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Quantification of methane emissions from Chinese rice fields (Zhejiang Province) as influenced by fertilizer treatment

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Abstract. Methane emissions from rice paddies were quantified by using an automatic field system stationed in Zhejiang Province, one of the centres for rice cultivation in China. The data set showed pronounced interannual variations over 5 consecutive vegetation periods; by computing average values of all experimental plots the annual emissions were 177 g CH_4 m^{-2} yr^{-1} in 1987, 50 g CH_4 m^{-2} yr^{-1} in 1988, and 187 g CH_4 m^{-2} yr^{-1} in 1989.

The field preparations encompassed 4 different treatments: (1) no fertilizers, (2) mineral fertilizer (KCl, K_2SO_4), (3) organic manure (rape seed cake, animal manure), (4) mineral fertilizer plus organic manure. The methane emission rates of the different fertilizer treatments did not show significant differences. The mean emission rates, calculated over the entire observation period of 5 seasons, were 30.4 mg CH_4 m^{-2} h^{-1} (non-fertilized plot) and 28.3 mg CH_4 m^{-2} h^{-1} (mineral fertilizers). These values indicate a high level of methane production even without additional input of organic material into the rice-soils. In the other plots, the organic fertilizers were added once per vegetation period at app. 1 t fresh weight per ha, a relatively low application rate by agronomical standards. The mean emission rates were 35.1 mg CH_4 m^{-2} h^{-1} when manure was applied as sole fertilizer and 27.5 mg CH_4 m^{-2} h^{-1} when applied jointly with potassium fertilizers.

Based on the results of this study we estimate a range of 18—28 Tg CH_4 yr^{-1} as the total methane emission from Chinese rice fields. However, more field data from representative sites in China are needed to reduce the uncertainties in this estimate.

Introduction

The tropospheric background concentration of methane has increased over the past century from a preindustrial level of 0.7 ppm to a current mixing ratio of 1.7 ppm in the northern and 1.6 ppm in the southern hemisphere (Blake et al. 1982; Khalil & Rasmussen 1983). Methane absorbs outward-bound infrared radiation and thus, significantly contributes to the earth's radiation budget. The contribution of methane to the anthropogenic greenhouse effect is estimated to be 15% (Rodhe 1990). In addition to the impact on global warming, increasing atmospheric methane concentrations will also influence the stratospheric ozone budget (Isaksen & Stordal 1986). Photochemical oxidation of methane in the stratosphere leads to water vapor and subsequently to the formation of stratospheric clouds, which have been identified as major agents in the depletion of stratospheric ozone in polar regions (Rowland 1988).

The increase of methane abundance in the atmosphere can be attributed to an enhancement of the global methane source strength (Bolle et al. 1986; Cicerone & Oremland 1988). The total amount of methane emitted into the atmosphere has been estimated to be 505 $\pm$ 105 Tg per year (Crutzen 1991). However, great uncertainties remain in the present estimates of the source strengths of different individual sources of methane. Two types of 'environments' act as the main sources of methane in the global context: wetland soils and the digestive tract of ruminants. Agricultural activities, i.e. expansion of rice cultivation and lifestock breeding, have led to a considerable extension of both 'environments' in recent years (Harriss 1990; Bouwman & Sombroek 1990).

The total area planted with rice comprises 143×10^6 ha, of which 18×10^6 ha were planted under upland conditions and will not be prone to methane emissions (Neue et al. 1989). Wetland rice cultivation can be differentiated into three different rice ecologies: irrigated, rainfed and deepwater rice. Irrigated rice paddies represent 51% of the rice fields on a global scale; in East Asia this category comprises 92% of the rice fields (Neue et al. 1990). In addition to water regimes, rice fields vary within a broad range of soil properties and climatic factors (Wassmann et al. 1993). Neue et al. (1990) evaluated the impact of soil characteristics and temperatures on methane emission and estimated that only 56% of the total area planted with rice may emit significant quantities of methane.

Recent attempts to accomplish more reliable extrapolations of methane emission on a global and regional scale lead to geographical information systems (GIS) of rice cultivation (Matthews et al. 1991; Bachelet & Neue 1993). However, the high resolution of the geographical distribution of rice cultivation, available from the GIS, is contrasted by a very low number of field data on methane emission. The latitude expansion of rice cultivation reaches from equatorial to temperate zones, but only a few sites and climates have been investigated so far (Rennenberg et al. 1992;

Wassmann et al. 1993). A relative large proportion of methane emission rates have been recorded in regions with minor rice growing areas in the global context, i.e. Italy (Schütz et al. 1989a), Spain (Seiler et al. 1984), California (Cicerone & Shetter 1981; Cicerone et al. 1983), and Texas (Sass et al. 1991). In these regions, the ambient conditions and the cropping systems deviate considerably from the situation in Asia, where 91% of the rice cultivation is located (IRRI 1990). Only a small number of field measurements have been conducted in China, the country with the world's largest rice production (Wang et al. 1990; Schütz et al. 1990; Khalil et al. 1991). Therefore, the first objective of the present investigation was to provide a more comprehensive data set of the *in situ* emissions of CH_4 in one of the main rice growing areas of China.

Chinese rice cultivation can roughly be subdivided into 5 major rice growing areas with relatively uniform climatic and edaphic conditions (see Gong & Xu 1990). The Northern part of Zhejiang Province including the city of Hangzhou marks the south-eastern edge of one of the main rice growing areas that stretches along the middle and lower Changjiang (Yangtsekiang) River. The Changjiang region, as defined by Gong & Xu (1990), comprises ca. 20% of the Chinese rice growing area. This percentage was calculated by adapting statistical data of rice cultivation area on province level (IRRI 1991) to the map given by Gong & Xu (1990). The other regional centres for rice cultivation are located in Central China (mainly Hunan Province), in Southern China (mainly Guangdong Province), in the Southern Central China (mainly Sichuan Province) and in Northern China, where the rice growing area is widely scattered.

With respect to the effects of fertilizer applications on CH_4 emissions from rice paddies, previous field experiments yielded partly contradictory results by applying nitrogen containing amendments, whereas the application of organic manure, e.g. rice straw, generally enhanced methane emission (Cicerone & Shetter 1981; Schütz et al. 1989a; Yagi & Minami 1990). Therefore, the second objective of the present investigation was to gather information on the effect of different fertilizer treatments on methane emissions from rice paddies. The experiments included two commonly used fertilizers that have not been investigated so far: potassium compounds as mineral fertilizer and rape seed cake as organic fertilizer.

Site location and field preparation

The measurements were conducted at the Zhejiang Agricultural University of Hangzhou (30°19'N, 120°12'E), PR China. The soils of the experimental site are of alluvial origin representing a common soil type of the area at the lower course of the Changjiang river (Soil Map of China 1990). The experimental field was planted with vegetables prior to the experiment and was converted to wetland rice cultivation in 1987. The mechanical composition of the soil at the experimental station of Zhejiang Agricultural University was: 79.8% > 0.01 mm and 20.2% < 0.01 mm; the clay content (< 0.002 mm) was 12% (Yang, pers. communication). The pre-flooded soil contained 2.5% total organic matter, 0.19% total nitrogen, 1.74% potassium, and 0.101% phosphorous. Compared to other rice soils in China (Lu 1981) these nutrient contents have to be regarded as moderately high. The pH of the soil prior to flooding was 6.5; during waterlogging the soil pH varied inbetween 6.5 and 7.5.

The experimental field was routinely managed according to local farming practice. Winter crops were vegetables and green manure. Prior to flooding the fields were ploughed and raked. The field site was parcelled into 4 experimental plots, which were separated by concrete boards: Plot 1: no fertilizer; plot 2: mineral fertilizers (early rice: 694 kg KCl ha^{-1}; late rice: 694 kg K_2SO_4 ha^{-1}); plot 3: organic manure (1042 kg fresh weight ha^{-1}: rape seed cake in 1987 and 1988; animal manure in 1989); plot 4: mineral fertilizers plus organic manure (corresponding to treatments of plots 2 and 3).

The fertilizers were ploughed into the soil before flooding. The type of mineral fertilizer in plots 2 and 4 varied according to local practice (KCl in early seasons and K_2SO_4 in late seasons). The application rates corresponded to 438 kg K_2O ha^{-1} (early rice) and 375 kg K_2O ha^{-1} (late rice), respectively, representing a very high dosage of potassium fertilization. The addition of organic manure in this experiment corresponds to app. 10% of the common application rate when used as sole fertilizer.

The plots were flooded in April (early rice) and July (late rice). Rice seedlings were raised in nursery beds. At transplanting the water cover in the fields was kept shallow (ca. 1 cm); after the rice plants reached sufficient height the water level was maintained inbetween of 5—10 cm.

The pattern of plant development in the two annual vegetation periods can be derived from Table 1. Prior to harvesting the field was drained for several days. According to local practice the cultivars planted in the early vegetation periods were early-maturing and short-straw varieties (Oryza sativa L. Subsp. indica) with a growing period of 85 days; hybrid cultivars (Oryza sativa L. Subsp. japonica) with a growing period of 100 days were selected for the late vegetation periods. The names of the cultivars planted were in the early seasons Zhefu-3 (early rice 1988) and Zaolian 31 (early rice 1989) and in the late seasons T8340 (late rice 1987 and 1988) and T8528 (late rice 1989).

Table 1. Plant development (small letters) and farming practices (capital letters) as recorded in 1988.

Stage/PRACTICE	Early rice	Late rice
Seedling development	Apr. 5—Apr. 30	July 5—July 30
TRANSPLANTING	Apr. 25—May 5	July 25—Aug. 5
Tillering	May 5—May 15	Aug. 5—Aug. 15
Panicle initiation	May 15—June 10	Aug. 15—Sept. 10
Grain filling	June 10—July 10	Sept. 10—Oct. 10
Ripening	July 10—July 30	Oct. 10—Nov. 11
HARVEST	July 20—July 30	Nov. 1—Nov. 11

Sampling and analytical procedure

The experimental set up for the determination of methane emission used for this study was basically as described by Schütz et al. (1989a). The 16 gas collector boxes were made of colorless, smooth plexiglass. The edges were fixed by aluminum profiles and were sealed with silicone on the inside. The boxes covered a surface area of 0.42 m^2 and were 0.9 m high. Each box was fitted with a removable plexiglass cover. The position of the glass cover was regulated by a pneumatic pressure cylinder. A fan mounted on the inner side of the cover caused rapid replacements of air inside the box with ambient air when the box was open. During closure the fan ensured rapid mixing of the air within the box. Therefore, vertical CH_4 gradients inside the box could not build up.

The gas collector boxes were exposed in the field using stainless steel frames which had been sunk into the soil prior to flooding and remained at the same position during the entire vegetation period. The inner volume of the box was separated from the ambient atmosphere but the water beneath the box could exchange with the surrounding water body.

An automatic sampling system consisting of two completely separate units was used; each unit included a gas chromatograph connected to a personal computer via an interface designed for integration. The magnetic valves controlling the gas fluxes in the system were operated by a second personal computer that also stored temperature data (soil, water and air). The system was started by closing 8 of the 16 boxes, while the other boxes remained open. After a measuring period of 90 minutes, the closed boxes were opened and the open boxes were closed. Air samples from the inside of the closed boxes were taken periodically by two metal bellows pumps and transported through stainless steel capillary tubing into the laboratory located next to the experimental field. Water vapor was removed from the air by passing the sample through a cooling trap. Air samples were pumped into sampling loops connected to an 8-port-valve. By switching this valve the air samples were flushed to the columns of the gas chromatographs for CH_4 analysis (Shimadzu Mini II equipped with Flame Ionization Detector; column $\frac{1}{4}'$ stainless steel tube of 1.5 m length packed with molecular sieve 13 × 60/80 mesh; oven temperature 80 °C; detector temperature 110 °C; carrier gas: 120 mL/min hydrocarbon free synthetic air, Messer-Griesheim, Germany; burning air provided by a clean air generator, JUM GmbH, Germany; hydrogen of 99.999% purity, Messer-Griesheim, Germany).

After each 90-min measuring period, the GC system was calibrated automatically using CH_4 calibration gas stored in a steel cylinder under high pressure (9.8 ppm CH_4). The described automatic system allowed the determination of CH_4 emission rates for each box eight times a day.

The CH_4 emission rate E was calculated from the increase with time of the CH_4 mixing ratio inside the boxes by

$$E = h \times p \times \mathrm{d}m/\mathrm{d}t$$

where h = height of the box, p = density of CH_4, and $\mathrm{d}m/\mathrm{d}t$ = linear increase of the CH_4 mixing ratio. The lower detection limit of CH_4 emission rates was 0.05 mg CH_4 m^{-2} h^{-1}.

Results

The data set on methane emissions from Chinese rice fields in the present paper consists of app. 250,000 single measurements of methane concentrations and app. 30,000 computed emission rates covering 5 consecutive vegetation periods.

The seasonal, annual and total averages of methane emission rates from 4 different fertilizer treatments are shown in Table 2. The most evident finding that can be derived from Table 2 is the pronounced interannual variation of emission rates. The early and late vegetation period in 1988 yielded significant lower values than observed at other seasons. This pattern was found in all experimental sites independent from the fertilizer treatment. For each treatment the annual averages recorded in 1988 represented less than 40% of the corresponding values found in 1987 and 1989. Based on the annual mean emission rates given in Table 2 the overall average of the four plots accounted for 38.8 mg CH_4 m^{-2} h^{-1} in 1987, 10.9 mg CH_4 m^{-2} h^{-1} in 1988, and 41.4 mg CH_4 m^{-2} h^{-1} in 1989 (data not shown).

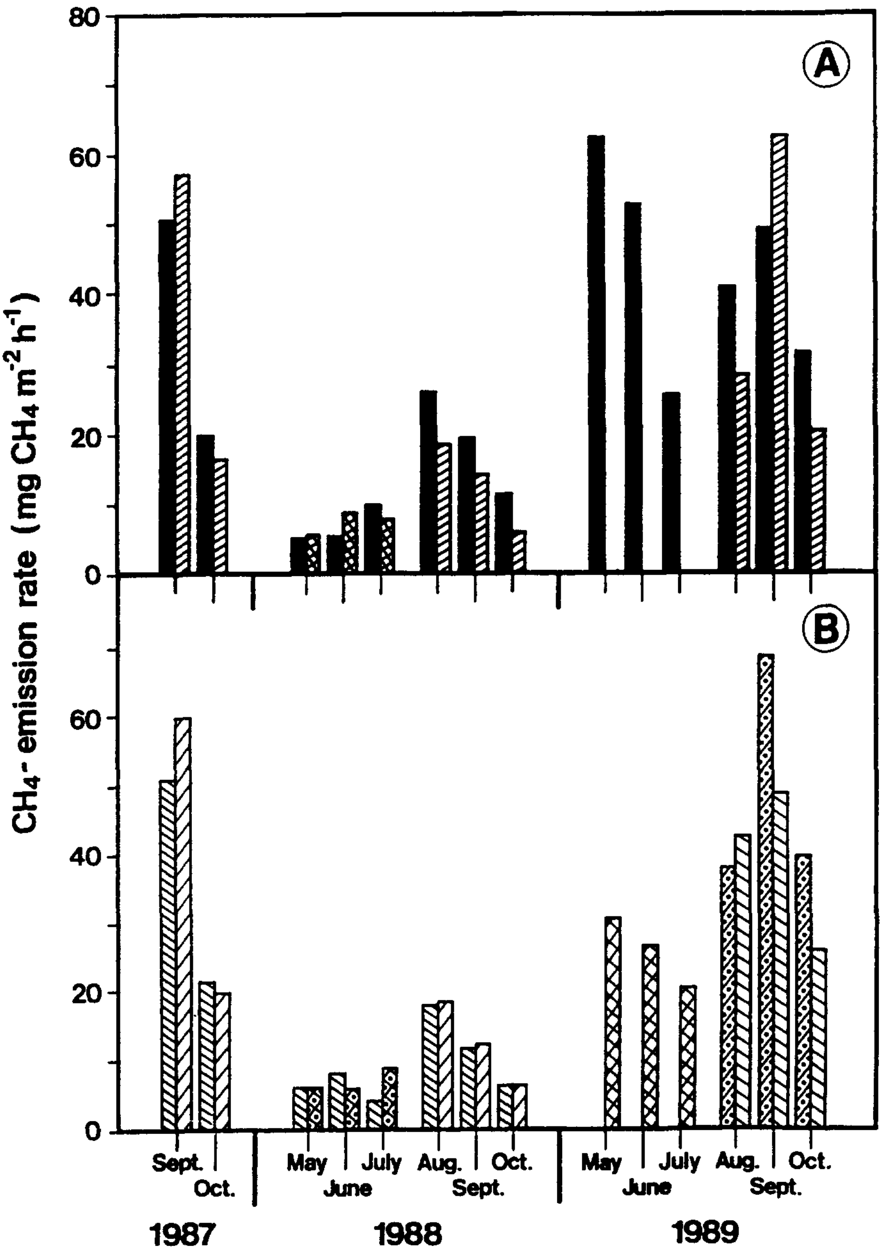
Table 2. Seasonal, annual and total averages of methane emission rates from experimental plots 1–4 [mg CH_4 m^{-2} h^{-1}].

Plot No.	1	2	3	4
Fertilizer:	None	Mineral[1]	Organic[2]	Min.[1] + Org.[2]
Seasonal average:				
1987				
late rice	33.7 ± 3.3*	36.6 ± 0.3	45.0 ± 2.6	39.7 ± 0.1
1988				
early rice	7.5 ± 1.1	7.7 ± 2.2	6.9 ± 0.2	7.9 ± 1.4
late rice	18.9 ± 2.1	13.1 ± 0.2	12.7 ± 2.5	12.0 ± 0.1
1989				
early rice	47.0 ± 7.8	no data #	no data #	26.2 ± 3.2
late rice	41.8 ± 2.5	35.4 ± 11.2	50.6 ± 6.2	44.0 ± 1.7
Annual average:				
1987	33.7	36.6	45.0	39.7
1988	13.2	10.4	9.8	10.0
1989	44.4	35.4	50.6	35.1
Total average:				
	30.4 (100%)	27.5 (90%)	35.1 (115%)	28.3 (93%)

[1] Early rice: 694 kg KCl ha^{-1} (equiv. to 438 kg K_2O ha^{-1})
Late rice: 694 kg K_2SO_4 ha^{-1} (equiv. to 375 kg K_2O ha^{-1})
[2] 1042 kg fresh weight ha^{-1} of rape seed in 1987 and 1988; 1042 kg fresh water ha^{-1} of animal manure in 1989)
* standard deviations derived from the mean seasonal flux rate of the individual boxes
\# lack of data due to technical problems

Monthly averages of the emission rates in the 4 different plots are displayed in Figs. 1A and 1B. One column approximately represents the mean methane emission rate of $\frac{1}{3}$ of a vegetation period. With this temporal resolution the four fertilizer treatments did not show pronounced differences in mean methane emission rates during the observation period. Generally, the maximum and minimum values within one vegetation period were found on all plots in identical months.

However, the comprehensive presentation of the emission rates in Figs. 1A and 1B required a broad temporal resolution, that may conceal virtual patterns occuring in smaller time scales, e.g. a short termed maximum of emission rates. In order to demonstrate the similarity of the seasonal courses in all plots, the daily averages of the emission rates obtained in



Figs. 1A, 1B. Monthly averages of methane emission rates from plot 1 (A, left columns), plot 2 (A, right columns), plot 3 (B, left columns), and plot 4 (B, right columns) treated with different fertilizers. ▮ no fertilizer; ▨ K_2SO_4; ▨ KCl; ▨ rape seed cake; ▨ animal manure; ▨ K_2SO_4 + rape seed cake; ▨ KCl + rape seed cake; ▨ K_2SO_4 + animal manure; ▨ KCl + animal manure.

late rice 1988 are shown as a typical example (Figs. 2A, 2B). The seasonal patterns obtained in all 4 plots were almost identical: generally, in the first half of the vegetation period two peaks of emission rates were observed, whereas during the second half of the vegetation period the emission rates were on a low level without pronounced peaks.

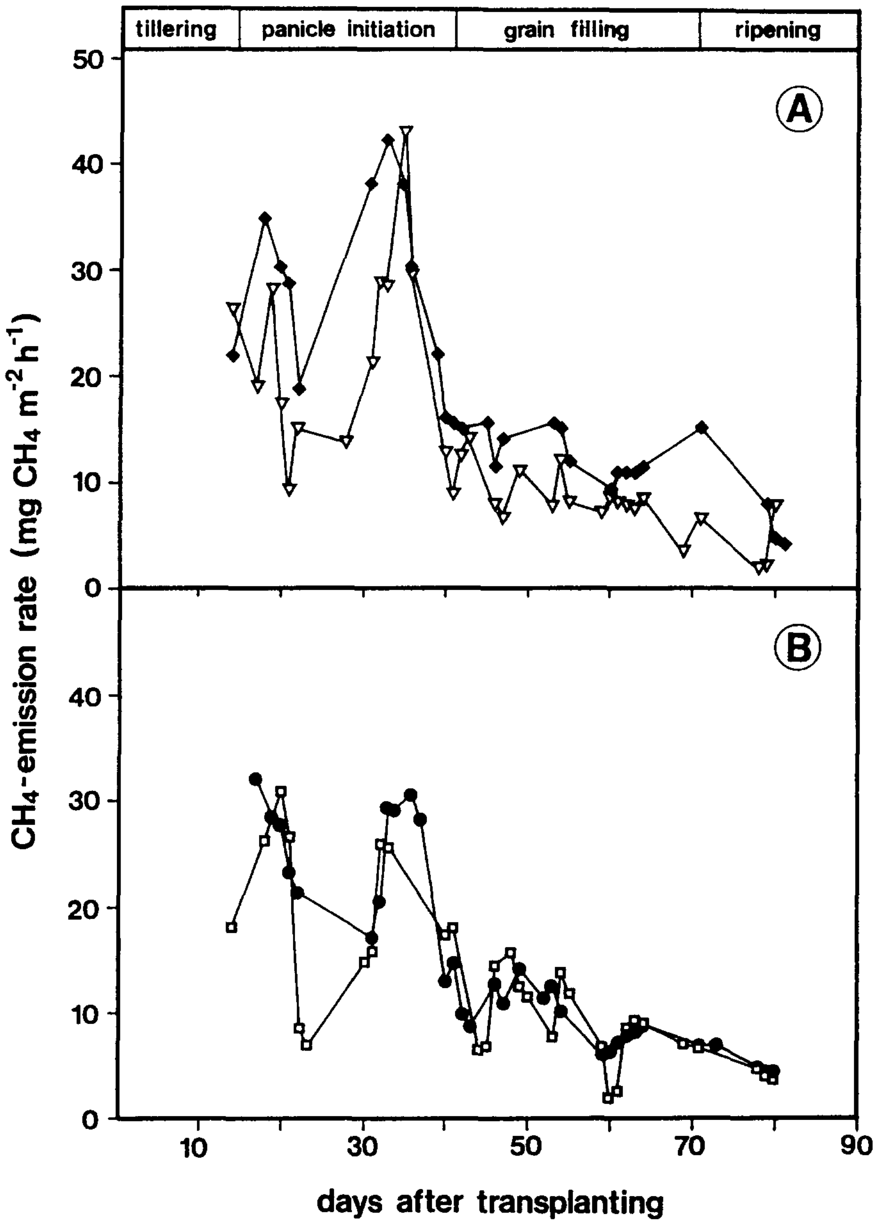
With regard to the 5 vegetation periods, the emission rates did not show a distinct seasonal pattern. This is illustrated by showing the results obtained in the unfertilized plot (Figs. 3A—3C), which is given as a typical example for the time-course of the methane emission rates found on all differently treated plots. Based on the daily averages of the emission rates, the early rice period in 1988 was characterized by a low level of emission rates with short-term peaks, e.g. 20 d and 40 d after transplanting, whereas the time-course in early rice 1989 basically consisted of an initial phase with high emission rates followed by persistent lower emission rates during the rest of the season (Fig. 3A). In the late vegetation periods the seasonal courses of the emission rates did also not exhibit a general pattern (Figs. 3B and 3C). Related to the dates of transplanting the maximum values in the late seasons were found after 54 d (1987), 20 d (1988), and 51 d (1989). The late stages of both early and late rice were generally characterized by lower methane emission rates compared to the early stages of the same vegetation period. In Figs. 3A—3C the daily minima and maxima are also given, in order to illustrate the ranges of emission rates observed during periods of 24 hours.

Integrated over the entire observation period, the mean emission rate of plot 1 was 30.4 mg CH_4 m^{-2} h^{-1} (Table 2). Related to this value the mean emission rates of the other plots corresponded to 90% (plot 2), 115% (plot 3), and 93% (plot 4), respectively (Table 2). However, a reduction of methane emission rates by pure mineral fertilizers (plot 2) was only observed in 2 of 4 seasons. Similarly, an enhancement of total methane emission rates from plot 3 (exclusively treated with organic fertilizer) was observed in only 2 of 4 vegetation periods; combined fertilization (plot 4) resulted in higher emission rates in 3 seasons and lower emission rates in 2 seasons (Table 2). Due to these inconsistencies the available data are insufficient to unequivocally verify fertilizer-specific effects related to the amendments as realized in this field experiment.

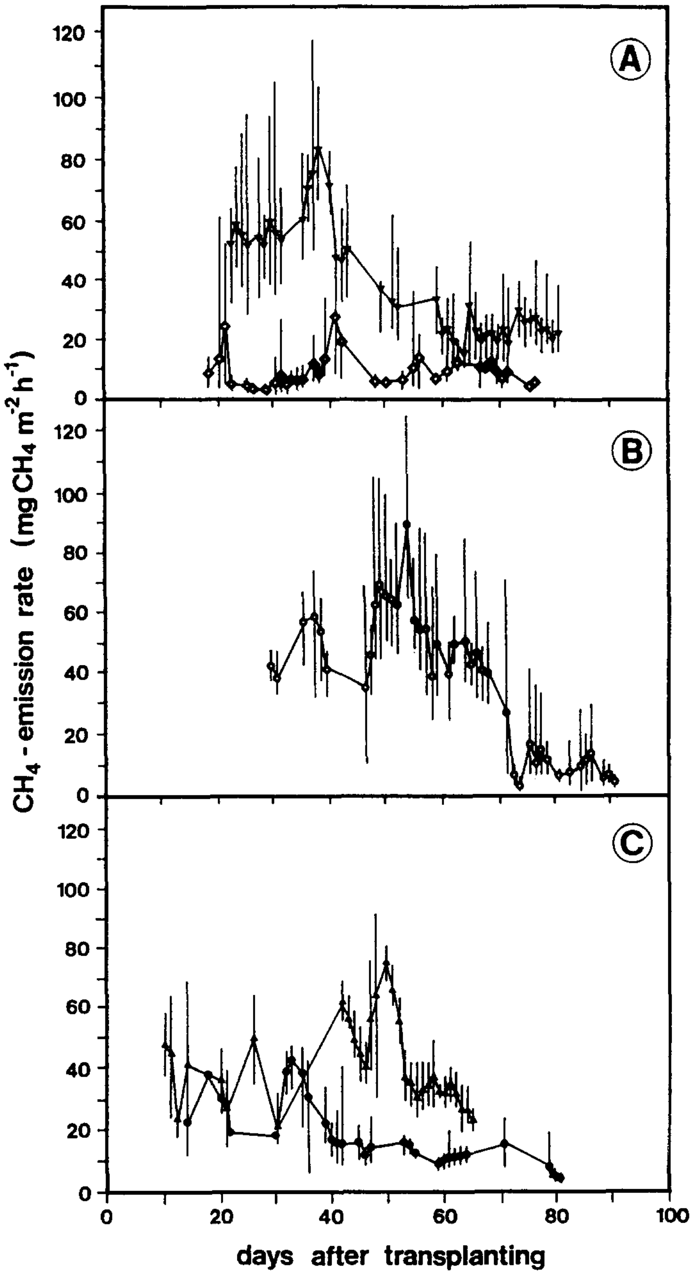
Discussion

1. *Temporal variation in methane emission rates*

The pronounced interannual variations (Table 2, Figs. 1A, 1B)) can be



Figs. 2A, 2B. Daily averages of the emission rates obtained in late rice 1988. ◆ plot 1; ▽ plot 2; □ plot 3; ● plot 4.



Figs. 3A, 3B, 3C. Seasonal variation of daily averages of methane emission rates [mg CH_4 m^{-2} h^{-1}] in field 1 (bars indicate daily minimum and maximum values). ◇ early rice 1988; ▼ early rice 1989; ○ late rice 1987; ▲ late rice 1988; ● late rice 1989.

attributed to a multitude of different factors. The different magnitude in emission rates found in different years could be caused by (1) climatic variations, (2) different rice cultivars, and (3) succession in soil properties triggered by the land conversion in 1987.

The seasonal average-values of three important meteorological parameters of the local climate, that determine plant growth, are displayed in Table 3. The mean air temperatures of the late vegetation periods in 1987, 1988, and 1989 were almost identical; in the two early seasons the mean air temperature differed by 1.4 °C. It is very unlikely that such a small difference in temperature is responsible for a 6-fold increase of methane emission rates as observed e.g. in plot 1 (Table 2). Furthermore, the low emission rates occured during the warmer season, when enhanced methane emission rates would be expected. The same applies to sunshine hours and cloud coverage (Table 3).

The changes of rice cultivars planted in the experimental field may have contributed to the interannual differences between the emission rates, but one can exclude this factor as the sole reason for the observed variations. The rice plants used in the two late vegetation periods 1987 and 1988 were of identical genotype (T8340) and yielded high methane emission rates in 1987, but low emission rates in 1988.

The seasonal differences may also be attributed to site-specific conditions. The field used for the experiments was planted with vegetables prior to the measurements. The biological and abiological conditions in the soil will differ at some extent from perennial wetland rice areas. The changes involved in a conversion to wetland rice could result in relatively unstable

Table 3. Seasonal averages of temperature [°C], daily sunshine hours [h d^{-1}] and proportional cloud coverage [%] at Hangzhou (data from Meteorological Observation Station Hangzhou compiled by D. Chen).

	Temperature [°C]	Sunshine hours [h d^{-1}]	Cloud coverage [%]
Season			
1987			
late	22.7	5.45	71
1988			
early	25.1	6.32	65
late	22.0	4.92	73
1989			
early	23.7	5.35	86
late	22.0	4.28	77

soil parameters, e.g. the stock of organic material, before the conditions in the soil will reach a stable level/persistent temporal pattern.

The time-course of emission rates from the differently treated plots were almost identical within a given vegetation period (Figs. 2A, 2B). Apparently, the modulation of emission rates is determined by factors that are not related to fertilization. The seasonal course of emission rates within the different vegetation periods differed totally and, therefore, it is not possible to define a distinct seasonal pattern (Figs, 3A—3C).

This finding contrasts with results from Italian rice paddies, where a relative stable sequence of phases with high and low methane emission rates was observed in three consecutive vegetation periods (Schütz et al. 1989a). The reasons for these differences between Italian and Chinese rice paddies remain presently unknown.

2. *Magnitude of methane emission rates*

The application of fertilizers into rice soils can affect all processes involved in methane emission, i.e. production, oxidation and vertical transport of methane (Wassmann et al. 1993). The interpretation of field data has to identify the fertilizer-specific impacts on the different levels of the methane budget in rice fields (Rennenberg et al. 1992).

The results obtained in the present study showed a relatively large conformity of different fertilizer treatments with respect to the magnitude of methane emissions. The results can be interpreted as follows: (1) high methane emission rates from rice paddies are not necessarily linked to organic fertilization; (2) a supplementary application of potassium fertilizers does not affect the methane emission rates; (3) organic amendments applied with a rate of 1 t ha^{-1} have only a minor impact on the methane emission rates.

In order to understand the results of this field study, the specific nutritional situation of the rice fields in Hangzhou has to be considered. In all four plots the nutrient content in the soil was not sufficient to achieve highest yields for the rice plants. This conclusion is based on the nature or quantities of the fertilizer amendments. The excess of potassium in plots 2 and 4 is unlikely to enhance plant growth as long as the amendment of K^+ is not accompanied by other essential nutrients, i.e. nitrogen and phosporous. The dosage of organic manure in plots 3 and 4 (ca. 1 t ha^{-1}) was about one order of magnitude less than commonly used when applied as sole fertilizer. The amount of nutrients derived from the mineralization of this organic material is unlikely to meet the requirements of the rice plants for achieving optimum yields.

Considering the complex interactions between the rice plants and the microbial community, nutrient limitations could have played a crucial role in determining the methane budget.

The relatively high levels of emission rates observed in plots 1 and 2 indicate an intense methane production derived from autochthonuous organic material. In rice fields, plant residues and root exudates represent internal sources of methanogenic substrates (Schütz et al. 1989b; Rennenberg et al. 1992). Various plants respond to a lack of nutrients by enhanced root exudation (Marschner et al. 1991); for rice plants increased exudation rates were shown as a response to potassium deficiency (Trolldenier 1971). However, the soil in Hangzhou was characterized by a moderately high K^+-content (1.74%), and the results from plot 2 indicate, that potassium did not play a crucial role in determining methane emission.

The mineral fertilizers used in this study were KCl (early rice) and K_2SO_4 (late rice). Previous investigations of mineral fertilizer effects focussed on nitrogen compounds and recorded pronounced differences between non-fertilized and nitrogen-fertilized plots. Cicerone & Shetter (1981) observed a five fold increase of the CH_4 emission after incorporation of ammonium sulfate. Extended field experiments in Italy indicated a 50% decrease of methane emission by deeply-incorporated urea or ammonium sulfate (Schütz et al. 1989a). In a field study in India, methane emission was recorded in a non-fertilized plot and a plot that was fertilized with mineral compounds according to the N-P-K requirements for optimal growth and yield (Parashar et al. 1991). The emission rates from the non-fertilized plot was 6.6—8.1 mg CH_4 m^{-2} h^{-1}, whereas the addition of 150 kg N ha^{-1} (as urea), 90 kg P ha^{-1}, and 90 kg K ha^{-1} yielded a range of 3.2—4.0 mg CH_4 m^{-2} h^{-1} (Parashar et al. 1991). In the present study the comparison of methane emission rates in early and late rice did not give any evidence for pronounced differences between the effect of KCl and K_2SO_4 on methane emission. There are some indications from other studies that sulfate in paddy soils can lead to a significant reduction in methane emission (Seiler et al. 1984; Schütz et al. 1989a), though other studies found the inverse effect (Cicerone & Shetter 1981). Apparently, the amount of sulfate (0.17 Mol per m^2) applied in Hangzhou was not sufficient to result in a significant alteration of methane emission.

In Hangzhou, the amendment of 1 t organic material per ha did not result in a detectable increase of the emission rates over the entire observation period. The sole use of farmyard manure, that is mainly based on animal excrement, requires an appliction rate of 5—10 t ha^{-1} to achieve high rice yields (Patnaik & Rao 1979). Plant residues, such as rape seed cake, generally contain less nitrogen (Stangel 1979) and have to be applied in even higher dosages. Previous experiments showed increased methane emission rates by organic manure, but the fertilizers were applied in considerably larger amounts than in the experiment in Hangzhou. A pronounced stimulation of methane emission rates by rice straw was observed by using application rates of 12 t ha^{-1} (Schütz et al. 1989a; Sass et al. 1991) and 6 t ha^{-1} (Yagi & Minami 1990; Yagi et al. 1990), respectively. The application of green leaf manure corresponding to a dose of 150 kg N ha^{-1} yielded 80 mg CH_4 m^{-2} h^{-1} in a single experiment conducted in Indian rice paddies (Parashar et al. 1991).

In addition to the application rate, the type of organic manure has an impact on methane emission (Yagi & Minami 1989). The organic fertilizers used in Hangzhou (rape seed cake and animal manure) can be characterized as readily decomposable material. This category of organic matter was shown to stimulate methane production rates in incubation experiments (Tsutsuki & Ponnamperuma 1987). Derived from the finding in this study, that the application of rape seed cake and animal manure did not result in increased methane emission rates, we conclude that the quantity of 1 t per ha applied in Hangzhou was not sufficient to prompt significant effects.

3. *Evaluation of methane emission from Chinese rice paddies*

The average CH_4 flux rate obtained over the entire observation period in all 4 plots corresponded to ca. 30 mg CH_4 m^{-2} h^{-1} in both, early and late vegetation period, and was equivalent to an annual emission of app. 130 g CH_4 m^{-2} yr^{-1}. Data obtained by another field experiment in Sichuan Province indicate an even higher level of methane emission rates in this region (Khalil et al. 1991). The experimental field in Sichuan was fertilized with organic material according to local practice; the average emission rates obtained over 2 consecutive years was 58 mg CH_4 m^{-2} h^{-1} (Khalil et al. 1991).

Results from this study and from Sichuan province give some evidence that rice fields in China are generally characterized by a high level of methane emission rates compared to other regions of the world. Only by intensive use of organic fertilizers rice paddies in Italy and Japan were recorded with average values of more than 25 mg CH_4 m^{-2} h^{-1}, whereas the absence of organic fertilizers generally yielded considerably lower values (Schütz et al. 1989a; Yagi & Minami 1989). A field study conducted in India (Parashar et al. 1991) recorded methane emission rates from mineral fertilization that were in the range of 1.6—19.3 mg CH_4 m^{-2} h^{-1}.

The total rice area harvested in China is actually 32.9 $\times$ 10^6 ha (IRRI 1991). In addition to acreage the varying durations of vegetation periods of rice in China must be taken into account in order to estimate the regional source strength with a reasonable accuracy. The largest portion of the rice area in China has the potential to yield 2 crops per year, in some areas even 3 rice crops are possible (Gao et al. 1987). The durations of early and late rice vegetation periods vary between 75—95 days and 80—140 days, respectively (Wang et al. 1990). Cropping systems with alternate rice and upland crops, e.g. wheat, are widely distributed in Central and Southern China (Gao et al. 1987). In Northern China the arable land yields only one crop per year, the vegetation period for a rice crop is 120—160 days (Gao et al. 1987).

Preliminary data of the field campaign in Hangzhou have been used by Wang et al. (1990) to extrapolate the regional source strength for China. This extrapolation was performed by differentiating between single and double cropping systems as well as early and late rice. On the basis of the average values for measurement campaigns in 1987 and 1988, i.e. 9.58 mg CH_4 m^{-2} h^{-1} (early rice) and 20.32 mg CH_4 m^{-2} h^{-1} (late rice), Wang et al. (1990) calculated a source strength of 13.8—22.8 Tg CH_4 yr^{-1}. However, using the same approach, but a mean emission rate of 30 mg CH_4 m^{-2} h^{-1} (representing the average value of 5 consecutive vegetation periods) we calculate a total emission from Chinese rice fields of 18—28 Tg CH_4 yr^{-1}.

However, this estimate is associated with considerable uncertainties that mainly derive from two biases for a more reliable extrapolation: (1) even by compiling all data available on methane emission from Chinese rice fields, the data base covers only a few sites over a few years, and (2) the observation periods in all measurements conducted so far were confined to the flooded phases in the rice fields and did not consider methane fluxes, i.e. emission or deposition, during fallow periods.

The previous estimates of the methane release from Chinese rice vary in the range from 13.8 to 30 Tg CH_4 yr^{-1} (Table 4). Two different types of approaches have been conducted so far in order to estimate the Chinese methane source strength (Table 4): extrapolation of emission data (Khalil et al. 1991; Wang et al. 1990) and geographical information systems using various classification criteria (Neue et al. 1990; Matthews et al. 1991; Bachelet & Neue 1993). The GISs provide high geographical resolutions, but the area-indices used to quantify the methane emissions are not yet validated by field observations. As long as an inventory of emission data *per se* is not available, the conventional extrapolation — as performed also in this study — has to be regarded as an appropriate procedure to accomplish regional estimates.

Chinese rice cultivation comprises 24.9% of the rice land located in Asia (acc. to IRRI 1991). In China, 92% of the area planted with rice is irrigated, whereas this type of water management accounts for ca. 50% of the rice land in entire Asia (acc. to IRRI 1991). Irrigated rice is

Table 4. Compilation of estimates of methane emission from Chinese rice fields [Tg CH_4 yr^{-1}] and China's contribution to the total methane emission from Asian rice land [%].

Estimated emission [Tg CH_4 yr^{-1}]	Contribution to CH_4-emission from Asian rice	Approach	Reference
13.8—22.8	n.d.	extrapolation	W
30	n.d.	extrapolation	K
21.32	33.8%	GIS (C-balance)	N
14.71	31.5%	GIS (soil emission potential)	N
21.62	47.1%	GIS (area index)	M
14.92	24.5%	GIS (soil potential)	M
21.6	26.4%	GIS (constant emission rate)	B + N
13.46	23.7%	GIS (C-balance)	B + N
21.3	33.8%	GIS (grain production)	B + N
18—28	25—35%	extrapolation	this study

W = Wang et al. 1990
K = Khalil et al. 1991
N = Neue et al. 1990
M = Matthews et al. 1991
B + N = Bachelet and Neue 1993

characterized by a higher potential for methane emission than the other rice ecologies, i.e. rainfed, deepwater and upland rice (Bachelet & Neue 1989). Therefore, the Chinese contribution to the total methane emission from rice land in Asia will be higher than the percentage of China's harvested rice areas. We estimate a range of 25—35% (Table 4) as China's contribution related to the methane release from rice land of entire Asia. Since more than 90% of the global rice area is located in Asia, the Chinese contribution in the global context will only be slightly smaller. We estimate that 22% to 32% of the methane emitted from the global rice are attributed to rice in China.

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