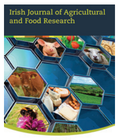

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Ammonia emission factors from cattle production systems in Ireland – a review

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Abstract

Ammonia (NH_3) emissions from livestock production contribute to environmental pollution. To address this challenge, the European Union (EU) National Emission Reduction Commitments Directive 2016/2284 (NECD) sets NH_3 reduction targets for EU member states. In order to achieve these targets, several strategies have been evaluated under Irish conditions. A compilation of emission factors (EFs) from studies which evaluated these strategies is necessary to assess their effectiveness. This paper reports NH_3 EFs from cattle production under Irish conditions. The results from the review show that the mean EFs from the deposition of dung, urine and urea applied to urine patches on grasslands were 4%, 9% and 8% total nitrogen (TN), respectively. EFs from the application of urea to urine patches were reduced by 28% after the addition of the urease inhibitor N-(n-butyl) thiophosphoric triamide (NBPT) to urea. The mean EF of 28% TN reported for urea fertiliser was almost 7 times higher than calcium ammonium nitrate (CAN). The inclusion of urease inhibitors with urea fertilisation on grassland led to EF reduction of up to 86%. The mean EFs from cattle houses, concrete yards, slurry storage pits and slurry landspreading were approximately 13%, 35%, 60% and 59% total ammoniacal nitrogen (TAN), respectively. The most effective NH_3 abatement strategies for concrete yards and slurry storage were immediate cleaning of concrete floors (up to 89% reduction) after excreta deposition and the application of chemical amendments (sulphuric acid, acetic acid, alum and ferric chloride) to slurry in storage pits (up to 98% reduction), respectively. Low-emission spreading strategies and slurry acidification were effective at abating EFs after slurry application to land.

Keywords

Ammonia • ammonia emission factors • ammonia mitigation • livestock excreta • manure management

Introduction

Agriculture is responsible for 99.4% of ammonia (NH_3) emissions from the Republic of Ireland (ROI) (EPA, 2022). The national total emissions in the ROI increased by 12.4% from 109.80 kt in 1990 to 123.4 kt in 2020 with livestock production accounting for the majority of national total emissions (EPA, 2022). In the year 2020, application of manure to soil, manure management and deposition of urine and dung by grazing animals together contributed to 90.1% of the ROI's national total emissions (EPA, 2022). Other factors contributing to the increase in the ROI's emissions include the increase in fertiliser use, continued use of urea as an inorganic nitrogen (N) fertiliser as well as the increase in dairy cattle and other cattle populations (EPA, 2022).

Similarly, the agricultural sector remains the main source of NH_3 emissions in Northern Ireland (NI) with cattle manure management contributing to at least 35% of the emissions (NAEI, 2022). Emissions in NI have increased since 2011 mainly due to increasing dairy herd size and emissions associated with dairy manure management (NAEI, 2022). The trend has remained fairly stable since 2017 even though the slight declines in dairy cattle numbers and in mineral fertiliser are being offset by an increase in poultry production (NAEI, 2022).

Atmospheric NH_3 contributes to acid depositions and high concentrations of nutrients in aquatic and terrestrial ecosystems which poses a threat to the sustainability of these ecosystems (Wilkins *et al*., 2016; Payne *et al*., 2017). According to DEFRA (2022), most of NI including designated sites and other priority habitats are now receiving levels of nitrogen (N) which are significantly above their critical loads, or the concentration at which significant and irreparable ecological damage occurs. Similarly, the risk of environmental degradation as a result of NH_3 deposition on terrestrial ecosystems is quite high in the ROI due to intensive agricultural production systems. Indeed,

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after monitoring NH_3 concentrations in 12 Natura 2000 sites in the ROI, Kelleghan *et al*. (2021) observed that 11 sites had exceeded either their critical level or load whereas 10 sites had exceeded both. The MARSH model also identified 80.7% and 5.9% of Natura 2000 sites in the ROI may exceed critical levels of 1 μg/m^3 (i.e. habitats where lichens and moss are important features) and 3 μg/m^3 (all other habitats), respectively (Kelleghan *et al*., 2019).

The European Union (EU) has set limits for national emissions of NH_3 under the National Emission Reduction Commitments Directive (NECD) 2016/2284. According to the NECD, the ROI and the United Kingdom (UK) are required to reduce NH_3 emissions relative to 2005 baseline by 1% and 8% by 2020 and 5% and 16% by 2030, respectively (EPA, 2022; NAEI, 2022). The ROI has exceeded its NH_3 limit since 2016 and though recent predictions estimate that the ROI can meet its 2030 target, immediate adoption and implementation of emission reduction strategies is required (Buckley *et al*., 2020).

Strategies for abating NH_3 emissions in livestock production include dietary manipulation (e.g. reducing the crude protein content in animal diet), housing strategies (e.g. low-emission flooring types, regular cleaning of housing floors, the application of urease inhibitors on floors), storage strategies (e.g. reducing slurry pH and NH_4^+ content, covering of slurry stores, inducing crust formation), field strategies (e.g. adopting low-emission spreading strategies [LESS]), rapid incorporation of slurry after application) and the use of protected urea (i.e. urea with urease inhibitor) as a mineral fertiliser (Bittman *et al*., 2014; Buckley *et al*., 2020; Bobrowski *et al*., 2021). Besides Ireland's agriculture being dominated by livestock production which has bearings on emissions, livestock production systems are mainly grass-based, making additional mitigation difficult. Even though some NH_3 abatement techniques are available to Irish agriculture, their adoption and implementation may be limited at the farm level due to their associated cost. An assessment of NH_3 mitigation strategies and their associated costs in Ireland (i.e. Teagasc's NH_3 marginal abatement cost curve [MACC]) shows that the most cost-effective (i.e. 80% reduction) strategy for abating NH_3 in Ireland is the implementation of LESS and the substitution of urea fertilisers with protected urea (i.e. urea with the urease inhibitor N-(n-butyl) thiophosphoric triamide [NBPT]) (Buckley *et al*., 2020).

A number of studies have been conducted to evaluate the effectiveness of NH_3 abatement strategies available to Irish agriculture. To ascertain whether these NH_3 abatement strategies are indeed contributing to NH_3 emission reductions in ROI and NI, a compilation of NH_3 EFs derived from these studies is required. Furthermore, a compilation of EFs will provide information that can be used to improve national inventories and potentially identify any new abatement strategies that may be relevant to Irish agriculture. This review presents a narrative of NH_3 EFs generated from cattle production systems under Irish environmental conditions.

Methodology

Ammonia emissions data reviewed for this study were retrieved from the digital libraries of Teagasc, Ireland and the bibliographical database Scopus. Articles published in scientific journals were retrieved using the following keyword combination: cattle AND slurry OR urine OR dung AND ammonia AND emissions AND grazing OR housing OR concrete OR yards OR storage OR land AND application AND Ireland. Data search from both databases yielded 45 results. Eligibility criteria for the study selection were predefined to eliminate publication bias. That is, the study should report NH_3 emissions from cattle production systems (i.e. slurry, dung, urine and fertiliser application to grassland) during grazing, housing, storage and landspreading under Irish (both ROI and NI) environmental conditions. The studies which did not fulfil the above criteria were excluded from this review or analysis. Out of the 45 assessed for eligibility, 16 articles fulfilled our criteria and were selected.

Data reviewed in this paper relates to NH_3 measurements in which results were either reported as experimental results or NH_3 emission factors (EFs). Emission factors from the deposition of urine and dung during grazing as well as mineral fertiliser application to grasslands are presented as a percentage (%) of total nitrogen (TN). As the TN is more representative of the N present in urine and urea fertiliser, EFs from dung, urine and mineral fertilisers are expressed as %TN to facilitate comparison of EFs between these N sources. In the case of concrete yards, slurry storage pits and slurry landspreading, EFs are reported as % of total ammoniacal nitrogen (TAN) as NH_3 emissions from slurry at these stages mainly emanate from the TAN in the slurry. If EFs were already presented as %TAN, EFs were included unchanged. If EFs were presented as TN and the TAN content of the slurry was reported, EFs were converted into %TAN applied. In instances where EFs were reported as TN without any information on the slurry TAN content (i.e. Bourdin *et al*., 2014; Burchill *et al*., 2019; McIlroy *et al*., 2019), EFs were converted into %TAN based on the assumption that 60% of the TN excreted in cattle slurry is TAN (EPA, 2022). Emission factors generated from models (e.g. Cahalan *et al*., 2015) were not included in the analysis of this review.

Grazing

Urine and dung deposition

Irish agriculture is dominated by pastoral bovine livestock production, with approximately 93% of the utilisable agricultural area composed of permanent grassland (CSO, 2021). Therefore, in these grass-based production systems, cattle and sheep spend most of the year at pasture. Their excretion of urine and dung during grazing leads to emitting approximately 13.63 kt NH_3 per annum, or 10.9% of the total agricultural NH_3 emissions (EPA, 2022). This is a small proportion of overall emissions considering the extensive length of the grazing season at over 220 d per annum (Läpple *et al*., 2012), compared to 47.4% of NH_3 produced by manure management (EPA, 2022). Indeed, extending grazing length is in itself a category 1 NH_3 abatement measure based on a simple principle that quick infiltration of urine in the soil leads to lower NH_3 emissions during grazing compared to housing, storage and landspreading of manure (Bittman *et al*., 2014).

Fischer *et al*. (2016) reported NH_3 EFs in Ireland between 2.8% and 5.3% TN and between 8.7% and 14.9% TN for dung and urine, respectively, depending on the season of application (spring, summer, autumn). Surprisingly, the largest emissions were observed in spring despite lower air temperatures compared to summer, which is typically associated with higher emissions (Clay *et al*., 1990; Lockyer & Whitehead, 1990; Sommer *et al*., 1991). This clearly highlights the importance of other factors such as rainfall aiding infiltration of the excreted material. Emissions were lower from dung than urine, reflecting both a lower N application rate and a higher proportion of organic N in this excreta form. Burchill *et al*. (2017b) investigated the effect of applying urea fertiliser (with and without N stabilisers) after urine patch deposition on NH_3 emissions on two occasions (i.e. May and June). Burchill *et al*. (2017b) found no significant differences in EFs between treatments receiving the two N stabilisers (dicyandiamide [DCD] and NBPT) in May and June relative to the other treatments (i.e. urine, urea and urine plus urea). The EFs generated from urine patches in May and June were 6.3% and 7.1% TN, respectively, representing spring/summer applications. However, both of these studies used wind tunnel methodology (Lockyer, 1984) which is more suitable for replicated treatment comparisons at small scales rather than the provision of reliable data for the development of country-specific emission factors. As a result, the Irish air pollutant inventory still uses a Tier 1 NH_3 EF for grazing animals of 6% TAN (EMEP/EEA; 2019; EPA, 2022). Figure 1 shows the mean NH_3 EFs from Irish studies for cattle excreta (i.e. urine or dung) and subsequent fertilisation of urine patches with urea (with and without N stabilisers). As expected, mean EF from dung was lower than urine as most of the N in dung is usually in the organic form and needs to be converted to NH_4^+ in order to generate NH_3 emissions. The addition of DCD to urine alone or in combination with urea to urine patches tended to increase EFs relative to the urine or urine plus urea treatments. As DCD acts as a nitrification inhibitor and not a urease inhibitor, it decreases the rate of conversion of NH_4^+ to

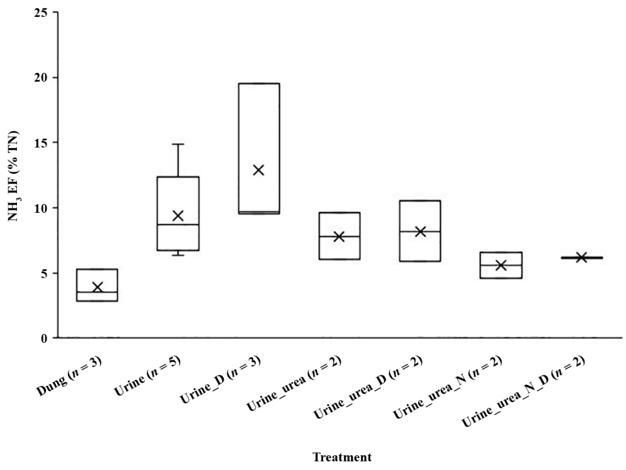

Figure 1. Mean EFs from Irish studies for cattle excreta (i.e. urine or dung) and subsequent fertilisation of urine patches with urea (with and without N stabilisers) on grassland. D = dicyandiamide (DCD); EFs = emission factors; N = N-(n-butyl) thiophosphoric triamide (NBPT).

nitrate (NO_3^-) and therefore it is not surprising that its inclusion led to increased NH_3. Such increases in NH_3 emissions after application of N stabilisers such as DCD to urea have also been observed in non-Irish studies (Prakasa Rao & Puttanna, 1987; Asing *et al*., 2008). In contrast, the addition of the urease inhibitor NBPT and NBPT plus DCD to urea after application to urine patches led to mean reductions in EFs by 28% and 21%, respectively, relative to when urea was applied to urine patches alone. The reduction in EF through the application of NBPT can be attributed to the inhibitory effect of NBPT on urea hydrolysis.

In general, Irish experimental data are comparable with or slightly lower than results from other countries. For example, Zaman *et al*. (2013) reported NH_3 EFs from urine in the range of 4.9% and 12% TN depending on the year and season of application in New Zealand, while in another New Zealand-based study, Laubach *et al*. (2013) found urine and dung emissions to be 25.5% TN and 11.6% TN, respectively. A study by Misselbrook *et al*. (2014) found NH_3 from urine patches in the UK to be 25.2% of the applied N and in the US, NH_3 EFs were between 10% and 35% TN for urine and 5% and 7% TN for faeces from pasture and shortgrass steppe rangeland (Nichols *et al*., 2018). However, reviews from Selbie *et al*. (2015) and Cai & Akiyama (2016) recorded wide ranges of NH_3 emission factors driven by animal, soil and climatic conditions as well as measurement methodology, further proving difficulty but also a need for providing a reliable basis for country-specific emission factors.

While grazing is already considered an NH_3 abatement measure, additional technologies have been suggested to further reduce emissions from this activity such as reducing crude protein intake in animal diet (Smith *et al*., 2008; Zaman & Blennerhassett, 2010), sodium chloride (NaCl) supplementation (Liu & Zhou, 2014) and use of urease inhibitors (Saggar *et al*., 2013). However, evidence is still limited regarding overall efficacy and the effects of season, soil temperature and moisture, rainfall and soil organic carbon on the variability in the efficacy and practicability of the implementation of some of these technologies. Table 1 summarises Irish studies on NH_3 EFs as %TN from the deposition/application of urine, dung and subsequent fertilisation with urea (with and without N stabilisers) on pasture. The findings from the studies show that NH_3 EFs from urine and dung deposited at pasture vary to a considerable extent owing to differences in the characteristics of excreta, environmental conditions at the time of application and limited experimental data specific to Irish conditions.

Mineral fertiliser application to grassland

The use of mineral N fertilisers to meet N demand in agricultural soils is a common practice in most parts of the world including Ireland. While the above practice is required to ensure adequate fodder for grazing animals, the practice is considered as an important source of NH_3 in Ireland accounting for 9.7% of the total NH_3 emissions (EPA, 2022). Figure 2 shows the mean EFs generated from (i) mineral fertiliser (i.e. calcium ammonium nitrate [CAN] and urea) application (with and without N stabilisers) and (ii) different timing of mineral fertiliser application to grassland. The results show that the mean EF from urea (27.9% TN) was higher than CAN (4.2% TN) which is consistent with the high availability of urea in urea fertiliser which hydrolyses rapidly to NH_3 after application. While the use of CAN often leads to lower NH_3 EFs relative to urea irrespective of the time/year of application, CAN is expected to generate higher emissions of nitrous oxide (N_2O), which is a potent greenhouse gas, compared to urea. This is due to CAN's higher nitrate content which becomes susceptible to denitrification after soil application (Harty *et al*., 2016). Results also show that besides the application of urea with maleic and itaconic acid polymer (MIP) which led to an increase in EF by 24%, the application of urea with

Table 1: Ammonia EFs as percentage (%) of TN after the deposition of cattle excreta (urine and dung), urine plus N stabilisers and urine plus urea (with or without N stabilisers) on grassland (perennial ryegrass and white clover mixture) in Ireland

Treatment	Time of application	NH_3 lost as %TN	% change in EF relative to control	Measurement method	Reference
Urine (control)	May	6.3	NA	Wind tunnel	Burchill *et al.* (2017b)
Urine + urea	May	6.0	5		
Urine + urea + DCD	May	5.9	6		
Urine + urea + NBPT	May	4.6	30		
Urine + urea + NBPT + DCD	May	6.2	2		
Urine (control)	June	7.1	NA		
Urine + urea	June	9.6	35[1]		
Urine + urea + DCD	June	10.5	48[1]		
Urine + urea + NBPT	June	6.6	7		
Urine + urea + NBPT + DCD	June	6.1	14		
Dung	April	5.3	NA	Wind tunnel	Fischer *et al.* (2016)
Urine (control)	April	14.9	NA		
Urine + DCD	April	19.5	31[1]		
Dung	July	2.8	NA		
Urine (control)	July	9.8	NA		
Urine + DCD	July	9.7	1[1]		
Dung	September	3.5	NA		
Urine (control)	September	8.7	NA		
Urine + DCD	September	9.5	9[1]		

Soil type/classification: Luvic gleysol.
DCD = dicyandiamide; EF = emission factor; NA = not applicable; NBPT = N-(n-butyl) thiophosphoric triamide; TN = total nitrogen.
[1]Represents an increase in emission factor.

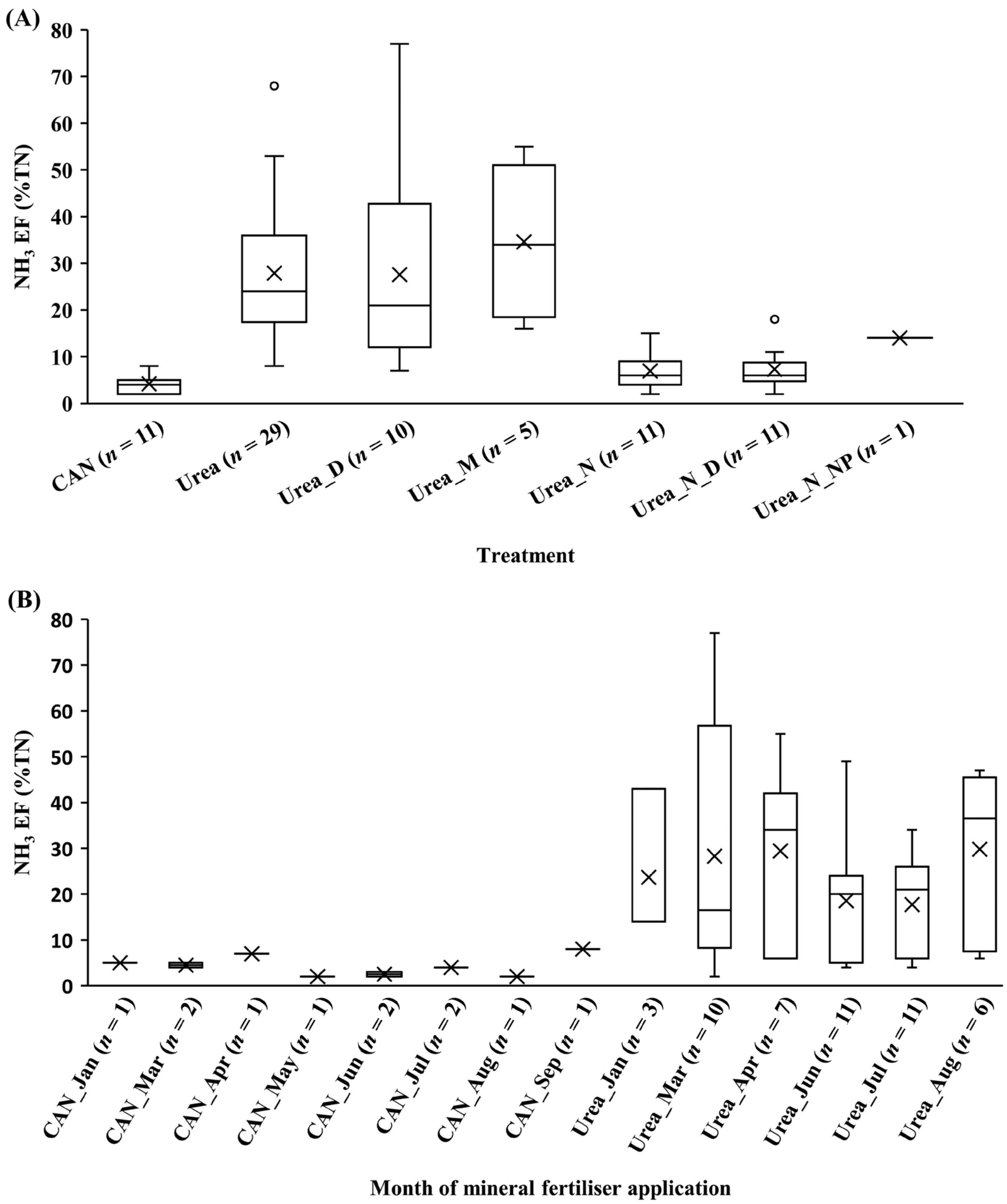

Figure 2. Ammonia emission factors after (A) mineral fertiliser (with or without N stabilisers) application to Irish grasslands and (B) timing of mineral fertiliser application. CAN = calcium ammonium nitrate; D = dicyandiamide (DCD); M = maleic and itaconic acid polymer (MIP); N = N-(n-butyl) thiophosphoric triamide (NBPT); NP = N-(n-propyl)-thiophosphoric triamide (NPPT).

N stabilisers such as NBPT, N-(n-propyl)-thiophosphoric triamide (NPPT) and the combined application of N stabilisers (i.e. NBPT + DCD, NBPT + NPPT) with urea led to a mean reduction in EF by 50% relative to urea application alone. The addition of DCD alone to urea appeared not to have any effect on EFs from the urea fertiliser. As mentioned earlier, DCD is a nitrification inhibitor and could rather increase NH_3 than decrease. In the case of the time of application, EFs for urea were generally higher than CAN irrespective of the time of application. The lowest mean EFs after urea application were obtained in the months of June and July. In the case of CAN, EFs produced from the different months or time of application were fairly similar although emissions tended to increase in the months of April and September.

The findings from the studies summarised in this section clearly indicate that the use of N stabilisers such as NBPT together with urea fertilisers can reduce N losses in the form of NH_3 and consequently increase N availability in the soil for grass uptake. In contrast, the use of DCD showed variable results relative to its impact on NH_3 EFs which agrees well with other studies (Kim *et al*., 2012). Thus, it may be assumed that the positive impact of the combined application of urea with NBPT and DCD is most likely a result of the action of NBPT in slowing urea hydrolysis through the inhibition of the urease enzyme. Even though the combined application of DCD and NBPT can have multiple benefits on the environment particularly mitigating NH_3 and subsequent N_2O emissions as well as NO_3^- leaching, the use of only NBPT may be economical to farmers if the overall goal is to abate NH_3. Table 2 summarises published literature in Ireland on NH_3 EF from mineral fertiliser (urea and CAN) application to grassland.

Cattle buildings and concrete farmyards

Housing is an essential component of animal agriculture. The primary goal for providing housing for animals is to promote good health and welfare in order to maximise productivity. In Ireland, cattle are mainly housed in naturally ventilated buildings during the winter period. In the winter season, cattle are generally housed either on straw bedding, in sheds over slatted tanks or in cubicle/loose sheds with floors scraped regularly into open tanks (Lanigan *et al*., 2015). In the straw-based systems, straw is added frequently and either allowed to accumulate or removed frequently as farmyard manure. The effluents (i.e. mixture of urine, spilled water, faeces, food particles, etc.) collected below the slatted floor are stored as slurry. The slurry-based systems (i.e. systems where effluents are collected below slatted tanks) predominate the dairy sector with about 94% of dairy housing in Ireland being slurry based (McIlroy *et al*., 2019). Cattle buildings are important sources of NH_3 emissions. Indeed, manure management from cattle (i.e. dairy and non-dairy) generates 50 kt NH_3 per annum which represents about 40% of national total emissions in Ireland (EPA, 2022).

Table 3 summarises NH_3 EFs generated from cattle buildings and concrete yards used by cattle under Irish conditions. Emissions from cattle buildings can vary to a considerable extent due to variations in factors such as building design/size, number of cattle and wind velocity/direction (Burchill *et al*., 2017a). Burchill *et al*. (2017a) quantified NH_3 EFs from four cattle buildings which varied in size, floor type as well as the type and number of livestock housed. The mean EF reported from the study was 13% TAN. The overall mean EF of 13% TAN reported in the Irish study is somewhat lower than the applied EF of 28% TAN in liquid or slurry-based housing in the Irish NH_3 inventory (EPA, 2022). The above observation suggests an over-estimation of EFs used in the Irish NH_3 inventory. There is therefore the need to generate additional datasets on NH_3 emissions from cattle buildings to validate the findings from Burchill *et al*. (2017a). In the case of concrete farmyards or hard standings, mean EFs generated from the analysis of this review was 35% TAN.

A number of strategies have been proposed as being effective at abating NH_3 from cattle buildings or sheds and concrete farmyards. These strategies include regular cleaning of sheds or yards, adsorption of urine using materials such as straw and the use of chemical additives such as urease inhibitors (Burchill *et al*., 2019; Bobrowski *et al*., 2021). Ammonia abatement techniques such as application of chemical additives and regular cleaning of yards/shed floors by washing with water or scraping have been evaluated under Irish conditions. Burchill *et al*. (2019) reported a reduction in EFs of up to 89% after cleaning (i.e. either pressure washing with water or scraping) of excreta. Greater reductions in EFs were noted when cleaning was done immediately (i.e. 1 h post deposition/application) after excreta deposition than later (i.e. 3 h post deposition/application). Similar observations were made in other studies conducted in the UK (Misselbrook *et al*., 1998, 2001) and were attributed to the removal of the N source (urine and dung) from the emitting surface area after cleaning. In the case of chemical additives, McIlroy *et al*. (2019) reported NH_3 reductions of 67%, 58%, 54% and 33% after application of alum, calcium chloride, actisan (disinfectant) and sulphuric acid, respectively, from slurry applied to a concrete surface under typical Northern Irish cattle housing conditions.

Storage pits

Slurry storage is an important step in managing manure as it allows slurry to be kept until conditions are suitable for landspreading. However, a substantial proportion of slurry N is emitted as NH_3 during storage, which reduces the amount of readily available N for plant uptake after landspreading. Research findings on EFs from slurry storage are summarised in Table 4. The mean EF for the control or un-amended slurries was 60% TAN and values ranged between 42–68% TAN. The mean EF (60% TAN) obtained in the present analysis can be considered higher than the EF of 5% and 28% TAN used by the Irish EPA in the Informative Inventory Report for covered slurry during storage and slurry-based cattle houses, respectively (EPA, 2022). It is of note that these studies (Table 4) were small-scale incubations measuring NH_3 in a dynamic chamber mode so the airflow is expected to be higher than above a typical tank in the shed. Thus, the values reported in the studies in Table 4 likely overestimate emissions and cannot be used to provide reliable EFs from storage, but rather

Table 2: Ammonia EFs as percentage (%) of TN applied from urea and calcium ammonium nitrate (CAN) application to grassland

Fertiliser type	Soil type/ classification	Time of application	NH_3 Lost as % TN	% change in EF relative to control	Measurement method	Reference
Urea	Haplic Cambisol	September	17	NA	Wind tunnel	Burchill *et al.* (2016)
Urea	Albic Gleyic	September	21	NA		
Urea	Haplic Cambisol	September	14	NA		
Urea	Cutanic Luvisol	September	22	NA		
Urea	Luvic Stagnosol	September	18	NA		
Urea	Luvic Gleysol	September	13	NA		
Urea	Luvic Gleysol	May	8	NA	Wind tunnel	Burchill *et al.* (2017b)
Urea	Luvic Gleysol	June	17	NA		
CAN	MDSL	March	5	NA	Wind tunnel	Forrestal *et al.* (2016)
Urea (control)		March	53	NA		
Urea – high rate		March	68	28[1]		
Urea + NBPT		March	15	72		
Urea + DCD		March	77	45[1]		
Urea + NBPT + DCD		March	18	66		
Urea + MIP		April	55	4[1]		
CAN		April	7	NA		
Urea (control)		April	25	NA		
Urea – high rate		April	38	52[1]		
Urea + NBPT		April	6	76		
Urea + DCD		April	42	68[1]		
Urea + NBPT + DCD		April	6	76		
Urea + MIP		April	34	36[1]		
CAN		June	2	NA		
Urea (control)		June	20	NA		
Urea – high rate		June	24	20[1]		
Urea + NBPT		June	5	75		
Urea + DCD		June	22	10[1]		
Urea + NBPT + DCD		June	4	80		
Urea + MIP		June	16	20		
CAN		July	4	NA		
Urea (control)		July	26	NA		
Urea – high rate		July	24	8		
Urea + NBPT		July	9	65		
Urea + DCD		July	22	15		
Urea + NBPT + DCD		July	5	81		
Urea + MIP		July	21	19		
CAN		August	2	NA		
Urea (control)		August	30	NA		
Urea – high rate		August	43	43[1]		
Urea + NBPT		August	6	80		
Urea + DCD		August	45	50[1]		

Table 2: (continued)

Fertiliser type	Soil type/ classification	Time of application	NH_3 Lost as % TN	% change in EF relative to control	Measurement method	Reference
Urea + NBPT + DCD		August	8	73		
Urea + MIP		August	47	57[1]		
CAN	IDCL	March	4	NA		
Urea (control)		March	21	NA		
Urea – high rate		March	14	33		
Urea + NBPT		March	2	90		
Urea + DCD		March	9	57		
Urea + NBPT + DCD		March	6	71		
CAN		May	2	NA		
Urea (control)		May	8	NA		
Urea – high rate		May	19	138[1]		
Urea + NBPT		May	2	75		
Urea + DCD		May	7	13		
Urea + NBPT + DCD		May	2	75		
CAN		June	3	NA		
Urea (control)		June	33	NA		
Urea – high rate		June	49	48[1]		
Urea + NBPT		June	4	88		
Urea + DCD		June	20	39		
Urea + NBPT + DCD		June	7	79		
CAN		July	4	NA		
Urea (control)		July	31	NA		
Urea – high rate		July	34	10		
Urea + NBPT		July	4	87		
Urea + DCD		July	13	58		
Urea + NBPT + DCD		July	6	81		
CAN		September	8	NA		
Urea (control)		September	33	NA		
Urea – high rate		September	43	30[1]		
Urea + NBPT		September	9	73		
Urea + DCD		September	19	42		
Urea + NBPT + DCD		September	11	67		
CAN	MDSL	July	5	NA	Wind tunnel	Krol *et al.* (2020)
Urea (control)		July	43	NA		
Urea + NBPT		July	14	67		
Urea + NBPT + NPPT		July	14	67		
Urea		July	47	NA	IHF method	

CAN = calcium ammonium nitrate; DCD = dicyandiamide; EFs = emission factors; IDCL = imperfectly drained clay loam; IHF = integrated horizontal flux; MDSL = moderately-drained sandy loam; MIP = maleic and itaconic acid polymer; NA = not applicable; NBPT = N-(n-butyl) thiophosphoric triamide; NPPT = N-(n-propyl)-thiophosphoric triamide; TN = total nitrogen.
[1]Values represent an increase in EF.

Table 3: Ammonia EFs as percentage (%) of TAN from cattle buildings and hard standings or concrete yards

Treatment	NH_3 lost (%TAN)	% change in EF relative to control	Measurement method	Reference
Cattle buildings	13	NA	Passive flux samplers (Ferm tubes)	Burchill *et al.* (2017a)
Concrete standings				
Slurry (control)	24	NA	Dynamic flow-through chamber	McIlroy *et al.* (2019)
Slurry + alum	8	67		
Slurry + calcium chloride	10	58		
Slurry + Actisan	11	54		
Slurry + sulphuric acid	16	33		
Slurry + Agrotain	22	8		
Slurry + Envirobed	23	4		
Slurry + double UI	21	13		
Slurry + clinoptilotite	21	13		
Slurry + eugenol	24	0		
Slurry + sawdust	22	8		
1 kg dung + 0.67 L urine	36	NA	Wind tunnel	Burchill *et al.* (2019)
1 kg dung + 1 L urine	42	NA		
1 kg dung + 2 L urine	48	NA		
Excreta without pressure wash (control)	36	NA		
Excreta + pressure wash 1 h post application	4	89		
Excreta + pressure wash 3 h post application	7	81		
Excreta without scraping (control)	46	NA		
Excreta + scraping 1 h post application	9	80		
Excreta + scraping 3 h post application	22	8		

EFs = emission factors; NA = not applicable; TAN = total ammoniacal nitrogen.

provide an indication of the efficacy of abatement strategies. Future studies should aim to measure emissions at a scale representative of current farming practice.

Regarding NH_3 abatement measures, the application of chemical amendments (i.e. sulphuric acid, acetic acid, alum and ferric chloride) to slurry during storage delivered the greatest emission reduction of up to 98%. The amendments of slurry with industrial and agricultural waste and by-products (e.g. brewers grain, apple pulp, maize silage effluent, sugarbeet molasses) also showed promising NH_3 abatement potential. However, greater reductions in EFs were observed through slurry acidification with commercial acids (i.e. acetic acid, sulphuric acid) and other acidifying additives (i.e. alum and ferric chloride) relative to the industrial/agricultural wastes and by-products. This is possibly due to the direct effect of acidification and chemical additives in reducing slurry pH. The impact of industrial waste products appeared to be influenced by their inclusion rate. On the whole, the greater the inclusion rate, the greater the NH_3 emission reduction potential.

Treatment of slurry using commercial additives generally tended to increase NH_3 emissions. Even though the mode of action and the composition of commercial additives are often not revealed by manufacturers, the proposed benefits of these commercial additives are often attributed to the role they play in stimulating biological processes and altering the chemical composition (i.e. TAN, NH_4^+ and pH) in slurries/manures. In the case of biological additives (i.e. microbial-based), NH_3 mitigation may be achieved either through (i) stimulation of NH_4^+ immobilisation which reduces NH_4^+ concentration in slurry or (ii) enhancement of fermentation which reduces slurry pH through the formation of organic acids. While chemical additives, typically acidifiers (e.g. sulphuric acid, hydrochloric acid) mitigate NH_3 losses by directly inducing a decline in pH, physical additives (e.g. biochar, clinoptilotite, peat) on the other hand act by adsorbing NH_4^+ onto their surfaces.

Although important reductions in NH_3 emissions have been reported in the literature with these additives during slurry storage, there is also evidence of either no effect (Matulaitis

Table 4: Ammonia EFs as percentage (%) of TAN from slurry storage pits in cattle houses

Treatment	Dry matter (%)	Inclusion rate of additive	NH_3 lost (%TAN)	% change in EF relative to control	Storage period (days)	Measurement method	Reference
Storage							
Slurry (control)	4	—	68.0	NA	83	Dynamic chamber	Kavanagh *et al.* (2019)
Slurry + acetic acid	4	35.7 g	22.0	68	83		
Slurry + alum	4	53.1 g	6.0	91	83		
Slurry + ferric chloride	4	41.3 g	1.4	98	83		
Slurry + sulphuric acid	4	20.8 g	10.0	85	83		
Slurry (control)	7	—	63.1	NA	83		
Slurry + acetic acid	7	43.5 g	12.9	80	83		
Slurry + alum	7	88.5 g	8.7	86	83		
Slurry + ferric chloride	7	70.1 g	1.6	97	83		
Slurry + sulphuric acid	7	19.2 g	8.9	86	83		
Slurry (control)	7	—	42.2	NA	70	Dynamic chamber	Kavanagh *et al.* (2021)
Slurry + apple pulp	7	7.0%	22.7	46	70		
Slurry + apple pulp	7	15.0%	29.2	31	70		
Slurry + brewers grain	7	7.0%	33.3	21	70		
Slurry + brewers grain	7	15.0%	29.8	29	70		
Slurry + dairy washings	7	7.0%	47.2	12	70		
Slurry + dairy washings	7	15.0%	39.5	6	70		
Slurry + dairy waste	7	10.0%	27.4	35	70		
Slurry + grass silage	7	15.0%	23.9	43	70		
Slurry + grass silage	7	7.0%	32.4	23	70		
Slurry + maize silage	7	15.0%	34.6	18	70		
Slurry + maize silage	7	7.0%	61.3	45	70		
Slurry + sugarbeet molasses	7	3.0%	25.1	41	70		
Slurry + sugarbeet molasses	7	5.0%	11.3	73	70		
Slurry + sugarbeet molasses	7	7.0%	40.5	4	70		
Slurry (control)	7	—	66.1	NA	116		
Slurry + commercial additive 1	7	RA	60.7	8	116		
Slurry + commercial additive 2	7	RA	66.2	0	116		
Slurry + commercial additive 3	7	RA	67.0	1[1]	116		
Slurry + commercial additive 4	7	RA	61.2	7	116		
Slurry + ferric chloride	7	0.4	47.2	23	116		
Slurry + ferric chloride	7	0.9	43.9	7	116		
Slurry + ferric chloride	7	1.1	20.2	54	116		

EFs = emission factors; NA = not applicable; RA = recommended application rate; TAN = total ammoniacal nitrogen.
[1]Values represent an increase in EF.

et al., 2013; Provolo *et al.*, 2016; Owusu-Twum *et al.*, 2017) or some increases (Van der Stelt *et al.*, 2007; Wheeler *et al.*, 2010) in NH_3 emissions after application of some additives. Studies (Wheeler *et al.*, 2010; Matulaitis *et al.*, 2013; Brennan *et al.*, 2015) also reveal that the use of these additives might lead to an increase in other pollutant gases or pollution swapping during storage or after land application which can undermine the sustainability of their use. Therefore, commercial additives should be used cautiously and based on empirical evidence that their use is not accompanied with any negative consequences on the environment.

In addition to the use of slurry amendments and acidification, emissions from slurry storage can also be reduced through slurry management and engineering solutions. Practices with the potential to abate emissions include covering slurry storage tanks, inducing or allowing natural crust formation on slurry surfaces, reducing disturbances in storage tanks such as aeration or mixing, reducing the length of housing to reduce the surface area-to-volume ratio and spreading earlier in the year to reduce manure storage length (Kupper *et al*., 2020).

Slurry landspreading

Landspreading of slurry is a practice common in livestock production systems and offers an opportunity to reutilise the nutrients present in the slurry. In areas of intensive livestock production, slurry nutrients can be used to supply all or a substantial amount of the nutrients needed on farms and consequently reduce the share of mineral fertilisers utilised in agriculture. The most common slurry-spreading technique in Ireland is broadcast application via splash plate although other techniques such as band spreading/trailing hose, trailing shoe and injection have been introduced as low-emission spreading techniques. Spreading of cattle slurry generates almost 37 kt of NH_3 per annum, representing approximately 30% of NH_3 emissions in Ireland (EPA, 2022). Figure 3A shows the mean EFs generated from low-emission spreading strategies (LESS) and the splashplate technique. The mean EFs generated for the LESS and splashplate were 50% TAN and 63% TAN, respectively. Therefore, the splashplate EF obtained in this meta-analysis is similar to that used by the Irish EPA in the NH_3 inventory (64% TAN). Even though the use of LESS led to reductions of approximately 21% in EFs relative to the splashplate, greater reductions were expected. Indeed, the NH_3 abatement potential used by the Irish Informative Inventory prepared by the EPA for LESS methods such as bandspreading and trailing shoe are 30% and 60%, respectively. The lower reductions observed in the presented meta-analysis may be attributed to (i) low level of data availability, (ii) high variability of results and (iii) methodology used to obtain data, which was based mostly on small-scale plot work with hand-simulated slurry applications. Therefore, there is a need to conduct further studies to generate data from large-scale plots evaluating LESS techniques such as bandspreading and trailing shoe using standard farm machinery in order to validate the findings of this review.

Figure 3B shows EFs generated from the splashplate and LESS in the spring (March, April and May), summer (June, July and August) and autumn (October). As expected, EFs were highest in the summer months relative to the spring and autumn months irrespective of the slurry application method used, which confirms the positive correlation between NH_3 emissions and ambient temperature. The EFs generated using the splashplate were 60% TAN, 70% TAN and 52% TAN for spring, summer and autumn seasons, respectively. In the case of LESS, EFs generated in the spring and summer seasons were 48% TAN and 64% TAN, respectively. There is no apparent information regarding EFs generated using the LESS in the autumn season in Ireland and this needs to be considered in future studies. The above observation indicates the potential to reduce EFs to a great extent by applying slurries in cool temperature conditions rather than in warm weather, ideally accompanied by LESS.

Table 5 summarises NH_3 EFs after landspreading of slurry under Irish conditions. Studies which assessed NH_3 abatement potential of LESS methods such as bandspreading and trailing shoe reported emission reduction of up to 45% relative to the splashplate. While the practice of reducing slurry pH either through acidification or the use of chemical amendments led to a reduction in EFs of up to 96% relative to the untreated slurry, slurry separation with sieves on the other hand led to emission reduction of up to 67% relative to the un-sieved or un-separated slurry. Slurry separation results in a low DM slurry which facilitates slurry infiltration after soil application and consequently reduces NH_3 emissions (Owusu-Twum *et al*., 2017). The reduction in EFs in the sieved or separated slurries can be related to enhanced infiltration of slurry in low DM slurries relative to high DM slurries even though some contradictory evidence, particularly regarding the spreading of low DM digestate, has been observed by Pedersen *et al*. (2021). In addition, the application of slurry in the evening using the splashplate and trailing shoe led to a mean reduction in EFs of 70% and 67%, respectively, relative to the daytime. This observation can be attributed to the lower solar radiation and less wind at night. In general, NH_3 emission factors obtained after slurry landspreading from Irish-based studies vary to a great extent with values ranging from 0.8% to 108% of TAN applied (Table 5). These variations can be attributed to variations in factors such as slurry N application rates, slurry DM content, treatment techniques, season/time of application, slurry-spreading technique, soil type and environmental conditions at the time of application.

Synthesis and challenges

Analysis of the data summarised in this paper indicates that in storage, manure amendments, particularly those that induce a decline in slurry pH, are effective at abating NH_3 emissions considerably. Waste products generated from on-farm practices and food processing such as silage effluent

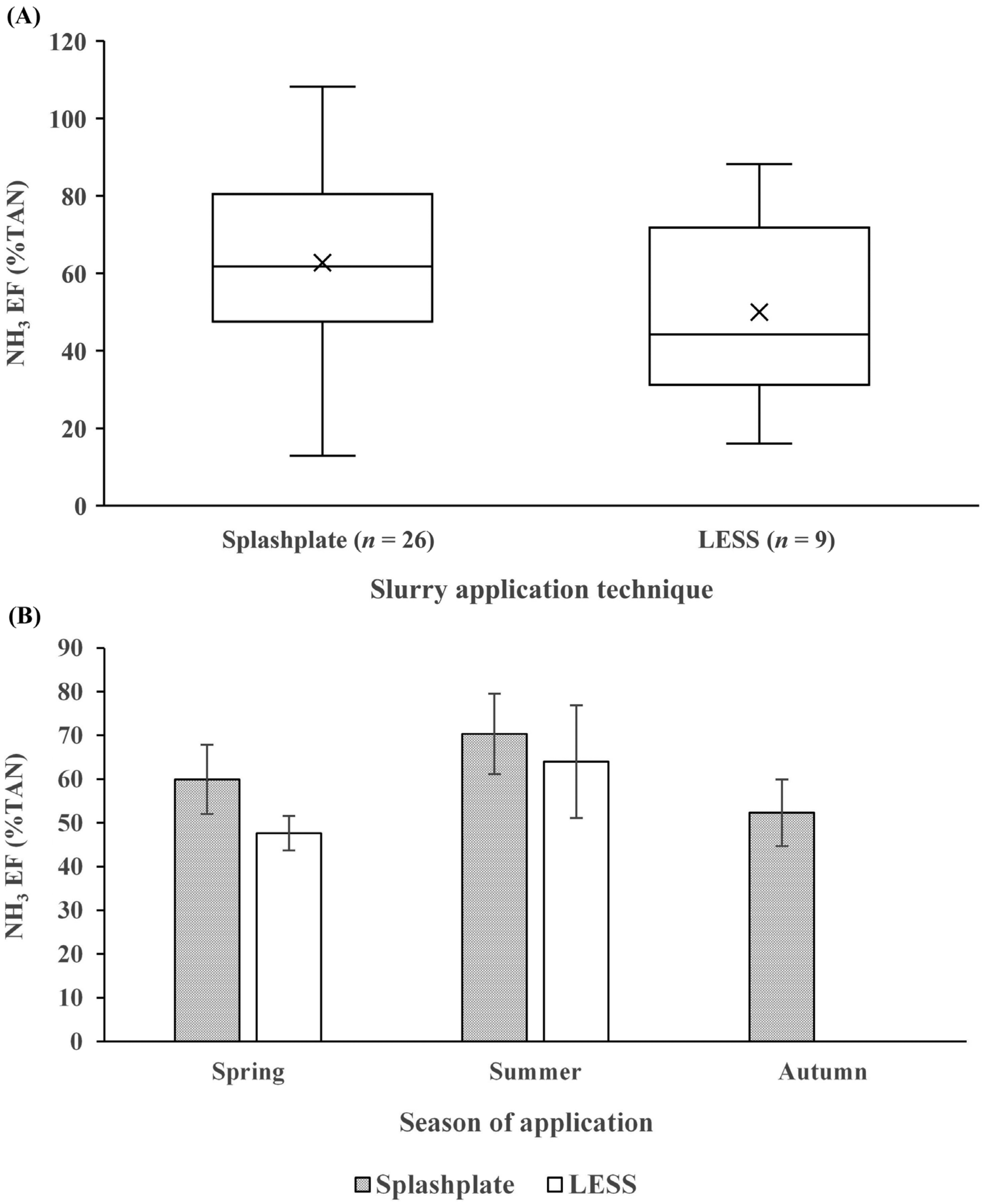

Figure 3. Ammonia EFs for (A) slurry application techniques and (B) slurry application techniques at different seasons of application. EFs = emission factors. Spring = March, April and May. Summer = June, July and August. Autumn = October. LESS = Low-emission spreading strategies. Bars represent standard errors of the mean.

and dairy washings may also be used as alternatives to chemical acids to reduce slurry pH (through the formation of lactic acid) and consequently reduce NH_3 emissions. While the use of such wastes may support the circular economy concept, their use may have limited benefits in terms of mitigating greenhouse gas emissions (Kavanagh *et al*., 2021). Furthermore, commercial additives reviewed in this study did not show an ability to reduce NH_3 emissions as observed in other studies (Matulaitis *et al*., 2013). Hence, the selection and use of commercial additives and waste amendments should be done based on scientifically proven evidence that (i) significant reductions in NH_3 emissions will

Table 5: Ammonia EFs as percentage (%) of TAN applied from slurry-spreading techniques in Ireland

Mitigation strategy	Spreading method	Time of application	Slurry type	Dry matter (%)	Soil texture	Sward height (high/short)	NH_3 Loss (% TAN)	% change in EF relative to control	Measurement method	Reference
1. Application technique and time of application	SP (C)	Day spreading	Cattle	7.6	Loam	Short grass	54.5	NA	IHF	Dowling (2012)
	TS	Day spreading		7.6		Short grass	35.0	36		
	SP (C)	Day spreading		7.6		High grass	49.6	NA		
	TS	Day spreading		7.6		High grass	27.5	45		
	SP (C)	Evening spreading		7.6		Short grass	24.5	NA		
	TS	Evening spreading		7.6		Short grass	16.0	35		
	SP (C)	12 July 2006		8.3	Loam	Short grass	61.5	NA	IHF	Dowling *et al.* (2008)
	TS	12 July 2006		8.3		Short grass	44.2	28		
	SP (C)	14 May 2007		8.3		Short grass	73.1	NA		
	TS	14 May 2007		8.3		Short grass	43.7	40		
	SP (C)	March	Cattle	6.0	Loam		68.0	NA	ALFAM model	Cahalan *et al.* (2015)
	BS	March		6.0			39.3	42		
	SP (C)	June		7.1			72.3	NA		
	BS	June		7.1			41.8	42		
	SP (C)	October		5.7			55.6	NA		
	BS	October		5.7			32.0	42		
	SP (C)	March		7.6			62.0	NA		
	BS	March		7.6			35.7	42		
	SP (C)	June		7.1			104.7	NA		
	BS	June		7.1			60.5	42		
	SP (C)	October		6.5			63.5	NA		
	BS	October		6.4			36.7	42		
	SP	March		5.6	Sandy clay loam		35.9	NA		
	SP	June		5.9			51.2	NA		
	SP	October		3.2			37.8	NA		
	SP (C)	April	Grass fed	7.4 (HDM)	Well-drained coarse loam	NA	58.0	NA	IHF	Bourdin *et al.* (2014)
	SP (C)	April	Grass fed	3.8 (LDM)			84.2	NA		
	SP (C)	April	Maize fed	7.3 (HDM)			41.3	NA		
	SP	April	Maize fed	4.0 (LDM)			79.2	NA		
	TS	April	Maize fed	4.5 (LDM)			51.6	21		

Table 5: (continued)

Mitigation strategy	Spreading method	Time of application	Slurry type	Dry matter (%)	Soil texture	Sward height (high/short)	NH_3 Loss (% TAN)	% change in EF relative to control	Measurement method	Reference
	SP (C)	July	Grass fed	7.4 (HDM)			108.2	NA		
	SP (C)	July	Maize fed	7.6 (HDM)			93.5	NA		
	TS	July	Maize fed	4.8 (LDM)			88.2	13		
	SP (C)	August	Grass fed	5.3 (HDM)			54.1	NA		
	SP (C)	August	Grass fed	2.1 (LDM)			103.0	NA		
	SP (C)	August	Maize fed	6.0 (LDM)			63.7	NA		
	SP (C)	August	Maize fed	3.3 (LDM)			95.8	NA		
	TS	August	Maize fed	3.5 (LDM)			59.6	25		
2. Chemical amendment	BS				Sandy loam	NA			DC + AT	Brennan *et al.* (2015)
Slurry (C)				10.1			84.0	NA		
Alum				9.4			3.0	96		
$FeCl_2$				10.1			18.0	79		
Lime				8.2			40.0	52		
Poly aluminium chloride (PAC)				9.6			13.0	85		
3. Separation and acidification	SP		Cattle	NA	Clay loam	NA			DC + AT	Frost *et al.* (1990)
Slurry (C)		9 May					19.1	NA		
SS sieved 5 mm		9 May					13.4	30		
SS sieved 3 mm		9 May					11.5	40		
SS sieved 0.25 mm		9 May					9.9	48		
SS sieved 0.015 mm		9 May					6.7	64		
Acidified slurry (C)		9 May					1.7	NA		
Acidified SS sieved 5 mm		9 May					2.3	35[1]		
Acidified SS sieved 3 mm		9 May					3.5	106[1]		
Acidified SS sieved 0.25 mm		9 May					1.6	6		
Acidified SS sieved 0.015 mm		9 May					1.5	12[1]		
Slurry (C)		27 June					23.1	NA		
SS sieved 5 mm		27 June					23.7	3[1]		
SS sieved 3 mm		27 June					21.8	6		
SS sieved 0.25 mm		27 June					15.2	34		
SS sieved 0.015 mm		27 June					13.7	41		
Acidified slurry (C)		27 June					5.5	NA		

Table 5: (continued)

Mitigation strategy	Spreading method	Time of application	Slurry type	Dry matter (%)	Soil texture	Sward height (high/short)	NH_3 Loss (% TAN)	% change in EF relative to control	Measurement method	Reference
Acidified SS sieved 5 mm		27 June					5.3	4		
Acidified SS sieved 3 mm		27 June					5.8	5[1]		
Acidified SS sieved 0.25 mm		27 June					3.0	45		
Acidified SS sieved 0.015 mm		27 June					3.7	33		
Slurry (C)		15 August					12.9	NA		
SS sieved 5 mm		15 August					6.9	47		
SS sieved 3 mm		15 August					7.4	43		
SS sieved 0.25 mm		15 August					4.2	67		
SS sieved 0.015 mm		15 August					1.1	91		
Acidified slurry (C)		15 August					1.9	NA		
Acidified SS sieved 5 mm		15 August					1.8	5		
Acidified SS sieved 3 mm		15 August					1.3	32		
Acidified SS sieved 0.25 mm		15 August					1.3	32		
Acidified SS sieved 0.015 mm		15 August					0.8	58		

Control treatments are assigned the letter C in brackets.
ALFAM = ammonia loss from field-applied manure; AT = acid trap; DC = dynamic chamber; EFs = emission factors; HDM = high dry matter; IHF = integrated horizontal flux method; LDM = low dry matter; NA = not applicable; PA = phosphoric acid; SP = splashplate; SS = separated slurry; TAN = total ammoniacal nitrogen; TS = trailing shoe; TN = total nitrogen.
[1]Values represent an increase in EF.

be guaranteed and (ii) the additives will not lead to pollution swapping during storage or at a different stage in the manure management continuum. This review also highlights that the practice of keeping concrete yards used by cattle clean from excreta particularly through pressure washing with water at short intervals should be encouraged on farms as the above practice has the potential to reduce emissions greatly from concrete yards used by cattle (Burchill *et al*., 2019).

Regarding the use of mineral fertiliser, our analysis shows that NH_3 emissions from the application of urea fertilisers to grasslands can be reduced appreciably through the use of urease inhibitors. Indeed, Teagasc's NH_3 MACC shows that the use of urease inhibitors such as NBPT can provide a reduction in national NH_3 emissions of approximately 20%. Similarly, the addition of urease inhibitor to urea is considered the most feasible solution to abate NH_3 from urea in other countries such as Germany (Hu & Schmidhalter, 2021). The UK has proposed an even stronger NH_3 abatement solution in the form of a legislative ban on urea fertilisers (DEFRA, 2020). However, a ban on urea fertilisers implies that farmers may have to resort to other forms of mineral fertilisers such as ammonium nitrate. A shift to ammonium nitrate fertilisers will also pose a threat to the environment as ammonium nitrate fertilisers could stimulate the release of N_2O and increase the risk of nitrate leaching. Nitrous oxide emissions from mineral fertiliser usage particularly from urea can be curtailed through the addition of urease inhibitors.

The adoption of abatement strategies upstream in manure management, such as covering slurry stores, slurry acidification and treatment of slurry by amendments during slurry storage will result in more N being retained in the slurry. High levels of slurry N imply that more N could be lost as NH_3 downstream or during landspreading of slurry. Therefore, low-emission spreading strategies (LESS) will play a vital role in abating NH_3 emissions in animal agriculture. Even though EFs summarised in the present study for slurry application techniques varied to some extent, Irish studies which compared EFs from LESS and the conventional splashplate showed that appreciable reduction in EFs can be achieved by substituting the splashplate technique with LESS such as trailing shoe and band spreading. This review highlights the positive impact of LESS in abating NH_3 EFs is consistent with Teagasc's NH_3 MACC, which shows that the implementation of LESS can deliver a reduction in national NH_3 emissions of approximately 60%. In France, it is estimated that slurry injection and direct incorporation of slurry will contribute to 60% of the total abatement potential (Mathias & Martin, 2013). Similarly, low-emission manure application is expected to form approximately 60% of the total technical abatement potential in Germany (Wulf *et al*., 2017). Research conducted under Irish conditions shows that LESS does not only have the potential to reduce NH_3 but also increases the fertiliser value of slurry, if applied at the right time (Lalor *et al*., 2011). Thus, the implementation of LESS should be a sustainable practice since its adoption could reduce the demand for mineral fertilisers and consequently reduce the proportion of greenhouse gas emissions associated with the production of mineral fertilisers. The reduction in the use of mineral fertilisers such as urea fertilisers could further reduce NH_3 emissions.

This review showed that important reductions in EFs after landspreading can be achieved within application timing management (Table 5). For instance, Dowling (2012) reported appreciable reductions in EFs after evening spreading of slurry relative to daytime spreading under Irish conditions, irrespective of the slurry application method (splashplate and trailing shoe). Similarly, studies conducted in France and Denmark show that slurry application between evening and early morning has the potential to reduce emissions by up to 50% (Moal *et al*., 1995; Sommer & Olesen, 2000). The lower reductions after evening spreading relative to daytime spreading can be attributed to factors such as higher humidity, lower temperatures and lower wind velocity at night. These findings indicate that additional reductions in EFs can be achieved within application timing management without imposing high technology adoption costs on farmers.

Prediction of NH_3 emissions in Ireland using the ammonia loss from field-applied animal manure (ALFAM) model shows higher emissions in the summer months relative to spring, autumn and winter (Lalor & Lanigan, 2010). Irish-based studies summarised in this review show that considerable emissions can also occur in the spring months after slurry landspreading and urea fertiliser application. Indeed, environmental conditions which stimulate NH_3 emissions such as high wind speed and solar radiation have been observed in the spring months in Ireland. Thus, besides the restrictions in the time/month of slurry application in Ireland, the management of slurry and mineral fertiliser applications based on weather conditions within each month will also be crucial in reducing NH_3 emissions in Irish livestock production systems.

A major challenge associated with NH_3 abatement in livestock production is the already mentioned potential increase in NH_3 emissions downstream after the implementation of an abatement strategy upstream. Besides the adoption of LESS, the above challenge can be mitigated by reducing the amount of N excreted from livestock through a reduction in the crude protein content in livestock diets. According to Bittman *et al*. (2014), a percentage point reduction in the protein content of animal feed content results in total NH_3 emission (from animal housing, manure storage and the application to land) reduction by 5–15% as a result of the reduced ammoniacal N in the manure produced. The amount of N excreted from livestock can also be reduced through genetic improvements, particularly the selection of livestock with high feed efficiency (De Verdal *et al*., 2013).

Another challenge is the pollution swapping potential of NH_3 abatement strategies. Emmerling *et al*. (2020) used a meta-analysis to evaluate the impact of major agricultural management practices on NH_3 and its pollution swapping effect. The result showed that for treatment, storage and application stages, only slurry acidification was effective at reducing NH_3 emissions with no pollution swapping effect with greenhouse gases such as N_2O, methane (CH_4) and carbon dioxide (CO_2). Even though the rest of the management techniques evaluated in the meta-analysis such as slurry separation, different storage types, biological treatment and variable field applications were also effective at abating NH_3 at varying degrees, they led to an increase in the emission of at least one other greenhouse gas. These findings indicate that the issue of pollution swapping could be addressed through slurry acidification or by implementing a combination of abatement strategies.

While LESS is the main strategy for abating NH_3 in Ireland, slurry acidification is the preferred technology for reducing NH_3 from the animal sheds to the landspreading stage in Denmark. Danish regulation of nutrients in agriculture requires high levels of nitrogen-use efficiency from slurry (MEFD, 2017), thus making it important to conserve ammoniacal nitrogen in the slurry before application to land. Slurry acidification is rarely used in Ireland and other EU countries owing to issues such as its associated cost as well as health and safety and soil health concerns.

The adoption of LESS or other landspreading abatement technologies may be limited on Irish farms due to its associated cost. This may be the case, particularly for farmers in low-income categories who have already invested in splashplate systems. The provision of incentives especially in the form of subsidies to farmers in the above category to cover the cost of implementation of the technology can facilitate the adoption of LESS on Irish farms. According to the EPA (2022), the adoption rate of LESS increased from 3% in the year 2015 to 36% in the year 2020. The adoption rate of LESS is expected to further increase in the coming years due to the restrictions imposed by the current Nitrates Action Programme in Ireland which requires all farmers operating above 170 kg N/ha to use LESS on their farms.

For further studies

Although a number of studies have been conducted on NH_3 mitigation strategies under Irish climatic conditions, there is the need to investigate novel strategies for abating NH_3 emissions to add to the suite of options already available to Irish agriculture.

Most of the Irish studies which quantified NH_3 emissions from slurry landspreading were conducted using small-scale plots with hand-simulated slurry applications. Additional studies are required to generate data from large-scale plots evaluating different slurry-spreading techniques available to Irish agriculture using standard farm machinery in order to validate the findings of this review.

The application of urease inhibitors to manure or urea-based mineral fertilisers is regarded as a potent strategy to abate NH_3 emissions in agriculture. Although protected urea offers a low NH_3 emission alternative to straight urea (Forrestal *et al*., 2016; Krol *et al*., 2020), the potential to further utilise urease inhibitors to reduce emissions from livestock wastes has not yet been made commercially feasible (Sigurdarson *et al*., 2018). The reason for the limited adoption in animal agriculture is attributed to factors such as the absence of efficient and automated application systems, the need for reapplication, limited chemical stability of the inhibitors and the associated cost to the farmer (Sigurdarson *et al*., 2018). Further studies are required to improve the above aspects to facilitate the adoption of urease inhibitors on a commercial scale.

Slurry separation is used in some countries as a tool to improve manure management on farms. The technique is often employed in situations where there is the need to export slurry nutrients from farms with excess nutrients to those in deficit of slurry nutrients. The technique also facilitates the transport of organic matter in slurry over long distances to large/central biogas plants. Research shows that the application of the low DM liquid fraction obtained after separation to soil can enhance infiltration of slurry and TAN and consequently minimise NH_3 losses (Owusu-Twum *et al*., 2017). Unlike other European countries such as Spain, Italy and Denmark where the technique has been studied to a great extent, there is limited research findings on the impact of the above technique in Irish agricultural production systems. Slurry separation could offer a cost-effective approach to improve manure management and facilitate the production of biogas which is still in its infant stages in Ireland. An assessment of the environmental (i.e. NH_3, greenhouse gas emissions and nutrient leaching), economic and agronomic consequences of the above technique under Irish agricultural production systems is necessary in order to recommend the technique to farmers.

While anaerobic digestion (AD) is regarded as an environmentally friendly approach for reutilising the DM present in manure, the resulting effluents obtained after the AD process (i.e. digestate) might lead to increased NH_3 emissions either during storage and/or landspreading due to the increase in effluent pH and TAN after AD. A review of the literature shows limited findings regarding the environmental impacts (i.e. NH_3 and greenhouse gas emissions) of applying digestates obtained from AD of slurry to Irish soils. Further studies are required to adequately document the relative benefits of AD under Irish environmental conditions.

Recent research findings reveal the potential application of algae as a biofertiliser (Baweja *et al*., 2019; Al-Myali, 2021;

Zou *et al*., 2021). These organisms could also play a critical role in managing nutrients in slurry prior to landspreading as algae can be cultured using the elements (e.g. nitrogen, carbon and phosphorus) in slurry/manure as substrates. The increasing cost of fertilisers and the need to identify low-emission alternatives to land application of slurry could result in this being a potentially viable fertiliser in the future. Nevertheless, information regarding the environmental benefits and economic viability of the above strategy is still lacking in the literature and needs to be investigated.

In order to facilitate the adoption of slurry acidification in Ireland, research is required to generate ample scientific data, particularly regarding issues such as its NH_3 abatement potential, associated cost, impact on other pollutant gases from slurry as well as ways to address its health and safety concerns. Even though this review highlights the potential to abate NH_3 emissions during slurry storage through the use of slurry amendments, there is limited information regarding their impact on other N loss pathways such as N_2O and NO_3^- leaching under Irish conditions and should be studied. As previously mentioned, the majority of dairy system in Ireland are slurry-based. As a result, the slurry-based system has been the focus of most research on NH_3 mitigation. Information regarding NH_3 mitigation techniques from the straw-based system where excreta is collected as farmyard or solid manure is rare and needs to be investigated.

Limitations of the study

The main limitations in the study is the low number of samples and large data variability. This resulted in our data having a low statistical power to detect effects. For example, in sections relating to slurry storage, EFs were generated from small-scale incubations which measured NH_3 in a dynamic chamber mode which means that the airflow is expected to be higher than above a typical tank in the shed. Thus, EFs generated from the storage stage in the present study can only be used to provide an indication of the efficacy of abatement strategies and cannot be used to provide reliable EFs as they may overestimate NH_3 emissions. Further studies are required to generate additional datasets in order to perform a comprehensive assessment of NH_3 EFs generated from cattle production systems in Ireland and the efficacy of NH_3 abatement strategies available to Irish Agriculture.

Conclusion

This review provides an overview of NH_3 EFs in Ireland. EFs generated from the deposition of excreta during grazing, housing/storage and slurry landspreading varied considerably. The urease inhibitor NBPT was effective at reducing EFs after urea application to grassland. Acidifying additives or chemical amendments were effective at abating emissions from concrete yards or hard standings used by cattle and during slurry storage. Emissions from landspreading can be reduced within daily application timing management. Studies which compared LESS with the splashplate generally reported lower NH_3 EFs for the LESS compared to the splashplate. Further studies are required to generate additional data to validate the findings from this review. Further research is also needed to evaluate the impact of these NH_3 abatement strategies on other forms of N such as N_2O and NO_3 to ensure that the abatement of one form of N does not lead to pollution swapping either at the same stage or at a different stage in animal agriculture. Finally, research on the efficacy of new abatement strategies and mitigation options for solid manure systems in Irish agriculture is also required.

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Declaration of competing interest

The authors declare they do not have any personal relationships or known competing financial interests that could have appeared to influence the findings of this paper.

References

Al-Myali, A.A.H. 2021. Effect of algae and Bio-fertilizers on some growth characteristics and yield of wheat *Triticum aestivum* L. *Journal of Kerbala for Agricultural Sciences* **8**: 36–47.

Asing, J., Saggar, S., Singh, J. and Bolan, N.S. 2008. Assessment of nitrogen losses from urea and an organic manure with and without nitrification inhibitor, dicyandiamide, applied to lettuce under glasshouse conditions. *Soil Research* **46**: 535–541.

Baweja, P., Kumar, S. and Kumar, G. 2019. Organic fertilizer from algae: a novel approach towards sustainable agriculture. In: "Biofertilizers for Sustainable Agriculture and Environment", pages 353–370.

Bittman, S., Dedina, M.C.M.H., Howard, C.M., Oenema, O. and Sutton, M.A. 2014. "Options for Ammonia Mitigation: Guidance from the UNECE Task Force on Reactive Nitrogen". NERC/Centre for Ecology & Hydrology.

Bobrowski, A.B., Van Dooren, H.J., Ogink, N., Hagenkamp-Korth, F., Hasler, M. and Hartung, E. 2021. Reduction of ammonia emissions by using a urease inhibitor in a mechanically ventilated dairy housing system. *Biosystems Engineering* **204**: 115–129.

Bourdin, F., Sakrabani, R., Kibblewhite, M.G. and Lanigan, G.J. 2014. Effect of slurry dry matter content, application technique and timing on emissions of ammonia and greenhouse gas from cattle slurry applied to grassland soils in Ireland. *Agriculture, Ecosystems & Environment* **188**: 122–133.

Brennan, R.B., Healy, M.G., Fenton, O. and Lanigan, G.J. 2015. The effect of chemical amendments used for phosphorus abatement on greenhouse gas and ammonia emissions from dairy cattle slurry: synergies and pollution swapping. *PLoS One* **10**: e0111965.

Buckley, C., Krol, D., Lanigan, G.J., Donnellan, T., Spink, J., Hanrahan, K., Boland, A., Forrestal, P., Humphreys, J., Murphy, P. and NiFhlatharta, N. 2020. "An Analysis of the Cost of the Abatement of Ammonia Emissions in Irish Agriculture to 2030". Teagasc.

Burchill, W., Forrestal, P.J., Kavanagh, I. and Lanigan, G.J. 2017a. Ammonia Emissions From Pasture-Based Production Systems: a collation of Irish research. *Proceedings of the International Fertiliser Society*, pages 1–36.

Burchill, W., Lanigan, G.J., Forrestal, P.J., Reville, F., Misselbrook, T. and Richards, K.G. 2016. A field-based comparison of ammonia emissions from six Irish soil types following urea fertiliser application. *Irish Journal of Agricultural and Food Research* **55**: 152–158.

Burchill, W., Lanigan, G.J., Forrestal, P.J., Misselbrook, T. and Richards, K.G. 2017b. Ammonia emissions from urine patches amended with N stabilized fertilizer formulations. *Nutrient Cycling in Agroecosystems* **108**: 163–175.

Burchill, W., Reville, F., Misselbrook, T.H., O'Connell, C. and Lanigan, G.J. 2019. Ammonia emissions and mitigation from a concrete yard used by cattle. *Biosystems Engineering* **184**: 181–189.

Cahalan, E., Ernfors, M., Müller, C., Devaney, D., Laughlin, R.J., Watson, C.J., Hennessy, D., Grant, J., Khalil, M.I., McGeough, K.L. and Richards, K.G. 2015. The effect of the nitrification inhibitor dicyandiamide (DCD) on nitrous oxide and methane emissions after cattle slurry application to Irish grassland. *Agriculture, Ecosystems & Environment* **199**: 339–349.

Cai, Y. and Akiyama, H. 2016. Nitrogen loss factors of nitrogen trace gas emissions and leaching from excreta patches in grassland ecosystems: a summary of available data. *Science of the Total Environment* **572**: 185–195.

Clay, D.E., Malzer, G.L. and Anderson, J.L. 1990. Ammonia volatilization from urea as influenced by soil temperature, soil water content and nitrification and hydrolysis inhibitors. *Soil Science Society of America Journal* **54**: 263–266.

CSO. 2021. "Census of Agriculture Preliminary Results 2020". Available online: https://www.cso.ie/en/csolatestnews/pressreleases/2021pressreleases/pressstatementcensusofagriculture2020/ [Accessed 15 February, 2023].

De Verdal, H., Mignon-Grasteau, S., Bastianelli, D., Même, N., Le Bihan-Duval, E. and Narcy, A. 2013. Reducing the environmental impact of poultry breeding by genetic selection. *Journal of Animal Science* **91**: 613–622.

DEFRA. 2020. "Impact assessment (IA)". Available online: https://consult.defra.gov.uk/airquality/implementation-of-cazs/supporting_documents/161012%20%20CAZ%20Impact%20Assessment%20%20FINAL%20consultation.pdf [Accessed 22 January 2023].

DEFRA. 2022. "Trends in Critical Load and Critical Level Exceedances in the UK". Available online: https://ukair.defra.gov.uk/assets/documents/reports/cat09/2208301034_Trends_Report_2022.pdf [Accessed 22 December 2022].

Dowling, C. 2012. Quantification and control of ammonia emissions from the landspreading of cattle slurry and cattle wintering facilities. PhD dissertation.

Dowling, C., Curran, T. and Lanigan, G. 2008. The effect of application technique and climate conditions on ammonia emissions from cattle slurry. *Proceedings of the Agricultural Research Forum, Tullamore, Ireland*, pages 12–13.

Emmerling, C., Krein, A. and Junk, J. 2020. Meta-analysis of strategies to reduce NH3 emissions from slurries in European agriculture and consequences for greenhouse gas emissions. *Agronomy* **10**: 1633.

Environmental Protection Agency (EPA). 2022. Ireland's Informative Inventory Report 2021. Available online: https://www.epa.ie/publications/monitoring--assessment/climate-change/air-emissions/Ireland-IIR-2022_mergev2.pdf [Accessed 6 December 2023].

European Environment Agency. 2019. EMEP/EEA air pollutant emission inventory guidebook 2019. Technical guidance to prepare national emission inventories. Available online: https://www.eea.europa.eu/publications/emep-eea-guidebook-2019 [Accessed 22 January 2023].

Fischer, K., Burchill, W., Lanigan, G.J., Kaupenjohann, M., Chambers, B.J., Richards, K.G. and Forrestal, P.J. 2016. Ammonia emissions from cattle dung, urine and urine with dicyandiamide in a temperate grassland. *Soil Use and Management* **32**: 83–91.

Forrestal, P.J., Wall, D., Carolan, R., Harty, M.A., Roche, L., Krol, D., Watson, C.J., Lanigan, G. and Richards, K.G. 2016. Effects of urease and nitrification inhibitors on yields and emissions in grassland and spring barley. *International Fertiliser Society, Cambridge, UK*.

Frost, J.P., Stevens, R.J. and Laughlin, R.J. 1990. Effect of separation and acidification of cattle slurry on ammonia volatilization and on the efficiency of slurry nitrogen for herbage production. *The Journal of Agricultural Science* **115**: 49–56.

Harty, M.A., Forrestal, P.J., Watson, C.J., McGeough, K.L., Carolan, R., Elliot, C., Krol, D., Laughlin, R.J., Richards, K.G. and Lanigan, G.J. 2016. Reducing nitrous oxide emissions by changing N fertiliser use from calcium ammonium nitrate (CAN) to urea based formulations. *Science of the Total Environment* **563**: 576–586.

Hu, Y. and Schmidhalter, U. 2021. Urease inhibitors: opportunities for meeting EU national obligations to reduce ammonia emission ceilings by 2030 in EU countries. *Environmental Research Letters* **16**: 084047.

Kavanagh, I., Burchill, W., Healy, M.G., Fenton, O., Krol, D.J. and Lanigan, G.J. 2019. Mitigation of ammonia and greenhouse gas emissions from stored cattle slurry using acidifiers and chemical amendments. *Journal of Cleaner Production* **237**: 117822.

Kavanagh, I., Fenton, O., Healy, M.G., Burchill, W., Lanigan, G.J. and Krol, D.J. 2021. Mitigating ammonia and greenhouse gas emissions from stored cattle slurry using agricultural waste, commercially available products and a chemical acidifier. *Journal of Cleaner Production* **294**: 126251.

Kelleghan, D.B., Hayes, E.T., Everard, M. and Curran, T.P. 2019. Mapping ammonia risk on sensitive habitats in Ireland. *Science of the Total Environment* **649**: 1580–1589.

Kelleghan, D.B., Hayes, E.T., Everard, M., Keating, P., Lesniak-Podsiadlo, A. and Curran, T.P. 2021. Atmospheric ammonia and nitrogen deposition on Irish Natura 2000 sites: Implications for Irish agriculture. *Atmospheric Environment* **261**: 118611.

Kim, D.G., Saggar, S. and Roudier, P. 2012. The effect of nitrification inhibitors on soil ammonia emissions in nitrogen managed soils: a meta-analysis. *Nutrient Cycling in Agroecosystems* **93**: 51–64.

Krol, D.J., Forrestal, P.J., Wall, D., Lanigan, G.J., Sanz-Gomez, J. and Richards, K.G. 2020. Nitrogen fertilisers with urease inhibitors reduce nitrous oxide and ammonia losses, while retaining yield in temperate grassland. *Science of the Total Environment* **725**: 138329.

Kupper, T., Häni, C., Neftel, A., Kincaid, C., Bühler, M., Amon, B. and VanderZaag, A. 2020. Ammonia and greenhouse gas emissions from slurry storage-A review. *Agriculture, Ecosystems & Environment* **300**: 106963.

Lalor, S.T.J. and Lanigan, G.J. 2010. The potential of application timing management to reduce ammonia emissions following cattle slurry application. *Proceedings of the 14th International RAMIRAN Conference, Lisboa, Portugal*.

Lalor, S.T.J., Schröder, J.J., Lantinga, E.A., Oenema, O., Kirwan, L. and Schulte, R.P.O. 2011. Nitrogen fertilizer replacement value of cattle slurry in grassland as affected by method and timing of application. *Journal of Environmental Quality* **40:** 362–373.

Lanigan, G.J., Donnellan, T., Hanrahan, K., Burchill, W., Forrestal, P., McCutcheon, G., Crosson, P., Murphy, P., Schulte, R., Richards, K. and Browne, P. 2015. "An Analysis of the Cost of the Abatement of Ammonia Emissions in Irish Agriculture to 2030". Teagasc Submission to the Department of Agriculture, Food and the Marine. Teagasc, Oak Park, Carlow. Available online: https://www.teagasc.ie/media/website/publications/2015/An-Analysis-of-the-Cost-of-the-Abatement-of-Ammonia-Emissions-in-Irish-Agriculture-to-2030.pdf [Accessed 30 December 2021].

Läpple, D., Hennessy, T. and O'donovan, M. 2012. Extended grazing: a detailed analysis of Irish dairy farms. *Journal of Dairy Science* **95**: 188–195.

Laubach, J., Taghizadeh-Toosi, A., Gibbs, S.J., Sherlock, R.R., Kelliher, F.M. and Grover, S.P.P. 2013. Ammonia emissions from cattle urine and dung excreted on pasture. *Biogeosciences* **10:** 327–338.

Liu, H. and Zhou, D. 2014. Mitigation of ammonia and nitrous oxide emissions from pasture treated with urine of sheep fed diets supplemented with sodium chloride. *Animal Feed Science and Technology* **192**: 39–47.

Lockyer, D.R. 1984. A system for the measurement in the field of losses of ammonia through volatilisation. *Journal of the Science of Food and Agriculture* **35**: 837–848.

Lockyer, D.R. and Whitehead, D.C. 1990. Volatilization of ammonia from cattle urine applied to grassland. *Soil Biology and Biochemistry* **22**: 1137–1142.

Mathias, E. and Martin E. 2013. "Analyse du potentiel de 10 actions de réduction des émissions d'ammoniac des élevages Français aux horizons 2020 et 2030, ADEME, Décembre 2013". Available online: https://unece.org/fileadmin/DAM/env/documents/2020/AIR/WGSR/Final_Assessment_Report_on_Ammonia_v2_20201126_b.pdf [Accessed 20 January 2023].

Matulaitis, R., Juškienė, V. and Juška, R. 2013. Gaseous emissions from manure as affected by microbial-based additive and temperature. *Veterinary Medicine and Zootechnics* **64**: 86.

McIlroy, J.P., McGeough, K.L., Laughlin, R.J. and Carolan, R. 2019. Abatement of ammonia emissions from dairy cow house concrete floor surfaces through additive application. *Biosystems Engineering* **188**: 320–330.

Ministry of Environment and Food of Denmark (MEFD). 2017. "Overview of the Danish regulation of nutrients in agriculture and the Danish Nitrates Action Programme". Available online: https://eng.mst.dk/media/186211/overview-of-the-danish-regulation-of-nutrients-in-agriculture-the-danish-nitrates-action-programme.pdf [Assessed 23 January 2023].

Misselbrook, T.H., Cardenas, L.M., Camp, V., Thorman, R.E., Williams, J.R., Rollett, A.J. and Chambers, B.J. 2014. An assessment of nitrification inhibitors to reduce nitrous oxide emissions from UK agriculture. *Environmental Research Letters* **9**: 115006.

Misselbrook, T.H., Pain, B.F. and Headon, D.M. 1998. Estimates of ammonia emission from dairy cow collecting yards. *Journal of Agricultural Engineering Research* **71**: 127–135.

Misselbrook, T.H., Webb, J., Chadwick, D.R., Ellis, S. and Pain, B.F. 2001. Gaseous emissions from outdoor concrete yards used by livestock. *Atmospheric Environment* **35**: 5331–5338.

Moal, J.F., Martinez, J., Guiziou, F. and Coste, C.M. 1995. Ammonia volatilization following surface-applied pig and cattle slurry in France. *The Journal of Agricultural Science* **125**: 245–252.

National Atmospheric Emissions Inventory (NAEI). 2022. "Air Pollutant Inventories for England, Scotland, Wales, and Northern Ireland: 2005-2020". Available online: https://uk-air.defra.gov.uk/assets/documents/reports/cat09/2210251052_DA_Air_Pollutant_Inventories_2005-2020_FINAL_v1.2.pdf [Accessed 15 January 2023].

Nichols, K.L., Del Grosso, S.J., Derner, J.D., Follett, R.F., Archibeque, S.L., Delgado, J.A. and Paustian, K.H. 2018. Nitrous oxide and ammonia emissions from cattle excreta on shortgrass steppe. *Journal of Environmental Quality* **47**: 419–426.

Owusu-Twum, M.Y., Polastre, A., Subedi, R., Santos, A.S., Ferreira, L.M.M., Coutinho, J. and Trindade, H. 2017. Gaseous emissions and modification of slurry composition during storage and after field application: Effect of slurry additives and mechanical separation. *Journal of Environmental Management* **200**: 416-422.

Payne, R.J., Dise, N.B., Field, C.D., Dore, A.J., Caporn, S.J. and Stevens, C.J. 2017. Nitrogen deposition and plant biodiversity: past, present, and future. *Frontiers in Ecology and the Environment* **15**: 431–436.

Pedersen, J., Andersson, K., Feilberg, A., Delin, S., Hafner, S. and Nyord, T. 2021. Effect of exposed surface area on ammonia emissions from untreated, separated and digested cattle manure. *Biosystems Engineering* **202**: 66–78.

Prakasa Rao, E.V.S. and Puttanna, K. 1987. Nitrification and ammonia volatilisation losses from urea and dicyandiamide-treated urea in a sandy loam soil. *Plant and Soil* **97**: 201–206.

Provolo, G., Finzi, A., Perazzolo, F., Mattachini, G. and Riva, E. 2016. Effect of a biological additive on nitrogen losses from pig slurry during storage. *Journal of Environmental Quality* **45**: 1460–1465.

Saggar, S., Jha, N., Deslippe, J., Bolan, N.S., Luo, J., Giltrap, D.L., Kim, D.G., Zaman, M. and Tillman, R.W. 2013. Denitrification and N_2O:N_2 production in temperate grasslands: Processes, measurements, modelling and mitigating negative impacts. *Science of the Total Environment* **465**: 173–195.

Selbie, D.R., Buckthought, L.E. and Shepherd, M.A. 2015. The challenge of the urine patch for managing nitrogen in grazed pasture systems. *Advances in Agronomy* **129**: 229–292.

Sigurdarson, J.J., Svane, S. and Karring, H. 2018. The molecular processes of urea hydrolysis in relation to ammonia emissions from agriculture. *Reviews in Environmental Science and Bio/Technology* **17**: 241–258.

Smith, L.C., deKlein, C.A.M. and Catto, W.D. 2008. Effect of dicyandiamide applied in a granular form on nitrous oxide emissions from a grazed dairy pasture in Southland, New Zealand. *New Zealand Journal of Agricultural Research* **51**: 387–396.

Sommer, S.G. and Olesen, J.E. 2000. Modelling ammonia volatilization from animal slurry applied with trail hoses to cereals. *Atmospheric Environment* **34**: 2361–2372.

Sommer, S.G., Olesen, J.E. and Christensen, B.T. 1991. Effects of temperature, wind speed and air humidity on ammonia volatilization from surface applied cattle slurry. *The Journal of Agricultural Science* **117**: 91–100.

Van der Stelt, B., Temminghoff, E.J.M., Van Vliet, P.C.J. and Van Riemsdijk, W.H. 2007. Volatilization of ammonia from manure as affected by manure additives, temperature and mixing. *Bioresource Technology* **98**: 3449–3455.

Wheeler, E.F., Adviento-Borbe, M.A.A., Brandt, R.C., Topper, P.A., Topper, D.A., Elliott, H.A., Graves, R.E., Hristov, A.N., Ishler, V.A. and Bruns, M.A.V. 2010. Amendments for mitigation of dairy manure ammonia and greenhouse gas emissions: preliminary screening. *Proceedings of American Society of Agricultural and Biological Engineers, Pittsburgh, Pennsylvania*, page 1.

Wilkins, K., Aherne, J. and Bleasdale, A. 2016. Vegetation community change points suggest that critical loads of nutrient nitrogen may be too high. *Atmospheric Environment* **146**: 324–331.

Wulf, S., Rösemann, C., Eurich-Menden, B. and Grimm, E. 2017. Ammoniakemissionen in der Landwirtschaft Minderungsziele und – potenziale Aktuelle rechtliche Rahmenbedingungen für die Tierhaltung, Thünen, Hannover 30.05.2017. Available online: https://www.ktbl.de/fileadmin/user_upload/Allgemeines/Download/Tagungen-2017/Rechtliche_Rahmenbedingungen_Tierhaltung/Ammoniakemissionen.pdf [Accessed 20 January 2023].

Zaman, M. and Blennerhassett, J.D. 2010. Effects of the different rates of urease and nitrification inhibitors on gaseous emissions of ammonia and nitrous oxide, nitrate leaching and pasture production from urine patches in an intensive grazed pasture system. *Agriculture, Ecosystems & Environment* **136**: 236–246.

Zaman, M., Zaman, S., Nguyen, M.L., Smith, T.J. and Nawaz, S. 2013. The effect of urease and nitrification inhibitors on ammonia and nitrous oxide emissions from simulated urine patches in pastoral system: a two-year study. *Science of the Total Environment* **465**: 97–106.

Zou, Y., Zeng, Q., Li, H., Liu, H. and Lu, Q. 2021. Emerging technologies of algae-based wastewater remediation for bio-fertilizer production: a promising pathway to sustainable agriculture. *Journal of Chemical Technology & Biotechnology* **96**: 551–563.