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Biosphere-atmosphere-exchange of C and N trace gases and microbial N turnover processes in irrigated agricultural systems of the Aral Sea Basin, Uzbekistan

ABSTRACT

Land-use and agricultural practices affect the soil microbial carbon (C) and nitrogen (N) turnover and hence the biosphere-atmosphere exchange of greenhouse gasses (GHG), namely N_2O, CH_4 and CO_2. In view of the global importance of irrigated agriculture, it is crucial to understand how and to which extent this land-use system interferes with the terrestrial N and C cycles and contributes to the global source strength of atmospheric GHG. Up to now, knowledge of trace gas exchange and N turnover from irrigated agriculture in arid and semiarid regions is much less developed than in other climate zones. Therefore, this study aims at providing more detailed insights into the biosphere-atmosphere exchange of trace gases and the underlying soil microbial transformation processes of the irrigated agricultural systems in the Aral Sea Basin (ASB), Uzbekistan. A two-year field study was carried out to quantify and compare emissions of N_2O and CH_4 in various annual and perennial land-use systems dominating in the study region Khorezm in western Uzbekistan: irrigated cotton, winter wheat and rice crops, a poplar plantation as well as a natural Tugai (floodplain) forest.

Irrigated agricultural production in the ASB was shown to be a relevant source of GHG. Seasonal variations in N_2O emissions during the annual cropping of wheat and cotton were principally controlled by fertilization and irrigation management. Very high N_2O emissions (> 3000 µg N_2O-N m^{-2} h^{-1}) were measured in periods directly following N fertilizer application in combination with irrigation events. These “emission pulses” accounted for 80-95% of the total N_2O emissions over the cropping season for cotton and wheat. Cumulated emissions over one season varied from 0.5 to 6.5 kg N_2O-N ha^{-1}. The unfertilized poplar plantation showed high N_2O emissions over the entire study period (30µg N_2O-N $m^{-2}h^{-1}$), whereas only negligible fluxes of N_2O (< 2µg N_2O-N $m^{-2}h^{-1}$) occurred in the natural Tugai forest. Observations of significant CH_4 fluxes were restricted to the flooded rice fields, with mean flux rates of 32 mg CH_4 $m^{-2}d^{-1}$ and a seasonal total of 35.2 kg CH_4 ha^{-1}. The global warming potential (GWP) of the N_2O and CH_4 fluxes was highest under rice and cotton, with seasonal changes between 500 and 3000 kg CO_2 eq.ha^{-1}. The biennial cotton-wheat-rice crop rotation commonly practiced in the region averaged a GWP of 2500 kg CO_2 eq.ha^{-1} $year^{-1}$.

In addition, laboratory incubation studies were conducted to assess the aggregated gaseous N losses composed of NO, N_2O, and N_2 from fertilized and irrigated agricultural fields in the ASB. NO_3^- fertilizer and irrigation water were applied to the incubation vessels to assess its influence on the gaseous N emissions. Under the soil conditions, naturally found after concomitant irrigation and fertilization, denitrification was the dominant process and N_2 the main gaseous product of denitrification. Based on the results of these laboratory incubation studies, the magnitude of N_2 emissions for the different field research sites of irrigated cotton could be estimated to be in the range of 24±9 to 175±65 kg-N $ha^{-1}season^{-1}$, while emissions of NO were only of minor importance (between 0.1 and 0.7 kg-N ha^{-1} $season^{-1}$). The findings demonstrate that under the current agricultural practices in the irrigated dryland soils of the ASB, denitrification is a major pathway of N losses and that beside N_2O extensive amounts of N fertilizer are lost as N_2 to the atmosphere.

Moreover, the experimental design of this study allows assessing the potential for reducing GHG emissions from these land-use systems. It is argued that there is wide scope for reducing the GWP of this agroecosystem by (i) optimization of fertilization and irrigation practices and (ii) conversion of annual cropping systems into perennial forest plantations, especially on less profitable, marginal lands.

KURZFASSUNG

Biosphäre-Atmosphäre Austausch von C/N Spurengasen und mikrobielle N Umsetzungsprozesse in bewässerten, landwirtschaftlichen Produktionssystemen des Aralseebeckens, Usbekistan.

Die mikrobiellen Umsetzungsprozesse von Kohlenstoff (C) und Stickstoff (N) in Böden und der damit verbundene Austausch von Treibhausgasen zwischen Biosphäre und Atmosphäre werden maßgeblich von der Landnutzung und den landwirtschaftlichen Methoden beeinflusst. Angesichts der weltweiten Bedeutung von bewässerter Landwirtschaft ist es äußerst wichtig zu verstehen, in wie weit diese landwirtschaftlichen Systeme die globalen N und C Kreisläufe beeinflussen und zu den globalen Treibhausgasemissionen beitragen. Im Gegensatz zu den landwirtschaftlichen Systemen der temperaten Klimazonen ist über N und C Spurengasemissionen aus bewässerter Landwirtschaft in ariden und semiariden Gebieten nur sehr wenig bekannt. Um einen wesentlichen Beitrag zur Schließung dieser Forschungsdefizite zu leisten, konzentrierte sich diese Studie auf den Austausch von strahlungsaktiven Spurengasen zwischen Biosphäre und Atmosphäre und die hiermit assoziierten mikrobiellen N Umsetzungsprozesse in den Böden der bewässerten landwirtschaftlichen Systeme im Aralsee-Becken (ASB) von Usbekistan. Dafür wurde über einen Zeitraum von zwei Jahren in verschiedenen einjährigen und mehrjährigen Landnutzungssystemen die Emissionen der Treibhausgase Lachgas (N_2O) und Methan (CH_4) untersucht. Ausgewählt wurden Landnutzungsysteme die typisch für das Untersuchungsgebiet Khorezm, in West-Usbekistan, sind: bewässerter Baumwoll-, Winter Weizen- und Reisanbau sowie eine Pappel-Plantage und der natürliche „Tugai“ Auenwald entlang des Amu Darya Flusses.

Es konnte festgestellt werden, dass der bewässerte Landbau im ASB insbesondere aufgrund von hohen N_2O Emissionen aus dem Baumwoll- und Weizenanbau eine maßgebliche Quelle von Treibhausgasen darstellt. In den einjährigen Anbausystemen wurden mittlere N_2O Emissionsraten zwischen 10 und 150 µg N_2O-N $m^{-2}h^{-1}$ festgestellt, wobei die höchsten Emissionen in Baumwollfeldern gemessen wurden. Über die gesamte Saison wurden die N_2O Emissionen hauptsächlich von Düngung und Bewässerung beeinflusst. Dabei traten extrem hohe N_2O Emissionen (bis zu 3000 µg N_2O-N m^{-2} h^{-1}) auf, wenn mineralischer N-Dünger direkt vor der Bewässerung appliziert wurde. Diese „Emissionsspitzen“ hatten einen Anteil von 80-95% an den Gesamtemissionen von N_2O bezogen auf die Vegetationsperiode von Baumwolle und Weizen. Insgesamt variierten die N_2O Emissionen über eine Saison von 0,5 bis 6,5 kg N_2O-N ha^{-1}. In der ungedüngten Pappel-Plantage wurden über den gesamten Messzeitraum hohe N_2O Emissionen (30 µg N_2O-N $m^{-2}h^{-1}$) gemessen, wohingegen in dem Tugai Wald lediglich äußerst kleine Flüsse von N_2O (< 2 µg N_2O-N $m^{-2}h^{-1}$) festgestellt wurden. Bedeutende CH_4 Emissionen traten nur in den gefluteten Reisfeldern auf, mit einer durchschnittlichen Flussrate von 32 mg CH_4 $m^{-2}d^{-1}$ und einer Gesamtemission über die Vegetationsperiode von 35,2 kg CH_4 ha^{-1}. Das Treibhauspotenzial der N_2O und CH_4 Flüsse, dargestellt als CO_2-Äquivalent, war am höchsten für den Reis- und Baumwollanbau, wobei auf den verschiedenen Messflächen die Gesamtemission einer Saison von 500 bis zu 3000 kg CO_2 eq.ha^{-1} variierte. Für eine

zweijährige Rotation von Baumwolle-Weizen und Reis, wie sie typisch für das Untersuchungsgebiet ist, konnte ein durchschnittliches Treibhauspotenzial von 2500 kg CO_2 eq.ha^{-1} Jahr^{-1}ermittelt werden.

Zusätzlich wurden im Labor Inkubationsversuche an intakten Bodensäulen durchgeführt um die gasförmigen Stickstoffverluste, bestehend aus NO, N_2O, und N_2, der gedüngten und bewässerten Anbausysteme des ASB zu erfassen. Ammoniumnitrat Dünger wurde zusammen mit Wasser auf die Bodensäulen appliziert, um den Einfluss von gleichzeitiger Düngung und Bewässerung zu simulieren. Es konnte gezeigt werden, dass nach synchroner Düngung und Bewässerung Denitrifikation der vorherrschende Prozess in den Böden ist, und dass der größte Teil des Nitrats vollständig zu molekularem Stickstoff (N_2) denitrifiziert wird. Aufgrund dieser Ergebnisse war es möglich für Baumwolle die Größenordnung der gasförmigen N Verluste von den verschiedenen Messflächen abzuschätzen. Demnach wurden von den einzelnen Baumwollfeldern zwischen 24±9 und 175±65 kg-N ha^{-1}Saison^{-1} als N_2 emittiert, während nur geringe Mengen von NO freigesetzt wurden (zwischen 0,1 und 0,7 kg-N ha^{-1} Saison^{-1}). Diese Studie konnte somit zeigen, dass unter den gegenwärtigen landwirtschaftlichen Methoden im ASB, erhebliche Mengen von Stickstoff durch Denitrifikation als N_2 an die Atmosphäre abgegeben werden.

Ferner erlaubte das experimentelle Design dieser Studie Möglichkeiten einer Reduktion des Ausstoßes von Treibhausgasen aus diesen Anbausystemen abzuschätzen. Abschließend kann festgestellt werden, dass durch (i) eine Optimierung der Dünge- und Bewässerungsmethoden und (ii) einen Wechsel von einjährigen Feldfrüchten auf mehrjährige Baumplantagen, insbesondere auf unrentablen, marginalen Boden, das Treibhauspotential dieses landwirtschaftlichen Produktionssystems wesentlich reduziert werden kann.

АННОТАЦИЯ

Землепользование и сельскохозяйственная практика оказывают своеобразное влияние на трансформацию углерода и азота почвенной микрофлорой и, соответственно, на биосферно-атмосферный обмен парниковых газов. Принимая во внимание огромное значение орошаемого сельского хозяйства, очень важно понимание роли системы земледелия в глобальных циклах N и C, её влияние на всеобщий баланс атмосферных парниковых газов. Однако вопросы о газовых потоках и трансформации азота в орошаемых почвах аридных и полуаридных зон все ещё недостаточно изучены в сравнении с другими регионами.

Целью данных исследований является изучение внутреннего цикла биосферно-атмосферного обмена радиативно-активных газов и процессов микробной трансформации на орошаемых почвах бассейна Аральского моря в Узбекистане.

Двухлетние полевые опыты по изучению указанных выше вопросов проведены в Хорезмской области, которая расположена на северо-западе Узбекистана. На основе результатов опытов проведена количественная оценка и сопоставление размеров эмиссии N_2O и CH_4 в системах земледелия с доминирующими однолетними (хлопчатник, озимая пшеница и рис) культурами и многолетними (плантация тополя и естественный тугайный лес) насаждениями. Изыскания были дополнены инкубационными опытами в лабораторных условиях для оценки общих газообразных потерь N и его составляющих в форме NO, N_2O, и N_2 с орошаемых почв в бассейне Аральского моря.

Выявлено, что орошаемое земледелие в бассейне Аральского моря является важным источником парниковых газов вследствие огромных размеров эмиссии N_2O с пшеничных и хлопковых полей. В системах земледелия с однолетними культурами эмиссия N_2O колеблется в пределах от 10 до 150 μg N-N_2O с квадратной площади в течение часа (м^2/час), где пик эмиссии наблюдается на полях занятых хлопчатником, что совпадает с результатами ранее проведенных исследований. Сезонные колебания эмиссии N_2O контролировались, главным образом, путем внесения удобрений и орошения. Высокий уровень эмиссий N_2O, который достигал 3000 μg/м^2/час N-N_2O, наблюдался в периоды вслед после внесения азотных удобрений в сочетании с вегетационными поливами. Эти "пики" составляют 80-95% от общей эмиссий N_2O в период вегетации хлопчатника и пшеницы, а кумулятивная эмиссия за вегетационный сезон равна 0,5-6,5 кг/га N-N_2O. Неудобренная азотом плантация тополя показала высокий уровень эмиссии N_2O в течение всего эксперимента (30 μg/м^2/час N-N_2O), в то время, как в тугайном лесу потоки N_2O были незначительными (< 2μg μg/м^2/час N-N_2O). Существенные размеры эмиссии CH_4 были отмечены только на затопляемых рисовых чеках, где скорость потока составляла в среднем 32 мг/м^2/день CH_4, а общий сезонный показатель был равен 35,2 кг/га CH_4. Потенциал глобального потепления (ПГП) потоков N_2O и CH_4 был наивысшим на рисовых и хлопковых полях, который был подвергнут сезонным колебаниям в пределах 500-3000 кг/экв.CO_2/га. ПГП севооборота хлопчатник-пшеница/рис с двухгодичным циклом, который часто практикуется в регионе, составляет в среднем 2500 кг/экв.CO_2/га в год.

Размеры эмиссии N_2, N_2O, NO из орошаемых хлопковых полей определены новым методом: использование потока атмосферного газа He/O_2,

техника почвенного монолита и инкубации. Азотное удобрение в нитратной форме и поливная вода подавались в инкубационный сосуд с целью оценки их влияния на размеры газообразных потерь N. Под влиянием почвенных условий, как они естественно формируются после сопутствующего орошения и внесения удобрений, денитрификация была доминирующим процессом, а N_2 - основной конечный продукт денитрификации. Основываясь на результатах лабораторных инкубационных исследований было рассчитано, что размеры эмиссии N_2 с хлопковых полей за вегетационный сезон могут достигать от 24±9 до 175±65 кг N/га, в то время, как эмиссия в форме NO была незначительной (0,1-0,7 кг N/га). Результаты опытов показывают, что при существующей практике ведения сельского хозяйства на орошаемых землях бассейна Аральского моря денитрификация является основным путем потерь N, и значительная часть азотных удобрений непроизводительно теряется путем улетучивания N_2 в атмосферу.

К тому же, экспериментальный дизайн данного исследования позволил раскрыть потенциальные возможности сокращения размеров эмиссии парниковых газов при существующей системе землепользования. Также определено, что существует ряд возможностей для сокращения потенциала глобального потепления данной агроэкосистемы путем оптимизации использования удобрений и оросительной воды, а также внедрения многолетних древесных насаждений взамен однолетних сельскохозяйственных культур, в особенности, на очень низкоплодородных, маргинальных землях.

TABLE OF CONTENTS

ACRONYMS AND ABBREVIATIONS

ANNAMOX	Anaerobic ammonium oxidation
ASB	Aral Sea basin
ATG	Amir Temur Garden farm
ATC	Amir Temur Cum farm
C	Carbon
CDM	Clean Development Mechanisms
CH_4	Methane
CO_2	Carbon dioxide
DNDC	DeNitrification-DeComposition model
DNRA	Dissimilatory nitrate reduction to ammonium
EF	Emission factor
FAO	Food and Agriculture Organization (UN)
GHG	Greenhouse gas
GWP	Global warming potential
HI	High intensity irrigation
LI	Low intensity irrigation
MMO	Methane monooxygenase
N	Nitrogen
N_2	Dinitrogen
N_2O	Nitrous oxide
NH_2OH	Hydroxylamine
NH_3	Ammonia
NH_4^+	Ammonium
NO	Nitric oxide
NO_2^-	Nitrite
NO_3^-	Nitrate
O_2	Oxygen
PP	Poplar plantation
IPCC	Intergovernmental Panel on Climate Change
SOC	Soil organic carbon
SOM	Soil organic matter
TF	Tugai forest
WFPS	Water filled pore space
WHC	Water holding capacity
UNESCO	United Nations Educational, Scientific and Cultural Organization
UNFCC	United Nations Framework Convention on Climate Change
URDU	Urgench State University
ZEF	Center for Development Research

1 GENERAL INTRODUCTION

1.1 Irrigated agricultural systems

Over the last century irrigated areas have seen an unprecedented growth and have helped to increase food security for a growing world population. Worldwide, irrigated land has increased from 50 million ha in 1900 to 277 million ha in 2003, and 70% of freshwater diverted for human purposes goes to agriculture (FAO 2000). Irrigation does not only increase the amount of land under cultivation, but also leads to increased productivity on existing cropland. Irrigating land decreases the uncertainty of the water supply from rainfall and enables many farmers to move from one annual crop to two or three. Moreover, irrigation is commonly used to produce high-yield varieties of crops. In Asia, yields of many crops have increased 100-400% after irrigation (FAO 1996). There is undisputable evidence that irrigated agriculture is instrumental in ensuring the world's food supplies. Globally, 40% of the world's food production is produced on irrigated land, which makes up only 17% of the land being cultivated (FAO 2000), and the role of irrigation is expected to grow significantly. For example, the FAO (2002) predicted that all developing countries will need to expand their irrigated area from 202 million ha in 1999 to 242 million ha in 2030 to meet their food demands.

Despite all the benefits of irrigation in increasing agricultural productivity and improving rural welfare, there have also been many negative impacts and failures of irrigated agriculture. In addition to high water use and low efficiency, the environmental problems are subject of concern. Environmental problems of irrigation include excessive water depletion, water quality reduction, waterlogging, and soil salinization (Cai et al. 2003). Water logging and salinization often occur together; generally, when excessive amounts of irrigation water are applied to the crops without efficient drainage. The ensuing high groundwater table eventually leads to secondary salinization of the soil, as the water evaporates increasing the concentrations of salts remaining in the soil. This is particularly a problem in arid and semiarid regions with high rates of evaporation. Estimates are that 20-50% of irrigated soils worldwide are affected to varying degrees by waterlogging and salinity (Pitman and Lauchli 2004), which severely affects the agricultural production. On a global scale it has been estimated that salinized soils show an average loss in productivity of 40%, accounting for approximately US $ 10 billion of yield losses per year worldwide (Dregne and Chou 1992). In Central Asia yield reductions of 20-30 % for cotton have been reported already at medium salinity levels of the irrigated soils (WARMAP and EC-IFAS 1998).

1.2 Irrigated agriculture in the Aral Sea Basin

One example of the most serious human-induced environmental disasters caused by excessive irrigation is that of the Aral Sea region. The Aral Sea Basin (ASB) is located in Central Asia and covers an area of about 1.9 million km^2 in the former Soviet Republics of Uzbekistan, Kazakhstan, Turkmenistan, Kyrgyzstan and Tajikistan and the northern part of Afghanistan. In the late 1950ies, the Aral Sea ranked as the fourth largest lake in the world. During the Soviet Union period, the region's primary agricultural role was to produce cotton largely for export to other Soviet republics. To achieve this in a region covered to 75% by deserts, the area under irrigation in the ASB was increased from 2.0 to 7.2 million ha between 1925 and 1985. This has dramatically reduced the inflow of the two major rivers that naturally terminated in the Aral Sea, i.e., the Amu Darya and the Syr Darya, and this in turn resulted in a loss of more than 80 % of its volume and 70 % of its surface area over the last decades (Micklin 2007). This desiccation of the Aral Sea and the desertification of its adjacent areas is known worldwide as the "Aral Sea Syndrome" (WBGU 1999).

In Uzbekistan, agriculture is still the key sector of the economy with a share in the GDP of over 30%, whereas over 50% of the labor force is employed in this sector. Cotton cultivation was continued after independence from the Soviet Union. However, as part of the government's policy to achieve national food sufficiency, nowadays also staple crops such as wheat and rice make up for a large share of the agricultural land (Martius and Wehrheim 2008). Cotton, as an exported arable crop, has a high significance for the national budget, and the country ranks as the fifth largest cotton producer in the world (Bremen Cotton Exchange 2007). Current agricultural production systems are characterized by crop rotations of cotton-wheat-rice under heavy inputs of water and fertilizers. Water is delivered via extensive irrigation systems that were created during Soviet times from 1925-1985. High amounts of irrigation water are applied to the fields via rather inefficient surface furrow irrigation, while the water management is poor and efficient drainage lacking. As a result, rising groundwater tables have led to severe problems of waterlogging and salinization. Therefore, land degradation is increasing in many irrigated areas of Uzbekistan.

1.3 Greenhouse gas emissions from agriculture

Warming of the Earth's climate system is unequivocal, as is now evident from observations of increases in global average air and ocean temperatures, widespread melting of snow and ice, and rising global average sea level. Most of the observed increase in global average temperatures since the mid 20th century is very likely due to the observed increase in anthropogenic greenhouse gas (GHG) concentrations (IPCC 2007) via the greenhouse effect. The greenhouse effect is a natural phenomenon in which the emission of radiation by the atmosphere warms the Earth's surface. Without the greenhouse effect, the Earth would be uninhabitable. In its absence, the average surface temperature would be -18 °C and about 33°C cooler than the present average surface temperature of 15°C. The most important GHG are water vapor (H_2O), carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O) and ozone (O_3). Due to human activities, the atmospheric concentration of the trace gases CO_2, CH_4 and N_2O has increased significantly over the last centuries to values by far exceeding the pre-industrial concentrations.

CO_2 is the most important anthropogenic greenhouse gas, and the increase in the atmospheric concentrations results primarily from the use of fossil fuels. However, agriculture also contributes significantly to the releases of CO_2, CH_4 and N_2O to the atmosphere. CO_2 is produced largely from the burning of plant residues and the microbial decomposition of soil organic matter. Despite large annual exchanges of CO_2 between atmosphere and agricultural land, the net exchange rate is estimated to be fairly balanced. However, changes in land-use, e.g. the conversion of natural unmanaged vegetation to agricultural land, can release substantial amounts of CO_2 to the atmosphere by reducing C storage in soil and vegetation (Robertson and Grace 2004). CH_4 is formed when organic materials are decomposed under anaerobic conditions by methanogenic microorganisms, mainly from enteric fermentation by ruminant livestock, from stored livestock waste, and from rice grown under flooded conditions (Mosier et al. 1998). N_2O is released during microbial transformations (nitrification and denitrification) in soils and livestock waste. It occurs whenever excess soil N is available, especially under wet conditions and high temperatures.

Table 1.1: Most important anthropogenic greenhouse gases (IPCC 2007). GWP = global warming potential.

Species	pre-industrial concentration (1750)	actual concentration (2005)	change since 1998	atmospheric lifetime [yr]	GWP (100 years)	share in anthropogenic greenhouse effect [%]
CO_2	~ 280 ppm	379 ± 0.65 ppm	+11 ppm	50-200	1	63
CH_4	~700 ppb	1,774 ± 1.8 ppb	+5 ppb	12 ±3	25	18
N_2O	~275 ppb	319 ± 0.12 ppb	+5 ppb	127	289	6

To be able to compare the different GHG contributions to global warming, the concept of global warming potential (GWP) has been developed. It allows assessing the radiative forcing of different GHG relative to the reference gas, in this case CO_2, over a specific time horizon. Based on a 100-year time frame, the GWP of CH_4 and N_2O are, respectively, 25 and 289 times higher than that of CO_2 (Forster et al. 2007). The net GWP is expressed in kilograms of carbon dioxide equivalents per hectare per day of the respective GHG.

According to estimates of the IPCC (2007), agriculture accounted for an emission of 5.1 to 6.1 Gt CO_2-eq/yr in 2005, corresponding to 10-12 % of total anthropogenic GHG emission. CH_4 and N_2O are the major contributors to agricultural GWP impacts, as the agricultural sector produces about 50 and 60%, respectively, of the total anthropogenic emissions of these gases. When land-use changes involving biomass burning and soil degradation are included in this estimate, the overall emissions account for one-third of the total anthropogenic GHG release. However, these estimates need improvement as the magnitude of gas flux from the agricultural sector still shows large knowledge gaps for several agroecosystem (Johnson et al. 2007).

The high impact of agriculture on the anthropogenic greenhouse effect suggests that the most significant GHG mitigation could be achieved in the agricultural sector. In many cases, these agricultural GHG mitigation options are cost competitive with non-agricultural options in achieving long-term (i.e., 2100) climate objectives and often may have synergy with sustainable development policies and improvement of environmental quality (Smith et al. 2007a). The global technical mitigation potential from agriculture (excluding fossil fuel offsets from biomass) by 2030 is estimated to be 5.5 to 6.0 Gt CO_2-eq/yr, outranging economic potentials, which are estimated to be 1.5-1.6, 2.5-2.7, and 4.0-4.3 GtCO_2-eq/yr at carbon prices of up to 20, 50 and 100 US$/t$CO_2$-eq, respectively (Smith et al. 2007a).

1.4 Greenhouse gas emissions in Uzbekistan

The Republic of Uzbekistan is a signatory of the United Nations Framework Convention on Climate Change (UNFCC) and thus, obliged to submit GHG inventories as part of the National Communications to the UNFCC. Uzbekistan submitted its initial National Communication in 1999 using the 1996 IPCC guidelines (IPCC 1996) and activity data generated by the National Commission of the Republic of Uzbekistan on Climate Change (s. http://unfccc.int/resource/docs/natc/uzbnc1.pdf). According to these estimates total GHG emissions in Uzbekistan reached 163.2 Mt in CO_2-equivalent in 1990, decreasing to 154.2 Mt in 1994 (Figure 1.1), corresponding to 8.0 t CO_2-equivalent per capita in 1990 and in 6.9 t per capita in 1994 (NCRU 1999). This reduction in emissions was attributed to the specific situation of the social and economic development that occurred in Uzbekistan after independence from the Soviet Union. In particular, the total volume of energy supplied to consumers dropped by 6.3% during this period, except for natural gas, which increased 2.7 fold, from 4.2 to 11.3 billion cubic meters.

In 1990, carbon dioxide accounted for the largest proportion of emissions (70.2%), while methane emissions accounted for the second largest (23.1%) and nitrous oxide emissions the smallest (6.7%). In 1994, the proportion of carbon dioxide declined to 66.3%, while the proportion of methane increased slightly (27.1%), and the proportion attributed to nitrous oxide remained relatively constant (5.6%). The major source of greenhouse gases in Uzbekistan is the power sector, which accounted for about 83% of the overall emissions in 1990 and 1994, followed by agriculture (11%), industrial production (4%) and the waste sector (2%) (Figure 1.2). In terms of GHG sinks, the annual sequestration of carbon by forests was estimated at 0.4 Mt CO_2 in 1990 and 1994.

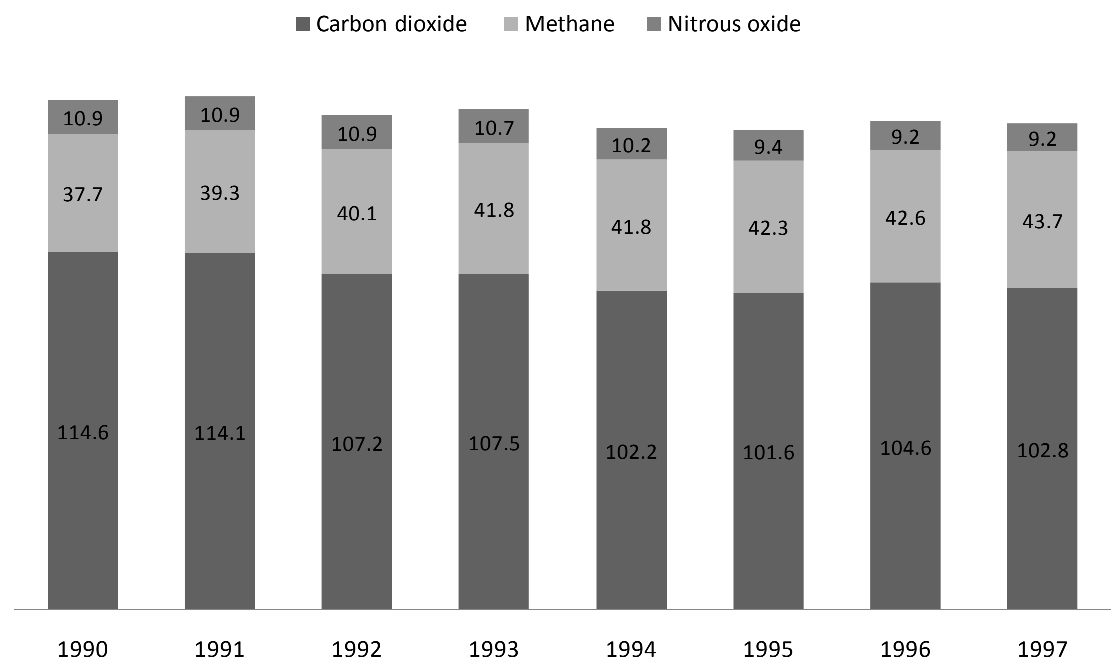

Figure 1.1: Greenhouse gas emissions in Uzbekistan (Megatons CO_2-equivalents) during the 1990-1997 period (NCRU 1999).

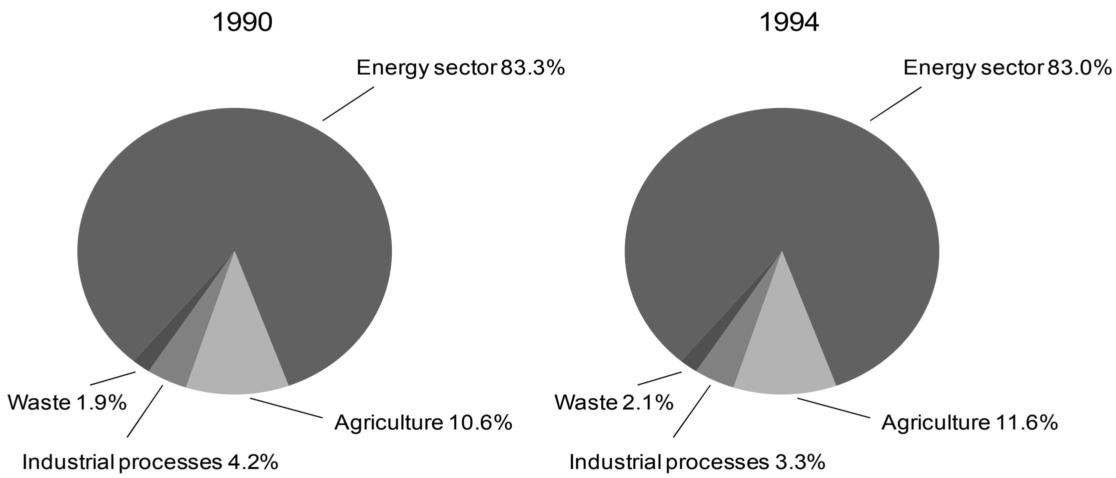

Figure 1.2: Sources of greenhouse gas emissions in 1990 and 1994 (NCRU 1999).

The major source of methane emissions is the power sector (oil and gas industry), which accounted for 73.5% of the overall volume of methane emissions, followed by agriculture (8.3%) and waste (8.2%). From 1990 to 1994 an increase of methane emissions of 10.7% was reported, which was attributed as the result of an increase in natural gas production and consumption and an increase in rice production (NCRU 1999).

The principal sources of nitrous oxide are agricultural soils, which accounted for 96.8% of the overall nitrous oxide emissions in 1994. Manure (2.5%) and chemical substances (0.5%) account for relatively minor fraction of nitrogen oxide emissions. Total nitrous oxide emissions declined only by 6.5% between 1990 and 1994, with reductions in the agricultural sector accounting for 3.8% of this decrease. Nitrous oxide emissions in industrial processes reduced 5-fold, due to a sharp drop in the production of nitric acid.

2 PROBLEM STATEMENT AND RESEARCH APPROACH

With 7.2 million ha of irrigated agriculture, land-use decisions in the five central Asian countries have a high impact on the anthropogenic greenhouse effect, and the potential for GHG mitigation is extremely important. In particular, in this region the trace gas emissions from irrigated agriculture are postulated as considerable, but are yet unknown. Therefore, this study aimed at providing insights into soil microbial processes and trace gas emissions from this important agroecosystem. The study was carried out within the framework of a development project conducted by the German Center for Development Research (ZEF). This project has been developed by ZEF since the year 2000 in close cooperation with its partners UNESCO, German Aerospace Centre (DLR), and the State University Urgench in Uzbekistan. It started in 2001, and has operated with funds of the German Ministry of Education and Research (BMBF) in the context of the Aral Sea crisis to provide sound, science-based policy recommendations for sustainably improving the natural resource use in the Khorezm region, Uzbekistan (Martius et al. 2006). For more information and a detailed description of the project, please refer to http://www.khorezm.uni-bonn.de/index.html.

2.1 Research objectives

The overall goal of this study was to investigate biosphere-atmosphere exchange of radiatively active trace gases and microbial transformation processes in soils, and to identify sustainable land-use strategies that reconcile low trace gas emissions with high fertilizer and water use efficiencies of the irrigated agricultural systems in the ASB.

The specific research objectives were to:

(i) Assess the impact of the dominating cropping systems in the irrigated areas of ASB - cotton, winter wheat and rice - on the emissions of N_2O and CH_4;

(ii) Compare these to fluxes of N_2O and CH_4 from perennial cropping systems, especially forest plantations and natural floodplain forests;

(iii) Evaluate the global warming potential (GWP) of N_2O and CH_4 fluxes from the entire irrigated dryland agriculture in the study region;

(iv) Identify the main processes and site-specific regulating parameters for GHG emissions from irrigated agriculture in the ASB;

(v) Assess the potential of management and irrigation practice for mitigating GHG emissions from irrigated agriculture in the ASB;

(vi) Quantify aggregated gaseous N losses composed of NO, N_2O, and N_2 from differently managed (water regime/ fertilizer management) agricultural systems throughout the vegetation cycle.

2.2 Thesis outline

The thesis consists of eight different chapters. The general introduction (Chapter 1) is followed by this outline of the problem approach and the research objectives (Chapter 2). Chapter 3 describes the key issues of soil biogeochemical cycles in irrigated agriculture. A characterization of the study region (Chapter 4) provides an insight into the geographical, agro-climatic and other main characteristics of the Khorezm Region and puts the results and conclusions of the subsequent chapters into a regional context. Chapters 5 to 7 are separate publications, each with a brief introduction into the topic to be dealt with and the discussion of the results. Chapter 5 presents the results of a two-year field campaign on N_2O emissions from irrigated cotton fields and the influence of irrigation and fertilization practices. Chapter 6 reports on CH_4 and N_2O fluxes in different annual and perennial land-use systems, evaluates the GWP and discusses options for mitigating GHG emissions of these irrigated agricultural systems. Chapter 7 presents and discusses the findings of a laboratory incubation study on the aggregated gaseous N losses consisting of NO, N_2O, and N_2 from the irrigated cotton fields. Chapter 8 summarizes the main findings of this research, provides overall conclusions and presents recommendations for further research.

3 BIOGEOCHEMICAL CYCLES IN IRRIGATED AGRICULTURE

3.1 Nitrogen turnover in soils and its impact on atmospheric trace gases

Trace gases are those that make up less than 1% of the earth's atmosphere. Soils contribute to the budgets of many atmospheric trace gases by acting as sources or sinks. The most important nitrogen trace gases include N_2O and NO. The principal forms of N in the soil are organic N compounds and the mineral N compounds ammonium (NH_4^+) and nitrate (NO_3^-). Most of the N in a surface soil is present as organic N; only a small proportion, in most cases less than 5%, is directly available to plants, as soluble forms of N. The change from one nitrogen compound to another depends primarily to the environmental conditions and is mainly subject to different microbial transformation processes (Figure 3.1).

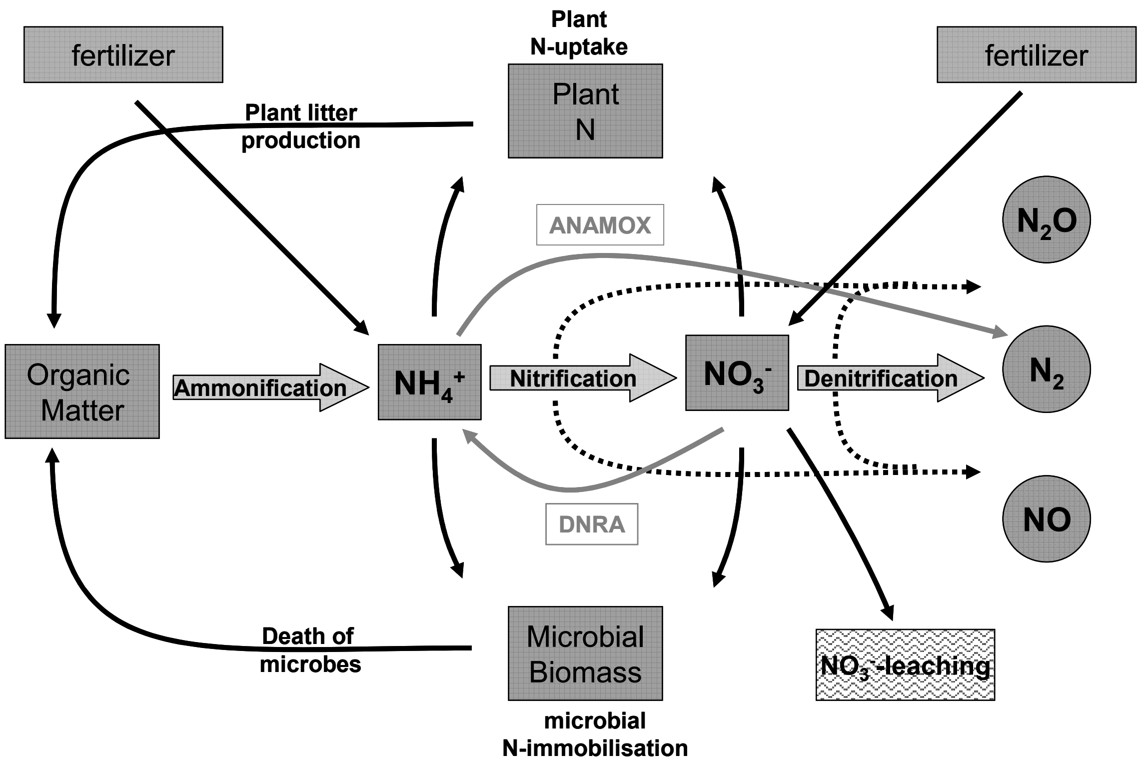

Figure 3.1: Schematic representation of the most important processes of the soil N cycle displaying the N trace gas exchange from nitrification and denitrification (dashed line). (Modified after Breuer 1999).

3.1.1 Mineralization (ammonification) and immobilization

The dead organic material of plants, animals, and microorganisms contains large concentrations of organically bound nitrogen in various forms, such as proteins and amino acids. Mineralization is a key process in the nutrient cycle of ecosystems and refers to the decomposition of organic material into soluble, plant accessible compounds (Figure 3.1). It is carried out by a great diversity of bacterial and fungal decomposers. A particular aspect of the more complex process of mineralization is called ammonification and denotes the microbial conversion of organic nitrogen back into ammonia (NH_3) or ammonium (NH_4^+). Immobilization stands for the consumption of soluble N by the microbes resulting in retention of plant available N (Figure 3.1). The rate between N mineralization and immobilization is primarily controlled by the quality of the soil organic material (SOM), especially by the availability of C relative to the available N (C:N ratio) (Robertson and Groffman 2007). When the C:N ratio of the SOM is relatively low, the microbes have no trouble obtaining N from the substrate, and as C is consumed, plant available N increases in the soil. On the other hand, when the C:N ratio of the SOM is high, N is retained by the microbes as C is consumed, resulting in a decrease in plant available N in the soil. Mineralization and immobilization occur at the same time in the soil, and it is important to make a distinction between gross and net mineralization and immobilization. Gross mineralization refers to the total amount of soluble N produced, while immobilization is the amount of soluble N taken up by microorganisms. Net mineralization is the difference between the two.

3.1.2 Nitrification, denitrification and related processes

The two most important processes involved in the production of NO and N_2O are nitrification and denitrification (Davidson et al. 1986). A summary of the key issues of these processes within the terrestrial N cycle is given below.

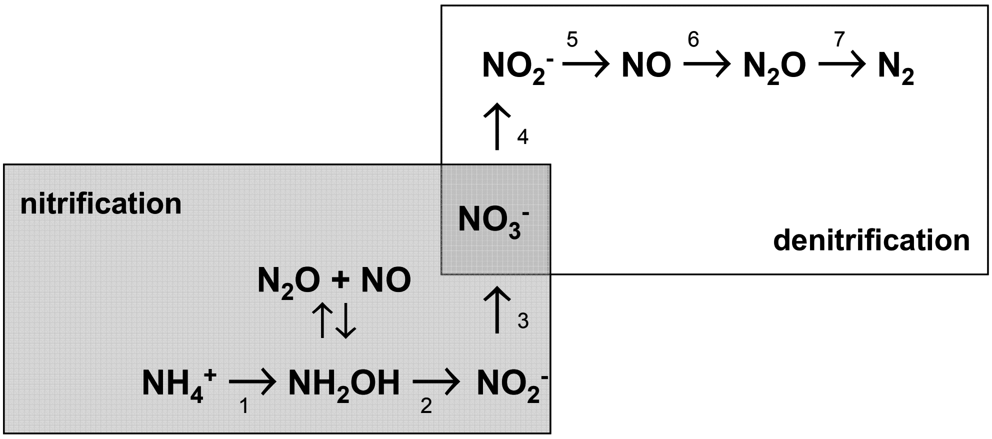

Figure 3.2: Nitrification and interaction with denitrification. The numbers indicate enzyme reactions, i.e., 1: ammonium monooxygenase; 2: hydroxylamine oxidoreductase; 3: nitrite oxidoreductase; 4: nitrate reductase; 5: nitrite reductase; 6: NO reductase; 7: N_2O reductase (Modified after Conrad 2001; Hofman and Van Cleemput 2004).

Autotrophic nitrification

Nitrification is a two-step process performed by different groups of micro-organisms. In the first step, the so called ammonium (NH_4^+)-oxidizing nitrifiers convert NH_4^+ via hydroxylamine (NH_2OH) into nitrite (NO_2^-), using as enzymes the ammonium monooxygenase and the hydroxylamine oxidoreductase, respectively. In the second step, NO_2^--oxidizing nitrifiers oxidize NO_2^- to NO_3^- with the nitrite oxidoreductase (Figure 3.2). The release of NO and N_2O may be a by-product of the nitrification process resulting from incomplete oxidation of intermediates, such as NH_2OH and NO_2^-. Nitrification is an aerobic process that requires O_2. Consequently, the soil moisture content has a great influence on the nitrification rate, since soil water reduces the diffusion of air into the soil. At a water-filled pore space (WFPS) above 80%, the oxygen content in the soil is low and nitrification ceases, whereas highest nitrification activity is expected at 30–60% WFPS (Firestone and Davidson 1989a). Nitrification is also slow under acid conditions with an increasing rate as pH rises. A number of different types of nitrifying bacteria have been identified. The oxidation of ammonia is mainly attributed to the genera *Nitrosomonas* and *Nitrosospira*, while the oxidation of nitrite is performed by *Nitrobacter* and *Nitrospira* species (Jetten 2001). So far, predominately obligate autotrophic bacteria have been considered to be the most important contributors to aerobic ammonia oxidation, but recent studies show that

Archaea may predominate among ammonia-oxidizing prokaryotes in soils (Leininger et al. 2006).

Heterotrophic nitrification

Heterotrophic microorganisms are also known to carry out nitrification, using organic carbon (C) as source of C and energy (Robertson and Kuenen 1990). Among those heterotrophic nitrifiers, fungi are considered to be more common than bacteria, but some heterotrophic bacteria can also nitrify (Papen et al. 1989). Although heterotrophic nitrification is thought to happen at much lower rates than accomplished by the autotrophic bacteria, it might produce significant amounts of N_2O under a set of circumstances, such as low pH, high oxygen amount and availability of organic material (Papen et al. 1989).

Denitrification

Under anaerobic conditions, NO_3^- produced via nitrification can be reduced to N oxides (NO, N_2O) and molecular N (N_2) by microorganisms (denitrification). Denitrification refers to the stepwise reduction of NO_3^--N to these compounds (Figure 3.1 and Figure 3.2). These gaseous products are not available for plant uptake. Denitrification is primarily carried out by heterotrophic, facultative anaerobic microorganisms that are able to use NO_3^- as electron acceptor in order to cope with low-oxygen or anaerobic conditions. Enzymes catalyzing denitrification are NO_3^- reductase, NO_2^- reductase, NO reductase and N_2O reductase. Basically, the production of N_2O is therefore controlled by the relative activity of the enzymes, NO reductase and N_2O reductase, the production of NO by the relative activity of the enzymes NO_2^- reductase and NO reductase, irrespective of which type of microorganisms they occur in (Conrad 2001). Denitrifiers are widely distributed across the bacterial taxa, including *Psudomonas*, *Bacillus*, *Thiobacillus* and *Alcaligenes*, some of which are also capable of heterotrophic nitrification under aerobic conditions. In addition, several fungi were shown to have denitrifying abilities (Kobayashi et al. 1996), but many fungi lack N_2O reductase; therefore, N_2O is the dominant product of fungal denitrification (Shoun et al. 1992). Accordingly, fungal denitrification can be the major source of N_2O emissions in some ecosystems (Laughlin and Stevens 2002).

In contrast to nitrification, during denitrification NO and N_2O are produced as regular intermediates. The regulation of the denitrification rate and the release of NO and N_2O from soils depend on several parameters like oxygen, moisture level, NO_3^- content, C supply, temperature, pH, soil texture, etc. The quantity and quality of incorporated C (harvest residues, organic manure and waste material), weather conditions (drying/wetting, freezing/thawing) and management practices (physical disturbance, soil compaction, drainage, irrigation) are especially important. One of the most important regulating parameters is the soil water content. Under conditions of high soil moisture levels, the diffusion of O_2 into the soil will decrease, and bacteria capable of denitrification may use nitrate as an alternative electron acceptor (Firestone and Davidson 1989b). Consequently, denitrification levels have been reported to increase with increasing soil moisture (Scholefield et al. 1997), and highest denitrification rates were found at a WFPS above 60% (Davidson 1991; Bouwman 1998). However, high rates of denitrification can also occur in predominantly aerobic soils where denitrification takes place in anaerobic microsites, so-called hotspots (Parkin 1987; McClain et al. 2003).

Chemodenitrification

In soils with low pH, chemical decomposition of NO_2^- to NO and N_2O can lead to the formation of NO and N_2O. This non-biological process, usually described by the term chemodenitrification, refers to the same reduction pattern and end products, but it is not carried out by microorganisms. In acid soils the major product of these reactions is NO, and chemodenitrification can be a major source of NO (Davidson 1992), whereas N_2O release from chemodenitrification is negligible (Bremner 1997).

Nitrifier denitrification and aerobic nitrification

Nitrification and denitrification have long been regarded as separate phenomena performed by different groups of bacteria under different soil conditions. It has now been established that strict segregation in place and time of the two processes is not necessary and that both denitrifiers and nitrifiers have versatile metabolisms (Jetten 2001). For example, autotrophic nitrifiers were found to be able to denitrify as well. In this so-called nitrifier denitrification, NH_4^+ is oxidized to NO_2^- followed by the stepwise reduction of NO_2^- to NO, N_2O and N_2. The rates of nitrifier denitrification are typically quite low but may be an important source of N_2O under certain circumstances, i.e., high N content, low organic C content, low O_2 pressure and maybe also low pH (Wrage et al. 2001). Moreover, some heterotrophic nitrifiers have been reported to be capable of aerobic denitrification by simultaneously using oxygen and nitrate as electron acceptors (Jetten 2001). However, the rates described for aerobic denitrifiers are very low compared to the rates observed under anoxic conditions.

Dissimilatory nitrate reduction to ammonium (DNRA)

A different pathway for NO_3^- reduction in soils is the dissimilatory nitrate reduction to ammonium (DNRA) (Figure 3.1). DNRA is carried out by fermentative bacteria that are capable of reducing NO_3^- through NO_2^- to N_2O or to NH_4^+; but these bacteria are not able to reduce N_2O to N_2. It is an anaerobic process that can proceed at the same time as denitrification and may be a fast and significant process in N transformations in C-rich soils (Yin et al. 2002). DNRA might also be an important source of N_2O, especially in agricultural soils with pH > 6.5 (Stevens et al. 1998).

Anaerobic ammonium oxidation (ANAMMOX)

Another recently discovered process is the anaerobic ammonium oxidation (ANAMMOX) (Jetten et al. 1998), in which ammonium serves as the electron donor for denitrification of nitrite into dinitrogen gas (Figure 3.1). The ANAMMOX process is catalyzed by a specialized group of planctomycete-like bacteria, which use a complex reaction mechanism involving hydrazine as an intermediate (Op den Camp et al. 2006). ANAMMOX was shown to be very important particularly in the suboxic conditions of marine sediments. Until the discovery of ANAMMOX, denitrification was thought to be the only substantial sink of reactive nitrogen. It now seems clear that ANAMMOX is a second important sink, and it is estimated to contribute up to 50% of oceanic nitrogen loss (Brandes et al. 2007). Some of the reported high nitrogen losses in soil might also be attributed to anaerobic ammonium oxidation.

3.1.3 Abiotic N losses

Ammonia (NH_3) volatilization

Ammonium N (NH_4^+) in the soil is either formed by mineralization of soil organic N and applied inorganic N or after hydrolysis of urea. In the soil, NH_4^+ is in physico-chemical equilibrium with ammonia gas ($NH_3 + H_2O \leftrightarrow NH_4^+ + OH^-$), and much nitrogen may be lost to the atmosphere in this form. NH_3 volatilization will be more pronounced in soils with high pH, but it depends on several other factors such as soil moisture, soil temperature, soil composition, soil texture and structure, weather conditions, etc. The dominant NH_3 source is animal manure and about 30% of N in urine and dung can be lost as NH_3. The other major source is surface application of urea or ammonium bicarbonate and, to a lesser degree, other NH_4^+ containing fertilizers (Hofman and Van Cleemput 2004). Usually, NH_3 is deposited rapidly within the first 4-5 km from its source, and NH_3 can contribute significantly to high total N deposition rates that can have negative effects on vulnerable terrestrial ecosystems (Krupa 2003).

Nitrate leaching

The negatively charged NO_3^- ions are, in contrast to the positively charged NH_4^+ ions, not absorbed by the negatively charged colloids that dominate most soils. Therefore, NO_3^- can move downward freely with drainage water, and can thus be leached from the rooting zone. The amount and intensity of rainfall, quantity and frequency of irrigation, evaporation rate, temperature, soil texture and structure, type of land-use, cropping and tillage practices and the amount and form of fertilizer N are all parameters influencing the amount of NO_3^- movement to the groundwater and surface waters (Hofman and Van Cleemput 2004). Such leaching does not only represent a loss of N from the ecosystem, but also can cause serious environmental problems. High levels of nitrate in groundwater are a significant health hazard and may cause degradation of aquatic ecosystems due to eutrophication (Jenkinson 2001).

3.2 Carbon turnover in soils and its impact on atmospheric trace gases

Soils are the third largest carbon (C) pool within the global carbon cycle, comprising two different components: soil organic carbon (SOC) and soil inorganic carbon (SIC). In fact, more carbon is stored in the soil than in the world's vegetation and atmosphere combined (Lal 2004b). The rate at which the soil C pool either increases or decreases is determined by the balance between the input and output of C into the soil. C input mainly refers to plant and animal residues at various stages of decomposition and of microbial by-products that are transformed into soil humus via the process of humification. C output occurs largely via oxidation of SOC into CO_2 by microorganisms (soil respiration), although CH_4 efflux and hydrologic leaching of dissolved and particulate carbon compounds can also be important. Enhancing the SOC pool can be an important strategy of abiotic C sequestration; practices that are known to increase C sequestration include afforestation and reforestation, conservation tillage, plant residue management and integrated nutrient management (Lal 2004a). At the same time, enhancing the SOC content can have beneficial influences on soil properties, site-water relations, and nutrient cycles in many soils. On the other hand, soils can be a major source of radiatively active C trace gases, as SOC is depleted as a result of land-use change and the release of CH_4 from natural wetlands and lowland rice fields. The main soil-microbial processes that are responsible for the release of CO_2 and the consumption/production of CH_4 are soil respiration, CH_4 production and CH_4 oxidation.

3.2.1 Soil respiration

CO_2 produced at the soil surface can result from autotrophic respiration (rhizosphere and mycorrhizal respiration) and heterotrophic respiration, e.g., respiration from microbial communities that use SOM as an energy substrate (Ryan and Law 2005). Soil respiration is mainly controlled by the supply and quality of SOM, but other environmental factors, such as soil moisture, oxygen supply, and soil temperature are also important. The annual global flux of CO_2 from soils to the atmosphere is estimated at 70-80 Gt C, which exceeds the emissions from fossil fuels (5.5 Gt C) by a factor of 12-15 (Raich and Potter 1995; Schlesinger and Andrews 2000). Thus, small changes in the magnitude of soil respiration could have a strong effect on the concentration of CO_2 in the atmosphere; if soils will be a net source or sink for carbon under changing climate is of major concern. Whether the effect of climate change will result in a warming-induced acceleration of soil C decomposition or increased carbon input to soils because of enhanced net primary production is not clear yet (e.g., Liski et al. 1999; Giardina and Ryan 2000; Knorr et al. 2005; Davidson and Janssens 2006). Such positive or negative feedback loops in the carbon cycle could significantly accelerate or slow down climate change over the 21st century, and it will be essential to accurately represent such feedbacks in order to successfully predict climate change over the next 100 years (Cox et al. 2000).

3.2.2 Methane oxidation and production

The net exchange of CH_4 between soils and the atmosphere is the result of (i) production of CH_4 through degradation of organic matter by methanogenic bacteria and (ii) the simultaneous consumption of CH_4 through oxidation by methanotrophic bacteria (Butterbach-Bahl 2002). CH_4 production occurs only under highly anaerobic conditions which typically occur in natural wetlands and lowland rice fields. The major pathways of CH_4 production in flooded soils is the reduction of CO_2 with H_2, with fatty acids or alcohols as the hydrogen donor, and the transmethylation of acetic acids or methyl alcohols by CH_4 producing bacteria (Conrad 1989; Mosier et al. 2004). Anaerobic soils are a major source for atmospheric CH_4, and natural wetlands and rice agriculture are estimated to comprise 35-45% of the total global source strength of CH_4 (Mikaloff Fletcher et al. 2004; Wang et al. 2004; Chen and Prinn 2006).

Consumption of CH_4 occurs in aerobic soil conditions where methanotrophic bacteria oxidize CH_4 using the enzyme methane monooxygenase (MMO), which requires both O_2 and reducing equivalents for activity (Conrad 1996). It is estimated that the oxidation of CH_4 in aerobic soils accounts for 5-7% of the global CH_4 sinks (Mikaloff Fletcher et al. 2004; Wang et al. 2004). CH_4 oxidation occurs primarily in the upper layer of the mineral soil, and the magnitude and rate of oxidation are controlled by soil type, aeration, environmental parameters and N availability (Topp and Pattey 1997; Le Mer and Roger 2001). Application of fertilizer N has been shown to inhibit CH_4 oxidation in soil due to competition between NH_3 and CH_4 for MMO enzymes (Steudler et al. 1989; Hutsch 1998).

4 STUDY REGION

4.1 Geographical and demographical setting

The study was conducted in Khorezm, a region situated in the so-called Turan Lowland of the Aral Sea Basin (ASB), a vast low-lying desert basin region stretching from southern Turkmenistan through Uzbekistan to Kazakhstan. Khorezm is located in the northwest of Uzbekistan in the Amu Darya delta, about 350 km south of the remains of the Aral Sea (Figure 4.1).

The region is situated between 41°08′ and 41°59′ N latitude and 60°03′ and 61°24′ E longitude at 90-138 m above sea level. It is bordered by the Kara Kum and Kyzyl Kum deserts to the south and east, the Amu Darya River to the northeast, the Autonomous Republic of Karakalpakstan to the north and the Republic of Turkmenistan to the southwest. Khorezm is one of the two administrative districts in Uzbekistan that belong to the lower reaches of the Amu Darya and is divided into ten administrative sub-units with the capital city of Urgench, which was inhabited by 137,600 people in 2003. Khorezm province is one of the areas most intensively used for agriculture. It covers an area of 6800 km^2 of which roughly 270,000 – 300,000 ha are used for irrigated agriculture (Conrad 2006). The population numbered 1.4 million in 2003, with approx. 80% of the population living in rural areas (OBLSTAT[1], 2005), with incomes largely depending on irrigated agriculture. Cotton production is still the main sector of the economy, but nowadays also staple crops such as wheat and rice make up for a large share of the agricultural land in Khorezm. All irrigation water originates from the Amu Darya and is key for agriculture in Khorezm. In view of its downstream location on the Amu Darya, Khorezm is especially vulnerable to water shortage and droughts. Moreover, due to population growth and the extension of the irrigated area, the probability of receiving an adequate supply of water has decreased considerably (Müller 2006).

[1] *OblStat* is the local Branch of Uzbekistan's Statistical Office in Khorezm region.

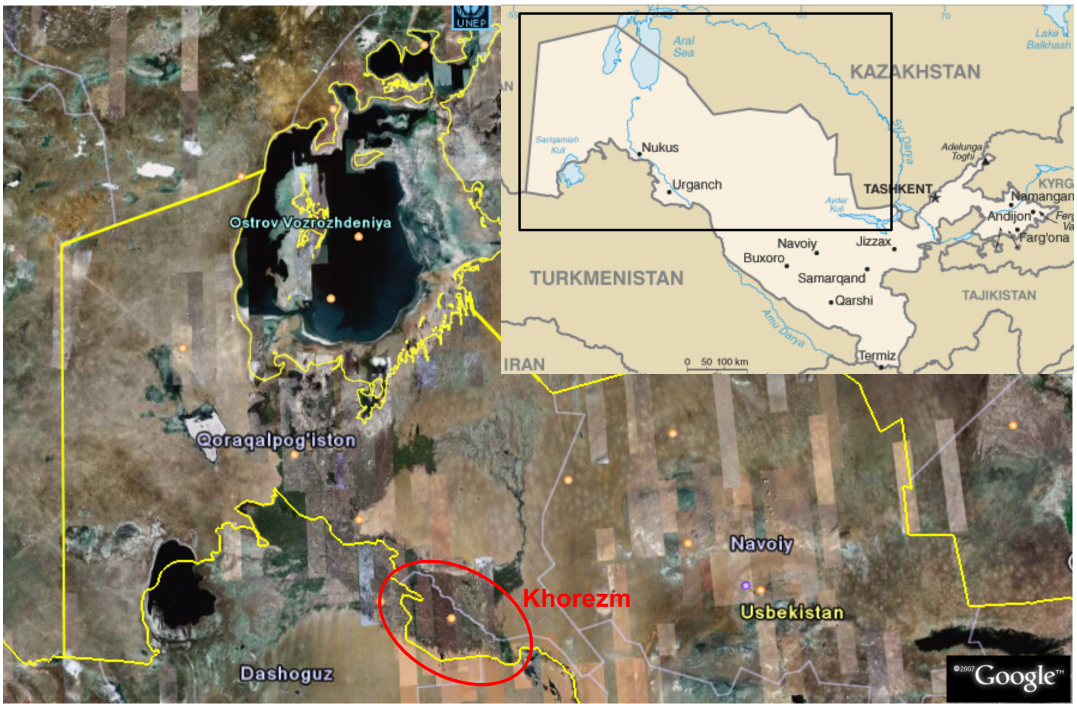

Figure 4.1: Location of the study region (Khorezm Province) in the north-west of Uzbekistan.

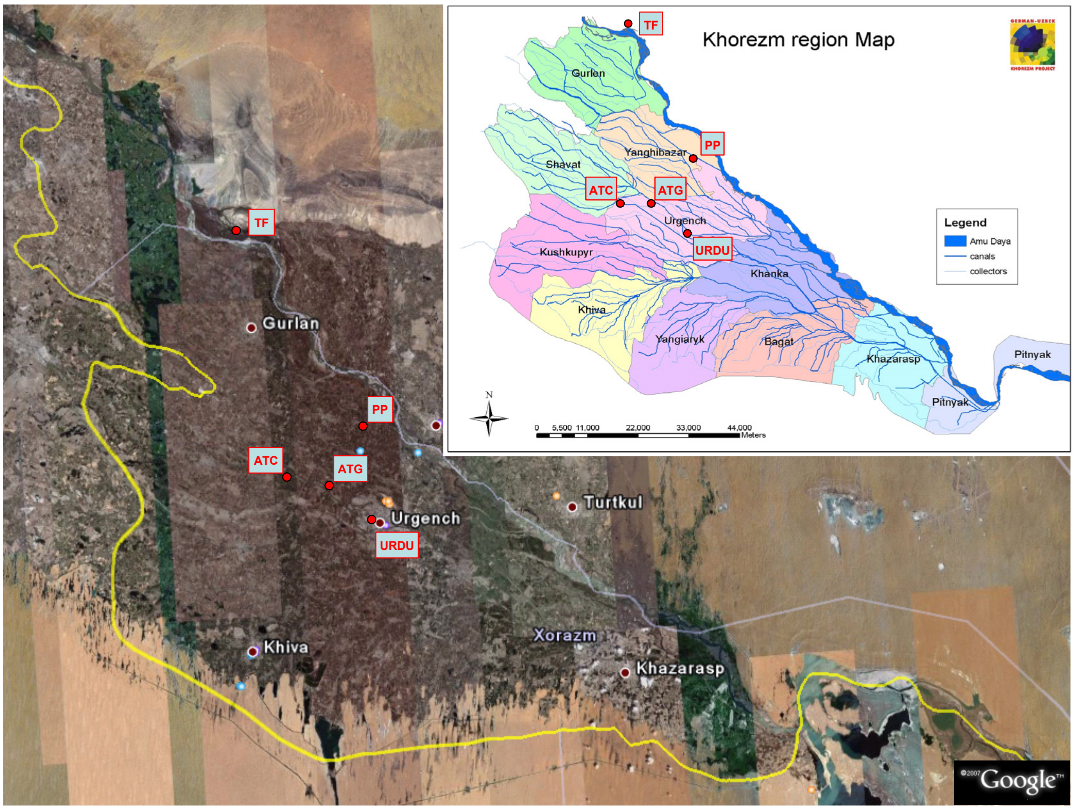

Figure 4.2: Khorezm Province and location of the five research sites. (Amir Temur Garden Farm (ATG), Amir Temur Cum Farm (ATC), Urgench State University (URDU), Poplar Plantation (PP), Tugai Forest Reserve (TF))

In 2000 and 2001, agricultural producers in Uzbekistan were severely affected by the worst drought of the last two decades. In addition, the extensive irrigation in Khorezm has drastically increased salinization and degradation of the soils, affecting the ecological and socio-economic situation of the region substantially.

The field experiments were conducted at five experimental sites of the ZEF/UNESCO project in Khorezm's Urgench and Yangibazar districts and in the Amu Darya floodplain north of Khorezm (Figure 4.2). Cropping systems of the region's most dominant soil textures were selected for the flux measurements. Two study sites were part of the Amir Temur Shirkat[2] close to the research station. Cotton and winter wheat flux rates were measured on experimental fields of the Amir Temur Garden (ATG) farm (41º36'N, 60º31'E); cotton and rice flux rates were measured on experimental fields of the Amir Temur Cum (ATC) farm (41º38'N, 60º25'E). The third site was situated on the campus of the Urgench State University (URDU) (41º33'N, 60º37'E). Measurements in perennial land-use systems were conducted in a plantation of poplar trees (PP) (41º43'N, 60º35'E) in the Yangibazar district and in a natural Tugai forest reserve (TF) located in the Amu Darya floodplain north of Khorezm (41º58'N, 60º24'E), which represents the native riparian vegetation along the banks of the Amu Darya River. More details of the different research sites and the experimental set up are shown in Table 6.1 and Table 6.2.

4.2 Climate

Climatic conditions in Khorezm region can be described according to Koeppen's Climate Classification System as dry arid desert climate (BWk), where precipitation is greatly exceeded by the local evapotranspiration potential of about 1,600 mm year^{-1}. It is a typically arid continental climate with long, hot, dry summers, erratic rains in winter-spring and very cold temperatures during the winter. According to data from the Main Administration on Hydrometeorology (Glavgidromet) of the Republic of Uzbekistan, average precipitation at the Urgench Meteorological Station for the period 1980-2000 was 97 mm and the mean annual temperature was 13.0ºC (Glavigdromet 2003). During the observation years 2005/2006, the average precipitation was 74 mm

[2] *Shirkat* = a collective farm established from a Soviet kolkhoz or sovkhoz farm after independence

and mean annual temperature was 13.3ºC. The coldest month was January with an average temperature of -1.8ºC, the hottest month July with an average temperature of +28.7º C. Mean daily temperatures vary from -25°C in winter to +45°C in summer. A characteristic of the region is the north-easterly wind during the vegetation season (from April until October) with an average wind velocity of 1.4 to 5.5 ms^{-1} with maximum velocities reaching 7-10 ms^{-1} (Forkutsa 2006). Radiation is high throughout the year, with actual sunshine hours ranging from 2700 to 3000 per year (Meteo-infospace 2004).

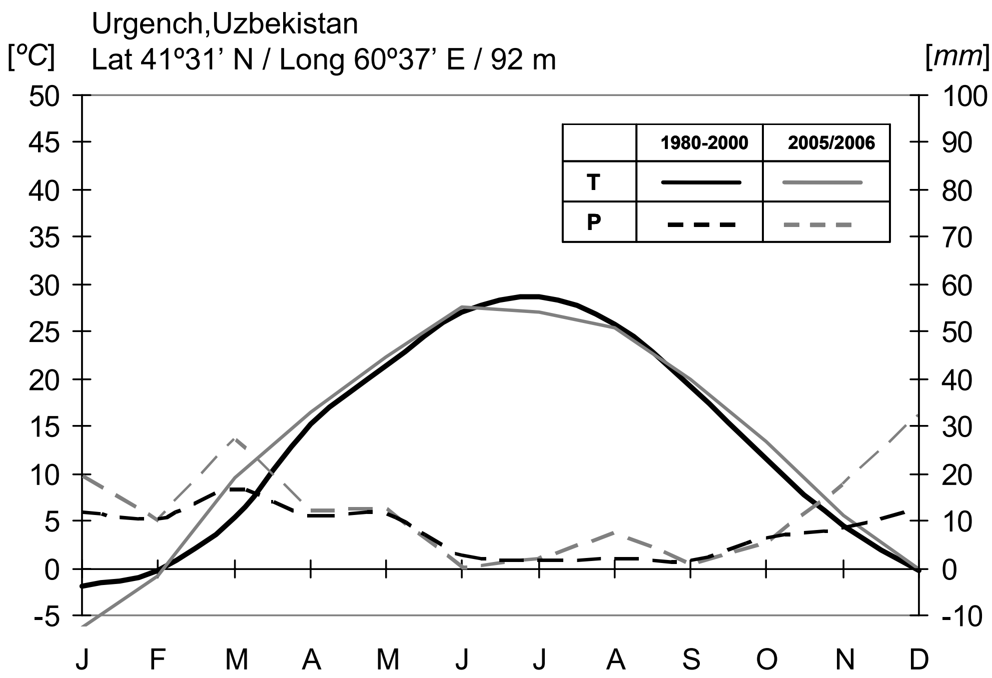

Figure 4.3: Mean monthly air temperature (solid line) and precipitation (dashed line) for 1980-2000 (black) and the observation years 2005/2006 (grey) of the Urgench meteorological station, Uzbekistan (Glavgidromet, 2003)

Due to the desiccation of the Aral Sea, which acted as a buffer protecting the region from the fierce Siberian winter winds and harsh summer temperatures, a change in the climatic conditions over the last decades has been observed. As a tendency, the climate appears to have become more continental, with shorter, hotter, rainless summers and longer, colder, almost snowless winters. In some areas, the growing season has been reduced to an average of 170 days, falling short of the 200 frost-free days needed to grow cotton with the varieties developed (Sdasyuk and Mamayeva 1995; Vinogradov and Langford 2001). However, the differences are not so pronounced for the comparison of the periods 1980-2000 and the study period 2005/2006 (Figure 4.3).

4.3 Relief, geomorphology and soils

The area of Khorezm is mostly flat showing a difference in elevation from east to west of about 40 m over 100 km. It belongs to the so-called "ancient river delta" and its lithological profile was mainly formed by the meandering Amu Darya River and its powerful ancient channels Dar'alik and Daudan that carried and deposited sediments along the riverbanks and in depressions. Coarse-textured particles were deposited along the river banks, while depressions were mainly filled with loam and clay. After flooding periods, the temporary streams changed into still lake water, resulting in lacustrine sediments with mostly heavy clay texture. Accordingly, the soils originated in these areas reveal a highly stratified non-homogeneous structure, with the area that currently is used for agriculture dominated by clayey, loamy and sandy-loamy textures (Nurmanov 1966).

According to the FAO classification, three major soil types can be identified within Khorezm (FAO 2003): (i) yermic regosols, soils that were formed of alluvial rock debris deposits outside the irrigated areas and of dunes of the Kara Kum desert mainly in the south of Khorezm; (ii) calcaric fluvisoils, meadow soils commonly found along the Amu Darya River in the eastern part of Khorezm; (iii) calcaric gleysoils, meadow soils in the irrigated areas characterized by a shallow groundwater table often with elevated groundwater salinity and secondary salinization in the upper soil.

However, the soil maps of the FAO classification are rather wide-ranging and do not capture the more detailed characteristics of the Uzbek classification. On the basis of the Uzbek classification, the major soil type found in the region is the so-called irrigated alluvial meadow soil, which covers 61% of the area (Rasulov 1989; ZEF 2006). Other dominant soils are "boggy-meadow" (covering 16% of the area), "takyr-meadow" (15%), "boggy" (5%), "grey-brown" and "takyr" (2%) soils (Sabirov 1980; Rasulov 1989). According to the Uzbek classification soil texture of the Khorezmian soils has recently been described as light, medium and heavy loams (Rizayev 2004). A comparison shows that an Uzbek medium loam corresponds to a silt loam and loam, while the local heavy loam matches best the silt clay loam (WARMAP and EC-IFAS

1998). According to Rizayev (2004), the prevailing soil particle sizes in most parts of the region are light and medium loams. For a more detailed description of the soils in Khorezm refer to Kienzler (forthcoming).

The natural fertility of the soils in Khorezm is considered as rather low. The supply of plant available N, P and K in the top 30 cm can be characterized as moderate to low (WARMAP and EC-IFAS 1998; ZEF 2006). Organic matter in these soils ranges from 10.5 to 80.5t/ha, whiles the total nitrogen (N) content varies between 1.1 and 4.9 t/ha. Total phosphorus (P) and potassium (K) contents in Khorezm soil are typically ranging between 3.3 - 12.9 t/ha and 19.8 - 168.7 t/ha, respectively (Ibragimov, Soil Science Institute Tashkent, pers. comm.). As a consequence high inputs of chemical fertilizers are used for the cultivation of many agricultural crops.

Caused by irrigation during the growing period, the groundwater tables are generally shallow (1.2-1.5 m below ground surface) and the average salinity of the groundwater ranges between 1.68 g/l in October and 1.81 g/l in April (Ibrakhimov 2004). As a result, secondary soil salinization has become a major problem in the irrigated areas of Khorezm; according to official government data (1999-2001), the entire irrigated area in the Khorezm region shows secondary salinization problems, and 81% of the area has problems with waterlogging (Abdullaev 2003). To flush salts from the soil, huge amounts of irrigation water are applied to the fields in spring, prior to crop planting. This, in turn, causes the groundwater table to rise, as an efficient drainage system is not in place in most of the areas, which increases the risk of re-salinization in the root zone.

4.4 Irrigation and drainage network

The extensive network of irrigation channels and the complementary drainage collectors in Khorezm were mainly built during the Soviet era from 1950 to 1970 (Katz 1976). Water is diverted from the Amu Darya River and supplied to the agricultural fields through a complex, hierarchical irrigation network consisting of main, inter-farm and on-farm canals. The total length of the network is 16,233 km and every year between 3.5 km^3–5km^3 of water from the Amu Darya is supplied to the region and used mainly for agricultural purposes (SIC-ICWC 2006), whereas in individual years withdrawals of 5.38 km^3 irrigation water in the vegetation period 2005 were recorded (Conrad 2006).

The predominant irrigation technique in Uzbekistan is surface irrigation, which includes 64% furrow, 31% strip and 5% basin irrigation (Abdullaev 2002). These days, however, due to the lack of maintenance and governmental investment, the capacity of the irrigation and drainage systems is considerably reduced, and about 2,500 km of canals and drainage collectors in Uzbekistan urgently require reconstruction. Moreover, due to the fact that only 11% of these canals are lined (Vodproject 1999) the seepage losses are high and it is estimated that only about 45-50% of the water reaches the farmers' fields.

The drainage network in the Khorezm region is mainly open horizontal. The total length of the network was 9,255 km in 1997, while the length of the tile drains was 414 km (Vodproject 1999). Drainage water is conveyed via a complex network of collectors from the irrigated fields into numerous small lakes and depressions outside the irrigated areas. The main receiver of drainage water is the Sarykamish Depression, which was formerly connected with the Aral Sea.

4.5 Natural vegetation

Previous to the introduction of the extensive irrigation systems, the natural vegetation in the Amu Darya Delta consisted mainly of Tugai riparian forests and widespread reed communities that typically occurred as narrow belts (from a few hundred meters up to several kilometres width) along the Amu Darya River, and white and black saxaul (*Kaloxylos persicum* L. and *Haloxylon aphyllum* L.) communities in the transition zones to the Kara Kum desert. Tugai forests are fast-growing, deciduous forests along river reaches or canals (Novikova et al. 2001). For their growth they need regular flooding and therefore occur mainly on sand banks, islands and low terraces (Kuzmina and Treshkin 1997). The Tugai vegetation is characterized by high tolerance to both very wet soil and very dry air, resistance to drought (however, somewhat limited) and salts, and high transpiration intensity (Treshkin et al. 1998). With more than 230 plant species, the Tugai forest is considered to be one of the most diverse vegetation types of arid regions in Central Asia (Novikova et al. 2001). The main tree species of the woody Tugai are poplar (*P. euphratica* L., *P. pruinosa* L.), oleaster (*Eleagnus turcomanica* L., *E. angustifolia* L.), and willow (*Salix songarica* L., *S. wilhelmsiana* L.). They are accompanied by bushes and tall grasses, such as tamarisk (*Tamarix ramosissima* L., *T. laxa* L.), Halimodendron (*H. halodendron* L.), reed (*Phragmites australis* L.), reed grass (*Calamagrostis dubia* L., *C. epigeios* L., *C. pseudophragmites* L.), and herbs, e.g., liquorice (*Glyrrhiza glabra* L.) and *Apocymum scabrum* L. (Kuzmina and Treshkin 1997).

Due to anthropogenic pressure and overexploitation of Amu Darya water for irrigation, approx. 90% of the original 300,000 ha Tugai forest along the lower Amu Darya disappeared during the last century (Kuzmina and Treshkin 1997; Novikova et al. 1998; Treshkin 2001). Nowadays, Tugai vegetation in Khorezm is only found in small patches along the Amu Darya north of the capital Urgench. Reed communities and different halophytes can be found at small lakes and along the irrigation canals.

4.6 Land use

Agriculture has been practiced in the Khorezm Oasis for thousands of years, and millet, wheat, barley, water melons, honey melons, and gourds have been grown under irrigation since historical times (Forkutsa 2006). Following the creation of the extensive irrigation systems by the Soviets in the mid 20th century, agricultural practices that had been sustained for over two thousand years under irrigation practices solely based on gravity were replaced by an agricultural system with the diversion of massive amounts of water from river valleys to the surrounding steppes and deserts, primarily for cotton cultivation (Vinogradov and Langford 2001). Currently, 270,000 – 300,000 ha of land are used for irrigated agricultural production. The fiber crop cotton is the most important cash crop in Khorezm (*Gossipum hirsutum* L.; 46% of the irrigated area in 2005) followed by winter wheat (*Triticum aestivum* L.; 23%) and rice (*Oryza sativa* L.; 21%) (Conrad 2006). Fodder and garden crops occupy the remaining irrigated areas, while tree plantations and shelterbelts cover a rather small proportion of the landscape (about 2.3 % of the irrigated land; Tupitsa et al. 2006).

Agricultural production in Uzbekistan is largely state controlled, and three major farm types have emerged in different steps during the transition process after independence from the Soviet Union: as a first step, the agricultural cooperatives (*shirkat*) were created as a transitory successor of former *kolhozes* and s*ovkhozes*. In a second step, the *dehqon* farms, which are the Uzbek version of the small, subsistence-oriented household plots that represent an important contribution to household food security and the *fermer* enterprises were created from dissolving the *shirkat*s. The *fermer* enterprise is a new type of farm that has emerged during the past five years, is established on the basis of long-term leases and has a commercial orientation (Martius and Wehrheim 2008). By 2005, these reforms had resulted in the establishment of 13,839 *fermer* enterprises cropping about 188,329 ha or an average of 13.6 ha per farm and 247,840 *dehqon* farms cropping about 48,912 ha or an average of 0.2 ha per farm (Khorezm hokimiat 2005).

Hence, the Khorezm region is representative of various regions in the Aral Sea Basin that suffer from a high dependency on irrigated agriculture and chemical fertilizer use on the most dominant crops such as cotton and wheat. In particular, the combination of excess soil N, wet conditions and high temperatures have been shown worldwide to provoke or enhance GHG emissions (Bouwman 1996; Dobbie et al. 1999), which however has never been validated in the central Asian regions and under the arid continental climate.

5 NITROUS OXIDE EMISSIONS FROM FERTILIZED, IRRIGATED COTTON (*GOSSYPIUM HIRSUTUM L.)*: INFLUENCE OF NITROGEN APPLICATIONS AND IRRIGATION PRACTICES

5.1 Introduction

The role of irrigated agriculture in food production is significant; although only 17% of global cropland is irrigated, it provides 40% of the world food production (FAO 2000). Moreover, irrigation is globally responsible for approximately 70% of anthropogenic water consumption (FAO 2000). Irrigation not only stimulates plant growth, but also accelerates microbial C and N turnover in the soil (e.g., Andren et al. 1992; Davidson 1992). To obtain optimal irrigation benefits, additional crop management practices to optimize nutrient inputs and mode of tillage must be adapted. Any modification of crop management and irrigation practice will affect the carbon and nitrogen cycles of these agricultural systems.

In the Aral Sea Basin (ASB), intense agricultural irrigation has reduced the river discharge to the Aral Sea, resulting in more than 80% loss of its volume over the past decades (Micklin 2007). This demise has led to the “Aral Sea Crisis”, which denotes a complex combination of ecological consequences of regional and global dimensions. In Uzbekistan, cotton cultivation was continued after independence from the Soviet Union, and the country still ranks as the fifth largest cotton producer in the world (Bremen Cotton Exchange 2007). The current agricultural production systems are characterized by crop rotations of cotton-wheat-rice under heavy inputs of water and fertilizers. High inputs of mineral N (150-300 kg N ha^{-1} yr^{-1}), in combination with high topsoil moisture levels after irrigation, are conducive to significant N losses to the environment either in the form of nitrate or as gaseous N (NH_3, NO, N_2O, N_2) to the atmosphere.

Although N_2O fluxes under different cropping systems have been investigated (Bouwman et al. 2002), only limited information is available for irrigated agriculture. A few studies reported a strong stimulation of N_2O fluxes by irrigation, but these studies were conducted in temperate semiarid agricultural systems (Jambert et al. 1997a; Hao et al. 2001) or semiarid subtropical rice/wheat rotation systems (Aulakh et al. 2001; Majumdar et al. 2002). For irrigated cotton, various studies identified denitrification as the main pathway of fertilizer losses from the soil-plant system (Chua et al. 2003), but did not report on N_2O fluxes (Hou et al. 2007). In Australia, denitrification losses of 40-60% of the applied N fertilizer were reported (Rochester et al. 1996). This is in accordance with findings by Mahmood et al. (2000) who measured denitrification losses of 65 kg Nha^{-1} using the acetylene inhibition. This corresponded to 40% of the applied fertilizer during one season in the semiarid subtropical climate of Pakistan. Based on a ^{15}N balance approach, Rochester (2003) estimated that roughly 2 kg Nha^{-1} (~1.1% of the N applied) was lost as N_2O during the cotton-growing season.

However, to the best of our knowledge, no investigations have been published on N_2O emissions from irrigated agricultural systems in an arid environment based on in-situ flux measurements. Given the 2.7 millions of ha of irrigated cotton in the five central Asian countries alone (FAOSTAT 2007), this topic is of great importance. The aims of this study were therefore (i) to identify the site specific regulating parameters for N_2O emissions from irrigated cotton fields in an arid area of Uzbekistan; (ii) to quantify losses of N_2O emissions from variously managed (water regime/ fertilizer management) cotton fields throughout the vegetation cycle; (iii) to assess the potential of management and irrigation practice for reducing the emissions of nitrous oxide.

5.2 Material and methods

5.2.1 Study sites

A field experiment was carried out on research sites of the ZEF/UNESCO project in the Khorezm Region, Uzbekistan, between April 2005 and October 2006. The research station was located at 41º55’ N latitude, 60º61’ E longitude and at an altitude of 92 m a.s.l. The climate is typically arid continental with long hot dry summers and very cold temperatures in winter. Average precipitation during 1982 to 2000 was less than 100 mm per year and the mean annual temperature was 13.6 ºC (Glavigdromet 2003).

Three sites differing in soil texture were selected for the flux measurements. Two sites were part of the Amir Temur Shirkat (a collective farm established from a Soviet kolkhoz or “sovkhoz” farm after independence) situated in the vicinity of the research station. In 2005 and 2006, N_2O emissions were measured during the entire cotton growing period, which lasted from April to October, on experimental fields of the Amir Temur Garden (ATG) farm situated in the central part of the Shirkat. In addition, fluxes were measured in 2005 during May-October on a field of the Amir Temur Cum (ATC) farm in the western part of the Amir Temur Shirkat. The soils were classified as calcaric gleyic Arenosols (FAO 1998) with silty loamy texture. The land had previously been in a rice/cotton/winter wheat crop rotation. The ATG and ATC sites were completely managed by local farmers following common practice fertilizer and irrigation strategies, which allowed for monitoring of the impact of different local farm management strategies on the emissions of nitrous oxide.

Table 5.1: Soil properties of the different research sites.

Research Site	Amir Temur Garden (ATG)	URDU research Site (URDU)	Amir Temur Cum (ATC)
SOC (%)	0.61	0.58	0.53
N (%)	0.10	0.10	0.07
pH(H_2O)	6.9	6.5	6.6
Bulk density (g cm^{-3})	1.59	1.51	1.41
Texture (USDA)	Sandy loam	Sandy loam	Silt loam
Clay (%)	14.6	15.5	17.9
Silt (%)	42.8	32.1	55.4
Sand (%)	42.6	52.4	26.7

In 2006, N_2O fluxes were recorded on an experimental site at the campus of Urgench State University (URDU), located on a calcaric gleyic Arenosol (FAO 1998) with a sandy loam soil texture. The experiment was a split-plot design with a total of 48 subplots each 2.5x2.5m in size, where irrigation was applied. This experiment included two types of irrigation management practices: (i) high intensity irrigation (HI) meaning after the first irrigation of the cotton in June the next irrigation took place when the soil moisture level was 75% of field capacity, and (ii) low intensity irrigation (LI): after the first irrigation of the cotton in June, the next irrigation occurred when the soil moisture decreased to 65% of field capacity. This allowed the effect of two soil moisture regimes on N_2O emissions to be investigated on one field, characterized by higher soil moisture content and more frequent irrigation applications. Soil characteristics and the experimental set-up of all sites are shown in Table 5.1 and Table 5.2.

Table 5.2: Experimental set up of agricultural fields.

Site	Amir Temur Garden (ATG)		URDU Research Site (URDU)		Amir Temur Cum (ATC)
Observation period	Apr-Oct 2005	Apr-Sep 2006	Apr-Sep 2006		May-Oct 2005
Irrigation intensity	Local farming practice		High	Low	Local farming practice
Area (ha)	1	1	0.5	0.5	2
Planting Date	21/04/2005	21/04/2006	22/04/2006	22/04/2006	15/04/2005
Harvesting Date	Sep/Oct 2005	Sep/Oct 2006	Sep/Oct 2006	Sep/Oct 2006	Sep/Oct 2005
Fertilizer (kg N ha^{-1})	Total: 250 May, 26: 75 Am.Nitr. June, 25: 87.5 Am.Nitr. July, 12: 87.5Am.Nitr.	Total: 200 May, 15: 42 Am.Sulph. June, 5: 42 Am.Phos. June, 12: 115 Urea	Total: 250 April, 19: 75 Am.Nitr. June, 17: 87.5 Am.Nitr. July, 5: 87.5 Am.Nitr	Total: 250 April, 19: 75 Am.Nitr. June, 17: 87.5 Am.Nitr. July, 7: 87.5 Am.Nitr	Total: 162.5 April, 13: 75 Am.Nitr. June, 26: 87.5 Am.Nitr.
Irrigation (mm)	Total: ~300 May, 27 June, 25 July, 12 July, 28 August, 15	Total: ~400 June, 7 June, 18 July, 8 July, 25 August, 14 August, 29	Total:463 June, 19 June, 26 July, 5 July, 12 July, 19 August, 5 August, 14 August, 23	Total:373 June, 19 June, 28 July, 7 July, 15 July, 28 August, 15 August, 24	Total:~300 July, 2 July,11 August, 11 August, 20

5.2.2 Determination of N_2O fluxes

Nitrous oxide emissions were measured using the closed chamber technique (IAEA 1992). This method uses a gas-tight chamber enclosing soil and plants over a given interval. The chamber consists of a frame inserted a few cm into the soil and a polypropylene box that is fixed to the frame throughout the sampling period. Chamber enclosure is achieved by a sealed gasket at the lower edge of the box and metal brackets that press the box onto the frame, air-tight. The frames remained on the plots during the entire experimental period and were only removed short-time during soil tillage operations.

The volume of each chamber was approximately $0.08m^3$ and the cross-sectional area was $0.21m^2$. Due to the chamber size, the cotton plants were only enclosed for measurement during the beginning of the growing cycle of cotton. At a later stage, the cotton plants were too high for the chambers. To check whether the effect of the plants can be ignored, we conducted an experiment on several sampling days in July 2007 using bigger chambers to compare N_2O emissions above the plants with the normal measuring chambers. No significant effect of the plants on N_2O emissions could be determined.

Fluxes were measured by collecting air samples from the chamber head space. 20 ml of headspace air was drawn through a septum into gas-tight 20 ml polypropylene syringes at 0, 10, 20, and 30 min after the soil was covered. The syringes were closed with a Luer Lock valve immediately after air sampling to prevent gas exchange. The syringes were tested for leaks on several occasions during the field campaign using calibration gas.

Table 5.3: Special characteristics of the used GC system

GC- System	Shimadzu GC 14B
Detector	^{63}Ni Electron capture detector (ECD)
Carrier gas	N_2
Pre column	Ascrite
Column	12ft Porapak Q 80/100 mesh
Flux of carrier gas [ml min^{-1}]	30
Oven temperature [°C]	40
ECD –Temperature [°C]	340

The N_2O content of the air samples was analyzed on the day of sampling at the laboratory of the ZEF Khorezm project in Urgench. The analytical system consisted of a Shimadzu G 14A Gas Chromatograph (GC) that was equipped with an 'electron capture detector' (ECD). For details of the GC set up refer to Table 5.3. The samples were injected into a sample loop of 3ml Volume and then inserted into the column using an 8 port Valve (Valco Sys., USA). Because of the similar molecular characteristics of N_2O and CO_2 we used a pre column filled with Ascarite (Sodium Hydroxide Coated Non-Fibrous Silicate) (Aldrich, Milwaukee, USA) to eliminate CO_2 from the sample air. In addition, because of the hygroscopic property of Ascarite, H_2O vapour in the sample was also removed. Since the filter capacity of Ascarite will decrease with the uptake of water vapour, these columns were exchanged every 2 weeks.

For calibration a gas mixture of synthetic air containing 397 ppb of N_2O (Messer Griessheim, BRD) was used. The chromatograms were evaluated using Class

VP integration software and the area of the N_2O peaks determined. The concentration of N_2O in the sample air was then calculated using equation (5.1):

$$\left[N_2O_{sample}\right] = \frac{PA_{sample} \cdot \left[N_2O_{CG}\right]}{PA_{CG}} \tag{5.1}$$

Where: $\left[N_2O_{sample}\right]$: N_2O concentration of the sample [ppb]

$\left[N_2O_{CG}\right]$: N_2O concentration of the calibration gas [ppb]

PA_{sample}: Peak area of the sample [msV]

PA_{CG} : Peak area of the calibration gas [msV]

N_2O emissions were calculated from the linear increase of the gas concentration at each sampling time (0, 10, 20 and 30 min during the time of chamber closure). The slope b [ppb/min] of the increase in concentration of N_2O inside the measuring chamber was calculated via linear regression. Besides the increase b also the correlation coefficient R was calculated and used as a quality check for the measurement; for $R^2 < 0.9$ ($R^2 < 0.7$ for small flux rates) the measurement was rejected. The flux rate F_{N2O} was then calculated using equations (5.2) and (5.3). All N_2O flux rates were corrected in order of the actual air temperature during the measurement and quoted as [$\mu gNm^{-2}h^{-1}$].

$$F_{N2O} = \frac{b \cdot V_{CH} \cdot MW_{N_2O-N} \cdot 60 \cdot 10^6}{A_{CH} \cdot MV_{corr} \cdot 10^9} \tag{5.2}$$

Where: V_{CH} : Basal area of the measuring chamber [m^2]

b : Increase of concentration [ppb min^{-1}]

MW_{N_2O-N} : Molecular weight of N-N_2O [28 g mol^{-1}]

MV_{corr} : Temperature corrected molecular Volume [m^3 mol^{-1}]

V_{CH} : Volume of the measuring chamber [m^3]

Nitrous oxide fluxes were measured at four replicated plots within each experimental site at least twice each week throughout the crop growing season. Measurements were conducted three to four times a week immediately after each fertilization and irrigation event. Fluxes were measured twice each sampling date in 2005 (between 8:00h and

11:00h in the morning and 13:00h and 15:00h in the afternoon) and once per sampling date in 2006 (between 8:00h and 11:00h).

$$MV_{corr} = 0.02241 \cdot \left(\frac{273.15 + T}{273.15} \right) \tag{5.3}$$

Where: MV_{corr} : Temperature corrected molecular Volume [m^3 mol^{-1}]

T : Air temperature during the measurement [°C]

To assess the diurnal pattern of N_2O emissions, two intensive measurement campaigns were conducted in the HI plots at the URDU research site in 2006, where fluxes were measured every 2 hours during the day and every 4 hours at night for a 48-hour period directly after irrigation.

5.2.3 Auxiliary data

Soil temperature was measured at the same time as gas sampling at a depth of 10 cm for each treatment. Air temperature and precipitation data were supplied by the meteorological station (Khiva Meteorological Station). During the gas sampling period, soil samples were taken at the 0 to 10 cm depth. Soil samples were cooled for transportation and processed the same day. Soil moisture was determined gravimetrically after drying for 24 hours at 105 ºC or until constant weight. Water-filled pore space (WFPS) was calculated using the measured soil bulk density data (arithmetic means of four samples) using a particle density of 2.65 g cm^{-3}.

In addition, at the beginning and end of the growing season, bulk soil samples were taken from each site by combining 5-10 soil cores (0-10cm depth). The samples were air-dried in the shade for 6 days, sent to the Tashkent Soil Science Institute, Uzbekistan, and analyzed for soil texture, total carbon (C %), total nitrogen (N%), mineral nitrogen content (NO_3^-, NH_4^+), and bulk density (see Table 5.1).

Statistical analyses were calculated using SPSS 8.0 (SPSS Inc., Chicago, USA). For analyzing the normal distribution of the data the Kolmogorov-Smirnov test was performed. As N_2O fluxes were non-normally distributed, the non-parametric Mann-Whitney test was performed. Data presented in figures are transformed values and presented as means ± 1 standard error. Connecting lines are included to clarify the data points.

5.3 Results

5.3.1 Seasonal pattern of N_2O emissions

Amir Temur Garden Farm

The temporal course of N_2O emissions at the ATG site for the years 2005 and 2006 is characterized by high emission levels following fertilization and irrigation (F+I) events and lower fluxes in the intermediate periods (Figure 5.1).

In both years, the plots were not irrigated during the first weeks after sowing, which explains the low soil water content (approx. 20% WFPS). During these initial periods characterized by WFPS values of approx. 20%, N_2O emissions were <10 µg N_2O-N m^{-2} h^{-1} (Figure 5.1). After fertilizer application with subsequent irrigation, N_2O emissions increased up to 3000 µg N_2O-N m^{-2} h^{-1} (Figure 5.1). These “emission pulses” accounted for 90% of the total N_2O emissions throughout the entire cropping cycle. The episodic emission pattern was similar for both years.

In 2005, the first fertilization at the end of May was directly followed by an irrigation event. N_2O emissions increased to values > 200 µg N_2O-N$m^{-2}h^{-1}$. The second F+I event on June 27th resulted in extremely elevated N_2O fluxes with an average daily mean greater than 2000 µg N_2O-N$m^{-2}h^{-1}$. The magnitude of this second emission pulse was 5 to 10 times higher compared to the first post F+I event. The third F+I event in mid-July showed a similar pattern to the second. They were followed by an emission pulse one or two days after fertilization and irrigation and stayed elevated for about seven days It is noteworthy that irrigation events without preceding fertilization (end of July and mid of August 2005) did not result in increased N_2O emissions, so that N_2O emissions were low (<50 µg N_2O-N$m^{-2}h^{-1}$) during crop maturity.

In 2006, ammonium (ammonium sulphate and ammonium phosphate) combined with urea was applied, while in 2005 pure ammonium nitrate was applied (Table 5.2). The seasonal emission pattern showed similar characteristics to those observed in 2005, although the magnitude of the emission pulses was lower than in 2005 (Figure 5.1). During the first weeks after sowing without irrigation and fertilization, only very low (<10 µg N_2O-N m^{-2} h^{-1}) N_2O emissions were measured.

These emissions remained low after the first fertilizer application on May 15th which, in contrast to 2005, had no accompanying irrigation. The first emission pulse (>200 µg N_2O-N m^{-2} h^{-1}) was therefore triggered by the first combined application of fertilizer and irrigation on June 6th. Emissions stayed elevated for approx. seven days and were followed by an even higher emission pulse (> 1000 µg N_2O-N$m^{-2}h^{-1}$) after the 2nd F+I event on June 22nd The emissions stayed slightly elevated for more than 14 consecutive days, including during a second irrigation. After subsequent irrigations without fertilizer application (July 26th, August 12th and 26th), N_2O emissions <10 µg N_2O-N m^{-2} h^{-1} were detected.

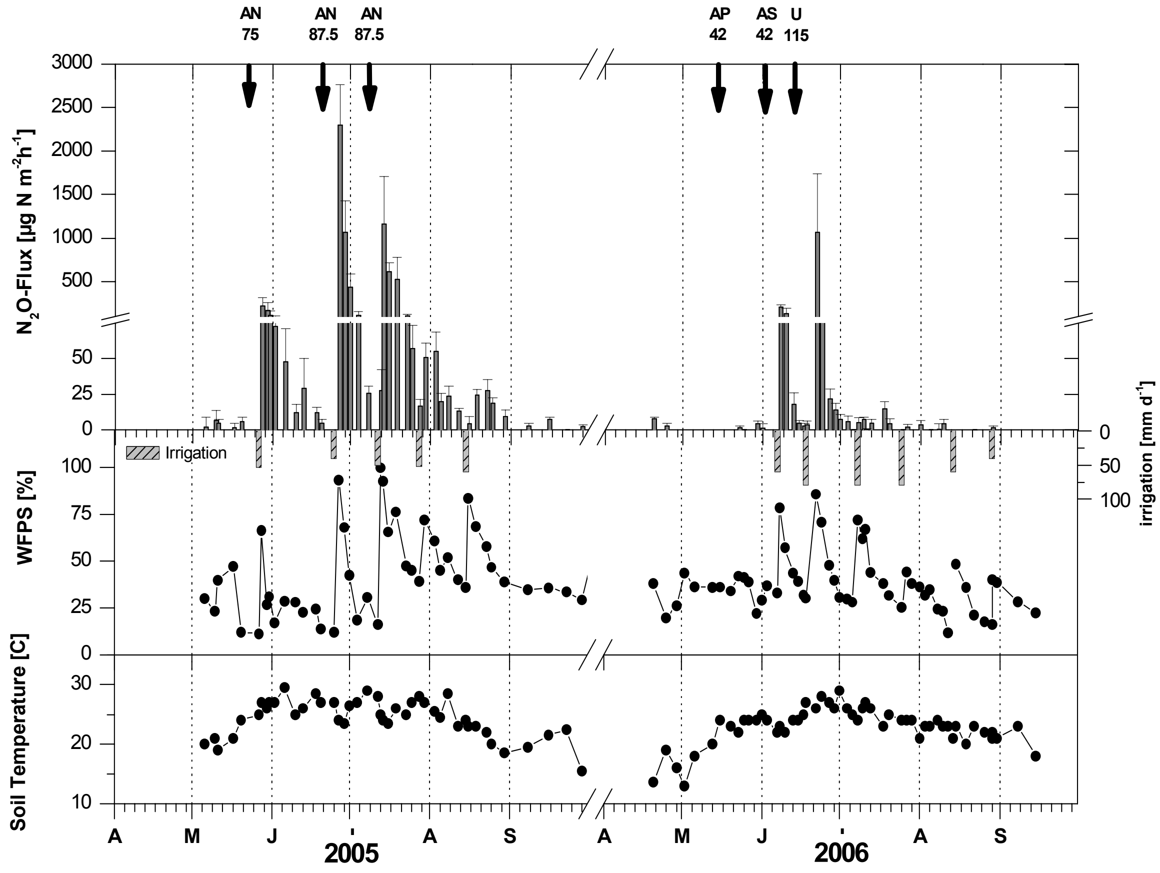

Figure 5.1: N_2O flux rates, irrigation rates, WFPS and soil temperature of the cotton field at Amir Temur Garden from April to September 2005 and 2006. Arrows indicate the events of N (kg N ha^{-1}) application to the plots (U = Urea, AN = Ammonium Nitrate, AP = Ammonium Phosphate, AS = Ammonium Sulphate), whereas bars provide information about the amount of irrigation. Error bars indicate the standard error. Connecting lines are inserted for showing the data points more clearly.

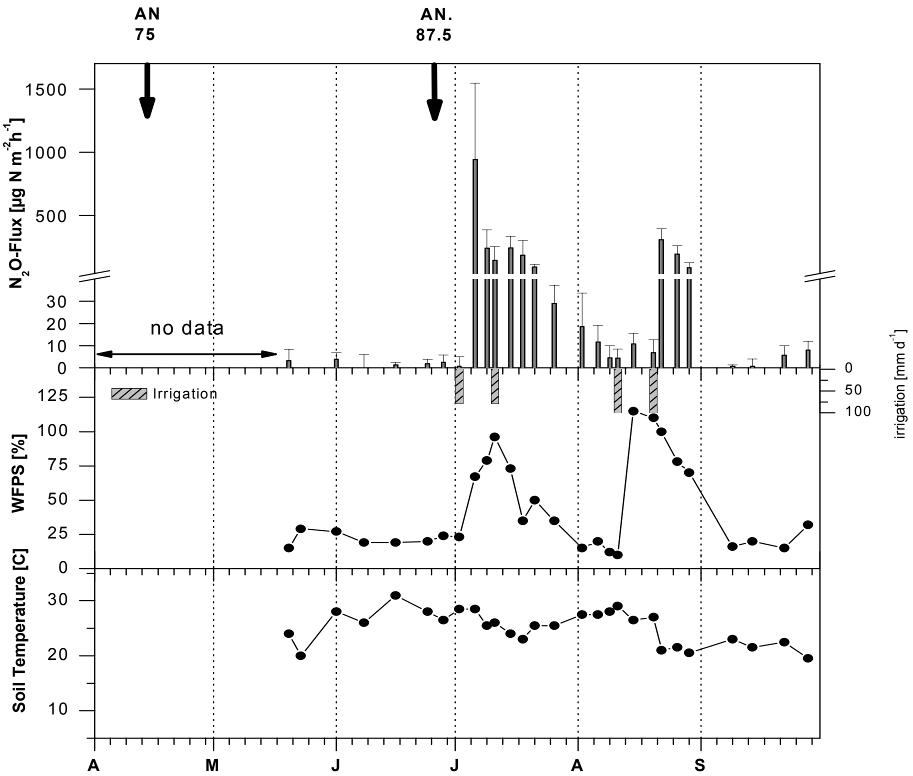

Figure 5.2: N_2O flux rates, irrigation rates, WFPS and soil temperature of the cotton field at Amir Temur Cum from May to September 2005. Arrows indicate the events of N (kg N ha^{-1}) application to the plots (AN = Ammonium Nitrate), whereas bars provide information about the amount of irrigation. Error bars indicate the standard error. Connecting lines are inserted for showing the data points more clearly.

Amir Temur Cum Farm

The temporal course of N_2O emissions at the ATC site showed similar characteristics to those observed at the ATG site. The temporal course of the emission rates of this site in 2005 is shown in Figure 5.2. Fertilizer was first applied before sowing cotton (mid April). Since the field was not irrigated until the onset of July, the topsoil water content stayed low (<30% WFPS) resulting in very low N_2O emissions (<<10 µg N_2O-Nm^{-2}h^{-1}) during the first measurements 30 days after sowing. The 1[st] F+I event in July caused a N_2O emission pulse of almost 1000 µg N_2O-N m^{-2} h^{-1}. The second irrigation (seven days later) resulted in elevated emissions but of a significantly lower magnitude (<250 µg N2O-N m^{-2} h^{-1} maximum flux). With the gradual desiccation of the topsoil, the emissions decreased to <10µg N_2O-Nm^{-2}h^{-1}. Following two consecutive irrigations in August, also without N fertilization, N_2O emissions increased again to values of up to

300µg N_2O-N$m^{-2}h^{-1}$. During this period, N_2O emissions stayed elevated for approximately seven days. During the following month irrigation ceased, resulting in low topsoil water content, and only very small N_2O emissions (<10 µg N_2O-N$m^{-2}h^{-1}$) were detected.

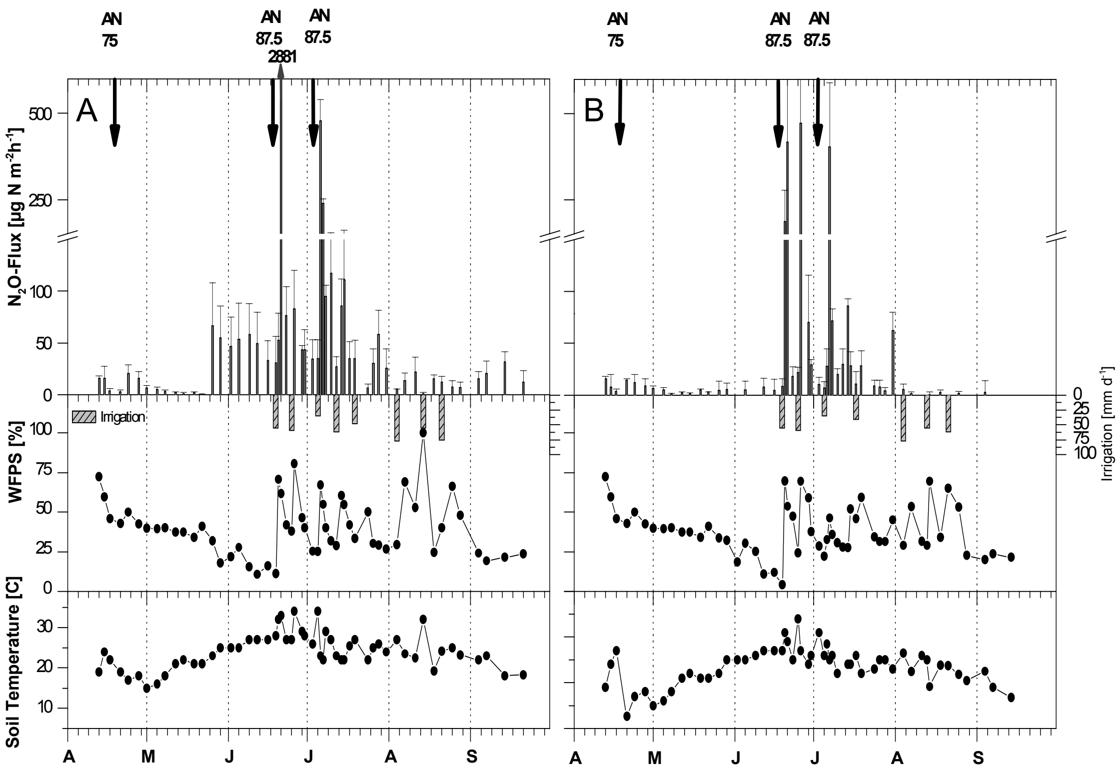

Figure 5.3: N_2O flux rates, irrigation rates, WFPS and soil temperature of the high (A) and low (B) irrigated cotton field at the URDU research site from April to September 2006. Arrows indicate the events of N (kg N ha^{-1}) application to the plots (AN = Ammonium Nitrate), whereas bars provide information about the amount of irrigation. Error bars indicate the standard error. Connecting lines are inserted for showing the data points more clearly.

URDU research site

Flux measurements at this site encompassed one plot with high irrigation intensity (HI) and one with low irrigation intensity (LI). Both plots were fertilized with 250 kg-N ha^{-1} of ammonium nitrate. A total of 463 mm was irrigated during eight events on the HI plot, whereas during the 7 irrigation events at the LI plots, a total of 373 mm was applied (Table 5.2)

As at the Amir Temur sites, the emission rates at the URDU site showed a seasonal pattern following the fertilization and irrigation events (Figure 5.3). More than

90% of the total N_2O emissions were observed during June and July when fertilization and irrigation was most intense. The significant increase of the flux rates at the HI plot at the end of May occurred after the measuring frames had been removed and installed in different locations of the plots and were not related to any management practice. While the single fertilizer application in April showed no impact on N_2O emissions, the first F+I event in June triggered elevated emissions at both plots. At the HI plot, this emission pulse reached an average value of almost 2900 µg N_2O-N m^{-2} h^{-1}, However, high standard deviations, caused by the extremely high emission rates of almost 10000 µg N_2O-N m^{-2} h^{-1}, were measured in only one of the four chambers. A subsequent single irrigation had no impact on N_2O emissions, but the subsequent F+I irrigation event in early July caused elevated N_2O emissions with values of >400µg N_2O-N$m^{-2}h^{-1}$. The following irrigation events in July and August (without fertilization) only slightly increased N_2O emission rates. In the LI plot, two emission pulses could be observed following F+I events and one pulse after irrigation only. However, the absolute values of these pulses varied from 400 to 500 µg N_2O-N m^{-2} h^{-1} and did not reach the magnitude of the highest emission in the HI plots.

5.3.2 Cumulative N_2O emissions and emission factors

During the whole observation period, 200-250 kg N ha^{-1} was applied as mineral fertilizer to the different fields (Table 5.2). The cumulative fluxes of the different research sites show that the average seasonal emission flux was 20.6 ± 7.32 µg N_2O-N m^{-2} h^{-1} for the Amir Temur site in 2006 and 149.8 ± 10.0 µg N_2O-N m^{-2} h^{-1} for the same site in 2005 (Figure 5.4). The maximum flux rate observed occurred after fertilization and irrigation of the HI plot at the URDU site on June 21st and reached 9612.7 µg N_2O-N m^{-2} h^{-1}. For the entire cotton vegetation period, cumulative N_2O emissions varied from 0.9 to 6.5 kg N_2O-N ha^{-1} at the different sites, corresponding to emission factors (EF), uncorrected for background emission, varying from 0.4 to 2.6% of the total amount of mineral N applied to the fields with an average value of 1.48% (Figure 5.4).

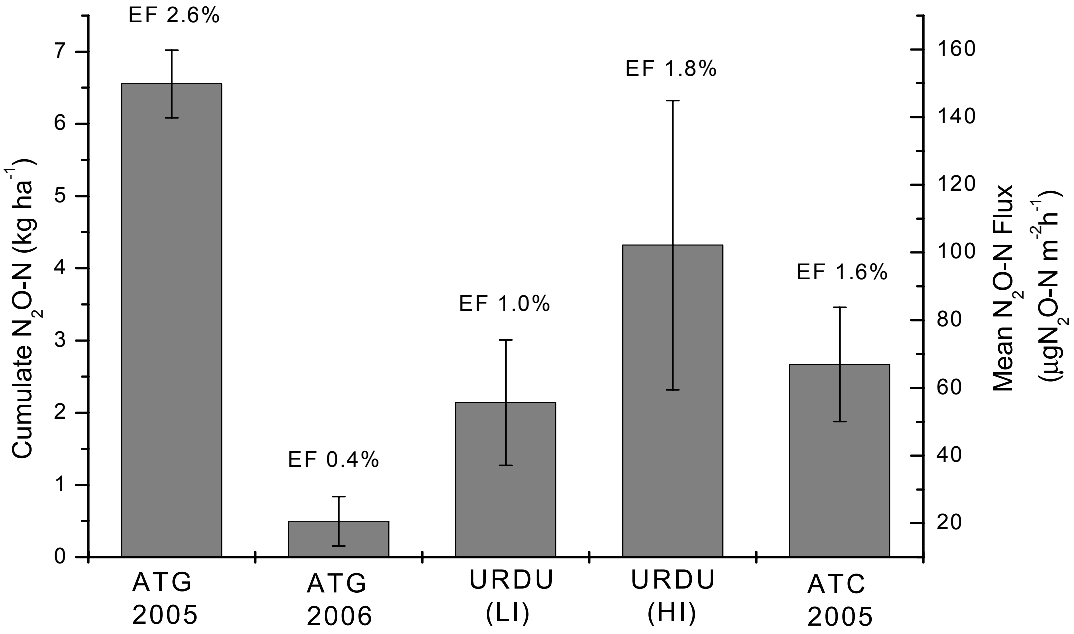

Figure 5.4: Cumulative N_2O emissions, mean N_2O-N flux and emission factors (EF) of the different research sites: ATG, URDU, and ATC. Vertical bars indicate the standard error.

5.3.3 Diurnal pattern of N_2O emission

To assess if diurnal variations in N_2O emissions occurred after irrigation and fertilization events, two intensive measurement campaigns with subdaily flux measurements were conducted at the URDU site. On the 5th of July, after concomitant fertilization and irrigation, fluxes were measured (87.5 kg N ha^{-1} Ammonium-Nitrate, 33 mm irrigation water). On the 13th of July fluxes were measured after irrigation alone (60 mm irrigation water).

The N_2O fluxes observed on July 5th were in general three to four times higher than the N_2O fluxes from July 13th (Figure 5.5). On both observations dates, the fluxes constantly increased during the first 12 hours, despite decreasing soil temperatures. After these initial 12 hours, the diurnal pattern of N_2O fluxes followed the soil temperature. On July 6th and 7th, the maximum flux occurred at 14:00, which coincided with the daily maximum soil temperature. The magnitude of both flux maxima declined from the first to the second day after irrigation. On July 6th a maximum flux of 646.3 µg N_2O-N m^{-2} h^{-1} occurred, compared to a maximum flux of 343.6 µg N_2O-N m^{-2} h^{-1} on July 7th. On July 14th and 15th, a peak flux occurred at 16:00 and 14:00, respectively, which also followed the pattern of the daily temperature pattern in the topsoil. During the second observation period, the magnitude of the flux rates was lower, while the spatial variability of the flux rates was higher. Only one of the four replicate measurements showed a clear diurnal pattern and maximum flux rates greater than 400 µg N_2O-N $m^{-2} h^{-1}$ while at the other 3 measurements no diurnal pattern were observed, with flux rates ranging from 0 to 150 µg N_2O-N $m^{-2} h^{-1}$.

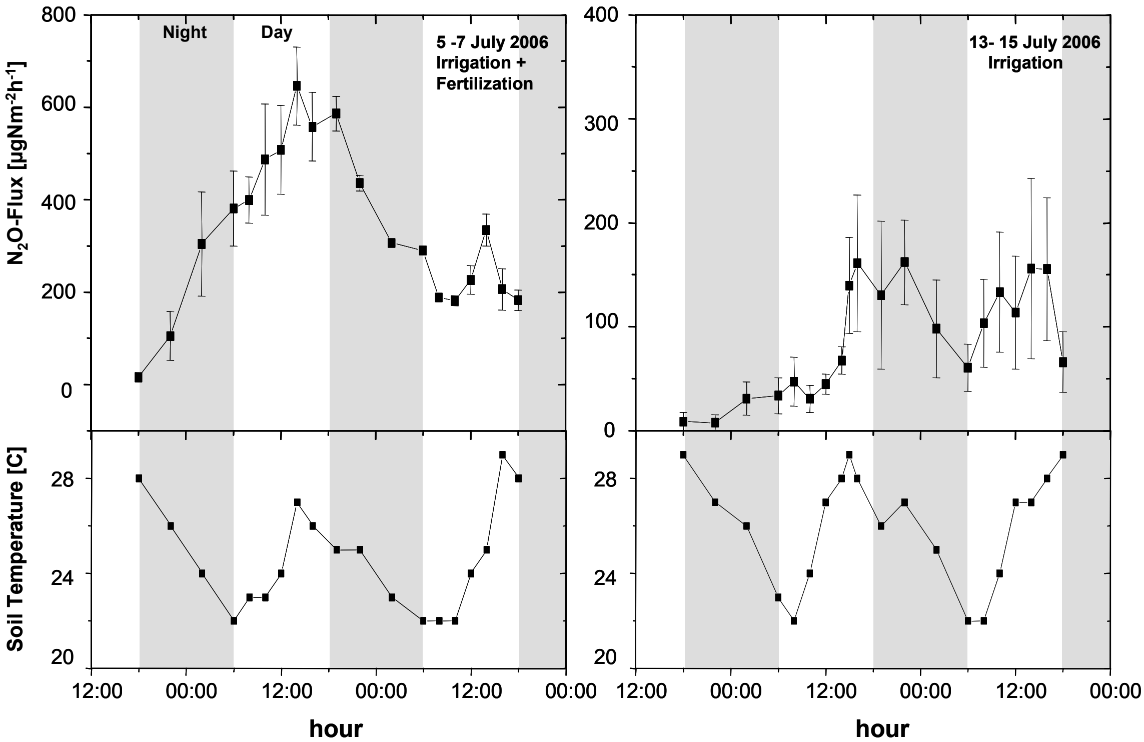

Figure 5.5: Diurnal patterns of N_2O fluxes and soil temperature (10 cm depth) at 48h cycles directly following irrigation events. Error bars indicate the standard error of the means (n =4). Connecting lines are inserted showing the data points more clearly.

5.3.4 Mean fluxes following irrigation

At all observation sites, highest flux rates were found within the first three days after an F+I event, while irrigation without fertilization resulted only in slightly increased N_2O emissions (Figure 5.6). The mean flux during the first three days after an F+I event reached 500µg N_2O-N $m^{-2} h^{-1}$, compared to a mean flux of only 32 µg N_2O-N $m^{-2} h^{-1}$ after irrigation alone. After an F+I event the flux rates stayed elevated for 7 to 10 days and decreased exponentially with the desiccation of the soil.

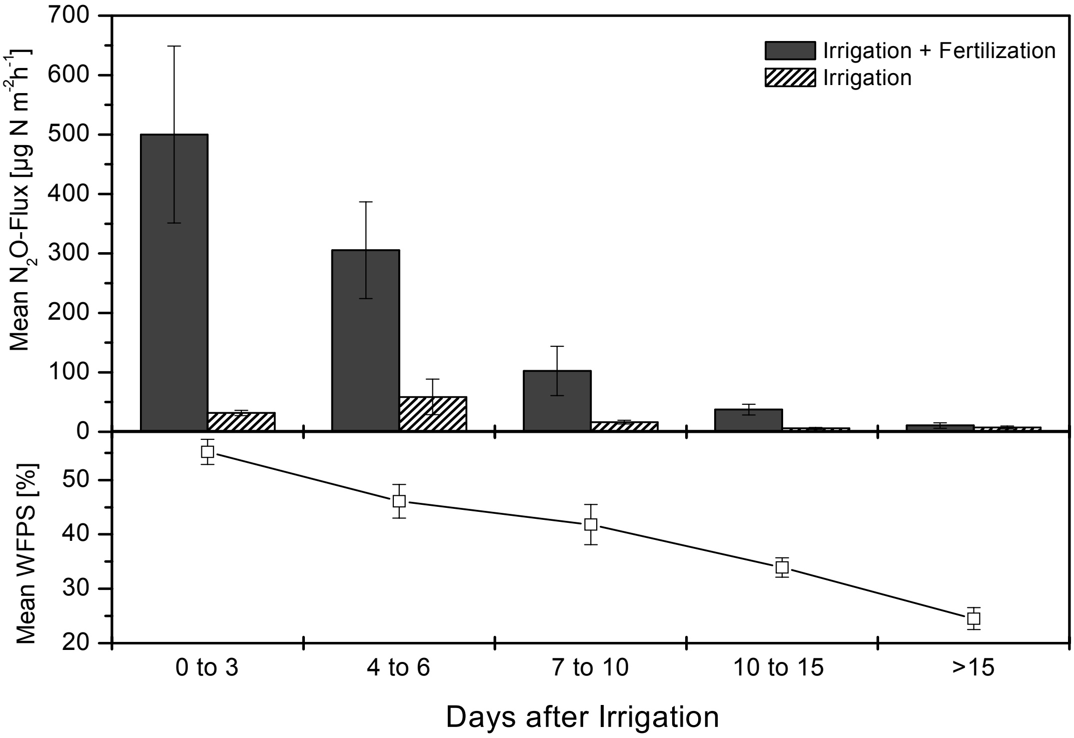

Figure 5.6: Temporal course of the mean N_2O flux rates and mean WFPS of all observation sites following fertilization + irrigation and sole irrigation events. Error bars indicate the standard error of the means.

5.4 Discussion

Our measurements represent the first field data set of N_2O emissions from an irrigated, agricultural system in an arid region. To date only field measurements of semiarid agricultural systems have been published (e.g., Jambert et al. 1997b; Hao et al. 2001; Pathak et al. 2002). Some research has reported strong stimulation of denitrification rates in irrigated cotton based on incubation studies or ^{15}N balance techniques (Rochester et al. 1996; Mahmood et al. 1998b; Mahmood et al. 2000). Our results corroborate that nitrous oxide emissions are primarily controlled by the fertilization and irrigation practices in cotton. Irrespective of the differences in experimental sites and management, pulses of nitrous oxide emissions were triggered chiefly by the combined impact of irrigation and fertilization as evidenced by the highest emissions following irrigation immediately after fertilizer N application. These findings agree with N_2O emissions research on irrigated agricultural ecosystems (e.g., Jambert et al. 1997a; Majumdar et al. 2002; Xiong et al. 2006) and other studies on N_2O emissions from N fertilized soils in temperate regions where highest N_2O emissions occurred following rainfall soon after fertilizer N application (Smith et al. 1998; Dobbie et al. 1999; Hyde et al. 2006). Thus we could identify soil moisture and soil mineral N content as the main parameters effecting N_2O emissions at our sites.

The impact of soil moisture content on N_2O emissions has frequently been described for agricultural soils (e.g., Davidson 1993; Lessard et al. 1996; Zheng X. et al. 2000). Various authors (e.g., Smith et al. 1998; Simojoki and Jaakkola 2000; Dobbie and Smith 2001; Sehy et al. 2003) measured highest flux rates at WFPS between 60 and 90% in soils with fertilized grassland. In our study, 60% of the flux rates higher than 50µg N_2O-N m^{-2} h^{-1} were detected at a WFPS greater than 60%, but all of the high N_2O flux rates were restricted to a maximum period of ten days after an irrigation event (Figure 5.6). Under these conditions, inner aggregates of alluvial meadow soils with loamy or silty loamy texture usually are waterlogged and sustain denitrification, even if a large proportion of pores between aggregates is already air filled (Smith 1980). We thus have to assume that elevated N_2O flux rates after irrigation result largely from denitrification.

The diurnal pattern of N_2O fluxes after the combined fertilization and irrigation events showed that N_2O emissions are also partly temperature limited (Figure 5.5). Diurnal patterns of N_2O emissions driven by changes in topsoil temperature were also observed in other studies where the mineral N content as well as soil moisture conditions favored microbial N_2O production by either coupled nitrification-denitrification or denitrification (Dittert et al. 2005).

When the fields were irrigated without preceding fertilization, the N_2O emissions stayed generally low (<50 µg N_2O-N m^{-2} h^{-1}), which demonstrated that under these conditions mineral N availability was the limiting factor and not the soil water content. This implies that at this stage most of the previously applied fertilizer was already taken up by plants, leached with the irrigation water, or lost to the atmosphere. Unfortunately, due to soil laboratory analytical problems, we do not have information on the soil mineral N content during the growing season to corroborate this.

At our observation sites, averaged daily flux rates ranged up to 3000 µg N_2O-N m^{-2} h^{-1} and cumulative N_2O emissions from the different observation sites varied from 0.9 to 6.5 kg N_2O-N ha^{-1} over the entire vegetation period. These values would rank within the medium to higher range of previously reported N_2O emissions from irrigated agriculture (e.g., Matson et al. 1998; Hao et al. 2001; Pathak et al. 2002). Matson et al. (1998) found flux rates up to up to 6000 µg N_2O-N m^{-2} h^{-1} and large N_2O emissions (~ 6 kg N_2O-N ha^{-1}) for an irrigated wheat production system in Mexico.

The high variation encountered in cumulative emissions between the different sites arose mainly as a consequence of different management practices. The different practices applied by the local farmers resulted in variable fertilizer and water application rates. The magnitude of emission pulse increased with fertilizer and water input levels, as observed at the ATG site in late June and early July (Figure 5.1).

We assume that the sharp interannual contrasts between the observed fluxes in 2005 and 2006 at the ATG site were caused not only by the different fertilizer application rates but in particular by the use of a different N fertilizer type. For example, only combined ammonium fertilizer (ammonium sulphate and ammonium phosphate) together with urea was applied in 2006, whereas pure ammonium nitrate was applied in 2005. In the study region, irrigation water is commonly retrieved from the river and applied in a rather inefficient way through furrow irrigation, allowing few irrigation events but with high doses of irrigation water. Subsequently, the cotton fields are completely flooded and high soil moisture contents can be observed for several days. We propose that under these soil conditions, denitrification is highly elevated and comprises the main source of N_2O. Consequently, the availability of nitrate during an F+I event should be a main factor controlling N_2O emissions.

This assumption is corroborated by a simultaneous study in the same field on fertilizer use efficiency using ^{15}N-labeled urea and ammonium nitrate. Highest soil ^{15}N was recovered from the Urea-only fertilizer regime at any time during the growing season, proving urea to be the more immobile than NO_3^- fertilizer under irrigated conditions (Kienzler pers. comm.). These findings, along with previous conclusions from McTaggert et al. (1994) and Velthof et al. (1996) for fertilized grassland and Li et al. (2005b) for legume plantations in a temperate region, indicate that the use of NH_4^+ fertilizer instead of NO_3^- fertilizer may reduce N_2O emission and denitrification losses during the wet soil conditions following an F+I event.

Not all our hypotheses were supported by the data. It remains unclear what caused the significant difference in N_2O fluxes between the HI and the LI plots of the URDU site during the first two months after sowing, since there was no difference in irrigation and fertilization until the 26th of June. Moreover, we observed a significant increase in emission rates at the HI plot at the end of May after the measuring frames had been removed and installed at different locations of the plots. This was most likely caused by a high spatial variability of flux rates within the plot (Figure 5.3). Significantly higher emission rates were detected in the HI plots than the LI plots, which confirm the importance of soil moisture. But a time series analysis revealed that the differences were largely caused by the extremely high emission pulse in the HI plot on one single day (June 21). We also observed a high spatial variability among the different measurements, owing most likely to natural soil heterogeneity rather than differences caused by different irrigation and tillage. Therefore, the influence of the different irrigation intensities on flux rates is difficult to assess. We propose that the soil heterogeneity and spatial variations in the soil N content due to N from previously applied fertilizer shrouded the influence of the different irrigation intensities at this experimental site.

Treatment-induced N_2O-N losses ranged from 0.4 to 2.6% of total N applied, corresponding to an average EF of 1.48% (Figure 5.4). The measured value agreed well with the IPCC value of 1.25% of applied N used for N_2O emissions inventory reporting purposes, and shows that the simple IPCC method for calculating N_2O emissions is appropriate and effective (IPCC 2000a). But the high EF variability at the different observation sites showed that there were large differences in the relative and absolute emissions from irrigated cotton owing to soil properties and agricultural management practices. In this study, the EF differences were mainly caused by fertilizer type and its application time and amount, as well as the irrigation frequency, its timing and amount, and the management thereof. But the differences between similarly managed sites and the high inherent spatial variability of the different sites showed that more detailed, comparative studies in irrigated cotton are required to produce a robust mean EF.

5.4.1 Spatial vs. temporal variability and sampling strategy

Various studies on N_2O emissions from croplands showed that emissions varied significantly in space and time (Veldkamp and Keller 1997). In this study we observed an inherent spatial variability in the four sites, with an average coefficient of variation (CV) of 87%. However, the observed temporal variation, with an average CV of 342%, was much higher than the spatial variation. This shows that the sampling frequency was more important than increasing the number of spatial replicates.

In this study, 80-95% of the total N_2O emissions occurred in pulses after concomitant fertilization and irrigation. A weekly sampling frequency (Simojoki and Jaakkola 2000; Xiong et al. 2006) was thus not suitable to reproduce the temporal variability of these pulses and would therefore over or underestimate the weekly emission rates under the prevailing arid climatic conditions. However, a representative sampling strategy is key for estimating annual N_2O flux rates. Consequently, we postulate that under irrigated conditions in arid regions, a daily sampling frequency during five consecutive days directly following fertilization and irrigation is necessary to capture the dynamics of the emission pulse. Between the June and July emission pulses, a sampling frequency of at least three times per week is recommended. For the periods with generally low emission rates, such as in April and June before the first irrigation, as well as in August and September, a sampling frequency of once per week is necessary.

The diurnal emission patterns showed that daily point measurements cannot represent the N_2O daily flux rates. Especially for the emission pulses after F+I events, the dependency of the flux rates on soil temperature must be accounted for (Figure 5.5). Therefore, we suggest taking two samples at 8:00 to 10:00 and/or from 18:00 to 20:00 to best represent the daily flux. This agrees with the work of Parkin and Kaspar (2003) who observed similar sampling times as representative of the temperature dependent CO_2 daily flux for a no-till corn-soybean system in the USA. Sherlock et al. (2002) found point measurements at 12:00 to best represent a daily flux in a New Zealand pasture. However, this sampling time was based only on two point measurements (10:00 and 16:00), thus only representing a portion of the whole day and neglecting fluxes during the night. Other studies did not observe a correlation between diurnal patterns of soil temperature and N_2O flux from no-till irrigated corn and manure amended soil under maize in North America (Lessard et al. 1996; Ginting and Eghball 2005). We presume that the difference between our result and those from other studies was due to different field conditions such as WFPS and/or the availability of mineral nitrogen. We only observed a soil temperature impact on the N_2O flux when WFPS and mineral nitrogen content were not limiting.

5.4.2 Mitigating options of N_2O emissions

Because nitrous oxide emissions primarily were controlled by fertilization and irrigation, we can postulate that management practice could be modified to mitigate emissions of nitrous oxide and sustain higher fertilizer use efficiency. Across all our field experiments, 80-95% of the total N_2O was emitted directly after a combined irrigation-fertilization event. Therefore, mitigating options must focus on the management options that will affect directly the magnitude and length of these post-treatment pulses and if possible concomitant fertilization irrigation should be avoided.

5.5 Conclusion

Two years N_2O emissions measurements from irrigated cotton fields in arid Uzbekistan demonstrated that these fields are a significant source of N_2O emissions.

Given the high impact of fertilizer and irrigation water management on N_2O emissions, there is wide scope for mitigating N_2O emissions and denitrification losses. Concomitant N fertilization and irrigation should be avoided as much as possible and, as our findings of temperature dependent N_2O fluxes suggested, conducting F+I at cool weather may reduce N_2O emissions. Our results support the hypothesis that replacing NO_3^- fertilizer with NH_4^+ fertilizer could be an option to lower N_2O emission pulses, especially in combination with nitrification inhibitors.

However, feasible mitigating options will always rely on win-win opportunities when emissions can be reduced with a concomitant financial benefit for the farmers. Improved irrigation and optimized fertilizer management could be an alternative to reduce both N_2O emissions and financial losses to farmers without affecting yield and crop quality. In general, optimization of the fertilizer use efficiency should also result in mitigation of N_2O fluxes. Hence management practices that increase the fertilizer use efficiency in irrigated systems, such as sub-surface fertilizer application, fertigation and drip irrigation (e.g., Thompson et al. 2000; Ibragimov et al. 2007), will most likely also reduce the N_2O emissions.

For further evaluation of mitigation strategies, a better understanding of the regulating factors of N_2O fluxes from irrigated dryland agriculture is needed. This can only be achieved by fully-automated N_2O flux measurements, allowing continuous subdaily temporal resolution and, additionally, continuous monitoring of physical and chemical soil parameters.

6 METHANE AND NITROUS OXIDE FLUXES IN ANNUAL AND PERENNIAL LAND-USE SYSTEMS

6.1 Introduction

Irrigated agriculture is crucial to the world's food supplies; although only 17% of the global arable land is irrigated, it provides 40% of the world's food production (FAO 2000). Moreover, the role of irrigation is expected to grow in significance. For example, the (FAO 2002) predicted that all developing countries will need to expand their irrigated area from 202 million ha in 1999 to 242 million ha by 2030 to meet their food demands. The impacts of irrigation, however, go far beyond the issue of food security, since irrigation affects virtually all biogeochemical cycles at the field and landscape levels. Apart from the direct interference in natural water cycles, irrigation also accelerates microbial C and N turnover in the soil (e.g., Andren et al. 1992; Davidson 1992). Since irrigation-based agriculture in drylands often is associated with tillage and heavy inputs of mineral fertilizer, a strong increase of N_2O fluxes is caused by the combined use of irrigation and N fertilization (e.g., Mahmood et al. 1998a; Aulakh et al. 2001; Scheer et al. 2008). In addition, irrigated lowland rice cultivation is now acknowledged as a major source of atmospheric methane (CH_4).

N_2O and CH_4 are important to the radiative balance of the earth owing to their long atmospheric life times (~10 yr for CH_4 and 120 yr for N_2O) and infrared absorption properties in the troposphere (Mosier et al. 2004). Next to CO_2, N_2O and CH_4 are the most important greenhouse gases (GHG), accounting for approximately 20% of global warming (Rohde 1990). In the soil, N_2O originates from inorganic N (ammonia and nitrate) during microbial nitrification and denitrification (Firestone and Davidson 1989b; Hutchinson and Davidson 1993), while CH_4 is produced by microorganisms that rely on the highly anaerobic conditions typically found in flooded rice fields (Conrad 1996). More than 50% of the global source strengths of N_2O and CH_4 are attributed to soil-atmosphere exchange. Hence, changes in land-use, and thus agricultural practices, infer major changes in the atmospheric budgets of these gases (IPCC 2000b).

It is crucial to understand the extent to which various agroecosystems contribute to GHG budgets, but, gas flux records from irrigated agricultural systems are limited except for rice. This highlights the need for field measurements in irrigated agriculture in drylands dominated by cotton (Rochester 2003), winter wheat (Majumdar et al. 2002), and rice (Aulakh et al. 2001) since these crops rank at the top within irrigated land-use systems. Given the 260 million ha of irrigated agriculture worldwide, and in particular the 8 million ha in the five central Asian countries that cover most of the Aral Sea Basin (FAOSTAT 2007), it seems highly relevant to improve the estimate of N_2O and CH_4 emissions from this important agroecosystem.

The Aral Sea Basin is located in Central Asia and covers an area of about 1.9 million km^2 in the former Soviet Republics of Uzbekistan, Kazakhstan, Turkmenistan, Kyrgyzstan and Tajikistan and the northern part of Afghanistan. During the Soviet era, the region served as the main supplier of cotton, and agriculture in Uzbekistan was, therefore, almost totally dedicated to cotton production. To achieve this in a region covered to 75% by deserts, the area under irrigation in the Aral Sea basin was increased from 2.0 to 7.2 million ha between 1925 and 1985. This has dramatically reduced the river discharge to the Aral Sea, resulting in a loss of more than 80 % of its volume and 70 % of its surface area over the last decades (Micklin 2007). This demise has led to the 'Aral Sea Crisis', a complex combination of ecological consequences with regional and global dimensions (Micklin 2007).

Currently, a dominant cropping system in Uzbekistan is cotton-wheat-rice rotation with heavy inputs of water and fertilizers. Forests naturally occupy a rather small proportion of the landscape (e.g. the 30,000 ha of *Tugai* floodplain forests along the Amu Darya River but there are tree plantations and shelterbelts (about 2.3 % of the irrigated land; Tupitsa, forthcoming)), and recent research findings underscore the use of trees for 'phytoremediation' of marginal agricultural land (Lamers et al. 2006). Moreover, the importance of trees for satisfying future domestic energy demands may increase (FAO 2006) although whether irrigated lands are suitable remains to be seen.

As the importance of GHG emission levels caused by these agricultural systems has largely been ignored, the aims of this study were to (i) assess the impact of the prevailing cropping systems in the Aral Sea basin such as cotton, winter wheat and rice, on the emissions of N_2O and CH_4; (ii) compare these to fluxes of N_2O and CH_4 from perennial cropping systems, especially forest plantations and natural floodplain forests; (iii) evaluate the global warming potential (GWP) of N_2O and CH_4 fluxes from the entire irrigated dryland agriculture in the study region.

6.2 Material and methods

6.2.1 Description of the study region

This study was carried out on research sites of a development project , conducted by the German Center of Development Research (ZEF) and UNESCO in the Khorezm Region, Uzbekistan, between April 2005 and October 2006. The climate is typically arid continental with long, hot, dry summers and very cold temperatures during winter (Figure 4.3). Annual average precipitation for the period from 1980 to 2000 was 97 mm and the mean annual temperature was 13.0 ºC (Glavigdromet 2003).

Table 6.1: Main characteristics of the different research sites (all soil variables are given ± standard error for 0-20 cm soil depth).

Study Site	Amir Temur Garden (ATG)	Amir Temur Cum (ATC)	Urdu Research Site (Urdu)	Poplar Plantation (PP)	Tugai Riparian Forest (TF)
Soil Type	Calcaric gleyic Arenosol	Calcaric gleyic Arenosol	Calcaric gleyic Arenosol	Gleyic Solonchak	Gleyic Fluvisol
pH(H_2O)	6.9 ± 0.3	6.5 ± 0.5	6.6 ± 0.3	6.7 ± 0.4	7.4 ± 0.1
Bulk density (g cm^{-3})	1.59 ± 0.03	1.51 ± 0.05	1.41 ± 0.02	1.44 ± 0.03	1.31± 0.04
Soil organic carbon (%)	0.61 ± 0.03	0.58± 0.05	0.53± 0.02	0.60± 0.01	0.40 ± 0.03
N (%)	0.10± 0.005	0.10± 0.01	0.07± 0.004	0.10± 0.002	0.07± 0.004
Texture (USDA)	Sandy loam	Sandy loam	Silt loam	Sandy loam	Loamy sand
Clay (%)	14.6	15.5	17.9	13.1	4.7
Silt (%)	42.8	32.1	55.4	32.9	24.1
Sand (%)	42.6	52.4	26.7	55.1	71.2

The annual land-use systems

Cropping systems on the region's most dominant soil textures were selected for the flux measurements. Two sites were part of the "Amir Temur" Farmer Association close to the research station. Cotton and winter wheat flux rates were measured in 2005 and 2006 on experimental fields of the Amir Temur Garden farm (41º36'N, 60º31'E), and cotton (2005) and rice (2005 and 2006) flux rates were measured in the Amir Temur Cum farm (41º38'N, 60º25'E). The soils were classified as calcaric gleyic Arenosols (FAO 1998) with a loamy, respectively silty loamy texture. The land had previously been cultivated by a cotton-wheat-rice crop rotation. The third site was situated on the campus of the Urgench State University (41º33'N, 60º37'E), located on a calcaric gleyic Arenosol (FAO 1998) with a sandy loam soil texture. The main characteristics of the different research sites are shown in Table 6.1.

Table 6.2: Experimental set up of agricultural fields (AN = Ammonium Nitrate, AS= Ammonium Sulfate, AP = Ammonium Phosphate; U = Urea).

Land-use System	Research Site	Observation period	Planting - Harvest	Irrigation	Fertilization ($kgha^{-1}$), Type
Winter Wheat (HI)	ATG	Oct 2005- Jun 2006	20/10/2005 21/06/2006	High (HI) 900mm	192, AN
Winter Wheat (LI)	ATG	Oct 2005- Jun 2006	20/10/2005 21/06/2006	Low (LI) 800mm	192, AN
Rice	ATC	May-Oct 2005	21/05/2005 15/09/2005	Continuously flooded	270, AS, AN, U
Rice	ATC	May-Sep 2006	24/05/2006 11/09/2006	Continuously flooded	200, AS, AN, U
Cotton	ATG	Apr-Oct 2005	21/04/2005 Sep/Oct 2005	Farmers practice ~300mm	250, AN
Cotton	ATG	Apr-Sep 2006	21/04/2006 Sep/Oct 2006	Farmers practice ~400mm	250, AS, AP, U
Cotton	ATC	May-Oct 2005	15/04/2005 Sep/Oct 2005	Farmers practice ~400mm	162.5, AN
Cotton (HI)	URDU	Apr-Sep 2006	22/04/2006 Sep/Oct 2006	High (HI) 463mm	250, AN
Cotton (LI)	URDU	Apr-Sep 2006	22/04/2006 Sep/Oct 2006	Low (LI) 373mm	250, AN

These farmer-managed sites were run under commonly practiced fertilizer and irrigation strategies, which allowed assessing the impact of conventional land management on N_2O and CH_4 fluxes. For comparison, cotton and winter wheat flux rates were measured in 2006 within an experiment on irrigation management. This experiment included two types of irrigation management practices: (i) high intensity irrigation (HI) meaning after the first irrigation the next irrigation took place when the soil moisture level was 75% of field capacity, and (ii) low intensity irrigation (LI): after the first irrigation the next irrigation occurred when the soil moisture decreased to 65% of field capacity. This allowed analyzing the effect of these two soil moisture regimes on N_2O emissions. The HI field was characterized by higher soil moisture content and more frequent irrigation applications.

The fertilizers consisted of mineral fertilizers only (urea, ammonium phosphate, ammonium nitrate), with fertilizer rates ranging from 160 to 270 kg-N ha^{-1}.

More details on the experimental setup of the different land-use systems are summarized in Table 6.2.

Description of the perennial land-use systems

Poplar plantation: In 2005 and 2006 flux measurements were conducted in a forest plantation (41º43'N, 60º35'E) on a gleyic solonchak soil (FAO 1998) with a sandy loam texture. The entire plantation covered 75 ha, including 70 ha of 10-year old *Populus nigra* L. and 5 ha of 6-year *Ulmus pumila* L. trees. Trees had been planted at a density of 3,300 trees ha^{-1}. Measurements were conducted particularly in the *Populus nigra* L. section of the plantation. At the onset of the experiment in 2005, occasional irrigation events occurred, but due to the shallow groundwater table of 1-2 m below ground surface, the site provided enough water for the trees over the entire summer, without any additional irrigation. The site was furthermore characterized by a slight to medium soil salinity (Khamzina et al. 2005).

Tugai Forest: Flux measurements were conducted in 2005 in the Tugai forest reserve located in the Amu Darya floodplain North of Khorezm (41º58'N, 60º24'E). The reserve spans 6,400 ha and was set aside in 1971 (UNDP/GEF 1998). The forest in the reserve has drastically suffered from decreased natural flood frequency caused by the diversion of the Amu Darya water into irrigation, and by deforestation.

Tugai vegetation is characterised by high tolerance to both very wet soil and very dry air, resistance to drought and salts, and high transpiration intensity (Treshkin et al. 1998). With more than 230 plant species, the Tugai forest is considered to be one of the most diverse vegetation types of arid regions in Central Asia (Novikova et al. 2001). It is estimated that 90% of the original 300,000 ha Tugai forest along the lower reach of the Amu Darya disappeared during the last century (Kuzmina and Treshkin 1997; Novikova et al. 1998; Treshkin 2001).

Flux measurements were conducted beside the Amu Darya River within the forest Tugai vegetation. In the experimental area, *Populus ariana* L. dominated the woody species next to *Populus pruinosa* L., *Salix sp.* and *Elaeagnus angustifolia* L..

6.2.2 Gas flux measurements

Nitrous oxide and methane fluxes were monitored by using the closed chamber technique as described in Mosier (1989). At each research site, four metal sampling frames were pressed into the soil surface and remained throughout the observation cycle in the field as footing for the chambers during sampling periods. The frames were randomly distributed over the field, at a maximum distance of 5 m between two adjacent frames, which eased the sampling procedure.

The 1.6 m high chambers installed in the rice fields had a basal area of 1 m^2, the chambers used for the cotton and wheat measurements were 0.4 m high and covered 0.21 m^2 of soil, while the chambers used in the Poplar plantation and the Tugai forest were 0.15 m high and covered 0.02 m^2. The frames were temporarily removed from the plots during tillage operations only. At the beginning of each sampling period, the chamber was fixed to the frame gas-tight and fluxes were measured after extracting air samples from the chamber head space. Precisely 20 ml of headspace air was drawn through a septum into a gas-tight 20 ml polypropylene syringe at 0, 10, 20, and 30 min after the installation of the chamber. The syringes were closed with a Luer Lock valve immediately after air sampling to prevent gas exchange. On the same day, air samples were analyzed for N_2O and CH_4 concentrations using a Shimadzu G 14A Gas Chromatograph (Shimadzu, Kyoto, Japan). For special characteristics of the GC system, please refer to Chapter 5.2.2. Before and after every four injections of chamber air samples, a reference gas was injected using a gas mixture containing 397 ppb N_2O and 3.88 ppm CH_4 in synthetic air (Messer Griesheim, Munich, Germany).

N_2O and CH_4 emissions were calculated from the linear increase of the gas concentration at each sampling time (0, 10, 20 and 30 min during the time of chamber closure) as described in detail in Chapter 5.2.2. All flux rates were corrected for temperature and air pressure. The correlation coefficient (R^2) for the linear regression was calculated and used as a quality check for the measurement. For $R^2 < 0.9$ ($R^2 < 0.7$ for small flux rates) the measurement was rejected.

Gas fluxes were measured at four replicated plots within each experimental site each week throughout the crop growing season and each month in the poplar plantation during winter. At the cropping sites, measurements were conducted three to four times a week immediately following fertilizer and/or water applications. Fluxes were measured twice each sampling day in 2005 (between 8:00h and 11:00h in the morning and 13:00h and 15:00h in the afternoon) and once on each sampling day in 2006 (between 8:00h and 11:00h). In order to estimate the accumulated seasonal emissions of the investigated land-use systems flux rates for days with no measurements were extrapolated by calculating a daily mean value based on the previous and successive sampling date.

6.2.3 Global warming potential (GWP)

The concept of GWP allows assessing the radiative forcing of different GHG's relative to the reference gas, in this case CO_2, over a specific time horizon. The study only refers to the GWP of CH_4 and N_2O emissions from the investigated agroecosystems. Soil-borne CO_2 emissions – as well as off-site emissions stemming from the energy required for irrigation, farm operations and fertilizer production – were not taken into account for the calculation of the GWP. Based on a 100-year time frame the GWP of CH_4 and N_2O are, respectively, 23 and 296 times higher than that of CO_2 (IPCC 2001). The net GWP is expressed in kilograms of carbon dioxide equivalents per hectare per day of the investigated land-use systems. The GWP concept allows comparing N_2O and CH_4 fluxes and assessing the relative contribution of these two gases on a reference scale and was calculated according to following equations (6.1) and (6.2):

$$GWP(N_2O)\left[\frac{kgCO_2eq}{ha \cdot day}\right] = x_1 \cdot \frac{\mu gN_2O-N}{m^2 \cdot h} \cdot \frac{44\mu gN_2O}{28\mu gN_2O-N} \cdot \frac{10^4 m^2}{1ha} \cdot \frac{24h}{1day} \cdot \frac{1kg}{10^9 \mu g} \frac{296kgCO_2}{1kgN_2O} \tag{6.1}$$

$$GWP(CH_4)\left[\frac{kgCO_2eq}{ha \cdot day}\right] = x_2 \cdot \frac{mgCH_4}{m^2 \cdot day} \cdot \frac{10^4 m^2}{1ha} \cdot \frac{1kg}{10^6 mg} \frac{23kgCO_2}{1kgCH_4} \tag{6.2}$$

where x_1 = average daily N_2O-N emission rate (µg N_2O-N $m^{-2}h^{-1}$)

x_2 = average daily CH_4 emission rate (mg CH_4 $m^{-2}d^{-1}$)

6.2.4 Soil data

Soil temperature at 10 cm depth was measured at each retrieval date with a hand held digital thermometer. The mean of initial and final air temperature was used in the gas flux calculations. Daily minimum and maximum air temperature and precipitation data were taken from the meteorological station at the research station in Khiva. Soil moisture at each sampling time was determined from bulk samples retrieved from the upper soil layer (0-10 cm depth). The samples were cooled during transportation and processed the same day; soil moisture was gravimetrically determined after drying for 24 h at 105º C or until constant weight. Water-filled pore space (WFPS) was calculated from soil bulk density (arithmetic means of four samples) using a particle density of 2.65 g cm^{-3}.

In addition, at the beginning and at the end of the growing season, bulk soil samples were taken from each site by combining 5-10 soil cores (0-10cm depth). The samples were air-dried in the shadow for 6 days, sent to the Tashkent Soil Science Institute, Uzbekistan, where they were analyzed for soil texture, total carbon (C %), total nitrogen (N%), mineral nitrogen content (NO_3^-, NH_4^+) and bulk density (see Table 6.1).

6.2.5 Statistical analyses

For analyzing the normal distribution of the data the Kolmogorov-Smirnov test was performed. As N and C trace gas fluxes were non-normally distributed, the non-parametric Mann-Whitney test was performed. Statistical analyses were calculated using SPSS 8.0 (SPSS Inc., Chicago, USA). Data in Figure 6.1 to Figure 6.5 are presented as means ± 1 standard error.

6.3 Results

6.3.1 Selected annual land-use systems

Rice: CH_4 flux was only observed in the flooded rice plots. There was no measurable CH_4 flux above the detection limit of 50 CH_4 $m^{-2}h^{-1}$ from the cotton and winter wheat fields (Table 6.3). The seasonal patterns of the CH_4 and N_2O emissions rates from rice over the two year measurement period are shown in Figure 6.1.

In both 2005 and 2006, CH_4 emissions rapidly increased during the first 40days after planting. In 2005 the flux rates then remained on a level between 50 and 90 mg CH_4 $m^{-2}d^{-1}$ while in 2006 the flux rates decreased again and stayed generally below 10 mg CH_4 $m^{-2}d^{-1}$ in August and September. The mean seasonal flux rates were 48 ± 19 mg CH_4 $m^{-2}d^{-1}$ in 2005 and 16± 9 mg CH_4 $m^{-2}d^{-1}$ in 2006; the cumulated flux amounted to 52.9 kg CH_4 ha^{-1}in 2005 and 17.5 kg CH_4 ha^{-1}in 2006, respectively.

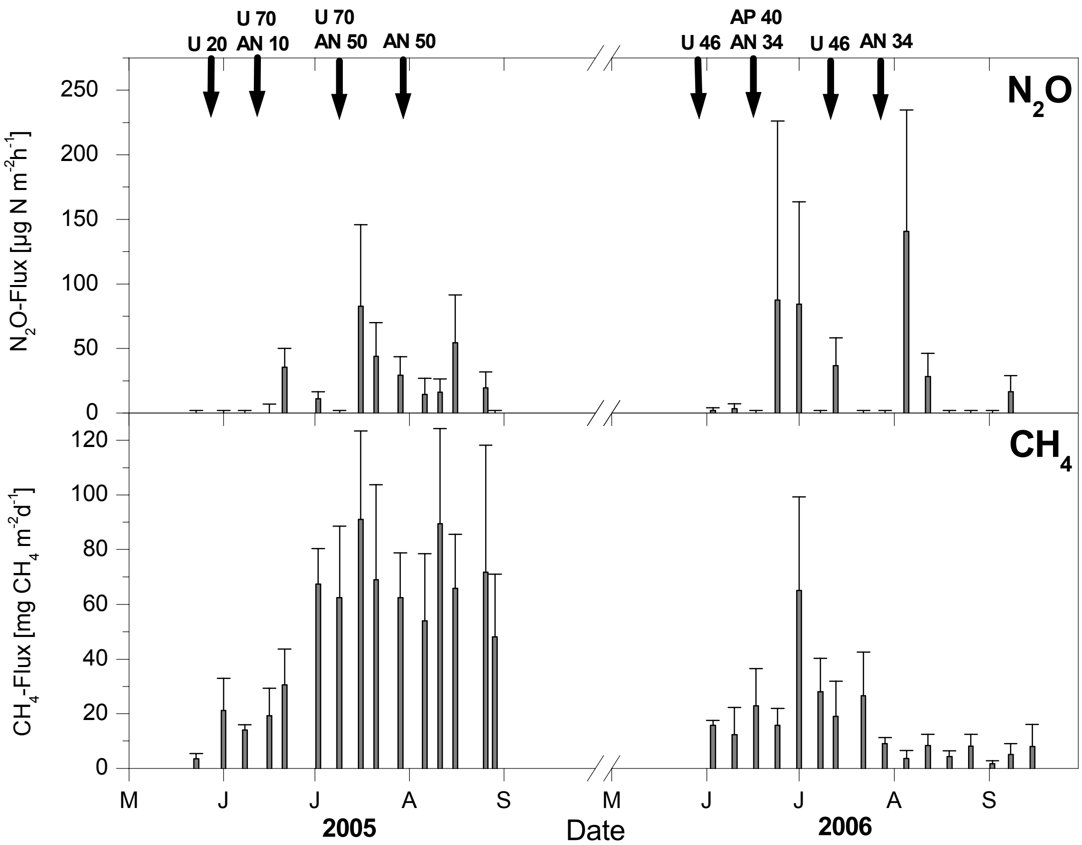

Figure 6.1: CH_4 and N_2O flux rates of the rice plot from May-Sept. 2005 and 2006. Arrows indicate the events of N (kg N ha^{-1}) application to the plots (U = Urea, AN = Ammonium Nitrate, AP = Ammonium Phosphate). Error bars indicate the standard error.

N_2O emissions from the flooded rice field were generally very low (<10 µg N_2O-N $m^{-2}h^{-1}$) except for the high peaks that occurred following the application of ammonium nitrate fertilizer. These emissions peaks reached values of more than 80 µg N_2O-N $m^{-2}h^{-1}$. Average flux rates and seasonal values are shown in Table 6.3.

Cotton and winter wheat: The temporal courses of N_2O emissions in the cotton and the winter wheat fields were very similar; the combined fertilization and irrigation events caused high emission levels in the early season which were followed by lower flux rates in the intermediate periods. The temporal course of N_2O emissions of winter wheat is displayed in Figure 6.2; the temporal course of N_2O emissions of several cotton sites is shown in Chapter 5.3. This emission pattern could be observed at all experimental sites of cotton and winter wheat. The emission pulses reached averaged daily values of up to 360 µg N_2O-N m^{-2} h^{-1}at the winter wheat sites and up to 2300 μg N_2O-N m^{-2} h^{-1} at the cotton sites. When accumulated, these peak emissions accounted for 80-95% of the total N_2O emissions over the entire cropping season for both crops. During periods of low soil moisture (< 50% WFPS) only reduced N_2O emissions were detected, even following fertilization. Similarly, irrigation events during the cropping cycle without concomitant N-fertilization did also not show elevated N_2O fluxes. The dynamics and regulating parameters of these fertilizer/irrigation induced pulses for cotton have been described in detail in Chapter 5.3 and 5.4.

Table 6.3: Average, seasonal N_2O emission, emission factor and seasonal CH_4 emissions (± standard error) of the different research sites over the April 2005- September 2006 measurement period. (bd = below detection limit).

	Research Site	Year	Average N_2O flux *[μg N m^{-2} h^{-1}]*	Seasonal N_2O Emission *[kg N_2O-N/ha]*	Emission Factor *[% of applied fertilizer]*	Average CH_4 flux *[mg CH_4 $m^{-2}d^{-1}$]*	Seasonal CH_4 Emission *[kg CH_4 /ha]*
Winter Wheat (HI)	ATG	2005/ 2006	15.7 ± 2.1	0.9 ± 0.1	0.5	bd	bd
Winter Wheat (LI)	ATG	2005/ 2006	10.0 ± 1.9	0.6 ± 0.1	0.3	bd	bd
Rice	ATC	2005	19.2 ± 13.3	0.5 ± 0.3	0.2	48.1 ± 19.4	52.9 ± 21.4
Rice	ATC	2006	25.0 ± 23.9	0.7 ± 0.7	0.3	15.8 ± 8.6	17.5 ± 9.8
Cotton	ATG	2005	149.8 ± 41.6	6.5 ± 1.8	2.6	bd	bd
Cotton	ATG	2006	20.6 ± 7.3	0.9 ± 0.3	0.5	bd	bd
Cotton	ATC	2005	67.0 ± 16.9	2.9 ± 0.7	1.8	bd	bd
Cotton (HI)	URDU	2006	102.2 ± 42.7	4.4 ± 1.8	1.8	bd	bd
Cotton (LI)	URDU	2006	55.7 ± 18.5	2.4 ± 0.8	1.0	bd	bd
Poplar plantation	PP	2005/ 2006	30.0 ± 14.7	2.6 ± 1.3	-	bd	bd
Tugai forest	TF	2005	1.4 ± 0.4	0.1 ± 0.03	-	bd	bd

Average N_2O fluxes of the different cropping sites ranged from 10.0 μg N_2O-N $m^{-2}h^{-1}$ at the 'low intensity' winter wheat site to 149.6 μg N_2O-N $m^{-2}h^{-1}$ at the Amir Temur Garden cotton site in 2005. Cumulative seasonal N_2O emissions varied from 0.5 to 6.5 kg N_2O-N $ha^{-1}season^{-1}$, which corresponded to emission factors, uncorrected for background emission, of 0.2 to 2.6% of the total amount of mineral N applied (Table 6.3).

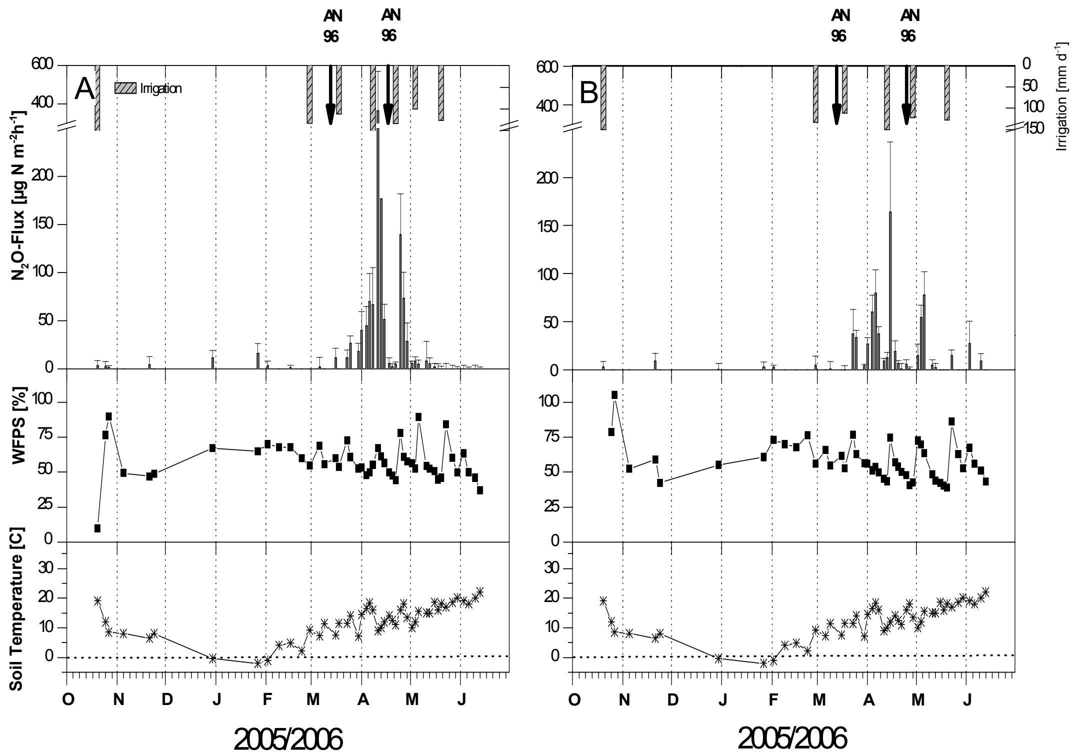

Figure 6.2: N_2O flux rates, irrigation rates, WFPS and soil temperature of the high (A) and low (B) irrigated winter wheat field at the Amir Temur Garden farm in 2005/2006. Arrows indicate the events of N (kg N ha^{-1}) application to the plots (AN = Ammonium Nitrate), whereas bars provide information about the amount of irrigation. Error bars indicate the standard error. Connecting lines are inserted for showing the data points more clearly.

6.3.2 Perennial land-use systems

Poplar plantation: No CH_4 fluxes, but substantial N_2O emissions were observed in the poplar plot over the whole measurement period (Figure 6.3). Due to a very shallow groundwater table 1-2 m below ground level throughout the season, the soil water content at this observation site remained constantly high, at approx. 80% of WFPS, even without the application of irrigation water. In 2005, the mean flux rate at the onset of the vegetation season in April was 20 µg N_2O-N $m^{-2}h^{-1}$. Throughout the observation period, emissions rose gradually, peaked at 75 µg N_2O-N$m^{-2}h^{-1}$ at the end of July and early August, and dropped to 20 µg N_2O-N $m^{-2}h^{-1}$ towards the end of the growing season in October 2005. In 2006, monthly measurements started in January and on January 13 and February 10 extraordinary high N_2O emissions occurred during the thawing of the upper soil which had been previously frozen. Throughout the observation period of 10 months after these events, flux rates remained below 50 µg N_2O-N $m^{-2}h^{-1}$ while varying a lot without any apparent or detectable reason.

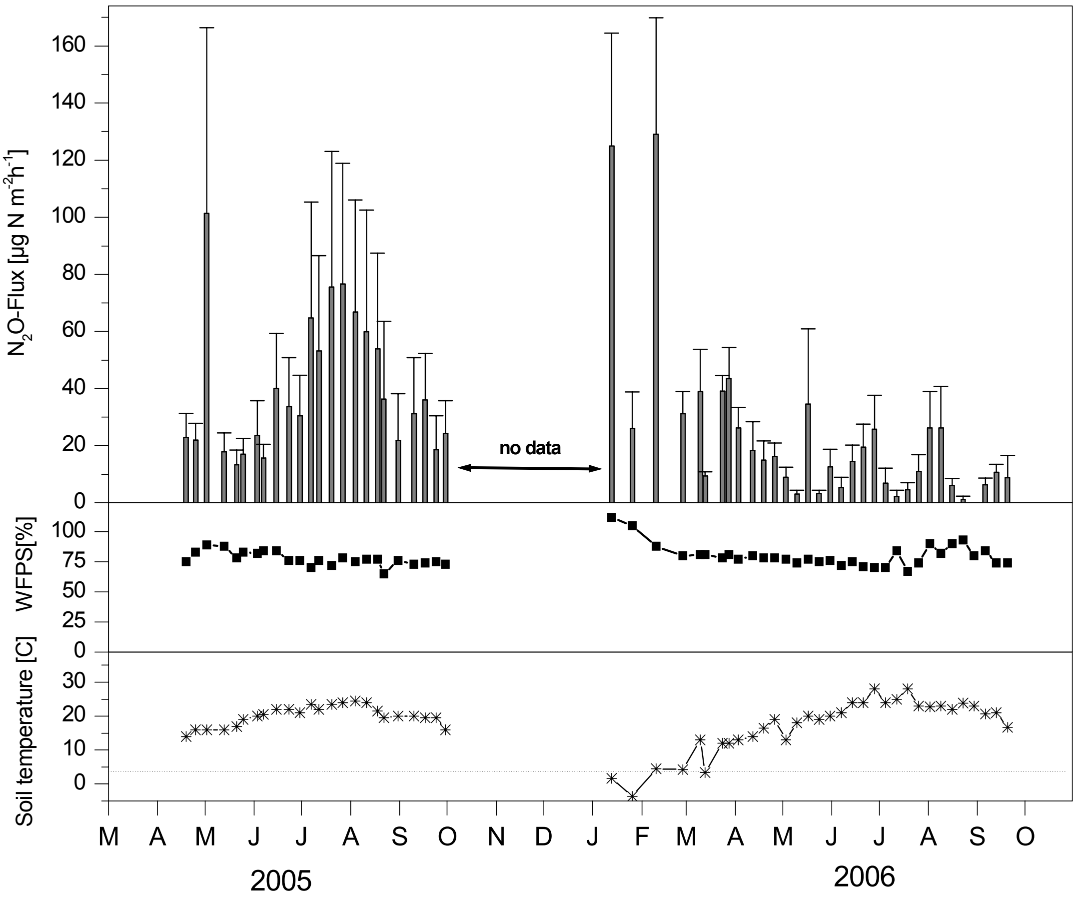

Figure 6.3: N_2O flux rates, Water-filled pore space (WFPS) and soil temperature of the poplar plantation plot over the April 2005- September 2006 measurement period. Error bars indicate the standard error. Connecting lines are inserted for showing the data points more clearly.

The 2006 summer months' emissions were lower compared to the same period in 2005. In 2005, a mean flux of 39.8 ± 18.2 µg N_2O-N $m^{-2}h^{-1}$ was observed during the summer months, compared to a mean flux of 22.9 ± 8.7 µg N_2O-N $m^{-2}h^{-1}$ in 2006, differences that were highly significant. The cumulative annual emission amounted to 3.5 kg N_2O-N ha^{-1} y^{-1} and 2.0 kg N_2O-N ha^{-1} y^{-1} in 2005 and 2006, respectively.

Tugai Forest: From May to September 2005 flux rates were measured at the Tugai forest plot at regular intervals. No natural flooding occurred in this riparian area during the observation period, and subsequently the soil water content of the top soil was very low (< 40% WFPS). No CH_4 fluxes and very low N_2O fluxes (< 2 µg N_2O-N

$m^{-2}h^{-1}$) were detected during the sampling effort. Maximum flux was measured in one chamber on May 12 and reached 17.6 µg N_2O-N $m^{-2}h^{-1}$ (Figure 6.4). The arithmetic mean of the N_2O flux for all measurements in 4 measuring chambers was 1.4 ± 0.75 µg N_2O-N $m^{-2}h^{-1}$, corresponding to a cumulative annual emission of 0.1 kg N_2O-N ha^{-1} y^{-1} (Table 6.3).

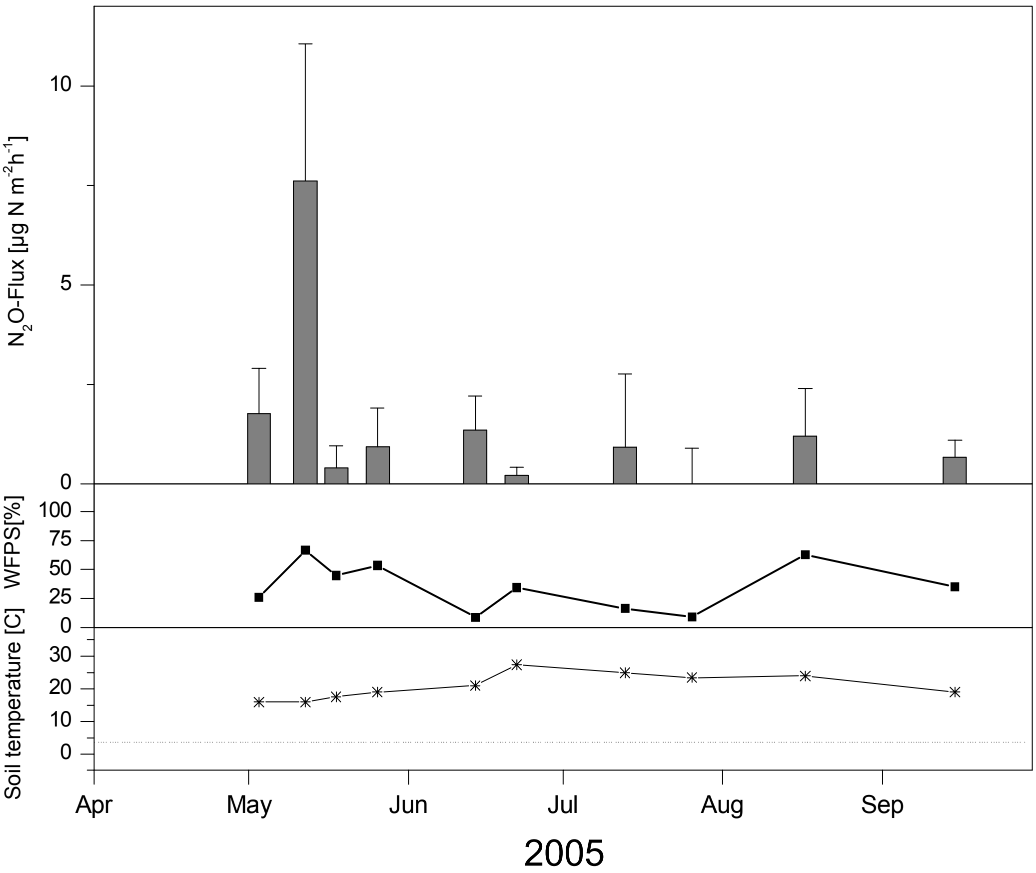

Figure 6.4: N_2O flux rates, Water-filled pore space (WFPS) and soil temperature of the Tugai forest plot over the April- September 2005 measurement period. Error bars indicate the standard error. Connecting lines are inserted for showing the data points more clearly.

6.3.3 Global warming potential (GWP) of the different land-use sites

N_2O and CH_4 emissions were converted into mean daily CO_2 flux rate equivalents to compare the different GHG emissions from the various sites and years (Figure 6.5). Highest GWP was found in the rice fields with an average daily flux of 10.1 kg-CO_2 eq.$ha^{-1}d^{-1}$, N_2O accounting for 20% and CH_4 for 80% of the total GWP. At the other land-use sites only N_2O was responsible for the GWP, since no fluxes of CH_4 were detected. In the fertilized cotton site, an average flux of 8.8 kg-CO_2 eq.$ha^{-1}d^{-1}$ was measured, whilst in the winter wheat fields an average of 1.4 kg-CO_2 eq.ha^{-1}d^{-1}was emitted as N_2O over the cropping cycle. Accordingly, a biennial crop rotation system of cotton-wheat-rice as it is commonly found in the study area, would amount to a flux of 6.8 kg-CO_2 eq.ha^{-1}d^{-1} with N_2O accounting for 60% and CH_4 for 40% of the total GWP. The unfertilized poplar plantation had a mean daily flux of 3.4 kg-CO_2 eq.ha^{-1}d^{-1}, while in the Tugai forest the GWP averaged only 0.2 kg-CO_2 eq.ha^{-1}d^{-1}.

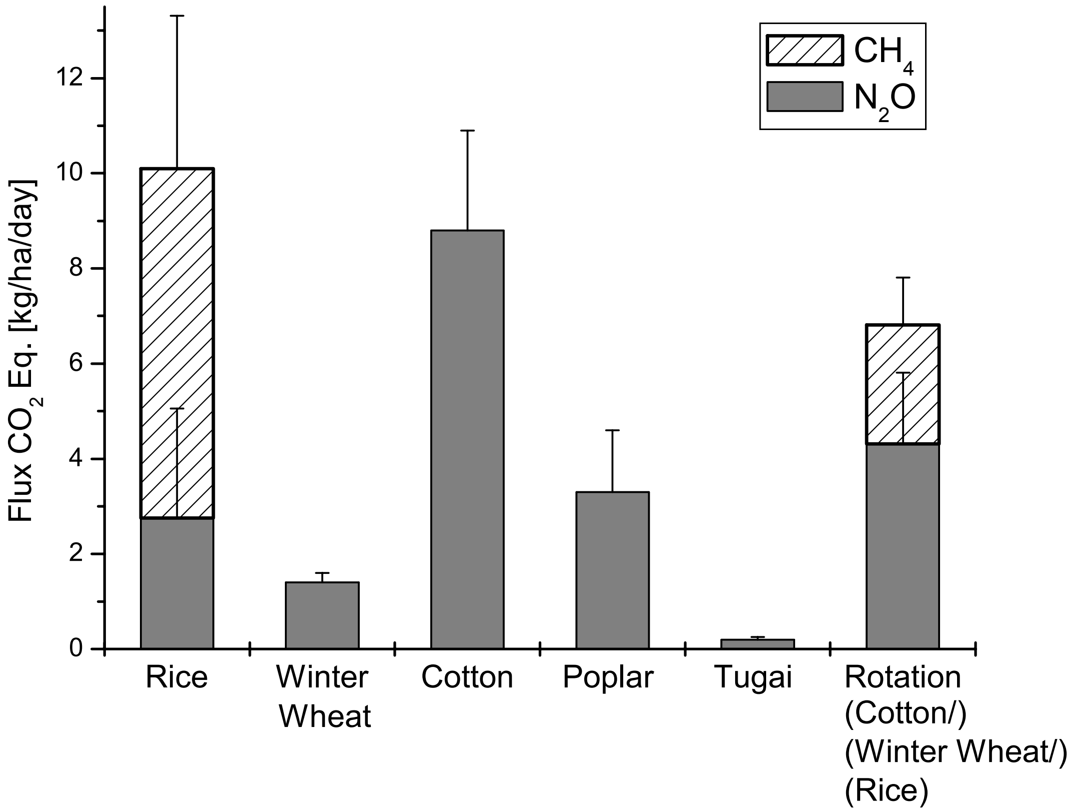

Figure 6.5: Mean daily flux rates in CO_2 equivalents of N_2O and CH_4 for the 6 different land-use and rotation systems.

6.4 Discussion

To our knowledge the reported N_2O and CH_4 fluxes represent the first detailed comparative analyses of soil-atmosphere trace gas exchange from irrigated agricultural systems in an arid environment. As expected, significant and sustained emissions of methane could only be measured in the permanently flooded rice land-use system, while in the cotton and winter wheat systems neither methane emissions nor soil methane deposition was detected. However, the sensitivity for CH_4 fluxes in our system (~50 µg CH_4 m^{-2}h^{-1}) was below the CH_4 uptake rate previously recorded in aerobic environments

(Robertson et al. 2000; Mosier et al. 2005). As a consequence, our data set only corroborates the high methane emissions from the anaerobic rice fields but does not allow a definite statement on methane uptake in the aerobic cropping and forest sites. However, even if we had detected methane fluxes with more sensitive equipment they would have been very low and hence insignificant within the net GWP budget of agricultural systems (Robertson et al. 2000).

6.4.1 Annual land-use systems

The observed methane emissions and the resulting seasonal flux were at the lower end of reported methane emissions from irrigated rice. For example, Wassmann et al. (2000b) reported seasonal fluxes up to 600 kg CH_4 $ha^{-1}season^{-1}$ from irrigated rice in tropical and subtropical Asia. One reason for the relatively low seasonal emissions in the arid regions of the Aral Sea Basin may have been the exclusive use of mineral fertilizers, in contrast with the use of organic amendments such as rice straw, which had been associated with enhanced emissions during the rice growing season (Setyanto et al. 2000). On the other hand, results from studies with management practices similar to those of Khorezm region (i.e. continuous flooding and no organic fertilizers) report emission rates up to 200 kg CH_4 $ha^{-1}season^{-1}$ (Wassmann et al. 2000a). Another reason for the low emission rates could have been the high percolation rates in the sandy-loamy soil of the research site, which consequently required frequent irrigation. It is acknowledged that such conditions result in a constant inflow of oxygen into the soil and reduce methane emissions from irrigated rice (Jain et al. 2000). However, since Oxygen fluxes were not measured, we would need further evidence to verify this explanation. Furthermore, we assume that the low emission rates were caused by the high salt content in the irrigation water (1-2 g l^{-1}) in Khorezm, which also has been reported to result in a strong inhibition of the methanogenesis and a decrease in the CH_4 emission rate (Holzapfel-Pschorn et al. 1985).

In contrast, the observed N_2O emission pulses of more than 100 µg N_2O-N $m^{-2}h^{-1}$ in the flooded rice fields were unexpectedly high; previously only very low N_2O emissions from flooded rice fields have been reported (Abao et al. 2000). These high fluxes were most likely caused by the use of ammonium nitrate fertilizer in Khorezm, whereas urea is the predominant fertilizer in other rice growing regions. Thus, effectively all previous field experiments measuring N_2O from rice encompassed urea treatments. In saturated rice soils, NO_3^- concentrations are immediately depleted by denitrification. Although anaerobic rice is capable of using both ammonium and nitrate forms of nitrogen, nitrate containing fertilizers are not recommended for flooded rice fields because of the very low fertilizer use efficiency due to high denitrification losses. This and the results from the irrigated cotton fields (Chapter 5) point to an exclusive use of urea and/or NH_4^+ fertilizers in the Aral Sea Basin for increased fertilizer use efficiency and mitigated N_2O emissions.

Emission patterns of N_2O were similar among cotton and winter wheat, underlining that N_2O emission in these land-use systems can be controlled by fertilization and irrigation practices. Emission pulses triggered by the combined activity of irrigation and fertilization events superseded any crop-specific or seasonal effect, which is in accordance with other studies on N_2O fluxes from irrigated agricultural ecosystems (e.g., Jambert et al. 1997a; Matson et al. 1998; Pathak et al. 2002; Majumdar et al. 2002). A strong influence of soil WFPS on the temporal variability of N_2O emissions has been described in many studies and denitrification was found as the principal source of N_2O at soil water contents above 60% WFPS (e.g. Keller and Reiners 1994; Zheng X. et al. 2000). This indicates that the emission pulses of N_2O following combined fertilization and irrigation events at our sites largely resulted from denitrification under waterlogged conditions.

The interaction between N fertilization and irrigation appeared as the predominant driver of N_2O production. This is substantiated by the very small N_2O emissions when N fertilizer was added to dry soil, or when irrigation occurred without a preceding N fertilization, which did not result in elevated N_2O emissions even at a later stage of the cropping cycle (Chapter 5). Hence, depending on the management, either mineral N availability or soil water content can become the limiting factor, and thus a controlling factor, for N_2O production.

The seasonal N_2O flux rates of cotton and winter wheat (0.6 - 6.5 kg N_2O-N ha^{-1} $season^{-1}$) and the corresponding emission factors (0.3 - 2.6%) are within the range of previous studies from irrigated cropping systems. Matson et al. (1998) reported flux rates from 0.7 to 6.9 kg N_2O-N $ha^{-1}season^{-1}$ and emission factors ranging from 0.4 to 2.8% for an irrigated wheat production system in Mexico. In the irrigated wheat systems of the semiarid subtropical Indo-Gangetic Plain, seasonal fluxes averaged between 0.5 and 1.4 N_2O-N kg $ha^{-1}season^{-1}$, equivalent to emission factors from 0.4 to 0.6% (Mahmood et al. 1998a; Pathak et al. 2002; Majumdar et al. 2002). However, there is no comparable data available for irrigated agriculture in an arid environment, or from irrigated cotton based on field measurements.

There was a significant difference in the cumulative N_2O emission and the resulting emission factor between the winter wheat and the cotton land-use systems. From the cotton systems an average of 3.4 kg N_2O-N $ha^{-1}season^{-1}$ was emitted, 400% higher than the averaged cumulative emissions of 0.75 kg N_2O-N $ha^{-1}season^{-1}$ in winter wheat. This difference seemed to be associated mainly with the higher soil temperature and accordingly higher soil microbial activity during the concomitant fertilization and irrigation events. In cotton, N fertilization took place in the hot summer months (June-July), while winter wheat received N fertilizers in March and April. Consequently, the emission pulses were far more pronounced in cotton, with soil temperatures above 25ºC during N fertilization and irrigation, compared to soil temperatures below 20ºC during N-fertilization and consequent irrigation of winter wheat.

The high frequency irrigation (HI) plots of winter wheat showed higher N_2O emissions than the low frequency irrigation (LI) plots, but the differences were statistically insignificant ($p > 0.05$) due to high spatial variability among the different measurements within one land-use system. We also observed high variability among emission pulses following a combined irrigation/ N fertilization with equal amounts of water and fertilizer (Figure 5.4 and Figure 6.2). This difference is explained by the natural soil heterogeneity among the different sites and the different demand and uptake of N by the crops growing at the time of irrigation and N-fertilization events. On the other hand, our sampling frequency of 3-4 times a week may have missed the peak gas fluxes and, therefore, caused an underestimation of emission pulses, which should be verified in subsequent research.

6.4.2 Perennial land-use systems

Poplar plantation: The observed flux rates of N_2O in the poplar plantation showed interannual differences. In 2005, high N_2O emissions were observed (39.8 µg N_2O-N $m^{-2}h^{-1}$) with a clear peak during the summer months (July/August), whereas in 2006 the mean flux rates were significantly lower (22.9 µg N_2O-N $m^{-2}h^{-1}$) with highest emissions in winter and lower flux rates in summer. Throughout 2005, the soil water content stayed at a constant high level of approx. 80% WFPS, and no N fertilizer was added. Thus, we consider the soil temperature to be the main regulating parameter of the emissions in 2005.

In January and February 2006, extraordinary high N_2O emissions were observed at two occasions, corresponding to thawing after a frost period in the upper soil. Such emission peaks have previously been described as freeze-thaw events and are known to induce an emission pulse of N_2O at, or shortly after, thawing during winter and spring (Papen and Butterbach-Bahl 1999; Morkved et al. 2006). This phenomenon was primarily attributed to enhanced denitrification due to increased biological activity (Oquist et al. 2004) and was quantitatively significant since winter emissions may exceed 50% of the annual emission (Rover et al. 1998). After these initial events in the mid winter months, the flux rates stayed below 50 µg N_2O-N $m^{-2}h^{-1}$ and followed no clear temporal trend.

Compared to results of other studies on N_2O emissions from forest plantations the measured flux rates from this site were extraordinary high. Ferre et al. (2005) investigated a poplar plantation in northern Italy that was fertilized with 300 kg ha^{-1} $year^{-1}$ of Urea and observed a mean flux rate of 11.4 ± 7.5 µg N_2O-N $m^{-2}h^{-1}$. Robertson et al. (2000) reported an average flux of 2.5 ± 0.3 µg N_2O-N $m^{-2}h^{-1}$ [1] for a poplar plantation in the midwestern United States fertilized with 10 kg N ha^{-1} y^{-1}.

We presume that one reason for the extraordinarily high flux rates at our observation site was the constantly high soil water content in combination with the high soil temperatures during the summer months. The soil water content was in the range of 65% to 89% WFPS during the whole measuring campaign. In different studies the WFPS has been described as the determining factor to regulate N_2O emissions (Simojoki and Jaakkola 2000). Largest fluxes occurred at high WFPS (70-90%), indicating that denitrification was the major factor responsible (Dobbie et al. 1999). Therefore, soil water content was always high enough (> 60%) for elevated flux rates to occur. However, no fertilizer was added at our site and soil nitrogen (0.1%) and organic matter content (0.6%) of the upper soil was rather low. Consequently, the limiting factor for N_2O emissions should have been the availability of mineral N in the soil. The area of the tree plantation was previously used for agriculture but, due to the continuous problems with water logging, it was always marginal land. Therefore, the poplar plantation had been installed more than a decade ago and had not been fertilized since. For that reason, we are quite sure that the high N_2O emissions do not come from previously applied fertilizer N on the land itself. The other option could be that from the adjacent agricultural fields with high fertilizer application rates significant amount of N was transported via the groundwater or via atmospheric deposition into this site. We took only sporadic groundwater samples from April-June 2006. The groundwater N levels were always low (0.5-2.5 mg-N/l) and, thus could not explain the high N emissions. But since we have measured only sporadically it could be well possible that we have missed this N influx. Therefore it remains unclear where the N leading to these high emissions came from which should be further investigated in subsequent research.

Tugai Forest: Given the decisive impact of WFPS to regulate N_2O emissions (Simojoki and Jaakkola 2000), and given that under low soil moisture conditions only very low emissions have been observed (Zheng, 2000), the very low N_2O emissions in the Tugai land-use system are explained by the low water content in the upper layer of the soil throughout the observation periods. In the absence of rain and irrigation of the forest, the sole water source was the shallow groundwater table. As a consequence, the soil water content of the top soil remained very low at < 40% WFPS. It should be noted, however, that this type of forest which typically flanked the Amur Darya River, used to be regularly inundated. As a result of the extensive use of river water for irrigation, this natural flooding ceased. We assumed that a natural flooding regime, with deposition of sediments and nutrients along the river banks, would have resulted in a high peak of the N_2O emissions after inundation events. Hence, the "natural" N_2O flux rates of the Tugai forest, corresponding to a historical situation before man-made interference, should be higher than observed in this study.

6.4.3 Global warming potential (GWP) of the different land-use sites

When aggregating the GWP of N_2O and CH_4 fluxes, highest GWP were found with the rice and cotton land-use systems, which are explained in the cotton land-use systems by the high input of mineral N and resulting high N_2O emissions, and by mainly the CH_4 fluxes in the flooded rice fields. CH_4 emissions in rice exceeded N_2O emissions (in terms of GWP) by more than four times, while the high GWP of cotton was exclusively caused by the high emissions pulses of N_2O following N fertilization and irrigation. The GWP of winter wheat was significantly lower than the GWP of cotton and rice, owing to the relatively low seasonal N_2O fluxes that were observed in the winter wheat systems. The unfertilized poplar plantation also showed a significant GWP owing to substantial N_2O emissions over the whole season, whereas N_2O and CH_4 fluxes from the Tugai forests were negligible in terms of GWP.

The results of the GWP estimates reflected the broad range of GWP due to the land-use systems and, thus the significance of the regional GWP budget on the actual land-use. The seasonal GWP of the different cropping sites ranged from 270 kg CO_2 eq.ha^{-1} to 3000 kg CO_2 eq.ha^{-1}; at the Tugai forest the annual GWP was below 50 kg CO_2 eq.ha^{-1}. If we assume biennial crop rotation systems of cotton-wheat-rice, which is the predominating system in the study area, the GWP from the annual cropping systems would average a flux of 6.8 kg-CO_2 eq.ha^{-1}d^{-1}, corresponding to 2.5 t CO_2 eq.ha^{-1} year^{-1}. If the entire irrigated arable land of Uzbekistan (4.3 million ha) had a similar GWP as our observed cropping sites, the annual N_2O and CH_4 fluxes would total 10.5 Mt C equivalents. This value agrees well with the estimations of the National Commission of the Republic of Uzbekistan (NCRU 1999), who reported annual fluxes of N_2O from agricultural soils and CH_4 from rice cultivation of 10.2 Mt C equivalents for the years 1994-1997, using the guiding principles set by the IPCC in 1996 for undertaking national GHG inventories (IPCC 1996).

Overall, this would contribute approximately 7% of the GWP economy in Uzbekistan, compared to total anthropogenic caused GHG emissions of 155 Mt C equivalents (NCRU 1999). On the one hand, N_2O and CH_4 fluxes from different land-use systems play a minor role in the total GHG budget of Uzbekistan. On the other hand, mitigation of these fluxes may be accompanied by additional benefits such as increased fertilizer use efficiency and carbon sequestration, both of which are additional arguments for GHG mitigation.

Improved irrigation and optimized fertilizer management should reduce losses of N fertilizer and irrigation water and minimize N_2O release. Conversion of less profitable, marginal agricultural land into forest could help to reduce land degradation and, at the same time mitigate N_2O and CH_4 fluxes and cut energy costs for fertilizer and fuel. Recent research also shows the profitability of establishing small scale forests on marginal land (Lamers et al., forthcoming). In addition, forest plantations offer mitigation options by carbon sequestration, owing to a high rate of soil C storage and the accumulation of C in unharvested wood. With the creation of a market for trading carbon dioxide emissions within the Kyoto Protocol this could provide an additional income for farmers.

6.5 Conclusion

The reported data sets represent a unique and detailed N_2O and CH_4 trace gas exchange for different land-use systems from an irrigated agricultural system in an arid region. Seasonal variations in N_2O emissions of the annual cropping systems were principally affected by N fertilization and irrigation management. Pulses of N_2O emissions occurred after N fertilizer applications in combination with irrigation events, both in the cotton and in the winter wheat land-use systems. An unfertilized plantation of poplar trees showed elevated N_2O emissions over the entire study period. Significant CH_4 fluxes could only be determined from the flooded rice field, but were at the lower end of emissions reported worldwide, whereas the natural Tugai forests showed negligible small fluxes of N_2O and CH_4.

These findings underscore the importance of land-use change concerning inventories and mitigation options for greenhouse gases. Increasing the irrigated area in desert environments such as throughout Central Asia will result in elevated GHG emissions via enhanced N_2O and CH_4 fluxes from all land-use systems dedicated to the dominant crops of cotton, rice and winter wheat. Mitigation opportunities exist, (e.g.) taking land out of irrigated agricultural production, or changing land-use from an annual cropping system to perennial forest plantations, in particular on marginal lands where annual crops are no longer profitable (Khamzina et al. 2006b; 2008). Moreover, forest plantations offer the potential of carbon sequestration by soil C storage and the accumulation of C in unharvested wood. Further research is needed to assess the net GWP of the different annual and perennial land-use systems and the interdependency of irrigation and cropping practices with changes in soil organic carbon and mitigation of N_2O and CH_4 emissions.

7 THE RELATIONSHIP BETWEEN N_2O, NO, AND N_2 FLUXES FROM FERTILIZED AND IRRIGATED DRYLAND SOILS

7.1 Introduction

Emissions of gaseous N compounds (N_2, N_2O, NO) from soils to the atmosphere reduce the N availability in the soil and at the same time represent a substantial loss of applied N fertilizers (Mosier et al. 1986; Weier et al. 1991). Furthermore, N_2O and NO are recognized as primary and secondary greenhouse gases, respectively, so that soil-borne emissions of N_2O and NO gasses have, apart from agronomic concerns, a significant influence on the atmospheric chemistry and global warming (e.g., Crutzen 1979; Crutzen 1981; Cicerone 1987).

Global cycling of reactive nitrogen has been sharply accelerated over the last century due to the (1) widespread cultivation of N fixing crops, (2) NO_x production during anthropogenic fossil fuel combustion, and (3) inorganic N fertilizer production (Galloway et al. 2003). Hence, fertilized agricultural soils have been identified as the main source of the anthropogenic N_2O emissions (Isermann 1994). However, despite substantial research over the last decades on N_2O and NO fluxes from different ecosystems, the contribution of individual sources is still uncertain (IPCC 2001). In the soils, N_2O and NO are produced as by-products and intermediate compounds during nitrification and denitrification. Denitrification, i.e., microbial respiratory reduction of NO_3^-or NO_2^- to dinitrogen (N_2), is the only significant process by which molecular N_2 can be produced in soils, whereas NO and N_2O can be produced by nitrification, i.e., microbial oxidation of NH_3 to NO_3^-, as well as by denitrification (Kuenen and Robertson 1994; Butterbach-Bahl et al. 2002). Denitrification is a key process in the global nitrogen cycle, as it removes reactive nitrogen by the production of inert N_2. Despite this outstanding ecological significance of denitrification, however, extremely little information is available on the emission of N_2 from upland soils, since it requires a reliable quantification of actual N_2 emission from soils against the high background concentration of 78% in the atmosphere (Groffman et al. 2006). Thus, we still lack a comprehensive, quantitative understanding of N_2 fluxes and its regulating factors across a wide range of ecosystems, and studies on N_2 fluxes, especially from upland terrestrial areas, are urgently needed (Groffman et al. 2006; Davidson and Seitzinger 2006).

A relatively new method for measuring N_2 emissions from soils is the gas flow soil core technique, in which the soil atmosphere is substituted with a N_2-free atmosphere, allowing direct estimation of N_2 emission from the soil due to denitrification. This method has been validated successfully for the simultaneous measurements of N_2 and N_2O emissions from intact soil cores (Butterbach-Bahl et al. 2002). Studies using this technique are few, however, and are difficult to perform because they require a rather elaborate experimental set up.

For irrigated agriculture, several studies reported high N_2O emissions from different cropping systems (e.g., Matson et al. 1996; Mahmood et al. 1998a) and identified denitrification as the main pathway of fertilizer losses from the soil-plant system (e.g., Chua et al. 2003; Hou et al. 2007). In Australia denitrification losses of 40-60% of the applied N fertilizer were reported (Freney et al. 1993; Rochester et al. 1996). Mahmood et al. (2000) measured denitrification losses of 65 kg Nha^{-1} (corresponding to 40% of the applied fertilizer) in a semiarid subtropical climate of Pakistan. It was assumed that these losses occurred mainly as N_2, however, due to methodological constraints no information was given on N_2 emissions from these irrigated agricultural systems.

In experiments on irrigated cotton fields, high N_2O emission pulses were monitored following concomitant irrigation and fertilization events (see Chapter 5). These pulses were ascribed to elevated denitrification rates due to the reduced availability of O_2 and enhanced availability of fertilizer NO_3^- in the soil after irrigation and fertilization. However, the extent of nitrogen emitted in form of N_2 or NO gasses into the atmosphere and hence the magnitude of total N gas flux remained unclear.

To understand how irrigation and fertilization influence the total N gas flux balance, a laboratory study was conducted with undisturbed soil cores taken from a cotton field using the gas flow soil core technique for evaluation of N_2O and N_2 fluxes and an incubation assay to determine N_2O and NO fluxes. The objectives of this study were thus (i) to determine N_2, N_2O, NO emissions and the ratio of N_2/N_2O and N_2O/NO under the soil moisture conditions as observed in the field after irrigation and (ii) to quantify the aggregated gaseous N losses composed of NO, N_2O, and N_2 from the irrigated cotton fields.

7.2 Material and methods

7.2.1 Study site

The soil used for this experiment was sampled at research sites of the ZEF/UNESCO project in Khorezm, Uzbekistan (Martius et al. 2006). The research station is located at 41º55' N latitude, 60º61' E longitude and at an altitude of 92 m. The climate is typically arid continental with long hot dry summers and very cold temperatures throughout the winter. Average annual precipitation during the period from 1980 to 2000 was 97 mm and mean annual temperature was 13.0 ºC (Glavigdromet 2003). The soil was classified as a calcaric gleyic Arenosol (FAO 1998) with a sandy loamy texture (Table 7.1). In the study region, it is under conventional agricultural practice with rice/cotton/winter wheat crop rotation.

Table 7.1: Main characteristics of the study site (all soil variables are given for 0-10 cm soil depth)

Study Site	
Location	41º60' N 60º51' E
Soil Type	calcaric gleyic Arenosol
pH(H2O)	6.9 ± 0.3
Bulk density (g cm 3)	1.59
SOC (%)	0.61
N (%)	0.10
Texture (USDA)	Sandy loam
Clay (%)	14.6
Silt (%)	42.8
Sand (%)	42.6

7.2.2 Sampling of soil cores and experimental design

The gas flow soil core experiment was conducted on undisturbed soil cores (diameter 0.1225 m, height 0.2 m). The soil cores were sampled at the end of July 2006, with a stainless-steel corer into which a Plexiglas cylinder was inserted. Each soil core was driven into the Plexiglas cylinder by pushing the corer into the soil. After removing the corer from the soil, the Plexiglas cylinder contained the intact soil core. At the same time, soil samples were taken from the topsoil (0-20 cm depth) for the NO/N_2O incubation measurements. The samples were air dried and transferred to the laboratory for incubation studies. Additionally, eight soil samples were taken to determine the maximum water holding capacity (WHC), which was used to adjust the moisture content of the incubated soil cores to a specific WHC.

During the cropping period prior to the soil core sampling, the *in-situ* measured soil moisture ranged from 120% WHC immediately after irrigation to 5% WHC during dry periods.

To emulate soil conditions as they occurred in the field after concomitant irrigation and fertilization, an equivalent fertilizer rate of 75 kg-N ha^{-1} of ammonium nitrate was applied to the soil cores and incubation samples. According to the conventional practices in the study region, three soil moisture levels were imposed for the incubation studies, which corresponded to wet (70% WHC), saturated (100% WHC) and flooded conditions (130% WHC). Emissions were measured at an incubation temperature of 25 ºC, which was similar to the mean soil temperature observed in the field during the cropping cycle.

7.2.3 N_2O and N_2 measurements (gas flow soil core technique)

For the determination of the N_2 and N_2O fluxes, a gas flow soil core system was used as described by Butterbach-Bahl et al. (2002). The soil cores were placed in stainless-steel incubation vessels and sealed gas tight. The soil atmosphere was changed to a N_2-free atmosphere by flushing the soil columns with a purge gas mixture of 20% O_2 5.5 and 80% He 5.0 (Basi, Rastatt, Germany). The columns were flushed for 24 h to ensure that the original soil cores' atmosphere was completely exchanged with the artificial N_2-free He/O_2 atmosphere. Next, the headspace of the incubation vessel was flushed for another 4-6 h to allow re-establishment of a "natural" gradient of gases within the soil core. Then the flushing was stopped, and the increase in N_2O and N_2 concentrations was monitored over the incubation time (6-10 h) in the headspace atmosphere by sampling the headspace every 60 minutes. The samples were analyzed using an electron capture detector (ECD, Shimadzu, Germany) for the detection of N_2O and a non-radioactive pulsed discharge helium ionization detector (PDHID, Vici AG, Switzerland) for the detection of N_2. Gas chromatographic conditions for N_2O analysis were: packed stainless-steel column Hayesep N (3 m, 1/8", 60/80 mesh); oven temperature: 25 ºC; detector temperature: 320 ºC; carrier gas: N_2 (30 ml min^{-1}). Gas chromatographic conditions for N_2 analysis were: GS capillar column (60 m); oven temperature: 25 ºC;

carrier gas He 6.0 (5 ml min^{-1}); ionization gas: He 6.0 (30 ml min^{-1}); detector temperature: 120 ºC.

After pre-incubation, the soil cores were fertilized and irrigated by adding 260 mg of ammonium nitrate (equivalent of 75 kg Nha^{-1}) dissolved in distilled water via a septum to the soil cores. Based on the information of initial soil water content and total WHC, cores were adjusted to the specific WHC. After water addition, the soil cores were purged again for at least 12h to ensure that all N_2 that could have been introduced into the vessel by this procedure was completely removed. The flux rates from the soil core were calculated from the increase in the gas concentration in the headspace over the incubation time under consideration of headspace volume and temperature.

7.2.4 N_2O and NO measurements (incubation study)

For the N_2O and NO incubation measurements, 100g of air-dried top soil was put into glass vessels (Schott Duran GmbH, Wertheim/Main, Germany). The soil was fertilized by adding 15 mg of ammonium nitrate (equivalent of 75 kg Nha^{-1}) dissolved in distilled water to the soil samples, and the soil samples were adjusted to the specific WHC. The samples were incubated at 25ºC with three replicates for each of the three soil moisture values. The samples were incubated for 6 days and analyzed for N_2O and NO on day 1, 2 and 6 after incubation.

For N_2O analysis, the glass vessels were closed gas-tight, and 5 ml of headspace air was drawn through a septum into gas-tight 10 ml polypropylene syringes at 0, 10, 20, and 30 min after the vessel had been closed. These air samples were analyzed for N_2O with a Shimadzu G 14A Gas Chromatograph (Shimadzu, Kyoto, Japan) equipped with an electron capture detector (ECD) and a Haysep N column (stainless steel 3m, 1/8", 80/100 mesh column). N_2O emissions were calculated from the linear increase in the gas concentration over time.

For NO analysis, the glass vessels acted as dynamic chambers. Soil NO efflux was determined with an automated system consisting of a sampling device, an air pump and a flow controller assuring constant sample air flow of approximately 500 ml min^{-1}, and a chemoluminescence detector (CLD 88p, Eco Physics AG, Duernten, Switzerland). Synthetic air (Basi, Rastatt, Germany) was used as carrier gas. NO concentrations of each run were averaged over 30 min. An empty glass vessel (reference chamber) was clamped to the tubing system between two runs. Details on the analytical set up and calculation of the flux rates are described in detail by Rosenkranz et al. (2006).

7.2.5 Estimating NO and N_2 field emissions

To estimate NO and N_2 field emissions from cotton sites in the ASB, the mean ratio of N_2/N_2O emissions from the gas flow soil core study and the mean ratio of N_2O/NO emissions from the incubation study were used. These ratios were multiplied with the seasonal N_2O emissions observed for the different cotton locations to calculate the corresponding seasonal emission of NO and N_2.

7.2.6 Statistical analyses

Statistical analyses were conducted with SPSS 8.0 (SPSS Inc., 1998). The effects of soil moisture content on N_2, N_2O and NO production rates were examined by analysis of variance (ANOVA) and Tukey's test ($p < 0.05$). Values in the figures represent means ± standard errors of the mean.

7.3 Results

7.3.1 N_2O and N_2 fluxes

A great variation in N_2O and N_2 emissions was recorded during the gas flux soil core experiments (Table 7.2). N_2 emissions from individual soil cores ranged from 37 to 2700 µg N_2-N m^{-2} h^{-1} and responded noticeably (although not significantly different) to soil moisture content (Figure 7.1). Average N_2 flux rates increased from 280 µg N_2-N m^{-2} h^{-1} under wet soil conditions (70% WHC) to 580 µg N_2-N m^{-2} h^{-1} under flooded soil conditions (130% WHC).

Table 7.2: N_2 and N_2O emission of the gas flow soil core experiment at different soil moisture contents (µg N_2O-N m^{-2} h^{-1} ± standard error of the mean). Means denoted by a different letter in the same column differ significantly, according to the ad-hoc Tukey test at alfa = 0.05.

WHC	n	N_2O emission [µg N_2O-N m^{-2} h^{-1}]	N_2 emission [µg N_2-N m^{-2} h^{-1}]
70%	6	61.6 ± 14.4^{a}	280.5 ± 110.1^{a}
100%	6	61.1 ± 32.9^{a}	360.7 ± 179.3^{a}
130%	6	47.0 ± 19.1^{a}	514.6 ± 368.7^{a}

N_2O emissions from individual soil cores ranged from 0.4 to 191 µg N_2O-N m^{-2} h^{-1}, but no significant effect of the soil water content on N_2O emissions was observed. Average N_2O flux rates were nearly constant at 70% and 100% WHC, but decreased slightly at 130% WHC (Table 7.2). The N_2/N_2O ratio increased with increasing soil moisture from 5 ± 2 at 70% WHC to 55 ± 27 (Figure 7.1).

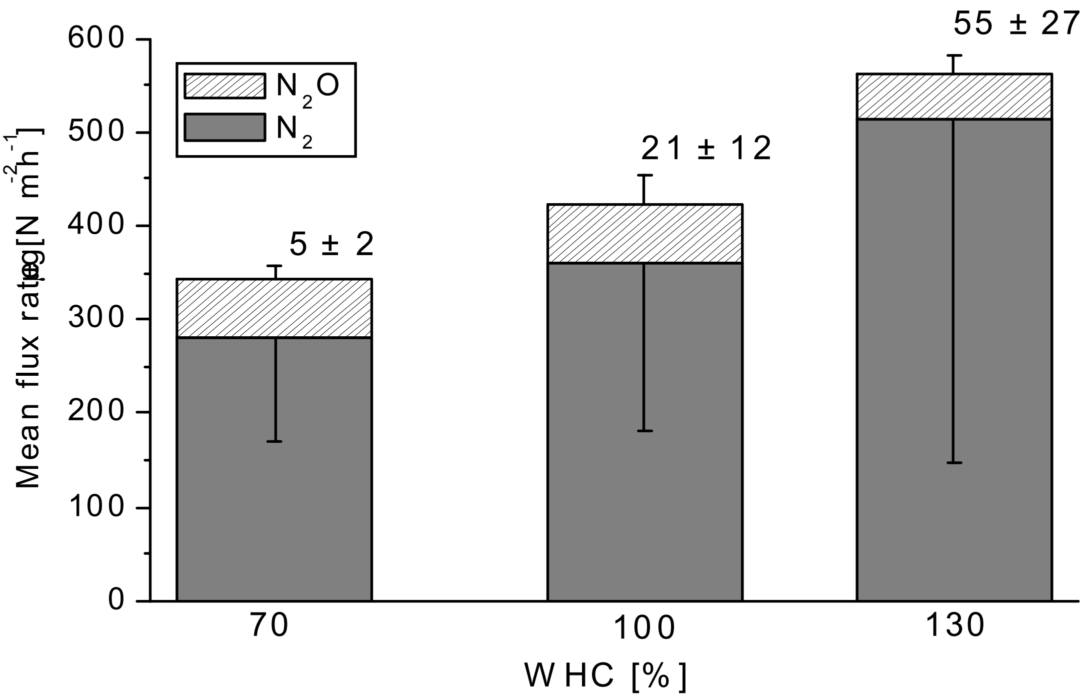

Figure 7.1: Dependency of N_2 emissions (-SE) and N_2O emissions (+SE) of the gas flow soil core experiment on changes in gravimetric water content. Values shown above the columns represent the N_2/N_2O ratio (± standard error of the mean).

7.3.2 N_2O and NO fluxes

A large daily variation of the emissions was recorded during the 6 days of the experiment (Table 7.3). NO emissions increased significantly ($P<0.05$) with time and hence peaked at day 6. At the same time, N_2O emissions decreased over time, which resulted in lowest emissions at day 6, although the differences were not significant ($P<0.05$) NO emissions were significantly ($P<0.05$) affected by soil moisture. Mean NO fluxes at 70% WHC were 10-20 times greater than NO fluxes at 130% WHC. Fluxes up to 2.94 ng h^{-1} dry $soil^{-1}$ were observed at 70% WHC, while under flooded soil conditions only trace (< 0.15 ng h^{-1}dry $soil^{-1}$) NO emissions could be detected.

Table 7.3: Daily N_2O and NO emission during the incubation experiment at different soil moisture contents (ng h^{-1} dry soil $^{-1}$± standard error of the mean). Means denoted by a different letter in the same column differ significantly according to the ad-hoc Tukey test at alfa = 0.05.

WHC	n	Day 1	Day 2	Day 6
		N_2O-N production [ng h^{-1} g dry soil $^{-1}$]		
70%	6	8.21 ± 3.34 ^{a}	8.72 ± 5.31 ^{a}	4.24 ± 0.93 ^{a}
100%	6	9.19 ± 4.32 ^{a}	12.29 ± 5.70 ^{a}	8.27 ± 2.64 ^{a}
130%	6	18.61 ± 3.36 ^{a}	20.42 ± 5.14 ^{a}	17.47 ± 3.83 ^{b}
		NO-N production [ng h^{-1} g dry soil $^{-1}$]		
70%	6	0.37 ± 0.04^{a}	1.03 ± 0.10 ^{a}	2.94 ± 0.56 ^{a}
100%	6	0.09 ± 0.02^{b}	0.24 ± 0.07 ^{b}	0.56 ± 0.07 ^{b}
130%	6	0.03 ± 0.00^{c}	0.06 ± 0.01 ^{b}	0.13 ± 0.02 ^{c}

N_2O fluxes ranged from 4.2 to 20.4 ng h^{-1}g dry soil^{-1} and notably (although not statistically significantly, P<0.05) increased with increasing soil water content. Production at 130% WHC was 2-4 orders of magnitude greater than the N_2O production at 70% WHC.

Table 7.4: Ratio of N_2O and NO emissions (± standard error of the mean) during the incubation experiment at different soil moisture contents.

	N_2O/NO Ratio		
Days after treatment	70% WHC	100% WHC	130% WHC
1	24 ±11	102 ± 52	644 ± 108
2	9 ± 6	49 ± 9	316 ± 48
6	1.5 ± 0.4	16 ± 7	136 ± 43

Average N_2O/NO emission ratios increased with increasing soil moisture and decreased over time with the lowest values at day 6 of the measurements (Table 7.4). The emission ratio of N_2O/NO ranged from 1.5 ± 0.47 at 70% WHC at day 6 after treatment to 644 ± 108 at 130% WHC at day 1 after treatment.

7.4 Discussion

7.4.1 N_2 fluxes

Soil moisture has frequently been reported as one of the major factors regulating N gas emission from croplands (e.g., Torbert and Wood 1992; Colbourn 1992; Dobbie and Smith 2001). Under conditions of high soil moisture levels, the diffusion of O_2 into the soil will decrease, and bacteria capable of denitrification may use the nitrate as an alternative electron acceptor (Firestone and Davidson 1989b), explaining thus the increased denitrification levels with increasing soil moisture (e.g., Weier et al. 1993; Scholefield et al. 1997). The results of the present study corroborate these findings since increasing N_2 flux rates with increasing soil moisture, and consequently highest denitrification activities under flooded soil conditions were observed (Figure 7.1). However, the enormous variability in N_2 emissions between the different soil cores resulted in large standard errors of the mean N_2 flux rates. This can be explained by the fact that denitrification often occurs in microsites, so called hotspots, where small areas of soil account for a very high percentage of areal denitrification (Parkin 1987; McClain et al. 2003). These hotspots result in a high spatial and temporal variability of N gas fluxes as has also been found in the field measurements of irrigated cotton (Chapter 5) and other laboratory studies on denitrification using intact soil cores (Parkin 1987; Speir et al. 1995; Butterbach-Bahl et al. 2002).

7.4.2 N_2O fluxes

The emissions of N_2O recorded during the soil core experiments were at the lower end of the N_2O emission pulses observed in field measurements at the same sites, where emissions pulses of up to 3000 µg N_2O-N m^{-2} h^{-1} occurred following N fertilizer applications in combination with irrigation events. The range of N_2O emissions observed in the incubation experiments was close to those observed by others in fertilized soil incubations (0.3-600 ng h^{-1}g dry soil^{-1}) (Bateman and Baggs 2005). Based on these findings, it appears that the temporal pattern during the incubation experiments was linked to an associated decrease in soil NO_3^- and/or soil C concentrations. Since pure mineral fertilizer was added, the microbial C immobilization may have limited the C source for denitrification. Furthermore, the process of denitrification itself could have been the cause of decreasing soil NO_3^- contents, since there was no mechanism for leaching or plant uptake during the incubation experiments. However, the effect of soil moisture on N_2O emission rates differed in the incubation experiments as compared to the soil core experiments. In the incubation experiment, emissions of N_2O were proportional to soil water content, although no statistical differences were found, whereas in the soil core experiment N_2O emissions remained constant under different soil water contents.

As previously reported for soils fertilized with mineral fertilizer, N_2O emissions increased when mineral N content of the soil was not limiting and the soil water content was above a certain threshold (e.g., Skiba et al. 1992; Dobbie and Smith 2003; del Prado et al. 2006). Highest N_2O emissions were reported when high rates of nitrification and denitrification occurred simultaneously. This optimum water content, however, seems soil specific and values varying from 60% to 100% WFPS have been reported previously (e.g., Zheng et al. 2000; Schmidt et al. 2000; del Prado et al. 2006). Consequently, above the optimum water content, denitrification will become the dominant process resulting in increased N_2O production with increasing soil water content, and as the soil becomes more anaerobic, emissions of N_2 will increase while N_2O emissions will decrease. This response was monitored during the soil core experiment with an increase in N_2 and a decrease in N_2O emissions for the 130% WHC treatment, indicating that under flooded soil conditions N_2O is completely reduced and N_2 becomes the predominant end product of denitrification. In contrast, this effect was not observed during the incubation experiments where highest N_2O emissions occurred under flooded soil conditions. It can be presumed that the comparatively small soil layer in the incubation jars (~ 1cm) allowed the diffusion of oxygen into the soil surface, which created less anaerobic conditions during the incubation experiments compared to the soil core experiments, where an intact soil core of 20 cm height was used. This may explain the different correlation between N_2O emissions and soil moisture for soil core and incubation experiments. However, the *in-vivo* field conditions should be better represented by the undisturbed soil core than the incubation experiments.

7.4.3 NO fluxes

The emission pattern of NO during the incubation experiment was similar to that found in other studies (Skiba et al. 1993), which indicate an increasing nitrification activity over time. The magnitude of these NO fluxes was in most cases lower than the NO production rates observed in other incubation studies. For example, for two agricultural soils in Germany, Bollmann and Conrad (1998) reported NO production rates of 1.5- 3 ng h^{-1}dry $soil^{-1}$ at soil water contents ranging from 80 to 100% WHC. For anaerobic incubation conditions, high NO release rates by denitrification of 100-200 ng h^{-1}dry $soil^{-1}$ were detected (Bollmann and Conrad 1998). In the present study, however, denitrification was not a significant source of NO, as shown by extremely low NO and high N_2 flux at 130 % WHC. This shows that the NO source in this study was dependent on nitrification, as has been reported in different studies (e.g., Davidson 1992; del Prado et al. 2006).

7.4.4 N_2/N_2O ratio

N_2O reductase is the enzyme responsible for N_2O reduction to N_2 in soils. The activity of the N_2O reductase enzyme is directly related to the N_2/N_2O emission ratio and is thought to increase with increasing pH values, decreasing nitrate concentrations and decreasing oxygen partial pressure (Chapuis-Lardy et al. 2007). The observed ratio of N_2/N_2O emissions was consistent with the assumption that the N_2/N_2O ratio will increase under increasingly anaerobic conditions (Firestone and Davidson 1989b; Davidson 1991; Scholefield et al. 1997). The high N_2/N_2O ratios (>>1) indicate that under soil conditions similar to field conditions after irrigation and fertilization, denitrification was the dominant process in the soil and N_2 emissions were the dominant form of gaseous N losses. This is in agreement with the field findings of irrigated cotton, where pulse emissions of N_2O after concomitant irrigation and fertilization were ascribed to highly elevated denitrification rates.

Moreover, the overall rate of production of nitrogen gases and the N_2/N_2O ratio is considered to be highest in neutral or slightly alkaline soils (Simek and Cooper 2002; Simek et al. 2002). Consequently, the high N_2/N_2O ratios may be explained by the neutral soil pH (6.9 ± 0.3) of the investigated soils in the present study resulting in high N_2O reductase activities. However, an inhibition of N_2O reductase under high NO_3^- contents to a level where N_2O becomes the predominant product of denitrification as it has been reported in several studies (Speir et al. 1995; Scholefield et al. 1997; Ruser et al. 2006) could not be underlined by the findings, where N_2 was the primary end product of denitrification under the chosen experimental conditions (high nitrate availability, high soil moisture).

7.4.5 N_2O/NO ratio

The emission ratio of N_2O/NO can be used as an indicator for the relative importance of nitrification and denitrification in producing NO and N_2O. Emission ratios < 1 are typically associated with active populations of nitrifiers and soil conditions favorable for nitrification, whereas emission ratios > 10 are associated with denitrification and restricted aeration (Skiba et al. 1992; del Prado et al. 2006). During the incubation measurements, average N_2O/NO emission ratios from 1.5 ± 0.47 at 70% WHC and 644 ± 108 at 130% WHC (Table 7.4) indicate clearly that the observed N-fluxes were mainly associated with denitrification. This underscores the hypothesis that, after concomitant fertilization and irrigation, denitrification was the dominant process in the investigated soils. However, the decreasing N_2O/NO ratios over the period of the laboratory experiment (6 days) indicates that nitrification was proceeding rapidly. Yet, under irrigated field conditions, the temperatures of the top soil are usually rather elevated during the growing season, thus enhancing the rapid drying out of the top soil after irrigation and in turn constraining the nitrification process.

7.4.6 Estimation of N_2 and NO emissions from cotton sites in Khorezm, Uzbekistan

The experiments were set up to emulate soil conditions as they occurred during the field surveys and in particular following concomitant irrigation and fertilization (see Chapter 5). The applied fertilizer rate corresponded to good agricultural practices in the study region, and during field observations 80 to 95% of the total N_2O emissions were observed in the aftermath of irrigation and fertilization events and at soil moisture contents similar to the ones used in this study. Therefore, the observed N_2/N_2O and N_2O/NO ratio can be used to estimate N_2 and NO losses from cotton over an entire cotton growth cycle.

The mean N_2O emission rate was 56.5 ± 11.7µg N_2O-N m^{-2} h^{-1} for the soil core experiments and thus fell within the range of the reported mean N_2O field emission rates of 20.6 – 149.8 µg N_2O-N m^{-2} h^{-1} under irrigated cotton. The N_2/N_2O and N_2O/NO ratios declined with increasing soil moisture with a mean value of 27 ± 10 for N_2/N_2O, respectively 144 ± 71 for N_2O/NO. Assuming seasonal N_2O emissions as were observed in the cotton fields (0.9 to 6.5 kg N_2O-N $ha^{-1}$$season^{-1}$; see Figure 5.4), the N losses for the different cotton sites could be estimated (Table 7.5). Hence, N_2 emissions from the different cotton fields in 2005 and 2006 ranged between 24 and 170 kg-N $ha^{-1}$$season^{-1}$, while NO emissions were of minor importance (between 0.1 and 0.7 kg-N ha^{-1} $season^{-1}$).

Table 7.5: Observed N_2O-Flux (see Chapter 5) and estimated N-fluxes (± standard error of the mean) for the different research sites.

Research site	Year	N_2O-flux [kg/ha/season]	N_2-flux [kg/ha/season]	NO-flux [kg/ha/season]	∑ Sum [kg/ha/season]	Applied Fertilizer [kg/ha]	% of fertilizer
ATG	2005	6.5	175 ± 65	0.7 ± 0.5	182.2± 65.5	250	73
ATG	2006	0.9	24 ± 9	0.1 ± 0.06	25.0± 9.1	250	10
Urdu LI	2006	2.4	65 ± 24	0.3 ± 0.17	67.7± 24.2	250	27
Urdu HI	2006	4.4	119 ± 44	0.5 ± 0.3	124.8± 44.3	250	50
ATC	2005	2.9	78 ± 29	0.3 ± 0.2	81.2 ± 29.2	162.5	50

However, the enormous heterogeneity of denitrification activity in the different soil cores indicates a high uncertainty of these estimates and shows that more measurements are required to produce robust mean emission ratios. Moreover, between the emission pulses of N_2O following irrigation and fertilization, field soil moisture was low, and nitrification might have become the dominant process producing larger amounts of NO that were not taken into account in the calculations. Overall N_2O, NO, and N_2 fluxes amounted to 25 to 182 kg-N ha^{-1} $season^{-1}$, corresponding to 10% to 72% of the applied N fertilizer. Nevertheless, these estimated N_2O, NO, and N_2 fluxes do not represent the total gaseous N losses, which would include the amount of NH_3 volatilization. Consequently, the total gaseous N losses can be even higher depending on soil properties and agricultural management practice.

Assuming a mean denitrification loss of 40% of the applied fertilizer for the 1.4 millions of ha of irrigated cotton in Uzbekistan (FAOSTAT, 2007) and a mean fertilization rate of 200 kg N ha^{-1} (FAO 2003), approximately 110,000 tons-N of fertilizer would annually be lost from irrigated cotton cultivation via denitrification alone. Considering the high heterogeneity of the measurements, this can only be seen as a rough estimate, but such losses are consistent with previous studies on denitrification in irrigated cotton, e.g., in Australia denitrification losses of 40-60% of the applied N fertilizer were reported by different studies using the ^{15}N balance techniques (Freney et al. 1993; Chen et al. 1994; Rochester et al. 1996). In the semiarid subtropical climate of Pakistan, Mahmood et al. (2000) measured denitrification losses of 65 kg N ha^{-1}, which corresponded to 40% of the applied fertilizer during one season; it has to be noted that they used the acetylene inhibition method, which usually even underestimates denitrification rates (Bollmann and Conrad 1997).

7.5 Conclusion

The findings for the first time provide estimates of the magnitude of N_2 losses from irrigated dryland cotton, using a gas flow soil core method that allowed direct measurements of N_2 emissions following irrigation and fertilization from intact soil cores. The findings demonstrate that under the current agricultural practices in irrigated cotton in Uzbekistan, denitrification is the major pathway explaining the high N losses as N_2 to the atmosphere from N fertilizers. Total annual denitrification losses of irrigated cotton may amount to 40% of the annually applied N fertilizer, which would correspond to 110,000 tons-N of fertilizer for entire Uzbekistan. Emissions of N_2O contributed to about 5-10% of the aggregated N loss, whereas only minor emissions of NO were observed. These losses not only represent a financial loss to farmers, but also concurrently influence the atmospheric chemistry and global warming. The wide range of the estimated N losses for the different cotton fields indicates that there is scope for reducing these losses by modified irrigation and fertilization practices. An improved understanding is needed of the pathways of N cycling in irrigated dryland agriculture, and of how denitrification can be controlled by fertilization and irrigation. Further research should be combined with projects on soil fertility management in order to identify management options that could increase N use efficiency and mitigate emissions of N_2O and NO.

8 OVERALL CONCLUSIONS AND OUTLOOK

Irrigation is indispensable to ensure food security for a growing world population. Over the last century, worldwide the area under irrigation has seen a five-fold increase and currently irrigated agriculture provides 40% of the world's food production (FAO 2000). The impacts of irrigation, however, go far beyond the issue of food security, since irrigation affects the microbial C and N turnover in the soil and hence the biosphere-atmosphere exchange of greenhouse gasses (GHG). In view of the global importance of irrigated agriculture, it is crucial to understand the extent to which this agroecosystem interferes with the global N and C cycles and contributes to the global source strength of atmospheric GHG budgets.

This research investigates the impact of common practices for irrigated cotton (*Gossipum hirsutum* L.), winter wheat (*Triticum aestivum* L.) and rice (*Oryza sativa* L.) as well as a perennial poplar plantation (*Populus nigra* L.) and the natural Tugai riparian forest in the Aral Sea Basin (ASB) on biosphere-atmosphere exchange of radiatively active trace gases and the microbial transformation processes in soils. The study aims to identify feasible concepts for ecologically sustainable land-use and management options.

The conclusions drawn from the two-year field study are summarized hereafter and focus on emissions of N_2O and CH_4 in various vegetation systems and a laboratory incubation study to assess the aggregated gaseous N losses composed of NO, N_2O, and N_2 from fertilized and irrigated agricultural fields in the ASB. Implications of the research findings and suggestions for further studies are given at the end of the chapter.

8.1 Biosphere-atmosphere exchange of greenhouse gases and N turnover in the soil of irrigated agricultural systems in the Aral Sea Basin

Irrigated agricultural production in the ASB turned out to be a significant source of GHG due to high emissions of N_2O during the annual cropping of wheat and cotton, which annually comprise about 70% of the total irrigated area in the study region Khorezm. Seasonal variations in soil N_2O fluxes depended upon fertilizer and irrigation management, and extraordinarily high N_2O emissions pulses were observed after N fertilizer applications but only in combination with irrigation events. These emission pulses accounted for 80-95% of the total N_2O emissions in cotton and winter wheat.

Significant CH_4 fluxes could only be determined from flooded rice fields, but these were at the lower end of CH_4 emissions reported from flooded rice worldwide. The unfertilized plantation of poplar trees showed elevated N_2O emissions over the entire study period, whereas the natural Tugai forests showed negligible fluxes of N_2O and CH_4.

For a global warming potential (GWP) assessment for the different land-use systems, CO_2 fluxes from the soils were not taken into account in this study. These fluxes generally show enormous variations in space and time, and net fluxes of CO_2 are typically derived indirectly from long-term changes in soil carbon content. Detection of changes in soil C, however, would require long-term studies of the content and composition of the soil organic matter. There were no indications that the soils in the monitored farm fields were progressively depleted in soil C, hence CO_2 fluxes were deemed small in the land-use systems studied. This study thus only refers to the GWP of CH_4 and N_2O emissions from the investigated agroecosystems. Soil-borne CO_2 emissions, as well as off-site emissions stemming from the energy required for irrigation, farm operations and fertilizer production, were not taken into account for the calculation of the GWP.

The corresponding GWP of the N_2O and CH_4 fluxes varied significantly among the different land-use sites ranging from below 50 kg CO_2 eq.ha^{-1}season^{-1} for the Tugai forest to 3000 kg CO_2 eq.ha^{-1}season^{-1} for an irrigated cotton site. In general, the GWP was highest for the rice and cotton systems. For N_2O this was due to the high input of mineral N and resulting high N_2O emissions in cotton; for CH_4 emissions, it was due to the CH_4 fluxes in the flooded rice fields. CH_4 emissions in rice exceeded N_2O emissions (in terms of GWP) more than four-fold, while the GWP of cotton was primarily caused by the high N_2O emissions pulses. A biennial crop rotation system of cotton-wheat-rice, which is the predominating system in the study area, would amount to an average GWP of 2.5 t CO_2 eq.ha^{-1} year^{-1}.

Consequently, if for the entire irrigated arable land of Uzbekistan (4.3 million ha) a similar GWP as found in the observed site is assumed, the annual N_2O and CH_4 fluxes from irrigated agriculture would total 10.5 Mt CO_2 equivalents. However, this estimate needs improvement, as it is based on measurements in Khorezm only, and soil CO_2 fluxes of land-use change and soil degradation were not taken into account. But the estimate agrees well with those of the National Commission of the Republic of Uzbekistan (NCRU 1999), where annual fluxes of N_2O from agricultural soils and CH_4 from rice cultivation of 10.2 Mt CO_2 equivalents for the years 1994-1997 were reported, using the guiding principles set by the IPCC in 1996 for undertaking national GHG inventories (IPCC 1996). Compared to total anthropogenic driven GHG emissions of 155 Mt C equivalents per year, this would contribute to approximately 7% of the GWP economy for entire Uzbekistan (NCRU 1999). This estimate shows that on the one hand, N_2O and CH_4 fluxes from different land-use systems play a minor role in the total GHG budget of Uzbekistan. On the other hand, mitigation of these fluxes may lead to synergies with sustainable development policies, improve environmental quality and can be accompanied by additional benefits such as increased fertilizer use efficiency.

A laboratory study conducted with intact soil cores showed that under the soil conditions naturally found during the emission pulses, denitrification was the dominant process responsible for gaseous N emissions, whereas molecular nitrogen (N_2) was the main end product of denitrification. Emissions of NO were only of minor importance, which indicates restricted nitrification activities after irrigation and fertilization. The mean ratios of N_2/N_2O emissions increased with the soil moisture content, and N_2 emission exceeded N_2O emission by a factor of 5 ± 2 at 70% WHC and a factor of 55 ± 27 at 130% WHC, while the mean ratios of N_2O/NO emissions varied between 1.5 ± 0.4 (70% WHC) and 644 ± 108 (130% WHC).

With the observed emission ratios, the gaseous N losses could be estimated for the different field research sites of irrigated cotton. N_2 emissions from the cotton fields in 2005 and 2006 ranged between 24 and 170 kg-N $ha^{-1}season^{-1}$, whereas NO emissions were of minor importance (0.1 to 0.7 kg-N ha^{-1} $season^{-1}$). Overall N_2O, NO, and N_2 fluxes amounted to 25 to 182 kg-N ha^{-1} $season^{-1}$, corresponding respectively to 10% N and 72% N of the N applied as fertilizer. However, the high heterogeneity of denitrification activity in the different soil cores during the incubation experiment suggests a high uncertainty of these estimates. Therefore, more measurements are recommended to assess more accurately the gaseous N_2 losses from these irrigated agricultural systems.

On the other hand, the outcomes for the first time provide estimates of the magnitude of N_2 losses from irrigated land-use systems in an arid environment. When assuming a mean denitrification loss of 40% N of the applied fertilizer for the 600,000 tons of mineral N fertilizer used annually in Uzbekistan (FAO 2003), at an average market price of US$ 150 per ton N (FAO 2003), every year an equivalent of US$ 36 Million of N applied as fertilizer would be lost through denitrification alone from the agricultural sector in Uzbekistan. Although this obviously is a very rough estimate as it did not account for e.g., fertilizer type, irrigation management, soil texture and various other parameters that influence denitrification activity, the computations nonetheless illustrate that reducing gaseous N losses of the dominating land-use systems offers the potential of cutting production costs and increasing income of farmers, while concurrently mitigating the environmental impact.

8.2 Greenhouse gas mitigation options for irrigated agricultural systems in the Aral Sea Basin

The findings demonstrate the importance of agricultural management and land-use change for mitigation options and inventories of GHG. The wide range of N_2O and CH_4 fluxes from the different land-use systems and the strong impact of the currently dominating practices of fertilizer and irrigation water management on N_2O emissions and N losses indicate a wide scope for reducing GHG emissions. This could be reached by changing the present land-use strategies and modifying the present irrigation and N fertilization practices. However, mitigating options will always rely on “win-win” opportunities, and the measures to reduce GHG emissions should offer additional benefits for the productivity and environmental integrity of the agricultural ecosystem. Given the present poverty of the producers in the study region, these will be unlikely to implement alternatives that only benefit the environment. In addition, mitigation options should ideally entail a positive adaptation to the impacts of climate change (e.g., carbon sequestration projects that concurrently implicate better drought preparedness). Agricultural GHG mitigation should also be addressed in the context of sustainable development, as it has social, economic and environmental effects that have to be considered. Worldwide evidence underscores that on a global scale the actual level of GHG mitigation is far below the technical potential of these measures, which is mainly due to economical and political constraints (Smith et al. 2007).

Nevertheless, in the study region sustainable mitigation measures do exist such as improvement of the N-use efficiency and measures by which the extreme N_2O emission peaks are avoided that were observed in the annual cropping systems immediately after fertilization. In particular, concomitant N fertilization and irrigation should be avoided whenever possible and, as the findings of temperature-dependent N_2O fluxes suggest, even conducting fertilization and irrigation in cool weather may reduce N_2O emissions. Moreover, the results suggest that replacing NO_3^- fertilizers with NH_4^+ fertilizers could be another option to lower N_2O emission pulses, especially in combination with nitrification inhibitors. In general, management practices that increase the fertilizer use efficiency in irrigated systems, such as sub-surface fertilizer application, fertigation and drip irrigation (Thompson et al. 2000; Ibragimov et al. 2007) are expected to reduce the N_2O emissions. This perhaps can be confirmed in subsequent research on the flood and furrow type of irrigation dominating in the study region. In view of the potentially high costs of the N fertilizer losses via denitrification, various management measures will most likely also be economically attractive.

Removing land from irrigated agricultural production, or changing land-use from an annual cropping system to perennial forest plantations, in particular on marginal lands where annual crops are no longer profitable (Khamzina et al. 2006b, 2008), could help reducing land degradation and, at the same time, mitigate N_2O and CH_4 fluxes and cut energy costs for fertilizer and fuel, which presently are among the highest expenses of farmers (Rudenko and Lamers 2006). In addition, forest plantations offer the potential of carbon sequestration by soil C storage and the accumulation of C in unharvested wood given the high biomass production which can reach 20-32 tons per hectare and year after five years depending on the species (Khamzina et al. 2006b). With an emerging market for carbon trading from the Kyoto Clean Development Mechanisms (CDM) and other types of certificates this could provide an additional income for farmers.

Modified land-use and tillage management certainly could mitigate GHG emissions and enhance favorable impacts on the environment. A mix of optimal crop rotations with reduced tillage and modified residue management would promote carbon sequestration and could also improve soil fertility and prevent soil degradation (Smith et al. 2007). However, the overall effect of reduced tillage on short- and long-term GHG fluxes has not yet been systematically assessed (Six et al. 2004), and it is not clear whether greater N_2O emissions offset the benefits resulting from carbon sequestration (Li et al. 2005a). It is, therefore, crucial to investigate further the effect of different N-management strategies together with different tillage and residue management on C and N fluxes in drylands.

8.3 Implications of the research for further studies

The substantial emissions of GHG from the annual cropping systems, the significant impact of the agricultural management on these fluxes, and the broad range of GWP of the different land-use systems, demonstrate the potential of enhancing the environmental integrity of the agroecosystems in Khorezm by modifying both the land-use patterns and the management practices. However, this conclusion is based on a two-year field study in selected land-use systems - albeit the predominant ones - of the ASB. Obviously more and long-term field studies of different land management strategies are required for an improved scientific assessment of the effect of modified management strategies and new production systems on GHG emissions in the long run. The influence of improved land- and water-use technologies, such as those anticipated by the ZEF/UNESCO project (Martius et al. 2006) on trace gas emissions and N turnover in soil, seems especially important.

8.3.1 Carbon and nitrogen trace gas emissions from alternative land-use and improved production systems

New land and water technologies and conservation agriculture practices are being introduced, such as bed planting, minimum and zero tillage practices and residue incorporation in cotton and wheat to increase production and productivity, in order to improve soil physical, chemical and biological properties and to increase irrigation water use efficiency in this region. Moreover, the introduction of alternative crops (e.g., potatoes, sorghum, indigo), the improvement of crop rotation and fertilizer management (Kienzler, forthcoming) and the potential of agroforestry and afforestation of degraded lands (Lamers et al. 2006; Khamzina et al. 2006a; Khamzina et al. 2006b) have been assessed in depth. The results clearly indicate options for reducing in the short term irrigation water consumption under conservation agriculture and in the longer term for improving soil fertility and preventing soil degradation (Egamberdiev 2007). Khamzina et al. (2006b) showed that afforestation of marginal lands reduced land degradation, and that tree plantations showed good productive potential on degraded land.

Follow-up research should thus assess the overall short-term and long-term changes of GHG emissions under different land- and water-management regimes. It is especially important to investigate the long-term, as well as the immediate effect of these new technologies on the fluxes of the three GHG (CO_2, CH_4 and N_2O) together. Furthermore, the effect of different N management strategies aiming to reduce N_2O fluxes and increase fertilizer use efficiency needs to be assessed. To identify feasible mitigation strategies, a better understanding of the regulating factors of GHG fluxes is needed. This can be done, e.g., with fully automated GHG flux measurements, together with a continuous monitoring of physical and chemical soil parameters. Increasing the soil organic matter pool concomitant with low GHG emissions, high fertilizer and water use efficiencies would represent the very desirable “win-win” situation.

8.3.2 Microbial key processes of soil N and C turnover and trace gas exchange

Another focus of subsequent research could be the complex interactions between microorganisms and plants with respect to C and N cycling and to quantify the microbial key processes of soil inorganic N production and consumption (gross ammonification, gross nitrification, microbial immobilization of inorganic N), C turnover and trace gas exchange under the specific conditions of the various land-use systems within an irrigation-dominated landscape. An improved understanding of the balance of microbial production and microbial consumption of inorganic N as well as plant uptake, gaseous losses and N losses along the hydrological pathways, the consequent seasonal dynamics of C and N cycling in soils, and the main factors controlling these fluctuations could help to improve management strategies and to better match soil N supply to crop N demand and thus mitigate emissions of GHG. In addition, this will help improving process-oriented biogeochemical models to better simulate GHG emissions from the cropping systems.

To achieve this, follow-up research should monitor the mineral N content of the agricultural soils continuously over the vegetation cycle together with the N trace gas emissions and losses of NO_3^-, including those via the groundwater as recently postulated (Kienzler, forthcoming). To assess gross rates of N mineralization, immobilization and nitrification, the ^{15}N pool dilution technique (Davidson et al. 1991), and to quantify net rates of ammonification and nitrification the buried bag incubation technique (Gasche et al. 2002) seem most appropriate.

8.3.3 Application of a biogeochemical model to assess C and N trace gas emissions on a larger scale

The results of a two-year field measurement campaign provide valuable insights into emissions of N_2O and CH_4 and their triggering and regulating parameters in several annual and perennial land-use systems of the ASB. The advocacy of an improved resource use that is financially attractive to farmers and ecologically sound and the extrapolation of the site-specific results to larger areas such as farm, regional, national or maybe even on global scale is a next step that however is time and resource demanding. But considerable progress can be expected from the use of a process-oriented biogeochemical model for simulating the coupled cycles of C and N in terrestrial ecosystems and the associated biosphere-atmosphere exchange that is linked to detailed geographic information systems (GIS).

Process-oriented models give an intricate description of the main biogeochemical processes and parameters involved and environmental impacts such as land-use type, agricultural activities, mitigation options, etc., on trace gas emissions. Several models (e.g., DAYCENT, DNDC) have been developed to simulate soil organic carbon levels and GHG fluxes for various ecosystems (Li et al. 2000; Del Grosso et al. 2002).

In the meantime, a data set from this study (cotton/ ATG) has already been used for validating the DNDC model version 8.6L (C. Li, pers. comm.). While previous versions of DNDC showed a rather poor match with the empirical N_2O flux rates, the modified DNDC version showed a reasonably good fit for the initial N_2O peaks, i.e. those triggered by joint water and N application. (Figure 8.1). On the other hand, the model clearly overstated N_2O emissions in the later stages of the vegetation periods, i.e., when the fields were irrigated without N fertilization. Thus, the total emission rate computed by the model was 40% above the measured values of 6.5 kg N_2O-N $ha^{-1} season^{-1}$.

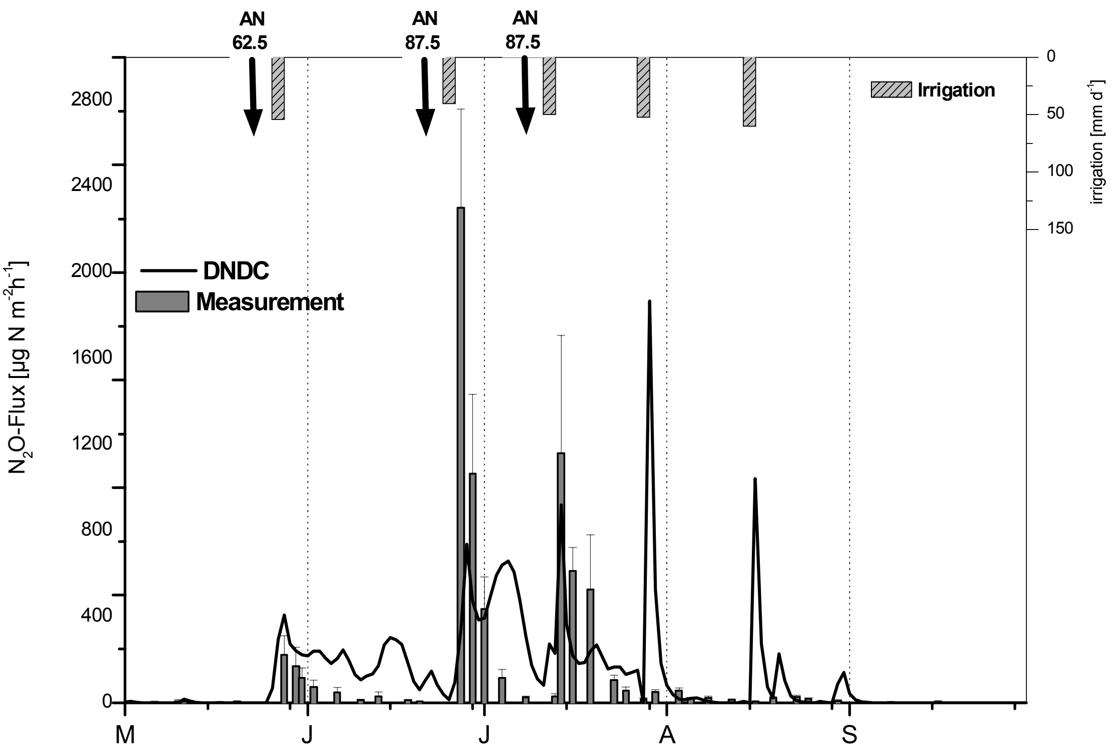

Figure 8.1: Comparison of measured and simulated N_2O flux rates using the data set from cotton/ ATG experiment in 2005 and DNDC model version 8.6L (C. Li, pers. comm.).

However, given the dynamic development of the DNDC model over recent years, the predictive capability of DNDC simulating these semiarid systems appears to have deteriorated due to changes done since the model version 8.6L. The most recent DNDC version 9.2 (downloadable from http://www.dndc.sr.unh.edu/Models.html as of March 2008) failed to reproduce the emission peaks and resulted in seasonal emissions <1 kg N_2O-N $ha^{-1}season^{-1}$ (using the identical data set as for the simulation shown in Figure 8.1). Irrespective of these more recent changes, the results shown in Figure 8.1 illustrate that the DNDC model has – in principal – the capacity to simulate N_2O emissions from irrigated dryland during periods when water and N are applied simultaneously. Apparently, the model version used in this study overstated the amount of available N in the soil during later stages when no more N is applied. While a modification of the model to simulate this persistence at low N_2O flux rates seems technically feasible, a universal model for this agroecosystem in the entire ASB will certainly require further model development and thus, considerable time input that was beyond the scope of this study.

In addition, the DNDC model can in future be linked to a GIS database of the ASB containing various parameters, such as soil, climate and land use. Once the model is successfully validated for the specific conditions of this region, it will allow upscaling of GHG emissions under different management options as previously done for agricultural systems in China and India (Li et al. 2004, 2006; Pathak et al. 2005; Babu et al. 2006). In the next step, present and future climate conditions can be integrated into DNDC to assess climate change impacts on GHG emissions as done for forests (Kesik et al. 2006). Such a model could become a powerful tool to identify GHG mitigation options for irrigated agricultural systems and provide a decision support system for policy makers by juxtaposing emission scenarios over a range of different scales.

8.3.4 Socio-economic assessment of the implementation of GHG mitigation projects

When the implementation of GHG mitigation projects is anticipated, the farmers' acceptance and the political feasibility of such measures have to be ensured. For example, for the implementation of agroforestry and conservation agriculture practices, farmers have to change their attitude to and understanding of cultivation methods while increasing their concern for the environment and future benefits. This is a point of concern worldwide, but mainstreaming such considerations is particularly challenging in the ASB with the ongoing gradual transition from the centralized, planned management to a market-oriented economy. Many farmers are not prepared for these changes and have insufficient management knowledge and experience in agriculture to run private farms as recently concluded (Rudenko and Lamers 2006). Therefore, the knowledge, experience and motivation of farmers to adopt such agriculture practices as well as the socio-economic benefits of anticipated GHG mitigation projects need to be evaluated.

The emerging market for carbon trading might offer options for Clean Development Mechanisms (CDM) or other types of certificate projects that are cost effective and economically attractive to the farmers. Implementation of such projects will require coordination across relevant ministries and institutions and a favorable institutional and legal environment. Furthermore, it has to be assessed if the existing land-tenure systems and the set up of the production systems favor such practices.

8.4 Summary and outlook

This is the first detailed study of N_2O and CH_4 trace gas exchange between different agroecosystems and the atmosphere carried out in irrigated agricultural systems in an arid environment. The measurements of trace gas emissions in five different land-use systems show that:

(i) irrigated agricultural production in the ASB is a significant source of GHG due to high emissions of N_2O and CH_4 from the annual cropping systems;

(ii) seasonal variations in N_2O emissions of the annual cropping systems are principally affected by N-fertilization and irrigation management;

(iii) extraordinarily high emission N_2O pulses occur after N fertilizer applications in combination with irrigation events, both in cotton and in winter wheat land-use systems;

(iv) significant amounts of CH_4 are emitted from the flooded rice field;

(v) in the natural, perennial Tugai riparian forests, fluxes of N_2O and CH_4 are comparatively low.

A laboratory incubation study and the use of a novel gas flow soil core method allowed direct measurements of N_2 emissions following irrigation and fertilization from intact soil cores, and provided for the first time estimates of the magnitude of N_2-losses from irrigated dryland cotton. It could be demonstrated that under the current agricultural practices in irrigated cotton in Uzbekistan, losses from N fertilizers as N_2 to the atmosphere are high, and that denitrification is the major pathway. As a consequence, total annual denitrification losses from irrigated agriculture may amount to 40% of the annually applied N fertilizer, leading to economically relevant losses of N from the cropland (estimated at US$ 36 Million for entire Uzbekistan). Emissions of N_2O contributed to about 5-10% of the aggregated N loss, whereas only minor emissions of NO were observed.

The corresponding GWP of N_2O and CH_4 fluxes varied significantly among the different land-use sites and ranged from below 50 kg CO_2 eq.ha^{-1}season^{-1} for the Tugai forest to 3000 kg CO_2 eq.ha^{-1}season^{-1} for an irrigated cotton site. The GWP per growing season was highest for the rice owing to the high CH_4 fluxes in the flooded rice fields caused by the high input of mineral N, and to the cotton land-use systems as a result of the high N_2O emissions triggered by the combined input of N as fertilizers and irrigation water. The predominating cropping system in the study area, a biennial rotation of cotton-wheat-rice, would amount to an average GWP of 2.5 t CO_2 eq.ha^{-1} year^{-1}. On the basis of the observed flux rates, the annual N_2O and CH_4 fluxes from irrigated agriculture in Uzbekistan were estimated to contribute 10.5 Mt CO_2 equivalents in GHG emissions, which is about 7% of the anthropogenic GHG emissions in Uzbekistan.

It was argued that GHG emissions from these agricultural systems can significantly be reduced by different land-use strategies and modified management practices. Recommendations for mitigating GHG emissions could be:

(i) matching fertilization better to crop N demand and avoiding concomitant N fertilization and irrigation as much as possible;

(ii) replacing NO_3^- fertilizer with NH_4^+ and using nitrification inhibitors

(iii) using modified management practices that have been shown to increase the fertilizer use efficiency in irrigated systems, such as sub-surface fertilizer application, fertigation and drip irrigation;

(iv) conducting fertilization and irrigation in cool weather;

(v) removing marginal lands where annual crops are no longer profitable from irrigated agricultural production or changing the land-use to perennial forest plantations.

This study underscores the importance of agricultural management and land-use change for the biosphere-atmosphere exchange of GHG gases and C and N turnover processes in irrigated soils. To improve estimates of GHG fluxes from irrigated dryland agriculture, a dual approach needs to be followed. First, a better understanding of the complex interactions between microorganisms and plants with respect to C and N turnover and their regulating parameters is needed. This can only be achieved by fully automated GHG flux measurements together with monitoring of physical and chemical soil parameters continuously over the vegetation cycle. On the basis of the results of these measurements, it will be necessary to develop new tools for upscaling from the site-specific results to larger areas to assess GHG emissions and to identify GHG mitigation potentials on a larger scale. At present, the best prospects for this task are biogeochemical models that are able to simulate all processes and mechanism involved in N and C trace gas emissions from soils. Furthermore, more studies on GHG emissions from different irrigated agricultural systems are required to validate whether the conclusions drawn in this study are also representative for other irrigated agroecosystems.

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