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TRACE GAS EXCHANGE AND CLIMATIC RELEVANCE OF BOG ECOSYSTEMS, SOUTHERN GERMANY

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PREFACE – VORWORT

Die vorliegende Arbeit ist aus einem „kumulativen“ Ansatz entstanden. Der bearbeiteten Thematik wird auf internationaler Ebene in den letzten Jahren verstärkt Aufmerksamkeit geschenkt. Daher habe ich die einzelnen Kapitel meiner Arbeit in Form von eigenständigen Artikeln für Fachzeitschriften in englischer Sprache verfasst. Da aber die Thematik mit regionalem Bezug bearbeitet wurde und die Ergebnisse für die konzeptionelle und praktische Naturschutzarbeit von Bedeutung sind, habe ich die Einleitung und die Abschlussdiskussion auf deutsch geschrieben. In diesen Kapiteln sind die Konzeption und die wesentlichen Ergebnisse der Arbeit zusammenfassend dargestellt.

Am Entstehen dieser Arbeit waren viele Menschen beteiligt, denen ich allen ganz herzlich danken möchte. Zu Beginn möchte ich mich bei meinem Doktorvater Prof. Dr. Jörg Pfadenhauer sehr herzlich bedanken, der mir mit Weitsicht die notwendige Freiheit gelassen hat, um dieses Thema eigenständig zu entwickeln. Ein großer Dank geht auch an Dr. Jan Sliva, der maßgeblich zur Absicherung der Arbeit in ihrer Anfangsphase beigetragen hat, und an Dr. Ulrich Wild, der die Türen zur Gasanalytik an der GSF öffnete. Dr. Dieter Maas möchte ich für anregende Diskussionen zu Beginn der Arbeit ebenso danken wie Dr. Kathrin Kiehl und Dr. Heike Beismann. Dr. Stefan Glatzel sei herzlich gedankt für den Einblick in die kanadische Moorforschung. Dr. Thomas Grünwald und Dr. Georg Wohlfahrt danke ich für die Kooperation bei den Eddy-Vergleichen. Besonderer Dank gilt einer engagierten Gruppe an wissenschaftlichen Hilfskräften, ohne deren tatkräftigen Einsatz auch bei widrigsten Wetterbedingungen das umfassende Messprogramm nicht möglich gewesen wäre. Stellvertretend seien Thomas Probst, Andreas Kühn, Michael Heinl, Gerhard Overbeck, Matthias Urmetzer und Matthias Muth genannt. Bedanken möchte ich mich des Weiteren bei Ingrid Kapps für die Unterstützung bei der Wasseranalytik und bei logistischen und technischen Problemen. Bei Prof. Dr. Munch, Dr. Kamp und Herrn Schilling, Lehrstuhl für Bodenökologie der GSF in Neuherberg, möchte ich mich für den Zugang zum GC und die Unterstützung bei der Gasanalytik bedanken. Prof. Dr. Torben Christensen sei herzlich gedankt für seine hilfreichen Anregungen im Endstadium der Arbeit. Herrn Strohwasser vom Landratsamt Traunstein danke ich für die Unterstützung bei der Flächenauswahl und die Berücksichtigung der Forschungserfordernisse in laufenden Renaturierungsmaßnahmen im Untersuchungsgebiet. Herrn Eschenbeck, Bund-Naturschutz, Kreisgruppe Traunstein danke ich für die Kooperation bei der Einrichtung der Untersuchungsflächen auf BN-Grundstücken. Bei der Justizvollzugsanstalt Bernau bedanke ich mich für den Materialtransport beim Stegebau. Ein herzliches Dankeschön geht auch an Katie McKendrick und Joe Schwantes für die hilfreichen Englisch-Korrekturen.

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Einführung

1. Kohlenstoffakkumulation: Das Funktionsprinzip von Mooren

Das charakteristische Funktionsprinzip natürlicher Moore ist die Akkumulation von Kohlenstoff (Pfadenhauer, 1997, Succow & Joosten, 2001). Torflager werden aufgebaut, da der Abbau der organischen Substanz durch die anaeroben Bedingungen in der dauerhaft wassergesättigten Schicht (Katotelm) langsamer abläuft als die Biomasse-Produktion im Akrotelm, der durchlüfteten, oberflächennahen Wachstumsschicht (Göttlich, 1990). Als Maß für das langfristige Mittel der Kohlenstoffakkumulation über ein gesamtes Torfprofil wird die long–term rate of Carbon accumulation (LORCA) aus Profil-Datierung (C^{14}) und Bestimmung des Kohlenstoffgehaltes ermittelt (Tolonen & Turunen, 1996). Die Spannweite der Werte für (boreale) Regenmoore liegt zwischen 6,3 g C m^{-2} a^{-1} (Turunen & Turunen, 2003) und 31,4 g C m^{-2} a^{-1} (Botsch et al., 1995), wobei die 24 g C m^{-2} a^{-1}, die Turunen & Tolonen (1996) auf der Basis von mehreren Tausend Profilproben finnischer Moore ermittelt haben, meist als durchschnittliche Akkumulationsrate für Regenmoore angegeben wird. Modellierte Akkumulationsraten von Mooren prognostizieren einen Wachstumsstillstand, da die Abbauprozesse über die zunehmende Profiltiefe zu insgesamt immer höheren Kohlenstoffverlusten führen, die Biomasseproduktion an der Oberflächenschicht aber gleich bleibt (Clymo, 1990). Die ermittelte LORCA auch der ältesten Regenmoore (älter als 10.000 a) ergab aber immer noch durchschnittliche Akkumulationsraten von ca. 15 g C m^{-2} a^{-1} (Turunen & Tolonen, 1996). Damit sind die Moore die einzigen Ökosysteme, die langfristig und dauerhaft Kohlenstoff akkumulieren.

Die Moore der nördlichen Hemisphäre haben über diese dauerhaften Akkumulationsraten seit der letzten Eiszeit 200 bis 455 Pg (Gorham, 1991) bzw. nach aktualisierter Schätzung 270 bis 370 Pg (Turunen et al., 2002) Kohlenstoff akkumuliert. Dies entspricht ca. 20 % des gesamten Bodenkohlenstoffgehaltes der Erde, obwohl die Moore nur ca. 3 % der Oberfläche bedecken (v. Post et al., 1982). Des Weiteren entspricht der in Mooren gespeicherte Kohlenstoff der Hälfte des in der Atmosphäre vorhandenen CO_2-C (Houghton et al., 1990). Damit spielen die Moore global eine wichtige Rolle als Kohlenstoffspeicher.

Die zugrundeliegenden Prozesse für den Aufbau der organischen Substanz und die Kohlenstoffakkumulation sind (i) die Photosynthese (GPP) und (ii) die aerobe Ökosystematmung (R_{ECO}), deren Differenz der direkt messbare Netto-Ökosystem-Austausch (NEE) an CO_2 darstellt (Buchmann & Schulze, 1999). Daneben ist aber (iii) als Produkt der anaeroben Bodenatmung gerade in naturnahen Mooren die Methanemission für die Kohlenstoffbilanz von Bedeutung (Joabsson et al., 1999). Letztes Element der Kohlenstoff-Bilanzierung sind (iv) die flüssigen Verluste gelösten organischen (DOC) und anorganischen (DIC) Kohlenstoffs (Alm et al., 1999). Aus diesen Komponenten kann die Kohlenstoffbilanz z.B. als Flussschema berechnet werden (Francez & Vasander, 1995). Die flüssigen Verluste DOC/DIC sind bei komplexen Grundwasserflüssen nur unzureichend bestimmbar und werden daher i.d.R. aus der Literatur geschätzt, wie z.B. von Alm et al. (1999) auf der Basis von Sallantus & Kaipainen (1996) und Fraser et al. (2001) durchgeführt. Damit kann die Kohlenstoffbilanz über das NEE von CO_2 und die Methan-Bilanz, d.h. über die Erfassung der Spurengasflüsse von CO_2 und CH_4, berechnet werden. Die ermittelte C-Bilanz entspricht der LORCA unter den aktuellen Standort- und Klimabedingungen und spiegelt den aktuellen C-Haushalt eines Moorstandortes wider, der sensibel auf Standorteinflüsse wie Entwässerung oder Renaturierung reagiert. Damit ist die aktuelle über den Spurengasaustausch ermittelte C-Bilanz potenziell geeignet, als funktioneller Indikator für den Störungsrad eines degradierten Moorstandortes und für die stoffhaushaltliche Erfolgkontrolle von Renaturierungsmaßnahmen zu dienen. Ein erstes Ziel der vorliegenden Arbeit ist es, die Eignung dieses stoffhaushaltlichen Indikators zu klären.

2. Austausch von Spurengasen und Klimarelevanz von Mooren

Die erheblichen Kohlenstoffmengen von 270 - 370 Pg C (Turunen et al., 2002), die der Atmosphäre durch Moorwachstum seit der letzten Eiszeit entzogen wurden, bergen aber zugleich das Potenzial von verstärkenden Rückwirkungen auf die Klimaerwärmung, falls diese Kohlenstoffspeicher freigesetzt werden (Byrne et al., 2004). Gefährdet sind die Kohlenstoffspeicher primär durch eine Veränderung des Wasserhaushaltes der Moore. Ein Absinken des Wasserstandes führt zu einer Vergrößerung der durchlüfteten Schicht im Profil und damit zu beschleunigten aeroben Abbauprozessen, also höheren Raten der ökosystemaren Atmung (R_{ECO}). Diese Systematmung bestimmt maßgeblich den Netto-Ökosystem-Austausch (NEE), da dem Aufnahme-Pfad über die Photosynthese (GPP) enge physiologische Grenzen gesetzt sind (Reichstein et al., 2003). Ein Absinken des Wasserstandes kann als Folge von Trockenheit eintreten oder durch Dränagemaßnahmen bewusst herbeigeführt werden, um einen Moorstandort beispielsweise für den Torfabbau oder eine Aufforstung vorzubereiten. Alm et al. (1999) zeigten, dass ein ombrotrophes Moor in einem trockenen Jahr den gespeicherten Kohlenstoff der vorangegangenen fünf Jahre verlieren kann. Es stellt sich allerdings die Frage, inwieweit sich die erwarteten Klimaänderungen auf die Moore und deren Wasser- und Stoffhaushalt auswirken werden: Vier verschiedene Klimamodelle (HadCM3, CSIRO, CGCM2, NCAR) berechnen für Nordeuropa eine Zunahme von Temperatur und Niederschlag, während für Mitteleuropa sowohl Zu- als auch Abnahmen des Niederschlages bei steigender Temperatur vorhergesagt werden (Fischer et al., 2002). Für Süd-Bayern, das Fokusgebiet der vorgelegten Arbeit, wird bei steigender Temperatur mit steigenden Niederschlägen gerechnet (BayFORKLIM, 1999). Allerdings ist mit stärkeren Extremereignissen (z.B. Hitzeperioden und unregelmäßigerer Niederschlagsverteilung zu rechnen (ebd.). Ob sich aus diesen prognostizierten Klimaveränderungen eine Gefährdung oder sogar Stabilisierung der Regenmoore ableiten lässt, ist nicht eindeutig abschätzbar. Das Atlantikum als Hauptentstehungsphase der Moore mittlerer Breiten (Pfadenhauer 1997) war ebenfalls wärmer und feuchter als unser gegenwärtiges Klima.

Für die Einschätzung der derzeitigen Klimarelevanz der Moore in Bezug zur aktuellen Landnutzung ist die Ermittlung der Kohlenstoffbilanz aber nicht ausreichend (Byrne et al., 2004). Vielmehr sind die aktuellen Flüsse aller drei biogenen klimarelevanten Spurengase zu ermitteln und mit ihrem jeweiligen globalen Erwärmungspotenzial (GWP) zu verrechnen (IPCC 1996). Dazu gehört neben Kohlendioxid (CO_2) und Methan (CH_4), die die Basis für die über den Gashaushalt ermittelte Kohlenstoffbilanz bilden, auch Lachgas (N_2O). Auf europäischer Ebene sind allerdings die Spurengasflüsse von Mooren im Vergleich zu anderen Ökosystemen verhältnismäßig schlecht untersucht (Byrne et al., 2004). Es fehlt insbesondere an Messprogrammen, die in der Lage sind, Gesamtjahresbilanzen für alle drei klimarelevanten biogenen Spurengase CO_2, CH_4 und N_2O zu liefern (ebd.). Darüber hinaus haben die bisher durchgeführten Untersuchungen zu ombrotrophen Mooren ihren räumlichen Schwerpunkt in den borealen und subarktischen Gebieten Nordeuropas (s. z.B. Martikainen 1995, Wüthrich et al. 1999, Tuittila et al. 1999, Heikkinen et al. 2002, Christensen et al. 2004). Insbesondere besteht nach wie vor ein Mangel an Untersuchungen, wie sich die Renaturierung von Regenmoorökosystemen im Vergleich zur degradierten und naturnahen Situation der Standorte auf die Klimarelevanz auswirkt (Byrne et al., 2004). Es gibt erste Hinweise, dass eine Renaturierung die Kohlendioxid-Emissionen von degradierten Standorten reduziert, aber andererseits die Methan-Emissionen verstärkt (Tuittila, 2000, Marinier et al., 2004). Das Verhältnis der Flüsse ist abhängig von dem Zusammenspiel von Standort und Vegetation, und erst durch die Ermittlung von Jahresbilanzen und eine Verrechnung mit dem Klimaerwärmungspotenzial kann die potenzielle Klimaentlastung durch die Renaturierung nachgewiesen werden. In ersten Abschätzungen zeigten Höper & Blankenburg (2000) die derzeitige Klimarelevanz der Moore Niedersachsens auf und berechneten das Entlastungspotenzial durch Renaturierung. Eine Abschätzung der Renaturierungseffekte auf die Klimarelevanz der Grundwassermoore von Deutschland wurde von Meyer et al. (2001) vorgelegt. Allerdings wurden in beiden Untersuchungen naturnahe Standorte nicht in die Messungen miteinbezogen, und direkte Messungen des NEE konnten mangels entsprechender Messtechnik nicht durchgeführt werden. Darüber hinaus wurde der Berechnung des GWP ein Zeitraum von 500 Jahren zugrundegelegt, wohingegen für die nationale Berichterstattung zur Klimarahmenkonvention das GWP von 100 Jahren (nach IPCC, 1996) als Berechnungsbasis definiert wurde.

Es besteht daher Bedarf an aktuellen Untersuchungen zum Austausch klimarelevanter Spurengase und zur Klimarelevanz von Moorökosystemen, die sowohl vollständige Reihen von naturnahen über degradierte zu renaturierten Standorten vergleichend betrachten als auch den Netto-CO_2 Austausch mittels geeigneter Messtechnik direkt bilanzieren.

3. UNFCCC-Berichterstattung, Kyoto Protokoll und Biologische Senken

Die Bundesrepublik Deutschland hat das Kyoto-Protokoll ratifiziert, das am 15.02.05 in Kraft trat. Die beteiligten Länder sind verpflichtet, jährlich über ihren Ausstoß an klimarelevanten Spurengasen zu berichten und innerhalb der ersten Verpflichtungsperiode (2008-2012) die Reduktion des CO_2-Ausstoßes entsprechend der übernommenen Verpflichtung (Deutschland -21% gegenüber 1990) zu erfüllen. Die Berichterstattung über biogene Quellen ist freiwillig; aber falls das Konzept der „biologischen Senken“ als eine Maßnahme mit in den Katalog der Klimaschutzaktivitäten aufgenommen werden soll, ist Voraussetzung, dass über die biogenen Quellen und Senken Bericht erstattet wird (IPCC 2003). Die Basis für die Berichterstattung sind aber möglichst aktuelle Flächenschätzungen und Messungen des Spurengasaustausches der relevanten Ökosystemtypen (IPCC 2003).

Der Einsatz von biologischen Senken für den Klimaschutz wird kontrovers diskutiert. So kritisieren Missfeldt & Haites (2001) die Wirkung als negativen Anreiz für technische Lösungen der Emissionsreduktion. Ein systemimmanentes Problem ist das sogenannte „slow in fast out“-Phänomen, das besagt, dass der Kohlenstoff in biologischen Senken langsam aufgebaut wird, aber bei entsprechenden Einflüssen (z.B. Waldbrände) schnell wieder freigesetzt werden kann (Noble & Scholes, 2001). Des Weiteren wird die Schwierigkeit thematisiert, die Kohlenstoffzunahmen ausreichend sicher zu verifizieren (Noble & Scholes, 2001). Allerdings sind hierfür mit den neuen Good Practice Guidance des IPCC Verfahrensregeln aufgestellt worden (IPCC 2003). Für den Einsatz von biologischen Senken spricht andererseits, dass sie in der Regel schnell und zu einem günstigen Preis etabliert werden können (Smith 2004c), was insbesondere für Regenmoore gelten kann. Ihr größtes Potenzial liegt daher in der sogenannten „Brückenfunktion“, bis technisch aufwendige und teure Umstellungen des fossilen Energiesektors vollzogen werden können (Noble & Scholes, 2001). Nachdem jede potenzielle Klimaentlastungsmaßnahme so schnell wie möglich umgesetzt werden sollte, sind biologische Senken in allen Maßnahmenpaketen zum Klimaschutz mindestens für die nächsten 20-30 Jahre zu berücksichtigen (Smith 2004c).

Es ist daher ein übergeordnetes Ziel dieser Arbeit, zur fachlichen Klärung des Potenzials von Regenmoorrenaturierungen zur Klimaentlastung und für den Einsatz als biologische Senken beizutragen.

4. Bayerisches Klimaschutzprogramm und Bayerisches Moorentwicklungsprogramm

Der besondere Charme der Renaturierung von Regenmooren für den Klimaschutz, falls tatsächlich ein positiver Effekt nachzuweisen sein wird, ist die erwartete Synergie zwischen den Zielen des Artenschutzes, des Landschaftsschutzes und des Klimaschutzes. Darüber hinaus ist der land- und forstwirtschaftliche Nutzungsanspruch an Regenmoore derzeit sehr gering, was den Zugriff auf die Flächen erleichtern sollte.

Bayern hat ein regionales Klimaschutzprogramm aufgestellt, das landeseigene Ziele für die Reduktion des CO_2-Austoßes formuliert. Mit einem differenzierten Mix an Klimaschutzmaßnahmen soll eine Reduktion der CO_2-Emissionen auf 80 Mio. t bis 2010 erreicht werden (Bayerische Staatskanzlei, 2003). Ergänzend hierzu soll mittels eines regionalisierten Ansatzes abgeschätzt werden, welchen Beitrag zur Zielerreichung die Regenmoorrenaturierung leisten könnte.

Das jüngst fertiggestellte Moorentwicklungskonzept Bayern (= MEK, LFU 2003) formuliert als ein übergeordnetes Ziel die Erhaltung der Klimaschutzfunktion der Moore. Unabhängig davon, dass dieses Ziel ohne regionalisierte Untersuchungen über die Rolle der bayerischen Moore für den Klimaschutz aufgestellt worden ist, hat Bayern damit Stellung bezogen, sich für den Klimaschutz durch Moorschutz einzusetzen (Schnappauf, 2003). In der Ausformulierung des MEK wurde allerdings eine Prioritätenliste an zu schützenden und zu renaturierenden Mooren ausgearbeitet, die offensichtlich vorrangig an den Zielen des Artenschutzes orientiert ist. Hier soll die vorliegende Untersuchung in Bezug zu Regenmooren Empfehlungen liefern, wie die Moorrenaturierung für den Klimaschutz in das MEK eingebunden und die Synergien zwischen dem Klimaschutzprogramm und dem Moorschutzprogramm genutzt werden können.

5. Forschungsbedarf

Aus der Darstellung des derzeitigen Kenntnisstandes zur Thematik der Klimarelevanz von Mooren und der regionalen Untersuchungsdichte lässt sich der Forschungsbedarf in folgende Themen gliedern:

(1) Schließung von Kenntnislücken zum Austausch klimarelevanter Spurengase der Regenmoore, repräsentativ für die mitteleuropäischen gemäßigten Breiten nördlich des Alpenbogens (Erstuntersuchung)

(2) Parallele Ermittlung des Netto-Ökosystem-Austausches (NEE) aller drei biogenen klimarelevanten Spurengase (CO_2, CH_4, N_2O) mit entsprechender Messtechnik

(3) Ermittlung von vollständigen Jahresbilanzen für die Flüsse aller drei klimarelevanter Spurengase (CO_2, CH_4, N_2O) zur Berechnung der Klimarelevanz der jeweiligen Standorte

(4) Vergleich der Flüsse entlang eines repräsentativen Gradienten von naturnahen über renaturierte zu degradierten Standorten

(5) Abschätzung des Potenzials von Regenmoorrenaturierung zur Klimaentlastung unter dem bayerischen Klimaschutzprogramm und einer potenziellen Anrechenbarkeit unter dem Kyoto-Protokoll

6. Ziele der Arbeit

Ziel dieser Arbeit ist die Untersuchung des Austauschverhaltens der drei klimarelevanten Spurengase CO_2, CH_4 und N_2O in unterschiedlichen Regenmoorökosystemen am Alpenrand. Mit dieser Untersuchung sollen die oben identifizierten thematischen und räumlichen Kenntnislücken aufgefüllt werden.

Dabei soll einerseits aufgeklärt werden, wie sich die Renaturierung im Vergleich zu naturnahen und degradierten Bedingungen auf die Gasflüsse auswirkt. Das Potenzial von stoffhaushaltlichen Untersuchungen für eine funktionelle Erfolgkontrolle von Renaturierungsmaßnahmen steht dabei auf dem Prüfstand. Hierfür wird die Kohlenstoff-Bilanz der Standorte aus den gashaushaltlichen Messungen ermittelt.

Andererseits sollen der lückige Kenntnisstand über die Klimarelevanz von Regenmoorökosystemen für eine potenzielle nationale Berichterstattung zur UNFCCC verbessert und des Weiteren die Eignung von Regenmoorrenaturierungen zur dauerhaften Emissionsminderung und zum Aufbau von biologischen Senken überprüft werden. Dafür ist es erforderlich, die Bilanzen aller drei Gase zu ermitteln und mit dem jeweiligen globalen Erwärmungspotenzial (GWP) zu verrechnen.

Zur Aufklärung der zu Grunde liegenden Prozesse des Gasaustausches und zur Ermittlung von Steuerfaktoren des Gashaushaltes sollen die Flüsse und Bilanzen mit standörtlichen und Vegetationsparametern in Beziehung gesetzt werden, um Eingangsparameter für die Modellierung der Bilanzen zu erhalten.

Aus diesen Zielen ergeben sich folgende übergeordnete Fragestellungen für diese Arbeit:

(1) Welche Eigenschaften muss eine Messtechnik haben, die in einem kleinräumigen Mosaik von unterschiedlichen Regenmoorökosystemen erlaubt, Netto-CO_2-Flüsse und parallel dazu CH_4- und N_2O-Flüsse zu erfassen (Kap. 2)?

(2) Inwieweit kann ein Störungsgradient von naturnahen über renaturierte zu degradierten Regenmoorökosystemen mittels Vegetation und standörtlichen Faktoren abgebildet und indiziert werden (Kap. 3)?

(3) Wie unterscheiden sich die naturnahen, renaturierten und degradierten Regenmoorökosysteme hinsichtlich der CO_2-Flüsse und Bilanzen, inwieweit spiegelt dieses gashaushaltliche Verhalten den Störungsgradienten wider und kann mittels Vegetations- oder Standortfaktoren erklärt und modelliert werden (Kap. 4)?

(4) Wie unterscheiden sich die naturnahen, renaturierten und degradierten Regenmoorökosysteme hinsichtlich der CH_4- und N_2O-Flüsse und Bilanzen, inwieweit spiegelt dieses gashaushaltliche Verhalten den Störungsgradienten wider und kann mittels Vegetations- oder Standortfaktoren erklärt und modelliert werden und wie stehen diese Flüsse in Relation zu den CO_2-Flüssen (Kap. 5)?

(5) Wie wirkt sich die Renaturierung von Regenmooren auf die Flüsse der klimarelevanten Spurengase aus, inwieweit können renaturierte Regenmoorökosysteme als biologische Senken einen Beitrag zur Klimaentlastung leisten und wie ist die Klimarelevanz der Regenmoore am Alpenrand im Vergleich zur durchschnittlichen Emission pro Kopf einzuschätzen (Kap. 6)?

7. Gebietsauswahl

Die Auswahl eines Untersuchungsgebietes bestimmt sehr wesentlich die zu erwartenden Ergebnisse einer Untersuchung. Aus den formulierten Zielen und Fragestellungen wurden folgende Kriterien für die Gebietsauswahl aufgestellt:

(1) Vorhandensein einer für den voralpinen Regenmoorgürtel möglichst repräsentativen Reihe von naturnahen über degradierte bis renaturierte Regenmoorökosysteme

(2) Räumlich enges Muster der verschiedenen Ökosystemtypen zur Erleichterung der Logistik

(3) Verschiedene Intensitäten der Degradierung (Torfabbau und Dränage) und verschiedene Zeitabstände seit der Renaturierung

(4) Leichte Zugänglichkeit

(5) Informationsbasis zum Projektgebiet aus Vorläuferprojekten

Nach einer ausführlichen Recherche zu potenziellen Untersuchungsgebieten kamen drei Gebiete in die engere Wahl: Die Kendlmühlfilze / Lkr. Traunstein, die Koller- und Hochrunstfilze / Lkr. Rosenheim und das Breitfilz / Lkr. Bad-Tölz-Wolfratshausen. In allen drei Gebieten sind Renaturierungsprojekte durchgeführt worden. Schließlich konnte im Zentrum der Kendlmühlfilze ein Teilgebiet identifiziert werden, das die genannten Kriterien am besten von allen drei Gebieten erfüllte (zur Gebietsübersicht, siehe chapter 3, Fig. 3.1)

8. Bearbeitungszeitraum und Präsentation von Ergebnissen

Diese Arbeit wurde in folgenden Phasen durchgeführt: Im Juni 1998 wurde das endgültige Thema formuliert und die Bearbeitung begann. Der Prototyp der neuen Messhaube wurde bis August 1998 entwickelt und gebaut. Testmessungen mit dieser Haube liefen im September 1998 sowohl im Labor als auch im Gelände (Weidfilz bei Mooseurach). Parallel dazu lief im September 1998 die Gebietsauswahl für die Untersuchungen. Im Oktober wurden 15 Hauben „in Serie“ gebaut und die zugehörigen 42 Bodenrahmen fertiggestellt. Im November 1998 wurden die Untersuchungsplots in der Kendlmühlfilze als repräsentative Beispiele für naturnahe, degradierte und renaturierte Regenmoorstandorte ausgewählt und die Messplots eingerichtet. Die Versuchseinrichtung wurde ebenfalls im November mit dem Bau der Stege (insgesamt 250 m) abgeschlossen (unterstützt durch die Strafvollzugsanstalt Bernau). Damit war die Standortvorbereitung mehr als einen Monat vor dem Beginn der Messungen abgeschlossen.

Da es das Ziel war, für das gashaushaltliche Monitoring ein volles Kalenderjahr zu erfassen, wurde das wöchentliche Messprogramm am 5.01.1999 gestartet und bis zum 28.12.1999 in diesem Rhythmus durchgeführt. Die Spurengase wurden immer im Lauf der beiden Folgetage nach der Probenerfassung am GC des Institutes für Bodenökologie der GSF-Neuherberg bestimmt. Ab April 1999 wurden speziell für CO_2 zusätzliche Messtage eingeführt, die der besseren Kalibrierung von Lichtabhängigkeits- und Atmungskurven dienten. Hierfür kam ein online-CO_2-Analysator zum Einsatz (Li-800, Licor, Lincoln, NE, USA), der eigens für den Geländeeinsatz konfiguriert wurde. Diese prozessorientierten Kampagnen liefen bis Juli 2000 weiter.

Erste Auswertungen konnten bis August 2000 vorgelegt werden. Eine Fortsetzung der Datenauswertung und Bearbeitung war dann, aufgrund anderer Beschäftigungsschwerpunkte des Autors, erst wieder zwischen September 2001 und März 2002 bzw. danach bis zum Abschluss dieser Arbeit nur in Teilzeit möglich. Von Mitte März 2002 bis Ende 2004 hat der Autor für die Universität Bayreuth im Rahmen des EU-Projektes CARBOMONT die Koordination des Messstandortes Berchtesgaden übernommen. Zusätzlich wurden im Jahr 2004 Messkampagnen für das EU-Projekt CARBOEUROPE-IP durchgeführt. Diese Tätigkeiten haben für die Fertigstellung der Dissertationsarbeit zusätzliche inhaltliche Impulse gegeben.

Dennoch wurden schon frühzeitig erste Ergebnisse auf nationalen und internationalen Tagungen vorgestellt, die zu einer internationalen Wahrnehmung der stoffhaushaltlichen Aktivitäten am Lehrstuhl für Vegetationsökologie führten:

Drösler M. & Pfadenhauer J, 1999: Quelle oder Senke? Ein Beitrag zur Bilanzierung der Spurengasflüsse von naturnahen, degradierten und renaturierten Mooren. Vortrag 10.07.1999. Fachtagung Klimaschutz durch Moorschutz? 10.-11.06.1999. NNA, Schneverdingen.

Drösler M. & Pfadenhauer J, 1999: The sink- and source-function of bog ecosystems for climatic relevant trace gases – first results from south Germany. Talk at 42nd IAVS Symposium 26.-30.07.1999. Bilbao, SP.

Drösler M., Kamp T. & Pfadenhauer J, 1999: Die Quellen und Senkenfunktion von Regenmoorstandorten für klimarelevante Spurengase – Erfassung mittels neuentwickelter „closed chamber“-Technik. Poster. 29. Jahrestagung der GFÖ. 13.-18.09.1999, Bayreuth, DE.

Drösler M., Kamp T. & Pfadenhauer J, 2000: Relationship of C-balance with GWP-balance: An evaluation tool for trace gas fluxes of wetland ecosystems. Talk at the INTECOL invited papers Symposium Global Wetlands and Greenhouse Gases. Within the Quebec2000: Millenium Wetland Event.6.-12.08.2000. Quebec, CA.

Drösler M. & Pfadenhauer J, 2001: Restoration effect on C-balance and GWP-balance of bog ecosystems in southern Germany. Talk at the XIIth Global Warming Conference. 8.-11.04.2001, Cambridge, UK.

Drösler M. 2001: Klimaschutz durch Moorschutz? Entwicklungspotenzial von biologischen Senken durch Hochmoorrenaturierung. Vortrag. ANL-Lehrgang: Ökologie und Management von FFH-Lebensräumen: Hoch- und Übergangsmoore. 1.-20.6.2001. Laufen.

Drösler, M. 2001: The potential of bogs in climate mitigation – evaluation by C-balance and GWP-balance. Poster, 44th IAVS, 29.07.-04.08.2001, Freising; DE.

Drösler M. & Pfadenhauer J, 2001: Biodiversity of vegetation as indicator for trace gas exchange in bog-ecosystems - species matter. Talk, 31. GFÖ-Tagung, 17.-23.09.2001Basel, CH.

Drösler M. & Pfadenhauer J, 2002: Net ecosystem productivity of natural, degraded and restored bog ecosystems, Talk. 7. Jahrestagung des Arbeitskreises Experimentelle Ökologie der GfÖ 7. - 10. April 2002, Freising, DE.

9. Einsatzgebiete der entwickelten Haubentechnik

Die im Rahmen dieser Arbeit entwickelte Haubentechnik hat über das Messprogramm der Arbeit hinaus fortlaufende Anwendung in Projekten zum Austausch von klimarelevanten Spurengasen gefunden. Einen Überblick gibt Abb. 1.1.

1. 1999-2000: Kendlmühlfilze, Regenmoorökosysteme, Messprogramm für CO_2–, CH_4-, N_2O-Flüsse im Rahmen der Doktorarbeit (volle Jahresbilanzen), Vegetationsökologie TUM mit IBÖ-GSF

2. 2001: Stubaital (Österreich), hochmontanes Grünland: 2 Demonstrationskampagnen für CO_2–Flüsse, Vegetationsökologie TUM für die Uni- Innsbruck in Vorbereitung des EU-Projektes CARBOMONT

3. 2001: Cottbus, Renaturierung von Tagebauflächen, Messkampagne für CO_2–Flüsse, Vegetationsökologie TUM mit BTU-Cottbus

4. 2002-2004: Nationalpark Berchtesgaden, hochmontanes bis subalpines Grasland, Messkampagnen für CO_2–Flüsse, für das EU-Projekt CARBOMONT, volle Jahresbilanzen; Pflanzenökologie Uni Bayreuth mit Vegetationsökologie TUM.

5. 2002-2004: Stubaital (Österreich) hochmontanes bis subalpines Grasland, Messkampagnen für CO_2–Flüsse, für das EU-Projekt CARBOMONT, Uni Innsbruck mit Pflanzenökologie Uni Bayreuth und Vegetationsökologie TUM.

6. 2003: Seebodenalp (Schweiz), Messkampagne für CO_2–Flüsse, im Rahmen des EU-Projekts CARBOMONT, Uni Innsbruck mit PSI, Schweiz

7. 2004: Tharandt, Grasland, Messkampagnen für CO_2–Flüsse, im Rahmen des EU-Projektes CARBOEUROPE-IP, Pflanzenökologie Uni Bayreuth mit Meteorologie TU Dresden

8. 2005: Donauried, Grundwassermoor, Vergleich Degradierung-Renaturierung, Messprogramm für CO_2–, CH_4-, N_2O-Flüsse, volle Jahresbilanzen, Vegetationsökologie TUM mit Max-Planck Institut für Biogeochemie, Jena und Uni Hohenheim.

9. 2005: (1) Bayreuth, Klimakammerversuch mit verkleinerter Version der Haube (Effekt von Austrocknung von Graslandökosystemen auf CO_2-Flüsse) (2) Grasland, Effekt von Klimaveränderungen (Freilandversuch) auf CO_2-Flüsse Pflanzenökologie Uni Bayreuth

10: 2006 (geplant): Poznan (Polen) naturnahes Grundwassermoor. CO_2–, CH_4-, N_2O-Flüsse, Vergleich mit Eddy-Kovarianz. Vegetationsökologie TUM mit Department für Agrometeorologie, Universität Poznan.

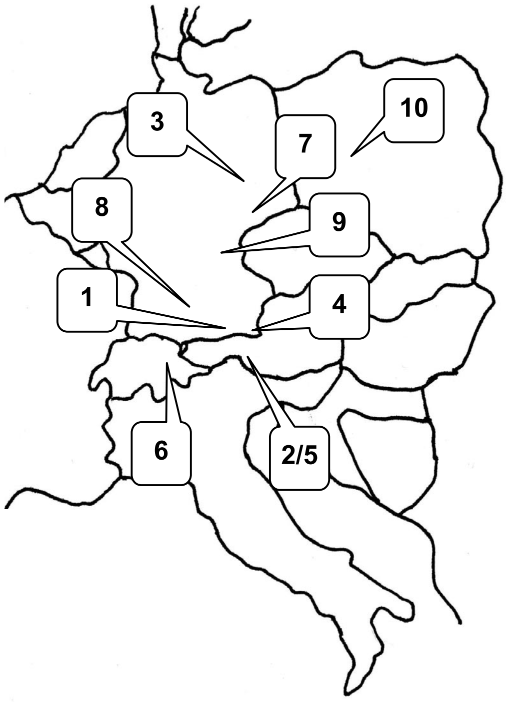

Abb. 1.1: Einsatzgebiete der im Rahmen dieser Arbeit entwickelten Haubentechnik

10. Aufbau der Arbeit

In **Kap. 2** wird die Gaswechsel-Haube beschrieben, die für die Messungen entwickelt und gebaut wurde. Die Ergebnisse von Eignungstests für gashaushaltliche Messungen insbesondere in Hinblick auf NEE und parallele Flussmessung aller drei klimarelevanter Spurengase werden ausführlich dargestellt. Durch Vergleiche mit der etablierten Eddy-Kovarianz Methode wird die Zuverlässigkeit der Haube überprüft. Die Standortbeschreibung erfolgt in **Kap. 3**. Die Identifikation eines Störungsgradienten von naturnahen über degradierte bis hin zu renaturierten Standorten wird in einem vergleichenden Ansatz mittels Vegetations- und Standortparametern dargestellt. **Kap. 4** zeigt, wie sich diese Standort- und Vegetationsunterschiede in den CO_2-Flüssen widerspiegeln und inwieweit die CO_2-Bilanzen als funktionaler Indikator Renaturierungserfolge anzeigen können. Die Auswirkungen der Standort- und Vegetationsunterschiede und der Einfluss der Renaturierung auf die CH_4- und N_2O-Flüsse werden in **Kap. 5** erläutert. Die Identifikation von Steuerfaktoren für die Erklärung der Gasflüsse und als Basis für die Modellierung nimmt in den **Kap. 4** und **5** breiten Raum ein. Zudem werden in **Kap. 5** die Beziehungen zwischen den CO_2- und CH_4-Bilanzen erläutert. In **Kap. 6** wird schließlich die Kohlenstoff-Bilanz ermittelt und der Klimarelevanz der Standorte gegenübergestellt. Die Abschätzung der Wirkungen von Regenmoorrenaturierungen auf die Klimarelevanz der Standorte dient zur Beurteilung des Einsatzpotenzials als Klima-Entlastungsmaßnahme und als biologische Senken im Klimaschutz. Abschließend werden in **Kap. 7** in Form einer Synthese die wesentlichen Ergebnisse der Einzelkapitel zusammengestellt.

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A PORTABLE CHAMBER SYSTEM FOR MEASURING NEE OF LOW STATURED VEGETATION

Summary

The presented chamber allows to study Net Ecosystem Exchange (NEE) of CO_2 on a system level especially under those conditions when eddy covariance is limited (hill-slopes, advection, heterogeneity of the fetch with small scale mosaic of vegetation types, lack of mains power, stably-stratified night-time, process-studies of small-scale experimental management measures).

For NEE-measurements it is crucial that chambers are transparent and climate-controlled: Light transmission through the chamber is 95 % PAR. The cooling system limits temperature and moisture differences between the inside and outside of the chamber to a maximum 1 °C difference in air temperature and 20 % relative humidity (RH) difference at the end of an enclosure. This leads to a VPD relation inside to outside between 0.7 to 0.9.

Without climate control, steep temperature and moisture gradients up to 3°C min^{-1} and 20 % RH min^{-1} will appear and lead to overestimations of CO_2-uptake up to 35 %.

Crosschecks between eddy covariance and the chamber system proved the comparability of the techniques. Up to 94% of the chamber fluxes could be explained by eddy covariance. Therefore this chamber is confirmed as complementary technique for eddy covariance.

It runs as a static (gas analysis via GC in the lab) as well as a dynamic system (online gas analysis via IRGA). As a static system, the chamber allows to sample the three biogenic trace gases CO_2, CH_4 and N_2O simultaneously.

Introduction

To assess the role of ecosystems in global change, it is important to obtain realistic data on the rates of trace gas exchange between the biosphere and the atmosphere. A better knowledge of these fluxes for specific ecosystems is necessary to strengthen the reporting of trace gas emissions in relation to national commitments under the Kyoto Protocol (Schulze et al., 2002).

Project-clusters like CARBOEUROPE are specifically focusing on establishing long term databases of greenhousegas (GHG)-exchange of the land surface. The basic tool is the eddy covariance technique, standardized within the projects with common equipment and methodology design, which allows site intercomparison (Aubinet et al., 2000).

Eddy covariance is a non-invasive and non-destructive micrometeorological method which reveals continuous integrated signals with low spatial (typically 0.05 to 1 km^2) but high time resolution (usually 0.5 hours; Fowler & Duyzer 1989). Gas exchange is measured at ecosystem or even landscape scale. It does however normally need appropriate topography (flat, homogeneous and extensive terrain) and power supply to work properly. At night, the frequent lack of turbulence in the boundary layer may affect the accuracy and reliability of the processed fluxes (Lenschow 1995). The advantage of obtaining an average signal over an area up to several hectares, may contrast with the need to investigate a small scale mosaic within the source area, caused for example by micro-topography, disturbance or management practices.

To overcome these limitations, chambers may be used as complementary tool to eddy covariance systems. Their high spatial resolution (normally < 1 m^2) allows distinctive measurements of ecosystems in a small scale mosaic and offers comparison of the influences of even minor differences in key factors, such as e.g. vegetation cover, floristic composition, water level or management effects. They are generally insensitive to topographic settings. Problems when using chambers for NEE measurements are principally (Livingston & Hutchinson, 1995) a) soil and atmosphere temperature changes under the chamber b) alteration or elimination of the natural atmospheric pressure fluctuations c) boundary layer resistance at the soil-atmosphere interface may be higher inside than outside the chamber d) concentrations build up or reduction to levels inhibiting the normal emission or uptake rates and e) inaccurate headspace volume. Often, leakage is an additional practical problem, especially in sealing the chamber to the substrate.

A number of different chamber types have been developed for soil flux measurements (Edwards & Sollins, 1973, Edwards, 1974, Denmead, 1979, Conrad & Seiler, 1985, Mosier, 1989, Schütz & Seiler, 1989, Bekku et al., 1995, Grau, 1995, Fang & Moncrieff, 1996, Norris et al., 1996, Norman et al., 1997, Rochette et al., 1997, Ambush & Robertson, 1998, Chan et al., 1998, Fang & Moncrieff, 1998, McGinn et al., 1998, Dantec et al., 1999), for photosynthesis measurements, especially on a leaf basis (see e.g. Jarvis et al., 1971, Strain, 1975, Field et al., 1989, Willert et al, 1995) and for measurements which combine these two approaches (Bartlett et al. 1990; Whiting et al. 1992). Pumpanen et al. (2004) tested recently in a comparative approach a wide range of chambers used for soil respiration measurements. The chamber of Schürer & Plesser (1997) is not applicable to NEE measurements, because of the limited transmission for PAR.

For the use as complementary technique to eddy covariance, the chamber developed by G.J. Whiting and used in several studies (Bartlett et al., 1990; Whiting et al., 1992, Carrol & Crill, 1997) and the chamber used in the study of Tuittila et al. (1999) appeared most suitable. Both chamber types were transparent and climate-controlled, accommodating the approaches of ecophysiologists (CO_2-balance) and soil scientists (CO_2-, CH_4- and N_2O-efflux), and suitable for the assessment of GWP (global warming potential)-balances. Both chambers however, are equipped with sophisticated cooling systems which provoke elevated costs and high weight of the whole system, with the need of two persons to run one chamber.

The focus for the use of the presented chamber was to compare several ecosystems within a small scale mosaic simultaneously. It should be possible to apply nine chambers parallel and one person had to run three chambers at the same time. For this approach it was not possible to use the above cited chambers (costs / manpower relation). Therefore the development of the new chamber was necessary and was oriented on following criteria: a) measurements should be possible in remote areas without mains electrical power supply b) measurement equipment should be easily portable between the plots c) gas exchange should be obtained by differential measurements of CO_2 uptake (NEE, transparent chamber) and release (R_{eco}, darkened chamber) under temperature-controlled conditions (development of a modular cooling system for field conditions without power supply) d) fluxes of all climatically relevant trace gases from ecosystems (CO_2, CH_4 and N_2O) should be measured simultaneously and e) the system should be easy to manage and should work even under difficult terrain conditions.

This chamber is designed to work as complementary technique under limiting conditions for eddy covariance. To check the comparability of the systems, a comparison between eddy covariance and chamber techniques under appropriate topographic conditions was undertaken. Up to now, only few studies compared the techniques directly via simultaneous measurements of fluxes by eddy covariance and chambers (e.g. Norman et al., 1997, Dore et al., 2003).

Material and Methods

Chamber design

The size of the chambers should cover the typical vegetation types of low statured grassland and mire ecosystems and, at the same time, be manageable for one person. As the spatial variability of trace gas fluxes is pronounced (e.g. Rochette et al., 1992), the bigger the enclosed area the better: Fluxes should reflect the average situation within the selected ecosystem. The base of the chambers was 78 cm x 78 cm. The height of two chamber types were 40 and 50 cm, respectively, taking into account the average heights of the typical bog vegetation types in the foothills of the German Alps. These dimensions fit well to cover, for example *Eriophorum* and *Sphagnum* hummocks, and allow to derive distinctive gas balances for these individual ecosystem types. With this size, and a weight around 10 kg each, the chambers are still easy for one person to handle. To follow vegetation growth, or rising water level in restoration ponds, 10, 30 and 40 cm chamber extensions are available. This shape fits well with almost all non-forest ecosystems, and even in forests it is possible to enclose low-statured understory herbs and shrubs (Fig. 2.1).

For chamber material, 3 mm thick Plexiglas XT type 20070 was selected as it is shock resistant, frost stable, light weighted and affordable. A Licor quantum sensor (190 SA) was used to test light transmission in the wavelength of PAR (390 – 700 nm). The tests revealed a transmission rate of 95 %.

75 cm x 75 cm square collars (Fig. 2.1) were built using 3 mm thick PVC, a light weight, inexpensive and shock-resistant material. The material shows no weathering effects and resists frost. Reduced heat absorption due to the white colour of the collars limits adverse effects on the vegetation and soil processes. Two types of collars have walls of 10 or 20 cm high and are surrounded by a 3 cm wide platform, 3 cm below the top (Fig. 2.1). The chamber with its sealing is placed on this platform during the measurement. The higher collars are used for plots with pronounced micro topography, to eliminate exchange between the atmospheres inside and outside the chamber. The collars are inserted to the soil till the platform (between 7 and 17 cm, depending on the height of the collar walls and the micro topography) to guarantee gas tightness. The platform of the collars is levelled with the soil and vegetation surface. These collars are generally placed days before the measurement campaigns start and, once in place, remain there over the whole measurement season. The plot area of the collars of 0.56 m^2 approaches a proposed minimum surface for chambers to avoid too many effects from spatial variation (Heinemeyer et al., 1995). It is important to fix the collars in the soil to avoid frost movements and to sustain the collar in wet and smooth wetland soils. To secure the collars 2 – 4 plastic rods (80 cm) are pushed crosswise in the soil on every side and fixed to the platform with screws from outside.

The sealing between the chamber and the collar has to be gas tight under all weather and seasonal conditions. A rubber tube (Sahlberg, Germany) of around 2 cm in diameter was used, which is gas

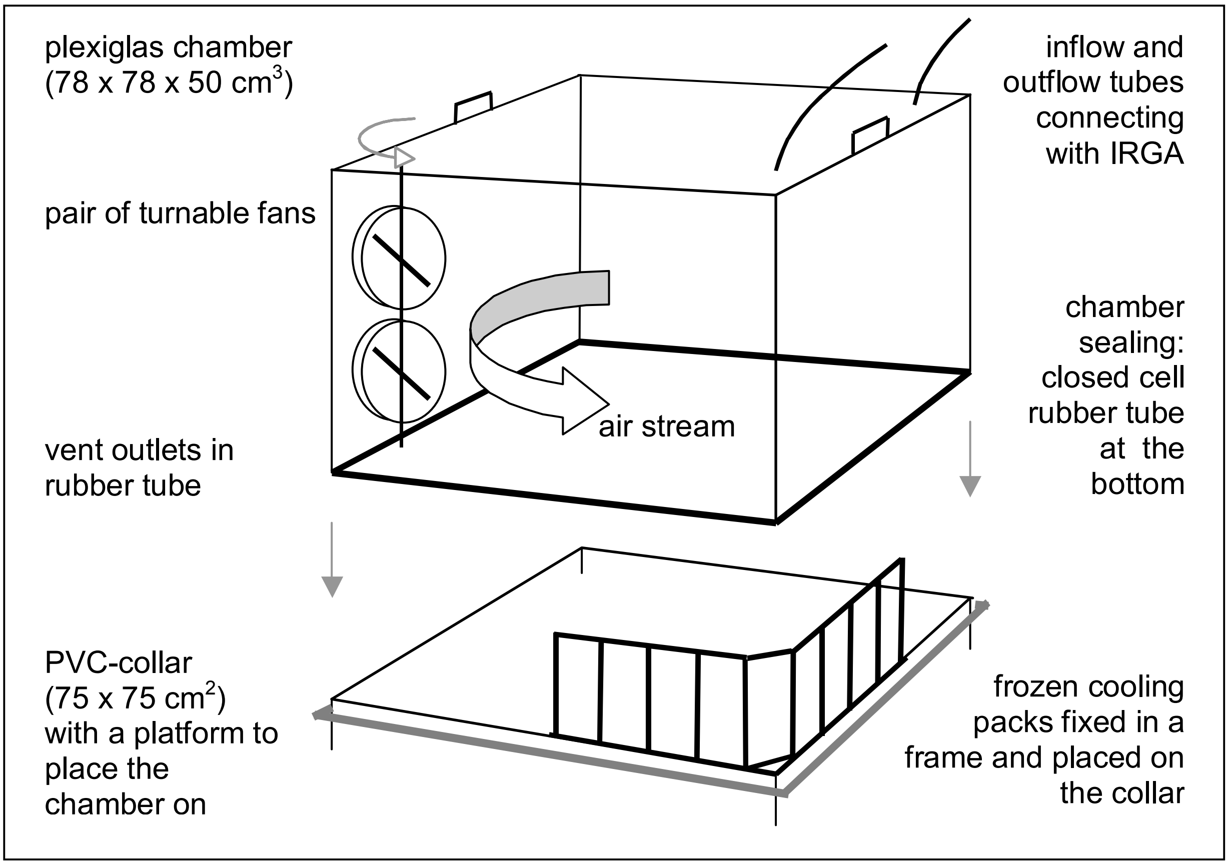

Fig. 2.1: Transparent closed chamber system. The fans can be turned from outside the chamber to regulate the amount of air passing over the cooling packs. By this and the quantity of cooling packs, the power of the cooling system can be regulated to match the instant conditions of the measurement

tight, resistant to frost and to UV-radiation. It is fixed to the chamber walls by a clamp-like extension of the tube. The tube-like shape provides certain elasticity and therefore chamber placement on the collar is cushioned. This helps to avoid unexpected emissions of trace gas bubbles from the soil surface while measuring. Finally two elastic cords are used to secure the chamber to the collar, reinforcing the gas-tight seal. To avoid pressure differences inside versus outside the chamber, a vent was installed in the rubber tube. A 80 cm PVC tube 4 mm wide was used, oriented on Hutchinson & Mosier (1981).

Ventilation is necessary to guarantee constant air circulation inside the chamber since only 100 ml of gas are sampled in the standard program from a total of 250 – 560 l depending on the chamber height. Each chamber is equipped with a pair of small fans powered by small electric motors (Igarashi, 3 V). Each motor runs by a set of two rechargeable batteries (1.5 V, 700 mAh), which power the fans for more than 30 min, which is the maximum enclosure period for a measurement cycle. Fan direction is adjusted during the measurement from outside the chamber, such that a circulation stream is established in the chamber, but the velocity of the air stream is highest along the walls and lowest in the center of the plot (Fig. 2.1). The most critical factor in transparent closed chamber use for CO_2 flux measurements (or Net Ecosystem Exchange, NEE), is the temperature inside the chamber. Peltier cooling elements could not be used because chamber volumes and power requirements were too large. Therefore a simple cooling system was developed (see Fig. 2.1): It consisted of ordinary cooling packs placed inside the chamber, exchanging heat with the air according to the adjustment of the fans (Fig. 2.1). During the measurement, the direction of the fans regulate the cooling power, which is very sensitive to the direction of the air stream. With this method, dynamic radiation during sunny and cloudy weather with subsequent temperature effects on the chamber atmosphere could be balanced. Tests of the temperature effect on the measured fluxes were undertaken, via comparison of climate controlled and non-climate controlled runs measured consecutively on the same plot and under the same outside temperature and climate conditions.

When measuring NEE by chamber technique, it is necessary, to sample under night and day conditions to record uptake as well as respiration fluxes. In the standard measurement program, placing a blackout shroud on the chamber simulates night conditions. These night simulations were used together with real night measurements for the model parameterisation of the ecosystem respiration-model. A silver-colored lid in PVC-quality was used, which is insensitive to UV radiation and blankets the chamber in absolute darkness.

Measurement of gas-fluxes

The chamber can be used as a static or dynamic closed chamber system. As a static system, a rubber valve enables extraction of gas samples using a syringe, connected to a type of vacutainer (evacuated glass bottles of ca. 100 ml volume). Because the volume of the chamber headspace is 250 to 560 l (depending on the chamber and extensions used), the effect of extracting 5 bottles with 100 ml each, during one measurement cycle on e.g. pressure or gas concentration, is negligible. The gas samples are analyzed in the lab by gas chromatography (ECD- and FID-detector) to determine the contents of CO_2, CH_4 and N_2O.

As a dynamic system, the chamber is connected by a flexible inflow and outflow tube (Fig. 2.1) to an infrared gas analyzer (IRGA; Licor-800 or Licor-6262, Licor, Lincoln, NE, USA), creating a closed circle with a constant through flow of 2 l/min (membrane pump, KNF, Germany). The entire volume of the inflow/outflow tubes and the tubes within the analyzer is around 0.1 l. Thus the cycling time to the analyzer is less than 3 sec and constant over time so that concentration changes in the chamber atmosphere are tracked nearly “real time”. The use of the IRGA offers instant CO_2 values directly in the field. In this way, the dynamic behaviour of CO_2 can be detected in a much higher time resolution than with the gas chromatography approach (taking samples and analyzing them in the lab).

The principle of closed chambers is the enrichment or rarefaction of gases in the chamber atmosphere. The concentration change over time is to be detected. Careful assessment of linearity in the increase or decrease of concentrations within the first minutes determines the undisturbed section of the graphs as a database for the flux calculation. Nevertheless Conen & Smith (2000) showed, that closed chambers may reduce emission rates by changing the concentration of trace gases in the soil pores after employment of the chamber and showed that this effect can not be detected by linearity of the concentration changes in the headspace of the chamber. However, as the developed chambers are high (40 to 90 cm, depending on the extension used) the ratio of an estimated soil air volume to headspace volume was on every measured site far below 0.1. Thus using a relation of Conen & Smith (2000) between this ratio and the missed flux, the missed flux in the presented approach should be lower than 1 %. Therefore, a linear increase was taken first as proof of undisturbed conditions by the chamber upon the fluxes and second as representing the fluxes realistically.

Results and Discussion

Air temperature and humidity inside versus outside the chamber

The temperature control system allows to maintain the chamber temperature within a range of 1 °C compared to the outside temperature (Fig. 2.2). Only some measurements at the end of the enclosure time (measurement No. 4 and 5), which means after 16 to 32 min of enclosure, were slightly outside this range. Ambient air temperatures were recorded as –13 °C to +34 °C during gas measurements over the field period and were paralleled without greater difficulties, inside the chamber. Unexpectedly, the most challenging situations for the cooling system were not bright summer days with maximum air temperatures over 30 °C, but springtime with moderate air temperatures around 15 °C and high radiation energy. The reason being the lower the ambient air temperature, the smaller the temperature difference is to the surface of the cooling packs and thus, the relative power of the cooling system is reduced as well.

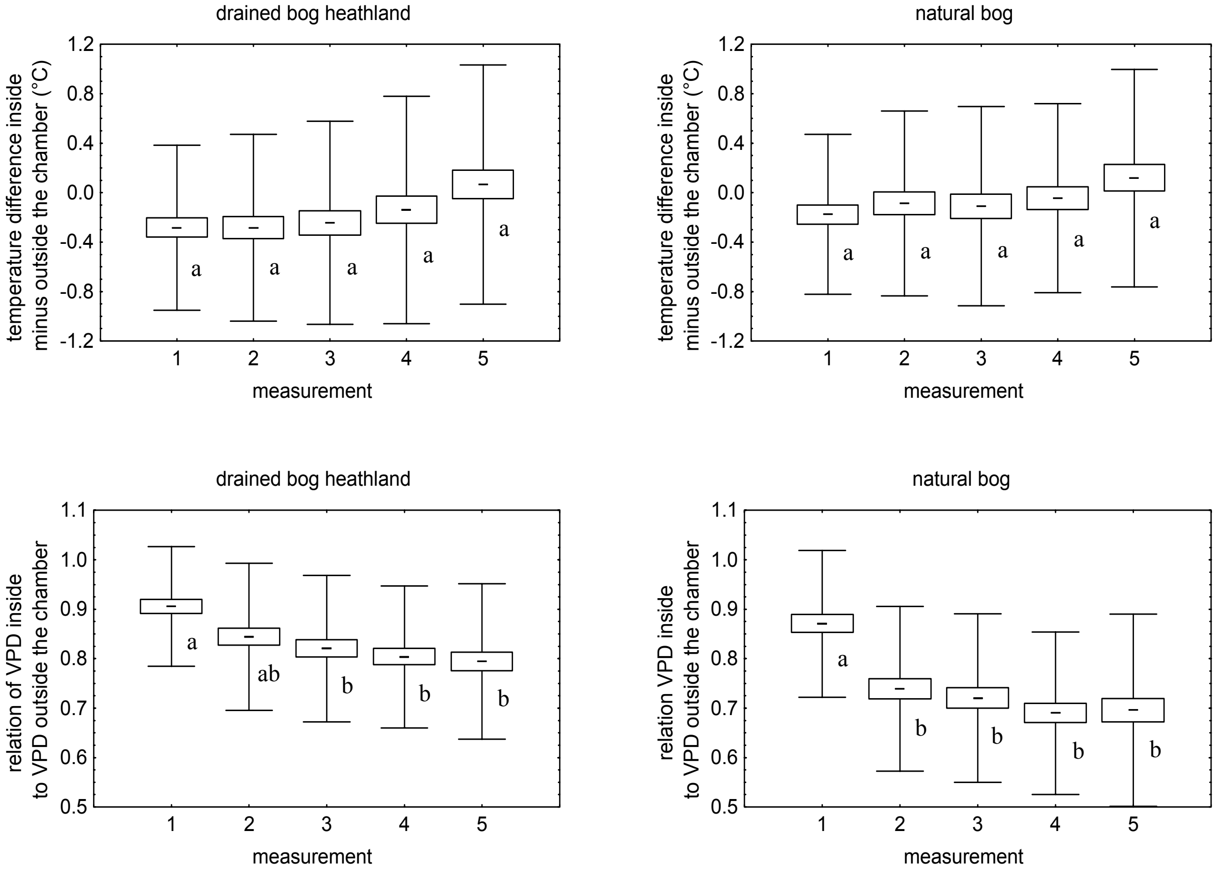

Fig. 2.2: Temperature and humidity (expressed as vapour pressure deficit, VPD) differences inside versus outside the chamber. Data were from a drained bog heathland and a natural bog site over the whole measurement period 1999 (n = 35). The first sample (No. of measurement) was taken immediately after closing the chamber. The fifth, at the end of the enclosure time (30 min). The remaining were taken in between with time steps between 3 and 8 min. For CO_2-flux calculation just data from the first minutes were used. For the CH_4- and N_2O-flux calculation the whole enclosure period was used, as for these gases normally longer enrichment times were needed than for CO_2. Data show mean, SE and SD values. Different letters indicate significant difference according to ANOVA, Tukey-test ($p<0.05$), n=66.

The air above a layer of vegetation is constantly loaded with humidity through evapotranspiration. When turbulence and transport of the ambient air are stopped by closing a chamber, humidity inside the chamber atmosphere rise immediately. But, ecophysiological properties of plants, such as the stomatal conductance, are sensitive to changes in vapour pressure deficit (VPD). Humidity was recorded over the whole enclosure time. Under non-climate-controlled test conditions humidity rose with up to 20% per minute in the first minutes even steeper than the temperature and reached after some minutes levels around 85-90%, which produced a film of condensed water inside the chamber. This water film reduced, on the other hand, the radiation by 20-30%, which, as a secondary effect, influenced ecosystem photosynthesis. Under climate controlled conditions, the humidity difference was lower than 20% over the whole enclosure period. The effect on VPD was expressed by the ratio between inside to outside the chamber (Fig. 2.2). Here, the difference between the relatively dry drained bog to the wet natural site is obvious, with higher VPD difference in the latter. However, a mean difference of 30% was not exceeded even at the end of the enclosure period (up to 32 min). This range of differences and the maximum differences in the warmer month of June, July and August were found as well by Johnson et al. (2000). The reason for the notably limited rising of humidity in comparison to non-climate-controlled runs is, that the air stream humidity condenses on the surface of frozen cooling packs. The condensed water remains in the frames and does not affect soil processes by punctually rising soil humidity, for example.

Temperature increases under enclosures may alter normal photosynthesis rates notably (Wagner & Reicosky, 1992). Matthias & Hernandez (1998) showed that soil temperature under opaque enclosures (without cooling system) changes in the upper 3 to 5 cm with degrees depending on soil humidity. Tests of the presented chamber without cooling system applied, showed that air temperature inside the chamber rose by steep gradients, with 3°C/min under bright conditions. This differed the temperature more than 20°C at the end of the enclosure period and the humidity rose to around 90 %. The same magnitude of temperature gradient is reported by Grau (1995). The effect of rising temperature on CO_2-C exchange rates was tested via a comparison of temperature controlled with non-controlled measurements (Fig. 2.3). In the non-controlled runs (which followed within 15 min after the controlled runs) the cooling packs were removed from the cooling system. Radiation and starting temperature was similar in the pair of measurements to compare. The comparisons were done on natural bog vegetation. Rising temperature inside the chamber led up to 30 % higher CO_2-C uptake. Bellisario et al. (1998) found this same magnitude.

The effect of at maximum 1 °C temperature difference (mean of the measurement) inside to outside the chamber provoke a maximum impact of the system upon the flux rates of less than 3 % (see Fig. 2.3). For instant quality checks of the performance of the measurements in the field, this mean 1 °C range was set as quality range and could be guaranteed under all weather conditions. For Q-10 analysis CO_2-C changes were expressed as relation (in percent) of the non-climate-controlled versus the climate-controlled measurement, and temperature changes were expressed as non-controlled minus controlled measurement (see Fig. 2.3). This test revealed a Q10 of 1.8.

The maximum detected differences of 35 % of the fluxes, when measuring without cooling system, are notable and provoke significant impact on the reliability of the calculated fluxes. This is one major source of errors when applying chambers and underline the necessity of the cooling system, not only to avoid short-term disturbance of the vegetation but also to obtain realistic numbers for gas-exchange (NEE) in ecosystems.

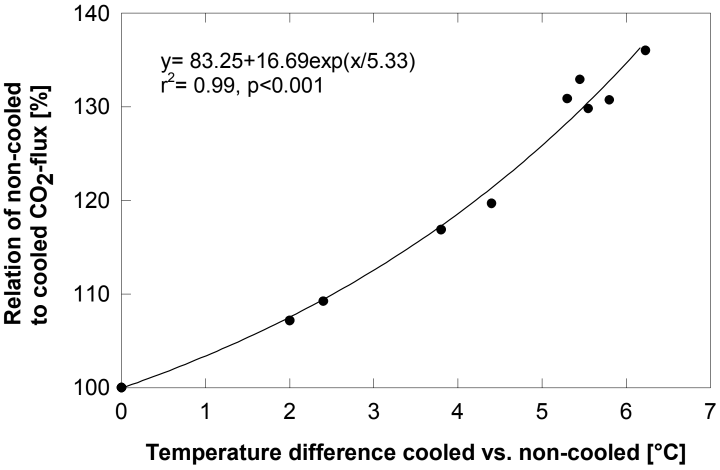

Fig. 2.3: Effect of temperature rise on NEE flux (transparent chamber). Fluxes were first measured with climate control and after 15 min under similar ambient conditions without climate control. Data are based in pair comparison with the climate controlled fluxes as 100% baseline.

Pressure

Pressure fluctuation is an important driver for the changing flux rate of trace gases between the biosphere and the atmosphere and may be altered by the enclosures (Mosier 1989, Fang & Moncrieff 1996). Pressure changes between inside to outside the chamber were recorded by a pressure transducer sensitive to pressure from 2 to 20000 mPa (MKS Baratron, Type 223B). During chamber placement only in the first 2 to 3 seconds a pressure difference inside to outside the chamber (at maximum 1 Pa) could be found. After five seconds, pressure differences were not longer detectable or remained below 0.1 Pa. This proves the pressure equilibration via a rubber valve, which is always kept open during placement. Five seconds after the placement, the valve is closed again and the measurement starts. During the enclosure period, the vent (designed after Hutchinson & Mosier, 1981) in the chamber sealing guarantees pressure equilibration. Lund et al. (1999) state, that even pressure differences of 1 Pa may have influences on the gas-exchange and can reduce notably the normal emission rates. However, such differences could be detected only in the first seconds after placement and before the measurement starts with closing of the valve. Thereafter pressure equilibration is reached via the vent. Conen & Smith (1998), however, point out a problem when using vents for pressure equilibration, the so-called venturi-effect, where an ambient air stream over the outlet of the tube may even provoke pressure changes (depressurisation) in the chamber. The authors found up to five times higher emissions of N_2O in vented enclosures compared to non-vented enclosures, with test plots on permeable soils respectively (Conen & Smith, 1998). The authors state, therefore, that the potentially beneficial effect of venting chambers is overweighed by the disadvantages. But vents should not be doubted principally, as the venturi-effect is relatively easy to avoid by positioning the outlet tube close to the soil/vegetation surface, where the wind-speed is generally very low. The vent in the presented chamber is placed in the sealing tubes (Fig. 2.1) and, therefore, is positioned close to the soil surface. Hence, the benefits of the vent in the way they are applied here should be greater than the risks of counter effects.

Gas-tightness

The different materials for the chamber (Plexiglas), sealing (closed-cell rubber) and collar (PVC) were gas tight. The tightness of the system as a whole, however, had to be tested, which means the chamber connected with the sealing and placed on the platform of the collar. Leakage tests in the field were done by assessing the linearity of the concentration changes with darkened chambers (see Fig. 2.4). Carbon dioxide and methane were analysed from the same gas samples. Therefore the linearity test could be done with both gases, allowing to avoid misinterpretation of single values. Assuming the underlined emission processes to be linear, these graphs with $r^2 > 0.95$ prove tightness of the system under field conditions.

As the basic assumption for the test in the field is necessarily a constant emission process, the gas tightness was tested as well in the lab. One collar was placed in a water basin, to avoid gas exchange below the collar. Then the chamber was set on the collar. Respiration air was blown into the chamber to elevate the CO_2 concentration before placing the chamber on the collar. The range of elevation reflected the common enrichment (less than 50 ppm) or rarefaction (less than 30 ppm) during a single NEE measurement cycle in the field. The IRGA signal was monitored for 20 min after placing the chamber to detect leakage effects. The detected concentration change of CO_2 (dc/dt) per min, was correlated with the level of CO_2 enrichment in the chamber air in comparison to ambient ($y=-1.578 + 200.91x^{0.74}$, $r^2=0.82$, n=6 $p<0.001$). This is consistent, as the very small leakage effect seems to be a function of diffusion processes going on through the vent. The leakage was less than 0.16 ppm min^{-1} at the plus 50 ppm level and less than 0.02 ppm min^{-1} at the plus 10 ppm level above ambient respectively.

This diffusion leakage is negligible in comparison to the concentration changes over time normally measured within the chamber. In contrast, measuring without pressure equilibration revealed significantly erroneous flux rates (Fang & Moncrieff, 1998).

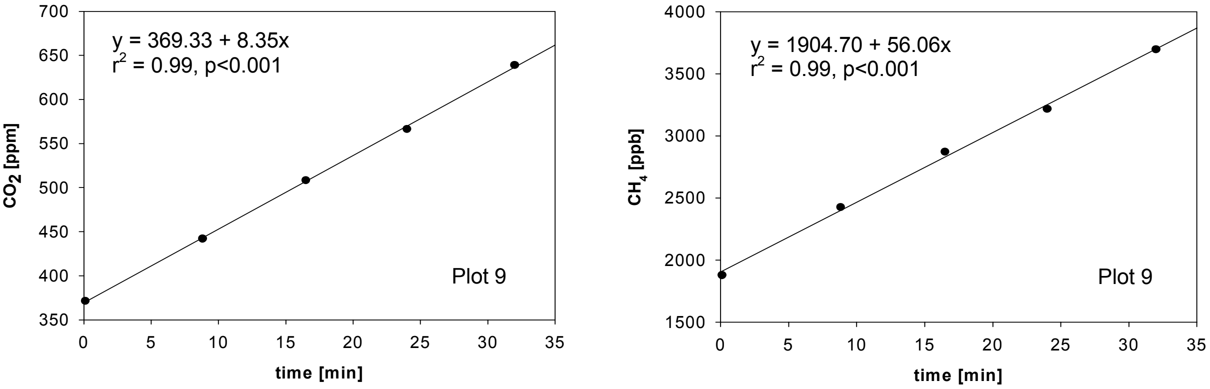

Fig. 2.4: Leakage tests under field conditions via assessment of linearity of concentration changes. Darkened chamber, CO_2 and CH_4 analysed by GC in the Lab from the same gas sample.

Headspace volume

The correct chamber headspace volume is important for the flux calculation. However, several conditions do limit the accuracy in determining the enclosed volume, especially a pronounced micro-topography of the soil or vegetation surface, air filled pores in the soil, air filled lacunas within the vegetation layer (like in *Sphagnum* lawns) or variable volumes due to snow cover and the changing water table. Rayment (2000) tested on permeable soils with air-filled pores, that the calculated flux rates with closed chamber systems underestimated soil CO_2 efflux by 10%. But, the effective volume of the chamber is not easy to assess exactly, as the integration in the gas-exchange processes of soil pores is decreasing with diameter and volume of the pores. Thus simple determination of bulk density will, especially on soils with dominating fine pores, not give satisfying results (Rayment 2000). Furthermore, destructive methods are not applicable to plots in running monitoring programs. Therefore in this study detailed micro-topographical measurements of the soil- or vegetation-surface were applied to determine the headspace volume. This height measurements were done on every point of 100 cell grid, fixed at a defined height over the collar. The height of the chambers, which are 40 cm up to 90 cm (with extension), limit the problem of over or underestimation, as the volume of air-filled pores and lacunas is low in comparison to the volume of the chamber air as a whole.

Air-mixing

Turbulence inside the chamber is necessary for a homogenous mixing of the air. To prove the mixing effect on the gas concentration gradient, cross tests were performed with and without ventilation. Fig. 2.5 shows an exemplary behaviour of a darkened chamber (covered with the black out shroud) with the signal being constant with ventilation but not so stable without ventilation. The latter proves that the air does not mix sufficiently in chambers with such large volumes without fans. The velocity of the air stream in the chamber was lower than 1 m s^{-1} along the chamber walls and less in the centre of the chamber (< 0.8 m s^{-1}). Such wind speed is rather common at the surface of vegetation outside the chamber. Wind-speed at the soil surface is merely affected by the fans due to vegetation cover, such that accelerated CO_2-emission from the soil surface because of higher wind speed in the boundary layer is avoided. The average aerodynamic resistance (ra) over all sites was 49.6 (15.1) s m^{-1}, with minimum values on the degraded heathland (D1 see chapter 3; 29.3 s m^{-1}) and maximum on the natural hummocks (N4, see chapter 3; 79.6 s m^{-1}). These are common values for free air conditions in low statured vegetation, thus the effect of the chamber air-stream on gas-exchange of the vegetation is considered as negligible.

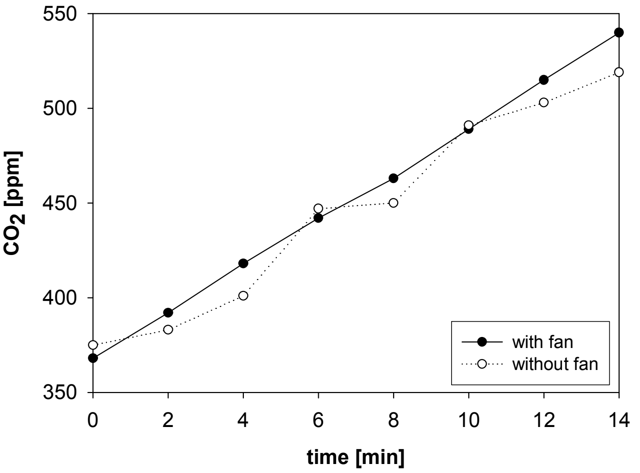

Fig. 2.5: Effect of turbulence on CO_2-flux rates: Homogenisation of the chamber-air via mixing with fan guarantees realistic tracking of fluxes.

Comparison of static and dynamic system

As CO_2 is measured both in static and dynamic manners, the data from the different analytical approaches should match absolutely to allow comparisons of the gas fluxes between different samplings. Rochette et al. (1992) compared static and dynamic chambers on different sites close together. The absolute values differed, but the authors found consistent relations between the tests. The presented test however, was done on the same plot at the

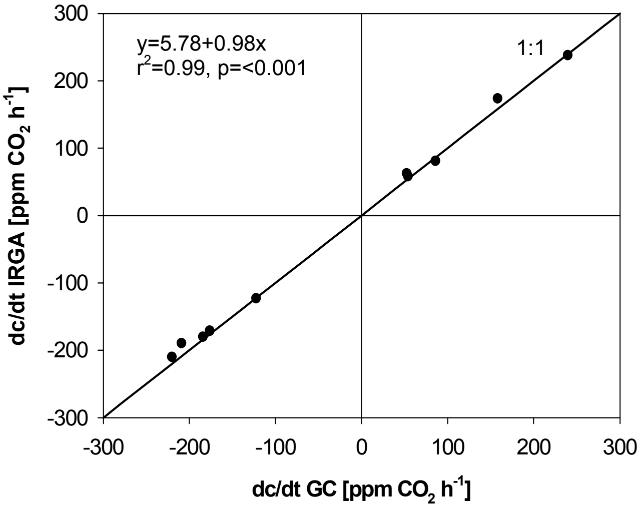

Fig. 2.6: Comparison between static (CO_2-GC) and dynamic (CO_2-IRGA) chamber systems. The comparative measurements for both analytic procedures were made at the same time and plot (wet *Sphagnum* hollow).

same time. The results maintained that even absolute values were very close together (Fig. 2.6), and the two measurement procedures did highly correlate. This is not only a comparison of analytic procedures but of different performance of the chambers, as in the dynamic version, a constant flow of 3 l/min through the chamber is established and in the static version singular gas-samples were extracted.

Comparison with eddy covariance

The high variability in time and space of gas exchange makes it difficult to compare different techniques. But as the scope of the presented chamber is to sample realistically absolute flux data, it is necessary to prove, as best as possible, the comparability of the measurements with other techniques (Janssens et al. 2000). Therefore, the developed chamber system was tested against eddy covariance, as the latter is used as reference system for global flux measurements (Aubinet et al., 2000) and the

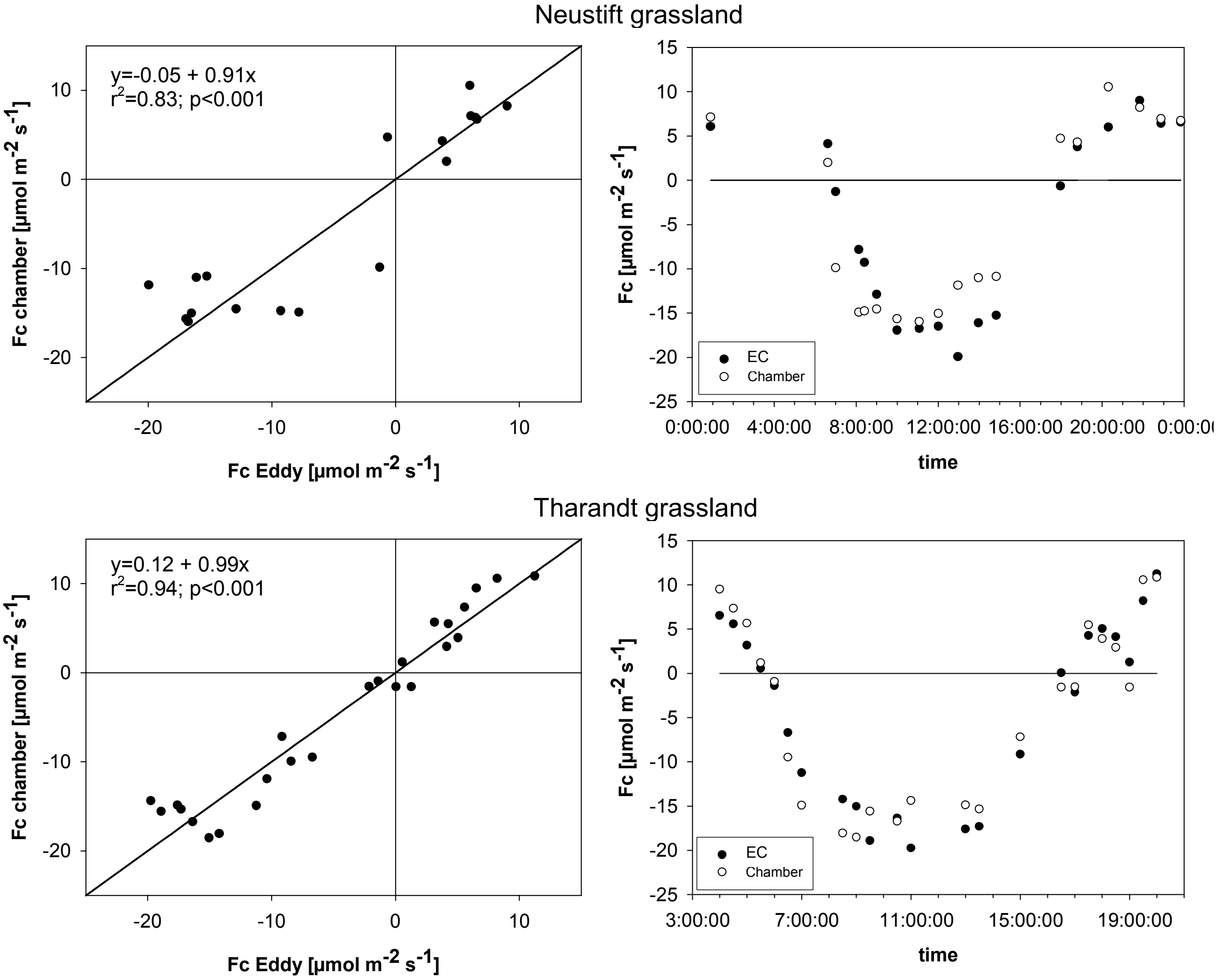

Fig. 2.7: Comparison of eddy covariance (EC) and chamber flux measurements. Chambers were placed within the fetch of the eddy covariance tower. Measurements were undertaken at both sites in June before first cut with peak LAI (~6). Neustift grassland was an alpine meadow (970 m a.s.l.). Tharandt grassland was a temperate lowland meadow (385 ma.s.l.). Eddy covariance was measured with open path Li-7500.

chamber should be used as complementary technique when eddy covariance is limited. Comparison was done in two campaigns on two different sites and under optimal conditions for eddy covariance measurements. Chambers were run as dynamic system with IRGA for online CO_2 analysis and maximum enclosure times of 6 min, depending on the flux. The chamber plots (n = 3 in Neustift, n = 4 in Tharandt) were installed within the fetch areas of the eddy covariance systems in flat meadow ecosystems. Special emphasis was given to carefully selected chamber plots representing the average development status of the vegetation within the fetch. The comparisons were done just before the first cut with LAI´s around 6 in both meadow ecosystems. The test areas were not the same as the bog area of the overall study, because for the test area the homogeneity of the sites and the appropriateness for eddy covariance was a prerequisite.

The Neustift grassland (Fig. 2.7) was located in a valley bottom (970 m a.s.l.). An up- and down-wind system is ruling the turbulence characteristic of the site. During the campaign typically around sunset and before sunrise the boundary layer showed a lack of turbulence. However the fit of the correlation between eddy covariance and chamber measurements showed satisfying results: Eddy covariance was able to explain 83 % of the variability of the chamber fluxes, at highly significant level. However, the two outliers (see Fig. 2.7) with close to zero fluxes from eddy covariance during sunset and sunrise, hampered a better fit of the two techniques. Therefore another campaign was undertaken during turbulent situations during night and day in the Tharandt-grassland (385 m a.s.l.; see Fig. 2.7). In this campaign the conditions for eddy covariance were perfect and therefore the chambers showed an even better fit to the eddy covariance data as in the Neustift experiment: Eddy covariance could explain 94 % of the variability of the chamber data. Considering the different properties of the techniques (more than 10.000 m^2 fetch for the eddy covariance against 0.5 m^2 for the chamber, sensitivity of eddy covariance to lack of turbulence and widely different time resolutions in the measurements), the range of coincidence is very satisfying (r^2=0.83 to 0.94) and assumed to be evidence, that this chamber-system can successfully measure realistic CO_2-fluxes in ecosystems with low statured vegetation. Law et al. (1999) however found less correlation between the two techniques, measuring in a ponderosa pine ecosystem. Dore et al. (2003) found similarly a good fit of night-time fluxes between the two techniques. During the day, however, their open chambers overestimated the flux of eddy covariance up to 26 %, as the result of higher temperatures (mean 4°C, max. 13°C) inside to outside the chamber (Dore et al., 2003). These findings are close to the detected 30 % overestimation of NEE, when using the presented chamber without cooling system, and demonstrate the necessity of a climate-control for making realistic and eddy covariance comparable NEE measurements with chambers.

Conclusion

Perspectives for application in complex trace gas exchange studies

Trace gas exchange studies normally must look at a lot of different processes and scales at the same time. For every research question a specific technique seems to be appropriate. Because the two most widely-used techniques, like eddy covariance and chambers, show contrasting limitations and benefits, the focus in future studies should be to combine these two approaches. This chamber may therefore allow sampling data complementary to eddy covariance measurements under difficult terrain conditions, during periods lacking turbulence (night-time) and under small-scale mosaic ecosystem patterns. As well, if methane and nitrous oxide are included in the set of sampled gases. In the last three years, this chamber was used for these purposes in the European research project CARBOMONT, which focused on NEE-measurements in mountain grasslands partially in sites too steep to allow the application of the eddy covariance systems.

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INDICATION OF A DISTURBANCE GRADIENT FROM NATURAL TO DEGRADED TO RESTORED BOG ECOSYSTEMS BY VEGETATION AND SITE-FACTORS

Summary

Ecosystems are intricately involved in the global biogeochemical cycles of matter exchange. The assessment of the role of ecosystems as possible sinks or sources of climatically relevant trace gases is of importance in climate change research and has recently had increasing policy implications (Kyoto-Protocol). Site selection is crucial for the validity of the measurements. The primary goal in site selection was a representation of the major bog-ecosystem types, together with a complete disturbance gradient from natural, to degraded, to restored sites. A total of 39 plots were identified and grouped as 12 different sites in the field, which represented a complete disturbance gradient. Site selections were made based on expert field assessments of vegetation composition, structure and micro-topography as indicators for site conditions. Additionally, information regarding land use history and management and a comprehensive vegetation map of the area were considered during site selection. However, neither a detailed quantitative vegetation assessment nor an in-depth analysis of site factors, like water-table or nutrient status, could be conducted before site selection had to be decided. Therefore the set of these predetermined sites represents the best estimate of a representative selection of bog ecosystems, oriented to a complete coverage of a disturbance gradient. These selected plots were however analysed in a post-hoc approach in terms of vegetation composition and site-factors, to assess the results of the field based selection. Detrended correspondence analysis (DCA) revealed a clear separation of the bog species of the plots. The plots could be separated via a detrended correspondence analysis on the basis of species composition, but several sites exhibited a spreading picture. A comparative approach using cluster analysis of species data and site-factor data was able to explain the pattern and diversity of the predetermined sites. Species data were analysed together with site-factors in a canonical correspondence analysis (CCA), which revealed a clear differentiation for almost all of the selected sites, closely following an expected disturbance gradient. A conceptual development scheme finally illustrated the link between the sites via management and time or succession effects. A field-based approach for site selection, based on vegetation composition and additional site information proved to be effective, and allowed for a realistic representation of an underlying disturbance gradient.

Introduction

Trace gas exchange of ecosystems has been studied in international clusters such as CARBOEUROPE, in order to assess the role of the biosphere in the global carbon cycle and in global climate change. Of particular interest is the determination of possible sinks for and sources of greenhouse gases (GHG). These GHG-fluxes and balances depend primarily on the unique composition and status of the studied ecosystems, together with climatic factors like temperature and radiation, which are drivers in the functioning of the system. Hence, the selection of study sites is of major importance for the measured fluxes and balances. Site selection should be guided principally by a site's ability to represent a specific ecosystem type or a regional and/or thematic gap (e.g. a poorly studied ecosystem type). Logistical criteria, such as site access are also very important. On-site conditions determine the appropriateness of the measurement techniques; for example, the need for wide flat terrain for eddy covariance measurements.

The majority of studies on trace gas exchange of bog ecosystems have been reported on northern areas (e.g., Roulet et al., 1993, Silvola et al., 1996a, Waddington et al. 1998, Nykänen et al., 1998, Tuittila et al., 1999, Lafleur et al., 2001, Christensen et al. 2004). Typically more than one ecosystem type was studied using a comparative approach. The selection of the studied ecosystems was generally driven by vegetation composition and additional information like management and land history (Alm et al., 1997, Bellisario et al., 1998, Bubier et al., 1998, Tenhunen et al., 1995, Tuittila et al., 1999). The differentiated ecosystem types were also described based on selected site factors. These site factors, like water-table and nutrient status, are functionally linked to trace gas metabolism and transport, especially in wetlands. Therefore they can notably influence the level and dynamics of trace gas exchange. However, a comparison between a vegetation based separation of the ecosystems and a grouping based on the underlying site factors was not shown in these studies (e.g. Waddington et al., 2001).

The dominating site factors correlated with bog development are a high water table, a wet but not too cold climate and a scarcity of nutrients (Ellenberg, 1986). Classification systems recognize the specific role of the water as a factor for classifying wetlands, like the hydrogenetic mire types from Succow & Joosten (2001). Bogs are classified as rain-fed peat-forming wetlands, characterised by a high water level in the peat and formed by nutrient poor rainwater. The nutrient input into the system is principally driven by rainwater and dry deposition, which were the preconditions for establishing nutrient poor growing conditions by specific ombrothrophic bog-peat forming vegetation types (Maltby & Proctor, 1996, Pfadenhauer, 1997). Bogs are therefore limited to climate zones which experience rainfall amounts that are higher than the evapotranspiration rate, such as the mire belt in the forelands of the Bavarian Alps of southern Germany (Maltby & Proctor, 1996). A disturbance is a change of vegetation or site-factors provoked by an external factor (Walker, 1996, White & Pickett, 1985). Disturbances in these bog ecosystems in the forelands of the Alps were mainly due to changes in the biophysical environment, caused by severe impacts like drainage or peat cutting. These disturbances were reflected by changes in the vegetation composition. Drainage as a preparation for, but without subsequent peat cutting, is by far the most widespread disturbance. Drainage impacts an estimated 70 % of the total bog cover in the forelands of the Bavarian Alps (50.000 ha; Drösler et. al in prep.), which ends up as a heathland dominated bog ecosystem. A reference status is needed to evaluate in a comparative approach the effects of disturbances on structure and functions of the system (Laska, 2001).

An overall goal of the study was, to assess the effect of disturbances on the trace gas exchange-function of southern German bog ecosystems. This goal was derived from the need to close a regional and thematic gap in trace gas exchange studies. Prior to this initiative, no trace gas exchange study focusing on net ecosystem exchange (NEE) and climatic relevance (GWP) simultaneously and comparing a complete set of natural, degraded and restored ecosystems had been undertaken in southern German bogs. The selection of the sites focused on a vegetation and structure based approach, guided by the criteria cited above. Vegetation composition was interpreted as indicator of site conditions, which were not available as measured site factors prior to site selection. After a 1.5 year field programme on trace gas exchange, measured data for a set of site factors were available. This data was used to evaluate and explain the vegetation based site selection via a post-hoc approach.

The principal question was, if ecosystem types in bogs, selected and differentiated via a vegetation-based field appraisal along an obvious structural disturbance gradient, could be confirmed in a more detailed post-hoc evaluation of vegetation composition and if they were sufficiently reflected by a differentiation of the underlying site factors.

The specific goals of this chapter are i) to differentiate the sites via vegetation composition, ii) to differentiate the sites via the site factors, iii) to compare the grouping of the sites via vegetation and micro-topography with the grouping via the site factors in terms of an underlying disturbance gradient, iv) to explain the vegetation composition of the sites via the site-factors and v) to sketch the developmental links via management activities and time between the sites.

Material and Methods

Study area

Bogs are among the few ecosystem types in southern Germany, which still represent sites with nearly natural conditions. The bog area which was selected for the study (Kendlmühlfilze ca. 900 ha, southern Bavaria, Germany, 47°20’ N, 12°25’ E; see Fig. 3.1) can be generally divided into a northern part, with impacts of former land use that have left a mosaic of different disturbance levels, and a southern part, which represents an almost natural bog ecosystem complex in the forelands of the Alps.

Certain disturbances caused by farmers, like extensive grazing or the removal of the moss layer to be used in the stables, have already been documented in the historical record (Frankl, 1996). It was impossible to specifically define both the exact timing of these management activities and the precise areas of influence. It can be assumed however, that these low-intensity impacts were widespread over the study area, at least until the beginning of the 19th century. This generalized impact over all sites

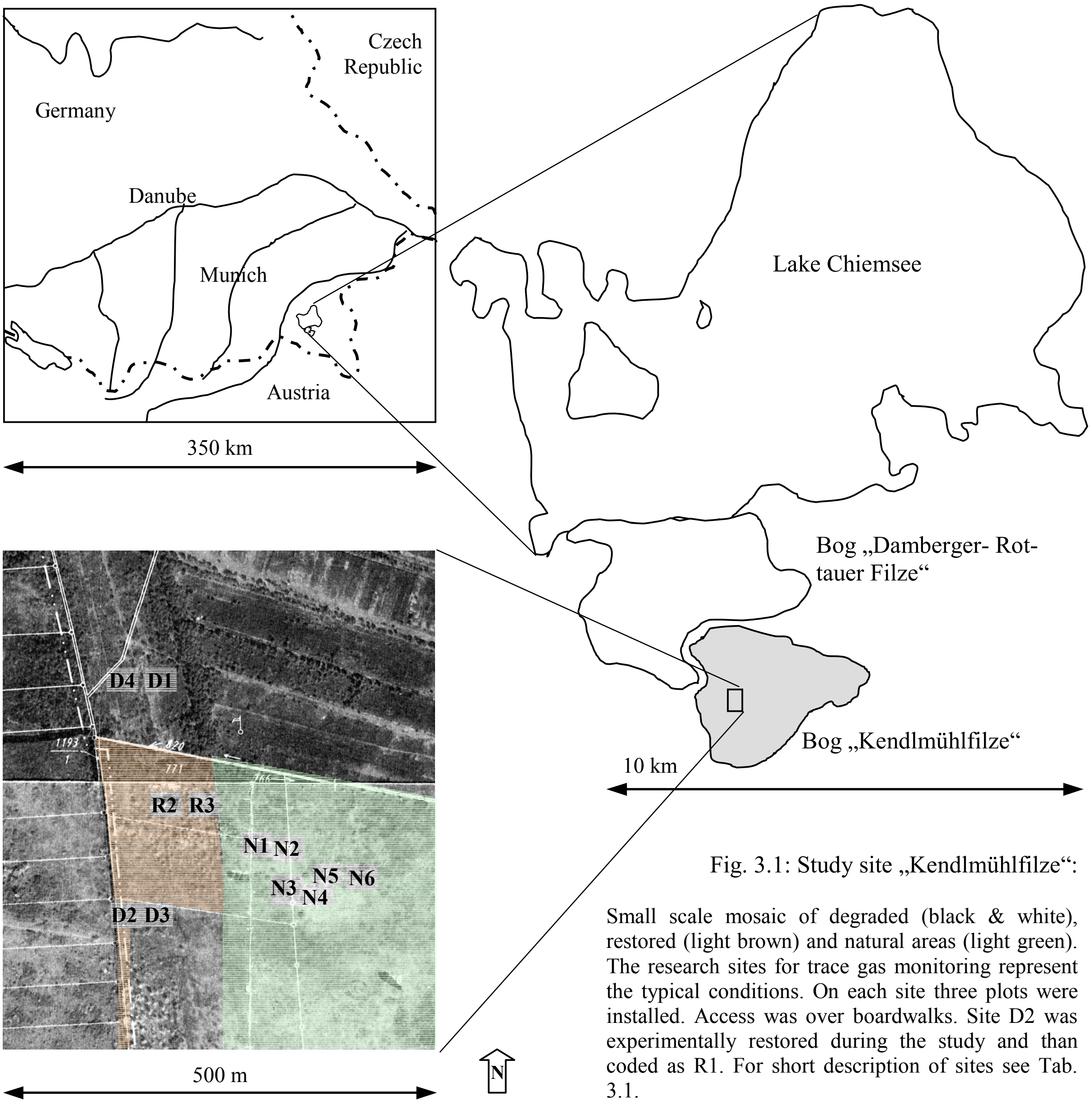

Fig. 3.1: Study site „Kendlmühlfilze“:

Small scale mosaic of degraded (black & white), restored (light brown) and natural areas (light green). The research sites for trace gas monitoring represent the typical conditions. On each site three plots were installed. Access was over boardwalks. Site D2 was experimentally restored during the study and than coded as R1. For short description of sites see Tab. 3.1.

does not allow for any of the sites to be considered as untouched. However, because these impacts were of very low intensity and ceased approximately a century ago, the most natural ecosystem types in the southern part of the study area, defined by vegetation composition and site factors, were taken as natural reference sites. Small-scale peat cutting by local farmers for heating purposes started in the middle of the 19th century on the fringe of the area. By the end of the 19th century, a major drainage system was already established and consequently the major impacts on the area started in 1895, with systematic exploitation of the northern part for peat cutting. Between 1976 and 1985 industrial peat cutting took place on part of this area (30 ha), and 1985 marked the endpoint of exploitation (Schmeidl, 1976, Sliva, 1997).

The individual study sites were selected to represent both the most common bog ecosystem types found in the mire belt in the forelands of the Bavarian Alps and a more or less complete gradient from natural to degraded to restored sites (Fig 3.1). As industrial and large scale peat cutting is of very limited importance for the entire bog area, these management-types were not included in the study.

site code	orig.site code	Plots	Status	short description
D1	2	4, 5, 6	degraded	dry *Calluna vulgaris* (L.) heathland, abandoned peat cut area (~50 years prior to the study)
D2	6d	16, 17, 18	degraded	dry *Calluna vulgaris* (L.) heathland, abandoned peat cut area (~50 years prior to the study); restored in the middle of the measurement period and thereafter coded as site R1
D3	5	13, 14, 15	degraded	dry *Calluna vulgaris* (L.) heathland, drained (~50 years prior to the study), but never peat cut (drained-only)
D4	1	1, 2, 3	degraded	dry *Calluna vulgaris* (L.) heathland with *Pinus sylvestris* (L.) and *Betula pubescens* (Ehrh.), drained (~50 years prior to the study), but never peat cut,
R1	6r	16, 17, 18	restored	flooded former peat cut dry *Calluna vulgaris* (L.) heathland (site D2), flooding during the measurement year (17th of July)
R2	4	10, 11, 12	restored	secondary moist *Calluna vulgaris* (L.) heathand, on former drained-only dry *Calluna vulgaris* (L.) heathlands (D3, D4), rewetted below surface ~ 10 year prior to the study
R3	3	7, 8, 9	restored	secondary *Sphagnum* (L.) lawn, on former peat cut dry *Calluna vulgaris* (L.) heathlands (D1, D2), rewetted till surface ~ 10 year prior to the study
N1	7/8	19, 20, 21, 22, 23, 24	natural	moist *Calluna vulgaris* (L.) heathland, due to microtopography and position on the slopes of the bog with a slight natural drainage effect
N2	9	25, 26, 27	natural	Bog shrubs (*Pinus mugo* (Turra)), due to microtopography and position on the slopes of the bog with a slight natural drainage effect
N3	11	31, 32, 34	natural	*Sphagnum* (L.) lawn
N4	12	33, 35, 36	natural	*Sphagnum* (L.) – *Eriophorum vaginatum* (L.) hummocks
N5	13	37, 38, 39	natural	transition between *Sphagnum* (L.) – *Eriophorum vaginatum* (L.) hummocks and *Sphagnum* (L.) lawns
N6	14	40, 41, 42	natural	*Sphagnum* (L.) – *Scheuchzeria palustris* (L.) hollows

Tab. 3.1: Summary description of the study sites. Originally the sites were coded with rising numbers from the entrance to the end of the research area. The renumbering was done, to express already the status of the sites with codes for degraded (D), restored (R) and natural (N) sites. The order within the subgroup “D” was decided along the similarity of the sites from peat cut to drained-only, in the subgroup “R” in terms of time since restoration and in subgroup “N” rising from the edge to the centre of the bog (parallel to the original rising numbering). For plot pictures see Annex-Fig. 3.1 of this chapter.

The widespread abandoned small-scale peat cut sites (D1, D2) of the local farmers and the sites with drainage in preparation for peat cutting (D3, D4) therefore represented the most degraded sites in the study area. Restoration was done via management of the water level, by flooding (R1) or rewetting to the surface (R2, R3). These sites received no further restoration management, such as the introduction of matrix species or liming and fertilization.

Aerial photographs (scale 1: 5.000) and a vegetation map (Pfadenhauer et al., 1990) were used as tools for site selection and field orientation. However, the final decision on the selection of a site was made in the field by using the vegetation composition as a primary indicator for the sites, along with visually obvious site factors, like micro-topography as an effect of former peat cutting activities. Furthermore, the best location of the sites was decided based on the placement of three sampling plots (75 x 75 cm) per site and the distance between and accessibility of the sites. For access to the sites, boardwalks had to be established. Therefore the location of the plots was decided in view of an economical design, in addition to plant composition and micro-topography. Access to the plots was always from the north and the plots were oriented north to south. An image of one plot at each site is illustrated in the annex of this chapter (Annex-Fig. 3.1). All other important site factors such as water-table, nutrients and pH could not be included in the site selection, as no detailed data were available for the sites. These factors were subsequently monitored together with the trace gas-exchange over 1.5 years after site installation.

The sites were each composed of three plots (site N1 was an exception with six plots), which were grouped together (see Tab. 3.1). The criteria for the selection of the plots and the grouping of sites were principally homogeneity and their representation of the site. The plots composing single sites with diverse micro-ecosystems, like the restored site R2 and in part D1 and D4, were less homogeneous but offered the best representation of the range of conditions of the site. For the rest of the sites, homogeneity was the primary criteria for plot selection.

Climate

The climate is cool-temperate with an annual mean temperature of 8.3 °C and 1483 mm of precipitation annually. Snow cover is common from mid-December until late February. In 1999, the annual average temperature was >1°C warmer than the long-term (30 years) mean annual temperature. The

month	01	02	03	04	05	06	07	08	09	10	11	12	Mean
temp 1999	0.6	-0.5	6	9.3	15.2	16.1	18.9	17.9	16.5	9.5	1.9	1.0	9.4
temp 30a	-1.4	0.1	4	8	12.8	15.9	17.9	17.1	13.8	8.9	3.5	-0.5	8.3
precip 1999	63.8	182.1	96.2	121.2	228.9	139.2	161.1	118.6	91.3	49.6	133.7	128.8	1514.5
precip 30a	95.9	77.2	91.1	117.9	151.7	185.9	176.1	174.6	113.8	94.4	103.5	101.5	1483.5

Tab. 3.2: Climate data for 1999 in comparison to the mean (30a) climatic conditions (data from the nearest weather stations to the measurement area, Bernau and Rosenheim)

precipitation regime differed only slightly from the long-term average. Additionally to the permanent weather stations in the surroundings of the study area, an individual weather station was established in the centre of the sites and was run from 1999 until the middle of 2000. This individual weather station was necessary to get specific site data, especially for radiation and soil temperature. The measured parameters were air temperature (T_{air}) and relative humidity (RH) at 10 cm and 2 m (Thermohygrograph), soil temperature at depths of 2, 5, 10, 20, 50 and 100 cm (Sensor: Siemens thermistor) and PAR (photosynthetic active radiation, Licor SA-190). Precipitation was recorded as a weekly sum with a totalisator.

Vegetation analysis

Vegetation sampling was conducted once in June, during the measurement period and at a stage of full vegetation development. As a result, vegetation data for site inter-comparison represented the same time window. Phenology typically was not pronounced at the sites, which is a normal feature in nutrient poor bog ecosystems. Therefore, the single sample data should represent a good base for comparing the vegetation characteristics of the sites. The gradient of the sites from natural, to degraded, to restored, offered the opportunity to interpret the vegetation data in relation to the time series, as the time since degradation (drainage and peat-cutting) and restoration (blocking the ditches, flooding) was known. This one-time vegetation assessment could be used to analyse the effects of the measures on the vegetation development, without directly following successional processes over several years, as was done in other studies that focused mainly on vegetation development after restoration with various methods (Sliva, 1997, Tuittila, 2000).

Vegetation cover and composition

Vegetation was sampled on every plot by placing a 100-cell grid over the plot. Vegetation cover was assessed in 1% steps and, if necessary, grouped on the 10 % Londo scale (Londo, 1976). This detailed procedure for sampling vegetation was useful for post-processing with multivariate methods. The grid was constructed of a thin rope, fixed crosswise on a wooden frame of the same dimensions as the sample plots (quadratic 0.56 m^2). The frame could be adjusted in height above the plot, to cope with pronounced micro-topography, as was found on *Eriophorum vaginatum* (L.) and *Sphagnum* (L.) – *Calluna vulgaris* (L.) hummocks. The plot dimensions allowed for viewing from directly above the plot, thus the varied height of the grid over the vegetation surface did not influence the cover estimates. Moreover, detailed sampling of the micro-topography was done by measuring from every point of the rope crossings down to the vegetation or soil surface. These data were used for describing the plot micro-topography as well as for determining the plus or minus volume of the plot over the soil frame, which was an important piece of input data for the trace gas flux calculations (see chapters 4 & 5).

Leaf area index (LAI)

The leaf area index of green leaves was measured by a leaf area meter (Li-3000; Licor). LAI was determined once (July) during the season, as the dynamic over the course of the year is very low in bog-ecosystems. For mosses, the single green leaves were picked off the capitula and separately measured; otherwise, the complex architecture would have led to underestimations of the LAI.

Aerenchymous plants

The presence of plants with aerenchymous tissue was recorded on every plot, since some literature implies that there is a functional link between these vascular plants and methane emission rates (Joabsson et al., 1999, Joabsson & Christensen, 2001). It is unclear which part of the plants should best represent the plants function as chimney for methane; the quantity of roots below the water-table, the surface area of the roots, or the amount of upper plant parts as shoot or leaves. It is most probable that the root surface, together with the diameter of the aerenchyma in the roots, are the best functional parameters for estimating the efficiency of the vegetation's methane emission. As the root surface area could not be determined in a non-destructive way, the quantity and the average cross-cut surface area (cut at the leaf base) of leaves were sampled as indicators for the total bypassing surface. The two species with aerenchyma on the studied plots were *Eriophorum vaginatum* (L.) and *Scheuchzeria palustris* (L.). There was no specific information indicating that the gas diffusion rate through the aerenchyma of the two species varied. Therefore, the relation of the cross-cut surface of the two species to transform the *Scheuchzeria palustris* (L.) leaves (1.5 to 3 times the surface of *Eriophorum vaginatum* (L.)) to a *Eriophorum vaginatum* (L.) leaf standard, was used as an indirect expression of the total leaf cross-cut surface. This was done because the simple counting of leaves was much more feasible for field sampling, than a determination of the sum of the surface areas, and the destructive method of directly determining the total root surface area was not possible because of the monitoring program.

Peat cores

Peat coring was done on every site using the Edelmann-digger, digging out half section cores of 5 cm diameter and 50 cm length. The peat profile was described in layers by individual peat type, decomposition degree H after v. Post (Grosse-Braukmann, 1990), colour and plant remnants. Peat cores were taken down to the mineral horizon below the peat. Peat samples were taken from the corresponding profile steps for further laboratory analyses on C and N content (Elemental analyser, VARIO-EL, Elementar, Hanau) to determine the C/N ratio. The C/N-relation is a common indicator of the trophic and disturbance status of the peat and is a very stable parameter. A single C/N value for the corresponding layer of the entire profile was determined. The mean C/N-ratio, as well as the mean H-value, for the profile was calculated as a weighted average for the acrotelm, defined by the minimum recorded water table. This section of the peat column is of major importance for the production and decomposition processes and as an indicator of the disturbance intensity of a site.

Water-table

Every plot was installed with a well (4 cm PVC-tube) at a constant position of 10 cm NW of the soil frames. Water table measurements were taken instantly during gas-exchange measurements every week. These data were sampled for the differentiation between the sites and for the assessment of site-specific water-table dynamics, as well as to test the strength of water-table conditions as a driving force for gas-exchange behaviour.

Water-chemistry

Within the wells, pH (with WTW pH 196 sensor) and Electric Conductivity (with WTW LF 191) were measured instantly during the gas measurements over the whole measurement period. For further determination of the nutrient status of the sites, Ca^{++}- and K^{+}-contents were calculated from water samples three times a year with a flame photometer (EELEX 6361, Eppendorf).

Statistical analysis

Normality of the distribution was tested with STATISTICA software, and when necessary log or root transformation was used. After transformation, normality could be achieved for any variable, so parametric procedures were applied for further analysis. Multivariate statistics, using PC-ORD, were applied to explain gradients in site factors and to find relationships between site factors and vegetation composition. First, a detrended correspondence analysis (DCA) was used to indirectly explore the composition of the vegetation, comparing the individual plots. DCA was applied because the gradient length was 2.8. For comparing the grouping of the sites, a cluster analysis (Euclidean, Ward) was applied both to the species data and to the site factor data. Finally, as a direct ordination technique, a canonical correspondence analysis was applied to the vegetation (primary matrix) and site factors (secondary matrix). CCA was used to clarify the relationships between the selected site factors and the vegetation composition of the plots.

Results

Vegetation

Plot and species ordination

The total species number over all sites was limited to 13 vascular plants and 6 moss species (liver-mosses and lichens were not determined), which is typical for bog ecosystems. Cover percentages for the species within the plots are illustrated in Annex-Tab. 3.1. A detrended correspondence analysis (DCA) was used to inspect the relationships between species composition and cover for the studied plots. The dataset exhibited a unimodal behaviour (gradient 2.8 of the first axis). Total variance in the species data was 2.240. The eigenvalues for the first axis were 0.718, for the second axis were 0.223 and for the third axis were 0.109 (see Fig. 3.2).

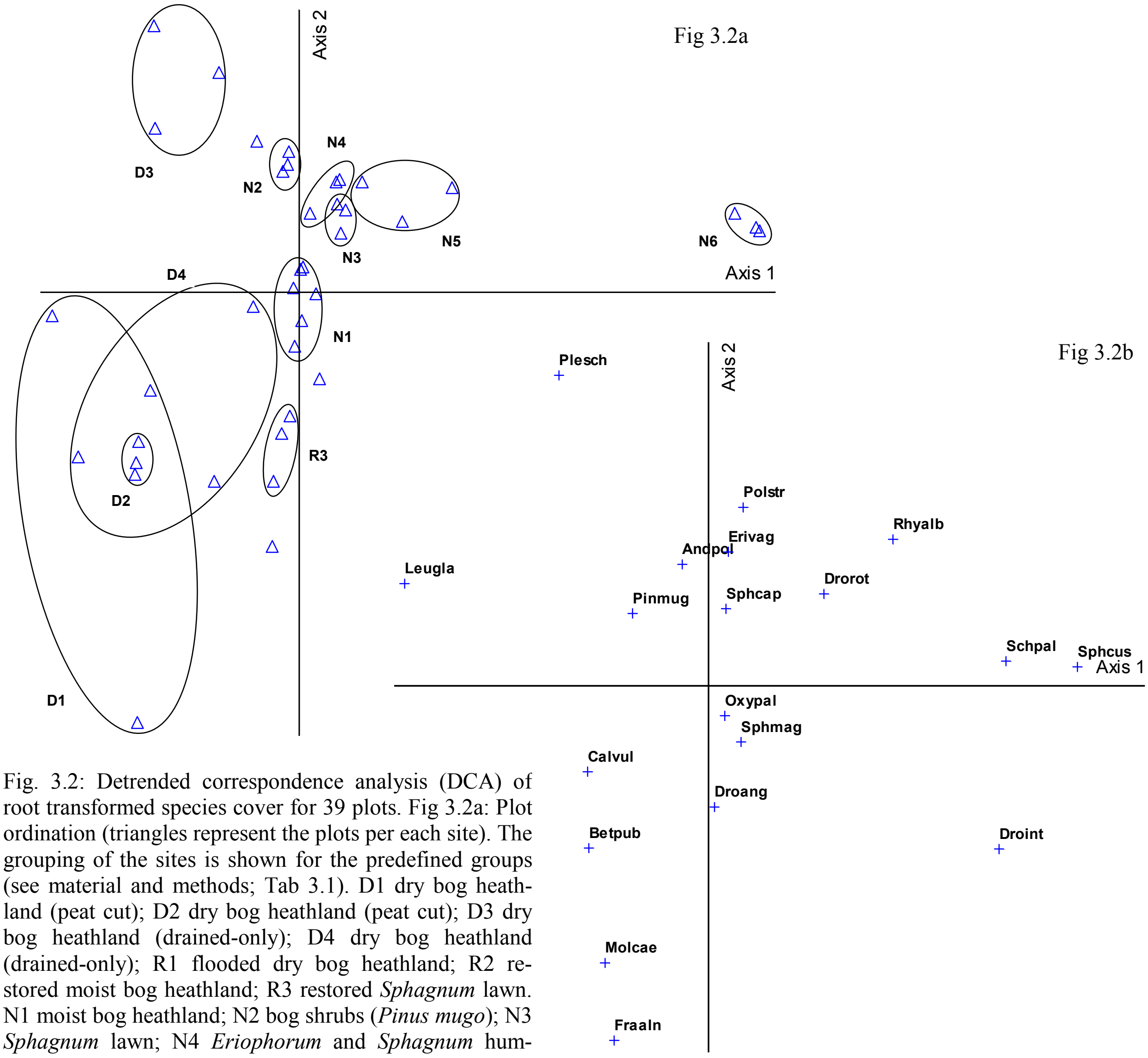

Fig. 3.2: Detrended correspondence analysis (DCA) of root transformed species cover for 39 plots. Fig 3.2a: Plot ordination (triangles represent the plots per each site). The grouping of the sites is shown for the predefined groups (see material and methods; Tab 3.1). D1 dry bog heathland (peat cut); D2 dry bog heathland (peat cut); D3 dry bog heathland (drained-only); D4 dry bog heathland (drained-only); R1 flooded dry bog heathland; R2 restored moist bog heathland; R3 restored *Sphagnum* lawn. N1 moist bog heathland; N2 bog shrubs (*Pinus mugo*); N3 *Sphagnum* lawn; N4 *Eriophorum* and *Sphagnum* hummocks; N5 transition hummocks – hollows; N6 *Sphagnum* hollow. R2 is represented in the DCA by the three isolated single triangles along axis 2.
Fig 3.2b: Species ordination.

Plot ordination revealed a differentiated separation between the sites (Fig. 3.2a). The natural sites (N1-N6) were positioned from the centre to the right, primarily above the first axis. The degraded sites (D1-D4) were located on the opposite side of the second axis, but with a wider span both above and below the first axis. The restored sites were located in between, with site R2 consisting of three single plots spread along the second axis. The isolated position of N6, underlined the unique vegetation composition with almost no overlap to other sites; *Sphagnum cuspidatum* Hoffm. and *Scheuzeria palustris* L. were the dominating species. Both species were similarly separate at the right margin of the species ordination (Fig. 3.2b).

The positions of the plots in relation to the site-grouping (Fig 3.2a) generally showed that the natural sites were more homogenous than the degraded ones. The plots of sites N2 (natural bog shrubs), N3 (*Sphagnum* lawn) and N6 (*Sphagnum* hollow) were very close together, followed by N4 (*Sphagnum* – *Eriophorum* hummocks) and N1 (moist *Calluna* heathland). This confirmed the homogeneity of the plots, one of the principal criteria in the selection of these plots for the corresponding sites. The restored *Sphagnum* lawn (R3) also presented a homogeneous picture, with the plots being close together. Finally, site D2 (dry *Calluna* heathland, former peat cut) was selected under the homogeneity criteria, which is reflected in the close positioning of the composing plots.

In more heterogeneous ecosystems, especially on the degraded sites, it was necessary to guide the selection by how representative the plots were for the range of micro-sites. Therefore, several sites demonstrated a more diverse pattern in terms of the location of plots in the DCA (Fig 3.2a). This is reflected by sites N5 (transition between hummocks and hollows), D1 (degraded dry *Calluna* heathland, former peat cut), D3 (degraded dry *Calluna* heathland), D4 (degraded dry *Calluna* heathland, with *Pinus* and *Betula* shrubs) and finally site R2 (restored moist *Calluna* heathland). The differentiated grouping in the DCA plot was in concordance with the applied selection criteria of the composing plots for the sites.

Looking to the species ordination, (Fig. 3.2b) *Scheuchzeria palustris* (L.) and *Sphagnum cuspidatum* (Hoffm.) appeared to behave very similarly and were clearly separate as dominant species on the hollow plots (N6), with the water-table at the surface. The group of *Sphagnum magellanicum* (Brid.), *Oxycoccus palustris* (Pers.) and *Drosera anglica* (Huds.) represented the dominating vegetation on the *Sphagnum* (L.) lawns (N3). The group of *Calluna vulgaris* (L.), *Betula pubescens* (Ehrh.), *Molinia cearulea* (L.) and *Frangula alnus* (Mill.), which appeared with varied composition and changing coverage on all but the wettest natural sites, extended along Axis 2. Above the center of the biplot, *Pinus mugo* (Turra), *Andromeda polifolia* (L.), *Sphagnum capillifolium* (Ehrh.), *Eriophorum vaginatum* (L.), *Polytrichum strictum* (Menz.) and *Drosera rotundifolia* (L.) composed the largest and most diverse group. These species were most abundant as part of the natural bog vegetation on hummocky sites. *Rhynchospora alba* (L.) linked towards the hollow species. The two drought tolerant mosses, *Leucobrium glaucum* (Hedw.) and *Pleurozium schreberi* (Brid.), were separate at the left margin of the biplot (Fig. 3.2b) and were found exclusively on the drier degraded sites (D1, D3) (see Annex-Fig.

3.1). The separation of the species reflected the underlying gradients caused by site-factors. The agreement of these indicator groups with those of Pfadenhauer et al. (1990) will be analysed together with the results of the site-factor ranges in the discussion chapter.

Leaf area index (LAI)

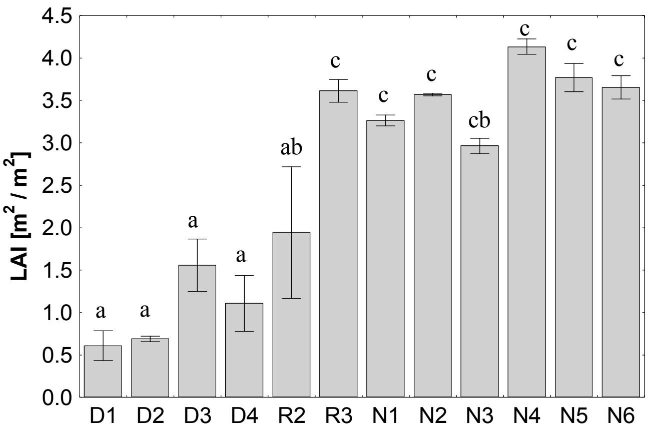

Fig. 3.3: Leaf area index (LAI) of the sites (mean; error bars are SE). For site codes see Tab 3.1. Different letters indicate significant difference according to ANOVA, Tukey-test ($p<0.05$), but n=3 limited differentiation strength.

Mean site values were between 0.61 ± 0.17 on the degraded site D1 and 4.13 ± 0.09 on the natural site N4 (see Fig. 3.3). The LAI was generally low on the degraded sites and rose steeply on the restored and natural sites. The natural sites had an average LAI level of 3.5. Here especially the mosses were responsible for higher LAI. This effect can be identified by the changing LAI from R2 to R3, as the moss cover on site R3 was rising to levels similar to the natural sites.

Aerenchymous leaves

The plants with aerenchymous tissue, (*Eriophorum vaginatum* (L.) and *Scheuchzeria palustris* (L.)), appeared when mean water-table depths did not drop below approximately 15 cm. Consequently, aerenchymous plants were absent on the most degraded sites within the study area. The maximum leaf quantity, with 2071 leaves per m^2 (calculated as *Eriophorum vaginatum* (L.) standard, see materials & methods), was reached at the natural *Sphagnum* (L.) hollow (site N6). The quantity of aerenchymous leaves rose in a fairly exponential fashion along the gradient from degraded, to restored, to natural sites (see Fig. 3.4). Only the natural site N3 (229 leaves per m^2) and the restored site R2 (437 leaves per m^2) did not conform to this shape. The restored moist *Calluna vulgaris* (L.) heathland (site R2) was characterised by the inclusion of one plot dominated by *Eriophorum vaginatum* (L.), with the rational being that the restored sites in particular still experience a variable mix of different vegetation types. The large amount of aerenchymous leaves on R2 was a product of the domination of this plot by *E.vaginatum*. Natural *Sphagnum* (L.) lawns (site N3) are contrastingly characterised as hosting a limited number of aerenchymous plants, due to vegetation type composition.

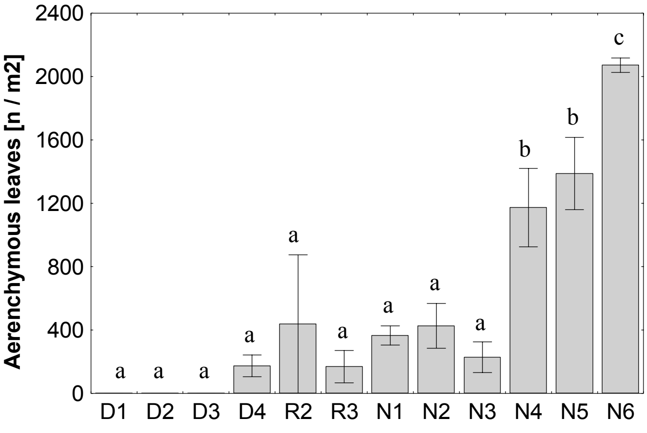

Fig. 3.4: Quantity of Aerenchymous leaves (Eriophorum vaginatum (L.) standard, within the sites (mean, error bars are SE). Site codes see Tab 3.1. Different letters indicate significant difference according to ANOVA, Tukey-test ($p<0.05$), but n=3 limited differentiation strength.

Water-table (WT)

Lowest mean WT and largest range of depths were found on the degraded site D1 (dry bog heathland, former peat cut), with –29 cm mean WT and 54 cm difference between the maximum and minimum WT. Mean WT at the natural *Sphagnum* (L.) hollow (N6) was recorded as 0 cm, as the floating vegetation layer (mainly *Sphagnum cuspidatum* (Hoffm.) and *Scheuchzeria palustris* (L.)) was moving with the water-table fluctuations. The restored site R2 (secondary moist *Calluna vulgaris* (L.) heathland) was in between with a mean WT of –11.7 cm and a fluctuation range of 23 cm max-min WT (see Tab 3.3). The differences in the dynamic behaviour over the course of the year cannot be described by mean WT and total fluctuation (max-min WT) alone (Tab. 3.3). The year-course of weekly WT is shown in Fig. 3.5. The driest site D1 demonstrated a dynamic pattern over the course of the year, with pronounced reactions to precipitation. The wet site N6 on the other hand exhibited a smooth dynamic, oscillating around the zero-line. Rainfall, however, could still be identified in the dynamic of the WT course. Summer drought during weeks 27, 31 and 37 led only to a smooth decline in the WT of N6, in contrast to the strong reaction of site D1. The WT dynamics of the degraded sites D1 and D4 was high during the first half of the year (till week 26) and continued with even more variation toward the end of the year. Site R3 was characterized by very distinctive behaviour between the first and second half of the year, with very small changes until week 26 and sharp oscillations during the second half. This was an intermediate course for the year, falling between the degraded sites D1 and D4 and the natural sites N3 to N6. However, sites R2, N1 and N2 experienced similar patterns of WT-year course, but the difference between the first and second half of the year was less notable. According to the ANOVA analysis, R2 and D3 did not differ significantly in the mean of the WT (Tab. 3.3), but demonstrated different behaviour when the first and second half of the year were compared (Fig. 3.5).

	WT [cm]		EC [µS cm^{-1}]		pH	
	Mean ± SE	max-min	mean ± SE	max-min	mean ± SE	Max-min
D1	-29.0 ± 0.9 a	54	68.3 ± 2.7 a	137	4.03 ± 0.03 a	1.43
D2	-20.2 ± 0.9 b	36	55.3 ± 1.4 bc	51	3.97 ± 0.05 a	1.23
D3	-11.6 ± 0.7 c	37	60.0 ± 2.5 c	136	4.14 ± 0.03 ab	2.46
D4	-17.0 ± 0.6 b	32.5	51.1 ± 1.4 bd	98	4.09 ± 0.02 ac	1.12
R1	44.5 ± 0.8 d	24	49.4 ± 1.1 cd	21	3.95 ± 0.04 ad	0.58
R2	-11.7 ± 0.4 c	23	51.1 ± 2.0 bd	100	4.37 ± 0.03 e	1.89
R3	-5.3 ± 0.7 f	37	37.0 ± 2.2 eg	128	4.57 ± 0.03 g	1.07
N1	-8.2 ± 0.3 gh	20.5	40.5 ± 1.0 e	54	4.20 ± 0.03 bc	1.72
N2	-8.4 ± 0.4 gh	24.5	39.0 ± 1.3 e	81	4.35 ± 0.03 ef	1.65
N3	-6.3 ± 0.3 fg	17	29.0 ± 0.7 f	47	4.26 ± 0.03 eb	1.69
N4	-9.5 ± 0.4 ch	27	29.6 ± 0.6 f	32	4.25 ± 0.03 eb	1.65
N5	-3.8 ± 0.4 f	28.5	30.0 ± 0.6 fg	33	4.23 ± 0.03 bf	2.32
N6	0.00 ± 0.3 i	17.5	31.0 ± 0.7 fg	37	4.18 ± 0.03 bcd	1.88

Tab. 3.3: Site factors of water-table (WT), electrical conductivity (EC) and pH-value for all sites. Data represent means over the whole measurement period (n between 60 and 140 depending on variable and site). Different letters indicate significant differences (ANOVA, Tukey test, $p<0.05$), but should be interpreted with caution, due to dynamic behaviour over the year (see corresponding paragraphs). For site codes see Tab 3.1.

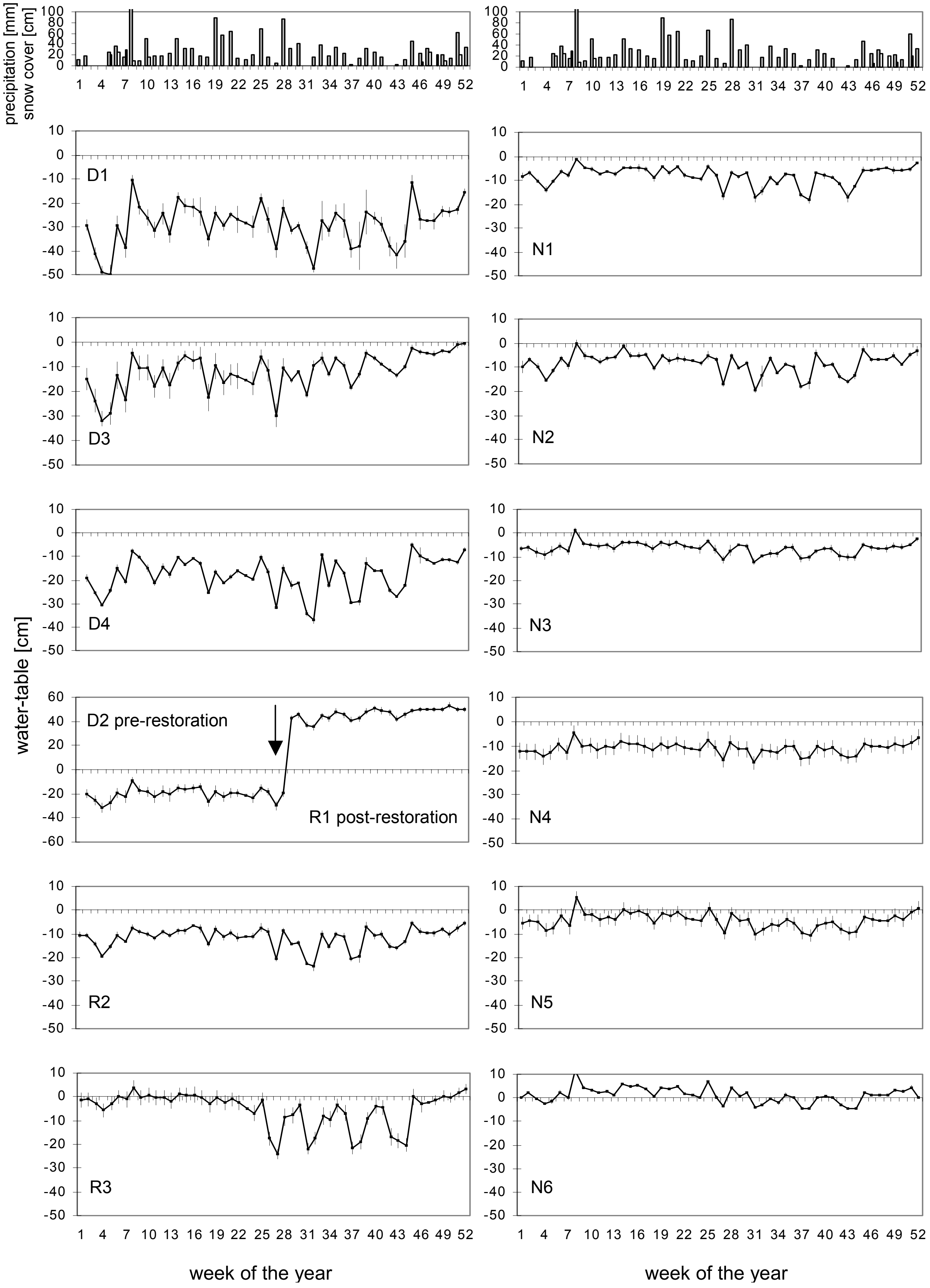

Fig. 3.5: Year courses of weekly water-table depth for all sites. Data represent means, error bars are SE. Precipitation is plotted as sum of the week, snow height as actual height at the weekly measurement day. The arrow at site D2/R1 marks the restoration activity at 17th of July. For site codes see Tab 3.1.

Electrical Conductivity (EC)

The means of electrical conductivity (EC) measurements showed a clear trend from higher to lower values along a gradient from degraded, to restored, to natural sites (Tab. 3.3). Maximum means were recorded on the degraded former peat cut site D1 with 68.3 ± 2.7 µS cm^{-1}. Low values were obtained on the natural sites, with a minimum mean of 29.0 ± 0.7 µS cm^{-1} for the natural *Sphagnum* (L.) lawn (site N3). Generally the restored sites fell in between. However, the natural sites N1 (moist *Calluna vulgaris* (L.) heathland), with a mean of 40.5 ± 1.0 µS cm^{-1}, and N2 (natural bog shrubs), with a mean of 39.0 ± 1.3 µS cm^{-1}, were significantly separated from the rest of the natural sites (N3 to N4) and were even higher than the restored *Sphagnum* (L.) lawn (site R3; 37.0 ± 2.2 µS cm^{-1}) (see Tab. 3.3).

It is obvious when looking at the dynamics over the year (Fig. 3.6) that EC tends to increase over the year until fall and then drops relatively sharply, ending up close to starting levels. The difference between the sites was noticeable both in the range and the level of the dynamic. Sites D1 and D3 reached the highest end of summer and early fall levels, with means greater than 100 µS cm^{-1} (Fig. 3.6) and were characterised by the widest range between maximum and minimum levels, which were up to 137 µS cm^{-1} (Tab. 3.3). The group of the natural sites N3 to N6 experienced very similar patterns of slowly rising levels over the year and a limited range between maximum and minimum values. The effects of restoration in the middle of the measurement year were clearly observable in the year course of site D2/R1 (see Fig. 3.6): Until restoration, site D2 behaved similarly to site D1 with an oscillating and rising curve. Restoration led to a sudden drop and the dynamic was reduced. The curve with the most variable pattern was obtained on site R3 (restored *Sphagnum* (L.) lawn), with a low-level baseline but very distinct individual values jumping out of the mean course of the curve. These maximum levels fell significantly during periods of low water table (see Fig. 3.5, box R3).

pH

All mean pH-values fell in a relatively small range between 3.95 (site R1) and 4.57 (site R3), which is typical for nutrient poor bog ecosystems (see Tab. 3.3). The pH-values did not experience a similarly pronounced dynamic over the year as water-table (WT) or electric conductivity (EC). In contrast to WT and EC, no simple gradient from degraded, to restored, to natural sites could be found. The lowest values were detected in the group of the degraded sites (D1-D4; 3.97 ± 0.05 to 4.14 ± 0.03), together with the restored site R1 (3.95 ± 0.04). The maximum values were obtained on the restored sites R2 (4.37 ± 0.03) and R3 (4.57 ± 0.03). The natural sites N1 to N6 behaved relatively uniformly, with pH values in between those for the degraded and restored sites, with 4.18 ± 0.03 to 4.35 ± 0.03.

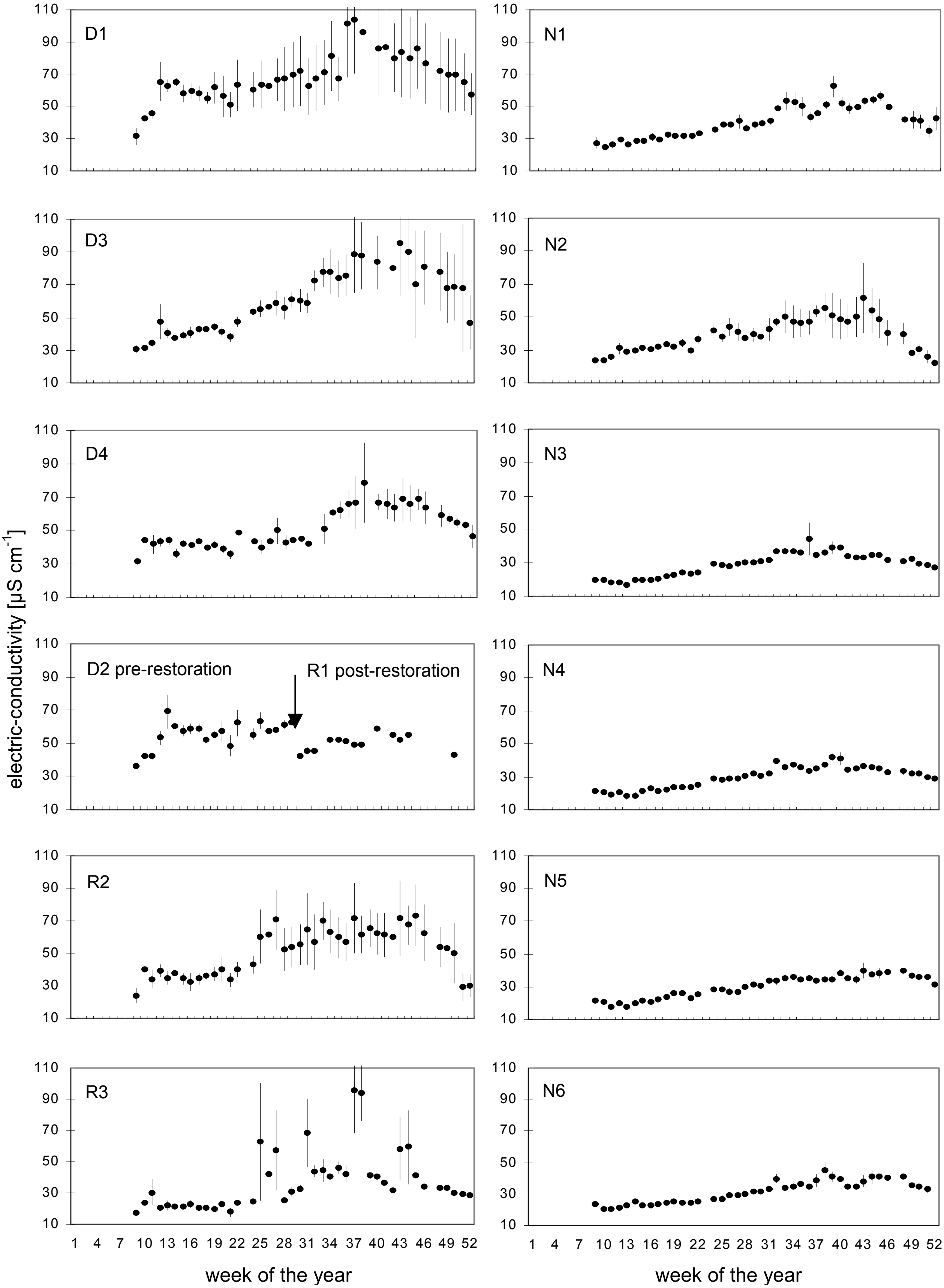

Fig. 3.6: Year courses of weekly electrical conductivity for all sites. Data represent means, error bars are SE. The arrow at site D2/R1 marks the restoration activity at 17th of July. For site codes see Tab 3.1.

C/N- and H-value

The C/N ratios were between 25.9 (site D2) and 40.5 (site N6), generally indicating a nutrient poor bog peat. The comparison of the sites revealed a clear separation between the natural sites N3 to N6 with the largest C/N ratios (39.1 to 40.5) and the rest of the sites (see Tab. 3.4), including the natural sites N2 (32.0) and N1 (28.0). Even in the group of the degraded sites, C/N values up to 30.3 (site D3) were obtained.

The H-values (v. Post method; Grosse-Brauckmann 1990) were calculated as means for the same section of the profile (acrotelm) as the C/N-values (Tab. 3.4). In this study, the most degraded former peat cut sites (D1 and D2) had the highest values, with 6.4 and 7.1 respectively. A diverse group showed a reduced span between 3.3 and 3.8 (D3, D4, R2, N1, N2). Site R3, which was restored after formerly being cut for peat had an intermediate position (H value 4.6), compared to the two former groups. The lowest H values of 1.0 to 1.6 were recorded on the natural sites N3 to N6. The peat cut sites (D1, D2) and the former peat cut site (R3), which was restored 10 years prior to the study, could be identified by higher H-values. On the other hand, the almost completely undecomposed peat on the sites with high WT (N3-N6) was also clearly identifiable.

	C/N	H
D1	29.0	6.4
D2/R1	25.9	7.1
D3	30.3	3.6
D4	27.2	3.8
R2	26.3	3.3
R3	28.0	4.6
N1	28.0	3.5
N2	32.0	3.4
N3	39.1	1.6
N4	39.1	1.6
N5	39.8	1.4
N6	40.5	1.0

Tab. 3.4: C/N and H-values. Means from the acrotelm. Profiling was done once at site D2 and represents R1 as well. For site codes see Tab 3.1

Calcium and Kalium

The Ca^{++}- and K^{+}- values should be interpreted with caution, as only three replicates at three different times during the year were sampled per site. This is in accordance with the approach of others (e.g. Sliva 1997), but the wide ranges (max-min) were a hint of higher year course dynamics, which could not be sufficiently tracked with this reduced program. However, as this was a common approach and literature data were comparable and have a similar time resolution, these data were presented as estimates for the relation between the sites, but were not used in further steps of data-analysis (multivariate statistics). The highest mean Ca-values were found on the restored sites, with 12.76 ± 1.89 mg l^{-1} at site R2 and 15.17 ± 3.32 mg l^{-1} at site R3 (see Tab. 3.5). The Ca^{++}-values of the degraded sites ranged between 4.28 ± 0.76 mg l^{-1} (site D1) and 7.47 ± 2.60 mg l^{-1} (site D4), whereas the natural sites N2 to N6 had lower means between 3.72 ± 0.86 and 2.33 ± 0.36 mg Ca l^{-1} respectively.

	Ca^{++} [mg l^{-1}]		K^{+} [mg l^{-1}]	
	mean ± SE	max-min	mean ± SE	max-min
D1	4.28 ± 0.76 a	6.9	1.59 ± 0.17 a	1.55
D2	6.93 ± 1.31 ab	9.1	1.29 ± 0.11 ab	1.05
D3	6.10 ±1.04 ac	8.8	1.26 ± 0.09 ab	0.97
D4	7.47 ± 2.60 ad	25.8	1.16 ± 0.10 ab	0.93
R2	12.76 ± 1.89 bcde	14.1	1.54 ± 0.08 a	0.79
R3	15.17 ± 3.32 e	32.3	1.21 ± 0.18 ab	1.47
N1	4.56 ± 0.47 a	7	0.68 ± 0.08 cdf	1.45
N2	3.72 ± 0.86 a	8.1	0.96 ± 0.11 bc	1.06
N3	3.14 ± 0.56 a	4.3	0.28 ± 0.02 fe	0.16
N4	3.63 ± 0.56 a	5.0	0.19 ± 0.02 e	0.22
N5	2.33 ± 0.36 a	3.1	0.35 ± 0.05 de	0.45
N6	3.34 ± 0.48 a	3.9	0.48 ± 0.06 ce	0.52

Tab. 3.5: Ca- and K-values in the pore water. Means of three sampling dates (with three plots each site) over the year (n=9). Different letters indicate significant differences (ANOVA, Tukey test, $p<0.05$), but should be interpreted with caution, due to dynamic behaviour and reduced repetition over the year (see corresponding paragraphs). .For site codes see Tab 3.1 .

K^{+}-values on the other hand showed a trend, which more closely followed a disturbance gradient over the sites, already identified via WT- and EC-values, than did the Ca^{++}-values. The highest K^{+}-content was measured on the degraded site D1 (1.59 ± 0.17 mg l^{-1}), but was closely followed by the restored site R2 (1.54 ± 0.08 mg l^{-1}). The natural sites had the lowest values, between 0.96 ± 0.11 mg l^{-1} at site N1 to 0.19 ± 0.02 mg l^{-1} at site N4. In terms of largest range (maximum-minimum values), indicating the variability of the values within the sites, D1 (1.55 mg l^{-1}) was the greatest, followed by R3 (1.47 mg l^{-1}) and N1 (1.45 mg l^{-1}) (see Tab. 3.5).

Comparative Cluster Analysis of vegetation and site factors

Cluster analysis (Euclidean and Ward) was applied first to the dataset of vegetation cover and secondly to the site factors (see Fig. 3.7) for all individual plots. For the latter, the means per plot (39 in total) of the above presented site-factors; *water-table, electric conductivity, pH-value, C/N ratio* and *H-value* were used. The cluster analysis was done both for a post-hoc evaluation of the grouping of the sites, and for inspecting the indication value of vegetation composition for the ranges of important abiotic site factors via a comparison of the two results.

The vegetation based cluster diagram (Fig. 3.7a) generally revealed a separation between the group of the degraded sites (D1 to D4) together with restored site R2 and the natural sites (N1 to N5) together with restored site R3. The natural site N6 held an intermediate position, apparently because of a limited number of species and almost no overlap in species composition with other sites. In detail the natural sites seemed to be more homogeneous, as the grouping of sites N1, N2, N3, N5, N6 and of R3 as well, confirmed that the composing plots were almost exclusively the closest neighbours. Sites D2 and D3 behaved homogenously as well. A more diverse picture was detected for sites N4, D1, D4 and especially R2, where the plots were mixed in between neighbouring groups. On site D4, the plots were still close together, with one R2 plot (P11) and one D1 plot (P4) mixed in between. R2 on the other hand is spread over the graph, with two (separated) plots (P10, P11) in the upper group of the degraded sites and one (P12) in the lower group of the natural sites. This cluster analysis however, showed a satisfying coincidence between a grouping of the individual plots to the corresponding sites, and generally confirmed the grouping, which was decided prior to this detailed vegetation analysis on the basis of an exploratory approach in the field.

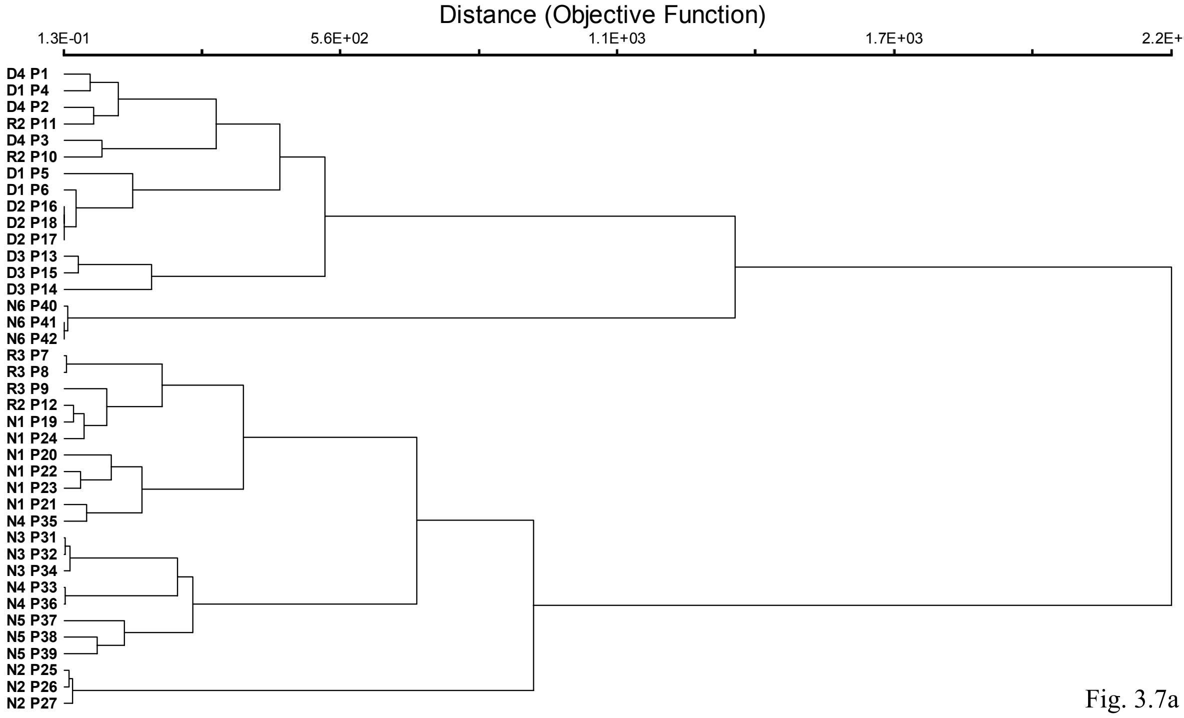

Fig. 3.7a

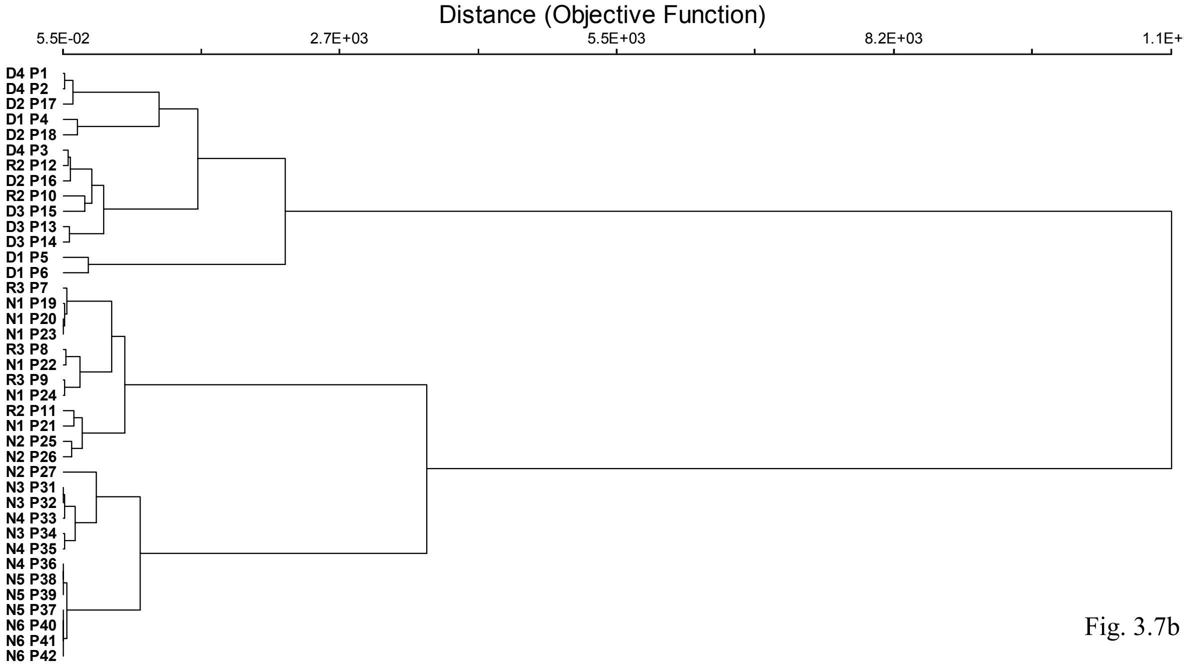

Fig. 3.7b

Fig. 3.7: Cluster analysis of vegetation (Fig 3.7a) and site factors (Fig 3.7b). Species cover was root transformed to meet normalized distribution. Site factors are WT, EC, pH, C/N, H. For site codes see Tab 3.1.

The site-factor based cluster analysis (Fig. 3.7b) principally followed a similar pattern in terms of overall separation of the sites, but the diverse feature of distances and the order of the plots resulted in a slightly different graph appearance. However, the overall grouping in the upper group of the degraded sites (D1 to D4) together with two plots of the restored group R2 (P10, P12) and the lower group of the natural sites (N1 to N6) together with the restored site R3, was similar to the vegetation-based cluster. Moreover, the separation between these two overall groups is more pronounced when looking at the distances. In contrast to the vegetation based analysis, site N6 now formed part of the lower overall group, marking the endpoint of the diagram. In terms of homogeneity, the picture was more diverse than in the vegetation-based cluster. In further detail, the separation of Plot 4 from Plots 5 and 6 of site D1 was very similar to the vegetation cluster, when the distances between the plots are compared. Also parallel to the vegetation cluster, was the grouping of the sites N3, N4, N5, N6. Site N1 was mixed up with plots from site R3, but still remained close together. Site N2 was more sharply divided, with a separation of plot 27. Site D2 also was split up within the upper group.

The general grouping was similar when comparing the results of the vegetation and site-factor based cluster analysis. However, some sites and plots behaved differently, especially sites D2 and N2 which had more distant positions between the individual plots, and N6 with a different location within the order of the sites. Consequently, a direct gradient analysis (CCA) was undertaken to analyse vegetation and site factors parallel.

Canonical Correspondence Analysis (CCA)

The canonical correspondence analysis (CCA) of species as the main matrix and site factors as a secondary matrix (Fig. 3.8) showed a general separation tendency from degraded (left), to restored (middle), to natural (right) sites. In more detail, the natural sites N3/N4 to N6 separate clearly, as does the group of degraded sites D1/D2, D3 and D4. However, in the centre of the biplot, groups N1 and N2 had a more diverse pattern and were found to partially overlap with R2 and especially with R3. Both the initial similarity of the restored sites with the natural sites N1/N2, as well as the clear distinction between the natural group as a whole (N1/N2 versus N3 to N6) was detected. The eigenvalue was 0.419 for the first axis, 0.186 for the second axis, and 0.069 for the third. Species variance was explained to 30.1 % cumulatively over the first three axes, with the first axis explaining 18.7 %, the second 8.3 % and the third 3.1 % of the variation. Pearson correlation was 0.885 for the first axis, 0.684 for the second and 0.640 for the third. The Monte Carlo permutation test (99 runs) was significant ($p<0.05$) with real data to randomised data for the eigenvalues of the first axis (0.419 / 0.130), the second axis (0.186 / 0.063) and the third axis (0.069 / 0.035). It was also significant for the Pearson correlation coefficient for the first axis (0.885 / 0.507), second axis (0.684 / 0.465) and the third axis (0.640 / 0.416). The first axis was significantly ($p< 0.01$) correlated with H (0.947), WT (-0.846), EC (0.805) and C/N (-0.805). The second axis was significantly ($p< 0.01$) correlated with pH (-0.531).

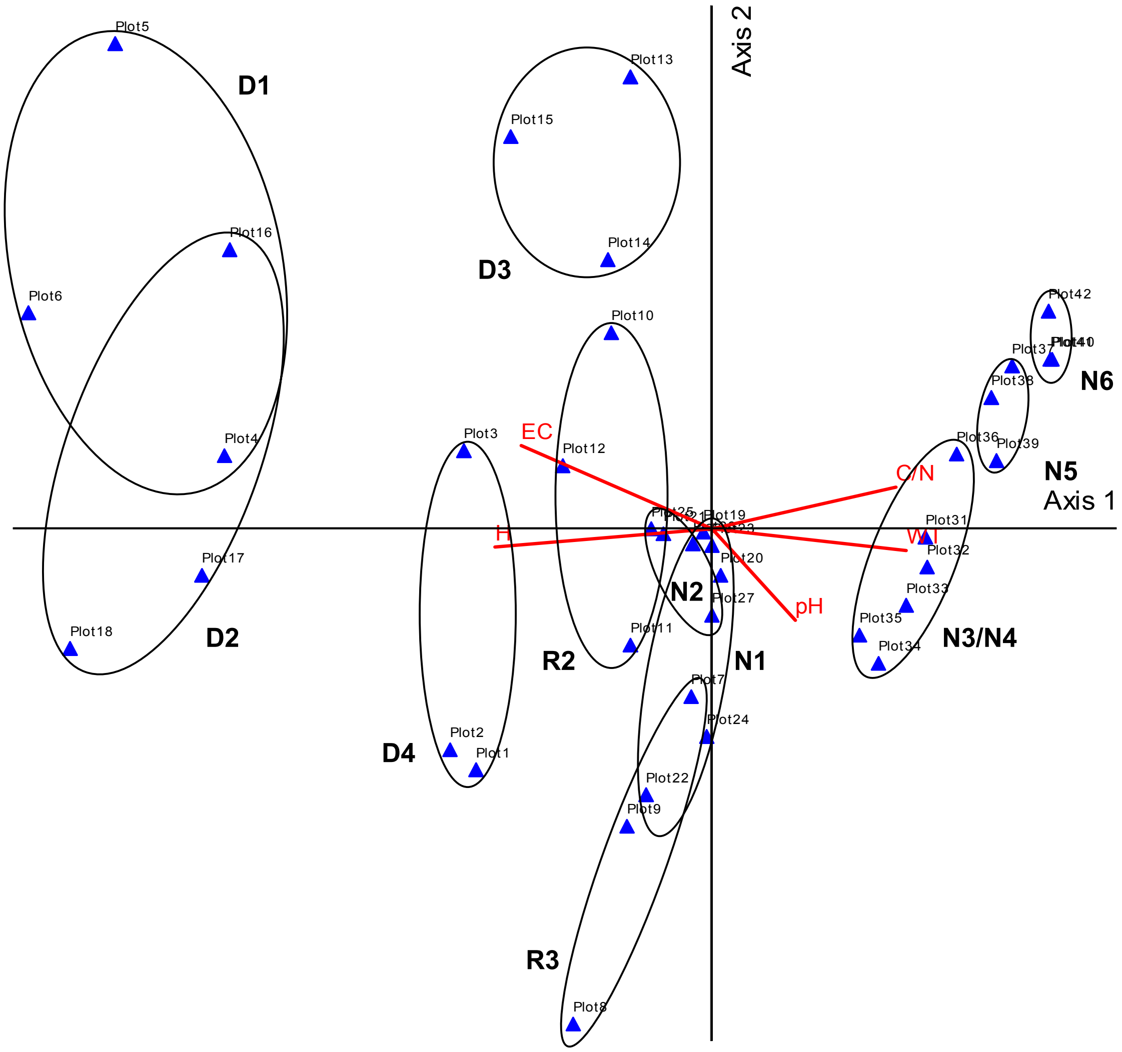

Fig. 3.8: Canonical correspondence analysis (CCA) of species as a main matrix, with the site factors of electrical conductiviy (EC), C/N-value, water-table (WT), pH-value and H-value as a secondary matrix. Species cover was root transformed to meet normalized distribution. The grouping of the sites is shown for the predefined groups (see material and methods; Tab 3.1), which is widely confirmed by the CCA results. D1 dry bog heathland (peat cut); D2 dry bog heathland (peat cut); D3 dry bog heathland (drained-only); D4 dry bog heathland (drained-only); R1 flooded dry bog heathland; R2 restored moist bog heathland; R3 restored *Sphagnum* lawn. N1 moist bog heathland; N2 bog shrubs (*Pinus mugo*); N3 *Sphagnum* lawn; N4 *Eriophorum* and *Sphagnum* hummocks; N5 transition hummocks – hollows; N6 *Sphagnum* hollow. For site codes see Tab 3.1.

Development scheme of sites

Management activities together with time since impact are important determining factors for the development of ecosystem types in bogs. These time and management effects could not be explicitly analysed in the former chapters (cluster analysis and CCA), as these approaches are based on the actual composition (as expression of the development). Hence, in order to understand the relationship between the natural, degraded and restored sites and potential development directions for the future evolution of the sites, a conceptual development scheme was sketched (see Fig. 3.9). This scheme was an adaptation of a broader scheme over all vegetation types of the Kendlmühlfilze area (Pfadenhauer et al. 1990) to the specific selection of the studied ecosystems. The positions of the sites were plotted relative to each other in terms of impact and water-table (dry to wet) and were linked via development lines. The former chapters revealed a separation of natural sites N1 and N2 from the rest of the natural sites (N3 to N6), which is reflected in their distinct positions in the scheme (Fig. 3.9). Site N6 was also separated from N3 to N5, as a result of significantly different WT and plant composition. Additionally, site R1, which represents the flooded bog heathland and could not be included in the cluster analysis and the CCA (see above), was added separately to the scheme. The interpretation of the stages of succession for the sites is full of uncertainties. However, the development scheme (Fig. 3.9) can help to visualize the past and potential future development of the sites and can help with interpreting the results of the former chapters from a development point of view.

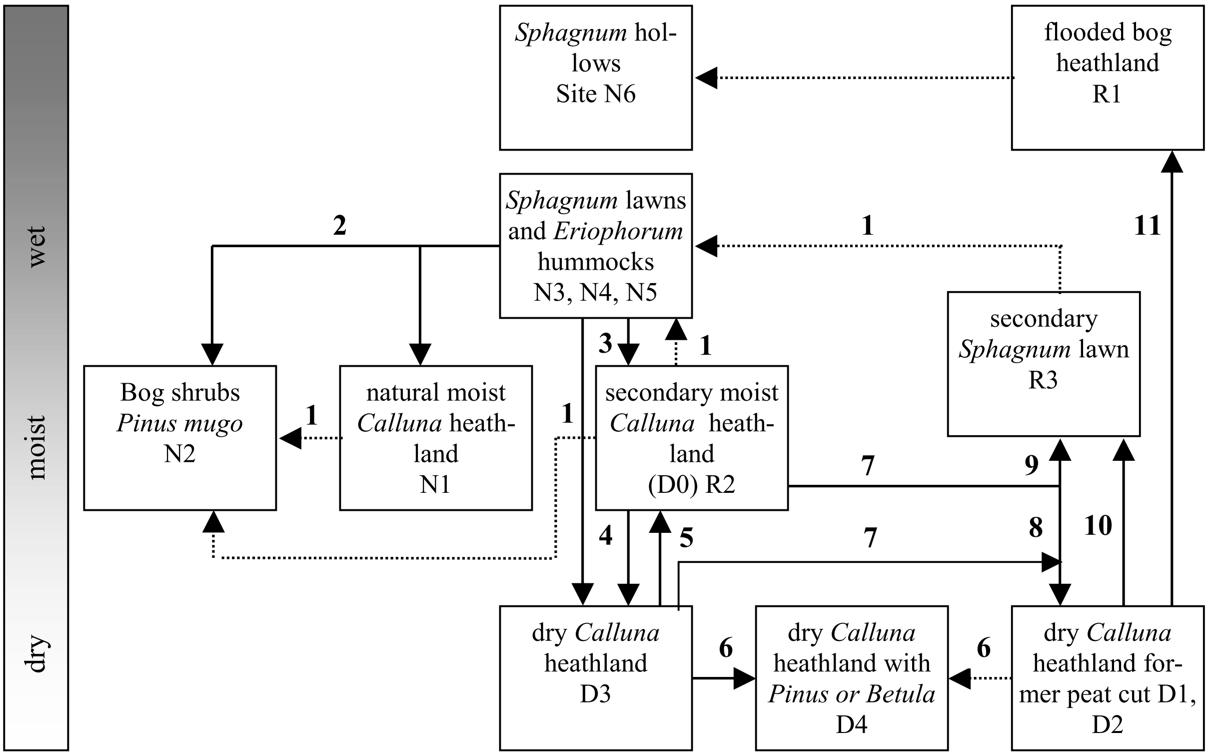

Fig. 3.9: Development scheme of the studied sites in the Kendlmühlfilze: 1 succession (natural or after restoration); 2 slight natural drainage due to the position at the marginal slopes of the bogs; 3 human induced slight drainage; 4 human induced severe drainage (establishment of D3 around 50 years prior to the study); 5 restoration by blocking the ditches – water-table close to the surface (10 years prior to the study); 6 succession – drainage still active; 7 peat cutting, filled with vegetation layer (last peat cut activities and establishment of sites D1, D2 around 50 years prior to the study); 8 drainage long-term active; 9 drainage not functioning after peat cutting; 10 restoration of D1, D2 by flooding till surface (10 years prior to the study); 11 restoration by flooding of D1, D2 sites over the surface (during the study). Full arrows: succession, development or relation between the types actually present in the area. Dotted arrows: potential further succession, development or relation between the types.

The natural sites N3, N4, N5 were likely to be linked with the sites N1 and N2, because the slight natural drainage effects at the marginal slopes and dryer former hummocky sites are typical site conditions for the development of bog shrubs (site N2) and natural moist *Calluna vulgaris* (L.) heathlands (site N1). If the natural sites N3 to N5 were slightly drained (human impact), the appearance of secondary moist *Calluna vulgaris* (L.) heathlands would be expected, which in the studied sites appeared as restored sites, developed from the more significantly drained dry *Calluna vulgaris* (L.) heathlands (site D3).

Therefore these secondary moist *Calluna* heathlands were classified as potential degraded sites (D0) but were actually present in the study area only as restored site R2. Depending on the position on the peat bog and the water-table fluctuations, the development of this type is expected to go to the bog shrubs (N2) or to the group of undisturbed natural sites N3 to N5. Dry *Calluna vulgaris* (L.) heathlands (site D3) evolved in the study area as degraded sites after severe human induced drainage around 50 years ago. This site developed directly from the natural sites N3-N6 or via an intermediate step over the secondary moist *Calluna vulgaris* (L.) heathlands (D0). With drainage active over a long term, the dry *Calluna vulgaris* (L.) heathland (site D3) developed into a dry *Calluna vulgaris* (L.) heathland with scattered *Pinus sylvestris* (L.) and *Betula pubescens* (Ehrh.) shrubs and trees (site D4). This latter type may also be the destination for the former peat cut dry *Calluna vulgaris* (L.) heathlands (D1, D2), if these sites are not restored. Restoration of these degraded types (D1, D2) via flooding up to the surface level 10 years prior to the study, led to the restored secondary *Sphagnum* lawn (site R3), which may continue to develop into the natural group of sites N3, N4 or N5. Finally, flooding over the surface of degraded site D2, experimentally effected during the study, resulted in a flooded bog heathland (site R1), which in dependence of water quality and nutrient input may directly develop in the direction of the natural *Sphagnum* (L.) hollows (site N6).

Discussion

Species composition

Species composition over all sites includes the indicator species for site conditions almost entirely, which were used for the differentiation of the vegetation units of the Kendlmühlfilze (Pfadenhauer et al. 1990). These indicator properties helped with the interpretation of the results of the indirect gradient analysis (DCA, Fig. 3.2.b) of the studied plots.

Ombrotrophic conditions (nutrient poor bog peat, WT close to the surface) are indicated by *Andromeda polifolia* (L.), *Eriophorum vaginatum* (L.), *Oxycoccus palustris* (Pers.), *Sphagnum magellanicum* (Brid.) and *Sphagnum capillifolium* (Ehrh.) (Pfadenhauer et al., 1990). In the studied plots, this group is split up around the centre of the biplot (DCA, Fig. 3.2b), illustrating the relationship within the group according to the specific composition of the underlying plots. *Scheuchzeria palustris* (L.) and *Sphagnum cuspidatum* (Hoffm.) were defined as indicators of “transition mires” (site characteristics between bogs and fens, see Pfadenhauer, 1997) and hollows (Pfadenhauer et al., 1990). The separation of these species in the DCA biplot (Fig. 3.2b), confirmed this distinct behaviour of these two species. *Rhynchospora alba* (L.) was described as a pioneer and as an indicator of partly compacted peat in bogs and transition mires (Pfadenhauer et al., 1990), which was in part reflected in the intermediate position near the hollow species within the DCA biplot (Fig. 3.2b). *Calluna vulgaris* (L.) was classified as an indicator of dried out bog-peat (Pfadenhauer et al., 1990). This species was present across the entire range of degraded sites as a dominant component. In the natural or secondary moist *Calluna* heathlands, (sites N1, R2) it was subdominant and indicated slight natural or human induced drainage effects. The singular position within the DCA biplot (Fig. 3.2b) reflected this drainage indicator value for the studied sites. *Molinia caerulea* (L.) indicates decomposed and heavily compacted bog-peat and oscillating water-tables (Pfadenhauer et al., 1990). Its appearance in sites R2, R3, and D4 was in accordance with this site-factor indication, as oscillating water-tables and decomposition were common features for these sites. According to the results of the species ordination (DCA, Fig. 3.2), *Molinia* clearly showed a behaviour that was separate from the other indicator species. The overall classification of the species to indicator groups (Pfadenhauer et al., 1990) was widely reflected in the position of the species in the DCA (Fig. 3.2b). A DCA biplot of southeastern alpine bogs had a very similar distribution for the corresponding plants and indicator groups (Bragazza & Gerdol, 2002).

LAI

Reported LAI´s from wetlands included the vascular plants but not the mosses, obviously because of the difficulty of making good estimates (Moore et al. 2002, Tenhunen et al., 1995). However, biomass data were available for different sections of the vegetation layer, including mosses in northern wetlands. So an LAI of 0.60 was reported for the graminoid plants of a riparian *Carex* meadow (Imnavit Creek, Alaska; Tenhunen et al. 1995), with a corresponding biomass of 54 g m^{-2} and a moss biomass of 216 g m^{-2} from the same plot (Oberbauer et al. 1992). Considering that the moss cover in the respective ecosystem type was more than 60 % (Tenhunen et al., 1992) and assuming a linear relationship between biomass and LAI, the moss-layer would account for an estimated maximum LAI of 4, calculated to a 100 % cover. So the maximum estimated moss-LAI of 3 for the studied plots, with 100 % moss cover, was within a conservative range.

Vascular LAI´s for northern tundra ecosystems were between 0.44 and 0.95 (Reynolds et al., 1996) and was 1.30 for a northern ombrotrophic bog (Moore et al., 2002). The maximum estimated vascular LAI of 1.13 found on the studied sites was within this range. The sites were clearly separated along a disturbance gradient from degraded, to the restored site R2, and finally reaching a total LAI of 3.6 for the restored site R3, a value close to the average over all natural sites. The importance of the underlying moss cover for LAI is reflected in this development, as site R3 already had a moss cover of nearly 100 % (see Fig. 3.3 and Annex-Tab. 3.1).

Aerenchymous leaves

The quantity of aerenchymous leaves was sampled as indicator for the plant mediated CH_4-production and emission, as several studies (Bellisario et al., 1999, Joabsson & Christensen, 2001) detected a strong relationship between aerenchymous plants and methane emission. In this study, a leaf count was used as a non-destructive alternative to biomass sampling. The *Scheuchzeria* leaves were transformed to an *Eriophorum*-leaf standard, according to the cross section diameter at the leaf base. The quantity of these standardized *Eriophorum* leaves rose nearly exponentially from 0 on the degraded sites D1, D2, and D3 to more than 2000 m^{-2} on the natural *Sphagnum* hollow (N6). As aerenchymous plants adapt to waterlogged sites, the increasing quantity of *Eriophorum* leaves was rising along with WT towards the natural sites. Mean WT explained 60% of the variation in leaf quantity of the sites (Leaves=2051*exp(-WT/-6.78); r^2=0.60; p<0.01). In addition to site differentiation, these quantities of aerenchymous leaves were used as an independent explaining variable for methane balances (see chapter 5).

WT

Water table is one of the most important factors in maintaining bog specific vegetation types and therefore was widely used as indicator of site conditions (Sliva, 1997). The differentiation of the studied sites via mean WT and span revealed clear differences for the majority of the sites (see Tab. 3.3). Ranges of the means and spans were in accordance with data reported by Frankl (1996), Sliva (1997), and Bragazza and Gerdol (2002) for the lawns (mean WT –9 cm) and the hollows (mean WT +1 cm), whereas much deeper mean WT (-26 cm) were reported for the hummocks. The mean of the natural sites did not drop below –10 cm and minimum values lower than –30 cm only occasionally appeared on site N4, which is widely reported as the lower limit for undisturbed bog sites (Dierssen & Dierssen, 1984, Ingram, 1983, Jeckel, 1986, Frankl, 1996).

Drainage provokes both a drop of the mean water-table and a more pronounced oscillation with faster response to rainfall and drought (Frankl, 1996, Sliva, 1997). This can be seen in the time-courses of sites D3 and D4, where WT instantly reacted to water deficits or rainfall. Lütt (1992) found that former peat cutting areas (even in the process of regeneration) still exhibited higher oscillations and lower mean water-table, than natural sites. The shape and dynamics of the WT-time-series for degraded sites D1 and D2, as along with the still oscillating curve of site R3 (restored, former peat cut site) confirmed these findings.

Functionally unique sites, such as R3 and N3 along with R2 and D3, were not significantly different in their mean WT, according to ANOVA results (see Tab. 3.3). Therefore, mean and span as used by Sliva (1997) did not seem sufficient to differentiate the sites by their individual behaviour. The dynamic over the course of the year was not sufficiently tracked by statistical parameters. ANOVA shows good performance for a single time comparison of different datasets, but was limited to differentiating the behaviour of sites, when time courses were underlying. Time series analysis (e.g. Stoyan et al., 1997) focused on smoothing and inspection of a trend in the dataset. Smoothing would have reduced the differentiation between the sites via the differences in the dynamic of the sites, especially for the WT dynamic. Finally, an explorative comparison of the year courses was used to identify the differences between the behaviour of the WT of the sites. This approach was also used by Lütt (1992), who found similarly pronounced dynamics during the course of a year for bog ecosystems, with minima in July and August.

One principal cause of the greater oscillation on drained and former peat cut sites seemed to be the structure of the peat and the development of the corresponding vegetation layer at the surface. Especially after peat cutting, but also after long-term drainage, the peat layers within the acrotelm are continuously decomposed (see H-values of the degraded sites respectively, Tab. 3.4). These layers loose their original structure and hence loose the capacity for capillarity processes to hold the WT high (Eggelsmann 1984). For this reason in particular, and because of the severely changed nutrient conditions, Pfadenhauer et al. (1998) proposed to start bog restoration at the peat surface of minerotrophic layers (fen peat), allowing the bog (in a long term process) to rebuild a structurally undisturbed peat profile.

EC, pH, Ca^{++} and K^{+}

Electrical conductivity (EC; in µS/cm) of the subsurface water was often used to indicate the nutrient status of the sites (Poschlod, 1990, Frankl, 1996, Sliva, 1997, Pfadenhauer, 1997). Onsite measurements are fast and easy, which allows for measurement of the dynamic behaviour of the nutrient contents over a longer period (see Fig. 3.6). As the bulk of soluble ions within the water are measured, EC is an integrating indicator of the nutrient status for the sites, although it does not allow for differentiation between the ions. Sjörs (1950) and Zimmerli (1988) stated the importance of correcting the electrical conductivity in relation to pH, because of the influence of H^{+}-ions on EC in lower pH ranges. However, using the raw data from a correlation analysis of Sliva (1997) between EC and pH in the low pH range (3.8 to 4.5), no significant impact on EC could be detected. Frankl (1996) also did not use corrected EC values when comparing natural sites on bog ecosystems in the forelands of the Alps, with a range of values between 40 and 80 µS/cm (means). The measured EC values in the presented study fall within the ranges reported for similar ecosystems (Frankl, 1996, Sliva, 1997), and they confirmed both the generally nutrient poor conditions of the bog peat and the indication of degradation processes with rising EC values (see Tab. 3.3). Time series on almost all sites demonstrate rising values until fall (Fig. 3.6), which may be a combined product of increasing nutrient availability (litter fall, favourable decomposition conditions with higher soil temperatures) and higher evapotranspiration (Sliva, 1997). At site R3, and to lesser extent at site R2, these continuously rising time courses formed just the baseline, whereas single values jumped notably above this level (Fig. 3.6). These jumps appeared almost exclusively together with low WT (see Fig. 3.5). Assuming these data not simply outliers, the coincidence with low water table may indicate that these two sites are influenced by up-welling calcareous subsurface water, which is a well known feature in these bog-types on the foothills of the Alps (Sliva 1997). Once the pressure of the rain-fed upper water-body is reduced because of drought, the lower water bodies can expand and locally well up and mix with the nutrient poor surface water. Consequently, means of WT are significantly correlated with means of EC at site R3 ($EC_{mean}=26.87 + EXP(-23.87/WT_{mean})$; $r^2=0.56$, $p<0.001$). The origins of the up-welling waters are normally calcareous groundwater streams below the peat layers of the bog, which explains, that the . mean pH-values of R2 and R3 were the highest of all the sites (Tab. 3.3). The influence of up-welling groundwater can be indicated by higher pH-values in the range up to 5 or 6 (Sliva, 1997). However, the dynamic of the pH-values over the year was not pronounced and did not follow a simple relationship to water table like EC. A further confirmation of the origin of the water during low WT was the higher ranges of Ca^{++} values (Tab. 3.5) within the corresponding sites (R2 and R3), although these values must be interpreted with caution (see below).

Acidity is a major characteristic of waterlogged peat bogs. Therefore, pH-values in the pore water were used as important indicators to differentiate the status of the peat sites. Succow (1988) classified pH-values lower than 4.8 as an indicator of acid site conditions. Poschlod (1990) found pH-values of 3.7 to 4.0 in oligotrophic bog-sites in the foothills of the Alps. The recorded values of the studied sites were in accordance with these prior studies (Tab. 3.3). Even at the sites with supposed influence of up-welling calcareous water (R2, R3), the maximum mean pH-value was 4.6, indicating an overall limited influence on pH, when compared to the ranges of 3.8 to 4.5 on nutrient poor bogs in the forelands of the Bavarian Alps without the influence of up-welling nutrient-rich groundwater (Sliva, 1997). Drainage can cause the pH to be lower than on natural sites, as the aeration of the peat enhances the redox-potential and provokes oxidation processes, which adds H^+ ions to the pore water (Lütt, 1992). This explains the low pH-values recorded on degraded sites D1-D4 respectively (Tab. 3.3).

Often Ca^{++} and K^+ are measured to express the nutrient status of the sites. Poschlod (1990) found Ca^{++} contents between 0.4 and 2.4 (mg/l) on oligotrophic bog sites and 5.1 to 25.4 (mg/l) for mesotrophic sites. Frankl (1996) reported Ca^{++} values between 0.33 and 1.33 mg l^{-1} (means) for a natural bog site close to the Bavarian Alps. Thus the minimum value of the presented study (2.33 mg l^{-1}, site N5) was higher than expected for nutrient poor bog-sites in the forelands of the Alps, and maximum values (15.17 mg l^{-1}, R3) reached the levels found on mesotrophic conditions. However, the range of 1.02 to 9.42 mg l^{-1} Ca^{++} found by Lütt (1992) in regenerating former peat cut areas corresponded relatively well with the sampled values, apart from the two restored sites that were apparently influenced by calcareous up-welling water. Sliva (1997) sampled K^{+} contents of nutrient poor degraded and restored bog ecosystems, which were generally less than 2 mg l^{-1}. Frankl (1996) found K^{+} values between 0.18 and 0.50 mg l^{-1} (means). Lütt (1992) reported K^{+} values between 0.63 and 3.67 mg l^{-1} (means) in regenerating former peat cut areas. Hölzer (1977) reported a range for K^{+} of 0.14 to 0.60 mg l^{-1}. So the K^{+} values of the studied sites in the Kendlmühlflze (see Tab. 3.5) seemed to be comparable with literature data, as seemingly the up-welling groundwater did not mask the underlying average level of the restored sites too much. For K^{+}-values, a clear tendency from higher to lower values along a disturbance gradient from degraded, to restored, to natural sites was identified despite limited repetitions (Tab. 3.5). Nonetheless, a longer time series could have better clarified the influence of short events, such as an up-welling of calcareous water on the average Ca^{++} content, especially for the restored sites.

C/N and H-value

The C/N ratio of the peat is commonly used to express the trophic level of the site and to indicate the potential decomposability of the peat (Göttlich, 1990). Peatlands with C/N ratios higher than 33 are considered to be oligotrophic, while those with ratios between 20 and 33 are classified as mesotrophic (Succow & Joosten, 1988). Since bogs are typically nutrient poor ecosystems, lower C/N ratios do normally not occur. The determined C/N values (see Tab. 3.4) represent mesotrophic site conditions for the degraded and restored sites along with natural sites N1 and N2. Sites N3 to N6 could be classified as oligotrophic sites (after Succow & Joosten 2001). This separation within the natural sites, as indicated earlier, is an effect of slight drainage on the trophic status of the upper peat layer, which was detected in parallel with the sharp differences of the decomposition status (H-value).

The decomposition status was estimated by the H-value of v. Post (Grosse-Brauckmann, 1990). The H-value normally rises from the youngest peat-layer at the surface to the older peat layers at the bottom of the profile (Grosse-Brauckmann, 1990). Drainage has the specific effect of sharply lowering and pronounced oscillating of water-tables, which provokes faster decomposition rates near the mean water-table level. Therefore, old drained areas can be detected via high H-values already in the acrotelm, as with sites D1 and D2, which had the highest values of the studied sites (6.4 and 7.1). Even the former peat cut site, which was restored ten years ago (site R3), experienced higher H-values (4.6) than the drained but never cut degraded sites D3 (3.6) and D4 (3.8). The H-values of a wide group (D3, D4, R2, N1, N2) did not differ significantly (3.3 to 3.8). The natural sites N3 to N6 were again very different from the rest, with the lowest values between 1 and 1.6. Thus the H-value reflected the gross differences of disturbance between peat cut, drainage and untouched sites; although a wide group (D3, D4, R2, N1, N2) with varied histories could not be separated further. The clear difference between the natural sites N1 and N2 and the rest of the natural sites (N3-N6) was a confirmation of the different site conditions, which were also identified by the C/N-value. The slight natural drainage obviously provoked enhanced decomposition rates in the peat; similar to the effects of artificial drainage on sites D3 and D4 (see Tab. 3.4).

Vegetation and sites

The sites were selected by considering the coverage of the most important bog ecosystems in the forelands of the Bavarian Alps, with special emphasis on representing a complete disturbance gradient from natural, to degraded, to restored sites. Site selection was based on vegetation, structure and micro-topography (methodologies like Succow, 1988 and Pfadenhauer et al., 1990) and was oriented on vegetation types, mapped in a prior study over the entire Kendlmühlfilze-area (Pfadenhauer et al., 1990). Six of ten vegetation types were represented in the core study area, but additional information about site management led to 12 sites with a differentiation of the dry *Calluna* heathlands (single type in the vegetation map) in areas of former peat cutting (D1, D2) and draining without subsequent peat cutting (D3, D4). Completely new to the list were the restored sites, with varied time since restoration (R2, R3 ten years prior to the study, R1 during the study) and different water management approaches (R1 flooding over the surface, R2 rewetting below surface, R3 rewetting till surface). This site composition offered a unique opportunity to analyse changing bog-ecosystem functions along a disturbance gradient (see chapters 4, 5 & 6). However, as site selection was based on a best estimation approach, without detailed vegetation-, soil- and water-analysis, a post-hoc evaluation was necessary to assess the grouping in terms of a sensible representation of the desired ecosystem types.

The grouping of the analysed plots via a direct gradient analysis (CCA, see Fig. 3.8) was taken as the most realistic picture of the relationship in a multidimensional space. Therefore this plot composition along the underlying site-factors represented the reference system for the post-hoc evaluation of the site grouping. Comparison by indirect gradient analysis (DCA, Fig. 3.2a) with the direct gradient analysis (CCA, Fig. 3.8) was first done in terms of the grouping of the plots to the sites and second in terms of the positions of the sites in relation to other sites in the biplot. The natural sites were characterized by their homogenous pattern with the corresponding plots close together in the species ordination (Fig. 3.2a). This was also true for the combined species and site-factor ordination (Fig. 3.8) for sites N2–N6, but N1 had a more diverse appearance. The underlying site factors had a correspondingly homogenous span over the sites N2-N6, but the variation of site N1 (Fig. 3.8) should be attributed to greater variation in the corresponding site factor scores. This can be confirmed by comparing the two cluster analyses. The plots of site N1 were more separate in the site-factor cluster (Fig. 3.7b) than in the species cluster (Fig 3.7a). On the other hand, the weakly clumped grouping of the degraded sites D1, D3 and D4 in the CCA (Fig. 3.8) was generally similar to the picture obtained via species ordination (DCA; Fig. 3.2a). The pattern of the plot grouping to the corresponding sites D2, R2 and R3 differed significantly between DCA (Fig. 3.2a) and CCA (Fig 3.8). Plots of site D2 were closely positioned in DCA (Fig. 3.2a), but appeared to be more distant in CCA (Fig. 3.8). Although vegetation composition seemed to be very similar for site D2 (see species cluster; Fig. 3.7.a), the values for the corresponding site factors varied according to the distance between the plots of site D2 in the site-factor cluster (Fig. 3.7b). The plots of site R2 were spread over a wide area in the species ordination (DCA; Fig. 3.2a), but appeared less distant in the combined species site-factor ordination (CCA; Fig. 3.8). Finally, site R3 was centred and homogenous in the species ordination (DCA, Fig. 3.2a), but had a wider span in the combined ordination (CCA, Fig. 3.8). Again, this can be explained by a more distant position of the corresponding plots in the site-factor cluster (Fig. 3.7b).

The relative position of the grouped sites was then compared between the species based ordination (DCA, Fig. 3.2) and the combined ordination (CCA, Fig. 3.8). The general order from degraded, to restored, to natural sites along Axis 1 was similar in both ordination biplots (Fig. 3.2a and Fig. 3.8). The span of the degraded sites along axis 2 and the isolated position of site N6 at the extreme right of the DCA biplot (Fig. 3.2a) however, were different in their appearance in the CCA (Fig. 3.8). The relative position of the natural sites N3 to N6 more closely represented a continuous underlying gradient in the CCA, which can be explained by looking at the site-factor cluster (Fig. 3.7a). Generally the separation between the sites was more pronounced in the combined ordination (CCA, Fig. 3.8) than in the species ordination (DCA; Fig. 3.2a). However, site R3 overlapped partially with site N1 as did site R2 and N2, which was interpreted as a hint of the compositional and hence functional similarities between these ecosystem types.

Finally these comparisons illustrated that the field based approach in selecting representative ecosystem types revealed a set of varied sites, which could be similarly identified in a post-hoc analysis via CCA. The indicator value of vegetation composition alone (Fig. 3.2.a and Fig.3.7a) for a specific expression of site-factor values (Fig. 3.7b) could be confirmed for the majority of the natural sites as well as for some of the degraded sites. However, the span and variability of the site-factors in the restored sites R2, R3 and degraded site D2, were not fully represented by the corresponding variability of the species plots. For the field based selection process therefore, the use of additional information on e.g. micro-topography and land-use history was strongly confirmed as essential to separate functional units, along a disturbance gradient.

Site development

The development scheme (Fig. 3.9) links the studied ecosystem types via succession and management impacts. This scheme represents an adaptation of a wider scheme for the entire set of vegetation types of the Kendlmühlfilze (Pfadenhauer et al., 1990). The development of the restored sites since restoration was a relatively fast process (sites R2 and R3 were restored 10 years prior to the study), although WT oscillations were identified as a constraint for *Sphagnum* development (Pfadenhauer, 1998). Therefore, the future speed and direction of the process is still uncertain, depending primarily on mean WT and oscillation and on nutrient input (Sliva, 1997). These sites are actively regenerating however, and the development towards natural site conditions with a building up of an active acrotelm is expected to proceed. The development of the acrotelm will in turn help to maintain a WT close to the surface and reduce oscillations. Site R1 was restored during the study year and the first colonizing and floating *Sphagnum cuspidatum* (Hoffm.) patches had already been detected a year after restoration. The colonization was possibly enhanced by the structures of dead *Pinus* and *Calluna* shrubs on the flooded site (Pfadenhauer, 1998). The direction of succession towards the conditions of hollow site N6 seemed to already be initiated, as was documented for several small-scale peat cutting sites in the region (Poschlod, 1990). The degraded site D1 is expected to develop slowly in the direction of a *Birch* or *Pinus* dominated peat forest, as these types of old Calluna shrubs on heavily drained bare peat are very un-hostile sites for invasion by other species (Pfadenhauer, 1998). The degraded site D3, which still has an entire peat profile, a notably higher WT than D1 and a more diverse vegetation cover already, is expected to develop towards the type D4, with sparse *Pinus* and *Birch* cover and ombrotrophic species in the understory. These slow growing forests are common vegetation types in degraded bogs in the foothills of the Alps. Without restoration, growing bog ecosystems will not develop from these degraded sites. The development of the natural sites is principally more difficult to predict, as both climate and nutrients will affect the maintenance of the sites. Global change models prognosticate wetter and warmer conditions (Fischer et al., 2002), which principally may not affect the bogs in an adverse direction, as the main growing period (Atlantikum, 8000 to 5000 a BC) was also wetter and warmer than the present day climate (see Pfadenhauer, 1997). Water-table fluctuations related to precipitation distribution over the course of the year will determine if growing conditions will become more or less favourable for bog development in the alpine foothills. Furthermore, changing atmospheric nutrient input is expected to have an influence on bog development. Frankl (1996) demonstrated that the N-input into a neighbouring bog ecosystem is between three and four times higher than 50 years ago. As a result of this extra nutrient load on the site, enhanced growth rates were found in the moss layer, separating the active growing horizon from the water table with subsequent invasion by woody species (heathland) Frankl (1996). Final judgements for the long-term future should be based on a detailed monitoring of the structural and functional development of bog sites along the disturbance gradient.

Conclusions

Disturbance provokes changes in the structure and function of ecosystems. The major disturbances that occur in bog ecosystems in the forelands of the Bavarian Alps are peat cutting and drainage, as a preparation for peat cutting. The actual vegetation cover of the bog belt reflects these long-term degrading activities. For the assessment of the role of these different sites as sinks for or sources of greenhouse gases, a study on trace gas exchange along a disturbance gradient was initiated. The important step of site selection was undertaken based on expert judgements of the vegetation composition together with micro-topography in the field and additional information about land-use history and management. A post-hoc evaluation of the differentiated sites in terms of vegetation composition and ranges of corresponding site-factors generally confirmed the field based pre-defined site grouping. However, single plots and even sites differed in terms of homogeneity when a vegetation based cluster analysis was compared with a site factor based analysis. Finally, a CCA-biplot illustrated the relationship between the sites, representing the expected disturbance gradient with a generally more homogenous appearance for the natural sites as compared to the degraded and restored ones. Hence the site selection was proven to cover a structural disturbance gradient. How far this structural disturbance gradient will be reflected in a functional gradient in terms of different trace gas exchange-rates will be outlined in the following chapters (4, 5 & 6). The differentiation of the sites as representative bog ecosystems for natural, degraded and restored conditions was successfully achieved by a field-based survey, which was assessed via a detailed post-hoc evaluation.

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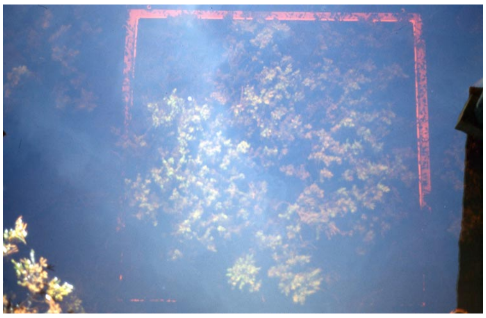
Annex – Fig 3.1

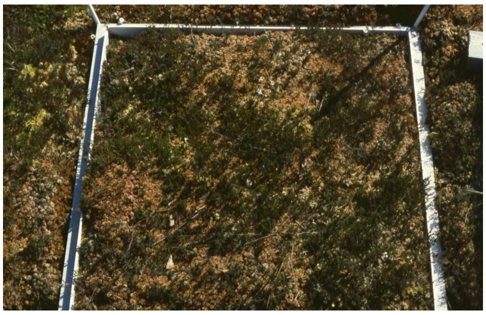


left:
Site D1
Plot 4

right:
after
flooding
site D2/R1
Plot 16

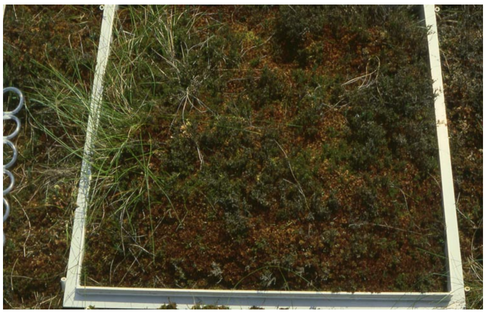
left:
Site D3
Plot 13

right:
site D4
Plot 3



left:
Site R2
Plot 12

right:
site R3
Plot 8

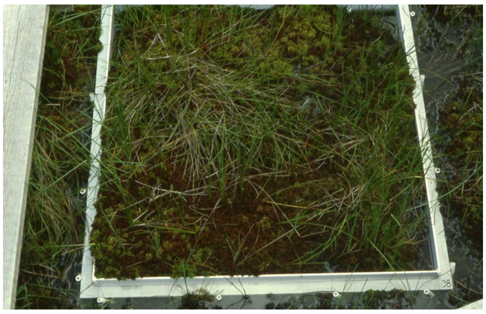


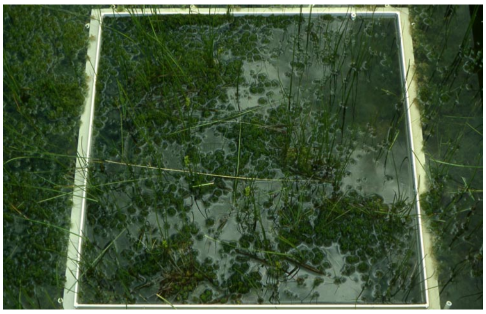
left:
Site N1
Plot 21

right:
site N2
Plot 26

left:
Site N3
Plot 31

right:
site N4
Plot 35





left:
Site N5
Plot 38

right:
site N6
Plot 41

Annex – Tab. 3.1

Site	Plot	Andromeda polifolia	Betula pubescens	Calluna vulgaris	Drosera anglica	Drosera intermedia	Drosera rotundifolia	Eriophorum vaginatum	Frangula alnus	Leucobrium glaucum	Molinia caerulea	Oxycoccus palustris	Pinus mugo	Pleurozium schreberi	Polytrichum strictum	Rhynchospora alba	Scheuchzeria palustris	Sphagnum capillifolium	Sphagnum cuspidatum	Sphagnum magellanicum
D1	4		1	30					2		4									
D1	5	1		60						18										
D1	6	1		40									1							
D2	16		1	65									4							
D2	17		2	70									2							
D2	18		1	60									2							
D3	13	1		38									1	30		1				
D3	14	2	1	60			1							36	11	1		16		1
D3	15	1	1	80										32		2				
D4	1	1		20						8	7									
D4	2	1		23					1		12				3			6		1
D4	3	1		15				8	1		3	1	1					40		1
R2	10	1		15				18			2		1	4	1	1		14		2
R2	11	1	1	13	2	1			2		9	1				2		2		5
R2	12	1		29		1										3		4		96
R3	7	3		22			2				19	1				1		10		90
R3	8	1		18			1				8	1				1		10		90
R3	9	1		26			1		2		5	1	1			1		0		100
N1	19	1		35	1		2	1				1	1			1		3		97
N1	20	1		40	1		1	10				1			10	1		10		90
N1	21	1	1	30	1		3	18				1	1			2		30		70
N1	22	1	1	30			1	9				1				3		3		97
N1	23	6	2	50	1		2	18				1	2			6		5		95
N1	24	1		40			2	1				1	2			13		3		97
N2	25	6	1	7			1	8				2	35			3		80		10
N2	26	6		9				5				7	40			3		70		20
N2	27	4		23			1	4				5	36			3		80		10
N3	31	8			2		2	1				4				1		40		60
N3	32	8			2		3	5				2				2		40		60
N3	34	12		1			3	5				2				2		30		70
N4	33	8					2	36				4	1			1		15		85
N4	35	6		16			2	36				4	3			2		20		80
N4	36	8					1	38				3				1		20		80
N5	37	3			1		6	12				4			12	1	1	20		80
N5	38						1	10				1			1	2	8	15		85
N5	39	1					6	5				1			1	10	18	28	4	68
N6	40					1	1									10	20		100	
N6	41					1										7	12		100	
N6	42					1										6	9		100	

Vegetation cover in percent; estimated with a 100 cell grid placed over the plots.

NET ECOSYSTEM CO_2 EXCHANGE OF NATURAL, DEGRADED AND RESTORED BOG ECOSYSTEMS

Summary

Accumulation of carbon is one of the key functions of undisturbed peatlands. Up to 30 % of soil carbon worldwide is stored in peatlands while they cover just 2.5 % of the worlds land surface. Therefore, the effects of disturbances, like peat extraction or drainage, and of restoration on the carbon balance of peatlands in comparison with their natural functioning are of interest in global change research. Net ecosystem CO_2 exchange (NEE) was studied with a chamber technique from January 1999 to July 2000 in natural, degraded and restored bog stands in Southern Germany. NEE measurements were done weekly to twice a week on 36 plots in 12 different sites ranging from degraded peat cut areas over restored *Sphagnum* lawns to natural *Eriophorum* hummocks and *Sphagnum* hollows.

Net ecosystem productivity (NEP) for the whole year cycle was calculated with a model using site-specific temperature-respiration and productivity-radiation relations as model parameters. Daily NEE CO_2-exchange rates over the whole year ranged between a loss of 3.3 g $m^{-2}d^{-1}$ CO_2-C in former peat cut areas and an uptake of 2.4 g $m^{-2}d^{-1}$ CO_2-C in the natural *Sphagnum* and *Eriophorum* dominated sites. Mean NEP for the whole year 1999 ranged from emissions of 443 ± 70 $m^{-2}a^{-1}$ CO_2-C in the most degraded site (former peat cut) to an uptake up to -157 ± 37 g $m^{-2}a^{-1}$ CO_2-C on a natural *Sphagnum* hollow. Restoration leads not directly to absolute C sinks but can reduce emission from degraded sites by 100 to 380 g $m^{-2}a^{-1}$ CO_2-C.

Introduction

Peatlands are terrestrial ecosystems which, under undisturbed conditions, have been continuously accumulating carbon (C) for thousands of years. The global C stock in peatlands is estimated to be between 250 (Augustin & Eschner 2001) and 455 Gt C (Gorham 1991), with more recent figures of 270 to 370 Pg C (Turunen et al., 2002). This amount of stored carbon represents approximately 12 to 30 % of the global C in soils, while peatlands cover only about 2.5 % of the world's land surface. Carbon accumulation in peatlands is the result of a gap between C uptake by photosynthesis and C- release by plant and soil respiration, since water saturation limits decomposition. Long-term accumulation rates for northern peatlands are reported to be around 24 g C m^{-2} a^{-1} (Turunen et al., 2001, 2002, Vitt, et al. 2000). Given the current climate change discussion, the role of ecosystems as sinks for or sources of greenhouse gases is an important topic. Forests and agricultural ecosystems have been the subject of many studies on carbon dioxide (CO_2), whereas wetlands (peatlands and swamps) have been studied less intensively (Buchman & Schulze, 1999). Peatlands however, in their natural functioning, have the potential to act as effective long-term sinks for C. It is also important to note that land-use practices like drainage and peat cutting, as well as changing climatic conditions, may severely affect this sink function. This is a major concern because huge amounts of carbon are stored in the organic soils.

Until now studies on C-balance and trace gas exchange of peatland ecosystems were undertaken primarily in the boreal region (e.g. Moore, 1986, Silvola, 1986, Tenhunen et al., 1992, Oechel et al., 1993, Whiting, 1994, Tenhunen et al., 1995, Oberbauer et al., 1996, Silvola, 1996a, Alm et al., 1997, Bellisario et al., 1998, Waddington et al., 1998, Bubier et al., 1999, McMichael, 1999, Sommerkorn et al., 1999, Wühtrich et al., 1999, Tuittila et al., 1999, Roulet, 2000, Soegaard et al., 2000, Waddington et al., 2001, Waddington & Warner, 2001, Frolking, 2002). The few studies that have been performed on peatlands across Europe's temperate zone have focused mainly on the emission of greenhouse gases (Flessa et al., 1998, Kamp et al., 1999, Meyer, 1999, Dirks et al., 2000, Augustin, 2001, Meyer et al., 2001). Studies on net ecosystem exchange (NEE) for peatland ecosystems in the temperate zone of Central-Europe have not been reported until now, partly because appropriate chamber techniques for measuring NEE together with other gases under these specific ecosystem conditions were not available until recently (see chapter 2).

Drainage and cutting of peat bogs have transformed many peatlands, especially in the temperate zone, into net sources of atmospheric CO_2 (Armentato & Menges 1986). In Germany, peat cutting has been reported since the first century AD, but large-scale, industrial harvesting did not take place until the 20th century (Richard 1990). In Southern Germany however, drainage for the preparation of peat cutting and non-industrial, small-scale peat cutting were the major degrading activities in bogs until recently. Fens are degraded to a greater extent, mainly because of agricultural practices. Restoration of peatlands has been a recent focus, but was initially aimed predominantly at meeting nature conservation goals (Pfadenhauer & Klötzli, 1996). Integration of matter fluxes and abiotic factors in the assessment of restoration effects was studied recently on southern German fens (Wild, 1997) and bogs (Sliva, 1997).

Trace gas exchange reflects the functioning of peatland ecosystems and can therefore be used to assess the degree of disturbance or the success of restoration on a site. For this reason, research that directly compares the CO_2 sink or source status of natural, degraded, and restored plots is of particular importance. However, recent reports on CO_2 efflux or on NEE for restored peat bogs (Komulainen et al. 1999; Tuitilla et al. 1999; Waddington & Price 2000; Waddington & Warner 2001; Waddington et al. 2001, 2002) have not compared the CO_2 exchange of all three "treatments".

Vegetation is an indicator of the status of an ecosystem, since it is the active layer for CO_2-exchange. Consequently, differences in vegetation types occurring within mire ecosystems should be reflected by their individual net ecosystem productivity (NEP). Bubier et al. (1999) found a gradient of increasing NEP, in the order of bogs, to rich fens, to intermediate fens and finally poor fens. Tuittila et al. (1999) found single species cover (*Eriophorum vaginatum*) to be related to the NEP of the sites and Bubier (1995) demonstrated that mosses (species specific cover classes) are a good indicator for predicting methane fluxes. Therefore vegetation and site parameters (see chapter 3) were tested as indicators for predicting the annual balance of CO_2- exchange.

The present study was initiated to fill a regional and thematic gap, by studying net ecosystem CO_2-exchange (NEE) in the southern German bog area in the forelands of the Bavarian Alps. Special emphasis was given to the representation of a complete disturbance gradient, covering a range from natural, to degraded, to restored bog stands. The comparison of site specific balances over the whole range of stands allowed for an assessment of the effects of degradation and restoration on the gas-balances, in comparison to natural sites. The specific objectives of the study were: i) to measure instantaneous net ecosystem CO_2- exchange with a climate controlled chamber system on a series of twelve sites with three plots each; representing natural, degraded and restored bog stands, ii) to use these instantaneous flux measurements for the parameterization of a NEE model, iii) to calculate whole annual balances for CO_2-exchange (NEP) and iv) to test if the modelled annual balances are related to and can be predicted by simple site factors, such as water table, nutrients or vegetation cover.

Materials and Methods

CO_2 sampling and analysis

The standard sampling schedule was run weekly from the 5th of January until the 28th of December 1999. On each field day, gas exchange was measured by employing a new climate controlled closed chamber method (see chapter 2) on 36 plots with 3 plots on each of the 12 different sites.

Sampling was conducted from boardwalks, which were established six weeks before the measurement period started in November 1998. Nine portable, climate controlled closed chambers (chapter 2) were applied at the same time. With four measurement cycles, all of the plots were thereby covered in a single day. One complete measurement cycle, which included moving and preparing the chambers, on one site lasted approximately one hour. The net enclosure time of the chamber lasted between 24 and 32 minutes, depending on the environmental and phenological conditions. However, CO_2-fluxes were processed from the gas samples of the first minutes of enclosure exclusively, whereas for the flux calculation of CH_4 and N_2O the gas samples of whole enclosure time were used. Measurements were generally taken between 9 am and 3 pm. Five gas-samples where transferred from the headspace of the chamber to evacuated glass bottles with a syringe. The gas samples were brought to the laboratory and analysed at the Institute for Soil Ecology, GSF Neuherberg Munich. The whole set of gas-samples were analysed within one day after sampling in the field, to avoid any effects of prolonged storage.

The gas-samples of the standard program were analysed by gas-chromatography (Shimadzu 14 A and B) and an electron capture detector (ECD). A system developed by Loftfield et al. (1997) at the Laboratory of the Institute of Soil Ecology of the GSF, Neuherberg, Germany was used. The Porapak Q columns (Millipore; 80-100µm) and the ECD of the gas chromatographs were heated to 60 and 280°C, respectively. The carrier gas was N_2 (Linde, Munich, Germany), with a flux rate of 20 ml min^{-1}. Calibration was carried out automatically every 20 samples with concentrations of 300, 1500, and 3000 ppm CO_2.

In order to measure light response curves, CO_2 concentrations were determined in the field via a portable infrared gas analysis (IRGA) system, which consisted of a Li-800 gas-analyser (Licor, Lincoln, NE, USA), a Li-1000 datalogger (Licor, Lincoln, NE, USA), and a membrane pump (3 l/min; KNF, Freiburg, Germany) combined with a rechargeable battery (7.2 Ah). The accuracy of the system was +/-2 ppm CO_2. The gas analyser worked as part of a dynamic closed chamber system, with the inflow and outflow tubes connected to the chamber. CO_2 concentrations were determined every 30 seconds to 2 minutes. These additional CO_2 measurements were conducted from end of March 1999 until July 2000. In 1999 the program included twelve field days on bright days, in addition to the standard weekly program.

Environmental variables

Parallel to the gas exchange measurements, a set of environmental variables was also tested for. Included were air temperature and relative humidity inside and outside the chamber, soil temperature at depths of 2, 5, 10, 20, 50, 100 cm, and water table depth in wells close to all collars. Photosynthetically active radiation (PAR) was sampled continuously at five minute intervals during the 1999/2000-measurement period. Furthermore, air temperature and relative humidity 2m above ground and in the vegetation layer were recorded during the whole measurement period continuously, while precipitation was determined on a weekly basis.

Flux calculation

The convention of negative values representing losses from the atmosphere to the ecosystem was used for this study. The enrichment or rarefaction graphs of the concentration versus time were accepted if r^2 was > 0.95, using at least three points for the linear regression model. Respiration measurements (dark chamber) normally resulted in a linear increase of CO_2, whereas uptake of CO_2 (transparent chamber) normally exhibited a descending curved slope. The slope was linear only during the first few minutes (summer), until the lowered concentration of CO_2 in the chamber limited the normal rate of photosynthesis. This effect was observed with CO_2 concentration lower than 300 ppm. Therefore, for the transparent measurements, only the first three to four data points of the curve, depending on linearity, were used for the flux calculation. From the concentration gradients (dc/dt), flux rates in mg m^{-2} h^{-1} were calculated by the following equation (see e.g. Flessa et al. 1998):

$$F_{CO2} = k_{CO2}\,(273 * T^{-1})(V * A^{-1})\,(dc * dt^{-1}) \tag{4.1}$$

F_{CO2} = flux rate of CO_2 (mg CO_2-C m^{-2} h^{-1})
k_{CO2} = gas-constant at 273.15 K (0.536 µg C μl^{-1})
T = instant air temperature during the measurement (K)
V = volume of the chamber (l)
A = surface area within the collar of the chamber (m^2)
$dc * dt^{-1}$ = concentration change in the chamber atmosphere over time (CO_2: ml $l^{-1}h^{-1}$)

NEE modelling

Instant NEE was measured according to the standard weekly program (from the 5th of January until the 28th of December) and additional experimental light response curves were measured over the entire snow-free period (end of March until the end of October) on a total of 12 bright days at every site. This mixed approach allowed for the coverage of changing weather conditions and phenology over the measurement period and for the parameterisation of the model along with changing conditions during the year. The model (4.2) is based on a rectangular hyperbola equation (Michaelis & Menten, 1913) and a respiration term (4.3; Lloyd & Taylor, 1994), thus calculating NEE as gross productivity minus respiration. This model (4.2; Michaelis & Menten, 1913) has been used by several authors (see e.g.

Whiting 1994; Bellisario et al., 1998) for the calculation of NEP- balances in similar ecosystems (mires) over the growing season.

$$NEE = ((GP_{max} * \alpha * PAR)*((\alpha*PAR)+ GP_{max}))^{-1})- R \qquad (4.2)$$

PAR = photon flux density of the photosynthetic active radiation [μmol m^{-2} s^{-1}]
GP_{max} = maximum rate of carbon fixation at PAR infinite [CO_2-C mg m^{-2} h^{-1}]
α = initial slope of the curve; light use efficiency [CO_2-C mg m^{-2} h^{-1}/μmol m^{-2} s^{-1}])
R = respiration model [CO_2-C mg m^{-2} h^{-1}]

The respiration model (RE) was developed separately, based on regressions between respiration rates, measured with darkened chambers, and soil temperature using Lloyd & Taylor (1994) equation (4.3). The darkened chamber measurements from the experimental light response curves were excluded from the data set. Similarly to other studies (Reichstein et al. 2003), E_o and R_{ref} of equation (4.3) were individually fit to the corresponding dataset, whereas T_{ref} (283.15 K) and T_0 (227.13 K) were set as constant parameters (Lloyd & Taylor, 1994):

$$R= R_{ref}\, e^{Eo\,(1/(T_{ref} - T_0)-1/(T_{soil} - T_0))} \qquad (4.3)$$

R = respiration [CO_2-C mg m^{-2} h^{-1}]
R_{ref} = respiration at the reference Temperature [CO_2-C mg m^{-2} h^{-1}]
E_o = activation energy [K]
T_{ref} = reference temperature: 283.15 [K]
T_0 = temperature constant for the start of biological processes: 227.13 [K]
T_{soil} = soil temperature at the depth of best fit with the dataset [K]

No effect of phenology on the respiration-temperature relationship over the year could be found. Therefore, the model parameterisation was done with the dataset from the entire year. The inclusion of water table measurements as a variable did not improve the fit of the respiration equation. This was in contrast to Mediterranean ecosystems, where soil water content is a major driver of soil respiration (Reichstein et al. 2003) and must be combined with soil temperature to realistically model soil respiration. For the respiration-temperature model, the soil temperature at the depth of best fit with respiration rates was individually identified for the sites. The soil temperatures used for the respiration-temperature relationships were measured individually, close to the plots during the gas-measurements, to guarantee the representation of the individual site conditions within the regression analysis.

NEE and gross productivity (GP), follows a pronounced dynamic through the year due to radiation, plant phenology and temperature changes during the growing period. After a test of a viable grouping of the data set, parameterisation of the model was performed for five consecutive time periods over the year, which could be differentiated sufficiently via the individual fit of the GP-radiation relationship. The grouping was as follows: months 1/2/11/12 (January, February, November and December), months 3 / 4 (March and April), months 5 / 6 (May and June), months 7 / 8 (July and August), and months 9 / 10 (September and October). The first group included the snow-covered period. The period specific parameters for the NEE-radiation relationship were subsequently interpolated between the periods, to adjust further for the phenology and climate driven changing character of the NEE-PAR relation. Finally, the model was run in 0.5-hour steps.

Statistical analysis

STATISTICA software was used to test the normality of distribution of the variables Simple and multiple linear regression analysis were done with STATISTICA. User defined equations were programmed and used in the TABLE CURVE 2D statistic program for the non-linear regression analysis, using the Michaelis & Menton (1913) and the Lloyd & Taylor (1994) equations.

Results

NEE- parameters for the model

The parameterisation of the respiration model (Lloyd & Taylor, 1994) was based on the data set from the entire year and significant regressions (r^2) between 0.43 and 0.91 (see Tab. 4.1) were derived. The individual behaviour of the respiration processes between the sites is reflected by the varied levels of the activation energy (E_0) and respiration at the reference temperature (R_{10}; $T_{ref} = 283.15$ K or 10 °C). The lowest activation energy (E_0) was found on the degraded site D3 (dry drained-only bog heathland), with 258 K (see Tab. 4.1). The highest activation energy (E_0) was obtained on site N6 of the natural group (*Sphagnum* hollow), with 499 K (Tab. 4.1). Mean activation energy (E_0) for the degraded sites (D1-D4) was 287 K, while on restored sites (R2, R3) it was 323 K. The mean activation energy was highest for the natural sites (N1-N6), at 363 K. Thus, the activation energy (E_0) rose along the gradient from degraded to restored to natural conditions. This sorting out of the sites was confirmed via the ranges of respiration rates at the reference temperature (R_{10}), where the highest rate was found on the degraded site D1 (dry bog heathland, former peat cut), with 86.7 ± 5.9 mg CO_2-C $m^{-2}h^{-1}$. The lowest value was again found on the natural site N6 (17.8 ± 3.6 mg CO_2-C $m^{-2}h^{-1}$). Grouping the sites along the disturbance gradient revealed considerable mean R_{10}-values of 74.9 ± 11.3 mg CO_2-C $m^{-2}h^{-1}$ on the degraded sites (D1-D4), 55 mg CO_2-C $m^{-2}h^{-1}$ on the restored sites (R2, R3) and 36 mg CO_2-C $m^{-2}h^{-1}$ on the natural sites (N1-N6). The natural sites represented half the R_{10} value of the degraded sites. Fig. 4.1 illustrates the differences between the sites showing the original data for the regression analysis. In all, graphs, positive values are fluxes from the soil or vegetation layer to the atmosphere (release of carbon from the ecosystem and negative values are fluxes from the atmosphere to the soil or vegetation (uptake of carbon dioxide by the ecosystem). The scaling is the same in every plot to provide a visual comparison of the graphs (see Fig 4.1). The fit of all curves over all sites to the data set (residuals) was very satisfying. The single case of degraded site D2 having a lower quality fit of the data set was a product of having

Site	Depth soil temp [cm]	$E_0 \pm SE$ [K]	$R_{10} \pm SE$ [mg m^{-2} h^{-1}]	n	r^2	Q_{10} [15-25°C]
D1	2	304.07 ± 29.51	86.66 ± 5.95	38	0.85***	2.02
D2	10	259.48 ± 90.11	74.89 ± 11.31	21	0.43**	1.82
D3	2	258.16 ± 24.62	63.49 ± 4.18	39	0.84***	1.71
D4	2	327.69 ± 35.88	72.32 ± 6.46	39	0.81***	2.13
R2	2	311.33 ± 21.86	48.70 ± 3.08	44	0.90***	2.05
R3	2	334.45 ± 30.50	61.72 ± 5.54	41	0.84***	2.16
N1	2	303.73 ± 27.79	47.52 ± 4.05	36	0.85***	2.02
N2	5	324.99 ± 59.73	43.52 ± 5.24	39	0.59***	2.12
N3	2	397.79 ± 32.43	28.93 ± 2.85	45	0.88***	2.50
N4	2	305.22 ± 20.81	44.25 ± 3.22	43	0.91***	2.02
N5	2	349.38 ± 45.83	33.28 ± 4.19	42	0.74***	2.24
N6	10	498.95 ± 82.79	17.84 ± 3.64	43	0.69***	3.16

Tab 4.1: Regression parameters of soil temperature vs. CO_2-flux: Lloyd and Taylor (1994) equation. E_0 and R_{10} were individually fit to the data set. Soil temperature was selected for best fit. Sites: D1 dry bog heathland (peat cut); D2 dry bog heathland (peat cut); D3 dry bog heathland (predrained); D4 dry bog heathland (predrained); R1 flooded dry bog heathland; R2 restored moist bog heathland; R3 restored *Sphagnum* lawn; N1 moist bog heathland ; N2 bog shrubs (*Pinus mugo*); N3 *Sphagnum* lawn; N4 *Eriophorum* and *Sphagnum* hummocks; N5 transition hummocks – hollows; N6 *Sphagnum* hollow.

less data. Since this site was restored during the measurement period (July), only 21 measurements could be used for the respiration - soil temperature relationship, compared to the 36 to 45 measurements taken on the other sites. The exceptionally shallow course of curve in the start of respiration graph for site N6 was primarily an effect of the high number of zero fluxes.

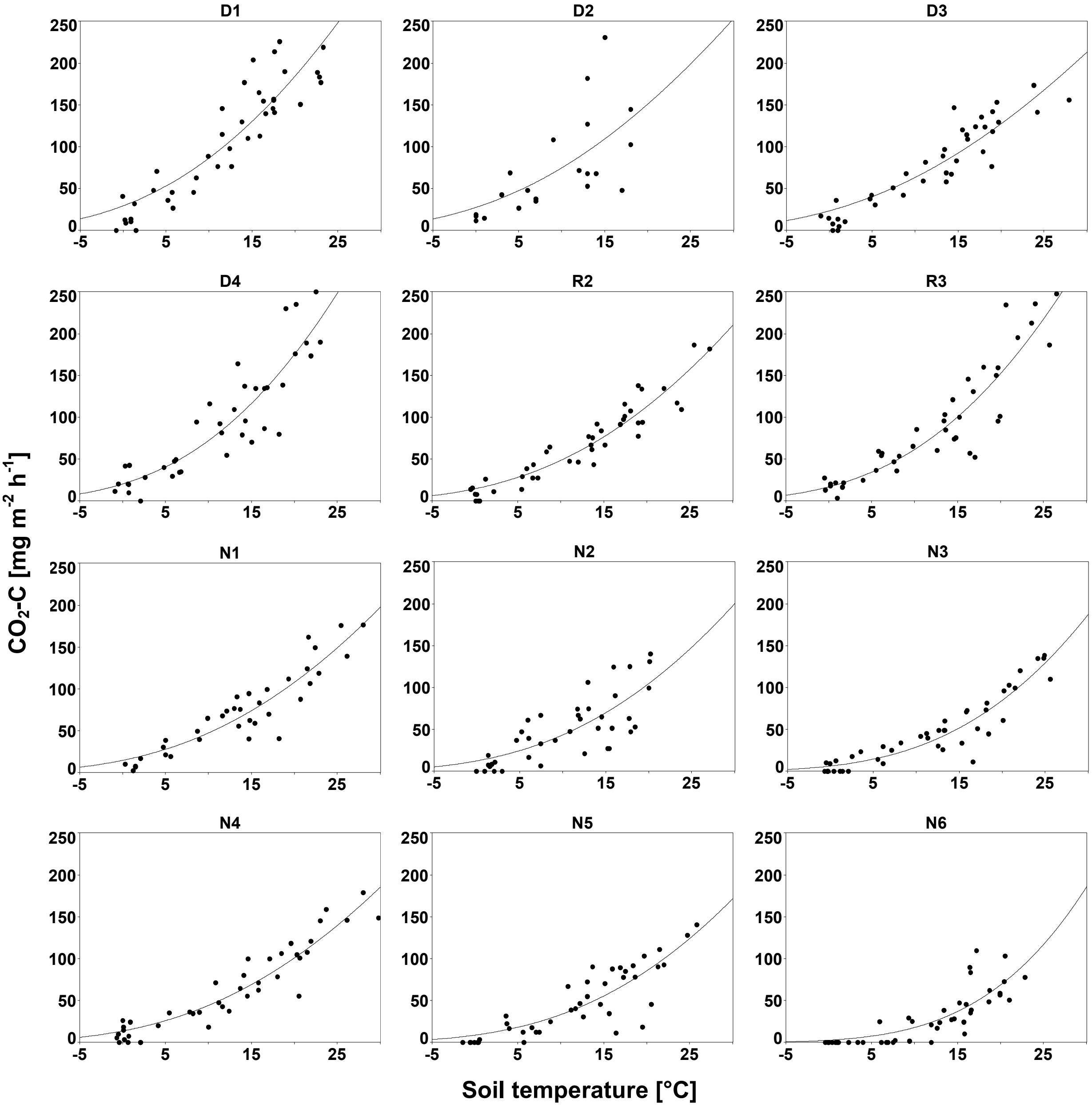


Fig 4.1: Relationship of Ecosystem Respiration (R_{ECO}) to soil temperature: Lloyd and Taylor (1994) graphs fit to the individual data set (see Tab.3). Soil temperature of best fit was selected: 2cm in all graphs, except N2 (5cm) , D2(10cm), and N6 (10 cm). For the corresponding parameters of equation (4.3) see Tab. 4.1.
Sites: D1 dry bog heathland (peat cut); D2 dry bog heathland (peat cut); D3 dry bog heathland (predrained); D4 dry bog heathland (predrained); R1 flooded dry bog heathland; R2 restored moist bog heathland; R3 restored *Sphagnum* lawn; N1 moist bog heathland ; N2 bog shrubs (*Pinus mugo*); N3 *Sphagnum* lawn; N4 *Eriophorum* and *Sphagnum* hummocks; N5 transition hummocks – hollows; N6 *Sphagnum* hollow. Further site description see chapter 3.

Tab 4.2: Site and period specific regression parameters of the rectangular hyperbola equation (Michaelis & Menten, 1916) for describing the NEE-PAR relation. Sites: D1 dry bog heathland (peat cut); D2 dry bog heathland (peat cut); D3 dry bog heathland (predrained); D4 dry bog heathland (predrained); R1 flooded dry bog heathland; R2 restored moist bog heathland; R3 restored *Sphagnum* lawn; N1 moist bog heathland ; N2 bog shrubs (*Pinus mugo*); N3 *Sphagnum* lawn; N4 *Eriophorum* and *Sphagnum* hummocks; N5 transition hummocks – hollows; N6 *Sphagnum* hollow.

Site	Period [month]	GPmax (+/-SE) $[mg\ m^{-2}\ h^{-1}]$	α (+/-SE) $[mg\ m^{-2}h^{-1}/\mu mol\ m^{-2}s^{-1}]$	R (+/-SE) $[mg\ m^{-2}\ h^{-1}]$	n	r^2
D1	1/ 2/11/12	-53.81 ± 51.47	-0.105 ± 0.102	16.54 ± 4.28	21	0.47*
D1	3 / 4	-135.03 ± 29.31	-0.193 ± 0.062	49.27 ± 5.73	36	0.78**
D1	5 / 6	-150.51 ± 18.02	-1.247 ± 0.651	121.34 ± 12.08	39	0.71*
D1	7 / 8	-345.07 ± 40.14	-0.966 ± 0.314	166.32 ± 18.73	41	0.78**
D1	9 / 10	-245.44 ± 45.91	-0.608 ± 0.265	108.26 ± 15.64	41	0.68*
D2	1/ 2/11/12	-43.29 ± 33.05	-0.112 ± 0.089	17.53 ± 2.18	22	0.66*
D2	3 / 4	-96.30 ± 12.21	-0.315 ± 0.083	53.43 ± 4.24	36	0.81***
D2	5 / 7	-142.43 ± 11.54	-1.005 ± 0.310	106.53 ± 7.53	44	0.83**
D3	1/ 2/11/12	-50.03 ± 29.30	-0.193 ± 0.182	18.17 ± 3.32	23	0.62*
D3	3 / 4	-100.72 ± 11.95	-0.292 ± 0.076	40.06 ± 4.05	36	0.84**
D3	5 / 6	-149.03 ± 13.26	-0.922 ± 0.329	84.31 ± 8.39	38	0.83**
D3	7 / 8	-214.10 ± 20.95	-0.859 ± 0.299	101.14 ± 12.24	41	0.80**
D3	9 / 10	-168.10 ± 20.58	-1.180 ± 0.507	76.32 ± 11.25	39	0.74*
D4	1/ 2/11/12	-82.04 ± 57.69	-0.088 ± 0.042	23.78 ± 2.76	32	0.56*
D4	3 / 4	-80.21 ± 10.29	-0.257 ± 0.077	42.61 ± 3.66	36	0.80**
D4	5 / 6	-281.47 ± 31.40	-0.598 ± 0.165	102.71 ± 12.48	39	0.83**
D4	7 / 8	-420.88 ± 30.55	-1.453 ± 0.300	134.24 ± 13.85	41	0.91***
D4	9 / 10	-494.12 ± 162.49	-0.390 ± 0.092	88.87 ± 8.85	41	0.88***
R2	1/ 2/11/12	-22.44 ± 4.78	-0.600 ± 0.496	14.68 ± 1.96	32	0.61*
R2	3 / 4	-194.28 ± 61.31	-0.108 ± 0.021	28.85 ± 2.84	34	0.88***
R2	5 / 6	-180.38 ± 20.66	-1.126 ± 0.519	68.82 ± 13.28	39	0.73**
R2	7 / 8	-201.35 ± 21.55	-0.799 ± 0.259	89.16 ± 11.07	42	0.79**
R2	9 / 10	-151.29 ± 20.69	-0.510 ± 0.160	59.27 ± 7.44	42	0.78**
R3	1/ 2/11/12	-105.10 ± 57.40	-0.346 ± 0.191	35.86 ± 4.53	43	0.55*
R3	3 / 4	-330.49 ± 53.55	-0.259 ± 0.034	34.22 ± 3.94	33	0.94***
R3	5 / 6	-279.13 ± 17.98	-1.417 ± 0.333	89.23 ± 10.65	39	0.91***
R3	7 / 8	-340.65 ± 29.49	-1.000 ± 0.247	115.94 ± 13.99	42	0.86***
R3	9 / 10	-301.79 ± 50.79	-0.760 ± 0.261	75.49 ± 15.65	44	0.73**
N1	1/ 2/11/12	-124.12 ± 103.24	-0.102 ± 0.040	16.46 ± 2.37	31	0.78*
N1	3 / 4	-137.64 ± 8.18	-0.403 ± 0.059	27.04 ± 3.38	37	0.94***
N1	5 / 6	-237.99 ± 19.40	-0.839 ± 0.222	66.30 ± 10.69	40	0.86***
N1	7 / 8	-320.33 ± 30.47	-0.553 ± 0.124	86.06 ± 10.79	43	0.88***
N1	9 / 10	-196.61 ± 17.81	-0.855 ± 0.225	58.57 ± 8.29	40	0.86***
N2	1/ 2/11/12	-23.80 ± 3.78	-0.901 ± 0.738	11.38 ± 2.10	26	0.72*
N2	3 / 4	-238.00 ± 25.82	-0.419 ± 0.083	29.31 ± 7.17	33	0.89***
N2	5 / 6	-196.04 ± 16.37	-0.960 ± 0.260	67.31 ± 8.90	35	0.87***
N2	7 / 8	-247.12 ± 23.69	-0.752 ± 0.216	89.09 ± 12.02	43	0.82***
N2	9 / 10	-278.56 ± 34.33	-0.595 ± 0.137	56.81 ± 9.49	42	0.85***
N3	1/ 2/11/12	-23.67 ± 4.49	-0.830 ± 0.792	8.01 ± 2.45	32	0.58*
N3	3 / 4	-81.35 ± 5.62	-0.674 ± 0.178	15.39 ± 3.37	35	0.89***
N3	5 / 6	-161.72 ± 12.03	-1.296 ± 0.398	44.49 ± 8.13	39	0.86**
N3	7 / 8	-198.03 ± 10.45	-0.799 ± 0.143	61.88 ± 6.18	43	0.93***
N3	9 / 10	-147.29 ± 10.21	-0.859 ± 0.215	39.09 ± 5.91	42	0.89***
N4	1/ 2/11/12	-37.01 ± 5.27	-0.636 ± 0.389	10.92 ± 2.24	42	0.67*
N4	3 / 4	-250.55 ± 28.64	-0.198 ± 0.024	23.28 ± 3.44	33	0.95***
N4	5 / 6	-218.20 ± 12.26	-1.327 ± 0.299	61.99 ± 8.45	43	0.90***
N4	7 / 8	-286.25 ± 10.09	-1.479 ± 0.194	77.73 ± 6.67	44	0.96***
N4	9 / 10	-223.62 ± 17.71	-0.897 ± 0.207	55.05 ± 7.80	40	0.90***
N5	1/ 2/11/12	-20.02 ± 4.24	-0.263 ± 0.241	6.44 ± 1.72	27	0.64*
N5	3 / 4	-159.18 ± 19.50	-0.309 ± 0.065	19.15 ± 4.70	34	0.88***
N5	5 / 6	-281.79 ± 22.67	-0.818 ± 0.178	48.52 ± 10.82	39	0.88***
N5	7 / 8	-327.66 ± 21.88	-1.551 ± 0.366	67.56 ± 12.50	44	0.89***
N5	9 / 10	-242.04 ± 31.11	-0.779 ± 0.231	40.75 ± 10.94	40	0.81**
N6	1/ 2/11/12	-1.00 ± 0.00	-0.01 ± 0.000	1.00 ± 0.00	19	n.s.
N6	3 / 4	-29.68 ± 3.69	-0.093 ± 0.0242	8.05 ± 1.09	34	0.83***
N6	5 / 6	-185.03 ± 23.71	-0.798 ± 0.322	35.48 ± 13.25	40	0.70**
N6	7 / 8	-356.59 ± 24.16	-1.033 ± 0.203	51.49 ± 11.16	43	0.92***
N6	9 / 10	-144.43 ± 16.06	-1.109 ± 0.431	30.72 ± 8.58	39	0.78**

Mean Q_{10}-values for system respiration (temperature range was 15-25 °C) differed between the sites following a disturbance gradient, which was observed in the individual values (Tab 4.1) and could be seen better when the group means were compared. The mean Q_{10}-value was 1.92 for degraded sites, 2.11 for restored sites and 2.34 for natural sites. The parameterisation of the photosynthesis model, based on a rectangular hyperbola equation (Michaelis & Menten, 1913), revealed site-specific values for the parameters GP_{max}, α and R (see equation 4.2 & Tab. 4.2).

The fit for the relationships was generally very satisfying. Although it is obvious that during the winter period (months 1/2/11/12), the diminished quantity of data and the lack of a variety of radiation intensities (most measurements taken on snow-covered sites) resulted in an inferior fit of the equations. For a comparison of the shape of the graphs, the NEE-PAR relationships for the spring period (5/6) are shown for each site in Fig. 4.2. Generally, the degraded sites D1-D3 are characterised by flat curves, and a light compensation point between 250 and 500 µmol m^{-2}s^{-1}. Towards the natural sites, the curves become steeper, more negative endpoints and light compensation points till 100 µmol m^{-2}s^{-1}.

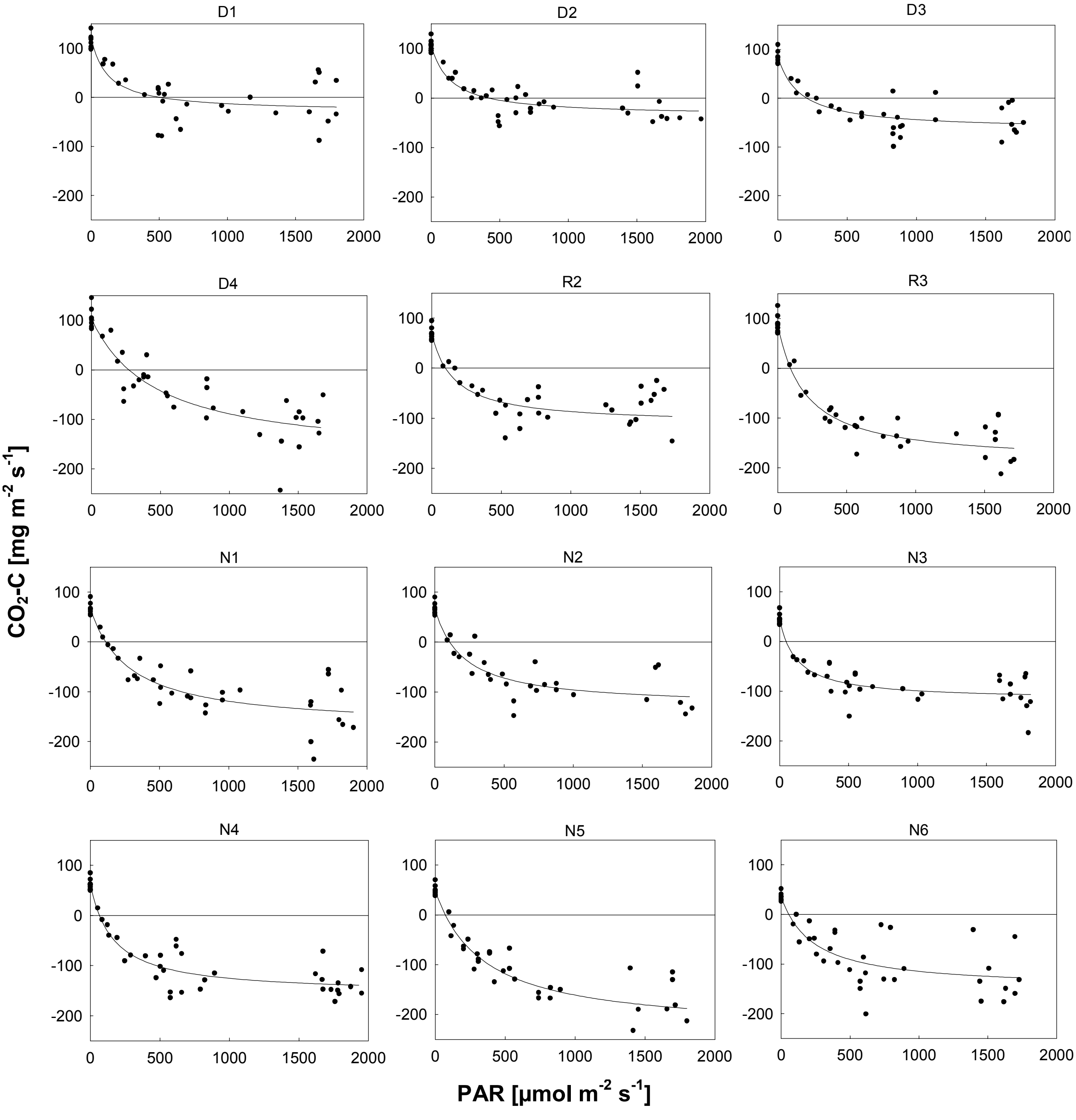


Fig 4.2: Light response curves of Net Ecosystem Exchange (NEE) for all sites in the period May to June. Data set is a mix of weekly measurement days and three days of experimental light response curve measurements. Graphs are rectangular hyperbola (Michaelis and Menten, 1913) fitted to the individual data set (see Tab. 4.2). Sites: D1 dry bog heathland (peat cut); D2 dry bog heathland (peat cut); D3 dry bog heathland (predrained); D4 dry bog heathland (predrained); R1 flooded dry bog heathland; R2 restored moist bog heathland; R3 restored *Sphagnum* lawn; N1 moist bog heathland ; N2 bog shrubs (*Pinus mugo*); N3 *Sphagnum* lawn; N4 *Eriophorum* and *Sphagnum* hummocks; N5 transition hummocks – hollows; N6 *Sphagnum* hollow. For details to the sites see chapter 3.

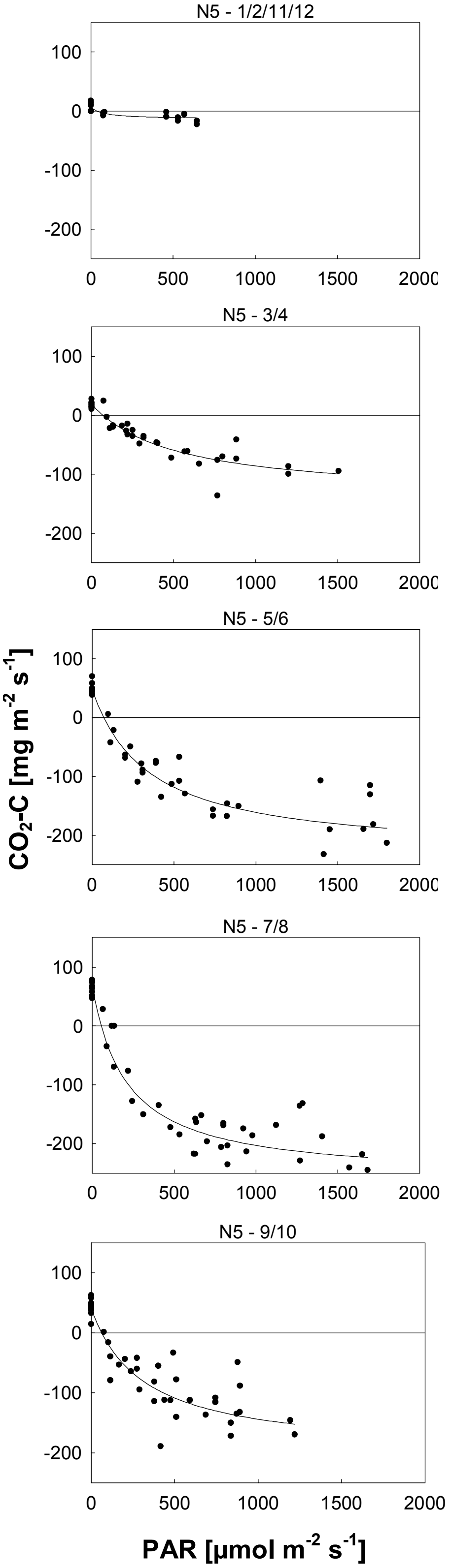


Fig 4.3: Time series of light response curves of Net Ecosystem Exchange (NEE) over the year at site N5 (natural transition between hummocks and hollows). Time steps: month 1/2/11/12, month 3/4, month 5/6, month 7/8 and month 9/10. For details to the sites see chapter 3.

One notable general trend was that the development of the driving variables was reflected by the evolution of the NEE-PAR relationship over the year (Fig. 4.3 & Tab. 4.2). Consequently the initial slopes (α) of the graphs, which represent the light use efficiency, had minimum values during the winter period (see 1/2/11/12 in Tab. 4.2) on all the sites. Additionally, the majority of the sites experienced site-specific maximum values during the late spring period (see 5/6 in Tab. 4.2). Minimum GP_{max} values were also found during the winter period on all sites (see 1/2/11/12 in Tab. 4.2), whereas the maximum values for each site were observed during the summer period (see 7/8 in Tab. 4.2) on the majority of the sites. The evolution of the graphs over the course of the measurement year shows Fig. 4.3 for site N5, which represented the most efficient site in terms of NEE in the group of the natural sites.

Modelled CO_2- fluxes over the course of the year

The model (4.2) was run at a 0.5 h resolution for every site, with individually fit model parameters for the NEE-PAR relationship (Tab. 4.2) and the respiration-temperature relationship (Tab 4.1). Continuous PAR measurements were taken from the climate station located at the centre of the measurement area, with a maximum distance of 150 m to individual sites. Data from the climate station was also used as a continuous data set for soil temperature. The model was run for all of 1999, including the winter period. Fig. 4.4 and Fig. 4.5 illustrate the modelled fluxes plotted as day balances over the course of the year. A special approach was followed to obtain year long results for sites D2 /R1, as D2 was a degraded site until its restoration (17th of July) during the measurement period. R1 represents the site after the restoration. The model for D2 was run in similar mode to the other sites until the restoration. After flooding, consistent respiration-temperature and NEE-PAR relationships could no longer be found.

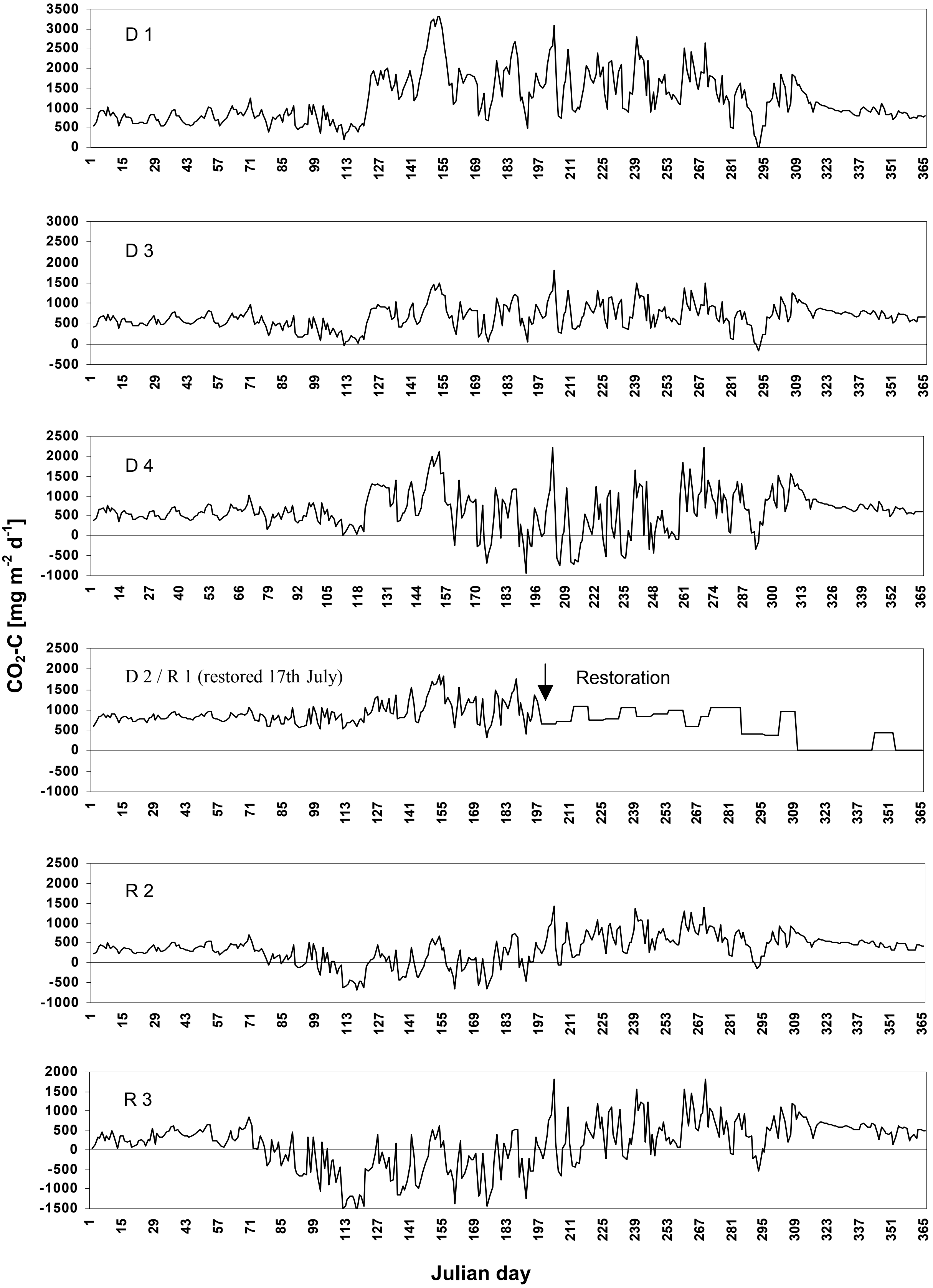


Fig 4.4: Modelled year course of NEE on degraded and restored sites. The model was run in half-hour time steps. Graphs show day balances for the 1999 model period.

Sites: D1 dry bog heathland (peat cut); D2 dry bog heathland (peat cut); D3 dry bog heathland (predrained); D4 dry bog heathland (predrained); R1 flooded dry bog heathland; R2 restored moist bog heathland; R3 restored *Sphagnum* lawn. For details to the sites see chapter 3.

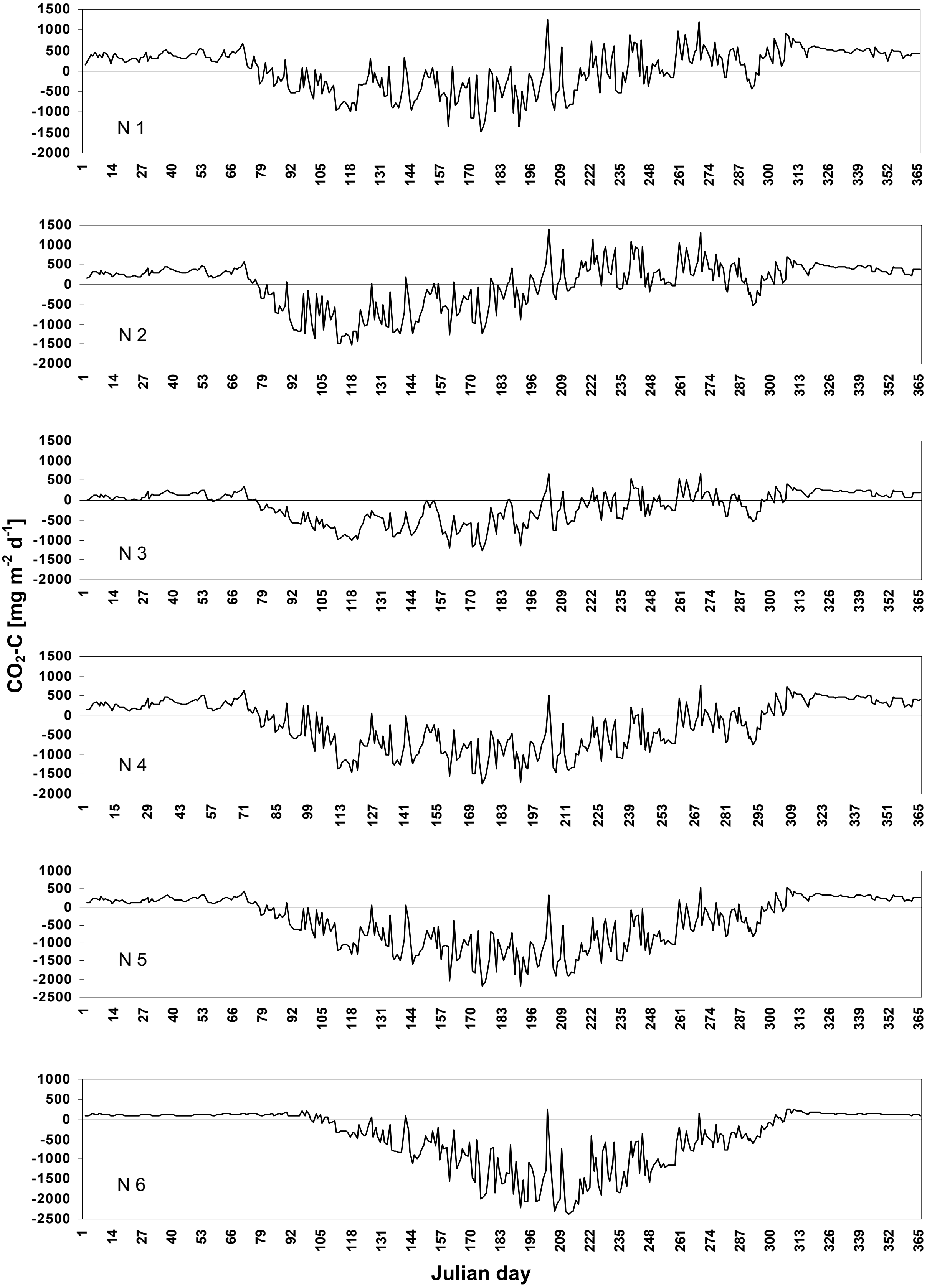


Fig 4.5: Modelled year course of NEE on natural sites. The model was run in half-hour time steps. Graphs show day balances for the 1999 model period.

Sites: N1 moist bog heathland; N2 bog shrubs (*Pinus mugo*); N3 *Sphagnum* lawn; N4 *Eriophorum* and *Sphagnum* hummocks; N5 transition hummocks – hollows; N6 *Sphagnum* hollow. For details to the sites see chapter 3. For details to the sites see chapter 3.

Therefore a simple approach of interpolating between the means of the weekly measurements for the restored and flooded site R1 (like Kamp 1999) was followed.

Generally the curves for the degraded sites D1, D2/R1, and D3 consist almost exclusively of positive values (Fig. 4.4). This means that the ecosystems lost carbon dioxide on almost every day. Site D1 was the extreme, experiencing several days with maximum releases of more than 3000 mg CO_2-C $m^{-2}d^{-1}$ and notably high carbon losses outside the growing season of around 1000 mg CO_2-C $m^{-2}d^{-1}$. During the summer (Julian days 160 to 260) site D4 had some negative flux values, representing days with net carbon dioxide uptake of up to -1000 mg CO_2-C $m^{-2}d^{-1}$ for the ecosystem; however, positive fluxes outside of the growing season remained high. This site acted as transition to sites R2 & R3 (Fig. 4.4) and N1 & N2 (Fig. 4.5), which experienced increasingly frequent uptake days. Days with net uptakes started relatively early in the season around Julian day 80 and reached notable ranges of up to -1500 mg CO_2-C $m^{-2}d^{-1}$. On these sites, the negative springtime values (uptake days) tended to switch back to positive values around Julian day 210 and then behaved similarly to degraded sites D3 and D4, but at a lower release rate of approximately 500 mg CO_2-C $m^{-2}d^{-1}$. The frequency of uptake days constantly rose from site N3, to N4, to N5, along with the amplitude of the negative values, which experienced maximum rates of more than -2000 mg CO_2-C $m^{-2}d^{-1}$. Finally, N6 had fewer uptake days, but demonstrated the lowest winter release rates of less than 200 mg CO_2-C $m^{-2}d^{-1}$ and the highest uptake values of nearly -2500 mg CO_2-C $m^{-2}d^{-1}$ (see Fig. 4.5).

	max uptake [mg $m^{-2}h^{-1}$]	mean flux [mg $m^{-2}h^{-1}$]	max release [mg $m^{-2}h^{-1}$]	max uptake [g m^{-2} d^{-1}]	mean flux [g m^{-2} d^{-1}]	max release [g m^{-2} d^{-1}]	year balance ± SE [g m^{-2} a^{-1}]	percent SE from mean year balance
D1	-154	51	261	-0.07	1.21	3.31	443 ± 70	16
D2	-77	38	176	0.31	0.92	1.86	360 ± 25	7
D3	-110	28	162	-0.16	0.67	1.79	246 ± 46	19
D4	-249	26	238	-0.95	0.62	2.21	225 ± 58	26
R1	0	24	45	0	0.57	1.07	192 ± 21	11
R2	-144	14	151	-0.67	0.35	1.42	127 ± 56	44
R3	-239	7	208	-1.55	0.17	1.82	62 ± 65	105
N1	-197	2	143	-1.47	0.04	1.25	14 ± 52	341
N2	-187	1	143	-1.51	0.01	1.42	4 ± 42	1050
N3	-159	-6	123	-1.26	-0.15	0.68	-54 ± 31	58
N4	-214	-10	134	-1.73	-0.24	0.75	-87 ± 33	38
N5	-241	-17	118	-2.19	-0.42	0.55	-152 ± 48	32
N6	-267	-18	93	-2.36	-0.43	0.25	-157 ± 37	24

Tab 4.3: Comparison of maximum, mean and minimum CO_2-C fluxes at the hourly and daily level and of the NEP-year balance (biophysical model) for natural, restored, and degraded bog plots.
Sites: D1 dry bog heathland (peat cut); D2 dry bog heathland (peat cut); D3 dry bog heathland (predrained); D4 dry bog heathland (predrained); R1 flooded dry bog heathland; R2 restored moist bog heathland; R3 restored *Sphagnum* lawn; N1 moist bog heathland; N2 bog shrubs (*Pinus mugo*); N3 *Sphagnum* lawn; N4 *Eriophorum* and *Sphagnum* hummocks; N5 transition hummocks – hollows; N6 *Sphagnum* hollow. For details to the sites see chapter 3.

The model fit with measured data is exemplarily shown (Fig 4.6) for site N5. The fit is very satisfying. However, both for NEE-uptake (negative values) and -release (positive values) the model is slightly underestimating the measured data, compensating each other for the balances.

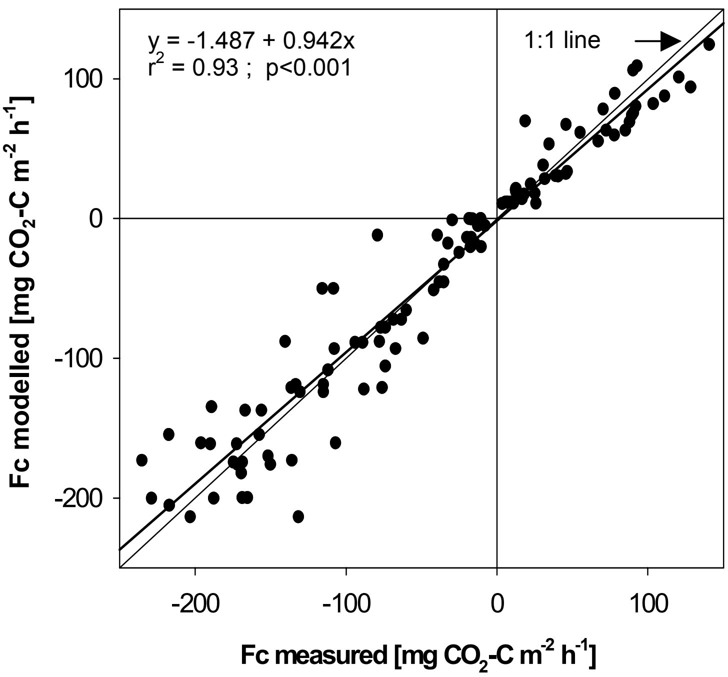


Fig 4.6: Fit of modelled with measured CO_2-C flux data, shown for site N5 (transition hummocks – hollows).

NEP- balance

The NEP-balance for the whole year was calculated by integrating the positive and negative values of the NEE curves from the model-output (see Fig. 4.4 and 4.5) for every site. For sites D2 and R1 the annual balances were obtained by extrapolating the balances before and after the restoration date (July 17th). As an extrapolation factor, the relation of the balance till and after this date of site D1 was used. Additionally, the days with frost on site R1 were determined to be similarly distributed (31 days in total before and after restoration). This is an important factor since CO_2-C emissions were blocked during frozen conditions on the flooded restored site R1 and on the natural hollow N6.

The release and uptake behaviour of the sites over the year (1999) followed a disturbance gradient (see Fig. 4.7 and Tab. 4.3). The degraded sites were the most important CO_2-C sources with emissions of up to 443 ± 70 g $m^{-2}a^{-1}$, while the majority of natural sites acted as CO_2-C sinks with uptake rates up to –157 ± 37 g $m^{-2}a^{-1}$. However, the natural moist bog heathland (N1) and the bog shrubs (site N2) oscillated around zero with 14 ± 52 and 4 ± 42 g CO_2-C $m^{-2}a^{-1}$ respectively. Restored sites still released carbon, but to a much lesser extent than the corresponding degraded sites (D3 -> R2; D1 -> R3). Restoration of site D2 reduced its role as a CO_2-C source notably within the first year by nearly 50%. Uncertainty about the balances was estimated by summing up the maximum and minimum standard errors (SE) of the respiration term and the gross productivity term of the models used (4.2, 4.3). Minimum and maximum SE were calculated by running the models with the single model parameters E_0,

R_{10}, GP_{max}, α, with plus or minus the SE of the parameter. The relative uncertainty (see Tab 4.3) was most pronounced on sites N1 and N2, where mean fluxes were close to zero and SE obtained positive or negative values, suggesting that the sites may act as sinks or sources for CO_2-C (see Fig. 4.7 As in similar approaches (Bubier et al., 1999) an ANOVA was not be applied to compare the NEP´s between the sites, as the presented dataset consisted by the mean and the calculated standard errors (by running the model with the parameters (a, GPmax, Eo, Rref) plus/minus their individual standard errors) exclusively.

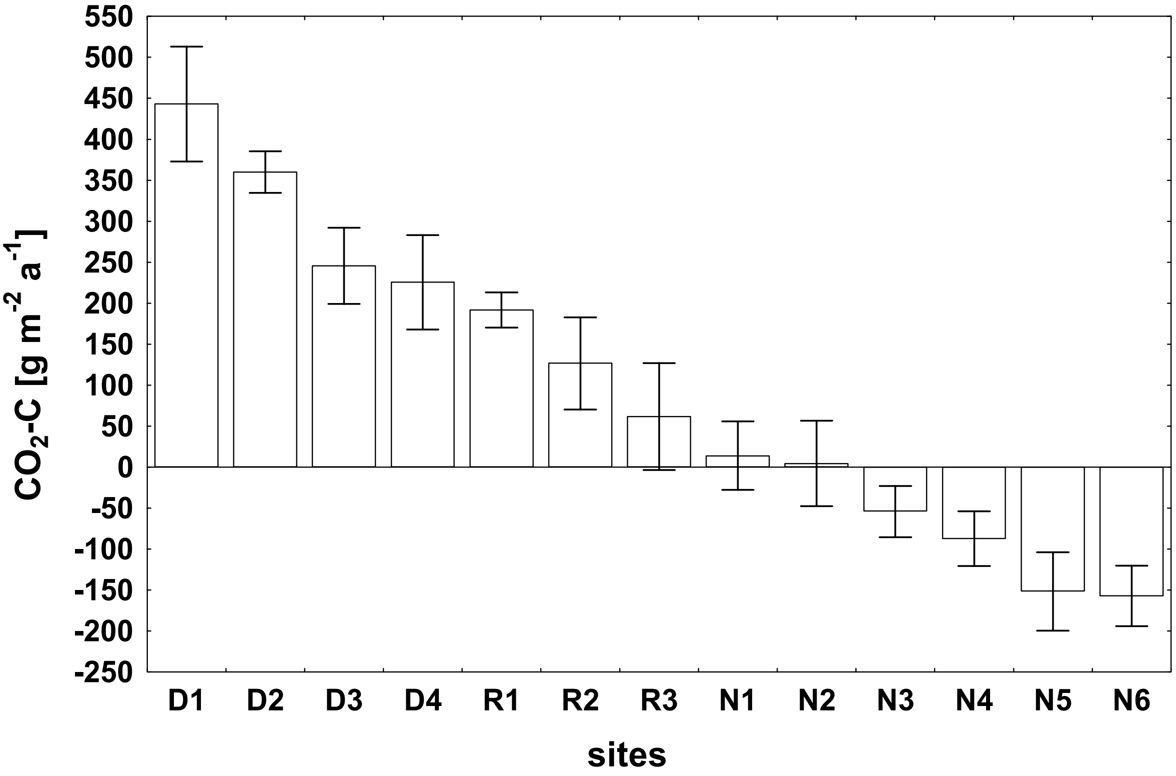


Fig 4.7: Annual balances for CO_2-C exchange: Integration of the modelled year courses of NEE (Figs. 4 & 5). Error bars are SE.
Sites: D1 dry bog heathland (peat cut); D2 dry bog heathland (peat cut); D3 dry bog heathland (predrained); D4 dry bog heathland (predrained); R1 flooded dry bog heathland; R2 restored moist bog heathland; R3 restored *Sphagnum* lawn; N1 moist bog heathland; N2 bog shrubs (*Pinus mugo*); N3 *Sphagnum* lawn; N4 *Eriophorum* and *Sphagnum* hummocks; N5 transition hummocks – hollows; N6 *Sphagnum* hollow. For details to the sites see chapter 3.

Explanation of NEP annual balance with site indicators

The modelled NEE fluxes (Fig. 4.4 and 4.5) over the course of the year and the summed NEP balances (Fig. 4.7) could be sorted out along a disturbance gradient. Therefore the regression between site factors and NEP was tested (Tab. 4.4). Further analysis was done with the three best correlating factors. Water table measurements, nutrient status and vegetation are often used as indicators for site conditions in bog-ecosystems. Disturbances severely influence these site factors; which proved to be highly correlated with NEP annual balances (Tab 4.4). The interaction between abiotic site factors and the vegetation composition is outlined in detail in chapter 3. Site-specific data on water table (WT), Electrical Conductivity (EC) and leaf area index (LAI) was taken from this chapter.

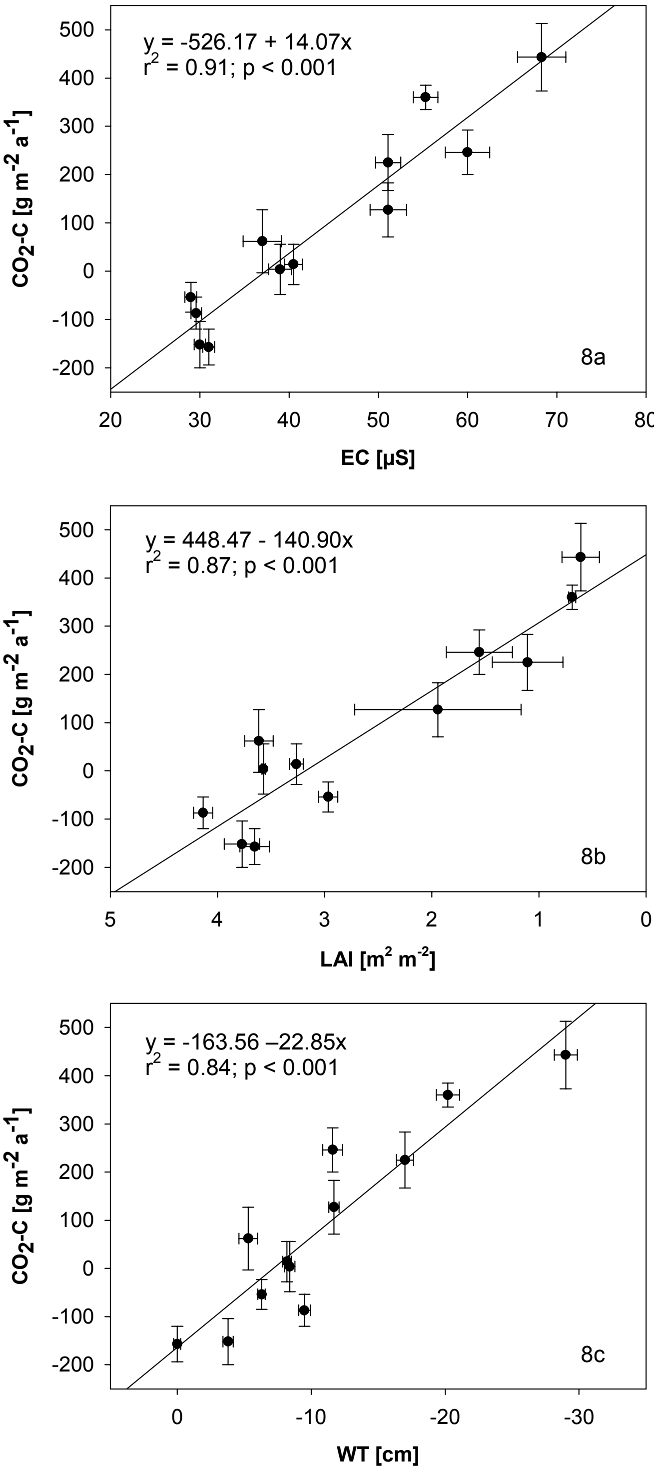


Fig 4.8: Explanation of CO_2- C balances with site parameters (mean values for the whole measurement year).
4.8a: Regression with mean electric conductivity (EC).
4.8b: Regression with leaf area index (LAI).
4.8c: Regression with mean water table (WT).
Error bars are SE.

	R^2	p<
EC	0.91	0.001
LAI	0.87	0.001
WT	0.84	0.001
H	0.81	0.01
K^+	0.72	0.01
C/N	0.59	0.05
pH	-	ns
Ca	-	ns

Tab. 4.4: Regression between site factors and the CO_2-C balance (NEP)

These best correlating parameters are functionally related to the gas-exchange process. Water-table determines the aerated horizon, especially in peat profiles, where the aerobic decomposition processes can take place, in addition to the zones with sufficient soil humidity for soil respiration. The nutrient status is important for the development of specific plant compositions, for the mineralisation processes, and generally can be indicated by the electrical conductivity of the pore water in peat soils. LAI is an indicator of the potential photosynthetic activity of the vegetation types on the sites. Since it was a goal to explain the NEP annual balance, the mean of these factors over the data set for the whole year was used. This is a viable approach as regression of instant fluxes with WT had no significant effect on the explanation level of the temperature-respiration relationship.

Electrical conductivity demonstrated a smooth dynamic over the year, and also did not provide an explanation value for instant fluxes. The leaf area index in moss and heath dominated bog ecosystems does not change significantly over the year, in contrast to more dynamic systems like broadleaf forests and grasslands. Single species, such as *Scheuchzeria palustris* on the wet site N6, show phenological development during the year, but LAI changes over the year were not pronounced. Therefore, as with coniferous forests, a single LAI-value was used for the entire year. Regression analysis revealed highly significant relationships between the selected site factors and the NEP annual balance (r^2 between 0.84 and 0.91), as shown in Fig. 4.8.

As a final step, the regression strength of a multiple linear regression analysis was tested, using all combinations of site factors to explain the NEP-balance. Only a multiple regression with the water-table (WT) and electrical conductivity (EC) factors was significant and could enhance the regression strength in comparison to the single factor relations (r^2= 0.95; see Tab. 4.5). So 95 % of the variability of the NEP annual balance could be explained by the prediction via the multiple linear regression with WT and EC (Fig. 4.9).

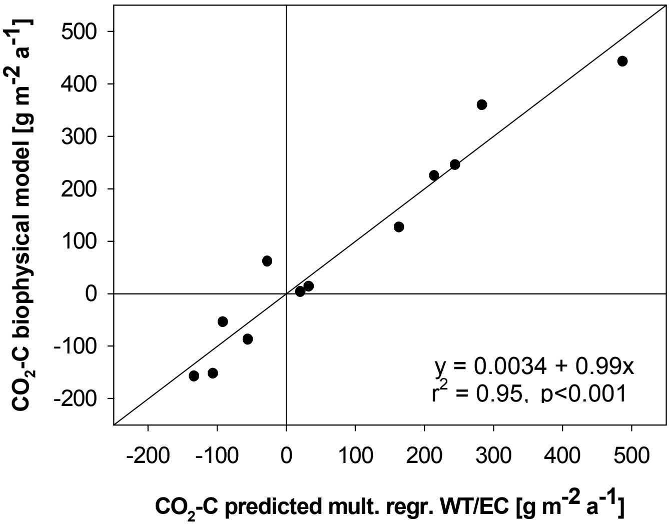


Fig 4.9: Relationship of modelled CO_2-C balance to predicted CO_2-C balance based on a multiple linear regression with Electric Conductivity (EC) and mean water-table (WT).

N = 12, r^2=0.95, p<0.001	Beta ± SE	B ± SE	t (9)	p <
constant		-418.78 ± 66.74	-6.27	0.001
EC	0.62 ± 0.15	9.21 ± 2.22	4.14	0.01
WT	-0.38 ± 0.15	-9.55 ± 3.75	-2.55	0.05

Tab 4.5. Multiple regression for the explanation of the CO_2-C year balance (NEP) with water-table (WT) and Electrical Conductivity (EC).

Discussion

NEE-fluxes

Some difficulties associated with the assessment of gas-balances are seasonality and spatial variation. In this study therefore, a weekly or twice weekly measurement rhythm was established for the complete year of 1999 and extended to mid 2000. This allowed for the parameterisation of the model with seasonality. Spatial variation was tried to fully include by a large set of compared sites (see chapter 3) along the disturbance gradient.

The individual NEE-behaviour of the studied sites during the year course (see Fig.4.4 and 4.5) was obvious not only during the growing season. Late and early winter fluxes were very low on natural sites, but rose to relevant levels for the annual balances on degraded and some restored sites.The natural sites typically had higher and less variable uptake values than the degraded and restored ones, which was detectable by both the instant and the modelled flux data. The absolute values were close to those found by other studies in natural and restored boreal peat bogs (Bubier et al. 1998; Whiting 1994; Tuittila 2000, Lafleur et al. 2001). However, the values were a bit lower than CO_2 uptake and respiration rates measured in a Canadian boreal fen (Suyker et al. 1997) indicating the importance of the peatland type. Maximum uptake rates of –5 to –6 µmol CO_2 $m^{-2}s^{-1}$ (representing -215 to -255 mg CO_2-C $m^{-2}h^{-1}$) found for western Russian and Siberian bogs (Arneth et al. 2002) were very similar to the maximum uptake rates measured at the studied sites (see Fig. 4.2, 4.3 and Tab. 4.3). Thus, at least during the growing period, site-specific factors and vegetation type seemed to be more important for the instant flux rates than the location of the sites in different climate zones. Rising R_{ECO} and GPP along latitudinal gradients compensate each other with the effect, that NEE remains comparable between similar ecosystem types (Christensen et al. 2003).

The daily NEE values between –2.36 g CO_2-C $m^{-2}d^{-1}$ to 3.3 g CO_2-C $m^{-2}d^{-1}$ had similar ranges to values obtained in other studies, which were primarily from northern latitudes (Whiting 1994; Alm et al. 1997; Frolking et al. 1998; Waddington et al. 1998). Differences in NEE between distinctive natural peatland ecosystems were reported by several authors (Whiting 1994; Waddington & Roulet 1996; Alm et al. 1997; Bellisario et al., 1998), with hummocks being found to be more efficient than lawns in their carbon fixation capacity. Waddington et al. (1998) found a similar relationship in daily NEE, with shallow hollows (corresponding to site N6) > lawns (corresponding to site N3/N5) > hummocks (corresponding to site N4) > large hummocks (corresponding partly to N1), as in the presented study, although the absolute values were higher for the first two groups. Tuittila et al. (1999) and Waddington & Warner (2001) compared degraded and restored sites, and they found notable shifts in NEE similar to the presented results. Again reaffirming that ecosystem type seems to be more important for the functioning than latitude or climate zone.

The curves derived from daily NEE values over the entire year of 1999 (Fig. 4.4 and 4.5) showed similar patterns when compared with (seasonal) NEE curves from other studies (Bellisario et al., 1998, Waddington & Warner 2001). It was obvious that respiration was lower on the natural sites compared to the degraded ones, where fluxes from the ecosystem to the atmosphere reached notable levels even during winter. As respiration is controlled primarily by soil temperature (Tab. 4.1), it can be expected that respiration on the degraded sites will be more sensitive to climate warming than on natural ones. Waddington et al. (1998) stated based on a modelling experiment, that wet (natural) peatland ecosystems may be insensitive to warmer and drier climate conditions in their functioning as a carbon sink, whereas dry (degraded) ecosystems may become a source of CO_2. Additionally, Turetsky & Wieder (1999) illustrated that *Sphagnum* refixes respired CO_2, which may help to balance higher CO_2 -emission.

In terms of respiration, it was apparent that the regression equations were site specific not only for the factors but also for the correlation with the corresponding soil temperature. Correlation was best with 2 cm soil temperature at almost every site. However, at site N2 correlation was best with the 5 cm soil temperature, and at sites D2 and N6 it was best with the 10 cm soil temperature. The correlation with different depths may be interpreted as an indicator of the location of the major CO_2 producing zones, if the best correlation depth is a functional indicator for the primary production zone. Without tracing the production zones directly some uncertainty remains, as a time-lag between soil temperature dynamics and emissions measured at the surface, affects regression and may have influenced the detection of a functional coupling of the two factors. Process studies have detected the participation of different zones and resources in soil respiration. In Finnish peat bogs, 35-70% of the respiration was derived from roots (Silvola et al. 1996b, Bhardwaj 1997) and between 50 and 70% of CO_2 efflux originated from recent photosynthates (Komulainen et al., 1999). For the present study it was not possible to include this process oriented approach.

To find the best soil temperature depth for a robust modelling procedure, soil temperature was sampled at six different depths parallel to the flux measurements. The modelling fit was best with the temperature used at depth of best relation with Ecosystem respiration. In other studies, respiration was modelled by correlation with soil temperature at a depth of 5 cm exclusively (Waddington & Warner 2001; Waddington et al. 2002) or it correlated best with soil temperature at 5 cm over all plots (Bubier et al. 1998).

A test of several different regression models for analysing the respiration-temperature relationship revealed that the Lloyd & Taylor (1994) equation provided the best fit. However, Bellisario et al. (1998) found linear regression of respiration with air temperature and water table over natural sites explaining best the variability of their specific dataset.

Natural sites (N1-N6) had the highest activation Energies (E_o, Lloyd & Taylor, 1994) with a mean value of 363. These high activation Energies (E_o) can be interpreted as an expression of the resistance of the sites to temperature induced respiration processes at low temperatures. The lowest activation Energies (E_o), representing less resistance to temperature induced respiration, were found on degraded sites (D1-D4) with a mean value of 287. Subke (2002) also found high E_o- values between 320 and 445, for soil respiration in temperate forest floors. Even lower values were found in drought influenced Mediterranean ecosystems with E_o- values between 185 and 263 (Reichstein et al., 2002).

NEP and its uncertainty

The NEP budget for the whole year was derived by integrating the modelled NEE over the course of the year. The model was run over the whole measurement year, as the measurement rhythm (weekly to twice a week) allowed for the coverage of seasonal fluxes, via parameterisation of the model in time steps (see Fig. 4.3, Tab. 4.1 & 4.2). This was an expansion in comparison to other studies, which applied the model only for the growing season (Whiting 1994, Bellisario et al., 1998, Bubier et al. 1998, Waddington & Warner 2001). The presented study also modelled the wintertime fluxes, based on a corresponding data set of directly measured fluxes during the snow-covered season.

Other studies have found wintertime CO_2 efflux (R_{ECO}) from arctic soils to contribute between 7 and 25 % to the annual balance (Mariko et al. 1994, Fahnestock et al. 1999, Jones et al. 1999, Alm et al. 1999a,b). This range is similar to the range found on degraded sites (D1-D4) and the restored site R2, which had contributions of 21-37 % from winter fluxes to the annual balance. However, winter efflux was of much greater importance for the annual balance on sites which were close to zero in their mean NEP annual balance. These sites consisted of R3 (78 % winter to annual), N1 (330%) and N2 (876%). On the natural sites N3 to N6 the mean NEP was negative (uptake to the ecosystem), therefore the relation of the winter-fluxes to the mean NEP was expressed as a negative value (-9 to –45 %). The substantial contribution of winter-fluxes suggests that without inclusion of the wintertime values in the calculation of the NEP, a notable over- or underestimation would result for the final balance. Subke (2002) also included wintertime respiration for the estimation of annual balances. The greater importance of winter fluxes on the annual balance for sites R3, N1 and N2 compared to the ranges found by Mariko et al. (1994), Fahnestock et al. (1999), Jones et al. (1999) or Alm et al. (1999a,b) can be explained by the low mean fluxes in relation to the SE in the presented study. The main reason for this was that the entire year NEP´s were compared to winter NEP (represented by respiration). In the cited studies, the estimates for the winter to non-winter relationship were done exclusively for respiration.

The mean NEP annual balances over all sites (Fig. 4.7 and Tab. 4.3) closely followed a disturbance gradient (in contrast to the findings of Marinier, 2003), with high emissions coming from degraded sites (225 ± 58 to 443 ± 70 g CO_2-C $m^{-2}a^{-1}$), moderate emissions on restored sites (62 ± 65 to 127 ± 56 g CO_2-C $m^{-2}a^{-1}$) and a range from small emissions to notable sequestration rates on natural sites (14 ± 52 to –157 ± 48 g CO_2-C $m^{-2}a^{-1}$). Considering the magnitude of the winter losses cited above, these data had values that were similar to other studies with a seasonal focus (Whiting 1994, Alm et al. 1997, Bellisario et al., 1998, Tuittila et al. 1999, Waddington & Roulet 2000). Thus the type of ecosystem seems to behave similar, but the latitudinal position may affect the annual balances, because of the different duration of the growth period.

The maximum uptake rates on natural sites N5 and N6 seemed relatively high, with an average CO_2-C balance for the natural sites (N1-N6) of -72 ± 40.5 g CO_2-C $m^{-2}a^{-1}$. However this rate was similar to the balance obtained by Lafleur et al. (2001) for a raised Canadian bog, measured with eddy covariance technique, which integrates the fluxes over areas that are several orders of magnitude larger than the plot area used in the presented study. Thus the average over all natural sites (N1-N6) may reflect best the CO_2-C exchange of a natural patterned bog surface.

The high level of CO_2 losses from the degraded sites corresponds with the findings of Waddington & Warner (2001) and is likely to remain elevated for many years after cessation of peat extraction (Waddington et al., 2002). This is of particular concern because even site D1, which was abandoned more than 50 years ago, demonstrated no self-restoration effects and was considered a severely degraded site, with a constant loss of carbon. Conversally, Liski et al. (1999) stated that the CO_2 emissions from soils are generally overestimated (if not measured during the entire year cycle in the field), as the decomposition of old soil organic matter reduced due to the persistence of the remaining C fractions. A cautious evaluation of the refractory components of the soil organic matter would be necessary, to determine the onsite decomposition risk of an individual peat layer. The magnitude of emissions demonstrated that the studied degraded sites were still acting as pronounced sources for carbon, in terms of absolute values when compared to the restored or even natural sites. The restored sites were still not acting as C sinks, but restoration may considerably reduce the amount of emissions in comparison to the degraded sites. For example, the restoration of site D1 to site R3 offered a reduction of around 380 g C $m^{-2}a^{-1}$. Restoration of sites D3/D4 to site R2 created a reduction of around 120/100 g C $m^{-2}a^{-1}$. Even within one year of the restoration of site D2 to site R1, a reduction of around 170 g C $m^{-2}a^{-1}$ was achieved. Restoration may not directly create an absolute carbon- sink, but it can offer a notable carbon-sink function relatively, when compared to the corresponding degraded sites. The possibility of actively mitigating CO_2-C emissions via relatively easy, cheap and effective restoration measures, has recently received special attention from a group assessing the status of the present greenhouse gas budget and mitigation in the German biosphere (Freibauer et al., in prep.).

For an estimation of the uncertainty, the models were first run with the mean parameter values plus or minus the parameter SE, and then the resulting maximum and minimum SE for the respiration term and the photosynthesis term were summed up to get the overall SE of the NEP estimation (see Fig 4.7 & Tab 4.3; this procedure was influenced by Waddington and Roulet (2000)). The uncertainty of the absolute NEP estimation was between 21 and 70 g $m^{-2}a^{-1}$, which is similar to but slightly higher than the value found by Bubier et al. (1999). The relative importance of the uncertainty rose with decreasing mean NEP values, and ranged from 7 % (site D2) to 1050 % (site N2; see Tab. 4.3). On sites R3, N1 and N2, uncertainty was greater than 100%, which means that these sites probably acted both as sinks or sources for CO2-C. On all other sites, uncertainty was between 7 % (site D2) and 58 % (site

N3); therefore, the general behaviour of these sites as sinks or sources was not influenced by the uncertainty term. The reason being for these relatively high uncertainties (which were in their absolute range not significantly different than in the other sites) that R3, N1 and N2, showed mean fluxes close to zero and under these conditions the SE was much more important relatively.

The modelled NEP values are based on an intensive single year study. Trace gas exchange of ecosystems reacts sensitive to climate and therefore interannual variability of fluxes may be significant (Arneth et al. 2002). However the climate of the measurement year was rather similar to the 30- average, with almost identical precipitation sums and just one 1 °C higher mean temperature. Arneth et al. (2002) showed that NEE curves did not vary significantly over three consecutive years in an Siberian bog, as climate was very similar between the three years as well. So the derived NEP-values should be taken as good estimate of an average behaviour of the sites. Moreover the relation between the sites should be relatively stable, as the NEP-balances are expressions of the site functioning and closely related to site factors (s. below).

Indication of the NEP balance with site parameters

The best correlating site factors with NEP were mean water-table, electrical conductivity and LAI of the plant cover. These parameters are strongly related to the functioning of a bog ecosystem (Göttlich, 1990, Succow & Joosten 2001). The water-table determines the site conditions for plant growth, and thus the specific vegetation composition and fitness. Water-table also controls the dynamic thickness of the aerated peat layer, creating the conditions necessary for aerobic decomposition (respiration). Electrical conductivity of the soil water is an easily measurable indicator for the nutrient status of a peat-land site (Pfadenhauer, 1997). The vegetation is important both as an active layer for gas exchange (photosynthesis and respiration) and as an indicator of the site conditions (water-table, soil-status). LAI (leaf area index) was used in the present study as a functional indicator of the vegetation status. Fig. 4.8 shows that the explanation range of the different variables was high, with r^2 values between 0.84 and 0.91, for the water-table, electrical conductivity and LAI variables. Just by using the simple indicator of LAI, 87% of the variability of the NEP over the sites could be explained. Bubier et al. (1999) also found NEP to be linked to vegetation types, increasing in the order of bog, to rich fen, to intermediate fen, and finally to poor fen. Even in restored cut-over bogs, the cover of the dominating single species (*Eriophorum vaginatum*) was determined to be a good predictor for the NEP of the sites (Tuittila et al., 1999).

Multiple linear regression methods were also applied to test if a combination of the different site parameters could enhance the explanatory value of the regression with NEP. Only the combination of water-table (WT) and electrical conductivity (EC) produced significant results (See Tab. 4.5). With this multiple regression approach, 95% of the variability of the NEP over the sites could be explained.

Conclusions

This study on the carbon exchange of bog ecosystems in southern Germany revealed that NEP was site specific and followed a gradient from natural, to restored, to degraded ecosystems. For the modelling of NEE, a parameterisation for five consecutive time periods throughout the year produced the best match with seasonality of the driving variables (radiation and temperature). Restoration was able to reduce the source function of degraded sites significantly, but it did not lead to the immediate development of CO_2-C sinks. Restoration was therefore a viable means of reducing CO_2–C emissions from degraded bog ecosystems. Highly significant relationships between NEP and simple indicators like leaf area index (LAI), water-table (WT) and electrical conductivity (EC) offer the option for spatial extrapolation of NEP via vegetation maps, or records of mean water-table and electrical conductivity, with input data derived from ground-sampling and remote sensing. As a result of inter-annual changes in gas-exchange behaviour, it is necessary to model the NEP with a model driven by the major forces of radiation and temperature. Once the NEP for a span of ecosystems was found, spatial extrapolation can be performed based on simple site parameters.

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METHANE AND NITROUS OXIDE FLUXES OF NATURAL, DEGRADED AND RESTORED BOG ECOSYSTEMS

Summary

Methane and nitrous oxide fluxes were measured weekly over an entire year period (1999) on 12 sites of bog ecosystems in Southern Germany using a newly developed closed chamber method. The aim was first, to identify the effect of restoration on trace gas fluxes via a comparison of degraded, restored and natural sites. The second goal was to explain the variability of the instant fluxes and of the annual balances with environmental variables, net ecosystem CO_2-exchange (NEE) and net ecosystem productivity (NEP). Natural sites, especially the natural *Sphagnum* hollow, had the highest mean instant methane fluxes at 4.63 mg CH_4-C $m^{-2}h^{-1}$, whereas on the drained and former peat cut sites, the lowest mean instant methane fluxes of 0.003 mg CH_4-C $m^{-2}h^{-1}$ were obtained. The means of the restored sites fell between (0.21 – 0.76 mg CH_4-C $m^{-2}h^{-1}$). The annual balances separated the sites according to a degradation gradient with the highest balances at the natural sites (up to 38.2 ± 2.2 g CH_4-C $m^{-2}a^{-1}$), over the restored sites (1.5 ± 0.2 – 7.1 ± 3.1 g CH_4-C $m^{-2}a^{-1}$), to the lowest balances at the degraded sites (minimum of 0.03 ± 0.02 g CH_4-C $m^{-2}a^{-1}$). The instant fluxes could be explained on the natural sites via multiple regression with NEE and soil temperature at 50 cm depth up to 89 % ($p<0.001$) at the *Sphagnum* hollow. But NEE was only a good predictor of methane-fluxes on the three major methane-emitting sites. On the other sites, soil temperature at different depths was the best predictor for the instant fluxes, whereas the low and scattered fluxes of the two most degraded sites (drained and former peat cut) could not be explained with any environmental variable. The variability of the annual balances between the sites could be explained best by a single exponential regression with NEP ($r^2 = 0.87$; $p < 0.001$). This indicates the important role of the carbon dioxide exchange for the methane balance of the sites. The explanation level by regression with water table and/or aerenchymous leaves was lower than this single NEP-regression.

At the nutrient poor bog ecosystems included in the study, measured nitrous oxide fluxes were low and scattered and could not be explained by any environmental variable. The annual balances for the natural and restored sites were oscillating around zero (1.8 to –2.3 mg N_2O-N $m^{-2}a^{-1}$), and were notable only on the degraded sites with calculated annual balances up to 168 ± 94 mg N_2O-N $m^{-2}a^{-1}$. The non-normal distribution of the annual balances over the sites, which could not be transformed to a normal distribution, allowed only non-parametric procedures to analyse the relation to environmental variables. The strongest relation was detected between nitrous oxide annual balance and electric conductivity (Spearman R=0.94) and NEP (Spearman R=0.87). Restoration leads to increased methane emissions but cuts off nitrous oxide emissions of bogs.

Introduction

Ecosystems take part in the global biogeochemical cycles by matter- and energy-exchange. The release and uptake of the greenhouse-gases carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O), especially from wetland ecosystems, has been identified as important for global climate change (Gorham, 1991). In molar matter fluxes methane and nitrous oxide are normally irrelevant in comparison to the dominating CO_2-fluxes. However, in mires, due to reduced decomposition rates that result from waterlogged conditions, methane emissions can rise to considerable levels, influencing the carbon balance of the ecosystems. Moreover, the individual climatic relevance (GWP) is 21 times higher for methane and 310 times higher for nitrous oxide than for carbon dioxide (100a timescale, IPCC, 1996). Therefore, methane and nitrous oxide should be included in studies on the gas exchange of ecosystems, with a focus on the assessment of the climatic relevance of the sites (Martikainen, 1996).

Methane production is a product of mostly acetate fermentation by methanogenic bacteria under anaerobic conditions (Lloyd et al., 1998, Yavitt & Williams, 2000). Methane production in the waterlogged peat column is reported to be much higher than the emission rate of methane measured on the peat or wetland surface (Edwards et al., 1998). That is a result of subsequent methane consumption via oxidation of methane to carbon dioxide by methanotrophic bacteria in the aerated zone of the peat profile (Sundh et al., 1995, Segers, 1998). However, the relation of produced to emitted methane is not only dependent on the efficiency of the oxidation processes, but on the emission pathways as well (Roura-Carol & Freeman, 1999, Joabsson et al., 1999). For methane emissions, three emission pathways were identified: i) diffusion from the methane producing area through the peat to the atmosphere, ii) ebullition in almost totally waterlogged situations and iii) diffusion through plant roots and shoots with aerenchyma (Walter & Heimann, 2000).

These emissions of methane from wetlands are reported to be controlled by a set of varied environmental factors and plant species. The major factors are peat temperature as a controller of the CH_4 production and oxidation rates (Daulat & Clymo, 1998, Hargraeves & Fowler, 1998, Tuittila et al., 2000), the water-table as an indicator of the potential zones of production/consumption due to aerobic and anaerobic conditions (Reeve et al., 1997, Tuittila et al., 2000, Walter & Heimann, 2000), and the appearance of species with by-passing effect for the CH_4 emission (Shannon et al., 1996, Frenzel & Rudolf, 1998). These are species like *Eriophorum vaginatum* and *Scheuchzeria palustris* with aerenchymous tissue, where methane, which enters the plants through the root surface, passes by without being oxidized by methanotrophic bacteria. Joabsson & Christensen (2001) found emission rates of methane driven by the appearance and cover of vascular plants. Besides the bypassing effect, this seems to be linked to the easily degradable root exudates in the rhizosphere of the active plant layer. Photosynthesis should play an important role in methanogenesis, as increased carbon assimilation leads to higher allocations of labile carbon to the roots, which finally can be found in the rhizosphere. Consequently, instant CO_2 net ecosystem exchange (NEE) rates appear to be correlated to methane emissions (Whiting & Chanton, 1993, Bubier, 1995, Waddington & Roulet, 1996, Bellisario et al., 1999, Christensen et al. 2000, Joabsson & Christensen 2001). Higher CH_4-concentration in the root zones of vegetated plots, in comparison to plots where vegetation was clipped, confirm the functional interactions between plant-productivity and methanogenesis (Joabsson et al. 1999). Therefore, NEE may be a useful indicator for methane fluxes or balances, but the correlations should be determined individually for every new study site (Joabsson & Christensen 2001).

As further predictive variable related to plant cover, the abundance of indicator species from the moss-layer following the water-table was found to be linked to methane gas-exchange (Bubier et al., 1995). The water table establishes the anaerobic zone available for methanogenesis, as well as the depth of the aerated zone, which determines the potential oxidation processes of methanotrophic bacteria, using CH_4 as substrate and releasing CO_2.

The two important processes for nitrous oxide production are autotrophic nitrification of Ammonia to Nitrate and heterotrophic denitrification of Nitrate to Nitrogen (Davidson, 1991). The "hole in the pipe" model set forth by Davidson (1991) resumes the different processes for N_2O production and emission. Granli & Bockmann (1994) listed several controlling factors for nitrous oxide emissions. Among those, ammonia and nitrate contents, soil temperature and soil humidity seem to be of major importance. However, Ruser (1999) could only find nitrate content as a correlating factor with nitrous oxide emissions in potato fields, and Glatzel & Stahr (2001) did not find any correlation with the aforementioned explaining variables for intensive and abandoned grasslands. Tilsner et al. (2003) on the other hand found the supply of nitrogen by fertilization and the type of fertilization as explaining variables for nitrous oxide emissions on extensively managed grasslands. The difficulty of finding simple relationships between nitrous oxide emissions and controlling factors is better understood since Brumme et al. (1999) developed a classification of N_2O flux patterns for forest ecosystems. Fluxes can be grouped as background, seasonal and event based fluxes. Consequently, background fluxes were not explainable by any environmental variable. Seasonal fluxes follow climatic effects like temperature and event based fluxes seem to be driven mainly by freeze-thaw effects and rain events. However, in bogs with normally high water tables and low nutrient contents, the identification of the controlling variables and the flux type is limited due to the scattered and low N_2O emissions.

The objectives of the study were i) to sample methane and nitrous oxide fluxes over a whole year period across a gradient of natural, degraded and restored bog ecosystems ii) to derive whole year exchange-balances for both gases and iii) to explain the instant fluxes with environmental variables measured in parallel and iv) to find simple but consistent predictors of the annual balances for the individual ecosystems in view of modelling options.

Materials and Methods

CH_4 and N_2O sampling and analysis

The standard sampling schedule was run weekly from the 5th of January until the 28th of December 1999. On each field day, gas exchange was measured by employing a newly developed closed chamber method (chapter 2) on 36 plots, with 3 plots on each of the 12 different sites (N1 as exception with 6 plots). When using the closed chamber method, the change in gas concentration over time and the chamber volume determine the gas flux. Sampling was conducted from boardwalks, which were established six weeks before the measurement period started in November 1998. Nine portable climate controlled closed chambers were applied at the same time, thus with four measurement cycles the entire measurement area was covered in a single day.

One complete measurement cycle, which included moving and preparing the chambers, on one site lasted approximately one hour. The net enclosure time of the chamber lasted between 24 and 32 minutes, depending on the environmental and phenological conditions. Five gas-samples where transferred from the headspace of the chamber to evacuated glass bottles with a syringe. The derived gas samples were brought to the laboratory and analysed on an automated GC-system at the Institute for Soil Ecology, GSF Neuherberg Munich. The whole set of gas-samples (180 per measurement day) were analysed within the next day after sampling in the field, to avoid any effects of prolonged storage.

The gas-samples of the standard program were analysed by gas-chromatography (Shimadzu 14 A and B), with a FID-detector for CH_4 and an ECD-detector for CO_2 and N_2O. The system that was used was developed by Loftfield et al. (1997) and was run at the Institute of Soil Ecology of the GSF, Neuherberg, Germany. The Porapack Q columns (Millipore; 80-100µm) and the ECD of the gas chromatographs were heated to 60 and 280°C, respectively. N_2 (Linde) was used as a carrier gas, with a flux rate of 20 ml min^{-1}. Calibration was carried out automatically every 20 samples with concentrations of 1000, 2000, 5500 ppb for CH_4 and 400, 1600, 4000 ppb for N_2O.

Environmental variables

Parallel to the gas-exchange measurements, a set of environmental variables was also measured, including; air temperature and relative humidity inside and outside the chamber, soil temperature at depths of 2, 5, 10, 20, 50, 100 cm, and water table characteristics in wells close to all collars. Electrical conductivity and pH were measured during the gas-exchange measurements in the wells. The NO_3 and NH_4 concentrations in the pore water was not determined because, as former studies showed, the large amount of humic substances can skew the measurement of low concentrations of nitrate and ammonia in the specific situation of bog pore-water.

Furthermore, air temperature and relative humidity 2 m above ground and in the vegetation layer were recorded during the whole measurement period continuously, and precipitation was determined weekly.

The quantity of leaves of the two species with aerenchyma tissue, *Eriophorum vaginatum* and *Scheuchzeria palustris*, was counted on every plot, as indicator of the total area through which methane could leave the system, thus by-passing the aerated zones in the peat profile with potential for methanotrophic methane consumption. Samples of *Eriophorum* leaves did not vary significantly in diameter between the sites. However, on the two sites with *Scheuchzeria* present (N5 and N6), a cross section of *Scheuchzeria* leaves covered 1.5 (N5) to 3 times (N6) the average surface area of *Eriophorum* leaves (mean of 1.0 mm^2). Assuming that the transport rate of CH_4 through the aerenchyma does not differ between the two species, the quantities of leaves were multiplied in relation to the average surface area for each species. This approach gives a normalised value for aerenchymous leaves with *Eriophorum* as a standard (see chapter 3).

Flux calculation

The convention of negative values representing a loss from the atmosphere to the ecosystem and vice versa was used for this study. The enrichment or rarefaction graphs of the concentration versus time were accepted if r^2 was > 0.95, using at least three points for the linear regression model. From the concentration gradients (dc/dt), flux rates in mg m^{-2} h^{-1} were calculated by the following equation:

$$F = k\,(273 * T^{-1})(V * A^{-1})\,(dc * dt^{-1}) \qquad (5.1)$$

F = flux rate of CH_4 (mg CH_4-C m^{-2} h^{-1}) or N_2O (µg N_2O-N m^{-2} h^{-1})
k = gas-constant by 273.15 K (0.536 µg C µl^{-1} for CH4, 1.25 µg N µl^{-1} for N_2O)
T = instant air temperature during the measurement (K)
V = volume of the chamber (l)
A = surface area within the collar of the chamber (m^2)
$dc*dt^{-1}$ = concentration change in the chamber atmosphere over the time (CH_4 and N_2O: µl l^{-1}h^{-1})

Cumulated annual balances

The annual balances for CH_4 and N_2O were obtained by extrapolation of the weekly measured fluxes between the measurement days and the accumulation of these time steps. This approach was followed because for modelling the methane fluxes at half hour resolution, the relationship to explaining environmental variables were only sufficiently strong on the natural but not on the degraded and restored sites. Due to the low fluxes and high dynamics of nitrous oxide, no relationship for modelling the fluxes at any site could be found. Therefore, a simple, but robust extrapolation procedure between the measurement days was used, taking the measurements as an expression of average fluxes for the intervals. This approach has proven its performance in earlier studies (e.g. Kamp, 1998, Ruser, 1999).

Statistical analysis

Tests of the normality of distribution for each of the variables were undertaken using STATISTICA software. Non-normally distributed variables were normalized via logarithmic or square root transformation, allowing parametric statistical analysis. Non-parametric correlation was applied with STATISTICA, when variables were not normally distributed even after transformation. Single and multiple linear regression analyses were done with STATISTICA, while non-linear regression analysis was done with TABLE CURVE 2D. ANOVA, Tukey test, was applied to the annual balances.

Results

Site specific methane and nitrous oxide fluxes

Measured methane fluxes showed high variability over the course of the year within the sites and were not normally distributed on any site. Fig. 5.1 shows the dynamic of the methane fluxes over the entire year for N6, while Fig. 5.2 illustrates the dynamic of the nitrous oxide fluxes for site D1. D1 was the only site with a more or less pronounced N_2O dynamic over the year. But even so, the uncertainty of the fluxes (range) was notable. All other sites showed scattered fluxes, which did not follow any clear pattern of dynamics over the year. For methane fluxes, the group of natural sites showed similar dynamics to site N6, but was most pronounced on N6.

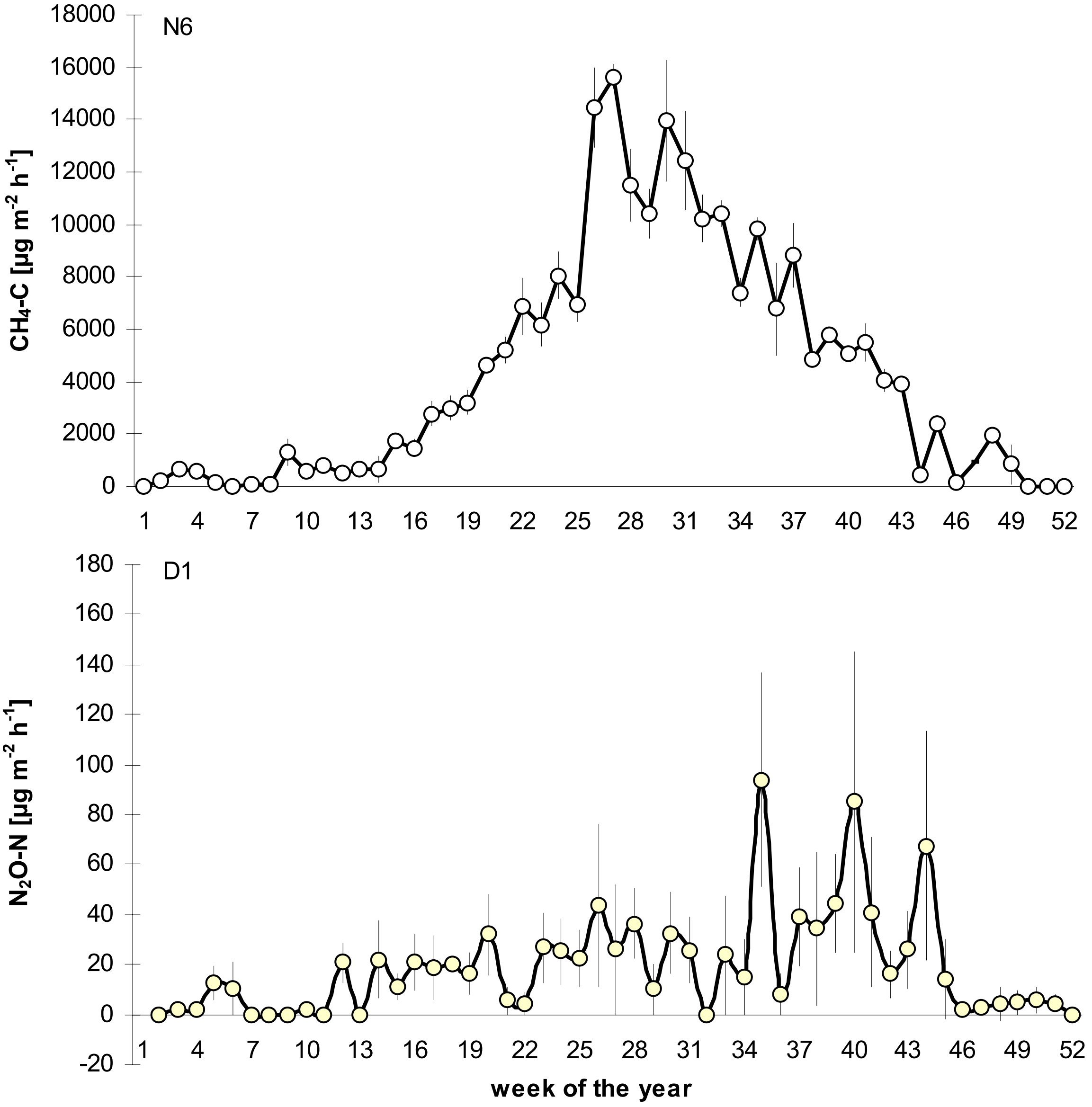

Fig. 5.1 (methane) and 5.2 (nitrous oxide): Seasonality of trace gas fluxes at exemplary sites. Error bars are SE. Methane fluxes at natural site N6 (natural *Sphagnum* hollow) and nitrous oxide fluxes at degraded site D1 (dry bog heathland). For further site description see chapter 3.

The degraded sites D1 and D2 (dry bog heathlands, former peat cut area) showed nearly zero methane fluxes (median of 0 and mean of 0.003 to 0.01 mg CH_4-C $m^{-2}h^{-1}$). Contrastingly, the natural sites N4, N5 and N6 (*Sphagnum-Eriophorum* hummocks to *Sphagnum* hollows) experienced the highest fluxes, with medians of 2.2 to 3.55 mg CH_4-C $m^{-2}h^{-1}$ and means of 3.25 to 4.63 mg CH_4-C $m^{-2}h^{-1}$. The next highest fluxes were found on a second group of natural sites N1, N2 and N3 (moist bog heathland, bog shrubs and *Sphagnum* lawn), with median fluxes from 0.32 to 0.74 mg CH_4-C $m^{-2}h^{-1}$ and mean fluxes of 0.63 to 1.16 mg CH_4-C $m^{-2}h^{-1}$. The restored sites R1, R2 and R3 (flooded bog heathland, restored moist bog heathland and restored *Sphagnum* lawn) showed fluxes close to the second group of natural sites, with medians of 0.11 to 0.28 mg $m^{-2}h^{-1}$ CH_4-C and means of 0.21 to 0.76 mg $m^{-2}h^{-1}$ CH_4-C. Tab. 5.5 illustrates the non-normal distribution of the methane fluxes within the sites, with the gross of the fluxes (median) at a relatively low level in comparison to the maximum fluxes. This was a general pattern over all sites, but was less pronounced in the high flux group (N4-N6) with closer to normal distribution of the fluxes. The fluxes generally followed along a disturbance gradient from degraded, to restored, to natural sites (Tab. 5.3).

	min [mg $m^{-2}h^{-1}$]	median [mg $m^{-2}h^{-1}$]	mean [mg $m^{-2}h^{-1}$]	max [mg $m^{-2}h^{-1}$]	winter balance ± SE [g m^{-2} per^{-1}]	annual balance ± SE [g m^{-2} a^{-1}]	percent SE from annual balance	percent winter balance from annual balance
D1	-0.06	0.00	0.003	0.09	0.01 ± 0.01	0.03 ± 0.02	67	33
D2	-0.01	0.00	0.01	0.28	0.06 ± 0.02	0.08 ± 0.03	38	75
D3	-0.09	0.03	0.19	2.98	0.13 ± 0.05	2.0 ± 0.3	15	7
D4	-0.05	0.07	0.10	0.42	0.11 ± 0.08	0.9 ± 0.3	33	12
R1	0.00	0.12	0.21	1.23	0.10 ± 0.03	1.5 ± 0.2	10	7
R2	0.00	0.28	0.76	10.30	0.83 ± 0.72	7.1 ± 3.1	44	12
R3	-0.10	0.11	0.23	1.54	0.39 ± 0.10	2.3 ± 0.4	18	17
N1	0.01	0.74	1.16	9.44	1.08 ± 0.14	10.7 ± 1.6	15	10
N2	0.00	0.32	0.63	7.99	0.49 ± 0.05	5.4 ± 0.5	9	9
N3	0.00	0.53	1.14	12.26	0.79 ± 0.18	10.1 ± 2.3	23	8
N4	0.00	2.20	3.25	12.86	2.28 ± 0.19	27.7 ± 2.1	8	8
N5	0.00	2.30	2.89	10.06	1.73 ± 0.07	24.1 ± 1.5	6	7
N6	0.00	3.55	4.63	18.54	1.67 ± 0.13	38.2 ± 2.2	6	4

Tab. 5.3: Comparison of minimum, maximum, median, and mean hourly CH_4- C fluxes and of the winter and entire annual balance for natural, restored, and degraded bog plots.
Sites: D1 dry bog heathland (peat cut); D2 dry bog heathland (peat cut); D3 dry bog heathland (drained-only); D4 dry bog heathland (drained-only); R1 flooded dry bog heathland; R2 restored moist bog heathland; R3 restored *Sphagnum* lawn; N1 moist bog heathland; N2 bog shrubs (*Pinus mugo*); N3 *Sphagnum* lawn; N4 *Eriophorum* and *Sphagnum* hummocks; N5 transition hummocks – hollows; N6 *Sphagnum* hollow. For further site description see chapter 3.

Nitrous oxide fluxes were generally very low, with less than 0.75 µg N_2O-N $m^{-2}h^{-1}$ mean-fluxes on almost all sites. Only the two most degraded sites D1 and D2 had notable hourly fluxes with means from 6.50 to 19.71 µg N_2O-N $m^{-2}h^{-1}$ (see Tab. 5.4). Generally, the N_2O-N fluxes were scattered and showed non-normal distribution on every site. It was notable that negative fluxes (N_2O-N uptake to the system) appeared on every site. However, the hourly means were negative almost exclusively in the entire natural group, but remained close to zero. By comparing the sites, the difference between the degraded sites and the rest of the sites could be identified. Within the degraded sites, a gradient from D1 to D4 could be found (Tab. 5.4).

	min [µg $m^{-2}h^{-1}$]	median [µg $m^{-2}h^{-1}$]	mean [µg $m^{-2}h^{-1}$]	max [µg $m^{-2}h^{-1}$]	annual balance ± SE [mg m^{-2} a^{-1}]	percent SE from annual balance
D1	-16.62	4.82	19.71	201.15	168.57 ± 93.70	56
D2	-11.44	0.00	6.50	78.48	50.05 ± 46.84	94
D3	-13.02	0.00	0.75	51.80	9.37 ± 4.89	52
D4	-12.04	0.00	0.49	21.61	4.01 ± 1.12	28
R1	-13.21	0.00	0.20	28.11	1.77 ± 4.34	245
R2	-21.62	0.00	-0.13	8.06	-1.21 ± 2.42	200
R3	-4.96	0.00	0.19	22.95	1.35 ± 1.44	107
N1	-13.25	0.00	-0.28	2.11	-0.81 ± 2.49	307
N2	-16.51	0.00	-0.13	15.62	-1.91 ± 1.62	137
N3	-13.24	0.00	-0.26	0.00	-2.29 ± 1.82	123
N4	-13.24	0.00	-0.15	16.49	-2.24 ± 1.13	50
N5	-9.46	0.00	-0.06	16.49	-0.49 ± 1.66	339
N6	-29.67	0.00	-0.24	12.49	-1.83 ± 1.05	57

Tab. 5.4: Comparison of minimum, maximum, median and mean hourly N_2O-N fluxes, and of the winter and entire year balance for natural, restored, and degraded bog plots.
Sites: D1 dry bog heathland (peat cut); D2 dry bog heathland (peat cut); D3 dry bog heathland (drained-only); D4 dry bog heathland (drained-only); R1 flooded dry bog heathland; R2 restored moist bog heathland; R3 restored *Sphagnum* lawn; N1 moist bog heathland; N2 bog shrubs (*Pinus mugo*); N3 *Sphagnum* lawn; N4 *Eriophorum* and *Sphagnum* hummocks; N5 transition hummocks – hollows; N6 *Sphagnum* hollow.

Explanation of instant methane and nitrous oxide fluxes by environmental variables and NEE

Simple and multiple linear regression analyses were performed on instantly measured methane fluxes as a dependent variable against explaining environmental variables. First, measured methane fluxes were correlated with simultaneously measured NEE (transparent chamber measurements during daytime, respectively), as recent findings revealed functional relationship between CO_2 uptake and CH_4 emission. Only on the natural sites could significant relationships be found, as illustrated in Tab. 5.5.

site	n	regression equation	r^2	p
N1	101	$_{Ln}CH_4 = 6.29 - 0.14 * NEE$	0.15	<0.001
N2	92	$_{Ln}CH_4 = 5.43 - 0.01 * NEE$	0.09	<0.01
N3	93	$_{Ln}CH_4 = 5.26 - 0.02 * NEE$	0.43	<0.001
N4	101	$_{root}CH_4 = 28.55 - 0.28 * NEE$	0.53	<0.001
N5	101	$_{root}CH_4 = 28.23 - 0.23 * NEE$	0.54	<0.001
N6	101	$_{root}CH_4 = 33.32 - 0.35 * NEE$	0.68	<0.001

Tab. 5.5: Explanation of methane fluxes with simultaneously measured CO_2-C fluxes (NEE, transparent chamber).

Within the natural sites, the strength of relation rose from sites N1/N2 up to site N6 (*Sphagnum* hollow), with a maximum explanation range of 68 % of the variability of the methane fluxes. Methane fluxes at sites N1 to N3 were Ln-transformed, and were root-transformed at the sites N4 to N6, to get normal distribution of the data set. Fig. 5.3 shows the corre-

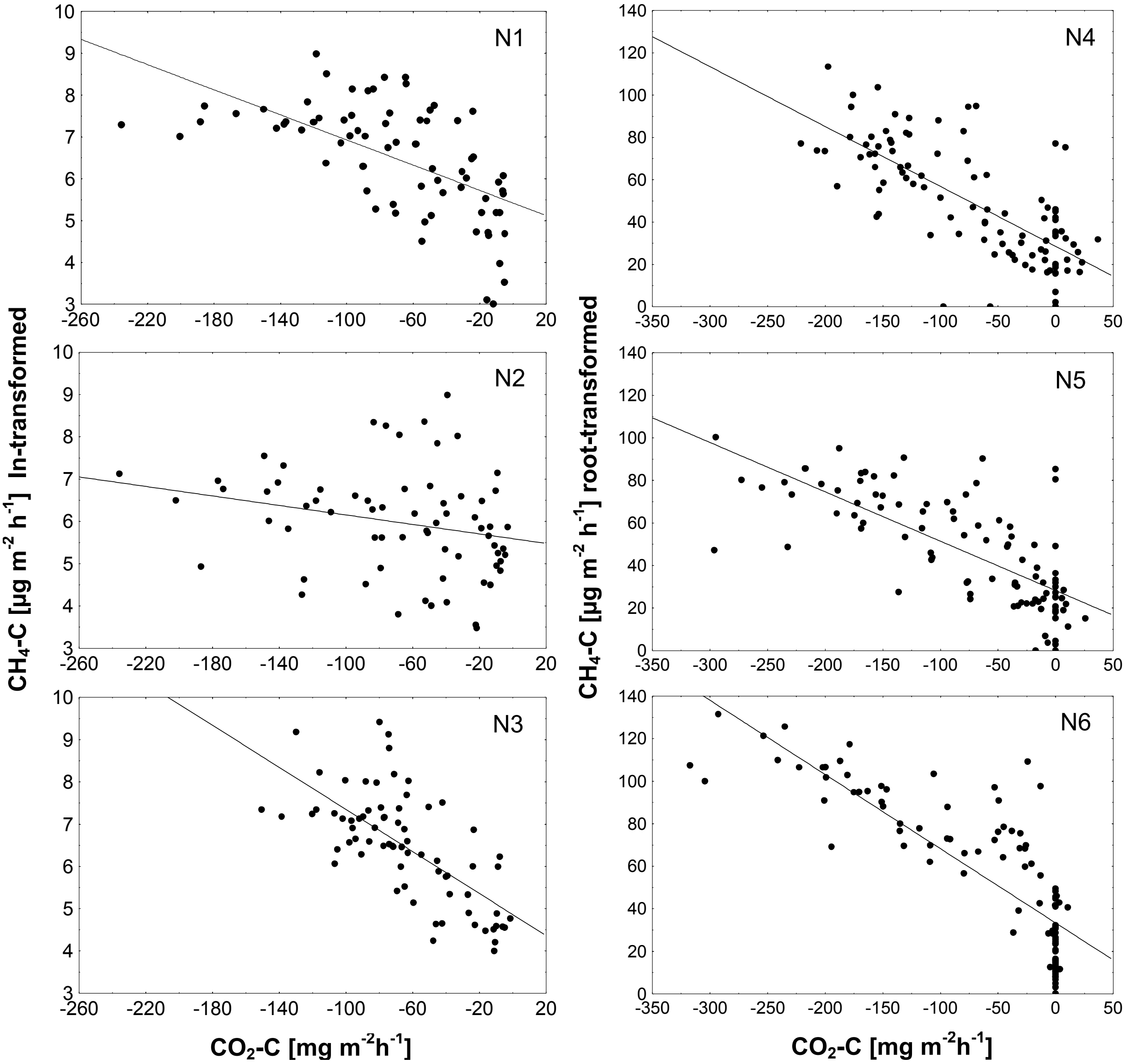

Fig 5.3: Correlation between instant methane-fluxes and NEE (net ecosystem exchange of CO_2-C), simultaneously measured with transparent chambers (data taken from chapter 4). Data had to be transformed to meet normalized distribution. NEE was normally distributed over all sites. N1 moist bog heathland; N2 bog shrubs (*Pinus mugo*); N3 *Sphagnum* lawn; N4 *Eriophorum* and *Sphagnum* hummocks; N5 transition hummocks – hollows; N6 *Sphagnum* hollow. For further site description see chapter 3.

sponding graphs. Since the single NEE regression could not explain the variability of the methane fluxes over all sites, other functionally linked variables were tested: water table and soil temperatures at different depths (2, 5, 10, 20, 50, and 100 cm). Both variables were sampled instantly along with the gas measurements. First, single regressions against methane fluxes were tested and second, multiple regressions with any combination of soil temperature (all depths), water table and NEE as independent variables were done to explain instant methane fluxes. Significant relationships could be determined for all sites, apart from the two degraded ones D1 and D2, which were the sites with close to zero-fluxes and a scattered pattern in their flux dynamic. The strength of explanation level did increase notably after the inclusion of other environmental variables (Tab. 5.6) compared to the single NEE relation (Tab. 5.5).

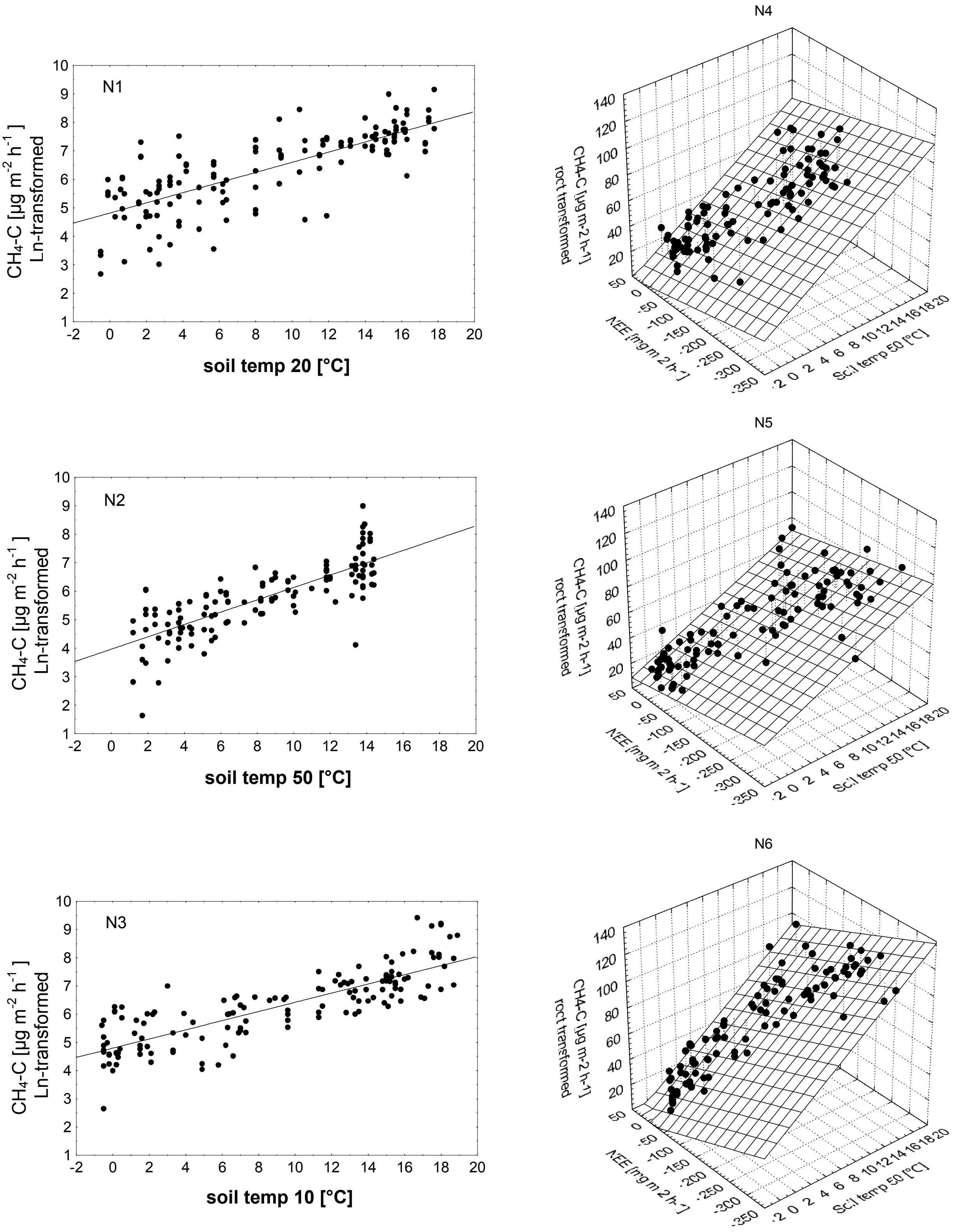

Fig. 5.4: Regression of instant methane fluxes with environmental variables. Methane fluxes at sites N1 to N3 were Ln-transformed, at N4 to N6 root-transformed to get normal distribution of the data set. At sites N1 to N3 a single linear regression with soil temperature at depth of best fit (10, 20 or 50 cm), showed highest strength of the regression. At sites N4 to N6 multiple linear regression with soil temperature (at 50 cm depth) and NEE (CO_2-C exchange measured simultaneously to CH_4–C exchange with transparent chamber) produced highest regression strength. For regression equations see Tab. 5.6. For further site description see chapter 3.

Sites: N1 moist bog heathland; N2 bog shrubs (*Pinus mugo*); N3 *Sphagnum* lawn; N4 *Eriophorum* and *Sphagnum* hummocks; N5 transition hummocks – hollows; N6 *Sphagnum* hollow.

The variability of the methane fluxes on the natural sites N4, N5 and N6 could be explained by multiple linear regression with NEE and soil temperature at 50 cm depth, at a range of 73 to 89 % (see Tab. 5.6 and Fig. 5.4), which was notably higher than in the single NEE relation (r^2=0.53-0.68, see Tab. 5.5). The incident CH_4 fluxes of the natural sites N1, N2, and N3 were explained best by linear regression with soil temperatures at depths between 10 and 50 cm (see Tab. 5.6 and Fig. 5.4). The strength of the relation (r^2=0.59 to 0.69) was notably higher than in the single NEE relation (r^2=0.09-0.43, see Tab. 5.5). Multiple regressions did not lead to a better correlation with any of the driving variables at sites N1 to N3.

The restored sites showed relatively weak regression coefficients with soil temperature as the sole variable (r^2=0.18-0.20, Tab. 5.6). Finally, the variability of the methane fluxes of the degraded sites D1 and D2 was not explainable to a significant level by any of the driving factors. The degraded site D3 was the only site, where the water table along with soil temperature at 2 cm led to a better explanation of the variability of the incident methane fluxes. And even here, the strength of explanation was weak (r^2 = 0.26; see Tab. 5.6). Instant nitrous oxide fluxes could not be explained at any significant level by any of the environmental variables.

site	n	regression equation	type of equation	r^2	p
D1		$_{Ln}CH_4$ =			ns
D2		$_{Ln}CH_4$ =			ns
D3	96	$_{Ln}CH_4 = 2.89 + 0.09 * ST2 – 0.04 * WT$	m	0.26	<0.001
D4	141	$_{root}CH_4 = 1.99 + 0.82 * ST50$		0.42	<0.001
R2	150	$_{root}CH_4 = 8.68 + 1.37 * ST20$		0.20	<0.01
R3	141	$_{root}CH_4 = 7.96 + 0.53 * ST10$		0.18	<0.05
N1	141	$_{Ln}CH_4 = 4.82 + 0.18 * ST20$		0.59	<0.001
N2	133	$_{Ln}CH_4 = 3.97 + 0.22 * ST50$		0.62	<0.001
N3	140	$_{Ln}CH_4 = 4.79 + 0.16 * ST10$		0.69	<0.001
N4	101	$_{root}CH_4 = 9.09 – 0.06 * NEE + 4.12 * ST50$	m	0.75	<0.001
N5	101	$_{root}CH_4 = 16.87 – 0.59 * NEE + 2.78 * ST50$	m	0.73	<0.001
N6	101	$_{root}CH_4 = 5.76 – 0.12 * NEE + 4.57 * ST50$	m	0.89	<0.001

Tab. 5.6: Explanation of methane fluxes with NEE and environmental variables. The equation of the best fit is shown. WT= water-table (in cm), ST= soil temperature (at 10 to 50 cm; in °C), NEE= instant CO_2-C fluxes (in mg $m^{-2}a^{-1}$), m=multiple linear regression. For further site description see chapter 3.

Methane and nitrous oxide annual balance

The annual balance of the fluxes of CH_4 showed a wide span over the sites and ranged between 0.03 ± 0.02 g CH_4-C $m^{-2}a^{-1}$ at the degraded site D1, to 38.2 ± 2.2 g CH_4-C $m^{-2}a^{-1}$ at the natural site N6 (Tab. 5.3, Fig. 5.5). The order of the sites based on the annual balances was generally parallel to the grouping of the instant fluxes. Nearly neutral in methane balance were the degraded sites D1 and D2 with

0.03 ± 0.02 to 0.08 ± 0.03 g CH_4-C $m^{-2}a^{-1}$. Low balances of 0.9 ± 0.3 to 2.3 ± 0.4 g CH_4-C $m^{-2}a^{-1}$ were measured on sites D3, D4, R1 and R3. Middle balances of 5.4 ± 0.5 to 10.7 ± 1.6 g CH_4-C $m^{-2}a^{-1}$ were obtained at the first group of natural sites N1 to N3, and the restored R2. Finally, the greatest balances were found on the natural sites N4 to N6 with 24.1 ± 1.5 to 38.2 ± 2.2 g CH_4-C $m^{-2}a^{-1}$. Uncertainties of the annual balances were expressed by the relation of SE to the balance (Tab. 5.3). At no site was uncertainty higher than 100 %, thus all balances were interpreted as different from zero. The highest uncertainties occurred on sites D1 (67 %) and R2 (44%). Generally the natural sites had lowest uncertainties between 6 % and 23 %.

The annual balances of the N_2O-fluxes were less certain due to the scattered pattern of the fluxes over the year, as well as the oscillation around zero especially at the restored and natural sites (Tab. 5.4, Fig. 5.5) Consequently, just for the degraded sites D1 to D4 and for the natural sites N4 and N6, year-balances which were different from zero could be confirmed (Tab 5.4). The range of the balances for the degraded sites was between 4 ± 1.1 mg N_2O-N $m^{-2}a^{-1}$ at site D4 and 168.6 ± 93.7 mg N_2O-N $m^{-2}a^{-1}$ at site D1, with uncertainties between 28 and 94 %. The natural sites N4 and N6 showed weak uptakes, of –2.24 ± 1.13 and –1.83 ± 1.05 mg N_2O-N $m^{-2}a^{-1}$, with an uncertainty of 50 and 57 % respectively. All other sites had both low fluxes (-2.29 to 1.77 mg N_2O-N m^{-2} a^{-1}) and uncertainties between 123 to 307 %. Consequently, these sites could act either as weak sinks or sources.

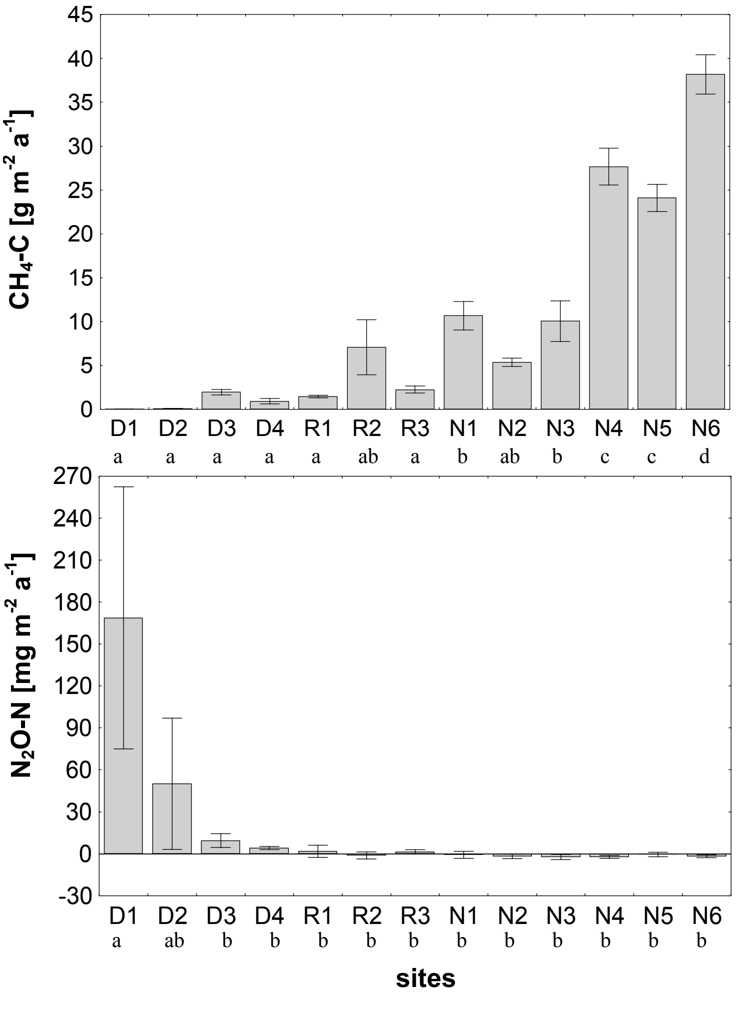

Fig. 5.5: Annual balances for methane (Fig. 5.5a) and nitrous oxide (Fig. 5.5b). Sites: D1 dry bog heathland (peat cut); D2 dry bog heathland (peat cut); D3 dry bog heathland (drained-only); D4 dry bog heathland (drained-only); R1 flooded dry bog heathland; R2 restored moist bog heathland; R3 restored *Sphagnum* lawn; N1 moist bog heathland ; N2 bog shrubs (*Pinus mugo*); N3 *Sphagnum* lawn; N4 *Eriophorum* and *Sphagnum* hummocks; N5 transition hummocks – hollows; N6 *Sphagnum* hollow. Error bars are SE. Different letters indicate significant difference according to ANOVA, Tukey-test ($p<0.05$), but n=3 limited differentiation strength. Site description see chapter 3.

Winter methane and nitrous oxide fluxes

The year-long measurement cycle allowed direct measurement of the winter-fluxes. Winter fluxes were considered the measurements taken before snow melt in spring (10.03.) and after snow fall in late autumn (16.11.). In 1999, this period was 115 days in total, corresponding to 32% of the year. Chambers were placed on the snow surface and were carefully sealed with snow without disturbing the snow cover over the plots.

Winter methane emissions followed a similar pattern as was recorded in the non-winter period, generally following a degradation gradient (see. Tab. 5.3). The degraded sites D1 to D4 had winter fluxes between 0.01 ± 0.01 to 0.13 ± 0.05 g CH_4-C $m^{-2}per^{-1}$. The winter fluxes of the restored sites R1 to R3 were between 0.10 ± 0.03 and 0.83 ± 0.72 g CH_4-C $m^{-2}per^{-1}$. Finally the natural sites N1 to N6 showed the highest winter fluxes, from 0.49 ± 0.05 to 2.28 ± 0.19 g CH_4-C $m^{-2}per^{-1}$. In the group of the natural sites, winter fluxes accounted for up to 4 to 10 % of the entire year fluxes. The relevance of winter fluxes for the degraded sites was generally higher (7 to 75%). The restored sites were in between, with winter fluxes contributing 7 to 17 % to whole year-fluxes.

The span of absolute values for the winter period (0.01 to 2.2 g $m^{-2}per^{-1}$) was an order of magnitude narrower than the span for the whole annual balances (0.03 to 38.2 g $m^{-2}a^{-1}$). Instant winter fluxes were scattered and dynamic, and could not be explained by any variables at a significant level. However, the winter methane balances could be explained at significant level with mean water table in wintertime (rootCH_4-C (winter) = 1.00+0.37*meanWT (winter) (r^2=0.63, $p<0.01$)). Winter methane balances could also be explained by the appearance of aerenchymous leaves, but with less strength (rootCH_4-C (winter) = 0.36+0.02*rootLEAVES (r^2=0.45, $p<0.05$)). Multiple linear regression combining water table and aerenchymous leaves was not found to be significant ($p>0.05$).

Due to the scattered patterns of the nitrous oxide fluxes, a differentiation of summer and winter fluxes did not reveal consistent results over the sites. Additionally, freeze-thaw cycles could not be identified as major emission periods.

Explanation of methane and nitrous oxide annual balances with environmental variables and NEP

The annual balances as integral signals of the methane emissions were tested against independent variables. To explain the methane and nitrous oxide balances, environmental variables were used, which are functionally related to the production or emission processes.

As explaining parameters for methane emissions, the mean water table (all plots), the quantitiy of aerenchymous leaves (all plots) and the CO_2-C balance (mean of the sites; NEP taken from chapter 4) were tested. Mean soil temperatures were similar at corresponding depths across the sites and could not explain the variability of the yearly methane balances between the sites at a significant level.

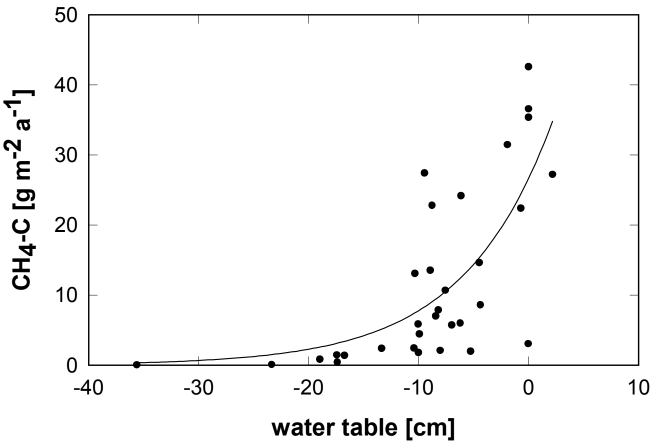

Fig. 5.6: methane balances vs. mean water table (all plots). Equation see Tab.5.7

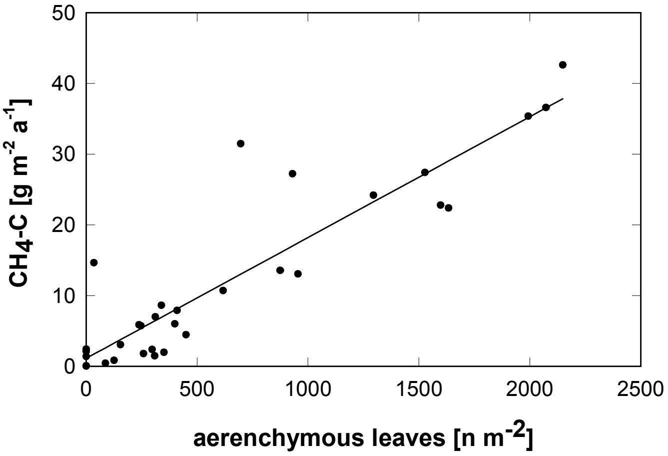

Fig. 5.7: methane balances vs. aerenchymous leaves (Eriophorum standard; all plots). Equation see Tab.5.7

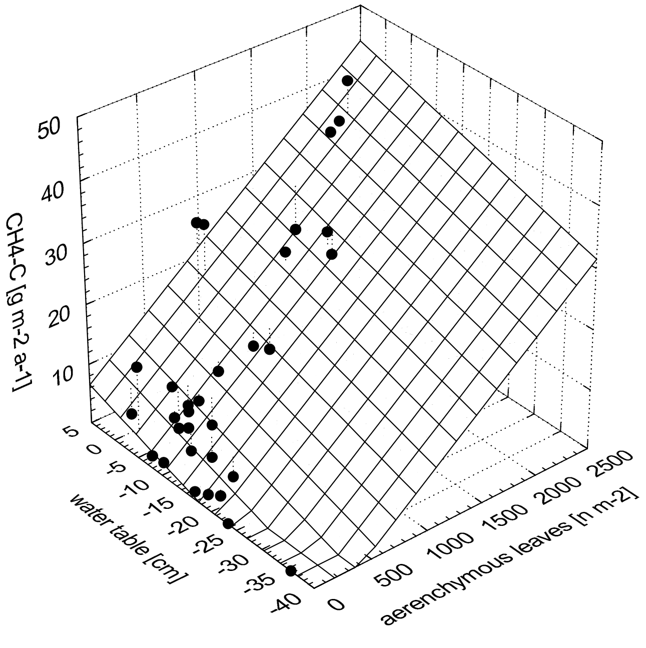

Fig. 5.8: methane balances vs. water table and aerenchymous leaves (Eriophorum standard; all plots). Equation see Tab. 5.7.

Initially, at the level of the plots single factor regressions were tested with mean water-table and the quantity of aerenchymous leaves as independent parameters. The relation between water table and methane balance was best described with an exponential equation (Fig. 5.6) which could explain 54% of the variability (Tab. 5.7) at significant level. However, a single factor regression with the quantity of aerenchymous leaves within one plot (Fig 5.7) was able to explain by a simple linear fit up to 82% of the variability of the methane balances at a significant level (Tab. 5.7). Quantity of leaves was expressed in Eriophorum standard (see materials and methods). So, with two single factors, water table and aerenchymous leaves, the great majority of the variability of the methane year-balances could be explained. Additionally, a multiple linear regression was tested for the combination of water table and aerenchymous leaves as explaining variables. Both parameters had significant *betas* and the regression could explain with 85% (Tab. 5.7) even more of the variability of the methane balances (Fig. 5.8) than the single factor explanation via aerenchymous leaves exclusively.

Additionally the regression with NEP as explaining variable was tested, because of the hints in the literature about the functional link between methane production and net ecosystem exchange. NEE was confirmed already as explaining variable for instant methane fluxes for the natural sites of the presented study (see Fig 5.4 and

Tab5.6). As for the NEP no single values were obtained for the individual plots but the model output for the annual balance was on the site level, the regression between methane and NEP was done as well at the site level. NEP showed an exponential relationship with the methane balances (Fig. 5.9) and was able to explain 87% of the variability of the methane balances of the sites (see Tab 5.7).

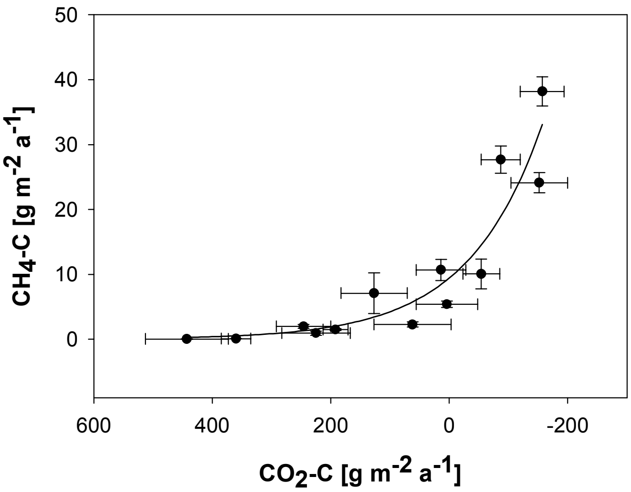

Fig. 5.9: NEP (CO_2-C annual balance) vs. methane balance. Equ. see Tab. 5.7. Error bars are SE

site	corresponding figure	n	regression equation	r^2	p
all sites (R1 excluded)	Fig. 5.6	33	CH_4-C = 26.64 * EXP (WT/8.11)	0.54	<0.001
all sites (R1 excluded)	Fig. 5.7	33	CH_4-C = 1.12 + 0.02*LEAVES	0.82	<0.001
all sites (R1 excluded)	Fig. 5.8	33	CH_4-C= 5.06 + 0.02*LEAVES + 0.29*WT	0.85	<0.001
all sites	Fig. 5.9	13	CH_4-C = 9.43 * EXP (NEP/125.08)	0.87	<0.001

Tab 5.7: Explanation of annual balances of methane by water table (WT), arenchymous leaves (LEAVES) and net ecosystem productivity (NEP).

For an explanation of nitrous oxide emissions, the relationship to electrical conductivity (EC), mean water table (WT), and the annual balance of CO_2-C flux (NEP; taken from chapter 4) was tested. Finally, the relationship between the nitrous oxide balance and the methane balance was tested. Since no transformation of the nitrous oxide balances was efficient to derive a normalized distribution, the correlation test was done with non-parametric procedures. Spearman R was highest for the correlation with mean electric conductivity (EC; 0.94), followed by the correlation with CO_2-C annual balance (NEP; 0.87), the CH_4-C annual balance (-0.80) and the mean water-table (WT; -0.75). All correlations were significant (see Tab. 5.8). Multiple linear regression analysis for explaining the N_2O-N annual balances was not undertaken, because of the non-normal distribution of the N_2O-N balances.

N_2O-N	n	Spearman R	t (N-2)	p
variable				
EC	12	0.94	8.48	<0.001
CO_2-C	12	0.87	5.51	<0.001
CH_4-C	12	-0.80	-4.18	<0.01
WT	12	-0.75	-3.57	<0.01

Tab. 5.8 Rank correlation N2O-N balance with site factors.

Discussion

Methane and nitrous oxide fluxes

The measured plots covered a wide range of site conditions and represented a nearly complete gradient of bog ecosystems in the southern German mire belt in the forelands of the Bavarian Alps, from severely degraded to natural conditions. This gradient is generally reflected in the instant methane fluxes. The measured fluxes were in the same range as those found by several authors for corresponding ecosystem types (Bubier et al., 1993, Yavitt et al., 1993, Bubier, 1995, Roulet & Moore, 1995, Granberg et al., 1997, Nykänen et al., 1998, Bellisario et al., 1999, Svensson et al., 1999, Christensen et al., 2000, Joabsson & Christensen, 2001) independent of latitudinal position from temperate to subarctic. However, Cao et al., (1998) modelled methane emissions on a global scale, with methane emissions rising at lower latitudes. In this model, natural wetlands were considered as a uniform group. Along this latitudinal gradient from north to south, a shift from bogs to fens to swamps occurs, so this latitudinal effect may also be a result of changing proportions of the major ecosystem types. For example, Bellisario et al. (1999) found 50 times greater emissions in fens than in bogs within the same study area. Christensen et al. (2000) obtained three times higher fluxes in hollows than hummocks of an arctic fen. The ecosystem type consequently seemed to be very important for the levels of the instant CH_4 fluxes. Rising CH_4-emission along a latitudinal gradient from north to south, however, can be attributed to rising average temperatures when the ecosystem type is comparable (Christensen et al., 2003).

The high degree of variability of the nitrous oxide fluxes within the sites may bring into question the viability of only using three replicates for each site, even though the space of the single plots ($0.56\ m^2$) was 8 times bigger than in comparative studies (Kamp, 1998, Ruser, 1999). The processes at the site level seem to vary at a higher spatial scale than does the cover of a single plot. However, the fluxes were generally very low (except on sites D1 and D2), so the variability could be due to the specific flux pattern of nutrient poor ecosystems. Nutrient poor bog ecosystems generally show very low fluxes of nitrous oxide (Martikainen, 1996) with high spatial variability.

Explanation of instant methane and nitrous oxide fluxes by NEE and environmental variables

The strength of relation between NEE and instant methane fluxes rose along the gradient from the natural sites N1/N2 to N6. This was interpreted as a tightening relationship between carbon dioxide uptake and methane emission along this gradient, unless at site N6 and partly N5 (Fig. 5.3) zero CO_2-fluxes do appear, together with significant CH_4-fluxes. These measurements originate from wintertime, with low temperatures and ice and snow-packs covering the sites. Limited detection of very small CO_2-fluxes under these conditions may be attributed to the GC-analysing system (Loftfield et al., 1997), as winter fluxes were not done with more sensitive IRGA-system in the field.

The chamber method only allows the measurement of surface fluxes as the sum of the production and consumption processes, and not the rates of methane production in the rhizosphere. Arah & Stephen (1998) found that methane concentrations within the soil profile are highest in the densely rooted area from a depth of 10 to 30 cm, where root exudates are available. A functional link between methane production and NEE was recently identified by several authors (Bubier, 1995, Bellisario et al., 1999, Christensen et al. 2000). For several sedge-species Ström et al. (2003) and Ström et al. (200X) found the concentration of acetate in the rhizosphere correlating with methane emissions, while NEE was correlated with the concentration of acetate. This functional relation explains the link between the NEE values and the CH_4 emission found for the natural sites (Tab. 5.5 and Fig. 5.3). The confirmation of this functional link on the studied sites depended on the viability of tracking this effect via measurements of the fluxes at the soil or vegetation surface. The ratio between produced versus emitted methane is mostly determined by the rate of oxidation of methane in the aerated soil layer by methanotrophic bacteria, which use CH_4 as a metabolic substrate and emit CO_2 (Sundh, 1995, Joabsson et al., 1999). This emitted CO_2, however, reduces the measurable NEE flux which is not in the same range as the methane flux, since the molar matter fluxes of CO_2 are several magnitudes higher than those of methane. The more pronounced the oxidization is, the less detectable the functional relation between methane production and NEE should be, when measuring the surface fluxes. Hence, it should be expected that the best relationship between methane fluxes and NEE can be found when oxidation is limited, like under waterlogged site conditions and in the presence of vascular plant species with aerenchymous tissue as bypass for methane fluxes (Shannon et al., 1996, Frenzel & Rudolf, 1998, Joabsson et al., 1999). Within the gradient of the analysed sites (N1-N6), the natural *Sphagnum* hollow (N6) was characterised by these site conditions. So it was assumed that no major oxidation occurred and the emission rates reflected more or less completely the production rates of methane. The instantly measured NEE at this site could explain 68 % (Tab. 5.5) of the variability of the methane fluxes, which was the highest strength of regression over all of the sites. This value fell in between the data of Bellisario et al. (1999) with 46 % and Christensen et al. (2000) with 79 % explanation level of the variability of CH_4-C fluxes by CO_2-C fluxes. The rest of the variability should be attributed to the fact that methane is not exclusively produced in the rhizosphere of the active plant layer, but through the peat profile as well. The sites N1-N6 are geographically close together and are settled on stratigraphically similar peat profiles. Consequently, it was assumed that the decreasing explanation percentages from site N6 to sites N1/N2 (Tab. 5.5) were a consequence of rising methane-oxidation, due to the lower water-tables and the decreasing abundance of aerenchymous plants (see Tab. 5.1). To reveal these relationships under field conditions, it is necessary to measure CH_4 and CO_2 simultaneously, as was done in the standard procedure at every site. However, the NEE-methane relationship was still only significant at the natural sites, and even there site N1 revealed only a weak relationship. Methane fluxes on degraded and restored sites however, could not be explained by NEE alone.

Several authors found CH_4 emissions to be related to temperature and water-table position (Roulet et al., 1993, Bubier, 1995, Saarnio et al., 1997, Granberg et al., 1997, Daulat & Clymo, 1998, Hargraeves & Fowler, 1998, McDonald et al., 1998, Nykänen et al., 1998, Bellisario et al., 1999). Therefore the contribution of these variables, together with NEE, were tested for explanations of the methane fluxes via single and multiple regression analysis (Tab. 5.6). The findings showed that deeper soil temperatures (10-50 cm) correlate best with the methane emissions. This behaviour was also found on Finnish peatlands (Kettunen et al., 1996; Nykänen et al., 1998). As these zones were waterlogged process oriented interpretation is possible, because the peat layers with the highest CH_4 production in the profile were reported at this depth. Here in the major rooting horizon, relatively young carbon substrates, which are easily decomposable, are abundant. Walter & Heimann (2000) found methane concentrations rising until a depth of 40 cm and Saarnio et al. (1997) sampled rising CH_4-production within the peat profile from the water-table down until 10 to 30 cm in depth, with decreasing production rates below this depth. However, the time lag between production and detection of methane after its passage through the peat provokes uncertainties in the field based correlation attempts. Laboratory experiments could help to strengthen the explanatory level of this relationship, which could not be included in this study. However, since the explaining variables are known to be functionally related to the production processes, the field based phenomenological regression approach has explanatory value.

The measured nitrous oxide fluxes could not be explained at a significant level ($r^2 < 0.10$; $p>0.05$) by any environmental variable. According to the system of Brumme et al. (1999), the fluxes at the study sites could be classified as background fluxes. Only at site D1, did the flux pattern seem to be influenced by seasonality as well (see Fig. 5.2), unless not explainable at significant level with e.g. soil temperature. This is obviously a result of the low absolute fluxes and the high variability in time and space of the fluxes, which was also found in other nutrient poor mires (Martikainen et al., 1996). Similarly, in ecosystems with higher fluxes, like potato fields, only Nitrate concentration could explain the scattered fluxes to a certain level (Ruser, 1999), or no relationship at all could be found between the fluxes and explaining environmental variables (Glatzel & Stahr, 2001).

Methane and nitrous oxide annual balances

Only a few studies have been reported with annual balances for CH_4-fluxes in mires, but similar rates were obtained by several authors studying methane balances of natural bogs (Dise, 1993, Shannon & White, 1994, Nilsson et al., 2001, Minkinnen, 2002). Whalen & Reeburgh (1992) found balances that were around the half of the presented ranges. The balances found by Roulet et al. (1992) were an order of magnitude lower, as the studied bog was seemingly much dryer than the natural sites N1-N6. The critical factor of mean water-table seems to rule the potential for methane emissions, once the water table is shallower than a threshold value (Christensen et al. 2003).

Looking at the pathways for methane emission, the balances should generally reflect the diffusion and plant mediated efflux, whereas ebullition appears stochastically and may not be fully included due to the once a week measurement program. Ebullition is only likely to take part in waterlogged sites and these conditions were restricted to site N6 and R1. On site N6 especially, instant fluxes could be explained at a high level by NEE and soil temperature (Tab. 5.6), so it was assumed that ebullition played a limited role in these waterlogged sites.

The study sites were generally characterised by nearly neutral emission or a weak uptake rate of nitrous oxide on the natural and restored sites. Similar behaviour was reported by Freeman et al. (1993) and Martikainen et al. (1995) for nutrient poor mire ecosystems. Only the degraded sites and the former peat cut areas (sites D1 and D2) here showed notable emissions, with 50 ± 47 to 168 ± 94 mg m^{-2} a^{-1}. This is close to the 222 mg m^{-2} a^{-1} nitrous oxide emissions found in situ experiments in fens (Martikainen et al., 1993). At the bog sites, drainage and peat cutting led to comparable emissions to drained fen sites, but the drained bog sites without former peat cutting (D3 and D4) produced much lower emissions (4.1 to 9.4 mg m^{-2} a^{-1}). Both the management regime and the ecosystem type seemed to be of importance for the nitrous oxide emissions.

Measured methane and nitrous oxide balances are based on a single year-round study. Trace gas exchange of ecosystems reacts sensitive to climate and therefore interannual variability of fluxes may be significant (Arneth et al., 2002). However the climate of the measurement year was very similar to the 30- average, with almost identical precipitation sums and just one 1 °C higher mean temperature. Arneth et al. (2002) showed that NEE curves of bogs did not vary significantly over three consecutive years, as climate was very similar between the three years as well. As NEE is climate driven and directly linked to the methane emissions, it can be assumed, that the average climate during the measurement year produced average fluxes as well. The relation of the sites in terms of their methane- and nitrous oxide -balances are expected to be robust, as these balances are expressions of the site functioning and closely related to site factors.

Methane and nitrous oxide winter-balances

Average daily fluxes on natural sites (6.4 - 24.8 mg CH_4-C $m^{-2}d^{-1}$) were higher than those reported by Alm et al. (1999), with 3.37 mg CH_4-C $m^{-2}d^{-1}$ as an average over lawn, hummock and hollow microsites. Contrastingly to the presented results, Alm et al. (1999) could not find significant differences between these mircosites. Dise (1992) reported daily winter fluxes on natural Minnesota bog sites of 2.25 to 12 mg CH_4-C $m^{-2}d^{-1}$. Winter methane fluxes in this study were scattered and could not be explained by environmental variables at significant levels. However, Alm et al. (1999) found a weak relationship between methane efflux and peat temperature, whereas Dise (1992) did not find consistent relationships with any explaining environmental variables.

Winter methane balances, as a percentage of the entire year methane balances, accounted for 7 to 33% of the year balance at the degraded sites, with D2 as an extreme (75 %). Winter balances for the restored sites ranged from 7 to 17 % and on the natural sites from 4 to 10 %. It was remarkable that site N6, with the highest whole year fluxes, experienced the lowest winter emissions and consequently the lowest percentage of winter to non-winter methane fluxes (4%). This was possibly a result of the fact that N6, as the only vegetated site with open water, was frozen during wintertime and the ice layer hindered methane emission. These calculated winter balances were generally notable contributions to the annual balances. Dise (1992) reported similar levels of winter contribution (4 to 21%) to the total methane annual balance. Winter CH_4-C efflux reported by Alm et al. (1999) for northern Finnland amounted to 5-33% of the annual methane efflux. Both the site-specific fluxes and the length of winter period determine the contribution to annual CH_4-C balances. The general ranges between the compared study sites matched well; however, the comparison was restricted to natural sites because no other data was reported on winter fluxes for restored or degraded bog ecosystems. Winter CH_4-C balances of restored sites (R1-R3) accounted with up to 17 % of non-winter efflux, for a significant part of the entire annual balance. This notable winter contribution confirms the need for an approach which records measurements weekly to twice a week throughout the whole year cycle. In contrast to the single fluxes, winter balances over all sites could be explained well by mean water-table (r^2= 0.63, $p<0.01$), which was at a higher level than the explanation of the whole year methane fluxes (r^2=0.54, $p<0.001$; see Tab. 5.7). However, the explanation level by correlation with the appearance of aerenchymous leaves (r^2=0.45, $p<0.05$) was much lower than in the whole year regression (r^2=0.82, $p<0.01$). This may result from the limited contribution of active aerenchymous plants in wintertime.

As the fluxes for nitrous oxide were generally low, the differentiation of summer and winter fluxes did not reveal significant results. Similarly, the recently suggested importance of freeze-thaw cycles as important emission periods for nitrous oxide, could not be confirmed for the studied sites. This may be again a result of the low and scattered fluxes, but even on the site of major emissions (D1), no such effect could be found. For natural boreal bogs, Alm et al. (1999) also reported no winter-fluxes but did find 90 mg N_2O-N m^{-2} a^{-1} efflux from intensively drained fens.

Restoration effects on methane and nitrous oxide balances

The sites were linked by a time series, comparing the situation before and after restoration. Flooding of D2 was done in the middle of the measurement year, with the restored site R1 as resulting ecosystem type. D1 represented the pre-restoration site conditions of R3, which existed less than ten years ago. Similarly, D4 represented the conditions that existed prior to the restoration of site R2. The ratio of the methane balance before and after restoration can be interpreted as an expression of the dynamics of methane evolution. The ratio of R1/D2 (restored within the measurement year) was 20, so restoration led to 20 times higher fluxes of methane already in the first year. The ratios of the sites R2/D4 and

R3/D1 were used as approximation for the time effect, as D4 and D1 could be taken as control sites for the situation on R2 and R3 before restoration, which took place ten years before the trace gas measurements. The R2/D4 ratio was 7.9 and the R3/D1 ratio was 76.7, so the instant ratio of 20 (R1/D2) fell in between. This may be interpreted as such, immediately after restoration, comparable ratios will be obtained, as in sites restored ten years ago. This could be confirmed by the ratio R3/R1 of 1.4 (similar former ecosystem before restoration). On site R3 where aerenchymous plants were absent and site R1 which was totally free of emergent plants, the methane fluxes were assumed to represent the methanogenic base flux of the sites, without being enhanced by aerenchymous plants. The narrow ratio of R3/R1 was possibly a product of the fact that methanogenic bacteria appear even in severely degraded bog soils and can take full advantage immediately after a change to anaerobic conditions via flooding the site. Consequently, Tuittila et al. (2000) found that constantly rising methane emissions over three years following restoration, were linked to increasing coverage of *Eriophorum vaginatum*. Rising methane emissions after restoration seemed therefore to develop almost immediately to a soil determined basic flux and were predominantly driven afterwards by the developing plant cover. The development of aerenchymous plants (leaf quantity or coverage) could be used as an indicator for the methane balance dynamics.

For nitrous oxide, the ratio of R1/D2 (restored within the measurement year) was 0.036, which means that restoration cut the nitrous oxide emissions to less than 4 % of the degraded situation existing before restoration. The R2/D4 ratio showed a 30% uptake of the former emissions and the R3/D1 revealed a 1 % emission on site R3 in relation to D1. Restoration has notable effects on reducing nitrous oxide emissions on sites, even within the first year of restoration, and can even switch a source to a weak sink for nitrous oxide.

Explanation of methane balances

The hourly fluxes could only be explained to a satisfactory level by environmental variables on the natural sites. Since the focus of this study was to also assess the sink or source function over an entire year cycle via comparison of natural sites with a set of degraded and restored ones, obtaining the annual balances for all sites was a prerequisite. Tuittila et al. (2000) developed a mechanistic model for predicting methane fluxes at a seasonal level, but did not apply the model to a whole year cycle. At a mechanistic level Walter & Heimann (2000) modelled methane fluxes from natural wetlands with a daily resolution, using water-table, soil temperature, NPP and the presence of vegetation types with CH_4 bypassing effects. This model was used for natural sites exclusively.

For calculating the annual balances of the presented sites, a simple extrapolation procedure (Kamp 1998, Ruser 1999; see Materials and Methods) was used, which was less sensitive to changing parameterisation strength between the sites. However, the annual balances of methane could be explained to 54 % by a regression with water table, to 82 % by a regression with the quantity of aerenchymous leaves, to 85% by a multiple linear regression with both parameters and even to a slightly higher strength of 87% by a single regression with the net ecosystem productivity (NEP). These regression results reflect functional relationships, as the three variables with major influence on the methane fluxes of these bog sites do in single or multiple form act as good predictors for the methane balances: The water table, determining the thickness of the aerated zone in the profile, where methane can be oxidised during the pathway towards the surface, the presence of aerenchymous plants as an indicator of the pathway for methane, bypassing oxidation processes and the plant related NEP, indicating the amount of available labile carbon in the root layer as substrate for methanogenic bacteria. Christensen et al. (2003) however found, that on a large scale the best predictor for methane balances was the soil temperature, whereas the water-table played an important role in switching on the methane emissions, once it was higher than –10 cm. These findings confirm the results shown in Fig. 5.6, with methane balances steeply rising beyond this water-table threshold.

The regression approach that was used, allows the methane balances to be predicted via a simple but functional relationship between NEP and aerenchymous plants. NEP on the other hand, can be modelled via a temperature and radiation driven model (see chapter 4), whereas recording the abundance of aerenchymous plants is a matter of applying simple sampling techniques. If estimates for the annual balances of methane fluxes are needed, as for the calculation of emission factors for reporting the behaviour of ecosystems in national GHG inventories, it may be sufficient to obtain sound NEP values and model the methane balance, using the amount of aerenchymous leaves and the mean water table as further independent variables.

Conclusion

Methane and nitrous oxide fluxes in bog ecosystems could be sampled successfully with a closed chamber technique in a weekly monitoring program and were converted to annual balances. These annual balances reflect more clearly than the individual fluxes the gradient over the restored, degraded and natural sites and the individual behaviour of the sites. Methane balances rose from degraded to restored to natural ecosystems. Nitrous oxide balances rose in the opposite direction from natural to restored to degraded ecosystems. Consistent relationships with predictors like soil temperature, NEP, aerenchymous leaves and water table allow for modelling the effects of changing site or land-use conditions or changing climatic conditions on the flux balances.

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CARBON BALANCE AND CLIMATIC RELEVANCE OF NATURAL, DEGRADED AND RESTORED BOG ECOSYSTEMS

Summary

Carbon accumulation is a key function of undisturbed bogs. The carbon balance was measured via a chamber-based technique as the difference between net CO_2-C fluxes (NEE), CH_4-C fluxes and estimated losses of DOC/DIC. Twelve bog sites in the forelands of the Bavarian Alps were studied along a disturbance gradient from natural to degraded (drainage and peat cut) to restored site conditions. The entire year carbon balance turned out to be a sensitive functional indicator of the site status. Degraded sites were severe carbon sources with losses between 244 ± 52 g C $m^{-2}a^{-1}$ for drained sites and 408 ± 47 g C m^{-2} a^{-1} for abandoned former peat cut sites. Restoration led to a significant reduction of these high emission values, but the sites were still acting as carbon sources. The range was between 71 ±65 g C $m^{-2}a^{-1}$ for ten-year-old restoration sites and 200.5 ± 21 g C $m^{-2}a^{-1}$ in the first year following restoration. Natural sites showed a wide range of balances, between high uptake rates of –121 ± 48 g C $m^{-2}a^{-1}$ and moderate release of 31.7 ± 52 g C $m^{-2}a^{-1}$. The two natural sites, which experienced carbon losses, were affected by a slight natural drainage, due to their position on the inclined border of the bog. The average uptake on the natural sites in the centre of the bog was –80.5 ± 37g C $m^{-2}a^{-1}$.

For assessing the climatic relevance of the bog ecosystems, the site-specific global warming potential (GWP) expressed as CO_2-C equivalents was calculated. At the 100 year timescale, degraded former peat cut sites contributed to global warming by a mean of 416.5 ± 48 g CO_2-C eq. $m^{-2}a^{-1}$, whereas drained but un-cut sites reached 247.4 ± 52 g CO_2-C eq. $m^{-2}a^{-1}$. Restored sites showed a significantly reduced climatic relevance (154.7 ± 49 g CO_2-C eq. $m^{-2}a^{-1}$). Natural sites, unless acting as carbon sinks, did still contribute to climate warming at the 100 year timescale (75 ± 43 g CO_2-C eq. $m^{-2}a^{-1}$). In comparison to degraded sites, restored ones acted as relative sinks with the potential of climate mitigation.

The climatic relevance of southern Bavarian bogs was estimated in a regionalized approach by a comparison of the site specific emission factors (EF), multiplied by the corresponding areas, with the Bavarian average per capita emission of the population of the mire belt: The actual status of the bogs showed a contribution of about 5 % to the CO_2 budget of the local population.

The Bavarian climate protection programme set a goal to reduce the emissions from 92 Mio t CO_2 to 80 Mio t CO_2. A scenario testing the effect of achievable technical measures revealed a 3 Mio t CO_2 gap in reaching this goal. The effect of bog restoration, calculated at a per capita basis for the entire mire belt, could help to reduce this gap by 27 – 36 % (regionalized for the counties within the mire-belt), depending on the accounting scheme. If Bavaria tries to fulfil the even stronger Kyoto commitments, bog restoration could still help to reduce the remaining gap by 10 – 12%.

Introduction

Mires are characterised in their natural functioning by the process of carbon accumulation (Turunen & Tolonen, 1996). Peat deposits are built up because the decomposition rate of organic substances is limited within the water saturated profile (catotelm) and is smaller than the production of new organic matter by the peat building plant community of the surface layer (acrotelm) (Göttlich, 1990). Thus the accumulation of organic matter leads to a continuous growth of a peat profile.

The distribution of bogs in southern Germany widely overlaps with the area that was glaciated during the last ice age in the forelands of the Alps and the higher ranges of the Bavarian forest. Natural bogs accumulated peat layers up to 10 m in depth. Peat deposits in Germany were recently estimated to contain 422 Tg C (Byrne et al., 2004), which represents approximately 1.5 % of the estimated carbon stored on a global scale in peatlands. Global estimates are in the range of 200 to 455 Pg C (Gorham, 1991), with more recent figures of 270 to 370 Pg C (Turunen et al., 2002). This huge quantity of carbon stored worldwide in peatlands represents approximately half of the carbon content of the atmosphere. The maintenance of these deposits depends strongly on the undisturbed functioning of mires as accumulating systems.

Modelled growth-rates of mires prognosticate equilibrium between carbon-accumulation and carbon-release via decomposition for old growth mires dependent of the peat deposit's thickness, once the accumulation within the catotelm and the decomposition along the whole profile reach the same magnitude (Clymo, 1990). However, the average long term rate of carbon accumulation (LORCA) calculated from an extended database of peat cores showed, even in the oldest mires, decreasing but still active accumulation rates of 15 g C $m^{-2}a^{-1}$ (Turunen & Tolonen, 1996). The recent accumulation rates (RERCA), calculated over the last 150 years, show significantly higher values (73 g $m^{-2}a^{-1}$; Turunen et al., 2004). The actual rate of C-accumulation however, can best be calculated from year- round gas-exchange measurements, but with the intrinsic uncertainty of interannual variability.

Peatlands can obviously play an important role in the climate system as carbon sinks and potentially huge carbon sources. However, the database on the greenhouse-gas (GHG)-fluxes of peatlands is still weak in comparison to other ecosystems (Byrne et al., 2004). It is an intrinsic feature of mires, that part of the carbon that is taken up as CO_2 via plant photosynthesis is lost via CH_4 emissions. Hence, for the assessment of the actual carbon accumulation rate with flux-measurements, it is necessary to include CH_4-exchange in estimating C-balances and assessing the carbon sequestration potential. Moreover, for the evaluation of the climatic relevance of an ecosystem, it is essential to measure the exchange of all three biogenic GHGs, CO_2, CH_4 and N_2O. Each of these gases has an individual radiative forcing capability.

Typically, flux clusters focus on CO_2 exchange and have until now only partly supported a full assessment of the climatic relevance of peatlands. Moreover, wetland sites at the European scale are severely underrepresented in flux-studies (Byrne et al., 2004). Only a few studies have reported on all three gases, including NEE for CO_2-C exchange, with the potential of a full assessment of the climatic relevance of the sites (e.g. Christensen, 1999).

The presented study was initiated to fill a thematic and regional gap, assessing net ecosystem exchange (NEE) of CO_2 as well as the CH_4- and N_2O-emissions along a disturbance gradient from natural to degraded to restored bog ecosystems in southern Germany.

Within the UNFCCC, the procedure for reporting emissions from the LULUCF sector and including biological sinks within the Kyoto protocol was recently clarified (COP9), and technical aspects of accounting were defined within the new good practice guidance (IPCC, 2003). Before this background, the climatic relevance of GHG-fluxes from bogs in southern Germany will be analysed via a synthesis of GHG-balances measured in 12 different ecosystem types ranging from natural, to degraded, to restored sites. These sites represent the majority of bog ecosystems in the extensive mire belt in the forelands of the Alps. Special emphasis was given to the potential of climate mitigation via bog restoration.

The specific goals of this chapter at the ecosystem level are i) to estimate the carbon balance of degraded, restored and natural bog ecosystems, ii) to estimate the GWP-balance of the sites, iii) to compare the time dependent climatic relevance with the carbon-balance and iv) to identify the effect of restoration on the carbon- and GWP-balance of the sites. The derived emission and removal factors will be used together with the estimated total bog area at the state level v) to assess the actual climatic relevance of the bogs in southern Germany for potential reporting to the UNFCCC and vi) to estimate the effectiveness of climate mitigation via bog restoration, as biological sinks helping to meet the Bavarian state climate protection programme and the Kyoto protocol commitments.

Material and methods

Balances of greenhouse-gas exchange

The annual balance for CO_2-C (NEP) was calculated in chapter 4. The annual balances for CH_4-C and N_2O-N were calculated in chapter 5. These balances were used for further calculations of the carbon balance and the climatic relevance of the sites within this chapter. For data-source see chapters 4 and 5. The carbon balance was calculated from the CO_2-C balance, the CH_4-C balance and the estimated loss of DOC/DIC. The balances of all three gases were further more used to calculate the overall climatic relevance of the individual sites by multiplying the greenhouse-gas (GHG)-balances with corresponding global warming potential (GWP).

Carbon balance

The carbon balance is a product of several interacting processes, which in their dimensions depend strongly on the status of the bog-ecosystem. The difference between carbon fixation by photosynthesis (GPP) and ecosystem respiration (R_{ECO}) is the net ecosystem exchange (NEE), which is expressed over the course of one year as the CO_2-C balance. Especially in wetlands, anaerobic respiration is an export pathway for carbon, via CH_4-emission. Additionally, carbon is lost in the liquid phase as dissolved organic and inorganic carbon (DOC and DIC) through the water saturated profile. The sum of these component fluxes determines the carbon balance for each individual peatland site and can be expressed with following equation (Waddington & Roulet (2000):

$$dC/dt = NEE + F_{CH4} + qDOC + qDIC \tag{6.1}$$

The CO_2-C balance and the CH_4-C balance, for the studied sites were derived from year round measurements at the different peatland-ecosystems (see chapter 4 & 5). DOC and DIC however, could not be determined, due to the very complex hydrology of the different sites. Moreover, Turunen & Moore (2003) showed, that DOC/DIC is partly re-fixed in the mineral subsoil below the peat body, so that not all of the dissolved carbon will leave the system. Therefore following Alm et al. (1999), the loss via the dissolved pathway was conservatively estimated to be 7 g m^{-2} a^{-1} (according to measurements in similar ecosystems from Sallantus & Kaipainen (1996), Waddington & Roulet (2000), and Fraser et al. (2001)). Fire is not an escape pathway for carbon in southern German mires.

Global warming potentials (GWP)

The global warming potential (GWP) is a simplified index based upon radiative properties of the gases. This index can be used for estimating the potential future impacts of emissions upon the climate system relative to a reference gas, which is normally CO_2 (IPCC, 2001). GWP´s are used as an expression of the climatic relevance of a gas-flux over a defined time period. For reporting under the UNFCCC and for biological sink assessments under the Kyoto Protocol, GWP´s based on the IPCC 1996 values (see Tab. 6.1) are typically used for the 100-year time horizon. However, since mire ecosystems function for a long period of time and are accumulating ecosystems, the 500-year time-scale is also important from an evaluation standpoint, and will therefore also be used here for further calculations.

	20 a	100 a	500 a
CO_2	1	1	1
CH_4	56	21	5.6
N_2O	280	310	170
CO_2-C	1	1	1
CH_4-C	20	7.6	2.4
N_2O-N	120	133	73

Tab. 6.1: Global warming potentials (GWPs) after IPCC(1996)

Climatic relevance of greenhouse-gas exchange

The climatic relevance (GWP-balance) of the greenhouse-gas exchange can be calculated with the following formula:

$$\text{GWP-balance} = NEE*GWP_{CO2} + F_{CH4}*GWP_{CH4} + F_{N2O}*GWP_{N2O} \quad (6.2)$$

As the GWP is time dependant (Tab. 6.1), the climatic relevance is not constant over time and depends on the time horizon used for the calculations. The use of GWP´s for the estimation of the climatic relevance of ecosystems over time was criticized (Frolking 2003) because the GWP concept is designed to cope with the effects of pulse emissions, rather than changing emissions over longer time periods. However, as no better method is currently available, the GWP´s were defined as the standard for the assessment of the climatic relevance of fluxes from ecosystems to be used for UNFCCC reporting (IPCC, 1996).

Area estimates

From beginning of the 20th century Bavarian mires were mapped in view of potential exploitation. Principally peat deposits were mapped following the definitions of the DIN (German industry norm) N° 4047, defining as key criteria an organic matter content of minimum 30 % and a depth of minimum 20 cm for drained peat and 30 to 50 cm for undisturbed peat. However, in practice peat deposits were mapped, once they were thicker than 50 cm (Kurz, 1998). The peatlands were differentiated in fens and bogs via different composition of the peat in the profile. The results of the mapping were displayed in 1:5000 cadastral maps. The first records go back to 1900 with continuation of re-mapping until recently. The mapping is still done with the original procedure and classification of peat, vegetation and land use. This approach was principally developed to estimate the exploitation potential of the peatlands, which contrasts with the actual vision of peatland management and conservation in Southern Germany. However, the constant application of the same methodology allows comparing the development of peatland status and extension, when maps over time series are available. The first complete dataset for the peat-areas was compiled in 1914 (Königlich Bayerische Moorkulturanstalt München, unpubl.). A georeferenced post processing and digitalisation of this map (Kurz, et al. 1999) led to coverage of approximately 55.000 ha of bogs in the southern Bavarian mire belt and 64.000 ha for total Bavaria. These figures can be taken as the best estimate for the original bog area. Several authors estimated the cover of bogs in Bavaria, like Schindler (1948; 61.392 ha), Friedrich (1958) and Kraemer (1958; 59.000 ha), Schuch et al. (1986; extrapolation from partial assessment 40 - 50.000 ha) and Günther (1988; 55.000 ha). The different approaches, the unclear integration of the available cadastral maps, or the extrapolation based on the study of a selected area (Schuch et al., 1986) determine the reliability of the estimates for the selected reference time. For the calculation of the climatic relevance of the bogs and the climate effect of restoration the actual distribution of peatlands, classified in natural, drained, peat-cut and restored sites is a necessary data base. However, the most recent re-mappings of cadastral-maps in the peatland area go back to 1990 and are far from being complete, comparable to the map from 1914. Thus, based on the different available estimates, an approximately 10% loss of the bog area in the southern Bavarian mire belt was estimated leading to some 50.000 has actual coverage. As none of the existing incomplete inventories differentiated sufficiently between different land cover-types according to a disturbance gradient, expert assessments were undertaken to estimate the actual cover of the different tpyes (see Tab. 6.1).

land cover status	area [ha]
Natural	~4.500
drained (forest included)	~35.000
Grassland	n.s.
Cropland	-
Peat cut	~10.000
Restored	~500
Total	~50.000

Tab. 6.2: Estimation of the current land coverage of bogs in the southern German mire belt in the forelands of the Bavarian Alps. (Drösler et al. in prep.)

Bavarian climate protection programme and mitigation commitments under the Kyoto protocol

Bavarian CO_2 emissions were 84.5 Mio tons in 1990 and rose to 92 Mio tons in 1998. The Bavarian state climate protection programme set a goal to reduce the emissions to 80 Mio tons CO_2 by 2010 (Bayerische Staatskanzlei, 2003). A study of the reduction potential possible via technical measures (Fahl et al., 2000) realistically claims emissions of some 83 Mio tons CO_2 by 2010 are achievable and economically viable with current technology. A gap of 3 Mio tons CO_2 will still remain. Blesl et al. (2001) studied the costs associated with the use of renewable energy options in comparison with the costs of improving building insulation, as a basis for an economical decision about filling this 3 Mio ton CO_2 gap. However, if the Kyoto commitments would be applied not only to the federal state of Germany as a whole, but also to individual states, like Bavaria, the goal would be a 21% reduction of the 1990 emissions, representing a target of 66.75 Mio tons CO_2 emissions. This was not the threshold fixed in the Bavarian state climate protection programme.

Expressed on a per capita basis, the yearly average Bavarian emission in 1990 was 6.85 t CO_2 per capita and had risen to 7.46 t CO_2 per capita in 1998. The Bavarian climate protection goal for 2010 would represent a reduction to 6.5 t CO_2 per capita. The gap of 3 Mio t between the IER scenario (Fahl et al., 2000) and the Bavarian climate protection programme would amount to 0.243 t CO_2 per capita. Meeting the Kyoto commitment however, would only allow mean emissions of 5.4 t CO_2 per capita, with a necessary reduction of 1.32 t CO_2 per capita, to cover the gap. On the other hand, the average per capita emissions for all of Germany in 1990 were 10.8 t CO_2. The already lower initial levels in Bavaria were the reason that the reduction goals of the Bavarian climate protection programme were not oriented strictly on the Kyoto plan, but on a less stringent domestic reduction plan.

Population in the peatland counties in the forelands of the Alps

For the assessment of climate mitigation effects of mire restoration, a regionalized approach was followed. The bog distribution is centred in a belt of several counties within the forelands of the Bavarian Alps (Oberallgäu, Ostallgäu, Weilheim-Schongau, Garmisch-Partenkirchen, Bad-Tölz, Miesbach, Rosenheim, Traunstein, Berchtesgadener Land). The population of the area is about 1.23 Mio people (Bayer. Landesamt für Statistik und Datenverarbeitung, 2005), which represents approximately 10 % of Bavaria’s population.

Mire conservation programme(MEK) and climate mitigation

The federal state of Bavaria recently published a mire conservation programme in which, together with biodiversity conservation, climate mitigation was set as an important goal of mire restoration. However, in a list of priority-areas for restoration, degraded sites like the abandoned peat cut areas did not appear. This is a result of the fact that the dominating selection criteria were focused predominantly on biodiversity conservation issues. Hence, the sites with the highest reduction potentials in terms of C-emissions and Global Warming effect were not included near the top of the list. This mismatch between the overall goals and the priority list was due mainly to a complete lack of basic information in the region about the efficiency of mire restoration for carbon sequestration and climate mitigation. Therefore in this study, for the first time, an estimation of the effects of potential bog-restoration within the southern German mire belt forelands of the Bavarian Alps will be presented. Bavaria has committed to taking action for climate mitigation via restoration through both the speeches of the Environmental Minister of Bavaria (Schnappauf, 2003) and the publishing of the mire conservation programme (LFU, 2003), which formulated the overall goal of climate mitigation via bog restoration.

Results

Carbon balance

On the degraded sites, the differences between the CO_2-C balance and the total C balance could be attributed almost exclusively to DOC and DIC loss, as methane emissions were close to zero under these drained conditions (see chapter 5 and Tab. 6.3). On these sites consequently, the CO_2-C balance combined with the DOC and DIC produced good estimates for the carbon balance. On the restored sites, the CH_4-C balance started to influence the carbon balance, especially at site R2, which experienced the highest methane emissions of this group (see chapter 5 and Tab. 6.3). The most pronounced effect of the CH_4-C balance upon the carbon balance was observed in the group of the natural sites (see chapter 5 and Tab. 6.3). The maximum influence of the methane balance was recorded on site N6, which lost 38.2 ± 2.2 g C m^{-2} a^{-1} via CH_4-C emissions (see chapter 5). The individual values for the single sites, as well as the averaged values over the groups of sites, underlined the increasing impact of methane emissions on the carbon balance converse to the disturbance gradient (Tab. 6.3). The highest carbon-loss was found on the former peat cut site D1 (450.1 ± 70 g C $m^{-2}a^{-1}$), whereas the highest uptake rates were recorded on the transition between hollows and hummocks, site N5 (-120.9 ± 48 g C $m^{-2}a^{-1}$) (see Tab. 6.3). To get a better impression of the average values relative to the site conditions, the sites were taken in groupings of degraded sites (D1-D4), restored sites (R1-R3) and natural (N1-N6) sites. The degraded and natural groups were split further into subgroups according to their management status (abandoned peat cut sites D1-D2 and drained sites D3-D4) and the vegetative and microsite similarities (natural sites N1-N2 and N3-N6; see chapter 3). The separation of the three major groups followed the disturbance gradient, with a maximum mean loss of 326.3 ± 49.8 g C $m^{-2}a^{-1}$ for the degraded group, a moderate mean loss of 137.6 ± 47.4 g C $m^{-2}a^{-1}$

Site	CO_2-C - balance ± SE [g $m^{-2}a^{-1}$]	C – balance ± SE [g $m^{-2}a^{-1}$] E
D1	443.0 ± 70.0	450.1 ± 70.0
D2	360.0 ± 25.0	367.1 ± 25.0
D3	246.0 ± 46.0	255.0 ± 46.0
D4	225.0 ± 58.0	232.9 ± 58.0
R1	192.0 ± 21.0	200.5 ± 21.0
R2	127.0 ± 56.0	141.1 ± 56.1
R3	62.0 ± 65.0	71.3 ± 65.0
N1	14.0 ± 52.0	31.7 ± 52.0
N2	4.0 ± 42.0	16.4 ± 42.0
N3	-54.0 ± 31.0	-36.9 ± 31.1
N4	-87.0 ± 33.0	-52.3 ± 33.1
N5	-152.0 ± 48.0	-120.9 ± 48.0
N6	-157.0 ± 37.0	-111.8 ± 37.1
D1-D4	318.5 ± 49.8	326.3 ± 49.8
D1-D2	401.5 ± 47.5	408.6 ± 47.5
D3-D4	235.5 ± 52.0	243.9 ± 52.0
R1-R3	127.0 ± 47.3	137.6 ± 47.4
N1-N6	-71.0 ± 40.5	-45.6 ± 40.5
N1-N2	9.0 ± 47.0	24.1 ± 47.0
N3-N6	-112.5 ± 37.2	-80.5 ± 37.3

Tab. 6.3: CO_2-C balance (NEP; taken from chapter 4) versus Carbon balance for the individual sites and as average for the site groups. D1-D2 were the former peat-cut degraded sites, D3-D4 represent drained-only degraded sites. N1-N2 were the moist natural Calluna heathland and N3-N6 was the wet group of the natural sites.

for the restored group, and a notable average uptake of -45.6 ± 40.5 g C $m^{-2}a^{-1}$ for the natural group (see Tab. 6.3). Splitting the primary groups into subgroups revealed the impact of management upon the C balance within the degraded sites: The former peat cut sites (D1-D2) experienced significantly higher emissions, with 408.6 ± 47.5 g C $m^{-2}a^{-1}$, than the drained sites (D3-D4), with 243.9 ± 52 g C $m^{-2}a^{-1}$. Within the natural sites, variations in vegetation and microsite characteristics also impacted carbon emissions. The naturally moist Calluna heathland (N1-N2) had an average emission of 24.1 ± 47 g C $m^{-2}a^{-1}$, while the wet part of the natural sites (N3-N6) had an average uptake rate of -80.5 ± 37.3 g C $m^{-2}a^{-1}$.

Climatic relevance (GWP balance)

To assess the climatic effects of the total GHG fluxes, the individual global warming potentials were calculated for every site. The UNFCCC calculations are based on a 100-year time scale and the 1996 IPCC values (see Tab. 6.1), so this time scale was used for the presented calculations. Additionally, GWPs were calculated for a 500-year time scale because of the long-term role of mires as continuously accumulating systems, a unique feature among terrestrial ecosystems. For the calculation, the site-specific emission-factors for carbon dioxide, methane and nitrous oxide were multiplied by the corresponding GWPs (Tab. 6.1) for the time horizons of interest (see Tab. 6.4). The GWP-balance for the sum of the gases is expressed as CO_2-C equivalents (CO_2-C eq.).

The principal effect of the different time scales is that the GWP´s of methane and nitrous oxide, relative to carbon dioxide, decrease because of shorter lifetimes of these two species compared to carbon dioxide (see Tab. 6.1). Thus the differences between the site-specific GWP-100 year balances and the GWP-500 year balances were the result of the time dependent relevance of the composing gas-balances per site. Within the degraded group, the differences between the GWP-100 and the GWP-500 were relatively small (at maximum 11 g CO_2-C eq. $m^{-2}a^{-1}$). These differences can be attributed in the former peat-cut sites D1 and D2, to the influence of the nitrous oxide fluxes, since methane fluxes were insignificant on these severely drained sites (see chapter 5). On sites D3 and D4, nitrous oxide emissions were less important but methane emissions of 0.9 ± 0.3 and 2.0 ± 0.3 g C $m^{-2}a^{-1}$ (see Tab. 5.3) started to influence the GWP´s (see Tab. 6.4). For all other sites, the importance of the nitrous oxide balances on the GWP-balances was weak, due to virtually neutral activity or very small uptake rates on the corresponding sites (see chapter 5). However, the differences between the GWP-100 and GWP-500 values for the restored and natural sites can be attributed to the contribution of the methane fluxes at these sites and the changing climatic relevance of methane over time.

In general, the climatic relevance of the sites was reduced over the long-term. There was a notable shift within the group of natural sites (N1-N6), from contributing to global warming (GWP-sources) in the 100 year time frame (75 ± 43 g CO_2-C eq. $m^{-2}a^{-1}$) to mitigating global warming (GWP-sinks) over the 500 year time frame (-25.6 ± 40.8 g CO_2-C eq. $m^{-2}a^{-1}$). This difference was even more pronounced when the group of wet natural sites (N3-N6) was assessed separately. As a subgroup, these sites had GWP-100 similar to the entire natural group (77.5 ± 40.5 g CO_2-C eq. $m^{-2}a^{-1}$), but in the long term the GWP sink strength was greater (-52.6 ± 37.6 g CO_2-C eq. $m^{-2}a^{-1}$). In terms of individual sites, site N6, which had the highest contribution of methane fluxes to the GWPs, consequently had the largest difference between the two timescales. In the 100-year timescale, the site acted as a strong GWP-source, with 133.1 ± 40.6 g CO_2-C eq. m^{-2} a^{-1}. Over the long-term 500 year timescale, there was a notable shift to a GWP-sink, with -65.5 ± 37.4 g C eq. $m^{-2}a^{-1}$.

Site	GWP 100 CO_2-C eq. ± SE [g $m^{-2}a^{-1}$]	GWP 500 CO_2-C eq. ± SE [g $m^{-2}a^{-1}$]
D1	465.6 ± 71.1	455.4 ± 70.3
D2	367.3 ± 25.8	363.9 ± 25.2
D3	262.4 ± 46.1	251.5 ± 46.0
D4	232.4 ± 58.0	227.5 ± 58.0
R1	203.6 ± 21.0	195.7 ± 21.0
R2	180.8 ± 60.8	143.9 ± 56.5
R3	79.7 ± 65.1	67.6 ± 65.0
N1	95.2 ± 53.4	39.6 ± 52.1
N2	44.8 ± 42.2	16.8 ± 42.0
N3	22.5 ± 35.6	-29.9 ± 31.5
N4	123.2 ± 36.7	-20.7 ± 33.4
N5	31.1 ± 49.3	-94.2 ± 48.1
N6	133.1 ± 40.6	-65.5 ± 37.4
D1-D4	331.9 ± 50.2	324.5 ± 49.9
D1-D2	416.5 ± 48.4	409.6 ± 47.8
D3-D4	247.4 ± 52.1	239.5 ± 52.0
R1-R3	154.7 ± 48.9	135.8 ± 47.5
N1-N6	75.0 ± 43.0	-25.6 ± 40.8
N1-N2	70.0 ± 47.8	28.2 ± 47.1
N3-N6	77.5 ± 40.5	-52.6 ± 37.6

Tab. 6.4: Global warming potential balances expressed as CO_2-C equivalents of all three gases for the 100 year and 500 year timescale. D1-D2 were the former peat-cut degraded sites, D3-D4 represent drained-only degraded sites. N1-N2 were the moist natural Calluna heathland and N3-N6 was the wet group of the natural sites. SE in brackets.

The climatic relevance of bogs depends as much on the magnitude of the corresponding gas fluxes as on the timescale used for the calculations. This differentiated approach is especially necessary when evaluating the impact of natural sites. As illustrated, the degraded and restored sites are not as sensitive to different time scales because methane plays a smaller role in the site specific greenhouse-gas budget.

Carbon balance vs. GWP balance

The carbon balance alone is an important tool for assessing the functions of a mire ecosystem, in terms of growth. To assess the climatic relevance of the growth processes however, on the basis of the exchange of GHGs, the calculation of the individual GWP-balances is important. In order to better understand the relation between these two factors, they were plotted crosswise (see Fig. 6.1a and 6.1b). The C-balance was taken as independent of time, while the GWP-balance depends on the time horizon is used in its calculation, once non-CO_2 GHG-gases are included in the emission profile of the site (see Tab. 6.1). The most notable differences between the carbon balances and the GWP-balances occurred on sites where methane and nitrous oxide fluxes reached the highest levels. For nitrous oxide, these were the degraded sites, D1 and D2, on the extreme left of the graphs (Fig. 6.1). For methane these were the natural sites, including the restored site R2 (see chapter 5 and 6.4). The difference from the 1:1 line in Fig. 6.1a represents this effect, with the impact of nitrous oxide on the sites toward the left side of the graph (degraded sites) and the impact of methane on the right side (partly restored and all natural sites). The most striking feature of Fig. 6.1a was that for the 100-year timescale, no single site acted as a GWP sink. Even the four natural sites toward the right side of the graph, which were acting as carbon sinks, had a positive GWP-balance (i.e. contributed to the warming effect on the atmosphere). Carbon sequestration

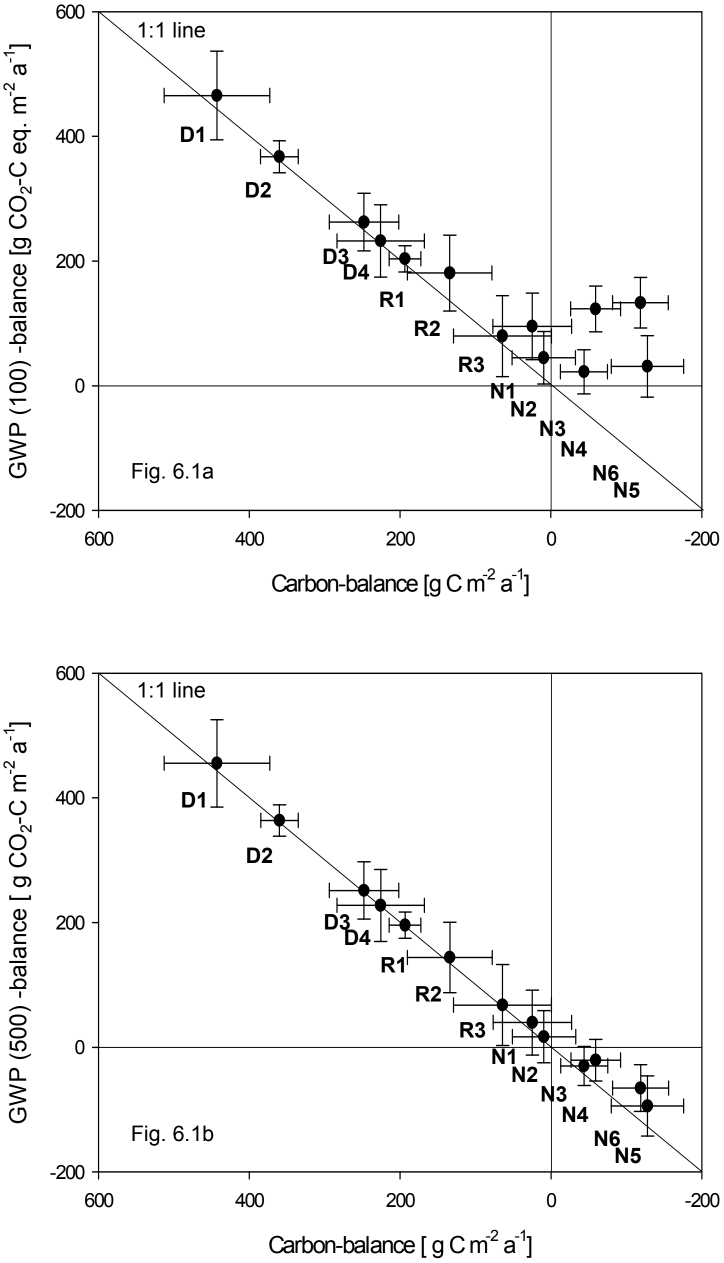

Fig. 6.1 Relation of carbon-balance with the GWP-balance. Error bars are SE.
Fig. 6.1a: GWP at the 100 years timescale
Fig. 6.1b: GWP at the 500 years timescale

alone cannot explain the climatic relevance of sites, where non-CO_2 gases play an important role on the GHG-balance.

Over the 500 year timescale, the effects of the non-CO_2 gases on the GWP-balance were reduced, due to their shorter lifetime relative to the reference gas CO_2 (Tab. 6.4). Therefore, in a comparison of the C-balance with the GWP-balance (Fig. 6.1b), the sites were close to a 1:1 line. In contrast to the GWP-balance for the 100-year timescale (Fig. 6.1a), the four natural sites were now acting as GWP-sinks. So in the long-term, the effect of methane is decreasing in importance and the natural bog ecosystems act both as C-sinks and as GWP-sinks, i.e. mitigating global warming.

Effect of restoration on the carbon balance and the GWP balance

The major goal of bog ecosystem restoration is to create site conditions, which in the short-term stop continuous degradation of the sites and in the long-term facilitate the newly growth of the bog-ecosystems. The carbon balance can be used as a functional indicator for assessing the effectiveness of restoration work and to identify the developmental stage of restored sites, in comparison to degraded and natural sites. The design of this study focused on detecting the effects of restoration on the C-balance and GWP-balance by comparing the restored sites with the degraded and natural sites. The arrows in Fig. 6.2 provide a visualization of the effect of restoration, on two degraded sites (D1 and D2), on the carbon balance (horizontal arrows) and on the GWP-balance (vertical arrows), by linking these sites to the corresponding restored sites. The horizontal arrows reflect significant reductions in C-release, which are already apparent in the first year (D2 to R1) and are even more pronounced after ten years of restoration (D1 to R3). The restored sites were still acting as C-sources; however, they emitted far less carbon compared to the degraded sites. Similar results were observed for the GWP-balance, as both vertical arrows indicate significant reductions in the climate warming effect of the degraded sites, once restored. Although still not acting as a C-sink, site R3 reached a similar GWP-balance level as the natural sites, which in the 100-year timescale were still being influenced by the importance of methane fluxes in

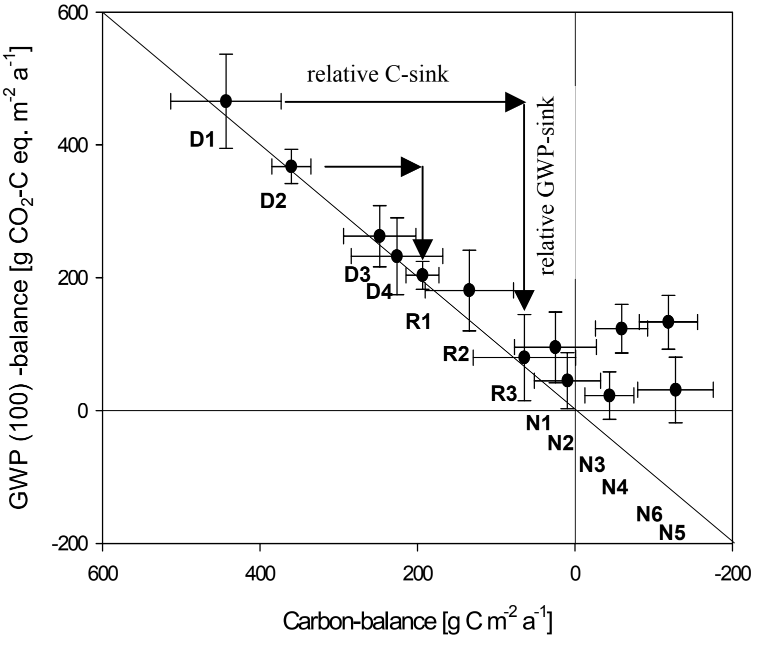

Fig. 6.2: Exemplary effects of restoration on the C-balance and GWP-balance (100 years timescale). Error bars are SE.

their climatic relevance (see Tab. 6.4 and Fig 6.2a). Restoration can therefore lead to both reduced emissions of carbon and to climate change mitigation. The assessment of the mitigation effects depends strongly on the targeted gases. Therefore, a differentiated approach is followed for (i) single CO_2-C balance, like in the Bavarian climate protection programme, (ii) the GWP-balance according to the good practice guidance (IPCC, 2003) without including CH_4-fluxes, and (iii) a GWP-balance for all three gases, which is scientifically the most appropriate approach, but still does not follow the good practice guidance and consequently is not included in the accounting scheme of the Kyoto protocol.

The restoration of abandoned former peat cut sites (D1, D2) led to a mean reduction of C-emissions of 339.5 ± 53.3g CO_2-C m^{-2} a^{-1} (see Tab. 6.5). The reduction of the global warming effect was 353.9 ± 54 g CO_2-C eq m^{-2} a^{-1} (for IPCC 2003) and 336.8 ± 54 g CO_2-C eq m^{-2} a^{-1} for all gases, respectively (see Tab. 6.5). The restoration of drained but uncut peat sites (D3, D4) led to a reduction of C-emissions of 108.5 ± 53.3 g CO_2-C m^{-2} a^{-1}, whereas the climate mitigation effect was 109.6 ± 53.3 g CO_2-C eq m^{-2} a^{-1} (for IPCC 2003) and 66.6 ± 55 g CO_2-C eq m^{-2} a^{-1} (for all gases; Tab. 6.5).

Degraded site	restored to	CO_2-C balance ± SE [g C $m^{-2}a^{-1}$]	GWP balance ± SE [g CO_2-C eq $m^{-2}a^{-1}$] (IPCC, 2003)[1]	GWP balance ± SE [g CO_2-C eq $m^{-2}a^{-1}$] (all gases)
D1	R3	- 381.0 ± 67.5	-403.2 ± 68.1	-386.0 ± 68.1
D2	R1 (first year)	-168.0 ± 45.0	-174.4 ± 45.4	-163.6 ± 45.5
mean D1/D2	R3	-339.5 ± 53.3	-353.9 ± 54.0	-336.8 ± 54.0
mean D3/D4	R2	-108.5 ± 53.3	-109.6 ± 53.3	-66.6 ± 55.0

Tab. 6.5: Effects of restoration on the CO_2-C balance and the GWP balance (100-year timescale) as a difference of the C balance and the GWP balances on the degraded and corresponding restored sites. Consistent with application of the good practice guidance, (IPCC, 2003) CO_2 and N_2O are reportable.

Climatic relevance of the bogs in the forelands Alps

For the estimation of the climatic relevance of bogs, a scientific approach base on the inclusion of all three gases and the whole bog area in all categories (natural, degraded and restored) for the calculations. However for reporting under the UNFCCC, natural areas are excluded, as well as no guidelines are available up to now for reporting methane emissions (IPCC 2003). For the assessment of the climatic relevance the per capita approach was used, where the bog related fluxes are compared to the average emission per person. As no differentiated data on the individual emission profile of the countys of the bog area were available, the average Bavarian state per capita emission values were used for the calculation. Background for the calculations for Tab. 6.6 were the areas of the different bog categories (Tab. 6.2) multiplied with the corresponding EF´s (Tab. 6.3 and 6.4) and divided by the population of the southern bavarian mire belt. Generally the climatic relevance of the bogs in the mire belt is around 5 % of the peak Bavarian per capita fossile CO_2 emission. The biggest differences between the calculation procedures appeared between "CO_2 fluxes, natural sites included" and "GWP 100 in CO_2 eq. (all gases), natural sites included", attributed mainly to the impact of methane emissions of the included natural sites. However, the differences expressed as percentage of the average Bavarian emissions are small (~0.05%).

	Per capita bog emission ± SE [kg CO_2 eq. $pers^{-1}a^{-1}$]	Relation of bog emission in percent to fossil Bavarian peak per capita emission (1998) of 7.462 (t CO_2 $pers^{-1}a^{-1}$). ± SE [%]
CO_2 fluxes, natural sites excluded	367 ± 68	4.91 ± 0.92
GWP 100 in CO_2 eq. (IPCC 2003), natural sites excluded	372 ± 69	4.98 ± 0.93
GWP 100 in CO_2 eq. (all gases), natural sites excluded	384 ± 69	5.14 ± 0.93
CO_2 fluxes, natural sites included	357 ± 74	4.78 ± 0.99
GWP 100 in CO_2 eq. (IPCC 2003), natural sites included	362 ± 75	4.85 ± 1.00
GWP 100 in CO_2 eq. (all gases), natural sites included	394 ± 75	5.28 ± 1.01

Tab. 6.6: Estimated contribution of bog emissions to the average per capita emissions in the southern German mire belt in the forelands of the Alps, under different calculation methods. Under the application of the good practice guidance (IPCC, 2003), only CO_2 and N_2O are reportable.

Climate mitigation via bog restoration to help meet Bavarian state climate protection programme and the Kyoto protocol commitments

The carbon sequestration and climate mitigation potentials of restoration of degraded bog areas in the southern German mire belt in the forelands the Alps (see Tab. 6.7) were estimated on a full area approach to get an impression of the possible contributions of restoration to climate protection measures. The areas of bogs with restoration potential were multiplied by the corresponding emission factor, which in the case of restoration was a removal factor (RF; Tab. 6.7), representing a reduction potential. The removal factor (RF) for carbon sequestration (calculated as t CO_2-C $ha^{-1}a^{-1}$) on abandoned peat cut areas is more than 3 times higher than the RF for sites with drainage in preparation of peat cutting. This relationship is maintained when comparing the RF for climate mitigation (calculated as t CO_2-C eq. $ha^{-1}a^{-1}$, IPCC 2003). The relationship is even more pronounced when comparing the RFs under the full gas accounting scheme (calculated as t CO_2-C eq. $ha^{-1}a^{-1}$, all gases), with a climate mitigation effect that is approximately 5 times more efficient (Tab. 6.7). On the other hand, the land area of drained but uncut bogs is 3.5 times greater than the area of abandoned peat cut sites. Consequently, the total amount of sequestered carbon (calculated as t CO_2-C) and climate mitigation values (calculated as t CO_2-C eq. $ha^{-1}a^{-1}$, IPCC 2003) for the entire restoration area were of similar magnitudes for both site-groups. However, when climate mitigation was calculated based on all three gases, the absolute effect of restoration on former peat cut sites is around 1.5 times higher than the effect of restoration on the drained areas (see Tab 6.7). These comparisons allow for future priorities to be set in terms of the effectiveness of restoration on climate mitigation.

bog area with restoration potential (estimation) [ha]		RF ± SE [t CO_2-C ha^{-1} a^{-1}]	RF ± SE [t CO_2-C eq. $ha^{-1}a^{-1}$] (IPCC 2003)	RF ± SE [t CO_2-C eq. $ha^{-1}a^{-1}$] all gases	carbon-sequestration ± SE [t CO_2-C]	climate-mitigation ± SE [t CO_2-C eq.] (IPCC 2003)	climate-mitigation ± SE [t CO_2-C eq.] all gases
abandoned peat cut areas	10.000	3.39 ± 0.53	3.54 ± 0.54	3.37 ± 0.54	33900 ± 5300	35400 ± 5400	33600 ± 5400
drained but uncut	35.000	1.11 ± 0.53	1.1 ± 0.53	0.67 ± 0.55	38850 ± 18550	38500 ± 18550	23450 ± 19250

Tab. 6.7: Potential carbon sequestration and climate mitigation (GWP 100) by bog restoration in the southern German mire belt in the forelands of the Alps. RF = removal factor. Under application of the good practice guidance (IPCC, 2003) CO_2 and N_2O are reportable.

Restoration of bogs to build up biological sinks cannot be done everywhere, as it is restricted to the area of bog distribution. This is in contrast to afforestation (Art. 3.3 Kyoto-protocol), which is not as strictly limited to a specific landscape. Therefore, the relevance of carbon sequestration and climate mitigation via bog restoration for climate protection should be assessed with a regionalized scope, where the target area matches the distribution of bogs. Consistent with these guidelines, a per capita approach was followed, based on the population within the Bavarian alpine counties, which entirely cover the distribution of bogs (see materials and methods).

The potential per capita reduction was calculated by dividing the carbon sequestration and climate mitigation potentials (Tab. 6.7) by the population in the mire belt ($1.23x10^6$ pers.), resulting in per capita reductions of 59.1 ± 19.4 kg CO_2-C $pers^{-1}a^{-1}$, 60.1 ± 19.5 kg CO_2-C eq. $pers^{-1}a^{-1}$ (IPCC 2003) and 46.5 ± 20 kg CO_2-C eq. $pers^{-1}a^{-1}$ (all gases) for the various accounting methods, respectively (see Tab. 6.8). These individual reduction potentials were then compared to the necessary per capita reductions needed to fill the gaps in the Bavarian climate protection programme and the more challenging goals of the Kyoto commitment (Tab. 6.8). Within the mire region, bog restoration can help substantially by filling 35.4 % of the gap of the Bavarian climate protection programme in terms of carbon sequestration, 35.6 % in terms of climate mitigation (IPCC 2003), and even 27.0 % when all gases are considered (Tab. 6.8). The Bavarian climate protection programme explicitly denies the application of reduction profiles according to the Kyoto commitments. However, the contributions towards filling the gaps via bog restoration when the Kyoto goals are applied, amount to 12.9 %, 13.0 % and 10.0 %, depending on the calculation method (Tab. 6.8).

	Potential per capita reduction via bog restoration in the mire belt ± SE	Necessary per capita reduction to meet the goals of the Bavarian climate protection programme ± SE	Necessary per capita reduction to meet the Kyoto commitments ± SE	Percentage of climate compensated population in the mire belt ± SE Bavarian climate protection programme	Percentage of climate compensated population in the mire belt ± SE Kyoto commitments
	[kg C $pers^{-1}a^{-1}$]	[kg C $pers^{-1}$] *	[kg C $pers^{-1}$] *	[%]	[%]
CO_2-C basis	59.1 ± 19.4	167.2	460.2	35.4 ± 11.6	12.9 ± 4.2
C-eq basis (IPCC 2003)	60.1 ± 19.5	168.6	461.6	35.6 ± 11.5	13.0 ± 4.2
C-eq basis (all gases)	46.5 ± 20.0	171.9	464.9	27.0 ± 11.7	10.0 ± 4.3

Tab. 6.8: Contribution of bog restoration to the Bavarian climate protection programme and the commitments of the Kyoto protocol, calculated by a regionalized approach for the southern German mire belt in the forelands of the Alps. *published climate protection goals are expressed in terms of CO_2. ± SE. Under the application of the good practice guidance (IPCC, 2003), only CO_2 and N_2O are reportable.

Discussion

Carbon balance

The carbon balance can be studied in undisturbed mires via a coring method which uses radiocarbon dating, providing average values for the long-term rate of carbon accumulation (LORCA). For northern bogs, the LORCA ranges from -6.3 g C $m^{-2}a^{-1}$ (Turunen & Turunen, 2003) to over -24.0 g C $m^{-2}a^{-1}$ (Turunen & Tolonen, 1996), with reports up to -31.4 g C $m^{-2}a^{-1}$ (Botch et al., 1995). However, for the assessment of recent rates of carbon accumulation, the LORCA is not appropriate. The long-term time interval, spanning the entire life history of a mire site, along with the decomposition over the whole profile, masks the actual accumulation rates. The calculation of the recent rate of carbon accumulation (RERCA) can help to fill this gap. The time window used by Turunen et al. (2004) was between 50 and 150 years. The average values over the last 50 years showed significant variation between sites with different nitrogen deposition levels. A mean RERCA of -137.7 g C $m^{-2}a^{-1}$ was obtained for the highest Nitrogen deposition class (mean 7.9 kg N $ha^{-1}a^{-1}$; Turunen et al., 2004). For the lowest Nitrogen deposition class (mean 3.2 kg N $ha^{-1}a^{-1}$), the RERCA was -110.1 g C $m^{-2}a^{-1}$ (Turunen et al., 2004). The RERCA is significantly higher than the corresponding LORCA and it can be assumed that the RERCA represents the best estimate of the recent rate of carbon accumulation.

The alternative means to assess the carbon balance via gas-exchange measurements of CO_2 and CH_4, is used to derive estimates for the actual carbon exchange. In principle, these fluxes represent the net-exchange of the entire peat-body, which should be more accurately reflected by the LORCA than the RERCA, which is based solely on the upper portion of the profile. Theoretically, gas-exchange based C-accumulation rates should reflect the actual functiong of the entire peat-body, meaning they should provide the LORCA under present day conditions. Indeed, Friborg (unpublished) found C-accumulation rates (CO_2 fluxes minus CH_4 fluxes) of -18 C g $m^{-2}a^{-1}$ in boreal bogs. Within a sub-arctic region, Aurela et al. (2002) assessed the C accumulation via gas-exchange measurements and observed rates of -15 g C $m^{-2}a^{-1}$. These data fall more in the range of the LORCA than the RERCA and seem to confirm the hypothesis. However, the data presented in this paper for southern German bog ecosystems are significantly higher, with a mean C accumulation of –45.6 g C $m^{-2}a^{-1}$ over all natural sites and even up to –80.5 g C $m^{-2}a^{-1}$ on the wettest natural sites N3-N6 (see Tab. 6.3). These values are closer to the RERCA values of Turunen et al. (2004) than to the LORCA. Moore et al. (2002) found similarly high C-accumulation rates of -60 g C m^{-2} a^{-1} for an ombrotrophic boreal Canadian bog, using chamber-based measurement methods. Campbell et al. (unpublished) found rates up to –200 g CO_2-C $m^{-2}a^{-1}$ for an undisturbed bog in New Zealand, but did not include losses via CH_4 and DOC/DIC. The apparent overestimation of the published C-accumulation (based mainly on boreal peat cores) by gas-exchange measurements may be interpreted as an effect of several interacting processes: Comparing the NEE of similar bog ecosystems in different latitudinal positions revealed similar ranges, as both GPP and R_{ECO} are rising towards lower latitudes almost compensating each other

(Christensen et al. 2003). The relatively high NEP values of the natural sites N3-N6 (-80.5 g C $m^{-2}a^{-1}$) may be attributed to the (i) longer growing period in the temperate climate zone, without limited water supply (precipitation surplus by orographic rains at the border of the alps), allows for a longer growing season and lower carbon losses during the winter period (snow cover duration of 65 days). Additionally (ii) the elevated nitrogen input (around 25 kg $ha^{-1}a^{-1}$) may have enhanced uptake rates, as Turunen et al. (2004) found the recent C-accumulation rising in conjunction with higher levels of nitrogen input.

However, interannual variability is levelled out by LORCA but may influence the NEP rates, deriving from gas-exchange measurements. Lafleur et al. (2003) demonstrated via multi-year eddy covariance measurements, that an ombrotrophic Canadian bog functioned as a net carbon sink of –60 g C $m^{-2}a^{-1}$ over three consecutive years, while in the fourth year, during a dry summer, the NEE dropped down to around 0 g C $m^{-2}a^{-1}$. The effects of an exceptionally dry summer on the carbon balance were also noted by Alm et al. (1999), which helps confirm the carbon balance's sensitivity to interannual differences. Arneth et al. (2002) on the other hand demonstrated, that the NEE curves of three consecutive years for a Siberian bog where very similar in dynamics and levels, closely reflecting the similar climate in these measurement years.

The climate during the measurement year of the presented study on southern German bogs however, was very similar to the long-term average; with an insignificant difference in precipitation and an average temperature that was only 1° C higher than average (see Materials and Methods). Therefore, the derived C-accumulation rates were taken as robust estimates of the actual average functioning of the studied bog ecosystems.

As for the natural sites, the effects of longer growing period, with higher average temperatures influences the NEP of the degraded and restored sites. The NEP´s are higher than recently published European average values, which were biased towards the north because the majority of the studies were undertaken in the boreal and subarctic regions (Byrne et al. 2004). The mean C-loss of the restored sites R1-R3 (137.6 g C $m^{-2}a^{-1}$; see Tab. 6.3) was higher than the average 76 g C $m^{-2}a^{-1}$ value measured in corresponding studies (Byrne et al., 2004). The average C-loss of the drained sites D3-D4 (243.9 g C $m^{-2}a^{-1}$; see Tab. 6.3) was also higher than the average across Europe, with a C-loss of 101 g $m^{-2}a^{-1}$ (Byrne et al., 2004). In addition to the varied influence of climate between north and south, the type of treatment may have influenced these lower values, as the studies from the north seemed to refer only to slightly drained sites. Finally, the abandoned former peat cut sites, D1-D2, with an average loss of 408.6 g C $m^{-2}a^{-1}$ (see Tab. 6.3), confirmed the trend towards higher emission in the temperate region, when compared to the boreal and subarctic region. The reported EU average C-loss was 238 g C $m^{-2}a^{-1}$ (Byrne et al., 2004). However, the tendency of increasing carbon loss from restored, to drained, to abandoned peat cut sites was consistent across Europe, but the average values were lower than exclusively in the temperate zone. According to the explanation of higher uptake rates in temperate zones, where the ecosystems can profit from longer growing seasons (with higher average temperatures and shorter periods of snow cover), the degraded sites were loosing more carbon for the same reasons. Here the rising C-loss via enhanced ecosystem respiration R_{ECO} seems to be the crucial factor. Longer periods with higher temperatures provoke higher respiration fluxes (R_{ECO}) and sparse or lacking vegetative cover with low GPP-rates at these sites could not compensate the rising soil borne C-emissions. It is interesting however, that the NEP rates for natural ombrotrophic bogs in Canada (e.g. Moore et al, 2002, Lafleur et al., 2003) match nicely with the values of this study from southern Germany. This may be a result of the naturally drier Canadian bogs, from which only a small amount of carbon was lost via the CH_4 pathway (approx. 1 g C $m^{-2}a^{-1}$; Roulet, pers. comm.). For this reason, the shorter growing season was sufficient to lead to C-accumulation rates similar to those in temperate bogs.

Relation of the carbon balance to the climatic relevance (GWP)

The carbon balance is a good indicator for growth assessments in mire ecosystems. The sensitive relationship to the status of an ecosystem (natural, degraded and restored), allows one to evaluate the functioning of bog sites (see Tab. 6.3). The success of restoration-works can then be estimated, by using a comparative approach between natural and degraded sites. For the assessment of the climatic relevance of ecosystems however, the fluxes of CH_4 and N_2O have to be measured in addition to the net CO_2-exchange.

The comparison of the carbon balance with the climatic relevance (GWP balance) of the bog sites revealed that the degraded sites, D1-D4, and the restored sites, R1 and R3, did not reveal significant differences (see Tab. 6.3, 6.4 and Fig. 6.1). The natural sites, N1-N6, and the restored site R2 showed pronounced differences between the two indicators. This was most obvious on sites N3-N6, which acted as carbon sinks, with uptakes ranging from –54 g C $m^{-2}a^{-1}$ to –120.9 g C $m^{-2}a^{-1}$ (Tab. 6.3), but were simultaneously acting as GWP-sources from 22.5 g CO_2-C eq. $m^{-2}a^{-1}$ to 133.1 g CO_2-C eq. $m^{-2}a^{-1}$ (Tab. 6.4; calculated for the 100-year timescale). This large difference was provoked by the increasing impact of methane fluxes on the GHG-balance for the natural sites. The wettest site N6 (*Sphagnum* hollow), experienced the maximum loss via CH_4-emissions (38.2 g C), which accounted for 24 % of the carbon uptake via CO_2 fixation. The relative extreme was found at site N4 (*Eriophorum* hummock), which experienced a loss of 27.7 g C via CH_4-emission, the equivalent of 32 % of the carbon uptake via CO_2 fixation. These methane fluxes contribute significantly to the 290 (N6) and 210 (N4) g CO_2-C eq $m^{-2}a^{-1}$ added to the global warming potential, but were cut in half by the CO_2-C uptake of the sites (-157 (N6) and -87 (N4) g CO_2-C $m^{-2}a^{-1}$). As a result of the warming impact of methane emissions, all studied natural bog ecosystems contributed to global warming for the 100-year timescale, unless they were acting as carbon sinks. This is in accordance with average EU results (Byrne et al., 2004), which report a GWP-source strength for natural bogs of 19.2 g CO_2-C eq $m^{-2}a^{-1}$. This value is lower than the average of all the natural bog sites (N1-N6) with 75 g CO_2-C eq $m^{-2}a^{-1}$(see Tab. 6.4) measured in this study. In contrast to higher C-accumulation rates, the temperate bogs demonstrated higher warming effect than the EU average, because the higher NEP of the temperate sites cannot compensate for the enhanced climatic effect of rising CH_4-emission. Whiting & Chanton (2001) showed that subtropical and temperate wetlands are contributing to global warming at a 100-year timescale, whereas northern wetlands are found to be around the "greenhouse compensation" point. However, here the ecosystem type seems to bias a strictly latitudinal interpretation, as from north to south the study compared arctic tundra, boreal bogs, temperate fens and subtropic swamps.

As bogs are ecosystems that function over the longer-term, with carbon accumulation over thousands of years, the climatic relevance over a longer period of time is of interest. For that reason, the difference of the climatic impact between the GWP-100 and the GWP-500 was tested. The GWP-average on the natural sites shifted from contributing to global warming in the 100-year timescale (75 g CO_2-C eq $m^{-2}a^{-1}$; see Tab. 6.4) to mitigating global warming in the 500-year timescale (-25.6 g CO_2-C eq $m^{-2}a^{-1}$; see Tab. 6.4). This tendency is similarly reflected in the EU-average values (Byrne et al., 2004), with GWP´s for natural bogs of 19.2 g CO_2-C eq $m^{-2}a^{-1}$ over the 100-year timescale, compared to -10.3 g CO_2-C eq $m^{-2}a^{-1}$ for the 500-year timescale. Over the long-term, temperate bogs have an even more pronounced cooling potential than boreal and sub-arctic EU-bogs.

Effects of restoration on the carbon balance and the GWP balance

Restoration of the studied bog ecosystems led to reduced C-emissions and reduced global warming impacts, but could not convert the sites absolutely to carbon sinks or GWP sinks (see Fig. 6.2 and Tab. 6.5). Restoration of former peat cut sites (D1-D2) led to a greater reduction of carbon emissions potential (-339.5 ± 53.3 g C $m^{-2}a^{-1}$, see Tab. 6.5), than did the restoration of the drained sites (D3-D4; -108.5 ± 53.3 g C m^{-2} a^{-1}, see Tab. 6.5). In order to integrate the current accounting scheme of the Kyoto Protocol, the effect on the GWP-balance was calculated according to IPCC 2003, without the inclusion of methane fluxes (unreportable until now). Hence, restoration of former peat cut sites (D1-D2) led to a GWP-reduction of –353.9 ± 54 g CO_2-C eq. $m^{-2}a^{-1}$ (see Tab. 6.5), while restoration on drained sites (D3-D4) led to a GWP-reduction of –109.6 ± 53.3 g CO_2-C eq. $m^{-2}a^{-1}$ (see Tab. 6.5). The latter value did not differ significantly from the reduction potential for carbon emissions (see above). However, from a scientific point of view, all gases should be included in the evaluation of the climate mitigation potential of bog restoration. Following this approach, the GWP-reduction for the restoration of former peat cut sites (D1-D2) amounts to -336.8 ± 54 g CO_2-C eq. $m^{-2}a^{-1}$ (see Tab. 6.5), which was not significantly different from the carbon emission reduction potential. The restoration of drained sites (D3-D4), led to a GWP-reduction potential of –66.6 g ± 55 CO_2-C eq. $m^{-2}a^{-1}$ (see Tab. 6.5) when all gases were finally included. This lower reduction potential was a result of higher methane emissions on the target site R2 (see chapter 5).

For restored Finnish cut-way peatlands, Tuittila et al. (1999) found that restoration can even lead to a switch from carbon sources to absolute carbon sinks within five years of rewetting. The EU-average for restoration of former peat-cut sites (Byrne et al. 2004) however, followed trends that were similar to the bog ecosystems studied across southern Germany, but at lower levels, with a relative carbon-sink potential of -171 C $m^{-2}a^{-1}$ and a relative GWP-sink potential of -176.5 C $m^{-2}a^{-1}$.

Restoration of bog ecosystems is a means of climate mitigation and the restored sites can be interpreted as relative C-sinks and relative GWP-sinks. According to the IPCC 2003 definitions, these restoration effects on the carbon balance and the GWP balance can be clearly attributed to human activities. Which means the effects can potentially be accounted for as direct human induced C-changes for climate mitigation. The reduction of a source can be accounted for as a climate mitigation measure.

Actual climatic relevance of bogs in southern Germany

The climatic relevance of the bogs that exist in the counties located in the forelands of the Alps in southern Germany was calculated via multiple approaches, with and without the inclusion of the natural sites, differentiated in (i) CO_2 only (comparable to the CO_2 per capita reporting), (ii) CO_2 and N_2O (according to IPCC 2003), and (iii) all three gases (scientific approach). Each calculation scheme led to a climatic relevance of about 5 % for the bog fluxes, when compared to the peak per capita emissions of the population in the corresponding counties (see Tab. 6.6). Uncertainties of the estimates were around 20% (see Tab. 6.6). Höper & Blankenburg (2000) estimated the climatic relevance of the bogs of Lower Saxony at approximately 3% of the entire emissions of Lower Saxony. Data regarding a regionalized approach are not available for Lower Saxony. Generally, the bogs seem to represent a limited contribution to global warming, both in Lower Saxony and in the forelands of the Bavarian Alps. Bogs have not been included in the national reporting scheme to the UNFCCC until now.

Climate mitigation via bog restoration - relation to the Bavarian state climate protection programme and the Kyoto protocol commitments

The ongoing discussion about the use of biological sinks for climate mitigation is controversial. Missfeldt & Haites (2001) criticise the negative incentive that is created for developing technical solutions to emissions reduction, once the sinks reach significant levels. Moreover, the so called “slow in, fast out” phenomenon can affect the permanence of the man-made or natural sinks, induced by impacts such as fires, storms etc. (which apply primarily to the forest sinks; Noble & Scholes, 2001). Soil respiration is of particular importance for projections regarding the feedback mechanisms of the climate system, with the biosphere and biological sinks. Global soil respiration as a flux is an order of magnitude higher than anthropogenic emissions, and this flux is sensitive to both soil temperature and humidity changes (Reichstein et al., 2003). Both of these are factors which are likely to change under common climate change scenarios. Principally, the changes in R_{ECO} (ecosystem respiration) determine the changing sink strength of an ecosystem, as GPP is physiologically limited. Therefore, the interannual variability of the carbon fluxes and the difficulties involved in measuring fixed carbon with sufficient accuracy were critical points of concern (Noble & Scholes, 2001).

On the other hand, biological sinks offer low-cost carbon sequestration and are relatively quickly available as mitigation options (Smith, 2004c). Their major potential should be recognized as potential bridge to reach the period when cost intensive changes in the energy sector are realized (Noble & Scholes, 2001). As any mitigation strategies should be undertaken as soon as possible, biological sinks should play an important role for at least the next 20-30 years (Smith 2004c).

The potential inclusion of bog restoration as biological sinks under the Kyoto protocol is still not fully defined. Article 3.4 (Kyoto Protocol) sets a possible frame for the inclusion of mires, once Germany decides to report landuse of wetlands. For exploring the climate mitigation potential via bog restoration, a comparative approach was followed, focusing on carbon dioxide exclusively (according to the Bavarian climate protection programme), on a combination of CO_2 and N_2O (according to the IPCC 2003 guidelines), and on all three gases (scientific accounting). The achievable reduction levels, on a per capita basis, were then compared to the goals of the Bavarian climate protection programme and to the more rigorous Kyoto commitments. As the results illustrated, (see Tab. 6.8) 27-36 % of the gap in the Bavarian climate protection programme (-170 kg CO_2-C per capita) can be filled for the local population in the mire belt via bog restoration exclusively. Even if Bavaria would target on fulfilling the Kyoto commitments (-460 kg CO_2-C per capita), bog restoration could contribute between 10 and 12% of the per capita reduction required for the mire belt population. Bog restoration can certainly be used as a tool for meeting Bavarian climate policy goals, once a regionalized approach is followed. Therefore, the Bavarian mire conservation programme should be re-evaluated in terms of prioritising the restoration of mires with highest sequestration potential.

Conclusions

Bog ecosystems receive attention in climate change topics because of their functioning as carbon-sinks in a natural status, and the potential to release big amounts of carbon once drained or peat cut. However, for assessing the climatic relevance the carbon balance is not sufficient, as the exchange of all three climatically relevant trace gases from ecosystems (CO_2, CH_4 and N_2O) must be included in the approach and their flux-balances have to be multiplied with the individual global warming potential (GWP). Natural southern German bogs are significant carbon sinks, but enhance climate warming (GWP balance) at the 100 years timescale. Former peat cut sites on the other hand act as strong carbon sources, at a level five times the sink effect of natural bogs. The climatic relevance (GWP balance) of the degraded sites is around four times higher than the one of the natural sites. Calculating the climatic relevance for the longer term perspective of the 500 years timescale, natural sites shift to act as GWP-sinks, thus mitigating climate warming. Restoration of degraded sites does not lead directly to C-sinks or GWP-sinks. However, the emissions of the degraded sites can be cut off significantly, offering a climate mitigation option by appropriate land management. The estimated climate mitigation effect of restoring the degraded Southern German bogs in the forelands of the Bavarian Alps could significantly help to meet the targets of the Bavarian climate protection program, regionalized for the population in the mire belt. Therefore mire restoration should be taken into account as a potential climate mitigation measure.

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DER AUSTAUSCH VON KLIMARELEVANTEN SPURENGASEN IN NATURNAHEN, DEGRADIERTEN UND RENATURIERTEN REGENMOOREN: KANN DIE RENATURIERUNG ZUM KLIMASCHUTZ BEITRAGEN?

Eine Synthese

Über den Austausch von klimarelevanten Spurengasen sind Ökosysteme in die Prozesse des Klimawandels eingebunden. Der Bedeutung dieser nicht-fossilen Spurengasflüsse wurde durch die Einbeziehung des Sektors „Landnutzung, Landnutzungswandel und Forst“ (LULUCF) in die Berichterstattung zur Klimarahmenkonvention Rechnung getragen. In den neuesten Good Practice Guidance (GPG; IPCC, 2003) werden Verfahrensweisen für die Verrechnung von Spurengasflüssen in Bezug zu Feuchtgebieten detailliert, wobei gerade die geringe Anzahl an Untersuchungen zum Einfluss von Nutzungsänderungen auf die Spurengasbilanzen zu großen Unsicherheiten in den ermittelten „default values“ (Standardwerte, die bei fehlenden eigenen Untersuchungen für die Berichterstattung verwendet werden sollen) führen. Für die Verrechung der CH_4-Flüsse sind zudem aufgrund des lückigen Kenntnisstandes noch keine Verfahrensregeln vorgelegt worden. Um die Güte der Berichterstattung zu verbessern, werden die Staaten daher aufgerufen, eigene aktuelle Erhebungen zur Kategorisierung und Ausdehnung der Ökosystemtypen und eigene Messungen und Modellierungen des Spurengasaustausches in diesen Typen durchzuführen.

Eine Zusammenstellung des derzeitigen Kenntnisstandes zum Spurengasaustausch in Mooren der EU (Byrne et al., 2004) zeigt, dass die Moore hinsichtlich ihrer Klimawirksamkeit relevante Ökosystemtypen sind und gerade für Deutschland eine besondere Rolle spielen: Deutschland steht hinsichtlich der Moorfläche an siebter Stelle im EU-Vergleich, hinsichtlich der Klimawirksamkeit aber an zweiter Stelle. Die abgeschätzten derzeitigen Gesamt-Emissionen aus Mooren entsprechen der gesamten Senke aus dem deutschen Wald (Benndorf, UBA, persönliche Mitteilung 2005). Die Untersuchungslücken sind allerdings nach wie vor groß.

Die vorliegende Arbeit ist die bisher einzige Untersuchung in süddeutschen Regenmooren, die (i) eine Messtechnik eingesetzt hat, die es erlaubt, den Netto-Ökosystem-Austausch (NEE) an CO_2 zu erfassen, die (ii) alle drei biogenen klimarelevanten Spurengase (CO_2, CH_4, N_2O) parallel erfasste und (iii) einen vollständigen Gradienten von naturnahen über degradierte zu renaturierten Standorten abgebildet hat. Übergeordnetes Ziel der Untersuchungen war es, die Kohlenstoffbilanz und die Klimarelevanz von naturnahen, degradierten und renaturierten Regenmoorstandorten im Vergleich zu ermitteln und die Eignung von Renaturierungsmaßnahmen für den Klimaschutz zu prüfen.

1. Klimatisiertes Haubensystem zur Messung des Nettoökosystemaustauschs (NEE)

Ein wesentlicher Grund für die bisher fehlenden Untersuchungen zum NEE von CO_2 und zum Austausch aller drei klimarelevanten Spurengase von Regenmooren war der Mangel an geeigneten Messtechniken. Zwar hat sich mit der Eddy-Kovarianz-Methode ein Standardverfahren für die Messung des NEE über Ökosystemen etabliert und bildet die Kernmesstechnik für weltweit aufgebaute Messnetze (FLUXNET; Aubinet et al., 2000), die Erfordernisse der Erfassung von NEE und Spurengasflüssen in Regenmooren entlang eines Degradierungs-Gradienten stellen aber folgende Anforderungen an die Messtechnik, die von der Eddy-Kovarianz nur unzureichend erfüllt werden können: (i) Möglichkeit der Messung eines kleinteiligen Mosaiks aus naturnahen, renaturierten und degradierten Ökosystemen (Eddy-Kovarianz integriert über große Flächen von mindestens einigen Hektaren und kann damit nicht eingesetzt werden, um die Kleinstandorte hinsichtlich des individuellen Austauschverhaltens ausreichend zu differenzieren), (ii) Unabhängigkeit von Strom, da insbesondere in Regenmooren eine Stromversorgung nicht in erreichbarer Nähe gegeben ist (Eddy-Kovarianz-Stationen sind in der Regel auf Strom angewiesen), (iii) Möglichkeit, alle drei klimarelevanten Spurengase parallel zu erfassen (Eddy-Kovarianz ist faktisch auf CO_2 beschränkt, da die Sensoren für Methan und Lachgas nicht schnell genug oder unverhältnismäßig teuer sind). Darüber hinaus ist der nächtliche Mangel an turbulenter Strömung in der Grenzschicht ein generelles Problem der Eddy-Kovarianz-Methode, das die zuverlässige Ermittlung von Tagesbilanzen erschwert.

Als Alternative bietet sich daher an, den Spurengasaustausch mit Haubentechniken zu erfassen. Die in vergleichbaren Studien in Nordamerika (Bartlett et al., 1990, Whiting et al., 1992) und Finnland (Tuittila et al., 1999) eingesetzten Haubentechniken sind zwar geeignet, das NEE zu erfassen. Allerdings handelt es sich um sehr teure Systeme, die zudem mindestens zwei Personen für die Bedienung erforderlich machen. Zur Erfassung eines umfassenden Standortgradienten war es aber erforderlich, neun Hauben parallel einzusetzen, wobei drei von einer Person parallel bedient werden mussten.

Daher wurde ein neues Haubensystem für die Erfassung des NEE aller drei klimarelevanten Spurengase mit folgenden Eigenschaften entwickelt: Ein einfaches modulares Kühlsystem, bestehend aus einer Kombination von Ventilatoren und Kühlaggregaten, hält die Haubeninnentemperatur gegenüber außen auf max. 1°C Differenz konstant und begrenzt den Anstieg der Luftfeuchte. Tests ohne Kühlsystem ergaben bei voller Einstrahlung einen Temperaturanstieg von >3°C min^{-1} und einen steilen Anstieg der relativen Luftfeuchte auf >85% bereits nach ein bis zwei Minuten Einschlusszeit. Die in der Haube eingeschlossenen Pflanzen können bei laufender Klimatisierung daher weitgehend unbeeinträchtigt Photosynthese treiben. Aber auch die temperaturgesteuerten Flüsse der Bodenatmung und der Methanemission bleiben vom Messsystem unbeeinflusst, da sich die Bodentemperaturen nicht verändern. Für die Dichtung zum Bodenrahmen wurde Gummi-Profil verwendet, das frost- und UV-stabil ist und eine zuverlässige Abdichtung der Haube auch im Winter gewährleistet. Tests ergaben eine vollständige Gasdichtigkeit des Messsystems. Ein Druckausgleichsventil verhindert Veränderungen des Luftdruckes innerhalb gegenüber außerhalb der Haube. Testmessungen, die mit und ohne Kühlungssystem durchgeführt wurden, ergaben bis zu 30% Überschätzung der CO_2-Aufnahme mit einem ungekühlten System.

Zur Absicherung der Erfassung realistischer Gasflussraten durch die Haube wurden Vergleichstests mit der etablierten Eddy-Kovarianz-Methode durchgeführt. Trotz der unterschiedlichen räumlichen Auflösung der Messsysteme ergaben die Tests eine sehr hohe Übereinstimmung der Eddy-Kovarianz und Hauben-Flüsse. Bis zu 94% der Variabilität der Hauben-Flüsse konnten durch die Eddy-Kovarianz-Werte erklärt werden mit nur minimalen Abweichungen von der 1:1 Linie. Damit konnte die entwickelte Haube in ihrer Eignung als komplementäre Technik zur Eddy-Kovarianz bestätigt werden.

2. Standortgradienten

Der Spurengasaustausch von Mooren ist bisher vorwiegend in den borealen Breiten untersucht worden (z.B. Silvola, 1996a, Alm et al., 1997, Bellisario et al., 1998, Tuittila et al. 1999, Joabsson & Christensen, 2001, Christensen et al., 2003). In den gemäßigten Breiten des voralpinen Moorgürtels fehlten dagegen Untersuchungen zum Austausch klimarelevanter Spurengase und insbesondere zum NEE von Moorökosystemen gänzlich. Mit dieser Untersuchung wurde daher eine räumliche und thematische Lücke geschlossen. Andererseits entstand daraus der Anspruch, die zu beprobenden Ökosystemtypen möglichst repräsentativ für die vorhandenen Moorökosysteme des Alpenvorlandes auszuwählen und einen vollständigen Standortgradienten von naturnahen über renaturierte zu degradierten Standorten abzubilden. Nach der Entscheidung für die Kendlmühlfilze / Lkr. Rosenheim, als geeignetstem Testgebiet, wurden die zu beprobenden Ökosysteme auf der Basis von Vegetationskarten (Pfadenhauer et al., 1990), Luftbildern und Informationen zur Nutzungsgeschichte vorausgewählt. Durch intensive Geländebegehungen mit Interpretation der aktuellen Vegetationsdecke und Standortausstattung (Mikrotopographie) wurden die zu untersuchenden Ökosystemtypen ausgeschieden. Insgesamt wurden sechs naturnahe, vier degradierte und zwei renaturierte Standorte ausgewählt, wobei ein degradierter Standort während der Untersuchungen renaturiert wurde und damit eine zusätzliche Renaturierungsvariante untersucht werden konnte. Die untersuchten Standorte sind in der Tabelle 7.1 aufgeführt. In einem umfassenden, die gashaushaltlichen Messungen begleitenden Untersuchungsprogramm wurden Vegetations- und Standortparameter für die einzelnen Plots erhoben. Diese Parameter dienten der Erklärung der ermittelten Spurengasflüsse und –bilanzen sowie der post-hoc Analyse des zugrundeliegenden Vegetations- und Standortgradienten.

Versuchsstandort	Plot-Nr.	Status	Kurzbeschreibung
D1	4, 5, 6	degradiert	Trockene Regenmoorheide (*Calluna vulgaris* (L.) dominiert), ehemaliger Torfstich, vor ca. 50 Jahren aufgelassen, aufgrund der topographischen Situation keine Vernässung
D2	16, 17, 18	degradiert	Trockene Regenmoorheide (*Calluna vulgaris* (L.) dominiert), ehemaliger Torfstich, vor ca. 50 Jahren aufgelassen, aufgrund der topographischen Situation keine Vernässung
D3	13, 14, 15	degradiert	Trockene Regenmoorheide (*Calluna vulgaris* (L.) dominiert), vor ca. 50 Jahren vorentwässert aber nicht torfgestochen, keine Vernässung
D4	1, 2, 3	degradiert	Trockene Regenmoorheide (*Calluna vulgaris* (L.) dominiert, mit *Pinus sylvestris* (L.) and *Betula pubescens* (Ehrh.), ca. 50 Jahre vor der Untersuchung vorentwässert, aber nicht torfgestochen, keine Vernässung
R1	16, 17, 18	renaturiert	Überstaute ehemalige trockene Regenmoorheide (ursprünglich Standort D2); die Überstauung wurde in der Mitte des Messjahres am 17. Juli durchgeführt
R2	10, 11, 12	renaturiert	Sekundäre feuchte Regenmoorheide (*Calluna vulgaris* (L.) dominiert) auf ehemaliger vorentwässerter Hochmoorheide, ca. 10 Jahre vor der Untersuchung angestaut bis unter die Geländeoberfläche
R3	7, 8, 9	renaturiert	Sekundärer bunter Torfmoosrasen (*Sphagnum* (L.) dominiert), auf ehemaligen torfgestochenen trockenen Hochmoorheiden, ca. 10 Jahre vor der Untersuchung angestaut bis zur Geländeoberfläche
N1	19, 20, 21, 22, 23, 24	naturnah	Feuchte Regenmoorheide (*Calluna vulgaris* (L.) dominiert), aufgrund der Position auf geneigtem Randstandort leichte natürliche Entwässerung
N2	25, 26, 27	naturnah	Latschengebüsch (*Pinus mugo* (Turra), aufgrund der Position auf geneigtem Randstandort und der Mikrotopographie leichte natürliche Entwässerung
N3	31, 32, 34	naturnah	Bunter Torfmoosrasen (*Sphagnum* (L.) dominiert)
N4	33, 35, 36	naturnah	Regenmoorbulte (*Sphagnum* (L.) – *Eriophorum vaginatum* (L.))
N5	37, 38, 39	naturnah	Übergangsvegetation zwischen Bulten und Torfmoosrasen
N6	40, 41, 42	naturnah	Schlenke (*Sphagnum* (L.) – *Scheuchzeria palustris* (L.) dominiert)

Tab. 7.1: Kurzcharakteristik der Untersuchungsflächen. Kendlmühlfilze, Lkr. Rosenheim.

Neben den kontinuierlich aufgezeichneten Klimaparametern (Luft und Bodentemperaturen, photosynthetisch aktive Strahlung) wurden als Standortparameter Wasserstand, pH-Wert und elektrische Leitfähigkeit wöchentlich parallel zu den Spurengasmessungen erhoben. Kalzium- und Kaliumgehalte des Porenwassers wurden dreimal im Verlauf des Messjahres erfasst. Vegetationsdeckung (nach Londo, 1976), Anzahl aerenchymhaltiger Blätter, Blattflächenindex, C/N-Verhältnis der Torfschichten und Zersetzungsgrad (H-Wert nach v. Post in Grosse-Braukmann, 1990) des Torfes wurden einmalig erfasst.

Als erster Schritt der post-hoc Analyse der Standortunterschiede wurde eine Korrespondenzanalyse (DCA) der Vegetationsausstattung der Plots durchgeführt. Die Ordination führte zu einer weitgehenden Bestätigung der durch die Geländebegehungen ausgewählten Standorte entlang eines Degradierungsgradienten. Die für das Untersuchungsgebiet in einer vorangegangenen Studie als Zeiger unterschiedlicher Standortbedingungen ausgeschiedenen Pflanzenarten (Pfadenhauer et al. 1990) differenzierten entsprechend der zugewiesenen indikatorischen Bedeutung innerhalb des Ordinationsdiagramms. Die Verteilung entsprach den Ergebnissen von Bragazza & Gerdol (2002) für vergleichbare Moorökosysteme in den Südostalpen. Durch eine vergleichende Clusteranalyse der Vegetation einerseits und der Standortfaktoren andererseits konnte der Einfluss der unterschiedlichen Ausprägung der Standortfaktoren auf die Standortunterschiede entlang des Degradierungsgradienten identifiziert werden. Mittels einer kanonischen Korrespondenzanalyse (CCA) mit der Vegetation als primärer Matrix und den Standortparametern als sekundärer Matrix wurde schließlich die Standortgliederung im Zusammenspiel von Vegetation und Standortfaktoren umfassend dargestellt. Die degradierten Standorte differenzierten deutlich von den naturnahen Standorten. Die renaturierten Standorte nahmen eine Mittelstellung ein und überlappten z.T. bereits mit den naturnahen Standorten der feuchten Regenmoorheide (N1) und des Latschengebüschs (N2). Diese mittels multivariater Methoden identifizierten Standortunterschiede bestätigten die Gliederung und Auswahl der Ökosystemtypen entlang des intendierten Gradienten von naturnahen über renaturierte zu degradierten Standorten.

3. Nettoökosystemaustausch (NEE) von Kohlendioxid

Die Rolle der Ökosysteme als Senke für Kohlendioxid ist seit der Formulierung des Kyoto-Protokolls zur Klimarahmenkonvention im Blickfeld des Klimaschutzes und wird nach wie vor kontrovers diskutiert (z.B. Noble & Scholes, 2001, Schulze et al., 2002). Zur Versachlichung der Diskussion und zur Schaffung von Entscheidungsgrundlagen für die Klimaschutzpolitik, sowie für die nationale Berichterstattung für die Klimarahmenkonvention sind aber dringend konkrete Messwerte und Modellierungen des CO_2-Austausches erforderlich. Der für die Berichterstattung relevante Wert ist der Nettoökosystemaustausch (NEE) bzw. die Nettoökosystemproduktivität (NEP), als Integral des NEE über längere Zeiträume (wie z.B. ein Jahr). Das NEE ist die Differenz zwischen der Bruttoprimärproduktivität (GPP) und der Ökosystem-Atmung (R_{ECO}) und wird als negativer Wert angegeben, wenn das Ökosystem Kohlenstoff aufnimmt (Buchmann 1999). R_{ECO} umfasst die autotrophe (Pflanzen) und heterotrophe (Bodenmikroorganismen) Atmung.

Manuelle Hauben können nicht wie die Eddy-Kovarianz kontinuierlich eingesetzt werden, um den Nettoökosystemaustausch in Form eines Monitoring zu erfassen. Daher ist es erforderlich, in einem kampagnenorientierten Ansatz die Abhängigkeiten des NEE und des R_{ECO} von sich ändernden Steuerfaktoren (vorwiegend Strahlung und Luft- und Bodentemperatur sowie Wasserstand) zu erfassen (s. z.B. Bellisario et al., 1998). Diese Abhängigkeiten dienen der Parametrisierung von Modellen, die zur Berechnung des NEE für den Jahreslauf eingesetzt werden (Whiting, 1994, Bubier et al., 1998, Tuittila et al., 1999).

Hinsichtlich der Abhängigkeit der Ökosystematmung (R_{ECO}) von der Temperatur als Steuergröße ließ sich aber für den Datensatz der Kendlmühlfilze kein jahreszeitlicher Effekt nachweisen. Die Parametrisierungen für das Respirationsmodell konnten daher mit dem jeweiligen Gesamtdatensatz pro Standort durchgeführt werden. R_{ECO} wurde nach Lloyd & Taylor (1994) aus der Temperatur modelliert, wobei die Aktivierungsenergie (Eo) und die Referenzatmung bei 10 °C (Rref) als Parameter ermittelt wurden. Die Variabilität der Ökosystematmung konnte über alle Standorte auf signifikantem Niveau ($p<0.01$) zwischen 43 % und 91 % durch die Bodentemperatur erklärt werden.

Für die Parametrisierung der Steuerfaktoren für NEE wurden Lichtabhängigkeitskurven ermittelt und mit einer rectangulär-hyperbolischen Funktion beschrieben (Michaelis & Menten, 1913). Im Gegensatz zur Systematmung wurden die Datensätze je Standort in fünf Abschnitte aufgeteilt (vier 2-Monatsgruppen und eine 4-Monatsgruppe über den Winter), die der jahreszeitlichen Entwicklung der Abhängigkeiten entsprachen. Für jede Periode wurden die Parameter Lichtnutzungseffizienz (a; Steigung der Lichtkurve bei Strahlung 0), die Systematmung R_{ECO} (bei Strahlung 0) und der maximale Fluss (GPmax bei Licht unendlich, bzw. bei Steigung 0 der Lichtkurven) ermittelt. Außerhalb der Winterperiode konnte die Variabilität CO_2-C-Flüsse von 68 bis 95 % mittels der Lichtkurven erklärt werden. Für die Modellierung der CO_2-Flüsse übers Jahr wurden die Parameterwerte der Lichtkurven zwischen den Perioden interpoliert. Mit diesen für jeden Standort individuellen Parameterwerten wurde dann das in 0,5-Stundenschritten laufende Modell betrieben, mit dem der Netto-Ökosystemaustausch des CO_2 über das Jahr modelliert wurde.

Hinsichtlich der modellierten Tagesbilanzen differenzieren die Standorte entlang des Degradierungsgradienten sowohl in der Höhe der Flüsse als auch der Dynamik über das Jahr. Die trockene Regenmoorheide (D1, ehemaliger Torfstich) erreicht an keinem Tag des modellierten Jahresverlaufs negative NEE Werte, d.h. der Standort weist ausschließlich Kohlenstoff-Verluste mit bis zu 3,5 g CO_2-C $m^{-2}d^{-1}$ auf. Beim renaturierten Torfmoosrasen (R3) überwiegen im Frühjahr die Tage mit negativen NEE Werten mit bis zu -1,5 g CO_2-C $m^{-2}d^{-1}$ (Aufnahme ins System), aber ab Mitte des Jahres dominieren die Verlusttage mit bis zu 1,8 g CO_2-C $m^{-2}d^{-1}$. Die naturnahe Schlenke (N6) zeigt außerhalb des Winters fast ausschließlich Aufnahmen ins System mit maximalen Tagesbilanzen von –2,4 g CO_2-C $m^{-2}d^{-1}$. Für die Gesamtjahresbilanzierung wurde der auf 0,5 h Basis modellierte NEE Verlauf pro Standort integriert. Die ermittelten CO_2-C-Bilanzen (NEP) differenzieren entlang des Degradierungsgradienten (s. Tab. 7.2) mit den höchsten Emissionen aus den degradierten ehemaligen Torfstichen, erheblich reduzierten Emissionen aus den renaturierten Standorten und schließlich Aufnahmen ins System auf vier von sechs naturnahen Standorten (s. Tab. 7.2). Die beiden naturnahen Standorte, die geringe CO_2-C-Verluste übers Jahr aufweisen, sind die aufgrund ihrer Position leicht entwässerten Standorte N1 und N2 (feuchte Hochmoorheide und Latschengebüsch). Der Mittelwert für die naturnahen Standorte liegt mit -71 ± 40,5 g CO_2-C $m^{-2}a^{-1}$ in der Größenordnung von Untersuchungen aus Kanada auf ombrothrophen Mooren (Lafleur et al. 2001). Damit ist der Nachweis erbracht, dass die untersuchten naturnahen Moor-Ökosysteme am Alpenrand effiziente Senken für Kohlendioxid darstellen. Die degradierten Standorte wirken dagegen mit durchschnittlichen Emissionen von 318,5 ± 49,8 g CO_2-C $m^{-2}a^{-1}$ als erhebliche Quellen für Kohlendioxid (mehr als das vierfache an CO_2-Emissionen im Vergleich zur mittleren Bindung auf den naturnahen Standorten). Durch die Renaturierung lässt sich

Versuchs-standort	CO_2-C [g m^{-2} a^{-1}]
D1	443 ± 70
D2	360 ± 25
D3	246 ± 46
D4	225 ± 58
R1	192 ± 21
R2	127 ± 56
R3	62 ± 65
N1	14 ± 52
N2	4 ± 42
N3	-54 ± 31
N4	-87 ± 33
N5	-152 ± 48
N6	-157 ± 37

Tab 7.2: Jahresbilanzen des netto CO_2-C-Austauschs (NEP). SF in Klammern.

die CO_2-Emission auf ca. ein Drittel der degradierten Standorte reduzieren (127 ± 56 g CO_2-C $m^{-2}a^{-1}$). Absolute CO_2-Aufnahme findet aber auf Renaturierungsflächen noch nicht statt.

Der Netto-Ökosystemaustausch ist ein Ausdruck der Summe der CO_2-C-Flüsse des Pflanze-Bodensystems im betrachteten Ökosystemausschnitt. Inwieweit die jeweiligen Jahresbilanzen mit Standort- und Vegetationsparametern der untersuchten Standorte erklärt werden können, wurde daher mittels Regressionsanalysen getestet. Der mittlere Blattflächenindex pro Standort konnte 87 % der Variabilität des NEP erklären, mit dem mittleren Wasserstand wurden 84% und mit der mittleren elektrischen Leitfähigkeit sogar 91% der Variabilität des NEP erklärt. Eine multiple lineare Regression mit mittlerem Wasserstand und elektrischer Leitfähigkeit als unabhängigen Variablen konnte schließlich 95% der Variabilität der Jahresbilanzen erklären. Damit zeigt sich einerseits, dass der Austausch an CO_2-C und die NEP-Bilanzen sensibel auf Standorteigenschaften und Vegetationsausstattung reagieren und andererseits einfach zu erfassende Parameter (z.B. Wasserstand, elektrische Leitfähigkeit, Blattflächenindex) geeignet sind, um die Jahresbilanzen zu erklären.

4. Methan- und Lachgasflüsse

Die Voraussetzung für die Einschätzung der Klimawirksamkeit eines Ökosystems ist die Erfassung aller drei biogenen klimarelevanten Spurengase. Daher wurde der Austausch von Methan und Lachgas parallel zu Kohlendioxid auf den Messplots im wöchentlichen Standardprogramm erfasst. Methan wird durch methanogene Bakterien vorwiegend aus Acetat im wassergesättigten, anaeroben Teil eines Torfkörpers gebildet (Lloyd et al. 1998). Die Methanproduktion im Torf ist generell höher als die Methanemission an der Vegetationsoberfläche, da ein Teil des gebildeten Methans bei der Bodenpassage durch die aerobe oberste Torfschicht (sofern vorhanden) zu CO_2 aufoxidiert wird (Sundh et al. 1995, Segers 1998).

Entscheidend für die Wirkung eines Moor-Standortes als Quelle oder Senke für Methan ist der mittels der Haubentechnik messbare Methan-Austausch mit der Atmosphäre, als Differenz der Produktions- und Oxidationsprozesse. Die untersuchten Moorstandorte unterscheiden sich hinsichtlich ihrer Methanflüsse entlang des Degradierungsgradienten deutlich. Die höchsten Flüsse wurden auf den naturnahen Standorten und hier insbesondere in der Schlenke (N6) gemessen (4,63 mg CH_4-C $m^{-2}h^{-1}$). Die geringsten Flüsse weist mit 0,03 mg CH_4-C $m^{-2}h^{-1}$ die trockene Regenmoorheide (D1, ehemaliger Torfstich) auf. Die renaturierten Standorte liegen mit 0,21 – 0,76 mg CH_4-C $m^{-2}h^{-1}$ zwischen diesen Extremen. Mit bis zu 89 % konnte die Variabilität dieser Flüsse auf den naturnahen Standorten N4-N6 durch multiple lineare Regression mit dem NEE und der Bodentemperatur in 50 cm Tiefe erklärt werden. Auf allen anderen Flächen ergab eine einfache Regression mit der Bodentemperatur die besten Erklärungsgrade, wobei einzig auf dem degradierten Standort (D3, trockene Regenmoorheide) durch die Einbeziehung des Wasserstandes eine Verbesserung der Regression erreicht werden konnte. Generell waren die Regressionen mit der Temperatur auf den degradierten und renaturierten Standorten gering ausprägt (r^2 0,18 - 0,42). Die sehr geringen Flüsse auf den beiden ehemaligen Torfstichen (D1, D2) konnten durch keinen Steuerparameter auf signifikantem Niveau erklärt werden.

Zur Bilanzierung der Methanflüsse übers Jahr wurde ein Verfahren der blockweisen Extrapolation eingesetzt (vgl. Kamp 1998, Ruser 1999), da sich kein Modell mit einheitlichem Parametersatz auf alle Standorte anwenden ließ. Kein Standort wirkte als Senke für Methan. Die geringsten Jahresemissionen mit 0,03 g CH_4-C $m^{-2}a^{-1}$ wurden auf der trockenen Regenmoorheide (D1, ehemaliger Torfstich) nachgewiesen. Die höchsten Jahresemissionen wurden auf den naturnahen Standorten erfasst, mit bis zu 38,2 ± 2,2 g CH_4-C $m^{-2}a^{-1}$ (Standort N6). Mit 7,1 ± 3,1 g CH_4-C $m^{-2}a^{-1}$ wurden aber auf der renaturierten Regenmoorheide (R2) Methan-Emissionen erreicht, die noch über dem Wert des naturnahen Standortes (Latschengebüsch; N2) mit den geringsten Emissionen dieser Gruppe lag (5,4 ± 0,5 g CH_4-C $m^{-2}a^{-1}$). In der Tendenz nahmen die Methan-Emissionen exponentiell von den nahezu neutralen degradierten Flächen über die renaturierten zu den naturnahen zu (s. Tab. 7.3).

Zur Erklärung der Bilanzen wurden Regressionen mit den funktional bedeutsamen Steuerfaktoren Wasserstand, Anzahl arenchymhaltiger Halme und der Jahresbilanzen des CO_2-C-Austausches (NEP) gerechnet. Die funktionelle Verknüpfung zwischen CO_2-Austausch und Methanbildung konnte bereits für die Flüsse nachgewiesen werden (s.o.), allerdings nur auf den naturnahen Standorten N4-N6. Mit dem Wasserstand alleine konnte 54 % der Variabilität der Methanbilanzen der Standorte erklärt werden, wohingegen die Anzahl der aerenchymhaltigen Halme bereits 82 % erklärte. Eine multiple lineare Regression mit Wasserstand und Anzahl aerenchymhaltiger Halme verbesserte die Regression auf einen Erklärungsgrad der Variabilität der Methanflüsse von 85 %. Der höchste Erklärungsgrad (87 %) konnte schließlich durch eine exponentielle Regression mit der CO_2-C-Bilanz (NEP) erreicht werden. Dabei waren die Methan- und Kohlendioxid-Bilanzen gegenläufig miteinander verknüpft: Eine hohe Kohlendioxid-Aufnahme, wie auf den naturnahen Standorten ging mit hohen Methan-Emissionen einher, während die degradierten trockenen Hochmoorheiden als größte Kohlendioxid Quellen sich hinsichtlich der Methanemissionen nahezu neutral verhielten.

Versuchs-standort	CH_4-C [g m^{-2} a^{-1}]	N_2O-N [mg m^{-2} a^{-1}]
D1	0.03 ± 0.02	168.57 ± 93.70
D2	0.08 ± 0.03	50.05 ± 46.84
D3	2.0 ± 0.3	9.37 ± 4.89
D4	0.9 ± 0.3	4.01 ± 1.12
R1	1.5 ± 0.2	1.77 ± 4.34
R2	7.1 ± 3.1	-1.21 ± 2.42
R3	2.3 ± 0.4	1.35 ± 1.44
N1	10.7 ± 1.6	-0.81 ± 2.49
N2	5.4 ± 0.5	-1.91 ± 1.62
N3	10.1 ± 2.3	-2.29 ± 1.82
N4	27.7 ± 2.1	-2.24 ± 1.13
N5	24.1 ± 1.5	-0.49 ± 1.66
N6	38.2 ± 2.2	-1.83 ± 1.05

Tab. 7.3 Methan- und Lachgasbilanzen. ± SF

Die zwei wichtigsten Prozesse für die Lachgasproduktion sind die autotrophe Nitrifikation von Ammonium zu Nitrat und die heterotrophe Denitrifikation von Nitrat zu Lachgas (Davidson, 1991). Bedeutende Steuerfaktoren für die Lachgasbildung sind die Ammonium- und Nitratkonzentration, die Bodentemperatur und die Bodenfeuchte (Granli & Bockmann, 1994). In verschiedenen Studien konnten die Lachgas-Emissionen aber nur mit einem (Ruser 1999) oder gar keinem (Glatzel & Stahr 2001) der Steuerfaktoren korreliert werden.

Einzig auf der degradierten, abgetorften Regenmoorheide (Standort D1), die die höchsten Flüsse aller Standorte aufwies, konnte ein saisonaler Trend über das Jahr identifiziert werden, der aber nicht signifikant mit der Temperatur zu erklären war. Die Lachgas-Emissionen waren generell sehr gering, entsprechend der Ergebnisse von Martikainen (1996) für nährstoffarme Ökosysteme. Die mittleren Flüssen lagen auf allen Standorten unter 0,75 µg N_2O-N $m^{-2}h^{-1}$, mit Ausnahme der degradierten Standorte D2 (6,50 µg N_2O-N $m^{-2}h^{-1}$) und D1 (19,71 µg N_2O-N $m^{-2}h^{-1}$), die höhere Flüsse aufwiesen. Die Lachgasflüsse waren auf keinem Standort auf signifikantem Niveau mit Steuerfaktoren erklärbar.

Die Jahresbilanzen (blockweise Extrapolation) erreichten nur auf den Standorten D1 (168 mg N_2O-N m^{-2} a^{-1}) und D2 (50 mg N_2O-N m^{-2} a^{-1}) nennenswerte Emissionen (s. Tab. 7.3). Diese Werte sind in der Größenordnung der Ergebnisse von Martikainen et al. (1993). Durch die Renaturierung konnten die N_2O-Emissionen auf minimale Niveaus gesenkt werden (s. R1-R3, Tab.7.3). Die stärkste Korrelation zwischen N_2O-Bilanzen und Standortfaktoren wurden mit der elektr. Leitfähigkeit erreicht (Spearman R=0.94), was die Bedeutung der Nährstoffversorgung des Standortes für die N_2O Bildung widerspiegelt. Die Methanbilanzen waren negativ mit den Lachgasbilanzen korreliert (Spearman R=-0.80).

5. Kohlenstoffbilanz und Klimarelevanz der Moorstandorte

Die Kohlenstoffbilanz eines Moores ist die Differenz von CO_2-C-Bilanz, CH_4-C-Bilanz und Verlust von flüssigem organischen und anorganischen Kohlenstoffs (DOC/DIC). Als klassische Methode zur Ermittlung der C-Bilanz werden die langfristigen Akkumulationsraten (LORCA) mittels Profildatierung und Bestimmung des Kohlenstoffgehaltes ermittelt. Turunen & Tolonen (1996) geben auf der Basis von mehreren tausend Profilbohrungen borealer Regenmoore 24 g C m^{-2} a^{-1} als langfristigen Akkumulationswert an. Die aktuelle Kohlenstoffbilanz wird aber über die LORCA wegen der geringen zeitlichen Auflösung der Profilschichten und der Mittelung der Raten über das gesamte Profil nur unzureichend erfasst. Daher sind alternativ gashaushaltliche Untersuchungen geeignet, die aktuelle Kohlenstoffbilanz aus der Differenz der Gasflüsse zu bestimmen, sofern der NEE zuverlässig erfasst werden kann (Byrne et al. 2004). Aus der Differenz der im zugrundeliegenden Messprogramm gewonnenen Bilanzen für CO_2-C und CH_4-C, konnte daher die aktuelle Kohlenstoffbilanz der untersuchten Moorstandorte abgeschätzt werden. Der flüssige Export über DOC/DIC wurde aus Literaturangaben (Sallantus & Kaipainen, 1996; Waddington & Roulet 1996; Fraser et al. 2001) auf 7 g C m^{-2} a^{-1} geschätzt, da die komplexe Hydrologie der Standorte eine direkte Bestimmung nicht möglich machte.

Versuchs-standort	C – balance [g m^{-2} a^{-1}]	GWP 100 CO_2-C eq. [g m^{-2} a^{-1}]	GWP 500 CO_2-C eq. [g m^{-2} a^{-1}]
D1	450.1 ± 70.0	465.6 ± 71.1	455.4 ± 70.3
D2	367.1 ± 25.0	367.3 ± 25.8	363.9 ± 25.2
D3	255.0 ± 46.0	262.4 ± 46.1	251.5 ± 46.0
D4	232.9 ± 58.0	232.4 ± 58.0	227.5 ± 58.0
R1	200.5 ± 21.0	203.6 ± 21.0	195.7 ± 21.0
R2	141.1 ± 56.1	180.8 ± 60.8	143.9 ± 56.5
R3	71.3 ± 65.0	79.7 ± 65.1	67.6 ± 65.0
N1	31.7 ± 52.0	95.2 ± 53.4	39.6 ± 52.1
N2	16.4 ± 42.0	44.8 ± 42.2	16.8 ± 42.0
N3	-36.9 ± 31.1	22.5 ± 35.6	-29.9 ± 31.5
N4	-52.3 ± 33.1	123.2 ± 36.7	-20.7 ± 33.4
N5	-120.9 ± 48.0	31.1 ± 49.3	-94.2 ± 48.1
N6	-111.8 ± 37.1	133.1 ± 40.6	-65.5 ± 37.4
D1-D4	326.3 ± 49.8	331.9 ± 50.2	324.5 ± 49.9
D1-D2	408.6 ± 47.5	416.5 ± 48.4	409.6 ± 47.8
D3-D4	243.9 ± 52.0	247.4 ± 52.1	239.5 ± 52.0
R1-R3	137.6 ± 47.4	154.7 ± 48.9	135.8 ± 47.5
N1-N6	-45.6 ± 40.5	75.0 ± 43.0	-25.6 ± 40.8
N1-N2	24.1 ± 47.0	70.0 ± 47.8	28.2 ± 47.1
N3-N6	-80.5 ± 37.3	77.5 ± 40.5	-52.6 ± 37.6

Tab. 7.4: Kohlenstoffbilanz und Klimarelevanz (GWP für 100 und 500 Jahreszeitraum). ± SF. D1-D4: Mittelwert für degradierte Standorte. D1-D2: Mittelwert für ehemalige Torstiche,.D3-D4: Mittelwert vorentwässert Hochmoorheiden, R1-R3: Mittelwert für renaturierte Standorte. N1-N6: Mittelwert über alle naturnahen Standorte. N1-N2: Mittelwert für naturnahe Standorte mit leichter natürlicher Entwässerung. N3-N6: Mittelwert für naturnahe Kern-Standorte.

Die degradierten Standorte (D1-D4) weisen mit 326,3 ± 49,8 g C m^{-2}a^{-1} erhebliche mittlere Kohlenstoffverluste auf (s. Tab 7.4). Die renaturierten Standorte (R1-R3) haben bereits deutlich geringere Verluste (137,6 ± 47,4 g C m^{-2}a^{-1}), stellen aber noch keine C-Senken dar. Die naturnahen Standorte (N1-N6) weisen dagegen im Mittel –45,6 ± 40,5 g C m^{-2}a^{-1} Netto-Kohlenstoffaufnahmen ins System auf (s. Tab 7.4). Gruppiert man die Standorte differenzierter hinsichtlich ihrer Ähnlichkeit in Standorteigenschaften und Vegetationsausstattung (s. Kap. 3), dann heben sich die ehemaligen Torfstichflächen (D1-D2) mit einem Kohlenstoffverlust von 408,6 ± 47,5 g C m^{-2}a^{-1} deutlich von den vorentwässerten Standorten (D3-D4) mit 243,9 ± 52 g C m^{-2}a^{-1} ab. Ebenso ist ein steiler Gradient der C-Bilanzen innerhalb der naturnahen Standorte feststellbar (s. Tab 7.4). Die aufgrund ihrer Position am Rande des Regenmoorkerns natürlicherweise leicht entwässerten Standorte der feuchten Hochmoorheide und des Latschengebüschs (N1-N2) sind im Mittel leichte Verluststandorte (24,1 ± 47 g C m^{-2}a^{-1}), während die naturnahen Flächen im Kern der Untersuchungsgebietes erhebliche (N3-N6) mittlere Aufnahmen zeigen (-80,5 ± 37,3 g C m^{-2}a^{-1}). Dies ist in der Größenordnung von Werten aus borealen kanadischen Regenmooren (-60 g C m^{-2}a^{-1}, Moore et al. 2002). Diese Werte liegen ca. dreimal höher als die für Regenmoore angegebenen und aus der Profilmethode bestimmten langfristigen Akkumulationsraten (24 g C m^{-2}a^{-1}; Turunen & Tolonen, 1996). Unter der Einschränkung der interannuellen Variabilität, die mit den gashaushaltlichen Untersuchungen nur über langjährige Messungen am gleichen Standort ausgemittelt werden könnte, können diese erhöhten Akkumulationsraten als Hinweis auf rezent verbesserte Wachstumsbedingungen der Moore interpretiert werden. Gründe hierfür können sowohl verlängerte Vegetationsperioden bei wärmerem und feuchterem Klima (wie von Klimamodellierungen für den Alpenrand prognostiziert) als auch erhöhte atmosphärische Stickstoffdüngung sein.

Für die Bewertung Klimarelevanz der Moorstandorte ist die Bilanzierung des CO_2-Austausches oder die Ermittlung der Kohlenstoffbilanz nicht ausreichend. Vielmehr müssen alle drei biogenen klimarelevanten Spurengase (Kohlendioxid, Methan und Lachgas) bilanziert und mit ihrem jeweiligen individuellen globalen Erwärmungspotenzial (GWP; IPCC 1996) verrechnet werden. Klassischerweise wird das GWP für den Zeitraum von 100 Jahren angegeben (Berechungsbasis für die UNFCCC Berichterstattung), wobei gerade für langfristig funktionierende Ökosysteme wie die Moore auch der Berechnungszeitraum von 500 Jahren zum Vergleich herangezogen wird. CH_4-C hat ein 7,6- (100a) bzw. 2,4-mal (500a) höheres GWP als CO_2-C. N_2O-N hat ein 133- (100a) bzw. 73-mal (500a) höheres GWP als CO_2-C. Nach Verrechnung der gasbezogenen Bilanzen mit dem jeweiligen GWP kann die gesamte Klimarelevanz des jeweiligen Ökosystems ermittelt und in CO_2-C-Äquivalenten (CO_2-C eq.) ausgedrückt werden (s. Tab. 7.4).

Die geringen Unterschiede zwischen den Kohlenstoffbilanzen und der Klimarelevanz in den degradierten Standorten D1-D4 (s. Tab 7.4) sind ein Ausdruck dessen, dass die Bedeutung der CO_2-Emission die der anderen beiden Gase auf diesen Standorten weit übersteigt. Einzig auf dem Standort D1 ist für den Berechnungszeitraum von 100 Jahren mit 15 g CO_2-C eq. $m^{-2}a^{-1}$ ein sichtbarer Effekt der erhöhten Lachgas-Emissionen feststellbar. Auf den renaturierten Standorten R1-R3 spielt die Lachgasemission für die Klimarelevanz keine Rolle mehr. Dagegen sind hier die ansteigenden Methan-Emissionen für die Unterschiede zwischen der C-Bilanz und der GWP-(100)-Bilanz mit bis zu 40 g CO_2-C eq. $m^{-2}a^{-1}$ (R2) verantwortlich (s. Tab. 7.4). Die Bedeutung der Methan-Emissionen für die Klimarelevanz nimmt bei den naturnahen Standorten noch weiter zu: Der Beitrag des Methan zur Klimarelevanz liegt zwischen 41 g CO_2-C eq. $m^{-2}a^{-1}$ (N2) und bis zu 290 g CO_2-C eq. $m^{-2}a^{-1}$ (N6). Dieser erhebliche Beitrag bewirkt aber, dass die naturnahen Standorte N3 bis N6, obwohl sie als Kohlenstoffsenken wirken (s. Tab. 7.4), im Bemessungszeitraum von 100 Jahren zum Klimawandel beitragen. Diese Ergebnisse zeigen, dass die Einschätzung der Klimarelevanz von Mooren unter alleiniger Betrachtung von CO_2-C-Flüssen oder C-Bilanzen zu erheblichen Fehlinterpretationen führen kann.

Der Vergleich der Klimarelevanz der Standorte zwischen dem GWP-100 und dem GWP-500 zeigt die abnehmende Bedeutung der Methan- und Lachgasemissionen für den längeren Berechnungszeitraum. So wirken die naturnahen Standorte N3-N6 nun auch in Bezug zur Klimarelevanz als Entlastungsstandorte, mit maximalen Aufnahmen von –94,2 ± 48,1 g CO_2-C eq. $m^{-2}a^{-1}$ (s. Tab 7.4; Standort N5).

Renaturierte Standorte wirken, wie auch hinsichtlich der C-Bilanz, nicht unmittelbar als Senken für CO_2-C-Äquivalente (s. Tab. 7.4). Die Bewertung, ob Renaturierungsmaßnahmen zur Klimaentlastung beitragen können, ist aber nur im Vergleich zur Klimawirksamkeit der ursprünglich auf diesen Standorten vorhandenen degradierten Situation durchführbar. Den renaturierten Standorten R1 und R3 gingen degradierte trockene Regenmoorheiden voraus, auf denen Torf gestochen wurde. Dieser Standorttyp wird von den aktuell vorhandenen Standorten D1 und D2 repräsentiert. Die Situation vor der Renaturierung von Standort R2 entspricht den Standorte D3 und D4 (vorentwässerte trockene Regenmoorheide). Für die Abschätzung des Klimaentlastungseffektes wurde die in CO_2-C eq. ausgedrückte Klimarelevanz der renaturierten Standorte von dem Mittelwert der entsprechenden degradierten Vorläuferstandorte abgezogen. Damit ergab sich ein Entlastungseffekt von 336,8 ± 54 g CO_2-C eq. $m^{-2}a^{-1}$ bei der Renaturierung von ehemaligen Torfstichen und von 66,6 ± 55 g CO_2-C eq. $m^{-2}a^{-1}$ bei der Renaturierung von vorentwässerten Regenmoorheiden. Für die aktuelle Berichterstattung zur Klimarahmenkonvention ist Methan aber aufgrund der ungenügenden Datenlage und damit einhergehender großer Unsicherheiten noch nicht aufgenommen. Wird das Entlastungspotenzial ohne Einbeziehung von Methan ermittelt, führt die Renaturierung von Torfstichen (D1-D2) zu einer Einsparung von 354 g CO_2-C eq. $m^{-2}a^{-1}$ und von vorentwässerten Regenmoorheiden (D3-D4) zu 110 g CO_2-C eq. $m^{-2}a^{-1}$. Für die getesteten Renaturierungsmaßnahmen konnte damit eine klimaentlastende Wirkung nachgewiesen werden.

Ob Renaturierungsmaßnahmen von Regenmooren einen relevanten Beitrag zum Klimaschutz leisten können, hängt aber einerseits von der Emissions-Bezugsgröße und andererseits von den zur Verfügung stehenden Flächen für eine Renaturierung ab. Bayern hat ein Klimaschutzprogramm mit dem Ziel aufgestellt, die Emission von 92 Mio. t CO_2 (1998) auf 80 Mio. t CO_2 (2010) zu reduzieren (Bayerische Staatskanzlei, 2003). Laut Fahl (2000) ist mit gängigen technischen Lösungen aber realistischerweise nur eine Reduktion auf 83 Mio. t erreichbar. Es verbleibt eine Lücke von 3 Mio. t CO_2, für deren Schließung der Beitrag durch Regenmoorrenaturierung abgeschätzt wurde. Regenmoorrenaturierung für den Klimaschutz kann nur in einer naturräumlichen Kulisse der Regenmoorverbreitung durchgeführt werden. Daher wird hier ein regionalisiertes Konzept des Einsatzes von biologischen Senken vorgeschlagen und hinsichtlich seiner Entlastungswirkung abgeschätzt. Als Emissions-Bezugsgröße diente der pro-Kopf-Anteil dieser Einsparungslücke (234 kg CO_2), multipliziert mit der Bevölkerung aller Landkreise am Alpenrand (ca. 1.23 Mio.), die den voralpinen Regenmoorgürtel umfassen. Nach aktualisierten Schätzungen (Drösler et al., in prep.) sind im gesamten voralpinen Moorgürtel ca. 10.000 ha ehemalige Torfstiche und ca. 35.000 ha vorentwässerte Regenmoorheiden vorhanden. Eine Renaturierung dieser Flächen würde zu einer Klimaentlastung von ca. 74.000 t CO_2-C eq. (Anrechnung von CO_2 und N_2O entsprechend IPCC 2003) bzw. ca. 57.000 t CO_2-C eq. (Anrechnung aller drei Gase) führen. Damit könnten 27 % - 36 % der notwendigen pro-Kopf Entlastung (Schließung der 3 Mio. t Lücke) in der voralpinen Moorregion durch Regenmoorrenaturierung übernommen werden. Würde Bayern darüber hinaus versuchen, die ehrgeizigeren Kyoto-Ziele einer 21 %-Reduktion der 1990er Emissionen anzuwenden, könnten immerhin noch 10-13 % pro-Kopf-Entlastung in der voralpinen Moorregion durch Regenmoorrenaturierung erreicht werden.

Bayern hat 2002 ein Moorentwicklungsprogramm (MEK) aufgelegt, in dem als ein übergeordnetes Ziel der Klimaschutz durch Moorschutz formuliert wurde. Für den Schutz und die Renaturierung von Mooren wurde eine Prioritätenliste der Moorgebiete Bayerns aufgestellt. Allerdings erscheinen die

Handlungsprioritäten vorwiegend an Artenschutzgesichtspunkten ausgerichtet, weniger am Klimaschutz. Wie in dieser Arbeit aufgezeigt, ist eine Entlastungswirkung für das Klima nur durch die großflächige Renaturierung von degradierten vorentwässerten Regenmoorheiden oder ehemaligen Torfstichen zu erreichen. Das MEK bietet die entscheidende Plattform für einer Operationalisierung des Konzeptes der biologischen Senken durch Moorrenaturierung. Dafür ist aber eine detaillierte Prüfung und Ergänzung der Prioritätslisten in Hinblick auf den Klimaschutz erforderlich. Das regionalisierte Entlastungspotenzial ist erheblich (wie oben aufgezeigt) und kann Ziele des bayerischen Klimaschutzprogramms sinnvoll unterstützen. Darüber hinaus bieten Renaturierungsmaßnahmen in Regenmooren Synergien zwischen Artenschutz und Klimaschutz. Der landwirtschaftliche Nutzungsanspruch auf vorentwässerte Regenmoorheiden bzw. ehemalige Torfstiche ist unter den gegebenen agrar- und energiepolitischen Rahmenbedingungen sehr gering, wodurch der Zugriff auf die Flächen erleichtert sein sollte.

6. Forschungsbedarf

Die vorgelegte Arbeit hat die Kohlenstoffbilanz und die Klimarelevanz von naturnahen, degradierten und renaturierten Regenmoostandorten am Alpenrand erfasst und das Klimaentlastungspotenzial durch Renaturierungsmaßnahmen abgeschätzt. Forschungsbedarf besteht auf der Basis dieser Untersuchungen in mehreren Ebenen:

- Vervollständigung der Untersuchungen zum Spurengasaustausch von weiteren Ökosystemtypen und Managementvarianten in Regenmooren (z.B. Erweiterung der Standortvarianten um vorentwässerte Hochmoorheiden mit unterschiedlicher Gehölzdichte und Renaturierungsvarianten) zur Verringerung der Unsicherheiten in den Abschätzungen aufgrund der räumlichen Variabilität.
- Umfassende Untersuchung der Grundwassermoore in einem vergleichbaren Ansatz wie in der vorgelegten Arbeit. Nach Byrne et al. (2004) sind auf gesamter EU-Ebene keine Daten zum Effekt von Grundwassermoor-Renaturierungen auf das NEE von CO_2 vorhanden.
- Einrichtung von Ankermessstellen, um in einem langfristigen Beobachtungsprogramm die Effekte von Klimaschwankungen zwischen aufeinanderfolgenden Jahren auf den Gasaustausch ermitteln zu können und damit die Parametrisierung der Austausch-Modelle zu verbessern und zur Verringerung der Unsicherheiten in den Abschätzungen aufgrund der zeitlichen Variabilität.
- Aktualisierung des Kenntnisstandes zu Flächenausdehnung und Status der verschiedenen Moorökosystemtypen. Hier liegen die größten Unsicherheiten in der Abschätzung der Gesamtwirkung der Maßnahmen.
- Modellbasierte Hochskalierung der Spurengasbilanzen als Pilotstudie zur Berichterstattung für das nationale Inventar, sowie Operationalisierung der Einbeziehung von Moor-Renaturierungsmaßnahmen in die Klimaschutzaktivitäten Bayerns bzw. der Bundesrepublik Deutschland.

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Zusammenfassung

Moore akkumulieren in natürlichem Zustand kontinuierlich organische Substanz und wirken als Kohlenstoffsenken. Weltweit sind in Mooren 20 % des gesamten Bodenkohlestoffs gespeichert, obwohl die Moore nur 3 % der Landfläche bedecken. Durch Entwässerung und Torfabbau werden diese Kohlenstoffspeicher abgebaut, und die Moore wirken als Kohlenstoffquellen. Untersuchungen zum Kohlenstoffhaushalt und zur Klimawirksamkeit wurden bisher vorwiegend in borealen und subarktischen Mooren durchgeführt. Daher wurde mit der vorliegenden Arbeit eine Lücke zur Kenntnis des Spurengashaushaltes von naturnahen, degradierten und renaturierten Regenmoorstandorten am Alpenrand geschlossen. Ziel war es, mittels gashaushaltlicher Messungen die Kohlenstoffbilanz und die Klimawirksamkeit verschiedener Moorstandorte zu erfassen und zu prüfen, inwieweit Renaturierungsmaßnahmen zum Klimaschutz beitragen können.

Da unter den Gegebenheiten der potenziellen Untersuchungsstandorte (insbesondere kleinteiliges Mosaik aus naturnahen, degradierten und renaturierten Standorten; fehlender Stromanschluss) und der Notwendigkeit, mehrere Untersuchungsflächen parallel zu beproben, keine etablierte Erfassungstechnik eingesetzt werden konnte, wurde ein eigenes Messsystem entwickelt (**Kap. 2**). Die wesentlichen Eigenschaften der Messhaube sind das transparente Haubenmaterial und die Klimatisierung mittels eines modularen Kühlungssystems, das ohne Stromanschluss funktioniert. Die Haubenluft (300 – 500 l Volumen je nach Haubenhöhe) kann selbst an Strahlungstagen auf ± 1°C gegenüber der Außenluft geregelt werden. Damit kann die Vegetation weitestgehend ungestört Photosynthese betreiben, und der Nettoökosystemaustausch von CO_2 ist differenziert in einem kleinteiligen Mosaik von Ökosystemtypen erfassbar. Vergleichstest mit der etablierten Eddy-Kovarianzmethode erbrachten weitestgehende Übereinstimmung bei den ermittelten CO_2-Flussraten ($r^2 = 0{,}94$, 1:1 plot). Parallel zu CO_2 erlaubt die Haube die Erfassung von Methan- und Lachgasflüssen als Voraussetzung für die Ermittlung der aktuellen Kohlenstoffbilanz sowie der Klimawirksamkeit der Ökosystemtypen.

Es wurden 12 Untersuchungsstandorte in der Kendlmühlfilze, Lkr. Traunstein, als repräsentative Beispielsflächen für die Ausprägung der Regenmoorökosysteme im voralpinen Moorgürtel ausgewählt. Die Standorte reichen von trockenen ehemaligen Torfstichen über vorentwässerte Regenmoorheiden, wiedervernässte Torfstiche und Regenmoorheiden bis zu einer naturnahen Reihe aus feuchten Regenmoorheiden, Latschengebüsch, *Sphagnen*-Rasen, *Eriophorum*-Bulten und *Scheuchzeria*-*Sphagnen*-Schlenken. Diese auf Vegetationsausstattung und Geländerelief beruhende Auswahl wurde in einem post-hoc Ansatz hinsichtlich der Differenzierung der Standorte überprüft (**Kap. 3**). Die Vegetationszusammensetzung und die im Messprogramms erhobenen Standortfaktoren wurden mittels multivariater Methoden auf Ähnlichkeiten untersucht. Eine Kanonische Korrespondenzanalyse (CCA) erbrachte eine klare Differenzierung der Standorte entlang des Degradierungsgradienten. Damit konnte die a-priori getroffene Standortauswahl bestätigt werden.

Auf diesen Standorten wurden im Standardmessprogramm über ein gesamtes Kalenderjahr hinweg wöchentlich Messungen des Austausches von CO_2, CH_4 und N_2O durchgeführt. Die erhobenen CO_2-Flüsse dienten der Parametrierung eines CO_2-Austauschmodells (**Kap. 4**). Als Ergebnis konnte der Nettoökosystemaustausch (NEE) von CO_2 in 0,5 Stundenschritten übers Jahr modelliert und zur Netto-Ökosystemproduktion (NEP) integriert werden (negativ, wenn Aufnahme ins System). So sind die ehemaligen trockenen Torfstiche mit mittleren Emissionen von 401,5 ± 47,5 g CO_2-C $m^{-2}a^{-1}$ erhebliche Quellen, während die naturnahen Standorte als Senken für CO_2 wirken (-71 ± 40,5 g CO_2-C $m^{-2}a^{-1}$). Die renaturierten Standorte liegen im Mittel bei 127 ± 47,3 g CO_2-C $m^{-2}a^{-1}$ und wirken damit noch nicht als Senken für Kohlendioxid, weisen aber nur ca. 30 % der Emissionen der degradierten Standorte auf. Das NEP differenziert entlang des Degradierungsgradienten. Die Bilanzen korrelieren signifikant mit der mittleren elektrischen Leitfähigkeit (r^2=0,91), dem Blattflächenindex (r^2=0,87) und dem mittleren Wasserstand (r^2=0,84).

Die Bilanzen der Methan und Lachgasflüsse (**Kap. 5**) differenzieren ebenfalls entlang des Standortgradienten. Die höchsten Jahresemissionen von Methan weist der naturnahe nasse Schlenkenstandort mit 38,2 ± 2,2 g CH_4-C $m^{-2}a^{-1}$ auf, während sich die trockenen ehemaligen Torfstiche hinsichtlich Methan neutral verhalten. Die renaturierten Standorte liegen mit 1,5 ± 0,2 – 7,1 ± 3,1 g CH_4-C $m^{-2}a^{-1}$ dazwischen. Die Methanflüsse können auf den naturnahen Standorten signifikant mit dem NEE erklärt werden (r^2 0,53 – 0,68), was die funktionelle Verknüpfung von Methanproduktion und Kohledioxidaufnahme widerspiegelt. Die Methan-Jahresbilanzen lassen sich signifikant mit dem mittleren Wasserstand (r^2 0,54), der Anzahl aerenchymhaltiger Blätter (r^2 0,82), einer multiplen Regression der beiden Faktoren (r^2 0,85) und schließlich am besten mit dem NEP (r^2 0,87) erklären. Alle diese Faktoren haben funktionelle Bedeutung für die Methanproduktion bzw. Emission.

Regenmoorökosysteme weisen aufgrund der nährstoffarmen Standorte sehr geringe Lachgasemissionen auf. Einzig auf den beiden ehemaligen trockenen Torfstichen konnten mit 50 ± 47 - 168 ± 94 mg N_2O-N $m^{-2}a^{-1}$ nennenswerte Emissionen nachgewiesen werden.

Zur Ermittlung der Kohlenstoffbilanz (**Kap. 6**) wurde die CH_4-C-Bilanz von der CO_2-C-Bilanz abgezogen, wobei die gelösten organischen Verluste aus Literaturangaben mit 7 g C $m^{-2}a^{-1}$ miteinbezogen wurden. Auf den degradierten Standorten wird die Kohlenstoffbilanz nur unwesentlich vom Methan beeinflusst, während auf den naturnahen Standorten die erheblichen Methanemissionen zu einem deutlichen Unterschied zwischen der mittleren CO_2-C-Bilanz (-71 ± 40,5 g CO_2-C $m^{-2}a^{-1}$) und der C-Bilanz (-45,6 ± 40,5 g C $m^{-2}a^{-1}$) führt.

Die Klimarelevanz der Standorte wurde über die Multiplikation der Bilanzen aller drei Gase mit dem jeweiligen globalen Erwärmungspotential (GWP) für 100 und 500 Jahre Bezugszeitraum ermittelt. Es zeigt sich, dass alle Standorte beim kürzeren Bezugszeitraum (100 a) zur Klimaerwärmung beitragen. Besonders eindrücklich ist der Unterschied zwischen C-Bilanz und GWP-Bilanz (Klimarelevanz) auf den nassen naturnahen Standorten, die als C-Senke wirken (-80,5 ± 37,3 g C $m^{-2}a^{-1}$), aber mit 77,5 ±

40,5 g CO_2-C Äquivalenten $m^{-2}a^{-1}$ (GWP-Bilanz) zur Klimaerwärmung beitragen. Dies liegt an der erheblichen

Emissionen von Methan, das eine 21-fach stärkere Klimawirkung aufweist als CO_2 (GWP 100). Die degradierten Standorte erreichen dagegen eine klimaerwärmende Wirkung bis zu 465,6 ± 71,1 g CO_2-C Äquivalenten $m^{-2}a^{-1}$. Wird das GWP 500 gerechnet, ändert sich die klimabelastende Wirkung für den degradierten Standort kaum (455,4 ± 70,3 g CO_2-C Äquivalenten $m^{-2}a^{-1}$), die nassen naturnahen Standorte dagegen wirken nun klimaentlastend (-52,6 ± 37,6 g CO_2-C Äquivalenten $m^{-2}a^{-1}$). Dieser Unterschied liegt an der viel höheren Beteiligung von Methan an den Gasbilanzen der naturnahen Standorte, aber des auf lange Sicht zurückgehenden globalen Erwärmungspotentials von Methan aufgrund der gegenüber CO_2 kürzeren Verweilzeiten in der Atmosphäre.

Die Renaturierung von ehemaligen Torfstichen trägt mit 339,5 ± 53,3 g CO_2-C $m^{-2}a^{-1}$ bzw. 336,8 ± 54 g CO_2-C Äquivalente $m^{-2}a^{-1}$ zur Klimaentlastung bei. Werden die vorentwässerten Hochmoorheiden renaturiert, ist immerhin noch mit einer Einsparung von 108,5 ± 53,3 g CO_2-C $m^{-2}a^{-1}$ bzw. 66,6 ± 55 g CO_2-C g Äquivalente $m^{-2}a^{-1}$ zu rechnen.

Bayern hat ein Klimaschutzprogramm aufgelegt, mit dem Ziel, die CO_2-Emissionen auf 80 Mio. t zu senken. Es wird allerdings davon ausgegangen, dass eine Lücke von 3 Mio. t verbleibt, die mit technischen Maßnahmen alleine nicht zu schließen sein wird. Auf der Basis der ermittelten Klimaentlastungswirkung durch Regenmoorrenaturierung und der potenziellen Gesamtflächen für die Renaturierung wurde das gesamte Klimaentlastungspotenzial durch Regenmoorrenaturierung abgeschätzt und dem pro-Kopf-Anteil der Einsparungslücke aus dem Klimaschutzprogramm gegenübergestellt. Daraus ergibt sich, dass durch Regenmoorrenaturierung für die Bevölkerung des voralpinen Moorgürtels der pro-Kopf-Anteil der Einsparungslücke zu 27-36 % geschlossen werden kann. Regenmoorrenaturierung kann damit einen relevanten Beitrag zur Umsetzung von Klimaschutzzielen auf regionaler Ebene leisten. Wenn Bayern sich darüber hinaus doch dazu entschließen sollte, die ehrgeizigeren Kyoto-Ziele umzusetzen, könnten immerhin noch 10-13 % der pro-Kopf Reduktionsverpflichtungen im voralpinen Moorgürtel durch Regenmoorrenaturierung erreicht werden. Das bayerische Moorentwicklungsprogramm wäre die geeignete Plattform für die Umsetzung der Synergien von Moorschutz und Klimaschutz.

Summary

Natural mires accumulate organic substances continuously and act as carbon sinks. At a worldwide scale, the amount of carbon stored in peatlands represents about 20% of the total soil carbon stock, unless peatlands cover just 3% of the worlds land-surface. Drainage and peat cutting provoke the decomposition of the carbon pools and convert peatlands to carbon sources. Research on carbon-exchange and climatic relevance were up to now mainly focused on boreal and sub-arctic peatlands. Therefore, this project was initiated to fill a regional and thematic gap, studying trace gas exchange of natural, degraded and restored bog-ecosystems in the southern German mire belt in the forelands of the Bavarian Alps. The overall goal was to assess the carbon balance and the climatic relevance via trace gas exchange measurements and to clarify, if bog-restoration is a viable means for climate mitigation. For that purpose, a new closed chamber system was developed, as existing techniques had limitations to be applied in the frame of this study (**chapter 2**). The prominent properties of the chamber are the transparency and the cooling system, working without line connection. The temperature inside the chamber can be controlled to ± 1°C even on bright days. Therefore, photosynthesis is not disturbed and the measurement of the net ecosystem exchange within a small-scale mosaic of ecosystems is possible. Crosschecks with the eddy covariance technique revealed highly coincident flux-rates for CO_2 ($r^2=0.94$, 1:1 line). As prerequisite to assess the carbon balance and the climatic relevance of the ecosystems, the chamber allows sampling CH_4 as well as N_2O, parallel to CO_2.

Site selection led to a total of 12 sites with 36 plots as representative examples of the southern German bog-ecosystems in the forelands of the Bavarian Alps. The selected sites were dry former peat cut areas, drained-only bog heathlands, restored *Sphagnum*-lawns, restored moist bog heathlands, natural bog shrubs, natural *Sphagnum*-lawns, *Eriophorum*-hummocks and *Scheuchzeria-Sphagnum* hollows. This field based selection was post-hoc assessed in terms of site differentiation (**chapter 3**). Vegetation composition and site factors were analysed with multivariate ordination techniques in view of inspecting similarities between the sites. A canonical correspondence analysis (CCA) revealed a clear differentiation of the sites along a disturbance gradient, confirming the field based site selection.

Gas exchange measurements for CO_2, CH_4 and N_2O were done weekly to twice a week at these plots. The determined CO_2 fluxes were used for the parameterisation of a NEE-model (**chapter 4**). NEE was modelled in 0.5 hours steps over the entire measurement year and the net ecosystem productivity (NEP) was integrated from the NEE curve, with the convention that negative values represent uptake to the system. As a result, former peat cut sites were detected to act as strong sources for carbon dioxide with mean emissions of 401.5 ± 47.5 g CO_2-C $m^{-2}a^{-1}$ whereas natural sites were notable sinks (-71 ± 40.5 g CO_2-C $m^{-2}a^{-1}$). Restored sites fell in between with 127 ± 47.3 g CO_2-C $m^{-2}a^{-1}$ still acting as sources for carbon dioxide but reduced to 30% of the amount of the former peat cut sites. The NEP differentiated the sites along the disturbance gradient. NEP correlated significantly with environmental

variables like electrical conductivity (r^2=0.91), leaf area index (r^2=0.87) and mean water-table (r^2=0.84).

Methane and nitrous oxide annual balances (**chapter 5**) do separate as well along the disturbance gradient. Maximum methane emissions were obtained at the natural *Sphagnum*-hollow (38.2 ± 2.2 g CH_4-C $m^{-2}a^{-1}$), whereas the dry former peat cut areas were almost neutral in methane emissions. Restored sites were found in between (1.5 ± 0.2 – 7.1 ± 3.1 g CH_4-C $m^{-2}a^{-1}$). Instant methane fluxes at the natural sites could be explained significantly with NEE (r^2 0.53-0.68), representing the functional link between carbon dioxide uptake and methane production. The annual methane-balances could be explained with water-table (r^2 0.54), quantity of aerenchymous leaves (r^2 0.82), multiple linear regression of both factors (r^2 0.85) and finally best with NEP (r^2 0.87). All these factors are functionally related to the production or emission of methane. Due to their low nutrient status and normally high water table, bog ecosystems show very small N_2O emissions. Consequently only the dry former peat-cut areas had notable emissions rates (50 ± 47 - 168 ± 94 mg N_2O-N $m^{-2}a^{-1}$).

The carbon balance (**chapter 6**) was calculated as CO_2-C balance minus the CH_4-C balance. Carbon losses via DOC/DIC however, had to be estimated based on literature, as the complex hydrology of the sites did not allow for studying this export path. The carbon balance of the degraded sites was only slightly influenced by the methane emissions. At the natural sites, however, rising methane emissions led to significant differences between the carbon dioxide balance (-71 ± 40.5 g CO_2-C $m^{-2}a^{-1}$) and the carbon balance (-45.6 ± 40.5 g C $m^{-2}a^{-1}$). The differences at the restored sites were significantly lower (127 ± 47.3 g CO_2-C $m^{-2}a^{-1}$ to 137.6 ± 47.4 g C $m^{-2}a^{-1}$). The climatic relevance was calculated via the multiplication of the balances of all three gases with their corresponding global warming potential (GWP) differentiated for 100 years and 500 years timescale. At the 100 years timescale, all sites contributed to global warming. This is remarkable especially for the wet part of the natural sites, which acted as notable carbon sinks (-80.5 ± 37.3 g C $m^{-2}a^{-1}$) but heated the atmosphere with 77.5 ± 40.5 g CO_2-C equivalents $m^{-2}a^{-1}$ (GWP-balance). This is an effect of the elevated methane emissions at the natural sites and the fact that methane holds a 21-times higher global warming potential than carbon dioxide. The degraded sites, however, contributed to global warming up to 465.6 ±71.1 g CO_2-C eq. $m^{-2}a^{-1}$. Calculating the climatic relevance for the 500 years timescale, results in similar global warming effects of the degraded sites (455.4 ± 70.3 g CO_2-C eq. $m^{-2}a^{-1}$), whereas the wet natural sites shift in the long-term to mitigate global warming at a rate of –52.6 ± 37.6 g CO_2-C eq. $m^{-2}a^{-1}$. The reason beeing the reduced global warming potential of methane for the long-term (5.6 times CO_2) because of shorter lifetimes of methane in the atmosphere.

Restoration of former peat cut sites contributes to climate mitigation with a reduction of 339.5 ± 53.3 g CO_2-C $m^{-2}a^{-1}$ or 336.8 ± 54 g CO_2-C g eq. $m^{-2}a^{-1}$ respectively. Restoration of the widespread drained-only bog heathlands leads still to climate mitigation in the range of 108.5 ± 53.3 g CO_2-C $m^{-2}a^{-1}$ and 66.6 ± 55 g CO_2-C g eq. $m^{-2}a^{-1}$.

Bavaria published a climate protection programme with the overall goal to reduce the CO_2 emissions to 80 Mio t per year. However, it is likely that a 3 Mio t gap in reaching this goal with technical measures exclusively will remain. Based on the above outlined climate mitigation effect of bog restoration together with the potential total area for restoration, the overall climate mitigation potential was calculated and divided by the per capita gap of the climate protection programme for the population in the mire belt. This leads to a range of 27 – 36 % for closing the per capita gap via the assessed restoration measures. Therefore, bog restoration can significantly contribute to the achievement of climate protection goals at a regional level. If Bavaria, however, targets to fulfil the stronger Kyoto goals, bog restoration can still help to meet 10 – 13% of the per capita commitments. The recently published Bavarian mire-development programme could serve as a suitable platform for the application of the synergistic goals of mire conservation and climate protection.