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Ryusuke Hatano, Jerzy Lipiec

EFFECTS OF LAND USE AND CULTURAL PRACTICES ON GREENHOUSE GAS FLUXES IN SOIL

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prof. dr hab. Jan Gliński, czł. rzecz. PAN

Adres redakcji

Instytut Agrofizyki im. Bohdana Dobrzańskiego PAN, ul. Doświadczalna 4, P.O. Box 201
20-290 Lublin 27, tel. (0-81) 744-50-61, e-mail: editor@demeter.ipan.lublin.pl
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Druk: Drukarnia *ALF-GRAF*, ul. Kościuszki 4, 20-006 Lublin

CONTENTS

1. INTRODUCTION

Long term records show increasing growth in anthropogenic greenhouse gas emissions, in particular during last decades [79]. There is a considerable uncertainty in the estimates of carbon dioxide (CO_2), nitrous oxide (N_2O), methane (CH_4) and nitric oxide (NO) emissions from soils [e.g, 104,135]. New estimations suggest that input of greenhouse gases to the atmosphere from agricultural production has been previously underestimated [123]. The gases may diffuse to the atmosphere directly from the soil or indirectly through subsurface drainage after leaching [156].

In this review we summarize current knowledge on the effects of land use and cultural practices on greenhouse gas fluxes with emphasis on recent literature. Rates of emission of greenhouse gas rates are reported in the literature in a variety of units. We present the results in their original units and also in the following converted units: g CO_2-C m^{-2} h^{-1}, g N_2O -N ha^{-1} day^{-1}, mg CH_4-C m^{-2} h^{-1}, ng NO-N m^{-2} s^{-1}.

2. CARBON DIOXIDE

Atmospheric CO_2 accounts for 60% of the total greenhouse effect [145] being the second largest flux in the global carbon cycle [177]. In general, past and present conversions of native soils to agriculture have contributed significantly to CO_2 emissions to the atmosphere [140]. Agricultural land-use types and cultural practices largely affect the emission and uptake of CO_2 and thereby play an important role in sequestering C in soil.

2.1. Effect of land use on carbon dioxide flux

2.1.1. Soil flux

The effect of agricultural land-use type and management on the soil CO_2 exchange is related to soil and climate conditions. The results from studies where different land uses were applied on the same site (Tab. 1) indicate enhancing effect of prairie vs. cornfield which can be associated with an extensive fibrous root system and greater microbial and earthworm populations and decomposition of usually present surface residues in prairies. Greater CO_2 soil flux from grazed than non-grazed prairie can be partly due to the effect of additional nutrients from livestock excrement.

Conversion of grassland supported by organic soil to cultivation in northern climate of Finland resulted in a substantial increase in mineralization of stored organic material and loss of carbon [116]. However, such conversion under tropical conditions of Venezuela did not cause significant changes in CO_2 flux [155].

Table 1. Effect of land use on CO_2 emission from soil

Land use	Location	Original units ($g\ CO_2\ m^{-2}\ h^{-1}$)	Converted unit number ($g\ CO_2\text{-}C\ m^{-2}\ h^{-1}$)	Reference
Prairie	Wisconsin, USA	2.0*	0.083	188
Cornfield		1.4-1.5	0.058-0.062	
Prairie (non-grazed)	Mandan, USA	3.5**	0.146	48
Prairie (grazed)		4.3	0.179	

* Based on yearly data; **Based on growing season's data.

2.1.2. Effect of vegetation

Net ecosystem exchange (difference between respiration and CO_2 uptake by plants) is largely regulated by vegetation cover [e.g. 30,38,47,116] and thereby influence the functionality of the various land uses and tillage systems as a sink or source for atmospheric CO_2. This regulation is mostly through capturing soil CO_2 through photosynthesis and partitioning of photosynthetic carbon to the roots [47,196] and depends on type of vegetation cover during the year. Table 2 presents some data on net CO_2 exchange from various ecosystems. Based on calculation of C sequestration rates using a global database, West and Post [194] showed that crop rotation other than continuous corn to corn-soybean sequestered $1.37 \cdot 10^{-3}$ g C m^{-2} h^{-1}. The sequestration is enhanced by high-intensity cropping [52], crop rotations with leguminous crops [35] and crop residue amendment [83].

The C partitioned to the roots can be partly lost by root and soil microbial respiration and sequestered [38,92]. The relative proportions of the losses and gains are associated with the size of the aboveground biomass and photosynthetic activity [49,116]. Maximum CO_2 daily fluxes in grasslands coincided with the period of maximum growth of the aboveground biomass when C highly partitioned to the roots enhanced respiration and was partly sequestered [38,49].

Ben-Asher et al. [15] showed that CO_2 flux is logarithmically related to the root size of corn and almond (Fig. 1). The relationship was closer for corn than almond. The root-associated respiration was considerably higher in grassland than in a barley field [116] and in native prairie than new bermudagrass and sorghum [38]. Measurements at the forest floor and above the trees showed that 77% of the carbon sequestered by tree canopy photosynthesis was lost to the atmosphere by root and soil microbial respiration [92]. Surface carbon dioxide fluxes were indicative of the shape and the size of root zone of almond [16].

Table 2. Effect of land use on net ecosystem exchange of CO_2 (g CO_2-C m^{-2} y^{-1})

Land use	Location	Study period	Fluxes	Reference
Grassland	Finland	Year	– 750*	[116]
Barley			– 400	
Brush	Arizona, USA	Year	– 144	[41]
Grass			– 128	
Brush	Arizona	Year(DT)	– 26	[41]
Grass	Arizona		+ 86	
Bermudagrass	Texas, USA	March-Nov.	–100 and +800**	[38]
Sorghum			+200 and -90	
Native prairie			+50 and +80	
Wheatgrass	Mandan, USA	Year	– 36 to +35***	[47]
Prairie			–19 to +52	
Wheatgrass	Mandan, USA	GS	+ 60	[47]
Prairie			+ 70	

DT day time; GS growing season; * – source, + sink;
** data for two years; *** range for three years.

Root respiration rates are largely influenced by temperature. Boone et al. [17] showed that respiration by roots plus oxidation of rhizosphere carbon is more temperature-sensitive (Q_{10} = 4.6) than the respiration of bulk soil (Q_{10} = 3.5). The authors indicated that if plants in a higher CO_2 atmosphere increase their allocation of carbohydrates to roots, as indicated in other literature, these findings suggest that soil respiration should be more sensitive to elevated temperatures, thus limiting carbon sequestration by soils.

In northern regions the carbon annual budget is affected by CO_2 flux during the dormant season (no photosynthesis) [47,48,116]. The data in Table 2 indicate that in certain regions [e.g. Mandan, USA] some land uses can either be a sink or a source for atmospheric CO_2 or near equilibrium, depending on the magnitude of the dormant season flux [47]. Dormant period fluxes in Northern Great Plains for the grasslands averaged 0.5 g CO_2-C m^{-2} d^{-1} (0.021 g CO_2-C m^{-2} h^{-1}) and 1.7-2.2 g CO_2-C m^{-2} d^{-1} (0.071-0.092 g CO_2-C m^{-2} h^{-1}) using respectively BREB [Bowen ratio/energy balance) and soil flux methods [47,50]. The dormant fluxes in Finland were from 1.13 in bare tilled soil to 1.56 g CO_2-C m^{-2} d^{-1} (0.047 to 0.065 g CO_2-C m^{-2} h^{-1}) in grasslands [116]. In case of over-winter cover crops and perennials such as alfalfa and poplar in south-east Michigan the fluxes were approximately 2 g CO_2-C m^{-2} d^{-1} (0.08 g CO_2-C m^{-2} h^{-1}) [139].

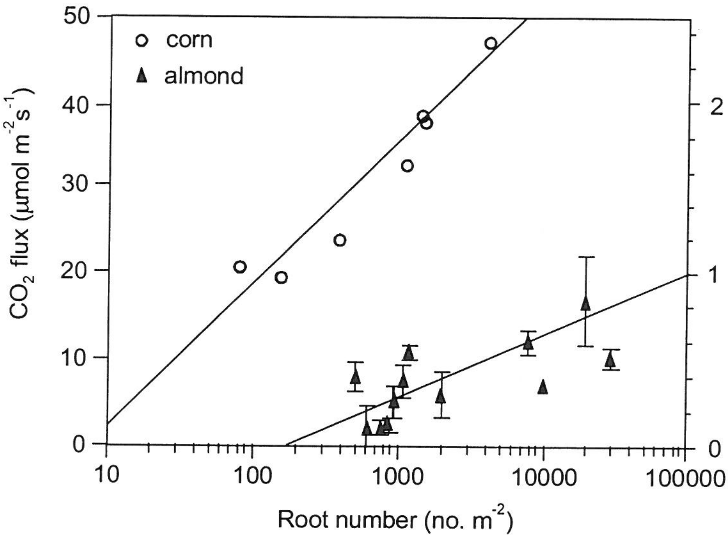

Fig. 1. CO_2 flux as a function of root number of almond (roots intersecting a cross section of 0.01 m^2 on the side wall) and corn (by a minirhizotron with 0.01 m length increment)[after 15]

2.1.3. Temporal and spatial variability

Carbon dioxide emissions vary temporally and spatially mainly according to land use and site conditions. In Hokkaido (Japan) soil respiration of volcanic ash soil supporting forest, grassland and cornfields showed high temporal variability and, on most occasions, was highest in grassland and lowest in forest (Fig. 2) due mostly to an effect of temperature accounting for 79-92% of the variability. The differences between the land uses were mostly pronounced during the summer period. As indicated by Q_{10} values soil respiration was the most sensitive to temperature in cornfield (4.8) and successively decreased in grassland (3.3) and forest (1.9). A substantial effect of temperature and, much less, of water content on temporal distribution of CO_2 have been reported in numerous studies [e.g. 48,102,112]. Under similar land uses in northern Europe, the CO_2 emission increased several times following wetting or thawing and this was most pronounced in grassland due to decomposition of carbon sources liberated from stressed by ice grass roots [143].

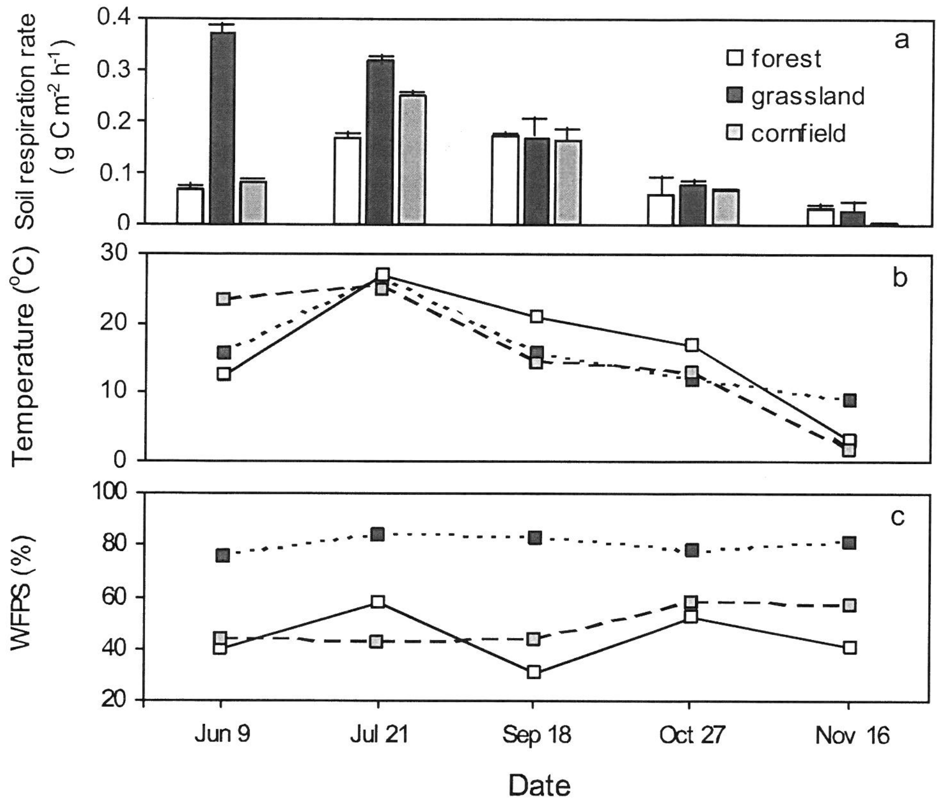

Fig. 2. Soil respiration, soil temperature at 5 cm depth and soil water filled pore space (WFPS) at 0-5 cm depth under different land uses; the temperature was measured during sampling time from 10 to 12 a.m. [after 74]

Spatial variation of emission of CO_2 from a bare acid oxisol in Brazil was correlated with soil carbon content, cation exchange capacity and free iron content and varied daily due to changes in weather conditions [101].

2.2. Cultural practices

2.2.1. Effect of tillage on carbon dioxide emissions

In general, no-till or reduced tillage in comparison to conventional tillage results in lower CO_2 emission [e.g. 103,109,146,154] and greater carbon sequestration in soil [13,117]. The gains in soil organic carbon under no-till increased linearly with increasing clay content and were greater in sub-humid than in semiarid sites [117]. Cumulative CO_2 fluxes after plowing were considerably greater from an established bermudagrass pasture than from a no-till sorghum field or a continuously cultivated sorghum field [146]. From a summary of 7 studies [148] reported that tillage-induced losses varied from 4 to 2000 kg CO_2-C ha^{-1} in the period from 1 to 66 days after tillage with rates from 2 to 147 kg CO_2-C ha^{-1} d^{-1} (0.0083 to 0.61 g CO_2-C m^{-2} h^{-1}) in northern America. The tillage effect on CO_2 emissions can be considerably greater in organic than mineral soils [116]. Also in organically-amended soils CO_2 emissions increased with earthworm populations [195]. Soil organic carbon exported in runoff and which depends on tillage system can be additional source of atmospheric CO_2 [82,100].

Lower rates of organic matter decomposition and CO_2 evolution with decreasing tillage intensity resulted in sequestration of crop-derived C and thereby increased the soil's ability to remove CO_2 from the atmosphere [35,52,62,193]. Using a global data-base of 67 long-term experiments in US, West and Post [194] indicated that a change from conventional tillage to no-till sequestered on average 57 g C m^{-2} y^{-1} (0.0065 g C m^{-2} h^{-1}). This figure was more than doubled (125 g C m^{-2} y^{-1} or 0.0143 g C m^{-2} h^{-1}) under Spanish conditions [154]. This impact can be enhanced when the C-building practices are strengthened because soil C sequestered due to reduced tillage is not stable and rapidly mineralized to CO_2 [35,117].

CO_2 emissions are substantially different before, during and after tillage. By monitoring CO_2 emission Wuest et al. [204] were able to detect peak emission in CO_2 during tillage and a rapid reduction immediately after tillage (Fig. 3a) and an increase afterwards. As shown in other studies the reduction was mainly during first day (Fig. 3b) and continued for a further fifteen days (Fig. 3c) but with much lower rate. The results are quite consistent despite different experimental conditions. Also in other studies [32,39,84,127,146] the CO_2 fluxes immediately and shortly after tillage were greatest. This effect is attributed to physical release of CO_2 by loosening and to an increase in respiration by organisms [84,204] and can be enhanced by warm temperature and availability of easily mineralizable organic matter [148]. Later during growing season the tillage effects on CO_2 emissions were less [13] and were related to crop type. For example in study of fluxes during growing season by Franzluebbers et al. [53], soil CO_2 evolution was greater under no-till compared with conventional tillage in sorghum and in soybean but the opposite was true in wheat. However, in experiment when intense rainfall occurred immediately after tillage and reduced greatly soil roughness, significantly greater CO_2-C emission in plowed compared to unplowed soil appeared only few days later [4].

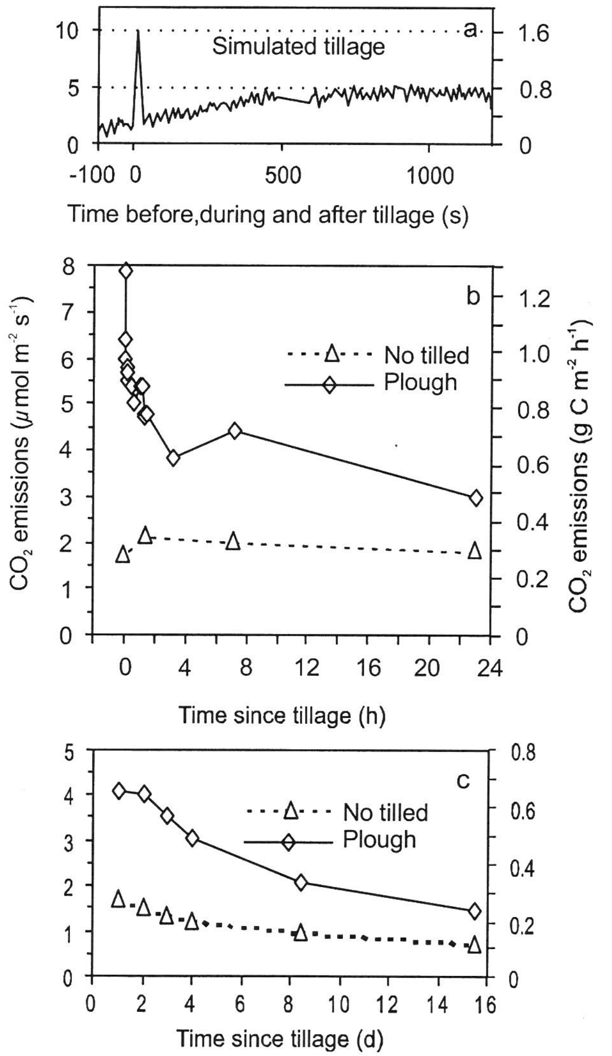

Fig. 3. Temporal variations of CO_2 emission before, during and after tillage: (a) silt loam WA, USA [after 204]; (b) sandy loam, Canada [after 148]; (c) dark red latosol, Brazil [after 103]. Note different time units in a, b and c.

In contrast to the above results, several workers showed a decrease in soil respiration after a soil disturbance [23,53,142]. In coarse soil this was attributed to low initial amounts of mineralizable C and N and microbial biomass [23].

Table 3. CO_2 emissions as related to tillage operations

Implement	Emissions (g CO_2-C m^{-2} h^{-1})	Reference
	Norfolk sandy loam, Alabama, USA	
None	0.079*	[144]
Kinze planter (5-8 cm)	0.101	
Coulter (10cm)	0.317	
Ro-til, coulters and rolling basket (45 cm)	0.554	
	Acid dark red latosol, Sao Paulo State, Brazil	
None	0.055**	[103]
Rotary tiller (20 cm)	0.087	
Chisel plough (20 cm)	0.131	
Disk plow (20 cm)	0.106	
Disk harrow (20 cm)	0.35	

*Means over 30-60 s after tillage, ** means over 2-week period.

The effect of tillage on CO_2 emissions depends on the type of implements used. The results presented in Table 3 indicate that CO_2 fluxes immediately (30-60 s) and shortly after tillage (two weeks) increased substantially with increasing depth and degree of soil disturbance. In addition, this increase can be associated with the bare soil in this period and associated greater emission of carbon dioxide [148].

Increased fluxes of CO_2 in disturbed soil were accompanied by enhanced water vapor fluxes and thus greater soil water losses [144,147]. On sensitive to compaction Norfolk loamy sand cumulative CO_2 emission under conventional tillage 80 h after was nearly three times larger than from no-till while corresponding H_2O losses was 1.6 times larger [147].

Tillage can affect soil surface CO_2 emissions by several mechanisms. In one mechanism the greater flux from the soil during and shortly after tillage was attributed to physical CO_2 release from soil pores and solution due to reduced resistance to gas transfer [146,204]. This explanation can be supported by observations of Calderón et al. [24] and Jackson et al. [84] indicating higher CO_2 emissions in tilled than non-tilled soils, despite lower or the same respiration rate. Other mechanisms of the tillage effects are associated with soil temperature and soil water content [4,53,188]. Kiese and Butterbach-Bahl [94] reported that CO_2 emission rates were positively correlated with changes in water filled pore space at dry to moderate soil water contents during the dry season, but were negatively correlated to the changes during the wet season.

Lower overall CO_2 emissions from no-till are also associated with reduced fossil-fuel use C emissions. Based on US average crop inputs, no-till emitted less CO_2 from agricultural operations than did conventional tillage, with 137 and 168 kg C ha^{-1} y^{-1} (156 and 192g CO_2-C m^{-2} h^{-1}), respectively [193]. In NE Italy minimum tillage contributed to the reduction of CO_2 emissions of between 200 and 300 m^3 ha^{-1} y^{-1} (0.45 and 0.67g CO_2-C m^{-2} h^{-1}) [18].

2.2.2. Soil aggregates

Soil tillage and other cultural practices may influence emission of CO_2 through their effect on soil aggregation and organic carbon content. Dexter *et al.* [34] reported that shallow (ploughless) tillage resulted in higher organic carbon (Corg) content in the surface layer and lower Corg content in the deeper layer than the ploughed treatment and basal respiration rates of the aggregates were positively correlated with the Corg contents. This study indicated that some of the Corg is "physically protected" against microbial activity primarily through interactions with clay particles. This protection can be greater in not tilled than tilled soil due to decreasing clay content in the latter, likely due to illuviation and surface runoff [99]. In general, increasing aggregate size was associated with decreasing carbon dioxide production and biomass C [51,159,165], availability of nutrients [157] and evaporation [199,200]. and increasing saturated water conductivity [199]. Reduced carbon dioxide production implies also increasing C sequestration.

3. NITROUS OXIDE

Nitrous oxide is a natural trace gas occurring in the atmosphere. In soils, it is mainly produced from mineral N during the microbial process of nitrification and denitrification [e.g. 14,114]. The annual global emission of N_2O from soils is estimated to be 10.2 Tg N or about 58% of all emissions [124]. Most of the nitrous oxide in the atmosphere, thought to be involved in global warming and the depletion of the stratospheric ozone layer is emitted from soil [19,31,160,168].

Soil N_2O emissions are mainly controlled by the availability of a suitable substrate (nitrogen), soil temperature and factors that reduce the redox potential, e.g. soil wetness, fine soil texture and organic carbon [153,173]. Many activities induced by land use and cultural practices affect these factors.

3.1. Effect of land use on nitrous oxide emissions

In general, N_2O emission from fertilized grasslands is greater than from crop-ped fields [58,170], forests [54,58] and woodlands [168]. The data in Table 4 illustrate this.

Table 4. Effect of land use on N_2O emissions

Land use	Study period	Emission	Original units	Converted unit (g N_2O-N ha^{-1} day^{-1})	Location	Reference
Grazed grassland	4-12 months	57-107	μg N_2O-N m^{-2} h^{-1}	14-21	UK	[170]
Cut grassland	Year	11-59		2.7-14		
Potatoes	May-December	2 8		6.7		
Cereal crops	April-December	5-14		1.2-3.4		
Grasslands	1-2 years	14-32	kg N $ha^{-1}y^{-1}$	38-87	Belgium	[58]
Arable lands	,",	0.3-1.5		0.8-4.1		
Forest	,",	1.3	kg N ha^{-1} $2y^{-1}$	1.78		
Permanent pasture	Year	0.03-0.99	g N_2O-N $ha^{-1}h^{-1}$	0.7-23	New Zeal.	[26]
Cornfield	,",	0.04-1.35		1-32		
Permanent pasture	Dec.-Sept	1.66	kg N_2O-N $ha^{-1}y^{-1}$	4.5	New Zeal.	[27]
Cropfields	,",	9.2-12.0		25-33		
Cropfield	Year	8.3-11	kg N_2O-N ha^{-1} y^{-1}	22.7-30.1	Finland	[115]
Forest	,",	4.2		11.5		
Bare soil	,",	6.5-7.1		17.8-19.4		

Under New Zealand conditions, however, N_2O emissions from the permanent pasture were significantly lower than under cropfields [27]. In the mowed grassland, denitrification is enhanced by anaerobic microsites in the surface horizon. Large emission of N_2O in the mown grassland was ascribed to enhanced release of carbon compounds from roots, which stimulates denitrification [10,168]. Under intensively managed and heavily fertilized (up to 500 kg N ha^{-1}) grasslands, N_2O-N loss per unit of fertilizer N applied can be larger than the 1.25% used for the global emission inventory [58].

Conversion of grassland to wheat field resulted in 8 times higher emission for 18 months and 25-50% higher after 3 years [125]. However, after returning the wheat field to grassland, mean N_2O emission rates were similar in both land uses. A modelling study by Mummey et al. [129] using more than 2900 cropland and grassland sites in USA showed that initial conversion of arable agricultural land to no-till resulted in greater N_2O in drier regions and similar or less in warm and wet areas compared with conventional tillage.

A large proportion of N_2O from agricultural soils is emitted during winter. Mosier *et al.* [125] showed that higher emissions from grassland than from arable fields during winter are due to greater snow accumulation and denitrification events. Disregarding the emission during the off-season period can lead to serious underestimation of the actual annual N_2O flux.

3.2. Cultural practices

3.2.1. Effect of tillage on nitrous oxide emissions

Increased N_2O emission from no-tilled compared to tilled soil has been reported in number of papers [e.g. 8,81,100,114,129] with maximum difference as much as several fold. This may be due to increased availability of C [136] and a greater contribution from large aggregates [107] with anoxic centres [67,70] and reduced air-filled porosity [7,13] under no-till. However, in experiment of Arah *et al.* [7] despite consistently higher denitrification rates in not tilled than ploughed soil N_2O emissions were very small due to low gas diffusivity in the soil near the surface. No differences in N_2O emission were found between conventionally tilled and not tilled soil with a short history of continuous tillage [26].

3.2.2. Effects of soil compaction

The risk of N_2O emissions increases with soil compaction. Accumulated denitrify-cations during 75 days from wheeled and unwheeled wheat field were 3-5 kg N ha^{-1} (40-67 g N ha^{-1} day^{-1}) and 15-20 kg N ha^{-1} (200 -267 g N ha^{-1} day^{-1}), respectively [9] and from a potato field up to 68% of total N_2O release was emitted from the compacted tractor tramlines [152]. N_2O emissions in compacted and uncompacted sandy loam corresponded to 5.3 and 3.9% of added NH_4NO_3-N, respectively [63]. In another study on the same soil, compaction effect was four times higher in the NPK-fertilized treatment compared to the unfertilized one [164]. In general, the increased N_2O-N emission from compacted soil was accompanied by greater N_2O concentration in the soil air. Enhanced N_2O emissions from compacted soil were attributed to increased water filled pore space (WFPS) or reduced air-filled porosity [37,110]. Therefore they were most pronounced after rain events [13,63,68,97] and poorly drained clay soils [166,171]. Another factor, similarly to no-till soil, was increased contribution of large aggregates (>20 mm) [9]. Figure 4 clearly illustrates increasing N_2O emissions from compacted potato inter-rows in periods of increased WFPS following precipitation. Increased emission from the ridges in August was attributed to herbicide killing of potato tops.

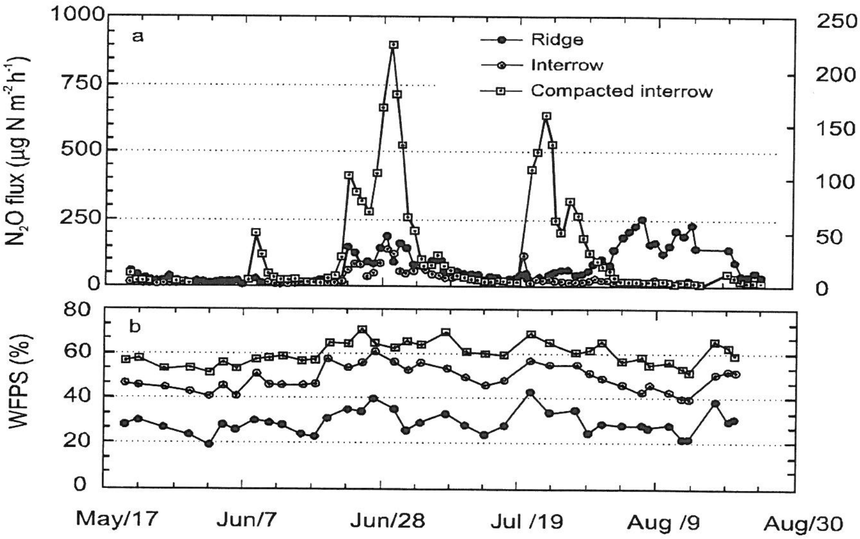

Fig. 4. N_2O fluxes and water-filled pore space (WFPS) during the potato growing period [after 46]

The presence of a compacted layer in deeper soil can also largely affect denitrification. This was shown in column experiments where the highest denitrification rate was from the 20-30 cm layer containing high organic carbon content and of high bulk density (Fig. 5). This is supported by results of Hatano and Sawamoto [68] showing significant N_2O flux from the subsoil of onion field with high rate of N fertilization. The above results imply that denitrification from the subsoil layer can significantly contribute to the emission from the whole soil profile provided that nitrate is available.

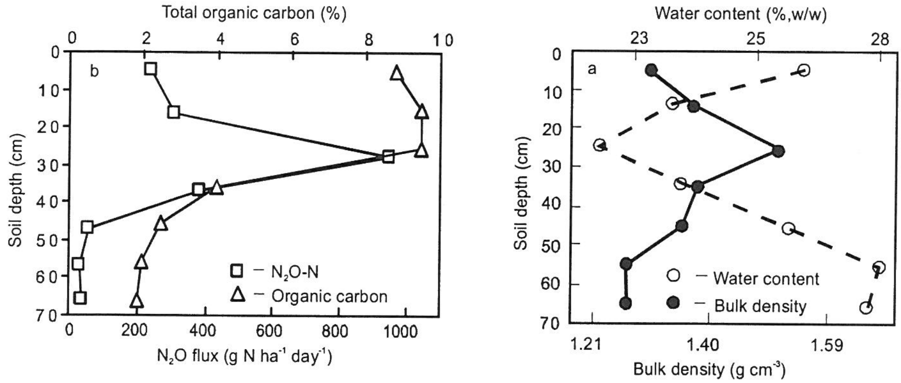

Fig. 5. Relation of N_2O flux to soil depth, carbon content, water content and bulk density [after 56]

Soil compaction by livestock hooves has been shown to increase N_2O emissions from grasslands. In addition uneven distribution of soil compaction and localized very high inputs of excretal N resulted in large temporal and spatial variability of the N_2O flux (Fig. 6). Intensive grazing by stock resulted in doubled N_2O emissions compared to occasional sheep grazing [26]. The highest fluxes from grazed grasslands predicted by regression model were 6-21 kg N ha^{-1} y^{-1} (16-57 g N ha^{-1} day^{-1}). The global contribution of grazing animals was estimated at 1.55 Tg N_2O-N per year and is more than 10% of the global budget [134].

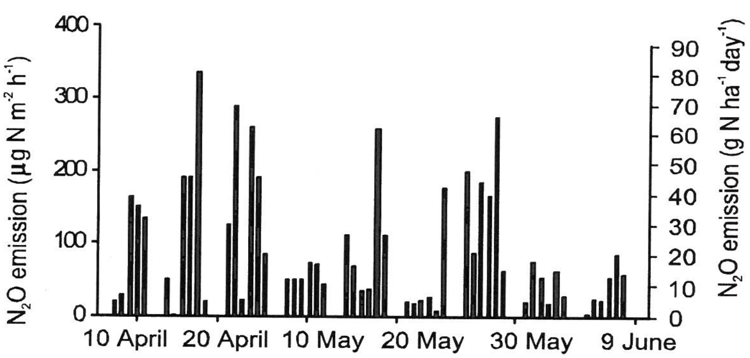

Fig. 6. The spatial and temporal variability in N_2O emission between six chambers at the same grazed grassland field. Chambers were removed in between flux measurements and were always relocated into the same position. Individual bars at each date correspond to the flux from chambers 1-6. [after 168]

3.2.3. Effect of fertilisation on nitrous oxide

In general, N_2O emission increases with increasing nitrogen fertilization [e.g. 68, 89,122]. Based on published measurements of nitrous oxide Bouwman [20] related the total annual N_2O-N emission (E) to the N fertilizer applied (F): $E = 1 + 0.0125 \cdot F$ with E and F in kg N ha^{-1} yr^{-1} ($r^2 = 0.8$) (Fig. 7). The prediction is independent of the type of soil fertilizer and can be appropriate for global estimates.

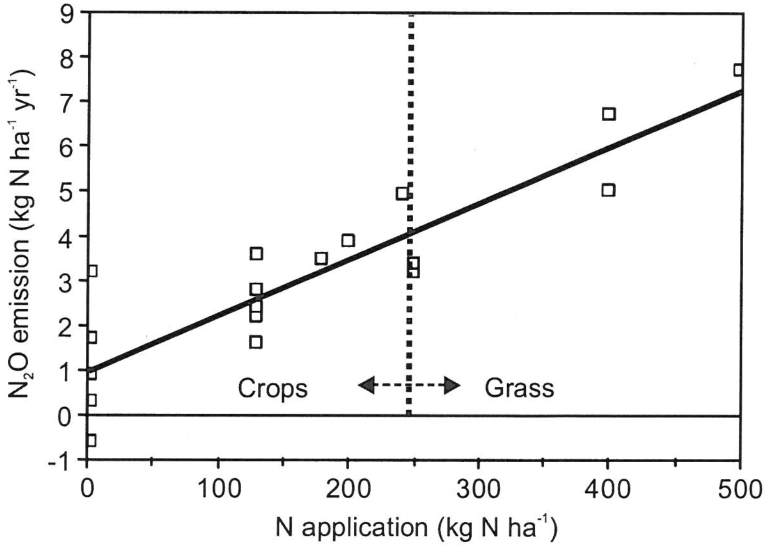

Fig. 7. Relationship between N fertilizer application and N_2O emission from mineral soils for N application rates <500 kg N ha^{-1} with a measurement period of one year [after 20]

In estimates of the emission at regional scale, effects of other factors such as fertilizer type, soil wetness and pH were considered. Akiyama and Tsuruta (3) showed that annual N_2O emissions from the Pac choi (Brassica spp.) fields fertili-zed with poultry manure, swine manure, and urea (15 g m^{-2} yr^{-1} in each case) were 1.84, 0.613, and 0.448 kg N ha^{-1} yr^{-1} (5.04, 1.68 and 1.83 g N ha^{-1} day^{-1}). Extensive studies in Germany [89,120] showed that annual N_2O losses ranged from 0.53 to 16.8 kg N_2O-N ha^{-1} with higher emissions from organically fertilized plots as compared to mineral or cattle slurry fertilized plots. This can be due to higher microbial biomass and suitable carbon-pools available for minerali-zation [120]. In the case of short term application of farmyard manure the inverse effect was predicted in a modeling study due to nitrogen use to build up soil organic matter [158].

Major effects of fertilization on N_2O emission were frequently observed in wet soil following rainfall or irrigation and this effect was mostly pronounced during the days immediately following fertilizer application [81,160,182]. For example soil fertilized with urea emitted during 120 days the highest amount of N_2O (1903 μg N_2O-N kg^{-1} soil) at field water capacity while that with NH_4NO_3 gave the highest emission (4843 μg N_2O-N kg^{-1} soil) when flooded [137]. In rice fields N_2O emission can be substantially reduced, down to less than 0.1% of the applied nitrogen, by applying fertilizer after rather than before flooding and thereby reducing a source of N_2O during wetting and drying cycles before permanent flooding [54]. Nitrous oxide emissions from unfertilized tropical sites were typically higher than those in temperate sites whereas those from fertilized sites were within the range for fertilized temperate sites [122]. In fertilized tropical grasslands the N_2O emission was almost three times greater from a Vertisol (130 μg N m^{-2} hr^{-1} or 31 g N ha^{-1} day^{-1}) than from an Ultisol and Oxisol (46 μg N m^{-2} hr^{-1} or 11 g N ha^{-1} day^{-1}) [122]. However, there was no significant difference across soils when they were not fertilized. In cooler climates the greatest N_2O fluxes occurring in association with freeze-thaw in spring may be minimized by applying N fertilizer and incorporating straw [64].

An important soil factor associated with N fertilization is pH. Tokuda and Hayatsu [183] reported that the application of more than 1 kg N ha^{-1} yr^{-1} of nitrogen fertilizer significantly enhanced the N_2O emission potential of acidic tea fields where a negative exponential relationship was found between the soil pH value and N_2O emission. Substantial reduction of N_2O emissions was observed when acid soils were both limed and fertilized [123].

3.2.4. Effects of crop type and rotations

The proportion of N-fertilizer released as N_2O depends on the type of crop, due to differences in growth and development, as well as crop rotation and crop residue management [81,87,88].

The N_2O emission during the growing season of cereals ranged from one to several percent depending on crop type and experimental conditions [86,97,208]. In general, the emission can be reduced in an actively growing crop and associated quicker nitrogen uptake [13,189]. For this reason Kusa *et al.* [97] revealed that about 70% of the annual emission occurred near harvesting of onions.

N_2O emission in a continuous cropping was higher for corn than for soybean [81,114] or alfalfa [114] which implies that legumes would reduce N_2O emission. However, in other study [54] legumes contributed in the emission due to denitrifying of nitrogen fixed by symbiotically living Rhizobia in root nodules.

3.2.5. Spatial and temporal variation of nitrous oxide emissions

Spatial distribution of zones with different potential of N_2O emission occurs in the scale of soil aggregates and of bulk soil. Using combined N_2O and O_2 microsensors [69] and technique of peeling the aggregates [70] allowed to show that soil aggregates contain anoxic centers with high denitrifying activity. These centers were much more pronounced when acetylene was injected to inhibit reductase of N_2O (Fig. 8a,b). The denitrifying activity was stimulated by organic matter [203], which greatly stimulated respiratory activity and caused anoxia-enhancing denitrification in hot spots [69]. The amount of N denitrified increased with aggregate diameter from 2 to 23 mm [70] or decreased when C substrate supply in larger aggregates was limited [159].

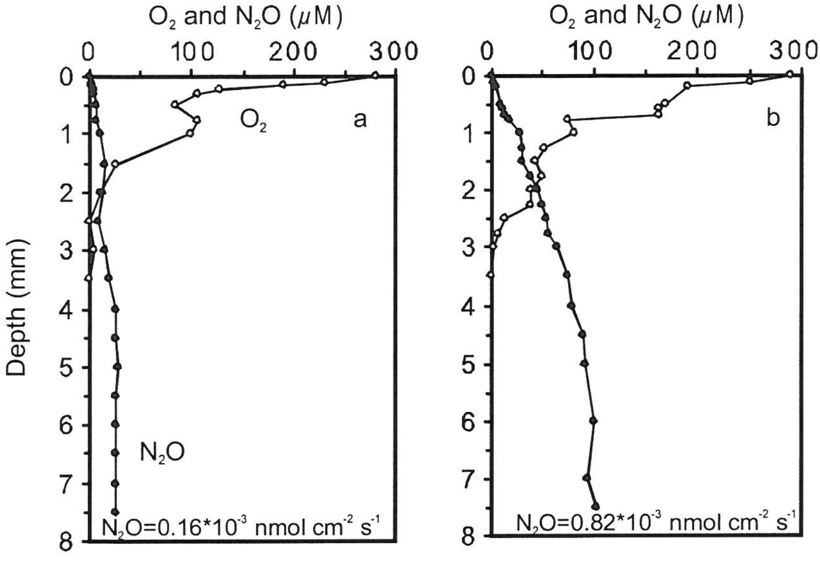

Fig. 8. Example of O2 and N_2O concentrations as a function of depth within soil aggregates measured at 6 mm from the center of a decaying clover leaf (6 mm diameter) placed at the aggregate surface, a: without acetylene addition; b: with acetylene. The diffusion fluxes indicated were calculated from the linear N_2O profile [after 69]

In arable soils spatial variability of N_2O emission was associated with extremely high N_2O emission rates from areas of a few square centimeters to a few square meters [10,27,151]. The hot spots were often attributed to high soil nitrogen concentration and tillage induced microreliefs [29,81,151], natural soil heterogeneity and the measurement technique used [27]. In no-tilled soil, substantially higher N_2O emission was recorded from soil enclosing a drill slit (458 μg N m^{-2} h^{-1} or 110 g N ha^{-1} day^{-1}) than between slits (207 μg N m^{-2} h^{-1} or 49 g N ha^{-1} day^{-1}) [13].

High local N_2O fluxes can coincide with wet depression in the ground. Ambus and Christensen [5] reported that patterns of N_2O flux at the scale beyond 7 m was controlled by soil moisture variability due to ground topography and at the scale below 1 m – by a patchy distribution of denitrifying microsites. Uneven N_2O flux can be also induced by earthworm casts that produce several times more nitrous oxide than bulk soil [40]. Knowledge of the small-scale spatial variability helps improve estimates of the emissions over a larger scale [128]. For representative N_2O loss estimation, Röver *et al.* [151] suggested measurements with a distance of 1 m between sampling points. However, Yanai *et al.* [211] showed that by combining principal component analysis with geostatistics, a map of predicted N_2O fluxes based on soil properties closely matched the spatial pattern of N_2O fluxes which was measured in 10 m by 10 m grids in an onion field. The N_2O fluxes were highly variable with an average of 331 μg N m^{-2} h^{-1} (79.4 g ha^{-1} day^{-1}) and CV of 217%.

A spatial patterns of N_2O fluxes often persists for short time and diurnal changes are largely influenced by cycles in soil temperature [27,170,197] and rainfall and irrigation events through the effects on air-filled porosity [173].

In northern areas with freezing/thawing cycles, approximately half of the annual N_2O emission occurs during winter and at thawing [89,115,181]. This effect is relatively high in farmed organic soils [143] and is enhanced by manure application and crop residue incorporation [87,88,150,189]. Peak emissions during soil thawing were explained by the physical release of trapped N_2O and/or enhanced denitrification with increasing temperature in the very wet soil [181]. In some other studies, however, fluxes during winter and other periods were comparable [86,197].

3.2.6. Main soil factors affecting nitrous oxide emissions

Studies conducted in more controlled conditions revealed that soil moisture, temperature and availability of NH_4^+ and NO_3^- are the most important factors regulating the N_2O emissions.

The effect of soil moisture was well demonstrated in Australia where the emissions were <20 μg N_2O-N m^{-2} h^{-1} (< 4.8 g N_2O-N ha^{-1} day^{-1}) in dry seasons and from 80 to 242 μg N_2O-N m^{-2} h^{-1} (19.2 to 58.1 g N_2O-N ha^{-1} day^{-1}) in wetter seasons [94]. With rice, high N_2O emission occurred when upland was converted to flooded fields [205,210]. The emission from flooded field was mostly through the rice plants (87%), while in the absence of floodwater – mainly through the soil surface.

Water filled pore space (WFPS), which includes differences in bulk density and particle density has often been used to characterize soil moisture conditions [e.g. 10,168,180]. In general, N_2O emissions were positively correlated with changes in WFPS up to 50-60% [94] and largely increased at WFPS >60% [107,160,170,182].

Many authors [e.g. 97,170] reported positive relation between N_2O emission and temperature. This relation was enhanced at high WFPS ($Q_{10} = 5$) [169].

Other important factors influencing N_2O emission include soil organic carbon [168,202,203], acidity [14,183] texture [57] and redox potential [80,173,201]. Włodarczyk *et al.* [202] reported that N_2O emission was about 4 times in organic (peaty-muck) than mineral sandy soil.

4. METHANE

The main land use contributors to the atmospheric methane concentration implicating global warming are cultivated wetlands (rice fields) and natural wetlands [104]. Global estimates of annual CH_4 emission from rice fields is 100 Tg. Methane is produced in the anaerobic zones through fermentation by methanogenic bacteria (domain Archea). Its transfer from the soil to the atmosphere occurs mostly through aquatic plants, but also by diffusion and as bubbles escaping from the wetland soils [104,130]. In the aerobic zones of wetland and upland soils methanotrophic bacteria using CH_4 as only a C and energy source can oxidize it to CO_2. Soil management may account for 20% of overall CH_4 emissions [130].

4.1. Land use

4.1.1. Wetlands

Methane emissions from peat wetlands have been shown to be very sensitive to soil temperature, the extent of standing water and the depth to the water table (Fig. 9), factors that determine the anoxic and unsaturated zones of CH_4 production and consumption. The mean methane emission approximately doubled with temperature in the range 7-11°C. A similar increase was observed in diffusion of methane when the temperature around roots in hydroponic culture increased from 15 to 30°C [71]. Adding annual soil temperature sum in a multiple regression model significantly accounted for a significant factor of variance of methane emission from a boreal mixed mire in northern Sweden [59].

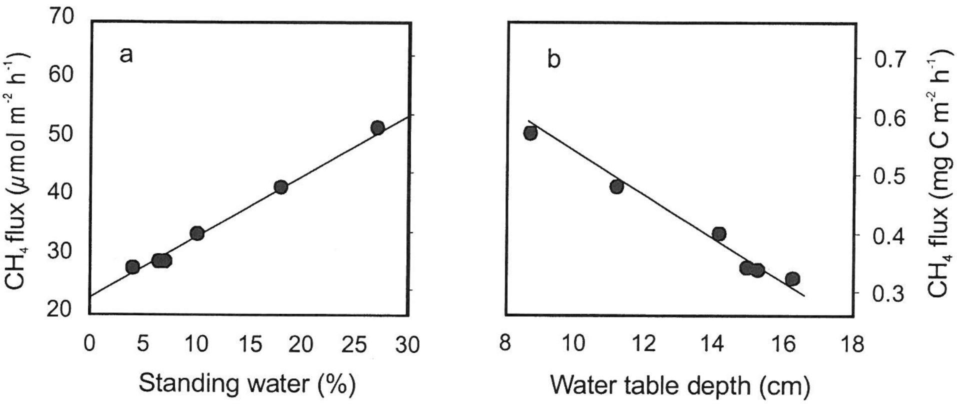

Fig. 9. Effect of proportion of standing water (pool area) (a) and water table depth (b) in a bog in Scotland [after 65]

Vegetation plays an important role in CH_4 emission from the wetlands. Higher CH_4 emission from the planted than from unplanted wetlands is attributed to the transport functions of rice plants from below ground to the atmosphere [85,95,130]. Well-developed intracellular air spaces (aerenchyma) in rice plants provide a transport system for the conduction of methane from the bulk soil into the atmosphere [119,131,132] by molecular diffusion and concentration gradient [71]. They act as chimneys for CH_4 transport to troposphere [163]. Also emission of N_2O was to high extent through the flooded rice fields (section 3.2.6). At the same time root exudates serve as a substrate for methanogen bacteria and the roots transport atmospheric O_2 to rhizosphere, stimulating CH_4 consumption [95,163]. From a 13 CO_2 applied pot experiment, Kimura [95] showed that 22-39% of photosynthesized carbon by rice plant was emitted as CH_4.

Transport of CH_4 to the atmosphere depends on specific transport abilities of rice cultivars. The relative per cent decrease in methane emission among six rice cultivars was found to be in the range of 1-42.6% [119]. Wang *et al.* [191] showed that the traditional variety Dular emitted 27% more CH_4 than IR72 and 177% more than IR65598. Also tall varieties emitted on average 62% more CH_4 than semidwarf varieties of which mean emission was 185 kg CH_4 ha^{-1} [105]. In general, emission increases with increasing rice plant biomass and root size [119,191,213].

Among the edaphic factors, redox potential (Eh) was often related to the action of methanogenic bacteria. Negative correlation between the soil Eh and methane emission was reported in several papers [e.g. 44,163,173,206]. Figure 10 illustrates this relationship during growing season of rice. Fiedler and Sommer [43] indicated that because of the linkage of ground water level and soil organic matter to

Eh, methane emissions may be estimated by coupling unit area emissions to soil maps based on soil morphology (e.g. soil taxonomy). Stępniewski and Stępniewska [175] reported that beginning of CH_4 production in a soil starts below 50 mV with maximum emission on the level of –150 mV.

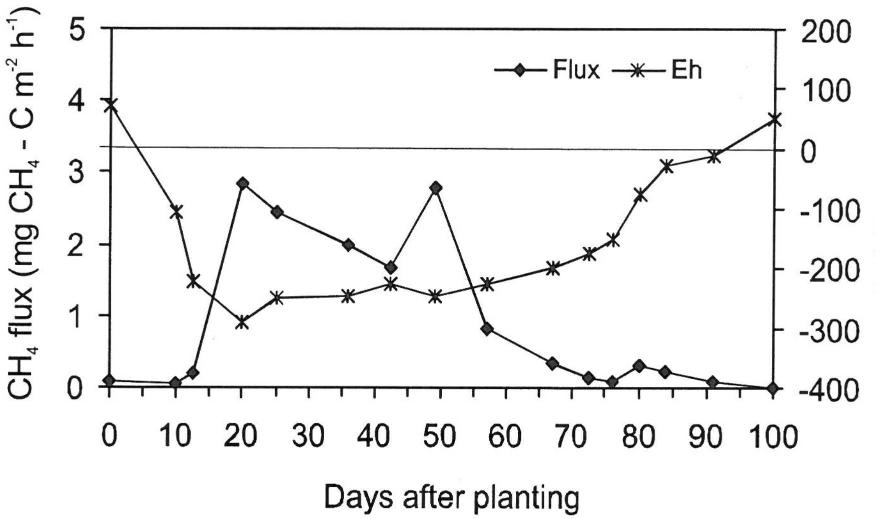

Fig. 10. Methane flux in relation to soil redox potential (Eh) during growing season of rice [after 119]

4.1.1.1. Effect of water management on methane emissions

Water management is a most important factor for CH_4 emission from rice fields [65,104,131,207]. Anoxic conditions and flooding favored methane release per unit area from different rice ecosystems in the order: deepwater rice>irrigated rice>rainfed rice [131]. Therefore appropriate drainage and associated drying significantly reduced CH_4 emission [130,207,212]. Yagi *et al.* [207] showed that intermittently draining practice in rice fields decreased CH_4 emission by more than 40% compared to continuous flooding practice, an effect which can be attributed to both low CH_4 production and high CH_4 oxidation [192]. Upon drying at harvest, large amounts of entrapped CH_4 escaped to the atmosphere when floodwater receded [130]. However, CH_4 flux in wheat was unaffected by the irrigation treatments [137]. In modeling study of Granberg et al. [59] mean water table level was the single most important predictor of simulated annual methane emission from from a boreal mixed mire ($r^2 = 0.58$).

4.1.2. Upland soils

In general, dry and aerated soils act as sinks for atmospheric CH_4. Numerous studies indicated that higher absorption potential occurs in forest or woodland than other land uses such as grasslands [76,123,186,198], cultivated soils [36,76, 113,197], moorland [113] set aside land [36]. Summary of results obtained in 27 studies under temperate climatic conditions made by Hütsch [76] indicate that the ranges in oxidation rates in mg CH_4 m^{-2} day^{-1} were 0-1.03, 0.03-1.16 and 0-6.9 (or 0-0.03, 0.0009-0.036, 0-0.22 mg C m^{-2} h^{-1}) for arable soils, grasslands and forest soils, respectively. Atmospheric CH_4 oxidation potentials in forest soils exceeded production potentials by up to 10-220 times [21]. There was not significant difference in CH_4 uptake between bare soil and soil cropped with onion [73].

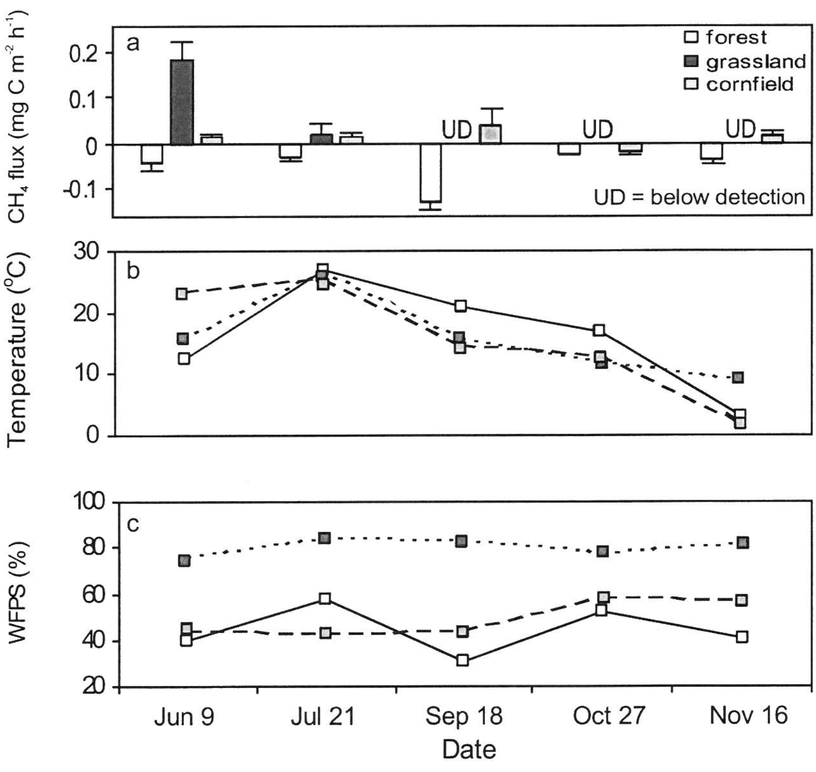

Fig. 11. CH_4 flux as related to soil temperature at 5 cm depth and soil water filled pore space (WFPS) at 0-5 cm depth under different land uses; the temperature was measured during sampling time from 10 to 12 a.m.[after 74]

Frequently, methane oxidation by aerobic upland soils is low (< 0.1 mg CH_4 m^{-2} h^{-1} or <0.075 mg C m^{-2} h^{-1}) relative to its emission in wetlands (approximately 10 g CH_4 m^{-2} h^{-1} or 7.5 g C m^{-2} h^{-1}) [73,104]. Measurements on several occasions during the year in Hokkaido (Japan) showed that CH_4 was emitted from grassland, absorbed and emitted from cornfield and absorbed by soil under forest (Fig. 11).

Tropical ecosystems play an important role in the production [186] and uptake [42,125,186] of atmospheric methane. In the course of one year, fertilized tropical pastures showed net CH_4 uptake (−0.34 mg CH_4 m^{-2} day^{-1} or –0.011 mg C m^{-2} h^{-1}) while traditional and legume pastures displayed net CH_4 emissions (+0.69 and +0.92 mg CH_4 m^{-2} day^{-1} or +0.022 and + 0.029 mg C m^{-2} h^{-1}) [186]. In other tropical sites pasture soils showed a net emission even during the dry season [42]. In the tropics the CH_4 uptake rates for forests were 10-fold higher than those from forage production sites [122].

The uptake rate of CH_4 did not appear to be related to any single variable. It was mostly ascribed to high relative gas diffusivity D/Do, air-filled pore space in the uppermost soil horizons [21,22,74,176] and air permeability [11]. This was due to enhanced gas exchange and favored development of methanotrophs that use methane as energy while oxidizing it to CO_2. In addition the CH_4 oxidation rate was negatively correlated with soil moisture content [36,42,161] and bulk density [113] that affect gas diffusion coefficient and air-filled porosity.

4.1.3. Effect of fertilization on methane exchange

Nitrogen fertilization either has no affect [e.g. 75,76] or reduces absorption and oxidation of CH_4 by the soil [e.g. 73,126,152]. The inhibitory effect of N inputs can be due to decreased soil oxygen content in organic matter reach sites that easily become anoxic [162] and increased population of nitrifies at the expense of methanotrophs [77,78]. Evidence for latter mechanism was supported by higher nitrate content in fertilized than unfertilized soil cropped with onion [73]. In addition, nitrogen fertilization enhances soil acidification and this may decrease CH_4 oxidizing activity [76]. Ruser et al. [152] reported that the annual CH_4 uptake was 140 and 118 kg C ha^{-1} (or 1.5 and 1.3 mg C m^{-2} h^{-1}) for low (50 kg N ha^{-1}) and high (150 kg N ha^{-1}) levels of fertilization, respectively. Fertilizer addition had a small negative effect on CH_4 uptake in the Vertisol, tended to enhance CH_4 uptake in the Ultisol and significantly decreased CH_4 uptake in the Oxisol in the tropics. In highly N-fertilized onion-cultivated soil, Sawamoto *et al.* [156] observed that large part (58% of direct emission) of the CH_4 was emitted indirectly through the subsurface drainage after leaching and dissolving [122].

Usually atmospheric CH_4 oxidation is reduced more by ammonium and urea fertilizer than nitrate [104,186,212] and by potassium nitrate than ammonium sulphate [108]. The negative effect of urea can be alleviated by its subsurface application in ploughed soils where atmospheric CH_4 uptake will not be hampered too much whereas on no-till soils urea should be avoided because of the highest CH_4 oxidizing potential at 5-15 cm depth [76].

Exogenous organic matter (OM) influences methane production. Addition of organic matter to flooded soil resulted in a several times increase of CH_4 emission relative to plots with mineral fertilizers [e.g.130,138,161]. The difference of the effect by addition of rice straw compared to urea was most pronounced two months after transplanting [130]. In another study [33] addition of OM resulted in 3-12 times higher CH_4 emission with largest emissions when OM was added deeper in the soil. The distribution of fresh organic material activating methanogenic activity was the most dominant factor for the microscale (1 cm) spatial variation in CH_4 production [190].

Combined use of inorganic N and organic manure increased the CH_4 emission from saturated rice soil to 172% compared to application of the entire amount of N through urea [138] and reduced its consumption from dryland rice field [161]. However, in the case of unflooded arable soils long-term farmyard manure application inhibited CH_4 oxidation less than application of the same amount of N as mineral fertilizer [76]. This effect was attributed with population of methanotrophic bacteria.

4.1.4. Effect of tillage and compaction on methane exchange

Some studies showed that methane oxidation potential of upland soils may best be preserved by no-tillage [13,76,93,104] because soil tillage disturbs methane-oxidising microorganisms by disrupting soil structure and releasing soil-entrapped methane. In some tropical soils, however, cultivation had little effect on soil methane uptake [90,124].

Transplanting 30-d-old rice seedlings, direct seeding on wet soil and direct seeding on dry soil reduced CH_4 emission by 5%, 13% and 37%, respectively, when compared with transplanting 8-d-old seedlings [96]. In the case of organic amendment, fall plowing compared with spring plowing was a more effective way of mitigating CH_4 emission from rice fields.

An important factor affecting methane oxidation and emission is soil com-paction [104,152]. In a potato-cropped field, the ridge soil and the uncom-pacted inter-row soil had mean CH_4-C oxidation rates of 3.8 and 0.8 μg m^{-2} h^{-1} (0.0038 and 0.0008 g CH_4-C m^{-2} h^{-1}), respectively, [152]. However the tractor-compacted soil in this study emitted CH_4-C at 2.1 μg m^{-2} h^{-1} (0.0021 g CH_4-C m^{-2} h^{-1}) due to anaerobic condition. Similar response was observed by Flessa *et al.* [46].

5. NITRIC OXIDE

Nitric oxide is produced in soils as a result of microbial activity through the processes of nitrification. NO emission to the lower troposphere leads to an increase in the concentration of photochemical oxidants, particularly O_3, which adversely affects human health, animals, and plants [e.g. 121,141] and reduces the CH_4 sink. It also reacts with water vapor to form nitric acid and nitrous acids, which acidify precipitation and increase N deposition [55,169,172]. Production and emission of NO are mostly controlled by environmental variables such as inorganic nitrogen availability, soil water content and soil temperature. Land use and cultural practices largely influence these variables.

5.1. Effect of land use and soil tillage on nitric oxide emissions

The effect of land use on nitric oxide emissions has not been intensively studied. From the review of Skiba *et al.* [167] nitric oxide emissions are relatively high in pastures and considerably higher than in forests. Under cropping systems the rate of NO emission is greater for crops with higher fertilizer requirements (e.g. maize vs. wheat). In tropical climate of Costa Rica NO emissions were several times greater in young pastures but lower in older pastures than in mature forests [91,187].

Tillage, in general, stimulates NO emissions, both in temperate and tropical soils. Skiba *et al.* [167] reported that conventional tillage and green manure incorporation are likely to increase NO emissions by a factor of 2 to 7 for mostly periods of 1 to 3 weeks compared to untilled soil. This was confirmed by more recent studies [28] where NO emission during summer on Maryland's Eastern Shore from an untilled field of 1.2 ng N m^{-2} s^{-1} was significantly lower than from a tilled field (8.6 ng N m^{-2} s^{-1}). This effect was attributed to an increased amount of soil exposed in the tilled field allowing easier physical transfer of NO out of the soil. These suggest that minimum cultivation strategies can significantly reduce NO emission. Analysis of data from various sites showed that on average tillage is likely responsible for emission 0.5 kg NO-N ha^{-1} y^{-1} (0.0016 ng NO-N m^{-2} s^{-1}) [167].

5.2. Effects of soil physical properties on nitric oxide emissions

Soil water as characterized by the water filled pore space (WFPS) is of critical importance to NO production and transport to the troposphere [141,167,178,179]. The threshold WFPS between water-limited (<60%) and oxygen-limited microbial processes or limited escape of NO to the atmosphere seems to be 60% in various soils [25,167]. This value was lower (about 43%) under controlled wetting in laboratory conditions. Peirce and Aneja [141] reported that more than 42% of the total NO flux comes from the top 1 cm of soil, with NO contributions decreasing exponentially with soil depth and very little from soil at a depth of 20 cm or greater. Tabachow *et al.* [179] showed that soils with increasing WFPS (3-40%) and temperature (15-28°C) generally produce greater quantities of NO. Skiba *et al.* [167] reported that soil temperature and soil NO_3^- concentration accounted for 60% of the variability in the NO emission, for a range of agricultural and seminatural soils. In dry soils of different climates, however, NO flux was inhibited and had no relationship with temperature [167,178].

These two physical parameters and soil nitrogen content are considered to have a reasonably consistent relationship with NO flux and were used as variables in modeling approaches [149,178]. One factor limiting accurate estimates of NO emission is high variability, incorporating the spatial and temporal effects of management practices could increase performance of computer modeling [6,61].

5.3. Effect of fertilization on nitric oxide emissions

From two comprehensive reviews results that fertilizer-induced nitric oxide emissions vary in a wide range from 0.003 to 11% of applied nitrogen (0-800 kg) [167,184] and average emissions (0.3-0.5%) were several times lower than the 2.5% used in modeling prediction of global NO emission by Yienger and Levy, quoted by Skiba *et al.* [167]. Using the data from field experiments covering at least a complete growing season and without legumes providing symbiotic nitrogen, Veldkamp and Keller [184] obtained a linear relation between rate of N application and NO emission (Fig. 12). Despite significance of the relationship the authors pointed out that care should be taken in extrapolating these results due to high uncertainty ($R^2 = 0.64$) associated with a variety of fertilizer types, soils, and climate conditions. NH_4^+ based fertilizers and urea caused often larger losses of NO than NO_3^- based fertilizers and the losses were more variable in tropical than temperate systems [167,184].

The effect of increasing N fertilization on NO fluxes can be enhanced by water filled pore space [6,105,179]. Figure 13 illustrates these effects on N fertilizer amended and on unamended soils. The trends in NO flux with moisture may change due to interactive effects of fertilization, soil parameters and crop development [6,105].

By thorough analysis processes that produce and regulate NO emissions, Skiba *et al.* [167] identified viable mitigation strategies of increasing fertilizer use efficiency with consideration of N availability and plant demand, selection of appropriate fertilizer type and cultivation, use of nitrification and urease inhibitors and their environmental and economic effects.

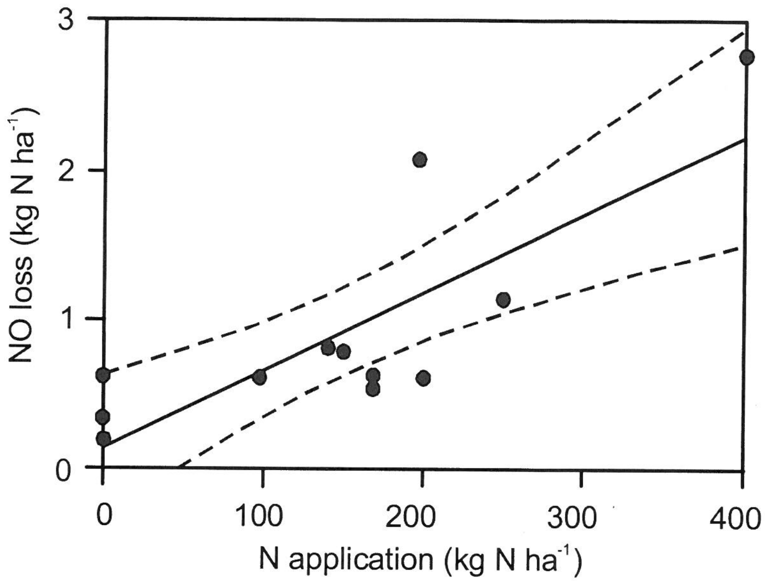

Fig. 12. NO flux in relation to N availability, data from field experiments covering at least a complete growing season [after 184]

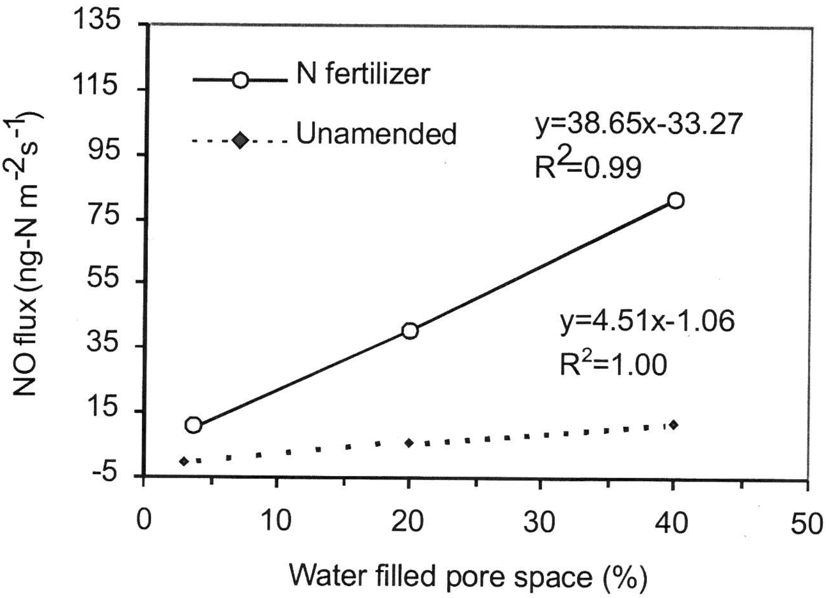

Fig. 13. NO flux in relation to water filled pore space in pot experiment [after 179]

5.4. Effects of crop type on nitric oxide emissions

The effect of crop type on NO emissions is mostly associated with fertilizer requirements. The summarized data of several experiments indicate that NO emission was greater from soil supporting corn (8.1-54.7 ng N m^{-2} s^{-1}) than cotton (1.8-4.3 ng N m^{-2} s^{-1}) and soybean (0.7-3.8 ng N m^{-2} s^{-1}) [178]. The crop type effect is related to site conditions. For instance, in one site in North Carolina (NC), NO fluxes were highest under corn (21.9 ng N m^{-2} s^{-1}) and much lower under cotton and soybean (4.3 and 2.1 ng N m^{-2} s^{-1}) whereas in another site they were greater for soybean than corn [6]. In Pennsylvania average NO flux was 1.2 ng N m^{-2} s^{-1} from a wheat field and 94 ng N m^{-2} s^{-1} from a maize field [197]. The inconsistent differences in NO emissions between the crops are attributed to various sampling dates and associated developmental growth stages [6] and interactive effect of WFPS and temperature [6,177].

With legumes Veldkamp and Keller [184] observed NO losses as high as 11% of applied N and this was many times higher than for other crops. This was attributed to excluding nitrogen fixed by the legumes which resulted in exaggeration of fertilizer derived NO emissions.

6. EFFECT OF LAND USE AND CULTURAL PRACTICES ON FLUXES OF MULTIPLE GREENHOUSE GASES

In this section are discussed the results from the experiments where fluxes of multiple greenhouse gases were studied. Such experiments allowed minimizing the effect of site conditions while comparing fluxes of various gases in response to management practices.

6.1. Carbon dioxide, methane, and nitrous oxide

Fluxes of different gases in response to land use are largely influenced by climate conditions in different way in respect to particular gases. The N_2O emission from peat soils was relatively high and ranged widely (8-38.4 kg N_2O-N ha^{-1} y^{-1} or 21.9-105 g N_2O-N ha^{-1} day^{-1}) in Finland [133] and the Netherlands [98]. However, in the tropical climate of Indonesia it was negligible or slightly positive or negative [80]. In the case of CH_4 the emission in the tropics (12 kg C ha^{-1} y^{-1} or 0.14 mg C m^{-2} h^{-1}) was one hundred times more that in the European countries while CO_2 emission was high in all sites ($11\text{-}22 \cdot 10^3$ kg CO_2 ha^{-1} y^{-1} or 0.034-0.068 g C m^{-2} h^{-1}). A wide range of N_2O emission in European countries was associated with nitrogen availability and high CH_4 emission rates in tropical areas were positively correlated with precipitation.

Converting tropical forest peatland to paddy field [80] increased annual emissions of CO_2 and CH_4 by 25% and 58%, respectively, while conversion of secondary forest to upland decreased emissions of these gases by 20% and 50%, respectively. However, no clear effect of the conversions was observed for N_2O. Under a cool moist climate in Scotland, periods of low CO_2 fluxes and very high N_2O fluxes under no-tillage were associated with reduced gas diffusivity and air-filled porosity following heavy rainfall [13]. The authors indicated that ploughing and control of compaction can minimize losses of CO_2 and N_2O whereas CH_4 oxidation may be preserved by no-tillage. In another long term study (25 years) in Germany mean integrated emission of the CO_2, CH_4 and N_2O was 4.2 and 3.0 Mg CO_2 equivalents for farm with conventional and organic farming, respectively [45].

6.2. Carbon dioxide and nitrous oxide

Studies in tropical rain forests (Northeast Queensland, Australia) showed that N_2O emissions were positively correlated with WFPS up to a threshold of 50-60% in the wet season. However, CO_2 emissions were positively correlated with WFPS during the dry season and negatively during the wet season [94]. So these imply that correlation of N_2O emissions depends on the range of the WFPS and climate.

Under flooding conditions diurnal N_2O production was positively correlated diurnal CO_2 emission and Corg [201] and the CO_2 and N_2O ratio in the gases evolved increased curvilinearly with redox potential and decreased with N_2O and CO_2 production [202].

6.3. Carbon dioxide and methane

Research in the tropics [42] showed that forest and pasture soils released more CO_2 during the wet than the dry season whereas forest soil CH_4 consumption was three times lower during the wet season. However, pasture soils showed a net emission of CH_4 even during the dry season. Similar response of the gases was observed in tropical peat land [80]. In diverse Alaskan soils, maximum CO_2 production occurred at maximum water holding capacity (WHC) whereas maximum atmospheric CH_4 consumption – at 34% WHC [60]. The relative effect of the fertilization in *Carex*-dominated peatland was considerably greater on CH_4 than CO_2 emission [1].

6.4. Nitrous oxide and methane

Combined measurements the exchange of both gases were done mostly in tropical soils since they notably contribute to the global soil source of N_2O and sink for atmospheric CH_4 [122,124,186,214]. Some cultural practices such as mineral fertilization, liming and land-use changes were adopted to reduce N_2O emission and increase CH_4 uptake. Mineral fertilization was used to mitigate the CH_4 emission but this practice led to high N_2O emission [185,186,214]. To mitigate emission of the gases it is suggested that N fertilizers should not be applied when WFPS is higher than approximately 75% [185]. Liming an acid Oxisol in the tropical grasslands increased N_2O emissions and CH_4 consumption [124]. Mosier et al. [124] observed that responses of both gases in the tropics are to some extent similar to those in central England [78].

Conversion of native grasslands to arable land decreased the uptake of atmospheric CH_4, and increased the emission of N_2O [122] and the inverse was true when converting arable land to woodland [12]. Afforestation of arable land is considered more beneficial to greenhouse gas exchange than conversion to organic production [12].

CH_4 and N_2O emissions were strongly correlated with changes in soil redox potential in paddy ricefields [72]. However, significant CH_4 emission occurred only at a soil redox potential of < -100 mv, while the emission of N_2O was not significant at potential $>+200$ mv. The results imply the possibility of using management practices to maintain the redox potential in a range where both N_2O and CH_4 emissions are relatively low.

6.5. Nitrous oxide and nitric oxide

Emission of both gases is largely influenced by soil conditions. In general, while N_2O emissions increased with the moisture content of the soil, NO emissions decreased with increasing soil moisture and rainfall [66,118]. In the UK the annual NO flux (0.79 kg N ha^{-1} or 0.0025 ng NO-N m^{-2} s^{-1}) was approximately half the corresponding N_2O (1.42 kg N ha^{-1} or 3.9 g N ha^{-1} day^{-1}) [66]. In the conceptual model 'hole in the pipe' of gaseous N loss [44] the relative amount of N_2O or NO that leaks from the pipe is mostly determined by soil water content that together with other factors determines relative rates of nitrification and denitrification. Based on the linear relationship between NO plus N_2O emissions and N availability and that between N_2O and NO ratio and % WFPS, Verchot *et al.* [187] developed formulae to predict N_2O and NO release, respectively. Predictions by the model agreed well with observed fluxes in different sites suggesting its applicability at a broader scale.

The release of both gases were positively correlated to temperature with Q_{10} values of 3.1 for N_2O and 8.7 for NO between 5-30°C [2]. Mean fluxes of N_2O decreased appreciably with increasing acidity, while those of NO showed little dependence on pH, with the highest mean flux from the plot at pH 5.9 [209]. N_2O emissions were greatest in the finer-textured soils, whereas NO emissions were greater – in the coarser-textured soils [118].

7. FUTURE TRENDS

Further studies are required to determine the seasonal effects of different land uses, cultural practices and type of vegetation on determining of soils are a sink or source of atmospheric carbon. To obtain accurate estimates of annual CO_2 fluxes in northern regions further research concerning the fluxes during dormant season and periods following wetting and thawing are needed.

Additional research are required to quantify the effects of fertilization on N_2O and NO formation and emission in relation with land use and cultural practices and fertilizer types. More studies are required to know better the effect of nitrogen fixed by legumes on N_2O and NO emission.

Better understanding of the effects of land use and cultural practices on rice root exudation and associated growth of methanogenic and methanotrophic bacteria and CH_4 exchange is needed.

There is a large uncertainty in regional and global inventory of greenhouse gas emissions. This motivates the need for better understanding of the spatial and temporal variability of greenhouse gas exchange in relation to management practices. Because most of the annual greenhouse gas losses result from a few maximal fluxes, defining and including these fluxes in a regular sampling schedule could improve the estimates and model predictions.

Further studies should also include experiments in which some parameters are controlled. Additional research is required on the effect of soil management on greenhouse gas emissions from deeper soil layers.

Research including measurements of multiple gas exchange is needed to weigh beneficial against negative effects of management practices with consideration site-relative importance of particular gases.

8. CONCLUSIONS

1. Land use and cultural practices largely affect greenhouse gas emission through changes in soil physical, chemical and biological properties of soil and crops. These effects depend on type of greenhouse gas and site conditions.
2. CO_2 exchange is largely affected by major land uses i.e. cropfield, grassland, forest and woodland. Most variations in the emission is accounted by soil temperature and plant cover. In northern regions whether given soil-plant system is a sink or a source for atmospheric CO_2 is mostly dependent on soil CO_2 capturing by photosynthesis and emission during dormant season. The direct effect of tillage on increase in CO_2 emission due to reduced resistance to gas transport and elevated temperature is most pronounced during and immediately after tillage. Increased tillage depth and

degree of soil disturbance enhance this effect whereas rainfall events diminish this effect. The long-term effect of tillage on CO_2 exchange is mostly through the changes in soil organic carbon content and soil structure.

3. Land use, fertilisation, tillage, and crop type largely influence N_2O emission. In most cases N_2O emission was higher from fertilized grasslands than other land uses (cropfield, forest, woodland). Enhancing effect of nitrogen fertilisation on N_2O emission often occurs after rainfall or irrigation. The proportion of N-fertilizer released as N_2O depends on crop type, crop rotation and crop residue management. Higher N_2O emission from no tilled than tilled soil is ascribed to increased availability of C, greater contribution of larger aggregates with anoxic centres and reduced gas diffusivity and air-filled porosity and greater denitrifying enzyme activity. The effect of soil compaction on the increase in N_2O emission is largely related to reduced pore volume and increased contribution of large aggregates. In northern regions peak N_2O emissions occur often during winter and at thawing.

4. Wetlands are the largest contributors to atmospheric methane concentration. CH_4 emission from wetlands is highly stimulated by addition of organic matter. Water management is the most important factor influencing CH_4 emission from ricefields. Drainage of water during the growing season may substantially reduce this emission. The emission from ricefields can be influenced also by properties of rice cultivars and their abilities to transport CH_4 to the atmosphere. In upland soils, CH_4 is mostly absorbed but can be also emitted depending mostly on air-filled pore space and fertilization. In general CH_4 uptake is higher in forests than grasslands cultivated fields. Redox potential (Eh) is the most important edaphic factor determining activity of methanogenic bacteria and absorption of CH_4.

5. There is a general trend of increasing NO emission with increasing N-fertilisation and soil water content. When soil water is not limiting NO flux increases exponentially with soil temperature. Interactive effects of fertilisation crop growth and soil parameters result in high temporal variability of NO. Nitric oxide emission can be reduced by application of N-fertilisers at low soil wetness (water filled pore space <20%).

6. Greenhouse gas emissions are very sensitive to spatial and temporal variability of soil and crop parameters, which are largely influenced by land use and cultural practices.

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8. SUMMARY

Land use and cultural practices play an important role in global green-house gas emission and uptake. Our objective is to review the effects of land use and cultural practices on carbon dioxide (CO_2), nitrous oxide (N_2O), methane (CH_4) and nitric oxide (NO) emission and uptake with emphasis on recent developments. The effects of land use and tillage on CO_2 emission are mostly due to changes in soil temperature affecting soil and root respiration. Soil-plant systems can either be a sink or a source for atmospheric CO_2 or at equilibrium, depending on the magnitude of captu-ring soil CO_2 through photosynthesis and dormant season flux. During and immediately after tillage CO_2 emission is enhanced by physical CO_2 release from soil due to reduced resistance to gas transfer. This effect increases with increasing tillage depth and degree of soil disturbance. N_2O emission is in general higher from grasslands than other land uses (crop field, forestland, woodland) and from not tilled than tilled soil. This emis-sion increases with increasing nitrogen fertilization, availability of C, contribution of larger aggregates and decreasing gas diffusivity and air-filled porosity. Large contributors of CH_4 to the atmosphere are cultivated (rice fields) and natural wetlands. The CH_4 emission from the rice fields can be reduced by drainage of water during growing season and by appropriate selection of rice cultivars and CH_4 emission from the wetlands is highly stimulated by increasing organic matter. In general, aerated upland soils act as sinks for atmospheric CH_4 with higher absorption potential in forests than grasslands and cultivated fields. Nitric oxide emissions increase with increasing N-fertilization, decreasing soil water content and soil temperature. Spatial and temporal variability of the greenhouse gas fluxes in relation to soil management practices and interrelations between fluxes of particular gases are discussed. The potential of some innovative techniques for measuring soil greenhouse gas concentration and emission at different scales is indicated. Large uncertainty in inventory of greenhouse gas fluxes implies the need for further measurements and modeling the fluxes under different management practices.

Keywords: land use, soil management, tillage, greenhouse gas fluxes, review

9. STRESZCZENIE

WPŁYW UŻYTKOWANIA ZIEMI I ZABIEGÓW AGROTECHNICZNYCH NA PRZEPŁYW GAZÓW SZKLARNIOWYCH W GLEBIE

Ryusuke Hatano[1], *Jerzy Lipiec*[2]

[1]Katedra Gleboznawstwa, Podyplomowa Szkoła Rolnictwa, Uniwersytet Hokkaido
Sapporo, 60-8589, Japonia
[I]nstytut Agrofizyki im. Bohdana Dobrzańskiego PAN
ul. Doświadczalna 4, 20-290 Lublin, Polska

Użytkowanie ziemi i zabiegi agrotechniczne wywierają duży wpływ na emisję i absorpcję gazów szklarniowych. Celem niniejszej pracy przeglądowej było omówienie emisji i absorpcji dwutlenku węgla (CO_2), podtlenku azotu (N_2O), metanu (CH_4) i tlenku azotu (NO) w zależności od użytkowania ziemi i zabiegów agrotechnicznych ze szczególnym uwzględnieniem ostatnich wyników badań. Sposób użytkowania ziemi i uprawa oddziałują na emisję CO_2 głównie poprzez zmiany temperatury gleby warunkującej oddychanie gleby i korzeni roślin. Ekosystemy rolnicze i leśne mogą być biorcą lub dawcą atmosferycznego CO_2 lub utrzymywać stan równowagi w zależności od ilości wiązanego w fotosyntezie CO_2 i od intensywności emisji tego gazu z gleby w okresie spoczynku zimowego roślin. Zwiększona emisja podczas i bezpośrednio po zabiegach uprawowych jest głównie rezultatem mniejszego oporu dyfuzyjnego przepływu i fizycznego uwolnienia CO_2. Szybkość tej emisji zwiększa się ze wzrostem głębokości uprawy i stopniem rozdrobnienia gleby. Na ogół emisja N_2O jest większa z użytków zielonych niż z pól uprawnych, lasów i zadrzewień oraz z gleby uprawianej niż nie uprawianej. Szybkość tej emisji rośnie wraz ze wzrostem nawożenia azotowego, dostępnością węgla, udziału dużych agregatów glebowych oraz zmniejszeniem oporu dyfuzyjnego przepływu gazów i porowatości powietrznej. CH_4 jest głównie emitowany przez pola ryżowe i naturalne obszary pod wodą. Wzrost zawartości materii organicznej w glebach zalanych wodą prowadzi do istotnego wzrostu emisji tego gazu. Emisję metanu z pól ryżowych można ograniczyć poprzez okresowe odwodnienie gleby podczas sezonu wegetacyjnego i dobór odpowiednich odmian ryżu. Generalnie, gleby dobrze natlenione są biorcami metanu atmosferycznego, przy czym większy potencjał absorpcyjny wykazują lasy niż użytki zielone i pola uprawne. Emisja tlenku azotu zwiększa się wraz ze wzrostem nawożenia azotowego i spadkiem wilgotności i temperatury gleby. Omówiono zmienność przestrzenną i czasową wymiany gazów szklarniowych w zależności od sposobów użytkowania gleby oraz wzajemne zależności pomiędzy wymianą poszczególnych gazów. Zwrócono uwagę na nowe metody pomiaru stężenia i emisji gazów szklarniowych w różnych skalach (agregaty glebowe, pola uprawne). Inwentaryzacja wymiany gazów szklarniowych obarczona jest dużą niepewnością. Stąd wynika potrzeba dalszych pomiarów i badań modelowych emisji i absorpcji tych gazów w zależności od sposobów użytkowania gleby i warunków klimatyczno-glebowych.

Słowa kluczowe: użytkowanie gruntu, zabiegi agrotechniczne, przepływ gazów szklarniowych, przegląd literatury

Addresses of authors:

Ryusuke Hatano
Soil Science Laboratory, Graduate School of Agriculture
Hokkaido University
Sapporo, 60-8589 Japan
e-mail: Hatano@chem.agr.hokudai.ac.jp

Jerzy Lipiec
Institute of Agrophysics, Polish Academy of Sciences
ul. Doświadczalna 4, 20-290 Lublin, Poland
tel. +48 81 7445061 fax +48 81 7445061
e-mail: Lipiec@demeter.ipan.lublin.pl