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The simulated environmental impact of incorporating white clover into pasture-based dairy production systems

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ABSTRACT

White clover (WC) offers an alternative source of nitrogen (N) for pasture-based systems. Substituting energy- and carbon-intensive synthetic N fertilizers with N derived from biological fixation by WC has been highlighted as a promising environmental mitigation strategy through the omission of emissions, pollutants, and energy usage during the production and application of synthetic fertilizer. Therefore, the objective was to investigate the effect of the inclusion of WC in perennial ryegrass (PRG) swards on the environmental impact of pasture-based dairy systems. Cradle-to-farm gate life cycle assessment of 3 pasture-based dairy systems were conducted: (1) a PRG–WC sward receiving 150 kg of N/ha per year (CL150), (2) a PRG–WC sward receiving 250 kg of N/ha per year (CL250), and (3) a PRG-only sward receiving 250 kg of N/ha per year (GR250). A dairy environmental model was updated with country-specific N excretion equations and recently developed N_2O, NH_3, and NO_3^- emission factors. The environmental impact categories assessed were global warming potential, nonrenewable energy, acidification potential, and eutrophication potential (marine and freshwater). Impact categories were expressed using 2 functional units: per hectare and per metric tonne of fat- and protein-corrected milk. The GR250 system had the lowest milk production and highest global warming potential, nonrenewable energy, and acidification potential per tonne of fat- and protein-corrected milk for all systems. The CL250 system produced the most milk and had the highest environmental impact across all categories when expressed on an area basis. It also had the highest marine eutrophication potential for both functional units. The impact category freshwater eutrophication potential did not differ across the 3 systems. The CL150 system had the lowest environmental impact across all categories and functional units. This life cycle assessment study demonstrates that the substitution of synthetic N fertilizer with atmospheric N fixed by WC has potential to reduce the environmental impact of intensive pasture-based dairy systems in temperate regions, not only through improvement in animal performance but also through the reduction in total emissions and pollutants contributing to the environmental indicators assessed.
Key words: life cycle assessment, milk production, perennial ryegrass, white clover, dairy

INTRODUCTION

Globally, dairy producers are faced with the challenge of supporting growing populations by providing food that is nutritious and economically viable while simultaneously increasing the resource use efficiency and sustainability of milk production (Miller and Auestad, 2013). In temperate regions, it is common practice for dairy producers to concentrate calving in spring to match herd feed demand with grass supply, therefore optimizing milk production from pasture and minimizing production costs (Dillon et al., 1995; Shalloo et al., 2004). Synthetic fertilizers are commonly applied to increase forage yield and optimize production. Although synthetic fertilizers are required to underpin the growth in the global population, the manufacture and application of these essential farm inputs are significant contributors to the environmental footprint of agricultural products (Brentrup et al., 2016). Synthetic nitrogen (N) fertilizer has been reported to be one of the main sources of nitrous oxide (N_2O) emissions and is considered to be one of the main causes of the 20% increase in atmospheric N_2O concentration from preindustrial concentrations of 270 ppb to 324 ppb (IPCC, 2013). This is of great concern

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as N_2O is a potent greenhouse gas (**GHG**) with global warming potential (**GWP**) 265 times stronger than that of carbon dioxide (CO_2) over a 100-yr time horizon (IPCC, 2013). In addition, excessive N fertilizer application can lead to a host of other environmental problems such as the eutrophication and acidification of terrestrial and aquatic ecosystems (Bouwman et al., 2002; Chislock et al., 2013).

White clover (**WC**; *Trifolium repens L.*) is the most common legume in temperate grasslands, where it fixates N from the atmosphere and releases it into the soil as plant-available N over time through exudates in the root structure to companion plants (Ledgard et al., 2009). In cool temperate regions, perennial ryegrass (**PRG**; *Lolium perenne* L.) swards with WC have been reported by Andrews et al. (2007) to produce similar pasture yields as PRG-only swards receiving 200 kg of N/ha. Incorporating WC into a PRG sward therefore poses an alternative N source and potential strategy to displace synthetic N fertilizers and mitigate their associated environmental impacts.

Measuring the environmental impact of agricultural processes (e.g., enteric fermentation of forage by cattle) is expensive, technically challenging, and often laborious. It is practically impossible to measure the environmental burden of every life stage of a product. Therefore, to address this type of problem and growing environmental concerns, a modeling technique known as life cycle assessment (**LCA**) was developed. The primary aim of the LCA technique is to compile the inputs and outputs for a system and simulate the potential environmental impacts of the processes related to the final product(s) of the system (ISO, 2006a; Guineé et al., 2011). It is widely used in research and has become a globally accepted methodology for quantifying the environmental impact of agricultural products (de Vries et al., 2015).

Some LCA studies of semiextensive dairy systems have highlighted that WC in PRG swards mitigates environmental impact (Schils et al., 2005; Yan et al., 2013). However, these findings may not be applicable for intensive pasture-based dairy systems. Yan et al. (2013) and Schils et al. (2005) largely focused on environmental impact in terms of GHG emissions, which is common. In a systematic review of 173 publications by McClelland et al. (2018), the most frequently included impact category was GWP (98%). Of the publications reviewed, 27% solely reported GWP. Along with being identified as key contributors to global agricultural GHG emissions, intensive livestock systems have accelerated the loading of limiting nutrients to water bodies, resulting in the deterioration of aquatic ecosystems in some areas (McDowell and Hamilton, 2013). Furthermore, livestock systems emit acidifying air pollutants; for example, 99% of Irish ammonia (NH_3) emissions are produced by the agricultural sector (Duffy et al., 2019a). Not only do acidifying air pollutants reduce air quality, they also negatively affect nearby sensitive ecosystems (Goulding, 2016). Therefore, it is essential to assess multiple environmental impact categories simultaneously to account for environmental impact trade-offs and to better understand the potential of proposed mitigation strategies.

This study hypothesizes that incorporating WC into PRG swards and reducing synthetic fertilizer application will mitigate the aforementioned environmental impacts of intensive pasture-based dairy systems. The specific objectives of this study were to (1) update and improve a pasture-based dairy systems LCA model, (2) evaluate the effect of the inclusion of WC on the environmental impact of intensive pasture-based dairy systems, (3) undertake an uncertainty analysis on key LCA variables, and (4) conduct a sensitivity analysis using a range of allocation methods.

MATERIALS AND METHODS

Description of Farm Systems

Data for this study were collated from a dairy farm systems experiment undertaken at Teagasc, Animal and Grassland Research and Innovation Centre, Moorepark, Fermoy, Co. Cork, Ireland (52°16′N, 8°25′W) from 2013 to 2016. Soil type was a free-draining acid brown earth of sandy loam to loam texture (pH 6.4). Average ambient temperature, annual rainfall, and average soil temperature were 10.9°C, 827 mm, and 12.5°C, respectively. The design of the study is described in detail by Egan et al. (2018) and Hennessy et al. (2018). Briefly, the goal of this farm systems experiment was to determine the herbage production and milk production from a PRG-only sward and from a PRG–WC sward receiving different levels of N fertilizer. The experimental production systems were a PRG–WC sward receiving 150 kg of N/ha per year (**CL150**), a PRG–WC sward receiving 250 kg of N/ha per year (**CL250**), and a PRG-only sward receiving 250 kg of N/ha per year (**GR250**).

In February, spring-calving Holstein-Friesian dairy cows were selected for each production system and blocked according to calving date, lactation number, milk solids yield, and 2-wk pre-experimental milk production. They were then randomly allocated to 1 of the 3 production systems described above. The GR250 sward was a 50:50 mix of the PRG cultivars Aston Energy and Tyrella (27.2 kg/ha). The CL150 and CL250 swards had the same PRG cultivars as GR250 with an additional 50:50 mix of WC cultivars Chieftain and

Crusader (5 kg/ha). Swards were reseeded when necessary (10% land area/yr). Criteria for reseeding included weed invasion, decline in PRG and WC content, and decline in herbage production. Between February and November 9, grazing rotations were completed in each production system. The chemical composition of herbage in each rotation is reported in Supplemental Table S1 (https://data.mendeley.com/datasets/38kf55bcn4/1). Nitrogen fertilizer was applied as urea (46% N) up until April for the first 2 rotations and as calcium ammonium nitrate (27% N) for the remaining rotations. Visual assessments were undertaken weekly for grass covers and input into PastureBase Ireland, an online grass management system (Hanrahan et al., 2017). Grass production exceeding requirements was removed and conserved as baled silage. The farm systems experiment reported an average sward WC content of 225 and 266 g/kg of DM for the CL250 and CL150 systems, respectively.

Concentrate supplementation [1.18 forage unit for lactation (UFL; 1 UFL = 7.11 MJ of net energy); CP = 151 g/kg of DM; NDF = 406 g/kg of DM; Supplemental Table S2, https://data.mendeley.com/datasets/38kf55bcn4/1] was offered to cows early in the grazing season (February–April), in late lactation (October–November), and during the main grazing season when herbage deficits occurred across all production systems. On average, 350 kg of concentrate DM was offered per cow per year. In the scenario where an herbage deficit occurred in an individual production system, conserved surplus grass silage from the same system was offered. The total quantity of herbage and milk produced by each production system is summarized in Table 1. Following the cessation of lactation at the end of November, dry cows were housed in slatted cubicle sheds. Over the winter period the majority (90%) of manure was stored in liquid manure storage below animal housing and the remainder in solid manure storage (straw bedding for cows before calving and for calves before being sold off-farm). Liquid and solid manure collected during housing was then returned to the farm area allocated to grass silage production (70% spring and 30% summer).

LCA

The principles and requirements specified by the International Organization for Standardization 14040

Table 1. Description of key farm characteristics for the 3 Moorepark[1] dairy production systems[2]

Item	GR250	CL250	CL150
Grassland production			
Land area (ha)	40	40	40
Cows calved (no.)	107	106	105
Replacement rate (%)	20	20	20
Stocking rate (LU[3]/ha)	2.44	2.42	2.40
Annual herbage production (kg of DM/ha)	11,787	12,153	11,803
Sward white clover content (g of DM/kg of DM)	0	225	266
First-cut silage, April to early June (% of total area)	40	40	40
Second-cut silage, early June to July (% of total area)	27	27	27
Milk production and live weight sales			
Milk production per cow (kg of milk/yr)	5,678	6,083	5,929
Milk protein (g/kg)	35.8	35.5	35.8
Milk fat (g/kg)	45.7	45.8	45.9
Total FPCM[4] (t)	649.6	689.8	667.3
Cull cow and surplus calf sales (kg of live weight)	16,363	15,714	15,578
Farm inputs			
Synthetic fertilizer (kg)			
N, CAN[5]	6,362	6,340	3,189
N, urea	3,402	3,420	2,960
P	374	375	373
K	396	399	392
Concentrate feed (kg of DM)	35,843	35,524	35,216
Electricity usage (kWh)	19,424	20,358	19,789
Diesel consumption (L)	1,532	1,532	1,532

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[2]GR250 = perennial ryegrass (PRG) sward receiving 250 kg of N fertilizer/ha; CL250 = mixed PRG and white clover (WC) sward receiving 250 kg of N fertilizer/ha; CL150 = mixed PRG and WC sward receiving 150 kg of N fertilizer/ha.

[3]Livestock unit; dairy cow = 1 LU, calf (0–12 mo) = 0.3 LU, yearling (12–24 mo) = 0.7 LU, and >24 mo = 1 LU.

[4]Fat- and protein-corrected milk.

[5]Calcium ammonium nitrate.

series (ISO, 2006a,b) were adhered to when applying the LCA methodology.

Goal and Scope

To meet growing global demand, it is projected that total meat and milk production is to increase by 1.3% and 1.1%/yr between 2006 and 2050, respectively (Alexandratos and Bruinsma, 2012). However, the agriculture, forestry, and other land use sector has been reported to contribute 24% of total anthropogenic GHG emissions, with livestock production systems being a major contributor (IPCC, 2014). Livestock production systems also release emissions in other sectors through the consumption of fossil fuels and industrial processes (e.g., fertilizer production; Crosson et al., 2011). Additionally, the intensification of livestock production systems has accelerated the loading of limiting nutrients (N and P) to water bodies, thus exacerbating the degradation of water quality and aquatic ecosystems through cultural eutrophication (McDowell and Hamilton, 2013). Along with reducing soil pH and fertility, the application of acidifying fertilizers and the atmospheric deposition of acidifying substances emitted from agricultural activities (e.g., animal manure and fertilizer application) can also lead to the degradation of surrounding ecosystems depending on climatic conditions and the sensitivity of surrounding ecosystems (Bouwman et al., 2002; Goulding, 2016). This is of particular concern for Ireland, where 34% of national GHG emissions and 98% of NH_3 emissions are sourced from the agricultural sector, which is dominated by ruminant production systems (Duffy et al., 2019a,b). Therefore, to meet the growing demand for animal products and reduce the negative environmental impacts associated with livestock production systems, it is pivotal that environmentally, economically, and socially sustainable management practices are identified and adopted. The goal of this LCA study therefore was to determine the effect of incorporating WC into intensive pasture-based dairy systems. To prevent environmental impact trade-offs and to determine the effect of incorporating WC on the environmental impacts discussed above, the impact categories selected were global warming, acidification, eutrophication, and nonrenewable energy use.

As milk processing data were unavailable, the system boundary was from raw material extraction to the point when the primary products (milk, meat) leave the farm (cradle to farm gate). Thus, the environmental impact of all on-farm processes and the impacts associated with the production of farm inputs (e.g., fertilizer and concentrate production) were quantified. Environmental impacts embodied in capital such as on-farm infrastructure and machinery were excluded as they did not differ between the production systems and due to a lack of environmental data for some capital goods. The production of medicine was excluded due to its small effect on the environmental impact of dairy production systems (Saunders and Barber, 2007). The impacts of a dairy system were scaled to a metric tonne of fat- and protein-corrected milk (**FPCM**), which is referred to as a functional unit (**FU**) in LCA. To ensure fair comparison between farms and treatments with different feeding regimens and breeds, FPCM adjusts the quantity of milk produced to that required to provide equal energy (3.14 MJ/kg) as milk with standardized fat and protein contents (4.0% and 3.3%, respectively; Rice et al., 2019). The equations provided by the International Dairy Federation (IDF, 2015) were applied.

Although the use of mass-based FU is ubiquitous in agricultural LCA studies, Salou et al. (2017) has highlighted that the sole use of a mass-based FU results in an unbalanced view on the environmental impact of a production system or management practice and could poorly advise decision makers when identifying mitigation strategies. Therefore, to prevent the potential paradoxical consequences of solely using mass-based FU, this LCA also selected on-farm land area (hectares) as an FU. By all systems occupying the same land area, a reduction in environmental impact per hectare indicates a reduction in total environmental impact. Area-based FU also contribute to better understanding the effect of management practices on the environmental impact of local environmental categories such as acidification and eutrophication (de Vries et al., 2015). As the economic relationship between coproducts reflects the socioeconomic demands of dairy systems, environmental impacts were allocated between co-products (milk, meat) in accordance with the economic method recommended by UK LCA guidelines (BSI, 2011; Ardente and Cellura, 2012).

Life Cycle Inventory

Inventory analysis was carried out using the Moorepark Dairy Systems Model (**MDSM**) developed by Shalloo et al. (2004) and a dairy environmental model (O'Brien et al., 2011). For the analysis, the models were amalgamated to create a hybrid model that allows for the interaction of various components from both models to occur. Milk and forage production data for each production system were averaged over the 4 yr of the trial and used to populate the MDSM LCA model. The temporal coverage of the MDSM LCA model was 1 yr. Based on forage and feedstuffs digestibility and animal output (live weight gain and milk production), DMI was simulated using the French net energy system adapted to Irish conditions (O'Mara et al., 1997).

On-Farm Impacts

Foreground data collected during the Moorepark PRG and WC trial were predominantly used to quantify the direct use of resources. Resources that were not measured, such as diesel consumption by contracted machinery, were estimated using secondary sources (Nemecek and Kägi, 2007).

Carbon Dioxide. On-farm CO_2 emissions were limited to the combustion of fossil fuels by machinery, lime application, and the hydrolysis of urea-based fertilizers following application (Supplemental Table S3). In accordance with the European Commission (2018) Product Environmental Footprint Category Rules, soil C sequestration was excluded from LCA results. However, several studies have reported that managed grasslands act as a C sink (Jaksic et al., 2006; Byrne et al., 2007, 2018). Net C sequestration rate in grasslands have been reported to range from −2 to 1 t of C/ha per year, with Irish pasture sequestering on average 0.5 t of C/ha per year. The wide range in C sequestration is dependent on soil type and management practice, with long-term research trials indicating that optimum fertilization, soil pH, and manure amendment have potential to increase soil organic C content (Byrne et al., 2018). Intensive Irish grasslands, similar to the systems simulated in the current study, have been reviewed by Soussana et al. (2010) to sequester on average 476 kg of C/ha per year. This study therefore reported GWP without soil C sequestration and with soil C sequestration using the former C sequestration rate of 476 kg of C/ha per year.

Methane. Enteric fermentation methane (CH_4) emissions from grazing cattle were estimated using the fixed CH_4 conversion factor of 6.3% of gross energy intake (medium-producing cow) as it is currently the best available and recommended method for calculating enteric CH_4 at pasture (Duffy et al., 2019a; IPCC, 2019). A country-specific equation developed by Yan et al. (2000) was used to calculate CH_4 from enteric fermentation when cattle are housed during winter and offered a grass silage-based diet (Supplemental Table S3). Methane from managed manure was quantified using the tier 2 emission algorithm and factors provided by IPCC (2019) guidelines. The same guidelines and tiers were used to quantify CH_4 emissions from unmanaged manure deposited during grazing.

N Emissions. Nitrogen emissions were caused by the application of synthetic fertilizers and by the storage and spreading or deposition of manure. Country-specific equations were used to calculate N excretion by livestock within the environmental model (Table 2). Nitrogen excreted in manure by lactating dairy cattle consuming fresh grass, grass silage, and concentrate was determined using equations taken from Yan et al. (2006) and Hynes et al. (2016). Recent equations developed by Stergiadis et al. (2015) were used to estimate the N excreted in manure and urine by dry dairy cattle. Fecal N was calculated as the difference between total N excretion and urine N excretion. In a similar manner, N excretion by livestock under 12 mo and between 12 and 24 mo was calculated using equations created by Jiao et al. (2014). For manure excreted in solid manure housing, the N content of the bedding material (cereal straw; 5.6 g of N/kg of DM) was included to determine total N in solid manure storage.

Nitrous oxide emissions from the deposition of excreta onto pastures vary greatly depending on climatic factors and soil conditions. To address this inherent variation, country-specific N_2O emission factors (**EF**) for deposition of excreta onto pasture developed by Krol et al. (2016) were added to the environmental model (Supplemental Table S3). The EF accounted for the effect of season and manure type. Nitrous oxide emis-

Table 2. Nitrogen excretion (g/d) equations and their references for lactating dairy cattle (milking cow), dry cattle, and cattle 0–24 mo on a predominantly pasture-based diet

Animal category	N form	Equation[1]	Reference
Lactating cow	Manure N		Yan et al., 2006
	Grass silage	$0.711\ \text{NI} + 7$	
	Fresh grass	$0.711\ \text{NI} - 21$	
	Concentrate		
	<160 g of CP/kg of DM	$0.689\ \text{NI} + 11$	
	160–200 g of CP/kg of DM	$0.689\ \text{NI} + 14$	
	>200 g of CP/kg of DM	$0.689\ \text{NI} + 25$	
	MUN	$-3.1 + 0.015\ \text{NI}$	Hynes et al., 2016
	Urine N	$139.1 + 0.0142\ \text{MUN}$	
Dry cattle	Manure N	$37.9 + 0.64\ \text{NI}$	Stergiadis et al., 2015
	Urine N	$7.907 + 0.613\ \text{NI}$	
0–24 mo	Manure N	$0.585\ \text{NI} + 0.489\ \text{BW}^{0.75} - 25.1$	Jiao et al., 2014
	Urine N	$0.366\ \text{NI} + 0.411\ \text{BW}^{0.75} - 31.6$	

[1]NI = N intake (g/d); MUN = milk urea N (mg/d); BW = live BW.

sions from the storage and application of solid manure were calculated using IPCC (2019) tier 1 default EF. Nitrous oxide emissions from liquid manure application were calculated using EF reported by Bourdin et al. (2014). Similar to the deposition of excreta, N_2O emissions from synthetic fertilizer application vary greatly depending on the form of N applied and climatic conditions. Country-specific EF developed by Harty et al. (2016) were subsequently added to the environmental model (Supplemental Table S3).

Ammonia volatilization, an indirect source of N_2O, occurs during grazing, manure storage, manure spreading, and synthetic fertilizer application. Following the redeposition of NH_3 onto soil or a water body, it contributes to acidification and eutrophication by releasing H^+ ions and by the loading of nutrients, respectively. Urea-based fertilizers are more susceptible to NH_3 volatilization than are nitrate (NO_3^-)-based fertilizers (Forrestal et al., 2016). The EF recommended in the Ireland Informative Inventory Report (Duffy et al., 2019b) for urea and other fertilizers were used to quantify NH_3 loss from this farm input (Supplemental Table S3). For manure management, NH_3 emissions were estimated using the tier 2 mass flow approach recommended by the same agency. To fully account for all N losses occurring during manure management, N_2 and NO_2 emissions were quantified using EF provided in the EEA (2019) guidebook. The N mass flow approach was also used for NH_3 emissions from the deposition of manure onto pasture.

Nutrient Leaching and Run-Off. Nutrient loading and climatic or seasonal conditions have a strong effect on NO_3^- leaching (Di and Cameron, 2002). To account for the effect of the latter on NO_3^- leaching from N inputs, seasonal EF developed by Dennis (2009) were incorporated into the model (Supplemental Table S3). The application of synthetic N fertilizer to mixed PRG and WC swards increases the amount of available inorganic N in the soil and the quantity susceptible to leaching. It is widely reported, however, that the increase in fertilizer application rate reduces WC content and N fixation, particularly with high soil inorganic N content (Lüscher et al., 2014; Enriquez-Hidalgo et al., 2016). To determine the amount of N fixed by WC and the quantity of forage N deriving from atmospheric fixation, equations developed by Enriquez-Hidalgo et al. (2016) were added to the model:

$$\text{N fix} = 8 \times \text{WC} - 77, \quad [1]$$

$$\text{NDFA} = 72 - 0.2 \times \text{N}, \quad [2]$$

where N fix = kilograms of N fixed per hectare per year; WC = WC content of sward (%); NDFA = N derived from the atmosphere (%); and N = kilograms of N applied per hectare.

The application of P and K fertilizer was based on soil P and K index requirement, which was index 3 for both nutrients. The quantity of P loss from P-saturated soils increases as the P surplus level rises. The potential loss of P through leaching and run-off was calculated using a methodology developed by Nemecek and Kägi (2007):

$$P_{gw} = 0.06 \times (1 + 0.2/80 \times P_2O_{5\ sl}), \quad [3]$$

$$P_{ro} = 0.25 \times (1 + 0.2/80 \times P_2O_{5\ min} + 0.7/80 \times P_2O_{5\ sl} + 0.4/80 \times P_2O_{5\ man}), \quad [4]$$

where P_{gw} = P leached to groundwater (kg of P/ha); P_{ro} = P run-off to surface water (kg of P/ha); $P_2O_{5\ sl}$ = quantity of P_2O_5 in liquid manure or slurry (kg/ha); $P_2O_{5\ min}$ = quantity of P_2O_5 in mineral fertilizer (kg/ha); and $P_2O_{5\ man}$ = quantity of P_2O_5 in solid manure (kg/ha).

Off-Farm Impacts

Inputs imported to the Moorepark PRG and PRG–WC dairy systems were recorded and used to calculate off-farm resource use and pollutants. Emissions of GHG and energy (MJ) embodied in electricity and synthetic fertilizers were sourced from national reports and international literature (Brentrup et al., 2016; Duffy et al., 2019a; SEAI, 2019). The LCA database Ecoinvent (2010) was used to quantify remaining GHG emissions and pollutants (Supplemental Table S4).

Life Cycle Impact Assessment

The inventory of emissions and resources was translated into midpoint environmental impacts using the characterization factors in Table 3. The IPCC (2013) characterization factors for a 100-yr time horizon was used to calculate GWP. The accumulated exceedance approach was adopted to determine acidification potential (**AP**) as it accounts for the dispersion of an emission in the environment as well as the sensitivity of ecosystems (Posch et al., 2008). The accumulated exceedance approach expresses AP in moles of H^+ released by acidifying substances (denoted as mol_c of H^+ eq; Posch et al., 2008). Downstream estuarine and coastal waters are physically, chemically, and biologically different from upstream water, where P and N have been identified as being the limiting nutrients that cause eutrophication in freshwater and marine waterbodies (McDowell and Hamilton, 2013). The ReCiPe

Table 3. Contributing substances, units, and characterization factors for selected environmental impact categories

Impact category	Unit	Contributing substance	Characterization factor	Reference
Global warming potential (GWP)	kg of CO_2 equivalent	CO_2	1	IPCC, 2013
		Biogenic CH_4	27.75	Muñoz and Schmidt, 2016
		Fossil CH_4	30.5	
		N_2O	265	
Freshwater eutrophication potential (FEP)	kg of P equivalent	P	0.7	Huijbregts et al., 2016
		PO_4	0.23	
Marine eutrophication potential (MEP)	kg of N equivalent	NH_3	0.82	Huijbregts et al., 2016
		N oxides	0.30	
		NO_3^-	0.23	
Acidification potential (AP)	H^+ equivalent	SO_2	1.1	Posch et al., 2008
		NH_3	1.2	
		N oxides	0.60	
Nonrenewable energy use (NRE)	MJ	MJ	1	—

2016 methodology disaggregates eutrophication potential based on these limiting nutrients into marine eutrophication potential [**MEP**; g of N equivalent (**eq**)] and freshwater eutrophication potential (**FEP**; g of P eq; Huijbregts et al., 2016). The subcategories should not be assessed in isolation as the response to nutrient loading in freshwater ecosystems may differ from the response in downstream marine ecosystems (Paerl, 2009). Nonrenewable energy (**NRE**) use refers to the consumption of finite fossil fuels for on-farm activities as well as fossil fuel consumption in production of farm inputs. Nonrenewable energy use is expressed in megajoules.

Uncertainty Analysis

Greenhouse gas emissions and pollutants from agricultural systems are inherently uncertain. They are influenced by spatial and temporal factors. Stochastic simulation was used to partly account for the inherent uncertainty in key LCA model parameters influencing the environmental impact of the 3 Moorepark production systems. Each production system was simulated deterministically to identify key parameters affecting the environmental impact categories investigated. Enteric fermentation, synthetic fertilizer application, manure management, and manure excreted during grazing were identified as hotspots for environmental impacts. Ten associated parameters were subsequently identified (Table 4). The probability distributions of the selected parameters and their minimum, maximum, and most likely values were taken from the studies from which they were obtained. Each parameter was assigned a probability distribution, and a series of Monte Carlo simulations were conducted (10,000 iterations per systems) using Palisade @Risk 7.5 (Palisade Corporation, 2013). All stochastic parameters were simulated simultaneously.

Sensitivity Analysis

Several approaches and criteria can be used to allocate an environmental impact between the co-products of dairy systems. The British Standards Institution (BSI, 2011) and ISO 14044 (ISO, 2006b) recommend evaluating the impact different allocation methods have on the outcomes of an LCA model. To comply with these guidelines and improve our understanding of the effect of allocation method on LCA results of dairy systems, a sensitivity analysis was conducted. In this analysis, the environmental impact of the GR250, CL250, and CL150 production systems was distributed between co-products using mass-based, protein-based, and physical causality allocation methods. The physical causality approach is reflective of energy intake requirements for milk and meat production and was applied according to the methods and equations described in the IDF LCA guidelines (IDF, 2015).

RESULTS

The life cycle impact assessment results for the PRG and mixed PRG–WC dairy systems are reported in Table 5. Regarding GWP, the production system with the lowest milk yield, GR250, had the greatest GWP per tonne of FPCM. On an area basis (per hectare), the highest yielding dairy system in terms of milk output, CL250, had the greatest GWP. For both FU, the CL150 system had the lowest GWP. The total GWP of the CL150 system was 65 kg of CO_2 eq (7.9%) lower than that of the GR250 system per tonne of FPCM and 1,010 kg of CO_2 eq (7.4%) lower than that of the CL250 system per hectare. The dairy system AP and NRE per tonne of FPCM and area followed the same order as their GWP ranking. The CL150 system AP was 0.45 mol_c of H^+ eq/t of FPCM and 7.9 mol_c of H^+ eq/ha lower than that of the GR250 and CL250

Table 4. Parameters and distributions used in the stochastic analysis of the environmental impacts of the Moorepark[1] perennial ryegrass and mixed perennial ryegrass–white clover dairy production systems

Source	Unit	Distribution	SD	Minimum	Most likely	Maximum	Reference
Enteric fermentation	Gross energy intake, %	Normal	0.612		6.3		IPCC, 2019
Fertilizer, CAN[2]	kg of N_2O-N/kg of N applied	Pert[3]		0.0081	0.0149	0.0381	Harty et al., 2016
Fertilizer, urea	kg of N_2O-N/kg of N applied	Pert		0.0010	0.0025	0.0049	Harty et al., 2016
Fertilizer, CAN	kg of NH_3-N/kg of N applied	Pert		0.0000	0.0080	0.0200	Duffy et al., 2019b
Fertilizer, urea	kg of NH_3-N/kg of N applied	Pert		0.0300	0.1550	0.4300	Duffy et al., 2019b
Grazing, dung	kg of N_2O-N/kg of N applied	Pert		0.0000	0.0031	0.0148	Krol et al., 2016
Grazing, urine	kg of N_2O-N/kg of N applied	Pert		0.0030	0.0118	0.0481	Krol et al., 2016
Liquid manure housing	kg of NH_3-N/kg of TAN[4]	Normal	14.41		27.7		Misselbrook et al., 2016
Liquid manure application	kg of N_2O-N/kg of N applied	Pert		0.0030	0.0067	0.0225	Bourdin et al., 2014
N applied susceptible to leaching	kg of NO_3-N/kg of N applied	Pert		0.0500	0.1000	0.1500	Duffy et al., 2019a

[1]Teagasc, Animal and Grassland Research and Innovation Centre, Moorepark, Fermoy, Co. Cork, Ireland.
[2]Calcium ammonium nitrate.
[3]Program evaluation and review technique.
[4]TAN = total ammonical nitrogen.

systems, respectively. Regarding NRE, the CL150 system used 3,840 MJ/ha less than the CL250 system and 250 MJ/t of FPCM less than the GR250 system. In contrast to GWP, AP, and NRE, the CL250 system had the greatest MEP for both FU. The MEP of the CL 150 system was 113 g of N eq/t of FPCM and 2,422 g of N eq/ha less than that of the CL250 system. With regard to FEP, there was no difference between the 3 dairy systems when expressed per tonne of FPCM and hectare.

GWP

Methane was the main contributor to GHG emissions across all dairy systems. The gas accounted for 66.8% to 71.2% of total GWP for dairy systems (Table 6). Enteric fermentation from cattle on-farm was the dominant source of CH_4 emission from dairy systems, emitting 441 to 457 kg of CO_2 eq/t of FPCM, representing on average 83% of gas emissions. Off-farm rearing of replacement heifers contributed to the majority of the remaining CH_4 emissions, with minor emissions sourced from manure management (Table 6). Nitrous oxide was the second largest GHG, representing 19.3% to 22.3% of total GWP for dairy systems (Table 6). The CL150 system had the lowest N_2O, emitting 147 kg of CO_2 eq/t of FPCM, compared with CL250 and GR250, emitting 177 and 180 kg of CO_2 eq/t of FPCM, respectively. Deposition of excreta during grazing was the largest source of N_2O for the CL150 system, followed by synthetic N fertilizer application (Table 6). In contrast to the CL150 system, synthetic N fertilizer was the main source of N_2O emissions from PRG and mixed PRG–WC systems that spread 100 kg more N/ha. On average, 20.2% of N_2O emissions were emitted indirectly through NH_3 volatilization and NO_3 leaching. Carbon dioxide represented the lowest share of total GWP for all dairy systems, with the GR250, CL250, and CL150 systems emitting 92.4, 87.6, and 73.7 kg of CO_2 eq/t of FPCM, respectively. The majority of CO_2 emissions were produced off-farm, where synthetic N fertilizer production was the dominant CO_2 emission source (23–39 kg of CO_2 eq/t of FPCM; Table 6). The productions of electricity and concentrate feed were other key CO_2 emission sources. The rearing of replacement heifers was a key source of GHG emissions, contributing 108.7, 97.2, and 99.5 kg of CO_2 eq/kg of FPCM to the GWP of the GR250, CL250, and CL150 systems, respectively.

AP

Ammonia was the dominant pollutant for AP, releasing 5.09, 4.94, and 4.83 mol_c of H^+ eq/t of FPCM for

Table 5. Environment impacts[1] expressed per farm area (hectare) and per metric tonne of fat- and protein-corrected milk (FPCM) for Holstein-Friesian cows within the Moorepark[2] dairy production systems

System[3]	Unit	GWP		GWP C seq		NRE		AP		FEP		MEP	
		On-farm	Total	On-farm	Total	On-farm	Total	On-farm	Total	On-farm	Total	On-farm	Total
GR250	Impact per hectare	10,498	13,417	8,917	11,836	1,222	21,326	75.2	100.9	274.0	523.6	105.3	126.7
CL250		10,849	13,696	9,254	12,102	1,233	21,563	78.3	104.0	276.3	526.8	115.8	137.2
CL150		10,156	12,686	8,565	11,095	1,231	17,723	72.4	96.1	275.8	523.8	96.9	117.2
GR250	Impact per tonne of FPCM	646	826	549	729	75	1,313	4.6	6.2	16.9	32.2	6.5	7.8
CL250		629	794	537	702	71	1,250	4.5	6.0	16.0	30.5	6.7	8.0
CL150		609	761	514	665	74	1,063	4.3	5.8	16.5	31.4	5.8	7.0

[1]GWP = global warming potential without soil carbon sequestration (kg of CO_2 equivalent); GWP C seq = global warming potential with soil carbon sequestration (kg of CO_2 equivalent); NRE = nonrenewable energy depletion (MJ); AP = acidification potential (mol$_c$ of H^+ equivalent); FEP = freshwater eutrophication potential (g of P equivalent); MEP = marine eutrophication potential (kg of N equivalent).

[2]Teagasc, Animal and Grassland Research and Innovation Centre, Moorepark, Fermoy, Co. Cork, Ireland.

[3]GR250 = perennial ryegrass (PRG) sward receiving 250 kg of N fertilizer/ha; CL250 = mixed PRG and white clover (WC) sward receiving 250 kg of N fertilizer/ha; CL150 = mixed PRG and WC sward receiving 150 kg of N fertilizer/ha.

the GR250, CL250, and CL150 systems, respectively (Table 7). On-farm NH_3 emissions from manure management (housing, storage, and application) were the main source of AP, releasing on average 2.1 mol$_c$ of H^+ eq/t of FPCM. Fertilizer application and the deposition of excreta at pasture were also key sources of on-farm NH_3 emissions, releasing on average 1.1 and 0.6 mol$_c$ of H^+ eq/t of FPCM, respectively. Rearing replacement heifers was the main off-farm NH_3 emission source, releasing on average 1.0 mol$_c$ of H^+ eq/t of FPCM. Minor quantities of NH_3 were emitted off-farm during the production of concentrate feed and fertilizers. The N oxides pollutants contributed the majority of the remaining AP, releasing 0.99, 0.97, and 0.82 mol$_c$ of H^+ eq/t of FPCM for the GR250, CL250, and CL150 systems, respectively. The gas was predominantly emitted after the application of organic and inorganic N fertilizer.

Eutrophication Potential

Nitrate and NH_3 pollutants accounted for the majority (94%) of MEP for dairy systems. The NH_3 sources that contributed to MEP were the same as AP. Synthetic fertilizer application and the deposition of excreta at pasture were the dominant sources of NO_3^- loss for GR250 and CL250, contributing 1.66 and 1.54 kg of N eq/t of FPCM, respectively. Due to reduced N fertilizer application rate, the deposition of excreta onto pasture was the largest contributor to NO_3^- leaching (1.52 kg of N eq/t of FPCM) for the CL150 system. Manure application contributed 0.24 kg of N eq/t of FPCM. The leaching of unutilized biologically fixed N presented an additional NO_3^- source, emitting 0.35 and 0.4 kg of N eq/t of FPCM for the CL250 and CL150 systems, respectively. Nitrate loss occurred off-farm during concentrate production and replacement heifer rearing. Similar to NH_3, the majority of NO_3^- loss occurred on-farm (83%). Nitrogen oxides were a minor contribution to MEP. Regarding FEP, the contribution of on- and off-farm sources to the impact category was similar (Table 7). The main on-farm FEP sources were deposition of excreta onto pastures and manure application. Concentrate feed and fertilizer applications were minor sources of FEP pollutants. Off-farm rearing of replacement heifers was the most important source of FEP (Table 7).

NRE Depletion

More than 90% of NRE used in the 3 productions systems was consumed off-farm. Fertilizer production was the dominant consumer of NRE for dairy systems (Table 7). The NRE used for fertilizer manufacture was similar in the GR250 and CL250 systems but lower in the CL150 system, which consumed 632, 600, and 400 MJ of NRE/t of FPCM, respectively. The other key consumers of off-farm NRE were replacement heifer rearing and concentrate and electricity production. The burning of fossil fuels for machinery operation was the sole on-farm activity that used NRE. The amount of on-farm NRE used to run machinery was the same for the 3 systems. Similarly, NRE used for concentrate production did not differ for the 3 systems as there was no difference in feeding rates. Total NRE used for electricity generation was influenced by milk production. The lower yielding GR250 system consumed 4.6% less NRE for electricity production than the CL250 system.

Uncertainty and Sensitivity Analysis

The parameters identified from the deterministic simulations affected CH_4, N_2O, NH_3, and NO_3 leaching. These emissions contributed to the uncertainty in GWP, AP, and MEP. The cumulative density functions in Figure 1 show the probability of possible GWP, AP, and MEP per hectare and tonne of FPCM for GR250, CL250, and CL150. Production systems with cumulative density function curves to the left have lower GWP, AP, and MEP. The CL150 production system had lower GWP per hectare (13,008 kg of CO_2 eq; 95% CI: 11,778–14,290) and tonne of FPCM (780 kg of CO_2 eq; 95% CI: 706–857) than the GR250 and CL250 production systems (Figure 1a and d). Uncertainty associated with CH_4 from enteric fermentation, N_2O from urine deposition during grazing, and N_2O from calcium ammonium nitrate fertilizer application contributed 53.4 to 62.4%, 22.6 to 27.0%, and 6.2 to 19.9% to the simulated variance for GWP, respectively (Supplemental Figure S1). Similarly, CL150 had lower MEP per hectare and tonne of FPCM than GR250 and CL250 (Figure 1c and 1f). Uncertainties associated with the proportion of N applied susceptible to NO_3 leaching, NH_3 from urea fertilizer application, and NH_3 from housing were responsible for 65.9 to 69.9%, 25.9 to 27.6%, and 3.7 to 6.3% of the simulated variance for MEP, respectively (Supplemental Figure S1). The CL150 system also had the lowest AP per hectare (98.7 mol_c of H^+ eq; 95% CI: 84.2–115.1 mol_c of H^+ eq) and tonne of FPCM (5.9 mol_c of H^+ eq; 95% CI: 5.1–6.9 mol_c of H^+ eq). The GR250 and CL250 systems had similar AP per hectare and per tonne of FPCM (Figure 1b and 1e). Uncertainties associated with NH_3 emissions from urea fertilizer application and slurry housing contributed 76.2 to 83.7% and 15.1 to 23.1% to the simulated variance for AP, respectively (Supplemental Figure S1).

The sensitivity of the dairy LCA model results to different methods for allocating environmental impacts between milk production (FPCM) and beef production (cull cows and surplus calves) is reported in Figure 2. The analysis showed that the order of dairy system environmental impacts was consistent across each alloca-

Table 6. Effect of white clover and N fertilizer level[1] on carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) emission sources (expressed as CO_2 equivalents/t of fat- and protein-corrected milk)

	GR250			CL250			CL150		
Source	CO_2	CH_4	N_2O	CO_2	CH_4	N_2O	CO_2	CH_4	N_2O
On-farm									
Enteric fermentation	—	456.9	—	—	441.4	—	—	449.3	—
Manure housing and storage	—	20.4	9.4	—	19.7	9.3	—	21.5	9.7
Manure during grazing	—	1.3	40.4	—	1.3	40.9	—	1.3	39.7
Manure spreading	—	—	9.8	—	—	9.7	—	—	10.1
Fertilizer application	7.5	—	59.9	7.2	—	56.7	6.4	—	31.2
Crop residues	—	—	1.6	—	—	1.5	—	—	1.5
Fuel use emissions	10.4	0.1	0.9	9.9	0.1	0.9	10.5	0.1	1.0
NH_3 emissions[2]	—	—	16.8	—	—	16.3	—	—	15.7
NO_3 leaching[3]	—	—	16.7	—	—	19.5	—	—	16.2
Off-farm	—	—	—	—	—	—	—	—	—
Fertilizer production	39.0	0.7	3.0	37.0	0.7	2.8	22.6	0.4	1.6
Fuel production	1.6	0.2	0.1	1.5	0.2	0.0	1.6	0.2	0.1
Electricity production	11.8	—	0.1	11.8	—	0.1	11.8	—	0.1
Concentrate production	8.6	0.0	0.0	8.1	0.0	0.0	8.3	0.0	0.0
Replacement heifers	13.3	73.8	21.7	11.9	66.0	19.4	12.2	67.5	19.8
Other	0.2	—	—	0.2	—	—	0.2	—	—
Total	92.4	553.4	180.4	87.6	529.3	177.3	73.7	540.2	146.9
Percentage	11.2	67.0	21.8	11.0	66.6	22.3	9.7	71.0	19.3

[1]GR250 = perennial ryegrass (PRG) sward receiving 250 kg of N fertilizer/ha; CL250 = mixed PRG and white clover (WC) sward receiving 250 kg of N fertilizer/ha; CL150 = mixed PRG and WC sward receiving 150 kg of N fertilizer/ha.

[2]NH_3 = ammonia.

[3]NO_3 = nitrate.

Table 7. Contribution of emissions, pollutants, and energy sources to environmental impacts[1] for 3 dairy production systems[2] (expressed as CO_2 equivalents/t of fat- and protein-corrected milk unless otherwise noted)

	NRE (MJ)			AP (mol$_c$ of H^+ equivalent)			FEP (g of P equivalent)			MEP (kg of N equivalent)		
Emission or energy source	GR250	CL250	CL150	GR250	CL250	CL150	GR250	CL250	CL150	GR250	CL250	CL150
On-farm				2.17	2.14	2.24						
Fertilizer application	—	—	—	1.50	1.43	1.17	1.6	1.5	1.6	2.7	2.5	1.6
Manure housing and storage	—	—	—	1.45	1.43	1.51	—	—	—	1.0	1.0	1.0
Manure application	—	—	—	0.72	0.71	0.74	7.4	7.2	7.6	0.7	0.7	0.7
Grazing	—	—	—	0.92	0.93	0.90	7.8	7.3	7.4	2.1	2.5	2.4
Fuel use	75	71	74	0.04	0.04	0.04	—	—	—	0.0	0.0	0.0
Off-farm												
Embodied in fertilizer	632	600	400	0.25	0.24	0.14	4.9	4.7	4.8	0.1	0.1	0.1
Embodied in concentrates	251	237	242	0.19	0.18	0.18	1.9	1.8	1.9	0.2	0.2	0.2
Embodied in fuel production	18	17	17	0.01	0.01	0.01	0.0	0.0	0.0	0.0	0.0	0.0
Embodied in electricity	144	144	144	0.04	0.04	0.04	0.0	0.0	0.0	0.0	0.0	0.0
Replacement heifers	193	182	186	1.08	1.02	1.04	8.5	8.0	8.2	1.0	0.9	0.9
Total	1,313	1,250	1,063	6.21	6.03	5.76	32.2	30.5	31.4	7.8	8.0	7.0
Pollutants (%)												
NH_3	—	—	—	81.9	81.8	83.9	—	—	—	44.6	42.4	47.0
N oxides	—	—	—	16.0	16.1	14.3	—	—	—	6.4	6.1	5.9
NO_3	—	—	—	—	—	—	—	—	—	49.1	51.5	47.1
SO_2	—	—	—	2.1	2.1	1.8	—	—	—	—	—	—
P	—	—	—	—	—	—	75.0	74.9	75.0	—	—	—
PO_4	—	—	—	—	—	—	25.0	25.1	25.0	—	—	—

[1]NRE = nonrenewable energy depletion; AP = acidification potential; FEP = freshwater eutrophication potential; MEP = marine eutrophication potential.

[2]GR250 = perennial ryegrass (PRG) sward receiving 250 kg of N fertilizer/ha; CL250 = mixed PRG and white clover (WC) sward receiving 250 kg of N fertilizer/ha; CL150 = mixed PRG and WC sward receiving 150 kg of N fertilizer/ha.

tion method. The baseline economic method allocated 90.4, 91.2, and 91.1% of the environmental burden created by the GR250, CL250, and CL150 systems to milk production, respectively. Physical causality reduced the environmental burden allocated to milk production to an average of 85.4% (Figure 2). In contrast, using mass-based allocation and, to a lesser extent, protein-based allocation increased the environmental burden allocated to milk production to an average of 97.6 and 93.5%, respectively. Mass-based and protein-based allocation methods therefore increased the environmental impact per category compared with the baseline economic allocation method. The results of the sensitivity analysis are reported in Supplemental Table S5.

DISCUSSION

To feed the world's growing population in a manner that is both environmentally and economically sustainable, greater emphasis needs to be placed on improving the efficiencies of current production systems and reducing their total environmental footprint via new technologies and management practices (Crosson et al., 2011). The incorporation of WC into PRG swards has been identified as an environmentally and economically sustainable management practice. The current study evaluated the application of this practice in intensive pasture-based dairy systems using an updated Irish dairy LCA model with a particular focus on N dynamics and emissions. The new LCA model applications indicated that reducing synthetic N fertilizer demand through the incorporation of WC into the sward of N-intensive spring-calving dairy system (GR250 vs. CL150) has potential to reduce the associated environmental burden. This was attributed to better animal performance on the PRG–WC swards compared with PRG-only swards and the reduction in synthetic N fertilizer consumption.

When comparing LCA results from different studies, several components need attention, as it has been well reported that variation in the methodology and assumptions can compromise the efficacy of comparing studies (Crosson et al., 2011). The current study took steps to ensure that the results would be assessable for comparisons. First, environmental impacts were reported using FU based on product (tonne of FPCM) and

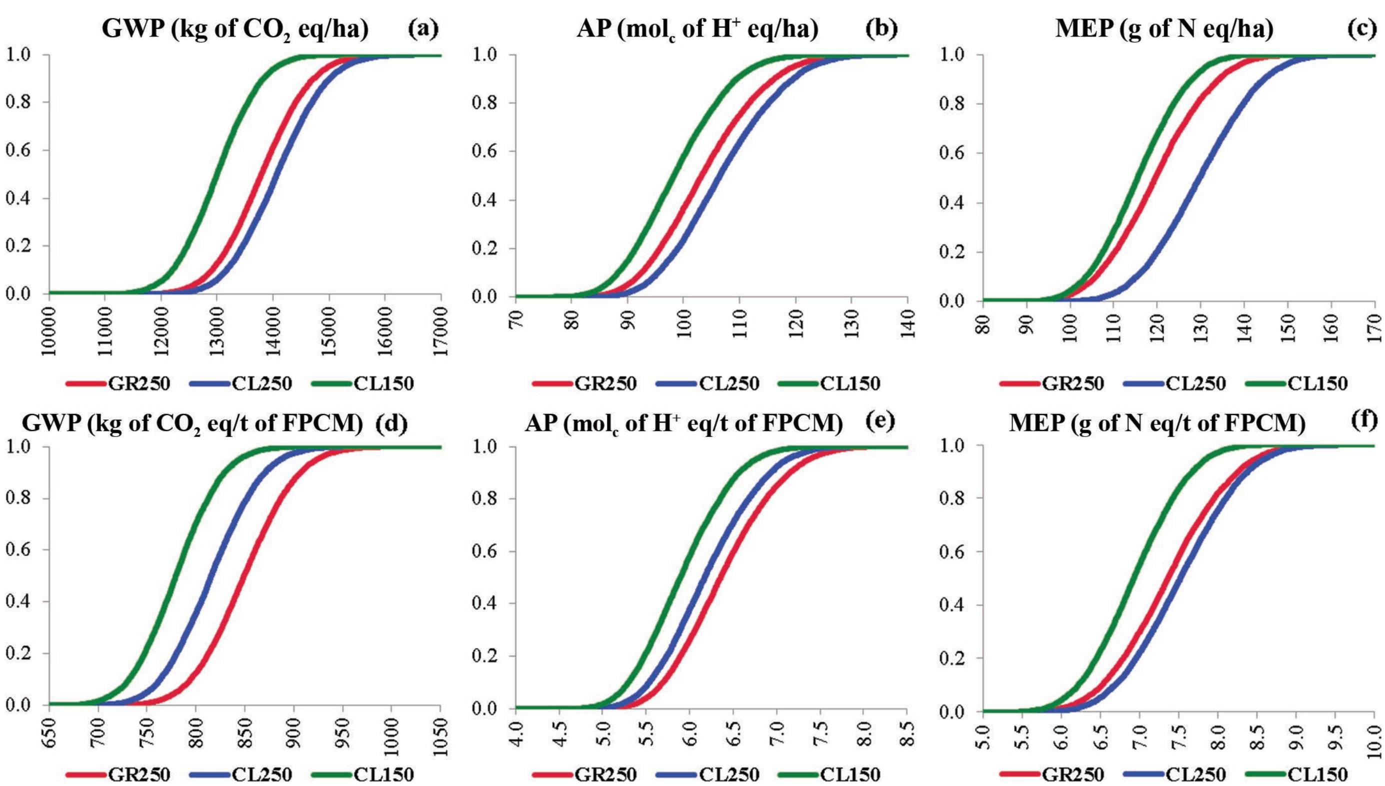
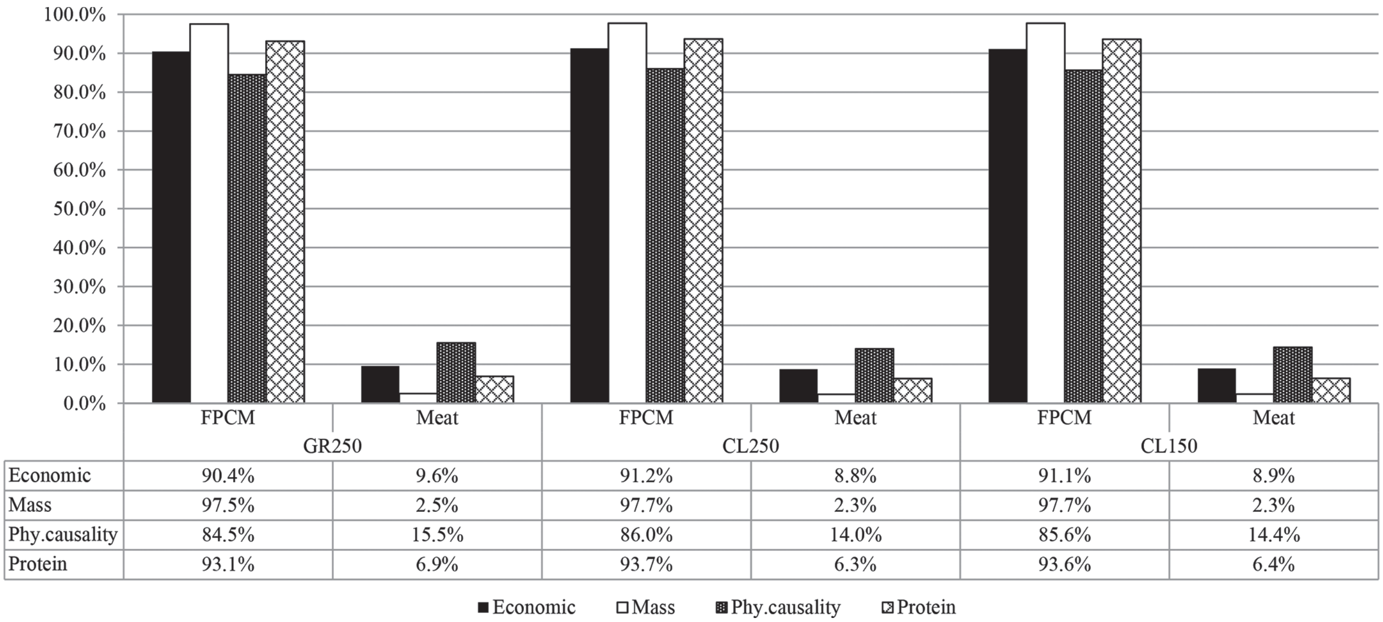


Figure 1. Cumulative density function of global warming potential (GWP), acidification potential (AP), and marine eutrophication potential (MEP) expressed per hectare and per metric tonne of fat- and protein-corrected milk (FPCM) for a perennial ryegrass dairy system receiving 250 kg of N fertilizer/ha per year (GR250) and mixed perennial ryegrass and white clover dairy systems receiving 150 (CL150) or 250 (CL250) kg of N fertilizer/ha per year. The dairy life cycle assessment model was run 10,000 times with the Monte Carlo method to calculate cumulative density functions.



	GR250 FPCM	GR250 Meat	CL250 FPCM	CL250 Meat	CL150 FPCM	CL150 Meat
Economic	90.4%	9.6%	91.2%	8.8%	91.1%	8.9%
Mass	97.5%	2.5%	97.7%	2.3%	97.7%	2.3%
Phy.causality	84.5%	15.5%	86.0%	14.0%	85.6%	14.4%
Protein	93.1%	6.9%	93.7%	6.3%	93.6%	6.4%

Figure 2. Allocation factors for fat- and protein-corrected milk (FPCM), meat produced by cull cows, and surplus calves based on economic return (Economic), mass of co-products sold (Mass), feed energy requirement to produce milk and meat (Phy.causality; IDF, 2015), and protein sold (Protein) for a perennial ryegrass dairy system receiving 250 kg of N fertilizer/ha per year (GR250) and mixed perennial ryegrass and white clover dairy systems receiving 150 (CL150) or 250 (CL250) kg of N fertilizer/ha per year.

area (hectare). Second, a sensitivity analysis was conducted to determine the effect of allocation method on the environmental burdens of dairy co-products. Third, an uncertainty analysis was conducted to account for the inherent variation in emissions and pollutants from livestock production systems.

Few LCA studies have investigated the environmental impact of the inclusion of WC in intensive PRG swards. Schils et al. (2005) reported a 10% reduction in GWP per kilogram of milk through the substitution of synthetic N fertilizer with the incorporation of WC into PRG swards. Similarly, Yan et al. (2013) reported a greater reduction in GHG intensity of 11 to 23% for extensive or semiextensive mixed PRG–WC swards relative to intensive swards. Consistent with our findings, the reduction was caused by reduced N_2O and CO_2 emissions. The magnitude of the reduction, however, was lower in the current study (7.9%; GR250 vs. CL150). This is partially attributed to greater use of synthetic N fertilizer in the mixed PRG–WC production system. The GWP results of our study were at the lower end or below the GWP range reported by Rotz (2018), 0.8 to 1.2 kg of CO_2 eq/kg of FPCM. Rotz (2018) concluded that the factors that have the greatest influence on GHG intensity are herd milk production and animal diet. Although increasing milk production normally increases total system emissions, it usually reduces GWP per unit of milk when cow fertility and health are not impaired. This dilution effect was seen in the current study when comparing the GR250 and CL250 systems. By contrast, the current study found total GWP, AP, MEP, and NRE from the CL150 system to be lower than that from the GR250 system despite an increase in milk production. This reduction was instigated by the substitution of synthetic N fertilizer with atmospheric-derived N. Interestingly, the CL250 system had the highest environmental impact per hectare across all categories reported, indicating that if WC is included in a production system and synthetic fertilizer application rate or stocking rate are not reduced, the environmental impact of the production system is likely to increase.

Schils et al. (2005) reported that total emissions per hectare for a Dutch dairy system based on PRG and WC swards were similar to those for the CL150 and CL250 systems despite having higher milk production per cow (8,294 kg of FPCM/cow). The lower stocking rate and greater concentrate consumption (~1,800 kg/cow) in the mixed PRG and WC system assessed by Schils et al. (2005) likely resulted in the lack of difference in total emissions per hectare but caused higher GWP per unit of product (1.04 kg of CO_2 eq/kg of FPCM) than our PRG–WC systems. The GWP per kilogram of FPCM of the 3 dairy systems in this study were lower than the GWP reported by O'Brien et al. (2012) for an intensive Irish pasture-based dairy system (0.87 kg of CO_2 eq/kg of FPCM). The minor difference is likely to be associated with the use of the most recent coefficients and country-specific EF in this study and minor differences in animal productivity and stocking rate. Chobtang et al. (2016) reported a GWP range of 0.78 to 0.82 kg of CO_2 eq/kg of FPCM for dairy farms in New Zealand using the physical causality allocation method recommended by IDF (2015). Based on the sensitivity analysis, the GWP of the production systems in the current study were slightly lower than those reported by Chobtang et al. (2016). However, it must be noted that the production systems in the current study were based in a research environment.

Recently, there has been an increasing focus on assessing GWP with more impact categories to provide a well-rounded perspective on the environmental performance of a product. The European Commission (2018) developed the Product Environmental Footprint guide with the aim of developing a harmonized European methodology to quantify the environmental impact of a product. The new guide is expected to set the basis for better reproducibility and comparability of LCA study reports. Chobtang et al. (2016) reported an average AP of 0.0153 (0.0148–0.0159) mol_c of H^+ eq/kg of FPCM from an analysis of 53 New Zealand dairy farms. In contrast, the current study found AP of 0.0058 to 0.0062 mol_c of H^+ eq/kg of FPCM. The large difference in AP was partly due to the characterization factors selected. The current study used country-specific characterization factors, whereas Chobtang et al. (2016) used standard classification factors, both provided by Posch et al. (2008).

Italian studies conducted by Famiglietti et al. (2019) and Battini et al. (2016) also reported greater AP per kilogram of FPCM than the current study (0.022 and 0.012 mol_c of H^+ eq/kg of FPCM, respectively). Differences between studies are largely attributed to variation in production systems and methodology. In contrast to the pasture-based systems simulated in the current study, all farms assessed by Famiglietti et al. (2019) housed animals full time. Furthermore, both Famiglietti et al. (2019) and Battini et al. (2016) used older versions of the EEA (2019) guidebook to calculate NH_3 emissions. Although adopting a similar N mass flow method, the current study used EF reported by Misselbrook et al. (2016) as used in the national inventory (Duffy et al., 2019b). The NH_3 EF for liquid manure storage, manure application, and manure deposition at pasture provided by Misselbrook et al. (2016) are notably lower than those reported in the EEA (2019) guidebook.

Despite having similar stocking rates, the MEP reported in the current study was greater than the average that Chobtang et al. (2016) reported (2.67 g of N eq/kg of FPCM). In contrast, Battini et al. (2016) reported MEP of 8.8 to 15.7 g of N eq/kg of FPCM for the indoor dairy systems in Po Valley in Italy using the physical causality allocation method. Differences in MEP are attributed to differences in farming systems as well as methodologies (i.e., life cycle inventory methodology, allocation methods, and characterization factors). Both Chobtang et al. (2016) and the current study agree that synthetic fertilizers were a key contributor to MEP. Concentrate feed was a minor contributor to MEP in this study and for the average New Zealand dairy system (6%) reported by Basset-Mens et al. (2009) as it was a small component of dairy cattle diet. Opposing these findings, Chobtang et al. (2016) found imported feed to contribute 12 to 25% of eutrophication potential, highlighting the environmental importance and benefits of the use of home-grown forage.

Consistent with Thomassen et al. (2008) and Basset-Mens et al. (2009), the majority of MEP occurred on-farm (82.2%) via N losses. The CL250 system had the greatest N leaching potential through the addition of N fixed by WC increasing the quantity of the soil N pool susceptible to leaching. The CL150 treatment used less N fertilizer and thus had the lowest potential to leach NO_3^-, which supports the findings of Ledgard et al. (2009), who identified an exponential rise in the rate of NO_3^- leaching with the increase in N fertilizer application and N_2 fixation by WC. Therefore, the impact assessment of the CL150 system indicates that there is scope to reduce the MEP of intensive pasture-based dairy systems without affecting milk production through the substitution of synthetic N fertilizer with the incorporation of WC into swards. The FEP reported in the current study was lower than the average of the pasture-based systems in Chobtang et al. (2016) and the indoor dairy systems reported in Battini et al. (2016). The lower FEP is attributed to the combination of the current study having lower P imported onto farm (i.e., concentrate feed and fertilizer production), lower synthetic P fertilizer application, and variation in life cycle inventory methodology.

With developments in synthetic fertilizer production technology, the energy required to produce a unit of N has decreased by approximately 20% (Brentrup and Palliére, 2011). Similarly, the development and implementation of renewable energy technologies has reduced the proportion of NRE embodied in electricity (SEAI, 2019). These improvements in energy use efficiency are evident when comparing the NRE for the GR250 (21.3 GJ/ha) with a similar production system in the study by O'Brien et al. (2012; 31.2 GJ/ha). Furthermore, comparing the NRE of the GR250 and CL250 (21.6 GJ/ha) systems with the CL150 (17.7 GJ/ha) system validates that reducing the reliance on synthetic fertilizer by including WC in swards reduces pasture-based dairy system dependence on NRE.

Allocation method is a major issue for LCA of multifunctional systems due to its effect on results and conclusions (Rice et al., 2017). The IDF (2015) guide for LCA methodology recommends allocation by physical causality using methodology outlined by Thoma et al. (2013). As previously reported by Rice et al. (2017) and Battini et al. (2016), the physical causality allocation method was biased in favor of milk production compared with other allocation methods. Rice et al. (2017) questioned the appropriateness of using an allocation method derived from a data set unrepresentative of many regions and the decision to allocate replacement heifer growth emissions to meat production. In contrast to physical causality, mass-based allocation method was biased against milk production. Although high-quality data are generally available for total milk and live weight sold, mass allocation assumes equal environmental impact per unit mass of co-product. In reality, meat is more valuable than milk per unit of mass and therefore should be allocated a greater share of impacts (Battini et al., 2016). Gerber et al. (2010) alternatively suggested protein-based allocation as it is reflective of the functionality of dairy systems, converting nutrients into high-value protein products. Furthermore, data on milk protein content are readily available as it is recorded to calculate milk prices; however, protein content of meat sold from farm is rarely recorded. Further standardization of allocation method and the adoption of a common harmonized LCA approach will reduce cases of inconsistencies in the application of LCA.

CONCLUSIONS

This LCA study shows that the inclusion of WC in PRG swards can reduce the environmental impact of intensive pasture-based dairy production systems at a given stocking rate compared with PRG-only swards. Comparing CL150 and GR250, the substitution of C emission-intensive synthetic N fertilizer with C-neutral N derived by WC reduced anthropogenic N_2O, CO_2, and NH_3 emissions along with NO_3^- leaching and NRE usage from pasture-based dairy systems. Moreover, the improvement in animal performance with the inclusion of WC further reduced the environmental impact of intensive pasture-based dairy systems, with the CL150 system having the lowest impact per hectare and metric tonne of FPCM. In turn, the inclusion of WC reduced the environmental intensity of dairy systems through improved animal performance as well as through the reduction in total environmental impact. This LCA study is one of few that investigated the effect of WC on the GWP of intensive pasture-based dairy systems and, to our knowledge, the only one that included more than 1 impact category. More research should be carried out using a common harmonized LCA methodology to confirm the environmental benefits of WC for pasture-based dairy systems.

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