

Ammonia volatilization from flooded soils

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Abstract. Ammonia volatilization from flooded soils has been studied for over half a century. In reviewing the literature on this subject, it becomes clear that there is no consensus on the importance given to this loss mechanism. In part, the differences of opinion can be explained by the fact that ammonia losses were studied in different environments, but to a great extent it seems due to the wide diversity of techniques used to study this loss mechanism.

The many factors that influence ammonia volatilization from flooded soils are chemical, biological, and environmental in nature. These various factors are reviewed in depth and discussed with respect to their implications for measurement techniques and for soil, fertilizer, and water management.

The major objective of this paper is to familiarize the reader with the most current developments in thinking about the mechanisms and extent of ammonia loss and hopefully to stimulate meaningful research on ammonia volatilization from flooded soils. Such research should be conducted in a wide range of agroclimatic conditions utilizing measurement techniques that are valid or for which the limitations are clearly understood. A better appreciation for the importance of ammonia volatilization will provide the impetus to research and development in fertilizer technology and management aimed at preventing such losses.

The various aspects of nitrogen transformation in paddy soils have been studied at least for half a century [20, 21]. Since that time the issue of the relative importance of NH_3 volatilization has been controversial. A number of Indian scientists maintained that NH_3 volatilization was a major component of the loss of nitrogen from flooded soil [35, 19]. Reports from Japan by Iwata and Okuda [23] and Mitsui et al. [29] claimed that, with the exception of strongly alkaline soils ($\text{pH} > 8.5$), ammonia volatilization was negligible. More recent research on NH_3 volatilization generally tends to down-play the importance of NH_3 loss from flooded soils [31, 11, 25, 27, 39, 46, 18]. In contrast, the findings by Ernst and Massey [15], Bouldin and Alimagno [4], Vlek and Craswell [40], and Sahrawat [34] suggest that NH_3 loss from flooded soil can be significant.

It would be futile to attempt to tabulate and compare the reported losses of NH_3 due to the difficulty in assessing the validity of the measurements. Moreover, even for valid measurements it remains difficult to determine for what conditions such data can be considered representative.

Ever since the pioneering work of Kelley [24], ammoniacal N sources have been favored for rice fertilization. At present, over 80% of the N fertilizer used on rice grown in the tropics is urea, which has largely replaced ammonium sulfate (AS) during the last two decades. As elaborated upon below, this change may have seriously increased the extent of ammonia volatilization losses from wetland rice soils. In this review, we summarize recent research on factors affecting the rate of ammonia loss from flooded soils and comment on the measurement and practical implications of these losses.

Floodwater chemistry

The main problems in comparing results from volatilization studies are the complexity of the process and the multitude of factors influencing it. The many factors that influence ammonia volatilization from flooded soils are environmental, chemical, and biological in nature. The NH_3 volatilization process can be regarded as a chain of events, the overall rate of which can be controlled at any point in the chain.

Figure 1 schematically represents the process and some of the key factors affecting the rate of various N fluxes. The floodwater ammonium may originate from the soil N pool or from extraneous sources such as fertilizer, organic amendments, and crop residues. The presence of algae may increase or decrease the floodwater NH_4 level, dependent on whether the algal biomass is decaying or growing [43].

It is now recognized that the concentration of ammoniacal N depends on the soil type (particularly texture and permeability); fertilizer rate, source, and mode of application; and floodwater level [40, 27]. Figure 2 summarizes some of the effects of soil type and fertilizer management on the floodwater dynamics following fertilization. Whether incorporated or not, a large fraction of surface-applied N is generally found in the floodwater. In the case of urea this floodwater N must be hydrolyzed before it can add to the floodwater NH_4^+ pool. Vlek et al. [43] found that for three soils varying in cation exchange capacity from 16.2 meq per 100 g to 49.2 meq per 100 g, 50%–60% of the urea was found in the floodwater, despite incorporation into the soil. This floodwater urea was found to be subsequently hydrolyzed at the soil-floodwater interface. Only a fraction (20%) of the formed ammonium was retained by the soil, while the remainder was found as ammoniacal N in the floodwater. One way to prevent these high ammoniacal N levels from developing is by deep placement of the fertilizer in the puddled soil. The effect of this method of fertilizer application is shown by curve E in Figure 2.

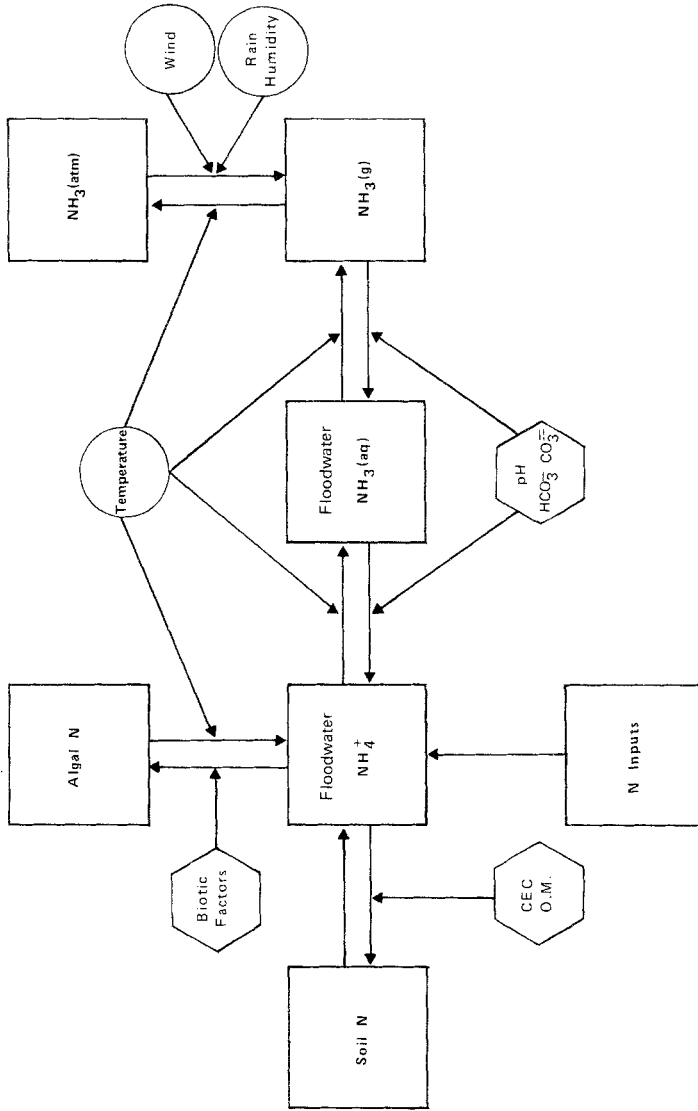


Figure 1. Schematic representation of the origin and fate of ammonium in floodwater and some of the key factors influencing its transformations

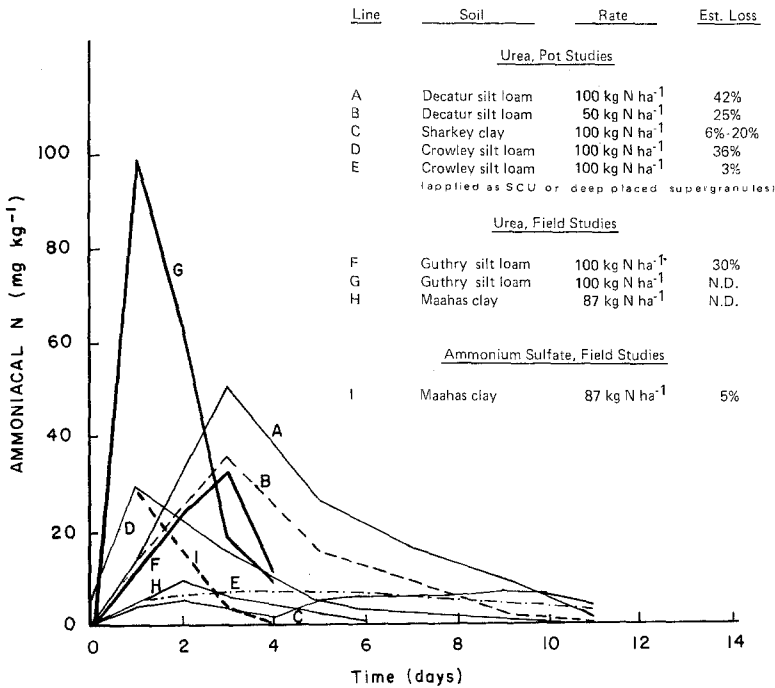


Figure 2. Changes in ammoniacal nitrogen concentration of the floodwater following fertilization of pot and field with urea or ammonium sulfate

The chemical dynamics of ammonia loss from floodwater was described by Vlek and Stumpe [41] as a consecutive reaction with an opposing step while ammonia volatilization per se follows first-order reaction kinetics [17, 30]. The rate of ammonia volatilization is thus directly related to the concentration of aqueous ammonia $[\text{NH}_3(\text{aq})]$ and therefore to the concentration of ammoniacal nitrogen, pH, and temperature of the floodwater, regardless of the source of NH_4 .

Mikkelsen and De Datta [26] derived a complicated set of equations and concluded that the pressure of ammonia in solution is a function of pH; the concentrations of ammonium, bicarbonate, and carbonate; and the pressure of CO_2 in solution. However, as is shown below, the partial pressure of ammonia in solution (P_{NH_3}) is in fact uniquely determined by the ammoniacal nitrogen concentration (AN) and pH of that solution. The ammoniacal N concentration in solution is equal to the sum of the various species:

$$\text{AN} = \text{NH}_4^+ + \text{NH}_3(\text{aq}) + \text{NH}_4\text{L} \quad (1)$$

Ignoring ion-pair formation (NH_4L) and activity corrections and substituting

$$\frac{\text{NH}_3(\text{aq}) \cdot \text{H}^+}{\text{NH}_4^+} = K, \quad \text{pK}^\circ = 9.24 \quad (2)$$

yields:

$$\text{NH}_3(\text{aq}) = K \frac{[\text{AN} - \text{NH}_3(\text{aq})]}{\text{H}^+} = \frac{K \cdot \text{AN}}{\text{H}^+ + K} \quad (3)$$

and thus

$$P_{\text{NH}_3} = \frac{\text{NH}_3(\text{aq})}{K_{\text{H}}} = \frac{K \cdot \text{AN}}{K_{\text{H}}(\text{H}^+ + K)}, \quad \text{pK}_{\text{H}}^\circ = 1.77 \quad (4)$$

Figure 3 summarizes the effect of pH on the distribution of ammoniacal species in solution. In these calculations all known ion pairs of ammonium and carbonaceous species were taken into consideration and plotted in Figure 3, if significant. Activity coefficients were assumed unity. The $\text{NH}_3(\text{aq})$ level in the floodwater is found by multiplying the ammoniacal N level with the mole fraction of $\text{NH}_3(\text{aq})$ for the appropriate pH. The mole fraction of $\text{NH}_3(\text{aq})$ increases rapidly above pH 7.5 and then declines when the formation of NH_4CO_3^- becomes an important species. However, for the pH levels common for floodwater, ion-pair formation is rather insignificant, and the aqueous ammonia level is virtually insensitive to fluctuations in CO_2

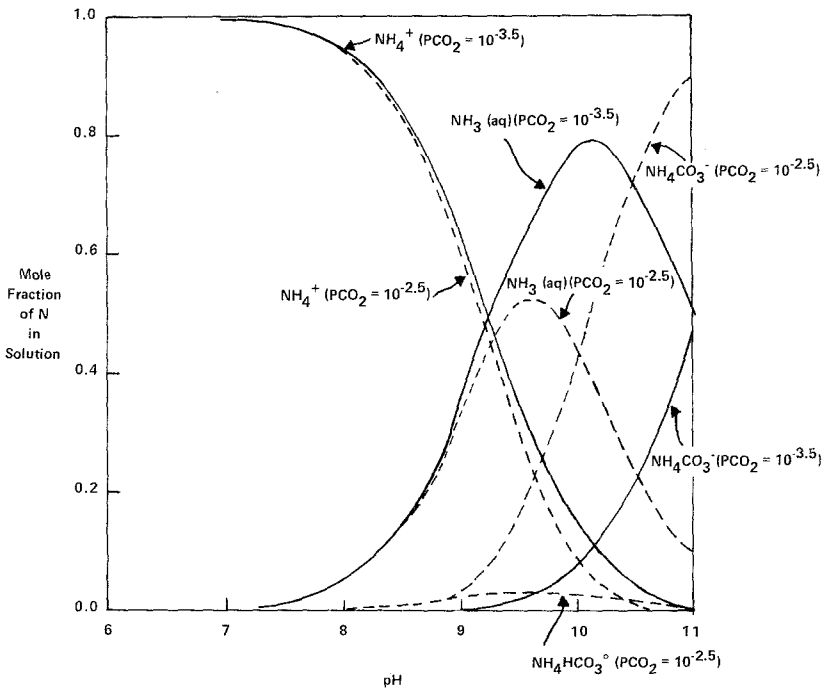


Figure 3. Fractions of ammoniacal nitrogen in solution as ions or ion-pairs

Conversion of NH_4^+ in solution to aqueous NH_3 is an extremely rapid (first-order) process with a rate constant of 24.6 sec^{-1} [14] and is thus rarely limiting to the overall volatilization process. The concentration of $\text{NH}_3(\text{aq})$ changes in direct proportion with ammoniacal N and increases about tenfold per unit increase in pH up to $\text{pH} \cong 9$ [41]. At a given total ammoniacal nitrogen concentration and pH, the $\text{NH}_3(\text{aq})$ concentration increases approximately linearly with increasing temperature [2], resulting in a nearly four-fold increase with a change in temperature from 10° to 40°C .

Following the hydrolysis of urea at the soil-water interface, the floodwater receives some of the ammonium and carbonate which is not absorbed by the soil and becomes a weak ammonium bicarbonate solution. Because of its inherent buffering capacity, this solution will tend to maintain a pH of about 8. The dilute ammonium bicarbonate solution is an ideal growth medium for green algae, though generally algal development is delayed for a few days before extensive algal blooms become apparent. The extent of algal biomass production will depend on soil characteristics and nutrient availability as well as climatic factors such as temperature and light. Algal biomass production in Senegal paddy fields was found to reach as high as 5 t ha^{-1} of wet material [32]; whereas, Watanabe et al. [44] report algal biomass production in excess of 20 t ha^{-1} . Photosynthetic and respiratory activities of such algal populations can cause significant diurnal fluctuation in the floodwater CO_2 level and, consequently, in the floodwater pH [4].

An example of the dynamics of floodwater pH following fertilization with urea is presented in Figure 4. The floodwater pH was buffered at a pH of nearly 8 for about 2 days after which time the strength of the ammonium bicarbonate solution had dropped sufficiently to lose its buffering characteristics and respond to CO_2 fluctuations due to algal activity. It is also clear that the pH fluctuations increased with time because of the increasing biomass of green algae.

In order that ammonia volatilization can proceed, buffering substances in the form of alkalinity must be present in the system in order to neutralize the production of H^+ ions that accompanies the production of aqueous ammonia [41]. Ammonia volatilization of applied ammonium is preceded by the reaction



The equilibrium constant is such that appreciable amounts of NH_3 can be formed only at $\text{pH} > 7.5$. While reaction (5) is going on, protons are released, and there is a tendency for acidification. In order to keep the pH high enough for volatilization to continue, the protons must be removed and the solution buffered at high pH. The only proton acceptor capable of that and present in appreciable quantities in the surface water of rice fields is bicarbonate, HCO_3^- :



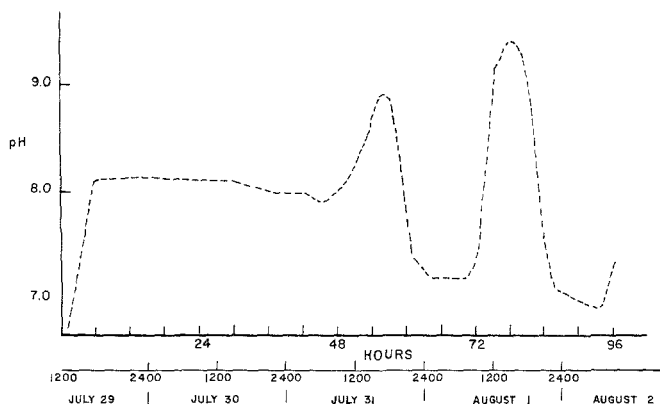
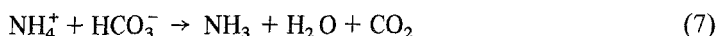


Figure 4. Dynamics of floodwater pH following broadcast and incorporation of 100 kg N ha^{-1} as urea. Floodwater depth 5 cm

Combining reactions (5) and (6) gives



The stoichiometry of reaction (7) is such that equivalent amounts of NH_4^+ and HCO_3^- are consumed in the reaction. Thus, the amount of HCO_3^- in the surface water sets an upper limit to the amount of NH_4^+ that can possibly be volatilized. Mikkelsen and De Datta [26] state that at low concentrations of HCO_3^- in the surface water the pH can become sufficiently high for ammonia volatilization if CO_2 is withdrawn by algae. Although this observation is correct, such high pH values are necessarily short lived because HCO_3^- will be consumed during the $\text{NH}_4^+ \rightarrow \text{NH}_3$ reaction and the pH will drop to levels too low for further volatilization. It is therefore not surprising that the largest fluctuations in floodwater pH occur once the ammoniacal N level has dropped substantially [10].

The effect of floodwater chemistry on the buffering capacity of floodwater is given in Figure 5 where the pH of various types of floodwater is plotted against the gain or loss of total carbon from the floodwater due to algal respiration or photosynthesis [37]. The world's average river water contains about $1 \text{ meq l}^{-1} \text{ HCO}_3^-$ while rainwater contains virtually none. Consequently, rainfed paddy water exhibits essentially no buffering capacity; large fluctuations in pH result even at relatively low algal activity. On the other hand, approximately 0.1 meq l^{-1} of carbon needs to be photosynthesized by algae to raise the pH of river water from 7.5 to 9.0. To accomplish the same pH change in floodwater derived from river water but fertilized with AS to contain 3 meq NH_4^+ (42 mg N l^{-1}) will require immobilization of $\cong 1.0 \text{ meq C l}^{-1}$. An even larger algal activity (1.4 meq C l^{-1}) would be required for a 'rainfed paddy' fertilized with 42 mg of urea N per Kg, once the urea was hydrolyzed to form the buffered ammonium carbonate

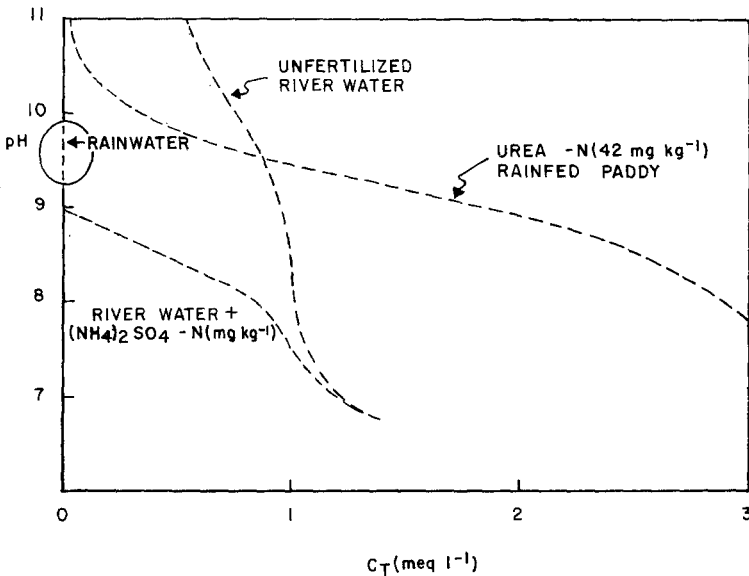


Figure 5. Effect of carbon assimilation or dissimilation by algae on the pH of floodwaters of different chemical compositions

solution. Extraneous sources of alkalinity such as irrigation with alkaline well water or $CaCO_3$ in the underlying topsoil may also prevent large diurnal fluctuations in floodwater pH due to algae.

Thus, in general, ammonia volatilization from urea-fertilized soils far exceeds that from ammonium sulfate-fertilized soils (Figure 6) because urea, upon hydrolysis, creates a solution ideally suited for volatilization; i.e., the solution has a high alkalinity and pH [41]. Ammonia volatilization losses from ammonium sulfate will become substantial only if the soil-floodwater system can provide the alkalinity necessary to buffer the system at a high enough pH to sustain the volatilization process. Such conditions may be found in areas with calcareous soils or where soils are irrigated with alkaline well waters (e.g., the International Rice Research Institute farm).

Environmental conditions

From a chemical standpoint, ammonia volatilization is thus largely determined by the floodwater dynamics with respect to pH, alkalinity, and ammoniacal N. The net result of these three factors on the dynamics of aqueous ammonia in the floodwater can be considered the ammonia volatilization potential, while actual ammonia volatilization loss rates will further be determined by environmental factors such as temperature, wind speed, and rainfall conditions. Some of these factors were studied and quantified for water impoundments of nonagricultural nature [17, 36, 45, 3]. The effects of

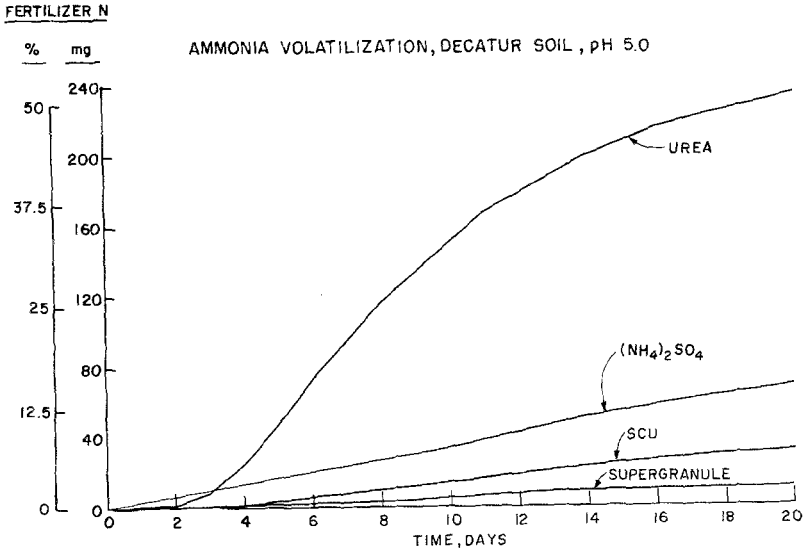


Figure 6. Apparent NH_3 loss of fertilizer nitrogen as affected by fertilizer source applied to an acid, silty-loamy, rhodic paleudult, as measured by the forced draft system

changing ammoniacal nitrogen concentration, pH, temperature, and wind speed were evaluated simultaneously using an ammonia diffusion model, and the results are presented in Figure 7 [5].

The importance of wind on the rate of ammonia volatilization has often been ignored by agricultural scientists and has led to a general underestimation of this loss mechanism. The importance of air movement in the NH_3 volatilization process is easily appreciated from the following simple calculations, modified after Berner [1].

Assume a hypothetical volume of air above a source of $\text{NH}_3(\text{g})$ (e.g., a head space) that is continually flushed with ambient air, and assume that the $\text{NH}_3(\text{g})$ concentration in the head space is spatially uniform. Conservation of mass requires that:

$$\frac{d\text{NH}_3(\text{g})}{dt} = R - \frac{F}{V} [\text{NH}_3(\text{g}) - \text{NH}_3(\text{ambient})] \quad (8)$$

where $\text{NH}_3(\text{g})$ = concentration of gas in head space (mass/volume)
 $\text{NH}_3(\text{ambient})$ = concentrations of NH_3 in ambient air (mass per volume)
 t = time
 F = flushing rate (volume per time)
 V = volume of head space
 R = rate of volatilization (mass per time)

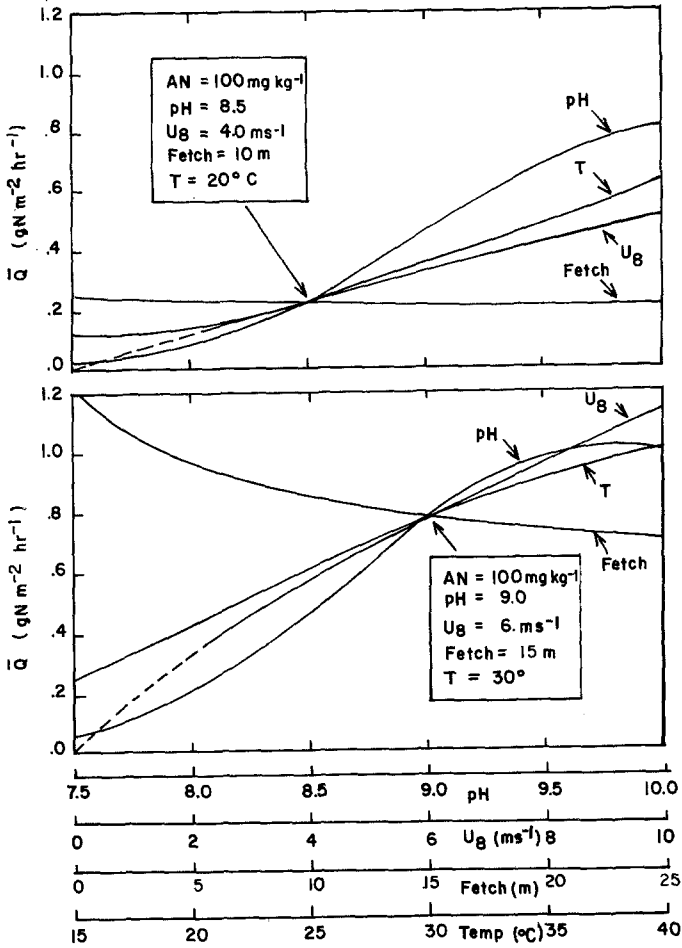


Figure 7. Variation of ammonia volatilization with wind velocity (U_B), temperature (T), fetch (F), and pH

For simplicity one may assume that the rate of volatilization is linearly proportional to the degree of undersaturation of the gaseous NH_3 in the head space $[\text{NH}_3(\text{g})]$ with respect to the gaseous NH_3 level above the floodwater if Henry's law equilibrium was maintained $[\text{NH}_3(\text{eq})]$, thus:

$$R = k [\text{NH}_3(\text{eq}) - \text{NH}_3(\text{g})] \tag{9}$$

If it is further assumed that steady state conditions exist in the fixed head space,

$$\frac{d\text{NH}_3(\text{g})}{dt} = 0$$

then

$$\text{NH}_3(\text{g}) = \frac{k \text{NH}_3(\text{eq}) + k_f \text{NH}_3(\text{ambient})}{k_f + k} \quad (10)$$

and,

$$R = \frac{k_f k [\text{NH}_3(\text{eq}) - \text{NH}_3(\text{ambient})]}{k_f + k} \quad (11)$$

where

$$\begin{aligned} k_f &= \text{flushing frequency } (F/V) \\ k &= \text{NH}_3 \text{ evasion constant} \end{aligned}$$

Equation (11) shows the rate of NH_3 volatilization to be dependent on $\text{NH}_3(\text{eq})$ as well as the ambient NH_3 level. However, in the case that $\text{NH}_3(\text{eq}) \gg \text{NH}_3(\text{ambient})$, the effect of the latter becomes insignificant and can be set equal to zero. With NH_3 levels of 2–6 ppb representative for unpolluted air, this condition requires levels of aqueous NH_3 in the floodwater of 0.5 ppm or higher. Where NH_3 volatilization is a problem, such levels of $\text{NH}_3(\text{aq})$ are easily reached, yielding:

$$R = \frac{k k_f \text{NH}_3(\text{eq})}{k + k_f} \quad (12)$$

From equation (12) it follows that an increase in flushing rate F , at constant k , $\text{NH}_3(\text{eq})$, and V , will cause an increase in the rate of NH_3 volatilization R . At very high flushing rates ($k_f \gg k$) equation (8) reduces to:

$$R \cong k \text{NH}_3(\text{eq}) \quad (13)$$

and the maximum rate of volatilization or volatilization potential is realized. The volatilization rate is no longer affected by the rate of flushing.

However, if the flushing rate is low, F decreases; as a result, R decreases. The ultimate case $k_f \ll k$ yields:

$$R \cong k_f \cdot \text{NH}_3(\text{eq}) \quad (14)$$

and the rate of volatilization is controlled entirely by the rate of flushing. These conditions appear to prevail in many of the NH_3 volatilization measurements using enclosures conducted in the past.

In a more detailed analysis, the ammonia transfer from paddy water to the atmosphere was found to be a diffusion-controlled process [5]. The effect of wind (U_8) on the rate of ammonia volatilization \bar{Q} is given in Figure 7 for different floodwater conditions. The diffusion model developed by Bouwmeester and Vlek [5] distinguishes the various rate-controlling factors: (a) the reaction rate of equation (5), (b) the transfer resistance in the liquid phase, and (c) the transfer resistance in the gas phase. To show the relative importance of these factors, volatilization rates (\bar{Q}) were calculated using the diffusion model and plotted in Figure 7. For low windspeeds the volatilization

rates are very small, and the gas phase resistance dominates. However, with increasing windspeed the volatilization rates increase, and the liquid phase resistance becomes more significant due to depletion of NH_3 in the surface film of the liquid phase. This shift from gas phase resistance to liquid phase resistance is more evident for high pH values [5].

Finally, the geometry of the rice paddy may influence the volatilization rate of NH_3 from the floodwater. The effect of fetch, as shown in Figure 7, is of little consequence for the size of most rice paddies and is, therefore, not discussed in depth.

Measurements of NH_3 volatilization

Techniques for measuring fertilizer losses of NH_3 are manifold and controversial, varying from closed systems with or without air turbulence to micrometeorological and aerodynamic techniques in undisturbed fields. To some extent, the differences in methodology employed can be explained by the differences in the objectives of the measurement programs. For instance, for the purpose of studying actual losses of ammonia from the field, micrometeorological and aerodynamic techniques are preferable [12, 13]. On the other hand, for studying the effect of environmental and N management factors on ammonia volatilization rates, wind tunnel techniques are most appropriate [5, 6]. To assess the effect of fertilizer management on the potential for ammonia volatilization, measurements under small enclosures can be successfully employed, provided the system is set up so that the gas phase resistance does not dominate the volatilization process [7].

Each of the methodologies has merits and demerits. Micrometeorological techniques are elaborate and provide information only for the place and time of the experiment. Due to the stochastic character of the environmental factors affecting NH_3 loss, such measurements are not easily extrapolated to different environmental conditions. The disturbance of the natural conditions under enclosures, however, causes these measurements to be inaccurate unless care is taken to simulate the gas phase resistance of the undisturbed field throughout the measurement [6]. Generally, the rate of airflow in such systems has been low, thus the rate of volatilization is limited. Drastic underestimation of the importance of this loss mechanism results. When airflow rates under enclosures are high, the measurement may, at best, indicate the upper limit of volatilization.

To demonstrate the importance of adequate airflow when studying ammonia volatilization from floodwater, data from two experiments conducted in 1979 and 1980 in the field are given in Figure 8. Ammonia was trapped from air flushing the head space of an enclosure permanently installed in the field following the broadcast application and incorporation of urea according to practices commonly recommended in Asian rice-growing areas. During the relatively cool summer of 1979, the flushing rate was such

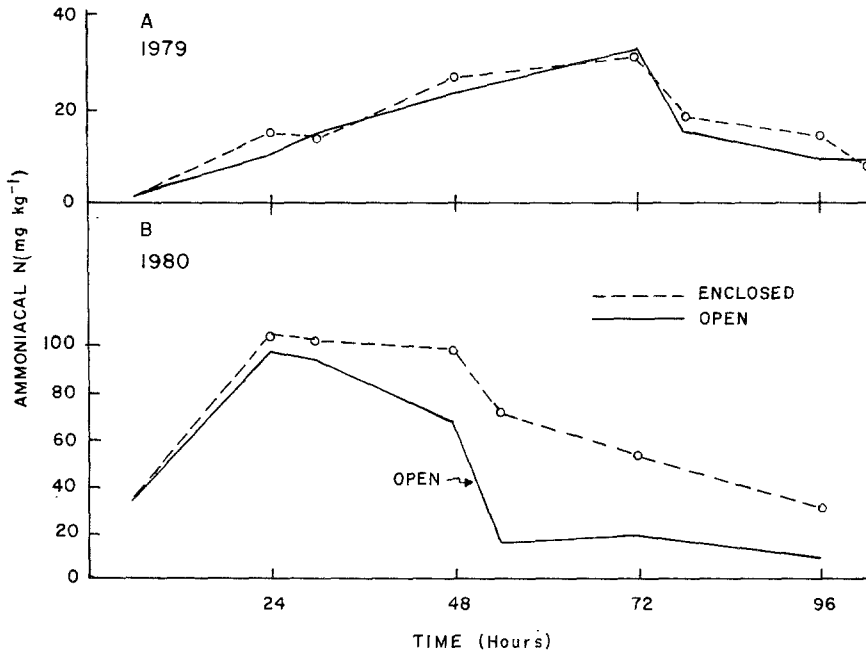


Figure 8. Dynamics of ammoniacal N in the floodwater of a field following fertilization with $100 \text{ kg Na ha}^{-1}$, under an enclosure with forced draft (broken lines), or in the surrounding open paddy (solid lines)
 a. Relatively cool summer of 1979
 b. Hot summer of 1980

that no temperature differential in the air or floodwater inside and outside the enclosure developed. As is seen in Figure 8A, the ammoniacal nitrogen concentration in the floodwater inside and outside the enclosure fluctuated in a like manner, indicating that the rate of ammonia loss inside the enclosure adequately simulated that of the bulk paddy. The total loss of NH_3 amounted to approximately 30% of the urea applied, 100 kg N ha^{-1} [22].

In contrast, during the exceptionally hot summer of 1980 (Figure 8B), the flushing rate was obviously inadequate to maintain an ammonia volatilization rate equal to that of the bulk paddy. The total loss as measured under the enclosure was again approximately 25%, but our confidence in the accuracy of this measurement is low due to the dissimilar behavior of the floodwater chemistry inside and outside the enclosure. Further research may improve the performance of this technique. One means of improvement suggested by Vallis and Harper [38] is to couple the flushing rate through an electronic feedback system to the fluctuating meteorological conditions outside the enclosure. Alternative measuring techniques under enclosures all suffer from one limitation or another.

The first attempts to assess NH_3 losses from flooded soils without disturbing the atmospheric environment were probably made by Bouldin and Alimagno [4] at the International Rice Research Institute (IRRI). The main approach used in their research was based on the disappearance of mineral N from small plots (20×20 cm) isolated from the bulk paddy by metal sleeves pushed into the soil down to the plow sole. Four such plots were sampled daily down to the plow sole, and the mineral N unaccounted for was assumed lost as NH_3 . To further verify the loss mechanism, a similar system was also maintained undercover throughout the study period of 5–10 days, thus preventing ammonia volatilization. Ammonia volatilization losses estimated by these methods appeared alarmingly high (up to 60% of broadcast AS).

Only very recently have efforts been made to apply micrometeorological measurement techniques to flooded soils [18]. The various micrometeorological methods have been adequately described elsewhere [12]. Bouwmeester and Vlek [6] evaluated aerodynamic methods of calculating ammonia volatilization rates with the so-called longitudinal flux method as the basis of comparison. The studies were carried out under simulated paddy conditions in a wind–water tunnel. They found that the longitudinal flux method [13] and the aerodynamic methods were in close agreement and suitable for flooded soils. Freney et al. [18] used the micrometeorological technique of Denmead and McIlroy [12] to calculate losses of AS applied to fields at IRRI. They reported losses of no more than 10.6% of the applied N, a drastic difference from the magnitude of loss reported for the same fields by Bouldin and Alimagno [4]. A typical pattern of N loss reported by Freney et al. [18] is given in Figure 9.

The discrepancy between the two reports may be due to differences in weather conditions during the experiments. Freney et al. [18] measured losses in the wet season, whereas Bouldin and Alimagno [4] worked in the dry season. In the wet season, the solar radiation is less, the wind is more variable, and the usage of irrigation water, which at IRRI is alkaline, is less than in the dry season. The enhancement of air turbulence above the microplots used by Bouldin and Alimagno was caused by the metal walls protruding from the water surface. This enhanced air turbulence may have caused an overestimation of the volatilization rate and, thus, could partially explain this discrepancy. Obviously, further work comparing these methods is warranted to clarify this point. Micrometeorological measurements of NH_3 losses from urea are also needed.

The wide variation in seasonal and agroclimatic conditions under which rice is grown may cause wide variation in measured NH_3 loss. A proper assessment of the relative importance of ammonia volatilization as compared to other loss mechanisms will require an extensive as well as intensive field measurement program. A single field measurement of the loss of fertilizer N through ammonia volatilization is therefore inadequate for the assessment of the magnitude of the problem. Hopefully, a relatively simple measurement

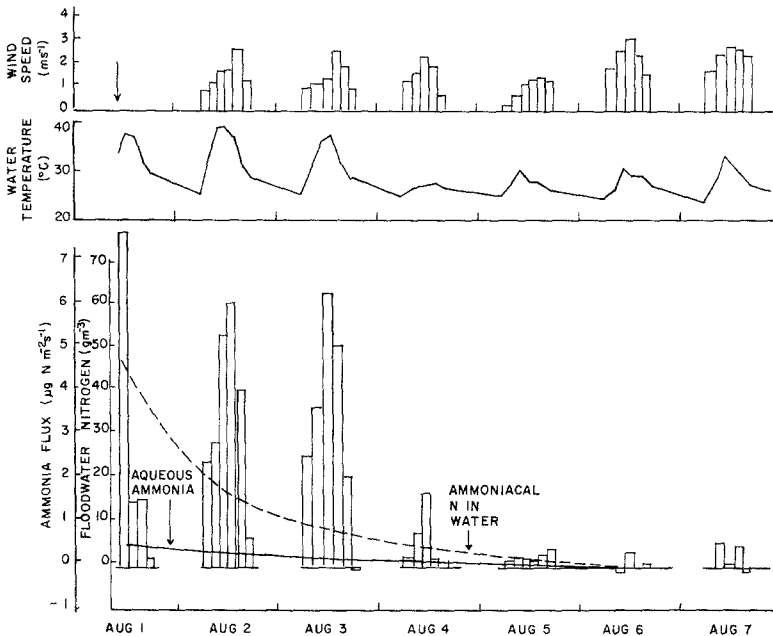


Figure 9. Ammonia and ammoniacal N in floodwater, water temperature, and wind speed at 0.25 m and the ammonia flux pattern following a preplanting application of ammonium sulfate (80 kg N ha^{-1})

technique will eventually be devised which, after proper calibration against a micrometeorological technique, can be used for extensive measurements under a wide range of conditions.

Practical implications

As stated earlier, the main interest in developing a better understanding of the NH_3 volatilization process stemmed from the need to reduce such losses. Several possibilities to reduce NH_3 losses have recently been reviewed by Craswell and Vlek [9]. The various methods presently under investigation are (1) deep placement, (2) slow-release fertilizers, and (3) urease inhibition. These methods supplement the traditional options of timing and splitting of fertilizer applications.

Deep placement. This concept relies upon placing the fertilizer in the reduced soil layer so that the concentration of urea and ammonium in the floodwater remains essentially zero. Originally, placement was thought to improve fertilizer efficiency primarily by reducing nitrification–denitrification [28]. However, our research has shown that deep placement also largely prevents ammonia volatilization losses [40]. Craswell and Vlek [8] showed that losses of ^{15}N -labeled supergranules under greenhouse conditions were negligible.

If the placement method effectively prevents urea from entering the floodwater, runoff losses would be prevented, and the immobilization of applied nitrogen by green algae would be suppressed. Another important advantage is that deep placement does not inhibit the growth and nitrogen-fixing activity of the blue-green algae in the floodwater and surface soil; thus a bonus of nitrogen to the soil-plant system is provided [33].

Slow-release fertilizers. The slow-release concept relies on delaying the availability of soluble nitrogen to the plant until the plant has a strong root system which can compete with the loss mechanisms and biological immobilization for the fertilizer nitrogen. Furthermore, if the release rate can be tailored to the needs of the plant, the tillers, panicles, spikelets, and grains will develop in the most efficient way to ensure high yield [42]. Less labor is required with slow-release fertilizer than for split application and more importantly in some countries, less technical skill is needed by the farmer.

Inhibitors. Part of the rationale for research on nitrogen losses has been that it is essential to know which loss mechanism must be blocked before effective means of preventing losses can be developed. This is particularly true in the case of inhibitors. Thus, if nitrification–denitrification is not a serious problem, nitrification inhibitors may not be necessary. Sufficient evidence has been gathered to suggest that broadcast urea is primarily subject to loss by ammonia volatilization [16] and that the use of urease inhibitors should also be investigated. Vlek et al. [43] recently showed that phenylphosphorodiamidate (PPD), added at a rate of only 2% (w/w of urea), delays the appearance of ammonia in floodwater after broadcasting urea. This promising result has suggested a need for further research to develop urease inhibitors.

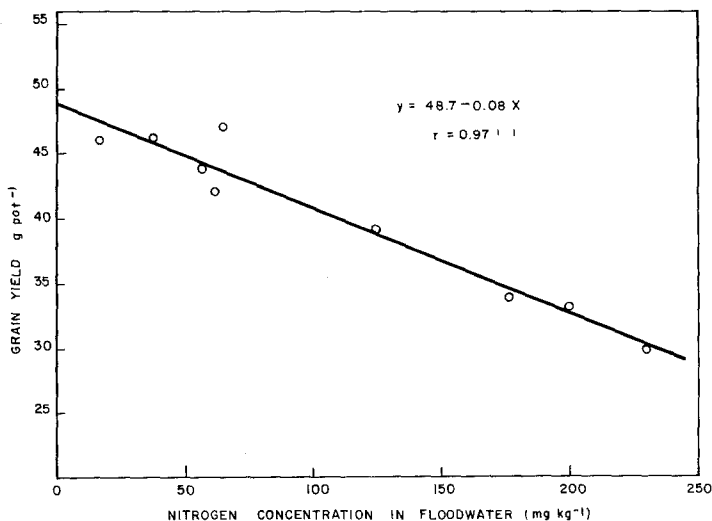


Figure 10. Relationship between grain yield and nitrogen concentration in floodwater one day after fertilization and transplanting. Greenhouse experiment 1980

All of these methods aim at reducing the development of NH_3 (aq) in the floodwater. Vlek and Craswell [40] found that under controlled conditions a good correlation existed between volatilization losses and the NH_3 (aq) concentration of the floodwater sampled between 1000 and 1100 hours each day. In a greenhouse experiment we subsequently were able to correlate the response of rice to various N sources and application methods with the mineral N level of the floodwater 1 day following fertilizer application. The excellent negative correlation found is presented in Figure 10 and may provide a basis for future evaluation of fertilizer sources and practices for rice.

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