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# Evaluating management effects on nitrous oxide emissions from grasslands using the process-based DeNitrification–DeComposition (DNDC) model

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ABSTRACT

The development of agricultural mitigation strategies to reduce greenhouse gas (GHG) emissions is urgent in the context of climate change – land use interactions. In this study the DNDC biogeochemical model was used to study nitrous oxide ($N_2O$) emissions from grazed grasslands in southern Ireland. The objectives of this study were: (1) to evaluate the DNDC model using a two year (2008–2009) data set of chamber measured $N_2O$ fluxes at eight grassland sites and (2) to investigate the impact of different management scenarios on $N_2O$ emissions including changes in i) inorganic nitrogen (N) fertilizer application rates ii) slurry application rates; and iii) animal density (livestock unit per hectare LU $ha^{-1}$). The comparison of modeled daily DNDC fluxes (using a combination of measured and default soil parameters) and measured fluxes resulted in an $r$ (coefficient correlation) = 0.48. To improve the model performance, the fluxes for 2008 were used in a calibration exercise during which the soil properties were optimized to obtain the best fit of $N_2O$ fluxes. This resulted in an improved model performance, with an $r = 0.62$. In a validation exercise using 2009 data, we used the model parameters set (e.g. soils) from the calibration exercise and this resulted in a model performance with an $r = 0.57$. The annual $N_2O$ fluxes (measured and modeled) were appreciably higher than those estimated using the IPCC emissions factor of 1.25%. In scenario analysis, the modeled $N_2O$ fluxes only increased/decreased on average ±6% and ±7% following a 50% increase/decrease of inorganic N and slurry N applications respectively. These modeled scenario % changes are much lower than the IPCC emission factor % changes of a 50% increase in $N_2O$ emissions for a 50% increase in nitrogen applied. An absolute change scenario (±50 kg) in inorganic N and slurry N resulted in greater change in $N_2O$ fluxes (±9% inorganic N and ±17% slurry N) as compared to the relative change scenario (above). Furthermore, DNDC $N_2O$ flux estimates were not sensitive to changes in animal density (LU $ha^{-1}$). The latter is a scenario limitation in the current model version. This study suggests that the calibration of soil parameters for Irish conditions is necessary for optimum simulation with DNDC and highlights the potential of management strategies for reducing $N_2O$ emissions from grazed grasslands. It further highlights the difference between DNDC and IPCC estimates that require further research.

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## 1. Introduction

Worldwide, agricultural soils, particularly grazed pastures, are considered a major source of $N_2O$ emissions contributing approximately 46–52% of the global anthropogenic $N_2O$ flux (IPCC, 2007). In Ireland, about 90% of the agricultural area or 58% of the total land area is grassland (Teagasc, 2010). Livestock production in grassland systems positively influence some soil characteristics by: reducing erosion and loss of organic matter associated with tillage; increasing biodiversity of microorganisms; and enhancing nutrient cycling. However, livestock farming can also have negative effects on soil quality including: increased compaction in the surface layers, water pollution (streams and rivers) and biodiversity losses. As Irish grasslands are forecast to become more intensively managed (Food Harvest, 2011), it is now important to assess the $N_2O$ fluxes from grasslands.

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Many countries use the Intergovernmental Panel on Climate Change (IPCC) default methodology for calculating $N_2O$ emissions from agricultural soils which does not account many of the management practices that could potentially reduce $N_2O$ emissions (e.g. fertilizer timing and splitting fertilizer applications). For these reasons the development of process-based models are desirable that are internationally acceptable and that quantify the $N_2O$ emissions under contrasting environments (Giltrap et al., 2010). The application of a process-based model not only allows the simulation of agricultural greenhouse gas emissions at a range of scales up to national or global level (Giltrap et al., 2010; Kim et al., 2010), but also the exploration of potential mitigation strategies along with a range of climate change − land use change scenarios. Several process-based simulation models for the estimation of $N_2O$ emissions at field scale have been used: e.g. DNDC (DeNitrification–DeComposition), (Li, 2000), DayCENT (Abdalla et al., 2010), PASIM (Lawton et al., 2006), MITERRA-EUROPE (Velthof et al., 2007) and Simile (Packham et al., 2006). DNDC was chosen for this study as its required inputs were readily available; the simulation time is short (Hu et al., 2011).

DNDC has been used in a few studies for Irish GHG emissions estimations (Hsieh et al., 2005; Abdalla et al., 2010). These previous studies only focused on one site at a time which may not be sufficient for accurate estimation of $N_2O$ using DNDC. There are very few other studies which focused on the management scenarios (Grant et al., 2004) but these studies only considered the relative change in management practices and did not consider the absolute changes. We believe there can be a significant difference in examining the relative and absolute management scenarios as the initial condition of the soil may have an effect. Furthermore, there is as yet no study of Irish grassland ecosystem which could identify the different ranges of the most influencing parameters (e.g. soil parameters). The direct application of this model to Irish soils presents a challenge; because the soils are distinctive and diverse within short distances and have high carbon (C) contents (Eaton et al., 2008; Xu et al., 2011). In addition, the long duration of the Irish grazed pastoral systems (up to 10 months per year outdoors in some regions of Ireland) and humid temperate climatic conditions (e.g. frequent rainfall throughout the year with continuously moist soils) are different to most other countries. In this study we applied the DNDC model to eight different study sites with different management practices which should enhance the understanding of the mechanistic approaches for the $N_2O$ estimations in Irish conditions.

Specifically, the objectives of this work were: 1) to calibrate and validate the DNDC model using a two year data set of chamber measurements of $N_2O$ fluxes at eight grassland sites in southern Ireland; and 2) to investigate the $N_2O$ mitigation measures including variations of inorganic N inputs; organic N (from slurry) input and grazing livestock density.

## 2. Materials and methods

### 2.1. Study sites

In January 2008, eight grasslands sites were selected in the South of Ireland for nitrous oxide flux measurements using the chamber technique (Skiba et al., 1998). The sites were selected to represent the major soil types of Ireland, a range of dairy management practices and a range of meteorological conditions. All the sites are pastures which are actively managed with regular grazing and fertilizer applications. The dominant grass species at all sites was perennial ryegrass (*Lolium perenne* L.), while site SH1 had a significant proportion (50% in summer) of white clover in the pasture. Across Ireland, the average annual temperature is approximately 10 °C, while the summer mean daily is approximately 19 °C and the winter mean daily is about 2.5 °C. The annual precipitation in the West of Ireland is ~1300 mm while in the East it is ~750 mm. In study sites, the annual rainfall ranged from 950 to 1605 mm. The sites D, CF, PK and SH (SH1 and SH2) experienced greater rainfall than other sites. The monthly average soil temperature ranged from winters 4.6 °C to high of 16.4 °C in summers. There was a little variation across all sites.

### 2.2. Nitrous oxide flux measurement techniques

Nitrous oxide emissions were measured using a manual closed chamber technique (Skiba et al., 1998). The measurements were carried out weekly from March to November and monthly from December to February. The $N_2O$ concentration in each sample was analyzed using a gas chromatograph (GC 3800, Varian, USA) fitted with a packed column (Porapak QS 80–100 MESH, Sigma Aldrich, USA) using an electron capture detector at 300 °C. Hourly $N_2O$ emissions (μg $N_2O$–N $m^{-2}$ $h^{-1}$) were calculated from the slope of the linear increase in $N_2O$ concentration during the chamber lid closure period (Rafique et al., 2011). The daily $N_2O$ flux at each site was estimated using the arithmetic mean of the fluxes from the individual chambers (Barton et al., 2008). The daily $N_2O$ emission values as g $N_2O$–N $ha^{-1}$ $d^{-1}$ were estimated from the concentration measured in the chambers over a measurement period of 1 h converted to those over a period of 24 h. Annual emission rates were estimated by integrating hourly rates with time (Rafique et al., 2011).

### 2.3. DNDC modeling

In this study the DNDC model version 9.3 (2010) was used. The DNDC model contains four main sub-models (Li, 2000); the soil climate sub-model calculates hourly and daily soil temperature and soil moisture fluxes; the crop growth sub-model predicts crop biomass accumulation and partitioning; the decomposition sub-model simulates decomposition, nitrification, ammonia ($NH_3$) volatilization, and $CO_2$ production; and the denitrification sub-model tracks the sequential biochemical reduction from $NO_3$ to $NO_2$−, NO and $N_2$ based on soil redox potential and dissolved organic carbon (DOC).

#### 2.3.1. Model input data

The site specific soil information, texture (% sand, silt, clay), bulk density (BD), porosity, pH, organic C and mineral nitrogen (N) are shown Table 1 which were measured in December 2007 (just prior the start of experiment). As no site specific air temperature data were available, data from the nearest climatological weather station provided by the Irish meteorological services (Met Eireann) within a distance of 30 km of the sites, was used. Only for one site CK, the rainfall data was taken from Met Eireann. The management information is summarized in Tables 2 and 3. The organic N applied was estimated from the number of grazing animals and the N excretion rate for Irish livestock (e.g. for dairy cows 1 LU = 85 kg N) (Rafique et al., 2011).

#### 2.3.2. Model calibration and validation

The model was calibrated using $N_2O$ flux data for year one (2008). Parameters that are sensitive with regard to the simulation of $N_2O$ fluxes (Hu et al., 2011; Smith et al., 2010) include: soil organic carbon (SOC), clay content, pH, bulk density (BD) and the C:N ratio. The model was first run in what we call default mode "DEF" which used measured soil parameters. It was then run in calibration mode "CAL" using values for soil parameters that gave the closest fit of $N_2O$ fluxes to the measured fluxes. The calibration was carried to examine the effect of the ranges of different soil parameters on the $N_2O$ fluxes and so determine the optimized

**Table 1**
General description of DNDC input data (measured on sites and default) for all 8 grassland sites at which $N_2O$ emissions were measured and molded from January 2008 to December 2009.

| DNDC input data/Site Name | BH | CK | D | CF | PK | KW | SH1 | SH2 |
|---|---|---|---|---|---|---|---|---|
| *Climate data* | | | | | | | | |
| Latitude (degree) (°N) | 51.47 | 51.36 | 51.58 | 51.58 | 51.44 | 51.37 | 51.35 | 51.35 |
| Yearly average maximum temperature (°C) | | | | | | | | |
| 1st yr | 18 | 17.44 | 18 | 18 | 19.55 | 19.95 | 18.5 | – |
| 2nd yr | 18.05 | 17.10 | 18.05 | 18.05 | 21.1 | 19.05 | 18.9 | – |
| Yearly average minimum temperature (°C) | | | | | | | | |
| 1st yr | 1.2 | 3.85 | 1.2 | 1.2 | 1.4 | −0.1 | 1.15 | – |
| 2nd yr | −0.6 | 1.32 | −0.6 | −0.6 | −2.45 | −2.75 | −3.25 | – |
| Yearly accumulated precipitation (mm) | | | | | | | | |
| 1st yr | 1019 | 1060 | 1570 | 1304 | 1140 | 1050 | 1405 | – |
| 2nd yr | 1340 | 1115 | 1760 | 1301 | 1091 | 1121 | 1432 | – |
| Atmospheric $NH_3$ conc. ($\mu$g-N $m^{-3}$) | 0.06[a] | 0.06[a] | 0.06[a] | 0.06[a] | 0.06[a] | 0.06[a] | 0.06[a] | 0.06[a] |
| Atmospheric $CO_2$ conc. (ppm) | 380[a] | 380[a] | 380[a] | 380[a] | 380[a] | 380[a] | 380[a] | 380[a] |
| *Soil properties* | | | | | | | | |
| Vegetation type | Moist Pasture | Moist Pasture | Moist Pasture | Moist Pasture | Moist Pasture | Moist Pasture | Moist Pasture | Moist Pasture |
| Soil texture | sandy clay loam | loam | silt loam | loam | loam | loam | loam | silty clay loam |
| Bulk density (g $cm^{-3}$) | 1.04 | 0.99 | 1.02 | 0.88 | 1.05 | 1.08 | 1.22 | 0.84 |
| Porosity | 0.62 | 0.63 | 0.63 | 0.68 | 0.61 | 0.59 | 0.63 | 0.69 |
| Clay fraction | 0.28 | 0.19 | 0.12 | 0.23 | 0.19 | 0.18 | 0.22 | 0.39 |
| Soil pH | 5.8 | 5.9 | 6.7 | 6.4 | 5.4 | 5.7 | 6.3 | 6.5 |
| Initial organic C content at surface soil (kg C $kg^{-1}$) | 0.0342 | 0.0477 | 0.0454 | 0.0567 | 0.0484 | 0.0417 | 0.0473 | 0.0782 |
| C/N ratio | 8.76 | 8.30 | 12.97 | 8.85 | 8.64 | 8.51 | 8.29 | 8.06 |
| $NO_3$–N (mg $g^{-1}$) | 3.7 | 74.05 | 5.3 | 36.1 | 22.9 | 23.2 | 30.6 | 11.45 |
| $NH_4$–N (mg $g^{-1}$) | 50 | 35.4 | 46.1 | 60.35 | 53.05 | 19.65 | 14.55 | 30.6 |
| Field capacity | 0.52 | 0.34 | 0.35 | 0.37 | 0.34 | 0.25 | 0.35 | 0.38 |
| Wilting point | 0.24 | 0.16 | 0.14 | 0.15 | 0.13 | 0.11 | 0.14 | 0.16 |
| Harvest | grazing | grazing | grazing | grazing | grazing | Grazing/cutting | grazing | grazing |
| Depth of water-retention layer (m) | 9.99[a] | 9.99[a] | 9.99[a] | 9.99[a] | 9.99[a] | 9.99[a] | 9.99[a] | 9.99[a] |
| Slope (%) | 2 | 7 | 0 | 2 | 2 | 2 | 2 | 0 |

[a] default values of DNDC.

parameter set values that resulted in the best fit of model fluxes to measured fluxes. The optimized values of these soil parameters are given in Table 4. The model was then validated using the flux data of the second year (2009) using the optimized parameters set from the calibration exercise. The validation exercise (using the optimized values of soil parameters) was used to determine the confidence level in the model.

The model performance was determined using statistical criteria e.g. coefficient of correlation ($r$), bias error (BE), absolute root mean square error (RMSE) and relative root mean square error (rRMSE) which were calculated using the following equations:

$$r = \frac{n\sum x_{meas}x_{mod} - (\sum x_{meas})(\sum x_{mod})}{\sqrt{n(\sum x_{meas}^2) - (\sum x_{meas})^2}\sqrt{n\left(\sum x_{mod}^2\right) - (\sum x_{mod})^2}} \quad (1)$$

$$\mathrm{BE} = \frac{1}{N}\sum(x_{mod} - x_{meas}) \quad (2)$$

$$\mathrm{RMSE} = \sqrt{\frac{1}{N}\sum(x_{mod} - x_{meas})^2} \quad (3)$$

$$\mathrm{rRMSE} = \sqrt{\frac{\sum(x_{mod} - x_{meas})^2}{\sum(x_{meas})^2}} \quad (4)$$

Where $x_{mod}$ is the simulated value, $\bar{x}_{mod}$ is the average of all simulated values, $x_{meas}$ is the value obtained from field data and $\bar{x}_{meas}$ is the average. These statistical parameters were computed for all eight sites for measured versus DNDC modeled daily $N_2O$

**Table 2**
Stocking rate (LU $ha^{-1}$), Animal type, inorganic, organic and total N input at 8 grassland sites at which $N_2O$ emissions were measured and modeled.

| Sites | Live Stock Unit (LU) $ha^{-1}$ | Ruminant types | Inorganic N (kg N $ha^{-1}$ $year^{-1}$) | | Fertilizer type (inorganic) | Organic N (kg N $ha^{-1}$ $year^{-1}$) through slurry | | Cattle grazing N (kg N $ha^{-1}$ $year^{-1}$) through excreta | | Total N (kg N $ha^{-1}$ $year^{-1}$) | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | 1st yr | 2nd yr | | 1st yr | 2nd yr | 1st yr | 2nd yr | 1st yr | 2nd yr |
| BH | 1.0 | Dairy cow/Sheep | 100.4 | 100.4 | CAN | NA | 40 | 53 | 91 | 153.4 | 231.5 |
| CK | 2.50 | Dairy cow | 124.8 | 124.8 | Urea/CAN | 77 | 77 | 175 | 228 | 376.8 | 429.8 |
| D | 3.0 | Dairy cow | 261.0 | 244.0 | Urea/CAN/Pasture Sward | 80 | 80 | 53 | 48 | 394.1 | 371.7 |
| CF | 1.80 | Dairy cow | 169.0 | 169.0 | CAN | 50 | 49 | 158 | 162 | 377.4 | 379.6 |
| PK | 1.80 | Dairy cow | 188.5 | 173.5 | Urea/CAN/Sweet Grass | NA | NA | 142 | 125 | 330.5 | 298.5 |
| KW | 2.46 | Dairy cow | 147.3 | 141.5 | Urea/CAN | NA | NA | 226 | 194 | 373.3 | 335.5 |
| SH1 | 1.5 | Dairy cow | NA | NA | NA | 240 | 240 | 105 | 96 | 345.0 | 336.1 |
| SH2 | 1.5 | Dairy cow | 40.0 | 52.0 | Urea | NA | NA | 81 | 87 | 121.0 | 139.0 |

NA = not applied, 1st yr = 2008, 2nd yr = 2009, Slurry N was estimated based on the standard value of 5.0 kg N present in 1000 L of slurry (Teagasc, 2010).

**Table 3**
Grazing pattern (animal number, grazing days) at 8 grassland sites at which N2O emissions were measured between January 2008 and December 2009.

| Month | Year | BH | | CK | | D | | CF | | PK | | KW | | SH1 | | SH2 | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | GD | AN | GD | AN | GD | AN | GD | AN | GD | AN | GD | AN | GD | AN | GD | AN |
| Jan | 1st | – | – | 21 | 50 | 02 | 51 | 25 | 60 | – | – | – | – | – | – | – | – |
| | 2nd | – | – | 13 | 70 | 21 | 50 | 20 | 60 | – | – | – | – | – | – | – | – |
| Feb | 1st | – | – | 11 | 75 | 25 | 51 | 28 | 60 | – | – | – | – | – | – | 28–29 | 18 |
| | 2nd | – | – | 05/26 | 80/102 | 20 | 50 | 23 | 60 | – | – | – | – | – | – | – | – |
| Mar | 1st | 10–30[a] | 20[a] | 03/31 | 90/93 | 30 | 51 | 03 | 60 | 18 | 159 | 23–26 | 5 | – | – | – | – |
| | 2nd | – | – | 19 | 120 | 25 | 50 | 20 | 60 | 13 | 150 | 24–28 | 5 | 14–15 | 18 | 01–02 | 18 |
| Apr | 1st | – | – | 17 | 95 | – | – | 02 | 60 | 05 | 200 | 23–25 | 5 | 23–27 | 18 | 20 | 18 |
| | 2nd | 15–30 | 25 | 08/30 | 140/160 | 19 | 50 | 24 | 60 | 10 | 190 | 22–25 | 5 | – | – | – | – |
| | | 20–30[a] | 15[a] | | | | | | | | | | | | | | |
| May | 1st | – | – | 12 | 95 | 02 | 51 | 04 | 60 | 05/25 | 200/203 | 18–20 | 5 | – | – | 24–25 | 18 |
| | 2nd | 01–05/15–31 | 25/25 | 21 | 170 | 21 | 50 | 22 | 60 | 20 | 190 | 13–18 | 5 | – | – | 01–02 | 18 |
| | | 17–30[a] | 15[a] | | | | | | | | | | | | | | |
| Jun | 1st | 25–30 | 15 | 03/27 | 95/95 | 05 | 51 | 05 | 60 | 14 | 191 | 02–05/21–23 | 5/5 | 06–10 | 18 | 23 | 18 |
| | 2nd | 01–05/10–20 | 25/25 | 18 | 170 | 17 | 50 | 26 | 60 | 18 | 180 | – | – | – | – | 21–23 | 18 |
| | | 15–25[a] | 15[a] | | | | | | | | | | | | | | |
| Jul | 1st | 01–15 | 15 | 21 | 95 | 15 | 51 | 01 | 60 | 08/31 | 191/173 | 03–07/28–31 | 5/5 | 13–19 | 18 | 20–21 | 18 |
| | 2nd | 01–07 | 25 | 09/30 | 170/170 | 25 | 50 | 24 | 60 | 20 | 190 | 25–27 | 5 | 19–21 | 18 | – | – |
| | | 07–30[a] | 15[a] | | | | | | | | | | | | | | |
| Aug | 1st | – | – | 19 | 95 | 20 | 51 | 20 | 60 | 23 | 163 | 24–25 | 5 | …. | …. | 11 | 18 |
| | 2nd | – | – | 20 | 170 | – | – | 23 | 60 | 20 | 160 | 24–30 | 5 | 19–20 | 19 | 03–04 | 18 |
| Sep | 1st | 25–30 | 15 | 17 | 95 | 10 | 49 | 21 | 60 | 20 | 155 | 27–30 | 5 | 16–18 | 29 | – | – |
| | 2nd | | | 15 | 95 | 15 | 52 | 21 | 60 | 15 | 140 | 25–30 | 5 | – | – | 09–11 | 17 |
| Oct | 1st | 01–15 | 15 | – | – | 15 | 51 | 22 | 60 | 18 | 155 | 01–03 | 5 | 24–26 | 25 | 04–09 | 25 |
| | 2nd | – | – | – | – | 10 | 55 | 18 | 60 | 10 | 140 | – | – | 08–11 | 16 | 09–11 | 19 |
| Nov | 1st | – | – | – | – | 20 | 54 | 21 | 60 | 10 | 148 | 07–13 | 4 | – | – | 08–09 | 27 |
| | 2nd | – | – | – | – | 18 | 54 | 20 | 60 | 15 | 150 | 08–13 | 5 | 29–30 | 17 | 24 | 19 |
| Dec | 1st | – | – | – | – | 25 | 54 | 25 | 50 | – | – | – | – | – | – | – | – |
| | 2nd | – | – | – | – | 25 | 54 | 20 | 50 | – | – | – | – | – | – | – | – |

GD = Grazing date, AN = Animal number on the day of grazing, 1st yr = Year 2008, 2nd yr = Year 2009.
[a] shows the sheep & cattle grazing.

fluxes and were used for inter comparison to compare the model performance between calibration and validation outputs.

### 2.4. Management scenario projection

We examined different management scenarios for their effects on $N_2O$ fluxes: (1) decreased management scenario (DMS) and (2) increased management scenario (IMS) using the parameters from calibration exercise. The alternative management scenarios included changes in the inorganic N fertilizer application rate, slurry application rate (organic N in slurry) and stocking density (LU $ha^{-1}$). The management change scenarios considered were as follows:

#### 2.4.1. Inorganic N fertilizer addition

In our decreased management scenario (DMS), the N application of inorganic N was reduced by 50% while for increased management scenario (IMS) the N application was increased by 50% of the recorded amount at all sites while the slurry application and animal density were kept constant. The SH1 site did not receive inorganic N. To determine the effect of inorganic N on $N_2O$ fluxes from the SH1 site, the same amount of N was applied as was recorded in the SH2 site (both the SH1 and the SH2 are adjacent fields on the same farm). The sites receiving inorganic N >100 kg N $ha^{-1}$ $yr^{-1}$ are characterized as heavily fertilized while other sites as less fertilized. Because the recorded rates of inorganic N application varied widely from 40 to 261 kg N $ha^{-1}$ $yr^{-1}$ (Table 2) and a 50% change therefore resulted in different absolute changes, we also ran an alternative scenario with constant increase/decrease of 50 kg inorganic N $ha^{-1}$ $yr^{-1}$ across all sites.

#### 2.4.2. Slurry application (organic N)

In the present study, four of the eight sites (CK, D, CF and SH1) received slurry application (Table 2). To examine the effect of slurry application, the slurry N was decreased by 50% for the DMS scenario while it was increased by 50% for IMS scenario. The sites which were not receiving slurry application, a typical amount (the average of all other sites ranged from 20 to 120 kg N $ha^{-1}$) were applied to determine the effect of increased slurry application while the inorganic N application and animal density were kept constant. For the same reason as described above, we also ran an alternative scenario with a constant increase/decrease of 50 kg slurry N $ha^{-1}$ $yr^{-1}$ across all sites.

**Table 4**
Calibrated values along with tested range of soil characteristics (bulk density, clay content, and soil organic carbon, pH and C/N ratio) used optimized simulation in calibration and validation.

| Sites | Bulk density (BD) (g $cm^{-3}$) | | Clay contents (%) | | Soil organic carbon (SOC) (kg C $kg^{-1}$) | | pH | | C:N ratio | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Optimized | Range tested | Optimized | Range tested | Optimized | Range tested | Optimized | Range tested | Optimized | Range tested |
| BH | 1.34 | 0.99–1.40 | 0.27 | 0.25–0.30 | 0.034 | 0.030–0.036 | 5.8 | 5.5–6.0 | 8.76 | 8.72–8.80 |
| CK | 0.97 | 0.95–1.00 | 0.20 | 0.17–0.22 | 0.045 | 0.040–0.050 | 5.8 | 5.5–6.0 | 9.21 | 8.20–9.50 |
| D | 1.42 | 1.0–1.45 | 0.14 | 0.11–0.18 | 0.047 | 0.040–0.050 | 6.6 | 6.2–7.0 | 9.60 | 9.50–13.00 |
| CF | 1.02 | 0.85–1.05 | 0.23 | 0.20–0.25 | 0.058 | 0.050–0.060 | 6.5 | 6.0–6.8 | 8.85 | 8.80–8.90 |
| PK | 1.43 | 1.0–1.45 | 0.19 | 0.16–0.23 | 0.046 | 0.040–0.050 | 5.6 | 5.2–6.0 | 8.64 | 8.50–8.80 |
| KW | 1.45 | 1.0–1.50 | 0.17 | 0.15–0.20 | 0.042 | 0.040–0.045 | 5.9 | 5.5–6.0 | 8.61 | 8.50–8.70 |
| SH1 | 1.11 | 1.05–1.25 | 0.24 | 0.20–0.26 | 0.045 | 0.040–0.050 | 6.1 | 6.0–6.5 | 8.31 | 8.25–8.40 |
| SH2 | 0.82 | 0.80–0.90 | 0.37 | 0.30–0.40 | 0.074 | 0.070–0.080 | 6.1 | 6.0–6.5 | 8.04 | 8.00–8.10 |

#### 2.4.3. Animal density (Livestock Unit: LU $ha^{-1}$)

In Ireland the LU $ha^{-1}$ varies from low ($\leq$1.0 LU $ha^{-1}$) to high (3.0 LU $ha^{-1}$). To examine the effect of decreasing and increasing LU $ha^{-1}$, the LU $ha^{-1}$ was decreased by 50% for the DMS scenario while for the IMS scenario the LU $ha^{-1}$ was increased by 50% of the present LU $ha^{-1}$ in all sites while the fertilizer N and slurry applications were kept unchanged. In the alternative analysis (absolute change scenario) we used absolute changes of $\pm$1 LU $ha^{-1}$.

## 3. Results

### 3.1. Climatic and soil characteristics

The annual rainfall and air temperature for eight study sites are shown in Table 1. The annual rainfall ranged from 1019 to 1760 mm. The sites D, CF and SH (SH1 and SH2) experienced greater rainfall than the other sites. The daily air temperature ranged from −2.75 °C to 21.1 °C with little variation between sites. The year 2009 was colder than 2008. Over the summer the soil WFPS ranged from 30.3 to 85.2%, while over the winters the range was 49.1–99.5%.

The measured soil characteristics are shown in Table 1. The soil bulk density (top 10 cm) ranged from 0.83 to 1.03 g $cm^{-3}$, while the porosity ranged from 59 to 69%. The soil organic C content ranged from 0.034 to 0.0782 kg C $kg^{-1}$. The C:N ratio was in the range of 8.06–12.97. The $NO_3$–N ranged between 3.7 and 74.0 mg $g^{-1}$ while the $NH_4$–N ranged from 4.55 to 60.35 mg $g^{-1}$ (Table 1). The optimized values of the soil properties for the calibration runs were in some cases different to the measured values. The ranges of values of the properties examined are given in Table 4. This range was based on the realistic values from the literature.

### 3.2. Model calibration

The measured daily $N_2O$ fluxes along with the DEF and CAL simulations from all sites (from January 2008 to December 2008) are shown in Fig. 1. The timings of N applications are shown as arrows in Fig. 1.

For all eight sites the measured daily $N_2O$ fluxes ranged from −8.54–132.77 g $N_2O$–N $ha^{-1}$ $d^{-1}$. The −ve sign means uptake and the +ve sign means emission. For the DNDC CAL simulation the range was 0.02–234.75 g $N_2O$–N $ha^{-1}$ $d^{-1}$ and for the DNDC DEF simulation the range was 0.03–375.75 g $N_2O$–N $ha^{-1}$ $d^{-1}$. During the summer, elevated measured $N_2O$ emissions were observed after heavy rainfall events, high surface soil temperature (data is not shown for rainfall and soil temperature) and antecedent N fertilizer application events. Corresponding elevated fluxes were also captured by the DNDC CAL and DEF simulations. The mean daily $N_2O$ fluxes varied considerably between most of the sites with maximum emission at site D for the measured data, and DEF and CAL simulations. Across all sites, the measured mean $N_2O$ flux ranged from 9.14 to 28.22 g $N_2O$–N $ha^{-1}$ $d^{-1}$; from 7.14 to 45.02 for the DEF simulation; and for 5.91–19.12 g $N_2O$–N $ha^{-1}$ $d^{-1}$ for CAL simulation (Table 5).

Occasional short term negative peaks (uptake) were also observed at all sites in the measured data but are absent in the modeled data (both CAL and DEF simulations). The highest instantaneous uptake rates of $N_2O$ were measured at SH1 and SH2. The comparison of measured and modeled daily fluxes for the DEF simulation had an $r$ amongst sites that ranged from 0.38 to 0.61 and after calibration the $r$ improved and ranged between sites from 0.50 to 0.80.

For annual fluxes, the BE, RMSE and rRMSE (Table 6) were also computed between the measured and modeled values. The sites with the highest RMSE and rRMSE values are where DNDC performs poorest. The more critical values of model performance, BE shows a general underestimation of the modeled fluxes except at sites D and SH2 where there is a positive BE. Furthermore, the RMSE and rRMSE suggest that the model with some limitations is able to estimate $N_2O$ fluxes for these sites (Table 6). Averaging the model performance for all sites resulted in an $r$ = 0.48, BE = −2.12, RMSE = 30.31, rRMSE = 0.79 for the DEF simulation while for the CAL simulation they were: $r$ = 0.62, BE = −2.61, RMSE = 20.48 and rRMSE = 0.70.

The annual emissions for the different sites estimated using the measured data were in the range of 2.73 ± 2.45–9.32 ± 2.75 kg

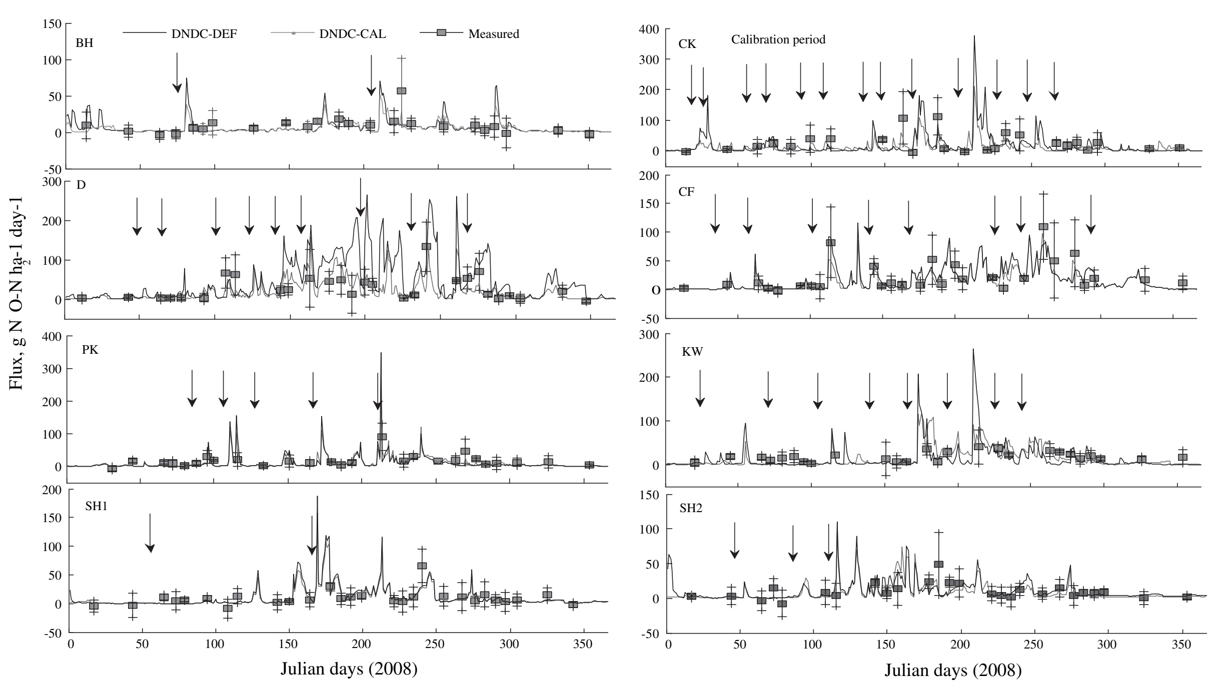


**Fig. 1.** $N_2O$ flux time series from January 2008 to December 2008 for 8 grassland sites in Ireland. The continuous line is for modeled data (DEF and optimized simulation) and squares are for the measured data (with their standard deviations). The arrows show the application of inorganic N (solid arrow) and slurry (dotted arrow).

**Table 5**
Compilation of minimum, maximum and mean $N_2O$ flux values from the 8 grassland sites as derived from model (calibration and validation) and from field measurements.

| Sites | Calibration period | | | | | | | | | Validation period | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Measured $N_2O$ fluxes ($g\ N_2O{-}N\ ha^{-1}\ d^{-1}$) | | | Modeled $N_2O$ ($g\ N_2O{-}N\ ha^{-1}\ d^{-1}$) (DEF simulation) | | | Modeled $N_2O$ fluxes ($g\ N_2O{-}N\ ha^{-1}\ d^{-1}$) (optimized) | | | Measured $N_2O$ fluxes ($g\ N_2O{-}N\ ha^{-1}\ d^{-1}$) | | | Modeled $N_2O$ fluxes ($g\ N_2O{-}N\ ha^{-1}\ d^{-1}$) | | |
| | Min | Max | Mean | Min | Max | Mean | Min | Max | Mean | Min | Max | Mean | Min | Max | Mean |
| BH | −4.03 | 56.59 | 9.14 | 0.09 | 74.64 | 7.14 | 0.11 | 51.27 | 5.91 | −13.53 | 64.51 | 9.70 | 0.09 | 211.77 | 8.58 |
| CK | −3.88 | 112.75 | 26.82 | 0.12 | 375.72 | 18.04 | 0.23 | 210 | 13.31 | −1.72 | 170.64 | 31.18 | 0.31 | 291.96 | 17.70 |
| D | −5.04 | 132.77 | 28.22 | 0.06 | 264.44 | 45.02 | 0.07 | 158.77 | 19.12 | 1.15 | 160.27 | 24.73 | 0.01 | 246.25 | 18.22 |
| CF | −1.87 | 109.30 | 22.76 | 0.04 | 115.64 | 17.11 | 0.04 | 97.66 | 15.16 | −44.78 | 122.11 | 22.64 | 0.03 | 115.69 | 18.83 |
| PK | −6.33 | 90.52 | 16.98 | 0.03 | 347.03 | 10.06 | 0.05 | 234.75 | 10.75 | −4.03 | 112.75 | 18.06 | 0.07 | 252.58 | 13.72 |
| KW | 3.32 | 40.65 | 13.36 | 0.05 | 264.82 | 13.36 | 0.02 | 116.32 | 14.76 | −6.91 | 126 | 15.70 | 0.02 | 156.27 | 8.02 |
| SH1 | −8.54 | 65.23 | 9.47 | 0.03 | 187.83 | 11.06 | 0.02 | 158.18 | 9.71 | −2.70 | 58.03 | 13.36 | 0.01 | 208.67 | 14.46 |
| SH2 | −7.53 | 49.39 | 10.05 | 0.13 | 110.36 | 10.27 | 0.09 | 100.49 | 8.04 | −4.89 | 36.48 | 10.92 | 0.01 | 151.15 | 10.03 |

$N_2O{-}N\ ha^{-1}\ yr^{-1}$. The maximum value was observed at site CK followed by site D. The DEF simulation resulted in an annual flux range of between 2.61 and 16.50 kg $N_2O{-}N\ ha^{-1}\ yr^{-1}$, and for the CAL simulation, an annual flux between 2.91 and 10.30 kg $N_2O{-}N\ ha^{-1}\ yr^{-1}$ (Fig. 2).

The linear equation between measured and modeled annual fluxes for all grassland soils showed a slope which is very near to 1:1 for the CAL simulation ($y = 1.024x + 0.129$) which is considerably better than the slope for the DEF simulation ($y = 1.219x + 0.504$) (Fig. 3). The DNDC DEF simulation for the eight sites (for annual $N_2O$ fluxes in kg $N_2O{-}N\ ha^{-1}$) resulted in BE = 1.62, RMSE = 3.95 and rRMSE = 0.61 (Table 5). Meanwhile, the CAL simulation resulted in BE = 0.25, RMSE = 0.55 and rRMSE = 0.08 (Table 5).

### 3.3. Model validation

For the validation period (2009) the predicted $N_2O$ emissions by the VAL simulation agreed reasonably well with the observed data (Fig. 4). For all eight sites the measured daily $N_2O$ fluxes ranged from −44.78 to 170.64 g $N_2O{-}N\ ha^{-1}\ d^{-1}$ and for the DNDC VAL simulation the range was 0.01–291.94 g $N_2O{-}N\ ha^{-1}\ d^{-1}$. Across all sites, the mean $N_2O$ flux ranged from 9.70 to 31.18 g $N_2O{-}N\ ha^{-1}\ d^{-1}$ for measured data and from 8.02 to 18.83 g $N_2O{-}N\ ha^{-1}\ d^{-1}$ for VAL simulations.

The sites BH, KW and SH2 resulted in the lowest $N_2O$ emissions in both measured and modeled data and the model was capable of simulating the highly dynamic changes in $N_2O$ emissions most of the time at these sites. During the dry period, the VAL simulation and field measurements agreed closely (Fig. 4). A comparison of measured and modeled $N_2O$ emissions at the CK site (Fig. 4) during the dry and wet periods showed that the model poorly captured the $N_2O$ emissions. The higher values of BE, RMSE and rRMSE and lower value of *r* show that the model performs poorly for this site (Table 6).

The model performance parameter analysis (Table 6) shows that the model is a poor fit for site D while for site SH1 it is a moderate fit. Similarly, for the sites CF and PK the model was able to estimate the $N_2O$ emission reasonably well. For the CF site, the model was found to overestimate fluxes on some occasions and underestimated on other occasions. For the PK site the model was able to capture the peaks most of the time. The *r*, BE, RMSE, rRMSE value show that the model is a reasonable fit for these sites. For daily fluxes, averaging the model performance across all eight sites resulted in $r = 0.57$, BE = 0.44, RMSE = 21.05 and rRMSE = 0.72 (Table 6). The values lie between those found for the DEF and CAL simulations. These results are in line with Beheydt et al. (2007) and Abdalla et al. (2010).

The measured annual emissions from 2009 were in the range of $3.05 \pm 2.57$–$11.49 \pm 3.26$ kg $N_2O{-}N\ ha^{-1}\ yr^{-1}$. The maximum value was observed at site CK followed by CF and D. In the DNDC validation (for 2009), the annual flux ranged between 3.22 and 10.25 kg $N_2O{-}N\ ha^{-1}\ yr^{-1}$. The annual $N_2O$ fluxes along with their standard errors are given in Fig. 5. The maximum differences between the modeled and measured fluxes were observed at sites D, KW and SH1. The linear equation for VAL simulations showed that the slope is lower than the 1:1 line ($y = 0.706x + 2.397$) (Fig. 3) for the annual fluxes. The VAL simulation for eight sites (for annual $N_2O$ fluxes kg in $N_2O{-}N\ ha^{-1}$) resulted in BE = 0.61, RMSE = 2.33 and rRMSE = 0.32 (Table 6).

### 3.4. DNDC simulations for different management scenarios

As the DNDC modeled fluxes in the calibration exercise agreed reasonably well with the measured fluxes in terms of general trend and annual fluxes, we considered that DNDC to be suitable to examine different management scenarios. Two different management scenarios i.e. DMS and IMS, to provide the lowest and highest impacts of management, were investigated.

**Table 6**
Statistics for DNDC simulations (DEF, calibration and validation) of $N_2O$ emissions shown in Figs. 1 and 3. For total $N_2O$ emissions the unit is kg $N_2O{-}N\ ha^{-1}\ yr^{-1}$.

| Sites | *R* | | | BE | | | RMSE | | | rRMSE | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | DEF 08 | CAL 08 | VAL 09 | DEF 08 | CAL 08 | VAL 09 | DEF 08 | CAL 08 | VAL 09 | DEF 08 | CAL 08 | VAL 09 |
| BH | 0.57 | 0.80 | 0.73 | −2.73 | −2.23 | −1.26 | 10.76 | 9.27 | 10.57 | 0.70 | 0.60 | 0.59 |
| CK | 0.43 | 0.55 | 0.49 | −12.30 | −10.91 | −15.03 | 34.70 | 35.75 | 35.73 | 0.74 | 0.76 | 0.74 |
| D | 0.38 | 0.50 | 0.48 | 21.45 | 14.24 | 16.78 | 52.72 | 33.48 | 30.14 | 1.31 | 0.83 | 0.95 |
| CF | 0.48 | 0.69 | 0.59 | −6.58 | −6.22 | 1.55 | 23.34 | 22.23 | 29.80 | 0.64 | 0.61 | 0.62 |
| PK | 0.43 | 0.52 | 0.50 | −5.75 | −4.63 | −0.10 | 19.36 | 18.07 | 19.78 | 0.77 | 0.72 | 0.76 |
| KW | 0.42 | 0.59 | 0.57 | −7.46 | −6.04 | −6.71 | 13.06 | 12.40 | 13.61 | 0.68 | 0.64 | 0.65 |
| SH1 | 0.61 | 0.74 | 0.63 | −3.91 | −4.12 | 8.03 | 19.38 | 19.60 | 15.63 | 0.75 | 0.76 | 0.75 |
| SH2 | 0.56 | 0.62 | 0.57 | 0.32 | −1.02 | 0.24 | 13.21 | 13.07 | 17.80 | 0.70 | 0.69 | 0.71 |
| Overall | 0.48 | 0.62 | 0.57 | −2.12 | −2.61 | 0.44 | 30.31 | 20.48 | 21.05 | 0.79 | 0.70 | 0.72 |
| Annual flux | 0.67 | 0.98 | 0.85 | 1.62 | 0.25 | 0.61 | 3.95 | 0.55 | 2.33 | 0.61 | 0.08 | 0.32 |

*R* = correlation coefficient, BE = biased error, RMSE = root mean square error, rRMSE = relative root mean square error.

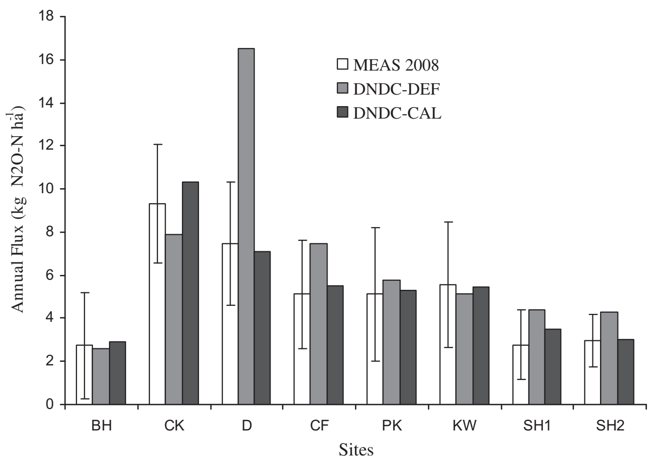


**Fig. 2.** Annual $N_2O$ fluxes of Calibration period year 2008. The measured fluxes are shown with standard error values.

### 3.4.1. $N_2O$ emissions under different inorganic fertilizer N application rates

The reduction of annual $N_2O$ flux was in the range of 2.07%–8.66% when the inorganic N fertilizer input was reduced by 50% (Fig. 6 A, B). The sites PK, D and CK showed the greatest difference. Similarly when the DNDC model was run with IMS (50% increase in N application), an increase of 2.89%–9.92% in annual $N_2O$ flux was found. This is very different to estimates bases on the linear increase/decrease as defined by the IPCC emission factor method, where for example a 50% increase in N results in a 50% increase in $N_2O$ fluxes.

For an alternative scenario with a constant change of ±50 kg inorganic N, the reduction potential of $N_2O$ fluxes was in the range of 2.11%–49.50% while the increase potential was between 4.24% and 20.31% from the baseline (i.e. recorded) application rates. In this scenario the maximum response was observed at the SH2 site. There was no clear trend in the response of $N_2O$ fluxes in both scenarios when ranked according to the recorded amount of N application (Fig. 6A) or the number of application dates (Fig. 6B). However, for the sites with N application <100 kg N $ha^{-1}$ $yr^{-1}$ and ≤3 application dates $yr^{-1}$ (called extensively managed sites: i.e. SH1, BH and SH2) the absolute change scenario indicated a greater response compared to the relative change scenario (Fig. 6C) although the differences was not statistically significant (t-test, $p > 0.05$). In contrast, the relative and absolute scenario resulted in

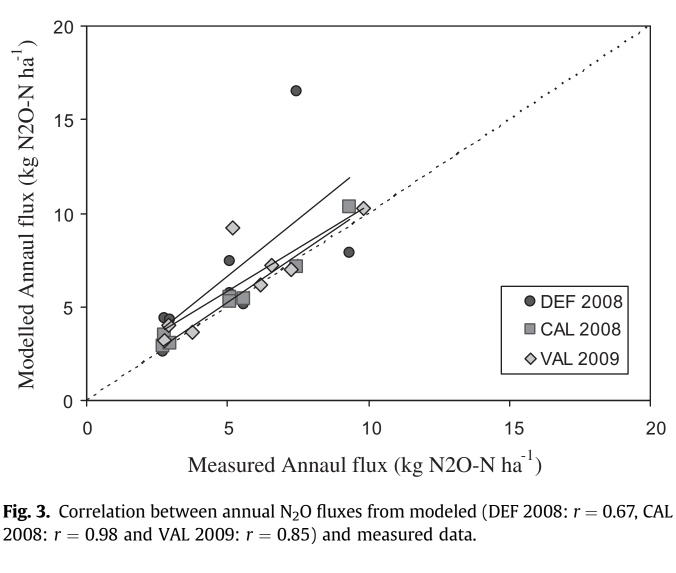


**Fig. 3.** Correlation between annual $N_2O$ fluxes from modeled (DEF 2008: $r = 0.67$, CAL 2008: $r = 0.98$ and VAL 2009: $r = 0.85$) and measured data.

similar responses for sites which initially received more inorganic N application >100 kg N $ha^{-1}$ $yr^{-1}$ in ≥5 application dates (called intensively managed sites) (Fig. 6C). Compared to intensively managed sites, responses in the extensively managed sites were slightly smaller for the relative scenario whereas somewhat greater in the absolute scenario (no statistically significant difference).

### 3.4.2. $N_2O$ emissions under different slurry application rates (organic N)

When the DNDC model was run to investigate the change in $N_2O$ fluxes under different slurry application rates (±50%), it was found to be more sensitive than inorganic N input (Fig. 7). The reduction was found to in the range of 6.34–28.57% with a maximum at SH1 and CF sites following ±50% changes in slurry N. Similarly the increase potential was found to be in the range of 2.7%–14.49% with maximum change at SH1 and PK. In an alternative scenario (with ±50 kg constant change in slurry N) the response was higher in most of the sites compared to the relative scenario. There is a clear trend in reduction potential in absolute change scenario (±50 kg change). Furthermore, among the sites which presently received slurry, the reduction response decreased with increasing present slurry application rates in the absolute scenario, whereas in relative change scenario this trend did not occur and the site which received the most slurry (SH1) showed the highest response (Fig. 7A). On average, the sites which were presently not receiving slurry showed three times higher response to increase in slurry N input when compared to the other sites in absolute change scenario (statistically not significant) (Fig. 7B). Meanwhile no such difference occurred in the relative change scenario.

### 3.4.3. $N_2O$ emissions under different animal density (LU $ha^{-1}$)

The DNDC model output for $N_2O$ emissions was found to be insensitive to both relative and absolute changes in animal density (LU $ha^{-1}$) (data therefore not shown). The annual flux was the same in both DMS (−50% or 1 LU $ha^{-1}$ decrease in animal numbers) and IMS (+50% or 1 LU $ha^{-1}$ increase in animal numbers) management scenarios compared to the observed fluxes.

## 4. General discussion

### 4.1. Model calibration and validation

In our study calibrating DNDC with soil parameters explicitly chosen to obtain the best fit of modeled fluxes (relative to measured fluxes), considerably improved the model performance. After calibrating, the daily modeled fluxes generally matched the data, but on certain days it tended to over or underestimates the fluxes. A possible explanation for these discrepancies may be the result of very high natural spatial variability in fluxes caused by the heterogeneities in the spatial distribution of excretal N (Haynes and Williams, 1993). Lacking high frequency flux measurements as would be available from eddy covariance systems (Scanlon and Kiley, 2003; Mishurov and Kiely, 2010) rather than with the less frequent chamber measurements also adds to the mismatch. Both the DEF and CAL simulations were capable of simulating the peak emissions at most of the sites. However, compared to DEF simulation the CAL simulation predicted the peak event more precisely in terms of magnitude and time of occurrence. This phenomenon is most obvious at site D which may be due to higher initial C:N ratio for this site compared to the other sites. However when the C:N ratio was lowered down the DNDC showed lower $N_2O$ fluxes which are reasonably close to measure annual $N_2O$ flux. This indicates that DNDC may overestimate the $N_2O$ fluxes in case of higher C:N ratio. This requires to be addressed by making changes in the algorithm of DNDC.

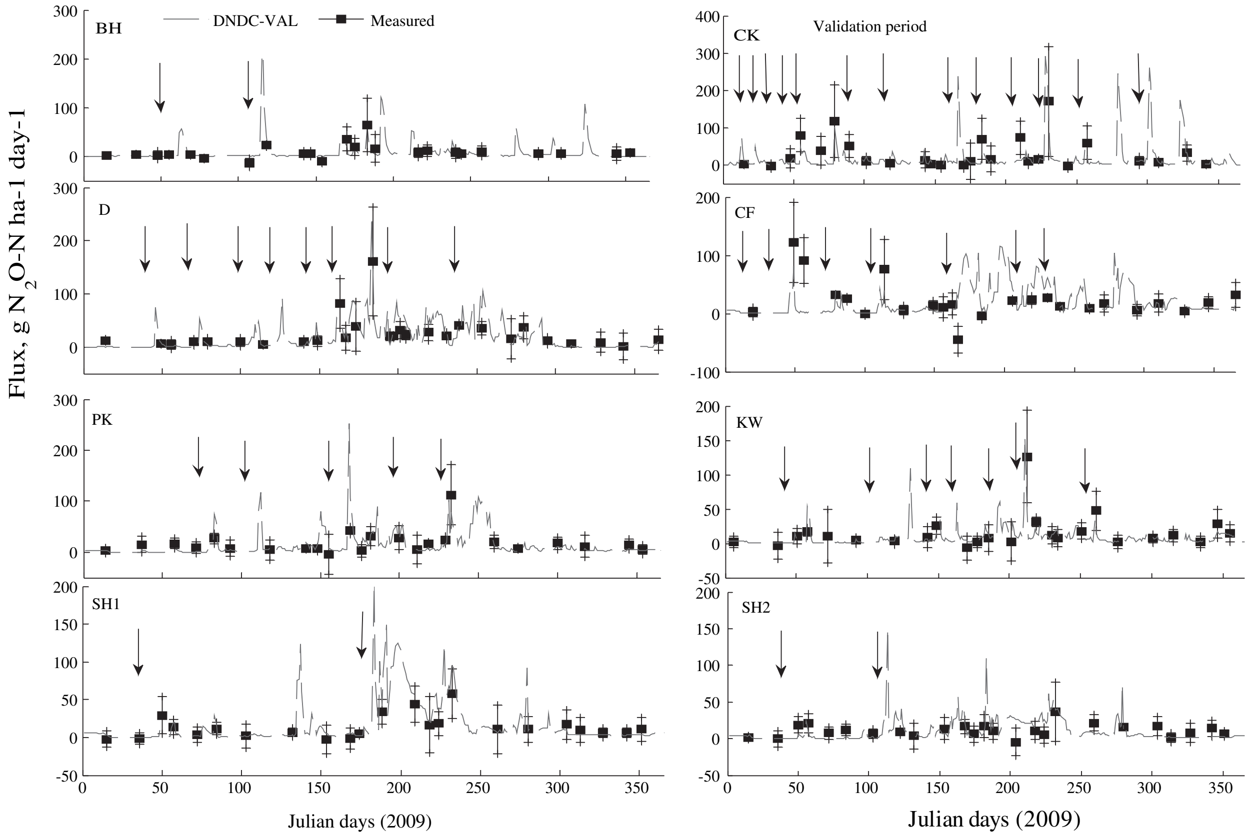


**Fig. 4.** $N_2O$ flux time series from January 2009 to December 2009 (modeled and measured) for 8 grassland sites in Ireland. The continuous line is for modeled data and squares are for the measured data (with their standard deviations). Arrows show the application of inorganic N (solid arrow) and slurry (dotted arrow).

During the validation period (2009), while the modeled fluxes were not as close to the measured fluxes as compared to the calibration period the agreement was still reasonable (Fig. 5). The model underestimated the annual $N_2O$ fluxes on the CK site which may be due to the peaks not being captured in the simulation. However, since the meteorological data for this site were obtained from a station approximately 30 km away, the difference between the simulated and measured $N_2O$ fluxes may also be related to spatial and temporal variability of rainfall between this site and the meteorological station (Kiese et al., 2005). The differences in the magnitude of modeled $N_2O$ emissions between CF and PK may be attributed to differences in SOC, $NO_3$ and $NH_4$ concentrations in soils (Table 1). DNDC did not show

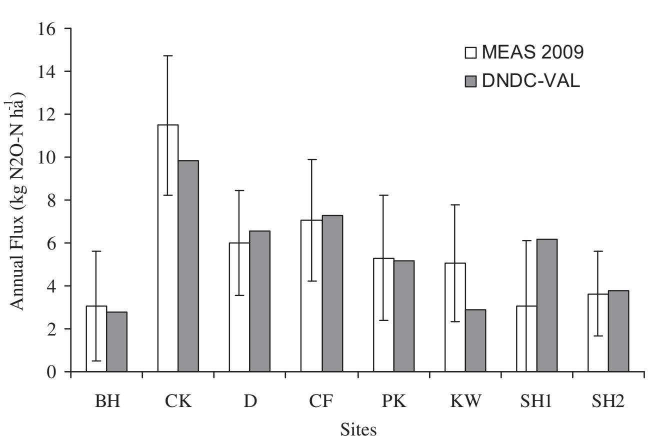


**Fig. 5.** Annual $N_2O$ fluxes of validation period year 2009. The measured fluxes are shown with standard error values.

any $N_2O$ uptake which may lead to the overestimation of modeled $N_2O$ flux and these results are not in line with other studies (Leahy et al., 2004; Rafique et al., 2011) where small $N_2O$ uptakes were recorded. $N_2O$ uptakes mostly occur when the WFPS is more than 80% as $N_2O$ is easily dissolved in water (Beauchamp, 1997).

Contrasting results between measured and DNDC modeled fluxes have also been observed by other authors (Cai et al., 2003; Beheydt et al., 2007). Divergence in annual sums may also occur because of the temporal integration and linear interpolation of the chamber methods (Freibauer and Kaltschmitt, 2003). Large inherent uncertainty of $N_2O$ emission data (Bouwman, 1996) is also always there which occurs because of the complex interactions of different controlling factors (e.g. land cover, hydrology, soil texture etc). The differences in daily, monthly and annual fluxes can also be attributed to land use management (e.g. fertilizer management, grazing regime etc) which is a key parameter in controlling C and N dynamics of landscape ecosystems (Priess et al., 2001).

The application of DNDC to the eight different sites with different soils and grassland management verifies that the model is able to capture the general trend and annual $N_2O$ emission with reasonable accuracy. The optimized values of the soil properties should be seen as effective values. They differ somewhat from the measured values (Table 1). In this study the calibration was restricted to five soil properties. The calibrations of other properties e.g. clay fraction, field capacity and wilting point may present an opportunity to further improve the model performance.

### 4.2. Scenario analysis

The DNDC model showed a limited response with the change in inorganic N input. However, the response from both relative and

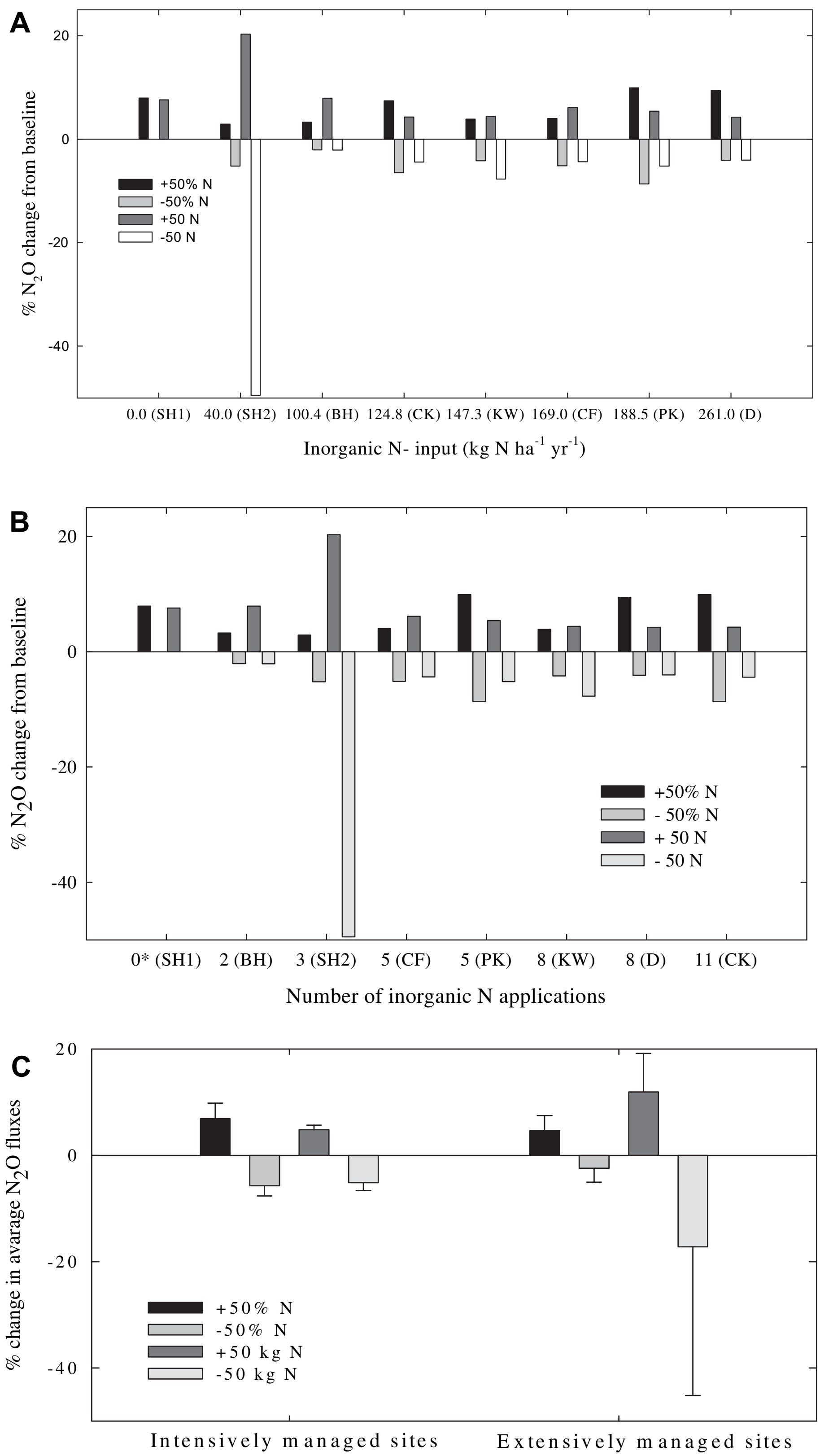


**Fig. 6.** Scenarios analysis under relative and absolute increases and decreases in inorganic N input (A) ranked according to present N application rates (B) ranked according to the number of N application dates. (C) Difference in response from intensively managed (IM: >100 kg N $ha^{-1}$ $yr^{-1}$ & >3 application dates) sites and extensively managed (EM: ≤100 kg N $ha^{-1}$ $yr^{-1}$ & ≤3 application dates) sites. Error bars show standard deviation values.

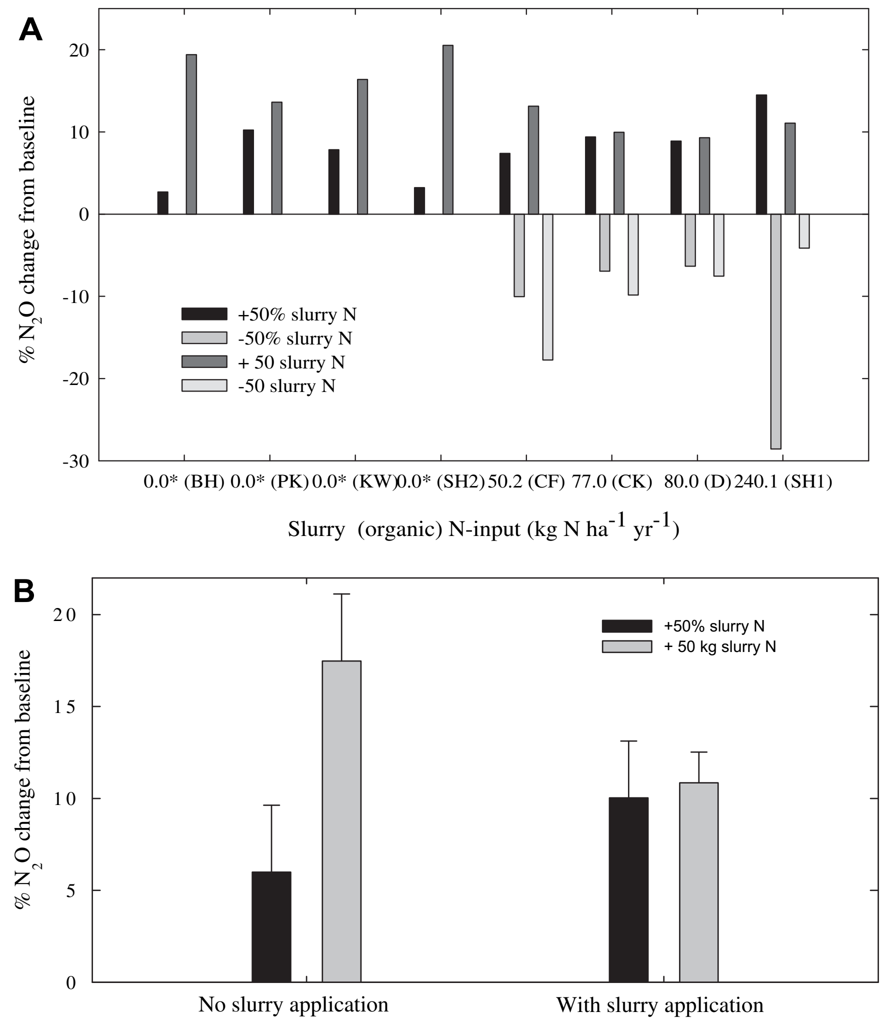


**Fig. 7.** Scenarios analysis under relative and absolute increases and decreases in slurry N input (A) ranked according to present slurry N application rates (C) Difference in response from no slurry application (presently receiving no slurry) sites and from slurry application (presently receiving slurry) sites. Error bars show standard deviation values.

absolute scenarios was different for both inorganic N and slurry N which show that the initial management conditions do matter. According to Beheydt et al. (2007), the slurry application has a high impact on $N_2O$ emissions especially after rainfall. The range of increase or decrease in $N_2O$ fluxes in both these scenarios are in good agreement with the relative change scenarios of Brown et al. (2002) and Saggar et al. (2007).

Previous studies (Saggar et al., 2007) suggest that the number of application dates may affect the response of $N_2O$ emissions. In theory, applying an increased amount of fertilizer N over a smaller number of application dates is likely to result in excess soil inorganic N which may favor a higher fraction to be returned to the atmosphere via $N_2O$ emissions, as opposed to spreading it over a larger number of application dates. In our analysis we found that if N applications dates are $\leq 3$, a considerable change in $N_2O$ fluxes may occur but if N application dates are $>3$ $yr^{-1}$ they do not make difference. However, the sites with $\leq 3$ application dates were also receiving lower amount of N input ($\leq 100$ kg N $ha^{-1}$ $yr^{-1}$).

In this study we observed that the absolute N change response is different to the relative change response. Previous studies (Brown et al., 2002; Grant et al., 2004) tested relative change in N input. However, as the absolute amount of N changes in the relative change scenario depends on and varies with baseline (i.e. measured) application rates and may therefore result in different fluxes for different sites. However the responses from both relative and absolute N change scenarios the $N_2O$ responses do not compare well with the IPCC emission factor estimate. For a 50% increase (decrease) in N (inorganic and slurry N) application, DNDC resulted in an $N_2O$ emissions increase (decrease) by less than 10%. This is appreciably lower than the IPCC guideline estimate for which results in a 50% increase in $N_2O$ for a 50% increase in N (IPCC, 2007).

DNDC flux estimates were unresponsive to an increase or decrease animal density. Previous studies showed that the number of grazing animals directly determines how much dung and urine is deposited on grasslands during grazing. Similarly, Velthof and Oenema (1995) found that the dung and urine patches from grazing animals can initiate small scale hot spots from which $N_2O$ emissions can persist for up to one month. When dung is produced, it is partitioned into the soil litter and humus pools where it will be mineralized and regulated by the decomposition routines which should be part in DNDC.

Our study shows that the DNDC model requires modification to include for variation in animal density and hence the calculation of animal excreta and its consequent $N_2O$ emissions rates. Saggar et al. (2007) found a notable response with change in animal density after applying necessary modifications in the New Zealand NZ-DNDC model. In contrast, scenario analysis for different N inputs from changing grazing animal numbers did not allow conclusive findings based on current DNDC model configurations. For an improved estimation of $N_2O$ emission from Irish grasslands using DNDC, modifications relating to N input from animals and soil moisture regime are required and should be addressed in future work.

## 5. Conclusion

We compared the DNDC modeled $N_2O$ fluxes with measured $N_2O$ data and conducted a series of scenario analysis to evaluate the effect of different management strategies (i.e. increase/decrease of inorganic N, slurry and animal density) on $N_2O$ emissions from eight different grazed grasslands. The application of DNDC (after it has been calibrated for soil parameters) to the eight different sites show that the model is able to capture the general trend and annual $N_2O$ emission with reasonable accuracy. The range of tested soil parameters in this study may serve as indicator for a range of uncertainty which can be used to change the algorithm in DNDC for better improvement for Irish conditions. Increasing or decreasing the total amount of N applied via fertilizer and slurry application resulted in small change (average values: $\pm$6% for relative inorganic change; $\pm$9% for absolute inorganic N change; $\pm$7% for relative slurry N change and $\pm$17% absolute slurry N change) in $N_2O$ emission which is much less than the estimated change of 50% using the IPCC default emission factor. In contrast, changes in animal density (LU $ha^{-1}$) resulted in no change of $N_2O$ emissions which may indicate a possible limitation of the present DNDC model configuration. This highlights weakness in DNDC which requires further research.

We conclude that relative versus absolute change scenario may result in contrasting findings if measured values differ and highlight the need for such consideration in the interpretation of change scenario modeling outputs. We also conclude that due to the poor performance at daily scales, the N input scenario and the lack of response to animal density, there is a need to improve DNDC for use in Irish conditions, especially with regard to long term prediction under different management as the soil parameters change with time. Incorporating such modifications relating to N input from animals and soil moisture regime into DNDC may improve estimates of $N_2O$ emissions within the context of management and climate change impacts for Irish grassland systems.

## Acknowledgment

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