Journal Pre-proofs

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PII: S0048-9697(19)34597-8
DOI: https://doi.org/10.1016/j.scitotenv.2019.134606
Reference: STOTEN 134606

To appear in: *Science of the Total Environment*

Received Date: 11 January 2019
Revised Date: 26 August 2019
Accepted Date: 21 September 2019

Please cite this article as: N.J. Hoekstra, R.P.O. Schulte, P.J. Forrestal, D. Hennessy, D. Krol, G. Lanigan, C. Müller, L. Shalloo, D. Wall, K.G. Richards, Scenarios to limit environmental nitrogen losses from dairy expansion, *Science of the Total Environment* (2019), doi: https://doi.org/10.1016/j.scitotenv.2019.134606

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Keywords: ammonia, nitrate, nitrous oxide, N footprint, N flow model

Journal: **Science of the Total Environment**

Abstract

Increased global demand for dairy produce and the abolition of EU milk quotas has resulted in expansion in dairy production across Europe and particularly in Ireland. Simultaneously, there is increasing pressure to reduce the impact of nitrogen (N) losses to air and groundwater on the environment. In order to develop grassland management strategies for grazing systems that meet environmental targets and are economically sustainable, it is imperative that individual mitigation measures for N efficiency are assessed at farm system level. To this end, we developed an excel-based N flow model simulating an Irish grass-based dairy farm, to evaluate the effect of farm management on N efficiency, N losses, production and economic performance. The model was applied to assess the effect of different strategies to achieve the increased production goals on N utilization, N loss pathways and economic performance at farm level. The three strategies investigated included increased milk production through increased grass production, through increased concentrate feeding and by applying a high profit grass-based system. Additionally, three mitigation measures; low ammonia emission slurry application, the use of urease and nitrification inhibitors and the combination of both were applied to the three strategies.

Absolute N emissions were higher for all intensification scenarios (up to 124 kg N ha^{-1}) compared to the baseline (80 kg N ha^{-1}) due to increased animal numbers and higher feed and/or fertiliser inputs. However, some intensification strategies showed the potential to reduce the emissions per ton milk produced for some of the N-loss pathways. The model showed that the assessed mitigation measures can play an important role in ameliorating the increased emissions associated with intensification, but may not be adequate to entirely offset absolute increases. Further improvements in farm N use efficiency and alternatives to mineral fertilisers will be required to decouple production from reactive N emissions.

1 Introduction

Agriculture is a major driver of the global nitrogen (N, Table 1 for abbreviations) cycle, with terrestrial N fixation increasing 400% since the industrial revolution (Sutton et al., 2013). There is increasing pressure on the agricultural sector to reduce the impact of N losses to air and groundwater on the environment. Average pasture-based livestock production systems have a low N-use efficiency (NUE) (<30%) resulting in large surpluses that are vulnerable to loss (Burchill et al., 2016). The principal sources of N pollution are associated with the application of organic or mineral N fertilizers and deposition of excreta in housing and fields. Denitrification converts much of this N back into unreactive N_2, but is also associated with the production of the potent greenhouse gas nitrous oxide (N_2O). The proportion of N lost as the reactive forms N_2O and nitric oxide (NO) is generally very low and while the proportion of the benign N_2 form is considerably higher, it represents an economic loss to the farmer resulting in lower N use efficiency. Ammonia volatilisation contributes to indirect N_2O emissions through increased N deposition (Martikainen, 1985) and is also related to eutrophication and acidification of natural ecosystems (Asman et al., 1998), while nitrate (NO_3) leaching also leads to the eutrophication of ground and surface waters (Sutton et al. 2013). At the same time, FAO projections suggest that increases in global population and wealth will increase demand for dairy and meat by more than 50% by 2050 compared to 2006 (Bruinsma, 2009). More recent assessments forecast an 80%

increase in dairy demand between 2000 and 2050 (Huang, 2010). Most importantly, there are significant concerns that this increase in food production will be associated with (among other impacts on natural resources) increased global greenhouse gas (GHG) and N emissions from agriculture. For example, Smith et al. (2007) estimated that, by 2020, global GHG emissions from agriculture will increase by 38% relative to 1990 (24% relative to 2005). In light of the sustained future demand for dairy and meat, it is essential that the N emissions per unit product (N emissions intensity) are reduced. Irish policies on agriculture are specifically aimed at addressing this challenge. In 2010, the Department of Agriculture, Food and the Marine (DAFM) published the Food Harvest 2020 Strategy (DAFM, 2010), which provided a roadmap for the industry to grow in the wake of the phasing out of EU milk quotas in 2015. Food Harvest 2020 identified potential to increase national levels of milk production by 50% between the reference period (2007-2009) and 2020. In 2015, this strategy was superseded by the Food Wise 2025 strategy (DAFM, 2015). Both strategies are based on the concept of 'Smart Green Growth', in which 'smart' refers to the leveraging of knowledge and technologies and 'green' refers to the imperative for growth to be sustainable within the context of environmental protection.

Sustainable intensification (Godfray *et al*., 2010) refers to increasing total food production from the current global agricultural land area, thus negating increased competition for land with ecological habitats, while reducing or at least decoupling the associated environmental impact (Schulte *et al*., 2014). This means that production is increased without increasing relative resource input rates, e.g. nutrients and pesticide quantities per unit food produced.

In practice, any increase in production must be aligned with the environmental policy context. Most of this context in Ireland, as in many EU countries is derived, either directly or indirectly, from EU policies. Examples of policies pertaining to reducing the impact of nutrient losses to the environment include the EU Nitrates Directive (EU, 1991), Drinking Water Directive and EU Water Framework Directive (EU, 2000) for nitrates, the EU Climate and Energy Package for 2020 (EU, 2009) and the EU Climate and Energy Framework for 2030 (European Council, 2014) for N_2O, and the National Emission Ceilings (NEC) Directive for ammonia (EU 2016).

While each of these policies relate to curtailing the loss of an individual N species, they are all equally progressive, in that they require increasingly stringent standards to be met over time. This means that the growth in agricultural production must be realised while reducing losses of N to the environment. Ireland provides a good example of a country that has the ambition to increase dairy production to meet the increased global demand and at the same time Ireland has challenging targets to reduce NO_3 leaching, N_2O and NH_3 emissions.

The quantity and form of N loss is highly dependent on farm management. Recent research has yielded increasing insight into the disaggregated N losses from different sources, i.e. urine and dung excretion in the field, fertiliser application, manure storage (Fischer et al. 2016; Krol et al. 2016; Forrestal et al., 2016ab; Hyde et al., 2016;; Harty et al., 2016; Roche et al., 2016). Research has shown significant potential of the use of nitrification inhibitors to limit these N losses and uncouple N emissions from level of activity (Smith et al., 2008; Monaghan et al., 2009; Harty et al., 2016; Roche et al., 2016).

In order to develop grassland management strategies that both meet environmental targets and are economically sustainable, it is imperative that individual mitigation measures for increased N use efficiency are assessed on a farm system level and evaluated for cost-effectiveness. The impact of N use associated with agricultural production can be constructed in a variety of ways that differ in terms of the functional unit or boundary of the analysis. Generally, farm-gate N pollution is assessed in terms of N input minus N output, which generates a farm N surplus (Oenema 2006, Mihailescu et al. 2014, Burchill et al. 2016). The 'N footprint' of an agricultural commodity on the other hand, can be assessed in terms of the' N surplus per unit product' which, as with the more traditional C-footprint , assesses the impact of N pollution for a commodity across the whole production chain, including upstream (feed / fertiliser production) emissions (Galloway et al. 2014, Mu et al. 2016). While the N footprint approach may provide a more holistic assessment of the N implications of production, it may mask the impact of individual mitigation measures on farm production systems (Einarsson et al. 2018). To this end, we developed an excel-based N flow model simulating an Irish grass-based dairy farm, to simulate the effect of farm management on N efficiency, N losses, production and economic performance.

In this paper we present the model description and we apply the model to assess the effect of different strategies to achieve the increased production goals of Food Wise 2025 on N utilization, N loss pathways and economic performance at farm level.

The strategies investigated include increased milk production through increased grass production, through increased concentrate feeding and by applying a high profit grass based system. Additionally, three mitigation measures are applied to the three intensification scenarios; 1)low emission slurry application, 2) the use of urease and nitrification inhibitors, and 3) a combination of 1 and 2.

2 Materials and methods

2.1 Model description

The €riN model is an excel-based model simulating (N) flows on an Irish grass-based dairy farm. Calculations are based on a monthly time-step and outputs are presented on a yearly basis. The model dynamically links the Moorepark Dairy System Model (MDSM) (Shalloo et al., 2004), with a grass growth module and a number of N loss modules in order to simulate the effect of farm management on N use efficiency, N losses and economic performance, as described below. The model divides the farm into three nominal (farm) areas based on the grassland management structure: 1) Grazing only, 2) One-cut silage + grazing and 3) Two-cut silage + grazing. Closing dates and silage cutting dates (inputs) are used to calculate the temporal and spatial proportion allocated to grazing and silage for each month.

2.1.1 Moorepark Dairy System Model (MDSM)

The MDSM is a budgetary simulation model of a dairy farm, integrating animal inventory and valuation, milk supply, feed requirement, land and labour utilisation and economic analysis (Shalloo et al., 2004). The model calculates the farm economic balance based on variable costs (fertiliser, concentrate, contractor charges, medical and veterinarian fees, artificial insemination, silage, and reseeding), fixed costs (machinery maintenance and running costs, farm maintenance, car, telephone, electricity and insurance) and prices (calf, milk and cow). In the default scenario for this analysis, the milk production system is spring calving (calving interval of 365 days with 15, 65 and 20% of the cows calving in January, February and March, respectively). The model assumes that 45% of female calves are reared for replacements (see Shalloo et al. (2004) for more details).

Feed demand is calculated based on the net energy (NE) requirements for milk production, maintenance and body weight change. Part of this NE requirement is met by concentrate, and the remainder by grass silage and grazed grass, based on the monthly proportion of the total grass NE derived from grazed grass (input variable). The mean yearly grass and silage demand per cow (including requirements for calves and yearlings) is used to calculate 1) the total number of cows that can be held on the farm, and 2) the proportion of the farm area required for first cut silage, taking into account grass losses associated with grazing and silage production (Table S1).

2.1.2 Grass growth

Potential grass dry matter yield production (DMY_{Pot}) is set at 17.5 ton DM ha^{-1} yr^{-1} (O'Donovan et al., 2011), and follows a typical pattern over the year (Table S1). The actual grass production (DMY_{Actual}) is derived from the following equation:

$$DMY_{actual} = DMY_{Pot} \times CF_{Sward} \times CF_P \times CF_N \qquad \text{Eq. 1}$$

Where

CF_{Sward} = correction factor for sub-optimal sward composition, assuming a 1% decrease in potential yield with every year increase in sward age, based on the effect of decreasing proportion of perennial ryegrass in the sward with increasing sward age (Creighton et al., 2010). For example, if the reseeding rate is 5% per year, the average sward age is 10 years, and CF_{Sward} is 0.9.

CF_P = Correction factor for sub-optimal P application rates resulting in soil P index < 3, set at 0.90 (P index 1) and 0.94 (P index 2) based on Herlihy et al. (2004).

CF_N = Correction factor for the DMY response to the application rate of available N, calculated from a quadratic plus plateau function derived from the N response curve for grassland by Hopkins (2000) and is calculated on a monthly basis.

$$CF_N = 0.22 + 0.0033 \cdot N - 3.4 \cdot 10^{-6} \cdot N^2, \qquad N < 474 \text{ kg N ha}^{-1} \text{ yr}^{-1} \qquad \text{Eq. 2}$$

$$CF_N = 1 \qquad N \geq 474 \text{ kg N ha}^{-1} \text{ yr}^{-1}$$

It is assumed that of the available N applied, 70% is available for the month of application and the residual 30% is available for the month following the month of application (Murphy et al., 2013). Grass growth is calculated separately for the grazing and silage areas of each farm area, taking into account the proportional time and area dedicated to grazing or silage, and grazing opening and closing dates.

2.1.3 Animal excreta: slurry, soiled water, dung and urine

2.1.3.1 N excretion

An overview of the calculation of the total N excretion and its partition over slurry, soiled water and grazing (dung and urine) are shown in Figure 1. Total $N_{Excreted}$ equals N inputs (N in intake and animals bought) minus N outputs (N in milk and animals sold or died). Soiled water is produced as the result of washings of the milking parlour and collecting yard. The calculation of $N_{Soiled\ water}$ is based on the assumption that one out of every 12 dung and urine depositions of dairy cows in lactation takes place in the yard or parlour (Figure 1). The N deposited on pathways by cows during transfer to and from the milking parlour ($N_{Pathways}$) is assumed to be equal to $N_{Soiled\ water}$ multiplied by the proportion of time spent grazing (see below). The remaining $N_{Excreted}$ is divided over slurry and grazing based on the proportion of grass Net Energy Intake in the form of grazed grass ($P\_NEI_{Grazed\ grass}$) which is used as a proxy for the proportion of time spent grazing (excluding milking time):

$$P\_NEI_{Grazed\ grass} = NEI_{Grazed\ grass} / (NEI_{Silage} + NEI_{Grazed\_grass}) \quad \text{Eq. 3)}$$

The proportion of time spent grazing is reduced when restricted grazing is applied, assuming that cows take only 50% of the usual time to achieve the same grass intake (Kennedy et al., 2011).

The proportion of N excreted in the form of urine (P_N_{Urine}) is used to determine the urine excretion during grazing and depends on total N excretion based on work by Kebreab et al. (2001) (Supplement S2.1)

2.1.3.2 Slurry and soiled water volume and N and P content

The total excretion volume produced on farm is calculated on a monthly basis, based on tabular values for the excretion volume per cow per week (Anon, 2014). Slurry volume equals the total excretion volume multiplied by the proportion of $N_{Excreted}$ in the form of slurry (see Figure 1). Soiled water volume is calculated by multiplying the volume per cow per day (26 L with optional additional 19 L per cow per day if the collecting yard is washed) with the number of cows in lactation. If soiled water is stored together with slurry, the total volume and N content are recalculated, accordingly, creating "dilute slurry". The N content in the slurry and soiled water for application is corrected for ammonia losses from the yard, storage and housing using emission factors based on Hyde et al. (2003) and Misselbrook et al. (2000) (Table S2).

2.1.3.3 Area affected by dung and urine

The grazing area affected by dung and urine patches during each month is calculated as the product of the number of cattle, the number of excretions per day, the area affected per excretion and the cumulative number of days spent on the grazing area. For lactating dairy cattle it is assumed that 10 out of 12 urinations and defecations per day are made onto the pasture, one on pathways during transfer, and one in the yard. The area affected per excretion has been estimated as 0.68 m^2 for urine and 0.25 m^2 for dung (Lantinga et al., 1987), which is in line with recent work in Irish grasslands, where Minet et al. (2016) reported urine patches with a median size of 0.62 m^2. This area is adjusted for different animal types (e.g. dairy cows and calves) based on the type-specific tabular organic N production (Anon, 2014). This total excretion area is multiplied by the proportion of time spent grazing (Eq. 3) to calculate the total grassland area affected by dung and urine. A Poisson distribution is used to correct this proportion of the area affected for overlap, assuming that the distribution of excreta is not concentrated in specific areas (Haynes and Williams 1999).

2.1.4 Fertiliser, slurry and soiled water application

2.1.4.1 Total N, P and K application

The total available N, P and K application rates per ha for each farm area are input parameters. P and K applications are made on a yearly basis. The available N applied to the grazing only farm area is divided over the months based on the proportional monthly fertiliser N distribution for grazing (Table S1). Nitrogen application rates for first and second cut silage are inputs, and the N rates applied for grazing before and after closing of the silage grounds are assumed to be equal to the grazing only rates.

2.1.4.2 Slurry and soiled water application

Slurry and soiled water (if applicable) application rates (m^3 ha^{-1}) can be assigned for individual farm areas. For slurry, a maximum number of two application dates per year can be selected, whereas it is assumed that soiled water is spread during the month it is produced, as is common practice in Ireland (Ryan et al., 2014). It is assumed that all of the P in slurry and soiled water is plant available, except when the soil has a P-index of 1 or 2, in which case the slurry P-availability is assumed to be 0.5 (to allow for rebuilding of soil P levels) (Anon, 2018). Slurry N availability is based on the proportion of N in the form of total ammoniacal N (TAN, default set at 0.6 kg kg^{-1} N) minus the proportion of TAN lost through ammonia volatilisation during application (see section 2.1.7), and soiled water N availability is set at 0.8 kg kg^{-1} N applied (Lalor et al., 2012).

2.1.4.3 Inorganic fertiliser N, P and K application

The inorganic fertiliser N, P and K requirement is calculated by deducting available nutrients from slurry and soiled water from the total (monthly for N) nutrient requirement. Monthly inorganic fertiliser N applications can be made in the form of calcium ammonium nitrate (CAN) or urea (input % of monthly inorganic N application). Fertiliser prices are set at 2 €, 1 €, 1.17 € and 0.86 € per kg P, K, CAN-N and urea-N, respectively (mean of 2014 and 2015 prices, CSO, 2016, available online at: http://www.cso.ie/multiquicktables/quickTables.aspx?id=ajm05).

2.1.5 Grass N content

Default silage N content is set at 14% crude protein (Keady, 2012). The herbage quality sub-model (Hoekstra and Schulte, 2008) predicts the effect of herbage management (N application rate, rotation length and cutting height) on the N content of the ingested herbage during grazing for early, mid- and late season, taking into account the effect of localised urine and dung deposition on herbage N content (Hoekstra and Schulte, 2008). The original model tended to underestimate or overestimate the herbage N content in late and early season, respectively. Therefore, we have applied a seasonal correction factor to ensure realistic seasonal variation in grass N content based on the validation results by Hoekstra and Schulte (2008).

2.1.6 Inhibitors: nitrification inhibitor Dicyandiamide (DCD) and urease inhibitor N-(n-butyl) thiophosphoric triamide (NBPT)

There is an increasing body of evidence that N losses from grassland systems can be reduced by applying nitrification inhibitors (most commonly Dicyandiamide) and / or urease inhibitors (NBPT) (Watson et al., 2008; Smith et al., 2008; Monaghan et al., 2009; Singh et al., 2013; Harty et al., 2016; Forrestal et al., 2016a). Nitrification inhibitors slow down the rate of the first step of the nitrification process, the conversion of ammonium (NH_4^+) to nitrite, and thus to nitrate (NO_3^-), by deactivating the responsible enzyme (Amberger, 1989). This can decrease NO_3^- leaching losses following applications of fertilizer N, livestock slurry or urine returns from grazing livestock. Additionally, N_2O emissions from both nitrification and denitrification will also be reduced by inhibiting nitrification, offering a potential mitigation strategy for greenhouse gas emissions from agriculture. Harty et al. (2016) observed a 73% reduction in N_2O emissions when calcium ammonium nitrate (CAN) was replaced by urea with NBPT. In addition the urease inhibitor NBPT reduces NH_3 (ammonia) loss from urea by delaying hydrolysis, Forrestal et al. (2016a) observed a 79% decrease from urea with NBPT compared to straight urea.

In the model, DCD can be applied as a spray on grassland, or mixed in with slurry, feed or CAN or urea fertiliser, whereas NBPT can be coated to urea granules (Table 2). The application of the inhibitors directly affects emission factors from the targeted sources (Table 2) as outlined in Table 3. The effects of the inhibitors on the agronomic response was set to zero, as experimental work to date show low and often inconsistent yield responses, particularly at the relatively low rates of application in practice (Forrestal et al., 2017; Harty et al., 2017). The N that is not lost as a result of these inhibitors is allocated to the soil N balance pool (section 2.1.8)

2.1.7 N losses

Ammonia volatilisation, N_2O emissions and nitrate leaching as a result of nutrient applications in the form of fertiliser (CAN, Urea), dung, urine, slurry, dilute slurry, and soiled water are calculated by multiplying the N load of the source by the best available emission factors for Irish moderately drained grasslands (Table 3). The N_2 to N_2O ratio is used to model the associated N_2 emissions (Table 3). The effect of the inhibitors DCD and NBPT on emissions is modelled by applying a reduction factor (Table 3). Ammonia losses from storage and housing are based on the ammonia emissions inventory by Misselbrook et al. (2000) adjusted for Irish conditions by Hyde et al. (2003) (Table S2). Emissions from N deposited on pathways is assumed to be 80% of TAN (Hyde et al., 2003). Background emissions refer to grassland emissions with no agricultural activity and are set at 2.4 kg N ha^{-1} (Li et al., 2011; Minogue, 2011) for N_2O emissions with a N_2 to N_2O ratio of 1.9 and at 1.7 kg N ha^{-1} for NO_3- leaching (Stark et al., 2007).

2.1.8 Soil N

The soil N balance is the difference between the field N inputs (fertiliser N, atmospheric N, soiled water and slurry N, dung and urine N), N outputs (animal N uptake) and N losses (leaching and gaseous emissions) and represents the change in soil N stock. However, it also includes the error of all other terms.

2.2 Sensitivity analysis

In the sensitivity analysis we assessed the effect of changes (range -20% to + 100% of original level in baseline scenario) in fertiliser N application rate and concentrate feeding rate on milk production, net farm profit and N loss.

2.3 Scenario analysis

We applied the model to test a number of different scenarios in which we simulate different approaches for increasing milk production in comparison to a baseline scenario. In this scenario analysis the farm area was held constant, and stocking rate was adjusted to reflect the change in feed availability whether in the form of herbage or concentrate (see also 2.1.1).

2.3.1 Baseline scenario

The baseline scenario represents a "typical" Irish dairy farm (Table 4), with 79 cows on 48 ha, resulting in a stocking rate of 1.83 livestock units ha^{-1} (equivalent to one cow ha^{-1}, Dillon et al. 2018). Average cow live weight is 561 kg, the mean calving date is the 15th of February and the culling % is 19.8. Fertiliser N input is 160 kg N ha^{-1} and P and K application rates were set at 5 and 13 kg ha^{-1}, respectively (Lalor et al., 2010). These P and K rates are below the recommended application rates (Coulter and Lalor, 2008), and as a result, on average, dairy farm soils have a P index of 2 (equivalent to 3.1 – 5.0 mg L^{-1} Morgan's P which is below the optimum range of 5.1 – 8.0 mg L^{-1}, P index 3), and this is assumed to reduce the potential DMY, the adjustment factor used is 0.92 (see CF_P in Eq. 1). The average reseeding rate of the grassland area is 4% per year (Creighton et al., 2011), resulting in an average sward age of 13 years and CF_{Sward} is set at 0.9 (see Eq. 1). Fifty % of the available slurry is applied to silage ground in April, and the remainder in June, using splash-plate application. From January to March, artificial fertiliser-N is applied in the form of urea and from April onward in the form of CAN. Milk yield was set at 5102 L cow^{-1} (Milk fat = 3.94 %; Milk protein = 3.44 %) (Dillon et al. 2018).

2.3.2 Scenario IG: Increase in milk production through increasing gGrass production

In scenario IG, we simulated increased milk production per ha through increased grass production as a result of optimised fertiliser application and sward management. The inorganic fertiliser N application rate was increased from 160 to 250 kg N ha^{-1} (Table 4). Additionally, P and K fertiliser applications were optimised based on stocking rate and soil P and K index (Teagasc, 2016a). It is assumed that it takes three years to build up soil P and K levels to index 3, after which recommended maintenance only P and K rates are applied. Finally, we introduced an annual reseeding of 11% of the grassland area as recommended.

2.3.3 Scenario IC: Increased milk production through increased concentrate feeding rates

In this scenario, milk production was increased by increasing the concentrate feeding rate by 100%, from 834 kg to 1668 kg cow^{-1}, other inputs are as in the baseline scenario (Table 4).

2.3.4 Scenario IP: Increased milk production in a high profit grass based system

This scenario is based on the research farm at Teagasc, Animal and Grassland Innovation Research Centre, Moorepark as described in the road map for the dairy production sector, aimed at increasing farm profit through an increase in grass utilised per hectare by dairy cows and higher genetic merit fuelled by an increased supply of high EBI dairy replacements (Teagasc, 2016b). Grass production was increased through increased N, P and K fertilisation rates and more frequent grassland renewal (10% reseeding rate) (Table 4). Through better grazing management, the utilisation of grazed grass was increased from 72% (for the baseline scenario, based on Table S1) to 87%. Additionally, during October and November restricted grazing was applied (achieving similar level of grass uptake in 50% of the grazing time (Kennedy et al., 2011). Finally, the milk production per cow was increased from 5100 to 5600 L cow^{-1} and milk fat and milk protein were increased from 4.0 and 3.4% to 4.7 and 3.7 %, respectively, reflecting the higher genetic merit of the dairy cows. This is reflective of the current research target and is being achieved on a cohort of extremely technically efficient and productive farms.

2.3.5 Mitigation of N losses

For each of the scenarios we applied three measures for the mitigation of N loss:

1) **TS:** In this measure, slurry was applied with trailing shoe application instead of splashplate in order to reduce N losses from ammonium volatilisation (Table 4) (Lalor, 2008). To maximise the mitigation potential, the application of artificial fertiliser N was decreased to compensate for the associated increase in available N from slurry. The extra cost associated with trailing shoe application compared to splashplate was 0.77 € m^{-3} slurry applied (Lalor, 2008). Additionally, all slurry was applied in April (as opposed to 50% in the baseline scenario) in order to reduce the ammonia N losses associated with weather conditions in June.
2) **INH**: In this measure, we added a suite of mitigation options in the form of urease and nitrification inhibitors. This included replacing CAN and urea fertilisers with urea stabilised

with NBPT and the addition of DCD to animal feed (Table 2). It is assumed that the extra N fed to the animals in the form of DCD is fully excreted, and fertiliser application rates are adjusted to compensate for this increased availability of N.

3) **INH + TS:** In a final measure we combined both the TS and INH mitigation options.

3 Results

3.1 Farm productivity

The modelled total herbage DM yield ranged from 10.1 ton DM ha^{-1} year^{-1} for the baseline scenario to 13.7 ton DM ha^{-1} year^{-1} for scenario IG and IP (Table 5). The increased use of concentrate feed (scenario IC) led to a 0.1 ton DM ha^{-1} year^{-1} increase in herbage yields due to increased N, P and K recycling in manure (as a result of increased animal numbers).

The stocking rate ranged from 1.8 livestock units ha^{-1} in the baseline scenario to 2.5 livestock units ha^{-1} for the IG scenario. Fat and protein corrected (FPC) milk production ranged from 8.3 tonne ha^{-1} for the baseline scenario to 13.3 tonne ha^{-1} for scenario IP, an increase in milk production of 61%. Milk production for scenario IG and IC was 11.5 and 9.6 ton FPC ha^{-1}, respectively, equivalent to a 38% and a 16% increase compared to the baseline scenario, respectively (Table 5).

The mitigation scenarios had little or no effect on farm productivity, as all potential effects on production (e.g. higher N utilisation from slurry with TS application) were offset by a reduction in fertiliser N application rates in order to maximise the mitigation potential (data not shown).

3.2 Farm net profit

Farm net profit of the baseline scenario was 301 € ha^{-1} (Table 5). The IG scenario increased the farm net profit to 634 € ha^{-1} (498 € ha^{-1} during the 3-year build-up phase for soil P and K levels), as the extra costs associated with fertilisers and reseeding were easily off-set by an increase in income from milk production. In contrast, increasing the concentrate feeding rate in the IC scenario resulted in a 67%

reduction to 99 € ha^{-1} in farm net profit. The IP scenario dramatically increased farm net profit to 1574 € ha^{-1}, representing an increase of 1272 € ha^{-1} above the baseline scenario mainly due to improved grass utilisation (which decreased concentrate use) coupled with increased milk yield per cow that resulted from higher genetic merit animals.

The increased costs of applying slurry by trailing shoe (compared to splash plate) was partly offset by the decrease in artificial fertiliser requirement (reduction of 6 - 8 kg N ha^{-1}), and the net effect on farm net profit was very small (Table 6). The application of inhibitors increased the fertiliser costs by 54 € ha^{-1} (IC), 72 € ha^{-1} (IP) and 74 € ha^{-1} (IG), and this was directly at the cost of the farm net profit, which decreased by 55%, 5% and 11% for IC, IP and IG, respectively.

3.3 Nitrogen input, output and losses

3.3.1 N inputs

Average farm-gate level N inputs (inorganic fertiliser and concentrates) ranged from 211 kg N ha^{-1} in the baseline scenario to 318 kg N ha^{-1} in scenario IG, an increase of 107 kg N ha^{-1} (Figure 3) as a result of higher fertiliser N application rates and increased concentrate feeding rates (due to the higher stocking rate). In scenario IC, the increased concentrate feeding levels resulted in an increased N input of 49 kg N ha^{-1}. The N input for the IP scenario was increased by 80 kg N ha^{-1}, as higher N application rates were partially off-set by lower concentrate feeding rates. The TS mitigation scenario resulted in a reduced mean N input level of 9 kg N ha^{-1}.

3.3.2 N output

Nitrogen output in milk and meat ranged from 56 kg N ha^{-1} in the baseline scenario to 79 kg N ha^{-1} in the IP scenario, roughly following the milk production (Figure 3). Farm-level N use efficiency (NUE = N output / N input) was 27%, 24%, 25% and 27% for the baseline, IG, IC and IP scenario, respectively. The mitigation scenarios had no effect on the N output. The soil N balance ranged from 47 kg N ha^{-1} for the baseline scenario to 80, 68 and 48 kg N ha^{-1} for scenario IG, IC and IP, respectively. The use of inhibitors increased the soil N balance by 36 (IC) to 46 (IG) kg N ha^{-1}.

3.3.3 Total N loss

Total N animal excretion ranged from 184 kg N ha^{-1} in the baseline to 276 kg N ha^{-1} in the IP scenario (Table S4). In the baseline scenario N excretion during grazing accounted for 109 kg N ha^{-1} but this increased substantially to 161 kg N ha^{-1} and 157 kg N ha^{-1} in scenario IG and IP, respectively. The quantity of N excreted applied in the form of slurry increased from 54 in the baseline to 87 kg N ha^{-1} in the IP scenario.

Total N loss (including ammonia, nitrate and nitrous oxide, but excluding the environmentally benign N_2) was 80 kg N ha^{-1} for the baseline scenario (Figure 3b). The N loss increased for all intensification scenarios to a maximum of 120 and 124 kg N ha^{-1} for IG and IP, respectively. Nitrogen losses from IC were intermediate at 93 kg N ha^{-1}. The TS mitigation scenario resulted in a reduction of, on average, 8% in total N loss, whereas the inhibitor scenario reduced total N loss (excluding N_2) by, on average, 13%. Even in the combined mitigation scenario, the total N loss (excluding N_2) for the IG (96 kg N ha^{-1}) and IP (99 kg N ha^{-1}) scenario remained above the baseline scenario losses. The milk N footprint (N loss expressed in kg N tonne^{-1} FPC milk produced) was 9.5 (12.9 incl. N_2) kg N tonne^{-1} FPC milk for the baseline scenario (Figure 3c). For the IG intensification scenario with no mitigation, the milk N footprint was higher than the baseline scenario, whereas for IC and IP the milk N footprint was at baseline level. The TS mitigation resulted in a small reduction of N footprint to around the baseline level, whereas the use of inhibitors resulted in a decrease in milk N footprint to below the baseline scenario for all intensification scenarios.

3.3.4 Ammonia

Ammonia volatilisation was the largest loss pathway and accounted, on average, for 48% of total N loss. The main sources of ammonia volatilisation were slurry application, housing and storage, grazing and fertiliser application (Figure 4a). Ammonia N loss ranged from 51 kg N ha^{-1} for the baseline scenario to 82 kg N ha^{-1} for the IP scenario (Figure 4a). The increase in ammonia N loss was 54%, 20% and 62% for the IG, IC and IP scenario, respectively. The TS mitigation scenario reduced ammonia losses by, on average, 11% (decrease in losses from slurry spreading), but total losses remained above the baseline scenario. The use of inhibitors had very little effect on total farm ammonia losses (1% decrease); the decrease in ammonia loss from urea fertiliser with NBPT was partially off-set by marginally increased ammonia losses as a result of DCD application. The ammonia N loss per tonne FPC milk was higher than the baseline scenario for all the intensification scenarios, but reduced to the baseline level or below, in the TS mitigation scenario (Figure 5a).

3.3.5 Nitrate

Nitrate leaching was 24 kg N ha^{-1} in the baseline scenario. Nitrate leaching comprised 18% to 23% of total N losses and was mainly the result of losses during grazing and fertiliser application (Figure 4b). Similar to ammonia, the intensification scenarios resulted in a strong increase in nitrate N loss, ranging from 14% for IC (to 27 kg N ha^{-1}) to 49% and 50% for IG and IP, respectively, (to 36 kg N ha^{-1}). Trailing shoe application resulted in a minor reduction in nitrate leaching (1%, due to lower fertiliser N application rates). The use of a nitrification inhibitor resulted in a substantial reduction in nitrate leaching of 37% to below the baseline level (Figure 4b). The intensification scenarios IP and IG resulted in a small increase in the nitrate N loss per tonne FPC milk. Application of the INH mitigation scenario resulted in a 36% reduction of the nitrate N loss per tonne FPC milk (Figure 5b), to well below the baseline level.

3.3.6 Nitrous oxide

Nitrous oxide (N_2O) losses ranged from 5.8 kg N ha^{-1} in the baseline to 7.4 kg N ha^{-1} in scenario IG and IP (Figure 4c), equivalent to, on average, 5% of total N loss. Direct N_2O losses were the dominant form of N_2O loss accounting for 3 to 5% of total N losses compared to <1% for indirect N_2O emissions from wet/dry deposition and leached N sources. The main sources of N_2O were fertiliser N, grazing losses, background emissions, and indirect emissions (Figure 4c). Trailing shoe application reduced N_2O emissions by 2% (due to lower indirect emissions), whereas the inhibitor scenario resulted in a 27% decrease (mainly through lower losses from grazing and fertiliser), to below the baseline scenario level

(Figure 4c).The N_2O emissions per tonne FPC milk produced was lower for all scenarios compared to the baseline scenario (Figure 5c).

3.3.7 N_2

After ammonia volatilisation and nitrate leaching, N_2 loss was ranked the third highest source of N loss from all scenarios. The N_2 loss ranged from 28.1 kg N ha^{-1} in the baseline scenario to 40.1 kg N ha^{-1} in scenario IG (Figure 4d). The main sources of N_2 loss was grazing with a relatively large contribution of background emissions. The trailing shoe application had no effect on N_2 emissions, but the inclusion of inhibitors reduced N_2 loss by 58% to below 17 kg N ha^{-1} for all scenarios, which is well below the baseline level.

4 Discussion

4.1 Model

The individual sub-models on which this model is based have been previously published (Hoekstra and Schulte, 2008; Schulte and Herlihy, 2007; Shalloo et al., 2004). The emission factors in the N loss modules for NH_3, NO_3^-, N_2O and N_2 are based on best available experimental data for Irish grasslands on moderately drained soils (Misselbrook et al., 2000; Fischer et al., 2016; Forrestal et al., 2016a; Misselbrook et al., 2014; Kim et al., 2012; Harty et al., 2016; Krol et al, 2016; Bourdin et al., 2014; Minogue, 2011; Clough et al., 2007; Cahalan et al., 2014; Li et al., 2014; Stevens and Laughlin, 2001; Monaghan and Barraclough, 1993; McGeough et al., 2012; Stark et al., 2007; Dennis et al., 2012). However, there is considerable uncertainty associated with these emission factors with the principal sources of variation being N form, soil type and climate. The current paper represents an "average" scenario in terms of losses. The model is designed in such a way that it is easy to update with the latest emission factors and to include (additional) effects of seasonal variation and for example soil drainage class on loss pathways and N utilisation if more detailed information becomes available

The model sensitivity to different emission factors (EF) was assessed by varying the main EF's associated with ammonia volatilisation, nitrate leaching and N_2O for the baseline scenario to high/low extremes observed for pasture-based bovine farms/soils (see Table 7) . Uncertainties associated with ammonia EF's were the largest source of variation for total N loss from farms, with EF's for urea fertiliser, slurry spreading and animal housing the largest source of variation (39.1 kg N ha^{-1}). To contextualise this value, it equates to the difference across all production systems, apart from the baseline in this study. The variation in EF's associated with nitrate leaching (fertiliser and urine/dung deposition) resulted in 67% difference (11.9 kg N ha^{-1}) in modelled N leaching, while varying N_2O EF's resulted in 231% difference in N loss (6.7 kg N ha^{-1}). This large variation in N_2O modelled outputs reflects the large variation observed for N_2O EF's in studies, particularly for CAN and urine deposition, where six-fold and three-fold variations in respective EF's have been observed (Stehfest & Bouwmann 2006, Kuikman et al. 2006, Harty et al. 2016). When the lowest and highest EF's for ammonia, nitrate and N_2O were applied for the IG, IC and IP scenarios, the total reactive N loss varied by 81% for IG and IP scenarios and 67% for the IC scenario.

While a complete model validation is not feasible, due to the lack of sufficient data for the total set of required input and output parameters, in general our model outputs were well within the range of values reported in the literature, as outlined below:

Burchill et al. (2016) completed a system N balance for an intensive grazed grassland (perennial ryegrass/white clover) dairy farm stocked at 2.3 cows ha^{-1} and a N input of 257 kg N ha^{-1} (divided between fertiliser and N fixed by white clover) (similar to scenario IG) on a poorly drained soil in Ireland. Burchill et al. (2016) estimated that 79 kg N ha^{-1} left the farm as product, 190 kg N ha^{-1} was lost to the wider environment and the NUE was 33%. This is similar to scenario IG which estimated N outputs of 78 kg N ha^{-1}, environmental losses of 160 kg N ha^{-1} and a NUE of 24%.

In our study NUE ranged from 24 to 28% for all scenarios, which is at the lower end of the range 18-40% reported for European and New Zealand dairy systems (Mihailescu et al. 2013).

Similar to scenario IG, Burchill et al. (2016) reported that N loss to the environment was dominated by NH_3 and N_2. Mean losses for NH_3 and N_2O were 69 and 7 kg N ha^{-1}, respectively, which is similar to 78 and 7.4 kg N ha^{-1}, respectively, for scenario IG. The losses of N_2 and NO_3 reported by

Burchill et al. (2016) were 56 and 22 kg N ha^{-1}, respectively, which differed from the losses in scenario IG which were 40 and 36 kg N ha^{-1}, respectively. This was related to the soil type in Burchill et al. (2016) being a poorly drained gley soil resulting in higher N_2 emissions and lower leaching losses (Jahangir et al. 2013, McAleer et al. 2017).

The modelled NH_3 emissions in our study ranged from 51 to 82 kg NH_3-N ha^{-1} yr^{-1}. These are similar to other dairy farm scale ammonia emission estimates which are reported to vary from 36 to 105 kg NH_3-N ha^{-1} yr^{-1} (Bussink and Oenema 1998, Jarvis and Ledgard 2002). In Ireland, Humphreys et al. (2008) reported emissions ranging from 31 to 59 kg NH_3-N ha^{-1} yr^{-1} in four grassland-based systems of dairy production. In general, the majority of ammonia emissions are sourced from housing and landspreading, 31% and 30% respectively (Hyde et al., 2016).

The NO_3 leaching rates for the scenarios ranged from 23 to 36 kg N ha^{-1}, which is in line with estimated N leaching reported in Ireland of 17-82 kg N ha^{-1} (Jahangir et al. 2013, McAleer et al. 2017) and New Zealand of 11-34 kg N ha^{-1} (Ledgard et al. 1999).

Estimates of N_2 emissions from grassland soils are very rare due to methodological limitations and high uncertainties (Selbie et al. 2015). Our study estimated that N_2 emissions ranged from 28 to 40 kg N ha^{-1} and that total denitrification (N_2+N_2O) emissions ranged from 34 to 48 kg N ha^{-1}. A few studies using the acetylene inhibition method have estimated similar total denitrification emissions range from 11 to 34 kg N ha^{-1} in New Zealand (Ledgard et al., 1999), 13 to 15 kg N ha^{-1} in Australia (Eckard et al., 2003) and 127 to 143 kg N ha^{-1} yr^{-1} on clay grassland soils in the Netherlands (van der Salm et al., 2007).

The baseline N surplus (soil N balance and N loss) was 154 kg N ha^{-1} which was at the upper end of the range reported by Van Grinsven et al. (2012) for the EU 27 of 56 to 176 kg N ha^{-1}. While the N loss to the environment is high at 107 kg N ha^{-1} in the baseline scenario, this is dominated by environmentally benign N_2 emissions. This is in line with the observations by van Grinsven et al. (2012) who noted that the high N surplus in Ireland did not breach the nitrate targets in the Nitrates Directive.

In a comparative study of NUE on dairy farms in New Zealand, the Netherlands and Ireland, Pinxterhuis et al. (2015) reported NUE ranged from 32 to 47%, NH_3 volatilisation ranged from 50 to 80 kg N ha^{-1}, NO_3 leaching ranged from 12 to 72 kg N ha^{-1} and total denitrification ranged from 15 to 64 kg N ha^{-1}. Within this study, systems with lower NO_3 leaching had higher total denitrification losses.

4.2 Scenario analysis

A previous study by Vibart et al. (2015) looked at a range of scenarios to reduce nutrient loss to water and greenhouse gas emissions for New Zealand systems. The scenarios they tested were improved nutrient management, improved animal productivity, and restricted grazing. In their study they highlighted that improved nutrient management and animal productivity reduced nitrate leaching by 9 and 18%, respectively, reduced greenhouse gas emissions by 2.5 and 5.2% and increased farm profits of 1 and 5%. Restricted grazing through removal of cows during the winter (normal farm management in the EU) reduced NO_3- leaching by 34% and greenhouse gas emissions by 7.2% but reduced farm profits by 14%. The scenarios tested in our work went beyond those in Vibart et. al. (2015) where we investigated the effect of moving from a grassland based system to a more European concentrate based system and included a range of technologies, as discussed below.

4.2.1 Milk production: achieving the Food Harvest 2020 and Food Wise 2025 targets

Our results show that the increase in milk production of 50% (Food Harvest 2020 and Food Wise 2025) cannot be achieved through the intensification measures of increased grass production (scenario IG, +38% milk production) or increased concentrate feeding levels (scenario IC, +16% milk production) based on the assumptions included in this analysis. Implementing a high profit grass based system aimed at increasing farm profit through higher grass production, grass utilisation and genetic merit of the dairy cows (scenario IP) resulted in an increase in milk production of 44%. This suggests that these targets may be at the upper limit of what can be realistically achieved, without increasing the total area for dairy production.

4.2.2 Economics

The IC scenario resulted in a strong reduction of the farm net profit, as a result of a strong increase in concentrate costs. In contrast, measures to increase the grass production (IG) resulted in a doubling of the farm net profit. As expected, the high profit grass based system (IP) had the largest effect on the farm net profit, which showed a five-fold increase. The TS mitigation scenario had a negligible effect on the farm net profit, as the increased cost associated with slurry application was more or less offset by a reduction in fertiliser costs. The extra costs associated with the application of inhibitors ranged from 54 to 72 Euro ha^{-1} and are much lower than the increase in farm net profit for scenario IG and IP.

4.2.3 Soil N balance

The soil N balance ranged from 47 kg N ha^{-1}(baseline) to 127 kg N ha^{-1} (IG + TS and inhibitors). An increased soil N balance may be an indication of potential long term effect on either N uptake or N losses. However, the soil N balance must be interpreted with caution, as the soil N balance is calculated as the 'unaccounted' mass balance of N in the model. This means that this fraction also includes the error terms of the preceding computations of each N loss pathway. In a recent publication, Uwizeye *et al*. (2016) presented a new framework for assessing the NUE of livestock supply chains, in which they reported on the virtual absence of complete empirical datasets that include measurements of all N flows and balances, including soil N balances. In most studies, at least *one* N flow or stock change is estimated, rather than measured, on the basis of mass balance. In most cases, as in the study reported here, this is indeed the Soil N Balance (see e.g. Velthof *et al*., 2009), but with notable exceptions (e.g. Özbek and Leip, 2015). The implications of this uncertainty were discussed in detail by Oenema *et al*. (2003).

4.2.4 N loss: consequences for the environment

All the intensification scenarios resulted in increased total N losses per ha compared to the baseline (Fig. 3a), ranging from 19 kg N ha^{-1} extra for scenario IC to 55 kg N ha^{-1} extra for scenario IP.

The increased losses compared to the baseline scenario tended to be highest for ammonia, both in absolute and relative terms. This was related to increased losses from slurry spreading, housing and grazing as a result of the increase in stocking rate. Additionally, in all scenarios except IC, a higher N fertilisation rate increased the ammonia losses from fertiliser application.

Slurry application by trailing shoe in spring reduced ammonia emissions by on average 11%, but the losses remained above the baseline scenario. The use of inhibitors had a minimal impact on the losses through ammonia volatilisation as the decrease in ammonia loss from urea fertiliser with NBPT was partially off-set by marginally increased ammonia losses as a result of DCD application.

For nitrate losses the increase compared to the baseline scenario was mainly related to increased losses from fertiliser N application and from grazing (as a result of the higher stocking density). However, the inhibitors were very effective in negating these increases, even to below the baseline level.

For N_2O emissions the increase compared to the baseline scenario was smaller, both in absolute and relative terms (partly due to the diluting effect of background emissions). Again, the inhibitors effectively counteracted these increased losses to below the baseline scenario levels, whereas the trailing shoe mitigation had a minimal impact (through reduced fertiliser N application rates).

If we look at the milk footprint of the scenarios (N losses per ton of fat and protein corrected (FPC) milk produced), all the intensification scenarios resulted in an increase in ammonia loss compared to the baseline, but the increase in N footprint was lower compared to the absolute increase, due to the increased milk yields (Figure 5a). The trailing shoe mitigation reduced ammonia losses to baseline level or just below (IC and IP). For nitrate, only scenario IP gave a lower milk footprint than the baseline scenario (Figure 5b), whereas the N_2O footprint was lower for all scenarios (Figure 5c). The application of inhibitors reduced the nitrate and N_2O footprint to well below the baseline scenario. This would imply that from an environmental point of view it is more efficient to achieve the increase in milk production through one of the proposed scenarios with the mitigation measures, as opposed to an alternative expansion scenario, in which the area for milk production is increased (without changing the production system). It should be noted that the current study focuses on the emissions within the farm gate and excludes emissions associated with imported feedstuffs, fertilisers and changes in fossil fuel use.

Scenario IC showed only a modest increase in N losses compared to the baseline. This relatively low environmental response to the intensification in this scenario can be explained by the reliance of scenario IC on increased concentrate imports. In contrast to scenario IC, scenario IP actually decreased the dependence on imported feedstuffs compared to the baseline scenario. The N losses associated with the production of both concentrates and fertiliser take place outside the system boundary of the €riN model, and as a result do not show up in Figure 3, which must therefore be interpreted with caution. Upstream emissions in terms of feed and fertiliser manufacture and downstream emissions (transport, refrigeration) in intensive livestock production (dairy, beef, pig meat) can account for 32%-24% of total livestock emissions, with approximately 40% arising from energy emissions and 60% from land-use emissions (Weiss & Leip, 2012) with the proportion of upstream/downstream emissions falling significantly for less intensive and more localised production systems (Mottet et al., 2017). As such, there is extra potential mitigation associated by reducing both concentrate feed and fertiliser inputs (Notarnicola et al., 2017). A life-cycle analysis assessment is required to establish the 'nutrient footprint' of farming systems that rely to a large extent on feed imports. Recently, Uwizeye *et al*. (2016) developed a comprehensive assessment methodology that allows for the segregation and quantification of nutrient losses associated with herbage production and both on-farm and off-farm production of animal feeds.

Further increases in NUE and concomitant reductions in fertiliser N use could be gained from optimizing soil nutrient status and adding N-fixing legumes onto swards. In terms of soil fertility, currently two-thirds of Irish soils are at sub-optimal pH. While targeting a similar grass yield, by increasing the soil pH from 5.5 to 6.3 with lime application the N fertiliser required could be reduced by up to 70 kg N $ha^{-1}yr^{-1}$ (Culleton et al., 1999). Additionally increasing the soil pH from 5.4 to 6.3 with lime application led to on average 5.3 kg ha^{-1} additional P uptake by the grass sward in the following three growing seasons (Fox et al., 2015). The inclusion of legumes (clover) into swards can also fix on average 80-120 kg N ha^{-1} yr^{-1}, with low associated N emissions (Burchill et al., 2016). This could allow for a further decoupling of emissions from production. Reduced crude protein content in bovine diets would also reduce N loading across the whole farm system. However, in a pasture-based system, there is less opportunity to control the protein content of diets, and therefore this option would be most impactful in a high concentrate scenario.

4.2.5 Implications for sustainable intensification

Our results suggest that stark choices may be required in the quest for sustainable intensification. Figure 3 suggests that it is technically possible to increase the milk output of a 'typical' dairy farm through a process of 'knowledge intensification', specifically in scenarios IG and IP. The environmental impact of this intensification is somewhat of a paradox: on the one hand, the environmental losses per litre of milk are projected to fall in these two latter scenarios, suggesting a smaller N footprint. On the other hand, total nutrient losses per hectare, or per farm, are projected to increase, even when both TS and inhibitor mitigations are applied.

This raises a difficult dilemma which type of sustainable intensification is required: reducing ‘emission intensities’ (losses per plate of food) or reducing total losses (per hectare, per farm)? Here we suggest that the answer is “a bit of both” and depends on the impact category and spatial scale that is of interest. Some environmental impacts take place at local level, for example: the impact of agriculture on local water quality. Technically, it is not possible to ‘offset’ a local decline in water quality with an increase elsewhere (Schulte *et al*., 2015). In this case, environmental pressure is best described by total nutrient losses per unit area. At the other extreme, the impact of greenhouse gases is global, which means that these emissions can be offset between regions in efforts to optimise land use to meet both food security concerns (i.e. increased production) and greenhouse gas reduction targets (*ibid*). In that case, the emission intensity may be the more appropriate indicator of sustainability (Schulte *et al*., 2011; O’Brien et al., 2012).

The use of urease and nitrification inhibitors resulted in a substantial decrease in N losses. However, the widespread use of the nitrification inhibitor DCD is problematic, as its transfer and persistence into agricultural products is not well understood and this has led to a potential problem of perception by the agricultural industry and the general public. Further research is required investigating the animal and human health risks associated with the use of DCD before it could be considered as an acceptable mitigation measure.

While increases in production efficiency is a win-win situation that leads to lower emissions per unit product and lower costs to the producer, in order to realise absolute reductions in N losses, activity levels (either production volume or animal numbers, N inputs) would have to be held constant. However, the supply response of farmers to increased profitability also needs to be considered and greater profitability may, in turn, lead to increased overall production, offsetting some of the improvement in emissions intensity. These rebound and backfire effects that result from increased efficiency have been documented for various sectors (Barker et al., 2009, Frondel et al., 2013). However, limiting production could also lead to significant ‘leakage’ of emissions to less efficient production systems. A recent study by the EU Joint Research Centre on the impact of 2030 GHG reduction targets on agriculture at an EU level found that implementation of a *pro-rata* reduction in agriculture across member states would result in a) adverse impacts on agricultural production in most member states and the EU as a whole and b) a net increase in global agricultural greenhouse gas emissions as production moved to less emissions efficient countries (Fellmann et al., 2018).

5 Conclusions

Increased global demand for dairy produce and the abolition of EU milk quotas has resulted in expansion in dairy production across Europe but most notably in Ireland. The resultant intensification of agriculture has the potential to reduce N emissions intensity, but also to increased absolute N emissions due to increased animal numbers, higher feed and/or fertiliser inputs, and also to potentially increase system profitability which may in turn fuel further farm expansion. Mitigation strategies, such as low ammonia loss landspreading methods and the use of urease and/or nitrification inhibitors can play a significant role in ameliorating this increase in emissions, but may not entirely eliminate absolute increases. Further improvements in farm nitrogen use efficiency and alternatives to mineral fertilisers will be required to decouple production from reactive N emissions.

Acknowledgments

This research was financially supported under the National Development Plan, through the Research Stimulus Fund, administered by the Department of Agriculture, Food and the Marine (Grant numbers RSF10-/RD/SC/716 and RSF11S138) and by the New Zealand Government to support the objectives of the Livestock Research Group of the Global Research Alliance on Agricultural Greenhouse Gases (MPI/AgResearch Contract No. 15811). Any view or opinion expressed does not necessarily represent the view of the Global Research Alliance.

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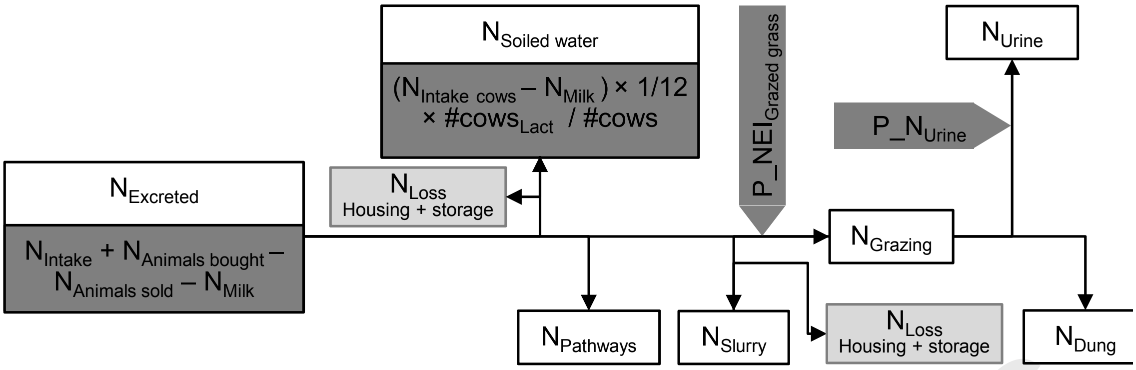


Figure 1. Overview of calculation of total N excretion (kg N ha^{-1}) and its partition over slurry, soiled water, N deposited on pathways and dung and urine patches during grazing as described in section 2.1.3. White boxes refer to N pools, light grey boxes indicate loss pathways, and calculations and loss factors are indicated in dark grey. For abbreviations see Table 1.

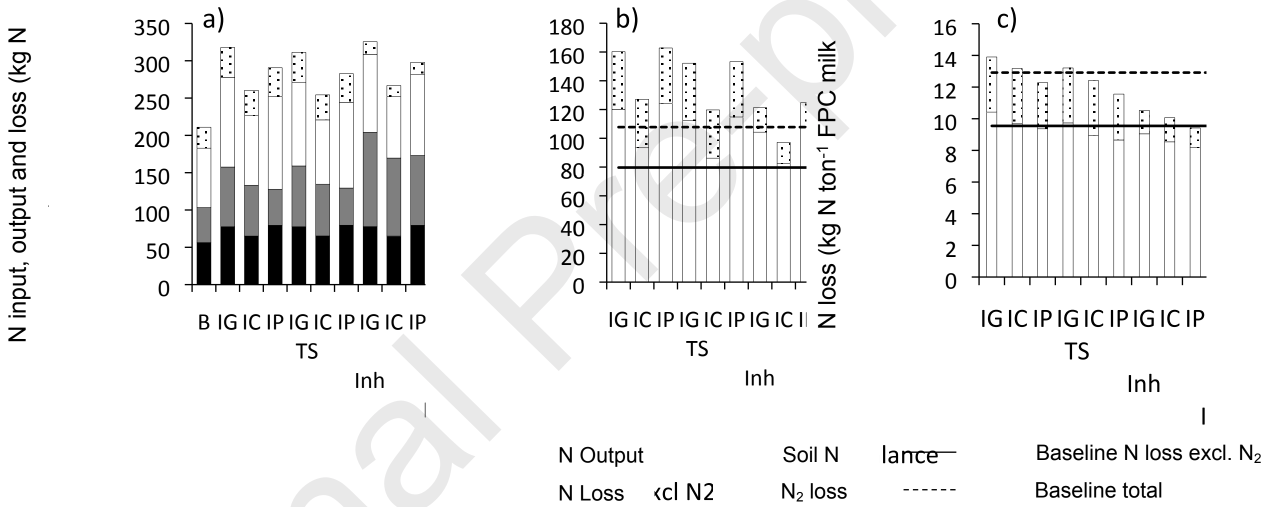


Figure 3. N output (in milk and meat), soil N balance and N loss (including or excluding the environmentally benign N_2) expressed in kg N ha^{-1} (a and b) and kg N per tonne fat and protein corrected (FPC) milk produced (c), for the different scenarios for increased milk production: B) baseline, IG) increased grass production, IC) increased concentrate feeding rates and IP) high profit grass based system (section 2.3 and Table 3) and the mitigation scenarios: TS) trailing shoe slurry application in spring, Inh) addition of urease and nitrification inhibitors to fertilisers and feed (For more details see section 2.3.5). N input is represented by the cumulative bars in Figure 3a.

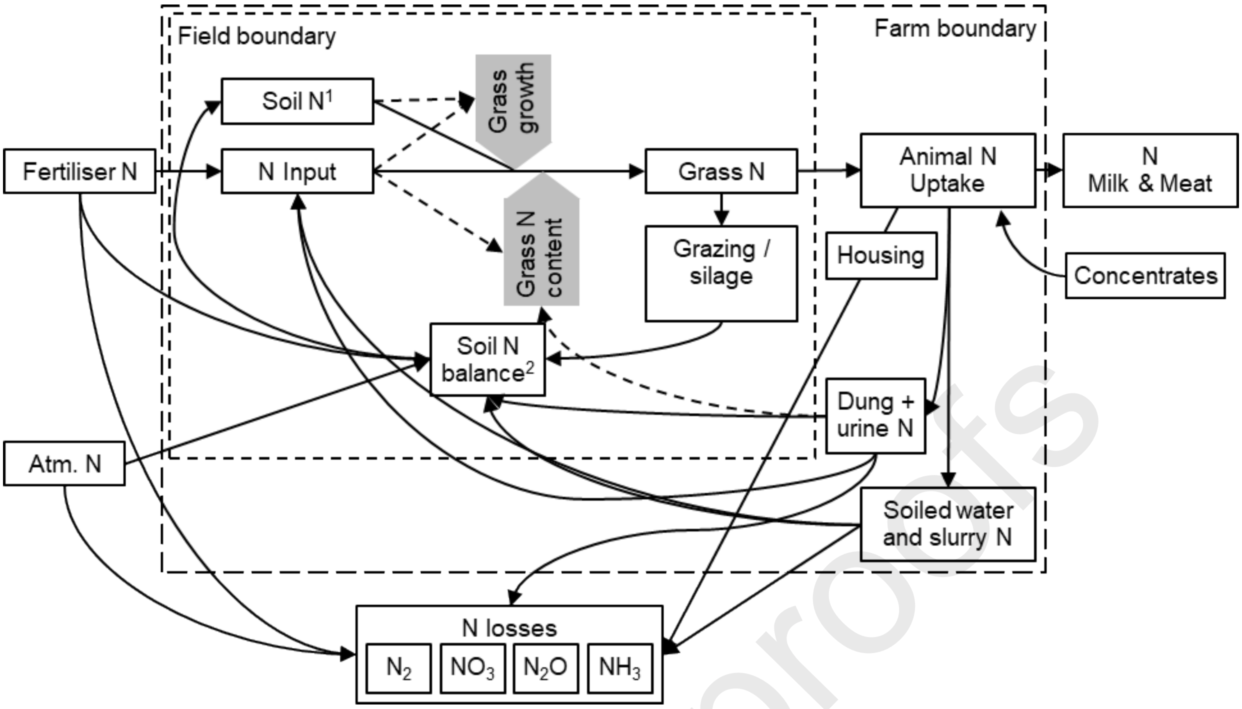
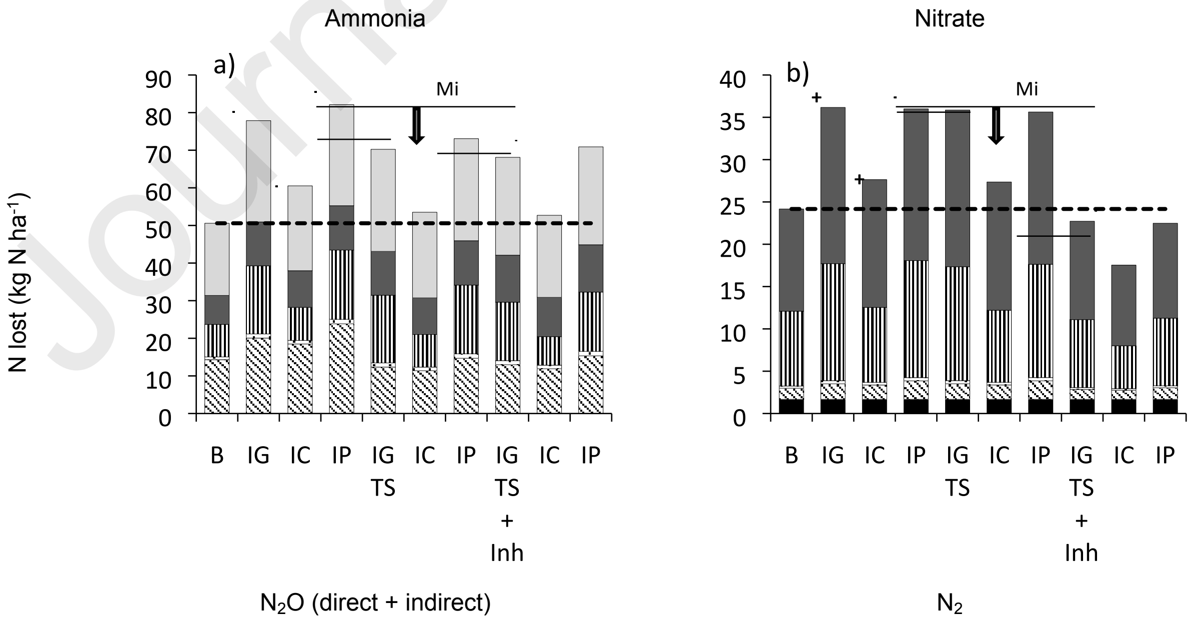


Figure 2. Overview of model N flow.

[1]Soil N available for grass growth is a “pull” function which depends on the intercept of the N response curve (eq. 2) and not on “modeled” soil N availability.

[2]The soil N balance is the difference between the field N inputs (fertiliser N, atmospheric N, soiled water and slurry N, dung and urine N), N outputs (animal N uptake) and N losses (leaching and gaseous emissions) and represents the change in soil N stock.



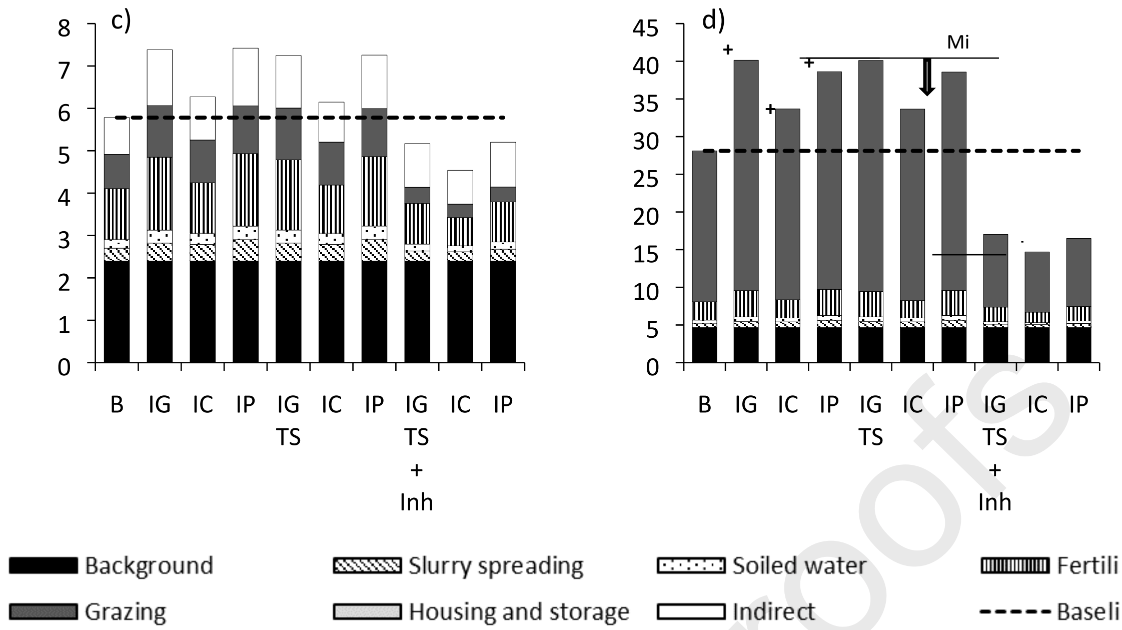


Figure 4. The N loss (kg N ha^{-1}) from different sources (Background, slurry and dirty water spreading, fertiliser spreading, grazing and storage and housing) in the form of a) ammonia, b) nitrate and c) N_2O for the different scenarios for increased milk production: B) baseline, IG) increased grass production, IC) increased concentrate feeding rates and IP) high profit grass based system (section 2.3 and Table 3) and the mitigation scenarios: TS) trailing shoe slurry application in spring, Inh) addition of urease and nitrification inhibitors to fertilisers and feed (For more details see section 2.3.5)

Relative change in N losses in relation of the scenarios compared to the baseline scenarios are indicated in bold. The relative change in N losses as a result of the mitigation options is also indicated.

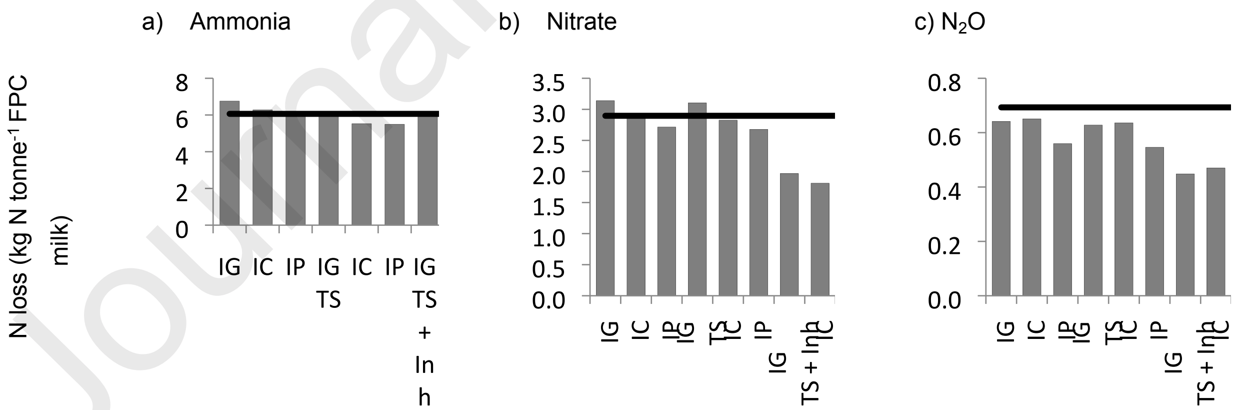


Figure 5. N loss per tonne fat and protein corrected (FPC) milk produced in the form of a) ammonia, b) nitrate and c) N_2O for the different scenarios compared to the baseline scenario (black line). Scenarios for increased milk production: IG) increased grass production, IC) increased concentrate feeding rates and IP) high profit grass based system (section 2.3 and Table 3). Mitigation scenarios: TS) trailing shoe slurry application in spring, Inh) addition of urease and nitrification inhibitors to fertilisers and feed (For more details see section 2.3.5)

Tables SUDEN model paper

Table 1. List of abbreviations

CAN	calcium ammonium nitrate fertiliser
DCD	Dicyandiamide (nitrification inhibitor)
CF	Correction factor
DMY	Dry matter yield
EF	Emission factor
FPCMilk	Fat and protein corrected milk production
GHG	Greenhouse gas
IG	Scenario in which milk production is increased trough increasing grass production
IC	Scenario in which milk production is increased trough increasing concentrate feeding rates
IP	Scenario in which milk production is increased through applying a high profit grass based system
INH	Mitigation strategy in which a suite of mitigation options in the form of urease and nitrification inhibitors are assessed.
MDSM	Moorepark Dairy System Model
N	Nitrogen
N_2O	Nitrous oxide
NBPT	N-(n-butyl) thiophosphoric triamide (urease inhibitor)
NE	Net energy
NEI	Net energy intake
NH_3	Ammonia
NO_3^-	Nitrate
NUE	N-use efficiency
Pot	Potential
P_NEI	Proportion of net energy intake
P_N_{Urine}	Proportion of N excreted in the form of urine
TAN	Total ammoniacal N
TS	Mitigation strategy in which slurry was applied with trailing shoe application instead of splashplate in order to reduce N losses from ammonium volatilisation

Table 2. Overview of the application options for nitrification inhibitors (DCD) and urease inhibitors (NBPT)

Inhibitor	Form	Application rate	Cost		Targets emissions from
			€ kg^{-1}*	€ kg N^{-1}	
DCD	Spray	20 kg ha^{-1} yr^{-1}	€3.30**		Urine, dung, fertiliser, slurry & soiled water (up to 3 months after application)
	Slurry	90 g kg^{-1} N applied	€3.30		Slurry

	Feed	15	g cow^{-1} day^{-1}	€3.30		Urine, dung, slurry, soiled water**
	Urea	35	g kg^{-1} N applied	€1.28	€0.045	Urea
	CAN	100	g kg^{-1} N applied	€1.28	€0.128	CAN
NBPT	Urea	2.4	g kg^{-1} N applied	€36.00	€0.087	Urea

*Prices are estimates based on New Zealand and the USA, as these products are currently not commercially available in Ireland

**Excluding contractor costs for spraying whole grassland area twice

***Extra N input in feed is assumed to be all excreted

Table 3. Overview of input values of emission factors for ammonia volatilisation, nitrous oxide and di-nitrogen emission and nitrate leaching (% of N applied) and the effect of inhibitors (DCD, NBPT) for the field application of the different N sources on moderately to well drained soils.

Source	NH_3 emissions			N_2O emissions				NO_3-leaching		
	EF (% N applied)	Change in emission factor (%) NBPT	DCD	EF (% N applied)	Change in emission factor (%) NBPT	DCD	N_2 ratio (N_2 / N_2O)	EF (% N applied)	Change in emission factor (%) NBPT	DCD
CAN	1.6 ,a		0[4]	0.87[6]		-39[6]	2[13]	5.5[8,16]		-40[17]
Urea	23	-78.5[3]	+3.6[5]	0.18[6]	+132[6]	-38[6]		5.5[8,16]	0[4]	-40
Dung: Spring	5.3			0.08[7]		-70	1.9	0		
Summer	2.8			0.23[7]		-70	1.9	0		
Autumn	3.5			0.62[7]		-70	1.9	0		
Urine: Spring	14.9		+3.6[5]	0.70[7]		-70[4,10]	30[14]	5-8[c]		-40[17]
Summer	9.8		+3.6[5]	0.76[7]		-70	30[14]	11-30[c]		-40[17]
Autumn	8.7		+3.6[5]	2.09[7]		-70	30[14]	15-46[c]		-40[17]
Slurry	49-58[b]	NA	0[4]	0.67[8]	NA	-47[11,12]	1.9[15]	2.9[4]	NA	-40
Dilute slurry	37-43[b]	NA	0[4]	0.93[8]	NA	-47[11,12]	1.9	2.9	NA	-40
Soiled water: Jan-Apr		NA	0	1.62[9]				0.3[9]		
May-Aug	9			0.97[9]	NA	-47[12]	1.9	0.3[9]	NA	-40
Sep-Dec				2.53[9]				5.4[9]		

[a]Numbers in superscript refer to the references below. Where no superscript is included, the emission factor from the most similar source is used

References:[1](Misselbrook et al., 2000); [2] Fischer et al., 2016; [3](Forrestal et al., 2016); [4](Misselbrook et al., 2014); [5](Kim et al., 2012); [6]Harty et al., 2016; [7]Krol et al, 2017; [8](Bourdin et al., 2014); [9](Minogue, 2011); [10] (Clough et al., 2007); [11](Cahalan et al., 2014); [12] (Li et al., 2014); [13](Stevens and Laughlin, 2001); [14](Monaghan and Barraclough, 1993) ;[15](McGeough et al., 2012); [16] (Stark et al., 2007); [17](Dennis et al., 2012)

[b]Ammonia emissions from slurry application are based on a modified version of the ALFAM model (Søgaard et al., 2002), taking into account weather conditions during 48 hrs following application, the application method (splash plate application, trailing shoe application or shallow injection) and slurry DM content (Sup. B2). The ranges in the table refer to the effect of month of application with splash plate application.

[c]Nitrate leaching from urine depends on the month of application based on data by (Stark et al., 2007) (Sup Figure B1).

Table 4. Overview of model inputs for baseline scenario (representing a typical Irish dairy farm) and the different scenarios for increased milk production. For more details see paragraph 2.3.

Scenario	Fertiliser N* (kg ha^{-1})	Fertiliser P* (kg ha^{-1})	Fertiliser K* (kg ha^{-1})	Concentrate (kg cow^{-1})**	Milk yield (L cow^{-1})
Baseline	160	5	13	834	5102
IG: Increased grass production	250	19 (37)***	22 (58)***	834	5102
IC: Increased concentrate	160	5	13	1668	5102
IP: Increased profit	250	20	18	392	5600

*Inorganic fertiliser only.

**Concentrate N content = 2.88%, P content = 0.16%

**Values in parentheses are for build-up phase based on average over first three years i.e. the time required to build up soil P and K levels

Table 5. Model outputs: Herbage DMY, stocking rate, milk production and farm net profit for the baseline scenario and the different scenarios for increased milk production (IG, IC and IP)**.

Scenario	Herbage yield (ton DMY ha^{-1})	Stocking rate (LU ha^{-1})	FPC Milk production*** (ton ha^{-1})		Farm net profit (€ ha^{-1})	Concentrate cost (€ ha^{-1})	Fertiliser & reseeding cost (€ ha^{-1})
Baseline	10.1	1.8	8.3		301	370	204
IG**	13.7 (13.2)	2.5 (2.44)	11.5 (11.0)	38%	637 (498)	510 (492)	336 (408)
IC	10.2	2.1	9.6	16%	99	819	204
IP	13.7	2.4	13.3	61%	1574	261	336

*Scenarios for increased milk production: IG) increased grass production, IC) increased concentrate feeding rates and IP) high profit grass based system. For more details see section 2.3 and Table 3.

**Values in brackets represent the build-up phase and are based on the average over the first three years i.e. the time required to build up soil P and K levels

***Milk production corrected for fat and protein content (FPC)

Table 6. The effect of the mitigation strategies on the absolute change in fertiliser N input, fertiliser and reseeding costs and farm net profit relative to the scenario without mitigation

Mitigation strategy*	Scenario**	Change in fertiliser N input (kg N / ha)	Change in fertiliser & reseeding cost (€ ha^{-1})	Change in farm net profit (€ / ha)
TS	IG	-6.7	4	-1
	IC	-6.2	3	-1

	IP	-8.0	2	3
Inh	IG	-0.9	74	-74
	IC	-0.7	54	-54
	IP	-0.9	72	-72

*Mitigation strategies: TS = trailing shoe, Inh = Inhibitors

**Scenarios for increased milk production: IG) increased grass production, IC) increased concentrate feeding rates and IP) high profit grass based system. For more details see section 2.3 and Table 4.

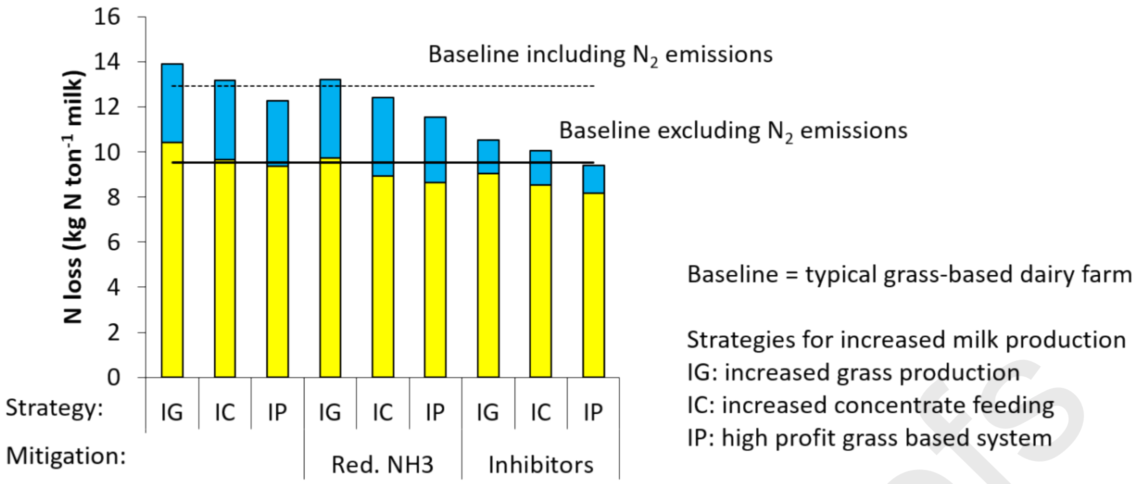
Table 7. The range of possible ammonia, nitrate and nitrous oxide emission factors that would be relevant in Irish bovine production systems and the impact on N loss generated from the model outputs

N type	Source	EF (%)	Reference	Range in N loss (kg N ha^{-1})
Ammonia	Urea	8% - 53%	1,2	34.8 - 73.9
	Slurry Spreading (SP)	45% - 65%	3,4	
	Housing	15% - 38%	4	
	PRP	8% - 16%	1,5	
Nitrate	Fertiliser	2% - 5.5%	6, 7	17.7 - 29.6
	Urine	5% - 10%	8	
	Slurry	2% - 5.5%	6,7	
N_2O	CAN	0.5% - 3.8%	9,10	2.9-9.6
	Urine	0.5% - 2.25%	9,11	
	Urea	0.1% - 0.4%	9,10	
	Slurry	0.3% - 2.25%	6,9	

[1]Fischer et al. 2016 [2]Peterson et al. 1998 [3]Lalor & Lanigan 2010 [4]Dowling 201 [5]Misselbrook et al. 2014 [6]Bourdin 2014 [7]Stark& Richards (in prep) [8]Dennis et al. 2012 [9]Kuikman et al. 2006 [10]Harty et al. 2016 [11]Krol et al. 2016

Highlights

- Intensification scenarios increased absolute N emissions compared to baseline
- Intensification scenarios may result in reduced N emission intensity
- Mitigation methods ameliorate increased emissions associated with intensification
- Mitigation methods may not offset absolute increase in N loss due to intensification


16
14
12
10
8
6
4
2
0
N loss (kg N ton-1 milk)
Baseline including N2 emissions
Baseline excluding N2 emissions
Strategy:
IG
IC
IP
IG
IC
IP
IG
IC
IP
Mitigation:
Red. NH3
Inhibitors
Baseline = typical grass-based dairy farm
Strategies for increased milk production
IG: increased grass production
IC: increased concentrate feeding
IP: high profit grass based system