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ORIGINAL PAPER

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N_2O emissions from different cropping systems and from aerated, nitrifying and denitrifying tanks of a municipal waste water treatment plant

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Abstract Nitrous oxide emissions, nitrate, water-soluble carbon and biological O_2 demand (BOD_5) were quantified in different cropping systems fertilized with varying amounts of nitrogen (clayey loam, October 1991 to May 1992), in an aerated tank (March 1993 to March 1994), and in the nitrification-denitrification unit (March to July 1994) of a municipal waste water treatment plant. In addition, the N_2O present in the soil body at different depths was determined (February to July 1994). N_2O was emitted by all cropping systems (mean releases 0.13–0.35 mg N_2O m^{-2} h^{-1}), and all the units of the domestic waste water treatment plant (aerated tank 0–6.2 mg N_2O m^{-2} h^{-1}, nitrification tank 0–204,3 mg N_2O m^{-2} h^{-1}, denitrifying unit 0–2.2 mg N_2O m^{-2} h^{-1}). During the N_2O-sampling periods estimated amounts of 0.9, 1.5, 2.4 and 1.4 kg N_2O-N ha^{-1}, respectively, were released by the cropping systems. The aerated, nitrifying and denitrifying tanks of the municipal waste water treatment plant released mean amounts of 9.1, 71.6 and 1.8 g N_2O-N m^{-2}, respectively, during the sampling periods.

The N_2O emission were significantly positively correlated with nitrate concentrations in the field plots which received no N fertilizer and with the nitrogen content of the aerated sludge tank that received almost exclusively N in the form of NH_4^+. Available carbon, in contrast, was significantly negatively correlated with the N_2O emitted in the soil fertilized with 80 kg N ha^{-1} year. The significant negative correlation between the emitted N_2O and the carbon to nitrate ratio indicates that the lower the carbon to nitrate ratio the higher the amount of N_2O released. Increasing N_2O emissions seem to occur at electron donor-to-acceptor ratios (C_{H_2O} or BOD_5-to-nitrate ratios) below 50 in the cropping systems and below 1200–1400 in the waste water treatment plant. The trapped N_2O in the soil body down to a depth of 90 cm demonstrates that agricultural production systems seem to contain a considerable pool of N_2O which may be reduced to N_2 on its way to the atmosphere, which may be transported to other environments or which may be released at sometime in the future.

Key words N_2O release · Cropping systems · Waste water treatment · Activated sludge · Nitrification · Denitrification · Carbon availability · Available carbon-to-nitrate ratio

Introduction

In soils and waste water, even if well aerated, anaerobic energy-conserving processes can occur inside aggregates and sewage flocculates in the sequence NO_3^-, MnO_2 and Fe_2O_3 respiration followed by SO_4^{2-} and CO_2 reduction (Ottow and Glathe 1973). Much work has been done to clarify the effect of the variables, water-soluble carbon (C_{H_2O}), BOD_5, nitrate concentration (NO_3^-), moisture content, temperature and oxygen partial pressure on the onset of anaerobic nitrate respiration (Benckiser et al. 1987; Ottow 1992; Ottow and Benckiser 1994; Sümer et al. 1996). Despite these endeavors, the role of the above parameters on the denitrification process remains unclear because of the high spatial and temporal variability in soil and waste water systems diffusional constraints and the numerous feedbacks of the process itself in the different environments ((Benckiser 1994; Sümer et al. 1995). Products of anaerobic energy-conserving processes are, in addition to CO_2, N_2, Mn^{2+}, Fe^{2+} and H_2S, the greenhouse gases CH_4, NO and N_2O. Nitrous oxides are well-documented gaseous products of the lithotrophic ammonia-oxidizers (Blackmer et al. 1980; Tortoso and Hutchinson 1990; Hooper et al. 1990; Martikainen and de Boer 1993; Sümer et al. 1996) and the heterotrophic denitrifiers (Abou-Seada and Ottow

Dedicated to Professor J.C.G. Ottow on the occasion of his 60th birthday

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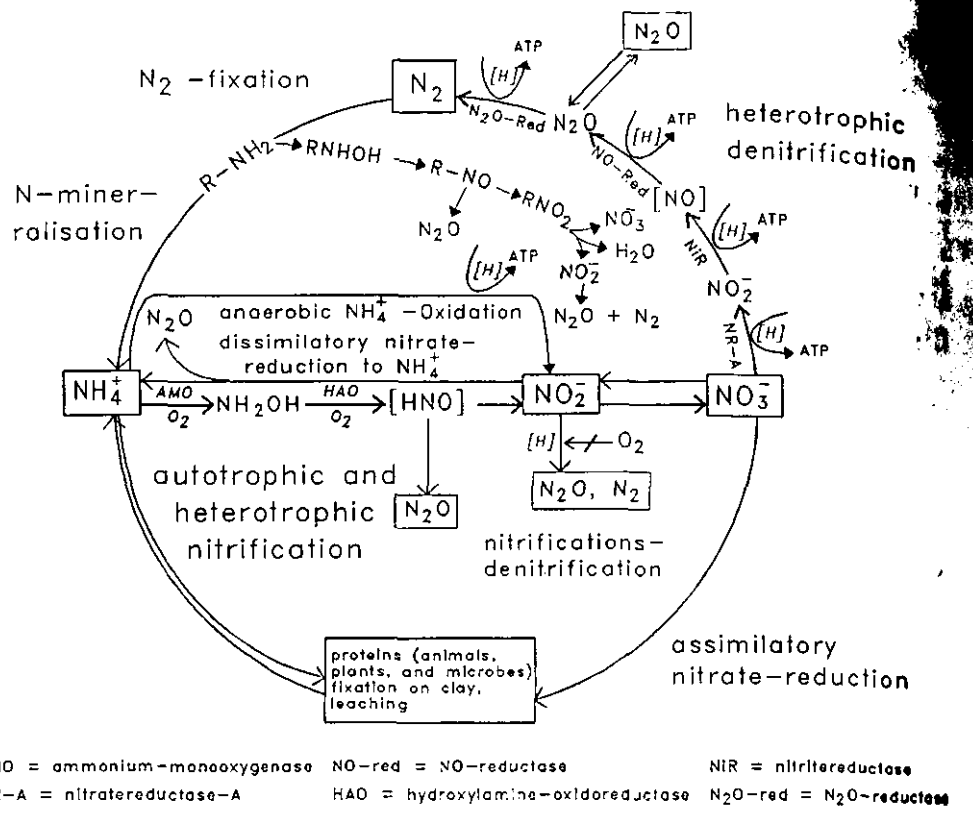

Fig. 1 Microbial N_2O-forming processes known so far

1985; Myrold and Tiedje 1985; Benckiser and Simarmata 1994). The main and so far known N_2O-forming processes are summarized in Fig. 1. "Nitrification-denitrification" and heterotrophic denitrification appear to be the dominant processes. Essential prerequisites for both energy-conserving processes are nitrate, nitrite and available carbon. In natural and agricultural used soils, nitrification and denitrification contribute to the N-availability of plants; and in waste water, which might be considered a model for soil solutions rich in organics and N, both processes are increasingly being used to avoid eutrophication by N-species in the "receiving" water. As side effects of soil and waste water management, increasing N_2O emissions are being reported (Simarmata et al. 1993; Lorch 1993; Wicht and Beier 1995). N_2O, which has a mean resistance time of 132 years in the troposphere combined with a considerable ozone-destroying capacity in the stratosphere, has a 260 times greater greenhouse effect potential than CO_2 (Papen and Seiler 1994). Consequently, this trace gas should not be underestimated as a potential hazard on a long-term scale. Unfortunately, local and global estimates of the quantity of N_2O from various terrestrial ecosystems, oceans, waste water treatment systems and combustion are scarce, if not totally lacking (Ottow and Benckiser 1994).

This paper compares the N_2O emissions from various cropping systems and municipal waste water treatment plant units (activated sludge, nitrification, denitrification tanks) in order to ascertain to what extent the N_2O releases depend on nitrate (end product of nitrification) and carbon availability (prerequisite of heterotrophic denitrification). Another objective of this paper was to estimate the amounts of N_2O still stored in the soil body in order to obtain information on the fate of N_2O during diffusion from its place of production to the atmosphere.

Materials and methods

Experimental design and field management

Cropping systems

The experiments to quantify N_2O emissions from different cropping systems were carried out at the Experimental Station of the Institute for Agronomy and Plant Breeding, Justus-Liebig-University, Gießen, Germany. The topsoil (0–35 cm) was a clayey loam (clay 31%, silt 60% and sand 5%) with a pH (H_2O) of 6.2. Total organic C was 1.41% and total organic N 0.15%. The soil was classified as an allochtone brown earth derived from river sediments. The average precipitation of the experimental site is ca. 640 mm year^{-1} and the mean temperature 8.9°C. The experimental design was a Latin Square (8×10 m, four replicates each), which was established in 1982 to obtain information about the biological N-supply of different cropping systems. The different cropping systems at this site have three rotation units in common, winter wheat, winter rye and barley, and one different one: A=*Vicia faba*, B=*fallow land*, H=*Avena sativa*, K=*Trifolium campestre* and M=*Zea mays*. Further treatments were achieved by different fertilizer applications (1–4). Treatments A-M1 received no fertilizer. Treatments A-M2–4 received 37 kg P and 100 kg K ha^{-1} year^{-1} and different rates of N as follows: Treatments A-M2 received no nitrogen, treatments A-M3 80 kg N ha^{-1} year^{-1} and the treatments A-M4 120 kg N ha^{-1} year^{-1} as ammonium nitrate split into three applications (40, 20, 20 or 60, 30, 30). During the growth of *Vicia faba* or *Zea mays* no nitrogen or 180 kg N ha^{-1} year^{-1}

was applied, respectively. The *Trifolium campestre* rotation was cut twice during the growing season followed by mulching each time.

Before starting with the N_2O-flux measurements in October 1991, the experimental site was ploughed (25 cm deep), levelled by rotary harrow and sown with winter rye (20 September 1991). Phosphorus and potassium were applied on 10 March and the first nitrogen was applied on 20 March 1992. Herbicides (Starane, 0.8 l ha^{-1}, and Pointer3, 25 g ha^{-1}, Dow Elanco, Germany) were spread on 7 April and the second N split application took place on 21 April. The fungicide Corbel (0.75 l ha^{-1}, BASF, Germany) and the growth regulator Terpal C (2.0 l ha^{-1}, BASF, Germany) were applied together with the third N split on 5 May. The N_2O surface flux measurements were carried out more frequently over the winter months (1 October 1991 to 14 May 1992). During the period 1 December 1991 to 1 March 1992 the soil temperature was below 5 °C. Consequently, the N_2O surface flux measurements were carried out mostly during October, November, March, April and May at temperatures between 5° and 15 °C. The soil humidity varied between 20% and 25% (w/w) during this period.

Waste water treatment plant

The in situ N_2O measurements in an aerated fluid-bed reactor, and a nitrification-denitrification tank, were carried out at the municipal waste water treatment plant of Gießen, Germany (Fig. 2). In this plant the waste water from about 150000 inhabitants is purified daily. One-third of the waste water, with a mean BOD_5 of 170 (10–510) mg l^{-1}, COD of 196 (32–756) mg l^{-1}, NH_4^+-N of 33 (5.7–52) mg l^{-1} and NO_3^--N of 2.5 (0.3–13.8) mg l^{-1} (Sümer et al. 1996), passes through a screen, a grit chamber (1170 m^3) and a sedimentation tank (2080 m^3, 45 min, to remove the coarse particles), a trickling filter filled with coarse and porous lava stones for biofilm development during irrigation with waste water (3900 m^3, 50 min) and a aeration tank (1700 m^3, 120 min, for BOD_5 degradation and NO_3^--formation) before it is discharged to a secondary sedimentation tank (2980 m^3, 18 min), a polishing pond (11000 m^3, a type of lake in which the biologically treated waste water is ameliorated during longer periods of residence) and the "receiving" water. The other two-thirds or 52000 m^3 day^{-1} pass the screen, the grit chamber (1170 m^3) and the sedimentation tank (1760 m^3, 45 min), but then goes through an anaerobic tank (4000 m^3, 70 min) and nitrifying and denitrifying units (16000 m^3, 140 min) before the clarified waste water is discharged into the secondary sedimentation tank (13450 m^3, 18 min), the polishing pond (11000 m^3) and the "receiving" water (Fig. 2; Linn et al. 1995).

Nitrous oxide flux measurements and N_2O quantification inside the soil body

The in situ N_2O fluxes to the atmosphere in the cropping systems were determined as described by Schwarz et al. (1994), but without flushing the soil with C_2H_2. Briefly, open chambers (50 cm×10 cm×15 cm, four in parallel) equipped with a sharpened steel base and a removeable lid were inserted about 5 cm deep into the soil between the plant rows. The plants under the boxes were removed. Before each N_2O-sampling period of 4 h day^{-1} the lid was made air tight and the open chambers continuously flushed with an air stream (20 l h^{-1}) using a vacuum pump (Vacubrand M2, Germany) and flow meters (Platon, Germany). After each N_2O measurement the lids were removed to avoid the development of a different microclimate under the cover boxes. The N_2O released into the air stream was collected on three 0.5-nm molecular sieve traps (19 cm in length, 2.5 cm in diameter; ca. 35-g 2-mm pellets; Merck, Germany).

The development of a method to quantify the N_2O retained by the soil body has been described in detail by Weiske et al. (1995). Briefly, soil samples from the cropping system K2 and M3 down to 90 cm depth were taken in the field (February–July 1994) with an auger (three parallel samples per treatment), and divided into subsamples (0–30 cm, 30–60 cm, 60–90 cm). To avoid major degassing of N_2O, portions of the parallel soil samples were transferred straight from the auger into 100-ml Erlenmeyer flasks (about 40–60 g fresh soil). The Erlenmeyer flasks were immediately closed and made air tight with a rubber septum seal (Venerett, France) and stored in a cooling box (ca. 4 °C) to keep the microbial activity at a low level during the transfer to the laboratory (ca. 15 min). On reaching the laboratory the flasks were heated in a water bath to 80 °C over a period of 80 min. Using this procedure, N_2O formation during incubation was completely suppressed (pasteurization) and a total release of the N_2O trapped in the soil body was achieved (Weiske et al. 1995).

The in situ N_2O emissions from the aerated fluid-bed reactor and the nitrification-denitrification tank of the waste water purification plant in Gießen were quantified with floating self-constructed open PVC covers (dimensions 60×40×20 cm, six in parallel) as recently described by Körner et al. (1993) and slightly modified by Sümer et al. (1995). For sampling dates see Sümer et al. (1995) and Linn et al. (1995). The atmosphere of the covers was transported continuously

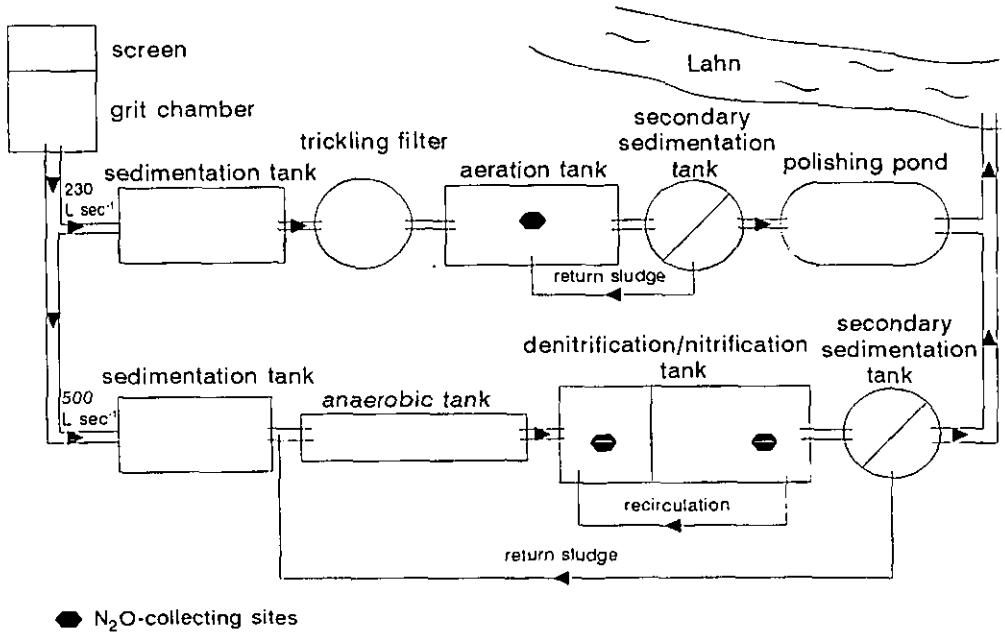

Fig. 2 Schematic diagram of the waste water treatment plant in Gießen, Germany, and the N_2O-measuring sites

by a membrane pump (air stream of 90 l h^{-1}) over silica gel and sodium hydroxide traps (to remove the H_2O and CO_2, respectively) into colums of 0.5-nm molecular sieve filled with 2-mm pellets (Merck, Germany, to absorb the N_2O quantitatively). To obtain a homogeneous air stream, two uniformly perforated PVC plates (with holes, each 0.8 cm in diameter) were fixed perpendicularly to the air stream inside the chambers. At each sampling the chambers were flushed for 2 h. To compensate the frequently changing air fluxes by the waste water aeration device, the total air stream of 90 l h^{-1} was subdivided into a 20-l-h^{-1} stream (for N_2O collection) and a 70-l-h^{-1} bypass (to avoid uncontrolled lifting up of the chambers). Both air streams were controlled by flow meters (Platon, Germany).

The N_2O released from the different cropping systems as well as from the different units of the waste water treatment plant, absorbed on the molecular sieve, was desorbed in evacuated Erlenmeyer flasks ca. 1 l in volume containing 150 ml water (Benckiser et al. 1995). Gas portions of the atmosphere of the Erlenmeyer flasks either used to desorb N_2O from the molecular sieve or to release N_2O from the soil solution were analysed gas chromatographically. A gas chromatograph equipped with an electron capture detector (ECD; Sigma 300, Perkin Elmer, Germany; Porapak Q column 2 m, 120 mesh; N_2 carrier gas; detector 300 °C; injector 150 °C; column 50 °C; flow 30 ml min^{-1}) was employed to quantify the N_2O. The amount of N_2O dissolved in the water was calculated according to Moraghan and Buresh (1977). The N_2O concentration in the surrounding air was determined in separate units and subtracted from the mreasurements. N_2O surface fluxes were calculated as milligrams N_2O per metre per hour.

Analytical procedure

Soil samples (ten samples per treatment) were collected with an auger (0–30 cm), combined, mixed and freed from roots and organic remains by passing through a 2-mm sieve and stored at 4 °C. NO_3^- was extracted following the procedure of Scharpf and Wehrmann (1976). Twenty grams of fresh soil was shaken with 100 ml extracting solution (1 *N* NaCl·to 0.1 *N* $CaCl_2$) for 1 h and filtered (595 1/2, Schleicher and Schüll, Germany). The filtrate was analysed for NO_3^- by ultraviolet absorption at 210 nm. Water-soluble carbon (C_{H_2O}) was determined following the procedure of Burford and Bremner (1975). Twenty grams of air-dried soil was shaken with 40 ml distilled water for 30 min at 130 U min^{-1}, which 10 ml of a 4 *M* Na_2SO_4 solution was added and by shaking poured through a filter (595 1/2, Schleicher and Schüll, Germany). From the filtrate an aliquot of 20 ml was transferred into 100-ml Erlenmeyer flasks and the water evaporated at 80 °C. Then 5 ml 2 *N* $K_2Cr_2O_7$ and 9 ml concentrated H_2SO_4 were added and allowed to oxidize for 90 min at 120 °C. After cooling down to room temperature, the samples were filled with distilled water to the 50-ml marker, centrifuged for 15 min (3000 U min^{-1}, Heraeus, Germany) and the reduced Cr^{1+} ions analysed photometrically at 578 nm (Hitachi, Germany).

Simultaneously to each N_2O measurement waste water samples were taken from a depth of 0–20 cm. NO_3^- was determined in 100-ml waste water samples by extraction with 100 ml of a 0.1 *N* NaCl and 0.01 *N* $CaCl_2$ solution for 1 h and analysis of the filtrate by ultraviolet absorption at 210 nm (Navone 1964). The biological oxygen demand after 5 days of incubation (BOD_5) was determined manometrically with a WTW BOD-analyser (Model 1002, Germany) or with a Sapromat BOD analyser (type B 12, Voith, Germany) in samples taken close to the N_2O-measuring sites. The unfiltered waste water samples (150–250 ml) were incubated over 5 days at 20 °C in a water bath and the O_2 demand l^{-1} was recorded directly.

Simple and multiple regressions for evaluating the soil and waste water parameters which might influence the microbial N_2O formation were carried out with the SPSS for Windows Release 5.01.

Results

Figure 3 compares the nitrate contents at the N_2O-sampling sites with the N_2O surface fluxes. N_2O was emitted by all the cropping systems studied and the domestic waste water treatment plant units. The cropping systems released mean amounts of 0.08, 0.14, 0.22 and 0.13 mg N_2O m^{-2} h^{-1} (Fig. 3 A–D) and the aerated, nitrifying and denitrifying tanks released mean amounts of 1.0, 19.5 and 0.5 mg N_2O m^{-2} h^{-1} (Fig. 3 E–G), respectively. The highest N_2O fluxes in the cropping systems occurred in the plots fertilized with 120 kg N ha^{-1} $year^{-1}$ and with maize in the rotation (Fig. 3 C). In the waste water treatment plant the highest N_2O releases occurred in the nitrification tank (Fig. 3 F). As frequently observed with field data, the correlations between soil or waste water nitrate and N_2O emissions were relatively poor. A positive correlation significant at the 95% probability level was found for the field site which received no mineral nitrogen fertilization (Fig. 3 D) and in the aerated tank which receives nitrogen almost exclusively in organic form or as NH_4^+. The correlation coefficients for the cropping systems (Fig. 3 A–D), in particular, but also for the waste water treatment plant (Fig. 3 E–G) suggest that decreasing N-fertilization or N-inputs improves the positive relationship.

Figure 4 compares the water-soluble or available carbon contents at the N_2O-sampling sites with the N_2O surface fluxes. In contrast to nitrate (Fig. 3) the correlation coefficients were generally negative. The cropping system with *Vicia faba* in the rotation and fertilized with 80 kg N ha^{-1} $year^{-1}$ showed a significant negative correlation (Fig. 4 A). This significant correlation, supported by the others, obviously shows that with increasing amounts of available carbon the N_2O emissions are lowered.

The water-soluble carbon- or BOD_5-to-nitrate ratios at the N_2O-sampling sites are compared with the N_2O surface fluxes in Fig. 5. A significant negative correlation is shown by the cropping system fertilized with 120 kg N ha^{-1} $year^{-1}$ and *Vicia faba* in the rotation (Fig. 5 B) and by

Table 1 Amounts of nitrous oxide (mg N_2O kg^{-1} dry soil) released from soil samples taken during 1994 from different depths of two cropping systems planted to winter rye (K2 = no N-fertilization, M3 = 80 kg N ha^{-1})

Cropping system	Soil depth (cm)	Sampling dates during 1994						
		9.2	17.2	1.3	22.4	19.5	21.6	27.7
K2	0–30	0	0	8.8±0.13	3.6±0.05	7.8±0.08	6.6±0.07	4.2±0.05
	30–60	0	0	0	1.4±0.08	2.6±0.04	0	0
	60–90	0	0	0	0	0	0	4.3±0.08
M3	0–30	11.0±0.13	6.1±0.1	35.4±0.2	45.8±0.9	21.5±0.3	19.5±0.22	6.8±0.15
	30–60	0	6.1±0.17	11.2±0.14	2.8±0.02	9.9±0.13	9.1±0.16	3.1±0.02
	60–90	0	3.5±0.05	0	0	3.3±0.02	1.4±0.03	0

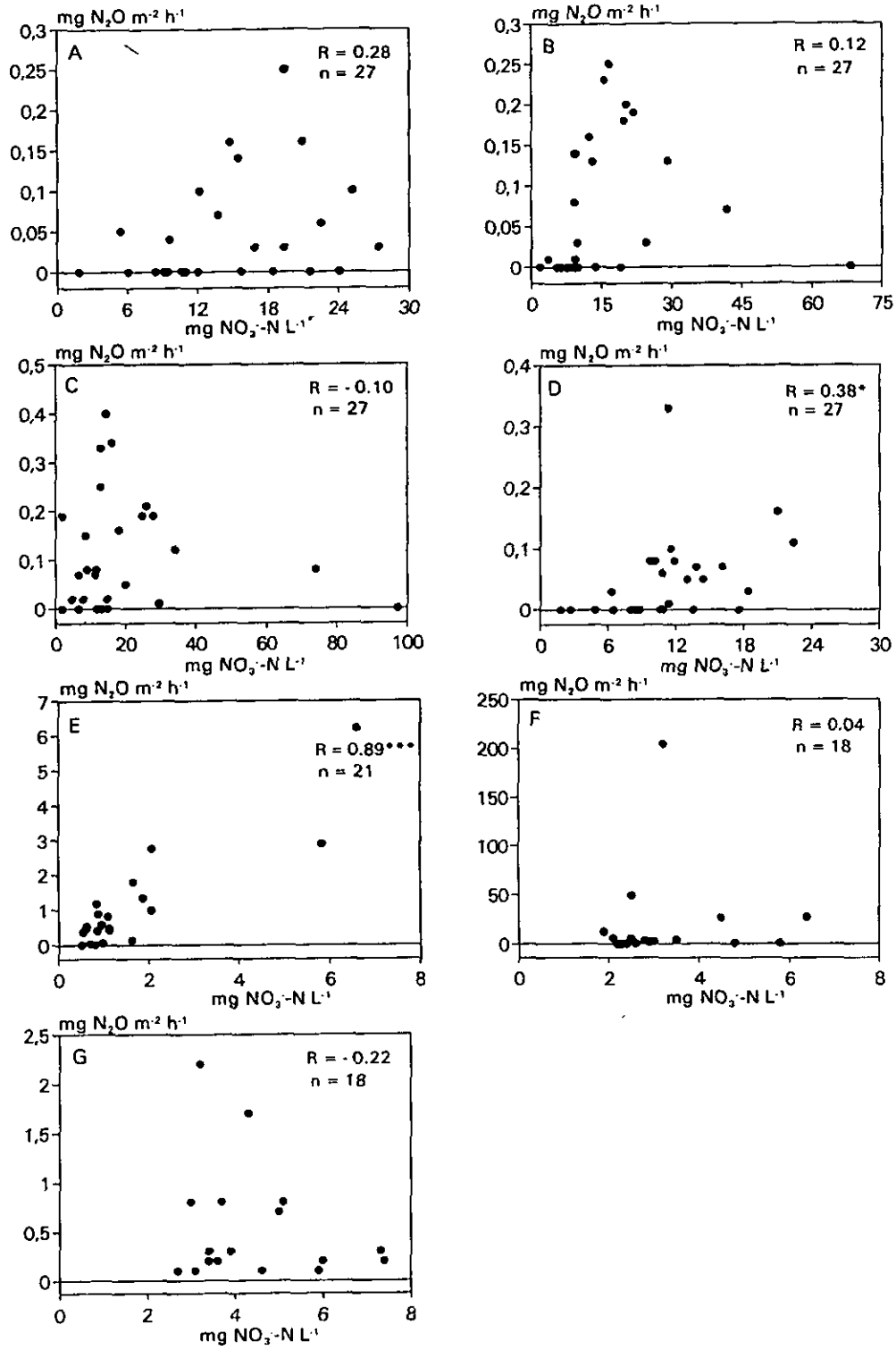

Fig. 3A–G Nitrous oxide surface fluxes from different cropping systems and the aerated, nitrifying and denitrifying tanks of the waste water treatment plant in Gießen, Germany, in comparison to the nitrate contents. A A3=*Vicia faba*, 80 kg N ha^{-1} kg; B A4 *Vicia faba*, 120 kg N ha^{-1}; C M4 *Zea mays*, 120 kg N ha^{-1}; D K2 *Trifolium campestre*, no nitrogen fertilization; E aerated tank; F nitrification tank; G denitrification tank

the aerated tank (Fig. 5E). From Fig. 5B, E as well as tentatively from the other investigated cropping and waste water systems, it can be concluded that the wider the carbon-to-nitrate ratio the lower the N_2O fluxes. This suggests that the cropping and waste water treatment systems can be potential N_2O sinks if organic carbon availability is high compared to nitrate. Water-soluble carbon-to-nitrate ratios above 60 in the cropping systems and BOD_5-to-nitrate ratios above 600–800 in the waste water treatment systems generally correspond to decreasing N_2O emissions.

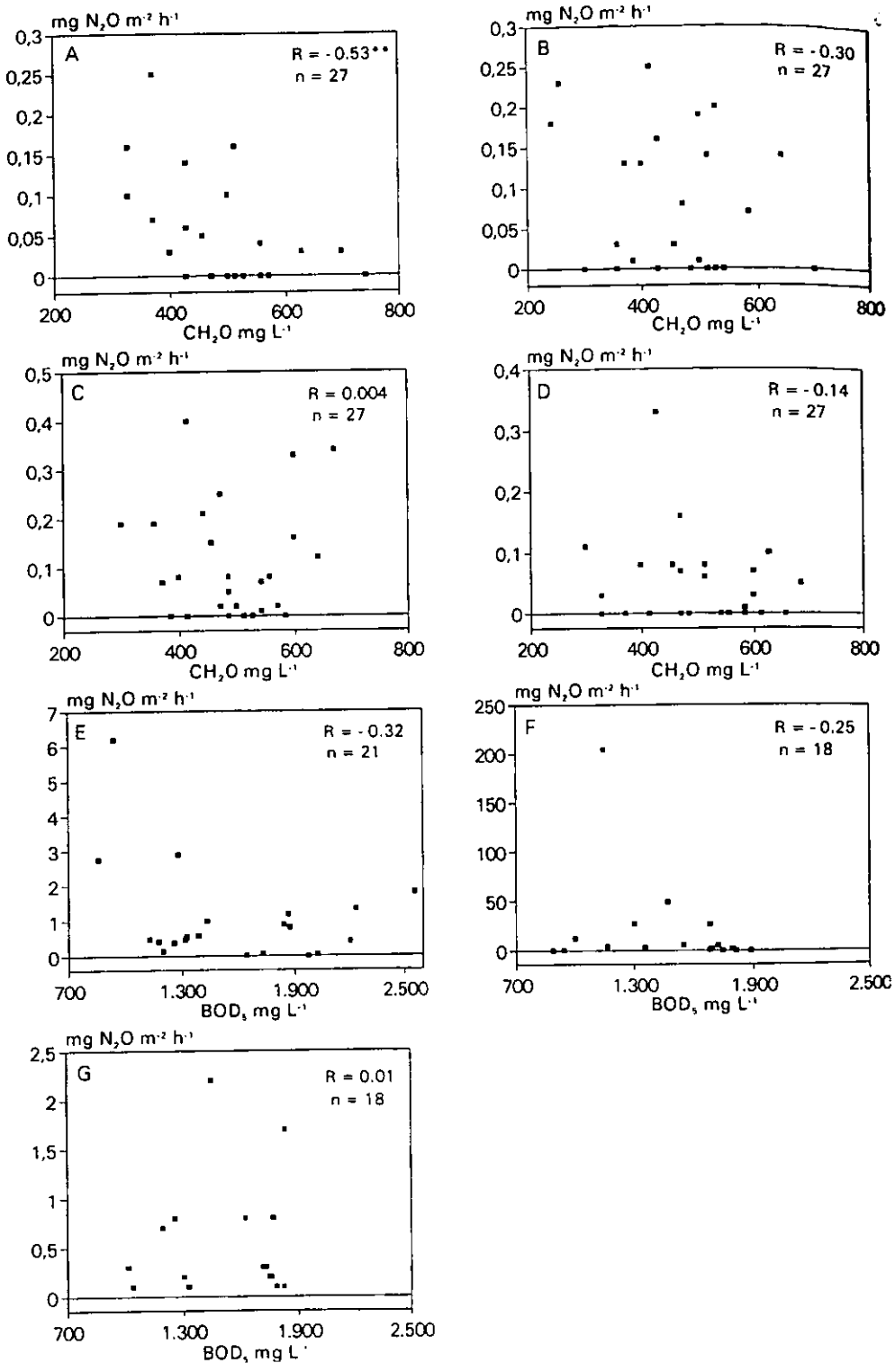

Fig. 4A–G Nitrous oxide surface fluxes from different cropping systems and the aerated, nitrifying and denitrifying tanks of the waste water treatment plant in Gießen, Germany, in comparison to the water-soluble carbon contents or the biological oxygen demand (BOD_5). For legend see Fig. 3

Table 1 shows the amounts of N_2O trapped during February to July 1994 in the soil bodies of the cropping systems M3 (80 kg N ha^{-1} $year^{-1}$) and K2 (no N-fertilization) down to a depth of 90 cm. Despite an unknown amount of N_2O which might be lost to the atmosphere during the transfer of the soil samples from the auger to the Erlenmeyer flasks, the following conclusions can be drawn from Table 1. First, both cropping systems cover significantly different amounts of N_2O in their soil bodies. The retained N_2O amounts of the variant M3 are about 40 times higher than those from the variant K2. Secondly, in both cropping systems the highest amounts of N_2O are in

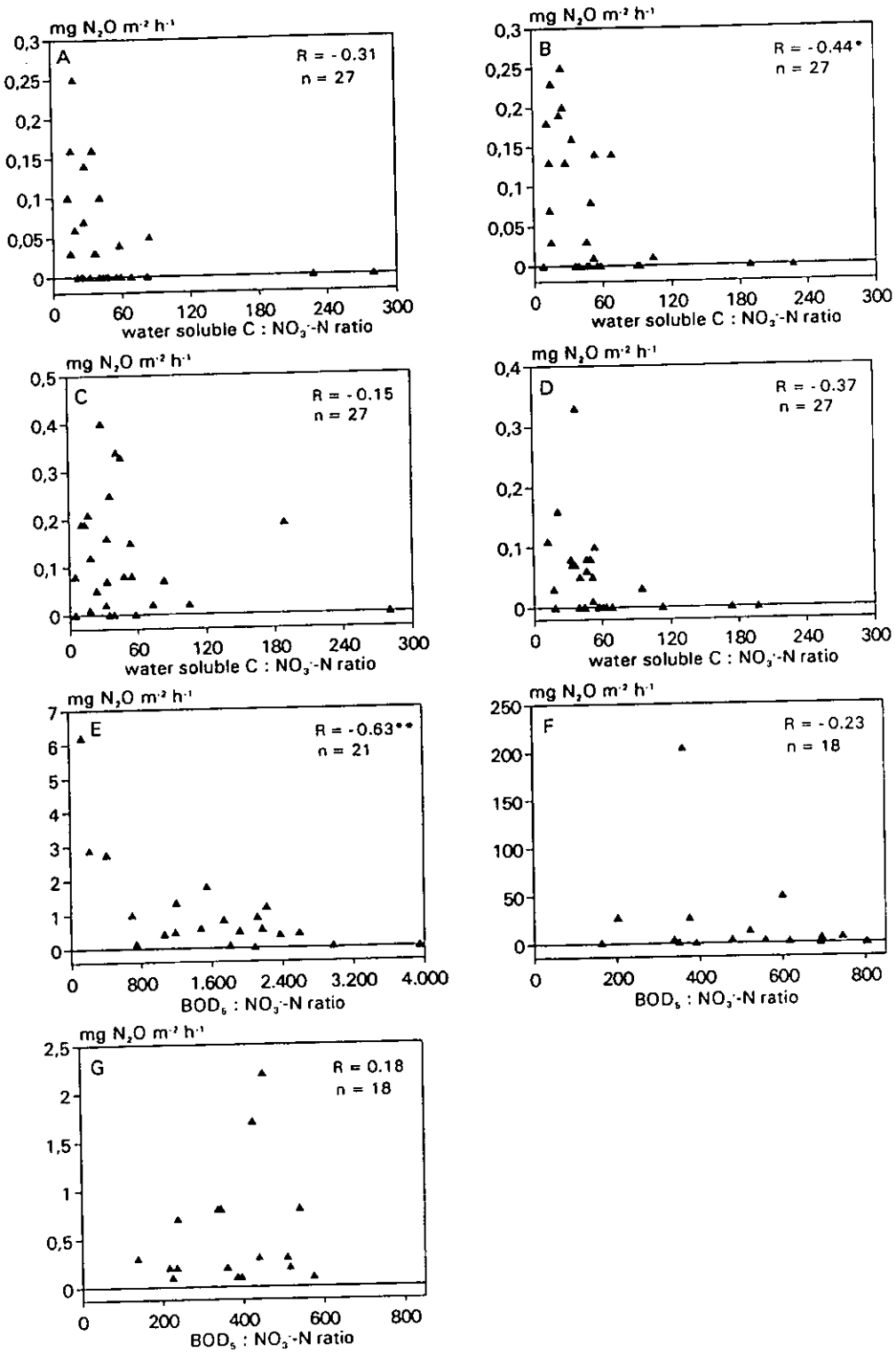

Fig. 5A–G Nitrous oxide surface fluxes from different cropping systems and the aerated, nitrifying and denitrifying tanks of the waste water treatment plant in Gießen, Germany, in comparison to the water-soluble carbon or biological oxygen demand (BOD_5)-to-nitrate ratio. For legend see Fig. 3

the 0- to 30-cm soil layer. Third, in the deeper soil layers (30–60 cm and 60–90 cm) the amounts of N_2O are still considerable. This N_2O formation seems to be restricted to distinct periods during the cropping season.

Discussion

The data presented show that the N_2O emissions from soils and waste waters seem to depend predominantly on the carbon-to-nitrate ratio. In some of the cropping and waste water systems studied it was even found that N_2O

release was highly affected by the availability of both carbon and nitrate. The various N_2O-forming processes (Fig. 1) can occur simultaneously in soil aggregates and sewage sludge flocculates, even if the soils and waste water are well aerated. Most of these N_2O-forming processes need available carbon as the electron donator and nitrate or nitrite as the electron acceptor. In the presence of oxygen, the availability of carbon and nitrate is generally improved. In reverse, oxygen diffuses more slowly than nitrate into the inner parts of the soil aggregates or sewage flocculates. Consequently, instead of O_2, nitrate and nitrite are preferentially used as e^--acceptors for energy conservation and N_2O may be formed. N_2O can now diffuse into the atmosphere of is dissimilated into N_2 by acting as the e^--acceptor. Depending on the nitrate-to-carbon ratio more or less N_2 will be released during denitrification. In this context, it is not surprising that the N_2O release from the well-aerated nitrification tank is about 40 times higher than from the un-aerated denitrification tank. Simple (Figs. 3–5 E) as well as multiple correlations between the spatially and temporally highly variable in situ N_2O emissions and factors such as water-soluble carbon (C_{H_2O}), BOD_5, NO_3^-, NO_2^- concentrations and pO_2, temperature and pH showed that in the aerated tank N_2O increased with NO_3^- and NO_2^- concentration as well as with the pH, but remained nearly unaffected by changes in pO_2, temperature and C_{H_2O} or BOD_5 (Sümer et al. 1995, 1996). Nitrite and nitrate has been identified as one of the proximal controls in the denitrification of nitrifiers (Blackmer et al. 1980; Bock et al. 1988; Tortoso and Hutchinson 1990; Robertson and Kuenen 1991; Hooper et al. 1990; Mosier and Schimel 1993; Sümer et al. 1996) and in the heterotrophic denitrification process (Tiedje 1988; Benckiser 1994; Ottow and Benckiser 1994). In well-aerated topsoils and waste water treatment systems N_2O emission should be ascribed essentially to nitrification-denitrification rather than heterotrophic denitrification. At reduced oxygen pressue the ammonium and nitrite oxidizers of the genera *Nitrosomonas* and *Nitrobacter* are able to use organic compounds and/or NH_4^+ for supplying reduction equivalents. With nitrite and/or nitrate as electron acceptors in order to continue their energy conservation (Körner et al. 1993; Sümer et al. 1995; Bock et al. 1995).

$$NH_4^+ + NO_2^- \rightarrow N_2 + 2H_2O \quad DG^0 = -360\,kJ\,mol^{-1} \quad (1)$$

This alternative strategy of nitrifiers of continuing ATP synthesis by using part of their own end products of aerobic metabolism at reduced oxygen supply may make a considerable to temporary contribution to the N_2O emissions from topsoils or waste water tanks. If more sophisticated biological waste water purification plants are installed world wide in the next 10–30 years, nitrate pollution and eutrophication of our waters will be reduced but as a side effect the global contribution of waste water purification plants to the atmospheric N_2O budget might be increased. At present the N_2O concentration in the atmosphere is increasing by 0.2–0.3% annually (Papen and Seiler 1994). An annual increase of 0.25% corresponds to an additional amount of 3.5 Tg N_2O-N $year^{-1}$ on a global scale. The amounts of N_2O released from terrestrial ecosystems (agricultural soils, grassland and forests) on a global scale have been budget at ca. 5 Tg a^{-1} and from Germany in the range of 0.075 Tg $year^{-1}$ (Davidson 1991; Dritter Bericht der Enquete-Kommision 1994). The aerated tank and the nitrification-denitrification units (Fig. 2) release about 0.001% N or 0.04% N, respectively, if N_2O-N is expressed as a percentage of the total amount of N received (Figs. 3–5 E–G; Linn et al. 1995; Sümer et al. 1995, 1996). German waste water treatment plants receive N-inputs from 161×10^6 inhabitant-equivalents. Based on the above N_2O percentage and the inhabitant-equivalents and considering that all German waste water treatment plants are equipped with N-elimination units (Fig. 2), the N_2O emission will increase from about 7 Mg N_2O $year^{-1}$ (roughly estimated from the data of the aerated tank; Sümer et al. 1995 a) to about 350 Mg N_2O $year^{-1}$ (roughly estimated from the data of the nitrification-denitrification tank; Linn et al. 1995). Compared to the estimates of total German anthropogenic N_2O emissions (206–276 Gg N_2O $year^{-1}$), these amounts of N_2O, should be considered insignificant. In view of the characteristics of N_2O that it is trapped in considerable amounts in the soil (Table 1; Weiske et al. 1995) and waste water (Linn et al. 1995) as well as being characterized by an atmospheric lifetime of about 100–200 years and a relatively high potential for IR adsorption, this trace gas should not be underestimated (Sümer et al. 1996). Our measurements of trapped N_2O are based on soil samples taken with an auger (Table 1). Compared with the amounts of dissolved N_2O, determined in soil solutions drawn up in a southeastern hardwood forest (USA) using high-flow porous cups (0.07–ca. 8 µg N_2O-N l^{-1}; Davidson and Swank 1990), the amounts of N_2O trapped in the soil body (76 mg N_2O-N l^{-1} in the K2 plots, which received no mineral N-fertilizer; 74.6 mg N_2O l^{-1} in the M3 plots, which received 80 kg ha^{-1} $year^{-1}$) are approximately 10^3–10^4 times higher. The quantities of N_2O trapped in the soil seem to depend on fertilization, seasonal constraints and soil depth. Subsoils have a potential for N_2O production which should be not neglected (Lehn-Reiser et al. 1991). In phreatic aerobic aquifers, nitrate enriched by diposal of human and animal wastes, the N_2O concentrations are up to 3 orders of magnitude higher than the concentrations expected as a result of equilibrium with the atmosphere (Ronen et al. 1988). This N_2O, which may be entrapped in soil aggregates (Kemper et al. 1985), dissolved in the soil water (Minami 1987) or sorbed on clay minerals and organic substances (Hazzard et al. 1985; Chalamet 1990) may dissimilated to N_2 and/or released somewhere at some future time into the atmosphere, potentially increasing the hazardous greenhouse effect.

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