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ORIGINAL ARTICLE

Emissions of nitrous oxide and ammonia after cauliflower harvest are influenced by soil type and crop residue management

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Abstract The decomposition of vegetable crop residues, e.g. from *Brassica* species, can cause substantial nitrous oxide (N_2O) and ammonia (NH_3) emissions due to their high nutrient and water contents. One promising approach to reduce these harmful emissions is optimizing post-harvest crop residue management. So far published results on the effects of different crop residue placement techniques on N_2O and NH_3 emissions do not give a consistent picture. One of the key issues is the diverse experimental conditions, in particular with respect to soil characteristics. Therefore, we studied the effects of cauliflower residue management, i.e. no residues (control), surface application (mulch), incorporation by mixing (mix), incorporation by ploughing (plough), on N_2O and NH_3 emissions in a 7.5-months field study, using a unique open-air facility featuring three different soils with contrasting soil texture (loamy sand, silt loam, sandy clay loam). Cauliflower residues caused the highest N_2O emissions after ploughing (2.3–3.4 kg N_2O–N ha^{-1}, 1.5–2.2 % of residue-N), irrespective of the soil type. In contrast, ammonia emissions were only affected by the residue placement technique in loamy sand, which exhibited the highest emissions in the mulch treatment (1.9 kg NH_3–N ha^{-1}, 1.2 % of residue-N). In conclusion, under the given conditions incorporating crop residues by ploughing appears to produce the highest N_2O emissions in a range of soils, whereas surface application may primarily increase NH_3 emissions in coarse-textured soils.

Keywords Nitrous oxide · Ammonia · Vegetable crop residues · Soil type · Cauliflower · Tillage

Introduction

Nitrous oxide (N_2O) and ammonia (NH_3) emissions from horticultural fields can be very high due to the high input of nitrogen (N)-rich crop residues (e.g. Glasener and Palm 1995; Velthof et al. 2002; Ruser et al. 2009; de Ruijter et al. 2010b). Upon decomposition these crop residues deliver ammoniacal N (NH_3, NH_4^+), which is both a source of NH_3 emissions and the starting point for a range of microbial processes that can produce N_2O, such as nitrification and denitrification (Butterbach-Bahl et al. 2013). In view of the detrimental effects of these gases, i.e. eutrophication and acidification of ecosystems caused by deposited ammoniacal N (Kuylenstierna et al. 1998)

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and the contribution of N_2O to the greenhouse effect (IPCC 2013), mitigation strategies need to be developed.

One promising approach to reduce these emissions is adjusting the placement (surface application or incorporation) or the technique of incorporating (e.g. rotary tillage or ploughing) crop residues. The few studies investigating the effects of different management options regarding vegetable crop residues on nitrogenous gas emissions have not provided consistent results. For instance, Janzen and McGinn (1991) and Glasener and Palm (1995) proposed that NH_3 emissions are a surface-phenomenon and disappear after incorporation, whereas Nett et al. (2015) found no differences in NH_3 emissions between surface-applied and incorporated residues. Similar ambiguity exists for reported results on N_2O. Some authors observed higher N_2O emissions after incorporation than after surface application (Ambus et al. 2001; de Ruijter et al. 2010a), while others reported similar or even higher emissions from surface-applied compared to incorporated vegetable residues (Baggs et al. 2003; Escobar et al. 2010).

Such apparent disagreement may be explained by differences in: (1) the investigated soils, especially regarding the soil texture; (2) the experimental procedures used, in particular with respect to the preparation and application of crop residues; and (3) the environmental conditions, either in the field (weather) or in the lab (incubation conditions). Systematic investigations into the soil-specific effects of crop residue management options on NH_3 and N_2O emissions in the field, without the perturbing effects of variable site and weather conditions, are scarce; however they are essential to elucidate the factors that control these emissions.

Here, we investigated the effects of crop residue management on in situ N_2O and NH_3 emissions after cauliflower harvest in three different soils at a single experimental site using common experimental procedures. We hypothesized that NH_3 emissions will decrease with soil texture fineness (loamy sand < silt loam < sandy clay loam) as a result of increased cation exchange capacity (NH_4^+ retention) and decreased NH_3 diffusivity (Sommer et al. 2003), as well as with the increasing distance of crop residues from the soil surface (surface application < rotary tillage < ploughing-under) due to the prolonged transport path to the atmosphere (Ni 1999). Nitrous oxide relationships were expected to be reversed. Hence, N_2O emissions should increase with soil texture fineness due to the higher water holding capacity as well as with increasing incorporation depth, both as a result of indirectly reduced O_2 availability at the location of the residues promoting denitrification (Davidson et al. 2000).

Materials and methods

Experimental design

Factor "soil type"

Experiments were performed at an open-air facility site, established in Großbeeren (52°21′N, 13°18′E, 42 m a.s.l.) in 1972, and hosting three different soils (Table 1), filled in concrete plots (2 m × 2 m base area, 0.75 m depth). For convenience, the soils are simply referred to as loamy sand, silt loam, and sandy clay loam on the basis of their particle size distribution (Table 1). All plots were uniformly cultivated with nasturtium (*Tropaeolum* sp.) in the two years prior to the experiment, with removing the aboveground biomass annually in autumn and without the use of fertilizers. At the site, the average annual precipitation is 500 mm $year^{-1}$ and the mean annual temperature is 9.8 °C.

Factor "crop residue management"

Cauliflower was planted on June 14, 2013 with intra- and inter-row distances of 0.5 m, irrigated according to common horticultural practice, and covered with protective nets. Application of fertilizer was based on the expected crop N uptake (251 kg N ha^{-1}) and aiming at a post-harvest soil mineral N (SMN) content of 80 kg N ha^{-1} (0–60 cm). Fertilization was split into an initial application of 200 kg N ha^{-1} to all soils at pre-planting (June 12–13) and a soil-specific second application as a top-dressing on Aug 01. The fertilization rate of the second application depended on the actual SMN contents in 0–60 cm, i.e. the average rooting depth at harvest. These SMN contents constituted 42, 165, and 392 kg N ha^{-1} (July 25) so that top-dressing application rates were adjusted to 164, 41, and 0 kg N ha^{-1} in the loamy sand, silt loam, and sandy clay loam, respectively.

Table 1 Soil characteristics

Original soil type	pH[a]	Particle size distribution (% sand /silt/clay)	Soil texture classification[b]	C_t (g C kg^{-1})[c]	N_t (g N kg^{-1})[c]
Arenic Luvisol	7.2	82/14/4	Loamy sand	8.4	0.77
Luvic Phaeozem	7.5	13/71/16	Silt loam	19.2	1.65
Gleyic Fluvisol	7.3	51/24/25	Sandy clay loam	18.0	1.60

Measurements refer to 0–30 cm topsoil

[a] 20 g dry soil in 50 mL 0.01 M $CaCl_2$ solution

[b] According to World Reference Base (WRB 2015)

[c] Total carbon (C_t) and total nitrogen (N_t) contents. C_t corresponds to organic C since soils are carbonate-free

On Aug. 21, cauliflower was harvested from part of the plots, all the aboveground biomass was removed, and chamber bases were inserted into the soil of these plots. A total of 36 chambers, twelve per soil type, were installed in groups of four, randomly assigned to three replicate plots. On Aug. 22, the cauliflower was harvested from the remaining plots and the respective crop residues were slashed using a lawn mower and homogenized. This produced residues very similar to those left by a flail mower. Four crop residue management treatments were established: an untreated control (control); residues applied to the surface and incorporated by rotary-tillage 24 days later (mulch); residues incorporated by rotary-tillage (mix); and residues incorporated by ploughing (plough). Rotary-tillage was simulated by mixing the residues with the soil to a depth of 15–20 cm using a bar spade and ploughing was simulated by turning blocks of soil with residues on top upside down so that residues were buried at a depth of 15–20 cm using a spade. In the control treatment, the blank soil was also mixed as described for the mix treatment. The residues were applied at a rate corresponding to the actual average amount of crop residues in this experiment, which was equivalent to 54.7 t fresh mass ha^{-1}, 6.7 t dry mass ha^{-1} (60 °C), 2627 kg C ha^{-1}, and 154 kg N ha^{-1} (CNS-Analyzer VARIO EL, Elementar, Hanau, Germany). The four residue placement treatments were randomly assigned to the four chamber bases per replicate plot and soil type. This means that the application was performed individually within the chamber bases, leaving the soil between the chamber bases fallow. Thus, the lay-out reflected a randomized block design within each of the three soil types. In addition to the plots for chamber measurements, two plots per treatment and soil type were entirely subjected to the residue management treatment for auxiliary soil measurements.

Gas flux measurements

Gas flux measurements started after cauliflower harvest and the establishment of the crop residue management treatments. A total of 23 (control, mix, plough) and 26 (mulch) gas flux measurements were conducted between Aug. 24, 2013 and Apr. 10, 2014, i.e. 2–231 days after residue application (DAA). The additional measurements in the mulch treatments were inserted to obtain a continuous observation of 4 days subsequent to mulch incorporation.

Chambers

The static closed chamber technique was applied using chambers made from PVC with a circular base area of 1152 cm^2 and a volume of 40.3–46.3 L (depending on the actual height after incorporation). Chambers were equipped with a vent (1.5 cm diameter, 40 cm length), reflective foil, a gasket seal and fasteners. The chamber bases were inserted to a depth of 20 cm so that 10 cm remained above the surface. Continuously running fans (wind speed 1–2 m s^{-1} at 15 cm distance) were attached to the inner wall of the chamber bases. This ventilation served two purposes, to produce a homogeneous gas concentration within the chamber during closure and to guarantee that air turbulence within the chamber bases was not limiting NH_3 emissions, especially during chamber closure.

N_2O and CO_2 samples

For measurement, four gas samples of 30 mL were taken through a septum-port using a gas-tight syringe and filled into pre-evacuated 20 mL glass vials sealed with butyl rubber stoppers at 0, 20, 40, and 60 min after chamber closure. Gas samples were analyzed for concentrations of CO_2 and N_2O using a gas chromatograph (Shimadzu GC-2014), modified according to Loftfield et al. (1997), and coupled to an electron capture detector. The GC system was tested regularly for stability: 10 repeated measurements of a standard with ambient concentrations always resulted in a coefficient of variation below 3 % and usually of about 2 %. The slope of concentration versus time was calculated using the approach prosed by Leiber-Sauheitl et al. (2014). In short, a linear regression (only three data points available), a robust linear model (RLM) with a Huber-M estimator (Huber 1981), or the HMR model with fixed κ (Pedersen et al. 2010) was used. The conditions for selecting HMR were a lower *p* value and a lower AIC (Akaike 1974) than RLM and a flux not exceeding four-fold of the RLM flux.

NH_3 samples

Cellulose filters were washed in deionized water, dried at 60 °C, impregnated with 2 % phosphoric acid solution (methanol/H_2O: 9/1; 2 mL filter^{-1}), and dried in pure N_2. For measurement, 1–3 filters were fixed to the inner wall of the chamber tops and exposed to the ventilated chamber air for the closing time. For analysis, filters were extracted with 40 mL of deionized water for 1 h in an ultra-sound bath, and the extract analyzed for NH_4^+–N concentration using an EPOS analyzer (Eppendorf, Hamburg, Germany). Unexposed filters were also analyzed on all measurement dates and values were subtracted from those of the exposed filters. The NH_3 emission rate was calculated from the amount of extracted NH_4^+-N on the basis of exposure time and soil surface area. A pre-experiment had shown that this method is suitable for trapping high amounts of NH_3 in the short time frame of one hour and that the quantitative extraction works well (Nett et al. 2015). It should be noted, that this approach can only be regarded as a semi-quantitative method to measure NH_3 fluxes since the artificial ventilation, and thus air turbulence, is a key determinant of NH_3 volatilization. The method may give estimates of NH_3 emissions unconstrained in terms of air turbulence but a calibration against alternative and absolute techniques, such as micrometeorological methods, is desirable.

Cumulative emissions

For calculation of cumulative CO_2, N_2O, and NH_3 emissions during the experimental period, hourly emission rates derived from the measurements were converted into daily emission rates and these were interpolated linearly. To also enable the comparison of temporal dynamics in gas emissions, the observational period was divided into two phases: an early phase of clearly enhanced emission rates in the amendment treatments (2–41 DAA, i.e. Aug. 22, 2013–Oct. 02, 2013) and a late phase comprising the remaining period (42–231 DAA, i.e. Oct. 03, 2013–Apr. 10, 2014). The cumulative emissions of N_2O and NH_3 were related to the total residue-N input to assess the significance of these fluxes. These emission factors for residues were calculated as $\text{EFR} = 100 \times (E_{\text{treatment}} - E_{\text{control}}) \times N_{\text{CR}}^{-1}$, where *EFR* is the emission factor for residues in %, regarding the considered gas and treatment, $E_{\text{treatment}}$ and E_{control} are the cumulative emissions [kg N ha^{-1}] of the considered treatment and the control treatment of the respective soil, and N_{CR} is the total N in cauliflower crop residues [kg N ha^{-1}]. To also assess the differences in C mineralization dynamics of cauliflower residues, the proportions of residue-C respired to CO_2 were estimated by subtracting the cumulative CO_2 emissions of the control from those of the amended treatments (i.e. neglecting potential priming effects) and relating this to the residue-C input.

Soil measurements

Soil mineral N

Sampling for soil mineral N ($\text{SMN} = NH_4^+$-N + NO_3^--N) determination was performed on nine dates after cauliflower harvest. For this, four soil cores were taken on each of the two replicate plots per treatment in the soil depth intervals of 0–30 and 30–60 cm using soil augers with inner diameters of 1.4 and 1.2 cm, respectively. The eight soil cores per treatment and soil depth were pooled in one composite sample, homogenized, and subsampled for further analysis. This means that SMN measurements were not replicated. For analysis, a dry mass equivalent of 25 g of moist soil was extracted with 100 mL of 0.0125 M $CaCl_2$ solution and concentrations of NO_3^--N and NH_4^+-N were determined colorimetrically in the extract using an EPOS analyzer (Eppendorf, Hamburg, Germany).

Soil bulk density and porosity

At the end of the experiment (Apr. 15 + 16, 2014), soil cores (100–250 cm^3) were taken from the undisturbed half of each chamber base at 5 and 15 cm soil depths, respectively. The dry soil bulk density (ρ_b) was calculated as the dry soil mass (105 °C) divided by the soil core volume. The porosity of the soil was then calculated as $\phi = 1 - \rho_b \times \rho_p^{-1}$, where ρ_p is the particle density, estimated according to Rühlmann et al. (2006).

Soil moisture and temperature

For soil moisture and temperature, time domain reflectometry (TDR) and temperature sensors were inserted into the soil (probe rods parallel to the soil surface) on one plot per treatment at soil depths of 5 and 15 cm. The TDR sensor output period (τ) in µs were corrected for temperature effects and then converted into volumetric water contents (WC_{vol}) in $cm^3\ cm^{-3}$ using the provider's standard calibration curves (quadratic in τ). Soil moisture was expressed in % water-filled pore space (WFPS), calculated as $WFPS = 100 \times WC_{vol} \times \phi^{-1}$, using the porosities ($\phi$) derived from the soil core measurements (see above).

Statistics

Statistical analyses were performed using R version 3.1.2 (R Core Team 2014). For parametric tests, the assumptions of normal distribution and homoscedasticity of within-group errors were tested visually (QQ-plots, histograms) and for the latter also by Levene's test. When obvious or statistically significant violations occurred, the non-parametric Kruskal–Wallis rank sum test or generalized least squares with variance function were used as indicated in the text. Multiple comparisons among groups were performed only when the preceding omnibus test yielded significant main effects, controlling the familywise type I error rate using Tukey's HSD test or a non-parametric multiple comparison procedure (Siegel and Castellan 1988) as indicated in the text. Unless stated otherwise, results are presented as arithmetic mean ±1 standard deviation. Statistical significance was stated at $p < 0.05$.

Results

Soil conditions

The undisturbed soil cores taken from within chamber bases at the end of the observation period indicated lower bulk soil densities and higher porosities in the silt loam than in the loamy sand and sandy clay loam (Table 2). Circa 7.5 months after residue application, neither residue management nor the interaction of soil type × residue management showed significant effects on the densities or porosities in two-way ANOVAs.

Soil temperatures (Fig. 1, top, orange/red) showed a typical seasonal pattern with a short period (6–12 days) of soil frost at the end of Jan. / beginning of Feb. 2014, when soil temperatures in 5 cm soil depth fell below zero. In contrast, WFPS (Fig. 1, top, turquoise/blue) only displayed very weak seasonal patterns with slightly higher levels in winter. More distinctly, average WFPS clearly differed between the soils (loamy sand < silt loam < sandy clay loam), and an increase with soil depth was very clear in the sandy clay loam (5 cm: 70 ± 12.8 %, 15 cm: 96 ± 8.0 %), weak in the silt loam (5 cm: 54 ± 8.7 %, 15 cm: 60 ± 7.4 %), and absent in the loamy sand (5 cm: 30 ± 5.6 %, 15 cm: 30 ± 5.2 %). Note that the values of WFPS in the sandy clay loam at 15 cm soil depth temporarily exceeded 100 %. This presumably resulted from underestimating soil porosity during earlier stages, since the soil cores taken at the end of the experiment reflected the soils after they had settled for 7.5 months.

The time courses of SMN content in the topsoil (Fig. 1, bottom) suggested that the initial NO_3^{-1} contents were higher in the loamy sand than in the silt loam and sandy clay loam. Apart from this, the courses of NO_3^{-1} contents reflected well the net N mineralization of the applied crop residues within a period of 2–3 months. The net N mineralization of cauliflower residues, estimated by subtracting SMN contents of the control treatment from those of the treatments receiving residues, was higher in the loamy sand than in the silt loam and the sandy clay loam (Fig. 1, bottom). Remarkably, only in the loamy sand,

Table 2 Soil bulk densities (g cm^{-3}) and porosities (cm^3 cm^{-3}) determined using soil cores from within chamber bases at the end of the experiment, averaged across residue management treatments

Soil type	Soil bulk density (g cm^{-3})		Porosity (cm^3 cm^{-3})	
	5 cm	15 cm[†]	5 cm	15 cm[†]
Loamy sand	1.31 (0.046) b	1.35 (0.035) b	0.50 (0.018) a	0.48 (0.014) a
Silt loam	1.16 (0.048) a	1.22 (0.062) a	0.55 (0.019) b	0.52 (0.024) b
Sandy clay loam	1.28 (0.056) b	1.36 (0.060) b	0.50 (0.022) a	0.47 (0.023) a

Values in brackets denote standard deviations ($n = 12$)

Values not sharing a common letter within a column were significantly different according to two-way ANOVA, followed by Tukey's HSD test or [†] Kruskal–Wallis rank sum test, followed by a non-parametric multiple comparison procedure according to Siegel and Castellan (1988; R function *pgirmess*::*kruskalmc*)

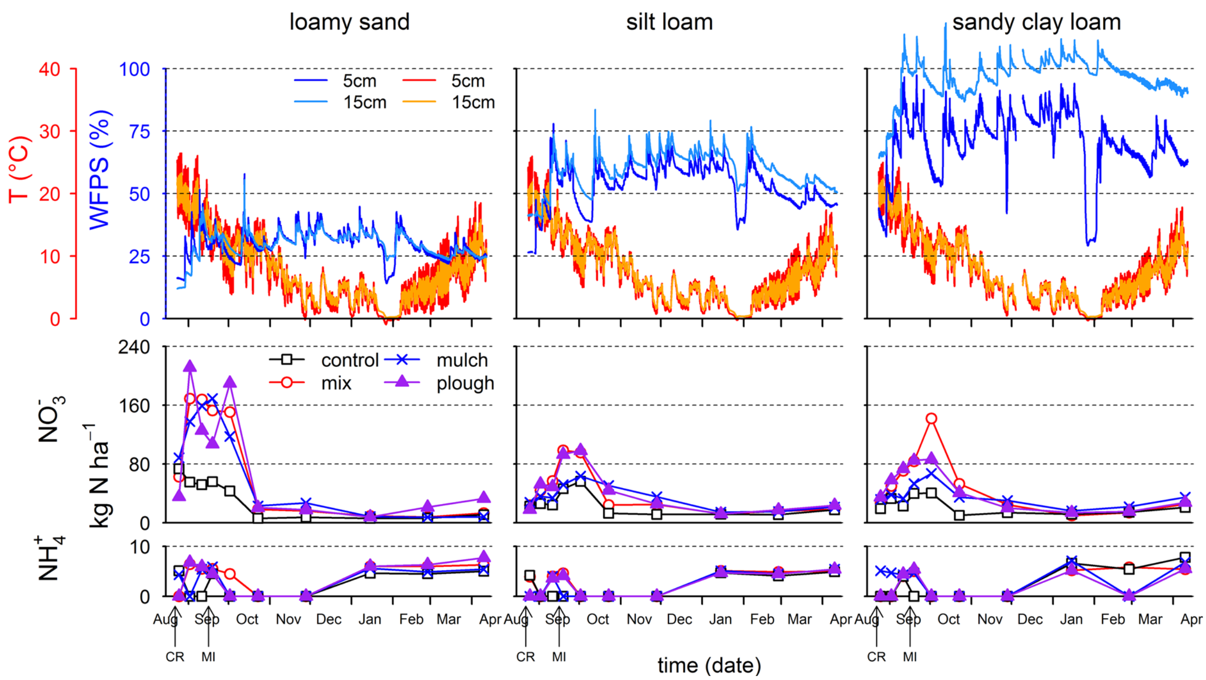

Fig. 1 *Top* time courses of soil water-filled pore space (WFPS) in % of porosity (*turquoise*/*blue*) and soil temperature (*T*) in °C (*orange*/*red*) at 5 and 15 cm soil depths averaged across crop residue management treatments ($n = 4$). *Bottom* time courses of soil nitrate (NO_3^-) and ammonium (NH_4^+) contents in kg N ha^{-1} in the soil depth interval 0–30 cm ($n = 1$, composite samples from eight soil cores). *Arrows* indicate the times of crop residue application (CR) and mulch incorporation (MI). (Color figure online)

considerable net N mineralization was observed in the mulch treatment before incorporation of residues. The SMN contents of all soils decreased during autumn and reached very similar levels in January. This was primarily a result of NO_3^{-1} leaching, as demonstrated by the similar but delayed and flattened courses of NO_3^{-1} in the soil depth interval 30–60 cm (not shown). In the loamy sand, NO_3^{-1} contents dropped dramatically between Oct. 02 and Oct. 23, a period comprising 110 mm of precipitation. Notably, NH_4^+ contents remained below 8 kg N ha^{-1} (0–30 cm) throughout the experiment in all treatments.

Carbon dioxide emissions and mineralization of cauliflower residues

The presented CO_2 emissions (Fig. 2) should only be regarded as a relative measure of microbial activity and residue mineralization dynamics, since 1-h chamber closing times are too long for unbiased measurements.

In the treatments that received crop residues, CO_2 emissions were highest at the first measurement date and generally decreased within the early phase; however the mulch treatments were delayed compared to the mix and plough treatments (Fig. 2).

For the control treatments, which reflect the soils' basal respiration, the cumulative emissions of CO_2 in the early period were significantly higher in the sandy clay loam (1462 ± 244.5 kg C ha^{-1}) than in the silt loam (747 ± 224.2 kg C ha^{-1}) and loamy sand (448 ± 5.5 kg C ha^{-1}), which were statistically indistinguishable (ANOVA, Tukey's HSD test). The same statistical results were obtained for relative cumulative CO_2 emissions per unit of soil organic C in 0–30 cm soil depth (SOC), which in the early period constituted 20 ± 3.3, 11 ± 3.2, and 13 ± 0.2 g CO_2–C kg^{-1} SOC in the sandy clay loam, silt loam, and loamy sand, respectively. In the late period, neither absolute nor relative cumulative CO_2 emissions in the control treatments were significantly affected by soil type (ANOVA). The C mineralization from cauliflower residues was not affected by soil type, residue management, or their interaction in any of the analyzed periods (two-way ANOVAs). On average ($n = 27$), the values constituted 41 ± 13.6, 16 ± 26.2, and 56 ± 35.9 % of residue-C in the early phase, late phase, and total experimental duration, respectively.

Nitrous oxide emissions

At the first measurement, 2 days after residue application, N_2O emissions (Fig. 3) were already elevated compared to the control, especially in the mix and plough treatments in the loamy sand and silt loam as well as the plough treatment in the sandy clay loam. Unlike CO_2 emissions, N_2O emissions were still on the rise, rather than generally decreasing, during the early phase. In some cases this increase first started after a few days, and in the sandy clay loam N_2O emissions even seemed to decrease temporarily. The following peak was most pronounced in the plough treatments, which reached maximum rates equivalent to 368, 215, and 323 g N ha^{-1} day^{-1} in the loamy sand (Aug. 30), silt loam, and sandy clay loam (both Sept. 06),

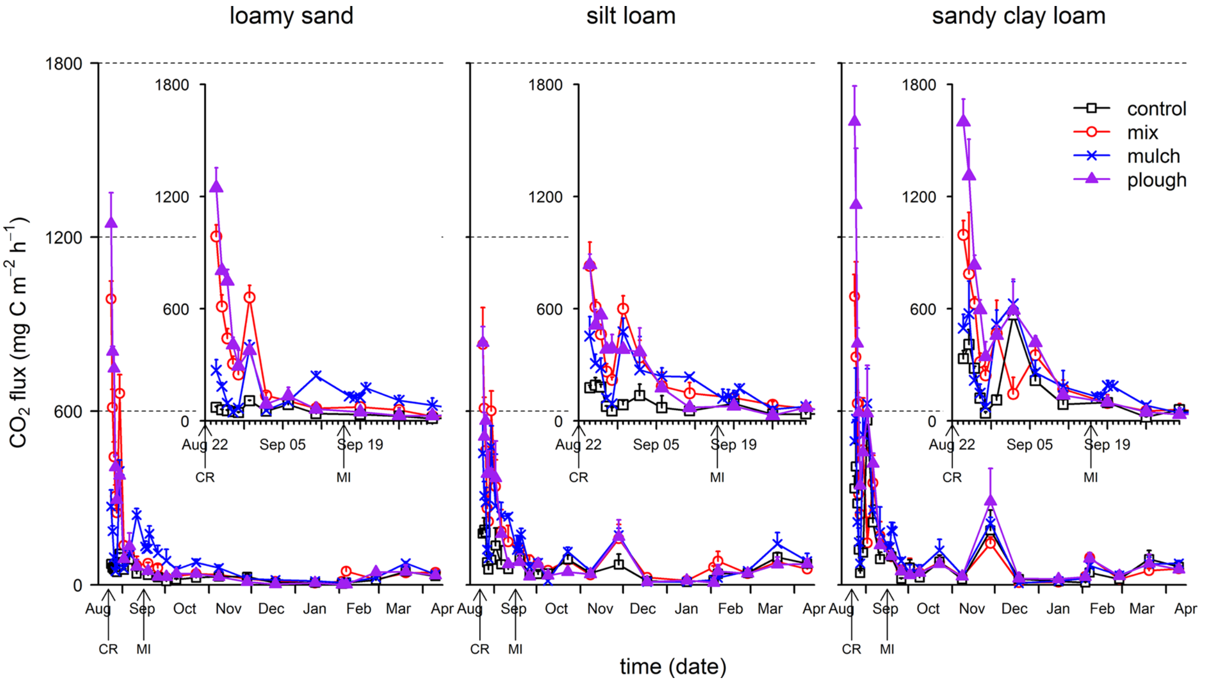

Fig. 2 Emissions of CO_2 in mg C m^{-2} h^{-1}. One-sided *error bars* indicate positive standard errors ($n = 3$). The *insets* show close-up views of the early phase (2–41 days after residue application, DAA). *Arrows* indicate the times of crop residue application (CR) and mulch incorporation (MI)

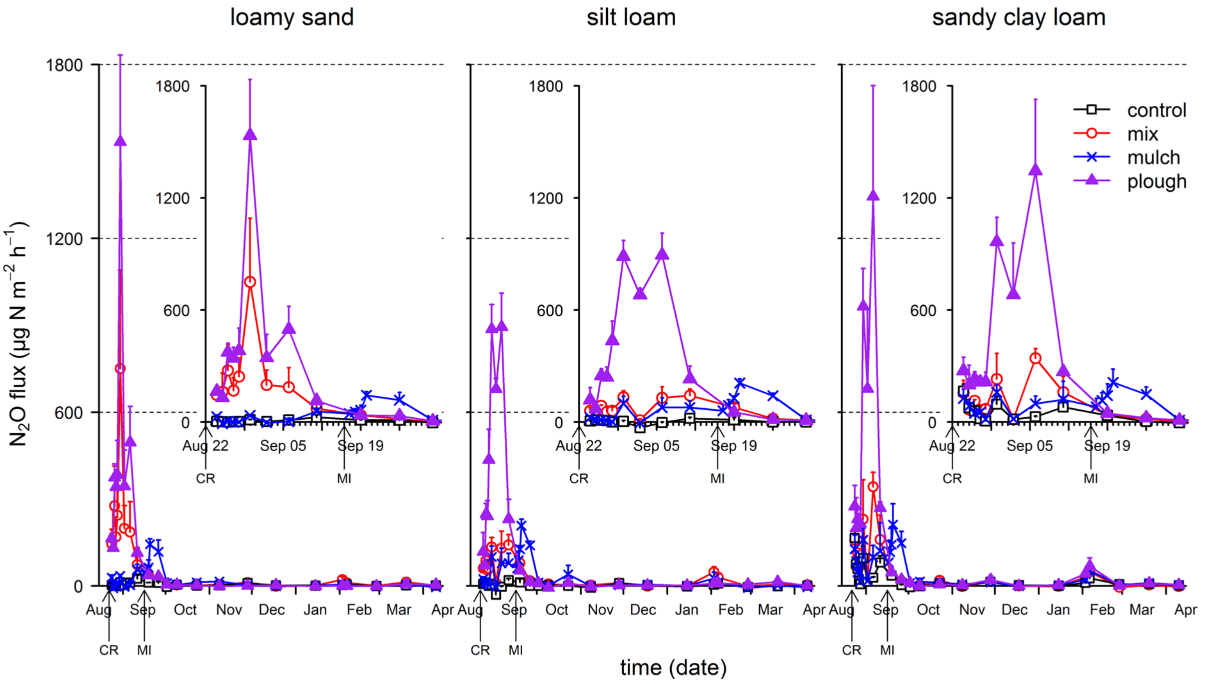

Fig. 3 Emissions of N_2O in µg N m^{-2} h^{-1}. One-sided *error bars* indicate positive standard errors ($n = 3$). The *insets* show close-up views of the early phase (2–41 days after residue application, DAA). *Arrows* indicate the times of crop residue application (CR) and mulch incorporation (MI)

respectively. In the loamy sand, this peak was somewhat higher but more transient than in the other soils.

Whereas in the silt loam and sandy clay loam the mix treatment only exhibited slightly higher emissions than the mulch treatment, a clear difference was obvious in the loamy sand (Fig. 3). Interestingly, N_2O emissions in the mulch treatments started to exceed those of the other treatments 1 or 2 days after incorporation of mulched residues and remained higher for 2 weeks. Concerning the control treatments, emissions during the early phase were also enhanced compared to levels observed in the late phase, especially in the sandy clay loam. Emissions during the late phase were minor in all treatments, not exceeding 5.2, 11.4, and 16.4 g N ha^{-1} day^{-1} in the loamy sand, silt loam, and sandy clay loam, respectively. A small peak in the emissions of all treatments in the silt loam and sandy clay loam occurred when soil temperatures fell below zero late Jan. / early Feb. 2014. While NO_3^- contents peaked around mid-September (loamy sand) or the beginning of October (silt loam, sandy clay loam; Fig. 1), the time of highest N_2O emissions was already two (loamy sand) to three (silt loam, sandy clay loam) weeks earlier (Fig. 3).

The cumulative emissions of N_2O (Table 3) were significantly affected by residue management in both the early phase and over the total experimental duration, however not during the late phase. The soil type significantly affected cumulative N_2O emissions in all phases, whereas no significant interaction effects between soil type and residue management occurred in any phase. The results indicated that during the early phase and over the total duration of the experiment the cumulative N_2O emissions, averaged across soil types, increased in the order: control $<$ mulch $<$ mix $<$ plough; however only the difference between control and plough were statistically significant. Indeed, early phase N_2O emissions in the plough treatment were 17, 4, and threefold higher than those of the control, mulch, and mix treatment, respectively. Concerning the soil type, the mean cumulative emissions consistently increased in the order: loamy sand $<$ silt loam $<$ sandy clay loam; but statistical evidence was only clear for the cumulative emissions in the sandy clay loam being higher than those of the silt loam in the late phase (2.7-fold). A trend (n.s.) of cumulative emissions in the control treatments was also evident following the order: silt loam $<$ loamy sand $<$ sandy clay loam, irrespective of the analyzed phase.

Table 3 Cumulative N_2O emissions (g N ha^{-1}) in the early phase (2–41 DAA), late phase (42–231 DAA), and over the total experimental duration (2–231 DAA)

Soil type	Residue management	Cumulative N_2O emissions (g N ha^{-1})		
		Early (2–41 DAA)†	Late (42–231 DAA)	Total (2–231 DAA)†
Loamy sand	**Mean**	1071 (1006.9) A	158 (159.9) A	1230 (1018.5) A
	Control	98 (44.3)	127 (75.3)	225 (89.8)
	Mix	1294 (781.5)	250 (166.3)	1544 (786.8)
	Mulch	468 (228.3)	177 (256.3)	645 (483.1)
	Plough	2425 (349.1)	79 (132.6)	2504 (481.6)
Silt loam	**Mean**	1094 (1120.0) A	201 (197.0) A	1294 (1173.9) A
	Control	37 (50.0)	51 (89.1)	87 (108.2)
	Mix	714 (113.5)	302 (204.9)	1017 (292.9)
	Mulch	750 (243.2)	253 (322.4)	1002 (565.4)
	Plough	2874 (56.6)	197 (79.1)	3070 (39.8)
Sandy clay loam	**Mean**	1448 (1383.9) A	432 (268.9) B	1880 (1528.4) A
	Control	371 (138.4)	295 (49.0)	666 (149.9)
	Mix	987 (248.3)	341 (162.3)	1328 (157.0)
	Mulch	986 (667.1)	467 (166.1)	1453 (583.3)
	Plough	3448 (1275.1)	623 (494.1)	4072 (1520.7)
Mean	Control	169 (172.0) A	157 (125.5)	326 (281.2) A
	Mix	999 (484.2) BC	298 (159.9)	1296 (484.9) BC
	Mulch	734 (435.4) AB	299 (257.4)	1033 (588.5) AB
	Plough	2916 (796.9) C	300 (358.4)	3216 (1053.1) C
ANOVA *p* values	Soil type (S)	**0.022**	**0.012**	**0.007**
	Residue management (R)	**<0.001**	0.439	**<0.001**
	S × R	0.670	0.621	0.557

Values in brackets denote standard deviations ($n = 3$). Results of the two-way ANOVAs are given at the bottom of the table with *p* values <0.05 printed in bold

Multiple comparisons among group means (upper case letters) and all individual groups (lower case letters) were only performed in the case of a significant main factor effects and interaction effects, respectively. Values not sharing a common letter within a column were significantly different according to Tukey's HSD test or (GLS) a non-parametric multiple comparison procedure according to Siegel and Castellan (1988; R function *pgirmess*::*kruskalmc*)

† Fitted using weighted generalized least squares with variance function depending exponentially on the response variable as covariate (GLS)

Calculations of EFR revealed that the cumulative N_2O emissions accounted for 0.3–2.2 % of cauliflower residue-N over the total experimental duration. Two-way ANOVA suggested that neither soil type ($p = 0.622$) nor the interaction between soil type and residue placement ($p = 0.283$) had an effect on EFR. In contrast, residue placement had a significant effect ($p < 0.001$), and Tukey's HSD test indicated that the plough treatments on average exhibited higher EFR (1.9 ± 0.61 %) than the treatments mix (0.6 ± 0.33 %) and mulch (0.5 ± 0.34 %).

Ammonia emissions

The period when NH_3 emissions (Fig. 4) were elevated due to residue application compared to the control lasted approximately 1 month. Within this period, emissions in the mulch treatments exceeded those from the other residue management treatments most of the time in all soils. In the loamy sand and silt loam, the mix treatments basically exhibited the same pattern as the mulch treatments but at a lower level. In contrast, the differences between plough and control treatments in the loamy sand and silt loam were negligible. At the time of mulch incorporation, NH_3 emissions had already been on the decline in all treatments. Subsequent to the incorporation, emissions in mulch treatments did not immediately stop, but rather continued to decline and showed a local minimum 3 days after incorporation in all soils. The highest average emissions corresponded to N loss rates of 223 (mulch; Sept. 06), 89 (mulch; Sept. 06), and 92 (mix; Aug. 28) g N ha^{-1} d^{-1} in the loamy sand, silt loam, and sandy clay loam, respectively. The negative emission rates that occurred especially during the late phase were a result of methodological artifacts, i.e. higher values for the unexposed blank filter traps compared to filters that had been subjected to 1-h exposure time during a chamber measurement. The lowest values corresponded to -29 g N ha^{-1} day^{-1}, a magnitude of uncertainty that has to be considered when interpreting these results.

Analogous to N_2O, in the late phase only the soil type had a significant effect on cumulative NH_3 emissions (Table 4), with loamy sand emissions being significantly higher than silt loam emissions. Regarding the early phase and total experimental duration, two-way ANOVA revealed both significant main factor effects and an interaction between the factors soil type and residue management. Accordingly, the effects of soil type depended on the effects of residue management and vice versa. Multiple comparisons revealed that in the early phase, emissions in the loamy sand followed the order: control = plough < mix < mulch; in contrast, in the silt loam, a clear distinction could only be made between emissions in the mulch treatment and emissions in the control and plough treatments. Finally, in the sandy clay loam, the only significant difference was between the mulch treatment and the control.

Only the mulch treatment showed a significant soil type effect occurring in the early phase (loamy sand > silt loam = sandy clay loam). Over the total experimental duration, no significant differences among residue management treatments were observed at all in the silt loam or sandy clay loam. In contrast, in the loamy sand the emissions of the mulch treatment were significantly higher than those in the control and plough treatments. Thus, the significant interactions for the most part translated into a strong effect of residue management in the loamy sand, a weak effect

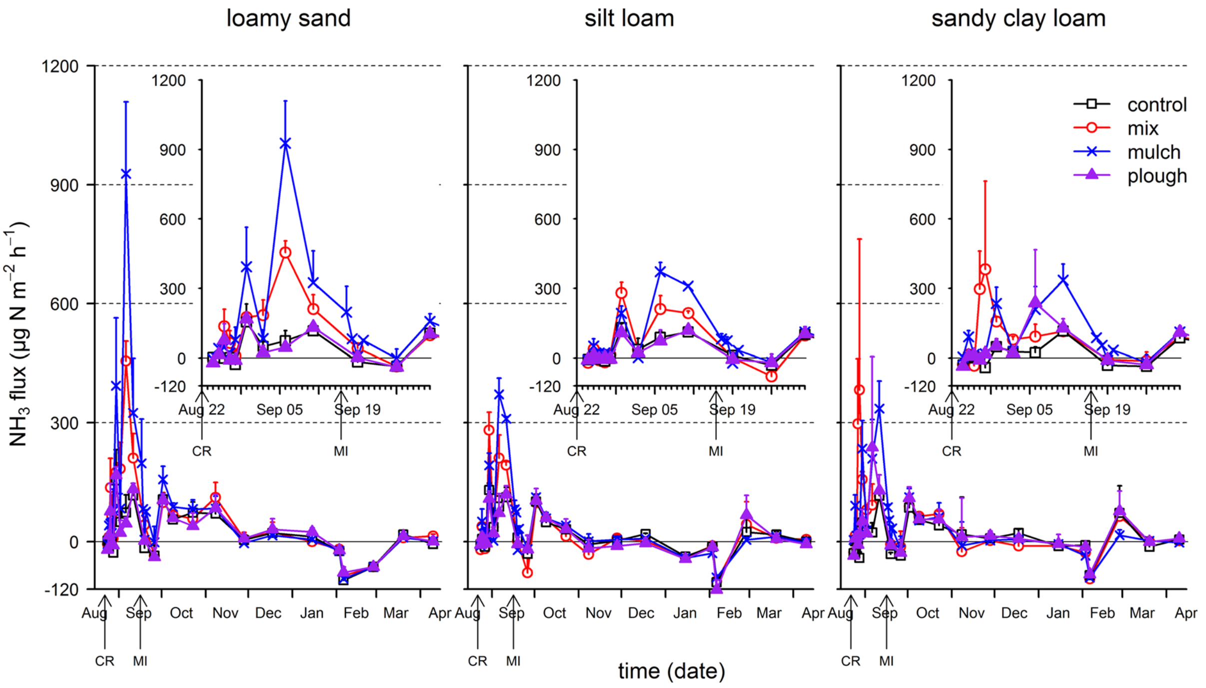

Fig. 4 Emissions of NH_3 in µg N m^{-2} h^{-1}. One-sided *error bars* indicate positive standard errors ($n = 3$). Note that the lower limits of *x* axes lie at –120 µg N m^{-2} h^{-1}. The *insets* show close-up views of the early phase (2–41 days after residue application, DAA). *Arrows* indicate the times of crop residue application (CR) and mulch incorporation (MI)

Table 4 Cumulative NH_3 emissions (g N ha^{-1}) in the early phase (2–41 DAA), late phase (42–231 DAA), and over the total experimental duration (2–231 DAA)

Soil type	Residue management	Cumulative NH_3 emissions (g N ha^{-1})		
		Early (2–41 DAA)	Late (42–231 DAA)	Total (2–231 DAA)
Loamy sand	**Mean**	1054 (779.3) B	508 (194.2) B	1562 (795.8) C
	Control	392 (44.8) ab	410 (149.7)	802 (189.9) a,b
	Mix	1237 (311.5) d	572 (213.8)	1809 (108.0) bc
	Mulch	2169 (203.3) e	496 (233.0)	2665 (47.7) c
	Plough	418 (139.2)	556 (246.3)	974 (363.5) ab
Silt loam	**Mean**	661 (324.9) A	−37 (395.6) A	624 (443.8) A
	Control	430 (67.2) ab	69 (480.5)	499 (476.3) a
	Mix	693 (159.0) abcd	−88 (596.9)	604 (457.7) a
	Mulch	1129 (153.0) cd	−42 (201.0)	1087 (50.6) ab
	Plough	391 (91.7) ab	−85 (457.1)	305 (383.6) a
Sandy clay loam	**Mean**	616 (425.1) A	411 (609.9) AB	1027 (416.6) B
	Control	198 (78.8) a	570 (657.3)	768 (585.5) ab
	Mix	681 (336.9) abcd	244 (675.0)	925 (369.0) ab
	Mulch	1044 (270.3) bcd	140 (715.7)	1184 (480.1) ab
	Plough	540 (498.7) abc	690 (582.9)	1230 (173.8) ab
Mean	Control	340 (121.9) A	350 (469.6)	690 (414.8) A
	Mix	870 (366.9) B	243 (544.2)	1113 (617.5) A
	Mulch	1447 (573.5) C	198 (455.8)	1645 (803.2) B
	Plough	449 (271.8) A	387 (530.2)	836 (498.1) A
ANOVA *p* values	Soil type (S)	**<0.001**	**0.023**	**<0.001**
	Residue management (R)	**<0.001**	0.819	**<0.001**
	S × R	**0.002**	0.896	**0.004**

Values in brackets denote standard deviations ($n = 3$). Results of the two-way ANOVAs are given at the bottom of the table with *p* values <0.05 printed in bold

Multiple comparisons among group means (upper case letters) and all individual groups (lower case letters) were only performed in the case of a significant main factor effects and interaction effects, respectively. Values not sharing a common letter within a column were significantly different according to Tukey's HSD test

in the silt loam, and a weak to negligible effect in the sandy clay loam.

The EFR for NH_3 ranged between −0.1 and 1.2 % of cauliflower residue N over the total experimental duration. Two-way ANOVA yielded both significant main factor effects (both $p < 0.001$) and a significant interaction ($p = 0.003$). The multiple comparisons (Tukey's HSD) revealed that across soil types, mulch treatment (0.6 ± 0.47 %) exhibited higher EFR than the mix treatment (0.3 ± 0.34 %) and plough treatment (0.1 ± 0.26 %). Across residue placement treatments, loamy sand (0.7 ± 0.49 %) exhibited higher EFR than silt loam (0.1 ± 0.29 %) or sandy clay loam (0.2 ± 0.22 %), which were indistinguishable. The multiple comparisons among all groups demonstrated that the significant interaction effect actually reflected a lack of residue placement effects in silt loam and sandy clay loam, whereas in loamy sand the EFR for mulch treatment (1.2 ± 0.03 %) was significantly higher than EFRs for the plough (0.1 ± 0.24 %) and the mix (0.7 ± 0.07 %) treatments.

Discussion

Here, we studied the effects of different post-harvest cauliflower residue management treatments on CO_2, N_2O, and NH_3 emissions across three contrasting soil types. Whereas C mineralization from crop residues was unaffected by residue placement or soil type, the highest N_2O emissions arose after incorporation of crop residues by ploughing in all three tested soil types. Higher NH_3 emissions resulted from surface application of residues, but only in coarse-textured loamy sand.

Carbon dioxide emissions and mineralization of cauliflower residues

Most of the CO_2 emissions occurred within a timeframe of 6 weeks. This was true for all residue treatments although the incorporation treatments (mix, plough) showed higher initial mineralization rates than the surface mulch treatments. The C mineralization from added residues estimated by subtracting the control values (i.e. neglecting priming effects) was not affected by soil type or incorporation techniques in any of the tested periods. This is not surprising since evidence suggests that when soil temperature and moisture are kept favorable short-term C mineralization from added plant residues does not depend on major soil properties, e.g. soil texture (Whitmore and Groot 1997) or microbial community structures (Stark et al. 2008). Measured emissions in the control treatments suggest that basal respiration was in part determined by differences in SOC contents, which were lower in the loamy sand than the other two soils. Also, the sandy clay loam showed the highest CO_2 emissions in relation to SOC stocks in 0–30 cm, presumably linked to the destruction of soil aggregates by the tillage action at the start of the experiment, evidenced by the initial CO_2 emission peak.

Nitrous oxide emissions

Cumulative N_2O emissions generally increased in the order: loamy sand < silt loam < sandy clay loam. This is due to the soil texture mainly determining the average water-filled pore space and hence oxygen availability, promoting denitrification in fine-textured soils (Weier et al. 1993). The relationship between WFPS and the N_2O/N_2 ratio (Davidson et al. 2000) also suggests that an unaccounted proportion of N was lost in the form of N_2, which presumably also increased in the order: loamy sand < silt loam < sandy clay loam. The fact that emissions of N_2O in the control treatments of loamy sand and silt loam were very low demonstrates that in well-aerated soils with low aggregation, N_2O emissions are minor even when N availability is high. This means that none of the many potential processes that can produce N_2O (Butterbach-Bahl et al. 2013) were supported by these soils, due to unsuitable conditions regarding the availabilities of C, N, and O_2 (Wrage et al. 2001).

The input of high amounts of readily available organic C and N in the form of crop residues presumably caused an increase in aerobic respiration, which consumed oxygen, and therefore promoted the formation of anaerobic microsites (Giles et al. 2012). Moreover, at preexisting or newly established anaerobic zones, the supplied C substrates could then be used as an electron donor in denitrification. However, it cannot be ruled out that processes other than ordinary denitrification played a role. Especially autotrophic nitrification may become an important N_2O source at lower levels of WFPS (Bateman and Baggs 2005; Li et al. 2016). The formation of anaerobic microsites in loamy sand still appears likely, since Flessa and Beese (1995) demonstrated that sugar beet residues (C/N:29) both reduce the redox potential and increase N_2O emissions in a well-aerated soil, although in that study soil moisture was higher than in our study (WFPS 63 %). Presumably fresh and coarse pieces of crop residues form their own microenvironment that differs from the bulk soil in terms of water and nutrient content. Consequently, the bulk soil conditions only play a minor role in N_2O emissions after recent crop residue input, explaining why no clear effect of soil type is seen in the early phase. Hypothetically, with increasing crop residue particle size such temporary “autonomy” of crop residues increases. The water content (88 % in this study) and the integrity of the crop residues probably play a critical role in this “autonomy”. Hence, dried and ground/chopped crop residues, which are most often used in lab experiments (e.g. Ambus et al. 2001), will likely produce different results. Yet, although Loecke and Robertson (2009) used dried residues, they found that especially at lower levels of WFPS (50 %) N_2O emissions increased with residue patch size.

Our residue placement treatment results are not surprising since both the concentration of water and nutrients in a layer of buried crop residues after ploughing is high and the oxygen availability at these locations is low. Therefore, this incorporation technique vigorously supports the formation of anaerobic denitrification “hot spots”, as corroborated by results from Harrison et al. (2002) and Carter et al. (2014). In contrast, Baggs et al. (2000) observed higher N_2O emissions after rotary tillage of lettuce residues (C/N:8) compared to conventional or deep ploughing, which they explained as the longer the distance N_2O has to travel to the soil surface after deep incorporation the greater the opportunity for N_2O to become reduced to N_2. This demonstrates the difficulty in interpreting N_2O emissions without also knowing the (difficult to measure) N_2 emissions. Effects on N_2O emissions can be causes by altered rates of both N_2O production and consumption (reduction to N_2), the latter becoming more relevant with increased soil anaerobicity.

Another possible reason for the relatively high N_2O emissions in the mix treatment of loamy sand are the higher SMN contents in loamy sand compared to the other two soils. This had no effect on N_2O emissions in the control treatment, but it may have caused higher N_2O emissions after input of available C with crop residues. In the treatments receiving crop residues, the differences in SMN contents were in part caused by the differences in initial SMN contents after harvesting cauliflower, but mostly by the higher net N mineralization from crop residues in the loamy sand compared to the silt loam and sandy clay loam. One explanation may be a faster overall net mineralization of crop residues in the loamy sand, presumably due to enhanced aeration. However, it should be noted that the high initial temporal variability in the NO_3^{-1} contents of the loamy sand demonstrates that these values should be interpreted with caution.

Unlike this study, several groups reported N_2O emissions being higher after surface application of crop residues compared to incorporation by mixing (Baggs et al. 2003; Escobar et al. 2010). The authors attributed these differences to conservation of soil moisture and the level of O_2 consumption as well as C and N availabilities in the upper mineral soil favoring the formation of microsites with high denitrification activity. In our study, most of the N_2O emissions in the mulch treatments occurred after the incorporation of surface-applied residues, even in the silt loam and sandy clay loam, which indicate that high soil moisture under the mulch layer was not sufficient to support denitrification. In contrast, probably due to a much higher application rate (286 kg N ha^{-1}), which allowed the formation of a wet and consistent mulch layer, Nett et al. (2015) observed very high N_2O emissions after surface application of cauliflower leaf residues (C/N:12) to a well-aerated sandy soil before incorporation. Finally, all of the observed EFRs for N_2O over the total experimental duration (7.5 months) ranged within the confidence interval (0.3–3 %) of the default IPCC emission factor of 1 % per year (IPCC 2006).

Ammonia emissions

The passive filter traps used in ventilated closed chambers reflect a semi-quantitative measurement of ammonia emissions, which may differ from the real emissions under natural turbulence conditions.

Compared with other similar studies, e.g. Glasener and Palm (1995) and Larsson et al. (1998), the NH_3 emissions in our study can be regarded as low, both in terms of absolute cumulative emissions and EFRs, which did not exceed 2.7 kg N ha^{-1} and 1.2 % in 230 days. de Ruijter et al. (2010b) established a relationship between the EFR and residue C/N ratio as well as between EFR and residue N content. Applying the properties of the cauliflower residues in the present study (C/N = 17, N content = 22.9 g N kg^{-1} DM) to these equations yielded an EFR of 3.3 % (5 kg N ha^{-1}), which is higher than our observed values. The lower NH_3 losses observed in our study may have resulted from relatively rapid drying-out of the residues, at least superficially, which hampered ammonification (Whitehead et al. 1988). CO_2 emissions also appeared to be lower in the mulch than in mix and plough during the first week, indicating delayed decomposition. But then, SMN contents during the first weeks were lower in the mulch than in mix and plough only in the silt loam and sandy clay loam. This was not the case in the loamy sand, which exhibited the highest NH_3 emissions. Perhaps the initial release of plant sap after slashing the crop residues and subsequent leaching into the upper mineral soil protected much of the NH_4^+ from being volatilized as NH_3. The fact that NH_4^+ contents remained below 8 kg N ha^{-1} in the topsoil (0–30) at all times demonstrates that freely available NH_4^+ was not abundant in the soil solution. Since the extracting agent used to determine SMN (0.0125 M $CaCl_2$) is not suitable for quantitatively measuring adsorbed NH_4^+, further interpretation of NH_4^+ contents remains to be addressed in the future.

The consistently higher NH_3 emissions from the loamy sand compared to the silt loam and sandy clay loam were presumably caused by a lower cation exchange capacity in the loamy sand, which does not allow efficient abiotic NH_4^+ immobilization. In addition, the higher proportion of air-filled macropores in the loamy sand, evidenced by the low levels of WFPS, and a lower specific surface area may have facilitated the diffusive transport of NH_3 in the gas phase.

Our results corroborate the findings of other studies, in that soil incorporation generally reduces NH_3 emissions substantially compared to surface application (Janzen and McGinn 1991; Glasener and Palm 1995; de Ruijter et al. 2010a, b). As discussed above, these reductions are likely due to an increasing probability that NH_3 is dissolved in soil water, followed by nitrification to NO_3^-, adsorption on mineral surfaces, and immobilization by microorganisms. In addition, the rise in pH associated with decomposition of plant residues (Kimber 1973) shifts the NH_3/NH_4^+ equilibrium towards NH_3 and this effect should be strongest where plant residues are concentrated in a layer. This was also the case in the plough treatments, but here the distance to the soil surface was too great for substantial NH_3 losses.

The different effects of residue placement treatments with increasing coarseness of the soil texture (soil type × residue placement interaction) may be interpreted as follows: under the conditions of our study, in particular decimating crop residues by slashing, NH_3 emissions can be expected to be low in fine-textured soils, even when surface-applied. In contrast, on a coarse-textured soil with low NH_4^+ retention capacity and high diffusivity the technique of residue application has a decisive impact on NH_3 emissions, which are high after surface application and decrease with the deepness of incorporation.

Conclusions

Our results clearly show that both management of vegetable crop residues and soil type greatly influence N_2O and NH_3 emissions. In fact, the results confirm our hypothesis that soil texture conversely affects N_2O and NH_3 emissions. Under the given conditions, fine-textured soils tend to produce higher N_2O, but lower NH_3 emissions than coarse-textured soils. Moreover, the results suggest that incorporation of crop residues by ploughing should be avoided to reduce N_2O emissions in a range of soils. In contrast, the effects of residue management on NH_3 emissions may strongly depend on the soil characteristics. Our findings suggest that surface application of slashed cauliflower residues only produces higher NH_3 emissions than residue incorporation in coarse-textured soil. Future studies could improve our understanding of how soil type and residue management affect gaseous N emissions by investigating the "autonomy" of plant residues from bulk soil conditions, e.g. in terms of soil moisture, the processes producing N_2O emissions under dry conditions, and the spatial distribution of NH_3 and N_2O sources after residue application.

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