Performance and methodological evaluation of a quantum-cascade-laser photoacoustic aerodynamic gradient system for field-scale NH_3 flux measurements

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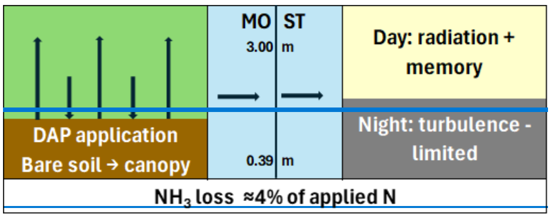
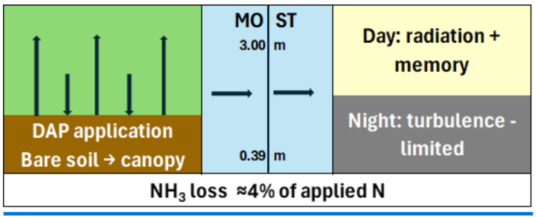
Abstract.
Reliable quantification of ammonia (NH_3) surface–atmosphere exchange remains challenging due to the compound's reactivity, inlet interactions, and the sensitivity of gradient-based flux estimates to instrumental response characteristics. We present the field evaluation and methodological assessment of a quantum-cascade-laser (QCL) based photoacoustic (PA) aerodynamic gradient system for half-hourly NH_3 flux measurements under agricultural conditions. The campaign covered a 54-day post-fertilization period under predominantly dry soil conditions, including the transition from bare soil to a developing winter rapeseed canopy. Instrumental performance was assessed through co-located inlet comparison experiments, yielding a random uncertainty of ±2 ppb (1σ) and demonstrating negligible systematic bias between sampling channels. The system operated continuously under field conditions with active thermal stabilization and humidity management. Fluxes were calculated using Monin-Obukhov similarity theory (MOST) across an ensemble of universal stability functions to evaluate methodological sensitivity. Sensitivity analysis indicated that variability attributable solely to stability-function selection remained small relative to observed diurnal flux amplitudes. ~~Mean~~The mean NH_3 ~~emission during~~loss over the investigated period was 1.85 nmol m^{-2} s^{-1}, corresponding to a cumulative loss of 1.21 kg N ha^{-1} or 4.0% of the applied fertilizer nitrogen. Based on instrumental uncertainty (±2 ppb) and literature-reported uncertainty ranges of MOST-based aerodynamic gradient flux calculations, the overall uncertainty is estimated to be approximately ±25%, corresponding to ±0.30 kg N ha^{-1}. A pronounced diurnal asymmetry was observed, with daytime emissions approximately one order of magnitude higher than nighttime values, reflecting strong coupling between turbulent exchange and radiation-driven surface processes. Complementary machine-learning analysis indicated that incorporating short-term temporal memory substantially improved the representation of NH_3 flux dynamics and revealed distinct daytime and nighttime exchange regimes. The combined QCL–photoacoustic gradient system demonstrated robust field performance and low instrumental bias, supporting its applicability for long-term field-scale studies of ~~reactive trace gas~~agricultural NH_3 exchange under post-fertilization conditions.





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1. Introduction

The accumulation of reactive nitrogen (N_r) in the Earth system is a major environmental consequence of modern agriculture. Since the industrial fixation of atmospheric dinitrogen, large amounts of N_r have been introduced into ecosystems, where a single nitrogen atom may contribute sequentially to multiple impacts – air pollution, eutrophication, acidification, biodiversity loss and climate forcing – described as the nitrogen cascade (Galloway et al., 2003).

Ammonia (NH_3) is a central component of this cascade. It is the dominant alkaline gas in the atmosphere and a key precursor of secondary inorganic aerosol formation, thereby contributing to $PM_{2.5}$ burdens and associated health impacts (Seinfeld and Pandis, 2016). Deposition of NH_3/NH_4^+ also perturbs nutrient balances in sensitive ecosystems and can reduce plant species richness under elevated nitrogen inputs (Bobbink et al., 2010). Volatilized NH_3 may also contribute indirectly to climate change as part of the nitrogen cascade, since redeposited reactive nitrogen can subsequently undergo nitrification and denitrification, leading to N_2O formation.

Agriculture accounts for the majority of global NH_3 emissions, with mineral fertilizer use being a major driver. Fertilizer-related NH_3 emissions have increased strongly since the 1960s and may further rise under future food demand and changing climate conditions (Sutton et al., 2013; Xu et al., 2019). From an agronomic perspective, NH_3 volatilization represents a direct loss of applied nitrogen and contributes to low nitrogen use efficiency at the global scale (e.g. Coskun et al., 2017; Bindraban et al., 2020).

Robust quantification of fertilizer-derived NH_3 losses is therefore essential both for environmental assessments and for evaluating mitigation options. However, NH_3 exchange measurements remain challenging because NH_3 is reactive and "sticky", shows strong interactions with wet surfaces, and commonly exhibits bidirectional exchange, with the surface acting as either a sink or a source depending on ambient concentrations and compensation point dynamics (Flechard et al., 2013). This complexity is amplified after fertilization, when soil emission potentials are high but canopy development increasingly promotes within-canopy uptake and recapture processes.

Micrometeorological flux methods (e.g., aerodynamic gradient, relaxed eddy accumulation, eddy covariance) are suitable for field-scale quantification, but they require accurate, fast-response concentration measurements. The performance of NH_3 instrumentation under field conditions has therefore been a long-standing focus, including large intercomparison efforts (von Bobrutzki et al., 2010). Besides chemical ionization mass spectrometry approaches enabling fast EC-type measurements (Sintermann et al., 2011), photoacoustic (PA) techniques have also been used in NH_3 monitoring and gradient applications (Pogány et al., 2010; Nelson et al., 2019; Kamp et al., 2020).

Photoacoustic approaches are attractive for long-term field deployment because of their sensitivity and continuous operation, but they are prone to practical issues relevant for gradient flux work: (i) humidity-related interferences, (ii) adsorption/desorption in sampling lines, and (iii) inlet switching and time-lag effects that can distort gradients if not handled carefully. Response-time and delay characteristics of PA multi-gas monitors have been explicitly investigated (Rom and Zhang, 2010), and diode-laser-based photoacoustic instruments have been developed and evaluated specifically for surface–atmosphere NH_3 flux applications (Pogány et al., 2010).

The primary objective of this study was to evaluate the field performance and methodological robustness of a ~~QCL~~ Quantum Cascade Laser (QCL) based photoacoustic aerodynamic gradient system for NH_3 flux measurements under agricultural conditions. NH_3 losses following mineral fertilizer application were quantified using aerodynamic gradient flux measurements combined with photoacoustic NH_3 mixing ratio observations. By covering the transition from bare soil shortly after fertilization to a developing crop canopy later in the season, the study targets a critical period when emission strengths and recapture processes change rapidly. Particular emphasis was placed on instrumental performance, methodological aspects of gradient-based flux calculations, and the robustness of Monin-Obukhov similarity-based flux estimates under field conditions. In addition, machine-learning analysis was applied as a complementary interpretative tool to investigate temporal dynamics and regime-dependent controls of the measured NH_3 fluxes.

2. Materials and Methods

2.1. Study Site and Experimental Conditions

Field measurements were carried out on an arable plot located near Kartal, Hungary, at the Gödöllő Experimental Farm. The observation period covered the late summer to early autumn growing season, extending from 28 August to 21 October 2025. The geographic coordinates of the site are 47° 39′ 42.56″ N, 19° 31′ 43.21″ E, with an elevation of approximately 153 m above sea level.

The region is characterized by a continental climate, with a long-term (1991–2020) mean annual precipitation of 550–600 mm. During the measurement campaign, cumulative precipitation amounted to 65 mm, resulting in predominantly dry soil conditions.

The soil at the site is classified as a Haplic Phaeozem (Humic, Pantoloamic) according to the World Reference Base for Soil Resources. Particle size analysis indicated a composition of 32.3% clay, 49.1% silt, and 18.6% sand, corresponding to a silty clay loam texture under the USDA classification scheme.

Field management prior to and during the experiment included mineral fertilization and crop establishment. On 25 August, the field was fertilized with 200 kg ha^{-1} of granular diammonium phosphate, equivalent to 30 kg N ha^{-1} and 80 kg P ha^{-1}. Winter rapeseed (*Brassica napus* L.) was sown two days later, on 27 August. As a result, the measurement period encompassed a transition from bare soil conditions immediately after fertilization to a developing crop canopy by mid-October.

2.2. Measurement Techniques and Instrumentation

2.2.1. Ammonia mixing ratio measurements

Ammonia mixing ratios were determined using a photoacoustic spectroscopy analyzer developed at the University of Szeged. One of the main components of the instrument is the data acquisition and control unit, which is a highly integrated embedded system developed by Videoton Holding Zrt. specifically for photoacoustic measurement systems.

The photoacoustic cell, thermostated at 50 °C, separates the internal volume of the instrument from the sample gas flow. Acoustic signal generation takes place in one of the two resonators machined into the cell. This configuration is referred to as a differential photoacoustic cell (Miklós et al., 2001).

Acoustic signal detection is performed using two MEMS microphones (SisonicTM SPU0410HR5H-PB, Knowles). The microphones are connected to a differential amplifier, which performs signal preamplification, filtering, and analogue-to-digital conversion using a 16-bit ADC. Lock-in detection is implemented digitally; the Fourier transformation is carried out by an FPGA. The sampling rate is 50 kHz, resulting in a frequency resolution of approximately 12 Hz.

A quantum cascade laser (QCL) operating at a wavelength of 10.39 µm was used. This wavelength is suitable for ambient ammonia measurements, as ammonia is one of the dominant atmospheric absorbers at this wavelength under typical ambient conditions, minimizing spectral interference effects. In photoacoustic measurements, VOCs may cause a positive bias in the 10.39 µm (≈ 962 cm^{-1}) region due to spectral overlap (Liu et al., 2020); however, in our case, the use of a narrow-linewidth, small-bandwidth QCL allows targeting a specific, isolated absorption line, thereby effectively eliminating this interference. The optical output power of the QCL ~~laser~~ (model: QD10500CM1, Thorlabs) is approximately 50 mW. A mirror was placed at the end of the photoacoustic cell to double the effective optical path length and thereby enhance signal generation. Laser driving and temperature control were provided by a Thorlabs ITC4002QCL unit, while the modulation signal was generated by the control electronics of the instrument.

Air was sampled at two vertical levels, positioned at 0.39 m and 3.00 m above the soil surface, to enable flux estimation using the aerodynamic gradient method. Each measurement point consisted of 1 minute of flushing, followed by 2 minutes of data acquisition, repeated for each sampling level. Level selection was achieved using two solenoid valves (Asco SCH284B015). The sampling system was primarily constructed from PTFE tubing with an outer diameter of 6 mm and an inner diameter of 4 mm. The length of each PTFE sampling line was 4 m. Given the inner diameter of 4 mm and the volumetric flow rate of 800 cm^3 min^{-1}, the internal volume of one sampling line was approximately 50 cm^3, corresponding to a residence time of about 3.8 s.

The photoacoustic cell itself is made of stainless steel and was coated with a sodium hydroxide (1.5 M) solution to reduce ammonia adsorption. The volumetric flow rate was set to 800 cm^3 min^{-1}. A flow meter, a solenoid valve, and a needle valve were installed between the pump and the cell to ensure a stable and constant flow rate.

The sensitivity of the instrument was 17.9 µV ppb^{-1}, and the minimum detectable mixing ratio (DL) was approximately 3 ppb with 2 minutes of signal averaging, determined from the standard deviation (SD) and the slope of the calibration line as DL = 3 × SD / b (Shrivastava and Gupta, 2011). Approximately 97.5% of the measured values fell within the range 3–87 ppb, with a mean value of 20 ppb.

The complete system was installed in a standard electrical enclosure with additional thermal insulation. The enclosure has an IP66 rating, allowing operation under various outdoor weather conditions. The internal temperature of the enclosure was actively controlled and set to 42 °C. During periods of high ambient temperature, a protective tent was installed above the instrument to shield it from direct solar radiation and prevent overheating.

Prior to the field deployment, both inlet lines were temporarily placed at the same height to assess instrumental and sampling-related measurement uncertainty. During this co-located inlet comparison, both sampling channels were positioned at 300 cm above the soil surface, so that no systematic vertical concentration gradient was expected. The comparison therefore reflects the combined uncertainty associated with the two sampling lines, valve switching, and instrumental response. Results are detailed in Section 3.1.

For the flux calculations, mixing ratios (ppb) were converted to concentrations (nmol m^{-3}) using the molar volume at the given temperature and pressure.

To minimize gradient attenuation due to inlet adsorption–desorption effects, identical tubing materials and lengths were applied for both sampling heights, and inlet switching was preceded by a dedicated flushing period. Under steady-state conditions, the effective equilibration time of the system was shorter than the 2 min averaging interval, thereby limiting gradient distortion during half-hourly flux calculations.

2.2.2. Auxiliary meteorological and surface measurements

A comprehensive set of ancillary measurements was conducted to support flux calculations and interpretation. Three-dimensional wind speed and turbulence parameters were measured at 10 Hz using a METEK USA-1 ultrasonic anemometer. Soil physical conditions were monitored with a Campbell Scientific CS650 reflectometer, providing soil temperature and volumetric water content, from which water-filled pore space (WFPS) was derived. Air temperature and relative humidity were measured with a CS215 probe; precipitation was recorded using an ARG100 tipping bucket rain gauge; photosynthetically active radiation (*PAR*) was measured with an SKP215 quantum sensor; and global radiation (Ig) was measured with a CM3 pyranometer. Crop development was characterized by repeated canopy transmittance measurements at seven permanent field locations using an AccuPAR LP-80 ceptometer, from which leaf area index (LAI) was derived. Leaf surface wetness was monitored with a PHYTOS-31 sensor. All instruments were connected to and synchronized by a Campbell Scientific CR3000 data logger, ensuring consistent time alignment across measurement systems.

High-frequency wind and scalar data from the sonic anemometer and a LiCor 7500 open-path infrared gas analyzer were processed using EddyPro software (version 7.0.9). Wind velocity, wind direction, friction velocity (u^*), temperature scale parameter (T^*), momentum flux, sensible and latent heat fluxes, and the Monin–Obukhov length were calculated from the 10 Hz measurements.

2.3. Flux Calculations

The turbulent ammonia flux was calculated using the aerodynamic gradient method with a 30-min averaging time interval at two measurement heights (z_1 = 0.39 m, z_2 = 3.00 m). During the calculations, the displacement height (d) was assumed to be 0.05 m. The friction velocity (u*) and the Monin-Obukhov length (L) were measured and/or derived using an eddy-covariance measurement system. Flux calculations were based on Monin-Obukhov similarity theory (MOST) using a range of universal similarity functions reported in the literature; additionally, turbulent fluxes were also determined using the simplest logarithmic profile approximation.

A widely used approach for flux calculations based on trace gas gradient measurements is the application of universal stability functions that depend on atmospheric stability. Since NH_3 can generally be treated as a passive scalar in the surface layer, universal similarity functions developed for sensible heat transfer are commonly applied in aerodynamic gradient studies. This methodology has been used, among others, by Phillips et al. (2004); Spirig et al. (2010); Toyota et al. (2016); Kułek and Weidinger (2023); Zahn et al. (2023); and Abdulwahab et al. (2025), and is also described in methodological overviews of modern micrometeorological measurement techniques (Trebs et al., 2021).

In the case of stable stratification, 30 universal functions were applied together with logarithmic profiles characterized by two different Prandtl numbers (Tables A1 and A3), whereas under unstable stratification 22 universal functions were used together with two logarithmic profile formulations (Tables A2 and A3). These were selected based on the synthesis studies of Högström (1988), Weidinger et al. (2000), Prueger and Kustas (2005), Kramm and Herbert (2009), Foken (2017), and a case study by Kułek and Weidinger (2023), as well as studies addressing turbulent exchange processes of trace gases. In addition to the classical logarithmic profile assumption (Prandtl number Pr = 1), several universal functions widely used in the literature were applied, such as those proposed by Businger et al. (1971) and Dyer (1974), later reformulated by Högström (1988).

The concentration gradient (Eq. 1) was expressed based on Monin-Obukhov similarity theory, adopting the commonly used assumption that the universal function describing trace gas transport has the same functional form as that for sensible heat transport $\varphi_c(\zeta) \equiv \varphi_H(\zeta)$ (Mayer et al., 2011; Zahn et al., 2023; Abdulwahab et al., 2025). The profile equation for concentration gradient is

$$\frac{\partial \bar{c}}{\partial \ln{(z-d)}} \cong \frac{\Delta \bar{c}}{\Delta \ln(z-d)} = \frac{\bar{c}_2 - \bar{c}_1}{\ln{(z_2-d)} - \ln(z_1-d)} = \frac{c_*}{\kappa}\varphi_c(\zeta_k), \tag{1}$$

where c_* is the dynamical concentration scale for the NH_3 flux calculation, $\bar{c}_1$ and $\bar{c}_2$ are the 30-min mean concentrations.

Assuming a near-logarithmic profile, the dimensionless height was defined as $\zeta_\mathrm{k} = z_\mathrm{k}/L$, where z_k denotes the geometric mean height of the two measurement levels:

$$z_\mathrm{k} = \sqrt{(z_1 - d)(z_2 - d)}. \tag{2}$$

Although the integral form of Eq. (1) could also be applied (Weidinger et al., 2000; Foken, 2017; Foken and Mauder, 2024), for two closely spaced measurement levels the direct calculation using Eq. (1) provides sufficiently accurate results ~~(Arya, 1991).~~. Comparative calculations were performed using the two most commonly applied universal functions

(Businger et al., 1971; Dyer, 1974), revealing negligible differences between the two approaches.

Universal functions have generally been validated within the interval $-1 \le \zeta \le +1$. In extremely stable ($\zeta > 1$) or extremely unstable ($\zeta < -1$) conditions – representing approximately 2% of the cases – the critical dimensionless heights ($\zeta_{krit} = 1$, if $\zeta > 1$, $\zeta_{krit} = -1$, if $\zeta < -1$) was applied in the calculations, while retaining the original stability classification.

The i-th type of dynamical concentration scale, c_{*i} was calculated based on the corresponding universal function, $\varphi_{ci}(\zeta_k)$ and von Karman-constant, κ_i from equation (1):

$$c_{*i} \cong \frac{\kappa_i}{\varphi_{ci}(\zeta_k)} \frac{\bar{c}_2 - \bar{c}_1}{\ln(z_2 - d) - \ln(z_1 - d)}. \quad (3)$$

The 30-min ammonia flux was obtained by averaging the individual flux estimates:

$$F_{NH_3} = \frac{1}{N} \sum_{i=1}^{N} (-u_* c_{*i}), \quad (4)$$

where N = 30 for stable or neutral stratification ($\zeta_k \ge 0$) and N = 22 for unstable ($\zeta_k < 0$) stratification, including the two logarithmic profile approaches in both cases (Appendix, Tables A1–A3).

To evaluate methodological robustness, fluxes were calculated using a comprehensive ensemble of universal similarity functions under both stable and unstable stratification. Sensitivity analysis showed that the variability attributable solely to stability-function selection remained small relative to observed diurnal flux amplitudes, indicating that the dominant uncertainty did not arise from similarity-function choice.

2.4. Machine Learning Analysis

Because ammonia flux time series inherently exhibit strong autocorrelation and non-stationarity, strict chronological validation was implemented to prevent temporal leakage. This design ensures that predictive performance reflects genuine forward extrapolation in time rather than artefacts arising from random data partitioning.

Half-hourly NH_3 fluxes (nmol m^{-2} s^{-1}) were analyzed over a multi-day post-fertilization period (n = 2590 observations). The dataset included a comprehensive set of candidate predictors encompassing structural variables (leaf area index), soil physical properties (volumetric water content, water-filled pore space, soil temperature), meteorological drivers (air temperature, relative humidity, precipitation, global radiation, photosynthetically active radiation), turbulent exchange parameters (friction velocity, momentum flux, sensible and latent heat fluxes, wind speed and direction, Monin-Obukhov length), chemical variables (NH_3 concentration at 3 m), management-related variables (days after fertilization), and derived aerodynamic and boundary layer resistances. All predictors were initially included in the model without subjective pre-selection.

The flux time series exhibited three dominant structural characteristics: a pronounced diurnal cycle with nighttime minima and daytime maxima, a progressive decay following fertilizer application, and strong short-term autocorrelation at sub-daily scale. Because these properties violate assumptions of independence and stationarity, a time-aware modelling framework was adopted.

To prevent temporal leakage, model validation was conducted using a strictly chronological split. The first 80% of days were used for training and the remaining 20% were reserved as an independent test set representing forward prediction in time. No random cross-validation was applied.

Model performance was evaluated exclusively on the final 20% of days, chronologically withheld from model training to represent forward prediction in time, using the coefficient of determination (R^2), root mean square error (RMSE) and mean absolute error (MAE).

To account for short-term dynamical persistence, autoregressive lag features were introduced to represent ~~intrinsic system memory.~~ short-term temporal persistence in the NH_3 flux time series.

Specifically, NH_3 flux values from the previous six half-hour intervals (NH_{3t-1} to NH_{3t-6}), corresponding to a three-hour memory window, were included as predictors. Lagged values crossing day boundaries were masked to avoid artificial carryover effects.

~~An Extremely Randomized Trees (ExtraTrees)~~The term "memory" is used here to describe the apparent temporal persistence of the NH_3 flux time series, rather than instrumental carry-over. Since the residence time in the sampling line was only about 3.8 s and each inlet switch was followed by a 1 min flushing period, instrumental memory cannot explain the observed persistence over several half-hourly intervals. The lag dependence is therefore interpreted as a property of the soil–surface–canopy exchange system, reflecting the gradual evolution of the surface NH_4^+/NH_3 reservoir, soil temperature and moisture conditions, adsorption–desorption processes, and persistent turbulence regimes.

A tree-based ensemble regression model was employed ~~due to its ability~~ to capture nonlinear relationships~~, resolve~~ and high-order interactions~~, and remain robust in the presence of correlated~~ among meteorological, surface, turbulence-related, and autoregressive predictors without assuming linearity. Missing values were handled using median imputation fitted on the training dataset and applied consistently to the test data.

To interpret model structure and underlying mechanisms, three complementary approaches were applied: impurity-based and permutation feature importance to identify dominant predictors, Partial Dependence (PDP) and Individual Conditional Expectation (ICE) curves to visualize nonlinear responses, and Friedman's H-statistic to quantify the strength of pairwise nonlinear interactions. Residual autocorrelation within days was additionally evaluated using autocorrelation functions (ACF) to assess remaining temporal structure after modelling. Finally, regime-specific analyses were performed separately for daytime ($PAR > 0$) and nighttime ($PAR = 0$) conditions to identify potential shifts in dominant environmental controls.

3. Results and discussion

3.1. Uncertainty Estimation and Measurement Reliability

To assess the measurement reliability and random uncertainty of the system, samples were collected at a single inlet height, applying two-minute temporal averaging. This averaging strategy suppresses short-term fluctuations and ensures that the comparison predominantly reflects instrumental and sampling uncertainties, rather than environmental variability.

During the comparison period, both sampling channels operated at an identical height of 300 cm. Under these conditions, the agreement between the two datasets was excellent. The Pearson correlation coefficient (r = 0.97) indicates a strong linear relationship. Linear regression yielded a = 0.97 × b + 0.60, demonstrating an almost one-to-one correspondence with only a minor offset~~.~~ (Fig. 1).

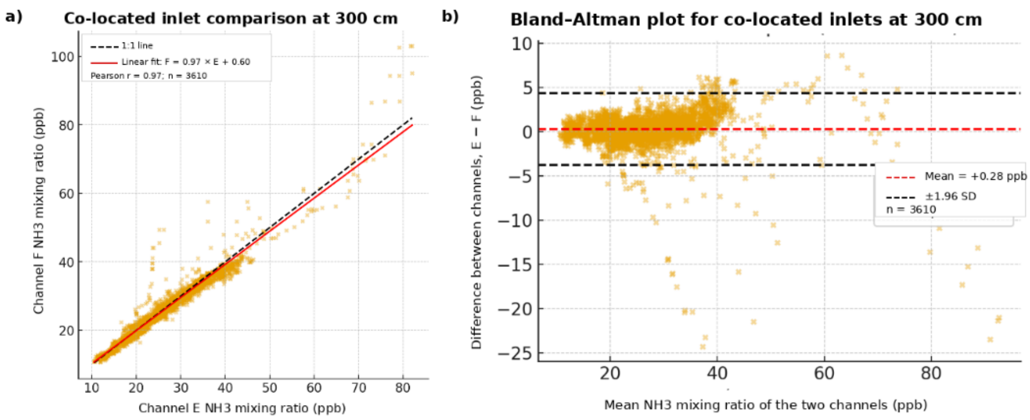


Figure 1. Co-located inlet comparison of the two NH_3 sampling channels at 300 cm above the soil surface. (a) Scatter plot with the 1:1 line and linear regression fit. (b) Bland–Altman plot of the channel differences. The mean difference ~~between the two channels~~ was +0.28 ppb~~,~~ and the standard deviation of the differences was 2.06 ppb~~.~~ (r = 0.97, n = 3610).

The ~~absence of a significant systematic bias suggests~~small mean offset of +0.28 ppb was negligible relative to the standard deviation of the differences (2.06 ppb), suggesting that the observed deviations ~~are~~were dominated by instrumental noise and sampling uncertainty.

Based on these results, the typical random measurement uncertainty between the two channels under identical sampling conditions is estimated to be ±2 ppb (1σ). Accordingly, the upper bound of the random error at the 95% confidence level (±1.96 σ) is approximately ±4 ppb.

Because inlet-related adsorption/desorption may represent a potential source of bias in NH_3 measurements, an additional in-situ inlet-length test was performed to evaluate its effect under the operating conditions of this study. The test showed that the dynamic inlet-induced bias was negligible compared with the measurement uncertainty; the test design and results are presented in Appendix B.

3.2. Diurnal Variation of Ammonia Flux and Turbulent Exchange

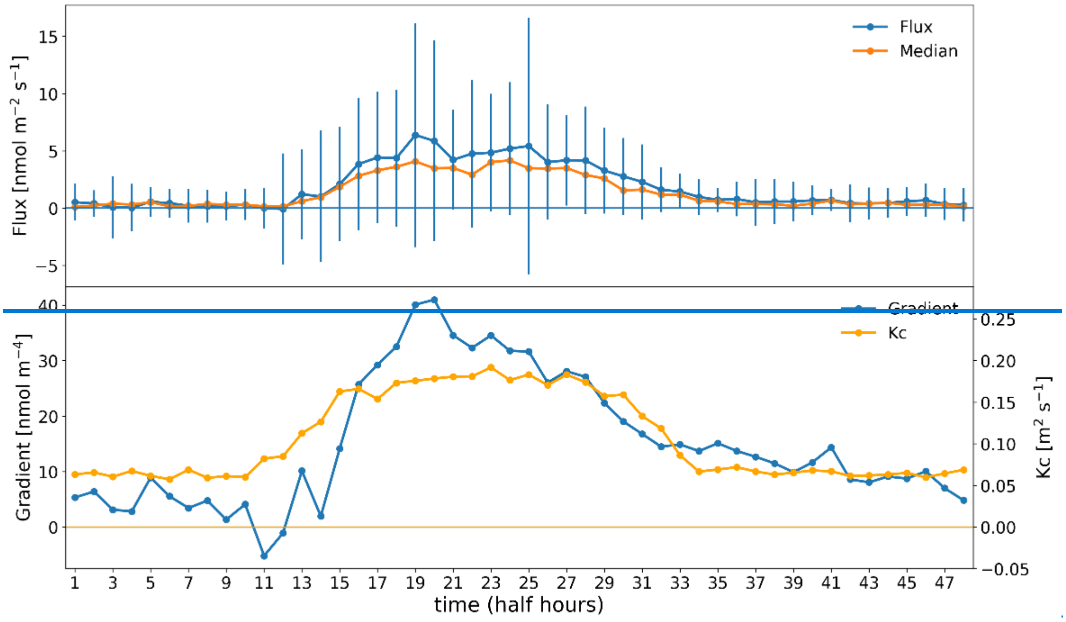
Ammonia fluxes during nighttime were relatively stable, ranging between −0.08 and 0.72 nmol m^{-2} s^{-1}. In contrast, the daytime period exhibited a pronounced diurnal pattern characterized by a distinct maximum in the late morning hours. This behavior can be attributed primarily to the diurnal dynamics of turbulent exchange.

Ammonia fluxes were calculated using Monin-Obukhov similarity theory (MOST) based on vertical concentration gradients and micrometeorological parameters. For diagnostic purposes, an effective turbulent transfer coefficient (K_c) was derived such that

$$F = K_c\left(\frac{\Delta C}{\Delta z}\right), \tag{5}$$

where K_c represents the stability-corrected exchange coefficient obtained from the MOST framework rather than an independently fitted parameter.

The turbulent diffusion coefficient (K_c), calculated from two-minute averaged flux and concentration gradient data, demonstrated a clear diurnal course. During the nighttime period (4:00 p.m.-03:00 a.m., UTC, zonal time is UTC+2), K_c averaged 0.06 $m^2\ s^{-1}$, indicating suppressed turbulent mixing due to limited surface energy input. In contrast, daytime values increased substantially, reaching approximately 0.17 $m^2\ s^{-1}$ (Fig. ~~1~~2), reflecting enhanced convective turbulence under higher radiation and thermal forcing.



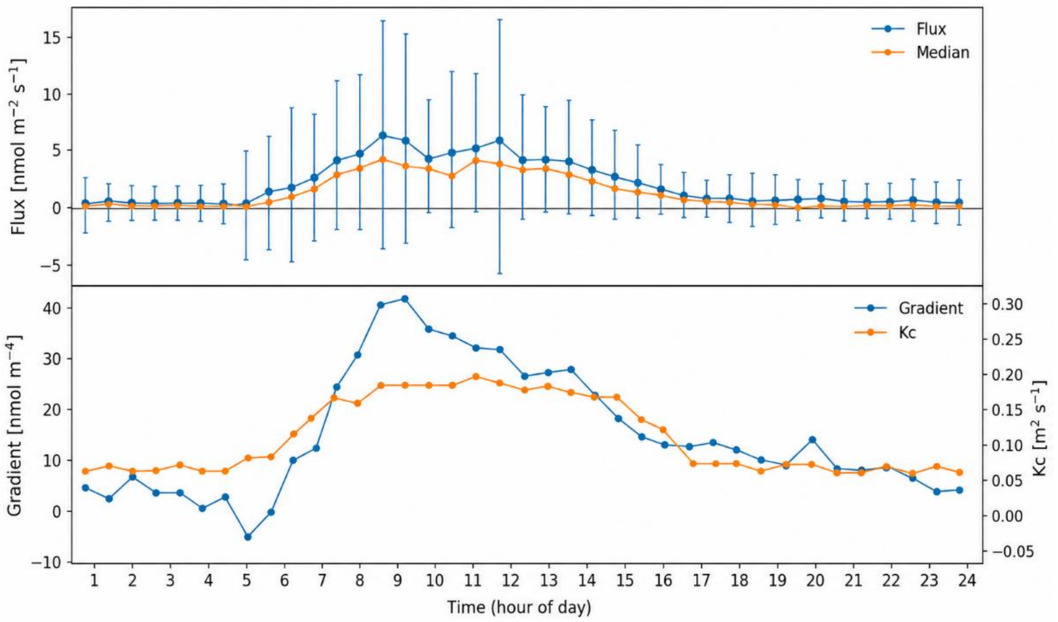


Figure ~~1~~2. Upper panel: Diurnal cycle of NH_3 flux. Vertical bars indicate ±1 standard deviation (SD) of half-hourly flux values. Lower panel: half-hourly gradients and diffusion coefficients. The horizontal axis represents half-hourly time steps (1–48). Time is given in UTC; local ~~standard~~civil time/CEST at the measurement site is UTC+2.

The concentration gradient exhibited a similar daytime maximum, and together with K_c, governed the diurnal evolution of ammonia flux. Fluxes peaked at approximately 09:00 (11:00 local time), followed by a local minimum around midday. This midday reduction may be consistent with commonly observed stomatal regulation phenomena.

Ammonia emission via plant stomata appears to be partially constrained during midday hours. Elevated temperature and high vapor pressure deficit increase transpiration demand, leading to transient plant water stress. In response, plants regulate stomatal aperture through abscisic acid (ABA)-mediated signaling, resulting in partial stomatal closure even under high irradiance conditions. Midday stomatal closure is primarily driven by high vapor pressure deficit and plant water stress (Oren et al., 1999), and stomatal regulation functions as an active, ABA-mediated control of photosynthesis (Flexas and Medrano, 2002).

Despite reduced turbulent exchange during nighttime, the concentration gradient remained small. This suggests that nocturnal ammonia emissions were limited, resulting in near-equilibrium conditions between the surface and the overlying air layer. Under such conditions, reduced mixing does not necessarily lead to concentration build-up, as the source strength is insufficient to generate a pronounced vertical gradient (Sutton et al., 1995; Nemitz et al., 2000).

In our study, as shown in Figure ~~1~~2, emission dominates during the daytime hours, with a maximum around noon, whereas during the nighttime it shows relatively constant, low values and occasionally shifts to deposition. When separated based on global radiation and photosynthetically active radiation, the mean emission was 3.11 nmol m^{-2} s^{-1} under $I_g > 0$ and $PAR > 0$ conditions, while during nighttime hours ($I_g = PAR = 0$) it was an order of magnitude lower, 0.448 nmol m^{-2} s^{-1}.

The daytime dominance of ammonia flux has long been recognized. For example, Harper et al. (1983) reported that the daily NH_3 cycle was pronounced, with substantial emission during the day and small emissions or even deposition at night. During the summer season, soil surface temperature was the factor most strongly correlated with NH_3 flux. During the remainder of the year, evapotranspiration showed the highest correlation, although the increased midday flux was likely attributable to the parallel increase in soil temperature and wind speed. All of the main influencing factors are interrelated through their dependence on solar radiation.

In a review study, Hargrove (1988) listed several factors determining ammonia flux. Environmental variables such as temperature, soil moisture content, and air exchange at the soil surface are primary determinants of the magnitude of NH_3 loss. Soil moisture and surface air exchange are particularly important. Under field conditions, these factors fluctuate widely on a daily basis due to the combined effects of dew formation and evaporation. Studies cited in Hargrove (1988) indicate that, on a daily scale, high NH_3 loss rates are generally associated with periods of rapid soil drying, when moist (near field capacity) surface soil is followed by several days with little or no rainfall (<0.5 inches). It is also evident that NH_3 loss is related to wind speed under field conditions, and that low air exchange rates at the soil surface—such as those expected within a plant canopy or in no-till production systems—may limit NH_3 loss.

Recent research has reinforced that ammonia emissions are strongly controlled by interactions among soil properties, meteorological conditions, and management practices. Advanced atmospheric emission models show that incorporating spatial and temporal variability in agricultural activities, such as manure and fertilizer application timing, as well as explicit representation of soil moisture and temperature, significantly improves simulation of observed NH_3 emission dynamics across Europe and beyond (Ge et al., 2020). Field and satellite-based studies further highlight that environmental drivers and atmospheric lifetime of NH_3 vary considerably with local conditions, affecting both emission rates and dispersion patterns (Xie et al., 2024). Reviews also emphasize the profound contribution of agriculture to global ammonia emissions and their impacts on particulate formation and human health, underscoring the need to account for complex environmental dependencies when estimating NH_3 fluxes (Wyer, 2022).

It is therefore clear that identifying the factors controlling emission is not straightforward, as their relative importance varies depending on the specific environmental conditions.

3.3. Temporal Variation of Ammonia Emission

During the investigated period (4.662×10^6 s), the mean ammonia emission was 1.85 nmol $m^{-2}\ s^{-1}$, corresponding to ~~1.21 kg ha^{-1}~~a cumulative loss of 1.21 kg N ha^{-1}. Based on instrumental uncertainty (±2 ppb) and literature-reported uncertainty ranges of MOST-based aerodynamic gradient flux calculations, the overall uncertainty is estimated to be approximately ±25%, corresponding to ±0.30 kg N ha^{-1}. when expressed as nitrogen, representing 4.0% of the applied fertilizer nitrogen.

Emission dynamics were strongly time-dependent (Fig. ~~2~~3). Immediately after fertilization, elevated fluxes were observed under bare soil conditions, reflecting high surface emission potential. In the following days, fluxes declined progressively, consistent with depletion of readily volatilizable ammonium and changes in soil-surface equilibrium conditions.

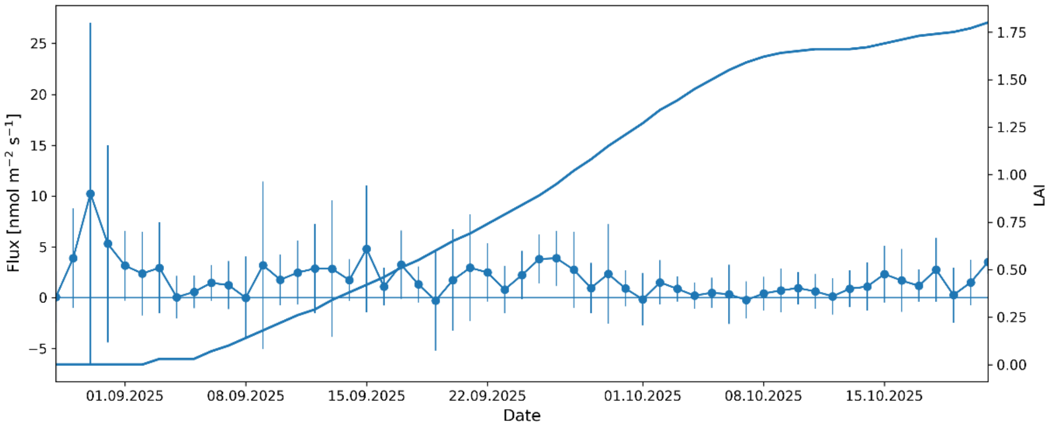


Figure ~~2~~3. Temporal variation of NH_3 flux and increase of leaf area index (LAI). Vertical bars indicate ±1 standard deviation (SD) of half-hourly fluxes for each day.

Superimposed on this seasonal-scale decay, pronounced short-term variability was observed, driven by diurnal changes in radiation and turbulent exchange as discussed in Section 3.2. The overall fertilizer-related nitrogen loss (4%) falls within the lower range reported in the literature for similar mineral fertilizer applications under relatively dry soil conditions.

Synthetic nitrogen fertilizers are applied in various forms, including urea, ammonium nitrate, ammonium sulphate, calcium ammonium nitrate, UAN, and phosphate-based ammonium fertilizers (Amhamed et al., 2022). Following application, a portion of ammonium is taken up by plants or transformed through nitrification, while a significant fraction may be lost via gaseous emissions – particularly as NH_3, which typically peaks shortly after fertilization and can account for 6-20% of applied nitrogen depending on fertilizer type, soil properties, climate, and management practices (Génermont and Cellier, 1997; Sommer et al., 2004; Zhan et al., 2020; Ma et al., 2021). Emission responses are strongly nonlinear and often increase exponentially with application rate (Jiang et al., 2017). Huang et al. (2017) reported that under mineral fertilizer treatments (NK/NP/NPK), total NH_3 volatilization losses were ≤4.2% of the applied nitrogen, whereas substantially higher losses were observed under combined organic and mineral fertilization.

In the present study, di-ammonium phosphate (DAP) was applied at a moderate rate and compared to urea-based fertilizers its volatilization potential is generally lower; consequently, the expected NH_3 loss response is less pronounced even under nonlinear dose–response relationships.

Regarding the operational range of the measurement system, the upper measurable NH_3 loss is not primarily limited by the fraction of fertilizer nitrogen lost through volatilization. Substantially higher volatilization losses than those observed in the present study could in principle be quantified, provided that the NH_3 mixing ratios remain within the calibrated response range of the analyzer or are measured using appropriate dilution/calibration. Thus, the upper limit is determined by the instrumental linear range and sampling configuration rather than by the percentage loss of applied nitrogen itself.

The lower quantification limit is more difficult to define, because it depends on ambient NH_3 mixing ratios, the magnitude of the vertical concentration gradient, atmospheric turbulence, averaging time, and the duration of the integration period. For aerodynamic gradient measurements, the critical factor is the detectability of the vertical NH_3 concentration gradient rather than the absolute concentration alone. Based on the inter-channel uncertainty of approximately ±2 ppb and the estimated overall uncertainty of about ±0.30 kg N ha^{-1} for the cumulative loss, seasonal NH_3 losses below approximately 0.3–0.5 kg N ha^{-1} would become increasingly difficult to distinguish reliably from measurement uncertainty.

3.4. Interpretable Machine Learning Analysis of NH_3 Flux Dynamics

To further interpret the temporal variability of the measured NH_3 fluxes, an exploratory machine-learning analysis was performed using the observational dataset. The half-hourly NH_3 flux time series exhibited a pronounced diurnal cycle, a progressive decay trend, and strong short-term autocorrelation. Therefore, model validation was conducted using a strictly chronological (day-aware) split, with the first 80% of days used for training and the remaining 20% reserved for forward evaluation.

Models based solely on contemporaneous meteorological predictors showed negligible predictive skill (R^2_test ≈ 0.01), indicating that instantaneous environmental drivers alone were insufficient to explain cross-day variability. Residual diagnostics revealed strong within-day persistence (lag –1 autocorrelation ≈ 0.33), motivating the explicit inclusion of autoregressive lag features up to six half-hour steps (3 hours). Incorporating this short-term memory structure markedly improved predictive performance (R^2_test ≈ 0.31) and reduced residual autocorrelation to near white-noise levels (lag-1 ACF ≈ 0.05). For the full dataset, inclusion of lag features increased predictive skill to R2_test ≈ 0.31. Subsequent regime-specific modelling further revealed distinct daytime and nighttime control structures.

In addition to the lagged-model analysis, the agreement between measured and model-predicted NH_3 fluxes was evaluated directly using the test subset. The predicted-versus-measured comparison showed a clear correspondence between the two datasets (Pearson r = 0.795, corresponding to $r^2 = 0.6322$, n = 575) (Fig. 4).

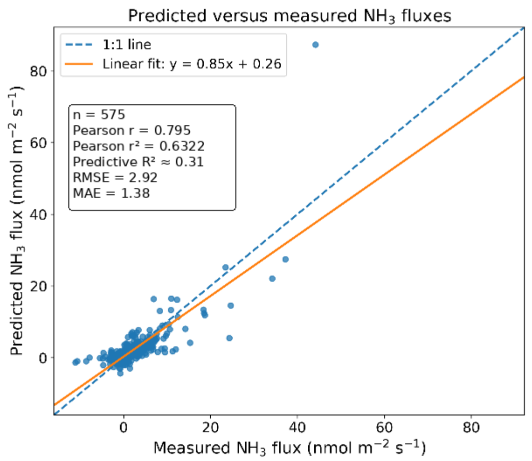


Figure 4. Comparison between measured and model-predicted NH_3 fluxes. The dashed line indicates the 1:1 relationship, and the solid line shows the linear regression fit.

Model interpretation revealed regime-dependent nonlinear controls. During daytime conditions (*PAR* > 0), radiation-related variables interacted strongly with autoregressive memory terms. ICE curves for NH3_lag_1 under daytime conditions (~~Figure 3~~Fig. 5) showed consistent positive persistence effects, while two-dimensional partial dependence surfaces (~~Figure 4~~Fig. 6) indicated that the influence of prior flux state was amplified under high *PAR*. This pattern suggests radiation-modulated volatilization from a dynamically buffered surface reservoir, where energy availability enhances release efficiency conditional on recent emission history.

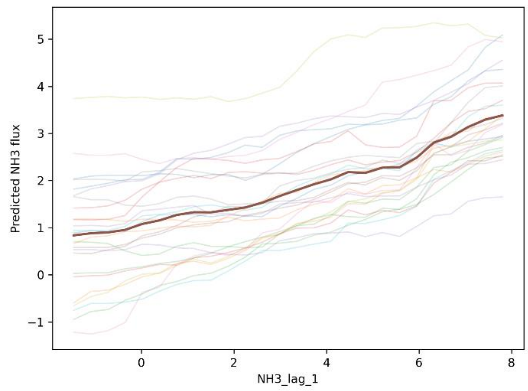


Figure ~~3~~5. Individual Conditional Expectation (ICE) curves for NH_3_lag_1 under daytime conditions (*PAR* > 0). Thin lines represent individual conditional responses, while the thick line shows the mean partial dependence.

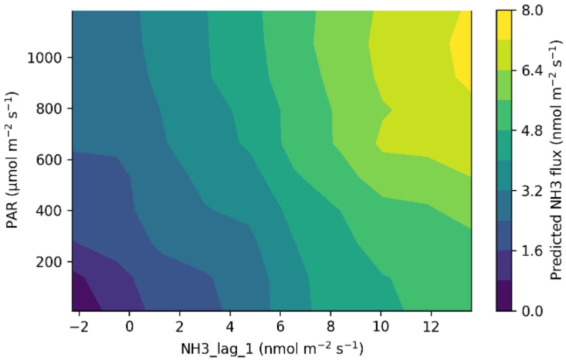


Figure ~~4~~6. Partial dependence surface of predicted NH_3 flux as a function of NH_3 lag-1 and *PAR* under daytime conditions (*PAR* > 0). Colors indicate predicted NH_3 flux (nmol m^{-2} s^{-1}) using the full value range of the model-predicted values for the daytime subset.

In contrast, nighttime conditions (*PAR* = 0) displayed a fundamentally different control structure. Radiation lost explanatory power, and turbulent transport metrics such as friction velocity (u*) and wind speed became dominant. ICE curves for u* (~~Figure 5~~Fig. 7) revealed threshold-like behavior, with suppressed flux under weak turbulence and rapid increases once mixing intensified. Two-dimensional interaction surfaces (~~Figure 6~~Fig. 8) confirmed that nighttime flux is primarily transport-limited, with strong nonlinear coupling between turbulence intensity and wind speed. Autoregressive memory remained relevant but interacted with mixing conditions rather than radiative forcing.

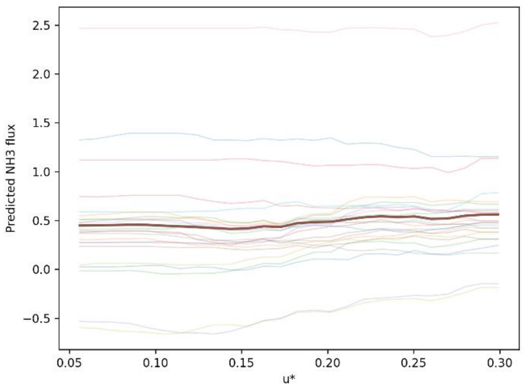


Figure ~~5~~7. Individual Conditional Expectation (ICE) curves for friction velocity (u*, m s^{-1}) under nighttime conditions (*PAR* = 0). Predicted NH_3 flux (nmol m^{-2} s^{-1}) shows threshold-like behavior, with suppressed emissions at low turbulence and rapid increases once mixing intensifies.

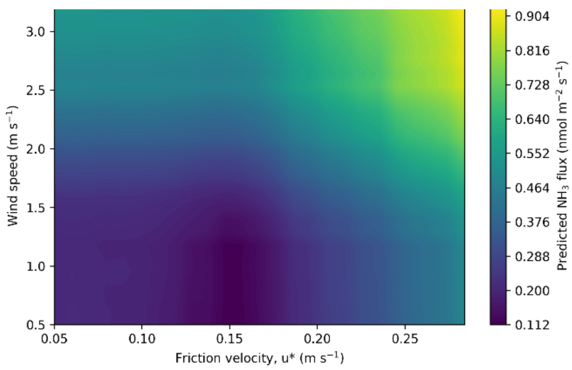


Figure ~~6~~8. Partial dependence surface of predicted NH_3 flux as a function of friction velocity (u*) and wind speed under nighttime conditions (PAR = 0). Colors indicate predicted NH_3 flux (nmol m^{-2} s^{-1}) using the full value range of the nighttime subset; the color scale is not shared with Figure ~~4~~6.

Together, these results demonstrate a clear regime shift between daytime and nighttime NH_3 exchange. Daytime flux is characterized by radiation-amplified persistence, whereas nighttime flux is controlled by turbulence-limited release. The ~~identified -~~temporal persistence represented by the 3-hour ~~system memory~~lag window and the regime-dependent nonlinear interactions indicate that NH_3 emissions reflect a transient soil–surface reservoir modulated by environmental forcing rather than a purely instantaneous response. Chronological validation further highlights that ignoring temporal structure can substantially overestimate predictive skill. These analyses suggest that incorporating temporal persistence and regime-specific environmental controls may improve the interpretation of NH_3 flux dynamics.

Despite the inclusion of multiple soil, meteorological, and turbulence-related predictors, permutation importance consistently highlighted PAR and friction velocity (u*) as dominant controls. The compensation point concentration of the soil-plant system was not directly measured. The inferred reservoir behavior is consistent with, but does not directly quantify, soil–plant compensation point dynamics. Finally, regime-specific analyses were performed separately for daytime (PAR > 0) and nighttime (PAR = 0) conditions to identify potential shifts in dominant environmental controls. Separate ~~ExtraTrees~~tree-based ensemble models were fitted for the ~~two~~daytime and nighttime subsets to resolve regime-specific controls.

4. Conclusions

This study presented the field evaluation and methodological assessment of a photoacoustic NH_3 measurement system combined with an aerodynamic gradient approach for quantifying bidirectional ammonia exchange following mineral fertilizer application under agricultural conditions. The measurement period covered the transition from bare soil to a developing winter rapeseed canopy, enabling the investigation of both emission intensity and evolving exchange dynamics under realistic field conditions.

The combined QCL-photoacoustic gradient system demonstrated robust and stable field performance during continuous long-term operation. Co-located inlet comparison experiments indicated low instrumental bias between sampling channels and a typical random uncertainty of approximately ±2 ppb (1σ). The implemented thermal stabilization, humidity management, and inlet-switching protocol proved suitable for minimizing gradient distortion and maintaining reliable operation under variable outdoor conditions.

Flux calculations based on Monin-Obukhov similarity theory (MOST) using a comprehensive ensemble of universal stability functions showed good methodological consistency. Sensitivity analysis demonstrated that variability attributable solely to stability-function selection remained small relative to the observed diurnal variability of NH_3 exchange, indicating that the dominant uncertainty did not arise from the choice of similarity function itself.

The cumulative NH_3 loss amounted to 1.21±0.30 kg N ha^{-1}, corresponding to 4.0% of the applied fertilizer nitrogen. A clear and quantitatively resolved diurnal asymmetry was observed, with mean daytime emissions (Ig > 0, PAR > 0) reaching 3.11 nmol m^{-2} s^{-1}, approximately one order of magnitude higher than nighttime values (0.448 nmol m^{-2} s^{-1}). This pronounced day–night contrast indicates that post-fertilization NH_3 exchange is strongly coupled to radiation-driven turbulent and surface processes during daytime, whereas nighttime exchange is primarily limited by turbulent transport conditions.

Complementary machine-learning analysis suggested that short-term temporal persistence plays an important role in NH_3 flux dynamics and revealed distinct daytime and nighttime exchange regimes. However, the primary outcome of the study is the demonstration that the presented QCL-based photoacoustic aerodynamic gradient system provides a reliable framework for long-term field-scale NH_3 flux measurements and methodological investigations of reactive trace gas exchange under agricultural conditions.

The combined QCL–photoacoustic gradient system demonstrated robust field performance and low instrumental bias under post-fertilization agricultural conditions, where NH_3 mixing ratios were generally above the instrumental detection limit. Thus, the system is particularly suitable for long-term field-scale studies of agricultural NH_3 exchange, whereas its applicability under relatively clean background conditions may be limited by the present detection limit.

Author Contributions

ZB and LH designed the experiments.

JF, CG, HH, ZN, KP, and AS carried out the field measurements and contributed to data acquisition.

LH prepared the manuscript with contributions from all co-authors.

TW performed the flux calculations.

The datasets are available in the ~~of~~ University of Szeged repository: https://doi.org/10.82570/SZTE_Datarepo_b7hfb-dms70

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Declaration of Competing Interest

The authors declare that they have no conflict of interest.

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Appendix A

Table A1. Universal functions for sensible heat flux in stable stratification for ($0 \leq \zeta \leq +1$) extended interval.

Reference	Karman constant	Universal function for the exchange of sensible heat
McVehil, (1964)	0.4	$\varphi_H(\zeta) = 1 + 6.3\,\zeta$
Tschalikov, (1968)	0.4	$\varphi_H(\zeta) = 1 + 5.17\,\zeta$
Zilitinkevich and Tschalikov, (1968)	0.43	$\varphi_H(\zeta) = 1 + 9.9\,\zeta, (\zeta \leq 0.4), \varphi_H(\zeta) = 4.96, (\zeta > 0.4)$
Zilitinkevich and Tschalikov, (1968)*	0.4	$\varphi_H(\zeta) = 0.95 + 8.9\,\zeta, (\zeta \leq 0.4), \varphi_H(\zeta) = 4.51, (\zeta > 0.4)$
Arya and Plate (1969)	0.4	$\varphi_H(\zeta) = 1 + 17.0\,\zeta$
Webb (1970)	0.41	$\varphi_H(\zeta) = 1 + 5.2\,\zeta$
Businger et al. (1971)	0.35	$\varphi_H(\zeta) = 0.74 + 4.7\,\zeta$
Businger et al. (1971)*	0.4	$\varphi_H(\zeta) = 0.95 + 7.8\,\zeta$
Badgley et al. (1972)	0.4	$\varphi_H(\zeta) = 1 + 7\,\zeta$
Sheppard et al. (1972)	0.41	$\varphi_H(\zeta) = 1 + 5.7\,\zeta$
Carl et al. (1973)	0.4	$\varphi_H(\zeta) = 0.74 + 0.963\,\zeta + 29.6\,\zeta^2$ $(\zeta \leq 0.08), \varphi_H(\zeta) = 1.2 + 6.1\zeta\ (\zeta > 0.08)$
Dyer (1974)	0.41	$\varphi_H(\zeta) = 1 + 5\,\zeta$
Dyer (1974)*	0.4	$\varphi_H(\zeta) = 0.95 + 4.5\,\zeta$
Munro and Davies (1978)	0.41	$\varphi_H(\zeta) = 1 + 4.3\,\zeta$
Lettau (1979)	0.36	$\varphi_H(\zeta) = 1 + 6.3\,\zeta^{\frac{3}{2}}$
Gavrilov and Petrov (1981)	0.4	$\varphi_H(\zeta) = 0.9 + 6\,\zeta$
Foken and Skeib (1983)	0.4	$\varphi_H(\zeta) = 1\ (\zeta \leq 0.125), \varphi_H(\zeta) = \left(\frac{\zeta}{0.125}\right)^2\ (\zeta > 0.125)$
Foken and Skeib (1983)*	0.4	$\varphi_H(\zeta) = 0.95\ \ (\zeta \leq 0.125), \varphi_H(\zeta) = 0.95\left(\frac{\zeta}{0.125}\right)^2\ (\zeta > 0.125)$
Fukui et al. (1983)	0.4	$\varphi_H(\zeta) = 1 + 4.7\,\zeta$
Hurtalova and Szabo (1985)	0.4	$\varphi_H(\zeta) = 1.0 + \zeta(0.118 + 0.405\zeta)^{-1}$ $(\zeta < 0.5), \varphi_H(\zeta) = 2.56\ \zeta \geq 0.5)$
Högström (1988)	0.4	$\varphi_H(\zeta) = 0.95 + 8\,\zeta$
Beljaars and Holtslag (1991)	0.4	$\varphi_H(\zeta) = 1 + \zeta(1 + a\zeta)^b + a\zeta(c - d\eta)e^{-d\zeta}$ where $a = 2/3, b = 1/2, c = 6, d = 0.35$
King et al. (1996)	0.4	$\varphi_H(\zeta) = 0.95\ + 4.99\,\zeta$
Handorf et al. (1999)	0.4	$\varphi_H(\zeta) = 1 + 5\zeta\ (\zeta \leq 0.6),\ \varphi_H(\zeta) = 4\ \ (\zeta > 0.6)$
Weidinger et al. (2000)	0.4	$\varphi_H(\zeta) = 1 + 7.5\,\zeta$
Chenge and ~~Brutseart~~Brutsaert (2005)	0.4	$\varphi_H(\zeta) = 1 + c\,\frac{\zeta + \zeta^d(1+\zeta^d)^{\frac{1-d}{d}}}{\zeta + (1+\zeta^d)^{\frac{1}{d}}}$, where $c = 5.3, d = 1.1$
Chenge and ~~Brutseart~~Brutsaert (2005)	0.4	$\varphi_H(\zeta) = 1 + 54\zeta,\ (\zeta \leq 0.8), \varphi_H(\zeta) = 5.32, (\zeta > 0.8)$
Edwards et al. (2005)	0.4	$\varphi_H(\zeta) = 1 + 4.7\,\zeta$

*recalculated by Högström (1988) after Foken (2017)

Table A2. Universal functions for sensible heat flux in unstable stratification for $(-1 \leq \zeta \leq 0)$ extended interval.

Reference	**Karman constant**	**Universal function for the exchange of sensible heat**
Swinbank (1968)	0.4	$\varphi_H(\zeta) = 1 + 3.478\,\zeta\ (\zeta \geq -0.1)$ (only extrapolation), $\varphi_H(\zeta) = 0.227(-\zeta)^{-0.44}\ (\zeta < -0.1)$
Zilitinkevich and Chalikov, (1968)	0.434	$\varphi_H(\zeta) = 1 + 1.45\,\zeta,\ (\zeta \geq -0.15),$ $\varphi_H(\zeta) = 0.417\,(-\zeta)^{-\frac{1}{3}}\ (\zeta < -0.15)$
Zilitinkevich and Chalikov, (1968)*	0.4	$\varphi_H(\zeta) = 0.95 + 1.31\,\zeta\ (\zeta \geq -0.15),$ $\varphi_H(\zeta) = 0.4\,(-\zeta)^{-\frac{1}{3}}\ (\zeta < -0.15)$
Dyer and Hicks (1970), Dyer (1974)	0.41	$\varphi_H(\zeta) = (1 - 16\,\zeta)^{-\frac{1}{2}}$
Dyer and Hicks (1970), Dyer (1974)*	0.4	$\varphi_H(\zeta) = 0.95\,(1 - 15.2\,\zeta)^{-\frac{1}{2}}$
Businger et al. (1971)	0.35	$\varphi_H(\zeta) = 0.74\,(1 - 9\,\zeta)^{-\frac{1}{2}}$
Businger et al. (1971)*	0.4	$\varphi_H(\zeta) = 0.95\,(1 - 11.6\,\zeta)^{-\frac{1}{2}}$
Carl et al (1973)	0.4	$\varphi_H(\zeta) = 0.74\,(1 - 16\,\zeta)^{-\frac{1}{2}}$
Lettau (1979)	0.36	$\varphi_H(\zeta) = (1 - 22.5\,\zeta)^{-\frac{1}{2}}$
Dyer and Bradley (1982)	0.4	$\varphi_H(\zeta) = (1 - 14\,\zeta)^{-\frac{1}{2}}$
Foken and Skeib (1983)	0.4	$\varphi_H(\zeta) = 1\ (\zeta \geq -0.06),$ $\varphi_H(\zeta) = (-\frac{\zeta}{0.06})^{-\frac{1}{2}}\ (\zeta < -0.06)$
Foken and Skeib (1983)*	0.4	$\varphi_H(\zeta) = 0.95\ (\zeta \geq -0.06),$ $\varphi_H(\zeta) = 0{,}95(-\frac{\zeta}{0.06})^{-\frac{1}{2}}\ (\zeta < -0.06)$
Fukui et al. (1983)	0.4	$\varphi_H(\zeta) = (1 - 8.5\,\zeta)^{-0.}$
Gavrilov and Petrov (1981)	0.4	$\varphi_H(\zeta) = 0.65\left[(1 - 35\,\zeta)^{-\frac{1}{2}} + \frac{0.25}{1 + 8\,\zeta^2}\right]$
Högström (1988)	0.4	$\varphi_H(\zeta) = 0.95\,(1 - 11.6\,\zeta)^{-\frac{1}{2}}$
Weidinger et al. (2000)	0.4	$\varphi_H(\zeta) = (1 - 12.5\,\zeta)^{-\frac{1}{2}}$
Edwards et al. (2005)	0.4	$\varphi_H(\zeta) = (1 - 15\,\zeta)^{-\frac{1}{2}}$
Maronga and Reuder (2017)	0.4	$\varphi_H(\zeta) = 1.1(1.5 - 17\,\zeta)^{-0.57}$
Maronga and Reuder (2017)	0.4	$\varphi_H(\zeta) = 1.1\,(1.5 - 14.5\,\zeta)^{-0.57}$

*recalculated by Högström (1988) after Foken (2017)

Table A3. Logarithmical profile function for all stability classes with widely used turbulent Prandtl (Pr) numbers.

Pr number	Karman constant	Universal function for the exchange of sensible heat
0.74	0.4	$\varphi_H(\zeta) = 0.74$
1	0.4	$\varphi_H(\zeta) = 1$

Appendix B
Evaluation of possible inlet-induced dynamic bias

Because NH_3 is prone to wall interactions, a separate in situ test was performed to evaluate whether the inlet lines used in the aerodynamic gradient measurements introduced a dynamic adsorption/desorption bias under the operating conditions of the study. The test was carried out between 6 and 9 July 2026. The two inlet lines were positioned at the same height so that both sampled the same ambient air. One inlet was maintained at its original length of 4 m, whereas the other was shortened to 0.5 m. This represented an eightfold difference in inlet length and internal wall surface area and therefore provided a sensitive test of possible inlet-induced dynamic effects.

Calibration of the photoacoustic NH_3 analyzer was performed using the same 4 m PTFE inlet configuration, flow rate of 800 cm^3 min^{-1}, and concentration range relevant to the field measurements. Consequently, after equilibration of the inlet walls, any steady-state inlet-related response was incorporated into the calibration function. The purpose of the additional inlet-length experiment was therefore specifically to evaluate whether dynamic, time-dependent adsorption/desorption or memory effects during alternating sampling could distort the measured NH_3 changes.

The test used the same flow rate, alternating channel sequence, and one-minute flushing period as the field campaign. For this dedicated experiment, however, each channel was sampled in a two-minute cycle consisting of one minute of flushing followed by one minute of averaging, whereas a two-minute data-acquisition period was used during the main field campaign. At the applied flow rate of 800 cm^3 min^{-1}, the internal volume of the original 4 m inlet was approximately 50 cm^3, corresponding to a nominal residence time of about 3.8 s. This residence time was short compared with the one-minute flushing period. The measured NH_3 mixing ratios during the test ranged from 33 to 75 ppb.

In total, 969 two-minute measurements were obtained for each inlet configuration. For the evaluation of dynamic inlet effects, the change in NH_3 mixing ratio relative to the preceding measurement from the same inlet was calculated separately for each inlet length, yielding 968 concentration-change values for each configuration. If the 4 m inlet had introduced a relevant dynamic adsorption/desorption or memory effect, this would be expected to appear as a systematic difference between the two inlet configurations in the mean concentration change, its variability, the overall distribution, or the relative occurrence and magnitude of positive and negative changes.

No meaningful difference in the average dynamic response was observed. The mean concentration change was −0.0202 ppb for the 4 m inlet and −0.0182 ppb for the 0.5 m inlet. The difference between the mean changes was therefore only 0.0020 ppb, which was negligible compared with the observed short-term variability and the measurement uncertainty reported in the main text. The corresponding standard deviations were also similar, amounting to 1.93 ppb for the 4 m inlet and 1.84 ppb for the 0.5 m inlet.

The distributions of positive and negative concentration changes were subsequently examined separately to determine whether the longer inlet produced an asymmetric dynamic response. For the 4 m inlet, 483 positive and 485 negative changes were recorded, whereas the corresponding numbers for the 0.5 m inlet were 493 and 475. The positive concentration changes had mean values of 1.460 and 1.381 ppb for the 4 m and 0.5 m inlets, respectively, with corresponding standard deviations of 1.174 and 1.158 ppb. The negative concentration changes had mean values of −1.495 and −1.471 ppb, with standard deviations of 1.319 and 1.173 ppb for the 4 m and 0.5 m inlets, respectively.

Table B1. Summary statistics and comparison of the NH_3 mixing-ratio changes measured with the 4 m and 0.5 m inlet lines. The p-values for the distributions were obtained using two-sided, two-sample Kolmogorov–Smirnov tests.

Concentration changes	**4 m inlet, n**	**4 m mean ± SD (ppb)**	**0.5 m inlet, n**	**0.5 m mean ± SD (ppb)**	**Kolmogorov–Smirnov p**
All changes	968	−0.0202 ± 1.93	968	−0.0182 ± 1.84	0.876
Positive changes	483	1.460 ± 1.174	493	1.381 ± 1.158	0.219
Negative changes	485	−1.495 ± 1.319	475	−1.471 ± 1.173	0.305

Non-parametric distribution tests supported the descriptive comparison. A two-sided, two-sample Kolmogorov–Smirnov test indicated no significant difference between the complete distributions of NH_3 mixing-ratio changes measured with the two inlet lengths ($p = 0.876$). No significant differences were found when the distributions of positive and negative changes were tested separately ($p = 0.219$ and $p = 0.305$, respectively). In addition, a two-sided Fisher's exact test applied to the 2 × 2 contingency table of positive and negative changes showed that their relative frequencies did not differ significantly between the two inlet configurations ($p = 0.682$).

Taken together, shortening the inlet from 4 m to 0.5 m did not produce a detectable change in the mean dynamic response, short-term variability, distribution of concentration changes, or balance between positive and negative changes. Under the applied flow rate, residence time, NH_3 concentration range, and flushing and averaging protocol, the dynamic inlet-induced effect was therefore below the detectable level of the measurement system and negligible relative to the uncertainty of the reported NH_3 fluxes.

The present experiment was designed to evaluate dynamic inlet effects during alternating measurements and does not constitute an independent determination of absolute NH_3 recovery. A dedicated laboratory recovery experiment using a certified NH_3 standard gas or permeation source would therefore remain valuable for further validation. Nevertheless, the calibration performed through the field inlet configuration, together with the in situ inlet-length comparison, provides no evidence that inlet-induced dynamic bias significantly affected the concentration gradients or the reported NH_3 fluxes.