Forschungszentrum Karlsruhe
in der Helmholtz-Gemeinschaft

Wissenschaftliche Berichte
FZKA 6898

N und C Spurengasaustausch und Erstellung eines N_2O-Emissionskatasters für tropische Regenwälder Australiens

R. Kiese

Institut für Meteorologie und Klimaforschung

Februar 2004

Forschungszentrum Karlsruhe
in der Helmholtz-Gemeinschaft

Wissenschaftliche Berichte
FZKA 6898

N und C Spurengasaustausch und Erstellung eines N_2O-Emissionskatasters für tropische Regenwälder Australiens

Ralf Kiese

Institut für Meteorologie und Klimaforschung

von der Fakultät für Forst- und Umweltwissenschaften
der Albert-Ludwigs-Universität Freiburg genehmigte Dissertation

Forschungszentrum Karlsruhe GmbH, Karlsruhe
2004

Impressum der Print-Ausgabe:

Als Manuskript gedruckt

Forschungszentrum Karlsruhe GmbH
Postfach 3640, 76021 Karlsruhe

Mitglied der Hermann von Helmholtz-Gemeinschaft
Deutscher Forschungszentren (HGF)

ISSN 0947-8620

N und C Spurengasaustausch und Erstellung eines N_2O-Emissionskatasters für tropische Regenwälder Australiens

Inauguraldissertation zur Erlangung der Doktorwürde
der Fakultät für Forst- und Umweltwissenschaften
der Albert-Ludwigs-Universität Freiburg i. Brsg.

vorgelegt von
Ralf Kiese

Freiburg im Breisgau
2003

Dekan:	Prof. Dr. K.-R. Volz
Referent:	PD Dr. Hans Papen
Korreferent:	Prof. Dr. E. Hildebrand

Zusammenfassung

N und C Spurengasaustausch und Erstellung eines N_2O-Emissionskatasters für tropische Regenwälder Australiens

In scharfem Kontrast zur Bedeutung tropischer Regenwälder als wichtige Quelle innerhalb des globalen N_2O-Budgets steht die derzeit nur äußerst begrenzte Anzahl von veröffentlichten N_2O-Flussraten aus diesen Ökosystemen. Problematisch ist auch, dass bislang zeitlich hoch aufgelöste Messungen von N_2O-Emissionen aus tropischen Wäldern über längere Zeiträume fehlten. Diese sind jedoch erforderlich, um einerseits die Unsicherheiten der Abschätzung der N_2O-Quellstärke auf Standortsebene zu verringern. Andererseits bilden solche Messungen die Grundlage für eine verlässliche Weiterentwicklung und Validierung prozessorientierter Modelle, die nach heutigem Stand des Wissens als das meist aussichtsreichste Werkzeug angesehen werden müssen, um die Abschätzung von N- und C-Spurengasemissionen auf regionaler und/ oder globaler Ebene qualitativ zu verbessern.

Im Rahmen dieser Arbeit wurden daher zeitlich hoch aufgelöste Datensätze des N_2O-, CH_4- und CO_2-Spurengasaustauschs zwischen verschiedenen tropischen Regenwaldböden Australiens und der Atmosphäre erarbeitet. Es ist dabei weltweit erstmals gelungen, den saisonalen Verlauf der N_2O-Emissionen und der CH_4-Aufnahme des Bodens eines Tieflandregenwaldes in mindestens täglicher Auflösung über ein komplettes Jahr zu bestimmen. Anhand begleitender Untersuchungen zu den Umweltbedingungen (z.B. Niederschlag, Bodenfeuchte, Bodentemperatur), der Bestimmung standörtlicher Unterschiede der Bodeneigenschaften (z.B. pH-Wert, C/N-Verhältnis, Textur) sowie durchgeführter Prozessanalysen (z.B. Messung von Brutto-Nitrifikationsraten) konnten wesentliche, die Höhe der N_2O-Emissionen auf Prozessebene regelnde Wirkungszusammenhänge aufgezeigt werden. Basierend auf dieser im Rahmen der vorliegenden Arbeit gewonnenen umfangreichen Datengrundlage wurde das prozessorientierte Modell PnET-N-DNDC, das ursprünglich zur Simulation von N- und C-Spurengasemissionen aus temperaten Wäldern entwickelt wurde, derart weiterentwickelt, dass es nunmehr auch zur Simulation von N_2O-Emissionen aus tropischen Waldökosystemen eingesetzt werden kann. Nach erfolgreicher Validierung des Modells wurde dieses mit einer GIS-Datenbank gekoppelt, um erstmals ein detailliertes N_2O-Spurengasinventar für ein ausgewähltes größeres tropisches Regenwaldgebiet berechnen zu können. Die für das Gebiet der Wet Tropics, Australien, durchgeführte Fallstudie zeigt, dass die tropische PnET-N-DNDC- Version ein elegantes wie zuverlässiges Werkzeug zur Regionalisierung von N_2O-Emissionen aus Böden tropischer Regenwälder darstellt.

Abstract

N and C trace gas exchange and calculation of a N_2O-emission inventory for tropical rainforest ecosystems in Australia

In contrast to the significant importance of tropical rainforests as one of the major single sources within the global N_2O budget the estimate of their source strength is still based on a small number of published data. Furthermore, long term measurements of N_2O-emissions from these ecosystems are still missing. However, more detailed measurements of N_2O-emissions from different tropical rainforest ecosystems are urgently required to better understand the major environmental drivers controlling the magnitude of N_2O-emissions. Such measurements are also needed to further develop and validate biogeochemical models, which simulate all processes and mechanism involved in N- and C-trace gas emissions from soils, and thus, at present provide the most promising tool to upscale emissions from the site to a regional or even global scale.

Hence, in the framework of this study the trace gas exchange of N_2O, CH_4 and CO_2 between tropical forest ecosystems and the atmosphere was recorded in high temporal resolution. This study shows for the first time world-wide measurements of the seasonality of N_2O-emissions and CH_4-uptake of a tropical lowland rainforest over an entire year in at least daily time intervals. With accompanying investigations on the influence of environmental conditions (e.g. precipitation, soil moisture, soil temperature) and soil properties (e.g. pH, C/N ratio, soil texture) as well as by the results of process studies, like site differences in gross nitrification rates, the main factors controlling the magnitude of N_2O-emissions on the process scale could be evaluated. On basis of the extensive database the PnET-N-DNDC model, which was originally developed to explicitly model biogeochemical cycling of C and N and associated biosphere-atmosphere exchange of N- and C-trace gases in temperate forest ecosystems, was further developed for its application to tropical rainforest ecosystems. After successful model validation the PnET-N-DNDC model was linked to a GIS database in order to calculate for the first time a detailed N_2O-emission inventory for a larger tropical rainforest area (Wet Tropics, Australia). The results of this case study revealed, that the further developed version of PnET-N-DNDC is a smart and reliable tool for up scaling N_2O-emissions from tropical rainforest soils.

Inhaltsverzeichnis

Danksagung

1. Einleitung und Problemstellung

Seit dem Umweltgipfel in Rio de Janeiro im Juni 1992 sind die Grenzen der ökologischen Belastbarkeit der Biosphäre Erde ein viel diskutiertes und öffentlich anerkanntes Problem geworden. Im Vordergrund dieser Diskussion stehen die zum Teil dramatischen Änderungen des globalen Klimasystems und die damit verbundenen ökologischen und ökonomischen Schäden auf regionaler und globaler Ebene. Schlagwörter, wie „Klimaveränderung", „Treibhauseffekt" und El Niño/ La Niña sind für große Teile der Bevölkerung längst keine fremden Begriffe mehr. Im Frühjahr 2001 stellte das Klimagremium der Vereinten Nationen, das Intergovernmental Panel on Climate Change (IPCC), im „IPCC Third Assessment Report - Climate Change 2001", die neuesten Ergebnisse der Klimaforschung vor. In diesem Report wird die Frage, ob sich das Klimasystem der Erde tatsächlich ändert, mit einem „eindeutigen Ja" beantwortet. Als Beleg hierfür wird in erster Linie der Anstieg der mittleren Temperatur an der Erdoberfläche um 0.6 ± 0.2 °C (IPCC, 2001) seit Ende des 19. Jahrhunderts herangezogen. Der Anstieg der mittleren Temperatur der Erdoberfläche ist dabei auf einen direkt oder indirekt durch menschliche Aktivitäten hervorgerufenen dramatischen Anstieg der Konzentration von Treibhausgasen in der Atmosphäre zurückzuführen, der seit dem Beginn der Industriellen Revolution Mitte des vorletzten Jahrhunderts, zu beobachten ist. Insbesondere betrifft dies die primär klimarelevanten Gase Wasserdampf (H_2O), Kohlendioxid (CO_2), Methan (CH_4), Lachgas (N_2O) sowie die halogenierten Kohlenwasserstoffverbindungen (FCKW) (Tab.1).

Tab. 1: Bedeutendste durch menschliche Aktivitäten beeinflusste Treibhausgase (IPCC, 2001).

Spurengas	Vorindustrielle Konzentration	Konzentration 1998	Zunahme der Konzentration	Atmosphärische Verweilzeit	Treibhaus-potential[2)]
CO_2	~ 280 ppmv	365 ppmv	1.5 ppmv/Jahr	5-200 Jahre	1
CH_4	~ 700 ppbv	1745 ppbv	7.0 ppbv/Jahr	12 Jahre	23
N_2O	~ 270 ppbv	314 ppbv	0.8 ppbv/Jahr	114 Jahre	296
FCKW[1)]	0	14-80 pptv	0.6–1.0 pptv/Jahr	260-50000 Jahre	5700 -12000

[1)] am Beispiel von Perfluoromethan (CF_4) und Hydrofluorocarbon-23 (HFC-23)

[2)] Bezugszeitraum 100 Jahre

Diese klimarelevanten Spurengase lassen die kurzwellige solare Strahlung (Wellenlänge < 4 µm) nahezu ungehindert bis zur Erdoberfläche passieren, wo sie absorbiert und z.T. als langwellige (Wellenlänge > 4 µm – 100 µm) Infrarotstrahlung reflektiert wird. Die langwellige Infrarotstrahlung wird von den Spurengasen in der Atmosphäre absorbiert und z.T. wieder zur Erdoberfläche zurückgestrahlt. Durch die gestiegenen Konzentrationen der Treibhausgase in der Atmosphäre ist der natürliche Treibhauseffekt, der eine Erhöhung der mittleren globalen Temperatur an der Erdoberfläche um 33 °C (von –18°C auf 15°C) bewirkt, signifikant verstärkt worden. Diese Temperaturerhöhung wirkt sich direkt auf den Energiehaushalt der Erde aus und kann so indirekt zur Veränderung von Meeresströmungen und der planetarischen Zirkulation führen, mit der Konsequenz globaler und regionaler klimatischer Veränderungen. Zur Prognose beispielsweise einer geänderten globalen Niederschlagsverteilung werden hoch komplexe Klimamodell eingesetzt. Die Güte der von diesen Modellen erstellten Voraussagen, ist jedoch eng an Kenntnisse über die zukünftige Entwicklung der Zusammensetzung der Atmosphäre geknüpft. Nicht nur für die Modellierung, sondern auch für die Entwicklung von Maßnahmen zur Emissionsreduzierung treibhausrelevanter Spurengase, ist die Erfassung der Senken und insbesondere auch die Erfassung der Quellen von zentraler Bedeutung.

Während die 30%ige Zunahme des Kohlendioxids in der Atmosphäre seit 1750 überwiegend auf die Oxidation von organischem Kohlenstoff durch Verbrennung fossiler Energieträger und Landnutzungsänderungen zurückgeht, sind die Kenntnisse über zeitliche und/ oder räumliche Veränderungen der Quell- und Senkenstärken der Spurengase CH_4 und N_2O seit der industriellen Revolution immer noch unbefriedigend.

Der Anstieg, der CH_4-Konzentrationen von ca. 700 ppbv im Jahr 1750 auf derzeit ca. 1745 ppbv, einer Steigerung um 150 %, ist im Vergleich zum atmosphärischen CO_2-Konzentrationsanstieg noch dramatischer. Neben den Feuchtgebieten gelten der Reisanbau und die Viehhaltung als wichtigste Quellen für atmosphärisches Methan (IPCC, 2001). Der Abbau von Methan wird von der Reaktion mit OH in der Troposphäre dominiert (Fung et al., 1991), jedoch stellen biologische Oxidationsprozesse methanotropher Bakterien in aeroben Böden (Conrad, 1994) eine weitere Senke in der Größenordnung von etwa 10 % dar. Dieser verhältnismäßig kleine Beitrag entspricht jedoch in etwa dem Betrag, um den die globalen CH_4-Emissionen die CH_4-Senken übersteigen (IPCC, 2001), woraus sich schließen lässt, dass Änderungen in der Senkenstärke der Böden, potentiell einen bedeutenden Effekt auf die zukünftige Entwicklung der CH_4 Konzentrationen in der Atmosphäre haben können (Ridgwell et al., 1999). Die wenigen Veröffentlichungen, die es derzeit über den CH_4-

Austausch zwischen tropischen Regenwaldökosystemen und der Atmosphäre gibt (Delmas et al., 1992; Keller und Reiners, 1994; Goreau und de Mello, 1998) zeigen, dass selbst diese, zumindest während der Trockenzeit signifikante Senken für CH_4 darstellen können. Im „Third Assessment Report" des IPCC (IPCC, 2001) wird jedoch darauf hingewiesen, dass gerade der Beitrag biogener Quellen und Senken zum globalen CH_4 Budget hoch variabel und deshalb die Voraussage zukünftiger atmosphärischer CH_4 Konzentrationen sehr schwierig ist.

Tab. 2: Abschätzung des globalen N_2O Budgets in Tg N $Jahr^{-1}$(verändert nach IPCC, 2001)

Zitat	Mosier et al. (1998) Kroeze et al. (1999)		Olivier et al. (1998)		IPCC, 1995	
Bezugsjahr	1994	Streuung	1990	Streuung	1980	Streuung
Natürliche Quellen						
Ozeane	3.0	1.0-5.0	3.6	2.8-5.7	3.0	1.0-5.0
Atmosphäre (NH_3 Oxidation)	0.6	0.3-1.2	0.6	0.3-1.2		
Tropische Böden						
Feuchtwälder	3.0	2.2-3.7			3.0	2.2-3.7
Trockensavannen	1.0	0.5-2.0			1.0	0.5-2.0
Temperate Böden						
Wälder	1.0	0.1-2.0			1.0	0.1-2.0
Grasland	1.0	0.5-2.0			1.0	0.5-2.0
Böden total			6.6	3.3-9.9		
Zwischensumme	9.6	4.6-15.9	10.8	6.4-16.8	9.0	4.3-14.7
Anthropogene Quellen						
Landwirtschaftliche Böden	4.2	0.6-14.8	1.9	0.7-4.3	3.5	1.8-5.3
Biomasseverbrennung	0.5	0.2-1.0	0.5	0.2-0.8	0.5	0.2-1.0
Industrie	1.3	0.7-1.8	0.7	0.2-1.1	1.3	0.7-1.8
Viehhaltung	2.1	0.6 -3.1	1.0	0.2 -2.0	0.4	0.2-0.5
Zwischensumme	8.1	2.1 -20.7	4.1	1.3 -7.7	5.7	3.7-7.7
Gesamtsumme	**17.7**	**6.7-36.6**	**14.9**	**7.7-24.5**	**14.7**	**8.0-22.4**
Senken						
Aufnahme in Böden	?		?		?	
Photolyse in der Stratosphäre	12.3	9.0–16.0	12.3	9.0–16.0	12.3	9.0-16.0
Atmosphärischer Anstieg	3.9	3.1 -4.7	3.9	3.1 -4.7	3.9	3.1-4.7
Gesamtsumme	**16.2**	**9.0-16.2**	**16.2**	**9.0-16.2**	**16.2**	**9.0-16.2**

Neben Kohlendioxid, das zu ca. 55% zum anthropogen verursachten Treibhauseffekt beiträgt und dem Methan sowie den FCKW, die zu 19 % bzw. 21 % beitragen, scheint das Spurengas N_2O mit einem Anteil von 5% eine auf den ersten Blick eher untergeordnete Rolle zu spielen (Myhre et al., 1998). Aufgrund des 296-fach höheren Treibhauspotentials von N_2O gegenüber CO_2 (Tab. 1) (IPCC, 2001), das durch seine vergleichsweise lange Verweilzeit in der Atmosphäre von 125 Jahren und die effektive Absorption von Infrarot-Strahlung hervorgerufen wird, sowie der starken anthropogen bedingten Änderung des globalen N-Kreislaufs (Galloway, 1998; Holland et al., 1999) ist jedoch damit zu rechnen, dass die Bedeutung von N_2O an der globalen Erwärmung zukünftig weiter zunehmen wird. Außerdem ist das in der Troposphäre inerte N_2O am Ozonabbau in der Stratosphäre beteiligt. In der Stratosphäre wird N_2O photolytisch gespalten und reagiert mit Sauerstoffradikalen zu Stickstoffmonoxid (NO), das letztlich katalytisch in den Abbau von Ozon involviert ist (Crutzen, 1970; Prather, 1998).

Bei der Betrachtung der Quellen von atmosphärischem N_2O wird deutlich, dass tropische Feuchtwälder mit 2.2-3.7 Tg N $Jahr^{-1}$ neben den Ozeanen (1.0-5.0 Tg N $Jahr^{-1}$), landwirtschaftlich genutzten Böden (0.6-14.8 Tg N $Jahr^{-1}$) und der Viehhaltung (0.6 – 3.1 Tg N $Jahr^{-1}$) die größte Einzelquelle darstellen und mit etwa 12-21% zum globalen N_2O Budget beitragen (Tab. 2). Als Senke für atmosphärisches N_2O wird hauptsächlich die Photolyse in der Stratosphäre aufgeführt (Tab. 2). Die Aufnahme und Reduktion von N_2O in Böden ist schwer zu quantifizieren, scheint jedoch nach jetzigem Stand des Wissens als Senke für atmosphärisches N_2O eher von untergeordneter Bedeutung zu sein (Smith, 1997).

Obwohl es keinen Zweifel an der Bedeutung tropischer Wälder als Quelle innerhalb des globalen N_2O Budget gibt, basieren die Abschätzungen der Quellstärken dieser Ökosysteme bisher auf vergleichsweise wenigen Messungen. Breuer et al. (2000) geben im Rahmen einer Literatur-Übersicht einen Überblick über die bis zum Jahr 2000 veröffentlichten Ergebnisse zu N_2O-Emissionen aus tropischen Regenwaldökosystemen. Diese insgesamt 49 Publikationen sind durch die in den letzten beiden Jahren publizierten Ergebnisse von Davidson et al. (2001), McSwiney (2001), Melillo et al. (2001), sowie Erickson et al. (2002) noch zu ergänzen. Allen Untersuchungen gemeinsam ist die Feststellung einer hohen räumlichen und zeitlichen Dynamik der N_2O-Emissionen, die eine exakte, nur auf Freilandmessungen basierende Quantifizierung der N_2O-Quellstärke tropischer Wälder ausschließt (Tab. 2). Die beobachtete hohe Dynamik der N_2O-Flussraten (0.1 – 207.0 µg N_2O-N m^{-2} h^{-1}) kann ihrerseits auf das komplexe Wirkungsgefüge der in die N_2O-Produktion, -Konsumption und -Emission involvierten Prozesse zurückgeführt werden (Abb. 1).

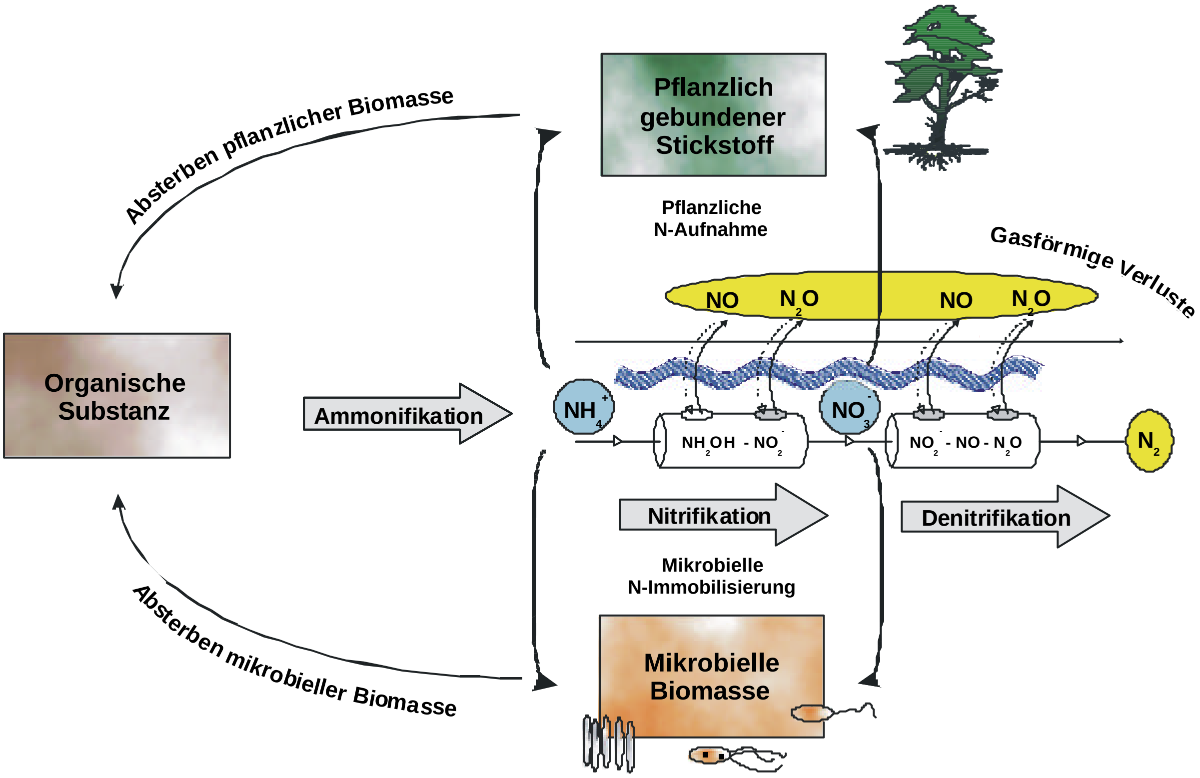


Abb. 1: N-Kreislauf und daran gekoppelte Wege der N-Spurengasfreisetzung aus Waldböden (verändert nach Firestone und Davidson, 1989; Breuer, 2000)

Dabei stellen die Nitrifikation und die Dentirifikation die zentralen mikrobiellen Prozesse dar, die zur Produktion, aber im Falle der Denitrifikation auch zur Konsumption von N_2O in Böden führen. Das im Zuge der Ammonifizierung (Mineralisierung) gebildete Ammoniak (NH_3)/ Ammonium (NH_4^+) wird in der Nitrifikation als Ausgangssubstrat verwendet und von nitrifizierenden Mikroorganismen über Hydroxylamin (NH_2OH) und Nitrit (NO_2^-) zu Nitrat (NO_3^-) oxidiert. An diesem aeroben Prozess können sowohl autotrophe (z.B. *Nitrosomonas* spec., *Nitrobacter* spec.) als auch heterotrophe Nitrifizierer (z.B. *Pseudomonas pudita*, *Alcaligenes faecalis*) beteiligt sein (Schimel et al., 1984; Papen et al., 1989; Daum et al., 1998). Dabei ist noch unklar, ob die Produktion des Spurengases N_2O, ausgehend von Nitrit durch Nitrifizierer-Denitrifikation bzw. Chemodenitrifikation (Reduktion) (Firestone und Davidson, 1989; Poth und Focht, 1995; Li et al., 2000) stattfindet und/ oder ob N_2O hauptsächlich bei der Oxidation von Hydroxylamin zu Nitrit als Nebenprodukt entsteht (Ritchie und Nicholas, 1972; Papen und Rennenberg, 1990). Unter anaeroben Bedingungen wird von Mikroorganismen innerhalb des Prozesses der Denitrifikation Nitrat als alternativer Elektronenakzeptor verwendet, das in Abhängigkeit der ökologischen Bedingungen unter der Bildung und teilweise Freisetzung der Zwischenprodukte Nitrit, Stickstoffmonoxid und

Lachgas bis zum molekularen N_2 reduziert werden kann (Firestone und Davidson, 1989). Neben den fakultativ anaeroben Gattungen *Pseudomonas* und *Alcaligenes* können auch die mit Pflanzenwurzeln in Symbiose lebenden N_2-fixierenden *Rhizobium* spec. und *Bradyrhizobium* spec. (O´Hara und Daniel, 1985) sowie auch Pilze (Zumft, 1997) - diese insbesondere bei niedrigen pH-Werten - am Prozess der Denitrifikation beteiligt sein. Der Beitrag der Chemo-Denitrifikation an der N_2O-Freisetzung durch chemische Umsetzungsprozesse von NO_2^-, die nur in Böden mit niedrigen pH-Werten auftritt, wird insgesamt als gering angesehen (Bremner, 1997).

Aus dem konzeptionellen „hole in the pipe“ Modell, das Firestone und Davidson (1989) zur Beschreibung der Produktion und Konsumption der Spurengase N_2O und NO während der Nitrifikation und der Denitrifikation entwickelten (Abb. 1), geht hervor, dass die Höhe der N-Spurengas Freisetzung, einerseits von der Höhe der Umsetzungsraten der Prozesse Nitrifikation und Denitrifikation (Durchfluss durch die Röhren), andererseits aber auch von der Höhe der dabei auftretenden Verlustraten während dieser Umsetzungsprozesse abhängt (Löcher in den Röhren). Diese Regelmechanismen sind ihrerseits stark abhängig von Umweltfaktoren wie z.B. Bodeneigenschaften (Bodentextur, pH-Wert, organischer Kohlenstoffgehalt), der Substratverfügbarkeit (NH_4^+ und NO_3^-), der inter- und intraspezifischen Konkurrenz um Nährstoffe oder der Bodenfeuchte und der Bodentemperatur (Firestone und Davidson, 1989; Davidson et al., 1993; Yamulki et al., 1997; Verchot et al., 1999; Breuer et al., 2000).

Die aufgezeigte Komplexität der Prozesse, die für die Emission von N_2O aus Böden verantwortlich sind, sowie die hohe zeitliche, wie auch räumliche Variabilität der sie steuernden biotischen und abiotischen Faktoren führen letztendlich dazu, dass die Regionalisierung, d.h. die flächenhafte Quantifizierung bodenbürtiger N_2O Emissionen trotz des z.T. guten Verständnisses auf der Prozess- und Standortsebene noch äußerst ungenau sind. Kroeze und Mosier (2002) stellen in diesem Zusammenhang fest, dass beispielsweise die Anwendung von Emissionsfaktoren, wie sie die IPCC Guidelines for National Greenhouse Gases (IPCC, 1997; IPCC, 2000) vorschlagen, aber auch einfache Interpolationsverfahren (Matson und Vitousek, 1990; Breuer et al., 2000) nicht geeignet sind, die Unsicherheiten beim Upscaling von N_2O-Emissionen auf regionaler und/ oder globaler Skala zu reduzieren. Es besteht daher der Bedarf an alternativen Methoden, wobei derzeit die Anwendung prozessorientierter Modelle, als das meist vielversprechende Werkzeug diskutiert wird, die Güte der flächenbezogenen Abschätzung von N_2O-Emissionen aus biogenen Quellen, wie beispielsweise landwirtschaftliche genutzten Böden oder Waldökosystemen, zu verbessern (Butterbach-Bahl et al., 2001). In den letzten Jahren wurden bereits mehrere biogeochemische Modelle zur Simulation des Stickstoff- und Kohlenstoffkreislaufs und des darin involvierten Austauschs von Spurengasen zwischen Böden und der Atmosphäre entwickelt, wie z.B. die Modelle CASA, CENTURY, DNDC und PnET-N-DNDC (Li et al., 1992; Parton et al., 1996; Potter et al., 1996; Li et al., 2000). Das letztgenannte PnET-N-DNDC Modell wurde explizit zur Simulation des N- und C-Kreislaufs in temperaten Waldökosystemen und des damit assoziierten N-Spurengasaustausches mit der Atmosphäre konzipiert. Validierungen zeigen, dass das PnET-N-DNDC-Modell N-Spurengasemissionen aus einer Vielzahl von temperaten Waldstandorten in Europa und den USA erfolgreich vorhersagen konnte (Stange et al., 2000; Butterbach-Bahl et al., 2002). In diesem Zusammenhang muss jedoch darauf hingewiesen werden, dass sowohl die Weiterentwicklung prozessorientierter Modelle als auch die Modellvalidierung, entscheidend von der Qualität der zugrunde liegenden Datensätze abhängt. Aufgrund der großen intra- und inter-annuellen Schwankungen der N_2O-Spurengasemissionen in temperaten (Ambus und Christensen, 1995; Zechmeister-Boltenstern und Menger, 1997; Papen und Butterbach-Bahl, 1999) und auch in tropischen Waldökosystemen (Keller und Reiners, 1994; Verchot et al., 1999; Breuer et al., 2000; Melillo et al., 2001), werden zeitlich hoch aufgelöste in-situ Messungen dringend benötigt, um die Weiterentwicklung und Validierung prozessorientierter Modelle verlässlich voranzutreiben. Die Qualitätsansprüche und Verfügbarkeit betreffen jedoch nicht nur die Emissionsdaten selbst, sondern auch die zur Modellinitialisierung und zum Modellantrieb benötigten Klimadaten, Boden- und Vegetationscharakteristika. Während die Datengrundlage für temperate Waldgebiete inzwischen stark verbessert ist, bestehen für tropische Waldökosysteme noch große Datendefizite.

Als Beitrag zur Reduzierung der Unsicherheiten bei der Abschätzung der regionalen N_2O-Quellstärke tropischer Regenwälder wurde im Rahmen dieser Arbeit ein kombinierter Forschungsansatz aus in-situ Messungen, Prozessanalysen und Modellentwicklung/ -anwendung verfolgt. Dabei sollte auf Grundlage einer verbesserten Datenbasis von N und C-Spurengasemissionen und der Erfassung der sie steuernden Parameter und Prozesse das mechanistische Modell PnET-N-DNDC für die Simulation von N_2O-Flüssen aus tropischen Regenwaldökosystemen weiterentwickelt, validiert und gekoppelt an eine GIS-Datenbank zur Erstellung eines N_2O-Emissionskatasters verwendet werden (Abb. 2). Als Untersuchungsgebiet dienten dazu die Wet Tropics in Australien, eine Region mit signifikantem Vorkommen tropischer Tiefland- und Bergwaldregenwälder, die aufgrund der stark ausgeprägten Klimagradienten, dem Vorkommen zahlreicher verschiedener Bodentypen, und nicht zuletzt wegen der Verfügbarkeit und hohen Qualität von GIS-Daten, ideal für diese Studie geeignet war (siehe Kapitel 2).

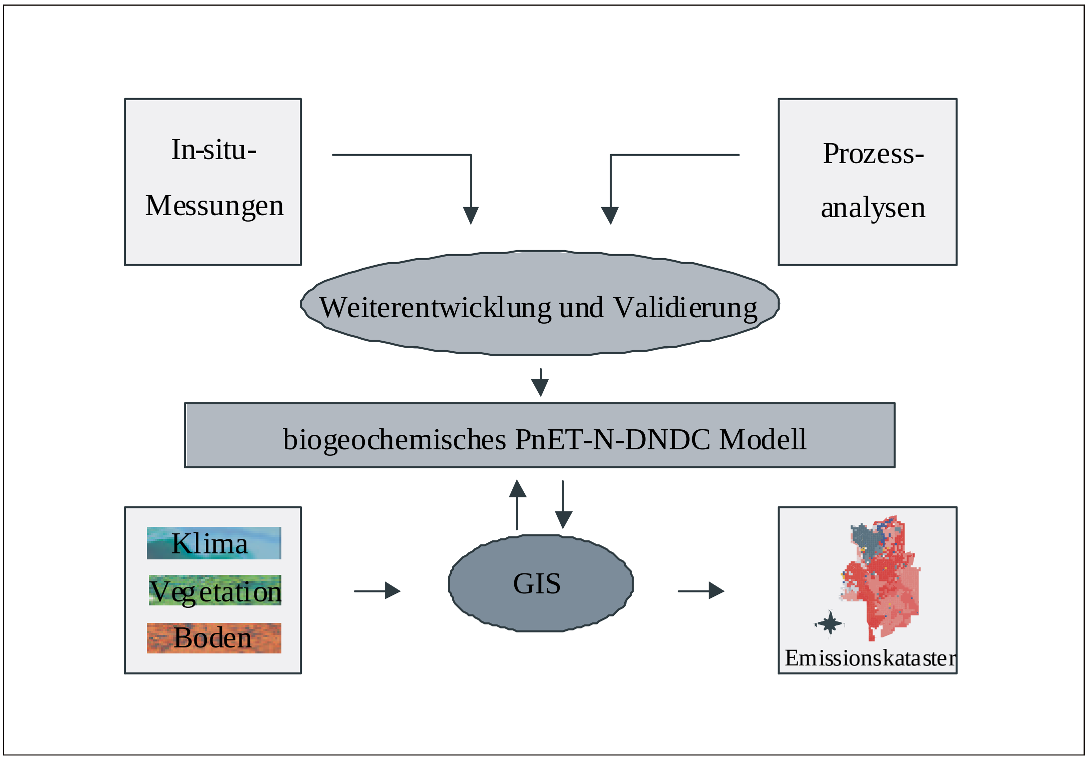


Abb. 2: Übersicht über die Arbeitsstrategie

Im einzelnen sollten folgende Ziele erreicht werden:

- Verbesserung der Datengrundlage zu N_2O-Emissionen aus tropischen Regenwaldökosystemen. Hierzu wurden zwei Forschungsansätze verfolgt:
  - Erfassung der räumlichen Variabilität von N_2O-Emissionen aus tropischen Regenwäldern in Abhängigkeit von klimatischen und edaphischen Faktoren.
  - Erfassung eines kompletten Jahresganges der N_2O-Emissionen in mindestens täglicher Auflösung für einen Tieflandregenwald-Standort in den Wet Tropics, Queensland, Australien.

- Bestimmung des Einflusses abiotischer (Bodentemperatur, Bodenfeuchte, C/N-Verhältnis, pH-Wert, NH_4^+- und NO_3^--Konzentrationen) und biotischer (Zusammensetzung und Größe der Population von Nitrifizierern und Denitrifizierern) Faktoren auf die N_2O-Emissionen.

- Erfassung saisonaler Unterschiede der Brutto-Nitrifikationsraten an einem Berg- und einem Tieflandregenwald-Standort und Evaluierung der Bedeutung der Nitrifikation für die N_2O-Emissionen aus tropischen Regenwäldern. Im Zuge dieser Arbeiten sollte zudem die Methode der Barometrischen Prozess-Separation zur Bestimmung von Brutto-Nitrifikationsraten für tropische Regenwaldböden validiert werden.

- Erfassung der Saisonalität von N_2O-Konzentrationen in verschiedenen Bodentiefen, des leaf area index (LAI), des Streufalls und der Abbauraten der Streuschicht im Hinblick auf die Weiterentwicklung und Validierung des PnET-N-DNDC Modells.

- Erfassung von CO_2-Emissionen als Indikator der Mineralisationsdynamik zu einem besseren Verständnis der mikrobiellen C- und N-Umsetzungen an verschiedenen Standorten.

- Erfassung eines kompletten Jahresganges der CH_4-Emissionen und CH_4-Konzentrationen in verschiedenen Bodentiefen als Beitrag zur Klärung der Frage, ob tropische Waldökosysteme eine Quelle oder Senke dieses wichtigen Treibhausgases darstellen.

- Weiterentwicklung und Validierung des biogeochemischen Modells PnET-N-DNDC zur Simulation von N_2O-Emissionen aus tropischen Waldökosystemen auf Grundlage der durchgeführten Untersuchungen und unter Berücksichtigung von Literaturdaten.

- Aufbau eines Geographischen Informationssystems (GIS) und Evaluierung der Anwendbarkeit von PnET-N-DNDC zur Berechnung eines N_2O-Emsissionskatasters für das Gebiet der Wet Tropics in Queensland, Australien.

2. Untersuchungsgebiet

Alle im Rahmen dieser Arbeit untersuchten Standorte liegen im Gebiet der von der UNESCO 1988 zum Weltkulturerbe erklärten „Wet Tropics“, einem bis zu etwa 50 km breiten Landstreifen, der sich entlang der Ostküste Australiens von Townsville über Cairns bis nach Cooktown (Abb. 3) erstreckt.

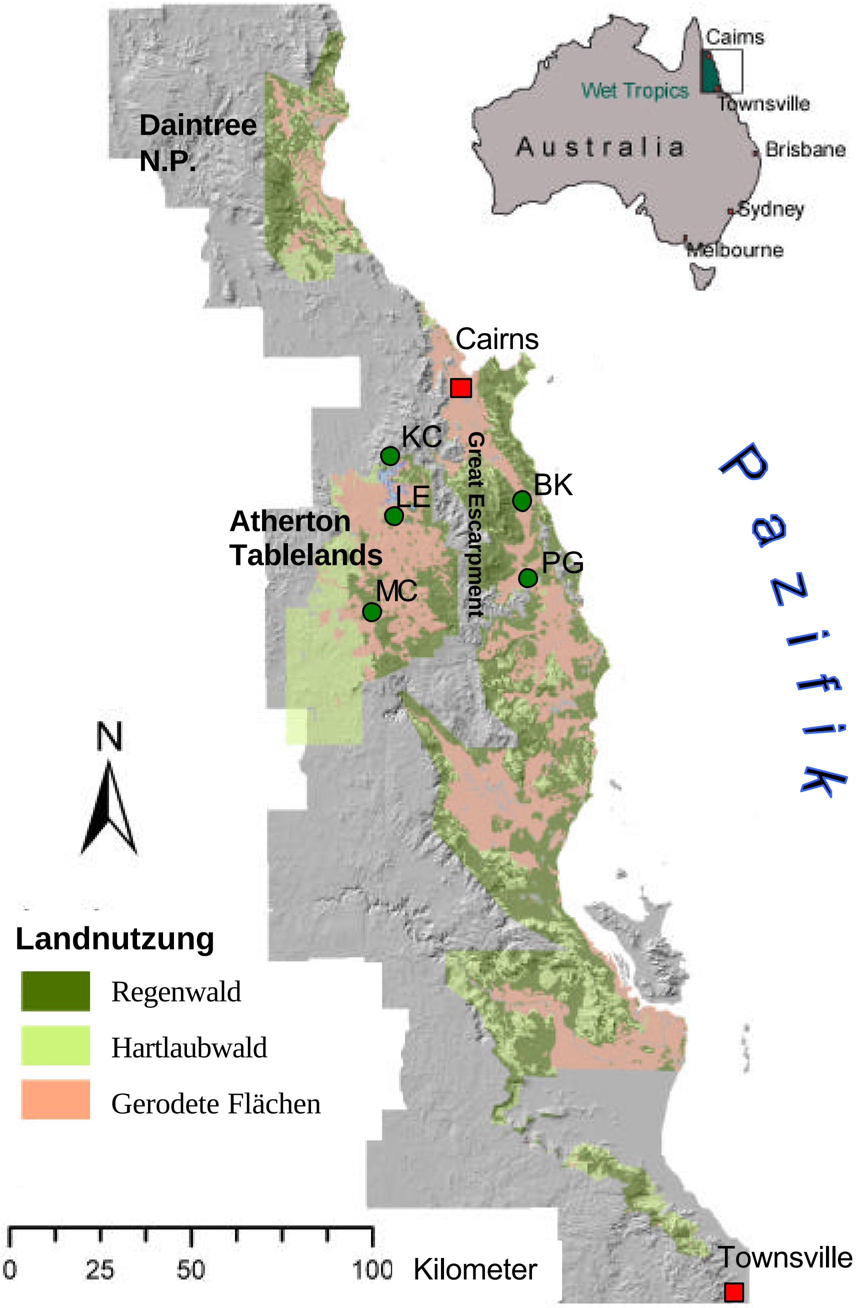


Abb. 3: Topographie und Landnutzung des Untersuchungsgebiets, sowie Lage der in dieser Arbeit genannten Untersuchungsstandorte (Tieflandregenwald: BK= Bellenden Ker, PG = Pin Gin Hill; Bergregenwald: KC = Kauri Creek, LE = Lake Eacham, MC = Massey Creek).

Die Wet Tropics mit ihren bedeutenden Vorkommen von Berg- und Tieflandregenwäldern umfassen eine Fläche von etwa 9000 km². Hauptsächlich bis zum Ende des 19. Jahrhunderts sind jedoch größere Waldflächen für landwirtschaftliche Nutzung (Mais, Zuckerrohr) und zur Viehhaltung, insbesondere auf den Atherton Tablelands, gerodet worden. Das Untersuchungsgebiet lässt sich naturräumlich in die küstennahen Tiefländer, die Hänge des Great Escarpments und die sich westlich daran anschließenden Atherton Tablelands, einer Tafelfläche zwischen 700 und 900 m ü. NN, gliedern (Abb. 3).

Tab.3: Charakterisierung der im Rahmen dieser Arbeit untersuchten Standorte

	Kauri Creek	Bellenden Ker	Pin Gin Hill
Lage	145°38’E	145°54’E	145°58’E
	17°17’S	17°16’S	17°33’S
Höhe über NN (m)	790	80	60
Mittlerer Jahresniederschlag (mm)	1594[a]	4395[b]	3609[a]
Jahresmitteltemperatur (°C)	20.9[a]	24.3[b]	24.1[a]
Bodentyp	Ustochrept[c]	Ustochrept[d]	Krasnozem[a]
Ausgangsgestein	Granit[c]	Granit[d]	Basalt[a]
Vegetationstyp	Bergregenwald	Tieflandregenwald	Tieflandregenwald
	Complex notophyll vine forest[e]	Complex mesophyll vine forest[e]	Complex mesophyll vine forest[e]
Hangneigung (°)	12.0–14.0	9.0-12.0	15.0-18.0
pH ± se 0-10cm	5.2 ± 0.08	4.1 ± 0.03	4.7 ± 0.06
Bodendichte 0-5cm (g cm^{-3})[d]	1.05 ± 0.05	1.09 ± 0.03	0.79 ± 0.02
C/N-Verhältnis 0-10cm	14.6	12.1	13.8
Organischer C-Gehalt 0-10cm (%)	2.93-3.51[c]	3.11	9.15
Bodenarten 0-10cm	Lts	Lts	T
Sand (%)	59-77[c]	57	7
Schluff (%)	8-10[c]	21	26
Ton (%)	13-33[c]	22	67

a) Spain et al. (1989)

b) Daten bereitgestellt vom Bureau of Meteorology, Brisbane

c) Laffan (1988)

d) Bob Hewett (unveröffentlichte Daten), Tropical Forest Research Centre Atherton

e) Australische Klassifizierung nach Tracey (1982)

Diese topographische Besonderheit des Untersuchungsgebiets führt zu einer großen räumlichen Variabilität der Temperatur und des Niederschlags (Kiese und Butterbach-Bahl, 2002 [**I**]; Kiese et al., 2003 [**IV**]). Letztere begründet sich in der überwiegend westlich orientierten Zugrichtung der Zyklonen, die beim Auftreffen am Great Escarpment zu Stauniederschlägen führen. Maximale Niederschläge treten daher in den Höhenlagen des Great Escarpment (Gipfelregion des Mt. Bellenden Ker 1592 m: bis über 9000 mm Jahr^{-1}) sowie in den küstennahen Tiefländern (etwa 4000 mm) nördlich und südlich der Stadt Cairns auf. Durch die hohen Niederschlagsgradienten von bis zu 20-70 mm km^{-1} nehmen die Jahresniederschläge im Untersuchungsgebiet in westlicher und südlicher Richtung auf Werte unter 1000 mm Jahr^{-1} ab (Tracey, 1982). Die mittlere Jahrestemperatur innerhalb des Untersuchungsgebiets bewegt sich wegen der Höhenabhängigkeit der Lufttemperatur in einem Bereich zwischen von 18 und 25°C, wobei ein signifikanter Unterschied in den Werten der mittleren Jahrestemperatur zwischen den höher gelegenen Standorten auf den Atherton Tablelands und denen des küstennahen Tieflands besteht (Tab. 3). Die hohe räumliche Variabilität der Bodentypen und Bodenarten in den Wet Tropics ist auf die Vielfalt der geologischen Ausgangsgesteine wie z.B. Granite, Metamorphite, Basalte und saure Vulkangesteine (Rhyolith) sowie die stark unterschiedlichen Rahmenbedingungen der Bodenbildung im Untersuchungsgebiet zurückzuführen (Kiese et al., 2002 [**I**], Kiese et al., 2003 [**IV**]). Aufgrund der oben aufgeführten Klimagradienten und der Heterogenität der Bodentypen bietet das Gebiet der Wet Tropics in Australien eine weltweit einmalige Möglichkeit zu einer äußerst differenzierten Untersuchung von N- und C-Spurengasfreisetzungen sowie der sie kontrollierenden Parameter in tropischen Waldökosystemen (Tab. 3).

3. Erfassung von N und C Spurengas-Emissionen aus tropischen Waldökosystemen Australiens

Im Rahmen von mehrmonatigen Intensivmesskampagnen (Januar - März 2000; Oktober 2000 - Januar 2001) wurden die N_2O- und CO_2-Emissionen aus Böden eines Bergregenwald-Standortes (Kauri Creek) sowie aus Böden zweier Tieflandregenwald-Standorte (Bellenden Ker und Pin Gin Hill) gemessen (Tab. 3, Abb. 3). Außerdem konnte im Rahmen dieser Arbeit am Tieflandregenwald-Standort Bellenden Ker, weltweit erstmals ein kompletter Jahresgang (November 2001 – Oktober 2002) der N_2O- und CH_4-Emissionen in zeitlich hoher Auflösung erfasst werden.

3.1. Messsystem und Messdesign

Die Erfassung der N_2O-, CH_4- und CO_2-Emissionen erfolgte durch die sogenannte „geschlossene Kammermethode“ (Butterbach-Bahl et al., 1998), wobei die Emissionen von 5 Messkammern durch ein mobiles vollautomatisches Messsystem simultan gemessen wurden (Breuer et al., 2000). Das Messsystem besteht aus einem Gaschromatographen mit einem ^{63}Ni Electron Capture Detector (ECD) zur Analytik von N_2O und einem Flame Ionisation Detector (FID) zur Bestimmung von CH_4-Konzentrationen, einem Infrarot-CO_2-Analysator sowie einem automatischen Bodengas-Probenahmesystem. Eine detaillierte Beschreibung des Messsystems findet sich in Breuer (2000) und Breuer et al. (2000). Die Höhe der Emissionen wurde als Mittelwert der linearen Regression (N = 4) des Konzentrationsanstiegs der einzelnen Gase in den 5 geschlossenen Messkammern berechnet, wobei alle bestimmten Flussraten für Temperatur und Druck korrigiert wurden. Die Konzentrationsmessungen jedes einzelnen Messzyklus sind direkt durch Aufgabe von Eichgas (N = 4) kalibriert worden. Während der beiden Intensivmesskampagnen wurde als Messzyklus ein Intervall von 60 Minuten Beprobung mit geschlossenen und weiteren 60 Minuten mit geöffneten Messkammern gewählt. Um einerseits Isolationseffekte zu vermeiden, wie z.B. das Ausschließen von Niederschlag und Streu während des geschlossenen Zustands der Messkammern, und um andererseits die räumliche Variabilität der Emissionen erfassen zu können, wurden alle fünf Messkammern einmal wöchentlich versetzt (Kiese und Butterbach-Bahl, 2002 [**I**]). Dieses Messdesign wurde prinzipiell auch für die Erfassung des kompletten Jahresganges der N_2O- und CH_4-Emissionen beibehalten, wobei das Messintervall von 60 auf 100 Minuten verlängert wurde und die Messkammern bis zum Start eines neuen Messzyklus für 200 Minuten geöffnet blieben. Die Versetzung der Messkammern wurde aus logistischen

Gründen auf einen zwei-wöchentlichen Rhythmus umgestellt (Kiese et al., 2003 [**III**]. Durch die zeitlich hohe Auflösung der Messungen, sowohl während der Intensivmesskampagnen in den Jahren 2000 und 2001 (60 potentielle Flussraten pro Tag) als auch während der Ermittlung des Jahresganges der N_2O- und CH_4-Emissionen von November 2001 bis Oktober 2002 (ca. 24 potentielle Flussraten pro Tag), konnte aufgrund nur äußerst geringer Datenausfälle eine sehr hohe Flussraten-Datendichte erreicht werden, die für tropische Regenwaldökosysteme einzigartig ist.

3.2. Zeitliche und räumliche Variabilität der N_2O-Emissionen

Die bereits für temperate Waldökosysteme (Brumme, 1995; Papen und Butterbach-Bahl, 1999) und erstmals auch von Breuer et al. (2000) für tropische Regenwaldökosysteme beobachtete hohe zeitliche Dynamik der N_2O-Emissionen konnte auch im Rahmen dieser Arbeit detailliert nachgewiesen und erfasst werden. An allen untersuchten Standorten war der Anstieg der N_2O-Emissionen auf bis das 10-fache des Ausgangswertes durch eine äußerst schnelle Reaktionszeit von nur 1-2 Stunden nach Niederschlagsbeginn gekennzeichnet (Kiese und Butterbach-Bahl, 2002 [**I**]; Kiese et al., 2003 [**III**]). Der schnelle Anstieg der N_2O-Emissionen konnte zusätzlich auch im Rahmen eines Boden-Befeuchtungsexperimentes am Standort Pin Gin Hill, das am Ende der Trockenzeit 2000 durchgeführt wurde, beobachtet werden (Kiese und Butterbach-Bahl, 2002 [**I**]). Dieser Versuch zeigte, dass die N_2O-Emissionen nach künstlicher Beregnung (40 mm) schnell ansteigen, nach etwa 4 Stunden ein Maximum erreichen und nach etwa 12 Stunden wieder auf das Ausgangniveau zurückfielen. Die in der vorliegenden Arbeit erzielten Ergebnisse zeigen weiterhin, dass an allen Standorten das Niveau der N_2O-Emissionen in der Regenzeit (Dezember – März) signifikant höher war als in der Trockenzeit (Juni – September). Die höchsten mittleren N_2O-Emissionen von 242 ± 7.4 µg N_2O-N m^{-2} h^{-1} und 140.8 ± 5.1 µg N_2O-N m^{-2} h^{-1} wurden während der Regenzeit 2000 an den Tieflandregenwald-Standorten Bellenden Ker bzw. Pin Gin Hill gemessen. Dabei wurden Flussraten >500 µg N_2O-N m^{-2} h^{-1} beobachtet. Diese Werte liegen im Bereich der höchsten aller bisher für tropische Waldökosysteme angegebenen N_2O-Flussraten (siehe z.B. Breuer et al., 2000). Die mittlere Höhe der N_2O-Emissionen am Bergregenwald-Standort Kauri Creek war mit 80.8 ± 3.3 µg N_2O-N m^{-2} h^{-1} etwa um den Faktor 2 niedriger als die N_2O-Emissionen an den beiden Tieflandstandorte Bellenden Ker und Pin Gin Hill (Kiese und Butterbach-Bahl, 2002 [**I**]). Während der Trockenzeit, wenn die mikrobiellen N- und C-Umsetzungsprozesse durch niedrige Werte der Bodenfeuchten (< 30 % wassergefüllter Porenraum (WFPS)) limitiert sind (Linn und Doran, 1984; Davidson et al., 1993; Kiese und

Butterbach, 2002 [**I**]; Kiese et al., 2003 [**III**]), sinkt das N_2O-Emissionsniveau an allen untersuchten Standorten auf Werte zwischen 5 und 20 µg N_2O-N m^{-2} h^{-1} ab (Kiese und Butterbach-Bahl, 2002 [**I**]; Kiese et al., 2003 [**III**]). Die Höhe der N_2O-Emissionen ist damit im Vergleich zur Höhe der N_2O Emissionen in der Regenzeit, je nach Standort, um das 8 bis 20-fache niedriger. Dieser saisonale Verlauf der N_2O-Emissionen mit signifikant niedrigeren Werten in der Trockenzeit wurde auch für andere tropische Regenwaldökosysteme beobachtet (Puerto Rico: Steudler et al., 1991; Mexiko: Garcia Méndez et al., 1991; Zentral–Afrika: Serca et al., 1994; Amazonien: Verchot et al., 1999, Melillo et al., 2001). Eine explizite Untersuchung des Effektes der Änderungen der Bodenfeuchte auf die Höhe der N_2O-Emissionen mittels einer Korrelationsanalyse erbrachte einen positiven linearen Zusammenhang (r = 0.58 – 0.74) für die Messdaten aus der Trockenzeit/ Übergangszeit (Oktober 2000 – Januar 2001), jedoch nur niedrige Pearson-Korrelationskoeffizienten für die Messdaten aus der Regenzeit (Kiese und Butterbach-Bahl, 2002 [**I**]). Wurden für die Korrelationsanalyse ausschließlich Werte <50-60% WFPS berücksichtigt, so konnte auch für die Regenzeit ein positiver linearer Zusammenhang zwischen N_2O-Emissionen und der Bodenfeuchte festgestellt werden. Den Effekt deutlich reduzierter N_2O-Emissionen bei hohen Wassergehalten konnten auch Riley und Vitousek (1995) für einige Bergregenwald-Standorte auf Hawaii beobachten. Linn und Doran (1984) zeigten, dass der Prozess der Nitrifikation bei Werten des WFPS von 50-60 % ein Optimum besitzt, da die Nitrifikation bis zu diesem Bereich noch durch eine zunehmende Substratverfügbarkeit mit höheren Bodenfeuchtegehalten profitiert, jedoch noch nicht durch eine herabgesetzte O_2-Verfügbarkeit, aufgrund des verminderten Bodengasaustausches mit der Atmosphäre, limitiert ist. Daraus lässt sich schließen, dass die Nitrifikation überwiegend unter 50-60 % WFPS und die Denitrifikation hauptsächlich über 50-60 % WFPS die Höhe der N_2O-Emissionen an den untersuchten Standorten dominiert. Ein Optimum der nitrifikatorischen N-Umsetzungsprozesse bei 50-60 % WFPS sowie eine zunehmende Bedeutung der Denitrifikation für die N_2O-Produktion in Böden bei Wassergehalten über 60 % WFPS wurde auch von Davidson (1991) beobachtet. Korrelationsanalysen zwischen der Bodentemperatur und der Höhe der in-situ N_2O-Emissionen erbrachten keinen signifikanten Zusammenhang. Da der Temperatureffekt auf die Höhe der N_2O-Emissionen bei Freilandmessungen nicht isoliert erfasst werden kann, sondern beispielsweise mit Änderungen der Bodenfeuchte (siehe oben) oder der Substratverfügbarkeit überlagert ist, wurde der Zusammenhang zwischen der N_2O-Produktion und der Temperatur in einem Laborexperiment zusätzlich überprüft (Kiese et al., 2002 [**I**]). Dabei wurden die Böden der drei Standorte für gleiche Wasserhaltekapazitäten jedoch unterschiedliche Temperaturen inkubiert. Nur für den Standort Pin Gin Hill, nicht jedoch für die Standorte Bellenden Ker und Kauri Creek konnte eine statistisch signifikante Zunahme der N_2O-Produktion im Temperaturbereich von 13 - 25°C festgestellt werden. In diesem Zusammenhang muss jedoch darauf hingewiesen werden, dass der Wassergehalt des tonreicheren Bodens des Standortes Pin Gin Hill (34.2% WFPS) (Tab. 3) wegen der höheren Wasserhaltekapazität signifikant höher war als der Wassergehalt der Bodenproben der Standorte Bellenden Ker (24.5% WFPS) und Kauri Creek (22.5% WFPS). Unter der Voraussetzung der aufgestellten These einer Dominanz der N_2O-Produktion durch die Nitrifikation bei Wassergehalten <50-60% würde dies bedeuten, dass eine Temperaturabhängigkeit der Nitrifikation erst ab moderaten Werten der Bodenfeuchte auftritt, darunter die Nitrifikation jedoch mehr durch die Bodenfeuchte limitiert ist. Diesen Zusammenhang konnten auch Breuer et al. (2000) in einer Literaturzusammenstellung von Brutto-Nitrifikationsraten in verschiedenen Waldökosysteme weltweit beobachten. Sie geben den Hinweise, dass die Nitrifikation in temperaten Waldböden durch niedrige Temperaturen, jedoch in Böden wärmerer Klimate eher durch niedrige Bodenwassergehalte limitiert wird.

Kein statistisch abzusichernder Zusammenhang konnte zwischen der Höhe der N_2O-Flussraten und den Ammonium- (NH_4^+) bzw. Nitrat- (NO_3^-) Gehalten im Boden (0-5cm) festgestellt werden (Kiese et al., 2003 [**III**]). Dies ist in Übereinstimmung mit den Arbeiten anderer Arbeitsgruppen an Standorten temperater wie auch tropischer Waldökosysteme (Keller und Reiners, 1994; Papen und Butterbach-Bahl, 1999; Verchot et al., 1999; Breuer et al., 2000).

Desweiteren konnte im Rahmen dieser Arbeit gezeigt werden, dass die Jahressumme der N_2O-Emissionen der Bergregenwälder der Atherton Tablelands generell niedriger sind (1.15-5.36 kg N_2O-N ha^{-1} $Jahr^{-1}$) als die N_2O-Emissionen der Tieflandregenwald-Standorte (6.89-7.45 kg N_2O-N ha^{-1} $Jahr^{-1}$) (Breuer et al., 2000; Kiese und Butterbach-Bahl, 2002 [**I**]). Dies ist in erster Linie auf die klimatischen Einflussgrößen Niederschlag und Temperatur zurückzuführen. Feuchtere und wärmere Verhältnisse, verursacht durch etwa doppelt so hohe Jahresniederschläge und eine etwa um 3.5 °C höhere mittlere Jahrestemperatur im küstennahen Tiefland (Tab. 3), begünstigen dort die Mineralisation, die mikrobiellen N-Umsetzungsprozesse sowie die daran gekoppelte Freisetzung von N_2O (Kiese und Butterbach-Bahl, 2002 [**I**]).

Ebenso tragen engere C/N-Verhältnisse der organischen Substanz zu einer Förderung des C- und N-Umsatzes und damit auch konsequenterweise zu einer Erhöhung der N_2O-Spurengasemissionen bei. So konnten Breuer et al. (2000) für Regenwaldstandorte mittels einer multiplen Regressionsanalyse nachweisen, dass das C/N-Verhältnis des Oberbodens eine wesentliche Größe war, die die Höhe der N_2O-Emissionen modulierte. Dieser Zusammenhang konnte im Rahmen der vorliegenden Arbeit bestätigt werden. Eine Korrelationsanalyse zwischen der Höhe der jährlichen N_2O-Emissionen und dem C/N-Verhältnis des Oberbodens (0-10 cm) aller im Rahmen dieser Arbeit untersuchten Standorte, ergänzt durch Daten von zwei weiteren Standorten innerhalb des Gebiets der Wet Tropics (Breuer et al., 2000), zeigte einen stark negativen Zusammenhang der betrachteten Größen (r = 0.89, P<0.01) (Kiese und Butterbach-Bahl, 2002 [**I**]). Für die Tieflandstandorte wurden dabei sowohl die höchsten N_2O-Emissionen als auch die engsten C/N-Verhältnisse gemessen.

Ein weiterer Faktor, der bei der Erklärung standörtlicher Unterschiede in der Höhe der N_2O-Emissionen berücksichtigt werden muss, sind die im Vergleich zum Bergregenwald-Standort Kauri Creek signifikant niedrigeren pH-Werte der Böden der Tieflandstandorte Bellenden Ker und Pin Gin Hill (Tab. 3). So wird in verschiedenen Publikationen darauf hingewiesen, dass bei niedriger werdenden pH-Werten die Aktivität der N_2O-Reduktase - das letzte Enzym in der Reihe von N-Transformationen innerhalb des Prozesses der Denitrifikation - abnimmt, was zu einer Verschiebung des N_2/N_2O Bildungsverhältnisses zugunsten von N_2O führt (Weier und Gillam, 1986; Granli und Bockman, 1994). Dies ist insbesondere von Bedeutung, da die Böden der niederschlagsreicheren Standorte des Tieflandes eher zur Denitrifikation neigen als die Böden der trockeneren Standorte auf den Atherton Tablelands (Kiese et al., 2002 [**II**]). Dass auch bei der Nitrifikation niedrigere pH-Werte die Produktion von N_2O fördern können (Martikainen und De Boer, 1993; Sitaula and Bakken, 1993), könnte eine weitere Erklärung der höheren N_2O-Freisetzung im Tiefland sein.

Begleitend zu den Messungen der N_2O-Emissionen wurde in mindestens wöchentlichen Abständen die N_2O-Konzentration in verschiedenen Bodentiefen erfasst. Dazu wurden Sonden in verschiedenen Bodenhorizonten (5, 10, 15 und 30 cm) eingebracht, mit gasdicht zu verschließenden Spritzen beprobt und die N_2O-Konzentrationen direkt am Gaschromatographen im Gelände bestimmt (Kiese und Butterbach-Bahl, 2002 [**I**], Kiese et al., 2003 [**III**]). Im Gegensatz zu den zeitlich versetzten Messungen der N_2O-Emissionen war es damit möglich, die N_2O-Konzentrationsverteilung innerhalb des Bodenprofils zeitgleich an allen drei Untersuchungsstandorten zu bestimmen. Die bereits erwähnten saisonalen Unterschiede der N_2O-Emissionen konnten auch durch die N_2O-Konzentrationsmessungen belegt werden. Der starke positive Zusammenhang zwischen der Höhe der N_2O-Emissionen und der N_2O-Konzentrationen im Boden spiegelt sich auch in den hohen Werten des Bestimmtheitsmaßes (r^2) von 0.73 – 0.82 (P<0.01) einer linearen Anpassung zwischen diesen

Bezugsgrößen wider (Kiese und Butterbach-Bahl, 2002 [**I**]; Kiese et al., 2003 [**III**]). Auch die Einschätzung standörtlicher Unterschiede mit höchsten saisonal gewichteten N_2O-Jahresemissionen an den Tieflandregenwald-Standorten Bellenden Ker (7.45 kg N_2O-N ha^{-1} $Jahr^{-1}$) und Pin Gin Hill (6.89 kg N_2O-N ha^{-1} $Jahr^{-1}$) sowie einer deutlich niedrigeren mittleren jährlichen N_2O-Emission am Bergregenwald-Standort Kauri Creek (4.36 kg N_2O-N ha^{-1} $Jahr^{-1}$) für das Jahr 2000 konnten durch ein signifikant verschiedenes N_2O-Konzentrations-Niveau im Boden der einzelnen Standorte in der Reihenfolge Bellenden Ker > Pin Gin Hill > Kauri Creek unterstützt werden (Kiese und Butterbach-Bahl, 2002 [**I**]).

Ein Vergleich der relativ hohen Jahressumme der N_2O-Emissionen am Standort Bellenden Ker von 7.45 kg N_2O-N ha^{-1} $Jahr^{-1}$ im Jahr 2000 mit dem Wert der jährlichen N_2O-Emission von 0.97 kg N_2O-N ha^{-1} $Jahr^{-1}$ für die Periode 01.11.2001 – 31.12.2002 unterstreicht, dass die N_2O-Quellstärke einzelner Standorten in verschiedenen Jahren große Schwankungen aufweisen kann. Dieser Unterschied ist hauptsächlich auf die annuelle Variabilität des Niederschlags und den damit verbundenen unterschiedlichen Bodenfeuchtebedingungen einzelner Jahre zurückzuführen. Aufgrund der feuchtelimitierten Produktion von N_2O bei niedrigen Wassergehalten (siehe oben) ist in trockenen Jahren eine insgesamt niedrigere N_2O-Freisetzung zu erwarten als in feuchten Jahren. So war auch am Standort Bellenden Ker die Zeitspanne von November 2001 bis Oktober 2002 mit 2678.2 mm Niederschlag eine sehr trockene Periode, während im Jahr 2000 mit einem Jahresniederschlag von 6649.0 mm sehr feuchte, die N_2O-Freisetzung fördernde Verhältnisse vorlagen (vgl. dazu langjähriges Jahresniederschlagsmittel am Standort Bellenden Ker 4395.0 mm, Tab. 3).

Neben den standörtlichen und saisonalen Variabilitäten der N_2O-Emissionen tritt demnach zusätzlich noch eine Variabilität der N_2O-Emissionen zwischen klimatisch unterschiedlichen Jahren auf, wie sie bisher nur für Wälder temperater Breiten beschrieben wurde (Butterbach-Bahl et al., 2002). Die durchgeführten Messungen belegen somit die hohe Komplexität des Wirkungsgefüges, das letztlich für die Höhe der N_2O-Emissionen verantwortlich ist und es so schwierig macht die Quellstärke tropischer Regenwälder genau zu erfassen und regional abzuschätzen. Nicht zuletzt deshalb sollte zukünftig der Fokus auf die Entwicklung neuer Methoden, beispielsweise der Anwendung von mechanistischen Modellen zur Simulation der N_2O-Emissionen aus tropischen Regenwaldökosystemen gelegt werden (siehe Kapitel 5). Diese haben nicht nur das Potential die komplexen Bildungsmechanismen von N_2O in Böden integrativ zu berücksichtigen, sondern können auch zur Simulation von N_2O-Emissionen für klimatisch unterschiedliche Jahre verwendet werden (Butterbach-Bahl et al., 2001).

3.3. Zeitliche und räumliche Variabilität der CO_2-Emissionen

Begleitend zu den N_2O-Emissionen wurde die Variabilität der CO_2-Emissionen an verschiedenen Standorten als Indikator der Mineralisationsdynamik und damit für ein besseres Verständnis der mikrobiellen C- und N-Umsetzungen erfasst. Die gemessenen CO_2-Flussraten schwankten zwischen Werten von 24.0 und- 247.7 mg C m^{-2} h^{-1} und lagen damit im Bereich anderer für tropische Regenwälder bestimmten CO_2-Emissionsraten (Holt et al., 1990, Maggs und Hewett, 1990; La Scala et al., 2000). Im Vergleich zu den N_2O-Emissionen waren die Unterschiede in den CO_2-Emissionen sowohl zwischen den Standorten als auch zwischen der Regenzeit und der Trockenzeit weniger stark ausgeprägt (Kiese und Butterbach-Bahl, 2002 [**I**]). Während für das N_2O prinzipiell höchste Emissionen an den Standorten im küstennahen Tiefland (Bellenden Ker und Pin Gin Hill) gemessen wurden, ergab sich für die CO_2-Emissionen kein entsprechend klares Muster. Höchste CO_2-Emissionen mit Werten >250 mg C m^{-2} h^{-1} zeigte der Bergregenwald-Standort Kauri Creek im Januar 2000 während einer Periode mit häufigem Wechsel zwischen trockenen Phasen und Phasen mit starken Niederschlägen. Hohe CO_2-Emissionen in Folge einer Stimulierung der C-Mineralisierung, respektive Bodenrespiration, während Zyklen mit Trocknungs- und Wiederbenässungsphasen wurden bereits in anderen Untersuchungen beobachtet (Sørensen, 1974; Vangestel et al., 1993). Am Standort Pin Gin Hill wurden sowohl zur Regen- als auch Trockenzeit die niedrigsten CO_2-Emissionen gemessen (< 100 mg C m^{-2} h^{-1}). Dies könnte mit dem im Vergleich zu den anderen Standorten signifikant höheren Tongehalt von 67% zusammenhängen (Tab. 3). Wie Van Veen und Kuikman (1990) sowie Koutika et al. (1999) für Böden im Amazonas-Becken zeigten, kann durch hohe Tongehalte ein Teil der organischen Substanz vor der Mineralisation geschützt werden, was die niedrigeren CO_2-Emissionen an diesem Standort erklären könnte. Das Argument verminderter Mineralisierungsraten und der damit verbundenen geringeren Substratverfügbarkeit für die mikrobiellen N-Umsetzungsprozesse Nitrifikation und Denitrifikation am Tieflandstandort Pin Gin Hill, kann ebenfalls zur Klärung der niedrigeren N_2O-Emissionen verglichen zum klimatisch ähnlichen Tieflandstandort Bellenden Ker beitragen (siehe oben).

Der Effekt des Bodenwassergehaltes auf die Höhe der CO_2-Emissionen hängt stark von den Feuchteverhältnissen des Untersuchungszeitraums ab und ist daher differenziert zu betrachten. Während Perioden mit moderaten Bodenwassergehalten (überwiegend Trockenzeit) konnte ein positiver Zusammenhang zwischen der Höhe der CO_2-Emissionen und steigenden Wassergehalten festgestellt werden. Über Werten von 50-60% WFPS kehrte sich dieser Trend jedoch um, so dass während der Regenzeit an allen Standorten negative

Korrelationen zwischen den CO_2-Emissionen und Werten des WFPS auftraten (Kiese und Butterbach-Bahl, 2002 [**I**]). Dieser Zusammenhang, den auch Davidson et al. (1998) für temperate Wälder beobachten konnten, unterstützt die bereits für die N_2O-Emissionen aufgestellte These einer limitierten Sauerstoffverfügbarkeit im Boden bei hohen Wassergehalten, die zu einer Minderung der heterotrophen Respiration bzw. in Bezug auf den Stickstoffkreislauf zu einer Förderung der Denitrifikation führt (Kiese und Butterbach-Bahl, 2002 [**I**]; Kiese et al., 2002 [**II**]). Dies kommt ebenso in den höheren negativen Korrelationskoeffizienten zwischen dem Bodenwassergehalt und der Höhe der CO_2-Emissionen (Pin Gin Hill: -0.71; Bellenden Ker: -0.61) der klimatisch feuchteren Tieflandstandorte gegenüber dem trockeneren Standort auf den Atherton Tablelands (Kauri Creek: -0.41) zum Ausdruck (Tab. 3).

Ein statistisch signifikanter positiver oder negativer Zusammenhang zwischen den in-situ-CO_2-Emissionen und der Bodentemperatur konnte analog zu den N_2O-Emissionen nicht festgestellt werden. Dies ist einerseits darauf zurückzuführen, dass die Tagesamplitude der Bodentemperatur in 1 cm Bodentiefe mit < 2°C sehr gering war und andererseits ein Überlagerungseffekt durch den Einfluss der Bodenfeuchte auf die CO_2-Emissionen im Freiland nicht auszuschließen war. Deswegen wurde in einem Inkubationsversuch im Labor unter konstanten Werten des Bodenwassergehalts die CO_2-Produktion im Temperaturbereich von 13 bis 32°C untersucht. Im Gegensatz zum N_2O, bei dem nur eine signifikante Zunahme der N_2O-Produktion mit höheren Temperaturen für Bodenproben des Standorts Pin Gin Hill nachgewiesen werden konnte, ergab sich für das CO_2 für Bodenproben aller drei untersuchten Standorte eine signifikante Zunahme der CO_2-Produktion im Temperaturbereich von 20 bis 32°C. Die berechneten Q_{10} Werte für den Temperaturbereich von 20 – 30 °C waren 3.0 für die Bodenproben des Bergregenwald-Standorts Kauri Creek sowie 3.6 und 5.6 für die Tieflandregenwald-Standorte Bellenden Ker und Pin Gin Hill (Kiese und Butterbach-Bahl, 2002 [**I**]). Daraus kann gefolgert werden, dass die Mineralisation in Böden der im Rahmen dieser Arbeit untersuchten Standorte weniger feuchtelimitiert ist als die Nitrifikation. Außerdem lässt sich aus der Übertragung der Laborergebnisse auf die Freilandbedingungen, zumindest bei Wassergehalten <50-60% WFPS, auf höhere Mineralisationsraten in den Böden der durchschnittlich 3.5°C wärmeren Tieflandregenwälder gegenüber den Böden der Bergregenwälder auf den Atherton Tablelands schließen. Die hieraus resultierende höhere Substratverfügbarkeit für die Nitrifikation und die Denitrifikation in den Tieflandböden könnte eine weitere Erklärung für die höhere N_2O-Freisetzung der Tieflandregenwälder im Vergleich zu den Bergregenwäldern sein.

3.4. Zeitliche Variabilität der CH_4-Emissionen

Im Gegensatz zur Datenlage für temperate Waldökosysteme, die mit Ausnahme ständig oder teilweise gefluteter Sumpf- und Moorwälder eine Senke für atmosphärisches Methan darstellen (Harriss et al., 1981; Steudler et al., 1989; Crill, 1991; Castro et al., 1995; Sitaula et al., 1995; Whalen und Reeburgh, 2000; Steinkamp et al., 2001; Butterbach-Bahl und Papen, 2002), sind Untersuchungen, die sich mit dem CH_4-Austausch zwischen tropischen Waldökosystemen und der Atmosphäre beschäftigen, sehr rar. Die bislang verfügbaren Datensätze zeigen, dass gut durchlüftete tropische Regenwaldböden als CH_4-Senken fungieren können, jedoch unter gesättigten Bodenwasserverhältnissen, wie z.B. während der Regenzeit, auch signifikante Quellen von CH_4 darstellen können (Keller et al., 1983, 1986; Devol et al., 1990; Delmas et al., 1992; Tathy et al., 1992; Wassmann et al., 1992; Keller und Reiners, 1994; Steudler et al., 1996). Jedoch basieren diese Ergebnisse meist auf sporadischen Messungen, was entweder die Länge des Messzeitraums (einzelne Tage), oder innerhalb längerer Untersuchungen den gewählten Messzeitschritt (Monatswerte) betrifft. Anhand der erstmaligen Erfassung eines kompletten Jahresganges des zeitlich hoch aufgelösten CH_4-Austausches zwischen einem australischen Tieflandregenwald-Standort (Bellenden Ker) und der Atmosphäre sollte dazu beigetragen werden, die Dynamik des CH_4-Austauschs in tropischen Regenwaldgebieten besser zu verstehen. Die Ergebnisse zeigten, dass der Boden während des gesamten Untersuchungszeitraums von 01.11.2001 bis 31.10.2002 am Standort Bellenden Ker eine Senke für atmosphärisches Methan war (Kiese et al., 2003 [**III**]). Wie schon für die Spurengase N_2O und CO_2 beobachtet, unterlag die CH_4-Aufnahme im Boden durch methanotrophe Bakterien einem saisonalen Trend mit höchsten CH_4-Aufnahmeraten von bis zu –65 µg CH_4 m^{-2} h^{-1} in der Trockenzeit und signifikant niedrigeren CH_4-Aufnahmeraten (Minimum: -5.0 µg CH_4 m^{-2} h^{-1}) in der Regenzeit. Besonders hervorzuheben ist, dass selbst während starker Niederschlagsperioden (z.B. Mai 2002) der Boden ausschließlich als Senke für CH_4 fungierte. Der enge Zusammenhang zwischen Änderungen in der Bodenfeuchte und der Höhe der CH_4-Aufnahme konnte durch eine negative lineare Regression ($r^2 = 0.54$) beschrieben werden (Kiese et al., 2003 [**III**]). Analog zu den Prozessen der Respiration, Nitrifikation und Denitrifikation, deren Raten über die Höhe der N_2O und CO_2-Freisetzung aus Böden entscheiden, stellt der Bodenwassergehalt damit auch eine wichtige Einflussgröße auf den Prozess der Methanoxidation dar. Wegen des stark herbabgesetzten Diffusionskoeffizienten von Gasen in Wasser gegenüber Luft ist deren Permeabilität im Boden stark von der Höhe des Bodenwassergehaltes abhängig. Insbesondere durch die daraus resultierende Beeinflussung der Sauerstoffverteilung im Boden und der damit verbundenen Höhe des Redoxpotentials bestimmt der Wassergehalt indirekt auf Prozessebene, welche biogeochemischen Reaktionen ablaufen können. Die parallel erfassten CH_4-Konzentrationen bis in eine Bodentiefe von 30 cm lagen stets unter der atmosphärischen Methankonzentration von 1745 ppmv, was die gute Wasserleitfähigkeit und Durchlüftung des Bodens am Messstandort Bellenden verdeutlicht. Daraus lässt sich schließen, dass selbst in tieferen Bodenschichten überwiegend Methan konsumiert wird, während die Methanproduktion durch methanogene Bakterien die an strikt anoxische Verhältnisse geknüpft ist, zumindest im Rahmen des gemessenen Jahresgangs (November 2001 bis Oktober 2002) unbedeutend ist.

Im Gegensatz zur signifikanten positiven Korrelation zwischen der Höhe der N_2O-Emissionen und der N_2O-Konzentrationen im Boden konnte für das Methan kein signifikanter Zusammenhang zwischen Konzentrationen und Emissionen festgestellt werden (Kiese und Butterbach-Bahl, 2000 [**I**]; Kiese et al., 2003 [**III**].

Auf Basis von 348 Tagesmittelwerten konnte die jährliche CH_4-Aufnahme im Boden des Tieflandregenwald-Standortes Bellenden Ker durch messtechnisch hohen Aufwand relativ genau quantifiziert werden. Die Aufnahmerate von 3.21 kg CH_4 ha^{-1} $Jahr^{-1}$ liegt in der Größenordnung jährlicher Aufnahmeraten von CH_4, die für Böden temperater Wälder bestimmt wurden (Borken und Brumme, 1997; Steinkamp et al., 2001). Unter Berücksichtigung des vergleichsweise geringen Niederschlagaufkommens im Messzeitraum November 2001 bis Oktober 2002 lässt sich schließen, dass tropische Regenwaldböden, zumindest in niederschlagsarmen Jahren eine bisher unterschätzte bzw. nicht berücksichtigte Senke für atmosphärisches Methan darstellen.

4. Erfassung und Bedeutung der Nitrifikation für die N_2O-Emission aus Böden tropischer Regenwälder

Neben der Mineralisation ist die Nitrifikation ein weiterer zentraler mikrobieller Prozess, der den N-Umsatz in Böden von Wäldern wesentlich bestimmt. Abgesehen von der Denitrifikation ist die Nitrifikation zudem entscheidend an der Freisetzung von N-Spurengasen beteiligt (Abb. 1). Die Bedeutung der Nitrifikation ist dabei nicht nur auf die direkte Produktion der primär bzw. sekundär treibhauswirksamen Spurengase N_2O und NO beschränkt, sondern liegt auch gerade darin, dass durch sie Nitrat, das Ausgangssubstrat für die Denitrifikation, gebildet wird. Trotz der wichtigen Rolle dieses Prozesses für den N-Umsatz in Waldböden liegen aufgrund methodischer Schwierigkeiten nur in begrenztem Umfang Informationen zur Höhe der in-situ Nitrifikationsraten, d.h. der Brutto-Nitrifikationsraten in Böden vor. Die einfachere Bestimmung der Netto-Nitrifikation, bei der über die zeitliche Änderung des Nitrat-Pools im Boden auf die Höhe der N-Umsetzungsraten in Böden geschlossen wird, ist jedoch um Größenordnungen kleiner als die tatsächlich ablaufende Brutto-Nitrifikation (Davidson et al., 1992; Stark und Hart, 1997; Verchot et al., 2001). Die damit verbundene Unterschätzung der wirklichen Höhe und Dynamik der N-Umsetzungen in Böden ist gerade im Hinblick auf die Freisetzung von N-Spurengasen und die (Weiter-)Entwicklung von Modellen zur Simulation der Emissionen von N-Spurengasen als problematisch anzusehen. Das herkömmlich eingesetzte Verfahren zur Bestimmung von Brutto-Nitrifikationsraten ist die ^{15}N pool dilution Methode. Bei dieser Methode wird der Nitratpool des Bodens mit $^{15}NO_3^-$ markiert und die zeitliche Veränderung der Isotopenzusammensetzung des ^{15}N markierten Nitratpools verfolgt. Geht man davon aus, dass im Zuge der Nitrat verbrauchenden Prozesse im Boden, wie die mikrobielle Immobilisierung von Nitrat oder die Denitrifikation keine Isotopendiskriminierung stattfindet, so führt die Nitrifikation durch die Nachlieferung von $^{14}NO_3^-$ zu einer Verringerung des $^{15}N/^{14}N$-Isotopenverhältnisses des Nitratpools. Aus der Kinetik der Verdünnung kann somit direkt auf die Höhe der Brutto-Nitrifikation geschlossen werden (Kirkham und Bartholomew, 1954; Barraclough et al., 1985; Barraclough et al., 1995). Da diese Methode arbeitsintensiv und zudem technisch sehr aufwendig ist, liegen bisher nur sehr wenige Untersuchungen von Brutto-Nitrifikationsraten, insbesondere aber aus tropischen Ökosystemen vor. Wesentlich einfacher ist hingegen die Bestimmung von Brutto-Nitrifikationsraten mit der neu entwickelten Methode der Barometrischen-Prozesssepararation (BaPS), was speziell für tropische Regionen aufgrund oft limitierter infrastruktureller Ausstattung von Bedeutung ist.

Die Methodik basiert auf einem einfachen Massenbilanzverfahren (Gl. 1), bei dem über die Konzentrationsänderungen von Kohlendioxid und Sauerstoff und der simultanen Erfassung der Gesamtdruckänderung mit der Zeit in einem gasdichten, isothermal inkubierten System, die Höhe der Brutto-Nitrifikationsrate berechnet werden kann (Ingwersen et al., 1999).

$$\left(\frac{\Delta n}{\Delta t}\right)=\left(\frac{\Delta O_2}{\Delta t}\right)+\left(\frac{\Delta CO_2}{\Delta t}\right)+\left(\frac{\Delta N_x O_y}{\Delta t}\right) \tag{1}$$

In gut durchlüfteten Böden sind die Nitrifikation, die Denitrifikation und die Respiration sowie die physiko-chemische Lösung von CO_2 im Bodenwasser die quantitativ bedeutsamen Prozesse, welche wesentlich die Gasbilanz im System beeinflussen und somit die beobachtete Gesamtdruckänderung bewirken. Die Nitrifikation, bei der durch Oxidation von NH_4^+ zu NO_3^- molekularer Sauerstoff aus der Umgebungsluft verbraucht wird, wirkt sich dabei druckmindernd, die Denitrifikation durch Produktion von CO_2 und N-Gasen druckerhöhend im System aus, während die Respiration, für die ein Respirationskoeffizient von 1.0 angenommen wird (Ingwersen et al., 1999), als druckneutral anzusehen ist. Da sowohl die Gesamtdruckänderung (?n) als auch die Änderungen der Partialdrucke von CO_2 (?CO_2) und O_2 (?O_2) gemessen werden, kann die Höhe der denitrifikatorisch gebildeten N-Gase (?N_xO_y) als einziger unbekannter Term durch Auflösung der Grundgleichung (1) berechnet werden. Aus dieser Größe lassen sich durch inverses Bilanzieren die CO_2-Produktion durch die Denitrifikation, die CO_2-Freisetzung und der O_2-Verbrauch durch die Respiration, und daraus der O_2-Verbrauch durch die Nitrifikation erfassen. Aus dem O_2-Verbrauch der Nitrifikation kann schließlich stöchiometrisch auf die Höhe der Brutto-Nitrifikation geschlossen werden (Details siehe Ingwersen et al., 1999).

Da die BaPS bisher nur für Böden temperater Waldökosysteme validiert wurde (Ingwersen et al., 1999; Heidenfelder, 2002), diese Methode jedoch im Rahmen dieser Arbeit auf die Erfassung von Brutto-Nitrifikationsraten von tropischen Regenwaldböden angewendet werden sollte, wurden vorab Validierungsexperimente der BaPS mit der ^{15}N pool dilution Methode durchgeführt (Kiese et al., 2002 [**II**]). Dazu wurden Bodenproben des Bergregenwald-Standortes Kauri Creek während des Übergangs Trocken/ Regenzeit 2000 und Bodenproben des Tieflandregenwald-Standortes Bellenden Ker zum Ende der Trockenzeit 2001 verwendet. Die Validierungsexperimente ergaben, dass keine signifikanten Unterschiede in der Höhe der Brutto-Nitrifikationsraten zwischen der BaPS-Methode und der ^{15}N pool dilution Methode nachweisbar waren. Mit der Methode der Barometrischen Prozessseparation liegt damit eine relativ einfach handhabbare Technik zur Erfassung von

Brutto-Nitrifikationsraten vor, die es ermöglicht, die bisher äußerst spärliche Datenlage zur Höhe der nitrifikatorischen N-Umsetzungen, insbesondere in tropischen Waldökosystemen, wesentlich zu verbessern.

Die durchgeführten Messungen zur Bestimmung der Brutto-Nitrifikationsraten in intakten Bodenkernen eines Bergregenwald-Standorts auf den Atherton Tablelands (Kauri Creek) und eines Tieflandregenwaldes (Bellenden Ker) ergaben Raten im Bereich von 8.4 – 24.0 mg N kg BTG^{-1} Tag^{-1}. Diese Raten bewegen sich in einer vergleichbaren Größenordnung wie die von Davidson et al. (1993) für einen saisonalen Trockenwald in Mexiko veröffentlichten Brutto-Nitrifikationsraten. Im Vergleich zu publizierten Brutto-Nitrifikationsraten anderer Regenwaldböden weltweit (Zou et al,. 1992; Riley und Vitousek, 1995; Neill et al., 1999) lagen die für den Standort Bellenden Ker und Kauri Creek bestimmten Brutto-Nitrifikationsraten jedoch im oberen Bereich veröffentlichter Werte (Kiese et al., 2002 [**II**]). Zum ersten mal konnte im Rahmen der vorliegenden Arbeit nachgewiesen werden, dass die nitrifikatorischen N-Umsetzungen in tropischen Regenwäldern einen ausgeprägten saisonalen Verlauf aufweisen. Höchste Raten der Brutto-Nitrifikation wurden während des Übergangs von der Trocken- zur Regenzeit (Bergregenwald: 24.0 mg N kg BTG^{-1} Tag^{-1}; Tieflandregenwald: 13.1 mg N kg BTG^{-1} Tag^{-1}) beobachtet, während signifikant niedrigere Raten sowohl in der Regen- als auch in der Trockenzeit auftraten (Kiese et al., 2002 [**II**]). Die Brutto-Nitrifikationsraten des Bergregenwald-Standortes waren dabei zu allen hygrischen Zeiten signifikant höher als die Raten des Tieflandregenwald-Standortes. Die von Breuer et al. (2002) für andere Bergregenwald-Standorte der Wet Tropics gezeigte positive Korrelation zwischen der Höhe der Brutto-Nitrifikationsraten und der Höhe der N_2O-Emissionen ist konträr zu dieser Beobachtung, da allgemein höhere N_2O-Emissionen am Tieflandstandort Bellenden Ker gemessen wurden (Kiese und Butterbach-Bahl, 2002 [**I**]). Dies lässt auf prinzipielle Unterschiede der für die N_2O-Produktion verantwortlichen Prozesse der einzelnen Standorte schließen. Am Tieflandregenwald-Standort können die wesentlich höheren N_2O-Emissionen in Folge des deutlich feuchteren und wärmeren Klimas auf eine höhere Aktivität der Denitrifikation zurückgeführt werden, während am kühleren und trockeneren Bergregenwald-Standort die Nitrifikation die Freisetzung von N_2O dominiert (Kiese et al., 2002 [**II**]; Kiese et al., 2003 [**IV**]). Analysen zur Abundanz von am Stickstoffkreislauf beteiligten Mikroorganismen-Populationen unterstützen diese Interpretation. Ein Vergleich beider Standorte ergab höhere Zellzahlen der Denitrifizierer am Tieflandregenwald-Standort, während im Boden des Bergregenwaldes die Population der nitrifizierenden Mikroorganismen größer war (Kiese et al., 2002 [**II**]). Die erzielten Ergebnisse, insbesondere die klimatisch bedingten Unterschiede der die N_2O-Emissionen dominierenden Prozesse zwischen den Standorten, bildeten wichtige Grundlagen für die im Folgenden beschriebene Weiterentwicklung und Validierung eines mechanistischen Modells zur Simulation von N_2O-Emissionen aus tropischen Regenwaldökosystemen.

5. Modellierung von N_2O-Emissionen aus Böden tropischer Regenwälder

5.1. Weiterentwicklung des biogeochemischen Modells PnET-N-DNDC

Eine verbesserte Abschätzung der N_2O-Emissionen auf regionaler oder gar globaler Ebene kann aufgrund der komplexen Interaktionen der in die N_2O-Produktion/ -Konsumption involvierten Prozesse und der räumlichen und zeitlichen Variabilität der die Prozesse beeinflussenden biotischen und abiotischen Faktoren (Kapitel 3) nur durch die Entwicklung und Anwendung prozessorientierter Modelle erwartet werden. Im Rahmen dieser Arbeit sollte zur Simulation von N_2O-Emissionen aus Böden tropischer Waldökosysteme das PnET-N-DNDC Modell verwendet werden. Dieses Modell, welches ursprünglich zur Simulation der biogeochemischen Umsätze von Kohlenstoff und Stickstoff sowie des daran gekoppelten Biosphäre-Atmosphäre-Austauschs von C- und N-Spurengasen in temperaten Waldökosystemen entwickelt wurde (Li et al., 1992, 2000; Stange et al., 2000; Stange, 2001; Butterbach-Bahl et al., 2001), musste für seine Übertragbarkeit und Anwendbarkeit auf tropische Regenwaldökosysteme weiterentwickelt und adaptiert werden. Das Modell besteht prinzipiell aus zwei Modulen. Das erste Modul, das die Unter-Module zur Simulation des Bodenklimas, des Pflanzenwachstums und der Mineralisierung beinhaltet, simuliert basierend auf Eingabe-Daten zu Klima, Bodeneigenschaften, Bestand (Art und Alter) die Gradienten der Umweltfaktoren Temperatur, Bodenwassergehalt, Redoxpotential oder Konzentration von Substraten innerhalb des Bodenprofils. Das zweite Modul, das sich aus den Unter-Modulen Nitrifikation, Denitrifikation und Fermentation zusammensetzt, berechnet die Flüsse von N_2O, NO, N_2, NH_3, CO_2 und CH_4 sowohl innerhalb des Bodenprofils als auch zwischen der Pedosphäre und der Atmosphäre. Die Simulation der N- und C-Umsetzungen, einschließlich der resultierenden Spurengasproduktion, basiert dabei auf der Aktivität der an den entsprechenden Prozessen beteiligten Mikroorganismen, deren Höhe als Funktion der oben aufgeführten Umweltfaktoren bestimmt wird.

Da die prozessorientierte Simulation der N_2O-Emissionen in PnET-N-DNDC, wie oben beschrieben, auf bio- und geochemischen Funktionen und Algorithmen beruht, wurde angenommen, dass diese im wesentlichen auf andere Klimazonen und Bodentypen übertragbar sind (Kiese et al., 2003, [**IV**]). Trotzdem mussten für die Anwendung von PnET-N-DNDC zur Simulation von N_2O-Emissionen aus tropischen Regenwaldökosystemen einige Änderungen an den bestehenden Algorithmen durchgeführt werden. Dies umfasste die Implementierung eines Parametersatzes „Tropischer Regenwald“ zur Simulation der Physiologie und Phänologie tropischer Regenwaldökosysteme, die teilweise aus

Literaturdaten (Hunt et al., 1996) abgeleitet wurde, aber auch auf einer Vielzahl von eigenen Messungen, z.B. zum C/N-Verhältnis der Blätter, zur saisonalen Variabilität des LAI, zum Streufall und zur Masse der Streuauflage, beruhten (Kiese et al., 2003 [**III**]). Desweiteren wurden Modifikationen in der Simulation des Bodenwasserhaushaltes vorgenommen sowie ein Aktivitäts-Index für die Population der Denitrifizierer eingeführt, um die Simulation der Saisonalität (Regen-, Trockenzeit) der N-Umsetzungen in tropischen Regenwaldböden in Abhängigkeit des Bodenwassergehaltes zu verbessern (Kiese et al., 2003 [**IV**]). Da in PnET-N-DNDC die biologische N_2-Fixierung bisher nicht berücksichtigt ist, diese aber in tropischen Regenwaldökosystemen im Gegensatz zu temperaten Wäldern nicht vernachlässigt werden kann (Cleveland et al., 1999), musste ein Algorithmus zur Quantifizierung der N_2-Fixierung in den bestehenden Programmcode eingebaut werden. Basierend auf einer detaillierten Literaturrecherche von Cleveland et al. (1999) wurde dazu eine einfache lineare Regressionsgleichung verwendet, in der die Höhe der N_2-Fixierung als Funktion der Evapotranspiration berechnet wird (Kiese et al., 2003 [**IV**]). Abbildung 4 stellt schematisch einen Überblick über das weiterentwickelte PnET-N-DNDC Modell dar.

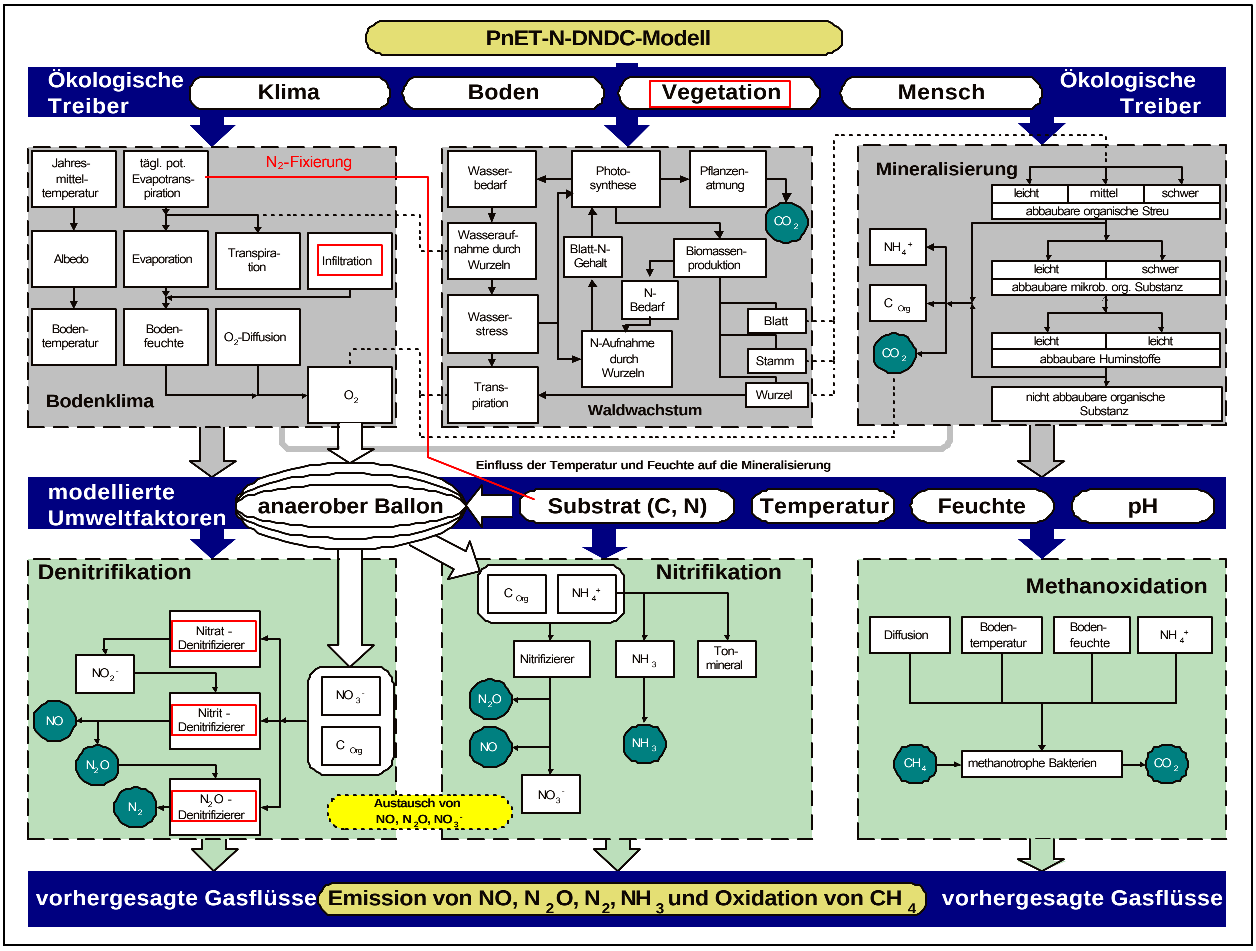


Abb. 4: Schematische Übersicht über das PnET-N-DNDC Modell. Die von Modifikationen betroffenen Modellteile sowie die Implementierung der N_2-Fixierung sind rot markiert.

5.2. Modell-Validierung

Um die Anwendbarkeit der neuen Version des PnET-N-DNDC Modells zur Simulation von N_2O-Emissionen aus tropischen Regenwaldböden bewerten zu können, wurde das Modell mit Datensätzen von sechs verschiedenen Standorten validiert (Kiese et al., 2003 [**IV**]). Vier der Standorte befinden sich im Gebiet der Wet Tropics, Australien (Breuer, 2000; Kiese et al., 2002 [**I**]; Kiese et al., 2003 [**III**]), zwei der Standorte in Costa Rica (Keller et al., 1993; Keller und Reiners, 1994; Liu et al., 2000). Die bewusste Beschränkung auf diese Standorte ergab sich daraus, dass für eine adäquate Modellvalidierung einerseits ausreichend lange Datensätze von N_2O-Emissionen benötigt werden, andererseits aber auch Klimadaten in täglicher Auflösung als Modelltreiber verfügbar sein müssen. Aus letzterem Grund konnten z.B. die N_2O-Datensätze von Verchot et al. (1999) und Melillo et al. (2001) für Standorte in Amazonien nicht zur Modellvalidierung verwendet werden. Die in der Validierung berücksichtigten Standorte deckten sowohl hinsichtlich der Bodeneigenschaften (z.B. Textur, pH-Wert, organischer Kohlenstoff) als auch der klimatischen Verhältnisse (Höhe der mittleren Jahrestemperatur und Jahresniederschlag) ein weites Spektrum ab (Kiese et al., 2003 [**IV**]). Die Ergebnisse aus den Untersuchungen zur Modellvalidierung zeigen, dass die modellierten N_2O-Emissionen den gemessenen saisonalen Verlauf der N_2O-Emissionen an allen Standorten sehr gut widerspiegelten. Das Modell PnET-N-DNDC war sogar in der Lage die hohe zeitliche Dynamik der N_2O-Emissionen in der Regenzeit, die hauptsächlich durch Änderungen in der Bodenfeuchte hervorgerufen werden, wiederzugeben. Einschränkungen hinsichtlich der Simulationsgüte ergeben sich jedoch für den sich vom 01.11.2001 – 31.10.2002 erstreckenden Datensatz der N_2O-Emissionen am Tieflandregenwald-Standort Bellenden Ker. Das PnET-N-DNDC Modell konnte zwar den beobachteten saisonalen Verlauf und die teilweise hohe Dynamik der gemessenen N_2O-Emissionen nachbilden, jedoch überschätzte das Modell insbesondere in der Regenzeit die gemessenen N_2O-Emissionen etwa um den Faktor 2 (Kiese et al., 2003 [**IV**]). Hingegen waren die Simulationsergebnisse für den gleichen Standort für die im Jahr 2000 beobachteten N_2O-Emissionen deutlich besser, so dass eine Diskrepanz von lediglich 25% zwischen der mittleren gemessenen und der mittleren simulierten N_2O-Emission festzustellen war. Die Modell-Effizienz nach Nash und Sutcliffe (1970) betrug $r^2 = 0.23$ (Kiese et al., 2003 [**IV**]). Eine sehr gute Übereinstimmung der simulierten mit den im Jahr 2000 verfügbaren gemessenen N_2O-Emissionen zeigte sich für den Bergregenwald-Standort Kauri Creek (Australien). Dabei ist anzumerken, dass das Modell PnET-N-DNDC die großen Unterschiede des N_2O-Emissionsniveaus zwischen Regen- (> 30 g N_2O-N ha^{-1} Tag^{-1}) und Trockenzeit (< 2 g N_2O-N ha^{-1} Tag^{-1}) relativ genau erfassen konnte, woraus die hohe Modell-Effizienz von $r^2 = 0.68$ resultierte. Die Güte der Modellvalidierungen mit Daten von N_2O-Emissionen anderer australischer Standorte (Lake Eacham und Massey Creek) war beachtlich, insbesondere in Hinblick auf die minimalen, maximalen und mittleren N_2O-Emissionen, aber auch in Hinblick auf den zeitlichen Verlauf der N_2O-Emissionen (Kiese et al., 2003 [**IV**]). Dieser Befund spiegelt sich auch in guten Werten für die Modell-Effizienz wider, die für diese Standorte im Bereich von $0.20 < r^2 < 0.60$ lag (Kiese et al., 2003 [**IV**]).

Die Ergebnisse aus den Untersuchungen zur Modellvalidierung für die beiden Standorte in Costa Rica müssen gesondert betrachtet werden, da für diese Standorte nur in monatlichen Abständen erfasste N_2O-Emissionen vorlagen, die Modellsimulationen aber in Tagesauflösung erfolgen. Zu diesen Datensätzen muss angemerkt werden, dass die N_2O-Emissionsmessungen in Costa Rica, abgesehen von den eigenen Messungen und den Messungen von Breuer et al. (2000), zu den besten publizierten Datensätzen von N_2O-Emissionen aus tropischen Regenwaldböden weltweit gehören. Für die beiden Tieflandstandorte La Selva und Guacimo simulierte PnET-N-DNDC während der Regenzeit eine hohe zeitliche Variabilität der N_2O-Emissionen, die allerdings aus den monatlich gemessenen N_2O-Flussraten nicht abzuleiten war. Die simulierten Tageswerte waren jedoch in derselben Größenordnung wie die monatlich gemessenen N_2O-Emissionen. Da die Simulationsergebnisse beider Standorte sehr gut mit der saisonalen Variabilität der gemessenen N_2O-Flüsse übereinstimmte, ergab sich aus dem Vergleich der aus den Tageswerten gemittelten monatlichen N_2O-Emissionen mit den monatlich gemessenen Werten eine sehr gute Übereinstimmung (La Selva: $r^2 = 0.68$; Guacimo $r^2 = 0.38$) (Kiese et al., 2003 [**IV**]). Insgesamt zeigen die Ergebnisse aus den Untersuchungen zur Modellvalidierung, dass die Simulationen sowohl die standörtlichen als auch die saisonalen Unterschiede der N_2O-Emissionen mit einer akzeptablen Genauigkeit wiedergeben konnten. Die Präzision, mit der im Rahmen dieser Arbeit die N_2O-Emissionen aus tropischen Regenwaldböden simuliert werden konnten, ist durchaus mit Modellergebnissen von PnET-N-DNDC für temperate Waldökosysteme zu vergleichen (Stange et al., 2000; Butterbach-Bahl et al., 2001). Damit war die Voraussetzung geschaffen, die Eignung des PnET-N-DNDC Modells zur Berechnung eines detaillierten N_2O-Emissionskatasters für tropische Regenwaldböden der Wet Tropics in Australien zu evaluieren.

5.3. Erstellung eines N_2O-Emissionskataster für tropische Regenwaldböden Australiens

Für die regionale Modellanwendung war es erforderlich, ein umfangreiches Geographisches Informationssystem aufzubauen, das die räumliche Verteilung der Modelltreiber (Tages-Minimum und -Maximum Temperaturen, Tages-Niederschlag), die Modell-Eingabeparameter (pH-Wert, organischer C-Gehlat, Bodenfruchtbarkeit, Tongehalt, Skelettgehalt, Bestandesart und Bestandesalter) und das Vorkommen von Regenwald innerhalb des Untersuchungsgebiets der Wet Tropics im Rahmen einer GIS Datenbank vorhielt.

Die Regionalisierung des Klimas im Untersuchungsgebiet erfolgte auf Basis meteorologischer Daten von insgesamt 61 Klimastationen, für die durch ein statistisches Verfahren repräsentative Polygone berechnet wurden (Kiese et al., 2003 [**IV**]). Zur Erfassung der Bodeneigenschaften konnte auf mehrere digitale Karten des Department of Primary Industries (DPI), Mareeba und Bodenzustandsberichte (Murtha, 1986; Laffan, 1988; Murtha, 1989; Wilson *et al*., 1990; Cannon *et al*., 1992; Heiner *et al*. 1994; Murtha *et al*., 1996; Malcolm *et al*., 1999) zurückgegriffen werden, die Informationen zur räumlichen Verteilung von Bodentypen und deren Eigenschaften enthielten. Die Verbreitung von Regenwäldern im Untersuchungsgebiet (Tracey, 1982) wurde aus einer digitalen Karte, die vom Tropical Forest Research Centre, Atherton, Australien, zur Verfügung gestellt wurde, abgeleitet (Kiese et al., 2003 [**IV**]). Durch die Kopplung von PnET-N-DNDC an die GIS Datenbank konnte schließlich eine Regionalisierung der N_2O-Emissionen für eine Fläche von etwas mehr als 4000 km^2 durchgeführt werden. Da das Modell für die Verhältnisse der Trockenzeit initialisiert wurde, erfolgte die Modellierung der N_2O-Emissionen für den Zeitraum 1. Juli 1997 bis 31. Juni 1998. Die Jahressumme der simulierten N_2O-Emissionen im Gebiet der Wet Tropics variierte dabei zwischen 1.3 - 4.6 kg N_2O-N ha^{-1} $Jahr^{-1}$ (Mittelwert für das gesamte Untersuchungsgebiet: 2.4 kg N_2O-N ha^{-1} $Jahr^{-1}$) (Abb. 5). Vergleichsweise hohe N_2O-Emissionen(> 3.0 kg N_2O-N ha^{-1} $Jahr^{-1}$) wurden in den küstennahen Tiefländern für Regionen mit hohen Jahresmitteltemperaturen und hohen Jahresniederschlägen (Daintree Nationalpark, und Bellenden Ker/ Bartel Frere Gebiet) simuliert. PnET-N-DNDC prognostizierte dagegen deutlich niedrigere N_2O-Emissionen für die trockeneren und kälteren Atherton Tablelands (1.3 – 3.0 kg N_2O-N ha^{-1} $Jahr^{-1}$) sowie für die trockeneren küstennahen Regionen im Süden des Untersuchungsgebiets (1.3 – 2.0 kg N_2O-N ha^{-1} $Jahr^{-1}$). Die signifikanten räumlichen Unterschiede der N_2O-Emissionen im Untersuchungsgebiet sind folglich hauptsächlich auf klimatische Unterschiede zurückzuführen. Die Variabilität der N_2O-Emissionen innerhalb eines klimatisch homogen zu betrachtenden Gebiets, z.B. die Atherton Tablelands, wird hingegen hauptsächlich durch die Heterogenität der Bodeneigenschaften hervorgerufen (Kiese et al., 2003 [**IV**]). Dieses Ergebnis stimmt sehr gut mit den beobachteten N_2O-Emissionen an den verschiedenen Untersuchungsstandorten überein (Breuer et al., 2000; Kiese et al., 2002 [**I**], Kiese et al., 2003 [**III**]).

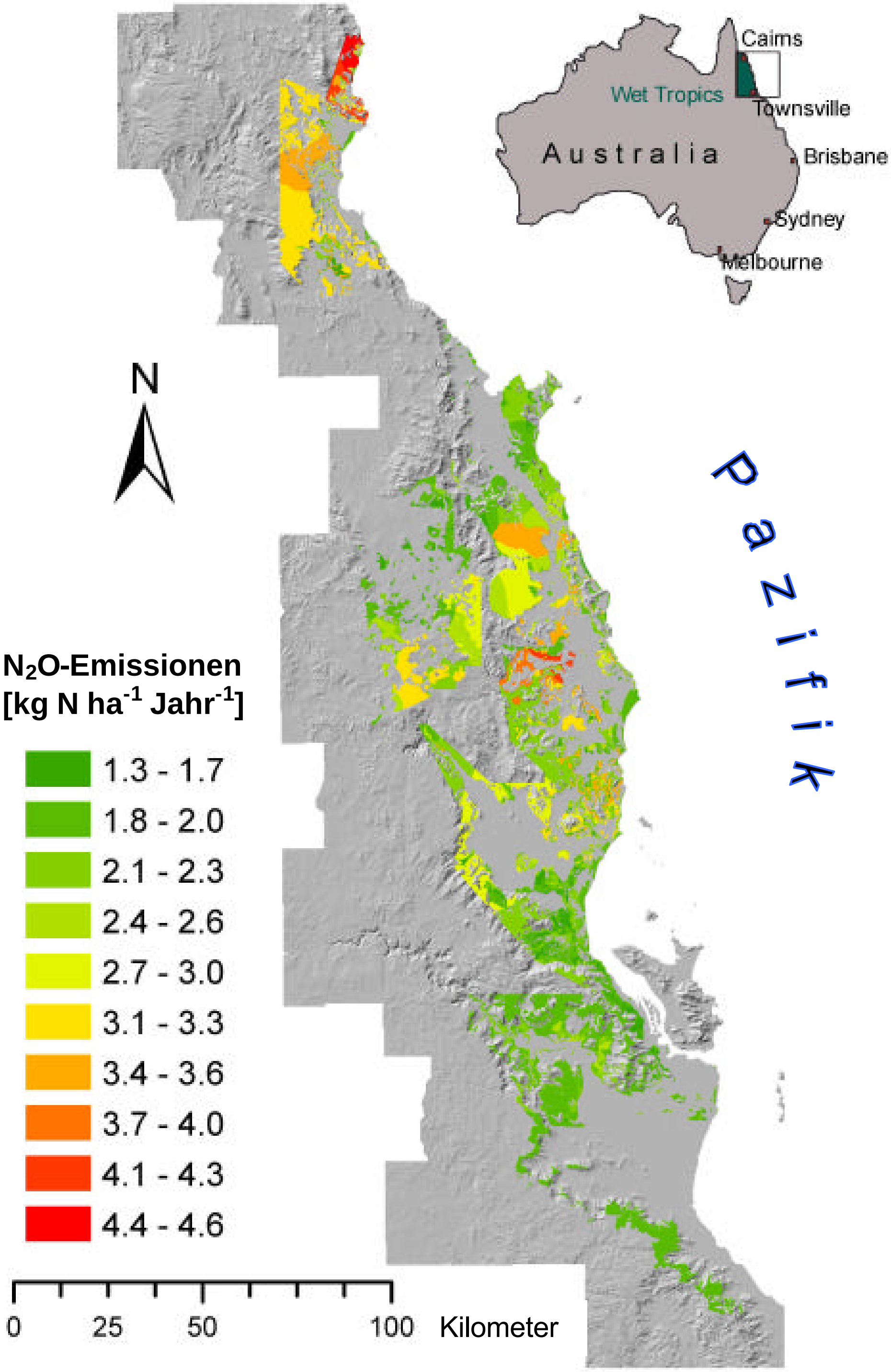


Abb. 5: N_2O-Emissionskataster für das Gebiet der Wet Tropics

Zur Untersuchung der Reaktionsstärke von PnET-N-DNDC auf Veränderungen der Eingabeparameter oder Modelltreiber wurden zahlreiche Sensitivitätsanalysen auf regionaler Ebene durchgeführt. Dabei wurden unterschiedlich initialisierte Modellläufe für das gesamte Gebiet der Wet Tropics gestartet und die Abweichung der aus den Ergebnissen berechneten mittleren flächengewichteten N_2O-Emissionen mit der Höhe der flächengewichteten N_2O-Emissionen der Originalparametrisierung (2.4 kg N_2O-N ha^{-1} $Jahr^{-1}$) als Kontrolle verglichen. Die Analyse ergab, dass die Bodentextur, der C/N-Gehalt der Blätter, die Biomasse und der organische C-Gehalt im Oberboden die Parameter waren, die die höchsten Abweichungen (±10-15%) der simulierten N_2O-Emissionen gegenüber der Kontrolle verursachten (Kiese et al., 2003 [**IV**]).

Die Untersuchungen zur Modellvalidierung, die Simulationen im regionalen Maßstab sowie die Sensitivitätsanalysen zeigen, dass das weiterentwickelte PnET-N-DNDC Modell logische, dem Prozess- und Raumverständnis nach erklärbare Ergebnisse liefert, die zusätzlich durch Messungen belegt sind. Es konnte damit erstmals weltweit ein prozessorientiertes Modell zur Regionalisierung von N_2O-Emissionen für tropische Waldökosysteme erfolgreich angewendet werden. Die erzielten Ergebnisse unterstützen nachhaltig die aufgestellte Hypothese, dass die Entwicklung und Anwendung prozessorientierter Modelle eine zukunftsweisende Strategie ist, um die Quellstärke von N_2O aus biogenen Quelle mit höherer Präzision quantifizieren zu können.

6. Zusammenfassung und Ausblick

In scharfem Kontrast zur Einschätzung der Bedeutung tropischer Regenwälder als der zumindest zweitwichtigsten Quelle innerhalb des globalen N_2O-Budgets steht die derzeit nur äußerste begrenzte Anzahl von veröffentlichten N_2O-Flussraten aus diesen Ökosystemen, auf die sich diese Einschätzung stützt. Problematisch ist in diesem Zusammenhang auch, dass bislang zeitlich hoch aufgelöste Messungen von N_2O-Emissionen aus tropischen Wäldern über längere Zeiträume fehlten. Diese sind jedoch erforderlich, um einerseits die Unsicherheiten der Abschätzung der N_2O-Quellstärke auf der Skala der Standorte zu verringern. Andererseits bilden solche Messungen die Grundlage für eine verlässliche Weiterentwicklung und Validierung prozessorientierter Modelle, die nach heutigem Stand des Wissens als das meist aussichtsreichste Werkzeug angesehen werden müssen, um die Abschätzung von N- und C-Spurengasemissionen auf regionaler und/ oder globaler Ebene qualitativ zu verbessern.

Im Rahmen dieser Arbeit wurden daher zeitlich hoch aufgelöste Datensätze des N_2O-, CH_4- und CO_2-Spurengasaustauschs zwischen verschiedenen tropischen Regenwaldböden Australiens und der Atmosphäre erarbeitet. Es ist dabei weltweit erstmals gelungen, den saisonalen Verlauf der N_2O-Emissionen und der CH_4-Aufnahme des Bodens eines Tieflandregenwaldes in mindestens täglicher Auflösung über ein komplettes Jahr zu bestimmen. Anhand begleitender Untersuchungen zu den Umweltbedingungen (z.B. Niederschlag, Bodenfeuchte, Bodentemperatur), der Bestimmung standörtlicher Unterschiede der Bodeneigenschaften (z.B. pH-Wert, C/N-Verhältnis, Textur) sowie durchgeführter Prozessanalysen, wie der Erfassung standörtlicher und saisonaler Unterschiede der Brutto-Nitrifikationsraten, konnten wesentliche, die Höhe der N_2O-Emissionen auf Prozessebene regelnde Wirkungszusammenhänge aufgezeigt werden. Basierend auf dieser im Rahmen der vorliegenden Arbeit gewonnenen umfangreichen Datengrundlage wurde das prozessorientierte Modell PnET-N-DNDC, das ursprünglich zur Simulation von N- und C-Spurengasemissionen aus temperaten Wäldern entwickelt wurde, derart weiterentwickelt, dass es nunmehr auch zur Simulation von N_2O-Emissionen aus tropischen Waldökosystemen eingesetzt werden kann. Nach erfolgreicher Validierung des Modells wurde dieses mit einer GIS-Datenbank gekoppelt, um erstmals ein detailliertes N_2O-Spurengasinventar für ein ausgewähltes größeres tropisches Regenwaldgebiet berechnen zu können. Die für das Gebiet der Wet Tropics (9000 km^2), Queensland, Australien, durchgeführte Fallstudie zeigt, dass die tropische Version von PnET-N-DNDC ein elegantes wie zuverlässiges Werkzeug zur

Regionalisierung von N_2O-Emissionen aus Böden tropischer Regenwälder darstellt. Jedoch muss gleichzeitig auch darauf hingewiesen werden, dass für eine weitere Präzisierung der simulierten N_2O-Emissionen sowie für eine zukünftige Validierung anderer N- und C-Spurengase, wie NO und CH_4 sowohl die Modellentwicklung von PnET-N-DNDC weiter vorangetrieben werden muss als auch weitere detaillierte Messungen von N- und C-Spurengasemissionen aus tropischen Waldökosystemen unabdingbar sind. Wie wichtig die Bereitstellung von Methoden zur detaillierten flächenbezogenen Erfassung klimarelevanter Spurengase ist, geht auch aus dem Artikel 5 des Kyoto-Protokolls hervor. Dieser Artikel fordert von allen Annex I Staaten, dass diese die N- und C-Spurengasemissionen in lückenloser, nachvollziehbarer Weise inventarisieren, da sie sonst zukünftig (2007) von den Mechanismen, beispielweise des Emissionshandels, ausgeschlossen werden.

Darüberhinaus haben prozessorientierte Modelle auch das Potential die Auswirkungen von „Global Climate Change“ auf die Wechselwirkung zwischen der Biosphäre und der Atmosphäre zu prognostizieren. Dies betrifft insbesondere Rückkoppelungs-Mechanismen zwischen Klimawandel und dem Emissions/ -Depositions-Verhalten klimarelevanter Spurengase. Dazu wird es allerdings erforderlich sein, die prinzipiell eindimensionalen Modelle, wie PnET-N-DNDC, an regionale Wasserhaushalts-, Vegetations-, und Klimamodelle zu koppeln und so zu komplexen Modellsystemen weiterzuentwickeln. Der hierdurch zu erzielende verbesserte Kenntnisstand der Biosphäre-Atmosphäre-Wechselwirkungen von der lokalen bis zur globalen Skala könnte einen wichtigen Beitrag zur Verbesserung globaler Klimamodelle und damit zur Abschätzung zukünftiger Klimaveränderungen leisten.

7. Literatur

Ambus P., Christensen S. (1995), Spatial and seasonal nitrous oxide and methane fluxes in Danish forest-, grassland-, and agroecosystems, *Journal of Environmental Quality*, 24, 993-1001.

Barraclough D., Geens E.L., Davies G.P., Maggs J.M. (1985), Fate of fertilizer nitrogen. III. The use of single and double labeled ^{15}N ammonium nitrate to study nitrogen uptake by ryegrass, *Journal of Soil Science*, 36, 593-603.

Barraclough D. (1995), ^{15}N isotope dilution techniques to study soil nitrogen transformations and plant uptake, *Fertilizer Researc*h, 42, 185-192.

Borken W., Brumme R.(1997), Liming practice in temperate forest ecosystems and the effects on CO_2, N_2O and CH_4 fluxes, *Soil Use and Management*, 13, 251-257.

Bremner J.M. (1997), Sources of nitrous oxide in soil, *Nutrient Cycling in Agroecosystems*, 49, 7-16.

Breuer L. (2000), N_2O-Freisetzung aus tropischen Waldböden Australiens – Anteil von Nitrifikation und Denitrifikation, Dissertation Albert-Ludwigs-Universität Freiburg, Schriftenreihe des Fraunhofer Instituts Atmosphärische Umweltforschung, Band 64, Deutschland.

Breuer L., Papen H., Butterbach-Bahl K. (2000), N_2O emission from tropical forest soils of Australia, *Journal of Geophysical Research,* 105, 26353-26367.

Breuer L., Kiese R., Butterbach-Bahl K. (2002), Temperature and moisture effects on nitrification rates in tropical rain forest soils, *Soil Science Society of America Journal*, 66, 834-844.

Brumme R. (1995), Mechanisms of carbon and nutrient release and retention in beech forest gaps, III. Environmental regulation of soil respiration and nitrous oxide emissions along a microclimate gradient, *Plant and Soil*, 169, 593-600.

Butterbach-Bahl K., Gasche R., Huber C., Kreutzer K., Papen H. (1998), Impact of nitrogen input by wet deposition on N-trace gas fluxes and CH_4-oxidation in spruce forest ecosystems of the temperate zone in Europe, *Atmospheric Environment*, 32, 559-564.

Butterbach-Bahl K., Stange F., Papen H., Li C. (2001), Regional inventory of nitric oxide and nitrous oxide emissions for forest soils of Southeast Germany using the biogeochemical model PnET-N-DNDC, *Journal of Geophysical Research*, 106, 34155-34166.

Butterbach-Bahl K., Papen H. (2002), Four years continuous record of CH_4-exchange between the atmosphere and untreated and limed soil of a N-saturated spruce and beech forest ecosystem in Germany, *Plant and Soil*, 240, 77-90.

Butterbach-Bahl K., Gasche R., Willibald G., Papen H. (2002), Exchange on N-gases at the Höglwald Forest – A summary, *Plant and Soil*, 240, 117-123.

Cannon M.G., Smith C.D., Murtha G.G. (1992), Soils of the Cardwell-Tully area, North Queensland, Divisional Report No. 115, Commonwealth Scientific and Industrial Research Organization, Melbourne, Australien.

Castro M.S., Steudler P.A., Melillo J.M., Aber J.D., Bowden R.D. (1995), Factors controlling atmospheric methane consumption by temperate forest soils, *Global Biogeochemical Cycles*, 9, 1-10.

Cleveland C.C., Townsend A.R., Schimel D.S., Fisher H., Howarth R.W., Hedin L.O., Perakis S.S., Latty E.F., Von Fischer J.C., Elseroad A., Wasson M.F. (1999), Global patterns of terrestrial biological nitrogen (N2) fixation in natural ecosystems, *Global Biogeochemical Cycles*, 13, 623-645.

Conrad R. (1994), Soil microbial processes involved in production and consumption of atmospheric trace gases, *Advances in Microbial Ecology*, 14, 207-250.

Crill P.M. (1991), Seasonal patterns of methane uptake and carbon dioxide release by a temperate woodland soil, *Global Biogeochemical Cycles*, 5, 319-334.

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

Daum M., Zimmer W., Papen H., Kloos K., Nawrath K., Bothe H. (1998), Physiological and molecular biological characterization of the heterotrophic nitrifier Pseudomonas pudita, *Current Microbiology*, 37, 281-288.

Davidson E.A. (1991), Fluxes of nitrous oxide and nitric oxide from terrestrial ecosystems, In: *Microbial production and consumption of greenhouse gases: Methane, Nitrogen Oxides and Halomethanes* (Hrsg. Rogers J.E., Whitman W.B.), American Society of Microbiology, Washington, USA.

Davidson E.A., Hart S.C., Firestone M.K. (1992), Internal cycling of nitrate in soils of a mature coniferous forest, *Ecology*, 73, 1148-1156.

Davidson E.A., Matson P.A., Vitousek P.M., Riley R., Dinkin K., Garcia-Méndez G., Maass J.M. (1993), Processes regulating soil emission of NO and N_2O in a seasonally dry tropical forest, *Ecology*, 74, 130-139, 1993.

Davidson E.A., Belk E., Boone R. (1998), Soil water content and temperature as independent or confounded factors controlling soil respiration in a temperate mixed hardwood forest, *Global Change Biology*, 4, 217-227.

Davidson E.A., Bustamante M.M., de Siqueira Pinto A. (2001), Emissions of nitrous oxide from soils of native and exotic ecosystems of the Amazon and Cerrado regions of Brazil, In: *Optimizing Nitrogen Management in Food and Energy Production and Environmental Protection*: Proceedings of the 2nd International Nitrogen Conference on Science and Policy, The ScientificWorld 1, 312-319.

Delmas R.A., Servant J., Tathy J.P., Gros B., Labat M. (1992), Sources and sinks of methane and carbon dioxide exchanges in mountain forest in Equatorial Africa, *Journal of Geophysical Research*, 97, 6169-6179.

Devol A.H., Richey J.E., Forsberg B.R., Martinelli L.A. (1990), Seasonal dynamics in methane emissions from the Amazon floodplain, *Journal of Geophysical Research*, 95, 16417-16426.

Erickson H., Davidson E.A., Keller M. (2002), Former land-use and tree species affect nitrogen oxide emissions from a tropical dry forest, *Oecologia*, 130, 297-308.

Firestone M.K., Davidson E.A. (1989), Microbiological basis of NO and N_2O production and consumption in soil, In: *Exchange of trace gases between terrestrial ecosystems and the atmosphere*, (Hrsg. Andrea M.O., D.S. Schimel), John Wiley & Sons Ltd., Chichester, UK.

Fung I., John J., Lerner J., Matthews E., Prather M., Steele L.P., Fraser P.J. (1991), Three-dimensional model synthesis of the global methane cycle, *Journal of Geophysical Research*, 96, 13033-13065.

Galloway J.N. (1998), The global nitrogen cycle: changes and consequences, *Environmental Pollution*, 102, 15-24.

Garcia-Méndez G.J., Maass J.M., Matson P.A., Vitousek P.M. (1991), Nitrogen transformations and nitrous oxide flux in a tropical deciduous forest in Mexico, *Oecologia* 88, 362-366.

Goreau T.J., De Mello W.Z. (1998), Tropical deforestation: Some effects on atmospheric chemistry, *Ambio* 17, 275-281.

Granli T., Bockman O.C. (1994), Nitrous Oxide from Agriculture, *Norwegian Journal of Agricultural Sciences*, 12, 1-128.

Harris R.C., Sebacher D.I. (1981), Methane flux in forested freshwater swamps of the southeastern United States, *Geophysical Research Letters*, 8, 1002-1004.

Heidenfelder A. (2002), Einfluß von Klima und Bewirtschaftung auf am N-Umsatz beteiligte mikrobielle Prozesse in Waldböden, Dissertation Albert-Ludwigs-Universität Freiburg, Forschungszentrum Karlsruhe in der Helmholtz-Gemeinschaft, Wissenschaftliche Berichte FZKA 6766, Deutschland.

Heiner I.J., Grundy M.J. (1994), Land resources of the Ravenshoe - Mt. Garnet area, North Queensland, Vol. 1 – Land resources inventory, Land Resources Bulletin, Queensland Department of Primary Industries, Brisbane, Australien.

Holland E.A., Dentener F.J., Braswell B.H., Sulzman J.M. (1999), Contemporary and pre-industrial global reactive nitrogen budgets, *Biogeochemistry*, 46, 7-43.

Holt J. A., Hodgen M. J., Lamb D. (1990), Soil respiration in the seasonally dry tropics near Townsville, North Queensland. *Australian Journal of Soil Research*, 28, 737-745.

Hunt R.E., Piper S.C., Nemani R., Keeling C.D., Otto R.D., Running S.W. (1996), Global net carbon exchange and intra-annual atmospheric CO_2 concentrations predicted by an ecosystem process model and three dimensional atmospheric transport model, *Global Biogeochemical Cycles*, 10, 431-456.

Ingwersen A., Butterbach-Bahl K., Gasche R., Richter O., Papen H. (1999), Barometric process separation: New method for quantifying nitrification, denitrification, and nitrous oxide sources in soils, *Soil Science Society of America Journal*, 63, 117-128.

IPCC (1995), Houghton J.T., Meira Filho L.G., Bruce J., Lee H., Callander B.A., Haites E., Harris N., Maskell K. (Hrsg.), Climate Change 1994: Radiative Forcing of Climate Change and an Evaluation of the IPCC IS92 Emission Scenarios, *Cambridge University Press*, Cambridge, UK.

IPCC (1997), Intergovernmental Panel on Climate Change, Guidlines for National Geenhouse Gas Inventories, The Organization for Economic Co-operatioin and Developement (OECD), Paris, Frankreich.

IPCC (2000), Intergovernmental Panel on Climate Change, Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories, Technical Support Unit IPCC National Greenhouse Gas Inventory Programme, Institute for Global Environmental Strategies, Kanagawa, Japan.

IPCC (2001), Houghton J.T., Ding Y., Griggs D.J., Noguer M., van der Linden P.J., Xiaosu D. (Hrsg.), Climate Change 2001: The Scientific Basis, Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change, *Cambridge University Press*, Cambridge, UK.

Keller M., Goreau T.J., Wofsy S.C., Kaplan W.A., McElroy M.B. (1983), Production of nitrous oxide and consumption of methane by forest soils, *Geophysical Research Letters*, 10, 1156-1159.

Keller M., Kaplan W.A., Wofsy S.C. (1986), Emissions of N_2O, CH_4 and CO_2 from tropical forest soils, *Journal of Geophysical Research*, 91, 11791-11802.

Keller M., Veldkamp E., Weitz A.M., Reiners W.A. (1993), Effect of pasture age on soil trace-gas emissions from a deforested area of Costa Rica, *Nature*, 365, 244-246.

Keller M., Reiners W.A. (1994), Soil atmosphere exchange of nitrous oxide, nitric oxide, and methane under secondary succession of pasture to forest in the Atlantic lowlands of Costa Rica, *Global Biogeochemical Cycles,* 8, 399-409.

Kirkham D., Bartolomew W.V. (1954), Equations for following nutrient transformations in soil, utilizing tracer data, *Soil Science Society of America Journal*, 18, 33-34.

Koutika L.S., Choné T., Andreux F., Burtin G., Cerri C.C. (1999), Factors influencing carbon decomposition of topsoils from the Brazilian Amazon Basin, *Biology and Fertility of Soils*, 28, 436-438.

Kroeze C., Mosier A., Bouwman L. (1999), Closing the N_2O Budget: A retrospective analysis, *Global Biogeochemical Cycles*, 13, 1-8.

Kroeze C., Mosier A. (2002), Progress in source strength estimates of nitrous oxide (N_2O). In: *Non-CO_2 greenhouse gases: Scientific understanding, control options and policy aspects*, (Hrsg. van Ham J., Baede A.P., Guicherit R., Williams-Jacobse J.G.), S. XXIX–XXXIII, Millpress, Rotterdam, Niederlande.

Laffan M.D. (1988), Soils and land use on the Atherton Tableland, North Queensland, Soils and land use series No. 61, Commonwealth Scientific and Industrial Research Organization, Melbourne, Australien.

La Scala N., Marques J., Pereira G.T., Cora J.E. (2000), Short-term temporal changes in the spatial variability model of CO_2 emission from a Brazilian bare soil, *Soil Biology & Biochemistry*, 32, 1459-1462.

Li C., Frolking S., Frolking T.A. (1992), A model of nitrous oxide evolution from soil driven by rainfall events: 1. Model structure and sensitivity, *Journal of Geophysical Research*, 97, 9759-9776.

Li C., Aber J., Stange F., Butterbach-Bahl K., Papen H. (2000), A process-oriented model of N_2O and NO emission from forest soils, 1. Model development, *Journal of Geophysical Research*, 105, 4369-4384.

Linn D.M., Doran J.W. (1984), Effect of water-filled pore space on carbon dioxide and nitrous oxide production in tilled and nontilled soils, *Soil Science Society American Journal*, 48, 1267-1272.

Liu S., Reiners W.A., Keller M., Schimel D.S. (2000), Simulation of nitrous oxide and nitric oxide emissions from a primary forest in the Costa Rican Atlantic Zone, *Environmental Modeling and Software*, 15, 727-743.

Maggs J., Hewett B. (1990), Soil and litter respiration in rainforest of contrasting nutrient status and physiognomic structure near Lake Eacham, north-east Queensland, *Australian Journal of Ecology*, 15, 329-336.

Malcolm D.T., Nagel B.K., Sinclair I., Heiner I.J. (1999), Soils and agricultural land suitability of the Atherton Tablelands, North Queensland, Land Resources Bulletin, Department of Natural Resources, Brisbane, Australien.

Martikainen P.J., De Boer W. (1993), Nitrous oxide production and nitrification in acidic soil from a Dutch coniferous forest, *Soil Biology & Biochemistry*, 25, 343-347.

Matson P.A., Vitousek P.M. (1990), Ecosystem approach to a global nitrous oxide budget, *Bioscience*, 40, 667-672.

McSwiney, C.P., McDowell W.H., Keller M. (2001), Distribution of nitrous oxide and regulators of its production across a tropical rainforest catena in the Luquillo Experimental Forest, Puerto Rico, *Biogeochem*istry, 56, 265-286.

Melillo J.M., Steudler P.A., Feigl B.J., Neill C., Garcia D., Piccolo M.C., Cerri C.C., Tian H. (2001), Nitrous oxide emissions from forests and pastures of various ages in the Brazilian Amazon, *Journal of Geophysical Research*, 106, 34179-34188.

Mosier A., Kroeze C., Nevison C., Oenema O., Seitzinger S., van Cleemput O. (1998), Closing the global N_2O budget: nitrous oxide emissions through the agricultural nitrogen cycle - OECD/IPCC/IEA phase II development of IPCC guidelines for national greenhouse gas inventory methodology, *Nutrient Cycling in Agroecosystems*, 52, 225-248.

Murtha G.G. (1986), Soils of the Tully-Innisfail area, North Queensland, Divisional Report No. 82, Commonwealth Scientific and Industrial Research Organization, Melbourne, Australien.

Murtha G.G. (1989), Soils of the Mossman Cape Tribulation area, North Queensland, Divisional Report No. 102, Commonwealth Scientific and Industrial Research Organization, Melbourne, Australien.

Murtha G.G., Cannon M.G., Smith C.D. (1996), Soils of the Babinda-Cairns area, North Queensland, Divisional Report No. 123, Commonwealth Scientific and Industrial Research Organization, Melbourne, Australien.

Myhre G., Highwood E.K., Shine K.P., Stordal F. (1998), New estimates of radiative forcing due to well mixed greenhouse gases, *Geophysical Research Letters*, 25, 2715-2718.

Nash J.E., Sutcliffe J.V. (1970), River flow forecasting through conceptual models, Part I-A, Discussion of principles, *Journal of Hydrology*, 10, 282-290.

Neill C., Piccolo M.C., Steudler P.A., Cerri C.C. (1999), Nitrogen dynamics in Amazon forest and pasture soils measured by ^{15}N pool dilution, *Soil Biology and Biochemistry*, 31, 567-572.

O´Hara G.W., Daniel R.M. (1985), Rhizobial dentirification: A review, *Soil Biology and Biochemistry*, 18, 559-568.

Olivier J.G., Bouwman A.F., van der Hoek K.W., Berdowski J.J. (1998), Global air emission inventories for anthropogenic sources of NO_x, NH_3 and N_2O in 1990, *Environmental Pollution*, 102, 135-148.

Papen H., von Berg R., Hinkel I., Thoene B., Rennenberg H. (1989), Heterotrophic nitrification by Alcaligenes faecalis: NO_2^-, NO_3^-, N_2O and NO-production in exponentially growing cultures, *Applied and Environmental Microbiology*, 55, 2068-2072.

Papen H., Rennenberg H. (1990), Microbial processes involved in emissions of radiatively important trace gases, *Transactions 14th international congress of soil scienc*e, S. 232-237, Kyoto, Japan.

Papen H., Butterbach-Bahl K. (1999), A 3-year continuous record of nitrogen trace gas fluxes from untreated and limed soil of a N-saturated spruce and beech forest ecosystem in Germany, 1. N_2O-emissions, *Journal of Geophysical Research*, 104, 18487-18503.

Parton W.J., Mosier A.R., Ojima D.S., Valentine D.W., Schimel D.S., Weier K., Kulmala A.E. (1996), Generalized model for N_2 and N_2O production from nitrification and denitrification, *Global Biogeochemical Cycles*, 10, 401-412.

Poth M., Focht D.D. (1985), ^{15}N kinetic analysis of N_2O production by Nitrosomonas europaea - an example of nitrifier denitrification, *Applied and Environmental Microbiology*, 49, 1134-1141.

Potter C.S., Riley R.H., Klooster S.A. (1997), Simulation modeling of nitrogen trace gas emissions along an age gradient of tropical forest soils, *Ecological Modelling*, 97, 179-196.

Prather M.J. (1998), Time scales in atmospheric chemistry: Coupled perturbations to N_2O, NO_y, and O_3, *Science*, 279, 1339-1341.

Ridgwell A.J., Marshall S.J., Gregson K. (1999), Consumption of atmospheric methane by soils: A process-based model, *Global Biogeochemical Cycles*, 13, 59-70.

Riley H.R., Vitousek P.M. (1995), Nutrient dynamics and nitrogen trace gas flux during ecosystem development in montane rain forest*,* *Ecology*, 76, 292-304.

Ritchie G.A., Nicholas D.J. (1972), Identification of the sources of nitrous oxide produced by oxidative and reductive processes in Nitrosomonas europaea, *Biochemical Journal*, 126, 1181-1191.

Schimel J.P., Firestone M.K., Killham K.S., (1984), Identification of heterotrophic nitrification in a sierran forest soil, *Applied and Environmental Microbiology*, 48, 802-806.

Serca D., Delmas R., Jambert C., Labroue L. (1994), Emissions of nitrogen oxides from equatorial rain forest in central Africa: Origin and regulation of NO emission from soils, *Tellus*, 46B, 243-254.

Sitaula B.K., Bakken L.R. (1993), Nitrous oxide release from spruce forest soil: relationships with nitrification, methane uptake, temperature, moisture and fertilization, *Soil Biology & Biochemistry*, 25, 1415-1421.

Sitaula B.K., Bakken L.R., Abrahamsen G. (1995), CH_4 uptake by temperate forest soil: effect of N input and soil acidification, *Soil Biology and Biochemistry*, 27, 871-880.

Smith K.A. (1997), The potential for feedback effects induced by global warming on emissions of nitrous oxide by soils, *Global Change Biology*, 25, 1527-1536.

Sørensen L.H. (1974), Rate of decomposition of organic matter in soil as influenced by repeated air drying-rewetting and repeated additions of organic material, *Soil Biology & Biochemistry*, 6, 287-292.

Spain A.V., Unwin G., Sinclair D.F. (1989), Soils and vegetation at three rainforest sites in tropical north-eastern Queensland, Division Report 15, Division of Soils, CSIRO, Melbourne, Australien.

Stange F., Butterbach-Bahl K., Papen H., Zechmeister-Boltenstern S., Li C., Aber J. (2000), A process oriented model of N_2O and NO emissions from forest soils: 2, Sensitivity analysis and validation, *Journal of Geophysical Research*, 105, 4385-4398.

Stange F. (2001), Entwicklung und Anwendung eines prozeßorientierten Modells zur Beschreibung der N_2O- und NO-Emissione aus Böden temperater Wälder, Dissertation Albert-Ludwigs-Universtiät Freiburg, Schriftenreihe des Fraunhofer Instituts Atmosphärische Umweltforschung, Band 69, Deutschland.

Stark J.M., Hart S.C. (1997), High rates of nitrification and nitrate turnover in undisturbed coniferous forests, *Nature*, 385, 61-64.

Steinkamp R., Butterbach-Bahl K., Papen H.(2001), Methane oxidation by soils of an N-limited and N-fertilized spruce forest in the Black Forest, Germany, *Soil Biology and Biochemistry*, 33, 145-153.

Steudler P.A., Bowden R.D., Melillo J.M., Aber J.D. (1989), Influence of nitrogen fertilization on methane uptake in temperate forest soils, *Nature*, 341, 314-316.

Steudler P.A., Melillo J.M., Bowden R.D., Castro M.S., Lugo A.E. (1991), The effect of natural and human disturbance on soil nitrogen dynamics and trace gas fluxes in a Puerto Rican wet forest, *Biotropica*, 23, 356-363.

Steudler P.A., Melillo J.M., Feigl B.J., Neill C., Piccolo M.C, Cerri C.C. (1996), Consequence of forest-to-pasture conversion on CH_4 fluxes in the Brazilian Amazon Basin, *Journal of Geophysical Research*, 101, 18547-18554.

Tathy J.P., Delmas R.A., Marenco A., Gros B., Labat M., Servant J.(1992), Methane emission from flooded forest in Central Africa, *Journal of Geophysical Research*, 97, 6159-6168.

Tracey J.G. (1982), The vegetation of the humid tropical region of North Queensland, CSIRO, Melbourne, Australien.

Vangestel M., Merckx R., Vlassak K. (1993), Microbial biomass and activity in soils with fluctuating water contents, *Geodinamica Acta*, 56, 617-626.

Van Veen J.A., Kuikman P.J. (1990), Soil structural aspects of decomposition of organic matter by micro-organisms, *Biogeochemistry*, 11, 213-233.

Verchot L.V., Davidson E.A., Cattânnio J.H., Ackermann I.L., Erickson H.E., Keller M. (1999), Land use and biogeochemical controls of nitrogen oxide emissions from soils in eastern Amazonia, *Global Biogeochemical Cycles*, 13, 31-46.

Verchot L.V., Holmes Z., Mulon L., Groffman P.M., Lovett G.M. (2001), Gross vs net rates of N mineralization and nitrification as indicators of functional differences between forest types, *Soil Biology and Biochemistry*, 33, 1889-1901.

Wassmann R., Thein U.G., Whiticar M.J., Rennenberg H., Seiler W., Junk W.J. (1992), Methane emission from the Amazon floodplain: characterization of production and transport, *Global Biogeochemical Cycles*, 6, 3-13.

Weier K.L., Gillam J.W. (1986), Effect of acidity on nitrogen mineralization and nitrification in Atlantic Coastal Plain soils, *Soil Science Society of America Journal*, 50, 1210-1214.

Whalen S.C., Reeburgh W.S. (2000), Methane oxidation, production, and emission at contrasting sites in a boreal bog, *Geomicrobiology Journal*, 17, 237-251.

Wilson P.R., Baker D.E. (1990), Soils and land suitability of the wet tropical coast of North Queensland: Ingham area, Land Resources Bulletin, Queensland Department of Primary Industries, Brisbane.

Yamulki S., Harrison R.M., Goulding K.W., Webster C.P. (1997), N_2O, NO and NO_2 fluxes from a grassland: Effect of soil pH, *Soil Biology and Biochemistry*. 29, 1199-1208.

Zechmeister-Boltenstern S., Meger S. (1997), Nitrous oxide emissions from two beech forests near Vienna, Austria, In: *Proceedings of the 7th Inernational Workshop on Nitrous Oxide Emissions*, S. 365-371, Köln, Deutschland.

Zou X., Valentine D.W., Sanford Jr. L.R., Binkley D. (1992), Resin-core and buried-bag estimates of nitrogen transformations in Costa Rican lowland rainforests, *Plant and Soil*, 139, 275-283.

Zumft W.G. (1997), Cell biology and molecular basis of denitrification, *Microbiology and Molecular Biology Reviews*, 61, 533-616.

8. Eigene Publikationen

Publikation I

Copyright (2003), with permission from Elsevier

R. KIESE UND K. BUTTERBACH-BAHL

N_2O and CO_2 emissions from three different tropical forest sites in the wet tropics of Queensland, Australia

Soil Biology and Biochemistry, 34, 975-987, 2002

Publikation II

Copyright (2003), with permission from Wiley-VCH

R. KIESE, H. PAPEN, E. ZUMBUSCH UND K. BUTTERBACH-BAHL

Nitrification activity in tropical rain forest soils of the Coastal Lowlands and the Atherton Tablelands, Queensland, Australia

Journal of Plant Nutrition and Soil Science, 165, 682-685, 2002

Publikation III

Copyright (2003), with permission from American Geophysical Union

R. KIESE, B. HEWETT, A. GRAHAM UND K. BUTTERBACH-BAHL

Seasonal variability of N_2O-emissions and CH_4-uptake from/ by a tropical rainforest soil of Queensland, Australia

Global Biogeochemical Cycles, 17, 12-1–12-13, 2003

Publikation IV

R. KIESE, C. LI, D.W. HILBERT, H. PAPEN UND K. BUTTERBACH-BAHL

Regional application of PnET-N-DNDC for estimating the N_2O source strength of tropical rainforests in the Wet Tropics of Australia

Global Change Biology, zur Veröffentlichung eingereicht

PERGAMON

Soil Biology & Biochemistry 34 (2002) 975–987

Soil Biology & Biochemistry

www.elsevier.com/locate/soilbio

N_2O and CO_2 emissions from three different tropical forest sites in the wet tropics of Queensland, Australia

Ralf Kiese, Klaus Butterbach-Bahl*

Department of Soil Microbiology, Fraunhofer Institute for Atmospheric Environmental Research (IFU), Kreuzeckbahnstraße 19,
D-82467 Garmisch-Partenkirchen, Germany

Received 27 July 2001; received in revised form 21 December 2001; accepted 31 January 2002

Abstract

Three different tropical rain forest sites in Northeast Queensland, Australia, two in the Coastal Lowlands (Pin Gin Hill and Bellenden Ker) and one on the Atherton Tablelands (Kauri Creek), were investigated for the magnitude of N_2O and CO_2 emissions from soils during the wet and the dry season. At all sites, mean N_2O emission rates were significantly higher during the wet season (Bellenden Ker: 242.0 ± 7.4 μg N_2O–N m^{-2} h^{-1}, Pin Gin Hill: 140.8 ± 5.1 μg N_2O–N m^{-2} h^{-1}, Kauri Creek: 80.8 ± 3.3 μg N_2O–N m^{-2} h^{-1}) as compared to the dry season when N_2O-emissions were markedly lower (<20 μg N_2O–N m^{-2} h^{-1}) due to limitations in soil moisture. During the wet season, mean N_2O emission rates of the Coastal Lowland sites Bellenden Ker and Pin Gin Hill were approximately twofold higher as compared to N_2O emission rates of the Atherton Tableland site Kauri Creek. These site differences were found to be due to differences in precipitation and soil moisture, the C-to-N ratio of the organic matter, soil pH and temperature. Site and seasonal differences in CO_2-emissions were not as pronounced as for N_2O-emissions. Mean CO_2 emission rates at the different sites were in a range of 92.2 ± 1.8 up to 137.3 ± 4.5 mg C m^{-2} h^{-1}. Correlation analysis revealed a strong dependency of N_2O and CO_2 emissions on changes in soil moisture, whereas changes in soil temperature did not significantly influence the magnitude of in situ N_2O and CO_2 emissions. N_2O emissions were positively correlated to changes in water filled pore space (WFPS) up to a threshold of 50% WFPS at the Bellenden Ker and Kauri Creek sites and up to a threshold of 60% WFPS at the Pin Gin Hill site. CO_2 emission rates were positively correlated to changes in WFPS at dry to moderate soil water contents during the dry season, but were negatively correlated to changes in WFPS during the wet season. Measurements of soil air N_2O-concentrations at the different sites revealed the following sequence in magnitude: Bellenden Ker > Pin Gin Hill > Kauri Creek, which is the same as found for the N_2O source strengths at these sites. © 2002 Elsevier Science Ltd. All rights reserved.

Keywords: Tropical rain forests; Soil N_2O- and CO_2-emission; Soil–air N_2O-concentrations

---

1. Introduction

Nitrous oxide (N_2O) is one of the most important radiatively active trace gases in the atmosphere contributing at least 5% to the observed global warming (Myhre et al., 1998) at present. The current atmospheric N_2O concentration is about 313 nl l^{-1} and is further increasing at a rate of 0.25% yr^{-1}. Atmospheric N_2O is estimated to have 310 times (Houghton et al., 1996) the radiative forcing per molecule relative to CO_2, partly as a consequence of its estimated long atmospheric life time of 127 yr. While the mechanism for the removal of N_2O from the atmosphere appears to be well understood (Bouwman and Taylor, 1996) and the atmospheric increase can be measured with reasonable precision, considerable uncertainty exists with regard to the sources of atmospheric N_2O. About 57% of the global atmospheric sources of N_2O is estimated to be related to emissions from soils (Mosier and Kroeze, 1998) mainly due to the microbial processes of nitrification and denitrification. Besides N_2O-emissions from agricultural soils—which are at present estimated to contribute 3.3 Tg N_2O–N yr^{-1} to the global atmospheric N_2O-budget (IPCC, 1997)—tropical rain forest soils are the major single sources for soil related N_2O emissions (3.5 Tg N_2O–N yr^{-1}, Breuer et al., 2000). Watson et al. (1992) estimated that, on a global scale, tropical rain forest soils contribute 14–23% to the atmospheric N_2O budget. Though there is no doubt about the importance of tropical rain forest soils to the global atmospheric N_2O budget, these estimates are still based on a relatively small number of field data. N_2O fluxes exist in a range of 4.2–70.0 μg m^{-2} h^{-1} for neotropic rain forests (Goreau and DeMello, 1987; Keller et al., 1993; Verchot et al., 1999), 1.7–207.0 μg m^{-2} h^{-1} for an African rain forest site in Kongo (Serca et al., 1994) and 11.3–123.4 μg m^{-2} h^{-1} for

---

* Corresponding author. Tel.: +49-8821-183136; fax: +49-8821-183294.

E-mail address: butterbach@ifu.fhg.de (K. Butterbach-Bahl).

0038-0717/02/$ - see front matter © 2002 Elsevier Science Ltd. All rights reserved.
PII: S0038-0717(02)00031-7

Table 1
Main characteristics of the different measuring sites (all soil variable are given for 0–10 cm soil depth)

	Kauri Creek	Bellenden Ker	Pin Gin Hill
Location	145°38′E 17°17′S	145°54′E 17°16′S	145°58′E 17°33′S
Height above sea level (m)	790	80	60
Mean annual precipitation (mm)	1594[a]	4395[b]	3609[a]
Mean annual temperature (°C)	20.9[a]	24.3[b]	24.1[a]
Soil type	Ustochrept[c]	Ustochrept[d]	Krasnozem[a]
Soil parent material	Granite[c]	Granite	Basalt[a]
Vegetation type[e]	Complex notophyll vine forest	Complex mesophyll vine forest	Complex mesophyll vine forest
Slope (°)	12.0–14.0	9.0–12.0	15.0–18.0
pH ± SE	5.2 ± 0.08	4.1 ± 0.03	4.7 ± 0.06
Bulk density 0–5 cm ($g\ cm^{-3}$)[d]	1.05 ± 0.05	1.09 ± 0.03	0.79 ± 0.02
C-to-N ratio	14.6	12.1	13.8
Organic C content (%)	2.93–3.51[c]	3.11	9.15
Soil texture	Sandy clay loam	Sandy clay loam	Clay
Sand (%)	59–77[c]	57	7
Silt (%)	8–10[c]	21	26
Clay (%)	13–33[c]	22	67

[a] Spain et al. (1989).

[b] Data from Bureau of Meteorology, Brisbane.

[c] Laffan (1988).

[d] Bob Hewett (personal communication, 2000), Tropical Forest Research Center, Atherton.

[e] Tracey (1982).

Australian rain forests (Breuer et al., 2000). In an effort to improve the data base for N_2O emissions from tropical rain forests, we carried out measurements in notogean (Australia) tropical rain forests in the region of the Wet Tropics, Northeastern Australia. Due to the high spatial variability in climate, soil and vegetation this area offers a unique opportunity to study factors influencing the magnitude of N_2O exchange between biosphere and atmosphere in tropical rain forest ecosystems. Thus, the focus of this paper is on the quantification of N_2O-emission rates at sites with pronounced differences in climate and soil characteristics, two of them on the Coastal Lowlands and the third one on the Atherton Tablelands of Queensland, Australia.

2. Materials and methods

2.1. Study sites

N_2O emissions from tropical forest soils were investigated at three different sites: the Kauri Creek site on the Atherton Tablelands and the Pin Gin Hill and Bellenden Ker sites in the Coastal Lowlands, Queensland, Australia. The Atherton Tableland forms an elevated plateau varying in altitude from 700 to 900 m above sea level (asl) 40 km southwest of the city of Cairns. The Coastal Lowlands, approximately 0–80 m asl, are divided from the Atherton Tablelands through the Great Escarpment (downward slope). These geomorphological characteristics in combination with predominantly easterly winds coming from the Coral Sea are the causes for significant gradients in climate, i.e. in precipitation and temperature. Furthermore, a huge variability in soil parent material, starting from granite, metamorphics (mainly schist and phyllit), basalt to acid volcanics (mainly rhyolite), can be found. The main characteristics of the sites investigated in this study are given in Table 1. There are significant differences in climate conditions between the Kauri Creek site at the Atherton Tableland and the sites at the Coastal Lowland. Due to difference in altitude, the mean annual temperature is about 3.5 °C higher at the Coastal Lowland sites Bellenden Ker and Pin Gin Hill as compared to the Kauri Creek site. Furthermore, pronounced differences in the amount of annual rainfall exist between the sites. Though normally precipitation increases with higher altitude, this effect is largely overridden by the marked decrease in precipitation with increasing distance from the sea. Consequently, mean annual precipitation at Bellenden Ker (4395 mm) and at Pin Gin Hill (3609 mm) are both more than two times higher than at Kauri Creek (1594 mm). Due to the differences in temperature and precipitation, changes in the predominating vegetation type can also be observed. At both Lowlands sites, the vegetation type is a complex mesophyll vine forest, whereas at the cooler and 'drier' Atherton Tableland site Kauri Creek, the vegetation type is a complex notophyll vine forest (Tracey, 1982). However, soil texture is comparable at Bellenden Ker and Kauri Creek, since soils at both sites were derived from granite. The soil type is a Ustochrept, characterized by a high sand fraction of up to 60–70%, and medium silt (10–20%) and clay (13–33%) contents. In contrast, soil texture at the Pin Gin Hill site, where soil formation started from a basaltic lava flow, is dominated by a high clay content of 67%. The soil type at this site is a Krasnozem. Soil pH values in the uppermost mineral layer

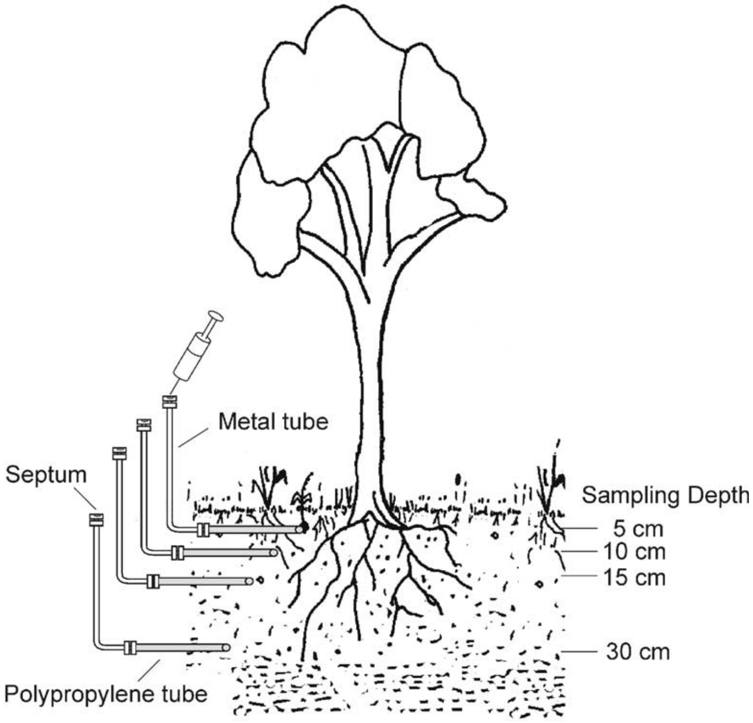


Fig. 1. Sampling design of N_2O concentration measurements in different soil depths.

varies from 5.2 ± 0.1 at the Kauri Creek site to more acidic conditions at the Pin Gin Hill (4.7 ± 0.1) and Bellenden Ker site (4.1 ± 0.1). Most likely, this is due to higher leaching rates of basic soil minerals because of higher annual precipitation values at the Coastal Lowland sites. Finally, a comparison between the sites shows that the C-to-N ratio of the top soil layer is narrower at the Lowland sites (Bellenden Ker, 12.1; Pin Gin Hill, 13.8) as compared to the Kauri Creek site (14.6).

2.2. Measurements of N_2O and CO_2-emissions

N_2O and CO_2 emissions were monitored during two field campaigns each lasting 3 months in the years 2000 and 2001. To quantify seasonal changes in the magnitude of N_2O emissions, measurements were carried out during the wet season and at the end of the dry season with continuing transition towards the wet season.

For determination of N_2O and CO_2-emissions, a mobile and fully automatic measuring system was used, which was described in detail by Breuer et al. (2000). In short, this system consisted of a gas chromatograph equipped with a ^{63}Ni electron capture detector (ECD) for N_2O analysis, an infrared gas analyzer for CO_2-measurements (BUSE Guardian II, Buse Anlagenbau, Germany), an automated gas sampling system, and five measuring chambers. For measurement of trace gas fluxes, the chambers were closed automatically for 1 h, and changes in N_2O and CO_2 concentrations were monitored every 15 min for each chamber. Rates of N_2O- and CO_2-emissions were calculated from the linear increase in N_2O and CO_2 concentrations within the chamber air. All flux rates were corrected for temperature and air pressure. Even in view to the destruction of sampling lines by rodents and short-term failure of the automatic measuring system under the humid and hot tropical conditions, more than 5780 emission rates—78% of the potential number—were obtained.

2.3. Determination of N_2O concentration at different soil depths

During the field campaign October 2000 to January 2001, N_2O concentrations were measured in triplicate at all sites at different soil depths (5, 10, 15 and 30 cm). For installation of the newly developed sampling tubes at each site, a 40 cm deep manhole was dug 2 months before the start of field measurements. From this manhole, holes with the same diameter as the sampling tubes, were drilled horizontally into the soil profile. A sampling tube consists of a gas permeable but hydrophobic Accurel®polypropylene tube (Akzo Nobel Faser AG, Germany) connected to an 1/8″ (3 mm diameter) steel tube (length approximately 50 cm), which were closed at the end with a rubber septum (Fig. 1). Soil air samples were taken every 3 d (at the actual measuring site) or weekly with 10 ml gas-tight syringes (Baton Rouge, Lousiana, USA) and analyzed for N_2O (Breuer et al., 2000) within a few hours after sampling.

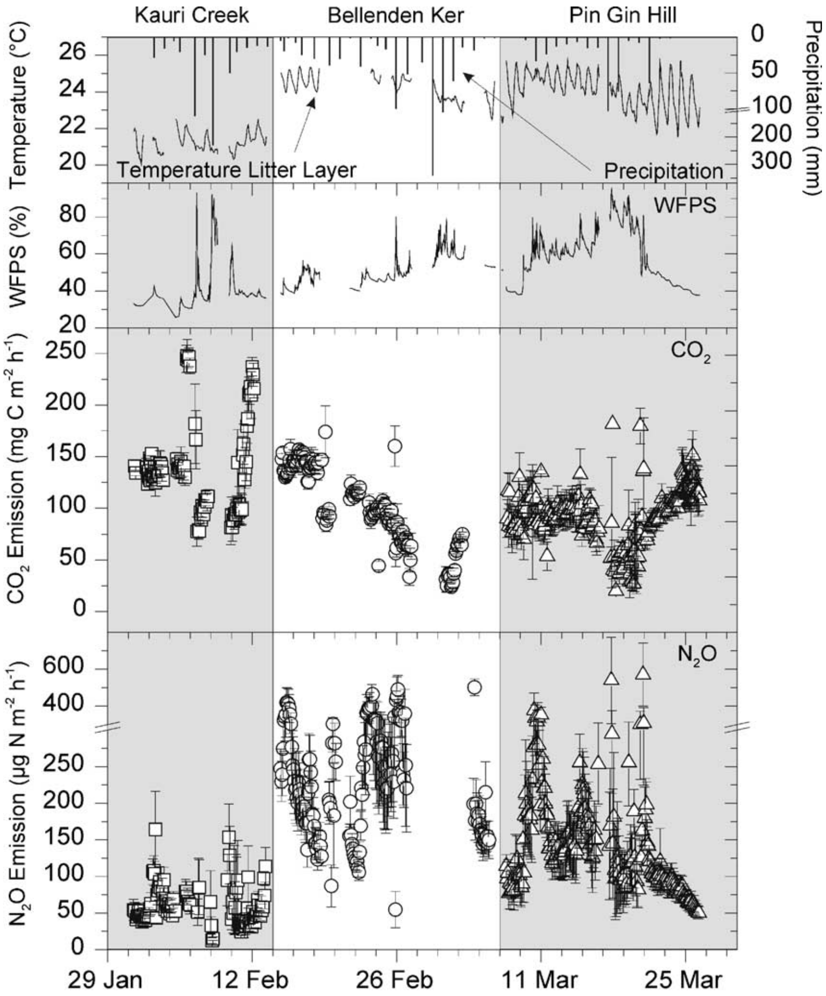


Fig. 2. Hourly/two hourly means of soil temperature (litter layer), water-filled pore space (WFPS), mean CO_2-emission rates ($N = 5$), N_2O-emission rates ($N = 5$), and daily precipitation at the Kauri Creek (□), Bellenden Ker (○), and Pin Gin Hill (△) sites during the wet season (January to March 2000).

2.4. Measurements of precipitation, temperature and soil moisture

Precipitation was measured daily as throughfall by means of three rain gauges (area: 50.3 cm^2). Measurements of temperature were carried out at 2 m above ground level (air) and at various soil depths (litter layer, mineral soil: 1, 5 and 10 cm) using PT100 thermocouples (IMKO, Germany). Changes in soil moisture were monitored by three TDR probes (UMS, Munich, Germany) which were placed inside the chambers. Further details on these measurements are described in detail by Breuer et al. (2000).

2.5. Statistical analyses

Due to non-normal distribution of N_2O emission rates, as tested by the Kolmogorov–Smirnov test, the non-parametric Mann–Witney test was used to identify differences between measuring chambers at a given site and for comparison of mean N_2O emissions between the three investigated sites. Correlation analyses were used to examine relationships between N_2O emissions, CO_2-emissions, N_2O concentrations in the soil profile, soil temperature and soil moisture. In order to determine the lag time of N_2O and CO_2 emission on changes in soil moisture, cross correlation analysis were performed. SPSS 8.0 (SPSS Inc., United States) was used for all statistical analyses.

3. Results

3.1. N_2O and CO_2 emission from soils

The use of the automatic measuring system resulted in a high temporal and spatial resolution of N_2O-emission rates for the two Coastal Lowland sites Pin Gin Hill and Bellenden Ker and for the Kauri Creek site at the Atherton Tablelands. The results obtained from measurements carried out during wet season are shown in Fig. 2. Diurnal variations in temperature in the litter layer was in maximum 4 °C, e.g. at the Pin Gin Hill site. Due to difference in

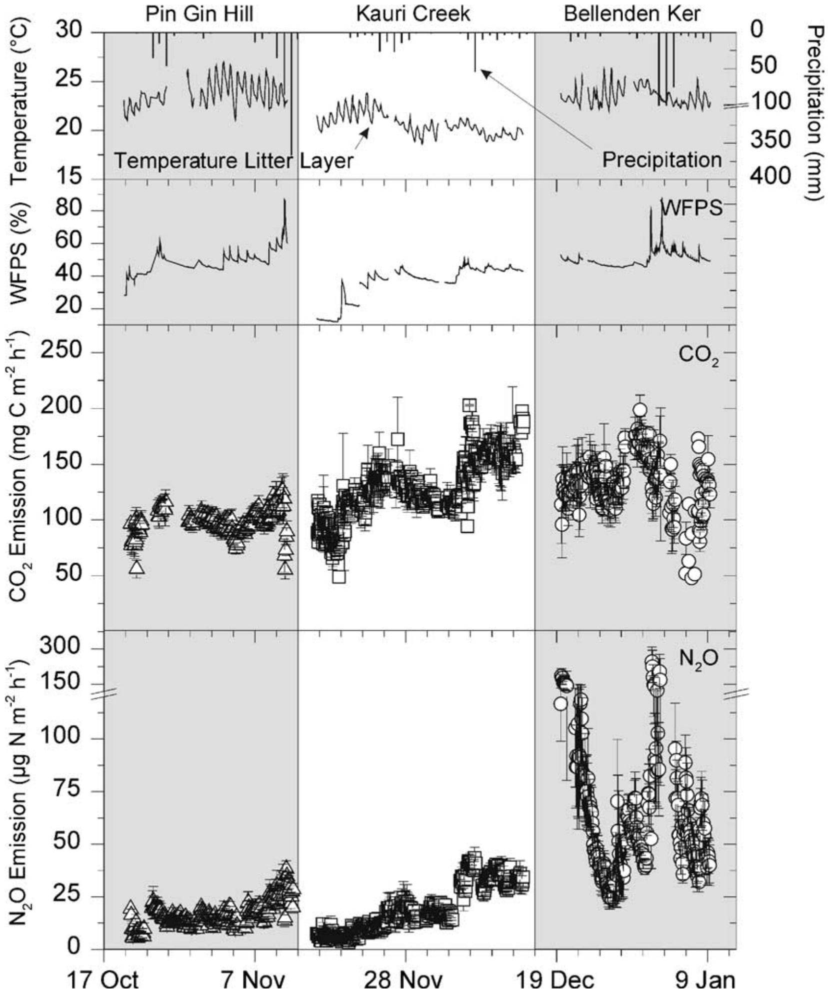


Fig. 3. Hourly/two hourly means of soil temperature (litter layer), water-filled pore space (WFPS), mean CO_2-emission rates ($N = 5$), N_2O-emission rates ($N = 5$), and daily precipitation at the Pin Gin Hill (△), and Kauri Creek (□) sites during the dry season and at the Bellenden Ker (○) site during the transition period to the wet season (October 2000 to January 2001).

altitude temperatures at the Coastal Lowland sites, Bellenden Ker and Pin Gin Hill are significantly higher as compared to the Atherton Tableland site Kauri Creek. At all sites, frequent heavy rainfall events resulted in huge short-term variations in soil moisture content (amplitude of WFPS at all sites: 30–80%). However, values of water-filled pore space were higher at the Coastal Lowland sites Bellenden Ker and Pin Gin Hill compared to the Atherton Tableland site Kauri Creek. N_2O emissions, at all sites, were strongly linked to changes in soil moisture resulting in the same pattern of huge short-term variation in N_2O emission rates with a significantly higher amplitude at the Pin Gin Hill and the Bellenden Ker site as compared to the Kauri Creek site. In contrast to N_2O emission, variation in CO_2 emission was significantly lower at the sites Bellenden Ker and Pin Gin Hill but was also very high at the Kauri Creek site.

During the dry season, at the sites in Pin Gin Hill and Kauri Creek, N_2O emission rates and values of WFPS were strongly reduced in magnitude and amplitude as compared to the results of the wet season (Figs. 2 and 3). However, this was not true for the Bellenden Ker site, since measurements of N_2O emissions—due to advanced re-wetting of the soil—have to be assigned to the transition period between the dry and the wet season.

Table 2 summarizes the results of N_2O emission measurements at all three sites during the investigation period 2000–2001. At all sites, highest N_2O emissions were recorded during the wet season lasting from January to April, due to a strong positive correlation of N_2O-emissions and rainfall events with increases in soil moisture. During this period of time, highest mean N_2O emissions at 242 ± 7.4 µg N_2O–N m^{-2} h^{-1} were found for the Bellenden Ker site, whereas N_2O-emissions at the Pin Gin Hill site were much lower (140.8 ± 5.1 µg N_2O–N m^{-2} h^{-1}), but still significantly higher than N_2O-emissions at the Kauri Creek site on the Atherton Tablelands (80.8 ± 3.3 µg N_2O–N m^{-2} h^{-1}, Fig. 2).

During the dry season, mean N_2O emissions dropped down to 18.9 ± 0.7 µg N_2O–N m^{-2} h^{-1} at the Kauri Creek and 16.6 ± 0.4 µg N_2O–N m^{-2} h^{-1} at the Pin Gin

Table 2
Means (±SE), minima and maxima of N_2O emissions at the different field sites for different field trips. Different superscripts above means indicate significant differences between different times and sites of field measurements ($P < 0.05$) (c_v: coefficient of variation; N: number of measured N_2O flux rates)

Site/measuring interval	N_2O emission rate ($\mu g\ N_2O–N\ m^{-2}\ h^{-1}$)				
	Mean	Minimum	Maximum	N	c_v (%)
Atherton Tablelands					
Kauri Creek					
31 January–14 February 2000[a]	80.8 ± 3.3[a]	12.3	167.7	108	42.5
15 November–15 December 2000[b]	18.9 ± 0.7[b]	3.3	43.5	292	60.2
Mean annual N_2O-emission[c]	4.36 kg N $ha^{-1}\ y^{-1}$				
Coastal Lowlands					
Bellenden Ker					
15 February–5 March 2000[a]	242.0 ± 7.4[c]	105.7	501.6	142	36.5
19 December 2000–9 January 2001[d]	66.6 ± 2.7[d]	23.9	247.3	209	57.6
Mean annual N_2O-emission[c]	7.45 kg N $ha^{-1}\ y^{-1}$				
Pin Gin Hill					
7–26 March 2000[a]	140.8 ± 5.1[e]	50.24	570.8	209	52.9
20 October–12 November 2000[b]	16.6 ± 0.4[b]	5.4	39.7	242	36.6
Mean annual N_2O-emission[c]	6.89 kg N $ha^{-1}\ y^{-1}$				

[a] Wet season.

[b] Dry season.

[c] Calculations of mean annual N_2O-emissions at all sites are seasonally weighted (6 months wet and 6 months dry season). Transition period at Bellenden Ker site is regarded as part of the wet season. For calculation of mean annual N_2O-emission for the Bellenden Ker site we assumed that mean N_2O-emissions during dry season were as high as at the Pin Gin Hill site.

[d] Transition period.

Hill site, respectively. As a result of more humid conditions in December 2000 to January 2001, the mean N_2O flux rate of 66.6 ± 2.7 $\mu g\ N_2O–N\ m^{-2}\ h^{-1}$ at the Bellenden Ker site had to be assigned to the transition period between dry and wet season. The intermediate flux rate of the transition period at the Bellenden Ker site was significantly different to wet and dry season emissions at all sites (Table 2, Fig. 3).

At all sites, the coefficient of variation (c_v) for N_2O emissions varied between 36.5 and 60.2%, indicating a moderate, though significant spatial variability (Table 2). For the Kauri Creek and Bellenden Ker sites, the c_v was higher under drier conditions, whereas at the Pin Gin Hill site, the c_v was higher under wet season conditions. The highest N_2O emission rate (570.8 $\mu g\ N_2O–N\ m^{-2}\ h^{-1}$) was determined in March 2000 at the Pin Gin Hill site, whereas the lowest N_2O emission rate (3.3 $\mu g\ N_2O–N\ m^{-2}\ h^{-1}$) was observed at the Kauri Creek site in November 2000. Uptake of atmospheric N_2O by the soils, i.e. N_2O-deposition, was never observed at any site.

Our investigations show that N_2O emission rates at Coastal Lowland sites (Bellenden Ker and Pin Gin Hill) during wet season conditions are approximately twice as high as compared to N_2O emission rates at the Atherton Tableland site Kauri Creek. During this investigation, temperature at the litter layer was 3 °C higher at both Coastal Lowland sites (23.6 °C) as compared to the more elevated Atherton Tableland site (20.6 °C). Furthermore, due to the climate effect of higher precipitation at the Coastal Lowlands mean values of water-filled pore space in the soil (WFPS) were higher at Bellenden Ker (50.4%) and Pin Gin Hill (57.9%) than at the Kauri Creek site (36.9%) (Fig. 2). During the dry season, when values for WFPS did not differ significantly between the Coastal Lowland site Pin Gin Hill and the Atherton Tableland site Kauri Creek (Fig. 3), N_2O emissions were not significantly different, even though mean temperature at the litter layer was approximately 3.0 °C higher at the Pin Gin Hill site (Pin Gin Hill: 23.7 °C, Kauri Creek: 20.6 °C).

Compared to the differences in mean N_2O emission rates between the three sites, variations in mean soil CO_2 emissions during wet and dry season were less pronounced (Table 3). Highest average CO_2 emissions of 137.3 ± 4.5 mg C $m^{-2}\ h^{-1}$ were observed at the Kauri Creek site in a period of frequent changes of heavy rainfall and drier periods during the wet season (Fig. 2). In contrast to the observation that CO_2 emission rates at Kauri Creek were highest in the wet season (137.3 ± 4.5 mg C $m^{-2}\ h^{-1}$), the Coastal Lowland sites showed highest mean CO_2 emission rates during the dry season (Pin Gin Hill: 98 ± 0.8 mg C $m^{-2}\ h^{-1}$) or during the transition period from the dry to the wet season (Bellenden Ker: 133.1 ± 1.8 mg C $m^{-2}\ h^{-1}$). Significant differences in the magnitude of soil CO_2 emissions could be identified between all three sites under wet as well as under dry conditions in the following sequence: Kauri Creek > Bellenden Ker ≥ Pin Gin Hill.

The coefficients of variation for CO_2 emission rates at the different sites were in a range of 11.5–36.7% and, thus, approximately only half of the coefficient of variation for

Table 3
Means (±SE), minima and maxima of CO_2 emissions at the different field sites for different field trips. Different superscripts above means indicate significant differences between different times and sites of field measurements ($P < 0.05$) (c_v: coefficient of variation; N: number of measured N_2O flux rates)

Site/measuring interval	CO_2 emission rate (mg C m^{-2} h^{-1})				
	Mean	Minimum	Maximum	N	c_v%
Atherton Tablelands					
Kauri Creek					
31 January–14 February 2000[a]	137.3 ± 4.5[a]	75.9	247.7	94	31.5
15 November–15 December 2000[b]	125.8 ± 1.5[b]	48.9	202.9	360	22.2
Coastal Lowlands					
Bellenden Ker					
15 February–5 March 2000[a]	102.2 ± 3.2[c]	24.0	174.1	135	36.7
19 December 2000–9 January 2001[c]	133.3 ± 1.8[a]	48.0	198.7	188	18.2
Pin Gin Hill					
7–26 March 2000[a]	92.2 ± 1.8[d]	20.3	181.6	230	28.9
20 October–12 November 2000[b]	98.4 ± 0.8[c]	55.4	126.4	180	11.5

[a] Wet season.
[b] Dry season.
[c] Transition period.

N_2O emissions (Table 3). In contrast to N_2O emissions, lower coefficients of variation of CO_2 emissions were always observed during drier periods.

3.2. Dependency of N_2O emissions on changes in soil temperature and WFPS

To identify the effect of changes in soil temperature and WFPS on the magnitude of N_2O emission, correlation analyses were performed. At all three sites, statistical analysis failed to reveal a significant influence of soil temperature on N_2O emissions. This is mainly due to only small daily changes (<2 °C) in soil temperature values at soil depth of 1 cm. Even with regard to the entire observation period at each site, the amplitude of soil temperature was only 2–6 °C. To further support the finding in the field that N_2O-emissions were rather insensitive to changes in temperature, soil samples were incubated at different temperatures in the laboratory. The results of these experiments are shown in Fig. 4A. The figure shows that N_2O production remained almost constant at approximately 10 μg N kg^{-1} SDW d^{-1} for the Bellenden Ker and Kauri Creek sites and did not change significantly even when temperature was increased from 14.5 to 27 °C. When the incubation temperature was further increased above 30 °C, N_2O production rates raised up to 41.5 ± 12.7 μg N kg^{-1} SDW d^{-1} at the Kauri Creek site and to 64.7 ± 10.2 μg N kg^{-1} SDW d^{-1} at the Bellenden Ker site. However, for the Pin Gin Hill site, a significant dependency between increases in temperature and N_2O-emissions could be demonstrated by laboratory incubation experiments ($r = 0.9$, $P < 0.01$). Compared to the other two investigated sites, the results of the incubation studies revealed that the Pin Gin Hill site showed highest N_2O-emissions (27 °C: 67.0 ± 30.0 μg N kg^{-1} SDW d^{-1}). It has to be mentioned that values of WFPS of incubated soil samples were highest from the Pin Gin Hill site (34.2%) due to a higher water holding capacity resulting from a significant higher clay content (Table 1) as compared to the Bellenden Ker (24.5%) and Kauri Creek sites (22.5%).

In contrast to temperature, a detailed correlation analysis revealed a close relationship between changes in in situ soil moisture (WFPS) and magnitude of in situ N_2O emission rates. During the dry season the strong positive relationship between soil moisture and magnitude of N_2O-emissions could be demonstrated for all sites (Kauri Creek: $r = 0.74$, $P < 0.01$; Pin Gin Hill: $r = 0.70$, $P < 0.01$; Bellenden Ker: $r = 0.58$, $P < 0.01$). However, during the wet season—taking the entire datasets—only for the Pin Gin Hill site, a significant positive correlation ($r = 0.44$, $P < 0.01$) between changes in WFPS and N_2O emission rates could be demonstrated (Table 4). At all sites, the correlation between changes in soil moisture and changes in N_2O-emissions could be significantly improved if data above a certain threshold of WFPS were excluded from the analysis. For the Pin Gin Hill site, exclusion of observations of N_2O-emissions above 60% WFPS improved the correlation to $r = 0.71$, $P < 0.01$. For the sites at Bellenden Ker and Kauri Creek, exclusion of data observed at values of WFPS > 50% revealed that, during the wet season too a strong positive correlation between changes in WFPS and changes in N_2O-emissions did exist (Bellenden Ker: $r = 0.56$, $P < 0.01$; Kauri Creek: $r = 0.48$, $P < 0.01$). The higher threshold for the Pin Gin Hill site can be explained by the significantly higher clay content causing differences in the distribution of pore geometry. Macropores, responsible for gas diffusion processes, e.g. oxygen, were blocked earlier with water in the sandy clay loams of the Bellenden

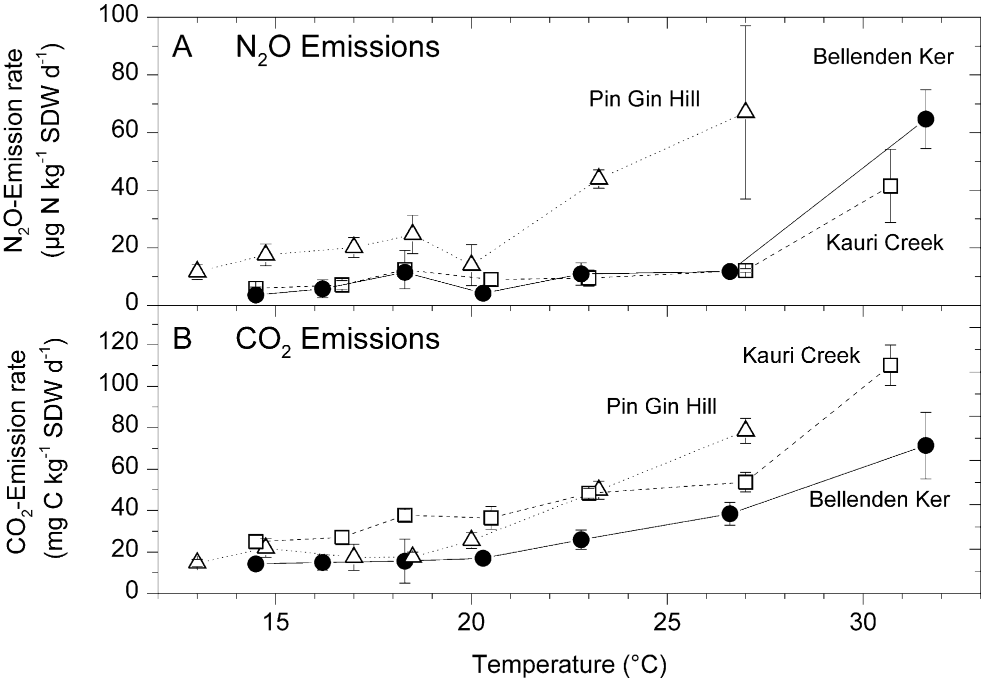


Fig. 4. Dependency of N_2O-emission rates (A) and CO_2-emission rates (B) (±SE, $N = 3$) on changes in temperature during laboratory incubation studies of soil samples. (△): Pin Gin Hill, (□): Kauri Creek, (●): Bellenden Ker. SDW: soil dry weight.

Ker and Kauri Creek site as compared to the clay soil of the Pin Gin Hill site (Table 4).

Cross correlation analysis revealed that increases in soil moisture almost immediately led to increases in N_2O-emissions at the Kauri Creek and Pin Gin Hill sites, but was delayed by 2 h at the Bellenden Ker site. The immediate response of N_2O emissions to increase in soil moisture was also found during an irrigation experiment carried out at the end of the dry season 2000 (2–4 November) at the Pin Gin Hill site. During this experiment, three of five chambers were irrigated daily with 40 mm water and changes in N_2O-emissions were compared to the two control chambers. Fig. 5 shows mean N_2O emission rates of the control and irrigated chambers. During the experiment, mean N_2O emissions of the control chamber decreased from 20 to 10 µg N m^{-2} h^{-1} in agreement with decreasing values of WFPS. In contrast to the control chambers, mean N_2O-emissions in the irrigated chambers almost doubled from a base emission of 12 to a maximum value of 22.9 µg N m^{-2} h^{-1} during this period. At the end of the experiment, on 6th November, N_2O-emissions in both chambers, i.e. in the irrigated as well as in the 'dry' chamber, were larger, due to increases in soil moisture caused by a rainfall event. However, mean N_2O-emission of the control chambers (32.3 µg N m^{-2} h^{-1}) were twice as high as mean N_2O emission of the irrigated chambers (16.8 µg N m^{-2} h^{-1}).

3.3. Dependency of CO_2 emissions on changes in soil temperature and WFPS

At all the three sites, correlation analysis failed to demonstrate a significant relationship between changes in soil temperature and changes in in situ soil CO_2 emissions. However, laboratory experiments showed a significant positive relationship between soil CO_2-emissions and changes in temperature (Fig. 4B). This discrepancy between field observations and laboratory investigation is due to an overlying effect in driving the magnitude of CO_2 emission

Table 4
Results of correlation analysis between changes in water filled pores space (WFPS) and changes in situ N_2O- and CO_2-emission rates, $^{**}P < 0.01$

Correlation WFPS with		Kauri Creek		Pin Gin Hill		Bellenden Ker	
		N_2O	CO_2	N_2O	CO_2	N_2O	CO_2
Wet season	All data	−0.21	−0.41**	0.44**	−0.61**	0.18	−0.71**
	Improved[a]	0.48**		0.71**		0.56**	
Dry season	All data	0.74**	0.79**	0.70**	0.60**	0.58**[b]	−0.27**[b]

[a] Exclusion of data above threshold of WFPS: Pin Gin Hill >60%, Kauri Creek and Bellenden Ker >50%.

[b] Classified as transition period.

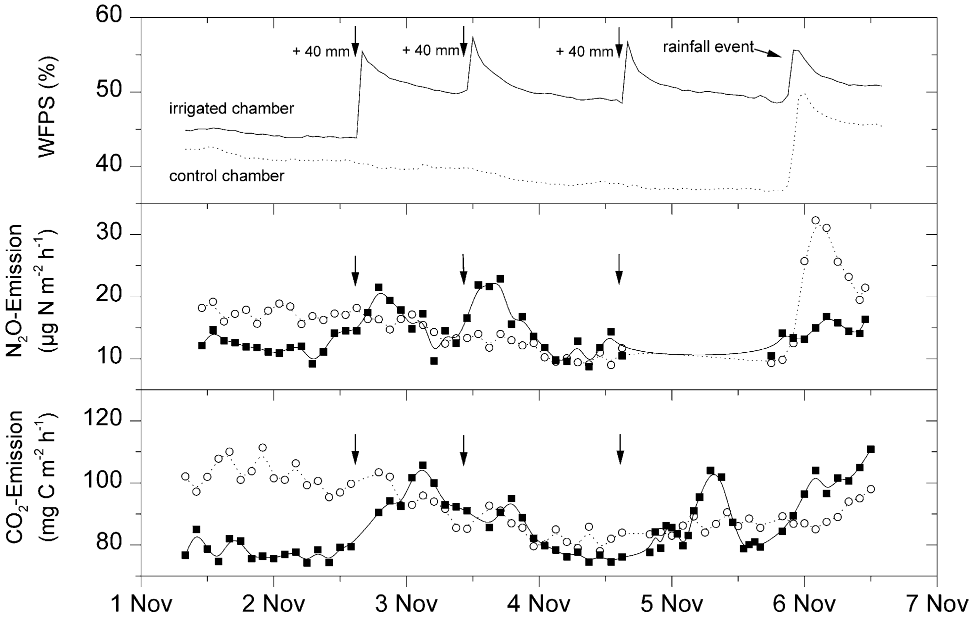


Fig. 5. Two hourly rates of N_2O-emissions, CO_2-emissions, and water-filled pore space (WFPS) of artificially irrigated (■) and control chambers (○), respectively, during the dry season at the Pin Gin Hill site (November 2000).

of soil moisture compared to soil temperature. The figure shows that soils taken from the Pin Gin Hill and Bellenden Ker sites did not show significant variations in CO_2 emission rates in a temperature range of 13–20 °C. For temperatures >20 °C, CO_2 production increased linearly from values of approximately 20 to values up to 78.5 ± 6.0 mg C kg^{-1} SDW d^{-1} (27 °C) at the Pin Gin Hill site and 71.4 ± 16.1 mg C kg^{-1} SDW d^{-1} (31.6 °C) at the Bellenden Ker site. In contrast to both lowland sites (Bellenden Ker and Pin Gin Hill), soils taken from the site at Kauri Creek already showed a positive relationship between increasing CO_2 emission and temperature starting from 15 °C with highest measured CO_2 emission rate of 110.1 ± 9.8 mg C kg^{-1} SDW d^{-1} (30.7 °C). Q_{10} values, calculated for the temperature range 20–30 °C, were 3.0 (Kauri Creek), 3.6 (Bellenden Ker) and 5.0 (Pin Gin Hill), respectively.

The effect of soil moisture on the magnitude of CO_2 emission strongly depended on the period investigated, i.e. dry season, intermediate period or wet season. During the wet season, CO_2 emission rates at all sites were negatively correlated with increasing WFPS (Table 4), whereas at the end of the dry season, re-wetting of the soil by rainfall events stimulated CO_2 emission e.g. at the Pin Gin Hill ($r = 0.6$, $P < 0.01$) and Kauri Creek ($r = 0.79$, $P < 0.01$) site.

3.4. N_2O concentration at different soil depths

At all three sites, N_2O-concentrations increased with soil depth from 5 to 30 cm. Furthermore, N_2O-concentrations in the soil were positively correlated to increases in soil moisture due to rainfall events (Fig. 6). The coefficients of determination for this relationship were: $r^2 = 0.93$ for the Bellenden Ker site, $r^2 = 0.91$ for the Pin Gin Hill site and $r^2 = 0.89$ for the Kauri Creek site. The increase in N_2O concentrations with soil depth was most pronounced at the Bellenden Ker site, where mean N_2O concentrations for the entire investigation increased by 38.5 ± 10.3 nl $N_2O\ l^{-1}\ cm^{-1}$ soil depth. This N_2O-gradient for the Bellenden Ker site was significantly steeper ($P < 0.001$) than for the sites at Pin Gin Hill (9.2 ± 4.4 nl $N_2O\ l^{-1}\ cm^{-1}$) and Kauri Creek (11.0 ± 1.4 nl $N_2O\ l^{-1}\ cm^{-1}$). This ranking is in excellent agreement with the ranking in N_2O-emissions for the same investigation period at the different sites, i.e. Bellenden Ker > Kauri Creek > Pin Gin Hill. A correlation analysis between N_2O-concentrations in the soil and magnitude of N_2O-emissions for daily values further supported this finding (mean $r^2 = 0.75$, $P < 0.01$).

4. Discussion

Using the mobile and fully automated measuring system for online analysis of N_2O emissions from tropical rain forest soils, we were able to obtain about 5800 emission rates at three sites during two field campaigns in 2000 and 2001. Based on this spatially as well as temporally highly resolved data set, significant differences in the magnitude of N_2O emissions of the three investigated sites, i.e. Kauri Creek at the Atherton Tablelands as well as Bellenden Ker and Pin Gin Hill in the Coastal Lowlands of North-eastern Australia, could be demonstrated. At all sites, nitrous oxide emissions varied seasonally with low emissions during the dry season and highest N_2O emissions during the wet season as was also observed by others (Garcia Méndez et al., 1991; Steudler et al., 1991; Verchot et al., 1999; Breuer et al., 2000). The highest mean

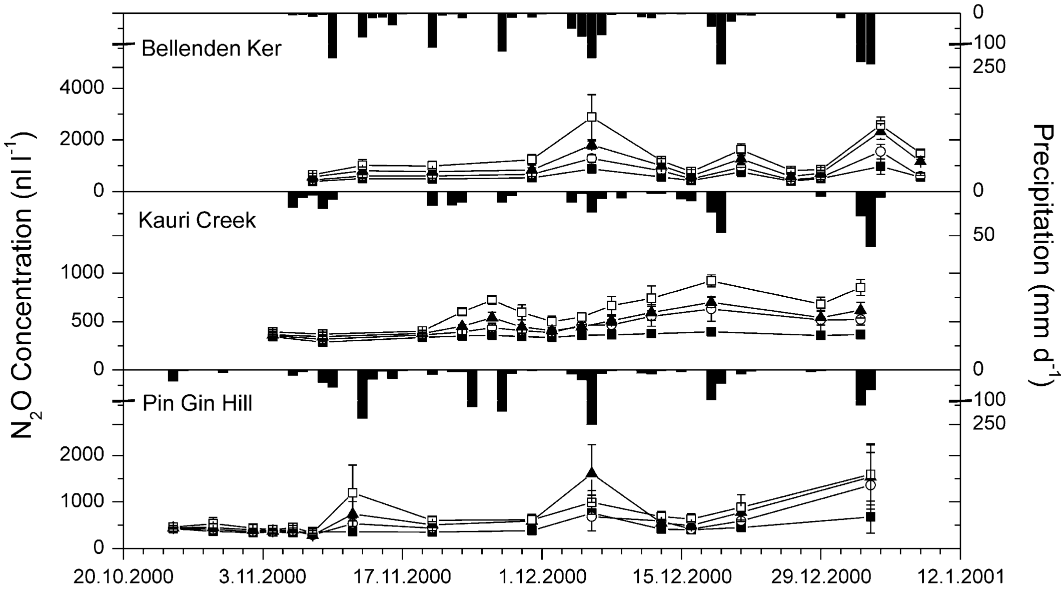


Fig. 6. Soil air N_2O concentration (±SE, $N = 3$) in (■) 5 cm, (○) 10 cm, (▲) 15 cm and (□) 30 cm soil depth and values of daily precipitation at the Bellenden Ker, Kauri Creek and Pin Gin Hill sites.

N_2O-emission rate was observed during the wet season at the Bellenden Ker site (240.0 ± 7.4 μg N m^{-2} h^{-1}) with maximum N_2O-emission rates of up to 501.6 μg N m^{-2} h^{-1}. These rates are among the highest N_2O-emission rates so far recorded for tropical rain forest soils. At the other two sites, i.e. Pin Gin Hill (Coastal Lowland) and Kauri Creek (Atherton Tablelands), mean N_2O emission rates during the wet season were lower (Pin Gin Hill: 140.8 ± 5.1 μg N m^{-2} h^{-1}; Kauri Creek: 80.8 ± 3.3 μg N m^{-2} h^{-1}). The coefficient of variation c_v for N_2O emission rates of all sites is low and in a range of 36.5–60.2% (Table 2). For N_2O emission from tropical rain forest soils, others (Vitousek et al., 1989; Verchot et al., 1999) reported values for c_v between 94 and 196%, and for a beech and spruce forest ecosystem in Germany, Papen and Butterbach-Bahl (1999) found even higher c_v values of up to 215.6%.

Our data show generally higher N_2O emission rates at the Coastal Lowland sites as compared to the Atherton Tableland site. This is mainly related to two environmental factors: (a) to higher values of mean annual temperature (approximately 3.5 °C) and (b) even more important, to higher values of mean annual precipitation at the Coastal Lowland sites causing higher values of WFPS at these sites at least during the wet season. A correlation between changes in soil temperature and in situ N_2O emissions could not be demonstrated. This is mainly due to an overlaying effect of changes in WFPS where correlation analysis revealed a strongly positively correlation with in situ N_2O-emissions. This observation is in good agreement with results from other field and laboratory studies (Garcia Méndez et al., 1991; Davidson et al., 1993). In contrast to the findings by Keller and Reiners (1994) who found an exponential relationship between N_2O-emissions and WFPS, N_2O-emissions at our sites were correlated linearly to changes in WFPS for values of WFPS <50–60%. For values of WFPS > 50–60%, a decrease in N_2O emission rates was observed, which is in good agreement with observations by Riley and Vitousek (1995), who described a reduction in N_2O-emissions during times with high soil moisture values for some montane rain forest sites in Hawaii. Linn and Doran (1984) found that, at a threshold of 50–60% WFPS, nitrification rates in soils are in optimum, since at this value increasing substrate availability and decreasing availability of oxygen with increasing soil water content still favor nitrification processes in soils. With regard to our sites, this would imply that fluxes below 50–60% are mainly due to nitrification, whereas above 50–60%, one has to assume that denitrification is the major process responsible for N_2O emissions. The decrease in N_2O-emissions for values of WFPS > 60% indicates that the main product of denitrification at our sites is N_2 rather than N_2O. Though there is no doubt, that soil moisture is one of the key driving variables for the observed site differences in N_2O-emissions, differences in additional site factors further modulate in situ N_2O-emissions at our sites. The lower N_2O-emissions at the Pin Gin Hill as compared to Bellenden Ker site are most likely due to lower mineralization rates, as indicated by lower values of soil respiration (lower CO_2 emission rates), thus limiting the availability of substrates for microbial N-turnover processes involved in N-trace gas production such as nitrification and denitrification. Furthermore, the high clay content at the Pin Gin Hill site (67%) might protect the organic matter from mineralization as was described for soils in the Amazon basin by Van Veen and Kuikman (1990) and Koutika et al. (1999). This explanation would also be supported by the fact that the organic C-content in the soil at the Pin Gin Hill site is

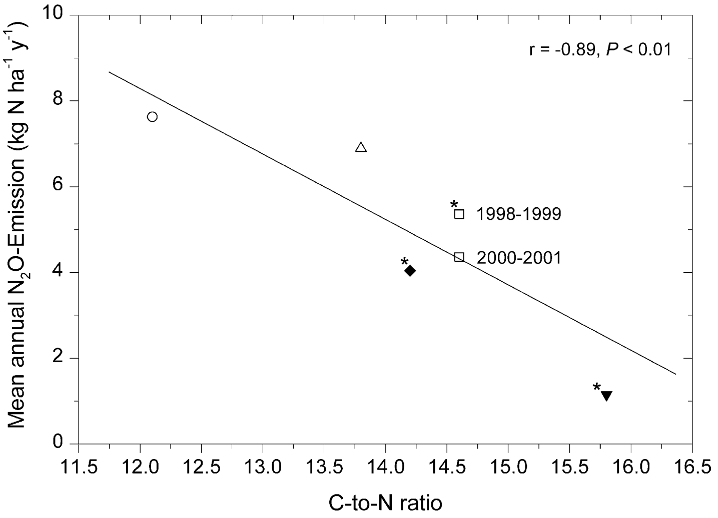


Fig. 7. Correlation between the C-to-N ratios and mean annual N_2O-emission rates for different Australian rain forest sites: Bellenden Ker = (○), Pin Gin Hill = (△), Kauri Creek = (□), Massey Creek = (◆), and Lake Eacham = (▼). * data from Breuer et al. (2000).

approximately threefold higher than at the Bellenden Ker site (Table 1).

When explaining site differences in N_2O-emissions from tropical forest soils, the C-to-N ratio of the organic matter has to be considered, as demonstrated by Breuer et al. (2000). Narrower C-to-N-ratios support higher rates of microbial C and N turnover in the litter and top soil layer and, consequently, support high rates of N_2O emissions from soils. Such a close relationship between C-to-N ratios in soils and the magnitude of N_2O-emissions can also be found for our data set. The ranking in the magnitude of annual N_2O-emissions between our sites, i.e. Bellenden Ker > Pin Gin Hill > Kauri Creek, is in excellent agreement with the ranking in the C-to-N ratio of the soil organic matter: Kauri Creek (14.6) > Pin Gin Hill (13.8) > Bellenden Ker (12.1) (Table 1). A strongly negative correlation ($r = 0.89$, $P < 0.01$) between C-to-N ratio and the mean annual N_2O-emission rate can be demonstrated, extending with data from Breuer et al. (2000) for the Kauri Creek site and two additional sites on the Atherton Tablelands (Fig. 7).

Another soil factor, which should to be taken into account to explain the higher N_2O-emissions at the Coastal Lowland sites as compared to the Atherton Tablelands, might be the lower soil pH values. It is well known that low pH values decrease the activity of N_2O-reductase—the last enzyme in the denitrification pathway—thereby increasing the release of N_2O from denitrification (Weier and Gillam, 1986; Granli and Bockman, 1994). In addition, for nitrification, it has been demonstrated that low pH values are in favor of increased loss rates of N_2O during nitrification (Sitaula and Bakken, 1993; Martikainen and De Boer, 1993).

The measured CO_2 emissions at our sites ranged from 24.0 to 247.7 mg C m^{-2} h^{-1}. These values are within the same range as recorded by others, e.g. La Scala et al. (2000), who reported CO_2 emission ranging from 54.4 to 107.2 mg C m^{-2} h^{-1} for a Brazilian bare soil, with coefficients of variation from 21.1 to 31.7%, values which are also very similar to ours (11.5–36.7%). In addition, values reported by Holt et al. (1990), who measured CO_2 emission rates from 9.6 mg C m^{-2} h^{-1} during dry season up to 133.6 mg C m^{-2} h^{-1} under wet season conditions for a rainforest site located 100 km south of our sites, are in good agreement with our observations.

During periods with moderate soil moisture values, CO_2 emission rates were positively correlated with increasing soil moisture. Whereas for values exceeding 50–60% WFPS, the relationship between soil CO_2 emissions and soil moisture suddenly became negative. This further supports our hypothesis that above values of WFPS of >60%, O_2 diffusion gets limited in the soil matrix, thus limiting heterotrophic respiration and favoring denitrification. A gradual decline in respiration due to increasing soil moisture during the wet season was also observed by Maggs and Hewett (1990) for two other tropical sites in North Queensland. In addition, Davidson et al. (1998) observed a positive correlation between CO_2 emission and WFPS under low soil moisture conditions, but a negative correlation between soil moisture and soil CO_2-emissions for periods with high soil moisture contents. Also the observation that highest CO_2 emission rates at Kauri Creek site were recorded in a period of frequent drying and re-wetting events with values >250 mg C m^{-2} h^{-1}, is in accordance with other investigations which show that C-mineralization and soil CO_2-emissions are stimulated by drying and re-wetting cycles (Sørensen, 1974; Vangestel et al., 1993).

The significant seasonal variability of mainly N_2O, but also of CO_2 emission clarifies the importance of temporal as well as spatial highly resolved data to minimize errors when calculating the annual source strength of tropical rainforest ecosystems for these gases. As many factors influence N_2O and CO_2 emissions, e.g. changes in WFPS, C-to-N ratio, changes in soil temperature, pH, soil texture, vegetation etc., future scientific work should focus on development or modification of process-oriented models (Li et al., 2000), which already have been successfully validated for temperate (Stange et al., 2000; Butterbach-Bahl et al., 2001) but not for tropical rain forest ecosystems.

Acknowledgements

The authors thank Martin Kock for his excellent assistance during the field trips, Bob Hewett, Andrew Graham, and many helping hands from the Tropical Forest Research Centre (CSIRO) at Atherton, QLD, Australia for their idealistic support and generous hospitality, and Hans Papen for critical reading and his valuable comments on the manuscript. This research was supported by contracts from the Deutsche Forschungsgesellschaft (DFG).

References

Breuer, L., Butterbach-Bahl, K., Papen, H., 2000. N_2O emission from tropical forest soils of Australia. Journal of Geophysical Research 105, 26353–26367.

Bouwman, A.F., Taylor, J.A., 1996. Testing high-resolution nitrous oxide emission estimates against observation using an atmospheric transport model. Global Biogeochemical Cycles 10, 307–318.

Butterbach-Bahl, K., Stange, F., Papen, H., Li, C., 2001. Regional inventory of nitric oxide and nitrous oxide emissions for forest soils of Southeast Germany using the biogeochemical model PnET-N-DNDC. Journal of Geophysical Research 106, 34155–34166.

Davidson, E.A., Matson, P.A., Vitousek, P.M., Riley, R., Dinkin, K., Garcia-Méndez, G., Maass, J.M., 1993. Processes regulating soil emission of NO and N_2O in a seasonally dry tropical forest. Ecology 74, 130–139.

Davidson, E.A., Belk, E., Boone, R., 1998. Soil water content and temperature as independent or confounded factors controlling soil respiration in a temperate mixed hardwood forest. Global Change Biology 4, 217–227.

Garcia-Méndez, G.J., Maass, J.M., Matson, P.A., Vitousek, P.M., 1991. Nitrogen transformations and nitrous oxide flux in a tropical deciduous forest in Mexico. Oecologia 88, 362–366.

Granli, T., Bockman, O.C., 1994. Soil water content. Nitrous Oxide from Agriculture, Wiig, M. (Ed.). Norwegian Journal of Agricultural Sciences (Suppl. 12), 34–42.

Goreau, T.J., De Mello, W.Z., 1998. Tropical deforestation: some effects on atmospheric chemistry. Ambio 17, 275–281.

Holt, J.A., Hodgen, M.J., Lamb, D., 1990. Soil respiration in the seasonally dry tropics near Townsville, North Queensland. Australian Journal of Soil Research 28, 737–745.

Houghton, J.T., Meira Filho, L.G., Callander, B.A., Harris, N., Kattenberg, A., Maskell, K. (Eds.), 1996. Climate Change 1995: The Science of Climate Change Cambridge University Press, New York.

IPCC, 1997. Intergovernmental Panel on Climate Change Guidelines for National Greenhouse Gas Inventories. Agriculture: Nitrous oxide from Agricultural Soils and Manure Management. OECD, Paris Chapter 4.

Keller, M., Reiners, W.A., 1994. Soil atmosphere exchange of nitrous oxide, nitric oxide, and methane under secondary succession of pasture to forest in the Atlantic lowlands of Costa Rica. Global Biogeochemical Cycles 8, 399–409.

Keller, M., Veldkamp, E., Weitz, A.M., Reiners, W.A., 1993. Effect of pasture age on soil trace gas emissions from a deforested area of Costa Rica. Nature 365, 244–246.

Koutika, L.S., Choné, T., Andreux, F., Burtin, G., Cerri, C.C., 1999. Factors influencing carbon decomposition of topsoils from the Brazilian Amazon Basin. Biology and Fertility of Soils 28, 436–438.

Laffan, M.D., 1988. Soils and Land Use on the Atherton Tableland, North Queensland. , Soils and Landuse Series, vol. 61. CSIRO, Melbourne.

La Scala, N., Marques, J., Pereira, G.T., Cora, J.E., 2000. Short-term temporal changes in the spatial variability model of CO_2 emission from a Brazilian bare soil. Soil Biology & Biochemistry 32, 1459–1462.

Li, C., Aber, J., Stange, F., Butterbach-Bahl, K., Papen, H., 2000. A process-oriented model of N_2O and NO emission from forest soils, 1. Model development. Journal of Geophysical Research 105, 4369–4384.

Linn, D.M., Doran, J.W., 1984. Effect of water-filled pore space on carbon dioxide and nitrous oxide production in tilled and nontilled soils. Soil Science Society American Journal 48, 1267–1272.

Maggs, J., Hewett, B., 1990. Soil and litter respiration in rainforest of contrasting nutrient status and physiognomic structure near Lake Eacham, north-east Queensland. Australian Journal of Ecology 15, 329–336.

Martikainen, P.J., De Boer, W., 1993. Nitrous oxide production and nitrification in acidic soil from a Dutch coniferous forest. Soil Biology & Biochemistry 25, 343–347.

Mosier, A., Kroeze, C., 1998. A new approach to estimate emissions of nitrous oxide from agriculture ant its implication to the global N_2O budget. IGACtivities NewsLetter 12, 17–25.

Myhre, G., Highwood, E.K., Shine, K.P., Stordal, F., 1998. New estimates of radiative forcing due to well mixed greenhouse gases. Geophysical Research Letters 25, 2715–2718.

Papen, H., Butterbach-Bahl, K., 1999. A three year continuous record of nitrogen trace gas fluxes from untreated and limed soil of a N-saturated spruce and beech forest ecosystem in Germany, 1. N_2O emissions. Journal of Geophysical Research 104, 18487–18503.

Riley, H.R., Vitousek, P.M., 1995. Nutrient dynamics and nitrogen trace gas flux during ecosystem development in montane rain forest. Ecology 76, 292–304.

Serca, D., Delmas, R., Jambert, C., Labroue, L., 1994. Emissions of nitrogen oxides from equatorial rain forest in central Africa: origin and regulation of NO emission from soils. Tellus 46B, 243–254.

Sitaula, B.K., Bakken, L.R., 1993. Nitrous oxide release from spruce forest soil: relationships with nitrification, methane uptake, temperature, moisture and fertilization. Soil Biology & Biochemistry 25, 1415–1421.

Sørensen, L.H., 1974. Rate of decomposition of organic matter in soil as influenced by repeated air drying–rewetting and repeated additions of organic material. Soil Biology & Biochemistry 6, 287–292.

Spain, A.V., Unwin, G., Sinclair, D.F., 1989. Soils and vegetation at three rainforest sites in tropical north-eastern Queensland, Division Report 15. Division of Soils, CSIRO, Melbourne.

Stange, F., Butterbach-Bahl, K., Papen, H., Zechmeister-Boltenstern, C., Li, C., Aber, A., 2000. A process-oriented model of N_2O and NO emission from forest soils, 2. Sensitivity analysis and validation. Journal of Geophysical Research 105, 4385–4398.

Steudler, P.A., Melillo, J.M., Bowden, R.D., Castro, M.S., Lugo, A.E., 1991. The effect of natural and human disturbance on soil nitrogen dynamics and trace gas fluxes in a Puerto Rican wet forest. Biotropica 23, 356–363.

Tracey, J.G., 1982. The Vegetation of the Humid Tropical Region of North Queensland. CSIRO, Melbourne.

Vangestel, M., Merckx, R., Vlassak, K., 1993. Microbial biomass and activity in soils with fluctuating water contents. Geodinamica Acta 56, 617–626.

Van Veen, J.A., Kuikman, P.J., 1990. Soil structural aspects of decomposition of organic matter by micro-organisms. Biogeochemistry 11, 213–233.

Verchot, L.V., Davidson, E.A., Cattânnio, J.H., Ackermann, I.L., Erickson, H.E., Keller, M., 1999. Land use and biogeochemical controls of nitrogen oxide emissions from soils in eastern Amazonia. Global Biogeochemical Cycles 13, 31–46.

Vitousek, P.M., Matson, P., Volkmann, C., Maass, J.M., Garcia-Méndez, G., 1989. Nitrous oxide flux from dry tropical forests. Global Biogeochemical Cycles 3, 375–382.

Watson, R.T., Meira Filho, L.G., Sanhueza, E., Janetos, A., 1992. Greenhouse gases: sources and sinks. In: Houghton, J.T., Callander, B.A., Varney, S.K. (Eds.). Climate Change 1992: The Supplementary Report to the IPCC Scientific Assessment. Cambridge University Press, New York.

Weier, K.L., Gillam, J.W., 1986. Effect of acidity on nitrogen mineralization and nitrification in Atlantic Coastal Plain soils. Soil Science Society of America Journal 50, 1210–1214.

Nitrification activity in tropical rain forest soils of the Coastal Lowlands and Atherton Tablelands, Queensland, Australia

Ralf Kiese[1], **Hans Papen**[1], **Elisabeth Zumbusch**[1], and **Klaus Butterbach-Bahl**[1]*

[1]Institute of Meteorology and Climate Research, Department for Atmospheric Environmental Research, Forschungszentrum Karlsruhe GmbH, Kreuzeckbahnstr. 19, D-82467 Garmisch-Partenkirchen, Germany

Accepted 31 August 2002

Summary – Zusammenfassung

Intact soil cores from a montane tropical rain forest site in the Atherton Tablelands (Kauri Creek) and from a lowland tropical rain forest site in the Coastal Lowlands (Bellenden Ker), Queensland, Australia were investigated during different hygric seasons for the magnitude of gross nitrification rates using the Barometric Process Separation technique (BaPS). Pronounced seasonal variations of gross nitrification rates were found at both sites with highest values during the transition period between dry and wet season (montane site: 24.0 mg N (kg SDW)$^{-1}$ d^{-1}; lowland site: 13.1 mg N (kg SDW)$^{-1}$ d^{-1}) and significantly lower rates of gross nitrification during the dry and wet season. Rates of gross nitrification were always higher at the montane site than at the lowland site, but the opposite was found for N_2O emissions. The results indicated that the high losses of N_2O at the lowland tropical rain forest site may be contributed largely by high denitrification activity due to its wetter and warmer climate as compared to the dryer and colder climate at the montane tropical rain forest site. This conclusion was supported by analysis of cell numbers of microbes involved in N-cycling. Higher numbers of denitrifiers were present at the lowland site, whereas higher numbers of nitrifiers were found at the montane site.

Key words: nitrification / denitrification / nitrifiers / denitrifiers / N_2O emission / tropical rain forest soils / BaPS method

Nitrifikationsaktivität in tropischen Regenwaldböden im küstennahen Tiefland und in den Atherton Tablelands, Queensland, Autralien

Mit Hilfe der Methode zur Barometrischen Prozess Separation (BaPS) wurden Brutto-Nitrifikationsraten intakter Bodenkerne von einem Bergregenwald-Standort auf den Atherton Tablelands (Kauri Creek) und einem Tieflandregenwald (Bellenden Ker) in den Coastal Lowlands, Queensland, Australien, während unterschiedlicher Jahreszeiten bestimmt. An beiden Standorten konnte ein ausgeprägter saisonaler Verlauf der Brutto-Nitrifikationsraten festgestellt werden, mit maximalen Werten während des Übergangs von Trocken- zur Regenzeit (Bergregenwald: 24,0 mg N (kg SDW)$^{-1}$ d^{-1}; Tieflandregenwald: 13,1 mg N (kg SDW)$^{-1}$ d^{-1}) und signifikant niedrigeren Werten in der Trocken- und Regenzeit. Während die Brutto-Nitrifikationsraten am Bergregenwald-Standort zu allen Jahreszeiten höher als am Tieflandregenwald-Standort waren, ergab sich für die Höhe der N_2O-Emissionen ein umgekehrter Trend. Die Ergebnisse zeigen, dass in Folge des deutlich feuchteren und wärmeren Klimas am Tieflandregenwaldstandort die im Vergleich zum trockeneren und kühleren Bergregenwald-Standort wesentlich höheren N_2O-Verluste auf eine höhere Aktivität der Denitrifikation zurückzuführen sind. Analysen zur Abundanz von am Stickstoffkreislauf beteiligten Mikroorganismen unterstützen diese Feststellung. Ein Vergleich beider Standorte zeigte höhere Zellzahlen der Denitrifizierer am Tieflandregenwald-Standort, während am Bergregenwald-Standort die Population der nitrifizierenden Mikroorganismen größer war.

1 Introduction

Recently, it was found that nitrification rates ranging from 0.01 to 21.0 mg N (kg SDW)$^{-1}$ d^{-1} in temperate and tropical forest soils were much higher than previously estimated (e.g. *Davidson* et al., 1993; *Hart* et al., 1994; *Riley* and *Vitousek*, 1995; *Ingwersen* et al., 1999; *Neill* et al., 1999; *Fisk* and *Fahey*, 2001; *Verchot* et al., 2001; *Breuer* et al., 2002). The magnitude of nitrification in forest soils indicates that nitrate cycling via the microbial biomass pool is inherent in these ecosystems (*Davidson* et al., 1992). Therefore nitrification is a key factor that must be investigated to understand the nitrogen (N) cycling and N losses at the ecosystem level (*Ingwersen* et al., 1999; *Breuer* et al., 2002). A study on gross nitrification in soils of montane tropical rain forests in the Atherton Tablelands, Queensland, Australia, revealed that nitrification was strongly affected by temperature, soil moisture, and the C : N ratio of the soil organic matter (*Breuer* et al., 2002). The authors also showed that nitrification activity was positively correlated with N_2O emissions. *Kiese* and *Butterbach-Bahl* (2002) reported that N_2O emissions at a lowland tropical rain forest site were higher than at a montane tropical rain forest site in the Atherton Tablelands despite their comparable soil properties and vegetation characteristics. Since pronounced differences in soil temperature and moisture between the investigated sites exist, this study was designed to find out if gross nitrification rates explain the differences in N_2O evolution observed between these tropical rain forest sites.

---

* Correspondence: PD Dr. K. Butterbach-Bahl; E-mail: klaus.butterbach@imk.fzk.de

1436-8730/02/0612-682 $17.50+.50/0

2 Materials and methods

Two tropical rain forest sites in the marginal outer tropics of North Queensland, Australia, were investigated. The lowland tropical rain forest site (approx. 80 m asl) at Bellenden Ker is covered with a complex mesophyll vine forest (*Tracey*, 1982). Annual rainfall is approximately 4395 mm and mean annual temperature is 24.3 °C. The montane tropical rain forest site at Kauri Creek (790 m asl) is located in the Atherton Tablelands and has a complex notophyll vine forest (*Tracey*, 1982). It has an average annual rainfall of 1595 mm and a mean annual temperature of 20.9 °C. Soils at both sites had developed from granitic parent material and were classified as Ustochrepts with a sandy clay loam texture. Groundwater levels at both sites are at approximately 6–8 m depth. The pH of the soil was 5.2 (Kauri Creek, montane site) and 4.1 (Bellenden Ker, lowland site) and the content of organic carbon (OC) at both sites was approximately 3 % in the uppermost 10 cm. Further details on site characteristics were given by *Kiese* and *Butterbach-Bahl* (2002).

Rates of gross nitrification in the top soil of both sites were determined by the Barometric Process Separation (BaPS) technique (*Ingwersen* et al., 1999; *Breuer* et al., 2002). In short, BaPS is based on the determination of the CO_2-, O_2- and total gas balance of well aerated soils inside an isothermal gas tight soil system. In such a system the processes of nitrification (net consumption of O_2 – pressure decrease), denitrification (net CO_2 production and net production of NO, N_2O, and N_2 here summarized as N_xO_y – pressure increase), soil respiration (pressure neutral if the respiration coefficient equals 1) and CO_2 dissolution in the soil water (pressure decrease) are the only pressure relevant processes.

$$\left(\frac{\Delta n}{\Delta t}\right) = \left(\frac{\Delta O_2}{\Delta t}\right) + \left(\frac{\Delta CO_2}{\Delta t}\right) + \left(\frac{\Delta N_x O_y}{\Delta t}\right) \tag{1}$$

Based on the total gas balance of such a system ($\Delta n/\Delta t$) an inverse balancing approach can be used to mathematically calculate gross nitrification rates. Theory and advantages of this technique as well as the validation of the BaPS technique with the ^{15}N pool dilution method for determination of gross nitrification rates were described extensively by *Ingwersen* et al. (1999) and *Breuer* et al. (2002).

For measurements of gross nitrification rates via the BaPS technique the BaPS instrument from Umweltanalytische Mess-Systeme GmbH, Munich, Germany, was used. This instrument combines sensors for CO_2, O_2, and pressure and allows to incubate undisturbed soil cores (Ø 56 mm, height 40 mm) or homogenized samples at defined temperatures.

At both sites 7 intact soil cores from the uppermost 5 cm were taken randomly from an area of 50 m to 50 m at least 3 times per week during the wet season (January to March), the transition period from dry to wet season (November to December) in 2000, and at the end of the dry season (October to November) in 2001. In total more than 40 measurements of gross nitrification (at least 7 replicates per site and season), comprising 7 soil cores each, were carried out.

In order to show that the BaPS-method for the determination of gross nitrification rates can also be used for rain forest soils, validation experiments were performed, in which simultaneously the BaPS technique and the ^{15}N pool dilution technique (*Kirkham* and *Bartholomew*, 1954; *Barraclough*, 1995) were applied. For these experiments soil samples were taken from the montane tropical rain forest site during the transition period 2000 and from the lowland tropical rain forest site during the dry season 2001. Air-dried soil samples from both sites were sieved (mesh width < 4 mm), vigorously mixed and re-wetted to a moisture content of approximately 30–35 % (w/w). After 2 days of pre-incubation 40 µg of ^{15}N labeled N (g soil fresh weight)$^{-1}$ was added to the soils by injecting aliquots of 11.0 mM KNO_3 solutions (a ^{15}N enrichment of 6 atomic-% was used), thereby slightly increasing the soil moisture content by approximately 1–2 % (w/w). After this procedure the soil was directly filled into the BaPS instrument and the system was closed gas tight and incubated at a temperature of 20 °C. Determination of gross nitrification rates via the BaPS technique lasted approximately 8–12 hours. At these intervals subsamples were taken simultaneously for determination of gross nitrification rates by ^{15}N pool dilution technique. All validation experiments were performed with 6 replicates for the BaPS as well as for the ^{15}N pool dilution technique. Nitrate concentration in the soil was determined in 1 M KCl extracts (1:5 soil:solution ratio) of 40.0 g fresh soil; the extracts were filtered through glass-fiber filter paper, and NO_3^- was determined using flow injection colorimetry. The ^{15}N : ^{14}N isotope ratios in the nitrate fractions were determined at the Risø National Laboratory, Roskilde, Denmark on a Carlo Erba EA1110 elemental analyzer coupled in continuous flow mode to a Finnigan MAT Delta PLUS stable isotope-ratio mass spectrometer (Thermo-Finnigan, Bremen, Germany) and referenced against IAEA ^{15}N quality control standard 205 (*Barraclough* and *Puri*, 1995). From the rates by which the ^{15}N enriched NO_3^- pool is diluted by oxidation of ^{14}N via nitrification, rates of gross nitrification can be determined (*Kirkham* and *Bartholomew*, 1954; *Barraclough* et al., 1985).

During the 2000 transition period from dry to wet season fresh soil samples were transferred directly to the microbiological laboratory at the Institute of Meteorology and Climate Research for determination of cell numbers of microbes involved in N-cycling and N_2O emissions. Enumeration of autotrophic ammonia and nitrite oxidizers, heterotrophic nitrifiers and denitrifiers was perfomed using the most probable number method. Techniques for incubation, determination, and media used for the different microbial groups are described in detail by *Papen* and *von Berg* (1998). At each site two soil horizons were investigated (0–5 cm and 5–10 cm) with five replicates. All given data are based on soil dry weight (105 °C).

Statistical analyses were performed with SPSS 8.0 (SPSS Inc., Chicago, USA) and Microcal Origin 6.0 (Microcal Software, Northampton, USA). In view of non-normal distribution of nitrification rates the non-parametric Mann-Whitney U test was used to identify differences between sites and seasons.

3 Results and discussion

The validation experiments indicated no significant differences between the BaPS and the ^{15}N soil pool dilution technique for quantifying the rates of gross nitrification at both sites (Fig.1). This finding further supported the previous validation experiments from other sites, sampling dates, and soils (*Ingwersen* et al., 1999; *Breuer* et al., 2002, *Heidenfelder*, 2002).

Gross nitrification rates at the montane tropical rain forest site showed a strong seasonal pattern. Significantly highest rates of gross nitrification (24.0 mg N kg^{-1} d^{-1}, Tab. 1) were found during the transition period from dry to wet season. This finding is in perfect agreement with previous findings by *Breuer* et al. (2002), who also demonstrated that nitrification at different rain forest sites at the Atherton Tablelands peaked during this hygric season. In contrast to the clear seasonal pattern of nitrification at the montane site, the lowland site showed a weaker seasonal shift in nitrification activity. However, a significantly higher gross nitrification rate during the transition period from dry to wet was also observed at this site (Tab. 1). In general, the magnitude of gross nitrification rates measured for both investigated sites (8.4–24.0 mg N kg^{-1} d^{-1}) were in the same range as observed by *Davidson* et al. (1993) for a seasonally dry forest in Mexico (6.7–21.0 mg N kg^{-1} d^{-1}), but higher

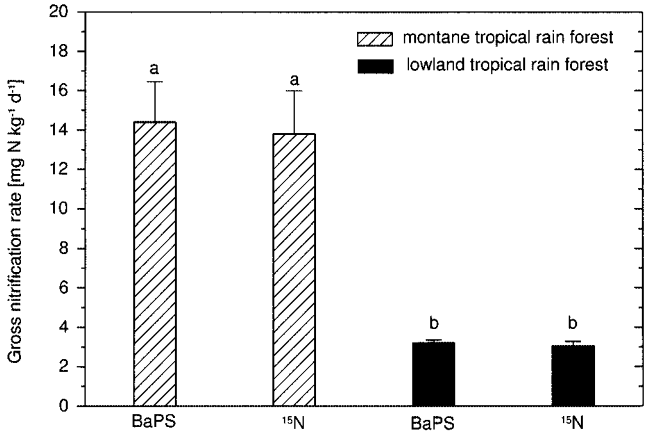


Figure 1: Comparison between BaPS method and ^{15}N pool dilution technique for measurements of gross nitrification rates from a lowland tropical rainforest site (dry season 2001) and a montane rainforest site (transition period 2000). Superscripts indicate significant differences between methods used. Means ± standard errors (n = 6) followed by the same letter are not significantly different (Mann-Whitney U test, $P < 0.05$)
Abbildung 1: Vergleich der Brutto-Nitrifikationsraten eines Tieflandregenwald-Standortes (Trockenzeit 2001) und eines Bergregenwald-Standortes (Übergang von der Trocken- zur Regenzeit 2000) gemessen mit der BaPS- und ^{15}N pool dilution-Methode. Indizes zeigen statistisch signifikante Unterschiede. Mittelwerte ± Standardfehler (n = 6) mit gleichen Buchstaben sind nicht signifikant verschieden (Mann-Whitney U Test, $P < 0.05$).

than other published rates of gross nitrification of rain forest soils (e.g. *Zou* et al., 1992: 4.0–5.0 mg N kg^{-1} d^{-1}, La Selva, Costa Rica; *Riley* and *Vitousek*, 1995: 1.0–4.0 mg N kg^{-1} d^{-1}, Hawaii; *Neill* et al., 1999: 0.01–2.8 mg N kg^{-1} d^{-1}, Amazon, Brazil).

Compared to the montane tropical rain forest site, nitrification rates at the lowland tropical rain forest site were significantly lower at all seasons (Tab. 1). This is contrary to our expectation based from the previous findings, whereby nitrification was positively correlated to N_2O emissions (*Breuer* et al., 2002), and N_2O emissions were higher at the lowland site than at the montane site (*Kiese* and *Butterbach-Bahl*, 2002; see also Tab. 1). However, the measured nitrification activity at these sites provided insights to dominating N_2O producing processes at each site. The much wetter (approximately 3-fold higher mean annual precipitation) and warmer (on average approximately 4 °C) climate at the lowland site than at the montane tropical rain forest site may have favored N_2O emissions by denitrification. That means a significant part of nitrate formed by nitrification may be denitrified and emitted as N_2O. On the other hand, the N-cycling at the dryer and colder montane rain forest site and other montane rain forest

Table 1: Rates of gross nitrification ($n \geq 7$) as well as N_2O ($n \geq 108$) and CO_2 ($n \geq 94$) emissions from a lowland and a montane tropical rainforest site. Superscripts indicate significant differences between sites ($^{A,\ B}$) and among seasons ($^{a,\ b}$). Means ± standard errors followed by the same letter are not significantly different (Mann-Whitney U test, $P < 0.05$)
Tabelle 1: Mittlere Brutto-Nitrifikationsraten ($n \geq 7$), N_2O ($n \geq 108$) und CO_2-Emissionen ($n \geq 94$) eines Tieflandregenwald- und eines Bergregenwald-Standorts. Indizes zeigen signifikante Unterschiede zwischen den Standorten ($^{A,\ B}$) und Jahreszeiten ($^{a,\ b}$). Mittelwerte ± Standardfehler mit gleichen Buchstaben sind nicht signifikant verschieden (Mann-Whitney U Test, $P < 0.05$).

Site	Dry season	Transition period	Wet season
Lowland tropical rain forest			
Gross nitrification [mg N kg^{-1} day^{-1}]	$8.4^{Aa} \pm 0.6$	$13.1^{Ab} \pm 1.5$	$10.8^{Aa} \pm 0.6$
N_2O emission[1] [µg N m^{-2} h^{-1}]	n.a.	$66.6^{a} \pm 2.7$	$242.0^{Ab} \pm 7.4$
CO_2 emission[1] [mg C m^{-2} h^{-1}]	n.a.	$133.3^{a} \pm 1.8$	$102.2^{Ab} \pm 3.2$
Montane tropical rain forest			
Gross nitrification [mg N kg^{-1} day^{-1}]	$13.2^{Ba} \pm 1.8$	$24.0^{Bb} \pm 2.0$	$15.0^{Ba} \pm 1.2$
N_2O emission[1] [µg N m^{-2} h^{-1}]	$18.9^{a} \pm 3.3$	n.a.	$80.8^{Bb} \pm 3.3$
CO_2 emission[1] [mg C m^{-2} h^{-1}]	$125.8^{a} \pm 1.5$	n.a.	$137.3^{Bb} \pm 4.5$

[1] Emissions rates are based on continuous measurements of N_2O- and CO_2 fluxes. For details see *Kiese* and *Butterbach-Bahl* (2002). n.a.: not available

Table 2: Cell numbers (n = 5; $P < 0.05$) of different functional bacterial groups in the top soil of a lowland tropical rain forest and a montane tropical rain forest site during the transition period from dry to wet season.
Tabelle 2: Zellzahlen (n = 5; $P < 0.05$) verschiedener funktioneller Gruppen von Bakterien eines Tieflandregenwald- und eines Bergregenwald-Standorts während des Übergangs von Trocken- zu Regenzeit.

Site / Soil depth	Aerobic heterotropic nitrifiers	Denitrifiers	Chemoautotrophic ammonia oxidizers	Chemoautotrophic nitrite oxidizers
	[cell numbers g^{-1}]			
Lowland tropical rain forest				
0–5 cm	5.4×10^{6}	2.7×10^{7}	3.0×10^{3}	9.5×10^{3}
	$(1.6 \times 10^{6}–1.8 \times 10^{7})$	$(8.2 \times 10^{6}–8.9 \times 10^{7})$	$(9.1 \times 10^{2}–9.9 \times 10^{3})$	$(2.9 \times 10^{3}–3.1 \times 10^{4})$
5–10 cm	5.3×10^{6}	1.1×10^{7}	4.5×10^{3}	1.3×10^{4}
	$(1.6 \times 10^{6}–1.7 \times 10^{7})$	$(3.3 \times 10^{6}–3.6 \times 10^{7})$	$(1.4 \times 10^{3}–1.5 \times 10^{4})$	$(3.9 \times 10^{3}–4.3 \times 10^{4})$
Montane tropical rain forest				
0–5 cm	5.4×10^{7}	5.4×10^{6}	1.4×10^{4}	9.5×10^{3}
	$(1.6 \times 10^{7}–1.8 \times 10^{8})$	$(1.6 \times 10^{6}–1.8 \times 10^{7})$	$(4.2 \times 10^{3}–4.5 \times 10^{4})$	$(2.9 \times 10^{3}–3.1 \times 10^{4})$
5–10 cm	2.6×10^{7}	1.7×10^{6}	1.9×10^{4}	1.2×10^{4}
	$(7.9 \times 10^{6}–8.6 \times 10^{7})$	$(5.2 \times 10^{5}–5.6 \times 10^{6})$	$(5.8 \times 10^{3}–6.3 \times 10^{4})$	$(3.6 \times 10^{3}–4.0 \times 10^{4})$

sites at the Atherton Tablelands (*Breuer* et al., 2000, 2002) may be tightly coupled, e.g. the higher nitrate formation by nitrification may be compensated by an effective microbial nitrate immobilization. High rates of microbial nitrate immobilization were reported for other tropical (*Davidson* et al., 1993; *Neill* et al., 1999) as well as temperate ecosystems (*Davidson* et al., 1992; *Hart* et al., 1994; *Stark* and *Hart*, 1997). The hypothesis that enhanced denitrification activity is the reason for the high N_2O emissions at the lowland tropical rain forest site is further supported by the observation that CO_2 emissions during the wet season were significantly lower at this site than at the montane site (Tab. 1). This indicates an increased demand for alternative electron acceptors.

Finally, the enumeration of cell numbers of soil microorganisms involved in nitrification and denitrification also supported the above hypothesis of higher rates of N_2O production via denitrification rather than via nitrification at the lowland tropical rain forest site. The cell numbers of denitrifiers (Tab. 2) were approximately one order of magnitude larger at the lowland site than at the montane site. On the other hand the cell numbers of autotrophic ammonia oxidizers and heterotrophic nitrifiers were larger at the montane tropical rain forest site than at the lowland site, which supported the similar pattern of gross nitrification rates.

In conclusion, the results clearly demonstrate the importance of combining measurements of N_2O emissions with determination of (1) rates of processes involved in N_2O production and (2) characteristics of the microbial population involved in the N-turnover processes, to improve our understanding of processes and mechanism governing N-trace gas emissions from soils.

Acknowledgment

The authors thank *Bob Hewett* and *Andrew Graham* from the Tropical Forest Research Centre (CSIRO) at Atherton, QLD, Australia, *Spiro Buhagiar* and *Jex Healey* from the Bellenden Ker Cablecar Inc. for their perfect support and generous hospitality during our stays at the field sites. This research was funded by the *Deutsche Forschungsgemeinschaft* (DFG) under contract BU 1173-2 and BU 1173-3.

References

Barraclough, D., *E. L. Geens*, *G. P. Davies*, and *J. M. Maggs* (1985): Fate of fertilizer nitrogen. III. The use of single and double labelled ^{15}N ammonium nitrate to study nitrogen uptake by ryegrass. J. Soil Sci. 36, 593–603.

Barraclough, D. (1995): ^{15}N isotope dilution techniques to study soil nitrogen transformations and plant uptake. Fert. Res. 42, 185–192.

Barraclough, D., and *G. Puri* (1995): The use of ^{15}N pool dilution and enrichment to separate the heterotrophic and autotrophic pathways of nitrification. Soil Biol. Biochem. 27, 17–22.

Breuer, L., *R. Kiese*, and *K. Butterbach-Bahl* (2002): Temperature and moisture effects on nitrification rates in tropical rain forest soils. Soil Sci. Soc. Am. J. 66, 834–844.

Davidson, E. A., *M. K. Firestone*, and *S. C. Hart* (1992): Internal cycling of nitrate in soils of a mature coniferous forest. Ecology 73, 1148–1156.

Davidson, E. A., *P. A. Matson*, *P. M. Vitousek*, *R. Riley*, *K. Dunkin*, *G. García-Méndez*, and *J. M. Maass* (1993): Process regulating soil emissions of NO and N_2O in a seasonally dry tropical forest. Ecology 74, 130–139.

Fisk, M. C., and *T. J. Fahey* (2001): Microbial biomass and nitrogen cycling responses to fertilization and litter removal in young northern hardwood forests. Biogeochemistry 53, 201–223.

Hart, S. C., *G. E. Nason*, *D. D. Myrold*, and *D. A. Perry* (1994): Dynamics of gross nitrogen transformations on an old-growth forest: the carbon connection. Ecology 75, 880–891.

Heidenfelder, A. (2002): Einfluss von Klima und Bewirtschaftung auf am N-Umsatz beteiligte mikrobielle Prozesse in Waldböden. Ph.D. Thesis, University of Freiburg, Germany.

Ingwersen, A., *K. Butterbach-Bahl*, *R. Gasche*, *O. Richter*, and *H. Papen* (1999): Barometric process separation: New method for quantifying nitrification, denitrification, and nitrous oxide sources in soils. Soil Sci. Soc. Am. J. 63, 117–128.

Kiese, R., and *K. Butterbach-Bahl* (2002): N_2O and CO_2 emissions from three different tropical forest sites in the wet tropics of Queensland, Australia. Soil Biol. Biochem. 34, 975–987.

Kirkham, D., and *W. V. Bartolomew* (1954): Equations for following nutrient transformations in soil, utilizing tracer data. Soil Sci Soc Am. Proc 18, 33–34.

Neill, C., *M. C. Piccolo*, *P. A. Steudler*, and *C. C. Cerri* (1999): Nitrogen dynamics in Amazon forest and pasture soils measured by ^{15}N pool dilution. Soil Biol. Biochem. 31, 567–572.

Papen, H., and *R. von Berg* (1998): A most probable number method (MPN) for the estimation of cell numbers of heterotrophic nitrifying bacteria in soil. Plant and Soil 199, 123–130.

Riley, R. H., and *P. M. Vitousek* (1995): Nutrient dynamics and nitrogen trace gas flux during ecosystem development in montane rain forest. Ecology 76, 292–304.

Stark, J. M., and *S. C. Hart* (1997): High rates of nitrification and nitrate turnover in undisturbed coniferous forests. Nature 385, 61–64.

Tracey, J. G. (1982): The vegetation of the humid tropical region of North Queensland. Commonw. Sci. and Ind. Res. Org., Melbourne, Vitoria, Australia.

Verchot, L. V., *Z. Holmes*, *L. Mulon*, *P. M. Groffman*, and *G. M. Lovett* (2001): Gross vs net rates of N mineralization and nitrification as indicators of functional differences between forest types. Soil Biol. Biochem. 33, 1889–1901.

Zou, X., *D. W. Valentine*, *L. R. Sanford Jr.*, and *D. Binkley* (1992): Resin-core and buried-bag estimates of nitrogen transformations in Costa Rican lowland rainforests. Plant Soil. 139, 275–283.

[P99/1B]

GLOBAL BIOGEOCHEMICAL CYCLES, VOL. 17, NO. 2, 1043, doi:10.1029/2002GB002014, 2003

Seasonal variability of N_2O emissions and CH_4 uptake by tropical rainforest soils of Queensland, Australia

Ralf Kiese

Institute of Meteorology and Climate Research, Atmospheric Environmental Research (IMK-IFU), Forschungszentrum Karlsruhe GmbH, Garmisch-Partenkirchen, Germany

Bob Hewett and Andrew Graham

Commonwealth Scientific and Industrial Research Organisation, Tropical Forest Research Centre, Atherton, Queensland, Australia

Klaus Butterbach-Bahl

Institute of Meteorology and Climate Research, Atmospheric Environmental Research (IMK-IFU), Forschungszentrum Karlsruhe GmbH, Garmisch-Partenkirchen, Germany

Received 21 November 2002; revised 28 February 2003; accepted 14 March 2003; published 9 May 2003.

[1] Over 1 year we followed the seasonal variations of N_2O and CH_4 fluxes at a tropical rain forest site in Australia. In addition, meteorological parameters, litter fall and decomposition, plant species composition, and concentrations of NH_4^+/NO_3^- in the soil and N_2O and CH_4 in the soil atmosphere were measured. N_2O emissions showed a pronounced seasonal pattern with highest rates in the wet season (108.6 μg N m^{-2} h^{-1}) and significantly lower rates during dry season (mostly <10 μg N_2O-N m^{-2} h^{-1}). N_2O emissions were positively correlated to N_2O-concentrations in the soil profile and to moisture, but not to concentrations of NH_4^+ and NO_3^-. The annual emission of N_2O (N = 6015) was 0.97 kg N ha^{-1} yr^{-1}, and, thus, approximately 7 times lower than a previous estimate for the year 2000. The marked differences in N_2O emissions between different years indicate that the interannual variability of N_2O emissions from rain forest soils cannot be neglected. With regard to CH_4 the soil functioned throughout the entire year as a significant sink. Rates of CH_4 uptake during the dry period (35–68 μg CH_4 m^{-2} h^{-1}) were higher as compared to the wet period (4–45 μg CH_4 m^{-2} h^{-1}). A close linear correlation between soil moisture and magnitude of CH_4 uptake was found. The calculated annual CH_4 uptake (N = 6090) is 3.2 kg CH_4 ha^{-1} yr^{-1}. This implies that tropical rain forest soils function as significant sinks for atmospheric CH_4 on a global scale. *INDEX TERMS:* 0315 Atmospheric Composition and Structure: Biosphere/atmosphere interactions; 1615 Global Change: Biogeochemical processes (4805); 1610 Global Change: Atmosphere (0315, 0325); 3322 Meteorology and Atmospheric Dynamics: Land/atmosphere interactions; *KEYWORDS:* N_2O emission, CH_4 uptake, tropical rainforest, seasonal variability, long-term measurements

Citation: Kiese, R., B. Hewett, A. Graham, and K. Butterbach-Bahl, Seasonal variability of N_2O emissions and CH_4 uptake by tropical rainforest soils of Queensland, Australia, *Global Biogeochem. Cycles*, *17*(2), 1043, doi:10.1029/2002GB002014, 2003.

1. Introduction

[2] Nitrous oxide (N_2O) and methane (CH_4) are two of the most important radiative active trace gases in the atmosphere. Since the industrial revolution the concentrations of these greenhouse gases have increased from 275 ppbv to 315 ppbv (N_2O) and from approximately 0.7 ppmv to 1.72 ppmv (CH_4), thus contributing at present approximately 5% and 12% respectively to the observed global warming [*Intergovernmental Panel on Climate Change* (*IPCC*), 1997]. For both gases it has been shown that soils are of significant importance as sources and sinks [e.g., *Steudler et al.*, 1989; *Bouwman*, 1990; *IPCC*, 1997; *Breuer et al.*, 2000]. With an estimated source strength of 2.2–3.7 Tg soils of tropical rain forest ecosystems are one of the major sources in the global atmospheric N_2O budget besides agricultural soils [*IPCC*, 1997]. The few published data sets on N_2O emissions from tropical rain forest soils show that emissions are in a range of 1.7 μg N m^{-2} h^{-1} to 570.8 μg N m^{-2} h^{-1} [e.g., *Kiese and Butterbach-Bahl*, 2002; *Breuer et al.*, 2000; *Keller et al.*, 1986; *Verchot et al.*, 1999; *Serca et al.*, 1994; *Melillo et al.*, 2001]. In contrast, well-aerated soils of tropical rain forests function at least during dry season conditions as sinks for atmospheric CH_4. The few publications available at present show that CH_4-exchange rates for these soils are in a range of −300 to + 550 μg CH_4 m^{-2} h^{-1} [*Keller et al.*, 1986; *Keller and Reiners*, 1994;

0886-6236/03/2002GB002014$12.00

Table 1. List of Tree and Vine Species and Frequencies of Diameter at Breast Height (DBH) > 10 cm Found at the Bellenden Ker Measuring Site Within the 50 m by 20 m Area in Which All Measurements Were Carried Out[a]

Tree Species	Number	DBH, cm	Height, m	Stem Area, cm^2	Biomass Index, m^3
Austrosteenisia stipularis	1	12.0	30	113	0.3
Backhousia bancroftii	17	11.5–90.0	11–31	33554	91.3
Brombya platynema	2	11.5–13.5	8–12	248	0.2
Canthium sp. (Whitfield Range BH1020 FRK)	1	23.5	20	433	0.9
Castanospermum australe	2	19.5–31.0	25–29	1053	2.9
Citronella smythii	1	14.0	8	154	0.1
Dysoxylum arborescens	1	15.5	16	189	0.3
Dysoxylum pettigrewianum	1	51.0	26	2043	5.3
Endiandra leptodendron	1	10.0	12	79	0.1
Endiandra sankeyana	1	13.5	13	143	0.2
Ficus pleurocarpa	1	20.0	28	314	0.9
Gomphandra australiana	1	14.0	13	154	0.2
Myristica globosa subsp. *muelleri*	9	11.0–39.5	10–32	3742	8.8
Pouteria obovoidea	1	45.0	26	1590	4.1
Rockinghamia angustifolia	2	11.0–11.5	6–7	199	0.1
Synima cordierorum	1	10.0	8	79	0.1
Tetrasynandra laxiflora	1	30.5	12.5	731	0.9
Waterhousea hedraiophylla	1	69.0	28.0	3739	10.5
Sum				48557	127.2

[a] This floristic composition is typical of well-drained soils along the foothills of the Bellenden Ker Range.

Goreau and de Mello, 1998; *Weitz et al.*, 1998; *Delmas et al.*, 1992; *Tathy et al.*, 1992; *Steudler et al.*, 1996].

[3] The magnitude of N_2O emissions, as well as of CH_4 uptake, is highly variable and strongly influenced by environmental conditions, like soil texture, substrate availability and, in most published data sets, also highly dependent upon changes in soil moisture [e.g., *Davidson*, 1993; *Kiese and Butterbach*, 2002; *Breuer et al.*, 2000; *Steudler et al.*,1996]. However, due to the harsh environmental conditions and the problems with infrastructure, most of the estimates of N_2O emissions or uptake of atmospheric CH_4 by tropical rain forest soils are based on sporadic measurements [e.g., *Matson et al.*, 1990; *Verchot et al.*, 1999], which do not fully reflect the spatial and temporal dynamics of emission/deposition of these gases as was recently demonstrated by *Kiese and Butterbach-Bahl* [2002]. The limited availability of data and particularly the absence of measurements covering entire years are the main reason why estimates of the source and the sink strength for N_2O and CH_4, especially for tropical rain forest soils, are still highly uncertain [*Breuer et al.*, 2000; *Kroetze and Mosier*, 2002]. To improve these estimates a dual approach needs to be followed. At first more measurements of rates of trace gas exchange between different tropical ecosystems and the atmosphere are required, fulfilling both representativeness and long-term coverage of measurements in order to understand seasonal variability and to identify environmental drivers which control the magnitude of trace gas emission/deposition. On the basis of the results of these measurements it will be necessary to develop new tools for upscaling emissions from a site to a regional scale. At present the best prospects for this task are biogeochemical models, which are able to simulate all processes and mechanism involved in N- and C-trace gas emissions from soils. However, even these models will need detailed data of a high temporal resolution, which should cover at least entire seasons or years, for further development and validation. On the basis of this background, we conducted a study that aimed to acquire a unique, long-term data set on N_2O emissions and CH_4 uptake by a tropical rain forest soil and to report also on other environmental parameters which may affect the magnitude of trace gas fluxes.

2. Material and Methods

2.1. Study Site

[4] The study was conducted at a typical tropical rain forest site (145°54′E, 17°16′S, 80 m a.s.l.) in the Coastal Lowlands of the "Wet Tropics," Queensland, Australia, close to the village of Bellenden Ker, approximately 70 km south of Cairns. Mean annual precipitation at the site is 4360 mm and mean annual temperature is 24.3°C. The precipitation regime is characterized by a pronounced seasonality with 70% of annual precipitation falling during the wet season, which normally lasts from November to April, and only 30% during the dry season lasting from May to September. The soil type is a Ustochrept derived from granitic parent material characterized by a high sand fraction (60%) and medium silt (20%) and clay (20%) contents. The soil pH in the uppermost layers of the mineral soil is 4.1 indicating acidic conditions due to intensive weathering and leaching processes. The organic carbon content is around 3.1% the nitrogen content is 0.26% resulting in a C/N ratio of 12:1 [*Kiese and Butterbach-Bahl*, 2002]. The complex mesophyll vine forest [*Tracey*, 1982] at our site has a basal area of almost 50 m^2 ha^{-1} and a average canopy height of 20 m. The forest structure is comparable to many of the lowland rain forests in South East Asia. Plant biodiversity is very high with over 130 plant species including 63 different kind of trees occurring in a defined plot of 20 m by 50 m. (Table 1). All experiments described below were conducted within this area. For further information about site properties, see *Kiese and Butterbach-Bahl* [2002].

2.2. Measurement of Soil-Atmosphere N_2O and CH_4 Exchange

[5] N_2O emissions and CH_4 uptake were monitored continuously over a one year period from November 1, 2001, to

October 31, 2002, thus, covering all seasons, i.e., wet, dry and intermediate conditions. For the measurements a fully automated recording system was used, which allowed simultaneous determination of N_2O emissions and CH_4 uptake with a 5-hour time resolution. The system consisted of a gas chromatograph (Texas Instruments, SRI 8610C) equipped with a ^{63}Ni electron capture detector (ECD, Vichy Valco, Switzerland) for N_2O analyses and a flame ionization detector (FID) for CH_4 analyses, an automated sampling unit for gas sampling and five measuring chambers. Details of the measuring system and modes of calibration have already been described by *Breuer et al.* [2000] and *Kiese and Butterbach-Bahl* [2002].

[6] For the flux measurements, two sampling positions were randomly selected for each chamber within the 20 m by 50 m study area. Combinations of two or three chambers were moved weekly between the different sampling positions to increase representativeness and to reduce the bias of the chambers to the measuring site, for example, by reducing rainfall or by excluding part of the litter fall. A full measuring cycle for the determination of N_2O emissions and CH_4 uptake lasted 300 min, during which the chambers were automatically closed for the first 100 min and were kept open for the last 200 min. On the basis of four gas concentration measurements in the headspace of the closed chambers, N_2O and CH_4 fluxes were calculated by linear regression and were corrected for temperature and air pressure. The system was automatically calibrated by standard calibration gases after each measuring cycle.

2.3. Measurements of N_2O- and CH_4-Concentration at Different Soil Depths

[7] Weekly measurements of N_2O- and CH_4-concentration at different soil depths (5 cm, 10 cm, 15 cm, 30 cm) were carried out during the investigation period from November 2001 to October 2002 at three different sampling positions within the defined 20 m by 50 m area. For that, sampling tubes, which consist of a hydrophobic gas permeable membrane (Akzo Nobel Faser AG, Germany), were fixed to 1/8″ stainless steel tubes. To allow gas sampling, the stainless steel tubes were closed with a rubber septum and probes were inserted horizontally into the soil at 5 cm, 10 cm, 15 cm and 30 cm soil depths. For further details, see *Kiese and Butterbach-Bahl* [2002]. Soil air samples where taken with 10-mL gas-tight syringes (Baton Rouge, Louisiana, USA) and analyzed for N_2O- and CH_4-concentrations by gas chromatography (described above) within a few minutes after sampling.

2.4. Measurement of Litter Mass, Litter Fall and Litter Decomposition Rates

[8] The litter mass on the soil was determined monthly from five replicates by randomly collecting all litter from quadrates of 0.125 m^2. Measurements of litter fall were carried out weekly by sampling five square litter traps each with a collecting area of 1 m^2. The litter traps consisted of a fiber-glass gauze with a mesh size of 2 mm which was fixed 50 cm above the ground level. All litter sampling was done within the 20 m by 50 m study area. For litter sampling we included twigs (<2 cm diameter), leaves, flower buds, flowers, bark, fruits and seeds. All sampling material was oven dried at 70°C for 24 hours before weighted.

[9] Litter decomposition rates were determined using wire mesh cages (1.0 cm mesh on metal frames) to exclude falling litter. Five cages, each 1 m by 1 m by 0.5 m high, were placed over five sampling positions. Litter from all of these positions was collected, bulked, mixed and then subdivided into five portions of equal mass. Each portion was then placed back on the ground beneath a cage. Any litter fall on the tops of the cages was removed weekly to maintain typical micro-climatic conditions. The decomposition of litter, determined as mass loss, was followed by taking 0.125 m^2 subsamples at establishment and at subsequent monthly intervals. The experiment was carried out for wet (November 2001 to April 2002) and dry season conditions (June to November 2002). To allow comparison with other published data sets we calculated the decomposition constants (k-values) based on the single exponential decay model by *Olson* [1963].

2.5. Measurements of Leaf Area Index (LAI)

[10] LAI data at the Bellenden Ker site were obtained using the standard methodology for the Licor LAI 2000 developed at the CSIRO Tropical Forest Research Centre, Atherton. Observation were made facing eastward either on clear days with the afternoon sun behind the western ranges, or on overcast days in the late afternoon. Readings were taken at 10 pegged locations along a 90-m transect spanning the study area. The sensor head was held at a standard height of 0.3 m above the ground, and at least 0.5 m from the nearest understorey foliage. No trunks of >30 cm DBH were located within 3 m of the sensor head. Large gaps were not encountered.

2.6. Measurements of Climate Parameters

[11] An automatic climate station (Campbell 21×) was installed at the top of a 25-m high tower 100 m from the measuring area. This meteorological station acquired hourly data on the following parameters: global radiation, air pressure, air temperature, relative air humidity, precipitation, wind direction and wind speed.

2.7. Measurements of Ammonia and Nitrate Concentrations in the Soil

[12] For ammonia and nitrate analyses and soil moisture content soil samples from the uppermost 5 cm of the mineral soil were taken weekly in three replicates. Each replicate soil sample was sieved to pass through a 2-mm mesh and divided into two subsamples. The first subsample was utilized for determining gravimetric soil moisture, being dried to constant weight at 105°C. Soil moisture content was expressed as percentage of soil dry weight. For NH_4^+ and NO_3^- determinations, 15 grams of the second subsample were extracted in 60-mL 2N KCl, shaken for 45 min, and allowed to stand for 15 min before being separated using a Whatman #42 filter paper prewashed with 2N KCl. For each subsample, 13 mL were frozen for future analysis in the Laboratory of CSIRO, Brisbane, where NH_4^+ and NO_3^- were determined colorimetrically [*Henzell et al.*, 1968].

2.8. Statistics

[13] All statistical analyses were performed with SPSS 8.0 (SPSS Inc., United States) and Microcal Origin 6.1. Tests of significance of differences were either performed by using

Table 2. Main Characteristics of the Meteorological Parameters Temperature, Precipitation, Radiation and Air Humidity at the Rain Forest Site at Bellenden Ker, Australia, During the Measuring Period November 1, 2001, to October 31, 2002

	Annual 11/01/01–10/31/02	Wet 11/01/01–05/15/02	Dry 05/16/02–10/31/02
	Temperature, °C[a]		
Mean	23.8	26.1	21.6
Mean daily maximum	31.4	31.4	28.6
Mean daily minimum	17.3	21.4	17.3
	Air Humidity, %[a]		
Mean	77	81	73
Mean daily maximum	100	100	95
Mean daily minimum	39	57	39
	Global Radiation, W m^{-2} day^{-1}[a]		
Mean	4412	4674	4171
Daily maximum sum	8052	8052	8360
Daily minimum sum	364	364	7616
	Precipitation, mm		
Sum	2678.2	2186.4	491.8
Daily maximum	178.1	178.1	24.2
20-year average (1980–1999)[b]	4360 ± 225		

[a] Measurements are missing for the following periods of time: 01/01/02–01/03/02, 01/06/02–02/23/02, 05/27/02–06/02/02, and 07/29/02–08/04/02.

[b] Data from Bureau of Meteorology, Brisbane, Australia.

the multiple range test (LSD) by ANOVA or by using a parametric t-test or nonparametric U-test (Mann-Whitney) (SPSS 8.0, SPSS Inc.).

3. Results

3.1. Environmental Conditions

[14] In the annual measuring period from November 1, 2001, to October 31, 2002, precipitation at the rain forest site at Bellenden Ker was in total 2678.2 mm (Table 2, Figure 1). This value is only 61% of the mean annual average precipitation for the 20 year period 1980–1999 at this site (4360 mm, Bureau of Meteorology, Brisbane, Australia). The 2001–2002 wet season was the driest observed in the Wet Tropics of Queensland for decades. Rainfall during the 2002 dry season (May/June 2002 to October 2002) also was much lower compared to previous years. However, the pronounced seasonal pattern which can be observed in this part of Australia was still evident in 2001/2002; that is, 80.7% of total rainfall was observed during the wet season (November 2001 to April 2002), whereas only 19.3% of total rainfall occurred during the dry season (May to October 2002). The seasonal pattern of soil moisture closely followed the pattern of rainfall; that is, highest values of soil moisture were recorded in the period from February to April 2002 (approximately 30%), whereas significantly lower values were measured during the period from June to October 2002 (16.5–20%) (Figure 1). Mean annual temperature for the observation period was 23.8°C. The lowest mean daily air temperature was recorded in July 2002 (17.3°C), whereas the maximum occurred in December 2002 (31.4°C) (Table 2). Daily sums of global radiation were highest with values of up to 8050 W m^{-2} day^{-1} from October to December 2001. Lower values were observed

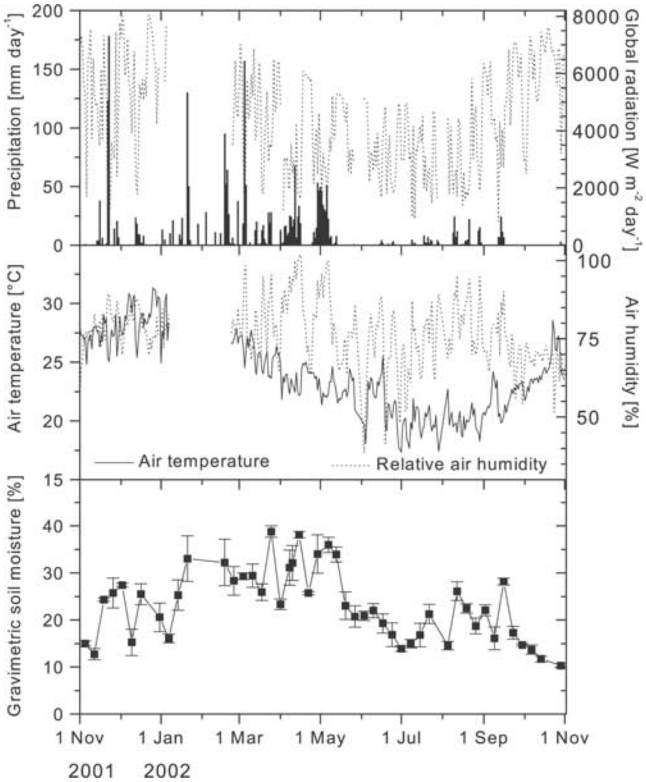


Figure 1. Seasonal course of daily totals for precipitation and global radiation, daily mean air temperature and air humidity, and weekly measurements of gravimetric soil moisture at the rain forest site at Bellenden Ker, Australia (November 1, 2001, to October 31, 2002).

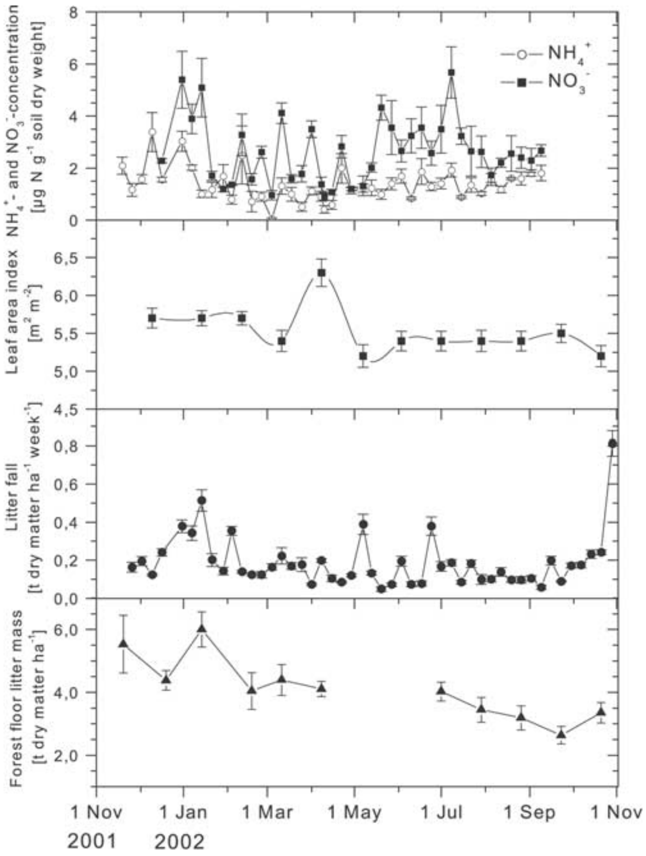


Figure 2. Seasonal changes in NH_4^+ and NO_3^- concentrations in the soil, leaf area index, litter fall and forest floor litter mass at the rain forest site at Bellenden Ker, Australia (November 1, 2001, to October 31, 2002).

from May–September 2002, when daily sums of global radiation were <5000 W m^{-2} day^{-1} (Figure 1).

3.2. Seasonal Variability of NH_4^+ and NO_3^- Concentrations in the Soil, Leaf Area Index, Litter Fall, Forest Floor Litter Mass and Litter Decomposition

[15] Over the entire observation period from November 1, 2001, to October 31, 2002, measurements of leaf area index (LAI) and forest floor litter mass were carried out at monthly intervals and litter fall and NH_4^+ and NO_3^- concentrations in the soil were measured at weekly intervals (Figure 2). NH_4^+ and NO_3^- concentrations in the soil varied in a range of 0.05–3.4 µg NH_4-N g^{-1} soil dry weight (SDW) or 0.9–5.7 µg NO_3-N g^{-1} SDW, respectively. Highest concentrations of NH_4^+ and NO_3^- in the soil were found at the beginning of the wet season. A second peak of high concentrations of NO_3^- in the soil (>5 µg N g^{-1} SDW) was observed at the start of the dry season, i.e., from mid-May 2002 to June 2002, whereas NH_4^+ concentrations in the soil decreased with time during the wet season but increased with time during dry season conditions (Figure 2). Mean average NH_4^+ and NO_3^- concentrations in the soil over the entire observation period were 1.4 ± 0.1 µg NH_4-N g^{-1} SDW and 2.6 ± 0.2 µg NO_3-N g^{-1} SDW.

[16] During the study period the LAI only varied in a very narrow range of 5.2–6.3 m^2 m^{-2} (mean: 5.5 ± 0.1 m^2 m^{-2}). Generally, higher values of LAI (mostly >5.7 m^2 m^{-2}) were recorded during wet season conditions, whereas the LAI in the dry season was slightly lower (<5.5 m^2 m^{-2}). The lowest value of LAI (5.2 ± 0.1 m^2 m^{-2}) was recorded at the end of the observation period in October 2002. The latter finding fits well with the observed maximum in litter fall (Figure 2). However, compared to seasonal variations in the LAI litter fall was much more variable with time. A first peak of litter fall was observed in January 2002 with 0.51 ± 0.06 t ha^{-1} week^{-1}, during a period of time with heavy storms and rainfall events. During dry season conditions litter fall was mostly lower only occasionally exceeding 0.2 t ha^{-1} week^{-1}. Except at the end of the observation period, after 3–4 months of dry season conditions, rates of litter fall increased and reached an absolute maximum of 0.81 ± 0.07 t ha^{-1} week^{-1} (Figure 2). During the entire observation period, litter fall was 9.0 ± 1.0 t ha^{-1} yr^{-1}. The average N-content of the leaf litter was 1.63% and annual total litter fall nitrogen was approximately 150 kg N ha^{-1} yr^{-1}. The amount of forest floor litter mass was highest in the early stages of the wet season in January 2002 (6.0 ± 0.6 t ha^{-1}). Thereafter, forest floor litter mass declined steadily and reached a minimum in September 2002 with 2.6 ± 0.3 t ha^{-1}. On average the litter mass of the forest floor was 4.1 ± 0.3 t ha^{-1}.

[17] With regard to the dynamics of litter decomposition significant differences were found between the wet and dry seasons (Figure 3). Litter decomposition was approximately two-fold higher during wet season conditions as compared to dry season conditions. The half-life of litter during wet season was 42 days (90% decay: 159 days) while the half-life of litter during the dry season was 83 days (90% decay: 276 days). The decomposition constants (k-values) were 6.1 yr^{-1} for the wet season and 3.0 yr^{-1} for the dry season, respectively.

3.3. Seasonal Variability of N_2O Emissions

[18] By the use of the fully automatic measuring system it was possible for the first time to measure in at least daily

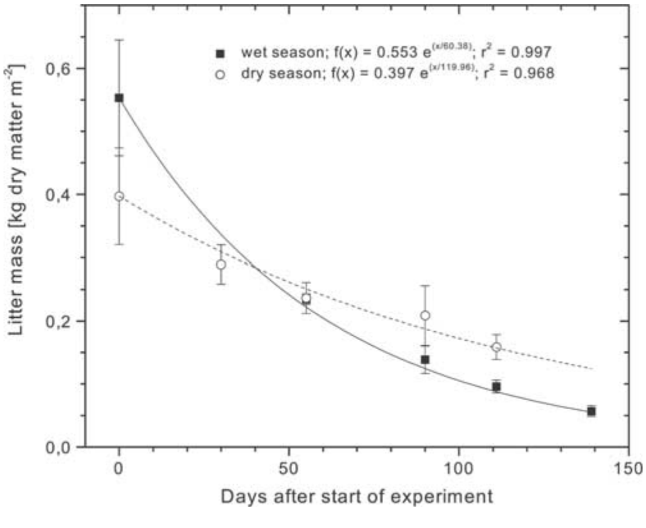


Figure 3. Dynamics of litter decomposition during the wet and dry seasons. Experiments started on November 20, 2001 (wet season), and July 1, 2002 (dry season).

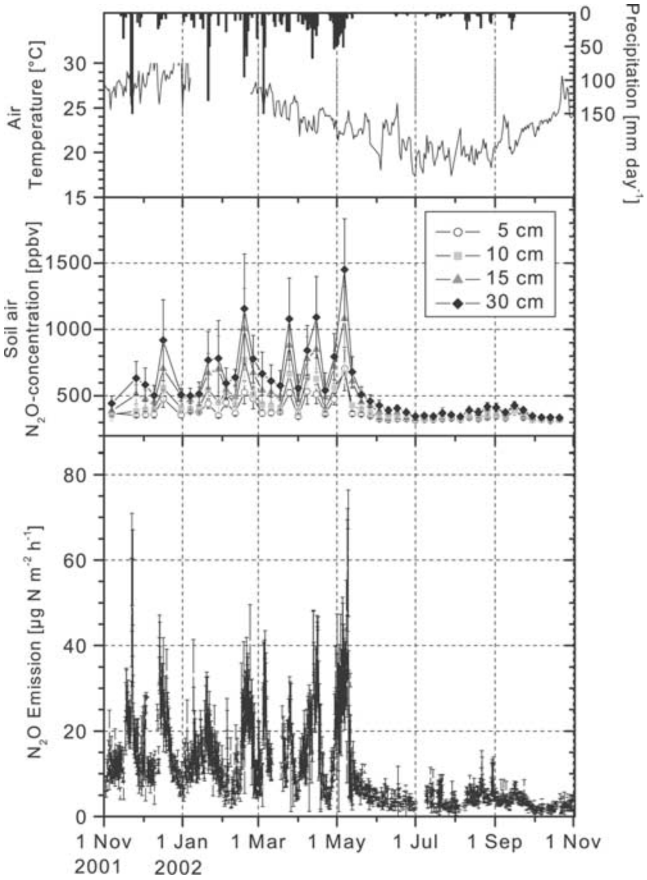


Figure 4. Seasonal variability of daily air temperature, daily precipitation, mean N_2O emissions (mean of five chambers ± SE) and weekly concentrations of N_2O (± SE) in the soil air at different depths at the rain forest site at Bellenden Ker, Australia. (November 1, 2001, to October 31, 2002).

resolution the rates of exchange of N_2O and CH_4 between the soil and the atmosphere in a tropical rain forest ecosystem continuously over an entire year (Figure 4, Table 3). During the 1-year observation period, more than 6000 valid N_2O emission rates were obtained, spanning 348 of 365 days. Mean N_2O emissions from the different positions varied by approximately a factor of 4 in a range of 5.6–20.1 µg N_2O-N m^{-2} h^{-1}. The maximum observed N_2O emission rate was 108.6 µg N_2O-N m^{-2} h^{-1}, whereas minimum N_2O emission rates at all positions were lower than the detection limit of the system (<1.0 µg N_2O-N m^{-2} h^{-1}) (Table 3). Highest N_2O emissions were observed during the wet season (Figure 4) and during the transition period from wet to dry season (May 2002). During this period of time, the mean N_2O emission from all positions only occasionally dropped below 10 µg N_2O-N m^{-2} h^{-1} (mean for November 1, 2001, to May 15, 2002: 16.3 ± 0.3 µg N_2O-N m^{-2} h^{-1}). Figure 4 clearly shows that rainfall events during this humid period, especially if they last for some days, directly lead to increased N_2O emissions. During such a period, after 2 weeks of continuous rainfall (sum of precipitation: 410 mm) in May 2002, when soil moisture reached a value of 34%, the highest mean (average of 5 chambers) N_2O emission of 62.0 µg N_2O-N m^{-2} h^{-1} was also observed.

[19] Compared to the relatively high and temporally variable N_2O emissions in the humid season, N_2O emissions during the dry season were almost constant and on a significantly lower level (mostly <10 µg N_2O-N m^{-2} h^{-1}) (Figure 4). The mean N_2O emission rate for the dry period (here May 15, 2002 to October 31, 2002) was 4.0 ± 0.1 µg N_2O-N m^{-2} h^{-1} (Figure 4). Even short-term increases of soil moisture due to sporadic rainfall events hardly led to increased N_2O emissions during the dry period, which was exceptionally long lasting and having unusual low rainfall. Only during a short rainy period in mid August 2002, when approximately 90 mm of rainfall was observed within 2 weeks and when soil moisture increased from approximately 15% to well above 20%, did N_2O emissions increase to the dry season maximum of 13.0 µg N_2O-N m^{-2} h^{-1} (Figure 4). The average N_2O emission for the entire observation period, i.e., November 1, 2001, to October 31, 2002, was 11.0 ± 0.2 µg N m^{-2} h^{-1}, which equates to an annual N_2O release of 0.97 ± 0.02 kg N ha^{-1} yr^{-1}.

[20] The data presented show that the magnitudes of N_2O emissions obviously correlate with changes in soil moisture.

Table 3. Means (± SE), Minimum and Maximum of N_2O Emissions and CH_4 Uptakes for the 10 Positions Investigated at the Rain Forest Site at Bellenden Ker, Australia[a]

	N_2O Emission, µg N m^{-2} h^{-1}					CH_4 uptake, µg CH_4 m^{-2} h^{-1}				
Position	Min.	Max.	Mean	N	c_v%	Min.	Max.	Mean	N	c_v%
1	<1.0	28.5	5.6 ± 0.2^{A}	667	70.2	9.7	67.2	40.8 ± 0.4^{A}	746	26.5
2	<1.0	51.1	7.3 ± 0.2^{B}	647	69.9	9.6	68.4	46.9 ± 0.4^{B}	669	21.5
3	<1.0	60.3	11.1 ± 0.5^{C}	556	100.2	6.1	50.5	28.5 ± 0.3^{C}	640	29.0
4	<1.0	108.6	17.2 ± 0.7^{D}	520	90.7	7.6	65.4	31.2 ± 0.4^{D}	594	30.4
5	<1.0	29.6	6.3 ± $0.2^{A,B}$	566	73.9	6.3	65.8	42.7 ± 0.3^{E}	572	17.2
6	<1.0	58.1	11.7 ± 0.4^{C}	696	82.6	9.6	63.1	36.5 ± 0.4^{F}	548	22.3
7	<1.0	83.1	20.1 ± 0.7^{E}	526	77.5	4.1	63.4	30.7 ± 0.5^{D}	543	35.6
8	<1.0	92.1	16.2 ± 0.5^{D}	652	82.8	4.2	57.3	31.0 ± 0.4^{D}	674	33.9
9	<1.0	62.0	14.0 ± 0.6^{F}	564	93.1	15.7	52.9	38.8 ± 0.3^{G}	507	16.4
10	<1.0	62.1	11.9 ± 0.4^{C}	621	85.2	13.8	57.8	36.6 ± 0.3^{G}	597	16.3
Total	<1.0	108.6	11.0 ± 0.2	6015		4.1	68.4	36.7 ± 0.1	6090	

[a] Different superscript (A–G) indicate significant differences between individual positions ($P < 0.05$). c_V: coefficient of variation; N: number of valid rates of N_2O emissions/CH_4 uptake.

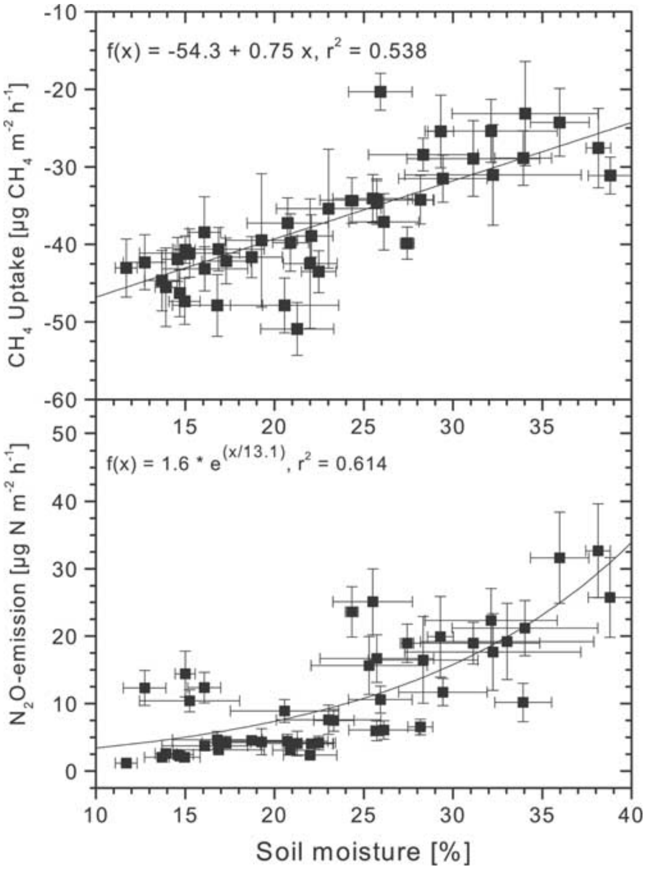


Figure 5. Dependency of the magnitude of mean N_2O emissions (mean of five chambers ± SE) and CH_4 uptakes (mean of five chambers ± SE) on soil moisture (± SE). This analysis utilized only those mean daily values of N_2O emissions and CH_4 uptakes, which were observed on the same day at which soil sampling for the determination of the gravimetric soil moisture was done.

This close positive relationship can be best described by an exponential equation (Figure 5). A correlation between N_2O emissions and NO_3^- or NH_4^+ concentrations in the soil could not be demonstrated, irrespective of which data transformations (e.g., log transformation of N_2O emission, mineral N concentrations, NO_3^- to NH_4^+ ratio or sum of inorganic N in the soil) were applied. Furthermore, also the amount of litter fall nitrogen was not correlated to the magnitude of N_2O emissions.

[21] The seasonal dynamic of N_2O emissions was highly correlated with changes in N_2O-concentrations in the soil air (Figures 4 and 6). Maximum mean N_2O-concentrations in all investigated soil depths were observed during rainy periods, for example, mid-February or beginning of May, when N_2O-concentrations at the 5 cm soil depth increased to values of up to 700 ppbv N_2O and at 30 cm soil depth of up to 1450 ppbv N_2O (Table 4). During the dry period from mid-May to end of October, N_2O-concentrations in all soil layers were relatively constant and close to the ambient atmospheric N_2O-concentration of approximately 315 ppbv (soil air 5 cm: 316–365 ppbv N_2O; soil air 30 cm: 339–390 ppbv N_2O). Only during the rainy period at the end of August 2002 did N_2O-concentrations in the soil air increase significantly from the very low levels observed before. During this period, N_2O-concentrations of up to 393 ppbv N_2O at 5 cm soil depth and 429 ppbv N_2O at 30 cm soil depth were observed.

3.4. Seasonal Variability of CH_4 Uptake

[22] During the entire observation period the soil at the rain forest site at Bellenden Ker was a sink for atmospheric CH_4 (Figure 7, Table 3); that is, rates of CH_4-oxidation were always higher than possibly occurring CH_4-production. Lowest rates of CH_4 uptake were observed at late April/early May, i.e., at the end of the wet season 2002, when uptake rates at individual measuring positions were as low as 4.1 $\mu g\ CH_4\ m^{-2}\ h^{-1}$ (Table 3). With the start of the dry season and with declining values of soil moisture, CH_4 uptake rates increased rapidly and reached a maximum in July 2002 with mean values in the range of 40–50 $\mu g\ CH_4\ m^{-2}\ h^{-1}$ (Figure 7). The coefficients of variation show that the seasonal variability of CH_4 uptake by the soil was much lower as compared to the seasonal variability of N_2O emissions (Table 3). For most of the individual positions the coefficient of variation of CH_4 uptake was <30% and, thus, approximately 3 times lower than the coefficient of variation for N_2O emissions. However, as was also shown for N_2O emission, the magnitude of CH_4 uptake was strongly linked to observed soil moisture values. Figure 5 shows that the relationship between soil moisture and rates of CH_4 uptake could be best described with a simple linear regression and that rates of CH_4 uptake decrease with increasing values of soil moisture. The mean rate of CH_4 uptake over the entire observation period, which is based on more than 6000 valid uptake rates, was 36.7 ± 0.1 $\mu g\ CH_4\ m^{-2}\ h^{-1}$ or 3.21 ± 0.01 $kg\ CH_4\ ha^{-1}\ yr^{-1}$.

[23] A weak but significant relationship was found between CH_4 uptake rates and concentrations of NH_4^+ in the soil ($r^2 = 0.15$, $p < 0.05$, $f(x) = 29.1 + 5.3\ x$), whereas a correlation between NO_3^- concentrations in the soil and CH_4 uptake could not be demonstrated. The function given illustrates that CH_4 uptake rates increased with increasing NH_4^+ concentrations in the soil.

[24] The comparable low seasonal variability of CH_4 uptake at our rain forest site was also mirrored by only small seasonal changes in CH_4-concentrations in the soil air at different depths (Table 4 as well as Figure 7). The lowest CH_4-concentrations in the soil air with values of 144–386 ppbv CH_4 were observed for the 30-cm layer, whereas CH_4-concentrations in 5-cm soil depth were significantly higher (1038–1446 ppbv CH_4). In contrast to the results obtained for N_2O, no significant relationship between soil air CH_4-concentrations and rates of atmospheric CH_4 uptake could be demonstrated (Figure 8).

4. Discussion

[25] The measured values for leaf area index of 5.2–6.3 $m^2\ m^{-2}$ at our rain forest site at Bellenden Ker, Australia, fall within the range of values reported for tropical forests elsewhere in the world [*Grace et al.*, 1995; *Vourlitis et al.*,

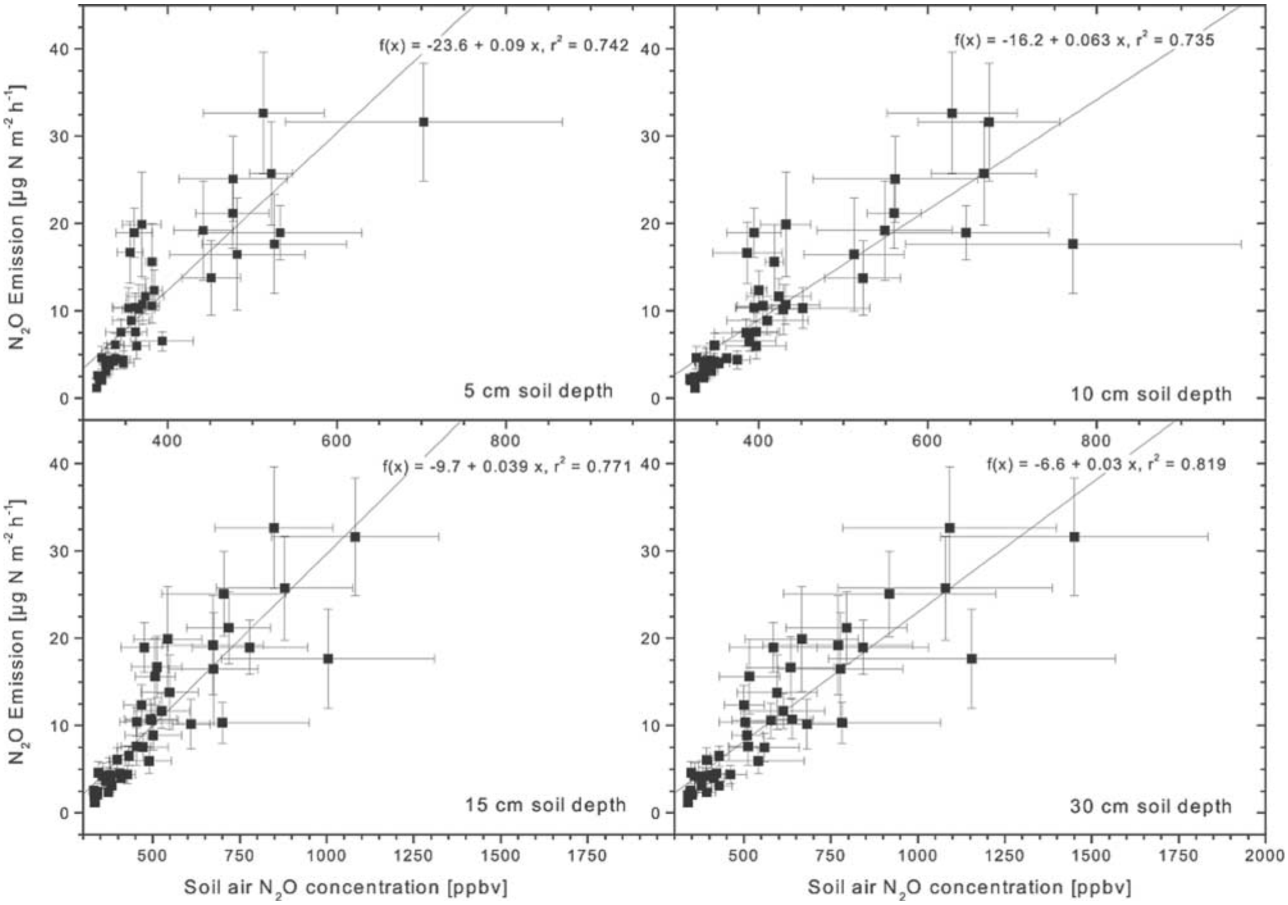


Figure 6. Linear regression analysis between soil air N_2O-concentrations (± SE) at different depths and mean N_2O emissions (± SE). The correlation analysis utilized only those mean daily N_2O emissions that were observed on the same day as soil air N_2O-concentration measurements.

Table 4. Means (± SE), Minimum and Maximum of N_2O- and CH_4-Concentrations (ppbv) in the Soil Air at Different Soil Depths at the Rain Forest Site at Bellenden Ker, Australia[a]

	Minimum	Maximum	Mean	N	c_v%
		N_2O-Concentration in the Soil Air			
11/01/01 – 05/15/02					
5 cm	344	702	$463 \pm 17^{A,1}$	72	20.5
10 cm	384	772	$494 \pm 23^{A,1}$	72	22.9
15 cm	454	1082	$632 \pm 36^{B,1}$	72	28.2
30 cm	500	1449	$740 \pm 50^{C,1}$	72	33.0
05/16/02 – 10/31/02					
5 cm	316	393	$333 \pm 4^{A,2}$	66	5.3
10 cm	319	396	$342 \pm 5^{A,2}$	66	6.1
15 cm	331	453	$375 \pm 7^{B,2}$	66	9.3
30 cm	339	511	$388 \pm 9^{B,2}$	66	11.3
		CH_4-Concentration in the Soil Air			
11/01/01 – 05/15/02					
5 cm	1038	1446	$1213 \pm 25^{A,1}$	54	2.1
10 cm	676	1137	$870 \pm 22^{B,1}$	54	2.5
15 cm	310	609	$403 \pm 18^{C,1}$	54	4.5
30 cm	144	640	$274 \pm 27^{D,1}$	54	9.9
05/16/02 – 10/31/02					
5 cm	1131	1317	$1254 \pm 10^{A,1}$	63	0.8
10 cm	747	993	$928 \pm 11^{B,2}$	63	1.2
15 cm	398	531	$450 \pm 7^{C,2}$	63	1.6
30 cm	186	397	$253 \pm 11^{D,1}$	63	4.2

[a] Different superscripts indicate significant differences ($P < 0.05$) in N_2O- or CH_4-concentrations between different soil depths (A – C) and periods of time (1,2). C_V: coefficient of variation; N: number of measurements.

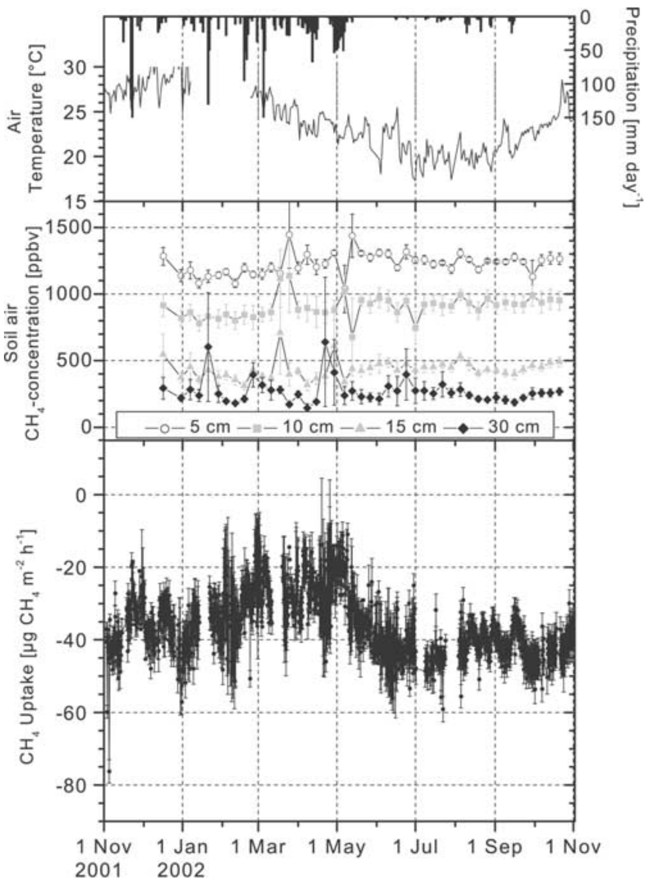


Figure 7. Seasonal variability of daily air temperature, daily sum of precipitation, mean CH_4 uptake (mean of five chambers ± SE) and weekly concentrations of CH_4 (± SE) in the soil air at different depths at the rain forest site at Bellenden Ker, Australia. (November 1, 2001, to October 31, 2002).

2001; *Kitayama and Aiba*, 2002]. Also, the annual rate of litter fall at our site (9.0 ± 1.0 t ha^{-1}) is in very good agreement with reported values for other tropical rain forest sites in Australia (7.3–11.0 t ha^{-1}) [*Spain*, 1984; *Stocker et al.*, 1995] or on other continents [*Erickson et al.*, 2002] (and see, for example, compilation of data by *Vitousek* [1984]). Furthermore, the mass of litter on the forest floor (2.6–6.0 t ha^{-1}) agrees well with other reports from tropical rain forests in Australia [*Spain et al.*, 1984], Brazil [*Klinge et al.*, 1975] or Nigeria [*Hopkins*, 1966]. The calculated decomposition rates with k-values of 6.1 yr^{-1} (wet season) and 3.1 yr^{-1} (dry season) are in the upper range of published values for tropical regions [e.g., *Kumar and Deepu*, 1992; *Muoghalu et al.*, 1994; *Vitousek et al.*, 1994; *Aerts*, 1997], but in very good agreement with reports by *Spain* [1984] for tropical rain forests in Australia. This may be due to the quite narrow C/N ratio of leaf litter at our site, which is in the range of 20–30 (unpublished data).

[26] This paper reports for the first time a complete data set of N_2O emissions from a tropical rain forest site with at least daily resolution. This huge dataset allowed us (1) to identify short-term changes in the magnitude of N_2O emissions, (2) to demonstrate seasonal variability of N_2O emissions with a hitherto unknown time resolution and (3) to calculate precise annual loss rates of N_2O from our site. The short-term variability of N_2O emissions from tropical rain forest soils and its dependency on rainfall events have already been highlighted by *Breuer et al.* [2000] and *Kiese and Butterbach-Bahl* [2002] for different rain forest sites in Australia and by *Vitousek et al.* [1989] during a rewetting experiment of a tropical forest soil in Chamela, Mexico. Our observations show, that it is essential to have continuous measurements of N_2O emissions, at least during wet season conditions, to reliably estimate annual loss rates. This statement can be further supported by a simple statistical analysis. Using the entire data set, we calculated mean annual N_2O emissions (CH_4 uptake) based on randomly selected data sets assuming daily, weekly or monthly measuring intervals. Such an analysis shows that the reduction of the measuring frequency from 4–5 flux rates per day to daily, weekly or monthly intervals would reduce the accuracy of the calculated annual N_2O emission (CH_4 uptake) by approximately 10% (5%), 20% (10%), and 40% (20%), respectively. From these figures it can be concluded that weekly measurement of trace gas fluxes provide a reliable basis for estimating annual budgets of N_2O- and CH_4-exchange at our site. However, it is necessary to point out that for further improvement and validation of biogeochemical models which simulate the biosphere-atmosphere trace gas exchange it is of utmost importance to have detailed data available such as daily or subdaily trace gas fluxes in order to be capable to proof that the processes involved in trace gas production, consumption and emission are reliably reproduced.

[27] The seasonal variability of N_2O emissions found in this study, with significantly higher N_2O emissions during wet season as compared to the dry season, is in general agreement with the seasonal changes in decomposition rates at our site and with previous reports about N_2O emissions from soils of tropical rain forests worldwide [e.g., *Garcia-Méndez et al.*, 1991; *Steudler et al.*, 1991; *Verchot et al.*, 1999; *Breuer et al.*, 2000; *Melillo et al.*, 2001]. The pronounced seasonal variations in N_2O-emissions are most likely associated with significant changes in the composition of the microbial community involved in N-cycling and N_2O-production, since results from *Kiese et al.* [2002] indicate that the abundance of denitrifiers at our site is approximately 5 times higher than at a montane rain forest site with a colder and dryer climate. However, at present data on seasonal changes in microbial community composition are not available to further support this hypothesis. The higher N_2O emissions in the wet season as compared to the dry season are also mirrored by higher concentrations of N_2O in the soil atmosphere. N_2O-concentrations in the soil air were highest with up to 1440 ppbv N_2O in 30 cm soil depth and lowest close to the surface [<702 ppbv N_2O]. However, these data do not allow to conclude that N_2O-production does mainly occur in the deeper soil horizons, since soil N_2O-concentrations do not only depend on N_2O-production but also on rates of N_2O-consumption and diffusion. *Breuer et al.* [2000] have shown for different

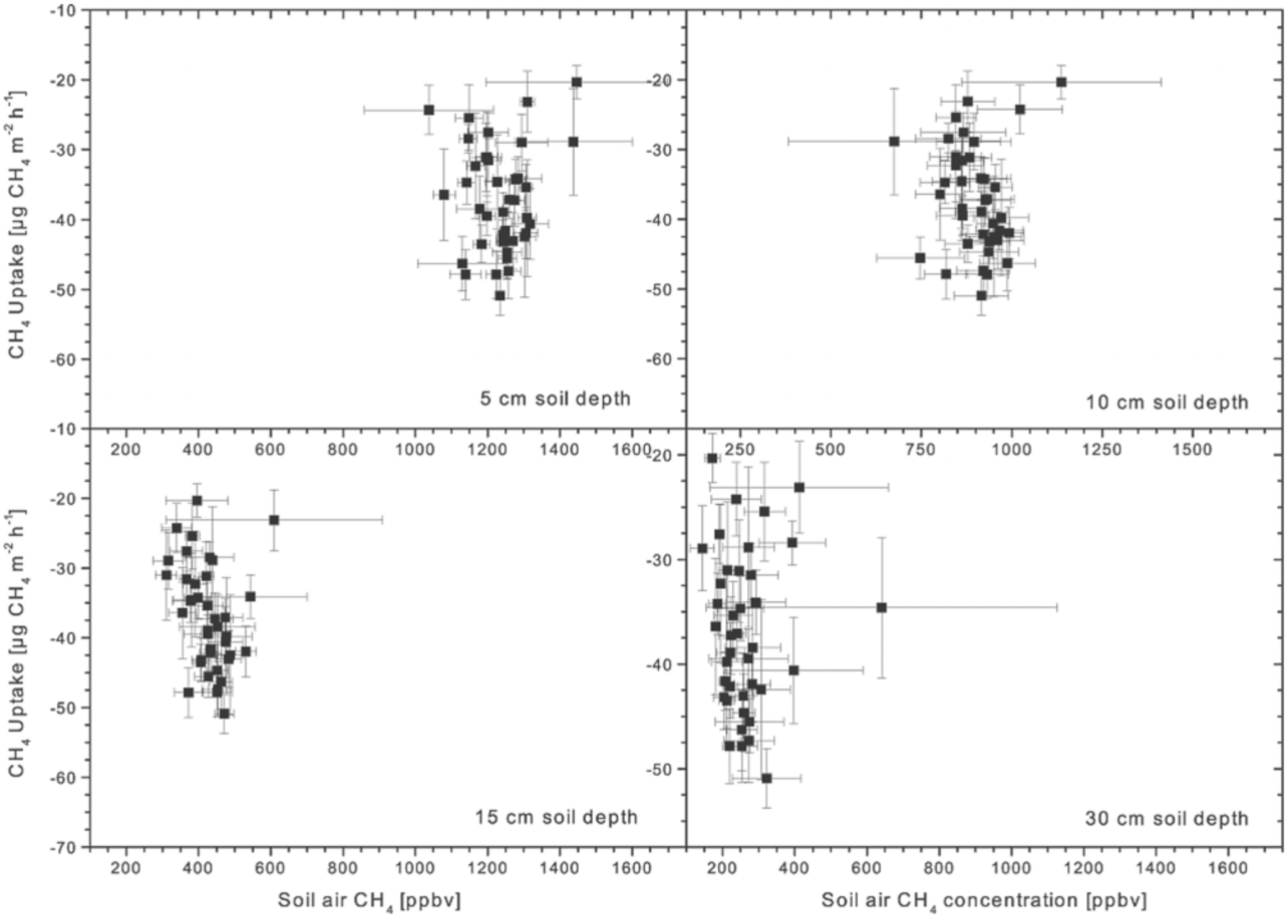


Figure 8. Relationships between soil air CH_4-concentrations (± SE) at different soil depths and rates of mean CH_4 uptake (± SE). This analysis utilized only those mean daily CH_4 uptake rates which were observed at the same day of soil air CH_4-concentration measurements.

montane rain forest soils that the N_2O-production in such soils is mainly concentrated in the uppermost centimeters of the mineral soil or even in the litter layer. The measured N_2O-concentrations in the soil air (340–1440 ppbv N_2O) are lower than reported by *Verchot et al.* [1999], who found N_2O-concentrations of up to 8200 ppv N_2O in a primary rain forest in eastern Amazonia, but in the same range of N_2O-concentrations as observed by *McSwiney et al.* [2001] in tropical rain forest soils in Puerto Rico. However, these values are difficult to compare since sampling depth and soil properties, and thus the permeability of the soil for gases, will be different between the sites. Nevertheless, the observed pronounced seasonal dependency of N_2O-concentrations in the soil atmosphere is in good agreement with results reported by *Verchot et al.* [1999]. NH_4^+ as well as NO_3^- concentrations in the uppermost 5 cm of the soil did not significantly correlate with the magnitude of N_2O emissions at all. This finding is in agreement with results from other studies for tropical forest soils and temperate soils, in which correlations between seasonal changes in N_2O emissions and soil mineral N concentrations could not be presented [e.g., *Keller and Reiners*, 1994; *Verchot et al.*, 1999; *Breuer et al.*, 2000; *Papen and Butterbach-Bahl*, 1999]. Only if mineral N concentrations and N_2O emissions across different sites are compared, do significant correlations between N_2O emissions or N_2O + NO emissions and NO_3^- and/or NH_4^- concentrations in the soil become obvious [*Keller and Reiners*, 1994; *Verchot et al.*, 1999; *Davidson and Verchot*, 2000; *Davidson et al.*, 2000]. From our point of view, the latter correlation only indicates differences in the nutrient status between sites. However, such a correlation does not allow the conclusion that seasonal changes in N_2O emissions at a given site are reflected by changes in soil NO_3^- and NH_4^+ concentrations. In this context, one should also be aware of the dynamics of microbial N turnover processes such as nitrification in temperate and tropical forest soils, which have been observed to be in the range of 20- >300 mg NH_4-N m^{-2} d^{-1} (see compilation of results from different groups by *Breuer et al.* [2002]). The observed rapid cycling of NH_4^+ and NO_3^- leads to short turnover times of the inorganic N pools in the soil. For example, *Verchot et al*. [2001] showed for different temperate forest sites in the United States that the mean turnover time of the NH_4^+ and NO_3^- pools in the soil was only 1.7 ± 0.4 d^{-1} and 4.4 ± 2.5 d^{-1}, respectively.

[28] In the recent past numerous estimates of annual loss rates of N_2O emissions from tropical rain forests have been published, which range from 0.3 to 7.5 kg N ha^{-1} yr^{-1}

[e.g., *Riley and Vitousek*, 1995; *Keller and Reiners*, 1994; *Verchot et al.*, 1999; *Breuer et al.*, 2000; *Kiese and Butterbach-Bahl*, 2002]. In this paper we calculated that N_2O emissions from the Bellenden Ker site in the period November 1, 2001, to October 31, 2002, were 0.97 kg N ha^{-1}, i.e., within the published ranges of N_2O emissions from tropical rain forest soils. However, the recent estimate is approximately 7 times lower than the estimate for the same site for the year 2000 [*Kiese and Butterbach-Bahl*, 2002], and 3–5 times lower than estimates for tropical rain forest sites at the Atherton Tablelands, Australia, for the years 1997, 1998 and 2000 [*Breuer et al.*, 2000; *Kiese and Butterbach-Bahl*, 2002]. As pointed out previously, the rainfall in the wet season of the years 2001–2002 was extremely low and the annual sum of rainfall was only 61% of the long-term average (4360 mm). Mainly due to this fact, N_2O emissions in the wet season 2001–2002 were on average only 16.3 ± 0.3 μg N_2O-N m^{-2} h^{-1}, whereas the mean of the transition period and the wet season of the year 2000, in which the annual sum of rainfall was well above the mean (6746 mm), was 154.3 ± 7.9 μg N_2O-N m^{-2} h^{-1} [*Kiese and Butterbach-Bahl*, 2002]. This finding, the first detailed description of interannual variability of N_2O emissions for tropical rainfall ecosystems, is extremely relevant for estimates of global N_2O emissions from this source. Since N_2O emissions are highly dependent upon soil properties, vegetation [*Keller and Reiners*, 1994; *Riley and Vitousek*, 1995; *Davidson et al.*, 2001; *Breuer et al.*, 2002; *Kiese and Butterbach-Bahl*, 2002; *Erickson et al.*, 2002] and, as demonstrated, on annual rainfall variability, the principle problem of reducing the uncertainty of global estimates becomes obvious. We are sure, that this problem can only be solved with an integrated measuring and modeling approach, in which detailed and long-term field measurements are used to further develop and validate process oriented models [e.g., *Parton et al.*, 1996; *Potter et al.*, 1996; *Li et al.*, 2000] and in which the models are used for upscaling site results to regional [*Potter et al.*, 1998; *Butterbach-Bahl et al.*, 2001] and global scales [*Potter et al.*, 1996].

[29] It is well known that well aerated forest soils are significant sinks for atmospheric CH_4 [e.g., *Crill*, 1991; *Castro et al.*, 1995]. Though there are many publications which have investigated the sink strength of temperate forest soils for atmospheric CH_4 [e.g., *Steudler et al.*, 1989; *Sitaula et al.*, 1995; *Butterbach-Bahl and Papen*, 2002], published data sets on rates of CH_4 uptake by tropical rain forest soils are still scarce. The few reports available show tropical rain forest soils can function as sinks as well as sources for atmospheric CH_4 and that fluxes are in a range of -300- + 550 μg CH_4 m^{-2} h^{-1} [*Keller et al.*, 1983, 1986; *Tathy et al.*, 1992; *Delmas et al.*, 1992; *Keller and Reiners*, 1994; *Steudler et al.*, 1996; *Verchot et al.*, 2000]. Our measurements on CH_4 uptake at the Bellenden Ker site are the first which report on a full annual cycle of CH_4 uptake by a tropical rain forest soil, thereby demonstrating that a pronounced seasonal variability exists, with significant higher rates of CH_4 uptake in the dry period as compared to the wet period. The range of CH_4 uptake observed within this study (4.1–68.4 μg CH_4 m^{-2} h^{-1}) is well within the span given above. However, it is remarkable that even in periods with heavy rainfall, as, for example, at the beginning of May 2002, the soil at our site still function as a weak, but still significant sink for atmospheric CH_4 and that during these periods of time the CH_4-concentrations in the soil air remained significantly below atmospheric concentrations at all soil depths monitored. This indicates the high drainage capacity of the soil at our site and the high gas permeability of the soil at least down to approximately 30 cm. However, soil moisture and, thus the gas permeability of the soil for atmospheric CH_4 and O_2, remains the main controller of the magnitude of CH_4 uptake at our site. This observation is in good agreement with many other reports on controls of CH_4 uptake by forest soils [e.g., *Crill*, 1991; *Adamsen and King*, 1993]. The negative correlation between N_2O emissions and rates of CH_4 uptake ($f(x) = 41.7 - 0.55 \times N_2O$ emission; $r^2 = 0.340$; $p < 0.01$) further support the finding, that CH_4 uptake rates at our site mainly depend on CH_4 and O_2 diffusion from the atmosphere to the soil, whereas N_2O-emissions are higher if O_2 diffusion is reduced due to, for example, increased soil moisture. The finding that higher rates of CH_4 uptake were found at times with higher concentrations of NH_4^+ in the soil is surprising, since results of other studies indicate that increased availability of inorganic N in forest soils inhibit CH_4-oxidation [e.g., *Adamsen and King*, 1993; *Schnell and King*, 1994]. However, since this correlation was very weak and NH_4^+-concentrations in the soil were at the lower end of observed inorganic N-concentrations in tropical and temperate forests [e.g., *Verchot et al.*, 1999; *Papen and Butterbach-Bahl*, 1999] and moreover highly correlated to soil moisture; that is, higher NH_4^+-concentrations were found at lower soil moisture values, we doubt that NH_4^+-concentrations in tropical forest soils are a good predictor for CH_4 uptake rates. The calculated annual CH_4 uptake at our site of 3.21 kg CH_4 ha^{-1} yr^{-1} is in the range of observed annual rates of CH_4 uptake for temperate forest soils [e.g., *Steinkamp et al.*, 2000; *Butterbach-Bahl and Papen*, 2002; *Borken and Brumme*, 1997]. In accordance with the work of *Verchot et al.* [2000] on CH_4 uptake in Amazonian tropical forest soils our results demonstrate that tropical rain forest soils are indeed significant sinks for atmospheric CH_4 on a global scale.

[30] **Acknowledgments.** The authors thank Hans Papen for many valuable discussions which helped to improve the design and evaluation of measurements and Niels Johannsen for his excellent assistance during the establishment of the automatic measuring system and the first weeks of measurements. We are in particular indebted to Spiro Buhagiar and Jex Healey from the Bellenden Ker Cablecar Inc. (BTS), Queensland, Australia, for their generous hospitality during our stays at the field sites and for their support in running the equipment successfully over an entire year. Furthermore the authors want to express their gratitude to Andrew Ford, Tropical Forest Research Centre, Atherton, Australia, for providing an extensive plant species list for the site. This research was funded by the Deutsche Forschungsgemeinschaft (DFG), Germany, under contract BU 1173-3.

References

Adamsen, A. P. S., and G. King, Methane consumption in temperate and subarctic forest soils: Rates, vertical zonation and responses to water and nitrogen, *Appl. Environ. Microbiol.*, *59*, 485–490, 1993.

Aerts, R., Climate, leaf litter chemistry and leaf litter decomposition in terrestrial ecosystems: A triangular relationship, *Oikos*, *79*, 439–449, 1997.

Borken, W., and R. Brumme, Liming practice in temperate forest ecosystems and the effects on CO_2, N_2O and CH_4 fluxes, *Soil Use Manage.*, *13*, 251–257, 1997.

Bouwman, A. F., Exchange of greenhouse gases between terrestrial ecosystems and the atmosphere, in *Soils and the Greenhouse Effect*, edited by A. F. Bouwman, pp. 61–127, John Wiley, New York, 1990.

Breuer, L., H. Papen, and K. Butterbach-Bahl, N_2O emission from tropical forest soils of Australia, *J. Geophys Res.*, *105*, 26,353–26,367, 2000.

Breuer, L., R. Kiese, and K. Butterbach-Bahl, Temperature and moisture effects on nitrification rates in tropical rain-forest soils, *Soil Sci. Soc. Am. J.*, *66*, 834–844, 2002.

Butterbach-Bahl, K., and H. Papen, Four years continuous record of CH_4-exchange between the atmosphere and untreated and limed soil of a N-saturated spruce and beech forest ecosystem in Germany, *Plant Soil*, *240*, 77–90, 2002.

Butterbach-Bahl, K., F. Stange, H. Papen, and C. Li, Regional inventory of nitric oxide and nitrous oxide emissions for forest soils of Southeast Germany using the biogeochemical model PnET-N-DNDC, *J. Geophys Res.*, *106*, 34,155–34,166, 2001.

Castro, M. S., P. A. Steudler, J. M. Melillo, J. D. Aber, and R. D. Bowden, Factors controlling atmospheric methane consumption by temperate forest soils, *Global Biogeochem. Cycles*, *9*, 1–10, 1995.

Crill, P. M., Seasonal patterns of methane uptake and carbon dioxide release by a temperate woodland soil, *Global Biogeochem. Cycles*, *5*, 319–334, 1991.

Davidson, E. A., and L. V. Verchot, Testing the hole-in-the-pipe model of nitric and nitrous oxide emissions from soils using the TRAGNET database, *Global Biogeochem. Cycles*, *14*, 1035–1043, 2000.

Davidson, E. A., M. Keller, H. E. Erickson, L. V. Verchot, and E. Veldkamp, Testing a conceptual model of soil emissions of nitrous and nitric oxides, *BioScience*, *50*, 667–680, 2000.

Davidson, E. A., M. M. C. Bustamante, and A. de Siqueira Pinto, Emissions of nitrous oxide from soils of native and exotic ecosystems of the Amazon and Cerrado regions of Brazil, in *Optimizing Nitrogen Management in Food and Energy Production and Environmental Protection: Proceedings of the 2nd International Nitrogen Conference on Science and Policy, Sci. World, 1*, 312–319, 2001.

Delmas, R. A., J. Servant, J. P. Tathy, B. Gros, and M. Labat, Sources and sinks of methane and carbon dioxide exchanges in mountain forest in equatorial Africa, *J. Geophys. Res.*, *97*, 6169–6179, 1992.

Erickson, H., E. A. Davidson, and M. Keller, Former land-use and tree species affect nitrogen oxide emissions from a tropical dry forest, *Oecologia*, *130*, 297–308, 2002.

Garzia-Méndez, G. J., J. M. Maass, P. A. Matson, and P. M. Vitousek, Nitrogen transformations and nitrous oxide flux in a tropical deciduous forest in Mexico, *Oecologia*, *88*, 362–366, 1991.

Goreau, T. J., and W. Z. De Mello, Tropical deforestation: Some effects on atmospheric chemistry, *Ambio*, *17*, 275–281, 1998.

Grace, J., et al., Fluxes of carbon dioxide and water vapour over an undisturbed tropical forest in south-west Amazonia, *Global Change Biol.*, *1*, 1–12, 1995.

Henzell, E. F., I. Vallis, and J. K. Lindquist, Automatic colorimetric methods for determination of nitrogen in digests and extracts of soil, paper presented at 9th International Congress on Soil Science, Adelaide, South Aust., Australia, 1968.

Hopkins, B., Vegetation of Olokemeji Forest Reserve, Nigeria: IV. The litter and soil with special reference to their seasonal changes, *J. Ecol.*, *54*, 687–703, 1966.

Intergovernmental Panel on Climate Change, Agriculture: Nitrous oxide from agricultural soils and manure management, in *IPCC Guidelines for National Greenhouse Gas Inventories*, chap. 4, Org. for Econ. Coop. Dev., Paris, 1997.

Keller, M., and W. A. Reiners, Soil atmosphere exchange of nitrous oxide, nitric oxide, and methane under secondary succession of pasture to forest in the Atlantic lowlands of Costa Rica, *Global Biogeochem. Cycles*, *8*, 399–409, 1994.

Keller, M., T. J. Goreau, S. C. Wofsy, W. A. Kaplan, and M. B. McElroy, Production of nitrous oxide and consumption of methane by forest soils, *Geophys. Res. Lett.*, *10*, 1156–1159, 1983.

Keller, M., W. A. Kaplan, and S. C. Wofsy, Emissions of N_2O, CH_4 and CO_2 from tropical forest soils, *J. Geophys. Res.*, *91*, 11,791–11,802, 1986.

Kiese, R., and K. Butterbach-Bahl, N_2O and CO_2 emissions from three different tropical forest sites in the wet tropics of Queensland, Australia, *Soil Biol. Biochem.*, *34*, 975–987, 2002.

Kiese, R., H. Papen, E. Zumbusch, and K. Butterbach-Bahl, Nitrification activity in tropical rain forest soils of the Coastal Lowlands and the Atherton Tablelands, Queensland, Australia, *J. Plant Nutr. Soil Sci.*, *165*, 682–685, 2002.

Kitayama, K., and S. I. Aiba, Ecosystem structure and productivity of tropical rain forests along altitudinal gradients with contrasting soil phosphorous pools on Mount Kinabalu, Borneo, *J. Ecol.*, *90*, 37–51, 2002.

Klinge, H., W. A. Rodrigues, E. Brunig, and E. J. Fittkau, Biomass and structure in the Central Amazonian rain forest, in *Tropical Ecological Systems: Trends in Terrestrial and Aquatic Research*, edited by F. B. Golley and E. Medina, pp. 115–122, Springer Verlag, New York, 1975.

Kroetze, C., and A. Mosier, Progress in source strength estimates of nitrous oxide (N_2O), in *Non-CO_2 Greenhouse Gases: Scientific Understanding, Control Options and Policy Aspects*, edited by J. van Ham et al., pp. XXIX–XXXIII, Millpress, Rotterdam, Netherlands, 2002.

Kumar, B. M., and J. K. Deepu, Litter production and decomposition dynamics in moist deciduous forests of the Western Ghats in Peninsular India, *For. Ecol. Manage.*, *50*, 181–201, 1992.

Li, C., J. Aber, F. Stange, and H. Papen, A process-oriented model of N_2O and NO emission from forest soils: 1. Model development, *J. Geophys. Res.*, *105*, 4369–4384, 2000.

Matson, P. A., P. M. Vitousek, G. P. Livingston, and N. A. Swanberg, Sources of nitrous oxide flux from Amazonian ecosystems, *J. Geophys. Res.*, *95*, 16,789–16,798, 1990.

McSwiney, C. P., W. H. McDowell, and M. Keller, Distribution of nitrous oxide and regulators of its production across a tropical rainforest catena in the Luquillo Experimental Forest, Puerto Rico, *Biogeochemistry*, *56*, 265–286, 2001.

Melillo, J. M., P. A. Steudler, B. J. Feigl, C. Neill, D. Garcia, M. C. Piccolo, C. C. Cerri, and H. Tian, Nitrous oxide emissions from forests and pastures of various ages in the Brazilian Amazon, *J. Geophys. Res.*, *106*, 34,179–34,188, 2001.

Muoghalu, J. I., O. M. Adeleye, and R. T. Balogun, Litter decomposition and inorganic element dynamics in a secondary rain forest at Ile-Ife, Nigeria, *Afr. J. Ecol.*, *32*, 208–221, 1994.

Olson, J. S., Energy storage and the balance of producers and decomposers in ecological systems, *Ecology*, *44*, 322–331, 1963.

Papen, H., and K. Butterbach-Bahl, A 3-year continuous record of nitrogen trace gas fluxes from untreated and limed soil of a N-saturated spruce and beech forest ecosystem in Germany: 1. N_2O emissions, *J. Geophys. Res.*, *104*, 18,487–18,503, 1999.

Parton, W. J., A. R. Mosier, D. S. Ojima, D. W. Valentine, D. S. Schimel, K. Weier, and A. E. Kulmala, Generalized model for N_2 and N_2O production from nitrification and denitrification, *Global Biogeochem. Cycles*, *10*, 401–412, 1996.

Potter, C. S., P. A. Matson, P. M. Vitousek, and E. A. Davidson, Process modeling of controls on nitrogen trace gas emissions from soils worldwide, *J. Geophys. Res.*, *101*, 1361–1377, 1996.

Potter, C. S., E. A. Davidson, S. A. Klooster, D. C. Nepstad, G. H. De Negreiros, and V. Brooks, Regional application of an ecosystem production model for studies of biogeochemistry in Brazilian Amazonia, *Global Change Biol.*, *4*, 315–333, 1998.

Riley, R. H., and P. M. Vitousek, Nutrient dynamics and nitrogen trace gas flux during ecosystem development in montane rain forest, *Ecology*, *76*, 292–304, 1995.

Schnell, S., and G. M. King, Mechanistic analysis of ammonium inhibition of atmospheric methane consumption in forest soils, *Appl. Environ. Microbiol.*, *60*, 3514–3521, 1994.

Serca, D., R. Delmas, C. Jambert, and L. Labroue, Emissions of nitrogen oxides from equatorial rain forest in central Africa: Origin and regulation of NO emission from soils, *Tellus, Ser. B*, *46*, 243–254, 1994.

Sitaula, B. K., L. R. Bakken, and G. Abrahamsen, CH_4 uptake by temperate forest soil: Effect of N input and soil acidification, *Soil Biol. Biochem.*, *27*, 871–880, 1995.

Spain, A. V., Litterfall and the standing crop of litter in three tropical Australian rainforests, *J. Ecol.*, *72*, 947–961, 1984.

Steinkamp, R., K. Butterbach-Bahl, and H. Papen, Methane oxidation by soils of an N-limited and N-fertilized spruce forest in the Black Forest, Germany, *Soil Biol. Biochem.*, *33*, 145–153, 2000.

Steudler, P. A., R. D. Bowden, J. M. Melillo, and J. D. Aber, Influence of nitrogen fertilization on methane uptake in temperate forest soils, *Nature*, *341*, 314–316, 1989.

Steudler, P. A., J. M. Melillo, R. D. Bowden, M. S. Castro, and A. E. Lugo, The effect of natural and human disturbance on soil nitrogen dynamics and trace gas fluxes in a Puerto Rican wet forest, *Biotropica*, *23*, 356–363, 1991.

Steudler, P. A., J. M. Melillo, B. J. Feigl, C. Neill, M. C. Piccolo, and C. C. Cerri, Consequence of forest-to-pasture conversion on CH_4 fluxes in the Brazilian Amazon Basin, *J. Geophys. Res.*, *101*, 18,547–18,554, 1996.

Stocker, G. C., W. A. Thompson, A. K. Irvine, J. D. Fitzsimon, and P. R. Thomas, Annual patterns of litterfall in a lowland and Tableland rainforest in tropical Australia, *Biotropica*, *27*, 412–420, 1995.

Tathy, J. P., R. A. Delmas, A. Marenco, B. Gros, M. Labat, and J. Servant, Methane emission from flooded forest in Central Africa, *J. Geophys. Res.*, *97*, 6159–6168, 1992.

Tracey, J. G., The vegetation of the humid tropical region of North Queensland, report, Commonw. Sci. and Ind. Res. Org., Melbourne, 1982.

Verchot, L. V., E. A. Davidson, J. H. Cattânnio, I. L. Ackermann, H. E. Erickson, and M. Keller, Land use and biogeochemical controls of nitrogen oxide emissions from soils in eastern Amazonia, *Global Biogeochem. Cycles*, *13*, 31–46, 1999.

Verchot, V. L., E. A. Davidson, J. H. Cattânio, and I. L. Ackerman, Land-use change and biogeochemical controls of methane fluxes in soils of Eastern Amazonia, *Ecosystems*, *3*, 41–56, 2000.

Vitousek, P. M., Litterfall, nutrient cycling, and nutrient limitation in tropical forests, *Ecology*, *65*, 285–298, 1984.

Vitousek, P. M., P. Matson, C. Volkmann, J. M. Maass, and G. Garcia-Méndez, Nitrous oxide flux from dry tropical forests, *Global Biogeochem Cycles*, *3*, 375–382, 1989.

Vitousek, P. M., D. R. Turner, W. J. Parton, and R. L. Sanford, Litter decomposition on the Mauna Loa environmental matrix, Hawai'i: Patterns, mechanisms, and models, *Ecology*, *75*, 418–429, 1994.

Vourlitis, G. L., N. Priante Filho, M. M. S. Hayashi, J. D. S. Nogueira, F. T. Caseiro, and J. Holanda Campelo, Seasonal variations in the net ecosystem CO_2 exchange of a mature Amazonian transitional tropical forest (cerradao), *Funct. Ecol.*, *15*, 388–395, 2001.

Weitz, A. M., E. Verchot, M. Keller, J. Neff, and P. M. Crill, Nitrous oxide, nitric oxide, and methane fluxes from soils following clearing and burning of tropical secondary forest, *J. Geophys. Res.*, *103*, 28,047–28,058, 1998.

K. Butterbach-Bahl and R. Kiese, Institute of Meteorology and Climate Research, Atmospheric Environmental Research (IMK-IFU), Karlsruhe Research Centre, Kreuzeckbahnstr. 19, D-82467 Garmisch-Partenkirchen, Germany. (klaus.butterbach@imk.fzk.de; ralf.kiese@imk.fzk.de)

A. Graham and B. Hewett, CSIRO, Tropical Forest Research Centre, Maunds Rd Atherton, QLD 4883, Australia. (andrew.graham@csiro.au; bob.hewett@csiro.au)

Regional application of PnET-N-DNDC for estimating the N_2O source strength of tropical rainforests in the Wet Tropics of Australia

RALF KIESE*, CHANGSHENG LI,[†] DAVID W. HILBERT,[‡] HANS PAPEN[*] and KLAUS BUTTERBACH-BAHL[*]

[*] Institute for Meteorology and Climate Research, Atmospheric Environmental Research (IMK-IFU), Karlsruhe Research Centre, Kreuzeckbahnstr. 19, D-82467 Garmisch-Partenkirchen, Germany, [†] Institute for the Study of Earth, Oceans, and Space, University of New Hampshire, Durham, NH 03824, USA, [‡] CSIRO, Tropical Forest Research Centre, Maunds Rd,, Atherton, QLD 4883, Australia

Keywords: biogeochemical model, PnET-N-DNDC, N_2O-emission inventory, tropical rainforest, sensitivity analysis, GIS

Abstract

Though there is no doubt about the importance of tropical rain forest soils as a dominant source within the global atmospheric N_2O budget, regional estimates for tropical forest ecosystems are still limited and highly uncertain. To contribute towards a higher reliability of estimates of the N_2O source strength of tropical rainforest ecosystems on a regional scale, we further developed the process oriented biogeochemical model PnET-N-DNDC, which was initially designed to simulate C- and N-turnover and associated N_2O-emissions in/ from temperate forest ecosystems. This required several model modifications affecting forest physiology and phenology, soil hydrology, denitrifier activity and biological N_2-fixation. In order to evaluate and validate the capability of the model to simulate N_2O-emissions from tropical rainforest soils the modified PnET-N-DNDC model was applied to different tropical rainforest sites in the Wet Tropics of Australia (three montane sites and one lowland site) and in Costa Rica (two lowland sites). Simulated N_2O-emissions for all sites were in very good agreement with field observations. The model was even able to simulate short-term changes in N_2O-emissions during the wet season, which were mainly driven by changes in soil moisture. Since the model could predict site and seasonal variations of N_2O fluxes with reasonable accuracy, we finally linked the PnET-N-DNDC model to a GIS database in order to calculate a regional inventory of N_2O-emission for rainforest soils of the Wet Tropics of Australia. The annual sum of simulated N_2O-emissions in this area varied

from 1.3 - 4.6 kg N_2O-N ha^{-1} y^{-1} (mean value: 2.40 kg N_2O-N ha^{-1} y^{-1}). This is in very good agreement with previous field observations within the study area. Regional sensitivity analyses revealed that the PnET-N-DNDC model was most sensitive to changes in soil texture, leaf C/N ratio, biomass and SOC content.

Introduction

In view of global climate change the precise quantification of sources and sinks of greenhouse gases like CO_2, N_2O and CH_4 for natural, agricultural and forest ecosystems is a crucial task. Due to the commitments to the Kyoto Protocol, Annex I states are obliged to report their greenhouse gas emissions by sources and removals by sinks in a transparent and verifiable manner. However, the current approach suggested to use, the IPCC assessment, does not provide sufficiently detailed spatial information for diffuse sources of greenhouse gases like soil emissions (Kroeze & Mosier, 2002). Furthermore, with regard to the global atmospheric budgets of greenhouse gases, it is essential to accurately quantify the role of soils as sinks and sources. Though tropical rainforest soils are one of the major sources for N_2O, contributing at least 14-23% to the global atmospheric N_2O budget (IPCC, 1997), their source strength is not well constrained, since available datasets are sparse, both spatially and temporally. Reported N_2O-emissions from tropical rainforest ecosystems are between 0.3 - 7.5 kg N ha^{-1} y^{-1} (e.g. Matson & Vitousek, 1990; Garcia Méndez *et al.*, 1991; Steudler *et al.*, 1991; Keller & Reiners, 1994; Riley & Vitousek, 1995; Verchot *et al.*, 1999; Breuer *et al.*, 2000; Kiese & Butterbach-Bahl, 2002; Kiese *et al.*, 2003).

N_2O-emissions from soils are mainly the result of the microbial processes of nitrification and denitrification (e.g. Firestone & Davidson, 1989; Conrad, 1996; Bremner, 1997). Both processes have been shown to be strongly dependent on soil properties (e.g. texture, pH, soil organic carbon), substrate availability, inter- and intraspecies competition for substrates, and environmental parameters such as soil moisture and soil temperature (Firestone & Davidson, 1989; Yamulki *et al.*, 1997; Verchot et al., 1999; Kiese & Butterbach-Bahl, 2002). Since both, the biotic and abiotic factors are highly variable in space and time, observed N_2O-emissions from rainforest soils show a substantial spatial and temporal variability (Breuer *et al.*, 2000), which results in the above mentioned uncertainty in estimating the source strength of N_2O for tropical rainforest soils. To improve these estimates it will be necessary to follow a dual approach. On the one hand more detailed measurements of rates of N_2O-emissions from different tropical rainforest ecosystems are required, which should fulfill both representativeness and long-term coverage of measurements in order to better understand seasonal variability and to identify the major environmental drivers which control the magnitude of N_2O-emissions (Kiese *et al.*, 2003). On the other hand, these data are needed to further develop and validate biogeochemical models, which simulate all processes and mechanism involved in N- and C-trace gas emissions from soils. These so called process oriented models provide the most promising tool to upscale emissions from the site to a regional scale (e.g. Davidson *et al.*, 1998; Li *et al.*, 2000; Butterbach-Bahl *et al.*, 2001). However, for regional calculation of N_2O emissions it is essential to link the biogeochemical models to Geographic Information Systems (GIS) databases, which hold the spatially differentiated information of all parameters needed for driving and initializing the models.

In the recent past several biogeochemical models (e.g. the CASA, CENTURY, DNDC and the PnET-N-DNDC models) have been developed, which focus on simulating C and N turnover in soils and on predicting soil-atmosphere exchange of N and C trace gases (Potter *et al.*, 1996; Parton et al., 1996; Li et al., 1992; Li et al., 2000). The PnET-N-DNDC model, was developed to explicitly model biogeochemical cycling of C and N and associated biosphere-atmosphere exchange of N-trace gases in temperate forest ecosystems The intention of the present study was to adapt and validate this model for tropical rainforest conditions, in order to allow the simulation of N_2O-emissions from these ecosystem. Finally, in the framework of a case study, we aimed to use the new version of the PnET-N-DNDC model to calculate a detailed N_2O-emission inventory for rainforests in the area of the Wet Tropics of Australia.

Materials and Methods

Adapting PnET-N-DNDC to tropical rainforests

The PnET-N-DNDC model was initially developed to predict soil carbon (C) and nitrogen (N) biogeochemistry, including N-trace gas emissions, in temperate forest ecosystems (Li *et al.*, 2000; Stange *et al.*, 2000; Butterbach-Bahl *et al.*, 2001). The model consists of two components. The first component, which includes the soil climate sub-model, the forest growth sub-model and the decomposition sub-model, predicts soil environmental conditions (e.g., temperature, moisture, Eh, pH and substrate concentration gradients) based on observed meteorological data, soil properties, forest type and age, and forest management. The second component includes the sub-models for nitrification, denitrification and fermentation and calculates fluxes of N_2O, NO, N_2, NH_3 and CH_4 based on the soil microbial activities driven by the soil environmental factors. The six interacting sub-models are linked to the primary drivers (i.e. climate, soil, vegetation and anthropogenic activity), to soil environment (e.g. temperature and oxygen profiles in the soil) and a series of biochemical and geochemical reactions, which finally determine the transport and transformation of C and N components. To adapt the model for simulating C and N turnover and associated N_2O-emissions to/ from tropical rainforest ecosystems, several modifications have been made to the original algorithms. A new forest type, “Tropical Rainforest”, has been introduced in PnET-N-DNDC to simulate the physiology and phenology of tropical rainforests ecosystems. The physiological parameters for the rainforest were partly adopted from the parameterization of the “Evergreen Broadleaf Forest“ used in the BIOM-BGC model (Hunt *et al.*, 1996), but are also based on our own measurements. For the tropical rainforest, we defined leaf N re-translocation to be 0.85, wood C/N ratio = 100, leaf retention = 1.5 years, and optimum temperature of photosynthesis to be 30°C. Leaf C/N ratio was set to be 30 based on the field measurements performed by Kiese *et al.* (2003, and unpublished data) who observed a leaf C/N ratio range of 25 to 44 for a variety of rainforest sites in the Wet Tropics of Australia. Kiese *et al.* (2003) and Spain (1984) reported that leaf litter fall was mainly driven by soil water stress and heavy rainfall events in the study area. During the dry season, leaf litter fall increased if water stress occurred, whereas during the wet season, peaks of leaf litter fall were caused by physical damage due to heavy rainfall (Kiese *et al.*, 2003). These observations were converted into two equations:

daily leaf litter fall = canopy leaf mass * 0.003 * (1- water stress factor);
daily leaf litter fall = canopy leaf mass * 0.005, if daily rainfall >100 mm.

Since most of the soil process-related functions or algorithms existing in the original PnET-N-DNDC are process-based, we assumed that they should be basically applicable across climatic zones or soil types. Modifications were necessary only in the soil climate sub-model. In the original version of PnET-N-DNDC, this sub-model features matrix flow for 12 texture classes (Li *et al.*, 1992 and 2000). Due to their aggregate structures and high bioturbation tropical forest soils often have more macropores as temperate forest soils. This leads to increased values of saturated hydraulic conductivity even in the heavier texture classes. In order to cope these phenomena, we increased the saturated hydraulic conductivity for all texture classes by a factor of approx. 2. This figure is based on own measurements and on investigations from Bonell *et al* (1983), who showed for a catchment in the center of the study area (see below) that values for the saturated hydraulic conductivity were up to 20 m d^{-1}.

To improve the simulation of the seasonality of N-turnover processes in tropical rainforest soils, an activity-index for the denitrifier population (denitrifier_activity_factor) was introduced. This index was allowed to vary in a range of 0.008 - 1.12. Under dry season conditions the activity index can be lower than 0.5 whereas the activity index will increase and approach to 1 under wet season conditions. Following the approaches utilized for nitrifier activity in the original PnET-N-DNDC model (Stange, 2000), we calculate the denitrifier activity based on modeled soil moisture values. If soil moisture (in PnET-N-DNDC: water filled pore space) is lower than 0.6, the denitrifier activity is reduced by 20%, otherwise it is increase by 30% at a daily time step.

Previously, PnET-N-DNDC has neglected biological N_2-fixation as a source of N input into forest ecosystems. This may be acceptable for well aerated temperate forest ecosystems, where biological N_2-fixation is assumed to be <2 kg N ha^{-1} y^{-1} (see literature review by Cleveland *et al.*, 1999), but biological N_2-fixation cannot be neglected in tropical forests. Several investigations demonstrate that biological N_2-fixation by epiphytes, lichens, non symbiotic N_2 fixing bacteria in soil and litter and by symbiotic plants in tropical rainforest ecosystems can make substantial contributions to the N cycling in tropical forests and is likely to vary in a range of <2 – 200 kg N ha^{-1} y^{-1} (Sylvester-Bradley *et al.*, 1980; Forman, 1975; Salati *et al.*, 1982; Goosem & Lamb, 1986). Based on an exhaustive literature review, Cleveland *et al.* (1999) estimated the mean N_2-fixation rate of evergreen tropical rainforests to be approx. 14.7 - 36.1 kg N ha^{-1} y^{-1}. Furthermore, these authors showed that the rate of biological N_2-fixation in terrestrial ecosystems was closely related to the ecosystem evapotranspiration. Based on this review , we introduced a simple linear regression to PnET-N-DNDC to quantify the N_2-fixation rate:

$$N_{fix} = R_{et} * 0.234 - 4.71 * 10^{-4},$$

where N_{fix} is the N_2-fixation rate (kg N ha^{-1} d^{-1}), and R_{et} is the ecosystem evapotranspiration rate (cm water d^{-1}).

With the above modifications for forest physiology/phenology, soil hydrology, denitrification, and N_2-fixation, the new version of PnET-N-DNDC is able to simulate C and N biogeochemical cycles for tropical rainforests.

Model application

In order to evaluate the capability of the modified PnET-N-DNDC model in simulating N_2O-emission from tropical rainforest soils a series of sensitivity tests were conducted with the major environmental factors, including climate, soil properties, forest biomass etc. In addition, the new version of the model was validated against field data sets observed at four tropical rainforest sites in the Wet Tropics of Australia and two sites in Costa Rica for which measurements of N_2O-emissions covering a reasonable time span (e.g. at least 1 month) were available. However, datasets covering longer periods of time on N_2O-emissions from tropical rainforest soils are sparse, and, at present, only the data set from Kiese *et al.* (2003) reports the seasonality of N_2O-emissions in at least daily resolution for an entire year. Furthermore, the availability of daily measured meteorological input parameters was a major restriction in applying the modified PnET-N-DNDC model to other datasets of N_2O-emissions from tropical rainforest soils. For this reason we could e.g. not apply the model to the sites of Verchot *et al*. (1999) and Melillo *et al* (2001) in Amazonia.

For documentation of model performance and for comparison between measured and simulated values, model efficiency values (r^2) were calculated by using the following equation (see also Stange *et al.*, 2000):

$$r^2 = 1 - \left(\frac{\sum \left(x_{mod} - x_{meas} \right)^2}{\sum \left(x_{meas} - \bar{x}_{meas} \right)^2} \right)$$

where x_{mod} is the value obtained from simulation runs using PnET-N-DNDC, x_{meas} is the value obtained from the measured field data, and $\bar{x}_{meas}$ is the average of measured field data.

Regional Model Drivers

The study area of the “Wet Tropics” (approx. 9000 km^2) is located in the Northeast of Queensland, Australia. This area includes lowland tropical rainforests along the coast and montane tropical rainforest on the Atherton Tablelands, an elevated plateau varying in altitude from 700 to 900 m a.s.l. The Coastal Lowlands and the Atherton Tablelands are divided by the Great Escarpement with summits going up to 1600 m (e.g. Mt. Bartel Frere, Queensland highest mountain) (Fig. 1).

Climate. Data on daily rainfall and temperature for the years 1997-1998 were obtained for 61 meteorological stations within the study area from the Bureau of Meteorology, Brisbane, Australia. Due to the rough and complex topography, the spatial interpolation of precipitation data was based on the map of mean annual rainfall within the study area, which is itself based on data from a large number of rainfall stations and a thin plate spline interpolation using a digital terrain model (Houlder *et al.*, 1999). In a next step this map was superimposed with the location of the available climate stations and representative polygons were calculated using a cost distance approach, which considers the gradient of spatial changes in annual rainfall within a given area. Since this approach used for the regionalization of daily rainfall amounts already considers aspects of topography and altitude, we used the same polygons for the spatial distribution of daily temperature values. Climate data were prepared in such a way that the model was initialized for dry season conditions, i.e. for July 1997.

For all 61 climate stations, the annual sum of rainfall for the period July 1, 1997 - June 30, 1998, varied within a range of 789.2 – 7839.6 mm (mean value: 2689.0 ± 155.3 mm). Highest rainfall values were obtained for the Bellenden Ker region (climate station at summit of Mt. Bellenden Ker), which is located in the center of the study area. Significantly lower rainfall was observed for areas on the Atherton Tablelands (mean value: 1685.5 ± 226.3 mm) and in the most Southern part of the study area (mean value: 2424.1 ± 117.4 mm). Highest mean annual temperature values were observed for the tropical lowland areas close to the coast (24.2 °C), whereas records for the mean annual temperature at the Atherton Tablelands were approx. 3.5 °C lower.

Seasonal variations in rainfall were pronounced. During the wet season, which lasted from 11/97 to 03/98, more than 68% of the annual rainfall was observed, whereas during the dry season (07/97-09/97 + 05/98-06/98) less than 21% of precipitation occurred. The seasonality of temperature was less pronounced. However, during the dry period the daily mean temperature was approx. 4.5 °C lower as compared to wet season conditions. This differences were more pronounced for the Atherton Tablelands than for the Coastal Lowlands.

Soils. Several digital maps on the distribution of soil series for specific regions of the study area, i.e. Townsville, Ingham, Cardwell-Tully, Babinda-Cairns, Mossman, Atherton and Ravenshoe, were provided by the Department of Primary Industries (DPI) at Mareeba, Queensland, Australia. Information on soil properties in the uppermost 10 cm of the mineral soil, i.e. soil organic carbon (SOC), C/N ratio, soil pH, skeleton fraction and texture were derived from various reports (Murtha, 1986; Laffan, 1988; Murtha, 1989; Wilson *et al*., 1990; Cannon *et al*., 1992; Heiner *et al.* 1994; Murtha *et al*., 1996; Malcolm *et al*., 1999) for each specific region. However, these reports mostly provide only one value, instead of a range, for soil properties for a given soil type within such a region. Therefore, we used these specific values for model initialization. Estimates of soil fertility were based on the total N-content of the soils. Areas with high soil fertility (N > 0.3%) covered approx. 28% of the study area, whereas soils with medium (0.15% < N < 0.3%) and low soil fertility (N < 0.15%) covered 42% and 30% of the study area, respectively.

According to the Australian soil classification, approx. 75 different soil series were found within the study area. Soil properties for these soil types varied significantly, e.g. pH values from 4.1 to 7.0 and SOC in a range of 0.5 to 5.9%. Table 1 summarizes soil characteristics of the different soil series found within the study area grouped after the classification of Soil Taxonomy 1975 (USDA).

Rainforest distribution. The distribution of rainforest within the study area was based on the work of Tracey (1982). For the GIS exercise we only considered the following forest classifications as true rainforests: complex notophyll, notophyll, simple notophyll, complex mesophyll, mesophyll, semideciduous mesophyll, simple microphyll vine forest, mesophyll vine forest with dominant palms, simple microphyll vine-fern forest, simple microphyll vine-fern thicket, deciduous microphyll vine thicket. In this study we excluded sclerophyll forests dominated by *Eucalyptus* spp as well as sites which were cleared during the last decades (Fig. 1). For this reason we simulated N_2O-emissions from rainforest soils for an area of 4011 km^2.

Results and Discussion

Model Validation for Different Field Sites

In order to assess the capability of the model to simulate N_2O-emissions from tropical rainforest soils, the PnET-N-DNDC model was applied to 6 different field sites. Four of the sites were located in the Wet Tropics of Australia and two were in Costa Rica (Table 2). Mean annual temperature for the different sites ranged from 20.1 °C for the montane rainforest at Massey Creek, Australia, to 25.8°C for the lowland rainforest sites in Costa Rica. Also the mean annual sum of precipitation was variable between the sites. Lowest rainfall is expected for the montane rainforest site at Lake Eacham, Australia (1500 mm), whereas the annual rainfall at the lowland rainforest site, Bellenden Ker, Australia, is approx. 3 fold higher (4395 mm) (Table 2). The six sites also differed substantially in soil characteristics. The pH varied from 3.6 (sites in Costa Rica) to 5.5 (Massey Creek, Australia) and the soil organic carbon from 2.3% (Lake Eacham, Australia) to 5.2% (Massey Creek, Australia).

Figure 2 shows a comparison of the seasonal variability of observed and simulated N_2O-emissions for four of the six sites. The dataset on N_2O-emissions from the lowland rainforest site at Bellenden Ker, Australia (Kiese *et al.*, 2003), is at present the most detailed available anywhere in the world. Simulated N_2O-emissions there, were in very good agreement with field observations of the seasonal pattern of N_2O-emissions. The model was even capable of simulating short-term changes in N_2O-emissions during the wet season (January – Mid-May), which were mainly caused by changes in soil moisture (Kiese *et al.*, 2003). However, during most periods, the model overestimated N_2O-emissions by approx. a factor of 2, so that in consequence simulated mean N_2O-emissions for the period 11/01 - 10/02 were also approx. 2 fold higher (4.6 ± 0.2 g N_2O-N ha^{-1} d^{-1}) than mean observed N_2O-emissions (2.5 ± 0.1 g N_2O-N ha^{-1} d^{-1}) (Table 3). Simulated N_2O-emissions at the Bellenden Ker site were in better agreement with field measurements for the period 02-03/00 & 12/00-01/01. For these periods, mean simulated (22.8 ± 2.9 g N_2O-N ha^{-1} d^{-1}) and mean measured N_2O-emissions (30.6 ± 3.7 g N_2O-N ha^{-1} d^{-1}) were not statistically different and the model efficiency r^2 increased to 0.23 (Table 3).

A comparison of measured and simulated N_2O-emissions at the Kauri Creek site shows that the model moderately underestimated emissions during wet season conditions (01/00 – 02/00), but the simulation only differed slightly from the field measurements during the transition period from the dry to the wet season at the end of the year 2000. For the entire period of the year 2000 for which N_2O-emission measurements were available, the mean value of simulated N_2O-emissions was 7.7 ± 0.9 g N_2O-N ha^{-1} d^{-1} and, thus only 15.4 % lower than the calculated mean of measured N_2O-emissions (9.1 ± 1.2 g N_2O-N ha^{-1} d^{-1}) (Table 3). However, since the meteorological data for this site were obtained from a station approx. 20 km apart from the measuring site, the relatively small deviations between the simulated and the measured N_2O-fluxes could also be related to the spatial variability of rainfall within this area. This problem induced from the representativeness of meteorological input data also holds for the sites at Massey Creek and Lake Eacham, both montane rainforests sites on the Atherton Tablelands. Taking the spatial variability of rainfall into account, especially with frequent occurrence of convective rainfall events in tropical regions, simulated and measured N_2O-emissions for the Massey Creek and Lake Eacham sites were generally in very good agreement with regard to mean, minimum and maximum values as well as with regard to observed and simulated site and seasonal differences (Table 3). Site differences in the magnitude of simulated N_2O-emissions between the Lake Eacham and the Kauri Creek site were mainly driven by differences in soil organic carbon, whereas the differences between the Kauri Creek and Massey Creek site were mainly due to lower temperatures but higher rainfall at the latter site. These findings are in excellent agreement with results obtained by field observations (Breuer *et al.*, 2000).

It is always a challenge to compare the modeled daily gas fluxes with field data sets measured with lower time frequencies. For example, PnET-N-DNDC predicted a huge temporal variability of N_2O fluxes from the two lowland rainforest sites in Costa Rica, though the monthly field data showed a lower variability (Figure 2) (Keller *et al.*, 1993, Keller &

Reiners, 1994). However, the daily predictions were generally in the same range as the monthly measured N_2O-emission and did reflect the seasonal pattern of N_2O-emissions at both sites (Fig. 2). This was also the case if daily simulated N_2O-emission were aggregated to mean monthly values. Furthermore, the good agreement of monthly simulated and measured values was mirrored by high values of model efficiency (La Selva, $r^2 = 0.68$, Guacimo, $r^2 = 0.38$). The mean value of simulated N_2O-emissions at the La Selva site is 12.4 ± 2.8 g N_2O-N ha^{-1} day^{-1}, and thus approx. 22.5 % lower than the mean value which was calculated from the results of the field measurements (16.0 ± 3.7 g N_2O-N ha^{-1} d^{-1}). The differences between mean monthly measured and simulated values at the Guacimo site were even lower (14.5 %). The model was also able to predict the differences in the magnitude of N_2O-emissions between both lowland rainforest sites at La Selva and Guacimo. However, in our modeling exercise these differences could be attributed to different climatic conditions, since the time span for the field measurements of Keller & Reiners (1994) at the La Selva site was from October 1990 to September 1991, whereas the time span for the field measurements at the Guacimo site was from March – November 1992 (Keller *et al.*, 1993; Liu *et al.*, 2000).

In general, the applications of PnET-N-DNDC to the six different field sites demonstrated, that the model is able to capture the spatial and temporal variations in soil N_2O fluxes at the site scale with reasonable accuracy, which was in the same range as previous results obtained for temperate forests (Stange *et al.*, 2000; Butterbach-Bahl *et al.*, 2001). The confidence gained from the site validations have set a sound basis for scaling up the model applications to regional scale.

Regional Application

In order to explore the applicability of the PnET-N-DNDC model as a tool for the calculation of regional inventories of N_2O-emissions from areas covered with tropical rainforests we linked the model to a GIS database which held all relevant model drivers and input parameters for the area of the Wet Tropics, Queensland, Australia. The regional distribution of N_2O-emissions in the study area was simulated for the time period July 1, 1997 to June 30, 1998 (Fig. 3). The annual sum of simulated N_2O-emissions varied within a range of 1.3 - 4.6 kg N_2O-N ha^{-1} y^{-1}(mean for the entire study area: 2.40 kg N_2O-N ha^{-1} y^{-1}), which is in good agreement with reported annual N_2O-emission rates for tropical rainforest ecosystems worldwide (see e.g. overview given by Breuer *et al.*, 2000). Highest N_2O-emissions were simulated for the northern region around Mossman and the Daintree National Park with N_2O-emissions rates of up to 4.6 kg N_2O-N ha^{-1} y^{-1}. Furthermore, high N_2O-emissions (> 3.0 kg

N_2O-N ha^{-1} y^{-1}) were predicted for the mountainous Bellenden Ker and Bartel Frere region, located approx. in the center of the study area. N_2O-emissions from both regions, which also receive the highest rainfall amounts within the study area, were significantly higher compared to N_2O-emissions from rainforest soils at the Atherton Tablelands (1.3 – 3.0 kg N_2O-N ha^{-1} yr^{-1}) and from the area south of Tully down to Townsville (1.3 – 2.0 kg N_2O-N ha^{-1} y^{-1}). The significant regional differences in N_2O-emission between various regions of the study area were mainly induced by differences in precipitation and temperature, whereas at relatively small scale, such as within the Atherton Tablelands, the spatial variations in N_2O-emissions were mainly triggered by the heterogeneity of soil properties. This finding is in excellent agreement with the observed regional variability of N_2O-emissions within the study area (Breuer *et al.*, 2000; Kiese & Butterbach-Bahl 2002; and Kiese *et al.*, 2003). The results of the regional differences in N_2O-emissions as calculated by the regional application of PnET-N-DNDC are further supported by reports about regional differences in gross nitrification and denitrification across the study area by Breuer (2000), Breuer *et al.* (2002) and Kiese *et al.* (2002). These authors found, that the differences in N_2O-emissions between sites were highly correlated to nitrification rates, and that the higher N_2O-emissions in lowland rainforest sites as compared to montane rainforest sites were mainly attributed to increased denitrification activity.

Modeled N_2O-emission from rainforest soils in the study area, where vegetation, soil as well as climate data were available in the GIS (4011 km^2), totaled in 962 t N_2O-N y^{-1}. This value is half of the value of N_2O-emissions estimated to come from the official "land use change and forestry" sector in Australia (2000 t N_2O-N y^{-1}), but much smaller than N_2O-emissions from the "agriculture" sector (82000 t N_2O-N y^{-1}) (Australian Greenhouse Office, 2000). In spite of the small area investigated our figure underlines the importance of tropical rainforest soils as a source of atmospheric N_2O on a regional as well as on a global scale.

Model Sensitivity

In order to investigate the sensitivity of the regional predictions of N_2O-emissions to variations in input parameters or model drivers, we initialized the model with moderately changed values (Fig. 4). Fig. 4 shows, that the model reacts most sensitive to changes in soil texture, aboveground biomass and the C/N ratio of the leaf litter. The latter two parameters determine the abundance and turnover of litter and soil organic carbon in the PnET-N-DNDC model, i.e. increases in C/N ratio or aboveground biomass will directly lead to increased rates of mineralization and thus to increased N_2O-emissions (Li *et al.*, 2000). In the sensitivity test, changes in soil texture classes to either lighter or heavier soils lead to a decrease in N_2O-emissions. It may not be very surprising to see the non-linear effects of soil texture on N_2O fluxes as soil texture has multi-effects on soil hydrology, aeration and thus on the processes involved in N_2O production, consumption and emission, i.e. on mineralization, nitrification and denitrification (Li *et al.*, 2000). For example increases in soil clay contents could often lead to increased production of N_2 in favor of N_2O via the denitrification pathway, whereas in soils with low clay content N_2O-emissions are often reduced due to limitations of microbial N- and C-turnover processes by reduced soil water availability (Li *et al*., 2000).

Conclusion

Through the modeling practices reported in the paper, we developed a tropical version of PnET-N-DNDC for quantifying N_2O-emissions from tropical rainforest soils at both site and regional scales. The validations and sensitivity tests have demonstrated that this model is capable of up-scaling N_2O-emissions from a site to a regional scale in order to improve regional and in the future global inventories of N_2O-emissions from tropical forest soils. However, to further increase the accuracy of model predictions, we need to continuously improve and validate the model in the coming years. This will require detailed laboratory experiments on processes governing N-trace gas emissions as well as more detailed field measurements in high temporal resolution on C- and N-cycling and associated N- and C-gas exchange in rainforest ecosystems across climatic zones soil types, vegetation properties, and management regimes. In order to improve our ability in modeling N-trace gas emissions from tropical soils, it must be stressed that the exclusive measurement of N_2O-emissions from soils will not be sufficient in the future. In this context it will be necessary to quantify emissions of all N-gases from soils, i.e. N_2O, NO and N_2 and to measure simultaneously the magnitude of microbial processes involved in N-cycling and N gas emissions, i.e. mainly the processes of nitrification, denitrification, microbial immobilization of N and biological N_2-fixation.

Acknowledgements

The authors thank Bertram Ostendorf, Andrew Graham, and Bob Hewett, Tropical Forest Research Centre, Atherton, Australia, for their support for this project and sharing their expertise about the Wet Tropics of Australia. Many thanks also to Christian Werner, IMK-IFU, Garmisch-Partenkirchen, Germany, for his excellent support with the GIS. This research was funded by the Deutsche Forschungsgemeinschaft (DFG), Germany, under contract BU 1173-3.

References

Bonell M, Cassells DS, Gilmour DA (1983) Vertical soil water movement in a tropical rainforest catchment in Northeast Queensland. *Earth Surface Processes and Landforms*, **8**, 253-272.

Bremner JM (1997) Sources of nitrous oxide in soils. *Nutrient Cycling in Agroecosystems*, **49,** 7-16.

Breuer L, Papen H, Butterbach-Bahl (2000) N_2O emission from tropical forest soils of Australia. *Journal of Geophysical Research*, **105**, 26353-26367.

Breuer L, Kiese R., Butterbach-Bahl K (2002) Temperature and moisture effects on nitrification rates in tropical rain-forest soils. *Soil Science Society of America Journal*, **66**, 834-844.

Butterbach-Bahl K, Stange F, Papen H, Li C (2001) Regional inventory of nitric oxide and nitrous oxide emissions for forest soils of Southeast Germany using the biogeochemical model PnET-N-DNDC. *Journal of Geophysical Research*, **106**, 34155-34166.

Cannon MG, Smith CD, Murtha GG (1992) Soils of the Cardwell-Tully area, North Queensland. Divisional Report No. 115, Commonwealth Scientific and Industrial Research Organization, Melbourne.

Cleveland, C.C., A.R. Townsend, D.S. Schimel, H.Fisher, R.W. Howarth, L.O. Hedin, S.S. Perakis, E.F. Latty, J.C. Von Fischer, A. Elseroad and M.F. Wasson. (1999) Global patterns of terrestrial biological nitrogen (N2) fixation in natural ecosystems. *Global Biogeochemical Cycles*, **13**, 623-645.

Conrad R (1996) Soil microorganisms as controllers of atmospheric trace gases (H_2, CO, CH_4, OCS, N_2O and NO). *Microbiological Reviews*, **60**, 609-640.

Davidson EA, Potter CS, Schlesinger P, Klooster SA (1998) Model estimates of regional nitric oxide emissions from soils of the Southeastern United States. *Ecological Application*, **8***,* 748-759.

Firestone MK, Davidson EA (1989) Microbiological basis of NO and N_2O production and consumption in soil. In: *Exchange of Trace Gases between Terrestrial Ecosystems and the Atmosphere* (eds Andreae MO, Schimel DS), pp. 7-21, John Wiley & Sons Ltd., Chichester, United Kingdom.

Forman RT (1975) Canopy lichens with blue-green algae: A nitrogen source in a Columbian rain forest. *Ecology*, **56**, 1176-1184.

Garzia-Méndez GJ, Maass JM, Matson PA, Vitousek PM (1991) Nitrogen transformations and nitrous oxide flux in a tropical deciduous forest in Mexico. *Oecologia*, **88**, 362-366.

Goosem S, Lamb D (1986) Measurements of Phyllosphere nitrogen fixation in a tropical an two sub-tropical rainforests. *Journal of Tropical Ecology*, **2**, 373-376.

Heiner IJ, Grundy MJ (1994) Land resources of the Ravenshoe - Mt. Garnet area; North Queensland, Vol1 – Land resources inventory. Land Resources Bulletin, Queensland Department of Primary Industries, Brisbane.

Houlder D, Hutchinson M, Nix H, McMahon J (1999) ANUCLIM Version 5.0 User Guide. Centre for Resource and Environmental Studies, Australian National University, Canberra.

Hunt RE, Piper SC, Nemani R, Keeling CD, Otto RD, Running SW (1996) Global net carbon exchange and intra-annual atmospheric CO_2 concentrations predicted by an ecosystem process model and three dimensional atmospheric transport model. *Global Biogeochemical Cycles*, **10**, 431-456.

IPCC (1997) Intergovernmental Panel on Climate Change Guidelines for National Greenhouse Gas Inventories, Chapter 4: Agriculture: Nitrous oxide from agricultural soils and manure management. OECD, Paris.

Keller M, Veldkamp E, Weitz AM, Reiners WA (1993) Effect of pasture age on soil trace-gas emissions from a deforested area of Costa Rica. *Nature*, **365**, 244-246.

Keller M, Reiners WA (1994) Soil atmosphere exchange of nitrous oxide, nitric oxide, and methane under secondary succession of pasture to forest in the Atlantic lowlands of Costa Rica. *Global Biogeochemical Cycles,* **8**, 399-409.

Kiese R, Butterbach-Bahl K (2002) N_2O and CO_2 emissions from three different tropical forest sites in the wet tropics of Queensland, Australia. *Soil Biology and Biochemistry*, **34**, 975-987.

Kiese R, Hewett B, Graham A, Butterbach-Bahl K (2003) Seasonal variability of N_2O-emissions and CH_4-uptake from/ by a tropical rainforest soil of Queensland, Australia. *Global Biogeochemical Cycles*, submitted.

Kiese R, Papen H, Zumbusch E, Butterbach-Bahl K (2002) Nitrification activity in tropical rain forest soils of the Coastal Lowlands and Atherton Tablelands, Queensland, Australia. *Journal of Plant Nutrition and Soil Science*, in press.

Kroeze C¸ Mosier A (2002) Progress in source strength estimates of nitrous oxide (N_2O). In: *Non-CO_2 greenhouse gases: Scientific understanding, control options and policy aspects*, (eds van Ham J, Baede AP, Guicherit R, Williams-Jacobse JG), pp. XXIX–XXXIII, Millpress, Rotterdam.

Laffan MD (1988) Soils and land use on the Atherton Tableland, North Queensland. Soils and land use series No. 61, Commonwealth Scientific and Industrial Research Organization, Melbourne.

Li C, Aber J, Stange F, Butterbach-Bahl K, Papen H (2000) A process-oriented model of N_2O and NO emission from forest soils, 1. Model development. *Journal of Geophysical Research*, **105**, 4369-4384.

Li C, Frolking S, Frolking TA (1992) A model of nitrous oxide evolution from soil driven by rainfall events: 1. Model structure and sensitivity. *Journal of Geophysical Research*, **97**, 9759-9776.

Liu S, Reiners WA, Keller M, Schimel DS (2000) Simulation of nitrous oxide and nitric oxide emissions from a primary forest in the Costa Rican Atlantic Zone. *Env. Mod. & Soft.*, **15**, 727-743.

Malcolm DT, Nagel BK, Sinclair I, Heiner IJ (1999) Soils and agricultural land suitability of the Atherton Tablelands, North Queensland. Land Resources Bulletin, Department of Natural Resources, Brisbane.

Matson PA, Vitousek PM (1990) Ecosystem approach to a global nitrous oxide budget. *Bioscience,* **40**, 667-672.

Melillo JM, Steudler PA, Feigl BJ, Neill C, Garcia D, Piccolo MC, Cerri CC, Tian H. (2001) Nitrous oxide emissions from forests and pastures of various ages in the Brazilian Amazon. *Journal of Geophysical Research*, **106**, 34179-34188.

Murtha GG (1986) Soils of the Tully-Innisfail area, North Queensland. Divisional Report No. 82, Commonwealth Scientific and Industrial Research Organization, Melbourne.

Murtha GG (1989) Soils of the Mossman Cape Tribulation area, North Queensland. Divisional Report No. 102, Commonwealth Scientific and Industrial Research Organization, Melbourne.

Murtha GG, Cannon MG, Smith CD (1996) Soils of the Babinda-Cairns area, North Queensland. Divisional Report No. 123, Commonwealth Scientific and Industrial Research Organization, Melbourne.

Parton WJ, Mosier AR, Ojima DS, Valentine DW, Schimel DS, Weier K, Kulmala AE (1996) Generalized model for N_2 and N_2O production from nitrification and denitrification. *Global Biogeochemical Cycles*, **10**, 401-412.

Potter CS, Matson PA, Vitousek PM, EA Davidson (1996) Process modeling of controls on nitrogen trace gas emissions from soils worldwide. *Journal of Geophysical Research*, **101**, 1361-1377.

Reiners WA, Bouwman AF, Parsons WF, Keller M (1994) Tropical rainforest conversion to pasture: changes in vegetation and soil properties. *Ecological Applications*, **4**, 363-377.

Riley RH, Vitousek PM (1995) Nutrient dynamics and nitrogen trace gas flux during ecosystem development in montane rain forest. *Ecology*, **76**, 292-304.

Salati ER, Sylvester-Bradley R, Victoria RL (1982) Regional gains and losses of nitrogen in the Amazon basin. *Plant and Soil*, **67**, 367-376.

Spain AV (1984) Litterfall and the standing crop of litter in three tropical Australian rainforests. *Journal of Ecology*, **72**, 947-961.

Stange F, Butterbach-Bahl K, Papen H, Zechmeister-Boltenstern S, Li C, Aber J (2000) A process oriented model of N_2O and NO emissions from forest soils: 2, Sensitivity analysis and validation. *Journal of Geophysical Research*, **105**, 4385-4398.

Stange F (2000) Entwicklung und Anwendung eines prozeßorientierten Modells zur Beschreibung der N_2O und NO-Emissionen aus Böden temperater Wälder. Ph.D. Thesis, University Freiburg, Germany, Schriftenreihe des Fraunhofer Institut für Atmosphärische Umweltforschung, 66, pp. 143.

Sylvester-Bradley R, De Oliviera LA, De Podesta Filho JA, St. John TV (1980) Nodulation of legumes, nitrogenase activity of roots, and occurrence of nitrogen-fixing Azospirillum spp. in representative soils of Central Amazonia. *Agro-Ecosystems*, **6**, 249-266.

Steudler PA, Melillo JM, Bowden RD, Castro MS, Lugo AE (1991) The effect of natural and human disturbance on soil nitrogen dynamics and trace gas fluxes in a Puerto Rican wet forest. *Biotropica*, **23**, 356-363.

Tracey J G (1982) The vegetation of the humid tropical region of North Queensland. CSIRO, Melbourne, pp 124.

Verchot LV, Davidson EA, Cattânnio JH, Ackermann IL, Erickson HE, Keller M (1999) Land use and biogeochemical controls of nitrogen oxide emissions from soils in eastern Amazonia. *Global Biogeochemical Cycles*, **13**, 31-46.

Wilson PR, Baker DE (1990) Soils and land suitability of the wet tropical coast of North Queensland: Ingham area. Land Resources Bulletin, Queensland Department of Primary Industries, Brisbane.

Yamulki S, Harrison RM, Goulding KWT, Webster CP (1997) N_2O, NO and NO_2 fluxes from a grassland: Effect of soil pH. *Soil Biology and Biochemistry*. **29**, 1199-1208.

Table 1 Soil characteristics of typical soil types found within the study area. Values given represent properties of the top soil (0 – 10 cm).

Soil order	Area covered (% of study area)	Clay (%)	Skeleton (%)	SOC (%)	C/N ratio	Soil pH
Alfisols	6.1	1 – 35	- 14	1.2 – 3.7	12.8 – 32	5.0 – 6.2
Inceptisols	11.2	9 – 60	- 30	1.1 - 5.7	6.2 – 30.6	4.1 – 7.0
Oxisols	1.0	20 – 50	- 15	0.7 – 5.9	9.1 – 75	4.7 – 6.2
Spodosols	4.4	1 – 24	- 10	0.5 – 2.7	11.0– 28.1	5.1 – 5.9
Ultisols	76.8	9 – 57	- 50	1.1 – 5.3	6.7 – 44.5	4.2 – 6.6
Vertisols/ Entisols	< 1.0					

Table 2 Characteristics of the different tropical rainforest sites used for validation of PnET-N-DNDC

	Bellenden Ker, Australia[a]	Kauri Creek, Australia[b]	Massey Creek, Australia[b]	Lake Eacham, Australia[b]	La Selva, Costa Rica[c,d]	Guacimo, Costa Rica[c,d]
Latitude	17°S	17°S	17°S	17°S	10°N	10°N
Rainforest type	Lowland	Montane	Lowland	Montane	Lowland	Lowland
Soil						
Type	Inceptisol	Inceptisol	Ultisol	Ultisol	Ultisol	Ultisol
Texture	Sandy clay loam	Sandy clay loam	Clay loam	Sandy clay loam	Clay	Clay
Clay fraction	0.22	0.23	0.34	0.21	0.75	0.75
pH mineral soil	4.1	5.2	5.5	4.8	3.6	3.6
SOC	3.1	3.2	5.2	2.3	3.0	3.0
Climate						
Mean Ann. Temp., °C	24.3	20.9	20.1	21.0	25.8	25.8
Annual Rainfall,mm	4395	1594	2350	1500	3962	3962
N in rainfall, ppm	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01

[a] Kiese & Butterbach-Bahl (2002)

[b] Breuer *et al.* (2000)

[c] Reiners *et al.* (1994)

[d] Liu *et al.* (2000)

Table 3. Compilation of results for N_2O-emissions from the different field sites as derived from model runs with PnET-N-DNDC and from field measurements.

Site	Duration of measurements	Number of data points / resolution	Measured N_2O-emission			Simulated N_2O-emission			Model efficiency r^2	Reference for dataset
			g N_2O-N ha^{-1} d^{-1}							
			Mean	Min	Max	Mean	Min	Max		
Lowland rainforest at Bellenden Ker, Australia	11/01 - 10/02	348/ daily	2.5 ± 0.1	0.3	12.4	4.6 ± 0.2	1.0	40.8	…	Kiese *et al.* (2003)
Lowland rainforest at Bellenden Ker, Australia	02/00 - 03/00 & 12/00 - 01/01	32/ daily	30.6±3.7	6.7	82.4	22.8±2.9	3.0	54.9	0.23	Kiese & Butterbach-Bahl (2002)
Montane rainforest at Kauri Creek, Australia	01/00 - 02/00 & 11/00 - 12/00	44/ daily	9.1±1.2	1.4	31.5	7.7±0.9	2.3	38.1	0.68	Kiese & Butterbach-Bahl (2002)
Montane rainforest at Kauri Creek, Australia	07/97	12/ daily	3.0±0.7	1.2	8.7	2.4±0.2	1.7	4.3	0.20	Breuer *et al.* (2000)
Montane rainforest at Massey Creek, Australia	12/98	14/ daily	7.1±1.1	3.1	16.1	7.5±1.3	3.8	19.1	0.48	Breuer *et al.* (2000)
Montane rainforest at Massey Creek, Australia	05/97 - 06/97	19/ daily	2.5±0.2	1.7	4.1	2.1±0.1	1.5	3.3	0.38	Breuer *et al.* (2000)
Montane rainforest at Lake Eacham, Australia	05/97	16/ daily	2.5±0.4	0.8	5.6	3.4±0.4	1.7	6.2	0.60	Breuer *et al.* (2000)
Lowland rainforest at La Selva, Costa Rica	10/90 - 09/91	12/ monthly	16.0±3.7	1.2	23.3	12.4±2.8	2.4	32.9	0.64	Keller & Reiners (1994)
Lowland rainforest at Guacimo, Costa Rica	03/92 - 11/92	7/ monthly	12.4±2.2	6.7	19.1	10.6±2.3	3.3	20.1	0.38	Liu *et al.* (1993)

FIGURES

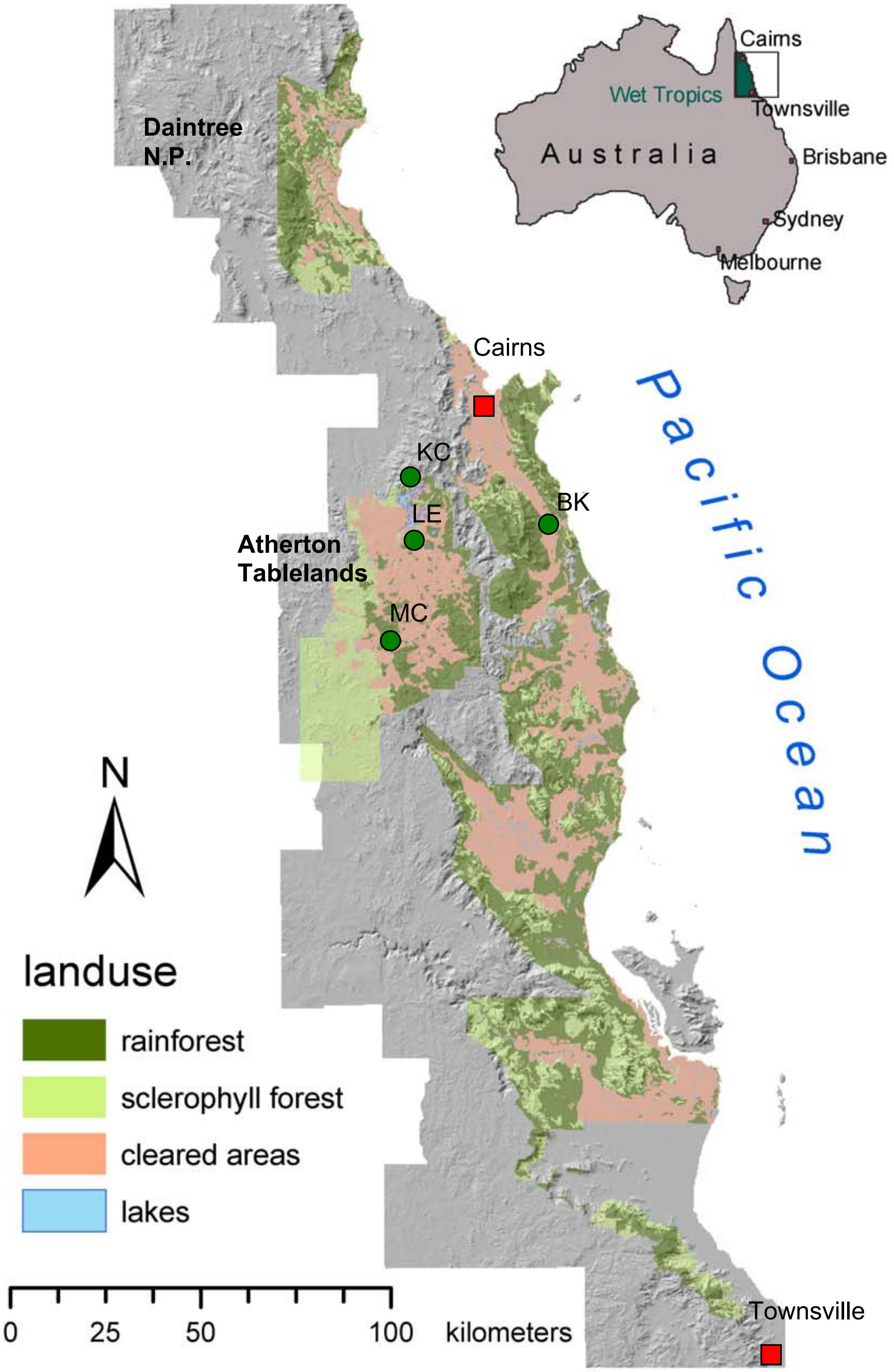
Fig. 1 Location, topography and forest distribution of the study area in North-East Queensland, Australia. (KC: Kauri Creek site; LE: Lake Eacham site; MC: Massey Creek site; BK: Bellenden Ker site)

Fig. 2 Measured and simulated N_2O-emissions (± SE, if available) from lowland rainforest soils at Bellenden Ker (Australia), La Selva and Guacimo (Costa Rica) and from a montane rainforest soil at Kauri Creek (Australia).

Fig. 3 Regional distribution of annual sums of N_2O emissions (kg N_2O-N ha^{-1} y^{-1}) from tropical rainforest in the Wet Tropics of Australia for the year 07/1997-06/1998.

Fig. 4 Sensitivity of regional estimates of N_2O-emissions from rainforest soils in the Wet Tropics of Australia from changes in the input parameters. N_2O-emissions in the baseline scenario are equivalent to 1082 t N_2O-N y^{-1}.

Figure 1



Cairns
Wet Tropics
Townsville
Australia
Brisbane
Sydney
Melbourne
Daintree N.P.
Cairns
KC
LE
BK
Atherton Tablelands
MC
Pacific Ocean
N
landuse
rainforest
sclerophyll forest
cleared areas
lakes
0
25
50
100
kilometers
Townsville

Figure 2

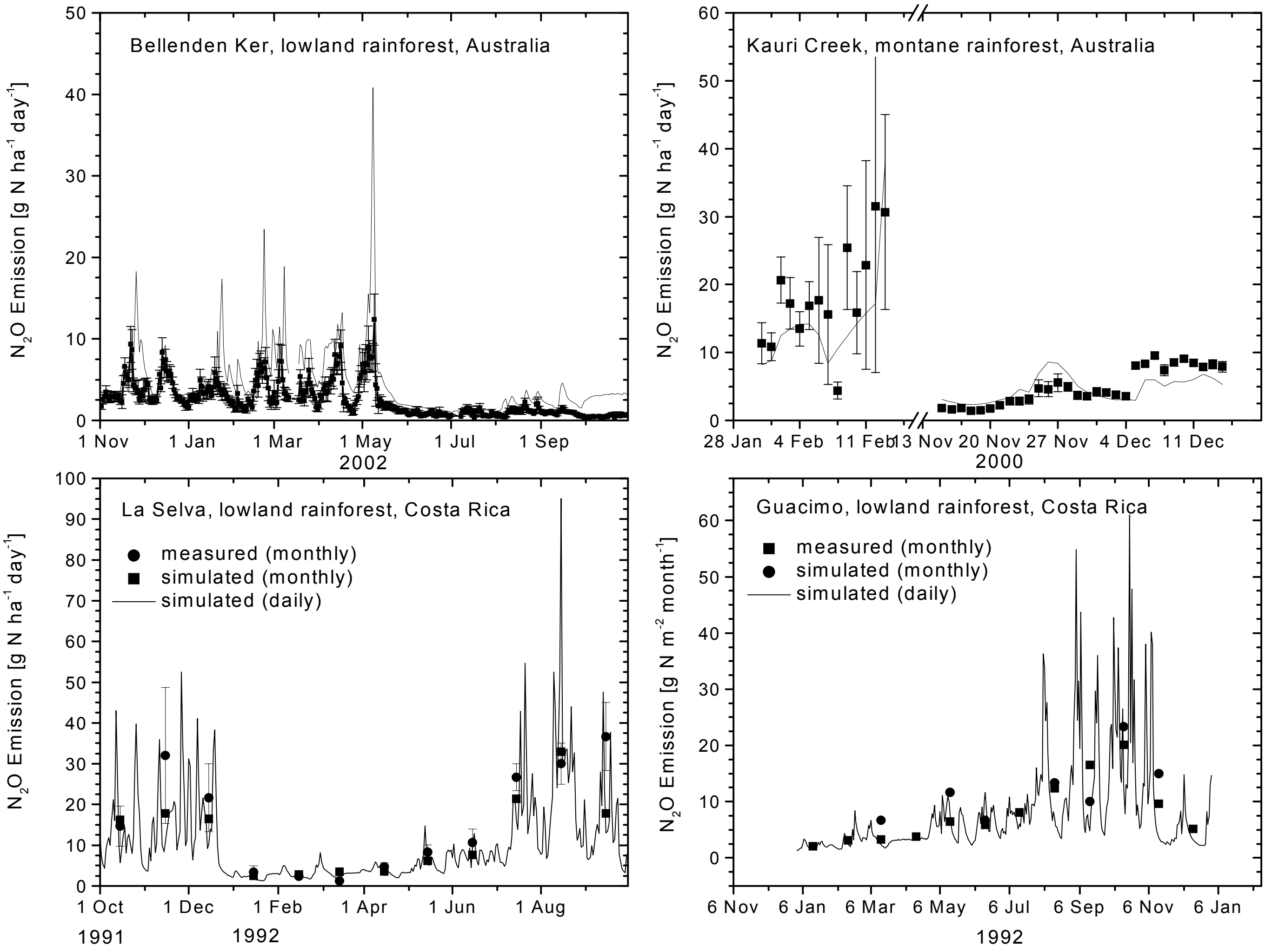

Bellenden Ker, lowland rainforest, Australia
N_2O Emission [g N ha^{-1} day^{-1}]
0
10
20
30
40
50
1 Nov
1 Jan
1 Mar
1 May
1 Jul
1 Sep
2002
Kauri Creek, montane rainforest, Australia
N_2O Emission [g N ha^{-1} day^{-1}]
0
10
20
30
40
50
60
28 Jan
4 Feb
11 Feb
13 Nov
20 Nov
27 Nov
4 Dec
11 Dec
2000
La Selva, lowland rainforest, Costa Rica
● measured (monthly)
■ simulated (monthly)
simulated (daily)
N_2O Emission [g N ha^{-1} day^{-1}]
0
10
20
30
40
50
60
70
80
90
100
1 Oct
1 Dec
1 Feb
1 Apr
1 Jun
1 Aug
1991
1992
Guacimo, lowland rainforest, Costa Rica
■ measured (monthly)
● simulated (monthly)
simulated (daily)
N_2O Emission [g N m^{-2} month^{-1}]
0
10
20
30
40
50
60
6 Nov
6 Jan
6 Mar
6 May
6 Jul
6 Sep
6 Nov
6 Jan
1992

Figure 3

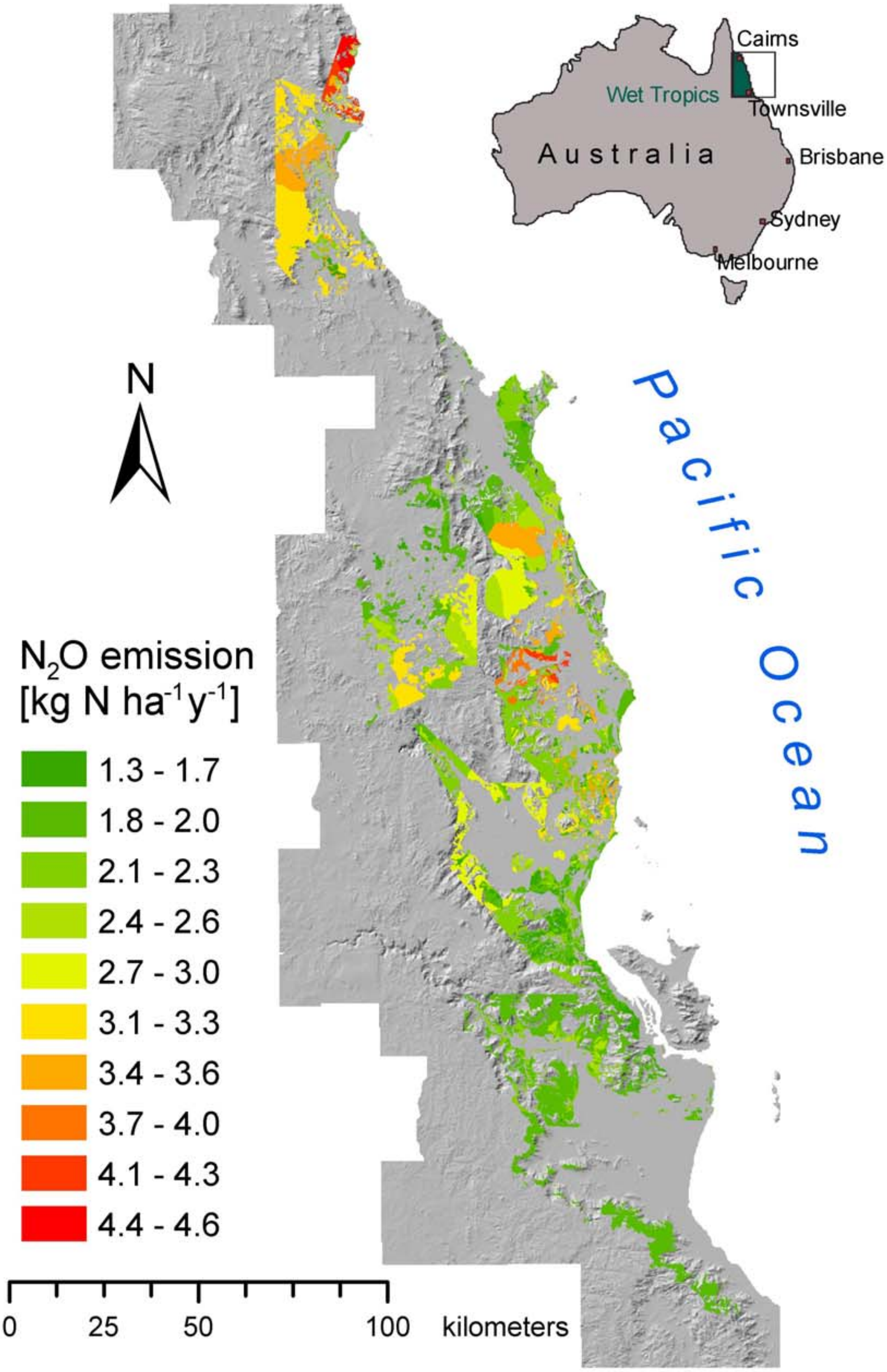
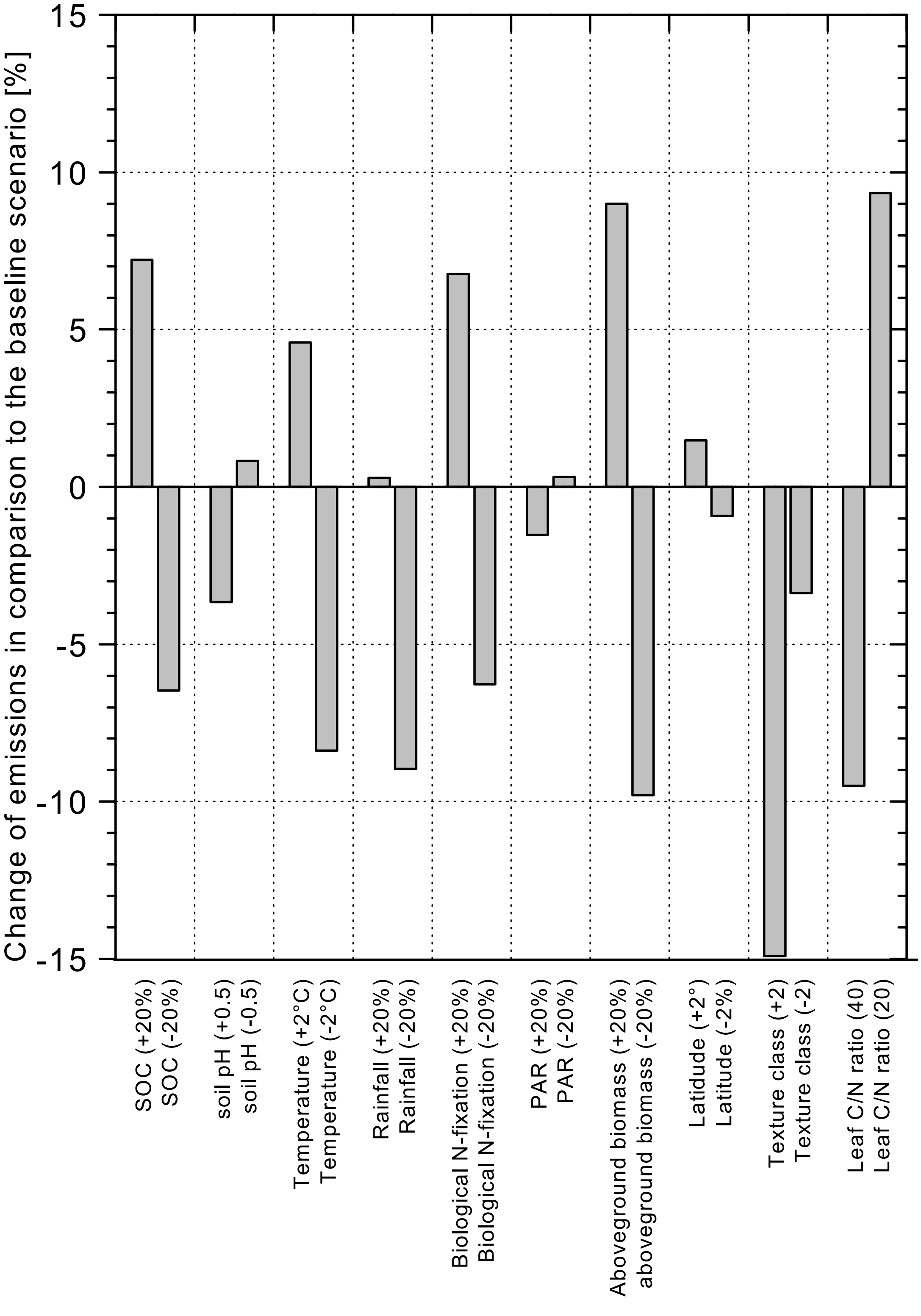

Cairns
Wet Tropics
Townsville
Australia
Brisbane
Sydney
Melbourne
N
Pacific Ocean
N_2O emission
[kg N $ha^{-1}y^{-1}$]
1.3 - 1.7
1.8 - 2.0
2.1 - 2.3
2.4 - 2.6
2.7 - 3.0
3.1 - 3.3
3.4 - 3.6
3.7 - 4.0
4.1 - 4.3
4.4 - 4.6
0
25
50
100
kilometers

Figure 4


15
10
5
0
-5
-10
-15
Change of emissions in comparison to the baseline scenario [%]
SOC (+20%)
SOC (-20%)
soil pH (+0.5)
soil pH (-0.5)
Temperature (+2°C)
Temperature (-2°C)
Rainfall (+20%)
Rainfall (-20%)
Biological N-fixation (+20%)
Biological N-fixation (-20%)
PAR (+20%)
PAR (-20%)
Aboveground biomass (+20%)
aboveground biomass (-20%)
Latidude (+2°)
Latitude (-2%)
Texture class (+2)
Texture class (-2)
Leaf C/N ratio (40)
Leaf C/N ratio (20)