Environmental efficiency of alternative dairy systems: A productive efficiency approach

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ABSTRACT

Agriculture across the globe needs to produce “more with less.” Productivity should be increased in a sustainable manner so that the environment is not further degraded, management practices are both socially acceptable and economically favorable, and future generations are not disadvantaged. The objective of this paper was to compare the environmental efficiency of 2 divergent strains of Holstein-Friesian cows across 2 contrasting dairy management systems (grazing and non-grazing) over multiple years and so expose any genetic $\times$ environment ($G \times E$) interaction. The models were an extension of the traditional efficiency analysis to account for undesirable outputs (pollutants), and estimate efficiency measures that allow for the asymmetric treatment of desirable outputs (i.e., milk production) and undesirable outputs. Two types of models were estimated, one considering production inputs (land, nitrogen fertilizers, feed, and cows) and the other not, thus allowing the assessment of the effect of inputs by comparing efficiency values and rankings between models. Each model type had 2 versions, one including 2 types of pollutants (greenhouse gas emissions, nitrogen surplus) and the other 3 (greenhouse gas emissions, nitrogen surplus, and phosphorus surplus). Significant differences were found between efficiency scores among the systems. Results indicated no $G \times E$ interaction; however, even though the select genetic merit herd consuming a diet with a higher proportion of concentrated feeds was most efficient in the majority of models, cows of the same genetic merit on higher forage diets could be just as efficient. Efficiency scores for the low forage groups were less variable from year to year, which reflected the uniformity of purchased concentrate feeds. The results also indicate that inputs play an important role in the measurement of environmental efficiency of dairy systems and that animal health variables (incidence of udder health disorders and body condition

score) have a significant effect on the environmental efficiency of each dairy system. We conclude that traditional narrow measures of performance may not always distinguish dairy farming systems best fitted to future requirements.

Key words: environmental efficiency, experimental dairy farm, nonparametric data envelopment analysis

INTRODUCTION

During the second half of the 20th century, the agriculture industry harnessed technology that has resulted in crop and animal productivity increases of around 2% per annum (Duvick, 1986; Ludena et al., 2007), mainly from advances in genetics, management practices, and forage improvements (Pryce et al., 2004). However, significant challenges are yet to be addressed, as global warming effects are apparent and unsustainable food production systems are causing environmental degradation (Foresight, 2011). Against a background of population growth and food security concerns at the global level, the UK population is forecasted to increase by 14% by 2033 (ONS, 2012); if productivity gains continue at the same rate, then UK domestic demands could be met. However, UK dairy farming has been in decline for over 30 yr. During this time, the national milking herd has contracted by 43% (since 1980), yet efficiency has increased as total UK production fell by 20% over the same time period (Defra, 2011). Average UK yields per cow continue to increase while the overall number of farms continues to decrease (DairyCo, 2011). In addition to population growth and environmental concerns, further challenges for the future of food production include increased competition for land, energy, and water resources; maintaining food security; and managing changes in consumer behavior. These forces of change require the food system to be radically redesigned to achieve production in the long term (Foresight, 2011), but which dairy systems are optimal for delivering milk production that is sustainable?

In global terms, milk production systems are responsible for approximately 2.7% of all anthropogenic greenhouse gas emissions. As demand for dairy and

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meat products is expected to double by 2050, it is crucial for the dairy industry to aim toward more sustainable low-carbon methods of production (FAO, 2010). Dairy systems are also a source of nutrient losses to the wider environment. Livestock excrete nitrogen (N) and phosphorus (P), which have ecological impacts (Erisman et al., 2007), such as water pollution caused by nitrate leaching, eutrophication of surface waters due to N and P enrichment, and soil acidification and plant damage from ammonia emissions (Amon et al., 2011). Moreover, the Haber process used to manufacture N is energy intensive, which incurs CO₂ emissions, and P is a nonrenewable, depleting resource.

Sources of pollutants arising from dairy farms and associated mitigation strategies are well researched (Defra, 2008; DairyCo, 2009; FAO, 2010) and should be implemented dependent upon system type, location, and historical land use (IPCC, 2007). The focus on food security and climate change has led to increased attention on the measurement of both economic and environmental performance of agricultural production systems. Evidence regarding differences in the overall environmental impact created by a range of dairy farming systems could be improved as there is a need for evaluation of the differences between total resource use across different dairy systems (Alvis et al., 2012). Comparative analysis on the effect of dairy management systems that favor livestock on pasture land or not would aid the understanding of livestock sector impacts (Anderson and Gundel, 2011).

This study applied data envelopment analysis (DEA) to measure relative differences in the environmental efficiency of grazing and nongrazing dairy systems across 2 genetic lines of cows to investigate any genotype $\times$ environment ($G \times E$) interaction and to determine the wider environmental impact of each of the livestock systems while taking milk production into consideration. Types and amounts of undesirable outputs associated with each of the systems were evaluated and compared to identify possible options for mitigation of pollutants, because future legislation may require environmental externalities to be accounted for and reduced. In addition, the analysis estimated changes in the environmental efficiency of each system over time and the effect of independent variables on the efficiency of each system.

MATERIALS AND METHODS

Langhill Feeding Experiments

Data were obtained from the Langhill herd (Crichton Research Farm, Dumfries, Scotland) of Holstein-Friesian dairy cows, which are part of a long-term experiment to assess genetic line $\times$ feeding system interactions (Pol-

lott and Coffey, 2008). The herd represents a range of dairy management methods in terms of breeding and feeding systems, and consists of 2 contrasting genetic lines, selected either for increased milk fat plus CP yield (select line, **S**) or to remain close to the average genetic merit for milk fat plus CP yield for Holstein-Friesians evaluated annually in the United Kingdom (control line, **C**; Pryce et al., 1999).

During the experiment, cows were allocated to either a low forage (**LF**) diet consuming 3.0 t of concentrate annually or a high forage (**HF**) diet containing approximately 1.2 t of concentrate (Chagunda et al., 2009). The LF cows remained indoors (i.e., nongrazing), whereas the HF cows were grazed when possible. Both groups were managed so that calving took place all year round. The feeding systems within the herd were defined as low forage control (**LFC**), low forage select (**LFS**), high forage control (**HFC**), and high forage select (**HFS**). Approximately 50 cows in each of the 4 groups were fed a TMR plus concentrate in the milking parlor. Forages fed in both LF and HF diets included grass silage, maize silage, and whole-crop wheat silage. Diets within LF herds consisted of approximately 47% forage, and in the winter months, the HF herd was housed and fed 75% forage on a DM basis (Chagunda et al., 2009). Diets were formulated to provide adequate ME and CP to meet the requirements of a cow within each genetic line $\times$ feeding system (Pollott and Coffey, 2008).

Data

The data set used in this analysis consisted of variables from all cows within the experiment between 2004 and 2010, irrespective of their health. Variables were extracted from the database for each cow and aggregated annually at the 4 system levels. Daily milk yields were the sum of 3 milkings daily, and 3-times-daily fat and CP concentrations of the milk were analyzed once per week and averaged. Cows were fed ad libitum, and feed intakes for those cows indoors were recorded for lactating cows using Hoko automatic feed measurement gates (Insentec BV, Marknesse, the Netherlands). Data regarding milk yield, fertilizer application, energy use, and diet were extracted directly from the database, and data for herd dynamics were taken from an annual inventory of the systems. The protocol of the experiment required that cows were kept in the herd for at least 3 lactations unless cow welfare dictated that culling was necessary. Cows could remain in the herd for more than 3 lactations if a suitable heifer of the same genetic line was not available. Culling time was defined as the date on which a cow's productive life ceased, and cows had a maximum of 7 chances to conceive before being culled.

Carbon Footprints. The PAS2050-accredited SAC Carbon Calculator v3.11 (SAC, 2011) was used to estimate greenhouse gas (GHG) emissions attributed to each of the 4 groups. The calculator uses IPCC (2007) Tier 2 methodologies, equations, and emission factors, and requires detailed information regarding farm inputs and outputs related to livestock, land and crops, purchased feeds, and energy use. Herd dynamic inputs required an annual reconciliation of all ages of livestock taking into consideration sales, purchases, and culling of cows. A summation of home-grown and purchased feeds was footprinted per type and manure management practices were expressed depending on the amount of time the cows spent grazing or housed. Hectares of land required and fertilizer applications were allocated on a per-crop and fertilizer-type basis, which allowed the calculation of emissions of direct and indirect CO₂ as well as nitrous oxide released following decomposition of crop residue in the soil. Use of petrol and diesel was recorded in the database; however, electricity (kWh) was estimated from milk yield (Sheane et al., 2010) as there was no on-farm electricity meter. Sprays of pesticides, fungicides, herbicides, and sequestration were not included in the footprints because their use was limited or data were not available. Table 1 shows a selection of emission factors applied within the model.

Results were expressed in terms of ECM, and the formula for determining ECM (kg) followed Sjaunja et al. (1990):

$$\text{ECM} = 0.25 \times \text{mass of milk} + 12.2 \\ \times \text{fat (kg)} + 7.7 \times \text{protein (kg)}. \quad [1]$$

Apportioning of emissions to the meat produced by the systems produces alternative emission totals depending on the method applied. Several publications, including Casey and Holden (2005), quantify these

reductions for using varied economic and mass allocation methods; however, in this study no allocation was applied. This was because replacement rates among the systems were similar, culled cows had a range of destinations, and the emissions from those calves that left the system for beef would be attributable to the farms that spend up to 12 mo raising them.

Farm Nutrient Balances. When considering agri-environmental sustainability, N and P balances are key indicators (Amon et al., 2011). Quantification of farmgate surpluses of N and P is a widely used method that provides an indication of the longer term risks of nutrient losses to the natural environment by leaching and volatilization (Jarvis and Aarts, 2000). A nutrient balance can be described as a measure of the inputs and outputs within a system, where a nutrient surplus is the difference between the input and output of each nutrient.

Absence of an international legal framework designed to account for N and P losses results in a lack of common standards and methodologies (Amon et al., 2011). After reviewing available methodologies and their data requirements, a farm N balance and farm P balance were calculated for each of the 4 groups annually. Farm balances were a broad indicator, similar to the Organisation for Economic Co-operation and Development's gross balance, with the only difference being that the balance applied here included imports of bedding, which can be significant (Amon et al., 2011). Although it is recognized that soil N and P balances are more representative of changes of nutrient storage within the soil and hence provide a more accurate indication of losses to the aquatic environment, the data required for these balances require estimations associated with greater ambiguity than the farm balances used here (Figure 1).

For each herd, the variables required for calculation of N and P balances were taken from the database where

Table 1. Emission factors and equations applied within the SAC Carbon Calculator v3.11 (SAC, 2001) to determine the carbon footprints of milk production systems

Source	Emission factor/equation	Reference
Manure CH ₄ /cow	Max 0.24 m ³ of CH ₄ /kg of volatile solids	IPCC, 2006 (Chapter 10, page 10.44)
Nitrogen fertilizer	7.11 kg of CO ₂ /kg of N	Carbon Trust Footprint Expert 3.1 ¹
Phosphate fertilizer	1.85 kg of CO ₂ /kg of P ₂ O ₅	Carbon Trust Footprint Expert 3.1
Electricity	0.59 kg of CO ₂ /kWh	DECC, 2011
Diesel	3.18 kg of CO ₂ /L	DECC, 2011
Compound feed ²	341 kg of CO ₂ /t ³	Carbon Trust Feeds Database v11(2010)
Blended feed LF ⁴	154 kg of CO ₂ /t ³	Carbon Trust Feeds Database v11(2010)
Blended feed HF ⁴	198 kg of CO ₂ /t ³	Carbon Trust Feeds Database v11(2010)
Nitrogen excretion	0.48 kg of N/1,000 kg of cow per day	IPCC, 2006 (Chapter 10, page 10.59)

¹Footprint Expert Product Footprinting Tool [Excel]; Carbon Trust, London, UK.

²Derived from typical industry average ingredients estimated by nutritionist Colin Morgan (SAC Consulting, Penicuik, UK; unpublished, 2011).

³Emission factors for individual ingredients were sourced from Carbon Trust Feeds Database v11 (2010; Carbon Trust, London, UK).

⁴LF = low forage diet; HF = high forage diet.

Table 2. Nitrogen fixation rates per crop type

Land use	Fixation rate	Reference
Agricultural land	90 kg of N/ha per year	Deacon, 2000
Winter wheat	2.5 g of N/ha per day for 100 d	Pederson et al., 1978
Pasture (white clover)	140 kg of N/ha per year	Leach et al., 2002
Maize (corn)	4 g of N/ha per day	Barber et al., 1976

possible, and factors were applied to estimate nutrient inputs and outputs for each system. No manure was imported onto the farm or exported from the farm and no crop products were sold during the experiment.

Farm N Balance. For farm outputs, the average protein content of milk produced was taken from the database and applied to the total yield of each group, with the percentage of N in CP taken as 16%. The amount of N in animals sold (N_{animal}) was calculated based upon the animal weight sold (ARC, 1980):

$$N_{\text{animal}} (\text{kg}) = 0.214 \times \text{live weight (kg)} + 0.299. \quad [2]$$

Data related to inputs of N in the form of fertilizer and feed were taken from the database and expressed in 100% N. The N content of imported feed such as sugar beet pellets was taken directly from feed labels on the farm or from the Feeds Directory (Ewing, 2004). Nitrogen within blended feeds was calculated from results of monthly CP analyses of the feedstuffs. The farm manager provided details of seed application rates (H. McLymont, Crichton Research Farm, Dumfries, UK, personal communication). The N content of seeds for each of the crops, which was taken from the literature (Triboi and Triboi-Blondel, 2001; Yu et al., 2004), and fixation rates, which were applied per crop type, are shown in Table 2.

Rates of atmospheric deposition (kg/ha) for the Dumfries area were available for each year except 2010 (Defra, 2012). Deposition for 2010 was taken as an average of the 6 yr before 2010. Table 3 outlines descrip-

tive statistics for variables used to calculate the farm nitrogen balance.

Farm P Balance. Calculation of farm P balance was relatively straightforward because fewer variables were included and there was no gaseous loss of P from manure management systems. Calculating the amount of P in the inputs and outputs of the farm balance was carried out in the same manner as described for the N balance. The amount of P in feed products was taken from the products themselves or the Feeds Directory (Ewing, 2004), and the P in milk was calculated using a factor provided by the Dairy Council (2002). The amount of P in animals sold (P_{animal}) was calculated from an equation based on the weight of animals sold (ARC, 1980):

$$P_{\text{animal}} (\text{kg}) = 0.00067 \times \text{live weight (kg)} + 0.055. \quad [3]$$

Table 4 shows descriptive statistics for variables used to calculate the farm P balance. Phosphate fertilizer was only applied in years 2004 to 2006.

DEA

Data envelopment analysis is a frequently used method to estimate the efficiency of production systems (Färe et al., 1994). It is presumed, according to traditional microeconomic theory based on the assumption of optimizing behavior, that producers optimize from a technical perspective by not wasting resources. This means that they operate somewhere on the boundary,

Table 3. Descriptive statistics for variables for each milk production system within farm nitrogen balance 2004–2010¹

Variable	LFC		LFS		HFC		HFS	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD
N Inputs								
Fertilizer (t)	2.99	0.3	3.01	0.39	4.44	0.42	4.39	0.39
Feeds (t)	6.90	1.2	7.00	1.39	2.48	0.47	2.32	0.49
Seeds (kg)	1.66	0.31	1.72	0.30	1.69	0.35	1.72	0.34
Deposition (t)	0.29	0.05	0.29	0.06	0.42	0.08	0.42	0.08
Fixation (t)	2.53	0.55	2.53	0.59	4.22	0.95	4.18	0.98
N Outputs								
Milk (t)	2.29	0.20	2.65	0.44	1.99	0.20	2.25	0.32
No. of cows sold	0.32	0.11	0.24	0.05	0.32	0.14	0.24	0.12
N surplus (t)	10.5	1.62	10.4	1.78	9.68	1.42	9.24	1.38

¹Feed systems: HF = high forage system, LF = low forage system; genotype: C = control, S = select.

Table 4. Descriptive statistics for variables for each milk production system within farm phosphate balance 2004–2010¹

Variable	LFC		LFS		HFC		HFS	
	Mean	SD ²	Mean	SD	Mean	SD	Mean	SD
P Input								
Fertilizer (kg)	15.9	26.2	15.1	23.8	23.7	40.9	23.4	39.9
Feeds (t)	1.65	0.23	1.61	0.23	1.05	0.23	0.94	0.23
P Output								
Milk (t)	0.42	0.047	0.46	0.077	0.36	0.036	0.39	0.054
Cows sold (kg)	99.1	35.2	74.6	16.4	101.6	43.6	75.6	36.1
P surplus (t)	1.26	0.17	1.21	0.15	0.72	0.15	0.62	0.14

¹Feed systems: HF = high forage system, LF = low forage system; genotype: C = control, S = select.

rather than on the interior, of their production possibility sets. The production possibility set is the set of all nonnegative outputs of goods and services that can be produced using the economy’s available factor inputs. The output combinations on the frontier of this set correspond to the Pareto-efficient allocation of factor inputs; that is, the allocation at which it is not possible, given the total factor endowment, to increase the production of one good without decreasing the production of some other good. For different reasons, not all producers succeed in solving the optimization problem in all circumstances. That is why it is important to have a way of analyzing the degree to which producers fail to optimize and the extent of the deviations from technical and economic efficiency (Färe et al., 1994). Several studies have applied DEA to measure the efficiency of dairy farms (Bravo-Ureta and Reiger, 1991; Barnes et al., 2011; Shortall and Barnes, 2013).

Production processes generate 2 major types of outputs: desirable outputs (e.g., milk) and undesirable outputs (e.g., pollutants associated with agricultural emissions from dairy farms). Desirable outputs are considered as strongly disposable; that is, it is always possible to reduce the production of a desirable output without increasing costs. On the other hand, undesir-

able outputs are weakly disposable; that is, it is not possible to reduce the production of an undesirable output without reducing the production of a desirable output or increasing the use of an input.

The extension of traditional efficiency analysis to account for undesirable outputs began with Pitman (1983) and has since been further developed along different lines. The basic idea is the estimation of efficiency measures that allow for the asymmetric treatment of desirable and undesirable outputs. Traditional DEA models have been adjusted by considering not only inputs and desirable outputs, but also the third type of factors—the pollutants or undesirable outputs. Färe et al. (1996) introduced the use of an activity analysis model based on the division of overall efficiency into productive efficiency and environmental efficiency. Piot-Lepetit et al. (1995) present information about DEA for identification of agricultural input adjustment which might result in improved technical efficiency and at the same time reduced environmental impacts. Tyteca (1996) investigated the possibilities of obtaining environmental performance indicators from an approach that would be analogous to that classically used to quantify output, input, or overall productive efficiency. He introduced alternative DEA models that differ in their complexity level and in the way in which they account for the 3 categories of factors (desirable outputs, inputs, and undesirable outputs). Various data envelopment techniques for environmental efficiency measurement have been presented in the literature (Tyteca, 1996; Kuosmanen and Kortelainen, 2005; Kortelainen, 2008).

We used 2 DEA models: the undesirable output-oriented model (UO) and the normalized undesirable output-oriented (NUO) model (Tyteca, 1996).

The UO model is as follows:
minimize

$$T \ (T \leq 1)$$

[4]

subject to

Figure 1. Constituents of a farm N balance. Atm. Dep. = atmospheric deposition. Source: Amon et al. (2011). © European Union, 2011. Reproduced with permission.

$$\begin{aligned}
\sum_{k=1}^K \mathbf{Z}^k v_m^k &\geq v_m^0 \\
\sum_{k=1}^K \mathbf{Z}^k w_j^k &= T w_j^0 \\
\sum_{k=1}^K \mathbf{Z}^k x_n^k &\leq x_n^0, n = 1, \dots, N, \\
\mathbf{Z}^k &\geq 0
\end{aligned}
\qquad
\begin{aligned}
&\frac{\sum_{j=1}^J c_j w_j^k}{\sum_{m=1}^M a_m v_m^k - \sum_{n=1}^N b_n x_n^k} \geq 1 \\
&a_m, b_n \geq 0
\end{aligned}$$

where m , n , and j are the numbers of desirable outputs, inputs, and undesirable outputs, taking values from 1 to M , 1 to N , and 1 to J , respectively; k is the number of observations (producers, time periods, or in our case, dairy systems by year and treatment), taking values from 1 to K ; v_m^0, w_j^0, x_n^0 are desirable outputs, undesirable outputs, and inputs, respectively. In case of observation 0, observation $0 \in \{1, \dots, K\}$ takes values from 1 to K , successively. Variable T represents the standardized indicator of environmental performance; $\mathbf{Z}$ is a vector that denotes the intensity levels at which each of the K observations are (or might be) conducted; $\mathbf{Z}$ is a vector that enables shrinking or expanding individual observed activities, for the purpose of constructing unobserved but nonetheless feasible activities. It provides weights that facilitate the construction of the linear segments of the piecewise linear boundary of the technology.

The model shows one key difference to the classical DEA formulation; namely, that instead of minimizing a ratio of inputs to outputs or maximizing a ratio of outputs to inputs, it minimizes a ratio of undesirable outputs to a weighted sum of desirable outputs and inputs. Thus, the undesirable outputs are considered as peculiar outputs that one tries to minimize with respect to the other factors of production (inputs and desirable outputs; Tyteca, 1996):

minimize

$$h_0 = \frac{\sum_{j=1}^J c_j w_j^0}{\sum_{m=1}^M a_m v_m^0 - \sum_{n=1}^N b_n x_n^0} \quad [5]$$

where h_0 represents the standardized indicator of environmental performance; and a_m , b_n , and c_j denote intensity levels, subject to

The problem assumes constant returns to scale; that is, in pollution terms, for efficient dairy systems, namely those showing a value of $T = 1$, a given increase in desirable outputs or inputs, or both, would result in a proportional increase in undesirable outputs (Färe, 1992). The problem can be modified to assume varying returns to scale by the addition of the constraint presented in Equation [6] (Lovell, 1993; Yaisawarng and Klein, 1994; Tyteca, 1996):

$$\sum_{k=1}^K \mathbf{Z}^k = 1 \quad [6]$$

One limitation of this approach is that all undesirable outputs have to vary according to the same proportion because of the uniqueness of the T factor affecting all undesirable output components (Tyteca, 1996).

The NUO model is as follows:
minimize

$$T \quad (T \leq 1), \quad [7]$$

subject to

$$\sum_{k=1}^K \mathbf{Z}^k v_m^k \geq v_m^0$$

$$\sum_{k=1}^K \mathbf{Z}^k w_j^k = T w_j^0, j = 1, \dots, J,$$

$$\mathbf{Z}^k \geq 0; k = 1, \dots, K.$$

Compared with the UO model, the NUO model is a simplified version of the DEA formulation in the sense that it no longer accounts for inputs. In the case of only one desirable output, we simply consider weighted sums of undesirable outputs scaled by the desirable output (Tyteca, 1996).

Estimating the models allowed us to assess the effect of inputs by comparing efficiency values and rankings between models. Four models were estimated: 2 UO mod-

els, where undesirable outputs were (1) GHG emissions (CO_2 equivalent; CO_{2e}) and N surplus ($\text{UO CO}_{2,N}$), and (2) CO_2 , N, and P surplus ($\text{UO CO}_{2,N,P}$); and 2 NUO models, where undesirable outputs were (1) CO_2 and N ($\text{NUO CO}_{2,N}$), and (2) CO_2 , N, and P ($\text{NUO CO}_{2,N,P}$). Milk production was the desirable output in all 4 models. Inputs included in the 2 UO models were land, nitrogen fertilizers, feed (home-grown and purchased), and cows (milking cows and young stock).

In building the relative environmental efficiency measure, we use the DEA endogenous weighting scheme (Farrell, 1957; Charnes et al., 1978; Tyteca, 1996); that is, instead of a priori specified (subjective) weights based on potential environmental damage, this technique endogenously selects the weights that present every evaluated decision-making unit (DMU) in the most favorable light.

Applying weights to environmental factors is an approach used by researchers dealing with pollutant inventories; however, it is not evident that deriving individual weights would properly quantify the combined effect of all pollutants in all possible site-specific contexts, as their effects are likely to vary as a function of the geographical site considered (Milford et al., 1989), and a lack of consensus exists in this regard (Tyteca, 1996). Moreover, it is even less evident that the combined effects of pollutants would result in linear interactions.

As an alternative to the weights based on impact on environment, cost coefficients that reflect the damage caused by the effluents to the environment could be used if and when available. This procedure has some recognized difficulties; namely, although there are some estimates of the damage costs associated with GHG and for carbon value at the UK level (Price et al., 2007; DECC, 2009), marginal damage cost values for the other pollutants (e.g., NH_3 , N leaching or P) are even more contested due to the uncertainties involved (Eory et al., 2013). Part of the reason is that the other pollutants are more localized than GHG, so their marginal damage value is highly dependent on (and varies with) the region where the emissions occur (based on the baseline pollutant load, the threshold for pollution effects, and what is being damaged in the specific area). Area-specific valuation of each pollutant's damage costs is rarely available. Another reason is that N and P surpluses are indicators of the N and P excess; however, no information is available on the different forms of these pollutants. For instance, N_2O , NO , NO_2 , NH_3 , and NO_3^- are pollutants, whereas N_2 is harmless (Sutton et al., 2011); they have different effects on the environment and therefore should be assigned different weights. Without this information, marginal

damage costs and, implicitly, correct weights cannot be assigned to these undesirable outputs.

Data envelopment analysis is a method that does not require the specification of a priori weights on the variables, thus it is ideally suited for our case. We estimated the models using the General Algebraic Modeling System (GAMS 22.8); GAMS programs for the DEA models are available on request from the authors.

We used DEA to account for temporal aspects; that is, not only to compare the dairy systems among themselves, but to quantify changes in environmental efficiency over time. The models consider each of the 4 systems (LFC, LFS, HFC, and HFS) as divided into 7 independent DMU (for each year between 2004 and 2010), which results in a set of 28 DMU (a similar approach was used by Färe et al., 1996 and Ball et al., 1994). Thus, the best-practice production frontier is composed of systems that were efficient in any of the 7 yr. The analysis allows us to provide a measurement of improvement (or deterioration) in environmental efficiency for each system over time.

One-Way ANOVA

We used a one-way ANOVA procedure to identify significant differences between efficiency scores by year and dairy system type. We tested for mean differences using post hoc tests [least significant difference (LSD) test for equal variances and Tamhane test for unequal variances].

Tobit Model

We use the Tobit model (Tobin, 1958) to estimate the effects of variables related to the physical condition of animals within the milking herd on the environmental efficiency of dairy systems. The Tobit model has been used in this context frequently in the literature (Coelli et al., 2002; Hansson, 2008; Manevska-Tasevska et al., 2011) because the dependent variable takes values varying from zero to unity.

The model is presented in Equations [8] and [9] (Greene, 2003):

$$y_i = \begin{cases} y_i^* & \text{if } y_i^* > 0 \\ 0 & \text{if } y_i^* \leq 0 \end{cases} \quad [8]$$

$$y_i^* = \beta x_i + u_i, \quad u_i \sim N(0, \sigma^2), \quad [9]$$

where y_i^* is a latent dependent variable; x_i is the independent variable; β is a parameter (vector) that deter-

Table 5. Mean N and P surpluses and greenhouse gas (GHG) emissions per liter of ECM for each milk production system 2004–2010¹

Output	Type	LFC	LFS	HFC	HFS
Nitrogen	Surplus/L (kg)	0.023	0.021	0.025	0.022
Phosphorus	Surplus/L (kg)	0.0028	0.0025	0.0018	0.0015
GHG	Direct kg of CO ₂	0.09	0.08	0.09	0.08
	Indirect kg of CO ₂	0.23	0.20	0.25	0.21
	kg of CO ₂ e from methane ²	0.56	0.49	0.74	0.63
	kg of CO ₂ e from nitrous oxide	0.16	0.13	0.25	0.22
	kg of CO ₂ e/kg of ECM	1.04	0.90	1.33	1.14
Total GHG					

¹Feed systems: HF = high forage system, LF = low forage system; genotype: C = control, S = select.

²CO₂e = CO₂ equivalents.

mines the relationship between the independent variable x_i and the latent variable y_i^* ; u_i is a normally distributed error term.

Quasi-maximum likelihood (Huber/White) for robust standard errors was used to correct for heteroscedasticity. The dependent variable was the environmental efficiency score, and the independent variables included in the model were incidence of udder health disorders and BCS (a herd management tool that ensures cows are in the correct condition for each stage in their cycle and if not, dietary changes can be made). Udder disorders include mastitis, *Escherichia coli*, blockages, and reduced milk. The Tobit model was estimated with econometric software EViews7.1 (<http://www.eviews.com>).

RESULTS

Carbon Footprint and Nutrient Surplus Results

Surplus nutrients were greater in the LF groups, which mainly stems from quantities of N and P in imported feeds. However, we did not observe substantial differences between groups when comparing nutrient surpluses per kilogram of ECM because the LF groups generally produced considerably more milk per year. Phosphate surpluses showed greater variation between feed groups because of less P input in the HF diet. Output of P was greater in the control groups because a higher conception rate increased the number of animals sold from these groups. Table 5 provides descriptive statistics of GHG estimates and nutrient balance calculations averaged from 2004 to 2010. Annual results for each pollutant type per feeding system and genetic merit represent undesirable outputs that should be minimized.

On average, the LFS system produced less GHG per kilogram of ECM. This group produced less manure-related methane and nitrous gases because of the continuous housing management. The LF diet generated a greater amount of indirect CO₂ emissions from purchased feeds than the HF group, although the HF system required more land and fertilizer, which resulted

in a similar per-liter indirect CO₂ totals. The HF herd emitted more enteric methane per liter of milk than the LF herd because of their diet composition. Table 6 shows descriptive statistics for inputs, desirable outputs, and undesirable outputs, subsequently used as variables in the DEA models.

DEA Results

Results of the DEA indicated that the HF systems were more volatile than the LF systems, which could reflect the influence of crop production and forage quality on the HF diet (Table 7). The GAMS listing of weights (values of z-variables) for all iterations in each model are presented in Tables S1 to S4 in the online data supplement (<http://dx.doi.org/10.3168/jds.2013-6911>).

The highest to lowest efficiency systems were LFS > HFS > LFC > HFC when scores of each system in each model were averaged. For HF systems, the best and worst years coincided, which highlights the dependency of the HF system on annual weather patterns. In the case of LF systems, the best and worst years differed, which could be due to annual variation in culling rates between S and C cows. When considering associations between the environmental efficiency and productivity, it is clear that high environmental efficiency is not confined to higher production systems. Figure 2 shows average efficiency scores plotted against annual milk output for each system and indicates that the most efficient systems were spread across a production range (368–602 m³/yr). This was the case with all models and agrees with results from other published work (Iribarren et al., 2011). High production systems have consistently high efficiency; however, in some years, low production systems also have high efficiency.

We tested for mean differences between environmental efficiency scores by year and dairy system types using a one-way ANOVA. Results of testing differences in mean efficiency scores by year are presented in Tables 8 and 9 and Figure 3.

Table 6. Descriptive statistics of variables for each milk production system 2004–2010¹

Variable	LFC		LFS		HFC		HFS	
	Mean	SD ²	Mean	SD	Mean	SD	Mean	SD
Inputs								
Land use (ha)	25.3	5.0	25.6	5.4	37.6	7.7	37.4	8.1
N fertilizer (t)	3.0	0.3	3.0	0.4	4.4	0.4	4.4	0.4
Grown feed (t)	712.0	100.1	718.5	109.4	717.0	79.1	700.4	76.4
Bought feed (t)	461.5	52.5	452.3	56.8	306.9	33.1	284.3	28.5
Number of milking cows	48	2.8	43	5.9	51	4.4	49	4.6
Number of young stock	70	13.1	60	8.24	77	13.6	61	10.5
Desirable outputs								
ECM (t)	441.8	45.2	500.0	87.0	392.3	34.9	436.2	58.0
Undesirable outputs								
GHG CO ₂ e ² (t)	460.0	56.4	451.8	65.7	517.8	59.9	497.9	66.1
N Surplus (t)	10.5	1.6	10.4	1.8	9.7	1.4	9.2	1.4
P Surplus (t)	1.3	0.17	1.2	0.15	0.7	0.15	0.6	0.14

¹Feed systems: HF = high forage system, LF = low forage system; genotype: C = control, S = select.²Greenhouse gases in CO₂ equivalents.**Table 7.** Environmental efficiency scores for the models applied to each milk production system from 2004 to 2010

System/year ¹	Model ²			
	NUO CO ₂ ,N,P	NUO CO ₂ ,N	UO CO ₂ ,N,P	UO CO ₂ ,N
LFC				
2004	0.915	0.904	0.915	0.912
2005	1	0.833	1	0.833
2006	0.876	0.833	0.88	0.854
2007	0.88	0.862	0.88	0.862
2008	0.902	0.866	0.902	0.866
2009	0.922	0.849	1	0.856
2010	0.8	0.791	0.8	0.791
LFS				
2004	0.996	0.986	1	1
2005	1	0.895	1	0.896
2006	1	0.966	1	1
2007	1	1	1	1
2008	1	1	1	1
2009	1	0.987	1	1
2010	1	1	1	1
HFC				
2004	0.82	0.804	0.82	0.804
2005	0.861	0.858	0.861	0.858
2006	1	1	1	1
2007	0.826	0.676	0.826	0.676
2008	0.723	0.688	0.723	0.688
2009	0.828	0.828	0.828	0.828
2010	0.731	0.724	0.731	0.724
HFS				
2004	1	0.973	1	1
2005	0.992	0.979	1	1
2006	1	1	1	1
2007	1	0.759	1	0.826
2008	0.868	0.824	1	1
2009	0.895	0.89	0.905	0.902
2010	0.986	0.97	1	1

¹Feed systems: HF = high forage system, LF = low forage system; genotype: C = control, S = select.²NUO CO₂,N,P = normalized undesirable output-oriented (NUO) model with undesirable outputs CO₂, N, and P; NUO CO₂,N = NUO model with undesirable outputs CO₂ and N; UO CO₂,N,P = undesirable output-oriented (UO) model with undesirable outputs CO₂ and N, P; UO CO₂,N = UO model with undesirable outputs CO₂ and N.

Figure 2. Average of efficiency scores versus raw milk production for each milk production system. Feed systems: HF = high forage system, LF = low forage system; genotypes: C = control, S = select.

The graphs (Figure 3) show equality of variance across groups. Similarly, the Levene statistic (Table 8) did not reject the null hypothesis that the group variances are equal for each model between years. The significance value of the *F*-test in the ANOVA (Table 9) was >0.05 ; thus, we cannot reject the hypothesis that average efficiency scores are equal across years (i.e., efficiency scores did not differ significantly between years). Additionally, we compared differences year-by-year with post hoc tests (LSD test for equal variances) and no significant differences were found. Results of the LSD test for equal variances are available online (<http://dx.doi.org/10.3168/jds.2013-6911>).

Results of testing differences in mean efficiency scores by system are presented in Tables 10, 11, and 12 and Figure 4. The graphs (Figure 4) show inequality of variance across groups. Similarly, the Levene statistic (Table 10) rejects the null hypothesis that the system group variances are equal for each model. The significance value of the *F*-test in the ANOVA (Table 11) was <0.05 ; thus, we reject the hypothesis that average efficiency scores were equal across years (i.e., average efficiency scores differed significantly between systems). To see precisely where the significant differences originated, we ran post hoc tests (Tamhane test for unequal variances; Table 12).

Table 8. Differences between environmental efficiency scores by year: Test of homogeneity of variances for models¹

Model ²	Levene statistic	df1	df2	Significance
NUO CO ₂ ,N,P	1.305	6	21	0.298
NUO CO ₂ ,N	1.214	6	21	0.338
UO CO ₂ ,N,P	1.535	6	21	0.215
UO CO ₂ ,N	1.169	6	21	0.360

¹df1 represents the degrees of freedom between groups [df1 = $y - 1$, where y equals the number of groups (i.e., 7 years)]; df2 represents the degrees of freedom within groups [df2 = $k - y$, where k equals the number of values in the set (i.e., 28 DMU)].

²NUO CO₂,N,P = normalized undesirable output-oriented (NUO) model with undesirable outputs CO₂, N, and P; NUO CO₂,N = NUO model with undesirable outputs CO₂ and N; UO CO₂,N,P = undesirable output-oriented (UO) model with undesirable outputs CO₂ and N, P; UO CO₂,N = UO model with undesirable outputs CO₂ and N.

Table 9. Analysis of variance for models

Model ¹		Sum of squares	df	Mean square	F-statistic	Significance
NUO CO ₂ ,N,P	Between groups	0.033	6	0.006	0.654	0.687
	Within groups	0.179	21	0.009		
	Total	0.213	27			
NUO CO ₂ ,N	Between groups	0.043	6	0.007	0.657	0.685
	Within groups	0.229	21	0.011		
	Total	0.273	27			
UO CO ₂ ,N,P	Between groups	0.023	6	0.004	0.397	0.872
	Within groups	0.200	21	0.010		
	Total	0.222	27			
UO CO ₂ ,N	Between groups	0.036	6	0.006	0.492	0.807
	Within groups	0.255	21	0.012		
	Total	0.291	27			

¹NUO CO₂,N,P = normalized undesirable output-oriented (NUO) model with undesirable outputs CO₂, N, and P; NUO CO₂,N = NUO model with undesirable outputs CO₂ and N; UO CO₂,N,P = undesirable output-oriented (UO) model with undesirable outputs CO₂ and N, P; UO CO₂,N = UO model with undesirable outputs CO₂ and N.

The results showed significant differences between systems in each model. The UO CO₂,N model presented the largest number of significant differences: between LFC and LFS, between HFC and HFS, between LFC and HFS, and between LFS and HFC. The NUO CO₂,N,P model presented significant differences between LFC and LFS, between HFC and HFS, and between LFS and HFC. The NUO CO₂,N model showed significant differences between LFC and LFS, and between LFS and HFC. The UO CO₂,N,P model presented significant differences between LFS and HFC, and between HFC and HFS.

The correspondence between the 2 DEA models (UO and NUO) was estimated using Spearman rank correlation and Kendall's Tau tests. According to the Spearman rank correlation test, the UO and NUO models were highly correlated, with a Spearman coefficient equal to 0.897 for NUO CO₂,N,P and UO CO₂,N,P models, and 0.897 for the NUO CO₂,N and UO CO₂,N models (correlations significant at the 0.01 level; 2-tailed). According to Kendall's Tau test (based on the number of concordant and discordant pairs of observations), the UO and NUO models showed a correlation coefficient of 0.850 for the NUO CO₂,N,P and UO CO₂,N,P models, and 0.842 for the NUO CO₂,N and UO CO₂,N models (correlations significant at the 0.01 level; 2-tailed). The 2 statistical tests proved that there were more differences between scores than between rankings obtained from the 2 types of models. Additionally, we ran the DEA models for each of the undesirable outputs (CO₂, N, and P). According to the Spearman rank correlation test, the UO and NUO models were significantly correlated in all cases, with a Spearman coefficient equal to 0.858 for the NUO CO₂ and UO CO₂ models, 0.968 for the NUO N and UO N models (correlations significant at the 0.01 level; 2-tailed), and 0.396 for the NUO P and UO P models (correlation significant at

the 0.05 level; 2-tailed). According to Kendall's Tau test, the UO and NUO models were significantly correlated in all cases, showing a correlation coefficient of 0.735 for NUO CO₂ and UO CO₂ models, 0.905 for the NUO N and UO N models (correlations significant at the 0.01 level; 2-tailed), and 0.321 for the NUO P and UO P models (correlation significant at the 0.05 level; 2-tailed). The 2 statistical tests confirmed that there were more differences between scores than between rankings obtained from the 2 types of models in all cases; that is, combined undesirable outputs and each undesirable output considered separately.

Tobit Model Results

Both of the independent variables (incidence of udder disorder and BCS) had a statistically significant effect on environmental efficiency scores in all models. These results highlight the lower incidence of udder health issues in the HF (grazed) herds. Body condition scores among the S cows tended to be lower, because they were more likely to give up their body condition to the production of milk. Results are presented in Table 13.

DISCUSSION

The UK dairy sector can be depicted by a diversity of management practices that, in recent years, have relinquished traditional all-summer grazing in favor of import-based feeding and year-round housing. Implementing efficiency improvements across a range of management types requires system-specific strategies. Western Europe and the United Kingdom are already among the most economically and carbon-efficient producers of milk in Europe and the world (O'Mara, 2011; DairyCo, 2012).

Table 10. Differences between environmental efficiency scores by system: Test of homogeneity of variances for models¹

Model ²	Levene statistic	df1	df2	Significance
NUO CO ₂ ,N,P	2.628	3	24	0.073
NUO CO ₂ ,N	4.202	3	24	0.016
UO CO ₂ ,N,P	3.014	3	24	0.050
UO CO ₂ ,N	3.511	3	24	0.031

¹df1 represents the degrees of freedom between groups [df1 = $y - 1$, where y equals the number of groups (i.e., 4 milk production systems)]; df2 represents the degrees of freedom within groups [df2 = $k - y$, where k equals the number of values in the set (i.e., 28 DMU)].

²NUO CO₂,N,P = normalized undesirable output-oriented (NUO) model with undesirable outputs CO₂, N, and P; NUO CO₂,N = NUO model with undesirable outputs CO₂ and N; UO CO₂,N,P = undesirable output-oriented (UO) model with undesirable outputs CO₂ and N, P; UO CO₂,N = UO model with undesirable outputs CO₂ and N.

Figure 3. Graphing means and standard errors to test homogeneity of variances of environmental efficiency scores by year for models: (a) NUO CO₂,N,P = normalized undesirable output-oriented (NUO) model with undesirable outputs CO₂, N, and P; (b) NUO CO₂,N = NUO model with undesirable outputs CO₂ and N; (c) UO CO₂,N,P = undesirable output-oriented (UO) model with undesirable outputs CO₂, N, and P; (d) UO CO₂,N = UO model with undesirable outputs CO₂ and N.

Table 11. Analysis of variance for models

Model ¹		Sum of squares	df	Mean square	F-statistic	Significance
NUO CO ₂ ,N,P	Between groups	0.121	3	0.040	10.481	0.000
	Within groups	0.092	24	0.004		
	Total	0.213	27			
NUO CO ₂ ,N	Between groups	0.128	3	0.043	7.067	0.001
	Within groups	0.145	24	0.006		
	Total	0.273	27			
UO CO ₂ ,N,P	Between groups	0.133	3	0.044	11.975	0.000
	Within groups	0.089	24	0.004		
	Total	0.222	27			
UO CO ₂ ,N	Between groups	0.167	3	0.056	10.714	0.000
	Within groups	0.124	24	0.005		
	Total	0.291	27			

¹NUO CO₂,N,P = normalized undesirable output-oriented (NUO) model with undesirable outputs CO₂, N, and P; NUO CO₂,N = NUO model with undesirable outputs CO₂ and N; UO CO₂,N,P = undesirable output-oriented (UO) model with undesirable outputs CO₂ and N, P; UO CO₂,N = UO model with undesirable outputs CO₂ and N.

The methods applied in this study evaluated undesirable outputs and compared relative efficiencies of 2 divergent dairy management systems to understand the differences between overall environmental performances. Langhill total farmgate emissions per system were comparable with a recent survey that reported an average of 1.31 kg of CO₂e per kg of ECM (DairyCo, 2012) and are consistent with a Scottish survey that reported 1.37 kg of CO₂e/L (Sheane et al., 2010). Even though these emissions per kilogram of ECM are much lower than the global average of 2.7 kg of CO₂ (FAO, 2010), the potential exists for lowering GHG emissions and nutrient surpluses within each system. The LF (housed) system was more carbon efficient, primarily because less manure-related emissions were generated; however, emissions from purchased feeds could offset this should the yield differences not be so great. System footprints could be reduced by introducing feeds with lower emission factors, such as by-products, or by introducing new technologies, such as installing an anaerobic digester. Lowering nutrient surpluses is limited by inefficiencies within the process of converting feed into milk (Dewhurst and Thomas, 1992); however, opportunities for reduction exist within each system. Nitrogen surplus could be reduced by introducing lower-protein forages in the diet, increasing milk yield per cow, reducing fertilizer application rates, and improving manure management (Roberts et al., 2007). Cost-effective approaches to reduce P surplus can be understood by whole-farm nutrient system analysis, and strategies include those tailored to land and crop management. Optimization of P in the diet has the potential to reduce surplus; however, this can lead to increased feed costs (Nousiainen et al., 2011). Nutrient losses resulting from slurry application can be controlled by regulations such as the EU Nitrates Directive (European Council, 1991),

which currently focuses on N. However, the introduction of the EU Water Framework Directive (European Parliament and Council, 2000), which regulates aquatic systems as a whole, may lead to the development of combined nutrient manure management plans.

Ranking efficiency scores and comparing DEA model types showed little discrepancy between the majority of system years, as most DMU were similarly ranked. However, exceptions were noted: HFS 2007 ranked considerably more favorably in models that contained P as a pollutant, because this system year generated the least amount of surplus compared with all groups in all years and performed comparatively well with regard to other pollutants; LFC 2005 also performed better when P was included in the DEA model because of a comparatively higher milk yield. These instances highlight the benefits of considering more than one type of pollutant as well as more than one type of model.

Results of the Tamhane test showed significant differences between the efficiency scores of the 4 dairy management systems, which arose from genetics or feed type factors. Across all DEA models, significance between system efficiency scores arose from genetic differences more often than feed system differences. These results also highlight an interaction between the dairy management systems, the choice of pollutants, and the type of DEA model implemented. The NUO models did not incorporate the effects of input use, and significant differences found here resulted mainly from genetics. When looking at UO models, an equal number of system differences caused by genetics were found to occur; however, there were greater instances of feed type differences. These results emphasize that resource inputs play an important role in the measurement of overall environmental performance of these dairy systems. This is a potentially important finding

Table 12. Tamhane test for unequal variances for models for each milk production system¹

Dependent variable ²	(I) system	(J) system	Mean difference (I – J)	SE	Significance	95% CI	
						Lower bound	Upper bound
NUO CO ₂ ,N,P	LFC	LFS	–0.10014*	0.02273	0.027	–0.1875	–0.0128
		HFC	0.07229	0.04165	0.511	–0.0627	0.2073
		HFS	–0.06371	0.03117	0.326	–0.1617	0.0343
	LFS	LFC	0.10014*	0.02273	0.027	0.0128	0.1875
		HFC	0.17243*	0.03491	0.016	0.0382	0.3066
		HFS	0.03643	0.02135	0.592	–0.0456	0.1185
	HFC	LFC	–0.07229	0.04165	0.511	–0.2073	0.0627
		LFS	–0.17243*	0.03491	0.016	–0.3066	–0.0382
		HFS	–0.13600*	0.04091	0.046	–0.2697	–0.0023
	HFS	LFC	0.06371	0.03117	0.326	–0.0343	0.1617
		LFS	–0.03643	0.02135	0.592	–0.1185	0.0456
		HFC	0.13600*	0.04091	0.046	0.0023	0.2697
NUO CO ₂ ,N	LFC	LFS	–0.12800*	0.01950	0.000	–0.1893	–0.0667
		HFC	0.05143	0.04503	0.872	–0.1105	0.2134
		HFS	–0.06529	0.03725	0.533	–0.1959	0.0653
	LFS	LFC	0.12800*	0.01950	0.000	0.0667	0.1893
		HFC	0.17943*	0.04536	0.030	0.0177	0.3411
		HFS	0.06271	0.03765	0.580	–0.0679	0.1933
	HFC	LFC	–0.05143	0.04503	0.872	–0.2134	0.1105
		LFS	–0.17943*	0.04536	0.030	–0.3411	–0.0177
		HFS	–0.11671	0.05536	0.300	–0.2921	0.0586
	HFS	LFC	0.06529	0.03725	0.533	–0.0653	0.1959
		LFS	–0.06271	0.03765	0.580	–0.1933	0.0679
		HFC	0.11671	0.05536	0.300	–0.0586	0.2921
UO CO ₂ ,N,P	LFC	LFS	–0.08900	0.02681	0.092	–0.1921	0.0141
		HFC	0.08400	0.04401	0.402	–0.0560	0.2240
		HFS	–0.07543	0.03005	0.185	–0.1765	0.0256
	LFS	LFC	0.08900	0.02681	0.092	–0.0141	0.1921
		HFC	0.17300*	0.03490	0.015	0.0388	0.3072
		HFS	0.01357	0.01357	0.929	–0.0386	0.0657
	HFC	LFC	–0.08400	0.04401	0.402	–0.2240	0.0560
		LFS	–0.17300*	0.03490	0.015	–0.3072	–0.0388
		HFS	–0.15943*	0.03745	0.018	–0.2903	–0.0285
	HFS	LFC	0.07543	0.03005	0.185	–0.0256	0.1765
		LFS	–0.01357	0.01357	0.929	–0.0657	0.0386
		HFC	0.15943*	0.03745	0.018	0.0285	0.2903
UO CO ₂ ,N	LFC	LFS	–0.13171*	0.02027	0.000	–0.1955	–0.0680
		HFC	0.05657	0.04519	0.822	–0.1053	0.2184
		HFS	–0.10771*	0.02980	0.033	–0.2074	–0.0080
	LFS	LFC	0.13171*	0.02027	0.000	0.0680	0.1955
		HFC	0.18829*	0.04553	0.023	0.0267	0.3499
		HFS	0.02400	0.03031	0.972	–0.0763	0.1243
	HFC	LFC	–0.05657	0.04519	0.822	–0.2184	0.1053
		LFS	–0.18829*	0.04553	0.023	–0.3499	–0.0267
		HFS	–0.16429	0.05050	0.051	–0.3293	0.0007
	HFS	LFC	0.10771*	0.02980	0.033	0.0080	0.2074
		LFS	–0.02400	0.03031	0.972	–0.1243	0.0763
		HFC	0.16429	0.05050	0.051	–0.0007	0.3293

¹Feed systems: HF = high forage system, LF = low forage system; genotype: C = control, S = select.

²NUO CO₂,N,P = normalized undesirable output-oriented (NUO) model with undesirable outputs CO₂, N, and P; NUO CO₂,N = NUO model with undesirable outputs CO₂ and N; UO CO₂,N,P = undesirable output-oriented (UO) model with undesirable outputs CO₂ and N, P; UO CO₂,N = UO model with undesirable outputs CO₂ and N.

* $P < 0.05$.

because earlier work suggested that high-genetic-merit cows did not lose their advantage on low-input feeding regimens; that is, there was no G×E based on gross energetic efficiency (Veerkamp et al., 1994). However, this study suggests that the situation is less clear when environmental efficiency is considered. It could be ex-

pected that no significant differences in efficiency score between years were found to occur, because the diets should have been as constant as possible to conform within the Langhill systems experiment.

This study found that udder health disorders and BCS had a significant effect on environmental effi-

Figure 4. Graphing means and standard errors to test homogeneity of variances of environmental efficiency scores by system for models: (a) NUO CO₂,N,P = normalized undesirable output-oriented (NUO) model with undesirable outputs CO₂, N, and P; (b) NUO CO₂,N = NUO model with undesirable outputs CO₂ and N; (c) UO CO₂,N,P = undesirable output-oriented (UO) model with undesirable outputs CO₂, N, and P; (d) UO CO₂,N = UO model with undesirable outputs CO₂ and N.

ciency scores, which indicates the importance of animal health within milk production systems. Analyzing the effect of various factors on efficiency is a frequently used approach in the literature (Barnes et al., 2011; Manevska-Tasevska et al., 2011).

Opportunities exist for UK dairy farmers to successfully compete in a global industry, because grasslands, primarily in the west of Britain, receive ample rainfall and the growing season is sufficient. Moreover, 2015 should herald a new era because output will no longer be restricted by milk quotas introduced in 1984 under the European Common Agricultural Policy. Regeneration of milk production should be carried out in a sustainable manner with an emphasis on increasing efficiency and lowering undesirable outputs.

It could be argued that research farms such as Langhill are generally more efficient in terms of resource use; they perform better in terms of average yields and, as such, could act as references for environmental policy. A changing climate may provide further opportunities in some parts of the United Kingdom as the growing season could be extended if predictions of warmer and wetter summers are realized, and UK dairy farmers could capitalize on natural resources as other regions may not have such opportunities.

The policy implications of this analysis would be enhanced if, instead of considering the best practice frontier, the DEA models used a frontier reflecting environmental quality targets. Thus, dairy systems would be ranked with respect to each other and with

Table 13. Tobit model results (coefficients, robust SE, z-statistic test, and probabilities)

Model and trait ¹	Coefficient	SE ²	z-Statistic	Probability
NUO CO ₂ ,N,P				
BCS	0.384373	0.021984	17.48418	0.0000**
Incidence of udder disorder	0.019139	0.007428	2.576629	0.0100**
NUO CO ₂ ,N				
BCS	0.377125	0.018913	19.94027	0.0000**
Incidence of udder disorder	0.010193	0.004387	2.323409	0.0202*
UO CO ₂ ,N,P				
BCS	0.377291	0.028120	13.41728	0.0000**
Incidence of udder disorder	0.034911	0.010827	3.224353	0.0013**
UO CO ₂ ,N				
BCS	0.372700	0.028083	13.27129	0.0000**
Incidence of udder disorder	0.021610	0.007911	2.731487	0.0063**

¹NUO CO₂,N,P = normalized undesirable output-oriented (NUO) model with undesirable outputs CO₂, N, and P; NUO CO₂,N = NUO model with undesirable outputs CO₂ and N; UO CO₂,N,P = undesirable output-oriented (UO) model with undesirable outputs CO₂ and N, P; UO CO₂,N = UO model with undesirable outputs CO₂ and N.

²Quasi-maximum likelihood (Huber/White) standard errors and covariance.

* $P < 0.05$; ** $P < 0.01$.

respect to a hypothetical system based on low carbon policy goals. If measures to mitigate GHG emissions are adopted alongside strategies for nutrient surplus reduction annually at a system level, then advances in environmental performance could be acknowledged and quantified. Alternatively, or additionally, a comparison between the efficiency of the mitigation measures used could be undertaken (Bosetti and Buchner, 2009). Other opportunities for improvements in the methodology would be to include pesticides within the footprints and water use as additional DEA variables. Indicators for animal welfare and biodiversity could also be included, if data are available.

CONCLUSIONS

The purpose of this paper was to compare the environmental efficiency of divergent strains of dairy cows within contrasting management systems and investigate any G×E interaction. Although the LFS system was most efficient in the majority of DEA models, cows of the same genetic merit on HFS could be just as efficient. Results indicated no G×E interaction, but do indicate that system efficiency is not limited to high milk production and that lower production systems are shown to be efficient in some years. Inputs play an important role in the measurement of environmental efficiency of dairy systems, and opportunities to reduce undesirable outputs varied dependent upon system type. Thus, mitigation of dairy pollutants should be customized by system. Efficiency scores for the LF groups were less variable from year to year, which reflected the uniformity of purchased concentrate feeds.

Significant differences were found between efficiency scores among the systems, mainly due to genetic rather than feed system differences. Furthermore, herd health was shown to be an important factor. Improving herd health by, for example, targeted health schemes would lead to improved environmental efficiency. Environmental efficiency demands a holistic approach to ensure that all attributes of dairy systems are properly matched to production circumstances and objectives. A focus on high outputs or single indices of technical efficiency alone may not address the multiple challenges food production systems face.

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