sci., 169, 341–351, 2006.

Kasimir Klemedtsson, Å., Klemedtsson, L., Berglund, K., Martikainen, P., Silvola, J., and Oenema, O.: Greenhouse gas emissions from farmed organic soils: a review, Soil Use Manage., 13, 1–6, 1997.

Klemedtsson, L., Svensson, B. H., and Rosswall, T.: Dinitrogen and nitrous oxide produced by denitrification and nitrification in soil with and without barley plants, Plant Soil, 99, 303–319, 1987.

Klemedtsson, L., von Arnold, K., Weslien, P., and Gundersen, P.: Soil CN ratio as a scalar parameter to predict nitrous oxide emissions, Glob. Change Biol., 11, 1142–1147, 2005.

Koops, J. G., Oenema, O., and van Beusichem, M. L.: Denitrification in the top soil of grassland on peat soil, Plant Soil, 184, 1–10, 1996.

Livingston, G. P. and Hutchinson, G. L.: Enclosure-based measurement of trace gas exchange: application and sources of error, in: Biogenic Trace Gases: Measuring Emissions from Soil and Water, edited by: Matson, P. A. and Harriss, R. C., Blackwell Science, Cambridge, 14–50, 1995.

Maljanen, M., Komulainen, V. M., Hytönen, J., Martikainen, P .J., and Laine, J.: Carbon dioxide, nitrous oxide and methane dynamics in boreal organic agricultural soils with different soil characteristics, Soil Biol. Biochem., 36, 1801–1808, 2004.

Maljanen, M., Sigurdsson, B. D., Gudmundsson, J., Óskarsson, H., Huttunen, J. T., and Martikainen, P. J.: Greenhouse gas balances of managed peatlands in the Nordic countries – present knowledge and gaps, Biogeosciences, 7, 2711–2738, doi:10.5194/bg-7-2711-2010, 2010.

Merino, P., Estavillo, J. M., Graciolli, L. A., Pinto, M., Lacuesta, M., Muñoz-Rueda, A., and Gonzalez-Murua, C.: Mitigation of N_2O emissions from grassland by nitrification inhibitor and Actilith F2 applied with fertilizer and cattle slurry, Soil Use Manage., 18, 135–141, 2002.

Messner, H.: Düngewirkung anaerob fermentierter und unbehandelter Gülle, Dissertation, Fakultät für Landwirtschaft und Gartenbau, TU München, 1988 (in German).

Möller, K. and Stinner, W.: Effects of different manuring systems with and without biogas digestion on soil mineral nitrogen content and gaseous nitrogen losses (ammonia, nitroux oxide), Europ. J. Agronomy, 30, 1–16, 2009.

Möller, K. and Müller, T.: Effects of anaerobic digestion on digestate nutrient availability and crop growth: A review, Eng. Life Sci., 12, 242–257, 2012.

Möller, K., Stinner, W., Deuker, A., and Leithold, G.: Effects ofdifferent manuring systems with and without biogas digestion on nitrogen cycle and crop yield in mixed organic dairy farming systems, Nutr. Cycl. Agroecosyst, 82, 209–232, doi:10.1007/s10705-008-9196-9, 2008.

Moiser, A. R.: Exchange of gaseous nitrogen compounds between agricultural systems and the atmosphere, Plant Soil, 228, 17–27, 2001.

Mounier, E., Hallet, S., Chèneby, D., Benizri, E., Gruet, Y., Nguyen, C., Piutti, S., Robin, C., Slezack-Deschaumes, S., Martin-

Laurent, F., Germon, J. C., and Philippot, L.: Influence of maize mucilage on the diversity and activity of the denitrifying community, Environ. Microbiol., 6, 301–312, 2004.

Ni, K., Pacholski, A., Gericke, D., and Kage, H.: Analysis of ammonia losses after field application of biogas slurries by an empirical model, J. Plant Nutr. Soil Sci., 175, 1–12, doi:10.1002/jpln.201000358, 2011.

Nykänen H., Alm, J., Lång, K., Silvola, J., and Martikainen, P. J.: Emissions of CH_4, N_2O and CO_2 from a virgin fen and a fen drained for grassland in Finland, J. Biogeogr., 22, 351–357, 1995.

Oenema, O., Wrage, N., Velthof, G. L., van Groenigen, J. W., Dolfing, J., and Kuikman, P. J.: Trends in global nitrous oxide emissions from animal production systems, Nutr. Cycl. Agroecosys., 72, 51–65, 2005.

Pacholski, A., Cai, G., Nieder, R., Richter, J., Fan, X., Zhu, Z., and Roelcke, M.: Calibration of a simple method for determining ammonia volatilization in the field – comparative measurements in Henan Province, China, Nutr. Cycl. Agroecosyst., 74, 259–273, 2006.

Pacholski, A., Gericke, D., Ni, K., and Kage, H.: Ammoniakemissionen nach Ausbringung von Gärresten im Vergleich zu Güllen, KTBL-Schrift, 483, p. 384, 2010 (in German).

Pain, B. F., Misselbrook, T. H., Clarkson, C. R., and Rees, Y. J.: Odour and ammonia emissions following the spreading of anaerobically-digested pig slurry on grassland, Biol. Wastes, 34, 259–267, 1990.

Quarkernack, R., Pacholski, A., Techow, A., Herrmann, A., Taube, F., and Kage, H.: Ammonia volatilization and yield response of energy crops after fertilization with biogas residues in a coastal marsh of Northern Germany, Agric. Ecosyst. Environ., 160, 66–74, doi:10.1016/j.agee.2011.05.030, 2011.

Regina, K., Syväsalo, E., Hannukkala, A., and Esala, M: Fluxes of N_2O from farmed peat soils in Finland, Eur. J. Soil Sci., 55, 591–599, 2004.

R Development Core Team: R: A language and environment for statistical computing, R Foundation for Statistical Computing, Vienna, Austria, ISBN 3-900051-07-0, available at: http://www.R-project.org (last access: November 2013), 2010.

Renger, M., Wessolek, G., Schwarzel, K., Sauerbrey, R., and Siewert, C.: Aspects of peat conservation and water management, J. Plant Nutr. Soil Sci., 165, 487–493, 2002.

Rodhe, L., Pell, M., and Yamulki, S.: Nitrous oxide, methane and ammonia emissions following slurry spreading on grassland, J. Soil Use Manage., 22, 229–237, 2006.

Roelcke, M., Li, S. X., Gao, Y. J., and Richter, J.: In situ comparisons of ammonia volatilization from N fertilizers in Chinese loess soils, Nutr. Cycl. Agroecosyst., 62, 73–88, 2002.

Rubæk, G. H., Henriksen, K., Petersen, J., Rasmussen, B., and Sommer, S. G.: Effects of application technique and anaerobic digestion on gaseous loss from animal slurry applied to ryegrass (Lolium perenne), J. Agric. Sci. Camb., 126, 481–492, 1996.

Ruser, R.: Möglichkeiten zur Minderung der Lachgasfreisetzung aus landwirtschaftlich genutzten Böden bei mineralischer Stickstoffdüngung, in: Emissionen landwirtschaftlich genutzter Böden, KTBL-Schrift, 483, p. 384, 2010 in (German).

Sänger, A., Geisseler, D., and Ludwig, B.: Effects of rainfall pattern on carbon and nitrogen dynamics in soil amended with biogas slurry and composted cattle manure, J. Plant Nutr. Soil Sci., 173, 692–698, 2010.

Schils, R. L. M, van Groeningen, J. W., Velthof, G. L., and Kuikman, P. J.: Nitrous oxide emissions from multiple combined applications of fertiliser and cattle slurry to grassland, Plant Soil, 310, 89–101, 2008.

Schothorst, C. J.: Subsidence of low moor peat soils in the Western Netherlands, Geoderma, 17, 265–291, 1977.

Senbayram, M., Chen, R., Mühling, K. H., and Dittert, K.: Contribution of nitrification and denitrification to nitrous oxide emissions from soils after application of biogas waste and other fertilizers, Rapid Commun. Mass Spectrom., 23, 2489–2498, 2009.

Smith, V. H., Tilman, G. D., and Nekola, J. C.: Eutrophication: impacts of excess nutrient input on freshwater, marine, and terrestrial ecosystems, Environ. Pollut., 100, 179–196, 1999.

Smith, K. A., Jackson, D. R., Misselbrook, T. H., Pain, B. F., and Johnson, R. A.: Reduction of ammonia emission by slurry application techniques, J. Agric. Engng. Res., 77, 277–287, 2000.

Smith, K., Cumby, T., Lapworth, J., Misselbrook, T., and Williams, A.: Natural crusting of slurry storage as an abatement measure for ammonia emissions on dairy farms, Biosyst. Eng., 97, 464–471, 2007.

Sørensen, P. and Amato, M.: Remineralisation and residual effects of N after application of pig slurry to soil, Eur. J. Agron., 16, 81–95, 2002.

Sommer, S. G. and Husted, S.: A simpel model of pH in slurry, J. Agric. Sci. Camb., 124, 447–453, 1995.

Sommer, S. G., Sherlock, R. R., and Khan, R. Z.: Nitrous oxide and methane emissions from pig slurry amended soils, Soil Biol. Biochem., 28 , 1541–1544, 1996.

Sommer, S. G. and Hutchings, N. J.: Ammonia emission from field applied manure and its reduction – invited paper, Eur. J. Agro., 15, 1–15, 2001.

Sonneveld, M. P. W. and Lantinga, E. A.: The contribution of mineralization to grassland N uptake on peatland soils with anthropogenic A horizons, Plant Soil, 340, 357–368, 2011.

Tambone, F., Genevini, P., D'Imporzano, G., and Adani, F.: Assessing amendment properties of digestate by studying the organic matter composition and the degree of biological stability during the anaerobic digestion of the organic fraction of MSW, Bioresource Technol., 100, 3140–3142, 2009.

Umweltbundesamt: Nationale Trendtabellen für die deutsche Berichterstattung atmosphärischer Emissionen seit 1990, available at: http://www.umweltbundesamt.de/daten/klimawandel/treibhausgas-emissionen-in-deutschland/distickstoffoxid-emissionen, 2014 (in German).

van Beek, C. L., Hummelink, E. W. J., Velthof, G. L., and Oenema, O.: Denitrification rates in relation to roundwater level in a peat soil under grassland, Biol. Fertil. Soils, 39, 329–336, 2004.

van Beek, C. L., Pleijter, M., Jacobs, C. M. J., Velthof, G. L., van Groenigen, J. W., and Kuikman, P. J.: Emissions of N_2O from fertilized and grazed grassland on organic soil in relation to groundwater level, Nutr. Cycl. Agroecosyst., 86, 331–340, 2010.

van Beek, C. L., Pleijter, M., and Kuikman, P. J.: Nitrous oxide emissions from fertilized and unfertilized grasslands on peat soil, Nutr. Cycl. Agroecosyst., 89, 453–461, 2011.

Velthof, G. L., Brader, A. B., and Oenema, O.: Seasonal variations in nitrous oxide losses from managed grasslands in the Netherlands, Plant Soil, 181, 263–274, 1996.

T. Eickenscheidt et al.: Short-term effects of biogas digestate and cattle slurry application **6207**

Velthof, G. L., Kuikman, P. J., and Oenema, O.: Nitrous oxide emission from animal manures applied to soil under controlled conditions, Biol. Fertil. Soils, 37, 221–230, 2003.

Weiland, P.: Biogas production: current state and perspectives, Appl. Microbiol. Biotechnol., 85, 849–860, 2010.

Whalen, S. C.: Biogeochemistry of methane exchange between natural wetlands and the atmosphere, Environ. Eng. Sci., 22 , 73–94, 2005.

WRB, 2006 – IUSS Working Group: World Reference Base for Soil Resources 2006, 2nd edition, World Soil Resources Reports No. 103, Rome, 2006.

Wulf, S., Maeting, M., and Clemens, J.: Application Technique and Slurry Co-Fermentation Effects on Ammonia, Nitrous Oxide, and Methane Emissions after Spreading: I Ammonia Volatilization, J. Environ. Qual., 31, 1789–1794, 2002a.

Wulf, S., Maeting, M., and Clemens, J.: Application Technique and Slurry Co-Fermentation Effects on Ammonia, Nitrous Oxide, and Methane Emissions after Spreading: II Greenhouse Gas Emissions, J. Environ. Qual., 31, 1795–1801, 2002b.

7.3 PUBLICATION III.

Eickenscheidt, T., Heinichen, J., Augustin, J., Freibauer, A., and Drösler, M.: Nitrogen mineralization and gaseous nitrogen losses from waterlogged and drained organic soils in a black alder (*Alnus glutinosa* (L.) Gaertn.) forest, Biogeosciences, 11, 2961–2976, doi:10.5194/bg-11-2961-2014, 2014.

Biogeosciences

Nitrogen mineralization and gaseous nitrogen losses from waterlogged and drained organic soils in a black alder (*Alnus glutinosa* (L.) Gaertn.) forest

T. Eickenscheidt[1]**, J. Heinichen**[1]**, J. Augustin**[2]**, A. Freibauer**[3]**, and M. Drösler**[1]

[1]University of Applied Sciences Weihenstephan-Triesdorf, Chair of Vegetation Ecology, Weihenstephaner Berg 4, 85354 Freising, Germany
[2]Leibniz Centre for Agricultural Landscape Research e. V., Institute of Landscape Matter Dynamics, Eberswalder Straße 84, 15374 Müncheberg, Germany
[3]Thünen Institute of Climate-Smart Agriculture, Bundesallee 50, 38116 Brunswick, Germany

Correspondence to: T. Eickenscheidt (tim.eickenscheidt@hswt.de)

Abstract. Black alder (*Alnus glutinosa* (L.) Gaertn.) forests on peat soils have been reported to be hotspots for high nitrous oxide (N_2O) losses. High emissions may be attributed to alternating water tables of peatlands and to the incorporation of high amounts of easily decomposable nitrogen (N) into the ecosystem by symbiotic dinitrogen (N_2)-fixation of alder trees. Our study addressed the question to what extent drainage enhances the emissions of N_2O from black alder forests and how N turnover processes and physical factors influence the production of N_2O and total denitrification. The study was conducted in a drained black alder forest with variable groundwater tables at a southern German fen peatland. Fluxes of N_2O were measured using the closed chamber method at two drained sites (D-1 and D-2) and one undrained site (U). Inorganic N contents and net N mineralization rates (NNM) were determined. Additionally a laboratory incubation experiment was carried out to investigate greenhouse gas and N_2 fluxes at different temperature and soil moisture conditions. Significantly different inorganic N contents and NNM rates were observed, which however did not result in significantly different N_2O fluxes in the field but did in the laboratory experiment. N_2O fluxes measured were low for all sites, with total annual emissions of 0.51 ± 0.07 (U), 0.97 ± 0.13 (D-1) and 0.93 ± 0.08 kg N_2O–N ha^{-1} yr^{-1} (D-2). Only 37 % of the spatiotemporal variation in field N_2O fluxes could be explained by peat temperature and groundwater level, demonstrating the complex interlinking of the

controlling factors for N_2O emissions. However, temperature was one of the key variables of N_2O fluxes in the incubation experiment conducted. Increasing soil moisture content was found to enhance total denitrification losses during the incubation experiment, whereas N_2O fluxes remained constant. At the undrained site, permanently high groundwater level was found to prevent net nitrification, resulting in a limitation of available nitrate (NO_3^-) and negligible gaseous N losses. N_2O flux rates that were up to four times higher were measured in the incubation experiment. They reveal the potential of high N_2O losses under changing soil physical conditions at the drained alder sites. The high net nitrification rates observed and high NO_3^- contents bear the risk of considerable NO_3^- leaching at the drained sites.

1 Introduction

Black alder (*Alnus glutinosa* (L.) Gaertn.) forests represent the natural vegetation in many semi-wet and wet regions, mostly on soils with high organic carbon contents (Dilly et al., 1999; Schäfer and Joosten, 2005; Kätzel, 2003). Unlike most other wetlands, alder forests on peatland represent non N-limited ecosystems, due to the ability of alder trees for symbiotic N_2-fixation of up to 85 kg N ha^{-1} yr^{-1} (Dittert, 1992; Augustin, 2003). To date, there are only a few studies on N turnover and N transformation processes

in black alder forests on peatland. First results indicate that these ecosystems show high N-transfer rates between the atmosphere, the alder trees and the soil (Augustin, 2003). Through the symbiotic N_2-fixation high amounts of easily decomposable N enter the ecosystem and lead to a low C/N ratio of the soil organic matter (Lõhmus et al., 2002). Semi-wet peat soils with low C/N ratios are known to be potential hotspots for gaseous and dissolved nitrogen losses via mineralization, nitrification, denitrification and nitrate leaching (Lõhmus et al., 2002; Gundersen et al., 1998; Ollinger et al., 2002; Klemedtsson et al., 2005). Enhanced N_2O emissions, in particular, are of great interest due to the fact that N_2O acts as a radiative forcing greenhouse gas (IPCC, 2007) and contributes to the chemical destruction of stratospheric ozone (Crutzen, 1979). The few annual observations of N_2O emissions from *Alnus* forest ecosystems on drained or undrained peatlands range between 0.1 and 72.0 kg N ha^{-1} yr^{-1} (Augustin et al., 1998; Brumme et al., 1999; Merbach et al., 2001; von Arnold et al., 2005). Most emissions exceed by far the IPCC default emission factor of 0.6 (0.16–2.4) kg N_2O-N ha^{-1} yr^{-1} (IPCC, 2006) for boreal and temperate nutrient-rich forest peatlands. The high range requires in-depth process understanding to narrow down the uncertainty and better understand N_2O formation, not least because of the attempt to reduce the climate impact of drained fen peatland by rewetting and afforestation with black alder trees (*Alnus glutinosa* (L.) Gaertn.) (Schäfer and Joosten, 2005; Wichtmann and Joosten, 2007).

The production of N_2O depends mainly on the two microbial processes of denitrification and nitrification (Davidson, 1986). Thus, the availability of soil mineral N and oxygen are the main controlling factors (Del Grosso et al., 2000; Jungkunst et al., 2004). Furthermore, soil temperature is known to be a key variable controlling both processes (Firestone and Davidson, 1989; Smith et al., 1998). Highest N_2O fluxes are mainly related to incomplete denitrification. This requires the presence of efficient electron acceptors (e.g., NO_3^-) to prevent the reduction of N_2O to N_2 (Bremner and Blackmer, 1981; Speir et al., 1995; Jungkunst et al., 2004). Due mainly to analytical difficulties in the determination of microbially produced N_2, to date the denitrification potential from waterlogged peat soils has rarely been estimated (Watts and Seitzinger, 2000; Mander et al., 2003; Teiter and Mander, 2005; Wray and Bayley, 2007; Roobroeck et al., 2010; Soosaar et al., 2011; Uri et al., 2011). The substitution of ambient N_2 by Helium (He) in laboratory studies possibly represents the most reliable method for the direct and simultaneous determination of N_2O and N_2 exchange rates (Butterbach-Bahl et al., 2002; Roobroeck et al., 2010; Butterbach-Bahl et al., 2013).

The objectives of our study were to quantify (i) the nitrogen mineralization and nitrogen transformation processes, and (ii) N_2O emissions and the factors regulating the N_2O emissions and N_2O/N_2 ratio from waterlogged and drained black alder forest on organic soil.

We hypothesize that (1) in drained black alder forests high N_2O emissions occur due to accelerated N turnover and N mineralization, and (2) N_2O is replaced by N_2 losses during periods of temporarily high water levels. In contrast, NO_3^- limitation in permanently waterlogged peat soils results in negligible N_2O and N_2 losses. To investigate our hypotheses, we selected three sites in an alder forest along a groundwater table gradient, representative of the small-scale heterogeneity of the forest stand.

2 Materials and methods

2.1 Study area and experimental design

The study was carried out in a drained 45-year-old black alder (*Alnus glutinosa* (L.) Gaertn.) forest stand (2.2 ha) on organic soil (Freisinger Moos, 48°22′ N, 11°40′ E; 449 m a.s.l.). According to the climate station in Weihenstephan, located 10 km northeast of the site, the 30-year mean annual temperature was 7.5 °C and the mean annual precipitation was 787 mm (1961–1990). At a German Level II monitoring area, located at 5 km distance from the alder forest, annual stand N deposition amounted to 17.67 kg N ha^{-1} yr^{-1} with a NH_4^+-N : NO_3^--N ratio of 46 : 54 in 2011 (data were collected and analyzed by the Bavarian State Institute of Forestry). The black alder stand was planted on grassland in 1965 and has not been managed. In 2009, stem density was approximately 1031 trees per hectare, with a mean height of 19.7 m and a mean diameter at breast height of 21 cm (S. Röhling, personal communication, 2012). Approximately 15 % of the forest area consists of undrained hollows whereas most parts are drained by ditches. In October 2009, we selected three sites which differed in groundwater table (Table 1) as well as in their physical and chemical soil properties relevant for N_2O processes (e.g., bulk density, carbon and nitrogen content) (Table 2). One site was situated in an undrained hollow (named U). The other two sites were situated in the drained part of the forest (named D-1 and D-2) along a gradient of drainage depth. Soils at the drained sites were classified as sapric histosols whereas the undrained site was classified as fluvisol (according to the World Reference Base for Soil Resources (WRB), 2006). The surface organic layer at the drained sites was a L-mull humus, and at the undrained site a submerged mud layer. At the site U, the dominant species in the herb layer were *Phragmites australis* and *Carex acutiformis*. Site D-1 was dominated by *Impatiens glandulifera, Impatiens parviflora, Deschampsia cespitosa* and *Circaea x intermedia*, whereas site D-2 was dominated by *Carex acutiformis, Circaea x intermedia, Rubus caesius* and *Carex elongata*.

At each site, three PVC-collars for greenhouse gas (GHG) measurements (inside dimensions 75 × 75 cm) were permanently inserted 10 cm into the soil. The collars of a site were grouped closely to each other (distance of 0.8 to

Table 1. Groundwater level, water-filled pore space (WFPS), extractable nitrate (NO_3^-) and ammonium (NH_4^+) contents of the three sites for the year 2011.

	Soil depth [cm]	U	D-1	D-2	n
Groundwater level [cm]		4 (−26/28)[a]	−41 (−71/−3)	−47 (−77/−5)	
		3 (−20/22)[b]	−42 (−65/−22)	−44 (−68/−25)	81
WFPS [Vol %]	0–10	97 (79/100) a	70 (56/87) b	63 (51/73) c	26
	10–20	99 (92/100) a	81 (65/95) b	78 (66/88) c	26
NO_3^- [mg N kg^{-1}]	0–10	7 (0/57) a	103 (36/196) b	60 (24/106) c	26
	10–20	4 (0/21) a	109 (55/193) b	62 (28/109) c	26
NH_4^+ [mg N kg^{-1}]	0–10	73 (0/209) a	18 (0/88) b	10 (0/58) b	26
	10–20	68 (0/169) a	11 (0/57) b	6 (0/24) b	26

Values are given as mean with minimum and maximum in parentheses. Means followed by the same lower-case letters indicate no significant differences between the sites within the year 2011. (Pairwise Wilcoxon rank sum test with Bonferroni correction for WFPS, NO_3^- and NH_4^+ at $P \leq 0.05$). [a] Values give the mean data from the water level loggers (15 min log interval). [b] Values give the mean groundwater level during the gas flux measurements conducted.

Table 2. Soil properties of the sites.

	Soil depth [cm]	U	D-1	D-2	n
Soil type (WRB* 2006)		Fluvisol	Sapric histosol	Sapric histosol	
Soil type (German classification KA5)		DY	KV-KM	KV-KM	
Peat depth [cm]		60	90	90	1
Organic carbon [%]	0–10	35.2 ± 1.4	14.6 ± 0.1	9.7 ± 0.4	3
	10–20	30.2 ± 0.6	13 ± 0.1	8.3 ± 0.2	3
Total nitrogen [%]	0–10	2.8 ± 0.1	1.8 ± 0.1	1.1 ± 0.0	3
	10–20	2.0 ± 0.0	1.5 ± 0.1	0.9 ± 0.0	3
C / N ratio	0–10	12 ± 0.2	8 ± 0.2	9 ± 0.6	3
	10–20	15 ± 0.1	9 ± 0.5	9 ± 0.3	3
Bulk density [g cm^{-3}]	0–10	0.14 ± 0.01	0.36 ± 0.02	0.47 ± 0.04	6
	10–20	0.13 ± 0.00	0.43 ± 0.01	0.60 ± 0.03	6
Porosity [%]	0–10	89 ± 3	79 ± 2	74 ± 1	6
	10–20	92 ± 1	79 ± 1	75 ± 1	6

Values give means ± standard error. *World Reference Base for Soil Resources.

2.5 m) due to large heterogeneity in soil properties. Boardwalks were installed to prevent oscillations of the peat through movements during the measurements. Plastic perforated tubes (JK-casings DN 50, Ø 60 mm, length 1 m) were installed close to each collar for plot-specific measurements of the groundwater table during gas flux measurements. We equipped one tube per site with a water level logger (Type Mini-Diver, Schlumberger water services), which logged the water table every 15 min. Additionally, climate stations were installed between the sites D-1 and D-2 at the beginning of March 2010 and at site U in April 2011. At each climate station, air temperature and humidity at 20 cm above soil sur-

face, soil temperatures at a depth of −2, −5 and −10 cm as well as soil moisture content at −5 cm were logged every 30 min. In addition to the recorded data, site-specific soil temperatures at three soil depths (−2, −5 and −10 cm) were determined with penetration thermometers at the beginning and end of each gas flux measurement. Moreover, plot-specific groundwater levels were measured inside the tubes without loggers during gas flux measurements.

2.2 Field measurements

2.2.1 N₂O flux measurements

We measured fluxes of N_2O every second week from December 2010 to January 2012 using the static manual chamber method with a chamber height of 0.5 m (Livingston and Hutchinson, 1995). At site U, in periods where the forest floor vegetation grew higher than the chamber height, extensions (same dimension as chamber) were installed between the collar and chamber (white, opaque; volume varied between 309 and 927 L). A detailed description of chamber dimension and configuration is given in Drösler (2005). Four gas samples were taken at four regular time intervals after chamber closure (enclosure time 60 min, in the case of two extensions 120 min). The samples were collected in 20 mL glass vials, each sealed with a butyl rubber septum. The vials were flushed with chamber air for 30 s using a portable micro pump (KNF Neuberger GmbH, NMP015B), so that the air in the vials was exchanged 32 times. In addition, the pump was used to build up an overpressure of approximately 550 mbar to protect the sample against fluctuations in atmospheric pressure during storage. Gas analyses were carried out with a gas chromatograph (Perkin and Elmer, Clarus 400 GC or Clarus 480 GC) equipped with a headspace auto sampler (Perkin and Elmer, TurboMatrix 110), a Porapack 80/100 mesh column and an electron capture detector (ECD) for N_2O (ECD temperature 380 °C). Gas flux rates were calculated from the linear change in gas concentration over time considering chamber air temperature and atmospheric pressure. Gas fluxes were accepted when the linear regression was significant ($P \leq 0.05$). In the case of small N_2O fluxes, fluxes were also accepted if the coefficient of determination was ≥ 0.90 and the regression slope was between -1 and 1 ppb min^{-1}. The cumulative annual mean exchange rate was calculated by linear interpolation between the measurement dates.

2.2.2 Soil sampling and laboratory analysis

Total carbon (C_{tot}), organic carbon (C_{org}) and total nitrogen (N_{tot}) were analyzed by the AGROLAB Labor GmbH (Bruckberg, Germany) on three mixed soil samples per site. Each sample was composed of nine individual samples collected close to each collar at two soil depths (0–10, 10–20 cm). After drying for 72 h at 40 °C, soil samples were sieved to 2 mm to remove stones and living roots.

For the determination of bulk density and porosity, three undisturbed core cutter samples (100 cm^3) were randomly taken at four depths (0–5, 5–10, 10–15, 15–20 cm) for each site.

During every gas flux measurement, one mixed soil sample consisting of nine individual samples was collected at two soil depths (0–10, 10–20 cm) for each site for the determination of mineral N ($N_{min} = NH_4^+-N + NO_3^--N$) contents.

Samples were immediately cooled in an ice box until further processing. Mineral N was extracted after shaking 40 g of fresh soil with 160 mL CaCl₂ (0.0125 M) for one hour. The extracts were filtered through a 4–7 μm filter paper (Whatman 595 1/2) and the first 20 mL of the extract were discarded (VDLUFA, 1997). The solution was frozen at -20 °C until analysis, which was conducted by the AGROLAB Labor GmbH (Bruckberg, Germany). A subsample of 20–30 g was used to determine the gravimetric water content, which was taken into account for the calculation of mineral N concentrations.

2.2.3 Net N mineralization

In 2011, net ammonification, nitrification and N mineralization rates (NNM) were determined in situ in four different periods in time (March, June, September and December). At each study site and period, three pairs of intact soil cores were taken randomly from the upper 20 cm of the soil layer using a cylindrical auger (diameter of 45 mm). In the case of flooding at the undrained site, disturbed soil samples were used. One core of each pair was sealed in low density polyethylene (LDPE) bags (19 μm) and returned into the soil for incubation (t1), which lasted 21 days (20 days in December) (e.g., Adams et al., 1989). The other soil cores were used as initial samples (t0). Mineral N contents were determined as described above after soil cores were divided into 0–10 and 10–20 cm sections. Net ammonification, net nitrification and net N mineralization were calculated according to the method described by Uri et al. (2003)

2.3 Laboratory incubation experiment

To examine the magnitude of potential total denitrification losses from the undrained and drained alder forest, a laboratory incubation experiment was performed; the experimental conditions were supposed to reproduce field conditions as closely as possible in respect to their temperature range and mean WFPS.

2.3.1 Soil core sampling and WFPS adjustment

In June 2011, eight intact soil cores with a diameter of 7.3 cm and a height of 6 cm were collected from a soil depth of 9–15 cm close to each collar of site D-2 for incubation in the laboratory (in total, 24 soil cores per site). Four samples per collar were immediately weighed and stored at 4 °C. The remaining four intact soil cores per collar were used to estimate the actual water-filled pore space (WFPS). This information was used to adjust the WFPS to 70 % and 100 % in the cores stored for laboratory incubation (six cores for 70 % and six cores for 100 % per site).

For the soil core sampling at site U, it had to be considered that the drying of the soil cores in order to adjust a WFPS of 70 % would lead to distinct shrinking of the waterlogged organic soil in the cores. To avoid this, three cubic blocks

(dimensions 40×25 cm, height 20 cm) were collected with plastic containers from which the bottom plate was removed beforehand. For transport, storage and drying to 70 % WFPS, the containers were placed on perforated metal plates. Six intact soil cores for site U were collected from these soil blocks after they reached the required WFPS level. Additionally, six soil cores per collar were directly collected in the field and treated in the same way as the samples from site D-2 (four samples for determining actual WFPS and two samples for incubation at 100 % WFPS level per collar). All soil cores were stored at 4 °C until they were placed in the incubation vessels.

After incubation the WFPS of the soil cores was determined to test the accuracy of the adjusted WFPS. The samples from site U and D-2 had exactly 100 % WFPS as well as 70 % WFPS at D-2 (on average 70.4 ± 2.2 % WFPS). For site U, however, WFPS had been adjusted to 83.0 ± 0.5 % instead of 70 %. We suspect that, due to the shrinking of the soil sample blocks, the bulk density must have changed; therefore the calculation of the required water content was not as accurate as needed to adjust a WFPS of 70 %.

2.3.2 Determination of gas fluxes

For measurements of N_2, N_2O, CO_2 and CH_4 fluxes, we applied the helium–oxygen (He–O) method (Butterbach-Bahl et al., 2002; Mander et al., 2003; Roobroeck et al., 2010; Uri et al., 2011) using four different incubation temperatures (0, 5, 15 and 25 °C). Five replications per site and moisture content were simultaneously placed in special gas-tight incubation vessels inside a climate chamber. The sixth one served as a reserve. Analyses were conducted in the laboratory of the Institute of Landscape Matter Dynamics, Leibniz Centre for Agricultural Landscape Research (ZALF), Müncheberg, Germany. Four substitution sequences with moderate evacuation (0.047 bar), followed by flushing the vessels with an artificial He/O_2 gas mixture (20.58 % O_2, 347.8 ppm CO_2, 1.780 ppm CH_4, 0.290 ppm N_2O, 3.04 ppm N_2, rest He), were conducted to remove ambient N_2. Subsequently, the air temperature of the climate chamber was set to 0 °C and a continuous He/O_2 gas flow rate of 15 mL per minute was adjusted to the vessel headspaces, followed by a 24 h period to establish a new flow equilibrium. From each vessel, we measured the N_2O, CO_2 and CH_4 headspace concentration once and the N_2 concentration three times. After gas flux measurements were done, we immediately increased the incubation temperature to the next level for 20 h and continued the measurements. Soil core samples lost approximately 5.0 ± 0.9 % and 2.7 ± 0.3 % of their initial water content at 100 % and 70/83 % WFPS during the incubation procedure of five days. Concentration of N_2 were analyzed by a micro-GC (Agilent Technologies, 3000 Micro GC), equipped with a thermal conductivity detector (TCD). Gas chromatograph settings were as follows: TCD temperature 60 °C, sample inlet 60 °C, molsieve capillary column (14 m), oven tempera-

ture 60 °C, carrier gas He 6.0 (1 mL min^{-1}). Concentrations of trace gases were analyzed by a GC (Shimadzu, Duisburg, Germany, GC–14B) equipped with a flame ionization detector (FID) for CH_4 and an ECD for N_2O and CO_2. GC settings were as follows: FID temperature 310 °C, ECD temperature 310 °C, column Porapack 80/100 mesh, oven temperature 60 °C, carrier gas N_2 6.0 (13 mL min^{-1}). Background N_2 concentrations varied between 3.5 and 4.5 ppm (ca. 3 ppm originated from the artificial He/O_2 gas mixture and 1 ppm from diffusion into the incubation measuring device). Flux rates were calculated from the actual gas concentration of the continuous flow rate from the vessels' headspaces after subtraction of a blank value from a vessel without a soil core, which is equivalent to concentrations from the artificial He/O_2 gas mixture. The lowest detectable flux rates were 200 µg CO_2-C m^{-2} h^{-1}, 0.2 µg CH_4-C m^{-2} h^{-1}, 0.5 µg N_2O-N m^{-2} h^{-1}, and 0.04 mg N_2-N m^{-2} h^{-1}.

In order to obtain the temperature effect on gas flux rates, Q_{10} values were calculated from the following equation:

$$Q_{10} = \left(\frac{R_2}{R_1} \right)^{\left(\frac{10}{T_2 - T_1} \right)},\qquad(1)$$

where Q_{10} is the temperature quotient; R_1 and R_2 are gas flux (CO_2, CH_4, N_2O, N_2) rates (mg C or N m^{-2} h^{-1}; µg C or N m^{-2} h^{-1}); and T_1 and T_2 are temperature (°C).

2.3.3 Estimation of available N_{min} and NO_3^- consumption

According to Schaufler et al. (2010) laboratory incubation experiments are limited by the fact that sample storage and incubation procedure decrease the substrate (e.g., NO_3^-) supply and thereby falsify the results. The amount of N_{min} in the solution from soil cores was not ascertained in the present study. However, a rough estimation of available NH_4^+ and NO_3^- was performed on the basis of t0 samples from the net N mineralization experiment conducted, which started three days before soil core samples were taken (Table 3). In the present study, significant amounts of gaseous N losses are only expected in the case of complete denitrification with N_2 as the end product. Thus only samples incubated at 100 % WFPS were potentially affected by NO_3^- limitation. According to our estimation, samples from site U at 100 % WFPS were presumably permanently NO_3^- limited at each temperature level during the incubation procedure, which reflects the natural site conditions. However, for samples from site D-2 at 100 % WFPS no NO_3^- limitation was expected. For soil core samples with 70/83 % WFPS, it has to be kept in mind that not only N consumption but also nitrification takes place. However, we can not totally rule out that reduced NO_3^- availability lowered the gaseous N fluxes at a later stage of incubation for samples from site D-2 at 100 % and 70 % as well as for site U at 83 % WFPS.

Table 3. Estimation of available N_{min} and NO_3^- consumption during sample storage and incubation time at 100 % WFPS.

	Time [h]	Site U	Site D-2
Mean bulk density from soil samples 9–16 cm soil depth [g cm^{-3}]		0.13	0.56
Estimated NO_3^- content [mg N kg^{-1}][a]		5.90	68.33
Estimated NH_4^+ content [mg N kg^{-1}]		23.30	0.00
Soil mass 250 cm^3 soil core [kg]		0.03	0.14
Estimated amount of NO_3^- per soil core [mg N]		0.19	9.57
Estimated amount of NH_4^+ per soil core [mg N]		0.76	0.00
Surface soil core [m^2]		0.00407	0.00407
Estimated mean N_2 flux during storage at 4 °C [mg N m^{-2} h^{-1}]		1.40	0.82
Mean N_2 flux at 0 °C [mg N m^{-2} h^{-1}]		1.55	1.03
Mean N_2 flux at 5 °C [mg N m^{-2} h^{-1}]		1.24	0.60
Mean N_2 flux at 15 °C [mg N m^{-2} h^{-1}]		1.36	1.37
Mean N_2 flux 25 °C [mg N m^{-2} h^{-1}]		1.27	6.02
Estimated mean N_2 exchange from soil cores at 4 °C [mg N]	264/432[b]	2.4527	0.8757
Mean N_2 exchange from soil cores at 0 °C [mg N]	24	0.1514	0.1006
Mean N_2 exchange from soil cores at 5 °C [mg N]	20	0.1009	0.0488
Mean N_2 exchange from soil cores at 15 °C [mg N]	20	0.1107	0.1115
Mean N_2 exchange from soil cores at 25 °C [mg N]	20	0.1034	0.4900
Sum N_2 exchange during sample preparation and incubation [mg N]		2.92	1.63
Proportion from estimated available NO_3^- [%]		> 100	17

[a] Values give start concentrations from the net N mineralization experiment, which started on the 7 June 2011. Soil core samples for incubation were taken on the 10 June 2011. [b] Samples from site D-2 (100 % WFPS) were stored for 264 h before the incubation experiment started. Samples from site U (100 % WFPS) were stored 432 h before the incubation started.

2.4 Statistical analysis

Statistical analyses were conducted using R 2.12.1 (R Development Core Team, 2013). The assumption of normality of residuals was tested using the Lilliefors or Shapiro–Wilk test and by plotting the quantile–quantile plots. Homogeneity of variances of residuals was checked with the Levene or Breusch–Pagan test and by plotting the residuals against the fitted values. If necessary, data were transformed (Box-Cox) prior to analyses. We used analysis of variance (ANOVA) (for NNM rates) or the nonparametric Kruskal–Wallis rank sum test (for WFPS, NO_3^-, NH_4^+) to compare means of samples. In case of significant differences between the means, we used Tukey's honest significant differences (TukeyHSD) or the nonparametric pairwise Wilcoxon rank sum test with Bonferroni correction for multiple comparisons. For testing two independent sample means (for WFPS, NO_3^-, NH_4^+), we used the nonparametric Mann–Whitney U test.

Due to temporal pseudoreplication of time series data (N_2O field measurements) and repeated measurements of vessels in the incubation experiment, we applied linear mixed effect models (Crawley 2007; Eickenscheidt et al., 2011; Hahn-Schöfl et al., 2011). For N_2O field measurements we set up a basic model with site type as a fixed effect and the spatial replication (individual plot) nested in time as a random effect. For the incubation experiment (N_2O, N_2, CO_2, CH_4), the fixed structure included the temperature, the site type and the soil water-filled pore space as well as all possible

interaction terms. Non significant terms were removed from the fixed structure. The individual vessel was set as a random effect. We extended the respective basic model by a variance function when heteroscedasticity was observed. In case of significant serial correlation in data, a first-order temporal autoregressive function was included in the model. Autocorrelation was tested with the Durbin–Watson test and by plotting the empirical (partial) autocorrelation structure (Eickenscheidt et al., 2011). The model extension was proven by the Akaike information criterion (AIC). For multiple comparisons, we conducted Tukey contrasts using the general linear hypothesis function from the "multcomp" package (Hothorn et al., 2013). We used nonlinear regressions to explain field N_2O fluxes and NNM rates. The model structure for mean N_2O fluxes included a linear term for the soil temperature in −2 cm soil depth and a cubic term for the mean groundwater (GW) level. Serial autocorrelation were not taken into account in the regression model. The nonlinear model for NNM rates was optimized by using the program Table Curve 3-D (version 4.0) (nonlinear equation is shown in Fig. 4). We accepted significant differences if $P \leq 0.05$. Results in the text are given as means ± 1 standard error.
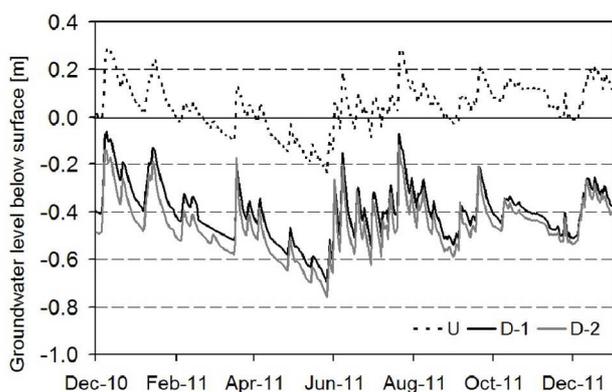
Figure 1. Annual variation in the groundwater level (U = undrained; D-1 = drained site 1; D-2 = drained site 2). Please note that the x axis starts on 1 December 2010.

3 Results

3.1 Site characteristics

In 2011, the forest air temperature in 20 cm height ranged from -12.3 to $27.8\,°C$ with an annual mean of $9.1\,°C$. Soil temperature in $-2\,cm$ soil depth averaged $9.6\,°C$ at the drained sites and was slightly lower with $9.2\,°C$ at the undrained site. The topsoil at the undrained site U consisted of a mud sediment while the topsoil at the drained sites D-1 and D-2 consisted of highly degraded fen peat. Total organic carbon and the total nitrogen concentrations in the 0–10 cm and 10–20 cm soil layers were two to three times higher at site U than at sites D-1 and D-2 (Tables 1 and 2). Due to much higher bulk density, however, the total N stock in the upper 20 cm of the soil was more than 50 % higher at D-1 and D-2 than at U. The C / N ratios of all sites were narrow (between 11 and 16), indicating nitrogen-rich conditions. No relation between soil properties and N fluxes was determined so that the results and discussion focus on groundwater table and climate variables.

The three sites showed similar dynamics in their annual hydrographs but on different levels (Fig. 1). The mean annual groundwater level was +4 cm at U, −40 cm at D-1, and −47 cm at D-2. At site U, WFPS remained permanently close to 100 %. At the drained sites, WFPS was close to 70 % (D-1) and 63 % (D-2) in the 0–10 cm layer and around 80 % in the 10–20 cm layer with some temporal variation (Table 1).

3.2 Nitrogen availability and net N mineralization

In 2011, extractable NO_3^- contents in both soil depths investigated differed significantly among the three sites (all $P < 0.03$) (Table 1). At the undrained site, nitrate was not detectable most of the time, whereas the drained sites showed high NO_3^- contents (Fig. 2a), in particular for the soil depth

of 10–20 cm. However, differences between the two soil depths were not significant.

In contrast to NO_3^-, significantly higher NH_4^+ contents were measured at the undrained site than at the drained sites (Table 1). NH_4^+ contents did not significantly vary between soil depths.

In line with the high contents of extractable inorganic N compounds, NNM rates were also high at the drained sites (Fig. 3a and b). For both soil depths, the drained sites had significantly (all $P < 0.005$) higher NNM rates than the undrained site, whereas differences between the drained sites were not significant. The NNM rates decreased with soil depth, but differences were not statistically significant. The mean NNM rates of the four time periods investigated and the upper 20 cm of the soil were $23 \pm 7\,mg\,N\,m^{-2}\,d^{-1}$ (U), $179 \pm 35\,mg\,N\,m^{-2}\,d^{-1}$ (D-1) and $142 \pm 32\,mg\,N\,m^{-2}\,d^{-1}$ (D-2). Assuming that the time periods are representative of the annual dynamics, mean annual NNM rates are estimated to be $84 \pm 26\,kg\,N\,ha^{-1}\,yr^{-1}$ (U), $653 \pm 128\,kg\,N\,ha^{-1}\,yr^{-1}$ (D-1) and $518 \pm 117\,kg\,N\,ha^{-1}\,yr^{-1}$ (D-2). The NNM rates of the upper 10 cm displayed a clear seasonality with high rates during the summer months and low rates during the winter months. However, only in June a significantly (all $P < 0.001$) higher NNM rate was observed compared to December and March. At the drained sites, net nitrification amounted to nearly 100 % of the NNM, whereas ammonification was the dominating process at the undrained site. Negative net nitrification rates (NO_3^- consumption) were detected at the undrained site in June and September in 0–10 cm soil depth. The NNM in the upper 10 cm of the soil depended on the WFPS and the mean temperature in $-5\,cm$ soil depth (Fig. 4).

3.3 Field N_2O fluxes

Annual N_2O emissions were unexpectedly low at all three sites (Fig. 5) with $0.51 \pm 0.07\,kg\,N\,ha^{-1}\,yr^{-1}$ (U), $0.97 \pm 0.13\,kg\,N\,ha^{-1}\,yr^{-1}$ (D-1) and $0.93 \pm 0.08\,kg\,N\,ha^{-1}\,yr^{-1}$ (D-2). The differences in N_2O fluxes between the sites were not statistically significant. At the drained sites the highest N_2O emissions were observed during the late summer and autumn, whereas at the undrained site highest emissions occurred during the dry period in May (Figs. 1 and 5). The temperature at $-2\,cm$ soil depth and the mean groundwater level represented the best explanation for mean N_2O fluxes; nevertheless only 37 % of the spatiotemporal variation could be explained ($P < 0.001$) (Fig. 6). Site-specific regression analyses explained 55 % of temporal N_2O variation at D-1 ($P < 0.001$), 32 % at D-2 ($P < 0.01$) and 20 % at U ($P < 0.04$).
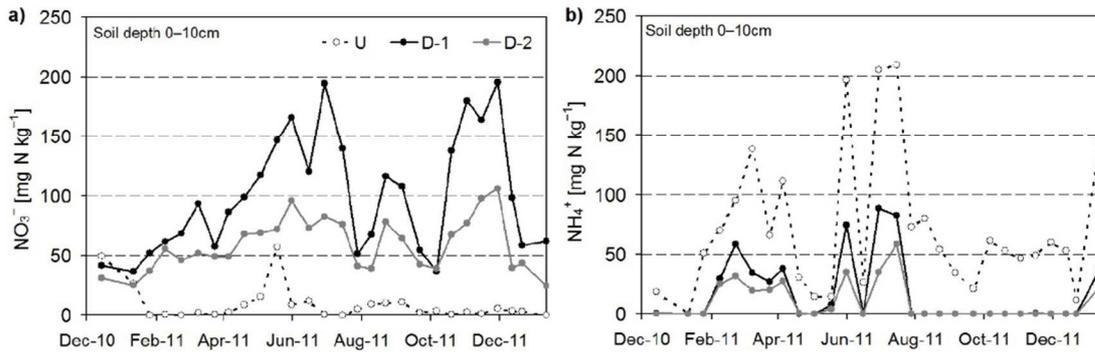
Figure 2. Annual variation in the extractable NO_3^- **(a)** and NH_4^+ **(b)** contents for the soil depth of 0–10 cm (U = undrained; D-1 = drained site 1; D-2 = drained site 2). Please note that the x axis starts on 1 December 2010.
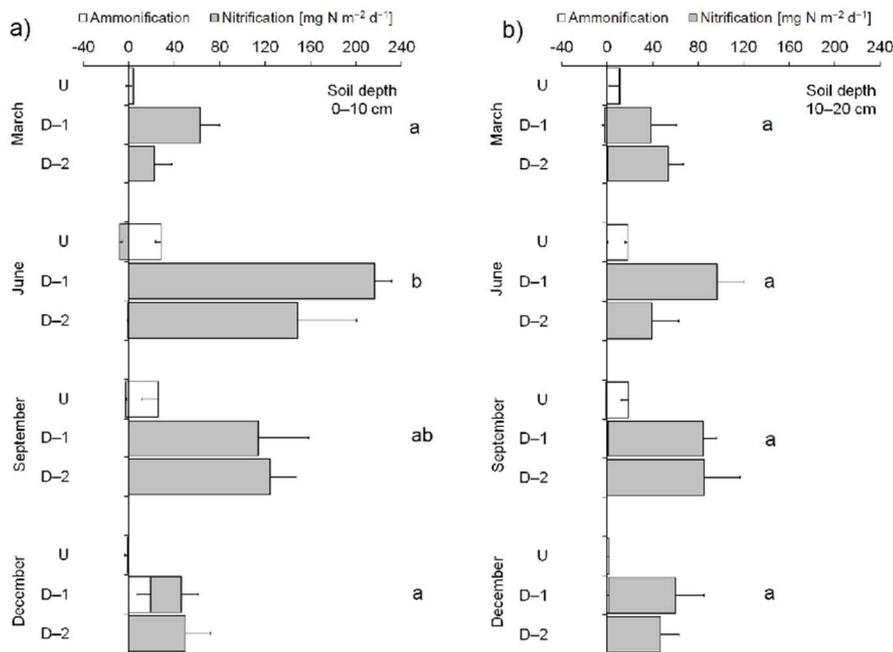

Figure 3. Net ammonification, net nitrification and net N mineralization rates (sum of net ammonification and net nitrification rate) for the soil depth of 0–10 cm **(a)** and 10–20 cm **(b)**. Means ($\pm$SE; $n = 3$) followed by the same letter indicates no significant differences in the net N mineralization rate between the months within one soil depth (two-way ANOVA, TukeyHSD test at $P \le 0.05$). Differences between both soil depths are not significant.

3.4 Effect of temperature and soil moisture on greenhouse and dinitrogen gas exchange under laboratory conditions

3.4.1 CO$_2$ exchange

CO_2 exchange indicates heterotrophic soil respiration in the incubation. No significant effect of soil moisture and site type on CO_2 exchange was detected (Fig. 7a), indicating comparable soil microbial activity in all incubated treatments. A statistically significant ($P < 0.0001$) quadratic effect of tem-

perature on CO_2 emissions was observed in all treatments with Q_{10} values between 1.82 and 17.80 mg C m^{-2} h^{-1} 10 °C^{-1}.

3.4.2 CH$_4$ exchange

No significant CH$_4$ emissions were observed in any treatment (Fig. 7b). Incubations at 100 % WFPS showed consistent near zero CH_4 fluxes or slight CH_4 uptake without significant effects of site type and temperature. At 70/83 % WFPS the uptake of CH_4 was significantly ($P < 0.0001$) higher at all temperature levels and for both sites compared
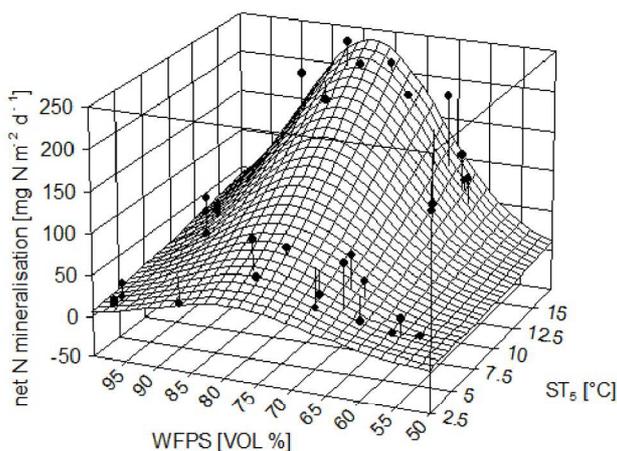
Figure 4. Relationship of net N mineralization in 0–10 cm soil depth (y) to mean water-filled pore space ($x1$) and mean soil temperature in 5 cm soil depth ($x2$) over 21 days of in situ incubation. Equation formula is $y = 304.28$ $(\pm 85.27) \cdot \exp(-0.5 \cdot ((((\text{WFPS} - 25.72\ (\pm 4.46)) /\ 11.47\ (\pm 1.14))^2) + (((\text{ST}_5 -\ 78.60\ (\pm 1.15)) /\ 11.47\ (\pm 1.14))^2)))$. ($n = 36$; adj$-R^2 = 0.75$; $P < 0.0001$).

Figure 6. Relationship of mean N_2O flux (y) to mean soil temperature in 2 cm soil depth ($x1$) and to mean groundwater level ($x2$). Equation formula is $y = -4.884\ (\pm 2.409) + 1.128\ (\pm 0.179) \cdot$ $\text{ST}_2 + (-0.322)\ (\pm 0.086) \cdot \text{GW} + 0.0004\ (\pm 0.000) \cdot \text{GW}^3$ ($n = 80$; adj$-R^2 = 0.37$; $P < 0.001$).

Figure 5. Mean ($\pm$SE, $n = 3$) N_2O fluxes of the undrained (U), drained 1 (D-1) and drained 2 (D-2) sites. Please note that the x axis starts on 1 December 2010.

For both WFPS levels, samples of the undrained site (U) only showed a weak, but significant (all $P < 0.03$) temperature response for the temperature range of 0 to 15 °C. The Q_{10} values of N_2O emissions for the undrained site are 2.07 µg N m^{-2} h^{-1} 10 °C^{-1} at 100 % WFPS for 0 to 15 °C and 1.51 µg N m^{-2} h^{-1} 10 °C^{-1} at 83 % WFPS for 0 to 25 °C.

For both sites a decline in nitrous oxide emissions at 25 °C and 100 % WFPS was observed. However, this pattern was also detected at 83 % WFPS for the undrained site. N_2O emissions observed at the undrained site U were on the same order of magnitude as the field measurements, whereas the drained site D-2 samples showed up to four times higher N_2O fluxes during the laboratory incubation than in the field at 70 % WFPS.

3.4.4 N₂ exchange

N_2 emissions were near zero at 70/83 % WFPS at low temperatures and never exceeded 0.4 mg N m^{-2} h^{-1} at high temperatures (Fig. 7d). N_2 emissions at 100 % WFPS were consistently at least 10 times higher than at 70/83 % WFPS ($P < 0.0001$). N_2 exchange also reacted to site type at 25 °C and to temperature. At the drained site, Q_{10} of N_2 emissions was 2.03 µg N m^{-2} h^{-1} 10 °C^{-1} at 70 % WFPS for the temperature range of 0 to 25 °C and 1.48 µg N m^{-2} h^{-1} 10 °C^{-1} at 100 % WFPS for the temperature range of 5 to 25 °C. At the undrained site, Q_{10} of N_2 emissions was 2.13 µg N m^{-2} h^{-1} 10 °C^{-1} at 83 % WFPS for the temperature range of 0 to 25 °C. N_2 emissions of the undrained site did not respond to temperature at 100 % WFPS. N_2 emissions increased most strongly from 15 to 25 °C in samples of

to 100 % WFPS. CH$_4$ uptake increased significantly with temperature only in incubated samples of the drained site ($P < 0.001$).

3.4.3 N₂O exchange

N_2O exchange differed in response to site type, WFPS and temperature. Significantly ($P < 0.0001$) higher N_2O fluxes at both WFPS levels were measured from samples of the drained site (D-2) compared to samples from the undrained site (U) (Fig. 7c). At the drained site, Q_{10} of N_2O emissions was 2.17 µg N m^{-2} h^{-1} 10 °C^{-1} at 70 % WFPS for the temperature range of 0 to 25 °C and 2.12 µg N m^{-2} h^{-1} 10 °C^{-1} at 100 % WFPS for 0 to 15 °C.

Figure 7. Mean ($\pm$SE, $n = 5$) of CO_2 (**a**), CH_4 (**b**), N_2O (**c**) and N_2 (**d**) exchange rates per water-filled pore space (vol. %; 70 % for D-2 and 83 % for U), temperature (°C) and site treatment. U = undrained; D-2 = drained 2. Please note the different scales for the N_2 exchange rates.

the drained site at 70 % and in particular 100 % WFPS and in samples of the undrained site at 83 % WFPS.

The high N_2 emissions observed at 100 % WFPS resulted in low N_2O/N_2 ratios, with mean values of 0.03 ± 0.01 (U) and 0.10 ± 0.02 (D-2). A similarly low N_2O/N_2 ratio (mean value of 0.12 ± 0.03) were found at 83 % WFPS for the samples from the U site, as a result of the low N_2O emissions. In contrast, the samples from the drained site D-2 showed a considerably higher N_2O/N_2 ratio, with a mean value of

0.62 ± 0.11 (note that in the case of zero N_2 fluxes no ratio was calculated). No relationship between temperature and N_2O/N_2 ratio was observed for either soil moisture contents.

4 Discussion

4.1 Nitrogen mineralization and transformation processes

For both drained sites, the NNM rates observed (518–653 kg N ha^{-1} yr^{-1}) are at the high end of NNM rates given in the literature. Janiesch (1978) and Janiesch et al. (1991) reported NNM rates of up to 500 kg ha^{-1} yr^{-1} (0–15 cm soil depth) and of up to 321.2 kg ha^{-1} yr^{-1} (0–10 cm soil depth) for different *Alnus glutinosa* forests from degraded fen peatlands in northern Germany. The significantly lower NNM rates from the undrained site are in the range of NNM rates reported for other seminatural black alder forests (Janiesch, 1978; Janiesch et al., 1991).

The compiled values from the literature and the results of the present study indicate an increasing NNM rate in black alder forests with increasing degradation of the fen peatland. In forest soils, high NNM rates are generally related to narrow C / N ratios (Gundersen et al., 1998; Ollinger et al., 2002), which are also characteristic for alder swamps (Mäkinen, 1979) and which were observed in the present study. Typically NNM rates for deciduous stands range between 50 to 150 kg N ha^{-1} yr^{-1} (Aber et al., 1989). The considerably higher NNM rates measured in the present study compared to those of other deciduous forests can partly be attributed to the high amount of easily decomposable organic nitrogen which becomes incorporated into the forest ecosystem through symbiotic atmospheric N$_2$ fixation by *Frankia* actinomycetes (Kätzel, 2003). Dittert (1992) estimated N$_2$ fixation rates of between 40 to 85 kg N ha^{-1} yr^{-1} for a black alder stand in northern Germany. Comparable and even higher values have also been reported for grey alder stands (e.g., Rytter et al., 1991; Lõhmus et al., 2002; Mander et al., 2005; Uri et al., 2011). Additionally, drainage of histosols supplies large amounts of N through mineralization of ancient organic matter (Klemedtsson et al., 2005). However, the distinct differences in the C and N contents of the drained sites did not result in significantly different NNM rates in the present study. Low retranslocation of nitrogen from senescing leaves to the alder tree (Mander et al., 1997; Lõhmus et al., 2002; Uri et al., 2011) causes high contents of N and narrow C / N ratios of the foliage, favoring the fast decomposition of the leaf litter (Struwe and Kjøller, 1990; Rytter et al., 1991; Lõhmus et al., 2002; Uri et al., 2011). The organic surface layer at the drained sites was classified as L-mull humus type, indicating the fast decomposition of the incorporated alder leaves.

Beside substrate chemistry, moisture content together with temperature are considered as the most important abiotic factors influencing biogeochemical transformation processes (Tietema et al., 1992). In general, increasing moisture contents stimulate the biogeochemical processes up to a threshold value where anaerobicity can limit microbial activity (Tietema et al., 1992). Temperature has a similar effect

on microbial activity, due to the fact that rates of enzymatic processes generally increase with increasing temperature (Michaelis and Menten, 1913). This finding becomes apparent in the Gaussian relationship observed between NNM rates in 0–10 cm soil depth, mean WFPS and mean temperature in −5 cm soil depth during the incubation period (Fig. 4). Significant effects of soil moisture and temperature on NNM or nitrification rates were also reported by other studies (e.g., Kowalenko and Cameron, 1976; Goodroad and Keeney, 1984; Gonçalves and Carlyle, 1994). The minor reduction in N turnover observed with increasing soil depth furthermore reflects the influence of aeration and temperature on mineralization processes. We attributed the significantly lower NNM rates at the undrained site to the limitation of microorganisms by anaerobicity due to the permanently high water levels. The oxygen deficit at the undrained site also explained why NNM was solely determined by net ammonification. Events of NO$_3^-$ presence and NO$_3^-$ consumption in the upper soil layer suggest that in periods of low water level net nitrification also occurs at the undrained site. In contrast, at the well-aerated drained sites, the NH$_4^+$ produced becomes immediately oxidized to NO$_3^-$, hence net nitrification entirely controls the NNM. However, using NNM rates as indicators of soil N dynamics bears the risk of considerably underestimating the intensity and complexity of the real N fluxes (Augustin, 2003); thus, further investigation should rather use the ^{15}N isotope dilution method (e.g., Davidson et al., 1991) for measuring gross rates of all relevant N turnover processes.

The permanently high mineral N contents detected in the probed soil depth probably indicate that the alder forest investigated has reached N saturation, due to the fact that the availability of N exceeds the biotic uptake capacity of the system (Aber et al., 1989). High contents of available NO$_3^-$ in the soil carry the risk of leaching or gaseous losses (Robertson, 1982). Several studies reported NO$_3^-$ losses by leaching from soils under alder stands (e.g., Binkley et al., 1992; Van Miegroet et al., 1992; Compton et al., 2003; Uri et al., 2011). The slightly higher NO$_3^-$ contents but lower NNM rates observed in the soil depth of 10–20 cm compared to the soil depth of 0–10 cm at the drained sites probably indicate leaching of NO$_3^-$ into deeper soil layers. Analysis of groundwater samples from the sites investigated showed nitrate contents of up to 36.2 mg L^{-1} and dissolved organic nitrogen (DON) contents of up to 2.8 mg L^{-1} (B. Tiemeyer, personal communication, 2013), confirming the earlier assumption of N leaching. Nitrate leaching into the anaerobic peat profile can result in enhanced denitrification (Regina et al., 1996), wherein the ratio of N$_2$O / N$_2$ depends on the residence time of N$_2$O on its way from the production site in the anaerobic subsoil to the atmosphere (van Groenigen et al., 2005).

4.2 Factors controlling temporal and site-specific variation in N_2O and N_2 fluxes

In the present study, field N_2O fluxes observed at the drained sites are very low. Different studies reported considerably higher values for N_2O emissions from drained black alder stands, varying from 5 to $75 \, \mathrm{kg \, N \, ha^{-1} \, yr^{-1}}$ (Mogge et al., 1998; Augustin, 1998; Brumme et al., 1999; Merbach et al., 2001; Hefting et al., 2003; von Arnold et al., 2005). According to Klemedtsson et al. (2005), NO_3^- availability is the main driver of N_2O emissions from drained histosols. However, despite distinct differences in the NO_3^- contents and NNM rates, significantly different N_2O emissions between the three sites were not found in the present study. This was in contrast to our hypothesis that accelerated N turnover and N mineralization leads to significantly increased amounts of N_2O emissions. We cannot rule out, however, that we may have missed high N_2O events on the drained sites due to our regular measurement intervals. In addition many wetland adapted trees such as alder can mediate N_2O fluxes from the soil to the atmosphere through the stem (Rusch and Rennenberg, 1998; Pangala et al., 2012; Machacova et al., 2013). This flux was not taken into account in the present study so that this study may underestimate the N_2O emissions from the alder ecosystem.

In the alder forest investigated, temperature in $-2 \, \mathrm{cm}$ soil depth and the GW level best explained field N_2O flux rates, whereas no relationship was found between field N_2O fluxes and the amount of NO_3^- in the soil. It is well known that temperature is a key variable affecting the emission rates of N_2O because both nitrification and denitrification are microbial processes (Firestone and Davidson, 1989; Maag and Vinther, 1996; Smith et al., 1998; Dobbie and Smith, 2001). Higher soil moisture content increases N_2O emissions through higher denitrification rates (Wolf and Russow, 2000). In our study, however, temperature and GW only explained 37 % of the spatiotemporal variation of mean N_2O flux rates. This demonstrates the complex interlinkage of the controlling factors for N_2O emissions, as reported for agricultural soils by Jungkunst et al. (2006). According to Couwenberg et al. (2011), mean annual GW levels below $-15 \, \mathrm{cm}$ show a wide range of N_2O emissions in fens; in contrast, Jungkunst et al. (2004) found highest annual N_2O emissions in a Norway spruce stand (black forest, southwest Germany) at a GW level of $-20 \, \mathrm{cm}$, whereas higher or lower GW levels decreased the emissions. However in the present study, differences observed in the GW levels were only small between the drained sites, resulting in comparable annual N_2O emissions. The observed cubic response to the GW level in our N_2O model reveals maximum N_2O emissions at a certain GW level. Perhaps it could be assumed that a mean GW level between the levels measured at the observed sites D-1 and U would result in enhanced N_2O emissions at the present black alder forest. Indeed, Brumme et al. (1999) reported that small seasonal fluctuations of the GW level seem to be more important than the mean GW level.

The overall results from the laboratory experiment lead to the conclusion that the incorrect setting of the WFPS to 83 % only marginally affected the responsiveness of samples from site U compared to 100 % WFPS treatments as well as to the samples from site D-2. Nevertheless, we cannot totally exclude the possibility that the incorrect adjustment lowered the gaseous N losses from these samples. However, in the laboratory experiment, significantly different N_2O emissions were not observed between the two adjusted soil moisture contents, but total denitrification was considerably higher at 100 % WFPS compared to 70/83 % WFPS for both sites. This was in line with investigations by Scholefield et al. (1997), who found a greater than 50-fold increase in denitrification activity with increasing WFPS from 70 % to 90 %. Beside soil moisture, the dependency of denitrification activity on temperature, as found for field N_2O fluxes, becomes even more apparent regarding the results from the incubation experiment, particularly for samples from site D-2. Additionally, our results reveal that the production of N_2 increased more with increasing temperature than the production of N_2O, which was in line with Maag and Vinther (1996). Higher denitrification activity with increasing temperature was also reported by several other studies (e.g., Dobbie and Smith, 2001; Schindelbacher et al., 2004; Schaufler et al., 2010). Butterbach-Bahl et al. (2013) reported that the Q_{10} of denitrification exceeds the Q_{10} of soil respiration and attributed this to a tight coupling between the microbial C an N cycles. Additionally, denitrification is indirectly affected by the temperature-induced respiratory depletion of soil oxygen concentration (Butterbach-Bahl et al., 2013).

The proportion of N_2O emissions at 100 % WFPS amounted to ~ 1 to 6 % and to ~ 0.7 to 2 % of total N release from denitrification for samples from site D-2 and U, indicating that in the present alder stand losses of N_2O are only of minor importance compared to N_2 losses at water-saturated conditions. This agrees with findings that, with increasing anoxic conditions, the percentage of N_2 increases until it becomes the major gas evolved (Rolston et al., 1978; Davidson, 1991). Indeed several studies reported that high contents of available NO_3^- result in the inhibition of N_2O reductase activity (Blackmer and Bremner, 1978; Firestone et al., 1979; Weier et al., 1993; Regina et al., 1996). However, for the present study it is not possible to state conclusively to what extend the probably higher NO_3^- availability inhibits the conversion of N_2O to N_2 at samples from site D-2. N_2O fluxes only declined in D-2 samples at water-saturated conditions and 25 °C with simultaneously exponentially increasing N_2 fluxes. While this mechanism supports our hypothesis that N_2O release is displaced by N_2 losses during periods of temporarily high water levels, the field observations never reached the point in which the decreasing $N_2O : N_2$ ratio overcompensated the increasing denitrification rate.

T. Eickenscheidt et al.: Nitrogen mineralization and gaseous nitrogen losses 2973

In contrast to the field observations, significant differences in the N_2O fluxes between the two sites investigated were found at both soil moisture contents in the incubation experiment conducted, mainly due to the distinct differences in the N_2O reaction to temperature. Our data have shown that denitrification was limited by temperature at the drained site, but limited by other factors at the undrained site, like the availability of labile organic carbon, available NO_3^- or soil moisture (Letey et al., 1980; Knowles, 1982; Weier et al., 1993; Klemedtsson et al., 2005). We can rule out that redox conditions are too anaerobic or microbial activity is less favorable for denitrification in samples from the undrained site in the incubation experiment because no CH_4 emissions were detected from water-saturated soil cores and aerobic and anaerobic CO_2 production was comparable and showed a distinct temperature response at both sites. However, at 100 % WFPS and 25 °C, the CO_2 to N gas emission ratio was 30 for site U and 13 for site D-2, indicating that anaerobic CO_2 production at site U was significantly more decoupled from denitrification activity than at the drained sites (Wang et al., 2011). Thus we attribute the small temperature response of N_2O and denitrification of the undrained site to NO_3^- limitation. The amount of NO_3^- in the solution from soil cores was not ascertained, but in the field net nitrification could never be detected at the undrained site. However, N_2O fluxes measured temporarily confirm our assumption that in periods of low water level net nitrification also occurs at the undrained site, albeit at a low level. Thus, we conclude that for most of the year denitrification activity is NO_3^- limited at the undrained site, confirming our hypothesis that NO_3^- limitation in permanently waterlogged peat soils results in negligible N_2O and N_2 losses.

For N_2 fluxes no comparable values from the field are available in the literature, but N_2 fluxes observed in the incubation experiment are in the range of other studies from drained and undrained fen ecosystems (e.g., Teiter and Mander, 2005; Wray and Bayley, 2007; Mander et al., 2008; Roobroeck et al., 2010; Soosaar et al., 2011).

In the present study less than 0.2 % of the nitrified N was emitted as N_2O–N at the drained sites. Similar or slightly higher values were reported by Klemedtsson et al. (1988) and Maag and Vinther (1996). In the incubation experiment N_2O fluxes that were up to four times higher were measured in samples for site D-2 compared to the field fluxes. This indicates the potential of high N_2O emissions from the drained alder sites under different soil physical conditions.

The low N_2O and N_2 losses, despite very high N mineralization rates, perhaps indicate that, for most of the year, aerobic or only slightly anaerobic soil conditions are predominant at the drained sites. Thus, a certain proportion of the gaseous N may have been lost as nitric oxide (NO), a by-product of nitrification and an intermediate product of denitrification (Davidson, 1991). Several studies reported that net nitrification rates and NO_3^- in the soil solution are positively correlated to NO fluxes in well-aerated soils (Skiba et al., 1998, 1999; Gasche and Papen, 1999; Venterea et al., 2003; Eickenscheidt and Brumme, 2012).

5 Conclusions

We studied N turnover processes in organic soils along a water table gradient in a nitrogen-saturated alder forest characterized by high mineral N concentrations. Drainage considerably increased net N mineralization and N turnover processes and shifted the dominant process from net ammonification to net nitrification, resulting in nitrate leaching. Surprisingly, N_2O emissions in the field remained low at the drained sites, but we cannot rule out that we have missed N_2O emission peaks. The incubation experiment supported the potential for high N_2O emissions at the drained alder sites. Temporarily water-saturated conditions enhanced N_2 emissions, whereas N_2O emissions remained unchanged. In contrast, the permanently high groundwater level at the undrained site prevented net nitrification and resulted in negligible gaseous N losses.

References

Aber, J. D., Nadelhoffer, K. J., Steudler, P., and Melillo, J. M.: Nitrogen saturation in northern forest ecosystems – Excess nitrogen from fossil fuel combustion may stress the biosphere, Bioscience, 39, 378–386, 1989.

Adams, M. A., Polglase, P. J., Attiwill, P. M., and Weston, C. J.: In situ studies of nitrogen mineralization and uptake in forest soils; some comments on methodology, Soil Biol. Biochem., 21, 423–429, 1989.

Augustin, J.: Erlenstandorte als Quelle und Senke klimarelevater Spurengase. (Alder forests as a source and sink of Greenhouse gases), in: Die Schwarz-Erle (*Alnus glutinosa* [L.] GAERTN.) im nordostdeutschen Tiefland, Eberswalder Forstliche Schriftenreihe, 17, 46–59, 2003.

Augustin, J., Merbach, W., and Rogasik, J.: Factors influencing nitrous oxide and methane emissions from minerotrophic fens in northeast Germany, Biol. Fert. Soils, 28, 1–4, 1998.

Binkley, D., Sollins, P., Bell, R., Sachs, D., and Myrold, D.: Biogeo-chemistry of adjacent conifer and alder-conifer stands, Ecology, 73, 2022–2033, 1992.

Blackmer, A. M. and Bremner, J. M.: Inhibitory effect of nitrate on reduction of N_2O to N_2 by soil microorganisms, Soil Biol. Biochem., 10, 187–191, 1978.

Bremner, J. M. and Blackmer, A. M.: Terrestrial nitrification as a source of atmospheric nitrous oxide, in: Denitrification, Nitrification and Nitrous Oxide, edited by: Delwiche, P. P., 151–170, Wiley, New York. 1981.

Brumme, R., Borken, W., and Finke, S.: Hierarchical control on nitrous oxide emission in forest ecosystems, Global Biogeochem. Cy., 13, 1137–1148, 1999.

Butterbach-Bahl, K., Willibald, G., and Papen, H.: Soil core method for direct simultaneous determination of N_2 and N_2O emissions from forest soils, Plant Soil, 240, 105–116, 2002.

Butterbach-Bahl, K., Baggs, E. M., Dannenmann, M., Kiese, R., and Zechmeister-Boltenstern, S.: Nitrous oxide emissions from soils: how well do we understand the processes and their controls?, P. T. R. Soc. B, 368, 20130122, doi:10.1098/rstb.2013.0122, 2013.

Compton, J. E., Church, M. R., Larned, S. T., and Hogsett, W. E.: Nitrogen export from forested watershed in the Oregon Coast Range: The role of N_2-fixing red alder, Ecosystems, 6, 773–785, 2003.

Couwenberg, J., Thiele, A., Tannenberger, F., Augustin, J., Bärisch, S., Dubovik, D., Liashchynskaya, N., Michaelis, D., Minke, M., Skuratovich, A., and Joosten, H.: Assessing greenhouse gas emissions from peatlands using vegetation as a proxy, Hydrobiologia, 674, 67–89, 2011.

Crawley, M. J.: The R Book, John Wiley and Sons Ltd, Chichester, 942 pp., 2007.

Crutzen, P. J.: the role of NO and NO_2 in the chemistry of the troposphere and stratosphere, Annu. Rev. Earth Pl. Sc., 7, 443–472, 1979.

Davidson, E. A.: Fluxes of nitrous and nitric oxide from terrestrial ecosystems. in: Microbial Production and Consumption of Greenhouse Gases: Methane, Nitrogen Oxides and Halomethanes, edited by: Rogers, J. E. and Whitman, W. P., American Society for Microbiology, Washington, D.C., 219–235, 1991.

Davidson, E. A., Swank, W. T., and Perry, T. O.: Distinguishing between nitrification ad denitrification as sources of gaseous nitrogen production in soil, Appl. Environ. Microb., 52, 1280–1286, 1986.

Del Grosso, S. J., Parton, W. J., Mosier, A. R., Ojima, D. S., Kulmala, A. E., and Phongpan, S.: General model for N_2O and N_2 gas emissions from soils due to denitrification, Global Biogeochem. Cy., 14, 1045–1060. 2000.

Dilly, O., Blume, H. P., Kappen, L., Kutsch, W. L., Middelhoff, U., Wötzel, J., Buscot, F., Dittert, K., Bach, H. J., Mogge, B., Pritsch, K., and Munch, J. C.: Microbial processes and features of the microbiota in histosols from a black alder forest, Geomicrobiol. J., 16, 65–78, 1999.

Dittert, K.: Die stickstofffixierende Schwarzerle- Frankia-Symbiose in einem Erlenbruch der Bornhöveder Seenkette, EcoSys. Suppl., 5, 1–98, 1992.

Dobbie, K. E. and Smith, K. A.: The effects of temperature, water-filled pore space and land use on N_2O emissions from an imperfectly drained gleysol, Eur. J. Soil Sci., 52, 667–673, 2001.

Drösler, M.: Trace gas exchange and climatic relevance of bog ecosystems, Southern Germany, PhD thesis, Chair of Vegetation Ecology, Department of Ecology, Technical University Munich, 179 pp., 2005.

Eickenscheidt, N. and Brumme, R.: NO_X and N_2O fluxes in a nitrogen-enriched European spruce forest soil under experimental long-term reduction of nitrogen depositions, Atmos. Environ., 60, 51–58, 2012.

Eickenscheidt, N., Brumme, R., and Veldkamp, E.: Direct contribution of nitrogen deposition to nitrous oxide emissions in a temperate beech and spruce forest – a ^{15}N tracer study, Biogeosciences, 8, 621–635, doi:10.5194/bg-8-621-2011, 2011.

Firestone, M. K. and Davidson, E. A.: Microbial basis of NO and N_2O production and consumption in soil, in: Exchange of trace gases between terrestrial ecosystems and the atmosphere, edited by: Andreae, M. O. and Schimel, D. S., John Wiley and Sons Ltd., Chichester, U.K., 7–21, 1989.

Firestone, M. K., Smith, M. S., Firestone, R. B., and Tiedje, J. M.: The influence of nitrate, nitrite, and oxygen on the composition of the gaseous products of denitrification in soil, Soil Sci. Soc. Am. J., 43, 1140–1144, 1979.

Gasche, R. and Papen, H.: A 3-year continuous record of nitrogen trace gas fluxes from untreated and limed soil of a N-saturated spruce and beech forest ecosystem in Germany – 2. NO and NO_2 fluxes, J. Geophys. Res.-Atmos., 104, 18505–18520, 1999.

Gonçalves, J. L. M. and Carlyle, J. C.: Modelling the influence of moisture and temperature on net nitrogen mineralization in a forested sandy soil, Soil Biol. Biochem., 26, 1557–1564, 1994.

Goodroad, L. L. and Keeney, D. R.: Nitrous oxide production in aerobic soils under varying pH, temperature and water content, Soil Biol. Biochem., 16, 39–43, 1984.

Gundersen, P., Emmett, B. A., Kjønaas, O. J., Koopmans, C. J., and Tietema, A.: Impact of nitrogen deposition on nitrogen cycling: a synthesis of NITREX-data, Forest Ecol. Manag., 101, 37–55, 1998.

Hahn-Schöfl, M., Zak, D., Minke, M., Gelbrecht, J., Augustin, J., and Freibauer, A.: Organic sediment formed during inundation of a degraded fen grassland emits large fluxes of CH_4 and CO_2, Biogeosciences, 8, 1539–1550, doi:10.5194/bg-8-1539-2011, 2011.

Hefting, M. M., Bobbink, R., and Caluwe, H.: Nitrous oxide emission and denitrification in chronically nitrate-loaded riparian buffer zones, J. Environ. Qual., 32, 1194–1203, 2003.

Hothorn, T., Bretz, F., Westfall, P., Heiberger, R. M., and Schuetzenmeister, A.: Simultaneous Inference in General Parametric Models: R package version 1.2-17, http://cran.rproject.org/web/packages/multcomp/index.html, 2013.

IPCC: Chapter 11: N_2O Emissions from managed soils, and CO_2 emissions from lime and urea application, in: IPCC guidelines for national greenhouse gas inventories, edited by: Eggleston, H. S., Buendia, L., Miwa, K., Ngara, T., and Tanabe, K., IGES, Hayama, Japan, 2006.

IPCC: Changes in atmospheric constituents and in radiative forcing, in: Climate Change 2007: The physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, edited by:

Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K. B., Tignor, M., and Miller, H. L., Cambridge University Press, Cambridge, UK and New York, USA, 2007.

Janiesch, P., Mellin, C., and Müller, E.: Die Stickstoff-Netto-Mineralisierung in naturnahen und degradierten Erlenbruchwäldern als Kenngröße zur Beurteilung des ökologischen Zustandes, Poster zu Verhandlungen der Gesellschaft für Ökologie (Freising-Weihenstephan 1990), Band 20, 353–359, 1991.

Jungkunst, H. F., Fiedler, S., and Stahr, K.: N_2O emissions of a mature Norway spruce (Picea abies) stand in the Black Forest (southwest Germany) as differentiated by soil pattern, J. Geophys. Res., 108, D07302, doi:10.1029/2003JD004344, 2004.

Junkunst, H. F., Freibauer, A., Neufeldt, H., and Bareth, G.: Nitrous oxide emissions from agricultural land use in Germany – a synthesis of available annual field data, J. Plant Nutr. Soil Sc., 169, 341–351, 2006.

Kätzel, R.: Zum physiologischen Anpassungspotenzial der Schwarz–Erle, In: Die Schwarz-Erle (*Alnus glutinosa* [L.] GAERTN.) im nordostdeutschen Tiefland, Eberswalder Forstliche Schriftenreihe, 17, 39–45, 2003.

Klemedtsson, L., Svensson, B. H., and Rosswall, T.: A method of selective inhibition to distinguish between nitrification and denitrification as sources of nitrous oxide in soil, Biol. Fert. Soils, 6, 112–119, 1988.

Klemedtsson, L., von Arnold, K., Weslien, P., and Gundersen, P.: Soil CN ratio as a scalar parameter to predict nitrous oxide emissions, Glob. Change Biol., 11, 1142–1147, 2005.

Knowles, R.: Denitrification, Microbiol. Rev., 46, 43–70, 1982.

Kowalenko, C. G. and Cameron, D. R.: Nitrogen transformations in an incubated soil as affected by combinations of moisture content and temperature and adsorption – fixation of ammonium, Can. J. Soil Sci., 56, 63–70, 1976.

Letey, J., Valoras, N., Hadas, A., and Focht, D. D.: Effect of air-filled porosity, nitrate concentration, and time on the ratio of N_2O/N_2 evolution during denitrification, J. Environ. Qual., 9, 227–231, 1980.

Livingston, G. P. and Hutchinson, G. L.: Enclosure-based measurement of trace gas exchange: application and sources of error, in: Biogenic Trace Gases: Measuring Emissions from Soil and Water, edited by: Matson, P. A. and Harriss, R. C., Blackwell Science, Cambridge, 14–50, 1995.

Lõhmus, K., Kuusements, V., Ivask, M., Teiter, S., Augustin, J., and Mander, Ü.: Budgets of nitrogen fluxes in riparian grey alder forests, Arch. Hydrobiol. Suppl., 141, 321–332, 2002.

Maag, V. and Vinther, F. P.: Nitrous oxide emission by nitrification and denitrification in different soil types and at different soil moisture contents and temperatures, Appl. Soil Ecol., 4, 5–14, 1996.

Machacova, K., Papen, H., Kreuzwieser, J., and Rennenberg, H.: Inundation strongly stimulates nitrous oxide emissions from stems of the upland tree *Fagus sylvatica* and the riparian tree *Alnus glutinosa*, Plant Soil, 364, 287–301, doi:10.1007/s11104-012-1359-4, 2012.

Mäkinen, A.: Peat quality and peat formation in Finnish alder swamps, in: Classification of Peat and Peatlands, edited by: Kivinen, E., Heikurainen, L., and Pakarinen, P., International Peat Society, Helsinki, 171–183, 1979.

Mander, Ü., Kuusemets, V., Lõhmus, K., and Mauring, T.: Efficiency and dimensioning of riparian buffer zones in agricultural catchments, Ecol. Eng., 8, 299–324, 1997.

Mander, Ü., Kuusemets, V., Lõhmus, K., Mauring, T., Teiter, S., and Augustin, J.: Nitrous oxide, dinitrogen, and methane emission in a subsurface flow constructed wetland, Water Sci. Technol., 4, 135–142, 2003.

Mander, Ü., Lõhmus, K., Kuusemets, V., Ivask, M., Teiter, S., and Augustin, J.: Budgets of nitrogen and phosphorus fluxes in riparian grey alder forests, Natural and Constructed Wetlands: Nutrients, Metals and Management, 1–19, 2005.

Mander, Ü., Lõhmus, K., Teiter, S., Mauring, T., Nurk, K., and Augustin, J.: Gaseous fluxes in the nitrogen and carbon budgets of subsurface flow constructed wetlands, Sci. Total Environ, 404, 343–353, 2008.

Merbach, W., Augustin, J., and Gans, W.: Nitrous oxide emissions from fen mires in dependence of anthropogenic activities, J. Appl. Bot., 75, 118–123, 2001.

Michaelis, L. and Menten, M. L.: Die Kinetik der Invertinwirkung, Biochem. Z., 49, 333–369, 1913.

Mogge, B., Kaiser, E. A., and Munch, J. C.: Nitrous oxide emissions and denitrification N-losses from forest soils in the Bornhöved lake region (northern Germany), Soil Biol. Biochem., 30, 703–710, 1998.

Ollinger, S. V., Smith, M. L., Martin, M. E, Hallett, R. A., Goodale, C. L., and Aber, J. D.: Regional variation in foliar chemistry and N cycling among forests of diverse history and composition, Ecology, 83, 339–355, 2002.

Pangala, S. R., Moore, S., Hornibrook, E. R. C., and Gauci, V.: Trees are major conduits for methane egress from tropical forested wetlands, New Phytol., 197, 524–531, 2012.

R Development Core Team: R: A language and environment for statistical computing, R Foundation for Statistical Computing, Vienna, Austria, ISBN 3-900051-07-0, available at: http://www.R-project.org, 2010.

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

Robertson, G. P.: Nitrification in forested ecosystems, Philos. T. R. Soc. Lond., 296, 445–457, 1982.

Rolston, D. E., Hoffman, D. L., and Toy, D. W.: Field measurements of denitrification: 1. Flux of N_2 and N_2O, Soil Sci. Soc. Am. J., 42, 863–869, 1978.

Roobroeck, D., Butterbach-Bahl, K., Brüggemann, N., and Boeckx, P.: Dinitrogen and nitrous oxide exchanges from an undrained monolith fen: short-term responses following nitrate addition, Eur. J. Soil Sci., 61, 662–670, 2010.

Rusch, H. and Rennenberg, H.,: Black alder (*Alnus glutinosa* (L.) Gaertn.) trees mediate methane and nitrous oxide emission from the soil to the atmosphere, Plant Soil, 201, 1–7, 1998.

Rytter, L., Arveby, A. S., and Granhall, U.: Dinitrogen (C_2H_2) fixation in relation to nitrogen fertilization of grey alder [Alnus incana (L.) Moench.] plantations in a peat bog, Biol. Fert. Soils, 10, 233–240, 1991.

Schäfer, A. and Joosten, H. (Eds.): Erlenaufforstung auf wiedervernässten Niedermooren. (Alnus afforestation on restored fen peatlands), Greifswald: Institute for Sustainable Development of Landscapes of the Earth, 68 pp. 2005.

Schaufler, G., Kitzler, A., Schindlbacher, A., Skiba, U., Sutton, M. A., and Zechmeister-Boltenstern, S.: Greenhouse gas emissions from European soils under different land use: effects of soil moisture and temperature, Eur. J. Soil Sci., 61, 683–696, 2010.

Schindlbacher, A., Zechmeister-Boltenstern, S., and Butterbach-Bahl, K.: Effects of soil moisture and temperature on NO, NO_2, and N_2O emissions from European forest soils, J. Geophys. Res., 109, D17302, doi:10.1029/2004JD004590, 2004.

Scholefield, D., Hawkins, J. M. B., and Jackson, S. M.: Use of a flowing helium atmosphere incubation technique to measure the effects of denitrification controls applied to intact cores of a clay soil, Soil Biol. Biochem., 29, 1337–1344, 1997.

Skiba, U., Sheppard, L., Pitcairn, C. E. R., Leith, I., Crossley, A., van Dijk, S., Kennedy, V. H., and Fowler, D.: Soil nitrous oxide and nitric oxide emissions as indicators of elevated atmospheric N deposition rates in seminatural ecosystems, Environ. Pollut., 102, 457–461, 1998.

Skiba, U., Sheppard, L. J., Pitcairn, C. E. R., van Dijk, S., and Rossall, M.: The effect of N deposition on nitrous oxide and nitric oxide emissions from temperate forest soils, Water Air Soil Poll., 116, 89–98, 1999.

Smith, K. A., Thomson, P. E., Clayton, H., McTaggart, I. P., and Conen, F.: Effects of temperature, water content and nitrogen fertilisation on emissions of nitrous oxide by soil, Atmos. Environ., 32, 3301–3309, 1998.

Soosaar, K., Mander, Ü., Maddison, M., Kanal, A., Kull, A., Lõhmus, K., Truu, J., and Augustin, J.: Dynamics of gaseous nitrogen and carbon fluxes in riparian alder forests, Ecol. Eng., 37, 40–53, 2011.

Speir, T. W., Kettles, H. A., and More, R. D.: Aerobic emissions of N_2O and N_2 from soil cores: factors influencing production from ^{13}N-labelled NO_3^- and NH_4^+, Soil Biol. Biochem., 27, 1299–1306, 1995.

Struwe, S. and Kjøller, A.: Seasonality of denitrification in waterlogged alder stands, Plant Soil, 128, 109–113, 1990.

Teiter, S. and Mander, Ü.: Emission of N_2O, N_2, CH_4 and CO_2 from constructed wetlands for wastewater treatment and from riparian buffer zones, Ecol. Eng., 25, 528–541, 2005.

Tietema, A., Warmerdam, B., Lenting, E., and Riemer, L.: Abiotic factors regulating nitrogen transformations in the organic layer of acid forest soils: Moisture and pH, Plant Soil, 147, 69–78, 1992.

Uri, V., Lõhmus, K., and Tullus, H.: Annual net nitrogen mineralization in a grey alder (*Alnus incana* (L.) moench) plantation on abandoned agricultural land, Forest Ecol. Manag., 184, 167–176, 2003.

Uri, V., Lõhmus, K., Mander, Ü., Ostonen, I., Aosaar, J., Maddison, M., Helmisaari, H. S., and Augustin, J.: Long-term effects on the nitrogen budget of a short-rotation grey alder (*Alnus incana* (L.) Moench) forest on abandoned agricultural land, Ecol. Eng., 37, 920–930, doi:10.1016/j.ecoleng.2011.01.016, 2011.

van Groenigen, J. W., Zwart, K. B., Harris, D., and van Kessel, C.: Vertical gradients of $\delta^{15}N$ and $\delta^{18}O$ in soil atmospheric N_2O – temporal dynamics in a sandy soil, Rapid Commun. Mass Sp., 19, 1289–1295, 2005.

van Miegroet, H., Homann, P. S., and Cole, D. W.: Soil nitrogen dynamics following harvesting and conversion of red alder and Douglas fir stands, Soil Sci. Soc. Am. J., 56, 1311–1318, 1992.

VDLUFA: Bestimmung von mineralischem (Nitrat-)Stickstoff in Bodenprofilen (Nmin- Labormethode), in: Methodenbuch Teil 2, edited by: VDLUFA, Speyer, Germany, 1997.

Venterea, R. T., Groffman, P. M., Verchot, L. V., Magill, A. H., Aber, J. D., and Steudler, P. A.: Nitrogen oxide gas emissions from temperate forest soils receiving long-term nitrogen inputs, Glob. Change Biol., 9, 346–357, 2003.

von Arnold, K., Nilsson, M., Hånell, B., Weslien, P., and Klemedtsson, L.: Fluxes of CO_2, CH_4 and N_2O from drained organic soils in deciduous forests, Soil Biol. Biochem., 37, 1059–1071, 2005.

Wang, R., Willibald, G., Feng, Q., Zheng, X., Liao, T., Brüggemann, N., and Butterbach-Bahl, K.: Measurement of N_2, N_2O, NO and CO_2 Emissions from Soil with the Gas-Flow-Soil-Core Technique, Environ. Sci. Technol., 45, 6066–6072, 2011.

Watts, S. and Seitzinger, S. P.: Denitrification rates in organic and mineral soils from riparian sites: a comparison of N_2 flux and acetylene inhibition methods, Soil Biol. Biochem., 32, 1383–1392, 2000.

Weier, K. L., Doran, J. W., Power, J. F., and Walters, D. T.: Denitrification and dinitrogen/nitrous oxide ratio as affected by soil water, available carbon, and nitrate, Soil Sci. Soc. Am. J., 57, 66–72, 1993.

Wichtmann, W. and Joosten, H.: Paludiculture: peat formation and renewable resources from rewetted peatlands, in: IMCG-Newsletter 3/2007, 24–28, 2007.

Wolf, I. and Russow, R.: Different pathways of formation of N_2O, N_2 and NO in black earth soil, Soil Biol. Biochem., 32, 229–239, 2000.

Wray, H. E. and Bayley, S. E.: Denitrification rates in marsh frings and fens in two boreal peatlands in Alberta, Canada, Wetlands, 27, 1036–1045, 2007.

WRB, 2006 – IUSS Working Group: World Reference Base for Soil Resources 2006, 2nd Edn., World Soil Resources Reports No. 103, Rome, 2006

7.4 Calculation approach to estimate net GHG fluxes of the alder forest

NEE estimation of the alder forest based on two different calculation approaches. The first calculation method based on the approach of von Arnold et al. (2005), who used dark forest floor CO_2 fluxes from static chamber measurements, which were also determined simultaneously to N_2O fluxes in the present study (unpublished data). It must be pointed out that several studies (e.g. Venterea, 2010) have shown that the static chamber measurement method significantly underestimate CO_2 exchange of ecosystems. However, despite of high uncertainty the site-specific mean CO_2 flux rates observed for U, D–1 and D–2 amounted to 56 ± 10 mg C m^{-2} h^{-1}, 81 ± 14 mg C m^{-2} h^{-1} and 76 ± 13 mg C m^{-2} h^{-1}, respectively. In the present study, referred CO_2 fluxes represent the sum of heterotrophic respiration (R_h), autotrophic respiration (R_a) from forest floor vegetation and autotrophic respiration from tree roots. Based on literature values, von Arnold (2005) assumed that 50% of the measured soil CO_2 released originated from root-based activity. Thus, in average 245 g C m^{-2} yr^{-1} and approximately 344 g C m^{-2} yr^{-1} are presumed to be mineralized at the undrained and drained site, respectively. Based on the conducted incubation experiment, collected annual leaf litter fall (unpublished data) and mineralization rates derived from a litterbag study, which lasted 374 days (unpublished data), a second calculation was accomplished to roughly estimate heterotrophic CO_2 release. There, R_h derived from the incubation experiment was determined assuming an annual average temperature of 10°C and a mean annual WFPS of 100% for the undrained site and 70% for the drained sites. According to these assumptions, estimated CO_2 flux rates amounted to approximately 15 mg C m^{-2} h^{-1} and 20 mg C m^{-2} h^{-1} for the undrained and drained site, respectively. These values are distinctly lower than values determined from forest floor measurements, even after considering R_a. However, it has to be kept in mind that mineralization rates from soil cores are solely representative for the soil depth 9–15 cm. From monitored leaf litter fall (4593 ± 764 kg DM ha^{-1} yr^{-1}) and the litterbag study, it was found that litter decay rates observed at the drained sites of the alder forest approximately doubled decay rates from the undrained sites (11.54 vs 6.88 kg DM ha^{-1} d^{-1}). This result is in line with Bragazza et al. (2009), who reported that the percentage of mass losses from vascular plant litter and *Sphagnum* litter was on average 2.3 times higher in oxic than anoxic layers. The average C content of collected leaf litter was 47%, leading to a mineralization rate of

approximately 118 g C m^{-2} yr^{-1} at the undrained site and 198 g C m^{-2} yr^{-1} at the drained sites. Taking together both, R_h determined from soil cores and leaf litter decay, annual mineralization rates amount to approximately 249 g C m^{-2} yr^{-1} and 373 g C m^{-2} yr^{-1} for the undrained and drained sites, respectively, which fits well with mineralization rates derived from forest floor release. Estimated net primary productivity (NPP) of the alder forest was -260.7 g C m^{-2} yr^{-1} and originated from tree biomass data and simulations of forest growth by the models BALANCE (Grote and Pretzsch, 2002) and SILVA (Pretzsch et al., 2002) (S. Röhling and T. Rötzer personal communication, 2013). Carbon accumulation from forest floor vegetation has not been considered in NPP calculation. However, the estimated NECB of the alder forest revealed a small C uptake ranging between -6.8 and -2.8 g C m^{-2} yr^{-1} at the undrained site and a small C release ranging between 83.1 and 112.1 g C m^{-2} yr^{-1} at the drained sites. For the drained sites the estimated values of C release seems quite low, considering that NNM rates revealed distinctly higher SOM turnover rates (e.g. NNM = 653 kg N ha^{-1} yr^{-1}; C/N ratio = 8.5 for the soil depth 0–20 cm; resulting in a C turnover of at least 556 g C m^{-2} yr^{-1}), but this can probably be attributed to the high uncertainty of estimated R_h. To assess the GWP$_{100}$ (reference period 100 years; IPCC, 2007) of the alder sites, net emissions of carbon equivalents of NECB and N_2O were summed, considering radiative forcing factors of 25 and 298 for the conversion of CH_4 and N_2O to CO_2 equivalents (Forster et al., 2007).

8 EIDESSTATTLICHE ERKLÄRUNG

Ich erkläre an Eides statt, dass ich die der Fakultät Wissenschaftszentrum Weihenstephan für Ernährung, Landnutzung und Umwelt der Technischen Universität München zur Promotionsprüfung vorgelegten Arbeit mit dem Titel

„Assessment and modeling of greenhouse gas emissions (CO_2, CH_4, N_2O) from different land use types and management practices on drained fen peatlands and associated organic soils – case study Freisinger Moos "

am Fachgebiet für Renaturierungsökologie unter der Anleitung und Betreuung durch Prof. Dr. Matthias Drösler ohne sonstige Hilfe erstellt und bei der Abfassung nur die gemäß § 6 Abs. 5 angegebenen Hilfsmittel benutzt habe.

Ich habe die Dissertation in keinem anderen Prüfungsverfahren als Prüfungsleistung vorgelegt.

Ich habe den angestrebten Doktorgrad noch nicht erworben und bin nicht in einem früheren Promotionsverfahren für den angestrebten Doktorgrad endgültig gescheitert.

Die Promotionsordnung der Technischen Universität München ist mir bekannt.

Freising, den _____   _____

                                    Tim Eickenscheidt

9 CURRICULUM VITAE

Tim Eickenscheidt
Born on 04.01.1982 in Leverkusen, Germany
Alte Poststr. 101
85356 Freising

EDUCATION

Since 03/2009	Ph.D. study, Chair of Restoration ecology, Faculty of Landscape Architecture and Landscape Planning, Research Department Ecology and Ecosystem Management, School of Life Sciences Weihenstephan, Technical University Munich.
2003–2008	Diplom-Ingenieur (FH) in Forestry, University of Applied Forest Sciences Rottenburg, Germany. Main focus: Geographic Information System and Landscape Management. Diploma thesis: "Quantifizierung der Kohlenstoff- und Humusvorräte der Encosta Atlântica im Waldforschungsgebiet Pró-Mata, Mata Atlântica, Rio Grande do Sul".
2002–2003	Civilian service, Protection Centre for Birds of Prey, Erftkreis.
2001–2002	School further education, Vocational College Märkischer Kreis, Iserlohn.
1998–2001	Vocational training forestry, National Forest Department, Arnsberg.
1992–1998	Comprehensive school, Fröndenberg.

RESEARCH EXPERIENCE

Since 09/2011	Scientist, Chair of Vegetation ecology, University of Applied Sciences Weihenstephan-Triesdorf.
03/2009–08/2011	Scientist, Chair of Restoration ecology, Research Department Ecology and Ecosystem Management, Technical University Munich.
02/2009	Scientific helper, Georg-August-University Göttingen; Department of Soil Science of Tropical and Subtropical Ecosystems.
03/2007–05/2007	Project thesis, Regional council Tübingen, Department of enviroment/nature conservation and University of Applied Forest Science Rottenburg, Project Titel: "Habitatsanalyse beim Weißstorch (*Ciconia ciconia*) im Donautal".
03/2006–09/2006	Data acquisition - Diploma thesis, Forest conservation and research area Pró-Mata, in Rio Grande do Sul, Brazil in cooperation with the Pontificia Universidade Católica do Rio Grande do Sul in Porto Alegre, Brazil and the Eberhard-Karls- University in Tübingen, Germany.
09/2004–11/2004	Internship, Swiss Federal Institut for Snow and Avalanche Research, Department of Alpine Environment in Davos/Schwitzerland.