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Whole farm modelling the effect of grass silage harvest date and nitrogen fertiliser rate on nitrous oxide emissions from grass-based suckler to beef farming systems

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ABSTRACT

The intensification of agricultural production systems to produce food for the growing world population is envisaged to result in the increase in nitrous oxide emissions (N_2O). The goal of this study was therefore to assess the effect of different management practices on greenhouse gas (GHG) emissions from contrasting grass-based suckler beef farms with a particular focus on N_2O emissions. The contrasting grass-based suckler beef systems evaluated were intensive (INT) and extensive (EXT) steer and heifer (SH) beef systems and bull and heifer (BH) systems. A whole farm model approach was taken to simulate GHG emissions from these baseline systems using data from a long-term research trial and a hybrid economic-GHG model. Several aspects of the hybrid model were updated. Default values for nitrogen (N) content of fresh and conserved grass were replaced with prediction equations. N excretion and partitioning prediction equations and emission factors (EF) for N_2O from grazing cattle and fertiliser were also updated. The four baseline systems were simulated to harvest first cut silage on May 24. The pasture fertiliser rate for the EXT and INT systems were 77 kg N ha^{-1} and 205 kg N ha^{-1}, respectively. To test the effect of changing management practices, the four baseline systems were simulated at earlier (May 5) and later (June 28) first cut silage harvest dates and 50% higher and lower pasture fertiliser application rates. In total, GHG emissions from four baseline systems and sixteen alternative scenarios were simulated. The carbon footprint of the baseline systems in kg CO_2-equivalent (CO_2e) per kg of carcass weight (kg CO_2e CW^{-1}) ranged from 17.7 for BH EXT to 19.4 for SH INT. This was lower than the latest published EU average of 22.2 kg CO_2e CW^{-1}. Across all scenarios, the increase in fertiliser application rate and earlier first cut silage harvest date increased the kg N_2O kg CW^{-1} of the four production systems. Due to younger slaughter age facilitating higher stocking rates and thus higher productivity per hectare, systems finishing males as bulls at 16 months had lower N_2O and total GHG emissions than production systems finishing males as steers at 24 months. Therefore, BH EXT with increased fertiliser application rate and earlier silage harvest date was the most sustainable suckler to beef production system while SH EXT with reduced fertiliser application rate and later silage harvest date was the least sustainable suckler to beef production system due to longer time to slaughter and consequently lower stocking rate.

1. Introduction

Nitrous oxide (N_2O) is produced naturally as part of the nitrogen (N) cycle during the nitrification and denitrification processes (Firestone and Davidson, 1989; Harty et al., 2016). It is influenced by soil condition and climatic factors with the main drivers being water filled pore space (WFPS), temperature, available organic carbon (C), mineral N content, and soil type (Conen et al., 2000; Signor and Cerri, 2013). Anthropogenic activities such as farming have significantly increased N_2O emissions by raising the amount of N available in soil through increases in both N excreted onto pastures by grazing animals and use of synthetic N fertiliser (Davidson, 2009; Bell et al., 2015). N_2O emissions from animal manure and synthetic fertiliser are rising at an annual rate of 1.1% and 3.9%, respectively (IPCC, 2014). The continuing

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increase in N_2O concentration combined with the long atmospheric lifespan of this gas (114 years) is having detrimental effects on the environment (IPCC, 2013).

The European inventory report states that 73% of total N_2O emissions are derived from the agricultural sector (EEA, 2018). It has been reported by van Groenigen et al. (2005) that managed grasslands emit higher levels of N_2O than arable and unmanaged land. This is of particular concern for grass-based countries like Ireland where 80% (92% including rough grazing) of agricultural land is devoted to permanent grassland (CSO, 2017). This is significantly higher than the European average of 38% (FAOSTAT, 2017). Consequently, as grass-based ruminant livestock production is the dominant agricultural system, grazing and manure related N_2O emissions have a proportionately dominant impact on the total greenhouse gas (GHG) emissions from the agricultural sector (Breen et al., 2010; Krol et al., 2016). Thus, the accurate measurement and the mitigation of N_2O emissions from livestock production systems is vital. However, it must be noted that although grass based ruminant production systems have high environmental impacts, it has been stated that they provide ecosystem services through the enhancement of biodiversity, conservation of cultural landscapes, consumer's perception of meat quality and animal welfare and contributes to the socio-economic viability of rural regions, particularly those with marginal ground (Bragaglio et al., 2018).

Consumption of N is typically considered the principal driver of N losses in ruminant livestock systems. For grass-based systems, the amount of consumed N converted into milk and meat protein rarely exceeds 30% because of the excessive protein concentration of managed pastures (Castillo et al., 2001; Zaman et al., 2013). Of the N excreted, 60–70% is typically in urine (Kebreab et al., 2001; Zaman et al., 2009). Urea is the dominant form of N in urine, constituting 52.1–93.5% of total N (Dijkstra et al., 2013). When excreted, urea is rapidly converted by urease enzymes to ammonia (NH_3), which is highly susceptible to volatilisation, resulting in the emission of NH_3 (Jarvis et al., 1989). The remaining urine-derived NH_3 undergoes nitrification resulting in the production of nitrates (NO_3), an available form of N to pastures. However, the excretion of urine onto pasture results in localized, high loading of N to the soil ($\approx 1000\,kg\,N\,ha^{-1}$) that surpasses the pasture's capacity to fully utilise it (Di and Cameron, 2002). The studies by Welten et al. (2013) and Selbie et al. (2015) reported that 20–30% of N excreted through urine is leached from the soil, similar to the IPCC (2006) default leaching factor of 30%. Irish studies have found lower leaching fraction rates of 7.3–11.2% of N applied (Ryan et al., 2006).

The high loading of N and wetting effect of urine patches onto soils create ideal anaerobic conditions for nitrification and denitrification, thus resulting in the emission of N_2O (Dijkstra et al., 2013). Hence, urine patches are considered “hot spots” for N loss in grass-based systems (Misselbrook et al., 2016). Therefore, diet manipulation is a widely reported strategy to reduce N_2O emission and achieve sustainable production (Dijkstra et al., 2013).

Many indoor studies have shown reductions in N excretion and NH_3 volatilisation without having major effects on animal performance, e.g. by offering concentrate feed and/or silage diets of lower protein content (Castillo et al., 2001; Mulligan et al., 2004; Gregorini et al., 2016; Hynes et al., 2016). However, for grass-based systems, grazed and conserved grass consist of up to 90% of a finishing animal's diet (Crosson et al., 2014). This is a challenge as the composition and quality of grass varies over the grazing season and with different management practices (Keady et al., 2000; Stergiadis et al., 2015), making it difficult to manipulate the diet to reduce N excreted. Therefore as silage quality is largely dictated by management, the efficacy of manipulating silage quality to reduce N excretion in grass-based production systems has been examined. The regrowth period between grazing events is an important determinant of grass quality as it defines what growth stage the grass is grazed. An increase in the regrowth period reduces the amount of N available in the diet and improves the N and energy balance within the rumen, thus reducing NH_3 losses in the rumen and subsequently N excreted through urine (Hoekstra et al., 2007). This may also have a mitigating effect on GHG emissions, in particular N_2O.

In addition to reducing emissions by manipulating animal diets, there is potential to mitigate emissions from fertiliser. Meta-analysis of fertiliser types showed that the application of Ammonium Nitrate (AN) or Calcium Ammonium Nitrate (CAN) had a higher N_2O emission factor than urea fertiliser (Bouwman and Boumans, 2002). Variation in N_2O emissions from fertiliser application occurs between seasons, climatic conditions and different soil types due to different soil characteristics such as bulk density and texture, both of which have large impact on WFPS (Harty et al., 2016). Similarly, for excreta at pasture, it has been shown that N_2O emissions vary greatly depending on type of excreta (dung or urine), soil type and time of application (Krol et al., 2016). Thus, targeting the application of N fertiliser and slurry to plant growth in spring may have some potential to reduce such losses and reduce farm costs. However, previous whole farm grass-based GHG models use tier 1 IPCC (2006) emission factors, which are based on a limited number of international studies, that may not represent country specific conditions (Bell et al., 2015) or cannot capture the mitigation potential of new practices or devices. Country specific N_2O EF have been developed by some researchers, e.g. Bell et al. (2015) and de de Klein et al. (2003), and should be incorporated into whole farm models where possible.

The objectives of this study were twofold: Firstly, to update a whole farm GHG model for beef farms using national research on beef cattle nutrition and agricultural N_2O emissions. Secondly, to use this updated model to determine the effect of the management practices of first cut silage date and fertiliser application rate, on N_2O emissions from grass-based suckler to beef production systems. The effect of these management practices on total GHG emissions, profitability and emission efficiency was also investigated.

2. Materials and methods

Similar to the studies undertaken by Foley et al. (2011) and Clarke et al. (2013) who evaluated GHG emission from a suckler beef production system, a hybrid modelling approach was used. This involved the use of both the Beef Greenhouse Gas Emission Model (BEEFGEM) and the Grange Beef System Model (GBSM) which were amalgamated into one model, allowing the various components of both to interact. A number of modifications were made to the research models to allow the new model to become more suitable for providing recommendations to commercial grass-based beef farmers. Data was inputted into the bio-economical GBSM model to define key system parameters. Such parameters include, herd size, farm structure, area, beef output per year and feed budget (grass, silage and concentrate). After calculating the system's net margin and defining parameters, the output variables from the GBSM model were used as the inputs into the BEEFGEM model which in turn estimated the GHG emissions from the system being simulated.

2.1. Grange beef systems model (GBSM)

The GBSM model (Crosson et al., 2006) is a single year static bio-economic model which runs on a monthly time step where it is assumed that the system is in a steady state throughout the year. For each animal category (cow, calves, yearlings (12–24 months), and 2-year olds), animal numbers, weight and feed intake are stated. Energy requirements and the energy intake from grass and silage for each animal group were calculated using the Irish net energy system created by O'Mara et al. (1997) which is a modification of the French net energy system Jarrige (1989) to suit Irish conditions. Supplementary concentrate feed was provided to cattle to meet target live weight gain when the energy provided by the forage diet was not sufficient. The proportion of forage from pasture and silage in the diet of a group of animals was determined using default values derived from the fraction of time animals spent outdoors or indoors per month. After inputting key data into the GBSM, financial performance data such as livestock revenues, farm net margin, fixed costs (machinery operation, depreciation, car, telephone, electricity, interest) and variable cost of production (veterinary bills, concentrate feed, silage making, fertiliser, slurry spreading, transport) were provided. The model assumes that the farm is family owned and run. It does not farm labour, land costs or EU farm support payments into consideration.

2.2. Beef systems greenhouse gas model (BEEFGEM)

The BEEFGEM model is a single year, static, Microsoft Excel based model, which integrates the farm production variables described in the GBSM model with various emission factors (Foley et al., 2011). BEEFGEM uses a cradle to farm gate life cycle assessment (LCA) modelling approach where the environmental impacts of all processes on farm along with the environmental impacts embodied in farm inputs are accounted for up to the point when the product (beef carcass) leaves the farm (ISO, 2006). It is therefore important to note that as the boundary is the farm gate, the implications of the management practices invstigated at an industry level is not considered.

The impact category investigated was global warming potential (GWP) where the GHG emissions estimated were converted to their 100 year GWP carbon dioxide (CO_2) equivalent (CO_2e) where 1 kg of methane (CH_4) and N_2O are equivalent to 25 kg CO_2e and 298 kg CO_2e, respectively (Forster et al., 2007). Direct GHG emissions from operational farming activities were CO_2, CH_4 and N_2O. Sources of direct emissions included manure storage, manure spreading, excreta deposition onto pastures by grazing animals, application of synthetic fertiliser, enteric fermentation and on farm fossil fuel use (Table 1). Indirect GHG emissions were those associated with the production of inputs brought into the farm such as electricity, synthetic fertiliser. Indirect GHG emissions also include N_2O emitted following nitrogen leaching, run off (NO_3) and ammonia (NH_3) volatilisation (Supplementary material). Inputs which have negligible environmental impacts, e.g. medicine, were excluded from the study. Emissions associated with farm machinery and buildings were omitted as the study presumed that there were no differences in infrastructure or equipment between farms. Although agricultural soils have the ability to sequester large quantities of carbon (Vleeshouwer and Verhagen, 2002), to comply with PAS:2050:2011 (BSI, 2011) it was assumed that soil carbon fluxes were in equilibrium as the majority of grasslands for Irish suckler farms are defined as semi or permanent pastures. The EF and equations used to estimate CH_4 and CO_2 emission from direct or indirect sources were largely similar to Foley et al. (2011) (Supplementary material) but the EFs for direct and indirect N_2O sources were updated using national research and reports (see 2.4 GBSM and BEEFGEM model updates). All slurry produced during the housing season was returned to the silage area at a rate which was proportional to the stocking rate for the grassland management system in place. Silage is to be sold off farm or purchased onto farm if stock is in excess or deficit. Emissions embodied in excess silage sold from farm are removed from the system's total GHG emissions. Consistent with previous cradle to farm gate LCA studies of beef farms, the functional unit of the BEEFGEM model was a kg of beef carcass weight (CW). For comparison reasons the functional units per hectare (ha) and livestock unit (LU) were also reported for the four baseline systems. Additionally conversion factors for kg of beef live weight (LW) were calculated.

2.3. GBSM and BEEFGEM model updates

Following a literature review, the feed, animal N excretion and N_2O emissions components of the GBSM and BEEFGEM models were updated.

2.3.1. Feed

The BEEFGEM model assumed a constant value of 19.4 g kg dry matter^{-1} (DM) for the N content of silage. However, the quality and N content of grass varies greatly depending in particular on grass maturity and fertiliser application (Keady et al., 2000; Dawson et al., 2001). The effect of grass maturity on silage quality (dry matter digestibility (DMD)) was modelled using the relationship reported by Keady et al. (2000):

$$\text{Silage DMD} = (769.8–4.19 \text{ x } HD)/1000 \quad (1)$$

where: DMD in g kg^{-1}, HD = days delay in harvest after 10 May.

All N fertiliser used for growing grass silage was applied on 24 March in the study by Keady et al. (2000). The GBSM model originally used constant weekly silage harvest dates from O'Kiely (2004) to determine silage harvest yield (t DM ha^{-1}) and did not account for the effect of fertiliser application rate on yield. This was modified so that the silage yield for first cut silage for a specific date between 1 May and 3 July was estimated. Similarly, the manner in which second cut yields were estimated was modified. Second cut yield was calculated based on the time interval (5–10 weeks) from first cut harvest date to second cut harvest date. For the current study, the time interval between harvests was set to 6 weeks. To account for the variability which occurs between farms, management practices such as silage harvest date and fertiliser application rate could be specified in the modified model. Additionally, an option was inserted to specify whether the fertiliser applied was CAN (27% N) or Urea (46% N).

The relationship between the N content of grass silage and silage

Table 1
Grassland area for silage production and fertiliser application plan for grazing and silage grassland area for the four production systems: Steer and Heifer Intensive (SH INT), Steer and Heifer Extensive (SH EXT), Bull and Heifer Intensive (BH INT) Bull and Heifer Extensive (BH EXT).

	SH INT	SH EXT	SH INT	SH EXT
Silage area (ha cow unit^{-1})[a]				
First harvest silage (24 May)	0.41	0.27	0.34	0.24
First harvest silage (10 June)	–	0.27	–	0.22
Second harvest silage (6 week regrowth)	0.27	–	0.20	–
Grazing area N fertiliser application				
Spring application (Urea 46% N) (kg N ha^{-1})	57	57	57	57
Summer application (CAN 27.5% N) (kg N ha^{-1})	57	0	57	0
Autumn application(Urea 46% N) (kg N ha^{-1})	34	0	34	0
Post final harvest application (Urea 46% N) (kg N ha^{-1})	57	20	57	20
Silage area N fertiliser application				
First harvest silage (24 May)	110	110	110	110
First harvest silage (10 June)	–	110	–	110
Second harvest silage (6 week regrowth)	80	–	80	–
Total N fertiliser application rate (kg N ha^{-1})	200	100	200	100

[a] Cow unit = Cow plus progeny to slaughter.

yield, and harvest date was developed using data from the studies of Dawson et al. (2001) and Keady et al. (2000):

$$\text{Silage N content} = 26.986–0.956\ \textit{Yield} + 0.0709\ N \quad (2)$$

where: Silage N content = $gN\,kg\,DM^{-1}$, Yield = tDM ha^{-1}, N = $kg\,N\,ha^{-1}$.

The BEEFGEM model previously assumed that the N content of grass was 35.2, 29.6 and 33.6 g kg DM^{-1} for spring, summer and autumn seasons, respectively. In the model, these estimates were not affected by changes in N fertiliser application rates even though there is a strong correlation with N fertiliser rate and the N content of grass (Gately et al., 1972). Grass N content was calculated using equations developed by Stergiadis et al. (2015) where metabolisable energy ME content of grass was used as the main predictor. Grass DMD was the main predictor for grass ME content which decreased from 820 g kg DM^{-1} in February to 730 g kg DM^{-1} in November. Using the results of Gately et al. (1972) the effect of fertiliser application rate on grass N content was included.

$$\text{ME} = 0.532 + 15.56\textit{DMD} \quad (3)$$

where: Metabolisable energy (ME) = MJ kg DM^{-1}, dry matter digestibility (DMD) = g kg DM^{-1}.

$$\text{Grass N} = (16.9 + 0.504\textit{ME}) + ((\textit{Fert N}–50.6) \text{ x } 0.0013) \quad (4)$$

where: Grass N = g kg DM^{-1}, ME = MJ kg DM^{-1}, Fert N = kg N ha^{-1}.

Previously the GBSM model utilised animal performance data such as average daily gain (ADG) and carcass conformation from the studies by Murphy et al. (2008a, 2008b). However for the new hybrid model, the ADG targets, carcass conformation and kill out percentage for each animal group was taken from the study by Drennan and McGee (2009) to give a better representation of the performances of these systems.

2.3.2. N excretion

BEEFGEM predicted N excretion using equations developed by the IPCC (2006) and AFBI (personal communication). These equations were based on animals consuming a wide variety of different diets and therefore did not correctly represent the diets of grass-based suckler beef herds. To improve the model estimates, N excretion prediction equations developed from cattle that predominately consumed grass were used. Manure N excretion was calculated using Eq. (5) derived by Yan et al. (2007). This equation was used to calculate manure N excretion for all animal categories. Urine N excretion by suckler cows and animals > 24 months was predicted using Eq. (6) developed by Zou et al. (2015). Urine N excretion for the remaining animal catagories was calculated using Eq. (7) developed by AFBI (personal communication).

$$\text{Manure N} = (0.567\textit{NI} + 0.651\textit{BW}\text{^}0.75–22.3)/1000 \quad (5)$$

$$\text{Urine } N = 0.691\textit{NI} - 5.13\textit{DMI} + 9 \quad (6)$$

$$\text{Urine } N = 0.567\textit{NI} - 0.0008 \quad (7)$$

where: NI = Nitrogen intake (g N kg DM^{-1}), BW = Body weight (kg BW), DMI = Dry matter intake (kg DM). These equations were developed from cattle fed grass diets and account for the effect of both dietary factors (NI, DMI) as well as animal factors (BW) on N excretion. Faecal N excretion was calculated as the difference between manure N and urine N excretion.

2.3.3. N_2O emissions

Based on the research of Sneath et al. (2006), it was originally assumed that no N_2O emissions were emitted from slurry storage. However, this was revised using the IPCC (2006) estimate that 0.002 kg N_2O-N was emitted for every kg N excreted when manure was stored in a slurry tank below a slatted floor in an animal confinement facility.

The original EF used in the BEEFGEM model for N_2O from manure excreted at pasture was updated using research by Krol et al. (2016) carried out at Irish field sites. The new N_2O emission factors for manure excreted at pasture varies by season (spring, summer and autumn) and soil drainage classes (good, moderate and imperfect). A similar approach was taken by Harty et al. (2016) who developed N_2O emission factors for different forms of N fertiliser (CAN and Urea) applied to soils of different drainage classes. These coefficients were also included (Supplementary material).

The volatilisation of NH_3 and the leaching and runoff of NO_3 results in indirect N_2O emissions (IPCC, 2006). The amount of NH_3 produced during housing, manure storage, animals at pasture and manure application was estimated using a N mass balance approach reported in chapter six of the Irish Informative Inventory Report (EPA, 2017). The amount of N lost as NH_3 from synthetic fertiliser application was also estimated using recommended EFs in the Irish Informative Inventory Report (EPA, 2017) (Supplementary material). IPCC (2006) methodology was used to estimate the amount of N applied that is leached as NO_3 as well as the amount of NH_3 emissions and leached NO_3 that is converted to indirect N_2O (Supplementary material).

Indirect and direct N_2O emissions were reported together using different categories namely, fertiliser application, urine excreted onto pasture, manure management, dung excreted onto pasture and crop residue. Manure management included direct and indirect N_2O emissions from cattle housing, manure storage and application. Nitrous oxide emissions from the input of nitrogen from crop residue in renewed pasture were considered and calculated using IPCC (2006) methodology (Eq. 11.6) assuming 5% of total land area is renewed annually.

2.4. Farm scenarios

2.4.1. Baseline farms

Four grass-based suckler to beef systems described by Drennan and McGee (2009) were simulated with GBSM and BEEFGEM. Drennan and McGee (2009) investigated the performance of spring calving suckler cows and their progeny up to slaughter on two grassland management systems; Intensive (INT) and Extensive (EXT). These systems were allocated the same total grassland area (66 ha), but differed in terms of N fertiliser application, stocking rate and number of silage cuts. The INT system had two grass silage cuts and the EXT system one staggered cut. As described in Drennan and McGee (2009), first cut silage harvest date was simulated to being cut on 24 May for INT systems. For EXT systems, first cut silage harvest was cut on 24 May and 10 June (Table 1). In early spring, N fertiliser was applied at 57 kg N ha^{-1} to the grazing area in both the INT and EXT systems. To facilitate the greater stocking intensity in the INT system, a further 57 kg N ha^{-1} and 34 kg N ha^{-1} was applied after the first grazing rotation and before 20 August, respectively. For both systems, cattle typically started to graze in late spring (April); weather permitting. The fertiliser application rate for first and second cut silage was 110 kg N ha^{-1} and 80 kg N ha^{-1}, respectively. The silage area for INT and EXT systems received fertiliser at 57 kg N ha^{-1} and 20 kg N ha^{-1} following the final silage cut, respectively (Table 1). Urea (46% N) was the primary form of fertiliser used apart from during dry warm weather when it was substituted with CAN (27.5% N).

Animals were housed in October or November and offered silage ad libitum throughout the winter. Concentrate feed was offered to cattle during housing when the energy provided by silage was not sufficient to meet the energy required to achieve target live weight gain. The composition of concentrate fed to both growing cattle and finishing cattle was based on a study by Kyne et al. (2001) and the Teagasc ration reckoner (Teagasc, 2016) (Table 2). Heifers and steers were turned out to grass for a second grazing season, but bulls remained in animal housing where they were finished at 16 months on a silage and concentrate diet. Heifers were finished at grass at 20 months of age. At the end of the second grazing season, steers were housed for the second time and finished on a silage and concentrate diet at 24 months (Table 3). The stocking rate of the systems simulated ranged from

Table 2
Composition of concentrate feed and kg CO_2 equivalent per kg of concentrate ingredient (Ecoinvent).

Ingredient	Composition (%)	kg CO_2e kg DM^{-1}	kg CO_2e
Barley	87.00	0.35	0.31
Soya bean meal	6.80	4.18	0.28
Molasses	4.70	0.11	0.01
Mineral/Vitamin	1.50	–	–
Total kg CO_2e kg DM^{-1} concentrate feed			0.59

Table 3
Description of farming system and feed intake of the four baseline systems: Steer and Heifer Intensive (SH INT), Steer and Heifer Extensive (SH EXT), Bull and Heifer Intensive (BH INT), Bull and Heifer Extensive (BH EXT).

	SH INT	SH EXT	BH INT	BH EXT
Farming system				
Farm area (ha)	66	66	66	66
Stocking rate (LU ha^{-1})	2.54	2.05	2.74	2.20
Replacement rate (%)	15	15	15	15
February calving (% cows calved)	10	10	10	10
March calving (% cows calved)	40	40	40	40
April calving (% cows calved)	50	50	50	50
Suckler cow (head)	92	74	113	90
0–1 year olds (head)	91	73	112	89
1–2 year olds (head)	82	66	63	51
Age at finishing (male) (months)	24	24	16	16
Age at finishing (female) (months)	20	20	20	20
Annual feed intake (t DM $year^{-1}$)				
Suckler cow	4.0	4.0	4.0	4.0
Calves (0–12 months)	1.0	1.0	1.0	1.0
Yearling (12–24 months)	2.9	2.9	2.0	2.0
Grass consumed (t DM ha^{-1})	7.9	6.2	7.8	6.1
Silage consumed (t DM ha^{-1})	2.8	2.3	3.0	2.5
Concentrates consumed (t DM ha^{-1})	0.9	0.7	1.7	1.3

2.05–2.70 LU ha^{-1} where 1 suckler cow = 0.9 LU, calf (0–12 months) = 0.3 LU, yearling (12–24 months) = 0.7 LU and > 24 months = 1 LU (Table 3) (Hennessy and Moran, 2016).

All animals were assumed to be adequately finished to industry specifications where carcass conformation, fat score and kill out proportion for steers, bulls and heifers were obtained from the study by Drennan and McGee (2009). The carcass conformation and fat score for cull cows was assumed to be O grade and 3, respectively (Commission of the European Communities, 1982). As progeny were born from February to April and bulls, heifers and steers were all finished at different ages, (16, 20 and 24 months, respectively), the three different animal types were sold within 3 month windows at different times of the year based on percentage of cows calved per month. Based on this, income from finished cattle and cull cow sales was estimated using 2–3 year monthly average prices during the sale period of each animal category as reported by the Department of Agriculture Food and the Marine (DAFM, 2018). The weighted average price received per kg CW for bulls, steers and heifers over their sale period is shown in Table 4. In contrast, cull cows are sold throughout the year where the price reported in Table 4 is the weighted average priced received per kg CW over 12 months. The most recent fertiliser and concentrate price estimates were also used to assess the economic performance of grass-based suckler to beef systems (Table 4).

2.4.2. Alternative farm scenarios

Each of the four baseline systems, SH INT (a), SH EXT (b), BH INT (c), BH EXT (d) were simulated under 4 scenarios; Scenario 1 = First cut silage harvested 5 May and high fertiliser rate; Scenario 2 = First cut silage harvested 28 June and high fertiliser rate; Scenario 3 = Same as scenario 1 except lower fertiliser rate and Scenario 4 = Same as scenario 2 except lower fertiliser rate. In total, 16 alternative scenarios were simulated using the modified model to assess the effect of silage harvest date and N application rate on the N content of feed and consequently N intake, N excretion and N_2O and GHG emissions. The four systems described by Drennan and McGee (2009) were simulated at two different silage harvest dates and two different N fertiliser application rates for grazing pastures (Table 5). First cut silage harvest date varies greatly in Ireland depending on location as well as management practices. Therefore, to determine the effect of first cut silage harvest date on emissions of the four systems, alternative scenarios were simulated to be harvested at an earlier (5 May) and later (28 June) date than the baseline system (24 May) representing both extremes of first cut silage harvest date (Table 5). For INT scenarios the second cut regrowth period of 6 weeks remained and the second staggered first cut harvest date of 10 June in EXT systems remained. The area allocated for silage for each of the four baseline systems remained the same throughout the alternative scenarios (Table 1).

The high and low N application rates applied to grazing area were 270 kg N ha^{-1} and 115 kg N ha^{-1} for the INT systems and 115 kg N ha^{-1} and 50 kg N ha^{-1} for the EXT systems (Table 1). The stocking rate ranged from 2.23 to 2.89 LU ha^{-1} for intensively fertilised scenarios and from 1.88 to 2.41 LU ha^{-1} for extensively fertilised scenarios. Farm area ranged from 60 to 63 ha for increased fertiliser scenarios and 71-75 ha for reduced fertiliser scenarios (Table 5).

3. Results

3.1. Cattle production system

The four baseline production systems described above were simulated in the modified hybrid model. Estimated animal numbers, weight, CW output ha^{-1} and net margin ha^{-1} are compared in Table 6. Carcass weights for bulls and steers were greater than heifers. Male CW reflected differences in age at finishing where steers slaughtered at 24 months were heavier than bulls slaughtered at 16 months. However, on an area basis the CW output and net margin ha^{-1} of the bull and heifer (BH) system were greater than the steer and heifer (SH) system. This was due to the higher stocking rates of the BH systems. Similarly, the CW output ha^{-1} and net margin ha^{-1} for SH and BH systems increased with intensification because of higher stocking rates, e.g. SH INT vs. SH EXT (Table 6).

Table 4
Beef (steer, bull, cull cow) sale prices and farm input (fertiliser and concentrate) costs used in the modified model.

Item	Prices	Reference
Beef (Steer R-grade) price	€4.00 kg carcass $^{-1}$ (February–April 2016–2018)	DAFM (2018)
Beef (Bull U-grade) price	€4.10 kg carcass $^{-1}$ (June–August 2016–2018)	DAFM (2018)
Beef (Heifer R-grade) price	€4.05 kg carcass $^{-1}$ (October–December 2016–2017)	DAFM (2018)
Beef (Cull cow O-grade) price	€3.25 kg carcass $^{-1}$ (2016–2017)	DAFM (2018)
Fertiliser price (CAN)	€248.50 t^{-1} (2016–2017 year average)	CSO (2018a)
Fertiliser price (Urea)	€350.00 t^{-1} (2016–2017 year average)	CSO (2018a)
Concentrate price	€238.00 t DM^{-1}	CSO (2018b)

Table 5
Farm area (ha), stocking rate (LU ha^{-1}), first cut silage harvest date and grazing area fertiliser application plan for alternative scenarios.

	High fertiliser				Low fertiliser			
	SH INT	BH INT	SH EXT	BH EXT	SH INT	BH INT	SH EXT	BH EXT
Farm area (ha)	63	63	61	61	75	75	72	72
Stocking rate (LU ha^{-1})	2.67	2.89	2.24	2.41	2.24	2.42	1.88	2.02
First silage harvest date (Early)	5 May		5 May		5 May		5 May	
First silage harvest date (Late)	28 Jun		28 Jun		28 Jun		28 Jun	
Grassland fertiliser application plan								
Spring (Urea 46% N) (kg N ha^{-1})	80		65		35		30	
Summer (CAN 27.5% N) (kg N ha^{-1})	80		0		35		0	
Autumn (Urea 46% N) (kg N ha^{-1})	40		0		20		0	
Post final harvest (Urea 46% N) (kg N ha^{-1})	70		50		25		20	

Table 6
Carcass weight, carcass dressing, carcass output ha^{-1} and net margin ha^{-1} for baseline systems: Steer and Heifer Intensive (SH INT), Steer and Heifer Extensive (SH EXT), Bull and Heifer Intensive (BH INT), Bull and Heifer Extensive (BH EXT).

	SH INT	SH EXT	BH INT	BH EXT
Male carcass weight (kg)	370	369	330	334
Female carcass weight (kg)	284	283	284	283
Carcass dressing percentage (%)	54.8	54.8	55.5	55.5
Carcass output (kg ha^{-1})	442	355	517	418
Net margin per ha (€ ha^{-1})	429	342	481	409

3.2. Greenhouse gas (CO_2e) emissions

Total GHG emissions per kg CW for the four baseline systems (SH INT, SH EXT, BH INT, and BH EXT) and the four systems under the four scenarios are reported in Tables 7 and 8, respectively. Within all scenarios, BH INT (18.65–19.82 kg CO_2e kg CW^{-1}) and BH EXT (17.74–18.49 kg CO_2e kg CW^{-1}) systems consistently had lower total emissions per kg CW than their SH system counterparts SH INT (19.17–20.2 kg CO_2e kg CW^{-1}) and SH EXT (18.36–18.93 kg CO_2e kg CW^{-1}). Additionally, SH INT and BH INT scenarios emitted 0.77–1.43 and 0.87–1.38 kg CO_2e kg CW^{-1} more than SH EXT and BH EXT, respectively. Alternative scenarios that had high rates of fertiliser application (1a–1d, 2a–2d) emitted higher GHG emissions per kg CW than scenarios with low rates of fertiliser application (3a–3d, 4a–4d). Alternative scenarios with a first cut of silage on 5 May (1a–1d, 3a–3d) emitted higher GHG emissions than scenarios with a first cut of silage about 8 weeks later on 28 June (2a–2d, 4a–4d) (Table 7). The contribution of each emission source to total GHG emissions for the four baseline systems is presented in Table 7.

For all of the scenarios simulated, CH_4 was the main GHG emission from grass-based suckler to beef systems. Methane accounted for 83.2–86.2% of direct GHG emissions and 60.5–68.5% of total GHG emissions (Table 8). The majority of CH_4 emissions were emitted from enteric fermentation (97.1%) with minor emissions being emitted from manure excreted at pasture (1.4%), silage effluent (1.2%) and manure management (0.3%).

Carbon dioxide was the least important direct GHG emission source and the second most important total GHG emission source from grass-based suckler to beef systems. Direct CO_2 emissions came from the burning of fossil fuels through slurry handling and feeding silage during housing. Only 3.0–4.0% of direct GHG emissions were due to CO_2. Regarding total CO_2 emission, only 8.5–16.2% was emitted directly through the burning of fossil fuels on farm. A substantial amount of CO_2 emissions were emitted indirectly, increasing CO_2 emissions contribution to 18.0–26.0% of total GHG emissions (Table 8). Indirect CO_2 emissions were largely from CO_2 embodied in fertiliser. The contribution of fertiliser to indirect GHG emissions was larger (41.4–66.1%) for INT scenarios than EXT (30.8–55.3%). The CO_2e embodied in concentrate feed varied depending on the suckler to beef system. Due to the intensive finishing of BH systems, CO_2e embodied in concentrates emitted 34.2–44.5% and 40.8–52.5% of total CO_2 emissions from BH INT and BH EXT systems respectively. In contrast, CO_2e embodied in concentrates emitted 22.6–33.5% and 28.5–42.8% of total CO_2e emissions from SH INT and SH EXT systems, respectively. As expected fertiliser contribution was greatest in scenario 1 (1a–1d) and scenario 2 (2a–2d) when fertiliser application rate was increased while concentrate feed contribution was greatest in scenario 3 (3a–3d) and scenario 4 (4a–4d) when fertiliser usage was reduced. Silage harvest date also had an effect on concentrate feed contribution to total CO_2 emissions, increasing with the extension of first cut silage date due to increase in concentrate requirement. Other indirect CO_2e emission sources were CO_2e embodied in diesel (1.4%) and electricity (1.0%).

Table 7
Contribution of emission sources (expressed as percentage) to total greenhouse gas emissions and total emissions per functional unit from the four baseline systems: Steer and Heifer Intensive (SH INT), Steer and Heifer Extensive (SH EXT), Bull and Heifer Intensive (BH INT), Bull and Heifer Extensive (BH EXT).

Direct emissions	SH INT	SH EXT	BH INT	SH EXT
	%	%	%	%
CH_4 from enteric fermentation	62.18	66.01	60.95	64.44
CH_4 from manure management	0.14	0.20	0.15	0.22
CH_4 from silage effluent	0.79	0.85	0.74	0.81
CH_4 from deposition at pasture	0.90	0.96	0.82	0.87
N_2O from urine at pasture	4.77	5.01	3.94	4.12
N_2O from dung at pasture	1.33	1.38	1.13	1.20
N_2O from fertiliser use	1.95	0.71	1.78	0.65
N_2O from manure management	1.44	1.61	1.65	1.78
N_2O from crop residue	0.23	0.26	0.20	0.23
CO_2 from fossil fuel use (diesel)	2.38	2.80	2.36	2.77
Total direct	76.10	79.80	73.73	77.08
Indirect emissions				
CO_2 equivalents from electricity	0.19	0.25	0.17	0.22
CO_2 embodied in fertiliser	12.98	8.81	11.87	8.06
CO_2 embodied in concentrate	5.94	6.36	9.51	9.95
CO_2 embodied in diesel	0.27	0.31	0.27	0.31
CO_2e from NH_3 grazing	0.22	0.23	0.22	0.23
CO_2 equivalent from NH_3 manure management	1.35	1.50	1.51	1.63
CO_2 equivalent from NH_3 N fertiliser application	1.54	1.50	1.41	1.38
CO_2 equivalent from NO_3 leaching	1.42	1.23	1.31	1.15
Total indirect	23.90	20.20	26.27	22.92
Emissions per functional unit				
Total GHG emissions (tCO_2e ha^{-1})	8.6	6.5	9.7	7.4
Total GHG emissions (tCO_2e LU^{-1})	3.2	3.1	3.4	3.2
Total GHG emissions (kg CO_2e kg $carcass^{-1}$)	19.4	18.4	18.8	17.7
Emission efficiency (€ net margin tCO_2e^{-1})	51.91	54.57	50.89	56.85

Table 8
Direct and total GHG emissions (kg CO_2e kg CW^{-1}), contribution of methane (CH_4), nitrous oxide (N_2O) and carbon dioxide (CO_2) to total GHG emissions and feed intake (tonne DM ha^{-1}) for the four systems; Steer and Heifer Intensive (SH INT) (a), Steer and Heifer Extensive (SH EXT) (b), Bull and Heifer Intensive (BH INT) (c), Bull and Heifer Extensive (BH EXT) (d) under four scenarios; Scenario 1 = First cut silage harvested 5 May and high fertiliser rate; Scenario 2 = First cut silage harvested 28 June and high fertiliser rate; Scenario 3 = Same as scenario 1 except lower fertiliser rate; Scenario 4 = Same as scenario 2 except lower fertiliser rate.

	Direct GHG emissions	Total GHG emissions	CH_4 (%)	N_2O (%)	CO_2 (%)	Grass consumed	Silage consumed	Concentrate consumed
1a	14.9	20.2	61.4	15.4	23.2	8.5	3.0	0.9
1b	14.7	18.9	66.0	14.4	19.6	6.9	2.6	0.7
1c	14.2	19.8	60.5	14.3	25.2	8.3	3.2	1.7
1d	13.9	18.5	64.7	13.3	22.0	6.9	2.8	1.4
2a	15.0	20.2	62.0	14.1	23.9	9.1	3.2	1.1
2b	14.7	18.8	66.9	12.8	20.3	7.4	2.5	1.0
2c	14.1	19.5	60.9	13.1	26.0	8.9	3.5	1.9
2d	13.7	18.1	65.3	11.7	22.9	7.4	2.7	1.7
3a	14.8	19.2	64.7	14.8	20.5	6.8	2.5	0.7
3b	14.7	18.4	67.9	14.1	18.0	5.6	2.2	0.6
3c	14.1	18.9	63.4	13.7	22.8	6.7	2.7	1.4
3d	13.9	18.0	66.3	13.0	20.7	5.5	2.4	1.1
4a	14.9	19.2	65.2	13.5	21.3	7.1	2.7	1.0
4b	14.7	18.4	68.5	12.5	19.1	5.9	2.1	0.9
4c	14.0	18.6	63.7	12.5	23.8	7.1	2.9	1.6
4d	13.7	17.8	66.7	11.4	21.9	5.9	2.3	1.4

3.3. Nitrous oxide (N_2O) emissions

For all scenarios, N_2O was the second most important direct GHG emission (9.8–13.8%) and the third most important overall GHG emission (11.4–15.4%). Fig. 1 shows the main source of N_2O across all scenarios was the deposition of urine onto pasture ranging from an average of 36.0% and 39.4% of total N_2O emissions from BH INT and SH INT systems to 40.6% and 44.4% of total N_2O emissions from BH EXT and SH EXT systems. The second highest source of total N_2O emissions fluctuated between fertiliser application and manure management. For the baseline SH INT and BH INT systems and INT systems across all four scenarios, fertiliser application was the second highest source of total N_2O emissions (25.4–32.3%). In contrast, manure management was the second highest source of total N_2O emissions for the baseline SH EXT and BH EXT systems as well as EXT systems across all four scenarios (18.5–31.4%). The excretion of dung onto pastures and crop residue on renewed grassland were minor sources of N_2O emissions emitting 7.5–11.1% and 1.4–2.1% of total N_2O emissions, respectively.

The effect of farming system intensity on N_2O emissions was consistent across the various management practices. For example within scenarios, emissions from SH INT and BH INT were higher than their respective EXT systems e.g. 1a vs. 1b (Fig. 2). Within scenarios, emissions from BH INT were lower than from SH INT and emissions from BH

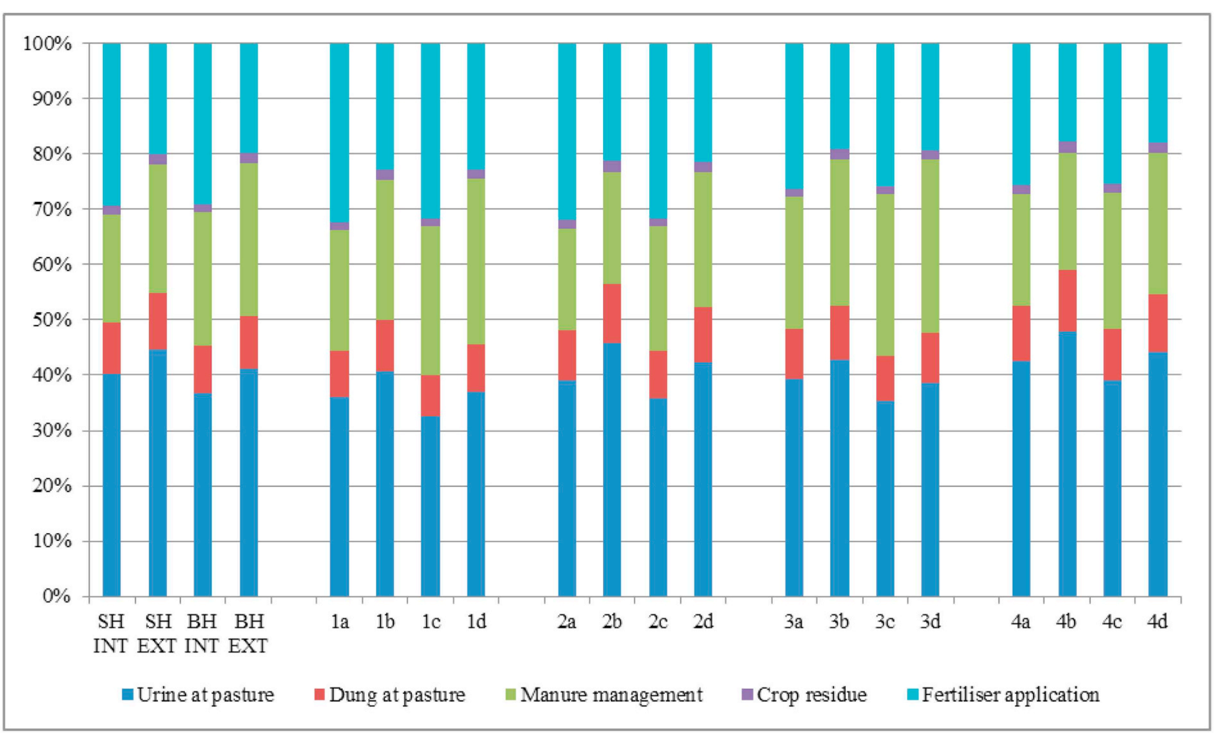

Fig. 1. N_2O emission sources (expressed as percentage) from the four systems; Steer and Heifer Intensive (SH INT) (a), Steer and Heifer Extensive (SH EXT) (b), Bull and Heifer Intensive (BH INT) (c), Bull and Heifer Extensive (BH EXT) (d) under four scenarios; Scenario 1 = First cut silage harvested 5 May and high fertiliser rate; Scenario 2 = First cut silage harvested 28 June and high fertiliser rate; Scenario 3 = Same as scenario 1 except lower fertiliser rate; Scenario 4 = Same as scenario 2 except lower fertiliser rate.

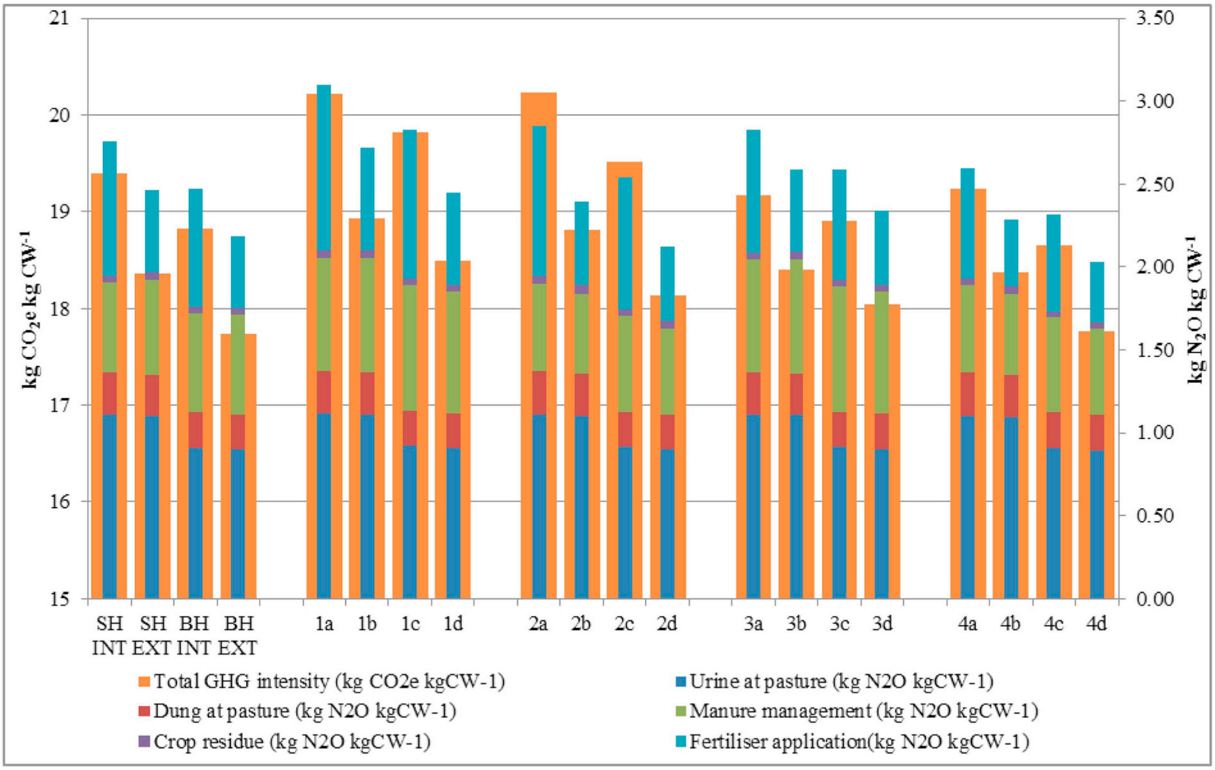

Fig. 2. Total GHG intensity (kg CO_2e kg CW^{-1}) and quantity of N_2O (kg CO_2e kg CW^{-1}) emitted from each N_2O source from the four baseline systems; Steer and Heifer Intensive (SH INT) (a), Steer and Heifer Extensive (SH EXT) (b), Bull and Heifer Intensive (BH INT) (c), Bull and Heifer Extensive (BH EXT) (d) under four scenarios; Scenario 1 = First cut silage harvested 5 May and high fertiliser rate; Scenario 2 = First cut silage harvested 28 June and high fertiliser rate; Scenario 3 = Same as scenario 1 except lower fertiliser rate; Scenario 4 = Same as scenario 2 except lower fertiliser rate.

EXT were lower than from SH EXT e.g. 3c vs. 3a (Fig. 2). This is due to the greater carcass productivity of BH systems in comparison to SH systems (Table 6). Despite producing more beef CW per farm system (table 6), INT systems emitted higher levels of N_2O kg CW^{-1} as a result of intensification. Intensification increased urine excretion, the volume of manure requiring management and the requirement for N fertiliser. This increased loss of direct N_2O and indirect N_2O through NH_3 volatilisation and NO_3 leaching at a rate greater than the rate of increase in productivity from intensification.

3.4. Effect of farm management practices on N_2O emissions

Fertiliser application rate had the largest effect on the amount of N_2O emitted from fertiliser application. INT and EXT SH and BH systems with increased fertiliser application rate (1a–1d, 2a–2d) fertiliser emitted on average 0.22–0.26 and 0.09–0.13 kg CO_2e kg CW^{-1} as N_2O more than INT and EXT scenarios that had a low level of fertiliser (3a–3d, 4a–4d), respectively. Fertiliser application rate had small impact on N_2O emissions from urine excreted onto pasture as a result of small increase in N content of grass with increase in fertiliser application rate and crop residue from the reduction in land area requirement due to positive relationship between grass growth per hectare and fertiliser application rate. No effect was found on N_2O emissions from manure management or dung excreted onto pasture.

First cut silage harvest date had the largest impact on N_2O emissions from manure management. A negative relationship between silage harvest date and N_2O emissions was identified where the amount of N_2O emitted from manure management reduced as silage harvest date was extended, e.g. 1a–1d vs. 2a–2d (Fig. 1). The largest effect was seen for EXT systems where N_2O emissions reduced on average 0.21 kg CO_2e kg CW^{-1}. A smaller effect was seen for INT systems (-0.17 kg CO_2e kg CW^{-1}). The small increase in N_2O emissions per kg CW from early harvesting of first cut silage was in part due to the higher N intake per kg DM during housing and thus N excreted at housing. This led to greater direct and indirect N_2O losses from the emissions during housing, manure storage and manure application. As silage harvest date was extended, the DMD and N content of silage reduced. This in turn increased concentrate DM intake to ensure ADG was maintained thus increased N content of diet and reducing the effect of silage harvest date on N excretion during housing. Additionally, as harvest date extends, the amount of silage produced per kg N applied increased. In turn, excess silage increased while inputs remain unchanged, resulting in the increase in emissions embodied in excess silage. As a result, manure management emissions associated with beef production reduced. Silage harvest date has the same effect on N_2O emissions from fertiliser application with early harvest date (5 May) emitting 0.078–0.091 and 0.085–0.113 kg N_2O kg CW^{-1} more than late harvest date (28 June) from INT an EXT systems, respectively e.g. 1a vs. 2a, 1b vs.2b. Neither silage harvest date nor fertiliser application rate had an effect on N_2O emissions from dung N.

3.5. Economic performance and N_2O emissions

The farm net margin (€ net margin ha^{-1}) and emission efficiency (€ net margin tCO_2e^{-1}) for each of the 4 baseline systems and the 16 alternative scenarios simulated are shown in Fig. 3. For all scenarios, INT systems were found to be more profitable than EXT systems. BH INT and BH EXT were found to be more profitable than SH INT and SH EXT across all scenarios, respectively. This was largely explained by the higher stocking rates and carcass output per ha of the BH systems, finishing males at 16 months rather than steers at 24 months. Although finishing of males as bulls led to the BH systems having a higher concentrate feed requirement ha^{-1} than SH systems (Table 8), the higher feed costs of BH systems were outweighed by the higher sales of CW output ha^{-1}, which lead to a greater farm net margin ha^{-1}. Overall, the BH INT scenario with high fertiliser application rate was the most

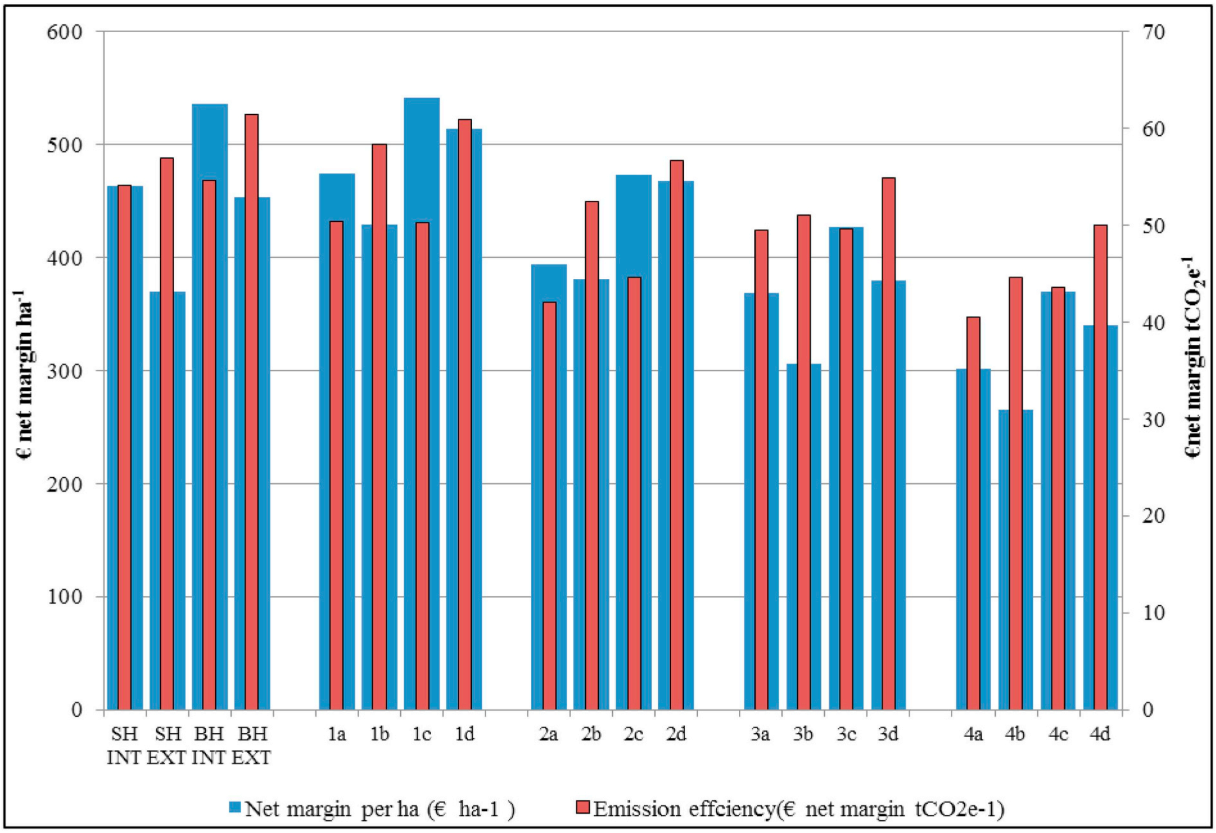

Fig. 3. Economic performance (€ net margin ha^{-1}) and emission efficiency (€ net margin tCO_2e^{-1}) of the four baseline systems; Steer and Heifer Intensive (SH INT) (a), Steer and Heifer Extensive (SH EXT) (b), Bull and Heifer Intensive (BH INT) (c), Bull and Heifer Extensive (BH EXT) (d) under four scenarios; Scenario 1 = First cut silage harvested 5 May and high fertiliser rate; Scenario 2 = First cut silage harvested 28 June and high fertiliser rate; Scenario 3 = Same as scenario 1 except lower fertiliser rate; Scenario 4 = Same as scenario 2 except lower fertiliser rate.

productive and most profitable system. The SH EXT scenario with low fertiliser application was the least productive and profitable.

For all systems, there was a positive relationship between net margin ha^{-1} and fertiliser application rate. Increasing fertiliser application rate increased grass yield (tDM ha^{-1}), stocking rates and output ha^{-1}, but had a minor negative effect on feed costs because of higher fertiliser input costs. Thus, profitability improved. In contrast, silage harvest date had a significant negative relationship with profit ha^{-1}. As grass matured with the delay of the silage harvest date, the DMD and energy value of grass silage decreased as yield increased. This increased the requirement for concentrate supplementation and feed costs (Table 8), because livestock ADG and thus total biological energy demand did not change. Additionally, harvesting silage long after grass maturity increased contractor, feed and energy costs, because, low quality grass silage was produced.

The emissions intensity and economic performance for each scenario was analysed simultaneously by expressing farm profit in terms of CO_2e emissions (€ net margin tCO_2e^{-1}). This is referred to as the emission efficiency. Within all scenarios, SH EXT and BH EXT consistently had higher emission efficiency that SH INT and BH INT, respectively. For example the emission efficiency of SH EXT ranged from €44.6–58.3 net margin tCO_2e^{-1} while SH INT ranged from €40.5–54.1 net margin tCO_2e^{-1} (Fig. 3). As first cut silage harvest date was extended, the emission efficiency of all four systems reduced e.g. 1a–1d vs. 2a–2d (Fig. 3). This was caused by the reduction in net margin with the extension of first cut silage harvest date. Similarly, emission efficiency was lower for all four systems in scenarios with reduced fertiliser application rate e.g. 1a–1d vs. 3a–3d.

Though BH INT systems resulted in a higher net margin per ha than BH EXT systems across all scenarios, when analysing GHG intensity per kg CW, net margin per ha, emission efficiency and system productivity simultaneously, it concluded that the most sustainable scenario was BH EXT systems with early silage harvest date and increased level of fertiliser application (1d).

4. Discussion

It is envisaged that GHG emissions are to increase as a consequence of the intensification of Agric. Syst. to provide food for the growing population and change dietary trends. Though it has been reported that CH_4, the dominant GHG from ruminant production systems can be reduced with the increase in the level of concentrates in the diet, the effectiveness of mitigation strategies for grass based production systems (improved grassland management, use of legumes) have been mixed. This is due to variation in grass composition and quality over the grazing season and with different management practices, thus making it difficult to manipulate the diet. In contrast, N_2O emissions from N fertiliser application and manure management are more controlled by management practices and therefore have greater mitigation potential.

The results of this study suggest that there is a considerable difference in N_2O emissions between contrasting suckler to beef systems and for different management practices. Although the studies by Clarke et al. (2013), Foley et al. (2011) and Casey and Holden (2006) did not report individual GHG emissions, they did provide comprehensive reports of emissions from different sources (e.g. manure management, fertiliser application, excreta at pasture) and stated their contribution to total GHG emissions. Based on key N_2O emissions sources, it was possible to estimate the amount of N_2O emitted per kg CW. The N_2O emissions estimated in this study were lower than reported by Foley et al. (2011) and Clarke et al. (2013) where the higher N_2O emissions from these two studies compared to the current study are largely explained by the differences in N_2O EF discussed earlier.

What is similar between our results and the findings of Foley et al. (2011), Clarke et al. (2013) and Casey and Holden (2006) is that SH suckler to beef systems emitted higher amounts of N_2O kg CW^{-1} than BH suckler to beef systems. The lower GHG intensity per kg CW for BH systems were related to bulls having a higher potential for carcass ADG relative to steers and BH systems operating at higher stocking rates, resulting in higher carcass output per ha in comparison to SH system (Drennan and McGee, 2009). This is due to bulls not being turned out for a second grazing season thus allowing more ground for the suckling phase of the production systems. In return, the younger slaughter age reduced enteric fermentation (CH_4) per finished animal while increased productivity had a mitigating effect on N_2O kg CW^{-1}, particularly on emissions from the deposition of manure on pasture and less from fertiliser application. Due to more manure being produced at housing, N_2O kg CW^{-1} was slighter greater in BH systems in comparison to SH systems.

Stocking rate is one of the most important factors affecting production, profitability and sustainability of a farm system (Foley et al., 2011; Horan, 2014). This is consistent with the findings of this study where the increasing stocking rate through fertiliser application was shown to increase carcass output ha^{-1} and net margin ha^{-1} but consequently resulted in the increase in the GHG intensity per kg CW and therefore total GHG emission of the production system e.g. 1c vs. 3c (Table 8), 1a vs. 3a (Fig. 3). However the study also found that increasing stocking rate by changing from a SH system to BH system not only increased carcass output ha^{-1} and net margin ha^{-1} but also reduced the GHG intensity per kg of beef and N_2O kg CW^{-1}.

It must be noted that in Ireland there is greater risk of financial loss with bull beef systems. Bulls are more reliant on higher cost feeds than steers to meet animal carcass specifications. Moreover, a bull finishing period is 3–4 months longer than steers and carried out indoors. Thus, bull systems are more specialized than steer systems requiring better housing facilities (e.g. lying space and ventilation) and excellent handling facilities to minimize safety risks. Bull finishing systems may also not qualify for higher value markets, e.g. "grass-fed".

In regard to food demand, the effect of stocking rate on productivity, profitability and environmental sustainability must be viewed from a global perspective. It has been predicted that global bovine meat demand is to increase by 1.8% per annum between 2012 and 2021 (OECD/FAO, 2012). Therefore, one option to meet such demand is the intensification of production. The current study has shown the ability of grass-based suckler to beef systems to increase productivity on the same land area with a marginal increase in N_2O emissions kg CW^{-1} through intensification and reduction in N_2O emissions kg CW^{-1} by converting from a SH system to a BH system. Furthermore, along with the improvement in grassland management, beef systems can become more efficient through the improvement in nutrient utilisation efficiency. This is of particular importance for N_2O emissions from fertiliser and manure application where variables such as fertiliser type, application rate and timing of application all have significant effect on nutrient utilisation and amount of N applied lost as N_2O.

4.1. Nitrous oxide emission factors

4.1.1. Manure deposition N_2O

All of the studies discussed previously used either IPCC Tier 1 emission factors (IPCC, 2006, 1997) or default EFs derived from a compilation of international data (Oenema et al., 1997) to calculate N_2O emissions from faecal deposition onto pastures. The original BEEFGEM model used the default IPCC EF of 2% for ruminant excreta onto pastures to estimate N_2O. The default EF was developed from numerous international studies, some of which were laboratory based. Therefore the default EF of 2% may not necessarily be representative of the conditions in a specific country (Bell et al., 2015). For example, Oenema et al. (1997) reported that the urinary N_2O EF used in developing the default EF of 2% varied widely from 0.1–1.4% (Yamulki and Jarvis, 1997) to 1.5–9.9% (Velthof et al., 1996). This proves that large variability occurs under different conditions and confirms the need for the development and adoption of Tier 2 country specific EF.

The amount and duration of N_2O emissions varies greatly with different soil types, climates, type of excreta and timing of deposition, factors not taken into consideration in the default EF (Krol et al., 2016). Additionally, de de Klein et al. (2003) state that the IPCC encourage the development of tier 2 country specific EF for key N emission sources. Along with being country specific, Krol et al. (2016) developed separate EFs for urine and dung for each season on different levels of soil drainage. When these estimates were averaged, the N_2O EF for dung and urine was 0.31 and 1.18%, respectively. This is considerably less than the default IPCC (2006) EF of 2%, validating the need for a more dynamic model.

The EFs developed by Krol et al. (2016) are similar to those developed by Bell et al. (2015) who reported EF of 0.2% for cattle dung and 1.1% for urine excreted onto Scottish pastures. In New Zealand, the IPCC (2006) default EF is not now used and has been replaced by a country specific average EF of 1%. Similar to the average of those developed by Bell et al. (2015), de de Klein et al. (2003) and Krol et al. (2016) state that soil and season specific N_2O EF should be used if possible. Therefore, the N_2O EF developed by Krol et al. (2016) for dung and urine excreted onto pastures were included in the model.

4.1.2. N fertiliser application N_2O

The majority of studies discussed used the IPCC tier 1 default EF of either 1.25% (Casey and Holden, 2006; Foley et al., 2011) or 1% (Clarke et al., 2013) to estimate N_2O emissions from fertiliser application. Meta-analysis showed that there was a difference in N_2O emissions between fertiliser types (Bouwman and Boumans, 2002) where AN and CAN had higher N_2O EFs than urea (Dobbie and Smith, 2003). Similar to excreta deposited onto pasture, N_2O emissions from synthetic fertiliser application varies with different types of fertiliser, timing of application, and soil and climatic conditions (Harty et al., 2016).

Despite the variation which occurs with N_2O emissions from fertiliser application, the IPCC utilise a default EF of 1% and previously 1.25% that disregarded the effect of soil conditions or fertiliser type (IPCC, 2006). To account for the variation which occurs in N_2O emission from fertiliser application, the BEEFGEM model IPCC default EF for this source was replaced with direct EFs developed by Harty et al. (2016) for CAN and urea applied to different soil types with different levels of drainage. The range of EFs used in this study (0.1–3.81%) was somewhat similar to the EFs developed by Dobbie and Smith (2003) on Scottish pastures (1–3%). On average, the direct N_2O EF for CAN and urea fertiliser application was 1.49% and 0.25%, respectively. This results in CAN and urea having an EF approximately 50% more and 75% less, respectively, than the default IPCC (2006) EF of 1%.

4.2. Total emissions

Comparison of modelling and LCA studies can be difficult due to differences in modelling methodologies, emission factors used and variation in production systems (de Boer, 2003; Crosson et al., 2011). Nevertheless, comparing studies can be useful as it helps to validate outcomes of a study and identify emission reduction strategies. Previous studies of emissions from Irish beef systems by Clarke et al. (2013), Foley et al. (2011), and Casey and Holden (2006), assessed production systems similar to the current study and used similar functional units, coefficients and emission factors. Foley et al. (2011) reported total GHG emissions of 19.7 and 22 kg CO_2e kg CW^{-1} and 18.9 and 20.4 kg CO_2e kg CW^{-1} for moderate and intensive steer and bull production systems, respectively. Clarke et al. (2013) investigated GHG emissions from grass-based suckler bull and steer beef systems at stocking rates ranging from 150 to 220 kg organic N ha^{-1} and reported similar CO_2e kg CW^{-1} to Foley et al. (2011), ranging from 20.1–20.9 CO_2e kg CW^{-1} and 22.3–23.1 kg CO_2e kg CW^{-1} for bull and steer suckler production systems, respectively. Casey and Holden (2006) reported that a typical Irish suckler to beef production system finishing cattle at approximately 24 months at 650 kg live weight produced 11.26 kg CO_2e kg live $weight^{-1}$ (LW). When converted into the functional units of the current study using the SH system conversion factor in Table 6 (54.8%), the GHG emissions were 20.55 kg CO_2e kg CW^{-1}, which was similar to average emissions reported by Clarke et al. (2013) and Foley et al. (2011).

In international studies, Beauchemin et al. (2010) conducted an LCA to estimate the whole farm GHG emissions of beef production in western Canada. Both cow-calf ranching and beef feedlot finishing operations were combined in one farm to account for emissions for an entire cycle. The study reported GHG emissions of 21.73 kg CO_2e kg CW^{-1}. This was within the ranges of the GHG emissions calculated for Irish grass-based studies discussed earlier. A similar study by Alemu et al. (2017) investigating the impact of grazing management strategy and stocking rate on the GHG intensity of beef from natural rangelands in western Canada reported a slightly higher GHG intensity of 24.1–26.6 kg CO_2e kg CW^{-1}. Greater GHG emissions were found by Pelletier et al. (2010) who conducted an LCA to compare the GHG emissions associated with three beef production strategies used in the Upper Midwestern United States, namely feedlot, backgrounding/feedlot and pasture. The outcomes of that study showed that when viewed on an LW production basis, feedlot and grass-based systems had GHG emissions of 14.8 kg CO_2e kg LW^{-1} ($\approx$27.00 kg CO_2e kg CW^{-1}) and 19.2 kg CO_2e kg LW^{-1} ($\approx$35.00 kg CO_2e kg CW^{-1}), respectively. This was consistent with Casey and Holden (2006) who found that by providing a higher quality diet to sustain a higher ADG and younger slaughter age, GHG emissions from key sources (i.e., CH_4 from enteric fermentation and N_2O from excreta at pasture) were reduced. Similar results were found in the current study when BH systems finishing bulls at 16 months were compared to SH systems finishing steers at 24 months within scenario, e.g. 1a vs. 1c, 1b vs. 1d (Table 8).

Pelletier et al. (2010) and Beauchemin et al. (2010) reported that the cow-calf phase of the cattle life cycle contributed the most GHG emissions (61% of total GHG emissions). Pelletier et al. (2010) also found that as forage utilisation rate increased, GHG emissions per kg of CW reduced. However, when the grass-based system simulated by Pelletier et al. (2010) had a utilisation rate similar to the production systems of the current study, GHG emissions were higher than those of the grass-based Irish studies ($\approx$33.6 kg CO_2e kg CW^{-1}) thus highlighting the effect of climate and region on GHG emissions from grass based beef production systems.

The GHG emissions in the present study were significantly lower than the national and international studies discussed previously. For example, our estimates for GHG emission intensity were 1.16–2.60 kg CO_2e kg CW^{-1} lower than the GHG emissions reported for the same research suckler systems by Foley et al. (2011). This difference was largely associated with N_2O emissions where Foley et al. (2011) predicted that 27–31% and 25–29% of GHG emissions emitted from SH and BH systems were from N_2O. In contrast, our results showed that 11.4–15.3% of total GHG emissions were from N_2O emissions. The significant reduction in our N_2O emission estimates can be explained by replacing BEEFGEM's N_2O EF from the IPCC (2006) with recently developed national N_2O EFs for fertiliser application (Harty et al., 2016) and deposition of manure onto pasture (Krol et al., 2016). When IPCC Tier 1 emission factors for synthetic fertiliser (1% N applied) and deposition of manure onto pasture (2% manure applied) were used in the current model, the GHG emissions for the four baseline systems, SH INT, SH EXT, BH INT and BH EXT increase to 21.2, 20.2, 20.6 and 19.3 kg CO_2e kg CW^{-1}, respectively, producing similar results previously reported by Foley et al. (2011), Clarke et al. (2013), Casey and Holden (2006) and Beauchemin et al. (2010). Therefore, the significant change in emission estimates highlights the need for the development of country specific EFs for key GHG sources to avoid the over or under estimation of GHG emissions and to reduce the reliance on IPCC tier 1 EFs that may not be representative of the conditions of a country, region or production system.

4.3. Profitability and emission efficiency

White et al. (2010) found a similar trend to the current study where the increase in fertiliser N application to support higher stocking rate resulted in an increase in GHG emissions per kg CW. It was stated that the main cause for the increase was the exponential increase in N_2O and CO_2 emissions with the increase in fertiliser application rate per ha. It has also been reported that higher stocking rates increase grass utilisation and thus increase profitability while only modestly increasing GHG emissions (Crosson et al., 2007; Shalloo et al., 2007; Clarke et al., 2013). This is in agreement with our study which consistently showed that for all four suckler to beef systems, the increase in stocking rate through the increase in fertiliser application rate resulted in an increase in profitability with marginal increases in N_2O emissions.

Delaying silage harvest date was shown to have a negative effect on profitability ha^{-1} (Fig. 3). As seen in the studies by Keady et al. (2000), Dawson et al. (2001) and O'Kiely (2004), as grass matures its digestibility and nutritional quality declines. Therefore, to ensure that animals consume adequate energy and nutrients to meet their ADG targets, concentrate feed consumption must increase. This is shown in Table 8 where systems with early silage harvest date having on average 0.26 t lower concentrate feed requirements per ha than systems with late silage harvest date e.g. 1a–1d vs. 2a–2d. Therefore, using the price of €238 per tonne of concentrate feed (Table 4), concentrate feed costs were approximately €61 greater per ha in systems with late silage harvest date (2a–2d, 4a–4d) than systems with early silage harvest date. This is consistent with the findings of Keady et al. (2013) who found that for each unit decline in silage digestibility, an additional 0.4 kg concentrates is required daily to sustain the performance of finishing cattle. An important factor with nutrient and energy intake is the substitution rate. This can have a significant effect on profitability where it has been shown by McGee (2014) that the cost per kg CW will rise with the increase in concentrate feeding level. This is depicted in the current study, as concentrate feed costs increased with later first cut silage harvest dates and profitability per ha decreased.

The emissions efficiency for each scenario was calculated to determine the effect of different management practices (fertiliser application rate and silage harvest date) on GHG emissions and economic performance simultaneously. Similar to the study by Clarke et al. (2013), the current study found that the emissions efficiency for all four systems (SH INT, SH EXT, BH INT, BH EXT) were higher at the increased fertiliser application rate scenarios than reduced fertiliser application rate scenarios e.g. 1a–1d vs. 3a–3d (Fig. 3). However as with Clarke et al. (2013), there was a diminishing returns response where a greater response was reported for the EXT systems than the INT systems across all four scenarios. For example the emission efficiency of BH INT and SH INT system with increased fertiliser application rate was on average 0.61 and 1.51 € net margin tCO_2e^{-1} greater that systems with reduced fertiliser application rate, e.g. 1a vs .3a, 2c vs. 4c (Fig. 3). In contrast the emission efficiency of BH EXT and SH EXT systems with increased fertiliser application rate was on average 6.66 and 7.78 € net margin tCO_2e^{-1} greater than systems with reduced fertiliser application rate e.g. 1b vs. 3b, 2d vs. 4d) (Fig. 3). However, the increase was marginal, where similar to Foley et al. (2011), the increase in GHG emissions from the increase in fertiliser application rate and hence stocking rate is offset by the increase in productivity associated with higher stocking rates. This in turn resulted in the slight increase in emission efficiency. Therefore, this indicates that the intensification of suckler to beef production systems must be appropriate to ensure that sustainable production is maintained.

The productivity of an average grass-based commercial suckler to beef farm is much lower than the research farm systems analysed in the current study. The average stocking rate for Irish suckler farms in 2016 was 1.34 LU ha^{-1} (Dillon et al., 2016). Along with this, based on National Farm Survey data, Foley et al. (2011) estimated that the average suckler to beef farm in Ireland is making a loss of €67 ha^{-1}. This consequently results in the emission efficiency also being negative. Based on the findings of this study, there is great scope for the improvement of the efficiency and profitability of the average suckler to beef systems through appropriate intensification.

5. Conclusions

The inclusion of new nutritional research findings into GBSM has greatly improved our understanding of the effect of production systems and management practices on the N content of fresh and conserved grass and cattle excretion and partitioning of N. Similarly, updating N_2O emission factors has enhanced our ability to assess the effect changes in farm performance can have on emissions from grazing cattle and fertiliser. Our modelling study showed that SH suckler beef systems emit more N_2O and total GHG emissions per kg carcass weight than BH systems. This was facilitated by the higher stocking rate and thus productivity per ha of BH systems in comparison to SH systems. When emission efficiency was analysed, it showed that BH systems consistently produced greater € net margin tCO_2e^{-1} than SH systems within scenario. Finishing bulls rather than steers resulted in higher carcass weight gain day^{-1} and output ha^{-1}, and hence higher profit ha^{-1} and emission efficiency. It is essential to note however that there are negatives with bull beef systems and not all farms are capable of adopting bull beef production systems. The study showed that improving farm management practices mitigated both SH and BH systems emissions. It was shown that applying fertiliser under appropriate conditions and at a higher application rate will increase grass growth ha^{-1}, allowing for higher stocking rate ha^{-1}, leading to improved productivity and efficiency for BH systems. However, a diminishing returns response was found between fertiliser application rate and emission efficiency, particularly in INT systems. Harvesting silage earlier in the season increased profit ha^{-1} and improved emission efficiency as a result of higher forage quality during housing, thus reducing the need for concentrate feed to achieve ADG targets. Overall, these examples demonstrate that implementing good farming practices can simultaneously improve farm income with little or no negative on GHG emissions.

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Appendix A. Supplementary data

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References

Alemu, A.W., Janzen, H., Little, S., Hao, X., Thompson, D.J., Baron, V., Iwaasa, A., Beauchemin, K.A., Kröbel, R., 2017. Assessment of grazing management on farm greenhouse gas intensity of beef production systems in the Canadian prairies using life cycle assessment. Agric. Syst. 158, 1–13.

Beauchemin, K.A., Janzen, H.H., Little, S.M., McAllister, T.A., McGinn, S.M., 2010. Life cycle assessment of greenhouse gas emissions from beef production in western Canada: a case study. Agric. Syst. 103, 371–379.

Bell, M.J., Rees, R.M., Cloy, J.M., Topp, C.F.E., Bagnall, A., Chadwick, D.R., 2015. Nitrous oxide emissions from cattle excreta applied to a Scottish grassland: effects of soil and climatic conditions and a nitrification inhibitor. Sci. Total Environ. 508, 343–353.

Bouwman, A.F., Boumans, J.M., 2002. Emissions of N_2O and NO from fertilised fields: summary of available measurement data. Global Biochem. Cycles 16, 6-1–6-13.

Bragaglio, A., Napolitano, F., Pacelli, C., Pirlo, G., Sabia, E., Serrapica, F., Serrapica, M., Braghieri, A., 2018. Environmental impacts of Italian beef production: a comparison between different systems. J. Clean. Prod. 172, 4033–4043.

Breen, J.P., Donnellan, T., Westhoff, P., 2010. Food for thought: EU climate change policy presents new challenges for agriculture. EuroChoices 9, 387–394.

British Standards Institution (BSI), 2011. PAS 2050:2011 Specification for the Assessment of the Life Cycle Greenhouse Gas Emissions of Goods and Services.

Casey, J.W., Holden, N.M., 2006. Quantification of GHG emissions from sucker-beef production in Ireland. Agric. Syst. 90, 79–98.

Castillo, A.R., Kebreab, E., Beever, D.E., Barbi, J.H., Sutton, M.A., Kirby, H.C., France, J., 2001. The effect of protein supplementation on nitrogen utilization in lactating dairy cows fed grass silage diets. J. Anim. Sci. 79, 247–253.

Central Statistics Office (CSO), 2017. Area under Crops and Pasture in June, Agriculture, Statistical Yearbook of Ireland 2017. Cork, Ireland. https://www.cso.ie/en/releasesandpublications/ep/p-syi/psyi2017/agri/cl/#d.en.132486, Accessed date: 9 November 2018.

Clarke, A.M., Brennan, P., Crosson, P., 2013. Life-cycle assessment of the intensity of production on the greenhouse gas emissions and economics of grass-based suckler beef production systems. J. Agric. Sci. 151, 714–726.

Commission of the European Communities, 1982. Commission of the European Communities (Beef Carcass Classification) Regulations. (Council Regulations 1358/80, 1208/81, 1202/82).

Conen, F., Dobbie, K.E., Smith, K.A., 2000. Predicting N_2O emissions from agricultural land through related soil parameters. Glob. Chang. Biol. 6, 417–426.

Crosson, P., O'Kiely, P., O'Mara, F.P., Wallace, M., 2006. The development of a mathematical model to investigate Irish beef production systems. Agric. Syst. 89, 349–370.

Crosson, P., O'Kiely, P., O'Mara, F.P., Wallace, M., 2007. Optimal beef production system in different concentrate price and grass utilisation scenarios. Proc. Agric. Res. Forum 2007, 51.

Crosson, P., Shalloo, L., O'Brien, D., Lanigan, G.J., Foley, P.A., Boland, T.M., Kenny, D.A., 2011. A review of whole farm systems models of greenhouse gas emissions from beef and dairy cattle production systems. Anim. Feed Sci. Technol. 166–167, 29–45.

Crosson, P., McGee, M., Fox, P., 2014. Technologies Underpinning Grass Based Suckler Beef Systems. Beef 2014, Grange. pp. 14–19.

CSO, Central Statistics Office, 2018a. Agriculture Fertiliser Price (Euro per Tonne) by Type of Fertiliser and Month January 2016–December 2017.

Central Statistics Office, 2018b. Agriculture Feed Stuff Price (Euro per Tonne) by Type of Feedstuff and Month January 2016-December 2017.

CSO, Central Statistics Office, Agriculture Feed Stuff Price (Euro per Tonne) by Type of Feedstuff and Month January 2016-December 2017, 2018 Davidson, E.A., 2009. The contribution of manure and fertiliser nitrogen to atmospheric nitrous oxide since 1860. Nat. Geosci. 2, 659–662.

Dawson, L.E., Kirkland, R.M., Ferris, C.P., Steen, R.W.J., Kilpatrick, D.J., Gordon, F.J., 2001. The effect of stage of perennial ryegrass maturity at harvesting, fermentation characteristics and concentrate supplementation, on the quality and intake of grass silage by beef cattle. Grass Forage Sci. 57, 255–267.

de Boer, I.J.M., 2003. Environmental impact assessment of conventional and organic milk production. Livest. Prod. Sci. 80, 69–77.

de Klein, C.A.M., Barton, L., Sherlock, R.R., Li, Z., Littlejohn, R.P., 2003. Estimating a nitrous oxide emission factor for animal urine from some New Zealand pastoral soils. Aust. J. Soil Res. 41, 381–399.

Department of Agriculture Food and the Marine (DAFM), 2018. Beef Factory Prices (Weekly Reports). https://www.agriculture.gov.ie/farmingsectors/beef/beeffactorypricesweeklyreports/ (accessed 12 October 2018).

Di, H., Cameron, K., 2002. Nitrate leaching in temperate agroecosystems: sources, factors and mitigating strategies. Nutr. Cycl. Agroecosyst. 64, 237–256.

Dijkstra, F.A., Oenema, O., van Groenigen, J.W., Spek, J.W., van Vuuren, A.M., Bannink, A., 2013. Diet effects on urine composition of cattle and N_2O emissions. Animal 7, 292–302.

Dillon, E., Moran, B., Donnellan, T., 2016. Teagasc National Farm Survey 2016 Results. Agricultural Economics and Farm Surveys Department, Rural Economy Development Programme, Teagasc, Athenry, Co Galway, Ireland.

Dobbie, K.E., Smith, K.A., 2003. Impact of different forms of N fertiliser on N_2O emissions from intensive grassland. Nutr. Cycl. Agroecosyst. 67, 37–46.

Drennan, M.J., McGee, M., 2009. Performance of spring-calving beef suckler cows and their progeny to slaughter on intensive and extensive grassland management systems. Livest. Sci. 120, 1–12.

Environmental Protection Agency (EPA), 2017. Ireland's Informative Inventory Report 2017. Air Pollutant Emissions in Ireland 1990–2015. Johnstown Castle Estate, Co. Wexford, Ireland.

European Environment Agency (EEA), 2018. Annual European Union Greenhouse Gas Inventory 1990–2016 and Inventory Report 2018. https://unfccc.int/process/transparency-and-reporting/reporting-and-review-under-the-convention/greenhouse-gas-inventories-annex-i-parties/national-inventory-submissions-2018 (accessed 01 November 2018).

FAOSTAT, 2017. Food and Agriculture Organisation of the United Nations Statistic Division, Domains, Land Use. http://www.fao.org/faostat/en/#data/RL (accessed 8 November 2018).

Firestone, M.K., Davidson, E.A., 1989. Microbiological Basis of NO and N_2O Production and Consumption in Soil. pp. 7–21.

Foley, P.A., Crosson, P., Lovett, D.K., Boland, T.M., O'Mara, F.P., Kenny, D.A., 2011. Whole-farm systems modelling of greenhouse gas emissions from pastoral suckler

beef cow production systems. Agric. Ecosyst. Environ. 142, 222–230.
Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D.W., Haywood, J., Lean, J., Lowe, D.C., Myhre, G., Nganga, J., Prinn, R., Raga, G., Schulz, M., Van Dorland, R., 2007. Changes in atmospheric constituents and in radiative forcing. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L. (Eds.), Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge United Kingdom and New York, NY, USA.
Gately, T.F., Ryan, M., Doyle, L., 1972. Effect of nitrogen on the yield, total-N and nitrate-N content of herbage over the growing season. Irish J. Agr. Food Res. 11, 63–75.
Gregorini, P., Beukes, P., Dalley, D., Romera, A., 2016. Screening for diets that reduce urinary nitrogen excretion and methane emissions while maintaining or increasing production by dairy cows. Sci. Total Environ. 551–552, 32–41.
Harty, M.A., Forrestal, P.J., Watson, C.J., McGeough, K.L., Carolan, R., Elliot, C., Krol, D., Laughlin, R.J., Richards, K.G., Lanigan, G.J., 2016. Reducing nitrous oxide emissions by changing N fertiliser use from calcium ammonium nitrate (CAN) to urea based formulations. Sci. Total Environ. 563–564, 576–586.
Hennessy, T., Moran, B., 2016. Teagasc National Farm Survey 2015 Results. Agricultural Economics and Farm Surveys Department, Rural Economy Development Programme, Teagasc, Athenry, Co Galway, Ireland. https://www.teagasc.ie/media/website/publications/2016/Income-Estimates-2015.pdf (accessed 10 May 2018).
Hoekstra, N.J., Schulte, R.P.O., Struik, P.C., Lantinga, E.A., 2007. Pathways to improving the N efficiency of grazing bovines. Eur. J. Agron. 26, 363–374.
Horan, B., 2014. Effects of stocking rate and calving date in dairy systems-Teagasc Technology Updates. accessed 10 February 2018. https://www.teagasc.ie/media/website/publications/2014/5891_Effect_of_SR.pdf.
Hynes, D.N., Stergiadis, S., Gordon, A., Yan, T., 2016. Effects of crude protein level in concentrate supplements on animal performance and nitrogen utilization of lactating dairy cows fed fresh cut perennial grass. J. Dairy Sci. 99, 8111–8120.
Intergovernmental Panel on Climate Change (IPCC), 1997. Greenhouse Gas Inventories. Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories. OECD.
Intergovernmental Panel on Climate Change (IPCC), 2006. IPCC guidelines for national greenhouse gas inventories. In: Egglestone, H., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), The National Greenhouse Gas Inventory Programme, Intergovernmental Panel on Climate Change. IGES, Japan.
Intergovernmental Panel on Climate Change (IPCC), 2013. Climate change 2013: the physical science basis. In: Hartmann, D.L., Klein Tank, A.M.G., Rusticucci, M., Alexander, L.V., Brönnimann, S., Charabi, Y.F.J. ... Zhai, P.M. (Eds.), The Fifth Assessment Report. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
Intergovernmental Panel on Climate Change (IPCC), 2014. Climate change 2014: mitigation of climate change. In: Smith, P., Bustamante, M., Ahammad, H., Clark, H., Dong, H., Elsiddig, E.A. ... Tubiello, F. (Eds.), Agriculture, Forestry and Other Land Use (AFOLU). Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge, United Kingdom, and New York, NY, United States.
International Organisation for Standardization (ISO), 2006. Environmental Management – Life Cycle Assessment – Principles and Framework. ISO 14040: 2006.
Jarrige, R., 1989. Ruminant Nutrition. Recommended Allowance and Feed Tables. John Libbey Eurotext, Montrougue, France.
Jarvis, S.C., Hatch, D.J., Roberts, D.H., 1989. The effects of grassland management on nitrogen losses from grazed swards through ammonia volatilisation; the relationship to excretal returns from cattle. J. Agric. Sci. 112, 205–216.
Keady, T.W.J., Mayne, C.S., Fitzpatrick, D.A., 2000. Prediction of silage feeding value from the analysis of the herbage at ensiling and the effects of nitrogen fertiliser, date of harvest and additive treatment on grass silage composition. J. Agric. Sci. 134, 353–368.
Keady, T., Hanrahan, S., Marley, C.L., Scollan, N.D., 2013. Production and utilization of ensiled forages by beef cattle, dairy cows, pregnant ewes and finishing lambs – a review. Agric. Food Sci. 22, 70–92.
Kebreab, E., France, J., Beever, D.E., Castillo, A.R., 2001. Nitrogen pollution by dairy cows and its mitigation by dietary manipulation. Nutr. Cycl. Agroecosyst. 60, 275–285.
Krol, D.J., Carolan, R., Minet, E., McGeough, K.L., Watson, C.J., Forrestal, P.J., Lanigan, G.J., Richards, K.G., 2016. Improving and disaggregating N_2O emission factors for ruminant excreta on temperate pasture soils. Sci. Total Environ. 568, 327–338.
Kyne, S., Drennan, M.J., Caffery, P.J., 2001. Influence of concentrate level during winter and date of turn out to pasture on the performance of cattle and the effect of grazing of silage ground on grass yield and quality. Irish J. Agr. Food Res. 40, 23–32.
McGee, M., 2014. Feed Efficiency in Beef Finishing Systems. Teagasc-IGFA Nutrition Conference – June 2014.
Misselbrook, T., Fleming, H., Camp, V., Umstatter, C., Duthie, C.A., Nicoll, L., Waterhouse, T., 2016. Automated monitoring of urination events from grazing cattle. Agric. Ecosyst. Environ. 230, 191–198.
Mulligan, F.J., Dillon, P., Callan, J.J., Rath, M., O'Mara, F.P., 2004. Supplementary concentrate type affects nitrogen excretion of grazing dairy cows. J. Dairy Sci. 87, 3451–3460.
Murphy, B.M., Drennan, M.J., O'Mara, F.P., McGee, M., 2008a. Post weaning growth, ultrasound and skeletal measurements, muscularity scores and carcass traits and composition of progeny of five beef suckler cow genotypes. Irish J. Agr. Food Res. 47, 27–40.
Murphy, B.M., Drennan, M.J., O'Mara, F.P., McGee, M., 2008b. Performance and feed intake of five beef suckler cow genotypes and pre weaning growth of their progeny. Irish J. Agr. Food Res. 47, 13–25.
Oenema, O., Velthof, G.L., Yamulki, S., Jarvis, S.C., 1997. Nitrous oxide emissions from grazed grassland. Soil Use Manag. 13, 288–295.
O'Kiely, P., 2004. Rates of changes in yield and digestibility of grasses grown for silage. Proc. Agric. Res. Forum 51.
O'Mara, F.P., Caffery, P.J., Drennan, M.J., 1997. Net energy values of grass silage determined from comparative feeding trials. Irish J. Agr. Food Res. 36, 110.
Organisation for Economic Co-operation and Development/Food and Agriculture Organisation (OECD/FAO), 2012. OECD/FAO Agricultural Outlook 2012-2021. OECD Publishing and FAO, pp. 163–170. https://doi.org/10.1787/agr_outlook-2012-en. (accessed 8 November 2018).
Pelletier, N., Pirog, R., Rasmussen, R., 2010. Comparative life cycle environmental impacts of three beef production strategies in the Upper Midwestern United States. Agric. Syst. 103, 380–389.
Ryan, M., Brophy, C., Connolly, J., McNamara, K., Carton, O.T., 2006. Monitoring of nitrogen leaching on a dairy farm during four drainage seasons. Irish J. Agr. Food Res. 45, 115–134.
Selbie, D.R., Buckthought, L.E., Shepherd, M.A., 2015. The challenge of the urine patch for managing nitrogen in grazed pasture systems. Adv. Agron. 129, 229–292.
Shalloo, L., O'Donnell, S., Horan, B., 2007. Profitable dairying in an increased EU milk quota scenario. In: Exploiting the Freedom to Milk: Proceedings of the National Dairy Conference, 21–22 November 2007, pp. 20–44 Kilkenny and Mayo.
Signor, D., Cerri, C.C., 2013. Nitrous oxide emissions in agricultural soils: a review. Pesquisa Agropecuaria Trop. 43, 322–338.
Sneath, R.W., Beeline, F., Hilhorst, M.A., Peu, P., 2006. Monitoring GHG from manure stores on organic and conventional dairy farms. Agric. Ecosyst. Environ. 121, 121–134.
Stergiadis, S., Allen, M., Chen, X., Wills, D., Yan, T., 2015. Prediction of metabolisable energy concentrations of fresh-cut grass using digestibility data measured with non-pregnant non-lactating cows. Br. J. Nutr. 113, 1571–1584.
van Groenigen, J.W., Kuikman, P.J., de Groot, W.J.M., Velthof, G.L., 2005. Nitrous oxide emission from urine-treated soil as influenced by urine composition and soil physical conditions. Soil Biol. Biochem. 37, 463–473.
Velthof, G.L., Brader, A.B., Oenema, O., 1996. Seasonal variations in nitrous oxide losses from managed grassland in the Netherlands. Plant Soil 181, 263–274.
Vleeshouwer, L.M., Verhagen, A., 2002. Carbon emissions and sequestration by agricultural land use: a model study for Europe. Glob. Chang. Biol. 8, 519–530.
Welten, B.G., Ledgard, S.F., Schipper, L.A., Judge, A.A., 2013. Effect of amending cattle urine with dicyandiamide on soil nitrogen dynamics and leaching of urinary-nitrogen. Agric. Ecosyst. Environ. 167, 12–22.
White, T.A., Snow, V.O., King, W.M., 2010. Intensification of New Zealand beef farming systems. Agric. Syst. 103, 21–35.
Yamulki, S., Jarvis, S.C., 1997. Nitrous oxide emissions from excreta from simulated grazing pattern and fertiliser application to grassland. In: Proceedings International Conference on Nitrogen Emissions from Grassland; Gaseous Transfers and their Interactions.
Yan, T., Frost, J.P., Keady, T.W.J., Agnew, R.E., Mayne, C.S., 2007. Prediction of nitrogen excretion in faeces and urine of beef cattle offered diets containing grass silage. J. Anim. Sci. 85, 1982–1989.
Zaman, M., Saggar, S., Blennerhassett, J.D., Singh, J., 2009. Effect of urease and nitrification inhibitors on N transformation, gaseous emissions of ammonia and nitrous oxide, pasture yield and N uptake in grazed pasture system. Soil Biol. Biochem. 41, 1270–1280.
Zaman, M., Zaman, S., Nguyen, M.L., Smith, T.J., Nawaz, S., 2013. The effect of urease and nitrification inhibitors on ammonia and nitrous oxide emissions from simulated urine patches in pastoral system: a two-year study. Sci. Total Environ. 465, 97–106.
Zou, C.X., Lively, F.O., Wylie, A.R.G., Yan, T., 2015. Estimation of the maintenance energy requirements, methane emissions and nitrogen utilisation efficiency of two suckler genotypes. J. Anim. Sci. 10, 616–622.